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Department of Fish and Game

Fish Bulletin 179

Contributions to the Biology of Central Valley Salmonids

Volume 2

Edited by

Randall L. Brown
Department of Water Resources
Sacramento, California

2001

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Contents

Preface

The Salmonid Symposium was organized by an ad hoc committee of state and federal fishery biologists concerned with the management of Central Valley (CV) salmon and steelhead trout (*Oncorhynchus* spp.) populations and their habitats. It was held at Bodega Bay, California on October 22–24, 1997. Topics covered included research on various CV salmon and steelhead populations, ocean fishery management, history of upper Sacramento River hatchery operations, and steelhead management policy.

Dedication

Fish Bulletin 179 is dedicated to the memory of Nat Bingham. Zeke Grader penned the text, but the feelings and inspiration come from the California community of fishermen, salmon biologists and managers.

It was about 10 years ago, the news had just come out that only 191 winter-run chinook had returned to the Sacramento River that year, when, in a call, Nat said something to the effect: "We've got to do something. This run will not go extinct on our watch." With that pronouncement, he set in motion a whirlwind of activity that, although we weren't certain in what direction, determined this magnificent run of salmon, spawning in the tributaries of the Upper Sacramento in the heat of the summer, those fish Livingston Stone chronicled more than a century before, would not be lost.

The campaign to save the winter-run began, and the eventual captive broodstock program and all of the products of that effort, was much like FDR's approach to the depression. That is, try something, do something, but just don't sit there. Nat Bingham, an ardent student of history may well have thought of that. Nat was going to do something. Initially, he considered a pen-rearing program at the National Marine Fisheries Service's Tiburon Laboratory, but after gathering the agencies and scientists together an alternate plan began to evolve. The fact that his original concept was rejected didn't bother him. He cared more that an action plan to save the run was now in motion.

Nat also knew that to save fish—again, as a student of history—the battle had to be engaged on many fronts. A captive broodstock program might prevent extinction of the winter-run, but action had to be taken to correct the problems that had led to the drastic decline of these fish. In a score of years the number of spawners had plummeted from almost 120,000 to less than 200. Litigation, lobbying Congress, cajoling farmers and water districts became Nat's almost daily activity until he died.

Nat had come from a famous old Connecticut family and started commercial fishing in the Bahamas as a teenager. He arrived in Berkeley in the sixties and shortly after that began commercial fishing salmon and albacore out of the East Bay. A few years later he ended up on California's north coast where, as a salmon troller, he began to take an interest in the factors affecting salmon productivity. He familiarized himself with the watersheds and the streams and was soon working with groups such as the Salmon Unlimited and the Salmon Trollers Marketing Association. He helped install and operate hatch box programs aimed at jump-starting runs that had nearly been extirpated from damage to the watershed. He saw first hand that logging, road building and a host of other land use activities were decimating the runs. Unlike most of his contemporaries, he would speak out. And, he railed against what he described as the "code of silence" among those in fisheries who would not actively defend the fish. "No more silence" was his mantra.

Outspoken yes, but Nat was also a gentle person who did not see those across the table as enemies but merely people who needed to be educated about the fish, who needed to understand what the fish needed. He never personalized a fight. He was never anti-logging, anti-grazing, anti-farming, or anti-urban water usage, he was just pro-fish. He never saw winning for the fish as defeating someone else. He was the practitioner of what many now call "win-win."

He was also tireless. In the early 1980s, at the height of an El Nino, he took over as president of a beleaguered Pacific Coast Federation of Fishermen's Associations (PCFFA), a more or less coastwide umbrella group of family-based fishing organizations. Ocean conditions associated with El Nino had devastated salmon production and left the group's coffers nearly empty. Over the next decade he found himself fishing less and spending more time helping with the organization and working on battles to save salmon from the Central Valley to the Columbia. He worked with tribes and ranchers in the Klamath Basin and with the timber industry in coastal watersheds—always trying to save, to rebuild salmon runs. He built alliances with conservation organizations and he looked for opportunities to work with those generally considered his adversaries—from timber industry executives, to power companies, to heads of agricultural and urban water districts. There were few meetings on salmon where Nat was not present.

In the early 1990s seeing no end to the fight for salmon survival, Nat decided to step down as President of PCFFA, a job he could very well have held for life, to sell his boat and dedicate himself exclusively to efforts to restore salmon habitat and rebuild the runs. PCFFA was able to cobble some monies together from government and private foundation contracts and grants and put Nat on the road. For the next seven years his beat-up Toyota pickup, held together it seems by bumper stickers, could be seen up and down the Central Valley, in the Sierra or the Trinity or in some coastal watershed. Nat the salmon disciple, the crusader would be working patiently and in his quiet way to convince people to do things differently so salmon could not only survive, but thrive.

In the spring of 1998, things were looking up for Nat. Quietly working behind the scenes he was able in six-month's time to help establish a winter chinook conservation hatchery on the mainstem of the Sacramento, just below Shasta Dam. Nat called it the Livingston Stone Hatchery, a name that has stuck. Moreover, negotiations with Pacific Gas & Electric were progressing for the removal of dams on Battle Creek to establish an additional "homestream" for the winter run. But it was also a tiring period, the Pacific Fishery Management Council meetings (to which Nat was appointed to a few years before) were particularly arduous. At the end of the April Council meeting Nat's wife Kathy was diagnosed with terminal cancer and by the end of the month she was gone. Nat kept his spirits up, but he was exhausted physically and mentally and within a week of Kathy's death, he was gone too.

Nat's life is the stuff of a great book. The important thing, however, for those of us left working for the survival of the salmon to remember what he did and how he did it—and, how he lived his life. With Nat's life as our inspiration, we will win.

Zeke Grader

In Appreciation

With the release of this Fish Bulletin, we extend our appreciation and those of our fellow biologists to its editor, Dr. Randall L. Brown. As local readers are aware, Randy retired last year from State service where he was employed for over 34 years by the California Department of Water Resources.

He will be forever remembered for his great devotion to improving our understanding of salmon biology in the Central Valley and San Francisco Bay-Delta Estuary of California. Randy's professionalism, support, encouragement and friendship to all of us in the salmon community is greatly respected and appreciated. His tireless efforts to enhance salmon monitoring and research as a coordinator in the Interagency Ecological Program, Chief Biologist for the Department, member of numerous committees related to salmon and their management, and as a leader in conducting multiple workshops, meetings, conferences, and symposiums on salmon has greatly improved our knowledge of salmon. Our progress in the area of salmon population genetics, salmon–hydrodynamics interactions, monitoring and evaluation techniques, population dynamics, data management and other fields are directly related to his personal efforts and accomplishments.

We join together to thank Randy as a friend and colleague for his excellent work and wish him the best in his retirement and all future endeavors.

Marty Kjelson
Terry J. Mills

Acknowledgements

Pulling this volume together would not have been possible without the support of Marty Kjelson and Terry Mills. We first discussed the concept over Chinese food a year or so before the Bodega meeting. Periodic meetings before and after Bodega kept me on track—to the extent that is possible.

Special thanks to the symposium presenters for converting their talks to papers. Joe Miyamoto of the East Bay Municipal Utility District receives the award for being, by far, the first to submit a manuscript.

I would also like to acknowledge several authors who did not present papers at Bodega but who were willing to contribute material to help make this a more balanced compendium.

Several anonymous peer reviewers took their valuable time to review the articles and their comments made for a better product.

L.B. Boydstun, of the California Department of Fish and Game, deserves recognition for allowing us to use the Department's Fish Bulletin series and to serve as the DFG sponsor. This is in keeping with L.B.'s long history of working with his agency, NMFS and the commercial and recreational fishing industry to scientifically manage a resource of special significance to California.

Finally, we should all thank Lauren Buffaloe (DWR) for a tremendous job of editing and formatting the articles and to Barbara McDonnell (DWR) and Sam Luoma (CALFED) for funding publication of the Fish Bulletin.

Randall L. Brown

Foreword

The impetus for publication of this Fish Bulletin came from conversations among several biologists working on salmonid issues in the Central Valley and the Sacramento-San Joaquin Estuary. These discussions centered on the idea that more information being developed about these economically, environmentally, and aesthetically important species needed to be available in the open literature. Marty Kjelson, Terry Mills and I developed the concept of a symposium followed by published proceedings. The Interagency Ecological Program's Central Valley Salmonid Team endorsed the concept and a successful symposium was held at the Bodega Marine Laboratory in October 1997.

Originally Marty and Terry agreed to co-edit the proceedings. Due to the press of other work, they were unable to take on much of the day-to-day work on the volume but did provide guidance and suggestions for ways to move the publication from concept to reality. I take responsibility for the final selection of papers and the final technical editing of the papers.

As you will find, I selected papers with varied writing styles. Some papers, such as the ones by Yoshiyama and others and by Black, are longer than would be typically found in journals. I believe they make a significant contribution to our understanding and decided to publish them without major revision. Others are more succinct and could be published in the open literature.

Those readers that attended the Bodega symposium will find that not all the papers presented have been included in this volume and that papers not presented are included. Several of the presenters were unable to find the time to prepare a manuscript. On the other hand, other authors had information of interest. The blend seemed to make the best sense in view of the objective of making a wide variety of information available to salmonid biologists and managers.

This volume also includes some material that could be considered duplicative in that two different papers may discuss the same question—for example, through-Delta survival of juvenile salmonids. I included these papers to provide different perspectives on important questions. I ask the reader to consider the papers, and the data, and reach his or her conclusions as to the interpretations. As with most difficult environmental issues, one must carefully consider all the available data before deciding to accept or reject a hypothesis.

I do recommend that you consider recommendations, made specifically by L.B. Boydstun, Peter Baker, Emil Morhardt, Wim Kimmerer and others, and John Williams about the need to (1) better coordinate salmonid related work in the Valley, the estuary and the ocean; (2) focus more on collecting and analyzing data that can be used to validate conceptual and mechanistic models; and (3) make the information more readily available in the open literature. Along those lines I suggest that symposium such as this be held every two to three years, including publication of the proceedings. Authors should not stop with publication in proceedings but should also publish in appropriate journals. Hopefully the next symposium will have more than one paper dealing with steelhead.

Randall L. Brown
Fair Oaks, California
September 1, 2001

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Chinook Salmon in the Lower American River, California's Largest Urban Stream

John G. Williams

Abstract

The American River now supports a mixed run of hatchery and naturally-produced fall-run chinook averaging about 30,000 spawners; the spring-run was lost to dams. Salmon in the river have been much studied over the last 20 years, largely because of litigation over proposed diversions, but much uncertainty remains about various aspects of their biology and about the environmental conditions needed to support them. This paper briefly reviews what is known and not known about salmon in the American River and makes recommendations for future work.

Introduction

The American River is the second largest tributary of the Sacramento and supports a mixed run of hatchery and naturally produced fall-run chinook salmon. Salmon in the American River have been intensively studied, largely because of litigation challenging a proposed diversion of water, but much remains to be learned. Here I review what is known about chinook salmon in the American River and give suggestions for future work.

Folsom Dam, a Central Valley Project facility completed in 1955 about 30 miles upstream from the Sacramento River, creates a 975,000 acre-foot reservoir and regulates flows in the reach now accessible to salmon. Salmon migration is blocked at river mile 23 by Nimbus Dam, a regulating facility for Folsom hydropower operations that also diverts a small amount of water into the Folsom-South Canal. Below Nimbus Dam, the lower American River flows through a parkway, surrounded by urban development and is a major recreational area for the Sacramento region. The American River is designated as a recreational river in the state and federal wild and scenic river systems. On average, tens of thousands of hatchery or naturally produced chinook salmon return each year to spawn in California's largest urban stream.

In natural conditions the American River supported spring, fall, and perhaps late fall chinook. Historical data on the upstream extent of salmon migrations are summarized in Yoshiyama and others (Volume 1). Salmon runs were devastated by hydraulic gold mining, and in 1886 the California Fish Commission reported that:

> *The American River is a shallow, muddy stream and empties into the Sacramento River at Sacramento City. But few fish are found in the lower parts of the stream. Trout are found in some of its branches above the mining districts—notably Silver River and the Rubicon. This river, prior to placer mining, was one of the best salmon streams in the state. Of late years no salmon have ascended it.*

Salmon can be resilient, however; 44 years later, G. H. Clark (1929) wrote that although the old Folsom Dam blocked passage for salmon the area downstream supported a large run.

> *The run of salmon into the American River has always been a late fall migration*[1] *and like the other rivers has known great runs. In 1927–1928 there was a very good run in the river, which has shown the inhabitants no noticeable decrease in the last twenty years. It was reported that the run of salmon in this river had been destroyed by the early mining operations. Such may have been the case, but since then the run has returned and has remained fairly constant, according to the observations of local residents.*

Clark reported that the old Folsom Dam, constructed in the late 1890s, effectively blocked salmon passage although it had a ladder that passed steelhead. Subsequent ladder counts showed a few spring-run chinook, but any prospect for restoring that run were dimmed considerably by construction of Folsom and Nimbus dams.

Physical Setting

The American River drains a roughly triangular watershed of about 1,900 square miles that is widest at the crest of the Sierra and narrows almost to the width of the river at its confluence with the Sacramento River at Sacramento. As described in USACE (1991):

> *The American River drainage basin above Folsom Dam is very rugged, with rocky slopes, V-shaped canyons, and little flat valley or plateau area. Elevations range from 10,400 feet at the headwaters to about 200 ft at Folsom Dam, with an average basin slope of 80 feet per mile. The upper third of the basin has been intensely glaciated and is alpine in character, with bare peaks and ridges, considerable areas of granite pavement, and only scattered areas of timber. The middle third is dissected by profound canyons, which have reduced the inter-stream areas to narrow ribbons of relatively flat land. The lower third consists of low rolling mountains and foothills.*

Below Folsom, the watershed flattens into the Central Valley, but the river remains confined or semi-confined by resistant Pleistocene fan deposits or by

1. Presumably these were fall-run chinook that spawned later than runs in some other rivers, like the current run, rather that late fall-run fish.

levees and has only a narrow flood plain that has been aggraded by debris from hydraulic mining. The channel of the lower American River is described in Snider and others (1992), and Beak Consultants and others (1992). Generally, the gradient of the river decreases over the 23 miles between Nimbus Dam and the Sacramento River, and the size of the particles making up the bed decreases from cobble and gravel to sand. This transition is not smooth, however, and there are large pools separated by steeper reaches along much of the lower river.

Snider and others (1992) divided the lower American River into three reaches (Figure 1). Reach 1, the 4.9 miles from the Sacramento confluence to Paradise Beach, has a very low gradient and sand bed. Depth is normally controlled by the stage in the Sacramento River, rather than discharge, and varies with the tide. Reach 2 includes the 6.7 miles of channel from Paradise Beach to Gristmill, with some slope (average gradient about 0.0005). The bed is mainly sand, but includes some gravel riffles. Reach 3 covers 11.1 miles from Gristmill to the weir at Nimbus Hatchery with more slope (average gradient about 0.001). The bed is mainly gravel, but the river is still characterized by long pools separated by riffles. The average width of the river at a flow of 1,000 cfs in the three reaches is 350, 375, and 275 feet.

The annual discharge in the river averages about 3,750 cfs, or about 2,710,000 acre-feet per year, but has varied from 730 to 7,900 cfs. Runoff comes from winter rains at lower elevations and from spring snowmelt at higher elevations, but very high flows all result from winter storms. Discharge is regulated by various dams, of which Folsom is the largest, with past and present direct diversions being relatively minor. The main hydrological effect of the dams has been to dampen variance in winter runoff and to store snowmelt for release in the spring to meet irrigation demand, mainly in the San Joaquin Valley, with the variance and timing of runoff being changed more than the total amount.

"Natural" mean monthly flows have been estimated by the Bureau of Reclamation (Figure 2A), and on average rise to a peak in May and drop to low levels in August through October. Flows reflecting diversions, regulations, and operating practices in effect in 1993 have been estimated by the Sacramento Area Flood Control Agency (Figure 2B) and show less variation over the year and within winter and spring months, but more variation within summer and early fall months. Comparison of daily flows from the moderately dry years 1908 and 1992 shows these effects in more detail (Figure 3). Because Folsom Reservoir is relatively small compared to the mean annual flow in the river; however, reductions in peak flows in wet years have been moderate (Figure 4), and geomorphically effective flows still occur with some frequency.

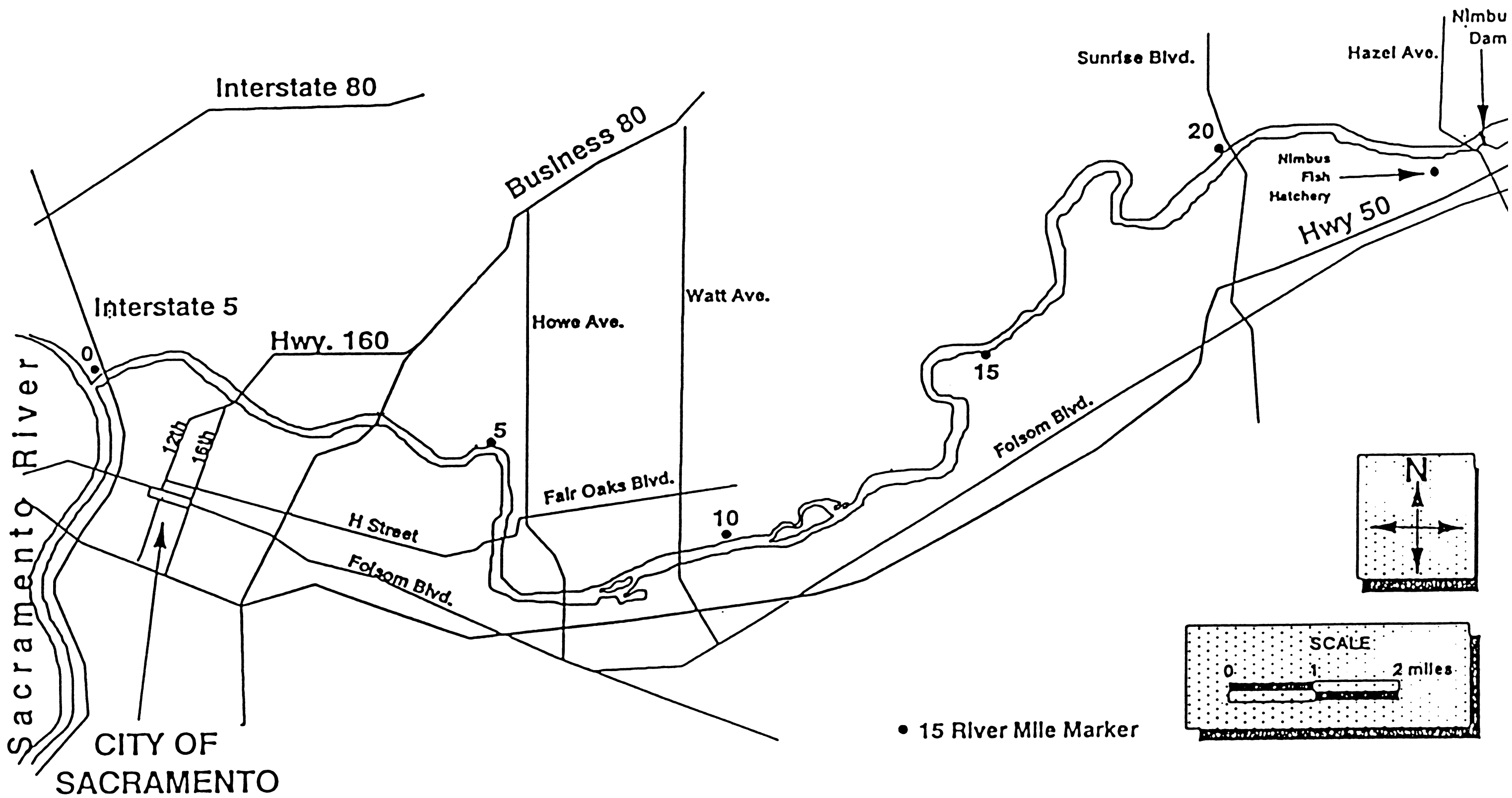

Figure 1 Map of the lower American River taken from Snider and others (1992)

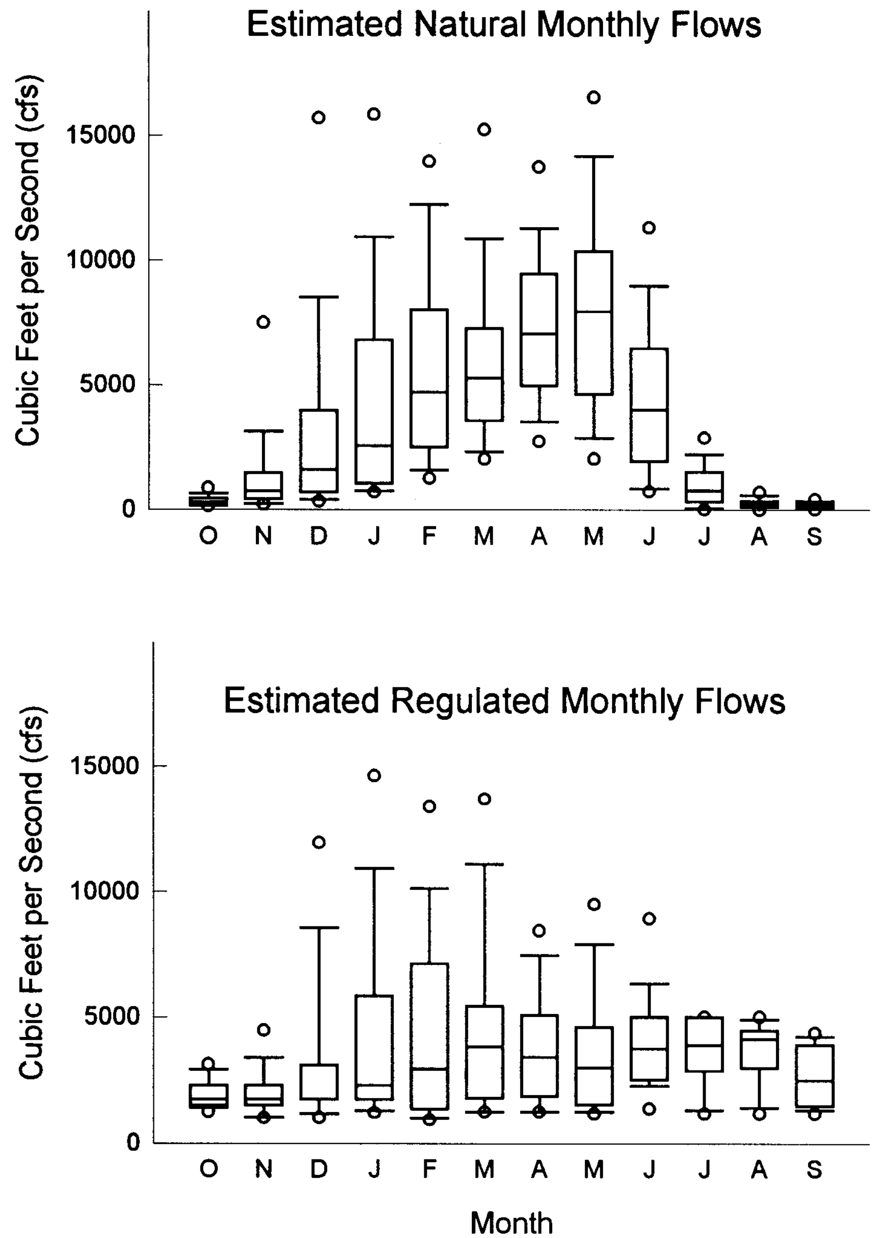

Figure 2 Comparison of the distributions of mean monthly flows in the lower American River for natural conditions (upper panel) and simulated 1993 conditions, assuming the same climatic conditions (lower panel). In the box plots for each month, the "box" covers the central 50% of the data, from the 25th to the 75th percentiles, the solid line across the box shows the median, and the dashed line shows the mean. The "whiskers" extend to the 10th and 90th percentiles, and the circles show the 5th and 95th percentiles. Note that the 1993 simulated flows do not reflect recent corrections to PROSIM, the operations model used for the simulations, or recent changes in CVP operations.

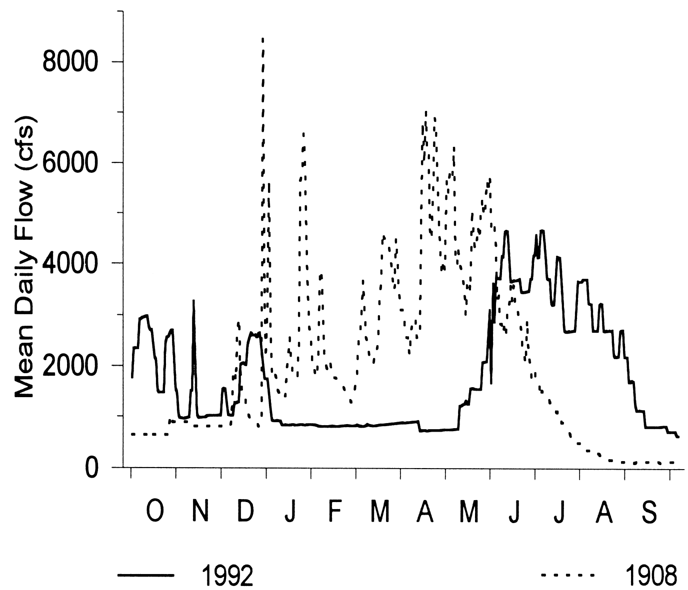

Figure 3 Comparison of flow in the American River in two dry years with approximately equal total discharge, illustrating the effects of regulation on the seasonality and variability of flow

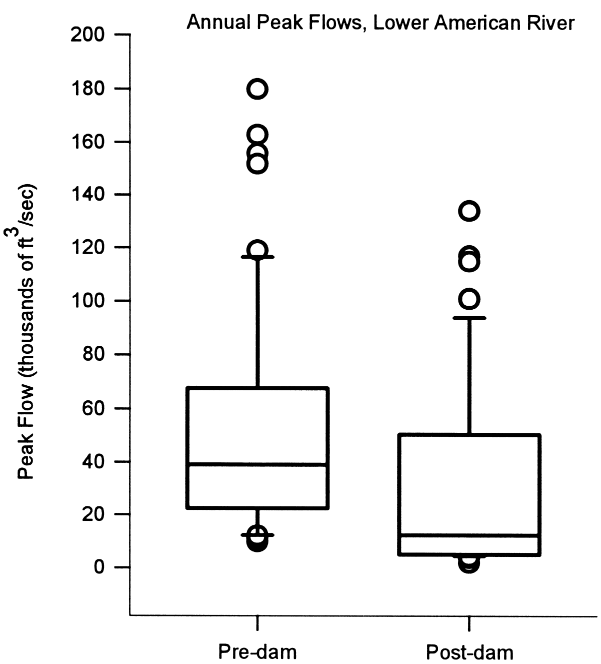

Figure 4 Comparison of the pre- and post-Folsom distributions of peak flows in the lower American River. Box plot conventions are as in Figure 2, except that circles show all values beyond the 5th and 95th percentiles. Data from USGS Fair Oaks gage.

The Hodge Decision

In 1970, the East Bay Municipal Utility District (EBMUD) negotiated a contract with the Bureau of Reclamation to take up to 150,000 acre-feet of water annually from the American River, through the Folsom-South Canal. The Environmental Defense Fund (EDF), Save the American River Association (SARA), and Sacramento County sued to block the contracts in 1972. Over the next 17 years, the California Department of Fish and Game (DFG) and the State Lands Commission (SLC) joined the litigation, the case went to the California Supreme Court twice, to the United States Supreme Court once, and to the State Water Resources Control Board (SWRCB) for a report of referee, before coming to trial in the Alameda County Superior Court of Judge Richard Hodge in 1989.

Simply put, the question was whether EBMUD could divert water through the Folsom-South Canal, or whether it must divert the water at some point farther downstream, so that the water could also serve instream uses. EBMUD wanted to divert through the canal because water quality decreases downstream. In its Report of Referee, the SWRCB recommended that EBMUD could divert through the canal, provided that certain instream flow standards were met. These standards were acceptable to EBMUD, but not to the plaintiffs. When the case went back to the Alameda County Superior Court, the substantive issues concerned the relation between water quality and public health on one hand and instream flow needs on the other.

Judge Hodge ruled that EBMUD could take water through the Folsom-South Canal, provided that enough water remained in the river to protect public trust resources. Based on the evidence in the record, Judge Hodge determined that "enough" meant: October 16 through February, 2,000 cfs; March through June, 3,000 cfs; July through October 15, 1,750 cfs. These flow standards, which apply to the whole 23-mile reach from Nimbus Dam to the Sacramento River confluence, are to remain in effect unless evidence is developed that justifies changes. The conditions apply only to diversions by EBMUD or by other parties to the litigation. Because the Bureau of Reclamation was not a party, the standards do not control the Bureau's operation of Folsom.

Judge Hodge emphasized that the evidence presented was inadequate to support a final determination of the flows necessary to protect public trust resources, however, so he retained jurisdiction, ordered the parties to cooperate in scientific studies to reduce the uncertainty regarding the necessary flows, and appointed the author as special master to supervise the continuing jurisdiction (Hodge 1990):

Perhaps the most salient aspect of the fishery/hydrology testimony consists of its large area or remaining uncertainty. …The task for this court is to recognize the fundamental inadequacy of existing studies as they relate to the American River, to extract from the 'consensus' and from the testimony those factors which can provide a guide for protecting fishery values, and significantly, to retain jurisdiction until the scientific community can provide definitive answers. (p 88, 95).

By emphasizing scientific uncertainty and framing a course of action that protects public trust resources while taking uncertainty into account, the Hodge Decision provides a good example of adaptive management (Castleberry and others 1996; Williams 1998).

Instream Flow Standards

In 1958, the SWRCB issued Decision 893, which granted the Bureau of Reclamation a permit for Folsom Dam, and set very low instream standards for the lower American River: 500 cfs from mid-September through October and 250 cfs otherwise. These remain the nominal state standards. The SWRCB set higher standards in Decision 1400, regarding Auburn Dam (for fish, 1,250 cfs from mid-September through June, 800 cfs otherwise), but since Auburn has not been constructed, these have not been binding. Nevertheless, the Bureau typically managed the lower American River to meet an approximation of the D-1400 standards called the "modified" D-1400 standards. [Why the SWRCB has never made the D-1400 standard applicable to Folsom is a fair question, but it has not. And as noted above, the Hodge standards only apply to diversions by the parties.] Since late 1997 the Bureau has operated Folsom with flow objectives set by the Anadromous Fish Restoration Plan (AFRP) (Table 1), developed under the Central Valley Project Improvement Act (CVPIA) which became law in 1992. Besides operating Folsom to meet the AFRP flows, the Bureau now meets regularly with the resource agencies and other interested parties to review details of dam operations.

Table 1 AFRP flow objectives for the lower American River

Month	*Wet*	*Above and below normal*	*Dry and critically dry*	*Critical relaxation*
October	2,500	2,000	1,750	800
November to February	2,500	2,000	1,750	1,200
March to May	4,500	3,000	2,000	1,500
June	4,500	3,000	2,000	500
July	2,500	2,000	1,500	500
August	2,500	2,000	1,000	500
September	2,500	1,500	500	500

As the AFRP flow objectives suggest, the amount of water available in the American River is limited in many years, and allocation of water to instream flows involves trade-offs among seasons, life-stages, and species, especially chinook and steelhead. Hence there is a need to understand the expected benefits of different seasonal flow regimes, typically in the face of uncertainty about future inflows to Folsom Reservoir and so about the amount of water that will be available to allocate in subsequent seasons. For example, a decision about how much water to allocate for spawning flows requires more than an understanding of the relation between flow and spawning habitat; it also requires an understanding of the importance of flows for juveniles and of the probability that water will be available for rearing flows, which will depend on post-spawning weather. One recent attempt to address this problem depended on the subjective assessment of biologists and did not spell out the rationale for the recommended allocation rules (Bratovich and others 1995), making it impossible to test the assumptions underlying the rules and to revise them in light of new information. A more transparent framework based on explicit assumptions and hypotheses is needed for guiding allocation decisions. Decision analysis (Peterman and Anderson 1999) seems well suited for this purpose.

Salmon in the American River

The EDF vs. EBMUD "Consensus"

With the agreement of the parties, Judge Hodge had the fish experts for both sides in the trial meet in closed session, without attorneys, to see how much agreement they could reach among themselves. The result was a "Report on Agreements and Recommendations," referred to elsewhere in the decision as the "consensus," that provides a useful summary of the understanding of chinook salmon at the time.

Life History Periodicities

1. Adult fall run chinook salmon are known to enter the lower American River from approximately mid-September through January. There is a high year-to-year variability; however, the bulk of the migration occurs from approximately mid-October through December.

2. Adult chinook salmon are known to spawn in the lower American River from approximately mid-October through early February. There is high variability from year to year; however, the bulk of the spawning occurs from approximately mid-October through December.

3. Chinook salmon egg and alevin incubation is known to occur in the lower American River from approximately mid-October through April. There is high variability from year to year; however, most incubation occurs from approximately mid-October through February.

4. Chinook salmon fry emergence is known to occur in the lower American River from January through mid-April.

5. Chinook salmon young-of-the-year juvenile rearing is known to occur in the lower American River from January to approximately mid-July. There is high year-to-year variability; however, the bulk of the rearing occurs from February through May. During March 1989, a few yearling chinook salmon were collected in the lower American River, suggesting that some fish may rear year round.

Water Temperature

1. Based on the scientific literature, the range of water temperatures for highest survival of incubating chinook salmon eggs appears to be between 43 °F to 58 °F. Prolonged (that is, more than a few days) exposure of eggs to temperatures in excess of 58 °F results in high egg mortality. 62 °F should be avoided.

2. Any definition of an "optimum" water temperature or temperature range for juvenile chinook salmon should include a synthesis of information on the effects of temperature on (a) growth rates; (b) effects on and availability of food supply ration; (c) predation; (d) disease; (e) stimulation of emigration; (f) physiological transformation to endure seawater; and (g) acclimation to the waters of the Lower Sacramento River and Delta when warmer than the American River.

Consensus on the optimum temperature range could not be reached.

Flow Needs

1. SWRCB Decisions 893 and 1400 are inadequate to meet the chinook salmon spawning habitat management objective for the lower American River.

2. The group could not reach consensus on the optimum spawning flow (or range of flows) needed to meet the fishery habitat management objective for chinook salmon in the lower American River.

3. Consensus could not be reached on the levels of flow required to provide optimum rearing habitat needed for juvenile chinook salmon in the lower American River.

4. SWRCB Decision 893 does not provide adequate rearing flows to meet the fish habitat management objective of maximizing the in-river production of juvenile chinook in the lower American River.

Recent Escapement Data

Both naturally and hatchery produced chinook salmon now spawn in the lower American River. Escapement has been estimated for several decades (Figure 5) and is highly variable but averages around 30,000. The data need to be regarded with considerable caution (Williams 1995). Returns to the hatchery are counts, but escapement to the river is estimated from mark-recapture methods applied to carcasses. Rich (1985) detailed problems with early estimates, and even recent estimates based on intensive carcass surveys involve great uncertainty, arising both from sampling errors and from the methods used to make estimates from the observations. Since 1976, DFG has used a modification of the Schaefer method, a multi-sample version of the Peterson method, but recently has reported estimates based on the Jolly-Seber method (e.g., Snider and Reavis 1996). For 1995, for example, the Schaefer estimate of escapement to the river was 70,096, while the Jolly-Seber estimate was 42,973, or 61% of the Schaefer estimate. The methods have been evaluated on Bogus Creek, a small tributary of the Klamath River for which weir counts are also available (Sykes and Botsford 1986; Boydston 1994; Law 1994), but conditions are less favorable for mark-recapture studies on larger rivers where a smaller percentage of marked fish are recaptured (Boydston 1994). Mark-recapture methods are also used to estimate escapement on other large rivers in the Central Valley and an evaluation by a competent statistician of their use on such rivers is sorely needed, as is a method for developing confidence intervals for the estimates.

The percentage of hatchery-produced fish among spawners in the American River is unknown, but presumably is large. Dettman and Kelley (1986) tried to evaluate this percentage; but as demonstrated by Hankin (1988) their calculations used so many approximate numbers and assumptions that it is hard to assign meaning to their results. Cramer (1992) applied a more sophisticated approach to the same question but the basic problem arises from the nature of the available data rather than the particular approach taken, so his estimates are also highly uncertain. For example, the results would depend on whether one used Schaefer or Jolly-Seber estimates of escapement. Cramer (1992, p 99) acknowledges this uncertainty:

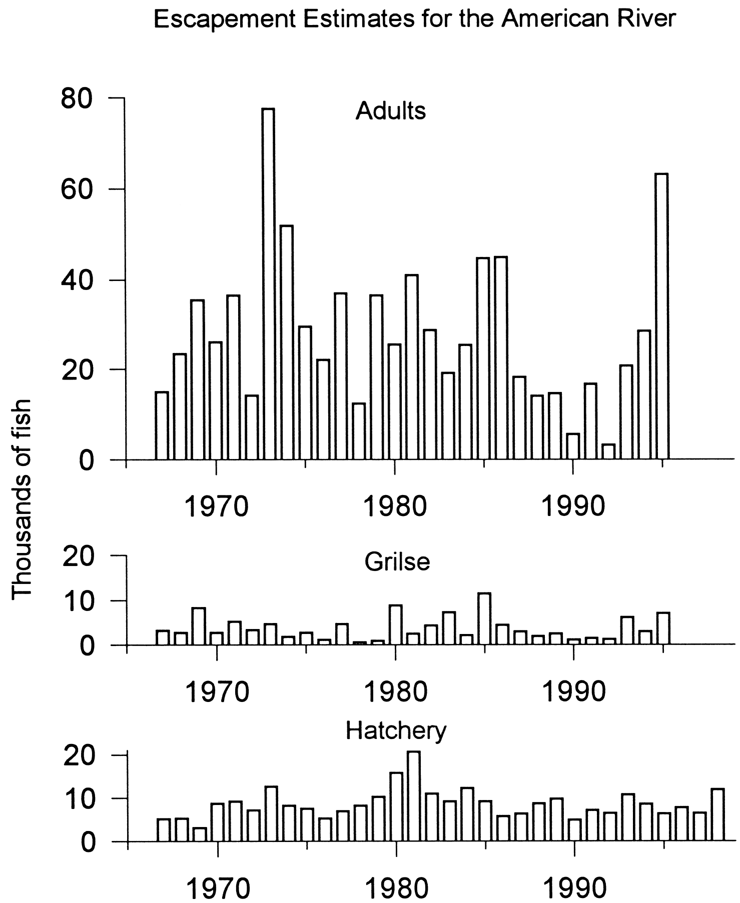

Figure 5 Escapement estimates for the lower American River from carcass surveys for "adults" and grilse, and counts at Nimbus Hatchery. Note that criteria for distinguishing grilse are not consistent and are of uncertain biological meaning (Williams 1995, p 100).

> *I conclude from these comparisons that Dettman and Kelley's predications of the escapement of hatchery fish are too high. However, evidence cited in this chapter also indicates escapement of hatchery fish predicted by run reconstruction may be too low. Clearly, hatchery and natural contributions cannot be estimated with confidence until a well designed marking program of hatchery fish and wild fish, extended to all release types, is initiated and systematic sampling is begun for all major spawning areas and river fisheries.*

It is remarkable that almost eight years after passage of the CVPIA, which calls for doubling the number of naturally produced anadromous fishes, the proportion of the salmon spawning in Central Valley rivers that are of hatchery origin remains unknown.

Hatchery Production

About half the chinook spawning habitat below Folsom was inundated by Nimbus Dam and Lake Natoma (USFWS and DFG 1953). Nimbus Hatchery was constructed to mitigate only for the spawning and rearing habitat inundated by Nimbus Dam and Lake Natoma, since passage of salmon was largely blocked by the Old Folsom Dam [loss of the opportunity to build a successful ladder over that dam apparently was not considered]. Nimbus now operates with a target of producing 4 million smolts for release in the estuary from May to July, for which it may collect up to 8 million eggs, distributed over the spawning season. The target size at release is 60 per pound (7.6 grams) or larger. Nimbus hatchery production of fingerlings for recent years is given in Table 2[2].

In the past, Nimbus Hatchery typically hatched more fry than it could rear, and over the period 1955–1967 released an average of almost 14 million fry annually. Emphasis then shifted toward producing larger juveniles, and average production of fry dropped to 3 million annually for 1968–1984 (Dettman and Kelley 1987). After 1990, fry were released into the Sacramento River at Garcia Bend so not to interfere with studies in the American River. But this too has recently ended; beginning with brood year 1998, DFG policy has been to rear to smolts all eggs hatched, and to limit egg take to meet smolt production goals (Bruce Barngrover, DFG, 1999, personal communication).

Table 2 Production of chinook salmon by Nimbus Hatchery [a]

Brood year	*Fingerlings (≤ 7.6 grams, 90 mm)*	*Advanced fingerlings (> 7.6 grams)*
1985	5,241,020	3,139,240
1986	3,167,680	3,040,375
1987	1,257,770	4,278,750
1988	---	3,210,570
1989	7,437,911	4,092,000
1990	6,069,505	1,244,800
1991	9,218,652	1,734,200
1992	7,930,390	1,988,700
1993	7,940,000	1,183,900
1994	8,103,143	1,378,100

[a] Data from California Department of Fish and Game.

2. Data for earlier years are available in Dettman and Kelley (1986) or Cramer (1992), but are given in different size categories.

The biological consequences of hatchery production for chinook salmon in the American River are unclear, but merit more attention. (General concerns about the effects of hatchery production on salmon populations are reviewed in NRC [1996]; see also Hilborn [1999]). Recent studies in New Zealand have shown that hatchery fish can replace naturally produced chinook rather than supplement them (Unwin 1997), probably because of density-dependent mortality in early ocean life, and some biologists believe that the same is true here (Walters 1997; Hilborn 1999). Hatchery production can lead to changes in life history patterns (Unwin and Glova 1977). Unwin (1997) also found that the size-adjusted mortality rates of hatchery fish were much higher than naturally produced fish, even though many of the naturally produced fish were progeny of hatchery fish.

One possible consequence of hatchery production on American River chinook may be decreased fecundity (discussed in the following paragraphs). Another possible indication of detrimental biological effects of hatchery production involves the composition of otoliths. The calcium carbonate in salmonid otoliths normally occurs as aragonite, which is opaque, and all the juvenile salmon sampled from the American River by Castleberry and others (1991, 1993) had opaque otoliths. However, some transparent otoliths were noted in juveniles from Nimbus Hatchery during supplemental work on marking otoliths with oxytetracyline (D. Castleberry, USFWS, 1995, personal communication). In transparent otoliths, the calcium carbonate occurs as vaterite. Such otoliths have been observed in high frequencies in some hatcheries in British Columbia, and there is concern that vaterite otoliths reflect inbreeding. Additionally, in British Columbia some of vaterite otoliths are also misshapen, raising concerns about how well they function (Blair Hotlby, June 1992, personal communication).

Life History Patterns

Chinook salmon remaining in the American River are fall-run, ocean-type fish that migrate to the ocean within a few months of emerging. Fish of this life history pattern simply avoid the period when flows in Central Valley rivers are naturally low and warm. Although late summer flows in the lower American River are now much higher and somewhat cooler than in natural conditions (Williams 1995), conditions are still unsuitable for chinook rearing, and water temperature in the lower Sacramento River often becomes very warm for juvenile chinook in late May or early June. Juveniles that fail to emigrate before the Sacramento River gets too warm probably have little chance of survival.

Spawning

Adult salmon appear in the American River in July, but many local biologists and fishermen believe that these early arrivals are hatchery strays from the Feather River, where spawning begins earlier than in the American. Spawning in the American River begins in October or November, typically when the water cools to about 15.5 °C (60 °F), approximately the temperature at which egg survival is possible. Facilities for controlling the temperature of releases from Folsom Dam were improved in 1996, and salmon responded by starting to spawn about two weeks sooner than had been common in the past. In 1997 water remained above 15.5°C until mid-November, however, and spawning was similarly delayed (Kris Vyverberg, DFG, 1999, personal communication). This variation in timing supports the hypothesis that water temperature rather than some correlated variable such as day length mainly controls the initiation of spawning.

Chinook redds normally show up well in aerial photographs of the American River because the water is usually clear and undisturbed gravel has a darkening surface layer of algae. Aerial photographs have been taken at intervals throughout the spawning season since 1991, producing a good record of where and when salmon spawn, at least for the early part of the season (Figure 6). Later, the popular areas are dug up so thoroughly that it is no longer possible to see individual redds or estimate the numbers of spawning fish from the photographs (Snider and Vyverberg 1996). Nevertheless, the approach should allow development of an empirical relation between flow and spawning habitat. The aerial photography also shows that spawning sites are related to geomorphic features in the channel that promote subsurface flow, as reported for the Columbia River by Geist and Dauble (1998).

Snider and Vyverberg (1996) report data on redd size, which is substantially smaller when measured on the ground (average 62 ft^2) than when measured from aerial photographs (average 196 ft^2). They discuss possible reasons for the difference, but until the matter is further clarified estimates of superimposition based on aerial photography should be viewed with some caution. Nevertheless, superimposition data (Table 3) indicate that density-dependent mortality can occur during spawning, and tends to vary inversely with flow (Snider and Vyverberg 1996).

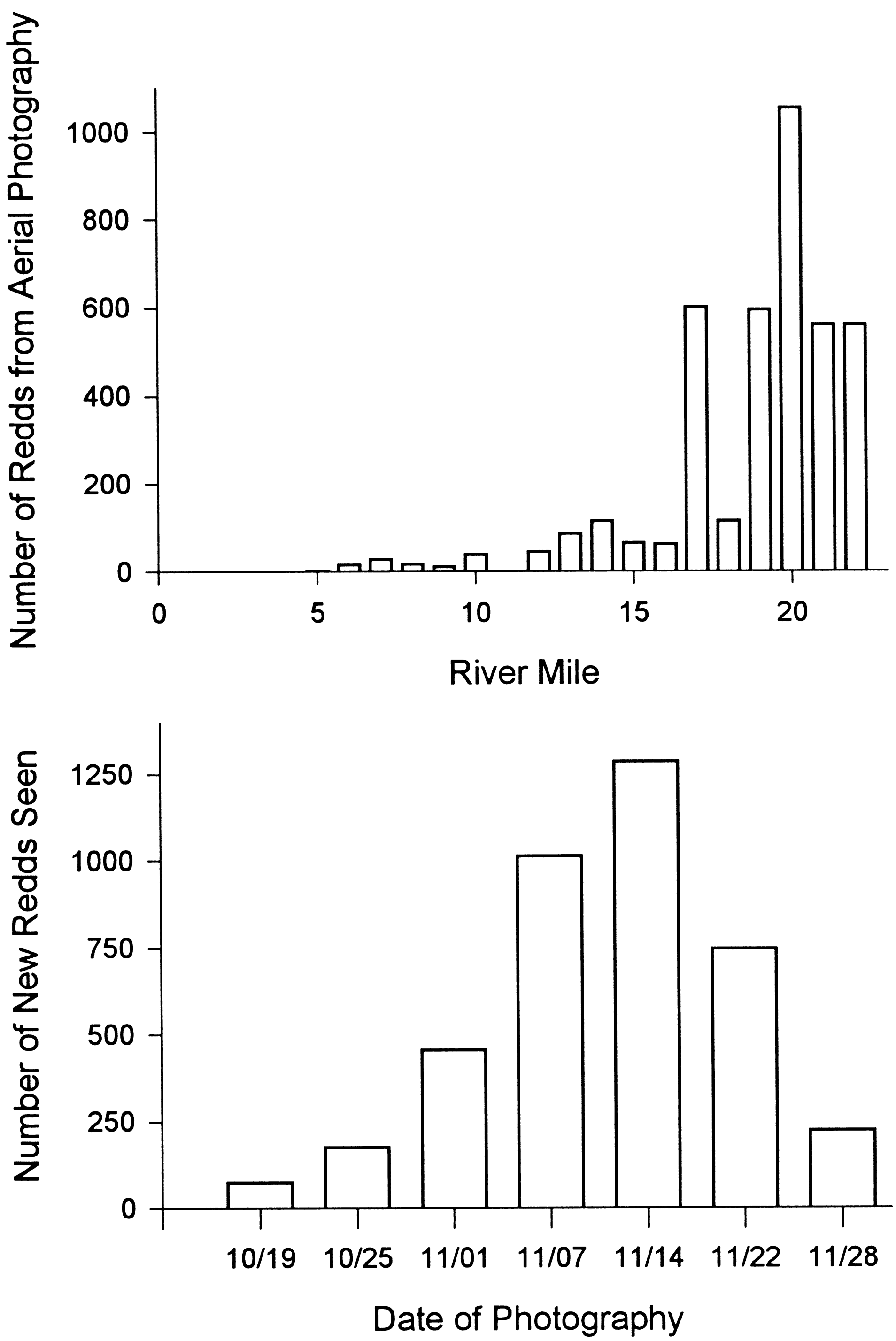

Figure 6 The spatial and temporal distribution of spawning in the lower American River in 1995. Data from Snider and Vyverberg (1996).

Table 3 Observed superimposition of redds, 1991–1995 [a]

Year	Percent of redds superimposed	Number of redds affected	Escapement estimate	Average flow (cfs)
1991	8	137	18,145	1,200
1992	42	474	4,472	500
1993	19	1,156	20,820 – 26,786	1,750
1994	17	450	26,881 – 31,333	1,500
1995	1.3	51	42,924 – 69,892	2,625

[a] Data from Snider and Vyverberg (1996); where two escapement estimate are given the first is a Jolly-Seber estimate, others are Schaefer estimates. There is a slight discrepancy between population estimates for 1995 given here for 1995 and those in Snider and Reavis (1996).

Spawning gravels in the lower American River are well described by Vyverberg and others (1997), who used both bulk sampling and pebble counts to estimate gravel size distributions and characterized intragravel conditions in terms of dissolved oxygen, water temperature, and hydraulic permeability. Gravel conditions are generally good but there are subsurface layers of coarse gravels that inhibit redd construction in some areas. These coarse gravels probably are deposits of stones too large for salmon to move during spawning in previous years. Vyverberg and others (1997) proposed that substrate conditions in these areas probably could be improved by "ripping" the gravel to break up the subsurface layers and to reduce compaction, which was done in late summer 1999 with an experimental design that includes pre- and post-project data collection in both treatment and control areas. Gravel was also added to the river as part of this project, funded through the CVPIA, despite a finding by Vyverberg and others (1997) that addition of gravel may not be necessary.

Vyverberg and others (1997) also showed that there is a good relation between the areas where salmon spawn and the permeability of the gravel and the estimated rate of subsurface flow, but the traditional microhabitat variables of depth and velocity do not distinguish areas that are used from those that are not (Figures 7 and 8). This should not be a surprise. According to Healey's review of chinook salmon life history (Healey 1991):

> *Provided the condition of good subgravel flow is met, chinook apparently will spawn in water that is shallow or deep, slow or fast, and where the gravel is coarse or fine.*

Nevertheless, the data provide further evidence that weighted usable area (WUA), the statistic calculated by the Physical Habitat Simulation Model (PHABSIM), is not a good measure of chinook spawning habitat because it ignores subsurface flow, the factor that seems most important to the fish. Gallagher and Gard (1999) reported statistically significant relations between WUA and the number of redds in PHABSIM "cells" in the American and Merced rivers, but the relations are not strong (r^2 = 0.40 and 0.38 respectively) and the study was conducted in areas that salmon were known to favor for spawning, and so presumably had good subsurface flow. Whether there is much of a relation between WUA and number of redds in randomly chosen areas of the river is unknown but doubtful in light of the results in Vyverberg and others (1997) on the American River and Geist and Dauble (1998) on the Columbia River.

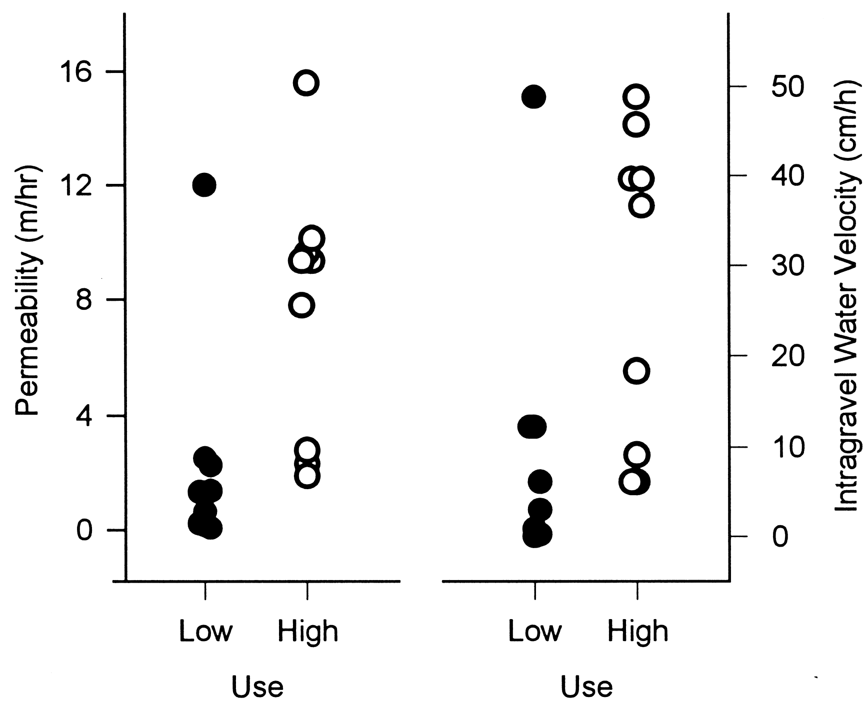

Figure 7 Permeability and estimated intragravel water velocity at ten sites that are selected (open circles) or avoided (closed circles) for spawning by chinook salmon. Data from Vyverberg and others 1997).

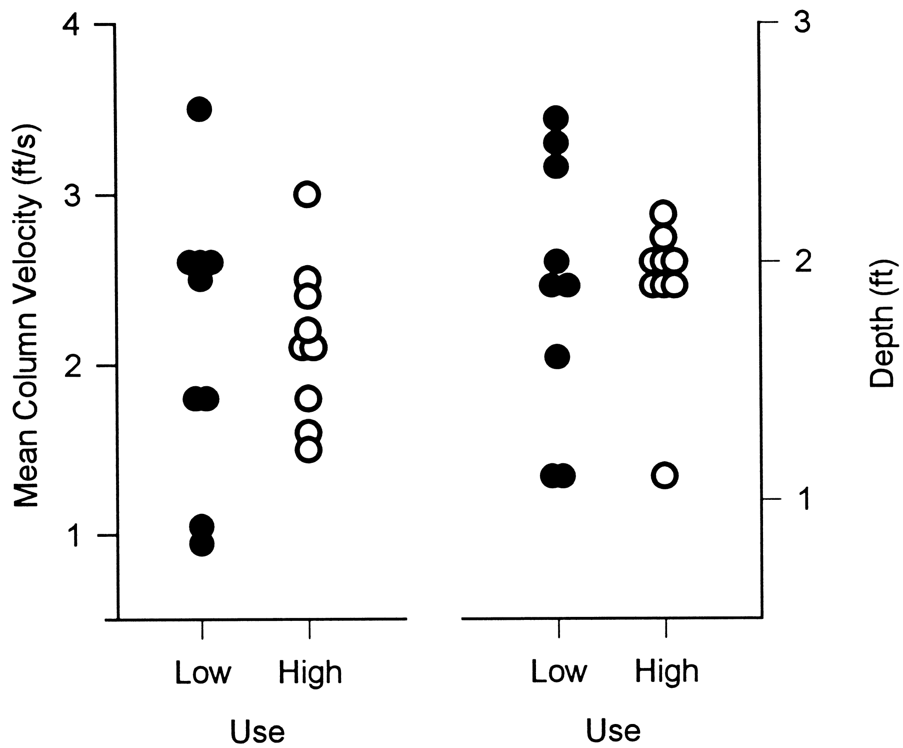

Figure 8 Mean column water velocity and depth at ten sites that are selected (open circles) or avoided (close closed circles) for spawning by chinook salmon. Data from Vyverberg and others (1997).

Pre-Spawning Mortality

The percentage of females that spawn completely before dying varies from year to year, ranging from 94% in 1993 to 68% in 1995 in samples of several hundred fish examined during DFG escapement surveys (Table 4) (Snider and others 1993, 1995; Snider and Bandner 1996; Snider and Reavis 1996). The reasons for the variation are not obvious; high proportions of unspawned carcasses were found in 1995 well into the spawning season, when water temperature should not have been a problem, and effective density as measured by redd superimposition was low. These data also illustrate the danger of drawing quick conclusions from short-term studies.

Table 4 Observed pre-spawning mortality (percent) from 1992 to 1995[a]

	1992	*1993*	*1994*	*1995*
Fully spawned	92%	94%	74%	68%
Partially spawned	3%	3%	9%	13%
Unspawned	5%	3%	17%	19%

[a] Data from Snider and Reavis (1996).

Incubation

Incubation is relatively rapid for fall-run chinook salmon in Central Valley streams because the water is warm compared to more northerly streams; in the lower American River water temperature usually averages between 6 and 9 °C in January, the coldest month. There are no available data on mortality during incubation on the American River. Emergence traps deployed in 1996 and 1997 were destroyed by high flows. However, Vyverberg and others (1997) estimated mortality using published relations between survival and gravel size (Tappel and Bjornn 1983) and between survival and intragravel water velocity (Gangmark and Bakkala 1960). There was no clear relation between the two estimates, which varied from 66% to 100% at 18 sites based on gravel size, and from 54% to 79% based on intragravel water velocity, except that estimates based on gravel size were always higher. Intragravel water velocity is directly related to the supply of oxygen to the eggs and alevins and the removal of metabolic wastes and seems a sounder basis for estimating survival.

Emergence

The timing of emergence depends on the timing of spawning and on water temperature, which strongly affects the rate of development of eggs and alevins. Chinook fry have been captured as early as late November in recent DFG studies (Snider and others 1998), earlier than suggested by the EDF vs. EBMUD "consensus." This change may reflect new sampling methods (rotary screw traps), and perhaps the relatively warm water temperature in the fall and winter of 1995–1996. Fry usually begin to emerge in large numbers in January and continue to emerge until April, or even later in some years (Snider and Keenan 1994).

Juvenile Rearing

Although most juvenile chinook leave the American River shortly after emerging, some rear in the river for a few months before emigrating. Even of this group, however, most are gone by mid-May and relatively few remain in June based on both trap (Snider and Titus 1995; Snider and others 1997, 1998) and seine data (Brown and others 1992; Snider and McEwan 1993; Snider and Keenan 1994; Snider and Titus 1996). Snider and others (1998) note that juvenile chinook now emigrate earlier in the year than when the USFWS operated fyke traps on the river in 1945–1947 (USFWS and DFG 1953). Warmer water during the incubation period resulting from the thermal effects of Folsom Reservoir seems the most likely explanation for this change (Rob Titus, DFG, 1999, personal communication).

Jackson (1992) observed habitat use by juvenile chinook in late April or early May at two flows, 350 cfs in 1991 and 3,700 cfs in 1989. Although his efforts were hampered by poor visibility, he summarized his observations as follows (p 104–105):

> *Juvenile chinook salmon in the lower American River exhibited trends in habitat selection and behavior similar to what has been observed by other researchers in other rivers. Juvenile chinook salmon occurred in groups of two fish to schools of thousands and ranged from 50 to 120 mm (FL), but predominantly were 50 to 80 mm in length. Schools were always associated with cover which provided visual and/or velocity shelter, the latter was utilized most often. As the juvenile chinook salmon became larger (80 to 120 mm), a progression toward deeper and faster water was observed. The larger fish were either paired or more often alone utilizing large cobble/boulder substrate as velocity cover and would move quickly from their shelter to feed on drift organisms. Individual chinook salmon were aggressive and territorial.*
>
> *During the high flow period a considerable amount of terrestrial vegetation was submerged and utilized extensively by juvenile chinook salmon. Root wad/debris jams were limited in quantity in the upper two reaches of the lower American River. These were utilized extensively and provided a significant juvenile chinook salmon microhabitat niche. On all occasions where root wad/woody debris jams were available as a cover type, except [for one], large schools of juvenile chinook salmon were observed. No juvenile chinook salmon were observed at either flow utilizing the one area surveyed … with riprap. During high flow juvenile chinook salmon were observed utilizing eddies and small microniches within undulating sandy substrate.*

While in the river the juveniles feed mainly on drifting invertebrates. Chironomids (midges) are most frequently eaten, but the larger caddisflies and mayflies make up most of the diet by weight (Brown and others 1991; Merz 1993).

Castleberry and others (1991; 1993) evaluated the physiological condition of juvenile chinook in the lower American River in 1991 and 1992, years with moderately low flows and warm water in late winter and early spring[3]. They found that non-polar lipid percentages for juveniles increased with length and tended to decrease with distance downstream, averaging about 6% to 8% dry weight for 40 to 49 mm fish, and 10% to 14% dry weight for fish 60 to 69 mm. This is in the low range for hatchery fish, but there are few comparable data for wild fish. They found that activity levels for Na^+-K^+ ATPase, an enzyme found in special cells in the gills that remove excess sodium and chloride ions from the blood, were high compared to published values. These data indicate that conditions in the river in 1991 and 1992 did not hinder the development of sea-water tolerance by juvenile chinook.

Approximate ages were determined from otoliths (Castleberry and others 1991, 1993, 1994), and showed that juveniles were growing well, averaging about 0.38 mm per day at 50 mm fork length (Williams 1995; estimates given in Castleberry and others 1991, 1993 are incorrect). Data on length by month suggest that juvenile chinook grew more slowly in 1993, when flow was higher and temperature lower, but this remains to be confirmed by analysis of the otoliths of fish collected and archived in 1993. DFG has this work underway (Rob Titus, DFG, 1999, personal communication).

Emigration

It has long been known that some ocean-type juvenile chinook emigrate as fry, shortly after emerging from the gravel, while others rear in the river for a few months and emigrate as smolts or large parr (Healey 1991). Based on the poor survival of coded-wire tagged fry released in the Delta (USFWS 1983), many biologists have assumed that the parr or smolt emigrants account for most returning adults. For example, the following assertion in Kelley and others (1985) was unchallenged in the trial of EDF vs. EBMUD:

> *Many of the small salmon are either washed, or voluntarily move, down into the estuary soon after they emerge from the gravel of the river bottom. The survival of these fish is very small, and fish that remain in the river and grow to a larger size have a much better chance of becoming adults.*

Some biologists argued that fry emigrants have continued to produce good returns in wet years; however, and a different view was expressed in the past. In the SWRCB hearings on Folsom in 1957, George Warner, a DFG biologist, argued the importance of fry emigrants:

3. See Williams (1995) for detailed temperature and flow data for 1991–1993.

Small fingerlings which are flushed rapidly out of the river to the rich feeding grounds in the Delta and in the ocean have a good chance of survival. A speedy downstream migration at high flows cuts down the loses from predation and losses in irrigation diversions. In addition these fish grow faster than fish which spend considerable time in the river. This has been amply proved in fingerling marking experiments and scale studies[4].

Recent investigations by DFG using screw traps near Watt Avenue (Snider and Titus 1995; Snider and others 1997, 1998) show that the overwhelming majority of fry leave the spawning areas in the lower American River shortly after emerging, with emigration usually peaking in February. Comparison of the size distribution of fish collected in the screw traps with that of fish collected with seines near the upstream limit of spawning suggests that this behavior has a temporal component, such that early emerging fry tend to emigrate directly (almost all fish are <50 mm before April), but later emerging fry are more likely to rear for some period before emigrating (Figure 9).

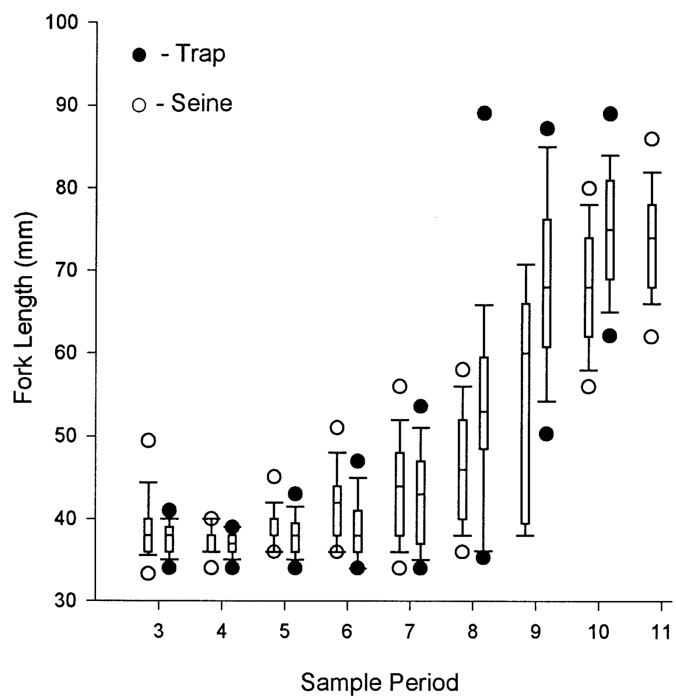

Figure 9 Size distributions of juvenile chinook salmon captured in the lower American River in screw traps (box plots with closed circles) and seines (plots with open circles) in 1995. Sample periods are two weeks: period 3 is 2/6–2/19, period 7 is 4/3–4/16, period 11 is 5/29–6/11. Box plot conventions are as in Figure 2. Data from DFG.

4. Unfortunately, he did not cite the studies; except for Clark's (1929) discussion of scale patters, I have not found any that fit his description.

There is controversy in the literature whether fry emigration is a forced, density-dependent behavior, or a volitional behavior (see Healey 1991 for a review). In the American River, the lack of larger juveniles in the seine samples early in the year when fish density is still low suggests early emigration is volitional, rather than a response to fish density or territorial behavior. Unpublished work relating length to otolith microstructure has developed no evidence that the fry captured in the traps are growing more slowly than others (Rob Titus, DFG, 1999, personal communication). More light could be shed on this issue by comparing the physiological condition of fry captured in the rotary screw trap with fry captured near the upper limit of spawning. Unfortunately, the traps were not effectively in service during the period that Castleberry and others (1991, 1993) were doing their work. Nevertheless, Castleberry and others (1993) found that ATPase activity increased downstream in fry <40 mm that were captured in seines, which is consistent with volitional emigration.

The large percentage of fry emigrants makes it seem likely that this is a viable life history pattern (Healey 1991). As noted by Snider and others (1998), the large proportion of fry emigrants emphasizes the importance of downstream rearing conditions for American River chinook salmon. Recent work by Sommer and others (2001) indicates that juvenile chinook in the Yolo Bypass grew more rapidly and had better survival to Chipps Island than fish in the Sacramento River, which supports the idea that natural floodplains along the lower Sacramento provided important habitat for juvenile chinook from the American River before the river was leveed.

Almost all juveniles leave the river before developing the full classic suite of smolt characteristics. DFG recently has classified juveniles collected in the screw traps as sac-fry, fry, parr, silvery parr, and smolts, (Snider and Titus 1995; Snider and others 1997, 1998) and reports less than 1% smolts and 74% or more fry or sac-fry (Table 5). Generally, however, the size distribution of fish collected in the screw trap is bimodal, with the great majority of the fish less than 45 or 50 mm, relatively few between 50 and 60 mm, and a second, much smaller group larger than 60 mm. The life stages. are not well correlated with length, however, in part because the length of parr and silvery parr tends to increase over the season (Snider and others 1998).

Table 5 Life stage statistics for emigrating chinook, 1994–1996 [a]

Life stage	*1994*	*1995*	*1996*
Yolk-sac fry	not distinguished	3.5%	22.6%
Fry	96.7%	70.5%	59.6%
Parr	1.6%	22.5%	17.4%
Silvery parr	1.4%	0.1%	4%
Smolt	0.3%	0.4%	0%

[a] Source: Data from Snider and others 1998.

Although the rotary screw trap data appear to provide good information on the timing of emigration and the nature of the emigrants, they do not provide good estimates of numbers of emigrants. Mark-recapture work by DFG shows that the capture efficiency of the rotary screw trap used by DFG is less than 1% (Snider and others 1998), and Roper (1995) argues that a capture efficiency of 10% or more is necessary for usefully accurate population estimates.

Age at Return

There are no data on the age or length at age of naturally produced chinook salmon returning to the American River, and very few data on hatchery fish, since fish from Nimbus are not normally coded-wire tagged. Recent information on length at age for Central Valley chinook generally is remarkably scarce, although it is commonly assumed that most spawners are three years old. Clark (1928) reported age data for salmon taken in the Delta gill net fishery in 1919 and 1921 (Figure 10), with ages determined by reading scales, showing more four- and five-year-old fish than three-year-old fish. However, chinook scales are hard to read (Godfrey and others 1968), and Clark may have overestimated ages (Frank Fisher, DFG, 1993, personal communication), but there is little doubt that the ocean troll fishery reduces that average age at return (Hankin and others 1994 and references therein). There is also good evidence that the size of returning adults has decreased from a comparison of the sizes reported by Clark and by a DFG survey in the American River (Figure 11). Hankin and others (1994) posit a genetically-influenced threshold size for maturation (see also Mangel 1994) that could be affected by inadvertent selection by the fishery and perhaps by hatchery practices.

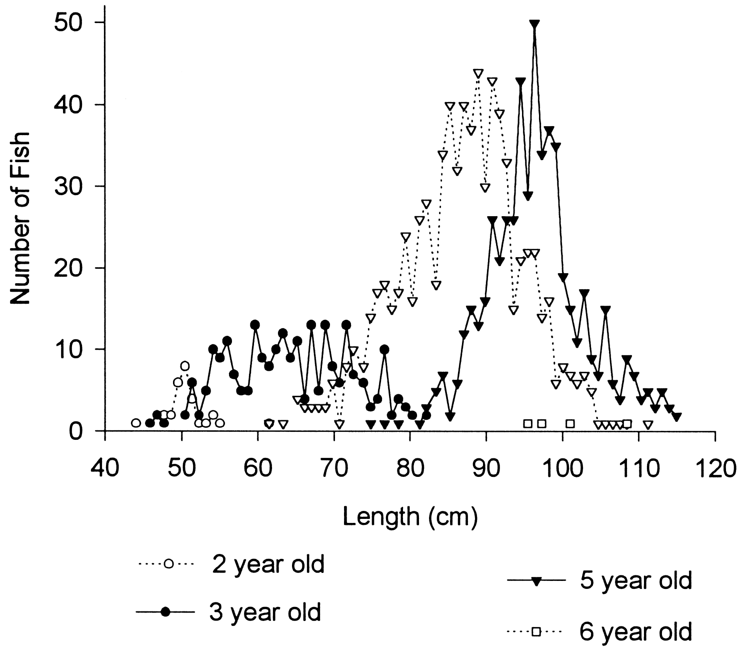

Figure 10 Ages of chinook salmon captured in the Sacramento gill net fishery in 1919 and 1921, estimated from scales. Data from Clark (1929).

Fecundity

There is substantial variation and a significantly declining trend in the average fecundity of females spawned at Nimbus Hatchery (Figure 12) from about 5,800 in the period 1955–1964 to about 5,100 for 1988–1997. Values for 1983 and 1984 stand out as low outliers, presumably reflecting poor ocean conditions associated with El Niño conditions. Unfortunately, the data were taken as the total number of eggs divided by the number of females, and there is information on the variance in fecundity among females and on the relation between fecundity and length for only one year, 1997. Fecundity of 135 individuals in 1997 varied from about 3,100 to 7,800 eggs, with length accounting for just over half the variation when fitted by fecundity = 6.385 (fork length)$^{1.564}$ (DFG 1998)[5]. Accordingly, the decline in average fecundity could reflect either a decline in fecundity at length, a decline in average length, or both. Fecundity is a basic biological parameter that deserves more attention.

5. A decline in average length probably accounts for the difference between the fecundity reported for Sacramento River chinook by McGregor (1923), which is cited by Healey and Heard (1984) and Healey (1991), and the fecundity at Nimbus in the late 1950s; in any event the fish measured by McGregor were large.

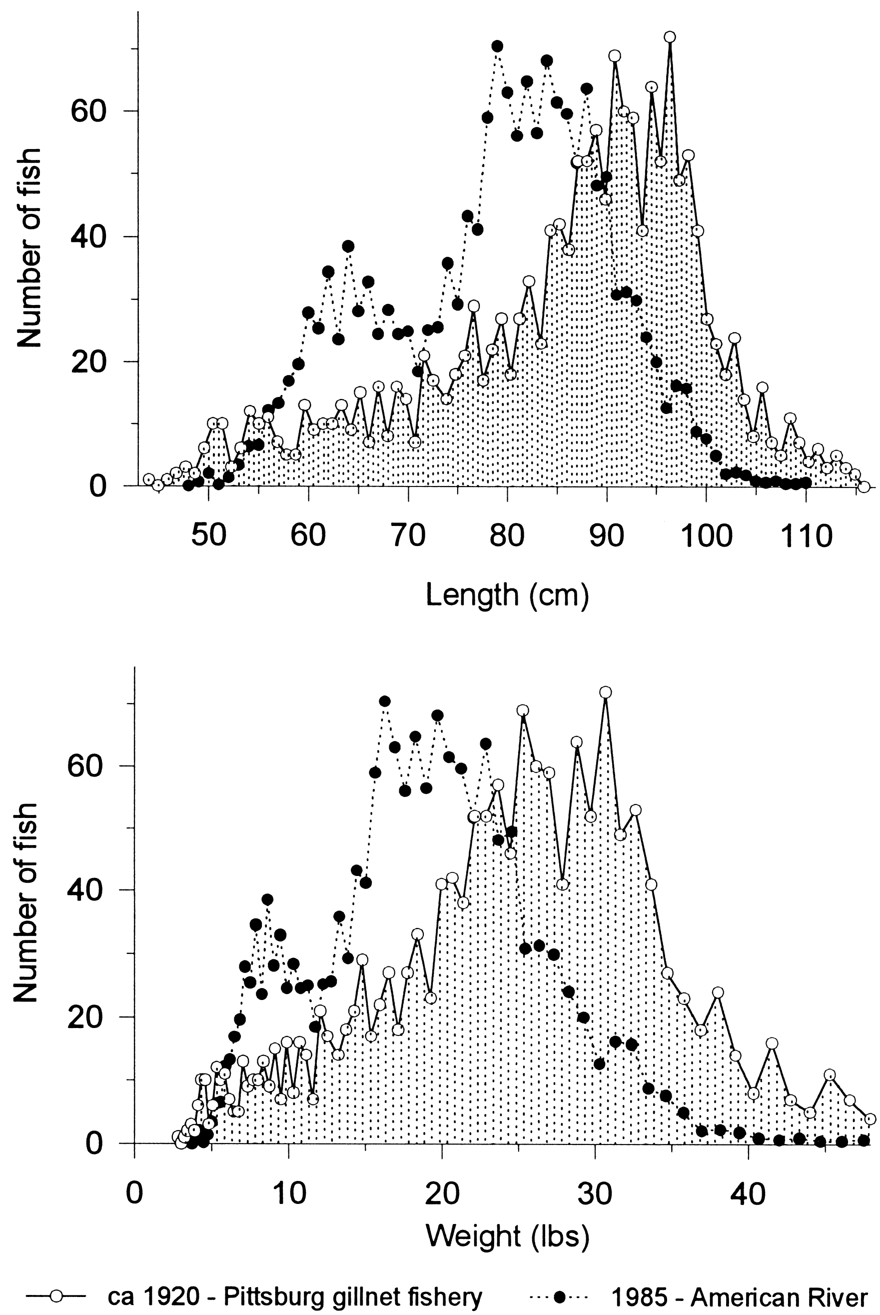

Figure 11 Length and weight distributions of chinook salmon captured in the Sacramento gill net fishery in 1919 and 1921, and from carcass surveys by DFG in 1985. Weights estimated with a length-weight relationship provided by Frank Fisher of DFG. Data from Clark (1929) and Fred Meyer of DFG.

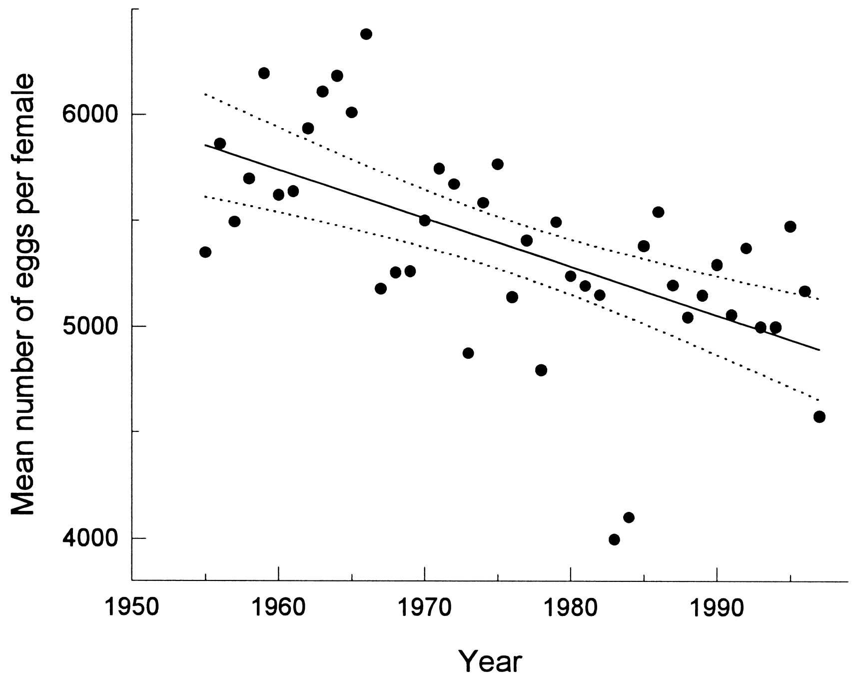

Figure 12 Declining trend in the average fecundity of chinook salmon at Nimbus Hatchery. Data from Fred Meyer and Terry West of DFG.

Salient Uncertainties and Research Needs

Several topics that deserve better understanding, such as fecundity and prespawning mortality, have been described above. Some additional topics follow.

Relation Between Flow and Rearing Habitat

The relation between flow and rearing habitat remains unclear. According to the consensus statement from a small workshop that discussed the American River at some length, "... currently no scientifically defensible method exists for defining the instream inflows needed to protect particular species of fish or ecosystems" (Castleberry and others 1996; Williams 1997). Methods such as PHABSIM suffer from measurement, statistical, and conceptual problems (Shirvell 1986, Shirvell 1994; Williams 1995, 1996; Campbell 1998; Bult and others 1999; Kondolf and others forthcoming). Simple empirical approaches that depend on measures such as smolts per spawner are confounded by measurement problems and density-dependent mortality (Williams 1999) and by the unknown percentage of hatchery fish. An adaptive approach that emphasizes measures of condition of juvenile fish, exemplified by the work of Castleberry and others (1991, 1993) on the American River, appears to be most promising, especially when linked to population-level responses by individual-based modeling (Osenberg and others 1994; Maltby 1999). More observations of habitat use like those of Jackson (1992) would be helpful, especially if they are directed toward developing a better understanding of the way juvenile chinook use habitat rather than "habitat suitability criteria" for PHABSIM studies. In any event, understanding the cause-and-effect relationships that underlie the responses of populations to habitat change seems crucial for effective management of habitats in regulated rivers (Jones and others 1996; Williams 1999).

The Importance of Fry Emigrants

The relative viability of fry that emigrate soon after emerging and fry that rear in the river for some time remains poorly known, as described above, but has important implications for management of the American River and investment in habitat restoration in the Delta. For example, there appears to be a trade-off between providing high flows for spawning in the fall and the risk of low carryover storage for flows the following spring, should the winter be dry. The optimal allocation of water to spawning probably depends on viability of fry emigrants, which in turn may depend upon habitat conditions in the lower Sacramento River and the Delta. DFG has work on otolith microstructure in progress that among other things aims to distinguish patterns associated with different juvenile life history patterns. If this can be done with even modest accuracy, then analysis of otoliths from adults should clarify the viability of fry emigrants. Monitoring the physiological condition of emigrating fry in the lower Sacramento River as well as in the American, and comparing these with fish remaining near upstream spawning areas in the American River, would be an alternative and complementary approach.

Density-dependent Mortality

Understanding the mechanisms of density-dependent mortality for chinook salmon in the American River should allow better management, even if measurement problems preclude quantifying the relationship accurately. As noted above, aerial surveys have provided some information on density-dependent mortality at spawning. Assuming that density-dependent mortality for juveniles works through mechanisms that also produce sub-lethal stress in juveniles, measures of condition such as lipids, otolith increment widths, or inter-renal distance (Castleberry and others 1991, 1993; Norris and others 1996) may be most useful. Otolith data on growth during early ocean life may provide evidence for density-dependence in that life stage, especially if combined with population data from streams where populations can be estimated more accurately than seems possible on the American River. Bold adaptive variation in hatchery production at a regional scale may be required to clarify this issue, however.

Temperature Tolerance of Juveniles

The temperature tolerance of juvenile chinook was much debated in the trial of EDF vs. EBMUD and despite recent progress remains unclear. Analyses of juvenile chinook and steelhead in the lower American River in 1991 and 1992 showed that they appeared to be growing well and be in good physiological condition, despite moderately low flows and warm water in late winter and early spring (Castleberry and others 1991, 1993; Williams 1995). Coded-wire-tagged fish in the Yolo Bypass grew more rapidly and showed better survival to Chipps Island than did paired releases of fish in the Sacramento River, where water temperature was lower (Sommer and others 2001). Juvenile chinook that move up relatively warm intermittent tributaries of the Sacramento River to rear grow rapidly (Moore 1997; Maslin and others 1997). Recent laboratory studies at the University of California at Davis (Marine 1999) showed that juvenile chinook from Coleman Hatchery grew as rapidly at 17 to 20 °C on full ration as they did at 13 to 16 °C. On the other hand, Clarke and Shelbourn (1985) described delayed mortality associated with scale loss in fish that were raised in freshwater at 16 or 17 °C, so freshwater growth and survival may not be the whole story. Paired coded-wire-tag releases like those of Sommer and others (2001), which will allow estimates of survival to catchable size from tag returns from the ocean fishery, could be especially useful in this regard. In any event, water temperature is an important predictor of the survival of coded-wire-tagged smolts, regardless of the statistical method used on the data (Ken Newman, University of Idaho, 1999, personal communication), while other variables such as flow seem important in some analyses but not in others. Assays for stress proteins (Iwama and others 1998) in fish collected at Chipps Island for the coded-wire tag studies could provide independent evidence of temperature stress. A literature review of the temperature tolerance of juvenile chinook that should clarify this issue is currently underway by Chris Myrick at the University of California at Davis.

The Importance of Hatchery Production

Intelligent management of chinook salmon in the American River depends on distinguishing fish of natural and hatchery origin. Hatchery fish can be marked easily and economically by manipulating water temperature in the trays in which larval fish (alevins) are reared. This creates visible bands of narrow and wide growth increments in otoliths (ear-stones) that mark fish as hatchery produced; the bands can even form bar-codes by which fish from different hatcheries or batches can be distinguished (Volk and others 1990, 1994). If all hatchery fish are marked, the proportion of naturally produced spawners could be estimated accurately from a relatively small sample, and the associated analysis of otoliths could also provide information on length at age of adults and perhaps information on year-to-year variation in ocean condition and on the life history patterns of fish that survive to spawn. A program for thermally marking the otoliths of hatchery fish is now being developed by DFG.

Quantitative Methods

Methods for analyzing biological data have developed rapidly in recent years (for example, Jongman and others 1987; Efron and Tibshirani 1991, 1993; Hilborn and Mangel 1997; Peterman and Anderson 1999). Unfortunately, these methods are unfamiliar to most Central Valley salmon biologists and even methods such as the bootstrap that are easy to implement are seldom used. Data analysis routinely should include the development and testing of models of the biological and sampling processes that generate the data (Elliott 1994; Hilborn and Mangel 1997). Besides guiding field studies to address the most relevant issues, this approach helps avoid the waste of resources on field studies that cannot generate useful information. The recent analyses of coded-wire tag data by Ken Newman and John Rice reveal a large gap between the quality of analysis that is possible and the quality that is typical in studies of salmon in the Central Valley, bearing out the observation of Effron and Tibshirani (1993) that "Statistics is a subject of amazingly many uses and surprisingly few effective practitioners."

Concluding Remarks

Much is known about chinook salmon in the American River and elsewhere, but much remains to be learned. Because of EDF vs. EMBUD, there have been many recent studies of chinook in the American River. In many respects, however, the American River is not a good study stream. Developing good population estimates for chinook salmon in the river does not seem to be practicable, especially for juveniles, mainly because the river is so big. The urban setting and heavy recreational use of the river create other problems, as does the heavy presence of hatchery fish. Efforts to understand density-dependent mortality or other aspects of chinook biology that require good population estimates probably should be focused on smaller streams such as Butte Creek or Clear Creek, or the Feather River side-channel where Castleberry and others (1994) confirmed that juvenile chinook form otolith increments daily. The low flow channel of the Feather River (see Sommer and others, Volume 1) probably is a better system than the American River for intensive studies on a larger scale because better experimental control of flows is possible.

Much could be gained by a regional perspective among salmon researchers that would allow a coordinated approach to addressing some questions and allow others to be addressed primarily in the parts of the system with the most favorable study conditions. Unfortunately, there is a tendency toward

Balkanization of salmon research in the Central Valley, with divisions among regions and agencies that discourages communication, let alone cooperation. Workshops such as the one giving rise to this publication are a step in the right direction, but much remains to be done to create an effective community of scientists in which the efforts and intelligence of those studying salmon in the Central Valley can realize their potential. (See also Kimmerer and others, this volume.)

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Juvenile Chinook Salmon Abundance, Distribution, and Survival in the Sacramento-San Joaquin Estuary

Patricia L. Brandes and Jeffrey S. McLain

Abstract

All four races of juvenile Central Valley chinook salmon migrate through and many rear in the Sacramento-San Joaquin Delta and Estuary. Delta residence and migration is considered important in determining adult production, as it is generally believed that density dependent effects are minimal after this life stage. Populations of winter run and spring run are presently listed as endangered and threatened species, while the remaining populations in the Central Valley are candidate species. Actions in the Delta to improve survival are likely important in the recovery of these depressed populations. The tidally influenced freshwater Delta also is an important area for water management in California, as it is where the Central Valley and State Water Project pump large volumes of water to southern California, the San Joaquin Valley and the Bay area. To document the effect of these various water management activities in the Delta on juvenile salmon, monitoring and special studies have been conducted since the early 1970s to the present. Changes in abundance in the Delta and estuary appear related to flow; high flows increase the use of the Delta and San Francisco Bay by fry. Relative survival of fry appears greater in the upper Sacramento River than in the Delta or bay, especially in the wetter years. Survival appears lower in the Central Delta relative to that in the North Delta in drier years for both fry and smolts. Fall-run smolt and late-fall-run yearling survival studies have found that diversion into the Central Delta via the Delta Cross Channel or Georgiana Slough reduces survival through the Delta. Experiments in the San Joaquin Delta have shown that survival appears greater for smolts that migrate down the mainstem San Joaquin River rather than through upper Old River. A temporary barrier in upper Old River was tested and found to improve survival for smolts originating in the San Joaquin basin. These specific experiments have identified management actions that could improve juvenile salmon survival through the Delta. In addition, indices of annual survival provide a way to compare survival through the Delta and could be used to assess restoration and management actions. This work demonstrates how long-term scientific studies can be applied to address management and restoration issues.

Introduction

The Sacramento-San Joaquin Estuary is one of the largest estuaries on the West Coast draining the majority of the Central Valley watershed of California. The Sacramento River from the north and San Joaquin River from the south converge in the freshwater, tidally influenced Delta (Figure 1). The Delta consists of nearly 1,200 km of freshwater channels, with most channels edged with riprap (Kjelson and others 1982). The bays downstream of the Delta are generally shallow, with salinities varying seasonally and affected by a combination of tidal flows and freshwater.

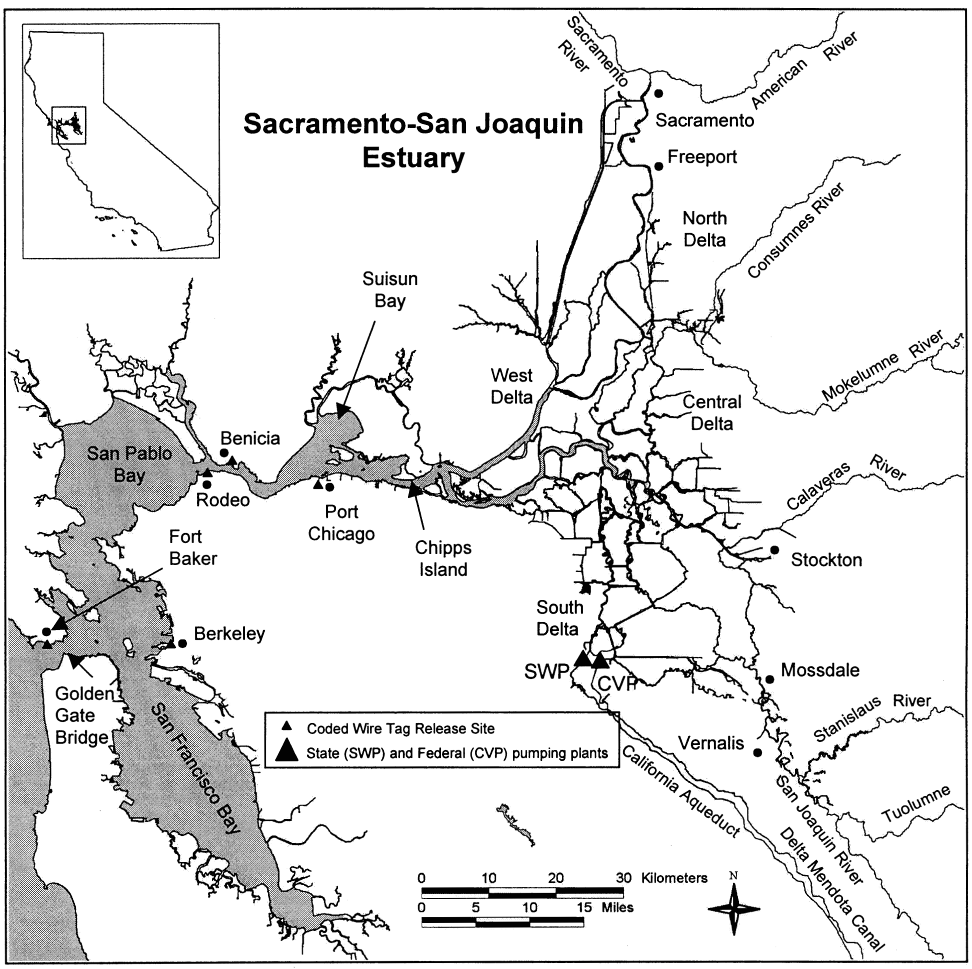


Figure 1 The Sacramento-San Joaquin Estuary, California

There are four races of chinook salmon in the Central Valley: fall, late-fall, spring, and winter. Races are based on their timing of return to freshwater for spawning (Fisher 1994). Historical documents indicate the start of the salmon fishery in California at about 1850 (USFWS 1995). Central Valley salmon continue to support valuable, economically important commercial and recreational fisheries.

During the past 30 years, overall escapement of Central Valley salmon has declined (Fisher 1994). Only the fall run continues to maintain stable spawning runs, likely because they are heavily supported by hatchery production (Fisher 1994). Winter-run chinook salmon were federally listed as threatened in 1990 and endangered in 1994 by the National Marine Fisheries Service. Spring run were recently listed as threatened in 1998 by the State of California. The remaining races and natural populations of chinook salmon in the Central Valley are presently considered candidate species under the Federal Endangered Species Act (NMFS 1999).

All of the various races of chinook salmon in the Central Valley use the Delta as a migration corridor to the ocean and many rear there before emigration. The survival of juvenile salmon through the Delta is considered critical to year class success, as density-dependent mortality after Delta residence is believed to be minimal (Junge 1970). Thus for any given set of ocean conditions, increasing the number of juveniles emigrating from the Delta will increase the production of adults. Actions in the Delta to improve survival are considered important in increasing the production of these Central Valley salmon populations.

In addition to the Delta being important to juvenile salmon, it is also critical to water management in California. Water resource project operations have altered the natural distribution, timing, and magnitude of flows in the Delta (Kjelson and others 1982). The State Water Project (SWP) and the Central Valley Project (CVP) use the Delta to move water from reservoirs in the North to the pumping plants located in the South Delta (Figure 1). The water is pumped (exported) into the State (California Aqueduct) and federal (Delta-Mendota Canal) aqueduct system for agriculture, municipal, and industrial use in the San Joaquin Valley, the Bay area, and southern California. Mean daily exports from the Delta have increased dramatically since the late 1950s and 1960s to peaks in the late 1980s (Figure 2). Due to population growth in California and other factors, there is a continued desire to increase exports further to meet the increased demands.

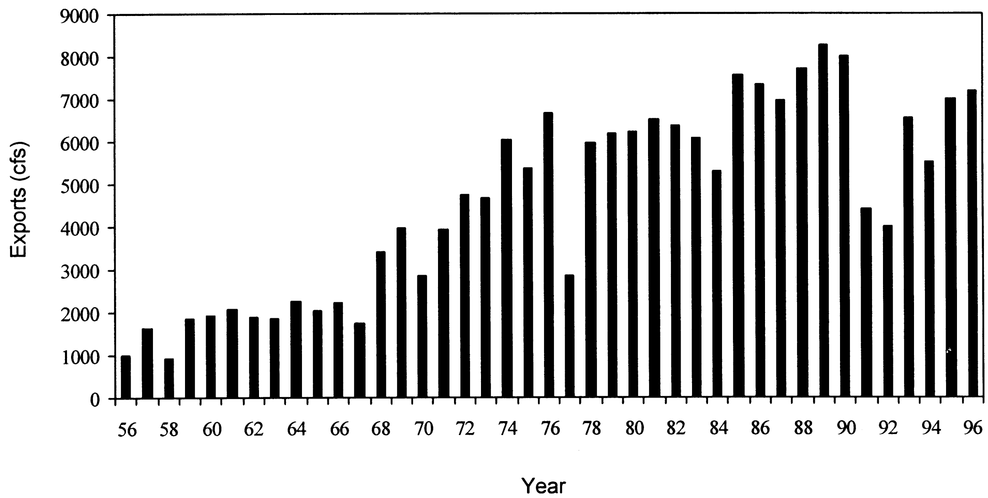


Figure 2 Mean daily combined (SWP + CVP) exports (cfs) between 1956 and 1996. Source: Department of Water Resources, DAYFLOW.

Although tidal fluctuations in the Delta are large relative to net downstream flows, an effect of the present export levels is that net flows in the South Delta often move upstream instead of downstream during periods of low Delta inflows. These net "reverse flows" occur when combined CVP and SWP export rates are higher than the net downstream flow in the San Joaquin River. The remaining water to meet the export needs originates from the Sacramento River. This process creates net flows in the South Delta that move upstream towards the pumping plants instead of downstream toward the ocean (Figure 3). For anadromous fish, such as chinook salmon, these reverse flows may cause confusion or divert them from their main migration routes to the sea. Delays in migration would expose juveniles to various mortality factors for a longer period of time and decrease their survival through the Delta.

Other habitat alterations by the two water projects are the construction of the Delta Cross Channel and the amount of water diverted from the mainstem San Joaquin River into upper Old River (Figure 3). The Delta Cross Channel, located in the North Delta, was built to increase the amount of water originating from the Sacramento River that flows into the Central Delta. The water in the Central Delta is then available by means of gravity to be pumped by the State Water and Central Valley projects located in the South Delta. In addition, the amount of water diverted into upper Old River from the San Joaquin River increases as project exports increase (Oltmann 1995). The CVP diverts water directly from Old River and the SWP diverts water from Clifton Court Forebay, and its intake is on Old River.

