SALMON MANAGEMENT IN THE 21ST CENTURY: RECOVERING STOCKS IN DECLINE

Proceedings of the 1992 Northeast Pacific
Chinook and Coho Workshop
Boise, Idaho
September 28-30, 1992

Presented by
Idaho Chapter of the American Fisheries Society
University of Idaho Water Resources Research Institute
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WELCOME

Karen Pratt
Idaho American Fisheries Society Chapter President

Idaho Chapter of the American Fisheries Society presents this 1992 Northeastern Pacific Chinook and Coho workshop. The chinook committee of the Idaho Chapter, with the leadership of Dave Cannamela, created this workshop for you.

Their objective was to bring new ideas forward and have an opportunity for active discussion. They gave the gift of their personal time, their creativity, and genuine concern for "Declining stocks." They offer this meeting as a way to refine the biological truths we use in our work, in the spirit of professional fishery scientists.

When the committee asked me to provide the welcome, I considered its purpose. I am here to verbally set the stage for what they prepared, to set the stage to discuss the theme: Salmon Management in the 21st Century: Recovering stocks in decline.

This theme, with minor modifications, might have been the theme of an American Fisheries Society meeting in 1892, 100 years ago. Yet we have a challenge like no other generation facing declining stocks. Over 100 years of professional effort did not prevent the decline of Pacific salmon stocks.

Is there hope for rebuilding? Dale Becker suggests there may not be, in a recent issue of "Fisheries". He states "The ultimate effect of human activity is extinction," and argues human population expansion will reduce species diversity on the planet.

We must believe there is some hope, or at least be willing to try, or we would not be here today. I would suggest Mr. Becker may be correct, but that the American public expects fisheries scientists to find a way to prove him wrong. Each year the public expects more refined information about how far they can squeeze the biological system.

Jim Lichatowich suggests fisheries management is first, an allocation process, between this generation and future generations. This allocation process includes what he calls a natural resource economics, in essence ecosystem management.

Any management or allocation plan requires a time frame, in this case a definition of "future generations." Does it mean your children or sometime beyond?. Larry Ecohawk, Idaho's attorney general, suggests that "future generations" means seven generations ahead of our time.

I suspect that if we look at fisheries management and the rebuilding of salmon stocks today and seven generations in the future, it clarifies our mission. It clarifies our work because we must examine what will endure. The only enduring contribution I can see for fisheries scientists is clearly defining and communicating biological truth.

I suggest we have three responsibilities as fisheries scientists:
1) to seek biological truth with rigorous review,
2) to be intolerant of partial presentation or misleading presentation of data,
3) communicate clearly the risks society takes when the public chooses to eliminate or compromise biological systems.

This workshop offers you two of the tools you need. It offers an opportunity to refine your understanding of several fisheries biology issues, and provides a forum to identify partial or misleading information. The members of the chinook committee created this agenda with time for discussion. I encourage you to take advantage of the discussion time.

Each of you gave up something to be here, your regular work duties, time with your family, a little cash. Because you gave up something to be here, I must assume you are committed to participate, to learn as much as you can. I applaud each of you, for you must give of yourself in some way to gain knowledge. I assume you are here in what I think of as the true spirit of the AFS.

You have come as a fisheries scientist. You have left your pride, and your politics at home. You've left your employer's agenda at work. And you look at each individual here as a peer, whether they are a supervisor, employee, professor, or student. I can only assume that your intent is to speak candidly from your experience, listen carefully to others, question everything, and offer critique to anyone as you would to your most respected colleague.

Use these next two and a half days well, as you refine the tools you need to rebuild declining salmon stocks in the 21st Century.

CONSERVATION BIOLOGY, SALMON RECOVERY, AND THE "REAL WORLD"

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INTRODUCTION

The great population geneticist Theodosius Dobzhansky once stated that "Nothing in biology makes sense except in the light of evolution". This statement exemplifies the fact that evolution is the single unifying concept in biology and has been so for well over 100 years. Evolution is the most powerful approach to understanding the incredible complexities and otherwise bewildering diversity of the natural world. This was emphasized by Ehrenfeld (1991) who stated that "The biosphere is a system, or a set of systems, with many millions of elements that are changing in time and are affected by myriad local irregularities and discontinuities and by countless historical singularities. As yet, there is no single comprehensive theory besides evolution that takes it all in. Quite possibly there never will be."

The decline of salmonid fishes is a biological problem, occurring in real ecological and evolutionary settings. Consequently, evolutionary and ecological considerations should be the centerpieces of any conservation and recovery program. First and foremost, salmonids must be viewed as biological entities with an evolutionary history and ecological requirements, rather than simply harvestable, semi-domesticated species that can be produced ad infinitum in hatcheries. They must be retained at some level in wild, self-perpetuating populations. To accomplish that, resource managers must understand the natural genetic structures and ecological requirements of populations, along with new pressures on natural systems imposed by harvest.

Genetic diversity is at the heart of ecological adaptation and evolutionary change, and understanding it is one of they keys to successful management with a single-species focus. Consequently, we should take a closer look at genetics--why genetic diversity should be conserved and what diversity really means.

WHY CONSERVE GENETIC DIVERSITY?

There are at least three biological reasons why genetic diversity should be conserved in natural populations. First, the Fundamental Theorem of Natural Selection states that "the rate of increase in fitness of any organism at any time is equal to its additive genetic variance in fitness at that time" (Fisher 1930). This means that the ability to adapt to changing circumstances is directly proportional to the amount of genetic diversity carried in a population. Loss of diversity is equated with loss of adaptability. A good working guideline, then, is that maintenance of genetic diversity in populations is good, and losses of diversity through drift, inbreeding or other means is bad.

Second, there is a consensus opinion (although not unanimous) among population geneticists that individual fitness increases with genetic variation (often measured as heterozygosity level). Thus, growth rate, fecundity, disease resistance, developmental stability, and metabolic efficiency may all be positively correlated with heterozygosity level (Allendorf and Leary 1986). Consequently, loss of heterozygosity may lead to a decline in fitness-related characters.

Third, a tremendous biotic resource is eroded as genetic variation disappears. The global pool of genetic diversity contains all of the information for all biological processes on the planet. Loss of this diversity will likely decrease the ability of organisms to respond to environmental change in a biologically meaningful way and will also discard biological processes potentially useful to humans. This may be the strongest argument for conservation, whether from a narrow and selfish utilitarian perspective or a more enlightened ecological outlook.

PARTITIONING OF GENETIC DIVERSITY

Species almost never exist as single, randomly interbreeding populations. Instead, genetic differences usually exist among populations. Such geographic distribution of genetic diversity is an important component of overall genetic diversity.

Consider a hypothetical species consisting of three populations (Figure 1). Genetic diversity in the species consists of within-population diversity (mean heterozygosity level within a population) and among-population divergence (mean genetic differences among populations). A simple genetic model of this diversity is $H_t = H_s + D_{st}$, where H_t = total genetic variation in the species, H_s = average diversity within populations, or average local heterozygosity, and D_{st} = average divergence among populations across the total species range (Nei 1973, 1975). Divergence among populations arises from random processes (founder events, drift, bottlenecks, mutation) and local selection.

The important feature here is that a species' total genetic variation may be hierarchically partitioned into component parts based on any objective geographic criterion; the most elementary criterion is within-verses among-population distinctions. From a hierarchical analysis, one can determine how variation is spatially distributed and thus define areas of particular conservation interest. If a high proportion of total diversity occurs among populations, then we need to maintain them as distinct units; if most of the variation occurs within populations, it is less important to maintain geographic distinctions and more critical to maintain high within-population heterozygosity.

CONSERVATION UNITS

If we can accept the importance of genetic variation to conservation of endangered species, a difficult question confronts the resources manager: What should we conserve? What are the units of genetic conservation? Obviously we cannot save everything, so where do we concentrate our resources to protect the greatest amount of genetic diversity?

A suggested solution indicates that a population is distinct if it represents an Evolutionarily Significant Unit (ESU) of the biological species (Waples 1991). An ESU is defined as a population that a) is reproductively isolated from other conspecific population units, and b) represents an important component in the evolutionary legacy of the species.

Adoption of the ESU definition is satisfying conceptually, because it recognizes the evolutionary role, importance, and fate of populations. However, it is operationally difficult for tow reasons. The first part of the definition is essentially a deme, which is difficult or impossible to

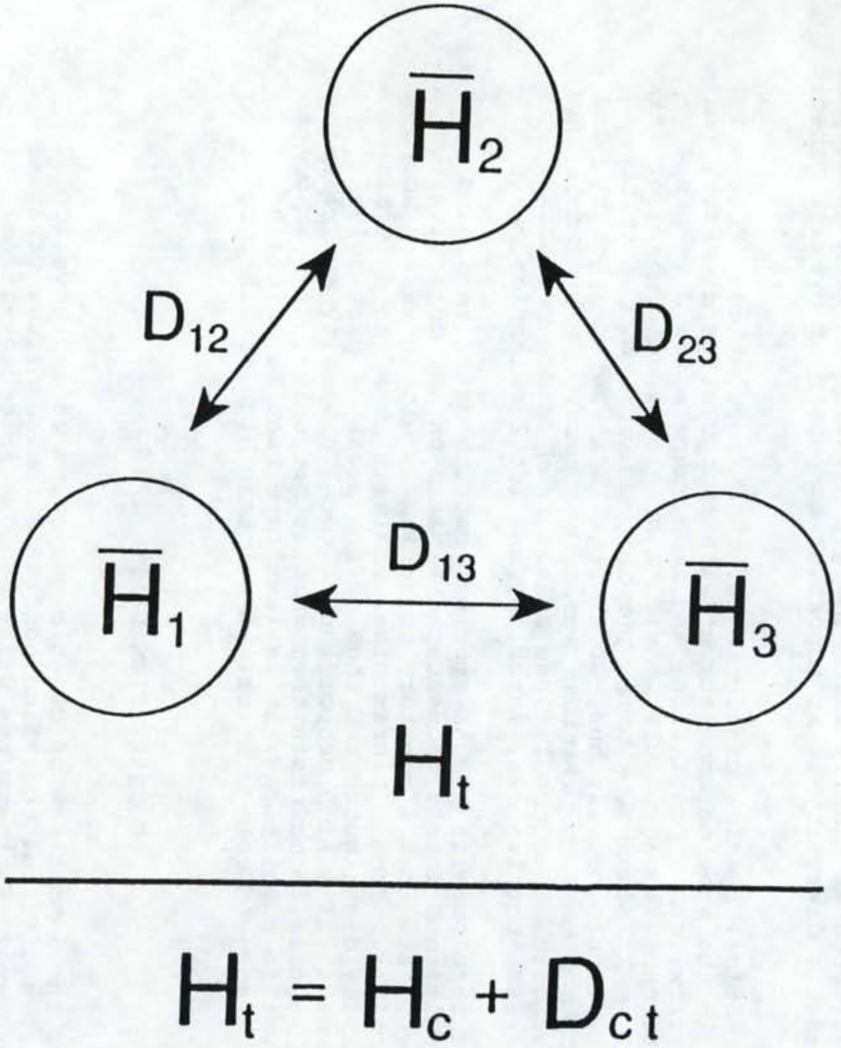


Figure 1. Partitioning of genetic diversity into within- and among-population variation. This hypothetical situation represents three populations of a species, each with some level of within-population heterozygosity (H_1 - H_3); mean heterozygosity is H_c. Among population divergence is represented by the arrows between populations; mean divergence is D_{ct}.

delineate in nature, especially in species like salmon, which do not occur in discrete and isolated habitat units. The second part requires a subjective assessment of the population of concern, relative to other populations of the species, which are equally undefined under the first part of the definition. However, an expansion of the simple genetic model presented above may offer a solution to these problems and allow adoption of the ESU approach in a functional, as well as conceptual, sense.

In the simple model of genetic diversity presented in Figure 1, the divergence component of diversity can be further subdivided based on any geographically reasonable hierarchy. All levels in a geographic hierarchy are potential genetic resources and may contribute to individual fitness and local population adaptation and should be so recognized in recovery (Currens et al., in review).

As an example, a clear and objective hierarchical geographic structure exists in salmon habitat in the form of stream orders. Every point in a watershed can be clearly identified in a stream order classification system (Figure 2). A likely hierarchical genetic structure also exists at the level of stream order, with the following hierarchical genetic model: total genetic diversity (H_t) consists of heterozygosity within local populations (H_c), plus average divergence among populations within first order streams (D_{p1}), plus average divergence among different first order within second order streams (D_{12}), plus average divergence among second order within third order streams (D_{23}) and so on through all stream orders. This approach would identify the magnitude of genetic diversity at each level of stream order and help to define the smallest reasonable conservation unit.

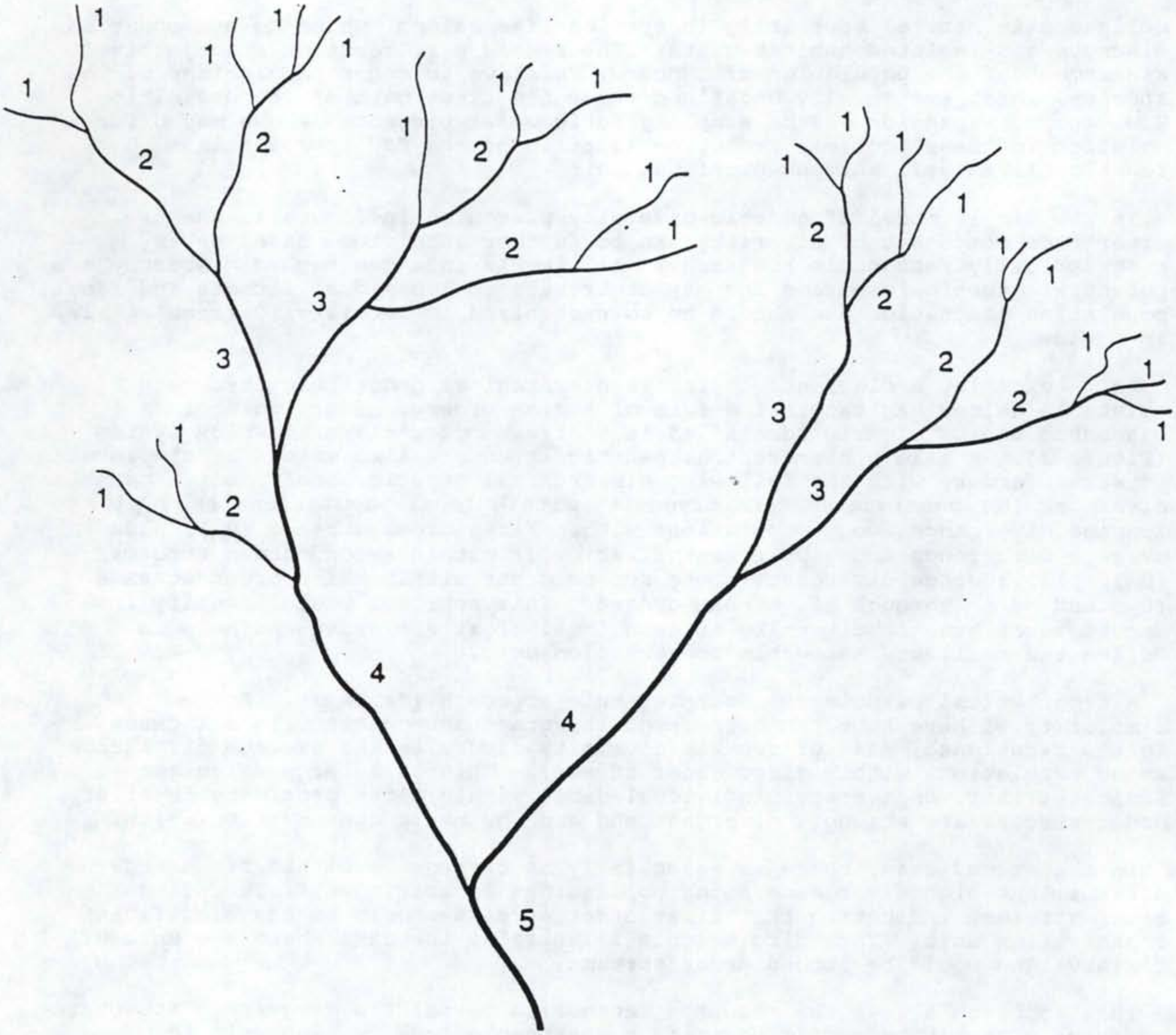
A hypothetical example demonstrates this approach (Table 1). For simplicity, I have kept the heterozygosity component constant in all cases. In the first case, 14% of genetic diversity occurs as the average divergence among populations within first order streams. This is a large value and indicates that, on average, individual demes within close proximity in first order streams are strongly divergent and are the basic conservation units.

In the second case, there is essentially no divergence within first order streams, but high divergence among populations inhabiting different first order streams, indicating that first order streams should be the significant conservation unit. The third example illustrates the case where the unit of conservation would be second order streams.

This approach allows the resource manager to reveal the geographic structure of genetic variation and to identify a reasonable conservation unit to concentrate efforts. The conservation unit would be the lowest level in the hierarchy at which a significant amount of genetic divergence occurs. "Significant" remains somewhat subjective, but I would suggest that genetic diversity of a few percent would be significant and necessary to protect. This method would also reveal any especially divergent populations or unique alleles that might need protection, and would allow maintenance of a natural hierarchical genetic structure.

THE ROLE OF HATCHERIES IN SALMON CONSERVATION

It is my firm belief that the hatchery production approach to salmon recovery is counter-productive to the goal of recovery and should be reconsidered. This position has been previously published elsewhere (Meffe 1992) and the interested reader should consult that source for complete details. Much of the information here has been extracted directly or slightly modified from that work.



$$H_t = H_c + D_{c\cdot 1} + D_{1\cdot 2} + D_{2\cdot 3} + D_{3\cdot 4} + D_{4\cdot 5}$$

Figure 2. A geographic hierarchy in streams based on objective stream order classification. Genetic diversity of salmon in these streams may also be structured in a hierarchical manner. Total genetic diversity may be partitioned into heterozygosity within populations (H_c), mean divergence among samples taken within first order streams (D_{cl}), mean divergence among samples taken within first order streams (D_{cl}), mean divergence among samples across first order within second order streams (D_{12}), and so forth.

Table 1. Three hypothetical outcomes of genetic hierarchy studies in a Pacific salmon species.

Case	1	2	3
H_e	63	63	63
D_{cl}	14	0.1	0.1
D_{12}	10	15	2
D_{23}	6	7	19
D_{34}	4	9	11
D_{45}	3	6	5

There are at least six reasons why a hatchery-centered approach to salmon conservation and recovery should be seriously re-examined and possibly discontinued in its present form:

1) The data demonstrate that hatcheries are not solving and probably will not solve the problem of salmon declines. Salmonids continue to decline throughout the Pacific Northwest, despite decades of hatchery production, and millions of dollars expended (Federal Register 1991; Hilborn 1991, 1992; Matthews and Waples 1991). It is obvious that this is not a reasonable solution to the problem, as it clearly is not working.

2) Hatcheries are enormously costly to run. Highly limited state and federal resources spent on hatcheries could be redirected to local and ecosystem-level habitat restoration, or prevention of further declines through land purchases. The latter would also benefit other species and maintain ecosystem services in the region.

3) Hatcheries are not sustainable in any sense of the word because they require continual energetic and monetary inputs, and are only a piecemeal, year-to-year approach to the problem. In 50, or 100, or 1,000 years, for economic or other reasons, hatcheries will cease to operate, and the system will collapse. A long-term, self-sustaining solution is needed.

4) Hatcheries are a biologically unsound approach to management that result in negative genetic changes in natural populations. The most basic concept in quantitative genetics is that an individual's phenotype reflects genotypic and environmental influences, plus interactions of these factors; hatcheries have never demonstrated the ability to properly manage either the genotype or the environment in any way that reasonably approximates nature. They have in fact been demonstrated to create genetic problems (Allendorf and Ryman 1987; Hindar et al., 1991).

5) Hatchery production leads to greater harvest of salmonids, including those from natural populations, resulting in decline of the stocks being protected. Hilborn (1991) stated, "There is wide concern throughout the Northwest that we have allowed our fisheries harvest rates to match the potential productivity of hatchery stocks, causing wild stocks to be overfished." He continues with an example: "Just north of Puget Sound, ... harvest rates on coho salmon are as high as 95%, sustainable only by the most successful hatchery stocks. The net result of these high harvest rates is that as hatchery production has increased, wild stocks have declined. But the

Canadians have no more coho now than they did 15 years ago. They have swapped hatchery fish for wild fish." Successful hatchery production provides a psychological license to increase harvest rates, which reduces wild stocks, thus defeating the initial purpose of hatcheries.

6) Hatcheries are at best a palliative that conceals the real problems and dangers facing this valued resource. This, I believe, is the most serious objection to the hatchery approach. By financially supporting hatchery production as a mitigation practice, the hydropower companies and other development projects that are largely responsible for environmental degradation can "buy out" of their moral responsibilities for salmonid losses and habitat destruction by demonstrating their concern for and dedication to the declining resource. They, along with the fishing industry, have created a popular mythology, which many fishery managers and the public have bought, that hatcheries are a viable solution to environmental destruction and loss of salmon. This is an insidious deception of the public trust, and this particular mythology must be challenged and exposed as a lie. The taxpayer and voter is deceived into believing that technological advances can simultaneously allow environmental degradation and sustained production of a valuable resource. The public is also led to believe that their native salmonids are in reasonable condition, in good hands, and that someone is watching over that resource.

The hatchery approach to salmon conservation is a good example of what Lewis Thomas (1974) called "halfway technology", a reference to medical practices that orient toward treating symptoms, rather than eliminating causes, of disease. Halfway technology in salmonid management recognizes the symptom (fewer fish) and treats that symptom (grow more fish) without concerted efforts toward identifying and eliminating the underlying causes (environmental destruction; overexploitation). Halfway technology for salmonids ignores the many causes of declines, focuses on reduced numbers, and invents technological methodologies to increase those numbers. This is the same problem that Nat Frazer (1992) recently pointed out relative to headstarting programs for sea turtles: hatchlings are grown in captivity and released by the thousands without addressing the major causes of turtle decline such as adult mortality in shrimp nets and destruction of or poaching on nesting beaches.

What then do I suggest for the management of salmonid fisheries? A reorientation of recovery efforts from the <u>symptoms</u> of decline to the <u>causes</u> of decline. Running a multitude of expensive hatcheries while ineffectively dealing with turbines and dams, dumping of mine tailings, sedimentation from road-building and logging, overgrazing of watersheds, overharvest, and genetic homogenization of populations, is halfway technology at its worst. Valuable and limited resources are being invested into a dead-end technology, while the causes of the problem continue unabated and even increase. Hatcheries may placate some individuals in the short-term, may please politicians, and may even sustain some fish populations for the present, but will not rejuvenate a dying system without a great deal of effort put into the fish's environment. This requires complete reevaluation of our basic philosophies with respect to nature, technology, and resource use. The only sensible basis for management of salmonid fisheries (or any species in nature) is a clear understanding and acceptance of the evolutionary history and ecological requirements of the species, and adoption of measures that work within the constraints of that history.

I am not implying that hatcheries are all bad; they may in fact be able to play a valid role in recovery of some salmonid populations. However, the purpose and operational philosophy needs to change from a production mission to that of genetic conservation (Currens et al., in review). Hatcheries can potentially play a critical role in genetic rehabilitation of depleted or genetically degraded stocks if they adopt strict genetic operational guidelines (Meffe 1986; Allendorf and Ryman 1987; Kapuscinski and Jacobson 1987; Kapuscinski et al. in review).

PHILOSOPHICAL FOUNDATIONS AND THE "REAL WORLD"

It is often stated that ideas such as those presented here are utopian and ignorant of how the real world operates. Proper genetic management, reassessment of expensive and detrimental hatcheries, and even removal of selected (and sacred?) dams will never be a reality because of powerful economic and political forces that cannot be overcome and are beyond challenge. People with crazy ideas such as these are accused of not living in the real world; they are simply mush-headed ivory tower academics who are divorced from reality. But all of this assumes that the "real world" is accurately represented by the political and economic forces that rule. That is the notion that should be under discussion.

I flatly disagree that the real world consists of unchangeable economic and political forces that we all must work around. This view implicitly, if not explicitly, denies the existence of natural laws or assigns them a secondary and diminutive role. Natural laws usually are ignored completely in standard economic models, for example (see Daly 1991 for a comprehensive critique of traditional economic models). In the economic models followed by the United States and other developed countries, resources are assumed to be infinite or totally substitutable, and waste products are assumed to be irrelevant. This is the "real world" under which developed countries operate, a nd the model which developing countries are trying to emulate. Even casual reflection reveals how patently absurd such a view is.

Our socio-economic systems are in fact new inventions, several hundred to 1,000 years old and are totally artificial constructs of humankind. They may work well in the short-term and under conditions of low human population density, with abundant resources and free ecosystem services (such as clean air and water or removal of pollutants by trees). With high human densities and fewer resources, however, natural systems begin to collapse, as we have seen over the last few decades, as evidenced by this and many similar symposia directed toward salvaging a few pieces of biodiversity.

I instead recommend that, as resource managers, we define the "real world" as the physical, chemical and biological laws that have operated for not hundreds, but billions of years. Natural selection is the real world. The first and second laws of thermodynamics are the real world. Material and energy flow through ecosystems is the real world. Heritable genetic variation is the real world. All of these have been operating in their present form for billions of years. By comparison, the so-called real world of politicians, economists and other managers of our world is a trivial and fleeting accident in evolutionary time, and is meaningless by comparison to natural laws that are incontrovertible and inviolate. I can in fact violate an economic or political law if I want: I can steal money from a bank and possibly not get caught; I can murder and maybe get away with it; I can cheat on my taxes. But I cannot violate natural laws: I cannot by-pass entropy; I cannot simply ignore gravity if it does not suit me; I cannot consistently destroy habitat, block rivers with dams, clear-cut old growth forests and expect salmon to maintain sustainable and profitable populations.

The human species has developed the notion that we can and should control nature, even re-model nature, to our own ends. Ultimately, of course, that is a ludicrous and even childishly naive concept. I can do no better than quote wise individuals who have argued this point before. Francis Bacon said "Nature is only to be commanded by obeying her." Rachel Carson closed the classic "Silent Spring" (1962) with these words: "The 'control of nature' is a phrase conceived in arrogance, born of the Neanderthal age of biology and philosophy, when it was supposed that nature exists for the convenience of man." Finally, in "The Arrogance of Humanism" (1981) David Ehrenfeld stated, "In no important instance have we been able to demonstrate comprehensive, successful management of our world, nor do we understand it well enough to be able to manage it in theory." All of these people emphasized limits to our attempted control of nature and technological mastery of it. The message once again is to manage within evolutionary and ecological constraints.

We cannot re-invent natural biological laws to suit our image of short-term economic gain, four-year political cycles and perpetually expanding economies. This flies in the face of everything we know about natural law, not to mention common sense. Yet, for some reason, we as a society have allowed politics and economics to emerge as THE ONE reality, the guiding principles that direct all facets of western life, while conveniently ignoring the natural world and natural laws.

The response of many is "this can't be changed; there is too great a momentum." I argue that it <u>must</u> be changed if we are to retain so much as a fraction of the biodiversity and natural processes necessary to maintain even modestly functioning ecosystems. I agree there is a huge momentum to be confronted, but it is minuscule compared with the implications of ignoring natural laws and their forces. Continued disregard of natural law simply cannot be sustained in the long run, and agencies and political systems can and must be changed to recognize this.

So what can we do? If we wish to retain a semblance of ecosystem function, biological diversity, and long-term sustainability, not only of natural resources but also for a reasonable quality of human life, we need a philosophical renaissance that recognizes supremacy of natural law over artificial, human institutions, or what those in economic and political power tell us is the "real world". We must recognize limits to our control of nature, and limits to the ability of natural systems to absorb abuse upon abuse yet still provide the services we expect of it, including abundant and healthy salmon populations.

How can this be done? The answer, I believe, lies in education at all levels: of low-, mid-, and high-level managers, of politicians, of economists, and especially of the public at large. This can best be done through absolute honesty by scientists and resource managers of the consequences of continued growth in human population and capital. We can no longer sugar-coat what we know is happening to the natural world, thinking that the public likes only good news and will only support us if that is what we provide.

The public makes incredible demands on resource managers, and wants a multitude of uses, often contradictory uses, satisfied simultaneously. However, the public and political leaders are usually ignorant of the ecological realities surrounding resource issues. Rather than accede to unrealistic demands based on ignorance, it is up to fisheries experts to inform of reality rather than conform to fantasy. It has always intrigued me that in the field of natural resource use, an ignorant public and their political leaders can so forcefully guide and drive decisions on specific approaches to be used. This would certainly not occur in medicine, physics, or even automobile repair. Can you imagine a Senatorial Committee deciding the best surgical approach for gall bladder removal, or a cancer treatment? Does the public tell General Motors how to repair a transmission? of course we leave that to the experts. Yet, we as a society let it occur, and even encourage it to occur, in resource use all the time. Why does economics, social pressure and politics wield such power over the real world of nature?

Salmon resource managers have a nearly unique opportunity and an obligation to lead the way in changing these perspectives. Salmon are a resource that people want to retain for economic, esthetics, and recreational reasons. This is not true for spotted owls or snail darters, that many people do not care about and even wish could be destroyed. Salmon are something important to most people, who may be willing to listen. The brutal truth about declining salmon populations needs to be told. We need to say, "Folks, we're losing them fast, and there's little more that technology can do. The answer is not another technological fix - we need to recapture habitats from abuse and destruction. We need a moratorium on further habitat destruction. Tough choices are to be made. You can have salmon back, but it will involve compromise and sacrifice." Ultimately, the price is stabilization and then reversal of human population growth. That topic is beyond the scope of this workshop, but human population control unequivocally must occur, or I believe all will be lost.

Philosophical meanderings are not the purpose of this workshop, and there are more immediate issues to deal with. However, some philosophical direction needs to form the base of any policy decisions; philosophical positions and value systems provide the compass that guides us through difficult territory. To paraphrase a quote I recently heard, "unless we change direction we might just get where we're going." I think it is well worth our time to spend a few moments to define ourselves and our intentions. The best philosophical guide with regard to all recommendations, policy decisions or other efforts on behalf of salmon is to ask yourselves "Is this where we want to go in the larger picture?" Decisions should be carefully made, because you might just get there.

I hope that I have made at least some salmon managers think about the broader view, even though they may disagree with the view or do not like it. I agree it is not a pretty sight, but it is the only view we have, and we need to deal with it honestly and head on if we want sustainable salmon populations in the 21st Century and beyond.

ACKNOWLEDGEMENT

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GENETICS AND POPULATION VIABILITY

SESSION LEADER: ALAN BYRNE,
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GENETIC CONSIDERATIONS IN RECOVERY EFFORTS FOR PACIFIC SALMON

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ABSTRACT

Restoring depleted Pacific salmon stocks to anything near historical levels of abundance is a challenging task. If recovery efforts are to be successful in the long run, it is essential to conserve the genetic resources that are the foundation of natural production. This can be done most effectively through comprehensive and timely measures to address the factors impeding natural recovery (e.g., habitat destruction, blockage of migratory routes, overharvest). Until these basic problems are resolved, efforts to conserve genetic resources should focus on three issues: 1) identifying the appropriate units for conservation; 2) minimizing genetic risks to natural populations caused by fishery enhancement programs; and 3) judicious use of intensive management measures for populations facing high short-term risks of extinction.

Failure of management plans to adequately reflect stock structure was a factor contributing to recent listings of several Pacific salmon populations under the U.S. Endangered Species Act (ESA). If additional listings are to be avoided in the future, it is essential that salmon restoration efforts focus on a scale at least as fine as the evolutionarily significant unit concept that has been developed for ESA evaluations of Pacific salmon. Identifying the appropriate units for conservation is a challenging effort that will require integrating diverse types of genetic, phenotypic, life history, and environmental information.

For many decades, salmon hatcheries have been seen as a means of enhancing fisheries to mitigate declines in natural populations. Although fisheries enhancement may in some cases be compatible with sustained natural production, large-scale artificial propagation programs can have a number of adverse direct and indirect genetic effects on natural populations. Current enhancement programs for Pacific salmon should be reevaluated for compatibility with long-term conservation of genetic diversity.

The risk of extinction can be evaluated using population vulnerability analysis (PVA) or other related methods. Genetic parameters such as effective population size are important in risk assessment, but it is difficult to formally integrate them into existing PVA models, which typically focus on the effects of demographic and environmental variability. Aggressive management actions, including the use of captive brood stocks or other forms of artificial propagation, should be considered for populations believed to be facing high short-term risk of extinction on the basis of genetic PVA.

ASSESSING RISKS OF DECLINE AND RECOVERY CHANCES WITH POPULATION VIABILITY ANALYSIS

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ABSTRACT

Both natural and managed populations undergo fluctuations induced by random variation in the environment. For most populations, these fluctuations can be very large, and may even result in local extinctions. This constitutes an essential feature of the population dynamics which must be addressed in any management effort. The efficacy of possible recovery plans or mitigation strategies can be predicted using tools developed in Population Viability Analysis (PVA), including the risk of population decline and the chance for probable recovery. These measures assess the vigor of the population using a probabilistic language that recognizes the importance of natural variation. With Monte Carlo simulations (made easy by recent software developments), one can make estimates of these PVA measures from quantitative models of the life history of the species. Assumptions about the density dependence or compensatory mechanisms can be taken into account, but we have found in many cases that omitting such effects leads to conservative results which may be suitable in planning efforts. The most important factors that determine risks of decline or extinction for a distributed population are related to inter-population dispersal.

GENETIC INFRASTRUCTURE IN SALMON POPULATIONS AND THE POTENTIAL FOR OUTBREEDING DEPRESSION

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ABSTRACT

Since 1979 we and our colleagues have studied the genetic variability and organization of pink salmon in a small population in Auke Creek, near Juneau, Alaska. Tagging studies indicated that adult fish segregated into early (August) and late (September) segments which spawned either intertidally or upstream. Using a genetic marker, we were able to confirm that Auke Creek pink salmon return to spawn at the same time and place within the stream as their parents do. We also observed that fry from early-run parents emigrated earlier (on the average) than fry from late-run parents. Using a hierarchical mating experiment, we estimated significant heritabilities for time of return within each (early or late) segment. There is evidence that timing of migration is important to survival; temperature records of Auke Creek suggest that variance in timing is also important.

The genetic influence on timing partitions the genetic variation in the population. As a result the population has a higher level of genetic organization or infrastructure. We believe that the adaptedness and productivity of salmon stocks are dependent on genetic infrastructure as well as genetic variability in general. Fisheries management practices, including enhancement of harvests by hatcheries and ocean ranching, potentially reduce genetic infrastructure as they may reduce other levels of genetic diversity.

The existence of genetic infrastructure invokes another concern, the possibility of outbreeding depression. Outbreeding depression is the decrease in fitness or fitness related traits that may accompany introgression or hybridization between populations. Hybridizing two populations that possess different sets of coadapted genomes (which have evolved in response to different environmental histories), may result in a decline of fitness related traits. We have evidence of outbreeding depression from hybrids between even- and odd-year pink salmon. In the F_1 generation, both hybrids and controls had the same survivals. The F_2 hybrids had very low survival and had increased levels of bilateral asymmetry, which is one indicator of a disrupted development and therefore, may reflect a disrupted coadapted genome.

The presence of a genetic infrastructure within populations increases the potential for outbreeding depression and loss of space- or time-based infrastructure. Rational resource management should seek to conserve genetic diversity at all levels. Treatment of salmon stocks as homogeneous units, neglecting with-in stock diversity or infrastructure, will not be adequate to conserve fitness and productivity of these commercially valuable resources.

INTRODUCTION

This manuscript represents parts of several publications in press (Gharrett and Smoker in press [A and B]; Smoker, Gharrett, and Stekoll in press; Gharrett et al. in press) or published previously (Lane et al. 1990; Gharrett

and Smoker 1991).

Genetic effects of hatchery stocks on wild stocks may no be as easily discerned as epizootic pathogen or as over fishing but may be as damaging in the long term. Genetic effects will accumulate over several generations as the genetic composition of the wild stock is modified indirectly or directly from hatchery and management practices. Indirect effects may result from altered natural section regimes, such as increased competition for finite resources between cultured and wild fish. Artificial selection may flow inappropriate harvest strategies that incidentally exploit a particular temporal segment of the natural population.

Direct genetic effects could result from introgression of hatchery-stock genes into the wild population if the two were genetically different. If the rate of introgression were large, the wild population would be "swamped" with alleles derived from hatchery fish. A continuous trickle of genes from the hatchery stock might also alter the genetic composition of the wild population, decrease the fitness of the population if they are maladaptive, or both.

Another potential effect of the introgression of cultured fish into adapted, wild populations is outbreeding depression, the disruption of adaptive gene complexes.

In these papers, we documented the existence of intra-populational genetic structure (infrastructure) and considered the importance of adaptive infrastructure of salmon populations. We rely largely on work on spawning in Auke Creek near Juneau.

PINK SALMON AS A MODEL

That pink salmon are less precise than their congeners in homing - more likely to stray - is a widely held opinion (eg. Glubokovsky and Zhivotovsky 1986), albeit supported by the little rigorous data. If this is true, genetic infrastructure observed for pink salmon should conservatively reflect the extent and importance of genetic infrastructure in the other species. Greater straying rates for pink salmon are supported by the rapid spread of pink salmon throughout the Great Lakes as compared to that of the other Pacific salmon species and the relatively lower interpopulational gene diversity of pink salmon in their native range as compared to other salmon species.

Genetic Infrastructure in Auke Creek Pink Salmon

We have observed temporal and spatial substructure of pink salmon populations in Auke Creek that appears to be both the cause consequence of genetic infrastructure. Timing of the return is bimodal, one peak spawns in August and the other in September; overlap occurs only in years of abnormally low stream flows during late August. Within each temporal component, the majority of fish spawned above tidal influence but a small component spawned intertidally. Progeny return to the same section of the stream and at the same time of the spawning season as their parents.

The genetic basis to the temporal and spatial substructure of the population has been confirmed with the use of a genetic marker. The marker was bred into the late, upstream segment in 1979. Over the next five generations there was no genetic exchange was detectable between the marked segment and the temporally and spatially distinct segments. The lack of exchange indicates that these segments are isolated, at least in the short term. That there is some degree of genetic influence on the time of return and the closely correlated time of spawning has been confirmed from a breeding experiment

designed to estimate the genetic segments of the return. Genetic determinants of return and spawning time probably exist in most salmon populations and have been exploited in salmon husbandry.

The importance of timing to adaptedness also has been demonstrated by observation over several generations that embryos from late-spawning parents survive incubation in Auke Creek substantially better than do embryos from early-spawning parents (Gharret, in prep). Marine survivals during those years has favored the early-spawning fish.

Outbreeding Depression in Auke Creek Pink Salmon?

The genetic structure we observed results from the environmental experience of the population. The life history characteristics of a population are central to its success. Such traits are quite complex and result from the expression of numerous loci. As a result of natural selection, only the most successful combinations of alleles survive in a population over time, only one or a few of the many possible combinations for the species. The variability that persists around these successful combinations ensures that the population will be able to meet the demands of populations that have found different genetic solutions to the environmental challenges may disrupt the successful combinations and decrease the fitness (productivity) of the population. This is outbreeding depression.

We tested the possibility that outbreeding depression could occur in salmon populations by hybridizing even- and odd-year pink salmon in Auke Creek. Because pink salmon have a rigid 2-year anadromous life cycle, there are two genetically isolated lines: one line spawns in even years and the other in odd years. Both lines spawn in Auke Creek and presumably experience very similar average environments.

We hybridized even- and odd-year pink salmon from Auke Creek and examined two generations of returns. If different coadapted genomes evolved in each of these two genetically isolated lines, one might expect to observe outbreeding depression in hybrids between these line (Figure 1). We observed decreased survival in the F_2 generation and increased fluctuating bilateral asymmetry; which may reflect destabilization of developmental canalization, symptoms of outbreeding depression.

IMPLICATIONS FOR OTHER SALMON POPULATIONS AND SPECIES

The genetic infrastructure in population results from directed processes such as selection (both natural selection and artificial selection related to fishing mortality), straying (gene flow), and mate selection and from random processes (sampling error in finite populations) acting on the available genetic material. We would expect, therefore, that in salmon populations, adaptive gene complexes evolve in a population in response to local selection pressures and random chance; each population or distinct segment of a population in response to local selection pressures and random chance; each population or distinct segment of population should be genetically distinct. Although allozyme studies often suggest that the genetic compositions of salmon in different streams within a locality are similar, this does not necessarily mean that intrapopulation structure is not significant, that there are no coadapted gene complexes in a population segment. A reasonable interpretation of these allozyme similarities is that gene flow among the streams slows divergence of these neutral biochemical genetic traits, but that gene flow need not prevent the evolution of unique coadapted genomes when a trait is acted on by natural selection.

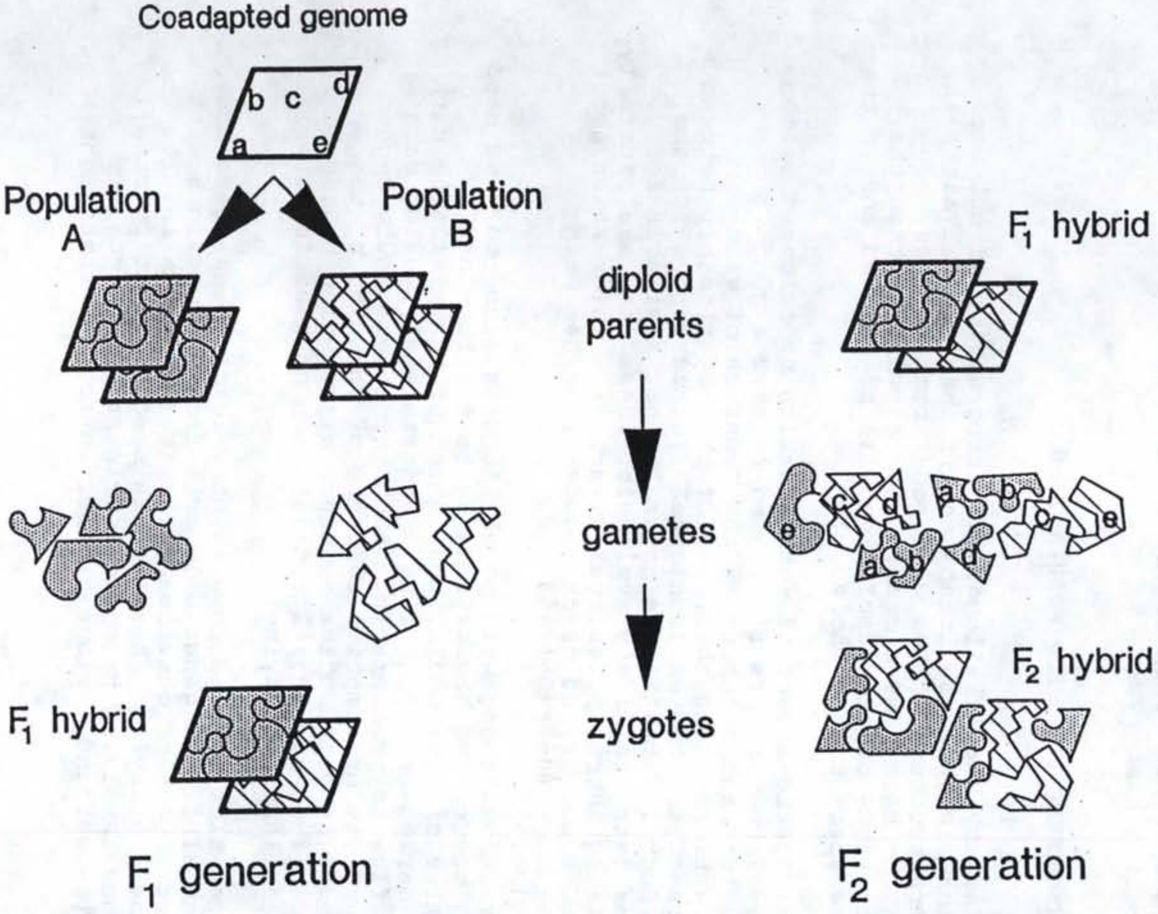

Figure 1. A conceptual example of outbreeding depression resulting from disrupted coadapted genomes. A particular set of interacting (epistatic) alleles at multiple loci (a-e) are selected jointly to produce a well adapted phenotype (coadapted genome) for a particular local environment or environmental series encountered by a salmon population. Alternative, coadapted genomes may be possible (Populations A and B). If two populations possessing different coadapted allele complexes are hybridized, the F1 hybrid individuals will receive two complete sets of coadapted alleles which may or may not affect fitness of the F1 population. However, independent assortment of alleles in the second generation disrupts coadapted genomes in individuals of the F2 and later generations. Disruption of coadapted genomes from outbreeding may reduce depression. (Gharrett and Smoker in press [B.].

Even within a coadapted genome, i.e., within a population segment, variation is important. Such variation is critical for the long term success of the Auke Creek populations; for example, the significant additive genetic variation for timing that we observed and the relationship between timing and survival in different years illustrate the importance of genetic variability to thepopulation. If individuals in a population return to spawn over a span of time, it is likely that at least some of the population will return to the stream when temperature and stream flow conditions are conducive to spawning and survival of developing embryos. This logic can be extended to other important portions of the life history.

CONCERNS FOR WILD POPULATIONS

The primary concern raised by the likelihood of introgression of cultured salmon into wild populations is that resultant genetic changes will alter the genetic structure and infrastructure of the wild population, reduce interpopulation variation, or both. As a result, it is probable that the average fitness - productivity - of the population will be reduced and in the extreme may compromise the species as a whole.

To determine the direct genetic impact of introgression by cultured salmon into wild populations, resource managers first need to have a good understanding of the structure and dynamics of the genetics of wild populations, of the interactions among them, and of the relationship between the genetic diversity present and the environmental fluctuations which the populations experience. Such information is generally not available for salmon. Although allozyme data are widely available, these data are generally inadequate for characterizing intrapopulational structure. Information about population infrastructure is not beyond reach, however, as has been demonstrated at Auke Creek.

Information about interactions among wild populations is also poor; we need to know how much straying occurs and how it relates to environmental variability and the dynamics of the populations. Much of the available data are from observations of marked or tagged fish. Unfortunately, using such markers to estimate gene flow is problematic. Although spawned-out, tagged fish on the spawning grounds is presumptive evidence of spawning by tagged fish on the spawning grounds is presumptive evidence of spawning by tagged salmon it is neither evidence of successful contribution of genetic material nor reliable evidence of homing or straying.

With insight into the genetics of wild populations, resource managers could begin to assess the potential direct genetic impacts from interactions with cultured stocks. The most important question for cultured stocks involve the extent and dynamics of their straying. Researchers need to quantify straying for a variety of situations involving broodstock origins (e.g. local or transplanted), remote releases, and various hatchery practices that might increase straying.

ACKNOWLEDGEMENT

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DNA SEQUENCE VARIATION IN CHINOOK SALMON POPULATIONS

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ABSTRACT

Recent technical advances, namely the polymerase chain reaction (PCR) have revolutionized our ability to collect comparative genetic information at the DNA sequence level. Its ease and utility on small, non-invasive tissue samples makes this approach particularly attractive to studies of the genetic variation in populations. Preliminary sequences of the mitochondrial genomes of different salmon species led to the identification of a hypervariable region suitable for analysis of genetic variation at the population level. A survey of 80 chinook from seven localities indicates that the sequence based approach provides better resolution of mitochondrial haplotypes than previous indirect methods. In addition, geographic structure was apparent in the data which may be useful in stock identification. This approach also opens possibilities hitherto unavailable with other techniques. One such opportunity is the use of long dead specimens from museum collections as a source of historic genetic information. Comparisons of the "old" populations with modern stocks offer an opportunity to directly address historical changes in genetic diversity.

INTEGRATING ARTIFICIAL AND NATURAL PRODUCTION

SESSION LEADER: ED BOWLES,
IDAHO DEPARTMENT OF FISH AND GAME

THE HATCHERY CHALLENGE

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ABSTRACT

The role of artificial propagation in fisheries management is at a crossroads. If hatcheries are to play a valuable role, artificial and natural production systems must be integrated to achieve survival advantage that minimizes adverse effects on natural populations and ensures long term fitness in the natural environment.

The current trend toward "natural" rearing and release strategies embraces this concept but lacks clear direction. An underlying theoretical premise is needed to guide our attempts to meet this new challenge. I believe success of hatchery programs are dependent upon our ability to circumvent some early life history mortality without compromising natural selection processes or incurring hatchery selective mortality. Hatchery programs should be designed to minimize mortality events operating randomly (non-selective) and duplicate mortality events operating selectively on fish in the natural environment. This, in essence, is the primary role of a supplementation hatchery, to reduce random mortality effects in order to produce a net gain in productivity. The difficulty in assessing and implementing this approach should not deter us from embracing this concept as a guiding premise.

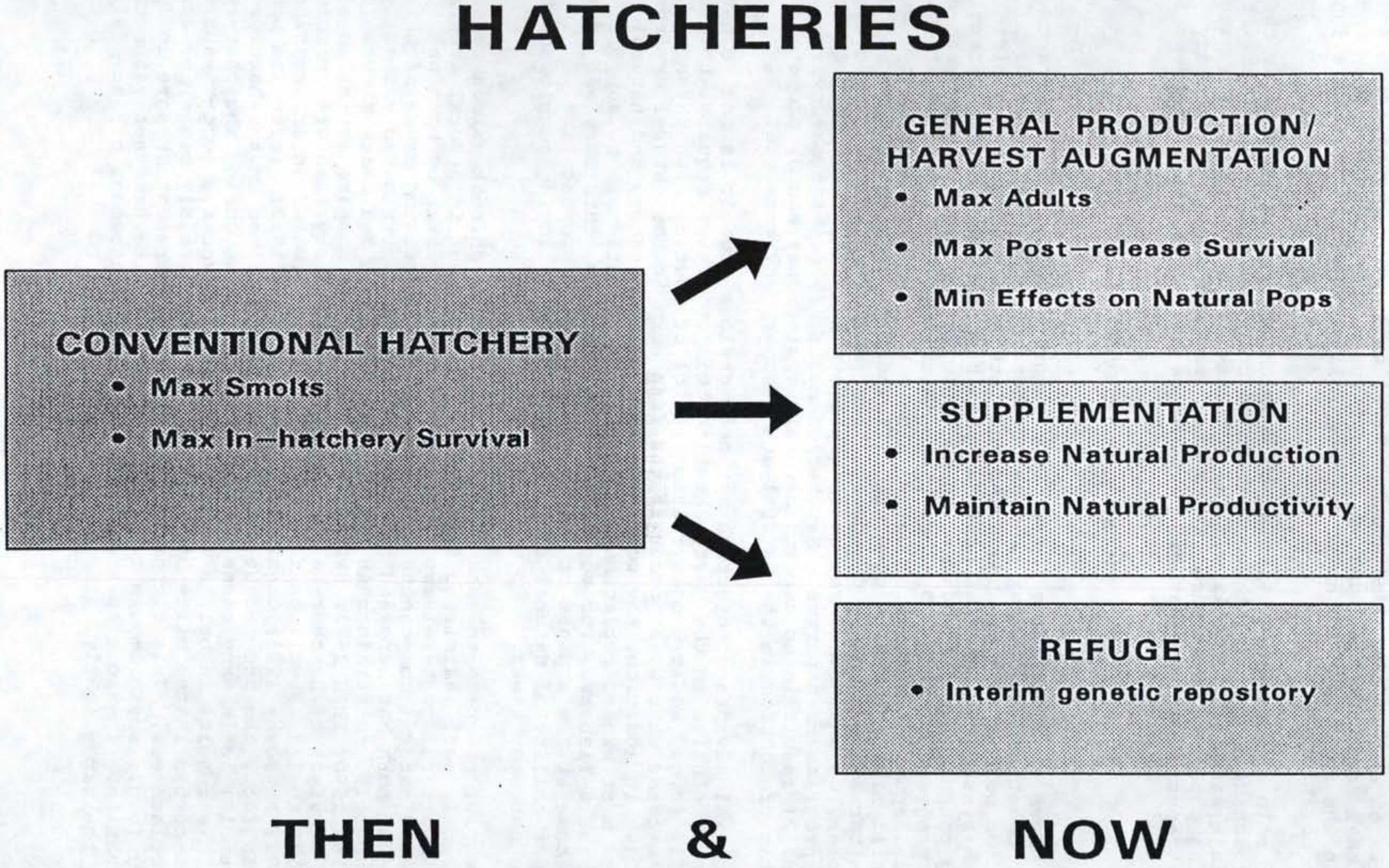
INTRODUCTION

Our success managing sustainable fisheries depends largely on how well we understand and integrate artificial and natural production systems. It is becoming increasingly clear that hatchery and natural production cannot be viewed and evaluated on two separate tracks. They inevitably touch each other, and often in profound ways (Smith et al. 1985; Miller et al. 1990; Steward and Bjornn 1990; RASP 1992). This paper provides ideas on what I view to be the hatchery challenge within this context.

PARADIGM SHIFT

The role of artificial propagation in fisheries management is changing dramatically (Figure 1). Conventional hatcheries of the past were primarily to augment harvest. Their success was often measured by smolt production and how well pre-spawn and in-hatchery survival was maximized. Little emphasis was placed on performance of hatchery fish in natural habitats and the effects of hatchery fish on existing natural populations. Given these objectives, hatchery managers did an excellent job.

The evolving demands on our hatcheries have a much different makeup today. These demands include continuing to provide harvest augmentation, as well as increasing natural production and providing a genetic refuge. Even the traditional role of harvest augmentation has changed dramatically. The objective to maximize smolt production has become subordinate to maximizing adults while minimizing adverse effects on naturally reproducing populations (Herrig 1990; IDFG 1991; NPPC 1992).



HATCHERIES

CONVENTIONAL HATCHERY
- Max Smolts
- Max In-hatchery Survival

GENERAL PRODUCTION/ HARVEST AUGMENTATION
- Max Adults
- Max Post-release Survival
- Min Effects on Natural Pops

SUPPLEMENTATION
- Increase Natural Production
- Maintain Natural Productivity

REFUGE
- Interim genetic repository

THEN & NOW

Figure 1. Changing roles and expectations of anadromous hatchery programs.

Supplementation is a somewhat newer role for hatcheries. Supplementation is the use of artificial propagation to maintain or enhance natural production while maintaining long-term fitness of target and non-target populations (RASP 1992). This objective has added another level to harvest augmentation programs because these adult returns must be able to spawn and produce progeny that persist in the natural environment and are fully compatible with existing natural populations.

Another role of hatcheries similar to supplementation is to provide refuge. This approach uses artificial propagation to hold and coddle fish until the "world" is a safer place to live. The objective is still to provide naturally reproducing populations at some future time, so this period of hatchery residence cannot alter the natural performance characteristics of these fish.

HATCHERY OBJECTIVES

All of these expanded roles for artificial propagation have important natural production objectives; success of these programs is, therefore, dependent on our ability to understand and integrate natural and artificial production systems. Fisheries professionals have focussed effort recently on assessing potential risks and developing criteria for measuring success of hatchery programs (CBFWA 1990; Bowles and Leitzinger 1991; Hard et al. 1992; RASP 1992; Kapuscinski et al. 1993). This is an important step, but I believe we are still missing a solid theoretical foundation or premise from which to develop alternative hatchery design and management strategies. To develop this premise, we must first recognize there are several common objectives that hatchery programs must pursue. Hatcheries must: 1) provide a survival advantage; 2) result in no harm to natural populations; and 3) promote sustainable production in the natural and hatchery environments.

The survival advantage objective may seem obvious, but it is one we have found pretty elusive in the past. Meeting this objective requires that the net benefit (e.g., adult progeny:parent ratio) of routing fish through a hatchery exceeds what would occur if the fish were in the natural environments (Figure 2). In addition, the combined survival of hatchery and natural components must exceed replacement (progeny:parent ratio >1) to support recovery and a fishery. For supplementation programs, this survival advantage must not come at the expense of the purely natural component (i.e., replacement ability of natural fish is not reduced following supplementation). These are not easy tasks.

For example, programs to enhance upper Snake River chinook salmon runs have done extremely well maximizing in-hatchery survival but in general are falling dismally short for post-release or smolt-to-adult survival of both the hatchery and natural components (Figure 3). The overriding cause of this low survival for both hatchery and natural fish has been attributed primarily to passage and flow constraints in the mainstem Columbia and Snake rivers (IDFG 1985; CBFWA 1990; IDFG 1991; NPPC 1992). But, if we compare estimated smolt-to-adult survival of hatchery and natural fish, hatchery fish are performing at only 1/5 to 1/3 as well as their natural cohorts (based on information from LSRCO 1991; Petrosky 1991; Cannamela and Kruse-Malle 1993; Kiefer and Lockhart 1993). In fact, the tremendous survival advantage gained in the hatchery (egg-to-smolt) is in many cases nearly completely lost during the post-release stage. This results in only a small net survival advantage (egg-to-adult) over what occurs in the wild--both of which are dangerously close, or even below, replacement. This disparity does not reflect recent changes in fish husbandry (e.g., erythromycin treatment) because adults have not yet returned for evaluation. I expect advances in fish health management may have already narrowed the gap slightly.

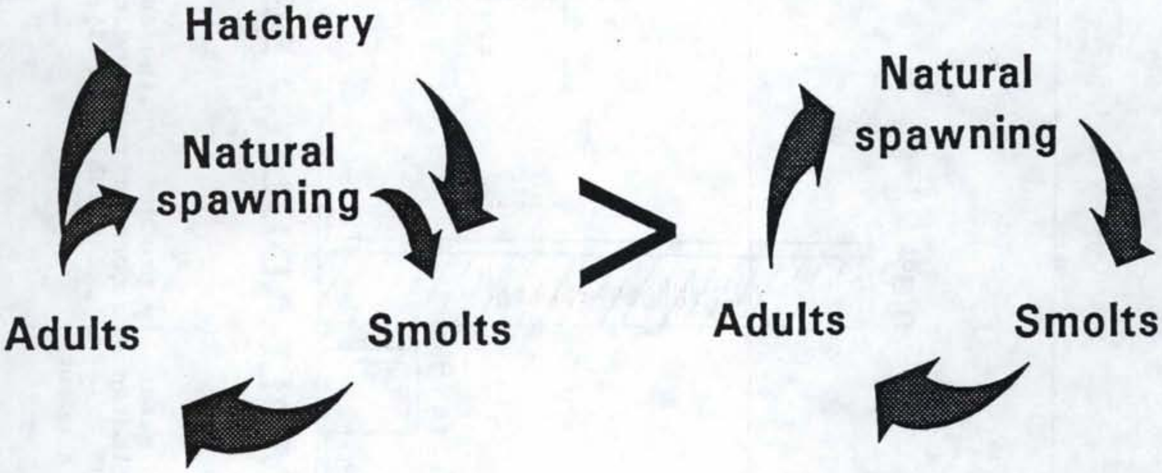


Figure 2. Fish life cycle showing that addition of hatchery component must provide a net survival advantage over what would occur without the hatchery component.

HATCHERY & NATURAL

SURVIVAL ADVANTAGES

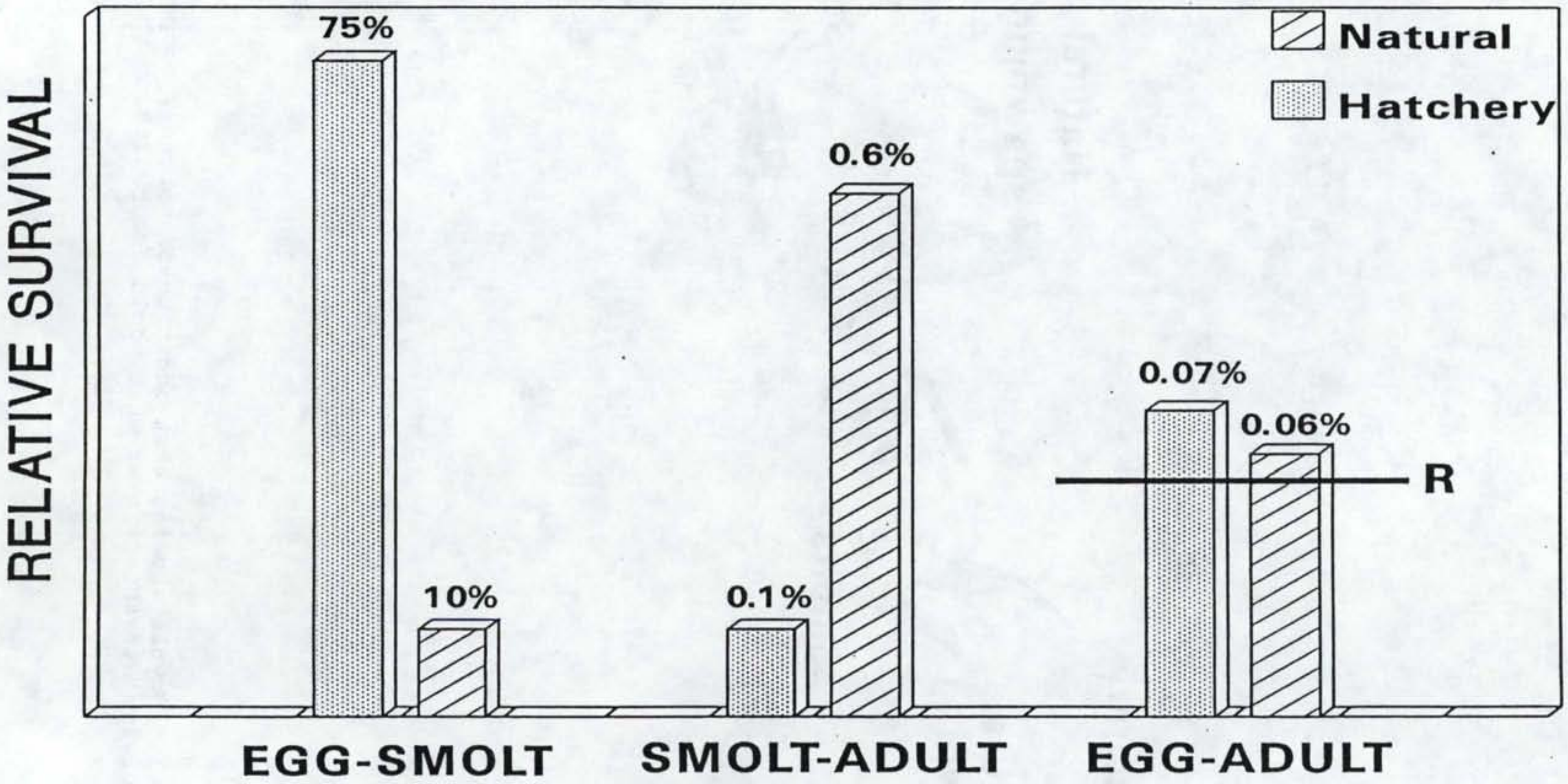


Figure 3. Relative survival of hatchery and naturally produced spring/summer chinook salmon from upper Snake River tributaries and hatcheries. Estimates are generalized from numerous sources (LSRCO 1991; Petrosky 1991; Cannamela and Kruse-Malle 1993; Kiefer and Lockhart 1993). Dark line (R) represents approximate survival rate necessary for replacement.

It is unrealistic to expect hatchery-reared fish to survive in the natural environment as well as wild/natural fish. But, to maintain a satisfactory survival advantage, hatchery fish do not have to. In some cases, only a slightimprovement in post-release survival is necessary because of the large survival advantage gained during hatchery residence. Perhaps a realistic target would be to have hatchery fish survive at half the rate of natural fish during the post-release stage. New hatchery management strategies to improve post-release survival may result in a reduction of in-hatchery survival, but this is worthwhile if the overall survival advantage (egg-to-adult) is enhanced.

As we work toward gaining the necessary survival advantage, we must remember that two additional objectives must also be met for success. One of these is to ensure that hatchery programs do not harm existing wild or natural populations. This objective is pertinent to all artificial propagation programs and is not an easy task to implement or to monitor and evaluate. Potential adverse impacts may occur through genetic, ecological, behavioral, and pathological avenues (Steward and Bjornn 1990; Busack 1990; Miller et al. 1990; Bowles and Leitzinger 1991; RASP 1992; Kapuscinski et al. 1993).

The last objective, ensuring sustainable production, is also crucial for success. Hatchery products must have the ability to persist in the natural environment, or the hatchery environment for harvest augmentation programs. Adults must have the ability to return to target production areas, spawn successfully, and produce viable progeny that will spawn successfully. Once again, this is not an easy task and may be impossible for upper Columbia River basin stocks, given the tremendous mortality occurring during juvenile emigration through the hydropower system.

HATCHERY MANAGEMENT ACTIONS

I believe hatchery management can be separated into two main areas of focus toward meeting these objectives, recognizing that benefits from these actions will be severely limited without concurrent improvements in mainstem Columbia and Snake rivers survival conditions. The first area is broodstock selection and spawning strategies. This has been, or is being, addressed through several forums (Bowles and Leitzinger 1991; RASP 1992; Kapuscinski et al. 1993), so I will not discuss it here.

The second area focusses on optimizing rearing and release strategies. This component has been recognized as important in addressing the three objectives to increase post-release survival, minimize adverse effects on existing natural populations, and ensure reproductive success and long-term fitness. The current trend toward "natural" rearing and release strategies (Olla and Davis 1989; YKPP 1990; Cannamela 1992; RASP 1992) embraces these objectives but lacks clear direction. Essentially, the concept is to mimic natural rearing (e.g., cover, predator, feeding, etc.) and emigration cues (e.g., timing, size, flows, smoltification, etc.) in the hatchery. The paradox of this approach becomes evident if we go to the extreme - the fish are back in the stream without a hatchery.

There must be a logical reason to pursue and maintain the survival advantage gained by routing fish through a hatchery. We need to develop a theoretical foundation from which to proceed. This will help avoid an overwhelming "shotgun" approach, i.e. where all possible aspects of natural rearing are applied in every conceivable combination. Without this premise to help identify and prioritize appropriate spawning, rearing, and release strategies, this objective becomes daunting.

THEORETICAL PREMISE

There are four main components of this hatchery premise. The first two are the most critical: remove random mortality occurring in the natural environment and mimic selective mortality operating naturally in streams. If these are accomplished, they actually incorporate the next two components: minimize artificial selection and provide training opportunities.

Operation of this idealized hatchery model is conceptually simple (Figure 4). Random mortality events occurring in the wild are removed from the fishes experience without producing any artificial selection in the hatchery. What we are trying to mimic in the hatchery are those events that either apply selective pressure in the natural environment (e.g. competition for food and space, thermal characteristics, etc.), or training opportunities that modify behavior in the natural environment (e.g., predator avoidance, natural feeding regimes, etc.). Thus, the role of most hatchery programs is not to maximize in-hatchery survival, or to make the hatchery as stream-like as possible; it is to remove as much random mortality effects as possible without having any other influence on the fishes natural life or experience.

This ideal is obviously impossible to achieve. But as a conceptual model, it provides a theoretical basis for why we are pursuing artificial propagation in first place, and can help us assess whether or not artificial propagation is an appropriate management strategy for a given situation. This premise also allows us to examine potential operations within this context and prioritize strategies based on their assumed importance in meeting these four components.

It is unrealistic to assume we can accurately describe the genetic and behavioral influence of every natural mortality event - and even more unrealistic to assume we can mimic the subtleties of these events in a hatchery environment. The difficulty in assessing and implementing this approach should not deter us from embracing this concept as a guiding premise from which to streamline and focus our efforts. This approach does require a better understanding of our natural production systems and the nature of natural selective pressures molding and chiseling anadromous stocks.

Adopting this approach should also help avoid implementing hatchery strategies that mimic non-selective natural conditions, simply because they are "easy" strategies to implement. Conversely, it will also help overlooking significant selective events simply because they are too difficult to implement.

I believe that as we allow hatchery managers and biologists to springboard from this premise, we will make steady progress toward meeting hatchery objectives to provide a survival advantage, produce no harm to existing populations and insure sustainable production.

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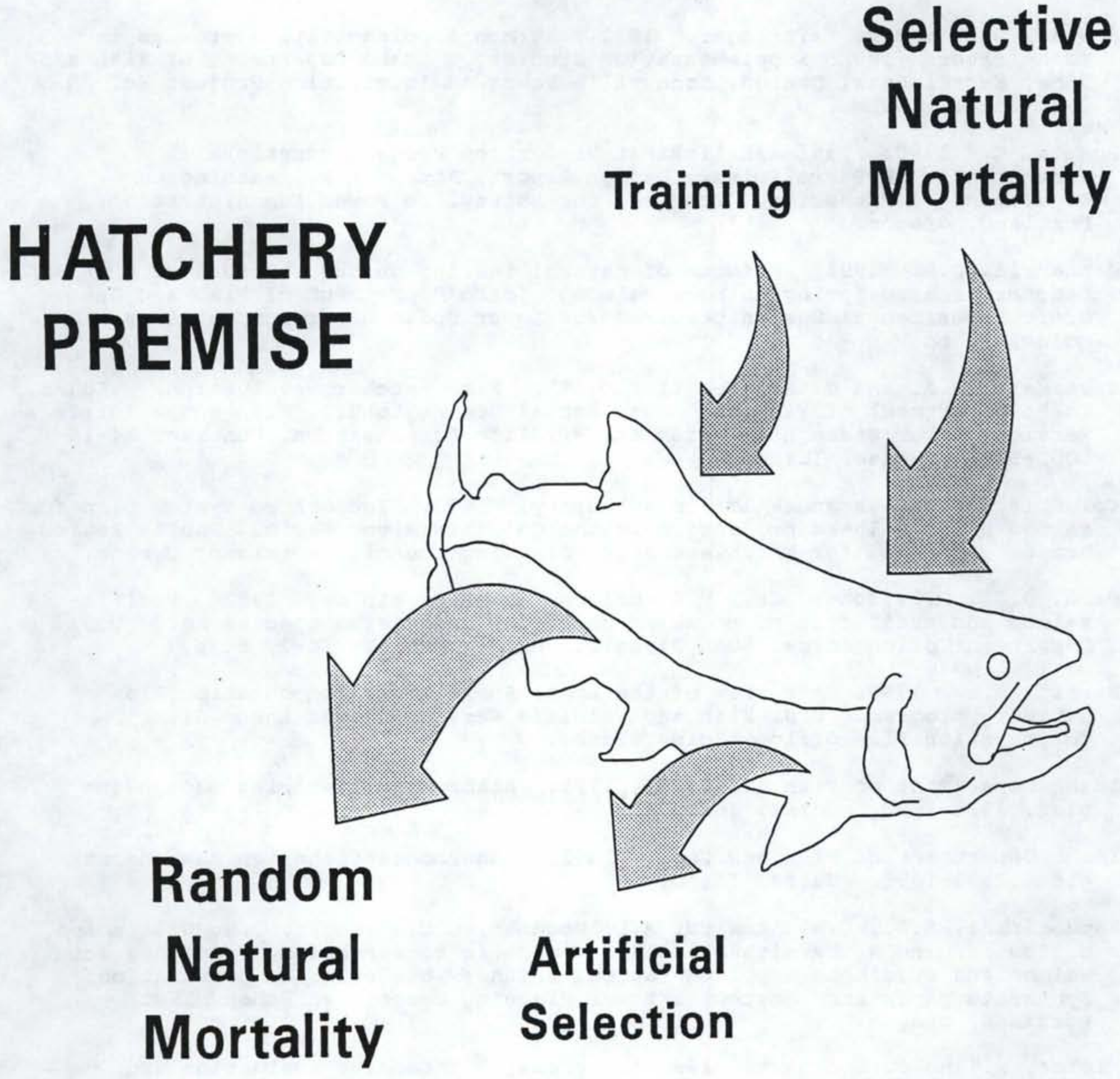

Figure 4. Idealized model depicting hatchery role in creating selective natural mortality and training opportunities while removing random natural mortality and artificial selection.

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PLANNING GUIDELINES
FOR SUPPLEMENTATION PROJECTS IN THE COLUMBIA BASIN

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ABSTRACT

Supplementation initiatives are a major element of the program to increase salmon and steelhead production in the Columbia Basin. These initiatives present managers with a new challenge: how to integrate natural and artificial production systems in the Columbia Basin in a way that yields sustainable increases in both natural and total production. The challenge is new in that it addresses an assumption that artificial propagation can be used to increase the production of naturally spawning populations without adverse genetic or ecological effects. In the past, the interactions between hatchery programs and wild stocks did not receive the levels of attention and understanding that are now required if supplementation is to succeed.

The Regional Assessment of Supplementation Project (RASP) was initiated, in part, to develop a framework for supplementation planning to help face this new challenge. Accordingly, RASP has formulated a set of planning guidelines to be used in developing supplementation plans. The guidelines are comprised of nine steps. In the first step goals are established, steps two to four are fact finding and descriptive, steps six and seven involve analysis of risks and benefits, and steps eight and nine address project evaluation.

A primary purpose of the guidelines is to stimulate thinking about the structure and function of the ecosystem to be manipulated through artificial propagation. Information about the natural production system or the lack thereof, is to be identified and considered in the planning process. While adaptive management allows projects to be implemented with information gaps and uncertainty, it also means that planning and evaluation are not a one time activity. Planning becomes an iterative process. New information is used to update a supplementation plan until uncertainties are resolved.

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USING "PATIENT/TEMPLATE ANALYSIS" IN THE DESIGN OF PROJECTS TO INCREASE NATURAL PRODUCTION OF ANADROMOUS SALMONIDS

(Title modified from workshop program)

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ABSTRACT

An important step in planning either a supplementation or habitat enhancement project is to identify the factors that constrain existing natural production as well as possible. The Regional Assessment of Supplementation Project (RASP) has developed an approach that attempts to identify these factors through an analysis of existing information and assumptions about a specific stock/stream system. The approach, called "Patient/Template Analysis" (PTA), contrasts the structure and function of an existing but debilitated stock/stream system (the "patient") with the structure and function of the system as it existed historically (the "template"). It is suggested that enhancement activities might most effectively take the form of attempts to restore or enhance suppressed or diminished elements of the historical system. The process is illustrated with the example of Yakima River spring chinook.

In PTA, populations are first subdivided by known or assumed substock, and them by known or assumed "life histories". "Life histories" are thought of as qualitatively distinct patterns of habitat use by life stage. In the example presented, analysis is restricted to juvenile/freshwater life states (egg to smolt). Alternative freshwater life histories are presented for both the existing population and, insofar as they can be reconstructed, for the historical, pre-development population. The environment underpinnings of life histories assumed in the template are contrasted with existing conditions and life history types, and the factors that have reduced production are tentatively identified for each life history in the template. This preliminary "limiting factors analysis" will provide focus for more definitive analyses, and will suggest potential environmental preconditions for successful supplementation. It is suggested that a combination of supplementation and strategic habitat enhancement focussing on the rehabilitation of historical life history types will frequently be required if a measure of the productivity and resiliency of the template population is to be restored.

INTRODUCTION

Patient/Template analysis (PTA), or some other approach to limiting factors analysis, must guide the design of any project with a goal of increasing *natural* production. This is so because natural production can be choked off by a "survival bottleneck" at any point in the life cycle, and a tool to identify such bottlenecks is essential if enhancement activities are to be focussed.

The PTA exercise forces the planner to interpret existing knowledge of habitat and life history in terms of a mechanism of natural production. Potential limiting factors are identified by estimating the ability of the environment to support each life stage in a network of life stages that collectively describes the population.

The analytic process focusses on ecological contrasts. Specifically, a healthy historical system -- the Template -- is contrasted with a dysfunctional contemporary system -- the Patient. Environmental differences and their biological consequences are then used to diagnose the causes of depressed production in the Patient.

The ultimate goal of PTA is to use a detailed, local diagnosis to design an optimal remedial program. In applying PTA to Yakima spring chinook, it became clear that an effective supplementation program might also require habitat enhancement within the subbasin. The perceived situation in the Yakima is probably not unique. Indeed, it is probable that any effective supplementation program will include some type of complementary habitat enhancement whenever *density-dependent* mechanisms significantly constrain natural production.

The following discussion will first address the general features and implications of PTA. With this background established, the process will be illustrated by showing how it was used to diagnose depressed production in one stock of spring chinook in the Yakima River.

The reader should understand that the Yakima spring chinook example represents only an initial attempt at PTA. Yakima planners are fully aware that many of their quantitative estimates are oversimplified and compromised by a lack of detailed field data. *Nevertheless, the Yakima PTA exercise was useful because it* provided an ecological perspective on natural production. This perspective lead to the formulation of testable hypotheses and the identification of a number of enhancement projects that could be implemented immediately with little risk and a high probability of producing measurable benefits. Importantly, it also highlighted a previously unrecognized problem that, if verified and not corrected, might compromise a planned supplementation program.

PTA METHOD

Definition of Terms

PTA entails the use of three specially defined terms:

1. ***Life history***: a series of seasonally and geographically connected *places* which support all life stages in the natural life cycle.
2. ***Population***: the aggregate of discrete life histories exhibited by a reproductively isolated group of fish (a "stock").
3. ***Life stage***: a discrete developmental phase, such as egg to emergent fry, emergent fry to late-summer parr, parr to late-winter pre-smolt, smolt, and so on.

Life Histories

Because it is central to the analytical process and is defined somewhat unconventionally, life history is probably the most important concept in PTA. A life history type is a group of fish within a population whose life cycle describes a unique "trajectory" in time and space. Put another way, different life history types use different combinations of tributaries, rivers and perhaps oceanic regions over their life cycle; *or* they use the same places, but at different times. We defined the alternative "places" which fish might use *as spawners and juveniles* as groups of tributaries or river reaches that are environmentally homogenous and distinct from other tributaries and reaches. Thus, for the "intra-subbasin" portion of the life cycle, life history types represent groups of fish that follow different spatio-temporal trajectories in completing spawning and incubation, fry-to-parr rearing, pre-smolt overwintering and smolting.

"Drainage units" was the term we used to describe the environmentally distinct places within the subbasin. Drainage units are qualitatively different from each other in ways that have a strong impact on juvenile production. The characteristics we used to define drainage units in the Yakima included thermal cycles, instream flow and hydrographic patterns, channel morphology and gradient, substrate character, riparian condition, predator densities and accessibility.

Four distinct drainage units were identified for the *upper Yakima stock* of spring chinook: Upper Mainstem, Upper Tributary, Lower Mainstem and Lower Tributary (See Fig. 1). [*To retain focus, analysis was restricted to a single stock, the "upper Yakima" stock". The entire Naches River Basin, and the two distinct stocks it supports, were omitted.*] The Upper Mainstem was defined as the Yakima River between Easton Dam and the Naches confluence. The Upper Tributaries were defined as all tributaries to the Upper Mainstem exclusive of the Naches drainage. The Yakima River itself above Easton Dam was classed with the Upper Tributaries because it has much more of the character of a tributary than a mainstem river. The Lower Mainstem was defined as the Yakima River below the Naches confluence, and the Lower Tributaries include all streams entering the Lower Mainstem exclusive of the Naches. The interpretation of subsequent material will be facilitated if the reader will scan the habitat summary of each drainage unit in Table 1.

Populations as Aggregates of Life Histories

Production entails access to food and space, and so depends on environmental opportunity. Salmonid life histories have evolved to maximize survival and take advantage of seasonal and spatial variations in resource availability. In a quasi-stable environment, the life stages of a species will reflect equilibrium conditions, and will be distributed throughout the system in patterns that maximize the fitness of the population.

However, streams and rivers inhabited by salmonids are distinguished less by environmental constancy than by short-term and often severe disturbances that can disrupt community structure, alter resource availability and modify the physical environment. Unpredictable natural disturbances and human activities may removeor diminish opportunities and reduce production potential. The existence of multiple life history types represents an adaptation by salmonid populations to spatial and temporal unpredictability in resource availability. Each life history type is a succession of life stages that collectively exhibits a unique pattern of movement and distribution within the environment. If critical habitats are destroyed or altered within relatively short periods of time (viz., periods that do not permit evolutionary response), the affected life histories are not likely to persist.

Steps of PTA

There are five steps or stages to PTA:

1. Describe life histories in Patient and Template.
2. Describe environmental requirements for each life stage in each life history.

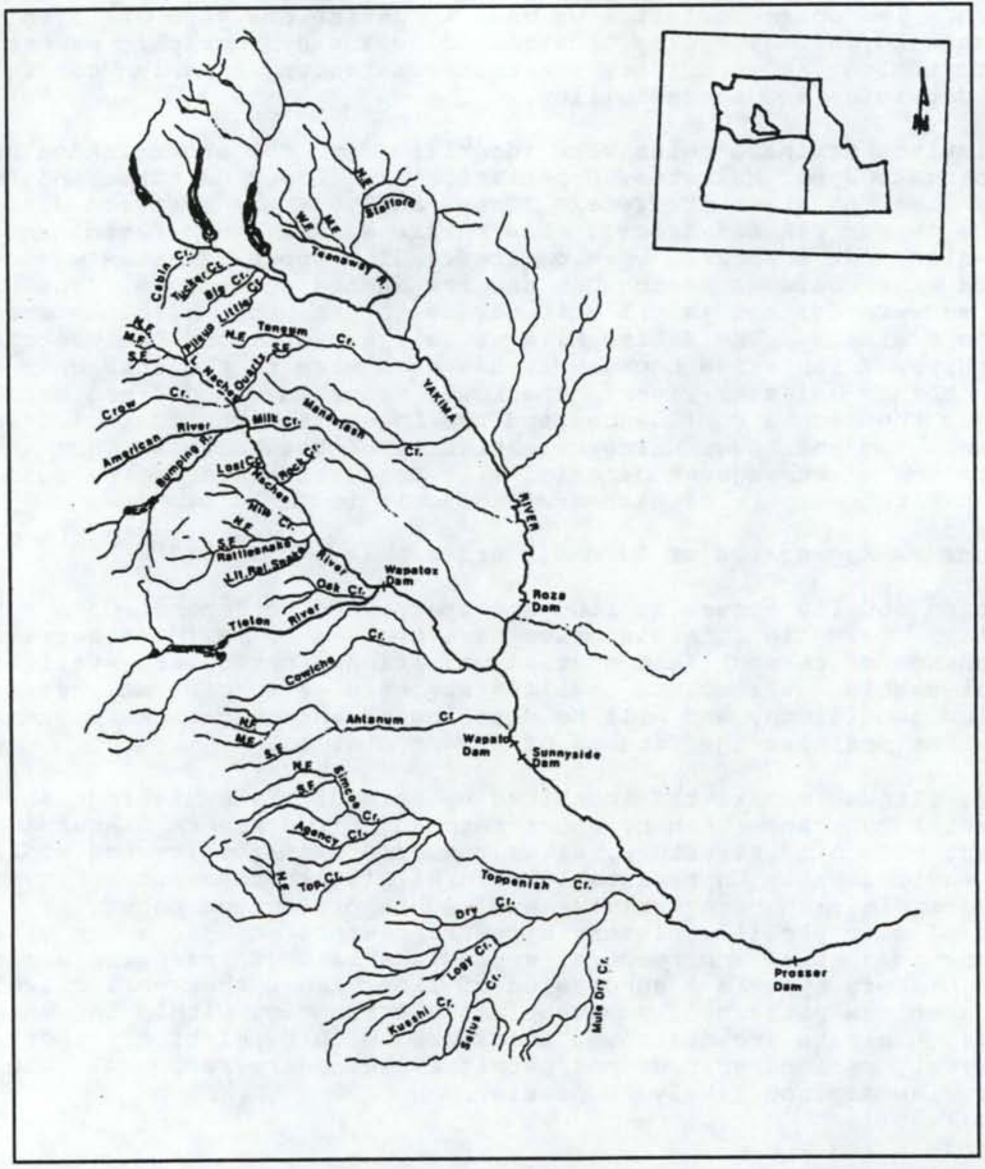

Figure 1. The Yakima Subbasin in south-central Washington.

Table 1. Summary of habitat characteristics of the four principle drainage units for the Upper Yakima stock of spring chinook.

DRAINAGE UNIT	SIZE, GRADIENT	TEMP.	HYDROGRAPH, INSTREAM FLOW	PASSAGE	SPAWNING and INCUBATION	REARING
UPPER TRIBS	Small to moderate sized streams (30-100 ft wide) of moderate gradient (0.4 - 2.0%) with larger streams between 0.4 - 0.7%.	Relatively cold except in lower reaches impacted by diversions, riparian damage.	"Flashy" hydrograph (headwaters in heavily-logged eastern Cascades): large spring discharge, low summer/fall discharge. Two streams heavily augmented with reservoir releases in summer and depleted by winter reservoir re-filling.	Fair-poor: Irrigation diversion dams on all streams restrict adult access either as a physical barrier or by dewatering. Smolts entrained in unscreened diversions.	Fair-excellent: abundant riffles with low to moderate embeddedness. Periodic dewatering problems during incubation as reservoirs re-filled.	Fair-excellent: Riparian conditions good-excellent above lower reaches, LOD abundant. Habitat is structurally complex. Predatory fish rare.
UPPER MAINSTEM	Medium-sized river (100-200 ft wide) with relatively low gradient (0.17 - 0.4%, mean of 0.25%).	Cool: summer temperature rarely exceeds low 60's.	Hydrograph radically altered by irrigation: summer reservoir releases increase discharge to unnatural levels; fall/winter reservoir refilling decreases flow to unnatural levels.	Excellent except for minor delays at Roza Dam ladder. Most diversions screened.	Many large gravel bars of moderate embeddeness above confluence of Cle Elum River. Similar bars exist below as well, but early fall flows may be excessive (irrigation releases from Cle Elum Reservoir). Redd dewatering not usually a problem.	Above Cle Elum, reasonably intact riparian, braids and side channels create good-excellent summer rearing. Below Cle Elum, river confined to canyon, LOD scarce, velocities excessive for fry/parr. Flow fluctuation cause side channel stranding in summer. Large decreases in winter flows eliminate most winter habitat. Squawfish congregate at diversion dams.
LOWER TRIBS	Smaller streams (30-60ft wide) with low gradient in most reaches (0.1-0.2%).	Warm: widespread riparian damage from overgrazing. Summer maxima in some reaches exceed 75°F.	"Flashy" hydrograph, with large flows in late winter and early spring, low to very low flows in summer and fall.	Poor: Except for one stream, passage impaired or blocked entirely by diversions (unladdered dams and/or dewatering). Most diversions unscreened.	Spawning gravel abundant only in middle-upper reaches; heavy siltation from grazing, irrigation returns in lower reaches. Incubation flows adequate in upper reaches.	Low summer flow, sedimentation and overgrazing has drastically reduced pool frequency. Summer temperature prohibitive in lower reaches. In lower reaches, squawfish are abundant and bass and other spiny rays are present.

DRAINAGE UNIT	SIZE, GRADIENT	TEMP.	HYDROGRAPH, INSTREAM FLOW	PASSAGE	SPAWNING and INCUBATION	REARING
LOWER MAINSTEM	Moderately large (300-500 ft wide) meandering river of very low gradient (~0.1%), with side channels and oxbows in some reaches.	Warm: riparian degradation, irrigation returns, low summer flows. Summer maxima reach 80°F.	Hydrograph similar to Upper Mainstem, although irrigation withdrawals limit summer peaks and contributions from Naches augment winter lows.	Excellent except for the odd year when high water temperature impacts the tail of the run (applies to all other drainage units also). All diversions screened.	Not applicable: spring chinook do not spawn in the lower mainstem, and probably never have.	The upper half of the unit meanders, has a forested riparian corridor, many oxbows, sloughs and side channels, and is heavily used by overwintering pre-smolts. The lower half is much less sinuous, lacks channel diversity and a forested riparian. Substrate is moderately to heavily embedded by sediment from irrigation returns. Piscivorous fish (squawfish, bass and channel catfish) are very numerous, especially in the lower reaches and in the vicinity of diversion dams.

3. Estimate existing, potential and historic carrying capacity, as well as existing production, for each density-dependent life stage in each life history in Patient and Template. Estimate historical and existing survival rates for density-independent life stages.

4. Identify the life stage at which each life history is limited and the mechanism of limitation.

5. Identify actions to remove or lessen severity of the limiting factors identified.

The first two steps entail the description of the existing and historical populations and the environmental conditions that shaped them. They end with a complete, qualitative picture of Template and Patient -- with a description of the complete historical and contemporary assemblages of life histories and associated environmental conditions.

The third and fourth steps attempt to quantify the description of Patient and Template. Step 3 requires the planner to classify all life stages in terms of density-dependent or density-independent regulation, and to estimate three kinds of carrying capacity for the density-dependent stages: "historical", "potential" and "existing". "Historical" capacity denotes an estimate of production capacity in historical times, when the environment was pristine. "Potential" capacity denotes an estimate of the production capacity that could occur now if all practicable habitat improvement projects were successfully implemented. "Existing" capacity simply refers to the capacity of the unimproved, existing habitat. Step 3 also requires an estimate of existing production by life stage; the mean number of fry, parr, pre-smolts, etc., produced in the drainage unit under study. The latter estimate is critical, because the ratio of existing production to existing capacity (the "utilization index") provides a crude index of limitation *by life stage*. **At its present stage of development, PTA assumes that life stages with high utilization indices represent periods during which overall production is constrained.** For such putative limiting life stages, the ratio of existing to potential carrying capacity provides another piece of useful information: the degree to which the problem might be resolved by habitat enhancement. Finally, the ratio of potential to historical carrying capacity provides an index of the productive capacity that is irretrievably lost. If the life stage under study is in fact a limiting phase of a major life history, this latter ratio can be used to gauge the maximum possible benefits of a supplementation project (expressed in terms of historical production).

The forgoing discussion emphasizes the importance of life stages regulated by density-dependent mechanisms. There are two reasons for this. First, nearly all life stages occurring within the subbasin are ultimately subject to density-dependent regulation. Pre-spawning survival, egg-to-fry survival, fry-to-parr survival and the overwinter survival of pre-smolts all decline as the quantity of usable habitat per fish declines. Second, in systems impacted by resource scarcity, supplementation alone might increase density-dependent mortality and limit or prevent a net increase in production. Supplementation under such conditions might have the unintended result of merely replacing wild fish with supplementation fish.

Step 3 also requires the supplementation planner to estimate survival rates for density-independent life stages. At least on the level of an individual population, the most commonly identified density-independent "life stage" is the "smolt-to-adult" portion of the life cycle. Ignoring for the moment the gross oversimplification inherent in such an expansive "life stage", the planner should compare smolt-to-adult survival with all utilization indices within the subbasin. The planner should, in particular, determine whether the targeted system combines low utilization indices with a low smolt-to-adult survival rate because, in such a case, supplementation alone could result in a significant increase in production.

Step 4, the identification of limiting life stages (in the Patient) and the description of the mechanism of limitation, has for the most part already been described. High utilization indices flag life stages limited by density-dependent mechanisms; a combination of low utilization indices and a low smolt-to-adult survival rate flags a system limited primarily by spawning escapement. The process of describing the particular *mechanism* of limitation, however, cannot be reduced to a simple formula. Detailed knowledge of the stream/stock system is required for this exercise. It is hoped that the Yakima spring chinook example will at least give planners an idea of some of the types of things to consider.

The final step in the process, the design of programs to resolve or circumvent limiting factors, is also idiosyncratic. Given an accurate diagnosis of a particular Patient, cost, risk, social acceptability and other considerations will determine which of a number of alternative "treatments" is optimal. The RASP spreadsheet model is specifically intended to be used at this point to assess the relative benefits and risks associated with alternative treatments.

