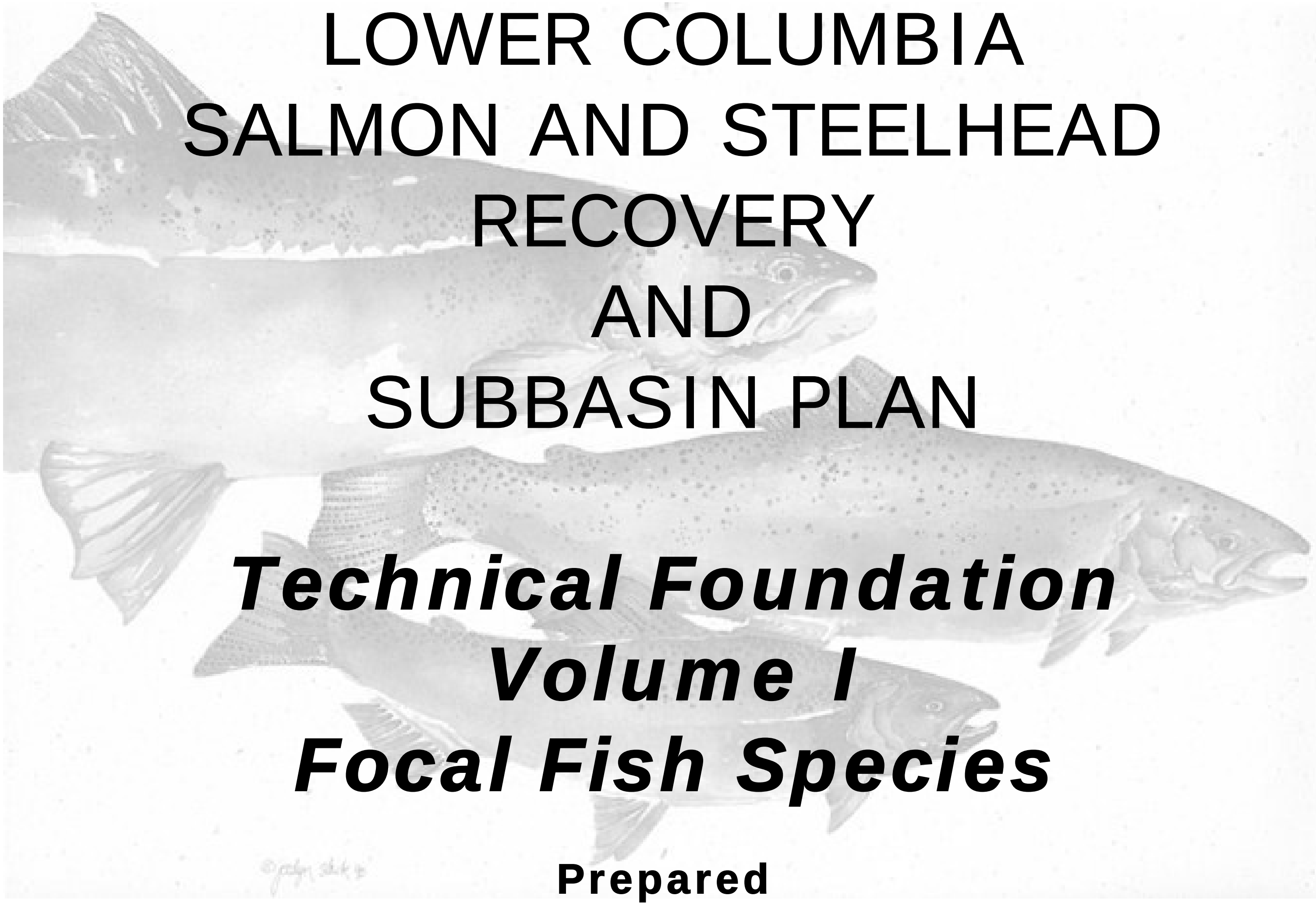

LOWER COLUMBIA SALMON AND STEELHEAD RECOVERY AND SUBBASIN PLAN

Technical Foundation Volume I Focal Fish Species

**Prepared
For
Northwest Power
And
Conservation Council**

MAY 28, 2004
DRAFT

Prepared By:
Lower Columbia Fish Recovery Board

Preface

This is number one of six volumes of a Technical Foundation for Recovery and Subbasin Planning prepared under direction of the *Washington Lower Columbia River Fish Recovery Board*. This information provides a basis for an integrated Salmon Recovery and Subbasin Plan prepared by the *Fish Recovery Board*. The Technical Foundation is an encyclopedia of information relating to focal and other species addressed by the plan, environmental conditions, ecological relationships, limiting factors, existing programs, and economic considerations. The Technical Foundation summarizes existing information and new assessments completed as part of the planning process. A separate Executive Summary document provides an overview of the entire Technical Foundation.

Technical Foundation volumes include:

Vol. I	Focal Fish Species	*Species overviews, limiting factors, recovery standards, and status assessments for lower Columbia River chinook salmon, coho salmon, chum salmon, steelhead, bull trout, and cutthroat trout*
Vol. II	Subbasins	*Fish populations and habitat conditions in each of 11 Washington lower Columbia River subbasins*
Vol. III	Other Species	*Descriptions, status, and limiting factors of other fish and wildlife species of interest to recovery and subbasin planning*
Vol. IV	Existing Programs	*Descriptions of Federal, State, Local, Tribal, and non governmental programs and projects that affect or are affected by recovery and subbasin planning*
Vol. V	Economic Assessment	*Potential costs and economic considerations for recovery and subbasin planning*
Vol. VI	Appendices	*Methods and detailed discussions of assessments completed as part of this planning process*

This work was funded by the *State of Washington* and the *Northwest Power and Conservation Council*. The Technical Foundation was completed primarily by the *Washington Lower Columbia Fish Recovery Board*, *Washington Department of Fish and Wildlife*, *S.P. Cramer and Associates*, and *The White Company*. This second draft of the Technical Foundation incorporates suggestions and revisions provided by a wide array of agency and public reviewers of an initial draft distributed in 2003. Additional opportunities for review and revision of the current draft will occur as part of ongoing recovery and subbasin planning processes

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1.0 Introduction and the Recovery Planning Process

1.1 Introduction

1.1.1 Technical Foundation

This document is a final draft of the Lower Columbia Fish Recovery Board's (LCFRB) subbasin planning Technical Foundation for the Washington side of the Lower Columbia. It is the first step in a process to develop a scientifically credible, socially and culturally acceptable, and economically and politically sustainable plan to:

- restore the region's four fish species listed as threatened under the federal Endangered Species Act (ESA) to healthy, harvestable levels, and
- protect and enhance other fish and wildlife species that have been adversely affected by the development and operation of the federal Columbia River Power System.

To complete the recovery/subbasin plan as intended by May 2004, the planning process is separated into two distinct, but related phases: the Technical Foundation and the Management Plan. Together, the two phases are envisioned to address five central questions about listed anadromous fish and other fish and wildlife species in the Lower Columbia Basin:

- Where are we now?
- How did we get here?
- Where do we need to go?
- How do we get there?
- How do we know when we're there?

The Technical Foundation is found in Volumes I through VI. It is a comprehensive collection and analysis of technical information relating to the plan's focal fish and wildlife species and the environmental and human activities and programs that affect their health and viability. The Foundation describes current conditions and sets forth recovery targets, biological goals, and proposed analytical approaches. While considerable data exists, significant gaps and variations remain across the region. To fill these gaps, analyses were designed to capitalize on the strengths and balance the weaknesses of existing fish, habitat, and program data.

The Technical Foundation covers an immense amount of complex information about very complicated issues. It was therefore necessary that the Foundation be strategically organized so that the five key questions could be addressed as efficiently as possible, as follows.

- Introduction and The Recovery Planning Process – Volume I, Chapter 1
  - *Where are we now? Where do we need to go? How do we get there?*
  - Chapter 1 describes the overall purpose and organization of this Technical Foundation. It also provides an overview of the recovery planning process, the statutory basis of the process, and the key organizations and their responsibilities and the recovery decision-making process.
- Species Overview – Volume I, Chapter 2
  - *Where are we now? How did we get here?*
  - Chapter 2 provides a basis for subsequent chapters through compilations and descriptions of each focal species' 1) definitions of populations, 2) life history, abundance, distribution, productivity characteristics, 3) genetic diversity, 4) and Evolutionarily Significant Unit (ESU) definition, and listing status.
- Limiting Factors – Volume 1, Chapter 3

- *Where are we now? How did we get here?*
- Chapter 3 serves as a thorough examination of all the factors that limit productivity and abundance of the focal species, with particular attention to human-induced factors. A thorough analysis of the effects of fisheries, hatcheries, and habitat alterations on production and abundance is presented.

• Conceptual Framework and Recovery Standards – Volume 1, Chapter 4
  - *Where do we need to go? How do we get there? How do we know when we're there?*
  - Chapter 4 presents a proposed approach for evaluating the status of focal species' populations and an analytical approach for recovery that includes defining recovery goals, using a life-cycle focus, addressing all factors affecting recovery, defining methods that lead to specific recovery actions, and maintaining flexibility. Chapter 4 also describes standards for developing recovery targets and addresses the issues surrounding defining the targets for each population, minimum viability versus harvestable surpluses, definitions of goals, targets, and metrics, and balancing biological and social goals.

• Assessments of Current Status and Limiting Factors – Volume 1, Chapter 5
  - *Where are we now? How did we get here? Where do we need to go? How do we get there?*
  - Chapter 5 is a broad examination of the current viability, status, and planning ranges for recovery targets of each ESU and its component populations. The chapter also broadly evaluates the impact of each of six humanly manageable factors (dams, hatcheries, fisheries, stream habitat, mainstem and estuarine habitat, and predation) on the populations. More detailed analyses of these factors are addressed specifically for each subbasin in Volume II.

• Subbasin Analyses – Volume II
  - *Where are we now? How did we get here? Where do we need to go? How do we get there?*
  - Volume II addresses specific conditions and factors that limit recovery of focal species in each subbasin. The first several chapters describe the analytical methods applied to the subbasin-by-subbasin analysis in the remaining chapters. Each subbasin analysis ends with recommended remedial actions for planners to consider.

• Other Species – Volume III
  - *Where are we now? How did we get here?*
  - Volume III provides descriptions of the status, impacts, and, as appropriate, population rebuilding challenges for a number of species that are also considered to be potentially affected by the same human impacts that have limited the focal species.

• Programs – Volume IV
  - *Where are we now? How did we get here?*
  - Volume IV inventories existing programs and activities that directly affect or are affected by salmon recovery activities. These include fish protection, restoration, and artificial production activities and programs implemented by Federal, Tribal, State, and local governments as well as significant non-governmental programs.

• Economics – Volume V
  - *Where are we now? How did we get here?*
  - This volume describes economic background information useful for evaluating tradeoffs, costs, and benefits of recovery actions. It describes the economic base for the region, the relationship between plan actions both adverse and beneficial, stakeholder types who might suffer adverse impacts, and unit cost and benefit information for anticipated actions.

• Technical Appendices – Volume VI
  - *Where are we now? How did we get here? How do we get there?*
  - Volume IV contains detailed descriptions of, and background for, the analytical methods applied in Volume II, particularly the methods for Integrated watershed Analysis; and ecosystem

Dignosis and treatment. Additionals analyses completed as part of this planning process are also included in this volume.

1.1.2 Subsequent Management Plan

In the Management Plan, to be reported separately, Federal and state agencies, tribes, local governments, and the people of the region will develop the path to the recovery goals through a collaborative process. That process will focus on various aspects of the basic questions: Where do we need to go? How do we get there? and How do we know when we're there? The Technical Foundation is intended to inform these decision-makers and the public and to assist them in shaping alternatives, understand potential tradeoffs, develop recovery strategies, identify necessary actions, and set priorities. Recovery targets and criteria for listed salmon and steelhead have been developed in consultation with the National Oceanic and Atmospheric Administration (NOAA) Fisheries Technical Recovery Team (TRT) and the Willamette/Lower Columbia ESA Executive Committee. During the second phase of the planning process, these targets will assist decision-makers in forging an effective and practical approach for recovering listed fish. Bull trout recovery goals and criteria will be taken from the U.S. Fish and Wildlife (USFWS) draft bull trout recovery plan. Biological objectives for the remaining focal species will be developed during development of the Management Plan. In recognition that a successful recovery/subbasin plan must meet the needs of both the focal fish and wildlife species and the people of the region, the planning phase will meld science with social, cultural, and economic considerations to produce an effective recovery program that can be implemented and sustained over the long term. In June 2004, the Plan will be submitted to NOAA Fisheries, USFWS, the state, and NPCC for review and approval. It is anticipated that the Plan will be finalized and approved by December 2004.

1.2 Recovery/Subbasin Planning Process

1.2.1 Overview

This section discusses the scope and context of the overall Washington Lower Columbia Recovery/Subbasin planning effort being led by the LCFRB. It explains how this planning process meets the needs of recovery planning for fish species listed as threatened according to the federal Endangered Species Act (ESA). It describes how the process addresses Northwest Power and Conservation Council (NPCC) subbasin planning requirements for rebuilding fish and wildlife adversely affected by the development and operation of the Columbia River hydropower system. Finally, it describes how the planning process relates to the state salmon recovery and watershed management planning processes. (These individual planning efforts are summarized below.)

The section also provides an overview of the decision-making process and describes the framework that brings different stakeholders and interested parties together as participants in the planning process. The section also discusses how the planning effort relates to other plans and processes, and how different entities and interests are working together to recover the diverse fish and wildlife resources that once defined the lower Columbia River landscape.

1.2.2 Scope and Context of Recovery Planning

The LCFRB is taking a collaborative approach to meet the needs for recovery and subbasin planning. This approach integrates ESA recovery planning, NPCC subbasin planning, State salmon recovery strategies, and state watershed planning into a single coordinated regional planning process. Through this process, by June 2004 eleven distinct subbasin plans will be rolled into a single comprehensive recovery/subbasin plan. The plan will address the recovery of four ESA-listed species (chinook, steelhead, chum, and bull trout) within the context of the 4Hs: habitat, hydroelectric, harvest, and hatchery impacts. Beyond this, the plan also will address selected anadromous and resident fish and wildlife of interest under the subbasin planning process. These additional focal species include coastal cutthroat trout, coho salmon, sturgeon, smelt, northern pikeminnow, American shad, warm water fish, Pacific lamprey, Caspian tern, Columbian white-tailed deer, dusky Canada goose, western pond turtle, sandhill crane, and selected neo-tropical birds. This approach provides significant benefits, including:

- ensuring consistency and compatibility of goals, objectives, strategies, priorities, and actions;
- eliminating redundancy in the collection and analysis of data; and
- establishing the framework for a partnership of federal, state, tribal and local governments under which agencies can effectively and efficiently coordinate planning and implement efforts for restoration of listed salmonids, as well as the enhancement of other focal fish and wildlife species.

In the end, the plan will provide common goals and a coordinated course of action that is scientifically sound, acceptable to the public, and economically sustainable. Protection, restoration, and enhancement actions will be prioritized to provide maximum benefit and ensure the efficient use of resources. The plan will focus on outcomes and allow implementing agencies and local governments the flexibility to craft innovative, yet scientifically sound, approaches that best fit local conditions and values.

1.2.3 The Planning Area

The 5,700 square mile planning area encompasses the entire Lower Columbia Salmon Recovery Region excepting the White Salmon basin, omitted at the request of Klickitat County. The planning area includes the Washington portion of the mainstem and estuary of the lower Columbia River as well as 18 major and a number of lesser tributary basins. These include the Chinook, Grays, Skamokawa, Elochoman, Mill, Abernathy, Germany, Cowlitz, Coweeman, Kalama, Lewis, Lake, Washougal, Duncan, Hardy, Hamilton, Wind, and Little White Salmon rivers, as well as the Columbia River Estuary, and their tributaries. In all, the tributaries total more than 1,700 river miles.

Approximately 464,000 people live in the planning area, which includes all of Clark, Cowlitz, Skamania, and Wahkiakum Counties and portions of Lewis and Pacific Counties. Thirteen cities are located in the planning area, as well as numerous unincorporated communities.

Several tribes have lands of interest in the planning area. Lands of interest to the Yakama Nation include areas in Cowlitz, Lewis, Clark, and Skamania Counties. The Cowlitz and Chinook tribes also have lands of interest within the lower Columbia region. Within these areas, reserved fishing and hunting rights are exercised, natural resources are co-managed, and tribal trust lands are inhabited.

1.2.4 ESA Salmon Recovery Planning

Chinook and chum salmon, steelhead, and bull trout in the lower Columbia have been listed as threatened under the ESA. As the listing agency for anadromous salmonids under the ESA, NOAA Fisheries is responsible for developing plans to recover chinook and chum salmon and steelhead. The USFWS is responsible for developing a bull trout recovery plan. An ESA recovery plan must include:

- site-specific actions necessary for recovery,
- measurable criteria (goals) which, when met, would result in removing the species from ESA protection (delisting), and
- estimates of the time and cost to carry out recovery actions.

There are several agencies and entities with direct responsibility in the recovery planning process.

1.2.4.1 NOAA Fisheries Recovery Planning

The basic unit used by NOAA Fisheries for listing and delisting salmonid species is the Evolutionarily Significant Unit (Waples 1991). An ESU is a distinctive group of Pacific salmon, steelhead, or sea-run cutthroat trout populations uniquely adapted to a particular area or environment and cannot be replaced. In the lower Columbia region of Oregon and Washington, there are currently three listed ESUs:

- Columbia River chum salmon—listed as threatened in 1999
- Lower Columbia steelhead—listed as threatened in 1998
- Lower Columbia chinook salmon—listed as threatened in 1999

A recovery domain is a collection of geographically proximate ESUs. The Willamette/Lower Columbia Domain includes the three lower Columbia ESUs and two Willamette ESUs. NOAA Fisheries has initiated efforts to develop a single recovery plan for the Willamette/Lower Columbia Recovery Domain. NOAA Fisheries desires to develop this domain recovery plan through a collaborative effort involving federal and state agencies, tribes, local governments, and the public. Under the proposed approach, the recovery plans being developed in Washington and Oregon for the lower Columbia ESUs will be combined with the recovery plans being developed for the Willamette ESUs to create a single domain plan. The Willamette/Lower Columbia ESA Executive Committee is coordinating the overall domain planning effort and will ensure 1) that the separate Oregon and Washington planning efforts are consistent and compatible, and 2) that they will result in a domain plan that meets ESA requirements. The committee comprises policy-level representatives from federal agencies, tribes, Washington and Oregon agencies, and local governments.

The LCFRB is coordinating the Washington recovery planning efforts for the lower Columbia region. The LCFRB recovery/subbasin plan will eventually be incorporated in the Willamette/Lower Columbia Domain Plan. It is expected that NOAA Fisheries will approve the LCFRB plan as the ESA recovery plan for those areas of the three listed lower Columbia ESUs in Washington, even if Oregon has not completed its plan for the Oregon portions of the ESUs.

In addition, NOAA Fisheries has established the Willamette/Lower Columbia Technical Recovery Team (TRT) to provide technical advice on recovery-related issues. The TRT comprises scientists from NOAA Fisheries, state agencies, academic institutions, and private consulting firms. The TRT has focused on developing guidelines for viability criteria for the listed species. These guidelines will assist NOAA Fisheries in identifying criteria for delisting species. These criteria will describe the conditions under which a listed species or ESU is no longer in danger of extinction (endangered) or likely to become so in the foreseeable future (threatened).

1.2.4.2 US Fish and Wildlife Recovery Planning

Bull trout was listed as threatened in the coterminous United States in November 1999. In December 1999, USFWS completed a draft bull trout recovery plan for five western states—Washington, Oregon, Idaho, Montana, and Nevada. The plan is broken down into four Distinct Population Segments, each of which is broken into recovery units and mini-recovery plans have been developed for each recovery unit. Much of the USFWS Lower Columbia Recovery Unit falls within the LCFRB planning area. The objectives of the draft USFWS bull trout plan are to:

- maintain the current distribution of bull trout within core areas and to restore distribution where possible,
- maintain stable or increasing trends in bull trout abundance,
- restore and maintain suitable habitat conditions for all bull trout life history stages, and
- conserve genetic diversity and provide opportunity for genetic exchange.

The recovery planning process being coordinated by the LCFRB will build on the provisions of the USFWS Lower Columbia Recovery Unit plan to refine bull trout recovery strategies for the lower Columbia and will ensure that bull trout recovery efforts are woven into the broader salmonid recovery strategies and actions for the lower Columbia.

1.2.4.3 NPCC Subbasin Planning

The Northwest Power and Conservation Council (NPCC) was created by Congress in 1980 to give Washington, Oregon, Idaho, and Montana a voice in how the region plans for its energy needs, while at the same time mitigating the effects of the federal Columbia River Power System on fish and wildlife resources.[1] To this end, the Council has developed the Columbia Fish and Wildlife Program. The program sets forth goals and strategies for the protection and enhancement of fish and wildlife resources. The Council uses the Program to solicit and evaluate proposals for on-the-ground projects and research. Priority proposals are forwarded to the Bonneville Power Administration (BPA) for funding.

The Council has initiated efforts to update its Columbia Basin Fish and Wildlife Program. A key element in this effort is the development of individual plans for the 62 subbasins within the Columbia basin. Eight of these subbasins fall totally within the lower Columbia region in Washington. Three others (Columbia Estuary, Columbia Lower, and Columbia Gorge) are shared with the state of Oregon. Subbasin plans:

- identify the goals for fish, wildlife, and habitat;
- define objectives that measure progress toward the those goals;
- establish strategies to achieve the objectives; and
- incorporate and build upon existing fish and wildlife information and activities.

[1] The Northwest Power and Conservation Council (NPCC) was formerly referred to as the Northwest Power Planning Council.

Completed subbasin plans will be adopted as part of the Council's Columbia River Fish and Wildlife Program and will help direct BPA funding of projects that protect, mitigate and enhance fish and wildlife that have been adversely impacted by the development and operation of the Columbia River hydropower system. The Council's effort is also linked to and accommodates the needs of other programs in the basin that affect fish and wildlife. Along with the NOAA Fisheries and the USFWS, the Council and BPA also intend to use the adopted subbasin plans to help meet the requirements of the 2000 Federal Columbia Power System Biological Opinion. The NOAA Fisheries and USFWS intend to use subbasin plans throughout the Columbia River basin as building blocks for recovery planning for threatened and endangered species. In addition, the Environmental Protection Agency (EPA), in cooperation with the states and tribes, intends to use subbasin and watershed planning initiatives to address concerns under the Clean Water Act (CWA).

Similar to the recovery planning process, the first phase of the NPCC's subbasin planning process calls for development of subbasin assessments. These assessments are intended to form the scientific and technical foundations for developing a management plan, which includes subbasin vision, objectives, and strategies. These plans will also include a research, monitoring, and evaluation plan and considerations to address the ESA and CWA. However, selection of fish and wildlife species in the subbasin planning process goes beyond that required for recovery of listed species. The planners identify focal species that have special ecological, cultural or legal status and use these species to evaluate the health of the ecosystem and the effectiveness of management actions. Subbasin plans are developed locally and in collaboration with fish and wildlife managers, local governments, interest groups and stakeholders, and other state and federal land and water resources managers.

Given the strong linkage between ESA recovery and NPCC subbasin planning, the LCFRB is working to integrate the two efforts into a single planning process. This will help to ensure consistency in goals, strategies, actions, and priorities and avoid potentially costly duplication of efforts. The NPCC has endorsed the concept and has contracted with the LCFRB to prepare a recovery/subbasin plan for the 11 NPCC lower Columbia subbasins that encompass the same geographical area of concern for ESA salmon and trout recovery. The LCFRB is working through the Lower Columbia River Estuary Partnership (LCREP) to prepare plans for the two subbasins that fall in both Oregon and Washington (Columbia Estuary and Columbia Lower).

1.2.4.4 Washington State Salmon Recovery Strategy

In September 1999, Washington published its statewide strategy to recover salmon. The goal of the state strategy is to return salmon and steelhead to healthy, harvestable levels. It calls for:

- collaborating on an incentive-based approach to salmon recovery, coupled with increased enforcement of environmental laws;
- identifying what actions must be taken immediately to prevent extinction;
- identifying clear performance measures for determining if restoration efforts are getting results; and
- establishing an action plan that can be put into place if restoration performance goals are not met on schedule.

The strategy strongly endorses cooperative regional recovery efforts. The Lower Columbia region is one of seven salmon recovery regions identified by the state. Regional recovery organizations have been established for five regions, and the LCFRB serves as the regional organization for the lower Columbia. The Washington Department of Fish and Wildlife (WDFW) has developed a recovery plan template in consultation with NOAA Fisheries and the USFWS and is participating and providing technical support to the regional organizations. The Governor's Salmon Recovery Office is coordinating state agency participation in recovery planning efforts and is helping to address recovery-related policy issues. The Salmon Recovery Funding Board has provided funding to the regional organizations to support recovery planning efforts.

1.2.4.5 Washington Watershed Management Planning

The state Watershed Management Act (RCW 90.82) provides local communities the opportunity to plan for the future use of their water resources in consultation with state agencies. To facilitate this planning, the state has been divided into Water Resource Inventory Areas (WRIAs). There are five WRIAs in the lower Columbia. Watershed planning efforts are underway in all five areas. The LCFRB coordinates watershed planning in four of the five lower Columbia WRIAs and is an active participant in planning for the fifth WRIA. Watershed plans for these WRIAs will address issues associated with:

- water quantity, including the availability and current use of water and actions needed to meet future needs for fish and people;
- water quality, including current water quality problems, priorities for addressing these problems, and water quality monitoring;
- stream flows, including the adequacy of existing flows for fish and other in-stream uses and measures to protect or enhance stream flows; and
- habitat, including the current condition of fish habitat and measures to protect or enhance habitat to support salmon recovery efforts.

Given the integral relationship between watershed management and salmon recovery, the LCFRB has integrated these two planning initiatives. Water quantity and quality and stream flow studies and data collected by the watershed planning initiatives will be incorporated in the regional recovery plan. Habitat data collected by the recovery planning effort will be shared with the watershed planning effort. Policies, strategies, actions, and priorities will be coordinated to ensure that they are compatible and complement each other.

1.2.5 Overview of Decision-Making Process

While the final recovery/subbasin plan will be a product of the LCFRB, it must meet the needs of, and be implemented through, the actions of multiple entities. For these reasons, the Lower Columbia Recovery Planning Steering Committee (RPSC) was convened to facilitate and oversee the plan's development. The committee's role is described below, but its basic functions include providing overall direction and oversight of the recovery planning initiative. Adopting the final plan will require the consensus of the organizations represented on the committee, as well as the approval of the LCFRB.

Public comments have been gathered during the planning process. The LCFRB coordinated and conducted public information and outreach efforts in concert with the participating agencies. Comments received during these efforts were used to develop this final draft plan.

The LCFRB will submit the final draft plan to the state, NOAA Fisheries, USFWS, and the NPCC for review and adoption. As part of the recovery planning process and coordinated by the LCFRB, recovery goals will be established in consultation with NOAA Fisheries, USFWS, and WDFW. The NPCC will conduct its own internal and public reviews before adopting the plan into its program.

1.2.6 Participants in the Planning Process

This integrated planning effort is built on effective working relationships among the participating governments, agencies, and organizations. These relationships will ensure that the recovery/subbasin plan meets the needs of the different entities and is implemented through their coordinated actions. Representatives from various agencies and organizations, tribes, private property owners, and other stakeholders are participating in the process through involvement on the LCFRB, Recovery Planning Steering Committee, planning working groups, public outreach, and other coordinated efforts.

The LCFRB leads the recovery and subbasin planning efforts and has three primary recovery planning functions. These include:

- coordinating, facilitating and administering the recovery/subbasin planning initiative;
- overseeing the development of the plan's habitat provisions; and
- approval of the final plan before its submission to NOAA Fisheries, USFWS, NPCC, and the state

The LCFRB comprises representatives from the state legislature, city and county governments, the Cowlitz Tribe, private property owners, hydro project operators, the environmental community and concerned citizens. The LCFRB is committed to finding solutions that restore fish and provide for the needs of the citizens of the region. Adoption of the final plan will require consensus of all Board members.

1.2.6.1 Recovery Planning Steering Committee

The Recovery Planning Steering Committee (RPSC) was created by the LCFRB to facilitate and oversee the plan's development. The committee is responsible for the overall direction and oversight of the recovery planning initiative. RPSC members represent the interests of their organizations and ensure that decisions are properly communicated and supported within their organizations. The committee makes decisions by consensus. Specific committee tasks include:

- establishing the goals and objectives of the recovery/subbasin plan;
- determining the scope and content of the recovery/subbasin plan to ensure that it meets the plan's goals and objectives;
- adopting and maintaining a workplan and schedule for the planning initiative;
- monitoring progress of planning efforts and adjusting scope and direction as necessary to achieve goals;
- approving a funding/resource strategy for the planning initiative;
- adopting and overseeing implementation of a public education and outreach program;
- addressing and resolving policy issues that arise during plan development;
- coordinating planning efforts with other planning initiatives in the region, such as the efforts of the ESA Executive Committee and the NOAA Fisheries TRT; and
- reviewing, commenting on, and concurring with plan elements as they are prepared and with the final draft plan, its goals, strategic priorities, and implementing actions prior to its submission to the state, NOAA Fisheries, USFWS, and the NPCC.

Current members of the RPSC include local governments and citizen representatives from the LCFRB, NOAA Fisheries, USFWS, NPCC, LCREP, WDFW, Governor's Salmon Recovery Office, Washington Department of Agriculture, Washington Department of Ecology, the US Forest Service (USFS), the Cowlitz Tribe, the Yakama Nation, and the Chinook Tribe. Adoption of the final plan will require the consensus of all the organizations represented on the Committee and will be sought before final approval by the LCFRB.

1.2.6.2 Work Groups

The RPSC is creating work groups to address specific issues and prepare recommendations or documents for RPSC review during the planning process. The work groups are used to secure the expertise or knowledge needed to complete the recovery/subbasin plan successfully as well as to broaden participation in the planning process. The composition of a work group depends on the issues to be addressed or the tasks at hand. Members are selected based on their knowledge or expertise. Work groups organized thus far include the following.

- Fish Work Group—provides technical assistance and advice to the RPSC regarding the development of plan elements dealing with recovery goals and biological objectives and the status, life history and environmental needs of salmonids.
- Factors Limiting Recovery Work Group—provides technical assistance and advice to the RPSC for developing plan elements dealing with factors limiting the recovery of salmonids and watershed assessment activities.
- Programs Work Group—provides assistance and advice to the RPSC for developing a plan element that identifies, inventories, and characterizes programs that affect fish resources and their recovery.

1.2.6.3 Partnerships

Other key partnerships have been formed with the NOAA Fisheries, USFWS, NPCC, the State of Washington, and with Native American tribes. NOAA Fisheries and USFWS have federal statutory responsibility for recovery planning and both agencies sit on the RPSC to ensure that planning work will result in a product meeting their requirements. The NPCC also is represented on the RPSC to ensure that its subbasin planning requirements are satisfied. The Governor's Salmon Recovery Office participates to coordinate state support for the recovery planning process and to help facilitate habitat elements of the plan as provided by state statute. Representatives of several Native American tribes are members of the LCFRB and RPSC. Tribal governments have specific legal rights and responsibilities related to the use, management, and stewardship of fish, wildlife, and cultural resources. Under treaties signed in 1855, the tribes reserve the right to fish, game, berries, roots, and associated plants and animals necessary to maintain their culture and religion. Maintaining these diverse resources requires healthy, interconnected, naturally functioning ecosystems.

Local stakeholders also are key participants in the planning process. Through their involvement on the LCFRB, the RPSC, technical work groups and other partnerships, local representatives play an active role during the recovery/subbasin plan process and in developing specific approaches that will improve fish status and achieve recovery goals. During Phase I, local representatives provided data and helped conduct analyses and define specific factors responsible for fish declines. In Phase II, they will help identify and evaluate scenarios, strategies, and actions consistent with recovery. Information gained during the recovery/subbasin planning process will help them identify the most effective and economically sound measures for fish recovery in their watersheds. They ultimately will be responsible for weighing this information with local needs to craft and implement their own innovative but scientifically sound approaches to best fit local conditions and values.

Volume I, Chapter 2
Species Overview

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2.0 Species Overview

In this section, we summarize life history and population characteristics of chinook, coho, and chum salmon, as well as steelhead, bull trout and cutthroat trout, in Washington tributaries to the lower Columbia River. We review the life history cycle and requirements of these species from gravel to gravel, and describe their distribution and genetic diversity within lower Washington tributaries. We also identify trends in abundance and factors that led to their decline in the lower Columbia watershed.

2.1 Chinook Salmon *(Oncorhynchus tshawytscha)*

Chinook salmon *(Oncorhynchus tshawytscha),* also commonly referred to as king, spring, tyee, or quinnat salmon, is the largest of the Pacific salmon (Netboy 1958). The species distribution historically ranged from the Ventura River in California to Point Hope, Alaska in North America, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (Healey 1991). Other chinook salmon have been reported in the Mackenzie River area of northern Canada (McPhail and Lindsey 1970). Of the Pacific salmon, chinook salmon exhibit the most diverse and complex life history strategies.

Chinook salmon generally follow one of two freshwater cycles: stream or ocean type. After emerging from the gravel, ocean-type chinook salmon migrate to the ocean within their first year (Figure 2-1). Stream-type chinook salmon reside in fresh water for a year or more before migrating to the ocean (Figure 2-2). These two types of chinook salmon have different life history traits, geographic distribution, and genetic characteristics. Ocean- type behavior and life history strategy is regarded as a response to limited carrying capacity of the freshwater environment of less productive streams, such as smaller watersheds, glacially scoured rivers, and systems with periodic flooding. Ocean-type chinook salmon occur primarily in coastal waters south of the 55th parallel, in Puget Sound, in the lower reaches of the Fraser and Columbia Rivers as well as California’s Central Valley (Gilbert 1913, Rich 1920, Healey 1983). Stream-type chinook emigrate as juveniles during their second, or more rarely, third year. This extended freshwater residency is characteristic of chinook that inhabit more productive watersheds where conditions are more stable, and water flows are not subject to dramatic changes. Since stream-type Chinook enter marine waters at a larger size, they are not as dependent on estuaries as ocean-type chinook for juvenile growth. In addition, stream-type chinook make more use of the open ocean environment far from coastal waters. Stream-type chinook populations are generally more predominant in waters north of the 55th parallel and in headwaters of the Fraser and Columbia rivers (Healey 1991).

Chinook in the lower Columbia River are further classified as fall or spring chinook depending on adult migration timing. Fall chinook dominate in the Washington tributaries of the lower Columbia River, though several tributaries also support spring chinook. Today, the once abundant natural runs of fall and spring chinook have been largely replaced by hatchery production. Although large chinook runs continue to return to many of their natal streams, they are mostly sustained by hatchery production with few sustained, naturally reproducing, native populations.

2.1.1 Life History and Requirements

Like other Pacific salmon, the life history of chinook involves spawning, incubation, and emergence in freshwater, migration to the ocean, and subsequent initiation of maturation and return to fresh water. Within this life history cycle, there may be a high degree of variability in response to freshwater environmental conditions and genetic imprinting.

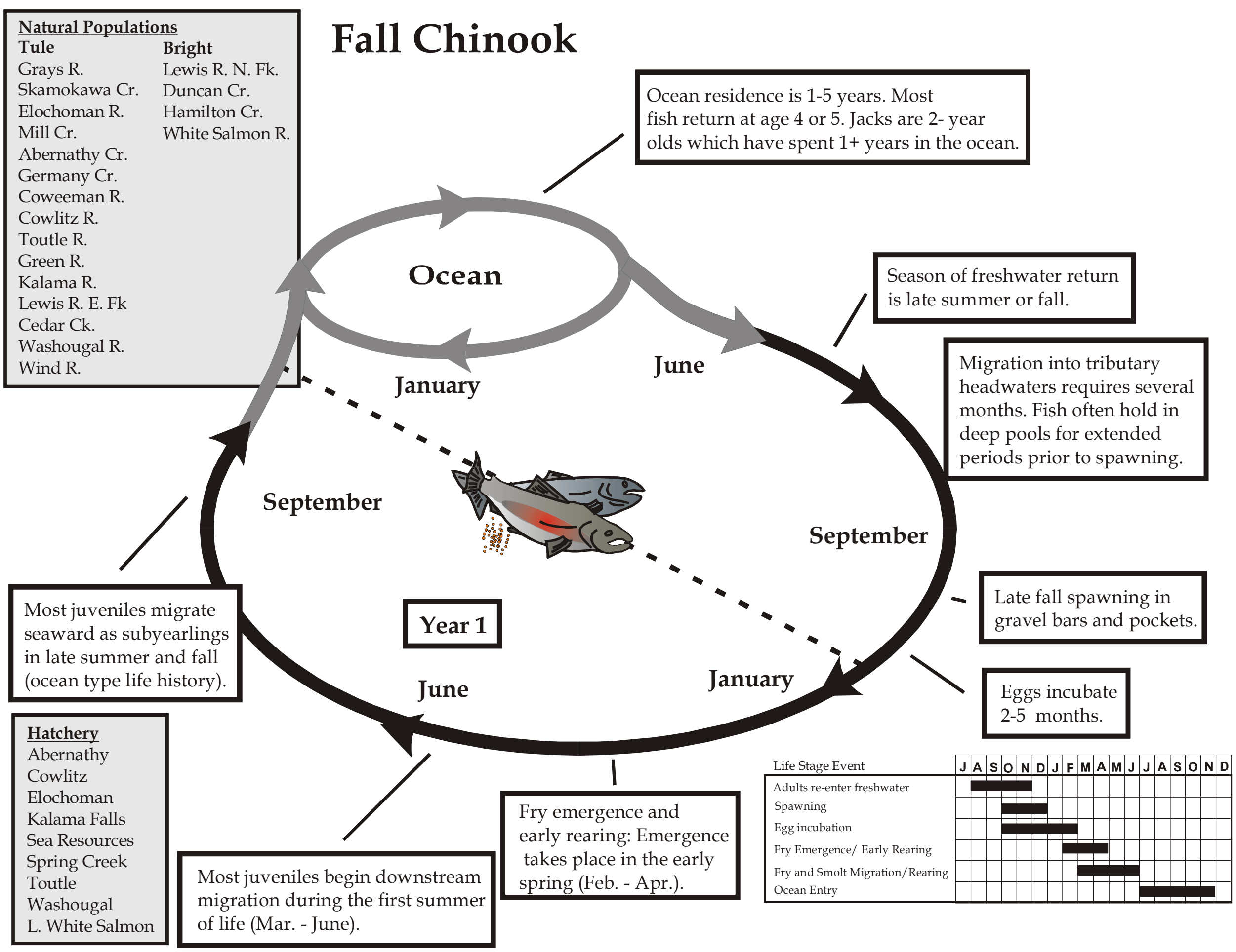

Figure 2-1. Washington lower Columbia fall chinook life cycle.