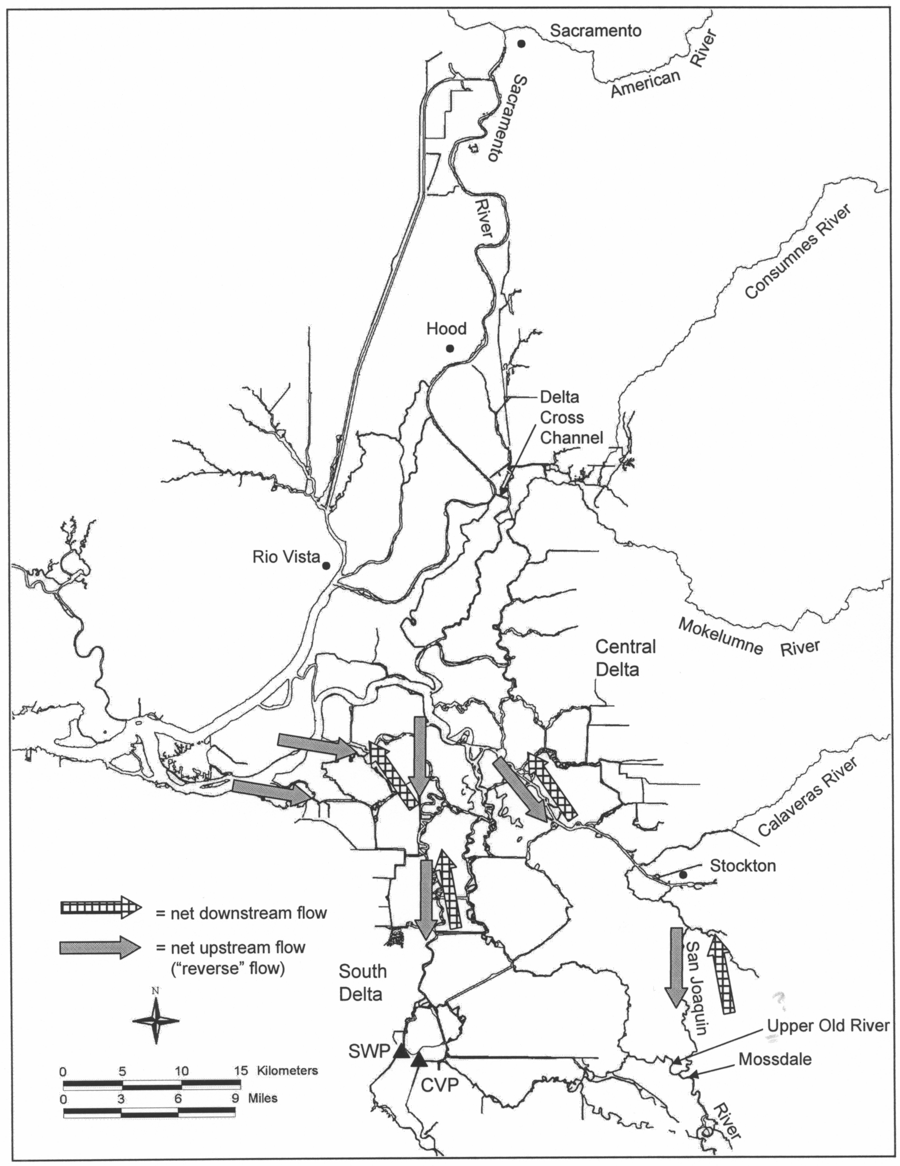


Figure 3 The Sacramento-San Joaquin Estuary, California. Arrows depict net downstream flow and "reverse flows."

The work presented in this report is derived from juvenile salmon monitoring and special studies conducted by the US Fish and Wildlife Service's (USFWS) Sacramento-San Joaquin Fishery Resource Office on behalf of the Interagency Ecological Program for the Sacramento-San Joaquin Delta (IEP). The IEP is a consortium of six federal and three State agencies charged with providing information on the factors that affect the ecological resources in the Sacramento-San Joaquin Estuary to allow more efficient management of the estuary. Agencies in the IEP, in addition to the USFWS, include the US Bureau of Reclamation, US Geological Survey, National Marine Fisheries Service (NMFS), US Army Corps of Engineers, US Environmental Protection Agency, California Department of Water Resources, California Department of Fish and Game, and California State Water Resources Control Board.

The IEP has been conducting juvenile salmon studies in the Delta since the early 1970s. The initial goals of the salmon studies were to define the impacts of water development on the estuarine salmon population and to document the water quality requirements (including flow standards) needed to both sustain and enhance salmon production (Kjelson and others 1981). The goals have been broadened since the program's inception and reflect an overall desire to gain information on what management actions can be taken to improve the survival of juvenile salmon rearing or migrating through the Delta.

The results of these studies have been shared in the past in various ways: workshops, *IEP Newsletter* articles, gray literature in the form of annual reports, testimony to the State Water Resources Control Board (USFWS 1987, 1992a) and peer-reviewed journal and symposium articles (Kjelson and others 1981, 1982; Kjelson and Brandes 1989). The purpose of this paper is to consolidate, update, and summarize the juvenile salmon information gained from the IEP salmon studies. Data from some of the studies are limited and do not provide statistically significant results. They are included to provide a more complete record of the results of the various studies. Many times inferences have been made based on limited data, but we acknowledge in that case there is a risk in drawing wrong conclusions. To lessen that risk, we have tried to draw on a variety of independent pieces of information to reach conclusions.

Specific studies were conducted on juvenile salmon abundance, distribution, and survival using beach seines, Kodiak and midwater trawls, and mark and recapture techniques. The beach seine and the trawls are size and habitat selective, with the beach seine targeting smaller fish (fry) near the shore and the midwater and Kodiak trawls generally capturing larger juveniles (smolts and yearlings) that migrate in the center of the channel. Mark and recapture experiments have been conducted with hatchery fry, smolts, and yearlings released in the upper Sacramento River, Delta, San Francisco Bay, and San Joaquin tributaries (Figures 1, 4 and 5) to estimate survival and examine the importance to survival of different environmental conditions (Kjelson and Brandes 1989).

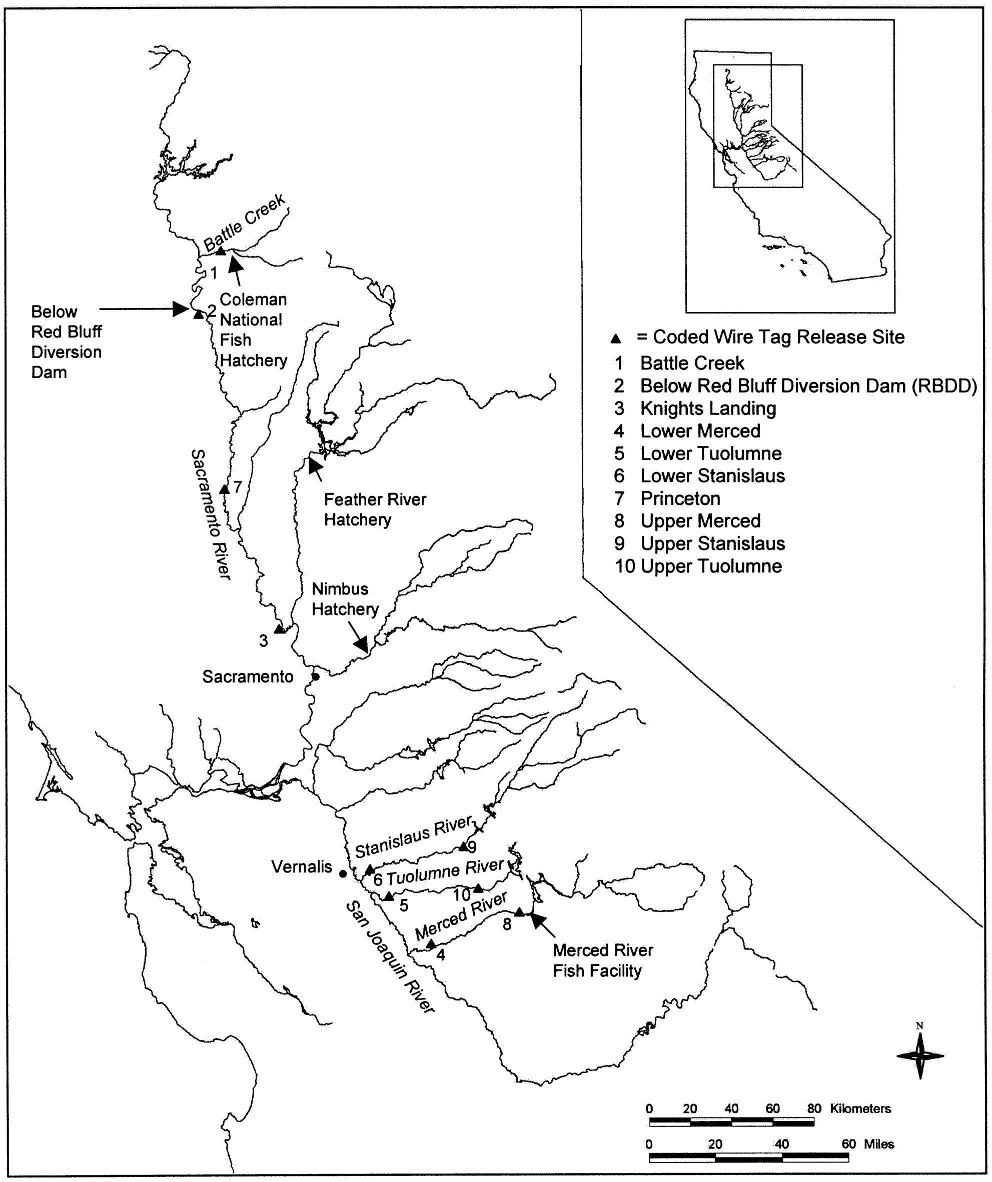


Figure 4 Map of coded wire tag release locations in the upper Sacramento River and San Joaquin River tributaries

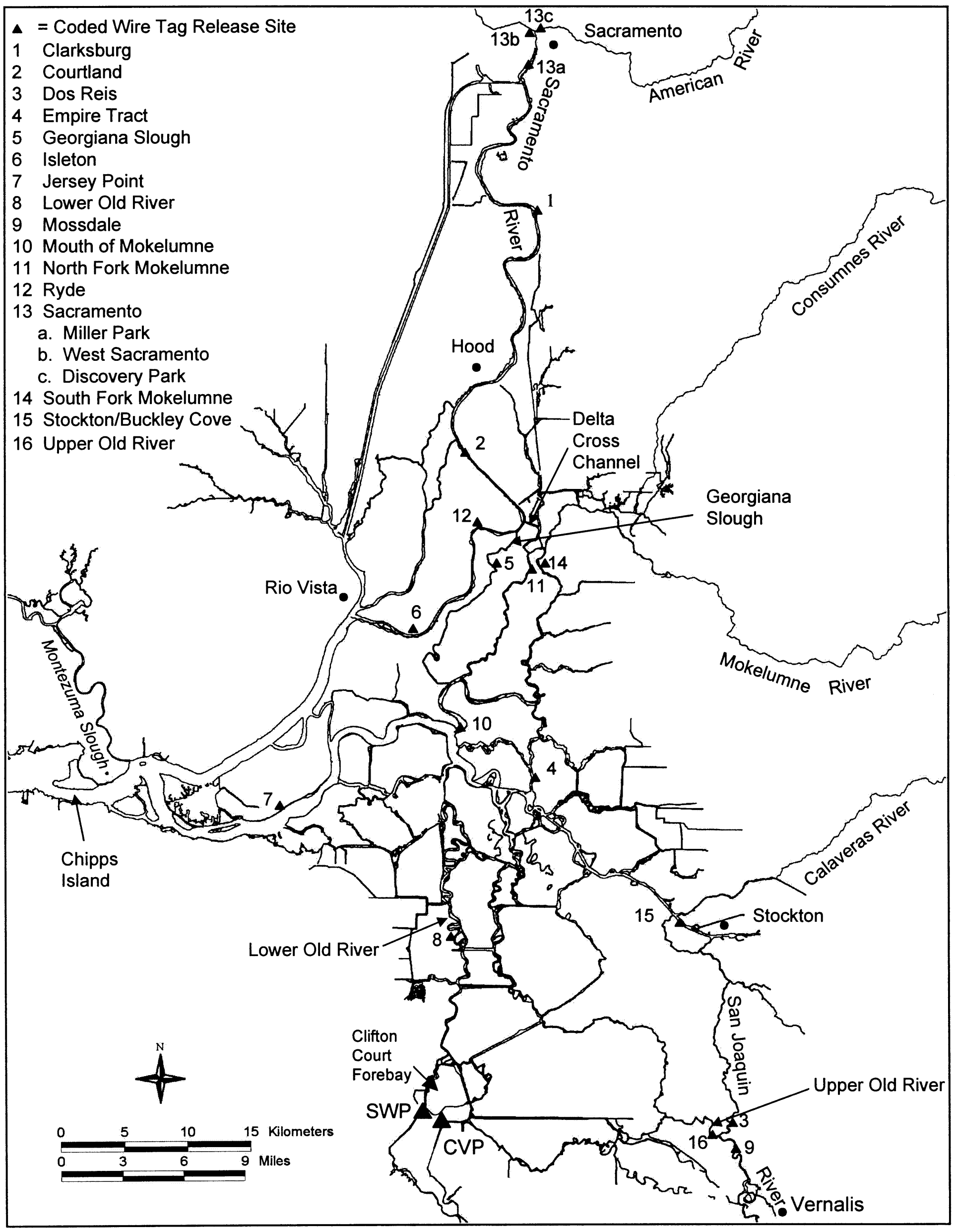


Figure 5 Detailed map of the Sacramento-San Joaquin Delta indicating coded wire tag release locations used between 1978 and 1997

There are many assumptions made in using hatchery fish to estimate the survival of wild fish. It is likely that wild fish survive at a greater rate than those released and reared at a hatchery (Reisenbichler and others 1992), but relative differences in survival of hatchery fish between different locations, times, sizes or other parameters can be informative. Using hatchery smolts to investigate factors affecting wild fish also seems appropriate (Kjelson and Brandes 1989) and we have found it useful in gaining information for managing and protecting wild juvenile salmon.

Chinook "fry," as defined in this report, is the life stage between emergence from the spawning gravel to the completion of upstream or estuarine rearing (<70 mm fork length). Juveniles that are starting to undergo behavioral and physiological changes to prepare for the transition to salt water are termed "smolts." In this report they are identified as juveniles equal to and greater than 70 mm fork length. Yearlings are defined as juveniles greater than 100 mm that have over-summered in freshwater.

Information contained in this paper is presented by topic: "Fry Abundance," "Smolt Abundance," "Fry Survival," and "Smolt Survival." Each topic includes methods, and results and discussion sections. The results and discussion sections under smolt survival are further sub-divided by basin (Sacramento and San Joaquin) and specific management issues.

The California Department of Water Resources provided flow and project export information via their DAYFLOW program. River flows were measured on the Sacramento River at "I" Street (in downtown Sacramento) and at Freeport, and on the San Joaquin River at Vernalis (Figure 1). River flows were estimated using calculations at Rio Vista and Stockton. Exports are the combined mean daily rate at the SWP and CVP in cubic feet per second (cfs).

A variety of statistical methods was used to evaluate relationships between abundance and survival and environmental conditions. Data used in the regression analyses were assessed for normality and heterogeneity of variance using the descriptive statistics function in SYSTAT 7.0 for Windows. Variables were transformed when necessary to meet the assumptions of parametric statistics.

Fry Abundance

Methods

Seasonal abundance and spatial distribution of juvenile salmon in the Sacramento-San Joaquin Estuary were estimated using beach seine surveys at sites in the Delta, lower Sacramento River and San Francisco Bay. Sites within the Delta and on the lower San Joaquin River were added in recent years to provide additional information on juvenile salmon distribution. Abundance and distribution data were collected to document the use of the Delta as a rearing area and evaluate its use relative to flow.

Beach seine sampling was made with a 15.2 by 1.2 m (50 ft by 4 ft) seine, with 3.2-mm (1/8-inch) mesh, during daylight hours. One seine haul was made at each sampling station. Thirty stations have been sampled weekly in the Delta and lower Sacramento River during the spring since 1979 and constitute core "historical" sites. Seven of the stations are located on the lower Sacramento River between Colusa and Elkhorn (10 miles north of Sacramento) and twenty-three sites are located in the Delta (Figure 6). The sites in the Delta were divided into three areas: the North Delta, Central Delta, and South Delta.

In addition, between 1981 and 1986, 16 stations were sampled twice a month in Suisun, San Pablo, and San Francisco bays (Figure 6) of which ten were re-sampled during the spring in 1997. Sites include boat ramps, mud banks, and sandy beaches. There were times when sampling was not possible due to changes in flow or other conditions that prevented site access. The beach seining sites added in recent years are located primarily in the South Delta and lower San Joaquin River (Figure 6). Additional sites on the Sacramento River have also been sampled in recent years, but discussion of these sites is not included in this report.

Water temperature was measured, and all fish species captured were identified and enumerated at each sample site. In each sample, up to 50 juvenile salmon were measured to the nearest millimeter fork length. All tagged salmon were kept for subsequent tag decoding.

Relative juvenile salmon abundance was compared within and between years using catch per haul or catch per cubic meter at the core "historical" sites sampled during similar periods between years. Average catch per haul is defined as the number of juvenile salmon caught divided by the number of seine hauls performed.

It became possible to calculate catch per cubic meter starting in 1985, when the depth, length, and width of the area swept by the beach seine were measured as part of the normal sampling protocol. Depth is the maximum depth swept by the seine haul. Length of the seine haul is the distance the haul was taken from shore and width is the measured scope of the seine haul, which is parallel to shore. The area of the seine haul was used to estimate the volume of water sampled, which was calculated by multiplying the depth of the sample by 0.5, then multiplying the product by the length and width of the seine haul. Catch per cubic meter (C/m^3) is estimated by dividing the catch by the volume of water sampled and yields a more robust density measurement than catch per haul.

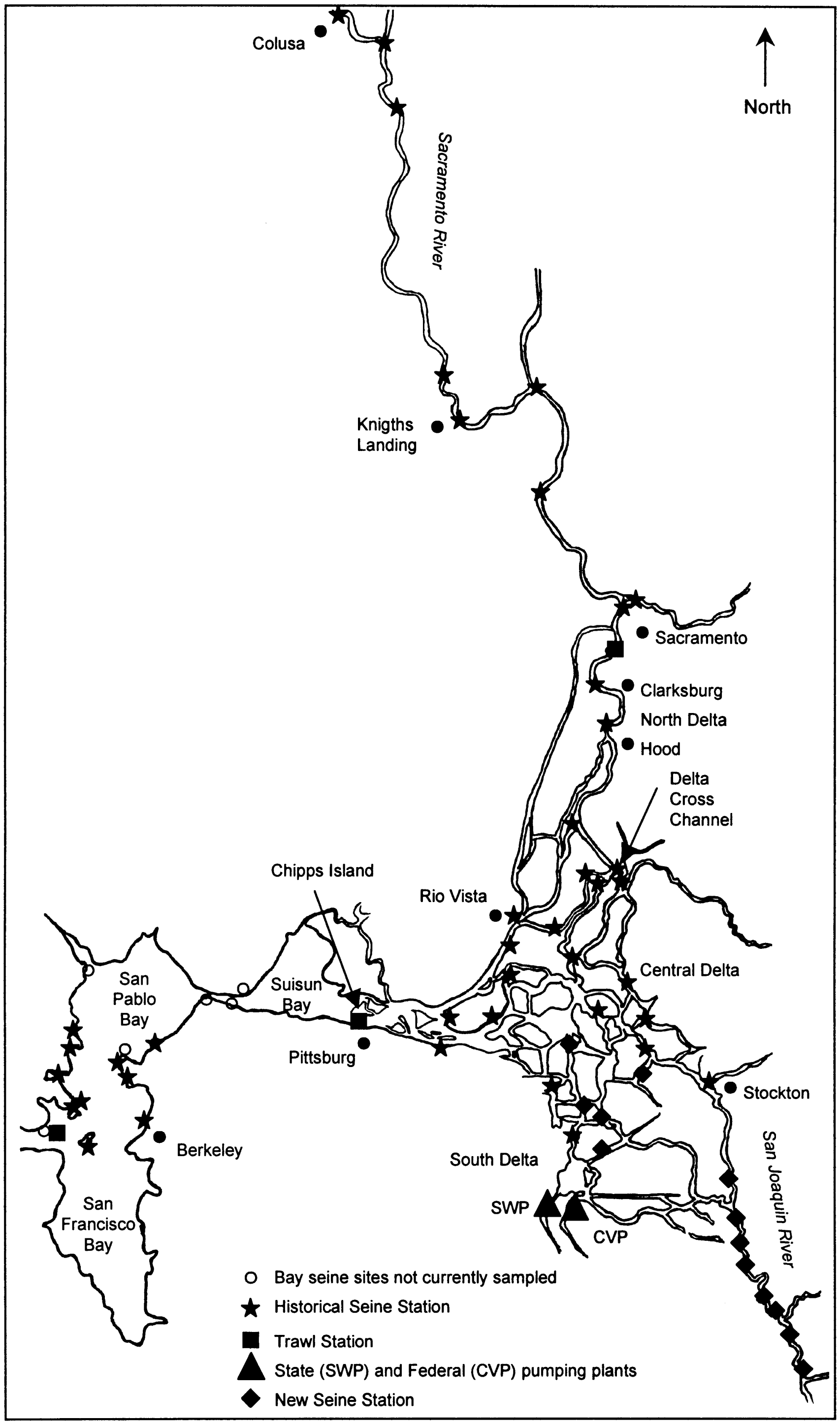


Figure 6 Sampling sites located in the Sacramento-San Joaquin Estuary, California

The average monthly C/m^3 and catch per haul, by area, was calculated by summing the average monthly C/m^3 or catch per haul for all sites within an area, and dividing by the number of sites sampled. The average monthly C/m^3 or catch per haul by site was estimated by summing the monthly C/m^3 or catch per haul for each site and dividing by the number of months sampled. Each monthly C/m^3 or catch per haul by site was estimated by summing the daily C/m^3 or catch per haul and dividing by the number of times the site was sampled within the month. The daily C/m^3 by site was estimated by dividing the catch by the volume of water sampled. Only one sample was taken at each site per day and generally each site was sampled once per week.

Simple linear regression analyses were used to determine if fry abundance in the North Delta and bay varied with flow. A constant 0.0001 was added to the catch per cubic meter in the bay before being log transformed. Sacramento River flow at Freeport was also log transformed for the regression analyses between catch per cubic meter in the bay and flow.

Results and Discussion

The number of fry in the estuary is influenced by the number of eggs deposited and environmental conditions during spawning, incubation, and rearing. Kjelson and others (1982) found that peak catches of fry in the Delta in the spring followed major runoff periods. We found that the annual spring abundance of fry in the Delta was also related to flow, with the highest abundance observed in wet years. Fry abundance in the North Delta between January and March, using catch per cubic meter in the beach seine, was significantly correlated ($r^2 = 0.69$, $P < 0.01$) to the mean flow in the Sacramento River at Freeport in February (Figure 7). Catch per cubic meter reduced the variability in the relationship even though some of the data from earlier years could not be included (Figure 8).

Based on sampling upstream of the Delta, it appears many fall run juveniles from the American and Feather rivers migrate to the Delta as fry in both wet and dry years (Snider and others 1998; Sommer and others 2001, this volume). Fry, originating from the San Joaquin tributaries, also were apparent in the Delta during the spring in the wet years (Figure 9). Sampling has not been conducted early enough in the season in dry years to determine if many fry move downstream into the Delta from the San Joaquin basin to rear in the drier years.

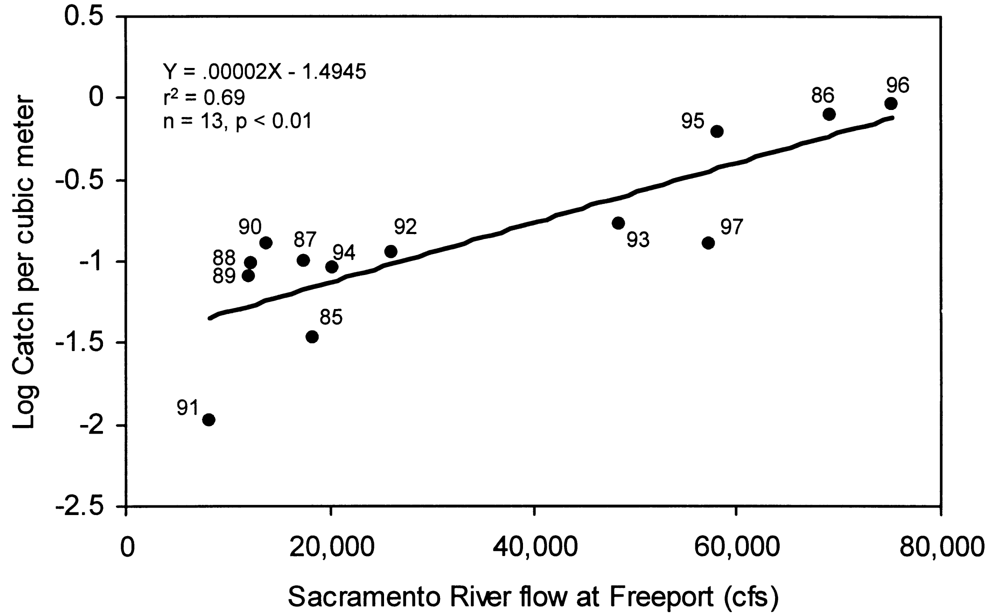


Figure 7 Catch per cubic meter of juvenile chinook salmon in the North Delta beach seine between January and March versus mean February flow on the Sacramento River at Freeport from 1985 to 1997

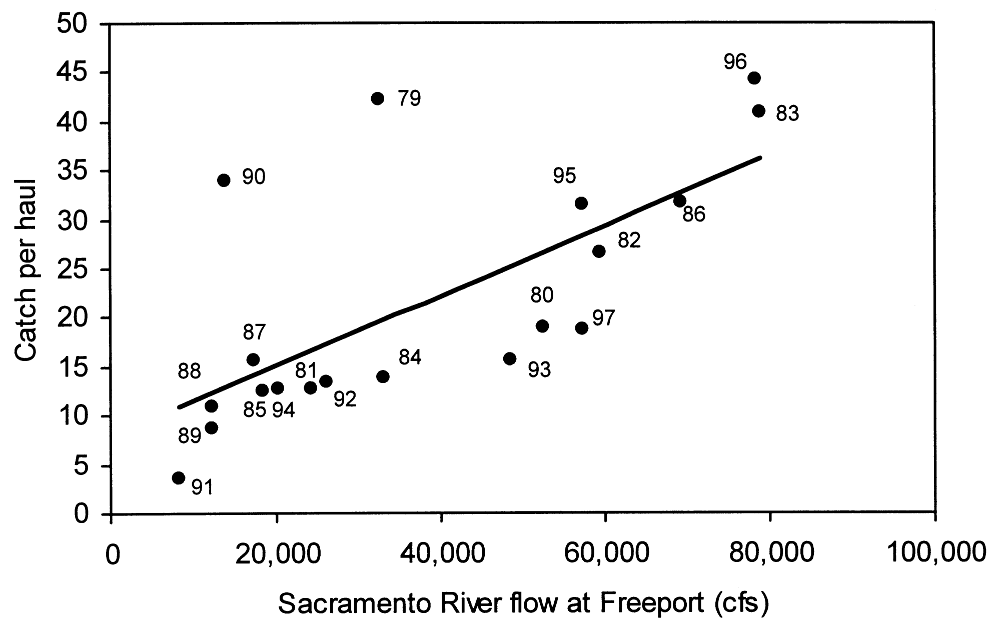


Figure 8 Catch per haul of juvenile chinook salmon in the North Delta beach seine between January and March versus mean February flow on the Sacramento River at Freeport from 1979 to 1997

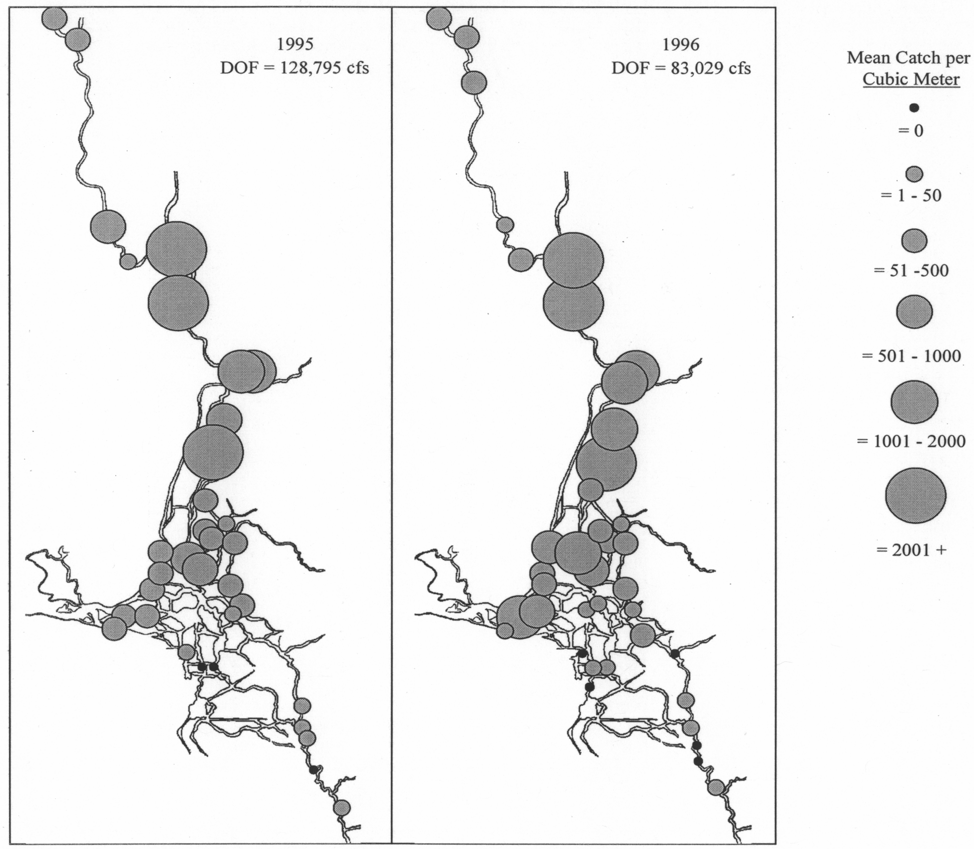


Figure 9 Mean monthly catch per cubic meter (x 1,000) of chinook salmon fry from beach seine sites and mean Delta outflow (DOF) between January and March in 1995 and 1996

Fry abundance during the spring in San Francisco Bay shows a similar effect of flow. We found that the average catch per cubic meter (plus 0.0001 and logged) in ten beach seine sites sampled in San Pablo and San Francisco bays (January through March) was positively correlated to the log of the mean daily Sacramento River flow at Freeport in February (r^2 = 0.98, P < 0.01) (Figure 10). Flow at Freeport was used, as most of the net flow moving from the Delta into the bay (Delta outflow) originates from the Sacramento River.

These results are consistent with Healy (1980) who observed increased chinook salmon fry catch during increased discharge in the Nanaimo River Estuary in British Columbia. Other studies have speculated that behavioral interactions and density dependent mechanisms were responsible for downstream migration (Healy 1991).

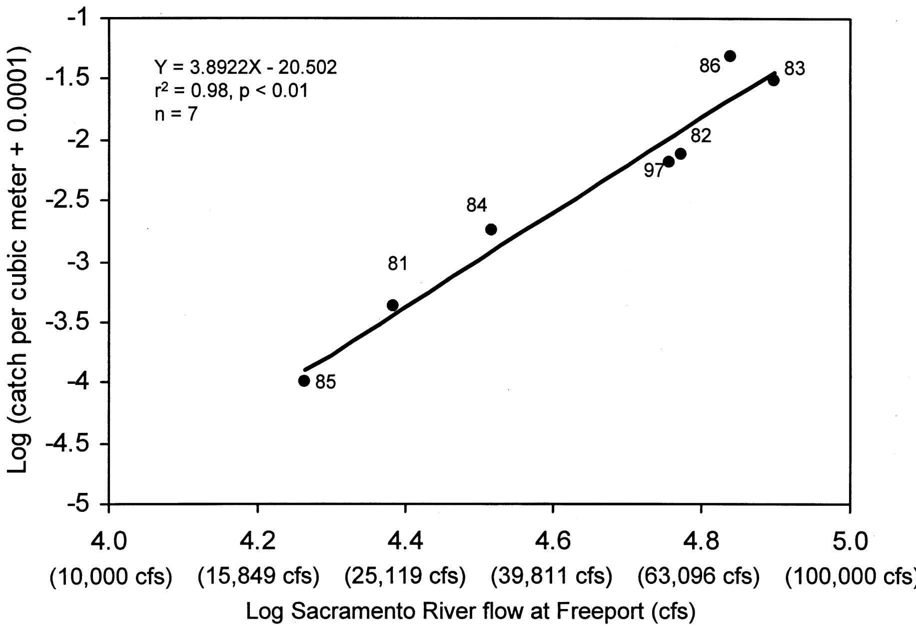


Figure 10 Log of catch of chinook fry per cubic meter (+ 0.0001) between January and March at beach seine sites in the San Francisco Bay versus log of mean flow at Freeport during February between 1981 and 1986 and in 1997. The equation without the 1986 outlier is $y = 3.6218x - 19.316$.

There were relatively few fry in the Delta during the other months of the year and likely reflected the lower abundance of the other races, lower Delta inflow, higher summer water temperatures and different life history strategies. Fry have been observed in the beach seining between April and July in some years; many were assumed to be late-fall run. They ranged in size between 30 and 53 mm. In addition, a nominal number of fry has been recovered in the Delta between November and January that ranged in size between 48 and 67 mm and were likely winter run. Overall, less than 300 fry have been observed in beach seining during the late spring and summer and late fall and winter between 1977 and 1997. In the earlier years, sampling was limited during the fall and winter months, but in recent years sampling frequency has generally been similar to that conducted in the spring.

Smolt Abundance

Methods

Since 1976, Kodiak or midwater trawls have been used near Sacramento and at Chipps Island (located near the city of Pittsburg) for a variety of purposes.

Initially, midwater trawling was conducted for approximately six weeks during the spring on the Sacramento River near Hood (1976–1981) and at Chipps Island (1976 and 1977) to recover marked fish released in those years (Kjelson and others 1982). Since 1978 at Chipps Island and 1988 at Sacramento, midwater trawling has been conducted between April and June to index the number of primarily fall-run smolts entering (Sacramento) and leaving (Chipps Island) the Delta (Figure 6).

Since 1992 at Sacramento and 1994 at Chipps Island, trawling has been conducted consistently between October and June and provides information on all races of juvenile salmon entering and leaving the Delta. Year-round trawling was conducted at Chipps Island in 1980 and at both locations in recent years (1996 and 1997). Starting in the fall of 1994, a Kodiak trawl replaced the midwater trawl at Sacramento during the fall and winter months to allow more intensive sampling of larger individuals from the less abundant races due to the larger net width and herding fashion of the Kodiak trawl (McLain 1998). The midwater trawling has continued at Sacramento between April and June to allow historical comparisons using the same gear.

Midwater trawling also was conducted in San Francisco Bay, near the Golden Gate Bridge (Figure 6) between 1983 and 1987. Sampling was conducted, primarily between April and July, to index the abundance of juvenile salmon migrating out of the bay during those months and to recover marked salmon released at Port Chicago in 1984, 1985, and 1986 (USFWS 1987). Only the survival information is presented in this report.

In general, 10 twenty-minute tows were done per sample day at each location, between three and seven days per week during the months sampling was conducted. Both the midwater trawl and Kodiak trawl fished the net at the surface. Occasionally, inclement weather, mechanical problems, or excessive fish catches required reducing tow times or the number of tows. All trawling at Sacramento was done in the middle of the channel facing upstream against the current within 1.5 km of the sample site. Trawling at Chipps Island also was done within 1.5 km from the sample site in both directions regardless of tide, and in three locations of the channel: north, south, and middle.

The midwater and Kodiak trawl nets at Sacramento, Chipps Island, and in San Francisco Bay varied in size and design. The midwater trawl net used at Sacramento had a mouth opening of 1.8 by 4.6 m (6 ft by 15 ft) (Figure 11a). The net tapered from the mouth to the cod end totaling 23.6 m (77.5 ft) to the beginning of the cod end. Net mesh varied from 102 mm (4 inches) to 6 mm (1/4 inch) at the cod end. Wings were constructed of 203-mm (8-inch) stretch mesh and attached to each of four corners of the net. Lead weights were attached to the bottom rib line of the net and floats attached to the top rib line. A metal depressor door was fastened to each bottom bridle line and an aluminum hydrofoil was fastened to each top bridle line. The midwater trawl at Chipps Island and in San Francisco Bay used a net with a mouth opening of 3.0 by 9.1 m (10 ft by 30 ft), was tapered from the mouth to the cod end, and totaled 25 m (82 ft) (Figure 11b). Net mesh and wings were similar to that used for the Sacramento midwater trawl. The Kodiak trawl net also was variable mesh with a fully expanded mouth opening of 1.8 by 7.6 m (6 ft by 25 ft) (Figure 11c). Net mesh varied from 51-mm (2-inch) stretch mesh to 6 mm (1/4 inch). A 1.8 m bar was attached to the front of each wing with lead and float lines on the bottom and top of the net respectively. The Kodiak trawl also incorporated a live box attached to the cod end of the net to avoid fish mortality. The live box consisted of perforated steel plating 6 mm (1/4 inch) in diameter.

Actual fishing dimensions of the nets varied and have been described in past reports (USFWS 1994). Based on these studies, the mean effective-fishing mouth size of the net at Sacramento was found to be 5.1 m^2 and 18.5 m^2 at Chipps Island. The estimated fishing net mouth size of the Kodiak trawl, based on these midwater trawl studies, was 12.5 m^2. The catch per cubic meter and mean amount of water sampled reported in this paper were based on these fishing mouth dimensions.

Cubic meters of water sampled with the trawls were estimated with a General Oceanics mechanical flowmeter (model 2030). Linear meters were calculated by multiplying meter rotations with the Standard Speed Rotor Constant (26,874) and dividing the result by a conversion factor (999999). The volume of water sampled was calculated by multiplying the number of linear meters traveled per tow by the mouth opening of the net.

Relative abundance was compared using average catch per cubic meter (C/m^3), where C/m^3 per tow equaled: catch per tow/net mouth area (m^2) x linear meters traveled through the water (m). Averages were calculated for each day, week and month. Each daily C/m^3 was calculated by averaging each C/m^3 per tow and dividing by the number of tows that day. Each weekly C/m^3 was calculated by summing the daily C/m^3 and dividing by the number of days sampled within the week. The monthly C/m^3 was the sum of weekly averages divided by the number of weeks sampled per month. Weeks were designated as Monday through Sunday and weeks which overlap months were split and included in their respective months.

Simple linear regression techniques were used to evaluate the relationships between C/m^3 and river flow. Mean C/m^3 between April and June at Sacramento was squared before regression analysis.

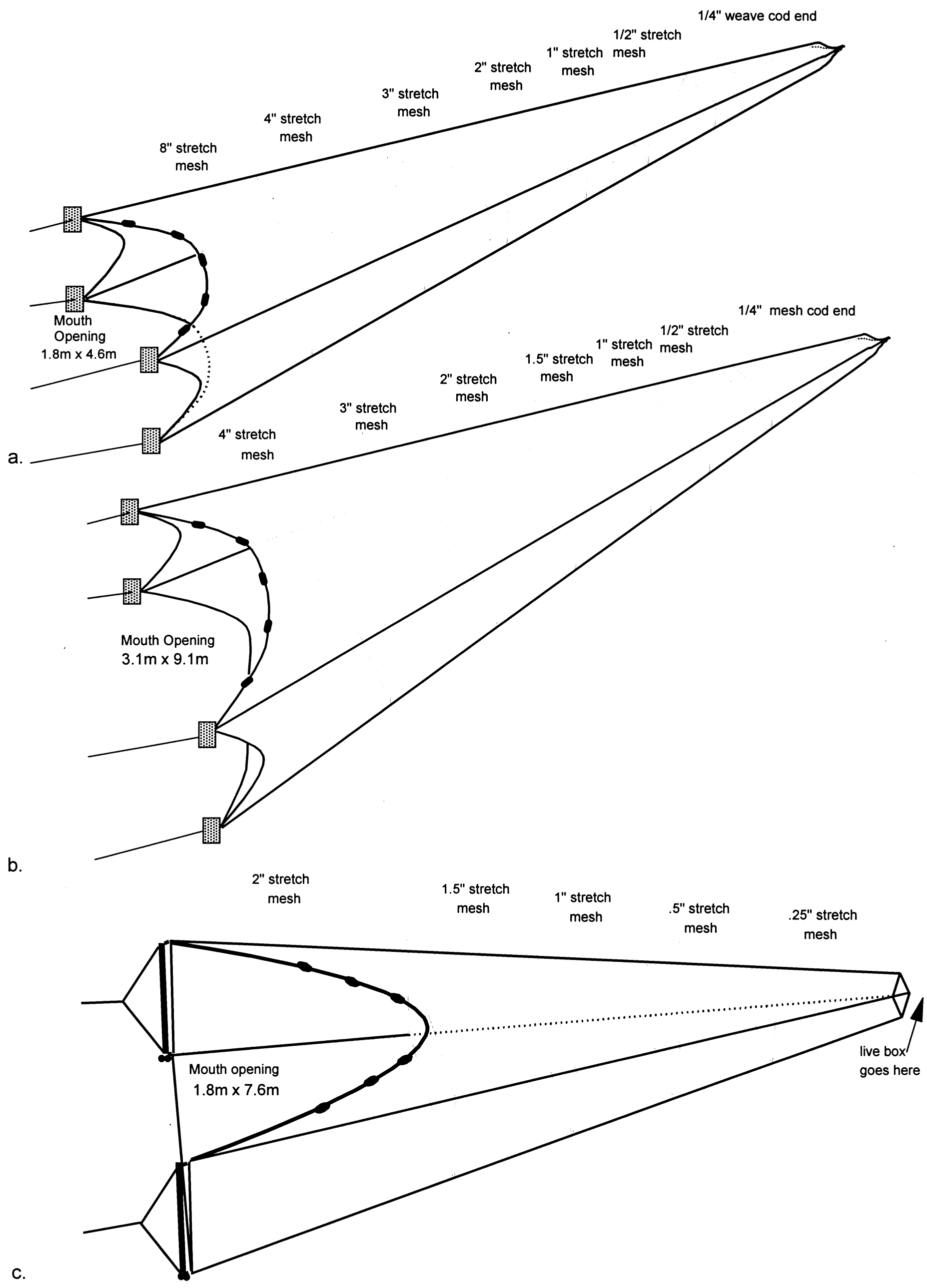


Figure 11 Schematic drawing of (a) midwater trawl net used at Sacramento, (b) midwater trawl net used at Chipps Island and in San Francisco Bay and (c) Kodiak trawl net used at Sacramento

Results and Discussion

The mean midwater trawl C/m^3 (squared) of unmarked smolts, primarily fall run, migrating past Sacramento between April and June was inversely and significantly (r^2 = 0.88, P < 0.01) related to mean Sacramento River flows in February (Figure 12). If this density measurement is a true index of abundance then it appears fewer smolts migrate into the Delta when flows are higher in the early spring (February).

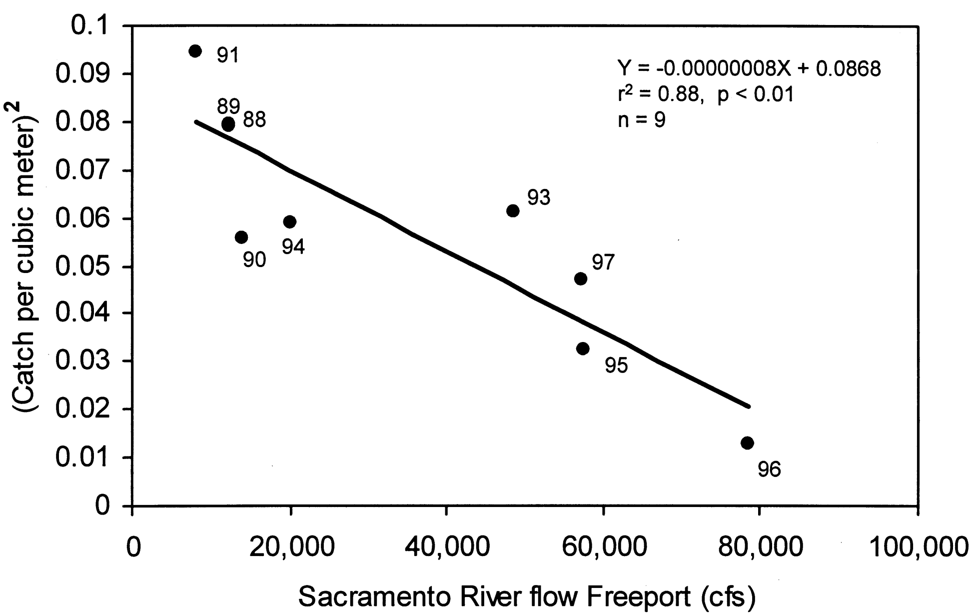


Figure 12 Mean catch of unmarked chinook salmon smolts per cubic meter (squared) in the midwater trawl at Sacramento between April and June of 1989 to 1997 versus mean daily flow (cfs) at Freeport on the Sacramento River during February. Data from 1992 were not included in the model because no sampling was done during April and late June in that year.

Catch of unmarked smolts in the midwater trawl at Chipps Island indicated that overall juvenile salmon production migrating from the Delta was greater in wet years. Mean catch per cubic meter between April and June at Chipps Island was positively correlated to flow at Rio Vista (r^2 = 0.78, P < 0.01), indicating that, overall, the density of juveniles leaving the Delta increases as flows increase (Figure 13). In addition, since many fry were observed downstream of Chipps Island in high flow years before April, the estimates of the juvenile production migrating past Chipps Island was underestimated in the high flow years. Stevens and Miller (1983) also found significant relationships between inflow and an index of abundance of fall run chinook in the Delta between April through June.

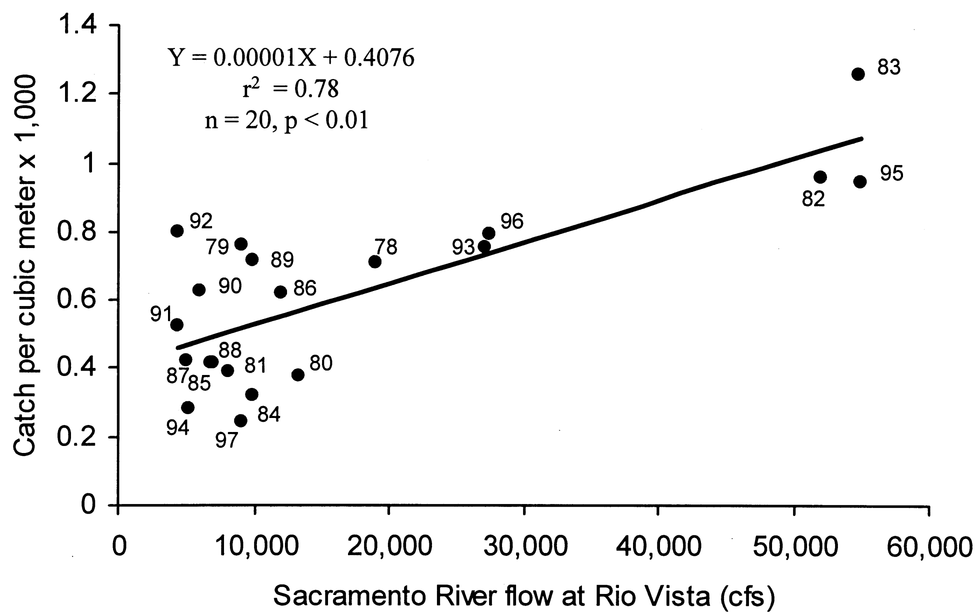


Figure 13 Mean catch of unmarked chinook salmon smolts per cubic meter (x 1,000) in the midwater trawl at Chipps Island between April and June from 1978 through 1997 versus mean daily Sacramento River flow (cfs) at Rio Vista between April and June

Catches at both Sacramento and Chipps Island include fall-run smolts released from Coleman National Fish Hatchery. Therefore, the Chipps Island abundance versus flow relationship incorporates flow effects on these hatchery fish as well as wild smolts. In recent years, about 12 million smolts have been released (Tom Nelson, personal communication, see "Notes"). Most other unmarked hatchery fish in the Central Valley are released downstream of Chipps Island.

Catches at Sacramento and Chipps Island during other months of the year indicated low abundance, until the December-January period when fall run fry enter the catches (Figure 14 and 15). Although, Figures 14 and 15 do not precisely show abundance, they show all unique lengths measured which illustrates this point.

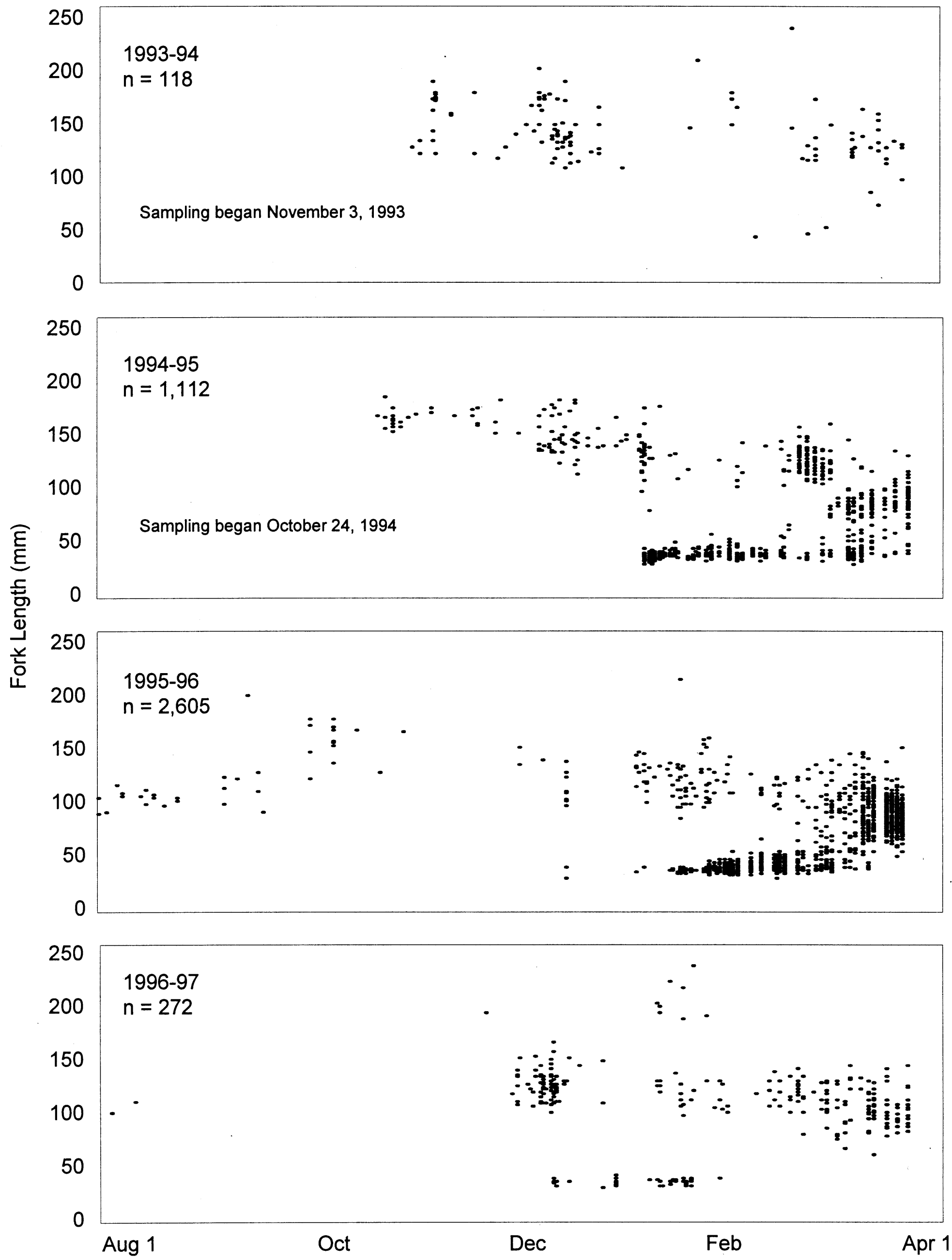


Figure 14 Measured juvenile chinook captured in the midwater trawl at Chipps Island near Pittsburg, California, between August 1 and March 31

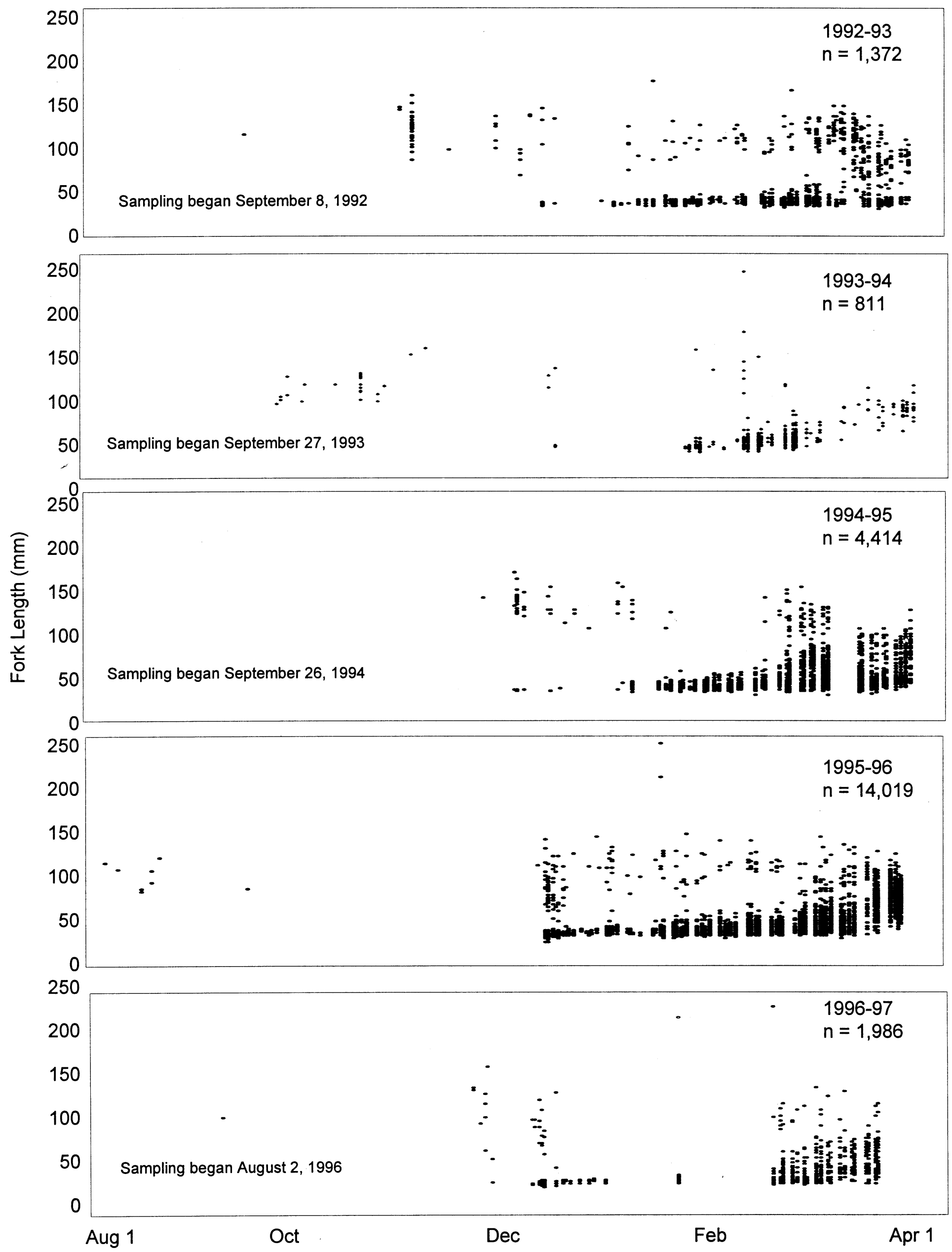


Figure 15 Measured juvenile chinook salmon captured in the midwater trawl and/or Kodiak trawl on the Sacramento River near Sacramento between August 1 and March 31

Fry Survival

Methods

Mark and recapture experiments with fry were conducted between 1980 and 1987 to (1) estimate survival in the upper Sacramento River, Delta and San Francisco Bay, under various river flows and (2) in later years, assess the impacts on survival of using existing Delta channels for water transport. Survival for fish released upstream in the Delta and in the bay was evaluated at various flows because river flows were anticipated to change with the operation of the proposed Peripheral Canal. The effects to juvenile salmon of using existing Delta channels for water transport were evaluated by estimating differential survival of marked fry released at locations in the North, Central and South Delta. Fry releases in the Delta were discontinued in 1988 to increase the number of marked smolts available for release.

Fry were obtained from Coleman National Fish Hatchery (Figure 4), adipose fin-clipped, and tagged in the snout of the fish with coded-wire half tags (CW½T). Recoveries of these marked fish were made in the beach seine, at the State and federal fish salvage facilities located at the respective pumping plant intake, and in the ocean fishery.

Ocean recovery rates are relative indices that were used to compare survival between locations within a year. The ocean recovery rate is the expanded number of recoveries in the ocean fishery divided by the number released (Kjelson and Brandes 1989). Catches in the ocean sport and commercial fishery were expanded based on the percentage of sampling conducted at the various ports (PSMFC 1998).

To compare survival between years, an estimate of absolute fry-to-smolt survival was obtained by comparing the recoveries in the ocean fishery of fry released in the Delta (or upstream) to those released at Port Chicago (or Benicia) in Suisun Bay (Kjelson and Brandes 1989). In some cases releases at Ryde were used as the downstream control group. We assume that the ratio between upstream and downstream groups factors out the smolt survival downstream of Suisun Bay from the upstream release group.

Ocean recovery rates for CW½T groups released on different days at the same location were averaged before analyses. Groups with different tag codes released at the same location on the same day were considered one group and recoveries were summed and divided by the total number released to represent the group. Two sample and student *t*-tests were used to test for significant differences between treatments at the 95% confidence level.

Results and Discussion

Ocean recovery rates indicated that relative survival was higher for fry released in the upper Sacramento River below Red Bluff Diversion Dam (RBDD) than for fry released in the North Delta, especially in the higher flow years (Figure 16). Those released in the bay had the lowest recovery rates in all years. The upper river release groups were recovered, on average, about five times greater than those released in the Delta in wetter years of 1980, 1982, and 1986 (Figure 16). We have defined the wetter years as those with mean February flows at "I" Street greater than 50,000 cfs. Although a dry year, 1987 also exhibited much greater survival upstream than in the Delta.

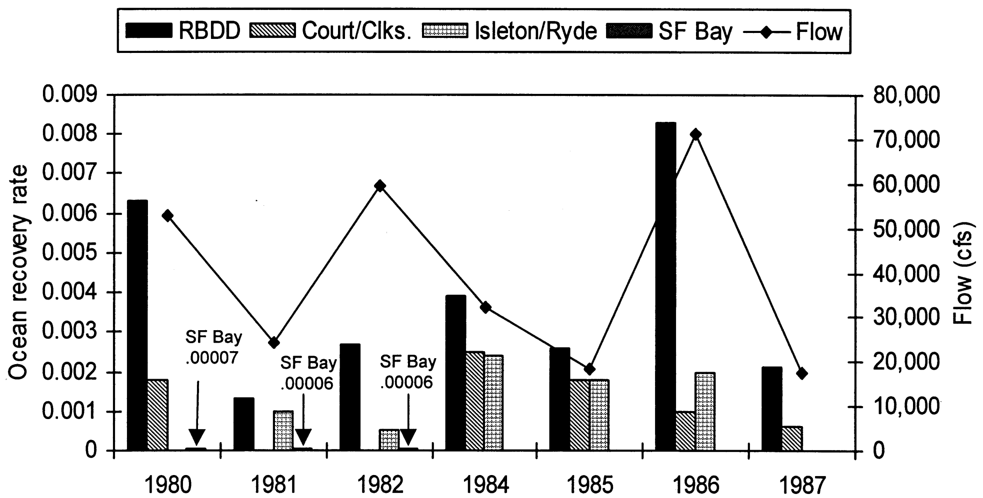


Figure 16 Ocean recovery rates of CW½T fry released in the upper Sacramento River below Red Bluff Diversion Dam (RBDD), in the Delta at Courtland or Clarksburg and Isleton or Ryde, and mean daily Sacramento River flow at "I" Street in February

Estimates of absolute survival provide additional support for the conclusion that survival is higher for upstream releases in the wet years. Absolute survivals of the RBDD release groups were significantly higher than the Delta release groups in wet years (two sample *t*-test, $t = 8.28$, $n = 3$, $P = 0.014$) (Table 1). In the drier years there was not a significant difference between fry released upstream and the Delta.

Table 1 Survival estimates for CW½T fry released below Red Bluff Diversion Dam in the Delta, mean daily Sacramento River flow at "I" Street during the month of February, and ocean recovery rates for smolts released at Port Chicago, Benicia or Ryde[a]

	Release location				
Year	*Red Bluff Diversion Dam*	*Courtland or Clarksburg*	*Isleton or Ryde*	*Port Chicago, Benicia, or Ryde*	*Mean daily Sacramento River flow at "I" Street (cfs)*
1980	0.29	0.08		0.022[b]	52,576
1981	0.05		0.04	0.028[b]	24,239
1982	0.39		0.07	0.009	59,432
1984	0.51	0.33	0.32	0.008[b]	32,949
1985	0.26	0.19	0.18	0.010[b]	18,376
1986	0.29	0.03	0.07	0.029[b]	69,306
1987	0.10	0.03		0.020[b]	17,404

[a] A Ryde release was used in 1987 because there were no groups released at Port Chicago or Benicia that year.

[b] Indicates Feather River Hatchery stock was used for the release. For all other releases, Coleman National Fish Hatchery stocks were used.

The observed wet year differences could be a result of increased survival of upstream fish or decreased survival in the Delta. The fact that Delta survival was not lower in wet years suggests that the trends are due to improved survival upstream. One hypothesis is that increased flows provide additional rearing habitat in the upper Sacramento River since there are large areas of floodplain (e.g. the Sutter and Yolo bypasses) that become accessible. Such habitat is not present along the Delta levees. Another explanation could be that some proportion of those released in the Delta moved downstream into the bay in the high flow years where observed survival was extremely poor making comparisons between those released in the Delta and those released upstream more difficult. Those released upstream also could have moved downstream into the Delta in the high flow years. Review of the recoveries by location in the beach seine survey indicated that some of those released upstream below Red Bluff Diversion Dam were recovered in the Delta soon afterwards, but recoveries were made in both dry and wet years (Table 2).

Table 2 CW½T fry released in the Delta and upper Sacramento River below Red Bluff Diversion Dam (Below RBDD) and recovered as fry (<70 mm) downstream of the Delta (Bay) and in the Delta, respectively, between 1980 and 1982[a]

Release site and date	*Recapture site (Delta or Bay)*	*Recapture date*
Clarksburg (Delta)		
26 Feb 1980	Crockett Marina (near Benicia) (Bay)	03 Mar 1980
07 Mar 1980	Montezuma Slough (Bay)	11 Mar 1980
Below RBDD		
12 Mar 1980	American River (near Sacramento) (Delta)	25 Mar 1980
	Brannon Island (near Rio Vista) (Delta)	02 Apr 1980
Isleton (Delta)		
12 Feb 1981	Montezuma Slough (2)	17 Feb 1981
	Montezuma Slough (2)	04 Mar 1981
04 Mar 1981	Montezuma Slough	17 Mar 1981
Below RBDD		
06 Feb 1981	Steamboat Slough (Delta)	12 Feb 1981
	Isleton (Delta)	26 Feb 1981
	Montezuma Slough (2)	04 Mar 1981
Isleton (Delta)		
02 Mar 1982	Antioch (near Chipps Island) (Bay)	30 Mar 1982
Below RBDD		
05 Feb 1982	Discovery Park (near Sacramento) (Delta)	09 Mar 1982
25 Feb 1982	Ryde (Delta)	09 Mar 1982
	Discovery Park	16 Mar 1982
	Discovery Park	30 Mar 1982

[a] No recoveries of fry released in the Delta or in the upper Sacramento River were made downstream of the Delta or in the Delta, respectively, between 1983 and 1987.

To evaluate growth as a potential mechanism for the higher survival observed upstream in these high flow years, we looked at growth rates of the CW½T fish released and recovered upstream and in the Delta in 1982, a high flow year. We did not find significant differences in growth between the two areas (using a student *t*-test to compare the slopes of the two lines) (Figure 17).

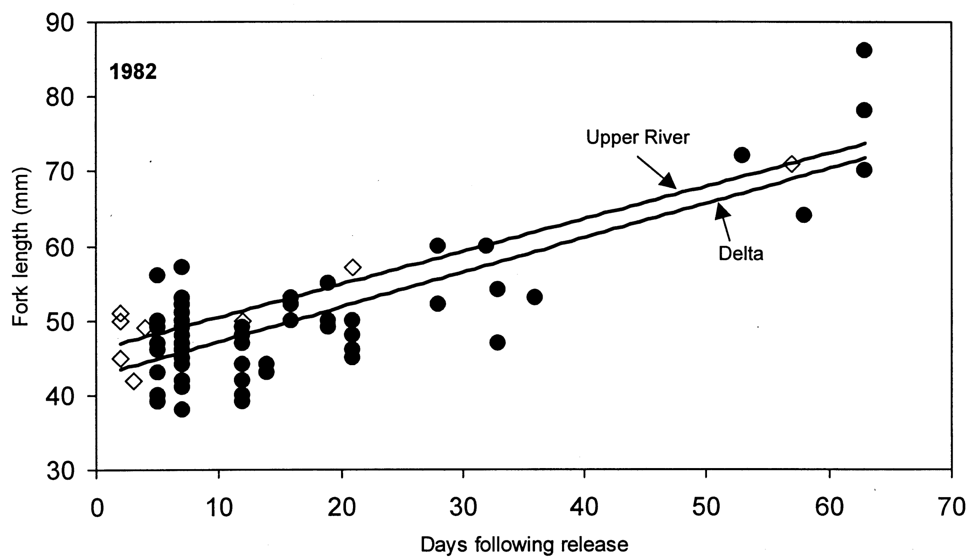


Figure 17 Growth curves for CW½T fry released and recaptured in the Delta (circles) and the upper Sacramento River (diamonds), between February 7 and April 28, 1982

Results, from additional CW½T fry released at various locations in the Delta between 1981 and 1985, indicated that in drier years survival was higher in the North Delta than in the Central Delta. Although not statistically significant, the ocean recovery rates were somewhat higher from CW½T fry released in the North Delta (Courtland, Ryde, or Isleton), relative to those released in Central Delta (at the mouth of Mokelumne River and in the North and South Forks of the Mokelumne) in the drier years (Figure 18). In the wetter years of 1982 and 1983, those released in the Central and South Delta (the mouth of the Mokelume River) appeared to survive at a similar rate as those released at Isleton (Figure 18). The lower Old River release even seemed to survive at a relatively high rate in 1983 (Figure 18). One mechanism for the lower survival of fry released in the Central Delta in dry years could be the greater effect of the pumping plants on hydrology in these years. In dry years (1981, 1984, 1985, and 1987), CW½T fry were recovered at the fish facilities, whereas in the wetter years they were not (1980, 1982, 1983 and 1986) (Appendix A).

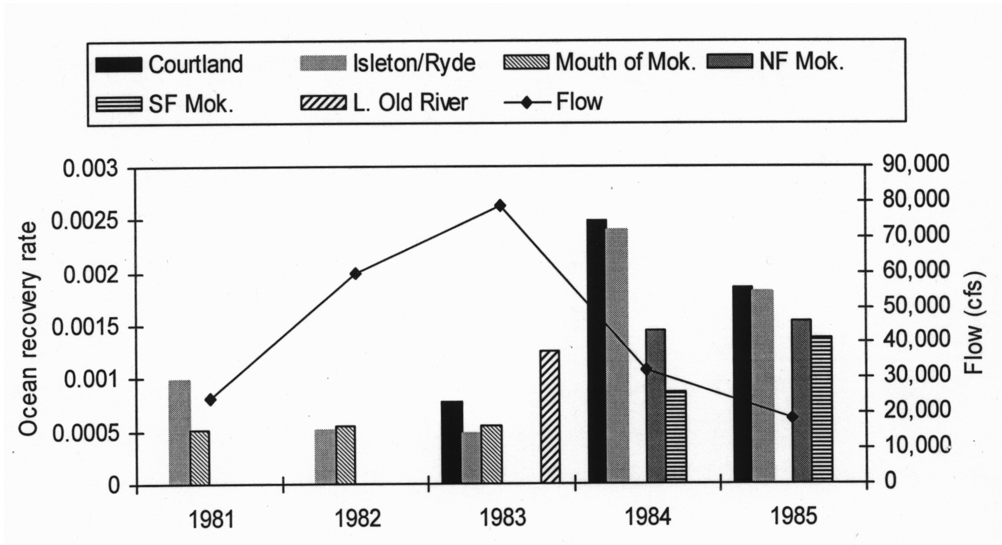


Figure 18 Ocean recovery rates for CW½T fry released at various locations in the Delta and mean daily Sacramento River flow at "I" Street in February

Smolt Survival

Methods

Mark and recapture studies also were conducted with fall run smolts starting in 1969. Survival through the Delta for smolts released near Sacramento was estimated between 1969 and 1971, 1976 and 1977, and 1978 and 1982 to document the importance of freshwater inflow on the survival of juvenile salmon migrating through the estuary (DFG 1976; Kjelson and others 1981, 1982). In 1983, the program was expanded to also examine the differential vulnerability to water project operations of marked smolts released at four locations in the Delta. These experiments were also used to evaluate the effect of movement into the Central Delta via the Delta Cross Channel and Georgiana Slough on the survival of juvenile salmon in the Delta. To separate the effects of flow from diversion into the Central Delta, experiments were conducted between 1987 and 1989 during low flows with the Delta Cross Channel gates open and/or closed. Prior to 1987, closure of the Delta Cross Channel gates only occurred in wet years. Between 1992 and 1997, survival was evaluated for fall-run smolts and late-fall-run yearlings released into Georgiana Slough in the Central Delta relative to those released on the mainstem Sacramento River. Late-fall-run yearlings were used as surrogates for winter-run juveniles to estimate the effects of diversion into the Central Delta on winter-run salmon.

Mark and recapture methodology also was used to evaluate survival in the San Joaquin Delta starting in 1985. Between 1985 and 1990, marked fish releases were made to evaluate the differential survival of smolts migrating through upper Old River relative to those continuing to migrate down the mainstem San Joaquin River. In 1992, 1994, and 1997, a temporary rock barrier was installed in upper Old River and marked smolts were released to determine if survival through the South Delta was increased with the barrier in place. In 1997, the rock barrier was changed to include two 48-inch culverts. In 1993, 1995, and 1996, the barrier was not installed because of high flows or lack of a permit, although survival through the South Delta was measured for comparison purposes. In addition, releases also were made in 1995 and 1996 to estimate the mortality associated with migration through upper Old River. Paired releases with smolts from both Feather River Hatchery and Merced River Fish Facility were made in 1996 and 1997 to address concerns that stock origin of the experimental fish had confounded previous results. In addition, physiological studies were conducted and subsets of fish were held in live cars to determine the potential cause of mortality or mortality differences between stocks if they were found. The role of exports was explored in 1989, 1990 and 1991 when releases were made at high, medium, and low exports.

Additional marked fish releases were made in the bay and upstream of the Delta. Survival through the bay was estimated to help develop outflow criteria to meet the needs of juvenile salmon migrating through San Pablo and San Francisco bays. Survival of smolts released from Coleman National Fish Hatchery into Battle Creek, at Merced River Fish Facility, and from the Feather River Hatchery released at the Feather River (Figure 4) has been measured in many years and provides an index of the survival of smolts migrating through the rivers and Delta.

For smolt and yearling mark and recapture experiments, hatchery fish were spray-dyed or fin-clipped and tagged with full sized coded-wire tags (CWT). Fall-run smolts used in the Delta experiments were obtained from Feather River Hatchery (FRH). Late-fall-run yearlings were obtained from Coleman National Fish Hatchery (CNFH). Hatchery smolts used in the San Joaquin Delta experiments originated from the Merced River Fish Facility (MRFF) between 1985 and 1987 and from the FRH between 1990 and 1995. In 1989, 1996, and 1997, both MRFF and FRH stocks were used. Smolts released at Jersey Point between 1989 and 1991, and 1994 and 1997 originated from FRH. In 1996 and 1997 releases also were made at Jersey Point with smolts from MRFF. Two groups of smolts released at Port Chicago and in San Francisco Bay in 1984 were from Nimbus Fish Hatchery. The location of the hatcheries is shown in Figure 4.

Water temperatures were measured in the transport truck (both at the hatchery and at the release site) and in the receiving water.

Recoveries of marked smolts and yearlings were made in the midwater trawl at Chipps Island, at the CVP and SWP fish salvage facilities, and as adults in the ocean fishery. (This report does not discuss inland adult recoveries.) Recoveries at the fish salvage facilities provided insight into the direct mortality of juvenile salmon within the Delta.

Sampling at the State and federal facilities generally occurred at ten-minute intervals every two hours, 24 hours per day, although the sampling protocol before 1985 was not as thorough or systematic. Marked salmon observed in the sampling were kept for tag recovery and were called unexpanded recoveries. To estimate the total number of marked salmon salvaged at the facilities (expanded salvage) those recovered in the sample are expanded by fraction of time sampled. (It should be noted that expanded salvage is not "loss." Loss would include mortality associated with pre-screen and screen efficiency losses.)

Relative and absolute survival were estimated using recoveries made at Chipps Island and in the ocean fishery. Survival indices to Chipps Island (relative survival) were estimated by dividing the number of fish recovered from each particular tag group by the number released, corrected for the fraction of time and channel width sampled using the midwater trawl at Chipps Island (Kjelson and Brandes 1989). Relative survival also was estimated using the recovery rate of marked fish as adults in the ocean fishery and was used to compare survival between locations within a year. Survival estimates (absolute survival) were obtained using the differential recovery rate of an upstream group relative to a downstream group, either at Chipps Island or in the ocean fishery and used to compare survival between years. This approach has the advantage of reducing variation due to differential gear or sampling efficiency between years. We have termed this absolute survival or a survival estimate, but it is more appropriately described as a standardized estimate of survival between two locations. The Chipps Island absolute survival estimates have the additional advantage of not incorporating the variability due to ocean residence and having the information available within months instead of years of release.

Several pieces of evidence indicate that our survival indices of hatchery fish do not have substantial bias. First, we show that smolt survival indices at Chipps Island were generally supported by similar trends of survival estimates using the ratio of ocean recovery rates. In addition, while recoveries at Chipps Island were relatively small, they seemed generally similar between separate tag codes from the same group (Appendix B). While these multiple tag codes within a group provided some assessment of the recapture variability both at Chipps Island and in the ocean fishery, true measurement of the variability in survival is not possible given the limits of releasing independent replicates each year. In addition, although in many years, especially on the San Joaquin River, survival is so low that determining true differences is problematic, we were able to detect large differences in survival between release locations, years and river basins.

Paired sample *t*-tests were used to test for significant differences with 95% confidence levels between survival indices of smolts released upstream and downstream of the Delta Cross Channel and Georgiana Slough with the cross channel gates open and closed. Simple linear regression analysis was used to explore the relationship between Georgiana Slough survival estimates and combined CVP and SWP exports. Regression analysis also was used to determine the relationship between survival estimates for smolts released at Dos Reis and river flow at Stockton.

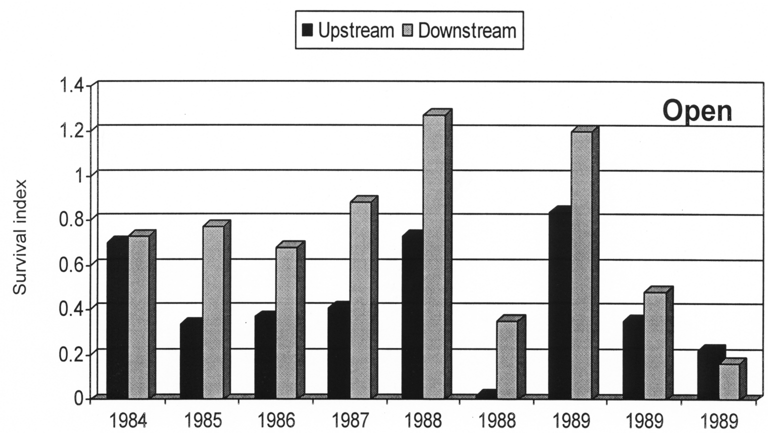
Results and Discussion

Sacramento

Role of Flow, Temperature and Diversion into the Central Delta on Survival. Kjelson and others (1982) reported a relationship between estimated CWT salmon survival rates and river flow, which suggested that river flows influenced juvenile salmon survival during downstream migration through the Delta. In 1982, they reported that survival (based on adult recoveries in the ocean fishery) in the Delta appeared to be influenced by water temperature and/or river flow rate: smolt survival decreased as flow rates decreased and temperatures increased. For trawl recovery data, smolt survival was related to water temperature only during June (Kjelson and others 1982). Almost total mortality was observed using both methodologies in 1978 and 1981 when temperatures were about 23° C (Kjelson and others 1982).

Data gathered between 1982 and 1987, using marked smolts released near Sacramento, further supported these relationships. In presenting the "State Water Resources Control Board with the Needs of Chinook Salmon, *Oncorhynchus tshawytscha* in the Sacramento-San Joaquin Estuary," USFWS (1987) shared relationships of survival with flow and survival with temperature using both the trawl and ocean indices of Delta survival. Maximum survival was reached with calculated flows between 20,000 to 30,000 cfs at Rio Vista and with temperatures less than 17° C. It also was shown that survival of smolts released in the North Delta (Sacramento or Courtland) using differential ocean recovery rates was correlated with the percentage of water diverted into the Central Delta from the Sacramento River at Walnut Grove (USFWS 1987). Determining which factor was most important to the survival of juvenile salmon was not possible because water temperatures and the percentage of water diverted into the Central Delta were higher in dry years. Prior to 1987 the Delta Cross Channel gates were only closed when flows in the Sacramento River at Freeport were greater than about 25,000 cfs.

Data collected between 1987 and 1989, combined with the data collected in earlier years, showed that smolts released on the Sacramento River, upstream of the entrances to the Delta Cross Channel and Georgiana Slough (Courtland), survived at a significantly lower rate than those released downstream (Isleton or Ryde), with the cross channel gates open (paired *t*-test, $t = 4.11$, $n = 9$, $P = 0.003$) (Figure 19). The results of these studies indicated that smolts were diverted into the Central Delta via the Delta Cross Channel and Georgiana Slough and entering the interior Delta decreased their survival. In addition, the data also showed that survival was significantly less for smolts released upstream relative to those released downstream, when the Delta Cross Channel gates were closed (paired *t*-test, $t = 10.75$, $n = 4$, $P = 0.002$) (Figure 19), indicating that diversion into Georgiana Slough also negatively affects survival. Smolt survival information obtained from the ocean fishery showed generally the same trends but was more variable and not statistically significant (Figure 20).



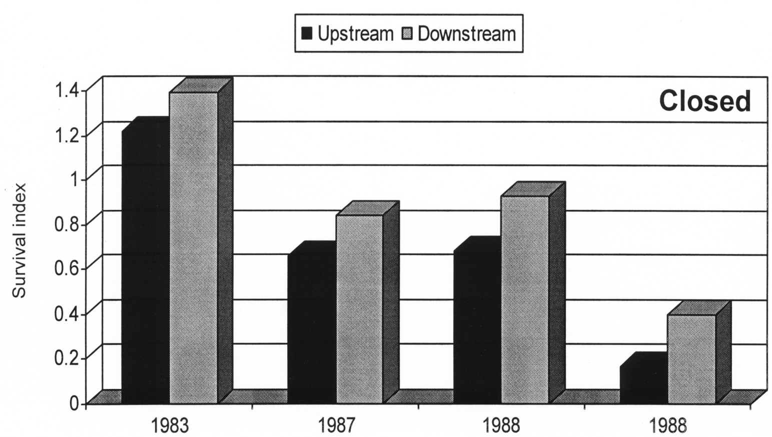
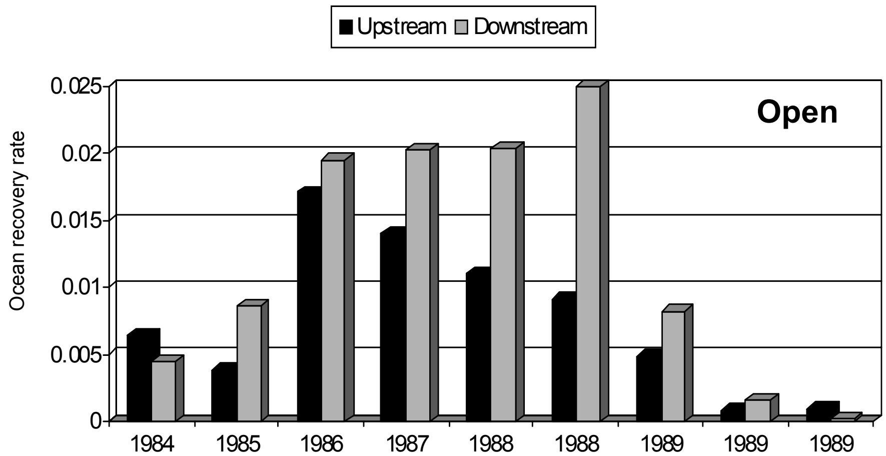


Figure 19 Survival indices of CWT fall run smolts released in the Sacramento River upstream (Courtland) and downstream (Ryde) of the Delta Cross Channel and Georgiana Slough with the gates open and closed



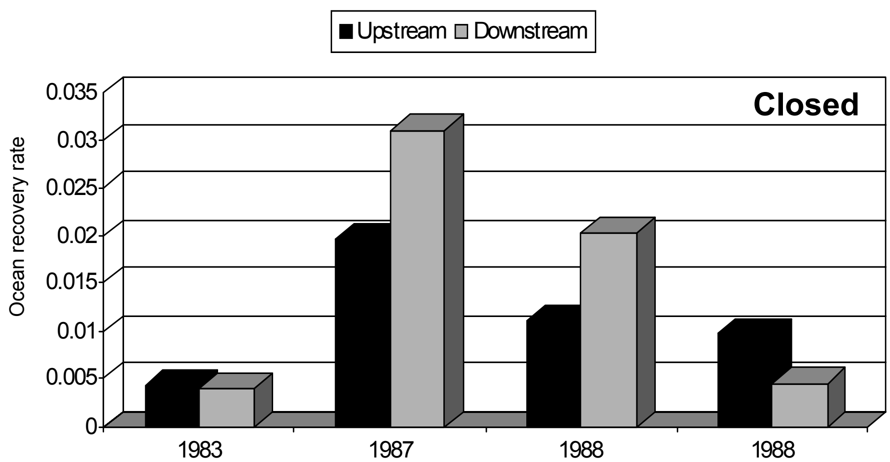


Figure 20 Recovery rates in the ocean fishery for CWT smolts released in the Sacramento River upstream (Courtland) and downstream (Ryde) of the Delta Cross Channel with the gates open and closed

The hypothesis that diversion into the Central Delta reduces juvenile salmon survival is further supported by the results of coded wire tagged, fall-run groups released into Georgiana Slough and in the main-stem Sacramento River at Ryde. The smolt survival indices and ocean recovery rates obtained from the two release locations indicated that fall run smolts survived at a significantly higher rate when released at Ryde rather than into Georgiana Slough) (Figure 21). (Paired t-tests were done for smolt survival indices (t = 3.14, n = 7, P = 0.019) and ocean recovery rates (t = 4.19, n = 7, P = 0.005).

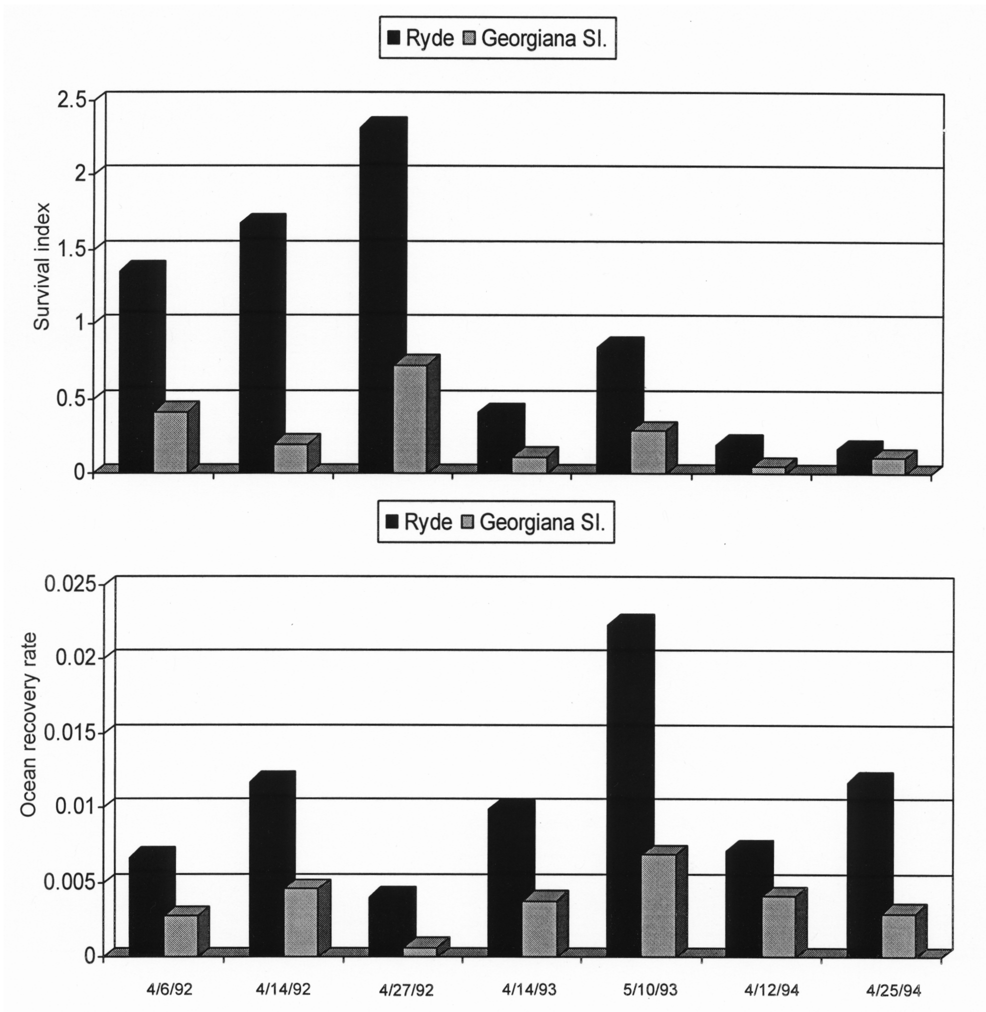


Figure 21 Survival indices to Chipps Island and ocean recovery rates for CWT fall-run smolts released at Ryde and in Georgiana Slough between 1992 and 1994

Between 1993 and 1998, studies using late-fall run juveniles were conducted to determine if survival also was higher for CWT late-fall yearlings released at Ryde than for those released into Georgiana Slough. Late-fall are larger and migrate through the Delta during the winter months when water temperatures are cooler. Despite the cooler temperatures and larger size of the fish relative to fall run, the results with late-fall yearlings were similar to those obtained with fall run smolts. Results indicated that the survival indices to Chipps Island and ocean recovery rates were significantly greater for fish released at Ryde than for those released into Georgiana Slough (Figure 22). Paired t-tests were done for smolt survival indices ($t = 3.60$, $n = 6$, $P = 0.015$) and ocean recovery rates ($t = 3.16$, $n = 4$, $P = 0.050$). Although the ratios between the groups released at Ryde versus those released into Georgiana Slough were similar for the fall and late-fall experiments, it is likely that true survival was less for the fall run groups which were smaller at release and experienced higher water temperatures. These data infer that once fish are diverted into the Central Delta via Georgiana Slough, high relative mortality occurs even for winter run juveniles migrating through the Delta in the late fall and winter months—a period when environmental conditions should be less stressful.

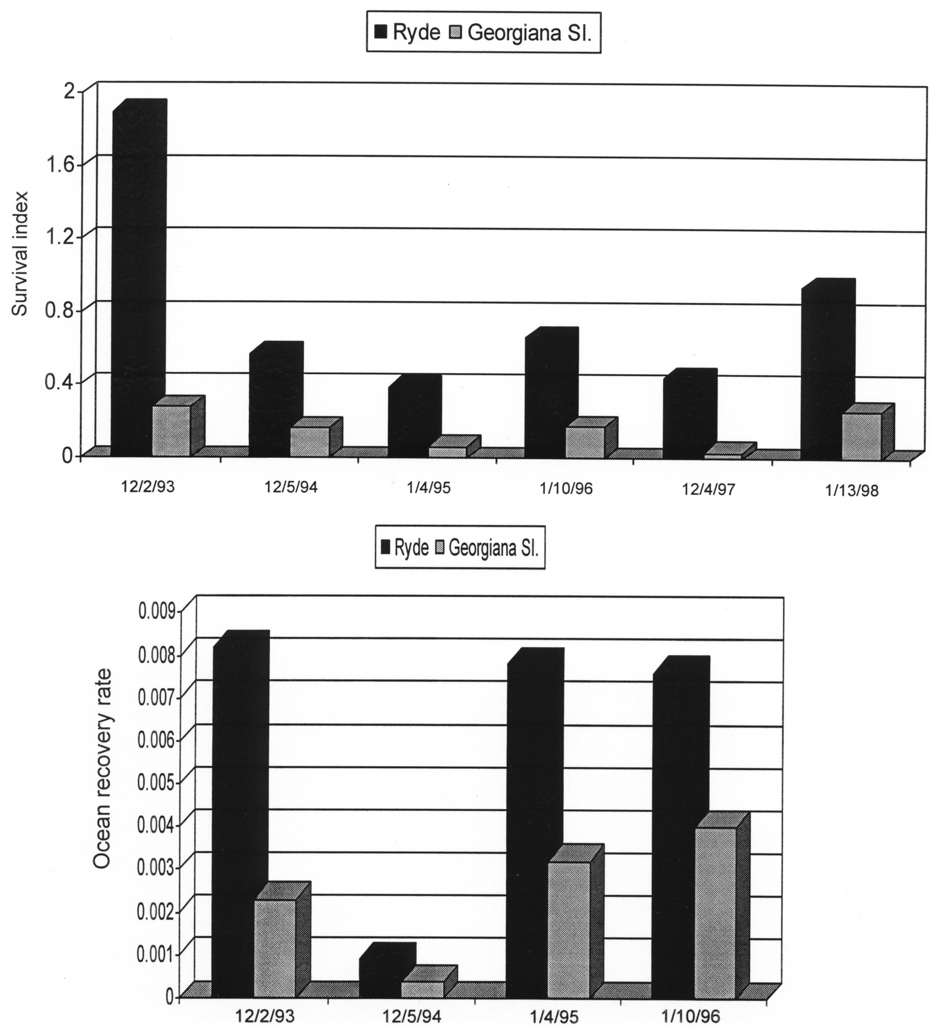


Figure 22 Survival indices to Chipps Island and ocean recovery rates for CWT late-fall-run juveniles released at Ryde/Isleton and in Georgiana Slough

Results from survival studies conducted to determine the relative vulnerability of juvenile salmon to project exports seem consistent with our hypothesis that diversion into the Central Delta is detrimental for juvenile salmon. Coded wire tagged smolts released in the North Delta (at Isleton or Ryde) appeared to have survived at higher rate than those released in the Central or South Delta (at the mouth, North and South Forks of the Mokelumne River and Lower Old River) in the drier years of 1985 and 1986 (Figure 23). This result is similar to that observed with fry released in the Central Delta relative to those released in the North Delta in the drier years.

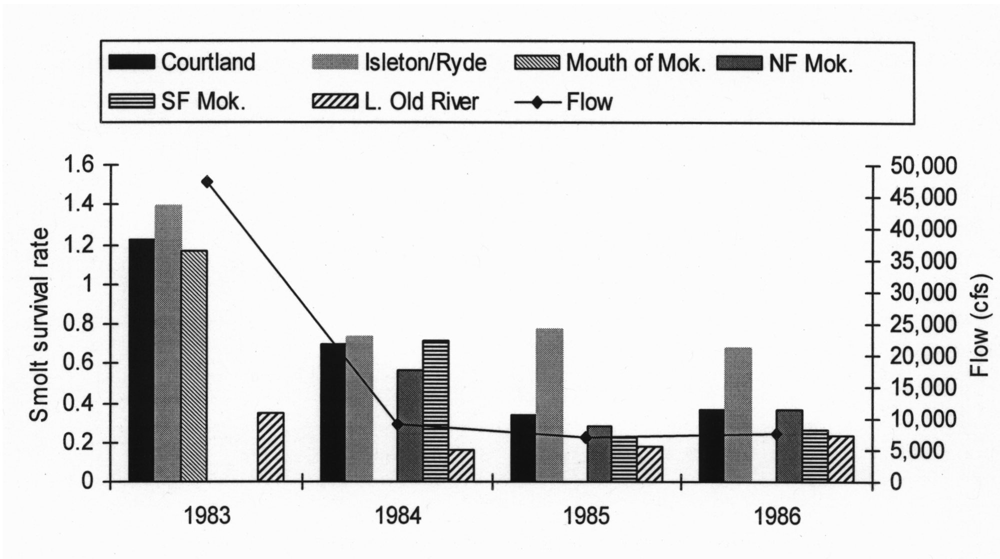


Figure 23 Survival indices of CWT smolts released at various sites in the Delta and mean Sacramento River flow (cfs) at Rio Vista. Flow at Rio Vista was the average during the recovery period of the Courtland releases at Chipps Island.

Although, we have found that diversion into the Central Delta increases juvenile salmon mortality, we have not been able to clearly separate the effects of flow and temperature from diversion impacts. The fact that relative mortality in the Central Delta appears to increase in the drier years, would indicate that there are combined effects. Two separate and independent models constructed using these coded wire tag data have found that temperature is likely the most important factor to fall run smolt survival in the Delta (Newman and Rice 1997; Kjelson and others 1989). Diversion into the central Delta via the Delta Cross Channel gates was also considered important in these models. Sacramento River flow was considered important in the Newman and Rice model (Newman and Rice 1997), but so was salinity (which was inversely correlated to Sacramento River flow) making interpretation difficult. In the Kjelson and others (1989) model Sacramento River flow was tied to the percent of water diverted into the Central Delta.

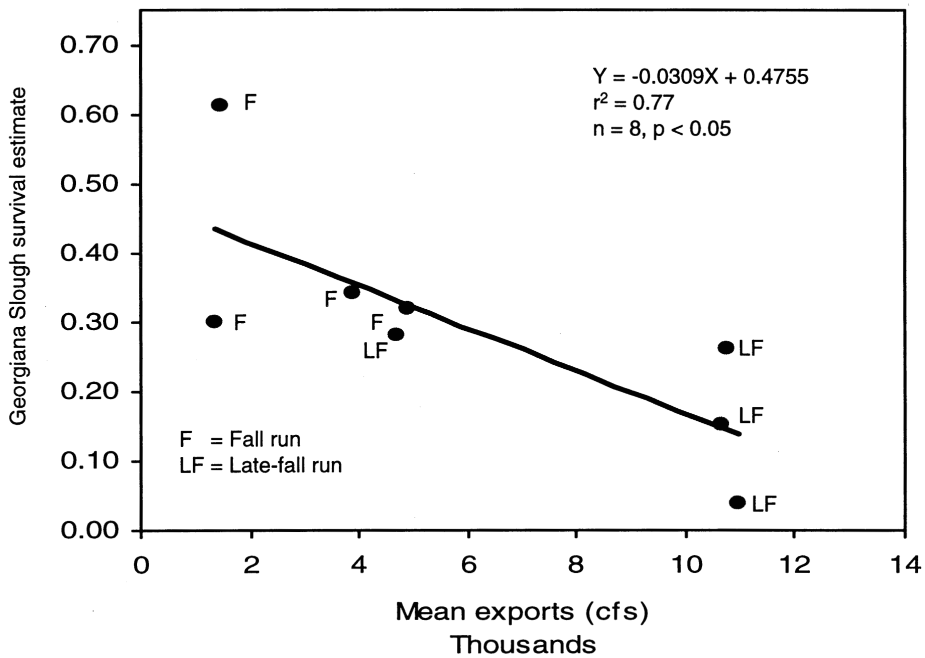


Figure 24 Estimate of survival to Chipps Island for CWT fall-run smolts and late-fall-run yearlings released into Georgiana Slough relative to those released at Ryde with the cross channel gates closed versus combined CVP+SWP exports from release date to 14 days later (fall run) or 17 days later (late-fall run).

