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Effects of Lost Creek Dam on Fall Chinook Salmon in the Rogue River

Phase II Completion Report

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EFFECTS OF LOST CREEK DAM ON FALL CHINOOK SALMON IN THE ROGUE RIVER. PHASE II COMPLETION REPORT.

Rogue Basin Fisheries Evaluation Project
Research and Development Section

Oregon Department of Fish and Wildlife

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FOREWORD

This report is the culmination of 13 years of research funded by the U.S. Army Corps of Engineers. A study of this duration has necessarily involved the collective effort of many people since its inception in 1974. For this reason, it is being presented as a staff report of personnel on the Rogue Basin Fisheries Evaluation Project. The completion report was drafted by Thomas Satterthwaite who was largely responsible for analyses contained in the report. Barry McPherson supervised the project through the initial stages of preparation. This report is the third of a series of completion reports planned for anadromous salmon and steelhead stocks produced in the Rogue River basin.

James Lichatowich was responsible for the original design and guidance of research on anadromous salmonids affected by the operation of Lost Creek Dam. These duties were subsequently assumed by Steven Cramer who served as program leader until 1985. Their leadership and insights on study designs were largely responsible for the ultimate success of research conducted by personnel in the Rogue Basin Fisheries Evaluation Project.

The mainstem and tributaries of the Rogue River collectively produce the largest population of wild anadromous salmonids in Oregon. The Rogue River supports recreational and commercial fisheries of immense importance to Oregon citizens and is nationally renowned for its diversity and productivity. Authorizing documents for Lost Creek Dam stipulate that fisheries enhancement is to be an important benefit of the dam, mainly through improved temperature and flow. We hope our studies will ensure that these benefits are achieved for present and future generations of Oregon citizens.

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24 February 1992

SUMMARY

In this report, we evaluate the effects of Lost Creek Dam on wild fall chinook salmon *Oncorhynchus tshawytscha* in the Rogue River. Field sampling began in 1974 and ended in 1986. Lost Creek Dam closed in February 1977, but the reservoir did not fill completely until spring of 1978. A summary of our findings follows.

Juveniles

1. Simulation modeling of water temperature by the United States Army Corps of Engineers indicated operation of Lost Creek Dam caused fry to emerge about 14 days earlier in the area near Grants Pass. We found emergence timing would have had to change by more than 40 days to be detected by field sampling.
2. Juvenile production was related to (1) intensity of peak flow during gravel incubation of eggs and alevins, (2) flow in tributary streams during adult migration and spawning, and (3) number of spawned females.
3. Operation of Lost Creek Dam increased juvenile production by reduction of peak flows. Operation of Applegate Dam increased juvenile production by augmentation of flow when adults migrated to spawning areas.
4. Spawning escapement limited juvenile production in most, and possibly all, years of the project.
5. Juvenile growth increased after operation of Lost Creek Dam. Juveniles grew faster, were heavier at a given length, and were larger at time of ocean entry because flow augmentation decreased water temperature to more optimal levels in summer.
6. Subyearling migrants from postimpoundment broods entered the ocean an average of 23 days earlier than counterparts from preimpoundment broods. Postimpoundment broods entered the ocean earlier because they grew at faster rates.
7. Most juveniles entered the ocean in the first year of life. Yearling migrants were less abundant within broods that grew at faster rates.
8. Growth rate in fresh water, age at ocean entry, and time of ocean entry differed among juveniles destined to mature as adults of different ages. Juveniles that matured at age 5 grew slower in fresh water and entered the ocean later compared with cohorts that matured at younger ages. Juveniles that matured at age 2 were less likely to be yearling migrants compared with cohorts that matured at older ages.

Adults

1. Estimates of freshwater returns derived from seine catches near the mouth of the Rogue River were positively related with counts of spawned carcasses in standard survey areas.

2. Estimates of freshwater returns averaged 43,800 wild adults and 1,100 hatchery adults in 1974-86.

3. We detected no difference in return of age 2 adults that originated from preimpoundment and postimpoundment broods. Returns for older ages were not tested because rates of fishing mortality and natural mortality varied greatly between years of ocean residence.

4. The El Niño event of 1982-83 increased rates of natural mortality among all ages of fall chinook salmon that reared in the ocean during 1982-83 and 1983-84. Increases in mortality rates were greater in 1982-83 compared with 1983-84.

5. Estimates of ocean abundance during spring of 1974-86 averaged about 750,000 age 2 fish, 260,000 age 3 fish, 77,000 age 4 fish, 8,000 age 5 fish, and less than 1,000 age 6 fish.

6. Production estimates ranged from 1 to 23 recruits per spawner for the 1974-80 broods. Age 2 jacks were excluded from these estimates. Data were insufficient to develop a stock-recruitment relationship.

7. Freshwater returns can be used to estimate fish abundance in the ocean during the succeeding spring, except during an El Niño event.

8. A minimum of six discrete populations of fall chinook salmon inhabited the Rogue River basin. These populations differed significantly in life history parameters.

9. Adults matured at ages 2-6. Estimates of maturity rates indicated the run could be classified as a midmaturing group of chinook salmon. However, fall chinook salmon of Applegate River origin matured at younger ages compared with the entire run. Other populations in the basin matured at older ages.

10. Operation of Lost Creek Dam may have affected maturity rates of adults because juveniles from postimpoundment broods entered the ocean earlier than preimpoundment broods. Small sample sizes prevented comparisons of maturity rates for preimpoundment and postimpoundment broods.

11. We estimated ocean fisheries in 1974-86 annually harvested an average of 163,000 wild fall chinook salmon that originated in the Rogue River basin. We estimated ocean harvest accounted for an average of 78% of the fall chinook salmon produced in the Rogue River basin.

12. Exploitation rates during the late 1970s and early 1980s exceeded optimum. High rates of harvest least affected recruitment for the population in the Applegate River. Recruitment from populations that matured at older ages were most affected.

13. Adults entered the Rogue River primarily in August-October. Populations differed in time of freshwater entry. Annual variations in relative abundance of populations affected estimates of migration timing for the entire run.

14. Operation of Lost Creek Dam affected run timing at river entry because of differential effects among populations in the basin. The entire run will appear to migrate earlier as decreased rates of prespawning mortality increase productivity of populations adapted for early freshwater return.

15. Large numbers of adults died prior to spawning before and after operation of Lost Creek Dam. We estimated mortality rates of 81% in 1979 and 57% in 1980. Annual losses in 1983-86 were less than 1%.

16. Rates of prespawning mortality in postimpoundment years were positively related with water temperature in summer. Increased flow augmentation in 1981-86 decreased rates of prespawning mortality.

17. Estimates of freshwater harvest averaged 2,900 fall chinook salmon in 1956-86. These estimates only included fish large enough to require entry on salmon-steelhead cards.

18. Estimates of harvest rates in the Rogue River averaged 10% in 1974-86. Harvest rate was negatively related with water temperature in late summer. Anglers harvested a greater percentage of the run in years of augmented flow and low prespawning mortality.

19. We detected no difference in spawning distribution or spawning time of adults that originated from preimpoundment and postimpoundment broods. However, flow in late summer was positively related with the percentage of spawners that passed Gold Ray Dam. Proportionally more fish spawned upstream of the counting station in years of high flow.

20. Sex composition of carcasses recovered during spawning surveys was related to age composition of the run. Returns dominated by older age classes had proportionately more females.

Simulation Model

1. We developed a simulation model that reflected major life history events in fresh water and in the ocean, but we were not able to simulate changes in maturity rates or rates of natural mortality in the ocean.

2. Simulations indicated recruitment of age 3 progeny was most sensitive to increases in water temperature of the Rogue River in late summer during years when parents returned to spawn.

3. Uncertainty associated with predictions of model output were greatest for prespawning mortality and recruitment of age 3 progeny.

4. Simulations indicated production of age 3 progeny decreases when water temperature at Agness averages more than 19.8°C in August-September.

5. Simulations suggested, for similar environmental conditions, rates of prespawning mortality are greater for larger runs than smaller runs.

RECOMMENDATIONS

Reservoir Management and Operation of Lost Creek Dam

1. Outflow from Lost Creek Dam should be managed so daily maximum water temperature averages less than 19.8°C at Agness in August-September. This goal can usually be attained with a minimum flow of 2,300 cfs at Agness from 10 August through 10 September. Outflow from Lost Creek Dam should be about 2,200 cfs to maintain a flow of 2,300 cfs at Agness in years of low water yield. This recommendation is designed to minimize effects of prespawning mortality on the subsequent production of progeny (*see* **Prespawning Mortality**, page 74 and **Application for Reservoir Management**, page 92).

2. Outflow from Lost Creek Dam should be managed so river flow is not augmented in October. Also, reservoir outflow during 10-30 September should not exceed outflow from 10 August through 10 September. These recommendations are designed to minimize migration of fall chinook salmon into spawning areas used by spring chinook salmon (*see* **Spawning Distribution**, page 81). Overlap in spawning distributions is not desirable because the spring race spawns in a small portion of the Rogue River basin.

3. Outflow from Lost Creek Dam should be managed to minimize intensity of peak flows in downstream areas in November-March. This recommendation is designed to increase survival rates of eggs and alevins that incubate in the gravel (*see* **Abundance**, page 39). Present strategies for reservoir operation decrease peak flows during operational seasons of flood control and conservation storage. We believe the intensity of peak flows can be further decreased in years of high water yield.

   Authorizing documents for the Rogue River basin project designate flood control as the first priority for reservoir management. Storage in excess of the rule curve decreases reservoir capability for flood control. However, maintenance of the reservoir level below the rule curve can provide for additional reductions in peak flows.

   The United States Army Corps of Engineers should develop criteria for reservoir level in operational seasons for flood control and conservation storage. We believe reservoir level can be scaled to estimates of water yield in the area upstream of the reservoir. Reservoir level should be reduced when water content of the snowpack is great. Implementation of this recommendation would increase reservoir capacity for flood control and decrease intensity of peak flows in downstream areas.

4. Release of water stored in the reservoirs during freshets should be managed so flow in downstream areas does not exceed the peak flow that previously occurred during the season. We recognize this recommendation may conflict with flood control operations. For example, managers may seek to return the reservoir level to the authorized rule curve for short periods between large storms. However, when potential for further flooding is minimal, reservoir level should be returned to minimum pool for flood control (or lower) so as not to produce a new peak flow in downstream areas.

Management and Evaluation of Fishery Resources

The following recommendations are directed primarily to the Oregon Department of Fish and Wildlife, the lead agency for management of fishery resources in the Rogue River basin. Cooperation of other state and federal agencies may be needed to implement these recommendations.

1. Index areas should be established to monitor long-term trends in production of juvenile fall chinook salmon in the Rogue River basin. Traps can be used to estimate juvenile production in smaller streams. Juvenile abundance in large streams can be effectively indexed with beach seines (*see* **Abundance**, page 39). At a minimum, juvenile abundance in the middle portion of the Rogue River should be monitored by weekly seining at Matson Park and Almeda Park from May through August.

2. A goal for spawning distribution should be developed for fall chinook salmon in the Applegate River (*see* **Abundance**, page 39). Reservoir storage currently allocated to increase the upstream distribution of spawners may or may not be entirely needed. Any excess storage could be allocated for other purposes of fishery enhancement.

3. Management plans and activities should recognize a minimum of six distinct populations of fall chinook salmon in the Rogue River basin. These populations are located (1) upstream of Gold Ray Dam, (2) in the Rogue River from Gold Ray Dam to Grave Creek, (3) in the Applegate River, (4) in tributaries of the lower Applegate, (5) in the Illinois River basin upstream of Klondike Creek, and (6) in the Rogue River basin downstream of Grave Creek and the Illinois River basin downstream of Klondike Creek. We believe differences in life history parameters (*see* **Age at Maturity**, page 66; **Migration Timing**, page 71; and **Spawning Time**, page 83) probably reflect genetic differences among populations.

4. Populations of fall chinook salmon in all major drainages of the Rogue River basin should be examined for unique genetic resources. Knowledge of the distribution of stocks within the basin is needed to develop effective strategies to maintain and enhance diversity among populations of wild fish. We believe maintaining diversity among wild fish is needed to sustain production of fall chinook salmon in the basin.

5. Exploitation rates in ocean and freshwater fisheries should be regulated to target for a spawning escapement of 23,000 age 3-6 fall chinook salmon in the Rogue River basin. Smaller escapements resulted in decreased juvenile production in the succeeding year (*see* **Abundance**, page 39). Effects of larger spawning escapements on juvenile production are not presently known. Additional sampling is needed to index juvenile abundance after years of large spawning escapement. The recommendation for a spawning escapement of 23,000 age 3-6 adults should be evaluated by an analysis of the production potential of habitat in the basin.

6. Effects of ocean harvest on populations of fall chinook salmon in the Rogue River basin that mature at older ages should be evaluated. Annual exploitation rates of 30% for age 3-6 fish in the ocean and 15% in the river may make it difficult to maintain late-maturing populations. We found indications that harvest in the ocean fisheries during the late

1970s and early 1980s may have accounted for more than 90% of the fish produced in late-maturing populations (*see* **Ocean Harvest**, page 68).

7. Techniques for estimation of freshwater return and spawning escapement of fall chinook salmon in the Rogue River basin require evaluation. Use of seine catches to estimate freshwater return should be evaluated with an independent method. Our mark-recapture experiments with fall chinook salmon failed because of high rates of tagging mortality. Hydroacoustic sampling may prove useful when techniques are developed to estimate returns of anadromous fish in large rivers.

   Indexing of spawning escapement from carcass counts in non-random survey areas also needs evaluation. Results from spawning surveys in the middle Rogue and Applegate River probably do not reflect spawning escapement in the Illinois River basin or tributaries of the lower Rogue. Rates of carcass recovery vary among survey areas because of differences in stream morphology, water clarity, and age composition of spawners. Until more reliable methods can be developed to estimate adult abundance and age composition, equal weight should be given to estimates of spawning escapement derived from seine catches and counts of spawned carcasses.

8. Salmon-steelhead cards should be modified to improve estimates of the spatial distribution of freshwater harvest of fall chinook salmon in the Rogue River (*see* **Freshwater Harvest**, page 78). The harvest area from the Rogue River estuary to Gold Ray Dam should be divided into three harvest areas (1) the estuary, (2) from the estuary to Agness, and (3) from Agness to Gold Ray Dam.

9. Habitat projects designed to maintain or increase production of fall chinook salmon in the Rogue River basin should be directed at gravel quality and quantity in streams with flows great enough to permit entry and migration of adults (*see* **Abundance**, page 39).

10. Hatchery programs designed to supplement populations of fall chinook salmon in the Rogue River basin should use wild broodstock. Juveniles of hatchery origin should be released near the site of broodstock collection to minimize the potential genetic exchange between populations.

11. Hatchery programs designed to increase ocean harvest of fall chinook salmon of Rogue River origin should develop broodstocks from late-maturing populations. Broods that mature at older ages contribute to ocean fisheries at greater rates than broods that mature at younger ages (*see* **Ocean Harvest**, page 68).

12. Hatchery programs designed to supplement wild populations should release juveniles no earlier than August to minimize competition with wild fish (*see* **Time of Ocean Entry:**, page 50). Hatchery programs designed to increase ocean harvest of fall chinook salmon should release juveniles no earlier than September. Later release will delay maturation and increase the contribution rate to ocean fisheries (*see* **Age at Maturity**, page 66).

13. Management plans for public and private lands in the Rogue River basin should identify and minimize activities that cumulatively affect intensity of peak flows in downstream areas (*see* **Abundance**, page 39).

INTRODUCTION

This report presents the findings of 13 years (1974-86) of work with fall chinook salmon *Oncorhynchus tshawytscha* in the Rogue River basin of southwestern Oregon. The Oregon Department of Fish and Wildlife (ODFW) conducted this study, funded by the United States Army Corps of Engineers (USACE), to (1) determine the effects of Lost Creek Dam on anadromous salmonids and (2) develop operating strategies that optimize the production and harvest of fishery resources in downstream areas.

The Congress of the United States of America authorized construction of Lost Creek Dam (Figure 1) at river kilometer (RK) 254 to create a reservoir to be used for multiple purposes, including the enhancement of fishery resources in downstream areas (United States Congress 1962). An updated economic review in 1971 indicated planners projected fishery enhancement to be the third largest benefit accrued annually from the operation of the dam (USACE 1972). Spawning and rearing habitat for salmon and steelhead *Oncorhynchus mykiss* blocked or inundated by the dam was to be mitigated by releases of fish reared at Cole M. Rivers Hatchery. Benefits to anadromous salmonids in downstream areas were expected to accrue by operating the dam to (1) decrease peak flow in winter, (2) increase flow in summer, and (3) decrease water temperature in summer.

To regulate the outflow temperature from Lost Creek Dam, the USACE designed an intake structure capable of withdrawing water from five different levels of the reservoir (Figure 2). Selective opening of intake ports allows for mixing of water from various temperature strata in the reservoir. Choice of outflow temperature is greatest in early summer when the reservoir is full and thermally stratified. Control of release temperature diminishes in late summer as reservoir level decreases and the highest intake ports become dewatered. Control of release temperature becomes minimal in autumn after the reservoir destratifies (USACE 1983).

Guidelines for the release of stored water were intended to be flexible, reflecting annual variations in water yield and user demand. When the reservoir fills, 180,000 acre-feet of storage is available for flow augmentation (USACE 1972). Of this total, 125,000 acre-feet were authorized for fishery enhancement (United States Congress 1962). The remaining 55,000 acre-feet of storage was dedicated to other uses: irrigation supply, municipal and industrial supply, and environmental enhancement. Dedicated storage that is not purchased is also available for downstream enhancement of fishery resources (USACE 1972).

The authorizing document identified flood control as the primary benefit associated with the construction of Lost Creek Dam. Other benefits would accrue by allocating conservation storage to irrigation, future water supply, and fishery enhancement. There was to be "No storage specifically for wildlife enhancement, power generation, water quality control, or recreation" (United States Congress 1962).

The authorizing document also outlined minimum outflow and maximum water temperature to be released from Lost Creek Dam, but clearly stated these guidelines should be modified as additional information became available: "It should also be noted that project operation plans must be sufficiently

Figure 1. Lost Creek Dam and Cole M. Rivers Hatchery.

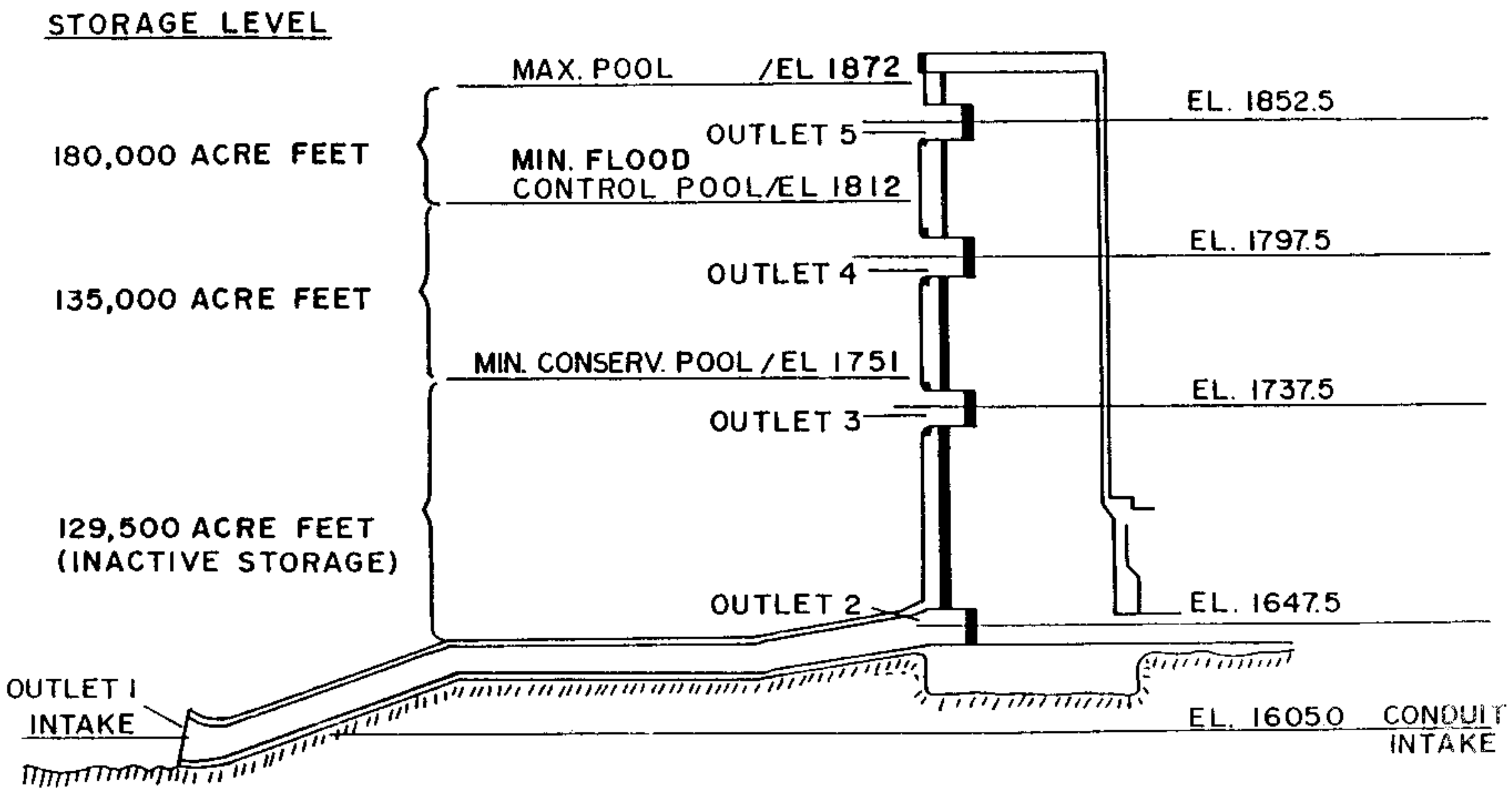

Figure 2. Schematic of the multiport intake structure for water withdrawal from five levels within Lost Creek Lake.

flexible to permit desirable modifications in scheduled fishery releases, within the limits of storage provided therefore, if experience and further study indicates such action to be desirable for overall project benefits" (United States Congress 1962). Including provisions for modifications of release strategies for fisheries benefits was farsighted as predictions of postproject responses are rarely accurate because of the complexity of aquatic ecosystems (Rosenberg et al. 1986).

Flexibility in scheduling temperature and flow releases from Lost Creek Dam provides an opportunity to implement an operating strategy that optimizes the production and harvest of anadromous salmonids in the river downstream of the dam. To identify the most appropriate operating strategy, we examined the effect of water temperature, flow, and turbidity on the biology and harvest of wild fall chinook salmon in the Rogue River basin. Preliminary findings from our work were reported in numerous annual progress reports and were summarized by Cramer et al. (1985).

We did not intensively study fall chinook salmon of hatchery origin because few hatchery fish were released during the study. Lost Creek Dam did not block any spawning areas used by wild fall chinook salmon. Consequently, there was no hatchery mitigation for habitat loss.

Fall chinook salmon produced in the Rogue River basin are an important fishery resource. Returns to the Rogue River constitute the largest run of chinook salmon on the Oregon coast (Nicholas and Hankin 1988). Also, fall chinook salmon that originate in the Rogue River basin contribute to commercial and recreational fisheries in coastal waters of Oregon and northern California (Garrison 1987). Production costs are minimal because wild fish account for almost all fall chinook salmon that originate from the Rogue River basin.

In this report, we estimate the effects of Lost Creek Dam on fall chinook salmon and present recommendations to enhance the production and harvest of fall chinook salmon. Use of water releases from Lost Creek Dam to increase stock productivity would be a low cost method of fishery enhancement. This report represents one of a series of completion reports for fisheries work in the Rogue River basin funded by the USACE.

STUDY AREA

The Rogue River basin encompasses 13,150 square kilometers of southwestern Oregon and a small portion of northwestern California (Figure 3). Approximately 13% of the basin is upstream of Lost Creek Dam. The Rogue River originates in the Cascade Mountains and flows west, breaching the Klamath Mountains prior to reaching the Pacific Ocean. Two major tributaries, the Illinois and Applegate rivers, originate in the Siskiyou Mountains and flow north where they enter the Rogue River at river kilometers (RK) 44 and 154, respectively.

The Rogue River estuary is relatively small, covering an area of about 630 acres at mean high tide. Ratti (1979) reported about 80% of the estuary could be classified as a riverine subsystem and 20% could be classified as a marine subsystem. Tideflats, marshes, and eelgrass beds are noticeably absent in the estuary.

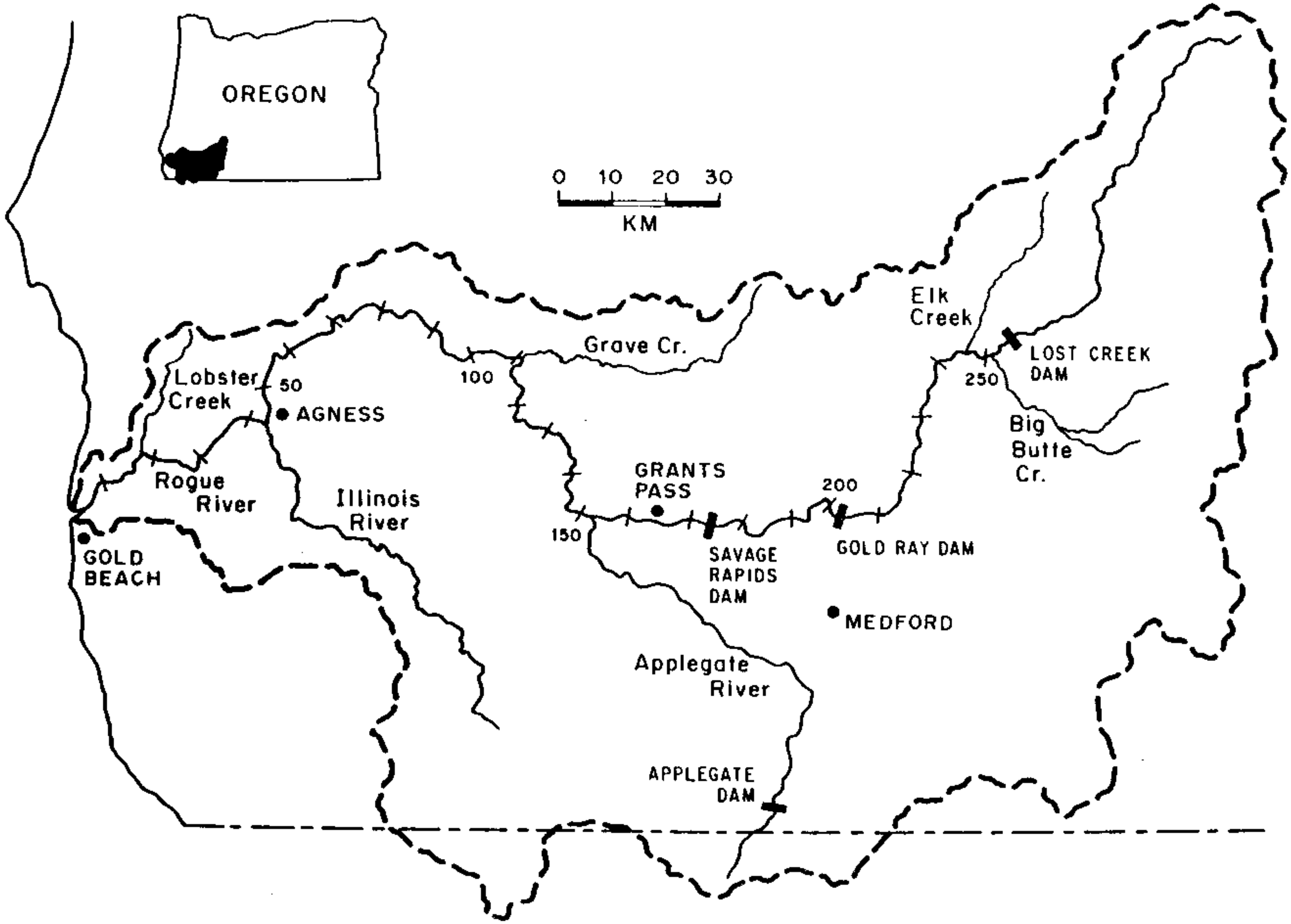

Figure 3. Map of the Rogue River basin. Numbers indicate kilometers from the river mouth.

Two USACE dams affect the timing of water yield in the Rogue River basin. Lost Creek Dam at RK 254 on the Rogue River began operating in February 1977. Applegate Dam, at RK 75 on the Applegate River, began operating in November 1980 and affects flow in the Rogue River downstream of Grants Pass. Operation of Applegate Dam has a lesser effect on flow of the Rogue River because the normally used storage capacity of Applegate Lake is one-third that of Lost Creek Lake.

The Rogue River basin yields an average of 7.4 million acre-feet of water annually (Friday and Miller 1984). The Illinois and Applegate rivers average approximately 40% and 7% of the water yielded annually in the basin, respectively. The Rogue River upstream of Lost Creek Dam accounted for an average of 18% of the water yielded annually in the basin.

In the lower portion of the basin, river flow varies markedly among seasons. Discharge upstream of the mouth of the Illinois River averages 1,400 cfs in September and 26,600 cfs in January. The variation in flow is less pronounced in the upper portion of the basin. Flow into Lost Creek Lake averages 1,000 cfs in September and 2,000 cfs in January (Moffatt et al. 1990). Reservoir inflow usually peaks when the snowpack in the Cascade Mountains melts at a rapid rate between April and June.

Weather patterns in the northeast Pacific greatly affect climate within the Rogue River basin. Wet, mild winters and dry, warm summers characterize the climate. Air temperature near Medford usually peaks between 32^{o}C and 35^{o}C in July and August. In December and January, air temperature usually peaks between 8^{o}C and 10^{o}C. Snow accumulates at the higher elevations in winter and is the principal source of water yield in spring and early summer. Annual precipitation averages about 50 cm in the inland valley surrounding Medford. Coastal and headwater regions receive an average annual precipitation of about 200 cm and 300 cm, respectively (ODWR 1985). About 50% of the annual precipitation falls from November through January. Less than 2% falls in July and August.

A large number of anadromous fish inhabit the Rogue River basin. Chinook salmon and steelhead are the most abundant salmonids. Coho salmon *O. kisutch* are present in tributary streams. Chum salmon *O. keta* and pink salmon *O. gorbuscha* are occasionally found in tributaries of the lower river. Resident salmonids include rainbow trout *O. mykiss*, cutthroat trout *O. clarki*, brown trout *Salmo trutta*, and brook trout *Salvelinus fontinalis*. Few resident salmonids inhabit areas accessible to anadromous salmonids. Other commonly seen game fishes include largemouth bass *Micropterus salmoides*, bluegill *Lepomis macrochirus*, green sturgeon *Acipenser medirostris*, white sturgeon *A. transmontanus*, American shad *Alosa sapidissima*, and brown bullhead *Ictalurus nebulosus*. Nongame fishes abundant in the basin include redside shiner *Richardsonius balteatus*, Klamath smallscale sucker *Catostomus rimiculus*, common carp *Cyprinus carpio*, prickly sculpin *Cottus asper*, riffle sculpin *C. gulosus*, and Pacific lamprey *Lampetra tridentata*. The distribution of northern squawfish *Ptychocheilus oregonensis* is rapidly expanding after an illegal introduction in 1979.

For discussion purposes, we divided the Rogue River into four general areas. The upper river refers to the area between Lost Creek Dam and Gold Ray Dam (RK 202). The middle river refers to the area between Gold Ray Dam and Grave Creek (RK 110). The canyon refers to the area between Grave Creek and Agness (RK 44). The lower river refers to the area between Agness and the estuary (RK 6). Stream gradient averages 2.3 m/km in the upper river, 1.6 m/km in the middle river, 2.4 m/km in the canyon, and 0.7 m/km in the lower river.

APPROACH

We chose not to use the instream flow incremental methodology (Bovee 1982) for the development of flow recommendations. Although this approach has proved useful in some instances, the assumed direct relationship between weighted usable area and fish production is not always appropriate (Mathur et al. 1985; Moyle and Baltz 1985; Irvine et al. 1987). Our work centered primarily upon assessing the biological implications of modifications in flow, water temperature, and turbidity. During planning of the study, changes in these physical factors were expected to be significant in the area of the river inhabited by fall chinook salmon.

The study comprised four objectives:

1. Determine the changes in temperature, flow, and turbidity that result downstream from Lost Creek Dam.
2. Determine the effects of Lost Creek Dam and develop operational criteria as related to the rearing and migration of juvenile fall chinook salmon.
3. Determine the effects of Lost Creek Dam and develop operational criteria as related to the abundance, migration, and life history of adult fall chinook salmon.
4. Determine the effects of Lost Creek Dam and develop operational criteria as related to the harvest of fall chinook salmon.

We devised three types of comparisons to meet project objectives. First, we compared biological parameters of fall chinook salmon that inhabited the Rogue River and tributary streams. Second, we compared biological parameters of fall chinook salmon that inhabited the Rogue River before and after the operation of Lost Creek Dam. Third, we estimated the relationships between biological and physical factors to simulate biological responses to changes in physical factors influenced by the operation of Lost Creek Dam. Each method had associated strengths and weaknesses.

Fall chinook salmon inhabit tributaries of the Rogue River for only a short period of time. Fish originating from individual tributary streams could only be differentiated at time of spawning and at time of fry emergence from the gravel. Marking experiments indicated extensive intermingling of tributary and mainstem fish at other life history stages. Consequently, use of tributary streams for spatial comparisons among populations of fall chinook salmon were of limited utility.

Temporal comparisons were of greater utility. Sampling conducted prior to the operation of Lost Creek Dam provided information on the interannual variability of life history parameters. Sensitivity analyses after the initial years of the study led us to terminate work with algal and invertebrate communities. High variability among the data meant there was a low probability of associating any changes in production or community structure with the operation of the dam. Life history parameters of chinook salmon exhibited less variability.

However, temporal comparisons had some limitations. Given the expected variability, many years of data are required to make effective comparisons. We had only five years of returns for adults that reared as juveniles in the postimpoundment period. Although the dam was operational in 1977, low water yield produced negligible storage for flow augmentation in 1978. Reservoir releases had little effect on physical factors of the Rogue River in downstream areas. Consequently, we treated data from 1978 as the first postimpoundment year.

Comparisons of conditions in preimpoundment and postimpoundment years were confounded by changes in factors other than the treatment. For example, water yield from the basin differed significantly before and after operation of Lost Creek Dam. Consequently, we attempted to identify the factor(s) responsible for concomitant changes in parameters.

Identification of factors associated with changes in biological parameters was approached by correlation and regression analyses. We reviewed the literature for background information on causative relationships among biological and physical factors. Factors that appeared to be most important were included as independent variables in regression analyses. We used regression analysis to estimate relationships between biological and physical factors. Limitations and potential problems with this approach are discussed in detail (*see* **Analytical Procedures**, page 31).

Quantitative estimates are particularly useful if reliable predictions of impacts can be generated (Rosenberg et al. 1986). We developed a simulation model from some of the important findings. We used the model to evaluate the effects of varied strategies of reservoir management on the production and harvest of fall chinook salmon. We believe that simulations were also useful in identifying optimal strategies for reservoir management and operation of Lost Creek Dam.

However, there are problems associated with this method. In particular, small sample sizes, irrelevant independent variables, estimation beyond the range of observed data, measurement errors, and intercorrelation among the stages of the simulation model may lead to erroneous conclusions from simulations. Potential biases, and the means we adopted to minimize biases, are discussed in greater detail (*see* **Analytical Procedures**, page 31).

Despite problems related to simulation analysis, we believe it is a practical tool for judging effects of varied strategies for reservoir management. Also, our simulation model should provide resource mangers with a useful description of the life history of fall chinook salmon in the Rogue River basin. Finally, predictions can be used to develop additional experiments within the context of active adaptive management (Walters 1986). For example, reservoir outflow was modified after 1981 in an attempt to minimize prespawning mortality. Sampling in 1982-90 showed minimal loss to prespawning mortality. To improve reliability of predictions, the model should be updated as new information becomes available.

METHODS

We converted Gregorian calendar dates to day-of-year and week-of-year **(APPENDIX A)** for data analyses. We present some findings as calendar dates to clarify results.

Physical Factors

The United States Geological Survey (USGS) operated automated gages at numerous sites in the Rogue River basin during the project. USACE personnel used data from USGS gages in the Rogue River basin to estimate effects of operation of Lost Creek and Applegate dams on water quality parameters in downstream areas. Hamlin and Nestler (1987) described the development of a QUAL II model specific to the Rogue River basin.

The QUAL II model was used to simulate flow, water temperature, and turbidity for regulated and unregulated conditions. Regulated conditions simulated the Rogue River with Lost Creek and Applegate dams operating. Unregulated conditions simulated the Rogue River as though the dams had not been built. Simulations encompassed the time periods of January 1978 through September 1986 for flow and January 1978 through December 1986 for water temperature and turbidity. Model simulations produced estimates of daily means for physical factors at six gages operated by the USGS (Table 1). Operation of Lost Creek Dam affected water quality and quantity at all USGS gages in downstream areas. After November 1980, operation of Applegate Dam affected physical parameters of the Rogue River at gages downstream of the Applegate River.

We used results from USACE modeling to estimate the effect of dam operations on water quality and quantity in downstream areas used by anadromous salmonids. We received data for flow simulations from Rock Peters, USACE, Portland District, on 24 April 1989. We received data for water temperature and turbidity simulations from Carla Haake, USACE, Portland District, on 25 May 1989.

Table 1. Stations with water quality parameters simulated by USACE.

Station	RK	Parameter simulated
Near McLeod	249	Flow, water temperature, and turbidity
Dodge Bridge	224	Flow, water temperature, and turbidity
Raygold	201	Flow, water temperature, and turbidity
Grants Pass	165	Flow, water temperature, and turbidity
Marial	78	Water temperature and turbidity
Agness	48	Flow

Juveniles

We sampled juvenile chinook salmon in the Rogue River at 10 sites downstream of RK 190, 2 sites in the estuary, and in 2 tributary streams during 1974-81 (Table 2). During 1983-86, we sampled juveniles at two sites in the middle river.

We sampled juveniles with beach seines. In the middle river and in the canyon, we used a 50-foot by 8-foot floating seine with 1/4-inch square mesh, double leaded, attached to a "many ends" bottom line. In the lower river, we used a 100-foot by 8-foot floating seine with 1/4-inch or 3/8-inch square mesh. We used the net with smaller mesh when juveniles were smaller than 5 cm. When juveniles exceeded 5 cm, we used the net with larger mesh. In the estuary, we seined the Coast Guard site with a 350-foot by 17-foot floating seine that varied from 3/8-inch mesh in the bag to 1-inch mesh in the wings.

We seined each site weekly, except we seined sites in the canyon on alternate weeks. We made two sets at sites in the middle river and in the canyon, and one set at sites in the lower river. Catch rates of juveniles were calculated from standard sampling effort. At times we made extra sets to meet sampling goals for lengths, weights, and scales.

Table 2. Sites seined for juvenile fall chinook salmon in the Rogue River basin, 1975-86.

Sampling site	RK	Time period	Frequency	Years
Rogue River:				
Valley of the Rogue	183	Jan-Oct	weekly	1976-81
Matson Park	148	Jan-Oct	weekly	1975-86[a]
Almeda Park	116	Jan-Oct	weekly	1975-86[a]
Whiskey Bar	105	Apr-Sep	biweekly	1975-81
Winkle Bar	85	Apr-Sep	biweekly	1975-81
Illahe	56	Apr-Sep	biweekly	1975-81
Agness	44	Mar-Oct	weekly	1974-81
Hideaway	24	Mar-Oct	weekly	1974-81
Canfield	8	Mar-Oct	weekly	1974-82
Estuary:				
Mail Boat Point	3	Apr-Oct	weekly	1974-81
Coast Guard	1	Apr-Oct	weekly	1974-82
Tributaries:				
Applegate River[b]	1	Jan-Jul	weekly	1975-86
Illinois River	1	Mar-Oct	weekly	1976-81

[a] *Not sampled in 1982. Sampled during May-August, 1983-86.*
[b] *Sampled as part of the Applegate Dam Fisheries Evaluation, 1978-86.*

We segregated juvenile fish by species and age class. We anesthetized juveniles with benzocaine or a mixture of tricaine methanesulfonate (MS-222) and quinaldine (Schoettger and Steucke 1970). We measured fork lengths, to the nearest 1 mm, of 30 juveniles from each age class captured weekly at each site. We weighed, to the nearest 0.1 g, 25 subyearlings monthly at each site. We sampled scales from 20 subyearlings weekly at Canfield. We removed approximately 10 scales from an area 4 rows above the lateral line on the left side of juveniles, immediately posterior to the dorsal fin.

We mounted scale samples on glass slides with a solution of 5% glycerin and 95% sodium silicate. We mounted 10 scales per juvenile and analyzed 2 of the larger, non-regenerated scales of regular shape. Scales were measured at a magnification factor of 88. We measured and counted circuli in the anterior region of the scale along the longer of two lines at 20^0 angles from the longitudinal midline of the scale. We used these data to estimate relationships between (1) scale radius and fork length and (2) circuli number and calendar day of the year.

We estimated parameters of juvenile life history from the freshwater component of scales taken from adult fall chinook salmon. We recognize parameters estimated from adult scales may not represent life history parameters of all juveniles that reared in fresh water. For example, juvenile salmonids that grow at slower rates are susceptible to increased rates of predation (Parker 1971). Ward et al. (1989) found juvenile steelhead migrated from the Keogh River, British Columbia, at a mean length of 17.6 cm while scales from adult cohorts indicated migrants averaged 19.3 cm in length.

We present estimates of life history parameters for individual age classes and completed brood years. We previously reported values for brood years that were weighted by the abundance of each age class at time of freshwater entry (Cramer et al. 1985). However, because rates of natural and fishing mortality varied between years, scale parameters weighted for freshwater returns produced biases in estimates of life history composition. For example, age 5 fall chinook salmon would compose greater proportions of freshwater returns if not harvested at ages 3-5 in ocean fisheries.

To circumvent age selective mortality, we estimated the age composition of each brood year as if there had been no ocean harvest and as if the El Niño event of 1982-83 had not occurred. Sequentially, the steps were:

1. Estimate the number of age 2 fish resident in the ocean during spring (*see* **Ocean Abundance**, page 24).

2. Set rates of fishing mortality in the ocean to zero.

3. Use observed estimates of maturity rates to predict freshwater returns.

4. Assume a natural mortality rate of 0.2 during winter, except assume a mortality rate of 0.5 between ages 2 and 3.

Results from these procedures are in Table 3. We later present a more thorough discussion of methods used to estimate ocean abundance and maturity rates (*see* **Ocean Abundance**:, page 24 and **Age at Maturity**, page 27).

Table 3. Estimated age composition of fall chinook salmon that would have returned to the Rogue River had there been no ocean harvest and no El Niño event of 1982-83.

Brood year	Composition of freshwater return (%)				
	Age 2	Age 3	Age 4	Age 5	Age 6
1972	6.7	15.3	56.7	21.3	0.0
1973	3.6	6.1	20.6	69.6	0.0
1974	2.7	3.7	70.0	23.6	0.0
1975	2.8	7.2	47.5	37.1	5.5
1976	5.3	6.3	42.4	44.1	2.0
1977	4.2	8.6	55.5	31.6	0.0
1978	2.9	15.0	48.5	25.9	7.7
1979	3.4	8.3	55.5	32.8	0.0
1980	3.4	8.0	62.8	24.2	1.6
1981	4.3	25.9	52.7	17.1	0.0

Emergence Timing

We designated the date of first capture of chinook salmon fry as the date of first emergence from the gravel. We designated the date after which mean lengths continuously exceeded 4.5 cm as the date of emergence completion. We chose this criteria because succeeding surveys showed few fry exhibited yolk sacs and mean lengths of fry increased at a steady rate.

Abundance

We were unable to develop population estimates for juvenile fall chinook salmon that reared in the Rogue River. Instead, we used catch rates at Matson and Almeda in May-August as an index of juvenile abundance. We selected these sites as representative sample areas because they were close to major spawning areas. We selected the interval of May-August because subyearlings reared in the Rogue River consistently during that period.

We assumed catch per unit effort was a reliable index of year class strength. To be a reliable index, annual variations in catch rates should reflect annual variations in juvenile abundance. We assumed annual changes in site morphology, sampling efficiency, and fish behavior accounted for minimal variation in annual catch rates of juveniles.

We believe seine catches reflected juvenile abundance because chinook salmon inhabit midwater habitat characterized by low water velocity and substrate composed of small particles (Everest and Chapman 1972). Beach seines effectively sample juvenile chinook salmon in this type of habitat (Parsley et al. 1989).

Growth Rate and Body Size

We used three parameters as indexes of freshwater growth by subyearlings: (1) average spacing of freshwater circuli on adult scales, (2) mean lengths of seined juveniles, and (3) mean weight predicted from regressions of $\log_{10}$ weight on $\log_{10}$ length.

We analyzed, if possible, a minimum of 30 scales annually from each age class of adults. We measured the distance from the nucleus margin to a check that we assumed represented the point of ocean entry. We also counted the number of circuli on the freshwater portion of the scales. These data were used to estimate the average spacing between freshwater circuli. We estimated the relationship between average spacing of circuli and the rate of increase in length by laboratory experiments. Results showed that these two variables were highly correlated (Cramer and Martin 1978).

We predicted the mean length of juveniles at three sites (Matson, Agness, and Canfield) from regressions of mean length on day of sampling (Cramer et al. 1985). This methodology accounted for annual variations in migration timing because we consistently caught juveniles at these three sites.

Age, Timing, and Size at Migration

We examined scales from adult fall chinook salmon to determine juvenile age at ocean entry. We classified migrants as subyearlings or yearlings based on the presence or absence of a freshwater annulus.

We estimated date of ocean entry from a combination of data derived from scales of juveniles and adults. We used regression analysis to estimate annual relationships between calendar day of sampling (*see* **APPENDIX A**) and number of circuli on juvenile scales. We found circuli number accounted for an average of 62% of the variation in sampling date among eight regressions developed annually in 1974-81 (Table 4). Circuli numbers were also highly correlated with capture date of juvenile spring chinook salmon in the John Day River, Oregon (Lindsay et al. 1985).

We compared pooled regressions for preimpoundment and postimpoundment broods and found that slopes did not differ significantly ($P = 0.522$), although elevations differed significantly ($P < 0.001$). We used pooled regressions from preimpoundment broods to estimate dates of ocean entry for fish that reared in fresh water prior to 1974 and used pooled regressions for postimpoundment broods to estimate dates of ocean entry for fish that reared in the river after 1981. Use of pooled data to estimate relationships for non-sampled years may be inappropriate because slopes of annual regressions differed significantly among preimpoundment broods ($P = 0.031$) and postimpoundment broods ($P < 0.001$).

Circuli counts to the point of ocean entry on adult scales were inserted into regressions to estimate date of ocean entry. Parameters from adult scales represented fish that survived to return to the Rogue River and probably do not reflect scale parameters of juveniles that entered the ocean (Ward et al. 1989).

Table 4. Regressions of day-of-year on number of freshwater circuli from scales of juvenile chinook salmon seined at Canfield, 1973-80 brood years. Only scales with more than 10 circuli were included in the regressions.