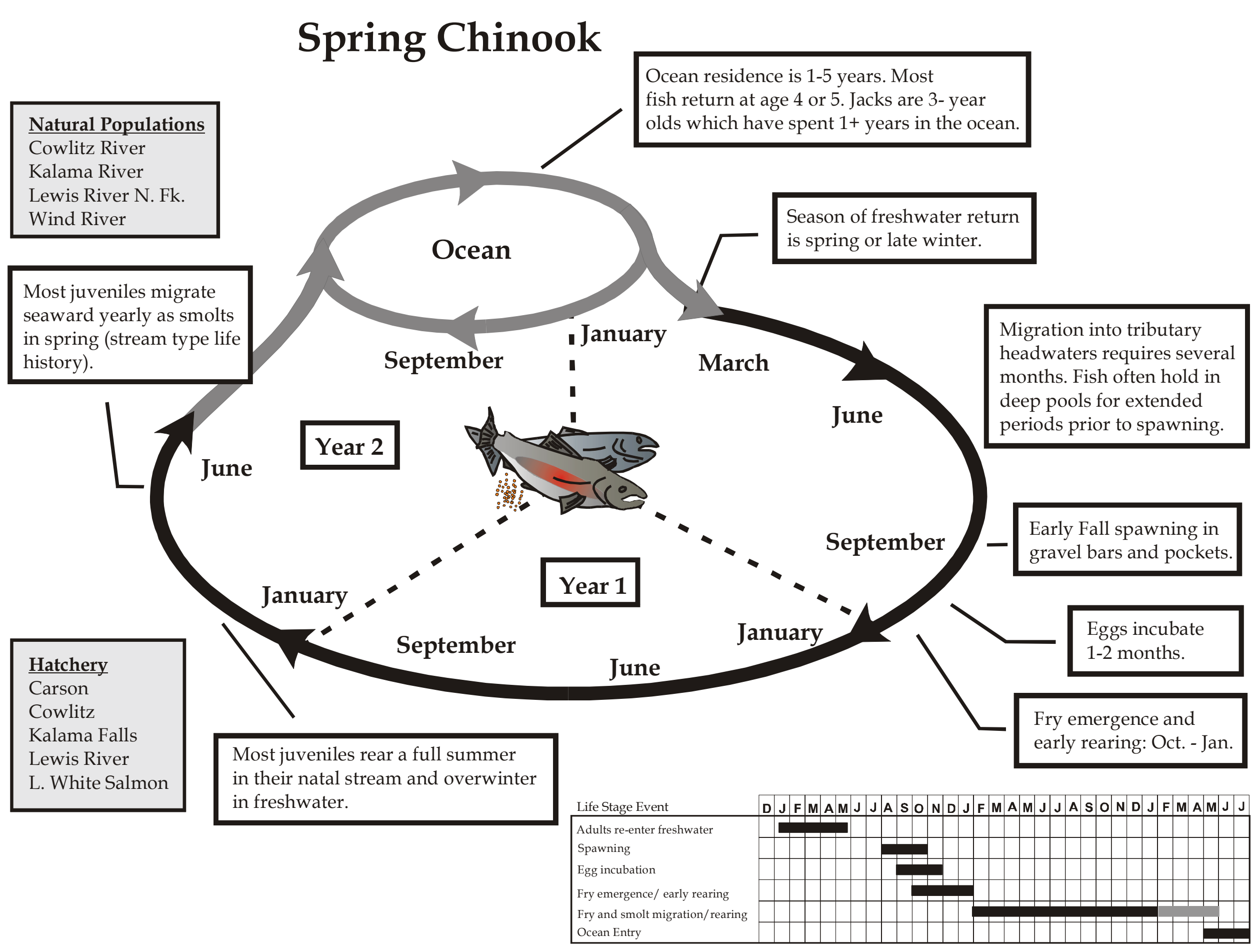

Figure 2-2. Washington lower Columbia spring chinook life cycle.

2.1.1.1 Upstream Migration Timing

The entry timing of individual runs into freshwater has evolved over thousands of years. Adult migrations were historically synchronized to streamflow characteristics and water temperatures in a particular river system. Freshwater entry and spawning timing are generally related to local temperature and water flow regimes (Miller and Brannon 1982). Migration timing was cued to the local thermal regime so that adults would reach spawning sites and deposit eggs in time to ensure that fry emerged during the following spring at a time when river or estuary productivity was sufficient for juvenile survival and growth. During lower flows, waterfalls, sandbars and other barriers can restrict migration. After entering freshwater, most fall salmon have a limited time to migrate and spawn; in some cases, as little as 2 to 3 weeks and delays can result in pre-spawning mortality or spawning in a sub-optimum location.

Three different major runs of chinook salmon presently return to Washington tributaries of the lower Columbia River. Adult spring chinook return to the Columbia River at 4 to 5 years of age. They enter the lower Columbia River from March through June, well in advance of spawning in August and September (Figure 2-2). Spring chinook typically spawn near headwater areas where higher gradient habitat exists. Historically, fish migrations were synchronized with periods of high rainfall or snowmelt to provide access to upper reaches of most tributaries where fish would hold until spawning (Fulton 1968, Olsen et al. 1992, WDF et al. 1993). Since spring chinook enter freshwater well before the time of spawning, survival until the spawning period is primarily a function of body fat reserves at the time of freshwater entry.

Fall chinook begin returning to the lower Columbia River in early to mid-August. One race of lower Columbia River chinook salmon are often called tules (pronounced "toolies") and are distinguished by their dark skin coloration, and advanced state of maturation at the time of freshwater entry. Tule fall chinook salmon populations may have historically spawned from the mouth of the Columbia River to the Klickitat River. Tule fall chinook return to the Columbia River at 3 to 4 years of age, although 5-year olds are common in some populations. They enter freshwater from August to September and spawning generally occurs from late September to November, with peak spawning activity in mid-October. Fall chinook spawn in the Grays River from late September to mid-November, but do not spawn until late October or November in the Washougal River. A later returning component of the fall chinook salmon run exists in the Lewis River (WDF et al. 1993, Kostow 1995, Marshall et al. 1995).

The other fall race, bright fall chinook, return to the Lewis River and several Bonneville area tributaries and the mainstem Columbia River. Their dominant age class varies by population and brood year, but is typically age 4. They enter the Columbia River in August to October, but spawning occurs in November to January, with peak spawning in mid-November. Because of the longer time interval between freshwater entry and spawning, these fall chinook salmon are less mature at freshwater entry than tule fall chinook salmon and are therefore commonly termed lower river 'brights' (Marshall et al. 1995) or lower river wild. A naturally produced, bright fall chinook run also exists in the area immediately downstream of Bonneville Dam, and in the Wind River basin. These fish likely originated from Bonneville and Little White Salmon bright fall chinook hatchery programs and are not included in the Lower Columbia chinook ESU.

2.1.1.2 Spawning

Successful spawning depends on sufficient clean gravel of the right size, in addition to the constant need of adequate flows and water quality. The driving force in redd site selection appears to be the presence of good subgravel flow; this need is likely greater in chinook than the other species of Pacific salmon. Chinook salmon have the largest eggs and therefore the smallest surface-to-volume ratio of Pacific salmon. As a result, their eggs are likely more sensitive to reduced dissolved oxygen levels and require a higher rate of irrigation.

Describing typical chinook spawning habitat is problematic as research has documented a broad range of water depth and velocity characteristics. Chinook have been documented spawning in streams as small as 7-10 ft (2-3 m) wide and only a few centimeters deep and as large as mainstem large rivers such as the Columbia and Sacramento. In addition, velocity measurements at redd sites have ranged from 0.33 ft/sec to 5 ft/sec (10 cm/sec to 150 cm/sec). There is no agreement as to whether depth and velocity characteristics of redd site selection differ between stream- and ocean-type chinook.

The reported depths at which chinook eggs are buried in the gravel also varied among researchers. Briggs (1953) reported egg depths of 7.9-14 in (20-36 cm) (average 11 in [28 cm]) for two small California streams. Vronskiy (1972) observed eggs buried from 4 to 31 in (10-80 cm) in the Kamchatka River, although few eggs were buried below 19.7 in (50 cm). The depth at which eggs are buried at a particular spawning site is partly dependent on water flow. Depth of redd excavation is negatively correlated with water velocity in the spawning area (Vronskiy 1972, Neilson and Banford 1983). Presumably, the higher mound in the tailspill of redds in low velocity areas improves subgravel irrigation of the eggs.

Although the measurements are not comparable among studies, the size of redds also appears to vary considerably among chinook populations. Chapman et al. (1986) measured redd size range of 22-482 ft^2 (2.1-44.8 m^2) for chinook spawning in the Hanford reach of the Columbia River.

Chinook salmon fecundity also varies within and among populations. Fecundity is correlated with size. However, size explained only 50% or less of the variation in fecundity between individuals within a population. There seems to be an unresolved trade-off between egg size and egg number; consequently, egg size varies more between chinook individuals than is usual for fishes. Latitudinal differences in fecundity may partly reflect a racial difference between stream- and ocean-type chinook rather than a latitudinal cline. For example, high fecundity populations near the northern limit of the chinook's range are all stream-type chinook while low fecundity populations in the south are mainly ocean-type chinook. However, if the data are segregated into stream- and ocean-type life histories, there is still a latitudinal cline in fecundity within ocean-type chinook. In the Columbia River, where fecundity data are available for both stream- and ocean-type chinook, stream-type chinook have a greater fecundity than ocean-type, although the difference is not statistically significant (Galbreath and Ridenhour 1966, Healey and Heard 1984).

2.1.1.3 Incubation and Emergence

Chinook eggs incubate throughout the autumn and winter months. In the lower Columbia River, spring chinook fry emerge from the gravel from November through March; peak emergence time is likely December and January. Fall chinook fry generally emerge from the gravel in April, depending on the time of egg deposition and incubation water temperature.

As with other salmonids, water temperature controls incubation time and affects survival. When incubation temperature is held constant, the upper and lower temperature limits for chinook salmon at 50% pre-hatching mortality is 61°F (16°C) and 36-37°F (2.5-3°C), respectively (Alderdice and Velsen 1978). The time to 50% hatch ranged from 159 days at 37°F (3°C) to 32 days at 61°F (16°C). Development rate and survival were better at low temperatures when water temperature varied with ambient temperature compared to when water temperature was constantly low. Presumably, the better performance reflected greater low water temperature tolerance after initial egg development (Alderdice and Velsen 1978). A simple thermal sum model appears to be adequate for predicting time to hatching (development rate = $468.7/T$, where T is the average temperature in Celsius during incubation). It is likely that lower Columbia chinook spawning begins in some locations where water temperatures approach the upper thermal limit (61°F [16 °C]), however, time of exposure to this temperature is likely brief as temperatures are typically dropping during the time of chinook spawning.

During incubation, clean, well-oxygenated water flow is critical. Eggs often do not survive in gravel choked with sediment (Shaw and Maga 1943, Wickett 1954, Shelton and Pollock 1966). Shaw and Maga (1943) observed that siltation resulted in the greatest mortality when it occurred early in the incubation period. In experimental stream channels, research has established a relationship between egg survival and both percolation rate and dissolved oxygen concentration: egg mortality increases with decreasing percolation rate and increases rapidly when dissolved oxygen concentration drops (Shelton 1955; Gangmark and Bakkala 1960).

Floods can have their greatest impact to salmon populations during incubation, as they can scour salmon eggs from the gravel or deposit sediment over spawning gravels (Wade 2002). Flooding has been documented as an important cause of high mortality of chinook eggs (Gangmark and Broad 1955, Gangmark and Bakkala 1960).

Estimates of egg to emergent fry survival are problematic because some fish migrate downstream as fry whereas others rear for a variable length of time in the river before migrating downstream. In Fall Creek, California, Wales and Coots (1954) and Coots (1957) found a 68-93% mortality from egg deposition to the emergent fry stage; average mortality was 85% and the high mortality estimates (93%) were associated with floods. Lower egg to fry mortality (40%) was observed in a controlled channel in Mill Creek, California (Gangmark and Bakkala 1960). Gravel conditions affect success of emergence. Shelton (1955) found that only 13% of hatched alevins emerged from fine gravel while 80-90% emergence was observed in coarse gravels. Success of emergence from fine gravels was influenced by egg deposition depth; eggs near the surface realized a greater success of emergence.

Dewatering can occur in regulated rivers where discharge is varied to satisfy domestic or industrial water needs but also occurs in natural systems. Becker et al. (1982, 1983) investigated the effects of dewatering on four different stages of chinook egg development based on accumulated thermal units. Alevins were most sensitive to periodic short-term and single prolonged dewatering; alevin survival was less than 4% in periodic dewaterings of 1 hour or a single dewatering of 6 hours. Cleavage eggs and embryos were the least sensitive to dewatering; embryos apparently suffered no ill effects from daily dewaterings of up to 22 hours over a 20-day period. Because dewatered eggs and embryos remained damp during dewatering, they probably suffered no shortage of oxygen, although metabolic waste elimination may have been a problem.

2.1.1.4 Freshwater Rearing

Fall chinook comprise most of the chinook populations in the lower Columbia River and they exhibit similar life history strategies to those observed in other fall chinook populations. Fry emergence is generally around April, depending on the time of egg deposition and water temperature. Fry spend 1–4 months in fresh water and emigrate in the summer as subyearlings. A few fall chinook remain in fresh water until their second spring and emigrate as yearlings (Chapman et al. 1994, Waknitz et al. 1995). Although the timing of emergence and downstream migration differs among lower Columbia fall chinook, there appears to be little divergence from the strategies of spring emergence and summer emigration. The earliest timing appears to be in the Wind River basin where fry emerge from January to March and emigrate in the spring. The early emigration timing for Wind River fall chinook may be a function of distance to the estuary, as the Wind River is further from the Columbia River estuary than most other lower Columbia basins. Early and late emergence and late emigration timing occurs in the Lewis River basin; the timing on the Lewis is a function of both late and extended spawn timing of the Lewis bright fall chinook stock, and warmer winter water temperatures for incubation then most basins. Consequently, fry emerge from early spring to early summer and seaward emigration occurs in the early to late summer.

Lower Columbia spring chinook exhibit juvenile life history characteristics similar to those observed in other spring chinook populations. They have more of a tendency to spend one full year in fresh water and emigrate to sea in their second spring than do fall chinook. However, some stocks migrate downstream from their natal tributaries in the fall and early winter into larger rivers, including the mainstem Columbia River, where they are believed to over-winter before emigration the next spring as yearling smolts.

Although there is some variation in timing, all populations of chinook appear to display similar migratory behavior. At the time of emergence, there is an extensive downstream dispersal of fry, although some fry are able to take up residence at the spawning site. For populations that spawn close to tidewater, this downstream dispersal carries fry to estuarine nursery areas, whereas in other locations it serves to distribute the fry among suitable freshwater nursery areas (Healey 1991). After spring and fall chinook fry leave their gravel nests, they generally move to suitable rearing habitat within side sloughs, side channels, spring-fed seep areas and along the outer edges of the stream. These quiet-water side margin and off-channel slough areas are vital for early juvenile habitat (Wade 2002). The presence of woody debris and overhead cover aid in food and nutrient inputs, and provide protection from predators primarily for the first 2 months of freshwater residence. As chinook fry grow, some gradually move away from the quiet shallow areas to rear in deeper, faster areas of the stream (Lister and Walker 1966, Chapman and Bjornn 1969). This movement to faster water often coincides with summer low flows that can constrain salmonid production.

Later in the spring, there appears to be a second dispersal that carries some populations to the sea or simply redistributes fry within the river system, presumably to suitable summer rearing areas. For those populations that spend a full year in fresh water, there is a third late fall redistribution to suitable overwintering habitat, usually from the tributaries to the river mainstem (Healey 1991). On the other hand, some overwintering juveniles need habitat to sustain their growth and protect them from predators and winter flows. Wetlands, off-channel habitat, undercut banks, rootwads, and pools with overhead cover are important habitat components during this time. During the late spring and fall distributions, fry tend to shift to deeper water and move seaward. The redistributions may punctuate developmental stages as well as achieve more efficient use of freshwater nursery habitat. Fry redistributions may have adaptive value by shortening the length of spring migration for yearling smolts, especially for headwater spawning populations in larger rivers (Healey 1991).

Survival rates from fry to subyearling migrant or fry to yearling migrant are mostly unknown, except for data collected on the Sacramento River by the USFWS (unpublished). Based on the ocean returns of chinook from the same brood year tagged as fry and smolts, survival from fry to smolt ranged from 3 - 34% for the 1980–82 year classes. These survival rates are similar to those for other Pacific salmon (Foerster and Ricker 1941, Hunter 1959, Parker 1965) so it is reasonable to assume that chinook in other river systems have similar survival rates. Predators are usually implicated as the principal agent of mortality among fry and fingerling of chinook and other species; heavy losses to predators have been documented (Foerster and Ricker 1941, Hunter 1959). However, on the Elochoman River, Patten (1971) observed 1-4% predation by sculpins of chinook released from the Elokomin Hatchery during 1962 and 1963. In this instance, the release of chinook fingerling occurred during a single night in 1962 and over three nights in 1963; thus, chinook were only available to predators for a brief period.

2.1.1.5 Juvenile Migration

The timing of parr-to-smolt transition seems to depend on a number of environmental and genetic traits that maximize individual survival (Myers et al. 1998). Differences in the timing of smoltification and emigration to the ocean may be affected by distance of migration to the marine environment, stream stability, stream flow and temperature regimes, stream and estuary productivity, and general weather regimes (Myers et al. 1998). Such environmental factors may be the reason why stream-type chinook—which usually spawn further inland than ocean-type chinook —appear unable to smolt as subyearlings. Ocean-type fish have been found to exhibit a faster growth rate relative to stream-type fish (Gilbert 1913, Carl and Healey 1984, Cheng et al. 1987).

Ocean-type juveniles enter salt water following one of three distinct strategies. Some fry migrate to the ocean soon after yolk resorption at 1-2 in (30-45 mm) in length (Lister et al. 1971, Healey 1991). In most river systems, however, fry migrate at 60–150 days post-hatching or as fingerling in the late summer or autumn of their first year. When environmental conditions are not conducive to subyearling emigration, ocean-type chinook salmon may remain in fresh water for their entire first year.

Stream-type chinook salmon migrate during their second or, more rarely, their third spring. The underlying biological bases for differences in juvenile life history appear to be both environmental and genetic (Randall et al. 1987). Distance of migration to the marine environment, stream stability, stream flow and temperature regimes, stream and estuary productivity, and general weather regimes have been implicated in the evolution and expression of specific emigration timing. Once stream-type chinook salmon leave freshwater, they usually move quickly through the estuary, into coastal waters, and ultimately to the open ocean (Healey 1983, Healey 1991). Thus, they are often more dependent on freshwater, rather than estuarine, ecosystems.

The majority of fall-run chinook salmon emigrate to the marine environment as subyearlings (Reimers and Loeffel 1967, Howell et al. 1985, Hymer et al. 1992a, Olsen et al. 1992, WDF et al. 1993). Most lower Columbia fall chinook exhibit the ocean-type life history, emigrating to saltwater within their first year (Myers et al. 1998). A portion of returning adults whose scales indicate a yearling smolt migration may be the result of extended hatchery-rearing programs rather than of natural, volitional yearling emigration. It is also possible that modifications in the river environment may have altered the duration of freshwater residence (Myers et al. 1998).

In the lower Columbia basin, spring chinook generally remain in the river for a full year. However, some stocks migrate downstream from their natal tributaries in the fall and early winter into larger rivers, including the mainstem Columbia River, where they are believed to over-winter before outmigration the next spring as yearling smolts. Cowlitz River spring-run chinook clearly exhibit yearling smolt pattern as revealed by scale analysis of returning adults (Table 5 in Myers et al. 2003). However, the natural timing of lower Columbia spring-run chinook salmon emigration is likely obscured by hatchery releases of spring-run chinook salmon juveniles late in their first autumn or early in their second spring (Myers et al 1998, 2003). Age analysis based on scales from naturally spawning spring-run adults from the Kalama and Lewis rivers indicated a significant contribution to escapement by fish that entered saltwater as subyearlings (Hymer et al. 1992a).

2.1.1.6 Estuary Rearing and Growth

Ocean-type chinook salmon reside in estuaries for longer periods as fry and fingerlings than do yearling, stream-type chinook salmon smolts (Reimers 1973, Kjelson et al. 1982, Healey 1991). Rivers with well-developed estuaries, such as the Columbia, are able to sustain larger ocean-type populations than those without (Levy and Northcote 1982). Juvenile chinook salmon growth in estuaries is often superior to river-based growth (Rich 1920a, Reimers 1971, Schluchter and Lichatowich 1977).

Since ocean-type chinook salmon spend more time in the estuary, they are more susceptible to changes in the productivity of that environment than stream-type chinook salmon. Estuaries may be 'overgrazed' when large numbers of ocean-type juveniles enter the estuary *en masse* (Reimers 1973, Healey 1991). The potential also exists for large-scale hatchery releases of fry and fingerling ocean-type chinook salmon to overwhelm the production capacity of estuaries (Lichatowich and McIntyre 1987). The loss of coastal wetlands to urban or agricultural development may more directly affect ocean-type populations than stream-type populations. For example, Thomas (1983) and Johnson et al. (2003b) have documented substantial loss of marsh and swamp habitat throughout the estuary and lower Columbia River mainstem; further, many researchers (Levy and Northcote 1982, Myers and Horton 1982, Simenstad et al. 1982, Levings et al. 1986, Bottom et al. 1984) have documented that small juvenile salmonids usually occupy shallow, protected habitats such as salt marshes, tidal creeks, and intertidal flats.

Diet of juvenile fall chinook varies considerably based on fish size and location in the river, estuary, and nearshore habitats (e.g. Craddock et al. 1976, McConnell et al. 1978, Sibert and Kask 1978, Kjelson et al. 1982, Levy and Northcote 1982, McCabe et al. 1983, Bottom et al. 1984, Dawley et al. 1986, McCabe et al. 1986, Bottom and Jones 1990, Sherwood et al. 1990, Healey 1991, Brodeur 1992, Miller and Simenstad 1997, Simenstad and Cordell 2000).). For young chinook in the lower Columbia River mainstem, Craddock et al. (1976) determined that diptera were the primary prey species during the winter and spring while zooplankton (primarily *Daphnia*) were the major prey item from July to October. Chironomids, *Daphnia*, amphipods (*Eogammarus* and *Corophium spp.*), *Neomysis*, small fish (juvenile herring, sticklebacks, other salmon), and crustacea larvae have all been identified as important food items in estuaries (Healey 1991). Bottom et al. (1984) and Bottom and Jones (1990) reported that young chinook in the Columbia River estuary primarily ate amphipods (*Corophium*), cladocerans (*Daphnia*), and diptera, with *Corophium* dominant in winter and spring and *Daphnia* dominant in summer. Seasonal changes in diet are typical, however, it is unclear whether this is related to seasonal abundance of food items or a result of diet shifts as chinook grow.

Growth in the estuary is correlated with food supply. As a result, growth rate varies between estuaries and between years within an estuary (Healey 1982, Neilson et al. 1985). Reported growth rate estimates range from 0.00275 in/d to 0.52 in/d (0.07 mm/d to 1.32 mm/d), although most estimates seem to fall near the range of 0.0197-0.295 in/d (0.5-0.75 mm/d) (Reimers 1971, Fedorenko et al. 1979, Healey 1980, Levy and Northcote 1981, Kjelson et al. 1982, Neilson et al. 1985, Levings et al. 1986). However, it is uncertain whether growth rate estimates are a measure of the true growth rate or are an artifact of sampling bias.

In the Columbia River estuary, subyearling chinook salmon were captured in every month of the year and were distributed throughout freshwater, estuarine, and marine regions (Bottom et al. 1984). Reimers (1973), working in the Sixes River, Oregon, suggested that estuarine rearing is critical to fall chinook survival. Subyearling chinook were one of the most abundant species collected in the Columbia River estuary; Bottom et al. (1984) suggested that subyearling chinook abundance was partially related to their slow migration through the estuary (i.e. subyearling chinook were available for long periods of time in a variety of estuarine habitats). For example, subyearling chinook tagged and released in April and May were captured in the estuary through October (Bottom et al. 1984). Subyearling chinook moved through the estuary slower than other salmonids; in fact, migration rate appeared to decrease for about half the hatchery groups when they entered the estuary (Bottom et al. 1984). Generally, juvenile hatchery subyearling chinook released further upstream in the basin migrated at a faster rate than juveniles released lower in the system (Bottom et al. 1984). Subyearling chinook abundance was highest in the spring and summer months; during spring and summer, subyearling chinook were most frequently associated with water column and nearshore habitats while in the winter, they were more frequently associated with nearshore, shoals, and bay habitats (Bottom et al. 1984). Subyearling chinook represented 68% of the total catch of juvenile salmonids in the estuary (Bottom et al. 1984).

Recent sampling of juvenile salmonids in the Columbia River plume has started to illustrate patterns of habitat use by salmonids in the plume and nearshore ocean habitats (Fresh et al. 2003), although limited years of data are currently available. First, juvenile salmon distance offshore appears to be positively related to river flow as measured at Bonneville Dam; generally, chinook and coho salmon yearling were captured further offshore in the plume environment as river flow increased (Fresh et al. 2003). Second, preliminary evidence suggests that some juvenile salmonids (chum, steelhead, and yearling coho) may preferentially utilize the plume front compared to other areas in the plume or adjacent ocean habitats, although this did not appear to be the case for yearling chinook salmon (Fresh et al. 2003). Although reasons for the apparent preference to the plume front are not clear, this area may be a more productive habitat than elsewhere in the plume and adjacent ocean.

2.1.1.7 Ocean Migrations

Ocean migrations of chinook salmon extend well into the North Pacific Ocean. Chinook salmon tend to be widely distributed and run deeper (to 110 m) than other salmon species (Major et al. 1978). Most chinook salmon remain at sea from 1 to 6 years (more commonly 2 to 4 years). Early maturing males returning to freshwater after 1 year at sea are commonly known as jacks. A small number of yearling males mature in fresh water or return after 2 or 3 months in salt water. (Rutter 1904, Gilbert 1913, Rich 1920a, Mullan et al. 1992).

Ocean migratory pattern differences between and within ocean- and stream-type chinook salmon stocks may be partly responsible for different fluctuations in abundance. They may also reflect long-term geographic and seasonal differences in marine productivity and estuary availability. In addition, differences in the ocean distribution of specific stocks may be indicative of resource partitioning and may be important to the success of the species as a whole. Current migratory patterns may have evolved as a balance between the relative benefits of accessing specific feeding grounds and the energy expenditure necessary to reach them. If the migratory pattern for each population is, in part, genetically based, then the efficiency with which subsequent generations reach and return from their traditional feeding grounds will be increased (NMFS 1998).

Actual oceanic migratory patterns are difficult to discern, especially in the vast marine areas where no fisheries are prosecuted and, hence, no tagged fish are recaptured. Coded-wire tag (CWT) data can help elucidate oceanic migrations, at least in areas where fisheries occur. Myers et al. (1998) stated that CWT recoveries of chinook from the lower Columbia River ESU (ocean-type) generally indicate a northerly ocean migration route, but with little contribution to Alaskan fisheries. For several specific examples, CWT recovery indicates that: Grays River Hatchery fall chinook are harvested primarily in southern British Columbia (51%), Columbia River (25%), and Washington ocean (12%) fisheries; Cowlitz River Salmon Hatchery fall chinook are harvested primarily in Washington ocean (30%), British Columbia (21%), Alaska (15%), Cowlitz River (11%), and Columbia River (8%) fisheries; and Kalama Hatchery fall chinook are harvested primarily in Alaska (38%), British Columbia (36%), Columbia River (14%), and Washington ocean (6%) fisheries. These three example stocks demonstrate that lower

Columbia fall Chinook can range far to the north and that the distribution is rather variable among stocks.

While collecting samples for genetic analysis of oceanic mixed-stock harvest from 88 locations extending from British Columbia to northern California, Utter et al. (1987) found that Columbia River tule fall chinook tended to be caught in the coastal waters of Washington, while upriver brights tended to be caught in Alaska and British Columbia commercial harvest.

2.1.2 Distribution

During the last 10,000 years, flow, water chemistry and physical features of specific habitats have shaped the characteristics of chinook salmon populations in the lower Columbia basin (Miller 1965). Since physical conditions varied between the different lower Columbia River tributaries, chinook once returned to individual spawning sites over a longer period than they do today. Chinook returning to hatcheries were originally divided into race based on time of arrival at the hatchery. Fish arriving before July 31, were categorized as spring chinook and after that date as fall chinook (Senn, H. 1993). This method, however, ignored the entry time of summer chinook adults. As a result, summer chinook have been mixed with both spring and fall races.

Fall chinook were predominant in the lower Columbia, with runs returning to the Cowlitz, Toutle, Coweeman, Lewis, Kalama, Chinook, Grays, Elochoman, Washougal, Big White Salmon and Little White Salmon rivers, as well as to some smaller Washington-side tributaries of the lower Columbia River (Figure 2-3). Chinook populations in many of these tributaries began declining by the early 1900s because of overharvest and poor land use practices. The Big White Salmon River (RKm 270) supported runs of chinook salmon prior to the construction of Condit Dam (RKm 4) in 1913 (Fulton 1968). Although some fall-run salmon spawning occurs below Condit Dam, there have been substantial introductions of non-native stocks (WDF et al. 1993), and the persistence of a discrete native stock is unlikely. Fall-run fish from the Big White Salmon River were used to establish the nearby Spring Creek National Fish Hatchery (NFH) in 1901 (Hymer et al. 1992a). Spring Creek NFH is one component of the extensive hatchery system in Washington and Oregon producing fall chinook salmon (Howell et al. 1985). Among other fall-run populations, a later returning component of the fall chinook salmon run exists in the Lewis and Sandy rivers (WDF et al. 1993, Kostow 1995, Marshall et al. 1995). Because of the longer time interval between freshwater entry and spawning, Lewis and Sandy river fall chinook salmon are less mature at freshwater entry than tule fall chinook salmon and are commonly termed lower river ‘brights’ (Marshall et al. 1995).

Historically in Washington, spring chinook returned to the Cowlitz, Lewis, Kalama, and Big White Salmon rivers (Figure 2-4). The Cowlitz, Kalama, Lewis, Clackamas, and Sandy rivers presently contain both spring and fall runs, while the Big White Salmon River historically contained both spring and fall runs but presently only contains fall-run fish (Fulton 1968, WDF et al. 1993). The Klickitat River probably contained only spring-run chinook salmon due to falls that blocked access to fall-run chinook salmon during autumn low flows (Fulton 1968). The spring run on the Big White Salmon River was extirpated following construction of Condit Dam (Fulton 1968), while a variety of factors may have caused the decline and extinction of spring-run chinook salmon on the Hood River (Nehlsen et al. 1991, Kostow 1995). Dams have reduced or eliminated access to upriver spring Chinook spawning areas on the Cowlitz, Lewis, Clackamas, Sandy, and Big White Salmon rivers.

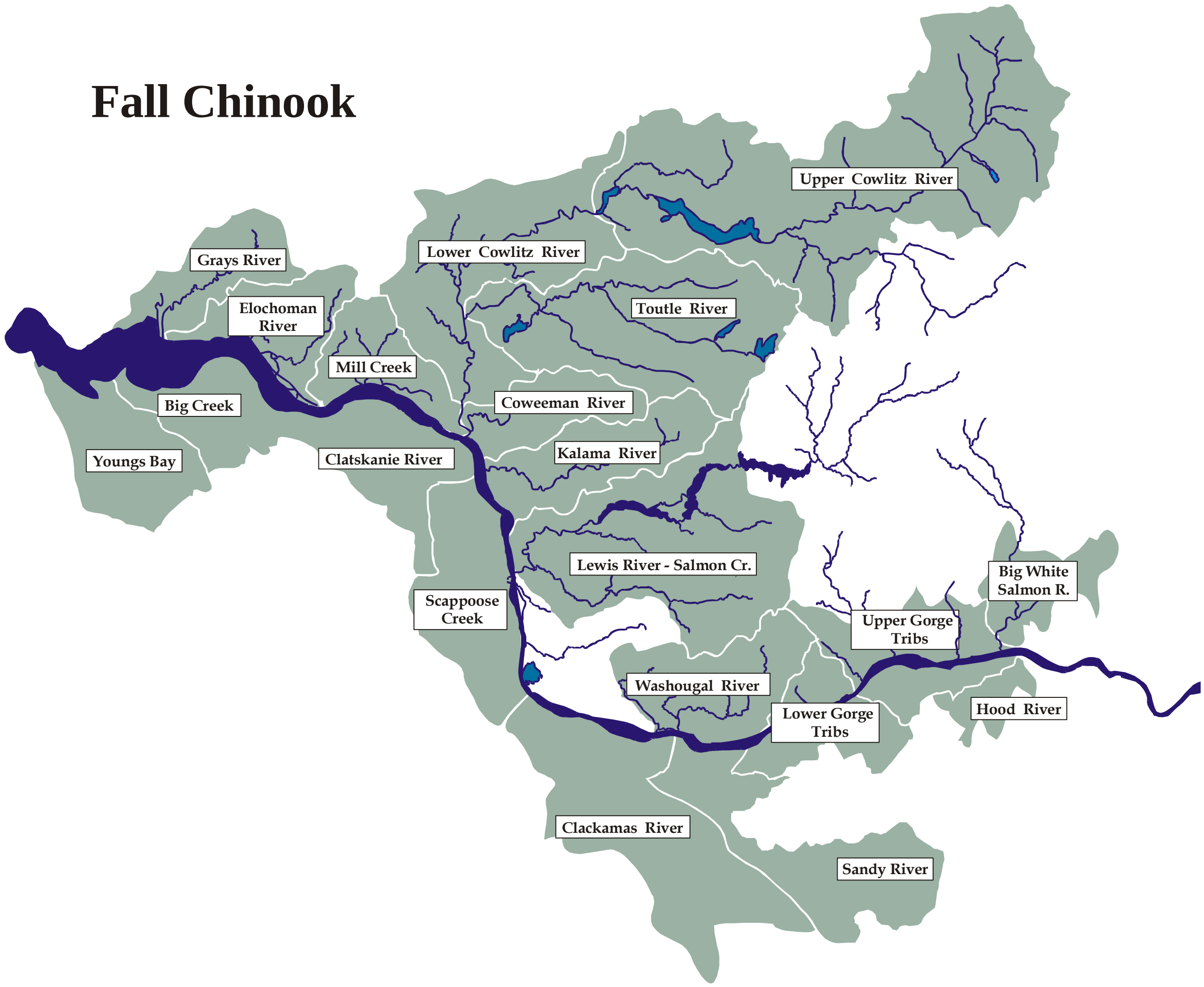

Figure 2-3. Historical demographically independent fall chinook salmon populations in the lower Columbia River ESU (Myers et al. 2002).

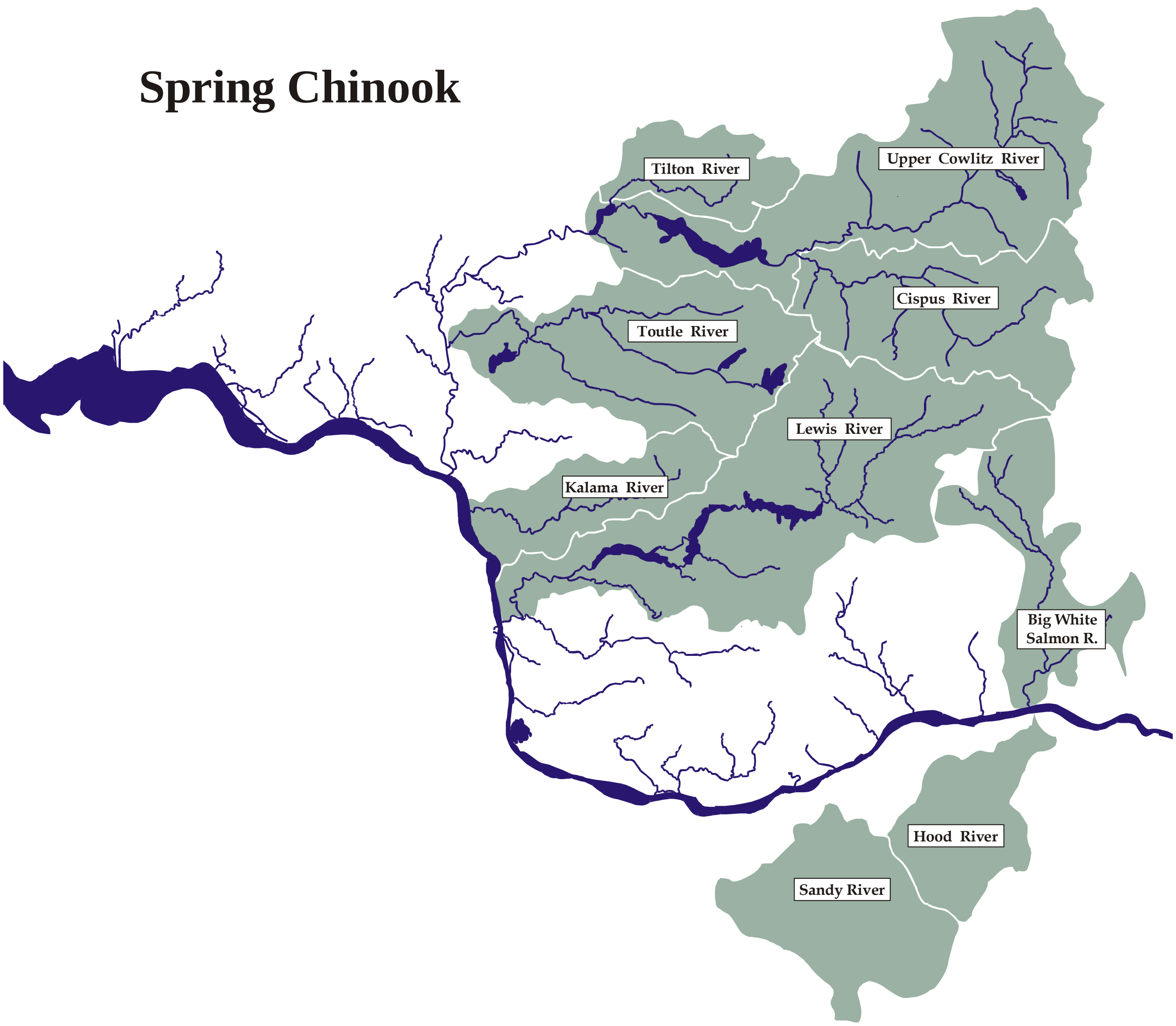

Figure 2-4. Historical demographically independent spring chinook salmon populations in the lower Columbia River ESU (Meyers et al. 2002).

2.1.3 Genetic Diversity

Utter et al. (1989) examined allozyme variability at 25 polymorphic loci in samples from 86 chinook populations extending from the Skeena River, British Columbia, to the Sacramento and San Joaquin Rivers, California. Their cluster analysis of genetic distances (Nei 1972) indicated the existence of nine genetically distinct regional groups of populations. Three groups were located in the Columbia River basin: lower Columbia River and its tributaries, populations above Bonneville Dam (except the Snake River), and the Snake River.

Schreck et al. (1986) examined allele frequency variability at 18 polymorphic loci to infer genetic relationships among 56 Columbia River Basin chinook salmon populations. A hierarchical cluster analysis of genetic correlations between populations identified two major groups. The first contained spring chinook salmon east of the Cascade Mountains and summer chinook in the Salmon River. This group contained three subclusters:

1. wild and hatchery run spring chinook salmon east of the Cascades,
2. spring run chinook in Idaho, and

3. widely scattered groups of spring chinook in the White Salmon River Hatchery, the Marion Forks Hatchery, and the Tucannon River.