Why survival in the Central Delta is lower than that on the main-stem Sacramento River has been hypothesized to be related to the amount of net lower San Joaquin river flow (QWEST), exports or just the longer route to the western Delta of smolts migrating through the central Delta. The survival index to Chipps Island, of fall run smolts and late-fall run yearlings released into Georgiana Slough, does not appear to be related to QWEST. However, the estimate of survival of the Georgiana Slough groups relative to the Ryde groups, for both the fall and late-fall run groups released when the cross channel gates were closed, may be related to exports, although there were large outliers in the relationship which transforming failed to resolve (Figure 24) (r^2= 0.77, $P <$ 0.05). A longer route through the Delta would expose the fish to various mortality factors for a longer period of time. However, the difference in distance, assuming the most direct routes for both groups, is only 37% greater for the Georgiana Slough group (White 1998). The Ryde groups survived between 1.5 and 22 times that observed for the Georgiana Slough groups (Table 3). Differences of between 2 and 7 times are observed in the ocean recovery rate data but some of the most recent releases have not yet been recovered in the ocean fishery (Table 4). These data would infer that the increased distance alone would not account for the differences in survival between the two groups, and exports may contribute, at least in part, to the observed differences.

Table 3 Survival indices to Chipps Island for fall-run smolts and late-fall-run yearlings released at Ryde and Georgiana Slough between 1992 and 1998 and the ratio of survival between the two paired groups

Date	*Ryde*	*Georgiana Slough*	*Ryde:Georgiana Slough ratio*
Fall run			
06 Apr 1992	1.36	0.41	3.3
14 Apr 1992	2.15[a]	0.71	3.0
27 Apr 1992	1.67	0.20	8.4
14 Apr 1993	0.41	0.13	3.2
10 May 1993	0.86	0.29	3.0
12 Apr 1994	0.20	0.05	3.7
25 Apr 1994	0.18	0.12	1.5
Mean			3.7
Late-fall run			
02 Dec 1993	1.91	0.28	6.8
05 Dec 1994	0.58	0.16	3.6
04 Jan 1995	0.39	0.06	6.5
10 Jan 1996	0.66	0.17	3.9
04 Dec 1997	0.67	0.03	22.3
13 Jan 1998	0.94	0.26	3.6
Mean			7.8

[a] The survival index and ocean recovery rate for the 1992 release made at Ryde has been corrected to account for 10,500 marked fish inadvertently released at Georgiana Slough instead of Ryde.

Table 4 Ocean recovery rates for fall-run and late-fall-run yearlings released at Ryde and Georgiana Slough between 1992 and 1996 and the ratio of survival between the two paired groups

Date	*Ryde*	*Georgiana Slough*	*Ryde:Georgiana Slough ratio*
Fall run			
06 Apr 1992	0.0067	0.0028	2.4
14 Apr 1992	0.0107[a]	0.0046	2.3[a]
27 Apr 1992	0.0041	0.0006	6.8
14 Apr 1993	0.0099	0.0039	2.5
10 May 1993	0.0224	0.0069	3.2
12 Apr 1994	0.0074	0.0042	1.8
25 Apr 1994	0.0118	0.0030	3.9
Mean			3.3
Late-fall run			
02 Dec 1993	0.0082[b]	0.0023	3.6
05 Dec 1994	0.0009	0.0004	2.3
04 Jan 1995	0.0078	0.0033	2.4
10 Jan 1996	0.0076	0.0040	1.9
Mean			2.6

[a] The survival index and ocean recovery rate for the 1992 release made at Ryde has been corrected to account for 10,500 marked fish inadvertently released at Georgiana Slough instead of Ryde.
[b] Actual release made at Isleton, about five miles downstream of Ryde.

Impacts of Migration Through Upper Old River and the Use of a Barrier in Upper Old River. Studies using marked fish released into upper Old River and on the San Joaquin River at Dos Reis found that smolts survived at a higher rate if they migrated to Chipps Island via the main-stem San Joaquin River instead of through upper Old River. Inter-annual survival rates at these two locations were highly variable and a significant difference was not found. Although not statistically significant, the survival difference is shown using both survival indices to Chipps Island and ocean recovery rates (Figure 25), suggesting that any wild smolts diverted into upper Old River have greater mortality than those migrating down the main-stem San Joaquin River.

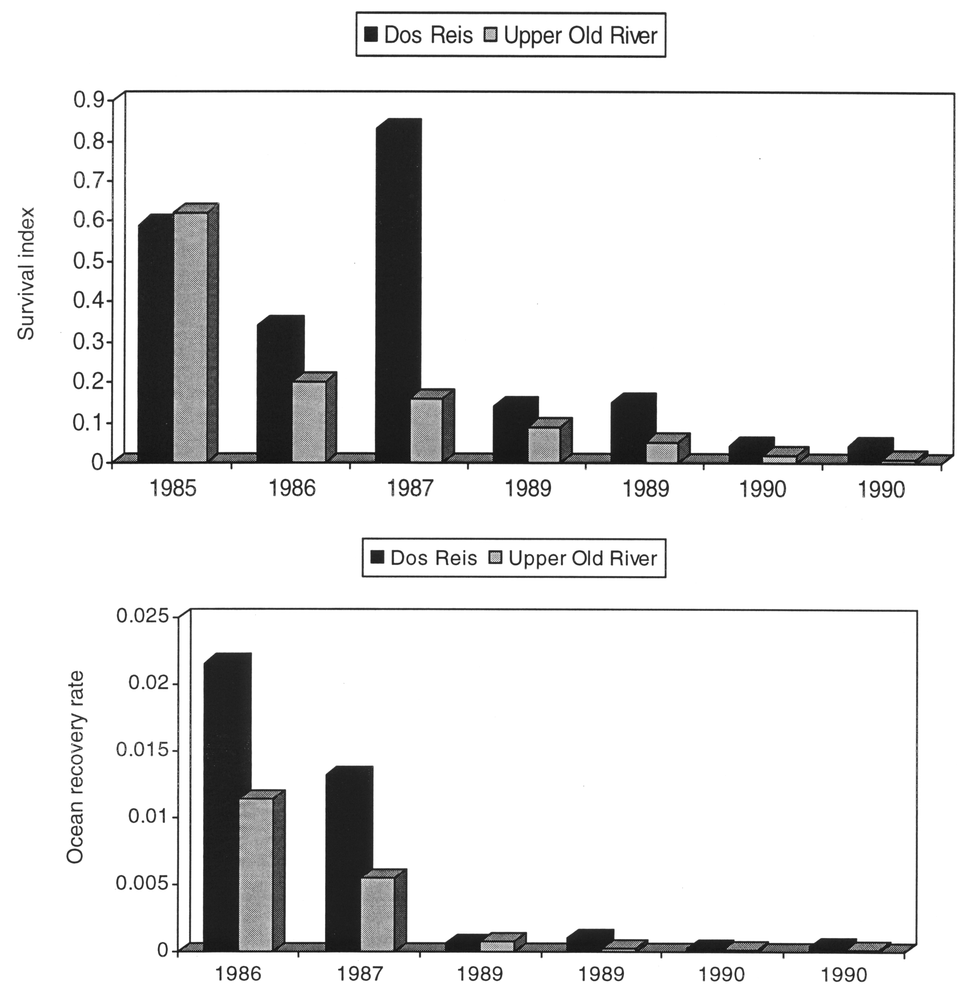


Figure 25 Smolt survival indices and ocean recovery rates of smolts released at Dos Reis on the mainstem San Joaquin River and into upper Old River. Ocean recovery rates are not available for spray-dyed smolts released in 1985.

Table 5 Results of studies comparing survival indices of Feather River Hatchery CWT juvenile chinook salmon from Mossdale to Chipps Island before and after the barrier at Old River was constructed in 1992

Date	*Water temperature (°F)*	*Survival*
Before barrier was constructed		
07 April 1992	64	0.17
13 April 1992	63	0.12
Mean	—	0.15
After barrier was constructed		
24 April 1992	69	0.08
04 May 1992	71	0.01
12 May 1992	72	0.02
Mean	—	0.04

Table 6 Results of studies comparing survival indices of Feather River Hatchery CWT juvenile chinook salmon from Mossdale to Chipps Island before and after the barrier at Old River was constructed in 1994

Date	*Water temperature (°F)*	*Survival*
Before barrier was constructed		
11 April 1994	63	0
After barrier was constructed		
26 April 1994	60	0.04
02 May 1994	66	0
09 May 1994	68	0.02
Mean	—	0.02

In 1992 and 1994, studies were conducted to evaluate the benefits to smolt survival of a full temporary rock barrier at the head of Old River. The study design included releasing CWT groups at Mossdale with and without the barrier in place. Due to logistical considerations, the without barrier scenario was the first experimental condition tested. In 1992, results showed survival indices to be less with the barrier in place than without counter to our hypothesis and earlier information. It is likely that the higher temperatures which occurred in the later part of the experimental period during the time the barrier was in place reduced the survival such that the benefits of the barrier were not observed (Table 5) (DWR 1992). Results in 1994 showed that smolt survival indices for all releases were extremely low and differences between the barrier-in and barrier-out groups were not large (Table 6) (DWR 1995). Neither the 1992 nor 1994 testing was adequate to confirm benefits to smolt survival of a barrier in upper Old River.

Table 7 Release temperatures, average fish size at release, average flow at Vernalis, average Delta exports, and survival indices for Delta CWT releases in 1993, 1995, and 1996[a]

Release date	*Release location*	*Temperature at release (°F)*	*Average size at release (mm FL)*	*Average flow at Vernalis (cfs)*	*Average Delta exports (cfs)*	*Survival index*
6 Apr 1993	Mossdale	63	59	3,293	6,968	0.04
28 Apr 1993	Mossdale	64	71	4,598	1,518	0.07
4 May 1993	Mossdale	61	72	4,349	1,516	0.07
12 May 1993	Mossdale	65	75	3,167	1,533	0.07
17 Apr 1995	Mossdale	57	70	20,558	3,915	0.22
17 Apr 1995	Dos Reis	57	70	20,698	3,924	0.15
5 May 1995	Mossdale	62	75–76	22,772	4,527	0.12
5 May 1995	Dos Reis	63	76	22,397	5,194	0.39
17 May 1995	Mossdale	63	76–79	23,269	4,700	0.07
17 May 1995	Dos Reis	65	77	23,012	4,993	0.16
15 April 1996	Mossdale	59.5	78	6,613	1,687	0.02
30 April 1996	Mossdale	64	81	6,296	1,571	0.01
1 May 1996	Dos Reis	63	83–84	7,714	1,566	0.02

[a] Average flows at Sacramento and Vernalis and average export values are from dayflow. Average flows at Vernalis are from date of release to last day of recovery, or for 14 days after release if no recoveries were made at Chipps Island (survival = zero). Average exports are for 14 days after release. In 1993, they were from release date to last recovery date at Chipps Island. All releases are from Feather River Hatchery stock.

Survival indices were low in 1993 and 1996, and somewhat higher in 1995, for smolts released at Mossdale, without a barrier at the head of upper Old River. Survival indices to Chipps Island ranged between 0.01 and 0.07 in 1993 and 1996 and between 0.07 to 0.22 in 1995. Complementary releases made at Dos Reis in 1995 and 1996 to estimate loss through Old River, indicated that survival was generally higher at Dos Reis than for releases made at Mossdale, suggesting that even in the higher flow years diversion into upper Old River reduces survival (Table 7).

In 1997 all releases at Mossdale were made with the barrier in place, to allow multiple measurements of survival to be generated with the barrier in place. Two 48-inch culverts included in the barrier in 1997 allowed approximately 300 cfs of water to flow from the San Joaquin River into upper Old River. Releases made at Dos Reis, relative to those released at Mossdale, were designed to evaluate the effects on smolt survival of the culverts in the barrier.

Table 8 Release temperatures, average fish size at release, average flow at Vernalis, average Delta exports, and survival indices for Delta CWT releases in 1997 with the head of Old River barrier in place[a]

Release date	*Release location*	*Temperature at release (°F)*	*Average size at release (mm FL)*	*Average flow at Vernalis (cfs)*	*Average Delta exports (cfs)*	*Survival Index*
28 Apr 1997	Mossdale	61	100	5,287	2,353	0.19
29 Apr 1997	Dos Reis	60	97	5,286	2,287	0.19

[a] Average flows at Vernalis and average export values are from dayflow. Average flows at Vernalis are from date of release to last day of recovery, or for 14 days after release if no recoveries were made at Chipps Island (survival = zero). Average exports are for 14 days after release. All releases are from Feather River Hatchery.

Survival indices to Chipps Island of the Feather River smolts released at Dos Reis and Mossdale in 1997 were similar indicating that no difference in survival attributable to the culverts was detected (Table 8). This would suggest that the impact of the culverts was minimal to smolts passing between Mossdale and Dos Reis.

Several pieces of evidence support our conclusion that the barrier improved smolt survival through the Delta in 1997. First, the similarity between the Mossdale and Dos Reis groups provides evidence that the barrier improved survival in 1997. Without a barrier we would have expected the Mossdale group to survive at a lower rate than those released at Dos Reis. Second, the smolt survival index, to Chipps Island for smolts released at Mossdale in 1997, was relatively high compared to past releases made at Mossdale since 1992 (Table 9). Third, the survival index to Chipps Island from smolts released at Mossdale was higher relative to past years of smolts released in the San Joaquin tributaries. In past years, survival of fish released at Mossdale was similar to that observed for fish released in the tributaries. For instance in 1996, the survival indices, to Chipps Island, of smolts released at Mossdale was 0.01 and 0.02, whereas for releases made on the upper Merced and Tuolumne it was 0.01 and 0.04, respectively - the same general magnitude (Table 9). Similarly in 1995, the survival index of smolts released at Mossdale was 0.22, and the groups released in the upper reaches of the tributaries survived at a rate of 0.15 and 0.25. Again, in the same general magnitude. In contrast, in 1997 the survival index from CWT fish released at Mossdale was 0.19 and the survival indices for upper reaches of the tributaries were 0.04, indicating that survival through the South Delta was higher relative to that in the tributaries in 1997, when the barrier was in place. Fourth, the survival index from the release made at Mossdale was closer to that of smolts released at Sacramento in 1997 than it had been in previous years. All of these data support our conclusion that the barrier improved survival in 1997.

Although it seems probable that the barrier increased survival in 1997, survival indices in the San Joaquin Delta still appeared low relative to earlier experiments conducted in 1985 and 1986 (Table 9). The use of non-basin, hatchery fish (those from FRH) or the affect of differences in water temperature between the hatchery truck and release site were hypothesized as possible causes. In both 1996 and 1997, paired releases were made at Dos Reis and Jersey Point with smolts from both FRH and MRFF to assess the potential affect of different stocks on the results of past experiments. Results showed that the survival estimate to Chipps Island, of the Dos Reis group relative to the Jersey Point group, was higher for the MRFF group in both 1996 and 1997 (Table 10). In 1997, smolts from FRH were significantly larger (average 88 mm fork length) and heavier than Merced River stock (average 74 mm fork length). However by standardizing survival, bias associated with recapture efficiency of the different sized fish between stocks should be factored out as long as sizes within a stock were similar, which they were in this case. Results from physiological tests conducted in 1996 and 1997, on subsets of fish (approximately 30) from paired groups released at Dos Reis, indicated there were no physiological reasons for the differences in survival between the two stocks (MRFF and FRH). In 1996 pathologists determined that the Merced stock was at an early infection stage of PKX, a myxosporean parasite, but it should not have affected their survival through the Delta, but could be a factor in adult survival of this stock (True 1996). Physiological tests conducted included those for internal parasites and bacterium and various other analyses (organosomatic analyses, ATPase assay, triglyceride level analyses and stress glucose response analyses). An additional group of 12 was used to assess osmoregulatory ability. In 1996 these tests were made on fish at release, while in 1997 they were made on fish that had been held in live cars for 48 hours.

Table 9 Survival indices of Merced Fish Facility, Feather River Hatchery, and Tuolumne River Fish Facility smolts released in the San Joaquin Delta and tributaries between 1982 and 1997

	Release sites									
Year	*Dos Reis*	*Mossdale w/o HORB*[a]	*Mossdale w/o HORB*[a]	*Upper Old River*	*Upper Merced*[b]	*Lower Merced*[b]	*Upper Tuolumne*[b]	*Lower Tuolumne*[b]	*Upper Stanislaus*[b]	*Lower Stanislaus*[b]
1997	0.19[a] 0.14[b]	–	0.19	–	0.04	0.14	0.04	0.17	–	–
1997		–	–	–	0	0.01	–	–	–	–
1996	0.02[a] 0.09[b]	0.01		–	0.01	0.01	0.04	0.07	–	–
1996		0.02	–	–	–	–	–	–	–	–
1995	0.15[a]	0.22	–	–	0.15	0.20	0.25	0.22	–	–
1995	0.39[a]	0.12	–	–	–	–	–	–	–	–
1995	0.16[a]	0.07	–	–	–	–	–	–	–	–
1994	–	0.00	0.02	–	0.06	0.02	0.03	0.04	–	–
1994	–	–	0.04	–	–	–	–	–	–	–
1994	–	–	0	–	–	–	–	–	–	–
1993	–	0.04	–	–	–	–	–	–	–	–
1993	–	0.07	–	–	–	–	–	–	–	–
1993	–	0.07	–	–	–	–	–	–	–	–
1993	–	0.07	–	–	–	–	–	–	–	–
1992	–	0.18	0.08	–	–	–	–	–	–	–
1992	–	0.12	0.01	–	–	–	–	–	–	–
1992	–	–	0.02	–	–	–	–	–	–	–
1991	0.16[a]	–	–	–	–	–	–	–	–	–
1990	0.04[a]	–	–	0.02[a]	–	–	0.04	0.01[c]	–	–
1990	0.04[a]		–	0.01[a]	–	–	–	–	–	–
1989	0.14[a]	–	–	0.09[a]	–	0.05	–	–	0.05	0.21
1989	0.15[b]	–	–	0.05[b]	–	–	–	–	–	0
1988	–	–	–	–	–	–	–	–	0.07	0.09
1987	0.83[a,d]	–	–	0.16[b]	–	–	0.05	0.18	–	–
1986	0.34[b]	–	–	0.2[b]	–	–	0.40	0.27	0.34	0.56
1985	0.59[b,e]	–	–	0.62[b]	–	–	–	–	–	–
1984	–	–	–	–	–	–	–	–	–	–
1983	–	–	–	–	–	–	–	–	–	–
1982	0.6[b,f]	–	–	–	0.62	–	–	–	–	–

[a] Stock source: Feather River Hatchery.
[b] Stock source: Merced River Fish Facility.
[c] Stock source: TRFF.
[d] Release temperature of 70 °F.
[e] Spray-dyed fish.
[f] May be biased low due to the lack of sampling at Chipps Island during the first week after release.

Table 10 Ratio of survival indices (absolute smolt survival) of smolts released at Dos Reis and Jersey Point and recovered at Chipps Island using Feather River Hatchery and Merced River Fish Facility stock in 1996 and 1997

Release date	*Hatchery stock*	*Release site*	*Survival index to Chipps Island*	*Absolute smolt survival*
01 May 1996	Feather River	Dos Reis	0.02	0.06
03 May 1996	Feather River	Jersey Point	0.35	
01 May 1996	Merced River FF	Dos Reis	0.10	0.14
03 May 1996	Merced River FF	Jersey Point	0.72	
29 Apr 1997	Feather River	Dos Reis	0.19	0.18
02 May 1997	Feather River	Jersey Point	1.03	
29 Apr 1997	Merced River FF	Dos Reis	0.14	0.27
02 May 1997	Merced River FF	Jersey Point	0.51	
08 May 1997	Merced River FF	Dos Reis	0.12	0.30
12 May 1997	Merced River FF	Jersey Point	0.40	

To address the concern that temperature shock reduced the survival of smolts released in the South Delta, smolts were held in live cars in 1996 and 1997. Approximately 200 fish from the paired Dos Reis releases in 1996 were held in live cages for 48 hours to assess immediate and short term mortality within and between groups. In 1997, this was expanded to include all release sites. Sub-samples of the 200 fish (25) were closely evaluated immediately after each release and after they had been held for 48 hours to assess their condition. Fish were evaluated based on eye condition, body color, fin condition, scale loss and gill color. All fish looked healthy both immediately after release and after 48 hours. Only minor mortality (6 dead fish) was observed, of which most (4) was attributed to one location in one year. Mortality (less than 1%) was observed at the release site for this group, which was released on May 12, 1997, at Jersey Point (Brandes 1996; Brandes and Pierce 1998). Considering that most of the fish were healthy after being held for 48 hours in the live cars, it did not appear that acute temperature shock or any other factor at the release site caused mortality within the holding period. Increased predation as a result of reduced avoidance to predators due to temperature stress or other factors can not be assessed holding fish in live cars.

The Role of Flow on Survival. To separate the role of flow in the San Joaquin River from the impacts of diversion into upper Old River, survival estimates of smolts released at Dos Reis relative to those released at Jersey Point were plotted against river flow at Stockton (Figure 26). The relationship between survival and river flow was statistically significant (r^2 = 0.65, P < 0.01), using stock from both Feather River Hatchery and Merced River Fish Facility. In 1989, a release was made at Dos Reis with Merced River stock. Since there was no corresponding Jersey Point release using Merced stock, a Feather River stock was used instead. Although the intercept changed slightly (0.0581), the slope of the relationship did not change when the data point was deleted from the regression (r^2 = 0.82, P < 0.01). If smolts from Dos Reis survive at a higher rate because of increased flows at Stockton, the barrier may have served as the mechanism to increase the flows. It could be that survival is improved via the barrier because of the shorter migration path, but also because it increases the flows down the main-stem San Joaquin River.

The relative importance of flow and the barrier to smolt survival through the Delta between Mossdale and Jersey Point is shown in Figure 27. The rock barrier cannot presently be installed at Vernalis flows of greater than 7000 cfs. To put the improvement in survival resulting from the barrier in perspective, survival indices were compared between 1996 and 1995 and 1996 and 1997. In 1995, the high flows, without a barrier, increased survival by about 16 times that of 1996. The barrier in 1997 improved the 1996 survival index about 4.5 times. The barrier improves survival at flows of less than 7000 cfs but flows greater than 10,000 cfs appear to improve survival even further (Figure 27).

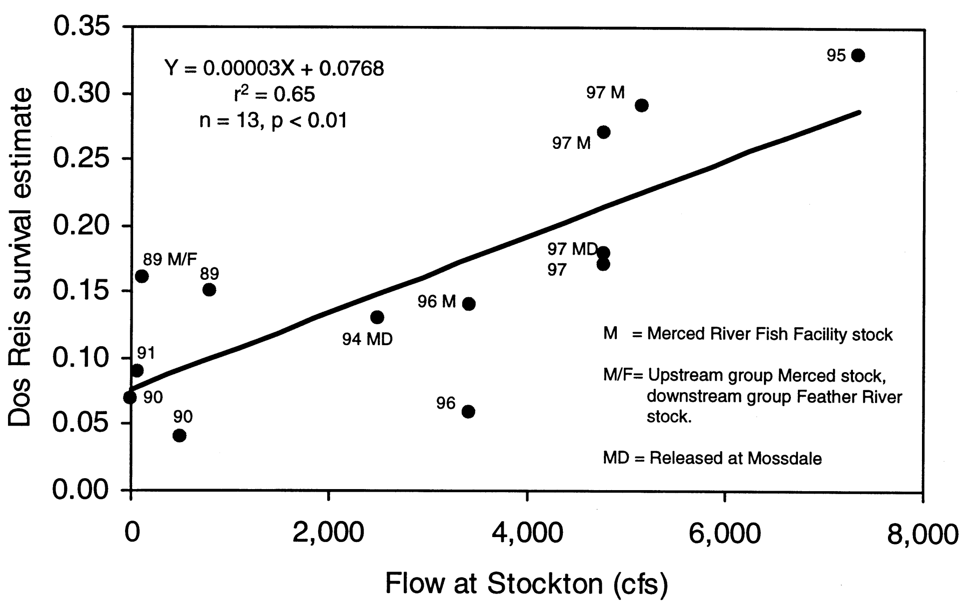


Figure 26 Estimate of survival between Dos Reis and Jersey Point or Mossdale and Jersey Point with the barrier in place using CWT smolts from Feather River Hatchery and Merced River Fish Facility

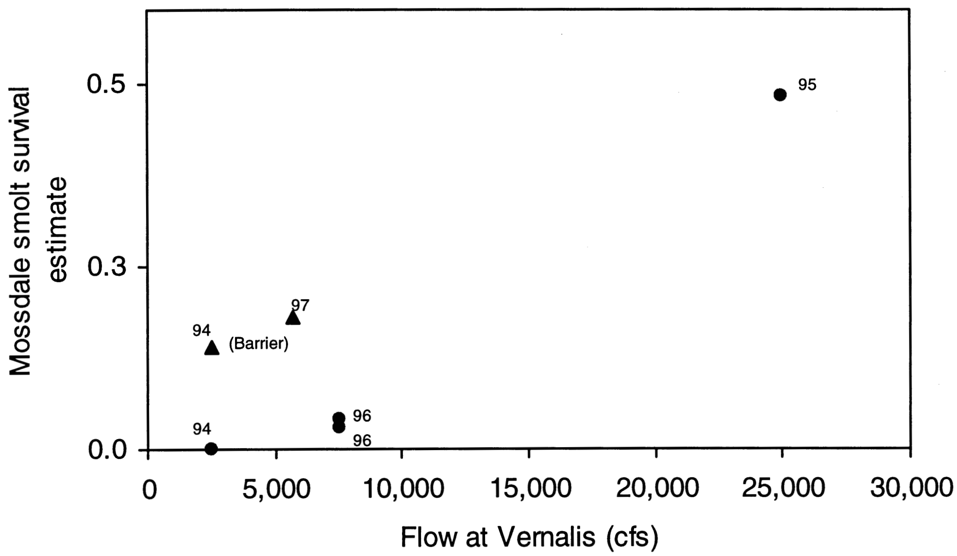


Figure 27 Absolute smolt survival for smolts released at Mossdale in relation to those released at Jersey Point versus flow (cfs) at Vernalis

The Role of Exports and Direct Entrainment on Survival. To determine if exports influenced the survival of smolts in the San Joaquin Delta, experiments were conducted in 1989, 1990 and 1991 at medium/high and low export levels. Results were mixed showing in 1989 and 1990 that survival estimates between Dos Reis and Jersey Point were higher with higher exports whereas in 1991 between Stockton and the mouth of the Mokelumne River (Tables 11 and 12) survival was shown to be lower (0.008 compared to 0.15) when exports were higher. One potential bias in the 1989 and 1990 data is that as mentioned earlier, smolts released at Dos Reis in 1989 were from the Merced River Fish Facility while those released at Jersey Point were from Feather River hatchery. Using different stocks to estimate smolt survival between two locations may introduce bias. In addition, results in 1989 and 1990 also showed that survival indices of the upper Old River groups relative to the Jersey Point groups were also higher during the higher export period, but overall still about half that of the survival of smolts released at Dos Reis (Table 11).

Table 11 Survival indices of smolts released at Dos Reis on the mainstem San Joaquin River, upper Old River, and Jersey Point based on Chipps Island recoveries[a]

Year	*Flow at Vernalis (cfs)[b]*	*CVP + SWP exports (cfs)[c]*	*Dos Reis*	*Upper Old River*	*Jersey Pt.*	*Dos Reis-Jersey Pt. ratio*	*Upper Old River:Jersey Pt. ratio*
1989	2,274	10,247	0.14	0.09	0.88	0.16	0.10
1989	2,289	1,797	0.14	0.05	0.96	0.14	0.05
1990	1,290	9,618	0.04	0.02	0.61	0.06	0.03
1990	1,665	2,462	0.04	0.01	1.05	0.04	0.01

[a] The ratio of survival indices between Dos Reis to Jersey Point and upper Old River to Jersey Point are included to compare absolute survival between years.

[b] Flows at Vernalis are ten-day averages in cubic feet per second after the Dos Reis groups were released.

[c] Exports are daily averages in cubic feet per second five days after release of upper Old River groups.

Contrary to the mixed results between survival and exports, direct entrainment losses at the fish facilities has been identified as a cause of juvenile salmon mortality in the Delta. Kjelson (1981) reported that records of salmon observed in salvage and respective spring export rates between 1959 and 1967 and 1968 to 1979 indicated that as exports increased more downstream migrating salmon are observed in the salvage. In USFWS Exhibit 31 (1987), it was reported that on average only 0.36% of the CWT smolts, released in the Sacramento River (above the Walnut Grove diversion) or in the forks of the Mokelumne River, were estimated to have been salvaged (expanded salvage) at the export facilities in the south Delta. These percentages are small, even when further expanded for screen efficiency and predation losses in Clifton Court Forebay, relative to the indirect mortality in the Delta that would occur to juveniles drawn off their normal migration path and exposed to other mortality factors for a longer period of time.

Table 12 Survival indices for CWT chinook released at various locations along the San Joaquin and Mokelumne rivers in the Delta in April and May 1991[a]

Month (mean exports) and release site	*River mile*	*Temp. (°F)*	*Survival index to Chipps Island*	*Survival rate per mile [a]*
April (4,283 cfs)				
Dos Reis	50	60	0.16	
Dos Reis to Stockton				0.06
Stockton	39	59	0.25	
Stockton to Empire Tract				0.05
Empire Tract	29	61	0.54	
Empire Tract to mouth of Mokelumne River				0.03
Mouth of the Mokelumne River	19	61	1.56	
Stockton to the mouth of Mokelumne River				0.008
Mouth of the Mokelumne River to Jersey Point				0.13
Jersey Point	12	63	1.70	
May (2,613 cfs)				
Stockton		65	0.19	
Stockton to the mouth of Mokelumne River				0.015
Mouth of the Mokelumne River		64.5	0.640	
Mouth of the Mokelumne River to Jersey Point				0.05
Jersey Point		61	1.69	

[a] Survival rate per mile and river miles to Chipps Island also are included for the reaches between Stockton and the mouth of the Mokelumne River and between the mouth of the Mokelumne River and Jersey Point.

Recoveries at the fish salvage facilities were much greater from releases made in the South Delta than in the North Delta. Many marked fish were observed at the fish facilities when they were released into upper Old River (average 19%). Smolts released at Dos Reis on the main-stem San Joaquin River had a lower salvage rate (averaged 3%) (USFWS 1990). These differences in salvage may, in part, account for the lower survival observed for the upper Old River group.

The number of marked fish recovered at the fish facilities from releases made in the San Joaquin Delta seems to be related to release location, whether or not there is a barrier in place and the rate of exports. In 1991, the greatest number of fish recovered at the fish facilities was from Dos Reis, Stockton and Empire Tract groups. The fewest recoveries were from the Mokelumne release groups and those released at Jersey Point (Table 13) (USFWS 1992b). Recoveries at the fish facilities from releases made at Mossdale were greatest when there was no barrier at the head of Old River (Table 14). This is further illustrated by the greater recoveries at both the CVP and SWP of smolts released at Mossdale relative to those released at Dos Reis (Table 15). In 1997, the number of expanded recoveries from two Dos Reis groups and the Mossdale group, were similar with the barrier in place (Table 16).

Table 13 Expanded fish facility recoveries during high (April) and low (May) export levels during spring 1991

				Expanded fish facility recoveries	
Release location	*Release date*	*Number released*	*Mean daily exports*[a]	*CVP*	*SWP*
Dos Reis	15 April	102,999	4,283	5,472	2,526
Stockton	16 April	99,341	4,283	338	2,635
Empire Tract	17 April	95,602	4,283	131	1,401
L. Mokelumne	18 April	47,289	4,283	0	276
Jersey Point	19 April	52,139	4,283	20	274
Stockton	06 May	99,820	2,613	52	64
L. Mokelumne	09 May	45,706	2,613	22	13
Jersey Point	13 May	49,184	2,613	6	0

[a] Mean daily exports (CVP and SWP combined) for April (16 April to 6 May) and May (6 May to 30 May) are the mean between the release date and final capture.

Table 14 Expanded SWP and CVP fish facility recoveries for fish released at Mossdale with and without the head of Old River barrier in place in 1992 and 1994

			Expanded fish facility recoveries	
Release date	*Barrier status*	*Number released*	*SWP*	*CVP*
07 April 1992	No barrier	107,103	71	5,380
13 April 1992	No barrier	103,712	106	3,385
11 April 1994	No barrier	51,084	100	648
04 May 1992	Barrier	99,717	8	28
12 May 1992	Barrier	105,385	6	0
26 April 1994	Barrier	50,259	0	0
02 May 1994	Barrier	51,632	24	12
09 May 1994	Barrier	53,880	13	0

Table 15 Expanded SWP and CVP fish facility recoveries for fish released at Dos Reis and Mossdale without the upper Old River barrier in place in 1995 and 1996

			Expanded fish facility recoveries	
Release location	*Release date*	*Number released*	*SWP*	*CVP*
Mossdale	17 April 1995	100,969	36	2,732
Dos Reis	17 April 1995	50,848	0	1
Mossdale	05 May 1995	102,562	74	1,859
Dos Reis	05 May 1995	52,097	0	0
Mossdale	17 May 1995	104,125	128	1,452
Dos Reis	17 May 1995	51,665	0	24
Mossdale	30 April 1996	99,656	24	110
Dos Reis	01 May 1996	206,780	0	0

In 1989 and 1990 expanded recoveries at the fish salvage facilities for both CWT groups released in upper Old River and Dos Reis during the high export were greater than those during the low export experiments (USFWS 1990) (Table 17). In 1991, it was observed that there were 25 times more marked fish recovered at the fish facilities from the Stockton group during the period of higher exports, than the releases made during the lower export period (USFWS 1992b) (Table 13). These pieces of information indicate that direct mortality is higher when exports are higher.

Table 16 Expanded SWP and CVP fish facility recoveries for fish released at Dos Reis and Mossdale in 1997 with the upper Old River barrier in place using Feather River Hatchery or Merced River Fish Facility stock

				Expanded fish facility recoveries	
Release location	*Stock*	*Release date*	*Number released*	*SWP*	*CVP*
Mossdale	Feather River Hatchery	28 April	48,774	34	204
Dos Reis	Feather River Hatchery	29 April	49,830	31	96
Dos Reis	Merced River Fish Facility	29 April	102,480	130	288

Table 17 Expanded SWP and CVP fish facility recoveries for smolts released in the San Joaquin River at Dos Reis and into upper Old River in 1989 and 1990 and mean daily exports five days after release of the upper Old River group

				Expanded fish facility recoveries	
Release location	*Release date*	*CVP + SWP mean daily exports (cfs)*	*Number released*	*SWP*	*CVP*
Dos Reis	20 Apr 1989	10,247	52,962	2,286	428
Upper Old River	21 Apr 1989	10,247	51,972	2,916	658
Dos Reis	02 May 1989	1,797	75,983	344	84
Upper Old River	03 May 1989	1,797	74,309	215	1,256
Dos Reis	16 Apr 1990	9,618	105,742	1,044	722
Upper Old River	17 Apr 1990	9,618	106,269	1,729	948
Dos Reis	02 May 1990	2,462	103,533	96	54
Upper Old River	03 May 1990	2,462	103,595	920	426

Bay Survival. In 1984, 1985 and 1986 smolts were released at Port Chicago in Suisun Bay and in San Francisco Bay near the Golden Gate Bridge to estimate survival through the Bay. The survival indices of marked fish released at Port Chicago and recovered in the midwater trawl at the Golden Gate were highly variable and ranged from 0.75 to 2.39 (Table 18). The extreme tidal fluctuations in San Francisco Bay most likely had a significant effect on the variability of the indices. Survival estimates through the Bay, from Port Chicago to the Golden Gate measured using the differential ocean recovery rates from the two release groups, showed survival ranged between 0.76 and 0.84 in the three years (Table 18). Delta outflows were low, and ranging from 6,690 cfs to 13,507 cfs, and did not appear to effect survival rates through the Bay. Since survival was observed to be relatively high through the Bay in the three years measured, CWT experiments in the Bay were discontinued to free up marked fish for use in the Delta where survival had been shown to be less and the need for information greater.

Table 18 San Francisco Bay (Golden Gate Bridge) tag summary, survival calculations and expanded ocean recoveries

Tag code	*Release site (stock)*	*Release date*	*Number released*	*Size (mm)*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Ocean recoveries*
6-62-51	Port Chicago (FRH)	06/02/86	47,995	75	06/05/86	06/18/86	15	0.75	1,382
6-62-52	Fort Baker (FRH)	06/03/86	49,583	73	---	---	0		1,807
6-62-45	Port Chicago (FRH)	05/13/85	48,143	76	05/17/85	05/29/85	22	1.54	465
6-62-44	Fort Baker (FRH)	05/14/85	47,158	N/A	---	---	0		537
6-54-51	Port Chicago (NFH)	07/23/84	50,114	N/A	07/26/84	07/31/84	36	2.39	1,159
6-54-52	Fort Baker (NFH)	07/25/84	48,677	N/A	---	---	0		1,461

Annual Indices of Survival. The survival indices to Chipps Island and survival estimates using differential ocean recovery rates, of smolts released near Sacramento, in the upper Sacramento River and in the San Joaquin tributaries, allows survival through the Delta and upstream to be compared between years.

Survival indices to Chipps Island of hatchery smolts released near Sacramento were compared to those released upstream in Battle Creek in Figure 28. The survival indices to Chipps Island from releases made into Battle Creek would include survival in the Delta as well as that in the upper river. Survival appeared to be relatively high between Battle Creek and Chipps Island in 1984 and 1993. Smolt survival through the Delta appeared highest in 1982 and 1983.

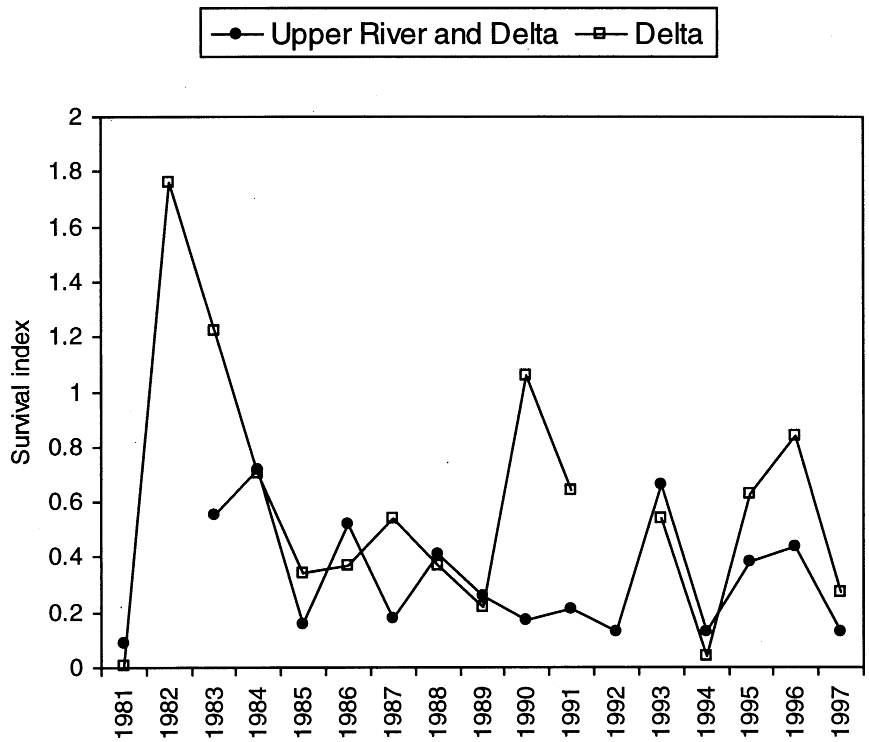


Figure 28 Survival indices of smolts released in the upper river (Battle Creek) and those released in the northern Delta (Courtland, Sacramento, Miller Park, Discovery Park) between 1981 and 1997. No release upstream in 1982 or at Sacramento in 1992.

Survival estimates using ocean recovery rates of smolts released at Battle Creek, relative to those released at Princeton, Knights Landing or Sacramento, were compared to the survival estimates of fish released at Sacramento, relative to those released at Port Chicago/Benicia, to allow upriver survival estimates to be separated from those in the Delta (Figure 29). These data show that survival was greatest upstream in 1987 and 1990, contrary to the conclusions based on Chipps Island survival indices. Variability in the ocean recovery rates or the poor survival of the downstream groups relative to the upstream groups likely account for the ratios of greater than 1.0 observed for several of the groups. Both sets of survival indices/estimates provide a rough estimate of the survival of smolts migrating through the River and Delta over time.

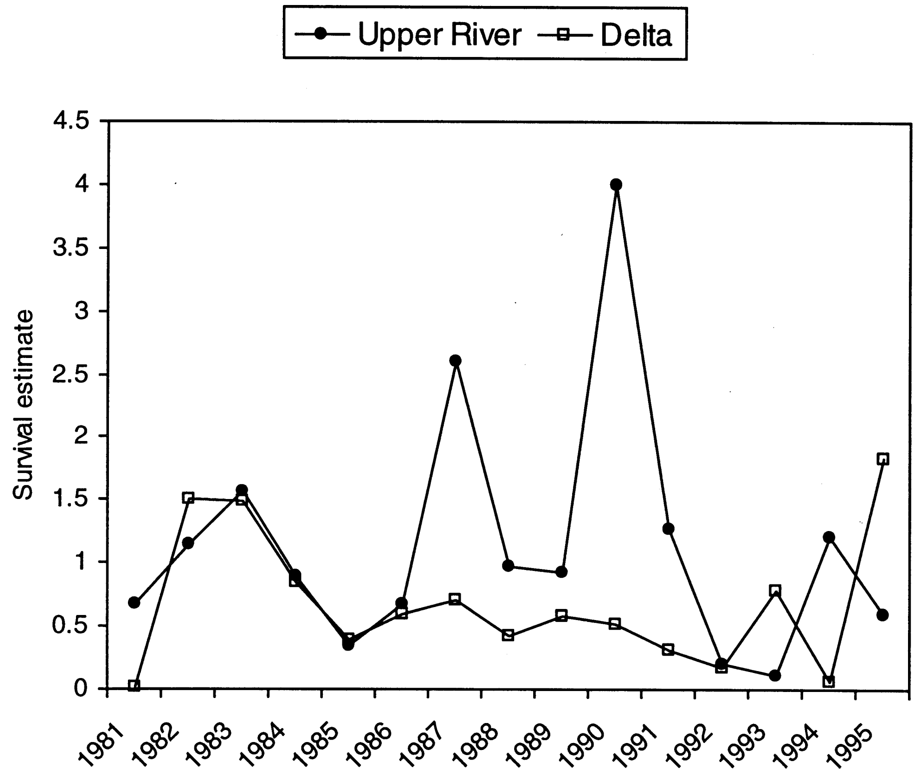


Figure 29 Estimates of survival in the upper river and Delta. A Ryde release was used as a downstream control in 1987 because a release was not made at Port Chicago or Benicia that year. In 1992, Princeton was used for the Delta survival estimate.

Marked releases have been made at sites in the San Joaquin tributaries during many years since 1982 and similar upstream and Delta comparisons were made between years. The survival indices from releases made in the lower reach of the tributaries would provide an index of survival through the lower San Joaquin River and Delta. Survival indices to Chipps Island from releases made in the upstream reaches of the tributaries would include both tributary and Delta survival. In many years (1986, 1988, 1994, 1995, and 1996) survival to Chipps Island from the upper reaches of the tributaries were similar to that from the releases made in the lower reaches indicating that most of the mortality occurred in the San Joaquin River main-stem and Delta (Figure 30). In other years, such as in 1987, 1989 and 1997, survival from releases made in the upper reaches of the tributaries was much less than that in the lower reaches, indicating that mortality in the tributaries was higher relative to that in the Delta and San Joaquin River.

Survival for smolts released in the lower reaches of the San Joaquin River tributaries also can be compared to that for smolts released near Sacramento. Survival is generally, substantially higher for smolts released at Sacramento than for those released in the lower tributaries of the San Joaquin River. Exceptions were in 1986, 1994 and 1997 when survival through the Delta from both basins was similar (Figure 30).

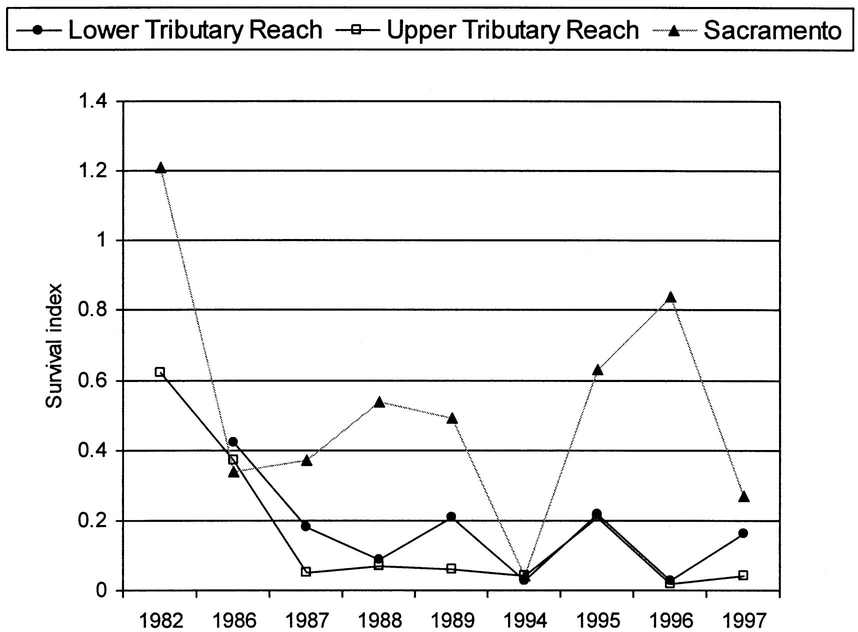


Figure 30 Survival indices for CWT smolts released at sites in the lower (L. Trib.) and upper (U. Trib.) reaches of the San Joaquin tributaries and near Sacramento (Sac.)

Summary and Recommendations

Analyses of the lower river and Delta beach seine data and the trawl data at Sacramento and Chipps Island, indicates that many juveniles enter the Delta as fry in wet years and that overall, juvenile production leaving the Delta is higher in wet years. The increase in juvenile production in wet years could be partially due to survival increases of fry upstream. Increased river flows appeared to increase fry survival upstream, but likely caused a greater proportion of them to migrate to the estuary where fry survival appears lower than upriver in the higher flow years. The survival of marked fry and smolts in the Central Delta appeared lower than in the North Delta, especially in the drier years. Both fry and smolts in the Central Delta may be more vulnerable to exports than those released in the North Delta in the drier years.

Studies using marked smolts in the Sacramento Delta indicated that migration into the Central Delta via the Delta Cross Channel or Georgiana Slough negatively affected the survival of juveniles migrating through the Delta not only in the spring, but in the winter months as well. Migration through upper Old River in the south Delta also appeared to negatively affect the survival of smolts originating from the San Joaquin basin. Direct losses, as indexed using expanded salvage recoveries, due to export pumping were generally low for smolts migrating through the Delta on the Sacramento River. Direct losses were higher for marked fish released in the San Joaquin Delta, with the greatest salvage from smolts released in upper Old River. Salvage also was higher for releases made at the same location when exports were increased. These long-term studies have helped identify actions that could improve juvenile salmon survival through the Delta.

Long-term systematic releases to measure survival through the Delta can be used as the basis for future modeling to further define ways to improve survival. Some models have been developed from CWT data generated from the Sacramento Delta (Newman and Rice 1997; Kjelson and others 1989). Additional models using this data are in the process of being generated. In such a complex system, it will take consistent releases over many years to refine models that will further define the factors important to juvenile salmon abundance and survival and identify additional ways to improve survival through the Delta.

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Notes

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Appendix A:
Unexpanded Fish Facility Recoveries for Coded Wire Half Tagged Fry Released in the Sacramento-San Joaquin Delta

Table A-1 Unexpanded fish facility recoveries for CW1/2T fry released in the Sacramento-San Joaquin Delta[a]

						Unexpanded recoveries	
Year	*Tag code*	*Release site (stock)[b]*	*Release date*	*Number released*	*Size (mm)*	*CVP*	*SWP[c]*
1987	H6-7-7	Below RBDD (CNFH)	13-Mar-87	52,977	52	0	1
1987	B5-4-13	Battle Creek (CNFH)	12-Mar-87	51,075	51	1	0
1986	H6-7-6	Courtland (NFH)	05-Mar-87	48,733	50	4	3
1986	H6-7-5	Below RBDD (CNFH)	19-Mar-86	51,426	50	0	0
1986	H5-7-7	Battle Creek (CNFH)	18-Mar-86	51,371	50	0	0
1986	H6-6-7	Courtland (CNFH)	27-Feb-86	50,961	45	0	0
1986	H6-7-3	Courtland (CNFH)	10-Mar-86	53,831	50	0	0
1986	H6-7-2	Ryde (CNFH)	14-Mar-86	52,635	47	0	0
1986	H6-7-4	Ryde (CNFH)	12-Mar-86	52,748	53	0	0
1985	H6-5-5	Below RBDD (CNFH)	14-Feb-85	49,155	47	0	2
1985	H6-6-5	Below RBDD (CNFH)	14-Mar-85	52,313	48	0	0
1985	H6-5-6	Courtland (CNFH)	19-Feb-85	51,201	48	0	0
1985	H6-6-4	Courtland (CNFH)	07-Mar-85	51,985	46	0	(3) 6
1985	H6-6-1	South Fork Mokelumne (CNFH)	26-Feb-85	50,052	48	2	(2) 1
1985	H6-6-2	North Fork Mokelumne (CNFH)	28-Feb-85	51,145	46	0	(1) 3
1985	H6-5-7	Ryde (CNFH)	21-Feb-85	49,183	47	0	0
1985	H6-6-3	Ryde (CNFH)	05-Mar-85	50,550	47	1	(2) 8
1984	H6-4-4	Below RBDD (CNFH)	01-Mar-84	43,883	45	1	0
1984	H6-5-4	Below RBDD (CNFH)	24-Mar-84	47,855	50	0	0
1984	H6-4-5	Courtland (CNFH)	05-Mar-84	48,460	45	0	0
1984	H6-5-3	Courtland (CNFH)	21-Mar-84	48,157	48	0	0
1984	H6-4-6	Ryde (CNFH)	08-Mar-84	45,465	49	4	0
1984	H6-5-2	Ryde (CNFH)	19-Mar-84	46,767	49	4	0
1984	H6-5-1	South Fork Mokelumne (CNFH)	14-Mar-84	45,036	49	3	0
1984	H6-4-7	North Fork Mokelumne (CNFH)	12-Mar-84	42,165	50	5	0
1983	H6-3-3	Isleton (FRH)	04-Mar-83	45,775	44	0	0
1983	H6-4-2	Isleton (FRH)	29-Mar-83	47,518	49	0	0
1983	H6-3-4	Courtland (FRH)	09-Mar-83	48,541	47	0	0
1983	H6-4-3	Courtland (FRH)	31-Mar-83	48,501	51	0	0
1983	H6-3-5	Mouth of Mokelumne (FRH)	14-Mar-83	45,960	N/A	0	0

[a] In some cases, average size was calculated from number of fish per pound using a conversion table (Source: USFWS 1982, Table I-6).

[b] CNFH = Coleman National Fish Hatchery, FRH = Feather River Hatchery.

[c] Salvage numbers in parentheses have an unknown location (either CVP or SWP).

Table A-1 Unexpanded fish facility recoveries for CW1/2T fry released in the Sacramento-San Joaquin Delta[a] (Continued)

						Unexpanded recoveries	
Year	*Tag code*	*Release site (stock)*[b]	*Release date*	*Number released*	*Size (mm)*	*CVP*	*SWP*[c]
1983	H6-4-1	Mouth of Mokelumne (FRH)	24-Mar-83	47,367	48	0	0
1983	H6-3-6	Lower Old River (FRH)	17-Mar-83	47,677	49	0	0
1983	H6-3-7	Lower Old River (FRH)	22-Mar-83	48,580	48	0	0
1982	H6-2-2	Below RBDD (CNFH)	05-Feb-82	41,753	44	0	0
1982	H6-2-6	Below RBDD (CNFH)	25-Feb-82	43,673	44	0	0
1982	H6-2-3	Isleton (CNFH)	11-Feb-82	43,248	44	0	0
1982	H6-2-7	Isleton (CNFH)	02-Mar-82	40,508	45	0	0
1982	H6-2-4	Mouth of Mokelumne (CNFH)	17-Feb-82	43,849	43	0	0
1982	H6-3-2	Mouth of Mokelumne (CNFH)	10-Mar-82	41,470	44	0	0
1982	H6-2-5	Berkeley (CNFH)	22-Feb-82	40,699	44	0	0
1982	H6-3-1	Berkeley (CNFH)	08-Mar-82	39,321	44	0	0
1981	H6-1-1	Below RBDD (CNFH)	06-Feb-81	35,905	41	0	0
1981	H6-1-5	Below RBDD (CNFH)	28-Feb-81	47,019	40	0	0
1981	H6-1-4	Berkeley (CNFH)	25-Feb-81	49,705	44	0	0
1981	H6-2-1	Berkeley (CNFH)	08-Mar-81	36,901	43	0	, 0
1981	H6-1-3	Mouth of Mokelumne (CNFH)	20-Feb-81	45,193	44	2	0
1981	H6-1-7	Mouth of Mokelumne (CNFH)	06-Mar-81	45,796	43	2	0
1981	H6-1-2	Isleton (CNFH)	12-Feb-81	40,916	45	3	0
1981	H6-1-6	Isleton (CNFH)	04-Mar-81	45,949	43	0	0
1981	H5-2-4	Berkeley (CNFH)		21,939	46	0	0
1981	H5-2-5	Berkeley (CNFH)		20,788	46	0	0
1981		**Total**	**28-Feb-80**	**42,727**			
1981	H5-2-6	Clarksburg (CNFH)		22,121	50	0	0
1981	H5-2-7	Clarksburg (CNFH)		21,624	50	0	0
1981		**Total**	**28-Feb-80**	**43,745**			
1981	H5-3-3	Clarksburg (CNFH)		23,908	46	0	0
1981	H5-3-4	Clarksburg (CNFH)		22,829	44	0	0
1981		**Total**	**31-Mar-80**	**46,737**			

[a] In some cases, average size was calculated from number of fish per pound using a conversion table (Source: USFWS 1982, Table I-6).

[b] CNFH = Coleman National Fish Hatchery, FRH = Feather River Hatchery.

[c] Salvage numbers in parentheses have an unknown location (either CVP or SWP).

Appendix B:
Chipps Island Tag Summary and Survival Calculations for Coded Wire Tagged Fish Groups with Multiple Tag Codes

Table B-1 1997 Sacramento-San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-1-6-2-11	West Sacramento (FRH)		25,641	22-Apr-97	07-May-97	14	0.52	
6-1-6-2-12	West Sacramento (FRH)		25,032	22-Apr-97	08-May-97	9	0.34	
	Total	15-Apr-97	50,673	22-Apr-97	08-May-97	23		0.43
6-1-6-3-2	Mossdale (w/ barrier) (FRH)		23,701	03-May-97	07-May-97	2	0.08	
6-1-6-3-3	Mossdale (w/ barrier) (FRH)		25,073	05-May-97	18-May-97	8	0.31	
	Total	28-Apr-97	48,774	03-May-97	18-May-97	10		0.19
6-1-6-3-4	Dos Reis (FRH)		25,084	06-May-97	11-May-97	7	0.27	
6-1-6-3-5	Dos Reis (FRH)		24,746	06-May-97	11-May-97	3	0.12	
	Total	29-Apr-97	49,830	06-May-97	11-May-97	10		0.19
6-25-45	Dos Reis (MRFF)		49,005	08-May-97	16-May-97	9	0.18	
6-25-46	Dos Reis (MRFF)		53,475	10-May-97	15-May-97	7	0.13	
	Total	29-Apr-97	102,480	08-May-97	16-May-97	16		0.15
6-1-6-2-13	West Sacramento (FRH)		25,829	05-May-97	15-May-97	15	0.55	
6-1-6-2-14	West Sacramento (FRH)		26,315	07-May-97	10-May-97	7	0.25	
	Total	01-May-97	52,144	05-May-97	15-May-97	22		0.40
6-1-6-2-7	Jersey Point (FRH)		24,815	03-May-97	10-May-97	27	1.03	
6-1-6-2-8	Jersey Point (FRH)		25,049	04-May-97	11-May-97	28	1.05	
	Total	02-May-97	49,864	03-May-97	11-May-97	55		1.03
6-1-6-2-9	West Sacramento (FRH)		25,152	---	---	0		
6-1-6-2-10	West Sacramento (FRH)		25,069	21-May-97	21-May-97	1	0.04	
	Total	15-May-97	50,221	21-May-97	21-May-97	1		0.02

[a] FRH = Feather River Hatchery, MRFF = Merced River Fish Facility.

Table B-2 1997 Upper San Joaquin River Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-1-11-5-11	Upper Merced (MRFF)		26,045	06-May-97	06-May-97	1	0.04	
6-1-11-5-12	Upper Merced (MRFF)		27,683	06-May-97	08-May-97	3	0.10	
6-1-11-5-13	Upper Merced (MRFF)		31,930	09-May-97	09-May-97	1	0.03	
6-1-11-6-12	Upper Merced (MRFF)		24,880	---	---	0		
	Total	20-Apr-97	110,538	06-May-97	09-May-97	5		0.04
6-1-11-5-15	Lower Merced (MRFF)		24,398	04-May-97	14-May-97	6	0.23	
6-1-11-6-1	Lower Merced (MRFF)		29,011	04-May-97	09-May-97	3	0.10	
6-1-11-6-2	Lower Merced (MRFF)		25,761	03-May-97	12-May-97	7	0.25	
6-1-11-6-3	Lower Merced (MRFF)		25,317	---	---	0		
	Total	22-Apr-97	104,487	03-May-97	14-May-97	16		0.14
6-1-11-6-7	Upper Tuolumne (MRFF)		31,112	18-May-97	18-May-97	1	0.04	
6-1-11-6-8	Upper Tuolumne (MRFF)		29,947	---	---	0		
6-1-11-6-9	Upper Tuolumne (MRFF)		24,551	24-May-97	24-May-97	1	0.04	
6-1-11-6-10	Upper Tuolumne (MRFF)		7,897	18-May-97	18-May-97	1	0.14	
	Total	22-Apr-97	93,507	18-May-97	24-May-97	3		0.036
6-1-11-6-4	Lower Tuolumne (MRFF)		25,241	11-May-97	18-May-97	6	0.23	
6-1-11-6-5	Lower Tuolumne (MRFF)		25,692	11-May-97	18-May-97	2	0.07	
6-1-11-6-6	Lower Tuolumne (MRFF)		21,531	08-May-97	21-May-97	4	0.19	
	Total	23-Apr-97	72,464	08-May-97	21-May-97	12		0.17
6-1-11-6-14	Lower Merced (MRFF)		33,064	---	---	0		
6-1-11-6-15	Lower Merced (MRFF)		28,294	28-May-97	28-May-97	1	0.03	
6-1-11-7-1	Lower Merced (MRFF)		24,943	---	---	0		
6-1-11-7-2	Lower Merced (MRFF)		5,856	---	---	0		
	Total	14-May-97	92,157	28-May-97	28-May-97	1		0.01

[a] MRFF = Merced River Fish Facility.

Table B-3 1996 Sacramento-San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-01-06-01-14	Mossdale (FRH)		49,024	---	---	0		
6-01-06-01-15	Mossdale (FRH)		51,718	25-Apr-96	27-Apr-96	2	0.04	
	Total	15-Apr-96	100,742	25-Apr-96	27-Apr-96	2		0.02
6-01-06-02-01	Mossdale (FRH)		50,462	07-May-96	07-May-96	1	0.02	
6-01-06-02-05	Mossdale (FRH)		49,194	---	---	0		
	Total	30-Apr-96	99,656	07-May-96	07-May-96	1		0.01
6-01-06-02-03	Dos Reis (FRH)		49,868	09-May-96	23-May-96	2	0.04	
6-01-06-01-10	Dos Reis (FRH)		48,770	11-May-96	11-May-96	1	0.02	
	Total	01-May-96	98,819	09-May-96	23-May-96	2		0.02
6-01-11-04-12	Dos Reis (MRFF)		25,530	08-May-96	10-May-96	2	0.07	
6-01-11-04-13	Dos Reis (MRFF)		28,079	05-May-96	07-May-96	2	0.07	
6-01-11-04-14	Dos Reis (MRFF)		18,459	06-May-96	06-May-96	1	0.05	
6-01-11-04-15	Dos Reis (MRFF)		35,893	08-May-96	12-May-96	5	0.13	
	Total	01-May-96	107,961	05-May-96	12-May-96	10		0.09

[a] FRH = Feather River Hatchery, MRFF = Merced River Fish Facility.

Table B-4 1996 Upper San Joaquin River Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-01-11-04-08	Upper Merced (MRFF)		21,011	---	---	0		
6-01-11-04-09	Upper Merced (MRFF)		21,069	---	---	0		
6-01-11-04-10	Upper Merced (MRFF)		22,638	08-May-96	08-May-96	1	0.04	
6-01-11-04-11	Upper Merced (MRFF)		21,693	---	---	0		
	Total	25-Apr-96	86,411	08-May-96	08-May-96	1		0.01
6-01-11-05-03	Lower Merced (MRFF)		21,705	---	---	0		
6-01-11-05-04	Lower Merced (MRFF)		22,019	05-May-96	05-May-96	1	0.04	
6-01-11-05-05	Lower Merced (MRFF)		20,613	---	---	0		
	Total	26-Apr-96	64,337	05-May-96	05-May-96	1		0.01
6-01-11-05-06	Upper Tuolumne (MRFF)		21,601	---	---	0		
6-01-11-05-07	Upper Tuolumne (MRFF)		22,861	02-May-96	06-May-96	2	0.08	
6-01-11-05-08	Upper Tuolumne (MRFF)	26-Apr-96	22,984	06-May-96	06-May-96	1	0.04	
	Total		67,446	02-May-96	06-May-96	3		0.04
6-01-11-05-09	Upper Tuolumne (MRFF)		22,789	02-May-96	02-May-96	1	0.04	
6-01-11-05-10	Upper Tuolumne (MRFF)		27,819	01-May-96	08-May-96	3	0.10	
	Total	27-Apr-96	50,608	01-May-96	08-May-96	4		0.07

[a] MRFF = Merced River Fish Facility.

Table B-5 1995 Sacramento-San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-1-14-5-1	Mossdale (FRH)		50,849	25-Apr-95	24-May-95	10	0.19	
6-1-14-4-14	Mossdale (FRH)		50,120	26-Apr-95	17-May-95	10	0.19	
	Total	17-Apr-95	100,969	25-Apr-95	24-May-95	20		0.19
6-31-50	Mossdale (FRH)		52,297	12-May-95	24-May-95	10	0.19	
6-31-51	Mossdale (FRH)		50,265	12-May-95	02-Jun-95	3	0.06	
	Total	05-May-95	102,562	12-May-95	02-Jun-95	13		0.12
6-1-14-5-4	Mossdale (FRH)		52,703	29-May-95	29-May-95	1	0.02	
6-31-48	Mossdale (FRH)		51,422	21-May-95	29-May-95	7	0.13	
	Total	17-May-95	104,125	21-May-95	29-May-95	8		0.07

[a] FRH = Feather River Hatchery.

Table B-6 1995 Upper San Joaquin River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-1-11-4-1	Upper Merced (MRFF)		28,349	08-May-95	29-May-95	5	0.17	
6-1-11-4-2	Upper Merced (MRFF)		27,961	21-May-95	13-Jun-95	3	0.10	
6-1-11-4-3	Upper Merced (MRFF)		26,839	28-May-95	12-Jun-95	4	0.14	
6-1-11-4-4	Upper Merced (MRFF)		28,138	23-May-95	06-Jun-95	6	0.20	
	Total	03-May-95	111,287	08-May-95	13-Jun-95	18		0.16
6-1-11-4-5	Lower Merced (MRFF)		27,318	21-May-95	03-Jun-95	7	0.24	
6-1-11-4-6	Lower Merced (MRFF)		27,643	21-May-95	09-Jun-95	4	0.14	
6-1-11-4-7	Lower Merced (MRFF)		28,054	20-May-95	29-May-95	7	0.23	
	Total	04-May-95	83,015	20-May-95	09-Jun-95	18		0.20
6-1-11-3-11	Upper Tuolumne (MRFF)		28,068	31-May-95	13-Jun-95	8	0.27	
6-1-11-3-12	Upper Tuolumne (MRFF)		27,132	21-May-95	25-Jun-95	6	0.22	
6-1-11-3-13	Upper Tuolumne (MRFF)		28,347	29-May-95	14-Jun-95	8	0.27	
	Total	04-May-95	83,547	21-May-95	25-Jun-95	22		0.26

[a] MRFF = Merced River Fish Facility.

Table B-7 1994 Upper San Joaquin River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-1-11-2-10	Upper Merced (MRFF)		28,315	01-May-94	02-May-94	2	0.07	
6-1-11-2-11	Upper Merced (MRFF)		25,328	01-May-94	01-May-94	1	0.04	
6-1-11-2-12	Upper Merced (MRFF)		28,532	01-May-94	08-May-94	2	0.07	
6-1-11-2-13	Upper Merced (MRFF)		17,390	15-May-94	15-May-94	1	0.05	
	Total	22-Apr-94	99,565	01-May-94	15-May-94	6		0.06
6-1-11-2-14	Lower Merced (MRFF)		35,017	---	---	0		
6-1-11-2-15	Lower Merced (MRFF)		23,324	01-May-94	03-May-94	2	0.08	
6-1-11-3-1	Lower Merced (MRFF)		23,750	---	---	0		
	Total	22-Apr-94	82,091	01-May-94	03-May-94	2		0.02
6-1-11-3-2	Upper Tuolumne (MRFF)		58,859	05-May-94	14-May-94	2	0.03	
6-1-11-3-3	Upper Tuolumne (MRFF)		4,281	12-May-94	12-May-94	1	0.22	
6-1-11-3-4	Upper Tuolumne (MRFF)		20,274	---	---	0		
	Total	23-Apr-94	83,414	05-May-94	14-May-94	3		0.03
6-1-11-3-5	Lower Tuolumne (MRFF)		36,429	06-May-94	06-May-94	1	0.03	
6-1-11-3-6	Lower Tuolumne (MRFF)		13,626	08-May-94	08-May-94	1	0.07	
	Total	24-Apr-94	50,055	06-May-94	08-May-94	2		0.04

[a] MRFF = Merced River Fish Facility.

Table B-8 1992 Sacramento-San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-1-14-2-12	Mossdale (FRH)		54,073	13-Apr-92	05-May-92	9	0.16	
6-1-14-2-13	Mossdale (FRH)		53,030	13-Apr-92	01-May-92	11	0.20	
	Total	07-Apr-92	107,103	13-Apr-92	05-May-92	20		0.18
6-1-14-2-14	Mossdale (FRH)		53,754	16-Apr-92	27-Apr-92	10	0.17	
6-1-14-2-15	Mossdale (FRH)		49,958	21-Apr-92	01-May-92	3	0.06	
	Total	13-Apr-92	103,712	16-Apr-92	01-May-92	13		0.12
6-1-14-3-3	Mossdale (w/barrier) (FRH)		53,294	03-May-92	06-May-92	7	0.12	
6-1-14-3-4	Mossdale (w/barrier) (FRH)		51,445	04-May-92	19-May-92	2	0.04	
	Total	24-Apr-92	104,739	03-May-92	19-May-92	9		0.08
6-31-31	Mossdale (w/barrier) (FRH)		51,262	16-May-92	16-May-92	1	0.02	
6-31-32	Mossdale (w/barrier) (FRH)		48,455	---	---	0		
	Total	04-May-92	99,717	16-May-92	16-May-92	1	0.01	0.01
6-31-33	Mossdale (w/barrier) (FRH)		52,454	---	---	0		
6-31-34	Mossdale (w/barrier) (FRH)		52,931	21-May-92	23-May-92	2	0.04	
	Total	12-May-92	105,385	21-May-92	23-May-92	2		0.02

[a] FRH = Feather River Hatchery, w/barrier = with barrier in upper Old River.

Table B-9 1991 Upper Sacramento River, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
5-18-45	Princeton (CNFH)		12,474	14-May-91	19-May-91	4	0.30	
5-18-47	Princeton (CNFH)		18,713	11-May-91	21-May-91	10	0.50	
5-18-48	Princeton (CNFH)		20,792	10-May-91	30-May-91	10	0.46	
	Total	03-May-91	51,979	10-May-91	30-May-91	24		0.44

[a] CNFH = Coleman National Fish Hatchery.

Table B-10 1991 Sacramento-San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-1-14-1-14	Dos Reis (FRH)		52,097	23-Apr-91	11-May-91	8	0.15	
6-1-14-1-15	Dos Reis (FRH)		50,902	23-Apr-91	02-May-91	9	0.17	
	Total	15-Apr-91	102,999	23-Apr-91	11-May-91	17		0.16
6-1-14-2-1	Buckley Cove (FRH)		51,128	24-Apr-91	06-May-91	15	0.28	
6-1-14-2-2	Buckley Cove (FRH)		48,213	25-Apr-91	02-May-91	11	0.21	
	Total	16-Apr-91	99,341	24-Apr-91	06-May-91	26		0.24
6-1-14-2-3	Empire Tract (FRH)		48,255	24-Apr-91	09-May-91	25	0.48	
6-1-14-2-4	Empire Tract (FRH)		47,347	24-Apr-91	12-May-91	29	0.58	
	Total	17-Apr-91	95,602	24-Apr-91	12-May-91	54		0.54
6-1-14-2-7	Sacramento (Miller Park) (FRH)		51,392	01-May-91	06-May-91	34	0.62	
6-1-14-2-8	Sacramento (Miller Park) (FRH)		51,272	30-Apr-91	09-May-91	50	0.91	
	Total	25-Apr-91	102,664	30-Apr-91	09-May-91	84		0.77
6-1-14-2-9	Sacramento (Miller Park) (FRH)		53,430	27-Apr-91	16-May-91	21	0.37	
6-31-24	Sacramento (Miller Park) (FRH)		51,086	04-May-91	10-May-91	34	0.64	
	Total	29-Apr-91	104,516	27-Apr-91	16-May-91	55		0.50
6-31-25	Buckley Cove (FRH)		49,393	11-May-91	30-May-91	7	0.13	
6-31-26	Buckley Cove (FRH)		50,427	11-May-91	16-May-91	13	0.24	
	Total	06-May-91	99,820	11-May-91	30-May-91	20		0.19

[a] FRH = Feather River Hatchery.

Table B-11 1990 Sacramento-San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-1-14-1-8	Dos Reis (FRH)		53,254	26-Apr-90	13-May-90	2	0.04	
6-1-14-1-7	Dos Reis (FRH)		52,488	12-May-90	15-May-90	2	0.04	
	Total	16-Apr-90	105,742	26-Apr-90	15-May-90	4		0.04
6-1-14-1-6	Upper Old River (FRH)		52,954	---	---	0		
6-1-14-1-5	Upper Old River (FRH)		53,313	24-Apr-90	01-May-90	2	0.041	
	Total	17-Apr-90	106,267	24-Apr-90	01-May-90	2		0.02
6-1-14-1-12	Upper Old River (FRH)		51,521	16-May-90	16-May-90	1	0.02	
6-1-14-1-13	Upper Old River (FRH)		52,074	---	---	0		
	Total	03-May-90	103,595	16-May-90	16-May-90	1		0.01

[a] FRH = Feather River Hatchery.

Table B-12 1990 Upper San Joaquin River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-1-11-1-14	Upper Tuolumne (MRFF)		24,134	12-May-90	12-May-90	1	0.04	
6-1-11-1-15	Upper Tuolumne (MRFF)		24,259	16-May-90	16-May-90	1	0.04	
6-1-11-2-1	Upper Tuolumne (MRFF)		23,494	12-May-90	12-May-90	1	0.04	
6-1-11-2-2	Upper Tuolumne (MRFF)		21,766	08-May-90	08-May-90	1	0.04	
	Total	30-Apr-90	93,653	08-May-90	16-May-90	4		0.04
6-1-11-2-3	Lower Tuolumne (TRFF)		27,263	12-May-90	12-May-90	1	0.03	
6-1-11-2-4	Lower Tuolumne (TRFF)		26,067	---	---	0		
6-1-11-2-5	Lower Tuolumne (TRFF)		24,905	---	---	0		
	Total	01-May-90	78,235	12-May-90	12-May-90	1		0.01

[a] MRFF = Merced River Fish Facility, TRFF = Tuolumne River Fish Facility.

Table B-13 1989 Sacramento-San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-1-11-1-11	Jersey Point (FRH)		27,758	29-Apr-89	15-May-89	24	0.81	
6-1-11-1-12	Jersey Point (FRH)		29,058	29-Apr-89	12-May-89	29	0.94	
	Total	24-Apr-89	56,816	29-Apr-89	15-May-89	53		0.88
6-1-11-1-7	Dos Reis (MRFF)		25,089	10-May-89	30-May-89	3	0.12	
6-1-11-1-8	Dos Reis (MRFF)		25,631	08-May-89	24-May-89	7	0.26	
6-1-11-1-13	Dos Reis (MRFF)		25,353	10-May-89	10-May-89	1	0.04	
	Total	02-May-89	76,073	08-May-89	30-May-89	11		0.15
6-1-11-1-4	Upper Old River (MRFF)		25,087	08-May-89	08-May-89	1	0.04	
6-1-11-1-5	Upper Old River (MRFF)		24,472	09-May-89	09-May-89	1	0.04	
6-1-11-1-6	Upper Old River (MRFF)		24,782	08-May-89	15-May-89	2	0.08	
	Total	03-May-89	74,341	08-May-89	15-May-89	4		0.05
6-1-11-1-9	Jersey Point (FRH)		27,525	08-May-89	25-May-89	33	1.13	
6-1-11-1-10	Jersey Point (FRH)		28,708	08-May-89	22-May-89	25	0.82	
	Total	05-May-89	56,233	08-May-89	25-May-89	58		1.0
6-31-15	Sacramento (Miller Park) (FRH)		44,695	17-Jun-89	19-Jun-89	11	0.23	
6-31-17	Sacramento (Miller Park) (FRH)		49,909	17-Jun-89	19-Jun-89	9	0.17	
	Total	14-Jun-89	94,604	17-Jun-89	19-Jun-89	20		0.20

[a] FRH = Feather River Hatchery, MRFF = Merced River Fish Facility.

Table B-14 1989 Upper San Joaquin River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
B6-14-9	Upper Stanislaus (MRFF)		52,445	29-Apr-89	11-May-89	3	0.05	
B6-14-10	Upper Stanislaus (MRFF)		51,506	30-Apr-89	16-May-89	4	0.07	
	Total	20-Apr-89	103,951	29-Apr-89	16-May-89	7		0.06
B6-1-1	Lower Stanislaus (MRFF)		25,525	24-Apr-89	02-May-89	11	0.40	
B6-14-11	Lower Stanislaus (MRFF)		48,695	24-Apr-89	27-Apr-89	6	0.12	
	Total	19-Apr-89	74,220	24-Apr-89	2-May-89	17		0.21
6-1-11-1-1	Lower Merced (MRFF)		25,357	03-May-89	10-May-89	3	0.11	
6-1-11-1-2	Lower Merced (MRFF)		25,276	15-May-89	15-May-89	1	0.04	
6-1-11-1-3	Lower Merced (MRFF)		23,832	---	---	0		
	Total	21-Apr-89	74,465	03-May-89	15-May-89	4		0.05

[a] MRFF = Merced River Fish Facility.

Table B-15 1988 Sacramento-San Joaquin Estuary Fall-run Releases

Tag code	Release site (stock)[a]	Date released	Number released	First day recovered	Last day recovered	Number recovered	Survival index	Group survival
B6-14-2	Courtland (FRH)		51,388	07-May-88	21-May-88	71	0.65	
B6-14-3	Courtland (FRH)		55,861	07-May-88	25-May-88	83	0.70	
	Total	03-May-88	107,249	07-May-88	25-May-88	154		0.68
B6-14-6	Sacramento (Miller Park) (FRH)		51,005	08-May-88	23-May-88	77	0.71	
B6-14-7	Sacramento (Miller Park) (FRH)		51,731	09-May-88	21-May-88	65	0.59	
	Total	05-May-88	102,736	08-May-88	23-May-88	142		0.65
B6-14-4	Courtland (FRH)		51,274	08-May-88	27-May-88	67	0.67	
B6-14-5	Courtland (FRH)		51,206	10-May-88	21-May-88	80	0.73	
	Total	06-May-88	102,480	08-May-88	27-May-88	147		0.73
6-62-59	Courtland (FRH)		54,997	23-Jun-88	29-Jun-88	25	0.21	
6-62-60	Courtland (FRH)		51,904	23-Jun-88	03-Jul-88	14	0.13	
	Total	21-Jun-88	106,901	23-Jun-88	03-Jul-88	39		0.17
6-62-61	Sacramento (Miller Park) (FRH)		49,245	26-Jun-88	07-Jul-88	7[b]	0.08	
6-62-62	Sacramento (Miller Park) (FRH)		48,647	26-Jun-88	03-Jul-88	7[b]	0.07	
	Total	23-Jun-88	97,892	26-Jun-88	07-Jul-88	14		0.08

[a] FRH = Feather River Hatchery.

[b] Total number recovered for both tag code 6-62-61 and 6-62-62 is reduced by 1, as they were recorded as being recovered at Chipps Island on the day of release.

Table B-16 1988 Upper San Joaquin River and Tributaries Fall-run Releases

Tag code	*Release site (stock)[a]*	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
B6-11-5	Upper Stanislaus (MRFF)		36,769	04-May-88	11-May-88	6	0.08	
B6-11-6	Upper Stanislaus (MRFF)		34,906	12-May-88	21-May-88	5	0.07	
	Total	26-Apr-88	71,675	04-May-88	21-May-88	11		0.07
B6-11-3	Lower Stanislaus (MRFF)		35,249	03-May-88	19-May-88	6	0.08	
B6-11-4	Lower Stanislaus (MRFF)		33,539	07-May-88	22-May-88	7	0.10	
	Total	26-Apr-88	68,788	03-May-88	22-May-88	13		0.09

[a] MRFF = Merced River Fish Facility.

Table B-17 1987 Sacramento San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-45-3	Upper Old River (MRFF)		31,099	30-Apr-87	06-May-87	8	0.23	
6-45-4	Upper Old River (MRFF)		29,253	30-Apr-87	05-May-87	5	0.16	
6-45-5	Upper Old River (MRFF)		30,600	30-Apr-87	03-May-87	3	0.07	
	Total	27-Apr-87	90,952	30-Apr-87	06-May-87	16		0.16
6-45-6	Dos Reis (MRFF)		30,919	05-May-87	13-May-87	30	0.93	
6-45-7	Dos Reis (MRFF)		31,634	05-May-87	20-May-87	22	0.66	
6-45-8	Dos Reis (MRFF)		30,059	05-May-87	12-May-87	28	0.89	
	Total	27-Apr-87	92,612	05-May-87	20-May-87	80		0.83
6-62-53	Courtland (FRH)		49,781	01-May-87	12-May-87	32	0.60	
6-62-54	Courtland (FRH)		50,521	01-May-87	14-May-87	39	0.72	
	Total	28-Apr-87	100,302	01-May-87	14-May-87	71		0.66
6-62-56	Courtland (FRH)		49,083	04-May-87	15-May-87	20	0.39	
6-62-57	Courtland (FRH)		51,836	05-May-87	22-May-87	23	0.42	
	Total	01-May-87	100,919	04-May-87	22-May-87	43		0.41

[a] MRFF = Merced River Fish Facility, FRH = Feather River Hatchery.

Table B-18 1987 Upper San Joaquin River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)[a]*	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-46-60	Upper Tuolumne (MRFF)		29,959	25-Apr-87	01-May-87	2	0.06	
6-46-61	Upper Tuolumne (MRFF)		30,601	---	---	0		
6-46-62	Upper Tuolumne (MRFF)		29,040	28-Apr-87	30-Apr-87	3	0.10	
	Total	16-Apr-87	89,600	25-Apr-87	01-May-87	5		0.05
6-45-1	Lower Tuolumne (MRFF)		31,866	22-Apr-87	04-May-87	5	0.14	
6-45-2	Lower Tuolumne (MRFF)		30,936	22-Apr-87	05-May-87	9	0.27	
6-46-63	Lower Tuolumne (MRFF)		30,709	26-Apr-87	05-May-87	4	0.12	
	Total	16-Apr-87	93,511	22-Apr-87	05-May-87	18		0.18

[a] MRFF = Merced River Fish Facility.

Table B-19 1986 Upper Sacramento River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
H5-4-2	Battle Creek (CNFH)		24,933	21-May-86	26-May-86	11	0.41	
H5-4-3	Battle Creek (CNFH)		28,659	21-May-86	27-May-86	19	0.62	
	Total	13-May-86	53,592	21-May-86	27-May-86	30		0.52
H5-4-4	Below RBDD (CNFH)		26,900	21-May-86	31-May-86	9	0.31	
H5-4-5	Below RBDD (CNFH)		27,606	20-May-86	01-Jun-86	17	0.58	
	Total	13-May-86	54,506	20-May-86	01-Jun-86	26		0.45
H5-4-6	Princeton (CNFH)		23,669	23-May-86	27-May-86	3	0.12	
H5-4-7	Princeton (CNFH)		22,719	19-May-86	22-May-86	7	0.29	
	Total	14-May-86	56,388	19-May-86	27-May-86	10		0.17

[a] CNFH = Coleman National Fish Hatchery.

Table B-20 1986 Sacramento-San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-46-58	Dos Reis (MRFF)		47,954	02-Jun-86	08-Jun-86	13	0.25	
B6-11-1	Dos Reis (MRFF)		47,641	02-Jun-86	07-Jun-86	22	0.43	
	Total	29-May-86	95,595	02-Jun-86	08-Jun-86	35		0.34
6-46-59	Upper Old River (MRFF)		49,434	01-Jun-86	06-Jun-86	10	0.19	
B6-11-2	Upper Old River (MRFF)		50,747	01-Jun-86	03-Jun-86	11	0.20	
	Total	30-May-86	100,181	01-Jun-86	06-Jun-86	21		0.20

[a] MRFF = Merced River Fish Facility.

Table B-21 1986 Upper San Joaquin River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-46-54	Upper Tuolumne (MRFF)		49,630	25-Apr-86	27-May-86	17	0.38	
6-46-55	Upper Tuolumne (MRFF)		49,518	23-Apr-86	21-May-86	18	0.43	
	Total	14-Apr-86	99,148	23-Apr-86	27-May-86	35		0.40
6-46-56	Lower Tuolumne (MRFF)		51,300	23-Apr-86	7-May-86	10	0.31	
6-46-57	Lower Tuolumne (MRFF)		52,174	25-Apr-86	10-May-86	10	0.26	
	Total	14-Apr-86	103,474	23-Apr-86	10-May-86	20		0.27
6-46-48	Upper Stanislaus (MRFF)		31,120	03-May-86	16-Jun-86	17	0.55	
6-46-49	Upper Stanislaus (MRFF)		31,148	05-May-86	20-May-86	11	0.33	
6-46-50	Upper Stanislaus (MRFF)		24,751	07-May-86	07-Jun-86	4	0.15	
6-46-53	Upper Stanislaus (MRFF)		21,254	08-May-86	11-Jun-86	5	0.22	
	Total	28-Apr-86	108,273	03-May-86	16-Jun-86	37		0.34
6-46-45	Lower Stanislaus (MRFF)		31,491	05-May-86	15-May-86	16	0.48	
6-46-46	Lower Stanislaus (MRFF)		31,310	05-May-86	25-May-86	18	0.54	
6-46-47	Lower Stanislaus (MRFF)		30,530	03-May-86	18-May-86	20	0.66	
6-46-52	Lower Stanislaus (MRFF)		12,768	05-May-86	18-May-86	6	0.44	
	Total	29-Apr-86	106,169	03-May-86	25-May-86	60		0.56

[a] MRFF = Merced River Fish Facility.

Table B-22 1985 Upper Sacramento River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
5-39-4	Battle Creek (CNFH)		11,484	21-May-85	23-May-85	2	0.16	
5-40-4	Battle Creek (CNFH)		10,698	22-May-85	25-May-85	2	0.17	
5-41-4	Battle Creek (CNFH)		10,330	21-May-85	23-May-85	3	0.27	
H5-1-5	Battle Creek (CNFH)		22,558	20-May-85	22-May-85	3	0.12	
5-6-16	Battle Creek (CNFH)		10,209	24-May-85	24-May-85	1	0.09	
	Total	14-May-85	65,279	20-May-85	25-May-85	11		0.16
5-9-47	Below RBDD (CNFH)		21,871	21-May-85	24-May-85	2	0.09	
5-42-4	Below RBDD (CNFH)		10,610	21-May-85	31-May-85	5	0.44	
5-43-4	Below RBDD (CNFH)		9,756	22-May-85	23-May-85	3	0.29	
H5-1-6	Below RBDD (CNFH)		23,378	22-May-85	24-May-85	9	0.36	
	Total	14-May-85	65,615	21-May-85	31-May-85	19		0.27
5-9-48	Princeton (CNFH)		21,943	21-May-85	24-May-85	3	0.13	
5-9-49	Princeton (CNFH)		20,460	21-May-85	22-May-85	3	0.14	
H5-1-7	Princeton (CNFH)		23,519	20-May-85	22-May-85	3	0.12	
	Total	15-May-85	65,922	20-May-85	24-May-85	9		0.13

[a] CNFH = Coleman National Fish Hatchery.

Table B-23 1985 Sacramento-San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-62-38	Courtland (FRH)		54,457	14-May-85	25-May-85	23	0.40	
6-62-39	Courtland (FRH)		14,731	14-May-85	25-May-85	2	0.13	
6-62-40	Courtland (FRH)		10,887	14-May-85	25-May-85	3	0.26	
6-62-41	Courtland (FRH)		20,551	14-May-85	25-May-85	9	0.41	
	Total	10-May-85	100,626	14-May-85	25-May-85	37		0.34

[a] FRH = Feather River Hatchery.

Table B-24 1984 Upper Sacramento River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-60-42	Battle Creek (CNFH)		50,742	16-May-84	23-May-84	19	0.77	
6-60-43	Battle Creek (CNFH)		49,479	16-May-84	23-May-84	16	0.66	
	Total	09-May-84	100,221	16-May-84	23-May-84	35		0.72
6-60-40	Below RBDD (CNFH)		51,948	15-May-84	23-May-84	29	1.06	
6-60-41	Below RBDD (CNFH)		50,921	15-May-84	23-May-84	29	1.08	
	Total	09-May-84	102,869	15-May-84	23-May-84	58		1.07
6-60-38	Knights Landing (CNFH)		49,400	13-May-84	23-May-84	19	0.85	
6-60-39	Knights Landing (CNFH)		49,351	15-May-84	23-May-84	17	0.65	
	Total	09-May-84	98,751	13-May-84	23-May-84	36		0.81

[a] CNFH = Coleman National Fish Hatchery.

Table B-25 1984 Sacramento-San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-62-28	South Fork Mokelumne (FRH)		41,371	16-Jun-84	26-Jun-84	25	0.72	
6-42-8	South Fork Mokelumne (FRH)		14,916	17-Jun-84	22-Jun-84	9	0.66	
	Total	12-Jun-84	56,287	16-Jun-84	26-Jun-84	34		0.72
6-62-29	Ryde (FRH)		44,818	16-Jun-84	26-Jun-84	30	0.66	
6-42-9	Ryde (FRH)		15,180	17-Jun-84	28-Jun-84	8	0.62	
	Total	13-Jun-84	59,998	16-Jun-84	28-Jun-84	38		0.73

[a] FRH = Feather River Hatchery.

Table B-26 1983 Upper Sacramento River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-60-36	Battle Creek (CNFH)		44,382	07-Jun-83	24-Jun-83	15	0.65	
6-60-37	Battle Creek (CNFH)		43,508	07-Jun-83	17-Jun-83	10	0.34	
	Total	02-Jun-83	87,890	07-Jun-83	24-Jun-83	25		0.55
6-60-34	Below RBDD (CNFH)		44,498	07-Jun-83	15-Jun-83	16	0.51	
6-60-35	Below RBDD (CNFH)		45,343	07-Jun-83	21-Jun-83	10	0.40	
	Total	02-Jun-83	89,841	07-Jun-83	21-Jun-83	26		0.52
6-60-32	Knights Landing (CNFH)		45,986	05-Jun-83	15-Jun-83	45	1.26	
6-60-33	Knights Landing (CNFH)		46,099	05-Jun-83	21-Jun-83	31	1.10	
	Total	02-Jun-83	92,085	05-Jun-83	21-Jun-83	76		1.34

[a] CNFH = Coleman National Fish Hatchery.

Table B-27 1982 Upper Sacramento River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-60-26	Battle Creek (CNFH)		42,964	13-May-82	07-Jun-82	37	1.65	
6-60-27	Battle Creek (CNFH)		41,738	13-May-82	24-May-82	27	0.76	
	Total	05-May-82	84,702	13-May-82	07-Jun-82	64		1.45
6-60-28	Below RBDD (CNFH)		44,308	13-May-82	15-Jun-82	34	1.24	
6-60-29	Below RBDD (CNFH)		43,817	10-May-82	24-May-82	35	1.06	
	Total	05-May-82	88,125	10-May-82	15-Jun-82	69		1.25
6-60-30	Knights Landing (CNFH)		44,735	10-May-82	24-May-82	50	1.48	
6-60-31	Knights Landing (CNFH)		44,540	10-May-82	24-May-82	41	1.22	
	Total	05-May-82	89,275	10-May-82	24-May-82	91		1.35

[a] CNFH = Coleman National Fish Hatchery.

Table B-28 1981 Upper Sacramento River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-60-20	Knights Landing (CNFH)		43,059	28-May-81	28-May-81	3	0.08	
6-60-21	Knights Landing (CNFH)		43,562	28-May-81	04-Jun-81	2	0.14	
	Total	18-May-81	86,621	28-May-81	04-Jun-81	5		0.18

[a] CNFH = Coleman National Fish Hatchery.

Table B-29 1981 Sacramento-San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*	*Survival index*	*Group survival*
6-62-14	Discovery Park (FRH)		71,932	10-Jun-81	10-Jun-81	1	0.02	
6-62-17	Discovery Park (FRH)		68,318	---	---	0	0	
	Total	04-Jun-81	140,249	10-Jun-81	10-Jun-81	1		0.01

[a] FRH = Feather River Hatchery.

Table B-30 1980 Upper Sacramento River and Tributaries, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*
H5-3-1	Below RBDD (CNFH)		25,618	---	---	0
H5-3-2	Below RBDD (CNFH)		22,560	---	---	0
	Total	28-Feb-80	48,178			
H5-3-5	Below RBDD (CNFH)		21,786	15-May-80	15-May-80	1
H5-3-6	Below RBDD (CNFH)		21,836	---	---	0
	Total	31-Mar-80	43,622			

[a] CNFH = Coleman National Fish Hatchery.

Table B-31 1980 Sacramento-San Joaquin Estuary, Fall-run Releases

Tag code	*Release site (stock)*[a]	*Date released*	*Number released*	*First day recovered*	*Last day recovered*	*Number recovered*
H5-2-4	Berkeley (CNFH)		21,939	---	---	0
H5-2-5	Berkeley (CNFH)		20,788	---	---	0
	Total	28-Feb-80	42,727			
H5-2-6	Clarksburg (CNFH)		22,121	---	---	0
H5-2-7	Clarksburg (CNFH)		21,624	---	---	0
	Total	28-Feb-80	43,745			
H5-3-3	Clarksburg (CNFH)		23,908	17-Apr-80	01-May-80	2
H5-3-4	Clarksburg (CNFH)		22,829	24-Apr-80	01-May-80	2
	Total	31-Mar-80	46,737			

[a] CNFH = Coleman National Fish Hatchery.

The Effects of San Joaquin River Flows and Delta Export Rates During October on the Number of Adult San Joaquin Chinook Salmon that Stray

Carl Mesick

Abstract

This report describes a two-part investigation of the effects of fall make-up pumping on straying of adult San Joaquin chinook salmon. The first part is a reevaluation of 1964 to 1967 data collected by Hallock and others (1970) on the migratory behavior of tagged and untagged adult San Joaquin salmon in the Delta. The second part is an evaluation of the recovery of adult salmon that were released in the San Joaquin basin as coded-wire tagged juveniles reared at the Merced River Fish Facility.

There are three important results from Hallock and others (1970) regarding their migration analysis. First, adult salmon are migrating through the San Joaquin Delta near Prisoners Point primarily during October, the period when they are probably most susceptible to low flows and high exports. Second, the fish migrate slowly and do not arrive in the San Joaquin tributaries until about four weeks after they pass Prisoners Point, even when flows, exports, and dissolved oxygen concentrations near Stockton are suitable for migration. And third, migration rates of adult salmon are substantially higher when Vernalis flows exceed about 3,000 cfs and total exports are less than 100% of Vernalis flows. Although most of the tagged fish migrated into the Sacramento and Mokelumne basins when Vernalis flows were less than about 2,000 cfs and total exports exceeded 150% of Vernalis flows, there is uncertainty as to whether these were San Joaquin fish that strayed or Sacramento River fish that were captured in the San Joaquin on their way to the Sacramento River.

The coded-wire-tag (CWT) recovery data may not have been appropriate for a straying analysis because there are no clear records of the number of fish examined for tags during the carcass surveys. Not all fish counted for the carcass survey were examined for tags. These recovery data are necessary to accurately compute the total number of adult salmon with tags in each river. A casual inspection of the CWT recovery data suggests that: (1) straying rates increased as the percentage of San Joaquin flow exported by the CVP and SWP pump-

ing facilities increased and (2) the critical period is between 1 and 21 October. Furthermore, pulse flows from the San Joaquin tributaries, or a reduction of Delta exports that result in no more than a 300% export rate of San Joaquin flows at Vernalis for eight to twelve days in mid-October, are sufficient to keep straying rates below 3%.

The results of these correlation analyses suggest that when more than 300% of Vernalis flow is exported over a ten-day period in mid-October adult San Joaquin chinook salmon stray to the Sacramento and eastside basins. However, further tests are needed due to the limitations of the existing data.

Introduction

To increase production of fall-run chinook salmon (*Oncorhynchus tshawytscha*) in the San Joaquin tributaries, exports at the State Water Project (SWP) and the Central Valley Project (CVP) and San Joaquin River flows were managed to provide a 1:4 ratio of exports to flow at Vernalis during spring 1996 when the salmon smolts were migrating through the Delta. The State Water Resources Control Board Order 96–6 permitted the SWP and the CVP to "make-up" the reduced volume of springtime exports by pumping at near maximum rates during fall, primarily October and November. Sustained high export rates during October and November were cause for concern, since this is the period when adult San Joaquin chinook salmon migrate upstream through the Delta to their spawning grounds. To do this, the salmon require the scent of San Joaquin River flow to return to their natal river. In October 1996, the combined SWP and CVP exports averaged about 9,600 cfs, whereas San Joaquin River flows at Vernalis averaged 2,650 cfs. Fall make-up pumping occurred again in fall 1997, and the combined SWP and CVP exports averaged about 9,700 cfs, while San Joaquin River flows at Vernalis averaged about 1,950 cfs. It is likely that when exports are relatively high compared to Vernalis flows, little if any San Joaquin River water reaches the San Francisco Bay where it is needed to help guide the salmon (see the literature review that follows). If true, a substantial portion of the adult salmon population of the San Joaquin tributaries could stray into the Sacramento and Mokelumne rivers, which provide a majority of the flow through the Central Delta during the fall, particularly when the ratio of exports to San Joaquin River flow is high.

This report describes a two-part evaluation of the possible effects of fall make-up pumping on the straying of adult San Joaquin chinook salmon. The first part is a reevaluation of the data collected by Hallock and others (1970) from 1964 to 1967 on the migratory behavior of tagged and untagged adult San Joaquin salmon in the Delta. The second part is an evaluation of the recovery of adult salmon that were released in the San Joaquin basin as coded-wire tagged juveniles reared at the Merced River Fish Facility. The recovery data are from Department of Fish and Game surveys made between 1983 and 1996.