Brood year	Regression[a]	Standard error	*N*	r^2	*P*
1973	Y = 79.6 + 8.71(X)	0.63	117	0.63	<0.001
1974	Y = 118.3 + 7.43(X)	0.31	194	0.75	<0.001
1975	Y = 86.4 + 9.39(X)	0.62	332	0.41	<0.001
1976	Y = 41.5 + 11.87(X)	0.56	407	0.52	<0.001
1977	Y = 76.9 + 7.83(X)	0.38	241	0.64	<0.001
1978	Y = 71.5 + 9.01(X)	0.37	356	0.63	<0.001
1979	Y = 70.9 + 8.02(X)	0.25	322	0.76	<0.001
1980	Y = 59.4 + 9.17(X)	0.38	366	0.61	<0.001
1973-75[b]	Y = 109.6 + 7.75(X)	0.26	643	0.57	<0.001
1977-80[b]	Y = 72.0 + 8.39(X)	0.18	1,285	0.64	<0.001

[a] *Y = day-of-year, X = number of circuli.*

[b] *Regressions used to predict date of ocean entry for preimpoundment and postimpoundment broods not sampled as juveniles.*

Also, scale parameters of seined juveniles may not reflect scale parameters of migrants that entered the ocean (Buckman and Ewing 1982). Juveniles seined relatively late in the season, particularly in early autumn, probably grew at slower rates compared with cohorts that migrated at earlier dates. Faster growing juveniles that migrated early would have exhibited fewer freshwater circuli. This type of bias would lead to underestimating the time of ocean entry by several weeks.

Alternatively, time of ocean entry could have been overestimated by several weeks. We believe the saltwater check does not develop on scales immediately upon ocean entry. We assumed the point of ocean entry was characterized by a transition from narrow circuli that were closely spaced to thicker circuli that were widely spaced. One or more evenly spaced circuli may be deposited prior to the formation of a saltwater check.

We concluded that although estimates for time of ocean entry may be biased one way or the other, comparisons of annual estimates of migration timing would remain valid, provided estimation errors were consistent among years.

We estimated juvenile length at ocean entry from a combination of data derived from scales of juveniles and adults. We used regression analysis to estimate annual relationships between fork length and scale radius for juveniles from the 1973-79 brood years. We found scale radius annually accounted for $\geq$ 85% of the variation in fork length for juveniles from the 1973-79 brood years (Table 5).

Table 5. Regressions of fork length on scale radius for juvenile chinook salmon seined at Canfield, 1973-79 brood years. Only data from fish 5-13 cm in length were included in the regressions.

Brood year	Regression[a]	Standard error	*N*	r^2	*P*
1973	Y = 2.93 + 0.178(X)	0.006	120	0.86	<0.001
1974	Y = 1.87 + 0.206(X)	0.003	260	0.93	<0.001
1975	Y = 2.67 + 0.193(X)	0.003	447	0.88	<0.001
1976	Y = 2.30 + 0.176(X)	0.003	497	0.88	<0.001
1977	Y = 2.29 + 0.188(X)	0.003	318	0.92	<0.001
1978	Y = 2.73 + 0.171(X)	0.004	390	0.85	<0.001
1979	Y = 2.24 + 0.181(X)	0.003	403	0.93	<0.001
1974-79[b]	Y = 2.52 + 0.181(X)	0.002	2,315	0.88	<0.001

[a] *Y = fork length (cm), X = scale radius (mm at 88X).*

[b] *Regression used to predict length at ocean entry for broods not sampled as juveniles.*

Analysis of pooled regressions for preimpoundment and postimpoundment broods showed no significant differences among slopes ($P = 0.825$) or elevations ($P = 0.087$). We used a pooled regression from all broods to estimate dates of ocean entry for fish that reared in fresh water prior to 1974 and after 1981. Use of pooled data to estimate relationships for years that were not sampled may be inappropriate because slopes of annual regressions differed significantly ($P < 0.001$).

Measurements of scale radii to ocean entry from adult scales were inserted into regressions to estimate length at ocean entry. Assumptions associated with this estimation procedure were comparable to assumptions previously discussed with estimation for time of ocean entry.

Adults

Abundance

Freshwater Return: We indexed freshwater returns of fall chinook salmon by beach seining in the lower river. We seined in the upper end of the estuary at RK 5 during 1974-75. We seined at Huntley Park (RK 13) in 1976-86, although low flow forced us to seine at Canfield (RK 8) in August 1977.

Seining began in early July and continued through late October. However, in 1974 we began sampling in early September. We seined 3-5 days weekly and effort ranged between 7 and 18 sets daily. Time of sampling varied annually. We sampled between 0900 and 1700 hours in 1974-76. During 1977-86, we began sampling 30 minutes after sunrise and usually finished by 1300 hours.

During 1976-86, we used a quick sinking beach seine 300 ft long with stretch mesh that varied from 2-inch for the center panel to 5-inch for the wing panels. Seine depth was 18 ft in the bag and tapered to 9 ft on outside edges of the wing panels. During 1974-75, we used two seines similar to the previously described seine, except one was 25 ft shorter and the other had only 3-inch stretch mesh.

We estimated freshwater returns of fall chinook salmon by determining the seining efficiency on coho salmon of hatchery origin. We estimated seining efficiency by comparing the seine catch of coho salmon with clipped adipose fins with the estimated number that entered the river. Freshwater return of marked coho salmon was estimated as the sum of (1) return to Cole M. Rivers Hatchery, (2) freshwater harvest, (3) prespawning mortality, and (4) strays that spawned in the wild. Estimation methods were described in detail by ODFW (1991a).

We assumed chinook salmon and coho salmon were equally vulnerable to capture by the beach seine. We know of no evidence that would either confirm or refute this assumption. Even if catchability differed between species, findings from our analyses would remain valid if bias was consistent among years. We recognize that variations in physical parameters of the Rogue River could differentially affect vulnerability of capture for each species.

We evaluated effects of seining time on catch rate of fall chinook salmon by comparison of bi-hourly catch rates within months. We ranked catch rates from six years of seining and used Friedman's Test (Zar 1984) to look for differences in catch rates among two-hour periods. We found catch rates did not differ, except in July (Appendix Table C-1). Because few fall chinook salmon entered the river in July, we concluded it was appropriate to compare monthly catch rates among years, 1976 excepted.

We excluded 1976 data from these analyses because time of seining differed from time of seining in later years. For example, we made 38% of the sets after 1300 hours in September 1976. In six succeeding years, sets after 1300 hours accounted for an average of only 3% of the effort in September. Seining time in July, August, and October 1976 also differed from seining time in later years. We only used catch rate data from 0600-1300 hours to make data from 1976 comparable to data from other years.

We multiplied weekly catch rates (Appendix Table C-2) by 45 to generate a standardized weekly catch. We selected 45 as a standard because this was the most common number of sets made weekly. We used weekly estimates of seining efficiency (ODFW 1991a) to expand standardized catches to estimate freshwater returns of fall chinook salmon on a weekly basis (Appendix Table C-3). We used the equation

$$\hat{E} = C/(e/100) \qquad (1)$$

where

$\hat{E}$ = estimated weekly return of fall chinook salmon to the Rogue River,
C = standardized seine catch based on 45 sets weekly, and
e = weekly estimate of seining efficiency (%).

We concluded that chinook salmon seined during 9-22 July should be classified as the spring race and those seined at later dates should be classified as the fall race. We commonly caught marked fish known to be spring chinook salmon during 9-22 July, but did not catch such fish after 22 July. Also, weekly estimates of freshwater returns usually decreased in late July, suggesting the end of the run of spring chinook salmon. The nadir in estimated passage of chinook salmon at Huntley Park averaged calendar week 30 (23-29 July) of the year (95% CI = ± 1 week) in 1976-86.

We estimated adults took an average of 26 days to reach the counting station at Gold Ray Dam based on an average migration rate of 7.3 km per day (Cramer et al. 1985). Because the separation date is 15 August for spring and fall chinook salmon that pass Gold Ray Dam, 22 July (the last day in calendar week 29, *see* **APPENDIX A**) appeared to be an appropriate date for segregating races of chinook salmon seined at Huntley Park.

We estimated freshwater returns of fall chinook salmon in 1974-75 from the number of spawned carcasses recovered in four survey areas. We found a significant relationship ($r^2 = 0.71$) between freshwater returns and carcass counts in 1976-85 (Figure 4). Prior to analysis, we adjusted the carcass count from 1978 because we estimated 12.5% of the run died prior to spawning (*see* **Prespawning Mortality**, page 28). Also, we excluded years when mortality rates exceeded 20%. We estimated freshwater returns of 42,656 in 1974 and 37,175 in 1975 based on carcass counts of 4,194 in 1974 and 3,359 in 1975.

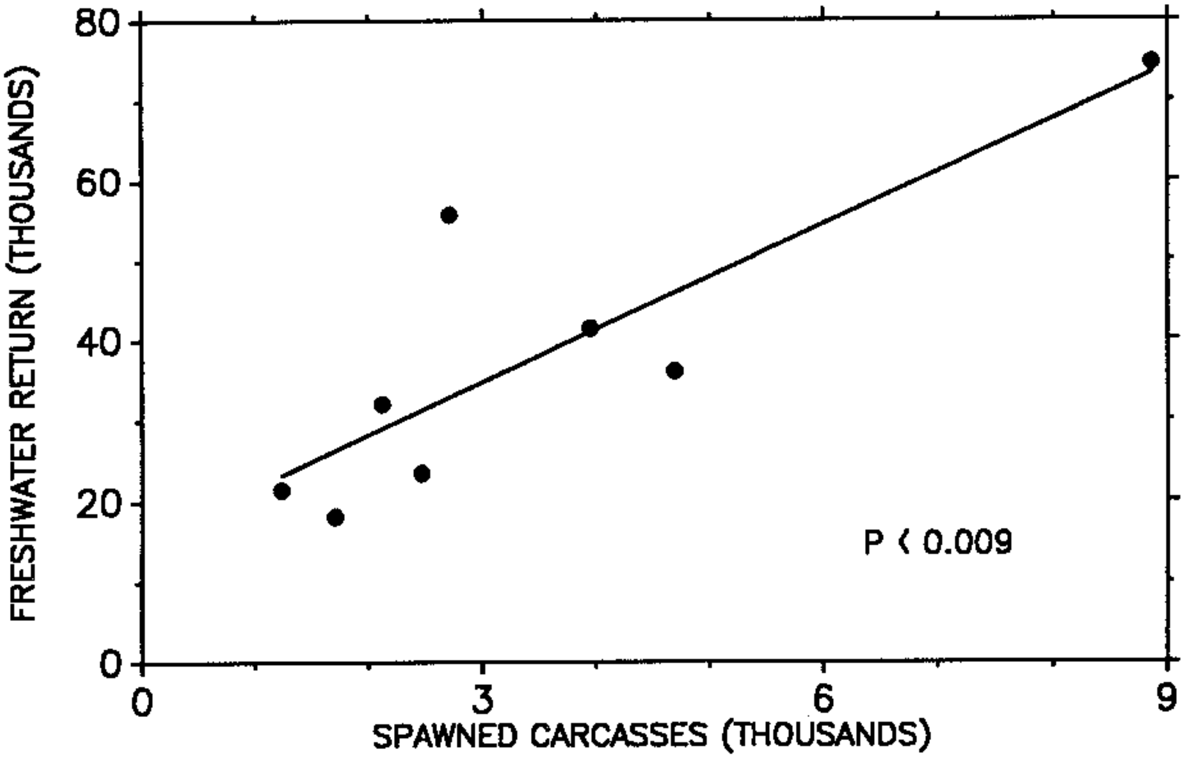

Figure 4. Relationship used to estimate freshwater returns of fall chinook salmon during 1974-75. Data from 1979-80 excluded because of extensive prespawning mortality.

This approach seemed plausible because we previously found that recoveries of spawned carcasses were highly correlated with returns of spring chinook salmon to Gold Ray Dam (Cramer and McPherson 1981). Similarly, population estimates of fall chinook salmon that entered Elk River accounted for 84% of the variation in counts of spawned carcasses within standard survey areas (memorandum dated 7 May 1984 from Timothy Downey, ODFW, Port Orford, Oregon). However, we were not able to determine whether carcass counts in the areas that we surveyed were representative of spawning escapement throughout the basin.

Ocean Abundance: We used cohort analysis (Ricker 1975) to estimate the number of wild fall chinook salmon that resided in the ocean during spring. Estimation procedures began with age 6 fish and ended with age 2 fish. At each age, cohorts in the ocean were estimated as the sum of (1) the number of fish that resided in the ocean during the succeeding year, (2) natural mortality, (3) the number of fish that returned to the river, and (4) harvest in the ocean fisheries. For each cohort, we used the equation

$$N_i(t) = (N_{i+1}(t+1)/(1 - A_i) + E_i(t))/\ 1 - u_i(t) \qquad (2)$$

where

$N_i(t)$ = number of age i fish resident in the ocean prior to fishing and freshwater return during year t,
$N_{i+1}(t+1)$ = number of age i+1 fish resident in the ocean during the next year,
A_i = rate of natural mortality for age i cohorts resident in the ocean after the fishery, between years t and t+1,
$E_i(t)$ = freshwater return of age i fish during year t, and
$u_i(t)$ = exploitation rate of age i fish in the ocean during year t.

Estimates of cohort abundance began with age 6 fish because fall chinook salmon that returned to the Rogue River matured at age 6 or younger. We made no attempt to estimate sources of error associated with cohort reconstruction. We were unable to reconstruct all broods because some cohorts remained in the ocean after termination of sampling in 1986. More thorough discussions of cohort analyses can be found in Pope (1972), Ricker (1975), Hankin and Healey (1986), Kope (1987), and Hankin (1990).

We assumed natural mortality occurred after termination of ocean fisheries and maturation of a portion of the cohort. Ricker (1975) termed this scenario as a type 1 fishery. We also assumed mature fish entered the river only after termination of ocean harvest. This assumption was violated to some extent for fall chinook salmon that originated from the Rogue River basin. We assumed a 50% rate of natural mortality in the ocean for age 2 fish and a 20% rate of natural mortality for older fish (Ricker 1976; Healey and Heard 1984).

We assumed ocean fisheries harvested fall chinook salmon of Rogue River origin at the same rates as fall chinook salmon of the same age that originated from hatcheries in the Klamath River basin of northern California (*see* **Ocean Harvest**, page 28). Ocean exploitation rates of fall chinook salmon that originated from the Klamath River basin were estimated by the Klamath River Technical Advisory Team (PFMC 1991). We also assumed ocean fisheries harvested no age 2 fish.

We estimated non-harvest fishing mortality (shaker mortality) with information supplied by the Klamath River Technical Advisory Team (telephone conversation on 27 May 1987 with Steven Cramer, ODFW, Corvallis, Oregon). Mortality of age 3 shakers was estimated as the ocean catch of age 3 fish multiplied by (1) 0.2 to account for fish released because they were too small to be legally retained and (2) 0.3 to account for mortality of released fish.

Mortality of age 2 shakers was estimated as the number of age 2 fish resident in the ocean multiplied by the contact rate and the mortality rate for released fish. Contact rate was estimated as 40% of the exploitation rate on age 4 adults in the same season. The assumed mortality rate of 30% for released age 2 and age 3 chinook salmon was similar to the 25% mortality rate reported by Wertheimer (1988).

The El Niño event of 1982-83 increased complexity of the cohort analysis. Fish resident in the ocean during El Niño grew at slower rates compared with counterparts that reared in the ocean during earlier years (Satterthwaite et al. 1985). We also found body condition of mature fish was less in 1983 compared with prior years. Adults that averaged 80 cm in length weighed an average of 7.4 kg (95% CI = ± 1.4 kg) in 1974-76, but averaged only 6.6 kg (95% CI = ± 2.0 kg) in 1983. Freshwater returns of fall chinook salmon in the Rogue River during 1983 and 1984 were also much lower than expected. These findings suggested that rates of natural mortality probably increased as a result of the El Niño event of 1982-83.

However, estimation of changes in mortality rates for chinook salmon was difficult to quantify because maturity rates may have also changed. Delayed maturity could account for freshwater returns that are less than predicted for chinook salmon, but sampling of freshwater returns in years after the El Niño event of 1982-83 did not suggest fish matured at older ages. These findings led us to conclude that rates of ocean mortality increased for fall chinook salmon of Rogue River origin as a result of the El Niño event of 1982-83.

We compared estimates and predictions of cohort abundance in the ocean to estimate changes in mortality rates that resulted from the El Niño event of 1982-83. Predictions were generated from regressions of ocean abundance on freshwater returns of cohorts in the previous year. For each age class, freshwater returns in 1982 and 1983 were substituted into regressions to predict cohort abundance in the ocean during 1983 and 1984. We used catch rates of juveniles seined in the Rogue River during the previous year to predict the ocean abundance of age 2 fish.

This methodology required assumption of constant rates of maturity for affected broods. Maturity rates affected the number of adults that returned to the river, the independent variable in the regressions. This assumption may be reasonable for adults that returned to the river in 1982 (used to predict ocean abundance in 1983). However, decreased rates of ocean growth in 1983 may have affected the proportion of fish that matured and returned to the river (used to predict ocean abundance in 1984) because size at age is positively correlated with maturity rate (Hankin 1990).

Despite this problem, we believe estimates of ocean abundance were less affected by errors in estimates of maturity rates compared with errors in estimates of survival rates. We also believe adjustment of survival rates for fish in the ocean resulted in more accurate estimates of fish abundance in the ocean prior to the El Niño event of 1982-83 compared with the alternative assumption of constant rates of survival.

Spawning Escapement: We estimated spawning escapement of fall chinook salmon by subtracting estimates of prespawning mortality and freshwater harvest from estimates of freshwater returns. We segregated age 2 jacks from older adults based on the lengths of fish seined at Huntley Park. Scale analyses indicated fish smaller than 60 cm were age 2 jacks, except in 1983-84 when 53 cm was an appropriate criteria to segregate jacks and older adults.

We also surveyed some areas for spawned carcasses of fall chinook salmon. During 1974-85, we counted spawned carcasses in four survey areas (RK 139-156 and RK 179-183 in the Rogue River, RK 0-6 in the Applegate River, and Slate Creek). We believe these areas, among the areas we sampled (Table 6), best reflected the spawning abundance of fall chinook salmon in the Rogue River basin. Sampling representative areas for spawned Pacific salmon results in good indexes of abundance for at least some streams (Cousens et al. 1982; Shardlow et al. 1987). However, we were not able to determine if carcass counts in the areas we surveyed were representative of spawning escapement throughout the basin.

We used driftboats to survey each side of the Rogue River on alternate weeks. However, we surveyed weekly in areas where carcasses congregated. Tributaries were surveyed weekly using rafts or were surveyed on foot, depending on the flow. We cut carcasses in half to avoid replicate counts of carcasses during succeeding surveys.

Table 6. Areas surveyed for spawned fall chinook salmon in the Rogue River basin, 1974-85.

Stream, area	Period	Years
Rogue River:		
RK 199-197	10/06-11/14	1974-81
RK 183-179	10/10-11/14	1974-85[a]
RK 169-158	10/15-11/23	1974-81
RK 156-139	10/19-11/28	1974-85
Applegate River:		
RK 17-13	10/15-12/21	1974-85[b]
RK 6-0	10/15-01/11	1974-85[b]
Evans Creek	10/19-11/28	1974-81
Slate Creek	10/19-01/24	1974-85[b]
Quosatana Creek	11/19-01/29	1974-81
Lobster Creek	11/14-01/19	1974-81

[a] *Not sampled in 1982.*
[b] *Sampled as part of the Applegate Dam Fisheries Evaluation, 1978-85.*

We also estimated spawning escapement of fall chinook salmon that passed Gold Ray Dam. We assumed all fish that passed the counting station survived to spawn because there was minimal prespawning mortality and no freshwater harvest of fall chinook salmon upstream of Gold Ray Dam.

Passage of chinook salmon at Gold Ray Dam has been estimated since 1942. Fish were counted eight hours daily, five days weekly. Partial counts were designed to estimate biweekly passage with an average error less than 10% (Li 1948). During 1942-67, fish were counted as they passed above a white board. Since 1968, fish have been counted as they passed an underwater viewing window. Samplers also recorded fin clips and classified fish by size. Samplers classified chinook salmon as jacks if fish were less than 20 inches (50 cm) in 1942-77 or less than 24 inches (60 cm) in 1978-86.

Age at Maturity

We collected scales from adults seined in the lower river to estimate the age composition of fall chinook salmon. We assumed no difference in catchability for adults of different ages. This assumption would not be appropriate if the seine captured fish in a size selective manner.

We collected scale samples from at least 35 fish, when possible, within each 5-cm size interval. Collection rates within common size intervals were limited to five scale samples weekly. We removed about 20 scales from each fish we sampled. Scales were taken immediately above the lateral line and immediately posterior to the dorsal fin. Four of the larger, non-regenerated scales of regular shape were mounted on gummed cards and impressed on acetate at 100°C under 5,000 psi for 3 minutes.

Scales were measured at a magnification factor of 88. We measured and counted circuli in the anterior region of each scale along the longer of two lines at 20° angles from the longitudinal midline of the scale. Scale radius to the nucleus margin, annuli, point of ocean entry, and outer edge of the scale were measured to the nearest 0.5 mm. Circuli were counted from the nucleus to points of annuli, ocean entry, and the distal edge of the scale. We used the terminology of Gilbert and Rich (1927) to present the age of fall chinook salmon. Age indicates the year of life at time of freshwater entry.

Scale analyses indicated that fall chinook salmon matured at ages 2-6. Consequently, we assumed maturity probability at age 6 was 1.0. Maturation probabilities for other age classes were estimated as

$$Q_i = E_i/S_i \tag{3}$$

where

Q_i = probability of maturation at age i,
E_i = freshwater return of maturing fish at age i, and
S_i = fishery survivors in the ocean at age i prior to freshwater return of maturing fish $((N_i(t)*(1-u_i(t))$-shaker mortality).

We were unable to distinguish sex of fall chinook salmon at time of scale removal, except we assumed that age 2 jacks were all male.

Ocean Harvest

We estimated ocean harvest of fall chinook salmon that originated from the Rogue River basin from cohort analyses described in **Ocean Abundance:**, page 24. Estimates of annual harvest were the product of age-specific estimates of cohort abundance in the ocean prior to opening of the fisheries and estimates of age-specific exploitation rates for fall chinook salmon that originated from hatcheries in the Klamath River basin of northern California.

Ideally, exploitation rates in ocean fisheries are estimated from wild juveniles tagged in natal streams. We were not able to directly estimate ocean exploitation rates because we did not tag wild juvenile fall chinook salmon. Nor were we able to use data from hatchery fall chinook salmon native to the Rogue River basin because few groups of tagged fish were released.

However, tag recoveries from those releases indicated fall chinook salmon that originated from the Rogue River basin contributed to the same ocean fisheries as fish from the Klamath River basin (Fraidenburg and Lincoln 1985). A similar distribution of landings suggested fall chinook salmon from both basins may be harvested at similar rates in the ocean fisheries. Hankin (1990) found similar age-specific exploitation rates for immature age 4 spring chinook salmon from the Rogue River and age 4 fall chinook salmon from the Klamath River basin.

A comparable analysis of ocean exploitation rates for age 3 chinook salmon produced different results (Hankin 1990). Age 3 fall chinook salmon that originated from Trinity River Hatchery in the Klamath River basin were often smaller than the size for legal harvest in ocean fisheries. Because age 3 fall chinook salmon that originated from the Rogue River basin were larger than some counterparts that originated from hatcheries in the Klamath River basin, we may have underestimated the ocean harvest of age 3 fall chinook.

Also, ocean fisheries harvested few age 5 or age 6 fall chinook salmon that originated at hatcheries in the Klamath River basin. The small numbers of tag recoveries made it impossible to estimate exploitation rates of these older age classes in the ocean fisheries. Consequently, we assumed age 5 and age 6 fish were harvested in the ocean at the same rate as age 4 fish.

Migration Timing

We tagged fall chinook salmon seined in the lower river during 1974-78. We used individually numbered Floy t-bar tags, except we used individually numbered Petersen disc tags in 1976. Floy tags were inserted so the t-bar wedged between the pterygiophore bones below the dorsal fin.

Prespawning Mortality

We surveyed the Rogue River canyon on alternate weeks from 1 August through 15 October in 1975-86. During periods of extensive mortality, we surveyed weekly. We also surveyed two other areas (RK 6-48 and RK 134-158) of the Rogue River weekly in 1975-81. We classified carcasses as age 2 jacks or older adults and examined carcasses for tags and fin clips.

We cut carcasses in half during 1975-79 to avoid counting mortalities more than once. In 1979, we evaluated the need to cut carcasses. Results indicated carcasses disappeared or decomposed prior to surveys in succeeding weeks. During 1980-86, we counted only those carcasses that appeared freshly dead, or where the majority of flesh was present.

To make survey results comparable between years, we interpolated data to estimate carcass counts during weeks not surveyed. Because we counted few carcasses during surveys on alternate weeks compared with weekly surveys, expansion of data for weeks not surveyed did not appreciably change carcass counts for years with low mortality.

We used the relationship between estimates of freshwater returns and carcass counts in standard spawning areas (*see* **Freshwater Return:**, page 21) to estimate rates of prespawning mortality in the Rogue River during 1979-81. Based on estimates of freshwater returns of 69,730 in 1979, 33,478 in 1980, and 41,420 in 1981, we predicted surveyors should have recovered 8,248, 3,401, and 4,463 spawned carcasses respectively (Figure 5). Because surveyors recovered 1,543, 1,474, and 3,944 spawned carcasses in those years, we estimated mortality rates of 81.3%, 56.6%, and 11.6%, respectively.

We indexed mortality rates by dividing counts of unspawned carcasses in the canyon by freshwater returns (Table 7). A comparison of mortality rate estimates and mortality indexes suggested there was a positive relationship

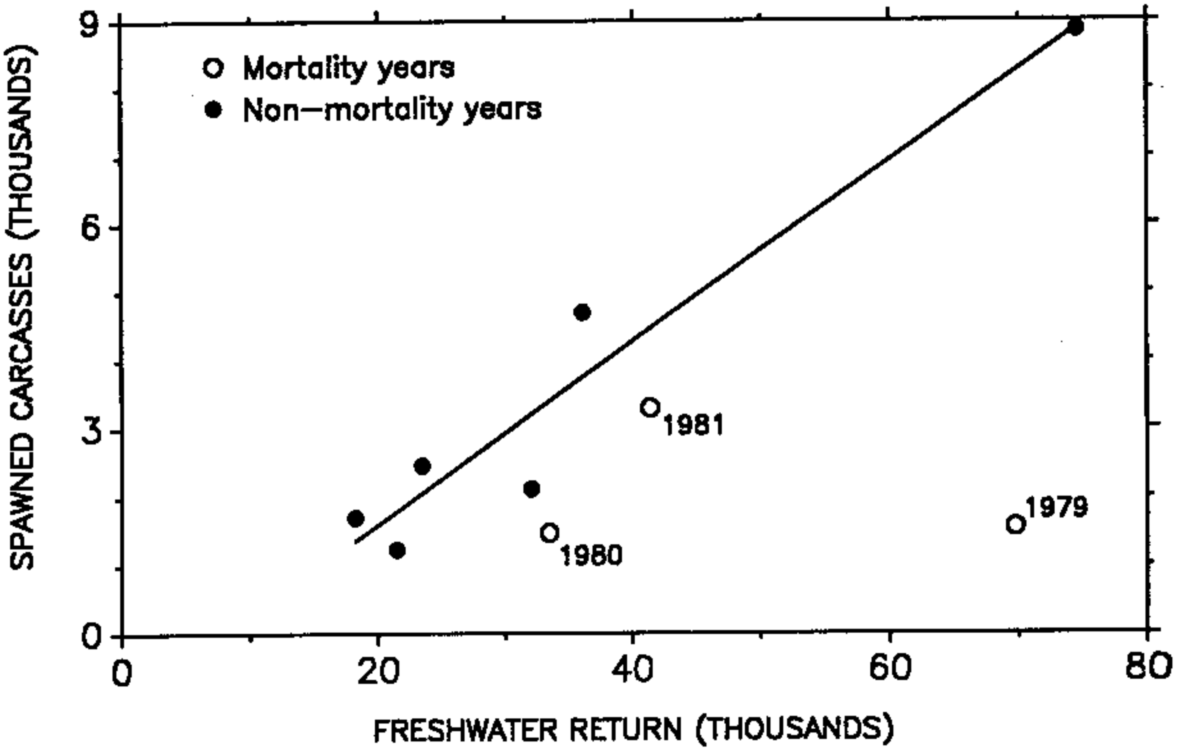

Figure 5. Relationship used to estimate rates of prespawning mortality of fall chinook salmon during 1979-81.

Table 7. Rate of prespawning mortality estimated for fall chinook salmon in the Rogue River, 1975-86.

Year	Carcass count[a]	Freshwater escapement	Mortality index[b]	Mortality rate (%)
1975	21	37,175	0.56	0.8[c]
1976	11	23,469	0.47	0.7[c]
1977	14	32,038	0.44	0.6[c]
1978	665	74,575	8.92	12.5[c]
1979	4,455	69,730	63.89	81.3
1980	936	33,478	27.96	56.6
1981	280	41,420	6.76	11.6
1982	160	55,735	2.87	4.0[c]
1983	2	21,464	0.09	0.1[c]
1984	5	18,212	0.27	0.4[c]
1985	15	36,109	0.42	0.6[c]
1986	42	98,314	0.43	0.6[c]

[a] *Unspawned carcasses found in the Rogue River canyon, 1 August - 15 October.*
[b] *(Count of unspawned carcasses/freshwater escapement) x 1,000.*
[c] *Estimated by regression of mortality index on mortality rate, 1979-81.*

between the two variables, although we had only three years (1979-81) of data. We calculated a regression through the origin to estimate the relationship between mortality index and mortality rate. The slope of this regression indicated mortality rates could be estimated by multiplying mortality indexes by 1.396. We used this result to estimate rates of prespawning mortality in 1982-86 (Table 7).

The 1978 count of spawned carcasses used in Figure 5 (8,860) and the 1978 estimate of mortality rate in Table 7 represents an adjustment to the count of spawned carcasses based on the mortality index for 1978. We deemed the adjustment to be necessary because surveys of unspawned carcasses in the canyon showed there was substantial prespawning mortality in the Rogue River during 1978.

Freshwater Harvest

We assumed harvest estimates derived from volitional returns of salmon-steelhead cards to ODFW were unbiased estimates for angler harvest of fall chinook salmon in the Rogue River. This assumption is likely erroneous because the adjustment factor used to account for non-response bias is appropriate only for an estimate of statewide harvest (Hicks and Calvin 1964). However, we believe harvest estimates from salmon-steelhead cards are reasonable data because the Rogue River is fished by a large number of anglers from throughout the region.

We estimated harvest rate in the river fishery by dividing harvest in August-October by freshwater returns of adults that were larger than the size of fish required to be recorded on the salmon-steelhead cards. We did not include harvest in November-December within estimates of harvest rate because these fish likely entered the Rogue River after we terminated sampling at Huntley Park. Most of the freshwater harvest in November-December occurs in the lower river (discussion on 15 May 1987 with Arvo Riikula, ODFW, Gold Beach, Oregon).

We assumed there was no difference in harvest rate of small adults not recorded on salmon-steelhead cards and large adults recorded on salmon-steelhead cards. This assumption may be erroneous. Surveys of other Oregon fisheries indicated catchability coefficients of jacks exceeded those of older fall chinook salmon (Nicholas et al. 1985; Boecheler and Jacobs 1987).

Spawning Distribution and Timing

Survey methods for spawned carcasses were previously described (*see* **Spawning Escapement:**, page 26). We indexed spatial distribution of spawners as the proportion of carcasses counted within each survey area. Also, we estimated the proportion of the run that spawned upstream of Gold Ray Dam (*see* **Spawning Escapement:**, page 26).

We indexed spawning time from weekly recoveries of spawned carcasses. We assumed surveyors recovered carcasses two weeks after spawning. We based this assumption on a postspawning longevity of nine days for coho salmon (van den Berghe and Gross 1986) and a postmortem period of five days until recovery by a surveyor (ODFW 1991b). We assumed carcasses were recovered at the site of spawning. This assumption appears reasonable for coho salmon in small streams (Cederholm et al. 1989), but may not be appropriate for chinook salmon in large streams.

Simulation Model

We used some findings to develop a stochastic simulation model of fall chinook salmon production in the Rogue River basin. The model is described in **Model Structure**, page 87.

We evaluated model sensitivity to changes in values of input variables. First, we ran a simulation with mean values for input parameters. Means were determined from empirical data gathered during the project. Second, we ran an array of simulations with mean values for all input parameters, although we progressively changed the mean value of one input parameter by ± 10%. We indexed model sensitivity as

$$S = Y(x_h)/Y(\bar{x}) \tag{4}$$

where

S = sensitivity of output to changes in values of input parameters,
$Y(x_h)$ = model output with a 10% change of one input parameter, and
$Y(\bar{x})$ = model output with mean values for all input parameters.

Evaluation of model behavior proved difficult. We were unable to compare predicted and observed values because (1) the model was only applicable to postimpoundment years and (2) we could not account for the increase in the rate of natural mortality during and just after the El Niño event of 1982-83.

Instead, we evaluated the model by estimating the precision associated with predicted output. To estimate output precision, we developed 95% confidence intervals from 500 simulations with one set of input parameters. We discarded the lowest 2.5% of the simulation outputs to designate the lower confidence boundary. We discarded the largest 2.5% of the simulation outputs to designate the upper confidence boundary.

A series of 11 regressions compose the model, including one multiple regression. Predicted output from each regression was allowed to vary stochastically within a normal distribution described by the estimated regression prediction and the estimated variance associated with that prediction:

$$Y_{ij} = Y_i + (\sigma_i) * (r_{ij}) \tag{5}$$

where

i = regression equation (1,...11),
j = simulation run (1,...500),
Y_{ij} = predicted output of regression i adjusted by random number j,
Y_i = predicted output of regression i with a x_i input value,
σ_i = variance of Y_i predicted from regression i, and
r_{ij} = random number j from a standard normal distribution used in regression i.

We estimated the variance of the predicted response from regressions with a single independent variable as

$$\sigma_i^2 = MSE_i * ((1/n_i) + (x_i - \bar{x}_i)^2/SS_i) \tag{6}$$

where

MSE_i = residual mean square for regression i,
n_i = sample size of data used to develop regression i,
x_i = value of independent variable for each of 500 simulations,
$\bar{x}_i$ = mean of independent variable used to develop regression i, and
SS_i = corrected sum of squares of independent variable x_i used to develop regression i.

We estimated the variance of the predicted response from the multiple regression in the model with procedures described by Neter and Wasserman (1974).

Analytical Procedures

Data we believed to exhibit a normal distribution were analyzed with parametric statistics, primarily using Microstat statistics software (Ecosoft Inc., Release 4.1). Data with distributions judged to be other than normal were assessed with nonparametric statistics.

Because many of the data sets contained less than 10 observations, the assessment of normality was frequently subjective. Uncertainty about the normality of the data led us to defer testing for homogeneity of variances. In general, we used $P \leq 0.05$ as the criteria for statistical significance. If sample sizes were small, we used $P \leq 0.10$ as a significance level. We followed analytical methods outlined by Snedecor and Cochran (1967) and Zar (1984).

Parametric methods most commonly used included analysis of variance, correlation analysis, and regression analysis. We used analysis of variance to test for differences between means of preimpoundment and postimpoundment variables. Where no difference was noted, we calculated the minimum detectable difference (Zar 1984) to estimate how much the postimpoundment mean would have had to change in order for the change to be detected. Proportional or percentage data were arcsine transformed prior to analysis of variance.

We used a two factor analysis of variance to test for differences between means of life history parameters for different age classes among multiple brood years. Where differences were noted, we used a Newman-Keuls multiple range test to evaluate differences between specific age classes.

To identify relationships among variables, we used correlation analysis and assumed data were independent observations with a bivariate normal distribution and common variance. We also used correlation analysis primarily to evaluate potential multicollinearity among independent variables considered for inclusion in multiple regression analyses. Percentage or proportional data were logit transformed prior to regression analysis.

To quantify relationships between dependent and independent variables, we used regression and multiple regression analyses. Independent variables were assumed to be measured without error. This may be a reasonable assumption for measurements of physical factors (such as flow and upwelling), but is certainly erroneous for some biological data. Associated errors were probably smallest for life history parameters reported as means (such as length at ocean entry, and scale measurements). Estimates or indexes of fish abundance almost certainly contained major sources of error, particularly where numerous estimation steps and assumptions were required to derive the data.

However, because fish abundance is of key importance to this evaluation, and other analytical procedures may be less robust, we used regression analysis to estimate the quantitative relationships between variables. Independent variables were included in regression analyses only when our previous findings (Cramer et al. 1985) or results from other research identified variables as probable causal factors associated with the dependent variable in question. Despite this step, there are probably multiple specification errors among regressions we present in this report.

Other potential problems associated with multiple regression analysis include autocorrelation, heteroscedasticity, and multicollinearity. We evaluated the potential for autocorrelation by the Durbin-Watson test. We evaluated the potential for heteroscedasticity by plotting residual values from a regression on the associated values of the dependent variable. We attempted to minimize the potential for multicollinearity by (1) use of correlation analysis to identify relationships between independent variables and (2) limiting the number of independent variables included in regressions.

We used regressions to predict values of a dependent variable from values of independent variable(s). Ricker (1973) and Jensen (1986) recommend use of predictive regression rather than functional regression if the objective is prediction rather than description of functional relationships. We chose predictive regression because our primary objective was to predict the response of dependent variables to variations in independent variables.

RESULTS AND DISCUSSION

Physical Factors

In 1988, USACE personnel simulated flow, water temperature, and turbidity for regulated (with dams) and unregulated (without dams) conditions in 1978-86. In this section of the report, we summarize some of the findings relevant to the production and harvest of fall chinook salmon in the Rogue River.

Flow

Operation of Lost Creek Dam affected flow in downstream areas. Storage of inflow occurred primarily from January through April and peaked in February (Figure 6). The reservoir reached full pool each year, usually by 1 June. Augmentation of natural flow usually began in the middle of June, peaked in July or August, and continued through November (Figure 6).

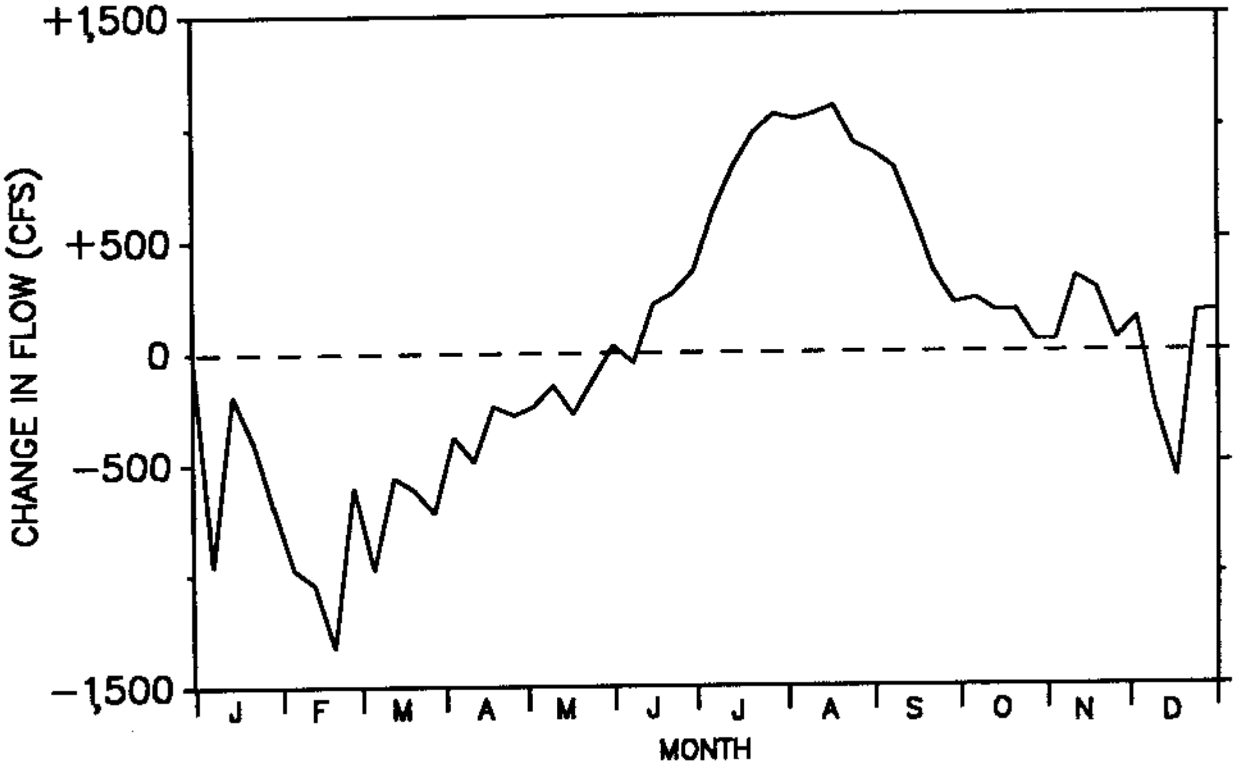

Figure 6. Change in mean weekly flow caused by the operation of Lost Creek Dam, 1978-86. The zero-line represents unregulated flow (inflow = outflow).

On a proportional basis, the effect of reservoir operation decreased with distance downstream (Figure 7). Regulated flow at Grants Pass generally ranged between 4,000 cfs and 8,000 cfs in January-April. At the same time, regulated flow downstream at Agness usually ranged between 7,000 cfs and 10,000 cfs. As tributary flow declined in late spring and early summer, flow in the lower river became similar to flow in the middle river.

Operation of Lost Creek Dam decreased the intensity of peak flows in downstream areas. Annual reductions in peak flows at Grants Pass averaged 6,300 cfs and ranged from 200 cfs to 10,900 cfs (personal communication dated 24 April 1989 from Rock Peters, USACE, Portland, Oregon).

Water Temperature

Operation of Lost Creek Dam increased water temperature in November-January and decreased water temperature in June-September throughout the river (Figure 8). Regulation increased average water temperatures at Grants Pass and Marial by about 1.0°C in November-January. During the thermal peak in summer, regulation reduced average water temperatures at Grants Pass and Marial by 3.4°C and 3.2°C, respectively.

Turbidity

Operation of Lost Creek Dam generally reduced turbidity in downstream areas. Regulation reduced average turbidity at Grants Pass by 5-8 Jackson Turbidity Units (JTU) in April-June and by 1-2 JTU in July-October (Figure 9). Operation of the dam did not appreciably affect turbidity in the middle river during freshets. Tributary streams increased turbidity in the mainstem, particularly when tributary flows increased after periods of high precipitation.

Juveniles

Emergence Timing

Fry of fall chinook salmon usually began to emerge from the gravel in early March in the Rogue River basin. First emergence at specific locations varied from late February to early April (Appendix Table B-1). Emergence completion usually occurred in late April, but ranged between late March and late May (Appendix Table B-2). Fall chinook salmon fry emerged at a similar time in other rivers within Oregon (Reimers 1973; Jonasson and Lindsay 1988).

Water temperature is a major determinate of development rate of eggs and alevins of chinook salmon (Alderdice and Velsen 1978; Heming 1982; Beacham and Murray 1990). Simulations of water temperature by the USACE suggested that reservoir operation affected the emergence timing of fall chinook salmon fry in the middle river. Simulations indicated mean daily water temperature at Grants Pass increased by an average of 0.75°C from 15 October through 25 March in 1977-78 through 1985-86. From USGS gage data, we found that mean maximum water temperature in November-March at Grants Pass averaged 6.3°C in 1973-74 through 1976-77 and 7.0°C in 1977-78 through 1985-86. The 0.7°C difference in means was significant at $P = 0.057$.

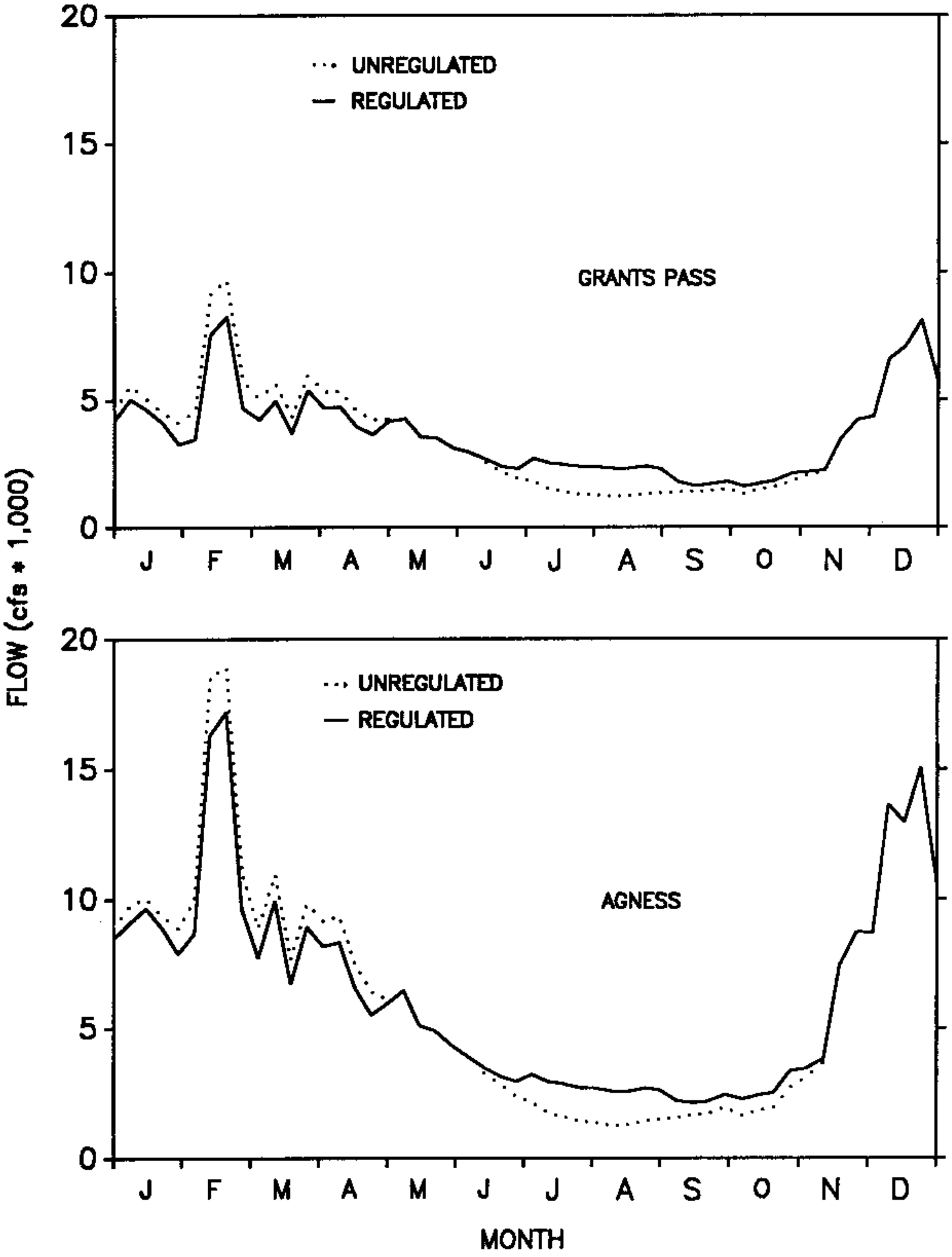

Figure 7. Mean weekly flow in the middle river at Grants Pass, and in the lower river at Agness, simulated for regulated and unregulated conditions during 1978-86.