A second major group consisted of spring chinook salmon west of the Cascade Crest, summer fish in the upper Columbia River, and all fall-run fish. Three subclusters also appeared in this group:

1. spring- and fall-run chinook salmon in the Willamette River,
2. spring- and fall-run chinook salmon below Bonneville Dam, and
3. summer- and fall-run chinook salmon in the upper Columbia River.

Winans (1989) estimated levels of gene diversity with 33 loci for spring, summer, and fall run chinook salmon at 28 localities in the Columbia River Basin. Fall-run chinook tended to have significantly greater levels of gene diversity than both spring and summer chinook salmon.

Waples et al. (1991) examined 21 polymorphic loci in samples from 44 populations of Columbia River Basin chinook salmon. An unweighted pair group method with arithmetic mean (UPGMA) tree of Nei's (1978) genetic distances between samples showed three major clusters of Columbia River Basin chinook salmon: 1) Snake River spring and summer chinook salmon, and mid- and upper Columbia River spring chinook salmon, 2) Willamette River spring chinook salmon, and 3) mid- and upper Columbia River fall and summer chinook salmon, Snake River fall chinook salmon, and lower Columbia River fall and spring chinook salmon.

In the NMFS status review, geneticists analyzed a set of allele frequencies for 31 loci in 55 samples from the Columbia and Snake rivers to depict population structure among these drainages. Samples in this analysis were separated into two distinct clusters: ocean-type populations and stream-type populations; except for a sample of spring chinook salmon from the Klickitat River, which was genetically intermediate between the two clusters. Results showed that additional genetic population structure was apparent within these two life history types. Within ocean-type chinook salmon, samples of spring and fall chinook salmon from the lower Columbia River were distinct from all inland samples. Furthermore, lower Columbia River spring-run fish were genetically more closely allied with nearby fall-run fish in the lower Columbia River than with spring-run fish in the Snake and upper Columbia rivers (Myers et al. 1998).

Taken together, the results of these studies indicate that the timing of chinook salmon returns to natal rivers is not necessarily consistent with genetic subdivisions. For example, spring chinook populations in the Snake, Willamette and lower, mid, and upper Columbia rivers were genetically distinct from each other, but had similar run timings. In addition, lower Columbia River tule fall chinook fish and upper Columbia River bright fall chinook have similar run timings, but were genetically distinct from one another. Conversely, spring and fall chinook in the lower Columbia River have different run timing, but were genetically similar (NMFS 1998). The large genetic groupings seem to be driven by geographic isolation more than run timing. Utter et al. (1989) stated that their clustering or gene diversity analyses did not support the concept that chinook salmon adult run times represented distinct 'races' with separate ancestries, rather that genetic divergence into temporally distinct runs tended to occur within an area from a common ancestry.

2.1.4 ESU Definition

The lower Columbia River chinook salmon ESU includes all native populations from the mouth of the Columbia River to the Cascade Crest, excluding populations above Willamette

Falls (Myers et al. 1998, 2003). Celilo Falls, which historically may have presented a migrational barrier to chinook salmon under certain flow conditions, is the eastern boundary of the ESU. Exclusions from the ESU are stream-type spring chinook found in the Klickitat River (mid-Columbia ESU) and the introduced Carson spring chinook. Tule fall chinook from the Wind and Little White Salmon rivers are included in the ESU, but introduced bright fall chinook salmon populations in the Wind, White Salmon, and Klickitat rivers are not included. Information suggests that spring chinook in the Clackamas and Sandy rivers are predominantly introduced chinook from the Willamette River ESU and are probably not representative of spring chinook historically found in these two rivers.

Chinook populations in this ESU are considered by NMFS to be ocean-type (Myers et al. 1998). However, some spring chinook populations have a large proportion of yearling migrants. Data for naturally reproducing spring chinook is limited and scale-based aging data, such as that collected by Hymer et al. (1992) may be biased by yearling hatchery releases. These populations exhibit a range of juvenile life history patterns that appear to depend on local environmental conditions. CWT recoveries for lower Columbia River ESU populations indicate a northerly migration route, but with little contribution to the Alaskan fishery. Populations in this ESU also tend to mature at ages 3 and 4, somewhat younger than populations from the coastal, upriver, and Willamette ESUs. Ecologically, the Lower Columbia River ESU crosses several ecoregions: Coastal, Willamette Valley, Cascades, and East Cascades (Myers et al. 1998).

2.1.5 Life History Differences

The obvious life history difference observed among chinook in the lower Columbia River basin is the presence of spring- and fall-run chinook. However, as described above, there is little evidence that spring and fall chinook in the lower Columbia basin are genetically distinct runs. Both spring and fall chinook in the region have been considered ocean-type chinook (i.e. migrate to the ocean during their first summer as subyearlings). However, recent scale analysis of juvenile spring chinook indicates that most lower Columbia spring chinook emigrate as yearlings. This analysis is heavily biased by the abundance of hatchery-released yearling spring chinook; it is unlikely that native spring chinook in the lower Columbia have adapted a stream-type life history.

Another difference among lower Columbia fall chinook is the observed rate of straying among chinook stocks in different regions. For example, fall chinook in the Coastal Range tributaries (i.e. Chinook, Grays, and Elochoman basins) have a high rate of straying, perhaps because of the relatively short length of these tributaries and/or because chinook mainly only use the lower rivers just above tidal influence. On the other hand, chinook in the western Cascade Range tributaries (i.e. Cowlitz, Kalama, Lewis, and Washougal) exhibit a high degree of spawning site fidelity, potentially because fish returning to larger-sized basins normally have a higher degree of homing fidelity. Of the hatchery releases analyzed in this region, more than 90% of the freshwater recoveries occurred in their natal river basin.

Among spring chinook populations in the lower Columbia River basin, there is little deviation in the life history strategies described above. There is little evidence documenting naturally produced juvenile spring chinook stream residence time. Spring chinook in the region may emigrate in the summer as subyearlings, however, documenting this is problematic when yearling hatchery spring chinook dominate the emigration.

Although fall chinook salmon populations are generally thought to be one widely mixed stock as a result of straying and egg transfers between hatcheries (Howell et al. 1985, WDF et al. 1993, Marshall et al. 1995), numerous life history differences can be observed among fall chinook populations throughout the lower Columbia basin. Many of the differences in life history strategies can be attributed to the presence of wild fish maintaining the historical characteristics of a population. Deviations from the typical life history pattern (described above) are observed in Abernathy/Germany, Cowlitz, NF Lewis, EF Lewis, Bonneville area tributaries, and Wind River fall chinook.

In Abernathy and Germany creeks, sexually mature 1-year old fall chinook have been found. In the Cowlitz basin, spawning generally occurs from September to November, over a broader time period than most fall chinook, and peak spawning activity does not occur until the first week in November, which is later that most fall chinook. The NF Lewis River has sustained a healthy natural population of bright fall chinook. These fish generally migrate from August through October, over a broader time period than other lower Columbia fall chinook. NF Lewis River bright fall chinook typically spawn from October through January, with peak activity in November. This spawn timing is substantially later than most other lower Columbia fall chinook stocks. Also, the dominant age classes of NF Lewis River bright fall chinook are 4- and 5- year olds. Furthermore, CWT data indicates that NF Lewis River bright fall chinook have a more northerly ocean distribution than other fall chinook from the region. On the EF Lewis River, fall chinook spawning occurs in two distinct segments; the early segment spawns in October and the late segment spawns from November through January. It is possible that the late segment is related to the bright fall chinook population on the NF Lewis River.

Dominant age classes of EF Lewis fall chinook include 3-, 4-, and 5-year olds. In the Wind River, tule fall chinook range from 2 to 4 years old, with 4-year olds predominating, while Wind River bright fall chinook range from 2 to 6 year olds, with 5-year old spawners predominating.

Wind River bright fall chinook likely originated from strays from Bonneville Hatchery and Little White Salmon NFH and are not indigenous to the Wind River. Some upriver bright fall chinook, spawn from mid-October to late November in the mainstem Columbia below Bonneville Dam. This stock was discovered in 1994 and is considered to have originated from hatchery strays from the Bonneville Hatchery upriver bright fall chinook program. These are not considered part of the Lower Columbia River chinook salmon ESU.

2.1.6 Abundance—Spring Chinook

There is widespread agreement that natural production has been substantially reduced over the last century. Chinook salmon in the region have been strongly affected by losses and alterations of freshwater habitat (Bottom et al. 1985, WDF et al. 1993, Kostow 1995). Large runs of spring chinook returned to the lower Columbia historically, most notably to the upper Cowlitz and upper Lewis basins. Both the Lewis and Cowlitz spring chinook are identified as depressed by WDFW in SASSI (2002). For example, in 1946, WDF estimated spring chinook escapement in the Cowlitz basin above the proposed Mayfield Dam site was 9,000 fish; when adjusted for harvest, this escapement represents a total spring chinook run to the Cowlitz of 32,490 fish (most produced from the Cispus River). From 1962 to 1966, an average of 9,928 spring chinook were counted annually at Mayfield Dam; from 1978 to 1985, only 3,894 spring chinook were counted annually at the dam. Historically, spring chinook were abundant in the upper Lewis basin, especially in the Muddy Fork and upper NF Lewis mainstem, with an estimate of at least 3,000 returning to spawn prior to the completion of Merwin Dam in 1932 (WDF 1951). The Merwin Dam was constructed downstream of the spring chinook habitat, and by 1950 only a remnant population of spring chinook (<100) remained. The spring chinook run to the Kalama may have been significant historically, but by the early 1950s, only a remnant population of spring chinook

(<100) existed in the Kalama. Kalama spring chinook spawning escapement has averaged 444 fish since 1980 and most spawners are considered first generation hatchery fish.

Spring chinook continue to return to the Cowlitz, Lewis, Kalama, Wind and Little White Salmon rivers, however these runs are almost entirely from hatchery production. Total runs (i.e. escapement plus catch) to the Cowlitz, Lewis, and Kalama rivers have ranged from 3,000 to 36,900 during 1980–2002 (Figure 2-5; WDF 1951).

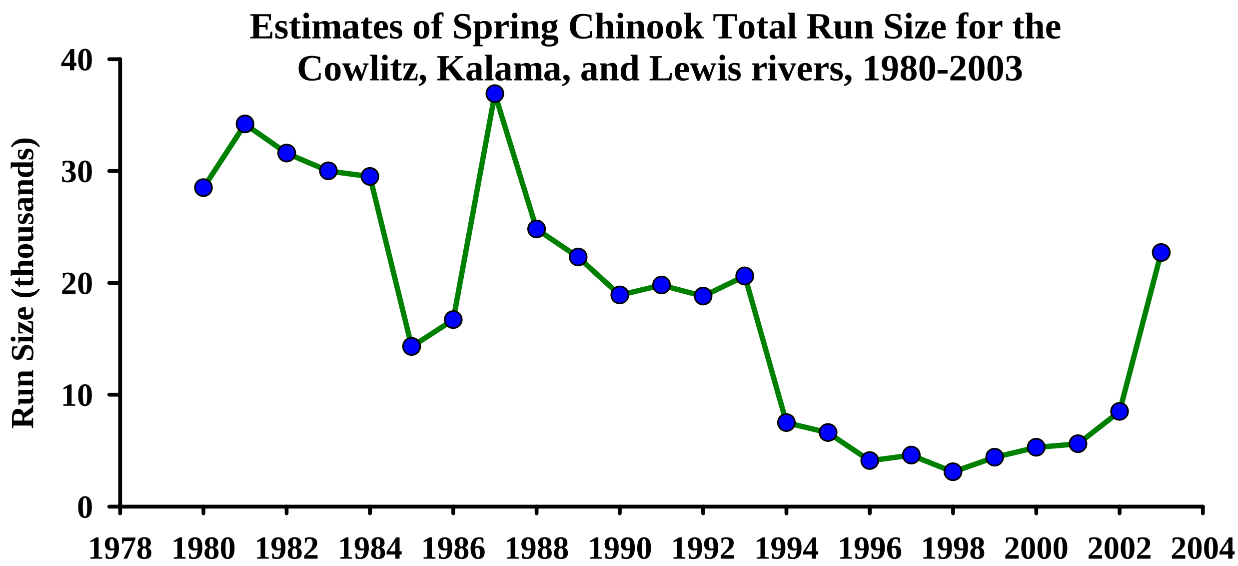

Figure 2-5. Total run size of spring chinook to the Cowlitz, Kalama and Lewis rivers.

In the Lewis River, the naturally spawning spring chinook population is considered healthy based on escapement trends (WDF/WDW 1993), but some research suggests that the native Lewis River spring chinook run is extinct (Myers et al. 1998) and that most natural spawners are resulting from hatchery programs. The Cowlitz River now produces very few spring chinook from natural spawning (average escapement of 338 fish since 1980), and these are generally considered hatchery strays (Hillson and Tipping 2000, cited in Wade 2000). The Kalama River spring chinook population is considered healthy, but shows signs of a severe short-term decline (WDF/WDW 1993). All naturally spawning of spring chinook in the lower Little White Salmon River stopped after the filling of the Bonneville Pool. In addition, hatchery spring chinook runs exist in the Little White Salmon and Wind rivers, however, spring chinook were not historically present in these basins. Spring chinook were historically present in the Big White Salmon River, but were extirpated after the construction of Condit Dam in 1917.

Overall, the number of naturally spawning spring chinook runs in the Lower Columbia River ESU is very low. The Biological Recovery Team (BRT) established by NMFS to evaluate the status of chinook was unable to identify any healthy native spring chinook populations in the ESU. Based on expanded peak fish counts in index areas, the 5-year (1992–96) geometric mean of spring run natural spawning escapement to the Lower Columbia River ESU was 11,200 fish. CWT accounting indicates that approximately 68% of natural spawners are first generation hatchery strays. Long-term escapement trends for spring chinook are positive or stable although short-term trends are negative. The BRT concluded that the pervasive influence of hatchery fish in almost every river in the ESU and the degradation of freshwater habitat suggested that many naturally spawning populations are not able to replace themselves (NMFS 1998).

2.1.7 Productivity—Spring Chinook

Very little data are available to assess the productivity of spring chinook in the lower Columbia River. In the absence of data, natural spring chinook production is believed to be quite low. The Northwest Power and Conservation Council's (NPCC)[1] smolt density model was applied to many systems in the lower Columbia to estimate potential spring chinook salmon smolt production (NPPC 1989). (The NPPC smolt density model produces optimistic smolt potential estimates compared to the EDT model.) In the Cowlitz basin, the model predicts potential spring chinook production of 329,000 smolts below Mayfield Dam, 788,400 smolts for the Toutle system, and 1,600,000 smolts for the basin above Mayfield Dam. In the Kalama basin, the model predicts smolt production of 111,192 smolts in the Kalama below Kalama Falls and 465,160 smolts above Kalama Falls. Wind River smolt production was estimated as 157,533 smolts, while the Little White Salmon River can produce an estimated 32,350 smolts. Smolt production estimates were not available for the Lewis River basin. Based on the smolt density model, the lower Columbia basins with existing populations of spring chinook (except the Lewis) could produce a total of 3,483,635 smolts. The vast majority of the lower Columbia production potential is in habitat upstream of the Cowlitz and Lewis hydro electric projects.

2.1.8 Abundance—Fall Chinook

Natural production of fall chinook has also dropped far below historical levels. Historically, the Cowlitz River was the primary producer of fall chinook in the lower Columbia River ESU; an estimated 100,000 adults once returned to the Cowlitz basin (WDF 1951). Although little historical information is available on tributary escapement, WDF and WDW estimated that the total Cowlitz run in 1948 was 63,612 fall chinook, with approximately 14,000 fish spawning above the proposed Mayfield Dam site. From 1961 to 1966, an average of 8,535 fall chinook were counted annually at Mayfield Dam. The natural spawning escapement goal of 3,000 fish was met in 2002, however, the escapement goal had not been previously met since the late 1980s. In the Coweeman basin, WDF estimated fall chinook escapement in 1951 as 5,000 fish. Since 1964, Coweeman fall chinook escapement has averaged 302 fish, although annual spikes in escapement have been observed periodically over the last 15 years (Figure 2-6) (WDFW 2002). In the Kalama River basin, chinook escapement in 1936 was estimated as 20,000 fish, although only 7,000 were allowed to spawn naturally and 13,000 were collected at the Fallert Creek Hatchery (operating since 1895). Fall chinook spawning escapements in the Kalama have averaged 5,514 fish since 1964. However, most natural spawners are likely first generation hatchery fish.

On the NF Lewis River, annual fall chinook spawning escapements have averaged 11,232 since 1964; most spawners in this basin are from natural production. The 5,700 fish escapement goal for NF Lewis fall chinook is met and exceeded in most years (Figure 2-7). Other basins in the lower Columbia River historically supported fall chinook runs of a few thousand fish, including the Grays, Elochoman, and Washougal rivers; escapement to these basins is currently below historical levels and sustained primarily by hatchery fish (although hatchery fall chinook are no longer released into the Grays River).

[1] The Northwest Power and Conservation Council (NPCC) was formerly known as the Northwest Power Planning Council (NWPPC)

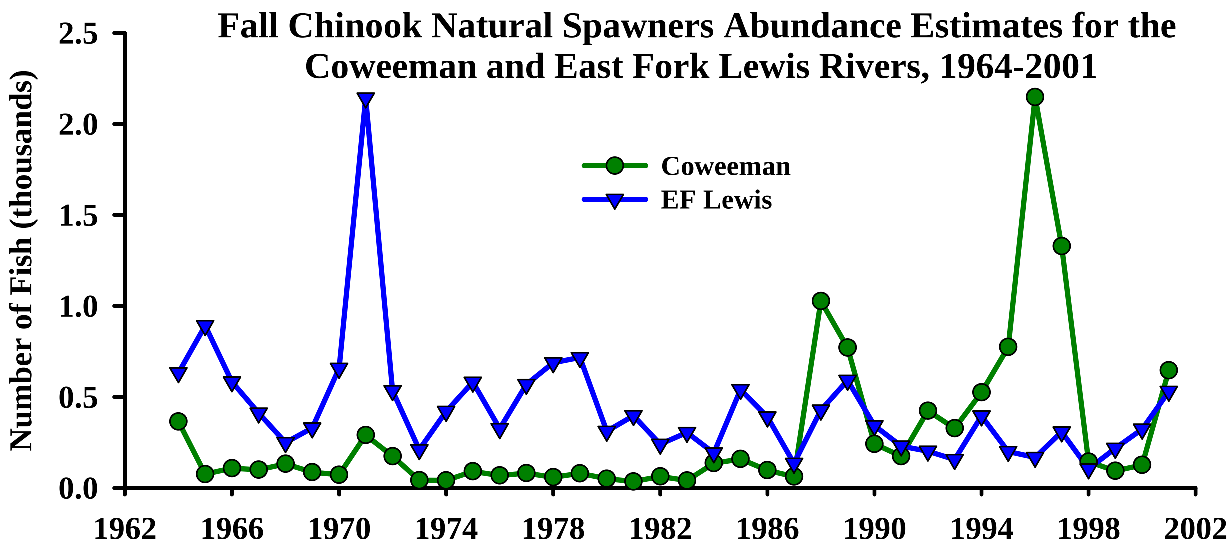

Figure 2-6. Natural spawner fall chinook abundance estimates for the Coweeman and EF Lewis rivers.

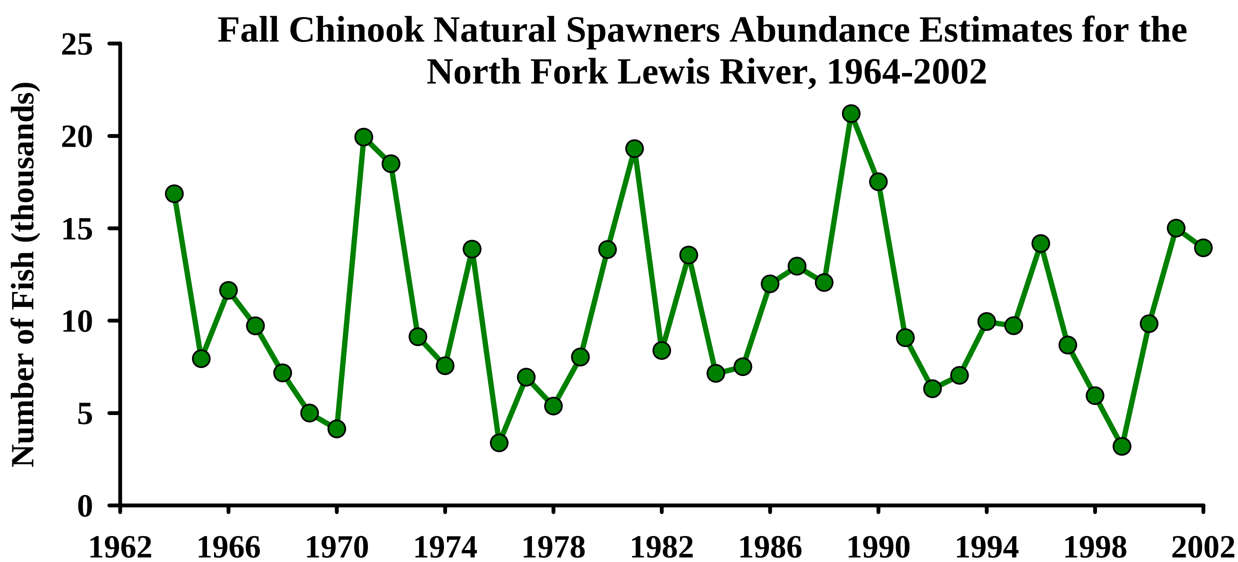

Figure 2-7. Fall chinook natural spawner abundance estimates for the NF Lewis River.

Today, fall chinook continue to return to the Cowlitz, Lewis, Kalama, Washougal, Grays, Chinook, and Elochoman rivers, as well as to several smaller lower Columbia tributaries. Only three fall chinook stocks, the North Lewis River, EF Lewis River, and Coweeman River fall runs, are considered to be of native origin and predominantly natural production. The Lewis River populations are considered healthy based on escapement trend (Wade 2000), however, recent analysis suggests that EF Lewis fall chinook are depressed based on low spawner escapement levels. Coweeman fall chinook are also considered depressed based on low spawner escapement levels.

Based on expanded peak fish counts in index areas, the 5-year (1991–95) geometric mean of fall run escapement to the Lower Columbia River ESU was 29,000 natural spawners and 37,000 hatchery spawners. Long-term escapement trends for fall chinook are mixed, with most larger stocks positive. However, short-term trends are negative.

Lower river hatchery stock is a management unit representing hatchery and natural production of tule chinook. Lower river wild stock is a management unit representing later-timed wild fall chinook production, primarily from the Lewis River. Figure 2-8 displays total Columbia River returns (fishery harvest and spawning escapement combined) for these stocks from 1984 to 2002.

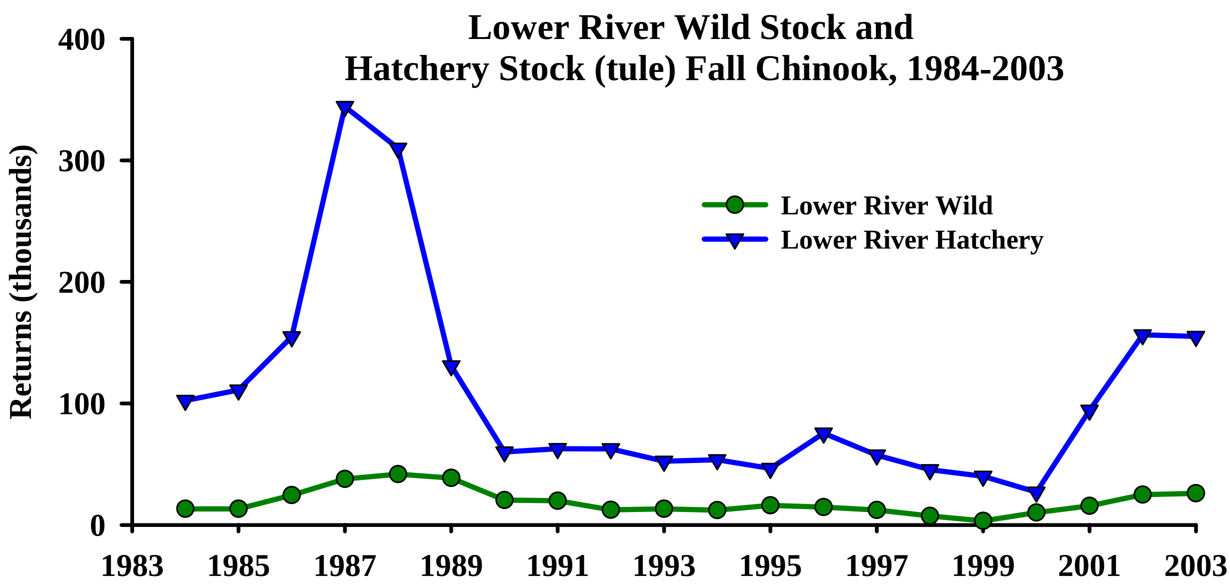

Figure 2-8. Returns of lower Columbia River hatchery and wild fall chinook stocks.

2.1.9 Productivity—Fall Chinook

Most fall chinook salmon populations in Washington tributaries to the lower Columbia River are thought to be one widely-mixed stock as a result of straying and egg transfers between hatcheries (Howell et al. 1985, WDF et al. 1993, Marshall et al. 1995). However, very few egg transfers have been made to Cowlitz and Kalama hatcheries, and the existing hatchery stocks are assumed to be similar to the original natural spawning populations in those rivers.

Cowlitz River fall chinook natural spawners are a mixed stock of composite production. Their status was listed as healthy by SASSI in 1993, but current fall chinook stocks are considered depressed by WDFW (Hillson and Tipping 2000 cited in Wade 2000). Mobrand Biometrics (1999) compared observed and estimated adult wild fall chinook returns to the Cowlitz River from about 1920 to 1999. Their results show that production, once estimated at 100,000 adults, declined to ~18,000 fish in the 1950s, ~12,000 in the 1960s and recently declined to less than 2,000 fish. An ecosystem diagnosis and treatment (EDT) analysis attributed the extreme loss in major production to mainstem dams that barred fish passage to historical habitat in the upper basin. They also attributed losses in the lower Cowlitz downstream of the Toutle River to major human-caused changes to the river channel, such as dredging, diking, and straightening. The EDT analysis states that "*uncertainty exists with all of the run-size estimates discussed, and the results must be applied with caution; however, the pattern is troubling*" (Mobrand Biometrics 1999).

Natural spawning occurs in the Washougal, Grays, Chinook, and Elochoman drainages, but the majority of returning fall chinook that spawn naturally are considered to be hatchery strays. There has been a concern regarding releases of Rogue River fall chinook at Youngs Bay, which are released into the lower Columbia River to increase harvest opportunities, and their documented straying into many tributaries in the lower Columbia River (NMFS 1998). In recent years, ODFW addressed the concern by eliminating Rogue stock releases from Big Creek Hatchery where they tended to home poorly. Rogue stock release now are entirely within Youngs Bay, Oregon weher an intensive gill-net fishery exists.

Today, the fall chinook run in the Lewis River appears to be the only healthy naturally produced population in the lower Columbia River ESU (NMFS 1998). NF Lewis River fall chinook represent about 80% to 85% of the wild fall chinook escapement to the lower Columbia River (WDF 1990). In a recent stock status inventory (SASSI 2002), WDFW grouped the lower Columbia River fall chinook populations into the following categories:

- Healthy - Elochoman, Abernathy, Toutle (Green), Kalama, NF Lewis, Washougal, and Wind (bright),
- Depressed - Grays, Skamokawa, Germany, Mill, Cowlitz, Coweeman, SF Toutle, and EF Lewis, and
- Critical - Wind (tule).

2.1.10 Listing Status

The BRT established by NMFS to examine the status of chinook determined in 1998 that the estimated overall abundance of chinook salmon in the Lower Columbia ESU is not cause for immediate concern. However, they found that apart from the relatively large, and apparently healthy fall-run population in the Lewis River, production in the ESU appears to be predominantly hatchery-driven with few identifiable native, naturally reproducing populations. Long- and short-term trends in abundance of individual populations are mostly negative, some severely so. About half of the populations comprising this ESU are very small, increasing the likelihood that risks due to genetic and demographic processes in small populations will be important. Numbers of naturally spawning spring-run chinook salmon are very low. The BRT cautioned that it is possible that some native spring chinook runs are now extinct, but that this loss is masked by the presence of naturally spawning hatchery fish. The BRT was particularly concerned about the inability to identify any healthy native spring run populations. The large numbers of hatchery fish in the ESU make it difficult to determine the proportion of naturally produced fish. While studies show that genetic and life history characteristics of populations in the Lower Columbia ESU still differ from those in other ESUs, the BRT identified the loss of fitness and diversity within the ESU as an important concern (NMFS 1998). The Lower Columbia River Chinook salmon ESU was listed as a threatened species under the Endangered Species Act (ESA) on March 24, 1999 (Fed. Reg., V64, N56, p.14308) .

2.2 Coho Salmon *(Oncorhynchus kisutch)*

Coho salmon (*Oncorhynchus kisutch)* is a widespread species of Pacific salmon, with production in most major river basins around the Pacific Rim from central California to Korea and northern Hokkaido, Japan (Laufle et al. 1986), as well as in many smaller independent tributaries through the region. In the lower Columbia River basin, coho salmon historically returned to spawn in all accessible tributary reaches. Until the early 1800s, these watersheds remained essentially untouched by human development. Heavy growth of coniferous trees and understory vegetation armored many river stretches. The Columbia River systems' cool streamflows, clean gravel beds, and deep pools supported healthy populations of coho, other salmon, and steelhead. These pristine environments began to change in the mid-1800s, often causing declines in salmonid production. Coho runs were further affected by hydro development and harvest pressure on the lower Columbia River. Harvest emphasis moved to coho as chinook abundance dropped; peak commercial catches of coho in the Columbia River occurred in 1925 (Lichatowich and Mobrand 1995).

Present coho populations in tributaries to the Washington side of the lower Columbia River have been heavily influenced by extensive hatchery releases. Investigations report that a number of local populations of coho salmon in the area have become extinct, and that the abundance of many others is depressed (Brown and Moyle 1991, Nehlsen et al. 1991, Frissell 1993, NMFS 1995).

2.2.1 Life History and Requirements

The freshwater life history cycle of lower Columbia coho salmon populations follows the timing of seasonal changes in river flow and water temperatures in lower Columbia River tributaries. The region generally has a mild climate with warm, relatively dry summers and cool, wet winters. The river environments coho enter are characterized by relatively low elevations in the headwaters (1,640-3,280 ft [500-1,000 m]), with moderate amounts of precipitation (80-95 in/year [200-240 cm/year]). These rivers display relatively low flows during late summer and early fall, increased river flows and decreased water temperatures beginning in early October, and a single flow peak in December or January.

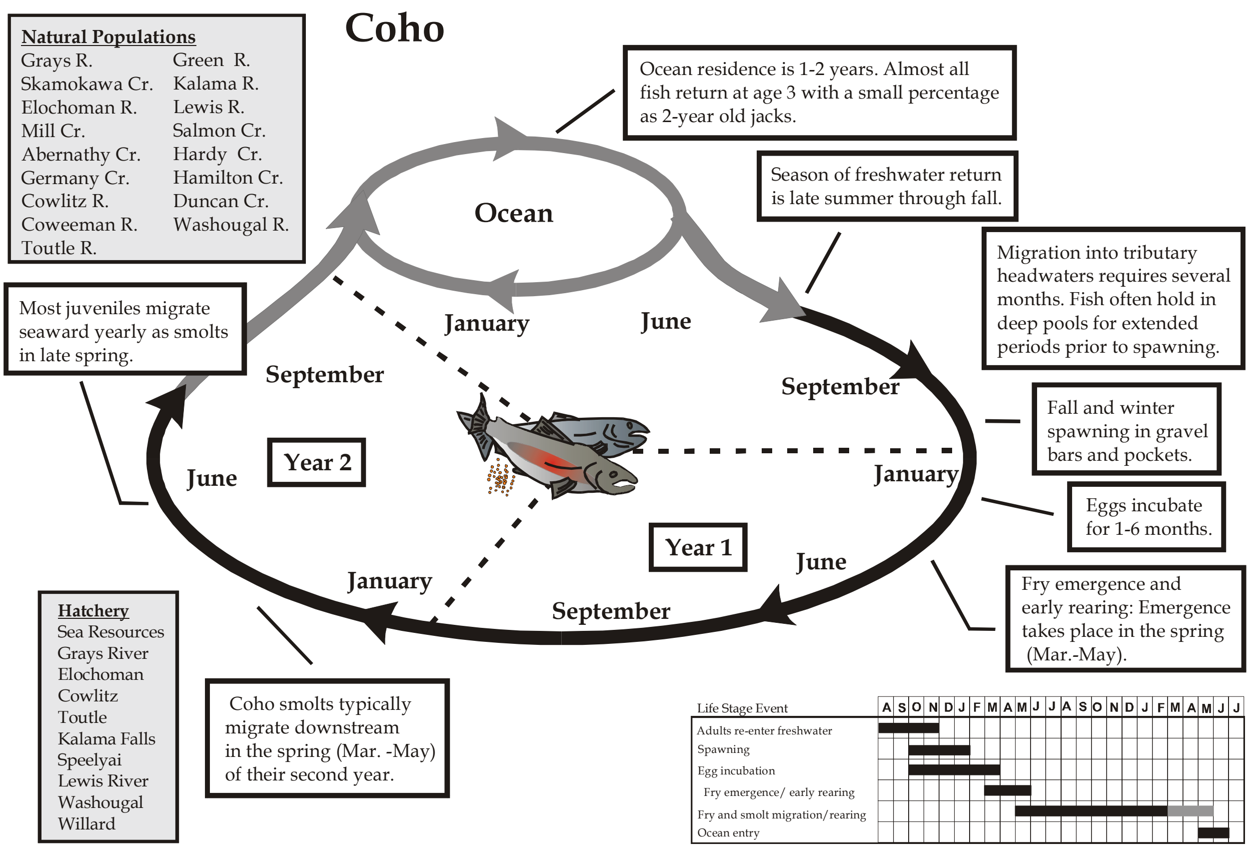

Figure 2-9. Coho salmon life history.

2.2.1.1 Upstream Migration Timing

Coho runs to the Columbia River show considerable temporal variability in river entry and spawn timing. Coho salmon begin to return to the Columbia River in August, continuing through December/ January and peaking in September/October. This variability resembles the pattern of river entry in other river systems, such as the Chehalis in southwest Washington, the Skagit in northern Washington, and the Klamath in northern California (Leidy and Leidy 1984, WDF et al. 1993).

Coho generally return in two runs:

- Early-returning (Type S) coho enter the Columbia River in mid-August and begin entering tributaries in early September, with spawning peaks from mid-October to early November.[2]
- Late-returning (Type N) coho pass through the lower Columbia from late September through December and enter tributaries from October through January.[3] Most spawning occurs from November to January, but some spawning ranges to February and as late as March.

In some regions, individual coho stocks show exceptionally early or late run timings; these stocks are often referred to as summer or winter runs, respectively (Godfrey 1965), and are thought to have evolved in response to particular flow conditions (Sandercock 1991). The relationship between populations with unusually timed runs and normally timed runs within the same basin is not well understood. For example, in some cases, such as the Soleduck

[2] referred to as Type S because their ocean migration is generally south of the Columbia River.
[3] referred to as Type N because of a more northern ocean distribution.

(Washington coast) and Clackamas (Willamette River) rivers, differently-timed, sympatric runs are thought to be largely reproductively isolated from each other (Houston 1983, Cramer and Cramer 1994), while in the Grays Harbor basin, there is believed to be reproductive overlap (WDF et al. 1993). Unusually timed runs are found in many geographic areas. However, because there is no evidence to suggest that all runs of a certain type are closely related, differently timed runs are considered to be a component of overall life history diversity within each area (NMFS 1995).

Total residence time of coho in freshwater streams is highly variable and dependent upon both environmental and population specific factors, and can range from a few days to months. Residence time in the spawning areas of the stream (i.e. survey life, or the number of days the average spawner is alive in a survey area) has ranged from 3-15 days across the Pacific coast region for 30 reported population observations, with an average of 11.4 days (Perrin and Irvine 1990).

2.2.1.2 Spawning

In general, earlier migrating fish spawn farther upstream within a basin than later migrating fish, which enter rivers in a more advanced state of sexual maturity (Sandercock 1991). Spawning usually occurs between November and early February, but generally peaks October–December and can extend into March (WDF et al. 1993). Tributary spawning extends from October through at least February, and into March in some river systems.

The timing of coho spawning can also reflect water temperature changes in a particular river system. Lister et al. (1981) found that spawn timing of coho salmon in tributaries of the Cowichan River (British Columbia) was strongly correlated to tributary water temperature. Coho salmon spawning in warmer tributaries spawned later than those spawning in colder tributaries. Such factors make determining and comparing when coho will enter a river or spawn difficult because of the temperature variability within basins (NMFS 1995).

Other environmental factors influence coho spawning as well. Adult coho returning to spawn need adequate flows and water quality, and unimpeded passage to their natal grounds. Thus, the onset of coho salmon spawning in lower Columbia tributaries is tied to the first significant fall freshet. They often mill near the river mouths or in lower river pools until freshets occur. They also need deep pools with vegetative cover and instream structures such as root wads for resting and shelter from predators. In addition, successful spawning and incubation depend on the presence of appropriate sized gravel. In research on the upper Toutle and Green rivers within the Cowlitz basin, Burner (1951) described coho redd characteristics. Coho prefer substrate of 6 in (15 cm) or smaller; only 10% of the coho redds were constructed in gravels greater than 6 in (15 cm) in diameter (Burner 1951).

While spawner size in naturally spawning populations normally shows considerable spatial and temporal variability, scientists have found that coho salmon, throughout their range, are, over time, becoming smaller, and the rates of decrease are specific to populations of fish or to certain areas (Ricker 1981, Bigler and Helle 1994). While the size of coho salmon adults is declining fastest in the Puget Sound/Strait of Georgia area, smaller fish are also returning to Columbia River tributaries. Results of a regression analysis of coho salmon size (FL [cm] or total weight oz [kg] over time, found that from 1952–92, the change in estimated weight for fish in the Columbia River fishery was statistically significant (P=0.00; NMFS 1995). Fecundity in coho increases with length (Salo and Bayliff 1958, Shaplov and Taft 1954).

It is not clear whether such size reductions are because of harvest practices, effects of fish culture, declining ocean productivity, density-dependence effects in the marine and freshwater environments attributable to large numbers of hatchery releases, or a combination of these factors. It is also not known whether there have been permanent genetic changes related to changes in fish size. Regardless of its cause or potential genetic effects, reduced adult size in itself poses a number of serious risks to natural populations of coho, and could be a sign of other factors placing the population at risk (NMFS 1995).

2.2.1.3 Incubation and Emergence

Coho eggs normally incubate for 1–6 months before hatching. During incubation, the eggs need stable gravel that is not choked with sediment, thus river channel stability is vital at this life history stage. Floods have their greatest impact to salmon populations during incubation, and flood impacts are worsened by human activities—particularly those related to timber harvest and urban development. In a natural river system, the upland areas are forested, and the trees and their roots store precipitation and slowly release stormwater into the stream. Trees within the riparian area also contribute large pieces of wood to the river that create habitat diversity and slow streamflow. Natural systems also have floodplains that connect directly to the river, and provide habitat and temporarily store floodwater. In a healthy river, erosion or sediment input is great enough to provide new gravel for spawning and incubation, but does not overwhelm the system, raising the riverbed and increasing channel instability. A stable incubation environment is essential for coho and other salmon, but is highly dependent on the functioning of nearly all habitat components in an ecosystem.