A Literature Review of Homing Behavior of Adult Pacific Salmon

Adult Pacific salmon rely on olfactory cues to guide their upriver migration to their natal stream, although other factors may be involved (Quinn 1990). It is generally believed that juveniles rearing and migrating downriver acquire a series of olfactory waypoints at every major confluence and retrace the sequence as adults when they return to spawn (Harden Jones 1968; Quinn and others 1989; Quinn 1990). Few adult coho (Wisby and Hasler 1954) and chinook salmon (Groves and others 1968) that had their olfactory pits plugged (to prevent them from sensing waterborne odors) were able to home to their natal stream. Most (67% and 89%) of the control fish in those studies were able to home to their natal stream. During both of these studies, blinded fish were able to home more successfully than were fish with occluded olfactory pits. Normal homing rates for chinook salmon probably range between 84% for 17,671 recovered fish that were reared at a New Zealand hatchery (Unwin and Quinn 1993) and 98.6% for 41,085 recovered fish that were reared at the Cowlitz River Hatchery, Washington (Quinn and Fresh 1984). Experiments have also shown that juvenile coho salmon exposed to artificial waterborne odors while they were reared in hatcheries, homed to waters that contained those artificial odors (Cooper and others 1976; Johnsen and Hasler 1980; Brannon and Quinn 1990; Dittman and others 1994; Dittman and others 1996).

Besides olfactory cues, there is evidence that compass orientation helps adult salmon to home to their natal stream. Adult Pacific salmon, particularly those that migrate long distances in the ocean to feed (stream-type populations), use compass orientation in ocean and coastal waters to locate the mouth of their natal stream, where they switch to olfactory clues (Quinn 1990). However, the mechanism of compass orientation and the transition from compass orientation in coastal waters and estuaries to olfactory-based upriver homing appear to be very complicated and not well understood (Quinn 1990). Furthermore, ocean-type populations of Pacific salmon, such as the fall-run chinook populations in the San Joaquin tributaries, may not have a well-developed means of navigation by compass orientation since they do not migrate far from the coast to feed. This would explain why most sockeye salmon, a stream-type population, that had their olfactory nerves severed in an experiment could still migrate in a homeward direction (Craigie 1926), whereas chinook salmon with plugged olfactory pits could not migrate homeward (Groves and others 1968).

There is contradictory evidence that hereditary factors also influence homing behavior. Bams (1976) and McIsaac and Quinn (1988) provided proof that a high proportion of displaced chinook salmon offspring homed to their ancestral spawning area even though the juvenile fish were never exposed to their ancestral waters. However, Donaldson and Allen (1957) provided evidence that coho juveniles relocated to two different locations prior to smolting would home to their release sites and not to their original hatchery site. The scent from siblings (population-specific odors) did not affect adult coho salmon homing behavior in Lake Washington (Brannon and Quinn 1990), and no other mechanism to account for a hereditary factor has been discovered.

When adult Pacific salmon do not return to their natal stream, they appear to select a new river for spawning based on the magnitude of streamflow. Two field studies conducted by Quinn and Fresh (1984) in Washington and Unwin and Quinn (1993) in New Zealand determined that adult chinook salmon strays selected rivers with the highest streamflow. An experimental study conducted by Wisby and Hasler (1954) also showed that when the scent of the fishes' natal river was not present, coho salmon moved into the arm of a Y-maze with the greatest flow. If true, then adult San Joaquin salmon that cannot use olfaction due to an absence of scent from their natal river would probably return to the Sacramento River, where flows are substantially greater.

A Review of Hallock's Study

The migration of adult fall-run chinook salmon in the Delta and lower San Joaquin River was studied by the Department of Fish and Game between 1964 and 1967. Adult salmon were captured with a trammel net (floating gill net, 23 feet deep and 1,378 to 1,804 feet long) at Prisoners Point in the San Joaquin River, which is about 2.5 miles upstream of the confluence with the Mokelumne River. The daily catch rate was recorded during each year except 1965. Sonic tags were attached to the dorsal surface of the fish just anterior to the dorsal fin with straps and pins. Stationary monitors that recorded the presence of the tags were used in the Sacramento River, Mokelumne River, San Joaquin River, and throughout the Delta to help determine the destination and migration rate of the tagged fish. The authors also presented the number of salmon captured at a trap operated in the Stanislaus River for hatchery stock from 1965 to 1967.

Results

For 1966 and 1967, when catch rates were estimated at both Prisoners Point and the Stanislaus trap, most fish arrived at Prisoners Point between 1 October and 20 October (some were caught through 21 November), whereas most fish were not caught at the Stanislaus River trap until after 5 November (Figures 1 and 2). Hallock reported that few of the tagged fish migrated past Stockton when dissolved oxygen (DO) levels were less than about 5 ppm (4.5 in 1967 and 5.5 in 1965). Furthermore, the catch at the Stanislaus trap tended to increase about one week after DO levels at Stockton stabilized at or above the critical level. The fish usually remained in the Delta for at least three weeks prior to entering the Stanislaus and Hallock reported that some remained in the Delta for up to two months. Therefore, an evaluation of fall make-up pumping on straying of adult fish must be conducted by monitoring fish in the Delta, not the San Joaquin tributaries.

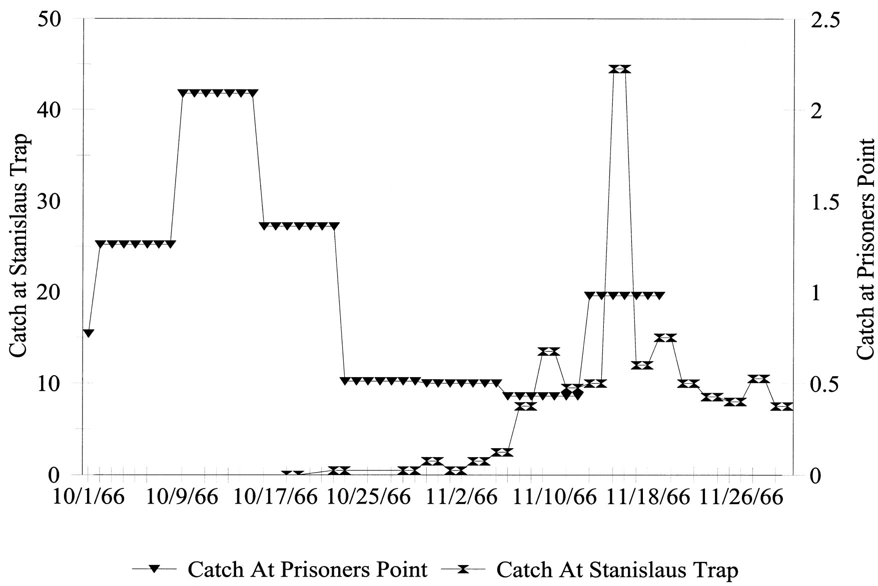

Figure 1 Catch rates of adult chinook salmon with a trammel net at Prisoners Point in the San Joaquin River (about 2.5 miles upstream of the confluence with the Mokelumne River) and at the Orange Blossom trap in the Stanislaus River in October and November 1966

I evaluated the effects of exports and San Joaquin flow on the number of strays using two sets of data collected by Hallock and others (1970). The first data set evaluated were straying rates for 35 to 77 adult salmon tagged at Prisoners Point each year and the second data set evaluated were catch rates of adult salmon at Prisoners Point relative to flows and exports.

During the first three weeks of October in 1965 and 1967, only 15% of the tagged fish migrated into the Sacramento and Mokelumne rivers when Vernalis flows ranged between 2,000 and 4,000 cfs and the proportion of Vernalis flows exported at Tracy ranged between 45% and 120%. In contrast, during the first three weeks of October, 54% of the tagged fish (35) strayed into the Sacramento and Mokelumne rivers in 1964 and 71% of the tagged fish (52) strayed in 1966 when Vernalis flow ranged between 700 and 1,500 cfs and the proportion of Vernalis flows exported at Tracy ranged between 150% and 250%. A solid rock barrier was installed at the head of Old River in 1964, but not during the other study years, and it is likely that the barrier increased the amount of San Joaquin flow that remained in the San Joaquin River.

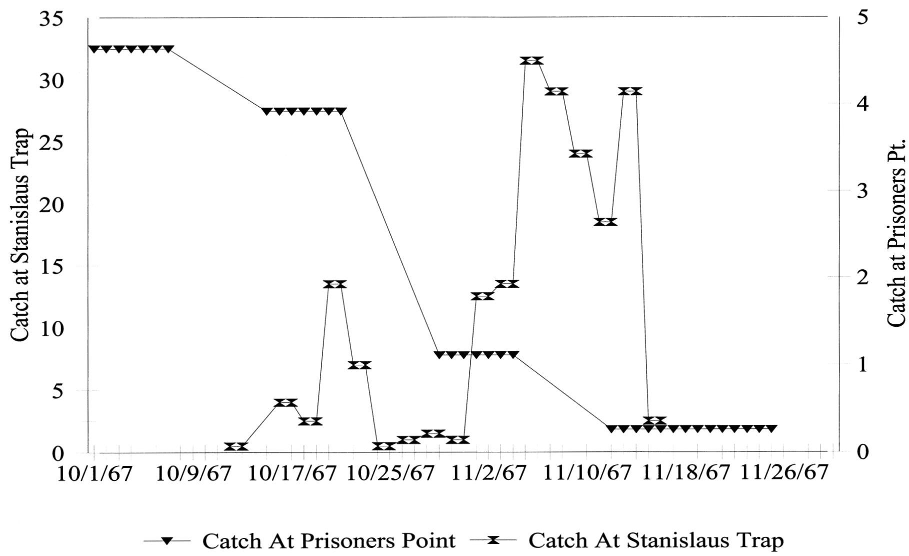

Figure 2 Catch rates of adult chinook salmon with a trammel net at Prisoners Point in the San Joaquin River, which is about 2.5 miles upstream of the confluence with the Mokelumne River and at the Orange Blossom trap in the Stanislaus River in October and November 1967

Hallock and others (1970) could not verify whether the adult salmon caught at Prisoners Point were actually of San Joaquin origin. They speculated the tagged fish that migrated into the Sacramento and Mokelumne rivers were not San Joaquin basin strays but were Sacramento basin fish guided 2.5 miles upstream of the mouth of the Mokelumne in the Delta (to Prisoners Point) by strong tidal flows. However, they could not reasonably explain why a high number of tagged fish migrated into the Sacramento and Mokelumne rivers in 1964 and 1966 when there was a high proportion of exported San Joaquin flow, but a low number of tagged fish migrated into the same rivers in 1965 and 1967 when there was a low proportion of exported San Joaquin flow.

The effects of Vernalis flows and exports on straying were also evaluated using the catch rate at Prisoners Point determined by Hallock and others (1970). In 1964, catch rates ranged between 0.63 and 1.25 fish/hour between 5 and 25 October, when Vernalis flow ranged between 1,100 and 1,500 cfs (Figure 3) and exports ranged between 130 and 225% of Vernalis flows (Figure 4). After Vernalis flows rapidly increased to about 2,000 cfs and exports began to decline to less than 100% of Vernalis flows on 29 October, catch rates at Prisoners Point increased to 5.36 fish/hour on 4 and 5 November.

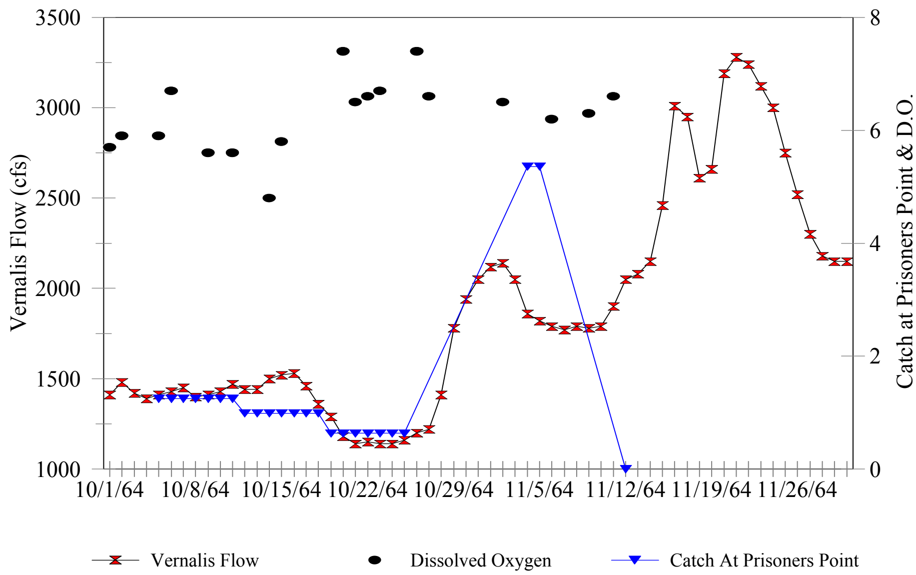

Figure 3 The catch rate of adult chinook salmon with a trammel net at Prisoners Point in the San Joaquin River relative to the flow in the San Joaquin River at Vernalis and DO levels (ppm) near Stockton in October and November 1964

In 1966, catch rates at Prisoners Point remained low throughout October and November when the straying rates of tagged fish were high (71%). A gradual increase in Vernalis flow from 700 cfs from 1 October to 1,350 cfs on 24 October had no obvious effect on catch rates (Figure 5). Likewise, declining export rates from 250% on 1 October to 100% of Vernalis flows on 24 October also had no effect on catch rates (Figure 6). When Vernalis flows increased to about 1,500 cfs on 8 November and exports decreased to 60% between 8 and 19 November, catch rates increased from a steady 0.5 fish/hour to 0.96 fish/hour on 14 November. This small increase in catch rates suggests most of the adults had already migrated into the Delta, and flow releases and/or export reductions after 8 November were already too late to substantially affect straying rates.

In 1967, catch rates at Prisoners Point remained high between 1 and 17 October when straying rates of tagged fish were low (15%). During high catch rates in early October, Vernalis flows ranged between 2,250 and 2,750 cfs (Figure 7), and exports ranged between 80% and 120% of Vernalis flows (Figure 8). When Vernalis flows increased to about 3,500 cfs and exports declined to 25% of Vernalis flows after 27 October, catch rates remained low suggesting most fish had completed their migration through the Delta.

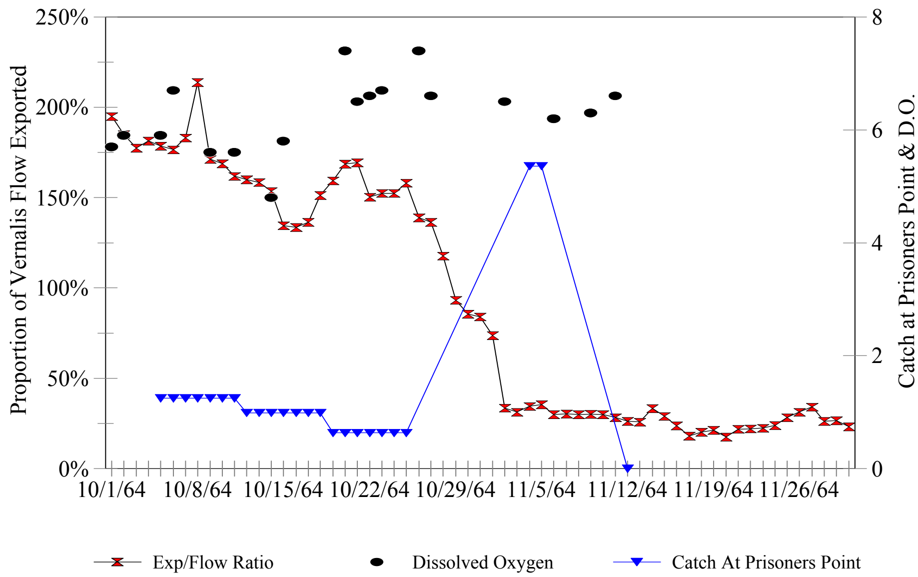

Figure 4 The catch rate of adult chinook salmon with a trammel net at Prisoners Point in the San Joaquin River relative to the proportion of the flow in the San Joaquin River at Vernalis that was exported at the SWP and CVP Delta pumping facilities and DO levels (ppm) near Stockton in October and November 1964

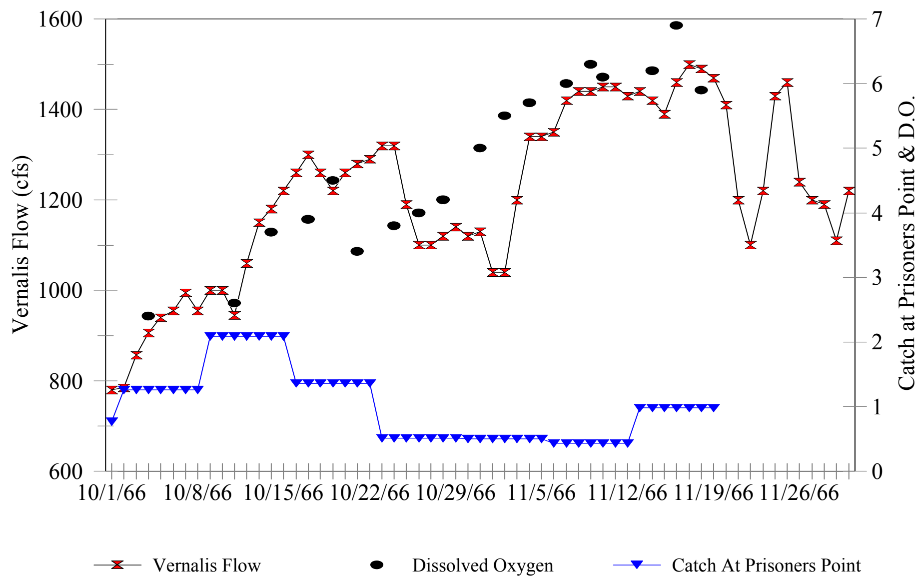

Figure 5 The catch rate of adult chinook salmon with a trammel net at Prisoners Point in the San Joaquin River relative to flow in the San Joaquin River at Vernalis and DO levels (ppm) near Stockton in October and November 1966

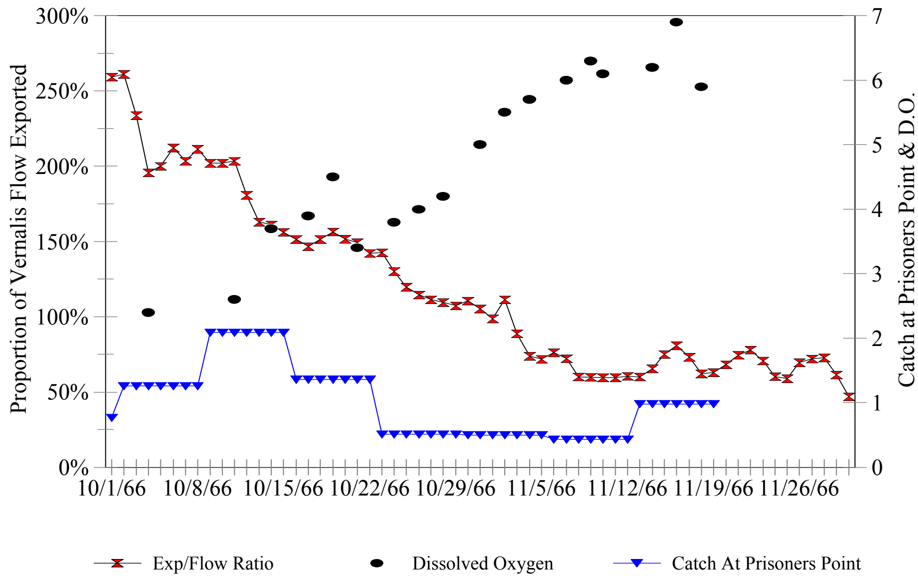

Figure 6 The catch rate of adult chinook salmon with a trammel net at Prisoners Point in the San Joaquin River relative to proportion of the flow in the San Joaquin River at Vernalis that was exported at the SWP and CVP Delta pumping facilities and DO levels (ppm) near Stockton in October and November 1966

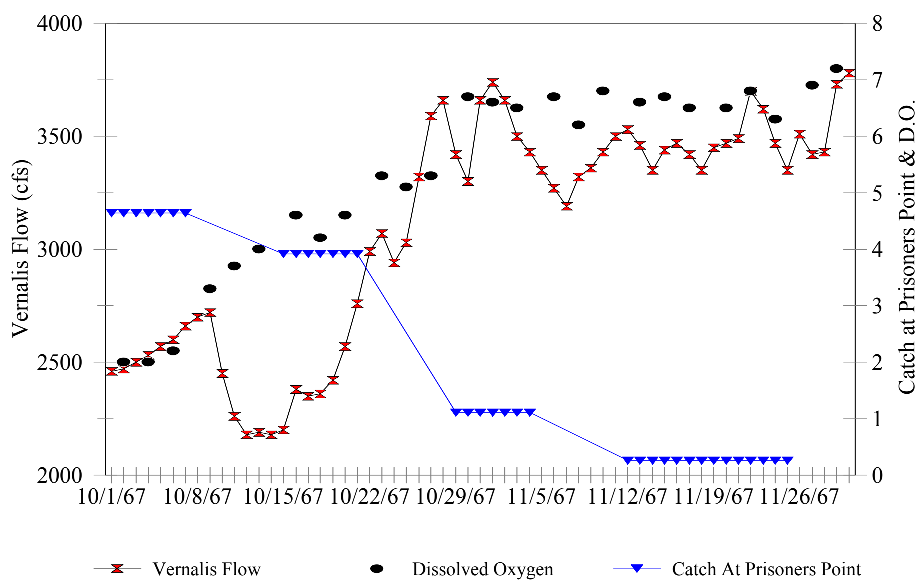

Figure 7 The catch rate of adult chinook salmon with a trammel net at Prisoners Point in the San Joaquin River relative to flow in the San Joaquin River at Vernalis and DO levels (ppm) near Stockton in October and November 1967

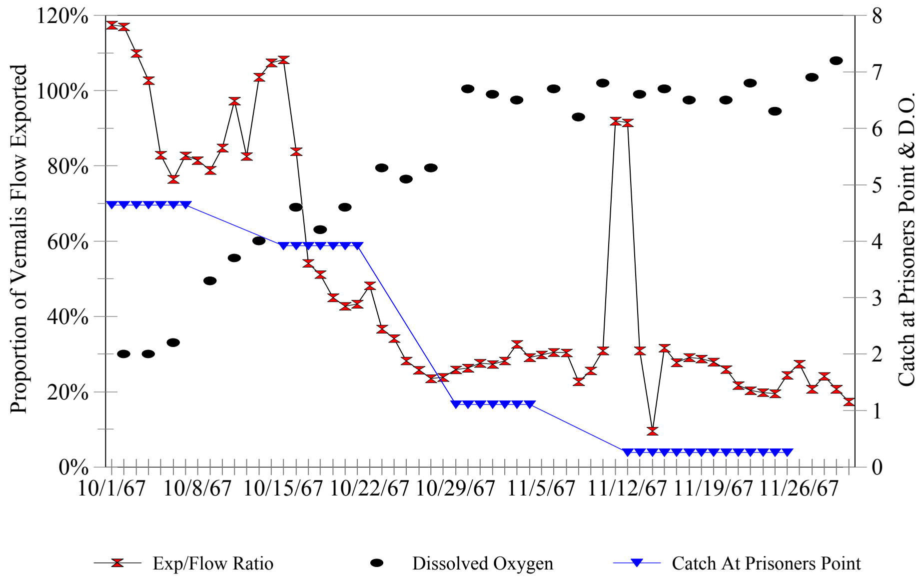

Figure 8 The catch rate of adult chinook salmon with a trammel net at Prisoners Point in the San Joaquin River relative to proportion of the flow in the San Joaquin River at Vernalis that was exported at the SWP and CVP Delta pumping facilities and DO levels (ppm) near Stockton in October and November 1967

There are three important results from Hallock and others (1970) regarding a straying analysis. First, adult salmon were migrating through the San Joaquin Delta near Prisoners Point primarily during October, the period when they are probably most susceptible to low flows and high exports. Second, the fish migrate slowly and do not arrive in the San Joaquin tributaries until about four weeks after they pass Prisoners Point, even when flows, exports, and dissolved oxygen concentrations near Stockton are suitable for migration. And third, migration rates of adult salmon are substantially higher when Vernalis flows exceed about 3,000 cfs and total exports are less than 100% of Vernalis flows. Although most of the tagged fish migrated into the Sacramento and Mokelumne rivers when Vernalis flows were less than about 2,000 cfs and total exports exceeded 150% of Vernalis flows, there is uncertainty as to whether these were San Joaquin fish that strayed or Sacramento River fish that were captured in the San Joaquin on their way to the Sacramento River. The US Fish and Wildlife Service reported that approximately 20% of the Sacramento River fall-run salmon returned to their natal streams by migrating through the lower San Joaquin, into the lower Mokelumne, and then through Threemile or Georgiana sloughs (Erkkila and others 1950). Evidence for this was based on the recapture of 44 adult salmon previously marked at the Coleman National Fish Hatchery as juveniles by the Paladini Fish Company in Pittsburg; nine of the recaptured fish were caught in gill nets drifted in the San Joaquin River below the mouth of the Mokelumne.

Recoveries of Coded-wire Tagged San Joaquin Chinook Salmon

This analysis is based on the number of recoveries of coded-wire tagged (CWT) juvenile chinook salmon in Central Valley streams that were originally reared at the Merced River Fish Facility and the Tuolumne Rearing Facility and released in the San Joaquin basin at Dos Reis Road and all upstream sites. The fish were recovered one to three years after their release when they returned to spawn. If these fish returned to one of the San Joaquin tributaries, they were judged to have successfully "homed." However, if they returned to the Sacramento River basin or one of the eastside streams, which include the Cosumnes, Mokelumne, and Calaveras rivers, they were judged to have strayed. The CWT recovery data were provided by Ralph Carpenter and Robert Kano, Inland Fisheries Division, California Department of Fish and Game (DFG), Sacramento (summarized in Table 1). Updated escapement estimates and the number of fish measured and sexed were obtained from the DFG's annual reports on chinook salmon spawner stocks in California's Central Valley from 1983 through 1989 (Reavis 1986; Kano and Reavis 1996, 1997, 1998; Kano and others 1996). The DFG identifies the 1995 and 1996 CWT recovery data and the escapement estimates from 1990 to 1996 as preliminary (Robert Kano, personal communication, see "Notes").

The accuracy of this analysis is limited because looking for a San Joaquin stray is like looking for a needle in a haystack. The number of spawners in the San Joaquin basin ranges between one and 10% of the numbers in the Sacramento and eastside basins. This means that even if half of the San Joaquin fish stray, the strays would constitute less than 5% of the populations in the Sacramento and eastside basins. Finding the strays is made more difficult because none of the fall-run fish spawning in the mainstem Sacramento River were examined for CWTs through fall 1995. Furthermore, surveys for tagged adults were not conducted every year in the Stanislaus and Merced rivers in the San Joaquin basin, or in the Yuba, American, or Mokelumne rivers in the Sacramento River basin. Overall, the percent of total spawners that was evaluated for tags ranged between 9% and 33% in the San Joaquin tributaries and between 6% and 22% in the Sacramento River basin and eastside rivers.

To compute the total number of salmon with CWTs in each river by survey year, the number of CWT recoveries was divided by the number of fish examined for tags and then multiplied by the escapement estimate. A comparison of the recovery data between the river surveys in the American, Feather, and Merced rivers and the recovery data at the hatcheries in those rivers suggests there are no accurate data to determine the number of fish examined during the escapement surveys. The hatchery data are assumed to be the most accurate, since all fish collected at the hatcheries were fresh and extensively handled, implying that there was a thorough inspection for adipose clips. When the percentage of fish handled during the river escapement surveys was calculated with the assumption that all fish in the carcass counts, those marked and chopped, were examined for adipose clips (and therefore CWTs), the percentage of fish with tags was much lower for the river surveys than those handled at the hatcheries (Table 2).

Table 1 The total number of coded-wire-tags (CWT) recovered by the Department of Fish and Game from adult San Joaquin hatchery reared fish during carcass surveys and at hatcheries in the Sacramento River basin, Eastside tributaries, and the San Joaquin tributaries, and the estimated number of strays and returns and the percent that strayed from 1979 to 1996 [a]

Year	*Total Recoveries*	*Estimated Number of Strays*	*Estimated Number of Returns*	*Percent Strays*
1979	10	7	85	7.6%
1980	26	8	106	7.4%
1981	32	0	361	0.0%
1982	14	4	153	2.2%
1983	300	0	3,129	0.0%
1984	180	32	2,419	1.3%
1985	138	101	1,570	6.1%
1986	149	27	1,519	1.7%
1987	245	680	3,298	17.1%
1988	232	239	1,951	10.9%
1989	120	58	432	11.8%
1990	62	2	137	1.7%
1991	16	6	66	7.9%
1992	74	2	269	0.6%
1993	157	5	269	1.9%
1994	135	10	495	1.9%
1995	237	0	–	0.0%
1996	784	114	2,657	4.1%

[a] Rivers and hatcheries surveyed for CWTs in the Sacramento Basin include Clear Creek, Coleman National Fish Hatchery, Battle Creek, Mill Creek, Red Bluff Diversion Dam, Tehama-Colusa Fish Facility, Feather River Fish Hatchery, Feather River, Yuba River, Nimbus Fish Hatchery, and American River. The Mokelumne River and the Mokelumne River Fish Installation were surveyed in the Eastside tributaries. The Tuolumne River, Stanislaus River, Merced River, Merced River Fish Facility, Los Banos Wildlife Area were surveyed in the San Joaquin basin.

Table 2 A comparison of the percentage of San Joaquin basin adult chinook salmon recovered with coded-wire-tags to the percentages observed at the hatcheries in the Merced, American, and Feather rivers[a]

Merced River								
River Escapement Survey Years	1983	1984	1985	1986	1987	1993	1994	1996
Number of Tags Recovered	5	49	14	13	0	22	38	263
Total Carcass Count	1634	2200	2200	781	426	532	1019	1220
Number Measured & Sexed	1124	448	535	291	138			
Number of Fresh Fish						294	324	147
Number of Fresh Fish & Decayed Adults						517	888	826
Number of Fresh, Decayed Adults & 50% of Decayed Grilse						525	954	1023
Escapement	16453	27640	14841	6789	3168	1995	4635	4599
Merced River Fish Facility								
Number of Tags Recovered	291	146	103	120	26	37	74	291
Number Examined	1795	2109	1211	650	958	409	943	1141
Percentage of Fish Recovered with Tags								
Based on the Hatchery	16.21%	6.92%	8.51%	18.46%	2.71%	9.05%	7.85%	25.50%
Based on Total Carcass Counts	0.31%	2.23%	0.64%	1.66%	0.00%	4.14%	3.73%	21.56%
Based on Number Measured & Sexed	0.44%	10.94%	2.62%	4.47%	0.00%			
Based on Fresh Fish Counts						7.48%	11.73%	178.91%
Based on Fresh & Decayed Adult Counts						4.26%	4.28%	31.84%
Based on Fresh, all Decayed Adults, & 50% of Decayed Grilse						4.19%	3.99%	25.71%

[a] Recovery data for escapement surveys are not presented when no tags were recovered.

Table 2 A comparison of the percentage of San Joaquin basin adult chinook salmon recovered with coded-wire-tags to the percentages observed at the hatcheries in the Merced, American, and Feather rivers[a] (Continued)

American River					
River Escapement Survey Years	1984	1985	1987	1988	1989
Number of Tags Recovered	1	1	3	0	0
Total Carcass Count	10027	4875	9451	3944	5550
Number Measured & Sexed	4875	857	649	908	1070
Escapement	27447	56120	39885	24889	19183
Nimbus Fish Hatchery					
Number of Tags Recovered	3	14	13	6	2
Number Examined	12249	9093	6258	8625	9741
Percentage of Fish Recovered with Tags					
Based on the Hatchery	0.02%	0.15%	0.21%	0.07%	0.02%
Based on Total Carcass Counts	0.01%	0.02%	0.03%	0.00%	0.00%
Based on Number Measured & Sexed	0.02%	0.12%	0.46%	0.00%	0.00%

Feather River				
River Escapement Survey Years	1984	1987	1988	1989
Number of Tags Recovered	4	5	13	4
Total Carcass Count	14603	21714	24099	9677
Number Measured & Sexed	3268	3566	4066	3719
Escapement	41769	67738	42556	40541
Feather River Hatchery				
Number of Tags Recovered	4	75	23	0
Number Examined	9288	10108	6480	7578
Percentage of Fish Recovered with Tags				
Based on the Hatchery	0.04%	0.74%	0.35%	0.00%
Based on Total Carcass Counts	0.03%	0.02%	0.05%	0.04%
Based on Number Measured & Sexed	0.12%	0.14%	0.32%	0.11%

[a] Recovery data for escapement surveys are not presented when no tags were recovered.

Therefore, it is highly unlikely that all of the fish in the carcass counts were closely examined for adipose clips. This is partly true for the San Joaquin tributaries as some of the chopped fish, particularly the grilse, have been too decayed to detect an adipose fin clip and few if any of their heads were taken for CWT evaluation (Jennifer Bull and George Neillands, personal communication, see "Notes"). The problem also appears to have occurred during the escapement surveys in the American and Feather rivers. Even though only fresh fish (clear eyed) were usually marked or chopped in these rivers according to DFG annual reports 1983 through 1989, the percentage of fish with CWTs was also usually much higher at the hatchery than for the river based on the total carcass count (Table 2). On the other hand, when the number of fish measured and sexed during the escapement surveys (DFG 1988–1997) were used to compute the percentage of fish with tags, there was better agreement between the hatchery estimates and the river estimates (Table 2). Therefore, the hatchery data were used to compute the expansion factor for both the hatchery and the corresponding river in most cases. However, in 1989 no CWT recoveries were made at the Feather River Hatchery, whereas four tags were recovered during the Feather River escapement surveys (Table 2). This was very unusual in that usually many more CWT fish were recovered at the hatchery than during the escapement surveys. It was assumed that the Feather River hatchery data were incorrect and the expansion factors for both the river and hatchery were based on the escapement survey for 1989. Whenever total CWT recoveries were estimated for the Feather River in 1989 and for the rivers without a hatchery (primarily the Stanislaus, Tuolumne, and Yuba rivers), the expansion factors was computed using the number of fish measured and sexed, which was available for the 1983 to 1989 surveys. For the 1990 to 1996 surveys, the expansion factor for the Stanislaus and Tuolumne rivers was computed as the number of fresh fish and decayed adults in the carcass counts; decayed grilse were usually not examined for CWTs (George Neillands, personal communication, see "Notes"). The number of fresh fish and decayed adults counted during the escapement surveys in the Merced River provided estimates of "Percentage of Fish Recovered with Tags" that were slightly more similar to the hatchery estimates than estimates computed with the total carcass counts (Table 2).

Since 1984, the DFG has used a trap at Los Banos to collect fish that try to enter the westside agricultural drainage system. DFG Region 4 assumes that approximately half the fish that enter the westside drainage system are recovered at the Los Banos trap (DFG 1988–1997). Therefore, the recoveries for the Los Banos trap were doubled in number to compute the total number of salmon with CWTs entering the westside system.

The total number of CWT strays was computed by summing the estimated total number of salmon with CWTs for each of the Sacramento and eastside rivers and hatcheries surveyed. Rivers and hatcheries surveyed in the Sacramento and eastside basin include Clear Creek, Battle Creek, Mill Creek, Feather River, Yuba River, American River, Mokelumne River, Coleman National Fish Hatchery, Tehama-Colusa Fish Facility, Feather River Hatchery, Nimbus Fish Hatchery, and the Mokelumne River Fish Installation.

The total number of CWT returns was computed by summing the estimated total number of salmon with CWTs for each the San Joaquin tributaries, the Merced River Fish Facility, and the Los Banos trap. No data on CWT recoveries are available for the Stanislaus River for the 1982, 1983, and 1986 surveys. Only the estimates for 1983 were used in the analysis of straying rates, because no strays were recovered in the Sacramento or eastside basins.

The percentage of CWT Merced hatchery fish that strayed was computed using the following equation:

Percent Strays = (Total CWT Strays)/ (Total CWT Returns + Total CWT Strays)

The effects of Vernalis flow and total Delta exports on the estimated Percent Strays was evaluated for four periods. The period from 15 September to 28 October was tested to evaluate whether flow and export conditions affected the homing ability of adult salmon in Suisun Bay that would be present in September and those at Prisoners Point that would be present in October. The period from 1 to 20 October was tested based on the assumption that Hallock's catch data reflected the time when most adult San Joaquin salmon migrated through the Delta. The period from 15 to 21 October was tested to evaluate the ability of short-term pulse flows in mid-October to affect homing behavior. The period from 9 to 15 October was tested to evaluate the peak time of migration based on Hallock's catch data.

The relationship between the estimate of Percent Strays and the ratio of Vernalis flow to total Delta exports for the four periods described above was evaluated for outliers. The estimate for the 1980 survey was relatively high and the estimate for 1981 was relatively low compared to the relationship of the other surveys to flows and exports. Since the number of tags recovered for the 1980 and 1981 surveys was 26 and 32 tags respectively (Table 1), no estimate with less than 33 recoveries was used in the analysis. This eliminated the surveys from 1979 to 1982 and 1991.

Results

The relationship between the estimated Percent Strays and the average ratio of SWP and CVP Export rates to Vernalis flows are shown for various periods in Figures 9 through 12. There are too few data to determine whether the relationship between the estimated Percent Strays and the export to flow ratios was linear, so regression analyses were not conducted. For example, if the

1989 estimate is assumed to be inaccurate, then the Percent Strays estimate appears to increase exponentially relative to the minimum export to flow ratio for both the 1 to 20 October period (Figure 9) and the 15 to 21 October period (Figure 10). However, if the 1987 estimate is assumed to be inaccurate, then Percent Strays appears to have a linear relationship with the minimum export to flow ratios for both periods. Rather than trying to determine the exact nature of the relationship based on the existing data, the uncertainty regarding the true number of fish examined for tags should be resolved first.

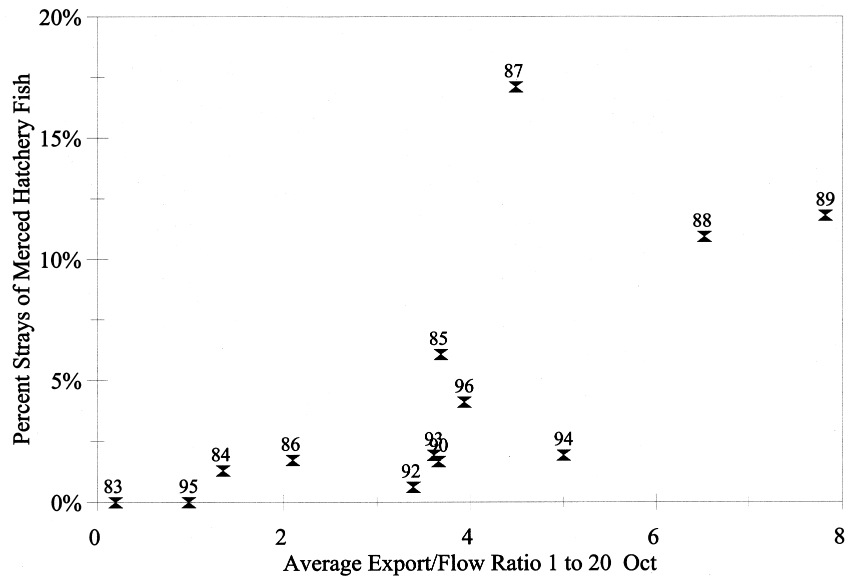

Figure 9 Estimated percent of adult CWT chinook salmon that were reared at the Merced River Hatchery, released in the San Joaquin basin as juvenile fish, and subsequently strayed to the Sacramento River and eastside tributary basins to spawn relative to the average ratio of the export rate at the CVP and SWP pumping facilities in the Delta to the flow rate in the San Joaquin River at Vernalis during 1 to 20 October from 1983 through 1996

A casual inspection of Figures 9 through 14 suggests the estimates of Percent Strays are accurate enough to reach several conclusions in spite of the above uncertainties. First, this analysis indicates that straying rates increase as the percentage of San Joaquin flow exported by the CVP and SWP pumping facilities increases, and the critical period is between 1 and 21 October. Furthermore, pulse flows from the San Joaquin tributaries or a reduction of Delta exports resulting in no more than a 300% export rate of San Joaquin flows at Vernalis for 8 to 12 days in mid-October is sufficient to keep straying rates below 3%. In October 1990, there were eight days when the export rate was less than 300% of Vernalis flows and the estimated straying rate was about 2%. Since 1991, a 300% export rate or lower occurred for at least 10 days in mid October. During most years evaluated when straying rates were less than 3%, San Joaquin River flows at Vernalis were at least 4,000 cfs. However in 1992, straying rates were estimated to be less than 1% when Vernalis flows averaged less than 700 cfs between 1 and 20 October, but Delta exports declined to less than 50% of Vernalis flows for four days and less than 100% of Vernalis flows for eight days. Conversely, straying rates were high, ranging between 11% and 17%, from 1987 to 1989 when between 400% and 700% of San Joaquin flows were exported and Vernalis flows ranged between 1,000 and 2,000 cfs.

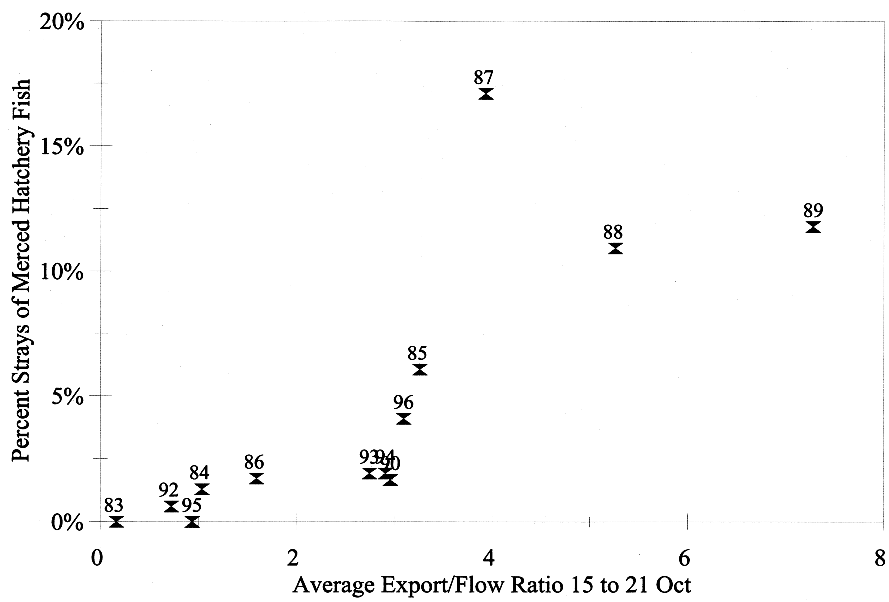

Figure 10 Estimated percent of adult CWT chinook salmon that were reared at the Merced River Hatchery, released in the San Joaquin basin as juvenile fish, and subsequently strayed to the Sacramento River and eastside tributary basins to spawn relative to the average ratio of the export rate at the CVP and SWP pumping facilities in the Delta to the flow rate in the San Joaquin River at Vernalis during 15 to 21 October from 1983 through 1996

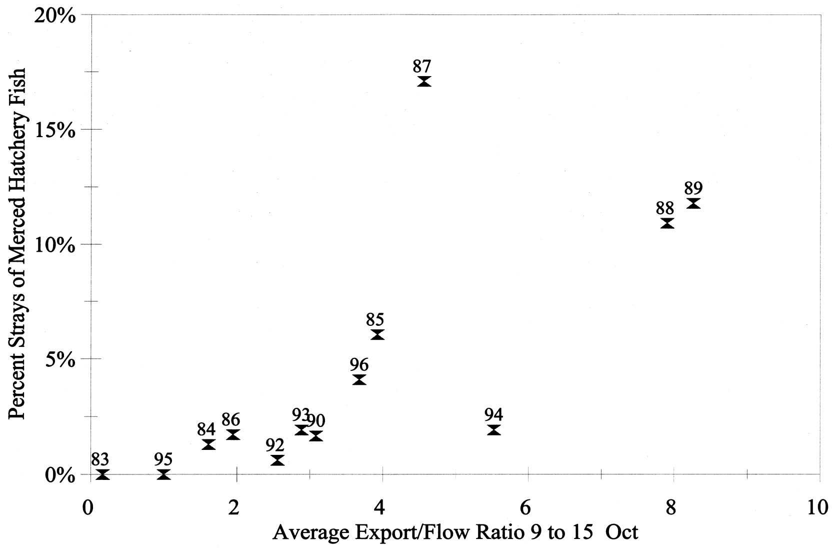

Figure 11 Estimated percent of adult CWT chinook salmon that were reared at the Merced River Hatchery, released in the San Joaquin basin as juvenile fish, and subsequently strayed to the Sacramento River and eastside tributary basins to spawn relative to the average ratio of the export rate at the CVP and SWP pumping facilities in the Delta to the flow rate in the San Joaquin River at Vernalis during 9 to 15 October from 1983 through 1996

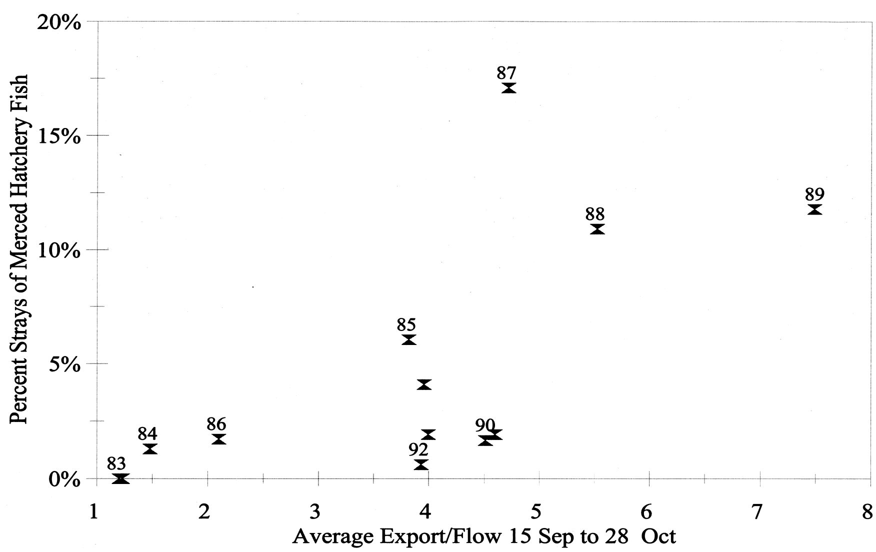

Figure 12 Estimated percent of adult CWT chinook salmon that were reared at the Merced River Hatchery, released in the San Joaquin basin as juvenile fish, and subsequently strayed to the Sacramento River and eastside tributary basins to spawn relative to the average ratio of the export rate at the CVP and SWP pumping facilities in the Delta to the flow rate in the San Joaquin River at Vernalis during 15 September to 28 October from 1983 through 1996

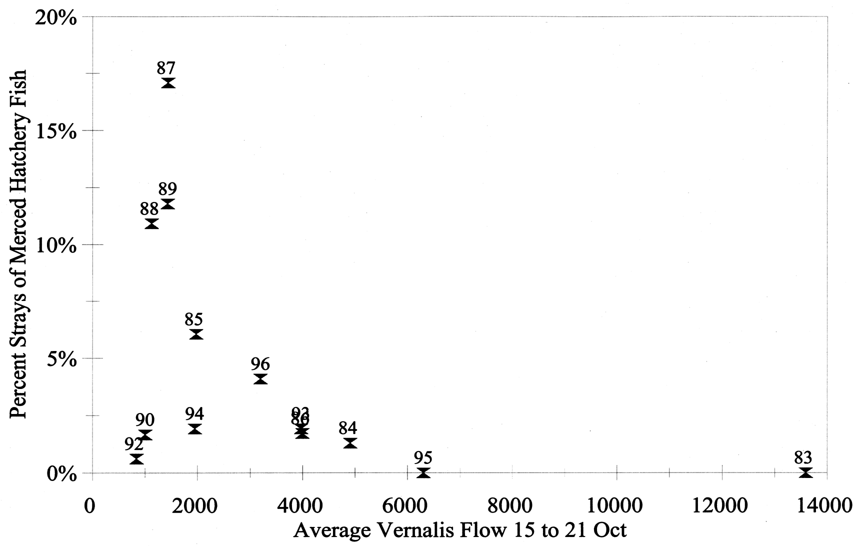

Figure 13 Estimated percent of adult CWT chinook salmon that were reared at the Merced River Hatchery, released in the San Joaquin basin as juvenile fish and subsequently strayed to the Sacramento River and eastside tributary basins to spawn relative to the average flow rate in the San Joaquin River at Vernalis during 15 to 21 October from 1983 through 1996

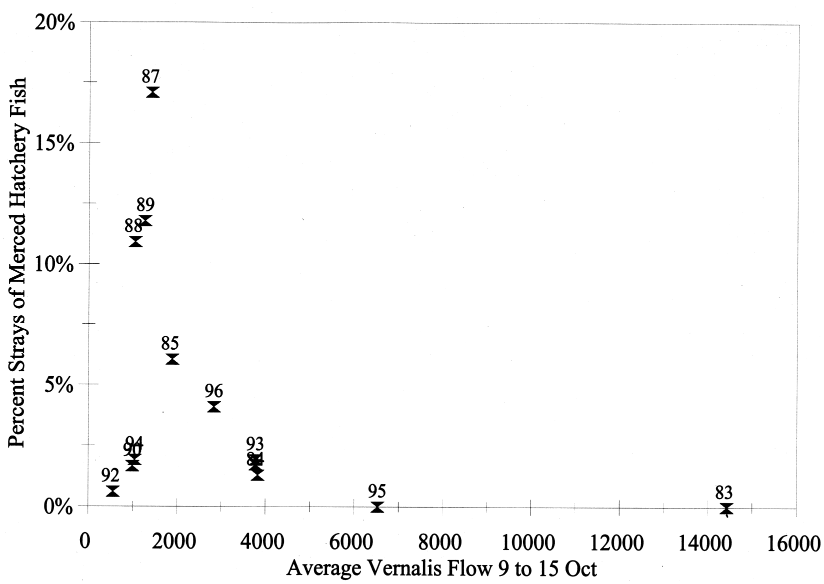

Figure 14 Estimated percent of adult CWT chinook salmon that were reared at the Merced River Hatchery, released in the San Joaquin basin as juvenile fish, and subsequently strayed to the Sacramento River and eastside tributary basins to spawn relative to the average flow rate in the San Joaquin River at Vernalis during 9 to 15 October from 1983 through 1996

Conclusions

The two-part investigation provided conflicting results. Reevaluation of the data collected by Hallock and others (1970) suggested that adult salmon that reared in the San Joaquin tributaries strayed when exports at the CVP and SWP pumping facilities exceeded about 100% of flow in the San Joaquin River at Vernalis and Vernalis flows were less than 2,000 cfs during the first three weeks of October. However, there is uncertainty about the origin of their study fish and data were collected in only four years.

The evaluation of the recovery of coded-wire-tagged fish suggests a maximum of about 20% of adult San Joaquin salmon strayed when Delta exports exceeded about 300% of Vernalis flows for a ten-day period in mid-October. Although the accuracy of the estimated number of strays is questionable, the estimates correlate strongly with the ratio of Delta exports to flows at Vernalis and with Vernalis flows.

Considering the results of these investigations, it is reasonable to assume that when more than 300% of Vernalis flow is exported over a ten-day period in mid-October that adult San Joaquin chinook salmon stray to the Sacramento and eastside basins. However due to the limitations of these analyses, further tests should be made by collecting the data needed to accurately evaluate the recoveries of coded-wire-tagged adults during future carcass surveys. These new data should include the results of annual surveys for adults with coded-wire tags in all major tributaries and the number of fish examined for the tags accurately recorded for each river surveyed. These data, along with accurate escapement estimates, records on the releases of tagged juvenile fish, and records on recovered adult fish with tags, will provide the information needed to accurately estimate the percentage of fish that stray.

Acknowledgements

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Survival of Chinook Salmon Smolts in the Sacramento-San Joaquin Delta and Pacific Ocean

Peter F. Baker and J. Emil Morhardt

Abstract

This paper summarizes current knowledge about the effects of river flow and water export on the survival of San Joaquin River Basin chinook salmon smolts migrating through the Sacramento-San Joaquin Delta. As will become clear, there are serious deficiencies in our understanding of the needs of smolts as they pass through this region, but there is a general agreement that mortality can be high and can probably be reduced by management actions. The potential for success of the various alternatives remains speculative; something needs to be done, but it remains unclear what will work best. For example, smolt survival is usually better at very high (flood) flows than at very low flows, but there is little solid information about the potential for improved survival in the range that might be managed regularly. Researchers have not really begun the search for optimal flows for smolt survival; analyses to date offer, at best, only the qualitative guidance that "higher" flows are "better" for salmon, without any indication of just how much better survival can be or should be. Similarly, although there is reason to believe that strategically placed barriers should improve smolt survival, by keeping smolts well away from the Delta export pumps; however, experiments to date have not been able to demonstrate or refute the effectiveness of such barriers directly.

San Joaquin Chinook Salmon Life History

Only one chinook salmon run, the San Joaquin fall run, is generally recognized in the San Joaquin basin. This run forms spawning populations in the Stanislaus, Tuolumne, and Merced rivers (hereafter called simply "the tributaries"). These populations are distinguished from other Sacramento runs not just by geography, but also in many details of life-history. In particular, the timing of the runs to the San Joaquin tributaries is quite distinct from that of the Sacramento system fall runs.

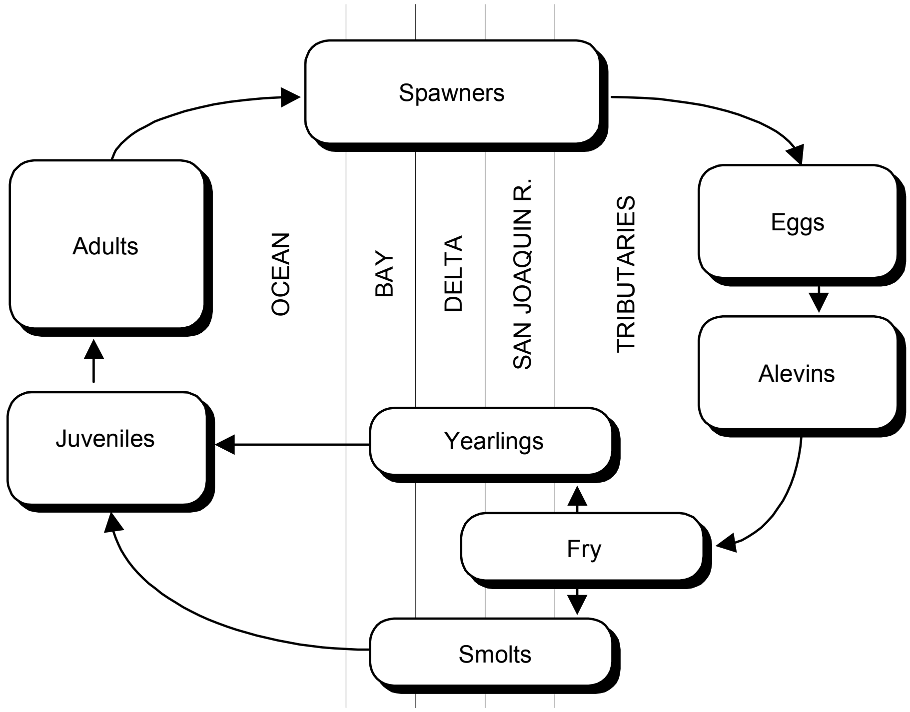


Figure 1 Schematic life history of San Joaquin fall-run chinook salmon. Salmon are vulnerable to export effects during upstream passage through the Delta as spawners, during emigration as smolts or yearlings, and as fry rearing in the Delta.

The life-history pattern of San Joaquin River chinook salmon is shown schematically in Figure 1. Adult chinook salmon typically migrate into the Stanislaus, Tuolumne, and Merced rivers as two-, three-, and four-year-olds. The age composition of the run varies considerably from year to year, but overall about half the migrants are three-year-olds, the remainder divided fairly evenly between two- and four-year-olds. Two-year-olds are disproportionately male, and are often reported separately as jacks, although the percentage of two-year-olds which are female is much higher for the San Joaquin runs than for other chinook salmon stocks, and such females contribute significantly to production in some years. The upstream migrants are collectively called the year's escapement.

The spawning run typically extends from October through December, with the bulk of the run appearing in the tributaries in November. Spawners are occasionally seen in September and are frequently reported in small numbers in January. They begin to construct nests, called redds, and spawn as soon as they arrive in the spawning reaches of the tributaries. Females defend their redds for seven to ten days after spawning. All adults die after spawning.

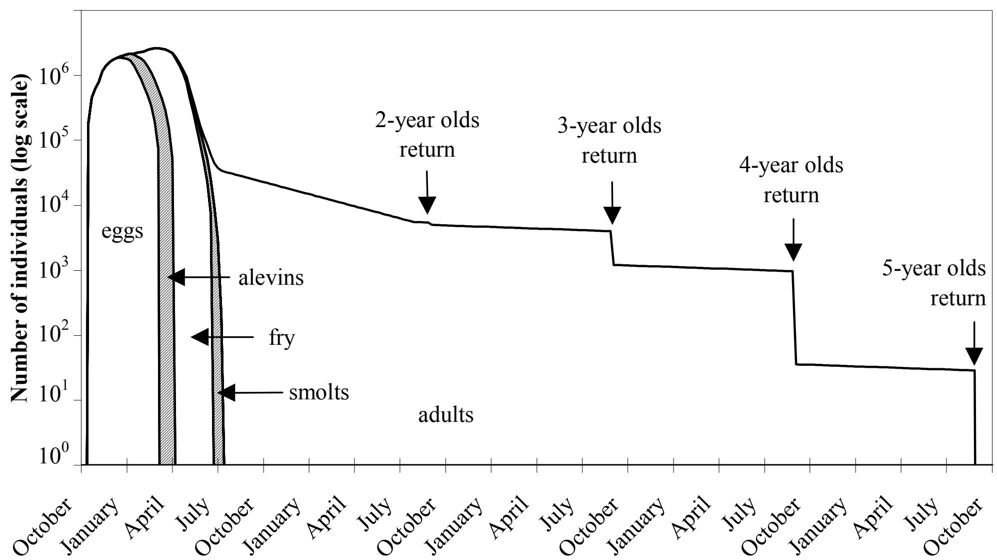


Figure 2 Representative numbers of individuals occurring in different life stages of a typical San Joaquin Basin cohort of chinook salmon. Estimates are derived from average numbers estimated by the EACH dynamic simulation population model.

The development of an idealized cohort over its lifetime is shown in Figure 2. The young fish emerge from the redds as fry from late December through April, with most emerging in February. Some fry soon migrate downstream into the San Joaquin River and the Delta, or are involuntarily displaced from the tributaries by high flows; whether such fry survive to contribute significantly to the total production is not known.

Most fry remain in the tributaries until spring, when they undergo smoltification, a set of physiological changes preparing them for ocean life, and begin their seaward migration. The smolt emigration peaks in April and May, but can extend from late February through June. Some fry do not join the spring emigration, but instead remain in the tributaries over the summer, emigrating in October and November as yearlings. Conditions in the tributaries for summer rearing have been highly variable until recently, however, and is not clear how important these fish have been to total San Joaquin Basin production.

Emigrating smolts experience considerable mortality in the lower reaches of the tributaries, the San Joaquin River, the Delta and San Francisco Bay, and during the first year of ocean life. Smolt mortality in the San Joaquin Delta, in particular, is known to be quite high in most years, and has become a principal focus of efforts to enhance San Joaquin salmon populations: this paper deals primarily with this issue.

Figure 2 illustrates the relative numbers of a typical cohort over the course of its life cycle based on average results from the EACH dynamic population simulation model (EA 1991). A few million eggs are produced in an average year. By the time the developing smolts reach the ocean, their number is reduced by two orders of magnitude. Comparatively minor improvements to survival in these early life stages can greatly improve the numbers of returning adults.

Sources of Information About Smolt Survival

Because of their complex life history, chinook salmon fall under multiple regulatory jurisdictions over their lifetimes. They are studied by many agencies; although there are many exceptions and much interagency coordination, the general tendency is for the California Department of Fish and Game (DFG) to study chinook salmon in the upstream tributaries, the U.S. Fish and Wildlife Service (USFWS) in the Delta, and the National Marine Fisheries Service (NMFS) in the ocean. The DFG's Region 4 annual reports are an important source of information about all stages of San Joaquin Basin salmon from spawning escapement to smolts in the San Joaquin River. The annual reports of the USFWS' Sacramento-San Joaquin Estuary Fishery Resource Office are a principal source of information about smolt survival in the Delta.

Since 1970, research activity by the State and Federal governments into environmental matters in the Delta has been consolidated under the Interagency Ecological Program (IEP). The IEP is a cooperative effort of the DFG, USFWS, NMFS, California Department of Water Resources, State Water Resources Control Board, U.S. Bureau of Reclamation, U.S. Army Corps of Engineers, U.S. Geological Survey, and U.S. Environmental Protection Agency. Activities under the IEP are reported in the quarterly *IEP Newsletter*. Bulk data generated by IEP studies are published electronically and can be accessed at the IEP web site (http://www.iep.water.ca.gov).

Although these are the primary "official" sources of data on San Joaquin salmon, many other entities have conducted studies or published analyses relevant to the needs of salmon in the Delta. Most such material has been presented at the Bay-Delta Hearings, and is part of the Administrative Record for the 1995 Water Quality Control Plan (SWRCB 1995a, 1995b). See also Brandes and McLain (this volume) for additional analyses and another view of survival of Central Valley juvenile salmon moving through the Delta.

Coded-wire-tag Releases Release and Recapture Studies

The principal source of information about smolt survival in the Delta is the recovery of coded-wire-tagged salmon. Coded-wire tags (CWTs) are short lengths of wire, encoded with a group serial number, which are inserted into the heads of the salmon. These tags are not visible externally, so tagged fish also have their adipose fins clipped for recognition. Normally, fish bearing the same tag number are released at the same time and place, although in the past, tags left over from one experiment were occasionally used in another. Adipose fins do not regenerate, so tagged fish can be identified visually throughout their lives. To read the tag number, however, the fish must be killed.

In principle, all tag recoveries are reported to the Pacific States Marine Fisheries Commission (PSMFC), which maintains the Regional Mark Information System database (RMIS). In practice, the conversion from older, local archives is not complete. Information about all CWT releases, and all information about ocean recoveries, is accessible through RMIS; however, inland recovery data from California are most easily obtained through the DFG or the USFWS, depending on the nature of recovery.

CWT experiments are of two sorts. Most commonly, two or more groups are released at approximately the same time, and treatment effects are estimated by comparing the numbers recovered at downstream locations. It is convenient to refer to these as "paired release" experiments, although more than two groups may be involved. The virtue of this approach is that if the releases are arranged so that both groups reach the recovery locations at approximately the same time, estimates of relative survival between groups can be calculated using only qualitative assumptions about the sampling procedures used. Sometimes it is necessary to estimate absolute, instead of relative, survival, in which case additional information is needed, such as the probability of capture. Such estimates are often referred to as "survival indices" to alert readers to the extra level of uncertainty. The CWT experiments most relevant to San Joaquin salmon issues can be grouped as follows.

Upstream Survival Experiments. In a long-standing series of experiments, CWT groups are released in the Merced, Tuolumne, and Stanislaus rivers to investigate in-river migration and survival. These are always arranged as paired releases; one group is released "upstream" (usually near the passage-blocking dam), and another group is released "downstream" (usually near the mouth) in the same river a few days later. These releases are often further coordinated with releases at Mossdale or Dos Reis, to provide paired-release data for survival in the San Joaquin River between the mouths of the tributaries and the Delta.

Old River-San Joaquin River Survival Experiments. In another long-standing series of experiments, CWT groups are released in the vicinity of the Old River-San Joaquin River split. These are usually arranged as paired releases, groups being released simultaneously in two of the following three locations: Mossdale on the San Joaquin River (upstream of the split), Dos Reis on the San Joaquin

River (downstream of the split), and Stewart Road on Old River (downstream of the split). These releases are often further coordinated with releases at Jersey Point.

Lower San Joaquin River Survival Experiments. In 1991, two sequences of CWT releases were made at locations along the San Joaquin River between the head of Old River and Jersey Point: groups were released at Dos Reis (River Mile 50), Buckley Cove (RM39), Empire Tract (RM29), Lower Mokelumne (RM19), and Jersey Point (RM12) on April 15, 16, 17, 18, and 19, respectively, and again at Buckley Cove, Lower Mokelumne, and Jersey Point on May 6, 9, and 13, respectively.

Interior South Delta Survival Experiments. In many years CWT groups are released in Old River at Palm Tract. These are usually coordinated with releases at Stewart Road on Old River or at Mossdale on the San Joaquin River.

Trawl Surveys

Since 1978, as part of IEP, the USFWS has monitored the relative abundance of chinook salmon smolts emigrating from the Central Valley with mid-water trawl surveys at Chipps Island. The sampling effort varies over course of the season, but during the peak of the emigration season is typically at its maximum level of ten 20-minute trawls per day, seven days per week. Smolts with adipose fin clips are killed and their CWTs are read. The number of smolts captured is expanded to account for the amount of time spent sampling and the ratio of the net width to channel width to form an estimate of absolute abundance. For CWT-bearing smolts, the expanded recovery for each tag group is divided by the number of smolts originally released and reported as a smolt survival index (SSI).

The Chipps Island trawls are of special importance for investigating questions of Delta smolt survival, because this trawl location can be loosely regarded as marking the transition from delta to bay environments, and because data have been gathered quite consistently at this location for two decades. In the spring of 1997, as part of the Vernalis Adaptive Management Plan, a new USFWS trawl survey location was added at Jersey Point on the San Joaquin River, to supplement the Chipps Island data with data more specific to San Joaquin salmon populations.

Since 1989, DFG has conducted similar monitoring in the San Joaquin River near Mossdale Landing County Park, just upstream of the head of Old River. Ten 10-minute trawls are conducted during a five-hour "index" period each day, typically for 5 days each week during the peak of the emigration season. The number of smolts captured is expanded by an efficiency index obtained by experiments in which smolts marked with subcutaneously-injected paint are released a short distance upstream of the trawl location. Sampling at this location is expected to become more consistent and intensive in future years.

Smolts Captured at the Delta Water Export Pumping Stations

Both the federal government's Central Valley Project (CVP) and California's State Water Project (SWP) export facilities in the South Delta include systems for the salvage of entrained fish: the Tracy Fish Collection Facility at the CVP's Tracy Pumping Plant and the John E. Skinner Fish Protection Facilities at the SWP's Harvey O. Banks Pumping Plant. In both cases, fish entrained at the facility are diverted by screens into a separate system of bypasses and holding tanks, from which they are loaded onto trucks for transport and release at one of two locations at Sherman Island. The salvage facilities are operated by USBR (Tracy) and by DWR and DFG (Skinner).

The salvage release locations are upstream from Chipps Island. Salvaged smolts are therefore vulnerable to recovery in the Chipps Island trawls, creating difficulties in interpreting Chipps Island trawl data: one doesn't know the route of the tagged smolts. Did they arrive through export salvage operations or on their own through Old and Middle rivers?

At both facilities, samples are taken at regular intervals by diverting the entire fish salvage flow into a separate holding tank. All salmonids in each sample are counted and measured, and used to estimate total salvage numbers. Salmon with clipped adipose fin are killed and their tags are read.

In addition to this regular sampling, the entire bypass system is flushed from time to time to remove predators that have taken up residence. A complete census is taken of the fish present, and all tagged smolts are killed and their tags are read.

Ocean Recoveries

Chinook salmon from the San Joaquin Basin are captured as adults in the commercial and sport fisheries. Detailed information about ocean recoveries in general, and CWT recoveries in particular, is collected by state, provincial, and federal agencies of the United States and Canada, and maintained by the PSMFC in the RMIS database.

Adult Escapement Estimates

From the size of the escapement it is possible to draw inferences about the survival of the adult salmon as smolts. In the San Joaquin Basin, all escapement estimates are based on carcass surveys, or returns to the Merced River Fish Facility, except for a few years in the early 1940s when counting weirs were used.

Management and Smolt Survival

Although it is generally recognized that considerable smolt mortality occurs between the mouths of the San Joaquin tributaries and the Delta, this mortality is not usually addressed directly. It is usually assumed that flow requirements upstream (for the benefit of fry and smolts in the tributaries), and downstream (for the benefit of smolts in the Delta), would equally benefit smolts in the San Joaquin River itself.

Smolt survival in the San Joaquin Delta is known to be poor, and there are many factors that could plausibly be manipulated to the benefit of survival. Foremost among these are the "usual suspects" in inland fisheries problems: flows, diversions, and water quality.

Flow and Export

As described above, the needs of smolts in the Delta have been studied by releasing large numbers of smolts marked with coded-wire tags upstream of the Delta and recovering them downstream of the Delta (near Chipps Island, in the ocean fisheries, or as returning adults). Researchers relate the observed recoveries to variables like flow and export in an attempt to determine empirical relationships that could be used to guide policy decisions.

This black-box approach, although it ignores the underlying mechanisms causing observed changes in survival, has merit. After all, the ultimate goal is to enhance salmon populations through management. If it could be shown that certain management actions would enhance survival, it would not be necessary to know why they did, or how survival depends on factors outside management control.

Unfortunately, this approach has not been entirely effective in the Sacramento-San Joaquin Delta. Although relationships between Delta smolt survival, flows, and exports have been the subject of investigation for many years, there is surprisingly little agreement on the value of management actions deriving from these relationships.

There are at least three reasons why these experiments have been so unsatisfactory:

- The data sets are small. Only a few points are added by each year's experiments.
- Recapture numbers are generally small, and expansion to survival indices is highly uncertain.

- Many potentially confounding factors cannot be satisfactorily controlled or taken into account.

The last reason is probably the most fundamental. The South Delta is a complex environment for smolts from the black-box point of view, some factors are simply distractions which contribute a great deal of noise. Increasingly, researchers been compelled to study the mechanisms by which flow and export affect smolt survival in an effort to divide the problem into more digestible pieces. Two major steps have been taken in this direction.

The first step has been to separate the dual role of export on smolt survival. Export affects smolts directly, by entraining fish at the facilities, and indirectly, by altering Delta flow patterns. The direct entrainment effects can be studied through mortality experiments, screen efficiency experiments, fish salvage records, and so on. The effects of export on Delta flow patterns are naturally treated in combination with those of inflow, with the help of hydrodynamic modeling.

The second step has been to divide in-Delta flow effects on smolt survival into two parts: first, the effects of these flows on the routes taken by smolts through the Delta, and second, survival along individual routes. This step is motivated by the fact that smolt survival often varies greatly from one part of the Delta to another. The clearest expression of this comes from a series of six experiments conducted by USFWS between 1986 and 1990 (Figure 3). In each of these experiments, two groups of smolts were released on the same date in the Lower San Joaquin River and in Old River. Both release sites are a short distance downstream of the Old River-lower San Joaquin River split, but the two groups would be expected to take different routes through the Delta. The lower San Joaquin River group survived better than the Old River group in all six experiments—a result which is already significant, with no further statistical assumptions, at the 98% confidence level. Overall, smolts released in lower San Joaquin River were more than twice as likely to reach the recovery site at Chipps Island than were smolts released in Old River.

Current efforts to understand the scope for improving smolt survival through flow and export management are thus organized around the following questions:

- How do San Joaquin River flow and CVP-SWP export affect in-Delta flows?

- How do in-Delta flows affect smolt migration patterns?

- How do in-Delta flows affect smolt survival along particular migration routes?

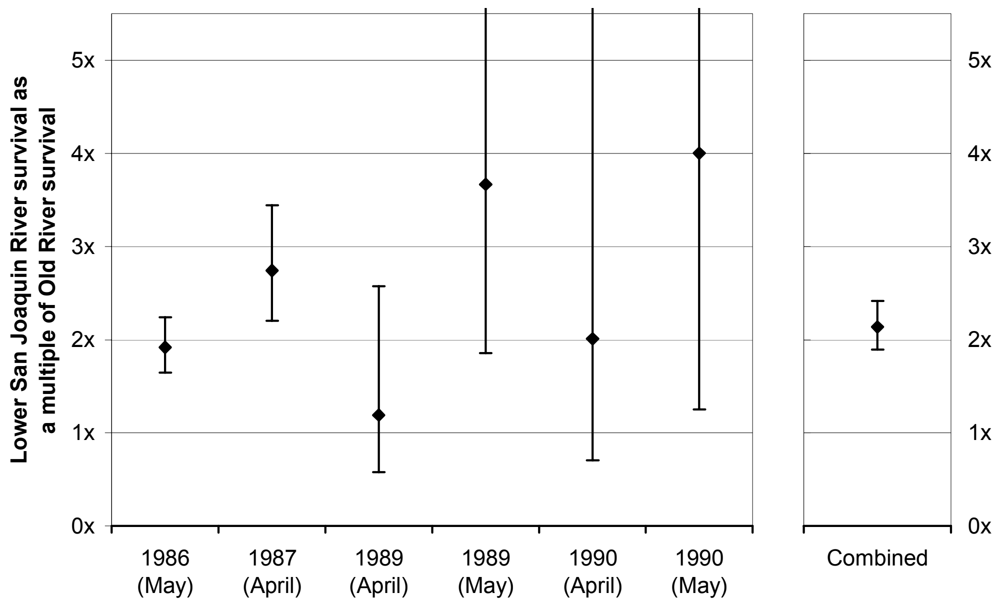


Figure 3 Survival of smolts released in lower San Joaquin River at Dos Reis, as a multiple of the survival of smolts released in upper Old River at Stewart Road, based on recoveries in trawls at Chipps Island and in the ocean fisheries. A value of 1x represents equal survival for both release. The survival ratio for all experiments combined was 2.14. The confidence intervals (95%) are calculated assuming that capture for each group at each location is a Poisson process and should be regarded as conservative.

San Joaquin River Flow, CVP and SWP Exports, and In-Delta Flows

In principle, the relationships between San Joaquin River flow, CVP-SWP export, and in-Delta flows are completely knowable, with the help of hydrodynamic models. There are several of such models in current use and more under development. Although there are important differences between these models, it may be safely said that the hydrodynamics of the Delta are understood far more thoroughly than are the effects of these hydrodynamics on Delta biota.

Two basic facts about Delta hydrodynamics important to emigrating smolts are (1) tidal flows are much larger than the tidally-averaged, or "net" flows, and (2) Old River is a principal channel through the Delta, typically receiving well over half the total flow of the San Joaquin River even in the absence of export.

It would be difficult to exaggerate the difference in magnitude between net and tidal flows. From water year (WY) 1940 through WY 1991, the average flow at Vernalis was 4,550 cfs, and the highest annual average flow over this period was 21,281 cfs (WY 1983). In the San Joaquin River near Columbia Cut and the mouth of Middle River, typical summer flows swing from roughly 50,000 cfs westward to 50,000 cfs eastward, and back again, each day (DWR 1993). At the confluence of the San Joaquin and Sacramento rivers, the typical daily excursion in each direction exceeds 300,000 cfs.

In-Delta Flows, Smolt Travel Time, and Smolt Migration Patterns

There is little theory available on the mechanisms by which smolts find their way through the estuary, or about how these mechanisms are affected by flow. Much of what is currently known about emigration mechanisms is negative. For example, the most straightforward model, that the movement of smolts mirrors the movement of water, has be shown to be incorrect. Smolts and water travel through the Delta at very different rates, and end up at very different places.

San Joaquin smolts pass through the Delta in a median time of 11 days, some arriving at Chipps Island as early as five days after release at the point where the San Joaquin River joins the Delta, and some taking as long as 26 days (Figure 4). This is considerably shorter than the transit time for neutrally-buoyant tracer particles, at least in hydraulic simulations. Figure 5 shows an example comparing the speed of smolt passage and the speed of tracer particles for a release made on April 4, 1987, in which 80% of the smolts were estimated to have been recovered after two weeks, but only 0.55% of the tracer particles were recovered after two months. (The estimated survival for this smolt group was atypically high, but the transit time was not. Typical survival estimates for smolts are still much larger than 0.55%.)

Not only do the tracer particles which reach Chipps Island take a long time to get there, but most of them go somewhere else. That somewhere else is the CVP and SWP pumps, at least for the hydraulic simulations available to us. Figure 6 shows that for the April 27, 1987 simulation, 77% of the tracer particles ended up at the export pumps, while only 13% of the smolts arrived there.

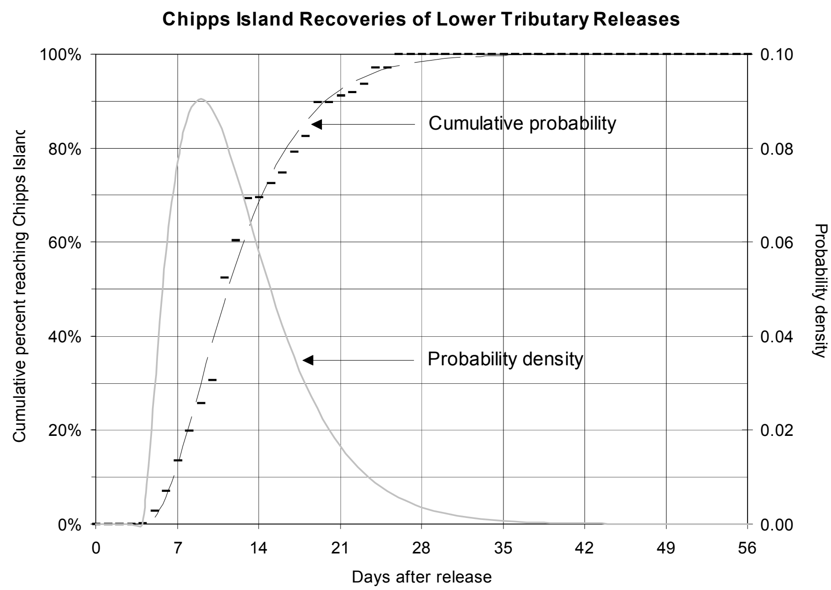


Figure 4 Empirical pattern of smolt recovery (cumulative) at Chipps Island as a function of days after release in Merced (1989), Stanislaus (1986, 1988, 1989), and Tuolumne (1986, 1987, 1990) rivers. The dashed (---) line indicates smoothed recovery (cumulative); the gray line indicates probability density of reaching Chipps Island based on smoothed recoveries. After release, the fastest smolts arrived at Chipps Island in five days and the slowest in 26 days. Peak recoveries occurred on the tenth day after release, and half of the fish had arrived within 11 days.

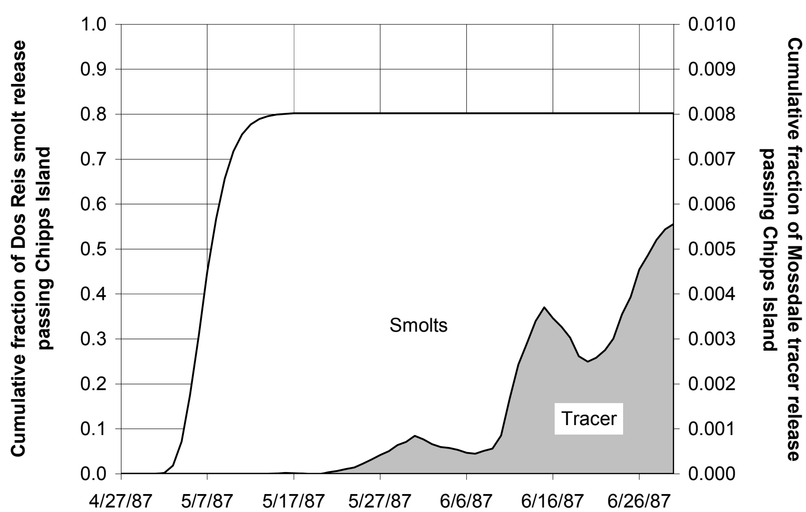


Figure 5 Comparisons of the movements of salmon smolts and passive particles released near the head of Old River on April 27, 1987. Cumulative recoveries at Chipps Island of smolts released at Dos Reis, and simulated mass flux past Chipps Island of tracer material released at Mossdale. The smolt recovery data have been fitted to an inverse gaussian distribution. Hydraulic simulations by Flow Science (1998).

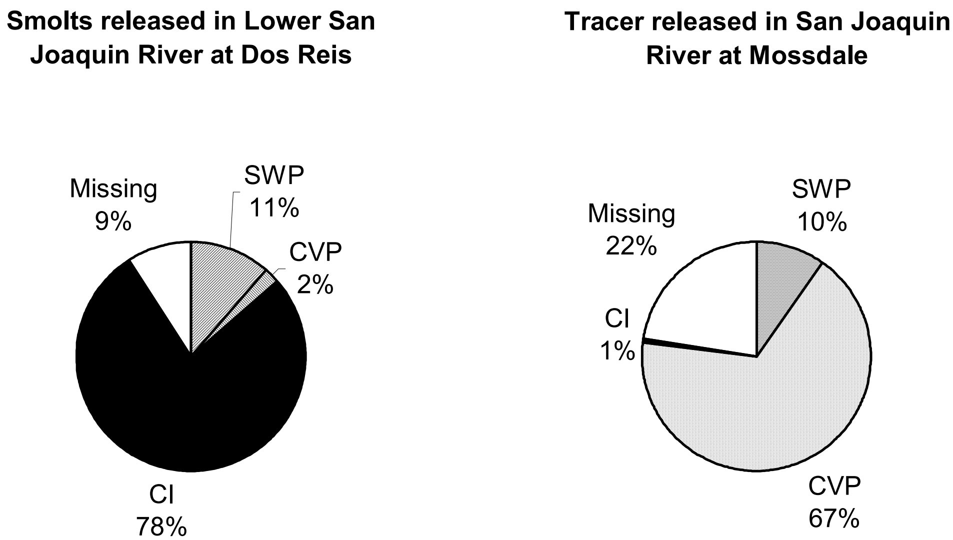
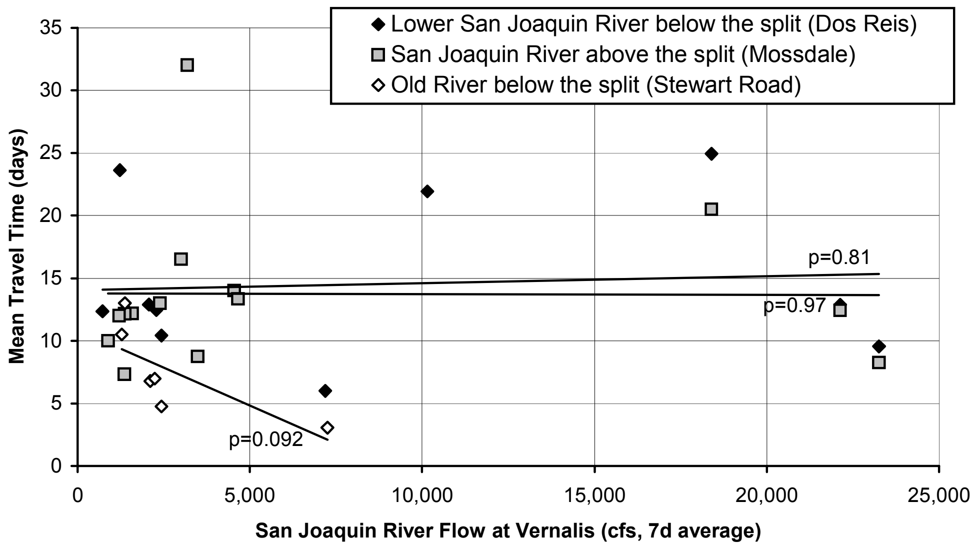


Figure 6 Comparisons of the final destinations of salmon smolts and passive particles released near the head of Old River on April 27, 1987. Estimated final disposition of tagged chinook salmon smolts released at Dos Reis and simulated disposition as of June 30, 1987 of tracer material released at Mossdale. For the smolts, the CVP and SWP values represent total entrainment, including estimates of screen inefficiency and mortality in Clifton Court Forebay, and the Chipps Island value represents successful emigration exclusive of release after salvage. Hydraulic simulations by Flow Science (1998).

Initially it seems intuitively reasonable that increased flows entering the Delta from the San Joaquin River at Vernalis would decrease travel times and speed passage, with concomitant benefits to survival. The data, however, show otherwise. Figure 7 (top) shows that Delta inflow has little if any effect on smolt travel time, probably because the large tidal flows swamp any passive effect of the incoming flows from the San Joaquin River, as suggested by the particle tracking results. On the other hand, Figure 7 (bottom) shows that the larger the smolts at the time of release, the shorter the travel time. This is in accordance with the striking difference between the passage time of smolts and passive particles: smolts actively swim toward the ocean, and the bigger they are the faster they do it.



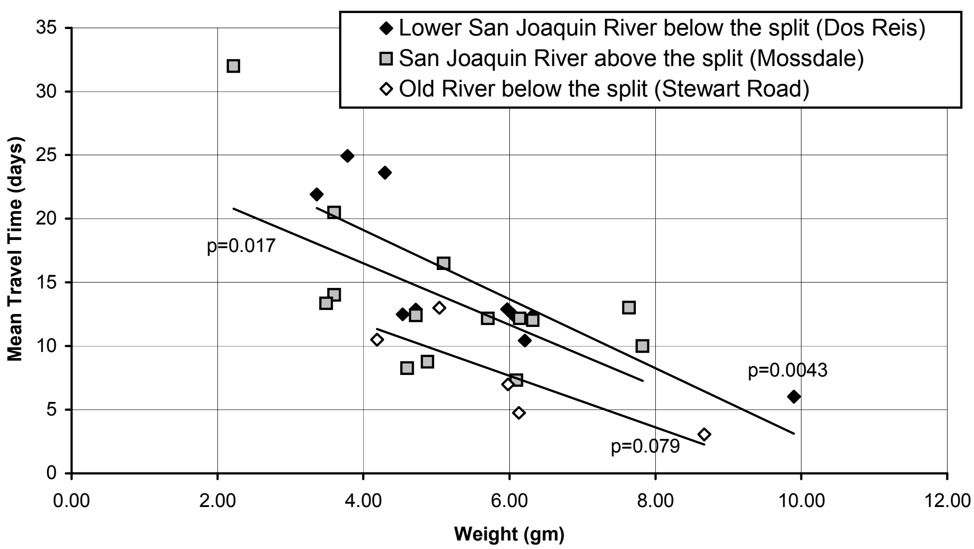


Figure 7 Mean smolt migration times from three locations near the Old River-San Joaquin River split to Chipps Island. The vertical ordering of the three trendlines in each plot agrees with the vertical ordering of the corresponding release locations in the legend. Top: Migration time and San Joaquin flow for the seven days following release. The regression for the Old River releases is significant only at the 90% confidence level, the other two are not significant at any acceptable confidence level. Bottom: Migration time and smolt weight at release. The regressions for both the San Joaquin and Lower San Joaquin releases are both highly significant (99% and 98% confidence levels, respectively). The regression for the Old River releases is only significant at the 90% level, but is still better than the corresponding regression with flow.

Choice of Routes Through the Delta

When arriving at the Delta from the San Joaquin River, smolts have a choice of routes, the initial decision of which is whether to remain in the larger channel, Old River, at the point that the San Joaquin River diverges toward the north. This decision is critical to their survival, because the Old River channel soon branches into two meandering through channels (Old and Middle rivers), a number of major canals (Grant Line, Fabian and Bell, Victoria), and various dead-ends (Paradise Cut, Tom Paine Slough). The through channels and canals all deliver smolts to or near the intake structures for the CVP and SWP pumping plants.

Under conditions of no export pumping, about 60% of the water arriving via the San Joaquin River goes down the Old River channel; as pumping increases, that proportion can increase to 100% (Figure 8). If smolts simply traveled at a fixed speed relative to the water they were in, one would expect 60% or more of them to go to the pumps as well. In fact, in the few experiments that have been done, the results show an even higher percentage of the smolts go down Old River than would be expected if they simply went with the flow. Figure 9 shows the results from a series of daily trawls in the San Joaquin River and in Old River below the flow split. The results are expressed as the number of naturally migrating smolts captured per 10,000 m^3 of water sampled. If the smolts were simply following the flow, their concentrations in the two rivers would be identical. In fact, most of the daily data points occur well above the line of equal concentration, showing a higher concentration of smolts in Old River.

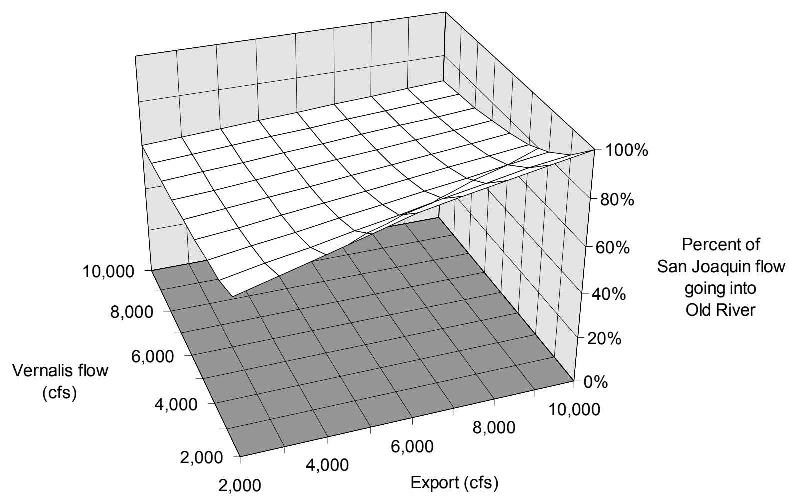


Figure 8 Percentage of net flow (calculated from 1986 DWR net flow equations) in the San Joaquin River at Vernalis flowing into Old River. At least 59% of the flow goes into Old River at any Vernalis flow, but as much as 100% can flow into Old River if Delta pumping is high.

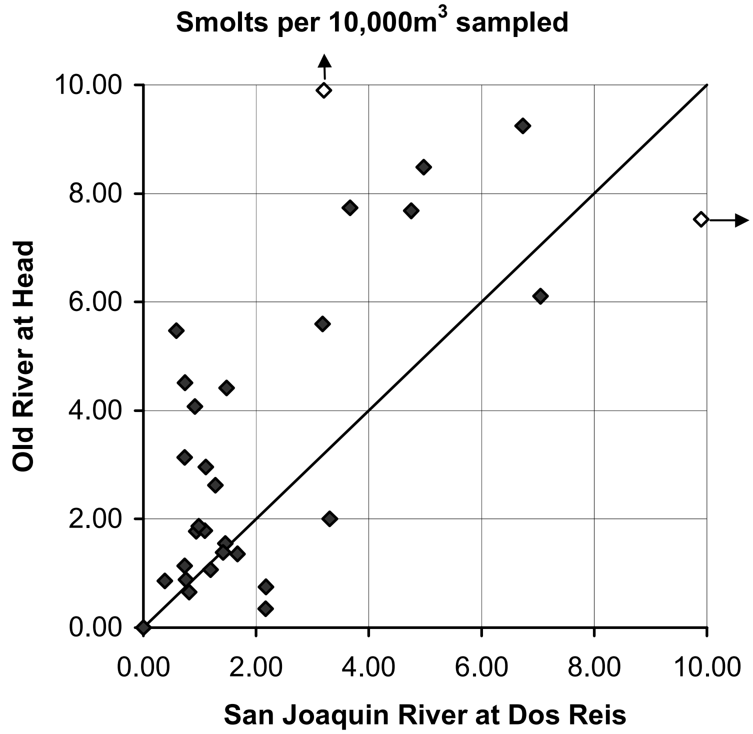


Figure 9 Daily smolt densities from 1996 real-time monitoring program from April 1 to May 6. These are unmarked natural smolts. If the proportion of smolts in each channel exactly followed flow, the data would all lie on the diagonal line. The data tend to lie well above the line, however, suggesting a preference on the part of the smolts for the Old River channel. The two open diamonds were well off-scale at 12.5 for the upper one and 18.7 for the one on the left axis, so we left them off to better visualize the majority of the data.

In-Delta Flows and Smolt Survival

Most of the USFWS CWT experiments in recent years have attacked the problem of relating survival along a given migration route to Delta hydrodynamics. In these experiments, two basic migration routes are recognized: down Lower San Joaquin River (past Stockton), or down Old and Middle rivers (past the export pumps). Delta hydrodynamics are represented by calculated net flows in Lower San Joaquin River at Stockton, and in Old River between its head and the split with Middle River, respectively.

This work has so far been inconclusive. There is a significant (P = 0.049) correlation between survival in Lower San Joaquin River and San Joaquin flow at Stockton. This relationship is no better (or worse) than that with San Joaquin flow at Vernalis, and thus sheds little light on what the underlying mechanisms for such a relationship could be. There is no empirical correlation at all between survival in Lower San Joaquin River and the rate of CVP-SWP export.

Results so far on survival in Old River have been even more unsatisfactory. Taken at face value, multiple regression of survival vs. flow in Old River and CVP-SWP export leads to the conclusion that increased export would improve smolt survival along this route (presumably an artifact of the strong contribution of export to Old River flow). As with the Lower San Joaquin River, the problem is that the degree of scatter, and lack of good controls, makes interpretation difficult.

Beginning in 1997, major changes have been made to the design of South Delta CWT experiments. These changes are expected to result in higher recapture numbers (leading to more precise estimates of survival), better control of flow and export conditions during individual experiments, and some degree of statistical design in the combinations of flow and export to be tested. It is too soon to tell whether these improvements will lead to a clearer understanding of the effects of flow and export on survival, but results so far are encouraging.

Vernalis Flows and Smolt Survival

Figure 10 shows the relationship between the USFWS smolt survival index for CWT tagged fish and the flows in the San Joaquin River at Vernalis, just before the flow split between the lower San Joaquin River and Old River. Shown on the figure is a simple linear regression and the 95% confidence intervals. The data points are grouped in the regions of moderately low flow and quite high flow, with no data at all between 11,000 cfs and 18,000 cfs. The flows over 18,000 represent periods when the tributaries are spilling from the dams, and are essentially at flood stage; such conditions are probably very important to fish, but cannot be provided on demand by reservoir operators. When only the data below 10,000 cfs are considered, there appears to be a negative relationship between flow and smolt survival.

There are two ways to think about these data. One school believes that there is, in fact, a linear positive relationship between flow and smolt survival and that, on average, one could expect to get a survival improvement through the Delta corresponding to the slope of the regression line in Figure 10. The other school suspects that different mechanisms are at work at flood flows than low or moderate ones, and there is little reason to believe that altering flows within the lower range will have much effect on smolt survival through the Delta. Data from the middle range of flows will help, but the data are very scattered and factors other than flow are obviously influential.

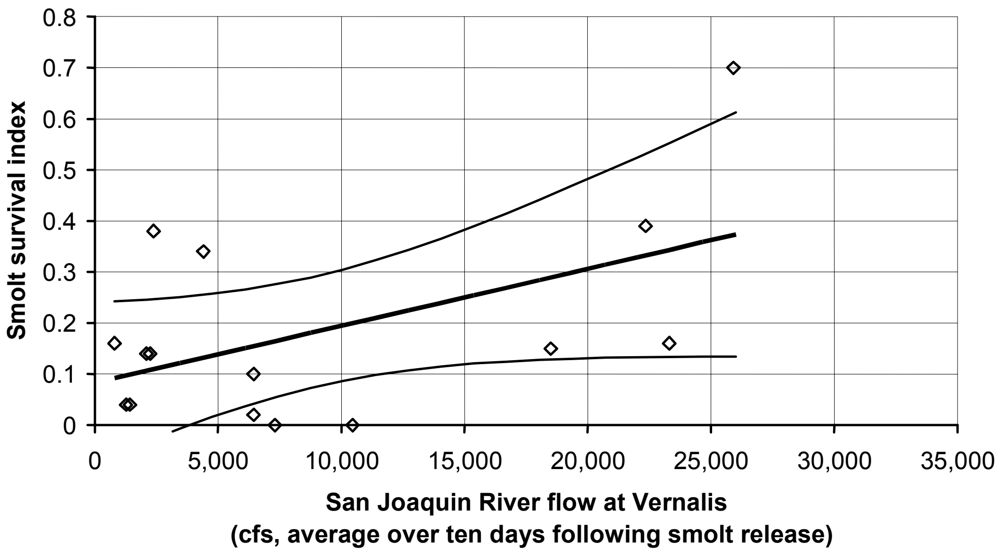


Figure 10 USFWS smolt survival index for tagged smolts released in the lower San Joaquin River at Dos Reis and average flow in the San Joaquin River at Vernalis over the 10 days following tag release. Fitted regression line and envelope of 95% confidence region for fitted line are shown.