YAKIMA EXAMPLE

In the material that follows, the qualitative picture of the upper Yakima spring chinook Template and Patient will be described first (steps 1 and 2). This will be followed by a description of the methods used to estimate carrying capacities and utilization indices for each life stage (step 3). Finally, five of the most significant factors limiting natural production of the upper Yakima stock will be described. Importantly, the reader must bear in mind that the analysis was restricted to those portions of the life cycle occurring within the subbasin. [Analyses covering more life stages and/or more stocks are possible. Focus is restricted here to increase clarity.]

Qualitative Description of Template and Patient

Tables 2 and 3 and Figures 2 and 3 represent the life histories comprising, respectively, the Template and Patient populations for upper Yakima spring chinook. Consider first Figures 2 and 3 and the spatiotemporal matrix within which life histories occur. Time, and the life stages generally occurring at a given time, appears as columns; space, in the form of the four environmentally distinct drainage units, appears as rows. Life histories appear as unique paths through this matrix.

Template life histories. The best justification for the six putative historical life histories identified in Fig. 2 is the fact that either they still exist, even if at greatly diminished levels of productivity; or solid documentation can be found for their existence historically. Moreover, the environmental characteristics necessary to support these life histories are not inconsistent with conditions that can reasonably be proposed for historical times.

As will be seen shortly, life histories II, III and V are responsible for most of the production observed today. The existence of life histories I and IV -- the life histories occurring wholly within the Upper and Lower Tributaries, respectively -- is based on a number of historical documents (e.g., Bryant and Parkhurst, 1950).

Table 2. Life histories present in upper Yakima spring chinook Template population.

NO.	SPAWNING LOCATION	SUMMER REARING LOCATION (fry to parr)	WINTER REARING LOCATION (pre-smolts)	SMOLT MIGRATION ROUTE (subbasin)	SMOLT AGE
I	Upper Tributaries	Upper Tributaries	Upper Tributaries	Entire drainage	I+
II	Upper Tributaries	Upper Mainstem	Upper Mainstem	~90% of drainage	I+
III	Upper Mainstem	Upper Mainstem	Upper Mainstem	~90% of drainage	I+
IV	Lower Tributaries	Lower Tributaries	Lower Tributaries	<50% of drainage	I+
V	All drainage units above Lower Mainstem	All drainage units above Lower Mainstem	Lower Mainstem & associated "sloughs"	<50% of drainage	I+
VI	All drainage units above Lower Mainstem	Lower Mainstem	Not Applicable	<50% of drainage	0+

Table 3. Life histories present in upper Yakima spring chinook Patient population.

NO.	SPAWNING LOCATION	SUMMER REARING LOCATION (fry to parr)	WINTER REARING LOCATION (pre-smolts)	SMOLT MIGRATION ROUTE (subbasin)	SMOLT AGE
I	Upper Tributaries	Upper Tributaries	Upper Tributaries	Entire drainage	I+
II	Upper Tributaries	Upper Mainstem	Upper Mainstem	~90% of drainage	I+
III	Upper Mainstem	Upper Mainstem	Upper Mainstem	~90% of drainage	I+
V	All drainage units above Lower Mainstem	All drainage units above Lower Mainstem	Lower Mainstem & associated "sloughs"	<50% of drainage	I+

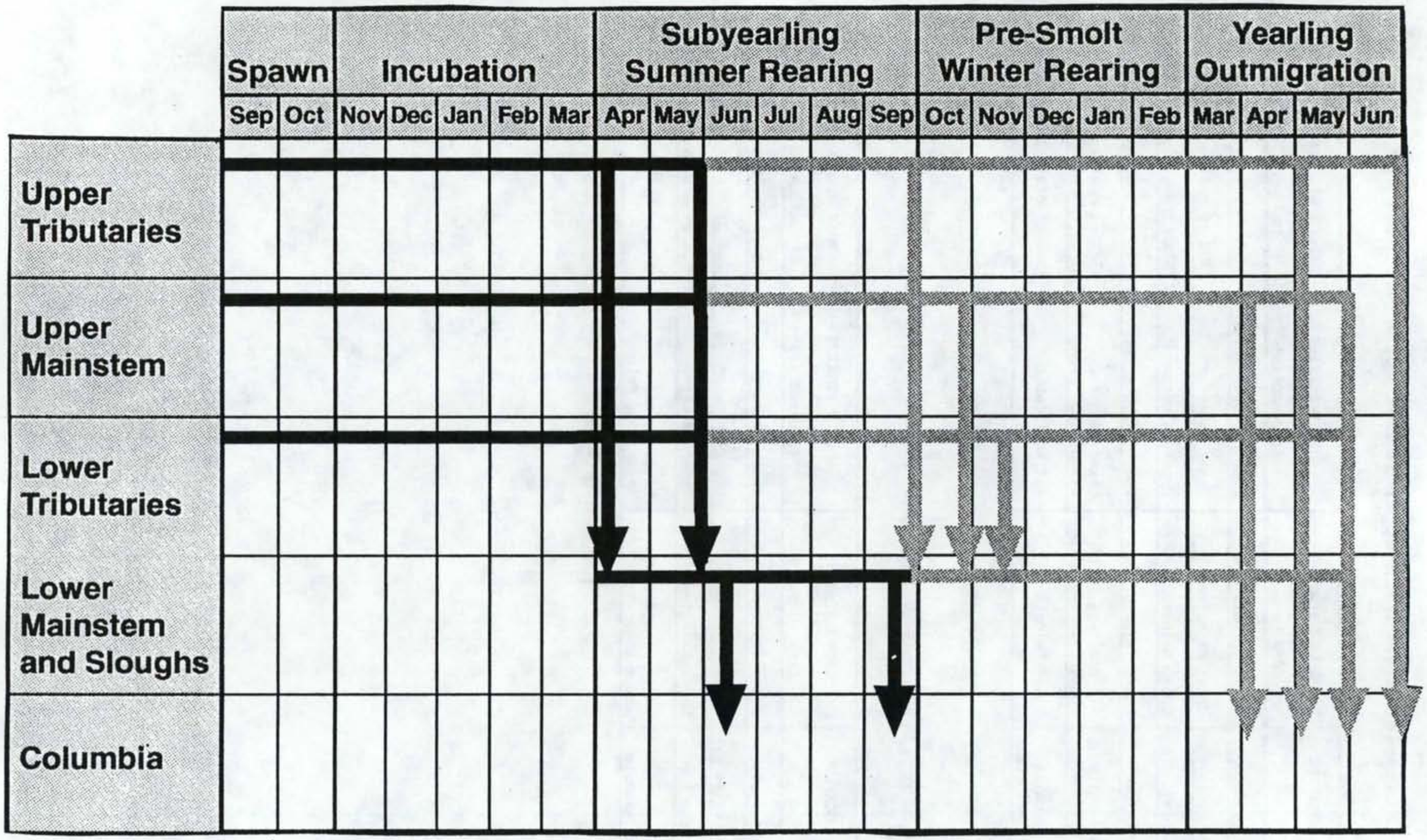

Figure 2. Template life histories, upper Yakima spring chinook.

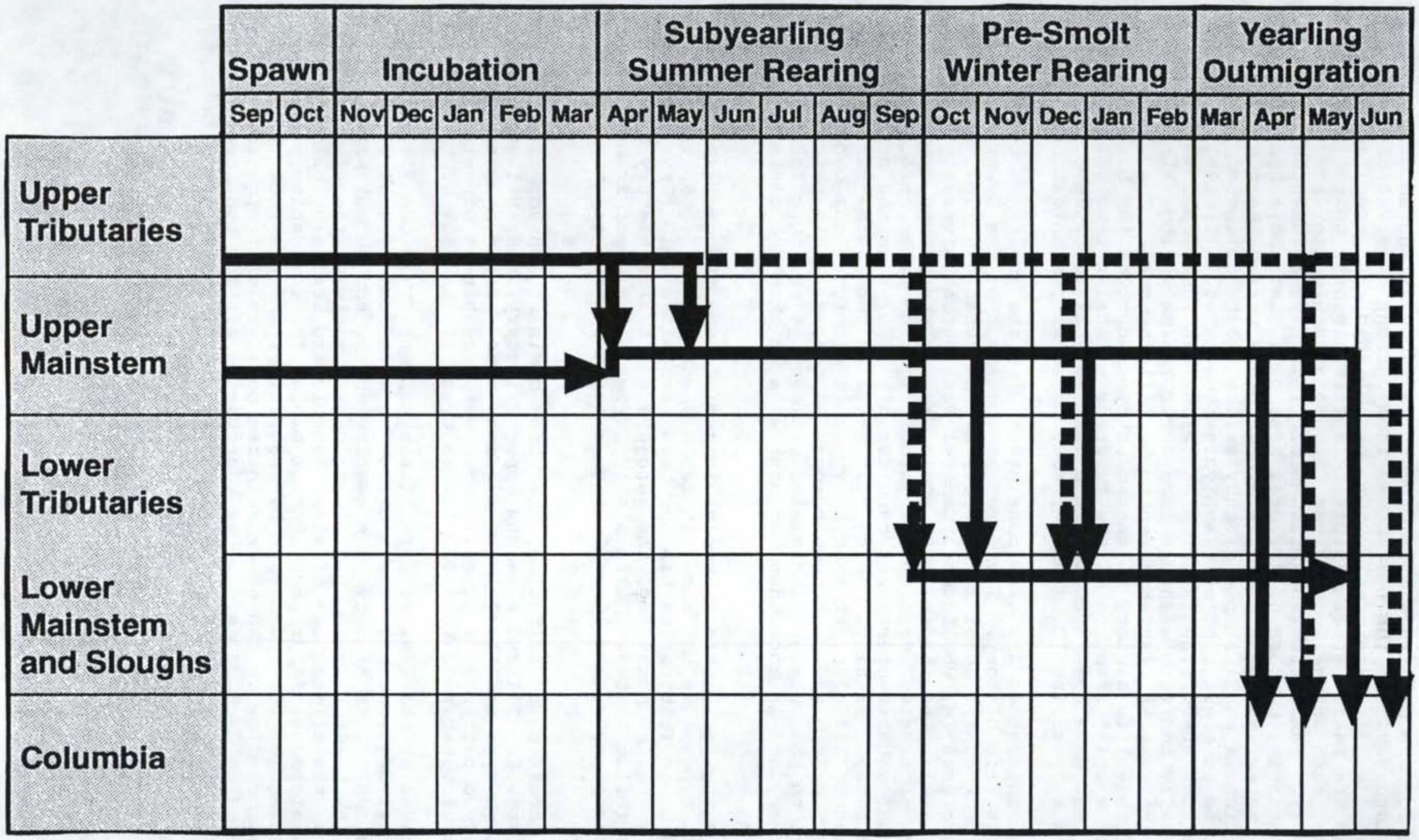

Figure 3. Patient life histories, upper Yakima spring chinook.

Justification for the existence of a productive "subyearling smolt" life history (life history VI) in the past comes from current observations and historical inference. It has recently been established that some few springchinook in the Yakima still exhibit this life history, which is dominant in other systems, such as the Rogue River in Oregon (Schluchter and Lichatowich, 1977). Yakima Indian Nation biologists have documented the movement of substantial numbers of fry over Roza Dam, just above the boundary of the Lower Mainstem, during the April-May emergence period (Fast et. al., 1991). In addition, a small sample of subyearling smolts collected in July of 1990 was subjected to electrophoretic analysis (Busack et. al., 1991). This analysis showed that 40% of the larger fish (>90mm) in the sample were electro-phoretically indistinguishable from upper Yakima spring chinook (almost all of the smaller fish appeared to be fall chinook). The contribution of this life history to contemporary production is, however, negligible; only a small fraction of the hundreds of scales from wild Yakima spring chinook that have been aged over the last several years have indicated ocean entry as a subyearling (C. Knudsen, WDF, personal communication, 1992).

Two historical inferences are also consistent with the existence of a subyearling smolt life history. The first is that much of the Lower Mainstem in historical times consisted of intricately braided channels flowing through dense riparian forests (M. Ubellacker, Central Washington University, personal communication, 1992). The shading that would have occurred then, combined with the absence of massive additions of warm irrigation water, probably resulted in summer water temperatures considerably lower than the prohibitive values observed today. Finally, biologists active in the Yakima during the late 1920s (Haggett, 1928) report that heavy smolt outmigrations began in June, peaked in mid-July and continued through September. This pattern is nearly identical to the outmigration timing of zero-age spring chinook smolts in the Rogue River, and is more than two months later than the contemporary pattern.

Patient life histories. Life histories IV and VI, the "Lower Tributary" and "zero-age smolt" types, are not shown in Table 3 and Figure 3. Spring chinook spawning in the Lower Tributaries has not been seen since 1910 (Bryant and Parkhurst, 1950) and, as mentioned, the zero-age smolt life history type has been reduced to relict status. Life history type I, the Upper Tributary type, is represented by dashed lines in Fig. 3 to indicate a greatly reduced level of productivity.

In order of increasing contribution, the three dominant contemporary life histories are types II, III and V -- the Upper Tributary/Upper Mainstem type, the Upper Mainstem type and the "winter migrant" type. Based on mean redd counts and observed patterns of juvenile movement, roughly 5% of existing production is attributable to type II, 15% to type III, and 80% to type V.

The primary cause of the loss of life history diversity in the Patient has been environmental degradation. The proximal cause of the virtual elimination of the zero-age smolt life history is excessive spring and summer water temperature in the Lower Mainstem. Ultimate causes include: the clearing of riparian forests; the elimination of side channels and braids by diking, agricultural development, "appropriation" of braids for irrigation purposes, and the "flattening" of the hydrograph by regulation; and the fact that virtually all summer flow in the Lower Mainstem consists of irrigation return water. The preceding factors as well as overgrazing afflict the Lower Tributaries (life history type IV), raising fall water temperatures, reducing instream flow and restricting adult passage. Production of the "Upper Tributary" fish (type I) has been greatly diminished by restricted adult access associated with irrigation diversions and, in some cases, by instream flows that are excessive in summer and inadequate in winter.

Quantitative Description of Template and Patient

Techniques of estimation. The life stages making up each life history were first classed as being subject to density-dependent or density-independent regulation. The life stages addressed were: newly-emergent fry (density-independent except at very high escapement); late-summer parr (density-dependent); late-winter pre-smolts (density-dependent); and outmigrant smolt within the subbasin (density-dependent). Carrying capacity for each life stage in each life history was then estimated for pristine historical conditions, "optimally enhanced" contemporary conditions, and unimproved contemporary conditions. An index of utilization was then estimated for each life stage under current conditions, as was the mean survival rate for each density-independent life stage.

The general approach to estimating carrying capacity for parr and pre-smolts was similar to the approach subbasin planners used in estimating smolt carrying capacity: over all reaches, we summed the product of accessible habitat area and "maximal density" of parr or pre-smolts supportable by the reach. Conditioned by acceptable values for temperature, velocity and depth, maximal density figures were estimated as a function of habitat structure. The habitat/density relationships used in the analysis were taken from a number of published studies conducted in other subbasins.

Given adequate knowledge of the distribution of habitat types, it is a straightforward task to use this procedure to estimate carrying capacity for parr and pre-smolts *for the Patient*. It is also straightforward to estimate "potential carrying capacity" so long as one is comfortable in predicting environmental responses to enhancement programs. It is considerably harder to apply it to the Template, because habitat characteristics are usually known on a much coarser scale. Planners will, however, not find the task of "reconstructing" historical habitat conditions impossible if they make use of all available information sources. In the hope that they will prove useful to others, the following six sources were (or will be) used by Yakima planners:

1. Old Government Land Office maps (excellent for riparian conditions and channel configuration).
2. A time series of aerial photographs (very useful for describing trends in channel morphology, riparian conditions, pool/riffle ratios, etc.)
3. Old instream flow and diversion records from the Bureau of Reclamation, local irrigation districts, the USGS or the Department of Ecology (adult and juvenile passage; spawning and incubation conditions).
4. Analyses provided by local geographers, geologists and hydrologists (virtually all habitat variables).
5. Archived records and reports from state and federal fish and wildlife agencies -- e.g., Bryant and Parkhurst (1950), Washington State's "Stream Improvement Files" (evidence for the existence of vanished life histories).
6. Affidavits of tribal elders submitted in court cases involving fishing rights (species/race distribution and abundance).

Fry capacity was determined by summing the product of three variables over all reaches in a drainage unit: the number of redds that can be accommodated by accessible gravel bars at "full seeding"; the mean fecundity of upper Yakima spring chinook; and reach-specific egg-to-fry survival rates. At "full seeding", the number of redds is equal to accessible gravel bar area divided by the average area of a spring chinook redd. Egg-to-fry survival rates have been empirically estimated in a number of reaches in the Upper Yakima (Fast et. al., 1991), and survival rates for unsurveyed drainage units and reaches were based on the Upper Yakima figures adjusted upwards or downwards depending on whether spawning and incubation conditions were relatively better or worse.

Existing production by life stage was not estimated by summing the product of habitat area and fish density because of a lack of density data in mainstem drainage units. Accordingly, smolt outmigration estimates from the Lower Mainstem were first allocated to various portions of the subbasin on the basis of relative brood year egg deposition, and estimated survival rates from parr to smolt and fry to parr were used to back-calculate drainage unit production at previous life stages.

The survival of outmigrant smolts from the point at which migration begins to the Columbia confluence -- a parameter termed "smolt-to-smolt survival" -- has been estimated a number of times and by various methods over the past decade (see Fast et. al., 1991 for a partial summary). Numerous groups of hatchery and wild spring chinook smolts have been marked with freeze brands and PIT-tags and released in the Upper Mainstem and monitored at a Lower Mainstem smolt trap (RM 47) and McNary Dam. Taken as a whole, these studies indicate that the smolt-to-smolt survival of wild, Upper Tributary and Upper Mainstem smolts is somewhat less than 50%; smolt-to-smolt survival for hatchery spring chinook released in the Upper Tributaries has averaged about 25-30%. Smolt-to-smolt survival for wild winter migrants has not yet been estimated because of the difficulty of discriminating them from other life history types at monitoring facilities during the spring outmigration. It is, however, believed that their survival rate is greater than Upper Mainstem and Upper Tributary types because their migratory path is shorter and they pass major diversion dams during the winter, when water is not diverted.

RESULTS

The results of the quantitative portion of the analysis are depicted in Fig. 4. To save time and sharpen focus, the figure presents data only for the three qualitatively different contemporary life histories: Upper Tributary (type I), Upper Mainstem (type III) and "winter migrant" (type V).

Each series of three bar charts in Fig. 4 depicts relative carrying capacity by life stage for the three major life histories. Potential and existing carrying capacity, as well as existing production, are scaled relative to historical carrying capacity, which is always assigned a value of 100%. From left to right, each bar in a life-stage cluster represents historic capacity, potential capacity, existing capacity and existing production, respectively. The utilization index (the ratio of existing production to existing capacity), called "percent use" in the figure, appears above each life stage. Finally, an estimate of the survival of outmigrating smolts while still in the subbasin ("smolt-to-smolt" survival) appears to the right of the pre-smolt cluster for each life history.

The discrepancy between existing production and existing capacity is an index of density-dependent limitation. Thus, Fig. 4 makes it clear that none of the three dominant life history types is spawning limited, because existing production never exceeds 3.5% of existing capacity. The utilization index for parr production never exceeds 14.4%, suggesting that natural production is not limited by summer rearing, either. Possible problems are, however, indicated for pre-smolt production. The utilization index for the two life histories that account for as much as 95% of existing production, types III and V, are 52 and 71%, respectively.

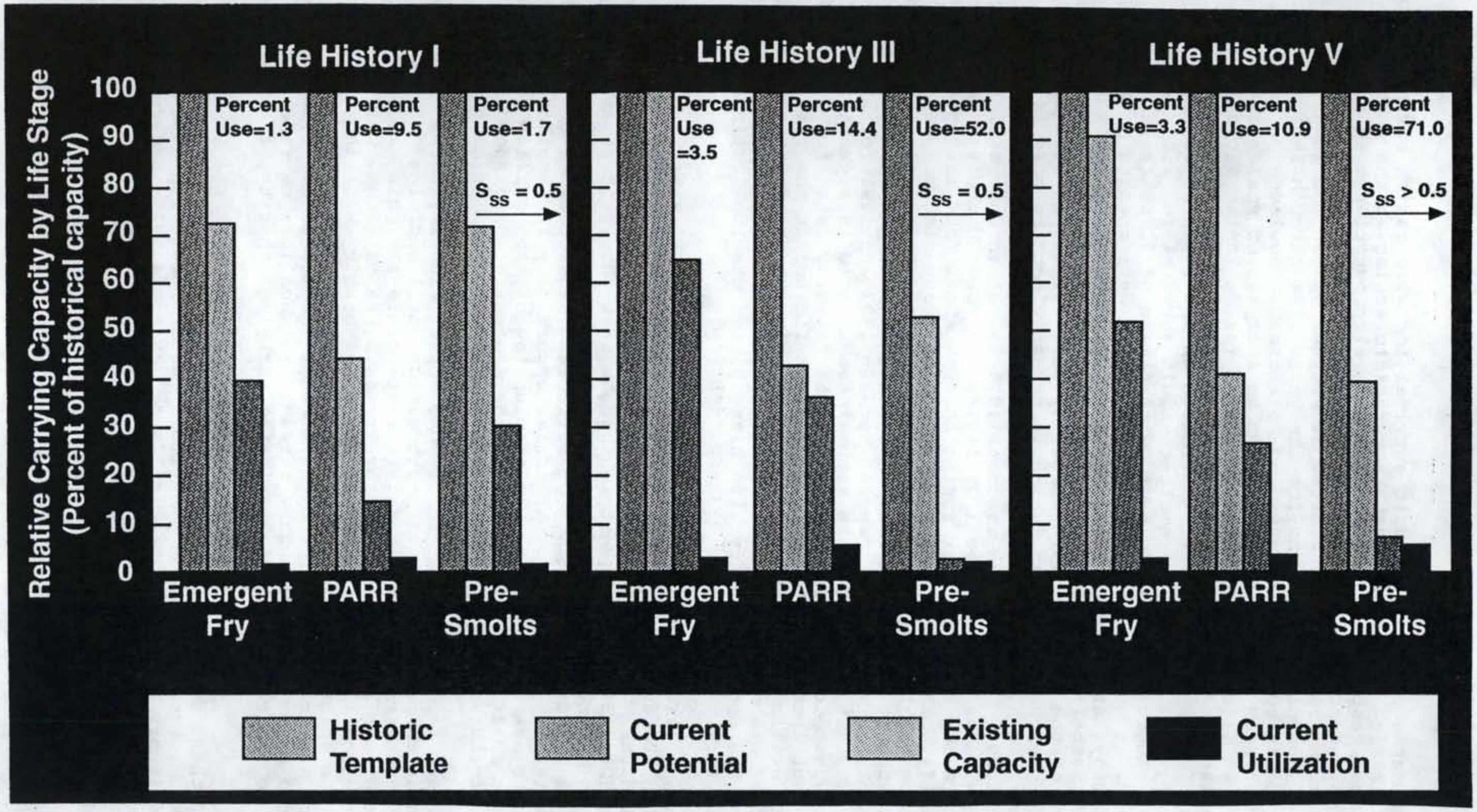

Figure 4. Relative historical, potential and existing carrying capacity, by life stage, for the three dominant life spring chinook salmon histories currently extand in the Yakima River.

Existing estimates of smolt-to-adult survival for wild Yakima spring chinook (not depicted in Fig. 4) also suggest that factors acting at the subbasin level may be more responsible for limiting natural production than had previously been suspected. Between the years 1983 and 1987, wild spring chinook smolt outmigration was estimated at Prosser Dam on the lower Yakima River (RM 47). Spawning escapement for the years 1984 through 1991 was also determined by adult counts at Prosser and Roza Dams. Knowledge of the age distribution for each return (obtained from an analysis of scales collected from carcasses in major spawning areas) enabled Yakima Indian Nation biologists to estimate the number of returning adults produced by the outmigrations of 1983 through 1987. Estimated as the ratio of returning adults to outmigrants at Prosser, smolt-to-adult survival for all Yakima spring chinook ranged from 1.77 to 6.0%, with a mean of 3.75% (Fast et. al., 1991). However, it must be pointed out that smolts were counted at Prosser, downstream of many hazardous reaches and after an undetermined amount of smolt-to-smolt mortality. If it is assumed that all smolts counted at Prosser were survivors of the ~50% in-basin mortality rate associated with Upper Tributary and Upper Mainstem fish (which is probably not the case if the winter migrant life history is dominant), mean smolt-to-adult survival would be halved, to 1.87%. Even if one assumes 1.87% is *not* an underestimate, it is still nearly twice as large as the 1.0% figure the Power Council's Monitoring and Evaluation Group estimated for spring chinook above four mainstem dams (Anonymous, 1987). The Monitoring and Evaluation Group also proposed a generic relationship between egg-to-smolt survival and "smolt seeding", which we have called the utilization index. According to this relationship, the mean egg-to-smolt survival rate observed between 1983 and 1991 for all Yakima spring chinook, 4.72%, indicates the subbasin as a whole is approximately 75% seeded with respect to smolt production. A 75% "seeding rate" for smolt production is not inconsistent with a population limited at the pre-smolt life stage by a 71% utilization index. Two final aspects of the data presented in Fig. 4 should be discussed: the absence of any indication of density-dependent limitation for the Upper Tributary life history, and the implications of a 50% smolt-to-smolt survival rate.

Upper Tributary life history type I would seem to be an ideal candidate for supplementation, because none of the density-dependent life stages have high utilization indices. Smolt-to-smolt survival rates are low (and may be even less than the 50% indicated), but it is thought that this feature is largely attributable to depensatory predation in the Lower Mainstem, the impact of which should be reduced by outplants of sufficient magnitude. Therefore, the Upper Tributaries may represent a particular opportunity for supplementation; once access problems are resolved, the outplanting of *high quality juveniles should increase smolt production provided the progeny of outplanted fish remain in the tributaries and do not migrate into the Upper Mainstem.*

Currently, very few fish remain in the one Upper Tributary stream (the Cle Elum River) that supports significant production until smolting. It is speculated that environmental factors are responsible for this exodus. Large summertime reservoir releases flush many fry into the mainstem, while drastically reduced flows during the period of reservoir refilling in the winter force the emigration of most of the remainder. Note that these conditions are ascribed primarily to one stream, the Cle Elum. Flows in the majority of the other Upper Tributary streams are much less extreme, contributing to the relatively high existing carrying capacity of the drainage unit.

But what if these downstream "rearing migrations" are not an artifact of a regulated system, but are instead *genetically* determined? The question of whether life history types are genetically or environmentally determined has not been addressed so far because it is a complex issue that has not yet been investigated in the Yakima. It is mentioned now only because, by itself, it need not be decisive in deciding whether a supplementation project is viable.

exists and is included among the outplanted fish, production of the targeted life history will increase if the program is continued long enough. In the present case, assume that life history types I and II are genetically determined, and that selective pressures have favored life history II. If both genotypes are included in the outplants in proportion to current abundance, the first outplants might have little impact; most of their progeny would move into the Upper Mainstem, where many would be lost in the winter because of overutilized/inaccessible overwintering habitat. However, some of the outplants would carry the type I trait, would remain in the tributaries, and their numbers would increase with each cycle of outplanting. Therefore, the point to be made on this issue is that genetic determination of life history types does not preclude a supplementation program so long as habitat is underutilized, some number of the targeted genotypes still exist, and some of these fish are included among the outplants.

Elements of the Yakima Diagnosis

Five factors have been identified as contributing to the depression of spring chinook production in the Yakima. These factors, in order of increasing concern to Yakima planners, are as follows:

1. Virtual elimination of the zero-age smolt life history because of excessive late-spring and summer temperatures in the Lower Mainstem attributable to riparian degradation, heavy diversions and low instream flows, and large inputs of irrigation return water.

2. Elimination of the Lower Tributary life history because of thermal blocks to spawners and/or restricted adult access caused by low flows and dams.

3. Underutilization of Upper Tributaries caused by restricted adult access attributable to low flows and impassible dams in the lower reaches many streams.

4. Low smolt-to-smolt survival for non-winter migrants because of depensatory predation in the middle and lower mainstem.

5. Limited overwintering capacity in the Upper and Lower Mainstem.

At the present time, this diagnosis has not been conclusively verified nor have the existing candidates for limiting factors been ranked. Until verification and ranking has occurred, the existing supplementation strategy will not be substantially modified.

The PTA exercise has, however, led to the proposal of a number of habitat improvement projects which, while desirable in themselves, may be more important for the light they might shed on the reality of putative limiting factors. In this regard, particular attention is being directed to item 5, the limitation of production at the pre-smolt stage because of the flow-mediated inaccessibility of appropriate habitat in the Upper and Lower Mainstem. Yakima planners hope to test the hypothesis of winter limitation experimentally, perhaps by monitoring the survival of PIT-tagged winter migrants released inside (test) and outside (control) "sloughs" providing good overwintering habitat.

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BEYOND PLANNING
STATUS OF SUPPLEMENTING CHINOOK SALMON IN THE IMNAHA RIVER BASIN

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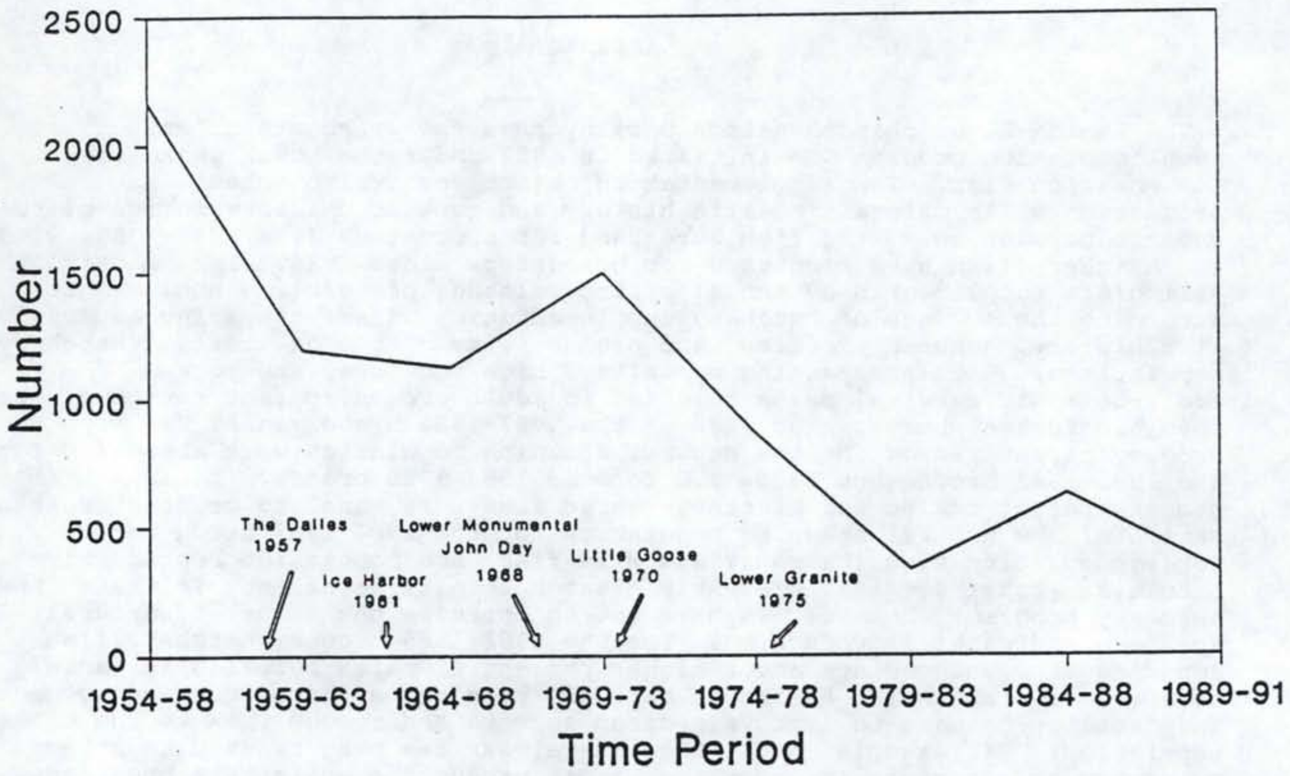
ABSTRACT

The Imnaha River chinook salmon *Oncorhynchus tshawytscha* hatchery supplementation program was initiated in 1982 under the Lower Snake River Compensation Plan. The supplementation objectives are to enhance natural production while maintaining life history and genetic characteristics of the endemic population. Wild fish were used for broodstock from 1982-1985, wild and hatchery fish have been used for broodstock since 1986. Imnaha chinook salmon are supplemented by annual spring releases of yearling smolts. To determine the success of hatchery supplementation we are comparing aspects of life history, genetic profiles, and productivity of the natural and hatchery populations. High prespawning mortality, high egg loss, and poor smolt-to-adult survival rates resulted in adult progeny:parent ratios of less than 1.0 for hatchery-reared fish of the 1982-1986 brood years. Progeny:parent ratios for the natural spawning population were above 1.0 for the 1982-1983 broods but below 1.0 for the 1984-1986 broods. Adult progeny:parent ratios for hatchery-reared fish were equal to or greater than ratios of the natural spawning population for the 1984-1987 brood years. Total population size (hatchery and wild fish) and population reproductive potential (total eggs) is presently greater than if we had not initiated the hatchery program. However, we have yet to increase the number of natural spawners and natural production. For the 1982-1985 broods, hatchery fish returned at a younger age and a higher percent of males returned as jacks. We have made a substantial number of hatchery production, broodstock management, and facility changes to improve program success and reduce risk to the endemic population. For example, smolt size-at-release has been reduced to attempt to shift age at return to an older age, smolt production goals have been reduced, a larger proportion of the hatchery and wild fish have been passed above the weir to spawn naturally, matrix mating protocols have been implemented, and new adult capture/holding and juvenile acclimation facilities have been constructed. Prespawning mortality and egg loss have been reduced and we expect improvements in smolt-to-adult survival.

INTRODUCTION

Summer chinook salmon *Oncorhynchus tshawytscha* populations in the Imnaha River basin have declined precipitously during the past three decades (Figure 1). Peak escapement in recent history was estimated as 3,459 in 1957 (Carmichael et al. 1990) and wild population levels have declined to levels below 150 individuals in recent years (Table 1). Population declines are principally attributed to reduced population productivity that has resulted from juvenile and adult mortalities that occur during migration at Snake and Columbia river dams and in the reservoirs. Historically, chinook salmon spawned in Lick Creek, Big Sheep Creek, and the mainstem Imnaha River. In recent years very few redds have been observed in Big Sheep and Lick creeks and counts and fish retained for hatchery broodstock.



Peak Escapement 3,439 in 1957

Figure 1. Five-year average of chinook salmon natural spawner escapement in the Imnaha River basin.

Table 1. Total escapement, number of broodstock collected, and number and origin of natural spawners in the Imnaha River for the 1962-1991 run years. All values include adults and jacks, therefore not all broodstock removed were actually spawned. W = Wild, H = Hatchery

Year	Total Escapement[a]	Broodstock Removed		Natural Spawners		% of Natural spawners of Hatchery Origin
		H	W	H	W	
1964	1,216	0	0	0	1,216	0
1965	921	0	0	0	921	0
1966	1,362	0	0	0	1,362	0
1967	1,055	0	0	0	1,055	0
1968	1,202	0	0	0	1,202	0
1969	1,297	0	0	0	1,297	0
1970	1,158	0	0	0	1,158	0
1971	1,750	0	0	0	1,750	0
1972	1,456	0	0	0	1,456	0
1973	2,097	0	0	0	2,097	0
1974	1,060	0	0	0	1,060	0
1975	656	0	0	0	656	0
1976	586	0	0	0	586	0
1977	551	0	0	0	551	0
1978	1,301	0	0	0	1,301	0
1979	192	0	0	0	192	0
1980	125	0	0	0	125	0
1981	307	0	0	0	307	0
1982	419	28	0	0	391	0
1983	397	64	0	0	333	0
1984	518	36	0	0	482	0
1985	692	116	15	0	561	0
1986	799	319	21	0	458	0
1987	479	83	22	0	374	0
1988	607	140	20	12	435	0
1989	415	111	126	0	178	0
1990	566	81	153	112	220	34
1991	683	50	262	163	208	56

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[a] Total escapement is the sum of total natural spawners estimated from redd counts and fish retained for hatchery broodstock.

The current spawning distribution is concentrated in about 18 miles of themainstem. Four dams (Ice Harbor, Lower Monumental, Little Goose, and Lower Granite) were constructed in the lower Snake River from 1961-1975. It was estimated that those four dams resulted in a 48% reduction in annual production of chinook salmon in all populations above Lower Granite Dam (USACOE 1975). Congress authorized the Lower Snake River Compensation Program (LSRCP) in 1976 to mitigate for losses of salmon, steelhead, and other fishery resources that resulted from construction of the four lower Snake River dams. Mitigation goals for the Imnaha spring/summer chinook salmon were established as 3,210 adults annually. Annual hatchery production goals of 490,000 smolts at 20 fish per pound (24,500 lbs) were established to compensate for the loss of 3,210 adults.

The use of artificial propagation to enhance salmon abundance for the purpose of sustaining or enhancing commercial and recreational fisheries has a long history of success. However, the use of artificial propagation to conserve or enhance natural production is highly debated and has not been demonstrated to be successful in many cases (Miller et al. 1990, Hard et al. 1992, RASP 1992). Much of the evidence for failure of artificial propagation programs in supplementing natural production has come from assessment of conventional hatchery programs designed for fishery augmentation and conventional hatchery impacts on natural production. Few hatchery programs have been designed, implemented, and managed for the primary purpose of enhancing natural production.

The Oregon Department of Fish and Wildlife initiated the hatchery program on the Imnaha River in 1982 under LSRCP. The Imnaha chinook salmon hatchery program was developed and has been managed under the guidance of the following four management objectives: Restore natural populations of chinook salmon in the Imnaha River basin to historic abundance levels; reestablish traditional tribal and recreational fisheries for chinook salmon; maintain genetic and life history characteristics of the endemic wild population while pursuing mitigation goals and management objectives; and operate the hatchery program to ensure that the genetic and life history characteristics of the hatchery fish mimic the wild fish.

We have been conducting research, monitoring, and evaluation since 1984. The objectives of our evaluation are: 1) Assess the effectiveness of the hatchery program in increasing production, progeny-to-parent ratios, escapement and reproductive potential; 2) estimate annual adult production (catch and escapement), smolt-to-adult survival, and smolt migration success of hatchery fish; 3) monitor and compare life history characteristics (age-composition, run timing, sex ratio, age-length relationships, juvenile migration characteristics) of natural and hatchery fish; 4) monitor and compare genetic characteristics of hatchery and natural populations; 5) make recommendations to improve the success of achieving mitigation goals and management objectives.

BROODSTOCK DEVELOPMENT AND MANAGEMENT

The biological uniqueness of the Imnaha River chinook salmon was recognized long before that hatchery program was initiated. This recognition led to a decision to use only the endemic stock to initiate the hatchery program and to use some natural fish for hatchery brood each year. Wild adults were collected for hatchery broodstock beginning in 1982. The weir is installed as early as physically possible; however, in all years fish pass above the weir prior to initiation of broodstock collection. Wild fish comprised a majority of the fish retained for broodstock and spawned for the hatchery program from 1982-1988 (Table 2). From 1989-1991 both wild and hatchery fish have been

Table 2. Stock source, origin, and number of females spawned for Imnaha River chinook salmon hatchery program.

Brood year	Stock source	Number of females spawned	Percent wild
1982	wild	10	100
1983	wild	31	100
1984	wild	11	100
1985	wild	32	100
1986	wild/hatchery	59	89.8
1987	wild/hatchery	39	97.4
1988	wild/hatchery	92	89.1
1989	wild/hatchery	83	56.6
1990	wild/hatchery	73	34.2
1991	wild/hatchery	39	38.5

utilized for broodstock. In addition, in all years a significant proportion of the Imnaha population spawns below the weir.

During the early years of this program, guidelines for broodstock collection and for passage of fish above the weir for natural production were not clearly defined. In the years 1982-1986 most of the fish that were collected at the weir were retained for hatchery broodstock. Since 1987 specific guidelines for broodstock collection, retention, mating, and passage have been utilized.

HATCHERY PRODUCTION PROGRAM

A temporary adult and juvenile facility was operated from 1982-1988. A permanent facility was completed in 1989. The Imnaha River facility serves as an adult collection/holding facility, spawning site, advance rearing pond for juveniles, and a release site. The facility resides in the lower section of the river reach that is used most for natural spawning. Currently, a picket weir is installed at the site as early in June-July as physically possible. Adults are trapped from June through early September. Fish are retained for broodstock and passed above the weir to spawn naturally. All fish that are trapped are anesthetized prior to being handled. Fish that are retained for hatchery broodstock are injected with antibiotics.

Spawning begins in mid August when the earliest fish ripen and ends in mid September. Eggs are transported back to Lookingglass Hatchery for incubation. At Lookingglass Hatchery eggs are incubated in pathogen free well water. All swim-up fry are ponded into Canadian style troughs inside the hatchery at standard densities. When fry reach approximately 1-2 grams they are transferred outside to concrete raceways. Fish are reared outside for about one year. Maximum density and loading factor reached just prior to release is 1.0 lbs/ft3 and 5.0 lbs/gpm respectively. Smolts are transferred to the advanced rearing pond on the Imnaha River on or near March 1 and are held for 30 days for acclimation. Fish are fed at the advanced rearing pond and are released by crowding the fish from the pond into the river. All fish released under the present program are yearling smolts.

The first release of hatchery reared smolts under LSRCP in the Imnaha River occurred in 1984. Smolt production levels have varied considerably on an annual basis (Table 3) and have ranged from a low of 24,920 in 1984 to a high of 444,500 in 1990. With the exception of the 1988 and 1989 broods released in 1990 and 1991 all hatchery smolts have been marked.

STUDY AREA

The Imnaha River is located in the northeastern corner of Oregon. The basin drains 2,461 km² of the eastern Wallowa Mountains and the plateau between the Wallowa River drainage and Hells Canyon of the Snake River. The watershed attains an elevation of 3,048 m. at the headwaters in the Eagle Cap Wilderness Area. The watershed undergoes a change from alpine mountains at the headwaters to semi-arid plateau in the in the lower mainstem. Stream discharge patterns are such that maximum flows generally occur in late spring to early summer and minimum discharge occurs during the fall. The Imnaha River weir and juvenile advanced rearing pond are located at km 85.3 on the Imnaha River (Figure 2). The Imnaha River enters the Snake River at km 309.3. Eight dams reside between the Imnaha River and the ocean.

METHODS

Egg-to-smolt survival, adult prespawn survival of fish collected and retained for broodstock, and total release numbers were estimated with standard hatchery inventory techniques. Smolt-to-adult survival rates for hatchery reared smolts were determined for each broodyear based on catch and escapement of coded wire tagged (CWT) fish. Groups of hatchery fish were marked with adipose clips and CWT each year in the fall prior to the spring of release. Ocean and in-river catch of marked fish was obtained from the Pacific States Marine Fisheries Commission database. We recovered marked fish that returned to the Imnaha River at the weir site and on spawning ground surveys.

Adult progeny-to-parent ratios for hatchery and natural populations were determined for the 1982-1987 brood years to assess relative performance of the hatchery program. Progeny-parent ratios for the hatchery population were estimated based on total adults collected for broodstock and the resulting adult returns produced back to the Imnaha River. Progeny-parent ratios for he natural population were determined from estimates of natural spawner escapement and resulting adult production. Natural spawner escapement was estimated as described by Carmichael and Boyce (1986). Because few, if any, hatchery fish spawned naturally until 1990 we conducted a series of model runs to assess and compare the actual total population size, total number of females, number of natural spawners, and the reproductive capacity of the population (total number of eggs) with the same parameters for return years 1982-1991 assuming we had never operated the hatchery program. To estimate parameters in the without hatchery scenario we used adult progeny-parent ratios of the natural population to determine additional natural production that would have occurred from the natural adults removed for broodstock.

We conducted spawning ground surveys over the entire spawning area in the Imnaha River from 1986-1991 to assess and compare spawning distribution without hatchery and with significant numbers of hatchery fish spawning naturally. Run timing was estimated and compared for hatchery and natural fish in all return years. Timing was determined from collection of fish at the adult collection weir. Age-composition of natural fish was determined for each broodyear by scale pattern analysis and for hatchery fish from known age marked fish. Age-specific fecundity of hatchery and natural fish was estimated based on total ovary weight and mean egg weight of individual fish

Table 3. Summary of releases of Imnaha chinook salmon hatchery reared smolts. 1985 brood Imnaha chinook were released into Lookingglass Creek because the fish were infected with erythrocytic inclusion body syndrome (EIBS).

Location, Stock, Brood	Hatchery of rearing	Number released	Size (fish/lb)	Date of release	Location of release
Imnaha Basin:					
Imnaha:					
1982	Lookingglass	24,920	32.0	03/22/84	Imnaha R.
1983	Lookingglass	56,235	24.4	09/14/84	Imnaha R.
1983	Lookingglass	59,595	17.4	03/22/85	Imnaha R.
1984	Lookingglass	35,035	10.8	03/28/86	Imnaha R.
1986	Lookingglass	101,929	9.9-11.0	03/21-03/22/88	Imnaha R.
1986	Lookingglass	97,137	8.8-8.9	04/20-04/21/88	Imnaha R.
1987	Lookingglass	142,320	16.0	04/05/89	Imnaha R.
1988	Lookingglass	364,547	15.3-19.0	03/31-04/04/90	Imnaha R.
1988	Lookingglass	79,953	18.4	04/02/90	Big Sheep
1989	Lookingglass	267,670	16.0-22.3	03/22/91	Imnaha R.
1989	Lookingglass	131,239	14.8-20.5	04/09/91	Imnaha R.
1990	Lookingglass	262,548	11.0-21.6	03/30/92	Imnaha R.
Lookingglass[a]					
1982	Lookingglass	4,258	31.0	03/22/84	Imnaha R.
1984	Lookingglass	518	10.8	03/25/86	Imnaha R.

[a] Lookingglass stock mixed with Imnaha stock during 12 December 1983 and 23 November 1985 ice-up.

Figure 2. Map showing location of Imnaha chinook salmon hatchery facility.

sampled in 1989 and 1991. Genetics monitoring is being conducted as described in Waples et al. (1991).

RESULTS AND DISCUSSION

High prespawning mortality of wild adults collected and held for spawning occurred during the initial years of this program (Figure 3). These high mortality rates were associated with the poor holding conditions at the temporary adult facility from 1982-1986. Because of high mortality we transported adults to Lookingglass Hatchery for holding and spawning in 1987-1989. Since 1990 the adults have been held in the new facility at the Imnaha River and prespawning mortality has been relatively low.

Egg-to-smolt survival rates have improved substantially through the years (Figure 4). For the last three complete production broods (1987-1990) egg-to-smolt survival rates have been equal to or greater than 80% and it appears that the rate for 1991 brood will be similar. Smolt survival rates have been estimated for each brood year based on catch and escapement of marked (Ad+CWT) fish. Survival rates, including all ages, have been highly variable and have ranged from 0.05 to 0.2% for completed returns of the 1982-1986 brood years (Figure 5). We have observed very little harvest of Imnaha chinook in ocean fisheries and the majority of harvest occurs in Columbia River treaty and non-treaty net fisheries that target on sockeye salmon (Table 4). Most of the harvest of Imnaha chinook salmon occurs on Age 3 males (Figure 6).

Because of high prespawning mortality, high egg loss, and poor smolt-to-returning adult survival the adult (excluding Age 3 fish) progeny-to-parent ratios for hatchery fish were below 1.0 for the 1982-1986 brood years (Figure 7). We have implemented numerous programmatic changes and with returns completed through Age 4 for the 1987 brood year the progeny-to-parent ratio was greater than 1.5 and will likely approach 2.0 with Age 5 fish. The progeny/parent ratio for wild fish was substantially better than hatchery fish for the 1982 and 1983 broods, was slightly less than the hatchery ratio for the 1984-1986 broods, and was substantially less than the hatchery ratio for the 1987 brood. Based on preliminary estimates the progeny/parent ratio for the 1988 brood hatchery fish will be equal to or greater than the ratio for the 1987 brood.

There was little difference in total population size (hatchery and natural fish combined) from 1982-1989, however, in 1990 and 1991 total population size was substantially greater with the hatchery than it would have been without the hatchery (Figure 8). However, when we examine the total number of natural spawners (hatchery and natural origin) we see that without the hatchery there would have been a greater number of natural spawners in all years except 1991 (Figure 9). This reduction in natural spawners has resulted in part due to poor hatchery performance during initial years (high egg loss, high adult prespawn mortality, and poor smolt-to-adult survival) and because prior to 1989 few if any hatchery fish were passed above the weir to spawn naturally to replace the wild adults taken for hatchery broodstock. Although we have not completed analysis it appears that in 1992 and future years (given the present hatchery survival rates and broodstock management guidelines) we will see a greater number of natural spawners relative to where we would be without the hatchery. The total number of females (hatchery and natural) in the population as well as the total reproductive capacity of the population (hatchery and natural) was greater with the hatchery for the 1989-1991 return years than it would have been without the hatchery (Figures 10 and 11).

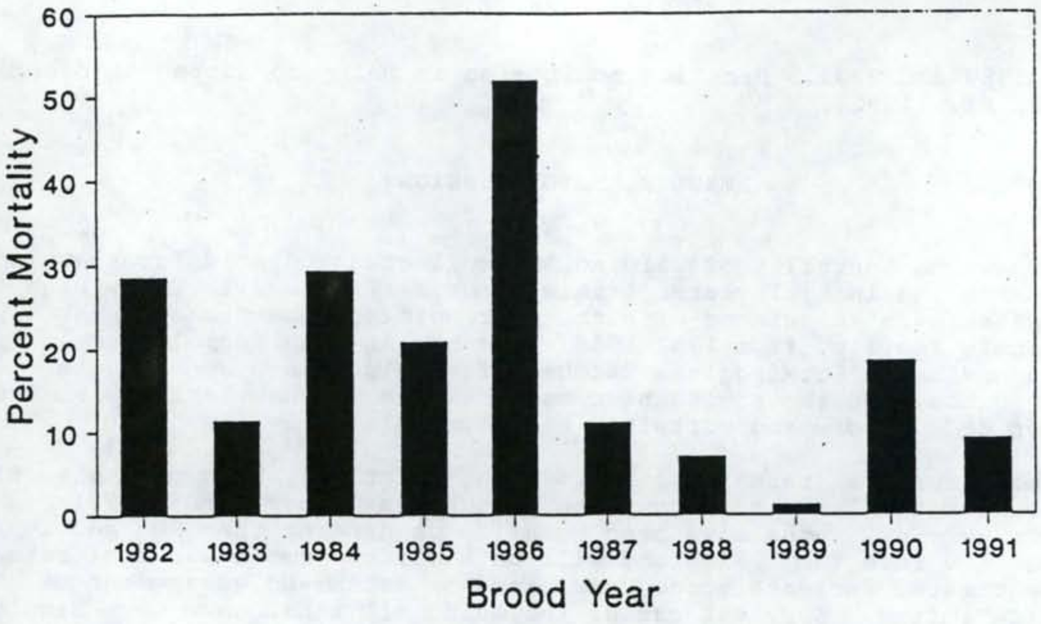


Figure 3. Mortality of Imnaha chinook salmon adults collected and held for hatchery broodstock, 1982-1991.

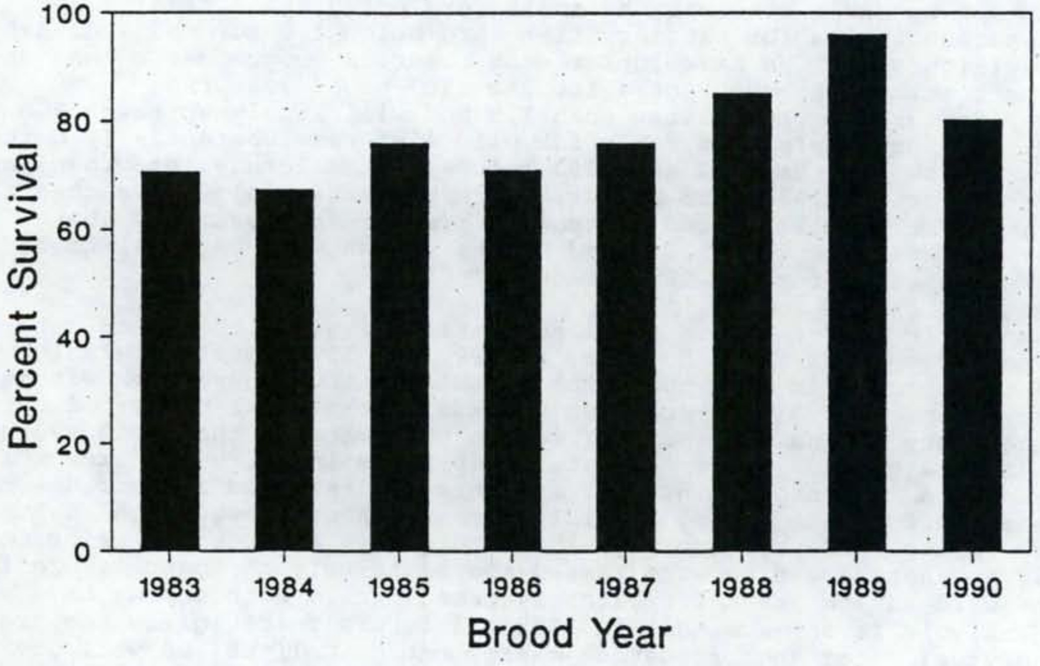
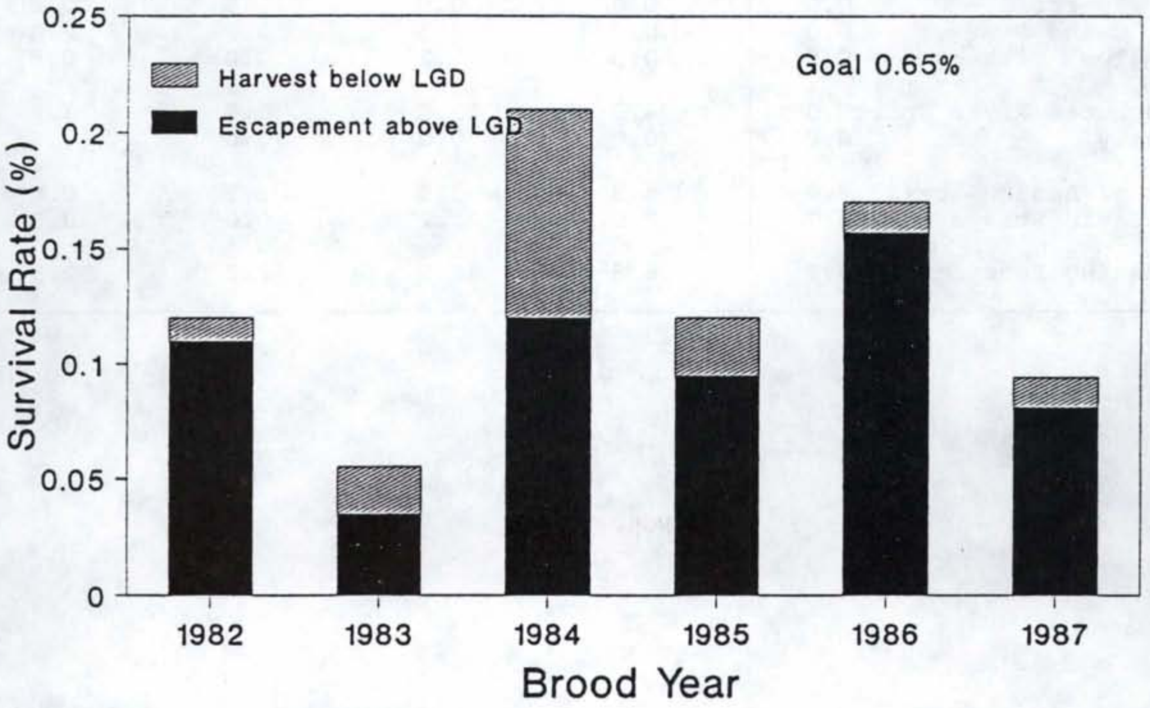


Figure 4. Egg-to-smolt survival rates for hatchery reared Imnaha River chinook salmon smolts at Lookingglass Hatchery, 1983-1990 brood years.



1985 brood released at Lookingglass Hatchery because of EIBS

Figure 5. Smolt-to-adult (catch and escapement) survival rates for hatchery reared Imnaha River chinook salmon. Catch and escapement of all ages are included. Survival rates based on recovery of coded-wire tagged fish.

Table 4. Catch and escapement distribution of Imnaha Hatchery chinook salmon. Distribution is based on catch and escapement of Ad+CWT fish and has not been adjusted for inter-dam adult losses. Values are preliminary because some ocean catch information has not been finalized.

Recovery location	1982 Brood	1983 Brood	1984 Brood	1985 Brood	1986 Brood
Ocean	0.0	26.1	0.0	0.0	0.6
Columbia River					
Treaty Net	0.0	8.7	7.0	21.8	0.9
Non Treaty Net	0.0	0.0	25.5	9.0	0.0
Sport	0.0	0.0	0.0	1.5	0.0
C and S	0.0	0.0	2.3	0.0	2.3
Test Fishery	0.0	0.0	1.2	0.0	0.9
Deschutes River Sport	0.0	4.3	12.8	0.0	1.6
Treaty	0.0	0.0	0.0	0.0	0.6
Out of Basin Strays	7.4	4.3	3.5	1.5	0.6
In Basin Strays	3.7	0.0	1.2	0.0	0.0
Spawning Escapement	88.9	56.6	46.5	66.2	92.5

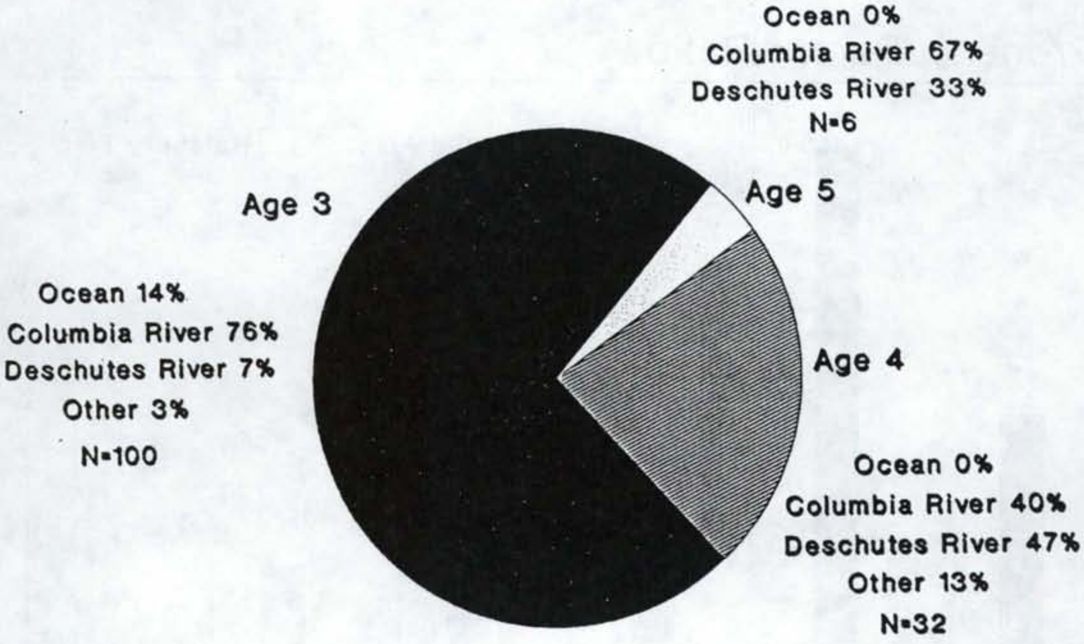
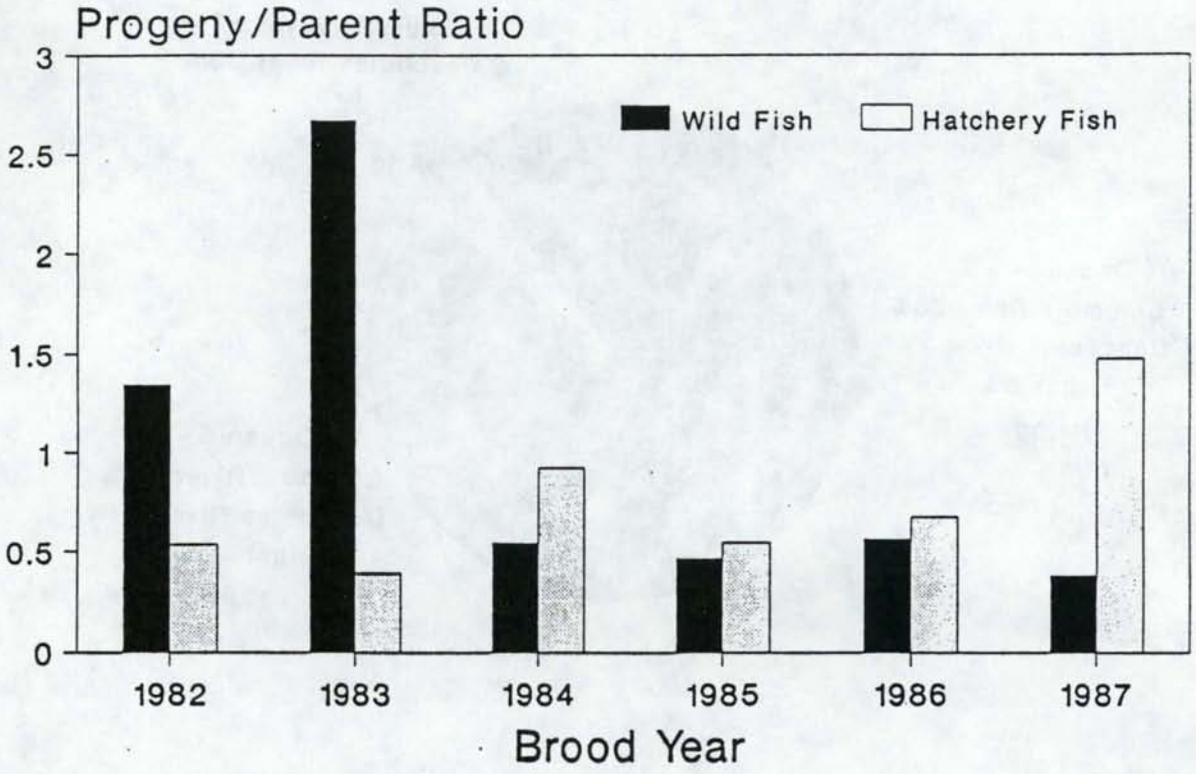


Figure 6. Mean age composition of 1982-1986 brood Imnaha River hatchery chinook that were harvested in ocean and in-river fisheries. Catch distribution within each age is also presented.



1987 brood age 4 returns only

Figure 7. Adult progeny-to-parent ratios for wild and hatchery Imnaha River chinook salmon, 1982-1987 brood years.

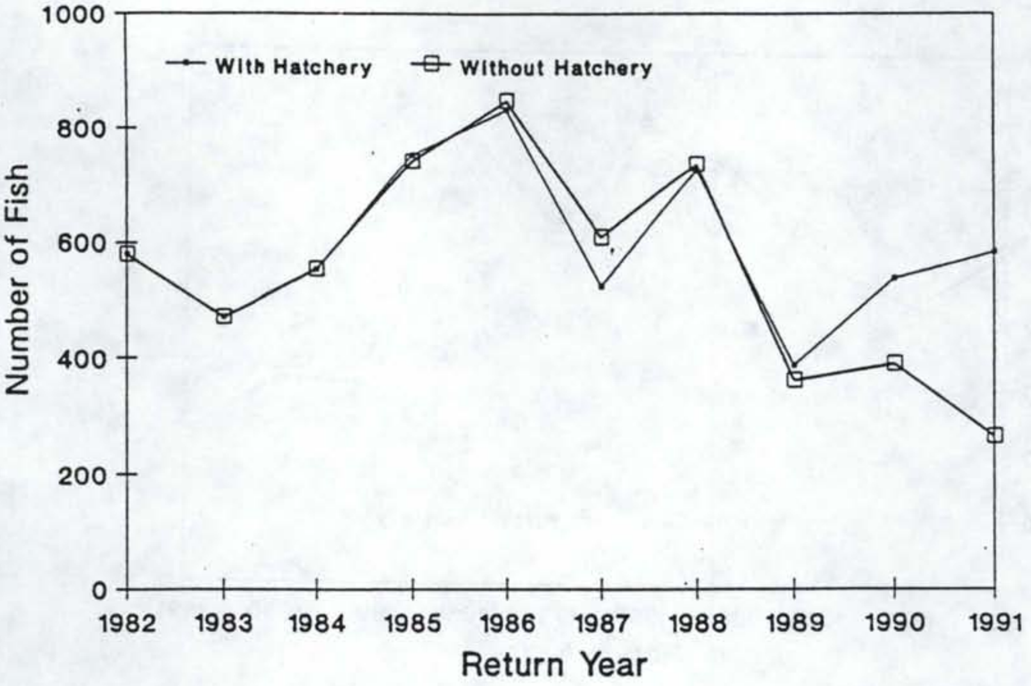


Figure 8. Comparison of actual total population size of Imnaha chinook salmon with hypothetical size had the hatchery program not existed.

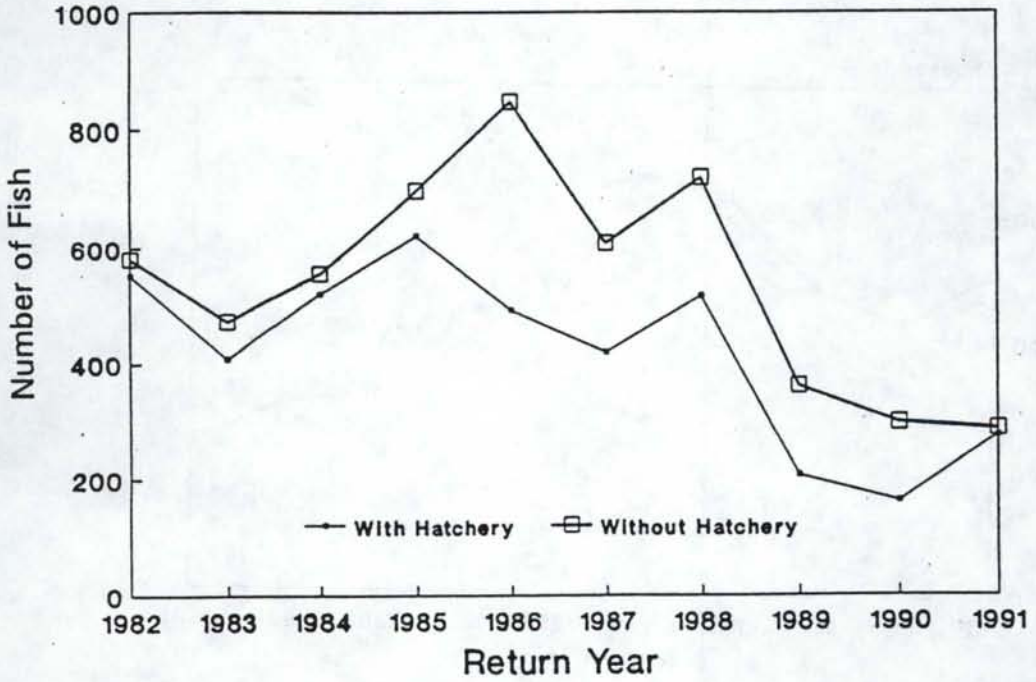


Figure 9. Comparison of the actual total number of natural spawners in the Imnaha River with hypothetical number of natural spawners had the hatchery program not existed.

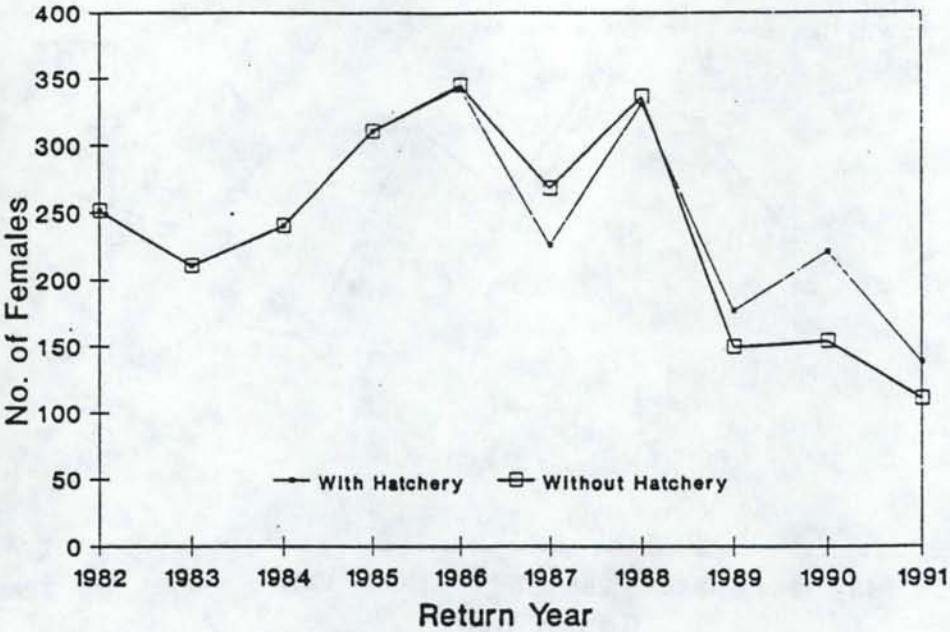


Figure 10. Comparison of the actual total number of females in the Imnaha River chinook population with hypothetical number of females had the hatchery program not existed.

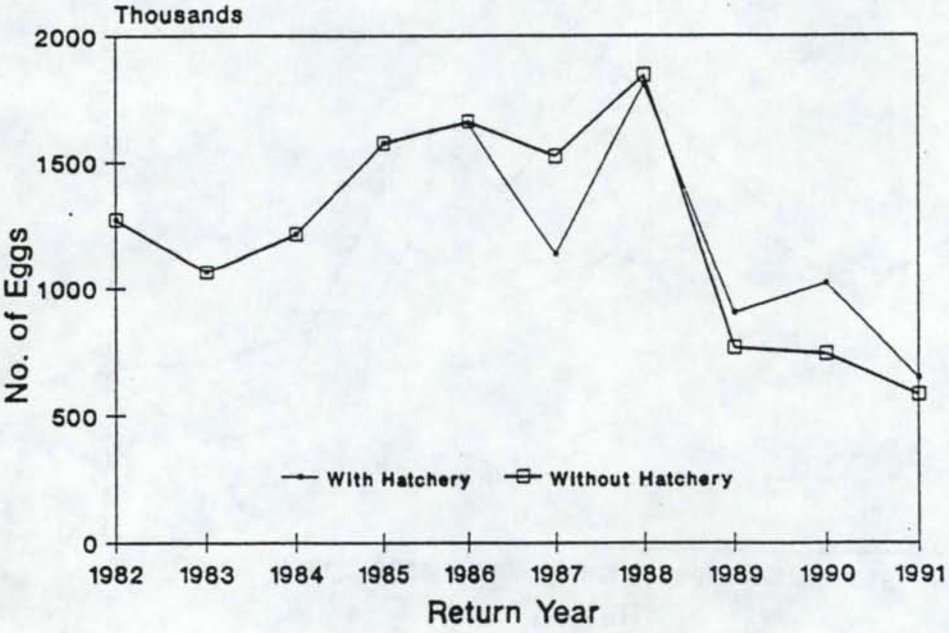


Figure 11. Comparison of the actual reproductive capacity (total eggs in population) of Imnaha chinook population with hypothetical reproductive capacity had the hatchery program not existed.

Because we release all Imnaha hatchery smolts at one location we are concerned that the natural spawner distribution may shift (concentrate near the facility) as the number of hatchery fish spawning naturally increases. We have conducted spawning ground surveys in the entire spawning area since 1986 in an effort to document shifts in spawner distribution. We saw little difference in spawning distribution in the 32.2 miles utilized for spawning between 1986-1989 (when very few, if any, hatchery fish spawned naturally) and 1990-1991 when a substantial number of hatchery fish spawned naturally (Figure 12). Marked hatchery fish carcasses were recovered throughout the entire spawning area during 1990-1992. Results of genetics monitoring are presented in Waples et al. (1991).

We have been monitoring and comparing aspects of the life history of hatchery and natural fish since 1986 when the first hatchery adults returned. As described earlier, during the initial years of trapping hatchery brood were collected from the later return component of the wild population. As a result, hatchery fish that returned from 1986-1988 exhibited a later run timing than the wild fish (see 1988 in Figure 13). However, from 1985 on a more representative cross-section of the wild run was collected for hatchery broodstock and this has resulted in nearly identical run timing of hatchery and wild fish for the 1989-1991 run years (Figure 14). We have seen a consistent pattern of earlier age-at-return for the hatchery fish (Figure 15). For example, the mean age-at-return for wild females was 39% Age 4 and 61% Age 5 while the mean age for hatchery females was 72% Age 4 and 28% Age 5. We believe the earlier age-at-return is a result of releasing the hatchery smolts at a larger size than wild smolts. We have decreased the size-at-release of a proportion of the hatchery production to a size similar to wild smolts to determine if we can shift age-at-return to older adults. We have been estimating and comparing age-specific fecundity of hatchery and wild fish and have found no differences (Table 5) to date.

The Imnaha chinook salmon supplementation program has been managed under the adaptive management philosophy. Extensive research, monitoring, and evaluation has been and is ongoing. A substantial number of programmatic changes have occurred since the initiation of this program in 1982 because of the information and knowledge gained. The goals and objectives that serve as the foundation of this program have shifted from emphasizing mitigation and hatchery production to emphasizing natural production enhancement. All aspects of this hatchery program have undergone substantial changes through time (Table 6) as discussed in previous text.

We have established the following broodstock management and mating protocols to guide the program in the future:

1. All fish that return to the weir site will be captured and we will retain no more than 30% of the natural fish by age and by sex for hatchery broodstock and the remainder will be passed above the weir to spawn naturally.

2. No more than 50% of the fish passed above the weir to spawn naturally will be hatchery origin fish.

3. Naturally produced fish will comprise a minimum of 30% of the fish used for hatchery broodstock.

4. The weir will be installed and broodstock collection initiated as early each year as physically possible.

5. We will use random split cross mating; split each female into two approximately equal groups and spawn each half female group with a different male. Thus each male and female contribute to two families.

River Miles	Location	Without Hatchery Fish 1986-1989	With Hatchery Fish 1990-1991
	Blue Hole		
2.0		19.4%	19.5%
	Indian Crossing		
7.7		55.5%	48.0%
	Mac's Mine		
4.5		4.3%	4.0%
	Weir		
3.5		13.2%	24.7%
	Crazyman Creek		
8.5		6.7%	3.4%
	Grouse Creek		
6.0		0.9%	0%
	Freezout Creek		

Figure 12. Comparison of natural spawning distribution in the Imnaha River in years when few or no hatchery fish spawned naturally (1986-1989) with years when a high proportion of spawners were hatchery fish (1990-1991).

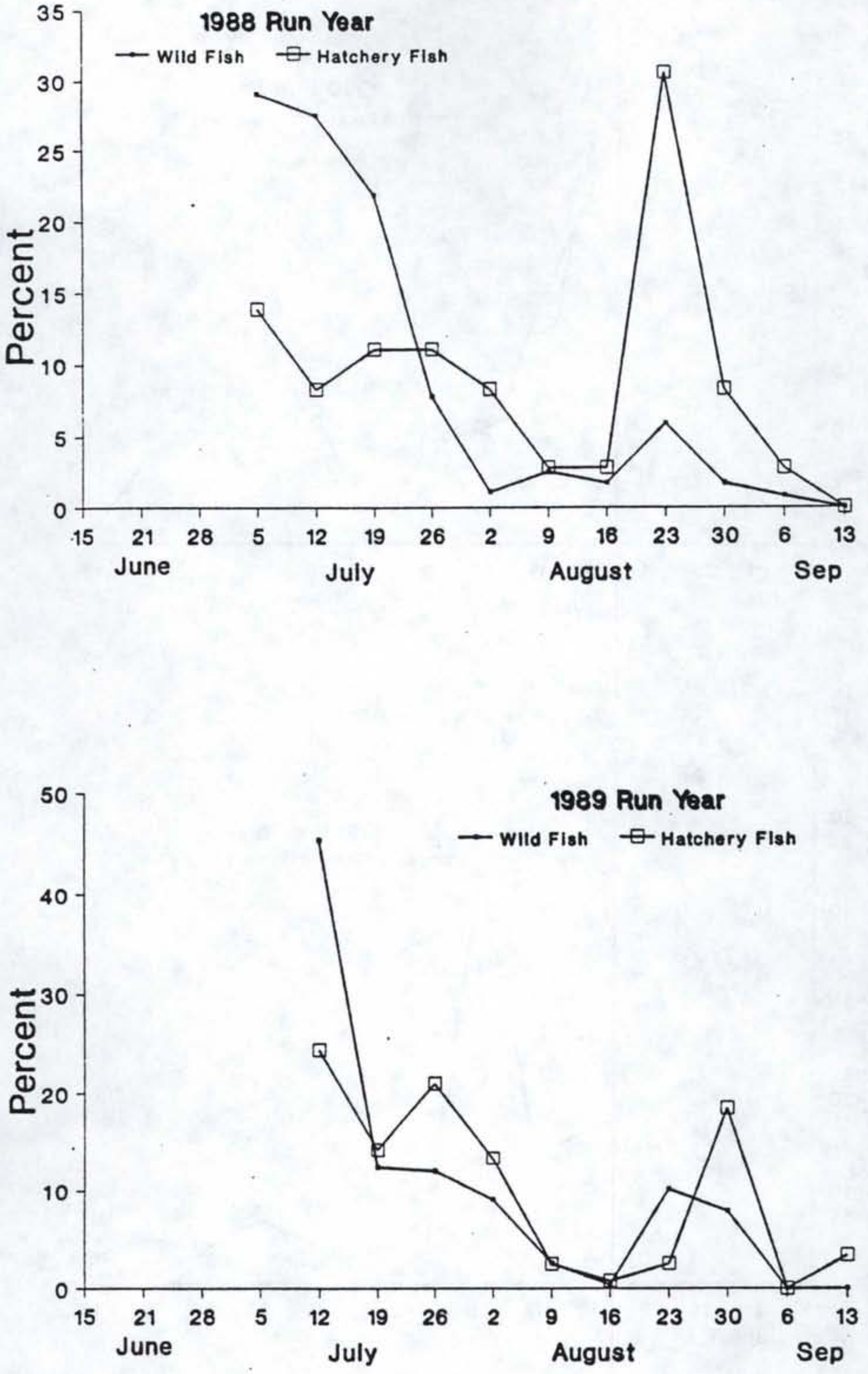


Figure 13. Run timing of wild and hatchery chinook salmon that returned to the Imnaha River weir in 1988 and 1989.

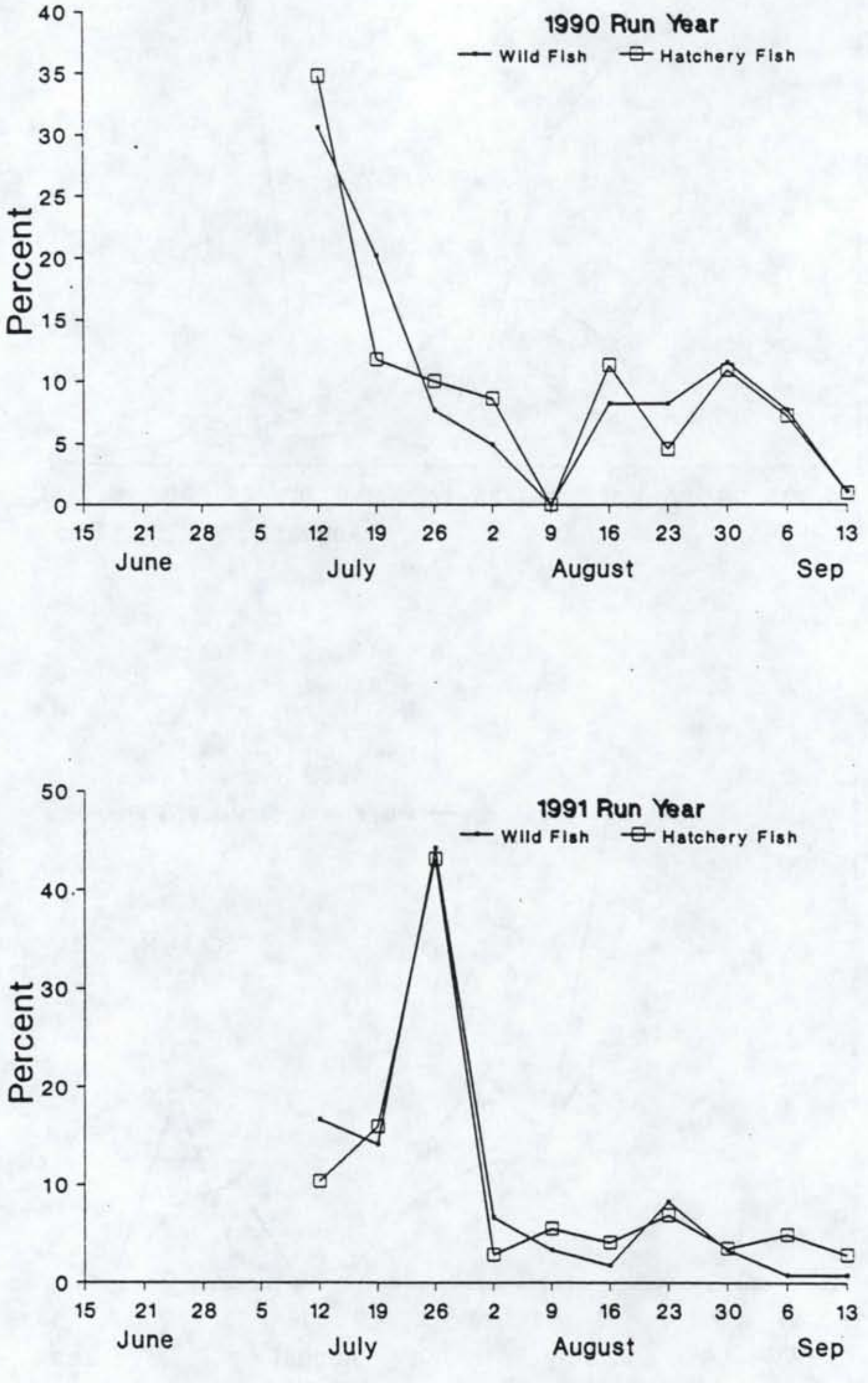


Figure 14. Run timing of wild and hatchery chinook salmon that returned to the Imnaha River weir in 1990 and 1991.

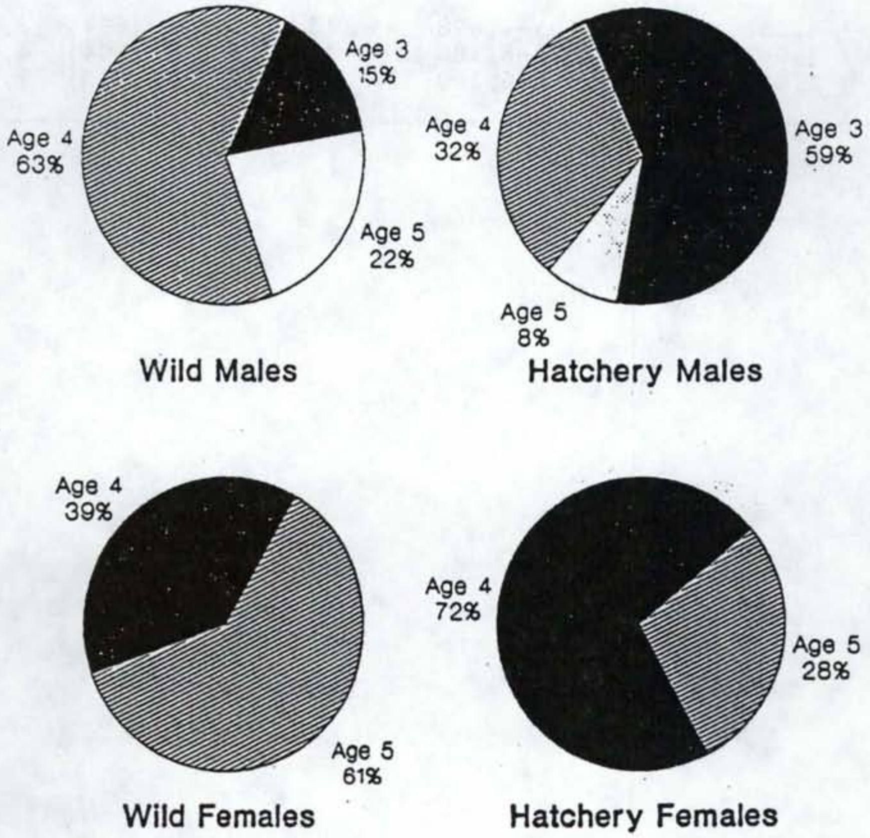


Figure 15. Mean age-composition of wild and hatchery chinook salmon that returned to the Imnaha River weir. Means are of the 1982-1986 brood years.

Table 5. Results of fecundity sampling of Imnaha stock chinook salmon, 1988 and 1991 brood years. Standard deviation is shown in parentheses.

Brood year, Origin	Age 4			Age 5		
	n	Mean	Range	n	Mean	Range
1988						
Wild	3	3,894 (286)	3,567-4,101	25	6,432 (889)	5,017-8,434
Hatchery	4	4,710 (768)	4,034-5,696	2	6,422 (950)	5,750-7,093
Combined	7	4,361 (716)	3,567-5,696	27	6,432 (874)	5,017-8,434
1991						
Wild	2	4,336 (483)	3,995-4,678	13	4,924 (667)	3,703-6,183
Hatchery	13	4,667 (864)	3,333-6,180	11	5,443 (904)	4,081-7,065
Combined	15	4,624 (818)	3,333-6,180	24	5,162 (810)	3,703-7,065

Table 6. Synopsis of programmatic changes made in the Imnaha chinook salmon hatchery program showing transition of objectives, guidelines, and operations from the original program to the present program.

Program Area	Original Program	Present Program
Production Goals	490,000	Based on broodstock guidelines and research needs, well below 490,000
Management objectives	Emphasized meeting mitigation goals	Emphasize natural escapement, natural production, and genetic principles
Hatchery broodstock	Kept most fish	Keep a maximum of 30% of natural fish and no more than 50% of natural spawning population will be hatchery fish
	Collected late in the run	Attempt to collect across entire run
	Limited treatment for BKD and fungus	Aggressive BKD and fungus treatment
Natural escapement above weir	Hatchery broodstock emphasized	Pass minimum of 70% of natural fish and equal number of hatchery fish
	No guidelines for hatchery fish	Hatchery fish not to exceed 50% of natural spawning population
Rearing/Release strategies	Standard densities	Reduced densities
	Large smolts	Large smolts and smolts of natural size
Monitoring and Evaluation	Focus on hatchery performance and mitigation success	Focus on hatchery and natural production performance

We believe it is too early in the program to determine if it is a complete success or a failure. The program is designed to minimize the genetic risks associated with the hatchery program while providing the maximum likelihood of providing a natural production enhancement benefit. Recent results indicate that we can increase total escapement and the number of natural spawners substantially by using the hatchery program. Major uncertainties remain including: how well the hatchery fish will perform in the natural system, what level of natural productivity will be achieved and sustained in the integrated population, and the long-term influence of the hatchery program on life history and genetic characteristics? We plan to continue conducting this program on an experimental basis and intensively monitor and evaluate program success.

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THE EFFECT OF NATURAL REARING ENHANCEMENT SYSTEMS ON THE POST-RELEASE SURVIVAL OF JUVENILE CHINOOK SALMON

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ABSTRACT

The use of conservation hatcheries to aid in restoration of depleted stocks of salmonids requires that techniques be developed to produce "wild-like" cultured fish. The literature we reviewed suggests that unintended selection and conditioning associated with standard culture practices generally produces salmonids which are less adept at foraging, less cryptic, and more aggressive than their wild-reared counterparts. The NATURES (Natural Rearing Enhancement Systems) program tests alternative fish culture techniques that may increase post-release survival.

To test the NATURES's hypothesis that rearing salmonids in structured raceways with cover, substrate, and artificial vegetation will increase post-release survival, groups of fall chinook salmon (*Oncorhynchus tshawytscha*) were reared in four conventional unstructured and eight NATURES structured 400-liter aquaria. Scan sampling techniques indicated that fish reared in tanks containing structure were more aggressive and ingested less decaying debris. At smoltification, the fish were PIT tagged, released into a nearby coastal creek, and challenged to survive outmigration to the estuary. Over 40% more fall chinook salmon reared under NATURES protocol survived outmigration from the test stream than fish reared under conventional raceway conditions.

FRY TO SMOLT STAGE: LIFE HISTORY ISSUES

SESSION LEADER: JIM CHANDLER,
IDAHO POWER COMPANY

SOME OBSERVATIONS ON SURVIVAL RATES AND THE FRY-TO-SMOLT STAGE OF CHINOOK AND COHO SALMON LIFE HISTORIES

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ABSTRACT

The various factors that act on salmon during their life cycle are ultimately expressed as mortality rates that can be depicted as survival curves and as recruit or reproduction curves for the various stages. Survival during the various stages can be variable, but usually falls within certain ranges for stocks that are self-sustaining. During the fry through smolt stage, a large fraction of the initial members of a cohort die, some from factors than can be influenced by managers and some that cannot. Mortality during the fry-to-smolt stage is most likely to be density related and lead to Beverton-Holt type stock-recruit curves. Not all mortality during the fry-to-smolt stage, however, is likely to be density dependent. The relative importance of survival rates for the various stages is discussed.

MARKING JUVENILE CHUM, SOCKEYE, AND FALL CHINOOK SALMON WITH STRONTIUM CHLORIDE

(Title modified from workshop program)

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ABSTRACT

Previous investigators have marked juvenile fishes by exposing them for several weeks to rearing waters or diets laden with elevated concentrations of trace elements. In this study, we immersed groups of newly emerged chum and sockeye fry into 9000, 1200, and 120 ppm solutions of strontium chloride for 24 h to ascertain whether such a short exposure period would produce recognizable marks. After being immersed, the fry were reared in freshwater for five wk. Vertebrae, opercula, and otoliths (sagitta) were collected from treated and control fish and analyzed with Inductively Coupled Plasma Mass Spectrometry (ICPMS). Control fish had about 200 ppm of strontium in their calcified tissues while chum salmon fry exposed to the two most concentrated strontium solutions had approximately 1600 ppm in their vertebrae and opercula and over 4000 ppm of strontium in their otoliths. Even the most dilute immersion treatment (120 ppm of $SrCl_2$) successfully marked the calcified tissues of chum and sockeye fry. We attempted to facilitate the marking process by using hyperosmotic baths (60 second immersions in a 4% NaCl solution was performed just prior to immersion), DMSO (some marking solutions were 0.005% DMSO) and heat (half the marking solutions were heated 4° C above ambient). All three increased the uptake of strontium, however, fry immersed in strontium solutions without these adjuvants retained almost as much strontium as their treated counterparts. Consequently, it is not necessary to use adjuvants to obtain good absorption and deposition of strontium in juvenile salmonids. A Scanning Electron Microscope (SEM) was used to take Back Scattered Electron Images of otolith thin sections obtained from treated fish. In all cases a highly visible ring of deposited strontium could be seen. Wave Dispersive Spectrometry disclosed that peak strontium counts in the bands depended upon the concentration of strontium chloride used in a marking bath. Fry immersed in 9000 and 1200 ppm baths had peak counts of 110,000 ppm while those held in 120 ppm baths had peak counts that ranged from 48,000 to 68,000 ppm. These data have led us to believe that multiple disparate strontium marks can be induced into salmon fry by varying the concentration or length of time a fish is held in a marking solution. Fingerling fall chinook were also placed into strontium solutions for 24 h and reared for 4 weeks. Scale and otolith samples taken from these fish were examined with WDS and prominent strontium marks were found in both tissues. The above studies demonstrate that salmonid fry and fingerlings can be quickly and permanently marked by immersing them into strontium chloride solutions. The potential use of such marks in survival, evaluation, growth, and life-history studies will perhaps only be limited by the imaginations of those who choose to use it.