As with other salmonids, the length of time required for coho eggs to incubate is largely dependent on water temperature, and to a lesser extent, on dissolved oxygen content. The colder the temperature, the slower the development rate; however, for a given temperature, hatching time often differs between eggs from different fish or even different eggs from the same fish (Shapovalov and Taft 1954). Consequently, a wide range of values have been reported for the mean number of days that elapsed from fertilization to hatching for coho salmon. Data indicates that coho require approximately 300-450 temperature units (TU) for hatching. In general, the hatching time is shorter for North American stocks than Asian stocks, even at northern latitudes.

After hatching, coho alevins migrate downward in the gravel; migration distance is related to gravel size (Dill 1969). If the gravel was ≤1.25 in (3.2 cm) in diameter, alevins migrated down about 4 in (10 cm). If the gravel diameter ranged from 1.25 to 2.5 in (3.2 to 6.3 cm), the alevins migrated down more than 8 in (20 cm). This migration appears to be an adaptation to prevent premature emergence of alevins located close to the gravel surface.

For coho salmon on the Big Qualicum River, Fraser et al. (1983) documented the total heat requirement for fry emergence was 1,036 ± 138 degree (°C) days. The total time from egg deposition to fry emergence averaged 167 days (range 149-188 d). In three Oregon coastal streams, Koski (1966) observed that the average time from egg deposition to fry emergence was 110 days (range 104-115 d).

Most mortality from the egg to fry stage is a result of winter flooding and the associated disruptive gravel movement. Coho have high egg to fry survival compared to other salmonids; Neave (1949) attributes this high survival to the selection of better spawning sites in areas of good flow stability and less crowding. Under average conditions, approximately 15-27% of eggs will survive to emergence (Neave 1949, Crone and Bond 1976). In three Oregon coastal streams, Koski (1966) reported that egg to fry survival ranged from 0-78% and averaged 27.1%. Also, survival to emergence was positively correlated with gravel sizes >0.1 in (3.35 mm) and <1 in (26.9 mm). However, survival decreases if the gravel bed has a high concentration (up to 50%) of fine sediment and sand (Tagart 1984). Furthermore, if the gravel is heavily compacted with fine sediment, fry may not be able to get out of the gravel (Koski 1966); when the gravel/sand mixture was 70% sand, survival to emergence was only 8% (Phillips et al. 1975).

Spawn timing affects time of fry emergence. In the lower Columbia River, peak spawning time for early run coho (Type S) is in late October, and for late run coho (Type N), peak spawning is generally from December to early January. In the Washougal River system, coho spawn from mid-October and continue through November. Incubation extends from late October through January, with emergence occurring in late January and early February (WDF 1990). On the Cowlitz River, fry emergence occurs from January to April (WDW 1990). In the Cowlitz and Lewis rivers at 50 °F (10°C), fertilization to eyed-egg stage takes about 3.5 weeks, eyed-egg to hatching about 2.5 weeks, and hatching to emergence about 8 weeks (Howell et al. 1985). On the Wind and Little White Salmon rivers, coho fry emerge in late winter/early spring, generally from mid-January to February (WDW 1990). These patterns likely typify the incubation timing occurring in other Washington tributaries to the lower Columbia River.

2.2.1.4 Freshwater Rearing

After emergence, coho fry move to shallow, low velocity rearing areas, primarily along the stream edges and in side channels. They congregate in quiet backwaters, side channels, and small creeks, especially in shady areas with overhanging branches (Gribanov 1948). The vast majority of coho juveniles remain in the river for a full year after leaving the gravel. They often rear in the same habitat areas as chinook, either commingled with or entering an area as the chinook fry are leaving. Stein et al. (1972) observed that coho juveniles at the head of riffles were able to defend the area against chinook fingerling. Although juvenile coho are found in both riffle and pool habitat, they are best adapted to holding in pools (Hartman 1965). They do not compete well with trout for rearing space in riffles. Godfrey (1965) noted that coho fry from 1.5-1.75 in (38-45 mm) may migrate upstream considerable distances to reach lakes or other rearing areas; in lakes, coho fry generally occupy the nearshore littoral zone (Mason 1974). However, the majority rear in streams. As they grow, the juveniles move into faster water and disperse into tributaries and areas that adults cannot access (Neave 1949). Coho fry are active during daylight hours and seem to tolerate a wide range of light intensities; this adapts coho well to the small, shallow streams they normally occupy where light conditions are highly variable (Hoar 1958).

The two most important factors for coho freshwater survival are water discharge rate and temperature. There is a correlation between summer flows and the catch of adult coho salmon 2 years later (Neave 1948, 1949, Smoker 1953). During summer months, the amount and quality of juvenile rearing habitat can decline due to low flows and high water temperatures. This may lead to a physical reduction of available habitat, increased stranding, decreased dissolved oxygen, and increased predation (Cederholm and Scarlett 1981). Coho fry production also has been shown to be a function of the stability of winter flows (Lister and Walker 1966, Seiler 2003).

The most productive coho streams are those with alternating pools and riffles of about equal area (i.e. 1:1 pool to riffle ratio; Ruggles 1966). Invertebrate production is maximized in the riffle area and the pool habitat is the optimum environment for coho fry holding and feeding (Mundie 1969). Coho tend to be more aggressive in defense of their territories where the current is fast and most of the available food is coming from upstream. In areas where the current is slow or slack, the food can appear from any direction and coho tend to move in loose aggregates while scrambling for food (Mundie 1969). As coho juveniles grow into yearlings, they become more predatory on fry of their own or other species (Gribanov 1948).

While juvenile coho are highly territorial and can occupy the same area for a long period of time (Hoar 1958), coho abundance can be limited by the number of suitable territories available (Larkin 1977). McMahon (1983) determined that pools of 10-80 m^3 (353-2825 ft^3) in size were optimum for coho production, provided there was adequate shading from streamside vegetation; however, if the canopy is very dense, then coho biomass will be reduced (Chapman and Knudsen 1980). Streams with more structure (logs, undercut banks, etc.) support more coho (Scrivener and Andersen 1982), not only because they provide more territories (useable habitat), but they also provide more food and cover. There is a positive correlation between the amount of terrestrial insect material in coho stomachs and the extent the stream was overgrown with vegetation (Chapman 1965). In addition, the leaf litter in the fall contributes to aquatic insect production (Meehan et al. 1977).

Coho fry are continually displaced downstream by freshets throughout the active juvenile growth period (Fraser et al. 1983). If the downstream area is unoccupied, displaced fry may take up residence; however, if fry already occupy the space, displaced fry will be displaced further downstream (Ruggles 1966). Evolutionarily, the displacement may distribute fry far from the spawning grounds, allowing them to make more effective use of the available habitat (Allen 1969). However, in many cases, fry are displaced to less favorable sites, where they become more vulnerable to predators or are prematurely driven to the estuary. Some coho fry displaced downstream may migrate back upstream, or they may move along the shore in low salinity water and enter other streams to continue rearing (Otto and McInerney 1970). Of those coho fry that are displaced to the ocean or voluntarily choose to migrate to sea in their first year, survival to the adult stage is uncommon (Crone and Bond 1976). However, Otto (1971) points out that the type of estuary has a substantial bearing on the ability of coho fry to survive. Crone and Bond (1976) demonstrated that coho fry could survive salinities as high as 29 ppt if they had been acclimated at lower salinities first. Regardless, adult production from coho fry that enter the sea during their first year is expected to be very low.

Pool habitat is also important during all stages of juvenile development. Preferred coho pool habitat includes deep pools with riparian cover and woody debris. In the autumn as the temperatures decline and juvenile coho feeding activity decreases, juvenile coho move into deeper pools and hide under logs, tree roots, and undercut banks (Hartman 1965). The fall freshets redistribute them (Scarlett and Cederholm 1984), and over-wintering generally occurs in available side channels, spring-fed ponds, and other off-channel sites to avoid winter floods (Peterson 1980). The lack of side channels and small tributaries may limit coho survival in some areas (Cederholm and Scarlett 1981). Fall freshets may cause considerable downstream migration before suitable overwintering habitat can be found. For example, in Washington's Clearwater River, coho move as much as 24 miles (38 km) downstream before entering a tributary (Scarlett and Cederholm 1984). Coho also have been observed overwintering in lakes, such as Tenmile Lake in Oregon.

High summer water temperatures can exceed the 77°F (25°C) upper lethal temperature for juvenile coho. Brett (1952) observed that juvenile coho preferred a temperature range of 54-57°F (12-14°C), which is close to optimum for growth efficiency. Rapid increases or decreases in temperature can also cause significant mortality in juvenile coho (Brett 1952). Most mortality in the fry stage occurs in the first summer. Godfrey (1965) reported an average fry to smolt survival of 1.27-1.71% for two British Columbia streams, one Washington stream, and one Oregon stream. British Columbia survival from egg to smolt has been estimated at 1-2% (Neave and Wickett 1953). In the Big Qualicum River, Fraser et al. (1983) estimated fry to smolt survival was 7.3%. The long freshwater residence time likely results in higher freshwater mortality, than juveniles with shorter freshwater residence time, but may contribute to a lower marine mortality because smolts are larger when they go to sea (Drucker 1972).

2.2.1.5 Juvenile Migration

Most juvenile coho, in the region south of central British Columbia, migrate seaward as smolts in late spring, typically during their second year. Factors that tend to affect the time of migration include; the size of the fish, flow conditions, water temperature, dissolved oxygen levels, day length, and the availability of food (Shapovalov and Taft 1954). The size of coho smolts is fairly consistent over the species' geographic range; a FL of 4 in (100 mm) seems to be the threshold for smoltification (Gribanov 1948). Generally, the timing of outmigration is earlier in the southern coho populations compared to northern populations. For example, coho smolt emigration in California starts as early as mid-March and peaks in mid-May (Shapovalov and Taft 1954), but in the Resurrection Bay area of Alaska, smolt migration begins in late May and has been observed into early September (McHenry 1981). In the lower Columbia River, coho smolt emigration likely occurs from March to June, with peak movement in April and May. In addition, the timing of smolt outmigration may respond to small-scale habitat variability, with smolts residing in ponds and lakes often having different outmigration timing and being a different size than smolts residing in streams within the same basin (Swales et al. 1988, Irvine and Ward 1989, Rodgers et al. 1993, Nielsen 1994).

Changes in the environment can also cue coho smolts to migrate. For example, Tripp and McCart (1983) observed the main peak of coho emigration coincided with a time of maximum stream discharge. In addition, a second peak of migration was observed during a time of increasing water temperature. For a given river system, there are annual variations in emigration timing that are related to these environmental factors. Changes in a tributary created by habitat degradation, habitat restoration and/or flow control often influence outmigration timing and though the relationships are not yet clear. Nearshore ocean conditions can also affect the timing of smolt outmigration in some tributaries to the lower Columbia River (NMFS 1995).

2.2.1.6 Estuary Rearing and Growth

Coho use estuaries primarily for interim feeding while they adjust physiologically to salt water. However, connectivity of available feeding and refuge areas may be important for species such as coho that move quickly through the estuary. For example, radio tagged coho in Grays Harbor estuary moved alternatively from low velocity holding habitats to strong current, passive downstream, movement areas (Moser et al. 1991). Additionally, Dittman et al. (1996) suggest that habitat sequences at the landscape level may be important even for species and life history types that move quickly through the estuary during the important smoltification process, as salmon gather the olfactory cues needed for successful homing and these cues may depend on the environmental gradients experienced during migrations.

Juvenile coho salmon were present in the Columbia River estuary from March to August of each year of sampling by Bottom et al. (1984); coho abundance was greatest in May and June and relatively low for other months (Bottom et al. 1984). Juvenile coho salmon comprised 18% of the total juvenile salmonid catch (Bottom et al. 1984). Coho juveniles were distributed throughout the freshwater, estuarine, and marine regions of the estuary; they were most frequently associated with water column habitats, however, tagged hatchery coho released in the lower Columbia (i.e. Grays River (rm 34) and Big Creek (rm 29)) were more likely to be found in shallow bays and intertidal areas than upriver coho (Bottom et al. 1984). Juvenile coho salmon moved through the estuary relatively quickly and appeared to increase their migration rate through the estuary (Bottom et al. 1984). As with other salmonids, juvenile hatchery coho released further upstream in the basin migrated at a faster rate than juveniles released lower in the system (Bottom et al. 1984).

Recent sampling of juvenile salmonids in the Columbia River plume has started to illustrate patterns of habitat use by salmonids in the plume and nearshore ocean habitats (Fresh et al. 2003), although limited years of data are currently available. First, juvenile salmon distance offshore appears to be positively related to river flow as measured at Bonneville Dam; generally, chinook and coho salmon yearling were captured further offshore in the plume environment as river flow increased (Fresh et al. 2003). Second, preliminary evidence suggests that some juvenile salmonids (chum, steelhead, and yearling coho) may preferentially utilize the plume front compared to other areas in the plume or adjacent ocean habitats (Fresh et al. 2003). Although reasons for the apparent preference to the plume front are not clear, this area may be a more productive habitat than elsewhere in the plume and adjacent ocean.

2.2.1.7 Ocean Migrations

Most research indicates that, upon entering the ocean, coho remain in nearshore environments over the continental shelf for a couple of months before they disperse on more seaward migrations; this holds true from California to Alaska (Shapovalov and Taft 1954, Milne 1964, Godfrey 1965). This pattern may help coho avoid pelagic predators and reduce feeding competition with immature salmon that are older by a year or more.

Some Washington and British Columbia stocks migrate only short distances to good feeding areas and remain there until they approach maturity (Godfrey et al. 1975). In the Strait of Georgia, coho smolts quickly disperse throughout the strait; the number of coho that remain in the strait depends on coho density and feeding conditions (Healey 1980). If fish find themselves in poor feeding areas within the strait, they move to outside waters; however, those fish that locate good feeding areas remain within the Strait of Georgia.

Coho salmon typically spend 18 months in the ocean before returning to fresh water. Thus, many returning coho are 3 years old and have spent 18 months in fresh water and 18 months in salt water. Jacks, however, return earlier at age 2. These sexually mature males return to fresh water to spawn after only 5 to 7 months in the ocean.

Data collected from CWT recovery studies shows that coho salmon released from Columbia River hatcheries are recovered primarily in Oregon (36-67%) and Washington (22-54%), with lower but consistent recoveries from British Columbia (2-16%) and California (1-15%). These ocean distribution patterns were determined from CWT recovery data for 66 North American hatcheries between 1973–92 from the PSMFC’s 1994 Regional Mark Information System. Compared to Oregon coast coho salmon, Columbia River fish are recovered less frequently in California and more frequently in Washington. Although they share the same general recovery pattern, coho salmon from Washington-side Columbia River hatcheries are caught more frequently in Washington and British Columbia, and less frequently in Oregon than are those from Oregon-side hatcheries. This is presumably the result of Washington hatcheries producing both Type S and Type N coho, while Oregon hatcheries produce only Type S coho. Washington has maintained both stocks in Columbia River hatcheries because early and late returning coho are indigenous to Washington streams and the mix of stocks provides fishing access off the Washington coast as well as in the Columbia River and Washington tributaries.

2.2.2 Distribution

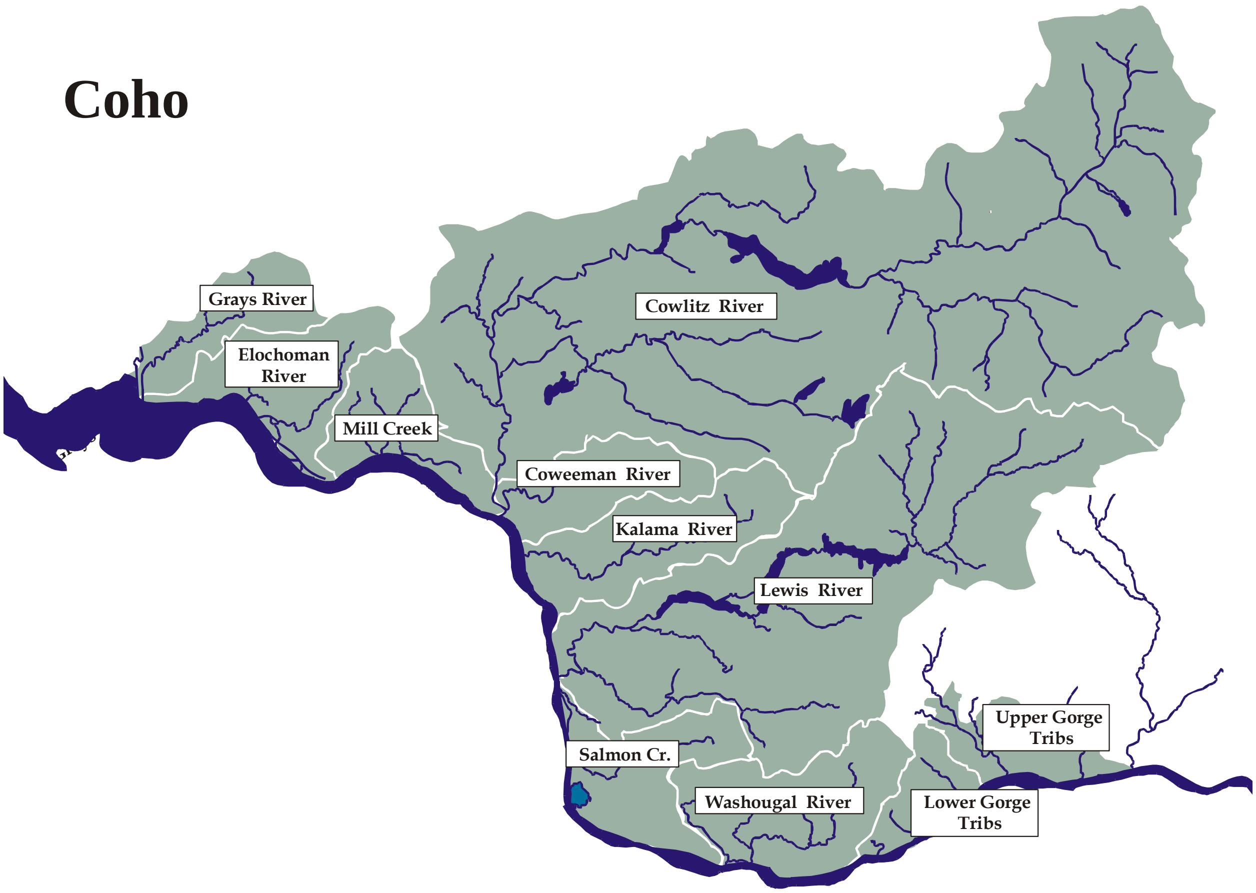

Figure 2-10. Distribution of historical coho salmon populations among Washington lower Columbia River subbasins.

Historically, coho were present in all lower Columbia River tributaries. Before the early 1800s, the watersheds remained essentially untouched by human development. Heavy growth of coniferous trees and understory vegetation armored many river stretches. The river systems' cool streamflows, clean gravel beds, and deep pools supported healthy populations of coho and other salmon and steelhead. These pristine environments began to change in the mid-1800s, often causing declines in salmonid production. Currently, very few wild coho salmon spawn annually throughout the lower Columbia River subbasins. Until recently, Columbia River coho salmon were managed as a hatchery stock. In some cases, coho salmon returning to Columbia River hatcheries above the brood stock needs for the hatchery are allowed to bypass the hatchery rack or collection facility and allowed to spawn naturally. Spawning is expected to occur in most areas accessible to coho, although production from naturally spawning hatchery fish is likely low.

Two general coho stocks are present in the lower Columbia River today: early (Type S) refers to an ocean distribution generally south of the Columbia River with an early adult run timing in the Columbia River; late (Type N) refers to an ocean distribution generally north of the Columbia River with a late run timing in the Columbia River. CWT data provides valuable information on fish distribution and harvest rate of stocks originating in Columbia River hatcheries; however, the fishery distribution does not necessarily reflect ocean migratory patterns as fishing effort levels vary in specific areas each year. CWT recoveries data are useful when describing catch distribution for a given time period. The recoveries illustrated by the following table represent fishery distribution during the late 1990s (Table 2-1). These fishery recoveries reflect years when Oregon ocean fisheries were minor compared to Washington ocean and Columbia River fisheries. Oregon fisheries were curtailed in the early 1990s in response to management of the presently ESA-listed Oregon Coastal natural coho.

Table 2-1. Coho salmon CWT recoveries in various fisheries.

	Sampling Areas			
Hatchery	**Columbia River**	**Oregon Ocean**	**Washington Ocean**	**California Ocean**
Early Coho				
Grays River[1]	58%	21%	19%	1%
Elochoman[2]	53%	7%	40%	—
North Toutle[2]	47%	1%	37%	—
Fallert Creek (Kalama)[2]	49%	9%	42%	—
Lewis River[2]	21%	21%	58%	—
Late Coho				
Elochoman[2]	59%	11%	29%	—
Cowlitz Salmon[3]	55%	1%	30%	—
Kalama Falls[2]	58%	10%	32%	—
Lewis River[2]	56%	21%	31%	—
Washougal[2]	57%	13%	30%	—

[1] 1994, 1996, and 1997 brood years; [2] 1995, 1996, and 1997 brood years; [3] 1994 and 1997 brood years

2.2.3 Genetic Diversity

The physical environments that support West Coast coho salmon—and the life history traits and genetic characteristics exhibited by these fish—indicate a substantial degree of ecological and genetic diversity. These environments range from the relatively dry climate in central California with strong and consistent upwelling offshore, to the extremely wet Olympic Peninsula with its snow and rain-fed rivers.

While information on historical runs is scarce, individual coho runs to the different tributaries probably showed a great deal of flexibility within their range. These runs became more conformed through selection by hatchery and harvest practices. In the 1940s, two separate runs of coho were reported to enter the Cowlitz River. The early run entered the Cowlitz from late August–September, with a spawning peak in late October. The late run entered from October–March, with a spawning peak in late November. Further, the Toutle River, a tributary to the Cowlitz River, historically produced an early-returning stock, with most fish returning from August–October. These early Toutle River coho are generally more southerly distributed in the ocean than the early component of the Cowlitz stock (WDW 1990).

Genetic diversity has largely been lost in the lower Columbia River because of widespread hatchery production with many out-of-basin (but mostly within-ESU) stock transfers. In the 1950s, salmon hatchery construction expanded on the lower Columbia River tributaries and hatcheries began to trap brood stock. Over time, transferring brood stock, eggs, and juvenile coho between hatcheries and planting hatchery fish off-station became commonplace throughout the watershed, resulting in a widely-mixed coho stock (WDF et al. 1993).

2.2.4 ESU Definition

In 1995, the BRT formed by NMFS concluded during its coho status review that, historically, there probably was an ESU that included coho salmon from all tributaries of the Columbia River below the Klickitat River on the Washington side and below the Deschutes River on the Oregon side. This ESU also would have included coho salmon from coastal drainages in southwest Washington between the Columbia River and Point Grenville (between the Copalis and Quinault rivers). The team based its determination in part on the similarities between the different physical environments. The Columbia River estuary, Willapa Bay, and Grays Harbor all have extensive intertidal mud and sandflats and similar estuarine fish faunas, and they differ substantially from estuaries to the north and south. Their similarity results from the shared geology of the area and the transportation of Columbia River sediments northward along the Washington coast. Other commonalties include:

- moving west to east, rivers that drain into the Columbia River have their headwaters in areas that are increasingly dry.
- Columbia River tributaries that drain the Cascade Mountains have proportionally higher flows in late summer and early fall than rivers on the Oregon coast.

Genetic analysis conducted since the 1995 status review, however, have led the BRT to conclude that SW Washington coho salmon form their own ESU, separate from LCR coho salmon. Geneticists from NMFS collected and compared allozyme data over 10 years from more than 100 coho salmon samples from locations ranging from California to Alaska, with a primary focus on Oregon, Washington, and southern British Columbia. Results from the study showed regional patterns of allele frequency.

The samples arranged into several different clusters. This cluster analysis placed SW Washington coast coho in a different cluster from LCR coho salmon, which form a supercluster with Oregon coast fishes (NMFS 2001). Within the SW Washington and LCR clusters, several subclusters and three branches have only one or two members.

The NMFS BRT concluded it could not identify any remaining natural populations of coho salmon in the lower Columbia River (excluding the Clackamas and Sandy rivers) or along the Washington coast south of Point Grenville that warrant protection under the ESA.

2.2.5 Life History Diversity

Little is known about the specific life history traits of wild coho salmon within the Washington tributaries of the lower Columbia River. There does not appear to be many significant life history differences among ‘same type’ coho stocks in the Washington subbasins of the lower Columbia River. Two general coho stocks are present in the lower Columbia River today; Type S (early) refers to an ocean distribution generally south of the Columbia River with an early adult run timing in the Columbia River and Type N (late) refers to an ocean distribution generally north of the Columbia River with a late run timing in the Columbia River. For early coho in the Washington subbasins of the lower Columbia, migration timing is generally from mid-August to September and peak spawning occurs in late October. For late coho stocks, migration timing is generally from late September to October and peak spawning occurs in December to early January. Dominant adult age class for lower Columbia coho is 1.2, indicating 1 year in fresh water and 2 years in salt water. Any natural spawning that does occur is thought to happen in all areas accessible to coho; specific spawning areas for each subbasin are noted in the Subbasin Chapters in Volume II.

For both stocks, fry emerge in the spring, spend 1 year in fresh water, then migrate to sea during the spring of their second year. In the Cowlitz River, there appears to be a late-late run component that may be an artifact of hatchery practices. Late stock hatchery programs throughout the lower Columbia have always taken brood stock for the late run from the early portions of the run. Once the annual brood stock needs were met, hatchery collection efforts ceased, leaving the later component of the run to spawn naturally. Therefore, a remnant late-late run from natural production exists in the Cowlitz basin and it is possible that other similar runs exist elsewhere in the lower Columbia.

There appears to be a significant difference in average length between the males and females of Columbia River hatchery coho stocks (Jim Ames, WDFW, personal communication). Based on CWT-marked coho recovered at hatcheries on the Washington side of the Columbia River, the females are larger in Type N coho, and to a lesser extent, in Type S coho. This length difference is not seen in other Washington hatchery coho stocks – if anything, the females tend to be smaller at most hatchery racks (Jim Ames, WDFW, personal communication).

2.2.6 Abundance

Since 1970, the Columbia River produced adult coho ocean population has ranged from a low of 96,700 in 1996 to over 3 million in 1971. The returns to the Columbia River mouth (after ocean harvest) have ranged from 74,800 in 1995 to over 1.5 million in 1986 (Figure 2-11).

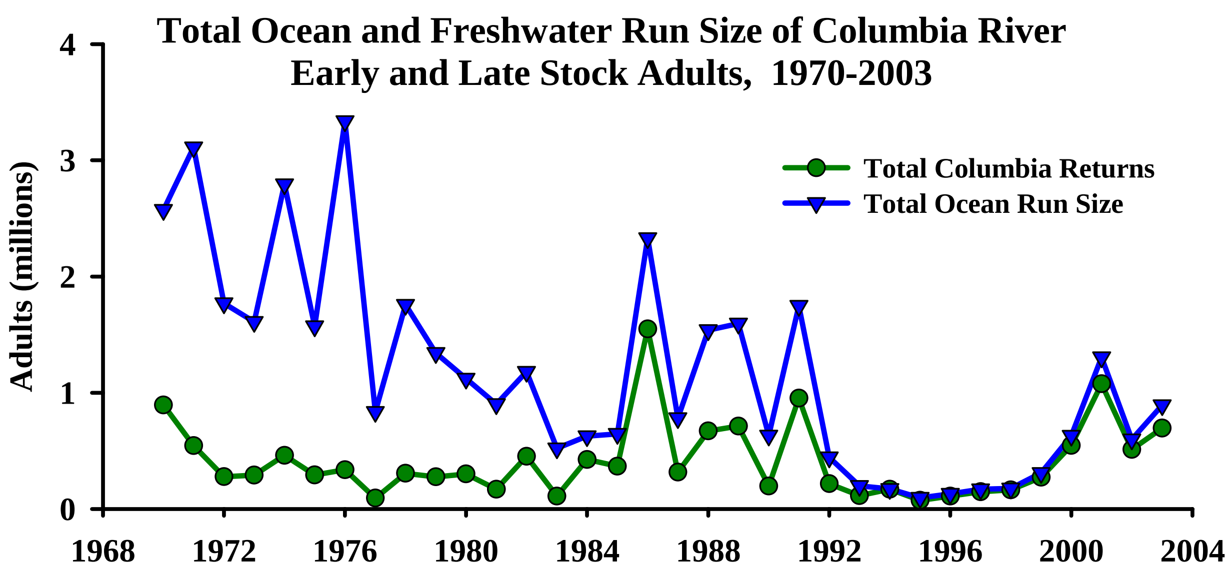

Figure 2-11. Columbia River coho ocean and freshwater run size.

While records of coho escapement from the 1940s and 1950s report runs as high as 77,000 coho to the Cowlitz River (WDF/WGC 1948) and 15,000 coho to the Lewis River, such large natural returning runs to lower Columbia River tributaries are now gone. Estimated coho escapement to other lower Columbia River systems in the early 1950s was generally in the range of a few thousand coho: Grays River (2,500), Kalama River (3,000), and the Washougal River (3,000). Today, coho stocks in these and other lower Columbia River tributaries in Washington are considered depressed, primarily because of chronically low escapement and production. Natural spawning is presumed to be quite low in most areas, and subsequent juvenile production is below stream potential. Much of this natural production has been replaced by hatchery production.

Natural coho production is now being reintroduced in some once productive habitat areas. For example, in the Cowlitz system, coho are now reseeding the highly productive habitat in the upper basin above the hydroelectric projects through a salmon reintroduction program. In 1994, a trap and haul program began with adults returning to the Cowlitz Falls Fish Collection Facility near Mayfield Dam; adult coho have been released in the upper Cowlitz, Cispus, and Tilton rivers and allowed to spawn naturally. Nearly 1 million coho smolts were estimated to have migrated out of the upper basin in the spring/summer of 2001. This shows that the upper river is still capable of producing significant numbers of fish, however, only a portion of the migrating smolts (375,000) were subsequently trapped and hauled below the dams. Downstream passage of juvenile coho continues to be a challenge for this reintroduction effort. Estimates of coho smolt production potential in the Cowlitz River basin above the dams range from 6,319 (Stockley 1961) to 261,254 fish (Easterbrooks 1980). In the Lewis, reintroduction of coho to the upper basin above the hydro-system is also being considered.

2.2.7 Productivity

The productive capacity of the freshwater environment for coho has been estimated by a number of researchers. In the Big Qualicum River, Lister and Walker (1966) determined that 19.1 smolts were produced per 100 m^2 of wetted stream area measured at low flow. In three Oregon coastal streams, Chapman (1965) reported 18-67 smolts produced per 100 m^2 over a 4-year period. In low gradient side channels of the Cowichan River, Armstrong and Argue (1977) determined that 125-141 smolts were produced per 100 m^2. In an evaluation of coho smolt production in ten Western Washington streams, average smolt production per square mile ranged from 417 to 1,798; additionally, average smolt production per $mile^2$ in select lower Columbia tributaries from 1997 to 2002 ranged from 17 to 765 (Jim Ames, WDFW, personal communication). In a study of hatchery coho in isolated headwater streams in Canada, Tripp and McCart (1983) found that the average production was 8.4-8.5 smolts per 100 m^2; this low production may be a result of hatchery smolt fitness or because high-gradient headwater streams are not usually productive areas. The average coho production likely falls between these extremes reported in the literature. In addition, smolt production in streams is 7-10 times greater than lakes (Foerster and Ricker 1953).

The NPCC’s smolt density model was run on many subbasins within the lower Columbia River to estimate potential coho smolt production. Estimates of coho smolt production for Washington subbasins include: 125,874 for the Grays River, 43,393 for the Elochoman, 123,123 for the lower Cowlitz, 131,318 for the Tilton River and Winston Creek, 155,018 above Cowlitz Falls, 142,234 for the Toutle, and 37,797 for the Coweeman.

2.2.8 Listing Status

In a 1995 status review of coho salmon, NMFS found that that if an evolutionarily significant unit of coho salmon (such as Clackamas River late-run coho) still exists in the lower Columbia River, it is not presently in danger of extinction, but is likely to become so (NMFS 1995). However, the Oregon Fish and Wildlife Commission conducted its own status review and concluded that lower Columbia coho produced in Oregon basins, including the Clackamas and Sandy Rivers, are at risk of extinction, and listed them as a state endangered species in 1998.

NOAA Fisheries was subsequently petitioned to list lower Columbia coho salmon on an emergency basis and to designate critical habitat. They determined that the petition presented substantial scientific information indicating that a listing may be warranted, but that there was insufficient evidence to support an emergency listing (Fed. Reg. V.65, N214, P. 66221). Lower

Columbia coho remain a candidate species for a potential ESA listing, with a listing decision pending.

2.3 Chum Salmon *(Oncorhynchus keta)*

Chum salmon (*Oncorhynchus keta*) have the widest natural geographic and spawning distribution of any Pacific salmonid, primarily because their range extends farther along the shores of the Arctic Ocean than other salmonids (Groot and Margolis 1991). They have been documented to spawn from Korea and the Japanese island of Honshu, east around the rim of the North Pacific Ocean, to Monterey Bay in southern California. The species' range in the Arctic Ocean extends from the Laptev Sea in the Russian Federation to the Mackenzie River in Canada (Bakkala 1970, Fredin et al. 1977). Chum salmon historically may have been the most abundant of all salmonids—Neave (1961) estimated that prior to the 1940s, chum salmon contributed almost 50% of the total biomass of all salmonids in the Pacific Ocean. Chum salmon also grow to be among the largest of Pacific salmon, second only to chinook salmon in adult size, with individuals reported up to 42.9 in (108.9 cm) in length and 45.9 lbs (20.8 kg) in weight (*Pacific Fisherman* 1928). Average size for the species is around 7.9 to 15 lbs (3.6 to 6.8 kg) (Salo 1991).

The species is best known for its canine-like teeth and the striking body color of spawning males; a calico pattern, with the anterior two-thirds of the flank marked by a bold, jagged, reddish lines and the posterior third by a jagged black line. Females are less flamboyantly colored and lack the extreme dentition of the males. The two most widely used common names, 'chum' and 'dog' salmon, reflect these traits. Chum salmon is the common name accepted by the American Fisheries Society, most likely derived from a word in the language of the Chinook peoples of the Columbia River area, *cam* (also translated as *sum* or *tzum*), which means calico.

In the Columbia River basin, chum salmon once migrated more than 310 miles (500 km) to spawn in the Walla Walla River (Nehlsen et al. 1991) and were productive in many lower Columbia River tributaries. Runs of nearly 1.4 million fish are believed to have returned annually to the Columbia River. The total minimum 2002 chum return to the Columbia River was estimated to be 19,914 fish, based on Washington tributary and lower Columbia mainstem spawning surveys (19,403), commercial fishery incidental catch (14), hatchery escapement (309), and the Bonneville Dam count (188). Production is generally limited to areas downstream of Bonneville Dam. All naturally produced chum populations in the Columbia River and its tributaries in Oregon and Washington were federally listed as threatened in August 1999.

Intensive monitoring of chum spawning escapement is conducted in three Washington tributaries in the lower Columbia basin—Grays River, Hardy Creek, and Hamilton Creek—and in the mainstem Columbia River near Ives Island. The latter three populations are located immediately downstream of Bonneville Dam. Chum salmon populations exist in other river systems of the lower Columbia, but have not been consistently monitored and abundances are assumed to be extremely low.

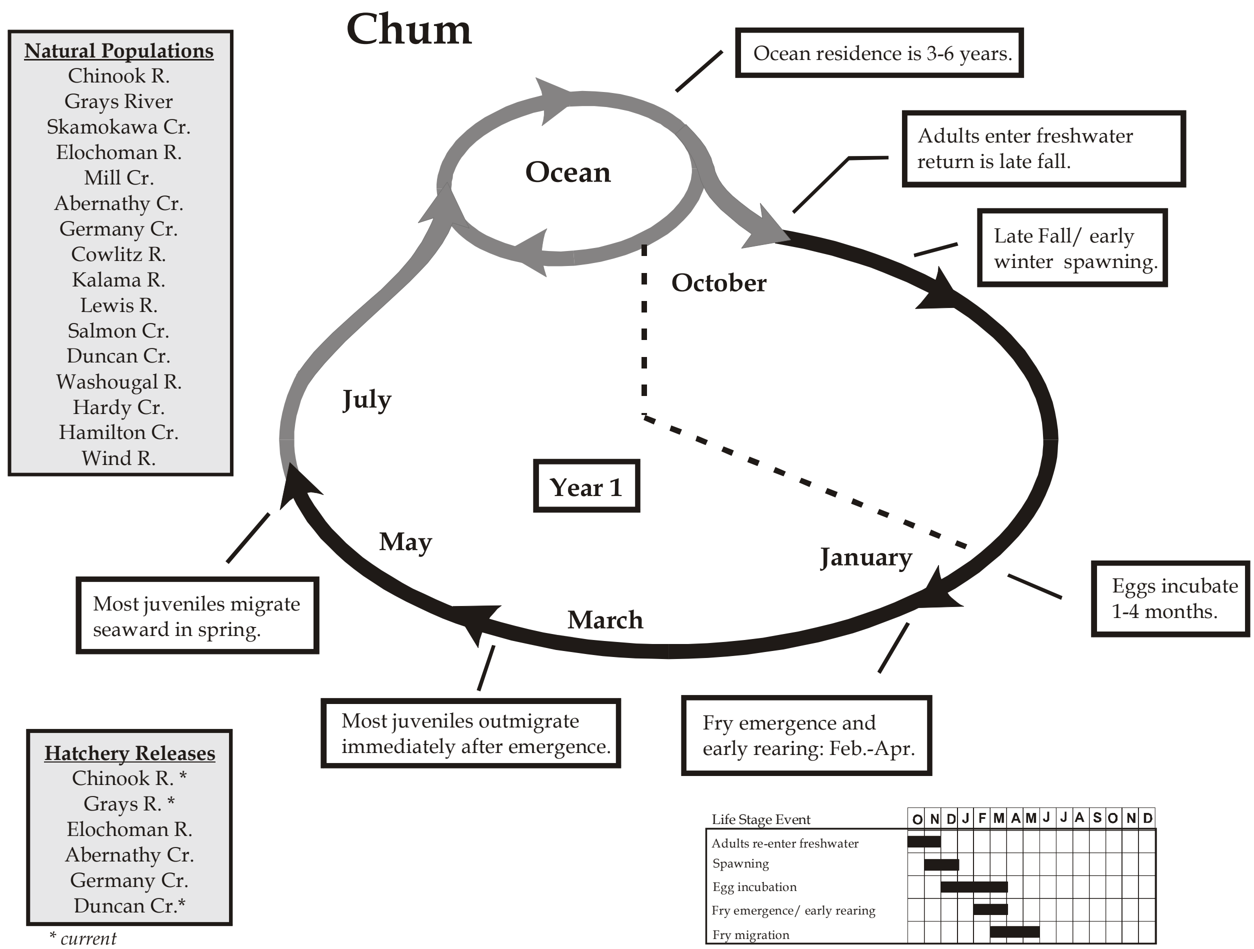

Figure 2-12. Chum salmon life cycle.