Vernalis Flows and Escapement

Another way to look at the effects of flow at Vernalis is to examine the escapement as a function of flows when the escapees were smolts. Figure 11 shows such a result, based on the simplifying assumptions that all adults returned 2.5 years after their emigration as smolts and that in every year there were the same number of smolts. The results are similar to those for the smolt survival relationship with Vernalis flow, but with considerably more data and consequently, with narrower confidence limits. As with the smolt data, there is a clear relationship when high flows are included in the analysis, but at flows below 10,000 cfs there is very little correlation between flows at Vernalis and escapement, and there is a very large amount of scatter in the data. The scatter is undoubtedly partly attributable to failure of the two assumptions, but efforts to correct for these assumptions have not been particularly successful, so there are likely to be other issues as well.

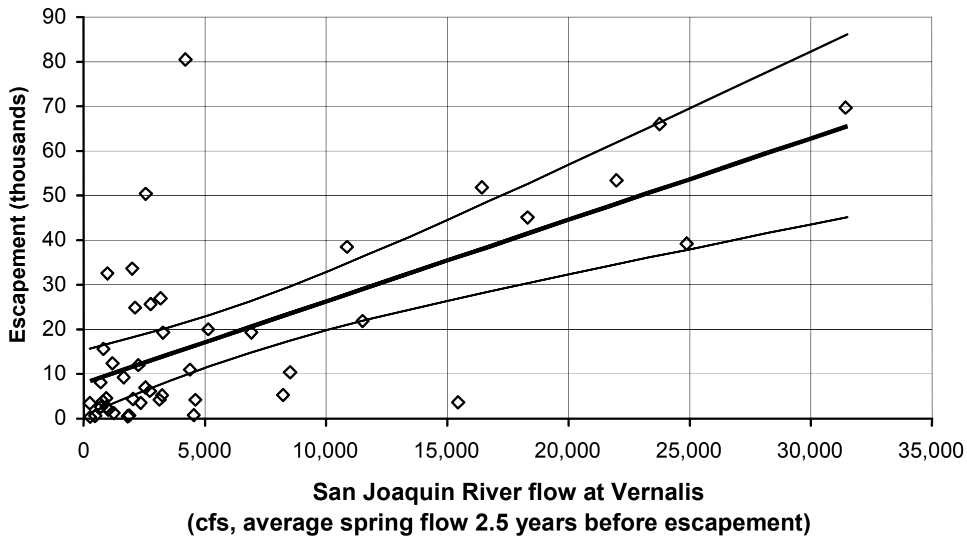


Figure 11 Total escapement to San Joaquin tributaries, 1951 through 1996, and spring flow in the San Joaquin River at Vernalis 2.5 years earlier. Fitted regression line and envelope of 95% confidence region for fitted line are shown.

Conclusions

Smolt survival through the Delta may be influenced to some extent by the magnitude of flows from the San Joaquin River, but this relationship has not been well quantified yet, especially in the range of flows for which such quantification would be most useful. Salvage records show clearly that export-related smolt mortality is a major problem, but no relationship between export rate and smolt mortality, suitable for setting day-to-day operating levels, has been found. Survival measured in the Delta using paired releases of tagged smolts shows a twofold better survival for individuals that travel past Stockton via San Joaquin River rather than past the export facilities via Old River. Since more than 60% of the smolts usually go down Old River, any measure that decreased this percentage would be expected to benefit smolts, however such a benefit has yet to be demonstrated empirically.

In general, current methods used to explore smolt survival in the Delta have not succeeded in clarifying basic technical and biological issues. Some of these methods are contributing useful information, but very slowly. New kinds of studies are needed, focussed on fundamental questions of salmon biology and survival methods, and designed with more concern for issues of statistical power and refutability.

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Ocean Salmon Fishery Management

L.B. Boydstun

Abstract

California ocean salmon fisheries are managed by the Pacific Fishery Management Council (Council) under the federal Magnuson-Stevens Fishery Conservation and Management Act. This chapter describes the ocean fisheries impacting California Central Valley (CV) chinook stocks, the federal regulatory process that is followed in managing these ocean fisheries, and discusses alternative management measures for protecting valuable natural resources. The CV supports fall, late-fall, winter, and spring chinook runs. The Council has adopted a spawning escapement goal for the fall run, while a federal rebuilding plan is used to regulate the fisheries to protect the winter run, an endangered species. The winter run plan is also protective of CV spring run, a threatened species. Some potential alternative management strategies include (1) a revised escapement goal for the Sacramento fall run, (2) a separate escapement goal for the spring run, (3) an escapement goal for the San Joaquin fall run, and (4) a selective ocean fishery for marked hatchery fish. The CV salmon management program is lacking in two areas: (1) river return estimates for coded-wire-tagged fish releases and (2) inconsistent tagging of hatchery fish releases, precluding estimation of hatchery fish contributions. I conclude that a comprehensive fishery management program should be implemented for CV chinook salmon under the Central Valley Project Improvement Act and that the Klamath Fishery Management Council be used as a model for developing such a program.

Introduction

Central Valley (CV) chinook salmon (*Oncorhynchus tshawytscha*) are primarily harvested in ocean fisheries off California between Point Sur and Point Arena, but are taken in significant numbers as far north as Cape Falcon in northern Oregon (Figure 1). Ocean fishing for salmon (*Oncorhynchus* spp.) off the Washington, Oregon, and California coasts is managed by the Pacific Fishery Management Council (Council) under the Magnuson-Steven Fishery Conservation and Management Act (Magnuson-Steven Act). Increasing concern for the protection of CV chinook stocks has led fishery and inland habitat managers to question the efficacy of current management strategies for ocean and river fisheries. This report describes (1) the ocean fisheries impacting CV chinook; (2) the process followed by the Council for managing CV chinook stocks; (3) alternative or complementary management measures aimed at providing additional protection for these valuable natural resources; and (4) recommendations for developing and implementing a comprehensive program for addressing fishery management concerns for CV chinook.

River fishery management, which comes under regulation of the Fish and Game Commission (Commission), is not discussed in this report. A Council "overfishing" review report provides a summary of CV sport fishery catch data through 1993 (PFMC 1994).

Fishery Resource

The Central Valley supports four runs of chinook salmon: fall, late-fall, winter, and spring, so named because of the time of year adults enter fresh water to spawn, which occurs within a few weeks or several months following river entry, depending on stock. CV chinook mature at ages 2 to 4, with a few individuals at age 5, except for winter-run chinook which mature at ages 2 to 3. Age 2 fish of all runs are primarily males (jacks).

The fall run is the more abundant and ubiquitous of the four runs, occurring in all suitable spawning areas. The other runs occur in the main stem or the various tributaries to the Sacramento River above the mouth of the American River. The winter run is listed as endangered under the State and federal Endangered Species Acts; the spring run as threatened under the two acts; and the fall and late-fall runs as candidates for listing under the federal act. All CV hatcheries (Coleman, Feather River, Nimbus, Mokelumne River, and Merced River) propagate fall chinook, while Feather River and Coleman also propagate spring and winter chinook, respectively. Hatchery production has a major effect on the number of fish available to ocean fisheries and that return to spawn each year to the hatcheries and natural spawning areas. Trucking of CV chinook salmon production from the State hatcheries for release below the Sacramento-San Joaquin Delta is done to bypass Delta water diversions. This practice increases hatchery fish survival but also increases straying of returning adults.

Figure 1 Map of coastal landmarks and Central Valley streams and hatchery locations

Ocean Fisheries

Salmon taken for commercial or recreational purposes may be taken only by hook and line (8210.1, Fish and Game Code and 27.80, Title 14, California Code of Regulations). Most salmon fishing is conducted by trolling a baited hook or lure behind a diesel or gasoline powered boat. In recent years, a baited hook fished from a drifting vessel (mooched) has become the most popular fishing method in the San Francisco Bay and Monterey Bay sport fisheries. Salmon are rarely harvested from shore although they are occasionally caught by sport fishing from the Princeton Pier, located just south of the Golden Gate.

Fishery Monitoring. The California Department of Fish and Game (DFG) aims to sample 20 percent of the salmon fishery landings to collect fishery management data by time, area, and fishery (and has done so since 1962). The heads from all ad-clipped salmon observed in the sampling are retained and the coded wire tag (CWT) contained in each head is extracted, decoded and the associated data are entered into the coastwide CWT data base maintained by the Pacific States Fisheries Commission. Fishery catch estimates are based on (1) State landings reports required from commercial and charterboat landings, and (2) random stratified sampling of the private boat fishery by the DFG. The actual sampling rates achieved in the respective fisheries (commercial, charterboat, and private boat) are used to develop the CWT expansion factors that produce estimates of CWT contributions, which are available by fishery, time, and area strata.

Commercial Fishery. The commercial fishery harvests about two-thirds of the chinook salmon taken off California. For example, commercial landings during 1995–1999 averaged 407,700 chinook compared to a sport catch of 200,000 fish (see PFMC 2000 for extensive data on California fisheries and spawning escapements).

Commercial fleet size (under limited entry) has dropped from nearly 6,000 vessels in 1982 to about 1,800 vessels in 1999. In 1999, 101 vessels landed 50 percent of the fish compared to 438 vessels in 1982 (PFMC 2000).

Commercial fishing in recent years has taken place primarily south of Point Arena because of conservation and allocation requirements for Klamath River fall chinook salmon. Commercial fisheries operate as far south as Point Conception but most landings occur in Monterey, Half Moon, and San Francisco bays. The commercial season takes place from May through September and the fishery has a 26-inch minimum size limit, although 27 inches has been used at times in recent years to protect winter chinook. Most chinook are landed from May through July.

Commercial fishing north of Point Arena has generally been limited to the month of September when Klamath chinook abundance is low.

Sport Fishery. The sport fishery has traditionally taken place between February and November and had a two fish per angler bag limit and 20-inch minimum size limit. In recent years, the season length has been reduced and higher size limits have been applied to fisheries south of Shelter Cove (Horse Mountain) to protect winter chinook. Beginning in 2000, the season opening south of Point Arena was delayed until April to protect CV spring chinook.

Chinook are taken in the sport fishery from Santa Barbara to the Oregon border, but most are landed in San Francisco and Monterey bays where most of the fishing effort originates. Charterboats take most of the fish south of Point Arena, while private boats or skiffs take most of the chinook harvested in the Fort Bragg, Eureka, and Crescent City areas. Coho fishing has been banned off California in recent years due to federal listing of Oregon and California coho stocks. This has led to salmon fishing closures north of Point Arena during most of July when coho are most abundant.

June, July, and August are the most important sport salmon fishing months off California. Since 1995, an average of 134 charterboats landed salmon in California. Annual salmon angler effort in those years (charterboat plus private) averaged 227,600 angler days. This effort produced a catch of 200,300 chinook for a catch per angler day of 0.88 chinook (PFMC 2000).

Ocean Fishery Management

The Council's Salmon Framework Plan (Plan) contains the management objectives that are followed in regulating the ocean fisheries. It specifies the area of jurisdiction, species, types of regulations, and procedures the Council must follow to make any changes. Amendment 14 to the Plan has been completed and is aimed at the meeting the requirements of the Magnuson-Steven Act as amended in 1997. It includes a recent escapement goal amendment for Oregon coho salmon and defines "Essential Fish Habitat" for salmon stocks that come under Council purview.

The Council has three advisory bodies that provide input on salmon amendment and regulatory issues. The Scientific and Statistical Committee (SSC) provides multi-disciplinary peer review of proposed fishery management actions. This includes review of stock assessments and assessment methodologies as well as review of biological, economic, and social impact analyses. The Salmon Technical Team (STT) provides the reports that summarize the previous fishing season, estimate ocean abundance for the coming season, and analyze the impacts of the Council's proposed and final management recommendations and Plan amendments. The Salmon Advisory Subpanel (SAS) develops annual regulation options and comments on all salmon issues that come before the Council, including habitat issues (PFMC 1996).

Each year the Council recommends ocean fishing regulations aimed to meet Plan escapement goals and jeopardy opinions for federally listed species. California fisheries are managed, in part, to meet escapement, allocation, and rebuilding goals for Sacramento River fall chinook, Klamath River fall chinook, Oregon and California coastal natural spawning coho salmon, and Sacramento River spring and winter chinook. A description of CV chinook salmon goals and Council stock management procedures follows.

Biological and Allocation Goals

Sacramento River Fall Chinook. The escapement goals for this stock is to achieve a spawning escapement in all years of 122,000 to 180,000 adults. The goal is based on historical river escapement levels and includes both hatchery and naturally produced fish. It should be noted the goal was modified in 1984 to establish a goal range because of the effect of Red Bluff Diversion Dam on upriver returns (PFMC 1984).

A predictor model has been developed to project CV chinook adult abundance. The model uses an index of abundance for CV chinook salmon runs (Central Valley Index or CVI), which is the sum of ocean fishery landings south of Point Arena and the adult CV spawning escapement in the same year (Table 1). The ocean prediction is based on the relationship between the CVI and the previous year CV jack estimate (Figure 2). The CVI harvest rate represents the sum of ocean fishery catches divided by the CVI for the same year. Recent years' CVI harvest rates and the proportion of adult fall chinook returning to the Sacramento River are used to project the Sacramento River fall chinook Salmon escapement under the proposed or adopted ocean fishing regulations (PFMC 2000).

The Sacramento River escapement goal has been met in all years since 1970 not including 1972, 1983, and 1990–1992 when the escapement declined to between 85,300 and 121,000 fish (Figure 2, PFMC 2000).

Table 1 Indices of annual abundance and ocean fishery impacts on California Central Valley chinook in thousands of fish

	Ocean chinook landings south of Point Arena			*Hatchery and natural escapements of Central Valley adults*				
Year	*Troll*	*Sport*	*Total*	*Fall*	*Other*[a]	*Total*	*CVI abundance*[b]	*CVI harvest index (%)*[c]
1970	226.8	111.1	337.9	190.5	55.6[d]	246.1	584.0	58
1971	150.7	166.3	317.0	190.6	62.0	252.6	569.6	56
1972	299.8	187.6	417.4	99.6	46.1	145.7	563.1	74
1973	422.5	180.9	603.4	227.1	27.1	254.2	857.6	70
1974	282.7	141.6	424.3	205.6	35.7	241.3	665.6	64
1975	234.4	92.7	327.1	159.2	47.6	206.8	533.9	61
1976	237.9	68.6	306.4	168.8	43.8	212.6	519.0	59
1977	263.8	76.6	340.4	148.7	42.8	191.5	531.9	64
1978	291.0	65.9	356.9	136.9	17.1	154.0	510.9	70
1979	234.1	108.5	342.6	167.9	11.3	179.2	521.8	66
1980	294.3	77.1	371.4	155.9	31.6	187.5	558.9	66
1981	289.9	73.8	363.7	189.3	18.8	208.1	571.8	64
1982	418.4	122.5	540.9	177.2	38.3	215.5	756.4	72
1983	178.2	53.0	231.2	121.0	12.8	133.8	365.0	63
1984	221.7	78.7	300.3	197.5	17.0	214.5	514.8	58
1985	212.3	121.8	334.1	308.9	18.1	327.0	661.1	51
1986	502.5	114.8	617.3	259.0	33.2	292.2	909.5	68
1987	446.8	152.8	599.7	188.0	25.5	213.5	813.2	74
1988	830.5	130.4	960.9	244.9	28.0	272.9	1,233.8	78
1989	363.8	130.9	494.7	149.6	17.9	167.5	662.2	75
1990	336.2	112.7	448.9	108.3	13.6	121.9	570.8	79
1991	254.6	62.1	316.7	112.3	15.3	127.6	444.3	71
1992	163.5	66.7	230.2	85.3	8.2	93.5	323.7	71
1993	259.7	99.3	359.0	131.5	10.4	141.9	500.9	72
1994	290.4	159.9	450.3	148.8	6.8	155.6	605.9	74
1995	670.6	354.6	1,025.2	272.0	16.2	288.2	1,313.4	78
1996	348.9	129.3	478.2	255.3	8.7	264.0	742.2	64
1997	482.5	208.4	690.9	350.8	17.4	368.2	1,059.1	65
1998	221.5	114.5	336.0	253.0	40.1	293.1	629.1	53
1999[e]	258.8	76.1	334.9	294.5	14.9[f]	309.4	644.3	52

[a] Spring run of the current calendar year and late-fall and winter runs of the following calendar year.
[b] Ocean landings + escapement.
[c] Ocean harvest landed south of Point Arena as a percent of the CVI.
[d] Percent of adults in 1970 spring run assumed the same as 1971 (72%, 5,500 total).
[e] Preliminary.
[f] Late-fall and winter contributions unknown—respective averages of 1995–1999 escapement used.

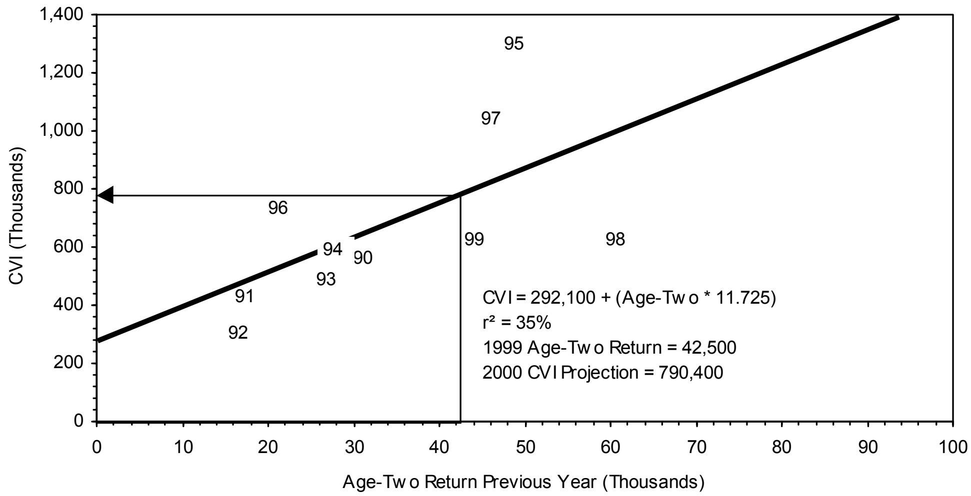

Figure 2 Linear regression of CVI on in-river age-two Central Valley chinook of the previous year, 1990–1999. Years shown are CVI year.

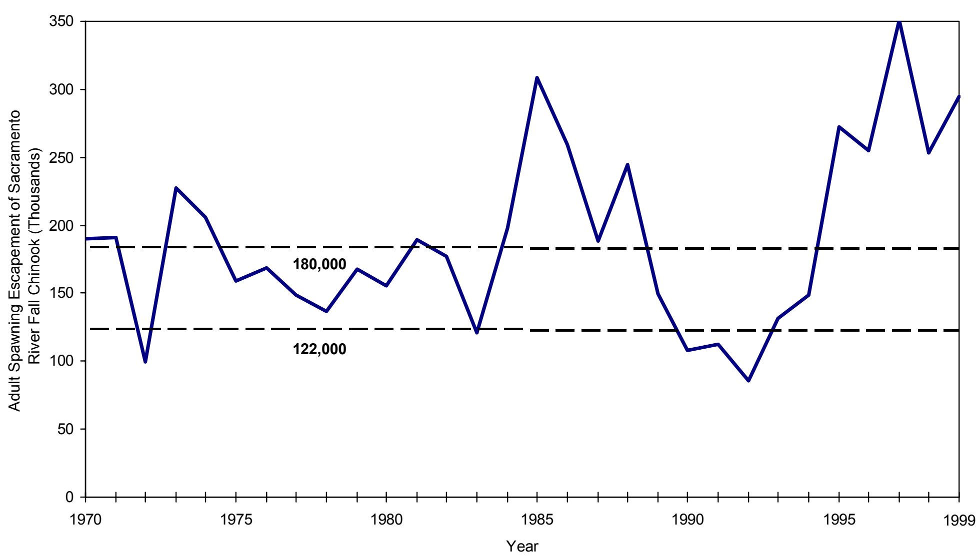

Figure 3 Spawning escapements of adult Sacramento River fall chinook, 1970–1999, and the goal range for the stock of 122,000 to 180,000 adult fish. Estimate for 1999 is preliminary.

Winter Chinook. The escapement goal for this stock is to achieve a 31 percent increase in escapement over the 1989–1993 mean rate. An ocean fishery model has been developed based on historical marked (fin-clipped) winter chinook data with which to compare proposed or actual ocean fishing regulations (DFG 1989). The model is stratified by the time and area and includes a length at age module to evaluate minimum size limits and the mortality associated with hook and release of undersized fish. It is noteworthy that the shift in recent years to mooching in the ocean sport fishery off central California has decreased the benefits associated with an increased minimum size limit (from 20 to 24 inches). This is because fish caught by mooching tend to swallow the hook, which is often fatal. Thus, in addition to an increased minimum size limit, time, and area closures have been required to meet the winter chinook harvest rate objective.

Spring Chinook. Spring chinook were listed as threatened under the State and federal acts in 1999. The NMFS recently issued a biological opinion regarding the effect of ocean fisheries on CV spring chinook. They concluded that recent action by the California Fish and Game Commission to delay the sport season opening south of Point Arena, in combination with the management measures in place to protect winter and spring chinook, should be sufficient to allow for stock rebuilding.

Administrative Process

Regulatory. The Council is advisory to the Secretary of Commerce (Secretary) who has the authority to implement federal salmon fishing regulations for ocean waters 3 to 200 miles offshore. The DFG Director has the authority under Section 7650 et seq. of the Fish and Game Code to conform State regulations affecting the commercial fishery in State waters (0 to 3 miles) to the Council's salmon fishery plan (or the actual federal regulations). The Commission retains regulatory authority over the sport fishery in State waters and must follow the State's Administrative Procedures Act in conforming State regulations to the PFMC plan. Each year the ocean salmon fishing regulations (federal and State) are adopted to be effective starting May 1.

Plan Amendment. The Salmon Plan contains the basic elements for regulating the ocean fisheries. The states generally have the lead with regard to recommending Plan amendments, and Council concurrence is required to proceed with any amendment proposals. The Council generally considers amendments at its September or November meetings, but can make an exception at its March, April, or June meetings. The amendment process requires the development of a document for public review, public hearings, final Council action, and publication (if approved) by the Secretary in the *Federal Register*. Extensive Council review is provided during the Plan development process, and the Secretary can reject the Plan or return it for additional development and public hearings. A Plan amendment generally takes a year or longer to complete. For species listed under the federal ESA, federal restrictions supersede the Council's goals.

Alternative Management Strategies

In response to the concern over the status of chinook stocks in the CV and elsewhere, the need may arise to implement additional or revised management objectives for CV chinook salmon runs. Alternative harvest strategies may also need to be considered. A discussion of possible Plan amendment options and the procedure to follow in implementing such changes through the Council process is presented in the following sections.

Revise the Escapement Goal for Sacramento River Fall Chinook. Raising the gates at Red Bluff Diversion Dam during most of the adult fall salmon run is expected to increase natural salmon production in the upper river. It follows that the Council goal range may no longer be appropriate for the stock and should be set at no less than 180,000 adult fish. This proposal would take a Plan Amendment and require the development of an analysis showing how a revised ocean fishing strategy would produce optimum yield to the U.S. fishing industry, as compared to the current goal (National Standard 1). Such an amendment would take a year or longer to complete. Listing under the federal ESA would supersede the Council's management goal for the stock.

Establish an Escapement Goal for Sacramento River Spring-run Chinook. The Council has approved Salmon Plan Amendment 14 in which a provision is included to allow for additional management goals for stocks not listed in the Plan as part of a two-meeting regulatory process. Such an action would have a time constraint, and would require a Plan amendment to complete the process. CWT spawning escapement estimates may not be available for CV hatchery spring-run chinook because a program has not been in place to make such estimates. The paucity of data could complicate the development of a fishery harvest strategy for the stock because it would be difficult to show the relationship between fishing and spawning escapement under historic or recent fishing regulations. A thorough review of available spring run CWT data is needed to assess the adequacy of available data for developing a spring run fishery model. Consideration should also be given to continuing or implementing a spring chinook CWT program at Feather River Hatchery, and to estimating river returns of CWT spring chinook beginning as soon as possible.

Establish an Escapement Goal for San Joaquin Fall Chinook. The original Salmon Plan developed in 1977 had an escapement goal for this stock, but the goal was removed in 1984 because Delta water withdrawals were affecting the run. The San Joaquin run has not been proposed for separate listing under the federal ESA, but was included as part of the fall and late-fall CV complex in the recent review of California chinook populations (Myers and others 1998). A separate goal could be established for the run under the Council's Plan amendment process. Any such proposal would have to show how goal attainment would affect the ocean fisheries, particularly with regard to their ability to access other chinook stocks when the San Joaquin run is depressed due to water diversion conditions. Analysis of CWT data might show a different ocean distribution pattern for San Joaquin chinook, which could ameliorate any reduction in harvest opportunity for other chinook runs. The amendment process would take a year or longer to complete.

Conduct Selective Fisheries for CV Hatchery Stocks. The Council has approved regulations since 1998 that allow for an ocean fisheries off Washington and Oregon for ad-clipped coho salmon. The fishery is for hatchery fish that were marked in the previous year for the purpose of providing for an ocean selective fishery. Post season analysis showed that the majority of fish encountered in the fishery were, in fact, ad-clipped hatchery fish. The ad mark historically was used as a "flag" for CWT salmon, but an exception was made in the case of Oregon and Washington hatchery coho releases. A selective fishery for hatchery-origin CV chinook salmon could be implemented in California fisheries. Hooking mortality of released (unmarked) fish would be an important consideration. Recent DFG studies show that hook and release mortality of sublegal chinook caught by mooching in the sport fishery is about 24 percent. Hooking mortality of chinook caught by trolling is lower in the sport fishery and about 30 percent in the commercial fishery. Use of the ad mark in a selective fishery could adversely impact the CV CWT program. This is because the tag detection rate, using currently available hand-held detection equipment, is much lower than it is for coho, stemming from the much larger head size of chinook.

Final Remarks and Recommendations

California ocean fisheries are regulated under a Plan developed by the Council and approved by the Secretary pursuant to the Magnuson-Steven Act. The process provides for thorough discussion of Council and State management objectives along with extensive scientific stakeholder input. The Plan amendment process is flexible and requires that proposed Plan amendment proposals are consistent with the National Standards of the Magnuson-Steven Act. The amendment process may take a year or longer to complete, but can be done in less than a year if the change is agreeable to the affected interest groups.

The Council's escapement goal for Sacramento River fall chinook has been met in all but five years since 1970. The goal does not differentiate between hatchery and natural production. Attainment of the winter chinook escapement goal is evaluated based on the adopted regulation structure and is not linked to the actual escapement.

NMFS has expressed concern that natural production in the CV is depressed and that hatchery production is masking the situation. NMFS has also concluded that CV chinook are subjected to excess fishing mortality in the ocean fisheries (NMFS 1998).

In my view, the major problem with current CV salmon fishery management is two fold: (1) the lack of river return estimates for CWT releases and (2) the lack of a comprehensive CWT program to estimate fishery and spawning escapement returns for all hatchery releases. To remedy this situation, I recommend using the Klamath River salmon management program as a model for developing a counterpart CV program. In addition to a comprehensive hatchery CWT and river spawning escapement estimation program; government, tribal, and stakeholder input is provided through a basin management advisory council (Klamath Fishery Management Council, KFMC). The KFMC has a scientific team that evaluates and analyzes biological data and fishery management options (Klamath River Technical Advisory Team). The opportunity is at hand to develop a comprehensive CV fishery management program as a main element of the fishery program to be developed under the Central Valley Project Improvement Act.

Thanks, Nat Bingham

In closing, I would like to recognize and pay tribute to former Council member and friend of many years Nathaniel (Nat) S. Bingham. Nat and I had agreed to prepare this report, but he passed away before we could actually begin work on the manuscript. Had he been around to help write the report, more would have been written about the importance of habitat protection and restoration to the sustainability of CV salmon populations. Nat was an important contributor to the management of California salmon fisheries, and, in particular, to the protection and conservation of the State's rivers and streams upon which our salmon resources depend. Nat was the consummate statesman, but he will mostly be remembered as the tireless advocate for the fish. He was a driving force behind the creation of the Council's Habitat Steering Committee, and was active with the Committee right up to the end. California salmon are better off today in large part because of Nat Bingham's motivation and desire to do what was right for the fish.

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Population Trends and Escapement Estimation of Mokelumne River Fall-run Chinook Salmon (*Oncorhynchus tshawytscha*)

Joseph J. Miyamoto and Roger D. Hartwell

Abstract

In 1990 the East Bay Municipal Utility District (EBMUD) began a program to monitor the fall-run chinook salmon (*Oncorhynchus tshawytscha*) populations in the lower Mokelumne River using video and trapping at Woodbridge Dam and weekly redd surveys.

Over the eight years of this monitoring program, the Mokelumne River fall-run chinook salmon escapement showed a trend of increased abundance of both hatchery and natural spawners. The 1997 estimated total spawning escapement (combined hatchery and natural run) was 10,175 compared to a spawning escapement of 497 in 1990 and the 57-year average escapement of 3,434 fish. The estimated natural spawning population fluctuated from a low of 369 in 1991 to a high of 3,892 fish (1,739.3 ± 1,384.9) in 1996. The percentage of natural spawners ranged between 31% to 90% (52.3 ± 19.9) of the total spawning escapement during the 1991–1997 period.

Significant correlations were observed between the number of redds and total escapement (R^2 = 0.941, P < 0.0001) and the hatchery returns and total spawning escapement (R^2 = 0.972, P < 0.001). The later correlation was used to determine the accuracy of past spawning escapement estimates based upon a similar correlation using a narrower dataset.

These results suggest accurate total spawning escapement estimates can be obtained from hatchery returns and from redd counts. Escapement estimates calculated from redd counts and compared with known estimates were accurate in the mid-range while those calculated from hatchery returns were accurate throughout the range of run sizes.

Introduction

East Bay Municipal Utility District (EBMUD) began daily monitoring of the fall-run chinook salmon (*Oncorhynchus tshawytscha*) population in the lower Mokelumne River in 1990. The focus of the monitoring was to document the timing and magnitude of adult salmon upstream migration and the number and distribution of salmon redds on the upstream spawning grounds.

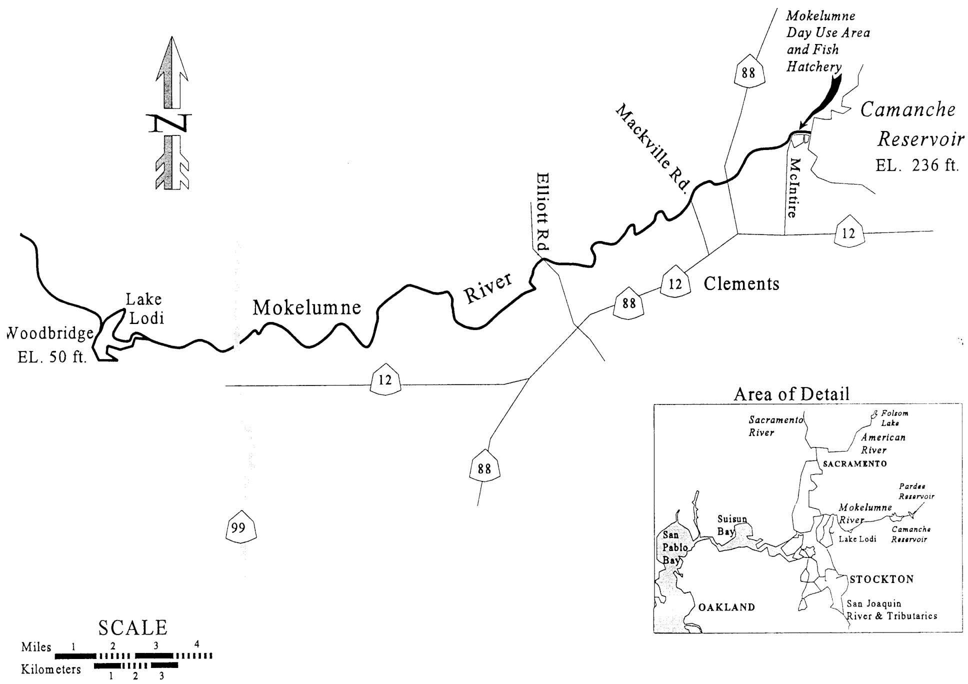

Figure 1 The Lower Mokelumne River between Camanche Dam and Woodbridge Dam, San Joaquin County, California

The Mokelumne River originates in the Sierra Nevada mountains at the Sierra Crest and flows through the Central Valley near the towns of Lockeford, Clements, and Lodi before entering the Delta forks of the Mokelumne just downstream of the Delta Cross Channel (Figure 1). The Mokelumne River watershed drains some 627 square miles. The average annual unimpaired runoff is 720,000 acre-feet with a range of 129,000 to 1.8 million acre-feet. The Mokelumne River watershed has a number of dams and reservoirs. In the upper watershed, Pacific Gas & Electric operates 19 dams, seven storage reservoirs, seven diversions, three regulating reservoirs and two forebays (FEIS 1993). Pardee Dam and Reservoir (river mile 39.6) is owned and operated by EBMUD to provide water for 1.2 million customers in Alameda and Contra Costa counties (EBMUD 1992). The reservoir also provides flood control storage, maintenance of the Camanche Reservoir hypolimnion, and water-based recreational opportunities including both coldwater and warmwater fisheries. Camanche Dam and Reservoir, completed by EBMUD in 1964, provides storage for flood control operations, water to meet agricultural and senior water rights, instream flows for fish needs and a number of water-based recreational opportunities. Camanche Dam (river mile 29.6) represents the upstream limit for anadromous salmonid migration. Historically, salmon and steelhead used the habitat to within one-half mile below Pardee Dam where a natural barrier existed at the Arkansas Ferry Crossing, a distance of some eight and one-half miles above Camanche Dam (EBMUD 1992).

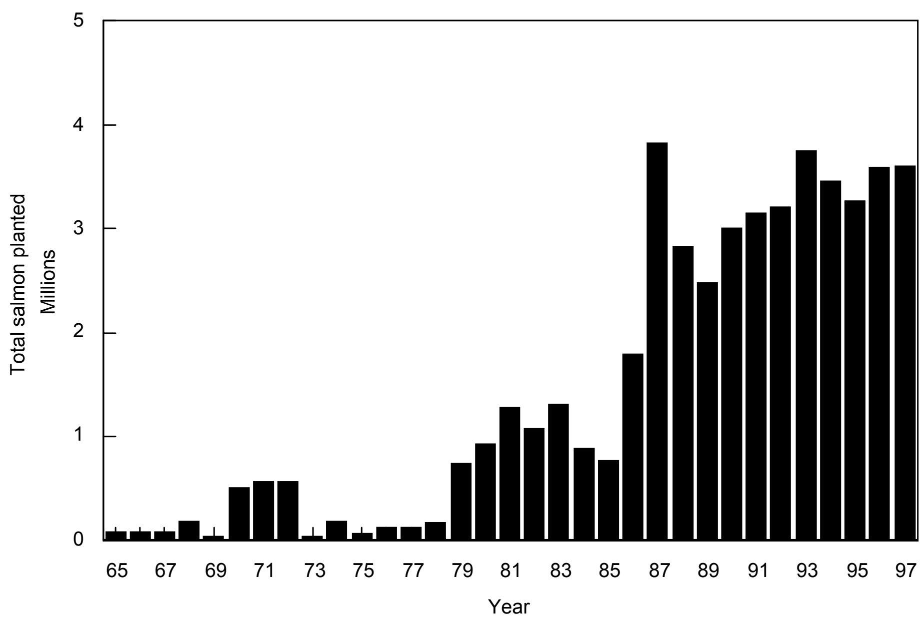

Figure 2 Mokelumne River Fish Hatchery production, 1965–1997. Source: Data from DFG reports. DFG corrections may have modified some previously reported yearly data.

To mitigate for lost habitat above Camanche Dam, the Mokelumne River Fish Hatchery was constructed in 1964 to produce both fall-run chinook salmon and steelhead trout (*Oncorhynchus mykiss*) (EBMUD 1992). Average production from the facility during the 1990s was 3.0 to 4.0 million fall-run chinook salmon smolts, 500,000 yearling chinook salmon, and 100,000 yearling steelhead (Figure 2). The source of most of the salmon broodstock was Feather River Hatchery fish. Two million salmon were raised to post-smolts each year for an ocean enhancement program. All of the enhancement salmon production was trucked around the Delta for release in San Pablo Bay (Figure 3). Salmon smolts that were Mokelumne origin fish were planted below Woodbridge Dam. In 1992 and 1993, yearling chinook salmon were planted in the

Mokelumne Day Use Area adjacent to the Mokelumne River Fish Hatchery just downstream of Camanche Dam (see Figure 1). After 1994, yearlings were released below Woodbridge Dam. During drought years, naturally produced juvenile salmon were collected at Woodbridge Dam and trucked around the Delta for release at Rio Vista or Carquinez (see Figure 1) (Bianchi and others 1992).

Woodbridge Dam spans the lower Mokelumne River near the City of Lodi and the town of Woodbridge (see Figure 2). Each year in March, flashboards are installed in the dam to create Lake Lodi and raise the water surface elevation to operate the Woodbridge Irrigation District diversion canal. Following the end of the irrigation season in late October or early November, the flashboards are removed to empty Lake Lodi. Fish passage past the dam under either mode of operation is provided by a pool-and-weir system that includes high-stage and low-stage fish ladders. The fish ladders provide a unique opportunity to obtain complete counts of fall-run chinook salmon passing Woodbridge Dam under nearly all flow and operating conditions.

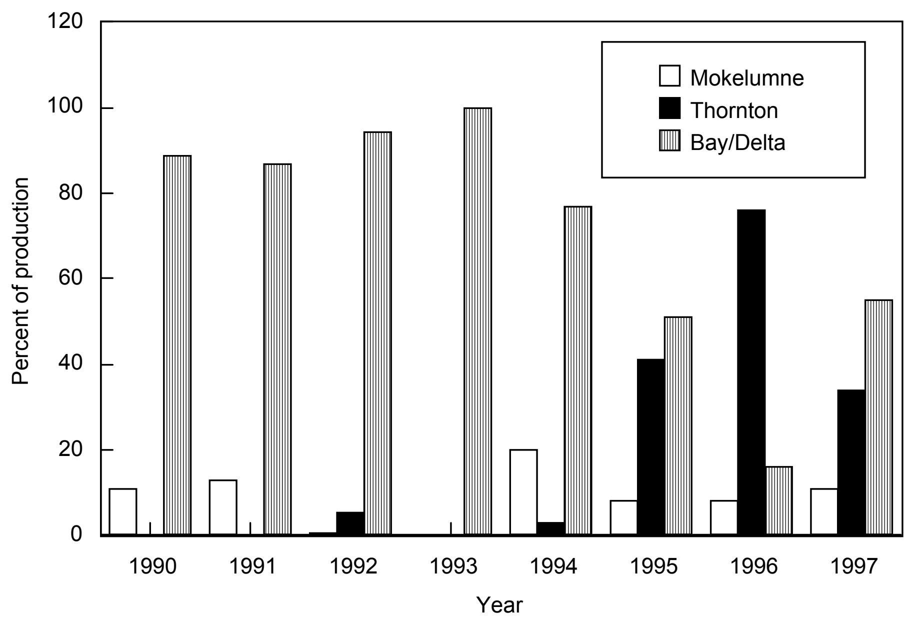

Figure 3 Release locations of Mokelumne River Fish Hatchery chinook salmon production 1990–1997. Production includes smolts, post-smolts, and yearlings. Source: Data from DFG annual reports.

Objectives

Daily video and trap monitoring at Woodbridge Dam provided a new, more reliable method to obtain salmon spawning escapement in the lower Mokelumne River. One of the objectives of this study was to compare results from this monitoring program to historical escapement estimators. These historical estimators are based on linear regression of hatchery return and estimated annual spawning escapement derived from carcass surveys.

Another objective of this study was to determine if alternate methods could be used to estimate spawning escapement based on the 1990–1997 dataset. The statistical relationships between the number of redds and total spawning escapement, and hatchery returns and total spawning escapement were examined for this purpose.

Methods

Escapement Estimation

From 1940 to 1990, the California Department of Fish and Game (DFG) estimated and/or counted the numbers of chinook salmon migrating upstream to spawn in the lower Mokelumne River. Several methods have been used to estimate spawning escapement (Table 1). These methods included carcass surveys of spawning grounds as well as projections of the natural run using linear regression equations based on the relationship between numbers of hatchery and natural spawners. Direct counting methods included observations of the number of salmon ascending the fish ladders at Woodbridge Dam (Fry 1961).

Table 1 Mokelumne River stock estimates

Period of sampling	*Sampling method*
1940 – 1942, 1945	Visual count at Woodbridge
1943, 1944, 1946, 1947	No estimate
1948 – 1971	Visual count at Woodbridge
1972 – 1981	Carcass survey
1982, 1983	Regression
1984 – 1990	Carcass survey
1990 – 1997	Video and trap monitoring

In 1990, a video and trap monitoring system was installed by EBMUD in both the upper and lower ladders of Woodbridge Dam. An overhead video camera was mounted in the high-stage fish ladder, and a waterproof enclosure housing a camera mounted for a side view was installed in the low-stage fish ladder. Both video cameras shot footage against a 1.2 m^2 plywood backboard covered with a white plastic sheet and marked with black grid lines spaced five centimeters apart. Four 150-watt flood lamps mounted above the water surface illuminated the backboard. Video camera recording was conducted 24 hours per day, seven days per week, throughout the fall upstream migration. The tapes were reviewed and count data were recorded. The start date of the video monitoring varied between 1 September and 26 October and the ending date each year was 31 December, except for 1997 when high flows ended operations on 10 December.

The sex ratios and age composition of the salmon spawning escapement at Woodbridge Dam were determined by reviewing the videotapes from the underwater camera in the low-stage fishway and collecting data from trapped fish. Sex ratios and age composition of hatchery fish were obtained from DFG Mokelumne River Fish Hatchery personnel.

Upstream migrant traps were installed each year between 1990 and 1997 and operated in the Woodbridge low-stage fishway in pool 8a (Figure 4). The traps were checked two to four times per day, depending on the number of fish captured. The two primary trap checks were one-half hour before sunrise and one-half hour after sunset. The traps were operated intermittently to verify results from the video monitoring program or when highly turbid conditions precluded the use of video cameras.

For a complete description of the video equipment, setup of the video monitoring stations, trap equipment and operations protocol see Bianchi and others (1992) and Marine and Vogel (1996).

Physical and environmental data collected included river flow, river temperatures from Campbell recorders at each gauging station and from a Ryan RTM 2000 thermograph in pool 6a of the Woodbridge low stage fishway (see Figure 4), National Weather Service data on barometric pressure from Stockton and local watershed precipitation from Camanche Dam, and water transparency measured by Secchi disk from pool 9a or from the left abutment of Woodbridge Dam (Marine and Vogel 1996).

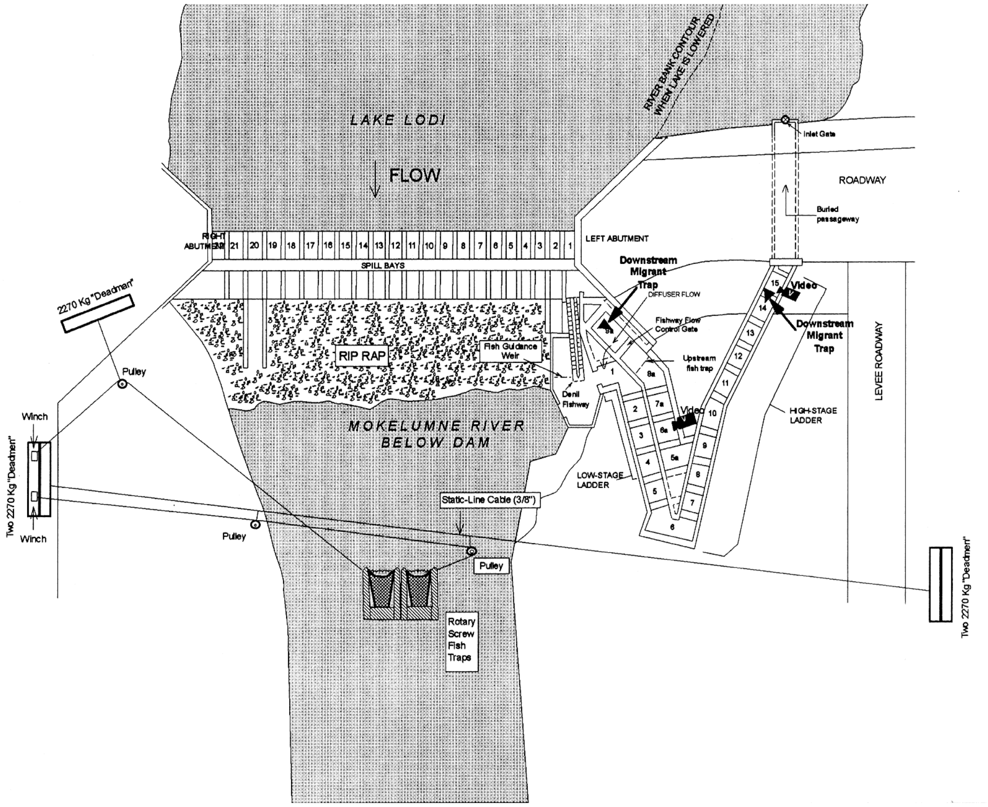

Figure 4 Plan view of Woodbridge Dam showing video monitoring sites and location of upstream migrant fish trap

The percentage composition of grilse and adult salmon in the run was based on a length criterion. A fork length of 61 cm was used to separate grilse from adults. Marine (1997) found this length to be conservative criterion for Mokelumne coded wire tagged hatchery fish recovered in Central Valley streams and hatcheries during 1992–1995. Using this criterion, Marine (1997) found that 20% of the two-year-old fish were greater than 61 cm and 5% of the three-year-old fish were less than 61 cm. The Mokelumne River Fish Hatchery used the 61-cm criterion, except in 1993 when a large number of grilse (57%) returned to the hatchery and the criterion was increased to 65 cm (Marine and Vogel 1994).

Salmon Redd Abundance Analysis

Salmon redd surveys were conducted weekly in the lower Mokelumne River from 1990 to 1997 (Hagar 1991; Hartwell 1996; Setka 1997). The surveys typically began in early to late October and ended the first week in January, except in 1996 when flood flows ended the surveys on 3 December. For recording the distribution of salmon redds, the lower Mokelumne River was divided into three reaches (Reach A: Camanche Dam to Highway 88, Reach B: Highway 88 to Mackville Road, and Reach C: Mackville Road to Elliott Road.) (Figure 5). The surveys involved teams of three biologists canoeing or boating and wading down the river in search of redds. Each redd was marked with a fluorescent colored brick and assigned a unique number. During the surveys, data were also collected on redd characteristics including the redd size, water depth, velocity, habitat characteristics, degree of redd superimposition, and usage of prior gravel enhancement sites. The different levels of redd superimposition were based upon the degree of overlap between adjacent estimated redd egg pockets and tail-spills (Hartwell 1996). Habitat types were characterized according to a modified Bisson system (Bisson and others 1981) and included glide, riffle, riffle-glide complex, side-channel glide, and side-channel riffle.

Physical and environmental data collected included water temperature, dissolved oxygen, and stream flow. Water temperatures were collected using hand held thermometers and Campbell data loggers at EBMUD gauging stations (see Figure 5). Total Camanche Dam and powerhouse releases were combined to determine streamflow in the spawning reaches (Hartwell 1996; Setka 1997).

To evaluate alternate methods for determining spawning escapements, linear regression equations were computed for the hatchery return and total escapement past Woodbridge Dam, hatchery return and natural spawning escapement, and total number of redds and total escapement.

The escapement at Woodbridge Dam includes both hatchery and naturally spawning fish. The natural spawning escapement estimate was derived by subtracting the hatchery fish return from the total escapement.

Results

Escapement Estimation

During the first year of the video and trap operations in 1990, the counts at Woodbridge Dam were compared with the DFG escapement estimate based upon the carcass survey. The results showed that substantially more salmon passed Woodbridge Dam than estimated by DFG using carcass survey data (497 actual count compared to 64 from carcass survey estimator) (Bianchi and others 1992). Because the accuracy of the carcass surveys was influenced negatively by environmental conditions such as streamflow and turbidity, DFG discontinued the carcass surveys on the lower Mokelumne River in favor of the more reliable daily video and trap monitoring.

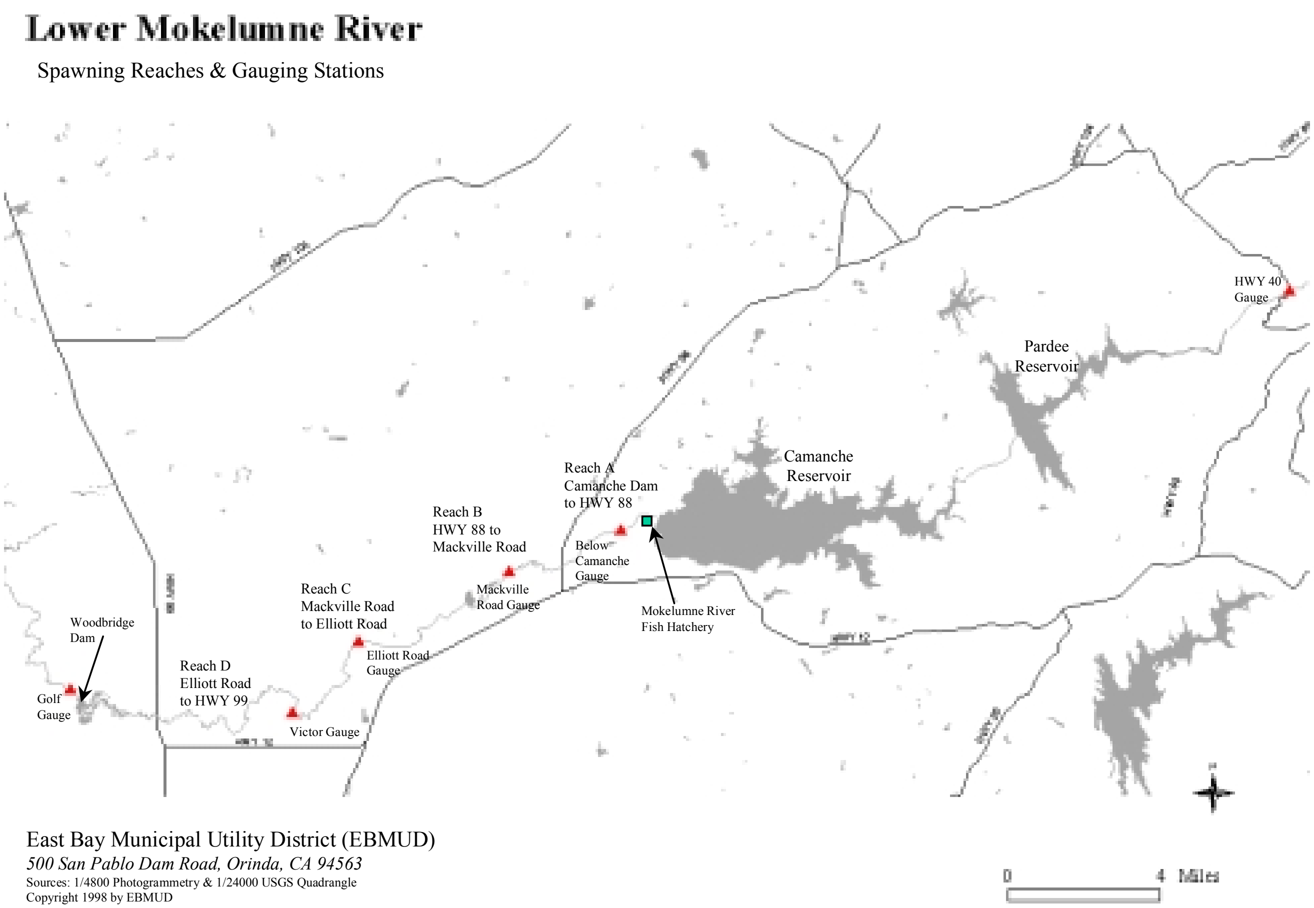

Figure 5 Location of US Geological Survey and EBMUD gauging stations

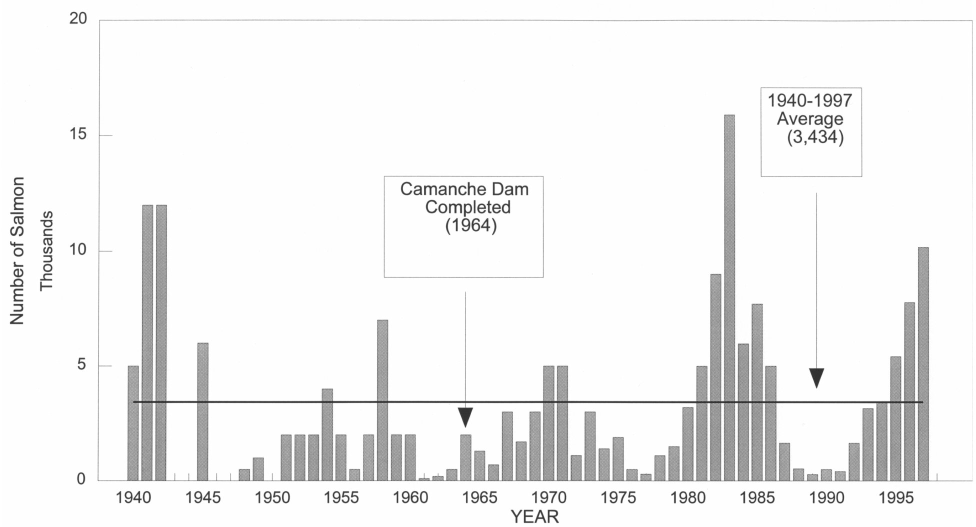

Figure 6 Lower Mokelumne River fall-run chinook salmon escapement, 1940–1997. Source: Data are from DFG, Biosystems, and NRS, Inc. Monitoring of salmon escapement in 1996 was discontinued early (on December 10, 1996) due to high flows. No data were collected in 1943, 1944, 1946, 1947, and 1950. Calculated from the average of the salmon escapement values from 1940 to 1997, excluding 1943, 1944, 1946, 1947, and 1950.

The estimated annual spawning escapement of fall-run chinook salmon over the 57-year period of record is shown in Figure 6. Estimates of spawning escapement during this period have varied from 100 fish in 1961 to 15,900 fish in 1983 (average = 3,434). The 1983 count was based upon an estimate projected from the regression between the hatchery returns and total escapement (Meinz 1983). This regression was based on hatchery return numbers ranging from 17 to 1,386 (average = 463). Over the course of the daily video and trap monitoring (1990 to 1997), the counts of fall-run chinook salmon have ranged between 410 and 10,175 fish (average = 4,062) (Marine 1997). So, the average spawning escapement estimated from 1940 to 1989 was 3,434 fish and average escapement counted by video trap and monitoring from 1990 to 1997 was 4,062.

During the daily video and trap monitoring at Woodbridge Dam (1990 to 1997), the percentage of total spawners ranged between 31% and 90%, with an average of 52.3% (Figure 7).

Adult salmon migrating into the lower Mokelumne River are primarily two- and three-year-old fish. The percentage of grilse in the spawning run has been highly variable, ranging between 7% and 57% over the eight-year monitoring period with an average of 26.8% (Figure 8) (Marine 1997).

The sex ratio of adult salmon counted at Woodbridge Dam over the 1990–1997 period varied between 46% and 57% female with an average of 50.4%. For the Mokelumne River Fish Hatchery, the sex ratio of adult fish varied between 33% and 53% females with an average of 44.9%.

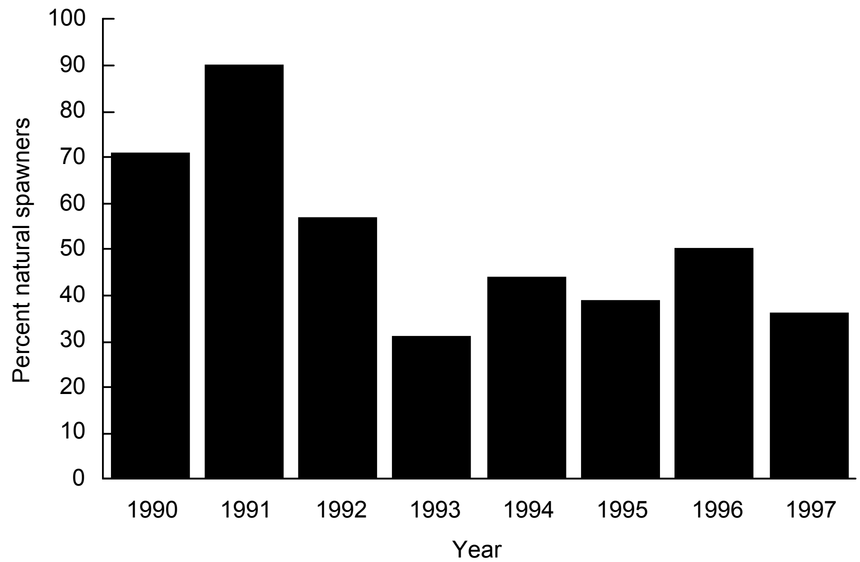

Figure 7 Percent of fall-run chinook salmon spawning naturally in the lower Mokelumne River. Source: Data from Biosystems and NRS, Inc. taken at WID.

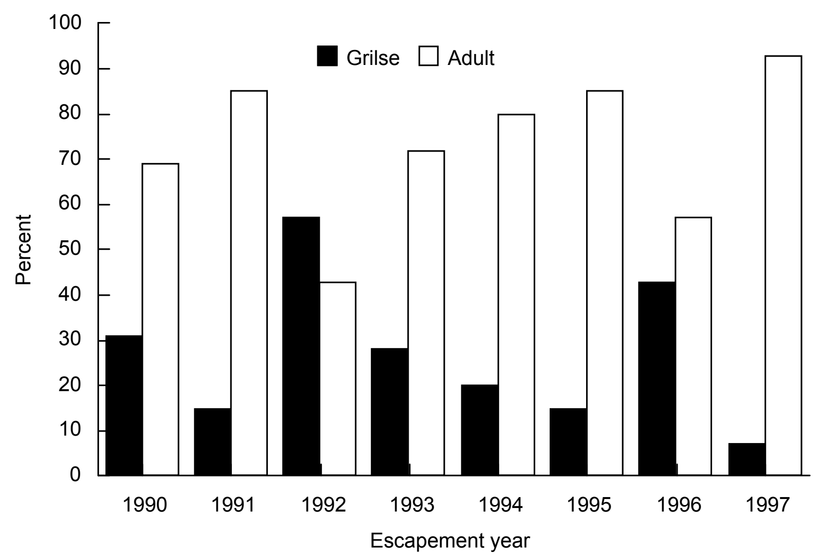

Figure 8 Percent of fall-run chinook salmon passing Woodbridge Dam that are grilse, 1990–1997

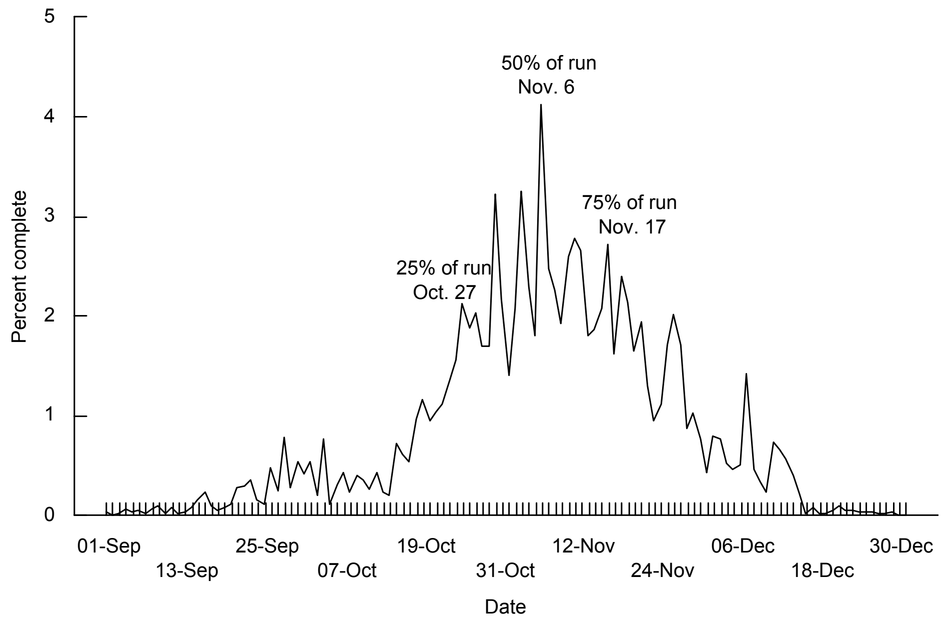

Figure 9 Daily average percent of fall-run chinook salmon escapement in the lower Mokelumne River, 1990–1997

The duration of the salmon run has expanded with increases in salmon spawning escapement from 1990 to 1997. Video and trap monitoring at Woodbridge Dam initially began in October, but was started in early September beginning in 1995 as spawning escapement increased. Except for 1996, when high flood flows ended monitoring on 3 December, the video and trap monitoring ended on 31 December of each year (Bianchi and others 1992; Marine and Vogel 1996; Marine 1997). The mean dates for the 10%, 50%, and 90% completion of the average upstream migration run timing were 27 October, 6 November, and 17 November, respectively. The average daily percentage of adult salmon migration past Woodbridge Dam from 1990 to 1997 show a peak in late October to mid-November (Figure 9).

Salmon Redd Abundance Analysis

Redd surveys show that chinook salmon use all three reaches from Camanche Dam to Elliott Road for spawning (Figure 10). The years with the greatest percentage of redds constructed in Reach A occurred during the highest spawning escapements (Hartwell 1996; Setka 1997).

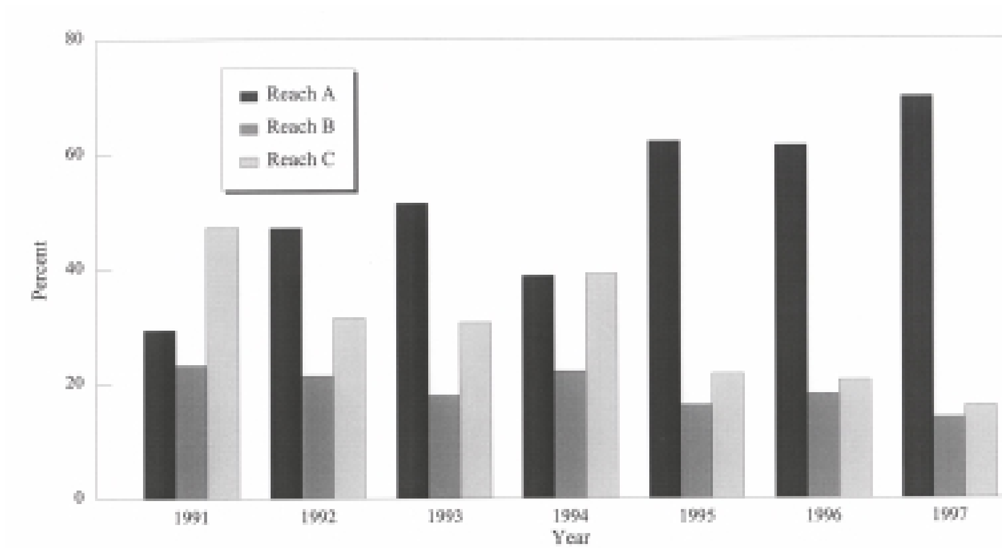

Figure 10 Lower Mokelumne River fall-run chinook salmon redd construction by reach, 1991–1997

The salmon redds in the lower Mokelumne River during the monitoring period increased from 71 redds in 1990 to 1,316 in 1997. The 1996 estimate of 1,284 redds is a projection based on the average percentage of redds completed on 3 December (the last date of the partial redd survey), from 1991–1995 (Setka 1997). The peak redd construction activity occurred from early November to mid-December.

The amount of redd superimposition increased with increased spawning escapement and ranged between 3% and 17%. There was a dramatic increase in redd superimposition from 3% in 1993 to 14% in 1994 (Hartwell 1996). Natural spawning escapement between these years increased from 993 to 1,503 fish (Figure 11). Between 1991 and 1997 no distinct relationship was evident between redd superimposition and other factors such as flow, temperature, or number of in-river females (Hartwell 1996; Setka 1997).

The relationship between the number of redds constructed and total escapement by linear regression (R^2 = 0.941, P < 0.0001) is shown in Figure 12. The database used to generate the linear regressions includes a range of salmon redds from 71 to 1,316 and total spawning escapements based upon video and trap counts of 410 to 10,175 salmon. The ratio of female spawners to redds during the study period ranged between 0.9:1 to a high of 2.3:1. The highest ratios were obtained at both lowest and highest spawning escapements during the study period (Figure 13).

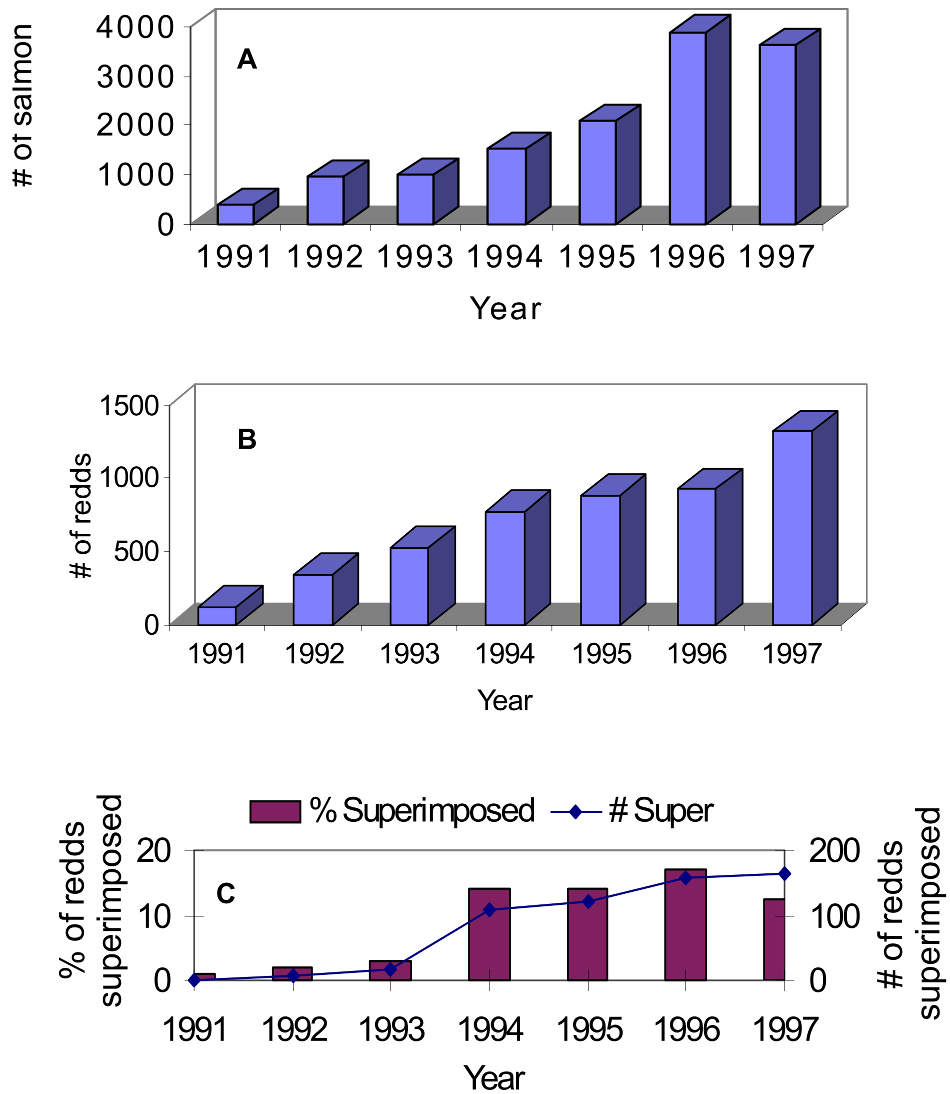

Figure 11 Comparison of salmon redd superimposition with spawning escapement, 1991–1997: (A) number of in-river spawners in Mokelumne River; (B) number of redds constructed per year in Mokelumne River; (C) percent and number of superimposed redds in Mokelumne River.

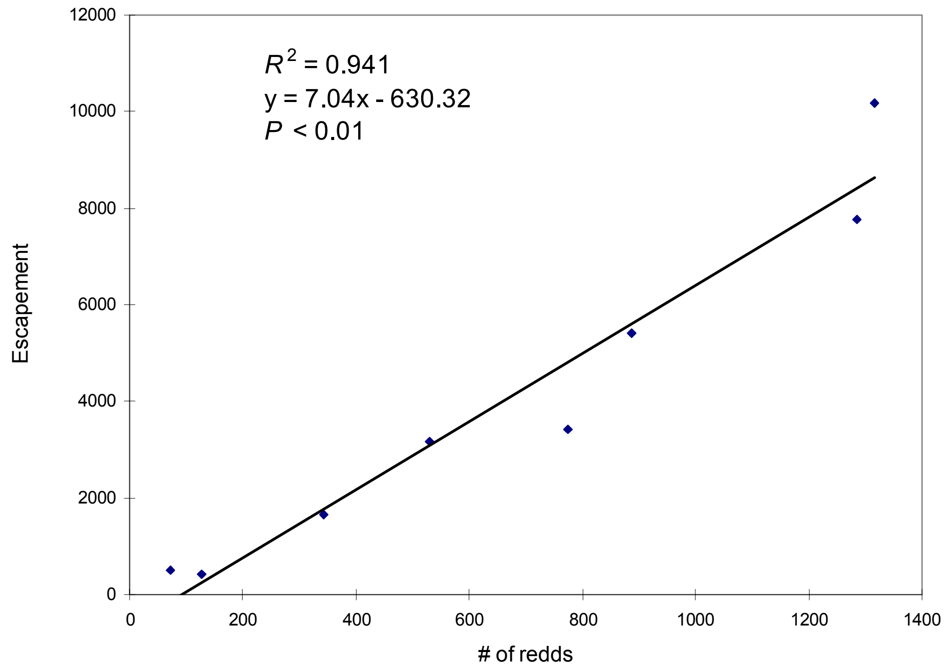

Figure 12 Linear regression of Mokelumne River total escapement compared with number of redds constructed, 1990–1997

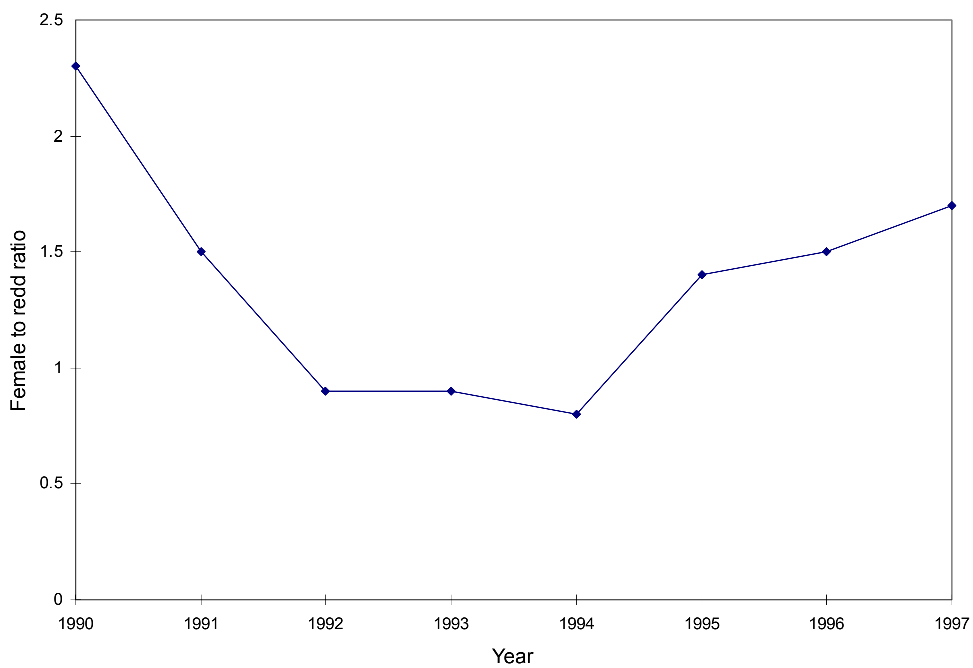

Figure 13 Female to redd ratio, 1990–1997

Discussion

Escapement Estimates

From 1972 to 1990 (with the exception of 1982 and 1983), DFG estimated the total salmon spawning escapement in the lower Mokelumne River from carcass survey recovery data. The recovery rates of marked carcasses were used to calculate the total number of spawners. The recovery rates ranged between 0% and 25%. In one-half of their surveys, DFG used the recovery rate for carcasses in the lower Mokelumne River. In 1976 and 1979, a rate of 20% was used when no carcasses were recovered. The 20% rate was based upon the average historic recovery rate for Sacramento River systems (DFG 1978). In 1990, DFG and EBMUD biologists conducted five carcass surveys between Camanche Dam and the Mackville Road Bridge (Reaches A and B). During the surveys, three carcasses were found including one grilse. The two adult carcasses were tagged and released and only one tagged carcass was recovered in subsequent surveys, resulting in a 50% recovery rate. Based upon this recovery rate, the 1990 spawning escapement estimate including 64 hatchery fish was 70 salmon (Fjelstad 1991). The daily count at Woodbridge Dam in 1990 totaled 497 fish.

One limitation of the 1990 carcass survey was that the survey was only conducted from Camanche Dam to Mackville Road. The shortened survey reach in 1990 may have contributed to the low spawning escapement estimate. Subsequent redd surveys conducted by EBMUD have shown that as many as 47% of the salmon redds are constructed below Mackville Road during low escapement years (Hartwell 1993). If as for 1976 and 1979, a Sacramento River system 20% recovery rate is used, the resulting escapement estimate of 94 salmon would still fall short of the total number counted at Woodbridge Dam.

During 1982 and 1983, Camanche flood control releases in excess of 2,000 cfs made it impossible to conduct carcass surveys (DFG 1986). The spawning escapement during these years was estimated using a statistical relationship between the number of salmon entering the Mokelumne River Fish Hatchery and the spawning escapement estimate from carcass survey data. A linear regression was established using data from the 1972 to 1981 runs (1977 was excluded because the ladder to the fish hatchery was closed). The 1982 and 1983 estimates were generated by extrapolating the hatchery returns to the regression line to obtain an estimate of total escapement (Meinz 1983; Figure 14). This methodology resulted in two of the highest spawning escapement estimates for the lower Mokelumne River (9,000 fish for 1982 and 15,866 for 1983). The hatchery returns for these years of 2,677 and 4,573 fish respectively were outside the range of the 1972 to 1981 database. In addition, the spawning stock estimates from carcass surveys used to generate the linear relationship may have been low because of the use of incomplete surveys in some of the years during the base period.

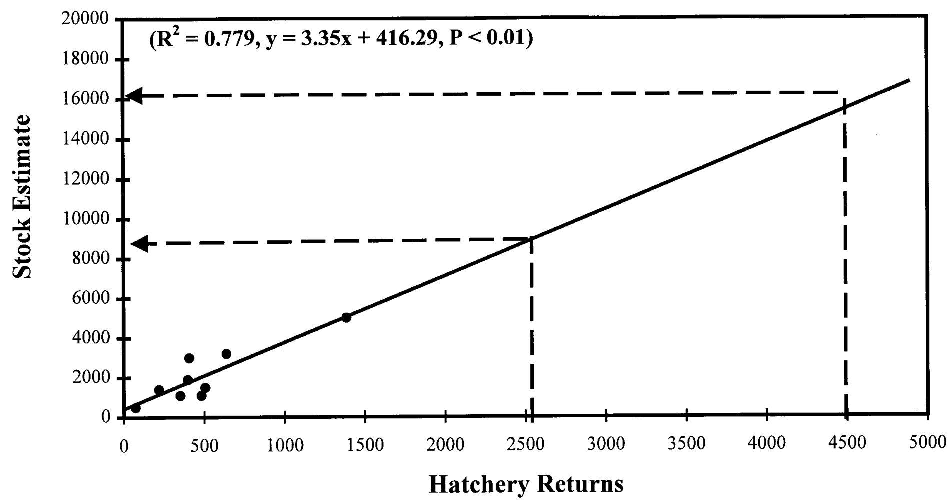

Figure 14 Mokelumne River stock estimates compared with hatchery returns, 1972–1981. 1997 data are omitted because no hatchery returns were reported.

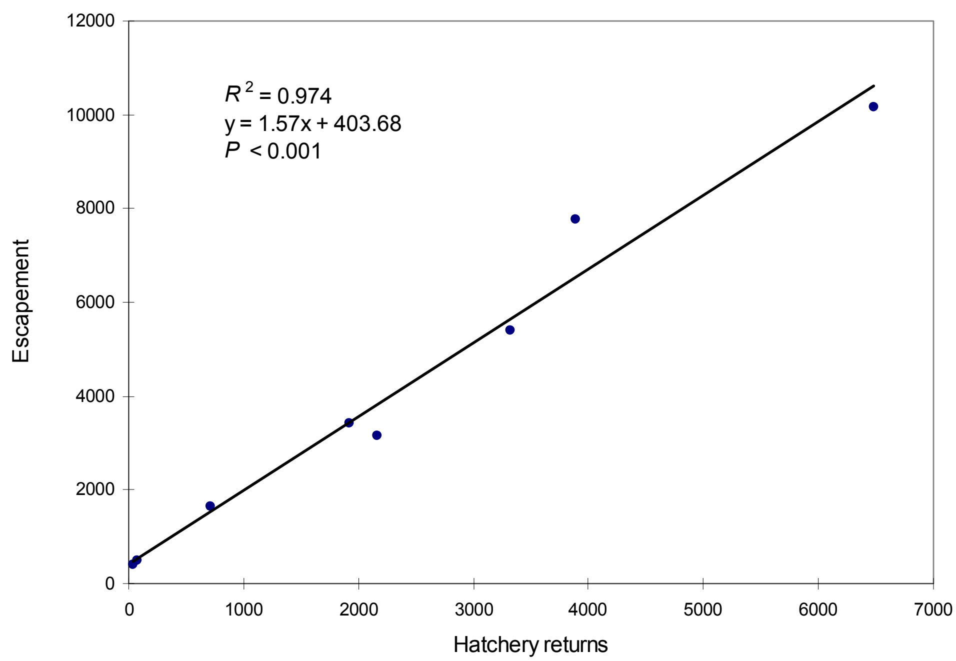

Figure 15 Linear regression of Mokelumne River escapement compared with hatchery returns, 1990–1997

Using the salmon spawning escapement data collected in the 1990–1997 monitoring program, a highly significant positive linear regression is obtained ($R^2 = 0.972$, $P < 0.001$) for the relationship between the number of salmon entering the Mokelumne River Fish Hatchery and the total spawning escapement (Figure 15). The Mokelumne River Fish Hatchery fish returns during this period ranged from 41 to 6,408 (DFG 1998). Using this relationship produces a spawning escapement estimate for 1982 of 4,590 and for 1983 of 7,548 (Table 2).

Table 2 1982 and 1983 spawning

Year	*Hatchery return*	*DFG regression estimate*	*1991–1997 regression equation estimate*
1982	2,677	9,000	4,590
1983	4,573	15,866	7,548

Salmon Redd Abundance Analysis

Fjelstad (1991) suggested that observations of the number of salmon redds could be used to generate a broad estimate of run size. Using the data collected in this monitoring program, the highly significant statistical relationship that was found between the number of redds and total escapement (see Figure 12) confirms this suggestion. This relationship regresses one empirical number on another, and because these values represent both the entire population and all salmon spawning in the river, includes spawning grilse in the estimation. Predictions of total escapement based on this relationship produces reasonable estimates at escapements in the 1,000 to 8,000 range. Multiple redds constructed by a single female, multiple superimpositions, and multiple female redds are all factors that may reduce the accuracy of these predictions. Linear regressions using redd counts and hatchery returns both provide reasonable estimates of the spawning escapements for the lower Mokelumne River (Table 3). The linear regression using hatchery returns provides the better estimate at both high and low levels of spawning escapement within the range of the database. Whether this relationship holds up in the future may depend upon the response of the natural population to habitat improvements or changes in the operation of future fish hatchery programs such as stocking levels, release locations or source of broodstock.

Table 3 Mokelumne River fall-run chinook salmon escapement, 1990–1997

Year	*Number of redds*	*Hatchery returns*	*Natural spawners*	*Escapement (WID)*	*Predicted escapement* [a]	*Predicted escapement* [b]
1990–1991	71	68	429	497	–131	511
1991–1992	127	41	369	410	264	468
1992–1993	343	711	934	1,645	1,784	1,523
1993–1994	530	2,164	993	3,157	3,100	3,810
1994–1995	774	1,918	1,503	3,421	4,817	3,422
1995–1996	888	3,323	2,094	5,417	5,620	5,634
1996–1997	1,284[c]	3,883	3,892	7,775	8,407	6,516
1997–1998	1,316	6,485	3,624	10,175	8,633	10,612

[a] Based on redds.

[b] Based on hatchery returns.

[c] 929 redds were observed through the first week of December 1996 when redd surveys were discontinued due to high flows. The value of 1,284 (±8.1%) is an estimate based on total 1992-1995 end-of-run average added to the observed number.

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Studies of Spawning Habitat for Fall-Run Chinook Salmon in the Stanislaus River Between Goodwin Dam and Riverbank from 1994 to 1997

Carl Mesick

Abstract

The spawning habitat of fall-run chinook salmon (*Oncorhynchus tshawytscha*) was studied in the Stanislaus River between Goodwin Dam and Riverbank between 1994 and 1997 to evaluate whether habitat quality was potentially limiting the population and whether two restoration projects improved spawning conditions. Redd surveys in 1994 and 1995 indicated that spawning was concentrated in the riffles located in the 12-mile reach between Goodwin Dam and Orange Blossom Bridge. Most of the spawning (73%) occurred upstream of the riffles' crests where the streambed gradient was positive (for example, the tail of a pool). Sample areas were divided into the upper, middle, and lower portions of riffles to determine why the salmon used the upper areas.

Substrate samples collected from the upper six inches of the streambed indicated that predicted survival probabilities for chinook salmon eggs using Tappel and Bjornn's (1983) laboratory study averaged 75.6% in the reach above the Orange Blossom Bridge, 58.6% in the lower spawning reach between the bridge and town of Riverbank, and 95.4% at two restoration sites near the U.S. Army Corps of Engineers' Horseshoe Road park where gravel was added in 1994. Predicted egg survival probabilities averaged 73.2% upstream of riffle crests and 62.1% downstream of riffle crests at four natural riffles with pronounced crests.

Intragravel dissolved oxygen (DO) concentrations were relatively constant at 32 piezometer sites in the 12 study riffles during five surveys conducted at 10-day intervals in November and December 1995. The DO levels declined markedly in early February 1996 at nine sites shortly after runoff from four major storms increased base flows from 300 cfs to as much as 800 cfs for several days after each storm. Prior to the storms in November and December, intragravel DO concentrations were less than 5 ppm at six piezometer sites (19%) and less than 8 ppm at eleven sites (34%). Immediately after the fifth major storm in early February, intragravel DO concentrations were less than 5 ppm

at 11 piezometer sites and less than 8 ppm at 16 sites (50%). Many of the sites where DO concentrations were low were associated with intragravel water temperatures that were between 1° and 6° F higher than surface temperatures. The elevated temperatures suggest the inflow of oxygen-poor groundwater. A high rate of groundwater inflow into the Stanislaus River's riffles would explain the unexpectedly positive vertical hydraulic gradients upstream of the riffle crests measured at most of the piezometer sites in fall 1996.

A regression model of the average intragravel DO concentrations in November and December 1995 had an adj-R^2 of 0.80 with significant ($P \leq 0.05$) variables that include an index of groundwater inflow, abundance of Asian clams (*Corbicula fluminea*), percent fines <2 mm, and mean column water velocity. A model for the February 1996 DO concentrations had an adj-R^2 of 0.68 with significant variables that include the groundwater index and the percent fines <2 mm. Although streambed gradient indexes were not selected for the regression models, DO concentrations that were greater than 80% saturation in February 1996 usually occurred where the gradient was positive 2% or higher.

Not all restoration sites in the Stanislaus River where clean gravel was added were used by spawning salmon. Two riffles constructed with imported gravel from the Merced River were used by very few fish for three years even though intragravel DO levels were near saturation and spawning occurred in the immediate vicinity. After high flows deposited a large berm of native rock at the crest of one of these riffles in spring 1997, a relatively high number of salmon began spawning in the new substrate in fall 1997. In Goodwin Canyon, where gravel was lacking, many salmon quickly spawned in newly added gravel from the Stanislaus' floodplain placed in late summer 1997.

Introduction

Two studies, one conducted in summer 1993 by the California Department of Water Resources (DWR 1994) and the other conducted in fall 1994 by Carl Mesick Consultants, Thomas R. Payne & Associates, and Aquatic Systems Research (CMC and others 1996), suggest that a majority of the spawning habitat in the Stanislaus River between Goodwin Dam and Riverbank is unsuitable for fall-run chinook salmon (*Oncorhynchus tshawytscha*). These studies reported that chinook salmon primarily spawn in the upper 30-ft sections of riffles where the streambed usually had an upward slope. The explanation for this pattern was not obvious as there were suitable water depths and velocities and an abundance of gravel in the unused, lower riffle areas.

The DWR (1994) study of 22 riffles between Goodwin Dam and Riverbank indicated that 45% of the substrate samples collected from the upper 30-ft section of the study riffles had high levels of fines (silt and sand). DWR did not sample the middle and lower sections of the riffles, but it is likely that the spawning activity concentrated in the upper sections would remove fines and make the upper sections relatively "clean" compared to the middle and lower riffle sections. If true, the percentage of riffle habitat with excessive amounts of fines would have exceeded 45%.

This report describes three years of spawning surveys from fall 1995 to fall 1997 that evaluated two questions:

1. Is habitat in the Stanislaus River's primary spawning reach unsuitable for spawning?

2. Did a riffle restoration project implemented in summer 1994 and a gravel augmentation project implemented in summer 1997 improve spawning conditions for salmon?

The first question was investigated by measuring the percentage of fines and monitoring intragravel dissolved oxygen (DO) concentrations and temperatures with piezometers buried in artificial and natural redds in the upper 30 feet, middle 30 feet, and lower portions of natural riffles between Goodwin Dam and Riverbank in fall 1995 and fall 1996. Measurements of vertical hydraulic gradient (an index of upwelling and downwelling of flow into the substrate), intragravel nitrate concentration, percentage of fines in the substrate, weight of Asian clams (*Corbicula fluminea*), streambed gradient, and the depth and velocity of the surface flow were also made to evaluate the cause of low intragravel DO concentrations.

Two restoration projects were also evaluated. The first involved two riffles that were reconstructed at the Horseshoe Recreation Area (river miles 50.4 and 50.9) by DWR in September 1994. At these sites, the streambed was excavated to a depth of 1.5 feet and then refilled with washed gravel from 0.5 to 4 inches in diameter to provide a uniform streambed (–0.2% to –0.5% gradient). The imported gravel was river-rock obtained from the Blasingame Quarry near the Merced River. Rock weirs were constructed at the upstream and downstream boundaries of each site to retain the imported gravel during high flows. The two riffles near the Horseshoe Recreation Area were surveyed for spawner use from 1994 through 1997 and intragravel conditions were monitored in fall 1995.

The second restoration project involved adding 2,000 tons of gravel to four locations in the Goodwin Canyon (near river mile 58) in summer 1997. There was almost no gravel in these areas before this project. The project was designed to create bars of introduced gravel that would be gradually transported to downstream spawning areas by high flows. At two locations, the gravel bars were placed in pools just upstream from the pool's tail at a depth of about ten feet. At the other two locations, the gravel bars were placed across the width of the river in shallow, moderately swift water. The imported gravel was river rock from 0.35 to 5 inches in diameter that was obtained from a quarry near the Stanislaus River. Shortly after the rock was placed, flows were increased to 1,200 cfs for ten days to help distribute the gravel and attract adult salmon to the river. Spawner use at these sites was surveyed in fall 1997.

This report presents a summary of the surveys conducted in the Stanislaus River from 1994 to 1997. The complete data sets and analyses for the fall 1994 and fall 1995 surveys are presented in CMC and others (1996) and for the fall 1996 survey in CMC (1997).

Methods

Surveys were conducted in the primary spawning reach of the Stanislaus River at approximately ten-day intervals in fall 1995 and 1996 to monitor spawner use and measure intragravel conditions at natural riffles and the restoration riffles at the Horseshoe Recreation Area. In 1995, six surveys were conducted between 2 November and 22 December while salmon were spawning and a seventh survey was conducted between 2 and 7 February 1996 after the fry had begun to emerge from the gravel. In 1996, three surveys were conducted between 31 October and 19 November. Flood control releases were begun on 21 November that made it impossible to continue the 1996 study. In 1997, spawner use was surveyed at the restoration sites and 12 natural riffles on 29 October and 3 December. Two of the riffles surveyed (R27 and R78) had a substantial amount of newly deposited gravel across the crest of the riffle as a result of the spring 1997 high flows (5,000 cfs with a maximum of 8,000 cfs compared to 300 to 400 cfs base flows).

Study Area

The spawning reach for fall-run chinook salmon in the Stanislaus River is about 25.5 miles long and extends from Goodwin Dam, which is impassible for salmon, downstream to the town of Riverbank (Figure 1). In a 4.2-mile, high-gradient canyon between Goodwin Dam and the Knights Ferry, U.S. Army Corps of Engineers (ACOE) Recreation Area, there are only four short natural riffles near the Two-Mile Bar Recreation Area at river mile 56.9 and several very small areas that have sufficient gravel for spawning. The largest natural riffle, which is identified as TM1, is the tail of a relatively wide pool that is just upstream of where the river divides into two channels. The double-channel riffle that begins at the pool's tail is high gradient, has no gravel, and was unused by salmon for spawning in 1994, 1995, and 1996. Two other riffles, which are identified as TM2 and TM3, are just downstream of TM1 and are relatively short, each about 30 feet long. The fourth natural riffle, which is about 150 yards upstream of TM1, was very armored and received few spawners.

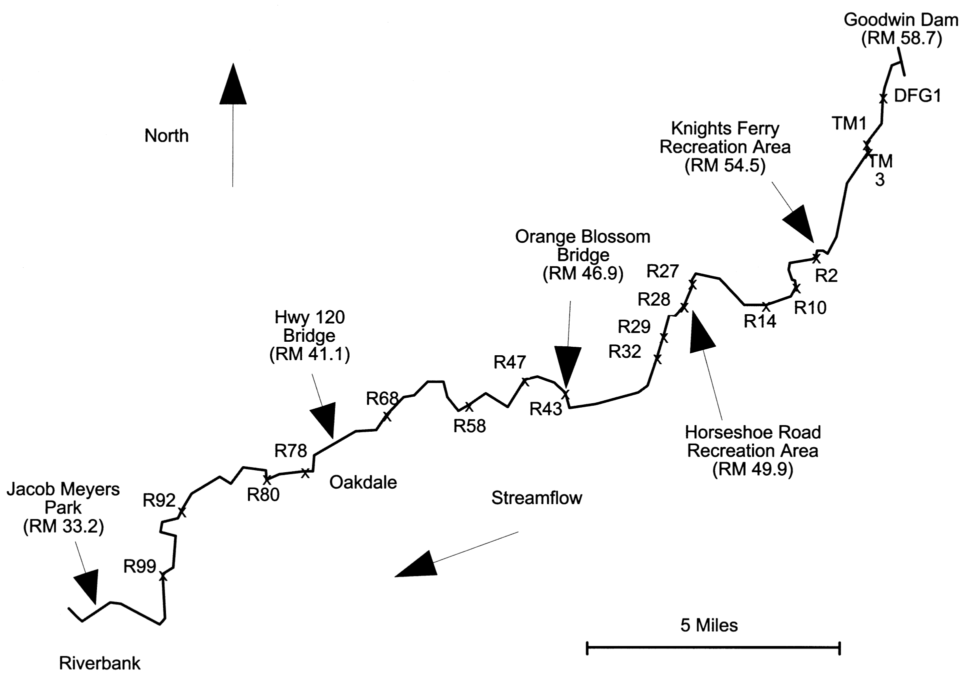

Figure 1 Location of riffles where spawning habitat for fall-run chinook salmon was studied in the Stanislaus River between Goodwin Dam and the town of Riverbank

During October 1995, 106 riffles between the ACOE Knights Ferry Recreation Area and Jacob Meyers Park in Riverbank were identified with a numbered 3-inch orange square that was nailed to either a tree or woody debris near the upstream boundary of each riffle. The riffle immediately upstream of the bridge at the ACOE Knights Ferry Recreation Area was identified as "R1." The other riffles were sequentially numbered in a downstream direction from there. During the fall 1995 surveys, salmon were observed at an additional 26 riffles and four small accumulations of gravel that were not numbered with orange squares. These areas were identified by adding a letter to the upstream riffle's number. For example, an unmarked spawning area downstream of Riffle R2 was called Riffle R2A.

Redd Distribution Within Riffles and the Spawning Reach, Fall 1995

Redds, "test-digs," and the number of live and dead adult salmon were counted at each of the 135 riffles between Goodwin Dam and Riverbank during the 1995 November and December surveys. Redds were identified as a disturbance in the substrate, approximately four feet wide by eight feet long, with a shallow pit or depression near the middle of the disturbed area and a tail spill. Test-digs were assumed to be unfinished redds that lacked eggs, because they lacked a pit and tail spill. Redds and test-digs were most conspicuous when first constructed because (1) the depth of the pit and the height of the tail spill were gradually reduced by the "smoothing action" of streamflow and (2) redd construction temporarily reduced the amount of algae and silt on the substrate's surface. Since some, but not all, of the redds became indiscernible during the study, it was necessary to distinguish new redds from previous redds. New redds were identified by comparing their appearance to old redds in the same riffle, the location within the riffle, and whether adult salmon were observed near the new redds. The total number of redds at each riffle was estimated as the cumulative total of new redds observed during each survey.

Redd counts for each riffle were subdivided into a maximum of three sections with the two uppermost sections being about 30 feet long each. For example, a 30-ft riffle had only an upper section, whereas a 120-ft riffle was subdivided into an upper 30-ft section, a middle 30-ft section, and a lower 60-ft section. The boundaries between riffle subsections were not measured or marked but visually estimated for each survey.

Surveys were conducted on foot and by canoe. The Two Mile Bar Recreation Area was accessed by road and observations were first made from the streambank and then by walking through the riffles. The reach between the Knights Ferry Recreation Area and Jacob Meyers Park in Riverbank was surveyed with two canoes, one on each side of the river. Visibility in the water column was usually about eight feet and so most of the streambed and all redds were easily observed in riffles, which ranged in depth between one and 3 feet. Streamflows releases at Goodwin Dam were consistent at about 305 cubic feet per second (cfs) during the 1995 surveys and 400 cfs during the 1996 and 1997 surveys.

Intensively Studied Riffles

Spawning habitat and redd distribution was intensively studied at 12 riffles in fall 1995 and at seven riffles in fall 1996 (Figure 1). The fall 1995 study riffles (TM1, R2, R10, R27, R28, R32, R43, R47, R68, R80, R92, and R99), were selected at approximately two-mile intervals between Goodwin Dam and Jacob Meyers Park in Riverbank. They were selected because they were highly used for spawning during fall 1994, a condition that was necessary to evaluate the relationship between redd distribution and the quality of incubation habitat. Riffles TM3, R10, R14, R29, R43, R58, and R78 were selected for the fall 1996 study because each had an upper section with an upward slope or positive streambed gradient, a relatively flat middle section, and a bottom section with a negative streambed gradient. These selection criteria made it possible to evaluate the effect of streambed gradient on the downwelling of surface water and intragravel conditions. Riffle R10 was studied during both fall 1995 and fall 1996. Different sections of Riffle R43 were studied in 1995 and 1996; a small concrete weir separated the lower section which was studied in 1995 from the upper section studied in 1996.

Intragravel Water Quality

Intragravel water samples were collected from piezometers buried 12 inches deep in the substrate. Piezometers were installed between 2 and 4 November in 1995 and between 25 and 27 October in 1996. One piezometer was installed at each of the top, middle, and lower sections of each riffle, except at Riffle R10 where two were installed in each section in fall 1996.