Note: Manuscript was submitted to the Canadian Journal of Fisheries and Aquatic Sciences and is in review.

PARTIONING SALMON SMOLT YIELD IN THE SITUK RIVER, ALASKA TO PREDICT EFFECTS OF GLACIAL FLOODING

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ABSTRACT

The Hubbard Glacier is expected to dam Russell Fiord and cause glacial flooding of the lower 20 km of the Situk River, one of the most productive salmon rivers in Southeast Alaska. To determine probable effects on salmonids, smolt yield was partitioned between areas inside and outside the predicted flood zone. Rotary-screw traps were fished at the upstream limit of the flood zone and near the river mouth. Fish were marked and released 1 km upstream of each trap; recaptures were used to estimate fish numbers at each trap and survival between traps. Bootstrap statistical methods were used to estimate variance. Estimated total smolt yield from the river was 168,000 coho, 67,000 chinook, 893,000 sockeye, and 26,000 steelhead. Estimated survival between traps was 49% for coho smolts, 46% for chinook, and 84% for sockeye. By difference between traps, 33% of coho smolts, 45% of chinook, 34% of sockeye, and 0% of steelhead migrated from inside the flood zone. Our study demonstrated that upriver/downriver traps and mark-recapture techniques can effectively partition smolt yield, but methods must account for fish mortality between traps to accurately assess contributions from different areas of a watershed.

INTRODUCTION

New methods for measuring smolt yield are needed to help evaluate effectiveness of habitat restoration or assess effects of habitat degradation. Smolt yield is considered the most direct measure of salmonid production from a watershed (Koski 1992), and studies on habitat restoration can measure smolt yield to assess response in salmonid production. Partitioning smolt yield between different areas of a watershed can be used to measure contribution from specific areas and assess treatment effects (Seiler et al. 1984). Restoration of part of a watershed, however, may increase smolt yield from that part, but if fish merely redistribute within the watershed, the net effect is zero (Koski 1992). Thus, to fully evaluate restoration efforts, smolt yield should be partitioned between treatment and control areas, as well as from the entire watershed. This paper reports on methods we used to partition smolt yield between different areas of a watershed threatened by glacial flooding. These methods should be useful in assessing effects of land uses or restoration projects in different parts of a watershed.

The Hubbard Glacier is expected to dam Russell Fiord by the year 2000; overflow from "Russell Lake" would flood the Situk River, one of Southeast Alaska's most productive salmon and trout rivers (Figure 1). This paper reports on part of a 4-year study involving the National Marine Fisheries Service, U.S. Forest Service, and Alaska Department of Fish and Game to predict effects of glacial flooding on salmonids in the Situk River. A full report is in Thedinga et al. (1992).

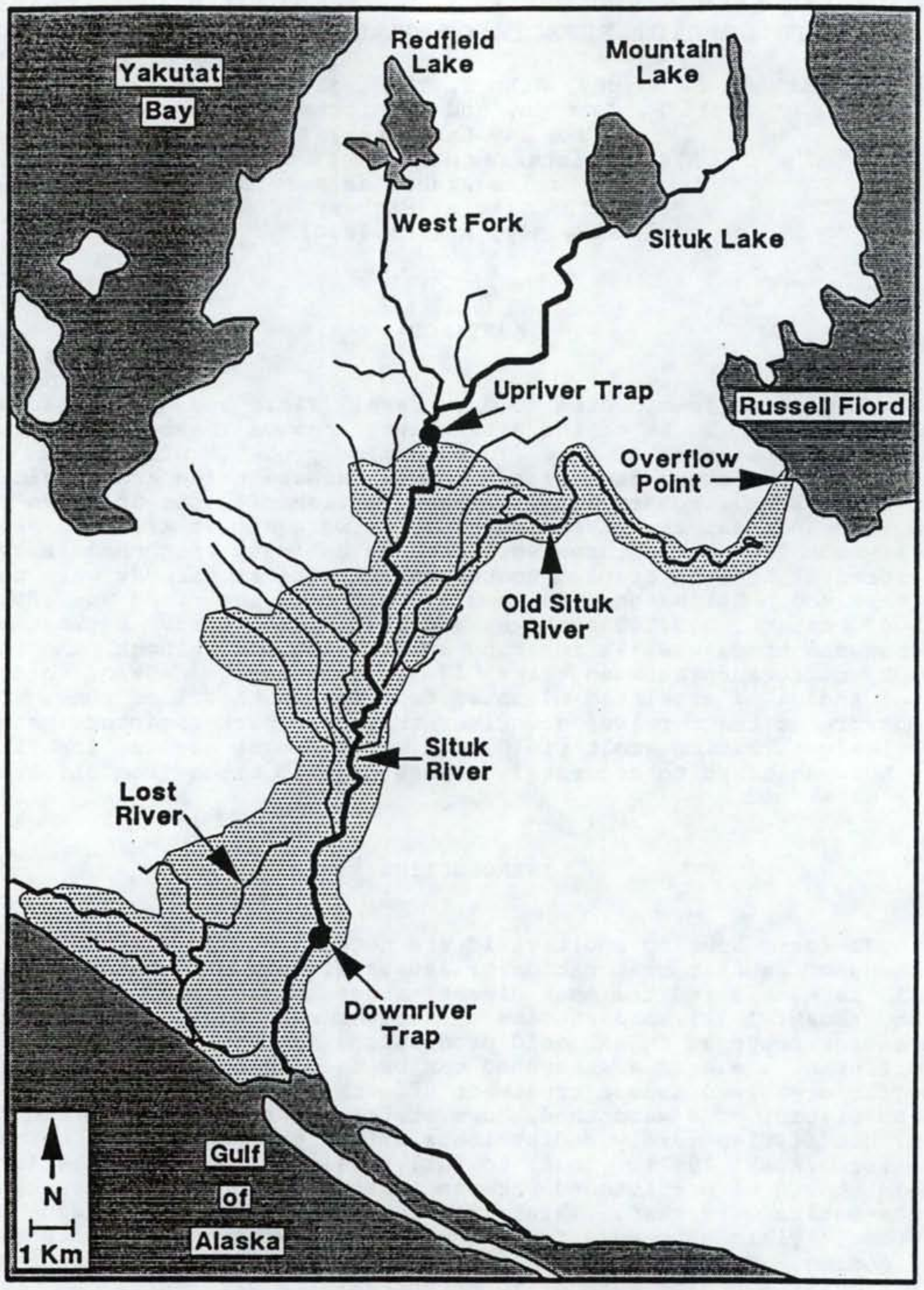

Figure 1. Map of the study area, showing location of traps used to catch juvenile salmonids on the Situk River. The predicted flood zone is stippled.

The Situk River, located near Yakutat, Alaska (Figure 1), flows across a glacial outwash plain and uplifted seabed and drains 200 km^2. The main stem is 25 m wide and 35 km long. Abundant groundwater and three headwater lakes help moderate river flow and temperature. Summer flow averages 6 m^3/s, and summer temperature ranges from 4 to 18°C. Flooding from "Russell Lake" would jeopardize commercial, recreational, and subsistence fisheries worth over $3 million annually to the Yakutat economy. Adult returns to the Situk River average about 150,000 sockeye (*Oncorhynchus nerka*), 60,000 coho (*O. kisutch*), 2,400 chinook (*O. tshawytscha*), 150,000 pink (*O. gorbuscha*), 500 chum (*O. keta*), 4,000 spring steelhead (*O. mykiss*), and 1,500 fall steelhead.

Glaciers have dammed Russell Fiord in the past, and as recently as 1860, overflow from "Russell Lake" flowed out the Situk River. The next flood would follow the old route down the Old Situk and main-stem Situk Rivers and out the Lost River to the Pacific Ocean (Figure 1), flooding 70% of present stream area. The "new" Situk River would flow 37 times larger and would be turbid with glacial silt and scoured sediments. The first years of flooding would be most destructive, but eventually the river would stabilize approximately in its former channel.

Salmonid smolt yield from the predicted flood zone provides a measure of possible impacts on production in the Situk River. Our objective, therefore, was to partition smolt yield between areas inside and outside the predicted flood zone.

METHODS

Two rotary-screw traps were fished from late March to mid-August 1990 at two sites: upriver, at the upstream limit of predicted flooding 20 km from the river mouth; and downriver, 3 km from the river mouth (Figure 1). The upriver trap fished the area outside the flood zone; the downriver trap fished the entire river; the difference between traps represented the area inside the flood zone. Each trap was a revolving stainless-steel, 2-mm-mesh cone on aluminum pontoons (Figure 2). The cone entrance was 2.4 m in diameter, and one-half (2.2 m^2) was submerged. An internal screw rotated the cone 3-6 rpm depending on water velocity. Fish passing through the cone collected in a live box where a revolving drum removed small debris. The traps were tied to shore and braced in the thalweg at river constrictions (16 m wide upriver and 24 m wide downriver; 1.2-2.4 m deep at both sites). They fished 6-11% of river cross-section upriver and 4-8% downriver. We built fences (5 m long, 6-mm mesh) to funnel fish into the traps. Water velocity (measured by meter) ranged 70-170 cm/s, mean daily water temperature ranged from 3°C in March to 16°C in July.

Trapped fish were removed each day and sorted by size groups (fry, parr, and smolts) into flow-through boxes with negligible water velocity. Fry were less than 45 mm fork length (FL); parr were 1 or more years old but did not look like smolts (i.e., silvered body, darkened fin tips). Up to 100 randomly selected fish per species and size group per week were measured for FL. Fish ages determined from scale samples from up to 50 fish per species per week were compared with FL frequencies to determine age composition.

Numbers of migrant smolts and parr were estimated by the trap-efficiency method by releasing marked fish upstream of each trap. At least 3 days per week, up to 1,000 smolts and 1,000 parr per species were marked with dye

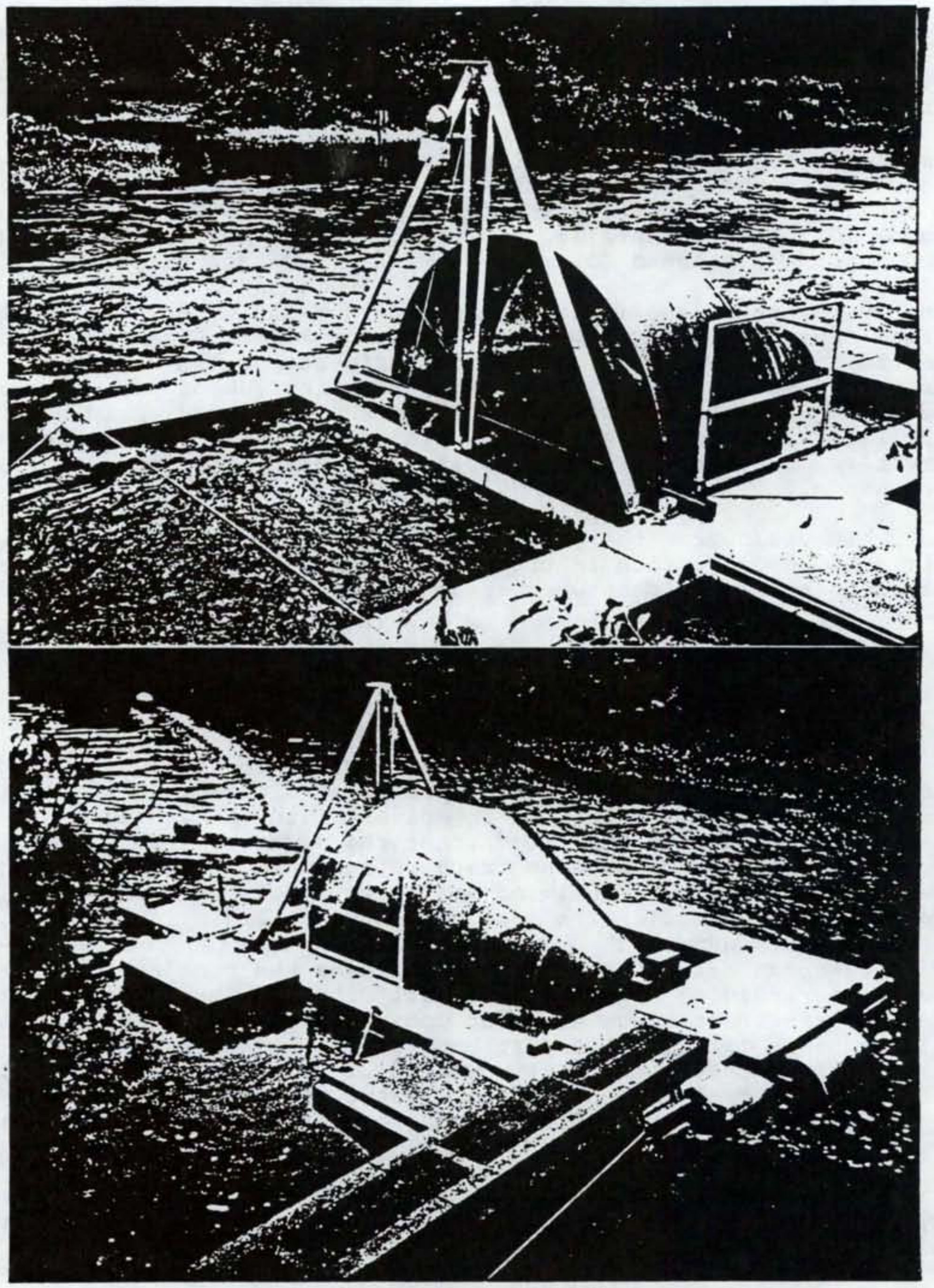

Figure 2. Rotary-screw fish trap on the Situk River in April 1990.

injected subcutaneously with a Panjet[1] medical instrument. We changed mark color on Monday and stopped marking on Thursday. Three colors (alcian blue, neutral red, and black India ink) were rotated the first 10 weeks. Neutral red was dropped after week 9 because of problems with retention and survival. Different marks were used at each trap. Upriver, salmon smolts were tattooed on the upper caudal fin or on both upper and lower caudal fin, steelhead smolts between the pelvic fins, and parr on the anal fin. Downriver, salmon smolts were tattooed on the lower caudal fin, steelhead smolts on the ventral caudal peduncle, and parr on both upper and lower caudal fin. Marked fish were held until dusk, moved in aerated tubs 1 km upstream, and released in quiet water. Each day, all trapped smolts and parr were checked for marks.

Short-term mark survival was tested to determine actual number of marks available for capture by periodically holding a random sample of 25 marked fish per species. Fish were held in aerated tubs or flow-through boxes in the river, and live fish with visible marks were counted after 24 h. Short-term mark survival was calculated as:

$$\hat{S} = s/h \; ; \qquad (1)$$

$\hat{s}$ is estimated survival and retention of marks; s is number of surviving fish with visible marks; and h is number of marked fish held. The number of surviving marks released was calculated as:

$$\hat{M} = m\hat{S} \; ; \qquad (2)$$

$\hat{M}$ is estimated number of surviving marks, and m is number of marks released. Short-term survival and mark retention after 24 h were generally high, but differed between species ($P < 0.001$; G test) and mark color ($P < 0.001$). Mark retention was 100% for coho and chinook, but 96% for sockeye and 97% for steelhead. Blue and black marks were retained better (98-99%) than red marks (90%). Mark retention was a problem in weeks 8 and 9 because of Panjet malfunction; data from weeks 7 and 10 were averaged to estimate fish numbers in weeks 8 and 9. Sockeye smolts were fragile, and their survival (mean, 95%) was lower ($P < 0.05$) than for other smolts (mean, 99%). Blue- or black-marked sockeye survived better ($P < 0.05$) than red-marked sockeye.

Trap efficiency was estimated by the equation:

$$\hat{E} = R/\hat{M} \; ; \qquad (3)$$

$\hat{E}$ is estimated trap efficiency, and R is number of marked fish recaptured. Number of migrants was estimated by the equation:

$$\hat{N} = U/\hat{E} \; ; \qquad (4)$$

$\hat{N}$ is estimated number of migrants, and U is unmarked catch. Trap efficiency and mark survival were first calculated separately for each week and then tested for differences between consecutive weeks. If not different ($P > 0.05$; Chi-square test), data were pooled. Recaptures were generally made soon after release: 2-28% within 1 d and 90% within 1 week.

Variance for $\hat{N}$ each week was determined by the bootstrap method (Efron and Tibshirani 1986) with 1,000 iterations. Each bootstrap iteration involved calculating $\hat{N}^*$ by equations (1-4) after drawing s^* from the binomial distribution ($h, \hat{s}$), R^* from the binomial distribution ($\hat{M}$, $\hat{E}$), and U^* from the binomial distribution ($\hat{N}$, $\hat{E}$), where asterisks denote bootstrap values.

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[1] Reference to trade names does not imply endorsement by the National Marine Fisheries Service, NOAA.

Variance of weekly $\hat{N}$ was summed to obtain variance for the total migration[2].

Because fish mortality between traps would cause an underestimate of the flood zone's contribution, we estimated fish mortality from the equation:

$$\hat{S} = R_d \,/\, (\hat{E}_d \hat{M}_u) \;; \tag{5}$$

$\hat{S}$ is estimated survival of marked fish between traps; R_d is number of upriver-marked fish recaptured downriver; $\hat{E}_d$ is estimated efficiency of the downriver trap; and $\hat{M}_u$ is number of marks released at the upriver trap. Variance for $\hat{S}$ can be obtained by bootstrap method. Important assumptions were that marking did not affect survival (other than initially), all surviving marked fish migrated past the downriver trap, and all recaptured marked fish were counted. Because many parr apparently remained in the area between traps and did not go to sea, their survival was not estimated.

RESULTS AND DISCUSSION

Trap efficiency differed between species and time period (Figure 3). Overall trap efficiency for smolts was greatest for chinook (24%), intermediate for coho (12%) and sockeye (7%) smolts, and least for steelhead smolts (3%). Trap efficiency depended on river stage, position of the trap and fences, and amount of debris on the trap. Differences between species probably reflected differences in migratory behavior and ability to avoid the trap. Efficiency generally increased during the study as we adjusted traps and fences. About 117,000 smolts and 3,000 parr were trapped upriver; 69,000 smolts and 22,000 parr were trapped downriver (Table 1). Based on estimated trap efficiency, a total of 1.1 million smolts and parr passed the upriver trap, and 1.3 million passed the downriver trap. Total yield from the river, therefore, was 1.3 million fish.

Coho smolts migrated mostly from mid-May to late June; peak migration was in late May and early June (Figure 4). Migration of coho parr was greatest in mid-June and in July during freshets. Mean FL of smolts was larger upriver than downriver (94-111 mm upriver and 86-95 mm downriver), and smolts were older upriver than downriver ($P < 0.05$; Kolmogorov-Smirnov test). Nearly 60% were age 2 or 3 upriver, compared to 17% downriver; 83% of smolts downriver were age 1. The decline in size and age of smolts downriver could be explained by predation mortality during migration down the main-stem river and by an influx of smaller, younger smolts from inside the flood zone.

The number of coho smolts was greater ($P < 0.01$; *t* test) at the upriver trap than at the downriver trap (Table 1): 230,000 upriver but only 168,000 downriver--a 27% decline. This decline would be greater if smolts originating from the flood zone were included with the upriver population estimate. The Old Situk River, for example, produced 26,000 coho smolts in 1989 (Thedinga et al. 1991). With this number added to the migrants passing the upriver trap, the loss of coho smolts between traps would be 34%. In contrast to coho smolts, coho parr were much more numerous downriver than upriver: 127,000 downriver and only 31,000 upriver.

Chinook smolts migrated in June and July, beginning 1 week earlier upriver than downriver and peaking at both traps in July (Figure 5). Mean FL of chinook smolts at the upriver trap increased gradually between June and August

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[2] A Fortran program for calculating bootstrap variance is available from the authors upon request.

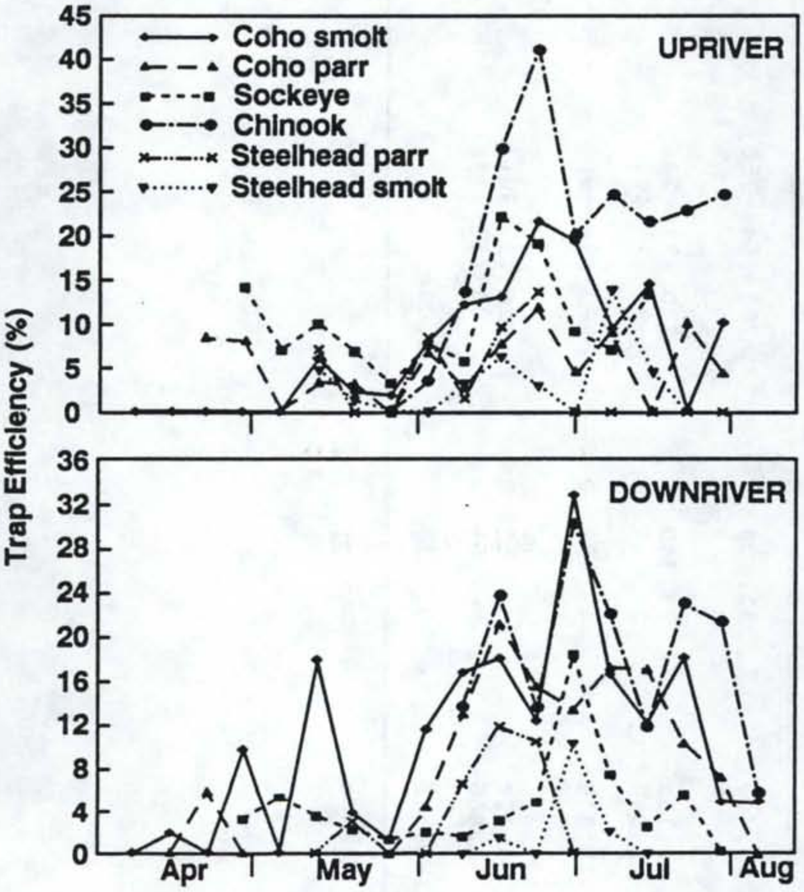

Figure 3. Trap efficiency for different species and size groups of juvenile salmon from the upriver and downriver traps on the Situk River, April to August 1990.

Table 1. Total catch and estimated number ($\hat{N}$) of juvenile salmonids at upriver and downriver traps on the Situk River, April to August 1990. The 95% confidence intervals are in parentheses.

$\hat{N}$ in thousands of fish (95% C.I.)		Catch		
Species and stage	Upriver	Downriver	Upriver	Downriver
Sockeye Smolts	74,460	31,304	701 (646-756)	893 (635-1,150)
Coho				
Smolts	22,131	23,740	230 (216-244)	168 (138-197)
Parr	1,997	20,941	31 (22-40)	127 (116-142)
Chinook Smolts	19,335	13,033	80 (74-85)	67 (59-68)
Steelhead				
Smolts	1,124	534	26 (15-38)	26 (0-72)
Parr	1,466	659	20 (15-41)	8 (5-12)
Total	120,513	90,211	1,088	1,289

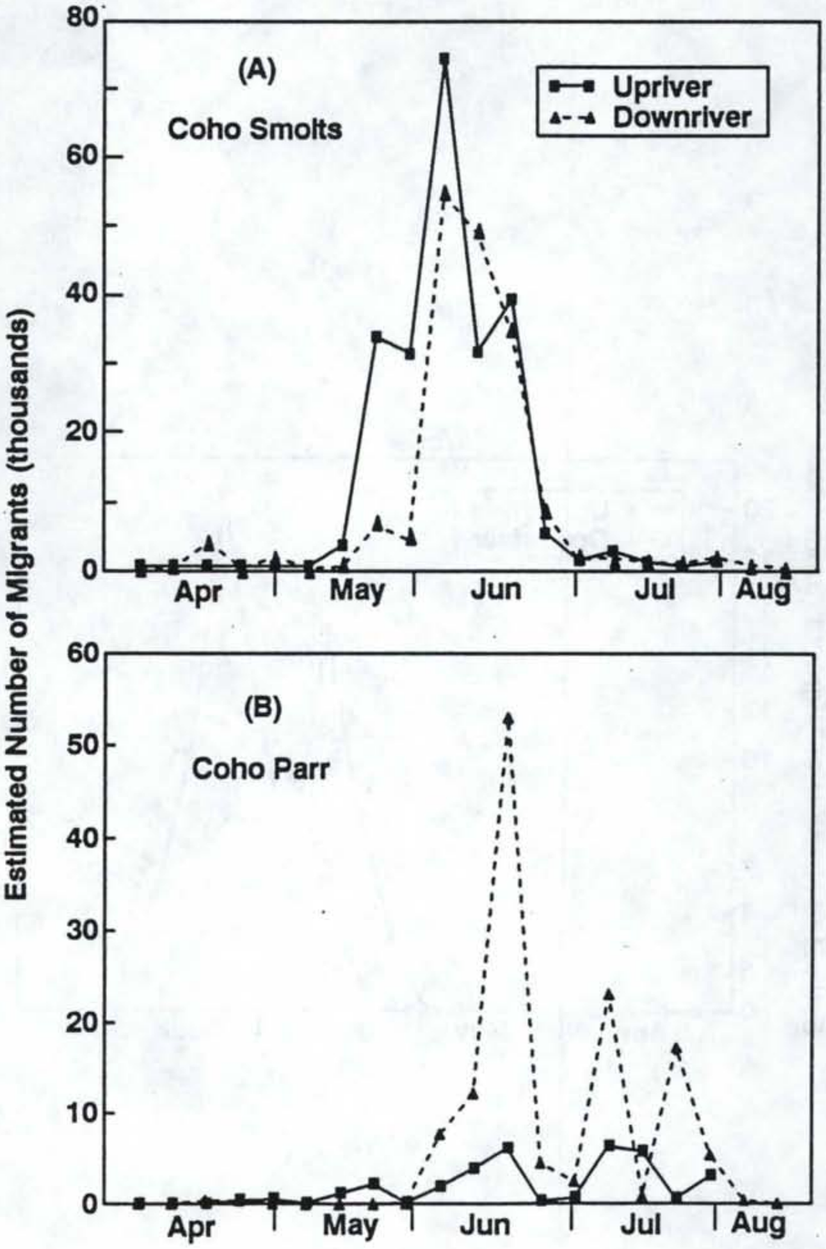

Figure 4. Estimated number of coho smolts (A) and parr (B) at upriver and downriver traps on the Situk River, April to August 1990.

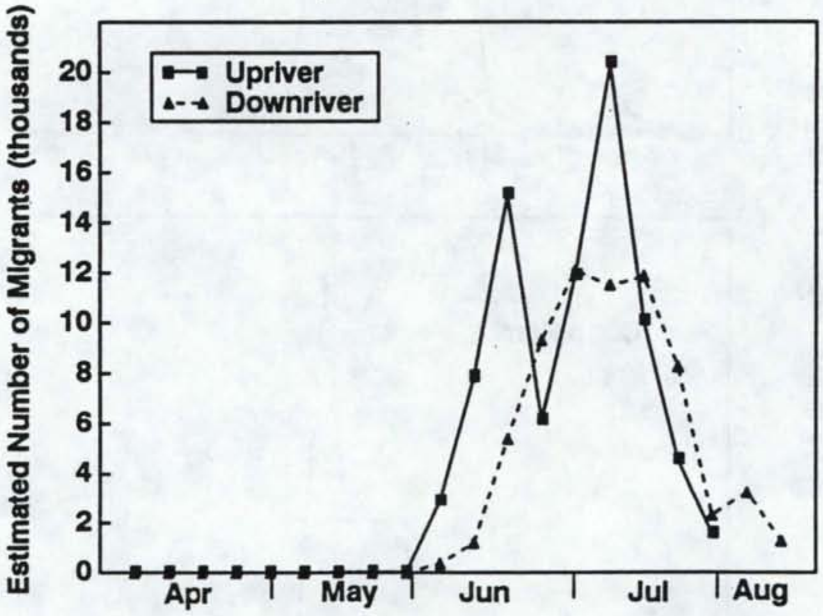

Figure 5. Estimated number of chinook smolts at upriver and downriver traps on the Situk River, April to August 1990.

(from 66 to 85 mm upriver and from 61 to 89 mm downriver). Nearly all (99.9%) smolts were age 0. Age and migration timing were unusual for Alaska chinook and resembled ocean-type chinook in the Pacific Northwest and British Columbia (Healey 1983). Except for the Deshka River (Delaney et al. 1982), Alaska chinook smolts are predominantly age 1 (Taylor 1990). Peak migration in other Alaska rivers is in late May (e.g., Murphy et al. 1991); in the Situk River, the peak was in July.

As with coho smolts, chinook were more numerous upriver than downriver: 80,000 upriver, but only 67,000 downriver--a 16% decline (Table 1). This decline would be greater if chinook fry that moved downstream in spring were added to the upriver population estimate. Chinook fry were not estimated by mark-recapture because of small size (<45 mm FL), but 2,149 chinook fry were caught by the upriver trap in April and May, and no fry were caught downriver. Based on likely trap efficiency of 5%, over 40,000 chinook fry entered the flood zone in spring. Thus, the loss of chinook smolts and fry between traps was about 44%.

Estimated survival of marked fish between traps corroborates the substantial decline in smolt populations between traps. Survival of marked fish between the traps was 49% for coho smolts, 46% for chinook smolts, and 42% for sockeye smolts; too few steelhead were caught to estimate survival (Table 2). Survival of coho and chinook stayed in a narrow range of only 38 to 42% during most of the migration. Chinook survival increased to 81-90% in the last 2 weeks. Estimated sockeye survival was variable, ranging from 4 to 69%.

Survival could have been underestimated because of delayed handling mortality. Overall handling mortality was negligible (<1%) in coho, chinook, and steelhead, but was nearly 3% in sockeye smolts (Table 3). Handling mortality in recaptured coho, chinook, and steelhead was also negligible, but about 6% in sockeye. The 6% mortality in recaptured sockeye indicated substantial delayed mortality from marking.

Problems identifying marks also contributed to underestimating survival. Mark recognition was tested in June by double marking sockeye on both upper caudal (the usual upriver mark) and lower caudal (the downriver mark) and releasing them at the upriver trap along with regular releases. Double marks were observed at nearly three times the rate of single marks at the downriver trap ($P < 0.001$; Chi-square test; Table 4), indicating that workers at the downriver trap were less efficient in observing marks applied at the upriver trap than marks applied by themselves. Because of this bias, sockeye survival may have been underestimated by two-thirds. An estimate of sockeye survival based only on double-caudal marks was 79% (Table 5). Mark recognition was not tested in coho and chinook.

Based on the difference in smolt populations at the two traps and estimated survival of smolts between the traps, the contribution from the flood zone to the river's total smolt yield was 33% of coho, 45% of chinook, and 34% of sockeye (Table 6). Because of problems in mark recognition, possible delayed handling mortality of marked fish, and possible increased vulnerability of marked fish to predators, smolt survival between traps may have been underestimated and the contribution from the flood zone may have been overestimated. Better estimates of smolt mortality during migration would provide a better assessment of the flood zone's contribution.

Predation could account for high smolt mortality. River otters (*Lutra canadensis*), mink (*Mustela vison*), common mergansers (*Mergus merganser*), belted kingfishers (*Megaceryle alcyon*), and great blue herons (Ardea herodias), as well as Dolly Varden (*Salvelinus malma*), are common in the Situk River and are potential predators of juvenile salmonids. Abundant salmonid fry and smolts may attract predators to the river, and such predator concentrations could cause high smolt mortality. Predation mortality of

Table 2. Upriver marks released, downriver recaptures, and estimated survival of marked fish between traps. Symbols refer to equation (5). Data included are for weeks with >100 marked fish released. Data for weeks 8 and 9 were omitted because of Panjet malfunction. Too few steelhead were caught to estimate survival.

Week	Marks Released ($\hat{M}_u$)[a]	Recaptures Downriver (R_d)[b]	Expanded Marks ($R_d/\hat{E}$)[c]	% Survival of Marks ($\hat{S}$)
		Coho Smolts		
10	1,836	146	1,209	66
11	1,528	107	630	41
12	1,409	99	558	40
13	638	33	251	39
15	114	8	48	42
Total	5,539	325	2,696	49
		Chinook Smolts		
11	159	7	35	22
12	1,177	103	483	41
13	834	66	354	40
14	879	100	338	38
15	762	63	320	42
16	769	48	307	40
17	444	59	400	90
18	194	20	157	81
Total	5,218	466	2,394	46
		Sockeye Smolts		
6	417	1	18	4
7	1,213	9	287	24
10	934	10	511	55
11	1,214	15	837	69
12	1,647	23	713	43
13	1,150	21	409	36
14	711	50	271	38
15	639	26	347	54
16	170	1	39	23
Total	8,095	156	3,432	42

[a] Estimated number after accounting for 24-h survival and mark retention.

[b] Total recaptures over 1-3 week period.

[c] Number of recaptures divided by downriver trap efficiency in week of recapture.

Table 3. Handling mortality of smolts and parr caught in the upriver trap.

	Released Alive	Died	% Mortality
		Unmarked Catch	
Coho	75,357		640.1
Chinook	20,104		440.2
Steelhead	3,239		230.7
Sockeye	73,018	2,016	2.7
		Recaptured Fish	
Coho	806	6	0.7
Chinook	1,398	6	0.4
Steelhead	55	0	0.0
Sockeye	1,021	64	5.9

Table 4. Comparison of the percentage of sockeye marked with single and double-caudal tatoos released at the upriver trap and later observed at the downriver trap on the Situk River, Alaska. Data are from 18 June-1 July 1990.

	Single Black	Single Blue	Double Black or Blue
Marks Released Upriver	1,313	982	503
Number Observed Downriver	14	13	16
% Observed Downriver	1.1	1.3	3.2

Table 5. An estimate of survival of double-caudal marked sockeye in the main-stem Situk River between 18 June and 1 July 1990, based on equation (5). Symbols are defined in the text.

Week	Upriver Marks Released ($\hat{M}_u$)	Downriver Recaptures R_d	Downriver Trap Efficiency $\hat{E}_d$	Expanded Recaptures ($R_d/\hat{E}_d$)	Survival $\hat{S}$
12	334	8	0.032	250	
13	164	7	0.050	140	
14	0	1	0.184	5	
Total	498	16		395	0.79

Table 6. Estimated contribution of the flood zone, based on difference in estimated number of smolts at upriver and downriver traps and estimated survival between traps. Smolt numbers are in thousands.

	Upriver $\hat{N}$	Survival %	Upriver Survivors	Downriver $\hat{N}$	Flood Zone Contribution
Coho Smolts	230	0.49	113	168	33%
Chinook	80	0.46	37	67	45%
Sockeye	701	0.84[a]	589	893	34%

[a] Survival based on estimated survival of double-caudal marks only (79%) and estimated 6% marking mortality (Tables 3 and 5).

Atlantic salmon smolts in two Swedish rivers was 50% (Larsson 1985). Mergansers caused up to 10% mortality in juvenile salmonids in a British Columbia stream (Wood 1987). At least 100 mergansers occur along the Situk River during the smolt migration (personal observation). If each merganser consumed 400 g of fish per day (Wood and Hand 1985) during the 7-week smolt migration, they would consume 200,000 10-g smolts. The combined effect of all predator species could explain the observed loss of migrating smolts.

Predation mortality in migrating smolts appears to be much greater than generally realized. The losses probably are more evident when smolt yield is quantified and partitioned between different areas of a watershed. In our study, we did not anticipate that more than one-half of the migrating smolts would disappear between upriver and downriver traps. Such heavy mortality not only affects results of studies like ours using mark-recapture methods, but may have important consequences for a river's salmon production and a manager's ability to conserve or restore depleted salmon stocks. More research is needed to further quantify of predation mortality in migrating smolts and assess its consequences for fisheries.

Although smolt yield is probably the best measure of salmonid production from a watershed as a whole, it may give only a partial measure of the contribution of specific areas within a watershed. Movement of fish between summer rearing, wintering, and spring staging areas complicates the assessment of an area's production. In the Situk River, for example, an estimated 70% of the river's juvenile salmonids rear in the flood zone in summer, but many move to other wintering areas from which they migrate to sea the following spring. Many parr migrate to staging areas in spring before they develop smolt characteristics. Complementary studies of summer rearing areas and surveys for residual parr should be conducted along with studies of smolt yield to fully evaluate contributions from different parts of a watershed.

Mark-recapture methods for estimating smolt yield have been developed in a number of recent studies (e.g. Macdonald and Smith 1980; Rawson 1984; Seiler et al. 1984). The simplest approach is to "expand" trap counts based on estimated trap efficiency, as in equations (3 and 4). The trap-efficiency approach is essentially a Petersen estimate:

$$\hat{N} = MC/R-M = U/(R/M) \; ; \qquad (6)$$

where C is the catch examined for marks and other symbols are as defined previously. The left-hand equation is the simple Petersen estimate (Ricker 1975), of the unmarked population (marked fish are released back upstream, so they are subtracted from the usual Petersen estimate), and the right-hand equation is the expansion of unmarked catch by trap efficiency. Thus, trap-efficiency and Petersen methods give the same estimate of fish numbers. Variance for $\hat{N}$ ($=U/E$) is not easily obtained because it involves the ratio of two binomial random variables (U and R): R follows the binomial distribution (E, M); U follows the binomial distribution (E, N). Thus, simply finding confidence interval for E and expanding U understates the true variance for $\hat{N}$. In simple situations, variance for $\hat{N}$ can be obtained by the Petersen method. The situation is more complicated, however, when additional binomial variables are included, as for example, when mark retention, mark recognition, and smolt mortality are quantified, or when marked smolts delay migration after release. Fortunately, bootstrap statistical methods allow simple calculation of variance with as many variables included as deemed necessary. In parametric bootstrap models (Efron and Tibshirani 1986), elements of sampling variability can be incorporated easily as binomial variables to provide realistic confidence intervals for $\hat{N}$. In our study, for example, we included short-term mark retention and survival, recapture rate, and catchability in a bootstrap estimate of variance.

Use of upriver/downriver traps and mark-recapture methods can be effective in partitioning smolt yield between different areas of a watershed, but methods must account for fish mortality between traps. Mortality can be estimated by equation (5), but delayed marking mortality and observer bias in mark recognition must be overcome. Tattooing has potential for rapidly marking smolts, but dyes must be fully evaluated for retention, recognition, and adverse effects on fish survival. Further development of tattooing methods and close quality control to ensure complete identification of recaptured fish should allow accurate partitioning of smolt yield between multiple areas of a watershed. This methodology should be useful in assessing impacts of habitat alteration on salmonids and evaluating effectiveness of habitat restoration.

SUMMARY

1. Estimated total smolt production from the Situk River watershed was 168,000 coho, 67,000 chinook, 893,000 sockeye, and 26,000 steelhead.
2. A 16-27% decrease in number of coho and chinook smolts between upriver and downriver traps was probably caused by predation.
3. Estimated survival of smolts between traps, based on recapture of marked fish, was 49% for coho, 46% for chinook, and 84% for sockeye.
4. Estimated contribution from the flood zone, based on difference between traps in estimated smolt numbers and estimated survival between traps, was 33% of coho smolts, 45% of chinook, and 34% of sockeye.
5. Use of upriver/downriver traps, mark-recapture techniques, and bootstrap statistical methods can be effective in partitioning smolt yield between areas of a watershed, but methods must account for fish mortality between traps.

ACKNOWLEDGEMENTS

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THE SEASONAL USE OF NATURAL AND CONSTRUCTED HABITAT BY JUVENILE COHO SALMON (*ONCORHYNCHUS KISUTCH*) AND PRELIMINARY RESULTS FROM TWO HABITAT IMPROVEMENT PROJECTS ON SMOLT PRODUCTION IN OREGON COASTAL STREAMS

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ABSTRACT

Our research project is directed toward developing models which will enable identification of stream habitat that is limiting the production of salmonid smolts in Oregon coastal streams, and to determine if habitat modification designed to increase limiting habitat actually increases smolt production. We have recently completed a limiting factors model for coho salmon (*Oncorhynchus kisutch*) which we are currently testing. To develop the model, our approach was to determine the average capacity of various types of stream habitat to rear juveniles during the spring, summer, and winter. By summing the capacities of all the available habitat in a stream for a particular season, and factoring in density independent mortality, the potential capacity of the stream to rear fish can be estimated. In determining the average capacities of habitat to rear fish we found that in the spring, coho salmon fry preferred slow velocity backwater habitat. In the summer, juveniles used pools of any type. Overwintering juveniles preferred slow velocity off-channel alcove pools and debris laden beaver ponds. Because the type of habitat preferred by juvenile coho salmon in the winter is not abundant, we believe that, given adequate spawning escapement, the amount of available winter habitat is limiting the production of smolts from many Oregon coastal streams.

To test the model and determine the effect of habitat modification on smolt production, we first determined the capacities of various types of constructed habitat to rear coho salmon during summer and winter. As expected, in the summer we found no difference between natural and constructed pools of any type. In the winter, constructed dam pools were not used as extensively as natural beaver ponds unless complex woody debris was added. With this information, we next began a study on two sets of paired streams designed to determine the effect of increasing the amount of winter habitat on smolt production. For three years on one pair of streams, and four years on another pair, we gathered baseline data on habitat, rearing population size, and smolt production. After this initial phase the habitat was modified in one stream in each pair while the other stream served as a control. Preliminary results showed increases in overwinter survival and size of migrating smolts in both treatment streams compared to the pretreatment streams and to the control streams. For more detail see Rodgers et. al. (In press) in Proceedings of the Coho Salmon Workshop, May 26-28, 1992, Nanaimo, B.C.

USE OF GROWTH DATA TO DETERMINE THE SPATIAL AND TEMPORAL DISTRIBUTION OF FOUR RUNS OF JUVENILE CHINOOK SALMON IN THE SACRAMENTO RIVER, CALIFORNIA

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ABSTRACT

Juvenile chinook salmon *Oncorhynchus tshawytscha* were captured monthly from 1981 to 1991 using beach seines at 13 sampling sites along a 134 mile stretch of the upper Sacramento River. These salmon were separated into one of four runs using a growth model developed from production records of fall-run chinook salmon reared on natural feed in the Tehama-Colusa Fish Facility from 1972 to 1981. Each of these runs exhibited different rearing strategies which could be explained by the availability of food, river flows, and water temperatures in the upper and lower river and San Francisco-San Joaquin Delta. Generally, fall, winter and spring-run chinook salmon move out of the upper river in one or two months after emergence, and are hypothesized to rear in the lower river or delta. Late-fall-run chinook salmon tend to reside four to six months in the upper river before moving out of the system.

INTRODUCTION

The Sacramento River and its tributaries in Northern California support fall, late-fall, winter, and spring runs of chinook salmon (salmon) *Oncorhynchus tshawytscha*. Runs are named according to the time of year adult fish enter San Francisco Bay and begin their migration upriver to spawn (Figure 1). These salmon runs have generally been in decline over the past 25 years (Figure 2). Decline in the threatened[1] winter run has been the most dramatic. Winter run escapement was over 117,000 in 1969, but had deteriorated to 191 in 1991. Fall run has steadily declined in recent years from 140,000 (1988) to 46,000 (1991), and has become increasingly dependent on hatchery production (Cramer 1991). Escapements of late-fall (7,089) and spring (773) run are substantially lower in 1991 than their 1969 highs of 37,000 and 26,000. The outmigration of juvenile salmon in the Sacramento River is possibly the least understood stage of their life histories. Information on spatial and temporal distribution of each of the four runs is critical to management, particularly in regard to operation of water diversions and other facilities. Also, it would enable resource managers to make informed decisions regarding release of hatchery fish that will maximize survival and minimize interference with natural production.

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[1] The winter run was listed as endangered by the California Fish and Game Commission in May 1989 (California Code of Regulations, Title XIV, Section 670.5, Filed 92289) and as threatened by the National Marine Fisheries Service in November 1990 (Federal Register, March 20, 1990, Volume 55, Number 54).

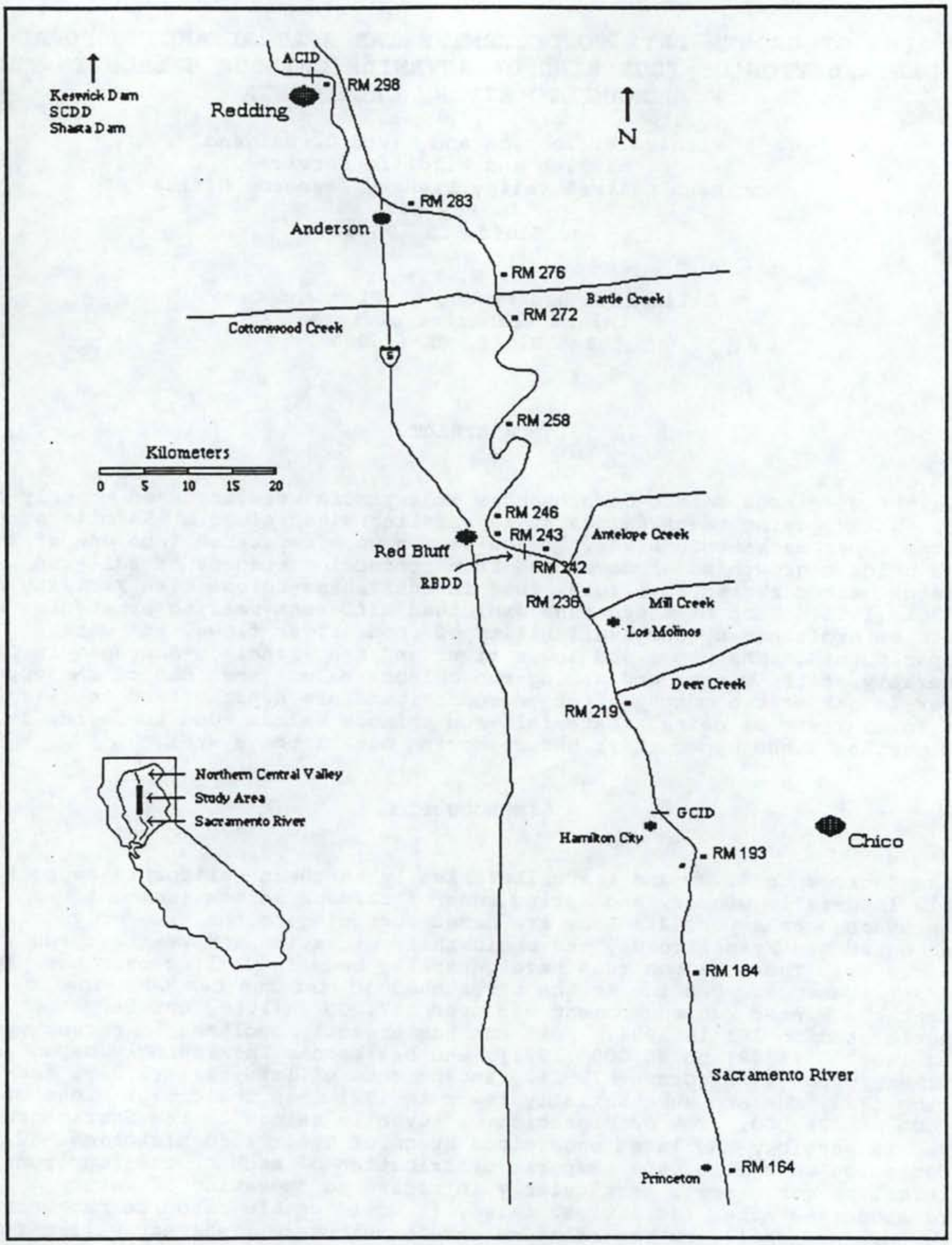

Figure 1. Location of 13 sampling sites on the Sacramento River from river mile 164 to 298, and with respect to Redding, Red Bluff, Chico, Anderson-Cottonwood Irrigation District Dam (ACID), and the Red Bluff Diversion Dam (RBDD), California.

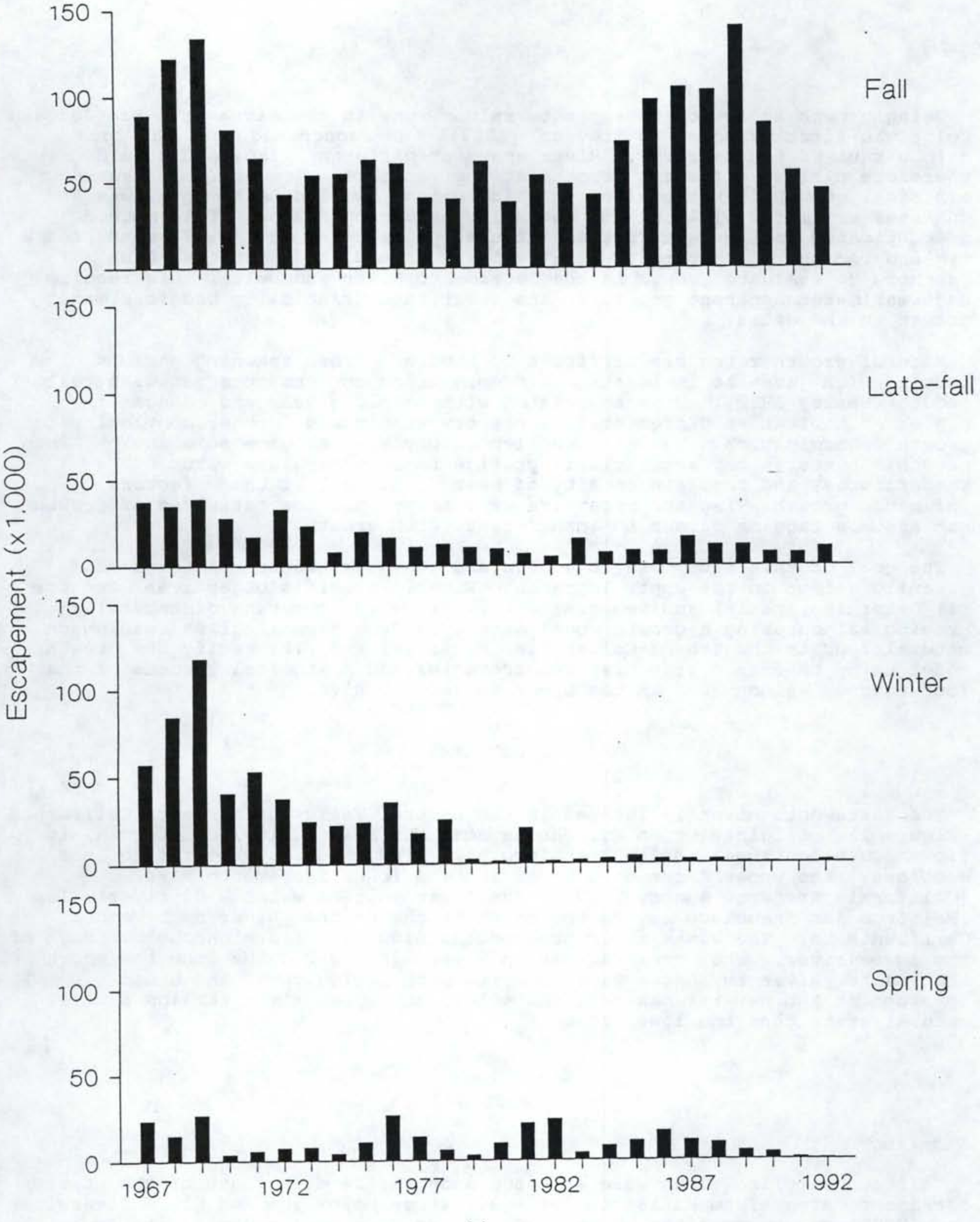

Figure 2. Estimates of escapement for fall, late-fall, winter, and spring runs of chinook salmon past Red Bluff Diversion Dam, 1967-1992. (California Department of Fish and Game, Red Bluff, California).

Using growth as a tool to separate salmon runs in the Sacramento-San Joaquin Delta was first proposed by Stevens[4] (1989). He recognized that the four salmon runs of the Sacramento River spawn at different times and should therefore exhibit different lengths at a given date. However, he assumed empirical growth in hatcheries, in which weight doubled each month, was an unbiased estimator of length in naturally occurring salmon. This method overestimated real growth in these stocks. Kjelson et al. (1982) conducted a tag and recapture experiment using fall-run from Coleman National Fish Hatchery to evaluate growth in the Sacramento River and Delta. His results underestimated apparent growth in the river, and drastically underestimated growth in the delta.

Natural growth rates are difficult to obtain. Broad spawning periods coupled with juvenile immigration and emigration confuse apparent with real growth (Healey 1980). Bias associated with sampling gear and changes in preferred habitat at different life history stages add further complexity to growth determination. Hatchery and laboratory studies give some insight into growth; however, they artificially provide food, manipulate water temperatures, and regulate density of test fish. All of these factors influence growth. Tag and recapture experiments provide estimates of growth but assumes tagging causes no interference with growth.

The goal of this study was to understand the outmigration patterns of juvenile salmon in the upper Sacramento River. Specific objectives were to: (1) Determine spatial and temporal distribution of four runs of naturally growing salmon using a growth model developed from juvenile fish rearing on natural feed in the Tehama-Colusa Fish Facility; and, (2) verify the growth model using theorized life history strategies and historical records of the four chinook salmon runs in the upper Sacramento River.

STUDY AREA

The Sacramento River is located in the Central Valley of Northern California (Figure 1). Originating on Mt. Shasta near Mt. Shasta City, California, it flows south-southwest to San Francisco Bay. The river may be divided into 2 sections: the upper Sacramento River and the lower Sacramento River (California Resource Agency 1989). The lower section extends 80 river miles (RM) from San Francisco Bay to the mouth of the Feather River near Verona, California to. The banks are riprapped and flow is uniform throughout much of the lower river. The upper Sacramento River extends 232 RMs from the mouth of the Feather River to Shasta Dam. Despite much agricultural and urban development and several man-made obstacles, the upper river remains a more natural state than the lower river.

METHODS

Sampling

Thirteen sampling sites were selected along a 134 mile reach of the upper Sacramento River from RM 164 to 298 - six sites below the Red Bluff Diversion Dam (RBDD) and seven sites above RBDD. Sites were selected empirically on the

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[2] Memorandum, June 19, 1989, To: H. K. Chadwick, Program Manager Bay-Delta Project, From: D. E. Stevens, California Department of Fish and Game, Subject: When do winter-run chinook salmon migrate through the Sacramento-San Joaquin Delta? California Department of Fish and Game, Red Bluff, California.

basis of current, substrate composition, accessibility, and relative separation from one another. Sites were either gravel bars or boat ramps. Gravel bars were shallow with high water velocities; while boat ramps were deep with low water velocities. Sites below RBDD were gravel bars (except RM 242); while boat ramps predominated above RBDD (except RMs 246 and 272). Sampling was conducted approximately once a month from 1981 to 1991. An 1/8 in mesh beach seine (4 x 75 ft) was used to capture juvenile fish. Fish were immediately removed from the bag of the seine and held in fresh water. Tricaine methanesulfonate (MS 222) was added to anesthetize the fish if salmon were present. Fork lengths (mm) were measured from 50 randomly selected salmon; additional salmon (if present) were counted. A second haul was conducted at gravel bars if less than 50 juvenile salmon were captured on the first. Second hauls were upriver of the first to minimize the chance of recapturing fish. All fish were released back into the river.

Run Determination

An estimate of apparent growth rate to establish run was made from 1972 through 1981 production records of fall-run chinook salmon < 90 mm fork length reared naturally (without artificial feed) at the Tehama-Colusa Fish Facility (Fisher 1992). A growth curve was fitted to fork length at age (days) using a linear function:

$$Log_e(Fork\ Length\ [mm]) = a + b(age)$$

where a and b are constants derived by regressing the logarithm (base e) of fork length and age. A table to predict run from length and capture date was developed and extrapolated to account for fish ≥ 90 to 270 mm (Table 1). The same model was used for late-fall, winter, and spring runs with adjustments made for spawning and incubation periods. Because minimum and maximum lengths of successive runs are the same, it was subjectively determined that these ambiguous fish be placed in the later spawning run.

RESULTS

Run Composition

Thirteen brood years of salmon (1980 - 1992) were represented over 10 sampling years. Fall run comprised the largest part of total catch (74%); followed by winter run (13%), late-fall run (7%), and spring run (6%).

Spatial and Temporal Distribution

Fall Run. Although present year-round, fall run were captured primarily between December and May (Table 2; Figure 3). They exhibited a fairly evenly distribution. By June, most had left the study area.

Late-Fall Run. Late-fall run were also captured year-round but predominately between April and January (Table 3; Figure 4). They tended to remain in the upper reach between April and September. None were captured below RM 219 between April and September but slowly spread to the lower reach (below RBDD) after October. There appears to be a dramatic movement downstream of late-fall juveniles in December, and were largely gone from the study area by February.

Winter Run. Winter run were captured between July and March (Table 4; Figure 5). They largely confined themselves to the upper reach of the study area from July through September and slowly spread to the lower reach from

Table 1. Growth table developed from fall-run chinook salmon reared naturally in the Tehama-Colusa Fish Facility (Fisher 1992). The table was developed using the linear function Log_e(fork length (mm))=3.516+0.007x*AGE*(days) to estimate the rate of apparent growth.

Emergence	Fall run Oct. 11-Apr. 2			Late-fall run Jan. 1-Jun. 27			Winter run Apr. 16-Oct. 18			Spring run Aug. 16-Dec. 9		
	Early	Peak	Late	Early	Peak	Late	Early	Peak	Late	Early	Peak	Late
Jan. mid-month	41		200	200	150	110	110	80	54	54	49	41
	45		219	219	166	122	122	89	59	59	54	45
Feb.	49	34	244	244	181	136	136	99	65	65	59	49
	54	37	270	270	200	150	150	110	73	73	65	54
Mar.	59	41			219	166	166	122	80	80	73	59
	65	45			244	181	181	136	89	89	80	65
Apr.	73	49	34	34	270	200	200	150	99	99	89	73
	80	54	37	37		219	219	166	110	110	99	80
May	89	59	41	41		244	244	181	122	122	110	89
	99	65	45	45	34	270	270	200	136	136	122	99
Jun.	110	73	49	49	37			219	150	150	136	110
	122	80	54	54	41			244	166	166	150	122
Jul.	136	89	59	59	45	34	34	270	181	181	166	136
	150	99	65	65	49	37	37		200	200	181	150
Aug.	166	110	73	73	54	41	41		219	219	200	166
	181	122	80	80	59	45	45	34	244	244	219	181
Sep.	200	136	89	89	65	49	49	37	270	270	244	200
	219	150	99	99	73	54	54	41			270	219
Oct.	244	166	110	110	80	59	59	45				244
	270	181	122	122	89	65	65	49	34	34		270
Nov.		200	136	136	99	73	73	54	37	37	34	
		219	150	150	110	80	80	59	41	41	37	
Dec.	34	244	166	166	122	89	89	65	45	45	41	34
	37	270	181	181	136	99	99	73	49	49	45	37

Table 2. Mean number of fall-run chinook salmon captured per month at the 13 sites between January 1981 and January 1991 (*N*=60,728). Means were rounded to the nearest whole number. Blanks indicate zero catches and dashes no sampling.

	River mile												
Month	298	283	276	272	258	246	243	242	236	219	193	184	164
December	17	<1	2	12	4	48	1	15	18	1	1	1	1
January	113	238	68	174	113	74	30	28	46	64	101	49	39
February	128	224	158	183	40	103	103	51	40	64	103	38	44
March	118	135	596	151	30	39	30	38	67	26	129	63	116
April	75	30	30	152	43	—	9	10	90	31	65	56	63
May	7	1	10	123	51	124	7	18	86	35	50	68	49
June			11	24	<1	—	<1	8	38	3	6	5	3
July		<1	6	9	<1	—		1	3	1	2	<1	1
August			<1	9		—		<1		<1	<1	1	<1
September		<1	<1	4		—		<1	<1	<1	<1	<1	
October				1		—	<1		<1		<1	<1	<1
November				<1		—					<1		<1

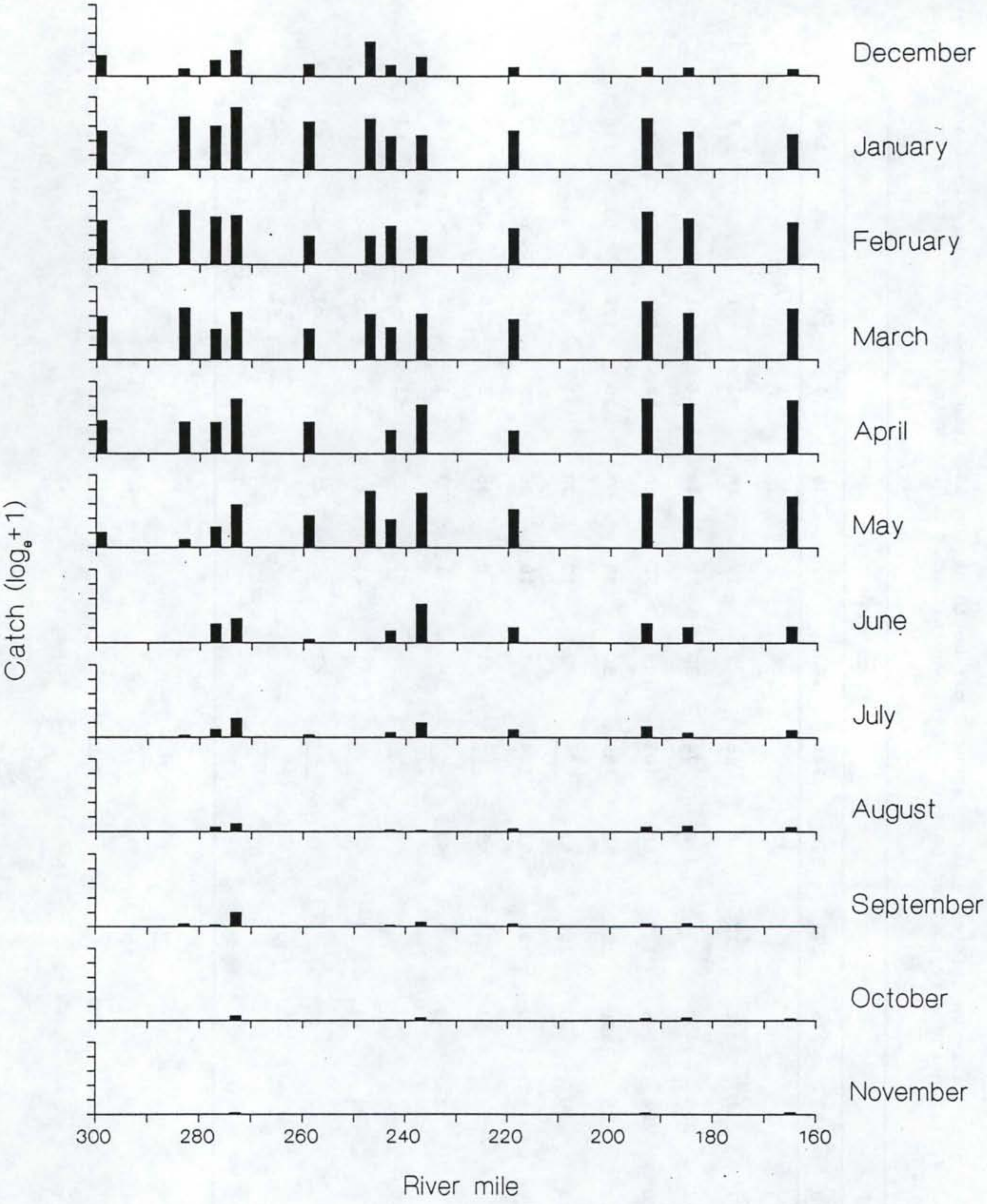

Figure 3. Spacial and temporal distribution of fall-run chinook salmon captured during beach seine sampling from 1981 to 1991. Because of the large range total catch has been rescaled using the transformation log$_e$(catch+1), so that values range from 1 to 5.

Table 3. Mean numbers of late-fall-run chinook salmon captured per month at the 13 sites between January 1981 and January 1991 (*N*=6,284). Means were rounded to the nearest whole number. Blanks indicate zero catches and dashes no sampling.

	River mile												
Month	298	283	276	272	258	246	243	242	236	219	193	184	164
April	16	17	7	7	5	—	<1	2	<1	<1		<1	
May	49	39	16	72	16	5	6	13	7	1		<1	
June	49	6	39	8		—	<1	2	9	<1		<1	<1
July	7	1	5	6	1	—		<1	<1				
August	4	1	3	42	1	—	<1	<1	1	<1	<1	<1	<1
September	26	4	2	46	11	—	<1	<1	4		<1	<1	<1
October	9	<1	2	18	1	—	<1	1	3	<1	3	<1	1
November	<1		<1	12	2	—		<1	<1	<1	6	<1	1
December	1			9	1	10	5	<1	4	1	8	1	3
January			<1	4	<1			1	<1	<1	2	<1	1
February													
March													

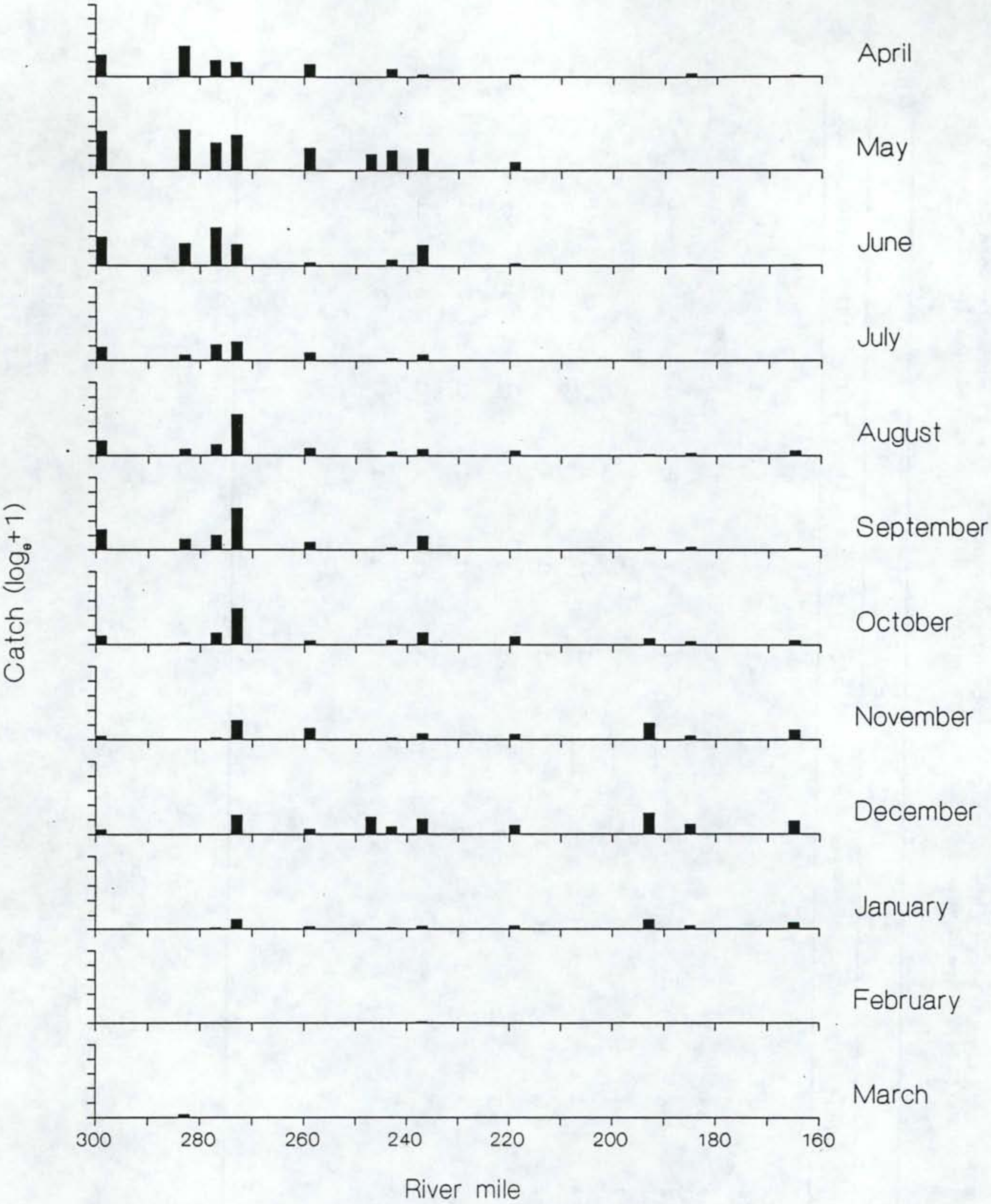

Figure 4. Spacial and temporal distribution of late-fall-run chinook salmon captured during beach seine sampling from 1981 to 1991. Because of the large range, total catch has been rescaled using the transformation log$_e$(catch+1), so that values range from 1 to 5.

Table 4. Mean numbers of winter-run chinook salmon captured per month at the 13 sites between January 1981 and January 1991 (*N*=10,778). Means were rounded to the nearest whole number. Blanks indicate zero catches and dashes no sampling.

	River mile												
Month	298	283	276	272	258	246	243	242	236	219	193	184	164
July	<1	<1	<1			—			<1				
August	23	<1	4	4	9	—	<1	1	3	<1			
September	264	3	43	193	67	—	7	9	57	<1		1	<1
October	76	1	28	59	29	—	9	8	11	3	6	<1	1
November	8	1	8	132	10	—	2	<1	2	1	2	<1	<1
December	4		1	15	1	4	6	1	11	<1	3	<1	4
January	1		<1	25		<1	1	24	2	2	16	4	9
February	<1	<1	<1	<1		4	<1	<1	1	3	11	<1	3
March	<1	<1	<1	<1		<1	<1	<1	1	<1	<1	2	3
April				<1	<1	—		<1					
May													
June						—							

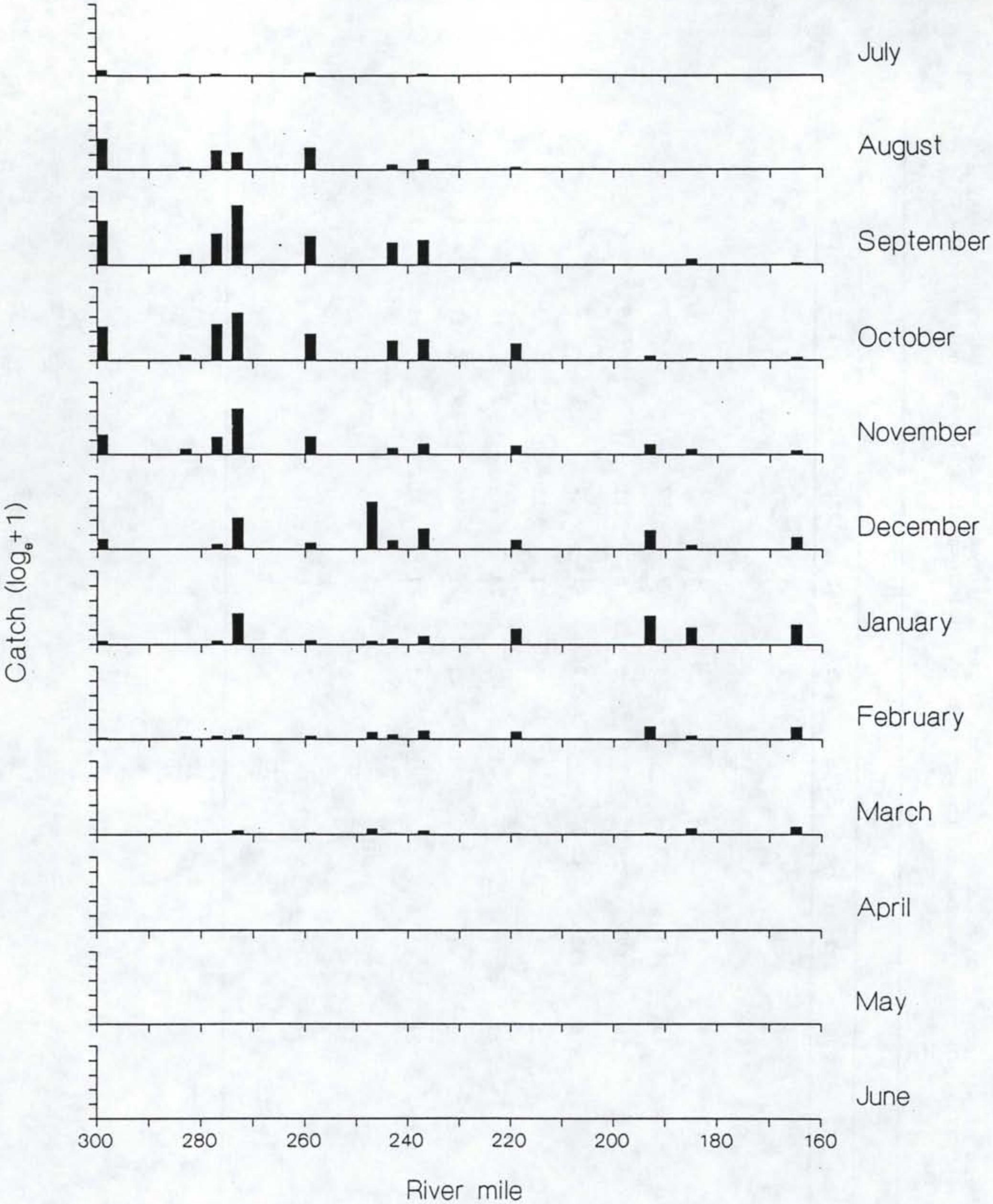

Figure 5. Spacial and temporal distribution of winter-run chinook salmon captured during beach seine sampling from 1981 to 1991. Because of the large range, total catch has been rescaled using the transformation $\log_e(\text{catch}+1)$, so that values range from 1 to 5.

October through March. By April they had completely left the study area. No winter run were captured in April through June.

Spring Run. Spring run were captured between October and May (Table 5; Figure 6). They were confined to the upper reach until December when they began to distribute themselves downstream. They became less abundant from January through May, and had completely left the study area by July. No spring run were captured in the study area in July through September.

DISCUSSION

Fall run spawn throughout the study area. Recently emerged (≤ 40 mm) fall run at sites above and below RBDD between December and March support this conclusion. Numbers of fall run captured above and below RBDD between December and March suggests 28% of the parental fish spawned in the lower reach of the study area. This estimate is high since a large number of small fish outmigrate almost immediately upon emergence (Frank Fisher, California Department of Fish and Game (CDFG), unpublished data, 1991). However, it is similar to the 30% estimated to have spawned in the lower reach in 1979 (Vogel et al. 1988).

Vogel et al. (1988) concluded that most fall run outmigrate past RBDD during winter in normal precipitaion years and spring in dry years. Results of our study support this conclusion. During the dry years 1989 to 1991, outmigration past RBDD did not occur until spring (April – June). In fact, in those years fall run were captured at RM 298 and 283 until May and above RBDD until October.

Late-fall run spawn mainly in the upper reach of the study area. Most recently emerged late-fall run were captured in the upper reach between April and June. Conversely, very few small late-falls were captured in the lower reach or even at lower sites of the upper reach (RMs 243, 246, and 258). It is further supported results from a 1985 redd distribution study in which 25% of the late-fall run spawned in a 3 ½ mile reach between RM 298 and Keswick Dam (RM 302; Hallock 1987).

Fisher (CDFG, unpublished data, 1991) identified two distinct outmigrant groups of late-fall run from capture data collected at RBDD between 1978 and 1989. One group outmigrated as fry in April and May and the second as smolts in October and November. A similar trend was observed in our study – one group in May and a second in October. Outmigration of two distinct groups may be related to precipitation. It is hypothesized that early outmigrants move with spring rains while late outmigrants move with fall rains. Precipitation in April and May in the years 1989 to 1991 was less than normal; consequently, a relatively large number of late-fall run reared in the upper river over the summer and outmigrated in the fall.

Large number of winter run begin to outmigrate almost immediately upon emergence. A substantial number of newly emerged winter run were captured in the lower reach of the study area in August and September. Since most winter run spawn in the upper reach of the study area (above RM 298; Vogel et al. 1988), these small fish are believed to have emerged above RBDD. This is supported by capture data collected at RBDD between 1978 and 1989 which demonstrates most winter run pass RBDD between August and October (Frank Fisher, CDFG, unpublished data, 1991). Early outmigrants are believed to rear somewhere in the system between RBDD and the Sacramento-San Joaquin Delta since water temperatures in the Delta during the summer are not suitable for juvenile salmon.

Table 5. Mean numbers of spring-run chinook salmon captured per month at the 13 sites between January 1981 and January 1991 (*N*=4,768). Means were rounded to the nearest whole number. Blanks indicate zero catches and dashes no sampling.

	River mile												
Month	298	283	276	272	258	246	243	242	236	219	193	184	164
October	1	<1	1		<1	—	<1		<1				
November	3	5	1	7	<1	—	2	<1					<1
December	14	<1	15	37	6	53	2	19	40	1	4	2	2
January	15	8	4	45	6	29	1	2	1	2	13	8	9
February	1		1	10	<1	2	<1	<1	1	3	10	3	13
March		<1	<1	2	<1	4	<1	2	2	1	11	3	19
April			<1	1	1	—	<1	<1	2	<1	6	4	10
May			<1	<1		50	<1	<1	<1	1		1	2
June						—							<1
July						—							
August						—							
September						—							

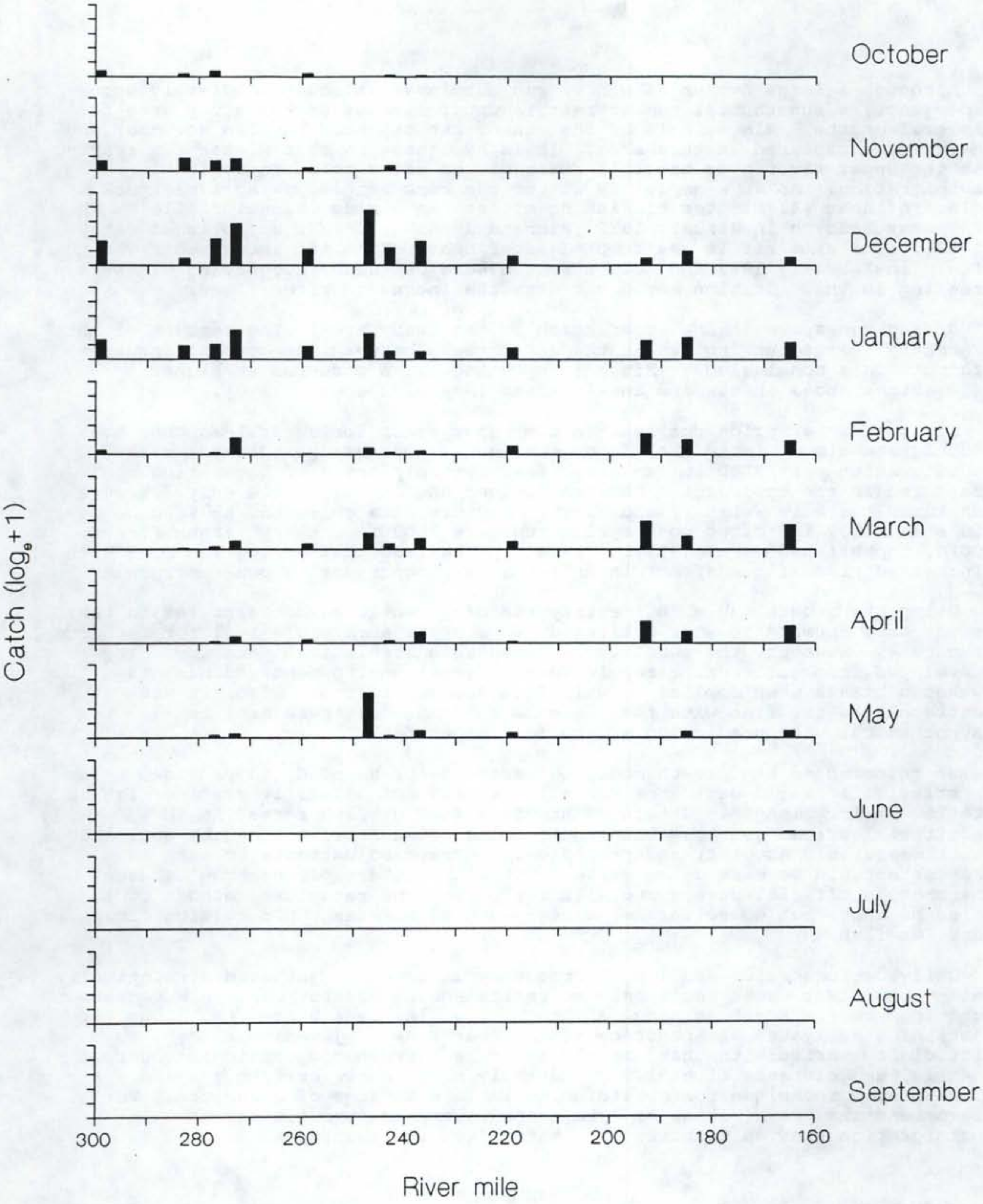

Figure 6. Spacial and temporal distribution of spring-run chinook salmon captured during beach seine sampling from 1981 to 1991. Because of the large range, total catch has been rescaled using the transformation $\log_e$(catch+1), so that values range from 1 to 5.

Although a large number of winter run outmigrate almost immediately upon emergence, a substantial number rear in upper reaches of the study area for several months. Almost half of the winter run captured between November and March were captured in this area. It is hypothesized that winter run rearing in the upper river over the fall await winter rains to begin their outmigration. As an example, 49 winter run were sampled using a backpack electrofisher (41 minutes of fishing effort) in a side channel riffle at RM 289 near Redding in January 1992 (Richard Johnson, USFWS, unpublished data, 1992). The same riffle was sampled after heavy rains and increased river flows in February 1992 and no winter run were captured, suggesting winter run rearing in this location moved out with the increased river flows.

Spring run spawn in the upper reach of the study area. The capture of most recently emerged spring run in the upper reach between October and January support this conclusion. Historic spawning areas occurred at higher elevations above Shasta and Keswick dams (Reynolds et al. 1990).

Although some spring run rear in the upper reach for several months, most outmigrate almost immediately upon emergence. Our study indicates peak outmigration past RBDD in January. Seventy-one percent of the spring run were captured in the upper reach between October and January while only 23% were captured there in February and March. Capture data collected at RBDD between 1978 and 1989 indicated most spring run pass RBDD in January (Frank Fisher, CDFG, unpublished data, 1991). It is hypothesized that spring run move with increased river flows after winter rains that occur during peak emergence.

Using the growth table to identify run of juvenile salmon captured in the study area appears to work well, and as such, is a valuable tool for resource managers. However, the model lacks measurable statistical veracity. It was developed from fall run reared in an artificial environment and harbors unknown biases when applied to wild fish and other runs. Also, it was extrapolated for fish with fork lengths ≥ 90 mm and; therefore, is hypothetical when predicting growth for larger fish.

We recommended the growth model be verified (or amended) to provide statistically valid estimates for all four runs of naturally rearing fish up to 250 mm fork length. Growth of hatchery fish or fish reared in an artificial stream (such as the Tehama-Colusa Fish Facility) can be estimated with measurable accuracy and precision. Perhaps adjustments to such an estimate could be made using scale or otolith pattern differences between hatchery/artificial stream and wild fish. Tag and recapture methods could also be used, but we reiterate, biases in growth rates could be significant in smaller fish.

Daily capture rates and length frequency information gathered strategically along the river would facilitate an understanding of the relative movements of the four runs. Beach seining, although valuable, is a biased technique for obtaining estimates of abundance of the four runs of juvenile salmon. Efficiency varies with changes in current, substrate composition, amount of debris, and distance of haul. Permanately based screw or fyke traps are a less biased technique for obtaining reliable estimates of abundance. We recommend the use of screw or fyke traps to expand juvenile salmon outmigration studies to better estimates juvenile abundance.

SUMMARY

1. A model was produced that described the spatial and temporal distribution of four runs of chinook salmon in the Sacramento River using a prototype growth table developed from juvenile fish rearing on natural feed in the Tehama-Colusa Fish Facility.

2. The model provided reasonable results based on theorized life history strategies and historical records.

3. We recommend the growth model be rigoruously verified (or amended) to provide statistically valid estimates of length by date by run, for all four runs of naturally rearing fish up to 250 mm fork length.

4. Daily capture rates and length frequency information gathered from screw traps strategically located along the river would facilitate a more complete understanding of the relative movements of the four runs.

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SMOLT TO ADULT STAGE: RIVER AND OCEAN ISSUES

SESSION LEADER: NANCY DAVIS,
UNIVERSITY OF WASHINGTON FISHERIES RESEARCH INSTITUTE

HIGH SEAS DISTRIBUTION OF COHO AND CHINOOK SALMON

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ABSTRACT

Chinook and coho salmon are concentrated in the western and central North Pacific Ocean and along the Gulf of Alaska coast. Coho from Asia and western Alaska are found further offshore than more southern stocks. Scale pattern analysis shows a mixture of Asian and North American coho and chinook in the area of the former high seas driftnet fisheries. Alaskan coho appear to make up the majority of the driftnet coho catches, but these estimates differ somewhat from tag return data. Alaskan chinook dominate in Bering Sea catches of chinook, but Asian fish are predominant in catches south of the Aleutians. Parasite analysis of chinook continental origins supports scale pattern results. From information on operations of squid driftnet fisheries and the distribution of west coast coho and chinook, we can infer that these fisheries probably had only minor effects on west coast coho and chinook stocks. New emphases in high seas salmon research will be carrying capacity, growth, and stock interactions.

INTRODUCTION

Research related to high seas salmon and squid driftnet fisheries has yielded data on the ecology and biology of salmonids in the North Pacific Ocean since the mid-1950s. We present here a broad overview of the distribution of coho and chinook, and information on potential future directions for high seas salmon research at the Fisheries Research Institute (FRI) of the University of Washington.

Brief History Of High Seas Driftnet Fisheries

Japanese fisheries regenerated and expanded after World War II. The International North Pacific Fisheries Commission (INPFC) was formed by the United States, Canada, and Japan to regulate Japanese high seas fisheries, primarily those for salmon (Jackson and Royce 1986). There were two major salmon fisheries of concern: a mothership fishery operating north of 46°N through the Aleutians and into the Bering Sea, and a landbased fishery operating south of 46°N (Fig. 1). The 1950s and 1960s were a period of expansion for the fisheries, which reached their peak in the late 1960s and early 1970s. The boundaries of the fisheries were severely cut back in 1977 and 1978 as the USSR and the United States adopted 200-mile zones. The following era, from 1978 to 1991, was a period of steady decline, dictated in large part by declining quotas set by Japan-USSR bilateral negotiations. Both fisheries ceased to operate after the 1991 season, due in part to USSR demands for zero quotas in 1992 and in part to a U.N. resolution calling for a voluntary cessation of all driftnet fisheries by the end of 1992.

The high seas squid driftnet fishery was started in 1978 by Japan, which already had coastal squid jigging fisheries (Yatsu 1990). Korea and Taiwan quickly followed in 1979 and 1980, respectively. The decline of the Japanese salmon driftnet fishery was partially responsible for the expansion of the squid driftnet fishery, as salmon driftnetters converted to squid boats. The

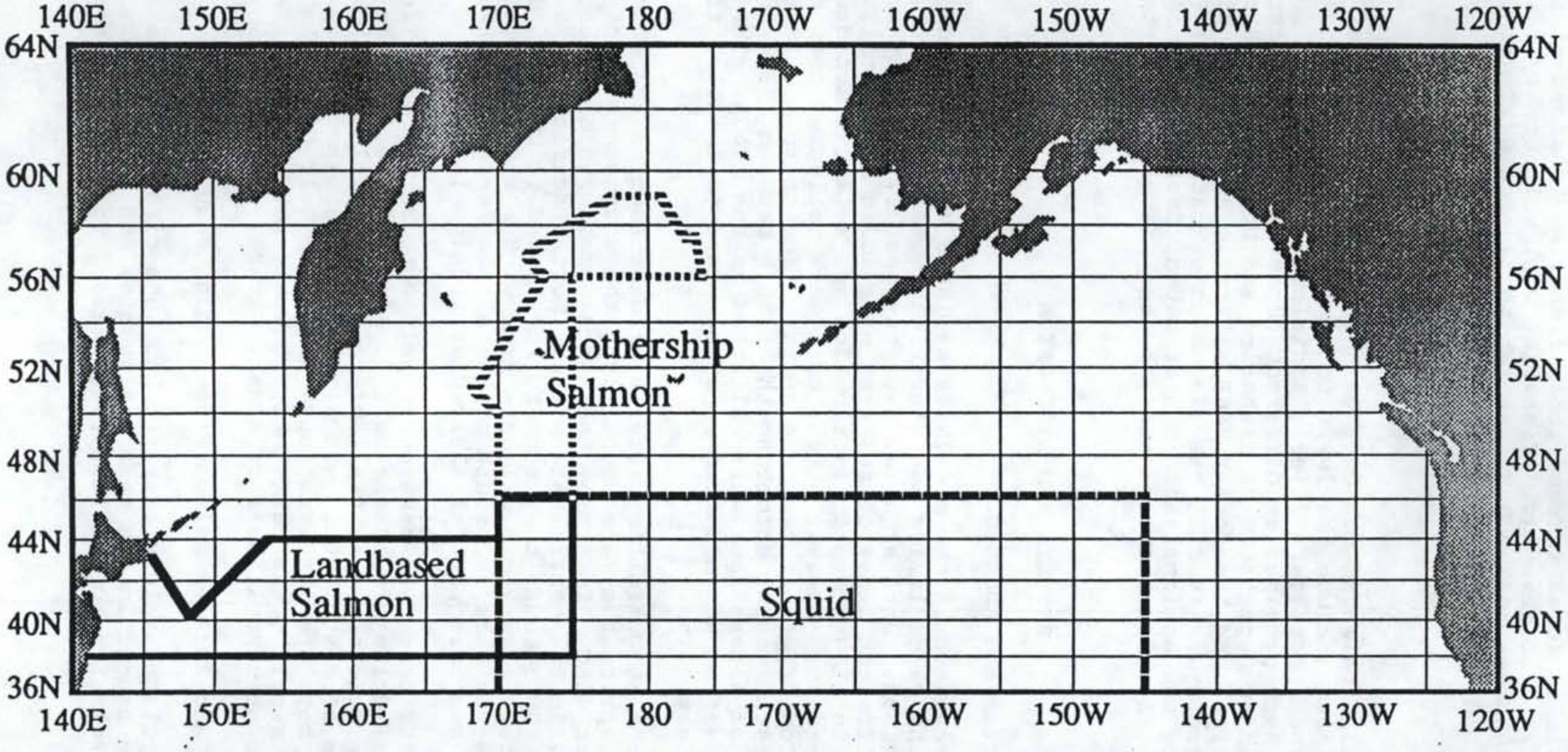

Figure 1. Fishing boundaries for the Japanese mothership and landbased salmon driftnet fisheries during the period 1978-1987, and maximum (September) boundaries for the Japanese squid driftnet fishery. The northern boundary of the Japanese squid fishery varies by month between 40°N and 46°N. Boundaries for the squid fisheries of the Republic of Korea and Taiwan are similar to the Japanese fishery, but allow fishing farther to the west.