2.3.1 Life History and Requirements

The freshwater adult life history cycle of lower Columbia chum salmon populations follows the timing of seasonal changes in river flow and water temperatures in lower Columbia River tributaries (Figure 2-12). In general, the region has a mild climate with warm, relatively dry summers and cool, wet winters. The river environments these fish enter are characterized by relatively low elevations (1,640-3,280 ft [500-1,000 m]), with moderate amounts of precipitation (80-95 in/year [200-240 cm/year]). These rivers display relatively low flows during late summer and early fall, increased river flows and decreased water temperatures beginning in early October, and a single flow peak in December or January. Upstream migration of chum to tributary spawning areas often coincides with changes in streamflow and water temperature. In the lower Columbia River, streamflows typically begin to rise in October with the onset of fall rains. Water temperatures also drop at this time, creating conditions that favor salmon migration and spawning activity.

2.3.1.1 Upstream Migration Timing

Chum salmon returning to the Columbia River are considered a fall run. Adult fall run chum salmon return to the Columbia River from mid-October through November, but apparently do not reach the Grays River until late October-early December. Spawning occurs in the Grays River from early November to late December. Fish returning to Hamilton and Hardy creeks begin to appear in the tributaries in early November and their spawn timing is more protracted (mid-November to mid-January). Chum salmon have been reported in October in the Washougal, Lewis, Kalama, and Cowlitz rivers in Washington and in the Sandy River in Oregon (Salo 1991).

Chum seldom show persistence in surmounting river blockages and falls, which may be why they usually spawn in lower river reaches. However, in some river systems, such as Washington's Skagit River, chum salmon routinely migrate distances of at least 105 miles (170 km). They swim even greater distances in at least two other rivers. In Alaska's Yukon River and the Amur River in the Russian Federation, chum salmon migrate more than 1,550 miles (2,500 km) inland. Both of these rivers have low gradients and are without extensive falls or other blockages to migration. Chum salmon that historically traveled up the Columbia River to spawn in the Umatilla and Walla Walla rivers, however, would have had to pass Celilo Falls and a web of rapids and cascades. The falls would have presented a considerable obstacle and probably were passable by chum salmon only at high water flows.

2.3.1.2 Spawning

Chum salmon spawn primarily in the lower reaches of rivers, digging their redds in the mainstem, tributaries or in side channels of rivers from just above tidal influence to nearly 60 miles (100 km) from the sea. They spawn in shallower, slower-running streams and side channels more frequently than do other salmonids. However, literature on selection of spawning sites and redd characteristics for chum salmon (reviewed in Bakkala 1970, Smirnov 1975, Salo 1991), indicates that under specific circumstances chum salmon spawn in a variety of locations.

Water velocity in spawning areas varies widely for chum salmon. In Washington, Johnson et al. (1971) measured water velocities near 1,000 chum salmon redds and found that velocities where fish spawned varied from 0.0 to 5.5 ft/sec (0.0 to 167.6 cm/sec), and that over 80% of the fish spawned in velocities between 0.7 and 2.7 ft/sec (21.3 and 83.8 cm/sec). This range is similar to that found in other species of salmon. For example, velocities of streams where chinook salmon spawn are reported to range from 0.3 to 4.9 ft/sec (10 to 150 cm/sec). Johnson et al. (1971) also attempted to correlate abundance indices of chum salmon in Washington with environmental variables such as stream discharge, velocity, and surface water temperatures, but found no relationship between run size and these variables. He concluded that he was unable to measure or to isolate the critical areas in which environmental factors influence run size.

Chum salmon in other parts of the world also choose spawning grounds with a variety of water velocities; for example, fall chum salmon spawned in pools where the velocity was reported to be quite insignificant (Soin 1954, Smirnov 1975). Working on Japan's Hokkaido Island, Sano and Nagasawa (1958) also found that fall chum salmon selected spawning areas with lower water velocities (0.3-0.7 ft/sec [10-20 cm/sec]) than did summer chum salmon in the Amur River area. These differences in the physical characteristics of spawning areas may act to isolate populations or runs in the same river (Salo 1991).

In some locations, subgravel flow (upwelled groundwater) may be important in the choice of redd sites by chum salmon. A summary of available information on Far Eastern chum salmon reported that throughout the Russian Federation and on Hokkaido Island, fall chum salmon "*utilize mostly spring areas of upper tributaries, [as] damage by freezing and other* *severe winter conditions is relatively minor in most years*." (Sano 1966). However, Sano also notes, based on studies by Smirnov in the 1940s, "*summer chum salmon spawn earlier in the season, and they do not particularly choose spring areas*."

Many Columbia River chum have been found to select spawning sites in areas of upwelling groundwater. New spawning grounds for chum were recently discovered along the Washington shoreline near the I-205 Glen Jackson Bridge where groundwater upwelling occurs. A significant proportion of chum returning to Hamilton Creek spawn in a spring-fed channel, and portions of the Grays River and Hardy Creek populations spawn in the area of springs. Hundreds of chum salmon once returned to spawn within spring-fed areas along Duncan Creek; efforts have been completed to restore passage to these productive areas and protect the springs that feed them. Adult and juvenile chum salmon from the Ives Island population are being released into newly rehabilitated habitat in Duncan Creek.

2.3.1.3 Incubation and Emergence

One of the earliest detectable differences between chum salmon populations in different areas is the time it takes for eggs to incubate, hatch, and emerge as alevins from the gravel. Differences between populations are caused by physical factors such as stream flow, water temperature, dissolved oxygen, and gravel composition, and by such biotic factors as genetics, spawning time, and spawning density, all of which can affect survival (reviewed in Bakkala 1970, Salo 1991).

Water temperature is believed to have the most influence on the rate of embryonic development in chum salmon (reviewed in Bakkala 1970, Koski 1975, Salo 1991). The amount of heat, measured in TUs, required by fertilized chum salmon eggs to develop and hatch is about 400-600 TUs, and the heat required to complete yolk absorption is about 700-1,000 TUs. Lower water temperatures can prolong the time required from fertilization to hatching by 1.5–4.5 months. For example, fertilized eggs hatch in about 100–150 days (400-600 TUs) at 39°F (4°C), but hatch in only 26–40 days at 59°F (15°C). Each salmonid has an optimal temperature range that maximizes egg to fry survival. Schroder et al. (1974) reported significantly higher mortality of chum salmon eggs, alevins, and fry when early incubation temperatures were below 34.7°F (1.5°C). Upper thermal limits for chum salmon incubation have not been reported.

The time to hatching also varies among populations and among individuals within a population (Salo 1991). Koski (1975) found differences in the time to hatching between early and late-returning chum salmon at Big Beef Creek, a tributary to Hood Canal. For 2 years (1968–70), early-returning (peak September) and late-returning (peak late November or December) fish spawned and their offspring were reared in spawning channels in the creek. Fry emerged from February to June, but the timing of fry emergence differed between early- and late-returning fish by an average of 35 days each year. Early-run fish took longer to hatch, and this difference between the two runs was consistent from year to year. However, the longer hatching time of early-returning spawners led to fry with lower average weight and less lipid content than fry of late-returning spawners. Lower weight and fewer food reserves in early-return fry may decrease their chances of survival during early life history. The difference in incubation times for eggs from these early- and late-returning fish suggested a genetic difference between the two runs, and Koski (1975) concluded that natural selection apparently acted on hatching times: fry tended to emerge when they had their best chances of surviving in streams and estuaries.

Changes in hatching (incubation) times due to adaptation to cold water also have been found for chum salmon in the Susitna River, Alaska (Wangaard and Burger 1983) and in the Amur River (Disler 1954 cited in Bakkala 1970). At low incubation temperatures, these populations demonstrated faster embryonic development than embryos in other populations at the same temperature. In Canada, however, Beacham and Murray (1986) failed to find differences in hatching times among eggs from adults with early, middle, and late spawning times that had been incubated at constant temperatures of 39, 46, and 54°F (4, 8, and 12°C). Nevertheless, the time of emergence in that study depended on the timing of spawning: earlier-spawning fish laid larger eggs that took longer to develop than did smaller eggs from later-spawning fish.

Factors such as dissolved oxygen, gravel size, salinity, nutritional condition, and even the behavior of alevins in the gravel can influence the time to hatching and emergence from the gravel. For example, Fast and Stober (1984) found that developing chum salmon embryos in small coastal streams required less oxygen than had been reported for either coho salmon or steelhead, but it is unknown to what extent chum salmon in different areas vary in their oxygen requirements. The relative importance of various factors influencing early development in different populations has not been evaluated.

Despite a large amount of variability in incubation environments, even over short distances, chum salmon display a variety of developmental responses that result in similar emergence and outmigration times among fry within a specific area. Variability in some of these responses appears to reflect differences among individual fish, but it also reflects differences among populations in adult run and spawning times, egg size, and temperature-development requirements.

Chum do not have a clearly defined smolt stage, but are nonetheless capable of adapting to seawater soon after emerging from gravel. Chum salmon usually retain parr marks when they first enter seawater. In Japan, chum salmon fry weighing less than 0.06 oz (2 g) maintained normal levels of plasma sodium (Na^+) when they moved from fresh water into sea water (Iwata 1982). This ability, however, declines slightly with continued residence in fresh water. The capability of chum salmon fry for early osmoregulation in seawater may be important for adults homing back to natal streams. For example, hatchery chum salmon were 10 times less likely to stray within a river system if they were released into the river as fingerlings rather than as smolts (McHenry 1981 cited in Lister et al. 1981).

2.3.1.4 Freshwater Rearing

Chum salmon do not typically have substantial freshwater rearing time. Most chum juveniles begin seaward migration with minimal time spent in natal streams.

2.3.1.5 Juvenile Migration

Less is known about chum salmon downstream migrations than juveniles of other salmonids (Salo and Bayliff 1958, Beall 1972, Koski 1975, Seiler et al. 1981, and reviewed in Salo 1991) because chum salmon outmigrants:

1. are smaller than outmigrants of other salmonids,
2. migrate at night,
3. usually have shorter distances to migrate to reach salt water than do other species, and

4. do not school as tightly as some other salmonids (e.g., pink and sockeye fry).

Nonetheless, several key facets of fry outmigration are known. Downstream migration may take only a few hours or days in rivers where spawning sites are close to the mouth of the river, or it may take several months, as in the Yukon and Amur rivers, where spawning sites are located hundreds of kilometers upriver. The timing of outmigration is usually associated with increasing day length, warming of estuarine waters, and high densities of plankton (Walters et al. 1978). Juvenile chum salmon at southern localities, such as those in Washington and southern British Columbia, migrate downstream earlier (late January through May) than do fry in northern British Columbia and southeastern Alaska (April to June).

Several factors influence the timing of downstream migration, resulting in considerable variability in migration timing throughout the species range. These factors include time of adult spawning, stream temperatures during egg incubation and after hatching, fry size and nutritional condition, population density, food availability, stream discharge volume and turbidity, physiological changes in the fry, tidal cycles, and day length (Simenstad et al. 1982, Salo 1991). In the Russian Federation, Soldatov (1912 cited in Smirnov 1975) found that chum salmon outmigrations did not always immediately follow emergence; juveniles in many rivers remained up to 4 months in the river and grew to a considerable size before outmigration (Kostarev 1970 as cited in Salo 1991). In Washington, chum may reside in fresh water for as long as a month (Salo and Noble 1953, Bostick 1955, Beall 1972). Juveniles have been found to reside in fresh water for more than a month in the mainstems of the Skagit (Dames and Moore 1976) and Nooksack (Tyler 1964) rivers.

Because chum fry generally emigrate shortly after emergence, predation mortality during downstream emigration can be significant. Coho juveniles, resident trout, and cottids have been implicated as the primary predators, however, the species composition in each system plays a significant role. The estimated mean freshwater mortality as a result of predation ranges from 22% to 58%. In general, predation on smaller chum fry is thought to be high and predation decreases as chum fry size increases (Beall 1972, Hiyama et al. 1972). To compensate for this predation mortality, chum fry form schools (Pitcher 1986) and synchronize their movements (Miller and Brannon 1982). Historical information concerning the timing of chum salmon emigration in the lower Columbia River is limited. One existing report, however, describes emerging fry outmigrating past the Mayfield Dam site on the Cowlitz River in March and May 1955 and 1956. Thompson and Rothfus (1969) reported the passage at the Mayfield Dam site of approximately 137,250 chum outmigrants past the site between March–May 1955, and about 8,200 fry during the same period in 1956. A wild chinook capture and tag project conducted by Washington Department of Fish and Wildlife (WDFW) in the North Lewis River during 1977–79 showed incidental capture of chum fry peaking in April and not present in the catch after mid-May. In recent years, seining projects conducted by WDFW in the Grays River and at Ives Island by WDFW and the Oregon Department of Fish and Wildlife (ODFW) indicate outmigration occurs from March through May and peaks from mid-April to early May. Similar activities are being conducted in Deep River to assure release of hatchery fish from net pens is timed to minimize predation.

2.3.1.6 Estuary Rearing and Growth

The period of estuarine residence appears to be the most critical phase in the life history of chum salmon and may play a major role in determining the size of the subsequent adult run back to fresh water (Mazer and Shepard 1962, Bakkala 1970, Mathews and Senn 1975, Fraser et al. 1978, Peterman 1978, Sakuramoto and Yamada 1980, Martin et al. 1986, Healey 1982, Bax 1983a, Salo 1991). Chum salmon juveniles, like other anadromous salmonids, use estuaries to feed before beginning long-distance oceanic migrations. However, chum and ocean-type chinook salmon usually have longer residence times in estuaries than do other anadromous salmonids (Dorcey et al. 1978, Healey 1982). Bax (1983b) determined that the extent of juvenile mortality within 4 days of a hatchery release into the Hood Canal estuary was 31-46%. The most important determinant of estuarine survival may be the timing of entry into salt water because plankton abundance in estuaries is highly seasonal (Gunsolus 1978, Helle 1979, Gallagher 1979, Simenstad and Salo 1982).

Because chum salmon spend more time in the estuary, they are more susceptible to changes in the productivity of that environment than stream-type salmonids. Estuaries may be ‘overgrazed’ when large numbers of ocean-type juveniles enter the estuary *en masse* (Reimers 1973, Healey 1991). The loss of coastal wetlands to urban or agricultural development may more directly affect ocean-type populations than stream-type populations. For example, Thomas (1983) and Johnson et al. (2003b) have documented substantial loss of marsh and swamp habitat throughout the estuary and the lower Columbia River mainstem; further, many researchers (Healey 1982, Levy and Northcote 1982, Myers and Horton 1982, Simenstad et al. 1982, Levings et al. 1986, Bottom et al. 1984) have documented that small juvenile salmonids usually occupy shallow, protected habitats such as salt marshes, tidal creeks, and intertidal flats.

Chum salmon juveniles of early-returning adults tend to enter estuaries before juveniles of late-returning fish (Koski 1975). Because the juvenile emigration timing of lower Columbia River chum salmon from the natal streams is generally from March to May with peak migration in April, chum salmon likely begin arriving in the Columbia River estuary in April. Juvenile chum salmon were a minor portion of the catch during Columbia River estuary sampling efforts of Bottom et al. (1984); chum, sockeye, and cutthroat collectively represented 1% of the total juvenile salmonid catch. Chum salmon juveniles were captured in the estuary during April and May during both years of the study; chum salmon were present in the estuary from February through June (Bottom et al. 1984). Juvenile chum salmon were primarily distributed within the freshwater or estuarine regions of the estuary, although there was one occurrence in the marine region (Bottom et al. 1984).

Residence times are known for only a few estuaries, even though residence timing has been studied since the 1940s (reviewed in Congleton 1979, Healey 1982, Simenstad et al. 1982, Bax 1983a). Observed residence times range from 4 to 32 days, with a period of about 24 days being the most common.

Migration patterns of juvenile chum salmon have been studied intensively in areas such as Hood Canal by following marked juveniles from hatchery populations of fall-run chum salmon and by monitoring outmigration (Bax 1982, 1983a, b; Bax et al. 1979, 1980; Bax and Whitmus 1981; Schreiner 1977; Whitmus and Olsen 1979; Whitmus 1985; Salo et al. 1980). Some fry remain near the mouth of their natal river when they enter an estuary, but most disperse within a few hours into tidal creeks and sloughs up to several kilometers from the mouth of their natal river. Movements of chum salmon fry in Hood Canal generally appear to follow a pattern that depends on the time of release from hatcheries, however, release time is not the only factor influencing migratory patterns (Bax 1982, 1983a). Chum salmon fry released into Hood Canal in early February and March spread out over a large area, but fish released in April and early May tended to remain inshore initially, moving offshore in summer. These movements were apparently associated with prey availability. Fish initially fed inshore on epibenthic organisms, then offshore on plankton later in the season.

In the Nanaimo and Fraser River estuaries, juveniles spend up to 3 weeks feeding in the inner estuary, with little local movement (Healey 1979, Levy et al. 1979). Chum salmon juveniles in the Nanaimo, Yaquina, Cowichan, and Courtenay estuaries are most abundant in nearshore areas during April and May, but are most abundant in the outer estuary during May and June (Myers 1980, Healey 1982). Chum salmon fry show daily tidal migrations in the Fraser and Nanaimo rivers, which have large deltas and marshlands (Healey 1982). However, fry in Hood Canal have not been observed to display daily tidal migrations (Bax 1983a), most likely because rivers entering Hood Canal do not have extensive delta or tidal marsh systems (with the exceptions of the Quilcene and Skokomish rivers).

Chum salmon spend more of their life history in marine waters than other Pacific salmonids. Chum salmon, like pink salmon, usually spawn in coastal areas, and juveniles outmigrate to seawater shortly after emerging from the gravel (Salo 1991). This ocean-type migratory behavior contrasts with the stream-type behavior of some other species in the genus *Oncorhynchus* (e.g., coastal cutthroat trout, steelhead, coho salmon, and most types of chinook and sockeye salmon), which usually migrate to sea at a larger size, after months or years of freshwater rearing. Again unlike stream-type salmonids, survival and growth in juvenile chum salmon depend less on freshwater conditions than on favorable estuarine conditions, except for those chum salmon that undergo lengthy migrations such as in the Yukon and Amur rivers.

Recent sampling of juvenile salmonids in the Columbia River plume has started to illustrate patterns of habitat use by salmonids in the plume and nearshore ocean habitats (Fresh et al. 2003), although limited years of data are currently available. For example, preliminary evidence suggests that some juvenile salmonids (chum, steelhead, and yearling coho) may preferentially utilize the plume front compared to other areas in the plume or adjacent ocean habitats (Fresh et al. 2003). Although reasons for the apparent preference to the plume front are not clear, this area may be a more productive habitat than elsewhere in the plume and adjacent ocean.

2.3.1.7 Ocean Migrations

Little is known about the seaward migration of juvenile chum salmon from the Columbia River. Generally, however, migration of chum salmon juveniles out of estuaries appears to be closely correlated with prey availability. Chum salmon move offshore as they reach a size that allows them to feed on the larger neritic plankton, and this movement normally occurs as inshore prey resources decline (Salo 1991). This transition has taken place at 1.75 in (45 mm) fork length (FL) in Puget Sound and Hood Canal, Washington, but at 2.33 in (60 mm) FL in Prince William Sound, Alaska (Cooney et al. 1978).

Studies have shown that chum salmon in Puget Sound, Washington, and southern British Columbia generally entered the ocean earlier than did more northern and western populations (Hartt 1980, Hartt and Dell 1986). Hartt (1980) and Hartt and Dell (1986) summarized available data on the distribution, migration, and growth of chum salmon in their first year at sea and found that chum, pink, and sockeye salmon juveniles tended to group together and remained nearer shore (within 22 miles [36 km]) than juvenile coho and chinook salmon and steelhead. As groups of chum salmon reached Alaska, they moved offshore in a generally southwestern direction, although movement was variable and appeared to be strongly influenced by currents

(Hartt 1980, Hartt and Dell 1986). A difficulty in these studies is that few numbers of tagged fish were recovered. In the tag recovery information summarized by Hartt and Dell, over 110,000 juvenile salmon and steelhead were caught and 35,259 tagged, of which 4,412 were chum salmon, although only 6 tagged chum salmon (0.1%) were recovered.

A second factor that obscures patterns of oceanic distribution and migration is the extent of delayed ocean migrations and residualism by chum salmon. In the tagging studies by Jensen (1956), juvenile chum salmon remained in nearshore waters beyond the usual time of ocean migration, although the extent of this residualism was unclear (Jensen 1956, Hartt 1980, Fresh et al. 1980, Hartt and Dell 1986). Not all of the chum salmon juveniles tagged in Hood Canal and Puget Sound moved northward toward British Columbia; some remained in Puget Sound throughout the summer, perhaps not leaving until the next spring (Jensen 1956). In November, Hartt and Dell (1986) found juvenile chum salmon in central Puget Sound and in Hecate Strait that averaged 9 in (230 mm) in length, an indication of good growth. It has been hypothesized that these fish may not make an extended northwest migration along the British Columbia/Alaska coast, but may instead proceed directly offshore into the North Pacific Ocean (Hartt and Dell 1986).

The International North Pacific Fisheries Commission (INPFC) has collected a large amount of information on the distribution and origins of high-seas chum salmon. These tagging and scale studies by the INPFC show that although chum salmon from both Asia and North America are distributed throughout the North Pacific Ocean and Bering Sea, Asian chum salmon apparently migrate farther across the Pacific Ocean than do North American fish. Neave et al. (1976) reported that North American chum salmon were rarely found west of the mid-Pacific Ocean beyond long. 175°E, while Asian chum salmon were often found far east of this line. Asian chum salmon have extended their distribution in recent years into the central and eastern North Pacific Ocean, perhaps because of the large increase in releases of hatchery fish in Japan (Kaeriyama 1989, Salo 1991), and because of the change from high-sea to inshore fisheries by Japan’s fishing industry (Kaeriyama 1989, Ogura and Ito 1994). Bigler and Helle (1994) and Helle and Hoffman (1995) suggested that the overlap of continental groups may be detrimental to North American chum salmon because maturing chum salmon in the North Pacific Ocean may be at or above carrying capacity.

Limited information exists on stock- or population-specific migration patterns and ocean distributions of chum salmon. Maturing chum salmon in the North Pacific begin to move coastward in May and June and enter coastal waters from June to November (Neave et al. 1976, Fredin et al. 1977, Hartt 1980). No region-specific information on chum salmon migrations to Washington and Oregon has been reported. Whether the large populations of chum salmon that once inhabited the Columbia River (Rich 1942) had oceanic distributions similar to Puget Sound chum salmon is unknown. As landings in coastal Oregon historically excluded landings on the Oregon side of the Columbia River (Henry 1953), these fish may have had a more southern distribution, like the present distribution of Columbia River coho salmon (Sandercock 1991), and may have returned northward along the Oregon coast.

2.3.2 Distribution

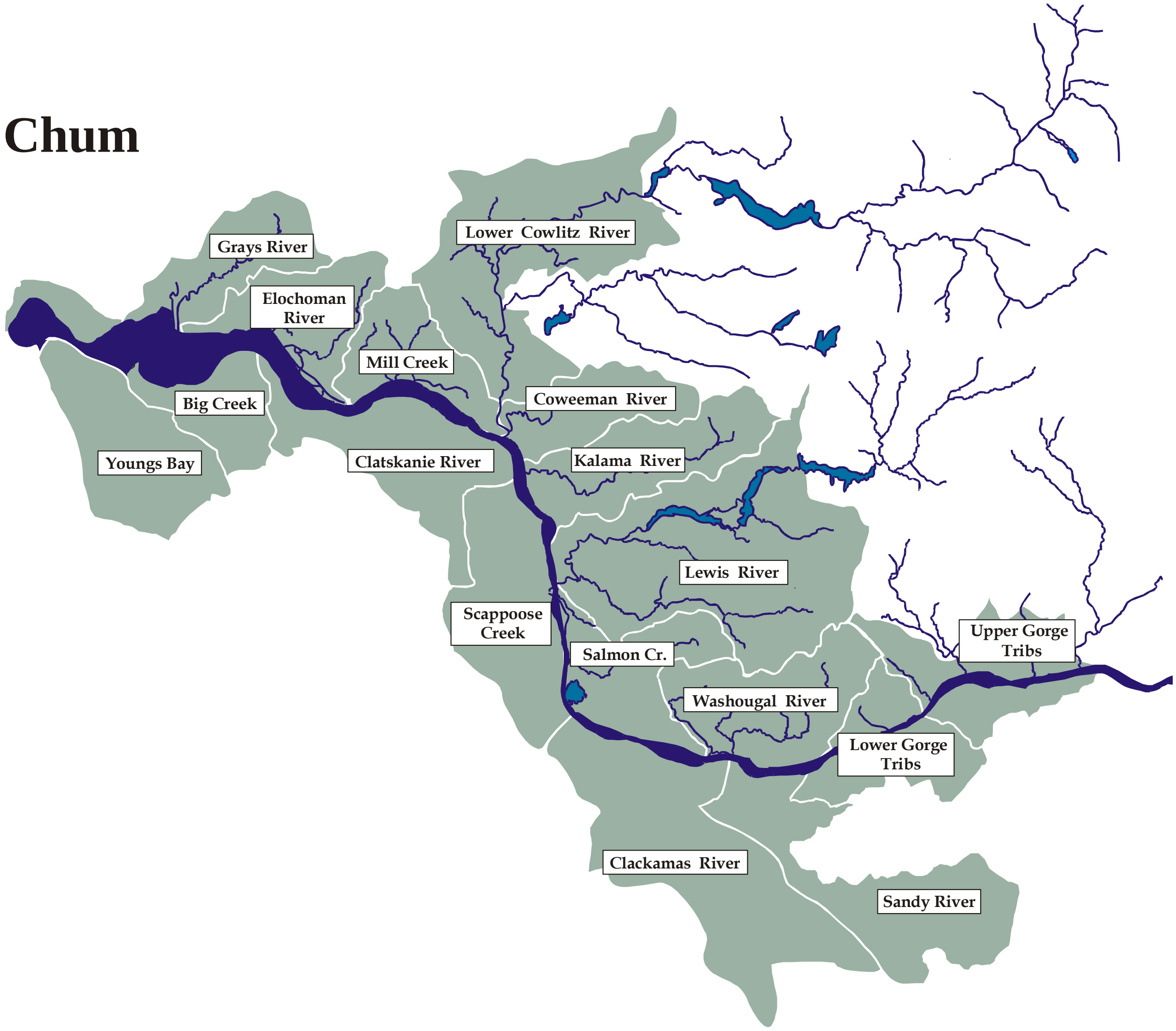

Figure 2-13. Historical demographically independent chum salmon populations in the lower Columbia River ESU (Myers et al. 2002).

Chum salmon spawn from Korea and the Japanese island of Honshu, east around the rim of the North Pacific Ocean, to the Columbia River. In the Arctic Ocean, they range from the Laptev Sea in the Russian Federation to the Mackenzie River in Canada.

Chum once were widely distributed in Columbia River tributaries below Celilo Falls (Figure 2-13). Some chum historically passed over Celilo Falls on the Columbia River to spawn in the Umatilla and Walla Walla rivers (Nehlsen et al. 1991).

The size and distribution of the Columbia River chum population dropped dramatically in the 1950s (National Marine Fisheries Service [NMFS] 1997), though the runs were still high compared to today. Estimates of chum escapement in 1951 included 3,000 fish to the Lewis River; 1,200 fish to the Chinook and Deep rivers and Crooked and Jim Crow creeks; 7,500 to Grays River; 3,000 to Skamokawa Creek; 1,000 to the Elochoman River; 1,000 to the Cowlitz River; 600 to the Kalama River; 1,000 to the Washougal River; and 2,700 chum to the Abernathy/Mill/Germany Creek area. These escapement estimates document that natural populations of chum salmon were present in basins throughout the lower Columbia River as recently as the 1950s.

Chum populations in many lower Washington tributaries continued declining into the 1970s. Between 1961–66, the Mayfield fish-passage facility on the Cowlitz River reported collecting only two adult chum (Thompson and Rothfus, 1969). Now, fewer than ten adults are usually collected each year at the Cowlitz Salmon Hatchery (Harza 1999). Chum populations also declined in the Lewis and Kalama rivers. In 1973, WDFW estimated the spawning population in the Lewis and Kalama basins as only a few hundred fish. According to a 1973 report the most dense observed chum spawning observed occurred in side channels and upwelling areas in the lower 6 miles (9.7 km) of the EF Lewis River (WDFW 1973). These declining chum populations illustrate that chum salmon distribution within the lower Columbia River was becoming localized by the 1970s; only a small portion of the historical lower Columbia River distribution possessed natural chum salmon populations.

Near Bonneville Dam, chum salmon return to Hardy and Hamilton creeks and to the lower reaches of Lawton, Good Bear and Duncan creeks. Chum salmon spawn primarily in the lower reaches of Hardy and Hamilton creeks. Annual escapement to these streams near Bonneville Dam averaged about 1,000 fish from 1967–1971. Bryant (1949) noted that a few chum spawned near the mouth of Woodward Creek in 1944. WDFW (1951) reported that chum use the lower portion of Gibbons, Walton, St. Cloud, Duncan, Woodward, Hardy, and Hamilton creeks.

Small numbers of chum salmon also return to other historical spawning tributaries in the lower Columbia River. Aside from the Grays River and Hamilton and Hardy creeks, chum salmon have been observed in Cowlitz, Lewis, Elochoman, Kalama, and Washougal rivers, and in Skamokowa, Germany, and Abernathy creeks. Biologists have monitored chum salmon populations in several of these river systems since 1998 and report that the populations remain extremely low (Uusitalo 2001). Monitoring was expanded in 2000-2002 to include repeat surveys in over 60 tributary streams. Significant spawning populations have been monitored for several years in the mainstem Columbia near Ives Island and Multnomah Falls in the lower Gorge and more recently chum spawning has been monitored in the mainstem Columbia at several spring seeps along the Washington shore near the I-205 Bridge. Some chum salmon may also return to areas above Bonneville Dam. In 1998 and 1999, about 195 and 135 chum salmon, respectively, were observed ascending the fish ladder at the dam (Keller 2001, NMFS 2000).

2.3.3 Genetic Diversity

While many streams in the lower Columbia River support small populations of chum salmon, large enough numbers to conduct a meaningful allozyme analysis have only been found in two regions, Grays River and just downstream of Bonneville Dam (Hamilton and Hardy creeks). Since 1992, collections of several hundred spawning adults have been made from these sites. Spawning has been observed recently in the mainstem Columbia River at the Pierce/Ives Island complex and in seep areas on the Washington shoreline near the I-205 Bridge. Chum adults and juveniles from these mainstem areas have been intermittently collected for genetic analysis from 1998 to 2001. Additionally, small numbers of chum also were collected in the Chinook and Cowlitz rivers in 2000.

The genetic analysis clearly separated the samples of spawning chum into three groups: the Grays River, the below-Bonneville area (Hamilton and Hardy creeks, Ives Island, and the I-205 seeps) and the Chinook River (Sea Resources Hatchery origin). When sampling occurred, the Sea Resources Hatchery was propagating a non-Columbia chum stock from Southwest Washington, but has since switched to Grays River stock. The maximum p-value between the Grays River collection and any other collection was 0.0001, showing good separation between this population and all others. There also were several high p-value comparisons among the Hardy and Hamilton Creek collections; however, there was no clear distinction among the below-Bonneville (F-test analysis) collections.

Statistical analysis results indicate that the Grays River and below-Bonneville populations are reproductively isolated to a large extent, but that there is no such evidence for isolation among the below-Bonneville areas. Similarities between the collections from the I-205 seeps and the more upstream collections likely indicate opportunistic colonization of a new area. Thus, there appear to be two Columbia chum groups: Grays River and below-Bonneville mainstem and tributaries, which agree with the GDU designations of Phelps et al. (1995).

The genetic samples also showed no apparent differences in age structure of the three aggregations, with 3-year old fish predominating (Keller 2001). These findings resembled findings from scale analysis for chum salmon returning to the Columbia River in 1914, which also indicated that 3-year old fish constituted the majority of the run, 70.4% (Marr 1943). Although scale samples from more recent returns in 2001 and 2002 show a relatively even split between age 3 and age 4 spawning chum (Figure 2-14).

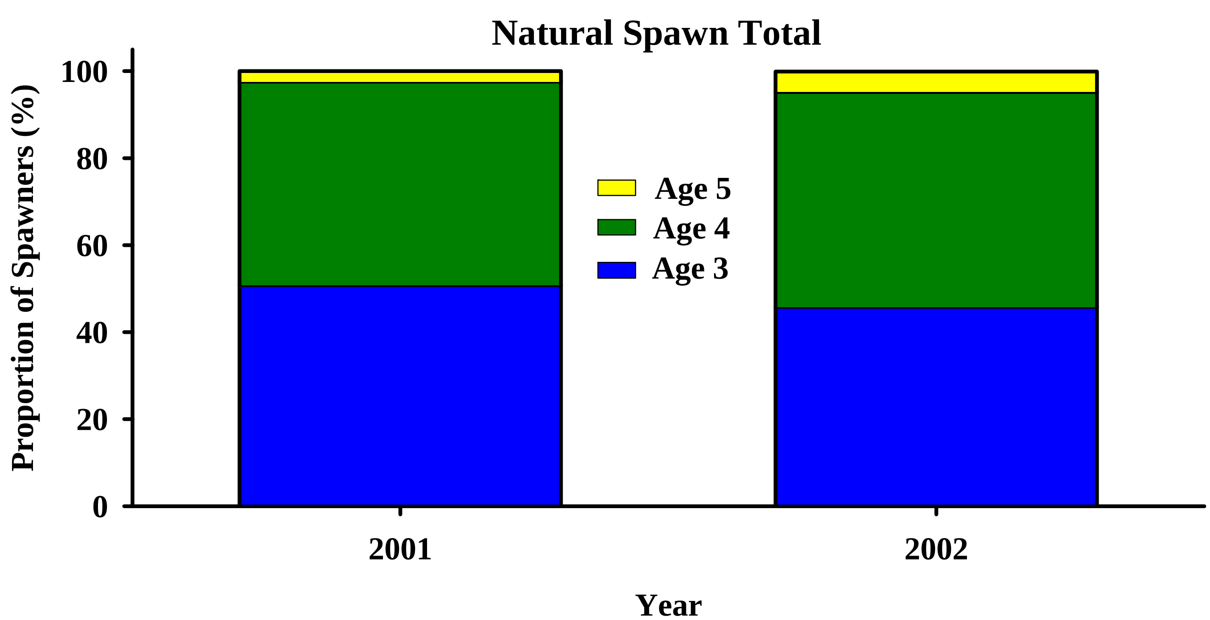

Figure 2-14. Age composition of natural spawning chum salmon.

2.3.4 ESU Definition

During the proposed listing process for chum salmon in the Pacific Northwest, NMFS (now NOAA Fisheries) received comments stating that chum salmon in the region represent one ESU (Fed. Reg., V64, N57, March 25, 1999, p. 14509). However, the NMFS Biological Review Team felt justified in separating Pacific Northwest chum salmon into 4 ESUs: Puget Sound/ Straight of Georgia, Hood Canal Summer-Run, Pacific Coast, and Columbia River. NMFS defined the Columbia River chum salmon ESU as including all naturally spawning populations in the Columbia River and its tributaries in Washington and Oregon (Fed. Reg., V64, N57, March 25, 1999, p. 14508). More recently, the specific historical populations have been identified by Myers et al. (2003) (Figure 2-13).

2.3.5 Life History Diversity

Runs to the Grays River, Hamilton Creek, and Hardy Creek return to the Columbia River basin in October and November (the peak is mid-November). This run time resembles that of chum salmon in rivers along the Washington coast (WDF et al. 1993). Small differences, however, do exist in the timing of spawning for these lower Columbia River populations. Barin (1886) observed that dog salmon (chum) appeared in the Clackamas River by November and spawned soon after. Peak spawning activity for chum salmon in the Grays River and Hamilton and Hardy creeks differs by about a month (November 8 and December 8 or 19, respectively), providing considerable geographic and temporal isolation (Keller 2001). The different spawn timing for these populations suggest that there are likely differences in time of emergence for these chum populations; however, time of emergence data are not available for naturally produced Grays River chum salmon. The spawn timing differences among these naturally producing chum populations supports the genetic analysis that suggests that the Grays River and below Bonneville area (Hamilton and Hardy creeks) are different stocks. Because of limited research focused on chum salmon in the lower Columbia, no other life history differences have been documented among chum salmon populations.

2.3.6 Abundance

The historical chum run size in the Columbia River has been estimated at nearly 1.4 million fish per year. Annual escapements to Washington waters of the lower Columbia mainstem and tributaries declined to an average of 3,000 after 1955 (WDFW 2001). The chum returns remained relatively stable at low levels from 1956-2000, but there were significant increases in returns to Washington waters during 2001-2002 as indicated in index area peak counts in Grays River, Hardy Creek, and Hamilton Creek areas (Figure 2-15).

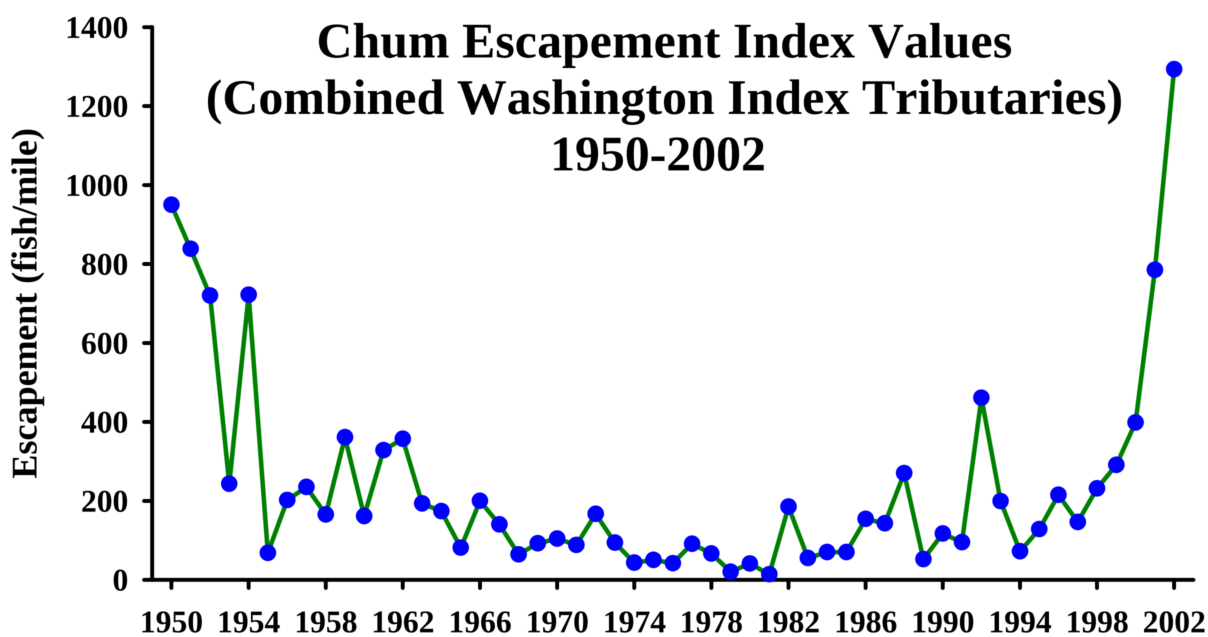

Figure 2-15. Chum escapement fish/mile peak counts for combined Washington index tributaries.