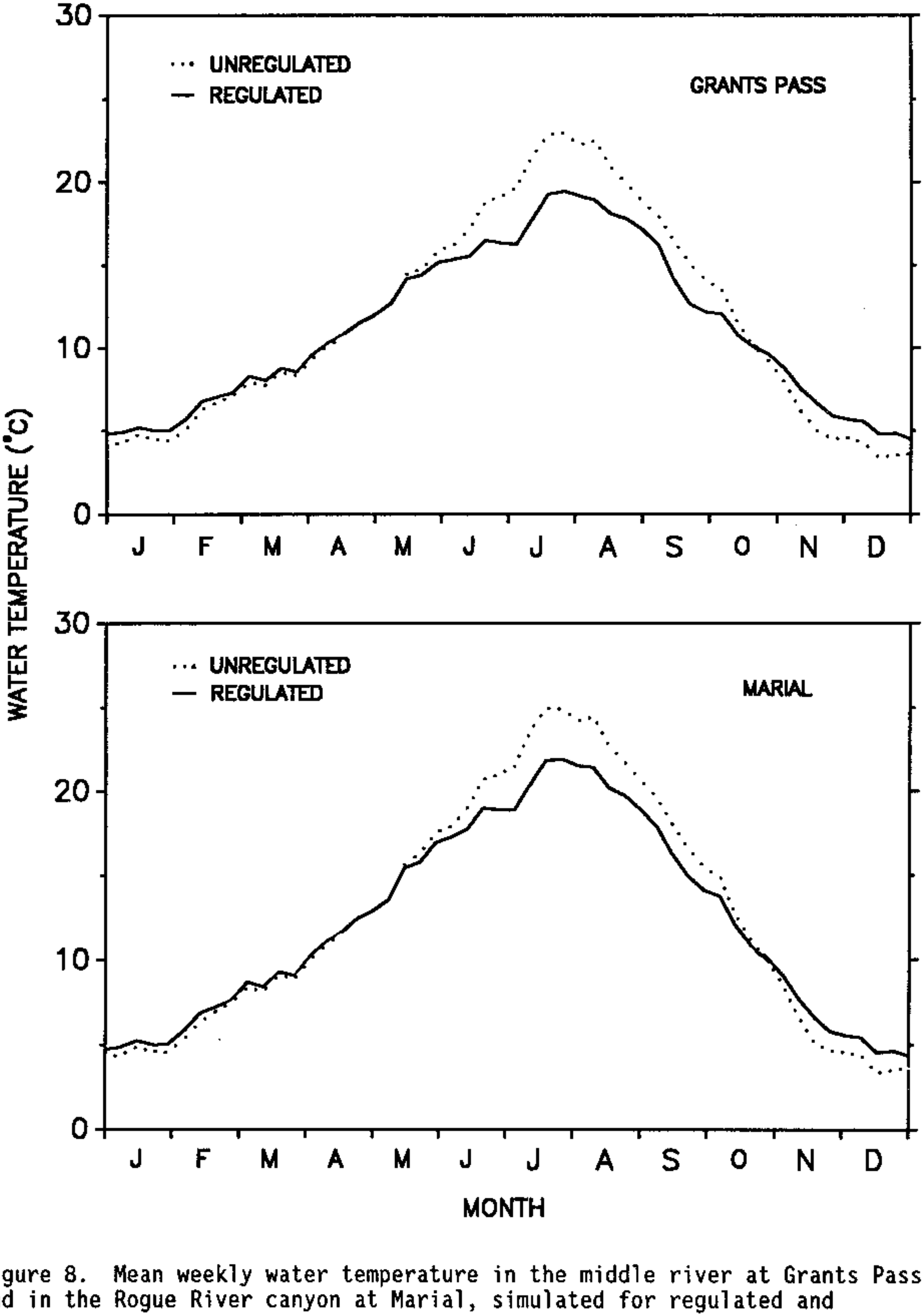

Figure 8. Mean weekly water temperature in the middle river at Grants Pass, and in the Rogue River canyon at Marial, simulated for regulated and unregulated conditions during 1978-86.

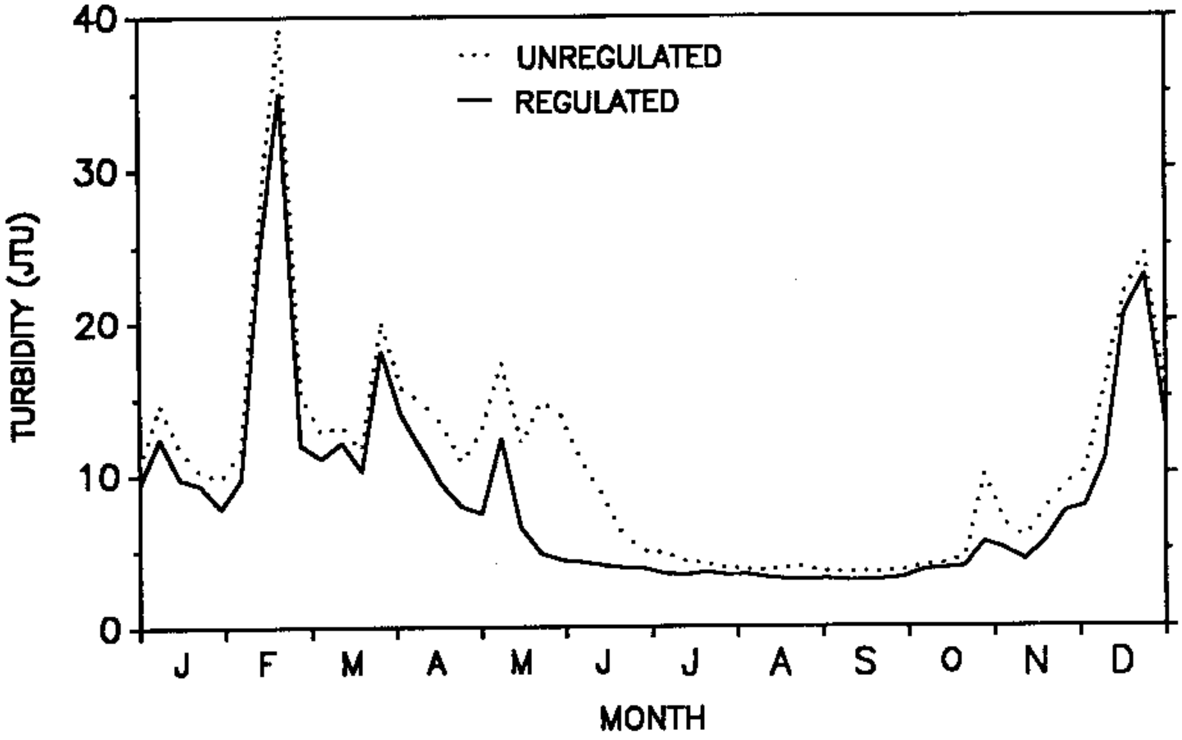

Figure 9. Mean weekly turbidity in the middle river at Grants Pass, simulated for regulated and unregulated conditions during 1978-86.

Based on the results of USACE simulations, we estimated that changes in water temperature caused fry to emerge an average of 14 days earlier in the middle river during 1978-86. To derive this estimate, we assumed (1) a mean spawning date of 15 October (*see* **Spawning Time**, page 83) and (2) 1,835 temperature units to the "button up" stage of fry development (personal communication dated 20 September 1991 from Michael Evenson, ODFW, Cole M. Rivers Hatchery, Trail, Oregon). USACE simulation of water temperature in the Rogue River indicated 1,835 temperature units accumulated at Grants Pass by 19 March for unregulated conditions and 3 April for regulated conditions. Estimates of temperature units represented averages for simulated conditions in 1978-86.

However, field sampling failed to detect any change in the emergence timing of fall chinook salmon fry. We found no significant differences in mean time of first emergence for preimpoundment and postimpoundment broods at any sample sites (Table 8). Also, mean time of emergence completion did not differ significantly for preimpoundment and postimpoundment broods (Table 8).

Annual variations in emergence timing decreased the chance that changes could have been detected by postimpoundment sampling. Sensitivity analyses indicated that time of first emergence at Matson would have had to change by 43 days to be detected. A second sensitivity analysis indicated time of emergence completion at Matson would have had to change by 46 days to be detected. Field sampling could not have detected the 14 day change in emergence timing estimated from USACE simulations of water temperature.

Table 8. Comparison of mean emergence timing of fry of fall chinook salmon in the Rogue River basin before and after operation of Lost Creek Dam, 1975-81.

Emergence index, sampling site:	Preimpoundment 1975-77	Postimpoundment 1978-81	*P* for difference
First emergence[a]:			
Matson	March 19	March 28	0.530
Almeda	March 20	March 11	0.517
Agness	March 17	March 14	0.770
Hideaway	April 2	March 24	0.520
Canfield	April 1	March 14	0.192
Applegate River[b]	March 11	February 28	0.286
Illinois River[b]	April 17	April 5	0.271
Emergence completion[c]:			
Matson	April 21	April 25	0.784
Almeda	May 2	April 26	0.696
Agness	April 26	April 23	0.871
Hideaway	May 6	April 23	0.533
Canfield	May 11	April 21	0.377
Applegate River[b]	April 25	April 15	0.682
Illinois River[b]	May 8	May 5	0.776

[a] *Date of first capture.*
[b] *Sampled at RK 1.*
[c] *First date when mean lengths continually exceeded 4.5 cm.*

Abundance

Catch rates of subyearling chinook salmon at Matson and Almeda in May-August averaged 119 fish per seine haul in 1975-86. Catch rates varied significantly between years and annually ranged between 20 and 273 fish per seine haul (Figure 10).

We assumed variations in seine catches at Matson and Almeda reflected variations in the abundance of juvenile fall chinook salmon throughout the Rogue River. We believe this assumption was reasonable because catch rates at Matson and Almeda were significantly correlated ($r = 0.86$, $P = 0.013$) with average catch rates for other sites sampled during 1974-81. Catch rates from all sites sampled in 1974-81 were reported by Cramer et al. (1985).

Results from branding experiments with juvenile spring chinook salmon (Cramer et al. 1985) suggested juveniles captured at Matson and Almeda were primarily progeny of fall chinook salmon. During 1976, we found no branded juvenile spring chinook salmon among 18,400 juveniles seined at Matson and Almeda. However, we found 22 branded juvenile spring chinook salmon among 47,078 juveniles trapped farther upstream at Savage Rapids Dam (RK 173).

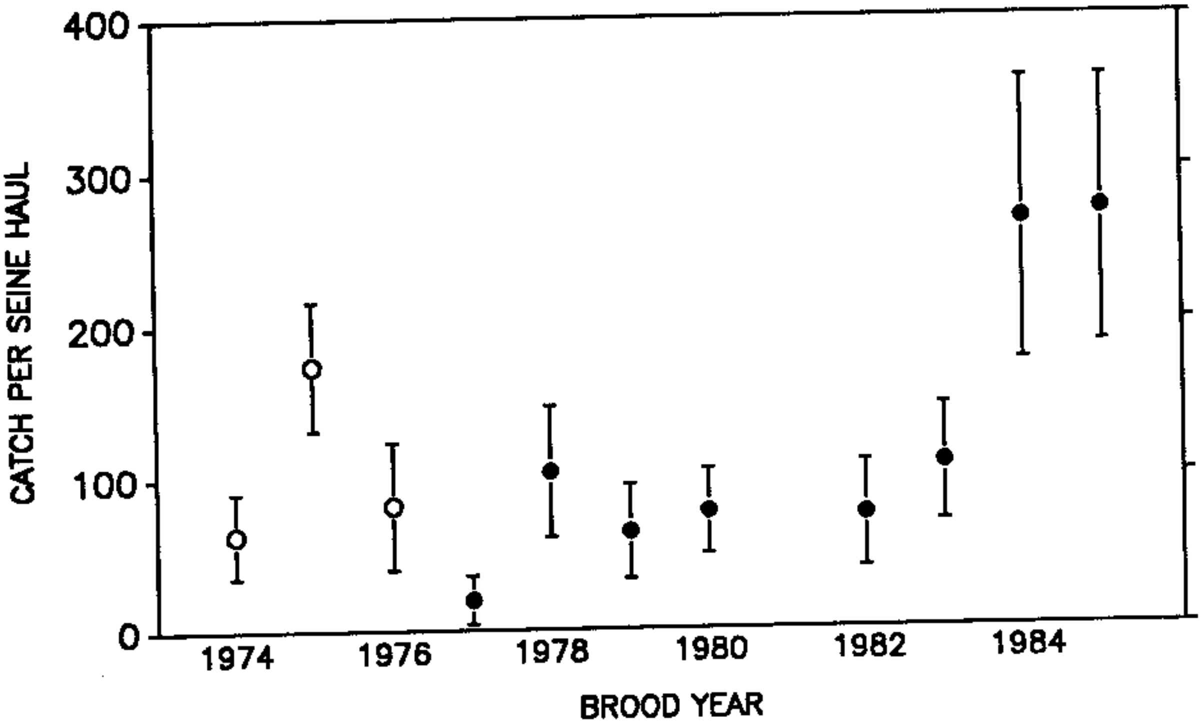

Figure 10. Mean catch rates of juvenile fall chinook salmon seined at Matson and Almeda during May-August, 1974-85 brood years. Open and closed circles represent preimpoundment and postimpoundment broods, respectively. Brackets represent 95% confidence intervals.

Juvenile chinook salmon caught at Matson and Almeda originated from the Rogue and Applegate rivers. During 1983-84, we captured juveniles previously branded in the lower portion of the Applegate River. Because we caught juveniles known to be progeny of fall chinook salmon and captured few juveniles known to be progeny of spring chinook salmon, we concluded progeny of the fall race dominated seine catches of chinook salmon at Matson and Almeda.

We hypothesized that four primary factors influenced production of juvenile fall chinook salmon: (1) number of female spawners, (2) flow when adults migrated into and spawned in tributary streams, (3) intensity of peak flow when eggs and alevins incubated in the gravel, and (4) water temperature when eggs and alevins incubated in the gravel. Data used to test this hypothesis are in Appendix Table B-3.

We found that tributary flow during adult migration and spawning, intensity of peak flows during gravel incubation of eggs and alevins, and abundance of female spawners were related to juvenile abundance. In a multiple regression analysis, each of these variables accounted for a significant amount of variation in annual catch rates of subyearling chinook salmon at Matson and Almeda (Appendix Table B-4).

Results indicated that juvenile abundance was positively related to spawning escapement and tributary flow in autumn, but was negatively related to the intensity of peak flow in winter. Of the three independent variables, tributary flow in autumn was most highly correlated (but not significantly) with juvenile abundance. However, residual variation from this relationship was significantly related to the intensity of peak flow when eggs and alevins incubated in the gravel (Figure 11). Residual variation from that relationship was significantly related with annual counts of spawned females in standard survey areas (Figure 11). The relationship between observed and predicted indexes of juvenile abundance is also presented in Figure 11.

Results of the multiple regression must be interpreted with caution. Inclusion of 3 independent variables is excessive for a regression analysis with only 11 years of data. However, we have some confidence in the results because (1) the updated analysis produced results similar to findings we reported with even fewer data (Cramer et al. 1985), (2) the effects of each independent variable on salmonid production is documented in the literature, and (3) independent variables in the regression were not intercorrelated (Appendix Table B-5). Also, carcass counts of spawned females, although an independent variable, were not measured without error. Efficiency of recovery of spawned carcasses varies in relation to flow (ODFW, unpublished data) and body size (Solazzi 1984). We included carcass counts as an independent variable because we found carcass counts were highly correlated with estimates of spawning escapement (*see* **Spawner Abundance:**, page 64).

This analysis suggested that operation of Lost Creek Dam increased production of juvenile fall chinook salmon in the Rogue River. We concluded that reductions in peak flow increased survival rate of eggs and alevins incubating in the gravel. Results from other research studies support this conclusion.

Egg-to-fry survival rates of chinook salmon increased from 0.2%-7% to 12%-20% after control of freshets by a dam on the Big Qualicum River in British Columbia (Lister and Walker 1966). Survival rates of chinook salmon eggs in Mill Creek, California, averaged 60% during controlled flow, while survival rates of eggs exposed to floods averaged 2% (Gangmark and Bakkala 1960). Wales and Coots (1954) found egg-to-fry survival rates for chinook salmon in Fall Creek, California, ranged between 7% and 32%, with greater survival during a winter of stable flow. Rivers (1964) and Cramer et al. (1985) found a negative relationship between the intensity of peak flow and returns of spring chinook salmon to the Rogue River.

Flooding reduced production of juvenile brook trout (Elwood and Waters 1969; Seegrist and Gard 1972; and Erman et al. 1988), sockeye salmon (Thorne and Ames 1987), and coho salmon (Holtby and Healey 1986; Scrivener and Brownlee 1989). High flows displace eggs and alevins from gravel nests, washing them into downstream areas where they may be injured (Harvey 1987) or consumed by predators (Wickett 1958).

The regression analysis also suggested operation of Applegate Dam increased production of juvenile fall chinook salmon in the Rogue River basin. Augmentation of flow allowed adults to migrate farther upstream and spawn in areas of the Applegate River not accessible to spawners in years of low flows (Fustish et al. 1988). Stevens and Miller (1983) reported a similar finding

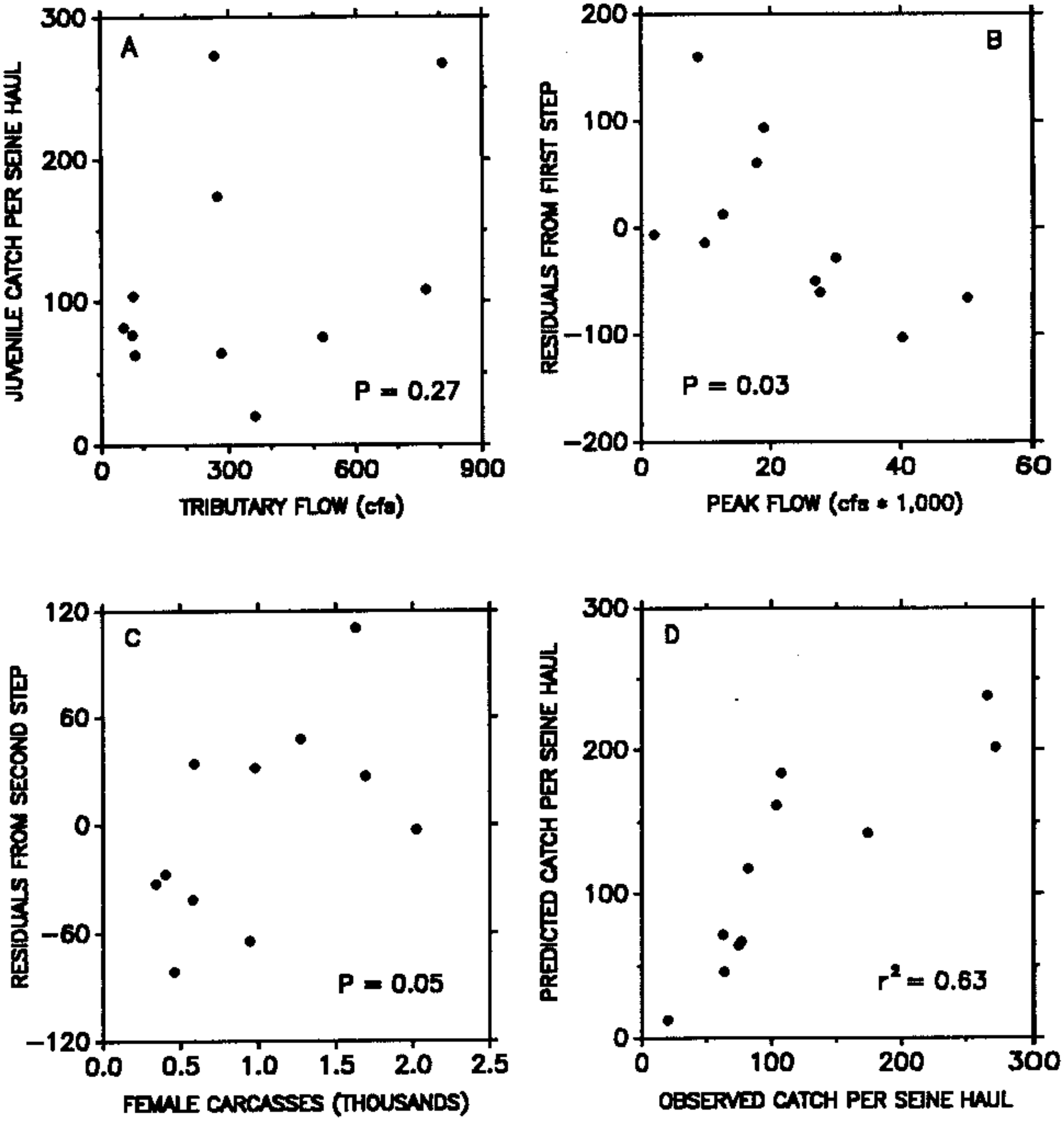

Figure 11. Steps in the analysis of factors that affected abundance of juvenile fall chinook salmon, 1974-85 brood years. Variables within plots are (A) mean catch rates of subyearlings at Matson and Almeda parks and mean flow in the Applegate River at Wilderville during the preceding October-November, (B) residual variation from relationship (A) and peak mean daily flow at Grants Pass during the preceding November-February, and (C) residual variation from relationship (B) and counts of spawned females in standard survey areas. Plot D compares values predicted from the regression with observed values.

for chinook salmon in the Sacramento-San Joaquin river system. Other research has documented the need for adequate flows to permit the upstream passage of chinook salmon (Smith 1973).

The regression analysis also indicated that spawning escapement of females (as indexed by carcass recoveries) affected the production of juvenile fall chinook salmon in the Rogue River basin during 1975-86. A positive correlation between spawner abundance and resultant juvenile production has been found for other stocks of chinook salmon (Major and Mighell 1969; Lindsay et al. 1989). Bjornn (1978) found that the number of eggs deposited by females accounted for 52% of the variation in the number of spring chinook salmon fry that migrated from the Lemhi River in Idaho.

We found no indication that abundance of juvenile fall chinook salmon reached or exceeded carrying capacity of the Rogue River basin during 1975-86. Because there is a finite amount of habitat, juvenile production must level out or even decrease at some level of spawning escapement (Ricker 1975). To identify the relationship between spawning escapement and production of juvenile fall chinook salmon in the Rogue River basin, additional data are needed for years of large spawning escapements.

Juveniles should be seined at Matson and Almeda when more than 2,000 carcasses of spawned females are recovered in standard survey areas during the preceding year. Until such sampling is conducted, we can only surmise that production of juvenile fall chinook salmon in the Rogue River basin will increase when spawning escapement exceeds levels observed in this study.

Our findings indicated that juvenile production increased as recoveries of spawned carcasses in standard survey areas increased to 1,500 females (Figure 11). Assuming females account for 50% of the age 3-6 spawners (*see* **Sex Composition**, page 86) and 13% of the age 3-6 spawners are recovered during surveys of standard areas (*see* **Spawner Abundance**, page 64), then at least 23,000 age 3-6 spawners appear to be required to completely seed spawning habitat available to fall chinook salmon in the Rogue River basin. Assuming no prespawning mortality and a freshwater harvest rate of 15% (*see* **Freshwater Harvest**, page 78), a freshwater return of about 27,000 age 3-6 fish is needed to completely seed the habitat. This assessment of spawning requirements for maximum production of juvenile fall chinook salmon in the Rogue River basin should be further evaluated by an analysis of the production potential of fresh water habitat.

Growth Rate and Body Size

Growth Rate: Scale analyses indicated that growth rates of juvenile fall chinook salmon increased after operation of Lost Creek Dam. Average spacing of freshwater circuli on adult scales increased from 2.13 mm (at 88X) for the 1972-76 broods to 2.22 mm (at 88X) for the 1977-81 broods. An analysis of variance indicated the change in mean spacing was significant at $P = 0.046$.

From the change in average spacing of freshwater circuli, we estimated that postimpoundment broods grew (cm per day) an average of 16% faster than preimpoundment broods. We derived this estimate from a relationship between circuli spacing and body length. In laboratory experiments, we that found mean circuli spacing was highly correlated ($r^2 = 0.92$) with the average daily increase in mean length (Cramer and Martin 1978). Growth rates of juvenile chinook salmon in the laboratory ranged between 0.015 cm and 0.030 cm per day. Significant relationships between circuli spacing and growth rate have also been reported for other species of fish (Bilton and Robins 1971; Doyle et al. 1987).

We found that growth rates in fresh water differed between brood years (Figure 12) and between age classes of adults (Table 9). A Newman-Keuls multiple range test indicated age 5 adults grew at significantly slower rates as juveniles in fresh water compared with cohorts that matured at younger ages. We found no significant differences among scale indexes of juvenile growth among cohorts that matured at ages 2, 3, or 4. Data included in these analyses are in Appendix Table B-6.

We hypothesized water temperature, flow, and juvenile abundance affected growth rate of juvenile fall chinook salmon in fresh water. However, no single variable, or combination of multiple variables, accounted for a significant (all $P > 0.10$) amount of variation in average spacing of freshwater circuli on adult scales. Data included in this analysis are in Appendix Table B-7. A correlation matrix that outlines relationships among all variables examined in this analysis is in Appendix Table B-8.

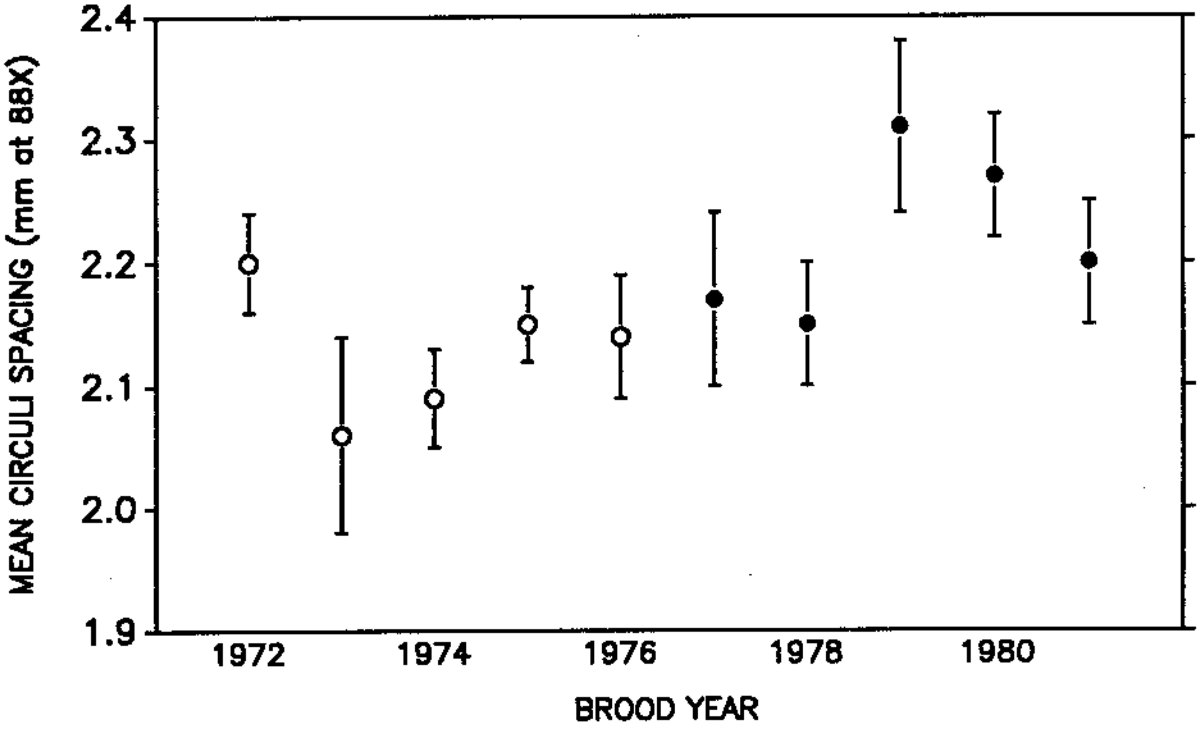

Figure 12. Freshwater growth rate of subyearling fall chinook salmon, 1972-81 brood years. Annual growth rates were indexed by mean spacing of freshwater circuli on adult scales. Open and closed circles represent preimpoundment and postimpoundment broods, respectively. Brackets represent 95% confidence intervals.

Table 9. Comparison of mean rates of freshwater growth for age 2-5 fall chinook salmon, 1972-81 brood years. Growth rate represented as the average spacing (mm at 88X) of freshwater circuli on adult scales.

Two factor analysis of variance					
Source of variation	Sum of squares	df	Mean square	*F*	*P*
Age class	0.134	3	0.045	9.72	<0.001
Brood year	0.217	9	0.024	5.24	<0.001
Residual	0.124	27	0.005		

Measurements from adult scales indexed growth rate of juveniles that reared throughout the Rogue River basin. Numerous adult fall chinook salmon reared as juveniles in the Applegate River. Trap catches in the lower portion of the Applegate River indicated juveniles entered the Rogue River mostly in late spring when 5-7 cm in length (Fustish et al. 1988). Consequently, a portion of measurements taken from adult scales represented growth in tributary streams rather than the Rogue River. We examined the size of juveniles caught in the Rogue River to further evaluate effects of changes in physical parameters of the Rogue River.

Body Length: We found that juvenile fall chinook salmon caught at sites downstream of Grants Pass were significantly larger in summer after operation of Lost Creek Dam. Mean lengths of juveniles captured at Matson, Agness, and Canfield in 1978-81 averaged 0.2-0.7 cm larger compared with counterparts caught at these sites in 1974-77 (Cramer et al. 1985). However, there were no significant changes in rate of change in mean lengths of juveniles seined during spring (Cramer et al. 1985).

Multiple regression analysis indicated that water temperature and abundance of the cohort were significantly related to lengths of juvenile chinook salmon that reared in the Rogue River during summer (Appendix Table B-9). Juvenile lengths increased with decreased water temperature and decreased abundance of juvenile chinook salmon (Figure 13). Data included in this analysis are in Appendix Table B-7. A correlation matrix that outlines relationships among all variables examined in this analysis is in Appendix Table B-8.

Regression coefficients, in conjunction with data gathered during the study, suggested annual variations in water temperature or juvenile production may singularly cause mean lengths of juvenile fall chinook salmon to vary as much as 0.9 cm in summer. We derived these estimates by multiplying the regression coefficients by differences in minimum and maximum values of each independent variable.

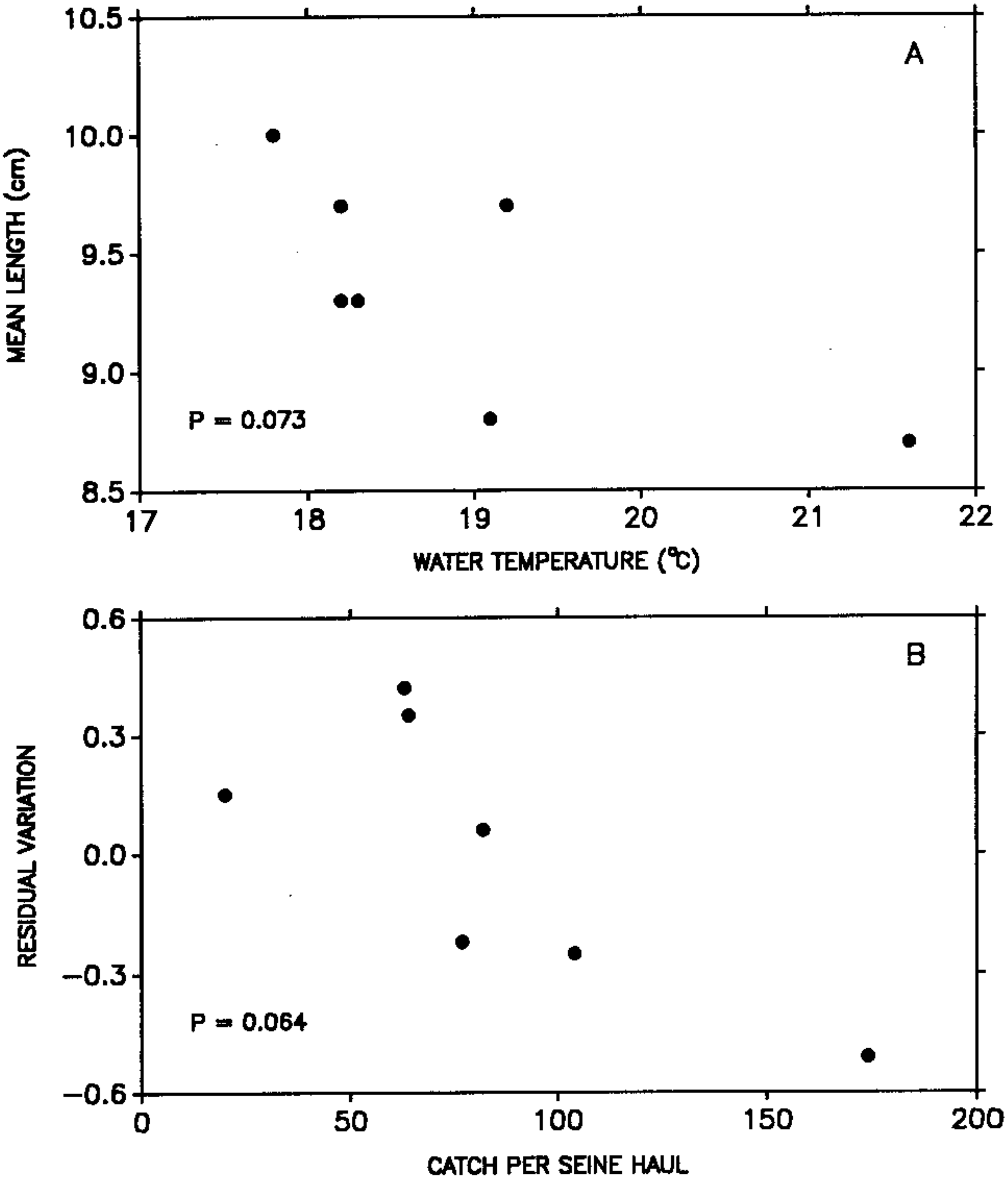

Figure 13. Steps in the analysis of factors that affected size of juvenile fall chinook salmon, 1974-80 brood years. Variables within plots are (A) mean lengths of subyearlings averaged for fish seined at three sites during summer and mean maximum water temperature at Grants Pass during June-August, and (B) residual variation from relationship (A) and mean catch rates of subyearlings seined at Matson and Almeda.

Body Weight: Weight of juvenile fall chinook salmon that reared in the Rogue River during summer increased after operation of Lost Creek Dam. Using length-weight relationships developed by Cramer et al. (1985), we predicted that weights of 10 cm juveniles averaged 11.1 g and 11.6 g for the 1974-76 and 1977-80 brood years, respectively. The difference in mean weight was significant at $P = 0.033$.

Juvenile chinook salmon weighed more because water temperatures in the middle river decreased after operation of Lost Creek Dam. We found that body weight was negatively related with water temperature in the middle river during summer (Figure 14). Juveniles 10 cm in length weighed more in years when water temperature was relatively low during summer.

Based on analyses of body weight, body length, and growth rate, we concluded the size of juvenile fall chinook salmon in the Rogue River increased with (1) decreased water temperature during summer and (2) decreased abundance of cohorts. Other research on salmon supported this conclusion.

Water temperature affected growth rates of juvenile chinook salmon to a greater degree than either salinity or photoperiod (Clarke et al. 1981). Banks et al. (1971) found juvenile chinook salmon grew faster at 15°C compared with 10°-13°C or 18°C. Juvenile chinook salmon also grew faster in a heated stream where temperatures peaked at 16°C compared with a control stream where temperatures peaked at 20°C (Bisson and Davis 1976). Brett et al. (1982) presented information that indicated optimum temperature for growth of juvenile chinook salmon was about 15°C. Greater than optimum temperature decreased efficiency of food conversion.

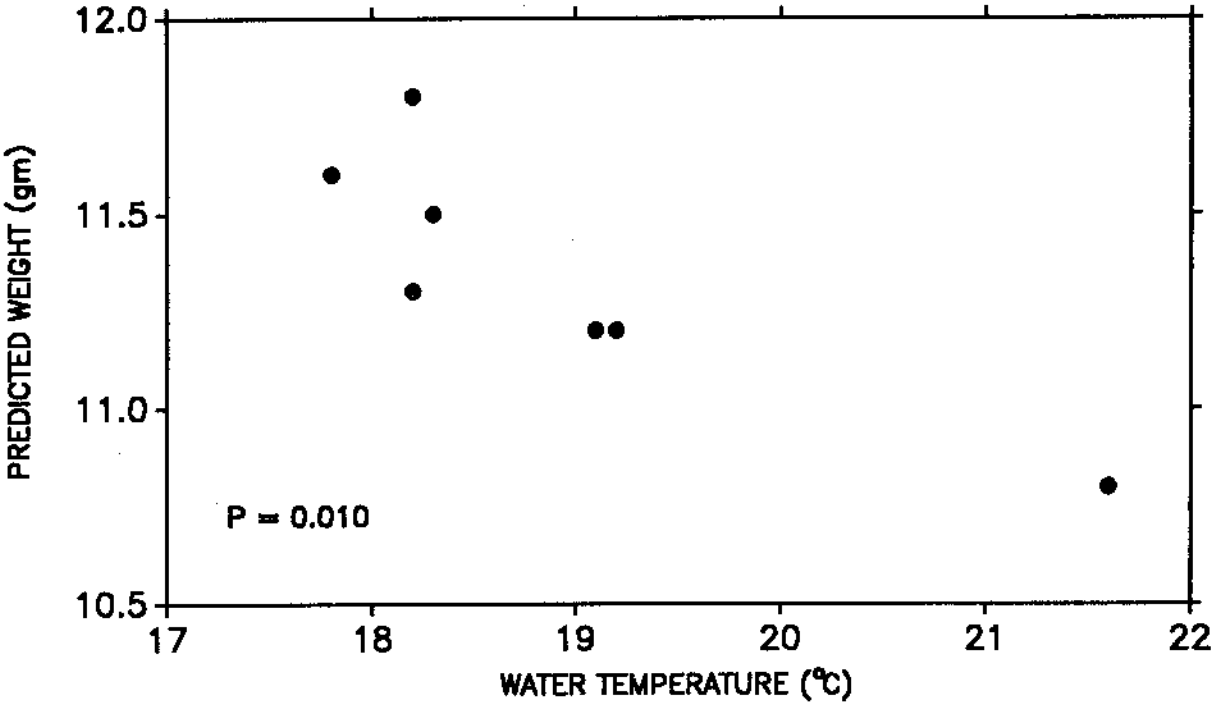

Figure 14. Relationship between predicted weight of a 10 cm juvenile chinook salmon and mean maximum water temperature at Grants Pass during June-August, 1974-80 brood years.

Effects of fish density on growth rate were less apparent in the literature. Weatherley (1972) concluded that density-dependent effects on growth rate are not always evident in fish populations. However, among Pacific salmon, density affected growth rates of juvenile sockeye salmon (Kyle et al. 1988) and coho salmon (Scrivener and Anderson 1984).

Among chinook salmon, Bjornn (1978) found a negative correlation between the length and abundance of juveniles that migrated from the Lemhi River, Idaho. Variations in juvenile abundance were associated with a 0.8 cm variation in migrant lengths. Lindsay et al. (1989) reported similar results for juvenile spring chinook that migrated from the Warm Springs River in Oregon. Neilson et al. (1985) found evidence that growth rate of juvenile chinook salmon that reared in the estuary of the Sixes River, Oregon, was density dependent. However, Carl (1984) found no relationship between growth rates and densities of juvenile chinook salmon that reared in two small tributaries of Lake Michigan. These studies indicated cohort abundance can affect growth rate of juvenile chinook salmon in some ecosystems.

Age, Timing, and Size at Migration

Age at Ocean Entry: We found that most juvenile fall chinook salmon in the Rogue River basin entered the ocean in the first year of life. Yearling migrants were relatively rare. At Almeda, where we caught most yearlings, catch rates of yearlings in spring averaged less than 2% of catch rates of subyearlings during the preceding summer. Scales taken from adults also indicated most juveniles migrated to the ocean as subyearlings.

Subyearling migrants are also the dominant life history type among populations of fall chinook salmon in the Columbia River basin (Koo and Isarankura 1968) and coastal streams of Oregon (Nicholas and Hankin 1988). Yearling migrants are more prevalent among spring chinook salmon (Rich 1925) and chinook salmon produced in streams at higher latitudes (Healey 1983).

A Kruskal-Wallis test (Zar 1984) detected a significant difference ($P < 0.001$) in the percentage of yearlings among age 2-5 fall chinook salmon. A nonparametric Tukey-type multiple comparison (Zar 1984) indicated yearling migrants were relatively less abundant ($P \leq 0.01$) among age 2 adults compared with older adults. We found no significant difference in the mean percentage of yearling migrants among adults that returned at ages 3, 4, or 5. We did not include age 6 adults in the analysis because small samples usually prevented estimation of the percentage of yearlings within this age class. Data included in this analysis are in Appendix Table B-10.

Yearling migrants accounted for an average of 4.6% of the adults that returned to fresh water from the 1972-81 brood years. We estimated yearling migrants would have averaged 7.8% of the adult returns without age selective mortality that resulted from ocean harvest and the El Niño event of 1982-83. We believe estimates adjusted for age-selective mortality more accurately represented the life history composition of fall chinook salmon produced in the Rogue River basin (*see* Juveniles, page 16).

An analysis of variance indicated no significant difference ($P = 0.276$) in the mean percentage of yearlings among preimpoundment and postimpoundment broods. Yearling migrants averaged 10.6% of the 1972-76 broods and 5.1% of the 1977-81 broods (Appendix Table B-10). The difference was not significant because of large variation in the percentage of yearling migrants among preimpoundment broods. A sensitivity analysis indicated the mean percentage of yearling migrants within postimpoundment broods would have had to decrease to 1.7% or increase to 28.9% in order to detect a change.

Juveniles produced after operation of Lost Creek Dam may have tended to migrate to the ocean at a younger age because they grew faster. We found the percentage of yearling migrants within broods was negatively related with growth rate in fresh water (Figure 15). Yearling migrants were relatively more abundant among broods that grew at slower rates in fresh water. Conversely, yearling migrants were relatively less abundant among broods that grew at faster rates in fresh water.

We concluded that a greater proportion of juvenile fall chinook salmon migrated to the ocean as subyearlings when they grew faster in fresh water. Other researchers have reported similar results for Atlantic salmon (Chadwick et al. 1978), coho salmon (Holtby 1988), and summer steelhead (Cramer 1986). Power (1981) found an inverse relationship between water temperature and length of the growing season for Atlantic salmon inhabiting rivers in eastern Canada. Implications of a change in the time of ocean entry are further discussed in the next section.

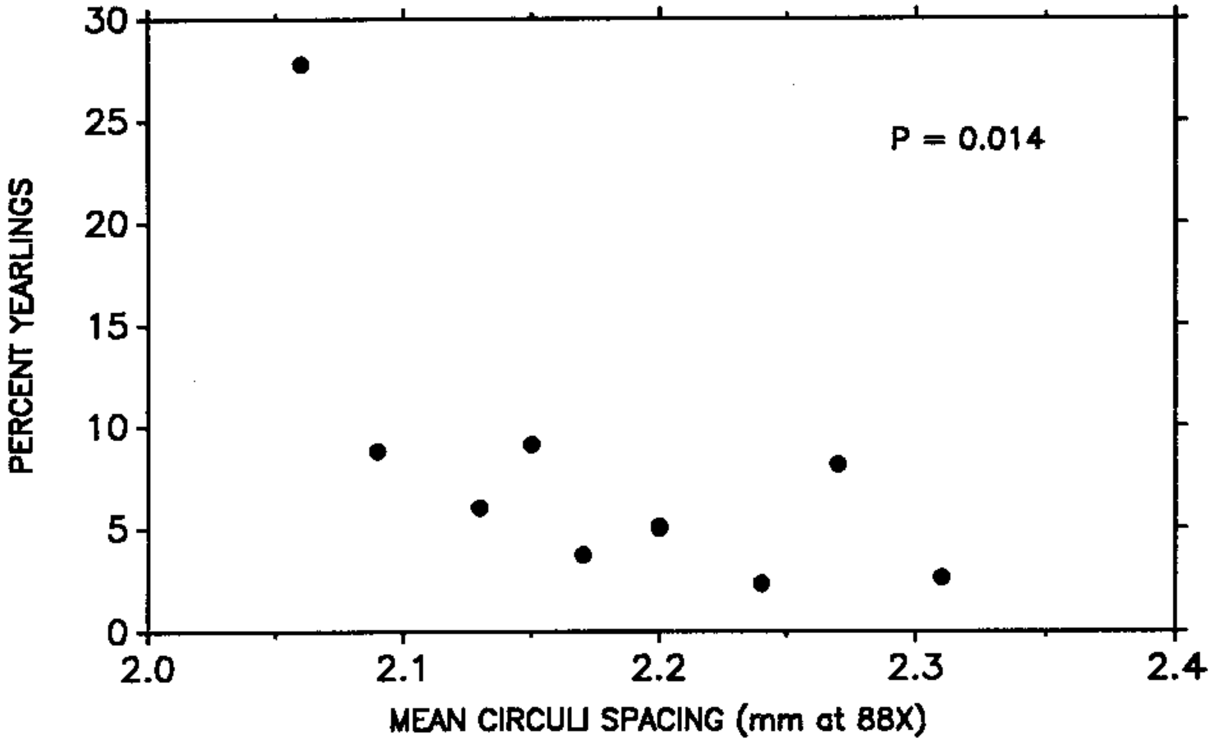

Figure 15. Relationship between the percentage of yearling migrants among broods of fall chinook salmon and mean growth rate of subyearlings in fresh water, 1972-81 brood years. Growth rates were indexed by mean spacing of freshwater circuli on adult scales. Significance level was calculated from percentages transformed to logits.

Time of Ocean Entry: We found that fall chinook salmon entered the ocean as subyearlings mostly in August-October. Catch rates of juveniles generally decreased at seining sites during this period. Analyses of scales taken from adult fall chinook salmon also indicated subyearlings entered the ocean in summer and early autumn. We estimated the mean date of ocean entry ranged between 9 August and 25 September for the 1972-81 broods (Figure 16). Brood years also differed significantly in mean time of ocean entry (Table 10).

Juveniles destined to mature at older ages entered the ocean later compared with cohorts destined to mature at younger ages. We found that the mean date of ocean entry differed significantly among age 2-5 adults (Table 10). A Newman-Keuls multiple range test indicated juveniles destined to mature at age 5 entered the ocean an average of 10 days later than cohorts that matured at younger ages. There was no significant difference in the mean date of ocean entry among subyearlings destined to mature at ages 2, 3, or 4. We did not include age 6 adults in this analysis because sample sizes were usually too small to estimate the date of ocean entry for this age class. Data included in these analyses are in Appendix Table B-11.

Fall chinook salmon that migrated as subyearlings entered the ocean earlier after operation of Lost Creek Dam. Mean date of ocean entry averaged 15 September (95% CI = ±7 days) for preimpoundment broods and 23 August (95% CI = ±12 days) for postimpoundment broods. The difference in means was significant at $P = 0.002$. Data included in this analysis are in Appendix Table B-11.