Japanese government instituted fishing boundary and season restrictions on their squid driftnet fishery in 1981 (Fig. 1; Yatsu 1990). The northern boundary shifts by month; the purpose is to prevent the bycatch of salmonids by keeping squid vessels south of the 15°C isotherm. U.S. negotiations with Taiwan and the Republic of Korea induced those fisheries to adopt similar boundary restrictions. All three squid fisheries will voluntarily cease by the end of 1992, in response to the U.N. resolution banning driftnet fishing.

METHODS

Distribution and relative abundance data were taken from databases maintained at FRI. These include the INPFC tag recovery database and FRI high seas research catch and effort data files. Additional data on high seas salmon research have been provided on different occasions by the Fisheries Agency of Japan, Canadian Department of Fisheries and Oceans, U.S. National Marine Fisheries Service, and the former USSR's Pacific Research Institute of Fisheries and Oceanography. Material has also been extracted from INPFC documents submitted by the National Sections of Japan, Canada, and the United States. Methods for the scale pattern analyses are described in cited references.

RESULTS AND DISCUSSION

Patterns Of Distribution And Abundance

Some of the early research in the mid-1950s by INPFC member nations was simply to ascertain the range limits and migration routes of the different species of Pacific salmon. Data acquired over the past 37 years by numerous vessels conducting different lines of research have added more detail, and the data can be combined to show oceanic distribution and relative abundance of coho and chinook by season. However, these data were collected over many years, and distribution of effort has been spotty. For example, there has been very little research vessel effort south of 50°N between 175°W and 155°W, and most of the Gulf of Alaska data were collected in the 1960s. The great majority of research effort has occurred in summer, and information from other seasons is much less extensive.

Both coho and chinook show similar patterns of distribution and abundance (Figs. 2 and 3). In spring they are found along the southern edge of their ranges, south of about 45°N. In summer there are concentrations along the Gulf of Alaska coast, in the central and western North Pacific, and along the Kamchatka Peninsula. By autumn, they seem to have moved to Gulf of Alaska and Kamchatka coastal waters. Very minimal data from winter show a few fish in the central eastern North Pacific, south of the Alaska Peninsula; there is no information from the southern part of their distribution in winter.

From 1964 to 1968 FRI used fine-mesh purse seines to sample juvenile (saltwater age .0) salmonids around the Gulf of Alaska. The sampling was conducted from May to October, nearly all of it in relatively nearshore waters. The pattern of distribution was similar to that shown by sampling adults: juveniles of both species were found along the Gulf of Alaska coast. However, because the sampling was nearshore, it would not have detected juveniles in offshore or central Gulf of Alaska waters.

Distribution Of Coho And Chinook Salmon From Tagging Studies

Much of the INPFC-related salmon research has been directed at the stock composition of salmon caught in the salmon fisheries and the delineation of ranges of stocks in relation to the fisheries areas. To this end, extensive

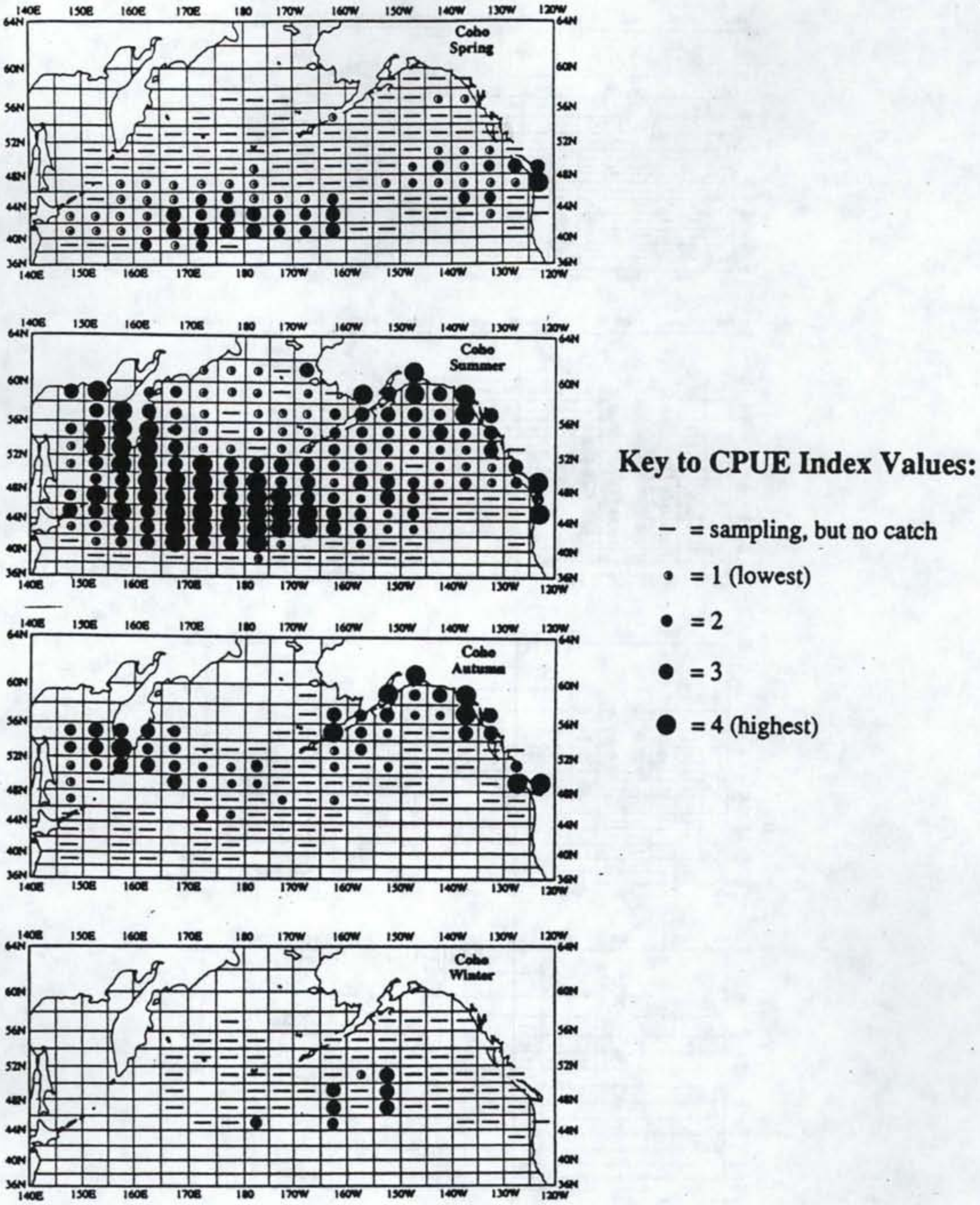

Figure 2. Distribution and relative abundance of coho salmon in the North Pacific Ocean and Bering Sea. Catch per unit effort (CPUE) data were combined from gillnet, longline, and purse seine gear used by Japanese, Canadian, Russian, and U.S. research vessels from 1955 to 1991. CPUE values were calculated for each gear type by season and location, and ranked within gear type. CPUE values were converted to a quartile position for that gear type, and quartile values for the different gear types were combined to yield an overall relative abundance for that stratum. Seasons were defined as March-May (spring), June-August (summer), September-November (autumn), and December-February (winter).

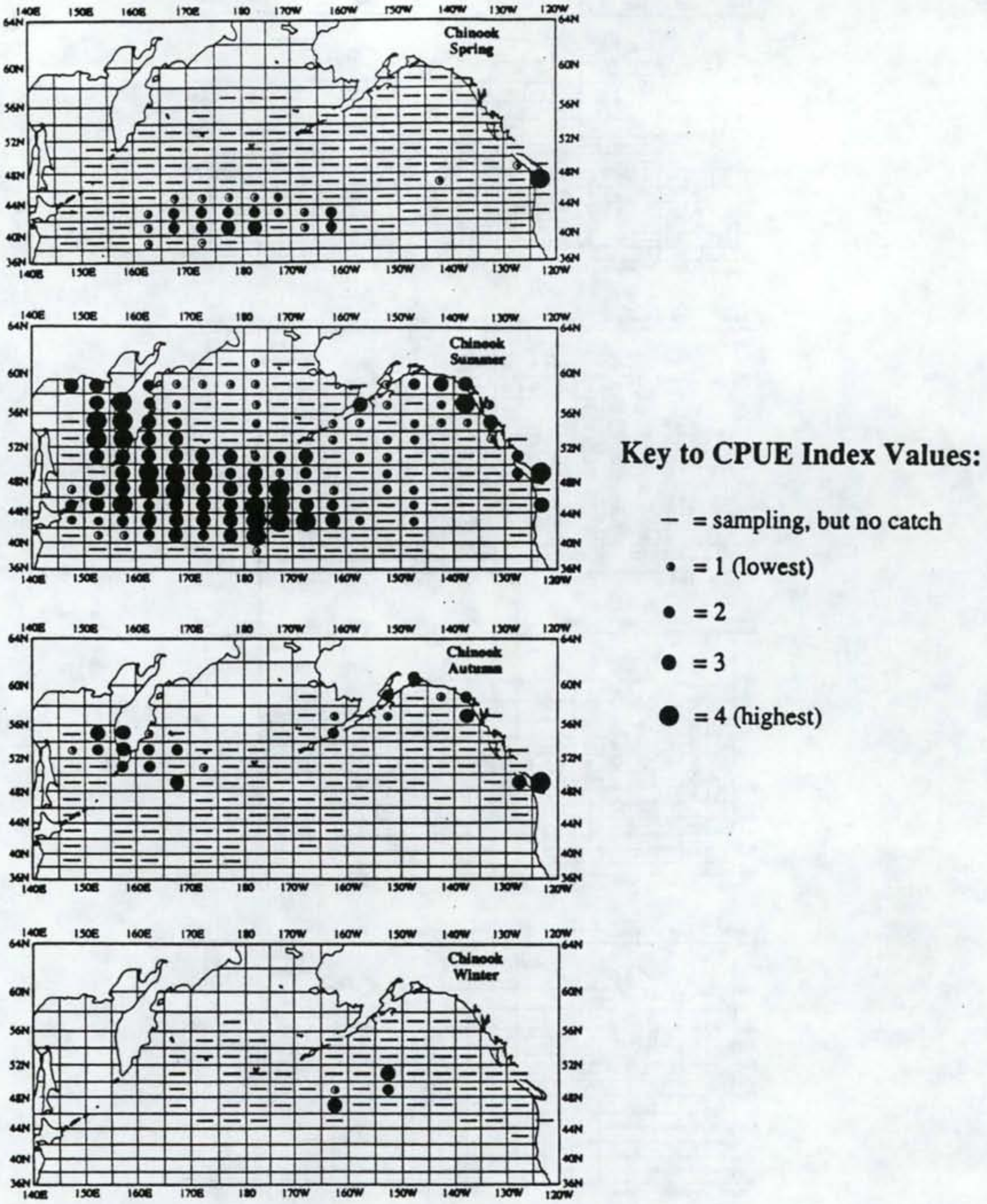

Figure 3. Distribution and relative abundance of chinook salmon in the North Pacific Ocean and Bering Sea. Catch per unit effort (CPUE) data were combined from gillnet, longline, and purse seine gear used by Japanese, Canadian, Russian, and U.S. research vessels from 1955 to 1991. CPUE values were calculated for each gear type by season and location, and ranked within gear type. CPUE values were converted to a quartile position for that gear type, and quartile values for the different gear types were combined to yield an overall relative abundance for that stratum. Seasons were defined as March-May (spring), June-August (summer), September-November (autumn), and December-February (winter).

tagging programs have been implemented. Tags have been applied to fish on the high seas for recovery inshore since 1955, and since 1981 the catches of high seas research vessels have been examined for salmon and steelhead with missing adipose fins to recover coded-wire tags (CWTs). In 1985 the search for CWTs was expanded to a sampling of the landings of the landbased salmon fishery.

Asian coho - Recoveries of high seas tags indicate fish from the northern coast of the Sea of Okhotsk and from both the east and west coasts of the Kamchatka Peninsula are found in a wide swath stretching southeast from Kamchatka, south of the Komandorskie and Aleutian Islands (Fig. 4). There is very little coho production, and very few tag recoveries, from other Asian areas.

North American coho - There are few recoveries from western Alaska, perhaps because runs are too late for fisheries and processors. Recoveries have come from fish tagged from the west central North Pacific to the Gulf of Alaska (175°E to 140°W), with a gap between 170°W and 155°W (Fig. 4). This gap coincides with an area of little tagging effort and may be related to that. From south central Alaska to California, there are progressively fewer recoveries from offshore tag releases and more recoveries from releases in coastal waters. Virtually all are from releases east of 160°W. However, in 1991 three coded-wire tagged coho from Oregon and Washington were recovered between 159°W and 173°E by observers on Japanese squid vessels (McKinnell et al. 1991).

There are interesting differences in the offshore migrations of northern (Asian and western Alaska) and southern coho stocks. Northern stocks are found farther offshore (averaging four times as far from recovery sites), compared to a more coastal distribution of southern stocks (Table 1). Travel rates also reflect this difference. Although days at liberty after tagging are about the same for both groups (around 50 days), Asian and western Alaska fish travel about four times faster, over 40 km/day compared to about 10 km/day for coho from southeastern Alaska and southward (Table 1). South central Alaskan stocks seem to be intermediate between the two extremes.

There are also differences between ocean areas in recovery rates of coho tagged on the high seas. West of 175°W and north of 46°N, the recovery rate is about 2.9%, compared to about 0.8% south of 46°N. East of 160°W (Gulf of Alaska and west coast waters), the rate is a phenomenal 26%, but many of those releases were coastal and close to recovery areas; offshore in the Gulf of Alaska, rates are in the 2-10% range, which is still higher than in the western Pacific. These recovery rates probably reflect differences in exploitation and tag reporting rates among inshore fisheries areas and may also be a function of mortality (tagging and natural) as related to the time and distance that the fish travel.

Asian chinook - There are only two Asian recoveries of chinook tagged on the high seas, one tagged off Hokkaido and one south of the central Aleutians (Fig. 5).

North American chinook - All recoveries of chinook from western Alaska have come from fish tagged in the Bering Sea or just south of the Aleutians (Fig. 5). Chinook tag recoveries from south central Alaska to California are primarily CWTs recovered from the bycatch of trawl fisheries along the coastal rim. There are also a few CWT recoveries of south central and southeastern Alaskan, British Columbian, and Oregon fish from trawl fisheries in the southeastern Bering Sea. One southeastern Alaskan and one Columbia River chinook have been recovered from high seas releases south of Aleutians.

Distribution From Stock Identification Studies

Tag recoveries have provided the outlines of oceanic distributions of salmon

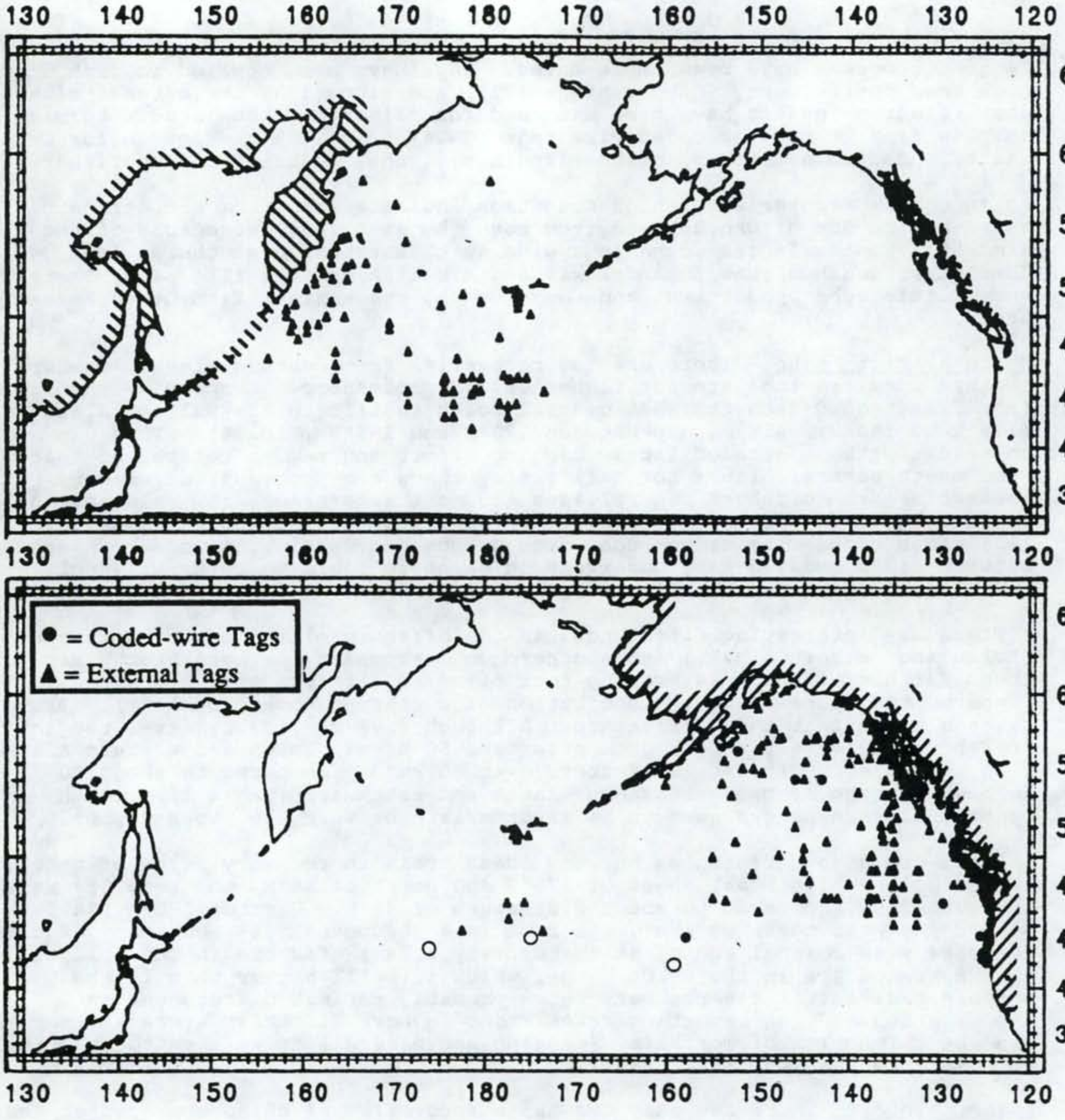

Figure 4. Release locations of inshore recoveries of tags applied to coho on the high seas (▲), and high seas locations of recovered of coded-wire tagged coho salmon (●), 1956-1989, from Asia (top) and North America (bottom). Adapted from Myers et al. 1990. Three high seas recoveries of coho salmon by observers in the Japanese squid fishery in 1991 have been added (○).

Table 1. Average days at liberty and estimated average distances and rate of return for coho tagged on the high seas and at liberty after tagging for at least 10 days. Distances were calculated as minimal straight line distances.

Region of Recovery	N	Mean days at liberty	Mean distance (km)	Speed (km/day)
Northern Okhotsk Sea coast	17	66	2,551	41.1
Western Kamchatka	26	54	1,915	37.0
Eastern Kamchatka	51	44	1,934	47.7
Western Alaska	21	50	2,062	47.3
South central Alaska	34	47	1,057	28.4
Southeastern Alaska	135	43	472	12.2
British Columbia	267	46	374	10.8
Washington/Oregon/California	153	54	301	7.0
Totals and overall averages	704	48	687	16.6

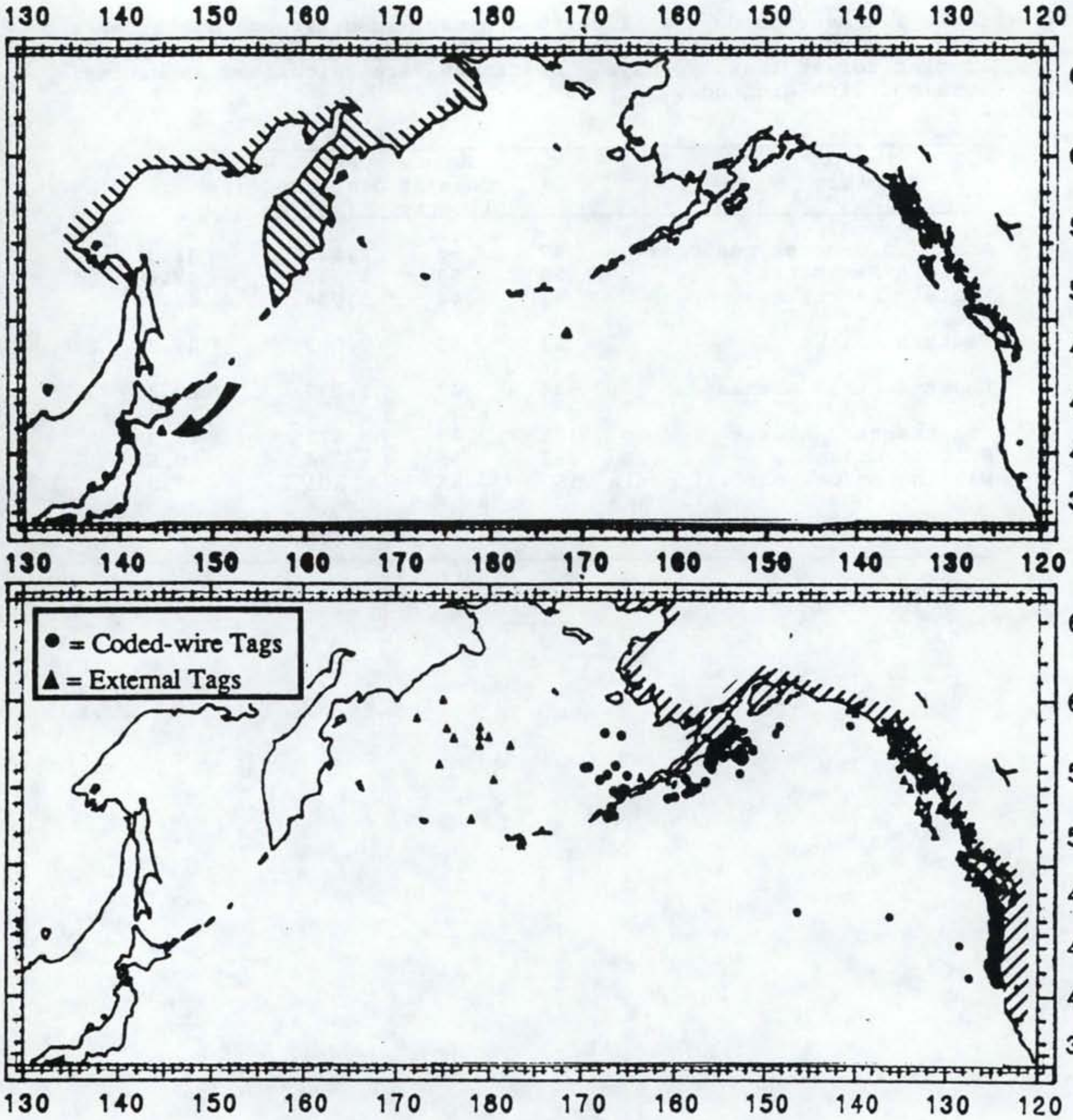

Figure 5. Release locations of inshore recoveries of tags applied to chinook on the high seas (▲), and high seas locations of recoveried of coded-wire tagged chinook salmon (●), 1956-1989, from Asia (top) and North America (bottom). Adapted from Myers et al. 1990.

from some stock regions, but there are not enough recoveries to make quantitative estimates of stock composition of salmon caught in driftnet fisheries.

Scale pattern analysis - Given the nature of sampling opportunities from the fisheries and the accessibility of known-origin reference materials from Asia and North America, scale pattern analysis (SPA) has proved to be the most useful approach for estimates of stock mixtures of salmon on the high seas. For coho, the few tags recovered from fish released in the fisheries areas have been of mostly Asian origin. SPA shows more of a mix of Asian and North American (primarily western Alaskan) coho. Early analyses showed that coho in the fisheries areas were mostly Asian, like the tag recoveries (Myers et al. 1981; Walker and Harris 1982). More recent analyses have shown progressively higher percentages of western Alaskan fish, but the quality of Asian scale samples in some of the analyses was not always good (not all scales appear to have been collected from the same body area as Alaskan and high seas samples) and this may have affected results (Walker and Davis 1983; Walker 1990, 1991, 1992, 1993).

For chinook there are virtually no tag recoveries from the fisheries areas. SPA shows western Alaska (particularly Yukon River) chinook predominate in the Bering Sea, but Asian chinook are also numerous. There is a mix of Asian and North American (western and south central Alaskan) stocks from south of the Aleutians down to 46°N, with wide variations in estimated proportions between years. South of 46°N, in the landbased salmon fishery area, the chinook are estimated to be predominately Asian (Myers et al. 1984, 1987; Myers and Rogers 1988; Davis 1990, 1991).

Parasites - Parasites acquired in freshwater and found only in one region or on one continent have also been used to determine the freshwater origin of chinook caught on the high seas. Japanese scientists examined two myxosporean brain parasites, *Myxobolus arcticus* and *M. neurobius*, in adult head and smolt samples from two Asian rivers and from North American rivers from the Yukon to the Sacramento (Urawa et al. 1990; Urawa and Nagasawa 1991). *Myxobolus arcticus*, the "Asian" indicator parasite, is also found in some North American chinook (as well as in other species) from British Columbia and southeastern Alaska, and North American fish are probably responsible for infections of *M. arcticus* found in chinook in the Gulf of Alaska. In the Bering Sea and the central and western North Pacific, parasite data generally agree with SPA estimates: there are few infected chinook in the Bering Sea (indicating mostly North American origins), and much higher infection rates south of the Aleutians (indicating predominantly Asian fish). *Myxobolus neurobius* has been found only in Columbia River chinook and so far has not been detected in high seas samples. However, it might be a useful indicator of Columbia River chinook in coastal mixed fisheries.

High Seas Squid Driftnet Fisheries

The distribution of observer coverage of the Asian squid driftnet fisheries in 1990 and 1991 appeared to be representative of the distribution of reported fishing effort and covered roughly 10% of the fisheries. The reported and observed effort of the squid fisheries indicate that most fishing occurs in the central and western North Pacific. Nearly all of the salmon bycatch noted by the observers in 1990 and 1991 was west of 165°W, with a few from further east (to about 151°W) (INPFC 1991, 1992).

Most of the salmonid bycatch of high seas squid driftnet fisheries is split between coho and chum; other species contribute only a few percent (INPFC 1991, 1992). An illegal Taiwanese salmon catch seized in 1989 also consisted mainly of coho (50.3%) and chum (38.6%) (Waples et al. 1989). Coho and chum appear to tolerate slightly warmer waters than do other salmon species. This is particularly true for coho, which are much less abundant overall than chum salmon in the North Pacific, but are nevertheless observed in slightly greater numbers in the squid driftnet fisheries.

Sightings of vessels fishing illegally beyond squid fishery boundaries have been mostly just north of the northern boundary, where they were probably fishing for larger squid which are found in cooler waters, and farther north up from the western edge of the fishery, where the vessels were probably targetting Asian pink and chum salmon (Fig. 6; Pella et al. 1991).

Aside from the recoveries of three Washington and Oregon coded-wire tagged coho from the squid driftnet fisheries in 1991, there are few recoveries of Pacific Northwest coho and chinook from in or near the squid or salmon fisheries areas. The range of western Alaskan coho as indicated by tagging has some overlap with the squid fishery areas.

In most years the legal squid driftnet fisheries probably had relatively small catches of salmon (compared to inshore fisheries) when vessels were operating south of the northern boundary, and the salmon stocks most affected were Asian and Alaskan, although Pacific Northwest and British Columbia steelhead were undoubtedly taken as well. It is difficult to judge the effect of the illegal fishery that targets on salmon, but observed transgressions seem to be mostly on the western side of the Pacific.

Future Research Plans

Because of the end of legal high seas salmon and squid driftnet fisheries, American, Japanese, Canadian, and Russian researchers are looking at new areas of research. We hope to direct our efforts at several important topics: carrying capacity questions, especially as related to potential competition between wild fish and large releases of hatchery fish; ocean growth; and the relation of salmon distribution, abundance, and growth to oceanographic conditions.

We are working with Russian scientists on cruise plans for a Russian research vessel in 1993, and we hope this becomes the start of a new series of long-term high seas salmon studies. We plan to establish permanent sampling stations and transects which would be revisited every year to accumulate a comparative data series on salmon in the North Pacific Ocean. The Russians would use the vessel from late March to May west of about 170°E to study the late winter and early spring distribution of Russian pink salmon stocks. FRI would sample in three areas: south of Adak Island in the Aleutians where we already have a long data series; along a diagonal transect from about 52°N,170°W to 38°N,155°W (which would cover an area not well studied in the past, where western and south central Alaskan salmon stocks mix with Asian chum salmon); and along a transect at 145°W in the Gulf of Alaska. We will continue tagging studies, as well as sampling for food habits, growth, and condition studies.

We also plan to continue placing biologists on Japanese and Canadian research vessels. On similar cooperative cruises in the past, we have conducted studies on salmon food habits, survival after different dehooking methods from longline gear, and scale collection studies. We want to extend the food habits studies to see what changes occur from year to year, and begin developing foraging and bioenergetics models.

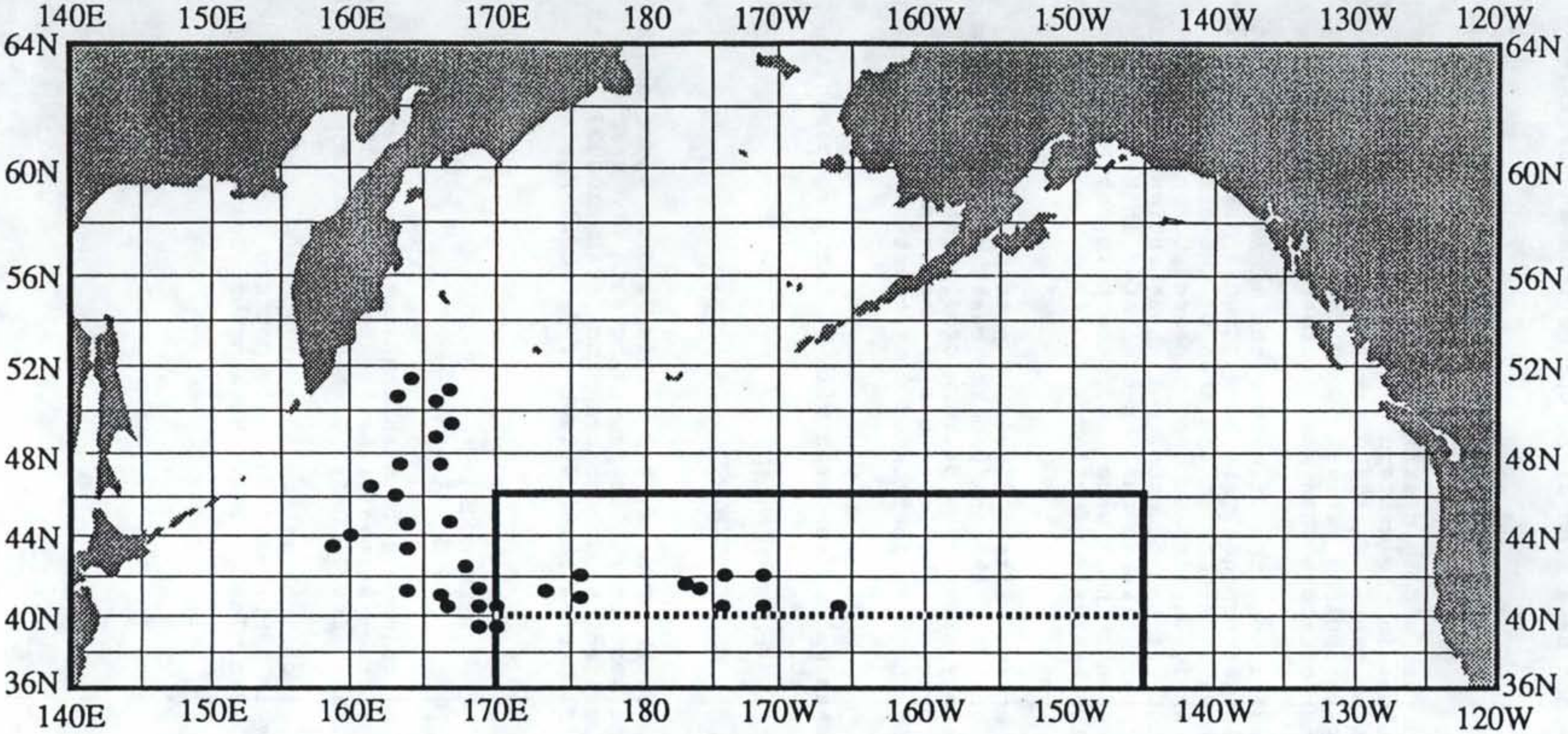

Figure 6. Approximate sighting locations of Japanese, Korean, and Taiwanese driftnet vessels operating outside of authorized areas in 1991, and maximum (September) boundary of the Japanese squid driftnet fishery. Sighting locations inside of the boundary were north of the authorized area in a month other than September; dashed line is the northern boundary in June and December. Adapted from Pella et al. 1991.

We have extensive biological databases and historical scale collections from high seas sampling over the last 35 years, although these resources have limitations in both time and space in their coverage of the North Pacific. We will use our historical scale collection to measure growth from different periods, and will compare length data from different periods and areas.

SUMMARY

1. Chinook and coho salmon are widely distributed in the North Pacific Ocean, with concentrations in the western and central North Pacific and along the Gulf of Alaska coast. Marine distribution of coho and chinook from different geographic regions varies. Coho from Asia and western Alaska are found further offshore than more southern stocks.
2. Tag recoveries are insufficient to estimate stock mixing in driftnet fishery catches. Scale pattern analysis have shown a mixture of Asian and North American coho and chinook in the area of the driftnet fisheries. In recent analyses Alaskan coho appeared to make up the majority of the driftnet catches of coho, but these estimates are at odds with tag return data. Alaskan chinook dominated in Bering Sea chinook catches, but Asian chinook were predominant in catches south of the Aleutians. Parasite analysis of chinook confirms the scale pattern results.
3. We can infer that the squid driftnet fisheries probably had minor effects on west coast coho and chinook stocks based on the location of the squid driftnet fishery areas, the pattern of fishing effort allocation and salmon bycatch, the locations of observed driftnet fishing violations, and the distribution of west coast coho and chinook salmon.
4. New emphases in high seas salmon research will be carrying capacity, growth, and stock interactions.

ACKNOWLEDGMENTS

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AN INTERVENTION ANALYSIS APPROACH TO MODELING THE LINKAGE BETWEEN NORTHEAST PACIFIC REGIME SHIFTS AND ALASKA SALMON PRODUCTION

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ABSTRACT

Alaska salmon stocks have exhibited enormous fluctuations in production during the 20th century. In this paper, we investigate our hypothesis that large scale salmon production variability is driven by north Pacific climatic processes. Using a time series analytical technique known as Intervention Analysis, we demonstrate that Alaska salmonids alternate between high and low production regimes. The transition from a high(low) regime to a low(high) regime is called an intervention. The timing of these interventions is shown to be nearly synchronous across different species of salmon as well as salmon from different origins. To test for interventions, we first fit the salmon time series to univariate autoregressive integrated moving average (ARIMA) models. Potential interventions were then identified, incorporated into the models and the resulting fit compared to the non-intervention models. A highly significant positive step intervention in the mid 1970s was found in the four major Alaska salmon stocks analyzed. The difference in mean production before and after the mid 1970's intervention was found to be 17.7 million for western Alaska sockeye, 4.8 million for central Alaska sockeye, 13.0 million for southeast Alaska pink, and 22.6 million for central Alaska pink. A smaller, negative, step intervention in the mid 1950s was identified for each of the four stocks. The drop in mean production was estimated at: 6.3 million for western Alaska sockeye, 2.0 million for central Alaska sockeye, 12.8 million for southeast Alaska pink, and 6.4 million for central Alaska pink. We review the evidence for synchronous climatic regime shifts in the early 1950s and mid 1970s that coincide with the shifts in salmon production. Potential mechanisms linking north Pacific climatic processes to salmon production are identified.

INTRODUCTION

In the mid 1970s, ocean conditions in the north Pacific underwent a dramatic and abrupt change (Graham 1992). Coincident with the physical regime shift, Alaskan salmonids entered an era of greatly increased production that has persisted into the 1990s (Figure 1). Throughout their long (100 yr +) commercial exploitation history, several of the Alaska salmon species have demonstrated "red noise" variability wherein periods of high (low) production tend to persist for a lengthy period before abruptly reversing to the opposite state. For example, in the 1930s and early 1940s, salmon landings were high, followed by an era of low catches from the early 1950s to mid 1970s. As Alaskan landings increased in the mid 1970s, several North American west coast stocks, notably Oregon coho (Pearcy 1992), went into a prolonged period of decline.

Much early research on variability in salmon survival (and, therefore, production and catch) focused on the freshwater stage of their life

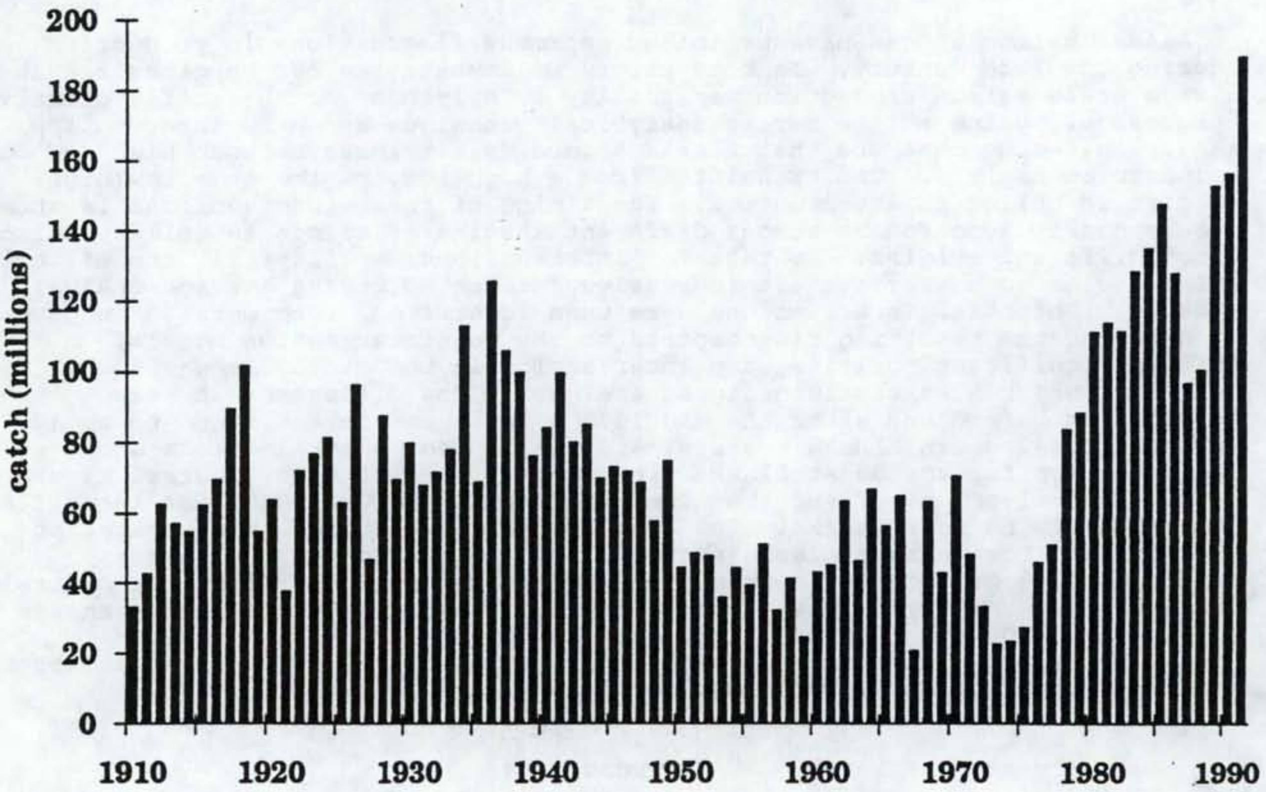

Figure 1. Trend in total Alaska salmon catch, 1910-1991.

cycle, attempting to link survival to conditions in their spawning and rearing habitat. The period spent at sea was regarded as relatively unimportant. There is now a growing awareness of the importance of the marine environment in determining salmon production (see, e.g., Pearcy 1984, Beamish and McFarlane 1989).

Variability in marine survival of salmon is poorly understood (Mathews 1984). Numerous studies have attempted to correlate survival with environmental factors, though few have proven useful in predicting salmon abundance or assisting in management decision making (Pearcy 1992). Part of the difficulty in elucidating the driving factors of survival is that the relationship between the environment and survival is clouded by many factors. Biotic (e.g., intra and interspecific competition, prey availability, predation) and abiotic (environmental variables, habitat) factors not only exhibit complex relationships with survival (non-linear, threshold) but are themselves often highly correlated.

Despite these drawbacks, the importance of attempting to understand the causes of variable survival is paramount (Francis and Sibley 1991). In particular, understanding large scale and long term variability would benefit both fishery managers and fishermen (Shepherd et al. 1984).

Large marine ecosystems fluctuate in response to physical forcings which occur over a number of time intervals. There appears to be a nested hierarchy of time relevant to their dynamics, ranging from relatively discrete events which occur over times on the order of a year or less (e.g., the 1970 North Pacific winter atmospheric circulation pattern - Hollowed and Wooster 1991), to epochs which occur at the inter-century level (Baumgartner et al. 1992). What we are most interested in identifying in this analysis are regimes which define points in time, separated by intervals on the order of decades, where major jumps or shifts in the level of abundance occur in large marine ecosystems. Therefore, in examining the interannual dynamics of various biological components of large marine ecosystems, what we see are responses to these nested hierarchies of interacting processes occurring at different time scales and working synergistically to create pattern. In this analysis, it is pattern at the regime level that we are trying to interpret.

We hypothesize that large scale variability in salmon production is driven by large scale climate change, reflected in north Pacific atmospheric/oceanic regime shifts. Under this hypothesis, salmon populations exhibit two characteristics: relatively stable production while a particular regime persists, followed by a rapid transition to a new production level in response to the physical regime shift. If large scale salmon production is closely related to north Pacific climate processes, we should find coherent shifts in mean production levels across both species and area. Using the salmon as a biological indicator, the timing of the changes in salmon production will then help pinpoint the physical regime shifts.

To test this hypothesis, we proceed by first statistically analyzing the historical production dynamics of four major Alaska salmon stocks: western and central sockeye, southeast and central pinks. While many researchers have noted the aforementioned swings in production, there remained the possibility that the changes might be either random processes or nonsignificant, in a statistical sense. Due to the high serial correlation (lack of independence between successive observations), the *t*-test for equality of means cannot be used to test for production shifts. We utilize a time series technique known as intervention analysis to identify the significance, timing and magnitude of structural shifts (interventions) in the four time series. Intervention analysis is a relatively recent statistical technique recommended as a method for detecting and quantifying non random change in an unreplicated experiment (Carpenter 1990).

To test for interventions, we fit univariate time series models of the Box-Jenkins (1976) autoregressive integrated moving average (ARIMA) class. These ARIMA models provide a baseline fit to the correlation structure exhibited by the time series. Interventions are subsequently identified by analyzing model residuals. Model parameters are reestimated incorporating the intervention(s) and the models compared on the basis of several criteria. On the basis of the identified interventions, we then review the evidence for large scale physical regime shifts in the north Pacific.

TIME SERIES MODELING AND INTERVENTION ANALYSIS

The use of time series analysis to model fish population dynamics has seen increasing use in recent years. Most theoretical development and initial application has taken place in the econometric and business forecasting literature. Recognition of the potential applicability to ecological problems appears to have begun with Moran (1949).

There are five classes of commonly applied time series models (Jenkins 1979). The simplest, and most widely known, are the so-called Box-Jenkins ARIMA univariate models. Simple ARIMA models utilize only the history of the time series to "explain" its observed variability. The second class are the transfer function noise (TFN) models which relate an output series variability to both its own history and that of one or more explanatory variables. A third class, related to TFN models, are intervention models which incorporate the effects of unusual events, natural or man-made, to modify ARIMA models. Two types of multivariate models comprise the other two classes. Multivariate stochastic models permit feedback among several time series, and are often referred to as vector ARIMA models. The final class includes explanatory variables giving a multiple input/multiple output mode, and are sometimes referred to as multivariate transfer function models.

In this paper, we use intervention models to examine if north Pacific regime shifts are reflected in Alaska salmonid time series. We provide a brief outline of the technique and explanation of time series terminology and notation. Those seeking a more theoretical development should consult one of the numerous texts available including the seminal works on ARIMA model formulation (Box and Jenkins 1976) and intervention analysis (Box and Tiao 1975). The notation utilized inthis paper follows the format of Wei (1990).

Model Development

Univariate time series model building, in the methodology of Box and Jenkins (1976), proceeds in the following fashion:

1) Model Identification. In this step, tentative models are identified. Determination of need for power transformation (for variance stabilization) and differencing (to render the series stationary in the mean) are first evaluated. Plots of the autocorrelation and partial autocorrelation functions (ACF and PACF respectively) of the possibly transformed series are examined to assist in determining the order of the AR and MA components (Box and Jenkins 1976). Several other identification tools are also available, such as the extended sample autocorrelation function (ESACF, Tsay and Tiao 1984), generalized partial autocorrelation coefficient (GPAC, Woodward and Gray 1981) and the prediction variance horizon (PVH, Parzen 1981).

2) Parameter estimation. Following selection of a potential model(s), estimates of the parameters are calculated. Access to time series software is almost essential as ARIMA model parameters must be fit via a nonlinear estimation routine (though the models themselves are usually linear). Maximum likelihood procedures, usually based on the Cholesky decomposition, or the

Kalman filter, have been developed as an alternative to the early methods of least squares and approximate likelihood utilized by Box and Jenkins (1976). Standard errors are also computed, and parameters judged to not be significantly different from zero can be dropped. The remaining parameters are then reestimated.

3) Model diagnostic checking. With a tentative model selected and parameters estimated, the adequacy of the model must be assessed to determine if model assumptions are met. One basic assumption is that the residuals a_t form a white noise series. A common test is the portmanteau test of Box and Pierce (1970), which uses the residual ACF to test the joint null hypothesis that all serial correlations are equal to zero. Secondly, it is common in time series analysis that several models may be adequate in the sense that the model residuals are reduced to white noise. Several model selection criteria have been developed to assist in model selection. In this analysis, we compared competing models using five criteria: mean absolute error (MAE) which measures the average one step ahead prediction error; the unbiased residual variance S^2_a, equal to the residual sum of squares divided by degrees of freedom; the coefficient of determination r^2, the amount of variance "explained" by the model; Akaike's Information Criterion (AIC, Akaike 1974); and Schwartz's Bayesian Criterion (SBC, Schwartz 1978). The AIC and SBC are performance statistics that balance statistical fit with model parsimony. The SBC utilizes a larger penalty function than the AIC thus often suggesting a less parameterized model. Formulas for the model diagnostic and selection criteria are contained in the Appendix.

Intervention Detection and Estimation

In intervention analysis, the correlation structure is initially assumed unaffected by the interventions which are modeled as deterministic functions of time. Once the "best" ARIMA model has been selected, the three step modeling sequence is repeated to identify and test the significance of interventions.

The original intervention methodology developed by Box and Tiao (1975) permitted estimation of intervention effects when the timing of the interventions were known *a priori*. To handle the situation where the number and timing of potential interventions is unknown, corresponding to the situation at the beginning of our analysis, Chang and Tiao (1983) proposed an iterative detection technique using a likelihood ratio test. Interventions are identified in a stepwise fashion begining with the residuals from the univariate model. Following detection and estimation of an intervention, model parameters are estimated and the resultant intervention model compared to the univariate model using the criteria cited above. The new model residuals can then be reanalyzed for evidence of other interventions.

A good general review of intervention models is contained in Wei (1990), while Noakes (1986) discusses the applicability of intervention analysis to fisheries problems.

There are two types of interventions, pulse and step, the first representing a discrete system shock, the second a permanent change in the mean level of a process. A myriad of delayed system responses can be modeled. In our analysis, we restrict ourselves to step interventions that result in abrupt, permanent shifts in the mean, corresponding to the type of transition we hypothesize between different production regimes. This method minimizes the number of estimated parameters, and reduces the probability of identifying spurious interventions. Two software packages, AUTOBOX (AFS 1992), and TIMESLAB (Newton 1988) were used for all analyses.

DATA

The salmon landings data used in this study were taken from an Alaska Department of Fish and Game annual report (ADFG 1991). We selected four of the major regional groups of stocks: western Alaska sockeye, central Alaska pink and sockeye, and southeast Alaska pink salmon. Landings data for these regional stocks are more likely to reflect actual production than other Alaskan salmon stocks as they have been the most intensively exploited stocks due to their high abundances and value. These four regional stocks accounted for an average of 80.4% of total Alaskan salmon catches (by number) for the period 1910-1988 (ADFG 1991). To more accurately reflect salmon production by area (Figure 2), we corrected the Alaska landings for interceptions using data provided in Shepard et al. (1985) and Pacific Salmon Commission (1991). Western Alaska salmon catches were adjusted for the number estimated to have been taken in Japanese high seas salmon fisheries. Southeast Alaska salmon landings were incremented by the estimated British Columbia (BC) catch and decremented by the numbers of BC salmon estimated taken in Alaska.

For three of the time series we analyzed (western and central sockeye, southeast pink) we used 1910-1989 data which we consider to be the period of full exploitation. The other series (central pink) we restrict to 1926-1989 to eliminate the early "fishing up" period prior to full exploitation. Analyses were also conducted on the full central pink salmon series and reported below, but we believe the results from the shorter period are more reflective of the true dynamics in the fishery.

RESULTS

Western Alaska Sockeye

The western Alaska sockeye data required a square root transformation to stabilize the variance. Differencing was not required. Examination of the ACF and PACF indicates rather complex dynamics in this time series (Figure 3). Lags 1, 4 and 5 in the ACF and lags 1, 4, 6 and 7 in the PACF were significant. A variety of models were fit and compared. Initial identification indicated three candidate univariate models: (6,0,0), (1,0,5) and the seasonal model $(1,0,0) \times (1,0,0)_5$. Diagnostics indicated residual serial correlation at lag 3 for the seasonal model, thus a moving average term was added and the resultant $(1,0,0) \times (1,0,0)_5 \times (0,0,1)_3$ model compared with the nonseasonal models. The seasonal model outperformed the two nonseasonal models in all diagnostics and the residuals pass the portmanteau white noise test. The final fitted model parameter estimates and standard errors for the univariate and subsequent intervention models are given in Table 1. Model diagnostics for the univariate and intervention models are given in Table 2.

Two potential step interventions were identified for western Alaska sockeye, a negative step in 1951 and a positive step in 1979. We fit two intervention models, one incorporating just the 1979 intervention and another incorporating both. Both models substantially outperformed the nonintervention model. The coefficient of determination, r^2, improved from .480 to .591 with the 1979 intervention and further increased to .643 with inclusion of the 1951 intervention[1]. Both the AIC and SBC decreased substantially with the addition of each intervention.

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[1] All model diagnostics reflect model fit in the transformed metric, thus for western Alaska sockeye, the statistics result from model fitting in natural logarithm space.

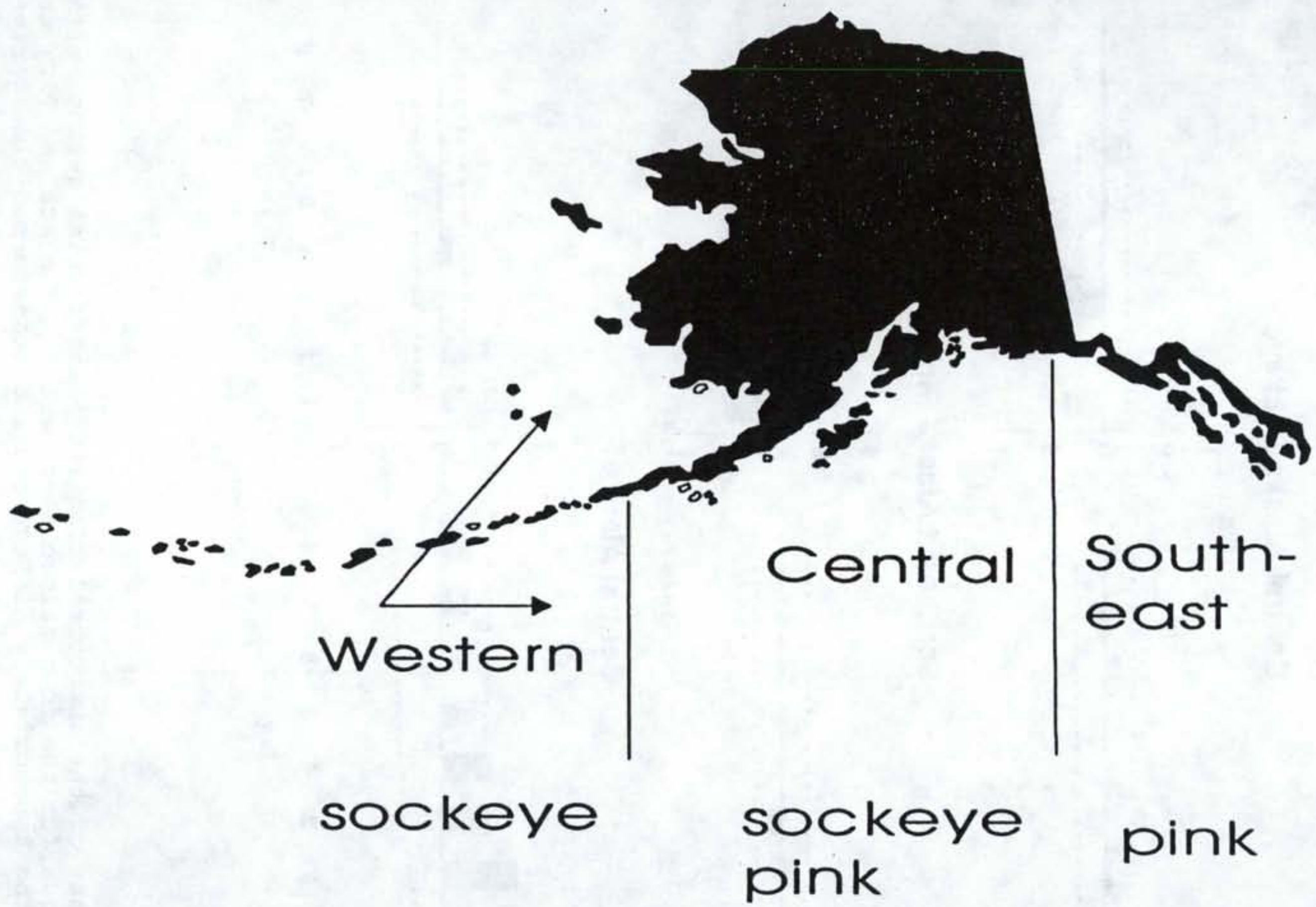


Figure 2. ADF&G statistical areas and regional salmon stocks utilized in this study.

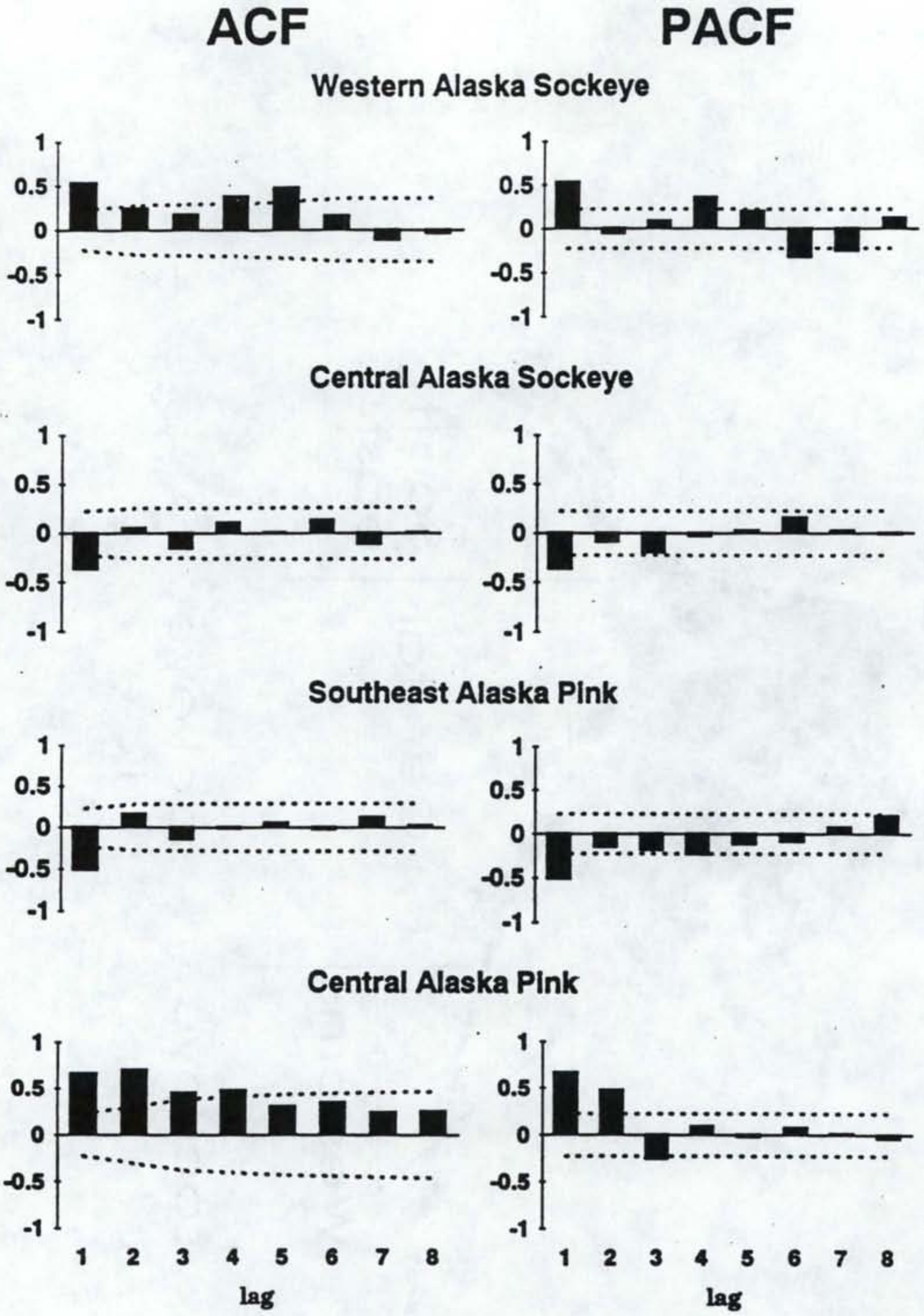


Figure 3. Plots of the autocorrelation (ACF) and partial autocorreltaion (PACF) functions for the four salmon time series. The ACF and PACF are computed for the appropriately differenced and transformed time series.

Table 1. Univariate and Intervention ARIMA models with parameter estimates and associated standard errors developed for western Alaska sockeye salmon.

Model	Parameter estimates and standard errors
Univariate	$(1 - \underset{(.102)}{.605}B)(1 - \underset{(.105)}{.490}B^5)\sqrt{Y_t} = \underset{(.219)}{.705} + (1 - \underset{(.127)}{.209}B^3)a_t$
1 Intervention (1979)	$(1 - \underset{(.110)}{.353}B)(1 - \underset{(.096)}{.577}B^5)\sqrt{Y_t} = \underset{(.236)}{.811} + (1 - \underset{(.119)}{.258}B^3)a_t + \underset{(.403)}{2.174}I_t^{1979}$
2 Interventions (1951, 1979)	$(1 - \underset{(.127)}{.223}B)(1 - \underset{(.104)}{.465}B^5)\sqrt{Y_t} = \underset{(.390)}{1.496} + (1 - \underset{(.114)}{.421}B)a_t - \underset{(.236)}{1.026}I_t^{1951} + \underset{(.275)}{2.355}I_t^{1979}$

Table 2. Summary statistics for univariate and intervention ARIMA models developed for western Alaska sockeye salmon. MAE is mean absolute error of fitted values, σ^2_a is the unbiased residual variance, r^2 is the coefficient of determination, AIC is Akaike's Information Criterion, SBC is Schwarz' Bayesian Criterion, Q is the portmanteau residual autocorrelation test (up to lag 20) and associated p-value. All statistics calculated in the transformed metric.

Model	MAE	σ^2_a	r^2	AIC	SBC	Q	p value
Univariate	.684	.763	.480	-16.1	-6.93	10.52	.838
1 Intervention	.616	.609	.591	-31.9	-20.4	8.57	.899
2 Interventions	.555	.538	.643	-40.1	-26.3	9.95	.766

Resultant model fits and pre and postintervention means for the three models are illustrated in Figure 4. For the one intervention (1979) model, estimates of the pre and postintervention means were 8.801 and 26.429 million respectively, resulting in an estimated step intervention of 17.628 million. In the two intervention model, the 1951 step was estimated at -6.335 million and the 1979 step at 17.671 million. The three means were estimated at: 12.965 (1910-1950), 6.630 (1951-1978) and 24.301 million (1979-1989).

The lag 1 and 5 autoregressive terms in the univariate model possibly indicate the existence of two concurrent phenomena in the population: persistence in the level of stock production and cyclic dominance. The effect of the interventions on the AR and MA parameters was to decrease the AR(1) and increase the MA(3) coefficients with little effect on the AR(5) coefficient. The decrease in the AR(1) coefficient is expected as the interventions accounted for part of the high serial correlation (persistence).

Central Alaska Sockeye

The central Alaska sockeye time series exhibited nonstationarity in both the variance and the mean, therefore the series was natural logarithm transformed and differenced. The resultant series ACF and PACF both showed only a lag 1 significant term (Figure 3) indicating either a (1,1,0) or (0,1,1) model. Model diagnostics indicated a slightly better fit for a (0,1,1) model. Parameter estimates for the univariate and intervention models are given in Table 3, model statistics in Table 4.

A large positive step intervention was identified in 1976. Overall model fit was significantly improved by incorporating the 1976 intervention. Inclusion of a second, negative, step intervention in 1957 further improved the model.

Because a (0,1,1) is nonstationary and possesses no single mean, quantifying the effect of a step intervention is less straightforward than for an undifferenced model. To provide an estimate of the effective change in mean catch following a step intervention, we estimated the magnitude of the intervention (in the differenced and transformed series), and then created a modified series by removing the effect of the intervention from the fitted series. This provided an estimate of how the series would have behaved had there been no intervention. Both series were then retransformed to the original metric. The change in mean catch was calculated as the difference between the means of the two series following the intervention (Figure 5). The effective change in mean catch for the one intervention model (1976) was 4.815 million. The estimated mean for the 1977-1989 period incorporating the intervention was 8.469, compared to an estimated mean of 3.654 million without the intervention effect. For the two intervention model, the interventions were estimated to have decreased mean catch by 1.996 million (from 5.082 to 3.086 million) for the 1957-1975 period, and increased mean catch by 4.789 million (from 3.646 to 8.435 million) for the 1976-1989 period.

There is no clean biological interpretation of the (0,1,1) model. The lag one MA parameter does indicate a high degree of serial correlation. The local mean of a (0,1,1) process is given by an exponentially weighted moving average (Box and Jenkins 1976). The amount of dependence on previous levels of the population is determined by the MA parameter q. A q value of one results in what is known as a random walk model where the best estimate of population size is simply the previous observation. As q approaches zero the past population sizes became increasingly important. In our models, q varied around 0.5, indicating a strong relationship to the previous three or four observations.

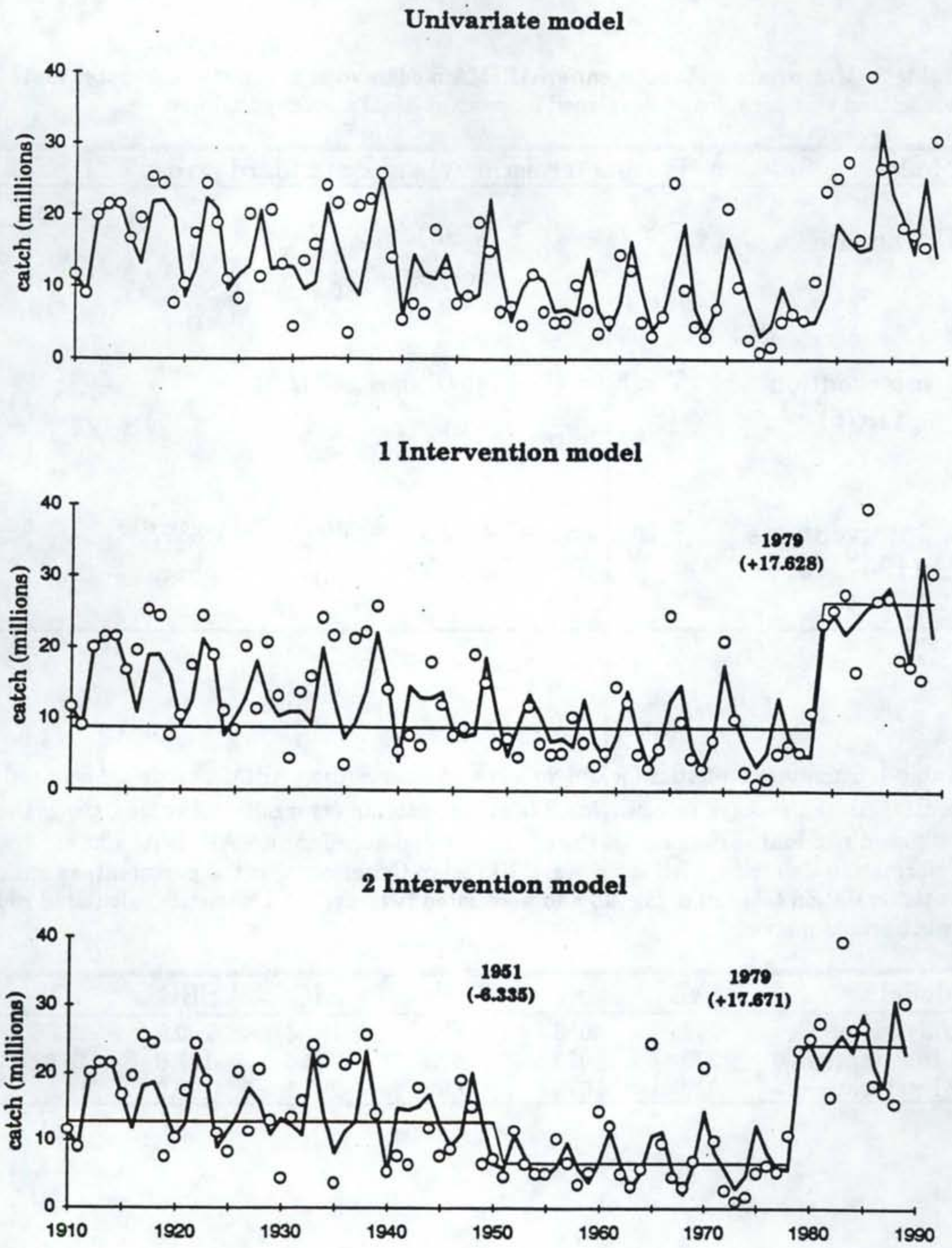


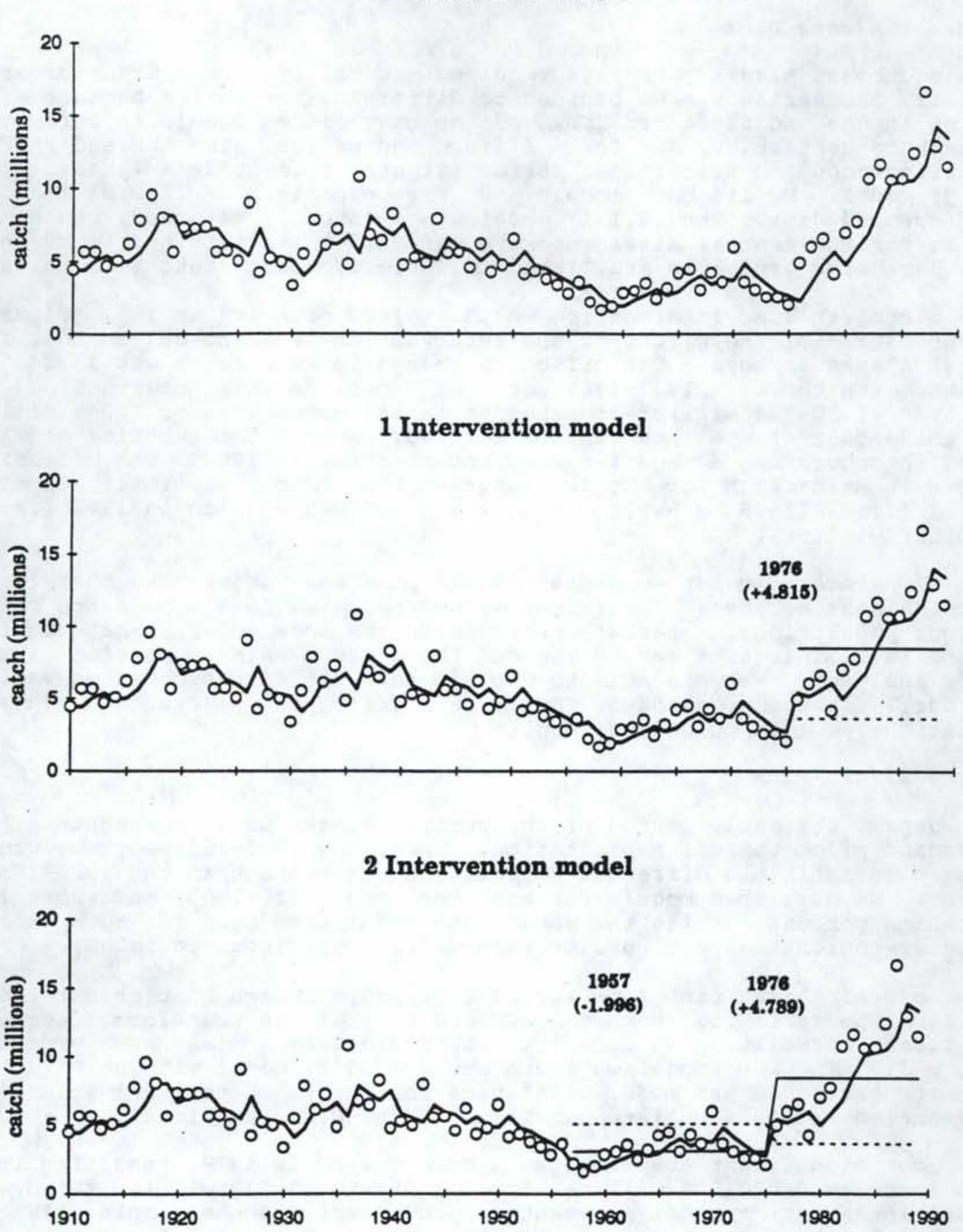
Figure 4. Plots of model fits for ARIMA and intervention models developed for western Alaska sockeye salmon time series, 1910-1989. Landings data are indicated by points, fitted values by thick line. Estimated means before and after interventions are shown by thin lines. Timing of the step interventions and resultant change in mean are also shown.

Table 3. Univariate and Intervention ARIMA models with parameter estimates and associated standard errors developed for central Alaska sockeye salmon.

Model	Parameter estimates and standard errors
Univariate	$\nabla \ln Y_t = (1 - \underset{(.101)}{.429}B)a_t$
1 Intervention (1976)	$\nabla \ln Y_t = (1 - \underset{(.101)}{.464}B)a_t + \underset{(.244)}{.841}I_t^{1976}$
2 Interventions (1957, 1976)	$\nabla \ln Y_t = (1 - \underset{(.072)}{.481}B)a_t - \underset{(.231)}{.499}I_t^{1957} + \underset{(.236)}{.839}I_t^{1976}$

Table 4. Summary statistics for univariate and intervention ARIMA models developed for central Alaska sockeye salmon. MAE is mean absolute error of fitted values, σ^2_a is the unbiased residual variance, r^2 is the coefficient of determination, AIC is Akaike's Information Criterion, SBC is Schwarz' Bayesian Criterion, Q is the portmanteau residual autocorrelation test (up to lag 20) and associated p-value. All statistics calculated in the transformed metric.

Model	MAE	σ^2_a	r^2	AIC	SBC	Q	p value
Univariate	.243	.086	.623	-193.3	-190.9	18.58	.484
1 Intervention	.229	.075	.673	-202.6	-197.8	19.02	.391
2 Interventions	.222	.072	.692	-205.1	-198.0	16.57	.484

 Univariate model

1 Intervention model

2 Intervention model

Figure 5. Plots of model fits for ARIMA and intervention models developed for central Alaska sockeye salmon time series, 1910-1989. Landings data are indicated by points, fitted values by thick line. Estimated mean following an intervention is illustrated by a thin solid line, estimate of the mean effect of the intervention removed is shown by a thin dashed line. Timing of the step interventions and effective change in mean are also shown.

Southeast Alaska Pink

The southeast Alaska pink data required natural logarithm transformation to stabilize the variance. We decided to difference the series because of a slow decline in the undifferenced ACF, and the best fitted model, in terms of performance statistics, was for a differenced series. The ACF and PACF for the differenced and transformed series (Figure 3) identify a likely (0,1,1) or (1,0,0) model. We fit both models and, for comparison, a (2,0,0) model to the undifferenced data. The (0,1,1) model was eventually selected, the same model form as for the central Alaska sockeye series. Univariate and intervention model parameter estimates are listed in Table 5, model statistics in Table 6.

The strongest step intervention we identified occurred in 1977 (Figure 6). Interpretation of the effect of the intervention is identical to that for the central Alaska sockeye. The effective change in mean catch was 13.345 million, with the mean 1977-1989 catch incorporating the intervention estimated at 20.394 million compared to an estimated mean of 7.049 million with the effect of the intervention removed. A two intervention model was fitted incorporating a negative step intervention in 1952. The effective changes in mean catch for the two intervention periods were: -12.781 million in 1952 (from 21.676 to 8.895 million) and +12.975 million in 1977 (from 7.154 to 20.129 million).

The MA parameter q for southeast Alaska pink was larger than that for central Alaska sockeye. The larger value indicates less dependence on previous populations, a factor reflected in the more volatile nature of this time series. This time series yielded the poorest ARIMA model fit of the four series analyzed. We were able to explain just 35% of the observed variance with our 2 intervention model. Separate modeling of the even and odd year populations yielded no better results.

Central Alaska Pink

We suspect the early period of the central Alaska pink represents a "fishing up" period prior to full exploitation. Thus, the 1910-1925 segment would be expected to exhibit a different correlation structure than the 1926-1989 segment. We developed models for both the long (1910-1989) and short (1926-1989) time periods. While the same ARIMA model was found for both data sets, the intervention analysis results were different (discussed below).

The central Alaska pink time series required a square root transformation to stabilize the variance. Both the ACF and PACF of the transformed series show significant correlation at lags 1 and 2, indicating a mixed ARMA process (Figure 3). The best model we found was a (2,0,1) model with no AR(1) term. Parameter estimates and model statistics for the short term univariate and intervention models are listed in Tables 7 and 8, respectively.

The most significant step intervention occurred in 1979, resulting in a mean level increase of 20.018 million, from 15.054 to 35.072 million (Figure 7). The one intervention model substantially outperformed the simple ARIMA model for both the short and long time series. A second, negative, step intervention was detected in 1951. Incorporation of this intervention produced mixed results in model diagnostic statistics. For the short time series, four of the five comparison statistics favored the two intervention model over the one intervention model. The SBC, however, increased, from 246.8 to 247.7, when the second intervention was included. In the two intervention model, the 1951 step was estimated at -6.427 million, and the 1979 step at +22.609 million. The three means were estimated at 18.883 million (1926-1950), 12.456 million (1951-1978) and 35.065 million (1979-1989). For the long time series, all diagnostics favored the one intervention model. This result can be attributed to the inclusion of the low average catch period from 1910-1925.

Table 5. Univariate and Intervention ARIMA models with parameter estimates and associated standard errors developed for southeast Alaska pink salmon.

Model	Parameter estimates and standard errors
Univariate	$\nabla \ln Y_t = (1 - \underset{(.081)}{.690}B)a_t$
1 Intervention (1977)	$\nabla \ln Y_t = (1 - \underset{(.079)}{.728}B)a_t + \underset{(.419)}{1.062}I_t^{1977}$
2 Interventions (1952,1977)	$\nabla \ln Y_t = (1 - \underset{(.072)}{.792}B)a_t - \underset{(.357)}{.891}I_t^{1952} + \underset{(.367)}{1.034}I_t^{1977}$

Table 6. Summary statistics for univariate and intervention ARIMA models developed for southeast Alaska pink salmon. MAE is mean absolute error of fitted values, σ^2_a is the unbiased residual variance, r^2 is the coefficient of determination, AIC is Akaike's Information Criterion, SBC is Schwarz' Bayesian Criterion, Q is the portmanteau residual autocorrelation test (up to lag 20) and associated p-value. All statistics calculated in the transformed metric.

Model	MAE	σ^2_a	r^2	AIC	SBC	Q	p value
Univariate	.505	.388	.253	-73.7	-71.4	15.79	.671
1 Intervention	.483	.364	.309	-77.8	-73.1	16.39	.566
2 Interventions	.470	.345	.353	-81.1	-74.0	13.70	.689

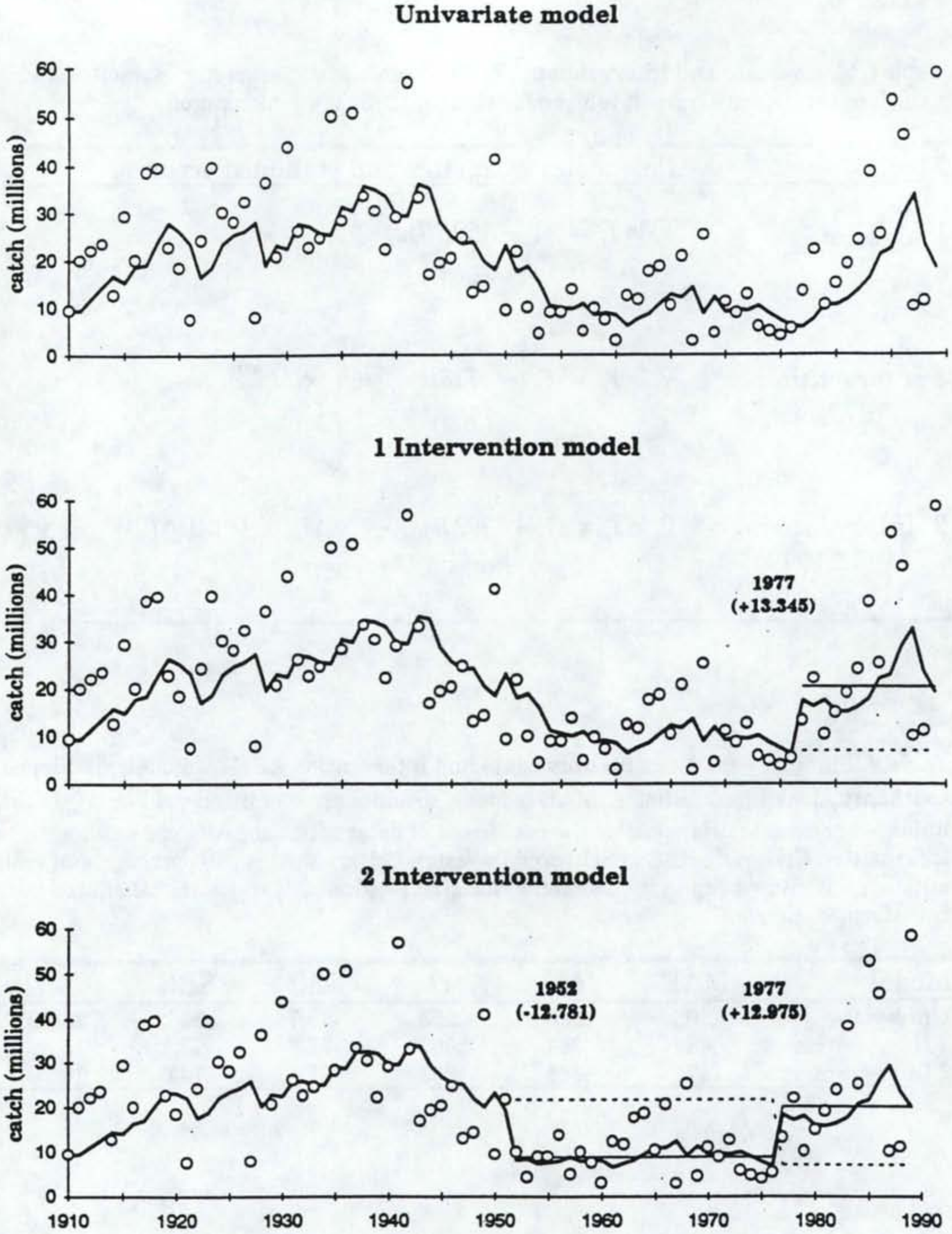


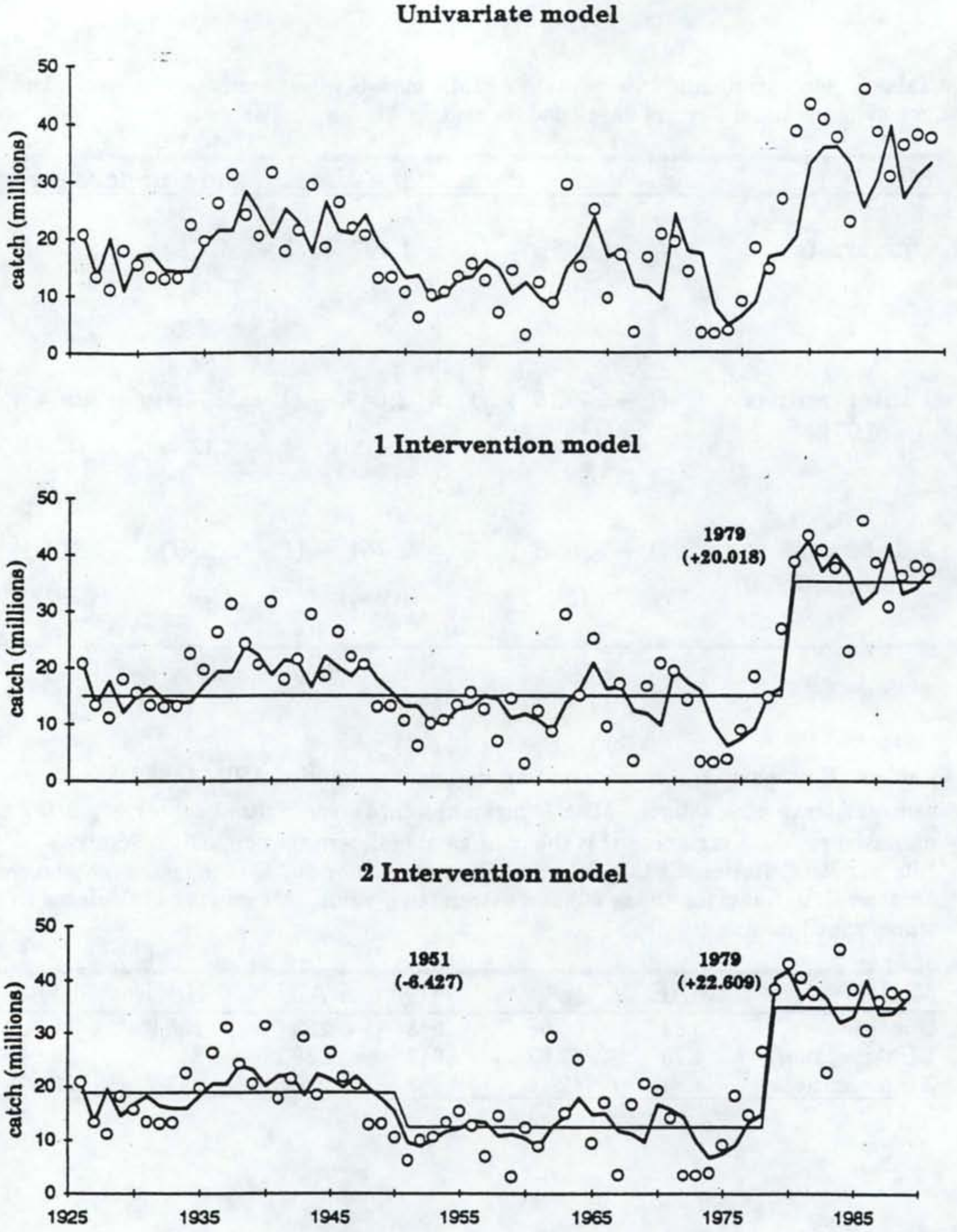
Figure 6. Plots of model fits for ARIMA and intervention models developed for southeast Alaska pink salmon time series, 1910-1989. Landings data are indicated by points, fitted values by thick line. Estimated mean following an intervention is illustrated by a thin solid line, estimate of mean with the effect of the intervention removed is shown by a thin dashed line. Timing of the step interventions and effective change in mean are also shown.

Table 7. Univariate and Intervention ARIMA models with parameter estimates and associated standard errors developed for central Alaska pink salmon.

Model	Parameter estimates and standard errors
Univariate	$(1 - \underset{(.107)}{.655}B^2)\sqrt{Y_t} = \underset{(.482)}{1.497} + (1 + \underset{(.118)}{.503}B)a_t$
1 Intervention (1979)	$(1 - \underset{(.118)}{.472}B^2)\sqrt{Y_t} = \underset{(.478)}{2.049} + (1 + \underset{(.127)}{.334}B)a_t + \underset{(.519)}{2.042}I_t^{1979}$
2 Interventions (1951, 1979)	$(1 - \underset{(.127)}{.365}B^2)\sqrt{Y_t} = \underset{(.599)}{2.761} + (1 + \underset{(.130)}{.253}B)a_t - \underset{(.397)}{.816}I_t^{1951} + \underset{(.468)}{2.392}I_t^{1979}$

Table 8. Summary statistics for univariate and intervention ARIMA models developed for central Alaska pink salmon. MAE is mean absolute error of fitted values, σ^2_a is the unbiased residual variance, r^2 is the coefficient of determination, AIC is Akaike's Information Criterion, SBC is Schwarz' Bayesian Criterion, Q is the portmanteau residual autocorrelation test (up to lag 20) and associated p-value. All statistics calculated in the transformed metric.

Model	MAE	σ^2_a	r^2	AIC	SBC	Q	p value
Univariate	5.84	56.06	.543	252.6	258.9	11.75	.815
1 Intervention	4.78	43.83	.649	238.2	246.8	18.84	.277
2 Interventions	4.84	42.19	.668	236.8	247.7	23.49	.074



Univariate model

1 Intervention model

2 Intervention model

Figure 7. Plots of model fits for ARIMA and intervention models developed for central Alaska pink salmon time series, 1910-1989. Landings data are indicated by points, fitted values by thick line. Estimated means before and after interventions are shown by thin lines. Timing of the step interventions and resultant change in mean are also shown.

Interpretation of the two univariate ARIMA parameters is relatively straightforward. They indicate that the central Alaska pink salmon catch is related to the catch two years previous and positive shocks from the previous year's catch. Incorporation of the interventions reduced both parameters as the change in mean accounted for part of the serial correlation.

DISCUSSION

Alaska salmon populations appear to alternate between high and low production regimes. We propose that Alaska salmon are responding to changes in north Pacific climate regimes. Under this hypothesis, each salmon population exhibits a unique smaller scale variability about some mean level of production during a climatic regime. The transition from one regime to another occurs relatively rapidly resulting in a shift in the mean production level of Alaska salmon populations.

In support of this hypothesis we have demonstrated nearly synchronous production shifts in four Alaska salmon stocks. These stocks include two different species from three widely separated geographic regions. Using the technique of intervention analysis, we identified three production regimes, defined by two major production shifts, one in the early 1950s, the other in the mid 1970s.

Alaskan pink and sockeye salmon spend the majority of their marine life cycle in the Central Subarctic Domain (CSD, Ware and McFarlane 1989) which encompasses the Gulf of Alaska (Figure 8). The principal feature within the CSD is the Alaska Gyre, an area of active upwelling at its core. The southern boundary of the CSD is defined by the Subarctic Current, whose latitudinal location varies yearly (Roden 1991). During the seaward and return migrations, pink and sockeye salmon pass through the Coastal Downwelling Domain, a region extending from Queen Charlotte Sound to Prince William Sound dominated by the Alaska Current.

Any attempt to link physical processes in the marine environment to Alaskan salmon production must involve oceanographic conditions within these two regions. We now examine the two production regime shifts in greater detail, summarize the change in production, and consider the evidence for concurrent climate regime shifts. We then discuss potential mechanisms linking the physics and biology.

Mid 1970s Shift

The increase in salmon production was highly significant in all four time series. In the 2 intervention models, the smallest t-value of the four mid 1970s step intervention variables was 2.817 ($p<.005$, southeast pink). The timing of the production shift varied slightly among the four series (1976 to 1979). Each of the four production groups is faced with a unique set of environmental conditions between their freshwater rearing habitat and entry into the marine feeding and migration grounds. The three geographic regions each contain numerous salmon bearing rivers. Localized factors may, therefore, delay or expedite the effect of the climatic regime on the population as a whole. It is clear, however, that the four stocks entered an era of increased production in the mid 1970s and have remained at that level into the 1990s. Combining the four series, we estimate the increased production resulted in an annual mean catch increase of greater than 58 million salmon.