Today, chum salmon are limited almost exclusively to habitats downstream of Bonneville Dam, with the majority of spawning occurring on the Washington side of the Columbia River. Chum spawning returns to the Grays River and Hamilton Creek have been monitored annually since 1944 and returns to Hardy Creek since 1957. Chum spawning in the mainstem Columbia River near the Pierce/Ives Island complex, Multnomah Falls, and I-205 Bridge has been monitored in recent years, and mainstem Columbia River chum spawning population estimates have been made using multiple methods (Rawding and Hillson 2003, Van der Naald et al. 2003, WDFW 2003). Chum salmon also return to spawn in several other lower Columbia tributaries, however most past years' chum surveys have not included tributaries other then the index streams. Beginning in 2000, BPA funded the expansion of WDFW chum surveys to include lower Columbia tributaries in addition to the index stream surveys.

The Grays River Index chum monitoring areas include the mainstem Grays River, West Fork Grays River, and Crazy Johnson Creek. Grays River chum are considered depressed by WDFW due to chronically low spawning escapement (WDFW 2002). Average fish-per-mile values in the survey indices show a sharp decline in spawning escapement beginning in about 1955, with an increase beginning in 2001. Average fish-per-mile values from 1955-2000 ranged from a low of 6 fish in 1958 to a high of 521 in 1959 (WDF et al. 1993). The past two years have exceeded the 1959 count with 759 in 2001 and 1,587 in 2002 (Figure 2-16).

The Bonneville chum monitoring index area includes Hardy Creek, Hamilton Creek, and man-made Hamilton Spring Channel. The Bonneville area tributary chum population is considered depressed due to chronically low spawning escapements (WDFW 2002). Average fish per mile in the Bonneville area survey indices also display a sharp decline beginning in 1955 (based on Hamilton Creek counts). During 1955-2000, Hamilton Creek counts have ranged from a low of 4 in 1979 to a high of 892 in 1963. The 2001 and 2002 fish/mile counts in Hamilton Creek improved to 987 and 888 respectively (Figure 2-16). During 1957-2000 the Hardy Creek counts ranged from a low of 1 in 1979 to a high of 636 in 1992. The 2001 and 2002 fish/mile counts in Hardy Creek improved to 711 and 416, respectively (Figure 2-16).

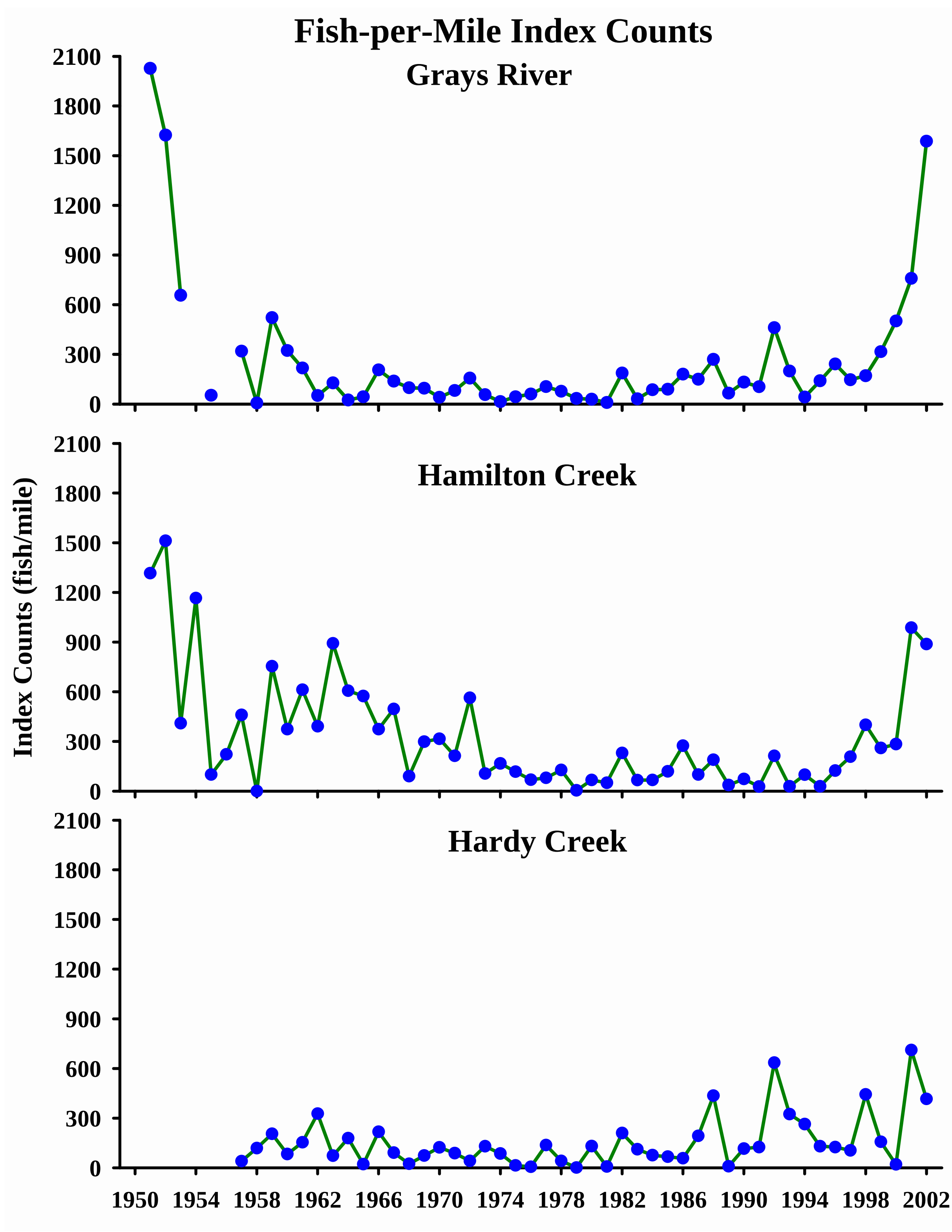

Figure 2-16. Fish-per-mile index area counts in Grays River, Hamilton Creek, and Hardy Creek.

Monitoring of lower Columbia chum returns has expanded in recent years to include mainstem spawners and over 60 non-index stream tributaries since. The lower Gorge area includes spawning in the mainstem Columbia near Ives Island and near Multnomah Falls. Mainstem Columbia spawning also occurs further downstream near I-205 Bridge near the Washington shore. Abundance estimates, distribution, and genetic research of mainstem spawners is currently a focus of federal and state agencies in an effort to determine if they are an independent population. The Bonneville mainstem and tributary spawning populations are also components of a reintroduction and low flow salvage plan. An effort is underway to reintroduce chum to Duncan Creek, just downstream of Hardy Creek, and also to 'salvage' adult chum in the mainstem Columbia and in the lower reaches of Hamilton and Hardy creeks during years when flows are too low for fish to spawn successfully or to access the spring-seep spawning areas.

Non-index area tributary monitoring includes chum spawning counts in watersheds from the Big White Salmon River downstream to the Chinook River. In 2002, WDFW used the spawning survey data to estimate chum spawning populations for index and non-index areas of the lower Columbia mainstem and Washington tributaries. The total spawning population estimate was 19,403, including 13,850 in index areas and 5,553 in non-index areas (Figure 2-17). When including the commercial fishery catch of 14 chum salmon, hatchery escapement of 309, and a Bonneville Dam count of 188 chum, the total minimum chum return to the Columbia River in 2002, excluding Oregon tributary spawning, which is considered to be low, is estimated to be 19,914 fish.

The vast majority of 2002 chum spawning occurred in the Grays River and lower gorge tributaries, and in the mainstem Columbia between I-205 Bridge and Bonneville Dam. However, notable spawning occurred in the Washougal, Lewis, and Chinook river basins, and in Skamokawa, Germany, and Abernathy creeks. The improved chum returns in the past two years has provided a unique opportunity to assess abundance (and presence and absence) in an expanded area of the Lower Columbia basin. This information can assist in discovery of areas which have maintained some capacity to produce chum under current conditions.

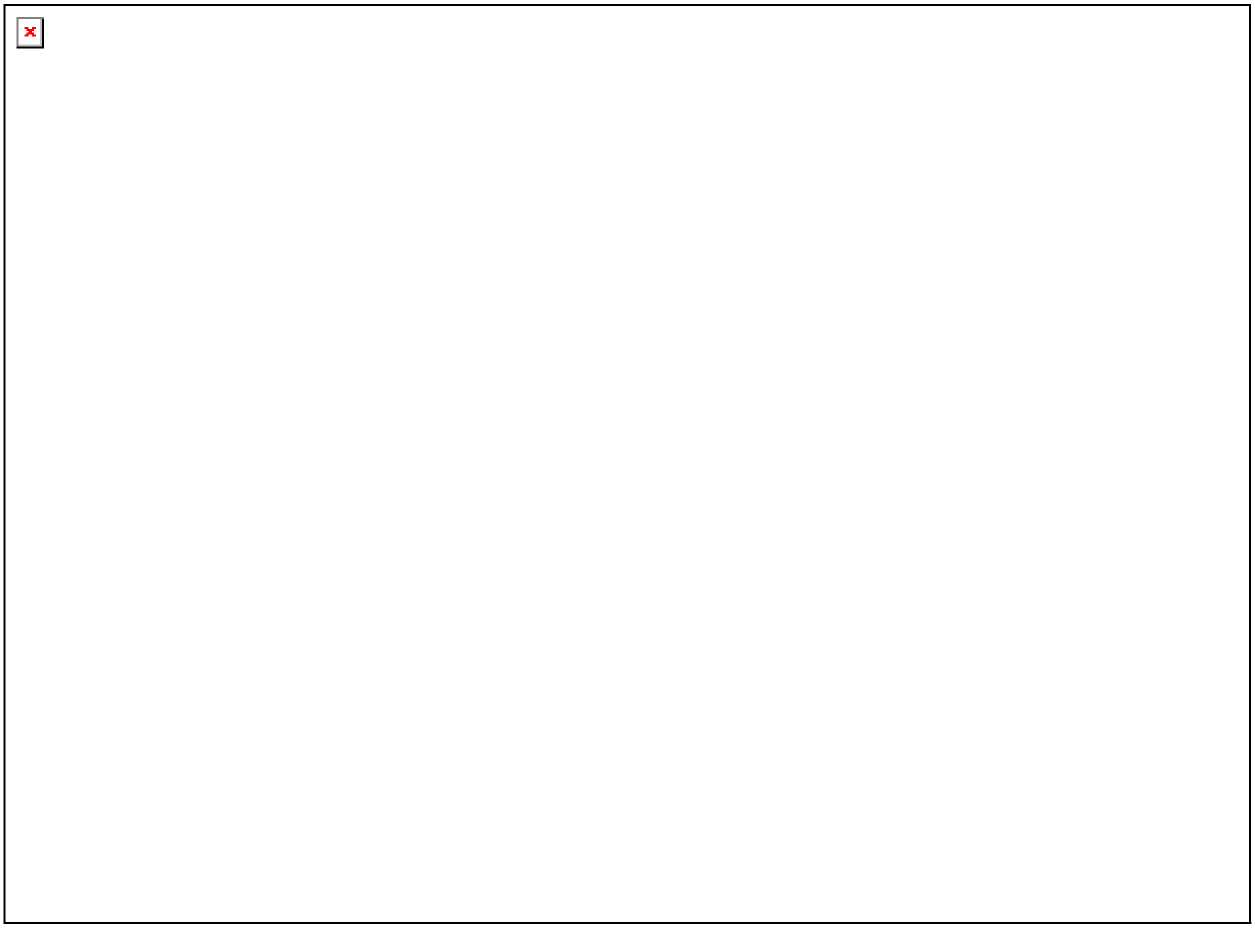

Figure 2-17. Distribution of Lower Columbia chum spawning populations in 2002.

2.3.7 Productivity

Little is known about the chum salmon production potential of subbasins in the lower Columbia River. Historically, many lower Columbia subbasins were capable of producing chum salmon runs in the thousands. Most watersheds have been negatively affected to some degree by human activity. The primary causes of habitat degradation, and hence salmon productivity, include urban development, dam construction and operation, channelization, riprapping, and timber harvest. For example, in the Cowlitz and Lewis rivers, dam construction has blocked chum salmon access to the majority of the productive habitat within the basin; the upper sections of these basins possess most of the productive salmon habitat. In the Grays River basin, the largest producer of chum salmon, habitat productivity and stability have been reduced as a result of logging road construction, timber harvest, and dike construction in the lower river. The productivity of the Grays River population was also reduced by the loss of Gorley Springs spawning channel that was destroyed in a 1998 flood. Mainstem Columbia chum productivity in the Ives and Pierce islands area is effected by flow operations at Bonneville Dam.

Chum salmon fecundity data are variable. In North America, literature-reported individual fecundity ranged from 2,018 to 3,977 eggs per female. No fecundity data are available for wild chum salmon in the lower Columbia River.

Chum salmon production is affected by the differential losses chum salmon experience during each stage of their life history; the magnitude of these losses varies geographically and temporally and is a reflection of complex interactions between biota and environment. Reported average egg to fry survival in natural streams can be quite variable. For example, Levanidov (1964) and Beacham and Starr (1982) reported egg to fry survival ranging from 6.1-14.2%. Meanwhile, Bakkala (1970) reported annual chum salmon egg to fry survival ranged from 0.1-34.4% and multi-year means ranged from 1.5-27.6%. In controlled stream environments, such as the spawning channel in Abernathy Creek, mean egg to fry survival can be as high as 82.1% (Bakkala 1970). Reported values for mean fry to adult survival range from 0.8-2.8% (Parker 1962, Wolcott 1978). Most mortality suffered by chum salmon in the marine environment occurs within the first few months. Fishing mortality further reduces the number of adults escaping to natal streams to spawn; however, for lower Columbia River chum salmon, fishing mortality has been a minor factor limiting production in recent years (see Chapter 3). Harvest in the lower Columbia River mainstem has been <100 chum/year since 1992. Retention of chum in tributary recreational fisheries is prohibited.

2.3.8 Listing Status

The BRT established by NMFS to examine the status of chum, concluded that the Columbia River ESU is presently at significant risk. The BRT believes the current abundance is probably only 1% of historical levels and the ESU has undoubtedly lost some (perhaps much) of its original genetic diversity.

The NMFS chum status review goes on to state:

Although the current abundance is only a fraction of historical levels, and much of the original inter-populational diversity has presumably been lost, the total spawning run of chum salmon to the Columbia River has been relatively stable since the mid 1950s and total natural escapement for the ESU is probably at least several thousand per year. Taking all of these factors into consideration, about half of the BRT members concluded that this ESU is at significant risk of extinction; the remainder concluded that the short-term extinction risk was not as high, but that the ESU is at risk of becoming endangered.

Lower Columbia chum salmon, including all naturally spawning populations in the Columbia and its tributaries in Washington and Oregon, were officially listed as threatened on March 25, 1999 (Fed. Reg., V64, N57, p. 14508).

2.4 Steelhead *(Oncorhynchus mykiss)*

Along with cutthroat trout, steelhead/rainbow trout (*Oncorhynchus mykiss*) has the greatest diversity of life history patterns of any Pacific salmonid. Variability in degrees of anadromy and plasticity of life history types between generations is common. Generally, anadromous forms of *O. mykiss* are referred to as steelhead, and resident forms are known as rainbow trout, though some interior subspecies of resident fish are known as redband trout (Behnke 2002). In coastal areas, anadromous and resident forms usually do not co-occur; they are most often separated by a natural or man-made barrier to migration (Busby 1996). Where anadromous and resident forms do co-occur, it is possible for the progeny of individuals of one life history form to exhibit a different life history form (Mullan et al. 1992, Busby 1996). Shapovalov and Taft (1954) reported *O. mykiss* maturing in fresh water and spawning before their first ocean migration, a life history variation that is also reported in cutthroat trout.

In the lower Columbia basin, migrating adult steelhead can occur in the Columbia River year-round, but peaks in migratory activity and differences in reproductive ecotype lend themselves to classifying steelhead into two races: summer and winter steelhead (Figure 2-18, Figure 2-19).

- Summer steelhead return to fresh water from May to November, and enter the Columbia River in a sexually immature condition, requiring several months in fresh water to reach sexual maturity and spawn.
- Winter steelhead enter fresh water from November to April; they are close to sexual maturation and generally spawn shortly after arrival in their natal streams.

Some rivers have both summer and winter steelhead, while others have only one race. Where both runs occur in the same stream, summer steelhead tend to spawn higher in the watershed than do winter forms, perhaps suggesting that summer steelhead tend to exist where winter runs do not fully utilize available habitat (Busby 1996, Withler 1966, Roelofs 1983, Behnke 1992).

In rivers where both winter and summer forms occur, they are often separated by a seasonal hydrologic barrier, such as a waterfall. Coastal streams are predominantly winter steelhead, whereas interior subbasins are dominated by summer steelhead. Historically, winter steelhead may have been excluded from interior Columbia River subbasins by Celilo Falls (Busby 1996).

Initial downstream migration of juveniles generally occurs at either of two life stages: parr (2.36-3.94 in [60-100 mm]), or smolts (5.91-7.87 in [150-200 mm]). Migration of parr leads to rearing in additional areas downstream. Within primary areas of steelhead production, the evidence suggests that capacity for steelhead is generally limited by summer rearing habitat for parr.

2.4.1 Life History and Requirements

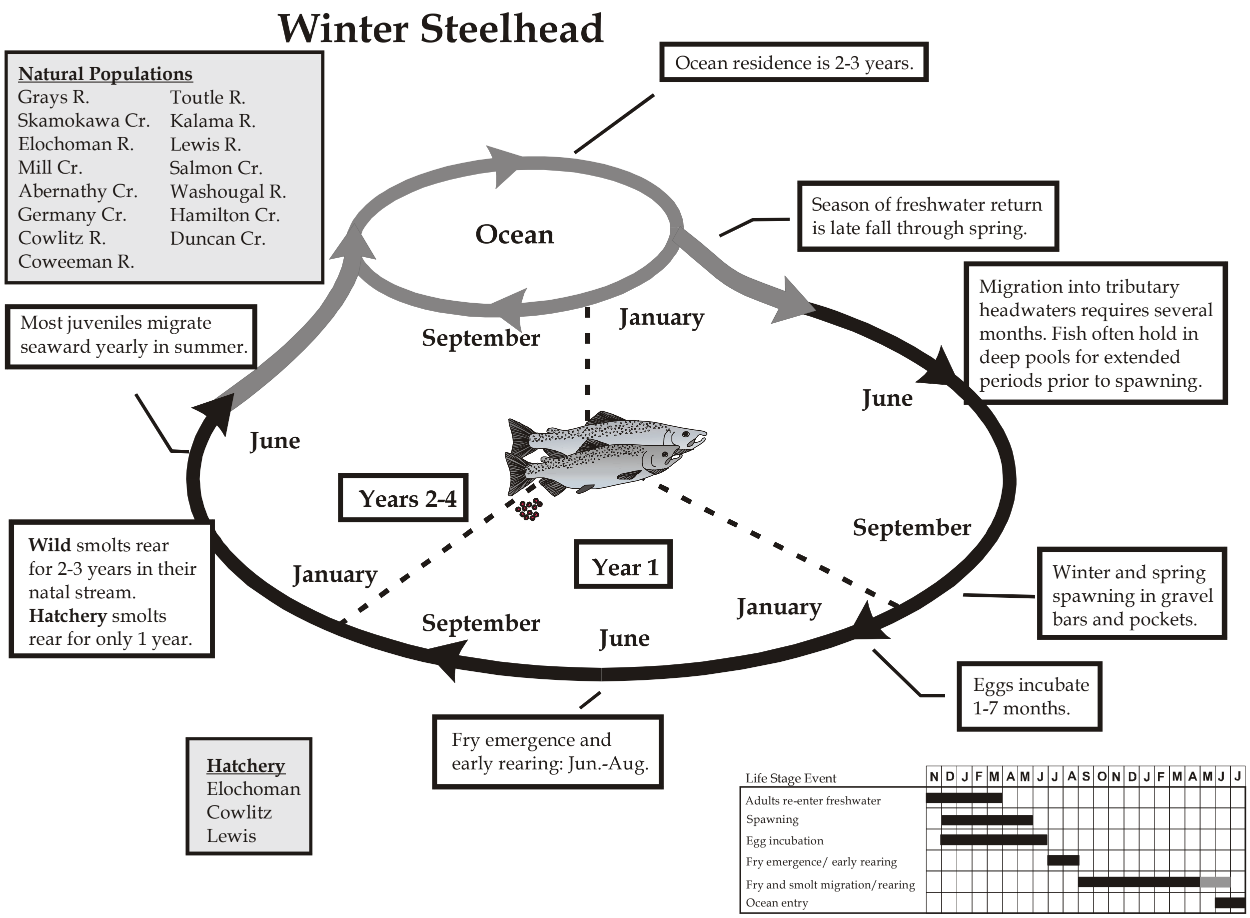

Figure 2-18. Winter steelhead life history.

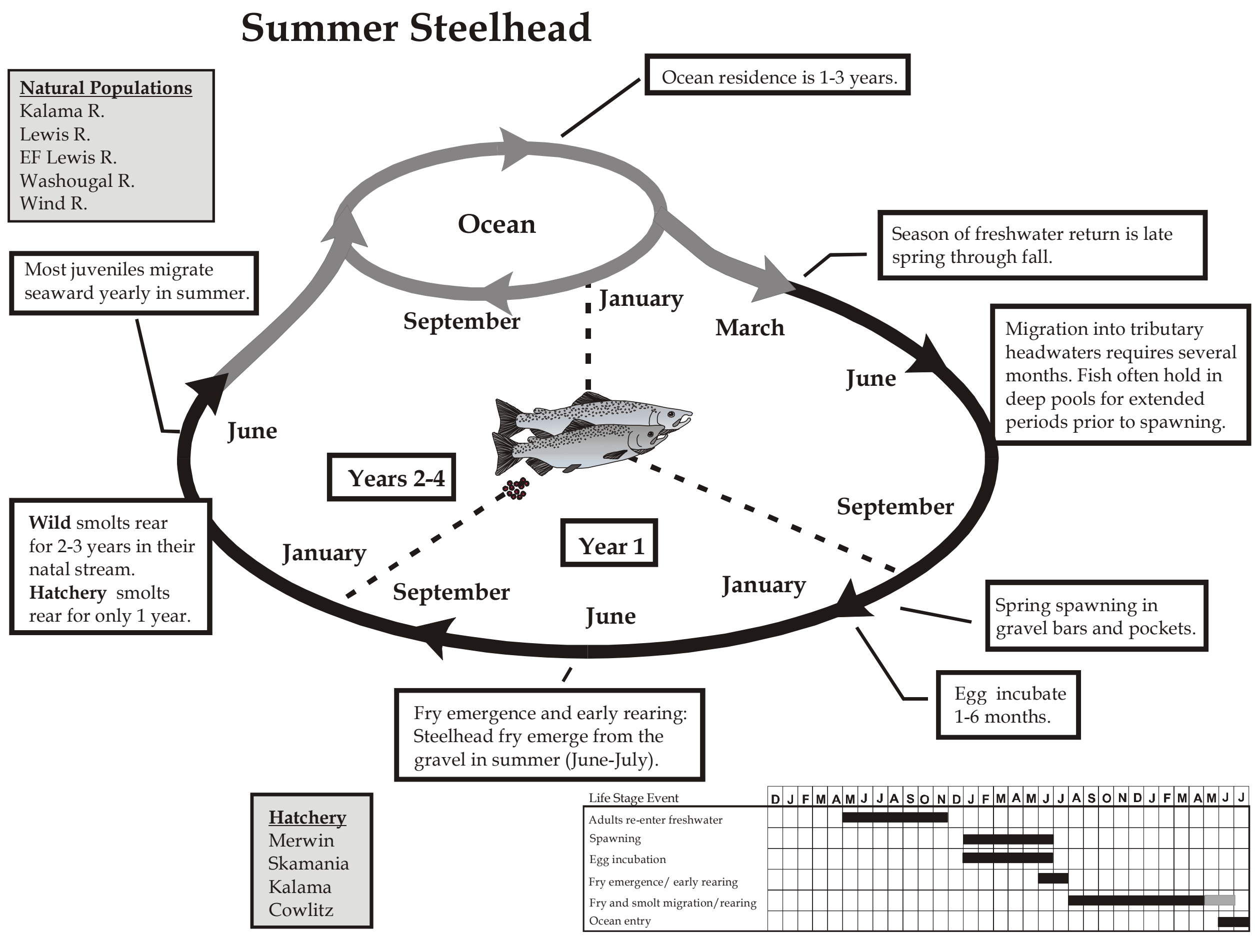

Figure 2-19. Summer steelhead life history.

2.4.1.1 Upstream Migration Timing

In the lower Columbia basin, summer steelhead return to fresh water from May to October, and enter the Columbia in a sexually immature condition, requiring several months in fresh water to reach sexual maturity and spawn. Winter steelhead enter fresh water from November to April, and return as sexually mature individuals that spawn shortly thereafter.

2.4.1.2 Spawning

Steelhead spawn in clear, cool, well-oxygenated streams with suitable gravel and water velocity. Adult fish waiting to spawn or in the process of spawning are vulnerable to disturbance and predation in areas without suitable cover. Cover types include overhanging vegetation, undercut banks, submerged vegetation, submerged objects such as logs and rocks, deep water, and turbulence. Spawning occurs earlier in areas of lower elevation and where water temperature is warmer than in areas of higher elevation and cooler water temperature. Spawning occurs from January through May, and precise spawn timing is related to stream temperature. Bovee (1978) reported a spawning temperature for steelhead of 39-55°F (4-13°C), with an optimum of 46°F (8°C).

Steelhead spawn in areas with water velocities of 1-3.62 ft/sec (30-110 cm/sec) but prefer velocities around 2 ft/sec (60 cm/sec) (Bovee 1978). Female steelhead bury their eggs at a depth of 2-12 in (51-305mm) in redds that occupy up to 60 ft^2 (5.57m^2). More than one redd may be constructed by each female in a season. Spawning sites typically require gravel (0.5-4.5 in [12.5-114mm] diameter) and well-aerated flow. Adult steelhead, unlike salmon, do not necessarily die after spawning but return to the ocean. However, repeat spawning is not common among steelhead migrating several hundred miles or more upstream from the ocean. Researchers have reported that the incidence of repeat spawning increases from north to south (Sheppard 1972, Barnhart 1986). Females have a higher survival rate after spawning that do males (Barnhart 1986). Spawning males usually mate with more than one female and remain in the spawning stream longer than do females (Jones 1974, Barnhart 1986).

2.4.1.3 Incubation and Emergence

Steelhead eggs hatch in 35–50 days depending on water temperature. Following hatching, alevins remain in the gravel 2 to 3 weeks until the yolk-sac is absorbed (Barnhart 1986).

Steelhead are spring spawners, so they spawn at a time when temperatures are typically cold, but increasing. Their spawning time must optimize avoidance of competing risks from gravel-bed scour during high flow and increasing water temperatures that can become lethal to eggs as the warm season arrives.

As with other salmonids, steelhead eggs show a distinct thermal range at which survival is optimum, and survival decreases at temperatures above or below that optimum. Incubation experiments by Kwain (1975) established that survival of steelhead eggs was near 100% at 44.6-50°F (7-10°C), was only about 25% at 59° F (15°C), and was about 50% at either 37 or 41°F (3 or 5°C). These data suggest that, while steelhead eggs have a narrower range of temperatures for optimal survival than either chinook or coho salmon, their rate of survival drops less at temperatures above or below optimum (Beacham and Murray 1990). While both steelhead and chinook eggs start dropping in survival as temperatures fall below 41°F (5°C), chinook egg survival drops to near 0% at 37°F (3°C), while steelhead egg survival drops to about 50% at that temperature. Survival of coho eggs remains near 100% down to 36°F (2°C).

Excessive amounts of fine sediment can reduce survival of salmonid eggs, rearing density of parr, and even survival of overwintering presmolts. Kondolf (2000) points out that although grains smaller than 0.39 in (10 mm) can reduce egg-to-fry survival, some streams with successful natural reproduction have egg-to-fry survivals of considerably less than 50%. Such occurrences indicate that egg-to-fry survival often is not the limiting factor to populations for which juveniles must rear at least a full year in fresh water.

Fry emergence is principally determined by the time of egg deposition and the water temperature during the incubation period. Fry emergence may occur from May through August in the Yakima River subbasin (YIN et al. 1990). In the lower Columbia, emergence timing differs slightly between steelhead races and among Washington subbasins. The different emergence times between races may be a function of spawning location within the watershed (and hence water temperature) or a result of genetic differences of the races. Generally, emergence occurs from March into July, with peak emergence time generally in April and May.

2.4.1.4 Freshwater Rearing

Following emergence, fry usually move into shallow and slow-moving margins of the stream, where they may aggregate in small schools of up to 10 individuals (Barnhart 1986) in waters 3-14 in (8 to 36 cm) deep (Bovee 1978). Fry tend to occupy shallow riffle habitats, but will occupy pool type habitats during periods of low flow (Barnhart 1986). As they grow, they inhabit areas with deeper water, a wider range of velocities, and larger substrate. Also as they grow older, fry cease schooling behavior and adopt and defend individual territories. Newly emerged fry are sometimes preyed upon by older juvenile steelhead (Barnhart 1986). When water temperatures fall below 39° F (4°C), fry become inactive and conceal themselves in the substrate (Chapman and Bjornn 1969, Barnhart 1986). Bjornn et al. (1977) found significant reductions in juvenile steelhead where gravel substrate was embedded in fine sediment. Sedimentation affects not only the availability of interstitial habitat, but also the productivity of invertebrates, a major food component of juvenile steelhead (Reiser and Bjornn 1979).

Excess fine sediment can reduce rearing density for salmonid parr. Newly emerged fry can occupy the voids of substrate made up of 0.8-2 in (2-5 cm) diameter gravel, but presmolts need cobble (>3 in (7.5 cm)) and boulder-sized rock to provide interstitial spaces they can occupy. Density of juvenile steelhead in summer and winter was reduced by more than half when enough sand was added to fully embed the large cobble substrate in an experimental stream (Bjornn et al 1977). Thompson and Lee (2000) found that probability of moderate to high densities of steelhead parr in Idaho streams decreased as the percentage of watershed with unconsolidated lithology increased ($P>0.05$). They deduced that this type of lithology was prone to sedimentation that could reduce parr survival.

Several studies have found that, although the stock-recruitment relationships show strong density dependence in survival over the length of a full generation from steelhead spawners to their mature offspring, the life stage at which the density dependence shows up is not the spawning-to-fry survival but rather parr-to-smolt survival. For example, Cramer et al. (1985) found from 6 years of data in the Rogue River that subyearling steelhead abundance was a positive linear function of spawner abundance. This finding remained the same after 2 more years of data were added (ODFW 1994). On the Keogh River in British Columbia, Ward and Slaney (1993), found from extensive sampling of all life stages since 1976 that the relationship of eggs-to-fry was linear, while the relationship between fry and smolts showed strong density dependence. Bjornn (1978) found that abundance of yearling steelhead migrants from the Lemhi River over 12 years approached an asymptote at which more age 0 steelhead did not produce more yearling migrants. Thus, carrying capacity of a stream for steelhead is determined by competition for space among parr. This density dependent mechanism affects overall abundance (Bjornn 1978, Everest et al. 1987, Ward and Slaney 1993), however, it appears to have little affect on steelhead growth (Bjornn 1978, Reeves et al. 1997).

Snorkel observations of rearing steelhead consistently show that they defend individual territories associated with the substrate, and are seldom found in schools (Everest and Chapman 1972, Hillman et al. 1987; Don Chapman Consultants 1989). By manipulating both the stocking density and the density of prey that rainbow trout eat in laboratory stream channels, Slaney and Northcote found that subyearling rainbow defended smaller territories and had fewer aggressive encounters as prey availability increased. Further, as prey density increased, the proportion of fry emigrating from the channels decreased. These behavioral traits lead to displacement of juvenile steelhead when space or food is limiting.

A distinct downstream movement of steelhead presmolts occurs in most streams in the fall, and this could be interpreted to indicate a shortage of winter habitat in their summer rearing area. Studies demonstrate that steelhead presmolts will migrate from an area in the fall where habitat for winter refuge is in short supply, but these fish typically find appropriate winter habitat farther downstream (Bjornn 1978, Tredger 1980, Leider et al. 1986).

Solazzi et al. (2000) found that construction of dammed pools and alcoves and adding woody debris to the constructed units substantially increased the number of steelhead smolts produced during the 4–5 years after restoration treatments in two streams. Solazzi et al. (2000) found that treatments did not increase the number of presmolts in the stream, so they deduced that the increased number of smolts must have resulted from higher survival of presmolts to smolts. While this may have been the case, the increased number of smolts also may have reflected a greater fraction of presmolts remaining in the treatment areas, rather than migrating downstream in search of winter habitat. The proportion of presmolts that migrate downstream in fall decreases as habitat complexity increases in the summer rearing areas.

Research has established that downstream movement of presmolts in the fall is not an indicator of increased mortality. In the Lemhi Basin, Bjornn (1978) found that 30-85% of steelhead presmolts in Big Springs Creek migrated downstream in the fall to overwinter in the Lemhi River, and that presmolt-to-smolt survival ranged from 6-41% for those that migrated, compared to 8-17% for those that stayed in Big Springs Creek. Similarly, Leider et al. (1986) found that most steelhead smolts originating from Gobar Creek had migrated downstream to the Kalama River as presmolts and reared successfully through the winter there. In Nuaitch Creek of British Columbia, Tredger (1980) estimate that 69% of steelhead smolts were actually produced downstream in the Nicoloa River where the fish had migrated to as fry or parr. Thus, evidence indicates that overwintering habitat is important to juvenile steelhead and such habitat may be equally valuable whether it is in the natal stream reach or in some reach downstream.

Observations indicate that juvenile steelhead habitat preferences in order of importance may be depth, velocity, and cover. In a study where effects of cover where held constant, Beecher et al. (1993) compared depth and velocity preferences of steelhead parr 3-8 in (75-200 mm) in a Washington stream that was uniformly lacking in cover; large boulders accounted for less than 1% of surface area and there was no large woody debris (LWD). The stream was believed to be fully seeded with juveniles, because goals for wild spawner escapement were met. Beecher et al. (1993) found that steelhead parr strongly avoided shallow habitats, but once depth was sufficient, velocity preference influenced habitat selection. The strongest effect that depth had on rearing distribution was that parr completely avoided areas with depths < 6 inches. Beecher et al. (1993) found the highest number of parr at depths of 1.6-2.5 ft (0.5-0.75 m), but parr showed the greatest preference for depth > 2.5 ft (0.75 m). Parr avoided depth < 0.8 ft (0.25 m) and velocities < 0.7 ft/sec (21 cm/sec). Beecher et al. (1993) found that most parr were observed at velocities of 0.9-1.1ft/sec (27.4-33.2 cm/sec), but velocities most preferred were 0.7-0.9 ft/sec (21.3-27.1 cm/sec). Preference of steelhead parr for these depths and velocities was also found in an Idaho stream by Everest and Chapman (1972), and was confirmed in an experimental setting by Fausch (1993). Additionally, Beecher et al. (1993) found a significant relationship between parr density and joint preference for depth and velocity combined in a fully-seeded Washington stream.

Fausch (1993) installed replicate experimental structures in a natural stream to evaluate steelhead preference for three types of cover; 1) velocity, 2) lateral cover, and 3) overhead cover. Fausch found that parr selected structures located adjacent to swifter velocities (>0.7 ft/sec [21 cm/sec]) and within 2.9 ft (0.9 m) of natural overhead cover. The preference for overhead cover was stronger than that for velocity cover or lateral cover, but preferences were additive when all were available. Overhead cover was located in or above the water, and presumably provided protection from air or land-based predators. Installation of experimental structures did not attract new fish to the area, but fish present organized around structures with consistent preferences as described. Shirvell (1990) found with experimental placement of rootwads that steelhead parr used velocity and shading created by rootwads, but were not attracted specifically to rootwads. The experiments of Shirvell (1990) concluded that steelhead preference for velocity ranked first, depth second, and light intensity (shade) third. These findings were still consistent with those of Beecher et al. (1993), because depths steelhead used were greater in the Shirvell study than the minimum depth that steelhead strongly avoided in the Beecher et al. (1993) study. Densities of steelhead parr in the stream studied by Shirvell (1990) averaged 1.7 parr/yd^2 (2 parr/m^2) at mean depth of 22 in (56 cm) and mean fish length of 48 in (124 cm).

Findings of steelhead micro-habitat preferences during snorkel surveys in natural streams were consistent with those in experimental settings. Don Chapman Consultants (1989) found that steelhead parr in Washington's Wenatchee River generally selected stations where adjacent velocities were 6-8 times their nose velocity. Stations chosen by parr increased in depth and velocity with fish size. The combined result of steelhead seeking cover from velocity and seeking stations adjacent to higher velocity, as described by Don Chapman Consultants (1989) and Fausch (1993), is that steelhead are often found in riffles or cascades behind boulders. Don Chapman Consultants (1989) found that steelhead concentrated in high gradient reaches (>5%) and usually stationed individually behind boulders where surface turbulence provided cover.

Ward and Slaney (1993) found that placement of boulders resulted in about one steelhead parr rearing per boulder, where none had reared previously. Dambacher (1991) found in the Umpqua River Basin, Oregon, "*Stream channels with relatively high (0.02/m^2) and low abundances (<0.02/m^2) of age >1 steelhead were separated, with some overlap, by the relative amount of large boulder substrate*." Although boulders (a form of substrate) can create velocity conditions that steelhead prefer, it appears that it is the velocity patterns, and not the boulders, that influence steelhead habitat selection. Snorkel surveys in the Wenatchee River by Don Chapman Consultants (1989) found that steelhead, when forced by sharp flow reductions to move, selected new stations with similar velocity patterns, but different substrates. Don Chapman Consultants (1989) concluded that associations with specific types of substrate were coincidental rather than causal.

Dambacher (1991) found that streams with greater average riffle depth also had greater densities of steelhead (R^2 =0.69). Similarly, Bisson et al. (1988) found a positive correlation of age 1+ steelhead use with habitat depth over the depth range of 0.4-1.6 in (10-40 cm). In smaller streams where riffles were too shallow, Dambacher (1991) found that age >1 steelhead showed strong electivity for pools, rather than riffles. Dambacher (1991) concludes, "*Stream size (as described by mean riffle depth) apparently creates an upper limit on density of age >1 steelhead rearing during the summer in stream channels of presumably good habitat quality*." Roper (1995) found in the South Fork (SF) Umpqua River and Jackson Creek that steelhead parr preferred riffles in the lower reaches and pools in the uppermost reach. In nine tributaries sampled in that study, steelhead tended to be more in pools than in riffles. Roper (1995) concluded that depth or other physical factors may be more important to steelhead preference than habitat selection for pools or riffles.