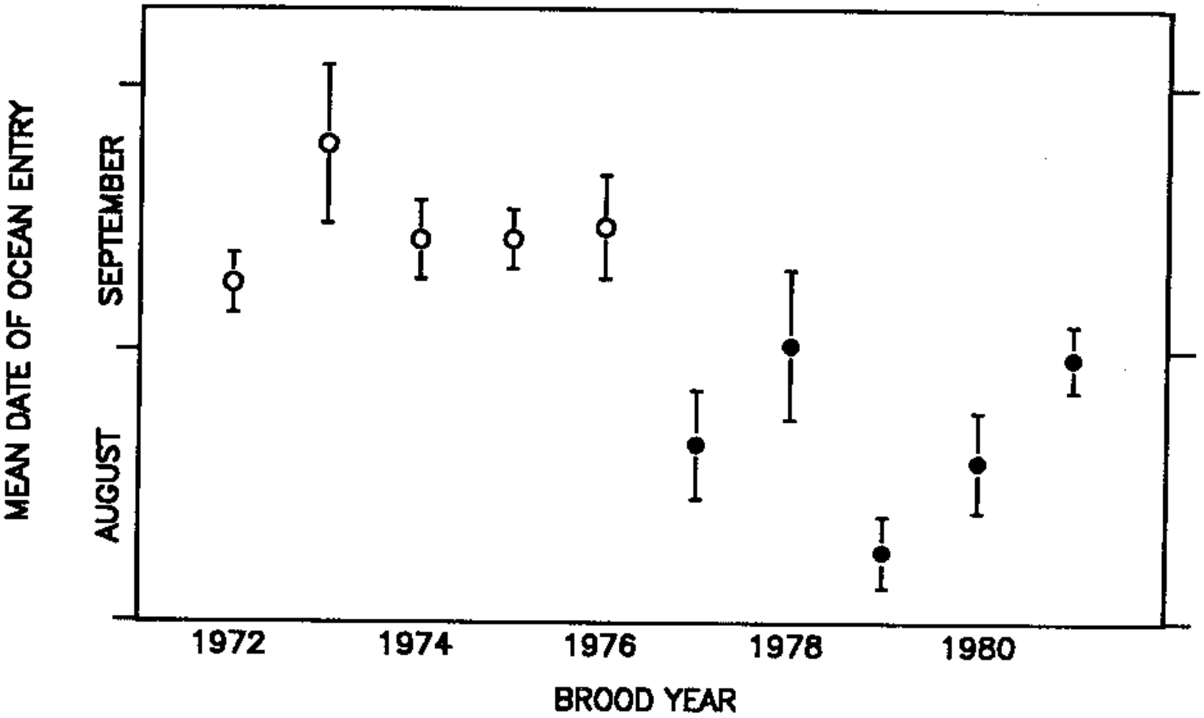

Figure 16. Mean date of ocean entry by subyearling fall chinook salmon, 1972-81 brood years. Open and closed circles represent preimpoundment and postimpoundment broods, respectively. Brackets represent 95% confidence intervals.

Table 10. Comparison of the mean dates of ocean entry for age 2-5 fall chinook salmon, 1972-81 brood years. Dates of ocean entry were estimated from counts of freshwater circuli on adult scales.

Two factor analysis of variance					
Source of variation	Sum of squares	df	Mean square	*F*	*P*
Age class	752	3	250.7	4.01	0.018
Brood year	7,146	9	794.0	12.70	<0.001
Residual	1,688	27	62.5		

We believe postimpoundment broods entered the ocean earlier because they grew faster in fresh water. We found mean date of ocean entry was negatively related with average spacing of freshwater circuli on adult scales (Figure 17). Subyearling migrants entered the ocean earlier in years of faster growth and later in years of slower growth. Data included in this analysis are in Appendix Table B-12. A correlation matrix that outlines relationships among variables examined in this analysis is in Appendix Table B-13.

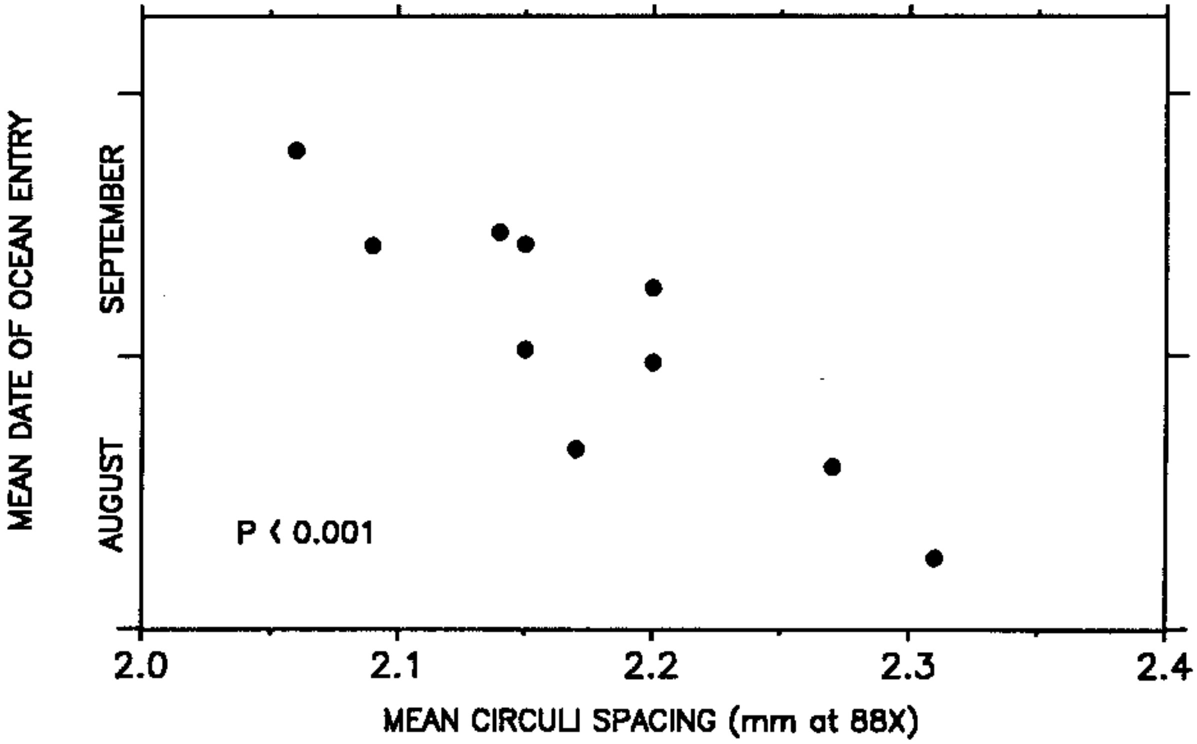

Figure 17. Relationship between mean date of ocean entry and mean growth rate in fresh water for subyearling fall chinook salmon, 1972-81 brood years. Growth rates were indexed by mean spacing of freshwater circuli on adult scales.

We found that juvenile fall chinook salmon migrated from the Rogue River at a time similar to other coastal populations of fall chinook salmon in Oregon (Reimers and Loeffel 1967; Reimers 1973). However, subyearlings in the Rogue River migrated later than other populations of chinook salmon. Juveniles that reared in estuaries of the Nanaimo (Healey 1980) and Campbell (Levings et al. 1986) rivers of British Columbia entered the ocean mostly during June-July. An August-October migration timing of juvenile fall chinook salmon in the Rogue River conforms to a definition of river smolts proposed by Carl and Healey (1984), except juveniles reared in the Rogue River basin for 4-5 months rather than 2 months.

Differences in time of ocean entry may be indicative of genetic variation among populations of chinook salmon. Migration timing is associated with physiological changes in juvenile chinook salmon. Buckman and Ewing (1982) found that specific activity of the (Na+K)-ATPase enzyme in gills of juvenile chinook salmon seined at Canfield peaked coincidentally with the peak in time of ocean entry as estimated from scales of adult spring chinook salmon. Increases in the activity of this enzyme are associated with physiological modifications required for the transition from fresh water to salt water (Folmar and Dickhoff 1980).

Peaks in specific activity of (Na+K)-ATPase in juvenile spring chinook salmon reared at Cole M. Rivers Hatchery were associated with attainment of a minimum size of 9 cm and onset of autumn equinox (Ewing et al. 1979). Bjornn (1971) postulated photoperiod and possibly growth were the primary factors that affected migration timing of juvenile chinook salmon in the Lemhi River of Idaho. Bjornn (1971) also concluded that water temperature, cover, food, and population density were less important determiners of migration timing. Schluchter and Lichatowich (1977) proposed that juvenile spring chinook salmon in the Rogue River needed to reach a minimum size threshold prior to migration from fresh water. Juvenile fall chinook salmon produced after operation of Lost Creek Dam would have reached a minimal threshold size for migration earlier in the year because they grew faster.

Size at Ocean Entry: We estimated that mean lengths of juvenile fall chinook salmon from the 1972-81 broods averaged 10-12 cm at the time of ocean entry (Figure 18). Mean lengths at ocean entry did not differ significantly among juveniles destined to mature as age 2-5 adults, but differed significantly among brood years (Table 11). We did not include age 6 adults in this analysis because sample sizes were usually too small to estimate the size at ocean entry. Data included in this analysis are in Appendix Table B-14.

Juvenile fall chinook salmon entered the ocean at a larger size after operation of Lost Creek Dam. At time of ocean entry, subyearling migrants from the 1972-76 broods averaged 10.5 cm and counterparts from the 1977-81 brood years averaged 11.1 cm. The difference between means was significant at $P = 0.028$.

We hypothesized that increased growth rate in fresh water or changes in river physical factors in summer were responsible for the increase in migrant lengths. We found that water temperature and flow were related to mean length of subyearlings at ocean entry (Figure 19). However, water temperature and flow were highly correlated in summer, making it impossible to identify a causative factor (Appendix Table B-13).

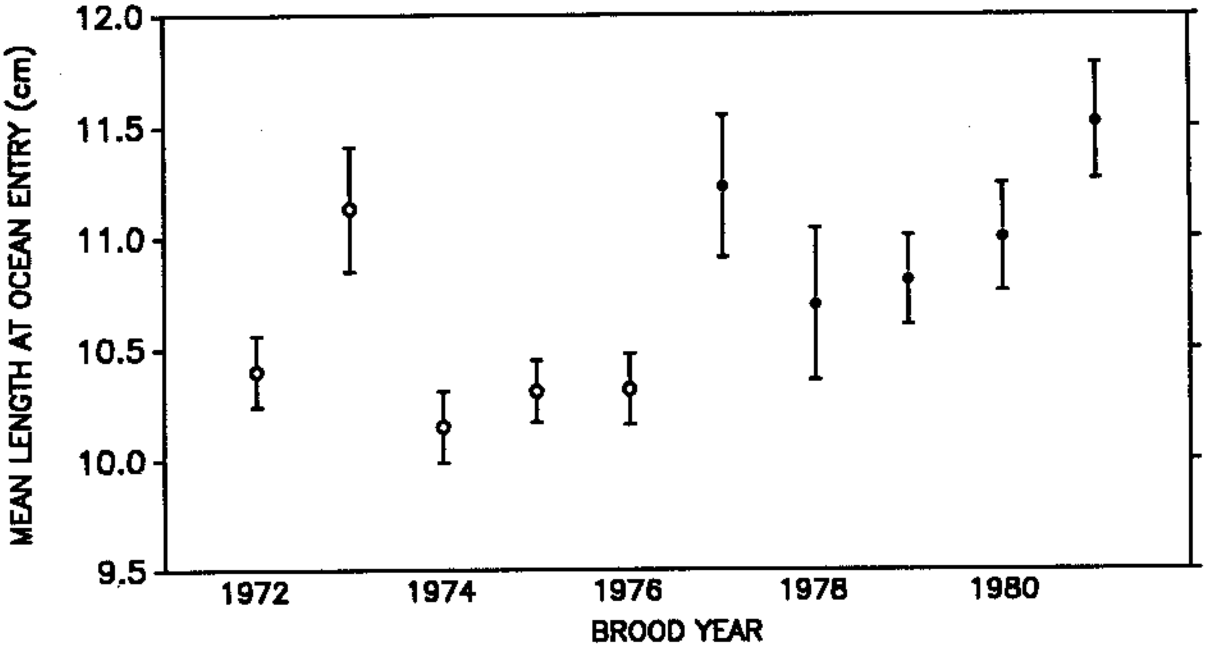

Figure 18. Mean length of subyearling fall chinook salmon at time of ocean entry, 1972-81 brood years. Open and closed circles represent preimpoundment and postimpoundment broods, respectively. Brackets represent 95% confidence intervals.

Juvenile fall chinook salmon migrated from the Rogue River at sizes comparable to those that migrated from other coastal streams of Oregon (Reimers and Loeffel 1967; Reimers 1973; Reimers and Concannon 1977; Nicholas and Hankin 1988). At higher latitudes, juvenile chinook salmon are smaller at time of ocean entry. Juveniles averaged 7.0 cm at time of migration from the estuary of the Nanaimo River in British Columbia (Healey 1980). Yearling chinook salmon that migrated from the Taku River, Alaska, averaged 7.3 cm in length (Meehan and Siniff 1962).

Table 11. Comparison of the mean lengths at ocean entry for age 2-5 fall chinook salmon, 1972-81 brood years. Lengths at ocean entry were estimated from the freshwater radius of adult scales.

Two factor analysis of variance

Source of variation	Sum of squares	df	Mean square	*F*	*P*
Age class	0.141	3	0.047	0.62	0.605
Brood year	8.677	9	0.963	12.82	<0.001
Residual	2.034	27	0.075		

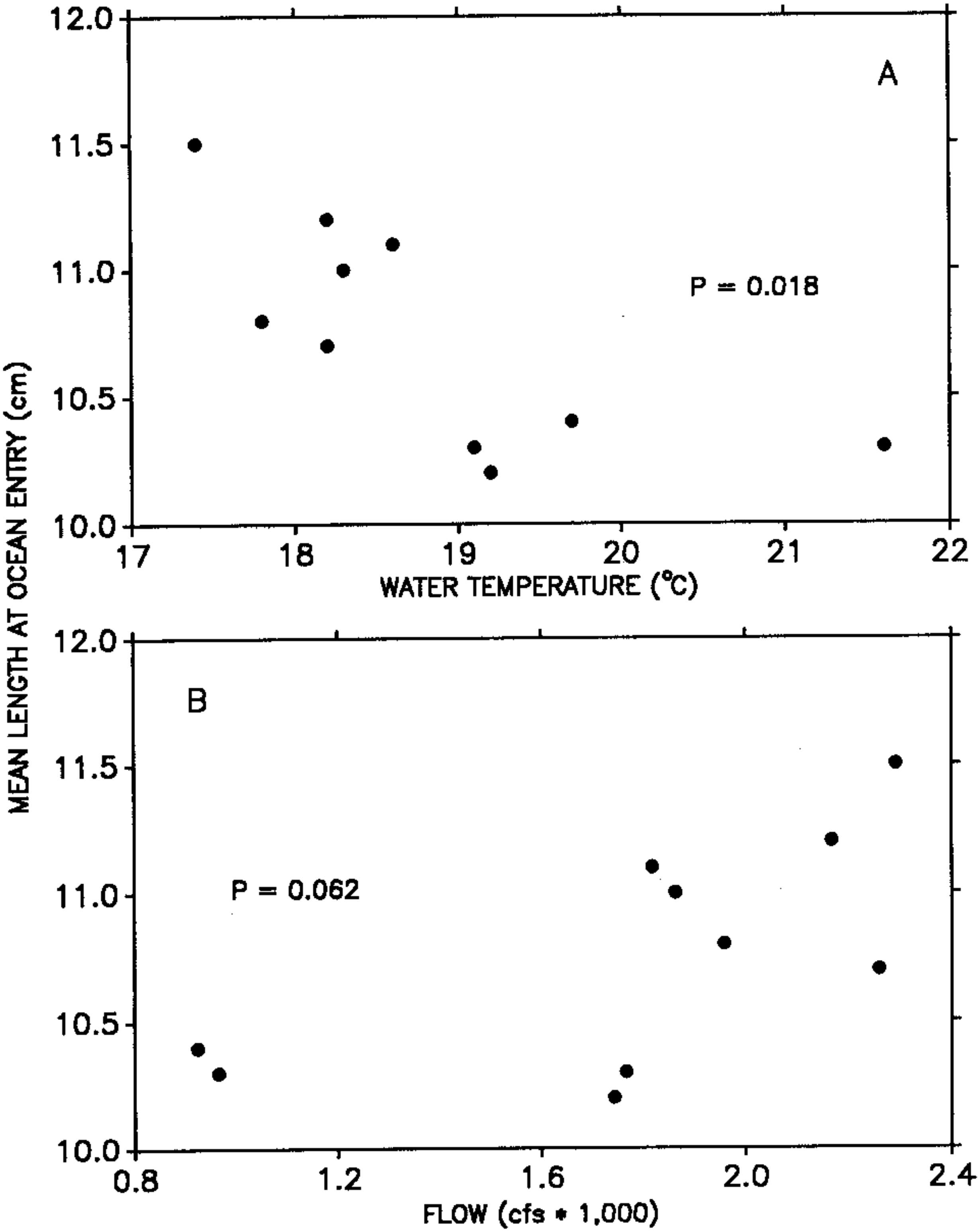

Figure 19. Relationships between mean length of subyearling fall chinook salmon at time of ocean entry and mean maximum water temperature (A), and mean flow (B), at Grants Pass during June-August, 1972-81 brood years.

Adults

Freshwater Return

The run of fall chinook salmon in the Rogue River is the largest on the coast of Oregon (Nicholas and Hankin 1988). We estimated that an average of 45,000 adults passed Huntley Park annually in 1974-86. Annual returns varied from less than 20,000 to almost 100,000 (Table 12). Returns of spring chinook salmon to the Rogue River exhibited a similar degree of variation (Cramer et al. 1985).

Wild fish predominated fall chinook salmon that returned to the Rogue River. Freshwater returns averaged 43,820 wild fish and 1,130 hatchery fish during 1974-86. No hatchery fish returned prior to 1981, but averaged 6.3% of the 1981-86 returns (Table 12). Freshwater returns of hatchery fish increased in 1981-86 because rearing programs started at Cole M. Rivers Hatchery and at Indian Creek in the lower river. Nicholas and Hankin (1988) listed releases of juvenile fall chinook salmon in the Rogue River basin. Because wild fish predominated returns, we limited our analyses to fish of wild origin.

We found freshwater returns of age 2 adults did not differ ($P = 0.79$) between broods produced before and after operation of Lost Creek Dam. Returns of age 2 fish averaged 11,186 and 10,139 for the 1972-76 and the 1977-80 brood years, respectively. We excluded data from brood years after 1980 because rates of natural mortality increased as a result of the El Niño event of 1982-83 (*see* **Effects of El Niño:**, page 61). Data included in this analysis are in Appendix Table C-4.

Table 12. Estimated number of wild and hatchery fall chinook that passed Huntley Park, 1974-86.

Year	Wild	Hatchery	Total	Percent hatchery
1974	42,656	0	42,656	0
1975	37,175	0	37,175	0
1976	23,469	0	23,469	0
1977	32,038	0	32,038	0
1978	74,575	0	74,575	0
1979	69,730	0	69,730	0
1980	33,404	74	33,478	0.2
1981	40,811	609	41,420	1.5
1982	54,197	1,538	55,735	2.8
1983	18,287	3,177	21,464	14.8
1984	17,004	1,208	18,212	6.6
1985	33,863	2,246	36,109	6.2
1986	92,506	5,808	98,314	5.9

We opted not to compare freshwater returns of older chinook salmon from preimpoundment and postimpoundment brood years for two reasons. First, sample sizes for postimpoundment broods were small if brood years affected by the El Niño event of 1982-83 were excluded from the analyses. Sample sizes for postimpoundment broods would have been 3, 2, 1, and 0 for ages 3, 4, 5, and 6 fish, respectively. Second, annual variations in rates of ocean harvest affected freshwater returns of age 3-6 adults. Instead, we focused on an assessment of factors that could have affected abundance of fall chinook salmon in the ocean.

Ocean Abundance

For age 3-6 fall chinook salmon, we found the number of fish resident in the ocean during spring were significantly correlated with freshwater returns of cohorts in the previous year. Also, the number of age 2 fish resident in the ocean was highly correlated with juvenile abundance in fresh water during the previous year. These correlations were significant only when brood years affected by the El Niño event of 1982-83 were excluded from the analyses. Data included in the analyses of ocean abundance are in Appendix Tables C-4 and C-5. A correlation matrix that outlines relationships among all variables examined in these analyses is in Appendix Table C-6.

Age 2: We estimated the number of age 2 fish resident in the ocean during spring averaged about 750,000 and ranged from 110,000 to more than 2 million. Recruitment at age 2 was lowest among broods that entered the ocean during the El Niño event of 1982-83. Annual recruitment among broods not affected by El Niño averaged 870,000 fish and varied by a factor of 10.

Our analysis indicated juvenile production was the primary factor that influenced ocean recruitment at age 2, except for broods affected by the El Niño event of 1982-83. We found that seine catches of juveniles in the Rogue River accounted for 78% of the variation in ocean recruitment of age 2 fish from the 1974-80 brood years. Large numbers of age 2 fish resided in the ocean when large numbers of juveniles were produced in the previous year (Figure 20). Residual variation from this relationship was not significantly related with (1) physical parameters of the ocean, (2) growth parameters during ocean residence, or (3) mean date of ocean entry as juveniles (Appendix Table C-7).

Age 3: We estimated the number of age 3 fish resident in the ocean during spring averaged about 260,000 and ranged between 35,000 and 892,000. Recruitment at age 3 was lowest among broods that entered the ocean during the El Niño event of 1982-83. Annual recruitment among broods not affected by El Niño averaged 299,000 fish and varied by a factor of 5.

Our analyses indicated cohort strength was the primary factor that affected ocean recruitment at age 3, except for broods affected by the El Niño event of 1982-83. We found that freshwater returns of age 2 fish accounted for 86% of the variation in ocean recruitment of age 3 fish from the 1972-82 brood years. Large numbers of age 3 fish resided in the ocean when large numbers of age 2 cohorts returned to fresh water in the previous year (Figure 21). Residual variation from this relationship was not significantly related with upwelling or water temperature of the ocean (Appendix Table C-8).

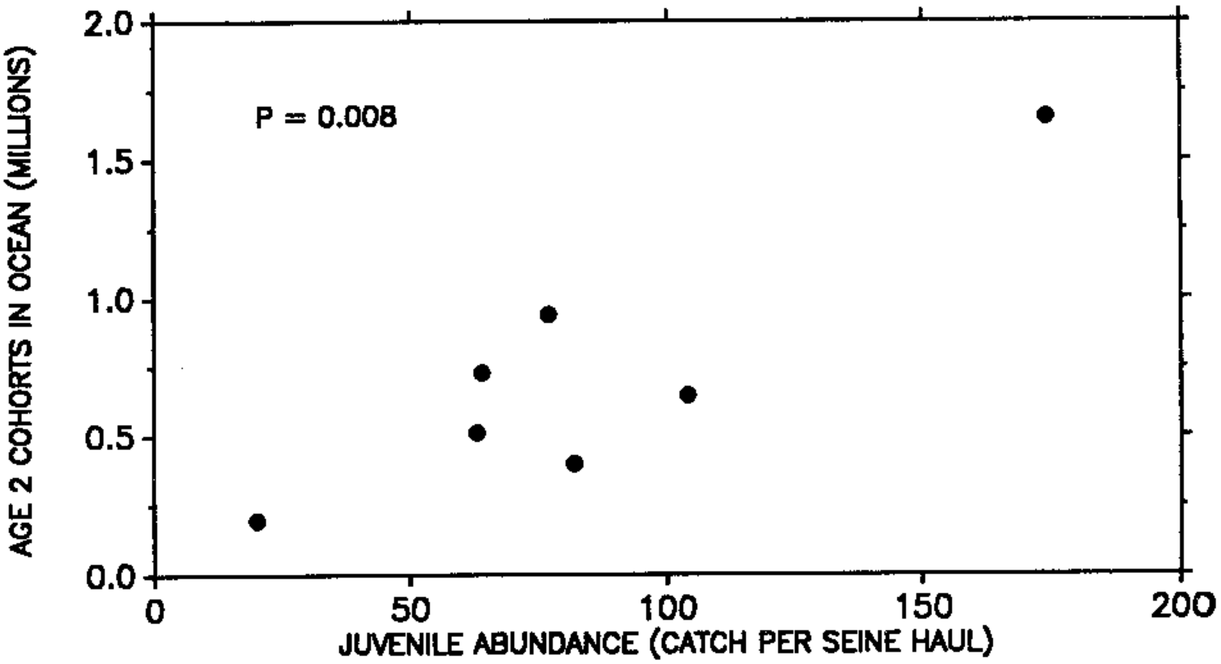

Figure 20. Relationship between estimated number of age 2 fall chinook salmon resident in the ocean during spring and mean catch rates of cohorts seined one year earlier at Matson and Almeda, 1974-80 brood years.

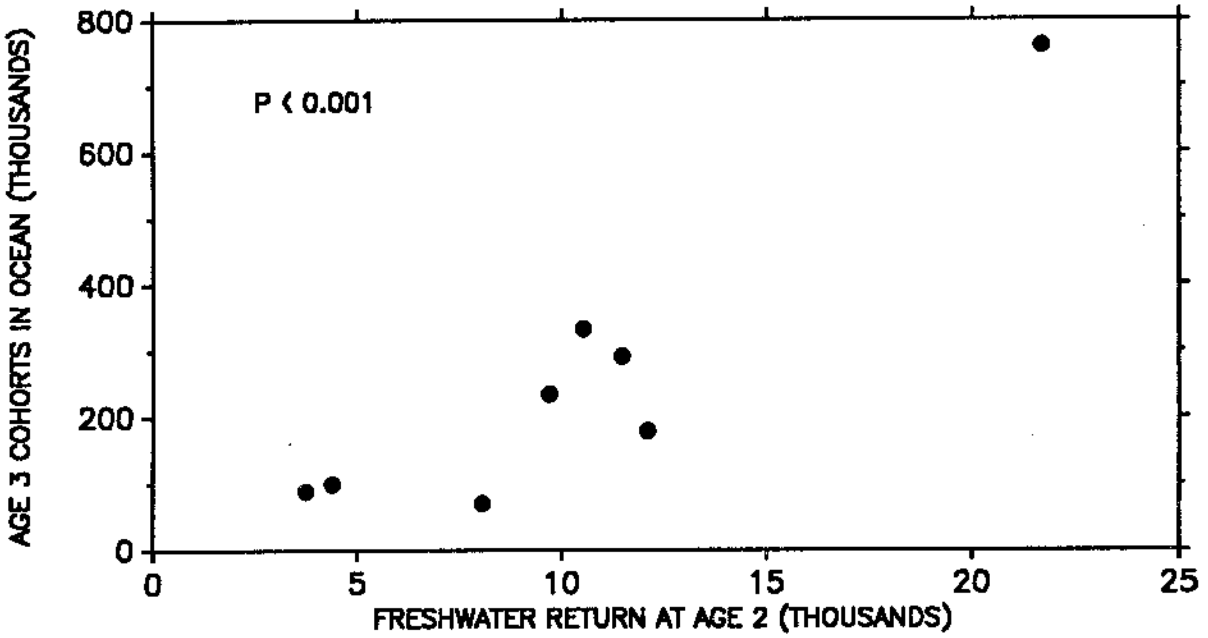

Figure 21. Relationship between estimated number of age 3 fall chinook salmon resident in the ocean during spring and estimated freshwater return of age 2 cohorts one year earlier, 1972-82 brood years. Data from 1980-81 broods were excluded because rates of natural mortality increased as a result of the El Niño event of 1982-83.

Age 4: We estimated the number of age 4 fish resident in the ocean during spring averaged about 77,000 and ranged between 16,000 and 334,000. Recruitment at age 4 was lowest among broods that resided in the ocean as age 3 fish during the El Niño event of 1982-83. Annual recruitment among broods not affected by El Niño averaged 88,000 fish and varied by a factor of 10.

Our analyses indicated cohort strength was the primary factor that influenced ocean recruitment at age 4, except for broods affected by the El Niño event of 1982-83. We found that freshwater returns of age 3 cohorts to the Rogue River accounted for 76% of the variation in ocean recruitment of age 4 fish from the 1971-81 brood years. Large numbers of age 4 fish resided in the ocean when large numbers of age 3 cohorts returned to fresh water in the previous year (Figure 22). The strong 1975 brood year was a primary influence on the slope of the regression. Residual variation from this relationship was not significantly related with upwelling or water temperature of the ocean (Appendix Table C-9).

Age 5: We estimated the number of age 5 fish resident in the ocean during spring averaged 8,000 and ranged between 900 and 40,000. Recruitment at age 5 was lowest among broods that resided in the ocean as age 4 fish during the El Niño event of 1982-83. Annual recruitment among broods not affected by El Niño averaged 10,000 fish and varied by a factor of 10.

Our analyses indicated cohort strength was the primary factor that influenced ocean recruitment at age 5, except for broods affected by the El Niño event of 1982-83. We found that freshwater returns of age 4 cohorts accounted for 75% of the variation in ocean recruitment of age 5 fish from the 1971-81 brood years. Large numbers of age 5 fish resided in the ocean when large numbers of age 4 cohorts returned to fresh water in the previous year (Figure 23). The strong 1975 brood year was the primary influence on the slope of the regression. Residual variation from this relationship was not significantly related with upwelling or water temperature of the ocean (Appendix Table C-10).

Age 6: Few fall chinook salmon that originated from the Rogue River basin matured at age 6. We estimated the number of age 6 fish resident in the ocean during spring averaged about 250 fish and ranged between zero and 1,000. Age 6 fish were absent from 4 of 10 broods we sampled. We found freshwater returns of age 5 cohorts in the previous year accounted for 55% of the variation in the ocean recruitment of age 6 fish from the 1969-80 brood years (Appendix Table C-11). As with some of the other analyses, the strong 1975 brood year was the primary influence on the slope of the regression (Figure 24).

Recruitment: We estimated that recruitment ranged from about 1 fish per spawner to about 23 fish per spawner among the 1974-80 broods (Table 13). Age 2 jacks were excluded from this analysis and recruitment estimates were not adjusted for increased rates of natural mortality during the El Niño event of 1982-83. With only eight years of estimates, we judged there was insufficient data for development of a stock-recruitment curve. We estimated recruitment rates averaged 11 fish per spawner. This level of production is comparable to other populations of chinook salmon on the Pacific coast of North America (Reisenbichler 1987).

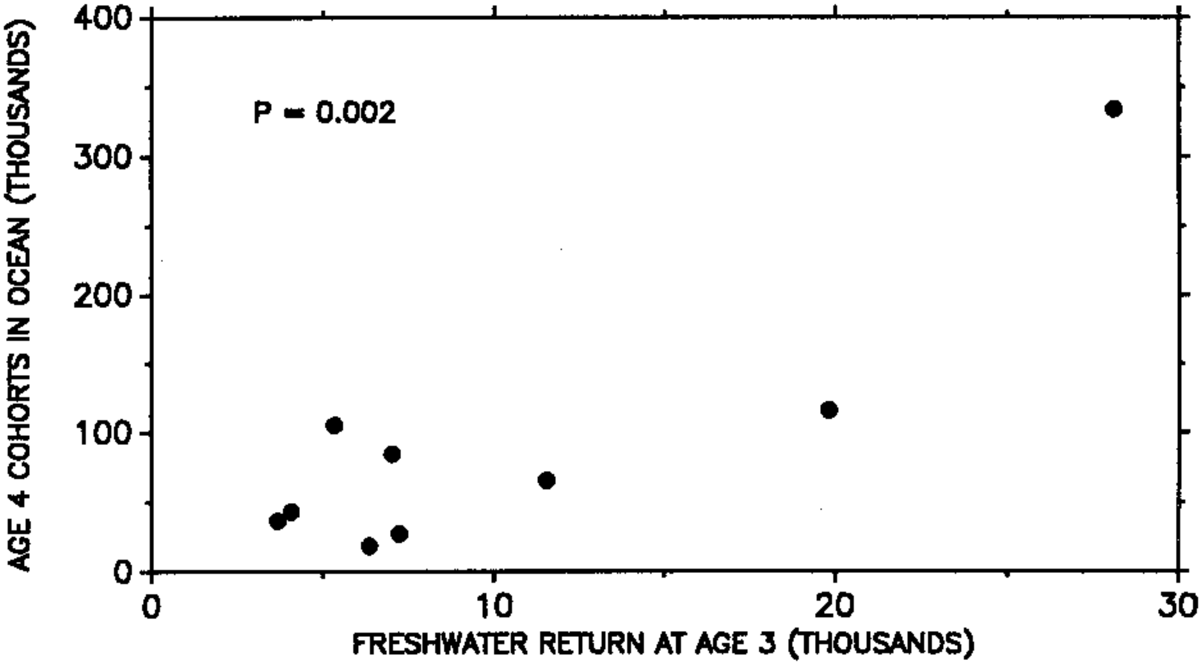

Figure 22. Relationship between estimated number of age 4 fall chinook salmon resident in the ocean during spring and estimated freshwater return of age 3 cohorts one year earlier, 1971-81 brood years. Data from 1979-80 broods were excluded because rates of natural mortality increased as a result of the El Niño event of 1982-83.

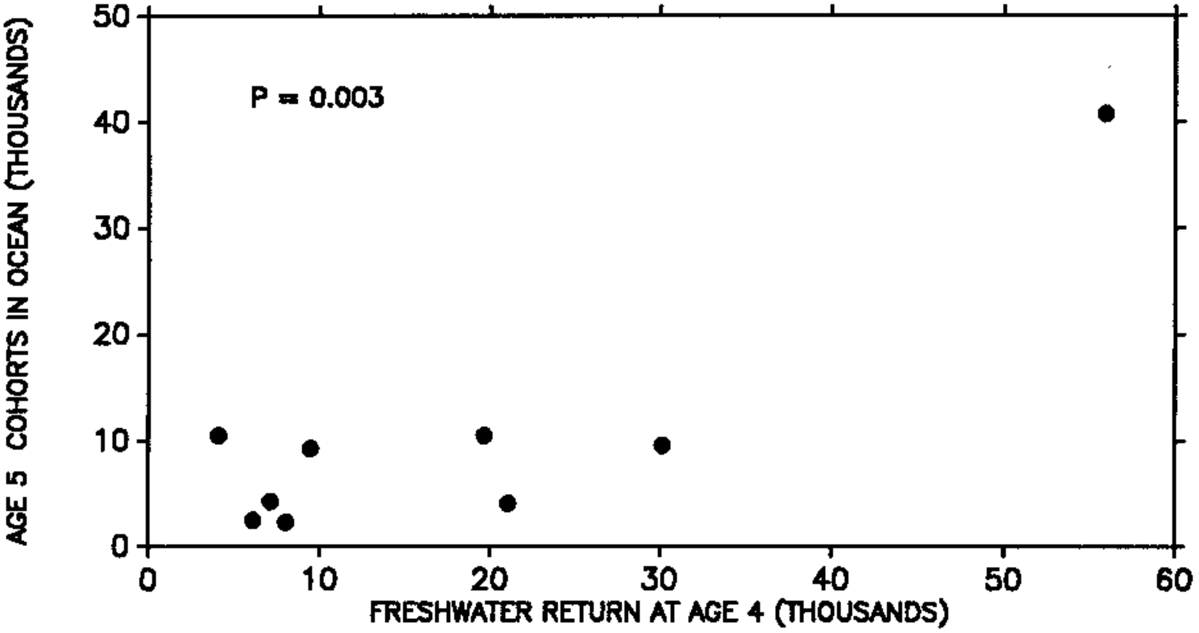

Figure 23. Relationship between estimated number of age 5 fall chinook salmon resident in the ocean during spring and estimated freshwater return of age 4 cohorts one year earlier, 1970-80 brood years. Data from 1978-79 broods were excluded because rates of natural mortality increased as a result of the El Niño event of 1982-83.

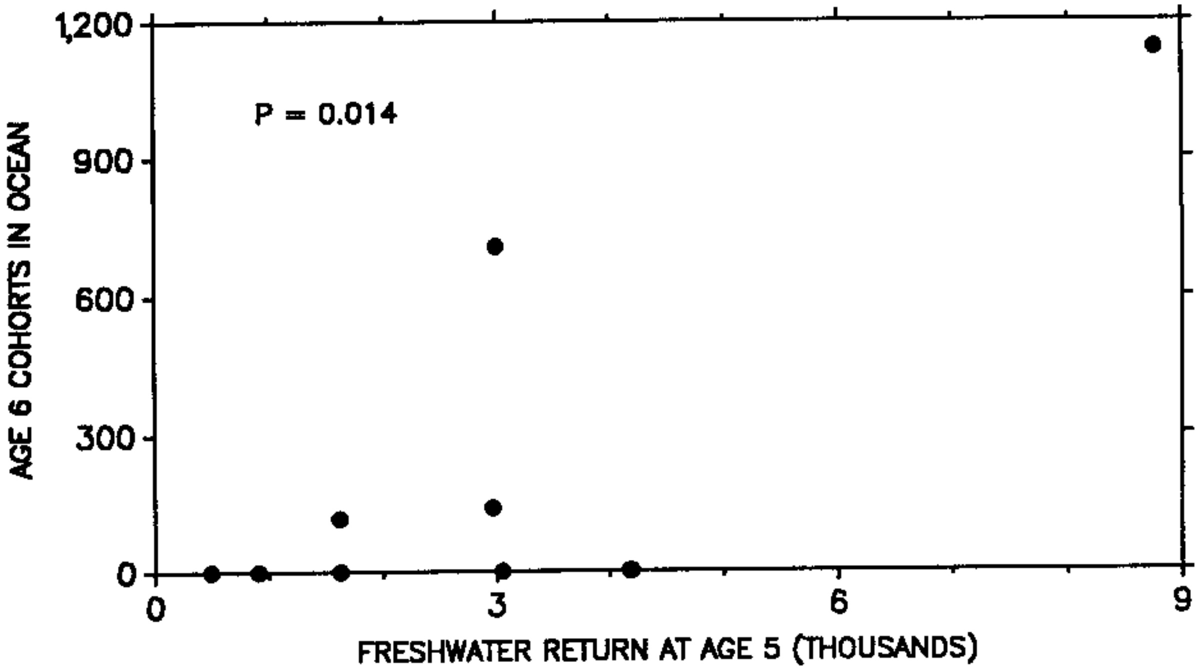

Figure 24. Relationship between estimated number of age 6 fall chinook salmon resident in the ocean during spring and estimated freshwater return of age 5 cohorts one year earlier, 1969-80 brood years. Data from 1977-78 broods were excluded because rates of natural mortality increased as a result of the El Niño event of 1982-83.

Table 13. Estimated rates of recruitment (freshwater return + ocean catch) for age 3-6 fall chinook salmon in the Rogue River basin, 1974-80 brood years. The 1981 brood year was assumed to produce no age 6 fish.

Brood year	Spawning escapement	Recruitment	Recruits per spawner
1974	29,160	199,481	6.8
1975	27,682	648,546	23.4
1976	10,366	152,029	14.7
1977	8,277	76,416	9.2
1978	48,392	240,829[a]	5.0
1979	10,762	188,409[a]	17.5
1980	8,384	91,660[a]	10.9
1981	24,294	32,323[a]	1.2

[a] *Estimates were not adjusted for increased mortality that resulted from the El Niño event of 1982-83.*

Effects of El Niño: We concluded that the El Niño event of 1982-83 increased rates of natural mortality for all ages of fall chinook salmon resident in the ocean. Estimates of ocean abundance for 1983 were less than predicted based upon freshwater returns of cohorts in 1982 for all age classes. Numbers of fish alive in spring of 1983, as estimated from the cohort analysis, represented only 10-41% of the number predicted from regression analyses (Table 14).

Because estimates of ocean abundance were affected by freshwater returns in later years, changes in maturity rates could have accounted for the low estimates of fish abundance in 1983. However, we believe increased rates of natural mortality accounted for the low estimates of fish abundance in 1983. We reached this conclusion based on the low number of cohorts that returned to the Rogue River in 1984-86.

We found that estimates of ocean abundance in spring of 1984 were less than predicted from freshwater returns of cohorts in 1983. Estimates of ocean abundance were 22-78% of predicted values (Table 14). This result indicated that rates of natural mortality remained high one year after the El Niño event of 1982-83. However, the effect of El Niño diminished as rates of natural mortality were less during 1983-84 compared with 1982-83 (Table 14).

Table 14. Comparisons of the predicted and observed number of wild fall chinook salmon alive in the ocean during 1983 and 1984. Observed numbers of fish taken from Appendix Table C-4. Predicted numbers were developed from regressions presented in Appendix Tables C-6 through C-10 and from freshwater escapement of cohorts during the previous year (see Appendix Table C-4).

Age	Brood year	Predicted ± 95% CI	Observed	Percent of predicted
		1983		
2	1981	1,004,633 ± 288,609[a]	101,128	10.1
3	1980	429,382 ± 97,933	177,143	41.3
4	1979	133,578 ± 48,769	34,449	25.8
5	1978	13,745 ± 6,234	2,534	18.4
		1984		
2	1982	650,577 ± 238,581	143,685	22.1
3	1981	45,246 ¯ 32,889-62,245[b]	35,396	78.2
4	1980	77,330 ± 44,032	15,905	20.6
5	1979	3,080 ± 7,375	853	27.7

[a] *Juveniles were not sampled in 1982. We predicted juvenile abundance with the regression in Appendix Table C-6 and data in Appendix Table B-3.*

[b] *Confidence limits shown because data were transformed to natural logarithms.*

We believe effects of the El Niño event of 1982-83 lingered through at least 1984-85. Freshwater return of age 2 fall chinook salmon in 1985 was greater than predicted from seine catches of juveniles in the Rogue River during 1984. However, older adults returned to the Rogue River in 1985 at rates commensurate with freshwater returns of cohorts in 1984. Predictions were derived from regressions presented in Appendix Table B-4 and Appendix Tables C-7 through C-10. We suspect juveniles from the 1983 brood year survived at an unusually high rate after ocean entry. Ocean populations of competitors, predators, parasites, or disease organisms could have decreased during the El Niño event of 1982-83 and recovered at slower rates than salmonid populations.

We concluded that increased mortality resulted in estimates of ocean abundance in 1983 that were 10-41% of values predicted for fall chinook salmon of Rogue River origin. Other populations of salmon were also significantly affected by the El Niño event of 1982-83. Johnson (1988) estimated the 1983 abundance of age 3 coho salmon in the Oregon Production Index Area was 42% of expected. Increases in natural mortality also probably occur during weaker El Niño events, which occur about twice as frequently as strong events (Mysak 1986).

Fisher and Pearcy (1988) concluded that increased rates of natural mortality of juvenile coho salmon that reared off the coasts of Oregon and Washington in 1983 and 1984 resulted from increased predation rather than a decreased food supply. They hypothesized mortality rates may have increased because predation increased. In general, predation affects anadromous salmonids in a size-dependent manner (Ricker 1969; Ricker 1976; Mathews and Buckley 1976; Healey 1982). Our data indicated predators would have had to have been large animals to account for the increased rates of natural mortality among older fall chinook salmon.

Estimated increases in rates of natural mortality during, and just after, the El Niño event of 1982-83 appeared independent of the age of fall chinook salmon (Table 14). This finding contradicts results from other research that showed younger fish are more sensitive to environmental perturbations compared with older fish (Ricker 1975). Effects of environmental factors other than El Niño are discussed further in the following section. Because the El Niño of 1982-83 is regarded as one of the strongest of the century (Mysak 1986), we deemed it to be an anomalous event.

Primary determinants of abundance: We concluded juvenile production was the primary determinant of ocean abundance of age 2 fall chinook salmon during most years of the study. If, as our data implied, relationships between successive age classes were linear, then juvenile production was usually the primary determinant that affected recruitment to the ocean fisheries.

Other research has documented linear relationships among anadromous salmonids of successive ages. Johnson (1988) described a linear relationship between freshwater returns of age 2 jacks and the abundance of age 3 coho salmon in the Oregon Production Index Area. Peterman (1982) found linear relationships for four stocks of age 3-6 sockeye salmon in Alaska and British Columbia. He also concluded there was a linear relationship between age 2-3 coho salmon in the Oregon Production Index Area. Correlation coefficients for these regressions averaged 0.72 and ranged between 0.40 and 0.96. The relationship between grilse harvest and two-sea winter Atlantic salmon in 15 Iceland rivers were judged to be linear or nearly linear by Scarnecchia (1984). Correlation coefficients for relationships of Atlantic salmon of successive ages averaged 0.73 and ranged between 0.52 and 0.93. Preseason forecasts developed with linear relationships between successive age classes for four populations of chinook salmon in the Columbia River basin differed, on the average, from actual returns by 17-24% (PFMC 1991).

We found correlation coefficients between successive ages of fall chinook salmon averaged 0.73 and ranged between 0.55 and 0.86 for age 3-6 fish. Although these correlation coefficients are comparable to values derived from other research, the strong 1975 brood year was primarily responsible for coefficient values between fish of ages 4-6. Non-linear models have been judged to be more accurate predictors of abundance for some populations of sockeye salmon (Bocking and Peterman 1988).

Environmental conditions in the ocean have been identified in numerous studies as important determinants for production of adult salmonids. Nickelson (1986) found negative relationships between ocean temperature in years of high upwelling and survival rates of wild and hatchery coho salmon in the Oregon Production Index Area. Ocean temperature also accounted for a significant amount of variation in the number of fall chinook salmon that returned to the Columbia River during 1938-46 (Van Hyning 1973).

Our finding of a linear relationship between the abundance of juveniles in fresh water and age 2 cohorts in the ocean suggested ocean recruitment of fall chinook salmon of Rogue River origin was primarily determined by juvenile production, except during unusual ocean conditions. Use of beach seines to index juvenile abundance may temper this conclusion. However, seine catches of juvenile fall chinook salmon in Elk River, Oregon, also correlated positively with returns of adults (personal communication dated 7 May 1984 from Timothy Downey, ODFW, Port Orford, Oregon). Seine catches of subyearling striped bass in Chesapeake Bay were significantly related to subsequent landings of adults (Goodyear 1985). These findings suggest that environmental determinants of year class strength act early in the life cycle of some species of anadromous fish.

Summer flow has been associated with production of salmonids. Increased flow in a small Maine stream was associated with increased survival rate of subyearling Atlantic salmon (Havey and Davis 1970). Standing crops of resident trout are positively related with flow in late summer for streams of the western portion of the United States (Binns and Eiserman 1979; Conder and Annear 1987).

Flow in the first year of freshwater residence has been related to the production of coho salmon (Mathews and Olson 1980; Anderson and Wilen 1985). Barton (1979) concluded greater flow during juvenile migration to the ocean increased returns of adult spring chinook salmon to streams in the upper portion of the Columbia River basin. Increased flow may have increased survival rate of juveniles that passed hydroelectric facilities. We found that flow of the Rogue River in the summer was not related with the subsequent abundance of fall chinook salmon in the ocean.

Biological characteristics of migrants appeared to have little affect on ocean recruitment of fall chinook salmon of Rogue River origin. However, failure to detect affects may have resulted from small sample sizes or narrow ranges observed within biological characteristics. For example, mean lengths of juvenile migrants varied less than 1.5 cm annually. Experiments designed to evaluate the influence of size on survival of hatchery fish used groups of more varied sizes.