Typical piezometers were 0.25-inch diameter copper tubes, each with eight 0.04-inch diameter holes at one end and a flexible tube at the other end that extended above the substrate surface (Figure 2). Redd construction was extensive at Riffle TM1 in October 1995, and so a different design, called a pipe-piezometer, was used which did not require streambed excavation. Pipe-piezometers consisted of 0.33-inch outside diameter hollow aluminum shafts that were driven straight down into the substrate so that eight 0.04-inch holes in the shafts were approximately 12 inches deep in the substrate and the top of the shaft extended about ten inches above the substrate. A 3-foot-long plastic tube was clamped to the upper end of the shaft for sample collection. Each pipe-piezometer was attached to a 4-foot-long, 0.5-inch diameter reinforcing bar with hose clamps to facilitate driving the shaft into the substrate. The pipe-piezometers at Riffle TM1 were left in place throughout the study, although they fell after the 20 December 1995 survey and had to be reinstalled on 3 February 1996. A pipe-piezometer was used at the top piezometer site in Riffle R2 because the buried piezometer was quickly vandalized. This pipe-piezometer was reinstalled during each survey.

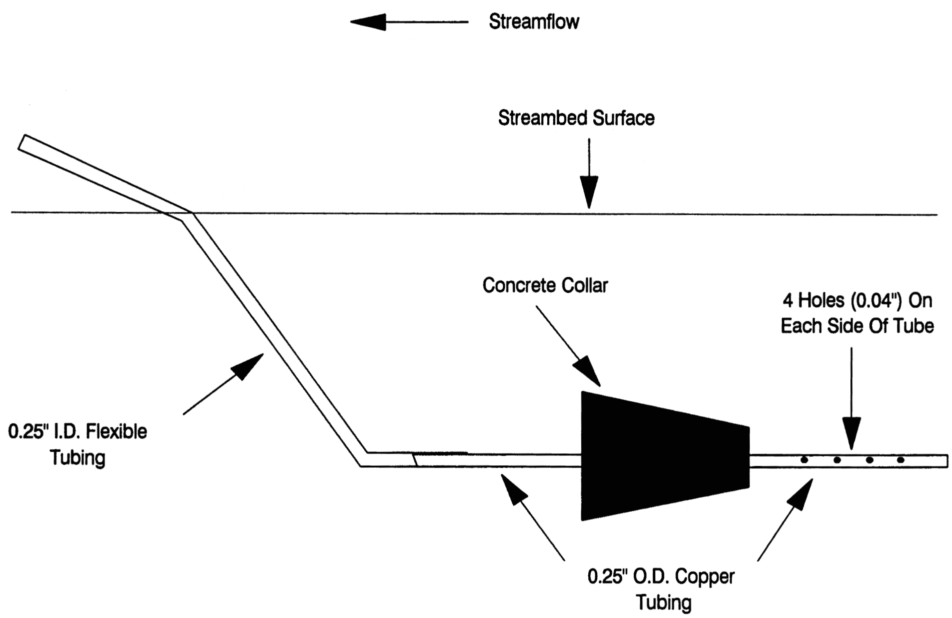

Figure 2 Typical piezometer used to collect intragravel water quality samples. The concrete collar was the shape and size of an eight-ounce Dixie cup. The copper tubing was about 11 inches long and the flexible tubing was about three feet long.

Piezometer sites were selected in areas where water depths ranged between 1.1 and 2.4 ft and mean column water velocities ranged between 1.6 and 4.2 ft/s, which were within the range used by spawning salmon. The typical piezometers (Figure 2) were installed to simulate sampling an egg pocket in a natural salmon redd. Pits were dug approximately 12 inches deep by 12 inches wide at the bottom with a hand-held hoe. The excavated substrate was piled downstream of the pit to simulate the tail spill formed in a natural redd. After the piezometer was placed in the pit, sediment was pulled into the pit in thin layers from the upstream areas using the hoe. The blade of the hoe was then fanned over each layer of gravel in the pit to flush most of the fines onto the tail spill. When completed, the piezometer was located at the upstream end of the tail spill which was raised several inches above the undisturbed streambed. An egg pocket would be expected to occur in this location in a natural redd. Immediately upstream of the tail spill, there was a two- to four-inch deep depression in the substrate that simulated the pit of a small, but natural-looking, redd. However, natural sediment transport filled the depression, and the tail spill was flushed away at most of the piezometer sites after approximately seven days. This smoothing of the streambed also occurred at natural redds.

Pipe-piezometers were also installed in the vicinity of the egg pockets of 21 completed salmon redds at study riffles R10, R14, and R43 on 11 and 12 November 1996. All of these redds had been observed on the previous survey when most had already been completed. These pipe-piezometers were driven approximately 12 inches into the upstream end of the tail spill of a completed redd. This was done by placing a 5/8-inch bolt into the upper end of the piezometer, inserting it into the bottom of a 4-foot-long, 0.5-inch ID steel pipe, placing a 3-foot-long, 0.25-inch diameter rod on top of the lag bolt, and then driving the pipe, rod, and piezometer into the redd. The pipe and rod were then removed and a plastic tube was fitted to the upper end of the piezometer, which extended about two inches above the substrate's surface to permit periodic collection of water samples. To avoid collecting surface water that may have been introduced into the redd during installation, the first measurements and samples were not collected until approximately five minutes after the pipe-piezometers had been installed. To minimize disturbance to the eggs, the pipe-piezometers were not removed between surveys.

Intragravel measurements of DO concentration, water temperature, and vertical hydraulic gradient were made at each piezometer in the artificial redds at approximately ten-day intervals. During the fall 1995 study, five measurements were made between 11 November and 22 December 1995 and a sixth measurement was made between 2 and 7 January 1996. During the fall 1996 study, three measurements were made between 31 October and 19 November 1996.

Two sets of measurements were taken from the redd-type piezometers between 11 and 19 November 1996. During 1997, measurements were made only at Riffles R27 and R78 on 3 December.

Intragravel water samples were collected to measure DO concentration and temperature using a 50-ml polypropylene, disposable syringe (Henke-Sass Wolf GmbH, Germany) fitted with a six-inch-long, 1/8-inch inside diameter polypropylene tube and a tapered connector that provided an airtight seal between the piezometer's tubing and the syringe's tubing. Samples were collected by first slowly withdrawing and discarding 50 ml of water, the approximate volume of water in the piezometer's tubing. Then a 70-ml sample was slowly withdrawn for a DO analysis using a LaMotte test kit, model EDO/AG-30. The LaMotte test kit uses the azide modification of the Winkler Method and a LaMotte Direct Reading Titrator for the final titration. The LaMotte Kit measures DO concentrations in 0.1 parts per million (ppm) increments. Kit reagents were replaced for each survey. During the fall 1995 and fall 1997 studies, samples were analyzed at the study riffles within five minutes of collection. During the fall 1996 study, the DO samples were fixed immediately after collection, placed in an ice chest, and analyzed at room temperature within ten hours.

After collecting the DO sample, a 100-ml sample was slowly collected and injected into a plastic sample bottle, which had been rinsed with surface water, for an immediate measurement of intragravel water temperature. Water temperature was measured with a Yellow Springs Instrument (YSI) model 55 meter in 1995 and with a mercury thermometer in both 1995 and 1996. A sample of surface water was also collected in the same 100-ml plastic bottle for a temperature measurement. The date and time that each water sample was collected were recorded for comparison with the thermograph recordings.

Nitrate concentration was determined for intragravel and surface water sampled during the 11–12 December 1995, 20–22 December 1995, and the 19 November 1996 surveys. During 1995, AquaCheck Nitrate/Nitrite test strips were used, which measure nitrate concentrations between zero and 50 mg/liter. The test strips were dipped into the samples collected for the LaMotte DO tests immediately prior to adding any of the LaMotte reagents. During 1996, one intragravel water sample was collected from one piezometer at each riffle for analysis of nitrate concentration. One surface sample was collected at Riffle R29 for nitrate analysis. These samples were immediately placed in an ice chest and analyzed by FGL Environmental in Stockton approximately 24 hours later.

The ratio of the differential head to the depth of the piezometer below the sediment-water interface (Lee and Cherry 1978; Dahm and Valett 1996) is known as the vertical hydraulic gradient (VHG). Negative VHG measurements indicate the downwelling of surface flow and positive values indicate the upwelling of intragravel flow. VHG was measured at each piezometer in fall 1996 and fall 1997. The differential head was measured with a manometer, which consisted of an 8-ft-long, 1/8-inch inside diameter, clear tube with one end attached to the piezometer’s tubing and the other held near the substrate surface (Lee and Cherry 1978; Dahm and Valett 1996). A silicone pipet bulb with emptying and filling valves was attached to the middle of the tubing with a T-connector to facilitate filling the manometer with water. Measurements were made by partially filling the manometer's tubing with water and then holding the middle of the tube above the water’s surface to form a loop with two vertical tubes and an air bubble, approximately 16 inches long at the top of the loop. After the water levels in both sides of the manometer’s tubes stabilized, the differential head was read as the difference in height (centimeters) between the water levels in the two tubes. Negative measurements occurred when the water level in the side of the tube connected to the piezometer was lower than the level in the side of the tube submerged in surface water. VHG was computed as the differential head divided by 30 cm, the depth of the piezometers below the substrate’s surface.

Streambed Elevation, Water Velocity, and Water Depth at Piezometers and Redds

Redds were identified by placing a numbered, three-ounce lead sinker with red flagging into the pit. The sinkers were replaced whenever redd construction buried or displaced the original sinker. The locations of redds and piezometers were mapped at each riffle using permanent headstakes (for example, 18-inch sections of reinforcing rods or nails partially driven into trees) on both sides of the river. The location of redds and water quality samples were determined by first running a tape measure from the redd or sample to the closest permanent transect at a perpendicular angle. The distance in feet from the right streambank along the permanent transect to the perpendicular line was recorded as the station. Then the distance in feet from the permanent transect to the redd or piezometer and the direction (upstream or downstream) from the permanent transect were recorded.

Water depth and mean column velocity were measured at the undisturbed substrate surface immediately adjacent to all piezometer sites, including those at redds, during 300-cfs releases in 1995 and 400-cfs releases in 1996. Mean column velocities were measured with a Marsh McBirney electronic flow meter and a top-set wading rod.

In fall 1995, a single longitudinal transect was used to measure the streambed elevation along the entire length of each study riffle. The transect was established by installing a three-foot piece of reinforcing bar about ten feet upstream of the riffle and then running a tape through the site to approximately ten feet downstream of the riffle. Relative streambed elevations were measured along the transect with an automatic level and a fiberglass stadia rod at five-foot intervals in steep gradient (>2%) sites or at ten-foot intervals in low gradient (<2%) sites. The absence of large structures in the study riffles produced a relatively uniform contour of the streambed across the river and it was assumed that the transect represented the entire width of the river.

In fall 1996, streambed elevations were measured at each piezometer site and at distances of 5 ft, 10 ft, and 20 ft immediately upstream of the piezometer to determine the streambed gradient.

Substrate Size Distribution, Crushed Rock, and Asian Clams

During 2–7 February 1996 substrate samples were collected at each of the piezometer sites after the water quality sample had been collected. Samples were taken with a six-inch diameter modified McNeil sampler. The sampler was placed over the approximate location of the piezometer and worked into the substrate to a depth of about six inches. If the sampler could not be inserted to a depth of six inches, the sampler was moved about one foot. The substrate inside the sampler was scooped into a plastic bag for transport to the laboratory for analysis. The water inside the sampler was not collected, because the sampler was too short to prevent a small amount of river water from entering at some sites, thereby resulting in the loss of some of the suspended sediments. Furthermore, it is likely that the total weight of the suspended sediments was small and would not have significantly affected the measured weight of fines <1 mm.

The samples were processed by first drying the samples, sorting the particles according to size, then determining the weight of each size class. The samples were dried by placing them in two-gallon metal buckets and occasionally stirring them as they were heated over a propane flame. After the sample had cooled, it was placed in the upper layer of a set of sieves of decreasing size. Sieve sizes used included 64 mm, 45 mm, 32 mm, 24 mm, 16 mm, 8 mm, 4 mm, 2 mm, 1 mm, 0.5 mm, and 0.25 mm. Sieves between 16 and 64 mm were shaken by hand for one minute, after which the size of the particles in each sieve was checked. Smaller sieves were shaken for about five minutes. After the sorting had been completed and before the sieve's contents were weighed, Asian clam (*Corbicula fluminea*) shells were weighed, counted, and removed from each sieve. Sieve contents were weighed on a triple-beam balance to the nearest 0.1 grams. The percentage of broken rock, 8-mm or larger particles with sharp edges, was determined for most of the samples, particularly those from Riffles R27 and R28, which are the 1994 restoration sites.

Percent fines were evaluated as two size classes, one as substrate particles less than 1 mm to correspond to the results of Young and others (1991) and the other as particles less than 2 mm, which corresponds to many other studies (Chapman 1988). Both of these size classes were evaluated as a percentage of the entire sample and as a percentage of the sample that excluded particles larger than 24 mm to minimize weight bias as recommended by Tappel and Bjornn (1983). To account for potential bias that would result if smaller sample volumes consisted of surface substrates that typically have lower percentages of fines, total sample weight was included in the statistical analyses of percent fines.

The predicted survival probability for chinook salmon eggs was estimated using the results of Tappel and Bjornn's (1983) laboratory study based on the percentages of substrate particles less than 0.85 mm and particles less than 9.5 mm in samples that exclude particles larger than 51 mm (largest size tested by Tappel and Bjornn). The percentages of particles less than 0.85 mm and 9.5 mm for Stanislaus River samples that excluded particles larger than 51 mm were estimated from a plot of the cumulative percentage of particles that passed through specific sieves versus the sieve apertures on a log scale. The following equation from Tappel and Bjornn (1983) was then used to compute predicted egg survival for chinook salmon:

$$\text{Percent Survival} = 93.4 - 0.171 S_{9.5} S_{0.85} + 3.87 S_{0.85}$$

Statistical Analyses

Multiple regression analyses were conducted to evaluate the environmental factors that appeared to influence DO concentrations during the November–December 1995 surveys and the February 1996 survey. The mean DO concentrations in percent saturation were used for the November–December surveys, since those concentrations were relatively stable and showed no increasing or decreasing trends during that period. DO concentrations measured during the February survey were substantially lower at some of the piezometer sites and so those data were evaluated separately.

The environmental factors evaluated for both analyses included six indices of percent fines, three indices of gradient, weight of clam shells, water depth, mean column velocity, two indices of nitrate concentration, a turbidity index, and the difference in temperature between the intragravel sample and the surface sample during the 20–22 December survey which served as an index of groundwater inflow (Tables 1 and 2). The variables for nitrate concentration, turbidity, and the temperature difference are described in further detail in the "Results and Discussion."

Transformations and tests were made on the assumptions of statistical analyses relating to normal distributions, linear relationships, and an absence of collinearity between independent variables (Sokal and Rohlf 1995). An arcsine transformation was made to variables consisting of percentages (which include DO concentrations, the absolute value of the gradient indexes, and substrate fines), to minimize bias resulting from the distribution of the variance being a function of the mean (Sokal and Rohlf 1995). "Wilk-Shapiro/rankit plots" were used to test the assumption of normality. Plots of standardized residuals versus fitted values of the independent variables were used to assess the assumptions of linearity and constant variances. A variance inflation factor was used to detect collinearity (Analytical Software 1994).

The regression analyses were affected by the absence of water depth and mean column velocity measurements at the piezometer sites at TM1 and the middle piezometer at Riffle R47. When the analyses included the depth and velocity variables, all variables from these sites were excluded from the analyses. To avoid this limitation, regression analyses were conducted with and without the depth and velocity variables.

Table 1 Habitat data from 32 piezometers at 12 riffles in the Stanislaus River, fall 1995

	Intragravel DO percent saturation [a]		*Groundwater Inflow Index (°C)* [b]			*Percent fines*					*Tappel and Bjornn Index* [d]		
					Asian clam shells (g) [c]	*Entire sample*		*Excludes >24 mm*			*Percent substrate*		
Site	*Nov-Dec*	*Feb*	*Dec*	*Feb*		*<1 mm*	*<2 mm*	*<1 mm*	*<2 mm*	*Sample Size (g)*	*<0.85 mm*	*<9.5 mm*	*Egg Survival*
TM1-TOP	94.3	100.0	0.1	1.1	0.0	2.3	5.9	3.7	9.3	7327.6	2.1	37.6	88.0
TM1-BOTR	95.1	100.0	0.05	1.3	0.0	2.6	7.1	4.6	12.5	9567.5	2.4	36.8	87.6
TM1-BOTL	99.6	100.0	0.05	1.3	0.0	1.5	3.2	5.2	11.5	8195.6	1.5	15.3	95.3
R2-TOP	96.8	100.0	0.3	1.4	0.0	5.3	7.3	22.9	31.7	4485.6	6.3	19.0	97.3
R2-BOT	88.8	54.8	0.3	1.2	7.0	3.0	8.8	7.7	22.5	7685.3	3.3	28.8	89.9
R10-TOP	94.4	84.3	0.2	1.0	0.0	3.7	7.6	7.8	15.8	9453.3	4.1	41.7	80.0
R10-MID	61.8	32.4	1.1	3.0	0.0	8.3	12.8	15.8	24.4	9597.4	7.7	40.4	70.0
R10-BOT	64.7	36.0	0.8	1.0	0.0	13.4	24.9	21.5	40.0	8147.5	11.7	50.7	37.2
R27-TOP	98.8	100.0	0.3	0.0	0.0	0.4	0.4	0.9	0.9	5376.4	0.4	3.6	94.7
R27-MID	97.4	100.0	0.3	0.5	0.0	0.5	0.5	1.8	1.8	5342.0	0.6	2.3	95.5
R27-BOT	97.8	100.0	0.4	0.5	0.2	0.1	0.1	0.2	0.3	4836.4	0.1	3.5	93.7
R28-TOP	99.4	98.6	0.5	1.0	28.7	9.2	9.9	27.3	29.4	7133.6	8.7	13.8	100.0
R28-MID	98.1	97.7	0.3	1.0	0.3	0.4	0.4	0.9	1.0	6113.1	0.4	3.4	94.7
R28-BOT	98.7	94.1	0.2	0.5	8.9	9.3	13.1	17.0	23.8	7057.1	7.6	22.4	93.7
R32-TOP	93.4	87.0	0.6	1.0	10.5	7.1	16.4	14.2	32.9	13483.0	7.8	48.5	58.9
R32-BOT	89.6	82.0	0.6	1.0	3.6	10.5	19.9	18.6	35.2	8263.9	10.9	48.9	44.4
R43-TOP	92.6	16.2	0.9	2.2	2.6	2.8	9.8	4.4	15.6	8383.0	2.7	47.3	82.0
R43-BOT	96.0	82.4	0.4	0.9	0.0	3.2	8.9	5.9	16.2	8887.1	3.0	35.4	86.8
R47-TOP	78.7	66.0	0.7	1.1	6.2	8.1	20.9	12.6	32.7	9240.6	6.8	47.1	64.9
R47-MID	42.3	41.2	0.8	0.7	95.1	12.1	19.7	19.7	32.0	10067.0	9.5	41.7	62.4
R47-BOT	44.1	34.1	1.1	1.1	79.5	21.0	28.9	26.8	37.0	9439.9	17.0	53.0	5.1
R68-TOP	39.0	40.1	1.1	1.1	57.9	12.0	23.1	14.3	27.5	8219.5	9.4	47.8	52.9
R68-MID	90.0	76.0	0.3	1.1	1.7	10.2	20.0	15.8	31.0	9459.6	9.0	45.9	57.6
R68-BOT	92.9	95.0	0.1	1.1	3.5	7.7	14.7	12.1	23.1	7423.4	6.3	41.4	73.2
R80-TOP	94.6	66.7	0.1	1.4	5.3	4.0	8.8	8.1	18.0	7651.1	3.4	24.4	92.4
R80-BOT	62.9	28.5	1.0	0.9	0.3	6.3	12.8	12.1	24.6	9396.5	5.5	31.9	84.7
R92-TOP	20.1	18.4	0.5	-0.2	147.0	20.8	33.4	27.4	43.9	7681.3	16.9	55.6	0.0
R92-MID	82.1	81.8	0.2	1.2	4.4	4.9	13.3	7.9	21.5	8620.7	4.0	39.2	82.1
R92-BOT	82.9	45.9	0.3	0.0	3.6	6.3	16.7	16.9	44.7	10394.0	9.2	52.2	46.9
R99-TOP	85.4	38.3	0.6	0.6	0.0	11.1	20.7	18.9	35.5	9551.9	10.9	45.4	51.0
R99-MID	94.5	86.2	0.4	0.3	1.8	4.0	8.8	8.7	19.3	9254.9	4.7	31.6	86.2
R99-BOT	53.1	27.7	0.8	0.1	2.8	8.0	18.2	11.9	26.9	8915.6	6.9	48.2	63.2

[a] Mean levels for five November and December 1995 surveys, including one measurement for February 1996.

[b] Computed as the difference in temperature between the intragravel sample and the surface sample during the 20–22 December 1995 survey.

[c] From February 1996 substrate samples.

[d] Source: Tappel and Bjornn 1983.

Table 2 Habitat data from 32 piezometer sites at 12 riffles in the Stanislaus River, fall 1995

Site	Streambed gradient upstream of samples [a]			Mean column velocity (ft/s) [b]	Water depth (ft) [b]	Surface water turbidity (JTU) [c]	Intragravel nitrate concentration (ppm) [d]		Channel width (ft)	Distance downstream of Goodwin Dam (mi)
	5 ft	10 ft	20 ft				Sample	Diff		
TM1-TOP	7.0%	3.6%	4.1%	-	-	3	1	0.5	99.0	1.5
TM1-BOTR	5.0%	7.6%	5.4%	-	-	3	0.5	0	99.0	1.5
TM1-BOTL	5.0%	7.6%	5.4%	-	-	3	0.5	0	99.0	1.5
R2-TOP	9.1%	8.7%	5.0%	3.1	1.8	3	0.5	0	92.0	3.9
R2-BOT	-5.8%	-5.2%	-2.7%	2.7	1.3	3	0.5	0.25	92.0	3.9
R10-TOP	4.8%	5.8%	2.7%	1.5	1.3	5	0.5	0	82.2	4.9
R10-MID	-1.2%	-0.8%	-0.0%	0.5	1.3	5	0.5	0	81.0	4.9
R10-BOT	-1.4%	-3.3%	-5.4%	1.3	1.1	5	0.5	0	80.0	4.9
R27-TOP	0.6%	5.5%	0.7%	2.1	1.5	10	0.5	0	91.0	7.4
R27-MID	9.8%	4.6%	1.6%	2.9	1.7	10	0.5	0	85.6	7.4
R27-BOT	-3.2%	-1.6%	-0.7%	1.4	1.6	10	0.5	0	75.0	7.4
R28-TOP	2.0%	2.1%	1.2%	3.0	1.3	10	0.5	0	84.0	7.9
R28-MID	-3.4%	-2.4%	-0.8%	3.2	1.3	10	0.5	0	86.0	7.9
R28-BOT	14.8%	8.3%	6.0%	2.4	1.0	10	0.5	0	88.0	7.9
R32-TOP	2.2%	4.1%	3.3%	3.4	1.7	7	0.25	0.25	63.4	8.9
R32-BOT	-4.2%	-2.2%	-2.3%	3.9	1.2	7	0.25	0.25	65.6	8.9
R43-TOP	0.6%	2.2%	6.8%	2.2	1.6	7	0.5	0	80.1	11.5
R43-BOT	1.0%	1.2%	1.0%	2.0	1.0	7	1.0	0	80.1	11.5
R47-TOP	0.8%	-1.3%	-1.6%	1.4	1.6	7	0.5	0	91.6	12.4
R47-MID	0.2%	0.2%	-1.0%	-	-	7	1.0	0.5	96.1	12.4
R47-BOT	0.3%	0.3%	0.2%	1.1	1.7	7	1.0	0	90.0	12.4
R68-TOP	0.8%	-0.2%	1.6%	1.6	1.2	5	5.0	2.8	143.0	16.3
R68-MID	-2.9%	-2.9%	-1.7%	1.8	1.2	5	1.5	0	70.7	16.3
R68-BOT	-0.6%	-0.6%	1.5%	1.7	1.9	5	0	0.3	62.5	16.3
R80-TOP	0.8%	1.3%	1.9%	2.5	1.4	10	2.0	0.5	64.0	19.1
R80-BOT	-3.6%	-1.5%	-0.4%	2.5	1.3	10	1.5	0	64.0	19.1
R92-TOP	-1.9%	-0.5%	0.4%	0.7	1.5	5	1.0	0	152.0	21.2
R92-MID	-1.2%	-1.2%	-0.5%	0.9	1.6	5	1.5	0.25	113.0	21.2
R92-BOT	3.4%	3.4%	1.5%	3.1	1.4	5	1.0	0	109.0	21.2
R99-TOP	0.6%	0.0%	0.1%	1.5	1.6	5	1.0	0	81.2	23.1
R99-MID	1.6%	1.6%	2.5%	2.5	1.5	5	1.0	0	86.5	23.1
R99-BOT	0.0%	0.0%	0.2%	1.3	1.5	5	2.0	0.8	82.2	23.1

[a] Measured 2–7 February 1996.

[b] Measured 2–4 November 1995.

[c] Measured 11 and 20 December 1995.

[d] Intragravel nitrate concentration and the difference between the intragravel sample and the surface sample. Collected 11–12 December 1995.

Habitat variables were selected for the final regression models by evaluating Pearson correlation coefficients (r), adjusted multiple coefficients of determination (adj-R^2) and Mallow's C_p statistic for all possible combinations of variables, and stepwise regression procedures using Statistix 4.1 software (Analytical Software 1994). The relative importance of the variables was evaluated with the t-statistic (Bring 1994).

Results and Discussion

The California Department of Fish and Game's preliminary estimate of chinook salmon escapement (grilse and adults) to the Stanislaus River was 1,079 for fall 1994, 611 for fall 1995, and 168 for fall 1996 (G. Neillands, personal communication, see "Notes"). Escapement during these studies was relatively low compared to an average of 4,800 salmon that occurred between 1967 and 1991.

Distribution of Redds in the Spawning Reach, Fall 1995

A total of 415 redds was observed during 1995, with the highest density (50) occurring at the uppermost Riffle TM1. In the downstream area between the Knights Ferry Recreational Area and the bridge at Orange Blossom Road, there was an average of five redds per riffle. Downstream of the Orange Blossom Bridge to Jacob Meyers Park in Riverbank, the average number of redds per riffle was less than two. No redds were observed in the lowermost three miles of this reach. During 1994, a total of 714 redds was observed between Two Mile Bar and Jacob Meyers Park that were distributed similarly to those in 1995.

As in 1994, most of the redds observed in 1995 were constructed near the head (upstream boundary) of the riffles, even though the entire riffle appeared to be suitable for spawning. Of the 337 redds that were observed at the 129 riffles between the Knights Ferry Recreation Area and Jacob Meyers Park, 72.6% (244 redds) occurred within the uppermost 30-ft section of the riffles, 13.4% (45 redds) occurred in the middle 30-ft section of the riffles, and 14.0% (47 redds) occurred in the lowermost section of the riffles, some of which were 200 ft long.

Redd Distribution at the Intensively Studied Riffles

The distribution and number of redds observed at most of the intensively studied riffles were similar from 1994 to 1997. The highest density typically occurred at Riffle TM1 (50 redds in 1995), moderate densities (6 to 18 per riffle) occurred at riffles R2, R10, R14, R32, R43, R47, R68, and R80 between Knights Ferry and Oakdale, while few or no redds were observed at riffles

R29, R58, and R78. Riffle TM3 had a moderate number of redds in 1995 (11) and 1997 (7), but only two redds had been completed by 19 November 1996. Riffles R92 and R99 each had nine redds in fall 1994, but none in fall 1995 when escapement was relatively low.

Of the 113 redds that were mapped with the longitudinal transects at the study riffles in fall 1995, 73 (64%) occurred where the streambed's gradient was increasing, 11 (10%) occurred where the streambed's gradient was decreasing, and 29 (26%) occurred where the streambed was relatively flat. Redd distribution and streambed configuration of Riffle R10 in fall 1995 were typical for the Stanislaus River (Figure 3).

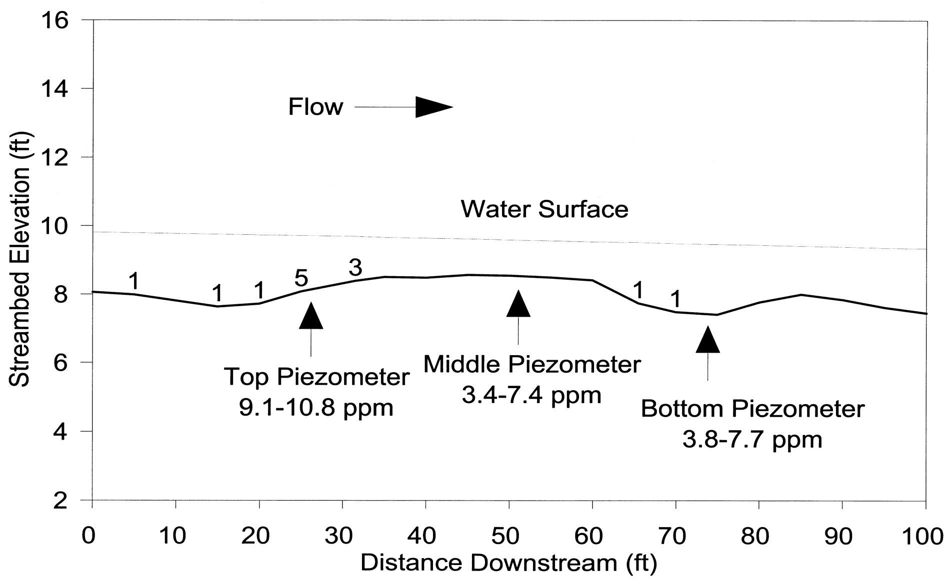

Figure 3 Map of Riffle R10 in the Stanislaus River showing the number of redds, locations of piezometers, range of DO concentrations observed between 11 November 1995 and 4 February 1996, and the water surface elevation relative to the streambed elevation recorded along a mid-channel longitudinal transect

1994 DWR Restoration Sites at the Horseshoe Recreation Area

At Riffles 27 and 28, few of the redds were constructed entirely in the added mitigation gravel obtained from the Merced River floodplain, whereas several redds were usually observed near the mitigation gravel in natural substrate. In fall 1994, all 11 redds observed at these riffles were constructed in the natural cobble substrate adjacent to the mitigation gravel. In fall 1995, only one redd at each riffle was constructed entirely in the mitigation gravel at R27 and R28. Two redds were observed immediately upstream of Riffle R27. Nine redds were constructed in cobble substrate upstream of the mitigation gravel, and two others were constructed in predominately cobble substrate where a fallen cottonwood tree had scoured away all of the mitigation gravel in the middle of Riffle R28 (Figure 4). No redds were observed in the large deposit of mitigation gravel approximately 20 ft downstream of Riffle R28 that had been moved by the high streamflows that occurred in spring 1995.

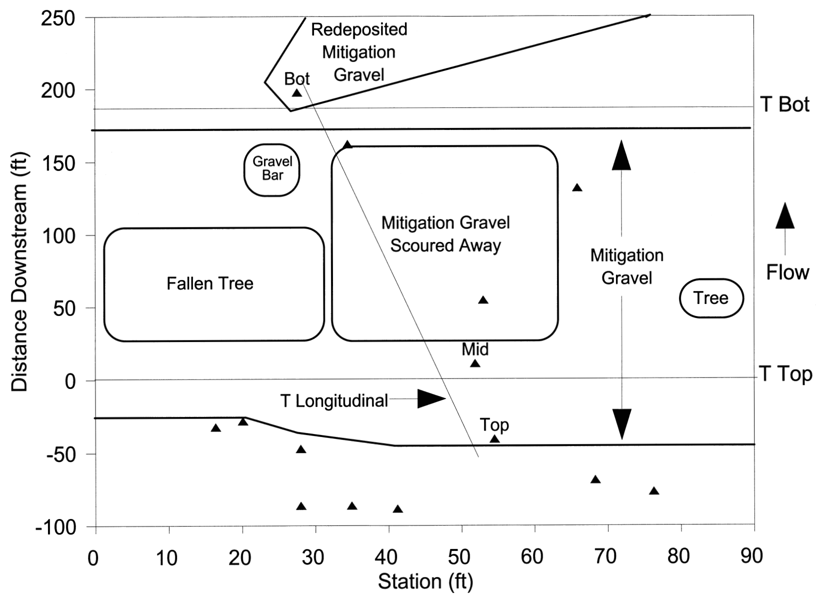

Figure 4 Map of Riffle R28, the lower mitigation site at the Horseshoe Road Recreation Area, in the Stanislaus River showing the locations of fall-run chinook salmon redds (black triangles), piezometers (black triangles identified as Top, Mid, and Bot sites), the permanent transects (T Top and T Bot) used to record the location of redds, the longitudinal transect (T Longitudinal) used to measure streambed elevations in fall 1995, and the location of mitigation gravel placed in summer 1994

By November 1996, all of the mitigation gravel from the lower one-third of Riffle R27 and from the entire riffle at R28 had been scoured away, presumably during high flows in spring 1996. The mitigation gravel that had been deposited immediately downstream of Riffle R28 in spring 1995 had been entirely scoured away in spring 1996. No redds or spawning fish were observed at Riffles R27 or R28 by 1 November 1996. Two redds were observed approximately 20 ft upstream of Riffle R28 on 1 November 1996.

In 1997, a 15-ft-long, 2-ft-high natural gravel berm had been deposited across the width of the river channel at the upstream boundary of Riffle R27, presumably as a result of the prolonged flows (5,000 to 8,000 cfs) in spring 1997. Nine redds and six pairs of spawning chinook salmon were observed in the newly deposited natural gravel on 29 October 1997; none were observed in the mitigation gravel that remained in the middle of the original riffle. On 3 December 1997, there were seven additional redds in the newly deposited natural gravel and one redd in the mitigation gravel. No redds were observed at

Riffle R28, although three adult salmon were observed near the top of the riffle on 29 October.

1997 DFG Restoration Sites in Goodwin Canyon

Chinook salmon spawned at all four locations where gravel from the Stanislaus floodplain had been added to the river in summer 1997. Most of the spawning occurred where gravel bars were placed across the width of the river in shallow, moderately swift water, but fish also spawned in the newly added gravel in pools where water velocities were quite low. On 29 October 1997, five redds were observed at both sites where gravel bars were placed across the width of the river in shallow (one to two feet deep), moderately swift water (about 2 ft/s). No redds were observed in gravel that had been mobilized and deposited in high gradient areas where the water was swift (>4 ft/s) or in the deep pools. On 3 December, there were three new redds at one shallow site, whereas the other shallow site appeared to be completely covered with redds (at least ten more redds). Two to three redds were also observed at each of the pool sites. However, no redds were observed where the new gravel had been redeposited in very swift water.

Evidence of Redd Superimposition

Seven piezometers had to be replaced in fall 1996, when salmon presumably constructed redds on top of them. The top piezometer at Riffle TM3 and the middle right piezometer at Riffle R10 and their thermographs had been excavated by adult salmon between the 31 October and the 11 November surveys. Pipe-piezometers were also lost from one redd (number 8) in Riffle R10, two redds (numbers 6 and 8) in Riffle R14, and from two redds (numbers 15 and 22) in Riffle R43 due to redd superimposition between the 11 November and 19 November surveys. In several cases, the pipe-piezometers were found several yards downstream of the original site indicating that the tail spills had been completely re-excavated. A superimposition rate of 24% (5 of 21 redds with pipe-piezometers) is surprisingly high, considering escapement was estimated to be only 168 fish. One possible explanation is that the substrate throughout the spawning reach in the Stanislaus River is cemented and the salmon construct redds in areas loosened by piezometer construction or redd building.

Intragravel Dissolved Oxygen

Fall 1995 Surveys

During the five surveys in November and December 1995, intragravel DO concentrations were relatively constant and suitable for egg incubation (>80% saturation) at most of the 32 piezometer sites (Table 1). The mean difference between the maximum and minimum levels of intragravel DO concentrations at the 32 piezometer sites during the five fall 1995 surveys was 1.3 ppm and none varied by more than a difference of 3.2 ppm. During the five surveys, intragravel DO concentrations were less than 5 ppm (about 50% saturation) at six sites (18.8%) and less than 8 ppm (about 80% saturation) at eleven sites (34.4%).

Mortality of chinook salmon eggs in Mill Creek, California, increased rapidly at oxygen concentrations below 13 ppm, averaging 3.9% at 13 ppm and 37.9% at less than 5 ppm (Gangmark and Bakkala 1960). Davis (1975), who reviewed the oxygen requirements of salmonids, reported a mean threshold of incipient oxygen response for hatching eggs and larval salmonids at 8.1 ppm, which was 76% of saturation. Silver and others (1963) reported that DO concentrations less than 11.7 ppm reduced the growth of chinook salmon embryos. A criterion of 8 ppm for intragravel oxygen concentration was adopted by the Environmental Protection Agency (EPA) and the U.S. Fish and Wildlife Service for the State Water Resources Control Board 1992 Water Rights Hearings for the Mokelumne River.

By the early February 1996 survey, intragravel DO concentrations had declined markedly at several piezometer sites (Table 1) after the runoff from four major storms in January 1996 increased base flows by a daily average of 200 to 500 cfs for several days after each storm. Intragravel DO concentrations at nine piezometer sites, declined to between 25% and 75% of the fall 1995 levels. During the February survey, intragravel DO concentrations were less than 5 ppm at 11 piezometer sites (34.4%) and less than 8 ppm at 16 sites (50%). The large declines were most frequently observed at the bottom piezometers, although the greatest decreases in DO concentration occurred at the top piezometers at Riffles 43 and 99. In addition, most of the piezometers where low DO concentrations occurred during the November and December surveys were located downstream of Riffle R32. However by February 1996, DO concentrations had substantially decreased at Riffles 2 and 10 and there were further declines at the downstream riffles. During the fall 1995 surveys, only one storm resulted in runoff (<80 cfs) between the fourth and fifth surveys and intragravel DO concentrations during the fifth survey were not low compared to the previous surveys at most of the piezometer sites. Gangmark and Bakkala (1960) demonstrated that flooding conditions in Mill Creek, California, were associated with low oxygen concentrations (<5 ppm) in spawning gravel and low intragravel flow rates.

DO concentrations remained high (>8 ppm) throughout all the 1995 surveys and February 1996 survey at all piezometer sites at Riffles TM1, R27, R28, and R32, which include both restoration sites at the Horseshoe Road Recreation Area. Other riffles where DO concentrations at individual piezometer sites remained above 8 ppm during all surveys include the top piezometers at riffles R2, R10, R80, and R99, and the middle or bottom piezometers at riffles R43, R68, R80, R99.

Fall 1996 Surveys

During the October and November 1996 surveys when no appreciable storm runoff occurred, a higher percentage of piezometer sites had intragravel DO concentrations below the EPA standard of at least 8 ppm compared to the percentages observed in November and December 1995. In 1996, eleven (46%) of the piezometer sites had DO concentrations less than 8 ppm, whereas three sites (13%) had concentrations below 5 ppm (Table 2). Low DO concentrations did not occur upstream of Riffle R14 and the lowest levels were usually observed at either the middle or bottom piezometers.

Intragravel DO concentrations at piezometers in the actual redds in riffles R10 and R14 were above 8 ppm during both November 1996 surveys (Table 2). However of the six redds sampled at Riffle R43, three had DO concentrations between 5.6 and 6.9 ppm and one had concentrations below 2 ppm during both surveys.

The distribution of salmon redds and intragravel DO concentrations at Riffles R10, R14, and R43 suggests while salmon typically spawned where intragravel DO concentrations were suitable for incubation (> 8 ppm), high densities of spawning salmon influenced the intragravel DO concentrations in nearby areas. Riffle R14 (Figure 5) provides an example where the fish appeared to avoid the area near the bottom piezometer where intragravel DO concentrations ranged between suboptimal and lethal. Almost all of the redds were constructed upstream of the riffle's crest where the piezometer in the artificial redd indicated that conditions were very suitable. DO concentrations were very similar between the two adjacent piezometers in the middle section of R14, one in an actual redd and the other in an artificial redd, which suggests the artificial redds provided a suitable surrogate for actual redds.

Riffle R10 provides an example where spawning activity slightly improved the intragravel conditions at some nearby piezometers (Figure 6). On 31 October 1996, the intragravel DO concentration at the bottom left piezometer in Riffle R10 was 8 ppm, whereas the concentrations at the other piezometers ranged between 9.5 and 12.1 ppm. By 11 November, after three new redds had been constructed within 80 feet upstream of the bottom left piezometer, intragravel DO concentrations increased to 10 ppm at the bottom left piezometer, whereas the concentrations at the other piezometers remained relatively constant or declined slightly. Intragravel DO concentrations remained high at the bottom left piezometer compared to the other piezometers during the 19 November survey (Table 2).

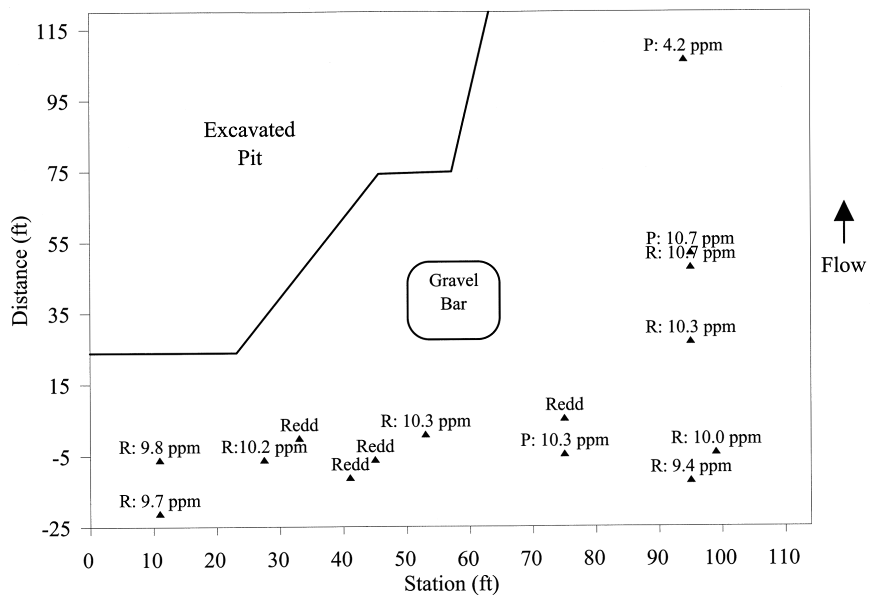

Figure 5 Map of Riffle R14 in the Stanislaus River showing redds (Redd) and the intragravel DO concentrations (ppm) at the piezometers in artificial redds (P) and fall-run chinook salmon redds (R) measured on 11 November 1996

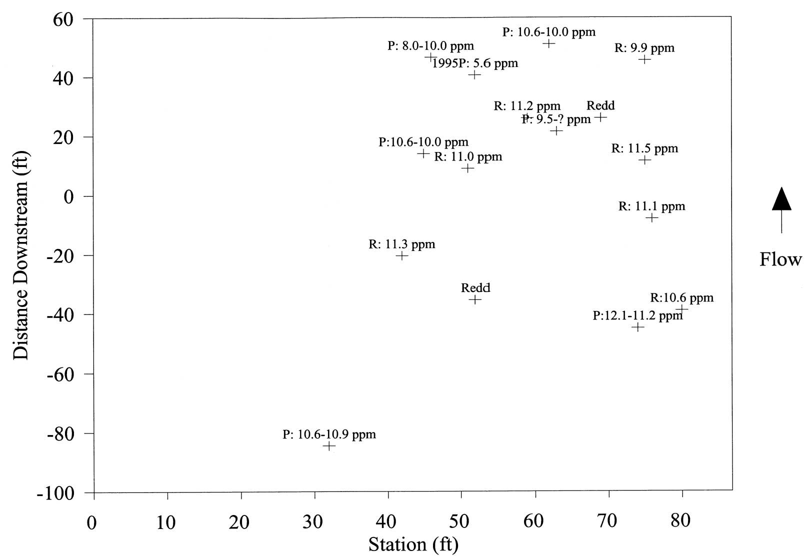

Figure 6 Map of Riffle R10 in the Stanislaus River showing the intragravel DO concentrations (ppm) at the piezometers in artificial redds (P) and fall-run chinook salmon redds (R) in fall 1996. The first DO concentration shown for the piezometers was measured during the 31 October survey and the second was measured during the 11 November survey. The DO concentrations for the redds were recorded during the 11 November survey. The DO concentration at the bottom piezometer installed in fall 1995 (1995P) was measured during the 19 November survey when atmospheric conditions reduced the concentrations by about 13% relative to the 11 November survey.

Riffle R43 provided an atypical example where redds were constructed in suitable areas that later became unsuitable (Figure 7). By 11 November 1996, intragravel DO concentrations below 8 ppm occurred at two salmon redds and one artificial redd located in the top and middle sections near the left streambank. By 19 November, another actual redd and artificial redd developed suboptimal intragravel DO concentrations. The area where the lowest intragravel DO concentrations occurred was adjacent to a grassy field with a large orchard about 100 feet from the water's edge. The left streambank of the bottom section was shielded from the orchard by a dense growth of willows. The ACOE recreation area was on the right bank of the riffle, which was vegetated with willows. The water depth gradually increased from about one foot along the left streambank to about four feet near the right streambank. These features suggest that willow growth may improve the quality or reduce the quantity of groundwater inflow from aquifers. The unusual pattern of intragravel DO concentrations in Riffle R43 also suggests that piezometers in artificial redds provide a suitable measure of conditions in actual redds.

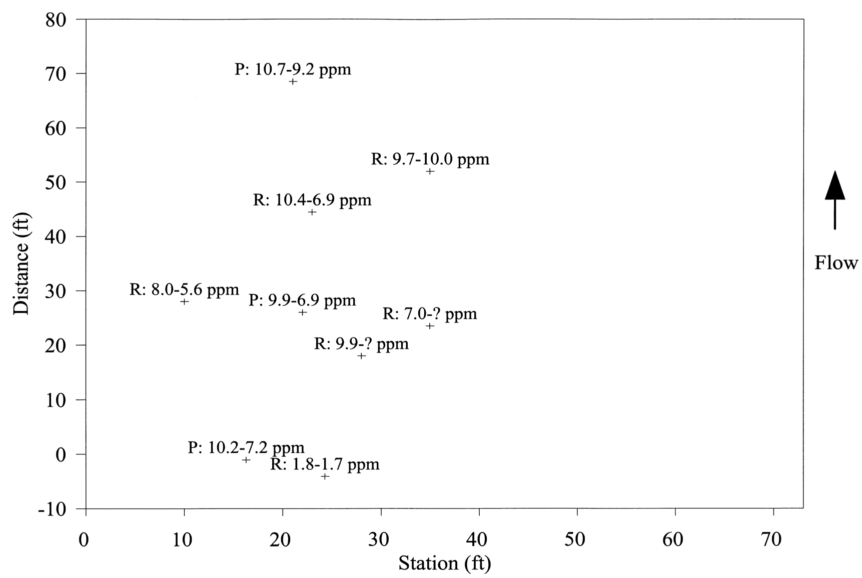

Figure 7 Map of Riffle R43 in the Stanislaus River showing the intragravel DO concentrations (ppm) at the piezometers in artificial redds (P) and fall-run chinook salmon redds (R) in fall 1996. The first DO concentration shown for the piezometers was measured during the 31 October survey and the second was measured during the 19 November survey. The first DO concentration shown for the redds was measured during the 11 November survey and the second was measured during the 19 November survey.

No redds were completed in riffles R29, R58, and R78, where intragravel DO concentrations at most piezometer sites gradually declined to minimally suitable levels (Table 2). One two-foot long salmon was observed constructing a redd in the vicinity of the top piezometer in Riffle R29 on 27 October 1996.

However, this redd and another were abandoned by 1 November when intragravel DO concentrations were measured at 90% and 92% at the top and middle piezometers respectively.

Fall 1997 Surveys

In fall 1997, intragravel DO concentrations and VHGs were measured at two pipe-piezometers installed in a newly deposited gravel berm at the head of Riffle R78. The gravel berm, deposited in a large horseshoe-shape during the spring 1997 flows, had an unexpectedly high proportion of fine substrates and no redds were observed there on 29 October or 3 December 1997. On 3 December, intragravel DO concentrations were 74.8% (8.9 ppm) and 79.8% (9.5 ppm) of saturation and the VHGs were +1.0 and +0.18 at the left and right piezometers, respectively. The high VHG at the left piezometer suggests that the low DO concentrations were caused by a very high rate of groundwater inflow. The presence of several large sewage treatment ponds adjacent to the right bank of this riffle probably contributed to this problem.

Additional evidence that intragravel DO concentrations were affected more by oxygen-poor groundwater from aquifers than by the intrusion of fine sediments was provided by samples collected in fall 1996 at two undisturbed piezometers originally installed in fall 1995. Since the substrate at these sites had been undisturbed since fall 1995, the accumulation of fine sediments and decomposing organic matter would be expected to have been greater in fall 1996 than in fall 1995. As expected, the intragravel DO concentrations at the bottom piezometers at riffles R2 and R10 in fall 1996 were 75.4% and 53.3% respectively, which is about 10% lower than the average concentrations observed in November and December 1995 (Table 2). Presumably this was due to the accumulation of fine sediments and decomposing organic matter. Conversely, the fall 1996 DO concentrations were about 20% higher than those observed in February 1996, when four major rainstorms presumably increased groundwater flow into the gravel (Table 2). Since the storm-influenced increase in groundwater flow would have been temporary, the increase in DO concentrations observed at both sites in November was probably in response to the reduction in groundwater inflow after the January storm effects subsided.

Intragravel Water Temperature

Fall 1995 Surveys

Temperature measurements made by withdrawing water samples from the piezometers with a 50-ml syringe, placing 100 ml of sample into a plastic bottle, and using a mercury thermometer, were inaccurate when compared to data recorded by thermographs. The inaccuracy was caused by the slow collection rate with a syringe and the plastic sample bottle, resulting in an overestimation of surface water temperatures by about 0.5 °F and an overestimation of intragravel water temperatures by about 1.0 to 2.5 °F as measured in fall 1996. Apparently the plastic sample bottle absorbed heat from the air, biologist's hands, thermometer, and the fanny pack used to hold the sample bottle. All readings from both thermographs and mercury thermometers used in the 1995 and 1996 studies were accurate as their measurements were within 0.2 °F when all were simultaneously placed in one gallon of water for ten minutes. The most highly suspect data were collected during 11–13 November 1995, when intragravel temperatures were measured to be 4 to 6 °F higher than surface water temperatures at all piezometer sites. The inaccuracy was probably worst during this survey because air temperatures were relatively high in early November compared to the other surveys and because more heat would have been transferred to the sample bottles by (1) using the large probe of the YSI meter to measure temperatures and (2) holding the plastic sample bottle by its neck. The variability in the overestimation observed in fall 1996 makes it impossible to determine whether intragravel water temperatures were actually elevated at any piezometer site in November 1995.

Temperature data collected during the 20–22 December 1995 survey may have been sufficiently accurate to show the effects of oxygen-poor groundwater inflow from warm aquifers. During this survey, heavy fog and low air temperatures may have reduced the temperature of the measuring equipment (and fingers) sufficiently to minimize heat absorption by the sample. Intragravel temperatures were less than 0.5 °F higher than the surface samples at many piezometer sites (Table 1), although intragravel temperatures were about 2 °F higher than surface temperatures a few of the piezometer sites. At most of the sites with elevated temperatures, intragravel DO levels were relatively low suggesting that relatively warm, oxygen-poor groundwater was influencing these sites.

The effect of groundwater inflow was apparent not only during the 20–22 December 1995 survey, when the first major rainstorm had just ended, but also during the 2–7 February 1996 survey, after five major storms had saturated the floodplain soils. The greatest difference in temperature occurred at Riffle R10 on 4 February about two hours after an intense rain storm had ended (Table 1). At Riffle R10, the intragravel temperature was 3.5 °F higher at the middle piezometer than at either the top or bottom piezometers at the same site. Two samples were taken at the middle site to confirm the difference in temperature on 4 February, whereas the temperature difference was only 0.5 to 1.0 °F higher at the middle site compared to the top and bottom sites at Riffle R10 during the previous surveys. Samples collected at the other riffles during the February survey occurred between one and two days after storm events and the observed temperature differences were lower than those observed at Riffle R10 (Table 1). This suggests substantial effects due to groundwater inflow may occur for only a few days after intense storm events. The differences in water temperature between the intragravel samples and the surface samples during the 20–22 December and 2–7 February surveys were both used as indices of groundwater inflow for the statistical analyses.

The effect of groundwater inflow on incubating salmonid eggs has been reported by other researchers. Curry and others (1994) reported that short-term declines in streamflow releases, which simulated a hydroelectric peaking regime, increased groundwater inflow into brook trout redds approximately 24 hours after the flow had declined. McNeil (1969) and Leitritz and Lewis (1980) reported that groundwater generally has a low DO concentration due to biochemical oxygen demand. This appeared to be true for the Stanislaus River as well, because the index of groundwater inflow based on water temperature differences was negatively correlated with intragravel oxygen concentration. Sowden and Power (1985) suggested the abnormally low egg survival (0.3% to 21.5%) observed for rainbow trout (*Oncorhynchus mykiss*) in a groundwater-fed streambed was partially caused by temporary declines in intragravel DO concentrations to lethal levels (minimums of 3.1 to 4.5 ppm). Unfortunately, they did not monitor groundwater-inflow throughout their study to determine whether pulses in groundwater inflow caused the temporary declines in DO concentration.

Vertical Hydraulic Gradient

The vertical hydraulic gradient (VHG) was atypical at most of the Stanislaus River piezometer sites. In typical rivers described by Lee and Cherry (1978), Creuze des Chatelliers and others (1994), and Dahm and Valett (1996), VHG is negative upstream of obstacles to surface flow, such as the crest of a riffle, and positive in areas downstream of flow obstacles, particularly at the tails of riffles. However, negative VHGs were observed only at the top piezometer of Riffle TM3 and the top right piezometer of Riffle R10 during the 31 October 1996 survey (Table 2); only one measurement was recorded at the top piezometer of Riffle TM3 because the device was displaced by spawning salmon. During the 11 November survey, VHGs were also positive at four new pipe-piezometers installed temporarily within 20 feet upstream of the riffle crest in riffles TM1 and TM3, where the streambed gradient was relatively steep (positive 7% to 9%). Unexpectedly, the middle left piezometer of Riffle R10 became negative on the last survey on 19 November. This site was on just upstream of the mildly upsloped riffle crest (positive 1.4% gradient), an unlikely area for downwelling. Based on these results, there appears to be relatively little downwelling of oxygenated surface water into the Stanislaus' riffles, but instead an upwelling of flow over a majority of the riffle's surface.

Changes in VHG at piezometer sites were unrelated to changes in intragravel DO concentrations, patterns in intragravel water temperatures, storm runoff, or precipitation (Table 2). Variability in VHGs has been known to result from changes in streambed permeability and morphology (Lee and Cherry 1978), streamflow (Lee and Cherry 1978), the depth of the water table (Price 1996), and atmospheric pressure in confined aquifers (Price 1996). Perhaps the effects of ongoing redd construction on streambed permeability and morphology that occurred throughout this study, and changes in groundwater flow from aquifers caused some of the variability in VHG measurements in the Stanislaus River.

Nitrate Concentration

Intragravel nitrate concentrations were usually higher than the surface water concentrations downstream of Orange Blossom Bridge (riffles R43 to R99) during both the 11–12 December and 20–22 December surveys 1995 (Table 2). The highest intragravel concentrations occurred at the top piezometers at Riffles R68 and R80 and the bottom piezometer at Riffle R99 during both of these surveys.

A few hours after an intensive rain storm had ceased, surface concentrations of nitrates were relatively high (1.0 ppm) at Riffle R68 and the downstream riffles on 11 December 1995 compared to Riffle R43 (the last site sampled on 11 December) and most of the riffles sampled on 12 December. The surface concentration was also high (1.0 ppm) at Riffle R47 compared to all the other riffles (0.25 ppm) during the 20–22 December survey. These results suggest nitrogenous compounds were entering the river primarily between the Orange Blossom Bridge and the town of Oakdale.

Two indices of nitrate concentration were used for the statistical analyses in fall 1995: the intragravel concentration and the difference in concentration between the intragravel sample and the surface sample during the 11-12 December 1995 survey (Table 2). The difference between the intragravel and surface concentrations was used to evaluate the relationship between groundwater and nitrate concentrations.

The high intragravel nitrate concentrations occurred further upstream in fall 1996 than during fall 1995. Intragravel nitrate concentrations in fall 1996 at riffles R29, R43, R58, and R78 were at least double (0.8 to 1.0 mg/L) the concentration of the surface water sample collected at Riffle R29 (0.4 mg/L) or the intragravel samples collected at the upstream riffle sites (0.5 to 0.6 mg/L).

Percent Fines, Crushed Rock, and Asian Clams in The Substrate in Fall 1995

The percent fines less than 2 mm in diameter for the entire substrate sample collected averaged 13% for the 32 piezometer sites and ranged between 0.13%

at the bottom piezometer site at Riffle R27 and 33.4% at the top piezometer site at Riffle R92 in fall 1995 (Table 1). The percent fines less than 1 mm for the entire substrate sample was strongly correlated (r = 0.95, P = 0.0000) with those less than 2 mm.

Predicted survival probabilities for chinook salmon eggs using Tappel and Bjornn's (1983) laboratory study averaged 76.5% in the reach above the Orange Blossom Bridge, 58.6% in the lower spawning reach between the bridge and Riverbank, and 95.4% at two restoration sites near the U.S. Army Corps of Engineers' Horseshoe Road park where gravel was added in 1994 (Table 1). Predicted egg survival probabilities averaged 72.3% upstream of riffle crests and 62.1% downstream of riffle crests at four natural riffles, Riffles R2, R10, R32, and R68, which had pronounced crests.

The total weight of substrate samples averaged 8,270 g and ranged between 4,486 g for the top piezometer at Riffle R2 and 13,483 g for the top piezometer at Riffle R32 (Table 1). There was a Pearson correlation coefficient (r) of 0.58 with a probability level of 0.0005 between sample weight and the percent fines less than 2 mm, which suggests that larger samples included deeper substrates, which contained a relatively high percentage of fines. However, including the sample weight in the multiple regression analysis had no significant effect on the final model.

The proportion of angular rock in the substrate samples was three to four times higher at the Horseshoe Road mitigation sites, Riffles R27 and R28, compared to the natural riffles. Approximately 60% of the rocks between 16 and 64 mm at Riffles R27 and R28 had sharp edges, indicating that they had been recently broken, whereas usually less than 20% of the rocks had sharp edges at the natural riffles. The amount of crushed rock at the Goodwin Canyon sites appeared to be similar to the substrate at the Horseshoe Road sites based on a casual comparison.

Some of the substrate samples at the piezometer sites at Riffles R47, R68, and R92 had high densities of Asian clams that ranged between 2 and 32 mm in diameter. At the top piezometer site at Riffle R92, there were 71 clams between 16 and 32 mm, 283 clams between 8 and 16 mm, and 15 clams between 4 and 8 mm per square-foot of streambed. Other sites, such at the top piezometer at Riffle R28 had high numbers of small clams: 306 between 2 and 4 mm and 868 between 4 and 8 mm per square-foot. Most of the clams in the samples had died prior to collection as evidenced by a strong putrid smell during sample collection. The clams, which normally live near the surface of the streambed, probably died as a result of becoming buried well below the surface during the installation of the piezometers. It is also likely that clams would be buried and die as a result of salmon building their redds. The total weight of the dried clam shells was used as an index of their biomass for statistical analyses.

Regression Analyses

Differences in DO concentration among the piezometer sites in fall 1995 were highly correlated with environmental factors. Based on the final multiple regression model, 80.4% (adj-R^2) of the variation in the mean DO concentrations during the November and December 1995 surveys was explained by groundwater effects, the weight of Asian clams, the percent fines less than 2 mm in diameter, and the mean column velocity at the piezometer site. The index for groundwater inflow, which was the difference in water temperature between the intragravel and surface samples during the 20–22 December survey, was the most important variable in the final model; it was negatively correlated with DO concentrations (t-statistic = –3.65, P = 0.001). Only slightly less important was the weight of Asian clam shells, which was also negatively correlated with DO concentrations (t-statistic = –3.41, P = 0.002). The percent fines less than 2 mm, as computed from the contents of the entire substrate sample recommended by Chapman (1988), was also negatively correlated with DO concentrations (t-statistic = -2.80, P = 0.010), although not as strongly as for groundwater inflow and clams. Although percent fines less than 2 mm based on the entire substrate sample was selected as the variable for the model, all four of the percent fines indexes were highly correlated (r) with DO concentrations (Table 3). Mean column water velocity was the least important variable in the final model and it was positively correlated with DO concentrations (t-statistic = 2.67, P = 0.014).

The final multiple regression model of DO concentrations during the February 1996 survey indicated that 68.3% (adj-R^2) of the variation was explained by groundwater effects and the percent fines less than 2 mm in diameter. Both of these variables were equally important to the model as they had similar t-statistics of about –4.1.

Although the gradient indexes were not selected for the regression models, low DO concentrations in February 1996 usually occurred where the streambed gradient immediately upstream of the piezometers ranged between –5% and 2% (Figure 8). This suggests that positive gradient, such as occurs at the tails of pools and at the heads of some riffles, minimized the occurrence of low DO concentrations to a greater degree than where streambed had the same gradient, but with a negative slope. This would explain why 77% of all redds observed throughout the spawning reach were located within the tails of pools and heads of riffles during both the 1994 and 1995 surveys. This skewed relationship also made it impossible to correctly evaluate the gradient indexes with linear regression analyses.

One exception occurred at the bottom piezometer at Riffle R92, where the gradient within 5 feet upstream of the piezometer was positive 3.4% and the DO concentration was low (4.5 ppm, 46% of saturation) during the February 1996 survey (Figure 8). This low DO concentration may be explained by an unusually high percentage of fines (<0.85 mm) and small gravel (<9.5 mm) which corresponds to a low gravel permeability and egg survival rate (Tappel and Bjornn 1983). High percentages of fines also occurred at the bottom piezometer of Riffle R10 and the top piezometer of Riffle R92, where DO concentrations were also low (<5 ppm) in February 1996.

Table 3 Dissolved oxygen concentrations (percent saturation), vertical hydraulic gradient, and stream gradient at 5-ft, 10-ft, and 20-ft distances upstream of piezometers at 29 piezometers in artificial redds (e.g., Top, Mid, Bot) and at 21 actual redds (e.g., Redd 20) at nine riffles in the Stanislaus River between 31 October and 19 November 1996

	D. O. concentration (percent saturation)			*Vertical hydraulic gradient*			*Streambed gradient*		
Piezometer Site	*10/31/96*	*11/11/96*	*11/19/96*	*10/31/96*	*11/11/96*	*11/19/96*	*5 ft*	*10 ft*	*20 ft*
TM1-BOTR	--	100.8	--	--	0.14	--	5.0	7.6	5.4
TM1-BOTL	--	99.2	--	--	0.14	--	5.0	7.6	5.4
TM3-TOP	95.0	95.0	97.2	-0.12	0.16	0.33	9.0	9.0	3.4
TM3-MID	96.7	96.7	92.6	0.33	0.20	0.13	5.0	10.0	0.0
TM3-BOT	93.3	92.5	92.6	0.40	0.41	0.32	2.8	0.1	-0.2
R2-TOP	--	100.0	--	--	0.27	--	9.1	8.7	5.0
R2-BOT	--	75.4	--	--	0.24	--	-5.8	-5.2	-2.7
R10-TOPR	97.6	93.3	90.5	-0.10	0.06	0.07	12.0	6.9	4.5
R10-TOPL	85.5	90.8	89.5	0.12	0.16	0.11	4.4	2.6	4.8
R10-MIDR	76.6	--	--	0.21	--	--	5.4	0.7	1.5
R10-MIDL	85.5	83.3	92.4	0.18	0.13	-0.17	-4.6	0.4	1.4
R10-BOTR	85.5	83.3	77.1	0.11	0.21	0.10	-1.8	-4.8	-2.5
R10-BOTL	64.5	83.3	88.6	0.13	0.11	0.16	-3.6	-2.8	-2.0
R10-BOT1995	--	--			53.3	0.05	-1.4	-3.3	-5.4
R10-REDD 20	--	88.3	85.7	--	0.05	0.04	12.6	6.8	7.2
R10-REDD 12	--	94.2	84.8	--	0.12	0.15	11.4	8.5	3.2
R10-REDD 14	--	82.5	85.7	--	0.27	0.23	12.0	9.6	2.5
R10-REDD 11	--	93.3	93.3	--	0.37	0.23	11.8	7.8	3.9
R10-REDD 10	--	95.8	85.7	--	0.25	0.13	1.2	1.1	0.3
R10-REDD 2	--	91.7	83.8	--	0.25	0.12	8.8	5.2	6.5

Table 3 Dissolved oxygen concentrations (percent saturation), vertical hydraulic gradient, and stream gradient at 5-ft, 10-ft, and 20-ft distances upstream of piezometers at 29 piezometers in artificial redds (e.g., Top, Mid, Bot) and at 21 actual redds (e.g., Redd 20) at nine riffles in the Stanislaus River between 31 October and 19 November 1996 (Continued)

	D. O. concentration (percent saturation)			*Vertical hydraulic gradient*			*Streambed gradient*		
Piezometer Site	*10/31/96*	*11/11/96*	*11/19/96*	*10/31/96*	*11/11/96*	*11/19/96*	*5 ft*	*10 ft*	*20 ft*
R10-REDD 9	--	92.5	--	--	0.26	---	9.6	6.5	3.3
R14-TOP	98.3	96.3	100.0	0.09	0.11	0.07	3.2	1.1	1.5
R14-MID	100.0	100.0	97.1	0.25	0.05	0.09	2.4	-0.8	-0.4
R14-BOT	69.5	39.3	37.9	0.07	0.09	0.04	-1.2	-1.0	-0.8
R14-REDD 12	--	90.7	90.3	--	0.08	0.08	5.8	5.5	7.2
R14-REDD 6	--	95.3	--	--	0.16	--	12.6	4.4	8.5
R14-REDD 1	--	96.3	99.0	--	0.11	0.05	10.8	6.7	4.4
R14-REDD 5	--	93.5	97.1	--	0.04	0.05	15.8	11.2	4.4
R14-REDD 10	--	91.6	91.3	--	0.19	0.23	6.2	3.6	4.6
R14-REDD 4	--	87.9	88.3	--	0.05	0.04	6.6	2.0	-0.4
R14-REDD 8	--	96.3	--	--	0.18	--	13.2	3.9	2.0
R14-REDD 17	--	100.0	97.1	--	0.23	0.10	0.0	-0.8	-0.4
R29-TOP	90.1	80.7	77.5	0.12	0.18	0.21	2.0	3.7	3.7
R29-MID	91.9	81.7	80.4	0.17	0.14	0.10	-1.8	-3.1	-0.7
R29-BOT	55.0	86.2	45.1	0.17	0.12	0.13	-2.0	-2.0	-1.1
R43-TOP	87.9	88.2	72.0	0.11	0.15	0.14	6.6	3.0	1.7
R43-MID	85.3	38.2	69.0	0.20	0.18	0.13	0.4	-1.1	0.1
R43-BOT	92.2	88.2	92.0	0.20	0.38	0.30	-4.2	-3.3	-5.8
R43-REDD 16	--	72.7	56.0	--	0.19	0.16	7.4	1.2	0.0
R43-REDD 22	--	63.6	102.0	--	0.14	0.22	17.4	7.9	5.0
R43-REDD 15	--	90.0	104.0	--	0.18	0.30	8.2	6.3	2.1
R43-REDD 18	--	94.5	69.0	--	0.29	0.33	9.0	7.6	2.7
R43-REDD 13	--	88.2	100.0	--	0.17	0.15	19.4	2.1	0.0
R43-REDD 30	--	16.4	17.0	--	0.13	0.11	5.4	2.8	2.1
R58-TOP	77.0	71.4	76.0	0.17	0.19	0.13	7.2	4.9	6.7
R58-MID	83.2	85.7	82.0	0.30	0.20	0.18	-1.8	-1.4	-0.6
R58-BOT	77.0	64.8	69.0	0.18	0.11	0.15	2.0	0.0	-0.8
R78-TOP	85.3	88.5	85.4	0.27	0.24	0.13	3.8	2.3	0.8
R78-MID	94.8	94.2	84.4	0.10	0.07	0.11	-4.0	-2.5	-1.5
R78-BOT	65.5	61.5	71.9	0.10	0.11	0.07	-1.6	-1.1	-0.8

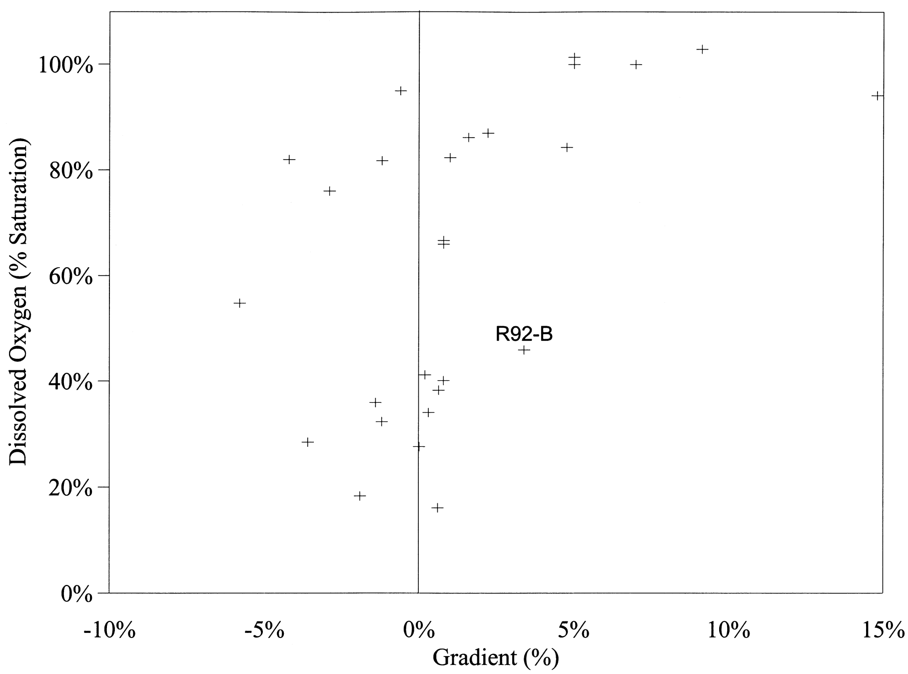

Figure 8 The relationship between intragravel DO concentrations measured at piezometer sites during the February 1996 survey and the gradient of the streambed within five feet upstream of the piezometer sites in the Stanislaus River study riffles TM1, R2, R10, R32, R43, R47, R68, R80, R92, and R99, and the bottom piezometer in Riffle R28

The most obvious explanation as to why high DO concentrations occur and salmon mostly spawn where streambed gradients are high and positive is that these areas are where downwelling of surface water typically occurs and downwelling would dilute the adverse effects of groundwater inflow and decaying clams. However, measurements of VHG upstream of riffle crests where the undisturbed streambed gradient at riffles TM1, TM3, R2, R14, R29, and R43 exceeded a positive 2% in fall 1996 indicated that upwelling occurred at these sites instead of downwelling (Table 2). Pipe-piezometers were also installed at two sites at Riffles R27 and R78 on 3 December 1997, and the VHG at these sites were also positive. Perhaps conditions, including the percentage of fines and groundwater inflow, are so extreme in the Stanislaus River that downwelling occurs at either a few, small locations or at a distance greater than 20 feet upstream from the riffle's crest.

Conclusions

Due to low intragravel DO concentrations and high concentrations of fine sediments, a majority of the riffle area in the Stanislaus River, particularly downstream of the Orange Blossom Bridge, was marginally suitable for spawning and incubation habitat for fall-run chinook salmon in 1995 and 1996. Survival of chinook salmon eggs would be expected to average 76.5% at the natural riffles at the Orange Blossom Bridge (Riffles TM1 to R43) and upstream, but only 58.6% at the riffles between the Orange Blossom Bridge and Riverbank (Riffles R47 to R99) based on a model from laboratory tests developed by Tappel and Bjornn (1983). These results are high in comparison to emergence trap studies on the Tuolumne River that indicate that egg-survival-to-emergence rates ranged between 0% and 68% (mean of 34%; EA Engineering, Science, and Technology 1992). Survival rates for Tappel and Bjornn's (1983) laboratory study may have been abnormally high for two reasons. First intragravel DO levels remained near saturation during all tests and so alevins would have been larger and stronger than those incubating in natural gravels where DO levels are lower. Second, embryos were incubated in a hatchery for 52 days before they were planted in test gravel mixtures to minimize handling mortality. Again this would have produced larger and stronger embryos compared to those incubated in natural gravels. Further research is recommended to accurately determine the relationship between gravel size and chinook salmon egg survival under natural conditions of intragravel flow and DO.

Intragravel DO concentrations were below EPA standards (80% of saturation) at 35% of piezometer sites in fall 1995 and at 42% of the sites in fall 1996. Intragravel DO concentrations declined to below EPA standards at 58% of the piezometer sites immediately following a February 1996 series of intense rain storms that made the river very turbid. Low intragravel DO levels were probably caused by the combined influence of groundwater inflow and fine sediment intrusion. If groundwater inflow increased as a result of the rain storms, the reduced intragravel DO levels associated with the February 1996 rain storms were probably temporary. On the other hand, it is also likely that the intrusion of fine sediments reduced gravel permeability, which reduced downwelling of surface flows. Regardless of the cause, intrusion of fine sediments and increased groundwater inflow did not reduce the DO levels at the 1994 restoration sites, which remained near saturation.

The predominance of nearly flat, silty riffles and fine sediment intrusion during rain storms when eggs are incubating may limit the production of chinook salmon in the Stanislaus River. A stock-recruitment analysis indicates that between 1945 and 1995, the number of spawners up to about 2,500 fish was directly correlated with the number of fish from their brood that returned to spawn in the Stanislaus River as adult fish (CMC 1996). However, once the number of spawners exceeded 2,500 fish, there was no corresponding increase in the number of returning fish. These results suggest that the Stanislaus River has enough suitable spawning habitat for only about 1,250 pairs of adult chinook salmon.

The restoration sites where gravel was added in 1994 and 1997 provided suitable incubation habitat, but only the 1997 sites were immediately used by spawners. It is possible that the source of the gravel used for restoration affected the salmon's use of spawning sites. Crushed gravel from the Merced River, 0.5 to 4 inches in diameter, that was placed at two sites in 1994 was not used by the spawners for three years until high flows washed natural Stanislaus River gravel into the site. In contrast, gravel obtained near the Stanislaus River (0.5 to 5 inches in diameter) and placed in Goodwin Canyon in 1997 was used immediately by many salmon. Although the source and type of rock used in the 1997 Goodwin Canyon project may have been more suitable for the spawners, the salmon may have used the sites because gravel is relatively scarce in Goodwin Canyon. Additional studies are needed to determine whether the salmon did not use the 1994 sites because the rock was imported from the Merced River, the rock was crushed, or if the gravel's size distribution was unnatural.

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Notes

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Distribution and Abundance of Chinook Salmon and Resident Fishes of the Lower Tuolumne River, California

Tim Ford and Larry R. Brown

Abstract

The Tuolumne River chinook salmon (*Oncorhynchus tshawytscha*) population represents one of the southernmost populations of the species and is of considerable management interest. This paper compiles and analyzes data available through 1997 for chinook salmon and other fish species occurring in the lower Tuolumne River. Estimates of adult fall-run chinook salmon varied from about 100 to 130,000 from 1940 to 1997 (mean: 18,300; median: 7,100). Age composition varied widely from 1981 to 1997; however, three-year-old fish usually dominated the population. The percentage of females in the population varied from 25% to 67% during 1971–1997. The percentage of tagged adult salmon increased from less than 2% before 1987 to an average of 20% during 1992–1997. Density of juvenile chinook salmon generally declined each year after a winter peak in fry abundance. Average, minimum, and maximum fork length of juvenile chinook salmon typically increased after February; although, declines occurred in some years because of large captures of fry in late spring. Few juvenile chinook salmon resided in the river over the summer during 1988–1993. A total of 33 taxa of fish (12 native and 21 introduced), including chinook salmon, was captured during various sampling programs. Native species were most frequent in upstream areas above river kilometer (rkm) 80. Introduced species dominated areas downstream of rkm 50. The resident fish community appeared to vary in response to annual differences in flow conditions with native species becoming more abundant in the year following a high flow year. There was no discernible seasonal change in fish communities when early summer (early June) and late summer (mid September) samples from the same sites were compared. Monitoring of the Tuolumne River chinook salmon population has provided valuable data on both chinook salmon and populations of other fish species.

Introduction

The chinook salmon (*Oncorhynchus tshawytscha*) populations in the tributaries to the San Joaquin River, including the Tuolumne River, constitute the southernmost extant populations of the species (Moyle 1976). The San Joaquin River tributary populations of fall-run chinook salmon, along with other Central Valley fall-run populations are presently considered candidate species under the federal Endangered Species Act (NMFS 1999). Even before candidate status, San Joaquin fall-run chinook salmon were of great management concern and were managed as a distinct stock. Historic declines in San Joaquin fall-run chinook salmon numbers and current threats to their survival have been attributed to a number of factors including habitat loss, habitat suitability, survival of emigrants, harvest, genetic effects of hatcheries, and water quality (USFWS 1995).

The earliest estimates of fall-run chinook salmon spawning escapement in the Tuolumne River date from 1940, with more detailed information collected since 1981. Since 1973, several other types of studies have been conducted within the lower 84 km of the Tuolumne River (from La Grange Dam to the confluence with the San Joaquin River) available for salmon spawning. Most of these studies have focused on winter-spring sampling when juvenile fall-run chinook salmon are abundant; but biologists have also gathered considerable data on the distribution and abundance of other fish species. A few studies have focused primarily on the resident fishes. The purpose of this paper is to compile and analyze data available through 1997 for chinook salmon and other fish species occurring in the lower Tuolumne River. The salmon data are clearly important to the proper management of Tuolumne River fall-run chinook salmon. Data on the other species can be used to develop understanding of interactions between salmon and other species, environmental conditions when salmon are not present, and the biology of species that become of management concern, such as splittail (*Pogonichthys macrolepidotus*, federally listed as threatened) and hardhead (*Mylopharodon conocephalus*, a California species of special concern) (Moyle and others 1995).

Methods

Adult Fall-run Chinook Salmon

Chinook salmon spawning runs in the Tuolumne River have been monitored to some degree since 1940, with estimates of adult escapement available for all years since 1951. Counts of migrating adult salmon were made at a weir in Modesto at river kilometer (rkm) 25.9 by the California Department of Fish and Game (DFG) in 1940, 1941 (partial count), 1942, and 1944, and by the U.S. Fish and Wildlife Service (USFWS) in 1946 (Fry 1961). DFG conducted carcass surveys for estimating escapements after 1946 (Fry 1961; Fry and Petrovich 1970), except that no estimate was made in 1950 due to an early flood. The results of spawning surveys since 1971 are described in a series of reports submitted by Turlock and Modesto irrigation districts (TID and MID) as part of the Federal Energy Regulation Commission (FERC) license process (EA 1991, 1997; TID and MID 1998). Tagging of some carcasses to obtain information on carcass recovery rates began in 1967, and since 1979, the DFG estimates are based on variations of Peterson or Schaefer mark-recapture formulas.

Carcass surveys were generally conducted in the reach of the Tuolumne River from La Grange at rkm 81.6 downstream to rkm 54.6 (Reed Rock Plant or Nielsen Ranch) just upstream of Waterford (Figure 1). Within this reach, data were segregated into three smaller sections that have varied over time. Since 1981 these sections have been divided at Basso Bridge (rkm 76.4) and Turlock Lake State Recreation Area (rkm 67.4). In some years, additional reaches were surveyed, including an upstream reach from rkm 81.6 to rkm 83.1 near La Grange Powerhouse and/or a downstream reach from rkm 54.6 to rkm 42.0 near Geer Road. Since 1981, population estimates for river sections not included in weekly carcass surveys were usually estimated by counting the number of redds in the section and then multiplying by the number of salmon per redd observed in the surveyed sections.

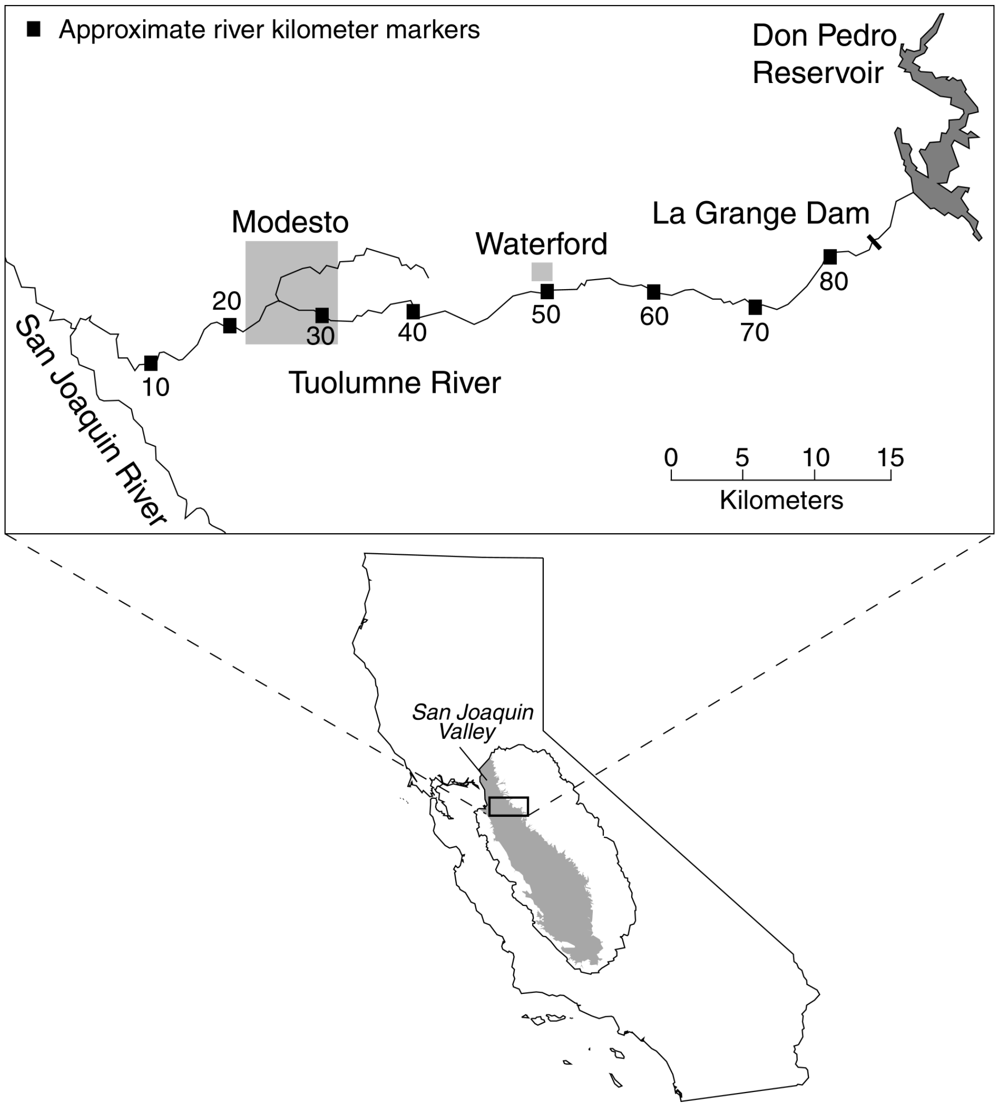


Figure 1 Location map for the lower Tuolumne River, California

DFG conducted weekly carcass surveys, generally by boat, using two- or three-person crews. Salmon carcasses were recovered by gaff for tagging and examination. Carcass mark-recapture sampling was conducted by attaching a marker to the upper jaw of some of the carcasses with a metal hog ring. Tagged carcasses were released in moving water for recovery during subsequent surveys. All other carcasses, including those marked with tags from earlier releases, were counted and chopped by machete to avoid double counts. Before 1988, only fresh carcasses were used for tagging and recovery. Beginning in 1988, both fresh (indicated by clear eyes) and non-fresh carcasses were tagged, with a distinction made between "adult salmon" and "grilse" (two-year olds). Carcasses under 60 cm fork length (FL) (considered grilse) were not tagged. From 1989 to 1991, fresh grilse carcasses were also tagged, but non-fresh grilse were not. Beginning in 1992, all carcasses were tagged. Fork length, sex, and condition (fresh or non-fresh) of measured carcasses were recorded. Snouts of carcasses possibly having coded-wire tags (CWT), externally indicated by a missing adipose fin, were saved for tag recovery. Redd counts for individual riffles and live salmon counts for the survey reach were also recorded. The annual survey periods are shown in Table 1.

Initial run timing was based on the first report, by TID and MID staff, of adult salmon near La Grange. Age composition of the run was estimated from visual examination of length frequency histograms for each sex. A spawning use index was calculated from redd counts in carcass survey sections using the following formula.

$$\text{Spawning use index} = \frac{\text{\% of total redds in a survey section}}{\text{\% of total stream length surveyed in that section}}$$

Juvenile Fall-run Chinook Salmon and Other Species

Winter-spring Seining Surveys

Winter-spring seining surveys for juvenile salmon were conducted annually by TID and MID in 1986–1997 (EA 1991, 1996; TID and MID 1998). The sampling interval and number of locations and sample periods varied depending on the year. These studies also documented the distribution and abundance of other fish species and represent the longest continuous juvenile salmon monitoring effort in the San Joaquin River system, upstream of the Sacramento-San Joaquin Delta.

The locations sampled each year are shown in Table 2. Seining was conducted with 1.2 to 1.8 m high, 3.2 mm mesh nylon seine nets in lengths of 6.1, 9.1, or 15.2 m. The same general areas were sampled during each visit during a given year to facilitate comparative analysis throughout the sampling period. Sample areas varied somewhat as a result of changes in flow. Seine hauls were generally made in the direction of the current and parallel to shore, although offshore-to-onshore hauls were sometimes used. In general, three hauls were made during each visit to a site. The three hauls sampled an area of approximately 140 to 186 m^2.

Table 1 Salmon survey periods, peak live counts, and arrival dates[a]

Year	Survey dates		Peak live count		Population estimate (x 1,000)	Peak live percentage (%)	Date fish first observed at La Grange
	Start	End	Date	No.			
1940	26 Sep	02 Dec	04 Nov	5,447	122.0	4.5	---
1941	21 Sep	18 Nov	13 Nov	2,807	27.0	10.4	---
1942	13 Sep	30 Nov	01 Nov	3,386	44.0	7.7	---
1944	30 Sep	30 Nov	06 Nov	10,039	130.0	7.7	---
1946	11 Oct	20 Nov	04 Nov	6,002	61.0	9.8	---
		--	**No data available from 1947 to 1956**			**--**	
1957	05 Nov	03 Jan	---	---	8.0	---	--
1958	06 Nov	09 Jan	---	---	32.0	---	---
1959	03 Nov	01 Jan	---	---	46.0	---	---
1960	12 Nov	13 Jan	---	---	45.0	---	---
1961	---	---	---	---	0.5	---	---
1962	08 Nov	04 Jan	---	---	0.2	---	---
1963	10 Feb	---	---	---	0.1	---	---
1964	04 Nov	18 Dec	---	---	2.1	---	---
1965	19 Nov	12 Jan	---	---	3.2	---	---
1966	08 Nov	18 Jan	09 Nov	271	5.1	5.3	---
1967	18 Oct	13 Jan	21 Nov	184	6.8	2.7	---
1968	11 Nov	15 Dec	22 Nov	1,490	8.6	17.3	---
1969	20 Nov	12 Jan	---	---	32.2	---	---
1970	19 Nov	20 Jan	20 Nov	1,517	18.4	8.2	---
1971	15 Nov	27 Dec	16 Nov		21.9	9.7	---
1972	13 Nov	23 Jan	27 Nov	349	5.1	6.8	---
1973	05 Nov	17 Jan	---	---	2.0	---	---
1974	---	---	---	---	1.2	---	---
1975	06 Nov	31 Dec	06 Nov	154	1.6	9.6	---

[a] Data for 1940–1946 are from Modesto; all later count data are from weekly carcass surveys in the spawning reach. Dashes (--) indicate no data. Population estimates are subject to revision.

Table 1 Salmon survey periods, peak live counts, and arrival dates[a] (Continued)

Year	Survey dates		Peak live count		Population estimate (x 1,000)	Peak live percentage (%)	Date fish first observed at La Grange
	Start	End	Date	No.			
1976	03 Nov	29 Dec	15 Nov	241	1.7	14.2	---
1977	29 Nov	20 Dec	---	---	0.5	---	---
1978	26 Oct	19 Dec	24 Nov	81	1.3	6.2	---
1979	05 Nov	17 Dec	02 Nov	153	1.2	12.8	---
1980	12 Nov	18 Dec	12 Nov	112	0.6	18.7	---
1981	04 Nov	16 Dec	---	---	14.3	---	14 Oct
1982	08 Nov	29 Nov	15 Nov	545	7.1	7.7	29 Sep
1983	07 Nov	01 Dec	15 Nov	263	14.8	1.8	13 Oct
1984	01 Nov	30 Nov	01 Nov	1,084	13.8	7.9	04 Oct
1985	29 Oct	20 Dec	12 Nov	2,986	40.3	7.4	24 Sep
1986	27 Oct	05 Dec	03 Nov	1,123	7.3	15.4	10 Sep
1987	28 Oct	16 Dec	17 Nov	2,155	14.8	14.6	06 Oct
1988	25 Oct	29 Dec	14 Nov	1,066	6.3	16.9	17 Oct
1989	24 Oct	29 Dec	09 Nov	291	1.3	22.8	15 Oct
1990	23 Oct	26 Dec	19 Nov	44	0.1	45.8	24 Oct
1991	22 Oct	02 Jan	25 Nov	24	0.1	45.3	06 Nov
1992	05 Nov	21 Dec	19 Nov	49	0.1	38.3	31 Oct
1993	14 Oct	18 Dec	06 Nov	94	0.4	24.2	26 Sep
1994	03 Nov	05 Jan	21 Nov	226	0.5	45.2	26 Oct
1995	27 Oct	30 Dec	03 Nov	270	1.0	27.0	05 Oct
1996	22 Oct	04 Dec	31 Oct	636	3.3	19.3	--
1997	14 Oct	23 Dec	12 Dec	1258	7.2	17.5	09 Oct
1971–1997 cumulative data							
First	14 Oct	29 Nov	31 Oct	---	---	---	10 Sep
Last	29 Nov	23 Jan	27 Nov	---	---	---	06 Nov
Median	02 Nov	20 Dec	11 Nov	---	---	---	11 Oct

[a] Data for 1940–1946 are from Modesto; all later count data are from weekly carcass surveys in the spawning reach. Dashes (--) indicate no data. Population estimates are subject to revision.

Table 2 Primary winter-spring seining locations for each year of sampling[a]

Location	*River kilometer*	*Year*											
		86	*87*	*88*	*89*	*90*	*91*	*92*	*93*	*94*	*95*	*96*	*97*
Old La Grange Bridge	81.3	X	X	X	X	X	X	X	X	--	--	X	X
Riffle 4B	77.9	X	X	X	X	X	X	--	--	--	X	X	X
Riffle 5	77.1	--	X	X	X	X	X	X	X	X	--	--	--
Tuolumne River Resort	68.2	--	--	X	X	X	X	X	X	X	X	X	X
Turlock Lake State Rec. Area	67.6	X	X	--	--	--	--	--	--	--	--	--	--
Reed Gravel	54.7	X	X	X	X	X	X	--	--	--	--	--	--
Hickman Bridge	50.8	X	X	X	X	X	X	X	X	X	X	X	X
Charles Road	40.1	--	X	X	X	X	X	X	X	--	--	--	X
Legion Park	27.7	X	X	X	X	X	X	X	X	X	X	X	X
Riverdale Park	19.8	--	X	X	X	X	X	--	--	--	--	--	--
McCleskey Ranch	9.7	X	X	X	X	X	X	X	X	X	--	--	--
Shiloh Bridge	5.8	X	X	X	X	X	X	--	X	--	X	X	X
Total locations		8	11	11	11	11	11	7	8	5	5	6	7
Mean interval (days)		9	7	10	11	10	21	28	18	19	19	21	15
Number of sample periods		18	21	14	13	14	8	5	7	7	8	8	10

[a] Mean interval is the mean number of days between samples. Dashes (--) indicate location not sampled.

Captured salmon were anesthetized, measured (FL in mm), and then revived before being released. If more than 100 salmon were caught, a random subsample of approximately 100 salmon was measured and the remaining salmon were counted and released. The number of fish caught, and fork lengths were recorded. Other fish species were counted and recorded separately.

Minimum, maximum, and average fork length of juvenile chinook salmon were plotted for each year and sample period. Density was calculated as the number of salmon captured per square meter of area seined. Seining data were stratified by river section and summed for the entire river. Three river sections were used for comparison: upper section (La Grange Powerhouse, rkm 83.7 to rkm 59.5), middle section, and lower section (Dry Creek, rkm 26.4 to mouth, rkm 0).

All fish species other than chinook salmon were included in the analyses of resident fish species. Total catch was summarized as species percentage abundance for all fish captured in all samples. Seining data were used for three types of analyses: frequency of occurrence of species at specific sites along the river, number of species captured per sampling effort, and resident fish assemblage structure.

Frequency of occurrence was determined on the basis of the number of samples collected at a site, from 1986 to 1997. Frequency of occurrence was the percentage of the total number of samples that included a particular species. The total number of samples at a site varied from 33 to 129. For each sample at each site, the number of species captured other than chinook salmon was determined. A mean value and standard deviation was then calculated for each site based on all samples from all years of sampling.

Assemblage structure of the resident fishes was described using detrended correspondence analysis (DCA). DCA is a multivariate ordination technique based on reciprocal averaging that results in an ordination of species based on occurrence at sites and an ordination of sites based on the species assemblage at each site. DCA was conducted with species percentage data. Only sites sampled consistently through the study were included. Review of the data resulted in selection of eight sites for analysis. These sites were sampled consistently from 1987 to 1993 and then more sporadically through 1997. Because of the low number of species captured per sample, all fish captured at a site each year were combined into a single sample and then the percentage of each species in the combined sample calculated. Years when a site was sampled fewer than four times were excluded from analysis. Species were only included in the analysis if they were present in at least 10% of the samples and comprised at least 5% of the fish captured in at least one sample. One-way analysis of variance (ANOVA) was used to test for annual differences in mean site scores on DCA axes 1 and 2.

Fyke Netting

Winter-spring fyke netting for juvenile salmon was conducted by the USFWS in 1973, 1974, 1977, and 1980. DFG performed the sampling in 1981, 1982, 1983, and 1986 (EA 1991). The locations and sampling periods are in Table 3. The fyke nets used were 7.6 m long with a 0.9 x 1.5 m opening and 12.2 m long with a 1.5 x 2.7 m opening. The variable mesh netting tapered to 0.3 x 0.3 m at the cod end into an attached aluminum holding box. Nets were usually deployed for two to four nights per week and checked once every 24 hours. The number and size of captured juvenile salmon were recorded as was the number of individuals of other fish species. Resident fish captured during fyke netting were summarized as percentage abundance of each species captured at each site for each year of sampling.

Table 3 Fyke net locations and sampling periods for each year sampled[a]

Location	Rkm	1973	1974	1977	1980	1981	1982	1983	1986
Turlock Lake SRA	68.2	2/15–6/8	2/13–6/7	2/14–5/18	1/28–6/13	2/17–5/14	1/19–4/30	1/26–6/1	2/05–3/28
Hickman Spill	52.3	--	--	--	3/27–6/13	--	1/19–4/30	--	--
Putnam Gravel	49.2	2/14–6/8	2/13–6/7	2/14–5/18	--	--	--	--	--
Charles Road	40.2	--	--	--	1/28 3/26	--	--	--	--
McCleskey Ranch	9.7	2/27–6/8	2/13–6/7	2/14–5/18	1/28–6/13	--	2/1–4/30	--	--

[a] Dashes (--) indicate location not sampled.

Rotary Screw Traps

Springtime juvenile salmon sampling was conducted with two 2.44-meter diameter rotary screw traps (RST) in 1995 (26 April to 1 June) by TID and MID and in 1996 (18 April to 29 May) by DFG at rkm 5.8 (Shiloh Road) (EA 1997). Only one trap was fished after 19 May in 1995 and after 17 May in 1996. The traps were located out of the main current in 1995 due to high flows and floating debris. The 1996 deployment was in the main current.