The preference of steelhead to station themselves where velocities are substantially higher on each side of their focal point results in higher densities as channel roughness (frequency of large boulders) increases. Johnson (1985) performed snorkel surveys of parr densities in a number of western Washington rivers. The published data show over a 10-fold variation between reaches in average parr densities within riffles, and much of that variation was related to whether boulders were the dominant substrate type. All of the reaches with higher than 3 parr/100m^2 (4 of 18 reaches) had boulders as dominant substrate, and the lowest density (1.5 parr/100m^2) in riffles dominated by boulder substrates was equal to the highest density observed in any other unit type. Parr densities in boulder-dominated riffles averaged about five times greater than in riffles with other substrate types as dominant.

Johnson et al. (1993) found that densities of steelhead parr within scour pools tended to increase with increasing structure complexity resulting from wood. They scored wood complexity from 1 to 5, with 1 being little to no wood present, 5 being a large amount of stable complex wood present. The average number of fish/pool during summer generally increased with increasing complexity from wood, and ranged from 2.6 parr/pool in 91 pools lacking complexity to 7.5 parr/pool in 27 pools with a complexity score of 4. Although there were fewer fish in pools with a habitat complexity of 5 compared to those rated as 4, the decrease in fish abundance at the highest pool complexity is likely to be an artifact of sampling bias. A different method of estimating fish abundance had to be used in the pools with the highest cover complexity, because both observation and capture of fish was obstructed. Recent studies have demonstrated that fish abundance in highly complex habitats is usually underestimated by conventional techniques. In winter, differences between complexity scores were not as consistent, and showed a gross indication of increasing fish numbers with greater wood complexity.

The USFWS (1988) studied winter habitat use by juvenile salmonids in the Trinity River. They found steelhead densities were greatest in areas containing cobble and boulder substrates, generally at shallow depths of 0.5-1.5 ft (0.2-0.6 m) with slow current (0-1.2 ft/sec [0-36.6 cm/sec]). Densities of steelhead in areas with high amounts of woody debris, but silt and sand substrate, had less than 1/10 of the steelhead densities found in side channels with cobble and boulder substrate. During winter, the USFWS (1988) found, "*Focal points were nearly always located underneath cobbles or boulders*." After 2 years of additional sampling for the same study in the Trinity River, USFWS (1990) concluded, "*For steelhead, by far the most important criterion of habitat utilization is the presence of cobbles from 6 to 12 inches in diameter free of sand or silt*." The preference of steelhead to overwinter in the interstices of cobbles was also reported by Bjornn (1971), Bustard and Narver (1975), and Hartman (1965).

Bjornn and Reiser (1991) reported from studies with steelhead and chinook in experimental stream enclosures, that more fish remained in pools with a combination of deep water, undercut bank, large rocks, and a bundle of brush than in pools with less cover. The number of yearling steelhead remaining in the stream section increased from 8 to 36 (>4-fold) when brush, large rock, and undercut banks were added. Thus, cobble or boulder substrate in semi-protected water is important as winter habitat for steelhead, either in their natal stream or in a larger channel downstream.

2.4.1.5 Juvenile Migration

Steelhead exhibit a great deal of variability in smolt age and ocean age. Most steelhead smolt at age 2, though British Columbia and Alaska populations exhibit a significant degree of age 3 smolting, and hatchery fish tend to smolt at age 1 (Busby 1996). Age at smolting tends to be younger in the southern part of the geographic range of steelhead.

Growth rate determines the size and age of smolts, and each of these has a strong influence on survival to maturity. Evidence from several studies shows that faster growing juveniles smolt at a younger age and that smolt-to-adult survival increases as smolt size increases. Ward and Slaney (1993) found that mean smolt age became younger in a linear relationship to increasing size of age 0 steelhead in mid summer ($R^2 = 0.45$; $P<0.05$). A similar trend for steelhead to reach smolting at a younger age as growth rate increased was found in the Rogue River (Cramer et al. 1985). Ward and Slaney (1988) found that steelhead smolting at a younger age tended to spend more years at sea than those that smolt at an older age (47% of age 2 female smolts stayed 3 years in the ocean, but only 20% of age 4 smolts stayed 3 years in the ocean). This trend was much stronger for females than males, and allowed a portion of females that grew quickly in fresh water to invest more years for growth in the ocean. Since fecundity is related to female size, females that spent longer time in the ocean produced more eggs (eggs increased by factor of 1.5 from 2-salt to 3-salt female). Additionally, fish that reached a smolt size earlier avoided an additional year of potential mortality in fresh water. A further advantage of earlier age at smolting is the reduced competition for space among the fish remaining in fresh water. Cramer (1986) found that the percentage of age 1 smolts was highly correlated ($R^2 = 0.64$) to growth during spring in the year of smolting, which in turn was highly and positively correlated ($R^2 = 0.71$) to stream temperature during February–April. Cramer (1986) found this same relationship was repeated in the following year to determine what proportion of the remaining cohort would smolt at ages 2 or 3. Ward and Slaney (1988) estimated that smolt-to-adult mortality rate dropped by more than half with each successive year in fresh water, probably as a function of fish size.

Size at smolting also has a strong influence on smolt-to-adult survival. Ward et al. (1989) studied wild steelhead from the Keogh River, BC, and showed that larger smolts survived at a higher rate. Size at smolting was highly correlated to age at smolting. Ward et al. (1989) compared the length at ocean entry for surviving adults (as determined from scales) to the length frequency distribution of smolts (same brood) passing a counting fence, and were able to reconstruct the relationship between smolt length and smolt-to-adult survival. That relationship showed that survival increased about 4-fold as smolt length increased from 6.3-8.7 in (160 mm-220 mm). However, a portion of the increased mortality at smaller smolt size would also have resulted from the increased tendency of those smaller (and younger) smolts to remain an extra year in the ocean. The classic study by Shapovalov and Taft (1954) also showed for wild steelhead in Waddel Creek, California, that larger smolts survived at a higher rate to maturity. Hatchery managers that raise steelhead are well aware that smolt-to-adult survival of hatchery steelhead increases dramatically as smolt size increases.

Available evidence indicates that a combination of the temperature regime and the size of a stream containing rainbow/steelhead trout (*O. mykiss*) determines whether the population will be predominantly anadromous or resident. Several large river basins on the West Coast have large populations of both resident and anadromous life-history types of *O. mykiss*, but in each case, the types occupy different portions of the basin. Further, breeding experiments indicate that the tendency to be anadromous or resident is an inherited trait, and that resident and anadromous fish in the same basin often breed as independent populations (Zimmerman and Reeves 1999, NMFS 1999). The separation of the primary rearing distributions of resident and anadromous *O. mykiss* within the same basin consistently occurs where there are strong differences in temperature regime. Resident rainbow occur upstream from anadromous forms in areas that are cooler in spring and summer. Resident trout that are larger than steelhead parr will competitively displace juvenile steelhead. The spatial patterns of stream temperature in basins where both the resident and anadromous forms are abundant are consistent with the theory that resident populations will prevail in streams where summer conditions are consistently favorable for growth and survival.

Baltz et al. (1987) studied the physical features that distinguished distribution of four fish species, including rainbow trout, in a 492 ft (150 m) reach of the Pit River, California, where sharp gradient of temperature occurs because of the inflow of cool water from a tributary. Their data show that rainbow were typically holding in water about 63°F (17°C), as were other species, but when river temperatures warmed to >68°F (20°C), rainbow held positions with stream water that averaged about 64°F (18 °C) while other species (sucker, northern pikeminnow, and hardhead) were at about 68°F (20°C). No rainbow were found in portions of the reach where temperature exceeded 68°F (20°C) on any of four sampling dates. These data suggest that temperatures of 64°F (18°C) or more would stimulate migration of *O. mykiss* out of the area, and if such temperatures were likely to occur at some time in most summers, then natural selection would probably favor anadromy over residency in that stream section.

Migrating smolts are particularly susceptible to predation because they may pass through areas of low cover and high predator concentration (Larsson 1985). Streamflow is important in facilitating downstream transport of outmigrating fish. Along with environmental cues such as photoperiod and temperature, flow is believed to be an important priming factor that triggers migratory behavior once a state of physiological readiness is achieved (Groot 1982). Flow may also influence the rate at which individuals move downstream, although some research indicates that flow may be a secondary factor to photoperiod, as faster-migrating individuals tend to occur at the peak of a run, regardless of low flow patterns that may exist at the time (Bjornn and Reiser 1991). Further, temperature influences the timing of freshwater migration by influencing the rate of growth and physiological development, and by affecting the responsiveness to smolts of other environmental stimuli (Groot 1982). Because of these relationships between migration behavior and flow or temperature, alteration of thermal and flow regimes can influence timing and rates of migration.

While it is likely that dissolved oxygen levels near saturation are required by smolts during the physiologically stressful period of outmigration, supersaturation of dissolved gases (especially nitrogen) has been found to cause gas bubble disease in outmigrating salmonids (Ebel and Raymond 1976). Reiser and Bjornn (1979) hypothesize that steelhead appear to be more susceptible to gas bubble disease because they seem to be less able to detect and avoid supersaturated waters (Stevens et al. 1980).

In the lower Columbia River, emigration of steelhead smolt generally occurs from March to June, with peak migration usually in April or May.

- On the Grays, Cowlitz, Lewis, and Washougal rivers, winter steelhead emigration is from April to May, with peak movement in early May.
- In the Kalama basin, emigration of summer and winter steelhead occurs from March to June, with peak migration from mid-April to mid-May.
- In the Lewis River, summer steelhead smolt emigration occurs from March through May, with peak migration in early May.
- In the Washougal River, summer steelhead smolt emigration generally occurs from April to May.

The dominant age class of emigrating steelhead smolts in the lower Columbia River is age 2.

- In the Grays River, juvenile rearing for the majority of wild winter steelhead lasts 2 years.
- Based on three years of data on the Cowlitz River, 91.1% of winter steelhead smolts resided for 2 years and 8.9% resided for 3 years before their emigration to salt water (Tipping et al. 1979, Tipping 1984).
- On the Toutle River in the Cowlitz basin, emigrating winter steelhead smolts were 86.5% age 2 and 13.5% were age 3 (Schuck and Kruse 1982).
- In the Kalama basin, winter and summer steelhead freshwater rearing primarily lasts 2 years (82.4%) before emigration, but some juveniles reside for 1 (6.2%) or 3 (11.4%) years prior to emigration (Loch et al. 1985).
- In the Lewis River, most winter steelhead juveniles rear for 2 years before emigration (83%), while others do not emigrate until age 3 (17%; Lavoy and Fenton 1983).

Lower Columbia River steelhead average smolt size was estimated at 6.3 in (160 mm). Emigrating steelhead smolts captured from the Kalama River and Gobar Creek ranged in length from 5.4-6.6 in (137.1-167.8 mm).

2.4.1.6 Estuary Rearing and Growth

Stream-type salmonid populations in the lower Columbia River include winter and summer steelhead. In general, stream-type juvenile salmon reach the lower mainstem and estuary at a relatively large size (> 100mm) and commonly spend less time than ocean-type salmonids rearing in the lower mainstem and estuary. Stream-type juvenile salmonids actively migrate through the lower Columbia River mainstem and estuary. Stream-type salmon are oriented to water column habitats and are typically found throughout the near-surface water column (i.e. top 6 m); they tend to avoid low-velocity areas and are not associated with any specific substrate type.

Juvenile steelhead were present in the Columbia River estuary from February to July of each year of sampling by Bottom et al. (1984); steelhead abundance was greatest in May and relatively low for other months (Bottom et al. 1984). Juvenile steelhead constituted 5% of the total juvenile salmonid catch (Bottom et al. 1984). Steelhead juveniles were distributed throughout the freshwater, estuarine, and marine regions of the estuary; they were most frequently associated with water column habitats (Bottom et al. 1984). Juvenile steelhead moved through the estuary more rapidly than other salmonids; based on catch data, they were present in the estuary for the shortest duration of any of the salmonid group (Bottom et al. 1984). Winter steelhead have been found to migrate at an average rate of 3.3 km/hr, traveling 134-143 km in 32-90 hours (Durkin 1982, Dawley et al. 1986 as cited in USACE 2001). Migration rate of many hatchery groups of juvenile steelhead increased through the estuary (Bottom et al. 1984). As with other salmonids, juvenile hatchery steelhead released further upstream in the basin migrated at a faster rate than juveniles released lower in the system (Bottom et al. 1984).

Steelhead in the Columbia River estuary consumed a relatively even proportion of *Corophium salmonis* (amphipod), *Corbicula manilensis* (bivalve), and adult *Diptera* (Bottom et al. 1984).

2.4.1.7 Ocean Migrations

In the ocean, steelhead migrate north and south along the continental shelf. Steelhead migrational patterns are generally believed to extend further out in the ocean than other salmonids; however, limited CWT recovery data is available to conclusively confirm this belief. Individuals grow rapidly in the ocean and their size at maturity depends primarily on how long they reside in salt water. Like other anadromous salmonids, steelhead smolt-to-adult survival can be dramatically affected by changes in ocean conditions. High variation in ocean survival between years is the norm for anadromous salmonids, and steelhead populations show some central tendency around a 10-fold range between smolt years (Cramer et al. 2003). Trends in ocean survival have been the driving force in years of low adult returns, where corresponding numbers of outmigrating smolts from previous years have been largely the same as years of higher ocean productivity (Cramer et al. 2003).

2.4.2 Distribution

Historically, steelhead were present throughout the lower Columbia River basins (Figure 2-20, Figure 2-21). Winter steelhead were distributed throughout most lower Columbia River tributaries from the Grays to the Wind rivers, while summer steelhead were found in the Kalama, NF Lewis, EF Lewis, Washougal, and Wind River basins. Steelhead continue to be produced naturally in most areas where native steelhead were found, although the abundance of most wild populations is thought to be low.

Spatial separation generally occurs in systems that have both summer and winter steelhead; summer steelhead usually are distributed within headwater areas of the basin, while winter steelhead spawn throughout the lower reaches. The headwater areas are often inaccessible to winter steelhead because of natural barriers that are not passable during the high winter water flows common during winter steelhead migration. These barriers are often passable during the lower flow conditions encountered by summer steelhead during upstream migration. Even in systems that do not have both summer and winter races of steelhead, summer steelhead generally use the upper river reaches, while winter steelhead generally spawn in the lower reaches.

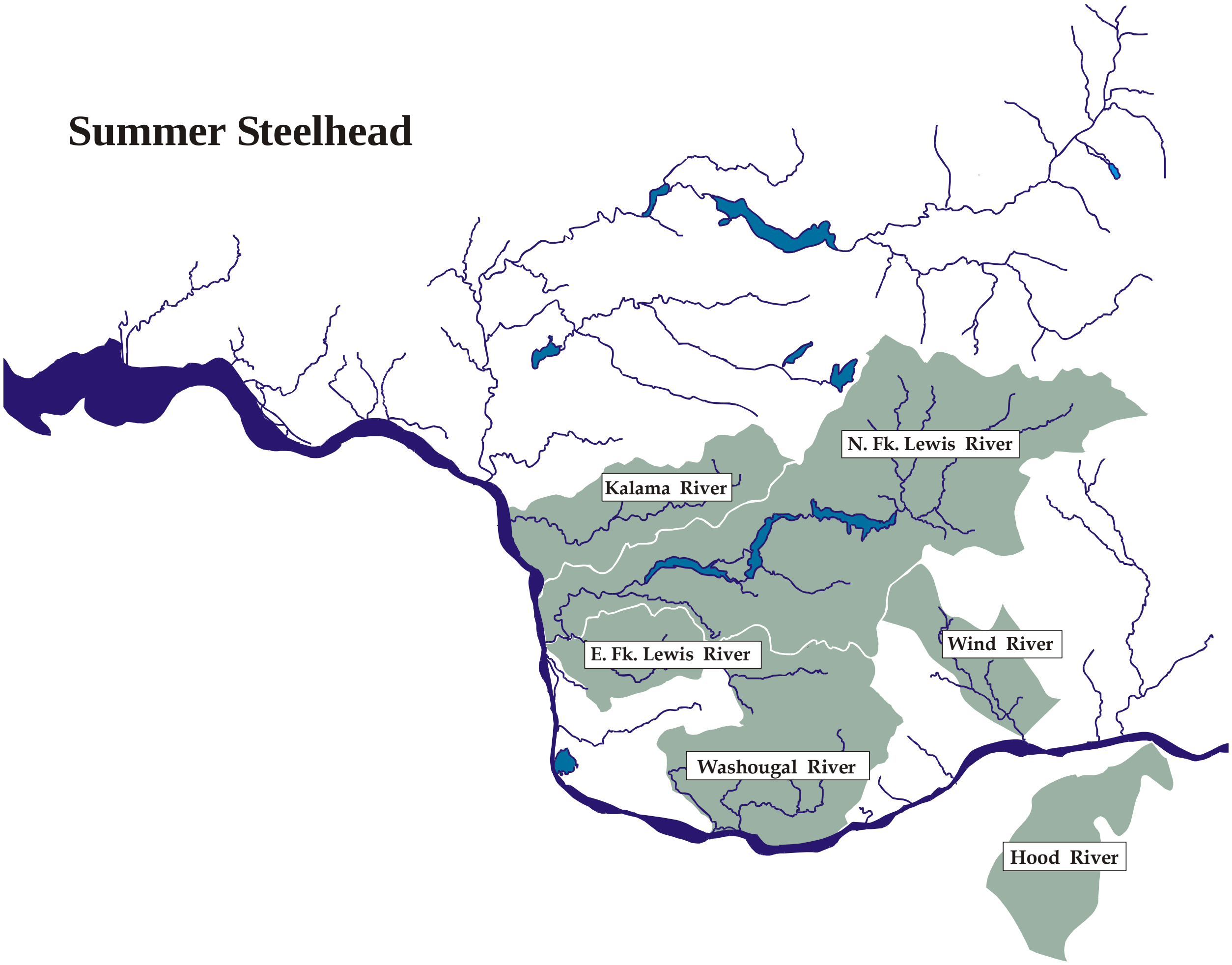

Figure 2-20. Historical demographically independent summer steelhead populations in the Lower Columbia River ESU (Myers et al. 2002).

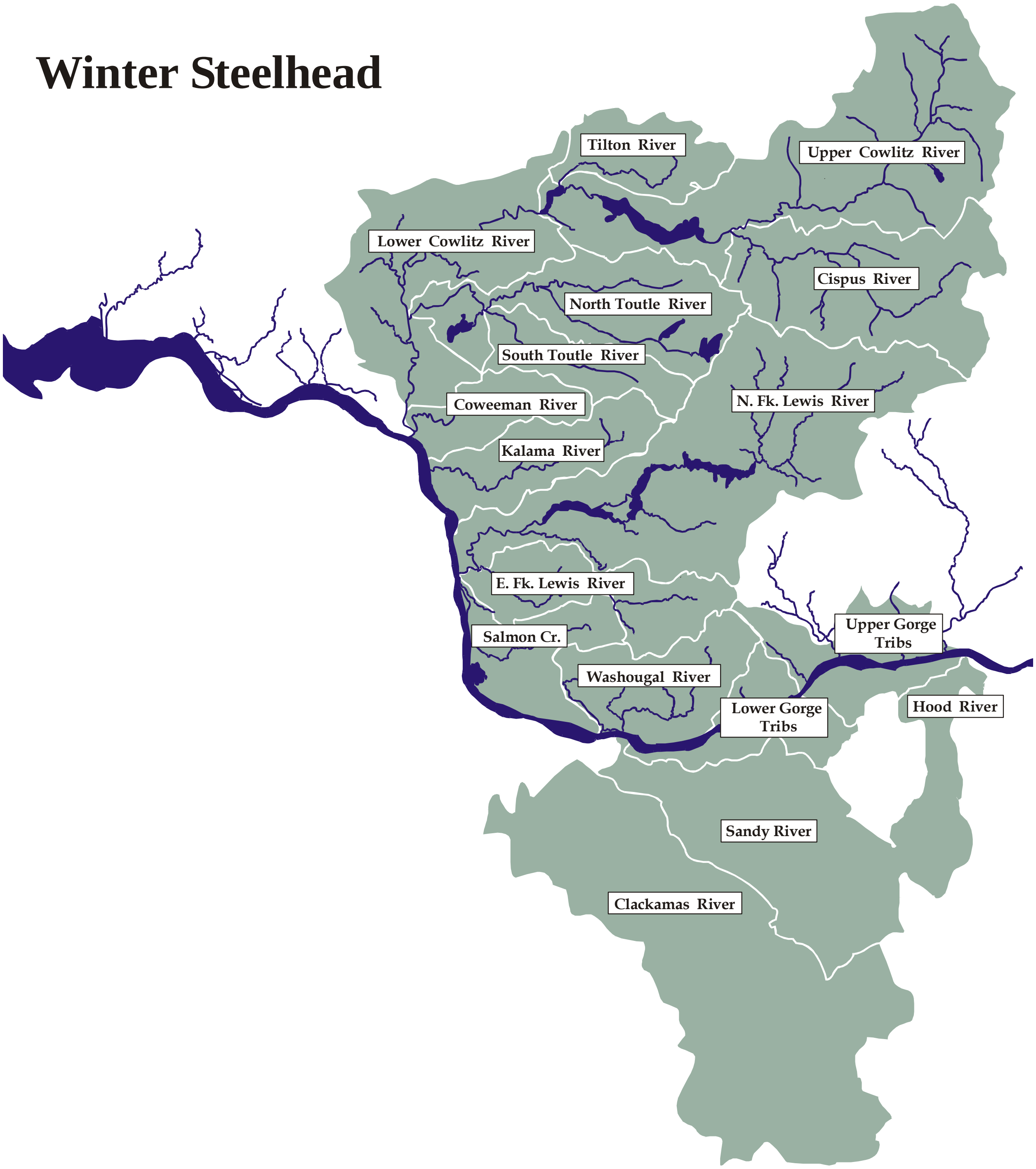

Figure 2-21. Historical demographically independent winter steelhead populations in the Lower Columbia River ESU (Myers et al. 2002).

2.4.3 Genetic Diversity

Multiple methods have been used to characterize West Coast steelhead genetic diversity; allozyme electrophoresis, DNA variations, and chromosomal karyotypes (Busby et al. 1996).

Allozyme frequencies have shown two genetically distinct groups of steelhead in Washington, Oregon, and Idaho; a coastal and an inland group (Allendorf 1975, Utter and Allendorf 1977, Utter et al. 1980, Okazaki 1984, Schreck et al. 1986, Reisenbichler et al. 1992). The geographic separation for the two groups appears to be the Cascade Crest, although some uncertainty remains in identifying the boundary. Based on genetic data alone (i.e. no geographical consideration), Leider et al. (1995) suggested that the boundary on the Columbia River is between the Wind and Big White Salmon Rivers. Similar differences have been identified between steelhead from interior and coastal regions of British Columbia (Huzyk and Tsuyuki 1974, Parkinson 1984).

Coastal steelhead have been further segregated into distinct groups; however, not all studies have delineated the same groupings. Hatch (1990) reported evidence for a north-south cline in allele frequencies for basins larger than 350 km^2. He suggested Cape Blanco as a geographic feature that limited straying in populations to its' north and south. Reisenbichler et al. (1992) observed that wild coastal steelhead clustered into a north coast group (from the mouth of the Columbia River south to coastal streams just north of the Umpqua River) and a south coast group (from the Umpqua River in central Oregon to the Mad River in northern California). During ESA status reviews, NMFS analyses (Busby et al. 1993, 1994) suggested three genetic population groups: Oregon coast north of Cape Blanco, Cape Blanco to the Klamath River basin inclusive, and south of the Klamath River basin. Leider et al. (1995) expanded the database for genetic data on the Washington coast; samples from certain geographic areas tended to be more similar to each other than they were to samples from other areas and established the following groupings: north Puget Sound (Stillaguamish River and basins north), south Puget Sound, Olympic Peninsula, southwest Washington, and the lower Columbia River (Kalama, Wind, and Washougal Rivers).

Reisenbichler and Phelps (1989) found some genetic variation among steelhead in nine northwest Washington (primarily Olympic Peninsula) basins. Genetic differences among steelhead populations in adjacent drainages were substantially smaller than those reported by Parkinson (1984) for steelhead populations in British Columbia. Reisenbichler and Phelps (1989) and Reisenbichler et al. (1992) suggested that the lower degree of variation in Washington drainages is a result of introgression of hatchery fish into naturally spawning populations. The use of hatchery steelhead in Washington has been extensive and most hatchery steelhead in Washington originated from two primary stocks (Chambers Creek and Skamania stock). However, it is possible that the Reisenbichler and Phelps (1989) study did not cover a large enough geographic area and is not comparable to research in British Columbia (Hatch 1990). Conversely, Phelps et al. (1994) found significant differences between all pairs of populations from Puget Sound and the lower Columbia River. Analyses of the Columbia River data suggests that summer run steelhead in the Wind and Washougal rivers are outliers in the analyses. For example, the Wind River sample contained an allele at a frequency of 15% that was not found in steelhead in any other sample.

In their investigation of genetic variation in steelhead from Idaho to northern California, Reisenbichler et al. (1992) observed that most hatchery populations were significantly different from wild fish. Phelps et al. (1994) also noted statistically significant differences between samples of hatchery and natural steelhead from Puget Sound and the lower Columbia River. These data suggest that at least some native population structure remains. Phelps et al. (1994) also investigated the effect of rainbow trout stocking on steelhead populations in Washington. Because there were large genetic distances between four widely used rainbow stocks and all steelhead populations sampled, Phelps et al. (1994) concluded that there has been little, if any, permanent genetic effect on steelhead populations from widespread rainbow trout stocking.

Using mtDNA analysis, Buroker (unpublished) examined 23 major river systems from Alaska to California and found no evidence for strong geographic structuring of populations because the most common clonal types were widely distributed. Thorgaard (1983) examined chromosomal karyotypes in steelhead from Alaska to Central California. The most common chromosome number was a 58-chromosome karyotype; however, two geographic regions were characterized by steelhead with 59-60 chromosomes: Puget Sound/Strait of Georgia and Rogue River/northern California. In contrast to the allozyme electrophoresis studies, Thorgaard (1983) did not find differences between coastal and inland steelhead populations.

Allozyme frequencies have been unable to demonstrate differences between distinct winter and summer runs of steelhead in the same drainage. Allendorf (1975) and Utter and Allendorf (1977) found that summer and winter steelhead of a particular coastal stream tended to genetically resemble one another more than they resembled populations in adjacent basins with similar run timing. Further allozyme studies support this conclusion in a variety of geographic regions (Chilcote et al. 1980, Schreck et al. 1986, Reisenbichler and Phelps 1989, Reisenbichler et al. 1992). Shreck et al. (1986) found allele frequencies to be similar for summer- and winter-run steelhead in geographically proximate streams in the Columbia River system. However, in more recent studies, the summer-run stocks have had some extent of hatchery introgression and may not represent the indigenous population. Also, interpretation of the results may be complicated by difficulties in determining run timing of sampled fish. Using chromosomal variability, Thorgaard (1983) was unable to demonstrate a difference in winter- and summer-run steelhead from two systems with limited hatchery introductions (Quinault River in Washington and Rogue River in Oregon). The chromosome number differed between the two systems but was similar in summer and winter steelhead within each river system. Reisenbichler et al. (1992) caution that the absence of difference in allozyme frequencies among groups of fish does not always provide a reliable basis for concluding that these groups are genetically homogeneous. The lack of evidence for genetic differentiation of steelhead within drainages observed in the aforementioned allozyme electrophoresis studies does not rule out the possibility that genetic differences exist in traits affecting survival. For example, genetic differences have been found between sympatric summer- and winter-run steelhead in number of vertebrae, gill rakers, parr marks, rate of maturation in salt water, and level of storage fat in juveniles and adults (Smith 1969).

No clear determination has been made regarding the genetic variation between steelhead and rainbow trout.

2.4.4 ESU Definition

Steelhead found in the lower Columbia River in Washington (as delineated by this recovery plan) fall into three separate ESUs defined by NMFS (Busby et al. 1996):

- Southwest Washington ESU includes steelhead from the Grays and Elochoman rivers, and Skamokawa, Mill, Abernathy, and Germany creeks,

- Lower Columbia ESU includes steelhead from the Cowlitz, Kalama, Lewis, Washougal, and Wind Rivers and Salmon and Hardy creeks, and
- Middle Columbia ESU includes steelhead from the Little White Salmon and Big White Salmon rivers.

On March 19, 1998 NMFS (now NOAA Fisheries) issued a formal notice listing the Lower Columbia steelhead ESU as threatened under ESA. The listed ESU includes only naturally spawned populations of steelhead (and their progeny) residing below naturally and man-made impassable barriers (e.g., impassable waterfalls and dams). The populations that have been identified as comprising the Lower Columbia ESU are shown in Figures 2-19 and 2-20. More detail on these populations is reported by Myers et al. (2003).

2.4.5 Life History Differences

Adult summer steelhead generally migrate in the lower Columbia River from May through November. Although limited age data are available, dominant age class of returning adults is 2.2 (i.e. 2 years in freshwater and 2 years in the ocean). Lower Columbia stocks of summer steelhead may spawn as early as January through early June in the year following their entry into fresh water. Wild steelhead fry emerge from April–July, depending on spawn timing and water temperature. Juvenile steelhead generally rear in fresh water for 2 years; juvenile emigration occurs from March–June, with peak migration usually in early May. NF Lewis, EF Lewis, Washougal, and Wind summer steelhead conform to these general life history strategies.

Kalama summer steelhead differ slightly in the timing of the adult run and spawning; their adult migration usually occurs over a shorter period (normally from June–October), and spawning occurs from mid-January through April, earlier than other lower Columbia stocks. Although considerable hatchery summer steelhead releases have occurred in the Kalama basin, Kalama summer steelhead wild stock appears to have retained genetic traits of considerable adaptive value relative to the transplanted hatchery stock (Leider et al. 1995). Also, Kalama summer steelhead have been observed spawning with Kalama winter steelhead; thus, genetic material has been shared to some extent among the steelhead races. These genetic differences of Kalama summer steelhead compared to other lower Columbia summer steelhead may provide some explanation of the different life history strategies of Kalama summer steelhead.

Winter steelhead adult migration timing in the lower Columbia River is generally from November through April. Although, Chambers Creek stock early-run winter steelhead arrive from November to December and spawn earlier than the listed late-run winter steelhead. Although limited age data are available, dominant age class of returning adults is 2.2. Winter steelhead spawn timing for lower Columbia stocks is usually from March through early June, with less time spent in fresh water before spawning than summer steelhead. Wild steelhead fry emerge from April to July, depending on spawn timing and water temperature. Juvenile steelhead generally rear in fresh water for 2 years; juvenile emigration occurs from March to June, with peak migration usually in early May. Winter steelhead in the basins of tributaries to the Grays, Elochoman, and Cowlitz rivers, as well as the lower Columbia Gorge tributaries, conform to these general life history strategies. Most of these basins are in the lower portions of the lower Columbia River basin and have had considerable hatchery influence from Elochoman and Cowlitz stocks.

Winter steelhead in the Kalama, NF Lewis, EF Lewis, Washougal, and Wind basins were identified as a distinct stock partially based on run timing, but the specific run timing for each stock was not provided. Most data suggests that the run timing for these stocks is similar to other lower Columbia River winter steelhead. All of these basins are in the upper portions of the lower Columbia River basin; most stocks have had substantial influence from hatchery stocks from either the Elochoman and Cowlitz or the Skamania Hatchery stock. Kalama winter steelhead are known to spawn from early January to early June; this is an earlier and longer spawning period than other lower Columbia River winter steelhead. Limited escapement surveys suggest that Salmon Creek winter steelhead spawn timing may be earlier than most lower Columbia River winter steelhead.

2.4.6 Abundance—Summer Steelhead

Summer steelhead abundance is naturally quite variable, with variation in ocean conditions believed to be the major driving force in the fluctuating sizes of Pacific salmonid runs and escapement (Pearcy 1992, Beamish and Bouillon 1993, Lawson 1993). Poor ocean conditions during the 1990s resulted in decreased steelhead abundance throughout the lower Columbia River basin.

During the early 1980s, steelhead abundance in the Lower Columbia River ESU (including the Upper Willamette ESU) was estimated at approximately 80,000 summer steelhead, although 75% of this estimate was thought to be of hatchery origin (Light 1987). Nehlsen et al. (1991) identified 19 stocks in the Lower Columbia River ESU at some risk of extinction or of special concern. The following designations were given to lower Columbia River stocks covered in this recovery plan:

- high risk of extinction—Cowlitz River summer steelhead, NF Lewis River summer steelhead, and Washougal River summer steelhead
- moderate risk of extinction—Wind River summer steelhead
- special concern—EF Lewis River summer steelhead

Historical (pre-1960) abundance estimates of steelhead populations in the lower Columbia River are scarce. Most summer steelhead stocks in the lower Columbia River are at low abundance levels compared to estimated historical levels as a result of hydro projects, habitat degradation from human activities in the basin (development, logging, etc.), and possible hatchery impacts.

Adequate long-term data is not available for most stocks in the lower Columbia River to address population trends, although available data indicates negative population trends and low abundance compared to historical estimates. Because most of the data sets are short-term, any determinations of population trends may be heavily influenced by short-term climate effects. Of the summer steelhead stocks identified by WDFW in the lower Columbia River in 2002, two were considered depressed (Kalama and Wind) and the status of the remaining three (NF Lewis, EF Lewis, and Washougal) was unknown (Table 2-2).

In a recent status report of steelhead in the Lower Columbia River ESU (unpublished), the TRT indicated that, of the six historical summer steelhead populations, not one population could be conclusively identified as naturally self-sustaining. Some degree of natural production was documented in three of six summer steelhead populations. The Ecosystem Diagnosis and Treatment (EDT) model used expected historical habitat conditions to estimate historical steelhead abundance in the Lower Columbia River ESU. A total historical abundance estimate of summer steelhead was 7,294 fish.

All summer steelhead populations in the Washington portion of the lower Columbia River are below WDFW's natural escapement goals. The Kalama summer steelhead escapement goal of 1,000 fish has not been met since 1995 (Figure 2-22). Although spawning escapement estimates of summer steelhead in the Lewis, Wind, and Washougal systems are not available, snorkel index escapement counts exist. Index snorkel counts are estimated to represent 25-70% of natural escapement in each stream (WDFW 1997). Based on these escapement index counts, summer steelhead escapement goals for the EF Lewis River (814 fish), Washougal River (1,210 fish), and Wind River (957 fish) have not been met for at least a decade and likely longer. An escapement goal has not been set for the NF Lewis summer steelhead stock.

Table 2-2. Lower Columbia River steelhead stock status as determined by SASSI 2002.

Basin	Stock	Winter Steelhead	Summer Steelhead
Grays River	Grays	Depressed	NA
Elochoman River	Elochoman/Skamokawa	Depressed	NA
	Mill	Unknown	
	Abernathy/Germany	Depressed	NA
Cowlitz River	Mainstem Cowlitz	Unknown	NA
	Coweeman	Depressed	NA
	NF Toutle/Green	Depressed	NA
	SF Toutle	Depressed	NA
Kalama River	Kalama	Healthy	Depressed
Lewis River	NF Lewis	Unknown	Unknown
	EF Lewis	Unknown	Unknown
Lower Columbia (Bonneville) Tributaries	Salmon	Unknown	NA
	Hamilton	Unknown	NA
Washougal River	Washougal	Depressed	Unknown
Wind River	Wind	Unknown	Depressed

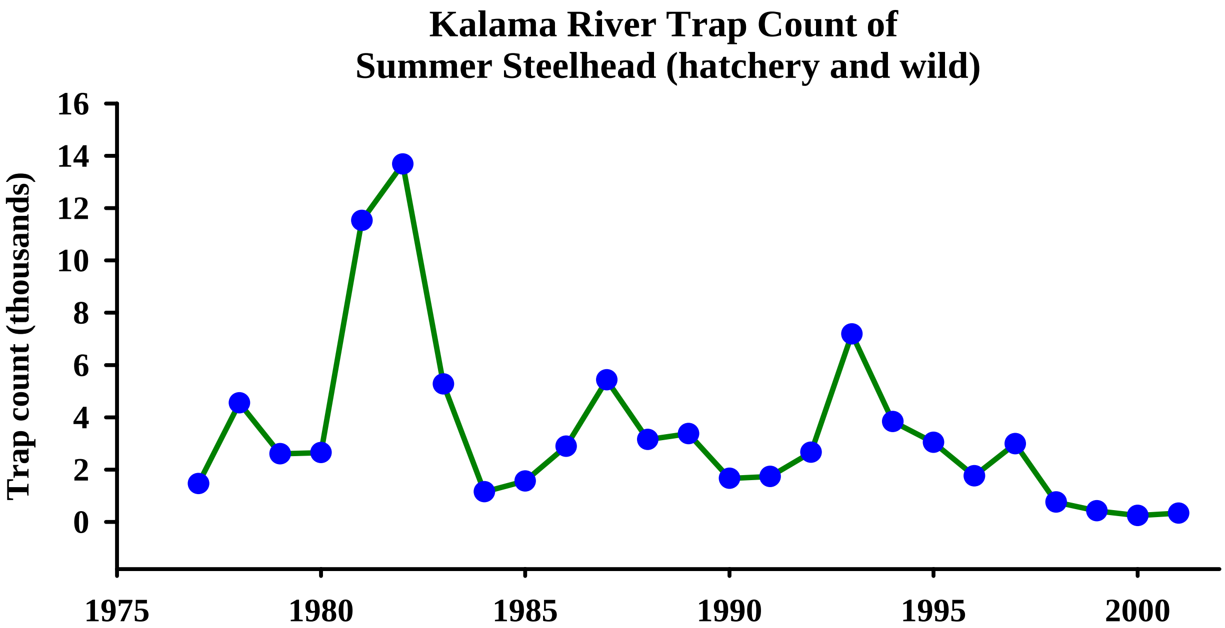

Figure 2-22. Counts of total wild and hatchery summer steelhead trapped at Kalama Falls Salmon Hatchery trap, 1976–2001.

2.4.7 Productivity—Summer Steelhead

As with other salmonids, steelhead productivity is highly influenced by ocean conditions. Steelhead in the ocean appear to follow a counterclockwise migration pattern in waters east of 167° East longitude, primarily within 27 feet of the surface. Poor ocean conditions during the 1990s resulted in decreased steelhead productivity throughout the lower Columbia River basin. Abundance may have been further decreased with increased predation by Caspian terns in the lower Columbia River during the late 1990s, although there is no basis to determine how the recent estimated tern predation differs from historic losses of juvenile salmonids.

Historically, steelhead production in Washington basins of the lower Columbia River was thought to be high, although most steelhead production was likely from the winter race. The production potential of most lower Columbia River basins is substantially reduced from historical conditions as a result of habitat degradation. Most habitat degradation has resulted from human activity, such as development or logging, although considerable habitat loss occurred in the Cowlitz basin, especially the NF Toutle River, as a result of the 1980 Mt. St. Helens eruption. Major hydro projects in the Cowlitz and Lewis basins have blocked access to approximately 80% of the historical steelhead spawning and rearing habitat within both basins.

The NPCC's smolt production model was applied to many systems within the lower Columbia River in the early 1990s to estimated steelhead production potential. The potential summer steelhead smolt production estimate was 62,273 smolts for the Wind River basin. In the Kalama basin, WDW estimated that potential winter and summer steelhead smolt production was 34,850; the NPCC's model estimated 64,860 (the NPCC model is generally optimistic and can overestimate production potential). From 1978–84, the number of naturally produced steelhead smolts migrating annually from the Kalama ranged from 11,175 to 46,659.