Bilton (1984) estimated freshwater returns to be five times greater for chinook salmon released at mean weights of 12 g compared with 6 g. Green and Macdonald (1987) concluded release time and size at release affected the number of chinook salmon that returned to Robertson Creek in British Columbia. However, unspecified variation in survival rates between brood years had a greater affect on adult returns than release time or size at release. This finding suggested environmental conditions after release from hatcheries strongly affect survival rates of chinook salmon.

We believe additional data, particularly from strong brood years, are needed to better identify factors that affect the ocean abundance of fall chinook salmon of Rogue River origin. Improved estimates of fish abundance are needed to identify factors that affect survival rates in the ocean, particularly during El Niño events. Such information would help identify factors that affect abundance of chinook salmon resident in the ocean.

Our findings indicated that Lost Creek and Applegate lakes can be managed to increase ocean recruitment of fall chinook salmon produced in the Rogue River basin (*see* **Abundance**, page 39). Our results indicated that ocean abundance at age 2 was usually limited by juvenile production in fresh water. We estimated the production of juvenile spring chinook salmon in the Rogue River during 1975-90 (Satterthwaite 1991) and fall chinook salmon in the Applegate River during 1979-88 (ODFW, unpublished data). These estimates of juvenile abundance should be compared to abundance estimates of age 2 cohorts in the ocean.

Spawner Abundance

Spawning escapement of fall chinook salmon in the Rogue River basin varied between years. Annual estimates of spawning escapement ranged between 700 and 36,000 age 2 jacks in 1974-86. Estimates of spawning escapement of age 3-6 adults ranged between 8,000 and 54,000 fish annually. Estimates of spawning escapement were lowest in 1979-80 when large numbers of adults died prior to spawning. Spawning escapement peaked in 1986 when broods produced after the El Niño event of 1982-83 dominated freshwater returns.

We found significant relationships between counts of spawned carcasses and estimates of spawning escapement (Figure 25). Regression analyses indicated carcass recoveries accounted for 56% of the variation in estimates of spawning escapement for age 2 jacks and 63% of the variation in estimates of spawning escapement for older adults (Appendix Table C-12). Significant correlations indicated both sets of data showed similar trends in spawning escapement of fall chinook salmon in the Rogue River basin. Jonasson and Lindsay (1988) reported a positive correlation between carcass counts and spawning escapement of fall chinook salmon in the Deschutes River of Oregon.

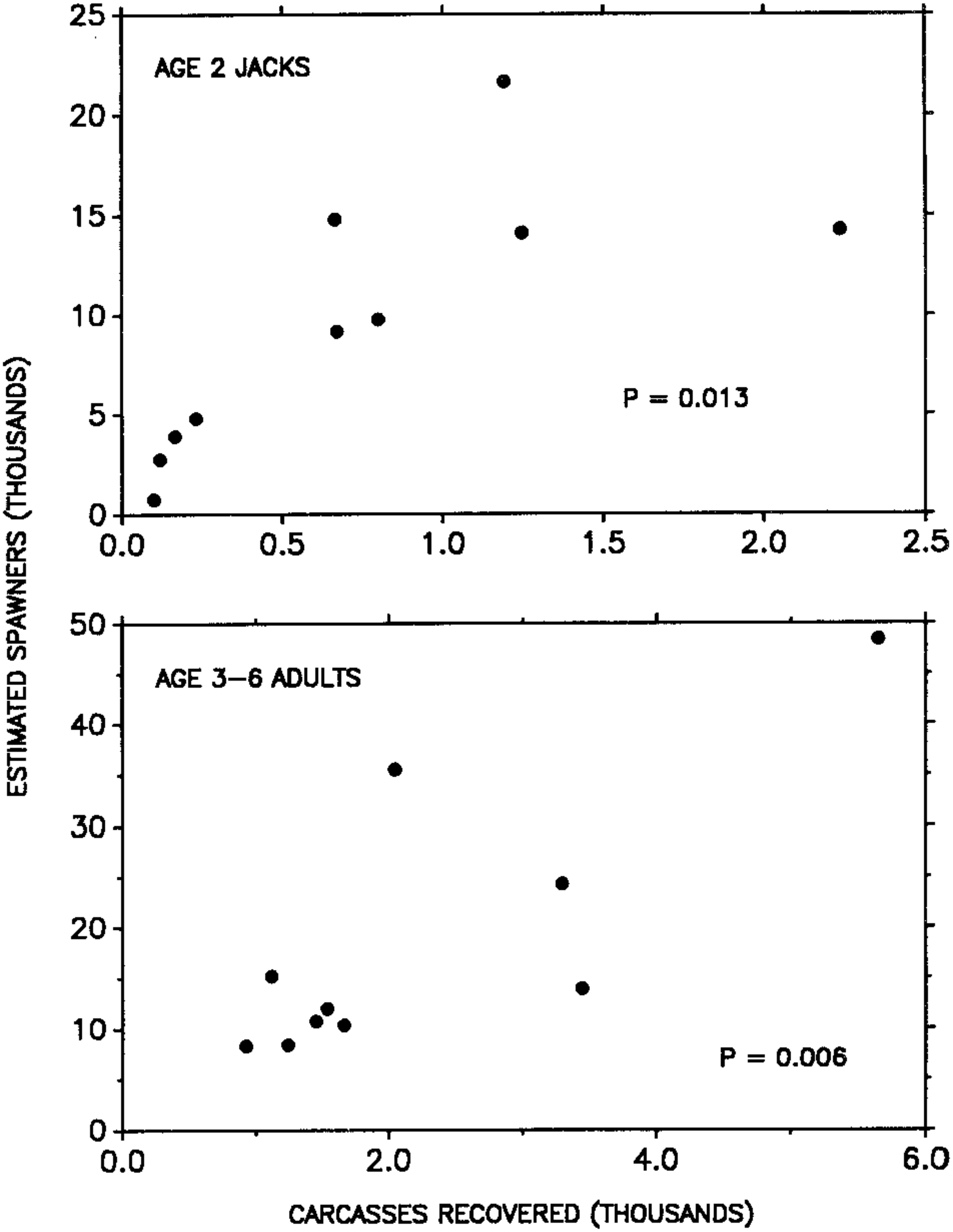

Figure 25. Relationships between estimated spawning escapement of fall chinook salmon and number of spawned carcasses recovered in standard survey areas, 1976-85.

Based on these findings, we concluded data from carcass surveys did not contradict estimates of spawning escapement derived from seine catches at Huntley Park. However, results from work with fall chinook in the Rogue River basin during 1987-91 appear to contradict this conclusion (ODFW, unpublished data). Additional data from years of large spawning escapements is desirable because we found a large number of carcasses in only one year of sampling.

We estimated survey personnel recovered an average of 10% of the fall chinook salmon that spawned in the Rogue River basin (Appendix Table C-13). Surveyors recovered age 3-6 adults at a greater rate ($P = 0.017$) than age 2 jacks. We estimated rates of carcass recovery averaged 7.7% for jacks and 13.1% for adults. Differential rates of recovery for salmonid carcasses of differing sizes have also been documented (Solazzi 1984; Boecheler and Jacobs 1987).

Management strategies for Lost Creek Lake affected spawning escapement of fall chinook salmon because changes in water quality affected rates of prespawning mortality and angler harvest. These findings are presented in other sections of the report (*see* **Prespawning Mortality**, page 74 and **Freshwater Harvest**, page 78).

Age at Maturity

Scale analyses indicated that fall chinook salmon returned to the Rogue River at ages 2-6. We estimated that freshwater returns in 1974-86 averaged 29% age 2 fish, 28% age 3 fish, 36% age 4 fish, 7% age 5 fish, and less than 1% age 6 fish. Estimates of the age composition of annual returns are in Appendix Table C-14.

We concluded age composition of freshwater returns did not truly reflect age composition of fall chinook salmon that originated from the Rogue River basin. Age-selective harvest in the ocean fisheries decreased the relative abundance of older adults among freshwater returns. Ocean fisheries harvested age 4 fish at greater rates than age 3 fish among chinook salmon that originated from hatcheries in the Rogue and Klamath river basins (Hankin 1990). Chinook salmon that mature at older ages are exposed to ocean fisheries for more years than cohorts that mature at younger ages (Ricker 1980; Hankin and Healey 1986). We concluded estimation of maturity rates was a more appropriate estimate of the age structure for a population of chinook salmon.

We estimated maturity probabilities of fall chinook salmon that originated from the Rogue River basin averaged 0.025 for age 2 fish, 0.091 for age 3 fish, 0.581 for age 4 fish, and 0.947 for age 5 fish. These estimates of maturity rates apply only to those broods not affected by increased rates of natural mortality that resulted from the El Niño event of 1982-83 (Table 15).

Age at maturity varies greatly among populations of chinook salmon. In general, age at maturity increases with increasing latitude (Healey and Heard 1984; Hankin and Healey 1986). An older age at maturity may be a genetic adaptation to slower rates of freshwater growth at higher latitudes (Roff 1984). Our estimates of maturity rates for fall chinook salmon approximate maturity rates of a midmaturing stock as defined by Hankin and Healey (1986).

Table 15. Estimated probability of maturity for wild fall chinook salmon that originated in the Rogue River basin, 1969-84 brood years.

Brood year	Maturation probability			
	Age 2	Age 3	Age 4	Age 5
1969	--	--	--	0.772
1970	--	--	0.600	1.000
1971	--	0.124	0.806	1.000
1972	0.054	0.175	0.739	1.000
1973	0.022	0.071	0.240	1.000
1974	0.020	0.039	0.717	1.000
1975	0.014	0.063	0.523	0.860
1976	0.033	0.062	0.451	0.944
1977	0.021	0.074	0.573	1.000[a]
1978	0.019	0.120	0.577[a]	0.736[a]
1979	0.016	0.080[a]	0.614[a]	1.000[a]
1980	0.017[a]	0.084[a]	0.664[a]	0.917[a]
1981	0.029[a]	0.288[a]	0.684[a]	0.959[a]
1982	0.029[a]	0.077[a]	0.612[a]	--
1983	0.011[a]	0.088[a]	--	--
1984	0.017[a]	--	--	--

[a] *Estimates affected by increased rates of natural mortality that resulted from the El Niño event of 1982-83 or by assumptions for maturation rates of immature cohorts remaining in the ocean after 1986* (see *Appendix Table C-4*).

Estimates of maturation rates were highly variable between brood years. Probability of maturation ranged from 0.014 to 0.054 for age 2 fish and ranged from 0.039 to 0.175 for age 3 fish. Probability of maturation ranged between 0.240 and 0.806 for age 4 fish and ranged between 0.772 and 1.0 for age 5 fish (Table 15).

Data were insufficient to test for changes in maturation rates with the exception of age 2 fish. Sample sizes for postimpoundment broods not affected by El Niño were N = 3, 2, 1, and 0 for age 2, age 3, age 4, and age 5 fish, respectively. We detected no significant difference (P = 0.33) in maturity rates of age 2 fish from preimpoundment and postimpoundment broods. We concluded the statistical power of this test was low because a sensitivity analysis indicated that average maturity rates for postimpoundment broods would have had to increase to 7.1% or decrease to 0.3% to be detected by parametric statistics.

We were unable to identify factors related to maturation rates of fall chinook salmon of Rogue River origin. Maturity rates of age 2 fish were negatively related with estimates of scale radius (a proxy of body length) at time of formation of the first annulus. However, we believe the correlation was spurious because faster growth should result in a younger age at maturity

(Hankin 1990). Date of ocean entry and growth during ocean residence were not significantly related to estimates of maturity rates for fall chinook salmon of any age. Data used in these analyses are in Appendix Tables C-15 through C-17. Correlation matrixes that outline relationships among all variables examined in these analyses are in Appendix Table C-18.

Contribution of multiple populations to the run of fall chinook salmon in the Rogue River complicated analyses of maturity rates. Our data indicated that, as a conglomerate, fall chinook salmon produced in the Rogue River basin matured at older ages compared with the component produced in the Applegate River. Hankin and Healey (1986) estimated maturity probabilities of the Applegate population as 0.042 at age 2 and 0.282 at age 3. We estimated maturity probabilities of pooled Rogue populations as 0.025 at age 2 and 0.091 at age 3.

We also found that jacks composed a significantly greater percentage of the spawned carcasses found in the lower portion of the Applegate River compared with nearby areas of the Rogue River. We believe differences in age distribution indicates fall chinook salmon in the Applegate River compose a population that is distinct from other fall chinook salmon in the Rogue River basin. Existence of multiple populations of chinook salmon in a single river basin have been documented from surveys of genetic resources (Kristiansson and McIntyre 1976; Beacham et al. 1989; Bartley and Gall 1990).

Results from other research showed that heredity and growth rate affect maturation rate of chinook salmon. Maturity age of progeny appears strongly influenced by parental age (Donaldson and Menasveta 1961; Downey et al. 1986; Beacham et al. 1989). Changes in growth rate can change maturation rates of chinook salmon. Later ocean entry and slower rate of freshwater growth delayed maturation among spring chinook salmon and fall chinook salmon released at hatcheries in the Rogue and Klamath river basins (Hankin 1990). Neilson and Geen (1986) found a negative correlation between length at time of first annulus formation and age at maturity for male chinook salmon from the Sixes River in Oregon. Bilton (1984) found a negative correlation between mean weight of juvenile chinook salmon at time of hatchery release and mean age of returning adults. The tendency for faster growing chinook salmon to mature at younger ages was also noted by Parker and Larkin (1959) and by Schluchter and Lichatowich (1977).

Operation of Lost Creek Dam could have affected maturity rates of fall chinook salmon produced in the Rogue River. Postimpoundment broods grew faster in fresh water and entered the ocean earlier than preimpoundment broods (*see* **Time of Ocean Entry:**, page 50). Because chinook salmon grow faster in the ocean than in fresh water (Hankin 1990), fish of postimpoundment origin may have matured at younger ages. However, we were unable to estimate the effects of reservoir operation on maturity rates because we could not differentially estimate freshwater growth and maturity rates for fish that reared only in the Rogue River versus fish that reared for some time in tributary streams.

Ocean Harvest

We estimated ocean fisheries harvested an average of 163,000 wild fall chinook salmon of Rogue River origin annually in 1974-86. Annual landings averaged about 101,000 age 3, 49,000 age 4, 6,000 age 5, and 200 age 6 fish. Age 3 fish composed the majority of the harvest because ocean fisheries harvested immature as well as maturing chinook salmon (Van Hyning 1973; Ricker 1980).

Ocean harvest of fall chinook salmon of Rogue River basin origin varied between years. Annual estimates of ocean landings ranged between 6,000 and 349,000 for age 3 fish, and ranged between 5,000 and 227,000 for age 4 fish (Appendix Table C-4). Individual broods contributed between 45,000 and 555,000 wild fish to ocean fisheries. The strong 1975 brood year accounted for the greatest harvest, while brood years affected by the El Niño event of 1982-83 accounted for the smallest harvests.

Fall chinook salmon from the Rogue River basin contribute primarily to ocean fisheries off the coasts of California and Oregon (Garrison 1987). Few are landed in Washington, British Columbia, or Alaska. Estimates of catch distribution were derived from recoveries of tagged fall chinook salmon landed at coastal ports. These experimental fish were reared and tagged at Cole M. Rivers Hatchery. Harvest appeared evenly distributed between California and Oregon. Landings of age 3 and age 4 fish exhibited similar distribution patterns. Landing patterns for age 2 and age 5 fish were not assessed because size regulations prevented retention of almost all age 2 fish and few fish of hatchery origin remained in the ocean to mature at age 5.

The area between Fort Bragg, California, and Coos Bay, Oregon, accounts for most of the landings of fall chinook salmon from the Rogue River basin (Nicholas and Hankin 1988). Within this area, commercial trollers average more than 90% of the harvest (McGie 1987; PFMC 1988). Because trollers accounted for most of the ocean harvest, we examined the influence of effort by the commercial fleet on the exploitation rate of chinook salmon.

We found that exploitation rate increased with increased effort in the troll fishery between Eureka, California, and Coos Bay (Figure 26). Boat-days of effort in the troll fishery were significantly correlated ($P = 0.053$) with exploitation rates estimated for age 3 chinook salmon. Ocean abundance of chinook salmon and size of age 3 chinook salmon were not related to exploitation rates of age 3 fish. Data from fall chinook salmon of Rogue River origin were used as a proxy for fish abundance in the ocean and the size of age 3 fish. Data included in these analyses are in Appendix Table C-19. A correlation matrix that outlines relationships among all variables examined in these analyses is in Appendix Table C-20.

We also found that exploitation rate of age 4 chinook salmon was significantly related ($P = 0.037$) with boat-days of effort in the troll fishery (Figure 26). Ocean abundance of fall chinook salmon of Rogue River origin was significantly related with fishing effort by trollers ($P = 0.060$). Regression analysis indicated exploitation rates increased as effort increased, but not on a proportional basis. For example, the analysis predicted that a doubling of effort from 20,000 to 40,000 boat-days would increase the exploitation rate on age 4 chinook salmon from 50% to 64%. When more restrictive harvest regulations were imposed in the mid-1980s to increase freshwater return of fall chinook salmon of Klamath River origin, effort in the troll fishery decreased and resulted in decreased exploitation rates on fall chinook salmon of Rogue river origin.

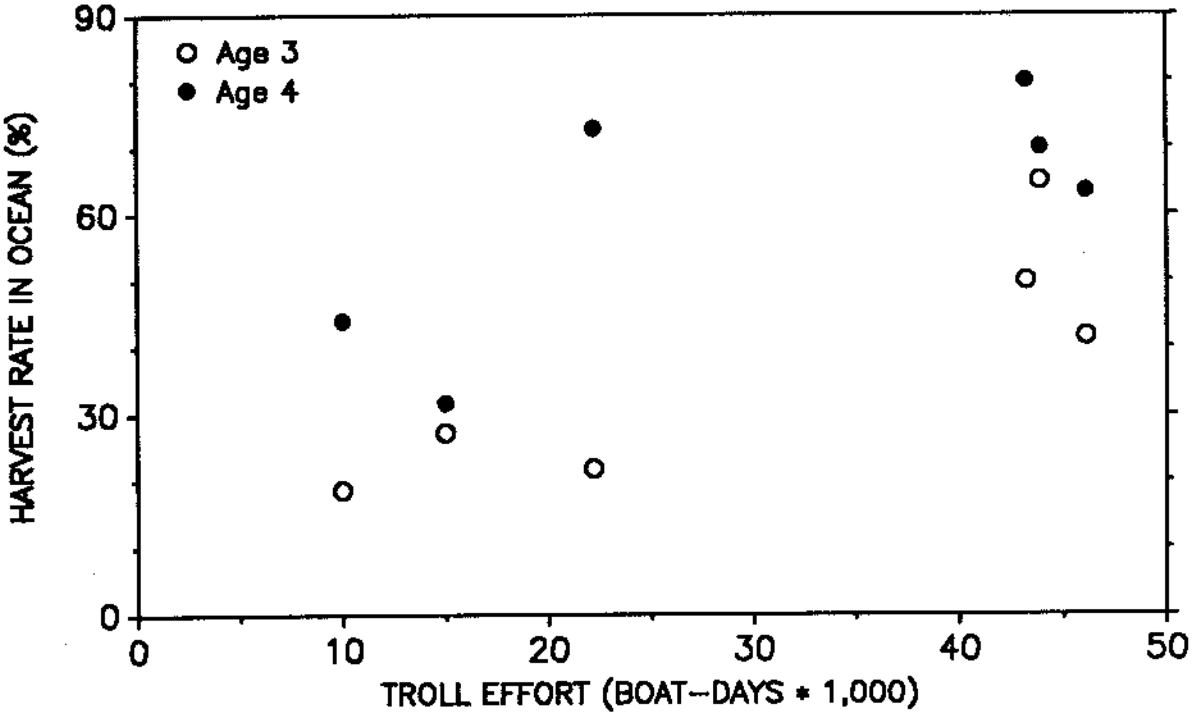

Figure 26. Relationships between ocean harvest rates of age 3 and age 4 fall chinook salmon and effort in commercial fisheries between Eureka, California, and Coos Bay, Oregon, 1980-85.

Exploitation rate is affected by factors other than fishing effort. Variations in fish abundance and ocean conditions affect availability of chinook salmon for harvest (Healey et al. 1990). Economic conditions affect number and duration of trips made by commercial operators. As more information becomes available, it may be possible to estimate how these factors affect exploitation rate of fall chinook salmon of Rogue River origin.

We estimated that ocean fisheries harvested an average of 78% of the age 3 and older fish produced from the 1972-80 broods of fall chinook salmon that originated from the Rogue River basin. Among the 1972-80 broods, ocean fisheries harvested 65-84% of the brood production (Table 16). Chapman (1986) concluded 68% was an optimum harvest for chinook salmon stocks of British Columbia and Columbia River origin. Reisenbichler (1987) concluded brood harvests of 60-70% are reasonable for many stocks of chinook salmon. A brood harvest of 67% was estimated for maximum sustained yield of fall chinook salmon in the Klamath River basin (personal communication dated 29 May 1992 from David Hankin, Humboldt State University, Arcata, California).

These findings indicated fall chinook salmon of Rogue River basin origin were harvested at greater than optimum rates in the late 1970s and early 1980s. Effects of excessive harvest on subsequent recruitment were least on the early maturing population in the Applegate River. Effects on recruitment of older maturing populations in other areas of the basin must have been greater and should be investigated. For example, excessive rates of ocean

Table 16. Estimated ocean harvest of wild fall chinook salmon that originated in the Rogue River basin, 1972-80 broods. Harvest estimates were derived from the cohort analysis in Appendix Table C-4.

	Age 2-6 fish		
Brood year	River returns	Ocean harvest	% of brood harvested
1972	24,261	45,242	65
1973	16,784	70,537	81
1974	48,272	160,916	77
1975	114,859	555,351	83
1976	31,635	132,498	81
1977	16,023	63,951	80
1978	55,325	196,572	78
1979	31,573	167,353	84
1980	31,528	74,966	70
1981	16,692	15,631	48
Average	38,695	148,302	75

harvest may be primarily responsible for depressed returns of fall chinook salmon to tributaries of the lower river. Estimation of age structure for populations of fall chinook salmon in the Rogue River basin would allow for better assessments of the effect of harvest on production.

Migration Timing

Fall chinook salmon entered the Rogue River primarily in August-October. Freshwater entry usually peaked between the middle of August and the middle of September (Figure 27). A few fall chinook salmon also enter the Rogue River in November and December (Rivers 1964).

Fall chinook salmon entered the Rogue River early compared with the time of freshwater entry for runs in other coastal streams of Oregon. Only the run in the Umpqua River enters fresh water at a time similar to the run in the Rogue River (Nicholas and Hankin 1988). In the Klamath River of northern California, fall chinook salmon enter fresh water earlier than the run in the Rogue River (Tuss et al. 1988). Fall chinook salmon in other coastal streams of northern California enter fresh water later compared with runs in the Klamath and Rogue rivers (Moyle 1976).

We found that variations in time of freshwater entry resulted primarily from variations in the relative abundance of populations that composed the run. Time of freshwater entry, coupled with results from spawning surveys, suggested there are a minimum of five distinct populations of fall chinook salmon in the Rogue River basin. Mean time of river entry differed significantly among some of these populations (Table 17).

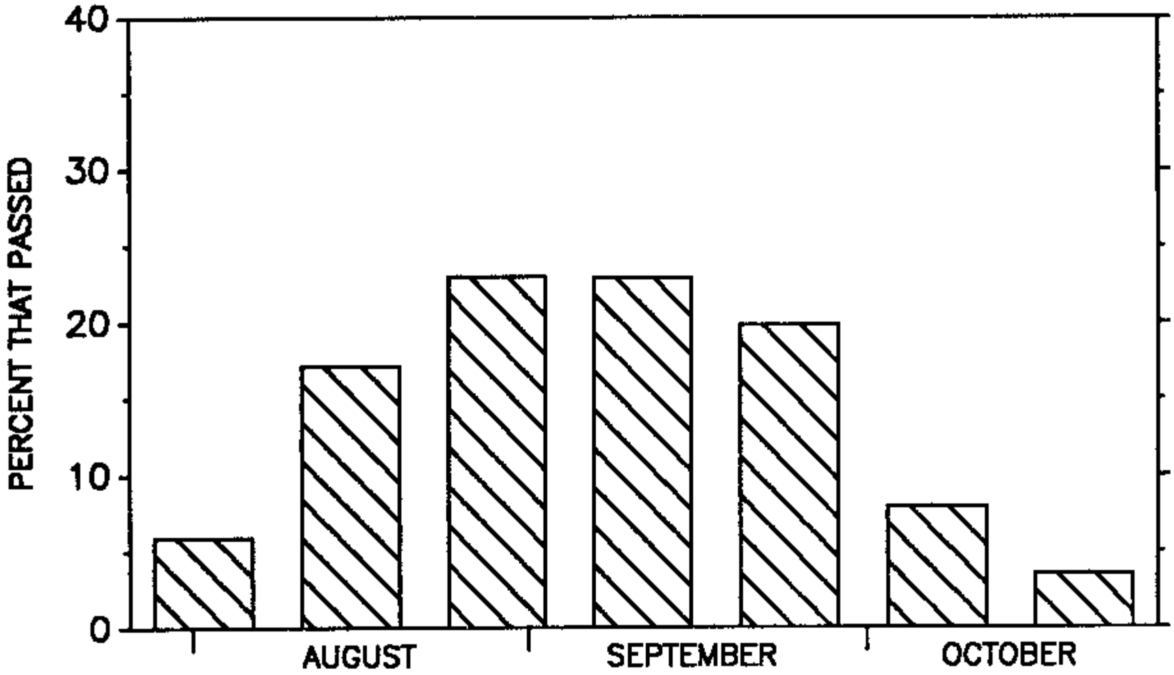

Figure 27. Migration timing of fall chinook salmon that passed Huntley Park, averaged for 1976-86.

Table 17. Mean date of freshwater entry for fall chinook salmon tagged in the lower river and later recovered as spawned carcasses, 1974-78.

Area of tag recovery	*N*	Mean date of tagging ± 95% CI
Rogue River, RK 204-228	5	08/21 ± 15 days
Rogue River, RK 139-199	41	08/27 ± 5 days
Applegate River, RK 0-21	102	09/13 ± 3 days
Illinois River basin	16	09/28 ± 4 days
Slate and Waters creeks[a]	14	10/01 ± 6 days
Lobster and Quosatana creeks[b]	29	10/02 ± 5 days

[a] *Tributaries of the lower portion of the Applegate River.*
[b] *Tributaries of the lower portion of the Rogue River.*

Adults destined to spawn in the Rogue River upstream of the canyon entered fresh water first. They were followed by adults destined to spawn in the Applegate River downstream of Murphy Dam (RK 21). Adults destined to spawn in the Illinois River basin and tributaries of the lower portions of the Rogue and Applegate rivers entered last.

Annual variations in run composition produced annual variations of run timing in the lower river. For example, spawning surveys indicated that adults that returned to tributaries of the lower river composed large portions of the 1974-75 runs, but composed a negligible portion of runs in the early 1980s. Coincidentally, estimates of freshwater returns indicated returns in October decreased in relative abundance compared with the remainder of the run. During 1974-75, 21% of the adults entered fresh water in October. During 1980-81, 9% of the run entered fresh water in October. What appeared to be early runs in the early 1980s probably resulted from a decrease in the relative abundance of populations adapted to enter fresh water late in the run.

Other research has identified differences in run timing between distinct populations within runs of chinook salmon (Burger et al. 1985; Merritt and Roberson 1986). Mundy (1982) hypothesized that populations of chinook salmon are genetically adapted for return time, but environmental conditions modulate the response. Tagged adult steelhead that made a repeat spawning run in the Kalama River, Washington, migrated at the same time as their initial run (Leider 1985). Coho salmon of hatchery origin in Oregon returned to fresh water earlier than wild counterparts because of broodstock selection practices at hatcheries (Nickelson et al. 1986). Heredity also appeared to strongly influence the migration timing of chinook salmon in Elk River, Oregon (Downey et al. 1986).

However, we also found differences in mean time of freshwater entry for individual components of the run. Tagging studies indicated adults destined to spawn in the middle river entered fresh water an average of 13 days earlier in 1975 compared with 1978. Mean annual dates of freshwater entry also differed significantly between years for adults destined to spawn in the Applegate River.

We were unable to evaluate the effects of environmental factors on the time of river entry because tag recoveries were sufficient to estimate migration timing for individual populations in only three of five years when tags were applied. Environmental factors were related to the migration timing of fall chinook salmon that migrated to the upper river.

Fall chinook salmon passed the counting station at Gold Ray Dam from 16 August through the middle of November. Adults that passed the counting station migrated later after operation of Lost Creek Dam (Figure 28). Passage at the counting station by 15 September averaged 58% of the 1970-77 returns and 41% of the 1978-86 returns. The difference in means was significant at $P = 0.035$.

Augmented flow in September may have encouraged a greater proportion of late migrating adults to pass Gold Ray Dam. We found a correlation ($r = 0.42$, $P = 0.048$) between mean flow at Raygold in September and the proportion of fish that passed the counting station by 15 September. Increased flow in September also appeared to increase the proportion of the run that spawned upstream of Gold Ray Dam (*see* **Spawning Distribution**, page 81).

Although we found that operation of Lost Creek Dam affected migration timing of fall chinook salmon destined to spawn farthest upstream in the Rogue River, we developed no recommendations for an operating strategy directed to migration timing. However, in the next section, we propose recommendations

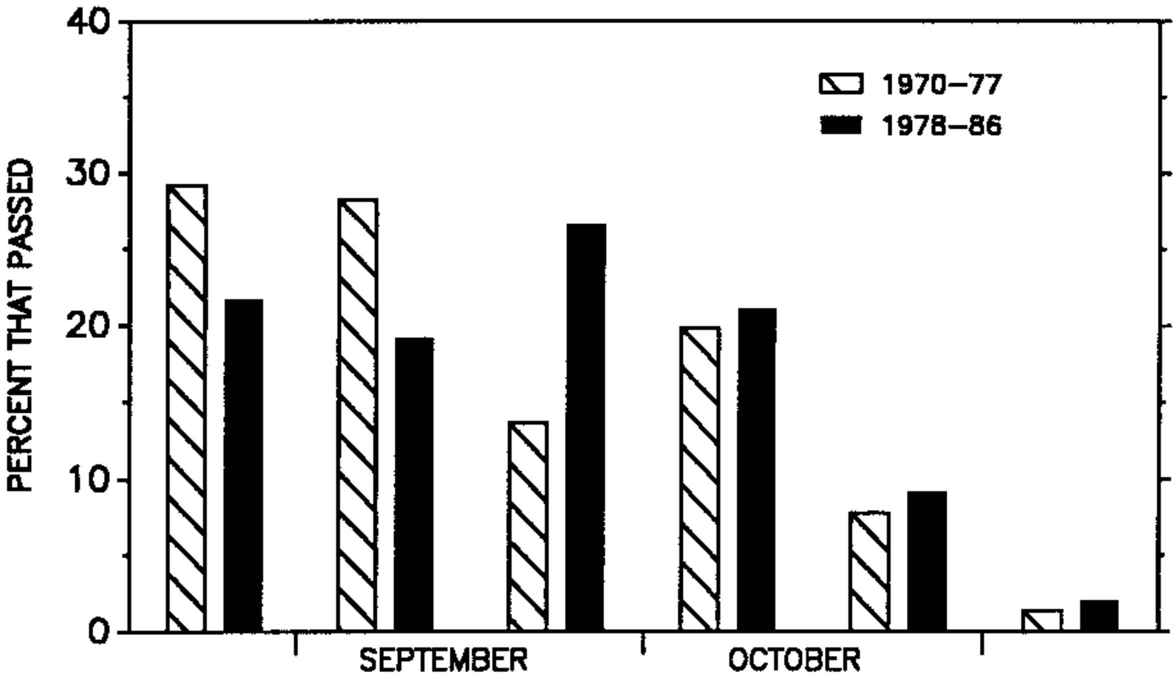

Figure 28. Migration timing of fall chinook salmon that passed Gold Ray Dam, before (1970-77) and after (1978-86) operation of Lost Creek Dam.

for reservoir management to decrease rates of prespawning mortality among populations that compose the early components of the run. An increase in the relative abundance of adults adapted to return early will give the appearance of an earlier migration time at freshwater entry.

Prespawning Mortality

We annually found unspawned carcasses of fall chinook salmon in the Rogue River. Counts of prespawning mortalities were greatest in 1978-80, the first years after operation of Lost Creek Dam. We found few unspawned carcasses in preimpoundment years and later postimpoundment years. Prespawning mortality of Pacific salmon has been observed in other streams (Murphy 1985).

Based on the relationship of freshwater returns and counts of spawned carcasses later in the year, we estimated that annual rates of prespawning mortality ranged from less than 1% to 81%. Except in 1979 and 1980, we estimated less than 13% of the adults died prior to arrival at spawning areas (*see* Table 7, page 30). Large rates of mortality in 1978-80 produced much concern about juvenile production in 1979-81.

Although we estimated that rates of prespawning mortality were less than 1% in the preimpoundment phase of our project, biologists with the Oregon Game Commission documented periodic mortalities of fall chinook salmon in 1944-60 (Appendix Table C-21) and later years. These reports indicated large prespawning mortality of fall and spring chinook salmon in at least three years prior to operation of Lost Creek Dam.

We found prespawning mortality of fall chinook salmon mentioned in 13 of the 30 years of reports written by fish management personnel. Records of prespawning mortality were associated with years of low flow in summer. We found that mean flow at Raygold during July-August averaged 1,247 cfs during years with mortality and 1,526 cfs in years without mortality. The difference of 279 cfs was significant ($P = 0.007$). We could not compare water temperatures because records of water quality were incomplete prior to 1960. Based on this result, we evaluated river physical parameters and rates of prespawning mortality estimated for postimpoundment years.

We found the rate of prespawning mortality during 1978-86 was positively related with water temperature in the canyon during late summer (Figure 29). Mortality rate increased as water temperature increased. Regression analysis (Appendix Table C-22) predicted mortality rates of <1% at 19°C, 15% at 20°C, and 89% at 21°C. We did not test for additional influence of flow because water temperature and flow were highly correlated ($P = 0.002$). The abundance of spring chinook salmon and fall chinook salmon correlated poorly with mortality rate ($P = 0.91$ and 0.63, respectively). This finding suggested mortality rate was less affected by fish density than by water temperature. Data used in this analysis are in Appendix Table C-23.

We also concluded that changes in water quality accounted for the difference in time of prespawning mortality in 1978 compared with 1979 and 1980. Carcasses seen after 9 September accounted for only 1% of the mortalities counted in 1978, even though large numbers of fish continued to pass Huntley Park (Appendix Table C-3). In contrast, 41% and 78% of the carcasses in 1979 and 1980, respectively, were observed after 9 September.

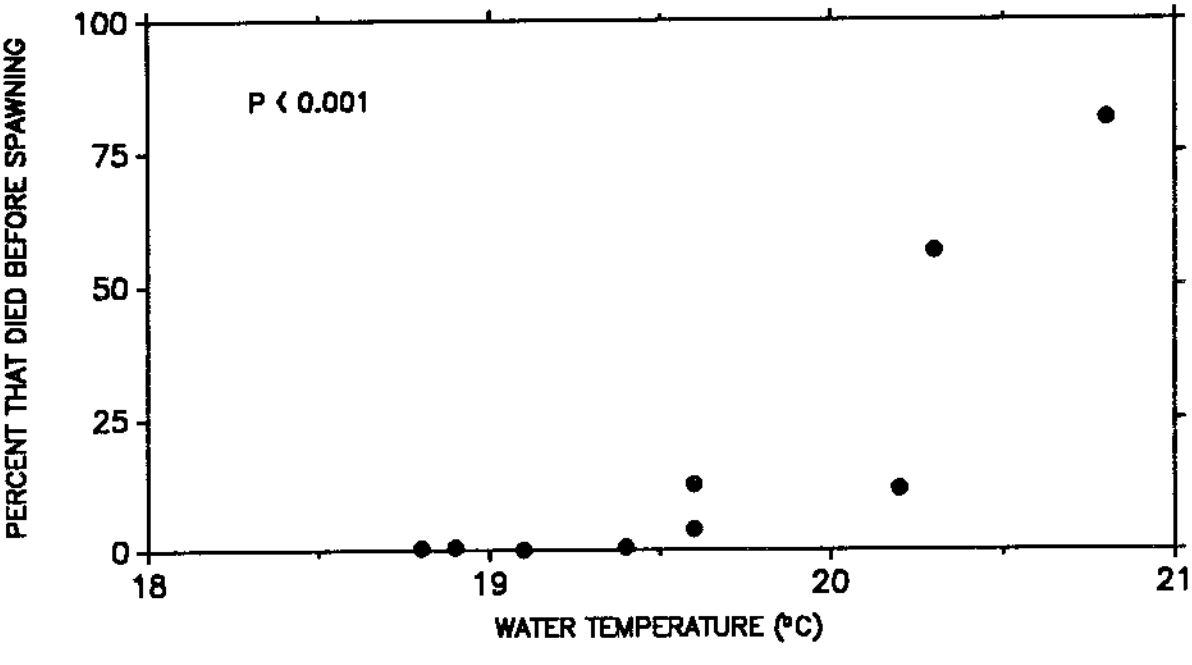

Figure 29. Relationship between the percentage of fall chinook salmon that died prior to spawning and mean maximum water temperature at Agness during August-September, 1978-86. Significance level was calculated from percentages transformed to logits.

Decreasing mortality in 1978 coincided with decreased water temperature and increased flow that resulted from a storm system moving through southern Oregon (Figure 30). During the same period in 1979, flow decreased, water temperature remained near 20°C, and carcass counts remained high through the end of September (Figure 30). Based on these results, we concluded decreased water temperature and maybe increased flow reduced prespawning mortality in 1978.

We believe disease probably caused prespawning mortality. No acutely toxic levels of chemicals or heavy metals were found in water samples or in tissue samples taken from adults (Cramer and McPherson 1983). Several disease organisms were isolated from dead and dying fish, but cause of death could not be attributed to any one organism (Amandi et al. 1982).

Flexibacter columnaris, a bacterial pathogen, was the disease organism most often isolated from dead and dying fall chinook salmon sampled in the Rogue River (Amandi et al.1982). Virulence of this bacterium varies among strains and epizootics may occur intermittently (Becker and Fujihara 1978). Mortality rates of juvenile chinook salmon infected with *Flexibacter columnaris* increase as water temperature increases between 12°C and 21°C (Holt et al. 1975; Becker and Fujihara 1978). Fall chinook salmon in the Rogue River are annually exposed to water temperatures at or above the upper end of this range.

Flexibacter columnaris was detected in resident fish in Lost Creek Lake and in juvenile chinook salmon held in the reservoir, but was not detected in reservoir water or reservoir outflow (Amandi et al. 1982). Greatest concentrations of the pathogen were detected in the outflow from Cole M. Rivers Hatchery. Spring chinook salmon in the hatchery also were found to be infected with the disease. We do not know whether adult salmon carried the disease into the hatchery or contracted the disease after entry. *Flexibacter columnaris* was also found in several species of fish sampled throughout the Rogue River basin, including the Applegate River (Amandi et al. 1982).

We were unable to determine whether operation of Lost Creek Dam, the reservoir, and the mitigation hatchery affected rates of prespawning mortality for fall chinook salmon in the Rogue River. In most cases, fish mortality associated with operation of reservoirs occurs in close proximity to the dams (Grizzle 1981). We found that mortalities of fall chinook salmon peaked almost 200 km downstream of Lost Creek Dam. However, *Flexibacter columnaris* can survive for at least five days in flowing water (Amandi et al. 1982). Also, fish throughout the river can become infected and release more pathogens that spread farther downstream.

Irrespective of the factor(s) responsible for prespawning mortality, our findings indicate Lost Creek Lake can be managed to minimize mortality of fall chinook salmon in the Rogue River. The reservoir should be managed so that average water temperature at Agness does not exceed 19.8°C in the last 3 weeks of August and the first 10 days of September. To meet this target for water temperature, outflow from Lost Creek Dam should be 2,200 cfs or greater during 11 August through 10 September.

River flow must be augmented to decrease water temperature in the Rogue River canyon because effects of outflow temperature diminish rapidly with distance downstream (USACE 1991). Improvements in water quality will decrease

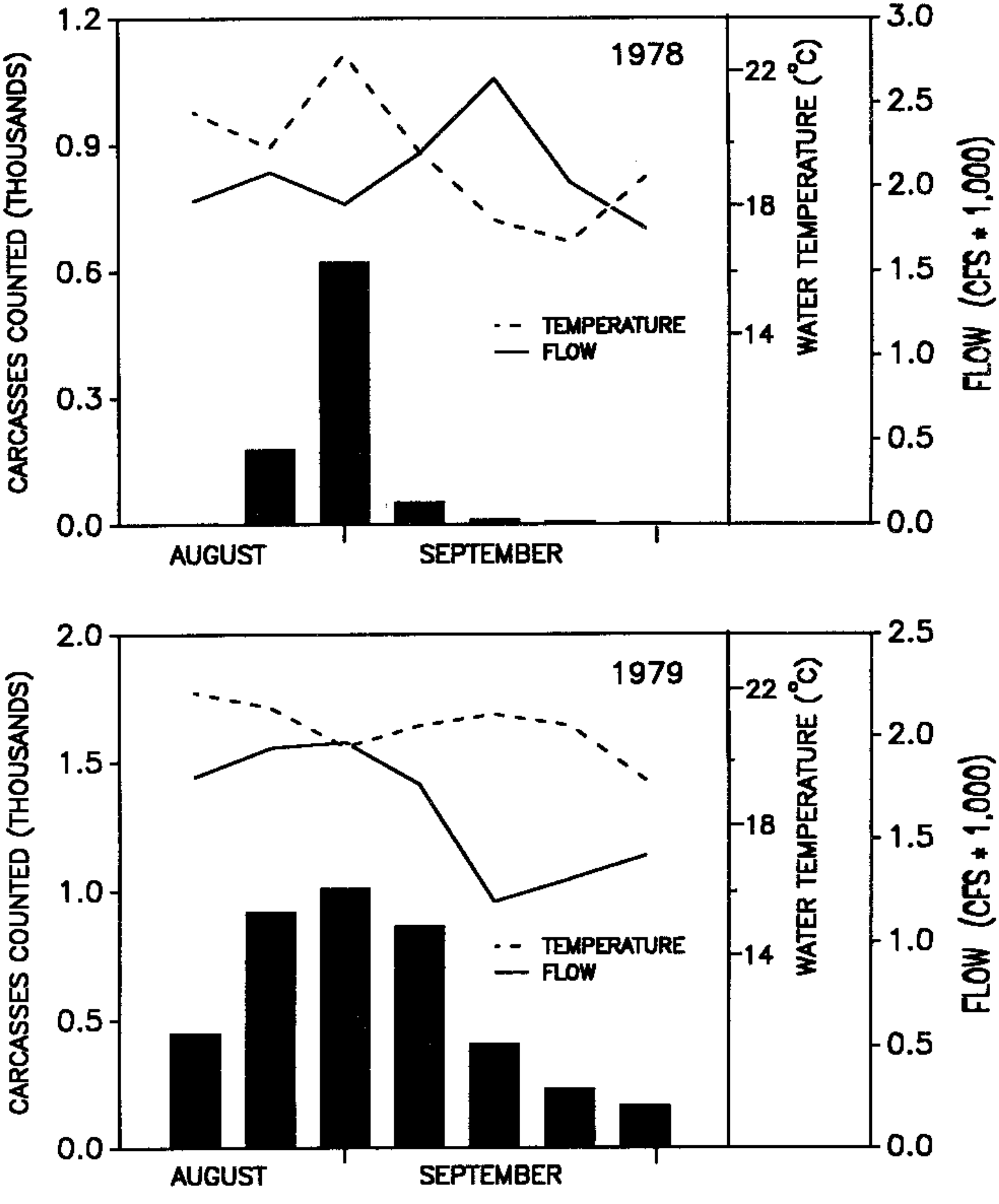

Figure 30. Timing of prespawning mortality during 1978 and 1979 in relation to weekly mean flow and weekly mean maximum water temperature at Agness.

stress on migrating fall chinook salmon. Decreased stress on returning adults should also reduce potential mortality from other infectious diseases (Wedemeyer 1970; Snieszko 1974).

Freshwater Harvest

Volitional returns of salmon-steelhead cards indicated anglers harvested an average of 2,913 fall chinook salmon annually in 1956-86. Harvest estimates ranged between 899 and 6,254 fish (Appendix Table C-24). Estimates of monthly harvest averaged 858 fish in August, 1,210 fish in September, 692 fish in October, and 176 fish in November. These estimates do not include adults smaller than 51 cm in 1956-77 or smaller than 60 cm in 1978-86.

We estimated anglers harvested an average of 9.5% of the fall chinook salmon that returned to the Rogue River in 1974-86. Recreational fisheries in other coastal streams of Oregon harvest fall chinook salmon at greater rates (Nicholas and Hankin 1988). Estimates of harvest rate in the Rogue River varied between 2% and 22% annually. We did not estimate harvest rates for earlier years because there were no estimates of freshwater returns. Nor did we evaluate data from 1977 because many fish harvested in August were spring chinook salmon (Cramer and Martin 1979).

We found that harvest rates varied among years. Harvest rates averaged 10.2% in 1974-76, 4.9% in 1978-82, and 14.7% in 1983-86 (Appendix Table C-25). We hypothesized harvest rates were low in 1978-82 because fish were stressed by disease. Harvest rate increased when prespawning mortality decreased to near zero in 1983-86. We hypothesized river physical factors and run size affected harvest rate of fall chinook salmon in the Rogue River.

We found that harvest rate was positively related with flow during the fishery (Figure 31). Harvest rates were greater in years of high flow compared with years of low flow. Angler harvest of spring chinook salmon in the lower river was also correlated positively with flow (Cramer et al. 1985). However, flow was also significantly related with water temperature (Appendix Table C-26), making it impossible to estimate the singular effect of each factor on harvest rate. Data included in analyses of freshwater harvest rate are in Appendix Table C-25.

Decreased water temperature in summer probably decreased stress among fall chinook salmon. Fish stressed by high water temperature or disease may have been less likely to strike angling gear. We previously reported a negative correlation of water temperature and weekly catch rates of age 2 jacks in the lower river fishery (Cramer et al. 1985). Also, we observed that anglers often released stressed fish, which were darker than healthy fish. Decreased water temperature probably improved the health of fall chinook salmon and increased desirability to anglers.

We also found that harvest rate in the river fishery correlated negatively with the abundance of fall chinook salmon (Appendix Table C-26). Harvest rate increased as freshwater returns decreased (Figure 31). This finding was expected because freshwater return is a also a component of harvest rate. Harvest rate averaged 14% (17% excluding years with prespawning mortality) when freshwater returns ranged between 10,000 and 20,000 fish.