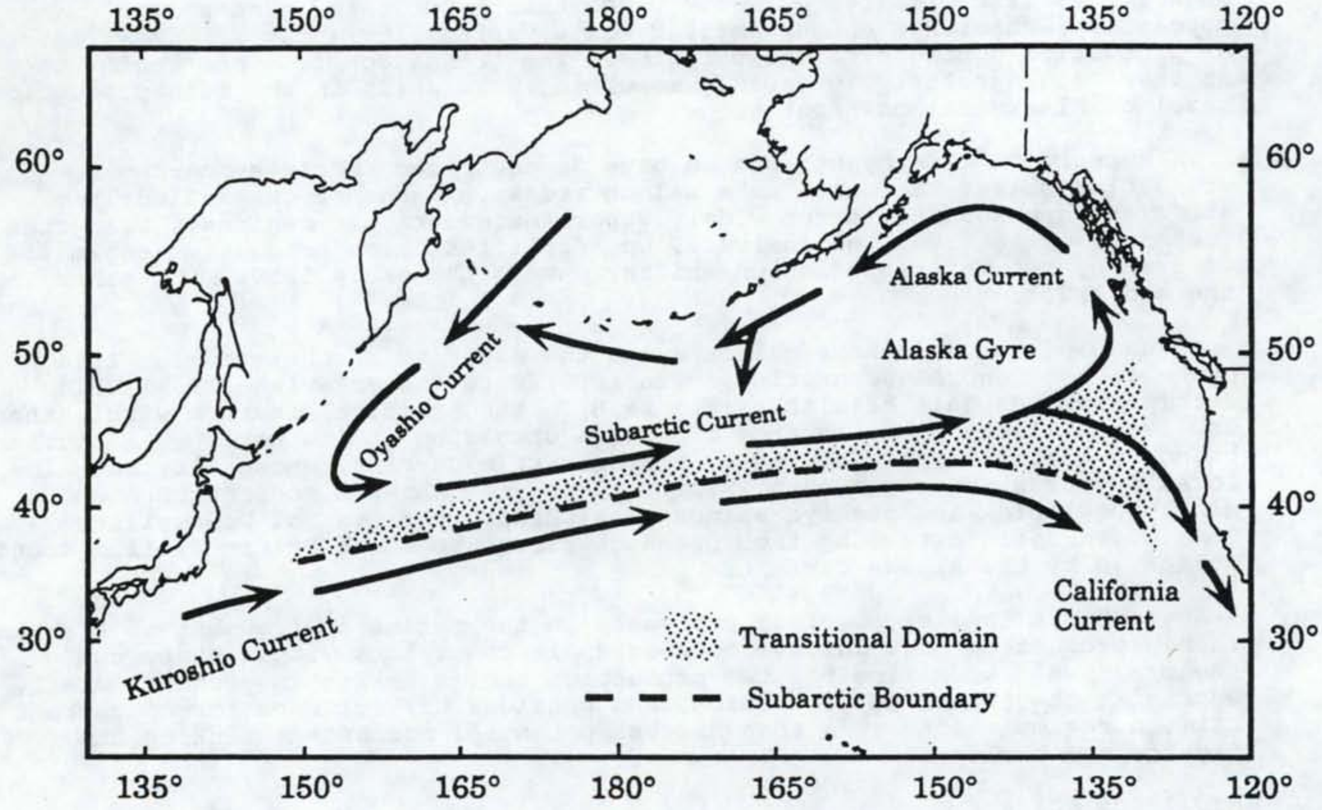


Figure 8. Summary of major oceanographic features of the north Pacific.

Evidence for the timing and strength of the mid 1970s "regime shift" (Hollowed and Wooster 1991) has been documented in numerous environmental and biological variables (Ebbesmeyer et al. 1991). The most obvious physical manifestations of the mid 1970s shift include a strengthening and eastward shift of the Aleutian Low (Trenberth 1990) and warming of the surface waters in the Gulf of Alaska (Royer 1989). Defining the event as the onset of a new regime rather than a temporary system shock reflects the persistence of the new state variables. Most evidence pinpoints the winter of 1976/77 as the critical transition period. The shift appears to have been forced by an increasingly vigorous winter circulation over the north Pacific (Graham 1992), leading to more severe and frequent winter storms (Seymour et al. 1984), decreases in mid Pacific SSTs and basin wide decreases in sea level pressure (Trenberth 1990). The large scale increase in central Pacific chlorophyll (and thus phytoplankton) during the 1970s has been attributed to persistence of warm SSTs into summer months (Venrick et al. 1987). The increase in Alaskan air and sea surface temperatures probably derived from warm air advected from the south by a strengthened Aleutian Low.

Hollowed and Wooster (1991) have hypothesized that the north Pacific alternates between two environmental states, with one transition occurring in 1977. The "cool" period prior to the transition, what they call a Type A regime, is characterized by a weak winter Aleutian Low, enhanced westerly winds in the eastern Pacific, decreased advection into the Alaska Current, and negative coastal SST anomalies. A "warm" era (Type B regime) is characterized by a strong winter Aleutian Low displaced to the east, enhanced southwesterly winds in the eastern Pacific, increased advection into the Alaska Current, and positive coastal SST anomalies.

The mechanisms driving the mid 1970s regime shift are the subject of much current intensive research. Several hypothesized mechanisms have suggested links between this regime shift in the Northeast Pacific and an abrupt climate shift in the tropical Pacific which occurred in the mid 1970s. Kashiwabara (1987) and Nitta and Yamada (1989) have hypothesized that changes in the tropical Pacific forced the change in north Pacific winter circulation patterns. Trenberth (1990) noted that in the period between 1976 and 1988, there were three warming El Niño, but no cooling La Niña events. Graham (1992) holds that the El Niño/La Niña cycle continued but the background state was set to a different state.

Early 1950s Shift

The negative production shifts identified in the early 1950s shifts were all significant, but of lesser magnitude than those of the mid 1970s. The t-values for the step interventions in the two intervention models ranged from 4.34 ($p<.001$, western sockeye) to 2.06 ($p<.025$, central pink). Timing of the shift (1951-1952) was coherent in three of the time series, but was not identified in the central sockeye series until 1957. Examination of the series, however, reveals a gradual decline in catch beginning in 1951. A ramp intervention could have been used to model this decline, though the addition of extra parameters to this model was not suggested by model statistics. We conclude therefore that the decline in catch was real and of the estimated magnitude and had its origin in the early 1950s. We estimate the combined drop in catch following the early 1950s intervention at approximately 27.55 million fish.

Evidence for an early 1950s regime shift is less confirmatory than for the mid 1970s. To some extent this may be due to the relative lack of data in comparison to that available for the later event. Also, if the salmon data are indicative of the physical data, the shift in physical variables is expected to be smaller.

Rogers (1984) presented average winter air temperatures for Kodiak and Bristol Bay from 1920-1983. A sharp drop in the temperatures occurred simultaneously in 1949/1950 and, with only a few exceptions, remained anomalously low until the 1976/77 winter. Surface temperature trends in the northern Hemisphere were shown by Jones (1988) to be in a cool period between the mid 1940s and mid 1970s. The frequency and intensity of ENSO events have undergone several changes in the past century (Trenberth 1990, Trenberth and Shea 1987) with strong events between 1880-1920 and 1950-present, and weak events between 1920-1950. Trenberth (1990) also noted the preponderance of Cold (La Niña) tropical events during the 1950-1977 period compared to the present (1977-1990) imbalance marked by a greater number of Warm (El Niño) events.

Several datasets we examined dated back only to the early 1950s. While not capable of demonstrating a shift in the early 1950s they do indicate a similarity of conditions for the 1950-1976 period. Between 1949 and 1976, Emery and Hamilton (1985) classified 22 of 28 north Pacific sea level pressure patterns as either weak or near normal. Hollowed and Wooster (1991) identified 24 of 31 winter atmospheric circulation patterns between 1946 and 1976 as Type A regimes (cool periods).

Potential Mechanisms

Establishing the mechanism whereby salmon production is driven by large scale climate processes can only be highly speculative at present. We alluded earlier to the general inability of most studies to establish predictable relationships between environmental variables and salmon survival and production that stand the test of time. Quinn and Marshall (1989), for example, found that inclusion of air and water temperature and freshwater discharge provided limited improvement to their time series models of southeast Alaska salmon variability.

At least two speculative mechanisms have been advanced to help explain the mid 1970s rise in Alaska salmon production. Rogers (1984) proposed that the increase in catch derived from increased marine survival of migrating salmon in their last winter at sea. Anomalously warm surface temperatures in the Gulf of Alaska altered both the migration paths and timing of returning salmon thus lessening their vulnerability to predators (principally marine mammals). Additional evidence for this hypothesis is provided by the 1970s and 1980s decline in northern fur seal (*Callorhinus ursinus*) and Steller's sea lion (*Eumetopias jubatus*) (Merrick et al. 1987, York 1987).

The second mechanism relates improved feeding conditions in the Alaska Current and Alaska Gyre to increased salmon production. Brodeur and Ware (1992) documented a two fold increase in zooplankton biomass between the 1950s and 1980s in the subarctic Pacific Ocean. They suggest that the primary beneficiaries of the elevated zooplankton biomass are juvenile salmon that migrate around the coastal margin of the CSD foraging upon zooplankton advected onto the oceanic shelf. Transport of zooplankton-rich waters derives from increased flow into the Alaska Current from the Subarctic Current (Pearcy 1992). Chelton (1984) has proposed that transport into the California and Alaska Currents fluctuates out of phase. This scenario suggests that the observed decrease in west coast salmon production may be due to poor feeding conditions resulting from decreased advection into the California Current (Pearcy 1992). Francis and Sibley (1991) illustrated opposite trends in production between Gulf of Alaska pink salmon and west coast coho salmon. The nature of the transitions from high (low) to low (high) production in both stocks suggests a single cause.

Perhaps the most interesting feature of the salmon regimes we have identified is the nature of the level of persistence exhibited by the different stocks. Hollowed and Wooster (1991) found synchronous recruitment patterns in several groundfish species corresponding to switches between Type A and Type B regimes. Strong year classes apparently derived from the onset of Type B regimes. Subsequent year classes, however, were much smaller. This appears to be quite different from the situation we have documented for Alaska salmon. In addition, the average duration of Type A and B regimes was 7-10 yrs, whereas we have identified much longer period regimes based on Alaska salmon dynamics. This suggests that different components of the North Pacific large marine ecosystem respond to different scale forcing factors.

Little is known about what causes low frequency shifts in thestructure and dynamics of large marine ecosystems. Margalef (1986) challenges us to develop a new paradigm in this regard. He suggests that infrequent and discontinuous changes in external (physical) energy are the most important factors affecting fluctuations in the biological production of these systems. These inputs, which he refers to "kicks", disrupt established ecological relationships within an ecosystem. Dr. John Steele (Woods Hole Oceanographic Institution, Woods Hole, MA 02543) puts it another way. He feels that, in the ocean, the variances of biological processes which respond to both physical and biological forcings are inversely proportional to their frequencies. If the variance of a process is forced beyond certain bounds or tolerances, that part of the system snaps, such as when an earthquake occurs, forcing repercussions throughout the ecosystem. As in the case of conditions leading up to an earthquake, many system variables which "snap" at the time of the earthquake demonstrate no aberrant behaviors prior to the occurrence of the earthquake itself. So, perhaps, it is with large marine ecosystems.

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APPENDIX

Time series model diagnostic and selection criteria.

Box-Pierce Portmanteau test

The joint null hypothesis H_o: $\rho_1 = \rho_2 = ... = \rho_K = 0$ is tested with the statistic

$$(1) \quad Q = n(n+2)\sum_{k=1}^{K}\frac{1}{n-k}\hat{\rho}^2(k)$$

The hypothesis of white noise is rejected if $Q > \chi^2_{\alpha,K-m}$, where K is the number of residuals calculated from the model and m is the number of estimated parameters.

Mean Absolute Error (MAE)

$$(2) \quad \mathrm{MAE} = \frac{1}{K}\sum_{k=1}^{K}|a_k|$$

unbiased residual variance σ^2_a,

$$(3) \quad \sigma^2_a = \frac{RSS}{K-m}$$

where RSS is the residual sum of squares and m is the number of estimated model parameters

coefficient of determination r^2

$$(4) \quad r^2 = 1 - \frac{RSS}{\sum(z-\bar{z})^2}$$

where z represents the (possibly) transformed and differenced observed values.

Akaike's Information Criterion (AIC)

$$(5) \quad \mathrm{AIC} = K\ln\tilde{\sigma}^2_a + 2m \qquad \text{where } \tilde{\sigma}^2_a = \frac{RSS}{K}$$

where RSS is the residual sum of squares, K is the number of residuals, m is the number of estimated parameters and $\tilde{\sigma}^2_a$ is the biased residual variance.

Schwarz Bayesian Criterion (SBC)

$$(6) \quad K\ln\tilde{\sigma}^2_a + m\ln K$$

where the parameters have the same interpretation as for the AIC.

EVALUATING CHINOOK SALMON ESCAPEMENT GOALS IN SOUTHEAST ALASKA AND TRANSBOUNDARY RIVERS

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ABSTRACT

In the mid-1970's the majority of chinook salmon stocks in the Southeast Alaska region were depressed relative to historical levels of production. As a result, a fisheries management program was implemented to rebuild depressed stocks of chinook salmon in southeast Alaska and transboundary rivers (rivers that originate in Canada and flow into southeast Alaska coastal waters). This program was formalized and expanded in 1981 to a 15-year (roughly 3 life-cycles) rebuilding program for the transboundary Taku, Stikine, Alsek, Unuk, Chickamin, and Chilkat rivers and the non-transboundary Blossom, Keta, Situk, and King Salmon Rivers. The objective of this program, which included regionwide, all-gear catch ceilings for chinook salmon, was to rebuild spawning escapements to goals by 1995. Interim goals were based on past observed levels of escapement in index tributaries multiplied by expansion factors to account for the proportion of spawners thought to be observed in surveys (survey expansion factor SEF) and the proportion of the entire system represented by the index tributary (tributary expansion factor TEF). The expansion factors were based largely upon professional judgement of the biologists familiar with the watersheds.

To assess the success of the rebuilding program requires evaluation of both the interim escapement goals and the escapement estimation methodology. Escapement goals based on spawner/recruit analysis require a long term database consisting of escapement and total return numbers by age class. ADF&G and DFO are compiling this data for most of the major chinook stocks in Southeast Alaska. Three examples are discussed, illustrating various levels of precision in escapement goal estimation. In one case, the Situk River, the database is complete enough that the escapement goal was revised downward in 1991, from 2,100 to 600 large spawners. A weir provides an accurate total escapement estimation for the Situk River and the harvest is small and accounted for. Total weir counts are not available for the Taku or Unuk Rivers and the escapement goals must be evaluated by less direct means. These consist of weir counts on major tributaries, aerial surveys, foot surveys, tagging studies and politics. The harvest rates of some stocks have been estimated by coded-wire tagging, while others are based only on limited tagging, hatchery exploitation rates or professional judgement.

Total escapement estimates based on survey data are evaluated by comparison with known escapements through weirs and tagging studies. Tributary expansion factors can be determined by tagging studies or radio-tagging distribution.

INTRODUCTION

Populations of chinook salmon are known to occur in some 34 river systems throughout Southeast Alaska, northwestern British Columbia, and the Yukon Territory, Canada (Figure 1). In the mid-1970's it became apparent

MINOR PRODUCERS	
Several hundred or less	
33	Martin
30	Big Goat
29	Rudyard
28	Walker
26	Klahine
24	Grant
23	Herman
22	Anan
21	Eagle
18	Tom
17	Aaron
15	Muddy
14	Farragut
13	Chuck
12	King Salmon
9	Dohn
8	East
6	Akwe
5	Italio
4	Dangerous
2	Lost
1	Ankau

MAJOR PRODUCERS	
10,00 or more in run	
16	Stikine
11	Taku
7	Alsek

MEDIUM PRODUCERS	
500 - 5,000	
32	Keta
27	Chickamin
25	Unuk
19	Harding
10	Chilkat
3	Situk
20	Bradfield
31	Wilson-Blossom

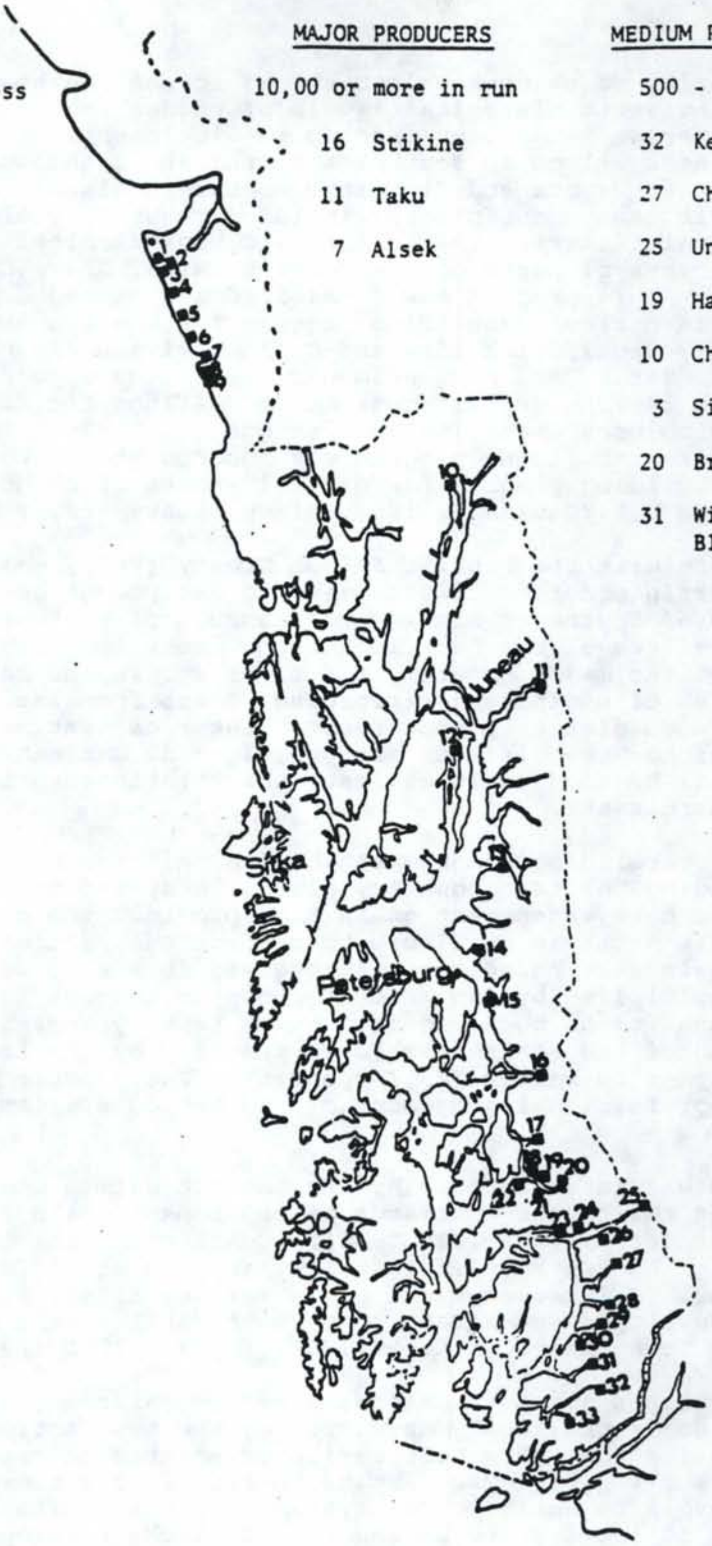

Figure 1. Chinook salmon systems in Southeastern Alaska.

that the majority of chinook salmon stocks in the Southeast Alaska region were depressed relative to historical levels of production (Kissner 1974). As a result, a fisheries management program was implemented to rebuild depressed stocks of chinook salmon in southeast Alaska and transboundary rivers (rivers that originate in Canada and flow into southeast Alaska coastal waters). Initially, this management program included regulatory closures of commercial and recreational fisheries in terminal and near-terminal areas and aerial escapement surveys of parts of the Taku, Stikine, Unuk, Chickamin, Keta and Wilson Rivers. This program was formalized and expanded in 1981 to a 15-year (roughly 3 life-cycles) rebuilding program for the transboundary Taku, Stikine, Alsek, Unuk, Chickamin, and Chilkat rivers and the non-transboundary Blossom, Keta, Situk, and King Salmon rivers. The objective of this program, which included regionwide, all-gear catch ceilings for chinook salmon, was to rebuild spawning escapements to interim goals by 1995. Then, in 1985, the southeast Alaska rebuilding program was incorporated into a broader, coastwide, rebuilding program for natural stocks of chinook salmon under the auspices of the U.S./Canada Pacific Salmon Treaty (PST).

In accordance with the Pacific Salmon Treaty (PST), escapement indices are used to ascertain progress towards meeting escapement goals for the chinook salmon stocks of Southeast Alaska and transboundary rivers. The Joint Chinook Technical Committee of the Pacific Salmon Commission combines the indices of escapements of the major, medium, and minor stocks and makes expansions to total estimates of escapements according to set formulas. These expansions are compared with similarly constructed historical estimates of escapement and appropriate fishery regulations are promulgated. Estimates of escapements by brood year will be used to investigate the relationship between spawners and subsequent recruitment.

The initial rebuilding program established minimum escapement goals for 9 systems including the transboundary Alsek, Taku, and Stikine Rivers. Although the aim was to have escapement goals that provided the optimal level of harvest, little data was available to produce such estimates. As a result, escapement goals were based on past observed levels of escapement in index tributaries multiplied by expansion factors to account for the proportion of spawners thought to be observed in surveys (survey expansion factor SEF) and the proportion of the entire system represented by the index tributary (tributary expansion factor TEF)(Appendix). The expansion factors were based largely upon professional judgement of the biologists familiar with the watersheds.

The program was later adopted by the Pacific Salmon Commission,and each Party provided their best estimates of escapement goals for each of the three major transboundary rivers (CTC 1986). Considering the limited data, it is not surprising the two Parties initially arrived at different escapement goals for each river. Joint escapement goals for the Alsek, Stikine, and Taku Rivers were developed and agreed to by both parties at a meeting of the Transboundary Technical Committee (TTC), in 1991 (TTC 1991).

The TTC developed joint recommendations for chinook escapement goal levels based on the goals previously developed by the two Parties. Escapement goals are developed for the index tributaries enumerated on each river rather than for the entire river systems. Expansion factors for converting index escapement levels to entire river system levels are often based on very little hard data and do not improve assessment of reaching escapement goals. The Chinook Technical Committee (CTC) has used expansion factors on index tributaries such that terminal catches could be added to the expanded escapement allowing total return to the system to be examined. However, in these three transboundary systems, terminal catches in relation to the escapement levels are very small and do not add much information beyond what the escapement data tells us of rebuilding.

In 1987, ADF&G calculated escapement goal ranges for 10 of the 11 index systems based on spawner-recruit analysis. A range of goals resulted from the use of two assumed life cycles (5 and 6 year) and three different exploitation rates (0, .2, .4). In nearly every case the estimates were much lower than the goals established in 1981. Spawner-recruit calculations using noisy data or bad assumptions often result in low escapement goal estimates (Hilborn, 1983).

Escapement goals estimated by spawner-recruit analysis are only as good as the database used. A good database must include or account for precise estimates of annual catches and escapements and precise estimates of the age composition. The database must include several lifecycles of the stock in question.

Three chinook stocks will be discussed as examples of the variation in the quality and duration of the database. Derivation of the escapement goals for the three stocks will be documented and weaknesses and strengths will be discussed.

METHODS

Of the 34 river systems with documented spawning populations of wild chinook salmon, three, the transboundary Taku, Stikine, and Alsek, are classified as major producers of chinook salmon with total run sizes potentially exceeding 10,000 fish. Nine systems are considered medium producers with run sizes between 1,500 and 10,000 fish. The remaining 22 rivers are placed in the minor production category with run sizes less than 1,500 chinook salmon. Although chinook salmon have been observed in small numbers in other southeast Alaska streams, successful spawning has not been documented.

Many index areas in the known chinook salmon spawning streams are surveyed annually to document escapements and to expand the database for southeast Alaska. Results from three major, seven medium, and one minor producing system are used to calculate an index of abundance for all southeast Alaska chinook salmon spawning streams. A detailed description of survey areas and spawning distribution in index tributaries can also be found in Mecum and Kissner (1989).

Since 1973 peak escapement surveys have been conducted from a Bell 206 or Hughes 500 helicopter during periods of peak spawning. In addition, weirs are operated on several systems and foot surveys are conducted where flying isn't possible. Peak spawning times, defined as the period when the largest number of adult chinook salmon actively spawn in a particular stream or river, are well documented from previous surveys.

An attempt is made to survey each of the index areas at least twice unless turbid water or unsafe flying conditions interfere. Pilots are directed to fly the helicopter from 6 to 15 meters above the river bed at a speed of 6 to 16 kilometers per hour. The helicopter door on the side of the observer is removed and the helicopter is flown sideways while observations of spawning chinook salmon were made from the open space. Only large (age 1.3, 1.4 and 1.5) chinook salmon are counted during aerial or foot surveys. No attempt is made to accurately count small age 1.1 and 1.2 chinook salmon (Mecum 1990). These small chinook salmon, also called jacks, are early maturing, precocious males that were considered to be surplus to spawning escapement needs. Male chinook salmon less than 2.3 kg (5 lbs) or less than 71 cm (28 inches) are considered jacks. These small chinook salmon are easy to visually separate from their larger age 1.3 and 1.4 counterparts under most conditions, due to their short, compact body configuration and lighter coloration. In early surveys it is not always clear whether the counts include jacks or not;

however, all escapements reported here are assumed to be adults and the escapement goals presented refer to adult returns.

DESCRIPTION OF STUDY AREAS

Situk River

The Situk River is located approximately 16 km east of Yakutat, Alaska (Figure 2). A weir was operated on the Situk River at the upper limit of the inter-tidal area from 1928 to 1955 to enumerate all five species of Pacific salmon spawning in the river. From 1957 to 1974, escapement was estimated from peak aerial surveys and float trips (Table 1). Since 1976, ADF&G has operated a weir each year, primarily to enumerate chinook and sockeye salmon.

Taku River

The Taku River originates in northern British Columbia and flows into the ocean 48 km east of Juneau, Alaska (Figure 3). The Taku River drainage covers over 16,000 km^2 and flows range from 787 to 2,489 m^3/sec. The drainage above the abandoned mining community of Tulsequah, British Columbia remains in pristine condition without any mining, logging, or other development activities. All of the important chinook salmon spawning areas in the Taku River are in upper tributaries in British Columbia. These include the Nakina, Nahlin, Dudidontu, Tatsamenie, Hackett, and Kowatua rivers and Tseta Creek.

Foot surveys of the Nakina River were conducted from 1951 through 1955 and during 1972 to enumerate and sample spawning chinook. From 1956 to 1959 and since 1973 a carcass weir was operated below the area containing the most concentrated chinook spawning in the drainage. The carcass weir does not provide a complete count but can be used as an index of abundance. From 1960 to 1971 intermittent aerial surveys with fixed wing aircraft were made. Factors affecting the comparability of these surveys include flying conditions, water conditions, missing the peak of spawning and questionable species composition. Since 1973 peak escapement surveys have been conducted from a helicopter during periods of peak spawning (Table 2). For the Nakina River the peak survey period is from August 1 to August 7 (Kissner 1982).

The Nahlin River is the second major clearwater chinook salmon spawning tributary of the Taku River. A foot survey in 1951 documented the importance of the Nahlin River for chinook spawning. Enumeration of the chinook salmon escapement to the Nahlin River has been conducted intermittently since 1951. Since 1974 helicopter surveys have been conducted each year during peak spawning.

The Tatsamenie River flows from the Tatsamenie Lakes system into Tatsatua Creek. This system also had fixed wing aerial surveys from 1965 to 1974 and since 1976 helicopter surveys during the peak spawning periods. The peak spawning period on the Tatsamenie is from August 20-26, the latest of any system in the Taku.

The Kowatua is a glacial stream that flows from the outlet of Little Trapper Lake into the Inklin River. The survey history of the Kowatua is similar to that of Tatsamenie, with fixed wing surveys from 1965 to 1974 and helicopter surveys since 1976. Visibility varies greatly with the amount of glacial melt and many sockeye salmon may be in the river at the same time. Peak spawning period is from about August 18 through 24.

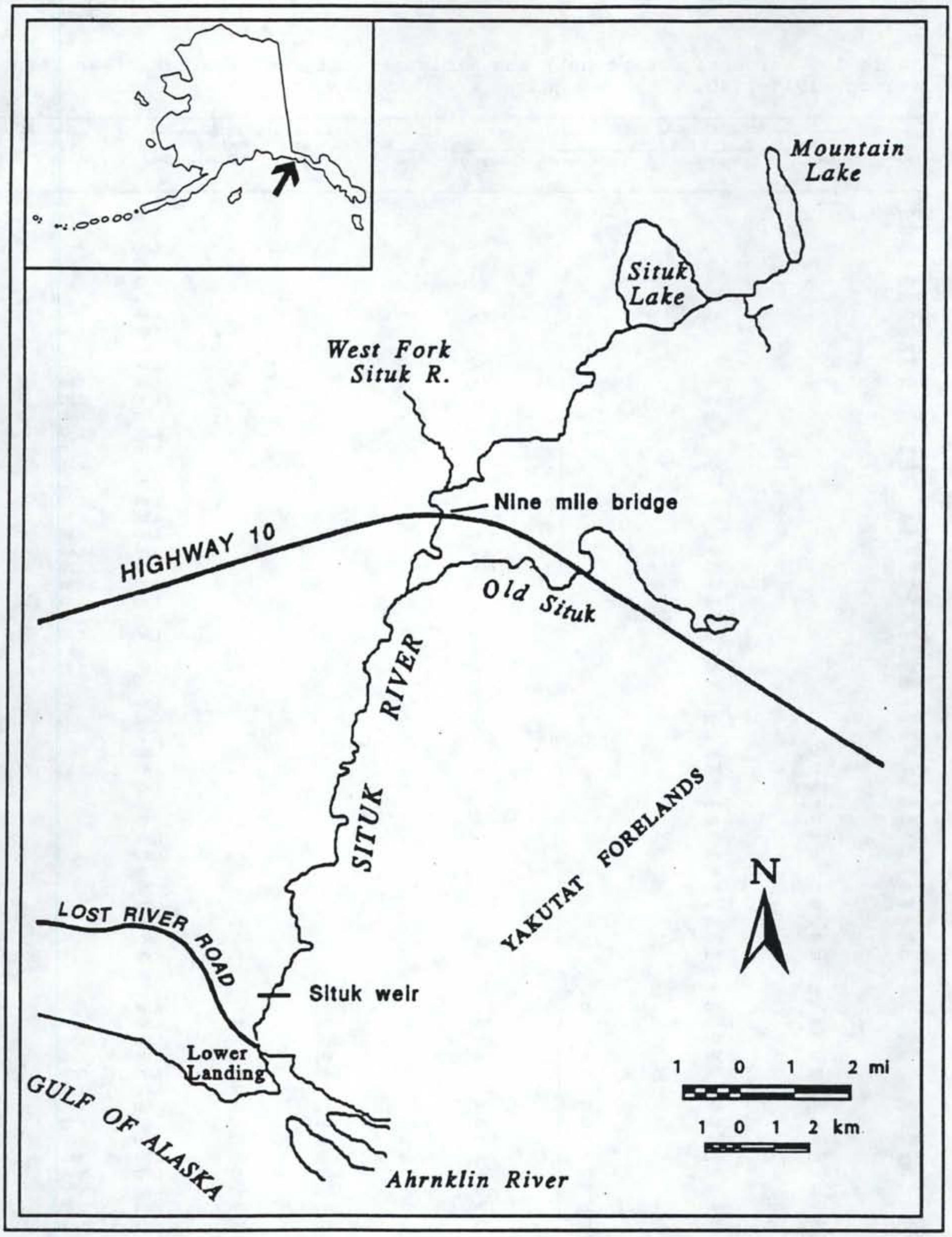

Figure 2. Situk River drainage, northern Southeast Alaska.

Table 1. Harvest, escapement, and minimum total run of Situk River chinook salmon, 1915-1990.

	Commercial Chinook Harvests					Escapement			Total Run Size[a]	
	Yakutat	Situk River		Recreational		Large	Small		Large	All
Year	Bay	Commercial	Subsistence	Large	Small	Chinook	Chinook	Total	Only	Chinook
1928	-	-	-	-	-	-	-	1,224	-	1,224
1929	-	-	-	-	-	-	-	3,559	-	3,559
1930	-	-	-	-	-	-	-	1,455	-	1,455
1931	-	-	-	-	-	-	-	2,967	-	2,967
1932	-	-	-	-	-	-	-	1,978	-	1,978
1933	-	267	-	-	-	-	-	-	-	267
1934	-	450	-	-	-	-	-	1,486	1,936	1,936
1935	-	558	-	-	-	-	-	638	1,196	1,196
1936	-	-	-	-	-	-	-	816	-	816
1937	-	-	-	-	-	-	-	1,290	-	1,290
1938	-	1,220	-	-	-	-	-	2,668	3,888	3,888
1939	-	495	-	-	-	-	-	2,117	2,612	2,612
1940	-	164	-	-	-	-	-	903	1,067	1,067
1941	-	390	-	-	-	-	-	2,594	2,984	2,984
1942	-	430	-	-	-	-	-	2,543	2,973	2,973
1943	-	947	-	-	-	-	-	3,546	4,493	4,493
1944	-	844	-	-	-	-	-	2,906	3,750	3,750
1945	-	692	-	-	-	-	-	1,458	2,150	2,150
1946	-	1,468	-	-	-	-	-	4,284	5,752	5,752
1947	-	885	-	-	-	-	-	5,077	5,962	5,962
1948	-	694	-	-	-	-	-	3,744	4,438	4,438
1949	-	410	-	-	-	-	-	1,978	2,388	2,388
1950	-	378	-	-	-	-	-	2,011	2,389	2,389
1951	-	948	-	-	-	-	-	2,780	3,728	3,728
1952	-	225	-	-	-	-	-	1,459	1,684	1,684
1953	-	378	-	-	-	-	-	1,040	1,418	1,418
1954	-	314	-	-	-	-	-	2,101	2,415	2,415
1955	-	740	-	-	-	-	-	1,571	2,311	2,311
1956	-	1,867	-	-	-	-	-	-	-	1,867
1957	-	1,796	-	-	-	-	-	1,500	-	3,296
1958	-	187	-	-	-	-	-	300	-	487
1959	-	426	-	-	-	-	-	-	-	426
1960	24	312	-	-	-	-	-	500	-	812
1961	28	367	-	-	-	-	-	400	-	767
1962	99	337	-	-	-	-	-	1,000	-	1,337
1963	141	466	-	-	-	-	-	-	-	466
1964	115	706	-	-	-	-	-	725	-	1,431
1965	86	442	-	-	-	-	-	1,500	-	1,942
1966	43	411	-	-	-	-	-	800	-	1,211
1967	241	203	-	-	-	-	-	200	-	403
1968	31	312	-	-	-	-	-	700	-	1,012
1969	29	1,089	-	-	-	-	-	2,500	-	3,589
1970	119	927	-	-	-	-	-	1,100	-	2,027
1971	106	473	-	-	-	-	-	964	-	1,437
1972	115	303	-	-	-	-	-	400	-	703
1973	79	752	-	-	-	-	-	510	-	1,262
1974	64	791	-	-	-	-	-	702	-	1,493
1975	41	562	27	-	-	-	-	1,180	-	1,769
1976	69	1,002	41	200	-	1,433	509	1,942	2,676	3,185
1977	53	833	24	244	-	1,732	148	1,880	2,833	2,981
1978	108	382	50	210	-	814	289	1,103	1,456	1,745
1979	51	1,028	25	282	-	1,400	367	1,767	2,735	3,102
1980	164	969	57	353	-	905	220	1,125	2,284	2,504
1981	151	858	62	130	-	702	105	807	1,752	1,857
1982	419	248	27	63	0	434	177	611	772	949
1983	371	349	50	42	10	592	257	849	1,033	1,300
1984	145	512	50	146	5	1,726	475	2,201	2,434	2,914
1985	240	484	81	294	217	1,521	461	1,982	2,380	3,058
1986	211	202	87	0	37	2,067	505	2,572	2,356	2,898
1987	329	891	22	76	319	1,884	494	1,884	2,873	3,192
1988	196	299	81	185	3	885	193	1,078	1,450	1,646
1989	297	1	29	0	0	652	1,217	1,869	682	1,899
1990	304	0[b]	na	0	0	700	631[c]	1,331	923	1,741

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[a] Total run = chinook escapement + Situk commercial, sport, and subsistence harvests.

[b] Non-retention regulation in effect for commercial fisheries in 1989 and 1990. Estimated harvest of 223 large chinook in 1990.

[c] Small chinook includes 486 medium fish (>450mm <660mm MEF).

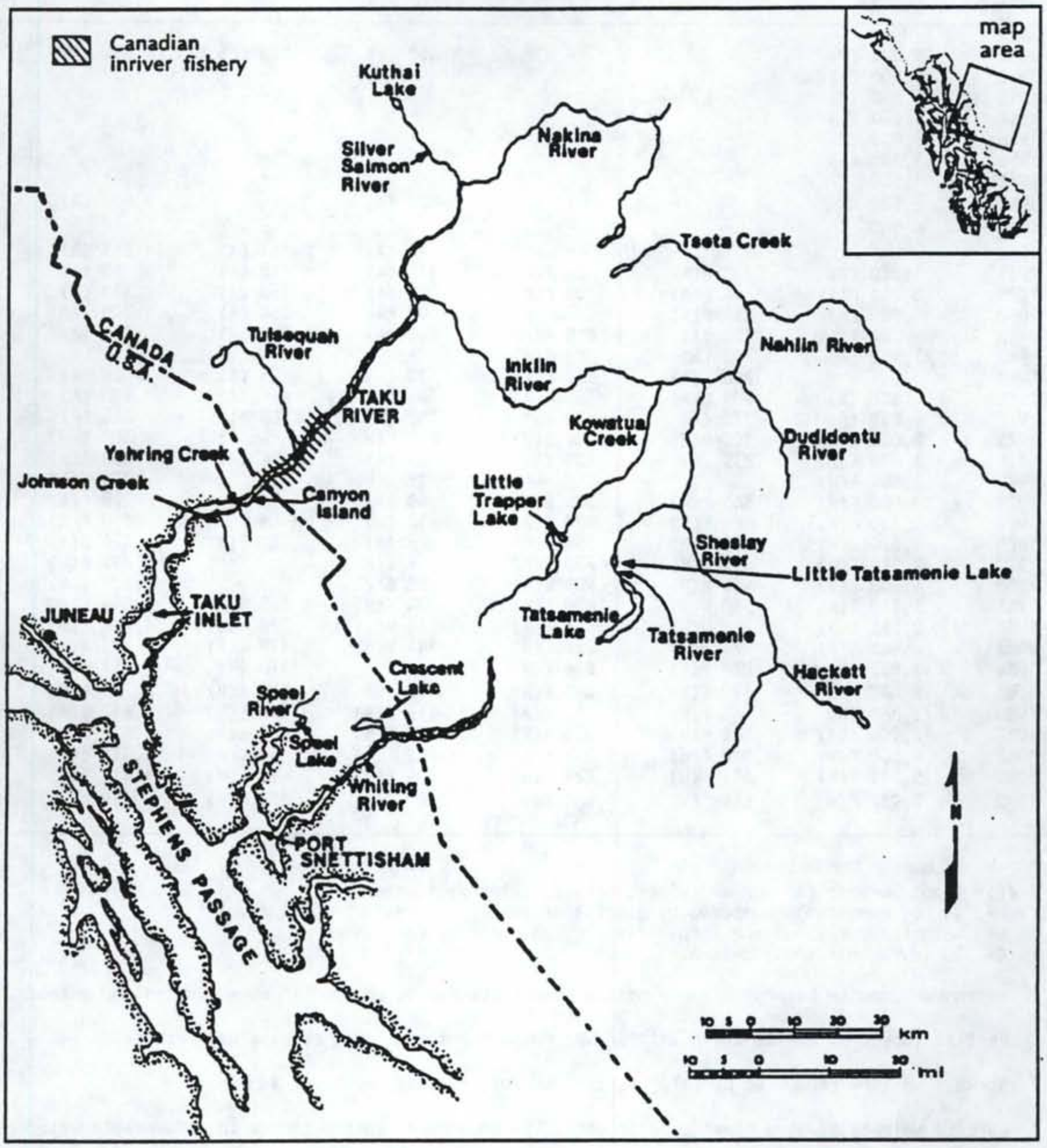

Figure 3. Taku River drainage, northwestern British Columbia and Southeast Alaska.

Table 2. Index escapement counts of chinook salmon for tributaries of the Taku River, 1951 to 1990.[a]

Year	Nakina River	Kowatua River	Tatsamenie River	Dudidontu River	Tseta Creek	Nahlin River	Total
1951	5,000 (F)	-	-	400 (F)	100 (F)	1,000 (F)	6,500
1952	9,000 (F)	-	-	-	-	-	9,000
1953	7,500 (F)	-	-	-	-	-	7,500
1954	6,000 (F)	-	-	-	-	-	6,000
1955	3,000 (F)	-	-	-	-	-	3,000
1956	1,380 (F)	-	-	-	-	-	1,380
1957	1,500	-	-	-	-	-	1,500[c]
1958	2,500	-	-	4,500 (A)	-	2,500 (A)	9,500[c]
1959	4,000	-	-	-	-	-	4,000[c]
1962	-	-	-	25 (A)	81 (A)	216 (A)	322
1965	3,050 (H)	200 P(A)	50 P(A)	110 (A)	18 (A)	35 (A)	3,463
1966	3,700 P(A)	14 P(A)	100 P(A)	252 (A)	150 (A)	300 (A)	4,516
1967	700 (A)	250 P(A)	-	600 (A)	350 (A)	300 P(A)	2,200
1968	300 P(A)	1,100 (A)	800 E(A)	590 (A)	230 (A)	450 (A)	3,470
1969	3,500 (A)	3,300 (A)	800 E(A)	-	-	-	7,600
1970	-	1200 P(A)	530 E(A)	10 (A)	25 (A)	26 (A)	1,791
1971	500 (A)	1,400 E(A)	360 E(A)	165 (A)	- (A)	473 (A)	2,898
1972	1,000 (F)	170 (A)	132 (A)	102 (A)	80 P(A)	280 (A)	1,764
1973	2,000 N(H)	100 N(H)	200 E(H)	200 E(H)	4 (A)	300 E(H)	2,804
1974	1,800 E(H)	235 (A)	120 (A)	24 (A)	4 (A)	900 E(H)	3,083
1975	1,800 E(H)	-	-	15 N(H)	-	274 E(H)	2,089
1976	3,000 E(H)	341 P(A)	620 E(H)	40 (H)	-	725 E(H)	4,726
1977	3,850 E(H)	580 E(H)	573 E(H)	18 (H)	-	650 E(H)	5,671
1978	1,620 E(H)	490 N(H)	550 E(H)	- (H)	21 E(H)	624 E(H)	3,305
1979	2,110 E(A)	430 N(H)	750 E(H)	9 E(H)	-	857 E(H)	4,156
1980	4,500 E(H)	450 N(H)	905 E(H)	158 E(H)	-	1,531 E(H)	7,544
1981	5,110 E(H)	560 N(H)	839 E(H)	74 N(H)	258 N(H)	2,945 E(H)	9,786
1982	2,533 E(H)	289 N(H)	387 N(H)	130 N(H)	228 N(H)	1,246 E(H)	4,813
1983	968 E(H)	171 E(H)	236 E(H)	117 E(H)	179 N(H)	391 N(H)	2,062
1984	1,887 (H)	279 E(H)	616 E(H)	-	176 (H)	951 (H)	3,909[d]
1985	2,647 N(H)	699 E(H)	848 E(H)	475 (H)	303 E(H)	2,236 E(H)	7,208
1986	3,868 (H)	548 E(H)	886 E(H)	413 E(H)	193 E(H)	1,612 E(H)	7,520
1987	2,906 E(H)	570 E(H)	678 E(H)	287 E(H)	180 E(H)	1,122 E(H)	5,743
1988	4,500 E(H)	1,010 E(H)	1,272 E(H)	243 E(H)	66 E(H)	1,535 E(H)	8,626
1989	5,141 E(H)	601 (W)	1,228 E(H)	204 E(H)	494 E(H)	1,812 E(H)	9,480[e]
1990	7,917 E(H)	614 (W)	1,068 N(H)	820 E(H)	172 N(H)	1,658 E(H)	12,249[e]

[a] -= No Survey Conducted
(F)= Foot Survey; (A) = Fixed-wing aircraft; (H) = Helicopter
P= Survey conditions hampered by glacial or turbid waters
N= Normal water flows and turbidities; average survey conditions
E= Survey conditions excellent

[b] Escapement counts before 1975 may not be comparable due to changes in survey dates and methods.

[c] Partial survey of Nakina River in 1957-59; comparisons made from carcass weir counts.

[d] Surveys in 1984 conducted by CDFO; partial survey of Tseta Creek and Nahlin.

[e] Carcass weir at Kowatua River used to partially enumerate escapement due to unfavorable water conditions.

The Dudidontu River and Tseta Creek are two tributaries of the Nahlin River with peak spawning periods of July 28 to August 4. Both systems had intermittent aerial surveys from 1958 to 1974. Peak helicopter surveys on the Dudidontu River began in 1975. In 1979 and each year since 1981 helicopter surveys of Tseta Creek have been conducted.

Unuk River

The Unuk, Chickamin, Blossom and Keta Rivers all flow into Behm Canal in southern Southeast Alaska and are all included in the Misty Fiords National Monument and Wilderness Area. The Unuk River originates in a heavily glaciated area of northern British Columbia and flows for 129 km to Burroughs Bay 85 km northeast of Ketchikan, Alaska; only the lower 39 km of the river are in Alaska (Figure 4). Most Unuk River chinook salmon spawn in tributaries in the lower 39 km of the U.S. portion of the river, including the Eulachon River and Cripple, Genes Lake, Clear, Lake, and Kerr creeks.

Intermittent surveys of various tributaries of the Unuk River by foot, fixed wing and helicopter have been conducted from 1960 through 1976. Since 1977, helicopter and foot surveys have been conducted on 6 index areas of the Unuk River (Table 3). Cripple Creek and Gene's Lake Creek, two of the most important spawning areas on the Unuk River, can not be surveyed from the air because of heavy vegetation so the escapements are counted by foot surveys. Peak spawning periods range from August 4-9 for Cripple Creek, to August 7-14 for Clear, Lake and Kerr Creeks, to August 14-21 for Genes Lake and Eulachon Creeks (Table 4).

RESULTS AND DISCUSSION

Situk River

Situk River chinook salmon are harvested by subsistence, commercial, and sport fisheries. Most harvests occur in or near the river and are accurately accounted for by age. In 1991, a spawner-recruit analysis was completed using data from 1976 to 1984 brood years (McPherson 1991). The biologically optimum escapement goal was estimated to be 600 large spawners with a 99% confidence interval of 450 - 870 (Figure 5). This was a significant decrease from the previous goal of 2,100 fish. The Situk River has the most complete database for chinook salmon in Southeast Alaska. It is the only system with continuous long-term total escapement enumeration through a weir. It is one of only a few systems with complete accounting of harvest. Catch and escapement counts are available inseason and the fisheries can be adjusted rapidly. Age composition of the escapement is the only weakness in the database. It is difficult to collect scales from live chinook at the weir and most age information has come from small numbers of spawning ground samples.

Taku River

The escapement goal established by ADF&G in 1981 for Taku River chinook salmon was based on the largest escapement ever observed on the major spawning grounds of the Nakina River. A count of 9,000 fish was recorded there in 1952 and has never been surpassed. The Nakina was assumed to contribute 40% of the total Taku system and 75% of the escapement was assumed to be observed thus, the total minimum escapement goal was 9,000/0.40/0.75 = 30,000 fish (Figure 6). The Nahlin River was added to the indicator stock prior to 1985. The peak escapement of 2,500 in 1958 was combined with the Nakina to give a goal of 11,500 fish. The two tributaries were assumed to contribute 60% of the Taku River chinook salmon, thus the goal was reduced to 25,550 (11,500/0.60/0.75 = 25,555).

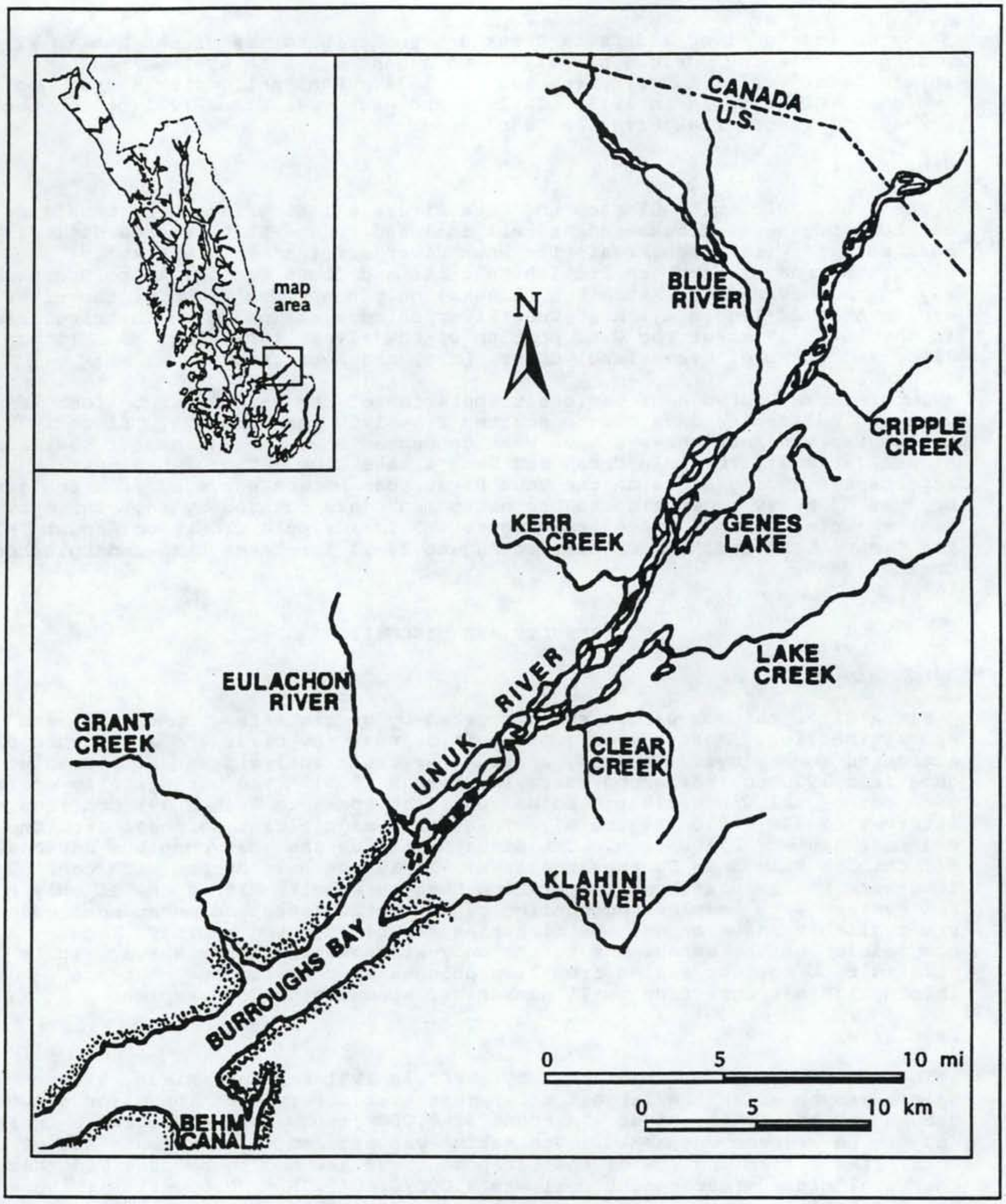

Figure 4. Unuk River drainage, northwestern British Columbia and southern Southeast Alaska.

Table 3. Proportion of the chinook salmon spawning in index areas in the Taku River as determined by distribution of radio-tagged fish, 1989 and 1990. Standard deviations are in parenthesis. (from J. Eiler, NMFS Auke Bay Lab, personal communication)

Stock	Percent of Run 1989	1990	Annual Variation
Nakina River	40.9 (2.9)	33.6 (3.7)	11.3
Kowatua Creek	2.7 (0.8)	0.8 (0.3)	1.8
Tatsatua Creek	2.6 (0.9)	1.0 (0.5)	1.3
Nahlin River	8.0 (1.5)	5.7 (1.7)	3.4
Dudidontu River[a]	0.1 (0.1)	2.1 (1.3)	1.7
Tseta Creek	1.1 (0.7)	0.4 (0.3)	0.3
Total	55.4	43.6	13.0

[a] Telemetry data suggest that fish movements through the lower portion of the river may have been restricted in 1989.

Table 4. Peak escapement counts of chinook salmon to index tributaries of the Unuk River, 1960-1990. [a,b]

Year	Cripple Creek	Genes Lake Creek	Eulachon Creek	Clear Creek	Lake Creek	Kerr Creek	Total
1960	-	-	250 (A)	-	-	-	250
1961	3 (F)	200 (F)	270 (F)	65 (F)	-	53 (F)	591
1962	-	150 (A)	145 (A)	100 (A)	30 (A)	-	425
1963	100 (A)	750 (A)	150 (A)	25 (A)	-	-	1,025
1964	-	-	25 (A)	-	-	-	25
1965	-	-	-	-	-	-	-
1966	-	-	-	-	-	-	-
1967	-	-	60 (H)	-	-	-	60
1968	-	-	75 (H)	-	-	-	75
1969	-	-	150 (H)	-	-	-	150
1970	-	-	-	-	-	-	-
1971	-	-	30 (A)	-	-	-	30
1972	95 (A)	35 (A)	450 (A)	90 (A)	55 (A)	-	725
1973	-	-	64 (H)	-	-	-	64
1974	-	-	68 (H)	-	-	-	68
1975	-	-	17 (H)	-	-	-	17
1976	-	-	3 (A)	-	-	-	3
1977	529 (F)	339 (F)	57 (H)	34 (H)	-	15 (H)	974
1978	394 (F)	374 (F)	218 (H)	85 (H)	20 (H)	15 (H)	1,106
1979	363 (F)	101 (F)	48 (H)	14 (H)	30 (H)	20 (H)	576
1980	748 (F)	122 (F)	95 (H)	28 (H)	5 (H)	18 (H)	1,016
1981	324 (F)	112 (F)	196 (H)	54 (H)	20 (H)	25 (H)	731
1982	538 (F)	329 (F)	384 (H)	24 (H)	48 (H)	28 (H)	1,351
1983	459 (F)	338 (F)	288 (H)	24 (H)	12 (H)	4 (H)	1,125
1984	644 (F)	647 (F)	350 (H)	113 (H)	32 (H)	51 (H)	1,837
1985	284 (F)	553 (F)	275 (H)	37 (H)	22 (H)	13 (H)	1,184
1986	532 (F)	838 (F)	486 (H)	183 (F)	25 (H)	62 (H)	2,126
1987	860 (F)	398 (F)	520 (H)	107 (H)	37 (H)	51 (H)	1,973
1988	1,068 (F)	154 (F)	146 (F)	292 (H)	60 (H)	26 (H)	1,746
1989	351 (F)	302 (F)	298 (H)	128 (H)	27 (F)	43 (H)	1,149
1990	86 (F)	284 (F)	81 (H)	103 (F)	26 (F)	11 (H)	591

[a] (F) = Escapement survey conducted by walking river
(A) = Escapement Survey conducted from fixed-wing aircraft
(H) = Escapement survey conducted from helicopter
\- = No survey conducted or data not comparable

[b] Escapement counts prior to 1975 may not be comparable due to differences in survey dates and counting methods.

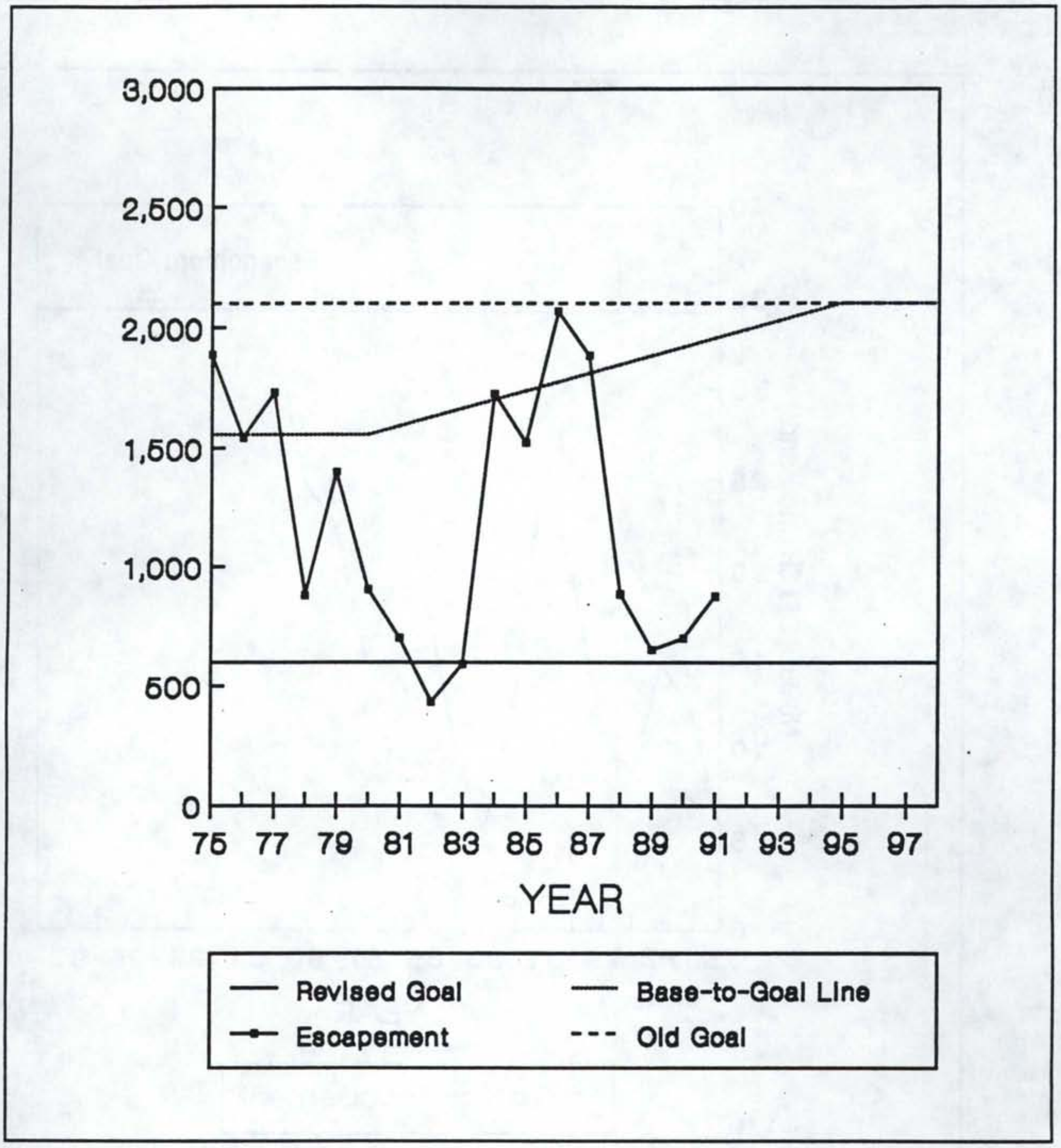

Figure 5. Estimated escapements of chinook salmon to the Situk River, 1975-1991. Base-to-goal line shows linear rebuilding trend, starting in 1981 at average escapement level during first cycle of rebuilding (1975-1980) and ending at management escapement goal of 2,100 large chinook salmon in 1995 (final year of the three-cycle rebuilding program).

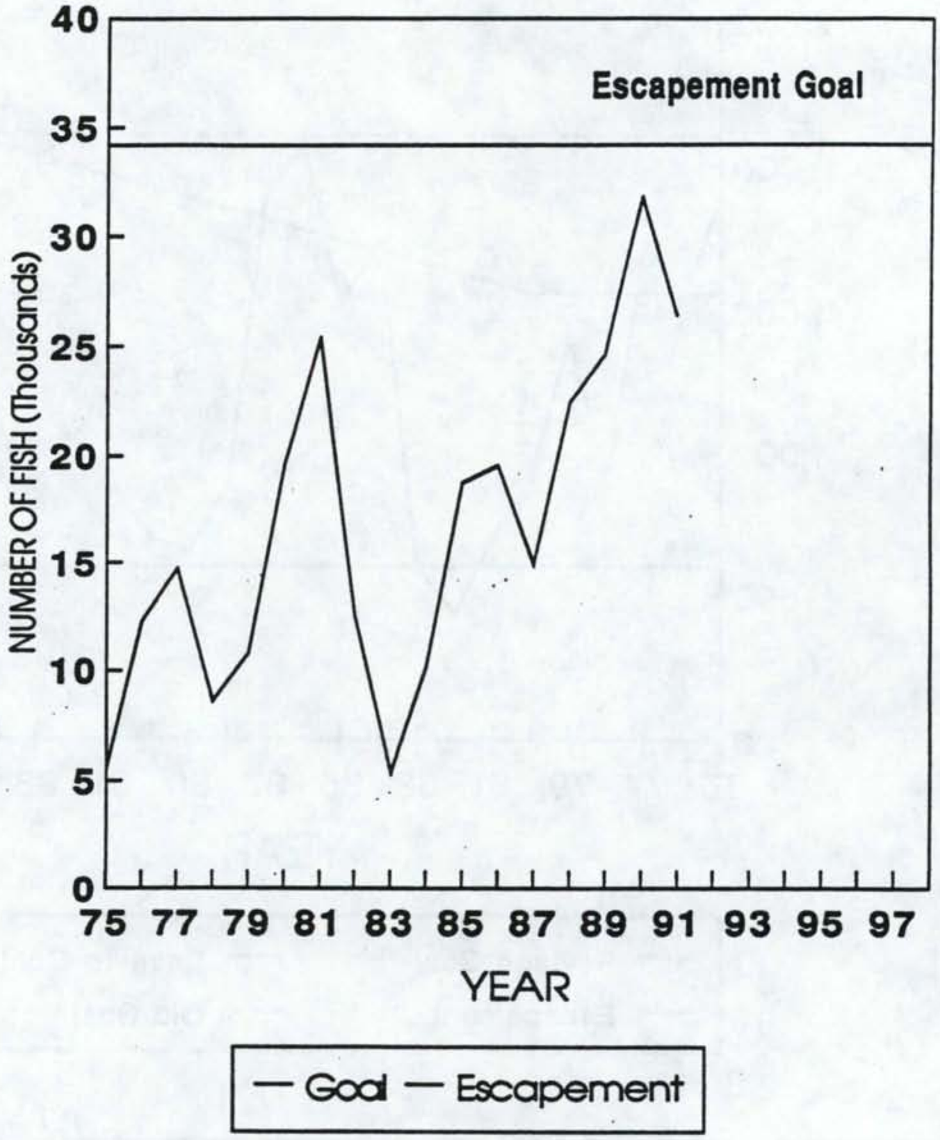

Figure 6. Estimated escapements of chinook salmon to the Taku River, 1975-1991.

Aerial surveys of escapement have been conducted fairly regularly on six index tributaries on the Taku River since 1965. Prior to 1991, the U.S. expanded counts from only the Nakina and Nahlin River index areas to estimate the escapement to the entire Taku River, while Canada expanded counts from all six index tributaries. The Canadian goal was 15,000 to the six index areas, which was then expanded by two to a total goal of 30,000 fish.

In October 1991, it was agreed by both Parties to use counts from all six tributaries when they are available. Not all tributaries are equally easy to survey and poor conditions could limit surveys in some years. In such cases counts of the surveyed tributaries will be expanded to represent the six tributaries based on the historical average proportions. A joint escapement goal of 13,200 for the combined counts of the six index tributaries was developed by summing each individual tributary's highest count between 1965 and 1981. This goal incorporates no expansion factors and refers to chinook actually observed on the surveyed tributaries.

Previously used expansion factors were not based on any scientific studies and differed between the two countries. A study conducted by the National Marine Fisheries Service on the Taku River using radio tagging of chinook salmon shows that the contribution of these six index tributaries to the entire Taku system was 55% in 1989 and 44% in 1990 giving an average of about 50% (Table 5). Expanding survey counts to reflect total numbers of fish in the streams surveyed is not so easy. The proportion observed of the spawning escapement of each tributary surveyed will vary from year to year depending on weather and tributary conditions. While it is noted on the recording sheets whether survey conditions were excellent, normal, or hampered, no estimates of annual variation in proportions counted is attempted. If conditions are very poor, escapement counts are not recorded. ADF&G survey biologists, based on stream characteristics, consider that about 75% of the fish present in the Nakina and Nahlin are counted while only 62.5% of the fish present in the other surveyed tributaries are counted. Since terminal catches are insignificant compared to escapement levels, the TBTC recommends that only escapement counts for the six index tributaries be used in assessing rebuilding. However, an expanded total escapement is still estimated for use by the CTC.

Mark/recapture studies conducted in 1989 and 1990 on returning adult chinook salmon estimated escapements of approximately 37,000 and 46,000 fish, over twice what was estimated using the old aerial survey expansion method.

Between 1977 and 1983, juvenile Taku River chinook salmon were captured and marked with coded-wire tags. Recovery of tagged adults was used to document migration patterns and harvest rates. Estimated harvest rates for 1981 - 1985 ranged from 9% to 32% (CTC 1990). Harvest rates are assumed to remain low due to restrictions on fisheries occurring in the migration corridor.

The 1987 spawner-recruit analysis which used expanded estimates of total escapement of Taku River chinook indicated an optimal escapement range of 4,100 to 8,500 3- and 4-ocean fish. The analysis only included age 3- and 4-ocean fish and looked at exploitation rates of 0, 20 and 40 %. Maximum r-squared value was only 0.46.

Analysis completed in 1991 included all ages and estimates of actual age composition and exploitation for the Nakina River. Two models were developed containing 11 data points corresponding to the return years 1980-1990; one included all spawners and another used only age 1.3 and older spawners. Optimum escapement estimated using all spawners was 2,800 large fish with r-squared of 0.69. In the model including only large spawners, optimum escapement was estimated to be only 1,400 large fish to the Nakina River, r-squared was only 0.27. The proportion of jacks in the escapement has varied greatly from year to year. The output of the model which included jacks is

Table 5. Comparison of weir counts and aerial survey estimates of chinook salmon escapements to the Little Tahltan River, 1985-1990.

Date	Weir Count	Low Level Helicopter Count	Percent of Escapement Observed From Helicopter
8/02/85	2,379	1,262	53.1
8/06/85	2,864	1,598	55.8
Final	3,146	1,598	50.8
8/01/86	2,323	1,101	47.4
8/05/86	2,646	1,143	43.2
Final	2,893	1,201	41.5
7/31/87	3,903	2,446	62.7
8/03/87	4,456	2,706	60.7
Final	4,781	2,706	56.6
7/30/88	5,573	3,484	62.5
8/05/88	6,822	3,796	55.6
Final	7,292	3,796	52.1
7/29/89	3,772	2,515	66.7
8/04/89	4,394	2,527	57.5
Final	4,715	2,527	53.6
7/31/90	3,780	1,658	43.8
8/07/90	4,232	1,678	39.7
Final	4,354	1,755	40.3
7/30/91	3,649	1,768	48.5
8/06/91	4,141	1,327	32.0
Final	4,506	1,768	39.2

driven by the 1978 escapement when over 8,000 1-ocean jacks made up 61 percent of the total escapement to the Nakina.

Unuk River

The Unuk River escapement goal of 1,800 was established in 1981 by using the maximum observed escapement between 1961 and 1980 which was 1,765 fish in 1978. The 1978 escapement count was revised in 1985 due to a double entry error. The resulting escapement was 1,106 fish, which is still the peak escapement prior to the rebuilding period. The survey goal remains 1,800 fish and has been met 4 times since 1984. The CTC expands the index count by 1/0.625 resulting in a total Unuk River goal of 2,880 (Figure 7).

Age composition of spawning Unuk River chinook has been estimated since 1985. Wild juvenile Unuk River chinook salmon were tagged with coded-wire tags from 1983 to 1988. Approximate exploitation rates have been estimated for brood years 1982-1986.

The 1987 spawner-recruit analysis using a range of assumed age composition and exploitation rates, estimated a range of optimum escapements of 900 to 1,200 age 3- and 4-ocean chinook salmon. Maximum r-squared values were 0.85 assuming a 5 year cycle and 20 to 60% exploitation rates. Additional spawner-recruit analysis will be attempted, however the errors in escapement estimation by age are probably too large for a useful relationship.

Expansion Factors

Escapement counts of selected index spawning areas are expanded by a factor which represents the estimated percentage of the total season escapement observed during the peak spawning period. These expansion factors vary according to the difficulties encountered in observing spawning chinook salmon due to overhanging vegetation, turbid water conditions, presence of other salmon species (i.e., pink sockeye, and chum salmon), or protraction of run timing. Survey expansion factors (SEF) range from 1/0.80 for Big Boulder and Stonehouse Creeks to 1/0.625 for most other systems (Appendix). Survey expansions are not necessary for those systems where upstream counting weirs are used to enumerate spawning chinook salmon. Peak aerial, foot, or weir counts are also expanded for the percentage of the total escapement to the entire drainage observed in index tributaries (i.e., not all tributaries or spawning areas were surveyed). Tributary expansion factors (TEF) range from 1/0.25 for the Stikine River to 1/0.64 for the Alsek River.

The expansion factors represent estimates whose validity is unknown for the majority of the index systems. The factors were arrived at in 1981 based on the subjective opinions of the researchers involved and on 5-10 years of experience with the index areas. In fact, comparison of aerial surveys with weir counts on some systems indicates the survey expansion factors for the larger systems may be too low. However, these expansion factors have been used since 1981 and have been adopted by the Joint Chinook Salmon Technical Committee (CTC) of the Pacific Salmon Commission. Therefore, a formal review of these index expansion methods by ADF&G, the Department of Fisheries and Oceans (DFO) and the CTC will be necessary before modifications can be made. The primary purpose of the index expansion program is to gauge progress in the rebuilding of chinook salmon stocks of the southeast Alaska region (Figure 8).

Weirs

Chinook escapement counts are also obtained from fish counting weirs operated by DFO on the Little Tahltan (Stikine), and Klukshu (Alsek) Rivers, and by ADF&G on the King Salmon River (Admiralty Island) and Situk River. Except for the Situk River, where aerial surveys were not practical due to overhanging vegetation, weir counts were compared with aerial or foot surveys

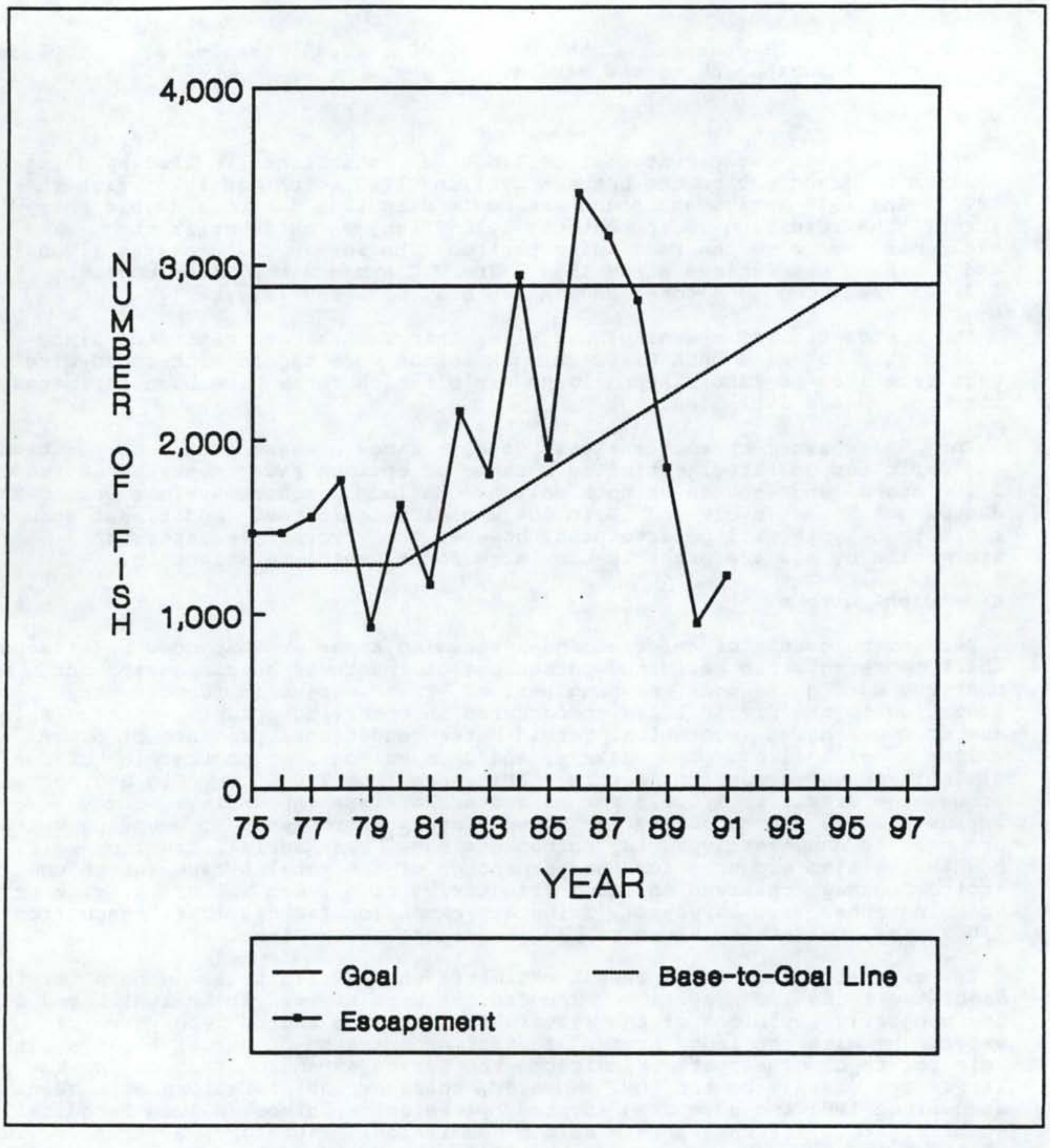

Figure 7. Estimated escapementsof chinook salmon to the Unuk River, 1975-1991. Base-to-goal line shows linear rebuilding trend, starting in 1981 at average escapement level during first cycle of rebuilding (1975-1980) and ending at management escapement goal of 2,880 large chinook salmon in 1995 (final year of the three-cycle rebuilding program).

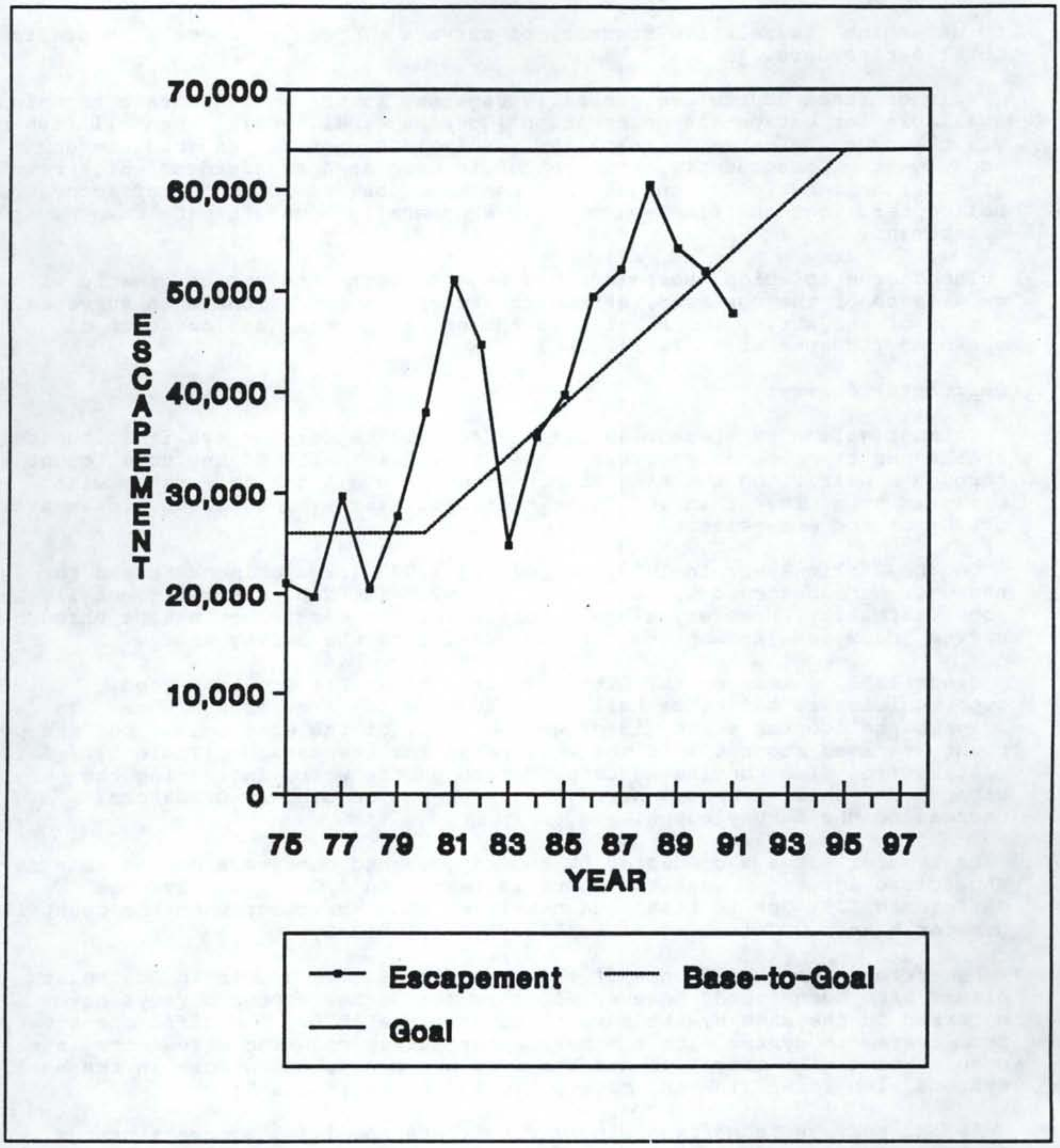

Figure 8. Estimated total escapement of chinook salmon to Southeast Alaska and transboundary river index systems, 1975-1991. Base-to-goal line shows linear rebuilding trend, starting in 1981 at average escapement level during first cycle of rebuilding (1975-1980) and ending at manaagement escapement goal of 64,000 large chinook salmon in 1995 (final year of the three-cycle rebuilding program).

to determine the relative accuracy of surveys of peak escapement in predicting total escapements.

Weir or fence counts are generally regarded as the most accurate technique available for escapement enumeration (Cousens et al 1982). When all fish pass via the counting gates or traps the result is supposedly an absolute count of escapement. Consequently, this method is used as a standard to which results of other enumeration techniques are compared for determination of accuracy. Helicopter, foot and fixed-wing surveys generally underestimate the escapement.

The degree to which escapement may be underestimated varies greatly with experience of the observer, stream character and conditions when surveyed, timing of the survey in relation to the spawning peak, and duration of spawning (Cousens et al 1982).

Helicopter Surveys

Chinook salmon escapement estimates from helicopter surveys in Southeast Alaska and transboundary rivers ranged from 27% to 85% of the total count through a weir . On the King Salmon River, a small coastal system with escapements of less than 300 fish; helicopter surveys accounted for an average of 70% of the escapement.

On the Nahlin River in 1990, 44% of the 1,773 large chinook passed through the weir were observed by helicopter survey. The Nahlin River generally has good visibility, however, a small portion of the escapement passes through the survey index area and spawns upriver outside of the survey area.

Since 1985, a weir on the Little Tahltan River has provided good opportunities to calibrate helicopter surveys of large escapements. In late July the helicopter survey is often above 60% of the weir count, but the peak count averages about 50% of the weir total for the season. (Table 5). This results from fish continuing to pass through the weir, increasing the escapement; while spawners are dying and being removed by predators, decreasing the survey count.

Helicopter surveys conducted by two experienced observers on the same day show close agreement when the count is less than 1,000 fish (average difference 1.9% or 10 fish) and nearly as good agreement when the count is greater than 1,000 fish (4.4% or 224 fish) (Table 6).

Few records of foot counts of chinook salmon above a weir in Southeast Alaska have been found, however, foot count and helicopter surveys have occurred on the same system several times since 1976. The differences vary from system to system with the helicopter survey counting a few more in the open systems with many fish and the foot surveys counting more in the small systems with fewer fish and more pools and trees (Table 7).

Aerial surveys from fixed wing aircraft are the least accurate of the methods used. Surveys of Andrew Creek averaged less than 30% of the known escapement through a weir (Table 8). Andrew Creek surveys are complicated by the presence of large numbers of other salmon species during the peak spawning period for chinook salmon.

Shardlow et al (1987) performed an experiment with chum salmon under the ideal survey conditions of clear shallow water and evenly distributed fish. They found an experienced observer counted 100% of the fish by helicopter survey, 93% by foot and 85% by fixed wing aerial survey. They also found that variation due to experience of the observer was much higher in both fixed wing and helicopter surveys than in foot and strip surveys. Use of strips, that is

Table 6. Variation in escapement counts by two different helicopter surveyors of chinook salmon in Southeast Alaska and transboundary rivers.

Date	River	First Count	Second Count	Percent Difference	Number
7/19/85	Nahlin	1334	1275	-4.4	-59
7/25/85	Nahlin	1626	1733	6.6	107
7/25/85	Dudidontu	52	49	-5.8	-3
8/01/85	Nahlin	833	811	-2.6	-22
8/01/85	Dudidontu	489	463	-5.3	-26
8/01/85	Tseta	309	296	-4.2	-13
8/02/85	Beatty Cr.	140	154	10.0	14
8/02/85	Little Tahltan	1262	1171	-7.2	-91
8/02/85	Nakina	1170	1129	-3.5	-41
8/26/85	Kowatua	665	732	10.1	67
8/26/85	Tatsamenie	730	791	8.4	61
7/24/87	Nahlin	621	643	3.5	22
7/24/87	Nahlin	160	169	5.6	9
7/24/87	Nahlin	341	267	-21.7	-74
7/31/87	Little Tahltan	2446	2234	-8.7	-212
7/31/87	Beatty	208	210	1.0	2
7/23/90	King Salmon	107	124	15.9	17
7/30/90	Nahlin	606	629	3.8	23
7/30/90	Nahlin	179	174	-2.8	-5
7/30/90	Nahlin	873	948	8.6	75
7/30/90	Dudidontu	757	780	3.0	23
7/31/90	Tseta	347	313	-9.8	-34
7/31/90	Little Tahltan	1755	1972	12.4	217
7/31/90	Beatty Creek	207	222	7.2	15
7/31/90	Nakina River	2490	2830	13.7	340
8/01/91	Takhanne	325	359	10.5	34
8/01/90	Klukshu	1381	1432	3.7	51
8/05/90	Nakina	6210	7917	27.5	1707

Average when counts less than 1000 = 1.9% or 10 fish.
Average when counts greater than 1000 = 4.4% or 224 fish.

Table 7. Variation in escapement counts by foot survey and helicopter survey of chinook salmon in Southeast Alaska and transboundary rivers.

Date	River	Foot Count	Helicopter Count	Difference %	Difference Number
7/30/76	King Salmon R.	59	62	5.1	3
8/14/84	Eulachon	319	350	9.7	31
8/07/85	Eulachon	293	275	-6.1	-18
8/29/85	King Creek	320	377	17.8	57
8/29/85	Humpy Creek	47	50	6.4	3
8/07/86	Clear Creek	183	86	-53.0	-97[a]
8/16/87	Clear Creek	86	92	7.0	6
7/28/89	Nahlin	900	1221	35.7	321
8/13/90	Andrew Creek	664	570	-14.2	-94

[a] Fish schooled in deep pool.

Table 8. Proportions of total escapement through weir observed in airplane and foot surveys of chinook salmon in Andrew Creek[a], Southeast Alaska.

Date	River	Weir Count	Air plane Count	Foot Count	Difference %	Difference Number
1976	Andrew Creek	468	60		12.8	-408
1978	Andrew Creek	400	110		27.5	-290
1982	Andrew Creek	910	340		37.4	-570
1984	Andrew Creek	355	120		33.8	-235
1979	Andrew Creek	382		221	57.9	-161
1981	Andrew Creek	629	75	275	43.7	-354
	Airplane survey average				27.9	-376
	Foot survey average				50.8	-258

[a] Andrew Creek is a small system which usually contains numerous pink, chum and sockeye salmon simultaneously with chinook.

standardized survey areas of high fish abundance, reduced variability due to observer experience and saved time over other methods. The Index Area method developed by the ADF&G is similar in some ways to the strip method. In large systems, only certain standardized index areas are surveyed in the same manner each year. Most of the index areas are on the spawning grounds where visibility is best and most of the fish are not actively migrating.

Estimates based on peak counts alone without further correction, are only reliable in small shallow systems where migration into spawning areas and spawning occur over a relatively short period of 7 - 10 days. Where spawning occurs in waves or continuously over a period longer than the redd life of individual spawners, a correction factor for turnover of spawners between counts is necessary, and variability is increased (Cousens et al 1982). Redd life or residence time on the spawning grounds was used by Neilson and Geen (1981) to calculate the total escapement of chinook salmon to the Morice River, British Columbia. Using helicopter counts and corresponding aerial photos they estimated that 96% of all fish present during a survey were observed. However, due to protracted spawning over a 30 day period, the peak aerial survey only counted 52% of the total escapement. Neilson and Geen 1981) found that as the spawning season progressed the redd life decreased from 13.1 days early in the season to 7.7 days late in the season. Redd life for chinook salmon spawning on the Nakina River has been observed for three seasons. The average was 10.8 days with no trend towards shorter life late in the season. Nakina River chinook salmon also spawn over approximately a 30 day period, which indicates that a peak survey estimate probably underestimates the escapement.

> "The usefulness of an index stream appears to be not so much in its value for predicting the size of salmonid runs to each and every individual stream in the same drainage basin or biogeoclimatic zone, but as a general indicator of what is taking place salmonwise in a river system or group of streams that the index stream has been chosen to represent.... Long time-series are essential, and consistency in escapement estimation is more important than accuracy. Long time-series of escapement are therefore of greater value when the estimates are made by the same individual year after year, using the same technique" (Symons and Waldichuk, 1984).

In summary, we are confident that the revised goal for the Situk River is a reasonable approximation of the optimum escapement goal. The analysis will continue to be updated periodically but we do not expect dramatic changes. The Taku River goal is based on estimates of both escapement and harvest rather than actual counts. It appears that the escapement estimates may be low and the escapement goal will need further revision as the database improves. Of the three systems discussed, we have the least confidence in escapement goal for the Unuk River. Escapements are estimated from helicopter and foot surveys and only small numbers are sampled for age composition. The harvest has been estimated by coded-wire tag recoveries, however that program has ended and harvests patterns have possibly changed significantly in the last decade. To construct an acceptable escapement goal based on spawner-recruit data will require continued improvement in escapement estimation and extensive increases in harvest estimation and sampling.

Escapement data are required by fisheries managers for a number of purposes which include: assessing exploitation and the effects of exploitation on stock status, estimation of stock-recruitment curves, forecasting future recruitment to the fishery, to achieve optimum escapement, to document timing of spawning runs (Symons and Waldichuk 1984). The required degree of precision and accuracy in the escapement data varies with the need. For example, the precision and accuracy required to assess the response to changes in exploitation (*ie* rebuilding) is much less than required to establish stock-recruitment curves and optimum escapements. Random errors or "noisy data" result in an apparently poor relationship between stock and recruitment.

Noisy data will always underestimate escapement requirements (Hilborn 1983). As previously stated, the primary purpose of the aerial survey index expansion program is to assess rebuilding of chinook salmon stocks in Southeast Alaska and transboundary rivers. Notwithstanding any errors in precision and accuracy, the long time-series of consistently arrived at estimates is sufficient for this task.

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APPENDIX

Appendix A1. Management escapement goals and survey and tributary expansion factors for southeast Alaska and transboundary rivers. The escapement goal for each category equals the sum of the survey escapement goal times the survey and tributary expansion factors times the category expansion factor. Using 1991 revised goals and preliminary expansions for transboundary rivers.

River System	Index Tributaries Surveyed	Survey Escapement Goal[a]	Survey Expansion Factor	Tributary Expansion Factor	System Escapement Goal	Category Expansion Factor	Category Escapement Goal
			Major Production Systems (Total = 3)				
Alsek	Klukshu	4,700 (W)	1/1	1/.64	7,300		
Taku	Six Tributaries	13,200 (A)	1/0.70	1/.55	34,300		
Stikine	Little Tahltan	5,300 (W)	1/1	1/.25	21,200		
Major Category Subtotal		23,200			62,800	3/3	62,800
			Medium Production Systems (Total = 9)				
Situk	All	600 (W)	1/1	1/1	600		
Chilkat	Big Boulder/Stonehouse		- UNDER REVIEW -				
Andrew Cr.	All	470 (A)	1/0.625	1/1	750		
Unuk	All	1,800 (A)	1/0.625	1/1	2,880		
Chickamin	All	900 (A)	1/0.625	1/1	1,440		
Blossom	All	800 (A)	1/0.625	1/1	1,280		
Keta	All	500 (A)	1/0.625	1/1	800		
Medium Category Subtotal		5,070			7,750	9/6	11,625
			Minor Production Systems (Total = 22)				
King Salmon	All	250 (W)	1/1	1/1	250		
Minor Category Subtotal		250			250	22/1	5,500
All Systems Total		28,520			70,800		79,925

[a] (W) = weir count; (A) = aerial survey peak escapement estimate.

APPLICATION OF RADIO TELEMETRY AND A MARK RECAPTURE EXPERIMENT TO ESTABLISH AN INDEX OF CHINOOK SALMON ESCAPEMENT TO A LARGE ALASKAN RIVER DRAINAGE

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ABSTRACT

Declining observations of chinook salmon (*Oncorhynchus tshawytscha*) spawning abundance in two clear-water tributaries of the otherwise silt-laden Chilkat River, near Haines, Alaska prompted systematic, quantitative research to determine the relationship between observed escapement to the index areas, and the total escapement to the system. Radio telemetry and a mark-recapture experiment were used to estimate spawning distribution and abundance of age 1.3 and older chinook salmon returning to the system.

Two hundred twenty-five large (age 1.3 +) upstream migrating chinook salmon were captured in fish wheels and gillnets between May 18 and July 19, 1991. The mean date of their immigration was July 5. One hundred and eighteen of these fish were implanted with radio transmitters, and 106 were tagged with solid-core spaghetti tags. One hundred of the fish with transmitters were tracked to areas of the drainage where it was assumed they spawned.

An estimated 54% (SE = 6.2%) of the chinook salmon spawning in the Chilkat River drainage in 1991 occurred in the Kelsall River system, 33% (SE = 6.0%) in the Tahini River, 8% in the Klehini River system, 4% in the mainstem Chilkat River, and 1% in Assignation Creek.

Between July 22 and September 12, 733 large chinook salmon spawning in the Chilkat River drainage, mostly on the Kelsall River, Nataga Creek, and Tahini River were randomly inspected for a tag to estimate abundance.

A simple Peterson model (n_1=224, n_2=733, m_2=27) was used to estimate that 5,897 (SE=1,005) chinook salmon age 1.3 and older returned to the Chilkat River in 1991. An unknown number of these fish died of natural causes, or were caught in a subsistence fishery, prior to spawning.

The two most important findings of this research were: a) that estimated escapement to the Chilkat River system was much greater (seven times) than the historical expansion estimator would have indicated, and b) that historic index areas, Big Boulder and Stonehouse Creeks, were not major spawning locations during the 1991 season.

INTRODUCTION

Direct enumeration of the escapement into a large river system is often difficult or impossible. Counting fish in river systems with turbid water is especially problematic. Thus, in Alaska, chinook salmon (*Oncorhynchus tshawytscha*) are counted in clear-water tributaries of large systems, to make indices of abundance. These indices of chinook salmon abundance have been obtained annually in Stonehouse and Big Boulder Creeks, two clear-water tributaries of the otherwise silt-laden Chilkat River system, near Haines

Alaska. During 1985 and 1986, these indices declined drastically, coinciding with the growth of the terminal chinook salmon recreational fishery and increased harvests of chinook salmon in the commercial gillnet fishery (Table 1). This abrupt decline prompted the restriction and ultimate closure of the directed chinook salmon fisheries in the vicinity from 1987 to the present. Under these circumstances, some user groups considered nomination of this stock for endangered or threatened status. Our research was motivated by concern that Chilkat River chinook salmon were either severely depleted, or that the peak survey counts in the index areas were an inaccurate indication of actual spawning escapement to the Chilkat River.