The two traps were fished side-by-side and were usually checked in the morning and evening, except when more frequent checks were required to remove debris. All fish and debris were removed from the RSTs each time they were checked. The fish were separated by species and counted. All of the juvenile salmon, or a subsample, were measured. Lengths of other fish species were estimated or occasionally measured.

Salmon data were summarized as daily catch per trap, because one or two traps were fished at a time. Resident fish captured during rotary screw trapping were summarized as percentage abundance of each species captured each year.

Summer Surveys

Summer surveys of resident fishes were conducted, generally during May to September, from 1988 to 1994. Unlike other sampling efforts, which focused on chinook salmon, these surveys were designed to document the distribution of all fish species throughout the river (Table 4). Two other sampling methods, electrofishing and snorkeling, in addition to seining, provided a greater likelihood of capturing other fish species. Seining was only conducted in the first year, 1988, because few species were captured. Snorkeling was sometimes limited due to water clarity and generally was not effective downstream of rkm 40. Only snorkeling was conducted in 1994. All years included both "early summer" and "late summer" sampling periods (Table 5) when intensive sampling was conducted to detect the presence of juvenile chinook salmon.

Table 4 Summer survey locations for each year sampled[a]

Location	*Rkm*	*1988*	*1989*	*1990*	*1991*	*1992*	*1993*	*1994*
Riffle A3	83.0	X	X	X	X	X	X	X
Riffle A7	81.6	X	X	X	--	--	--	--
Riffle 2	80.3	X	X	X	X	X	X	X
Riffle 5	78.7	X	X	X	X	X	--	X
Riffle 9	74.7	X	X	X	X	X	X	X
Riffle 23BC	68.1	X	X	X	X	X	--	--
Riffle 33	62.3	X	X	X	--	--	--	--
Riffle 39/40	57.8	X	X	X	X	X	X	X
Riffle 53	51.5	X	--	--	--	--	--	--
Riffle 58	50.7	--	X	X	X	X	X	X
Charles Road	40.1	X	X	X	X	X	X	X
Legion Park	29.3	X	X	X	--	--	--	--
Riverdale Park	19.8	X	X	X	X	X	--	--
McCleskey Ranch	9.7	X	X	X	--	--	--	--
Shiloh Bridge	5.8	X	X	X	X	X	X	--
Total number of locations sampled		14	14	14	10	10	7	7

[a] Dashes (--) indicate not sampled.

Table 5 Primary summer survey sampling periods

Year	*Early summer*	*Late summer*
1988	05 May – 02 Jun	20 – 22 Sep
1989	23 May – 02 Jun	05 – 15 Sep
1990	28 May – 06 Jun	18 – 28 Sep
1991	10 – 14 Jun	06 – 13 Sep
1992	01 – 10 Jun	21 – 29 Sep
1993	07 – 10 Jun	25 – 27 Oct
1994	13 – 14 Jul	03 – 04 Oct

Snorkeling was conducted by one or more persons. Observers would proceed through a specified area and record on dive slates the species, numbers, and sizes of all fish observed. In 1988, electrofishing was conducted with a Smith-Root Model 12 backpack electroshocker. In all other years, a gas-powered DC generator mounted on a tow barge with three hand-held anodes was used. Block nets were sometimes used to isolate sample areas. Stream reaches snorkeled and electrofished ranged from 50 to 150 m in length.

Salmon catch data from the primary sampling periods were summarized by sampling method, date, and location. For the other species, total catch for each sampling method was summarized as percentage abundance of species using data from all samples. Snorkeling and electrofishing data were used in additional resident fish analyses.

Frequency of occurrence was calculated as described for the winter-spring seining data. Only sites sampled at least five times were included in frequency of occurrence analyses. Analysis of the number of species captured per sampling effort was also calculated as described for the seining data.

Assemblage structure of the resident fishes was described using detrended correspondence analysis (DCA) of the electrofishing data, as described for the seining data. A total of 10 sites was sampled consistently and included in the analysis. To determine if there were any seasonal changes in fish assemblage structure, analyses were conducted using two samples per year. An early summer sample was defined as the sample collected closest to 1 June of each year. A late summer sample was defined as the sample collected closest to mid-September of each year. Two-way ANOVA was used to test for annual and seasonal (that is, early versus late summer) differences in site scores on DCA axes 1 and 2.

Results

Adult Fall-run Chinook Salmon

Since 1940, the salmon runs varied from about 100 to 130,000 with an average estimate of 18,300 and a median estimate of 7,100 (Figure 2, Table 6). The date of the first observation of adult salmon at La Grange ranged from 10 September to 6 November with a median of 11 October for the period 1981–1997 (Table 1). The peak weekly count of live salmon during 1971–1997 ranged from 31 October to 27 November with a median date of 11 November.

Age composition of the 1981–1997 runs varied widely (Figure 3). Occasionally a strong year class would dominate consecutive years (arriving as two-year olds the first year and three-year olds the second) such as occurred in 1981–1982, 1987–1988, and 1996–1997. From 1981–1997 there were six years when two-year olds were most abundant and 11 years when three-year olds were most abundant. Four-year olds were always less than one-third of the 1981–1997 runs and five-year olds were always less than 5%.

The percentage of females varied from 25% to 67% during the period 1971-1997 (Figure 4). Sex composition varied with the age composition. Years with a high percentage of two-year olds tended to have a lower percentage of females (Figure 5). The percentage of adult salmon with coded-wire tags increased from less than 2% before 1987 up to an average of 20% during 1992–1997 (Figure 6). Redd counts during 1981–1997 varied from 51 to 3,034 (Table 7). Spawning use indices varied from 2.85 to 0.27, declining in a downstream direction (Table 7).

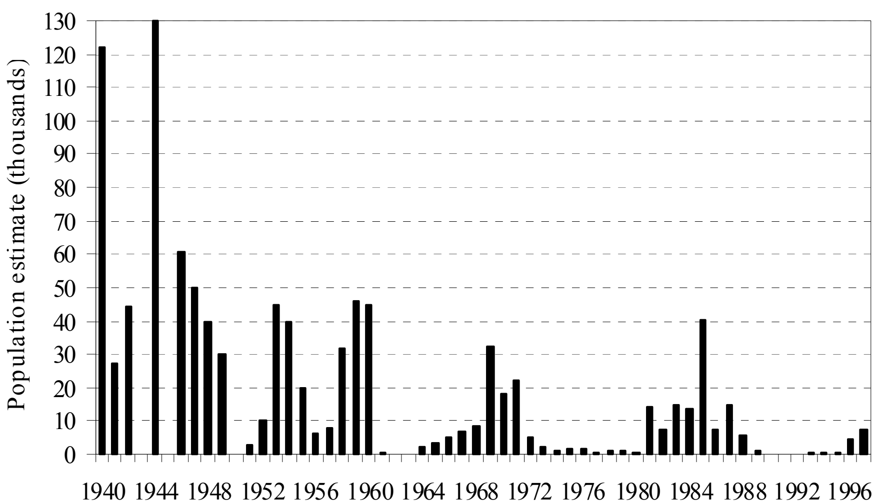
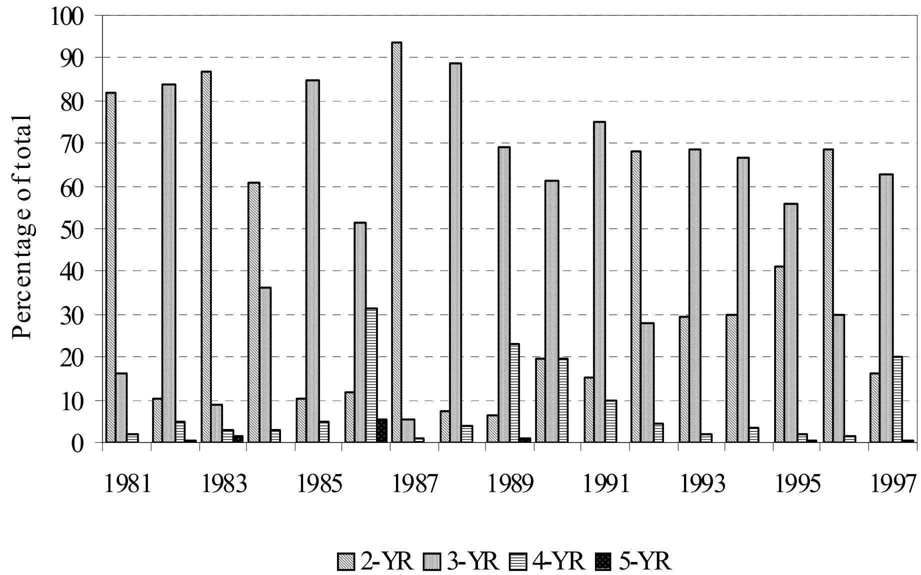


Figure 2 Estimates of adult fall-run chinook salmon in the Tuolumne River. There was only a partial count in 1941 and no counts in 1943, 1945, and 1950.

Table 6 Tuolumne River adult fall-run chinook salmon population estimates[a]

Year	Population estimate (x 1,000)	Year	Population estimate (x 1,000)	Year	Population estimate (x 1,000)
1940	122.0	1960	45.0	1980	0.6
1941	27.0	1961	0.5	1981	14.3
1942	44.0	1962	0.2	1982	7.1
1943	--	1963	0.1	1983	14.8
1944	130.0	1964	2.1	1984	13.8
1945	--	1965	3.2	1985	40.3
1946	61.0	1966	5.1	1986	7.3
1947	50.0	1967	6.8	1987	14.8
1948	40.0	1968	8.6	1988	6.3
1949	30.0	1969	32.2	1989	1.3
1950	--	1970	18.4	1990	0.1
1951	3.0	1971	21.9	1991	0.1
1952	10.0	1972	5.1	1992	0.1
1953	45.0	1973	2.0	1993	0.5
1954	40.0	1974	1.2	1994	0.5
1955	20.0	1975	1.6	1995	0.7
1956	6.0	1976	1.7	1996	4.6
1957	8.0	1977	0.5	1997	7.1
1958	32.0	1978	1.3		
1959	46.0	1979	1.2		

[a] There was only a partial count done in 1941 and no counts done in 1943, 1945, and 1950.



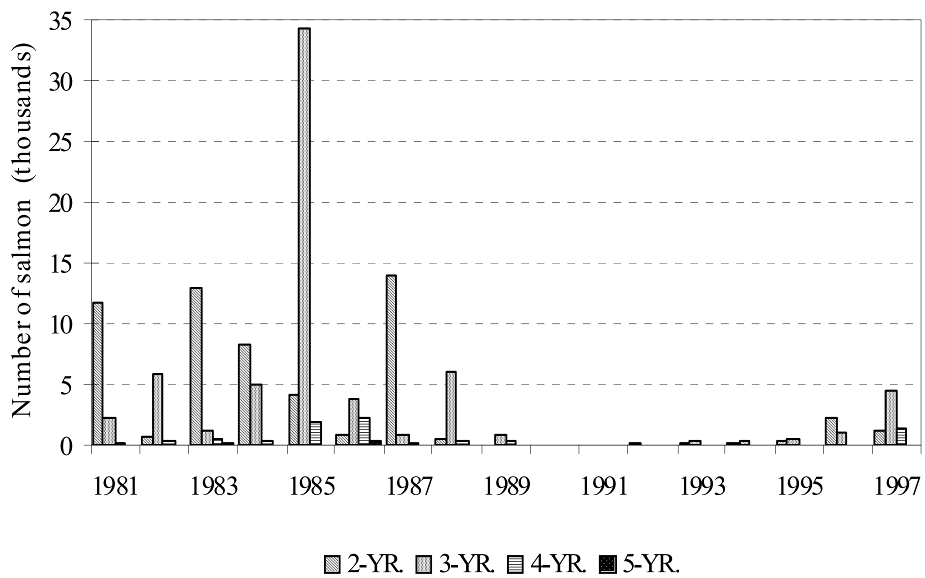


Figure 3 Estimated percentage and number of age classes in salmon runs based on fork length frequencies

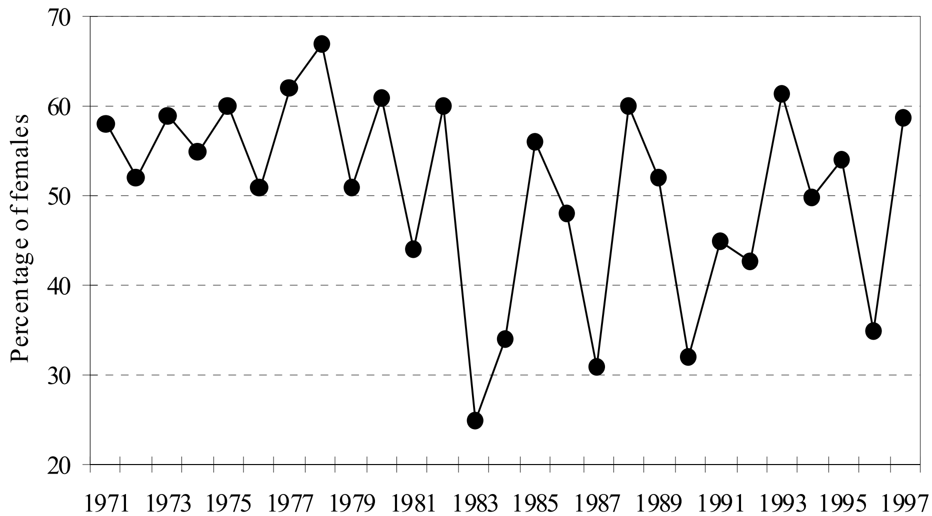


Figure 4 Percentage of females in Tuolumne River salmon runs

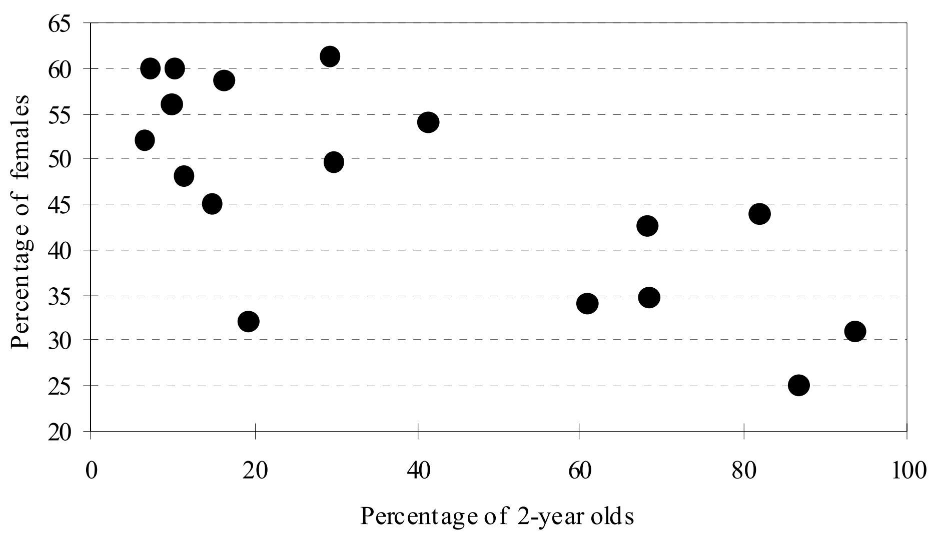


Figure 5 Percentage of females plotted against estimated percentage of two-year olds for 1981–1997 salmon runs

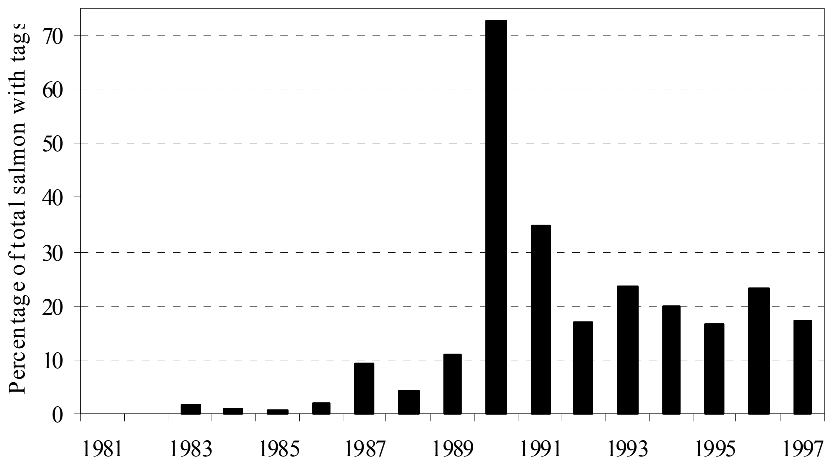


Figure 6 Estimated percentage of adult salmon with coded-wire tags in 1981–1997 salmon runs

Table 7 Total redd counts for each survey reach and the entire spawning reach[a]

Year	*Survey reach (rkm to rkm)* 42.0 – 54.6	54.6 – 67.4	67.4 – 76.4	76.4 – 81.6	81.6 – 83.1	No. of redds counted	Estimated no. of females	Females per redd
1981	137	440	461	510	128	1,676	6,292	3.8
1982	--	218	308	467	106	1,099	4,200	3.8
1983	18	155	180	110	2	465	3,700	8.0
1984	37	265	428	358	55	1,143	4,658	4.1
1985	140	605	874	1,230	185	3,034	22,568	7.4
1986	68	365	271	428	116	1,248	3,792	3.0
1987	77	209	216	246	102	850	4,619	5.4
1988	376	431	402	552	141	1,902	4,080	2.1
1989	76	149	130	181	48	584	676	1.2
1990	6	21	21	10	0	58	28	0.5
1991	7	13	9	16	6	51	27	0.5
1992	10	7	7	17	12	53	55	1.0
1993	17	49	61	78	45	250	238	1.0
1994	21	82	95	79	45	322	249	0.8
1995	25	56	61	48	39	229	522	2.3
1996	19	58	84	125	57	343	1,139	3.3
1997	26	171	272	404	108	981	4,224	4.3
Mean percentage of redds in survey reach								
	8.4%	23.7%	26.5%	31.0%	10.4%			
Spawning use index for survey reach								
	0.27	0.76	1.21	2.45	2.85			

[a] The ratio of female salmon to the number of redds is given for the entire spawning reach. The use index (% redds / % length) was calculated using data summed over all years.

Juvenile Fall-run Chinook Salmon

Density of juvenile salmon, as determined from winter-spring seining, declined each year after a winter peak in fry abundance (Figure 7). Juvenile salmon were abundant in the lower river section below Dry Creek (rkm 26.4) only in the high flow years of 1986, 1995, and 1997 (Figures 8 and 9). All measures of juvenile salmon size typically increased after February (Figures 10, 11, and 12), although in some years average size declined from April to May (Figure 10), because large numbers of smaller fry were captured.

The catch rate of the rotary screw traps was lower in 1995 than in 1996 (Figure 13). Peaks in juvenile salmon abundance were less obvious in 1995 compared to 1996.

No juvenile salmon were captured during the summer flow study in 1991, 1992, and 1994 (Table 8). Most juvenile salmon were captured in the early period with the largest catches upstream of rkm 74. Few juvenile salmon were captured in the late sampling as compared to the early periods. All but one of the juvenile salmon observed during the late period were found upstream of rkm 74.

Resident Fishes

A total of 33 taxa of fish, including chinook salmon, was captured during the various sampling programs (Table 9). Of the 33 taxa, 12 taxa are native to California and 21 are introduced. All lampreys captured were identified as Pacific lamprey; however, not every individual was examined in detail and it is possible that river lamprey (*Lampetra ayersi*) was also present. Similarly, several black bullheads (*Ameiurus melas*) were identified but the remaining *Ameiurus* species were combined into the general category of bullhead catfish.

The six methods of sampling used in the studies varied in effectiveness with regard to the capture of resident fish species. Winter-spring methods included seining, rotary screw trapping and fyke netting. Winter-spring seining generally caught few species in addition to chinook salmon during any single sampling effort (Figure 14). Mean number of species captured per sampling period varied from 1.0 to 2.4 species with standard deviations ranging from 0.8 to 1.2. However, over the course of the study winter-spring seining captured 28 of the 33 taxa present in the river (Table 10). Rotary screw trapping at rkm 5.6 resulted in a mean of 2.4 species (standard deviation 1.8) captured per sampling effort (usually daily), which was comparable to the seining results for that site (mean = 2.0, standard deviation = 1.2). Rotary screw trapping captured about 23 taxa; however, there may have been additional species included in some of the general categories used (Table 11). Fyke netting also captured few species during any single sampling effort with the mean number of species captured at the five sites ranging from 1.1 to 1.7. Standard deviations ranged from 1.0 to 1.5. Fyke netting captured about 27 taxa (Tables 12 and 13).

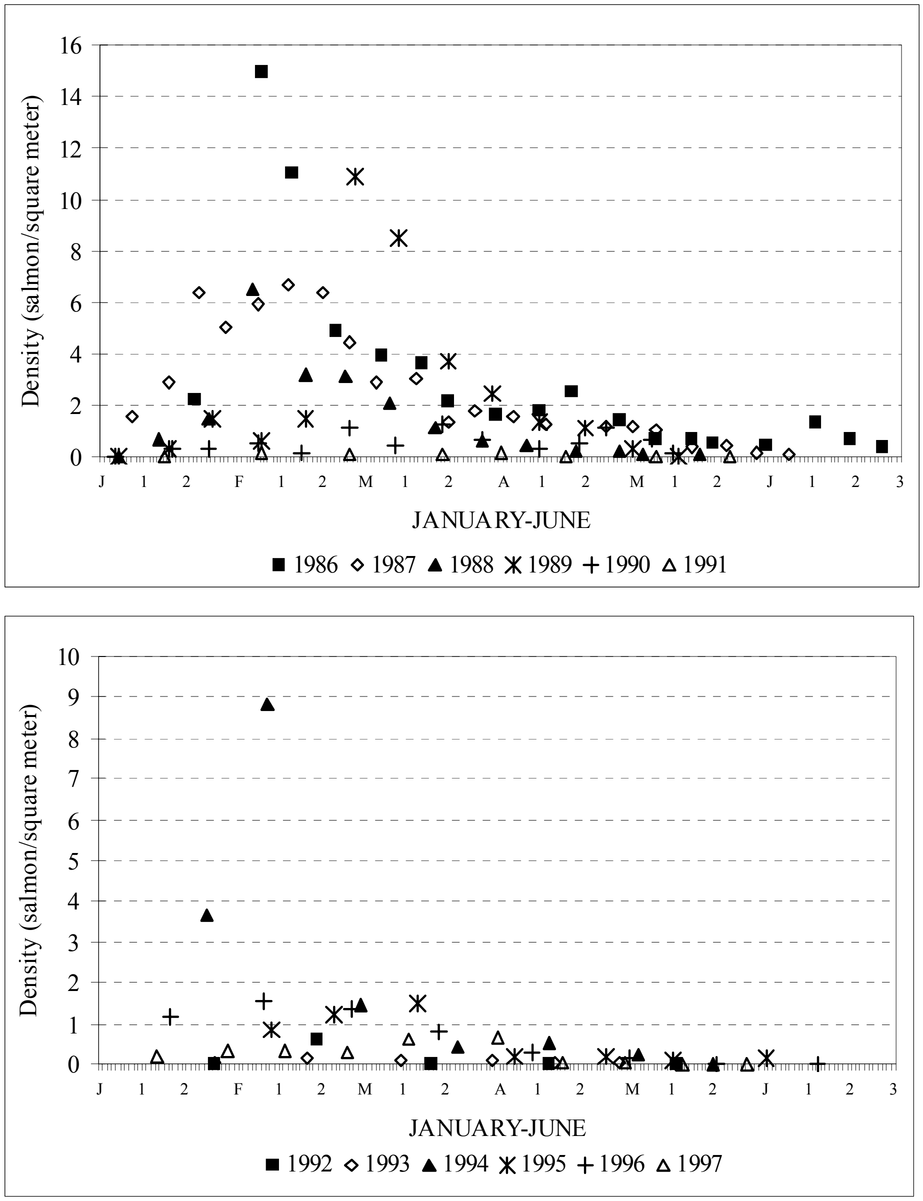


Figure 7 Densities of salmon from seining surveys from 1986 to 1997

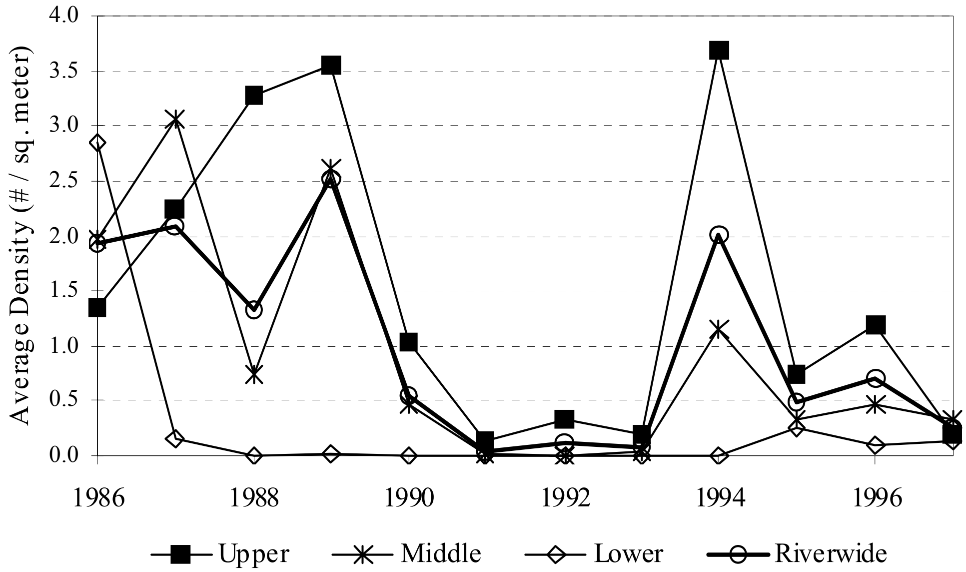


Figure 8 Densities of juvenile salmon captured during seining surveys for upper, middle, and lower sections of the river and for the entire river

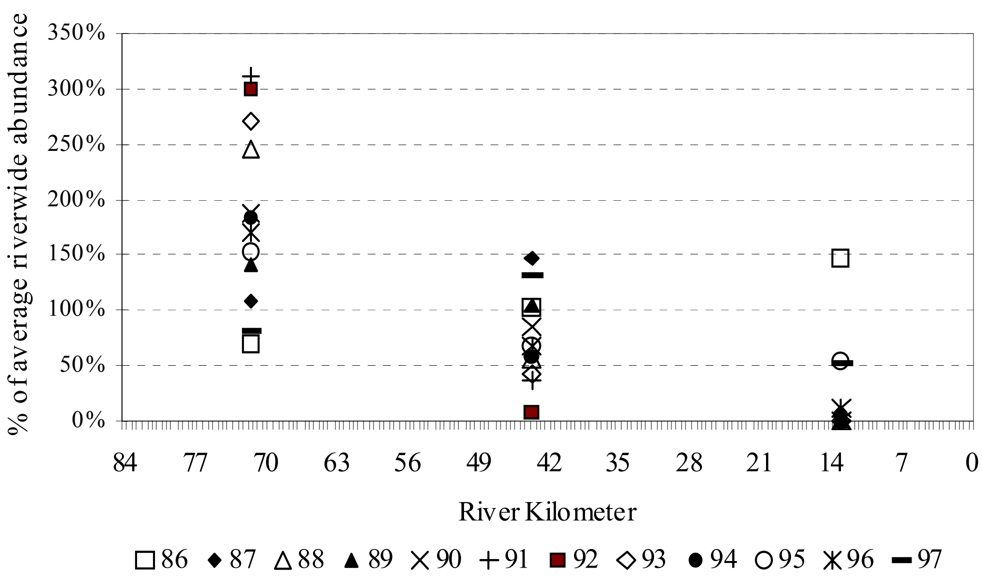


Figure 9 Densities of juvenile salmon in upper, middle, and lower sections standardized as percentage of the annual riverwide density and plotted at section midpoints

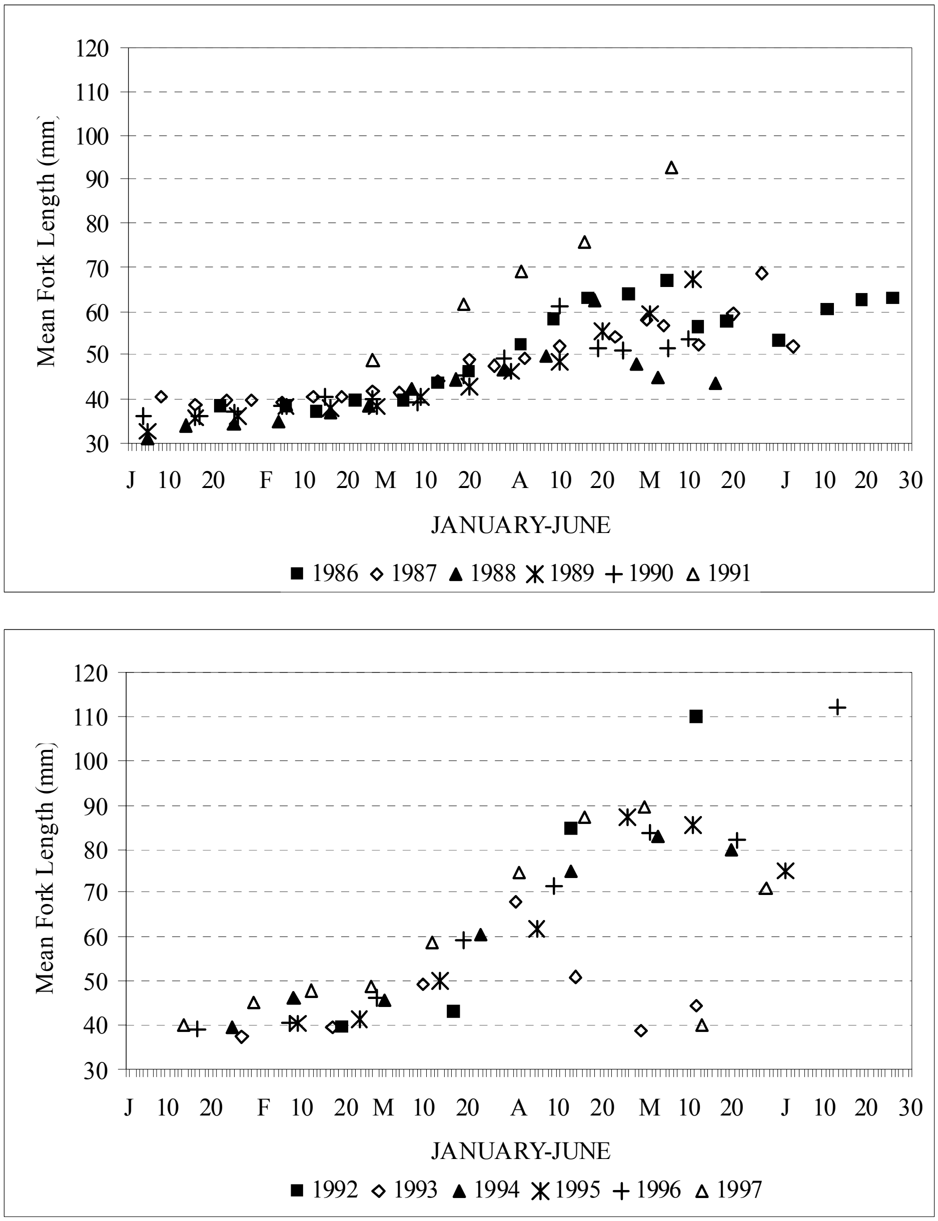
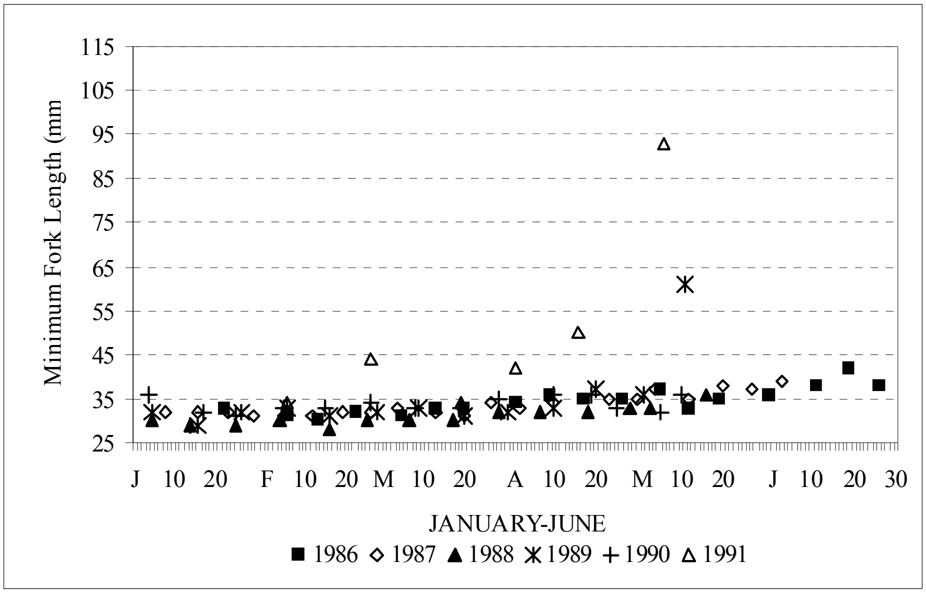


Figure 10 Mean fork length of salmon captured during seining surveys from 1986 to 1997



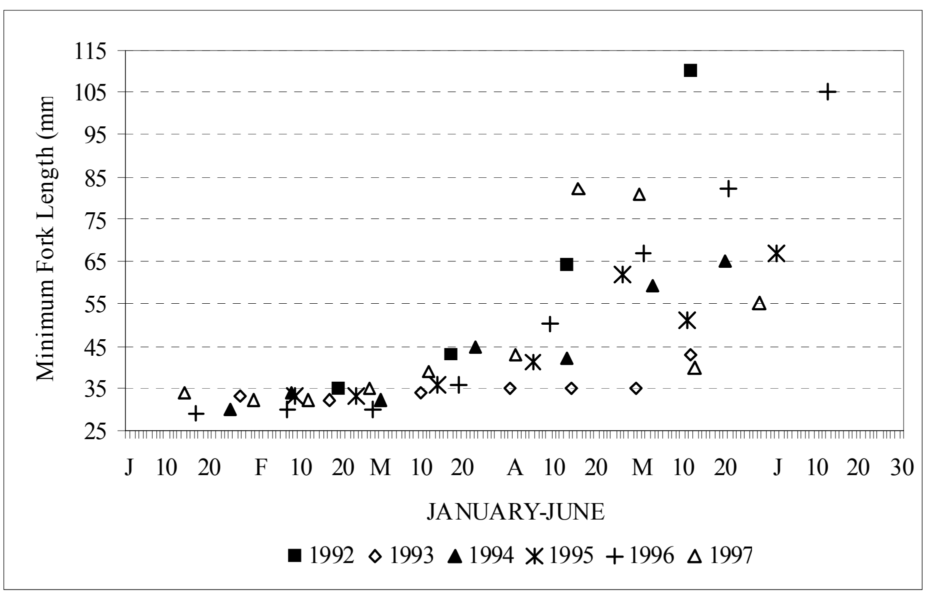
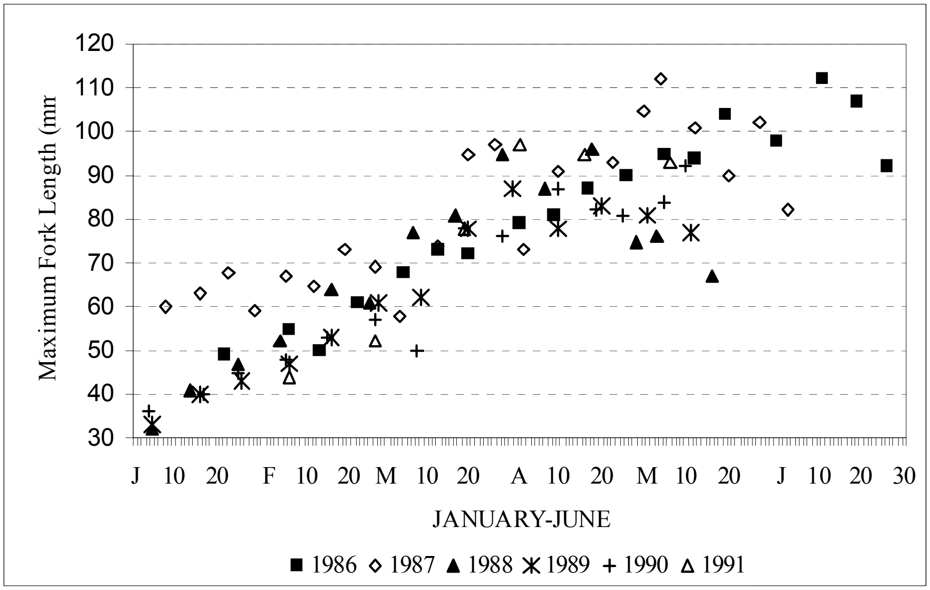


Figure 11 Minimum fork length of salmon captured during seining surveys from 1986 to 1997



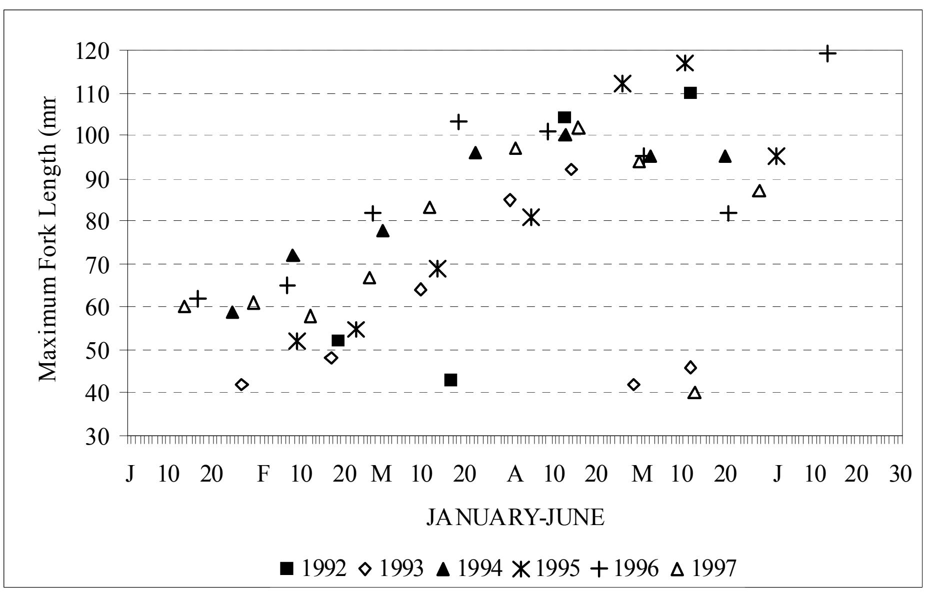
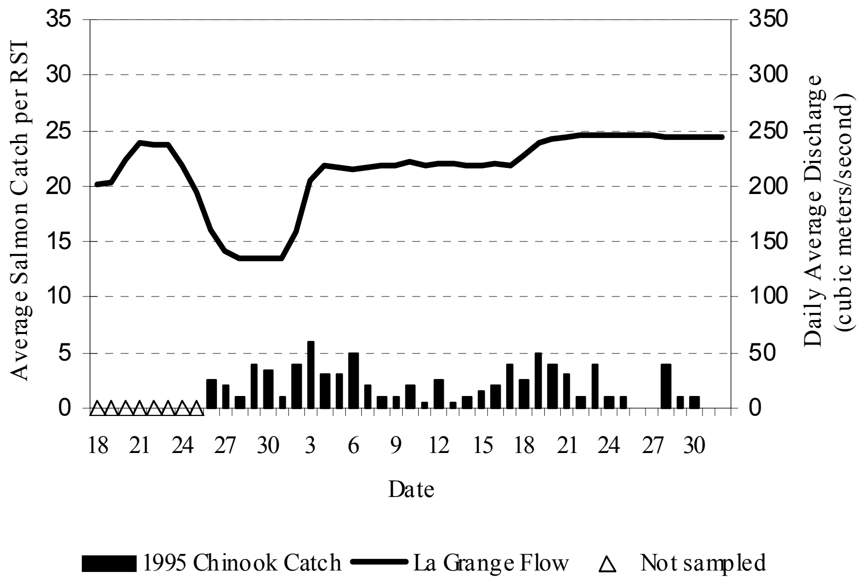


Figure 12 Maximum fork length of salmon captured during seining surveys from 1986 to 1997



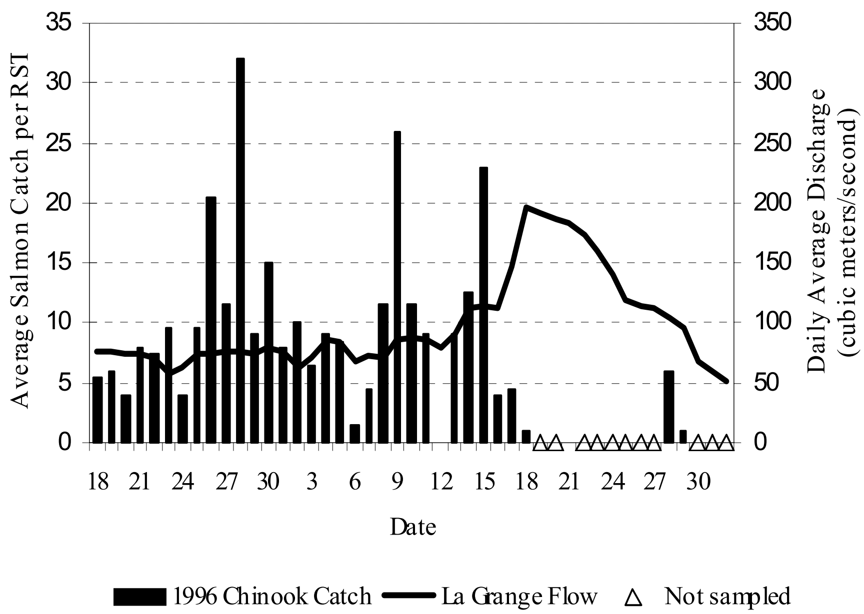


Figure 13 Rotary screw trap salmon catch data from 1995 and 1996 at Shiloh Road (rkm 5.8) during 18 April to 1 June. Days when sampling did not occur are indicated by a triangle.

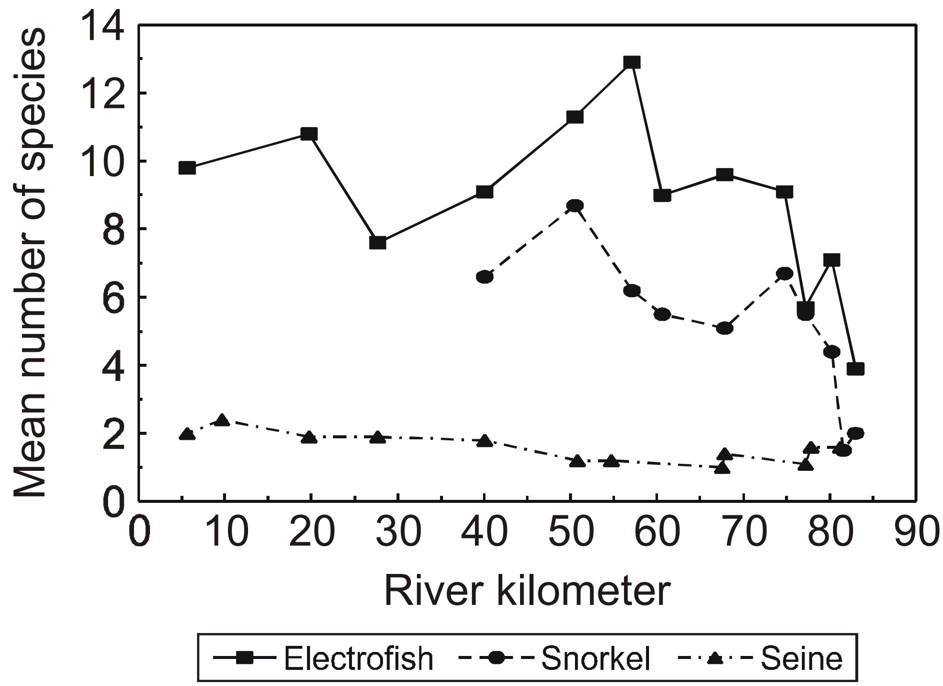


Figure 14 Mean number of species, excluding chinook salmon, captured during annual winter-spring seining and summer snorkeling and electrofishing

Table 8 Number of juvenile salmon captured during primary summer survey periods, listed by date, method, location, and river kilometer

Date	*Sampling method*	*Location*	*River kilometer*	*Number captured*
05 May 88	Seine	RA3	83.0	3
05 May 88	Snorkel	RA3	83.0	3
13 May 88	Snorkel	OLGB	81.3	22
06 May 88	Seine	R2	80.3	1
06 May 88	Snorkel	R2	80.3	1
13 May 88	Snorkel	R3B	79.0	1
13 May 88	Snorkel	R4A	78.5	1
13 May 88	Snorkel	R4B	77.9	1
24 May 88	Electroshocking	R5	77.2	1
13 May 88	Snorkel	R5	77.2	25
06 May 88	Electroshocking	R5	77.2	8
06 May 88	Snorkel	R5	77.2	104
11 May 88	Electroshocking	R9	74.8	3
11 May 88	Snorkel	R9	74.8	3

Table 8 Number of juvenile salmon captured during primary summer survey periods, listed by date, method, location, and river kilometer (Continued)

Date	*Sampling method*	*Location*	*River kilometer*	*Number captured*
11 May 88	Seine	TRR	67.9	1
12 May 88	Snorkel	R33	60.7	1
23 May 89	Electroshocking	RA3	83.0	2
23 May 89	Snorkel	RA3	83.0	127
23 May 89	Electroshocking	RA7	81.6	6
24 May 89	Electroshocking	R2	80.3	1
24 May 89	Electroshocking	R5	77.2	5
01 Jun 89	Electroshocking	R9	74.8	5
25 May 89	Electroshocking	TRR	67.9	2
25 May 89	Electroshocking	R33	60.7	10
01 Jun 89	Electroshocking	R39	57.1	2
02 Jun 89	Electroshocking	R58	50.5	2
26 May 89	Electroshocking	CROAD	40.1	1
09 Jul 89	Electroshocking	RA3	83.0	1
29 May 90	Electroshocking	RA3	83.0	20
05 Jun 90	Snorkel	RA3	83.0	12
29 May 90	Electroshocking	RA7	81.6	50
30 May 90	Electroshocking	R2	80.3	16
30 May 90	Electroshocking	R5	77.2	8
31 May 90	Electroshocking	R9	74.8	37
31 May 90	Electroshocking	TRR	67.9	4
01 Jun 90	Electroshocking	R33	60.7	4
01 Jun 90	Electroshocking	R39	57.1	3
02 Jun 90	Electroshocking	R58	50.5	13
18 Sep 90	Electroshocking	RA3	83.0	1
18 Sep 90	Electroshocking	RA7	81.6	2
08 Jun 93	Electroshocking	RA3	83.0	1
07 Jun 93	Snorkel	RA3	83.0	35
07 Jun 93	Snorkel	R2	80.3	2
09 Jun 93	Electroshocking	R58	50.5	1
08 Jun 93	Snorkel	CROAD	40.1	1
27 Oct 93	Snorkel	RA3	83.0	10
25 Oct 93	Snorkel	RA7	81.6	7
25 Oct 93	Snorkel	R1A	81.3	7
27 Oct 93	Snorkel	R2	80.3	11
25 Oct 93	Snorkel	R5	77.2	3
27 Oct 93	Snorkel	R9	74.8	7
25 Oct 93	Electroshocking	R58	50.5	1

Table 9 Common name, scientific name, origin, and code for species captured during Tuolumne River fish monitoring

Common name	Scientific name	Origin[a]	Code[b]
Petromyzontidae (lampreys)			
Pacific lamprey	*Lampetra tridentata*	N	LMP
Clupeidae (shad and herring)			
Threadfin shad	*Dorosoma petenense*	I	--
Salmonidae (salmon and trout)			
Chinook salmon	*Oncorhynchus tshawytscha*	N	--
Rainbow trout	*Oncorhynchus mykiss*	N	--
Cyprinidae (minnows)			
Common carp[c]	*Cyprinus carpio*	I	CP
Fathead minnow	*Pimephales promelas*	I	--
Golden shiner	*Notemigonus crysoleucas*	I	GSH
Goldfish	*Carassius auratus*	I	GF
Hardhead	*Mylopharodon conocephalus*	N	HH
Hitch	*Lavinia exilicauda*	N	--
Red shiner	*Cyprinella lutrensis*	I	RSH
Sacramento blackfish	*Orthodon microlepidotus*	N	--
Sacramento splittail	*Pogonichthys macrolepidotus*	N	--
Sacramento pikeminnow	*Ptychocheilus grandis*	N	SQ
Catostomidae (suckers)			
Sacramento sucker	*Catostomus occidentalis*	N	SKR
Ictaluridae (catfish)			
Black bullhead	*Ameiurus melas*	I	--
Bullhead catfish[d]	*Ameiurus spp.*	I	BCF
Channel catfish	*Ictalurus punctatus*	I	CCF
White catfish	*Ameiurus catus*	I	WCF
Poeciliidae (livebearers)			
Western mosquitofish	*Gambusia affinis*	I	GAM
Atherinidae (silversides)			
Inland silverside	*Menidia beryllina*	I	ISS
Percichthyidae (temperate basses)			
Striped bass	*Morone saxatilis*	I	--
Centrarchidae (basses and sunfish)			
Black crappie	*Pomoxis nigromaculatus*	I	--
Bluegill	*Lepomis macrochirus*	I	BG
Green sunfish	*Lepomis cyanellus*	I	GSF
Largemouth bass	*Micropterus salmoides*	I	LMB
Redear sunfish	*Lepomis microlophus*	I	RSF
Smallmouth bass	*Micropterus dolomieu*	I	SMB
Warmouth	*Lepomis gulosus*	I	--
White crappie	*Pomoxis annularis*	I	--
Percidae (perch)			
Bigscale logperch	*Percina macrolepida*	I	--
Embiotocidae (surf perch)			
Tule perch	*Hysterocarpus traski*	N	--
Cottidae (sculpins)			
Prickly sculpin	*Cottus asper*	N	--
Riffle sculpin	*Cottus gulosus*	N	RSCP

[a] N = native, I = introduced.
[b] Dashes (--) indicate no code was assigned.
[c] A single mirror carp, a variety of common carp, was captured.
[d] Because of difficulty in field identification of bullhead catfish, they were combined into a single category.

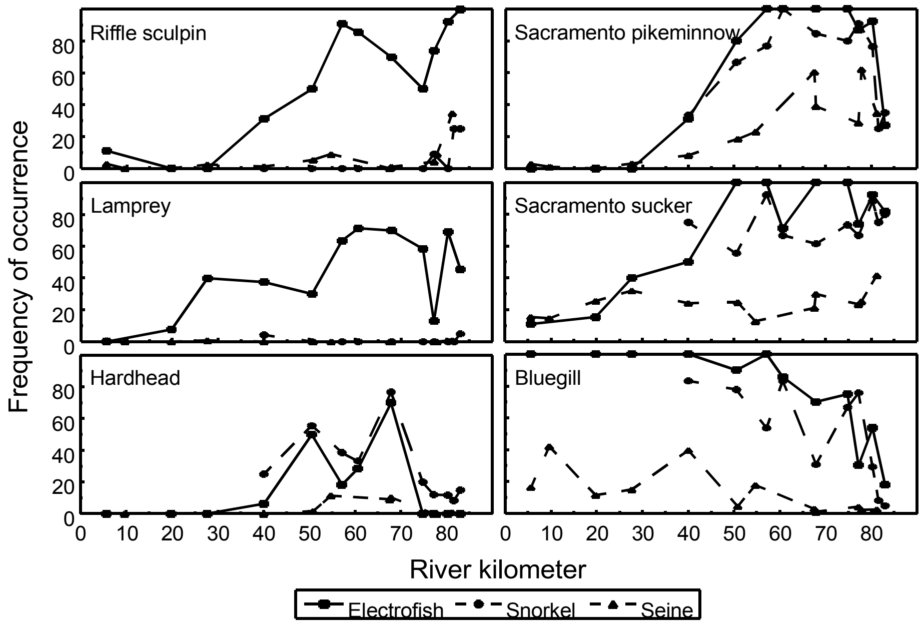


Figure 15A Frequency of occurrence plots for common native species (and bluegill) included in detrended correspondence analysis of annual winter-spring seining and summer electroshocking and snorkeling

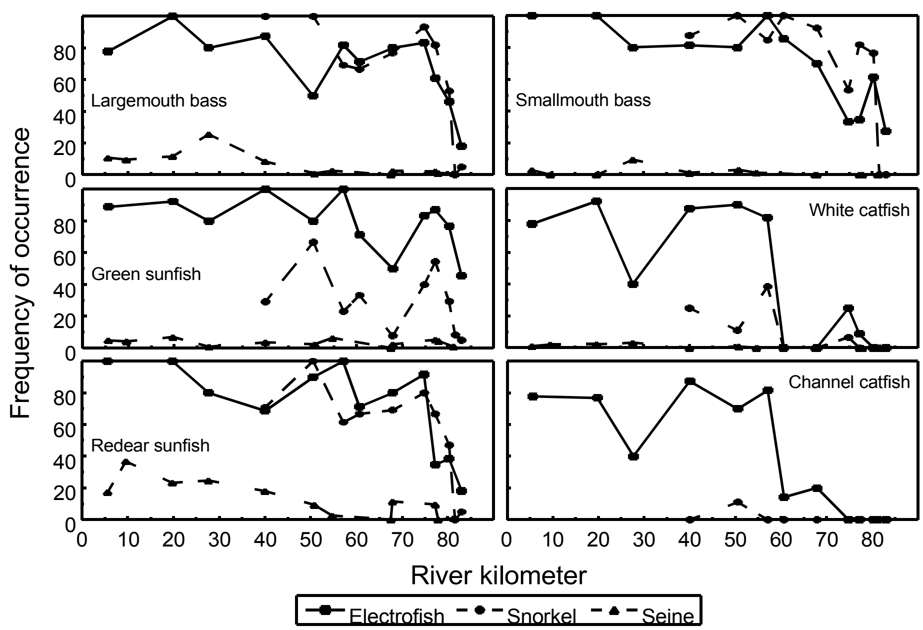


Figure 15B Frequency of occurrence plots for common centrarchid and ictalurid species included in detrended correspondence analysis of annual winter-spring seining and summer electroshocking and snorkeling

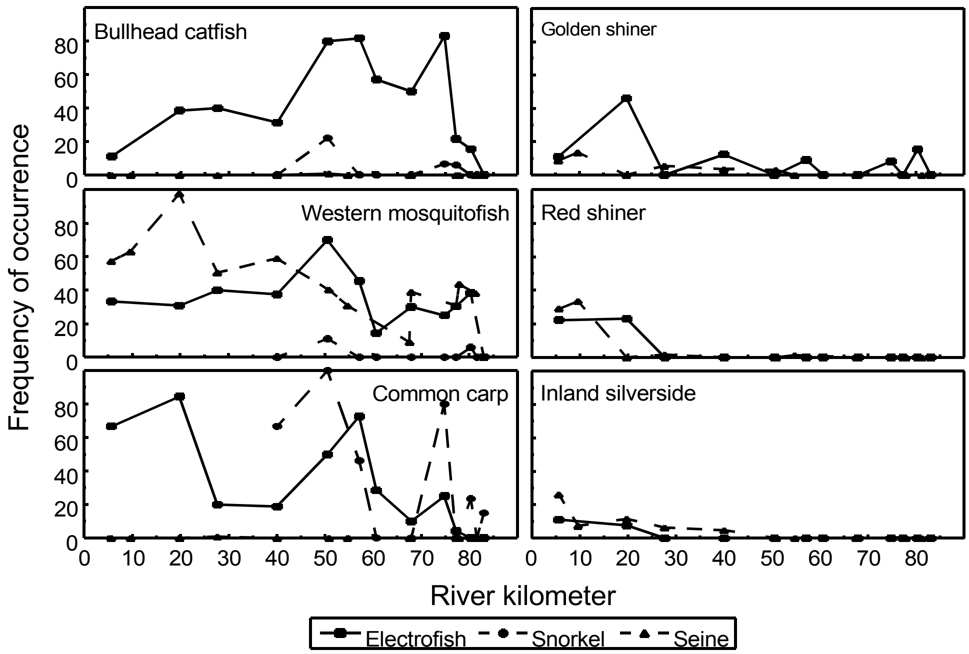


Figure 15C Frequency of occurrence plots for other common introduced species included in detrended correspondence analysis of annual winter-spring seining and summer electroshocking and snorkeling

Table 10 Percentages of fishes (excluding chinook salmon) captured during winter-spring salmon seining surveys (Jan–Jun, 1986–1997) and summer survey electroshocking (May–Oct, 1988–1993), snorkeling (May–Oct, 1988–1993), and seining (May–Sep 1988)

	Winter-spring survey	*Summer survey*		
Taxon	*Seining*	*Seining*	*Electroshocking*	*Snorkeling*
Bigscale logperch	<0.1	0	<0.1	0
Black bullhead	<0.1	0	<0.1	0
Black crappie	<0.1	0	0	0
Bluegill	2.4	9.5	10.6	7.3
Bullhead catfish	<0.1	0	1.6	0.7
Centrarchids (unknown)	0.6	0.1	1.9	2.0
Channel catfish	0	<0.1	1.9	<0.1
Common carp	<0.1	0	0.6	1.0
Cyprinids (unknown)	0.1	0	0	3.1
Golden shiner	2.1	0.1	0.2	0
Goldfish	<0.1	<0.1	0.7	0.8
Green sunfish	0.2	0.2	9.7	2.0
Hardhead	1.0	0	0.7	2.2
Hitch	0	0	0.1	<0.1
Inland silverside	1.3	0.6	<0.1	0
Largemouth bass	1.1	2.5	5.7	8.6
Pacific lamprey	<0.1	0	1.1	<0.1
Prickly sculpin	<0.1	0	0	0
Rainbow trout	0.1	0	<0.1	<0.1
Redear sunfish	5.0	2.0	8.0	17.1
Red shiner	6.2	0	0.1	0
Riffle sculpin	1.9	0.1	19.0	0.1
Sacramento blackfish	<0.1	0	0.1	<0.1
Sacramento pikeminnow	7.3	1.6	10.2	12.2
Sacramento splittail	0.1	0	<0.1	0
Sacramento sucker	35.4	0.9	13.3	36.9
Smallmouth bass	0.2	1.6	4.5	5.6
Striped bass	0	0	<0.1	0
Threadfin shad	0.3	0	<0.1	0
Tule perch	0	0	<0.1	0
Warmouth	<0.1	0	0.1	<0.1
Western mosquitofish	34.4	80.5	1.0	<0.1
White crappie	<0.1	0	0	0
White catfish	0.1	0.2	8.6	0.2
Number of samples	1,077	37	148	194
Total fish captured	21,736	3,611	23,774	26,371

Table 11 Percentage abundance of fish species, excluding chinook salmon, captured in rotary screw traps at river kilometer 5.6 in 1995 (25 April to 30 May) and 1996 (18 April to 29 May)[a]

Taxon	*1995*	*1996*
Native taxa		
Cottidae	0	1.0
Hardhead	0	0.3
Hitch	0	0.3
Sacramento blackfish	0	0.3
Sacramento pikeminnow	1.5	0.7
Sacramento sucker	5.5	3.9
Introduced taxa		
Black bullhead	0.1	0
Bluegill	0.1	8.5
Bullhead catfish	0	0.7
Centrarchidae	0.4	0.7
Channel catfish	0.1	0.3
Common carp	0.1	0
Golden shiner	0.3	3.6
Goldfish	4.5	3.9
Green sunfish	0.3	0.7
Ictaluridae	0	13.1
Inland silverside	0.4	33.4
Largemouth bass	0.3	18.4
Red shiner	1.7	0.7
Threadfin shad	0	0.3
Warmouth	0	0.3
Western mosquitofish	2.9	7.2
White catfish	2.0	0.7
White crappie	0	1.0
Unknown		
Cyprinidae	79.7	0
Total number of fish	715	305

[a] Some fish were not identified to species but were identified to the lowest possible taxon.

Table 12 Percentages of fish taxa (excluding chinook salmon) captured by fyke nets at various river kilometer locations, 1973–1980

Year	*1973*			*1974*			*1977*			*1980*			
Rkm	9.7	49.2	67.6	9.7	49.2	67.6	9.7	49.2	67.6	9.7	42.0	54.7	67.6
Native taxa													
Cottus sp.	0	0	0	0	0	0	0	0	0	0	0.4	0	0
Hardhead	0	0	0	0	0	0	0	0	0	0	0	1.9	0
Hitch	0	0	0	0	0	0	5.4	0	1.8	0	0	1.9	0
Pacific lamprey	0	0	0	2.5	2.9	12.6	6.8	9.9	0.9	5.2	1.6	28.8	10.8
Sacramento blackfish	0	0	0	0	0	0	1.4	0	0	0	0	0	0
Sacramento splittail	0	0	0	11.5	0	0	0	0	0	0	0	0	0
Sacramento pikeminnow	0	0	0	0	0	0	0	0	1.8	0	0	0	1.4
Sacramento sucker	0	0	0	0.6	86.0	16.5	0	60.6	2.7	0	0.4	0	17.6
Introduced taxa													
Bigscale logperch	0	0	0	0	0	0	0	0	0	0	0	0	0
Black bullhead	0	0	0	0	0	0	0	1.4	0	0	0	0	0
Black crappie	0	0	0	0	0	0	0	0	0	0	0	0	0
Bluegill	0	0	0	4.2	2.1	16.5	8.1	2.8	17.9	16.8	5.6	23.1	28.4
Bullhead catfish	0	50	20	0	0	1.6	0	1.4	0	0	0	3.8	0
Centrarchidae	0	0	0	0.3	2.4	3.1	0	0	20.5	0	0	7.7	0
Channel catfish	0	0	40	0	0	0	2.7	0	0	1.1	0	0	0
Common carp	12.5	0	0	0.6	0.1	0	0	0	0	0	0	0	2.7
Golden shiner	0	0	0	1.4	0.2	0	1.4	0	0	0.2	0	0	1.4
Goldfish	0	0	0	2.0	0.1	0	13.5	2.8	0.9	0	0	0	0
Green sunfish	0	0	0	0.6	0	3.1	1.4	0	0	0	0	0	0
Ictaluridae	87.5	0	20	40.3	1.6	6.3	1.4	0	1.8	69.7	0.8	17.3	14.9
Largemouth bass	0	0	0	6.2	0.7	28.3	1.4	0	2.7	0.2	0.4	0	9.5
Pomoxis sp.	0	0	0	0.6	0.2	0	1.4	0	0	0	0	0	0
Redear sunfish	0	0	0	0	0	0	0	0	2.7	0	0	0	0
Smallmouth bass	0	0	0	1.7	3.7	3.9	0	12.7	6.3	0	0	0	0
Striped bass	0	0	0	0.3	0	0	0	0	0	0	0	0	0
Threadfin shad	0	0	0	25.6	0	0	37.8	1.4	34.8	3.5	90.4	11.5	0
Warmouth	0	50	0	1.4	0	6.3	1.4	0	4.5	0	0.4	1.9	13.5
Western mosquitofish	0	0	0	0.3	0.1	1.6	0	0	0	0	0	0	0
White catfish	0	0	20	0	0	0	16.2	7.0	0.9	3.3	0	0	0
Days sampled	23	24	22	28	29	29	24	26	28	57	31	35	54
Total fish	8	2	5	355	1452	127	74	71	112	459	250	52	74

Table 13 Percentages of fish taxa (excluding chinook salmon) captured by fyke nets at various river kilometer locations, 1981–1986

Year	*1981*	*1982*			*1983*	*1986*
River kilometer	*67.6*	*9.7*	*51.5*	*67.6*	*67.6*	*67.6*
Native taxa						
Cottus sp.	0	0	0	0	0	0
Hardhead	0	0	0.3	10.4	0	2.9
Hitch	0	0.9	1.9	0.9	2.2	0
Pacific lamprey	100.0	50.4	52.9	39.6	28.3	82.2
Sacramento blackfish	0	0	0	0	0	0
Sacramento splittail	0	0	0	0	0	0
Sacramento pikeminnow	0	0.9	8.0	9.4	0	0
Sacramento sucker	0	0	0	0	0	0
Introduced taxa						
Bigscale logperch	0	2.6	0	0	0	0
Black bullhead	0	0	0	0	0	0
Black crappie	0	0	0.6	0	0	0.6
Bluegill	0	20.9	23.0	14.2	39.1	12.6
Bullhead catfish	0	0	0	0	0	0
Centrarchidae	0	0	0	0	2.2	0
Channel catfish	0	0	0	0	0	0
Common carp	0	0	1.1	0	0	0
Golden shiner	0	0	0	0	0	0
Goldfish	0	0	0	0	0	0
Green sunfish	0	0	0	0	0	0
Ictaluridae	0	14.1	3.3	16.0	26.1	0.6
Largemouth bass	0	0.4	0.3	3.8	0	0.6
Pomoxis sp.	0	0	0	0	0	0.6
Redear sunfish	0	0	0	0	0	0
Smallmouth bass	0	0	0	0	0	0
Striped bass	0	0	0	0	0	0
Threadfin shad	0	9.8	8.3	0	2.2	0
Warmouth	0	0	0.3	5.7	0	0
Western mosquitofish	0	0	0	0	0	0
White catfish	0	0	0	0	0	0
Days sampled	8	16	23	24	11	15
Total number of fish	4	234	361	106	46	174

Although the three winter-spring methods captured similar numbers of species, catches were dominated by different species. Seining catches were dominated by western mosquitofish (34.4%) and Sacramento sucker (35.4%) (Table 10). No other species exceeded 10% of the catch. The rotary screw trap catch was dominated by unidentified cyprinids (79.7%) in 1995 (Table 11). Of the fish identified to species, Sacramento sucker (5.5%) and goldfish (4.5%) were most common. The catch in 1996 was dominated by unidentified catfish (13.1%), inland silverside (33.4%), and largemouth bass (18.4%) (Table 11). Fyke netting results were variable among sites and years (Tables 12 and 13). Unidentified catfish commonly exceeded 10% of the catch in the lower river (rkm 6.0 and 26.1). Threadfin shad was common (>10%) in 1974 and 1977, as were bluegill in 1980 and 1982. Other species common in at least one year included common carp, splittail, goldfish, white catfish, and Pacific lamprey. Catfish of all kinds were common at more upstream sites. Pacific lamprey, Sacramento sucker, bluegill, warmouth, threadfin shad, and hardhead were common in some years.

Seining was initially included in the summer flow study but was suspended after the first year (1988) because the catch consisted primarily of western mosquitofish with few other species captured (Table 10). Summer seining only captured 15 taxa with only western mosquitofish exceeding 10% of the catch. Summer snorkeling and electroshocking captured many more species than winter-spring seining (Figure 14) and the other methods. Mean number of species (mean ± standard deviation) ranged from 1.5 ± 1.3 to 8.7 ± 2.2 for snorkeling and from 3.9 ± 1.6 to 12.9 ± 2.2 for electroshocking. Snorkeling and electroshocking captured 22 and 30 taxa, respectively. Snorkeling was limited to the more upstream reaches of the river where visibility was sufficient to identify and count the fish present.

Fish Species Distributions

Only the annual winter-spring seining and summer electroshocking and snorkeling surveys sampled enough sites to give good information on resident fish species distributions. Percentage abundance of species in the winter-spring seining and summer surveys indicates that a number of species were relatively rare in the system (Table 10). The native species hitch, prickly sculpin, rainbow trout, Sacramento blackfish, Sacramento splittail, and tule perch never exceeded 1% of the total catch with any of the methods used. The introduced species black crappie, bigscale logperch, goldfish, striped bass, threadfin shad, white crappie and warmouth were similarly rare.

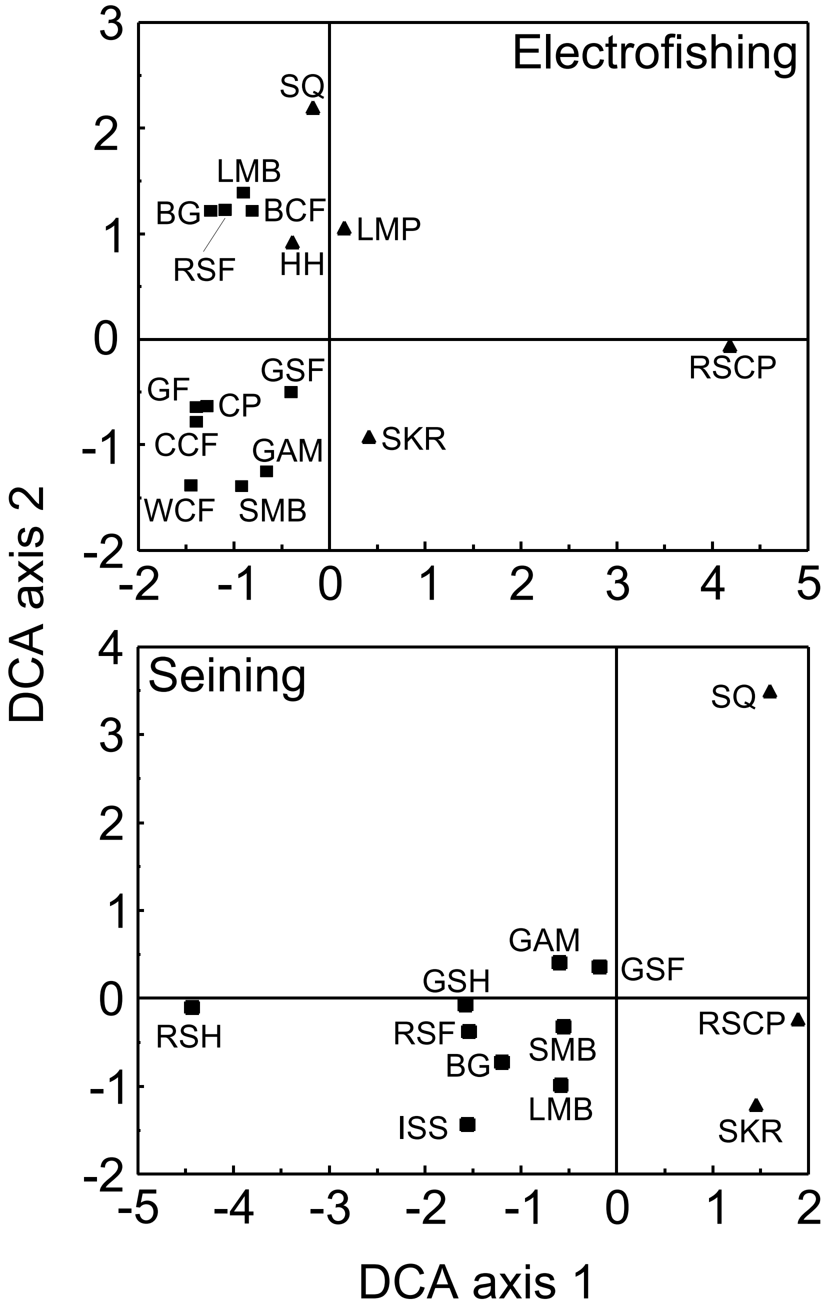


Figure 16 Species scores on DCA axes 1 and 2 resulting from analysis of the summer electrofishing data and winter-spring seining data. Species codes as in Table 9. Triangles indicate native species and squares indicate introduced species.

Frequency of occurrence plots for the common species included in DCA analyses indicated that the species were not evenly distributed in the river, particularly during the summer (Figure 15A). The common native species exhibited two basic patterns of distribution. In the summer electrofishing surveys, Sacramento sucker, lamprey, and riffle sculpin occurred most frequently at upstream sites above about rkm 50. Lamprey and riffle sculpin were rarely captured in the winter-spring seining or summer snorkeling. Sacramento suckers were fairly evenly distributed in the river in the winter-spring seining survey but in the summer surveys were most frequent upstream of rkm 50. The other two common native species, hardhead and Sacramento pikeminnow were most frequently captured upstream of about rkm 50, but there was a subsequent decline in frequency of occurrence around rkm 80.

The common introduced centrarchids exhibited very similar patterns in frequency of occurrence (Figures 15A and 15B). All of the species were well distributed throughout the river during the summer as indicated by both electroshocking and snorkeling. The occurrence of all species declined sharply around rkm 80 with somewhat lower frequencies of occurrence observed upstream of rkm 50. Only bluegill and redear sunfish were regularly captured during winter-spring seining. The winter-spring pattern was similar to the summer pattern with the species occurring most frequently downstream of rkm 50.

The remaining common introduced species exhibited a mixture of distributions. White catfish and channel catfish commonly occurred in summer electrofishing samples at downstream sites but became rare at about rkm 60 (Figure 15B). Both species were rarely captured during snorkeling or winter-spring seining surveys. Similarly, summer snorkeling or winter-spring seining rarely captured bullhead catfish (Figure 15C). Unlike the other catfish, bullheads were less frequently captured at the upstream and downstream ends of the study area compared to the middle section between about rkm 40 and 80. Warmouth, a centrarchid (not shown in Figure 15C), showed a very similar pattern of distribution. Red shiner and inland silverside were relatively rare, but were clearly most frequently captured in the downstream reaches of the river (Figure 15C). Red shiner was not captured upstream of rkm 30 and inland silverside was never captured above rkm 50. Western mosquitofish was fairly evenly distributed along the river in the summer electrofishing survey, but was captured most frequently at downstream sites in the winter-spring seining survey. The remaining common introduced species, common carp, goldfish (not shown but similar to carp), and golden shiner occurred sporadically at certain sites along the river. All occurred rarely at sites near rkm 80 and upstream sites.

Although the data are insufficient to determine distribution in the Tuolumne River, two additional native species deserve mention. A single tule perch was captured during a summer electrofishing survey at rkm 19.8 in June 1991. Splittail was occasionally captured below rkm 30 during winter-spring seining and summer electrofishing. Single individuals were captured during seining at rkm 9.7 in March of 1988 and 1989. In May 1987, seven splittail were captured at rkm 27.7, and five were captured at rkm 5.6. A single individual was captured in a May electrofishing survey at rkm 5.6. Forty-one splittail were captured during fyke netting at rkm 9.7 in 1974.

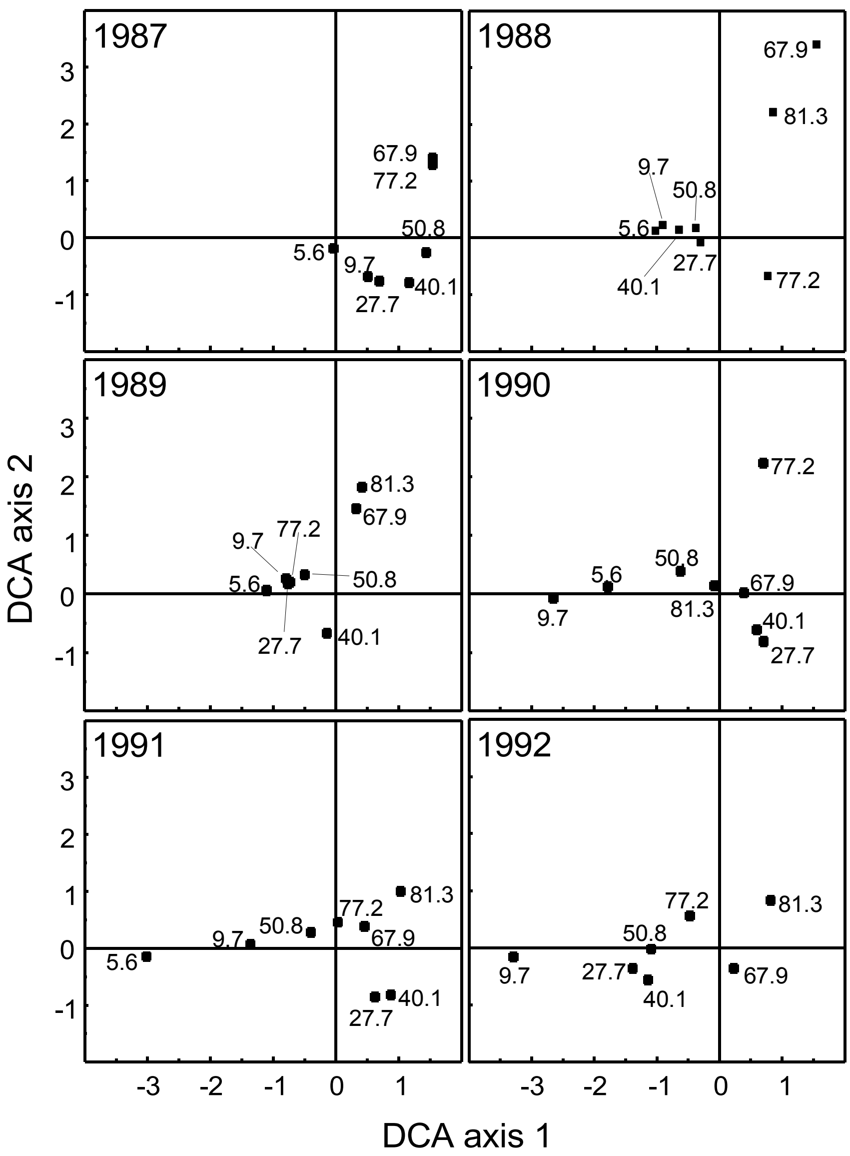


Figure 17A Site scores on DCA axes 1 and 2. Scores were derived from analysis of annual winter-spring seining data. Numbers indicate site location as kilometers from the San Joaquin River.

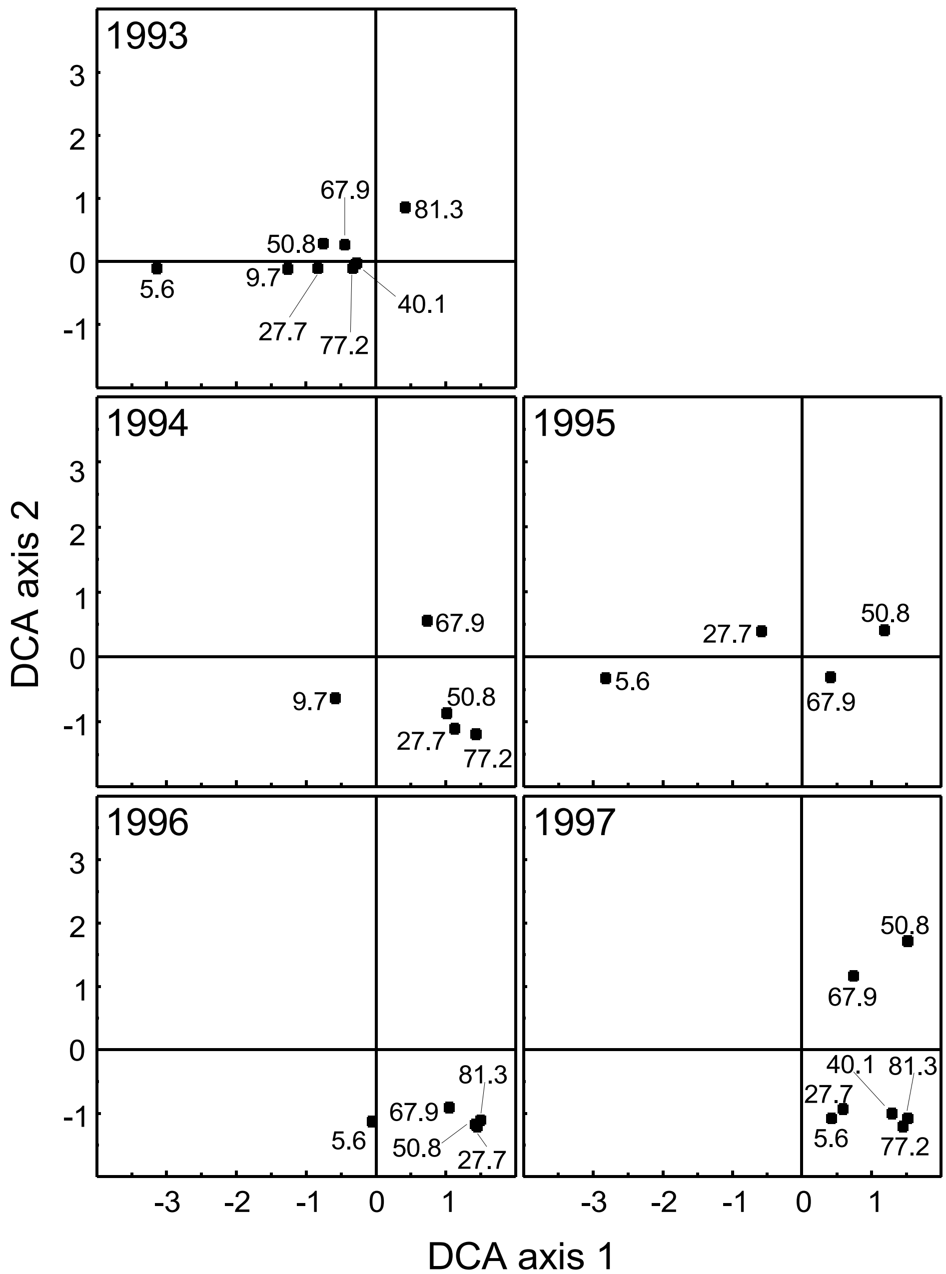


Figure 17B Site scores on DCA axes 1 and 2. Scores were derived from analysis of annual winter-spring seining data. Numbers indicate site location as kilometers from the San Joaquin River.

Fish Species Assemblages

The initial analysis of the winter-spring seining data was heavily influenced by a single sample collected at rkm 50.5 in 1987. Riffle sculpin dominated (94%) this sample. Because the high percentage of riffle sculpin was unusual compared to all other samples collected, it was omitted and the analysis conducted again.

The first four axes of the DCA of the winter-spring seining data explained a total of 51.6% of the variance in the species percentage abundances (Table 14). The distribution of species scores along DCA axis 1 suggests that the species form three groups based on similar percentage abundances (Figure 16). The native species are grouped to the right with positive scores, a large group of introduced species that occur together occurs near the center with scores between 0 and –2, and red shiner occurs alone to the left with the highest negative score. DCA axis 2 primarily separates Sacramento pikeminnow (positive score) from the two other native species (negative scores).

Table 14 Percentage of variance in species percentage abundances explained by detrended correspondence analysis of winter-spring seining data and summer survey electrofishing data

	Detrended correspondence axis			
Data set	*1*	*2*	*3*	*4*
Winter-spring seining	21.7	16.1	8.6	5.2
Summer electroshocking	26.2	9.9	5.6	4.6

The plots of site scores on DCA axes 1 and 2 indicate annual variability in winter-spring resident species assemblages (Figure 17). In 1987, all sites except rkm 5.6 were located to the right of the plot with positive scores on DCA axis 1. Native species were found at all sites, with high percentages of Sacramento pikeminnow at rkm 67.9 and 77.2. In 1988 and 1989, only sites above rkm 60 were found to the right of the plot with positive scores on DCA axis 1. The remaining sites clustered in the area of the plot characterized by the large group of introduced species with scores between 0 and –2. Western mosquitofish dominated the catch at these sites, but bluegill was commonly caught in both years and redear sunfish in 1989. Sacramento pikeminnow and suckers remained common at the upstream sites. From 1990 through 1993 the sites at rkm 5.6 and 9.7 were located to the left of the plot with the most negative scores on DCA axis 1 reflecting high percentages of red shiner. The sites above rkm 60 continued to have relatively high percentages of pikeminnow and sucker but redear sunfish became widespread resulting in a mixture of native and introduced species. Although not all sites were sampled after 1993, the assemblage appeared to shift back to the pattern seen in 1987. However, red shiner and occasionally inland silverside continued to be found in high percentages at the most downstream sites, particularly rkm 5.6. Redear sunfish became much less abundant and less frequent at the most upstream sites. The shifts in percentage abundances of the species indicated by the shifts in site scores are reflected in the annual mean percentage abundances of the three groups identified from the species plot (Table 15).

The one-way ANOVA supported the observed variability in assemblage structure. Significant differences among years were found ($P = 0.001$). Tukey HSD pairwise tests indicated that DCA axis 1 scores in 1987 were significantly higher than in 1992 and 1993 ($P < 0.05$). Similarly DCA axis 1 scores in 1997 were higher than 1993 ($P < 0.05$). The other years appear to represent transitional states between the high and low years. There were no significant differences for DCA axis 2.

Table 15 Mean percentage (± standard deviation) of species groups for all sites sampled in each year[a]

Year	*N*	*Red shiner*	*Introduced species[b]*	*Native species[c]*
1987	8	0	19.6 ± 25.4	74.7 ± 24.4
1988	8	0	66.5 ± 38.2	31.2 ± 35.5
1989	8	0.6 ± 0.7	82.0 ± 20.2	17.5 ± 20.6
1990	8	29.8 ± 29.0	54.9 ± 23.8	33.9 ± 27.7
1991	8	22.3 ± 29.2	52.1 ± 25.6	39.4 ± 32.2
1992	7	32.6 ± 46.1	72.8 ± 30.7	17.5 ± 27.0
1993	8	27.0 ± 33.4	74.1 ± 21.7	15.3 ± 15.4
1994	5	2.3 ± 3.2	26.2 ± 24.0	70.0 ± 27.4
1995	4	25.8 ± 36.5	54.7 ± 32.8	29.9 ± 35.1
1996	5	2.7 ± 3.7	12.0 ± 17.3	86.1 ± 21.1
1997	7	5.1 ± 6.4	12.4 ± 14.4	83.7 ± 18.9

[a] Means were calculated on the basis of all sites sampled (N), except for red shiner. Means for red shiner were calculated based on data from the three most downstream stations, the only sites where red shiner were captured during the study. Species groups were identified by DCA analysis of the annual winter-spring seining data.

[b] Introduced species include bluegill, largemouth bass, green sunfish, redear sunfish, smallmouth bass, white catfish, channel catfish, bullhead catfish, western mosquitofish, and common carp.

[c] Native species include Sacramento pikeminnow, hardhead, Sacramento sucker, lamprey, and riffle sculpin.

The first four axes of the DCA of the summer electrofishing data explained 46.3% of the variance in the species percentage abundance data (Table 14). Based on species scores on the first two DCA axes, the fish species appeared to form three groups (Figure 16). The native species tended to have scores near 0 with riffle sculpin clearly different with a high positive score. The other native species tended to occur in high percentage abundance with species of introduced fishes. Hardhead, Sacramento pikeminnow and lamprey were found in association with largemouth bass, bluegill, redear sunfish, and bullhead catfish. These species had positive scores on DCA axis 2. Sacramento sucker was associated with green sunfish, western mosquitofish, smallmouth bass, goldfish, carp, channel catfish, and white catfish. These species had negative scores on DCA axis 2.

Plots of site scores on the first two DCA axes indicated that summer fish assemblages were relatively stable on an annual basis but there appeared to be some seasonal variability in species assemblages at some sites in some years (Figure 18). The overall range of scores did not change dramatically from year to year, suggesting the diversity of fish assemblages was relatively constant on an annual basis. These observations were supported by results of the two-way ANOVA. For both DCA axis 1 and 2, there was no significant effect of year, season, or year-by-season interaction (all $P > 0.05$).

The sites at rkm 80.3 and 83.0 were consistently located to the right of the plot with positive scores on DCA axis 1, consistent with high percentages of native species, particularly riffle sculpin. Sites between rkm 60 and rkm 80 were generally located in the upper left quadrant of the plot with positive scores on DCA axis 2, consistent with high percentages of Sacramento pikeminnow and associated species. The remaining sites were generally located in the lower left of the plot with negative scores on DCA axis 2. Despite these general trends there were exceptions, particularly in 1992.

Comparisons of scores for the early and late samples from the same site, indicated significant seasonal changes at some sites in some years. For example, there was little change in the species assemblage at site rkm 80.3 in 1989 and 1991 but in 1990 and 1992, the site scores indicate that higher percentages of introduced species were present by late summer. The site at rkm 77.2 had similar seasonal scores in three out of four years. There was a large shift in the species assemblage only in 1989. The most seasonally stable fish assemblages were at rkm 19.8, 40.1, and 67.9.

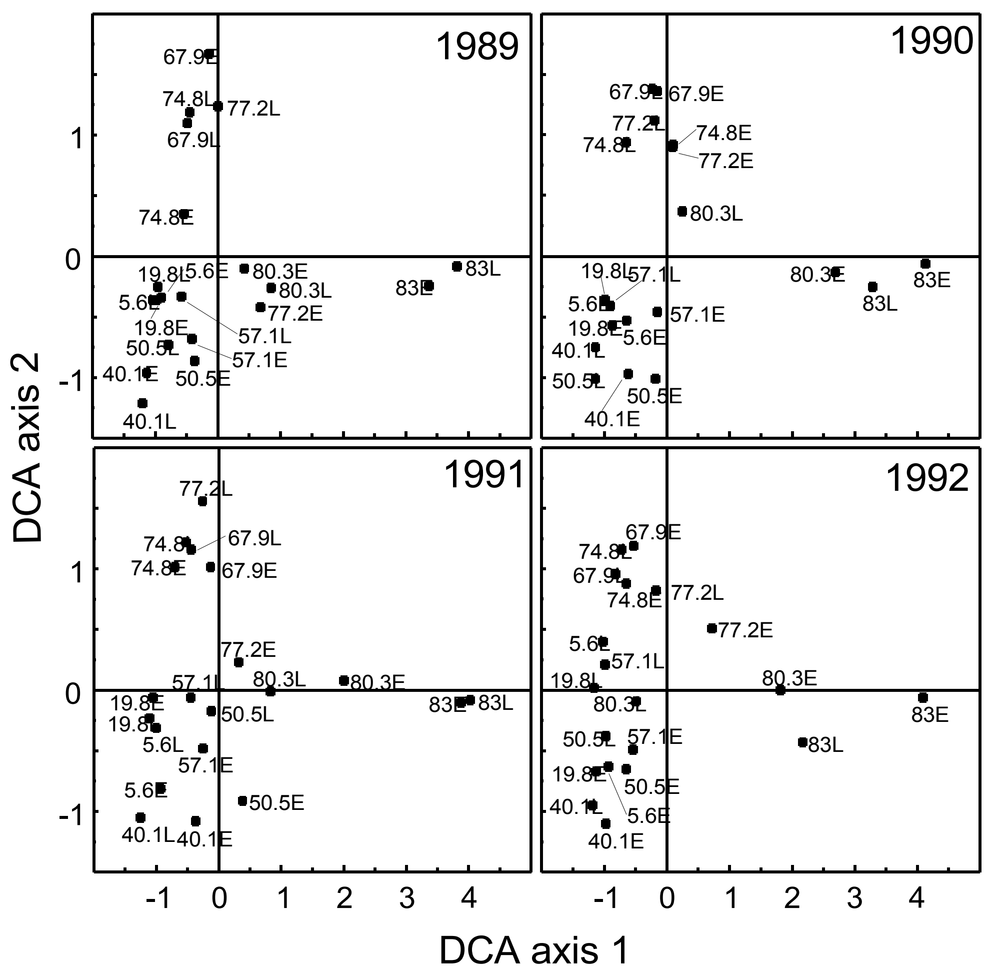


Figure 18 Site scores on DCA axes 1 and 2. Scores were derived from analysis of summer electrofishing data. Numbers indicate site location as kilometers from the San Joaquin River. The letters designate the early (E) or late (L) summer sample from each site.

Discussion

Adult Fall-run Chinook Salmon

The three years with run estimates greater than 50,000 occurred in the 1940s before completion of Friant Dam (1946) and the Tracy Pumping Plant (1951) in the Delta, both features of the Central Valley Project. The New Don Pedro Dam (1971) on the Tuolumne River and the State Water Project's Banks Pumping Plant (1968) in the Delta are other major water development factors affecting Tuolumne River salmon survival since the 1950s. Since that time, the runs have generally corresponded to overall hydrologic trends and streamflow conditions, with major declines following droughts in 1959–1961, 1976–1977, and 1987–1992. The high estimate of 40,300 in 1985 was associated with high juvenile survival in 1983, a very wet year. The effects of the ocean harvest on survival from juvenile to adult influence the trends.

The basis for the spawning run estimates has varied substantially over time, which means caution should be applied in considering their accuracy and comparability. The only direct counts were made at Modesto from 1940–1946 when fish passed over a weir. Since then all estimates are derived from carcass surveys in the upstream spawning reach. The estimates from 1947–1966 are questionable because no mark-recapture data were gathered. Carcass tagging began in 1967, but DFG estimates through 1978 are not entirely based on calculations from the tag recovery data. Methods have improved since 1979 due to the use of mark-recapture data, but various techniques and formulas have been used to calculate the population estimates and the variability of the estimates has not been fully analyzed. Expansions based on redd counts have been applied since 1981 to account for reaches not surveyed weekly for carcasses, but this was not done in prior years. The population estimates for recent years are still subject to revision as different statistical methods are applied.

The Tuolumne River is one of the few remaining major Central Valley salmon streams without a hatchery. However, hatchery salmon, as documented by the recovery of fish with CWTs, have become much more prevalent in the runs since 1986. Most of these CWT salmon originate from the Merced River Hatchery, with many returning from smolt survival releases made by DFG into the Tuolumne River. Others are mainly from releases into the Sacramento-San Joaquin Delta originating from other hatcheries in the Central Valley. The CWT recoveries represent a minimum for the hatchery salmon component of the runs because many unmarked Merced hatchery salmon are released as well. The determination of the status and dynamics of the wild population are not only complicated by the presence of hatchery fish, but the hatchery fish may also pose a threat to the long-term survival of the wild population (NMFS 1998; NRC 1996).

Juvenile Fall-run Chinook Salmon

Based on the maximum size of fry seined in January, fry began to emerge from the gravel in December in some years and continued in some years into April and May. The later fry emergence could be, in part, the result of spawning after December but there were no spawning survey data from later than 5 January since 1986. Maximum fork length data indicated that salmon >70 mm FL (potentially smolts) were present as early as March of most years.

The limited presence of salmon in the summer flow study suggests that few juvenile salmon reared for extended periods in the Tuolumne River; however, these studies were conducted during a series of low flow years and may not be representative of all conditions. The minimum summer flow requirements were increased since the sampling took place (FERC 1996) so the river reach with suitable temperatures for summertime rearing is now more extensive.

Resident Fishes

Although the results suggest the different sampling methods varied substantially in their ability to sample the resident fish communities, it is difficult to separate differences due to method from differences due to year, season, and location. Also, because the major purpose of the winter-spring sampling effort was to document the distribution and abundance of juvenile chinook salmon, sampling of the resident fish assemblage only had secondary importance. In contrast, the purpose of the summer flow study was specifically to document the resident fish assemblage.

The three winter-spring sampling methods were very successful at capturing juvenile chinook salmon but less successful at capturing other species. A number of factors likely contribute to the low catches. Low water temperatures during the winter-spring period are likely associated with reduced activity levels for most of the resident species, the majority of which are considered warm-water species. Resident fish populations are probably at their lowest abundance at this time of year due to cumulative mortality of small, young fish over the previous summer and fall. Flows are often high during the winter-spring period, increasing the size of the river, making it more difficult to sample a significant portion of the habitat. High flows are also often associated with reduced sampling efficiency because of high water velocities, greater depths, and increased debris in the river.

There were some obvious differences among the three winter-spring methods used. Fyke netting was clearly most effective for sampling bottom-oriented species, particularly catfish and lamprey (Tables 12 and 13). Rotary screw trapping emphasized pelagic species (Table 11). Seining emphasized stream-edge species, particularly western mosquitofish (Table 10). Although all the methods are somewhat biased as to the species sampled, seining has the advantage of simplicity. It is possible to sample many more locations by seining, making it possible to document species distributions as well as abundance. Electrofishing is another alternative but it was not used in winter-spring surveys and has the disadvantages of requiring expensive equipment and more likely causing mortality to captured salmon.

There were also some obvious differences among the three methods used during the summer flow study (Table 10). Seining was largely ineffective, except in capturing western mosquitofish, in the one year it was used. Presumably larger fish were able to detect and avoid the seine in the lower, clearer water present during the summer period. Electrofishing and snorkeling provided very similar data for larger more pelagic species. Snorkeling provided a more accurate assessment of large individuals, especially of the larger native species including Sacramento pikeminnow, hardhead, and Sacramento sucker. However, snorkeling tended to overlook bottom-oriented species such as catfish and sculpins and also small fishes such as red shiner and golden shiner. Snorkeling was also limited by water clarity to the upstream reaches of the river. Overall, of the three methods used, electrofishing appeared to provide the best data on the resident fish assemblage.