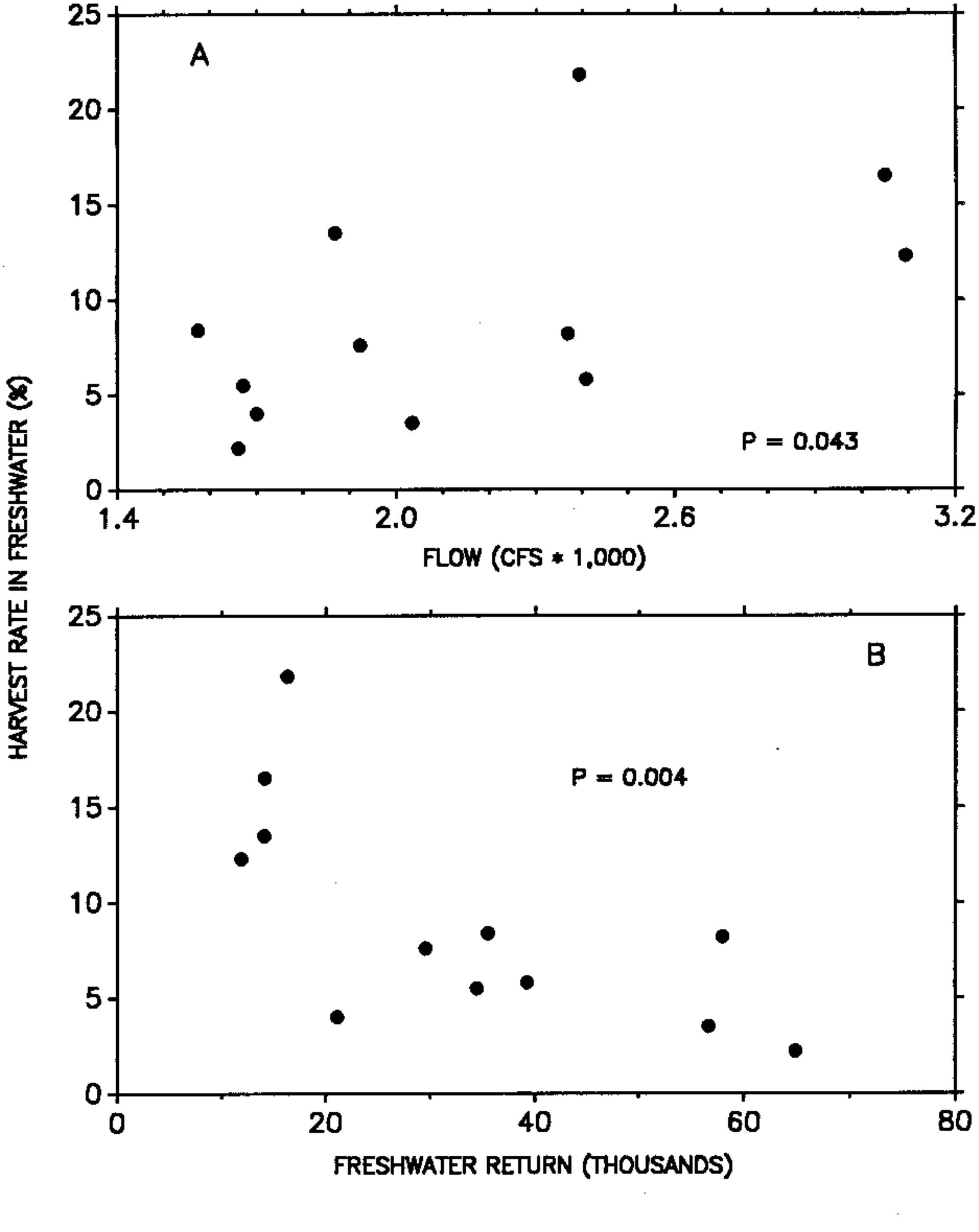

Figure 31. Relationships between rate of freshwater harvest of fall chinook salmon and (A) mean flow at Agness during August-September and (B) estimated return to the Rogue River, 1974-86. Data from 1977 excluded because migration delay resulted in harvest of spring chinook salmon during August. Significance levels were calculated from percentages transformed to logits.

When freshwater returns exceeded 20,000 fish, harvest rates averaged about 6% (8% excluding years with prespawning mortality). Negative relationships between run size and the harvest rate of chinook salmon in fresh water also have been documented by Peterman and Steer (1981).

Increased flow in late summer was related to the temporal change in distribution of angler harvest. We found a positive relationship between the proportion of fish harvested in September and flow in the middle river (Appendix Table C-27). Anglers harvested proportionally more fall chinook salmon in September during years of greater flow (Figure 32). We did not analyze harvest data prior to 1965 because construction of jetties at the mouth of the river modified an estuary fishery that primarily harvested the early component of the run (discussion on 15 May 1987 with Arvo Riikula, ODFW, Gold Beach, Oregon). Data included in the analysis are in Appendix Table C-28.

Augmented flow in August and early September may have stimulated adults to migrate faster through the canyon and enter the fishery in the middle river earlier. This fishery rapidly grew in the 1980s (discussion on 14 May 1987 with Michael Jennings, ODFW, Central Point, Oregon). Angler effort probably increased in response to increased angler success (Steinnes and Raab 1983; Cramer et al. 1985). Also, some anglers mentioned they considered fall chinook salmon caught in the middle river during recent years to be more desirable compared with fall chinook salmon caught in earlier years. We believe larger runs in future years will be harvested at greater rates if angler interest in this fishery continues to grow.

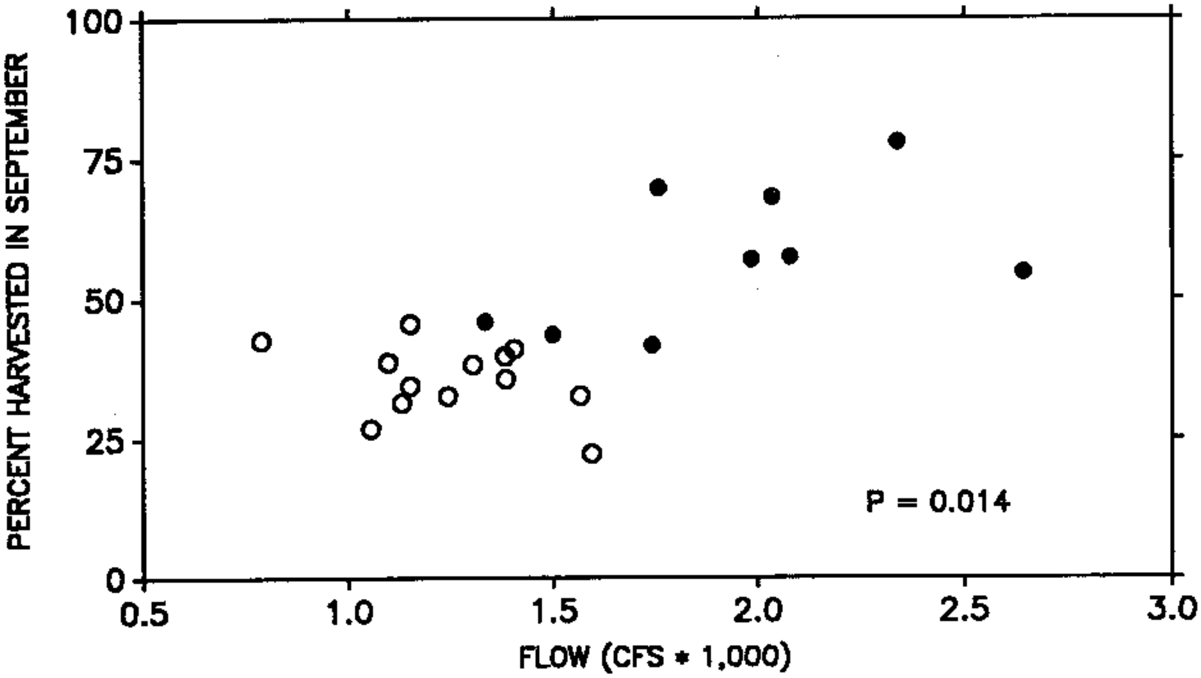

Figure 32. Relationship of timing of angler harvest of fall chinook salmon and mean flow at Grants Pass during September, 1965-86. Open and closed circles represent returns before and after operation of Lost Creek Dam, respectively. Significance level was calculated from percentages transformed to logits.

The increase in importance of the September harvest coincided with a decrease in the proportion of fall chinook salmon that were harvested in October and November. Decreased harvest in October-November was probably related to decreased abundance of populations that entered fresh water relatively late (*see* **Migration Timing**, page 71). We documented a decline in spawner abundance in these streams in the early 1980s (Cramer et al. 1985) and returns remained low in later years (discussion on 15 May 1987 with Arvo Riikula, ODFW, Gold Beach, Oregon).

Our findings indicated Lost Creek Lake can be managed to enhance the freshwater fishery for fall chinook salmon in the Rogue River. Strategies recommended to minimize the potential for prespawning mortality (*see* **Recommendations**, page 4) will increase contribution rates to the fishery in the middle river and may improve the quality of fall chinook salmon landed by anglers that fish areas farther downstream.

Spawning Distribution

We found fall chinook salmon spawned in widely distributed areas of the Rogue River basin. Although we did not survey all areas of the basin, spawning appeared concentrated in the middle river and in the lower portion of the Applegate River (Cramer et al. 1985). During some years, we observed large numbers of spawners in the middle portion of the Applegate River, Slate Creek, Lobster Creek, Quosatana Creek, Deer Creek, and the lower portions of the East Fork and the West Fork of the Illinois River. Large numbers of fall chinook salmon also spawn upstream of Gold Ray Dam (ODFW 1991b).

Operation of Lost Creek Dam may have affected the spawning distribution of fall chinook salmon. Among preimpoundment broods, we estimated an average of 9% of the fish spawned upstream of Gold Ray Dam. Among postimpoundment broods, we estimated an average of 15% of the fish spawned upstream of Gold Ray Dam. However, the difference in means was not significant ($P = 0.11$).

We concluded the power of this test was low. A sensitivity analysis indicated the mean percentage of the run that spawned upstream of Gold Ray Dam would have had to decrease to 2% or increase to 24% for a change to be detected by parametric statistics. Data included in these analyses are in Appendix Table C-29.

Further analysis indicated that flow augmentation stimulated an upstream shift in spawning distribution. We found that greater proportions of adults spawned upstream of Gold Ray Dam in summers of greater flow (Figure 33). Regression analysis indicated flow in September accounted for 55% of the variation in spawning distribution (Appendix Table C-30).

From the regression, we estimated an increase in flow from 1,500 cfs to 2,000 cfs would increase the percentage of fish that spawn upstream of Gold Ray Dam from 7% to 13%. Because spawning escapement averaged 34,000 fish, we estimated flow augmentation of 500 cfs would increase the number of fall chinook salmon that spawn upstream of Gold Ray Dam by about 2,000 fish. We did not attempt to evaluate the influence of water temperature on spawning distribution because water temperature correlated significantly with flow. Data included in these analyses are in Appendix Table C-29.

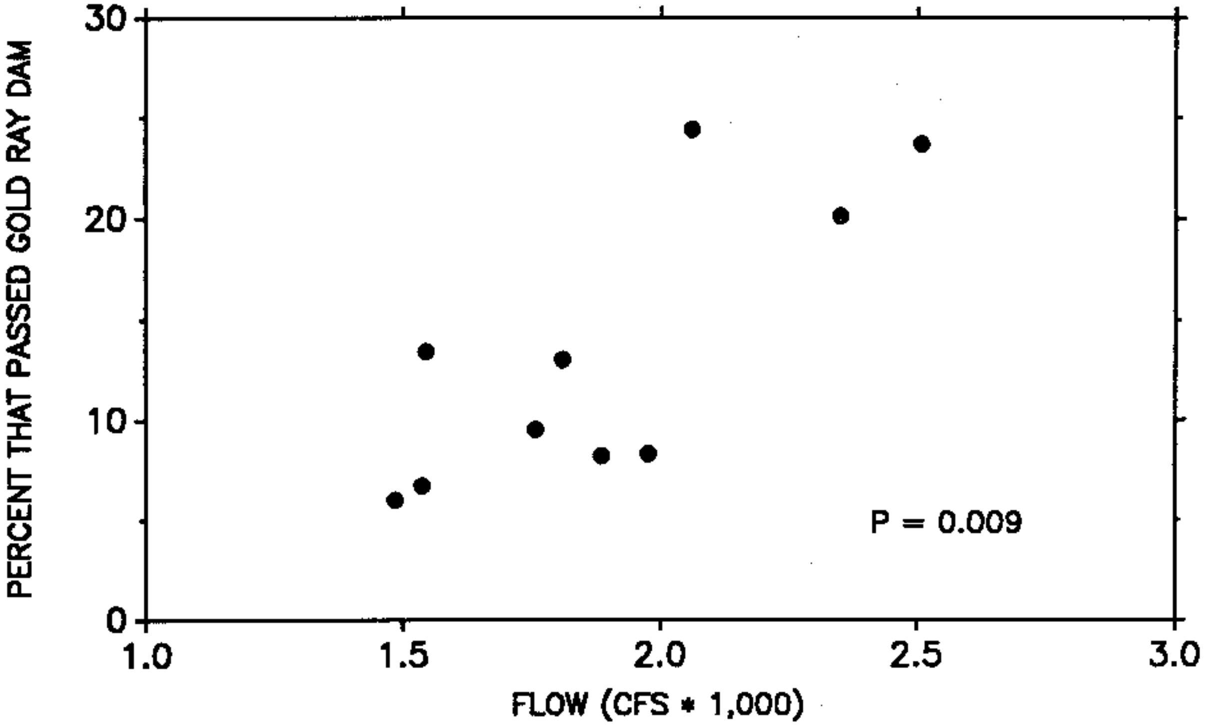

Figure 33. Relationship of the percentage of fall chinook salmon that migrated upstream of Gold Ray Dam and mean flow at Raygold during September, 1974-86. Spawning distribution could not be estimated for 1977 or 1979-80. Significance level was calculated from percentages transformed to logits.

In contrast, we could not detect any effect of reservoir operation on the spawning distribution of fall chinook salmon in the middle river. The proportion of spawned carcasses found in the middle river, compared with the lower portion of the Applegate River, did not change significantly after operation of Lost Creek Dam (*P* = 0.66). These were the only spawning areas surveyed consistently during the study (Appendix Table C-31). Carcasses in the middle river averaged 35% and 39% of the totals for preimpoundment and postimpoundment returns, respectively. A sensitivity analysis indicated the relative abundance of carcasses in the middle river would have had to decrease to 15% or increase to 60% for a change to be detected.

Fall chinook salmon spawned primarily in larger streams of the Rogue River basin. The tendency of chinook salmon to spawn in large streams compared with other anadromous salmonids was discussed by Burner (1951) and Briggs (1953). A large body size may be an adaptation for spawning in habitat susceptible to large freshets (Hankin and McKelvey 1986). Large females may be able to excavate deeper redds as a mechanism to shelter eggs and alevins from disruption of overhead gravel during small freshets. A positive relationship between body size and redd depth has been shown for female coho salmon (van den Berghe and Gross 1984). Larger streams can support more spawners, but there is greater potential for redd disruption by freshets.

Progeny tend to return and spawn in the vicinity of natal areas. Precise homing is characteristic of all Pacific salmon and may be a mechanism to insure progeny inhabit a suitable rearing environment (Ricker 1972; Brannon 1982). Olfactory stimuli provide the means by which maturing chinook salmon home to spawn in their natal areas (Groves et al. 1968). Chinook salmon of hatchery origin return to natal locations at rates greater than 90% (Quinn and Fresh 1984). We believe annual variations in spawning distribution of fall chinook salmon in the Rogue River basin probably resulted from variations in the relative abundance of adults that originated from different populations.

Environmental factors also affected spawning distribution of fall chinook salmon in the Rogue River basin. Drought in the winter of 1976-77 prevented entry of spawners into Slate Creek (Appendix Table C-31). Chinook salmon spawn farther upstream in years of high flow compared with years of low flow (Moyle 1976; McGie 1977; Fustish et al. 1988).

We found that fall chinook salmon in the Rogue River spawned farther upstream when adults migrated at greater flow. Augmented flow from Lost Creek Dam induced proportionately more adults to spawn upstream of Gold Ray Dam. We believe increased numbers of fall chinook salmon that spawn upstream of the counting station is not a desirable strategy for management of chinook salmon in the Rogue River basin. The area upstream of Gold Ray Dam is the only spawning habitat utilized by spring chinook salmon in the Rogue River basin (Cramer et al. 1985). Spring chinook salmon support a more intensive recreational fishery in the Rogue River compared with fall chinook salmon (ODFW 1991b). Protracted interracial spawning would likely decrease the freshwater harvest of spring chinook salmon (ODFW 1991b).

Our findings suggested decreased outflow from Lost Creek Dam in late summer will reduce the number of fall chinook salmon that enter spawning areas historically used by spring chinook salmon. To minimize passage of fall chinook salmon at Gold Ray Dam, the reservoir should be managed to minimize flow augmentation from the middle of September through the middle of October. This recommendation may be modified upon completion of ongoing research on factors that affect production of wild spring chinook salmon in the upper river (Satterthwaite 1991).

Spawning Time

Fall chinook salmon in the Rogue River basin exhibited a wide range in spawning time. We found spawning began in late September and continued through late January. The majority of fish spawned between early October and early December (Figure 34). This spawning time is similar to, or slightly earlier than, the spawning time of other runs of fall chinook salmon in other coastal streams of Oregon (Nicholas and Hankin 1988).

Spawning time varied significantly between different areas of the basin. Adults that migrated into tributary streams spawned later compared with adults that spawned in the Rogue River (Figure 34). Adults spawned latest in small tributary streams like Slate Creek. We observed little variation in spawning time within populations. Tagging studies showed spawning time was highly correlated with time of freshwater entry (Table 18). Early spawners entered the Rogue River significantly earlier than late spawners.

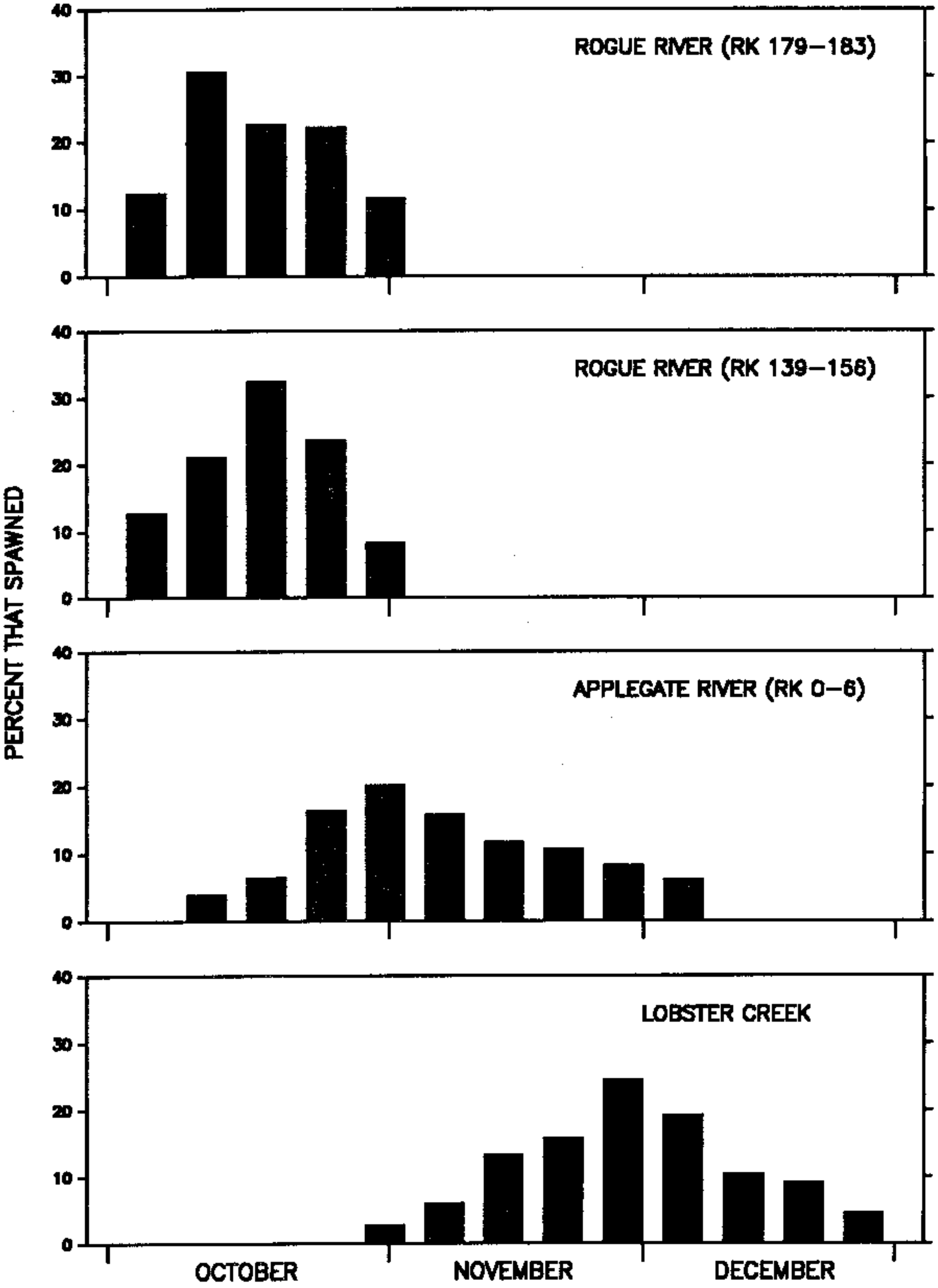

Figure 34. Estimated time of spawning for fall chinook salmon in the Rogue River basin, 1974-85.

Table 18. Relationship between date of tag recovery (Y) and date that fall chinook salmon were tagged (X) in the lower river, 1975-78. Relationship for 1974 not estimated because *N* = 5. Dates were represented as day-of-year (*see* **APPENDIX A**).

Year	*N*	Regression	*r*	*P*
1975	44	Y = -31.7 + 0.853(X)	0.66	<0.001
1976	38	Y = -77.0 + 1.018(X)	0.71	<0.001
1977	15	Y = -22.8 + 0.852(X)	0.83	<0.001
1978	107	Y = 59.9 + 0.604(X)	0.80	<0.001

Our data indicated operation of Lost Creek Dam did not affect spawning time of fall chinook salmon in the Rogue River. We found no significant differences in time of peak spawning before and after operation of Lost Creek Dam. Also, there were no significant differences in spawning time of preimpoundment and postimpoundment broods (all $P > 0.30$). During 1974-85, recoveries of spawned carcasses in the middle river peaked consistently in late October and early November (Appendix Table C-32). Spawning in the middle river peaked in the middle or latter portion of October.

Time of spawning in Pacific salmon reflects genetic adaptation to localized environments (Ricker 1972). Changes in photoperiod triggers maturation of chinook salmon (Zaugg et al. 1986). Physiological mechanisms that respond to changes in photoperiod are heritable. Taylor (1980) found progeny of early and late spawning pink salmon spawned at the same time as the parents.

Spawning times vary between populations because environmental parameters vary between habitats. Among Atlantic salmon stocks in Norway, Heggberget (1988) noted a strong negative correlation between spawning time and incubation temperature for eggs and alevins. Burger et al. (1985) found chinook salmon in the Kenai River basin of Alaska spawned later in areas downstream of lakes. Water temperature was judged to affect spawning time through natural selection. These researchers hypothesized spawning time differed between populations so progeny emerge from the gravel at times optimal for survival.

We were unable to detect any effect of reservoir operation on spawning time of fall chinook salmon. However, USACE simulations of water temperature in the middle river indicated reservoir operation should have accelerated the emergence time of fry by about two weeks (*see* **Emergence Timing**, page 35). This level of change in emergence timing may or may not affect a change in spawning time that is large enough to be detected by future sampling. If USACE simulations of water temperature are accurate, then future generations of fall chinook salmon may spawn at a later time in the middle portion of the Rogue River.

Sex Composition

A chi-square test indicated that sex composition of fall chinook salmon that spawned in the Rogue River basin varied significantly ($P < 0.01$) among years. We found that females accounted for an average of 55% of age 3-6 carcasses recovered in 1974-85 and ranged between 45% and 65% annually.

We found the percentage of females among spawners was positively related with mean age among age 3-6 fish that returned to fresh water (Figure 35). Returns with an older mean age had proportionately more females among spawners. Regression analysis indicated mean age of the run accounted for 49% of the variation in sex composition of spawners (Appendix Table C-33). Hankin and Healey (1986) found males matured at younger ages than females among fall chinook salmon that spawned in the Applegate River. Data included in our analyses are in Appendix Table C-34.

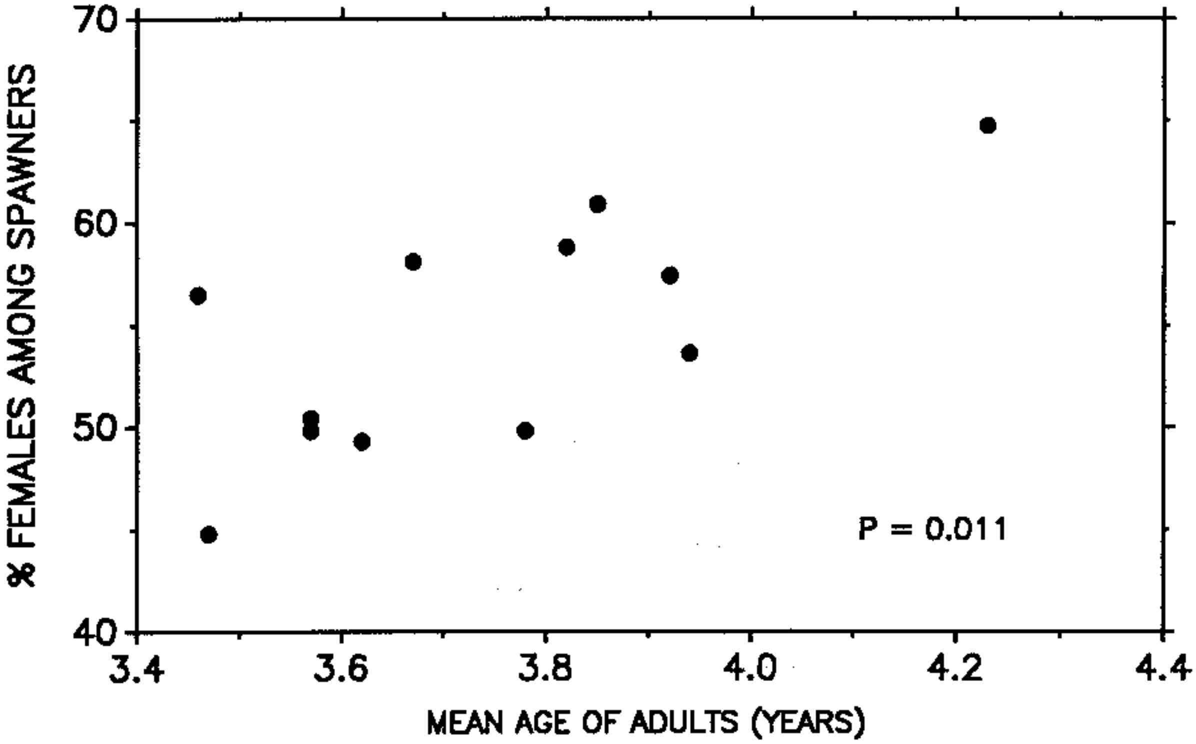

Figure 35. Relationship between the percentage of females among spawned carcasses of fall chinook salmon and mean age of age 3-6 adults at time of freshwater entry, 1974-85. Significance level was calculated from percentages transformed to logits.

Simulation Model

We developed a simulation model for fall chinook salmon in the Rogue River basin. We developed the model for two reasons. First, we believe simulation analysis will facilitate synthesis of the multitude of project findings, resulting in more effective recommendations for reservoir operation.

Second, we believe simulation analysis will identify areas where additional work would improve understanding of population dynamics of fall chinook salmon in the Rogue River basin. We recognize this type of model fails to account for interspecific affects of other aquatic organisms in ecosystems inhabited by fall chinook salmon.

Model Structure

Model structure reflects major life history events in the Rogue River basin and the ocean. The model simulates population and harvest parameters over a period of five years for one brood and that brood's parental components (Figure 36). The final product is a prediction of mean ocean recruitment of age 3 fall chinook salmon. We extended the model to this point because large numbers of fall chinook salmon from the Rogue River basin become available for ocean harvest at age 3 and only a small percentage mature at age 2.

The model begins with predictions of the number of immature age 2-6 fish resident in the ocean during spring of year i. Predictions are derived from inputs for juvenile abundance and freshwater returns of adults in year i-1.

Next, freshwater returns of adults in year i are predicted from the number of cohorts that escape ocean harvest or shaker mortality in year i. Ocean harvest is estimated from an input value for the annual exploitation rate of age 4 fish. Freshwater harvest is predicted from an input value for flow during the fishery. Prespawning mortality is predicted from an input value for water temperature of the river during adult migration. The number of spawning females is predicted from the number of fish that escape freshwater harvest and prespawning mortality, and the age composition of adults at time of freshwater entry.

Juvenile abundance in year i+1 is predicted from (1) the predicted number of females that spawn in year i, (2) an input value for tributary flow when parents migrated into spawning streams, and (3) an input value for peak flow when eggs and alevins incubated in the gravel. The number of age 2 fish resident in the ocean during spring of year i+2 is predicted from estimates of cohort abundance in year i+1. The number of age 3 fish resident in the ocean during spring of year i+3 is predicted from the estimated abundance of age 2 cohorts and loss to shaker mortality in year i+2.

Regression equations compose the principal components of the model. All regressions incorporated in the model were previously presented in this report (Table 19), with one exception. Regressions to estimate ocean abundance of immature age 3-6 fish in the ocean were developed with data transformed to natural logarithms. We assumed these regressions were independent of each other. In reality, regressions used to predict ocean abundance were probably interrelated because variations in maturity rates affected the ocean abundance of recruits that matured at older ages.

Sensitivity Analysis

We found that output response was most sensitive to increased water temperature of the Rogue River in late summer. Sensitivity to an increase in

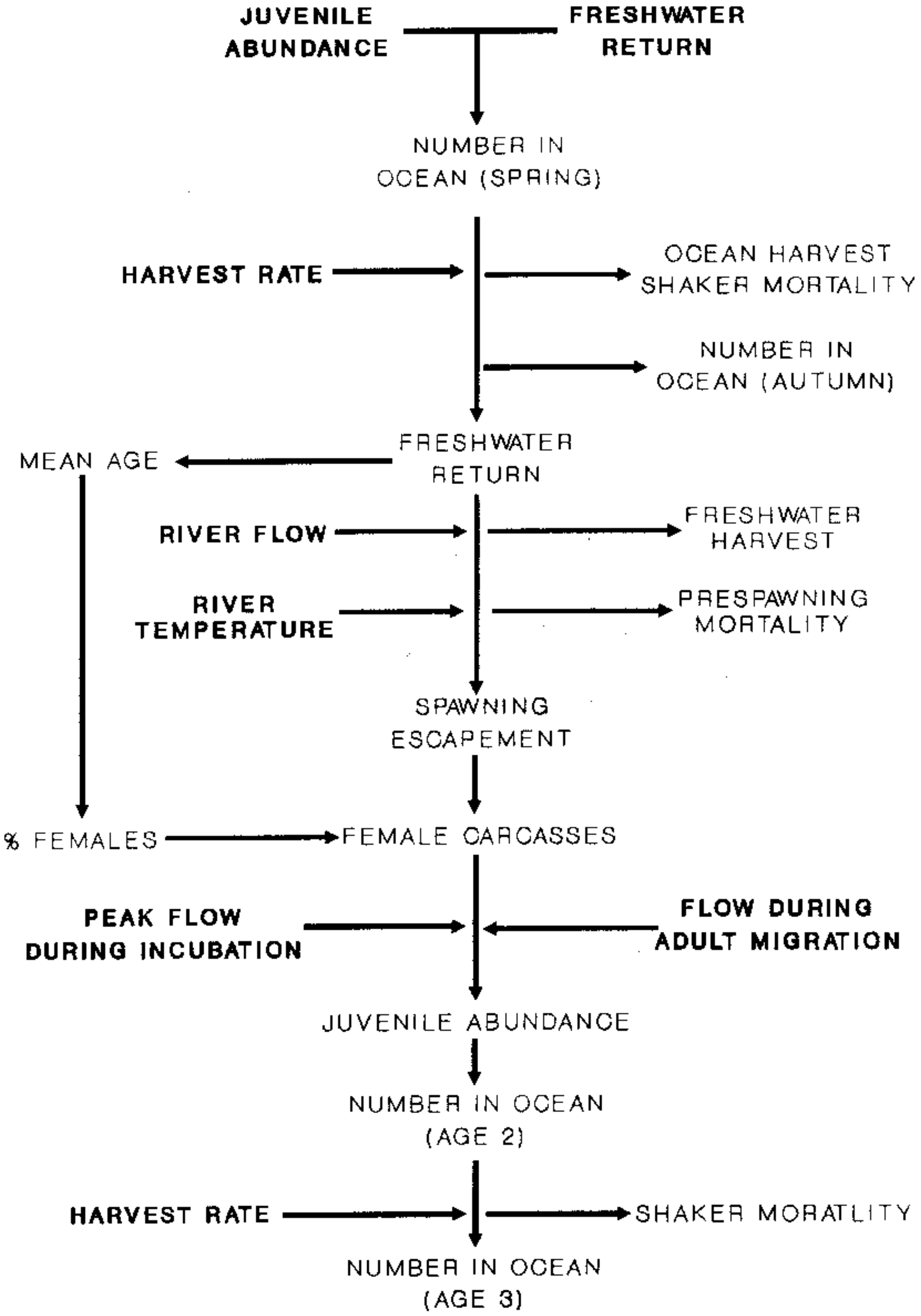

Figure 36. Schematic of the simulation model developed for fall chinook salmon in the Rogue River basin. Inputs for simulation analysis are in boldface.

Table 19. Components of the simulation model for wild fall chinook salmon of Rogue River origin.

Dependent variable	Description of estimator
Parental abundance in the ocean	*See* **Ocean Abundance**, page 56
Freshwater returns of parents	*See* **Ocean Abundance**, page 56
Prespawning mortality of parents	*See* **Prespawning Mortality**, page 74
Freshwater harvest of parents	*See* **Freshwater Harvest**, page 78
Abundance of female spawners	*See* (1) **Spawner Abundance**, page 64 (2) **Sex Composition**, page 86
Juvenile catch per seine haul	*See* **Juvenile Abundance**, page 39
Progeny abundance (age 2) in ocean	*See* **Ocean Abundance**, page 56
Progeny abundance (age 3) in ocean	*See* **Ocean Harvest**, page 68

water temperature was four times greater than sensitivity to any other change among input parameters (Table 20). Except for increased water temperature, sensitivities exceeded 0.1 only for decreased exploitation rate in ocean fisheries and decreased peak flow when eggs and alevins incubated in the gravel (Table 20). Output responses were least sensitive to changes in juvenile abundance, freshwater returns of age 4 and age 5 parents, and flow during the river fishery (Table 20).

Table 20. Sensitivity of simulated recruitment of age 3 fall chinook salmon to variations of input parameters. Inputs of average values predicted recruitment of 685,222 fish.

Parameter	Input value	Predicted recruitment		Sensitivity	
		Input-10%	Input+10%	Input-10%	Input+10%
Juvenile abundance[a]	119	689,384	683,163	<0.01	<0.01
Age 2 return	12,102	706,257	718,191	0.03	0.05
Age 3 return	12,784	662,015	737,080	0.03	0.08
Age 4 return	16,202	686,708	704,556	<0.01	0.03
Age 5 return	2,639	705,275	705,025	0.03	0.03
Ocean harvest rate	0.59	781,351	641,886	0.14	0.06
Water temperature	19.6°C	730,316	255,256	0.07	0.63
Fishery flow	1,797 cfs	691,580	690,183	<0.01	<0.01
Migration flow	347 cfs	673,962	727,171	0.02	0.06
Peak flow	24,312 cfs	759,499	675,687	0.11	0.01

[a] *Indexed as catch per seine haul.*

Model Evaluation

We could not compare values of model predictions with data observed during the project. We lacked appropriate data to make this type of comparison because (1) the relationship used to estimate prespawning mortality is only applicable for postimpoundment years (1978-86) and (2) relationships to estimate fish abundance in the ocean are applicable only to the 1969-77 brood years not affected by the El Niño event of 1982-83.

Instead, we evaluated the resolution, realism, and precision of the model. Resolution relates to the relative abundance of key ecosystem(s) components in a model. The greater the number of components, the greater the resolution of a model. Realism relates to the propriety of the equations used in a model. The better equations describe relationships between biological parameters, the greater the realism of the model. Precision relates to the degree of uncertainty associated with model output. Models with lower precision have greater uncertainty associated with output predictions.

We believe the model exhibits a relatively high degree of resolution with two major exceptions. The model assumes constant maturity rates and constant rates of natural mortality in the ocean. We found these parameters varied significantly between years, but were unable to identify factors responsible for the variation. Identification of factors responsible for variations in maturity rates and rates of ocean survival would require further research that exceeds the scope of this project. Variations in physical parameters and forage resources probably affect maturity rates of fish resident in the ocean (*see* **Age at Maturity**, page 66).

Also, variations in relative abundance of populations with different maturity schedules also affect estimates of maturity rates for a composite sample. We doubt collection of additional data will result in effective estimation of these parameters. Estimation of maturity schedules of individual populations would have to be approached experimentally, possibly with releases of hatchery fish that originate from different populations in the basin. Such experiments would require a great amount of effort.

Identification of factors that affect rates of natural mortality of fall chinook salmon in the ocean would also be difficult. Intensive work continues with coho salmon in the Oregon Production Index Area to attempt to answer that question (Peterman 1989). Estimation of mortality rates is even more complex for chinook salmon that mature at five ages rather than two ages as for coho salmon. We concluded attempts to improve resolution of the model would be a lengthy and expensive task. We recommend no additional work to improve model resolution unless findings from other research can be incorporated into the model with minimal effort.

Attempts to increase realism of the model would be less expensive. Seining at Huntley Park and spawning surveys continued during 1987-91. These data could be analyzed and incorporated into the model. Potential benefits would mostly accrue from revisions in relationships between freshwater returns and cohort abundance in the ocean during spring of the succeeding year. These relationships currently have only seven data points and are mostly dominated by the abundant 1975 brood year (*see* **Ocean Abundance** page 56).

We believe the model could be made more real by additional work on factors that affect juvenile production. Our findings indicated juvenile production increased ad infinitum with (1) increased spawning escapement and (2) increased flow in tributary streams during adult migration and spawning. These findings may be appropriate for the range of data we gathered, but are probably not appropriate for greater values of flow and spawning escapement. At some level, compensatory factors must act to limit juvenile production (Ricker 1975). In years of large spawning escapements, we recommend sampling to index juvenile abundance in the succeeding year.

We also recommend additional indexing of juvenile abundance to increase precision of model predictions. Simulation with average values for input parameters resulted in a 95% confidence interval that ranged between 85 and 333 juveniles per seine haul (Table 21). The relatively low level of precision resulted in part from a multiple regression with a small sample of data (*N* = 11).

The low level of precision that resulted from the juvenile component was carried along to the final step of the model. The coefficient of variation for predicted mean recruitment at age 3 was 41% for the simulation with mean values for input parameters. Confidence boundaries of the mean prediction for ocean recruitment at age 3 ranged between 0.3 million and 1.4 million fish.

Table 21. Variability estimates of model outputs for a simulation with average values for input parameters (*see* Table 20).

Output parameter	Mean prediction	CV[a]	95% confidence boundary Lower	Upper
Number of age 2 fish in ocean	965,753	17	662,177	1,345,866
Number of age 3 fish in ocean	277,889	17	189,020	376,510
Number of age 4 fish in ocean	98,212	27	52,931	159,144
Number of age 5 fish in ocean	6,607	22	4,126	9,668
Number of age 6 fish in ocean	254	119	0	995
River return of age 2 fish	22,434	17	15,382	31,264
River return of age 3 fish	15,610	17	10,618	21,149
River return of age 4 fish	23,395	27	12,609	37,910
River return of age 5 fish	2,565	22	1,637	3,754
River return of age 6 fish	104	119	0	408
Total return to river	64,109	12	49,159	80,526
Prespawning mortality[b]	1,668	42	692	3,393
Freshwater harvest[b]	2,917	30	1,189	4,660
Number of spawners[b]	37,089	17	25,639	50,008
Index of juvenile abundance	195	32	85	333
Number of age 3 fish in ocean	685,222	41	315,552	1,414,509

[a] *Coefficient of variation.*
[b] *Age 3-6 adults only.*

Variability among output responses were highest for predictions of age 3 recruitment and prespawning mortality (Table 21). Variability among output responses were least for predictions of spawning escapement and abundance of age 2 and age 3 fish in the ocean and at freshwater return. Coefficients of variation were the same for ocean abundance and freshwater returns of cohorts because we assumed no variation in maturity rates of fall chinook salmon resident in the ocean.

Precision of model predictions will further decrease as input parameters vary from historic means. We opted not to evaluate model precision with any data other than historic means for input values because simulations showed a lack of precision in the model. Precision of the model could be improved with additional sampling to index juvenile abundance and to estimate prespawning mortality. This work should be conducted only if ongoing sampling is designed to estimate freshwater returns and spawning escapement.

Application for Reservoir Management

Our findings indicated that operation of USACE reservoirs in the Rogue River basin affected production and harvest of fall chinook salmon. Principal effects resulted from (1) flow augmentation in summer when adults migrated in the Rogue River, (2) reductions in peak flow during reservoir filling and flood control operations when eggs and alevins incubated in the gravel, and (3) flow augmentation in autumn when adults migrated and spawned in the Applegate River (*see* **Prespawning Mortality**, page 74 and **Abundance**, page 39).

Reservoir operation in early summer also affected life history parameters of juvenile fall chinook salmon that reared in the Rogue River (*see* **Summary**, page 1). However, we were unable to identify subsequent effects on the production or life history of immature fish in the ocean and adults that returned to the Rogue River. Consequently, we did not simulate effects of variations in river physical factors in early summer.

Flow augmentation in late summer decreased water temperature in downstream areas (USACE 1991). Our simulation analyses indicated water temperature of the Rogue River in late summer affected subsequent recruitment of fall chinook salmon through changes in rates of prespawning mortality. Simulations indicated recruitment of subsequent broods decreased when water temperature at Agness exceeded an average of 19.8°C during August-September (Table 22). Simulations also suggested decreases in mean water temperature below 19.8°C had minimal effect on recruitment (Table 22).

Simulations indicated an increase in mean water temperature from 19.8°C to 20.7°C decreased subsequent recruitment by 32% in a year of low parental return, 45% in a year of average parental return, and 52% in a year of large parental return (Table 22). This finding suggested reductions in recruitment from loss to prespawning mortality would be greatest in years when large numbers of adults returned to fresh water. Freshwater density may affect transmission of disease among adult fall chinook salmon, but we were unable to identify the relationship between run size and mortality rate (*see* **Prespawning Mortality**, page 74).

Table 22. Mean recruitment for age 3 fall chinook salmon predicted from variations in parental returns to fresh water and physical parameters of the Rogue River during late summer. Other input parameters were set to average values listed in Table 20.

Fishery flow[a]	Water temperature[b]	Predicted recruitment (thousands) ± 1 SD		
		Low return[c]	Average return[d]	Large return[e]
1,500	20.7	335 ± 169	420 ± 210	517 ± 294
1,700	20.4	402 ± 180	540 ± 252	712 ± 428
1,900	20.1	457 ± 178	693 ± 326	888 ± 570
2,100	19.8	495 ± 184	764 ± 400	1,073 ± 691
2,300	19.6	504 ± 198	782 ± 385	1,074 ± 760
2,500	19.3	501 ± 195	820 ± 465	1,055 ± 594
2,700	19.1	499 ± 193	792 ± 386	1,091 ± 773
2,900	18.9	510 ± 198	799 ± 400	1,126 ± 991

[a] *Mean flow (cfs) at Agness during August-September.*

[b] *Mean maximum water temperature (°C) at Agness during August-September. Predicted from the fishery flow.*

[c] *Freshwater escapement during 1983 used as an example.*

[d] *Freshwater escapement during 1981 used as an example.*

[e] *Freshwater escapement during 1978 used as an example.*

From these findings, we concluded Lost Creek Lake should be managed so mean water temperature at Agness in August-September averages less than 19.8°C. To meet that goal for average conditions, we estimated that flow at Agness should average 2,035 cfs in August-September. We derived the flow estimate from a regression of water temperature and flow for the time period of August-September during 1978-86 ($r = 0.90$, $P = 0.001$).

However, maintenance of 2,035 cfs at Agness during the entire period is not the best strategy for reservoir management. Less augmentation is needed in early August because few fall chinook salmon enter the Rogue River prior to 10 August (see **Migration Timing**, page 71). Also, less augmentation is needed after the middle of September because water temperature at Marial naturally declines to an average of 15.5°C by the end of the month.

Flow augmentation in late September increases the potential that redds of spring chinook salmon will be dewatered during subsequent filling of the reservoir (Satterthwaite et al. 1985). Flow augmentation in September-October also reduces rates of angler success for summer steelhead in the Rogue River canyon (ODFW, unpublished data). Management strategies in recent years have minimized these effects by limiting reservoir outflow to a maximum of 1,200 cfs in October.

To maximize production of fall chinook salmon in the Rogue River basin, we recommend a minimum flow of 2,300 cfs at Agness during 10 August through 10 September. Attainment of this objective would require a release of about 2,200 cfs from Lost Creek Dam in years with low flows from tributary streams.

Less augmentation of flow would be required in years of greater water yield from tributaries. As previously discussed, resource managers should be particularly cautious about allocation of reservoir storage to minimize potential for prespawning mortality in years when large numbers of adults (i.e. >30,000) are expected to return to the Rogue River.

During years of low water yield in the Rogue River basin, we estimated as much as 100,000 acre-feet of reservoir storage would be needed to meet the recommended flow of 2,300 cfs at Agness during 10 August through 10 September. Use of this amount of storage may conflict with other needs for reservoir storage, including other priorities for fishery management. We will need to integrate this recommendation with findings from ongoing work with summer steelhead and spring chinook salmon.

We recognize there will be years when storage is insufficient to meet demand for flow in areas downstream of Lost Creek Dam. Development of annual plans for reservoir management helps identify optimal use of the finite amount of reservoir storage. Development of annual goals for management of fishery resources in the Rogue River basin also aids identification of optimal use for available storage. Goals for fishery management can vary between years based on expected returns of adult salmonids, management of ocean fisheries for chinook salmon, management of recreational fisheries in the Rogue River, and estimates of water yield from the basin.

We chose not to simulate alternative strategies for operation of Lost Creek Dam during flood control and conservation storage seasons (November-April). Although our findings indicated reservoir management in November-April affected production of juvenile fall chinook salmon (*see* **Abundance**, page 39), there is limited flexibility in operation strategies during this period. Flood control operations are a priority and maximum reservoir height is dependent on guidelines in authorizing documents for the project (United States Congress 1962). However, there may be flexibility in minimum height of the reservoir during flood control and conservation storage seasons (*see* **Recommendations**, page 4).

We also chose not to simulate alternative strategies for operation of Applegate Dam in autumn when adults migrated and spawned in the Applegate River. Our results suggested juvenile production increased linearly with increased flow in the Applegate River during autumn. Increased spawning in areas upstream of Murphy Dam (RK 21) is likely responsible (Fustish et al. 1988), but the relationship between spawning distribution and juvenile production in the Applegate River has not been estimated. We believe this analysis is needed to identify optimal use of water stored in Applegate Lake.

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REFERENCES

Alderdice, D.F., and F.P.J. Velsen. 1978. Relation between temperature and incubation time for eggs of chinook salmon (*Oncorhynchus tshawytscha*). Journal of the Fisheries Research Board of Canada 35:69-75.