The Chilkat River originates in Yukon Canada (Figure 1), covers about 1,000 square miles, and flows into Lynn Canal in Northern Southeast Alaska. Chinook salmon bound for the Chilkat River are harvested by troll, seine, and gillnet commercial fisheries, and recreational and subsistence fisheries. The Chilkat stock is affected by in-shore and off-shore gear groups and management plans, and is similar in such respects to other stocks on the west coast today.

The primary objective of our study (Johnson et al 1992) was to detect all spawning areas on the Chilkat River which received a substantial portion of the large (age 1.3 and older) chinook salmon spawning in 1991. A secondary objective was to test the feasibility of sampling fish on the spawning grounds to recover tags, and determine the feasibility of sampling for coded wire tags in the future.

METHODS

Adult chinook salmon were captured with fishwheels and drift gill nets as they immigrated into the lower Chilkat River from May 18 through July 19, 1991. Advance Telemetry Systems radio transmitters were implanted in large chinook salmon (>660mm mid-eye to fork), which were then tracked to spawning areas. Fish age was later estimated from scales collected from each fish. Results were then tabulated for "large" and "small" fish that were aged 1.3 and older, or <1.3 years of age, respectively. We initially estimated that the successful tracking of 80 radio transmitters would allow us to locate discrete areas receiving more than 7.5% of the escapement with 95% confidence.

We expected an escapement of 1,000 large chinook salmon, and attempted to tag fish in proportion to abundance as they immigrated into the river. However, radio transmitters were attached to captured fish at different rates during the immigration because many more fish were captured than were anticipated.

Chinook salmon captured but not implanted with a radio transmitter (radio-tagged) were marked with a spaghetti tag, and the posterior half of the adipose fin of each tagged fish was clipped as a secondary mark. Scales of each chinook salmon captured were collected for scale pattern analysis.

We searched for each radio transmitter once a week to determine the distribution of the radio-tagged fish. Transmitters were located from the road system when possible, then from a boat or aircraft as fish traveled to more remote areas.

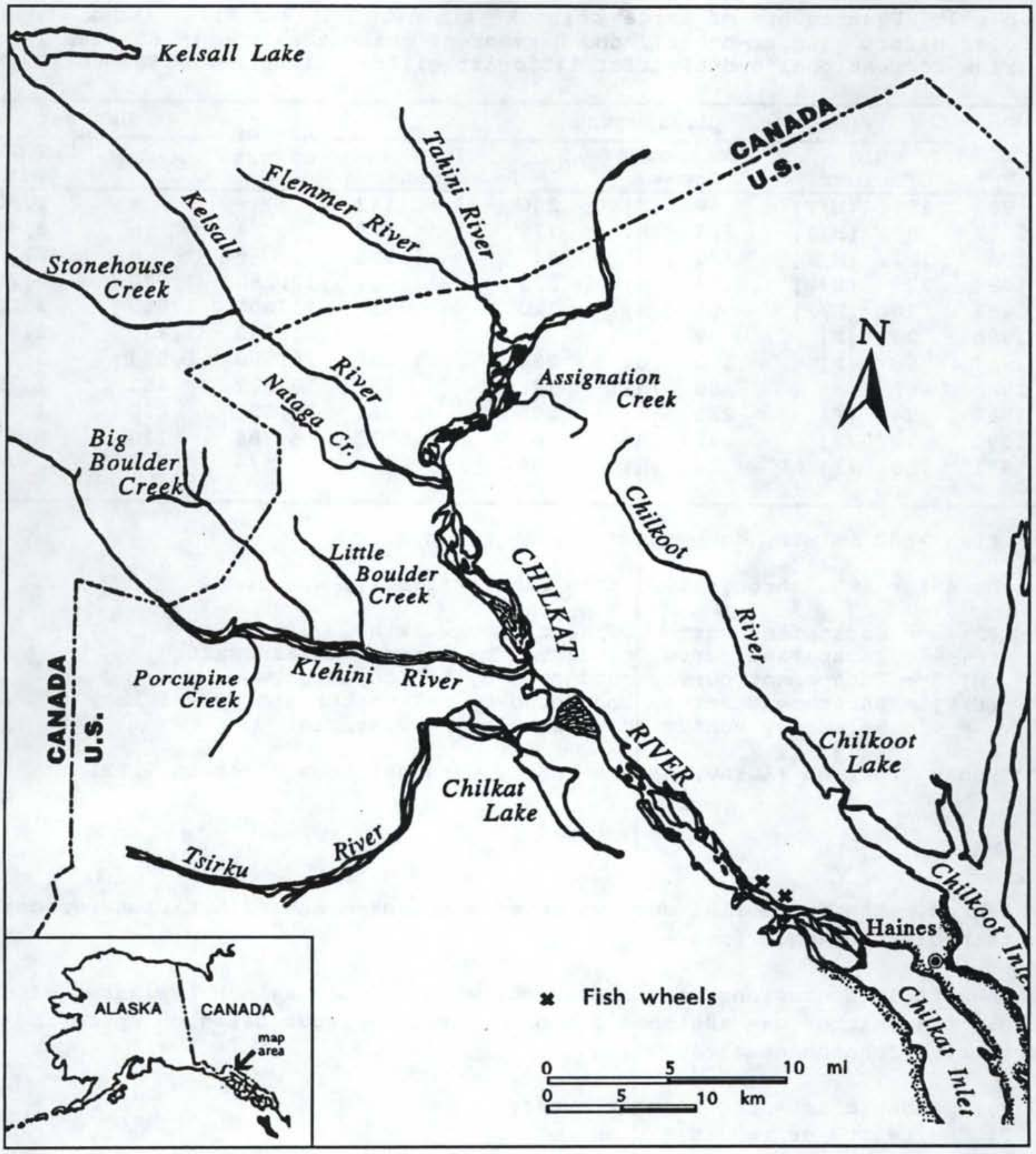

Figure 1. Map of the main features of the Chilkat River drainage.

Table 1. Peak counts of large[a] chinook salmon in Chilkat River index areas, angler effort (angler-hours), and harvest of chinook salmon in Chilkat Inlet marine recreational and District 115 drift gillnet fisheries, 1981 to 1991[b,c].

Year	Escapement				Angler effort	Harvest	
	Big Boulder	Stonehouse Creek	Total	Drainage expansion		Sport	115 Gill net
1981	187 (H/F)	69 (H)	256	1,143	-	-	1,300
1982	56 (H/F)	123 (H)	179	799	-	-	5,945
1983	121 (H/F)	126 (H)	247	1,103	-	-	2,119
1984	229 (H/F)	104 (H)	333	1,487	10,250	1,070	6,207
1985	70 (H/F)	50 (H)	120	536	21,600	1,615	3,260
1986	20 (F)	9 (H)	29	129	31,540	1,620	2,772
1987	98 (F)	190 (H)	288	1,286	26,590	1,094	3,223
1988	86 (F)	89 (H)	175	781	36,222	481	1,257
1989	74 (F)	231 (H)	305	1,362	10,526	252	1,995
1990	19 (F)	42 (H)	61	272	8,784	210	670
1991[c]	59 (F)	126 (H)	185	805	N/A	N/A	749

a Fish >660 mm MEF, determined during surveys.

b Data for 1981 through 1990 from Pahlke (1991a), where:

(F) = Escapement survey conducted by walking,
(A) = Escapement survey conducted by fixed-wing aircraft,
(H) = Escapement survey conducted by helicopter,
(H/F) = Escapement survey conducted by helicopter and by walking,
\- = No survey conducted or data not comparable.

c Randy Ericksen (ADF&G, Haines, AK.), personal communication, 1991.

Search paths for aerial surveys covered mainstem and tributaries reasonably attainable by tagged fish.

When field operations were concluded, each chinook salmon implanted with a radio transmitter was assigned one of 5 possible fates based on specific criteria (Johnson et al 1992):

1). Probable spawning in a tributary;
2). Mortality or regurgitation;
3). Probable spawning in the mainstem;
4). Captured;
5). Unknown.

The proportion of the large chinook salmon spawning in each area (P_a) was then estimated;

$$P_a = \frac{\sum_{t=1}^{3} \left(\frac{N_t}{n_t}\right) r_{a,t}}{\sum_{a=1}^{5} \sum_{t=1}^{3} \left(\frac{N_t}{n_t}\right) r_{a,t}} \qquad (1)$$

where $r_{a,t}$ is the number of large fish tagged with radios in period t that were tracked to and assumed to spawn in area a, N_t is the number of large fish captured in fish wheels and gillnets in period t, and n_t is the number of large fish radio-tagged in period t. Period (t) refers to distinct spans of time when the tagging fraction was constant. In 1991, we implanted radios in about 88% of the fish captured during the first 3 weeks of the experiment; then about 21% of the fish captured during the forth week, and finally about 33% of the fish captured during the final week of the experiment were implanted with radios. Fish were tracked to, and assumed to spawn in five areas (a=5) in 1991. Transmitters assigned fates not associated with successful spawning are accounted for in computing P_a, such that the sum of the proportions equals one. Because fish with unknown and captured fates were observed, the standard error of P_a was estimated using the bootstrap (Efron 1982).

The calculated proportions of large chinook salmon spawning in each area depends on an assumption of proportional tagging and the standard "and/or" assumptions for unbiased estimates (Seber 1982). Constant effort was expended to capture fish over time, and testing to detect a failure of this assumption was conducted, as described later.

After the spawning migration was complete, chinook salmon were sampled on the spawning grounds for tags, missing adipose fins, and age (scales). Large chinook salmon were captured primarily with custom-made dipnets that had long handles and sturdy frames. Captures were also made with gill nets, and bare hands. Dead or near-dead fish were captured with spears. Double sampling was prevented by punching a hole in the operculum of all captured fish released alive, and by slashing all sampled carcasses. Ages of the large chinook salmon captured were determined from scale pattern analysis. One recovery team was stationed on the Tahini River, and one team sampled as many sites as possible, but captured large numbers of salmon only on the Kelsall River and Nataga Creek (Figure 2). This was due in part to the difficulty of capturing fish in areas where the river was high and fast, and part to the inaccessibility of other areas.

RESULTS

Two hundred twenty-five large adult chinook salmon were captured near mile 8 of the Chilkat River from May 18 through July 19, 1991 (Table 3). Capture rates peaked on July 7, nearly one month later than anticipated on the basis of data from the local recreational marine boat fishery. Of the 225 chinook salmon captured, 118 were implanted with radio transmitters and 106 were given an external tag. Of the 118 large chinook salmon given radio transmitters, 100 were tracked to a spawning area. Sixteen other transmitters were thought to be regurgitated, lost because a fish died before spawning, or behaved in a way that defied assignment of a fate. One radio transmitter was returned from the subsistence fishery, and one transmitter was never located after tagging.

Adjusting for differential tagging rates, the proportion of large chinook salmon passing Chilkat River mile 8 and spawning were: 54% (SE = 6.2%) spawned

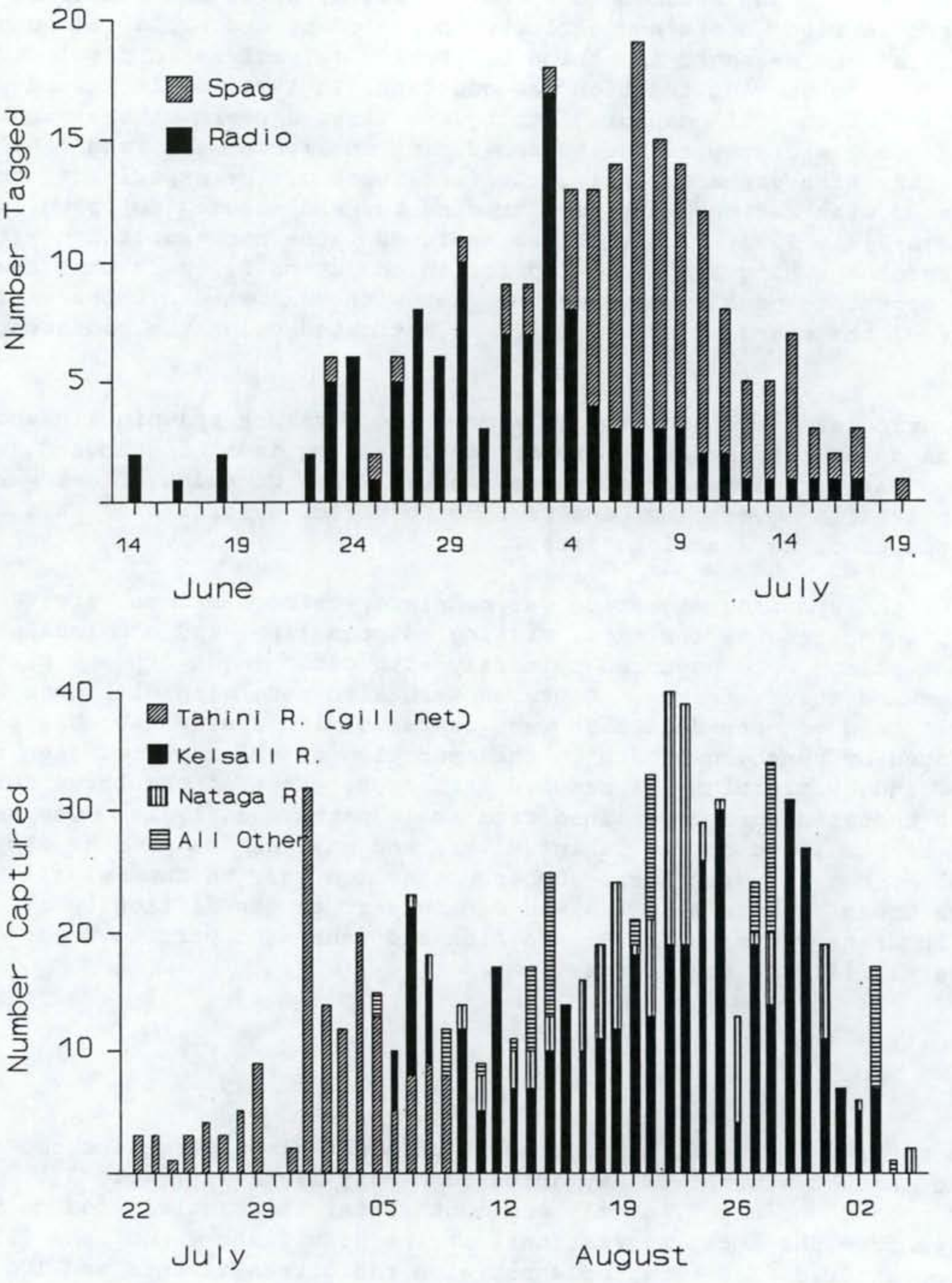

Figure 2. Numbers of chinook salmon age 1.3 and older marked with spaghetti (Spag) and radio transmitters (Radio) and released into the lower Chilkat River (top), and numbers of chinook salmon age 1.3 and older sampled in escapement surveys (bottom), by date, 1991.

Table 3. Frequency of capturing chinook salmon in fish wheels and gill nets during the tagging experiment, by size, sex, and time period, Chilkat River, 1991.

	Lower Fish wheel[a]				Upper Fish wheel[b]				Gill Net[c]		
	Large[d]		Small[e]		Large		Small		Large		
Time Period	M	F	M	F	M	F	M	F	M	F	Total
5/18-6/09	0	1	2	0	0	0	0	0	0	0	3
6/10-6/14	0	0	0	0	0	0	0	1	1	1	3
6/15-6/19	0	0	3	1	0	1	1	0	0	2	8
6/20-6/24	2	2	5	2	2	1	1	0	4	3	22
6/25-6/29	2	2	17	2	5	8	14	2	4	12	68
6/30-7/04	5	7	21	3	5	19	27	3	4	11	105
7/05-7/09	4	7	34	0	14	29	26	4	8	13	139
7/10-7/14	1	7	25	0	4	11	25	4	6	8	91
7/15-7/19	1	2	4	1	1	2	4	1	1	2	19
Total	15	28	111	9	31	71	98	15	28	52	458

[a] Fished 5/05 to 7/19.
[b] Fished 6/09 to 7/19.
[c] Fished 5/22 to 7/19.
[d] Fish aged 1.3 and older.
[e] Fish aged 1.2 and younger

in the Kelsall River system, 33% (SE=6.0%) spawned in the Tahini River, 8% (SE=3.7%) spawned in the Klehini River system, 4% (SE=1.4%) spawned in the mainstem ChilkatRiver, and 1% (SE=0.8%) spawned in Assignation Creek. Seven hundred thirty-three large chinook salmon spawning in the Chilkat River drainage (mostly in the Kelsall River, Nataga Creek, and Tahini River), were captured and inspected for tags between July 22 and September 12 (Figure 2). Of these fish, 14 carried spaghetti tags and 13 fish carried radio transmitters from the initial capture event near river mile 8. Age composition of fish sampled in Kelsall River and Nataga Creek was similar (x^2=3.5, P=0.18) so data for these neighboring systems was combined. The probability of recapturing large chinook salmon with spaghetti tags and radio tags was not significantly different (x^2=0.03, P=0.88), so sampling data on both types of tags could be pooled to estimate spawning abundance.

Additional hypothesis tests were then conducted to determine the best method to estimate abundance from the tagging data. First, we looked to see if differences in run-timing were apparent. Fortunately, two widely separated Chilkat River tributaries were heavily sampled in 1991; the Kelsall - Nataga, and the Tahini River systems. We tested the hypothesis that run timing for chinook salmon bound for the Tahini and Kelsall Rivers was equal, using an odds ratio (Agresti 1984)

$$\theta = \frac{\left(\frac{N_{e,t}}{N_{e,k}}\right)}{\left(\frac{N_{l,t}}{N_{l,k}}\right)}$$

(2)

where N represents the number of radio-tags implanted during the first (e) or second (l) half of the sampled immigration, which were tracked to the Tahini (t) or Kelsall Rivers (k). The hypothesis was accepted since θ=0.81 was well within the bounds of a 95% confidence interval for θ=1 (0.3 to 2.2). Secondly, we tested the hypothesis that the probability of recovering a marked fish in the two major spawning areas (Kelsall - Nataga) was equal. We accepted this hypothesis with low confidence (x^2=2.9, P=0.09).

Finally, since the escapement sampling was designed to give each fish in sampled areas similar probabilities of being inspected for marks, we felt reasonably justified in using a simple Peterson model to estimate abundance. In this experiment, the assumptions of the mark-recapture experiment (Seber 1982) could be investigated because a large number of fish were sampled on widely separated spawning grounds across the spawning season.

Thus, we used Chapman's modified Peterson estimator (Seber 1982, n_1=224, n_2=733, m_2=27) to estimate the immigration of large chinook salmon to the Chilkat drainage. The estimated abundance of 5,897 (SE=1,005) is germane to the time of tagging near mile 8, since an unknown component of mortality occurs (due to natural causes and a subsistence fishery) between the two sampling events. This estimate is more precise, and not significantly different from, an estimate using a Darroch's estimator (Seber, 1982) which does not assume equal probability of recovering marked chinook salmon by recovered area.

DISCUSSION

The two most important findings of this seasons research are: a) that estimated escapement to the Chilkat River system was much greater (seven times) than the historical expansion estimator would have indicated, and b) that historic index areas, Big Boulder and Stonehouse Creeks, were not major spawning locations during the 1991 season.

Based on these results, we will seek to determine if this was normal, or if it was a unique event. This can be addressed only after similar, future, studies are concluded. If it was indeed unique, we need to determine the cause.

In estimating the distribution of spawning escapement we assumed: a) catch of large chinook salmon during the tagging event was in proportion to their numbers immigrating over time; b) tagging did not change the spawning destination of a fish; and c) fates of tagged fish were accurately determined. Assumption b is unlikely to be untrue in this experiment, but we did not test for effects of the tagging on fish behavior. Since fishing effort was relatively constant, departures from proportional sampling would be related to time-dependent changes in catchability. Environmental conditions did, however, fluctuate greatly during the experiment (Appendix C), and the tagged to untagged ratios from the Tahini (0.03:1) and Kelsall-Nataga (0.06:1) spawning areas can be taken as weak evidence in support of non-proportional tagging (the ratios were estimated from 27 total recaptures and are not very precise).

We do not think gear selectivity (for size) was an important variable in this experiment; most fish were captured in fish wheels, and we could not demonstrate that age-composition (size) or timing of the stocks in the two major spawning areas was different in 1991.

Finally, errors probably exist in assigned fates of some radio-tracked fish, especially fish with fate code 2 (mortality and regurgitation), fate code 3 (mainstem spawning), and fate code 5 (unknown). For example, motion and mortality sensor signals can lead to ambiguous, inconsistent conclusions about a fate of a tracked fish (Bendock and Alexandersdottir 1992). Fates of fish having transmitters located repeatedly in the mainstem between Chilkat River miles 16 and 22, are most uncertain. Also, signals from several transmitters implanted during the later half of the experiment emitted weak signals, potentially contributing to the size of the "unknown" fate category. Since most fish with ambiguous tracking histories were not assigned to a spawning area, potential errors in the other assignments are essentially random, we assume, and are unlikely to significantly influence the estimated proportions for spawning in each area.

In estimating abundance we assumed: a) tagging of large chinook salmon was in proportion to their numbers immigrating over time, or that immigration timing of the stocks was similar and sampling for marks on fish spawning in the areas sampled was random; b) untagged fish did not recruit to the population between sampling events; c) tagged and untagged fish suffered similar mortality rates between sampling events; and d) that fish did not lose marks. Considerable efforts to catch and mark fish in proportion to their abundance were made. Weak evidence that this did not occur could be cited (from tag ratios on the Kelsall and Tahini Rivers). However, sampling effort for tags on the Kelsall and Tahini Rivers, where ≈87% of spawning occurred in 1991, was fairly constant across the time of the immigrations, and the complex-assumption a) is thus fairly robust for this experiment.

We reason a mechanism for failure of assumption b) does not exist. We have no direct evidence to disprove assumption c). We believe 11 of 117 radio transmitters (9%) tracked on large chinook salmon were regurgitated or associated with fish which died prematurely during the experiment. Some of these fish (and some of the 5 fish with unknown fates) certainly may have died due to the tagging procedure, the subsistence fishery, or to natural causes, and we cannot separate these fates. Similarly, we did not estimate tag loss during the experiment; but on the largest system sampled (Kelsall-Nataga), no tag loss was observed.

It is not known how the size of the escapement of chinook salmon in 1991 was influenced by closures of the recreational and commercial fisheries during the year. The closures were made because the primary age classes returning in 1991 would primarily come from escapements in 1985 and 1986 which were believed at the time to be very small (Table 1). Another problem lies in estimating the optimum escapement goal for the Chilkat River. Currently, the goal is 2,000 age 1.3+ chinook salmon (Pahlke, In Press) which was derived by expanding the number of fish observed in the two index stream sections. Prior to 1981, the expansion factor for Big Boulder Creek was 1/.11, and from 1981 to present (including Stonehouse Creek) it was (1/.8)x(1/.28), (Pahlke, 1991b). This "best guess" methodology was used because a better, systemic, method was not available. We propose that a new index method, based on catch at a setnet site on the Tahini River or a mark recapture experiment, will provide an accurate, precise, and cost-effective index of spawner abundance for the Chilkat River drainage. However, this will not work if enhancement activities continue on the Tahini River. An even better index could be made from data collected on the Kelsall River, if this were possible.

Application of these results to the more general problem of monitoring large chinook systems where escapements are poorly defined will hopefully occur in Alaska, where logistic problems and turbid water prevent other population and distribution research techniques.

ACKNOWLEDGEMENTS

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ABUNDANCE BASED CRITERIA FOR RECOGNITION OF DAMAGED SALMON POPULATIONS

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ABSTRACT

The question of how to measure the status of Columbia Basin salmon stocks is addressed by calculating several simple statistics for annual observations of chinook spawning escapement, primarily redd counts, from all available localities in that portion of the Columbia Basin within the states of Idaho, Oregon, and Washington. The records are an excerpt of the National Environmental Database (NED). The results for a total of 400 chinook salmon spawning escapement surveys (292 of wild, 67 of hatchery, and 41 of other origin) are reported. Less than 26% (103) of the most recent surveys were at seeding levels greater than 50%, with more than 30% (122) of those surveys being seeded at 10% or less. A total of 117 surveys (29%) were identified as badly damaged and decreasing. Among these surveys, 70% (82) were at or below the 20% seeding level. Among wild salmon surveys, more than half (53%; 155) were most recently at levels of annual abundance of no more than 20% of seeding. The forty-four wild chinook surveys identified as critically damaged were consistent with federal threatened species designations. Of the 67 hatchery observations, less than 24% (16) were most recently at, or below, 10% of their maxima during the time span of the surveys. Most of the chinook salmon survey areas in the Columbia Basin clearly are both in chronic decline and have critically low levels of spawning escapement.

INTRODUCTION

The question of how to measure the status of Columbia Basin salmon stocks has always been important, however the need to answer this question has taken on an air of urgency since stocks of chinook salmon in the Snake Basin joined the federal list of threatened species earlier this year. The federal listings, and the fact that salmon stocks in the Columbia Basin have been repeatedly declared to be at risk (i.e., NPPC 1986), add impetus to the discussion of appropriate measures of stock status.

The diversity of measurement techniques available in the form of visual counts at hydroelectric dams, and surveys of spawning activity made by foot and from the air, make the comparison of stock status for wild salmon populations problematic. Hatchery populations have counts of individuals returning to the facility that, in principle, are directly comparable. At present the need has not been met for a comprehensive survey of wild salmon populations that employs a quantitative measure of wild stock health and permits meaningful comparisons among stocks. The present effort is focused on chinook salmon, however the approach envisioned should be applicable to all species on which there are similar types of data.

METHODS

The data are annual observations of at least four kinds, all measures of chinook spawning escapements; redd counts on natural spawning areas, both foot and aerial, dam and weir counts, and hatchery rack counts at all available localities in that portion of the Columbia Basin within the states of Idaho, Oregon, and Washington. The records were obtained from an excerpt of the National Environmental Database, NED, provided by Mr. Duane Anderson (NPPC, Northwest Power Planning Council, Portland, OR). The NED is coordinated by Mr. Tom Pansky (Bonneville Power Administration, Portland, OR).

Each locality was examined in order to eliminate those annual time series of less than twenty five years of continuous records. For each series retained, a fifteen-year moving average was calculated for each year as the arithmetic mean of the year and the fourteen preceding years, starting with the fifteenth year of data, and progressing to the latest year available, usually 1990. Working with the fifteen-year moving averages, the parameters of a linear regression were calculated for the annual time series of each locality, and those localities having negative slopes that tested significantly different (alpha = 0.05) from zero were identified as, badly damaged and decreasing. Finally, the most-recent annual abundance observation, usually that of 1990, was taken as a percentage of the maximum annual observation in its time series in order to provide a rough measure of the most recent level of seeding of the spawning grounds. A spawning level that is 10% of the maximum observed is said to be at 10% of seeding. The ten percent level is arbitrarily defined as the threshold for being critically damaged.

RESULTS

A total of 400 chinook salmon spawning escapement surveys (292 of wild, 67 of hatcheries, and 41 of others that include mixed hatchery and wild data such as dam and weir counts) were found to have time series long enough for this analysis. In examining the most recent survey abundance, less than 26% (103) of the surveys were greater than 50% of the historical maximum, with greater than 30% (122) being 10% or less than the historical maximum as of 1990 (Figure 1).

In examining the most recent value for the 292 surveys of wild salmon alone, about 24% (69) were greater than 50% of the historical maximum abundance, however in more than 36% (106) of the surveys, the most recent annual abundance was 10% or less of the historical maximum survey abundance (Figure 2). After editing the list of wild salmon surveys at or below the 10% seeding level to exclude observations about which the authors have unanswered questions, there remain 44 biologically well known survey sites that are most

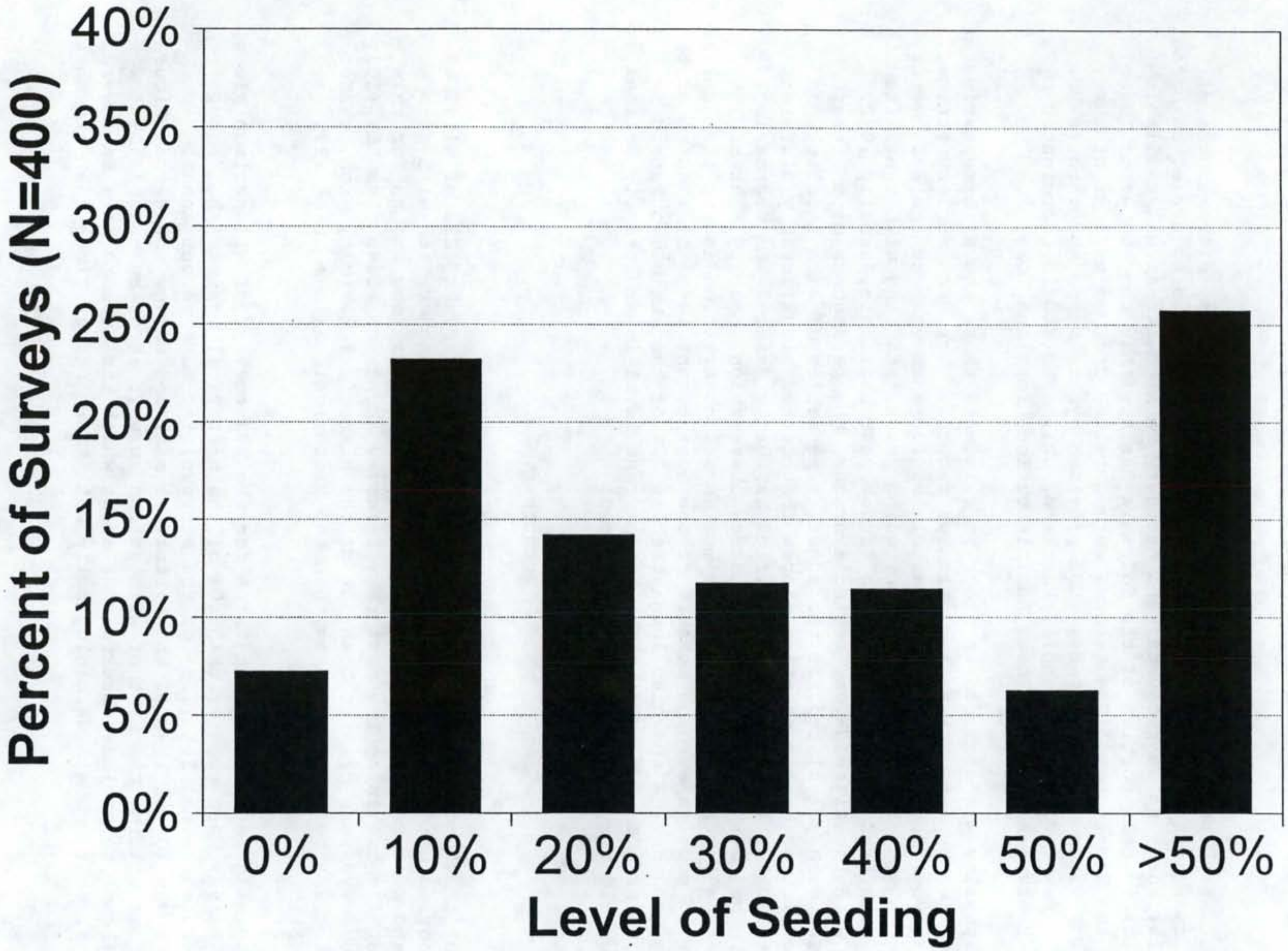

Figure 1. Frequency histogram of Columbia Basin chinook spawning escapement surveys at seeding levels grouped in ten percent increments.

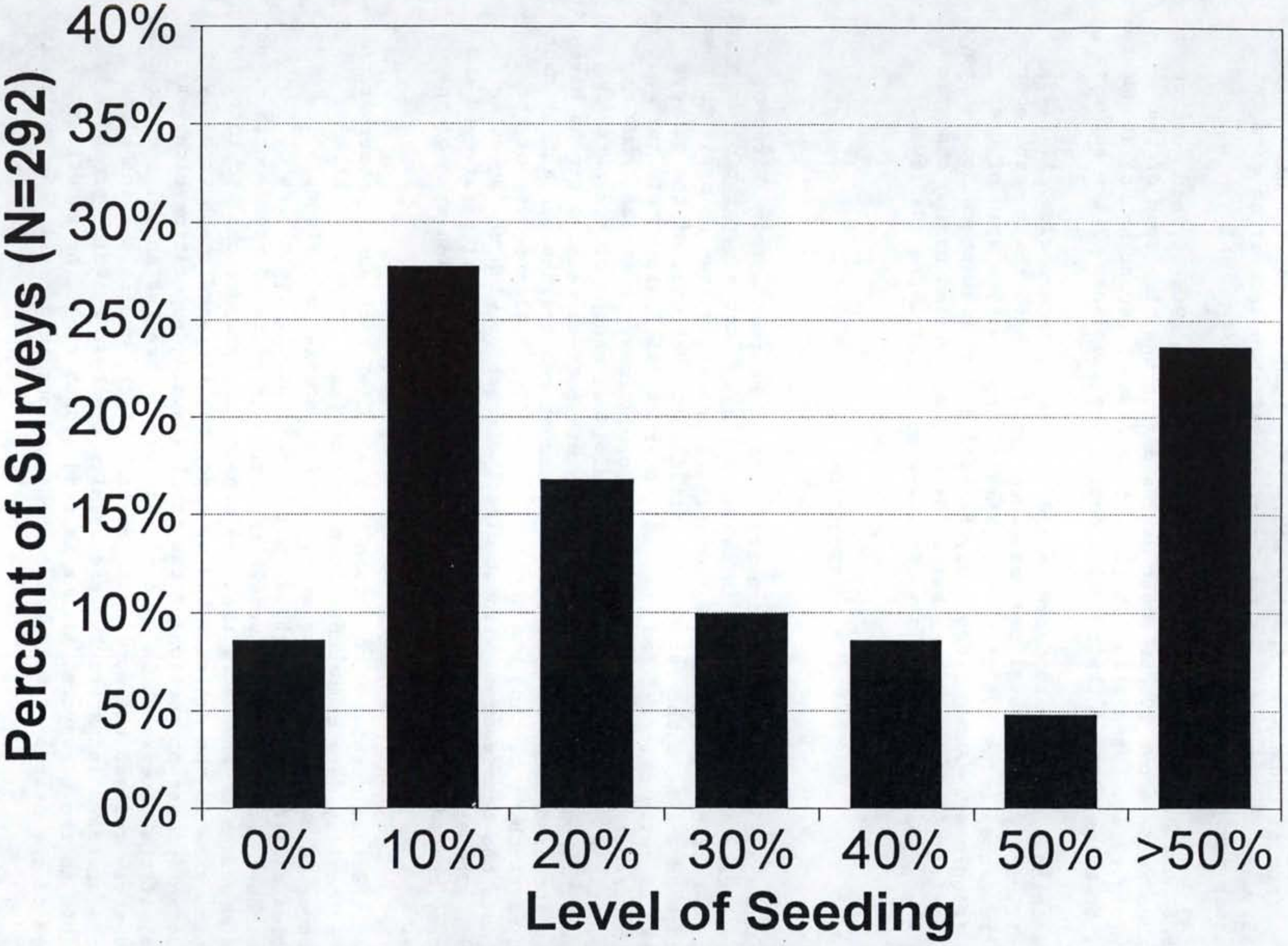

Figure 2. Frequency histogram of Columbia Basin chinook spawning escapement surveys of wild spawners at seeding levels grouped in ten percent increments.

likely to be indicative of critically damaged salmon populations (Table 1). The 44 surveys identified as critically damaged (Table 1) are also noteworthy in that all of the surveys in Idaho, and most of the surveys in Oregon, as well as one in Washington, contain individuals of species listed as threatened under the federal Endangered Species Act. More than half of the wild surveys (53%; 155) were most recently at levels of annual abundance of no more than 20% of their maxima during the time span of the surveys.

Of the 67 hatchery observations, slightly less than 39% (26) had most recent annual abundances above 50% of their maxima during the time span of the surveys (Figure 3). Less than 24% (16) of these most recent annual abundances were at, or below, 10% of their maxima during the time span of the surveys.

In examining trends in abundance in the 400 annual time series, 117 (29%) had significantly negative slopes, and were found to have their most recent annual abundances below 50% of their maxima during the time span of the surveys (Figure 4). Among these badly damaged and decreasing chinook surveys, nearly half (47%; 55) were at or below 10% of their maxima during the time span of the surveys, and fully 70% (82) were at or below the 20% level.

DISCUSSION

The nature and limitations of the study should be recognized. First of all, the time series of each locality is called a survey, not a population or stock (Simon and Larkin, 1972), because the authors presently have no basis to make this judgement. While the authors are optimistic that most of the surveys described as "wild chinook" and "hatchery chinook" will ultimately turn out to be indicators of self-sustaining breeding populations, or stocks, some may not. Questions of how to aggregate the individual groups of spawners described in these surveys (i.e., Table 1) into self-sustaining stocks have not been answered here, however such recommendations are the subject of much current research, including the authors. Secondly, the integrity of the survey data has been assumed without direct discussion with the agencies involved. The assumption of integrity means that the locality surveyed and the survey methods have remained the same throughout the time span of the surveys.

The statistical methods were chosen in the belief that annual observations within a locality were comparable from year to year. Further, while differences in the type of data among localities make comparisons among localities difficult or impossible, the methods permit the slopes of regressions to be compared. Surveys reporting in units of redd counts do not produce measures of abundance that are necessarily proportional to total population size in the drainage basin, since the standard reach probably contains an inconstant portion of the total spawning population each year. However, if the same reach is counted each year, then the annual redd counts do accurately reflect the population of that reach, and that reach alone. Since the standard index reaches are usually the best spawning grounds accessible to the surveyors, trends in redd counts should be excellent measures of the status of wild chinook populations in general. So, while it

Table 1. A list of the locations of chinook abundance survey areas that are most likely to contain critically damaged chinook salmon populations in the Columbia River Basin of Idaho, Oregon and Washington States.

Reach Name	State	Subbasin	Race	% Seeding	Ref Num	EPA Reach number
ALTURAS LAKE CR	ID	HEADWATER SALMON	SPRING	0.0	240	1706020107700
BIG SHEEP CR	OR	IMNAHA	SPRING	0.0	152	1706010200500
HURRICANE CR	OR	GRANDE RONDE	SPRING	0.0	181	1706010505000
LICK CR	OR	IMNAHA	SPRING	0.0	156	1706010202100
SHEEP CR	OR	GRANDE RONDE	SPRING	0.0	166	1706010404100
WALLOWA R	OR	GRANDE RONDE	SPRING	0.0	172	1706010500100
YANKEE FORK	ID	UPPER SALMON	?	0.0	235	1706020104000
SALMON R M FK	ID	MID FK SALMON	?	0.3	253	1706020600100
TOUTLE R	WA	COWLITZ	FALL	0.7	563	1708000501100
CISPUS R	WA	COWLITZ	SPRING	0.8	545	1708000400100
VALLEY CR	ID	UPPER SALMON	SPRING	0.9	239	1706020105200
KLICKITAT R	WA	KLICKITAT	FALL	1.3	355	1707010507200
COLUMBIA R	WA	UPPER COLUMBIA	SUMMER	1.5	1	1702000500100
ENTIAT R	WA	ENTIAT	SUMMER	1.9	53	1702001004300
SALMON R	ID	LOWER MAIN SALMON	SUMMER	2.2	264	1706020900100
VALLEY CR	ID	UPPER SALMON	SUMMER	2.3	238	1706020105200
KALAMA R	WA	KALAMA	FALL	2.6	492	1708000304600
YANKEE FORK W FK	ID	UPPER SALMON	?	2.8	237	1706020104600
HERD CR	ID	UPPER SALMON	SPRING	3.0	244	1706020112000
BULL RUN CR	OR	JOHN DAY	SPRING	3.1	395	1707020203901
SALMON R E FK	ID	UPPER SALMON	SUMMER	3.1	242	1706020109600
BEAR CR	OR	GRANDE RONDE	SPRING	3.6	178	1706010503100
LOON CR	ID	MID FK SALMON	SUMMER	3.8	252	1706020505000
BIG CR	ID	MID FK SALMON	SUMMER	4.3	255	1706020600700
LEMHI R	ID	LEMHI	SPRING	4.3	247	1706020400100
RUNNING CR	ID	SELWAY	SPRING	4.8	267	1706030100800
CLEAR CR	OR	JOHN DAY	SPRING	4.9	394	1707020203600
SALMON R	ID	LOWER MAIN SALMON	?	4.9	263	1706020900100
SULPHUR CR	ID	MID FK SALMON	SPRING	5.1	248	1706020502100
BIG CR	ID	MID FK SALMON	SPRING	5.2	256	1706020600700
SALMON R N FK	ID	UPPER SALMON	SPRING	5.2	246	1706020307100
YANKEE FORK	ID	UPPER SALMON	SPRING	5.2	236	1706020104000
BEAR VALLEY CR	ID	BEAR VALLEY	SPRING	5.7	249	1706020502300
SALMON R	ID	LOWER MAIN SALMON	SPRING	5.7	265	1706020900100
IMNAHA R	OR	IMNAHA	SPRING	5.8	151	1706010200100
MARSH CR	ID	MID FK SALMON	SPRING	8.0	251	1706020503200
CAMAS CR	ID	MID FK SALMON	?	8.1	257	1706020605100
CATHERINE CR S FK	OR	GRANDE RONDE	SPRING	8.1	169	1706010406500
YOUNGS R	OR	LOWER COLUMBIA	SPRING	8.1	591	1708000601900
CATHERINE CR N FK	OR	GRANDE RONDE	SPRING	8.2	170	1706010406600
WENAHA R	OR	GRANDE RONDE	SPRING	8.2	192	1706010600900
LOSTINE R	OR	GRANDE RONDE	SPRING	8.6	179	1706010504000
SELWAY R	ID	SELWAY	SPRING	9.2	270	1706030200100
GRANDE RONDE R	WA	GRANDE RONDE	?	9.4	188	1706010600100

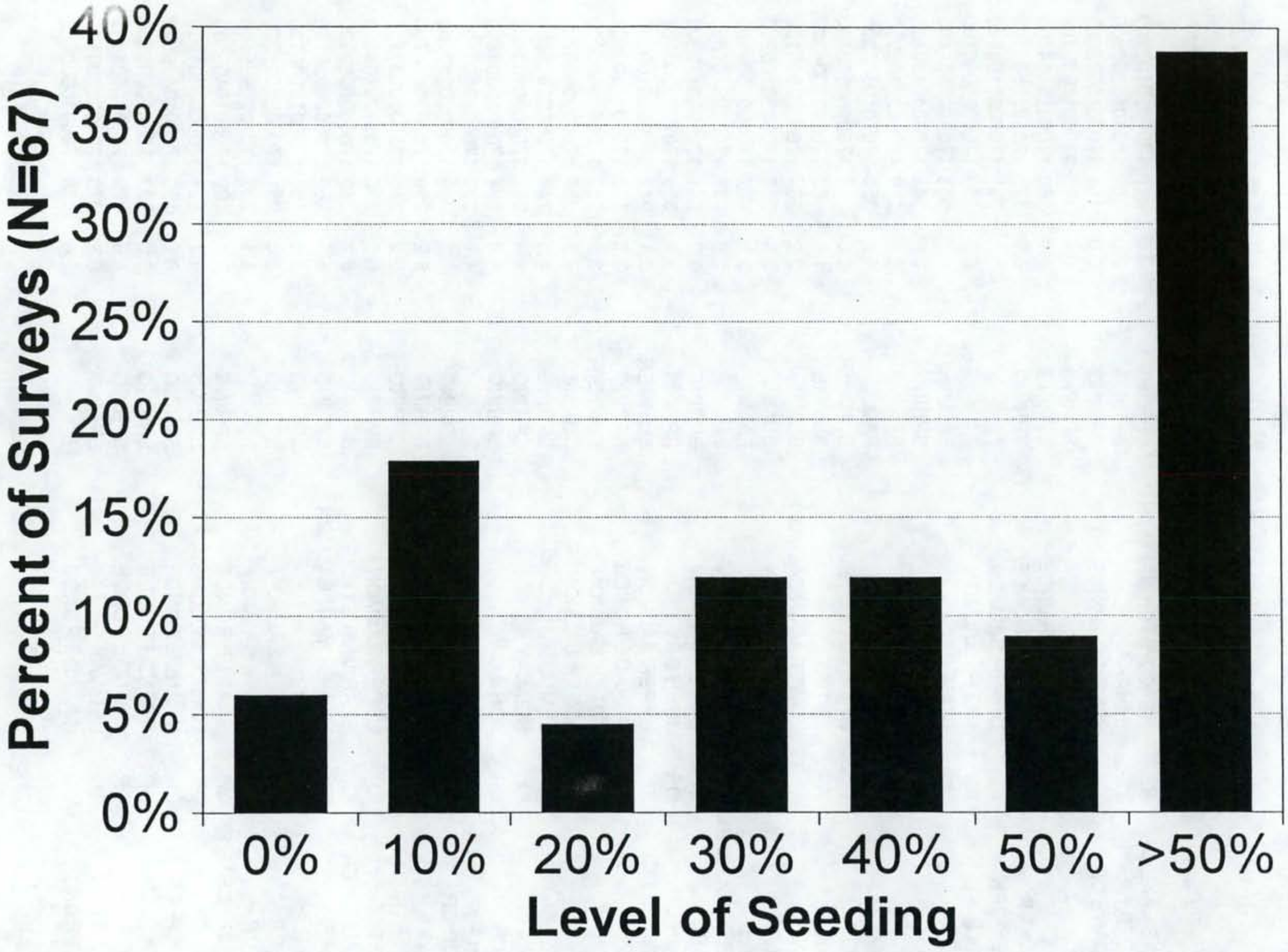

Figure 3. Frequency histogram of Columbia Basin chinook spawning escapement surveys of hatcheries returns at seeding levels grouped in ten percent increments.

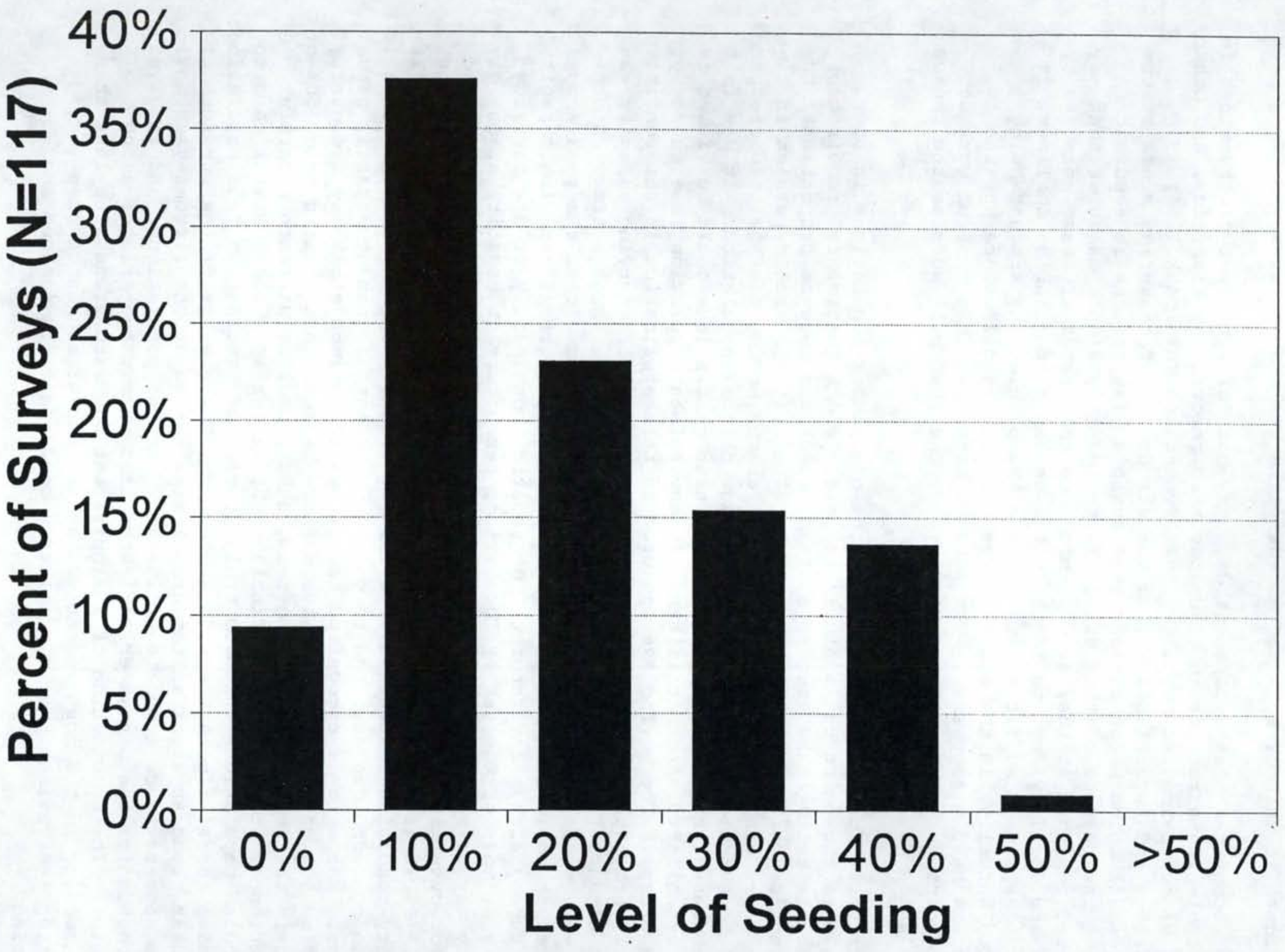

Figure 4. Frequency histogram of Columbia Basin chinook spawning escapement for those surveys classified as badly damaged and decreasing at seeding levels grouped in ten percent increments.

is recognized that the combined redd count data are not necessarily proportional to total chinook abundance in the Columbia River Basin as a whole, the redd counts are considered by the authors to be directly proportional to chinook spawning abundance in the reaches surveyed.

It is recognized that escapement data may not fully reflect fluctuations in total population size because of unaccounted removals by fisheries, and other sources of human-induced mortality (see Symons and Waldichuk 1984; Schwartzberg and Roger 1986). From the standpoint of measuring stock status of very small chinook populations, the authors view this as an academic concern. The purpose of this effort is to identify simple measures that may be related to stock status, and to inform on the status of these measures by species and locality, not to identify the causes of population declines. If numbers of spawners sufficient to sustain the populations are not being recruited annually, in the end it makes very little difference why the populations have disappeared. The authors hope that this type of work can help to point out where salmon recovery actions that will address the causes of population decline are most needed.

The criteria for classifying the status of chinook populations in the surveys were chosen to be as simple as possible. A fundamental requirement was that the criteria be abundance based, so that a measure proportional to annual egg deposition could be examined. The criteria should persist in time. Fifteen years was selected as the period for the moving average to correspond to three generation lengths for the average chinook population. It was felt that processes that were sustained over three generations could be viewed as firmly established, and significant, in any salmon population (see Cuenco et al., in press). The moving average used in this application is not sensitive to short term fluctuations in abundance. A measure of carrying capacity was seen as essential to determining the status of stocks through time. Since only a very crude measure was necessary, it was assumed that the peak annual survey count would be a minimum estimate of the historical carrying capacity of the survey site. Environmental degradation continually erodes carrying capacity at most of the survey localities, hence current capacity at any given locality may be very much lower than that of the last century. A threshold level for determining critical status is also an important measure of stock status. As an arbitrary rule, the authors chose seeding levels below 50% as a necessary condition indicative of a failing stock. Bear in mind that this is a screening technique designed to point out potential, not necessarily actual, problems. While it is recognized that a healthy salmon stock may frequently fall below 50% of full seeding, the authors believe that falling below 50% of seeding in the presence of a long-term, steady decline warrants a careful examination of stock status. Finally, the criteria of stock status need to distinguish populations that are truly at risk of extirpation from populations undergoing normal fluctuations in abundance. The measure of the strength of the linear trend apparent in the annual time series of the fifteen-year moving average appears to be adequate to identify damaged populations, and it gives error bounds for the parameter estimates that permit comparisons among localities. The combination of a significant downward trend with less than 50% of seeding in the most recently observed annual abundance pose very compelling indications that a population could be badly damaged and decreasing.

In future research we plan to add the most recent value of the fifteen-year moving average as a percentage of the maximum observed abundance to the list of criteria indicative of a damaged salmon population. A seeding level below 50%, when based on the most recent fifteen-year average, may be sufficient to identify a chronically depressed group of spawners without additional analysis of this kind of data. When based on only the most recent observation of abundance (e.g. Figs. 1 - 4, and Table 1), a seeding level below 50% needs to be combined with a significantly negative slope in the regression of average abundance on time to identify chronic damage.

Most of the chinook salmon escapement survey areas in the Columbia Basin clearly are both in chronic decline and have critically low levels of spawners. It is our inference that many of the chinook populations represented by the survey areas are in chronic decline. More than a quarter of the surveys are in the badly damaged and decreasing category (see Figure 4), while only slightly more than a quarter of all surveyed localities are above 50% seeding, regardless of trends in abundance (see Figure 1). Among the 117 badly damaged survey localities, most (82) are at, or below, 20% of seeding, the last time they were checked. Assuming that these surveys may represent chinook salmon populations, there are 82 chinook salmon populations in the Columbia Basin that have been steadily declining for the past three generations, and that have managed to achieve spawning levels of only one-fifth of the carrying capacity during the most recent spawning. If these surveys represent populations, or portions of populations, then they would clearly be candidates for accelerated recovery actions (see CBFWA 1991).

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FORECASTING AND ANALYTICAL TECHNIQUES

SESSION LEADER: PETER LAWSON*,
OREGON DEPARTMENT OF FISH AND WILDLIFE

**This session leader published information presented at the session summary. Please refer to: Lawson, P.W. 1993. "Cycles in ocean productivity, trends in habitat quality and the restoration of salmon runs in Oregon". Fisheries 18(8): 6-10.*

EVALUATING MANAGEMENT OPTIONS FOR KLAMATH CHINOOK

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ABSTRACT

Present management goals for Klamath chinook include a target harvest rate and a minimum number of natural spawners (escapement floor). I evaluated the effects of changes in management goals for the Klamath River stock of chinook salmon using a simulation model incorporating stochastic variation in recruitment, life-history, growth rate, stock assessments, and fisheries. Alternative management goals evaluated include elimination of the escapement floor, a partial escapement ceiling, and a constant escapement goal. Alternative goals were evaluated over an array of stock-recruitment parameters to examine sensitivity to current assumptions about stock productivity and equilibrium stock size. These strategies were compared to status quo management on the basis of total landings, variability in landings, the frequency with which management goals were met, and the frequency with which restrictions were placed on ocean fisheries to attempt prevention of overfishing.

Simulation results indicate that the constant escapement policy is far more sensitive to equilibrium stock size than is status quo management, and could provide a modest increase in harvest if equilibrium stock size is larger than presently assumed. The partial escapement ceiling could produce slightly increased landings if equilibrium stock size is larger, and the stock more productive than presently assumed. Elimination of the escapement floor would result in increased landings if the stock is more productive and has a smaller equilibrium size, but would lead to reduced landings if the stock is less productive than presently assumed.

INTRODUCTION

Klamath River chinook stocks comprise natural stocks from the Klamath River, Trinity River and a number of major tributaries, and hatchery stocks from two hatcheries. Current management of Klamath River stocks has been called harvest rate management and has two management goals. The first is to allow 33 to 34% of the potential spawners from each brood year to spawn, and the second is to provide a minimum of 35,000 natural spawners each year. The target harvest rate was based on the productivity from the fit of a Ricker stock-recruit relationship to spawner counts from the Shasta River (a Klamath tributary), and the escapement floor was arrived at by consensus of interested parties.

This management policy has been justified by a number of arguments. The rationale is that a constant escapement rate provides an expected harvest that is nearly as large as the MSY policy of a constant escapement goal while providing informative variation in spawning escapements that will help to better define the production of the Klamath basin in the future so the optimal number of spawners can ultimately be determined. At the same time, a constant harvest rate policy provides less variability in landings than a constant escapement policy, thus providing more stable supply to markets. The escapement floor was included as a safeguard to speed up recovery of the stocks if they should ever become depressed.

Two different changes to the current escapement goal have been proposed. In response to very large spawning escapements in 1986, 1987 and 1988, a partial ceiling on spawning escapement was proposed. Under this modified goal, when natural spawning escapement was forecast to be greater than 70,000 adults, 1/2 of the additional spawners would be allocated to harvest. Subsequently, spawning escapement has been below the escapement floor in 1990, 1991, and was projected to be below the floor in 1992. As a result of this, the utility of the escapement floor has been questioned.

The arguments supporting the current escapement goals are based on the assumptions that there is a fixed production function that can describe the Klamath basin, and that fishery managers can know the present status of the stocks and control the mortality inflicted on the stocks by the fisheries. I wanted to examine the performance of different management policies under more realistic conditions, with random variability in production, mortality rates, maturity rates, growth, and vulnerability, and managers have imprecise information about stock status and imperfect control over the fisheries.

METHODS

To evaluate these possible changes to current escapement goals, I constructed a detailed simulation model of the natural component of the Klamath basin chinook stock. The model includes components describing the population, fisheries, assessment and management processes. The population model operates with a monthly time interval using instantaneous rates. Recruitment is described by a Ricker SRR with multiplicative log-normal errors. The growth of each cohort is normally distributed about a modified Von Bertalanffy growth curve that includes seasonal growth (Pauly 1987) fitted by eye to the length distributions of aged spawners from the Sacramento River. In addition, each year there is a random deviation in growth rate. Maturation rate for each year class is drawn from a logit transform (Johnson 1987) of a normal distribution fitted to the variability in maturation rates of coded-wire-tagged (CWT) fish from basin hatcheries. A correlation between deviation in size and maturation rate is included (Hankin 1990), and maturing fish all leave the ocean at the end of August.

The fishery model contains commercial and sport troll fisheries in the ocean and a terminal fishery on the spawning run. The distribution of fishing effort within each year for the ocean fisheries is based on the average distributions of effort from Fort Bragg to Coos Bay, with independent deviations occurring in both fisheries on a monthly basis. Ocean fisheries use California size limits of 66.0 cm total length for the commercial fishery and 50.8 cm for the recreational fishery. Fishery contact rates and shaker mortality rates were taken from the management models presently used for Klamath stocks. River fisheries use average selectivities estimated from CWT fish from river fisheries on the Klamath River from 1983 to 1990 (KRTAT unpublished). Observational errors and sampling errors are included in data generated from the population and fisheries by introducing independent multiplicative lognormal errors to actual catches and spawning escapements and then ageing a random sample of specified size from each data series.

In the assessment model, cohorts are reconstructed from the observed data. This cohort model uses an annual time-step and estimates mortalities as fractions of the population. Because the assessment model is discrete and the population model is continuous, this builds up a distorted picture of the population. The model uses a data series of constant length, so as each new year's data is added to the data set, the oldest year's data is deleted. Harvest rates, selectivities and maturation probabilities are estimated, but no attempt is made to reassess the SRR or recalibrate the escapement goals. Age specific stock forecasts are made from the reconstructed stock abundances and the observed spawning runs.

The management model uses the stock forecasts and past performance of fisheries to set seasons for the ocean fisheries and quotas for the river fisheries. River fisheries are given priority as forecast abundance decreases so that, in order to protect the escapement goals, both ocean and river fisheries are reduced equally until the river fishery reaches a minimum subsistence level. Beyond this, the ocean fisheries are reduced to try to protect escapement goals. After ocean fisheries are eliminated, river fisheries are reduced until they reach a minumum level corresponding to harvest by Indians for ceremonial purposes.

Simulations were run for 1100 years with data from the last 1000 simulated years used to characterize the performance of the management strategy. Management strategies evaluated included: status quo, the partial escapement ceiling, elimination of the escapement floor, and a fixed escapement policy. Each management policy was applied to stocks driven by a variety of SRR parameters that bracketed the parameters for which the policy was intended to evaluate the sensitivity of the policy to errors in the SRR parameters.

RESULTS AND DISCUSSION

The constant escapement policy offers little improvement over status quo (Figure 1). If the current assumptions about stock production are correct, a constant escapement policy would offer about a 2% increase in total landings. In theory, the benefits from a constant escapement policy should be greater, but given the imperfect control over fishing mortalities in the ocean and the errors in river quotas that result from errors in stock forecasts, escapement is still quite variable under a constant escapement policy. If the current

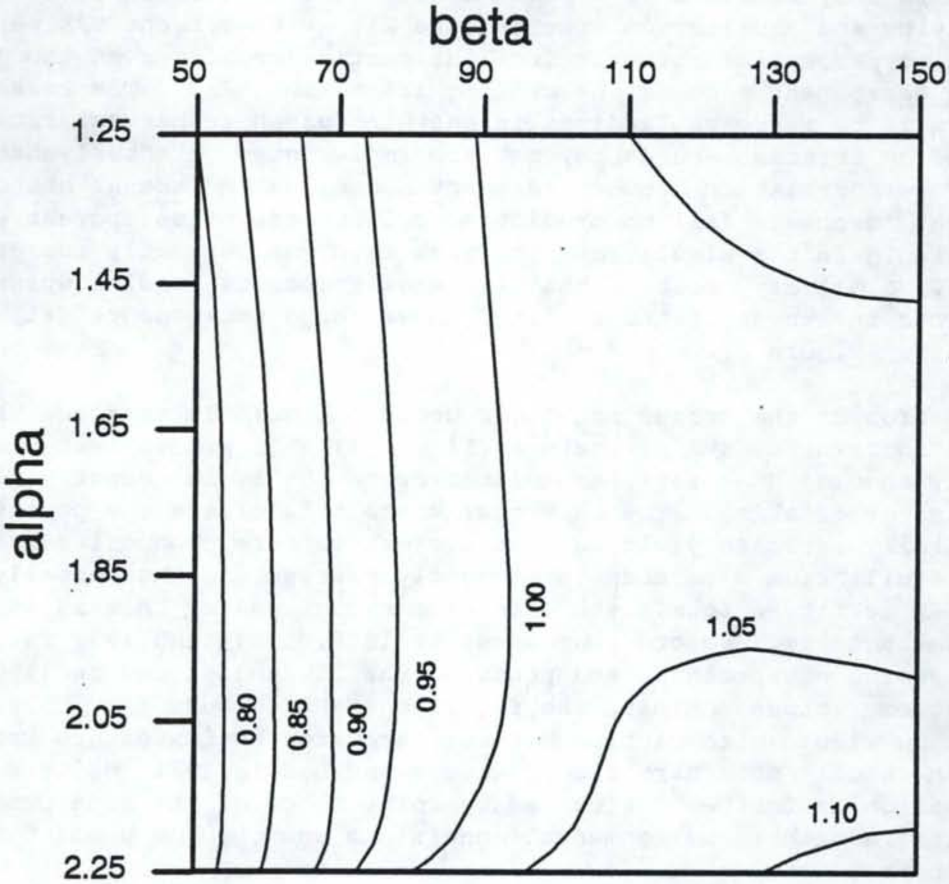

Figure 1. Expected yield from a constant escapement policy relative to status quo management. The escapement goal was chosen to maximize yield under current assumptions of stock productivity (a=1.76) and equilibrium stock size (Beta = 100).

assumptions about stock production are conservative, a constant escapement policy could increase landings by 5 to 10%. If the equilibrium stock size is smaller than presently assumed, a constant escapement policy with an escapement goal chosen as optimal for the current assumptions about stock and recruitment would result in less yield because of more frequently restricted seasons to try to meet an unrealistically high escapement goal.

The partial escapement ceiling has very limited potential to increase yield, and this is only realized if current assumptions underestimate stock productivity and equilibrium size (Figure 2). With current SRR parameters, there is no perceptible benefit from the partial ceiling even though the spawning escapement exceeds the ceiling 17% of the time. The reason that this policy fails to increase landings is that increases in harvest rate are triggered by forecast abundance, but are implemented on actual abundance. Because the correlation between forecast abundance and actual abundance is not very high, forecasts fail to predict most instances of escapement exceeding the ceiling. In the simulations, harvest rate was correctly increased 2.4% of the time. Similarly, most of the time when forecasts predict spawning escapements in excess of the ceiling, these large escapements fail to materialize (Figure 3).

Elimination of the escapement floor would potentially decrease the yield for most combinations of SRR parameters (Figure 4). If productivity is less that currently assumed, the escapement floor dramatically increases yield by preventing overfishing. The only case where this offers the potential to substantially increase yield is if the stock is more productive and with smaller equilibrium size than is presently assumed. Coincidentally, if a Ricker SRR is fitted to Klamath data from recent years, this is exactly where the parameters lie. Record abundances in 1986, 1987, and 1988 resulted from small spawning escapements, and produced the low abundances in 1990 and 1991. These extreme values dominate the fit of the SRR (Figure 5). This apparent SRR must be viewed with caution because parameter estimates are known to be biased in exactly this direction (Walters and Ludwig 1981, Walters 1985), and the situation is further confounded by coincidence of the high production years with favorable environmental conditions and the low production years with drought.

With current assumptions for SRR parameters, elimination of the escapement floor decreased the average yield from ocean fisheries by approximately 4%, inriver fisheries by 10%, and average spawning escapement by 25%. However, under the same scenario, restrictions on the ocean fisheries decreased from 27% of the years to 2.4% and complete closures of the ocean fisheries, which occurred 1.7% of the time under status quo management, were eliminated.

SUMMARY

Placing bounds, like escapement floors and ceilings, on the escapement goals of an otherwise constant harvest rate management strategy can increase the expected yield. This occurs because the hybrid strategy bears more resemblance to a constant escapement policy which theoretically produces MSY

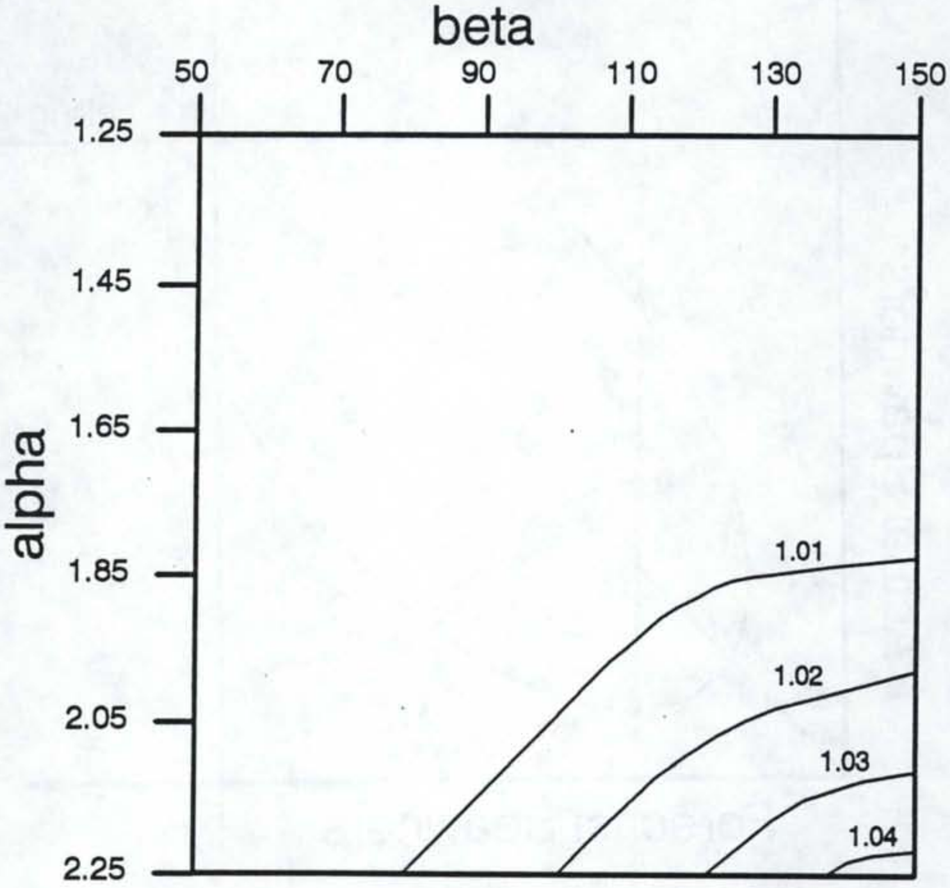

Figure 2. Expected yield resulting from adding a partial ceiling on spawning escapement relative to status quo management.

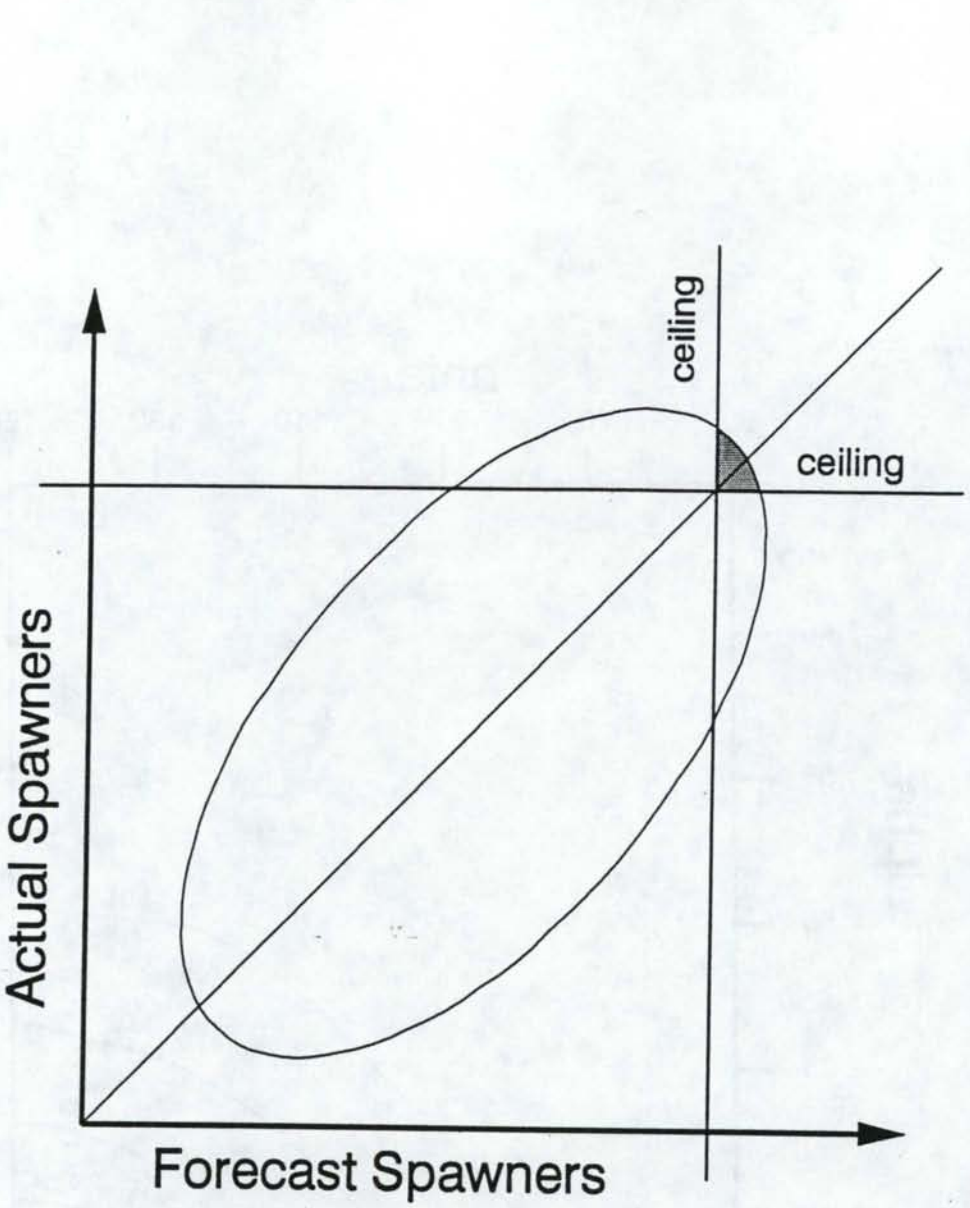

Figure 3. The effect of uncertainty on implementing an escapement ceiling. The goal is to increase harvest rates when actual escapement would be higher than the ceiling (left of the vertical line). Harvest rates are modified in response to forecast escapement (above the horizontal line). The imperfect relationship between forecasts and reality means that the fraction of the time that harvest rates are correctly altered may be very small (shaded region).

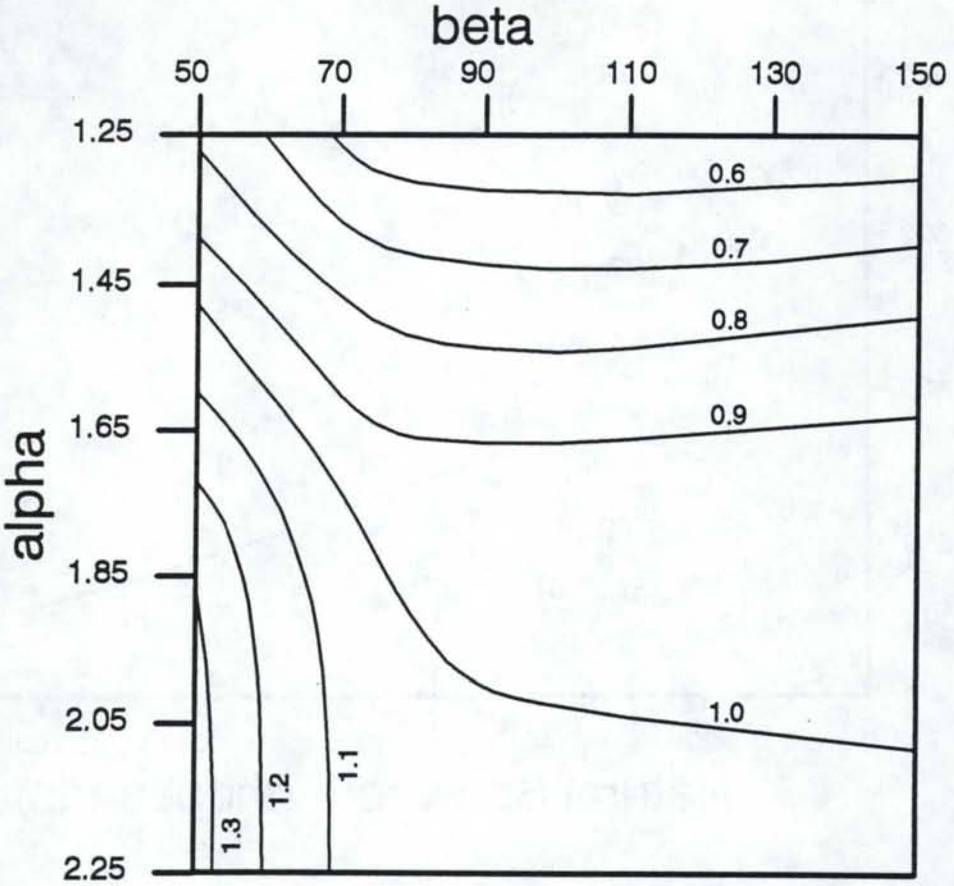

Figure 4. Expected yield resulting from removing the escapement floor relative to status quo management.

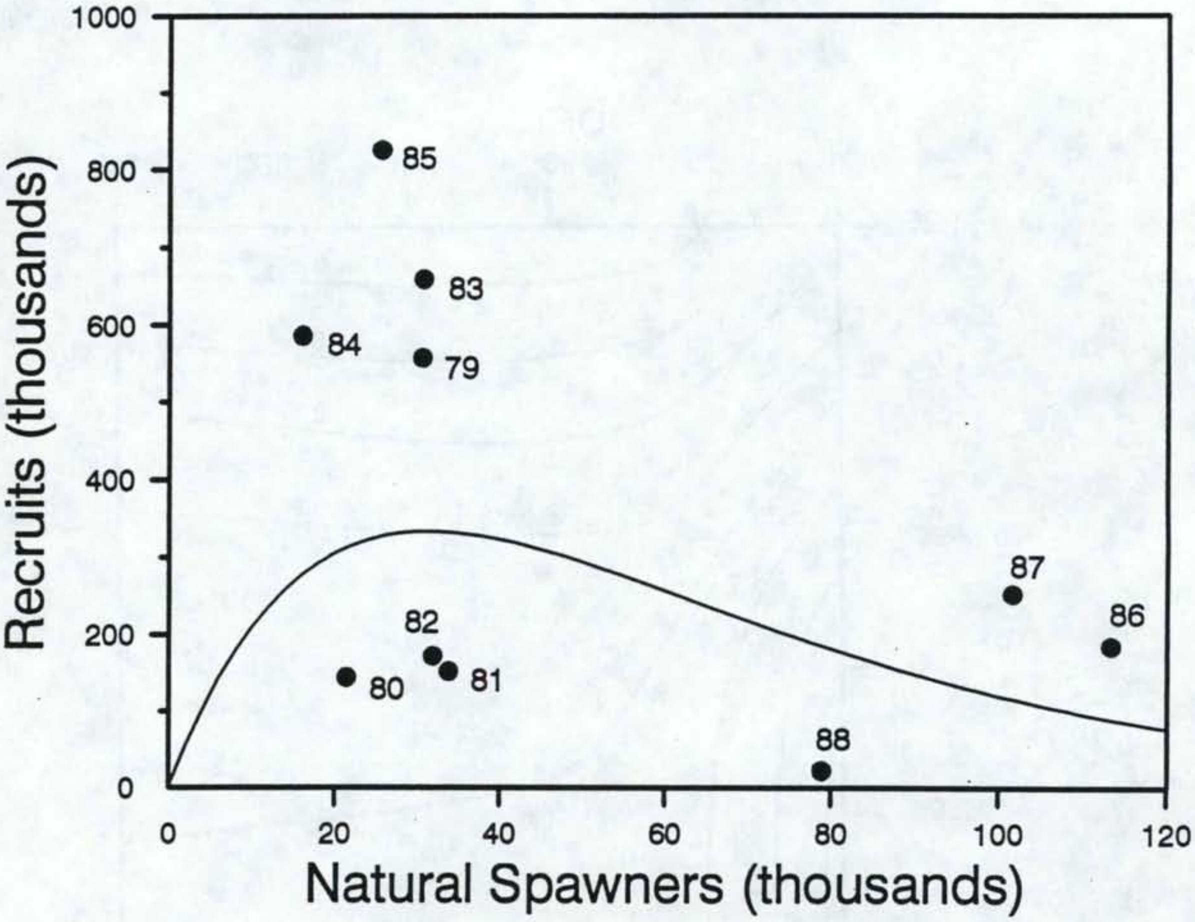

Figure 5. Apparent stock-recruit relationship for the natural component of the Klamath stock of fall-run chinook salmon. Recruits are age-2 ocean abundance of natural fish reconstructed by California Department of Fish and Game and data points are labeled by brood year.

if the goal is set at the correct level. A partial ceiling on escapement could increase landings if current estimates of stock productivity and equilibrium size are conservative, but would otherwise have no demonstrable effect. Eliminating the existing escapement floor would result in decreased landings unless the stock productivity is greater and the spawner capacity of the basin is less than presently assumed.

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MODELING THE EFFECTS OF TRANSPORTATION ASSUMPTIONS ON THE POPULATION TRENDS OF SNAKE RIVER CHINOOK SALMON: A RISK ANALYSIS

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ABSTRACT

The purpose of this paper is to address the effects of our uncertainty about the survival of transported smolts on both the downstream survival and overall population trends of Snake River spring, summer, and fall chinook salmon stocks which have been listed as threatened under the Endangered Species Act of 1972. I modeled the survival of the control releases for transportation experiments conducted from 1968 to 1986 by the National Marine Fisheries Service in the Snake and Columbia Rivers in the Columbia River Salmon Passage Model (CRiSP.0). I used the survival of the control fish and the Transportation Benefit Ration (TBR; ration of the number of transported adults captured to the number of control adults) to calculate the survival of the transported fish. I then attempted to relate the transport survival to environmental conditions such as flow and runoff volume. I used several alternative models of transport survival in CRiSP.0 to model the projected downstream survival of the smolts through 1998. I then used these survivals in the Stochastic Life Cycle Model (SLCM) to determine their effects on the projected population levels of wild Snake River spring chinook for the next 40 years. I then discuss the implications that our continually changing beliefs concerning transportation have on efforts to recover these stocks.

INTRODUCTION

Transportation of juvenile salmonid smolts has been used as a method to decrease mortality of the smolts through the hydrosystem on the Snake and Columbia Rivers since 1968. Smolts are generally intercepted at collector dams after they are guided into the bypass system by traveling screens across the turbine intakes or by other routes. The smolts are then held in raceways and loaded into barges or trucks and transported downstream to below Bonneville Dam on the Columbia River. Mass transportation has been occurring generally at McNary Dam on the Columbia River since 1979 and at Little Goose and lower Granite dams on the Snake River since 1975 and 1976, respectively. Throughout this period, researchers have attempted to evaluate the effectiveness of transportation by releasing experimental transport and control groups of smolts at various dams and comparing adult recoveries of these groups. This researcher provides a large dataset collected over a 21 year span for spring, summer, and fall chinook, and steelhead stocks on the Snake and Upper Columbia Rivers. Matthews (1992) provides a thorough review of transportation research in the Columbia River basin.

Over the years, researchers have established that, in general, transported fish contribute more adult returns to the ocean and in-river fisheries and the river recovery sites that non-transported fish. Researchers have attempted to relate the apparent mortality of transported smolts which has been indirectly observed in the transport experiments to their physiological condition (Maule et al. 1988), disease transmission in the barges (Pascho and Elliot 1989), straying of returning adults, and other factors. Another possible mechanism for explaining the survival of transported smolts is to relate their survival to the river conditions under which they were collected, since these conditions may influence their physiological condition. Knowledge of the relative survival of transported smolts is of crucial importance in the planning and execution of recovery plans for threatened stocks of Snake River chinook salmon. When any model simulating survival of smolts through the hydrosystem is used to evaluate the effects of actions to improve survival on these stocks, modelers need to simulate the survival of transported fish as realistically as possible, since by recent BPA estimates about 75% of all yearling chinook and steelhead smolts originating above Lower Granite Dam are transported. Therefore, if most of our eggs are in the transportation basket, so to speak, we need to fully take into account the survival of these transported smolts. If we do not, we risk putting all our eggs in the wrong basket, whether it be transportation or in-river migration.

Most of the juvenile hudrosystem passage models currently in use in the Columbia River basin model the survival of salmonid smolts as a function of river flow. This is primarily because Sims and Ossiander (1981) were successful in relating survival of test groups through most of the lower Snake and Columbia rivers to river flow. Therefore, for model purposes, I chose to attempt to relate the survival of transported fish to in-river migration conditions. I examined the relationship between survival and two factors which are assumed to affect the condition of migrating smolts: total runoff volume (April through July) and river flow. Total runoff was chosen since it presents a general picture of in-river migration conditions during the period when most salmonid smolts are migrating. Since this measure is not incorporated into our present hydrosystem passage models, I also used flow at the dam where the smolts were collected.

METHODS

Transportation Experiment Data

The experiments examining transportation of salmon and steelhead smolts on the Snake and Columbia River were carried out the by the National Marine Fisheries Service (NMFS) for the U.S. Army Corps of Engineers (USACE) from 1968 to 1989. the Researchers initially started experimenting with the transportation of chinook and steelhead smolts by tank truck from Ice Harbor Dam, the firs dam built on the Snake river, in 1968 (Ebel et al. 1973). They moved up the Snake River to Little Goose and Lower Granite dams as these projects were completed, and simultaneously started experimentation at McNary Dam on the columbia River below the Snake River confluence (Park and Ahteran 1985). The objective of these studies was to evaluate the effectiveness of transportation of smolts thorough recaptures of the adults of both transported smolts and smolts left to migrate in-river. To accomplish this, the researchers generally captured smolts at the dam which was being evaluated and divided the smolts into transported and non-transported groups. Smolts were divided into three general species: yearling chinook (age 1), sub-yearling chinook (age 0), and steelhead. For purposes of this analysis, yearling chinook were assumed to represent wild Snake River spring and summer chinook, and sub-yearling chinook were assumed to represent wild Snake River fall chinook.

All the smolts were then freeze-branded or otherwise visually marked and implanted with a coded wire tag (CWT). The smolts were transported and released to migrate naturally and roughly the same time. Adult counts were then gathered from three main sources: ocean and in-river fisheries, dam counts, and hatcheries. A very few recoveries were made from spawning areas. The transport benefit ration (TBR) was expressed as the ration of the proportion of transported smolts surviving to adulthood to the control (in-river) smolts surviving to adulthood. The molt-to-adult ration (SAR) was calculated as the proportion of either the transported or non-transported smolts captured as adults.

I used the data for all of the transport experimental groups when it was available for this analysis. Apparently the raw data for some of the experiments is no longer surviving. When available, I used the actual number of smolts released and the actual number of adults returning to re-estimate the TBR for many of the groups. This was done since researchers were not consistent in using the same recoveries of adults for their TBR estimates. Some researchers only used dam counts, some added in fishery recoveries, and some added in hatchery and spawning ground counts. For the purpose of consistency, I used data from all recoveries where possible. If the data were not available, I used the TBR which was reported by the researchers. I also summoned the TBR estimates from the same project in the same year when appropriate. For example, many experiments were conducted with three types of transportation, such as fresh water, brackish water, and full strength salt water as transport media. Since these were generally single truckloads of fish, there were often only a few adult recoveries from each group. Therefore, I combined such releases when conducted in the same time period into a single experimental unit. This was done in order to obtain a sufficient amount of adult recoveries for significance. I have not attempted in this analysis to re-analyze the statistical significance of the TBR data either as reported by the researchers or as combined into groups by myself.

Another issue which I have not attempted to resolve is the fact that the researchers used control groups which were not released at the same point in the river in each year. This may have been due to the lack of returns from some control groups. Control groups seem to have been released below the dam (in the tailrace) in some years and above the dam (in the forebay or even at the head of the pool) in other years. Theoretically, this should have an effect on the observed TBR, since fish released below a dam should have a higher survival rate than fish released above the dam. Since the calculation of a TBR relies on adult returns from the actual control and transport releases, I cannot adjust the survival of the control groups so that they all seem to originate at the same point for each project without compromising the integrity of the adult return data. Also, control fish may have been transported at downstream projects starting in about 1975 when mass transportation began. I have no way of knowing which control fish were transported and therefore cannot correct for this obvious bias in estimating control smolt survival.

Passage Survival Simulations

In order to investigate the relationship between transportation survival and river conditions, I had to estimate the survival of the transported fish. I used the Columbia River Salmon Passage Model (CRiSP.0; Hinrichsen et al. 1992) to simulate the smolt hydrosystem passage survival of the control groups which were left in-river. CRiSP.0 simulates the passage survival of smolts through the hydrosystem of the Snake and Columbia Rivers to below Bonneville Dam, the first dam on the Columbia. The control groups were input into the model as daily releases of smolts of the species, numbers, and releases sites reported in the literature. Again, if I could not locate the original release data, I excluded the control group and transported groups which were compared to those groups from the analysis.

River conditions were simulated on a yearly basis for the years in which experiments took place, which were most years between 1968 and 1989. Generally, I used flow and spill data for each dam from the USACE, dam instillation dates and number of turbines and screens from NMFS, and my own estimates of fish guidance efficiency from NMFS data for each dam in each year. For a complete description of dam and river parameters used in CRiSP.0 for this analysis see Fisher 1992.

Transport Survival Calculation

Once the survival of the control groups was known, I multiplied this survival by the TBR to obtain an estimate of the survival of transported smolts. This was calculated on a yearly basis by dam and species. The only exceptions to this were three releases of subyearling chinook smolts at Mcnary dam in both 1982 and 1983, which were not summed since they were designed to evaluate the effects of flow on TBR (Park et al. 1984). The transport survival then equals the predicted in-river survival times the TBR for each group of transported fish. In order to standardize transport survival for comparison with flow at different projects, I converted transport survival to survival/km, using the distance from the mouth of the Columbia to each project. Transport survival was set to 100% in cases where survival was calculated as greater than 100%.

Physical Variables

The two physical variables which were compared with transported survival were runoff volume and flow. Runoff volume in million acre-feet was obtained for the years 1968 to 1986 for the April - July period (when most smolts migrate) at Lower Granite Dam on the Snake River and at McNary Dam on the Columbia (BPA Division of Power Supply). Flow was calculated from USACE daily flow records for each dam. The period used for the flow calculation was the time during which the majority of the experimental transported smolts were released for each experiment. It was necessary to exclude some fish within a transport release since some groups were captured and released for an extremely protracted period (for example April 1 to July 31). Thus I excluded periods when only a relative few transport smolts were released from the calculation.

Analysis of the Relationship

I compared the transport survival (as survival/km), TBR, and the absolute survival of both transport and control fish (SAR) with the flow and runoff for each observation. A correlation analysis was performed between the two sets of variables with both standard parametric and non-parametric techniques. I then attempted to fit a series of regressions to the TBR, transport survival/km and flow data. I attempted to fit linear, log-linear, log-log, inverse, polynomial, and exponential functions to the data to obtain the best possible fit. I ran 50 CRiSP.0 simulations (one for each year of the simulated 1928 to 1977 50 year flow record) of 1990 hydrosystem operations (Fisher 1992) under each hypothesis I developed regarding transport survival. I then ran these same hypothesis for the future (through 1998) projected hydrosystem operation (Fisher 1992) for comparison of the effects of hydrosystem operations on smolt hydrosystem survival under each transport hypothesis.

Life Cycle Simulations

The smolt passage survival resulting from the two sets of CRiSP.0 runs was used in the Stochastic Life Cycle Model (SLCM) which was developed by Dr. Danny Lee to simulate the life cycle of anadromous salmonids (Lee and Hyman 1992). I simulated the life cycle dynamics of wild Snake River spring chinook under both 1990 conditions and future proposed conditions (Fisher et al. 1992) for 500 games during a 40 year span. The future conditions I modeled included predation control, additional screens and transportation sites, habitat improvements, and flow improvements, among others. These future conditions were chosen to provide a basis of comparison between two scenarios which result in markedly different population trends under almost any set of assumptions about transportation survival, I then compared the probability that the population would remain above 1,000 spawners under each of the hypotheses of transport survival.

RESULTS

Transportation Survival

I was able to obtain some information - either release or recovery data - representing 129 individual transport and control releases at four dams: McNary on the Columbia, and Ice Harbor, Little Goose, and Lower Granite on the Snake River. See the "Transport Experiments" section in the references for a complete list of the sources for this data. These data represented 84 separate transportation experiments: 15 for subyearling chinook, 37 for yearling chinook, and 32 for steelhead. After combining releases and excluding incomplete data, there were 63 observations: 15 for fall chinook, 22 for steelhead, and 26 for yearling chinook (Table 1). The TBR, SAR, and estimated in-river survivals of the control releases form CRiSP.0 are presented in Table 1.

Analysis of the Relationship

The mean and standard deviation of TBR, SAR, and transportation survivals for each species at each dam are shown in Table 2. The mean TBR is highest for steelhead, at 5.3:1, as was suggested by many of the researchers. The mean transport survival is highest for fall chinook, probably because all the fall chinook experiments were conducted at McNary Dam, which I expected to have higher survival than the upstream dams. I expected that the TBR would increase at upstream projects since the more dams a fish is transported around the greater the benefit to survival. this is generally apparent for spring chinook but not conclusive for steelhead (Figure 1). Tbr do increase up to Little Goose for steelhead but then decrease again at Lower Granite. The theory that TBR decrease as one moves upstream in the system is not supported by this analysis. The TBR may be smaller at points upstream of Lower Granite dam as suggested by some biologists; however, for modeling purposes, these TBR are not useful since the SLCM applies the TBR to the dams, not spawning areas.