There are two species that were not captured, but their presence is expected based on angler reports or known occurrence in the San Joaquin River. These species are the native white sturgeon and the introduced American shad. Their absence in this data set could be due to low susceptibility to the sampling methods employed and intermittent occurrence in the river.

Fish Species Distributions

Fish species distributions, based on frequency of occurrence, were much more distinctive during the summer than during the winter-spring seining surveys (Figures 15A, 15B, and 15C). Winter-spring distributions were usually similar to the summer distributions. However, differences with river kilometer were generally of smaller magnitude because high values rarely exceeded 50% for winter-spring seining, yet were often 100% for summer electrofishing and snorkeling.

The summer sampling indicated several distribution patterns for fishes (Figure 15A, 15B, and 15C). There was a very sharp transition for many species around rkm 80. Most species (except Sacramento sucker, riffle sculpin, and lamprey), occurred much less frequently at locations upstream of about rkm 80. These most upstream locations represent a very distinct habitat. Significant broad gravel riffles dominate the reach, as do cooler water temperatures.

All three of these native species are commonly associated with such habitats in other areas of California (Moyle 1976; Moyle and others 1982).

Another transition occurs at about rkm 50 (Figure 15A, 15B, and 15C). Downstream of this point the native species occur less frequently in samples and most of the introduced species reach their maximum frequency of occurrence. This location approximately corresponds to a reach of river that has been significantly affected by gravel mining. The gravel pits serve as a velocity refuge during high flows for many of the introduced species found in the river. When flows decrease the introduced species can re-invade both upstream and downstream areas. The area between rkm 50 and rkm 80 represents an area of overlap between the areas dominated by native and introduced species.

Red shiner, inland silverside, and golden shiner exhibited another pattern of distribution. These species were most commonly found at the most downstream stations. These results are consistent with Brown (2000) who described the former two species as San Joaquin River mainstem species because they were most abundant in that river and only entered tributaries such as the Tuolumne River for short distances. These results were interpreted to indicate that these species consistently invade the tributaries and perhaps maintain populations there but conditions in more upstream areas are unfavorable in some way. Brown (2000) did not capture golden shiners in his study (sampling 1993–1995), suggesting a different process may be occurring for this species.

The data on splittail and tule perch indicate that other native species do occasionally make their way into the Tuolumne River. The data on splittail were particularly interesting because previously published studies of fishes in the San Joaquin River drainage indicated splittail only occurred rarely in the system (Saiki 1984; Brown and Moyle 1993). Sommer and others (1997) noted those studies were based on summer sampling. It appears that splittail move into the upper San Joaquin River to spawn in some years (Sommer and others 1997) and that either additional spawning or young-of-year rearing occurs in the lower reaches of the tributary rivers including the Tuolumne River. Brown (2000) captured young-of-year splittail in the lower reaches of both the Tuolumne and Merced rivers in 1995. Brown (2000) found tule perch to be abundant in the Stanislaus River but not in the mainstem San Joaquin River or the other tributaries. Saiki (1984) observed tule perch in the San Joaquin River but did not sample the tributaries extensively. Brown (2000) suggested that the high summer flows in the Stanislaus River combined with extensive beds of aquatic vegetation provided a type of habitat not widely available in other streams in the lower San Joaquin River drainage.

Fish Species Assemblages

No other long-term data sets are available for winter-spring resident fish assemblages in the San Joaquin River system (Brown 1997), making this data set unusual. The results of the DCA indicate that there is significant annual variability in the winter-spring resident fish assemblage that appears to be related to flow conditions. Examination of daily flow records suggests high percentages of native species are associated with high stream discharge in the winter of the previous year. Native species dominated in 1987 after the wet winter of 1985–1986. Introduced species became more dominant during the drought (1988–1992) with native species returning to high percentages at many sites in 1994 after the wet winter of 1993–1994. Native species continually occurred in high percentages starting in 1996 after the wet winter of 1994–1995.

The mechanism causing this switching is unclear. The native species are all riffle spawners and many of the introduced species are nesting species (Moyle 1976). It is likely that high outflows provide more appropriate spawning conditions for the native riffle spawners and poorer conditions for the introduced nesters. A number of recent analyses has suggested that natural flow regimes, including high winter-spring discharges, benefit native California stream species over introduced species (Baltz and Moyle 1993; Moyle and Light 1996a, 1996b; Brown and Moyle 1997). The spawning success hypothesis also explains why winter-spring assemblage structure lags behind the wet winter by a year. The bulk of the seining occurs before or during the spawning seasons of the majority of the resident species. The effect is seen in the seining data the following year, after the young have become large enough to be susceptible to the seine.

Another complication is the importance of red shiner in the analysis (Figure 16). Red shiner is a recent introduction and the species was actively invading the San Joaquin River system in 1986 (Jennings and Saiki 1990). It is likely that the invasion process is complete (Brown 2000); however, there are no conclusive data to that effect. It is unknown if the same patterns of annual change would be apparent in the absence of red shiner; however, it seems likely that inland silverside, which exhibits a similar pattern in frequency of occurrence (Figure 15C), might assume similar importance in the absence of red shiner.

The summer resident fish assemblage did not exhibit significant annual change, but the data were not as extensive as the winter-spring seining data, being limited to four years during the 1987–1992 drought. There was also little change in the winter-spring assemblage during the years (1989–1992) of summer sampling (compare Figures 17 and 18). Brown's study (2000) did include years with very different flow conditions and there were obvious differences in the summer fish assemblages. In the wet year (1995), native species were present in downstream areas where they were absent or very rare during drier years (1993 and 1994). Despite the inability to use data from the present study to look at changes with flow conditions, the analysis did indicate some interesting patterns within the period analyzed.

In contrast to the winter-spring data, red shiner was only a minor component of the summer assemblage. As noted, this is consistent with Brown's (2000) observation that red shiner was rarely found in the large tributary rivers (Merced, Tuolumne, and Stanislaus rivers) to the San Joaquin River. Brown (2000) hypothesized that the low, clear water conditions prevalent in the tributaries during the summer are favorable for predators, resulting in heavy predation on red shiners that moved upstream during the winter and spring. Thus, the distribution of red shiners is a balance of invasion and predation mortality processes.

Native and introduced species appear to be more closely associated during the summer than during the winter and spring, with the exception of riffle sculpin (Figure 16). Riffle sculpin were found in high percentages at the most upstream sites probably for two reasons. The gravel riffle habitat they were associated with is most abundant in the most upstream areas and water temperatures are coolest there. Temperature has been found to limit the downstream distribution of riffle sculpin in other Central Valley streams (Baltz and others 1982).

The other native species were closely associated with introduced species (Figure 16). This is unusual compared to the Merced and Stanislaus rivers. Multivariate analyses presented in Brown (2000) indicate a close association of native and introduced species in the Tuolumne River, but in the Merced and the Stanislaus rivers, the most upstream sites were clearly dominated by native species. This difference may be related to the summer flow regimes and water diversion practices in the two rivers. In the Merced River, the native species dominate the river upstream of a series of diversion dams, but introduced species dominate downstream of the diversions. Flows in the Stanislaus River are relatively high all summer because of upstream releases to control water quality in the San Joaquin River and native species are dominant at several upstream sites. In the Tuolumne River, the major diversions are made at La Grange Dam with summer releases being relatively small (particularly during the period of study), and introduced species were present throughout the system. These results are also consistent with the hypothesis described earlier that natural hydrologic patterns appear to favor the native species (Baltz and Moyle 1993; Moyle and Light 1996a, 1996b; Brown and Moyle 1997). The recent implementation of new minimum summer flow requirements (FERC 1996) may change the pattern to one more similar to that observed in the other tributaries.

The comparisons between early and late samples indicate that significant changes can occur in resident fish assemblages over the course of the summer (Figure 18). It is unclear what process is causing these changes. There may simply be random events due to immigration and emigration. Changes might also result from physical or biological processes such as temperature avoidance as the river warms during the summer or competition or predation among species as low summer flows concentrate fishes into limited depth and cover refugia. More detailed field and laboratory work is necessary to clarify such processes and their interactions.

Monitoring of the resident fish community provides useful data on the effects of flow conditions on the river ecosystem. Continuation of the documentation of resident fishes in the winter-spring seining will provide a long-term database unmatched in any other Central Valley stream. Resumption of annual monitoring of summer fish assemblages could provide useful data on the positive or negative effects of changes in water management activities on native species of interest. Though resident species often appear to be of little management interest in the short term, they can often become critically important when populations reach low levels and threatened or endangered status becomes a possibility. The splittail, recently listed as a federal threatened species, is a good example. Effective monitoring of all species seems a worthwhile investment to reduce future uncertainty in management concerns.

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Building Models and Gathering Data: Can We Do This Better?

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Abstract

We are constructing a "second generation" model of chinook salmon for the Sacramento Basin to help investigate factors affecting salmon populations and the effects of management actions. We chose to build a new model rather than modify an older one to apply recent developments in computer interfaces and individual-based modeling and to incorporate a more detailed and flexible geographic representation. We also expected that substantial new knowledge had been developed that would enable us better to characterize the life cycle and influences on survival of chinook salmon. These expectations have not been met, and despite some recent progress we still find gaps between the knowledge available and that needed for successful modeling. Key examples of gaps in our knowledge include sublethal temperature effects, abundance of young fish, factors triggering migration, factors limiting rearing habitat, and survival of young salmon, particularly fry rearing in the mainstem or Delta reaches and early survival in the ocean. We believe these gaps arise for several reasons: (1) a mismatch in perceptions of what data are needed; (2) a lack of institutional commitment to long-term, broad-scale programs to provide knowledge useful in modeling; and (3) the fundamental difficulty of gathering information about environmental influences on fish populations.

Introduction

Models are representations of real-world objects or systems. Simulation models are formal mathematical representations of dynamic systems developed to examine the time course of system response to selected inputs. These models can be used as research or management tools or, if the underlying mathematics and parameters are known well enough, for prediction. Models of ecological systems are rarely suitable for prediction. Simulation models can be useful in investigating properties of a complex system, but are also useful as a framework for organizing knowledge and identifying knowledge gaps.

We are in the second phase of development of a chinook salmon simulation model for the Sacramento Basin. In the first phase we developed a conceptual model which has been distributed for review. In the second phase we are receiving and discussing comments on the conceptual model, while developing the code for the simulation model with the initial goal of producing a working prototype.

In this paper we briefly describe the model and some of its potential uses. We then discuss the more significant gaps in knowledge that have been identified during model development. Some of these gaps have been known for many years, yet little progress has so far been made to close them. We discuss some possible reasons for this and potential remedies that could lead to more effective allocation of effort devoted to research and monitoring on salmon biology and better understanding of the effects of human actions on salmon life histories.

Background

The Central Valley Project Improvement Act (CVPIA), an ambitious effort to increase production of chinook salmon in the Central Valley, mandates the development of "ecosystem models" to support understanding of potential measures needed to restore anadromous fisheries. The model described here is an element of the ecosystem modeling effort designed to assist with analyses and comparison of various alternatives for water and fisheries management. It is intended to build on both current understanding of the ecology of salmon and experiences of previous modeling efforts for chinook salmon in the Central Valley. These efforts include the following:

1. A simple stock-recruit model used to investigate effects of Delta conditions (Kelley and others 1986)

2. CPOP, a cohort simulation model of Sacramento Basin fall-run or winter-run chinook salmon (Kimmerer and others 1989), written in Fortran.

3. EACH, a simulation model for the San Joaquin Basin with similar structure to CPOP, written in Stella (EA Engineering, Science, and Technology 1991).

4. Two statistical models of the effects of Delta conditions on San Joaquin Basin chinook salmon (Speed 1993; Rein 1994: http://felix.vcu.edu/~srein/chinook.ASA/talk.html).

5. An individual-based simulation model of chinook salmon smolt production in the Tuolumne River (Jager and others 1996).

6. Statistical models of mark-recapture experiments using salmon smolts in the Delta (Kjelson and others 1982; Baker and others 1995; Rice and Newman 1997).

7. The CRiSP model of chinook salmon smolt passage, originally developed for the Columbia River by the University of Washington, modified for the Sacramento River in a student paper (http://www.cqs.washington.edu/papers/sacramento.html).

8. A survival model of winter-run salmon (Botsford and Brittnacher 1996).

We refer to our model as a "second-generation" model, because it builds on the results of previous modeling efforts. This model differs from previous models: it applies to all races in the Sacramento Basin; uses an individual-based approach; takes input from a variety of data sources, including flow and temperature data or model output; is designed in modules to simplify analyses of selected stages of the life cycle; will have a modern user interface so users can spend their time learning about the model rather than the program; and is being programmed in an object-oriented language that will make future modifications relatively straightforward.

The model is essentially a large combination of conditional statements about the salmon population. It contains various mathematical descriptions of attributes of habitat and individual fish, which determine responses of salmon to their environment. Many of the mathematical descriptions and the parameters and input variables used to develop numerical values for responses are based on limited data or expert opinion. Thus, it is extremely unlikely that all of them are accurate, so output of the model is not reliable as a prediction of future salmon population trends. Rather, the model will be most useful in a comparison among alternative scenarios. Provision will be made for varying important parameters and selecting alternative mathematical descriptions of functional relationships to determine the sensitivity of the conclusions based on model runs to the assumptions contained in the model.

Model Description

The model is capable of simulating the entire life cycle of all four races of chinook salmon in the Sacramento Basin (Figure 1). Conceptually the model can be divided into the four modules shown in the figure. Individual modules, corresponding to stages of the life cycle, can be run independently to simplify the model run for particular purposes. This will be a useful feature for investigating particular aspects of the life cycle such as spawning or ocean life.

We have chosen to use an individual-based modeling approach (DeAngelis and Gross 1995). Individual-based models (IBMs), also known as agent-based or multi-agent models, are a relatively recent development in modeling made possible by substantial advances in computer memory and speed. In an IBM, populations are represented by some number of individual entities, rather than by cohorts or other aggregates. Models written at the cohort or higher levels of aggregation have many advantages, but they do not accurately portray the population response to environmental change when the individuals in a cohort undergo different trajectories of growth or movement. This can happen when, for example, physical habitat is occupied at a small scale so that different fish experience different environments. A cohort model also suffers from the disadvantage that any nonlinear response of the fish to their environment distorts the statistical distribution of properties within the cohort (e.g., mean weight). Finally, some environmental influences act on individuals over a long period relative to the simulated time step; resolving variable temporal influences can be very complicated in a cohort or similar model.

In an IBM, there is no difficulty resolving whatever level of spatial or temporal resolution is of interest, and heterogeneity at the selected level of resolution is incorporated explicitly in the model. Any environmental influence requiring a "memory" of past conditions (e.g., thermal or toxic stress, feeding history) is easily represented. Nonlinearities in responses do not result in distortion of distributions of properties. Events occurring at the individual level, such as movement, growth, or death, are summed to arrive at the population response.

There are significant advantages in the individual-based approach: clarity and consistency of logic; unambiguous "currency" of the model (i.e., individual fish); ease of tracking movements and adding new features (e.g., energetic and genetic effects, interactions); ease of accumulating effects of past conditions (e.g., toxic body burden and condition factor); and straightforward simulation of responses to a spatially and temporally heterogeneous environment.

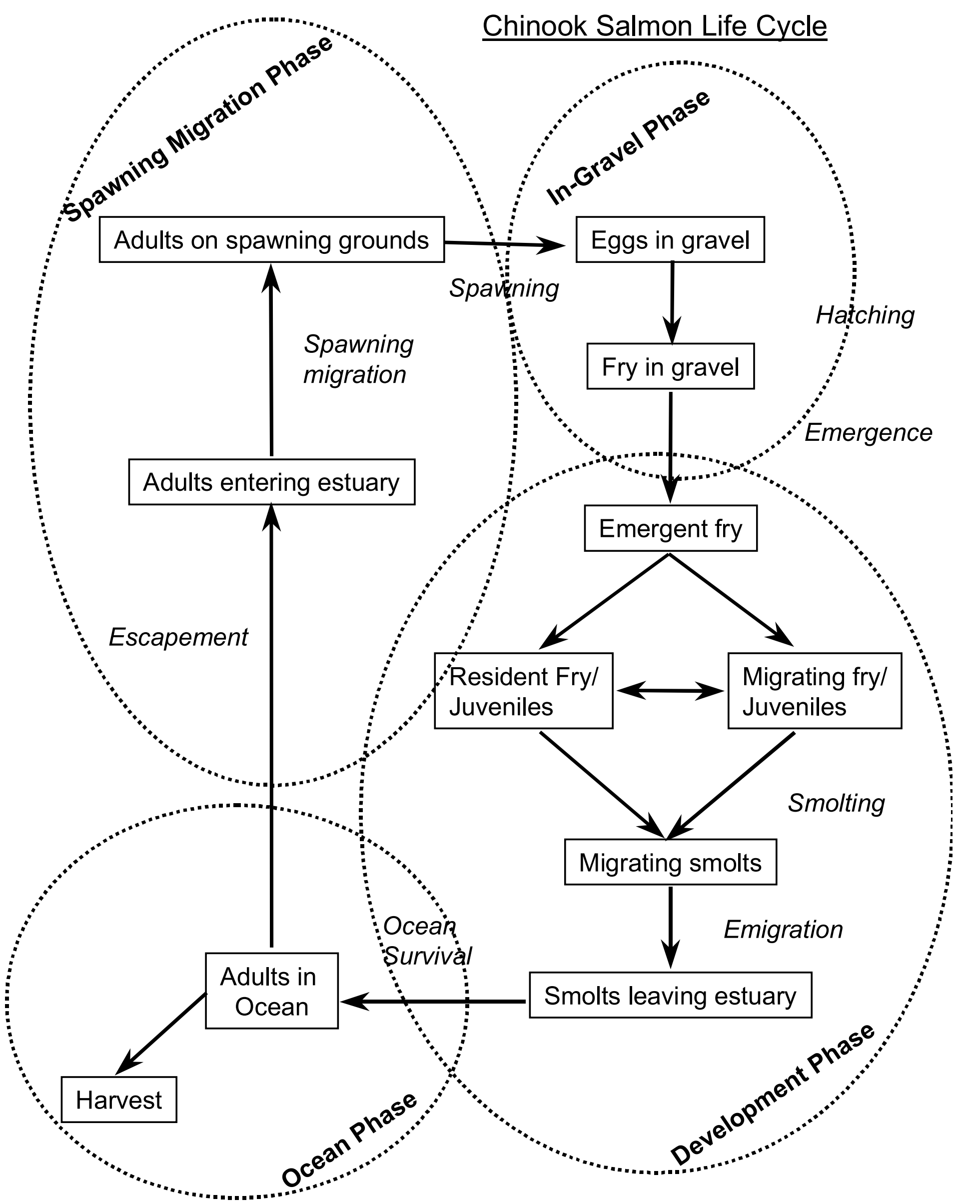

Figure 1 Key points in the life cycle of chinook salmon. The four oval areas represent the major life stages, represented by separate modules in the model. Arrows indicate a change of state of surviving salmon, with ocean harvest represented explicitly but other mortality not shown. Terms in italics indicate major life history events occurring in each stage.

The principal disadvantage of an IBM is that it is computationally intensive, and the computer power needed to run the model can be difficult to predict. Furthermore, simulating explicitly the hundreds of millions of fall-run juveniles in the Sacramento Basin would make the model unwieldy even with the fastest available computers. Therefore, the populations will be represented by a sample of the actual fish, and each model fish will be a "super-individual" representing some number of individuals (Scheffer and others 1995). This method, which is analogous to stratified sampling in opinion polls, should provide equivalent results to modeling every individual but at a manageable cost in computer time. It may be superior to the resampling method of Rose and others (1993), which can introduce bias if the number of model fish is too low (Scheffer and others 1995). It is also different from cohort modeling because sufficient numbers of sample fish are tracked to represent adequately the full range of variability in the population. The ratio of model to actual fish can be varied among life stages and races to keep the sample size large. Thus, abundant fall-run fry will be represented at a fairly small ratio of model to actual fish, while winter-run (and initially all) adults will be represented at a ratio of 1:1. Preliminary testing will ensure that the ratio selected does not bias the results. Clearly the selection of these ratios will represent a compromise between the speed at which the model runs and the amount of bias or error due to aggregation, and can change with type of model run and available computer power.

The individual-based approach lends itself directly to the use of object-oriented programming methods. In contrast to procedural languages (e.g., FORTRAN, C), an object-oriented language isolates elements of the program as "objects" which pass, receive, and respond to "messages." Objects may be any element of the program, but are most useful when they bear direct relationships to real objects, such as fish, river reaches, or computer windows. Thus, there is a direct correspondence between individuals in the model and objects in the program, making the transition from conceptual model to computer program as straightforward as possible. We have chosen to use the Swarm software package for multi-agent simulation of complex systems, developed by the Santa Fe Institute. This package comes with several ready-made objects and tools for input and analysis that will simplify coding and testing the model.

As noted previously, we have developed a draft conceptual model (Kimmerer and Jones & Stokes Associates 1998) and an annotated bibliography. We are proceeding on three parallel tracks in model development: (1) refining the conceptual model based on comments received, discussions with interested parties, and experience with submodels; (2) assessing the data available for model input; and (3) developing a model formulation in Swarm focusing initially on in-river life stages.

Principal Gaps in Knowledge

Significant advances have been made in understanding the biology of Sacramento Basin salmon since the previous population models were developed. However, our assessment of the available information gives little encouragement that the principal gaps have been filled. Although the model can be used to some degree to explore the consequences of different assumptions about these gaps, a lack of solid understanding may restrict use of the model for management purposes.

It is relatively easy to identify knowledge gaps, and several key ones are discussed below. However, a significant problem we have encountered in attempting to fill these gaps is that relatively few of the existing data are in the form of published reports or articles. Much of the information is either anecdotal, or has not yet been published or widely disseminated. Some data are presented in technical reports, but the data are not made available to the research community on a timely basis.

Thermal Effects Below Lethal Limits. When temperature exceeds lethal limits, mortality is expected to be rapid, but results of mark-recapture experiments in the Delta suggest effects at temperatures below these limits (Kjelson and Brandes 1989; Rice and Newman 1997). Although these effects could be artifacts from the use of hatchery smolts or other aspects of the experiments, it is also likely that similar effects apply to naturally-reared fish. If so, similar temperature effects should occur throughout the river system. They may arise through physiological changes that affect growth, disease resistance, predator avoidance, and smolting, through ecological effects such as increased predator activity or increased food requirement without an increase in supply, or through a combination of these effects. Since temperature in the system often varies within the range at which these effects seem to occur, these effects may be important influences on survival of young salmon. Available information on thermal effects, however, is largely confined to laboratory experiments on temperature above lethal limits, with abundant food (e.g., Brett 1952). The potential effects listed above remain virtually unexamined.

Abundance of Young Fish. There are reasonably good estimates of adult and redd abundance, although abundance of some adults has become more difficult to determine with the revised operation of the Red Bluff Diversion Dam, where dam gates are open during most of the upstream migration period. However, estimates of fry or smolt abundance in the rivers are uncommon, and although the data are available, estimates of abundance have not been made for the Delta. Many measurements of abundance in the river system provide only indices rather than actual abundance values. The problem is for many measures of abundance, no suitable method has been developed to calibrate the measures to the actual number of fish passing a point or residing in an area. Although these indices are adequate for comparing abundance data among years and investigating effects of local restoration actions, they fall short of the data needed to develop a comprehensive view of the salmon population. In particular, mortality values, essential for assessing population status, require accurate abundance estimates.

Availability of Rearing Habitat. Recent data suggest that most of the young salmon in some of the rivers leave their natal streams shortly after emergence (Snider and others 1997). Furthermore, beach-seining data show large numbers of salmon fry in the lower Sacramento River and the Delta (Kjelson and Raquel 1981; Brandes and McLain, this volume). This implies the existence of two very different life histories, that is, fish that rear largely in the natal streams and those that rear mostly downstream. The relative contribution to recruitment by these life histories needs to be assessed, and some effort needs to be made to determine the factors that induce the young salmon to migrate as early fry instead of rearing in the natal streams. This may relate to the carrying capacity of different parts of the system for rearing salmon, which may be a key element in density dependence and therefore population regulation (for example, Elliott 1989). However, existing data are insufficient to assess the importance of rearing in the natal river compared with the mainstem Sacramento River and Delta, the factors influencing the availability of rearing habitat, or the factors that stimulate movement of pre-smolt salmon. The principal issue is where and under what conditions the extent or quality of physical habitat limits the abundance or survival of rearing salmon. Although the model may be useful in testing the outcome of alternative conceptual models about rearing habitat, the ultimate answer to its importance must be obtained through hypothesis-driven field research. The importance of rearing habitat has obvious, large implications for the success of alternative restoration actions.

Survival of Young Salmon. A related issue about which little is known is survival during early life. Survival through hatching and emergence is at least qualitatively understood to be high except in cases of extreme changes in flow or high temperature. However, survival during rearing, seaward migration, and early ocean life is unknown, except for survival indices for smolts passing through the Delta. The location of rearing may have a big effect on survival: for example, density-dependent migration out of the natal stream combined with lower, density-independent survival in the Delta would result in density-dependent survival. Little is known about the influence of food supply on survival, nor is there good information on the abundance and activity of predators. Finally, the occurrence and locus of density dependence, a crucial ecological feedback to any biological population, is unknown; previous studies have shown evidence of density dependence in young salmonids both in streams (Neilson and Banford 1983; Elliott 1989) and in the ocean (Peterman 1984).

Filling these knowledge gaps will not be easy. Most of them would require a coordinated effort involving a variety of agencies and a long time frame. However, without this information the effects of restoration actions will be difficult to predict, and therefore the actions will be difficult to justify.

Filling the Knowledge Gaps

Why are these information gaps still present? We do not wish to understate the difficulty of gathering the kind of knowledge described above, nor to denigrate the efforts of the biologists investigating Central Valley salmon. Much of the difficulty lies with the complexity of the ecosystem and the populations to be investigated. Nevertheless, we believe there are some key impediments to filling these knowledge gaps, and removing or reducing these impediments may improve the rate at which the gaps are filled.

The first impediment is a mismatch in perception among modelers, fish biologists, and managers about what data are needed and how to use a model. Modelers tend to focus on the "big picture," with less attention to details and a tendency for excessive generalization. Fish biologists tend to have a deeper understanding of certain topics, but a narrower view, often constrained by their experience to certain aspects of geography or life history. Understandably, many fish biologists tend to view data needs in terms of their own research experience. Many managers prefer not to hear about uncertainty and tend to rely heavily on expert opinion or on well-presented (usually conceptual or statistical) models. Although managers often support status and trends monitoring, they may see little need for research aimed at fundamental questions, which can be expensive and risky. The perspectives of these three groups do not lend themselves to a coordinated attack on the key problems, because each group sees the key issues differently.

The second impediment is what we see as a lack of institutional commitment to resolving system-level uncertainties. Much of the work being done by fish biologists and other scientists in the system is focused on particular exigencies, mostly related to endangered-species protection. Thus, little time is available for consideration of larger issues. There is no agency whose mission is solely to investigate and understand the biology of salmon and the influences on it. Each of the resource agencies has significant other duties, particularly environmental or endangered-species protection, that may actually impede progress toward understanding at a system level. This impediment has been evident in the resistance of some agency biologists to adaptive management experiments designed to determine the effects of certain management actions on salmon populations when the experimental actions were seen as potentially (but not demonstrably) harmful.

There also seems to be a strong degree of territoriality in the Central Valley salmon biology field. Although the situation is improving (for example, with this Symposium), there is still a remarkable lack of collaboration among researchers. This situation is particularly alarming given the amount of work being done at public expense and the importance of salmon to the Central Valley's ecosystems and economy.

Several potential approaches may help to resolve these issues. The most direct is individual commitment by fish biologists to consider the "big picture" in what they do on a daily basis and to continually re-evaluate their contribution. Although such a commitment would seem consistent with the role and activities of scientists, it would be naive to expect individual scientists to deviate much from their immediate interests to the common good, at least without added incentive.

This indicates a need for institutional commitment to working toward answering large-scale questions. This commitment could be underwritten by one of the larger organizations (e.g., CALFED Bay-Delta Program, Comprehensive Monitoring, Assessment and Research Program, Interagency Ecological Program), but the individual agencies would still have to support the contributions of their own fish biologists to the larger view. This may be seen as contrary to the mission of resource agencies, which have immediate responsibilities for endangered-species protection and other activities that may preclude devoting adequate attention to large-scale issues. One mechanism for enlarging the view of agency biologists is to make publication in peer-reviewed journals a criterion for promotion. The process of preparing a paper and getting it through the review process is an excellent way of helping a researcher to put his or her work in a larger context.

An alternative method for filling gaps is to establish a small, dynamic research team whose sole mission would be to gather, analyze, and publish data specifically related to population-level issues. This team could be given the mandate to collect data from other researchers, and to initiate field research projects into areas outside of the interests of other agencies. Mechanisms would have to be established to ensure cooperation by agency biologists, and reciprocally to ensure partnerships between members of this team and agency biologists.

An additional aid to filling in knowledge gaps is to make data freely available. Although data are routinely published in annual and other reports, these data are not readily available to other researchers. Identifying and obtaining data has been one of the most time-consuming and frustrating activities in our modeling work. These data have been collected by public agencies with public funds, and the maximum possible use should be made of them. The prerogative of the investigators to publish their results can be upheld through a delay time of no more than one year from the date of collection to the date at which the data are made available on an Internet site. The salmon monitoring and research community would do well to follow the lead of the Interagency Ecological Program in terms of data dissemination and availability.

Regardless of the mechanism used, we urge managers and biologists to consider seriously the need for better use of the available information, better mechanisms for determining what information is gathered, and research targeted at a more comprehensive view of the biology and population dynamics of salmon.

In our model development to date, we have found it easy to identify significant gaps in the knowledge about salmon, as discussed above. No model runs were necessary to convince us that the gaps are serious impediments to understanding the complete life cycle of chinook salmon. As the simulation model is developed, we anticipate using sensitivity analysis to further delineate where significant gaps occur, and possibly to develop methods for filling the gaps. We hope that as this work progresses some of the impediments to knowledge discussed above can be removed, and progress can be made toward filling the gaps.

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Exploring the Role of Captive Broodstock Programs in Salmon Restoration

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Abstract

Severe population declines have occurred in many Pacific salmon stocks. Stock declines have been attributed to both anthropogenic and natural environmental causes. These declines have been so dramatic that resource agencies have not had the time or means to quantitatively describe stocks and develop rapid, reliable methods of conserving rare genes. One method to prevent extinction is gene banking by means of rearing broodstock in captivity for use in supplementing rare and endangered stocks. With varying degrees of success, several captive breeding programs have been initiated to provide "insurance" against genetic loss of imperiled stocks. Captive breeding is expensive, requiring long-term intensive fish husbandry. It is not an alternative to habitat restoration. In certain situations, such as small runs (20 to 100 spawning adults) combined with habitat undergoing restoration, captive breeding may be a desirable supplementation strategy. It is certainly a beneficial option for any stock on the edge of extinction. There are several salmon captive broodstock programs on the west coast of North America, each employing different approaches and technologies. Captive breeding techniques are evolved to a point where the progeny of wild fish can be reared with a high degree of success. However, this kind of intervention is costly and must be weighed against other factors that will determine stock recovery. It is incumbent upon managers and scientists to define the uncertainty, or risk, of captive breeding. Risk assessment is an essential component of any captive breeding program. Emerging captive breeding programs can benefit from the range of experience and technological development that has evolved over the past decade. Molecular genetics, captive broodstock technology, conservation biology, and fisheries supplementation risk assessment have matured to a stage where salmonid captive breeding can be planned as an intervention with a measured effect.

Captive Breeding as a Response to Declining Salmon Stocks

During the last four hundred years, approximately 490 described animal species are known to have become extinct (Magin and others 1994). Approximately 24,600 species of fish (in 482 families) exist worldwide, although this number may reach 28,500 as more species are described (Nelson 1994). The International Union for the Conservation of Nature (IUCN) has compiled a list of threatened or extinct fish species, documenting the downward trend in aquatic biodiversity (IUCN 1996). Moyle and Leidy (1992) estimated that 20% of the freshwater fishes of the world is at risk of extinction, yet this figure is likely very conservative (Leidy and Moyle 1998).

Anadromous salmonids, including many stocks of Pacific salmon along the west coast of North America, have experienced severe population declines. The Northwest Power Planning Council (1987) reported that annual returns of anadromous salmon and trout decreased from an estimated 12 to 16 million in the 1880s to 2.5 million fish in the 1980s. At least 106 major populations of salmon and steelhead have been extirpated from the West Coast (Nehlsen and others 1991). Nehlsen and others (1991) identified 214 stocks of Pacific salmonids from California, Oregon, Idaho, and Washington that face a high or moderate risk of extinction. Stock declines have been attributed to both anthropogenic and natural causes including land-use practices such as urbanization and logging, reduction of genetic diversity in native stocks and introduction of disease through hatchery production, overharvest, and flood and drought events (Nehlsen and others 1991; Pearcy 1992; NRC 1996).

The role of hatcheries in the conservation of wild salmon populations depends upon a keen appreciation of the reproductive interactions among and between hatchery and wild salmon (Fleming 1994). True "gene banking" efforts based on sperm cryopreservation to avoid the loss of valuable genotypes have been initiated for Snake River sockeye salmon (*Oncorhynchus nerka*) and chinook salmon (*O. tshawytscha*) in the Columbia River of the northwestern United States (Thorgaard and others 1998). Captive breeding can be considered a form of gene banking in that it relies on the captive rearing of the living genetic resource. Unlike conventional salmon hatcheries that rear animals to fry or smolts in single cohorts, captive breeding requires the maintenance of multiple age classes, in numerous family groups, to maturation. As such, captive breeding programs are costly and labor intensive.

Captive propagation is becoming accepted as one component of species enhancement (Gipps 1991; Johnson and Jensen 1991; Olney and others 1994). For example, the U.S. Fish and Wildlife Service uses captive propagation to enhance populations of nearly 30% of the non-anadromous North American fish species listed under the federal Endangered Species Act (USFWS 1990;

Johnson and Jensen 1991). With varying degrees of success, several captive breeding programs have been initiated to provide "insurance" against genetic loss of imperiled stocks. While captive breeding may be less cost-effective in the long-term than *in situ* preservation, it may provide the only mechanism to prevent extirpation of a stock, especially before or during the early implementation of an environmental recovery program. Indeed, the National Research Council (1996) now recognizes that long-term sustainability requires conservation of both wild populations and their natural habitats. Ecosystem-wide approaches are beginning to be recognized and adopted on both the theoretical and practical levels.

One aspect of the ecosystem approach to salmon restoration that is gaining attention is the role of salmon in the regeneration of forest-stream systems (Bilby and others 1996, National Research Council 1996). It is possible, given the multiple pathways salmon create for marine-derived nutrients to enter watersheds, that there is a critical abundance threshold necessary to stabilize runs. The precipitous stock declines witnessed during the past twenty years are likely some combination of ecosystem and population effects. If this is the case then supplementation becomes a more important part of the restoration equation.

Captive breeding may entail *in situ* gene banking ("insurance" only) or it may include a supplementation to the watershed. In the case of salmon and anadromous trout, captive breeding typically involves the propagation and early life stage rearing of a stock with subsequent release at the fry, parr, or smolt stage. Snyder and others (1996) described several limitations of captive breeding in endangered species recovery and asserted that it should be viewed as a last resort to avoid species extinction instead of a prophylactic or long-term solution. Artificial propagation in itself is not the remedy to stock declines. On the contrary, it may even contribute to the decline of native populations (Goodman 1990), risking further loss of genetic resources (Waples 1991). Case-specific economic, biological, and conservation-related variables must be considered in determining the appropriateness of captive propagation for a particular species (Balmford and others 1995; Snyder and others 1996). For example, *ex situ* conservation for the purpose of supplementing wild stocks depends on successful reintroduction, which in turn depends on the availability of suitable habitat (Griffith and others 1989; Wilson and Stanley Price 1994). For threatened and endangered species, artificial propagation and release may not assist in their recovery, particularly in instances where population declines are the result of altered or unsuitable habitat for self-sustaining reproduction. In the case of natural salmon populations, supplementation is appropriate in two scenarios: (1) when short-term extinction risk for the population is high, and (2) in re-seeding vacant habitat that is unlikely to be colonized naturally within a reasonable time frame (Robin S. Waples, personal communication, see "Notes").

Surprisingly, Balmford and others (1996) found that existing captive breeding efforts in zoos for mammals failed to focus on species subject to potentially reversible pressures such as overexploitation or small-scale habitat deterioration. Captive breeding efforts for fish have received similar criticism. For Pacific salmon, the use of hatchery techniques in conservation has been criticized as being a "halfway technology" since supplementation of wild stocks with hatchery produced fish addresses a symptom (declining fish stocks) but not the causes (Meffe 1992). The World Conservation Union's Conservation Breeding Specialist Group (CBSG) has developed a series of Conservation Assessment and Management Plans (CAMPs) calling for long-term captive breeding of numerous taxa (Seal and others 1994). In 1993, Tear and others reported that of the current 314 approved recovery plans for U.S. endangered and threatened wildlife, 64% recommended captive breeding. In the case of salmonids, the Forest Ecosystem Management Assessment Team (FEMAT) identified 314 native stocks as being threatened with extinction (FEMAT 1993). Yet with only limited resources for conservation and recovery measures, Allendorf and others (1997) asserted that priorities should be established for stocks which are candidates for preservation. Limited resources dictate that only a few stocks can be identified for intervention potential. Given the uncertainty in predicting extinction rate using measures of cohort replacement rate and population growth rate for Pacific salmon (Botsford and Brittnacher 1998), this task is indeed daunting.

Benefits and Risks of Captive Breeding

Captive breeding programs can serve multiple objectives in salmon restoration. The Sacramento River Winter-Run Chinook Salmon Captive Broodstock (WRCCB) project was developed with multiple goals. WRCCB's primary objective is to maintain broodstock in captivity as an insurance program in the event the remaining wild population is further reduced or is extirpated. In this situation captive broodstock could serve as a gene bank to assist in rebuilding the stock. Alternatively, this propagation program can provide gametes for supplemental breeding. The supplementation strategy is based on the premise that an appropriate genetics program, developed in parallel with the broodstock technology, could guide the spawning of wild caught broodstock in tandem with captive broodstock. In this manner captively reared spawning candidates could expand the spawning options of the supplementation program, which can be limited if dependent solely on wild trapped fish.

Captive breeding differs significantly from conventional hatchery practice. Sound captive breeding should be based on rules of conservation biology that recognize the potential effect of creating a population of captive progeny that, if released, will influence the genetic variation of the remaining wild stock it is intended to enhance. Models of effective population size described by Ryman and Laikre (1991) provide a means characterizing the interaction of two or more populations of salmon (in this case wild versus captive) at various production levels if the genetic variation is known. These models are essential if a captive broodstock program is going to operate as a true supplementation mechanism that enhances the genetic resource and contributes to species' recovery. Due to the high fecundity of salmon the risk of disproportionately supplementing the captively bred population can be serious and jeopardize the wild population. However, precise measures of genetic variation require sophisticated and expensive techniques such as molecular genetic analysis. This expense will limit application of these preferred methods of monitoring and evaluation. Without these techniques and proper evaluation captive breeding programs can easily introduce unacceptable risk to salmon recovery efforts aimed at assisting threatened and endangered populations. However, it should be recognized that integrating supplementation in a captive breeding program with interannual variation of the wild salmon counterpart links a captive breeding intervention with ecosystem function. This is a desirable model for the evolution of all hatchery practice.

If implemented as a basic element of stock recovery, captive breeding warrants assessment and evaluation to minimize risk posed to the stock it is addressing. Some of the ways risk can be manifested in captive breeding programs can be subtle. A major difference between conventional hatchery operation and captive breeding is that the gene banking aspect of captive rearing often includes rearing multiple cohorts of salmon from embryo to sexual maturation. This long-term husbandry increases the opportunity for artifacts of the captive setting to create differential mortality in the captive population or among captive family groups. Such artifacts lend themselves to various genetic sinks and are a cause for concern (Waples 1991). Allendorf and Waples (1996) have described genetic risks associated with supplementation programs including effects of broodstock collection on wild populations, reduction of fitness, and changes in reproductive potential in naturally spawning fish as a result of lack of control in restocking efforts.

In the last few years a few salmonid captive breeding projects have attempted to share information and develop methods based on sound science. A major obstacle of captive breeding is that by the time a population warrants such a serious intervention, the population is likely so reduced that true experimental approaches cannot proceed due to the limited number of fish available. And, for threatened and endangered species the protective nature of the federal Endangered Species Act (ESA) usually precludes invasive techniques such as tissue or serological assays and other conventional laboratory analyses of animal health and reproductive physiology. Given these unusual limitations captive breeding programs have been slow to evolve new techniques aimed at the special conditions of captively rearing and spawning undomesticated fish. However, new technologies have emerged that set captive breeding programs apart from conventional hatcheries.

The Sacramento River winter-run chinook was the first anadromous salmonid population to be protected under the ESA. In November, 1990, the National Marine Fisheries Service (NMFS) issued a final listing of the Sacramento River winter-run chinook salmon as a threatened species (54 Federal Register {FR} 32085), and in February 1994 the stock was listed as endangered (59 FR 440). The WRCCB Project, now in its ninth year, has made substantial progress in both fish survival and gamete production (Arkush and others 1995). Application of advanced technologies in systems design such as computer controlled salinity systems that create seawater environments for smoltification have increased fish survivorship and simplified maintenance. Veterinary techniques such as the use of ultrasonography to assess maturation and even predict spawning time have been developed in concert with this project (Petervary and others, forthcoming). Moreover, the project has enabled significant advances in the areas of fish health and genetics, particularly with the development of molecular markers for genetic discrimination among stocks that have wide application in fisheries management (Banks and others 2000).

All of these developments demonstrate a divergence from conventional hatchery practices and set the stage for new possibilities in salmon restoration. In this way sound captive broodstock conduct creates the potential for changes in future hatchery practice. Scientifically-based captive broodstock programs have the ability to serve as research hatcheries, which have been proposed as one of several requirements for salmon restoration (Moyle 1993). Research hatcheries, based on sound conservation biology and captive breeding advances, can balance the need to continue salmon supplementation while identifying the changes required to move towards a larger conservation strategy (Hilborn 1992).

Integration with Habitat Recovery Plans

Threatened and endangered species restoration requires implementation of a carefully designed and comprehensive recovery plan as the ultimate goal. Artificial propagation programs can play a pivotal role in preventing extirpation of stocks. If such an intervention is warranted, it is critical that implementation is initiated before, and during, the early phases of recovery plan action. However, restoring naturally sustaining populations is the only way to address ecosystem-wide concerns; supplementation provides no equivalent. In accordance with Section 4(f) of the ESA, a recovery plan must be developed for species listed as endangered or threatened, and this plan must be implemented unless it is found not to promote conservation of the species. A recovery plan must include: (1) a description of site-specific management actions necessary for recovery, (2) objective, measurable criteria, which when met, allow delisting of the species, and (3) estimates of the time and cost to carry out the recommended recovery measures.

The National Marine Fisheries Service (NMFS) identified several factors as major causes of the decline of the winter-run chinook salmon, such as elevated water temperatures in the upper Sacramento River and impediments to upstream and downstream migration at the Red Bluff Diversion Dam (Hedrick and others 1995; Botsford and Brittnacher 1998). However, there is a wide range of factors that affect winter-run chinook salmon survival, and all factors must be addressed to assist in its recovery. Hence, NMFS has concluded that no single action will suffice, and a comprehensive plan will require the participation of federal agencies, state and local governments, private industry, conservation organizations, and the public. Moreover, while the ESA is designed to recover individual species, the recovery plan for the winter-run chinook salmon must consider ecosystem restoration. Concurrent with the winter-run chinook salmon decline is the reduction of other native species of plants and animals in the Sacramento River ecosystem. Moyle and Williams (1990) described 46% of the native fish stocks of the Sacramento River drainage as extinct, endangered, or in need of special protection. The loss of native fish genetic resources is further complicated by the invasion of non-native species that increases the level of complexity in ecosystem restoration. Moyle and Light (1996) describe how invasive species and invaded systems interact in idiosyncratic ways that are difficult to predict. Further, the degree of integration of an invasive species will depend on the level and degree of human and natural disturbance to the aquatic system (Vermeij 1996). One hundred State and federal candidate, proposed, and listed plants and animals, and California Department of Fish and Game species of special concern occur in the present habitat range of the Sacramento winter-run chinook salmon (NMFS 1997). Clearly, recovery plans must incorporate some form of adaptive management plan to protect the endangered or threatened stocks as well as other flora and fauna identified as species of special concern during implementation. And, recovery plans need to be "plastic" so as to allow the inclusion of newly identified components of the ecosystem during the monitoring phase.

Conclusions

Captive breeding is an expensive and labor intensive effort. Programs such as the Winter-run Chinook Salmon Captive Broodstock Project have made significant contributions to the evolution of hatchery management practice while functioning as stop gap measures in the decline of natural stocks. Captive breeding programs that are defined by the rules of conservation biology can calibrate supplementation to increase abundance without loss of the genetic variation they are intending to preserve. Captive breeding programs that operate as conservation hatcheries can be a template for future hatchery practice.

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Are Juvenile Chinook Salmon Entrained at Unscreened Diversions in Direct Proportion to the Volume of Water Diverted?

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Abstract

Mark-recapture experiments were used to test the null hypothesis that juvenile chinook salmon smolts emigrating from the Sacramento River are entrained at unscreened water diversions in direct proportion to the water volume diverted. The experiments were conducted at the RD1004 Princeton Pumping Plant during June 1995, with a similar set of mark-recapture experiments conducted at the RD108 Wilkins Slough diversion. Results of four tests conducted at the RD1004 Princeton Pumping Plant showed an average of 0.05% of the marked salmon being entrained, compared to 1.03% of the Sacramento River flow diverted. Overall results at the RD108 Wilkins Slough diversion showed a similar pattern, with 0.08% of the marked salmon being recaptured compared to 1.1% of the Sacramento River flow being diverted. Based upon results of these tests the null hypothesis was rejected. The percentage of juvenile chinook salmon entrained was more than ten times lower than the corresponding percentage of Sacramento River flow diverted. Results of these tests have implications in the assessment of entrainment mortality of juvenile chinook salmon at unscreened diversions and the calculation of costs and biological benefits for intake screening projects. These study results are limited, however, due to the relatively low percentage of Sacramento River flow diverted during these 1995 tests, the assumption that hatchery-reared, spray-dyed salmon released a relatively short distance upstream of an unscreened diversion are representative of the behavioral patterns and distribution of wild salmon within the Sacramento River, and the size and configuration of water diversions tested.

Introduction

A large number of water diversions exist on the Sacramento and San Joaquin rivers and throughout the Delta (Herren and Kawasaki, this volume). The majority of these water diversions is unscreened. Concern has been expressed by resource agencies and other interested parties regarding the incremental increase in mortality to juvenile chinook salmon (*Oncorhynchus tshawytscha*), and other aquatic species, resulting from entrainment losses at these diversions. Data are not available, however, to quantify entrainment losses of juvenile chinook salmon at a majority of these sites. As part of the assessment evaluating diversion effects on juvenile chinook salmon, and benefits associated with positive barrier fish screens, an assumption has been made that fish entrainment is proportional to the volume of unscreened water diverted. To date few experimental tests have been performed within the Sacramento-San Joaquin system to verify or refute this fundamental assumption. Furthermore, no studies have been identified from the scientific literature that document the relationship between entrainment losses for juvenile salmon and steelhead in relationship to the volume of water diverted at unscreened intake structures located on west coast tributaries.

To test the null hypothesis that juvenile chinook salmon are entrained at unscreened diversions in direct proportion to flow diverted mark-recapture studies were performed in 1995 at an unscreened water diversion. The experiment was performed at the Reclamation District 1004 (RD1004) Princeton Pumping Plant. These tests were conducted as part of a more comprehensive investigation of the potential application of alternative fish protection devices (for example, acoustic barriers) in reducing juvenile chinook salmon entrainment losses at water diversions (Hanson 1996a).

Princeton Pumping Plant

The Princeton Pumping Plant is located on the east bank of the Sacramento River just north of the town of Princeton at river mile 164.4 (Figure 1). The pumping plant diverts water from the Sacramento River into Drumheller Slough, which serves as part of the RD1004 conveyance and distribution system. RD1004 provides water to approximately 15,000 acres of agricultural land and 10,000 acres of migratory waterfowl wetland habitat within the Butte Basin in Glenn and Colusa counties.

The Princeton Pumping Plant has been in operation since 1912, but was extensively rebuilt in 1981. The facility consists of four 150 hp, 36-inch diameter, vertical mix-flow pumps. The fifth pump is a 30-inch diameter, 100 hp, vertical mix-flow pump located on a separate platform. Each of the pumps has a separate 36-inch diameter flap-gate and steel discharge line entering Drumheller Slough. At the time of the 1995 investigations the pumping plant diversion was unscreened.

Peak seasonal diversions at the Princeton Pumping Plant occur during the spring irrigation of rice fields and other agricultural lands and during the fall flooding of seasonal managed wetlands. During the remainder of the irrigation season, the pumping plant provides water for agricultural operations. The spring peak typically occurs from April 15 to May 30, which coincides with the primary seasonal period of fall-run chinook salmon smolt emigration from the Sacramento River. The fall and early winter peak pumping typically occurs between October and mid-January, a time when juvenile winter-run chinook may be emigrating.

Peak diversion capacity at the Princeton Pumping Plant is approximately 290 cfs. During maintenance flow two to three pumps are typically in operation (120 to 180 total cfs), depending on water demand within the service area. Diversions occur both by active pumping and, when Sacramento River elevation is high, by gravity flow.

Methods

The experimental design of the field investigations was based on the release of spray-dyed marked juvenile chinook salmon into the Sacramento River upstream of the unscreened Princeton Pumping Plant diversion and subsequently monitoring the number of marked salmon recaptured at the water diversion over a 48-hour period. Results of more comprehensive fish investigations at two unscreened diversion sites (Hanson 1996a, 1996b) documented that a 48-hour sampling duration was appropriate for these mark-recapture tests.

Using release and recapture data, an estimate was calculated of the percentage of the marked salmon entrained at the unscreened diversion. Monitoring the volume of water diverted and the corresponding flow within the Sacramento River allowed calculation of the percentage of the Sacramento River flow diverted. The null hypothesis that juvenile chinook salmon are diverted in direct proportion to flow diversion could then be tested by comparing the estimated percentage of juvenile chinook salmon entrained with the corresponding estimate of the percentage of Sacramento River flow diverted during each test period.

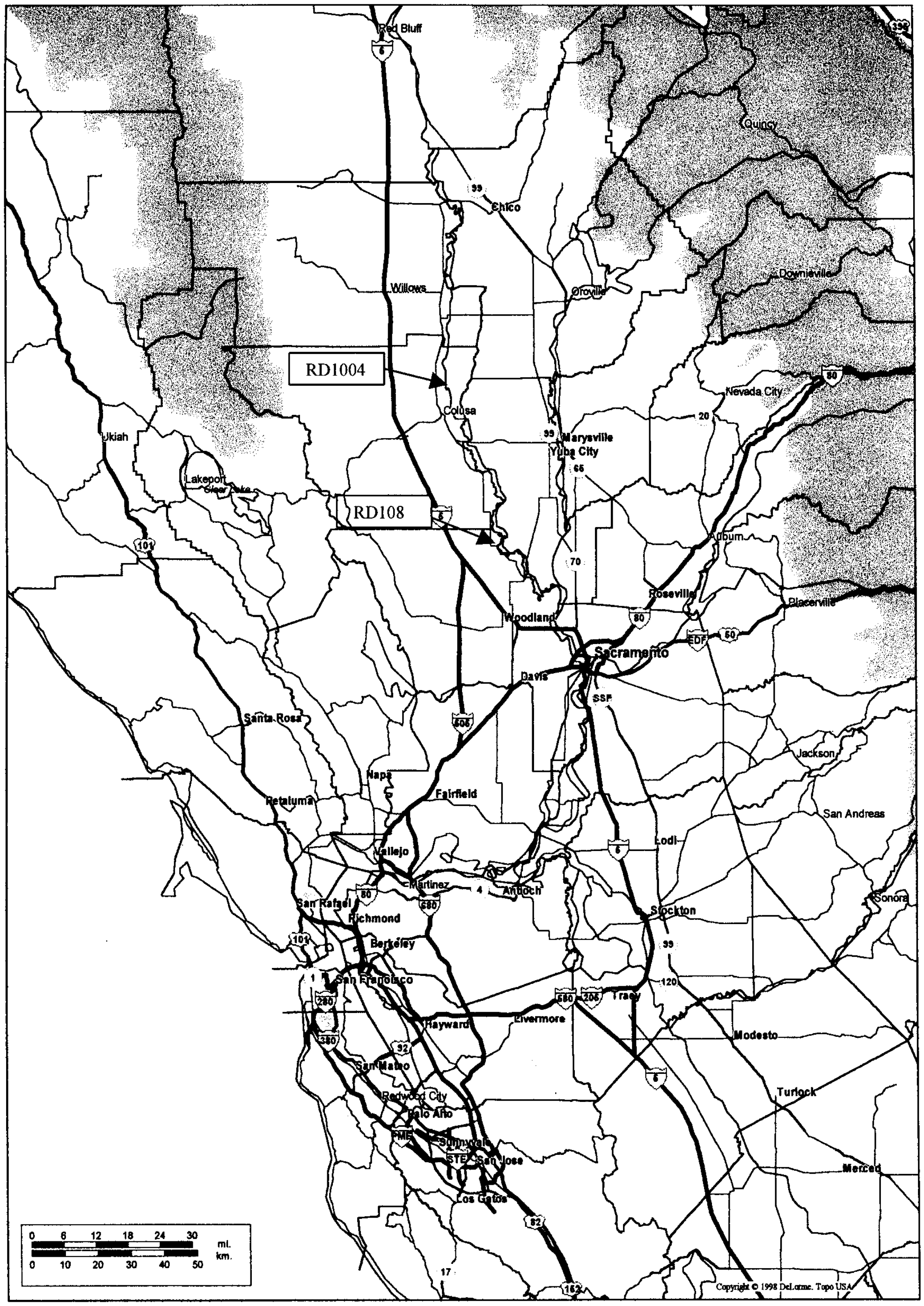

Figure 1 Location of the RD 1004 Princeton Pumping Plant and RD 108 Wilkins Slough diversion on the Sacramento River

Juvenile Salmon Spray-Dye Marking and Release

Juvenile chinook salmon used in these tests were obtained from the California Department of Fish and Game's (DFG) Feather River Hatchery. Juvenile salmon were marked using spray-dye (Scientific Marking Materials) at the hatchery. The number of fish marked was determined by weighing a subsample (number of fish per pound) and subsequently by weighing all marked fish within a test group. Juvenile salmon were marked without anesthesia and were retained in the Feather River Hatchery for a minimum of 72 hours after marking to recover from handling stress.

A sub-sample of approximately 100 marked fish from each release group was obtained from the transport truck and held on-site for a period of 48 hours, corresponding to the duration of the recapture collections for each test, to determine post-release survival. Fish held for post-release survival observations were inspected for dye retention as part of the quality assurance program.

Approximately 25,000 juvenile chinook salmon were marked for use in each release group. Mortalities occurring during and after marking were documented for each release group. After the hatchery recovery period, the marked group was loaded into a commercial hatchery truck for transport to the release location. Before release, fish within the transport truck were examined to determine the number of mortalities and the overall condition of the release group. Transport mortality ranged from 0.1% to 0.3%, while survival of a sub-sample from each release group 48-hours after release ranged from 98% to 100%. Inspection of the sub-sample of juvenile salmon held on-site from each release group confirmed 100% spray-dye retention and detection.

Dissolved oxygen and temperature were measured within the transport truck and Sacramento River at the time of release. Water temperature within the hatchery transport truck and Sacramento River at the release site were within 0 to 1.7 °C (0 to 3 °F), thereby avoiding significant temperature changes and thermal shock for fish at the time of release.

Marked fish were released at a location on the east side of the Sacramento River approximately 0.55 miles upstream of the RD1004 Princeton diversion. The release location selected for use in these tests was based upon access to a location sufficiently far upstream to provide the juvenile salmon an opportunity to disperse within the Sacramento River before encountering the unscreened diversion, yet sufficiently far downstream of identified sources of mortality, including other unscreened diversion locations.

Juvenile Salmon Entrainment Monitoring Using Fyke Nets

Monitoring the number of juvenile chinook salmon and other fish species entrained was performed using fyke nets approximately 35 feet long, mounted over the discharge of each pump. The fyke nets sampled 100% of the flow diverted from the Sacramento River. Fyke nets were constructed using 1/8-inch mesh equipped with a live box at the cod end. Collections were made from each live box to remove both fish and debris without the necessity of removing the entire net. Each live box was accessed from a floating dock located within the discharge canal of the Princeton Pumping Plant.

Fyke nets were processed to remove entrained fish and debris a minimum of twice per day (morning and afternoon), although more frequent processing was also performed as part of diel distribution studies. Although rips and tears in the fyke nets were uncommon, the nets were removed and inspected approximately every four to six days.

Direct release studies were performed to determine collection efficiency of the fyke nets. Collection efficiency studies were performed by releasing a known number of marked juvenile chinook salmon into the intake of diversion pump number one, and subsequently documenting the number of marked fish retained in the fyke net at the completion of the sampling cycle. Sampling cycles varied from 2 to 24 hours after release of marked fish into the diversion pump to determine the effects of sampling duration on net retention. Typically 40 juvenile chinook salmon were used in each collection efficiency test. Juvenile chinook salmon used in these tests ranged from 76 to 142 mm FL (mean length 102 mm). Salmon were alive at the time of release into the diversion pump. Fyke net collection efficiency studies had an overall recapture efficiency of 80%, with a range of 65% to 100%.

Data collected in association with each fyke net sample included identification and enumeration of all fish species collected. All salmon collected were examined using ultraviolet lights to determine the number and color of marked fish recaptured. Fork-length was measured for juvenile chinook salmon. Length measurements were made for a sub-sample of other fish species. Data were recorded for each collection identifying the individual fyke net where the collection was made, the start and end times of the sampling interval, and the water volume sampled. Mortality and damage to fish collected was also documented. After processing, live fish were released approximately 0.25 miles downstream of the diversion.

Sacramento River Flow and Diversion Operations

Data on daily Sacramento River flows in the vicinity of the Princeton diversion (Colusa Bridge) were obtained from the California Department of Water Resources (DWR) California Data Exchange Center Database (CDEC). Daily average Sacramento River flow during the study ranged from approximately 13,300 to 14,100 cfs.

The volume of water diverted from the Sacramento River by individual pumps at the Princeton facility was documented coincident with each fish collection. To the extent possible, diversion pump operations were held constant throughout each test to reduce effects attributable to variation in diversion operations. The diversion rate (cfs) and total volume diverted (acre-foot) were monitored for each individual diversion pump using a Sparling Inline flowmeter. Diversion rates for individual pumps typically ranged from 50 to 70 cfs. Diversion pump number five was not operational during the June 1995 study period. Diversion pump number three experienced operational problems and was removed from service in mid-June. Diversion pumps one, two, and four operated on a relatively consistent basis throughout the study period.

Results

Four mark-recapture tests were performed, which provided information on the percentage of juvenile chinook salmon entrained at the unscreened Princeton Pumping Plant. During the four mark-recapture tests included in this analysis (Table 1), a total of 124,394 salmon was released into the Sacramento River.

Spray-dyed chinook salmon were recaptured in low numbers in the RD1004 Princeton Pumping Plant fyke nets (see Table 1). The percentage of marked fish recaptured ranged from 0% to 0.1%, with an overall average for the four tests of 0.05%. The corresponding estimate of the percentage of Sacramento River flow diverted during each of these test periods ranged from 0.9% to 1.2% (see Table 1), with an overall average of 1.03%.

Table 1 Summary of juvenile chinook salmon mark-recapture test results used to determine the percentage of entrainment at the RD1004 unscreened Princeton Pumping Plant diversion in June 1995

Date	*Number released*	*Number recaptured*	*Expanded number recaptured* [a]	*Sacramento River flow (cfs)*	*Percentage of salmon entrainment*	*Percent Sacramento River flow diverted*
7–9 June	24,865	21	26	14,068	0.10	1.0
17–19 June	24,850	0	0	14,139	0	0.9
21–23 June	24,869	0	0	13,286	0	1.0
28–30 June	49,810	28	35	13,772	0.07	1.2
Total or Average	124,394	49	61		0.05	1.03

[a] Recaptures were expanded based on 80% fyke net collection efficiency (see text).

Discussion

The numbers of marked salmon entrained and recaptured at the RD1004 unscreened diversion was substantially and consistently lower than the percentage of Sacramento River flow diverted (Figure 2). The overall percentage of juvenile salmon recaptured was 0.05% (adjusted for net collection efficiency), compared with an average of 1.03% of the Sacramento River flow diverted during the period of these studies. The substantially lower percentage of fish diverted in these tests demonstrates that marked hatchery-reared juvenile chinook salmon are not entrained in direct proportion with the water volume diverted at the RD1004 intake and, the results suggest juvenile salmon are substantially less vulnerable to entrainment losses than would be expected based purely on a volumetric relationship. Factors such as the location of the diversion with respect to major flow patterns, topographic characteristics of the Sacramento River channel, the location of the diversion pump inlet within the water column, and the behavioral response of chinook salmon smolts to turbulence and velocity differences associated with operation of the intake may contribute to a reduction in the susceptibility of juvenile salmon to entrainment. In addition, the juvenile salmon used in these mark-recapture tests were hatchery reared and released a relatively short distance upstream of the diversion (0.55 miles) and may, therefore, not be representative of the behavioral patterns or distribution of wild salmon within the Sacramento River.

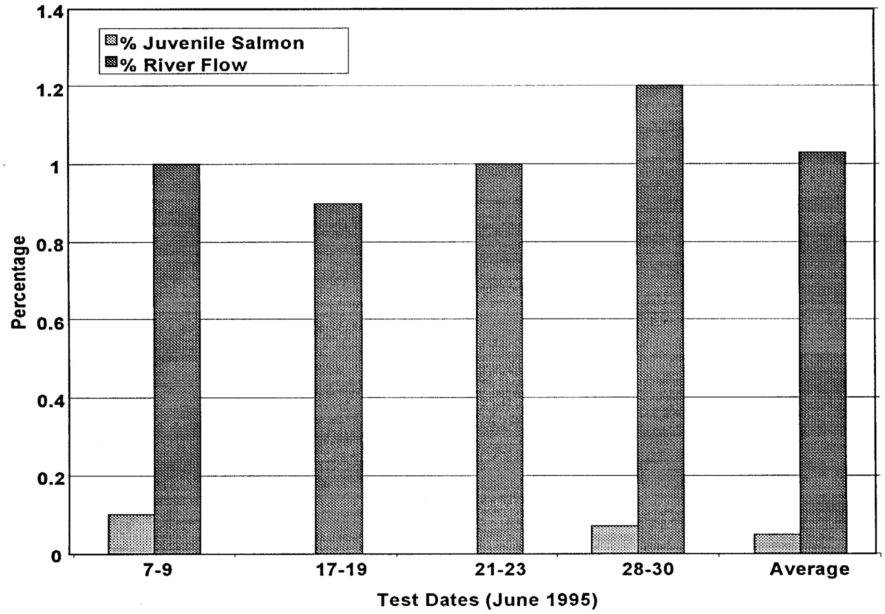

Figure 2 Comparison of the percentage of marked juvenile chinook salmon entrained and Sacramento River flow diverted at the unscreened RD 1004 Princeton Pumping Plant in June 1995

Results of the mark-recapture studies conducted at the RD1004 Princeton Pumping Plant are consistent with results of a similar mark-recapture investigation conducted in 1995 on the Sacramento River at the RD108 Wilkins Slough diversion (Hanson 1996b). Reclamation District No. 108 operates the Wilkins Slough Pumping Plant, located on the west bank of the Sacramento River at river mile 117.8, which was unscreened during 1995. A mark-recapture experiment was used at RD108 to evaluate entrainment of juvenile chinook salmon during four experiments conducted during June 1995. Spray-dyed salmon were released along the west bank of the Sacramento River at locations 0.45 to 0.65 miles upstream of the Wilkins Slough diversion. Marked salmon were subsequently recaptured within the Wilkins Slough diversion using fyke nets and processed in a manner similar to that described for the RD1004 investigation. During this study average daily Sacramento River flow ranged from approximately 13,000 to 14,500 cfs, while average daily diversion rates ranged from 118 to 221 cfs. Additional information regarding the four mark-recapture experiments conducted in 1995 at the RD108 Wilkins Slough diversion is documented in Hanson (1996b). Results of the RD108 mark-recapture tests are summarized below and compared to results from the RD1004 experiments.

Diversion location	*Number of marked salmon released*	*Expanded number recaptured* [a]	*Percentage of salmon entrainment*	*Percentage of Sacramento River flow diverted*
Princeton Pumping Plant [b]	124,394	61	0.05	1.03
Wilkins Slough Diversion [c]	99,419	75	0.08	1.1

[a] Fyke net collections were expanded assuming a collection and retention efficiency of 80%.

[b] Source: this study.

[c] Source: Hanson 1996b.

Results of the 1995 studies were consistent in demonstrating the low susceptibility of hatchery-reared juvenile chinook salmon to entrainment losses, and the fact that marked juvenile salmon were not entrained in direct proportion to the volume of Sacramento River water flow diverted by either the RD1004 Princeton Pumping Plant or the RD108 Wilkins Slough diversion. Results of these experiments provide useful insight into the vulnerability of juvenile chinook salmon to entrainment losses and can be used as part of the basis for assessing the risk of adverse impacts resulting from unscreened water diversion operations. Additional studies will be required, however, to provide data on the relationship between the vulnerability of hatchery-reared, marked salmon released a relatively short distance upstream from the diversion to entrainment losses and the vulnerability of wild salmon to entrainment losses at these unscreened diversion locations. The percentage of juvenile salmon entrained during mark-recapture studies should also be viewed in context with the flow occurring in the Sacramento River, diversion operations, and the percentage of Sacramento River flow diverted during these tests. Results of the 1995 tests may or may not be representative of the relationship between unscreened diversion operations and the susceptibility of juvenile chinook salmon to entrainment losses under other environmental conditions in which the Sacramento River flow may be reduced, and the percentage of river flow diverted may be higher than that observed during the 1995 tests.

Acknowledgements

Funding for this study was provided by Reclamation Districts 108 and 1004 as part of investigations to evaluate juvenile chinook salmon entrainment losses and the effectiveness of alternative behavioral barrier technologies in reducing entrainment mortality. The California Department of Fish and Game provided juvenile chinook salmon from the Feather River Fish Hatchery for use in mark-recapture tests. Field data collection and analysis was performed by the staff of Hanson Environmental, Inc. The author is grateful to Pete Rhoads, Joe Merz, and Nina Kogut for providing constructive comments on an earlier draft of this manuscript.

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Inventory of Water Diversions in Four Geographic Areas in California's Central Valley

Janna R. Herren and Spencer S. Kawasaki

Abstract

Water diversions in California, used primarily for agricultural, municipal, and industrial applications, have been considered a possible culprit in the decline of many California fishes. In 1991, the California Department of Fish and Game (DFG) initiated a study using the Global Positioning System (GPS) to inventory water diversions. The initial focus was on the Sacramento-San Joaquin Delta (Delta) and the Suisun Marsh, then continued to the Sacramento River and the San Joaquin River Basin, The inventory was to find, quantify, describe, and categorize diversions along waterways where California fish may be affected by water diversions. As of April 1997, 3,356 diversions have been located and mapped in a Geographical Information System (GIS). Approximately 98.5% of the diversions identified were either unscreened or screened insufficiently to prevent fish entrainment. The GPS data were post-processed to provide a horizontal accuracy of ±5 meters. The information was primarily collected by visual inspection of diversions on the stream bank. Information is maintained in a Microsoft Access database.

Introduction

California's Central Valley waterways support a rich diversity of fish species that are ecologically, economically, and recreationally important. Much of the water necessary for the survival of these species, however, is diverted out of the streams primarily for agriculture, but also for industry and municipalities.

A few researchers have attempted to estimate the number of Central Valley water diversions. Hallock and Van Woert (1959) estimated that there were 900 water diversions on the San Joaquin and Sacramento rivers above the Delta, which are used by anadromous fishes. Of the 900 diversions, only a small portion were specifically identified and described. Brown (1982) estimated 1,850 water diversions in the Delta based on an inventory conducted by the U.S. Bureau of Reclamation (USBR) in 1963–1964 and limited field observations.

These previous inventories were based on estimates and did not accurately assess the true number of diversions, nor did they maintain a database to monitor diversion modifications or relocations. In fact, water diversion inventories were not the objective of past studies, rather the objectives were to estimate water export volumes from geographic regions or fish losses due to entrainment. They did not identify all diversions in the area and map each diversion.

Past studies often located water diversions by visual observation while driving levee roads (Brown 1982). The corresponding odometer reading was then compared with river miles and river banks on maps to determine the locations of individual diversions (Hallock and Van Woert 1959; USBR 1963, 1964; Brown 1982). Another source of water diversion location and information in the Automated Water Rights Information Management System (AWRIMS) managed by the State of California Water Resources Control Board. AWRIMS gives locations of water diversion using the Public Land System, providing accuracies within 40 acres.

Comparisons of these earlier data did not correspond to GPS locations. DFG determined that a standardized and accurate database of water diversions was necessary before the magnitude of diversion-related fish losses could be recognized and addressed.

The GPS is a satellite-based positioning system maintained and operated by the U.S. Department of Defense (DOD). The use of GPS by the scientific community is growing due to its ease of use and high degree of accuracy. Examples of such uses include radiotelemetry studies of moose populations under various types of canopies (Moen and others 1996) and mapping and counting ponds used by breeding waterfowl (Strong and Cowardin 1995). With differential correction, the accuracy of GPS is usually within two meters of the true location 50% of the time, and within five meters 95% of the time (Trimble Navigation Ltd. 1992).

The objectives of our study were to find existing Central Valley water diversions, map them using GPS and GIS, and to identify and categorize them as screened or unscreened. The database of water diversions created by this program can easily be updated with future surveys to identify changes to location, size, and other features. Future objectives of the program will include prioritizing fish screen projects. GPS was selected as the survey method because of its ease of use, superior accuracy, application to various mapping programs and GIS compatibility.

Methods and Materials

Our study began with four regions established based on watershed drainages and similar geographic features (Figure 1). The initial focus was on the Sacramento-San Joaquin Delta (defined in California Water Code, Section 1220) and Suisun Marsh (defined in California Water Code, Sections 29101 and 29002–29003) since many ecologically, commercially, and recreationally popular fish species either reside in these areas or pass through them during some stage of life. These species include the chinook salmon (*Oncorhynchus tshawytscha*), striped bass (*Morone saxatilis*), white sturgeon (*Acipenser transmontanus*), delta smelt (*Hypomesus transpacificus*), Sacramento splittail (*Pogonichthys macrolepidotus*), and steelhead (*Oncorhynchus mykiss*). The survey for water diversions then continued to the San Joaquin River Basin (San Joaquin River mile 72.4 to the confluence with the Merced River, as well as the Stanislaus, Tuolumne and Merced rivers) and the Sacramento River (river mile 59.4 to Keswick Dam).

Field

A physical search to locate water diversions was made by boat, driving levee roads, or by walking the banks of waterways. A Pathfinder Basic Plus and a GeoExplorer, two portable GPS receivers manufactured by Trimble Navigation Limited, were used to geographically locate the position of water diversions. Topographical and navigational maps were used to systematically survey waterways, eliminating the possibility of data duplication.

Upon visual discovery of a water diversion, collection of data points was initiated using a GPS receiver. Data points were received via radio signals sent from 24 NAVSTAR satellites operated and maintained by the DOD (Trimble Navigation Ltd. 1982). Collection of points was made at, or as close to, the point of diversion as possible. Between 180 and 200 data points were collected at each site and stored in the receiver as individual rover files with unique file numbers. A location consisting of more than four diversions were treated as a single point of diversion.

Along with the GPS data, other attributes and a physical description of the diversion were recorded including type of diversion, intake size (outside diameter to the nearest inch, as measured with a logger's diameter tape), type of discharge, bank location, screen type (when present), river system or waterway, and likely primary use of the diverted water. Photographs were taken of each diversion or intake structure. Discharge outfalls or structures were only photographed if unique or uncommon to the region.

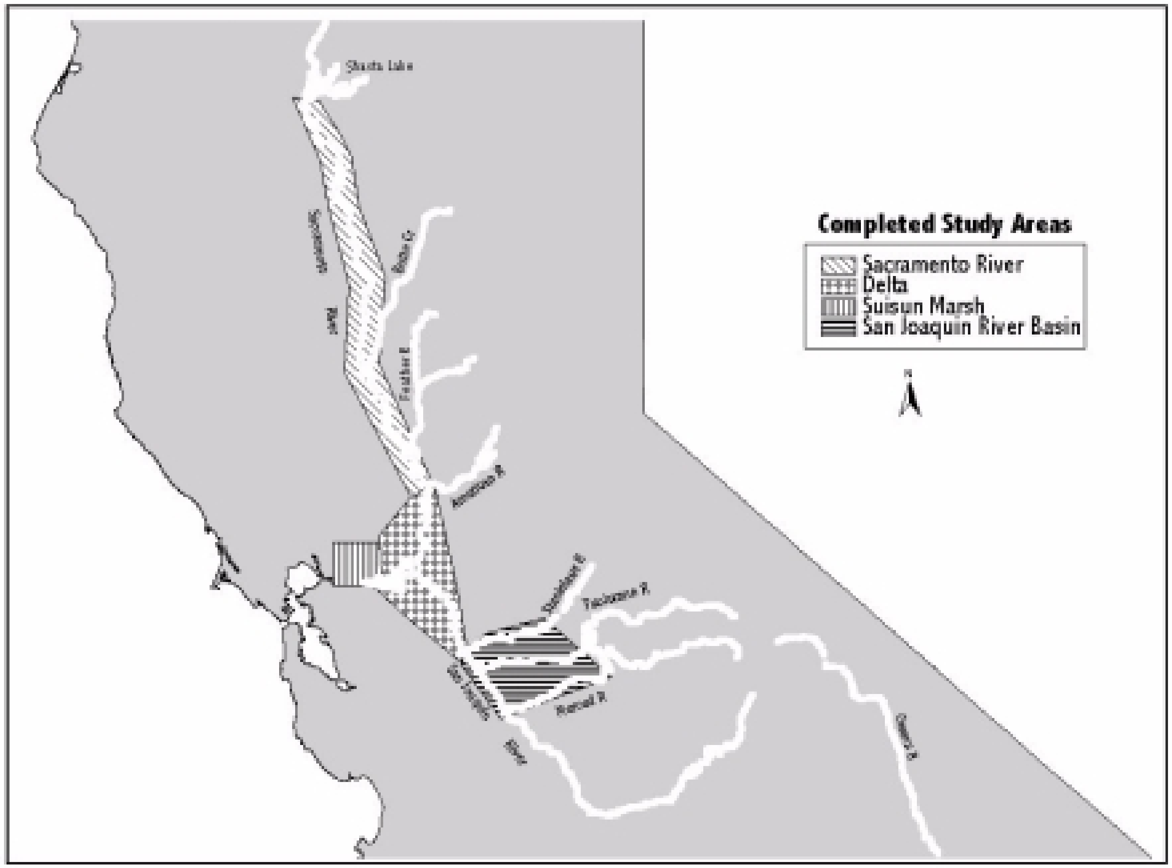

Figure 1 The water diversion study area showing the four geographic areas surveyed: mainstem Sacramento River, Sacramento-San Joaquin Delta, Suisun Marsh, and San Joaquin River basin watershed

Office

At the completion of each field day the rover files were uploaded to a personal computer. These files were then differentially corrected using the post-processing software, pFinder. Differential correction is the process of comparing the raw GPS positions (rover), to a known location (base station). Base station files were downloaded from various locations: USFWS offices in cities (including Sacramento, Susanville, Porterville, and Eureka); Trimble Navigation Ltd.; and other companies that operate base stations throughout the State. After post-processing, GPS data and associated attributes were entered, stored, and maintained in a Microsoft Access database. This information has been used to create a GIS layer output to a North American Datum 27 (NAD-27) Teale-Albers projection to be compatible with DFG's Arcview GIS system.

Each diversion is stored in the database as an individual record where it is assigned a unique identification number. Associated with the identification numbers are the map coordinates of the diversions, as well as its attributes and owner identification number. The owner identification relates to another Microsoft Access database containing the names and addresses of diversion owners. Determination of ownership is attempted through the research of water rights applications in AWRIMS, signs on the diversions, or through personal communication with the owners themselves.

Results

As of April 1997, 3,356 diversions have been located and mapped using GPS (Figure 2). Of these, 424 diversions were along the Sacramento River above the I Street Bridge in Sacramento (Figure 3), 298 diversions were found within the San Joaquin River Basin (Figure 4), 2,209 diversions were within the Delta, and 366 diversions were in the Suisun Marsh (Figure 5). Individual diversion sites containing a group of more than four diversions account for 31 points in our database. These points, if counted individually, add 144 diversions to the total number. These results have been placed on a layer of DFG's GIS as coverage files of 1:250,000 scale United States Geological Survey (USGS) topographic maps using ArcView (version 3.0).

Along with the locations of each diversion, we identified their attributes including the type of diversion and type of fish screen (if present) (Table 1). According to our data, a regional preference is evident for each diversion type. Floodgates are almost exclusively used in Suisun Marsh, while siphons are the preferred method of diversion in the Delta. Pumps are necessary in the Sacramento River and the San Joaquin River Basin because the land elevation is higher relative to water elevation. Furthermore, the Sacramento River study area contained the highest percentage of fish screens that are designed to meet current DFG criteria—almost 6%. The Delta, which had the highest density of water diversions, had fish screens on only 0.7% of the intakes.

The intake size of the diversions also varied based on region. Ninety percent of the diversion intake sizes in the Delta measured between 12 and 24 inches, whereas the Suisun Marsh was composed of 90% floodgates, with intake sizes between 36 and 48 inches. Fifty-four percent of the San Joaquin River diversions measured between 9 and 16 inches. Greater variability of diversion intake diameters for the Sacramento River and San Joaquin River Basin regions may be due to the higher horsepower pumps that are necessary to move water out of streams where more head differential exists. The largest water diversion in our database, to date, occur in the Delta where water is transported through large pumping plants into the California Aqueduct (State Water Project) and the Delta-Mendota Canal (Central Valley Project).

Discussion

Water diversions have been suggested as a significant cause of the loss and decline of many resident and migratory fish species. Most water diversions are unscreened, and to date, very little information has been reported on the entrainment losses of fish due to unscreened water diversions. Species such as the chinook salmon, steelhead, striped bass, white sturgeon, delta smelt, and Sacramento splittail, are valuable resources to California because of their ecological, commercial, and recreational importance or because they contribute to the rich biological diversity in California. Winter-run chinook salmon, delta smelt, Sacramento splittail, and steelhead are currently listed as endangered or threatened. Most small diversions do not entrain many young salmon and steelhead, however, collectively considerable numbers may be taken (Hallock and Van Woert 1959).

Other west coast states including Washington, Oregon, and Idaho, have undertaken similar inventories on water diversions (John Easterbrooks, personal communication), but on a smaller scale. Their surveys are mainly on watersheds where migrating anadromous fish may be adversely affected by water diversions. The data being collected are neither post-processed nor applied to a GIS. Our inventory of California water diversions is of much greater magnitude and accuracy than other west coast states.

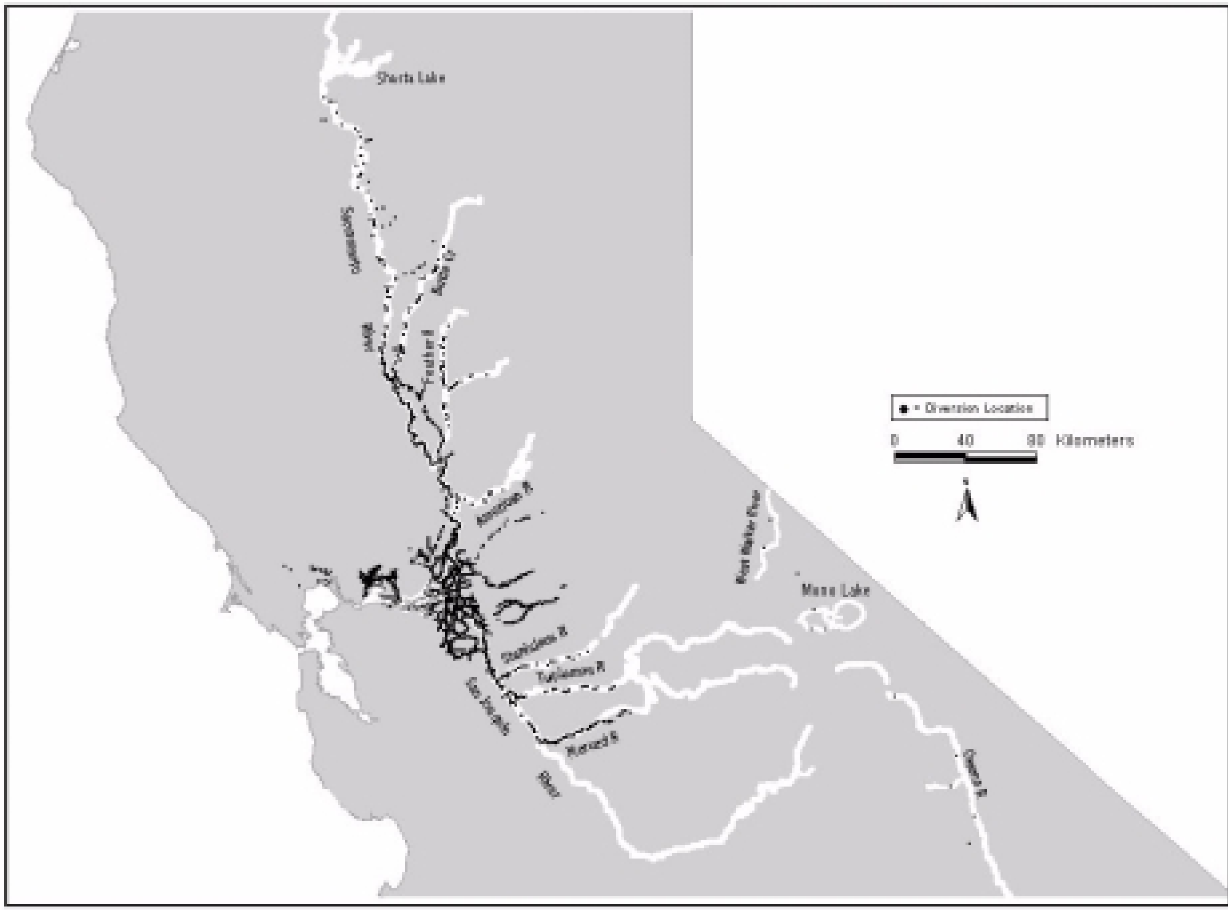

Figure 2 The Global Positioning System has been used to locate and map 3,356 water diversions in California as of April 1997

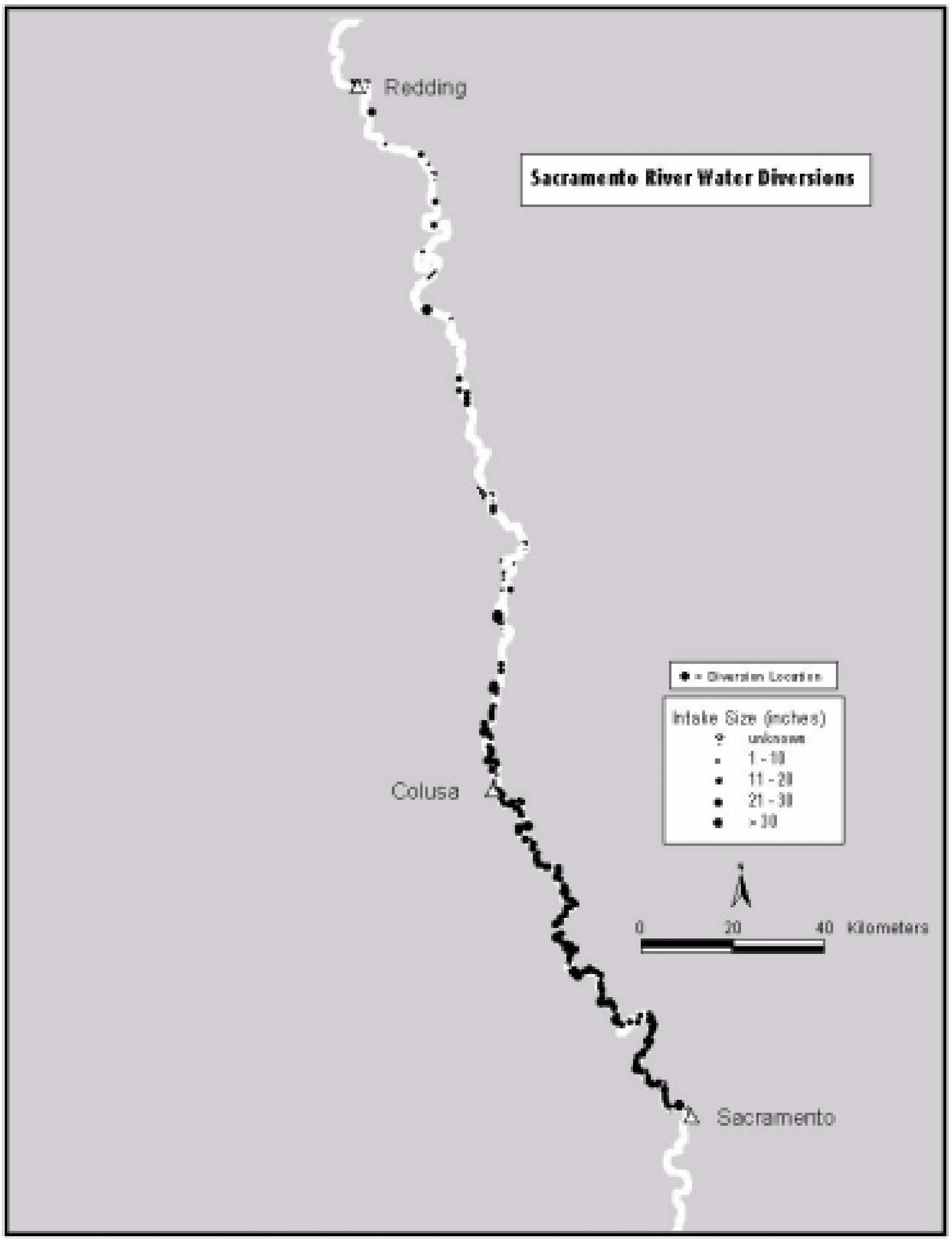

Figure 3 Four hundred and twenty-four water diversions have been identified on the Sacramento River between Keswick Dam and Sacramento at the I Street Bridge as of April 1997

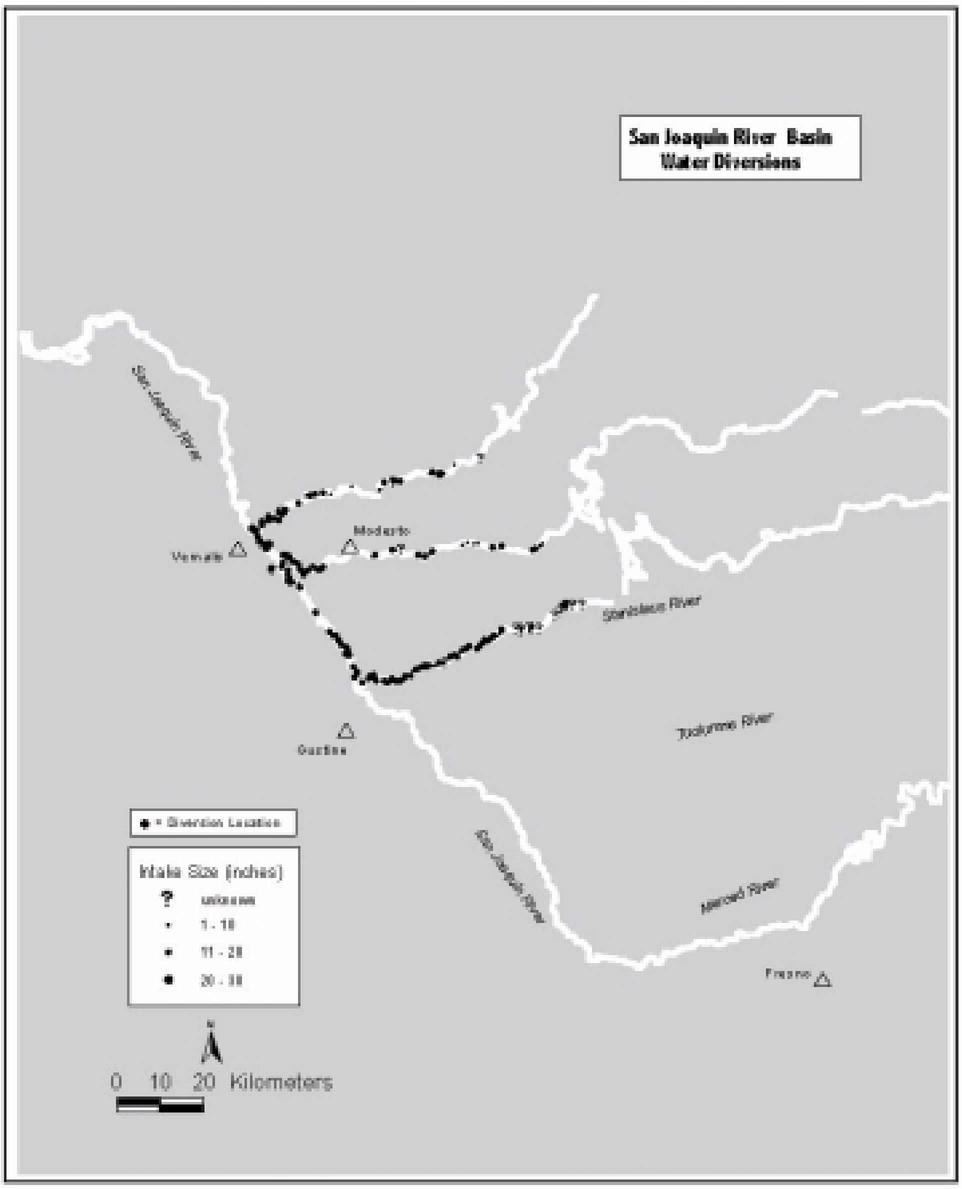

Figure 4 Two hundred and ninety-eight water diversions have been identified on the San Joaquin River between the lower boundary of the Sacramento-San Joaquin Delta and the mouth of the Merced River including the major tributaries as of April 1997

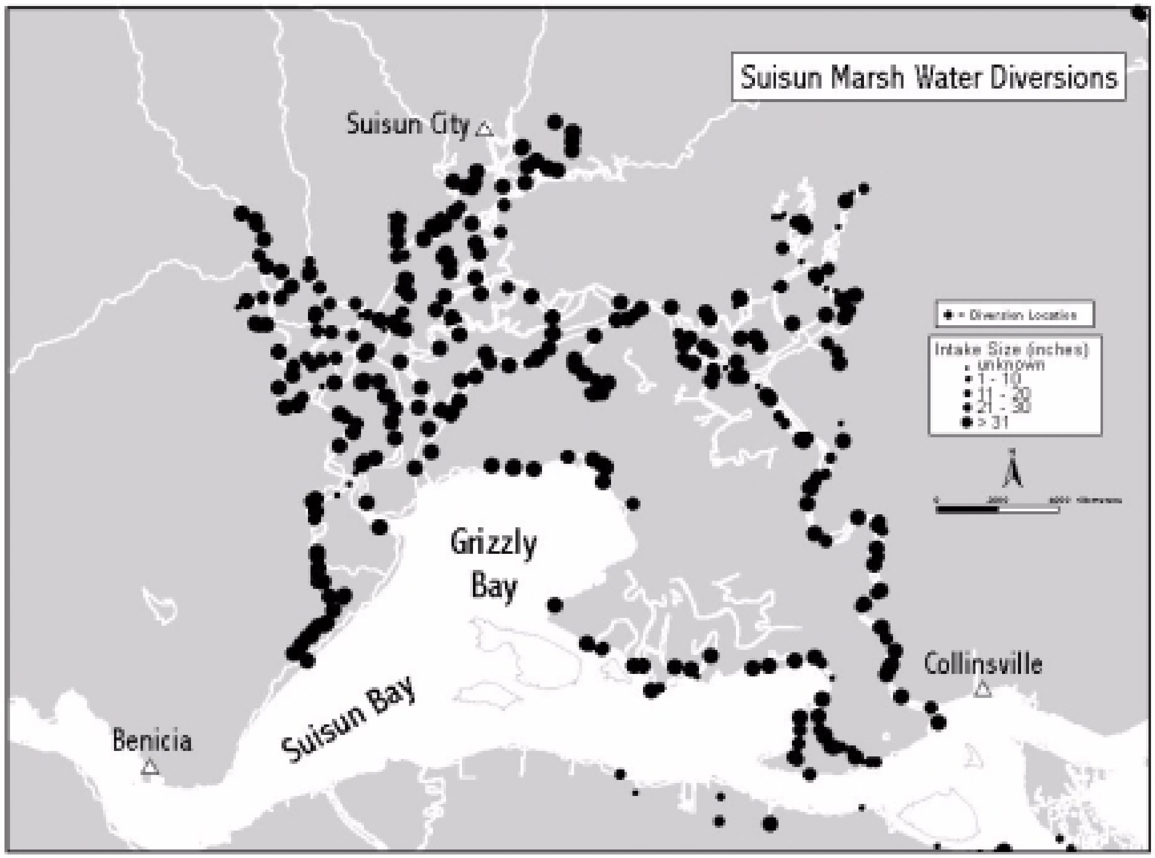

Figure 5 In the Suisun Marsh and the Sacramento-San Joaquin Delta, 366 and 2,209 water diversions, respectively, were identified and mapped using the Global Positioning System as of April 1997

Table 1 Percent and number of diversion types found in each of the four geographic study areas

	Delta		Sacramento River		San Joaquin River Basin		Suisun Marsh	
	%	No.	%	No.	%	No.	%	No.
Vertical pump	19.1	423	27	114	48	143	3	11
Slant pump	7.3	161	41	173	9	27	---	---
Centrifugal pump	17	375	19	80	34	100	<1	2
Siphon	45	994	<1	1	<1	1	---	---
Floodgate	3.6	79	---	---	---	---	79	328
Unknown[a]	1.3	28	5	21	3	8	1	4
Misc.[b]	6.7	149	8	35	6	19	17	69
Total		2209		424		298		414
Screened	1%	17	6%	25	1%	2	2%	8

[a] Diversions are classified as unknown when they cannot be definitively identified due to their location (private property or concealed by brush), missing parts, or hybridized pumps.

[b] Miscellaneous diversions are other devices used to move water. These include submersible pumps, Archimedes screw pumps, weirs, portable pumps, channels, culverts, and variable speed pumps.

We compared data from previous studies on water diversions conducted by the USBR (1963–1964) and Brown (1982) with our data for five selected Delta islands and noted several differences (Table 2). Approximately 21% of the diversions on the islands identified in the 1982 report had changed location. Some differences can be attributed to alternate methods, diversion relocation, or the consolidation of several small diversions into a centrally located large diversion. Since locations and sizes of water diversions could become an important source of information for issues including water pollution cases and fish screen planning, these discrepancies support the need for a comprehensive and standardized database. Water diversion GIS data should be kept in a format that is acceptable and easily accessible by various agencies and private individuals.

Table 2 A summary of changes in agricultural diversions on five islands between the DWR water diversion survey (Brown 1986–1987) and GPS data collected by DFG through April 1997

Island	*Total # of diversions in 1993–1994*	*Deletions since 1987*	*Additions since 1987*	*Intakes increased in size*	*Intakes decreased in size*
Bacon	30	14	7	8	2
Bouldin	38	11	11	11	---
McDonald	36	5	9	8	2
Twitchell	22	2	2	8	1
Venice	24	---	2	---	1
Total	150	32	31	35	6

Currently, along with the four geographic regions already surveyed, the American River and parts of the Mono Lake Basin have been surveyed (Figure 2). The study is ongoing to complete the San Joaquin River, the major tributaries to the Sacramento River, the coastal rivers and streams, and all watersheds containing migratory or resident fish populations that might be affected by water diversions.

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