Amandi, A., S.F. Hui, C.K. Arakawa, and J.S. Rohovec. 1982. Assessment of disease related mortality of adult salmonids in the Rogue River basin. Oregon State University, Department of Microbiology, Progress Report, Corvallis.

Anderson, J.L., and J.E. Wilen. 1985. Estimating the population dynamics of coho salmon (*Oncorhynchus kisutch*) using pooled-time series and cross-sectional data. Canadian Journal of Fisheries and Aquatic Sciences 42:459-467.

Banks, J.L., L.G. Fowler, and J.W. Elliot. 1971. Effects of rearing temperature on growth, body form, and hematology of fall chinook fingerlings. Progressive Fish-Culturist 33:20-26.

Bartley, D.M., and G.A.E. Gall. 1990. Genetic structure and gene flow in chinook salmon populations of California. Transactions of the American Fisheries Society 119:55-71.

Barton, A.C. 1979. Factors influencing the life history of spring chinook salmon (*Oncorhynchus tshawytscha*) spawning in the Columbia River watershed from 1960-77. Doctoral dissertation, Oregon State University, Corvallis.

Beacham, T.D., and C.B. Murray. 1990. Temperature, egg size, and development of embryos and alevins of five species of Pacific salmon: a comparative analysis. Transactions of the American Fisheries Society 119:927-945.

Beacham, T.D., C.B. Murray, and R.E. Withler. 1989. Age, morphology, and biochemical variation of Yukon River chinook salmon. Transactions of the American Fisheries Society 118:46-63.

Becker, C.D., and M.P. Fujihara. 1978. The bacterial pathogen *Flexibacter columnaris* and its epizootiology among Columbia River fish. A review and synthesis. American Fisheries Society, Monograph 2.

Bilton, H.T. 1984. Returns of chinook salmon in relation to juvenile size at release. Canadian Technical Report of Fisheries and Aquatic Sciences 1245.

Bilton, H.T., and G.L. Robins. 1971. Effects of feeding level on circulus formation on scales of young sockeye salmon (*Oncorhynchus nerka*). Journal of the Fisheries Research Board of Canada 28:861-865.

Binns, A., and F. Eiserman. 1979. Quantification of fluvial trout habitat in Wyoming. Transactions of the American Fisheries Society 108:215-228.

Bisson, P.A., and G.E. Davis. 1976. Production of juvenile chinook salmon, *Oncorhynchus tshawytscha*, in a heated model stream. Fishery Bulletin 74:763-774.

Bjornn, T.C. 1971. Trout and salmon movements in two Idaho streams as related to temperature, food, stream flow, cover, and population density. Transactions of the American Fisheries Society 100:423-438.

Bjornn, T.C. 1978. Survival, production, and yield of trout and chinook salmon in the Lemhi River, Idaho. University of Idaho, College of Forestry, Wildlife and Range Sciences Bulletin 27, Moscow.

Bocking, R.C., and R.M. Peterman. 1988. Preseason forecasts of sockeye salmon (*Oncorhynchus nerka*): comparison of methods and economic considerations. Canadian Journal of Fisheries and Aquatic Sciences 45:1346-1354.

Boecheler, J.L., and S.E. Jacobs. 1987. Catch and escapement of fall chinook salmon from Salmon River, Oregon, 1986. Oregon Department of Fish and Wildlife, Fish Research Project, Annual Progress Report, Portland.

Bovee, K.D. 1982. A guide to stream habitat analysis using the instream flow incremental methodology. U.S. Fish and Wildlife Service, Instream Flow Information Paper 12, Fort Collins, Colorado.

Brannon, E.L. 1982. Orientation mechanisms of homing salmonids. Pages 219-227 *in* E.L. Brannon and E.O. Salo, editors. Proceeding of salmon and trout migratory behavior symposium. School of Fisheries, University of Washington, Seattle.

Brett, J.R., W.C. Clarke, and J.E. Shelbourne. 1982. Experiments on thermal requirements for growth and food conversion efficiency of juvenile chinook salmon, *Oncorhynchus tshawytscha*. Canadian Technical Report of Fisheries and Aquatic Sciences 1127.

Briggs, J.C. 1953. The behavior and reproduction of salmonid fishes in a small coastal stream. California Department of Fish and Game, Fish Bulletin 94, Sacramento.

Buckman, M., and R.D. Ewing. 1982. Relationship between size and time of entry into the sea and gill (Na+K)-ATPase activity for juvenile spring chinook salmon. Transactions of the American Fisheries Society 111:681-687.

Burger, C.V., R.L. Wilmot, and D.B. Wangaard. 1985. Comparison of spawning areas and times for two runs of chinook salmon (*Oncorhynchus tshawytscha*) in the Kenai River, Alaska. Canadian Journal of Fisheries and Aquatic Sciences 42:693-700.

Burner, C.J. 1951. Characteristics of spawning nests of Columbia River salmon. United States Fish and Wildlife Fishery Bulletin 61:97-110.

Carl, L.M. 1984. Chinook salmon (*Oncorhynchus tshawytscha*) density, growth, mortality, and movement in two Lake Michigan tributaries. Canadian Journal of Zoology 62:65-71.

Carl, L.M., and Healey M.C. 1984. Differences in enzyme frequency and body morphology among three juvenile life history types of chinook salmon in the Nanaimo River, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 41:1070-1077.

Cederholm, C.J., D.B. Houston, D.L. Cole, and W.J. Scarlett. 1989. Fate of coho salmon (*Oncorhynchus kisutch*) carcasses in spawning streams. Canadian Journal of Fisheries and Aquatic Sciences 46:1347-1355.

Chadwick, E.M.P., T.R. Porter, and P. Downton. 1978. Analysis of growth of Atlantic salmon (*Salmo salar*) in a small Newfoundland river. Journal of the Fisheries Research Board of Canada 35:60-68.

Chapman, D.W. 1986. Salmon and steelhead abundance in the Columbia River in the nineteenth century. Transactions of the American Fisheries Society 115:662-670.

Clarke, W.C., J.E. Shelbourne, and J.R. Brett. 1981. Effect of artificial photoperiod cycles, temperature, and salinity on growth and smolting in underyearling coho (*Oncorhynchus kisutch*), chinook (*O. tshawytscha*), and sockeye (*O. nerka*) salmon. Aquaculture 22:105-116.

Conder, A.L., and T.C. Annear. 1987. Test of weighted usable area estimates derived from a PHABSIM model for instream flow studies on trout streams. North American Journal of Fisheries Management 7:339-350.

Cousens, N.B.F., G.A. Thomas, C.G. Swann, and M.C. Healey. 1982. A review of salmon escapement estimation techniques. Canadian Technical Report of Fisheries and Aquatic Sciences 1108.

Cramer, S.P. 1986. The influence of river temperature and flow on the production and harvest of summer steelhead in the Rogue River. Oregon Department of Fish and Wildlife, Fish Research Project DACW57-77-C-0027, Portland.

Cramer, S.P., and J.T. Martin, editors. 1978. Rogue basin fisheries evaluation program, juvenile salmonid studies. Oregon Department of Fish and Wildlife, Fish Research Project DACW57-77-C-0027, Annual Progress Report, Portland.

Cramer, S.P., and J.T. Martin, editors. 1979. Rogue Basin fisheries evaluation program, adult salmonid studies. Oregon Department of Fish and Wildlife, Fish Research Project DACW57-77-C-0027, Annual Progress Report, Portland.

Cramer, S.P., and B.P. McPherson, editors. 1981. Rogue Basin fisheries evaluation program, adult salmonid studies. Oregon Department of Fish and Wildlife, Fish Research Project DACW57-77-C-0027, Annual Progress Report, Portland.

Cramer, S.P., and B.P. McPherson, editors. 1983. Rogue Basin fisheries evaluation program, adult salmonid studies. Oregon Department of Fish and Wildlife, Fish Research Project DACW57-77-C-0027, Annual Progress Report, Portland.

Cramer, S.P., T.D. Satterthwaite, R.R. Boyce, and B.P. McPherson. 1985. Lost Creek Dam fisheries evaluation phase I completion report. Volume I. Impacts of Lost Creek Dam on the biology of anadromous salmonids in the Rogue River. Oregon Department of Fish and Wildlife, Fish Research Project DACW57-77-C-0027, Portland.

Donaldson, L.R., and D. Menasveta. 1961. Selective breeding of chinook salmon. Transactions of the American Fisheries Society 90:160-164.

Downey, T.W., G.L. Susac, and J.W. Nicholas. 1986. Research and development of Oregon's coastal chinook salmon stocks. Oregon Department of Fish and Wildlife, Fish Research Project NA-85-ABD-00115, Annual Progress Report, Portland.

Doyle, R.W., A.J. Talbot, and R.R. Nicholas 1987. Statistical interrelation of length, growth, and scale circulus spacing: appraisal of a growth rate estimator for fish. Canadian Journal of Fisheries and Aquatic Sciences 44:1520-1528.

Elwood, J.W., and T.F. Waters. 1969. Effects of floods on food consumption and production rates of a stream brook trout population. Transactions of the American Fisheries Society 98:253-262.

Erman, D.C., E.D. Andrews, and M. Yoder-Williams. 1988. Effects of winter floods on fishes in the Sierra Nevada. Canadian Journal of Fisheries and Aquatic Sciences 45:2195-2200.

Everest, F.H., and D.W. Chapman. 1972. Habitat selection and spatial interaction by juvenile chinook salmon and steelhead trout in two Idaho streams. Journal of the Fisheries Research Board of Canada 29:91-100.

Ewing, R.D., S.L. Johnson, H.J. Pribble, and J.A. Lichatowich. 1979. Temperature and photoperiod effects on gill (Na+K)-ATPase activity in chinook salmon (*Oncorhynchus tshawytscha*). Journal of the Fisheries Research Board of Canada 36:1347-1353.

Fisher, J.P., and W.G. Pearcy. 1988. Growth of juvenile coho salmon (*Oncorhynchus kisutch*) off Oregon and Washington, USA, in years of differing coastal upwelling. Canadian Journal of Fisheries and Aquatic Sciences 45:1036-1044.

Folmar, L.C., and W.W. Dickhoff. 1980. The parr-smolt transformation (smoltification) and seawater adaptation in salmonids: a review of selected literature. Aquaculture 21:1-37.

Fraidenburg, M.E., and R.H. Lincoln. 1985. Wild chinook salmon management: an international conservation challenge. North American Journal of Fisheries Management 5:311-329.

Friday, J., and S.J. Miller. 1984. Statistical summaries of streamflow data in Oregon. Volume 2. Western Oregon. United States Geological Survey, Open-file report 84-454, Portland.

Fustish, C.A., S.E. Jacobs, B.P. McPherson, and P.A. Frazier. 1988. Applegate Dam Evaluation. Phase I Completion Report. Impacts of Applegate Dam on the biology of anadromous salmonids in the Applegate River. Oregon Department of Fish and Wildlife, Fish Research Project DACW57-77-C-0027, Portland.

Gangmark, H.A., and R.G. Bakkala. 1960. A comparative study of unstable and stable (artificial channel) spawning streams for incubating king salmon at Mill Creek. California Fish and Game 46:151-164.

Garrison, R.L. 1987. Stock assessment of anadromous salmonids. Oregon Department of Fish and Wildlife, Fish Research Project AFC-130, Annual Progress Report, Portland.

Gilbert, C.H., and W.H. Rich. 1927. Investigations concerning red salmon runs to the Karluk River, Alaska. Bulletin of the United States Bureau of Fisheries Number 43 Part II.

Goodyear, C.P. 1985. Relationship between reported commercial landings and abundance of young striped bass in Chesapeake Bay, Maryland. Transactions of the American Fisheries Society 114:92-96.

Green, P.E.J., and P.D.M. Macdonald. 1987. Analysis of mark-recapture data from hatchery-raised salmon using log-linear models. Canadian Journal of Fisheries and Aquatic Sciences 44:316-326.

Grizzle, J.M. 1981. Effects of hypolimnetic discharge on fish health below a reservoir. Transactions of the American Fisheries Society 110:29-43.

Groves, A.B., G.B. Collins, and P.S. Trefethen. 1968. Roles of olfaction and vision in choice of spawning site by homing chinook salmon. Journal of the Fisheries Research Board of Canada 25:867-876.

Hamlin, D.E., and J.M. Nestler. 1987. Development of a model to predict longitudinal water temperatures for the Rogue River, Oregon. Miscellaneous Paper EL-87-15, United States Army Corps of Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Hankin, D.G. 1990. Effects of month of release of hatchery-reared chinook salmon on size at age, maturation schedule, and fishery contribution. Oregon Department of Fish and Wildlife, Information Reports (Fish) 90-4, Portland.

Hankin, D.G., and M.C. Healey. 1986. Dependence of exploitation rates for maximum yield and stock collapse on age and sex structure of chinook salmon (*Oncorhynchus tshawytscha*) stocks. Canadian Journal of Fisheries and Aquatic Sciences 43: 1746-1759.

Hankin, D.G., and R. McKelvey. 1985. Comment on fecundity of chinook salmon and its relevance to life history theory. Canadian Journal of Fisheries and Aquatic Sciences 42: 393-394.

Harvey, B.C. 1987. Susceptibility of young-of-the-year fishes to downstream displacement by flooding. Transactions of the American Fisheries Society 116:851-855.

Havey, K.A., and R.M. Davis. 1970. Factors influencing standing crops and survival of juvenile salmon at Barrows Stream, Maine. Transactions of the American Fisheries Society 99:297-310.

Healey, M.C. 1980. Utilization of the Nanaimo River estuary by juvenile chinook salmon (*Oncorhynchus tshawytscha*). U.S. National Marine Fishery Bulletin 77:653-668.

Healey, M.C. 1982. Timing and intensity of size-selective mortality of juvenile chum salmon (*Oncorhynchus keta*) during early sea life. Canadian Journal of Fisheries and Aquatic Sciences 39:952-957.

Healey, M.C. 1983. Coastwide distribution and ocean migration patterns of stream- and ocean-type chinook Salmon, *Oncorhynchus tshawytscha*. Canadian Field-Naturalist 97:427-433.

Healey, M.C., and W.R. Heard. 1984. Inter- and intra-population variation in the fecundity of chinook salmon (*Oncorhynchus tshawytscha*) and its relevance to life history theory. Canadian Journal of Fisheries and Aquatic Sciences 41:476-483.

Healey, M.C., R.E. Thomson, and J.F.T. Morris. 1990. Distribution of commercial fishing vessels off southwest Vancouver Island in relation to fishing success and oceanic water properties and circulation. Canadian Journal of Fisheries and Aquatic Sciences 47:1846-1864.

Heggberget, T.G. 1988. Timing of spawning in Norwegian Atlantic salmon (*Salmo salar*). Canadian Journal of Fisheries and Aquatic Sciences 45:845-849.

Heming, T.A. 1982. Effects of temperature on utilization of yolk by chinook salmon (*Oncorhynchus tshawytscha*) eggs and alevins. Canadian Journal of Fisheries and Aquatic Sciences 39:184-190.

Hicks, R.H., and L.D. Calvin. 1964. An evaluation of the punch card method of estimating salmon-steelhead sport catch. Agricultural Experiment Station Technical Report 81, Oregon State University, Corvallis.

Holt, R.A., J.E. Saunders, J.L. Zinn, J.L. Fryer, and K.S. Pilcher. 1975. Relation of water temperature to Flexibacter columnaris infection in steelhead trout (*Salmo gairdneri*), coho (*Oncorhynchus kisutch*) and chinook (*O. tshawytscha*) salmon. Journal of the Fisheries Research Board of Canada 32:1553-1559.

Holtby, L.B. 1988. Effects of logging on stream temperatures in Carnation Creek, British Columbia, and associated impacts on the coho salmon (*Oncorhynchus kisutch*). Canadian Journal of Fisheries and Aquatic Sciences 45:502-515

Holtby, L.B., and M.C. Healey. 1986. Selection for adult size in female coho salmon (*Oncorhynchus kisutch*). Canadian Journal of Fisheries and Aquatic Sciences 43:1946-1959.

Irvine, J.R, I.G. Jowett, and D. Scott. 1987. A test of the instream flow incremental methodology for underyearling rainbow trout, *Salmo gairdnerii*, in experimental New Zealand streams. New Zealand Journal of Marine and Freshwater Resources 21:35-40.

Jensen, A.L. 1986. Functional regression and correlation analysis. Canadian Journal of Fisheries and Aquatic Sciences 43:1742-1745.

Johnson, S.L. 1988. The effects of El Niño on Oregon's coho (*Oncorhynchus kisutch*) and chinook (*O. tshawytscha*) salmon. Fisheries Research 6:105-123.

Jonasson, B.C., and R.B. Lindsay. 1988. Fall chinook salmon in the Deschutes River, Oregon. Oregon Department of Fish and Wildlife, Information Reports (Fish) 88-6, Portland.

Koo, T.S.Y, and A. Isarankura. 1968. Objective studies of scales of Columbia River chinook salmon, *Oncorhynchus tshawytscha* (Walbaum). U.S. Fish and Wildlife Service Fishery Bulletin 66:165-180.

Kope, R.G. 1987. Separable virtual population analysis of Pacific salmon with application to marked chinook salmon, *Oncorhynchus tshawytscha*, from California's Central Valley. Canadian Journal of Fisheries and Aquatic Sciences 44:1213-1220.

Kristiansson, A.C., and J.D. McIntyre. 1976. Genetic variation in chinook salmon (*Oncorhynchus tshawytscha*) from the Columbia River and three Oregon coastal rivers. Transactions of the American Fisheries Society 105:620-624.

Kyle, G.B., J.P. Koenings, and B.M. Barrett. 1988. Density-dependent, trophic level responses to an introduced run of sockeye salmon (*Oncorhynchus tshawytscha*) at Frazier Lake, Alaska. Canadian Journal of Fisheries and Aquatic Sciences 45:856-867.

Leider, S.A. 1985. Precise timing of upstream migrations by repeat steelhead spawners. Transactions of the American Fisheries Society 114:906-908.

Levings, C.D., C.D. McAllister, and B.D. Chang. 1986. Differential use of the Campbell River estuary, British Columbia, by wild and hatchery-reared juvenile chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 43:1386-1397.

Li, J.C.R. 1948. A sampling plan for estimating the number of fish crossing Gold Ray Dam on the Rogue River. Unpublished report to the Oregon State Game Commission, Portland.

Lindsay, R.B., B.C. Jonasson, and R.K. Schroeder. 1989. Spring chinook salmon in the Deschutes River. Oregon Department of Fish and Wildlife, Information Reports (Fish) 89-4, Portland.

Lindsay, R.B., W.J. Knox, M.W. Flesher, E.A. Olsen, and L.S. Lutz. 1985. Study of wild spring chinook salmon in the John Day River System. Oregon Department of Fish and Wildlife, Fish Research Project DE-A179-83BP397996, Final Report, Portland.

Lister, D.B., and C.E. Walker. 1966. The effect of flow control on freshwater survival of chum, coho, and chinook salmon in the Big Qualicum River. Canadian Fish Culturist 37:3-25.

Major, R.L., and J.L. Mighell. 1969. Egg-to-migrant survival of spring chinook salmon (*Oncorhynchus tshawytscha*) in the Yakima River, Washington. Fishery Bulletin 67:347-359.

Mathews, S.B., and R. Buckley. 1976. Marine mortality of Puget Sound coho salmon (*Oncorhynchus kisutch*). Journal of the Fisheries Research Board of Canada 33:77-1684.

Mathews, S.B., and F.W. Olson. 1980. Factors affecting Puget Sound coho salmon (*Oncorhynchus kisutch*) runs. Canadian Journal of Fisheries and Aquatic Sciences 39:1444-1452.

Mathur, D., W.H. Bason, E.J. Purdy, Jr., and C.A. Silver. 1985. A critique of the instream flow incremental methodology. Canadian Journal of Fisheries and Aquatic Sciences 42:825-831.

McGie, A.M. 1977. Anadromous fish research in Oregon's coastal watersheds. Oregon Department of Fish and Wildlife, Fish Research Project 04-6-208-44040, Annual Progress Report, Portland.

McGie, A.M. 1987. Contribution and distribution of catch of 1969- and 1970-brood fall chinook salmon from Chetco River, Oregon. Oregon Department of Fish and Wildlife, Information Reports (Fish) 87-1, Portland.

Meehan, W.R., and D.B. Siniff. 1962. A study of the downstream migrants of anadromous fishes in the Taku River, Alaska. Transactions of the American Fisheries Society 91:399-407.

Merritt, M.F., and K. Roberson. 1986. Migratory timing of upper Copper River sockeye salmon stocks and its implications for the regulation of the commercial fishery. North American Journal of Fisheries Management 6:216:225.

Moffatt, R.L., Wellman, R.E., and J.M. Gordon. 1990. Statistical summaries of streamflow data in Oregon: Volume 1. Monthly and annual streamflow, and flow-duration values. United States Geological Survey, Open File Report 90-118, Portland, Oregon.

Moyle, P.B. 1976. Inland Fishes of California. University of California Press, Berkeley.

Moyle, P.B., and D.M. Baltz. 1985. Microhabitat use by an assemblage of California stream fishes: developing criteria for instream flow determinations. Transactions of the American Fisheries Society 114:695-704.

Mundy, P.R. 1982. Computation of migratory timing statistics for adult chinook salmon in the Yukon River, Alaska, and their relevance to fisheries management. North American Journal of Fisheries Management 4:359-370.

Murphy, M.L. 1985. Die-offs of pre-spawn adult pink salmon and chum salmon in southeastern Alaska. North American Journal of Fisheries Management 5:302-308.

Mysak, L.A. 1986. El Niño, interannual variability and fisheries in the northeast Pacific. Canadian Journal of Fisheries and Aquatic Sciences 43:464-497.

Neilson, J.D., and G.H. Geen. 1986. First-year growth rate of Sixes River chinook salmon as inferred from otoliths: effects on mortality and age at maturity. Transactions of the American Fisheries Society 115:28-33.

Neilson, J.D., G.H. Geen, and D. Bottom. 1985. Estuarine growth of juvenile chinook salmon (*Oncorhynchus tshawytscha*) as inferred from otolith microstructure. Canadian Journal of Fisheries and Aquatic Sciences 42:899-908.

Neter, J., and W. Wasserman. 1974. Applied linear statistical models. Irwin, Homewood, Illinois.

Nicholas, J.W., T.W. Downey, and L.A. Van Dyke. 1985. Research and development of Oregon's coastal chinook stocks. Oregon Department of Fish and Wildlife, Fish Research Project NA-85-ABD-00115, Annual Progress Report, Portland.

Nicholas, J.W., and D.G. Hankin. 1988. Chinook salmon populations in Oregon coastal river basins: description of life histories and assessment of recent trends in run strengths. Oregon Department of Fish and Wildlife, Information Reports (Fish) 88-1, Portland.

Nickelson, T.E. 1986. Influences of upwelling, ocean temperature, and smolt abundance on marine survival of coho salmon (*Oncorhynchus kisutch*) in the Oregon Production Area. Canadian Journal of Fisheries and Aquatic Sciences 43:527-535.

Nickelson, T.E., M.F. Solazzi, and S.L. Johnson. 1986. Use of hatchery coho salmon (*Oncorhynchus kisutch*) presmolts to rebuild wild populations in Oregon coastal streams. Canadian Journal of Fisheries and Aquatic Sciences 43:2443-2449.

ODFW (Oregon Department of Fish and Wildlife). 1991a. Effects of Lost Creek Dam on coho salmon in the Rogue River. Phase II Completion Report. Oregon Department of Fish and Wildlife, Fish Research Project DACW 57-77-C-0033, Completion Report, Portland.

ODFW (Oregon Department of Fish and Wildlife). 1991b. Effects of Lost Creek Dam on the distribution and time of chinook salmon spawning in the Rogue River upstream of Gold Ray Dam. Oregon Department of Fish and Wildlife, Fish Research Project DACW 57-77-C-0033, Special Report, Portland.

ODWR (Oregon Department of Water Resources). 1985. Rogue River Basin Study. Oregon Department of Water Resources, Salem.

PFMC (Pacific Fishery Management Council). 1985. Review of the 1984 ocean salmon fisheries. Pacific Fishery Management Council, Portland, Oregon.

PFMC (Pacific Fishery Management Council). 1988. Review of the 1987 ocean salmon fisheries. Pacific Fishery Management Council, Portland, Oregon.

PFMC (Pacific Fishery Management Council). 1991. Preseason report 1. Stock abundance analysis for the 1991 ocean salmon fisheries. Pacific Fishery Management Council, Portland, Oregon.

Parker, R.R. 1971. Size-selective predation among juvenile salmonid fishes in a British Columbia inlet. Journal of the Fisheries Research Board of Canada 28:1503-1510.

Parker, R.R., and P.A. Larkin. 1959. A concept of growth in fishes. Journal of the Fisheries Research Board of Canada 16:721-745.

Parsley, M.J., D.E. Palmer, and R.W. Burkhardt. 1989. Variation in capture efficiency of a beach seine for small fishes. North American Journal of Fisheries Management 9:239-244.

Peterman, R.M. 1982. Model of salmon age structure and its use in preseason forecasting and studies of marine survival. Canadian Journal of Fisheries and Aquatic Sciences 39:1444-1452.

Peterman, R.M. 1989. Application of statistical power analysis to the Oregon coho salmon (*Oncorhynchus kisutch*) problem. Canadian Journal of Fisheries and Aquatic Sciences 46:1183-1187.

Peterman, R.M., and G.J. Steer. 1981. Relation between sport-fishing catchability coefficients and salmon abundance. Transactions of the American Fisheries Society 110:585-593.

Pope, J.G. 1972. An investigation of the accuracy of virtual population analysis using cohort analysis. International Commission of Northwest Atlantic Fisheries Resources Bulletin 9:65-74.

Power, G. 1981. Stock characteristics and catches of Atlantic salmon (*Salmo salar*) in Quebec, and Newfoundland and Labrador in relation to environmental variables. Canadian Journal of Fisheries and Aquatic Sciences 38:1601-1611.

Quinn, T.P., and K. Fresh. 1984. Homing and straying in chinook salmon (*Oncorhynchus tshawytscha*) from Cowlitz River Hatchery, Washington. Canadian Journal of Fisheries and Aquatic Sciences 41:1078-1082.

Ratti, F. 1979. Natural resources of Rogue estuary. Oregon Department of Fish and Wildlife, Estuary Inventory Report 2-8, Portland.

Reimers, P.E. 1973. The length of residence of juvenile fall chinook salmon in Sixes River, Oregon. Fish Commission of Oregon Research Reports 4(2)3-43.

Reimers, P.E., and G.L. Concannon. 1977. Extended residence of hatchery-released juvenile fall chinook salmon in Elk River, Oregon. Oregon Department of Fish and Wildlife, Information Reports (Fish) 77-2, Portland.

Reimers, P.E., and R.E. Loeffel. 1967. The length of residence of juvenile fall chinook salmon in selected Columbia River tributaries. Fish Commission of Oregon, Research Briefs 13:5-19.

Reisenbichler, R.R. 1987. Basis for managing the harvest of chinook salmon. North American Journal of Fisheries Management 7:589-591.

Rich, W.H. 1925. Growth and degree of maturity of chinook salmon in the ocean. Bulletin of the U.S Bureau of Fisheries 41:15-90.

Ricker, W.E. 1969. Effects of size-selective mortality and sampling bias on estimates of growth, mortality, production, and yield. Journal of the Fisheries Research Board of Canada 26:479-541.

Ricker, W.E. 1972. Heredity and environmental factors affecting certain salmonid populations. Pages 27-160 *in* The stock concept in Pacific salmon. MacMillan Lectures in Fisheries, University of British Columbia, Vancouver.

Ricker, W.E. 1973. Linear regressions in fishery research. Journal of the Fisheries Research Board of Canada 30:409-434.

Ricker, W.E. 1975. Computation and interpretation of biological statistics of fish populations. Fisheries Research Board of Canada Bulletin 191.

Ricker, W.E. 1976. Review of the rate of growth and mortality of Pacific salmon in salt water, and non-catch mortality caused by fishing. Journal of the Fisheries Research Board of Canada 33:1483-1524.

Ricker, W.E. 1980. Causes of the decrease in age and size of chinook salmon (*Oncorhynchus tshawytscha*). Canadian Technical Report of Fisheries and Aquatic Sciences 944.

Rivers, C.M. 1964. Rogue River fisheries. Oregon State Game Commission, unpublished manuscript, Portland.

Roff, D.A. 1984. The evolution of life history parameters in teleosts. Canadian Journal of Fisheries and Aquatic Sciences 41:989-1000.

Rosenberg, D.M., R.A. Bodaly, R.E. Hecky, and R.W. Newbury. 1986. The environmental assessment of hydroelectric impoundments and diversions in Canada. *In* M.C. Healey and R.R. Wallace, editors. Canadian aquatic resources. Canadian Bulletin of Fisheries and Aquatic Sciences 215.

Satterthwaite, T.D. 1991. Evaluation of spring chinook salmon in relation to operation of Lost Creek Dam. Oregon Department of Fish and Wildlife, Fish Research Project (unnumbered), Annual Progress Report, Portland.

Satterthwaite, T.D., B.P. McPherson, P.A. Frazier, and S.P. Cramer. 1985. Rogue Basin fisheries evaluation program, Lost Creek Dam studies. Oregon Department of Fish and Wildlife, Fish Research Project DACW57-77-C-0027, Biennial Progress Report, Portland.

Scarnecchia, D.L. 1984. Forecasting yields of two-sea-winter Atlantic salmon (*Salmo salar*) from Icelandic rivers. Canadian Journal of Fisheries and Aquatic Sciences 41:1234-1240.

Schluchter, M.D., and J.A. Lichatowich. 1977. Juvenile life histories of Rogue River spring chinook salmon, (*Oncorhynchus tshawytscha* Walbaum) as determined by scale analysis. Oregon Department of Fish and Wildlife, Information Reports (Fish) 77-5, Portland.

Schoettger, R.A., and E.W. Steucke, Jr. 1970. Synergic mixtures of MS-222 and quinaldine as anesthetics for rainbow trout and in northern pike. Progressive Fish-Culturist 32:202-205.

Scrivener, J.C., and B.C. Andersen. 1984. Logging impacts and some mechanisms that determine the size of spring and summer populations of coho salmon fry (*Oncorhynchus kisutch*) in Carnation Creek, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 41:1097-1105.

Scrivener, J.C., and M.J. Brownlee. 1989. Effects of forest harvesting on spawning gravel and incubation survival of chum (*Oncorhynchus keta*) and coho salmon (*O. kisutch*) in Carnation Creek, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 46:681-696

Seegrist, D.W., and R. Gard. 1972. Effects of floods on trout in Sagehen Creek, California. Transactions of the American Fisheries Society 101:478-482.

Shardlow, T., R. Hilborn, and D. Lightly. 1987. Components analysis of instream escapement methods for Pacific salmon (*Oncorhynchus* spp.). Canadian Journal of Fisheries and Aquatic Sciences 44:1031-1037.

Smith, A.K. 1973. Development and application of spawning velocity and depth criteria for Oregon salmonids. Transactions of the American Fisheries Society 102:312-316.

Snedecor, A.W., and W.G. Cochran. 1967. Statistical Methods, 6th edition. Iowa State University Press, Ames, Iowa.

Snieszko, S.F. 1974. The effects of environmental stress on outbreaks of infectious diseases of fishes. Journal of Fish Biology 6:197-208.

Solazzi, M.F. 1984. Relationships between visual counts of coho, chinook, and chum salmon from spawning fish surveys and the actual number of fish present. Oregon Department of Fish and Wildlife, Information Reports (Fish) 84-7, Portland.

Steinnes, D.N., and R.L. Raab. 1983. A time series approach to forecasting angling activity at a recreational site based on past successes. Canadian Journal of Fisheries and Aquatic Sciences 40:2189-2193.

Stevens, D.E., and L.W. Miller. 1983. Effects of river flow on abundance of young chinook salmon, American shad, longfin smelt, and delta smelt in the Sacramento-San Joaquin river system. North American Journal of Fisheries Management 3:425-437.

Taylor, S.G. 1980. Marine survival of pink salmon fry from early and late spawners. Transactions of the American Fisheries Society 109:79-82.

Thorne, R.E., and J.J. Ames. 1987. A note on variability of marine survival of sockeye salmon (*Oncorhynchus nerka*) and effects on spawning success. Canadian Journal of Fisheries and Aquatic Sciences 44:1791-1795.

Tuss, C., J. Larson, T. Kisanuki, J. Polos, and C. Craig. 1988. Klamath River fisheries assessment program. United States Fish and Wildlife Service, Report FR1/FAO-88-14, Arcata, California.

United States Congress. 1962. Rogue River basin, Oregon and California. Public Law 87-874, 87th Congress, 2nd Session, House Document 566.

USACE (United States Army Corps of Engineers). 1972. Final environmental statement. Lost Creek Lake Project, Rogue River, Oregon. Portland District, Portland, Oregon.

USACE (United States Army Corps of Engineers). 1983. Water quality investigations--1981, Lost Creek Lake Project, Rogue River, Oregon. Portland District, Portland, Oregon.

USACE (United States Army Corps of Engineers). 1991. Elk Creek Lake environmental impact statement, supplement number 2. Portland District, Portland, Oregon.

van den Berghe, E.P., and M.R. Gross. 1984. Female size and nest depth in coho salmon (*Oncorhynchus kisutch*). Canadian Journal of Fisheries and Aquatic Sciences 41:204-206.

van den Berghe, E.P., and M.R. Gross. 1986. Length of breeding life of coho salmon (*Oncorhynchus kisutch*). Canadian Journal of Zoology 64:1482-1486.

Van Hyning, J.M. 1973. Factors affecting the abundance of fall chinook salmon in the Columbia River. Oregon Fish Commission, Research Briefs 4:1-87.

Wales, J.H., and M. Coots. 1954. Efficiency of chinook spawning in Fall Creek, California. Transactions of the American Fisheries Society 84:137-149.

Walters, C.J. 1986. Adaptive management of renewable resources. MacMillan Publishing Company, New York.

Ward, B.R., P.A. Slaney, A.R. Facchin, and R.W. Land. 1989. Size-biased survival in steelhead trout (*Oncorhynchus mykiss*): back-calculated lengths from adults' scales compared to migrating smolts at the Keogh River, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 46:1853-1858.

Weatherley, A.H. 1972. Growth and ecology of fish populations. Academic Press, New York.

Wedemeyer, G. 1970. The role of stress in disease resistance of fishes. American Fisheries Society, Special Publication 5:30-36.

Wertheimer, A. 1988. Hooking mortality of chinook salmon released by commercial trollers. North American Journal of Fisheries Management 8:346-355.

Wickett, W.P. 1958. Review of certain environmental factors effecting the production of pink and chum salmon. Journal of the Fisheries Research Board of Canada 15:1103-1125.

Zar, J.H. 1984. Biostatistical Analysis, 2nd edition. Prentice-Hall, Inc., Englewood Cliffs, New Jersey.

Zaugg, W.S., J.E. Bodle, J.E. Manning, and E. Wold. 1986. Smolt transformation and seaward migration in 0-age progeny of adult spring chinook (*Oncorhynchus tshawytscha*) matured early with photoperiod control. Canadian Journal of Fisheries and Aquatic Sciences 43:885-888.

APPENDIX A

Relation between Gregorian Day, Day-of-year and Week-of-year.

Gregorian day	Day-of-year	Week-of-year
1-7 January	1-7	1
8-14 January	8-14	2
15-21 January	15-21	3
22-28 January	22-28	4
29 January-4 February	29-35	5
5-11 February	36-42	6
12-18 February	43-49	7
19-25 February	50-56	8
26 February-4 March	57-64	9[a]
5-11 March	65-71	10
12-18 March	72-78	11
19-25 March	79-85	12
26 March-1 April	86-92	13
2-8 April	93-99	14
9-15 April	100-106	15
16-22 April	107-113	16
23-29 April	114-120	17
30 April-6 May	121-127	18
7-13 May	128-134	19
14-20 May	135-141	20
21-27 May	142-148	21
28 May-3 June	149-155	22
4-10 June	156-162	23
11-17 June	163-169	24
18-24 June	170-176	25
25 June-1 July	177-183	26
2-8 July	184-190	27
9-15 July	191-197	28
16-22 July	198-204	29
23-29 July	205-211	30

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[a] *Eight-day week during leap years.*

Gregorian day	Day-of-year	Week-of-year
30 July-5 August	212-218	31
6-12 August	219-225	32
13-19 August	226-232	33
20-26 August	233-239	34
27 August-2 September	240-246	35
3-9 September	247-253	36
10-16 September	254-260	37
17-23 September	261-267	38
24-30 September	268-274	39
1-7 October	275-281	40
8-14 October	282-288	41
15-21 October	289-295	42
22-28 October	296-302	43
29 October-4 November	303-309	44
5-11 November	310-316	45
12-18 November	317-323	46
19-25 November	324-330	47
26 November-2 December	331-337	48
3-9 December	338-344	49
10-16 December	345-351	50
17-23 December	352-358	51
24-31 December	359-366	52[b]

[b] *Eight-day week.*

APPENDIX B

Tables of Data Relating to Studies of Juvenile Fall Chinook Salmon

Appendix Table B-1. Dates of first capture for fry of fall chinook salmon in the Rogue River basin, 1975-81.

Sampling site	1975	1976	1977	1978	1979	1980	1981
Rogue River:							
Matson	03/07	03/28	03/21	04/05	04/16	02/27	03/30
Almeda	04/16	03/07	03/07	03/05	03/26	03/11	03/04
Agness	04/03	03/24	02/22	03/22	03/13	03/03	03/16
Hideaway	03/27	03/18	04/22	04/05	04/03	03/03	03/23
Canfield	03/18	03/18	04/28	03/15	03/13	03/10	03/16
Applegate River[a]	--	03/21	03/01	03/09	03/06	02/22	02/19
Illinois River[a]	04/14	04/15	04/22	04/19	04/17	03/17	03/30

[a] *Sampled at RK 1.*

Appendix Table B-2. Dates of emergence completion for fry of fall chinook salmon in the Rogue River basin, 1975-81. Emergence completion defined as the first date when mean lengths continually exceeded 4.5 cm.

Sampling site	1975	1976	1977	1978	1979	1980	1981
Rogue River:							
Matson	05/15	04/13	04/04	05/08	05/10	04/01	04/21
Almeda	05/15	04/27	04/25	04/10	05/21	05/06	04/07
Agness	05/20	04/21	04/06	04/19	05/15	04/29	03/30
Hideaway	05/29	04/21	04/28	04/12	05/29	05/06	03/23
Canfield	05/29	05/06	04/28	03/29	05/29	05/06	03/23
Applegate River[a]	--	05/16	04/04	04/09	05/09	04/30	03/12
Illinois River[a]	05/28	05/05	04/22	05/10	05/08	04/22	05/11

[a] *Sampled at RK 1.*

Appendix Table B-3. Data used to assess factors related to the abundance of juvenile fall chinook salmon in the Rogue River, 1975-86. Juveniles were not sampled in 1982.

Year	Juvenile abundance[a]	Flow at migration[b]	Peak flow[c]	Spawner abundance[d]	Winter temperature[e]
1975	63	80	30,000	1,692	5.8
1976	174	273	17,900	1,271	5.7
1977	82	52	1,950	945	5.9
1978	20	362	40,300	342	7.9
1979	104	76	12,700	2,022	5.6
1980	64	282	26,900	402	7.0
1981	77	74	9,820	579	7.0
1982	--	607	46,300	1,144	6.7
1983	75	520	50,400	980	6.5
1984	108	766	27,600	460	6.8
1985	267	806	19,000	588	5.6
1986	273	268	8,870	1,630	6.1

[a] *Mean catch per seine haul at Matson and Almeda during May-August.*

[b] *Mean flow (cfs) at Applegate during previous October-November.*

[c] *Largest mean daily flow (cfs) at Grants Pass during previous November-February.*

[d] *Count of female carcasses in standard survey areas during previous year.*

[e] *Mean maximum water temperature (^{o}C) at Grants Pass during previous November-February.*

Appendix Table B-4. Regression analysis of the abundance of subyearling fall chinook salmon in the Rogue River, 1974-85 brood years. Variables are described in Appendix Table B-3.

Independent variable	Regression coefficient	Standard error	*P*
Female spawners	0.0729	0.0317	0.055
Peak flow	-0.0039	0.0012	0.016
Migration flow	0.2812	0.0729	0.006
Constant	41.9		

Analysis of variance

Source of variation	Sum of squares	df	Mean square	*F*	*P*
Regression	52,053	3	17,351	6.76	0.018
Residual	17,968	7	2,567		

Variables tested	Partial r^2		
	Step 1	Step 2	Step 3
Female spawners	0.06	0.21	0.43
Peak flow	0.13	0.39	--
Migration flow	0.18	--	--

Appendix Table B-5. Correlation matrix for variables examined in the analysis of the abundance of juvenile fall chinook salmon, 1975-86. Variables are described in Appendix Table B-3.

Variable	Juvenile abundance	Flow at migration	Peak flow	Spawner abundance	Water temperature
Juvenile abundance	1.00				
Flow at migration	0.37	1.00			
Peak flow	-0.42	0.43	1.00		
Spawner abundance	0.25	-0.50	-0.29	1.00	
Water temperature	-0.52	0.12	0.38	-0.71[a]	1.00

[a] *Significant at* $P \leq 0.05$.

Appendix Table B-6. Growth rate of juvenile fall chinook salmon in fresh water as indexed by the width of spaces between freshwater circuli on adult scales, 1969-84 brood years. Data for age 6 fish were not reported because only five were sampled.

Brood	Mean width (mm at 88X) ± 95% CI				
year	Age 2	Age 3	Age 4	Age 5	Age 2-6[a]
1969	--	--	--	2.02 ± 0.11	--
1970	--	--	2.16 ± 0.05	2.17 ± 0.09	--
1971	--	2.18 ± 0.07	2.19 ± 0.04	2.04 ± 0.08	--
1972	2.13 ± 0.12	2.27 ± 0.07	2.21 ± 0.05	2.11 ± 0.11	2.20 ± 0.04
1973	2.16 ± 0.06	2.09 ± 0.06	2.08 ± 0.05	2.03 ± 0.14	2.06 ± 0.08
1974	2.15 ± 0.06	2.13 ± 0.05	2.12 ± 0.04	1.96 ± 0.15	2.09 ± 0.04
1975	2.19 ± 0.04	2.25 ± 0.05	2.19 ± 0.04	2.08 ± 0.05	2.15 ± 0.03
1976	2.19 ± 0.06	2.26 ± 0.07	2.15 ± 0.05	2.08 ± 0.10	2.14 ± 0.05
1977	2.23 ± 0.06	2.16 ± 0.07	2.18 ± 0.05	2.15 ± 0.21	2.17 ± 0.07
1978	2.10 ± 0.04	2.15 ± 0.04	2.13 ± 0.04	2.11 ± 0.16	2.15 ± 0.05
1979	2.25 ± 0.05	2.39 ± 0.05	2.32 ± 0.06	2.27 ± 0.20	2.31 ± 0.07
1980	2.39 ± 0.03	2.46 ± 0.03	2.32 ± 0.07	2.03 ± 0.10	2.27 ± 0.05
1981	2.35 ± 0.07	2.37 ± 0.08	2.12 ± 0.07	2.13 ± 0.07	2.20 ± 0.05
1982	2.19 ± 0.06	2.00 ± 0.06	2.08 ± 0.06	--	--
1983	2.14 ± 0.09	2.18 ± 0.06	--	--	--
1984	2.11 ± 0.06	--	--	--	--

[a] *Adjusted for age selective mortality that resulted from ocean harvest and the El Niño event of 1982-83.*

Appendix Table B-7. Data used in analyses of freshwater growth for juvenile fall chinook salmon that reared in the Rogue River basin, 1972-81 brood years.

Brood year	Circuli spacing[a]	Mean length[b]	Predicted weight[c]	Flow[d]	Water temperature[e]	Juvenile abundance[f]
1972	2.20	--	--	925	19.7	--
1973	2.06	--	--	1819	18.6	--
1974	2.09	9.7	11.2	1744	19.2	63
1975	2.15	8.8	11.2	1767	19.1	174
1976	2.14	8.7	10.8	966	21.6	82
1977	2.17	9.7	11.8	2167	18.2	20
1978	2.15	9.3	11.3	2258	18.2	104
1979	2.31	10.0	11.6	1959	17.8	64
1980	2.27	9.3	11.5	1864	18.3	77
1981	2.20	--	--	2292	17.4	--

[a] *Average spacing (mm at 88X) between freshwater circuli for the entire brood.*
[b] *Estimated from juveniles seined in summer (Cramer et al. 1985).*
[c] *Estimated from length-weight regressions reported by Cramer et al. (1985).*
[d] *Mean flow (cfs) at Grants Pass during June-August.*
[e] *Mean maximum temperature (°C) at Grants Pass during June-August.*
[f] *Mean catch per seine haul at Matson and Almeda during May-August.*

Appendix Table B-8. Correlation matrix of variables used in analyses of freshwater growth of juvenile fall chinook salmon in the Rogue River basin, 1972-81 brood years. Variables are described in Appendix Table B-7.

	Circuli spacing	Mean length	Predicted weight	Flow	Water temperature	Juvenile abundance
Circuli spacing	1.00					
Length	0.39	1.00				
Weight	0.54	0.74[a]	1.00			
Flow	0.11	0.61	0.80[a]	1.00		
Water temperature	-0.36	-0.71[a]	-0.87[a]	-0.94[a]	1.00	
Juvenile abundance	-0.18	-0.67[a]	-0.49	-0.15	0.17	1.00

[a] *Significant at $P \leq 0.05$.*

Appendix Table B-9. Regression analysis of the mean length for subyearling fall chinook salmon that reared in the Rogue River during summer. Variables are described in Appendix Table B-7.

Independent variable	Regression coefficient	Standard error	*P*
Water temperature	-0.2301	0.082	0.049
Juvenile abundance	-0.0057	0.002	0.064
Constant	14.19		

Analysis of variance

Source of variation	Sum of squares	df	Mean square	*F*	*P*
Regression	1.1339	2	0.567	8.61	0.036
Residual	0.2633	4	0.066		

Variables tested	Partial r^2	
	Step 1	Step 2
Water temperature	0.51	--
Juvenile abundance	0.44	0.62

Appendix Table B-10. Percentage of yearling migrants among fall chinook salmon that originated from the Rogue River basin, 1972-81 brood years.

Brood	Percent yearlings						
year	Age 2	Age 3	Age 4	Age 5	Age 6	Age 2-6[a]	Age 2-6[b]
1972	0.0	0.8	3.1	20.6	--	2.0	4.8
1973	0.0	19.7	6.2	40.2	--	16.7	28.1
1974	0.0	16.9	5.3	20.0	--	6.3	8.7
1975	0.5	7.7	1.1	14.1	78.3	4.0	9.0
1976	0.0	8.7	2.3	1.5	--	2.7	2.3
1977	0.0	4.0	4.4	3.1	--	3.2	3.8
1978	0.0	3.8	4.6	13.8	--	3.4	5.9
1979	0.0	2.4	4.1	0.0	--	1.8	2.6
1980	0.0	1.8	10.4	6.9	--	2.5	7.0
1981	0.0	3.7	2.5	13.9	--	3.6	4.4

[a] *Among observed returns to fresh water.*

[b] *Adjusted for age-elective mortality that resulted from ocean harvest and the El Niño event of 1982-83.*

Appendix Table B-11. Mean date of ocean entry by subyearling fall chinook salmon estimated from adult scales, 1969-84 brood years. Data for age 6 fish were not reported because only five were sampled.