The pattern of changes in transport survival is very consistent. One might expect survival to decrease as one moves downstream since the smolts are probably in worse conditions at the downstream projects; however, this is not apparent for any of the species. In fact, the calculated transport survival is highest at McNary for yearling chinook and lowest at Lower Granite. The same pattern is apparent for steelhead, although there is only one observation at Ice Harbor and none at Mcnary. There appears to be no consistent pattern to the control and transport SAR for yearling chinook. although one wold expect the transport SAR to be highest at the upstream projects. I also expected control SAR to decrease at the upstream projects; however, this pattern is not apparent for steelhead. Instead, the control SAR increased for steelhead at the upstream projects, and there is no consistent pattern for the transport SAR.

The results of the parametric correlation analysis are shown in Table 3. The transport SAR was significantly correlated at $\alpha = 0.1$ with both flow and runoff for spring chinook only. The control SAR was significantly correlated at $\alpha = 0.05$ with both flow and runoff for fall chinook and steelhead. the TBR was significantly correlated at α 0.05 with flow for spring chinook and with both flow and runoff for steelhead. Transport survival was significantly correlated at $\alpha = 0.05$ with runoff for spring chinook and steelhead and at $\alpha = 0.1$ with flow for steelhead. Transport survival per kilometer was significantly correlated at $\alpha = 0.05$ with flow and runoff for spring chinook and runoff for steelhead and at $\alpha = 0.1$ with flow for steelhead.

Table 1. Data used for examining the relationship between river conditions, transport survival, and benefit ratios for combined groups of transportation releases.

Species	Dam	Year	Flow (kcfs)	Runoff (maf)	Transport % Return	Control % Return	Control Survival	TBR	Transport Survival
Yearling chinook	Ice Harbor	1968	45	15.1	0.30%	0.15%	20.3%	2.07	42.06%
Yearling chinook	Ice Harbor	1968	45	15.1	0.16%	0.15%	20.3%	1.07	21.81%
Yearling chinook	Ice Harbor	1969	131	24.0	0.24%	0.19%	39.5%	1.26	49.64%
Yearling chinook	Ice Harbor	1969	131	24.0	0.13%	0.19%	39.5%	0.66	26.16%
Yearling chinook	Little Goose	1972	115	29.1	0.08%	0.08%	21.0%	1.08	22.61%
Yearling chinook	Little Goose	1972	115	29.1	0.09%	0.08%	21.0%	1.12	23.57%
Yearling chinook	Little Goose	1973	55	12.0	0.31%	0.02%	4.1%	13.76	56.43%
Yearling chinook	Little Goose	1973	55	12.0	0.42%	0.02%	4.1%	18.39	75.42%
Yearling chinook	Little Goose	1976	142	27.3	0.04%	0.02%	22.5%	1.78	39.99%
Yearling chinook	Little Goose	1976	142	27.3	0.03%	0.02%	22.5%	1.23	27.66%
Yearling chinook	Little Goose	1978	85	22.4	0.01%	0.01%	22.6%	0.74	16.67%
Yearling chinook	Little Goose	1978	85	22.4	0.00%	0.01%	22.6%	0.15	3.46%
Yearling chinook	Lower Granite	1975	129	28.3	0.64%	0.31%	18.7%	2.02	37.84%
Yearling chinook	Lower Granite	1976	143	27.3	0.02%	0.04%	15.6%	0.64	10.04%
Yearling chinook	Lower Granite	1976	143	27.3	0.04%	0.04%	15.6%	1.02	15.89%
Yearling chinook	Lower Granite	1973	89	55.4	0.12%	0.01%	2.7%	8.51	22.97%
Yearling chinook	Lower Granite	1978	89	22.4	0.07%	0.01%	2.7%	5.32	14.36%
Yearling chinook	Lower Granite	1979	82	17.0	0.04%	0.01%	6.4%	3.42	21.92%
Yearling chinook	Lower Granite	1986	111	22.8	0.16%	0.10%	26.5%	1.58	41.75%

Species	Dam	Year	Flow (kcfs)	Runoff (maf)	Transport % Return	Control % Return	Control Survival	TBR	Transport Survival
Yearling chinook	Lower Granite	1989	86	18.0	0.06%	0.02%	17.9%	2.46	44.10%
Yearling chinook	McNary	1979	203	66.2	0.02%	0.04%	31.5%	0.60	18.86%
Yearling chinook	McNary	1979	203	66.2	0.03%	0.04%	31.5%	0.67	21.24%
Yearling chinook	McNary	1980	209	80.6	0.01%	0.00%	37.9%	2.12	80.21%
Yearling chinook	McNary	1980	209	80.6	0.02%	0.00%	37.9%	4.55	100.00%
Yearling chinook	McNary	1986	265	78.3	0.02%	0.03%	52.1%	0.73	37.97%
Yearling chinook	McNary	1987	206	59.9	0.24%	0.14%	41.1%	1.73	71.11%
Subyearling chinook	McNary	1978	309	84.4	1.06%	0.20%	22.6%	5.16	100.00%
Subyearling chinook	McNary	1979	129	66.2	0.51%	0.08%	22.9%	6.29	100.00%
Subyearling chinook	McNary	1980	160	80.6	0.58%	0.16%	28.7%	3.63	100.00%
Subyearling chinook	McNary	1981	219	81.7	0.41%	0.06%	34.2%	6.64	100.00%
Subyearling chinook	McNary	1982	404	103.7	0.30%	0.30%	48.3%	0.99	47.87%
Subyearling chinook	McNary	1982	197	103.7	1.10%	0.22%	41.3%	4.88	100.00%
Subyearling chinook	McNary	1982	284	103.7	0.40%	0.31%	40.4%	1.30	52.46%
Subyearling chinook	McNary	1983	199	88.4	0.52%	0.27%	38.7%	1.90	73.48%
Subyearling chinook	McNary	1983	199	88.4	0.63%	0.27%	38.7%	2.30	89.16%
Subyearling chinook	McNary	1983	194	88.4	0.55%	0.17%	38.0%	3.22	100.00%
Subyearling chinook	McNary	1963	194	88.4	0.37%	0.17%	38.0%	2.18	82.89%
Subyearling chinook	McNary	1983	214	88.4	0.52%	0.10%	39.8%	5.44	100.00%
Subyearling chinook	McNary	1983	214	88.4	0.62%	0.10%	39.8%	6.46	100.00%
Subyearling chinook	McNary	1986	168	78.3	0.14%	0.07%	30.7%	2.05	62.89%

Species	Dam	Year	Flow (kcfs)	Runoff (maf)	Transport % Return	Control % Return	Control Survival	TBR	Transport Survival
Subyearling chinook	McNary	1987	110	59.9	0.52%	0.14%	18.6%	3.68	68.37%
Steelhead	Ice Harbor	1969	138	24.0	0.50%	0.18%	39.1%	2.75	100.00%
Steelhead	Little Goose	1977	44	8.5	0.12%	0.02%	2.7%	6.40	17.29%
Steelhead	Little Goose	1977	44	8.5	0.15%	0.02%	2.7%	8.48	22.89%
Steelhead	Little Goose	1973	56	12.0	1.92%	0.14%	4.7%	13.39	62.94%
Steelhead	Little Goose	1973	56	12.0	1.94%	0.14%	4.7%	13.50	63.47%
Steelhead	Little Goose	1978	89	22.4	2.03%	0.44%	22.3%	4.61	100.00%
Steelhead	Little Goose	1978	89	22.4	2.07%	0.44%	22.3%	4.70	100.00%
Steelhead	Little Goose	1972	115	29.1	1.27%	0.41%	22.3%	3.12	69.50%
Steelhead	Little Goose	1972	115	29.1	1.39%	0.41%	22.3%	3.43	76.45%
Steelhead	Little Goose	1976	142	27.3	1.15%	0.30%	24.5%	3.79	92.97%
Steelhead	Little Goose	1976	142	27.3	1.19%	0.30%	24.5%	3.92	96.04%
Steelhead	Lower Granite	1977	44	8.5	0.21%	0.02%	2.7%	11.71	31.63%
Steelhead	Lower Granite	1977	44	8.5	0.22%	0.02%	2.7%	12.33	33.29%
Steelhead	Lower Granite	1979	82	17.0	1.71%	0.96%	7.1%	1.78	12.66%
Steelhead	Lower Granite	1989	88	18.0	0.54%	0.25%	28.2%	2.19	61.64%
Steelhead	Lower Granite	1978	100	22.4	2.28%	0.44%	22.3%	5.17	100.00%
Steelhead	Lower Granite	1978	100	22.4	2.15%	0.44%	22.3%	4.86	100.00%
Steelhead	Lower Granite	1980	105	20.3	0.34%	0.20%	18.1%	1.71	31.01%
Steelhead	Lower Granite	1986	113	22.8	1.16%	0.58%	33.0%	1.99	65.72%
Steelhead	Lower Granite	1975	131	28.3	1.62%	0.45%	19.2%	3.61	69.30%

Species	Dam	Year	Flow (kcfs)	Runoff (maf)	Transport % Return	Control % Return	Control Survival	TBR	Transport Survival
Steelhead	Lower Granite	1976	143	27.3	0.56%	0.35%	18.8%	1.58	29.74%
Steelhead	Lower Granite	1976	143	27.3	0.47%	0.35%	18.8%	1.33	25.05%

Table 2. Mean and standard deviation of the transport benefit ratio and transport survival for yearling and subyearling chinook and steelhead overall and for each project.

Species	Dam	Variable	N	Mean	Std Dev
Yearling chinook					
	All				
		Transport SAR	26	0.13%	0.15%
		Control SAR	26	0.07%	0.08%
		Control surv.[a]	26	23.00%	13.13%
		TBR	26	3.03	4.29
		Transport surv.	26	36.30%	23.80%
	McNary				
		Transport SAR	6	0.06%	0.09%
		Control SAR	6	0.04%	0.05%
		Control surv.	6	38.67%	7.62%
		TBR	6	1.73	1.52
		Transport surv.	6	54.9%	33.63%
	Ice Harbor				
		Transport SAR	4	0.21%	0.08%
		Control SAR	4	0.17%	0.03%
		Control surv.	4	29.90%	11.09%
		TBR	4	1.27	0.59
		Transport surv.	4	39.92%	13.12%
	Little Goose				
		Transport SAR	8	0.12%	0.16%
		Control SAR	8	0.03%	0.03%
		Control surv.	8	17.55%	8.33%
		TBR	8	3.12	2.63
		Transport surv.	8	33.22%	23.18%
	Lower Granite				
		Transport SAR	8	0.14%	0.20%
		Control SAR	8	0.07%	0.10%
		Control surv.	8	13.26%	8.51%
		TBR	8	3.12	2.63
		Transport surv.	8	26.11%	13.28%
Subyearling chinook					
	McNary				
		Transport SAR	15	0.55%	0.25%
		Control SAR	15	0.17%	0.09%
		Control surv.	15	34.71%	8.30%
		TBR	15	3.74	1.95
		Transport surv.	15	85.14%	19.19%
Steelhead					
	All				
		Transport SAR	15	0.55%	0.25%
		Control SAR	15	0.17%	0.09%
		Control surv.	22	17.51%	10.64%
		TBR	15	3.74	1.95
		Transport surv.	22	61.87%	31.43%
	Ice Harbor				
		Transport SAR	1	0.50%	---
		Control SAR	1	0.18%	---
		Control surv.	1	39.10%	---
		TBR	1	2.75	---
		Transport surv.	1	>1	---
	Little Goose				
		Transport SAR	10	1.32%	0.72%
		Control SAR	10	0.26%	0.17%
		Control surv.	10	15.30%	10.04%
		TBR	10	6.53	3.97
		Transport surv.	10	70.15%	30.07%
	Lower Granite				
		Transport SAR	11	1.02%	0.79%
		Control SAR	11	0.37%	0.26%
		Control surv.	11	17.56%	9.77%
		TBR	11	4.39	4.00
		Transport surv.	11	50.91%	30.25%

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[a] surv. = survival

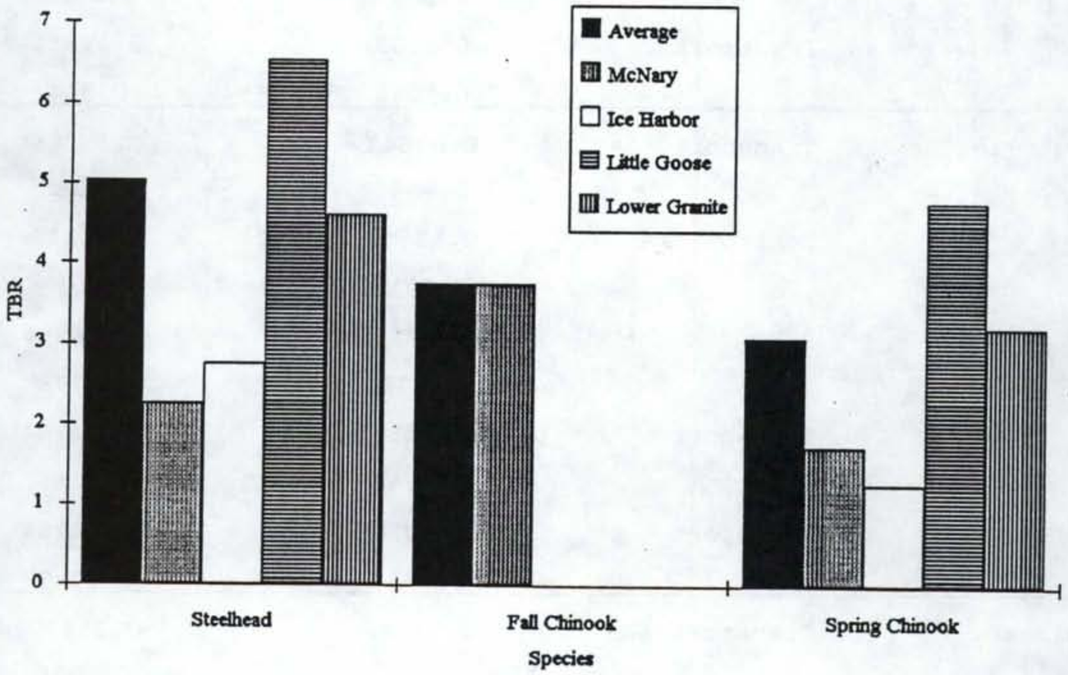

Figure 1. Transportation benefit ratios for each species for all projects and each individual project.

Table 3. Correlations between transport and control SAR, TBR, transport survival, and transport survival/km and flow and runoff for each species. The top number is the Pearson Correlation Coefficient $R = 0$. * = significant at $\alpha = 0.1$; ** = significant at $\alpha + 0.05$.

Species	Variable	Correlation with:			
		Flow		Runoff	
Yearling chinook	Transport SAR	-0.33992		-0.33856	
		0.0893	*	0.0907	*
	Control SAR	-0.08274		-0.18186	
		0.6878		0.3739	
	TBR	-0.40062		-0.28266	
		0.0425	**	0.1618	
	Transport	0.25937		0.40138	
		0.2007		0.0421	**
	Transport	0.5559		0.39811	
		0.0032	**	0.044	**
Subyearling	Transport SAR	0.02643		0.15286	
		0.9255		0.5865	
	Control SAR	0.55655		0.64119	
		0.0312	**	0.01	**
	TBR	-0.29898		-0.36594	
		0.279		0.1798	
	Transport	-0.38502		-0.22284	
		0.1564		0.4247	
	Transport	-0.38502		-0.22284	
		0.1564		0.4247	
Steelhead	Transport SAR	0.11585		0.31477	
		0.6077		0.1536	
	Control SAR	0.44647		0.55409	
		0.0373	**	0.0075	**
	TBR	-0.77048		-0.75622	
		0.0001	**	0.0001	**
	Transport	0.40539		0.51371	
		0.0612	*	0.0145	**
	Transport	0.41249		0.49992	
		0.0564	*	0.0178	**

The models TBR = flow + $flow^2$ ($P > F = 0.034$, $R^2 = 0.25$; Figure 2) and trans. surv./km = flow + $flow^2$ ($P>F = 0.046$, $R^2 = 0.24$; Figure 3) were significant but week for yearling chinook. The models TBR = flow + $flow^2$ ($P>F<0.0001$, $R^2 = 0.68$; Figure 2) and trans. surv./km = flow + $flow^2$ ($P>F+0.02$; $R^2 = 0.24$; Figure 3) were significant for steelhead. No model of transportation survival/km or TBR and flow was significant for fall chinook.

Transport Survival Hypotheses

I developed four hypotheses regarding transport survival form the above analysis. I only used spring (age 1) chinook, since the SLCM has not been parameterized for Snake River steelhead. Due to the fact that none of the models of either transport survival/km or TBR and flow were particularly significant for spring chinook, ny hypotheses dealt with constant rates of survival regardless of flow:

1. Transport survival = 95% (Table 4). From USACE sea water holding experiments (Matthews et al. 1985), direct mortality of barge transported fish was about 5% after 43 days. Therefore, I used 95% to simulate direct mortality effects only. The argument has been made that the survival we commonly accept for in-river fish, and which determines our transported fish survival, is greatly underestimated. This in turn underestimates the survival of the transported fish. This argument was recently reinforced by a re-analysis of the flow-survival relationship from Sims and Ossiander 1981 (C. Steward, Mobrand Biometrics, draft manuscript), which suggests that the researchers overestimated dam mortality rates in their release-recapture experiments.

2. Transport survival = average from 1986-1989 experiments (Table 4). Some researchers have suggested that the only transport experiments that are valid for today's collection and handling conditions are those performed in 1986 and later years (Matthews et al. 1990). Therefore, I used the average transport survival from those experiments for each species int he CRiSP.0 model. This hypothesis results in the highest transport survivals of the three hypotheses using the experimental data.

3. Transport survival = average by dam from all experiments (Table 4). I used this hypothesis as and alternative to hypothesis 2. This results in the lowest survival in the Snake but moderate survival from McNary of the last three hypotheses.

4. Transport survival = average from all experiments (Table 4). I used this hypothesis as the simplest model of transport survival, which sets the survival to the same value at each project, since some models of mainstem passage can only accept one value for the survival of transported fish. This results in the lowest survival at McNary and moderate survival in the Snake.

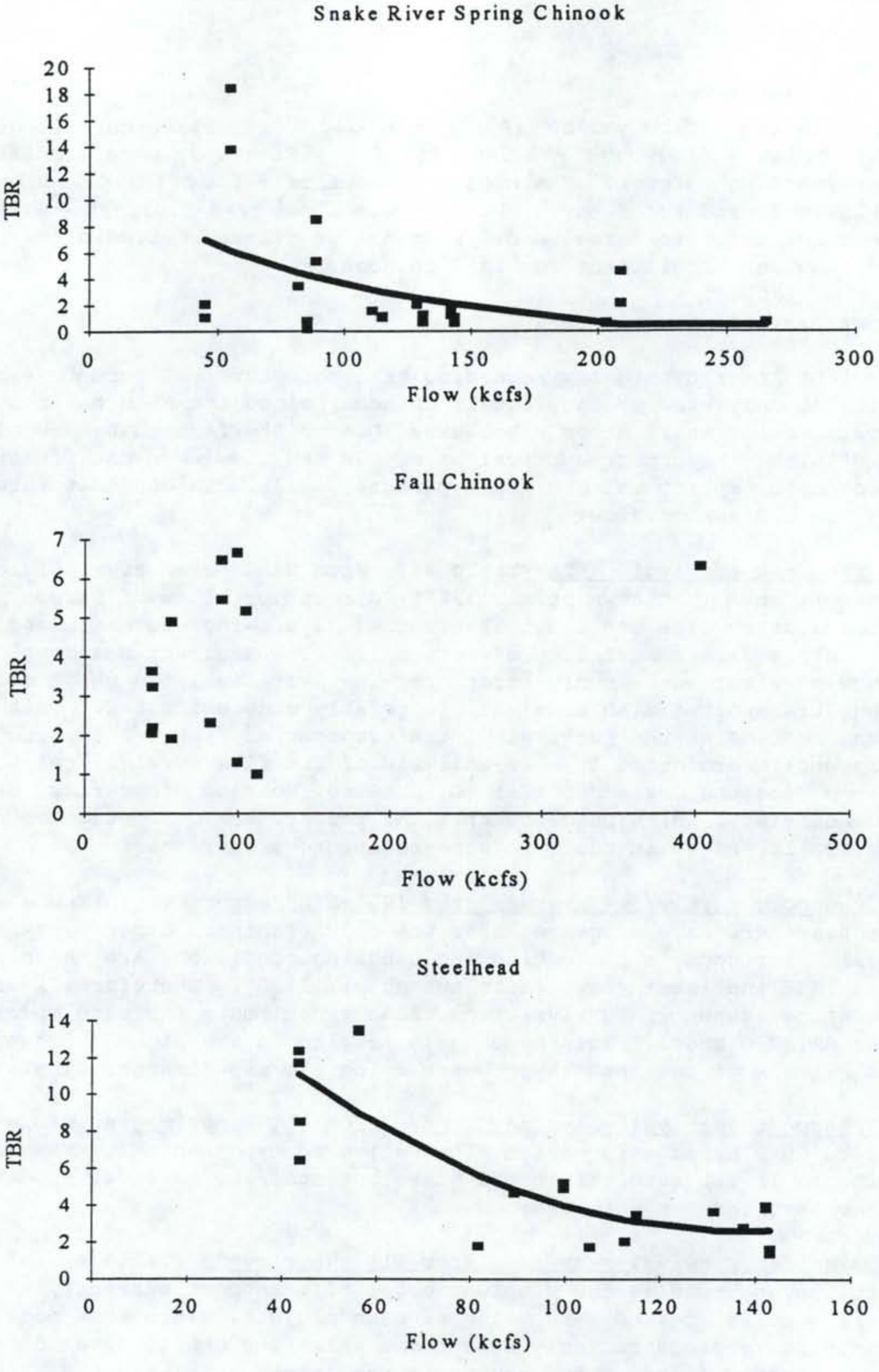

Figure 2. Transportation benefit ratio and flow for Snake River spring chinook and steelhead, and Columbia River fall chinook. The most significant regression mentioned in the text is represented by a solid line.

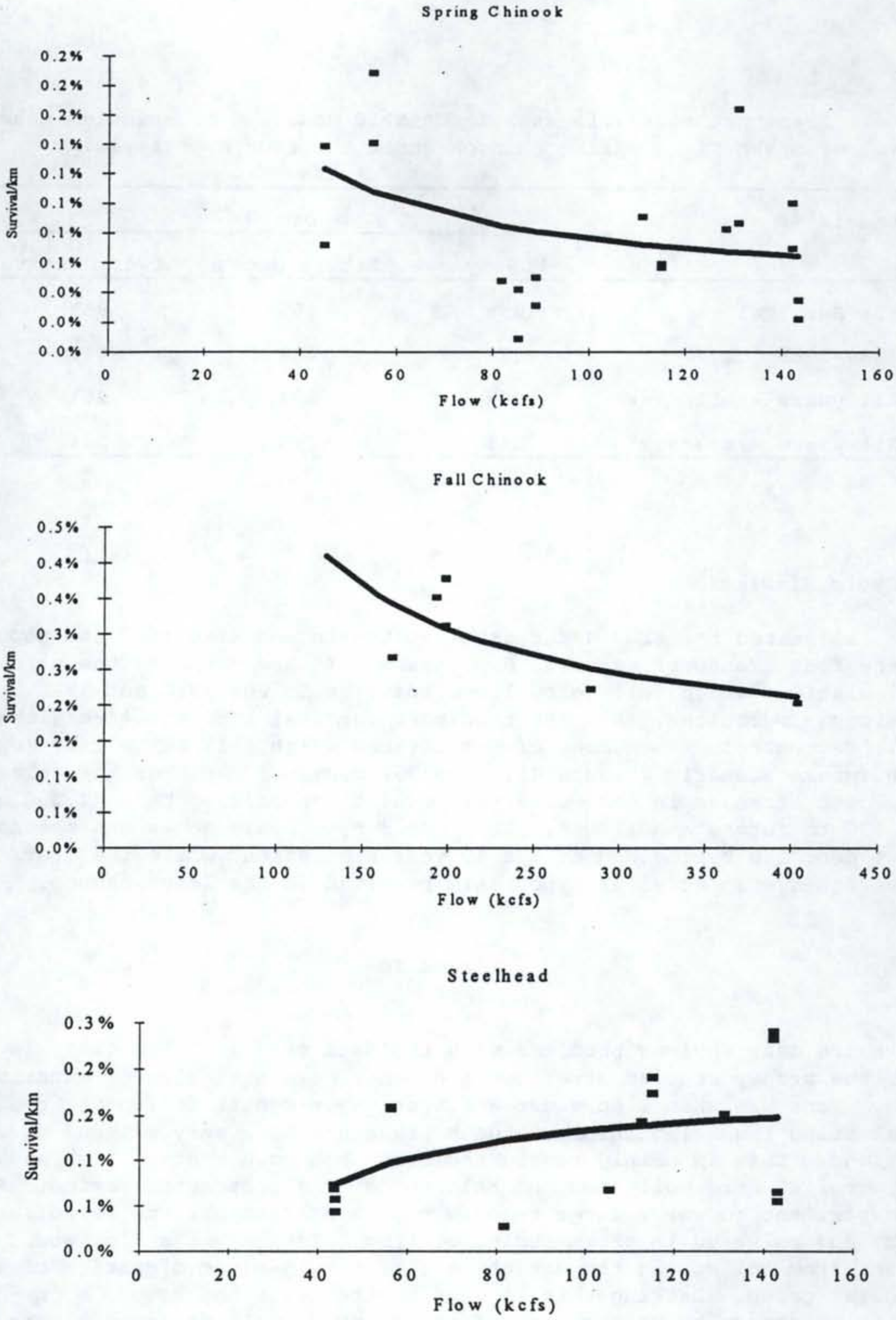

Figure 3. Transportation survival (percent survival per kilometer) and flow for Snake River spring chinook and steelhead, and Columbia River fall chinook. The most significant regression mentioned in the text is represented by a solid line.

Table 4. Transport survivals used in CRiSP.0 modeling of mainstem passage survival of Snake River spring chinook under the four hypotheses.

Hypothesis	Dam		
	McNary	Little Goose	Lower Granite
1. 95% Survival	95%	95%	95%
2. 1986-1989 Only	55%	43%	43%
3. All years - all dams	46%	33%	26%
4. All years - average	34%	34%	34%

Life Cycle Simulations

I re-calibrated the SLCM under each hypothesis and simulated the populations with the four transport survival hypotheses. I then compared the percent of 500 simulations which fell below 1,000 spawners in the 1990 and 1993 conditions simulations. All the transport survival hypotheses resulted in an overall decrease in the number of populations which fell below 1,000 spawners in the future scenario (Figure 4). The 95% survival hypothesis resulted in the largest decrease in the number of populations falling below 1,000 spawners from 1990 to future conditions. The modern hypothesis shows the second largest decrease by the end of the 40 year simulation, while the average and the average-by-dam survival hypothesis resulted in the least change.

DISCUSSION

There are many obvious problems with the data used for this analysis. First, the transportation experiments did not have a consistent experimental design. None was what I consider a robust experimental design for the purpose of evaluating the relationship between transportation survival and river conditions. This is mainly due to the fact that both control fish and transport fish were collected and released over a protracted period. For this type experiment to work, large release groups of fish need to be collected, marked, and released in short period of time. Ideally, the flow would be held constant from collection through the end of the in-river migration of a particular group. Lacking this kind of control over the river, a one-day delay from capture to release of and entire group wold be desirable so that similar river conditions would prevail for the transported fish as well as the controls. Since most of the control groups were released over a period of a month or more, the flows affecting these fish would be expected to vary greatly. This has a negative effect when computing and average flow for comparison with the transport survival.

Although combining the transport groups into single releases may have been necessary in order to add statistical significance to the TBR, it did result in a large decrease in the number of observations. By the time the data was broken down by species and dam, some cells had one to a very few observations. This made it impossible to compare TBR between dams within a species. I could no locate release and/or recapture data for many of the experiments, especially those between 1969 and 1974, so these experiments were not included in the analysis. Since I did not have the time to re-calculate TBR based on a common recapture point, some of the TBR values I used were calculated to all recapture points, some to only the dam of release, and some to all dams. Fortunately most of the TBR were calculated to a standard point for each species at each project. For example all TBR values reported for Snake River chinook releases were calculated to Lower Granite Dam.

Some problems also occurred in the modeling. The flow-survival relationship contained in CRiSP.0 determines the survival of the control fish which in turn determines transport survival. Therefore, the transport survival-flow relationship observed for some species may just be an artifact of the passage model. I also could not simulate free-flowing river reaches in the cases where dams did not yet exist (e.g. Lower Granite before 1975) because I do not know what the flow-mortality relationship wold be in free-flow reaches.

The analysis of transportation benefits and survivals indicates some consistent trends. In almost all cases both TBR and survivals were higher for the larger steelhead smolts than for spring chinook at the same dams, which suggests a relationship with smolt size. Most biologists have accepted the theory that survival of in-river migrants can vary with flow and runoff conditions; therefore, it should come as no surprise that the survival of in-river migrants can vary with flow and runoff conditions; therefore, it should come as no surprise that the survival of transported smolts can vary with river conditions as well, since the transported smolts experience the same river conditions prior to being collected. The significant correlations between TBR, SAR, flow and runoff for both spring chinook and especially steelhead lend support to the theory that transportation benefits can vary with flow. However, despite the large number of transportation experiments which took place at McNary Dam for fall chinook, I could only detect a significant relationship between river conditions and survival for the control releases. I cannot explain why fall chinook smolts would not be affected by transportation in the same manner as spring chinook smolts. I am also surprised that the transport SAR was significantly higher at lower flow and runoff for yearling chinook smolts. This does not support the theory of higher transport survival at higher flows.

One would expect that the adults from smolts transported from upstream dams would stray at a greater rate than those collected at downstream dams since they would be exposed to a smaller portion of the natural migration path. Therefore, I was surprised that I could not confirm the theory that TBR decrease at upstream dams. O the four hypotheses I proposed for modeling transportation survival, I believe that the survivals from the modern experiments have the most validity. Although a cursory time-series analysis did not show significant time trends in any of the measures of transportation success, I still believe that the conditions present in the collection and transportation systems have improved dramatically since the early (pre-1986)

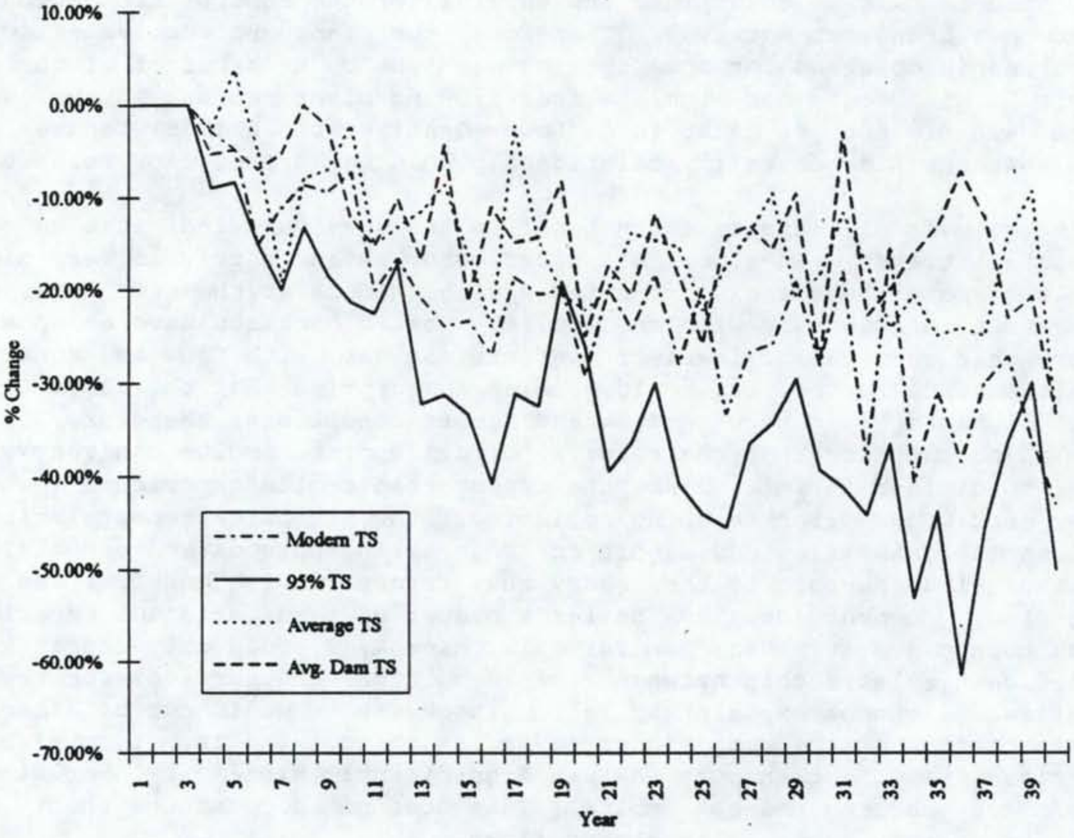

Figure 4. Percent change in number of Snake River spring chinook populations below 1,000 spawners in each year from 1990 conditions to future conditions in SCLM for each transportation survival hypothesis.

transportation experiments.

The results of the life-cycle simulations demonstrate that the transportation hypothesis used has a demonstrable effect on the molded trend of a composite Snake River wild spring chinook population. The number of simulations which fell below 1,000 spawners in each year of the simulation was almost always lowest for the 95% survival hypothesis. In fact, the order of the trends in Figure 4 corresponds closely with the absolute value of the transportation survival used. Thus the higher survivals showed the populations responding most favorably to the improvements I modeled.

CONCLUSIONS

1. This analysis supports the general conclusion that the benefits from transportation are greater at lower flows, but only for spring chinook and steelhead transported from the Snake River.
2. I cannot conclude that the actual transport survival was greater at higher flows, since I could not measure this parameter.
3. The weak correlations between both transport and control SAR and flow also do not support traditional flow-survival hypotheses for Snake River spring chinook.
4. The analysis supports the flow-survival hypotheses for in-river fall chinook and steelhead migrants.
5. The life-cycle modeling demonstrates the large effect that our beliefs about transportation can have on model predictions of changes in Snake River spring chinook populations over time. The greater the transportation survival I modeled, the greater the response of the population to the mitigation actions I modeled.
6. The greatest risk to the Snake River spring chinook population lies in modeling a transportation survival which is higher than the actual survival. This could lead one to believe that a set mitigation actions may be sufficient to recover the population when in fact they may not be.

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A COMPARISON OF PRODUCTIVITIES FOR SNAKE RIVER AND LOWER COLUMBIA RIVER SPRING AND SUMMER CHINOOK STOCKS

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ABSTRACT

Loss of productivity due primarily to hydropower development in the Columbia River Basin prompted the 1992 listing of Snake River spring/summer and fall chinook salmon under the Endangered Species Act. State fishery agencies in Idaho, Oregon and Washington and the Columbia River Intertribal Fisheries Commission are jointly developing an analytical framework to systematically evaluate the various combinations of proposed management actions to assess Snake River chinook rebuilding. The productivity of a stock must be considered directly to analyze the performance of alternative regulatory strategies. Therefore, we estimated potential productivity of Snake River spring and summer chinook using two methods. First, we reconstructed spawner and recruit information on a brood year basis for 12 Snake River spring and summer chinook populations, and fit the data to recruitment models for the period before major hydropower development (brood years 1957-69). We also estimated productivity for a downriver stock with similar attributes (Warm Springs River) that has not been exposed to the same high levels of density-independent mortality. Pre-development productivity of most Snake River stocks, on average, closely matched the recent productivity of the Warm Springs River (respective intrinsic rates of increase, 2.34 and 2.35). Productivity of South Fork Salmon River summer chinook, which suffered severe habitat degradation in the 1960s, was considerably lower (1.23). Recent recruitment of Snake River populations has been highly variable, and is related significantly to the mainstem velocities experienced during the smolt migration when density dependence is taken into account. Empirical recruitment data should be an essential element in validating the parameters, assumptions and performance of hydrosystem smolt survival models.

INTRODUCTION

The critical status of Columbia River Basin chinook salmon stocks has been the focus of many efforts and studies over the past 20 years. In 1980, passage of the Northwest Power Act mandated balance between fish and wildlife and power interests. In addition, in 1984 the U.S. and Canada signed the Pacific Salmon Treaty and formed the Pacific Salmon Commission (PSC) to institute a coastwide chinook conservation program. But in spite of measures contained in the Northwest Power Planning Council's Columbia River Basin Fish and Wildlife Program (NPPC 1987), and reductions in ocean chinook harvest rates the Snake River chinook stocks continued to decline to extremely low levels (Blum and Simrin 1991). In 1992 the National Marine Fisheries Service (NMFS) listed Snake River spring/summer chinook as one threatened species, and also listed Snake River fall chinook as threatened.

Improved smolt survival is critical to recovery and rebuilding of chinook runs in the Snake River (CBFWA 1991). The development and operation of the hydroelectric system on the Columbia and Snake rivers completely changed the hydrological conditions under which chinook have evolved. The building of dams dramatically increased the cross-sectional area of the rivers, increasing the mean and the variability of water particle travel time (Fig. 1.; Idaho Department of Water Resources). In addition to migration delay for smolts (Sims and Ossiander 1981; Bergren and Filardo 1991), hydroelectric projects caused site specific mortalities and delays (Raymond 1979, 1988).

An analytical framework is needed to systematically evaluate the various combinations of proposed management actions to assess Snake River chinook recovery and rebuilding. State fishery agencies in Idaho, Oregon and Washington and the Columbia River Intertribal Fisheries Commission are jointly developing a system of models that relies to a large degree on empirical recruitment data and estimates of stock productivity.

The productivity of a stock should be considered directly to analyze the performance of an alternative recovery or rebuilding objective (PSC 1989). Therefore, the recovery objectives and anticipated responses of a stock are going to depend on estimating potential productivity of the stock. Without estimates of stock productivity, the combinations of changes in mortality rates (at various life stages) needed to recover or rebuild stocks will be extremely difficult to determine.

One approach to estimating a stock's potential productivity is to reconstruct spawner and recruit information on a brood year basis and fit the data to recruitment models (e.g., Ricker 1954, 1975; Beverton and Holt 1957). However, large amounts of unaccounted independent mortality in these data sets (from hydroelectric development) will grossly underestimate the potential productivity. One solution is to estimate the productivity of a stock using the data set prior to the large increases in independent mortality. An alternative is to use the estimates of intrinsic productivity for a stock with similar attributes that has not been exposed to the same high levels of independent mortality (i.e., downriver spring chinook stock) to represent a Snake River spring or summer chinook stock. This approach has been used for evaluating Snake River fall chinook management actions relative to rebuilding (Schaller and Cooney 1992). The intrinsic rate of increase from a lower Columbia River stock, coupled with an estimate of spawning and rearing habitat capacity (i.e., spawners needed at either maximum sustainable yield (MSY) or maximum sustainable production (MSP)) would yield a reasonable approximation for a production function.

Smolt survival models used to evaluate mainstem management actions can be highly sensitive to both the form of the relationship between survival and

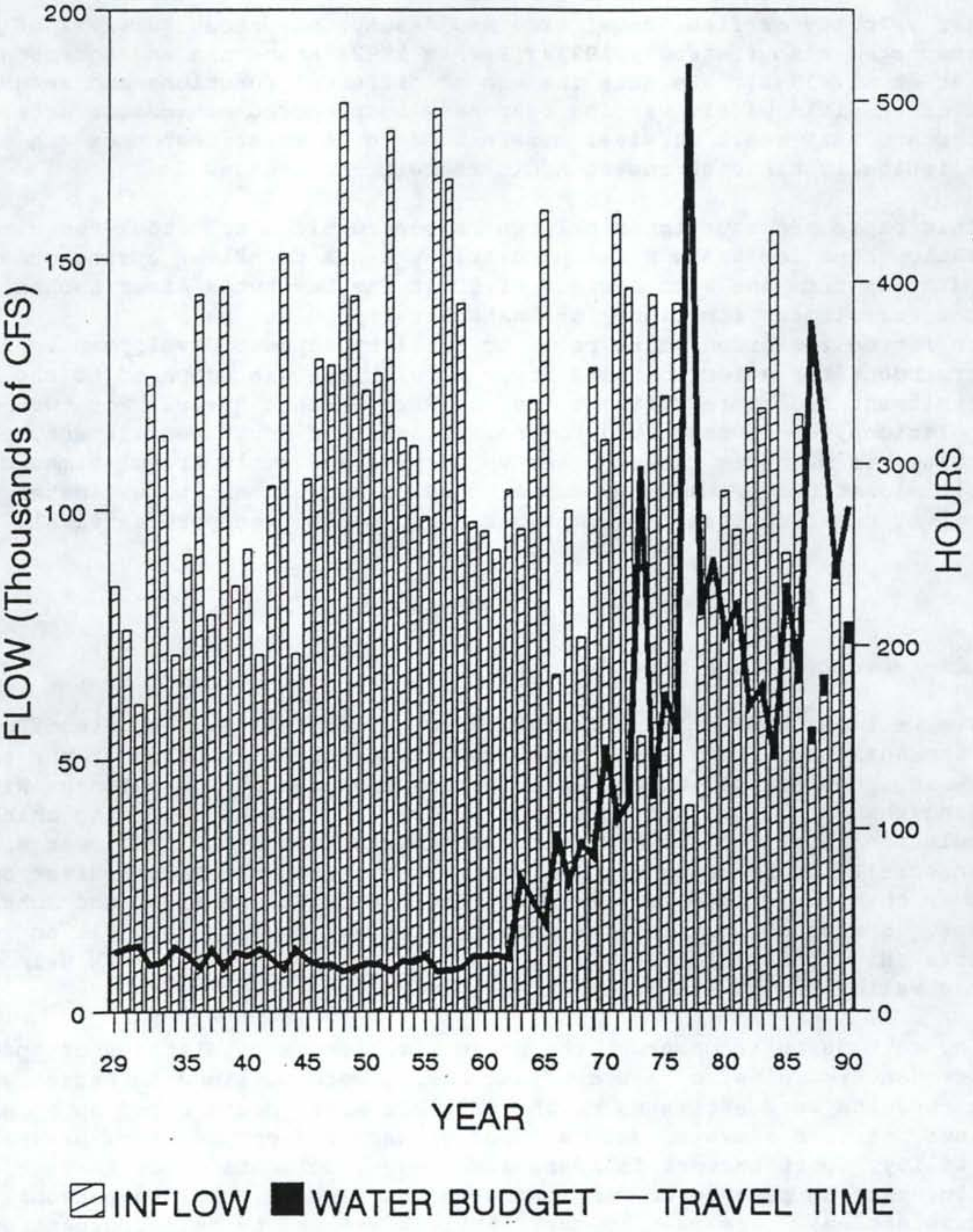


Figure 1. Mean Snake River flow (kcfs) and water particle travel time (hours) during the spring smolt migration period (April 15-June 15), 1960-90.

water velocity or fish travel time and assumptions about survival of transported fish (Petrosky 1991a; Fisher 1992; McConnaha and Anderson 1992; Weber et al. 1992). Because the use of different functions and assumptions can often yield widely varying responses to proposed management actions, it is important that smolt survival models used to forecast responses can also realistically hindcast recent adult recruitment trends.

This paper presents empirical run reconstructions and stock-recruitment relationships for Snake River populations and a downriver spring chinook population from the Warm Springs River in the Deschutes River Subbasin. Stock-recruitment functions for Snake River populations were fitted for brood years prior to final hydropower development. Recent recruitment for selected Snake River populations was compared to the baseline recruitment and to recruitment for the Warm Springs River. For two populations, we investigated the relationship of adult recruitment, at spawner density, to mainstem flows, water velocities and smolt travel times during the smolt migration. Finally we compared adult recruitment to estimates of smolt survival under different assumptions about smolt transport survival.

METHODS

Stock-Recruitment and Base Productivity

Twelve index populations of Snake River spring and summer chinook were used in the analysis. Most index areas represent only a fraction of the total spawning population. Oregon index populations included the Imnaha River spring/summer chinook and an aggregate Grande Ronde River spring chinook population (Lookingglass and Catherine creeks, and upper and lower Minam River transects). Idaho index populations were all from the Salmon River subbasin. Summer chinook indices were from the South Fork Salmon River and Johnson Creek. Spring chinook indices were from Marsh, Bear Valley, Elk and upper Big creeks (Middle Fork Salmon River), Lemhi River, upper East Fork Salmon River, upper Valley Creek, and upper Salmon River.

The methods to reconstruct the Snake River index populations of spawners and subsequent recruits, on a brood year basis, were outlined in Petrosky (1991b). The recruits were estimated to the Columbia River mouth based on annual redd counts, carcass surveys, estimates of spawner age composition, prespawning mortality, sport harvest in Idaho and Oregon, Columbia River harvest, and adult upstream passage conversion rates (Matylewich, CRITFC, personal communication). Prespawning mortality was assumed to be 10 percent as suggested by tributary weir counts of adults and redd count data in Salmon River streams (Ortmann 1968, Bjornn 1978, Kiefer and Forster 1991). In the run reconstructions, the effect of increasing the pre-spawning mortality rate would be to proportionately increase estimated recruitment to the Columbia River mouth.

Numbers of spawners and recruits were estimated for Snake River populations for brood years 1957-86. Lack of recent carcass data and/or inability to separate hatchery and natural adult returns precluded reconstruction of recent recruitment for four Salmon River populations.

Similar methods were used to reconstruct the Warm Springs River (downriver population) spring chinook spawners and recruits (Lindsay et al. 1989). The recruits were estimated based on annual redd counts, carcass surveys, estimates of spawner age and sex composition, prespawning mortality, fish taken for broodstock, and Deschutes River harvest (Lindsay et al. 1989 and Olsen 1992). Presence of a weir allowed for accurate accounting of escapements and age structure. The Lindsay et al. (1989) methods were updated to also account for Columbia River harvest and adult upstream passage conversion rates. Numbers of spawners and recruits were estimated for brood years 1975-86.

Spawner-recruit models were fitted to Snake River populations for the base period (brood years 1957-69) and to the Warm Springs population for the recent period (brood years 1975-86). Ricker (1975) models were fitted by least squares linear regression and nonlinear regression (Wilkinson 1989) techniques.

The recruit per spawner ratios (R/S) were compared relative to the percent of the maximum sustained production (MSP) spawning escapement levels achieved in each brood year. The %MSP escapement level, which accounts for density-dependent mortality, was approximated from the best fit stock-recruitment model for the Snake River base period and the Warm Springs recent period.

Post-Hydropower Recruitment

Recent recruitment of Snake River spring and summer chinook populations was assessed by comparing brood year success to recruitment predicted for the base period and for the downriver population. The ratio (observed R/S)/(predicted R/S) at spawner density (%MSP) produces a relative survival index for each brood year.

The relative survival index reflects primarily the density-independent mortality due to annual smolt survival conditions, as well as density-independent mortality in freshwater, estuary and ocean environments, and estimation errors. It excludes density-dependent mortality, for the most part, by expressing recruitment relative to seeding level (MSP).

Relationships to Smolt Migration Conditions and Modeled Smolt Survival

We investigated the influence of mainstem water velocity during smolt migration on recent recruitment for two Snake River populations, Marsh Creek spring chinook and Imnaha River spring/summer chinook (brood years 1975-86). First we regressed ln(R/S) against spawner seeding (% MSP) and mean Snake River flow during the peak chinook smolt migration period (April 15-May 5). We also estimated smolt travel times from the FLUSH Model (Weber et al. 1992) for the same period and substituted these in the multiple regressions for the flow variable.

We investigated the sensitivity of two assumptions about survival of transported fish by comparing estimated smolt survival from the FLUSH Model (Weber et al. 1992) with the relative survival index of adult recruitment for brood years 1975-86. Alternative transport survival assumptions were TBR-1 and TBR-2 used in the NPPC Phase 3 analyses (McConnaha and Anderson 1992). TBR-1 assumes that transport survival is independent of water velocities faced by smolts prior to collection and transportation. TBR-2 assumes that transport survival decreases under poor flows and velocities.

RESULTS AND DISCUSSION

Stock-Recruitment and Base Productivity

Stock-recruitment data from the Warm Springs River spring chinook population for brood years 1975-86 exhibited classic density dependence (Fig. 2). A Ricker function fit the data set exceptionally well ($r^2 = 0.96$; Table 1). This downriver index population appeared to be quite stable and moderately productive, with an intrinsic rate of increase of 2.35 (this is equivalent to an absolute rate of productivity of 10.49 progeny for each parent).

Base period (brood years 1957-69) productivity of Snake River populations, on average, closely matched the recent productivity of the Warm Springs stock (Fig. 3, Table 1). Excluding South Fork Salmon River summer chinook populations, which suffered severe habitat degradation in the 1960s (Platts and Megahan 1975), the intrinsic rates of increase ranged from 1.75 to 3.49. The average intrinsic rate of increase for these Snake River populations was 2.34 (10.38 progeny per parent), compared to 2.35 for the downriver index population.

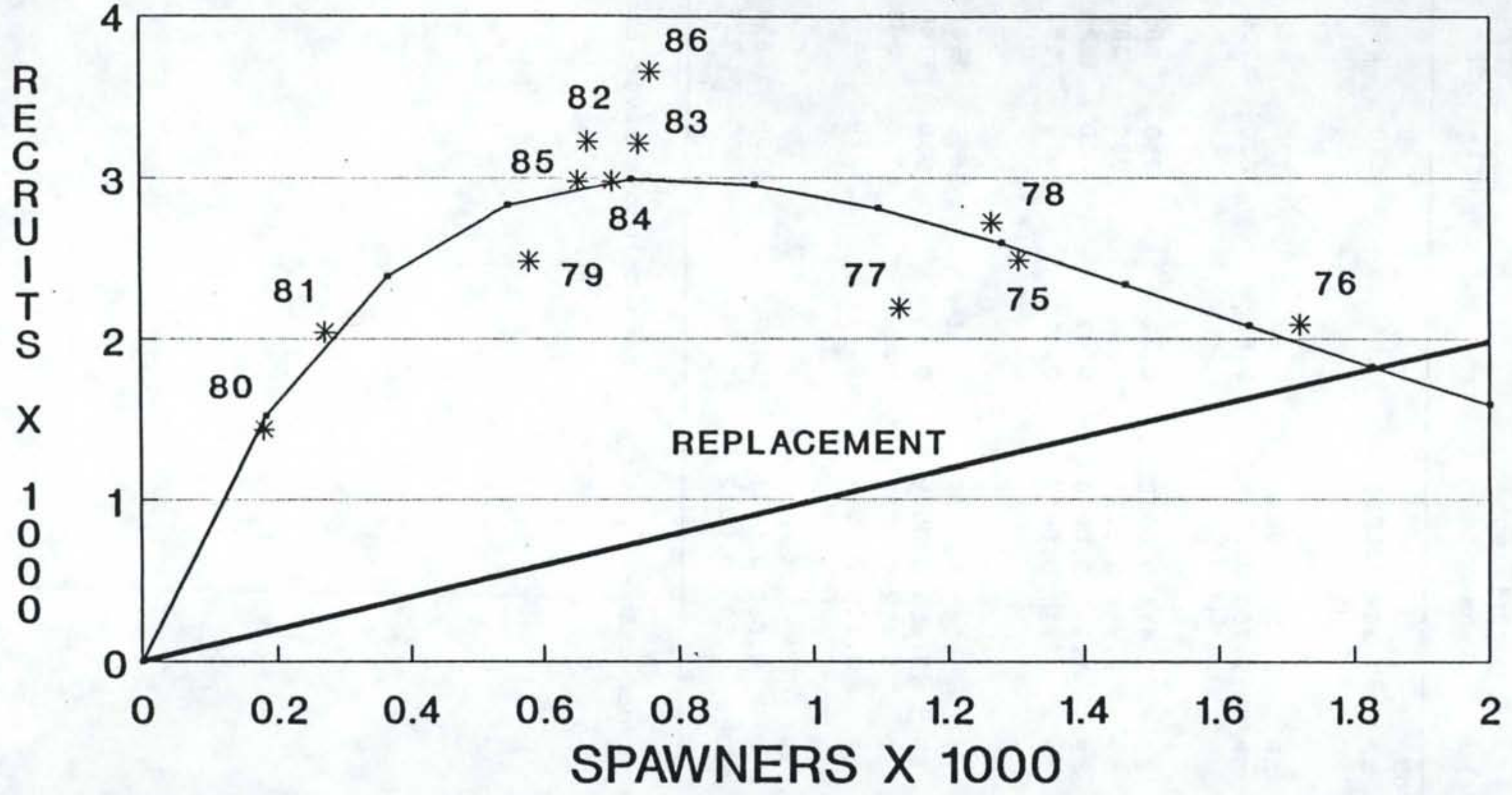
Base period productivity was considerably less in Johnson Creek (1.57) and South Fork Salmon River (0.88) index areas than in other Snake River populations. Productivity appeared to be declining for these two populations during the 1957-69 period, based on inspection of spawner and recruit plots of the individual brood years (T.E.C.H. 1992).

The fit of Ricker functions to base period stock-recruitment data was poor for some of the Snake River populations. Coefficients of determination were less than 0.25 for five index populations (Table 1), all of which had less-than-average productivity and some degree of habitat degradation during the base period.

Post-Hydropower Recruitment

Recent productivity of Snake River populations has been more variable and considerably lower than either the base period productivity or the recent productivity of the downriver index population (T.E.C.H. 1992). The recent recruit/spawner ratios (R/S) of Marsh Creek spring chinook, for example, were less than predicted for either the base period or the downriver stock in 11 of 13 years when density dependence is taken into account (Fig. 4). Similarly, recent R/S ratios for the Imnaha River spring/summer chinook were less than the base period prediction in all but one year, and less than the downriver population in all years (Fig. 5).

Density-independent survival of Snake River populations declined and became more variable following hydropower development. The relative survival index,



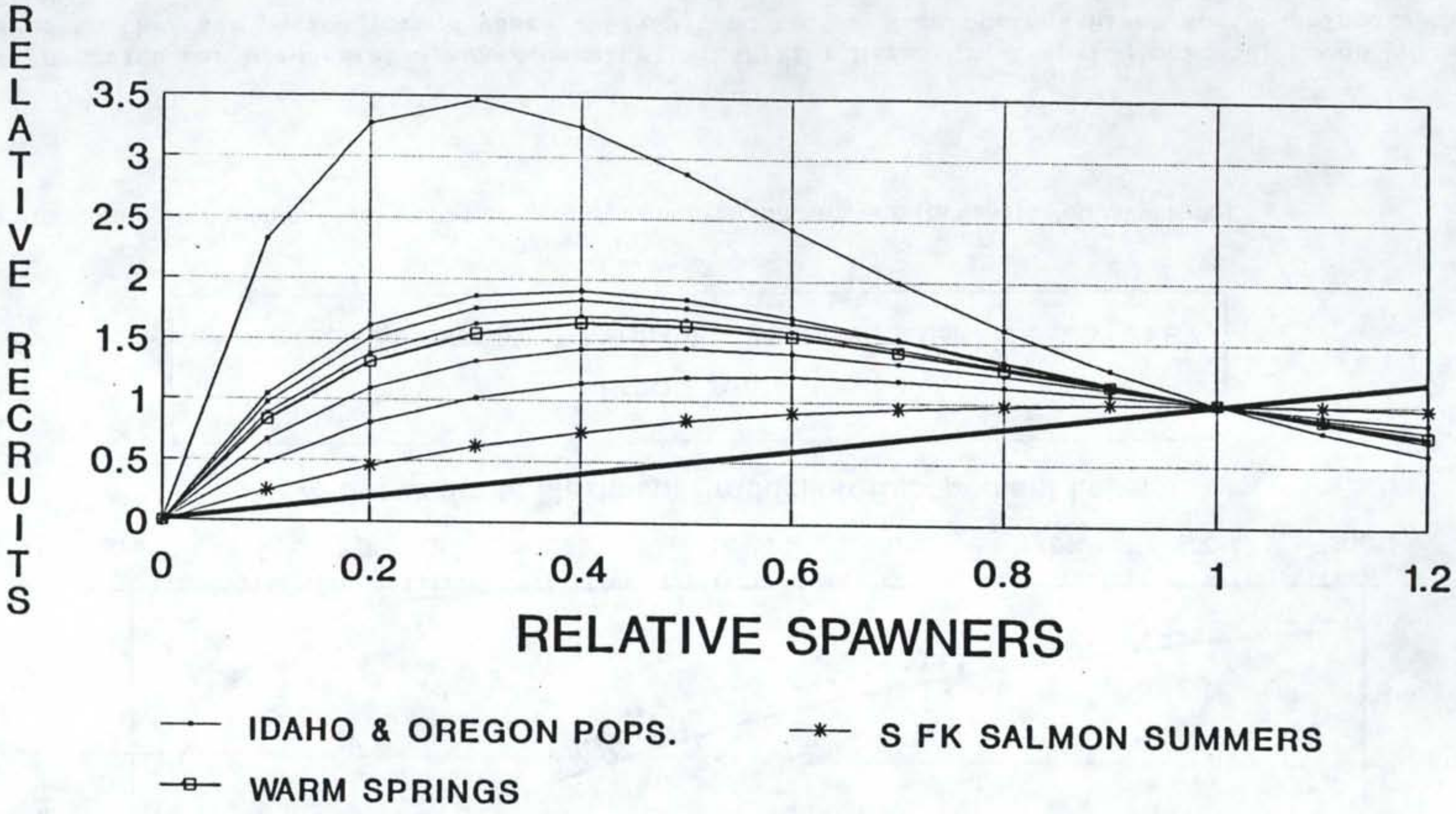
Alpha = 2.349; Beta = -0.00128
R^2 = 0.96
(Schaller, ODFW)

Figure 2. Stock recruitment of Warm Springs River, Oregon spring chinook, brood years 1975-88.

Table 1. Ricker stock-recruitment coefficients fit to Warm Springs River spring chinook population (downriver), brood years 1975-86, and to Snake River spring (SP) and summer (SU) chinook index populations, brood years 1957-69.

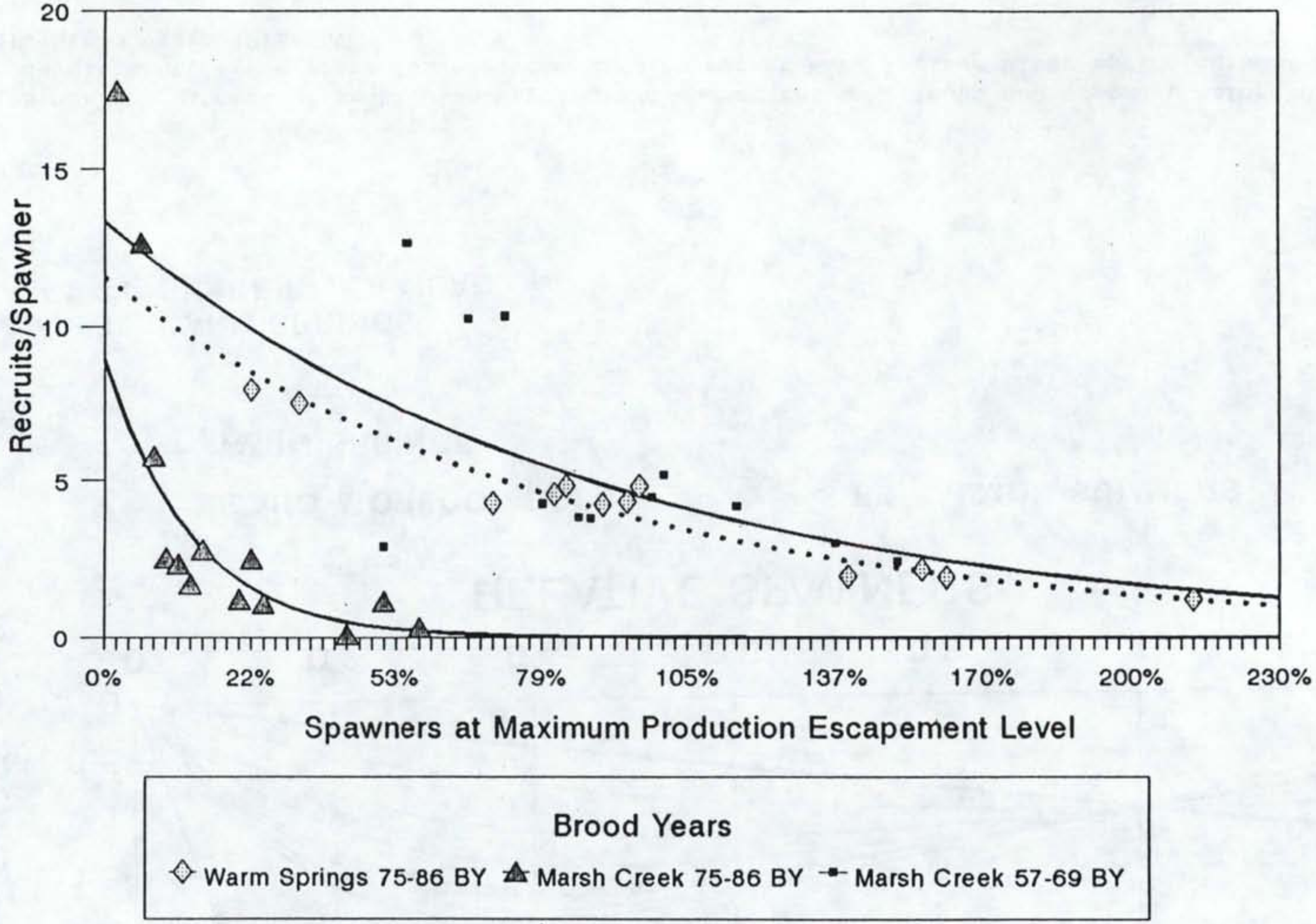
Area,		Coefficient			Spawners at		
Population	Run	alpha	beta	r^2	MSY	MSP	Replacement
Deschutes River, OR							
Warm Springs	SP	2.349	0.0013	0.96	630	805	1825
Snake River, OR							
Grande Ronde	SP	2.529	0.0012	0.80	700	848	2145
Imnaha[a]	SP/SU	1.780	0.0008	0.63	820	1219	2170
Middle Fk. Salmon, ID							
Marsh	SP	2.414	0.0012	0.42	710	861	2078
upper Big	SP	1.968	0.0026	0.34	290	385	758
Bear Valley	SP	1.947	0.0008	0.09	1000	1324	2579
Elk	SP	2.817	0.0017	0.41	520	582	1641
Upper Salmon, ID							
Lemhi	SP	1.747	0.0002	0.15	2650	3987	6963
upper E. Fk.	SP	2.603	0.0009	0.55	950	1140	2966
upper Valley	SP	3.492	0.0038	0.88	250	263	919
upper Salmon	SP	2.119	0.0004	0.22	1970	2642	5598
South Fk. Salmon, ID							
South Fk.	SU	0.880	0.0001	0.07	4150	10679	9401
Johnson	SU	1.571	0.0012	0.24	500	727	1083

[a] Model assumes summer chinook timing through Columbia River fisheries.



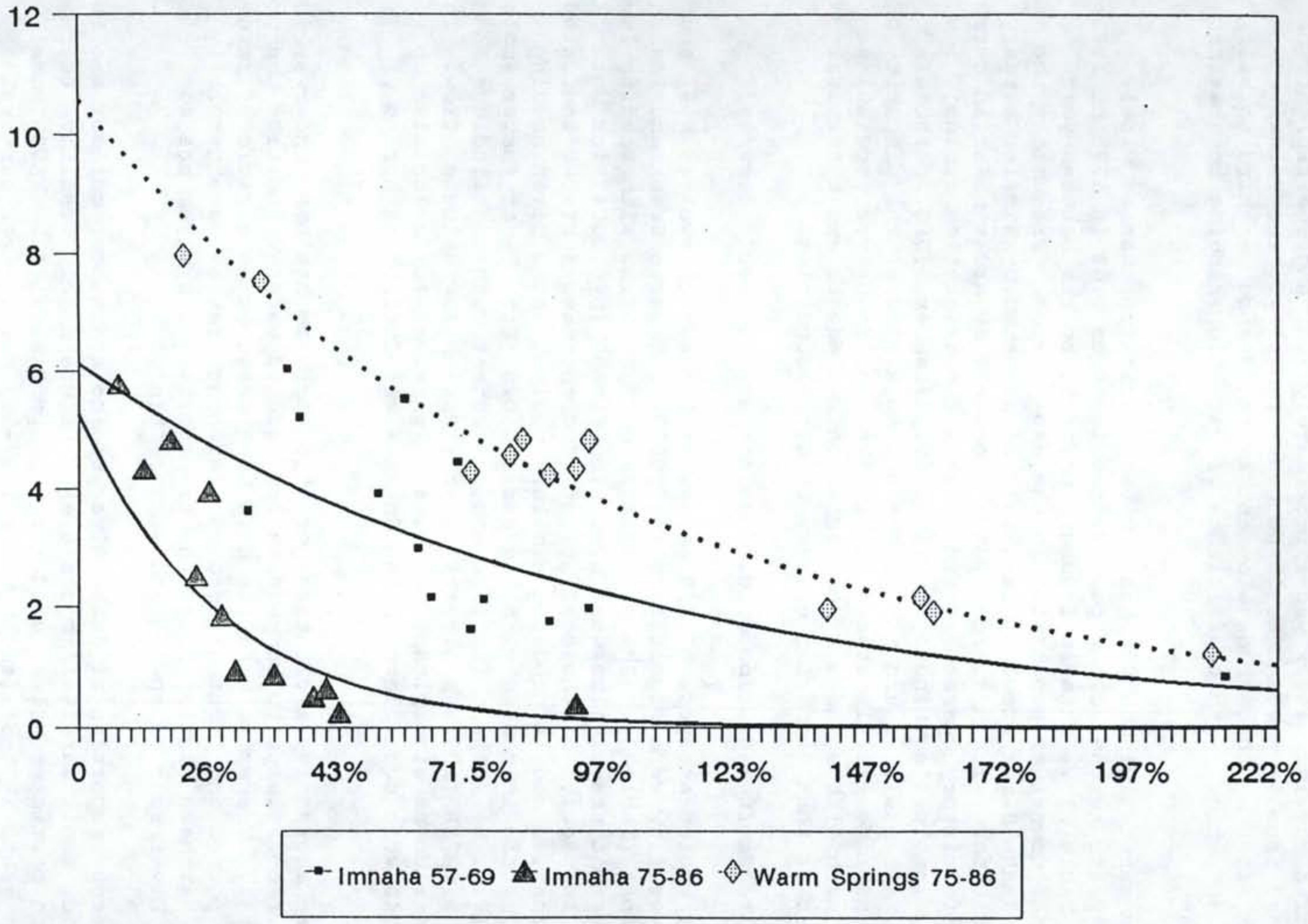
ALPHA = 2.35, WARM SPRINGS
ALPHA = 2.34, SNAKE RIVER MEAN

Figure 3. Stock recruitment of Snake River spring and summer chinook (Idaho and Oregon), before final hydropower development, brood years 1957-69, compared to recent Warm Springs River spring chinook (downriver population), brood years 1975-86.



Maximum Production Escapement Level estimated from Spawner/Recruit Analysis (Schaller&Petrosky 1992)

Figure 4. Recruits per spawner at spawner density (% MSP) for Marsh Creek spring chinook, brood years 1975-86, compared to the base period (brood years 1957-69) and to the Warm Springs River spring chinook, brood years 1975-86.



Maximum production escapement estimated from Spawner/Recruit analysis (Schaller & Petrosky 1992)

Figure 5. Recruits per spawner at spawner density (% MSP) for Imnaha River spring/summer chinook, brood years 1975-86, compared to the base period (brood years 1957-69) and to the Warm Springs River spring chinook, brood years 1975-86.

which expresses density-independent recruitment, equals 1.0 (by definition) when observed R/S for a brood year equals the predicted R/S for the base period. For the four Snake River index populations during the base period, the mean relative survival index was approximately 1.0, and coefficients of variation ranged from 33 percent to 52 percent (Table 2). The relative survival index after hydropower development averaged about a third to half that in the base period, with coefficients of variation ranging to greater than 100 percent.

The relative survival index for Snake River populations varied widely between brood years in the recent period, ranging from 0.04 in 1975 to 1.43 in 1980 (Fig. 6, Table 2) . This is in sharp contrast to the between-year stability of the downriver population in the same period. Presumably the downriver and Snake River populations should be affected by similar estuary and ocean conditions (within a brood year) because they are similar in terms of juvenile life history characteristics (age and outmigration timing), low ocean interception levels (Lindsay et al. 1989; Hassemer, IDFG, personal communication) and adult return timing the Columbia River. The disparity of density-independent survival between Snake River and downriver populations suggests that much of the Snake River variation was due to smolt survival conditions, rather than differing estuary or ocean conditions.

Relationships to Smolt Migration Conditions and Modeled Smolt Survival

Multiple regression analyses of R/S ratios with seeding level (% MSP) and mean Snake River flow were significant ($p \leq 0.0003$) for Marsh Creek and Imnaha River populations (Table 3). The regressions of R/S ratios with seeding level and estimated smolt travel time were also significant ($p \leq 0.0004$) for both populations. Inspection of the coefficients and response surfaces indicated that the alternative models produced similar results. At a given seeding level, the R/S ratio increased with increased flows (Fig. 7) or faster smolt travel times (Fig. 8). These findings are consistent with the findings outlined in the CBFWA (1991) biological justification for mainstem flow. That is, there appear to be significant benefits to survival for velocities up to at least the 140 kcfs equivalent in the Snake River during the spring migration.

Since 1977 an aggressive smolt transportation program has been underway in the Snake River to attempt to compensate for reduced water velocities and poor inriver survival. Transport benefits have been measured as a ratio of return rates of transported and "control" groups. Although ratios have generally exceeded 1:1 (Matthews et al. 1990), the transportation program has not rebuilt Snake River runs of spring and summer chinook.

Because a large proportion of Snake River spring and summer chinook smolts are transported, smolt survival models are by nature highly sensitive to the assumptions and parameter values used for transport survival. FLUSH model runs for brood years 1975-86 (smolt years 1977-88) for inriver survival (transportation turned off) yielded estimates of smolt survival that correlate significantly ($r^2=0.34$, $p<0.05$) with the relative survival indices for Snake River populations (Fig. 9). Use of both transport models TBR-1 and TBR-2 in FLUSH yielded total smolt survival estimates that deviated considerably from

Table 2. Mean relative survival index and coeffient of variation (C.V.) for Warm Springs River spring chinook population (downriver), brood years 1975-86, and four Snake River spring/summer chinook populations, brood years 1957-69 and 1975-86.

Population	Base Period (1957-69)		Recent Period (1975-86)	
	Mean	C.V.(%)	Mean	C.V.(%)
Downriver				
Warm Springs River	--	--	1.00	10.4%
Snake River				
Imnaha River	1.06	33.4%	0.48	75.3%
Marsh Creek	1.07	35.9%	0.43	117.5%
Bear Valley Creek	1.15	49.7%	0.48	113.5%
Elk Creek	1.02	52.0%	0.28	118.7%

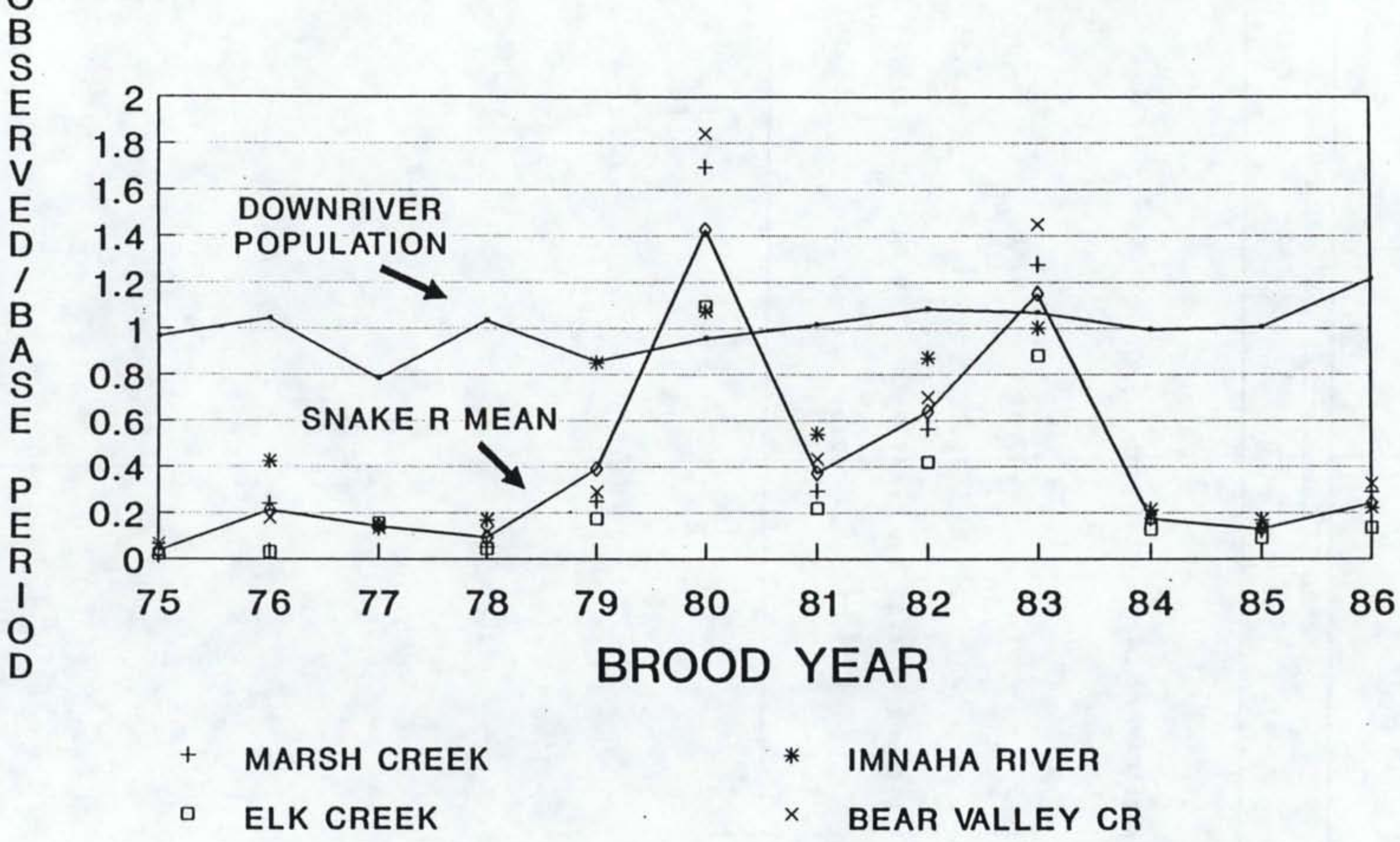


Figure 6. Relative survival index for four Snake River spring and summer chinook populations and for Warm Springs River (downriver population) spring chinook, brood years 1975-86.

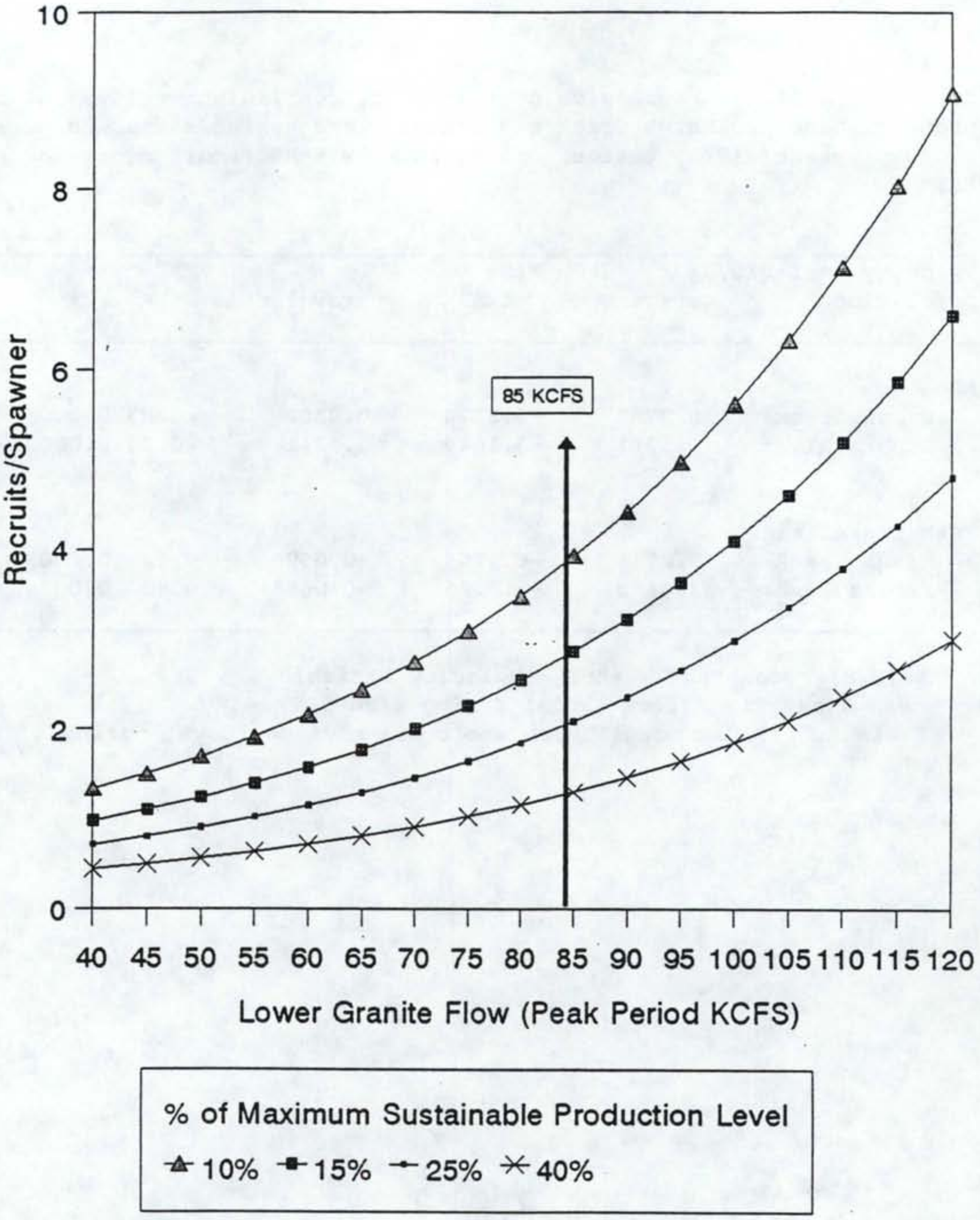
Table 3. Multiple regression coefficients, coefficients of variation and probabilities for Marsh Creek and Imnaha River spring/summer chinook recruit/spawner (R/S) ratios, brood years 1975-86 (smolt migration years 1977-88)[a].

Velocity variable,		Flow or			
population	Constant	%MSP	travel time	R^2	p =
Flow[b]					
Marsh Creek	0.2743	-6.3174	0.0202	0.83	0.0003
Imnaha River	-0.5151	-3.1849	0.0241	0.84	0.0002
Fish travel time[c]					
Marsh Creek	3.2715	-6.2705	-0.0599	0.83	0.0003
Imnaha River	3.0688	-3.3726	-0.0685	0.83	0.0004

[a] ln(R/S) = Constant + %MSP + Velocity Variable

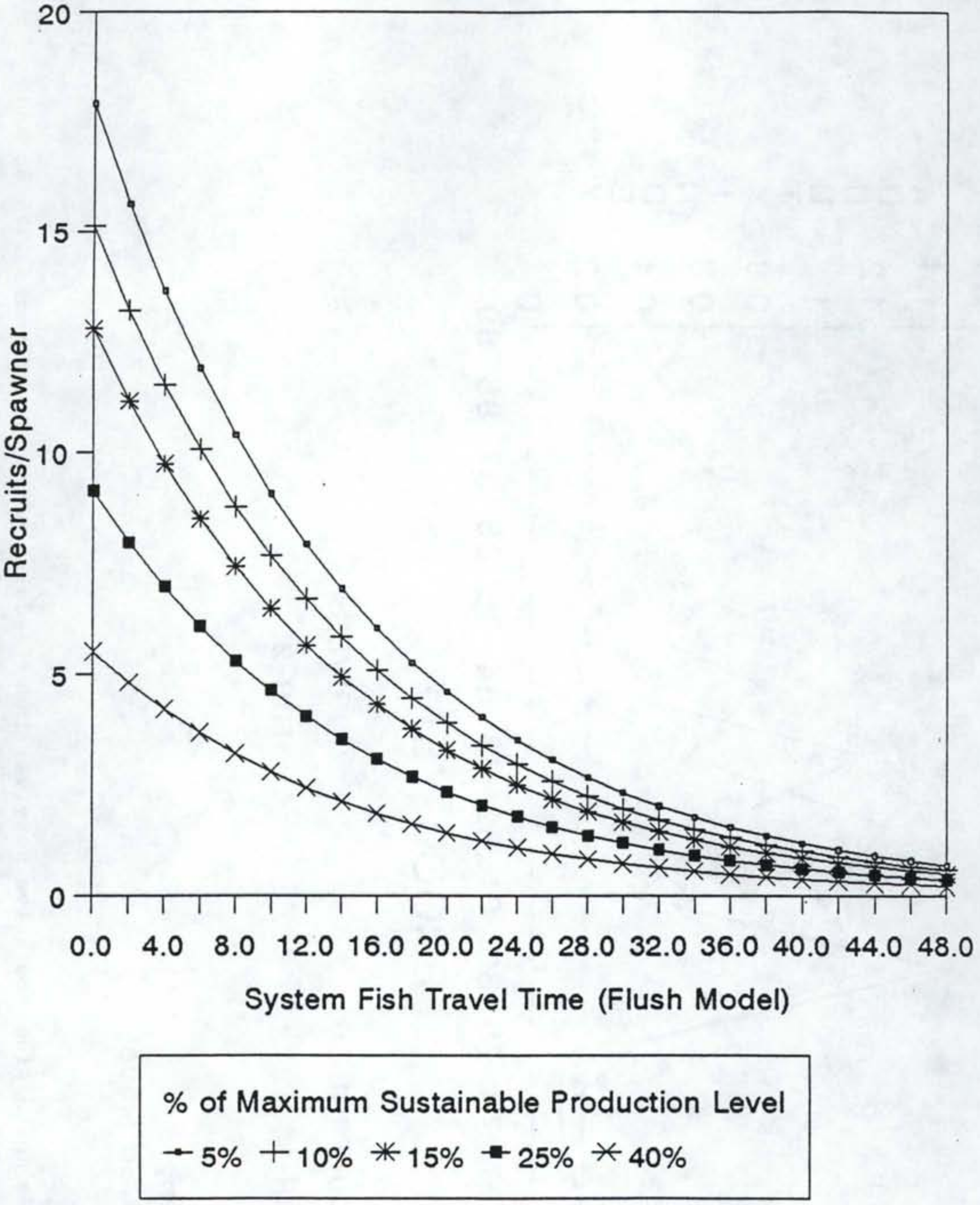
[b] Mean Snake River flow (kcfs) during 4/20-5/5.

[c] Estimated (FLUSH model) mean smolt travel time (days) during 4/20-5/5.



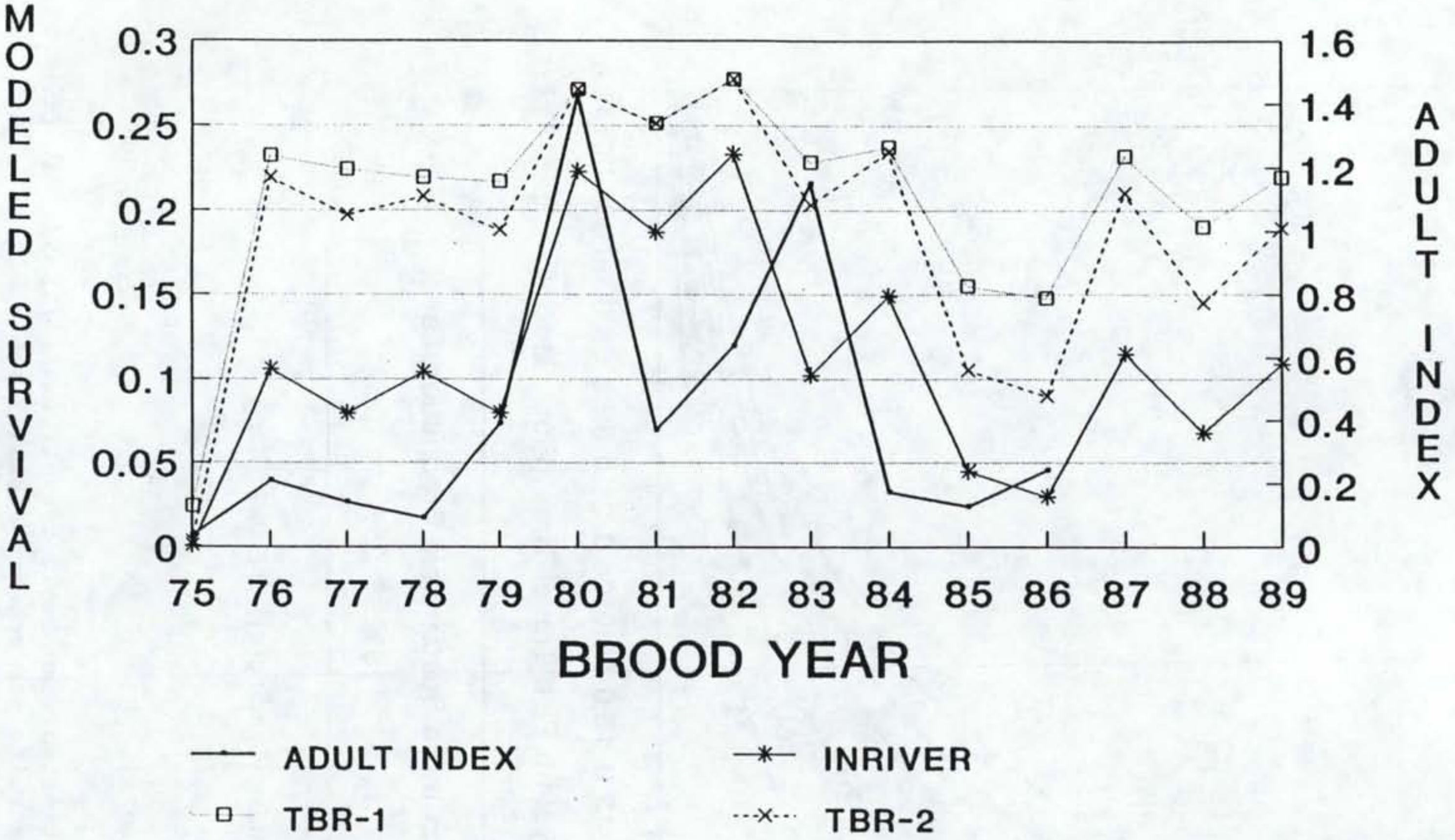
Migration Years 1977 - 1988
LN(Recruit/Spawner) = Constant + %Seeding(X1) +PFlow(X2)
R^2 = 0.84

Figure 7. Influence of mean Snake River flow (4/15-5/5) at four seeding levels (%MSP) on recruit per spawner ratios, Imnaha River spring/summer chinook, brood years 1975-86.



Migration Years 1977 - 1988
LN(Recruit/Spawner) = Constant + %Seeding(X1) + Fish TT(X2)
R^2 = 0.83

Figure 8. Influence of estimated mean smolt travel time (4/15-5/5) at five seeding levels (%MSP) on recruits per spawner ratios, Imnaha River spring/summer chinook, brood years 1975-86.



NPPC TBR Model 1 and 2
T.E.C.H. Workgroup FLUSH Model

Figure 9. Comparison of the Snake River relative survival index (adult index), brood years 1975-86 and modeled smolt survival using FLUSH, brood years 1975-89. The FLUSH model was run with transportation ignored (inriver survival), and with two alternative assumptions about survival of transported smolts (TBR-1 and TBR-2).

adult indices; neither correlated significantly ($p>0.10$). The models imply that the most optimistic assumptions used for transport survival will deviate farthest from empirically based adult recruitment data.

Smolt survival and life-cycle models have been used increasingly in Columbia River salmon planning and management. To the extent that smolt survival models are used as decision tools and in forecasting future trends, it is important that they can also realistically hindcast recent adult trends. Empirical recruitment data should be an essential element in validating the parameters, assumptions and performance of hydrosystem smolt survival models.

SUMMARY

1. Productivity of Snake River spring and summer chinook populations declined and became more variable since hydropower development.
2. Pre-development productivity of most Snake River populations closely matched recent productivity of a downriver population that has not been exposed to the same high levels of density-independent mortality.
3. Productivity of South Fork Salmon River summer chinook, which suffered severe habitat degradation in the 1960s, was considerably lower than in other Snake River populations.
4. Recent recruitment (brood years 1975-88) of Snake River populations has been highly variable.
5. Recent recruitment of Snake River populations is related significantly to the mainstem velocities experienced during the smolt migration when density dependence is taken into account. There appear to be significant benefits to survival for velocities up to at least the 140 kcfs equivalent.
6. The FLUSH model produced smolt survival estimates that correlated significantly with adult recruitment indices when transport was ignored; the more optimistic transport survival assumptions caused the modeled survival to deviate considerably from the empirically based adult data.
7. Empirical recruitment data should be an essential element in validating the parameters, assumptions and performance of hydrosystem smolt survival models.

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OREGON COASTAL NATURAL COHO SALMON ABUNDANCE MONITORING: AN EVALUATION OF PAST METHODOLOGIES AND IMPLICATIONS TO OCEAN FISHERY MANAGEMENT

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ABSTRACT

Naturally produced coho salmon (*Oncorhynchus kisutch*) in Oregon coastal watersheds are the cornerstone of regional ocean salmon fisheries management. These stocks, referred to as oregon coastal natural (OCN) coho, originate from coastal basins south of the Columbia River to the Oregon-California border. Ocean salmon fisheries off Oregon and California are regulated to achieve a specific escapement goal for OCN coho through the structuring of harvest quotas. Integral to this management program is the need for an accurate means of predicting OCN recruitment and estimating OCN spawning escapement. The Oregon Department of Fish and Wildlife (ODFW) has indexed the status of OCN stocks since 1950 by conducting spawning surveys. These surveys have provided the only direct measure of OCN abundance and therefore have been the basis of absolute recruitment and escapement estimates.

In 1990, ODFW began a study aimed at improving OCN abundance estimates. The goal of this study is to conduct a statistically based survey program alongside the traditional survey program, to assess estimation precision and bias associated with estimates obtained from traditional survey counts. The study incorporates a stratified random sampling design. Preliminary results indicate that OCN abundance estimates derived from traditional spawning surveys are substantially inflated. Implications of these findings to current quota management programs for ocean salmon fisheries are discussed.

REBUILDING AND RESTORING STOCKS

SESSION LEADER: ROY HEBERGER,
U.S. FISH AND WILDLIFE SERVICE

No abstracts or manuscripts submitted

POSTER SESSION

SESSION LEADER: ROB DILLINGER,
IDAHO DEPARTMENT OF FISH AND GAME

A LANDSCAPE ECOLOGY APPROACH TO FISHERIES HABITAT EVALUATION

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ABSTRACT

Management of anadromous fish habitat on Forest Service lands in the northern Rocky Mountains is directed to restore wild and naturally reproducing fish populations. Important first tasks in this restoration are to establish objectives for fish production capability for different streams and to describe desired future condition of riparian and aquatic habitats to meet those objectives.

Stream channels and associated riparian zones are highly dynamic in response to upstream processes, and disturbances on surrounding uplands. Their development is also constrained by local landforms. Fisheries habitat is affected by these physical processes that create valley floor landforms and their characteristic patterns of terrestrial vegetation.

The Clearwater District of the Nez Perce National Forest has initiated a landscape ecology approach to integrated resource assessment on a multiple watershed scale. Basinwide stream surveys are integrated using reach, landtype association, and watershed level attributes, both upland and riparian, as frames of reference.

PRECOCIOUS MATURATION IN COHO SALMON: GENETIC FACTORS

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ABSTRACT

Two generations of coho salmon were raised to examine the additive genetic and non-additive genetic factors influencing precocious maturation. In the first generation, produced by crossing parental fish in a paternal half-sib nested design, the heritability (the ratio of additive genetic variance to total phenotypic variance) of precocious maturation was estimated at between 0.05 and 0.13. Egg size of the female parent was identified as a significant non-additive genetic effect.

In the second generation, full-sib families were established by factorial crossing of F_1 fish from families with a <u>high incidence</u> of precocious maturation (range from 11% to 42%, H males and females) and families with a <u>low incidence</u> of precocious maturation (range from 0% to 1%, L males and females). Fish from F_2 families were also examined for incidence of precocious maturation. Precocious individuals were present in most F_2 families. The high incidence (25%) of precocious maturation in the HH families (H male crossed with H female) compared with the intermediate frequency in HL and LH families (16%) and low frequency (6%) in LL families supports the existence of an additive genetic component. The congruence in incidence of precocious individuals between HL and LH families indicates that both male and female parents contribute approximately equivalently to precocious maturation.

HISTORICAL ANALYSIS OF THE EFFECTS OF CHANGING LAND AND WATER MANAGEMENT PRACTICES ON FISH HABITATS IN THE SKAGIT RIVER, WASHINGTON

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ABSTRACT

Currently, there is a lack of detailed and site specific information on how riparian and fishery resources respond to cumulative land and water management practices on both public and private lands, especially from a watershed perspective. In particular, there is a lack of information on *when* and *where* historical management practices may have significantly impacted existing resources. Natural resource management regimes are currently limited because they do not integrate the historical effects of both biological and sociological factors which may have affected the resource.

Based on a synthesis of habitat requirements and distribution of the five major species of salmonids that utilize the Skagit River basin (Washington) and a synthesis of the historical land and water use patterns in the basin, a historical analysis is provided on how land and water management practices impact salmonid habitats. The historical analysis is taken both from the perspective of physical changes in fish habitat and the effects of management practices on fish habitat. Regulation and policy changes are also examined for their role in maintaining fish habitat. This analysis is conducted from a watershed perspective and will be used to determine "desired future condition" of fish habitat and to make recommendations, particularly with regards to habitat restoration.

STRAYING OF HATCHERY SPRING CHINOOK SALMON AND HATCHERY:WILD RATIOS OF NATURALLY-SPAWNING ADULTS IN THE GRANDE RONDE RIVER BASIN

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ABSTRACT

To estimate the hatchery and natural composition of populations and identify hatchery strays fishery biologists have recognized the importance of distinguishing hatchery from naturally-produced spring chinook salmon (*Oncorhynchus tshawytscha*). We used scale pattern analysis to identify the origin of unmarked adults that spawned naturally in the Grande Ronde River basin from 1986-1991. Discriminant functions were developed using circuli distances, bandwidths, and bandwidth ratios. A jackknife approach was used to determine the percent correct classification of scales from the hatchery and naturally-produced groups of known-origin fish (training populations). A different model was developed for each year to identify naturally-spawning fish of unknown origin. The first variable selected by each model was distance to the freshwater annulus which, used by itself, correctly identified greater than 87% of the scales in the training populations. We attained better than 95% correct classification of the training populations overall with additional variables. Results of the discriminant analysis of unmarked adults suggested that the hatchery component of the naturally-spawning populations of spring chinook salmon was much larger than previously assumed. Minimum yearly stray rates ranged from about 4 to 28%. Stray rates were higher than we expected and are reflected in the high percentages of hatchery fish in the natural spawning populations.

BACKGROUND

The hatchery program for spring chinook salmon (*Oncorhynchus tshawytscha*) in the Grande Ronde River basin began in the early 1980's under the Lower Snake River Compensation Plan. The three primary objectives that guided program development for the Lower Snake River Compensation Plan in Oregon were to: 1) enhance natural production; 2) restore sport and tribal fisheries; and 3) maintain wild endemic populations in the Minam and Wenaha rivers.

Throughout the 1980's smolts produced for the Grande Ronde River basin were released from Lookingglass Hatchery and, starting in 1984, from Big Canyon Facility. Smolts and/or adults were also outplanted into Catherine Creek, the upper Grande Ronde River, and the Wallowa River and its tributary, Hurricane Creek (Figure 1). Substantial numbers of hatchery fish began to return to the basin in 1985. Peak returns to Lookingglass Hatchery of over 2,000 fish per year occurred in 1987 and 1988 (Carmichael et al. 1988; Messmer et al. 1989). Adult returns in excess of hatchery brood stock needs in 1987 and 1988 (1,588 and 1,688) were outplanted in Catherine Creek and the upper Grande Ronde and Wallowa rivers. In 1989 about 100 adults were outplanted into the Wallowa River. The numbers of adults recovered on spawning surveys from 1986 to 1991 that were marked hatchery fish were much higher than expected in the Minam and Wenaha rivers where no hatchery fish have been released. This was of particular concern because these areas are managed for wild fish production.

Since the program was initiated there has been a major shift in the priority of management objectives. A greater emphasis on natural fish production and a heightened awareness regarding the potential genetic impact of non-local hatchery fish on indigenous populations has prompted requests for more accurate estimates of the percentage of hatchery fish in natural spawning areas of the Grande Ronde River basin. Concern was elevated with a "threatened" status being designated for Snake River spring/summer chinook salmon under the Endangered Species Act (1973) and the adoption of a revised Wild Fish Management Policy by Oregon Department of Fish and Wildlife (ODFW). Thus, quantitative estimates of the percentages of hatchery fish in the natural spawning populations were necessary to better assess stray rates, potential for genetic introgression, and compliance with the ODFW Wild Fish Policy criteria for the percent of hatchery adults in natural spawning populations.

Use of discriminant analysis is well-documented for distinguishing between salmon stocks in the ocean for some species, particularly sockeye salmon (Koo 1955; Clutter and Whitesel 1956; Henry 1961; Mosher 1963; Anas and Murai 1969; Bethe and Kranowski 1977; Cook and Lord 1978; Bethe et al. 1980). However, demonstration of the use of scale pattern analysis for chinook salmon is rather limited for ocean (Major et al. 1978) and in-river stock separation (Schwartzberg and Fryer 1989).

The specific objectives of this study were to: 1) develop a method to identify hatchery/wild origin of spring chinook salmon in the Grande Ronde basin using scale pattern analysis; 2) determine the percentage of naturally-spawning spring chinook salmon in six major tributaries of the

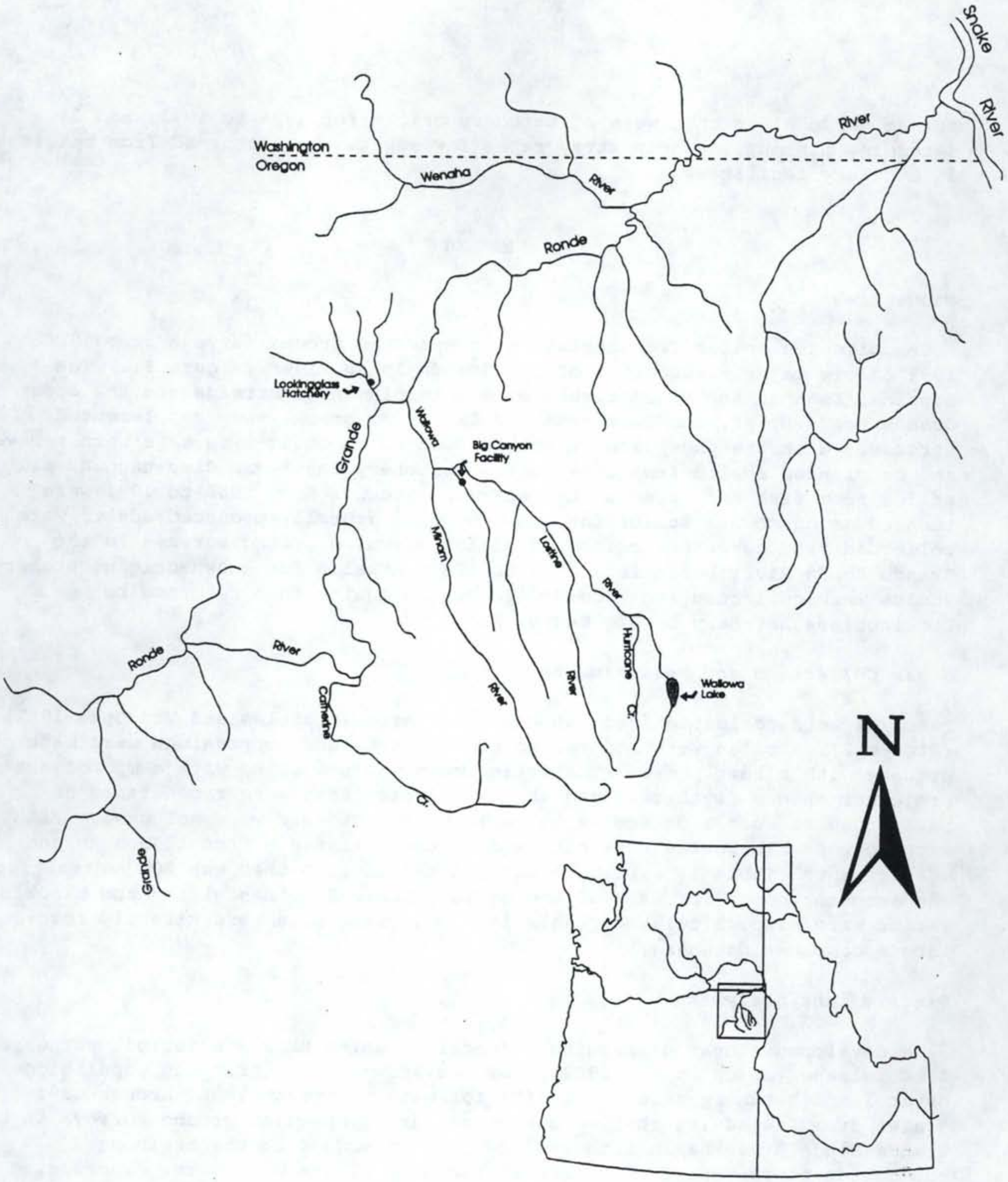

Figure 1. The Grande Ronde River basin, showing supplemented streams (Catherine and Hurricane creeks and the upper Grande Ronde River), unsupplemented streams (Wenaha, Minam and Lostine rivers), and hatchery facilities (Lookingglass Hatchery and Big Canyon Facility).

Grande Ronde River that were of hatchery origin for 1986 to 1991; and 3) determine minimum in-basin stray rates for adults that returned from releases at hatchery facilities.

METHODS

Study Area

We collected scales from carcasses on spawning ground surveys from 1986 to 1991 on six major tributaries of the Grande Ronde River (Figure 1). The Lostine, Wenaha, and Minam rivers were unsupplemented streams and the upper Grande Ronde River, and Hurricane and Catherine creeks were supplemented streams. With the exception of adults from which coded-wire-tags were removed and outplanted adults from Lookingglass Hatchery that were disc-tagged, all scales from fish recovered on the spawning grounds from 1986 to 1991 were treated as unknown. Scales for known-origin naturally-produced adults were collected from carcasses recovered during spawning ground surveys in the Grande Ronde River basin from 1976 to 1984. Scales for known-origin hatchery adults were collected from coded-wire-tagged adults that returned to Lookingglass Hatchery or Big Canyon Facility.

Scale Collection and Measurements

Scales were collected from the key scale area (Nicholas and Van Dyke 1982) (Figure 2). Scales were mounted on gummed cards and impressions were made on acetate with a heat press. The scale image was projected with a microfiche projector onto a digitizing pad at 88x. Scales that were regenerated or reabsorbed to such a degree as to make them unreadable were not used. Fish with no readable scales were not used in the analysis. Circuli counts and measurements (Table 1) were made along a radial line that was 20° ventral from the anterior-posterior axis of the scale (Figure 3). Bandwidths and bandwidth ratios were also calculated (Table 1). Digitized data were directly recorded into a computer database.

Discriminant Analysis

We developed linear discriminant functions using BMDP statistical software 1988 release (Dixon et al. 1988) from two known-origin training populations of natural and hatchery-produced adults for each year from 1986 through 1991. Scales from 144 adults that were sampled during spawning ground surveys in the Grande Ronde River basin from 1976 to 1984 were used as the training population to represent natural fish for each of the six years. The spring chinook salmon hatchery program under the Lower Snake River Compensation Plan was initiated in 1982, therefore very few hatchery-reared adults returned to the Grande Ronde River basin prior to 1985. Small numbers of juveniles released before the Lookingglass Hatchery program was established survived poorly and contributed little to adult returns in the basin.

Scales from 101 to 222 coded-wire-tagged adults that returned to Lookingglass Hatchery and Big Canyon Facility were used to represent adults of

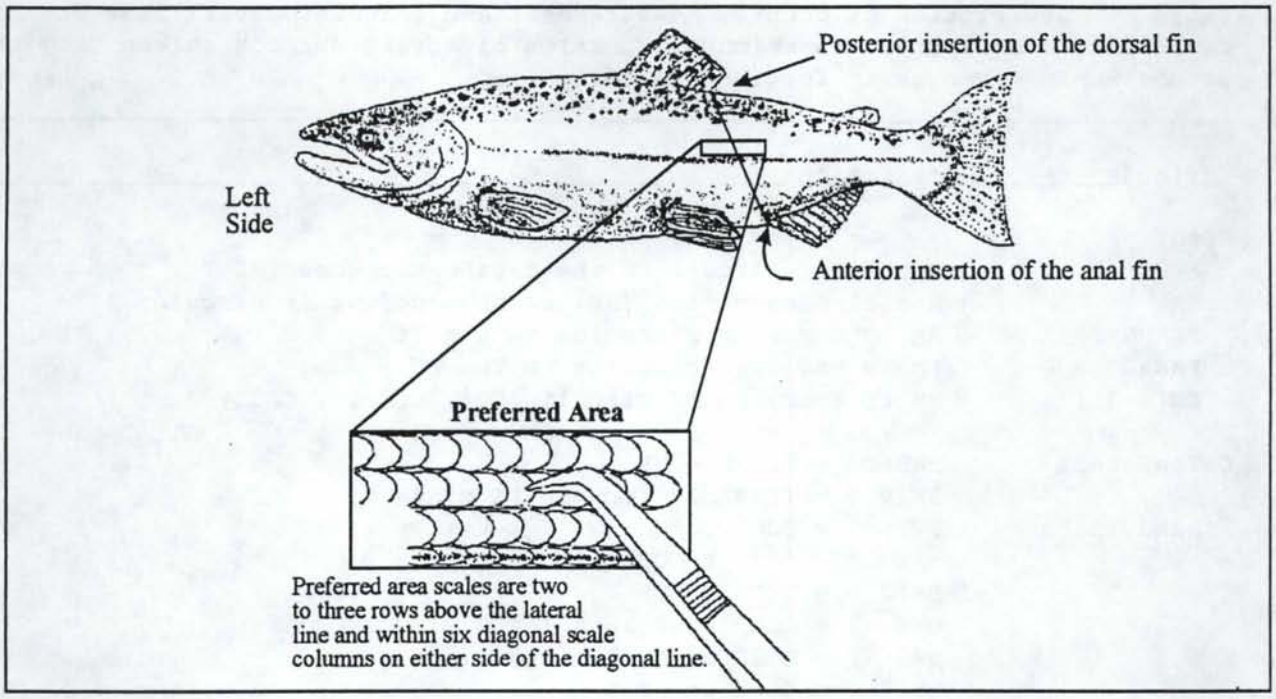

Figure 2. Preferred area of scale collection (Schwartzberg 1987).

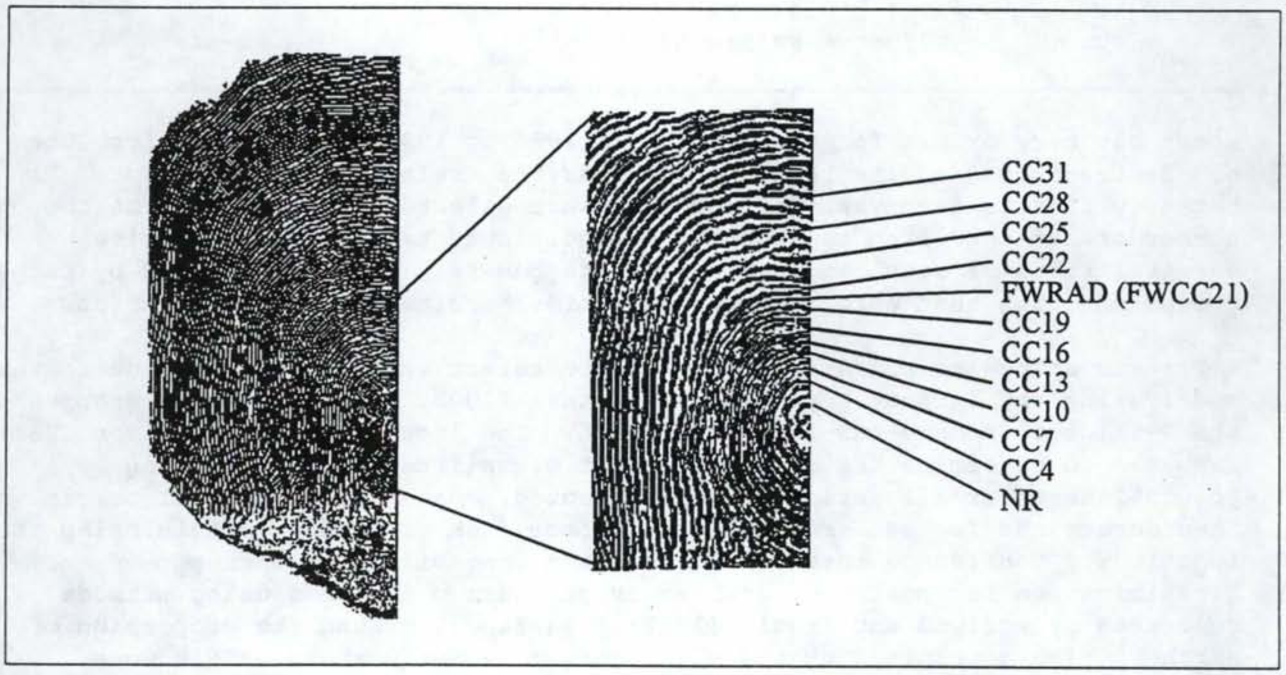

Figure 3. Example of scale measurements used for discriminant analysis.

Table 1. Description of counts, measurements and calculated variables used in developing discriminant functions for scales of spring chinook salmon from the Grande Ronde River basin for 1986 to 1991.

Variable	Description
Read:	
FWCC	Number of circuli in the freshwater zone (FZ)
NR	Radial measurement (RM) of the nucleus (= circulus 1)
FWANN	RM to the winter annulus in the FZ
FWRAD	RM to the last circulus in the FZ
CC(4-31)	RM to every third circuli (CC4, CC7 ... CC31)
Calculated:	FWRAD2 = FWRAD - NR
	FWAVSP = (FWRAD2)/(#CC in FZ minus 1)
Bandwidths	BW1 = CC4 to NR
	BW2 = CC7 to CC4
	BW12 = CC7 to NR
	BW6 = CC19 to CC16
	BW7 = CC22 to CC19
	BW67 = CC22 to CC16
	BW56 = CC19 to CC13
Ratios	R1D6 = BW1/BW6
	R1D7 = BW1/BW7
	R2D6 = BW2/BW6
	R2D7 = BW2/BW7
	R12D67 = BW12/BW67

known hatchery origin for each year from 1986 to 1991. The scales from the coded-wire-tagged adults that were used as the training population for hatchery fish in each year differed and were selected in proportion to the number of untagged fish represented by individual tagcodes and relative survival rates of each tagged group. Untagged fish were represented by tagged groups of fish that were released at a similar size, time, and location.

Forward step-wise regression was used to select variables to include in the model using the largest F-value greater than 4.000. Variables were dropped if the F-value to remove was less than 3.996. The jackknife method (Efron 1982) was used to determine the percent correct classification for training populations after all variables were selected. Hatchery or natural origin was then determined for each readable scale from fish of unknown origin using the functions. Confidence intervals around the probability of correct classification for scales of unknown origin were determined using methods described by Worlund and Fredin (1962). Variances around the proportion of hatchery fish determined by the model for scales of unknown origin were calculated using data for each tributary. Variances and 95% confidence intervals were applied to overall percentages of hatchery fish that were a summation of known hatchery fish (tagged fish) and those fish designated as hatchery by the models.

Minimum Stray Rates and Overall Percent Hatchery Fish

Estimates of escapement and the proportions that were of hatchery origin were used to estimate minimum stray rates defined as the percentage of hatchery fish that returned to locations in the Grande Ronde River basin other than the hatchery facility of release. Escapement was estimated based on an expansion factor of 2.4 fish per redd (Carmichael et al. 1986) and total redds observed during extensive spawning ground surveys. The number of strays returning to unsupplemented areas was estimated for each tributary by multiplying the proportion of hatchery fish recovered in that tributary by the estimated escapement to that tributary. The proportion of hatchery fish was calculated by adding the number of recovered hatchery fish that were identified by scale pattern analysis to the number of fish recovered that were of known hatchery origin, then dividing by the total number of recovered fish for each tributary. Because strays into supplemented areas could not be distinguished from outplanted fish, estimated stray rates do not include straying to supplemented areas and are assumed to be minimum estimates. Minimum escapement of hatchery fish destined for facilities was calculated as actual returns to Lookingglass Hatchery and Big Canyon Facility plus strays to unsupplemented areas. Minimum stray rates were determined by dividing the number of strays to unsupplemented areas by estimates of minimum escapement of hatchery fish destined for hatchery facilities in the basin.

RESULTS

Most adult spring chinook salmon designated as hatchery-origin were identified by scale pattern analysis rather than by the presence of a tag because only a small proportion of the fish were tagged at release. In all years the distance to the freshwater annulus (Figure 3) was the first variable selected for the model (Table 2). Distance to the freshwater annulus by itself was able to correctly classify a minimum of 87% of the scales in the training populations. Using all selected variables the mean percent correct classification of scales from the training populations was greater than 95% overall for every model year (Table 3).

The percentage of scales that was hatchery origin was often much larger than indicated solely by recovery of marked fish on the spawning grounds and was highly variable, ranging from 0 to 100%. Variability was evident between years and between tributaries (Figures 4, 5). Less than 26% of the fish recovered in 1986 were of hatchery origin in all tributaries except the Minam River (Figures 4, 5). The Minam River was the only tributary where coded-wire-tagged adults were recovered in 1986. Even with the relatively high percentage of hatchery fish in the Minam River in 1986, the overall percentage of hatchery fish in both supplemented and unsupplemented areas in 1986 was lower than in any other year (Figure 6). In 1986 and 1991 there was little difference between supplemented and unsupplemented areas in the estimated percentage of hatchery fish. Overall percent hatchery fish for

Table 2. Variables used in the discriminant function from 1986 to 1991 and the order in which they contributed to the model.

		Contribution order by model year					
Symbol	Variables	86	87	88	89	90	91
FWANN	Distance to freshwater (FW) annulus	1	1	1	1	1	1
NR	Nucleus radius (NR)	-	-	-	-	-	2
CC19	Distance to circulus (CC) 19	-	5	2	-	-	-
R12D67	Ratio of CC7-Nucl to CC22-CC16	-	2	6	2	-	4
BW67	Bandwidth between circuli 22 and 16	2	-	-	5	2	-
BW56	Bandwidth between circuli 19 and 13	-	3	5	-	-	-
CC13	Distance to circulus 13	3	-	-	3	4	6
BW12	Bandwidth between circuli 7 and 4	4	-	-	4	3	3
FWRAD2	Distance to last FW circulus w/o NR	-	4	4	-	-	-
FWAVSP	Average FW circuli spacing	-	-	-	-	5	-
CC25	Distance to circulus 25	-	6	3	-	-	5
FWCC	Freshwater circuli count	5	-	-	-	6	-
FWRAD	Distance to last FW circulus w/ NR	6	-	-	-	-	-
CC28	Distance to circulus 28	-	-	-	6	-	-

Table 3. Sample sizes and the percent correct classification of the models.

	Sample size		Percent correct classification		
Year	Hatchery	Natural	Hatchery	Natural	Overall
1986	101	144	99.0	97.2	98.1
1987	190	144	96.3	93.8	95.1
1988	222	144	95.5	95.1	95.3
1989	145	144	96.6	97.2	96.9
1990	147	144	97.3	95.1	96.2
1991	186	144	96.8	95.8	96.3

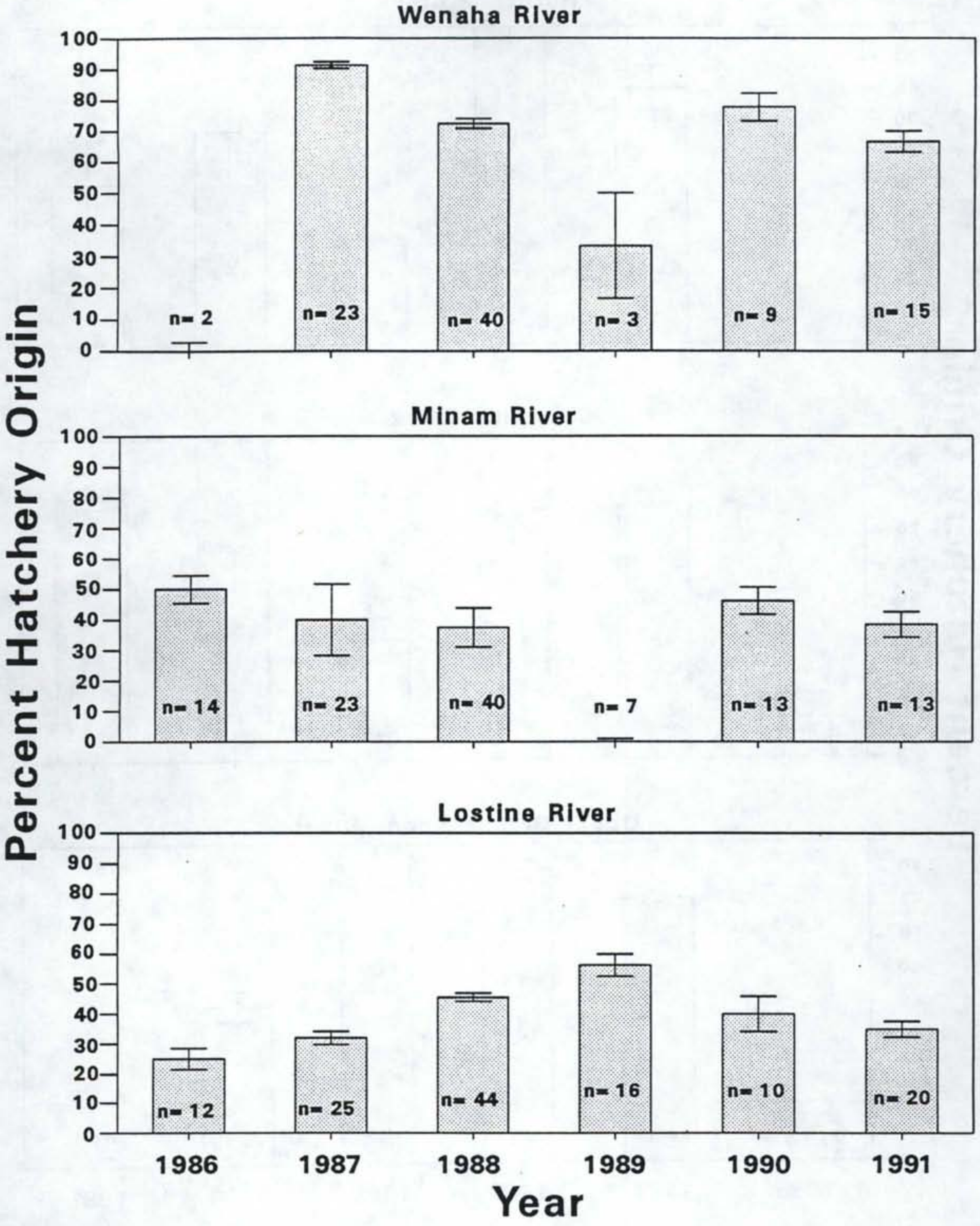

Figure 4. Percentage (and 95% C.I.) of carcasses recovered on unsupplemented tributaries of the Grande Ronde River from 1986 to 1991 that were hatchery origin. Sample size for each year is shown near the bottom of the bar.

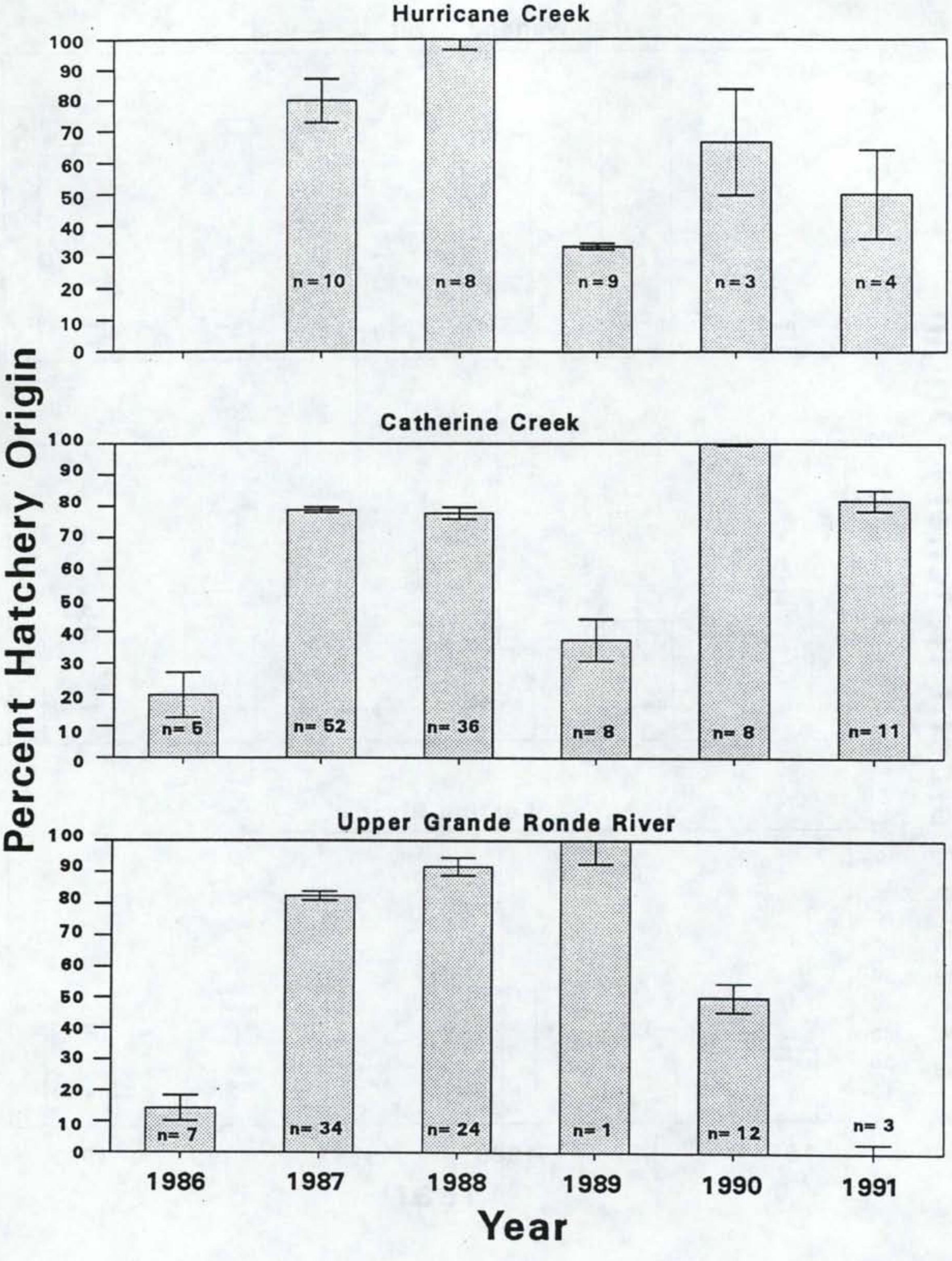

Figure 5. Percentage (and 95% C.I.) of carcasses recovered on supplemented tributaries of the Grande Ronde River from 1986 to 1991 that were hatchery origin. Sample size for each year is shown near the bottom of the bar.

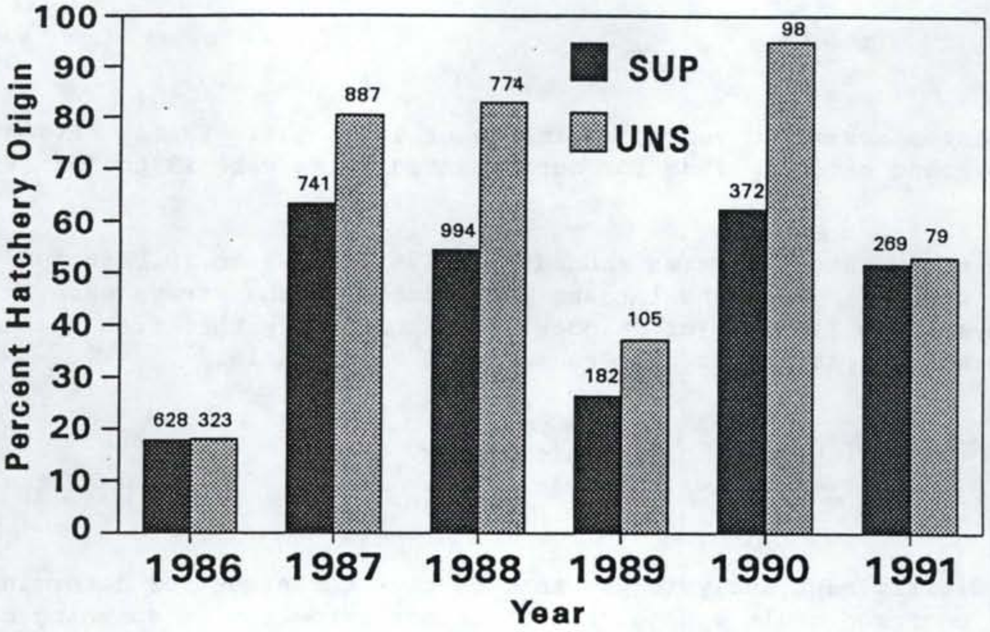

Figure 6. Percentage of fish that were hatchery origin estimated in supplemented (SUP) streams (Hurricane and Catherine creeks, and the upper Grande Ronde River) and unsupplemented (UNS) streams (Wenaha, Minam, and Lostine rivers). The estimated total number of spawners is above each bar.

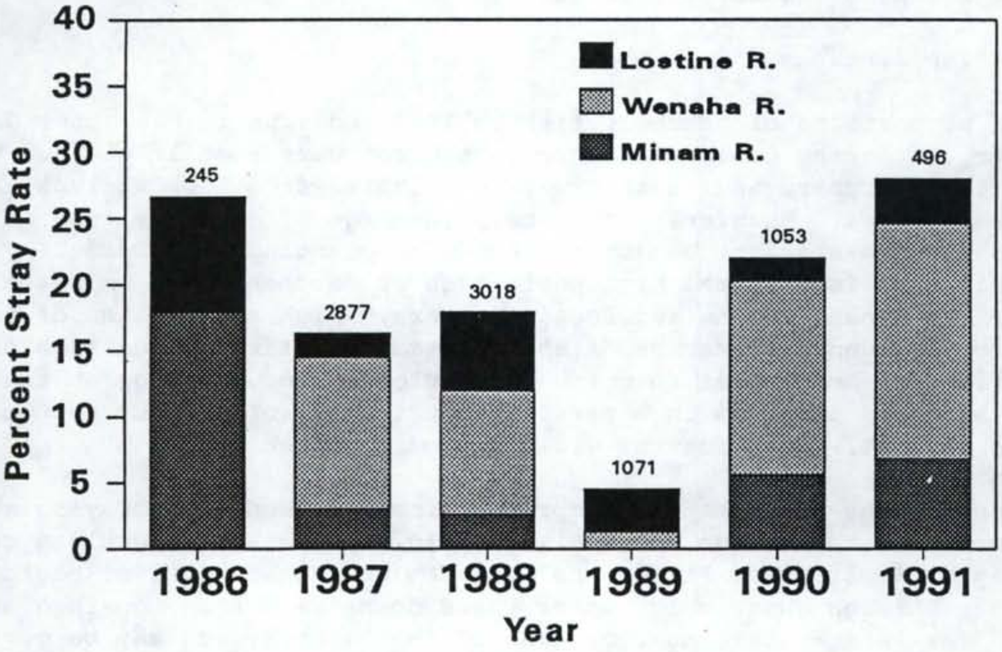

Figure 7. Minimum percent stray rates of hatchery spring chinook salmon estimated in unsupplemented streams (Wenaha, Minam, and Lostine rivers) in the Grande Ronde River basin. The estimated number of adults destined for hatchery facilities in the basin is above each bar.

unsupplemented areas had yearly estimates of 18 to 63%. Yearly estimates of overall percent hatchery fish for supplemented areas were 18 to 95% (Figure 7).

Minimum stray rate estimates ranged from 4.5 in 1989 to 28.0 in 1991 (Figure 7). In 4 of the 6 years the largest percentages of the strays were in the Wenaha River, the first major chinook salmon tributary that fish encounter while migrating upstream in the Grande Ronde River basin.

DISCUSSION

Technique

Linear discriminant analysis was an effective technique for determining the origin of unmarked adult spring chinook salmon recovered on spawning ground surveys in the Grande Ronde River basin from 1986 to 1991. This procedure may be useful for distinguishing between hatchery and naturally-produced fish in other subbasins and is currently being used to estimate the proportion of hatchery and natural spring chinook adults at Lower Granite Dam. A limitation of this technique is that it requires scales from known natural-origin adults. Our work has shown that scales from adults in unsupplemented areas can not be assumed to be fish of natural origin, making the collection of scales from known-origin natural adults difficult.

Management Implications

The high proportions of hatchery fish in 1987 and 1988 in the upper Grande Ronde River, Catherine Creek, and Hurricane Creek were most likely due to the more than 1500 hatchery adults that were outplanted from Lookingglass Hatchery each of these years. However, the large percentage of hatchery fish in unsupplemented areas cannot be attributed to outplanting. We were particularly surprised at the high percentage of hatchery fish on spawning grounds in the Wenaha, Minam and Lostine rivers. High stray rates of hatchery fish and low abundance of natural fish resulted in a large percentage of hatchery fish in the natural spawning populations. Results suggest that isolating hatchery programs in a particular tributary of a river system does not ensure that all adult returns will end up in target areas.

We are unsure why adult hatchery spring chinook salmon are straying at such high rates within the Grande Ronde River basin, however, we speculate that two factors may be implicated. First, the mainstem Grande Ronde River begins to warm rapidly through June in low-water years downstream from Lookingglass Hatchery. The later-returning component of the hatchery run may be moving into the Wenaha, Minam, and Lostine rivers because of cooler water temperatures in these tributaries. Secondly, smolts are released from Lookingglass Hatchery in late March or early April and have a prolonged migration period prior to arriving at Lower Granite Dam. Preliminary physiological and morphological data indicate that in general fish released from Lookingglass Hatchery in early April have not undergone complete transformation from parr to smolt. Thus they may not have imprinted well on Lookingglass Creek water prior to release.

Future Direction

We will continue to use scale pattern analysis, at least through returns of the 1989 brood, to identify the origin of natural spawners throughout the basin. All hatchery smolts from the 1990 brood and later will be marked for identification. We are planning to begin acquiring broodstock from local populations in the near future to reduce concerns of genetic introgression. Weirs have been proposed for the Wenaha, Minam and Lostine rivers to provide for the ability to remove hatchery strays if so desired. We plan to continue to investigate the potential causes for the high stray rates of hatchery fish produced at Lookingglass Hatchery and provide recommendations to reduce straying rates when desired.

ACKNOWLEDGMENTS

This work was funded by the U.S. Fish and Wildlife Service under the Lower Snake River Compensation Plan. We thank R.K. Bowden, E. Crateau, D. Herrig, R. Messmer, R. Mikus and T. Whitesel for their various contributions. Special thanks to M. Mclean for assistance in preparation of this document.

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LAKEWARD MOVEMENTS OF SUB-YEARLING CHINOOK SALMON IN THE KENAI RIVER, ALASKA

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ABSTRACT

Recent studies of juvenile chinook salmon in the mainstem Kenai River indicate that abrupt and significant declines in relative abundance occur during winter. Low densities of chinook salmon overwintering in the substrate of the Kenai River, or in tributary streams, do not adequately explain the observed disparity in seasonal abundance. The sudden decrease in catch rates and relative abundance during fall correlates with similar declines in stream discharge and water temperature.

We investigated the movements of sub-yearling chinook salmon inhabiting the mainstem by marking and releasing 90,105 fish in the lower river during July and August. Subsequent minnow and fyke trap catches from the lower 50 river miles were examined for the presence of marked fish. Marked chinook salmon were recovered both upstream and downstream from their point of release. Four marked fish were recovered below the tagging area, 24 marked fish were recovered at the outlet of Skilak Lake. Fish captured at Skilak Lake had migrated upstream approximately 35 miles during a 10 to 14 week period in the fall. These fish were significantly larger than those which remained in the lower mainstem. The use of Skilak Lake for overwintering by sub-yearling chinook salmon was indicated by their presence in catches from sites within the lake. The lakeward migration of juvenile salmon provides an explanation for the seasonal decline in juvenile salmon abundance in the mainstem, and has implications for land-use activities along the margins of the Kenai River.

THE IDENTIFICATION OF MORTALITY FACTORS AND MANAGEMENT SOLUTIONS FOR FALL RUN CHINOOK SALMON SMOLTS MIGRATING THROUGH THE SACRAMENTO/SAN JOAQUIN RIVER DELTA

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ABSTRACT

The Sacramento - San Joaquin River Delta is used as a nursery area and migration corridor for four races of chinook salmon in California's Central Valley, including the endangered winter run. Juvenile salmon, specifically the smolt stage, face a variety of problems during their migration through the Delta. These problems have been documented using mark and recapture methodology and have shown that fall-run smolt survival is negatively correlated to increases in water temperature, diversion, exports and reverse flows and to decreases in river flows. Five alternatives encompassing several measures to lessen these problems have been proposed to the State Water Resource Control Board which regulates water export operations in the Delta. Relative benefits of the selected alternatives were estimated using separate smolt mortality/survival models for the Sacramento and San Joaquin fall run populations. Relative costs are a function of reduced storage and project yield with some alternatives estimated to require over 1.0 million acre feet of water supply.

INTRODUCTION

Chinook smolt salmon face a variety of mortality factors migrating through California's Sacramento/San Joaquin River Delta on their way to the ocean. Most of these problems are caused by the present system of water management and are primarily related to changes in hydrology, whereby the timing, quantity, export and distribution of water flow through the Delta has been altered by the Central Valley Project (CVP) and State Water Project (SWP). These changes in the Delta environment have caused smolts to be diverted off their main migration pathways into less desirable regions where both direct and indirect losses from the CVP and SWP's water export facilities are high. Diversions as well as low and net reversed flows delay the migration of smolts through the Delta causing exposure time to mortality agents (such as higher water temperatures or predation loss) to be increased. The primary objective of this study was to determine the mortality factors and to identify potential protective measures for fall-run smolts migrating through the Delta.

METHODS

Mark and recapture methodology was used to measure fall-run smolt survival under various hydrologic and migration conditions. Coded wire tagged hatchery smolts were released at various sites within the Delta and recovered via midwater trawl near Chipps Island (Figure 1). A survival index for each group was calculated based on the number of recoveries, the number released and corrected for sampling effort, both temporally and spatially. Survival indices were correlated, using linear and multiple regression techniques, to various environmental variables to determine the factors important to smolt survival through the Sacramento/San Joaquin Delta. In some instances, the number of survival indices generated for the San Joaquin Delta were too few to develop meaningful multifactor relationships, thus an index of adult production (API) was used as a surrogate for smolt survival 2½ years earlier, since most adults return to the San Joaquin basin as three year olds (California Department of Fish and Game, 1987). This assumes a direct relationship between smolt survival in the Delta and adults in the spawning population 2½ years later. Environmental conditions correlated to the API were those the smolts were exposed to, 2½ years earlier as they migrated through the San Joaquin Delta.

RESULTS AND DISCUSSION

Smolt survival through the Sacramento/San Joaquin Delta has been shown to be negatively related to:

1) Diversions off the main rivers (Figures 2a and 2b)

2) Exports at the CVP and SWP pumping facilities (Figure 3a and 3b)

3) Reverse (upstream) river flows at Jersey Point (Figure 4a and 4b)

4) Increasing water temperatures at Freeport (Sacramento Delta) (Figure 5a)

5) Decreasing river flow at Vernalis (San Joaquin Delta) (Figure 5b)

Diversions off the main rivers: Diversion points off both the Sacramento and San Joaquin Rivers has been shown to be detrimental to smolt survival through the Delta. The Delta Cross Channel and Georgiana Slough divert water and smolt salmon) off the main Sacramento river into the Central Delta where migration to the ocean is delayed and exposure times to mortality factors (such as increased temperatures and predation) is increased. Diversion off the San Joaquin River is via Upper Old River, where water and salmon diverted are on a direct path to the two pumping facilities (SWP and CVP). Mortality down Upper Old River is higher than in the main river due to both the indirect (such as predation) and direct impacts (such as screen loss) of the projects. Comparisons of survival indices of those marked smolts diverted and those not diverted have shown a consistent (with the exception of the 1985 release in the San Joaquin Delta) benefit of staying in the main rivers (Figures 2a and 2b). Survival averaged about twice as great (ratio of 1 to 2.13 and 1 to 2.19) for smolts allowed to migrate downstream in the main rivers versus those that were diverted, on both the Sacramento and San Joaquin Rivers, respectively.

Exports at the CVP and SWP pumping facilities: The level of export (water transferred to the San Joaquin Valley) has been shown to be an important mortality factor for salmon smolts migrating through the Delta. Principally, on the Sacramento River, exports have been shown to impact the survival of marked salmon smolts in the Central Delta that have been diverted through the Delta Cross Channel and Georgiana Slough. After water temperature is removed from the variability associated with smolt survival, exports was found to be negatively correlated to survival in this area of the Sacramento river (Figure 3a). Since we did not have an adequate number of measurements of survival in the San Joaquin Delta, we used an index of adult production, 2½ years earlier to correlate to total exports. After the variability in survival due to river flow was removed, exports at the two facilities was significantly negatively correlated to our surrogate for San Joaquin smolt survival (adult production index, 2½ years later) (Figure 3b).

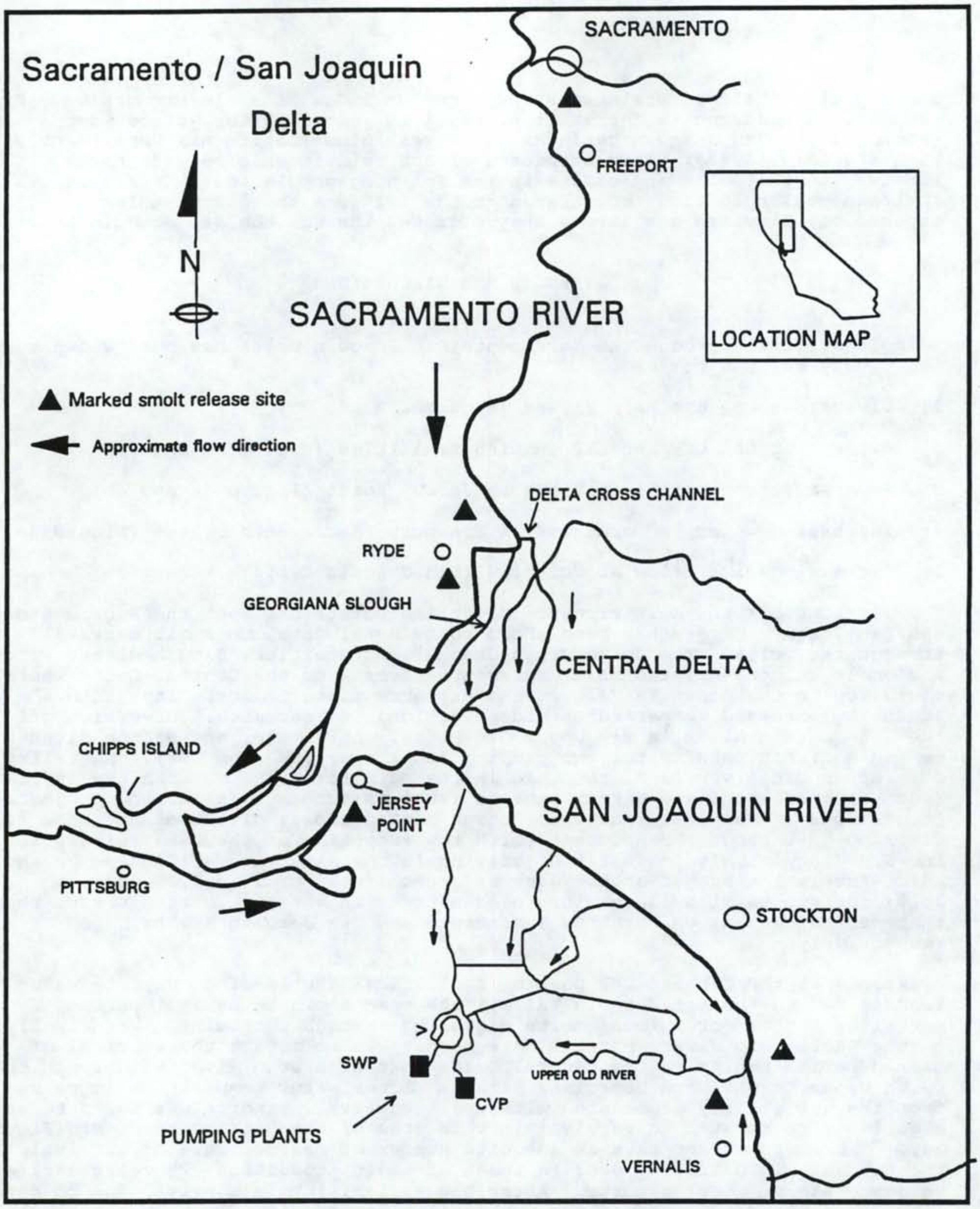

Figure 1. Map of the Sacramento / San Joaquin Delta of California with marked smolt release sites and approximate flow direction at certain areas.

Diversions

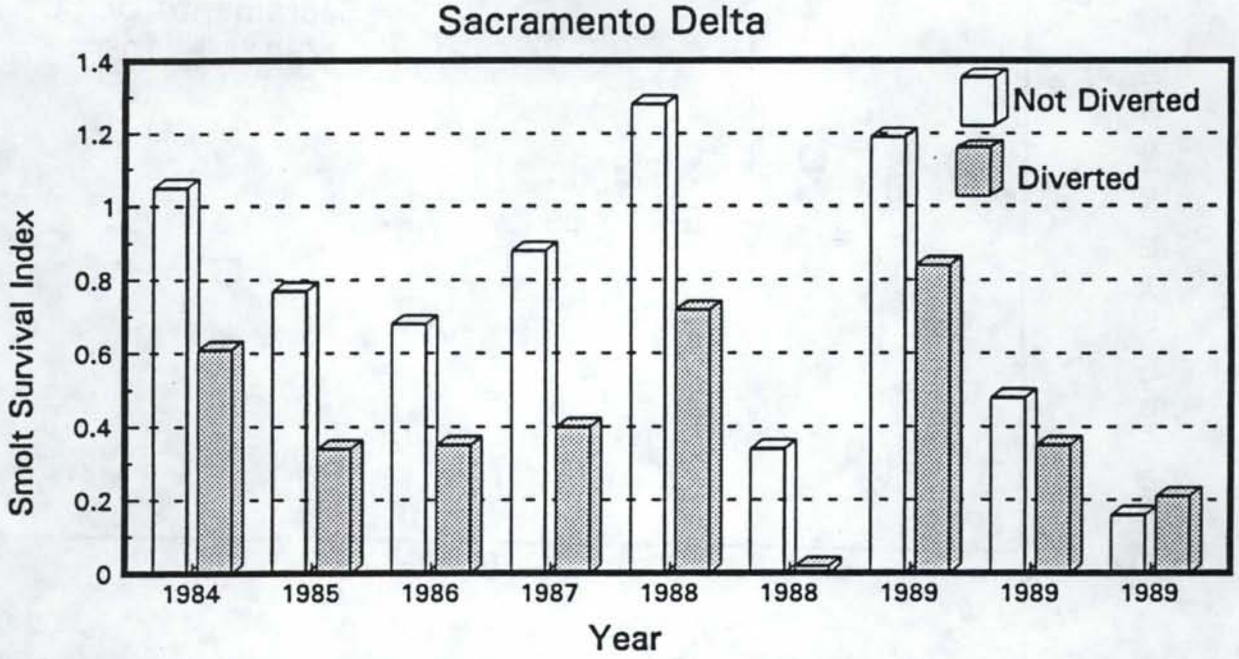

Figure 2a. Survival index for marked smolts released below (not diverted) and above two diversion channels (Delta Cross Channel and Georgiana Slough)

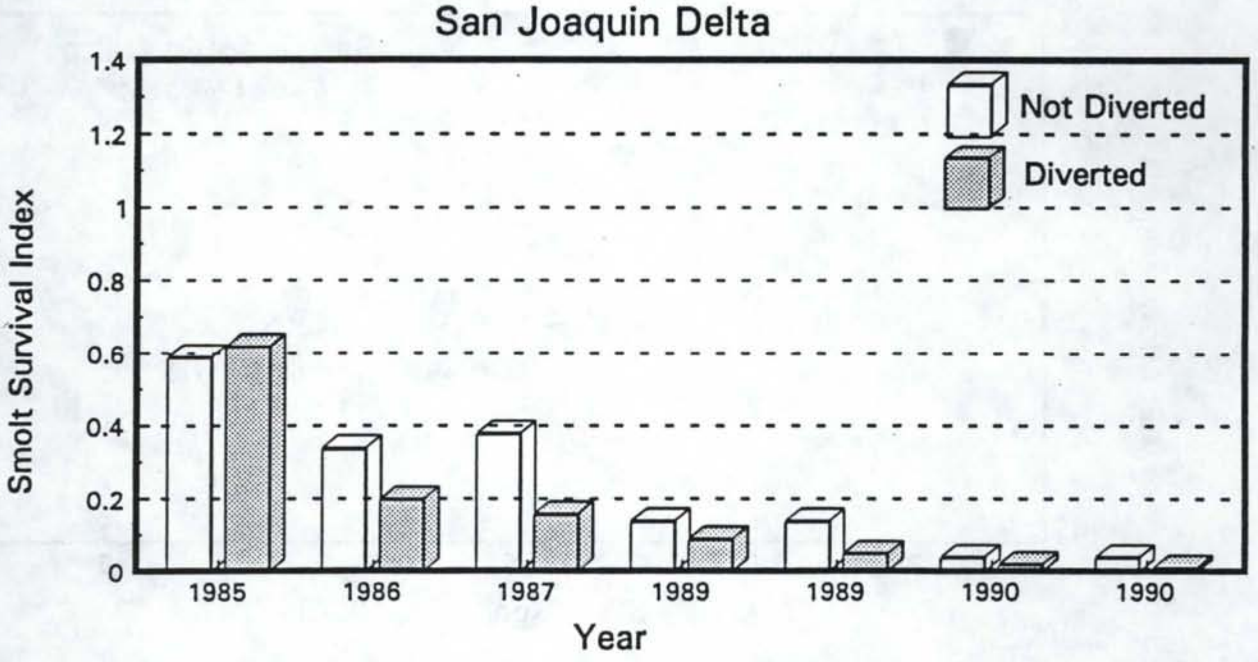

Figure 2b. Survival index for marked smolts released below (not diverted) and into the diversion channel.

Exports

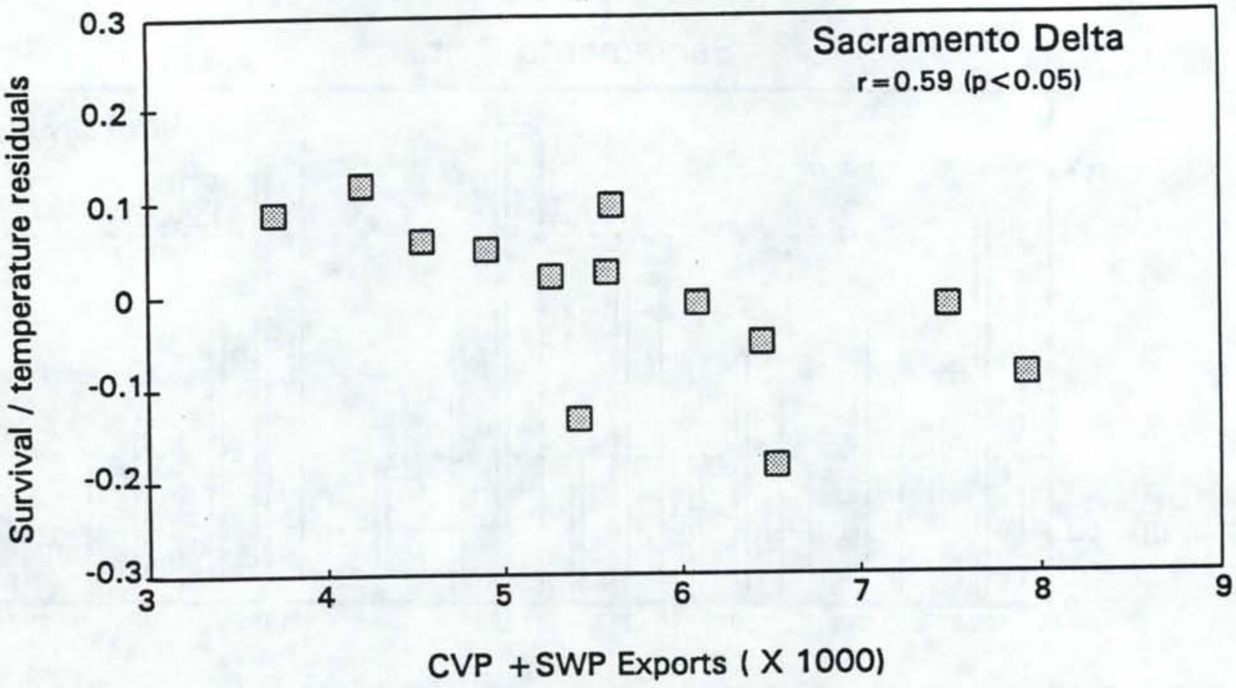

Figure 3a. The residuals between the survival of marked fish released into the Central Delta and water temperature versus exports.

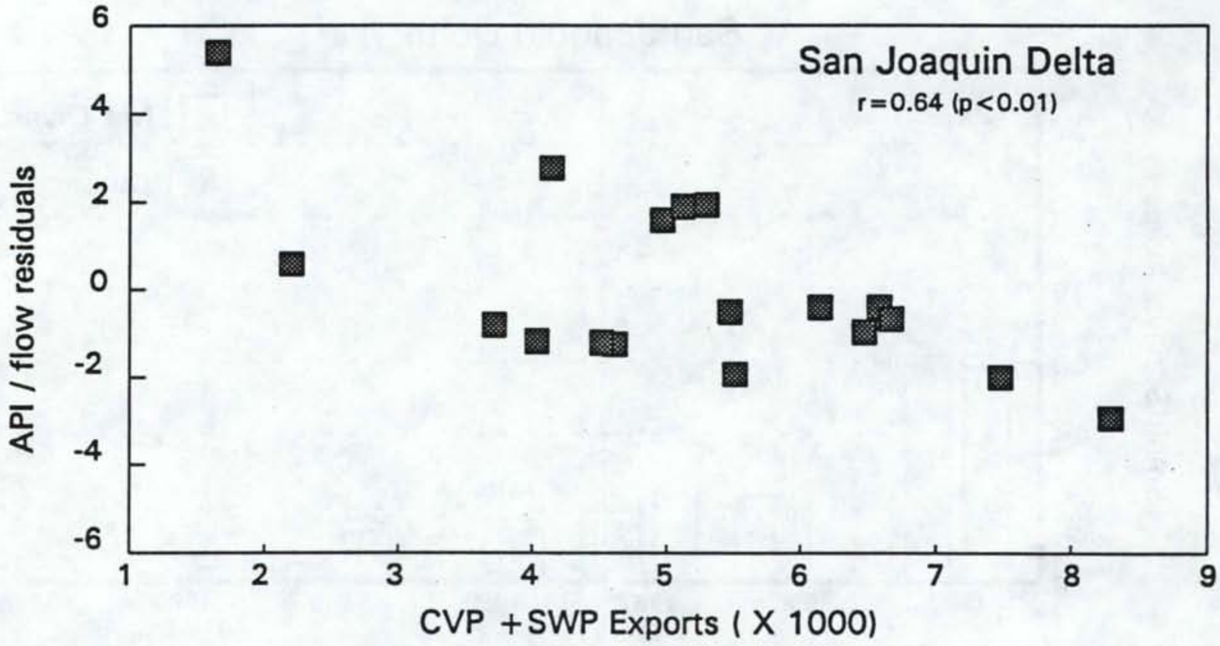

Figure 3b. The residuals between the API (adult production index) in the San Joaquin Basin and flow 2 1/2 years earlier, and exports.

Reverse Flows

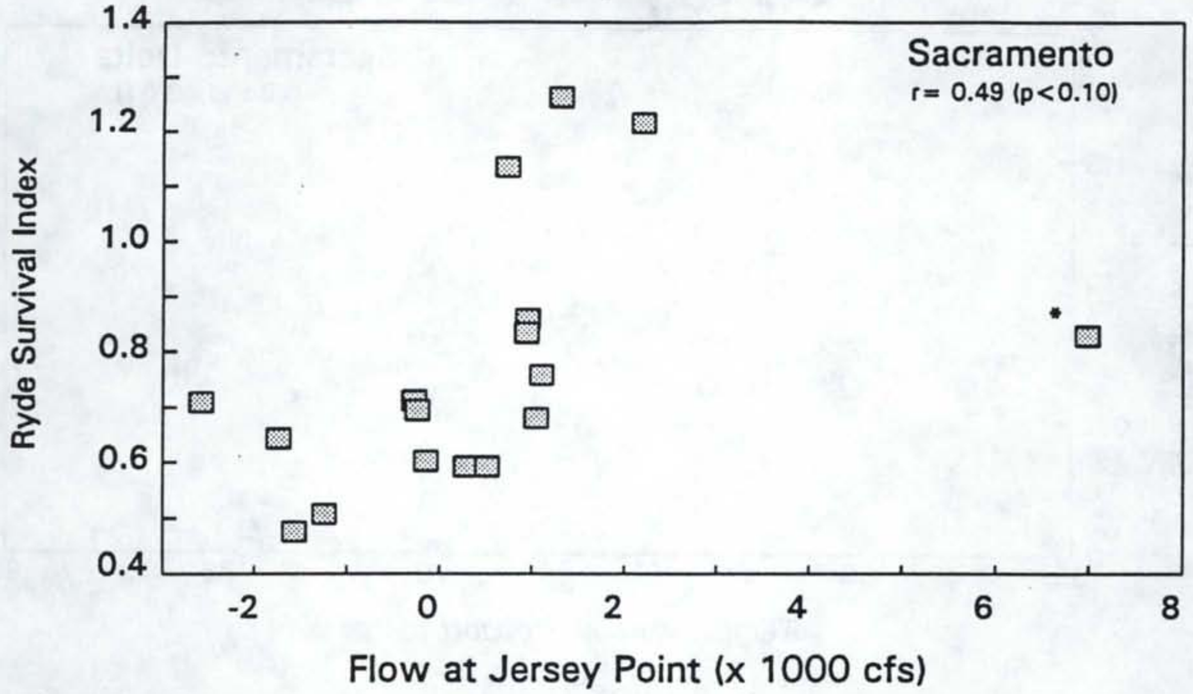

Figure 4a. Temperature corrected survival for fish released at Ryde versus San Joaquin river flow at Jersey Point. * = outlier

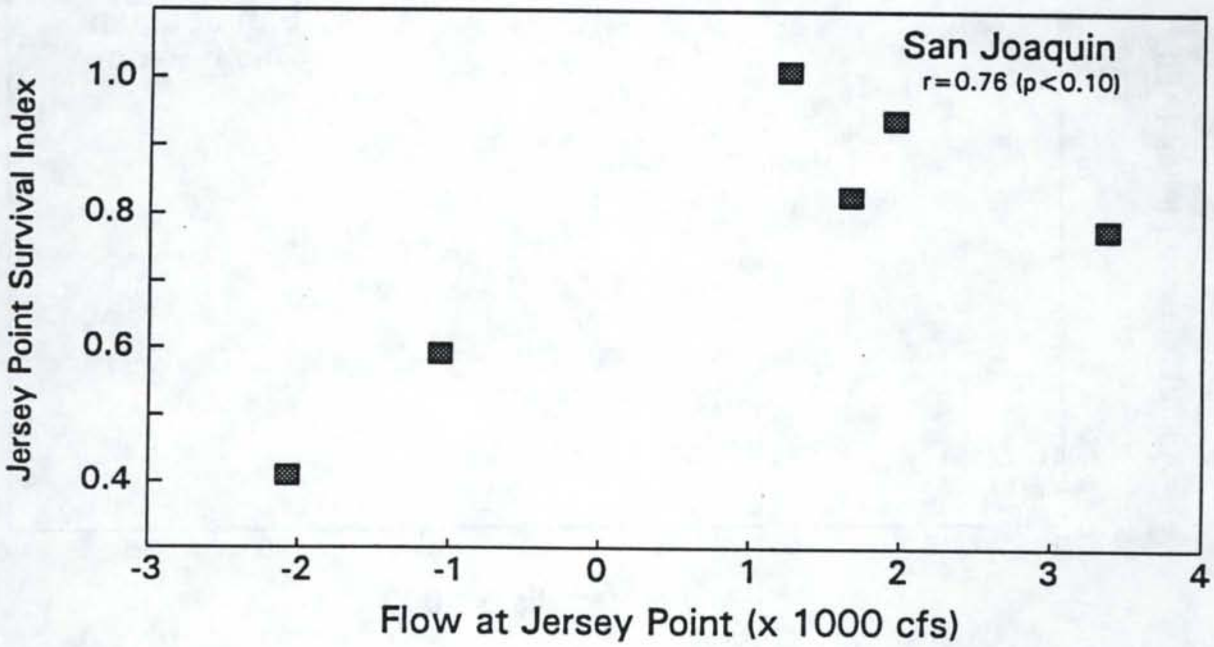

Figure 4b. Temperature corrected survival for fish released at Jersey Point versus San Joaquin river flow at Jersey Point.

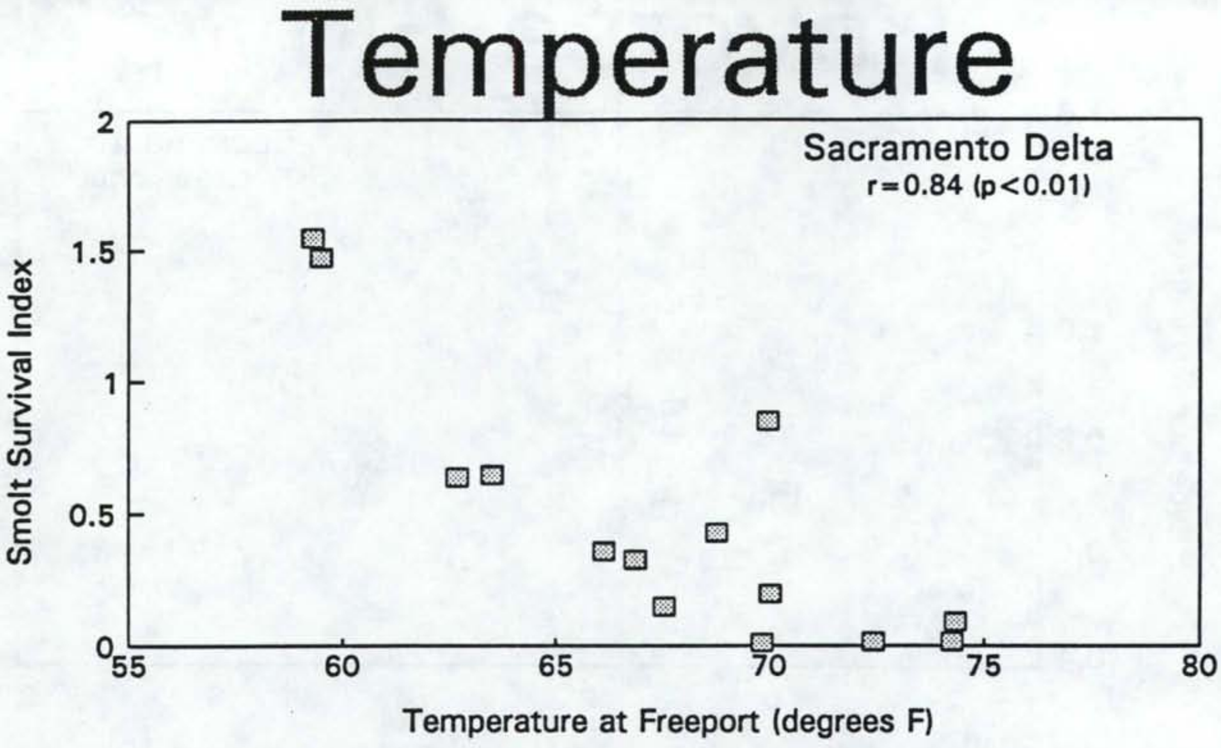

Figure 5a. Salmon smolt survival for marked fish released at Sacramento versus water temperature.

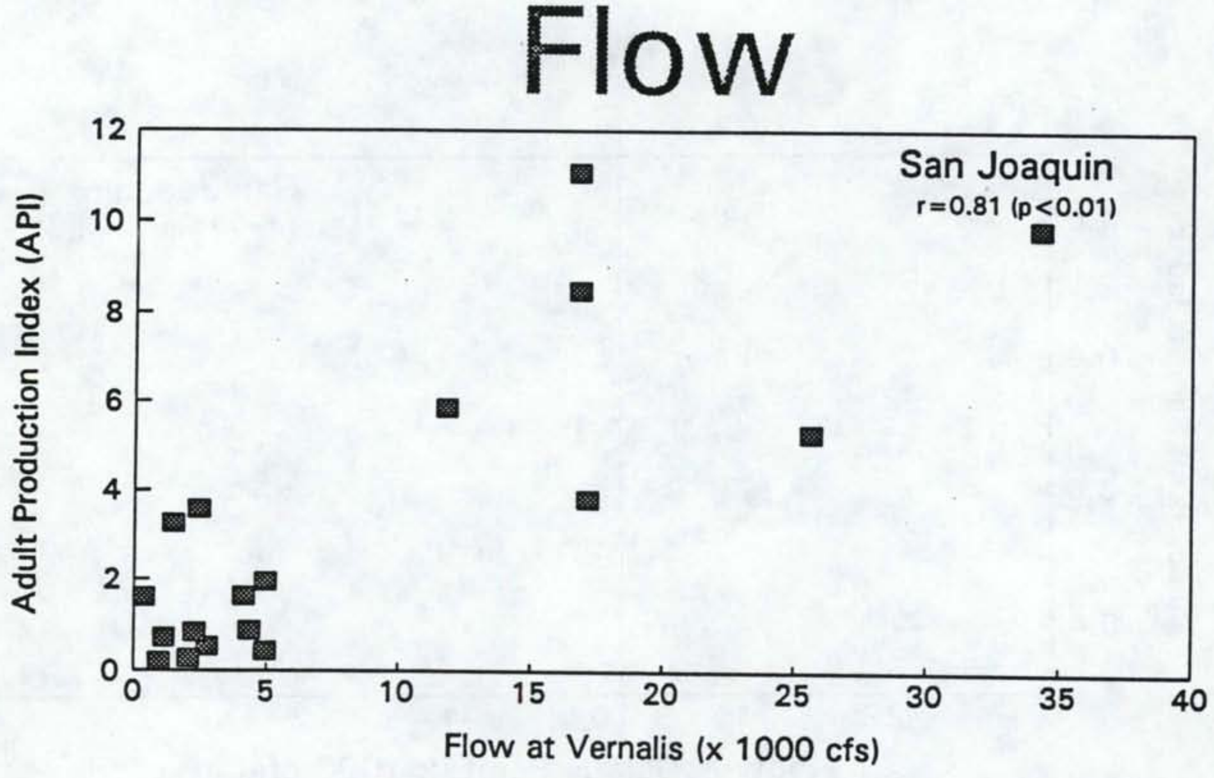

Figure 5b. An index of San Joaquin basin adult escapement versus river flow 2 1/2 years earlier.

Reverse (upstream) river flows at Jersey Point: Net river flows at Jersey Point vary depending upon inflow into the Delta, Delta water consumption and exports. Although this area of the river is massively effected by tidal flows, survival of marked fish released in both the Sacramento (at Ryde) and San Joaquin (at Jersey Point) have been shown to increase (significant at the 90% level) as net river flows increase from a negative 2000 to 3000 cfs to a positive 2000 to 3000 cfs (Figures 4a and 4b, respectively). The data generated from Ryde in 1986 was not used in the Sacramento Delta correlation as it was in a high flow year and an obvious outlier. The survival indices were standardized to reflect survival at a constant temperature (61 degrees fahrenheit), using a temperature/mortality relationship similar to that in figure 5a for the river reach between Ryde and Chipps Island.

High water temperatures at Freeport (Sacramento Delta): Water temperature at Freeport has been shown to be the most important variable in predicting smolt survival through out the Sacramento Delta. The relationship appears linear within the range of temperatures measured and mostly likely reflects the sublethal effects (such as increased metabolic rate, increased predation and stress) on salmon smolts of increased temperatures in the field (Figure 5a). Although, due to limited data, temperature has not been shown to affect smolt survival in the San Joaquin Delta, we have no reason to believe that the response of salmon smolts to temperature would be different between basins.

Decreasing river flow at Vernalis (San Joaquin Delta): Decreasing San Joaquin river flow at Vernalis has been shown in a multiple regression analyses to be negatively correlated to our index of adult production, 2½ years later (Figure 5b). Decreasing flows would tend to increase temperature, and the efficiency of sight feeding predators.

Potential protective actions to increase Delta smolt survival have been identified and include: barriers or gate closures to prevent diversion of smolts off the main rivers, export limitations and flow augmentation from upstream reservoirs during critical outmigration periods and various methods to reduce water temperatures (Table 1).

Protective actions were grouped to provide regulatory agencies (State Water Resources Control Board) with a series of management alternatives that encompass a range of protective levels for salmon smolts (Table 2).

Relative benefits of any of the proposed management alternatives were estimated using separate smolt mortality/survival models for the Sacramento and San Joaquin Delta's. A hydrologic model was used to estimate base hydrologic conditions and the proposed actions during identified time periods were added to estimate the modified hydrologic conditions based on the selected protective criteria. Smolt survival was then estimated using the environmental variables and the two models shown in Figures 6a and 6b.

Alternative A, although of relatively small survival benefit to smolts had the least cost in terms of lost yield or storage for the water projects. Conversely, Alternative E had the greatest benefit to smolts with the greatest water costs to the projects (Table 3) and has been estimated to be over 1.0 million acre feet.

CONCLUSIONS

Although it is very important to identify the specific problems facing the various life stages of salmon, it is equally important to identify management solutions and to estimate potential relative benefits and costs of any one or series of protective management alternatives.

Table 1. List of potential protective actions to increase smolt survival in the Sacramento/San Joaquin delta.

Potential Action	Problem Addressed
Close Delta Cross Channel Gates	Diversion
Barrier in Georgiana Slough	Diversion
Barrier in Upper Old River	Diversion
Export Limitations (during critical periods)	Exports Reverse flows
Positive flow at Jersey Point	Reverse flows
Increase flows at Vernalis	Reverse flows Low flows
Decrease ag drainage	High temperatures
Increase riparian vegetation	High temperatures
Reservoir releases	High temperatures

Table 2: Proposed selected alternatives to increase smolt survival in the Sacramento/San Joaquin Delta. Alternatives range from a low level of protection (Alternative A) to a high level of protection (Alternative E) for primary fall-run salmon smolts.

Alter.	Water Year Type	Close Delta Cross Channel	Close Georgiana Slough	Max Total CVP/SWP Exports[1]	Full Barrier Upper Old River	Minimum Flow Vernalis [1]	Minimum Flow Jersey Point[1]	
A	Wet	5/1-31	—	--	4/1 to 5/31 and 9/1 to 11/30 all water year types	--	0 cfs during time cross channel gates closed	
	Above Normal	5/1-31						
	Below Normal	5/1-31						
	Dry	2 wks May						
	Critical	2 wks May						
B	Wet	5/1 to 5/31 all water year types	--	May and June 5000 cfs all water year types	4/1 to 5/31 and 9/1 to 11/30 all water year types	3500 cfs 4/1-5/31	500 cfs during time cross channel gates closed	
	Above Normal					3000 cfs "		
	Below Normal					2500 cfs "		
	Dry					2000 cfs "		
	Critical					1500 cfs "		
							4/15-4/22 and 5/7-6/15	4/23-5/6
C	Wet	4/15 to 6/15 all water year types	--	6000 cfs 4/23-5/6	4/1 to 5/31 and 9/1 to 11/30 all water year types	10000 cfs 4/23-5/6	1000 cfs	3000 cfs
	Above Normal			5000 cfs "		8000 cfs "	1000 cfs	2500 cfs
	Below Normal			4000 cfs "		6000 cfs "	1000 cfs	2000 cfs
	Dry			3000 cfs "		4000 cfs "	1000 cfs	1500 cfs
	Cricial			2000 cfs "		2000 cfs "	1000 cfs	1000 cfs
							4/1-4/14 and 5/16-6/30	4/15-5/15
D	Wet	4/1 to 6/30 all water year types	4/15 to 6/15 all water year types	6000 cfs 4/15-5/15	4/1 to 5/31 and 9/1 to 11/30 all water year types	10000 cfs 4/15-5/15	1000 cfs	3000 cfs
	Above Normal			5000 cfs "		8000 cfs "	1000 cfs	2500 cfs
	Below Normal			4000 cfs "		6000 cfs "	1000 cfs	2000 cfs
	Dry			3000 cfs "		4000 cfs "	1000 cfs	1500 cfs
	Critical			2000 cfs "		2000 cfs "	1000 cfs	1000 cfs
E	Wet	2/1 to 6/30 all water year types	2/1 to 6/30 all water year types	4/1 to 6/30 zero export all water year types	2/1 to 6/30 and 9/1 to 11/30 all water year types	10000 cfs 4/1-6/30	3000 cfs 4/1-6/30	
	Above Normal					8000 cfs "	2500 cfs "	
	Below Normal					6000 cfs "	2000 cfs "	
	Dry					4000 cfs "	1500 cfs "	
	Critical					2000 cfs "	1000 cfs "	

[1] Flows and exports are mean daily averages.

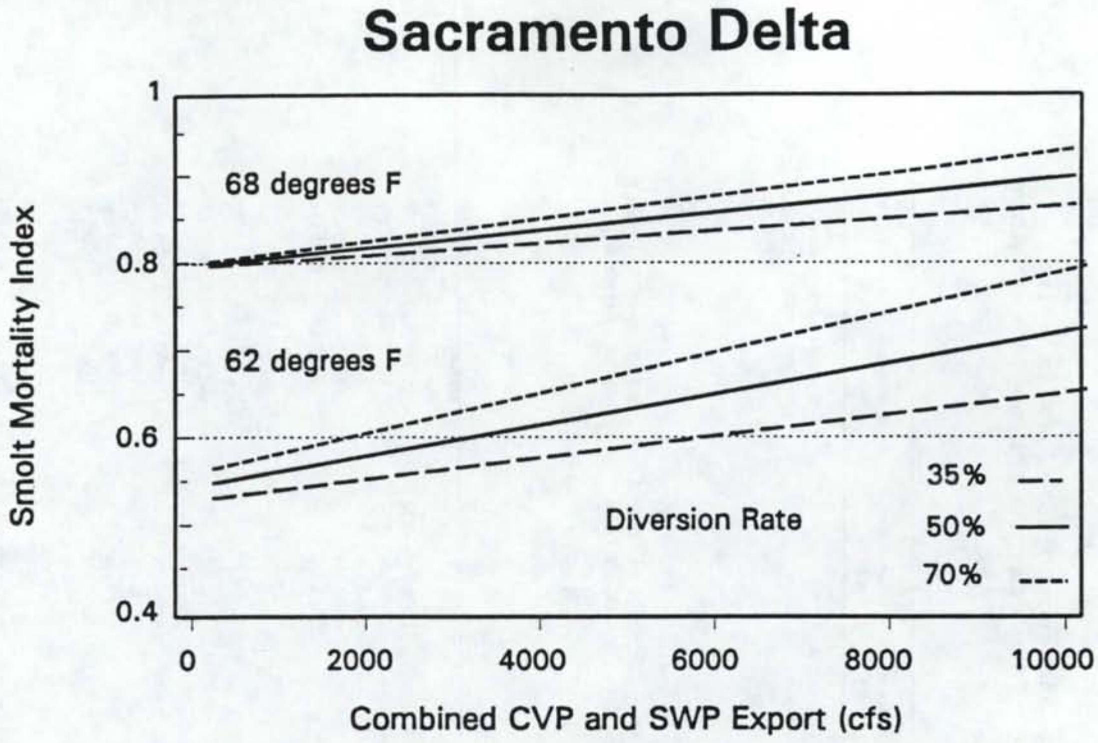

Figure 6a. Model between Sacramento Delta smolt mortality, water temperature, percent divided and exports.

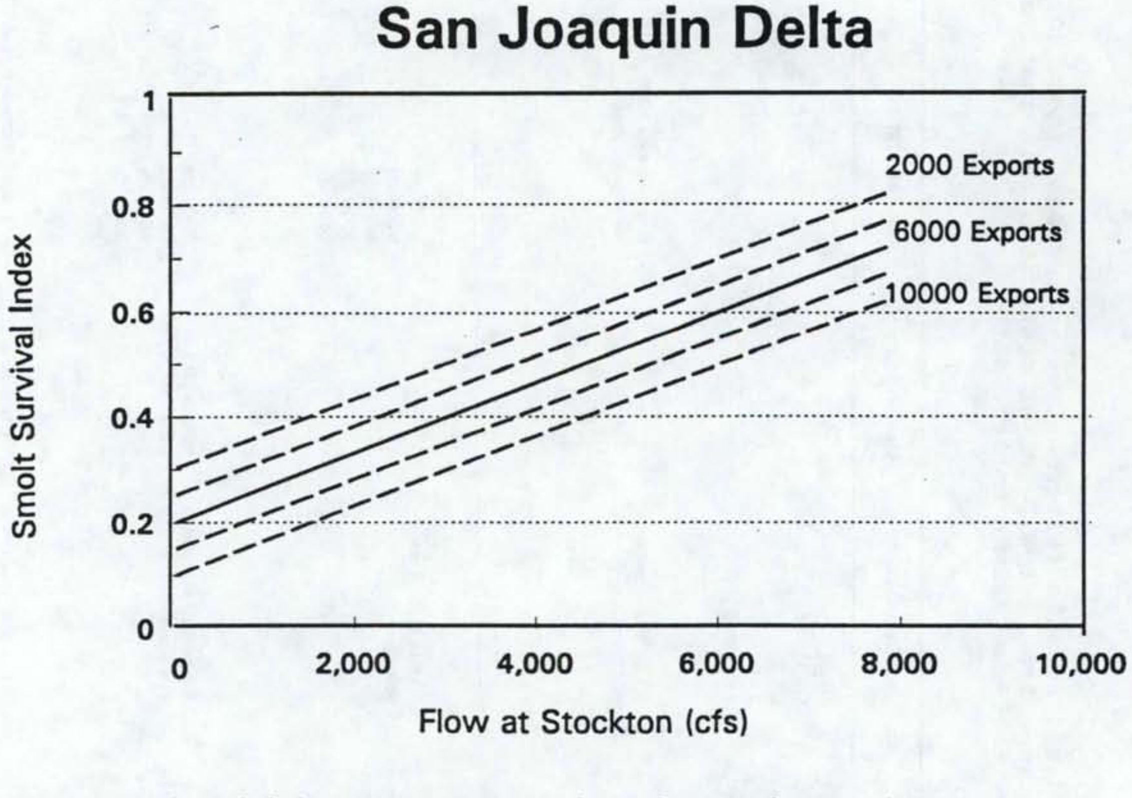

Figure 6b. Conceptual model between San Joaquin Delta smolt survival, river flow and exports.

Table 3: Estimated benefits in smolt survival of five selected alternatives for the five water year types.

Alternatives	Water Year Type	Fall-Run Smolt Survival Index Sacramento	San Joaquin
		.24	.10
		(Base Operational Study $\bar{X}$)	
	Wet	.39	.38
	Above Normal	.30	.19
A	Below Normal	.27	.16
	Dry	.21	.16
	Critical	.19	.16
		[1] $\bar{X}$ = .27	$\bar{X}$ = .21
	Wet	.40	.39
	Above Normal	.31	.21
B	Below Normal	.27	.19
	Dry	.23	.17
	Critical	.21	.16
		$\bar{X}$ = .28	$\bar{X}$ = .22
	Wet	.40	.42
	Above Normal	.32	.25
C	Below Normal	.30	.21
	Dry	.26	.19
	Critical	.24	.18
		$\bar{X}$ = .30	$\bar{X}$ = .25
	Wet	.48	.46
	Above Normal	.39	.30
D	Below Normal	.37	.26
	Dry	.32	.23
	Critical	.29	.20
		$\bar{X}$ = .37	$\bar{X}$ = .29
	Wet	.49	.62
	Above Normal	.40	.49
E	Below Normal	.38	.41
	Dry	.33	.34
	Critical	.30	.26
		$\bar{X}$ = .38	$\bar{X}$ = .42

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[1] Mean of the 5 water year types

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A PACIFIC SALMON PRODUCTION PLAN FOR THE COLUMBIA RIVER BASIN, USA

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ABSTRACT

A salmon restoration program to maintain, restore, and enhance Columbia River basin salmon populations above Bonneville Dam by means of supplementation and hatchery reprogramming is described. The program is being developed in accordance with both sound scientific principles of conservation management, and applicable laws and regulations such as United States Endangered Species Act and the United States and Canada Pacific Salmon Treaty.

Supplementation is the stocking of genetically and ecologically compatible fish into the natural habitat to increase the abundance of naturally reproducing fish populations. Hatchery reprogramming is the changing of operations of existing salmon hatcheries to directly support the maintenance, restoration and enhancement of salmon runs above Bonneville Dam. Supplementations and hatchery reprogramming are related activities in that existing hatchery facilities can be reprogrammed to support a supplementation program. Habitat restoration and protection are important components of a salmon production program.

Utilizing small scale and central supplementation facilities broodstock will be selected from natural fish returning to native spawning areas. Spawning protocol includes: implementation of "no genetic selection" protocol, maintenance of a large effective breeding size, targeted sex ratios, and minimizing genetic risks to the wild/natural populations. Rearing protocol includes natural rearing conditions with less intense aquaculture to mimic important natural rearing conditions: low loading levels, natural food, cover and predator avoidance conditioning, and stress reduction. Release protocols include low seeding levels, life stage, size and age of release, acclimation for stress reduction and imprinting, time of release and dispersal.

Habitat maintenance and improvement is considered essential to meeting the production goals. Spawning and rearing habitat has been negatively impacted by land use practices, and the migration corridor by dams.

A regional assessment of production projects will provide an overview of ongoing and planned activities. The assessment will construct a conceptual framework and model to estimate the potential benefits and risks of projects and to plan for regional coordination of research and monitoring.

BEAR VALLEY CREEK - BENEFIT OR BOONDOGGLE?

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ABSTRACT

Much of the focus on expenditures or ratepayer (i.e. Bonneville Power Administration) money has shifted away from habitat improvement in the tributaries to passage improvement in the mainstems. This change in emphasis has called into question the benefits of many, if not all, habitat projects - past, present, and future. An example of a project which has been questioned as to the benefits compared to the cost is the Bear Valley Creek habitat enhancement project.

Bear Valley Creek was extremely degraded due to dredge mining for rare earth metals in the 1950s. The mined section of the creek was rehabilitated in the late 1980s. Little was done to the stream itself; most work involved development of a new floodplain along 2-1/2 miles of stream. Total cost of the project was just under $3 million. Monitoring, both biological and physical, in the project site area has been continuous since 1984. Results from the monitoring effort in the form of photo-documentation are presented.

DIFFERENCES IN AGE STRUCTURE, SIZE, AND SURVIVAL OF LAKE AND STREAM-REARED COHO SMOLT POPULATIONS IN NORTHERN SOUTHEAST ALASKA

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ABSTRACT

Investigations of two coho salmon (*Oncorhynchus kisutch*) populations near Sitka show that lake-reared smolts have a multiple age-class structure (ages 2 through 6) while nearby stream-reared smolts are primarily age 2 with lesser numbers of age 3 individuals. The multiple age-class structure of smolt populations in lakes may buffer these coho populations during years when low escapements or other factors limit recruitment. Lake-reared smolts are larger at all ages than are stream-reared smolts. The increased size of lake-reared smolts provides a survival advantage to these lake-rearing populations.

Differential tagging of smolts by size class showed a much higher return of jacks (0-ocean adults) from smolts which emigrated at fork lengths over 150 mm. Smolt to adult survival ratios of the different age groups of lake-reared coho varied considerably within a given year. Smolt to adult survival of stream reared smolts was lower than survival of lake reared smolts in a given year.

IMPLEMENTING A HABITAT RESTORATION PLAN IN THE COOS AND COQUILLE BASINS, OREGON

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ABSTRACT

Beginning in June of 1990, we began a project to restore winter habitat for coho salmon. This involves placing large and/or complex material in pools, building off-channel or backwater ponds, replanting riparian areas with trees, constructing fence systems to allow riparian recovery as well as livestock production, working with timber companies through the Stream Enhancement Initiative, and evaluating our work to guide future projects. Current research has shown winter habitat to be limiting the production of coho salmon. We contract with Shutter Creek Correctional Institute to utilize a 10-man inmate crew to perform the manual labor. The results of the evaluation has been encouraging. Pools treated with structure held an average of 28 fish through winter compared to an average of 6 fish in untreated pools on one creek. One large treated pool holding 1,500 fish the first winter held 2,250 fish the second winter. The plan we developed can be used as a template through the region to conduct similar work.