Brood	Mean date ± 95% CI (days)				
year	Age 2	Age 3	Age 4	Age 5	Age 2-6[a]
1969	--	--	--	09/28 ± 20	--
1970	--	--	10/04 ± 6	09/23 ± 11	--
1971	--	09/29 ± 8	09/26 ± 5	10/20 ± 13	--
1972	09/10 ± 10	08/28 ± 5	09/10 ± 5	09/17 ± 11	09/09 ± 4
1973	09/16 ± 6	09/28 ± 9	09/22 ± 6	09/26 ± 16	09/25 ± 9
1974	09/13 ± 5	09/21 ± 5	09/10 ± 4	09/25 ± 20	09/14 ± 5
1975	09/13 ± 4	09/08 ± 4	09/14 ± 4	09/16 ± 6	09/14 ± 4
1976	09/04 ± 7	09/02 ± 10	09/04 ± 7	10/03 ± 13	09/15 ± 6
1977	08/23 ± 6	08/22 ± 8	08/21 ± 6	08/22 ± 19	08/22 ± 6
1978	08/27 ± 5	08/31 ± 4	08/26 ± 3	09/25 ± 47	09/02 ± 9
1979	08/10 ± 5	08/07 ± 4	08/16 ± 5	07/28 ± 12	08/09 ± 4
1980	08/15 ± 6	08/19 ± 4	08/12 ± 7	09/10 ± 10	08/19 ± 6
1981	09/01 ± 8	08/27 ± 7	09/04 ± 5	09/01 ± 9	08/31 ± 4
1982	09/09 ± 10	09/06 ± 8	09/11 ± 5	--	--
1983	08/26 ± 7	09/13 ± 7	--	--	--
1984	08/30 ± 5	--	--	--	--

[a] *Adjusted for age selective mortality that resulted from ocean harvest and the El Niño event of 1982-83.*

Appendix Table B-12. Data used in analyses of date of ocean entry and size at ocean entry for juvenile fall chinook salmon that reared in the Rogue River basin, 1972-81 brood years.

Brood	Ocean entry		Circuli	Mean		Water	Juvenile
year	Date[a]	Size[b]	spacing[c]	length[d]	Flow[e]	temperature[f]	abundance[g]
1972	253	10.4	2.20	--	925	19.7	--
1973	269	11.1	2.06	--	1,819	18.6	--
1974	258	10.2	2.09	9.7	1,744	19.2	63
1975	258	10.3	2.15	8.8	1,767	19.1	174
1976	259	10.3	2.14	8.7	966	21.6	82
1977	235	11.2	2.17	9.7	2,167	18.2	20
1978	246	10.7	2.15	9.3	2,258	18.2	104
1979	222	10.8	2.31	10.0	1,959	17.8	64
1980	232	11.0	2.27	9.3	1,864	18.3	77
1981	244	11.5	2.20	--	2,292	17.4	--

[a] *Mean day-of-year (see* ***APPENDIX A****) estimated for the entire brood.*
[b] *Mean length (cm) estimated for the entire brood.*
[c] *Average spacing (mm at 88X) of freshwater circuli for the entire brood.*
[d] *Juveniles seined during summer at three sites (Cramer et al. 1985).*
[e] *Mean flow (cfs) at Grants Pass during June-August.*
[f] *Mean maximum temperature (°C) at Grants Pass during June-August.*
[g] *Mean catch per seine haul at Matson and Almeda during May-August.*

Appendix Table B-13. Correlation matrix of variables used in analyses of date of ocean entry and the size at ocean entry for juvenile fall chinook salmon that reared in the Rogue River basin, 1972-81 brood years. Variables are described in Appendix Table B-12.

	Ocean entry		Circuli	Mean		Water	Juvenile
	Date	Size	spacing	length	Flow	temperature	abundance
Entry time	1.00						
Entry size	-0.40	1.00					
Circuli spacing	-0.88[a]	0.24	1.00				
Length	-0.69[a]	0.48	0.39	1.00			
Flow	-0.43	0.65[a]	0.11	0.61	1.00		
Water temperature	0.55	-0.72[a]	-0.36	-0.71[a]	-0.88[a]	1.00	
Juvenile abundance	0.50	-0.55	-0.18	-0.67[a]	-0.15	0.17	1.00

[a] *Significant at* $P \leq 0.05$.

Appendix Table B-14. Mean length of subyearling fall chinook salmon at time of ocean entry estimated from adult scales, 1969-84 brood years. Data for age 6 fish were not reported because only five were sampled.

Brood	Mean length (cm) ± 95% CI				
year	Age 2	Age 3	Age 4	Age 5	Age 2-6[a]
1969	--	--	--	10.8 ± 1.1	--
1970	--	--	11.6 ± 0.3	11.0 ± 0.5	--
1971	--	11.5 ± 0.4	11.2 ± 0.2	11.8 ± 0.6	--
1972	10.4 ± 0.4	10.2 ± 0.2	10.4 ± 0.2	10.5 ± 0.7	10.4 ± 0.2
1973	11.3 ± 0.3	11.4 ± 0.3	11.1 ± 0.2	11.1 ± 0.5	11.1 ± 0.3
1974	10.2 ± 0.2	10.6 ± 0.2	10.1 ± 0.2	10.3 ± 0.5	10.2 ± 0.2
1975	10.2 ± 0.2	10.3 ± 0.2	10.5 ± 0.2	10.2 ± 0.2	10.3 ± 0.1
1976	10.0 ± 0.2	10.2 ± 0.3	10.0 ± 0.2	10.8 ± 0.3	10.3 ± 0.2
1977	11.3 ± 0.3	11.0 ± 0.3	11.3 ± 0.3	11.2 ± 0.9	11.2 ± 0.3
1978	10.3 ± 0.2	10.8 ± 0.2	10.4 ± 0.1	11.1 ± 1.9	10.7 ± 0.3
1979	11.6 ± 0.2	11.2 ± 0.2	10.9 ± 0.2	10.4 ± 0.7	10.8 ± 0.2
1980	11.2 ± 0.2	11.1 ± 0.2	10.8 ± 0.3	11.3 ± 0.6	11.0 ± 0.2
1981	11.5 ± 0.4	11.8 ± 0.3	11.6 ± 0.3	11.6 ± 0.5	11.5 ± 0.3
1982	11.8 ± 0.5	11.2 ± 0.4	11.2 ± 0.4	--	--
1983	11.3 ± 0.4	12.4 ± 0.4	--	--	--
1984	11.4 ± 0.3	--	--	--	--

[a] *Adjusted for age selective mortality that resulted from ocean harvest and the El Niño event of 1982-83.*

APPENDIX C

Tables of Data Related to Studies of Adult Fall Chinook Salmon

Appendix Table C-1. Comparisons of catch rate for fall chinook salmon seined at Huntley Park during two-hour intervals beginning at 0600 hours, 1977-86. Time of catch data was not summarized for 1981, 1982, or 1984. Catch rate was ranked from 1 (highest) to 3 (lowest) on a monthly basis. Data from October were not included because we made few sets prior to 0800 hours.

	July			August			September		
Year	0600 -0800	0800 -1000	1000 -1200	0600 -0800	0800 -1000	1000 -1200	0600 -0800	0800 -1000	1000 -1200
1977	1	2	3	1	2	3	1	2	3
1979	3	2	1	2	1	3	3	2	1
1980	3	2	1	2	1	3	2	1	3
1983	3	2	1	3	2	1	1	2	3
1985	3	2	1	1	3	2	1	2	3
1986	3	2	1	3	2	1	1	3	2

Chi-square test for differences among times of day

Statistic	July	August	September
χ^2	5.33	0.33	3.00
df	2	2	2
P	0.070	0.846	0.223

Appendix Table C-2. Weekly catch rate (fish per seine haul) of chinook salmon seined at Huntley Park, 1976-86. Catch rates during 1976 were adjusted for differences in time-of-day sampling compared with other years. Week-of-year calendar is in **APPENDIX A**.

	Week-of-year							
Year	28	29	30	31	32	33	34	35
1976	0.470	0.394	0.268	0.514	0.785	1.197	0.554	0.609
1977	0.549	0.632	0.246	0.313	0.352	0.633	0.867	2.900
1978	0.750	0.062	0.146	0.500	0.021	10.417	4.938	1.125
1979	2.978	0.733	0.111	0.844	3.511	5.822	6.400	4.067
1980	0.222	0.133	0.067	0.556	1.356	2.244	1.489	2.044
1981	0.844	0.133	0.333	1.844	0.089	3.822	1.511	4.222
1982	0.267	0.111	1.289	1.289	4.156	1.933	4.000	2.756
1983	0.222	0.178	0.333	0.067	0.133	0.244	0.578	1.067
1984	0.000	0.000	0.100	0.133	0.600	0.422	0.778	0.644
1985	0.167	0.422	0.644	2.156	1.628	1.600	1.956	1.556
1986	0.911	1.022	0.600	0.533	0.356	4.244	8.200	8.600

	Week-of-year							
Year	36	37	38	39	40	41	42	43
1976	0.948	1.154	2.689	2.153	1.454	0.674	0.265	0.291
1977	0.700	3.033	2.617	2.500	1.983	1.483	0.467	0.611
1978	8.208	3.750	2.333	2.708	4.188	0.917	1.250	0.271
1979	5.333	1.422	3.533	3.311	2.000	0.067	1.289	0.733
1980	3.267	2.733	2.867	1.422	0.444	0.933	0.244	0.467
1981	2.444	4.089	2.200	1.000	1.267	0.206	0.167	0.126
1982	4.956	1.711	2.444	1.089	0.578	0.089	0.178	0.033
1983	0.667	0.267	0.778	0.622	0.400	0.378	0.333	0.133
1984	0.444	0.422	0.756	0.578	0.200	0.089	0.022	0.062
1985	1.800	2.800	0.822	0.511	0.756	0.200	0.022	0.067
1986	5.533	4.067	7.022	4.200	1.133	0.333	0.089	0.100

Appendix Table C-3. Weekly and annual estimates of wild and hatchery fall chinook salmon that passed Huntley Park, 1976-86. Week-of-year calendar is in **APPENDIX A.**

Year	Week-of-year														
	30	31	32	33	34	35	36	37	38	39	40	41	42	43	30-43
1976	432	885	1,486	2,686	1,005	979	1,524	1,854	4,691	3,460	2,492	1,083	425	467	23,469
1977	395	504	566	1,018	1,393	4,661	1,125	4,875	4,205	5,694	3,486	2,384	750	982	32,038
1978	293	897	36	18,205	8,950	1,959	15,390	8,945	4,164	4,645	7,174	1,473	2,009	435	74,575
1979	216	1,498	6,021	10,026	11,238	7,177	9,154	2,286	5,679	5,321	3,214	107	2,239	5,554	69,730
1980	125	1,010	2,460	4,068	2,551	3,286	5,250	4,393	4,607	2,286	714	1,500	430	798	33,478
1981	588	3,240	160	6,818	2,616	7,232	4,199	8,005	3,877	1,607	2,183	391	286	218	41,420
1982	3,062	3,039	9,346	4,422	7,831	5,213	9,801	2,982	5,081	2,864	1,274	201	375	244	55,735
1983	772	147	299	688	2,141	5,508	2,897	1,109	2,011	2,284	1,130	1,100	962	416	21,464
1984	467	644	2,766	1,810	3,002	2,617	1,844	1,444	1,711	1,078	389	229	59	152	18,212
1985	1,292	4,637	3,289	3,224	4,108	3,166	4,498	7,682	1,489	890	1,296	350	39	149	36,109
1986	1,153	1,005	666	7,969	15,517	16,680	10,705	7,136	13,279	21,143	2,135	594	155	177	98,314

Appendix Table C-4. Cohort analysis for wild fall chinook salmon that originated from the Rogue River basin, 1968-84 brood years. Estimates include (1) number of fish alive in the ocean during the spring, (2) harvest and non-harvest mortality during ocean fisheries, and (3) number of immature fish that remained in the ocean.

Age 6

Brood year	Freshwater return[a]	Survivors	Ocean catch	Harvest rate[b]	Number alive
1968	392	392	588	0.60	980
1969	283	283	425	0.60	708
1970	0	0	0	0.60	0
1971	0	0	0	0.60	0
1972	0	0	0	0.60	0
1973	0	0	0	0.68	0
1974	0	0	0	0.75	0
1975	388	388	751	0.66	1,138
1976	49	49	91	0.65	140
1977	0	0	0	0.71	0
1978	90	90	65	0.42	155
1979	0	0	0	0.29	0
1980	56	56	61	0.52	117

Age 5

Brood year	Immatures[c]	Freshwater return[a]	Survivors	Ocean catch	Harvest rate[b]	Number alive
1969	884	2,994	3,878	5,818	0.60	9,696
1970	0	4,190	4,190	6,285	0.60	10,475
1971	0	1,622	1,622	2,433	0.60	4,055
1972	0	910	910	1,365	0.60	2,275
1973	0	4,176	4,176	6,264	0.60	10,440
1974	0	3,054	3,054	6,490	0.68	9,544
1975	1,423	8,762	10,185	30,554	0.75	40,739
1976	175	2,967	3,142	6,099	0.66	9,241

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[a] *Derived from data in Appendix Table C-3 and Appendix Table C-14.*
[b] *Assumed to equal harvest rate of age 4 fish.*
[c] *Assumed an overwinter survival rate of 80%.*

Appendix Table C-4. Continued.

Age 5 continued

Brood year	Immatures[c]	Freshwater return[a]	Survivors	Ocean catch	Harvest rate[b]	Number alive
1977	0	1,485	1,485	2,578	0.65	4,243
1978	194	541	735	1,799	0.71	2,534
1979	0	495	495	358	0.42	853
1980	146	1,615	1,761	719	0.29	2,480
1981	65[d]	1,499	1,564[d]	1,694	0.52	3,258

Age 4

Brood year	Immatures[c]	Freshwater return[a]	Survivors	Ocean catch	Harvest rate	Number alive
1970	13,094	19,656	32,750	49,125	0.60[e]	81,874
1971	5,069	21,075	26,144	39,216	0.60[e]	65,359
1972	2,844	8,055	10,899	16,348	0.60[e]	27,247
1973	13,050	4,126	17,176	25,764	0.60[f]	42,940
1974	11,930	30,173	42,103	63,154	0.60[f]	105,257
1975	50,924	55,923	106,847	227,050	0.68[f]	333,897
1976	11,551	9,500	21,051	63,154	0.75[f]	84,206
1977	5,304	7,126	12,430	24,128	0.66[f]	36,558
1978	17,182[g]	23,408	40,590	75,382	0.65[f]	115,972
1979	3,851[g]	6,139	9,990	24,459	0.71[f]	34,449
1980	3,100	6,125	9,225	6,680	0.42[f]	15,905
1981	4,072	8,835	12,907	5,272	0.29[f]	18,179
1982	6,665[d]	10,490	17,155[d]	18,584	0.52[f]	35,739

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[d] *Data not available for older cohorts. Estimates were derived from average probability of maturity for broods not affected by the El Niño event of 1982-83.*

[e] *Data not available; harvest rates are assumed.*

[f] *Data received from Steven Jacobs, Oregon Department of Fish and Wildlife, Corvallis, Oregon, on 2 October 1990.*

[g] *Estimates were adjusted for effects of the El Niño event of 1982-83.*

Appendix Table C-4. Continued.

Age 3							
Brood year	Immatures[c]	Freshwater return[a]	Survivors	Ocean catch	Harvest rate	Shakers[h]	Number alive
1971	81,699	11,560	93,259	62,173	0.40[e]	3,730	159,162
1972	34,059	7,234	41,293	27,528	0.40[e]	1,652	70,473
1973	53,675	4,088	57,763	38,509	0.40[e]	2,311	98,582
1974	131,571	5,338	136,909	91,273	0.40[f]	5,476	233,658
1975	417,371	28,122	445,493	296,995	0.40[f]	17,820	760,308
1976	105,257	7,015	112,272	63,153	0.36[f]	3,789	179,215
1977	45,697	3,674	49,371	37,245	0.43[f]	2,235	88,850
1978	144,965	19,818	164,783	119,325	0.42[f]	7,160	291,268
1979	166,986[g]	14,422	181,408	142,535	0.44[f]	8,552	332,496
1980	96,689[g]	8,898	105,587	67,506	0.39[f]	4,050	177,143
1981	22,724	6,358	29,082	5,957	0.17[f]	357	35,396
1982	44,673	3,728	48,401	17,006	0.26[f]	1,020	66,427
1983	477,264[d]	45,938	523,202[d]	348,801	0.40[f]	20,928	892,932

Age 2						
Brood year	Immatures[i]	Freshwater return[a]	Survivors	Shakers[h]	Contact rate[h]	Number alive
1972	140,945	8,062	149,007	10,729	0.24	159,736
1973	197,164	4,394	201,558	14,512	0.24	216,071
1974	467,316	9,707	477,023	34,346	0.24	511,368
1975	1,520,616	21,664	1,542,280	111,044	0.24	1,653,324
1976	358,429	12,104	370,533	26,678	0.24	397,212
1977	177,701	3,738	181,439	14,805	0.27	196,244
1978	582,535	11,468	594,003	53,460	0.30	647,464
1979	664,992	10,517	675,509	53,500	0.26	729,009
1980	858,668[g]	14,834	873,502	68,133	0.26	941,635
1981	90,481[g]	2,708	93,189	7,940	0.28	101,128
1982	132,855	3,936	136,791	6,894	0.17	143,685
1983	1,785,863	19,681	1,805,544	62,833	0.12	1,868,377
1984	2,035,668[d]	34,514	2,070,182[d]	129,179	0.21	2,199,361

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[h] *Estimation described in* **METHODS**.

[i] *Assumed overwinter survival rate of 50%.*

Appendix Table C-5. Independent variables included in analyses of factors related to numbers of immature fall chinook salmon of wild origin alive in the ocean prior to the onset of fishing. Data on biological factors can be found in Appendix Table C-4.

Year	Date of ocean entry[a]	Ocean growth[b]	Water temperature[c]		Upwelling[d]	
			Autumn	Spring	Autumn	Spring
1974-75	260	178	11.7	10.2	214	196
1975-76	257	165	12.0	9.8	184	167
1976-77	257	183	11.8	10.6	103	107
1977-78	248	164	12.3	11.2	115	121
1978-79	236	179	12.7	10.9	114	116
1979-80	240	176	13.8	11.5	50	150
1980-81	223	181	11.6	11.6	205	109
1981-82	229	184	13.1	10.5	92	190
1982-83	245	150	12.4	12.3	65	-65
1983-84	253	165	12.7	11.4	104	30
1984-85	239	177	12.6	9.9	90	58
1985-86	243	159	11.6	11.2	85	42

[a] *Mean day-of-year (see* ***APPENDIX A****) estimated from scales.*

[b] *Mean scale radius (mm at 88X) from ocean entry to scale margin.*

[c] *Mean surface temperature (°C) at Coos Bay, Oregon. Autumn represents the period of August-October; spring represents the period of March-May.*

[d] *Sum of Bakun units near Crescent City, California. Autumn represents the period of August-October; spring represents the period of March-May.*

Appendix Table C-6. Correlation matrix for independent variables examined in regression analyses of the number of fall chinook salmon of wild origin alive in the ocean. Variables are described in Appendix Table C-5.

	Date of ocean entry	Ocean growth	Ocean temperature		Ocean upwelling	
			Autumn	Spring	Autumn	Spring
Date of ocean entry	1.00					
Ocean growth	-0.31	1.00				
Autumn temperature	-0.28	0.14	1.00			
Spring temperature	-0.30	-0.46	0.15	1.00		
Autumn upwelling	0.15	0.27	-0.64[a]	-0.38	1.00	
Spring upwelling	0.02	0.69[a]	0.09	-0.61[a]	0.48	1.00

[a] $P \le 0.05$.

Appendix Table C-7. Regression analysis of the number of fall chinook salmon in the ocean at age 2, 1974-80 brood years. Brood years 1981-84 were excluded from the analysis. Variables are described in Appendix Table C-5.

Independent variable	Regression coefficient	Standard error	r^2
Juvenile abundance	8,851	2,076	0.78
Constant	-13,278		

Analysis of variance

Source of variation	Sum of squares	df	Mean square	F	P
Regression	10.57×10^{12}	1	10.57×10^{12}	18.2	0.001
Residual	2.91×10^{12}	5	0.58×10^{12}		

Variables tested	Partial r^2 Step 1	Partial r^2 Step 2
Age 2 return to river	0.78	--
Date of ocean entry	0.06	0.16
Ocean growth	0.30	0.56
Autumn temperature	0.06	0.10
Spring temperature	0.02	0.08
Autumn upwelling	0.03	0.10
Spring upwelling	0.01	0.02

Appendix Table C-8. Regression analysis of the number of fall chinook salmon in the ocean at age 3, 1972-82 brood years. Brood years 1980-81 were excluded from the analysis. Variables are described in Appendix Table C-5.

Independent variable	Regression coefficient	Standard error	r^2
Age 2 return to river	36.380	5.517	0.86
Constant	-110,273		

Analysis of variance

Source of variation	Sum of squares	df	Mean square	F	P
Regression	3.34×10^{11}	1	3.34×10^{11}	43.5	0.001
Residual	0.54×10^{11}	7	0.08×10^{11}		

Variables tested	Partial r^2 Step 1	Partial r^2 Step 2
Age 2 return to river	0.86	--
Ocean temperature, autumn	0.01	0.12
Ocean temperature, spring	0.19	0.00
Ocean upwelling, autumn	0.01	0.17
Ocean upwelling, spring	0.01	0.00

Appendix Table C-9. Regression analysis of the number of fall chinook salmon in the ocean at age 4, 1971-81 brood years. Brood years 1979-80 were excluded from the analysis. Variables are described in Appendix Table C-5.

Independent variable	Regression coefficient	Standard error	r^2
Age 3 return to river	10.182	2.149	0.76
Constant	-13,274		

Analysis of variance

Source of variation	Sum of squares	df	Mean square	F	P
Regression	5.73×10^{10}	1	5.73×10^{10}	22.5	0.002
Residual	1.79×10^{10}	7	0.26×10^{10}		

Variables tested	Partial r^2	
	Step 1	Step 2
Age 3 return to river	0.76	--
Ocean temperature, autumn	0.10	0.00
Ocean temperature, spring	0.06	0.44
Ocean upwelling, autumn	0.05	0.04
Ocean upwelling, spring	0.00	0.20

Appendix Table C-10. Regression analysis of the number of fall chinook salmon in the ocean at age 5, 1970-80 brood years. Brood years 1978-79 were excluded from the analysis. Variables are described in Appendix Table C-5.

Independent variable	Regression coefficient	Standard error	r^2
Age 4 return to river	0.618	0.135	0.75
Constant	-711		

Analysis of variance

Source of variation	Sum of squares	df	Mean square	F	P
Regression	8.47×10^8	1	8.47×10^8	21.0	0.003
Residual	2.82×10^8	7	0.40×10^8		

Variables tested	Partial r^2	
	Step 1	Step 2
Age 4 return to river	0.75	--
Ocean temperature, autumn	0.42	0.09
Ocean temperature, spring	0.35	0.46
Ocean upwelling, autumn	0.14	0.07
Ocean upwelling, spring	0.04	0.01

Appendix Table C-11. Regression analysis of the number of fall chinook salmon in the ocean at age 6, 1969-80 brood years. Brood years 1977-78 were excluded from the analysis. Variables are described in Appendix Table C-5.

Independent variable	Regression coefficient	Standard error	r^2
Age 5 return	0.123	0.039	0.55
Constant	-170		

Analysis of variance

Source of variation	Sum of squares	df	Mean square	*F*	*P*
Regression	7.68×10^5	1	7.68×10^5	9.9	0.014
Residual	6.19×10^5	8	0.77×10^5		

Appendix Table C-12. Regression analysis of the number of spawned carcasses of age 3-6 fall chinook salmon recovered in standard survey areas, 1976-85. Data are described in Appendix Table C-13.

Independent variable	Regression coefficient	Standard error	r^2
Spawning escapement	8.74×10^{-2}	2.36×10^{-2}	0.63
Constant	602		

Analysis of variance

Source of variation	Sum of squares	df	Mean square	*F*	*P*
Regression	12.4×10^6	1	12.4×10^6	13.7	0.006
Residual	7.2×10^6	8	0.9×10^6		

Appendix Table C-13. Data used to assess relationships between estimated spawning escapements of fall chinook salmon and numbers of spawned carcasses recovered in standard survey areas, 1976-85.

Year	Spawning escapement[a]		Carcasses recovered[b]	
	Age 2	Age 3-6	Jacks	Adults
1976	9,770	10,366	801	1,664
1977	21,647	8,277	1,192	926
1978	14,251	48,392	2,235	5,650
1979	743	10,762	100	1,453
1980	4,807	8,384	232	1,242
1981	9,173	24,294	672	3,290
1982	14,785	35,488	665	2,046
1983	2,725	15,176	118	1,119
1984	3,884	12,015	166	1,537
1985	14,103	13,918	1,248	3,442

[a] *Freshwater return - (prespawning mortality + freshwater harvest).*
[b] *Data from Appendix Table C-31.*

Appendix Table C-14. Age composition of wild fall chinook salmon seined in the Rogue River (RK 5-13), 1974-86. Estimates were derived from scale samples, subyearling and yearling migrants combined.

Return	Fish sampled		Percentage of run ± 95% CI[b]				
year	Length[a]	Scales	Age 2	Age 3	Age 4	Age 5	Age 6
1974	220	210	18.90 ± 5.62	27.10 ± 6.32	46.08 ± 7.03	7.02 ± 3.82	0.92 ± 1.71
1975	565	365	11.82 ± 2.84	19.46 ± 3.44	56.69 ± 4.26	11.27 ± 2.78	0.76 ± 0.89
1976	1,048	331	41.36 ± 3.08	17.42 ± 2.39	34.32 ± 2.97	6.91 ± 1.63	0
1977	1,088	345	67.62 ± 2.87	16.66 ± 2.30	12.88 ± 2.08	2.84 ± 1.08	0
1978	1,947	399	16.23 ± 1.69	37.71 ± 2.20	40.46 ± 2.23	5.60 ± 1.07	0
1979	1,815	378	5.36 ± 1.09	10.06 ± 1.44	80.20 ± 1.89	4.38 ± 1.00	0
1980	918	348	34.33 ± 3.18	11.00 ± 2.13	28.44 ± 3.03	26.23 ± 2.95	0
1981	1,055	411	25.77 ± 2.73	48.56 ± 3.11	17.46 ± 2.29	7.27 ± 1.66	0.95 ± 0.68
1982	1,185	372	27.37 ± 2.62	26.61 ± 2.60	43.19 ± 2.90	2.74 ± 1.01	0.09 ± 0.25
1983	276	236	14.81 ± 4.55	48.66 ± 6.25	33.57 ± 5.93	2.96 ± 2.35	0
1984	228	214	23.15 ± 5.90	37.39 ± 6.71	36.02 ± 6.66	2.91 ± 2.61	0.53 ± 1.37
1985	690	285	58.12 ± 3.82	11.01 ± 2.48	26.09 ± 3.42	4.77 ± 1.73	0
1986	1,896	420	37.31 ± 2.23	49.66 ± 2.30	11.34 ± 1.48	1.62 ± 0.62	0.06 ± 0.16

[a] *Includes wild and unmarked hatchery fish.*

[b] *95% CI =* $t_{.05,df}\,(\sqrt{pq/n} + 1/2n)$.

Appendix Table C-15. Data included in analysis of factors related to the probability of maturity at age 2 for wild fall chinook salmon that originated from the Rogue River basin. Scale measurements were taken from age 2 adults.

Brood year	Maturation probability	Date of ocean entry[a]	Size at annulus 1[b]	Plus-growth[c]
1972	0.054	254	99	116
1973	0.022	260	123	104
1974	0.020	257	108	99
1975	0.014	257	142	84
1976	0.033	248	133	72
1977	0.021	236	143	85
1978	0.019	240	129	90
1979	0.016	223	136	93
1980	0.017	228	136	95
1981	0.029[d]	245	127	74
1982	0.029[d]	253	124	93
1983	0.011	239	138	88

[a] *Mean day-of-year (see **APPENDIX A**) estimated from scales.*

[b] *Mean scale radius (mm at 88X) from scale nucleus to first annulus.*

[c] *Mean scale radius (mm at 88X) from annulus 1 to the scale margin.*

[d] *Data excluded from analysis, maturation potentially affected by the El Niño event of 1982-83.*

Appendix Table C-16. Data included in analysis of factors related to the probability of maturity at age 3 for wild fall chinook salmon that originated from the Rogue River basin. Scale measurements were taken from age 3 adults.

Brood year	Maturation probability	Date of ocean entry[a]	Size at annulus 2[b]	Ocean growth	
				Age 2[c]	Plus-growth[d]
1971	0.124	273	245	114	75
1972	0.175	241	249	142	68
1973	0.071	272	244	123	64
1974	0.039	265	263	126	62
1975	0.063	252	256	125	55
1976	0.062	246	261	123	51
1977	0.074	235	254	117	58
1978	0.120	244	261	125	67
1979	0.080	220	270	123	68
1980	0.084[e]	232	272	122	35
1981	0.219[e]	240	230	104	78
1982	0.077	250	235	132	70

[a] *Mean day-of-year (*see ***APPENDIX A****) estimated from scales.*

[b] *Mean scale radius (mm at 88X) from scale nucleus to second annulus.*

[c] *Mean scale radius (mm at 88X) from annulus 1 to annulus 2.*

[d] *Mean scale radius (mm at 88X) from annulus 2 to the scale margin.*

[e] *Data excluded from analysis, maturation potentially affected by the El Niño event of 1982-83.*

Appendix Table C-17. Data included in analysis of factors related to the probability of maturity at age 4 for wild fall chinook salmon that originated from the Rogue River basin. Scale measurements were taken from age 4 adults.

Brood year	Maturation probability	Date of ocean entry[a]	Size at annulus 3[b]	Ocean growth	
				Age 3[c]	Plus-growth[d]
1970	0.600	278	309	78	71
1971	0.806	270	318	89	57
1972	0.739	254	320	83	58
1973	0.240	266	337	98	53
1974	0.717	254	345	98	49
1975	0.523	258	351	98	43
1976	0.451	248	335	84	40
1977	0.573	234	339	95	64
1978	0.577	239	361	102	51
1979	0.614[e]	229	336	82	34
1980	0.664[e]	225	301	62	62
1981	0.684	248	311	101	60

[a] *Mean day-of-year (see* ***APPENDIX A****) estimated from scales.*
[b] *Mean scale radius (mm at 88X) from scale origin to the third annulus.*
[c] *Mean scale radius (mm at 88X) from annulus 2 to annulus 3.*
[d] *Mean scale radius (mm at 88X) from annulus 3 to the scale margin.*
[e] *Data excluded from analysis, maturation potentially affected by the El Niño event of 1982-83.*

Appendix Table C-18. Correlation matrixes for variables examined in analyses of maturation rates of wild fall chinook salmon. Maturation rates were logit transformed prior to analysis. Variables are described in Appendix Tables C-15 through C-17. We excluded data or broods with maturation rates potentially affected by the El Niño event of 1982-83.

Age 2

	Maturity probability	Date of ocean entry	Size at annulus 1	Plus-growth
Maturity probability	1.00			
Date of ocean entry	0.36	1.00		
Size at annulus 1	-0.68[a]	-0.53	1.00	
Plus-growth	0.40	0.25	-0.77[a]	1.00

Age 3

	Maturity probability	Date of ocean entry	Size at annulus 2	Ocean growth	
				Age 2	Plus-growth
Maturation probability	1.00				
Date of ocean entry	-0.17	1.00			
Size at annulus 2	-0.29	-0.50	1.00		
Ocean growth, age 2	0.31	-0.21	-0.19	1.00	
Plus-growth	0.58	0.16	-0.39	0.10	1.00

Age 4

	Maturity probability	Date of ocean entry	Size at annulus 3	Ocean growth	
				Age 3	Plus-growth
Maturity probability	1.00				
Date of ocean entry	0.02	1.00			
Size at annulus 3	-0.36	-0.52	1.00		
Ocean growth, age 3	-0.18	-0.49	0.59	1.00	
Plus-growth	0.29	0.25	-0.64[a]	-0.31	1.00

[a] $P \leq 0.05$.

Appendix Table C-19. Data used to assess factors related to annual harvest rates of age 3 and age 4 chinook salmon in commercial and recreational fisheries off the coast of southern Oregon and northern California, 1980-85. Estimates of fishing effort not available prior to 1979.

Year	Harvest rate[a]		Fishing effort[b]	Number of recruits[c]	Age 3 length[d]
	Age 4	Age 3			
1979	0.68	0.36	56.0	523	68.0
1980	0.75	0.43	43.2	214	68.4
1981	0.66	0.42	43.9	337	72.1
1982	0.65	0.44	46.1	452	72.3
1983	0.71	0.39	22.2	214	64.4
1984	0.42	0.17	10.0	52	66.1
1985	0.29	0.26	14.9	86	69.0
1986	0.52	0.40	23.3	932	72.1

[a] *Data received from Steven Jacobs, Oregon Department of Fish and Wildlife, Corvallis, Oregon, on 2 October 1990.*

[b] *Boat-days in the commercial fishery between Eureka, California, and Coos Bay, Oregon (PFMC 1985; PFMC 1988).*

[c] *Estimated number (1,000s) of wild age 3-5 fall chinook salmon of Rogue River origin in the ocean prior to the onset of fishing (see Appendix Table C-4).*

[d] *Mean length (cm) of wild fall chinook salmon that returned to the Rogue River.*

Appendix Table C-20. Correlation matrix for variables examined in analyses of harvest rates of wild fall chinook salmon caught in the ocean fisheries. Variables are described in Appendix Table C-19.

	Harvest rate		Fishing effort	Number of recruits	Age 3 length
	Age 4	Age 3			
Age 4 harvest rate	1.00				
Age 3 harvest rate	0.78[a]	1.00			
Fishing effort	0.74[a]	0.70[a]	1.00		
Number of recruits	0.22	0.52	0.33	1.00	
Length at age 3	-0.01	0.48	0.39	0.59	1.00

[a] $P \leq 0.05$.

Appendix Table C-21. Incidence of prespawning mortality of fall chinook salmon in the Rogue River, 1944-60. Comments were found in the monthly reports of Cole M. Rivers, district fish biologist with the Oregon Game Commission. No mortality was reported during 1944-46.

Date of report	Comments
September 1947	Heavy mortality of fish in the main river is continuing, but declining with cooling waters. Heavy losses in lower canyon.
August 1949	On boat trip through canyon, 110 adult chinook examined for tags (all dead, no tags) believed to have died from columnaris (August 10-12).
September 1951	...losses have been found negligible this season. Since a study of annual losses was started in 1947, this year shows the lowest mortality for the Rogue.
August 1955	Reoccurrence of losses from columnaris from Galice to Agness. Heavy losses of both adult and immature steelhead and salmon. Concentrations heavy at mouths of tributaries in the canyon. Heaviest losses occur in areas of concentration.
September 1955	Losses this season comparable to 1947.
September 1956	Disease losses were light this season. A boat trip revealed less than 100 dead adult salmon had been cleaned up by black bear; through the lower canyon.
August 1958	In spite of warm waters in the main channel of the Rogue this summer, losses of fish from columnaris were found to be non-existent through the lower canyon on August 15-19.
August 1960	Wild fish losses from columnaris have been moderately heavy in the lower Rogue canyon and lower Applegate. One of the heaviest mortalities is occurring this year since this loss was first observed in 1945.

Appendix Table C-22. Regression analysis of the percentage of fall chinook salmon that died in the Rogue River prior to spawning, 1978-86. Percentage data were logit transformed prior to analysis. Variables are described in Appendix Table C-23.

Independent variable	Regression coefficient	Standard error	r^2
Water temperature	3.820	0.5934	0.86
Constant	-78.117		

Analysis of variance

Source of variation	Sum of squares	df	Mean square	F	P
Regression	53.99	1	53.99	41.44	<0.001
Residual	9.12	7	1.30		

Appendix Table C-23. Data used to assess factors related to the rate of prespawning mortality for fall chinook salmon in the Rogue River, 1978-86.

Year	Mortality rate[a]	Water temperature[b]	Flow[c]	Abundance of chinook salmon	
				Spring race[d]	Fall race[e]
1978	12.5	19.6	2,034	47,221	74,575
1979	81.3	20.8	1,661	38,207	69,730
1980	56.6	20.3	1,700	36,932	33,478
1981	11.6	20.2	1,922	17,213	41,420
1982	4.0	19.6	2,408	29,942	55,735
1983	0.1	19.1	3,048	12,511	21,464
1984	0.4	18.8	3,092	12,270	18,212
1985	0.6	18.9	2,393	41,039	36,109
1986	0.6	19.4	2,368	89,522	98,314

[a] *Percentage that died prior to spawning (see Table 7).*

[b] *Mean maximum water temperature (°C) at Agness during August-September.*

[c] *Mean flow (cfs) at Agness during August-September.*

[d] *Estimated passage at Gold Ray Dam.*

[e] *Estimated freshwater return.*

Appendix Table C-24. Angler harvest of fall chinook salmon in the Rogue River estimated from returns of salmon-steelhead cards, 1956-86. All years were adjusted for a non-response bias.

	Estimated harvest				
Year	August	September	October[a]	November[a]	Total
1956	1,012	1,414	865	187	3,478
1957	817	1,140	642	193	2,792
1958	--	--	--	--	--
1959	1,509	1,280	691	572	4,052
1960	868	909	886	233	2,896
1961	1,025	784	600	170	2,579
1962	1,212	1,110	338	121	2,781
1963	2,315	2,510	1,230	199	6,254
1964	1,948	1,596	816	226	4,586
1965	1,552	1,119	613	133	3,417
1966	851	1,176	1,166	216	3,409
1967	1,163	1,143	670	199	2,505
1968	1,208	1,268	394	106	2,976
1969	1,223	981	691	222	3,117
1970	1,020	536	313	124	1,993
1971	1,085	573	738	161	2,557
1972	594	821	892	205	2,512
1973	380	1,218	1,469	78	3,145
1974	627	1,258	1,105	537	3,527
1975	270	778	833	151	2,032
1976	423	862	621	194	2,100
1977	621	1,426	1,227	316	3,590
1978	412	962	591	345	2,310
1979	509	668	270	84	1,531
1980	206	413	231	49	899
1981	399	1,583	251	37	2,270
1982	538	1,306	419	33	2,296
1983	294	1,300	743	44	2,381
1984	38	1,151	273	16	1,478
1985	917	2,062	580	35	3,594
1986	698	2,942	608	80	4,328

[a] *Includes some coho salmon, 1956-71.*

Appendix Table C-25. Data used to assess factors related to harvest rates of fall chinook salmon in the Rogue River, 1974-86. Data from 1977 were excluded because numerous spring chinook were caught during August.

Year	Harvest rate (%)[a]	Flow[b]	Water temperature[c]	Freshwater return[a]
1974	8.4	1,573	17.8	35,481
1975	5.5	1,671	18.3	34,413
1976	13.5	1,868	18.0	14,086
1978	3.5	2,034	17.2	56,684
1979	2.2	1,661	17.9	64,891
1980	4.0	1,700	16.9	21,078
1981	7.6	1,922	16.2	29,561
1982	5.8	2,408	16.0	39,271
1983	16.5	3,048	16.0	14,153
1984	12.3	3,092	16.1	11,902
1985	21.8	2,393	15.8	16,328
1986	8.2	2,368	16.8	57,976

[a] *Data limited to fish that exceeded the minimum size required for entry on salmon-steelhead cards.*

[b] *Mean flow (cfs) at Agness during August-September.*

[c] *Mean maximum temperature (^{o}C) at Grants Pass during August-September.*

Appendix Table C-26. Correlation matrix for variables examined in the regression analysis of the harvest rate of fall chinook salmon in the Rogue River. Description of variables can be found in Appendix Table C-25.

	Harvest rate	Water temperature	Flow	Freshwater return
Harvest rate	1.00			
Water temperature	-0.43	1.00		
Flow	0.55	-0.75[a]	1.00	
Freshwater return	-0.76[a]	0.37	-0.39	1.00

[a] $P \leq 0.05$.

Appendix Table C-27. Regression analysis of the percentage of the annual harvest of fall chinook salmon in the Rogue River that occurred during September, 1965-86. Percentage data were logit transformed prior to analysis. Variables are described in Appendix Table C-28.

Independent variable	Regression coefficient	Standard error	r^2
Flow	9.039×10^{-4}	2.2187×10^{-4}	0.48
Constant	-1.612		

Analysis of variance

Source of variation	Sum of squares	df	Mean square	F	P
Regression	3.59	1	3.59	9.94	0.014
Residual	4.33	20	0.22		

Appendix Table C-28. Data used to assess the relationship between the timing of angler harvest of fall chinook salmon and flow in the Rogue River, 1974-86.

Year	% harvest in September[a]	Flow[b]	Year	% harvest in September[a]	Flow[b]
1965	32.7	1,242	1976	41.0	1,404
1966	34.5	1,152	1977	39.7	1,383
1967	45.6	1,152	1978	41.6	1,742
1968	42.6	790	1979	43.6	1,499
1969	31.5	1,131	1980	45.9	1,333
1970	26.9	1,056	1981	69.7	1,759
1971	22.4	1,594	1982	56.9	1,985
1972	32.7	1,567	1983	54.6	2,642
1973	38.7	1,098	1984	77.9	2,339
1974	35.7	1,384	1985	57.4	2,079
1975	38.3	1,304	1986	68.0	2,035

[a] *Percentage of harvested fish caught during September.*
[b] *Mean flow (cfs) at Grants Pass during September.*

Appendix Table C-29. Data used to assess factors related to the percentage of wild fall chinook salmon that migrated upstream of Gold Ray Dam, 1974-86. Passage data were not reported for 1977, 1979, and 1980 because returns were affected by >50% prespawning mortality in downstream areas or a delay in migration that caused spring chinook salmon to be counted as fall chinook salmon.

Year	Passage estimate[a]	Percent passage[b]	Flow[c]	Water temperature[d]
1974	2,309	6.0	1,484	16.4
1975	2,312	6.6	1,537	16.9
1976	2,648	13.2	1,545	17.5
1977	--	--	1,460	17.2
1978	5,878	9.4	1,757	15.0
1979	--	--	1,555	16.0
1980	--	--	1,434	15.2
1981	4,290	12.8	1,811	13.8
1982	4,072	8.1	1,976	14.3
1983	3,607	20.2	2,507	14.3
1984	2,978	18.7	2,350	14.2
1985	6,236	22.2	2,060	12.8
1986	7,140	8.0	1,886	13.5

[a] *Wild fish only.*

[b] *Percent passage = (passage/spawning escapement) x 100. Estimates of spawning escapement can be found in Appendix Table C-13.*

[c] *Mean flow (cfs) at Raygold during September.*

[d] *Mean maximum temperature (°C) at Raygold during September.*

Appendix Table C-30. Regression analysis of the percentage of wild fall chinook salmon that migrated upstream of Gold Ray Dam, 1974-86. Percentage data were logit transformed prior to analysis. Variables are described in Appendix Table C-29.

Independent variable	Regression coefficient	Standard error	r^2
Flow	0.0013	3.94×10^{-4}	0.55
Constant	-4.5303		

Analysis of variance

Source of variation	Sum of squares	df	Mean square	F	P
Regression	1.9131	1	1.9131	11.62	0.009
Residual	1.3166	8	0.1646		

Appendix Table C-31. Number of unmarked carcasses of fall chinook salmon found during surveys of spawning areas, 1974-85.

	Rogue River (RK)				Applegate Basin			Other tributaries	
					River (RK)		Slate	Lobster	Quosatana
Year	139-156	158-169	179-183	197-199	0-6	13-17	Creek	Creek	Creek
1974	526	246	335	30	3,204	852	129	650	264
1975	768	242	400	53	1,388	669	803	391	332
1976	900	377	474	65	1,091	328	0	297	14
1977	572	174	449	34	935	784	162	257	490
1978	1,174	595	756	119	4,807	1,007	1,148	748	1,337
1979	189	102	218	73	586	309	550	381	364
1980	242	132	170	54	826	280	236	57	24
1981	1,191	764	370	51	1,941	623	442	160	0
1982	1,130	--	454[a]	--	877	300	250	--	--
1983	476	--	184	--	324	308	253	--	--
1984	580	--	270	--	501	405	352	--	--
1985	1,254	--	557	--	2,073	2,099	806	--	--

[a] *Estimated by regression based on counts in other survey areas.*

Appendix Table C-29. Data used to assess factors related to the percentage of wild fall chinook salmon that migrated upstream of Gold Ray Dam, 1974-86. Passage data were not reported for 1977, 1979, and 1980 because returns were affected by >50% prespawning mortality in downstream areas or a delay in migration that caused spring chinook salmon to be counted as fall chinook salmon.

Year	Passage estimate[a]	Percent passage[b]	Flow[c]	Water temperature[d]
1974	2,309	6.0	1,484	16.4
1975	2,312	6.6	1,537	16.9
1976	2,648	13.2	1,545	17.5
1977	--	--	1,460	17.2
1978	5,878	9.4	1,757	15.0
1979	--	--	1,555	16.0
1980	--	--	1,434	15.2
1981	4,290	12.8	1,811	13.8
1982	4,072	8.1	1,976	14.3
1983	3,607	20.2	2,507	14.3
1984	2,978	18.7	2,350	14.2
1985	6,236	22.2	2,060	12.8
1986	7,140	8.0	1,886	13.5

[a] *Wild fish only.*
[b] *Percent passage = (passage/spawning escapement) x 100. Estimates of spawning escapement can be found in Appendix Table C-13.*
[c] *Mean flow (cfs) at Raygold during September.*
[d] *Mean maximum temperature (°C) at Raygold during September.*

Appendix Table C-30. Regression analysis of the percentage of wild fall chinook salmon that migrated upstream of Gold Ray Dam, 1974-86. Percentage data were logit transformed prior to analysis. Variables are described in Appendix Table C-29.

Independent variable	Regression coefficient	Standard error	r^2
Flow	0.0013	3.94×10^{-4}	0.55
Constant	-4.5303		

Analysis of variance

Source of variation	Sum of squares	df	Mean square	*F*	*P*
Regression	1.9131	1	1.9131	11.62	0.009
Residual	1.3166	8	0.1646		

Appendix Table C-31. Number of unmarked carcasses of fall chinook salmon found during surveys of spawning areas, 1974-85.

	Rogue River (RK)				Applegate Basin			Other tributaries	
					River (RK)		Slate	Lobster	Quosatana
Year	139-156	158-169	179-183	197-199	0-6	13-17	Creek	Creek	Creek
1974	526	246	335	30	3,204	852	129	650	264
1975	768	242	400	53	1,388	669	803	391	332
1976	900	377	474	65	1,091	328	0	297	14
1977	572	174	449	34	935	784	162	257	490
1978	1,174	595	756	119	4,807	1,007	1,148	748	1,337
1979	189	102	218	73	586	309	550	381	364
1980	242	132	170	54	826	280	236	57	24
1981	1,191	764	370	51	1,941	623	442	160	0
1982	1,130	--	454[a]	--	877	300	250	--	--
1983	476	--	184	--	324	308	253	--	--
1984	580	--	270	--	501	405	352	--	--
1985	1,254	--	557	--	2,073	2,099	806	--	--

[a] *Estimated by regression based on counts in other survey areas.*