In a recent status report of steelhead in the Lower Columbia River ESU (unpublished), the TRT compared the potential historical habitat available to steelhead to the potential current available habitat. For the entire Lower Columbia River ESU (including Oregon basins), 63% of the historical available steelhead habitat is available today. Most basins have over half the historical habitat still available and some basins still have the majority of the historically available habitat.

2.4.8 Abundance—Winter Steelhead

Winter steelhead abundance is naturally quite variable, with variation in ocean conditions believed to be the major driving force in the fluctuating sizes of Pacific salmonid runs and escapement (Pearcy 1992, Beamish and Bouillon 1993, Lawson 1993). Poor ocean conditions during the 1990s resulted in decreased steelhead abundance throughout the lower Columbia River basin.

Like steelhead populations throughout their range, lower Columbia winter steelhead have experienced declines in abundance during the past several decades. Wild winter steelhead of native origin exist in the Grays River, Skamokawa Creek, Elochoman River, Mill Creek, Abernathy Creek, Germany Creek, Cowlitz River, Coweeman River, Toutle River, Kalama River, Lewis River, Salmon Creek, Washougal River, Hamilton Creek, and Wind River systems (WDFW 1993).

During the early 1980s, winter steelhead abundance in the Lower Columbia River ESU (including the Upper Willamette ESU) was estimated at approximately 150,000, although 75% of this estimate was thought to be of hatchery origin (Light 1987). Nehlsen et al. (1991) identified 19 stocks in the Lower Columbia River ESU at some risk of extinction or of special concern. The following designations were given to lower Columbia River stocks covered in this recovery plan:

- high risk of extinction—Wind River winter steelhead,
- moderate risk of extinction—Cowlitz River winter steelhead, Washougal River winter steelhead, and
- special concern—Coweeman River winter steelhead, Toutle River winter steelhead, Kalama River winter steelhead, Lewis River winter steelhead.

No estimates of historical (pre-1960s) abundance specific to the Southwest Washington ESU are available. Nehlsen et al. (1991) identified three stocks within this ESU at risk of extinction or of special concern; all identified stocks were within the lower Columbia River portion of this ESU. Winter steelhead in the small lower Columbia River tributaries (including Mill, Abernathy, and Germany creeks) were designated with a moderate risk of extinction, and winter steelhead in the Grays and Elochoman rivers were designated as special concern.

Historical (pre-1960) abundance estimates of steelhead populations in the lower Columbia River are scarce. The largest steelhead population in the lower Columbia River was thought to be in the Cowlitz basin. WDF and WDG (1949) estimated that the steelhead spawning escapement past the Mayfield Dam site was 11,000 fish; considering harvest, the total run size was estimated at 22,000 fish. Naturally spawning populations of steelhead still exist in the lower mainstem Cowlitz, Coweeman, and Toutle River basins, although loss of habitat as a result of dam construction and the 1980 Mt. St. Helens eruption has significantly contributed to decreased abundance. Most other steelhead stocks in the lower Columbia River also are at low abundance levels compared to estimated historical levels as a result of hydro projects, habitat degradation from human activities in the basin (development, logging, etc.), and possible hatchery impacts.

Long-term data adequate to address population trends for most stocks in the lower Columbia River is not available, although available data indicates negative population trends and low abundance compared to historical estimates. Because most of the data sets are short-term, any determinations of population trends may be heavily influenced by short-term climate effects.

In the early 1990s, only two steelhead stocks in the lower Columbia River showed an increasing trend: the South Fork Toutle River and the Kalama River winter steelhead (Figure 2-23) (Busby et al. 1996). The increasing trend observed in Toutle/NF Cowlitz River winter steelhead stock was likely a result of continued rebuilding of the stock after the 1980 Mt. St. Helens eruption; abundance of this stock was still considered low.

Also in the early 1990s, only two steelhead stocks in the lower Columbia River were considered healthy: SF Toutle River winter steelhead and Kalama River winter steelhead (WDF et al. 1993). In 2002, WDFW identified 16 winter steelhead stocks on the Washington side of the lower Columbia River; one was healthy (Kalama winter steelhead), eight were depressed, and the status of the remainder was unknown because of a lack of escapement data (Table 2-2).

In a recent report on the status of steelhead in the Lower Columbia River ESU (unpublished), the TRT indicated that, of the 17 historical winter steelhead populations, not one population could be conclusively identified as naturally self-sustaining. Some degree of natural production was documented in 9 of 14 winter steelhead populations. The EDT model used expected historical habitat conditions to estimate historical steelhead abundance in the Lower Columbia River ESU. The total historical abundance estimate of winter steelhead based on the EDT model was 18,243 fish.

However, WDFW has estimated there were historically 20,000 winter steelhead in the Cowlitz system alone (Hymer et al. 1992, WDFW 1997). Estimates for winter steelhead escapement in Washington tributaries based on redd counts are presented in Table 2-3. No redd count data are available for the Cowlitz River. Recent year escapements for all winter steelhead populations in the Washington portion of the lower Columbia River are below WDFW's natural escapement goals. However, the most recent year (2002) return of winter steelhead to the lower Columbia basins was improved for most populations (Figure 2-24); also the North Toutle escapement was a post-eruption high.

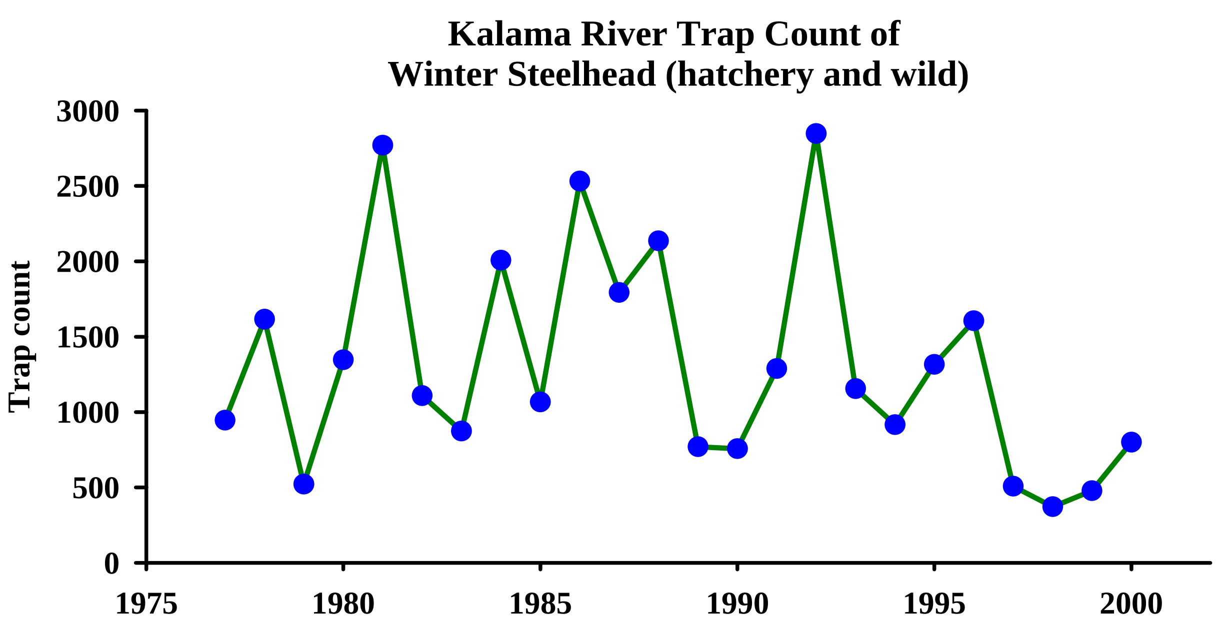

Figure 2-23. Counts of total wild and hatchery winter steelhead trapped at the Kalama Falls Salmon Hatchery trap, 1976–2001.

Table 2-3. Escapement of winter steelhead in lower Columbia Washington tributaries. Escapement estimates are from redd count data, and represent all natural production, including naturally spawning hatchery winter steelhead.

	Grays River	Elochoman River	Skamokawa Creek	Abernathy Creek	Germany Creek	Green River	NF Toutle River	SF Toutle River	Coweeman River	Kalama River	EF Lewis River	Total
1977										774		774
1978										694		694
1979										371		371
1980										1025		1025
1981										2150		2150
1982										869		869
1983										532		532
1984										943		943
1985						775		1807		632		3214
1986								1595		1081	282	2958
1987						402		1650	889	1155	192	4288
1988						310		2222	1088	1269	258	5147
1989						128	18	1371	392	588	140	2637
1990						86	36	752	522	419	102	1917
1991	716	166		280		108	108	904		1128	72	3482
1992	1224	278	304	246		44	322	1290		2322	88	6118
1993	1086	378	258	88	216	84	165	1242	438	992	90	5037
1994	704	230	208	58	108	128	90	632	362	853	78	3451
1995	426	132	92	34	42	174	175	396	252	1212	53	2988
1996	203	52	112	16	40		251	150	44	853		1721
1997	158	64	128	64	46		183	388	108	537	192	1868
1998	546	100	208	146	90	118	137	374	314	438	250	2721
1999	300	90	200	78	110	72	129	562	126	562	276	2505
2000						124	238	790	290	941	207	2590
2001						79	185	334	284	1085	79	2046
Average	*596*	*166*	*189*	*112*	*93*	*103*	*180*	*642*	*246*	*993*	*139*	*3138*

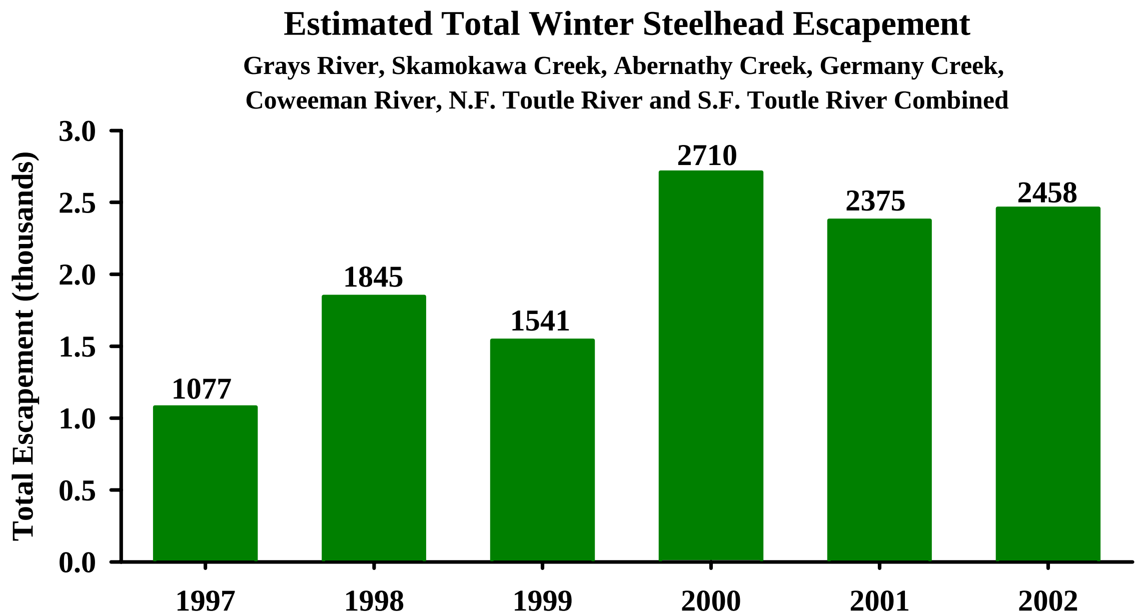

Figure 2-24. Total winter steelhead escapement (Grays River, Skamokawa Creek, Abernathy Creek, Germany Creek, Coweeman River, NF Toutle River, and SF Toutle River combined).

2.4.9 Productivity—Winter Steelhead

As with other salmonids, steelhead productivity is highly influenced by ocean conditions. Steelhead in the ocean appear to follow a counterclockwise migration pattern in waters east of 167° East longitude, primarily within 27 feet of the surface. Poor ocean conditions during the 1990s resulted in decreased steelhead abundance throughout the lower Columbia River basin.

Historically, steelhead production in Washington basins of the lower Columbia River was thought to be high. For example, total run size for steelhead in the Cowlitz River was estimated to be greater than 20,000 fish and, based on preliminary information developed in the process of Lewis hydro relicensing, 10,000 or more may have been produced in the Lewis basin. The production potential of most lower Columbia River basins is substantially reduced from historical conditions as a result of habitat degradation resulting mostly from human activity, such as development or logging. However, considerable habitat loss occurred in the Cowlitz basin, especially the NF Toutle River, as a result of the 1980 Mt. St. Helens eruption. Major hydro projects in the Cowlitz and Lewis basins have blocked access to approximately 80% of the historical steelhead spawning and rearing habitat within both basins.

The NPCC's smolt production model was applied to many systems within the lower Columbia River in the early 1990s to estimated steelhead production potential. Smolt production estimates were 45,300 winter steelhead smolts in the Grays River; 63,399 winter steelhead smolts in the Cowlitz River; 38,229 winter steelhead smolts in the Coweeman River; and 135,573 winter steelhead smolts in the Toutle River. In the Kalama basin, WDW estimated that potential winter and summer steelhead smolt production was 34,850; from 1978–84, the number of naturally produced steelhead smolts migrating annually from the Kalama ranged from 11,175 to 46,659.

In a recent status report of steelhead in the Lower Columbia River ESU (unpublished), the TRT compared the potential historical habitat available to steelhead to the potential current available habitat. For the entire Lower Columbia River ESU (including Oregon basins), 63% of the historical available steelhead habitat is available today. Most basins have over half of the historical habitat still available; some basins still have the majority of the historically available habitat (e.g. Columbia Gorge tributaries winter steelhead [100%], Kalama River winter steelhead [92%], SF Toutle River winter steelhead [89%], and Salmon Creek winter steelhead [88%]). Some notable exceptions—where very little historical habitat remains available to steelhead—include Cispus River winter steelhead (0%), Tilton River winter steelhead (0%), Upper Cowlitz River winter steelhead (2%), and NF Lewis River winter steelhead (22%).

2.4.10 Listing Status

NOAA Fisheries BRT concluded that the Southwest Washington steelhead ESU (which includes Columbia River populations downstream of the Cowlitz River and Grays Harbor and Willapa Bay tributary stocks) is not currently in danger nor is it likely to become endangered in the foreseeable future (Busby et al. 1996). Therefore, the Grays, Elochoman, Skamokawa, Abernathy, Mill, and Germany populations are not listed under the ESA. However, the BRT decision reflects the overall condition of the entire ESU and does not necessarily reflect the condition of each lower Columbia population within the ESU. All of the Columbia River populations in the Southwest Washington ESU were categorized as depressed by WDFW in 2002, with the exception of Mill Creek, which was listed as unknown.

The BRT concluded that the Lower Columbia steelhead ESU (which includes steelhead from the Cowlitz River upstream to the Wind River) is not presently in danger of extinction, but is likely to become endangered in the near future. Therefore, on March 19, 1998, NOAA Fisheries issued a formal notice listing the Lower Columbia steelhead ESU as threatened under ESA (Fed. Reg., V63, N53, p.13347).

WDFW categorized Kalama steelhead status as healthy, while Coweeman, NF Toutle/Green, SF Toutle, and Washougal steelhead were categorized as depressed, and NF Lewis, EF Lewis, Salmon Creek, Bonneville tributaries, and Wind River steelhead status were categorized as unknown.

The overall status of lower Columbia steelhead populations is generally low, but sustained natural production has been maintained in most areas in which steelhead were historically present. The most notable exceptions include areas in the Cowlitz and Lewis rivers where hydro development has blocked passage, and areas of the NF Toutle drainage where habitat was devastated by the eruption of Mt. St. Helens in 1980.

2.5 Bull Trout *(Salvelinus confluentus)*

Bull trout (*Salvelinus confluentus*) are a distinct species (Cavender 1978) that were previously considered to be a single species with Dolly Varden *(Salvelinus malma)* because of their overlapping ranges, similar appearance, and lack of sufficient analysis to discern the two species. Several genetic studies of the genus Salvelinus confirm the distinction between the bull trout and Dolly Varden (Phillips et al. 1989, Crane et al. 1994), and in fact show they are more closely related to other char species than to each other (Phillips et al. 1989, Phillips et al. 1991).

2.5.1 Life History and Requirements

Bull trout exhibit resident, freshwater migratory, and anadromous life history patterns (Rieman and McIntyre 1993; Figure 2-25). Resident and migratory forms are known to coexist in the same subbasin or even in the same stream. While it is unknown whether resident and migratory forms of bull trout can produce progeny exhibiting the alternate life history behavior, multiple life history forms of other char are known to give rise to one another (Rieman and McIntyre 1993).

Resident forms live out their lives in the tributary where they were born and in nearby streams. Freshwater migratory forms include both fluvial and adfluvial strategies (Fraley and Shepard 1989). The fluvial form migrates between main rivers and tributaries. The adfluvial form migrates between lakes and streams. Anadromous forms have been reported (WDFW 1997) in certain coastal areas, probably occur in the Puget Sound drainages and in the Squamish River, and may have occurred historically as far south as the Puyallup River (McPhail and Baxter 1996). Confirming the existence of anadromous bull trout populations is difficult because of the geographic overlap with Dolly Varden and the difficulty in discerning between the two species. In the lower Columbia River, bull trout may exhibit resident or freshwater migratory life history patterns; anadromous bull trout have not been observed.

Bull trout are found primarily in cold streams. Researchers consistently find that water temperature is a principal factor influencing distribution of bull trout in many streams (Rieman and McIntyre 1993, Baxter and McPhail 1996). Fraley and Shepard (1989) observed that water temperature above 59°F (15°C) may limit bull trout distribution. Studies in the John Day basin found bull trout present when maximum summer temperatures were 16°C or below, and maximum densities occurred where temperature maxima were 12°C or below (Buchanan and Gregory 1997). Bull trout do not compete well with introduced salmonids in degraded habitats (McPhail and Baxter 1996).

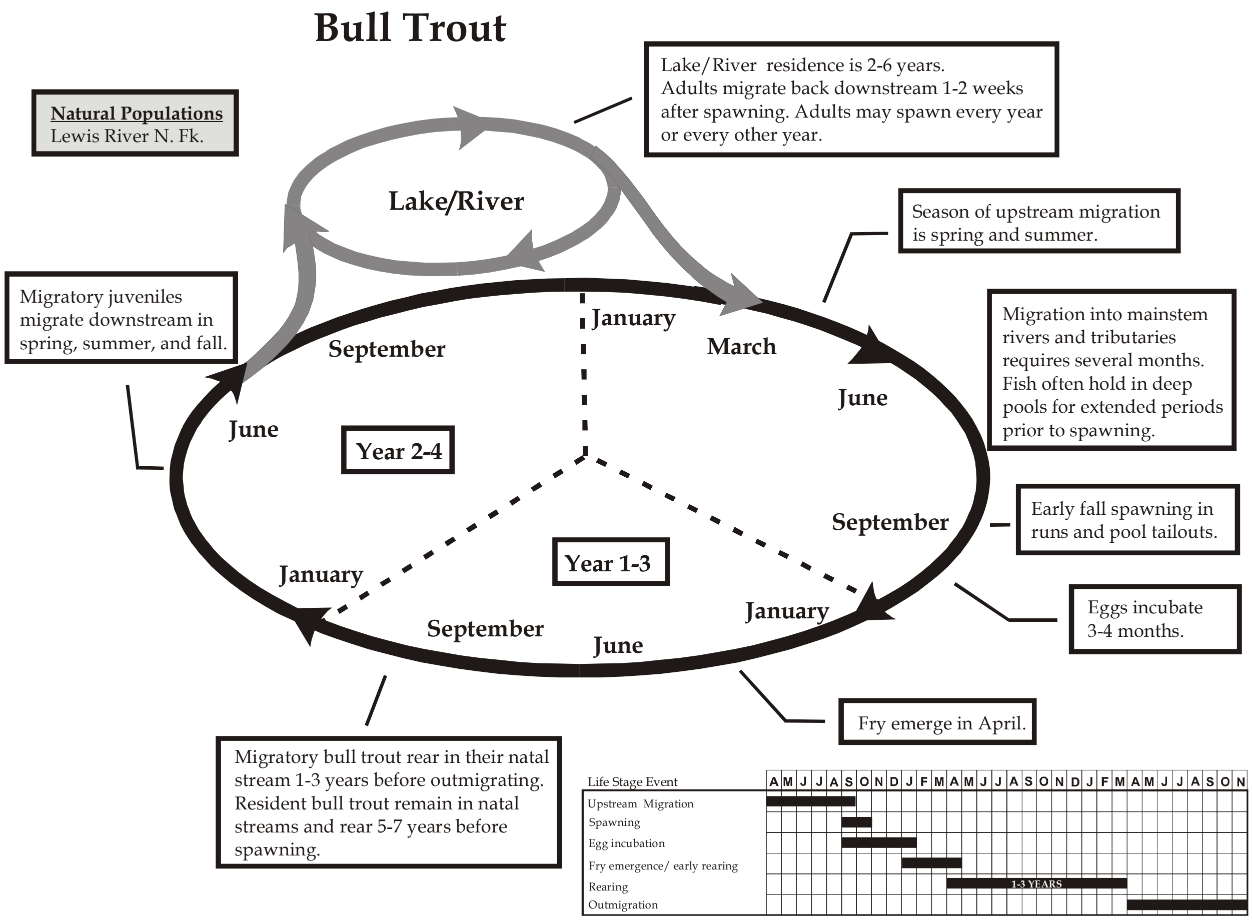

Figure 2-25. Bull trout life history.

2.5.1.1 Migration and Spawn Timing

Adults typically spawn from August to November during periods of decreasing water temperatures. Water temperature may be the proximate cue that initiates reproductive behavior. 48°F (9°C) appears to be the threshold temperature above which no spawning will occur (McPhail and Baxter 1996). Upstream migration begins in April and peaks during high flows in May and June (Pratt 1992). Spawners migrate upriver slowly, mostly at night, and enter tributary streams from late July through September (Pratt 1992). The above statements are broad generalities, and in fact migration patterns within and across basins can be varied and complex.

2.5.1.2 Spawning

Bull trout have spawning habitat requirements that may be more specific than those of other salmonids (Baxter and McPhail 1996). Spawners prefer areas of groundwater infiltration (Fraley and Shepard 1989, Baxter and McPhail 1996). Redds are relatively large, typically measuring about 1.5m x 2m (59 x 78 in) (Fraley and Shepard 1989, McPhail and Baxter 1996). Redd site selection across years may be remarkably consistent, and superimposition of redds has been observed (Baxter and McPhail 1999). Baxter and McPhail (1996) suggest that maintaining the quality of specific areas where high redd concentrations occur across years may be critical for survival in some populations. Buchanan et al. (1997) reported redd densities in bull trout spawning areas in the Umatilla and Walla Walla basins of 1.4- 8.6 redds/km, in study sections 5-16 miles (7.5-25.5 km) long.

Redd building and courtship behaviors occur mainly at night but have been observed during the day (McPhail and Baxter 1996). Often, only a single male is involved in mating, but jacks have been observed to surreptitiously mate with a female with which they have not courted (McPhail and Baxter 1996). It is likely, however, that reports of jacks may include cases of smaller, resident males mating with larger migratory females in those areas where the ranges of the two different life history forms overlap (McPhail and Baxter 1996). Because of different maturation rates and potential overlap of different life history patterns, it is possible to have four or more year classes compose any spawning population and as many as 12 to 16 age combinations in any spawning year (Shepard et al. 1984).

Preferred spawning habitats include stream reaches with loose clean gravel and cobble substrates, and temperatures 41-48°F (5-9°C) in late summer and early fall (Fraley and Shepard 1989, Goetz 1989). Optimal water depths for spawning are 12-24 in (30-60 cm) (Boag 1991, Baxter and McPhail 1996). Bull trout spawning areas are generally higher in the watershed than other salmonid species.

Some studies suggest that optimal water velocity for spawning bull trout are 0.33-1.6 ft/sec (10-49 cm/sec), with velocities greater than 2.3 ft/sec (70 cm/sec) unsuitable for spawning (Boag 1991, Baxter and McPhail 1996).

Repeat and alternate year spawning have been reported. Frequency of repeat spawning is not well documented (Fraley and Shepard 1989, Pratt 1992, Rieman and McIntyre 1996). Fraley and Shepard (1989) reported that 38-69% (average 57%) of adult sized bull trout stayed in Flathead Lake each year. These were presumed to be fish that skipped a year of spawning. Mushens and Post (2000) found that an average of only 13% (range of 9-17%) of spawners in Smith-Dorien Creek had skipped a year of spawning. These are adfluvial bull trout from lower Kananaskis Lake, Alberta.

There is also limited observations of post-spawning mortality rates, although it is generally presumed to be relatively low. Mushens and Post (2000) reported spawning related mortalities in Smith-Dorrien Creek of 0.7-5.2% for 4 years of observation.

2.5.1.3 Incubation and Emergence

Incubating and emergent bull trout require colder water than other salmonid species. Cool water during early life history results in higher egg survival and fry growth rates (Pratt 1992, McPhail and Murray 1979, Shepard et al. 1984). McPhail and Murray (1979) found that bull trout fry grew to larger sizes at lower temperatures, and reached largest size at 39°F (4°C). Goetz (1989) suggested optimum water temperatures for bull trout incubation of about 35-39°F (2-4°C).

Eggs are about 0.197-0.236 in (5-6 mm) in diameter (McPhail and Baxter 1996). Egg numbers deposited in redds increase with body size of females, and as few as 74 eggs to as many 6,753 eggs have been documented (see McPhail and Baxter 1996 and sources cited therein). Incubation is 100–145 days (Pratt 1992); development rate is temperature-dependent, but is also related to egg size, especially at low temperatures (Murray 1980, McPhail and Baxter 1996). Bull trout require around 350-440 thermal units (TUs) after fertilization to hatch (Weaver and White 1984, Gould 1987). McPhail and Murray (1979) investigated the relationship between egg survival and water temperature. They reported that at water temperatures of 46-50°F (8-10°C), 0-20% of eggs survived to hatching. At 42°F (6°C), 60-90% survived, and at 35-39°F (2-4°C), 80-95% survived.

The two major causes of egg mortality are siltation and freezing (McPhail and Baxter 1996). Weaver and White (1985) reported a negative relationship between intergravel fines and incubation survival in laboratory tests. Approximately 40%, 20%, and 1% of fertilized eggs survived to hatch when spawning substrate consisted of 20%, 30%, and 40% fines, respectively (fines defined as <0.37 in [9.5 mm] in diameter).

After hatching, juveniles remain in the substrate for up to 3 weeks before emerging from the gravel, and emergence may take place up to 200 days after the eggs have been deposited. Emergence is normally April–May, depending on water temperature and flow patterns (Pratt 1992, Ratliff and Howell 1992). Fraley and Shepard (1989) estimated that 50% of eggs survive to emergence of fry (Pratt 1992, Gould 1987). Size at emergence is usually around 1.0-1.1 in (25-28 mm).

2.5.1.4 Freshwater Rearing

In laboratory experiments, newly emerged fry did not fill their swim bladder for 3 weeks after emergence, were strongly bottom-oriented, and spent a great deal of time in the small spaces between pieces of gravel at the bottom of the water (McPhail and Baxter 1996).

Juvenile bull trout are associated with complex cover, including large wood, undercut banks, boulders, and pools (Fraley and Shepard 1989). In general, juvenile bull trout are associated with shallow water depths with good cover, near faster-flowing water that delivers food particles (Baxter and McPhail 1996). Fry stay close to the streambed; McPhail and Murray (1979) suggest this might be an adaptation to avoid being carried downstream before the fry are large enough to take up residence in a suitable feeding site. Mean distance above the stream bed increases as the fish get larger (Pratt 1984), although they tend to remain in the bottom 25% of the water column (McPhail and Baxter 1996). Age 0+ fish tend to hold in water depths of 0.1-1.5 in (2-40 mm) (Baxter and McPhail 1996, Baxter 1995, Tredger 1979, Ptolemy et al. 1977).

Fry frequently inhabit stream margins and side channels (Sexauer and James 1997). Martin et al. (1992) reported that age 0 bull trout densities in Mill Creek, Oregon, were highest in riffle- and cascade-type habitats and in the presence of woody debris. Goetz (1997) reported that age 0 bull trout were active at day while age 1 and older were most active during twilight. Paul (2000) found that age 0 bull trout were absent during their nighttime sampling in Smith-Dorrien Creek, Alberta.

Young bull trout exploit small pockets of slow water near higher velocity, food-bearing water (Shepard et al. 1984, Pratt 1992). These microhabitats may be created by cobble substrate, and Shepard et al. (1984) found highest densities of juvenile bull trout in reaches with highest cobble substrate percentages in the Flathead River basin. Densities decline as the small spaces between gravel are filled with fine particles (Enk 1985). Where unembedded cobble substrate is not available, woody debris, turbulence, and undercut banks take on increasingly important roles in providing suitable habitat (Pratt 1984). In general, complex forms of cover and high stream complexity are favored by young bull trout (Baxter and McPhail 1996).

Significant shifts in habitat use from one day to the next have been observed (Goetz 1994, Sexauer 1994, Baxter and McPhail 1996). At night, most bull trout juveniles observed are not associated with cover.

Juvenile bull trout < 4.3 in (110 mm) most commonly consume aquatic insects and fish. Once they are approximately 4.3 in (110 mm) long, they may begin feeding on smaller fish (Pratt 1992), and may consume prey items that are large in relation to their own body size. McPhail and Baxter (1996) report observing a 1.8 in (45 mm) rainbow trout in the stomach contents of a 3.5 in (90 mm) bull trout. Cannibalistic behavior is common. Fish species identified in the stomachs of juvenile bull trout include mountain whitefish, sculpins, salmon fry, and trout, including other bull trout (McPhail and Baxter 1996). Boag (1987) reports increased piscivory in adfluvial populations, as larger fish prey more exclusively on fish as they move downstream into larger water.

Juvenile and adult bull trout densities are typically low, and the species may be more sensitive to environmental degradation than other salmonids (McPhail and Baxter 1996). Late summer densities of bull trout ages 1-3 in pool and glide habitats in Jack, Roaring, Brush, Canyon and Candle creeks tributarie of the Metolius River, Oregon, were estimated to range from 2.0 to 20.6 fish per 100 square meters (Buchanan et al. 1997). In Mill Creek, juvenile bull trout densities were highest in plunge pools with woody debris (8.7 fish per 100 m^2) and run habitat with woody debris (8.4 fish per 100 m^2) (Martin et al. 1992). Fraley and Shepard (1989) and McPhail and Murray (1979) found juvenile bull trout densities to be higher in pools than in other habitat unit types. Carrying capacity of streams for juvenile bull trout is thought to be the major bottleneck in production (McPhail and Murray 1979, McPhail and Baxter 1996).

Paul (2000) found that densities of fluvial bull trout juveniles in Eunice Creek, Alberta over a period of 15 years fluctuated over two orders of magnitude (0.06 fish/100 m^2 to 19.57 fish/100m^2). During the same time, rainbow varied only from 0.45 fish/100m^2 to 3.29 fish/100m^2. Few fish were over 9.8 in (250 mm). The study also investigated interactions of juvenile bull trout and their role in population dynamics. The author used the sum of fork length squared as a measure of effective density (Walters and Post 1993, Post et al. 1999), because consumption rate is an exponential function of body size, with fish consuming less per body weight as they increase in size. He found that survival of age 1 and 2 juveniles was highly correlated to effective density (FL^2/m^2). The model was significantly improved by adding mean discharge (April-October), with a positive effect. Survival of age 3 bull trout was best correlated to density of age 3 bull trout. Survival of age 2 bull trout was positively related to summer flow. Density effects reduce survival of older juveniles up to 60%.

Paul (2000) also found significant negative relationship of juvenile bull trout growth to effective density in a resident bull trout population in Prairie Creek, Alberta. Growth rate was depressed more by abundance of larger juveniles than by abundance of smaller juveniles. The author concluded that competition for food occurred among all juveniles age 1 or older. This differs from other salmonids that tend to partition food between age groups as the fish exploit different habitat zones.

Ratliff et al. (1996) found no apparent relationship between redd counts and densities of age 1-3 juveniles in five spawning tributaries of the Metolius River, suggesting density dependence mechanisms affect juvenile abundance. Paul (2000) used experiments with stream enclosures to demonstrate that growth of age 1 bull trout was density-dependent. Over 42 days of the experiment, there were no differences in survival among treatments, but highly significant ($P<0.01$) differences in growth. Further, he found positive relationship between fish size at the beginning of experiment and survival to the end of experiment. Survival averaged 52% over 42 days for 4.33 in (110 mm) bull trout. He concluded that overwinter survival rate was determined by growth rate, which was determined by effective density. Larger fish at age survived better than smaller ones, and dominant age classes could be produced if larger fish were depleted, allowing rapid growth of age 0 fish and resulting in high overwinter survival. Survival rates through winter were based on bioenergetic modeling of lipid stores, which increase with body size. Thus, larger fish with more body fat withstand longer periods of starvation.

2.5.1.5 Juvenile Migration

Migratory juvenile bull trout typically rear in their natal streams for 2 to 3 years before migrating downstream. Although juveniles migrate in all months, most migration peaks in May and June. Migrating juveniles average about 8 in (200 mm) long. Juvenile bull trout migrate from the streams in which they are born to larger rivers and lakes throughout their range at ages 1, 2, and 3 (Pratt 1992).

2.5.1.6 Estuary Rearing/Ocean Migrations

Although bull trout are known to exhibit anadromous life history patterns, anadromy has not been observed in lower Columbia River bull trout. Thus, bull trout in the lower Columbia River do not have an estuarine rearing or ocean migration phase as part of their life cycle (Figure 2-25).

2.5.1.7 Adults

Size and age at maturity vary depending if the fish is resident, freshwater migratory, or anadromous. At maturity, resident fish are generally smaller and less fecund than migratory fish (Fraley and Shepard 1989). Bull trout normally reach sexual maturity from age 4–7, and may live longer than 12 years.

Because bull trout can be resident, freshwater migratory, or anadromous, it is not a simple task to come up with generalized habitat requirements for adult bull trout, but some themes are common. Bull trout are a cold water species. Bull trout are seldom found in areas where water temperatures frequently exceed 59°F (15°C). Forms of cover favored by adult bull trout include deep pools. Usually, adult fish migrate into a stream during spring or early summer freshets and may reside in deep pools up to 2 months before spawning (Baxter and McPhail 1996). This tendency makes adult bull trout particularly vulnerable to poaching or overfishing (McPhail and Baxter 1996).

Resident populations of bull trout usually are separated from other populations by some physical or thermal barrier (McPhail and Baxter 1996). Where present, resident populations are typically found in headwater streams in mountainous areas, and in higher gradients than other forms. They are usually associated with deep pools and complex cover, and are much smaller than individuals that migrate into larger rivers during adulthood. Resident fish average about 7.87 in (200 mm) in length, and mature from 1 to 2 years earlier than migratory fish in the same geographic area (McPhail and Baxter 1996). Suitable overwinter sites are critical to the viability of stream resident populations. Research has found that the most suitable overwinter habitats are areas of groundwater upwelling and deep pools (McPhail and Baxter 1996).

Fluvial forms of bull trout live as adults in large rivers but return to small tributary streams to spawn. Fluvial individuals usually reach sexual maturity by age 5, and can attain large sizes. Individuals up to 35 in (900 mm) have been reported (Baxter 1995, McPhail and Baxter 1996). In many instances, fluvial bull trout densities in larger rivers are higher near the mouths of smaller spawning tributaries that deliver colder water to the system (Buckman et al. 1992). Fluvial adults are generally associated with deep pools and instream cover (Shepard et al. 1984). In some systems where water is more turbid, adult bull trout are less associated with cover and more widely distributed, perhaps reflecting their status as top predators (Bishop 1975, McPhail and Baxter 1996).

Adfluvial bull trout live as adults in lakes and return to small tributaries to spawn. In some cases, spawning may occur in the lake outlet. Spawning migrations may be quite short, or as long as 124 miles (200 km) (Fraley and Shepard 1989). Evidently, lake dwelling adult bull trout may use different parts of lakes at different times of the year. In Flathead Lake, Montana, Goetz (1989) reported that bull trout forage in the littoral zone in fall and spring and move to deeper water in summer, likely because of temperature considerations. Sexual maturity is usually reached by age 5, and individuals can attain large sizes (up to 27.5 in [700 mm] long; McPhail and Murray 1979).

Beauchamp and van Tassell (1999) conducted a thorough diet study of an adfluvial bull trout population in Round Butte Reservoir on the Deschutes River. Kokanee, bull trout, rainbow trout, mountain whitefish, other salmonids, nonsalmonid fishes, and invertebrates were all important in adult bull trout diets, with diets changing seasonally and by size class of bull trout. Small bull trout (FL < 300mm) consumed primarily age 0 mountain whitefish in all seasons, as well as age 0 kokanee during summer; length of consumed prey items increased in a seasonal progression. Subadult bull trout (FL 300-450mm) ate age 0 kokanee during summer and fall and transitioned to age 1 kokanee during the winter and spring; intermediate sized mountain whitefish (~150mm) were an important food item during summer. Adult bull trout (≥ 450 mm) ate age 0 and age 1 kokanee from winter through summer but shifted to age 2 and age 3 kokanee during the fall; mountain whitefish were also an important prey item during the winter and spring. Fraley and Shepard (1989) sampled bull trout stomachs in Flathead Lake during November and January and found that kokanee composed 8.9% of the diet by weight, while various species of whitefish composed 48.1% and non-game fish composed 22.1%.

Beauchamp and Tassell (1999) used bioenergetics modeling to estimate the total predation impact by age 3 to 7 bull trout in Round Butte Reservoir. They estimated that the adult bull trout population annually consumed 11-49% of the available age 0 bull trout, 4-18% of the age 1 bull trout, 5-11% of the age 0 kokanee, 1-2% of the age 1 kokanee, and 13-74% of the age 2-3 kokanee. Larger bull trout ate larger prey, resulting in adults contributing the most to kokanee predation losses while subadults contributed the most to juvenile bull trout cannibalism losses.

2.5.2 Distribution

The Columbia River basin supports a total of 141 subpopulations of bull trout, 20 located in the lower Columbia River region downstream of the Klickitat River (Figure 2-26). Of these 20 subpopulations, two are located in the Lewis River (Federal Register, Vol. 63, No. 111, June 10, 1998). Bull trout have never been reported in the Wind River above Shipherd Falls (RM 2.0) (Byrne et al. 2001). Bull trout have been reported in the Little White Salmon basin but never above Little White Salmon National Fish Hatchery. Byrne et al. (2001) conclude after sampling in the subbasin that bull trout are not present in Lava Creek of the mainstem Little White Salmon above the hatchery, but could not confirm absence of bull trout in Moss Creek due to equipment failure and extensive instream debris. Reports of White Salmon River bull trout are rare, and it is unclear where preferred spawning areas are located. It is doubtful that bull trout in the White Salmon system venture into the mainstem Columbia due to temperature considerations (WDFW 1998). Byrne et al. (2001) note that groundwater contributions in the canyon area of the White Salmon River and in Spring Creek could make these areas possible bull trout habitat. Bull trout populations were also suspected to historically inhabit the Cowlitz and Kalama subbasins, but the current distribution of bull trout in these subbasins is unknown.

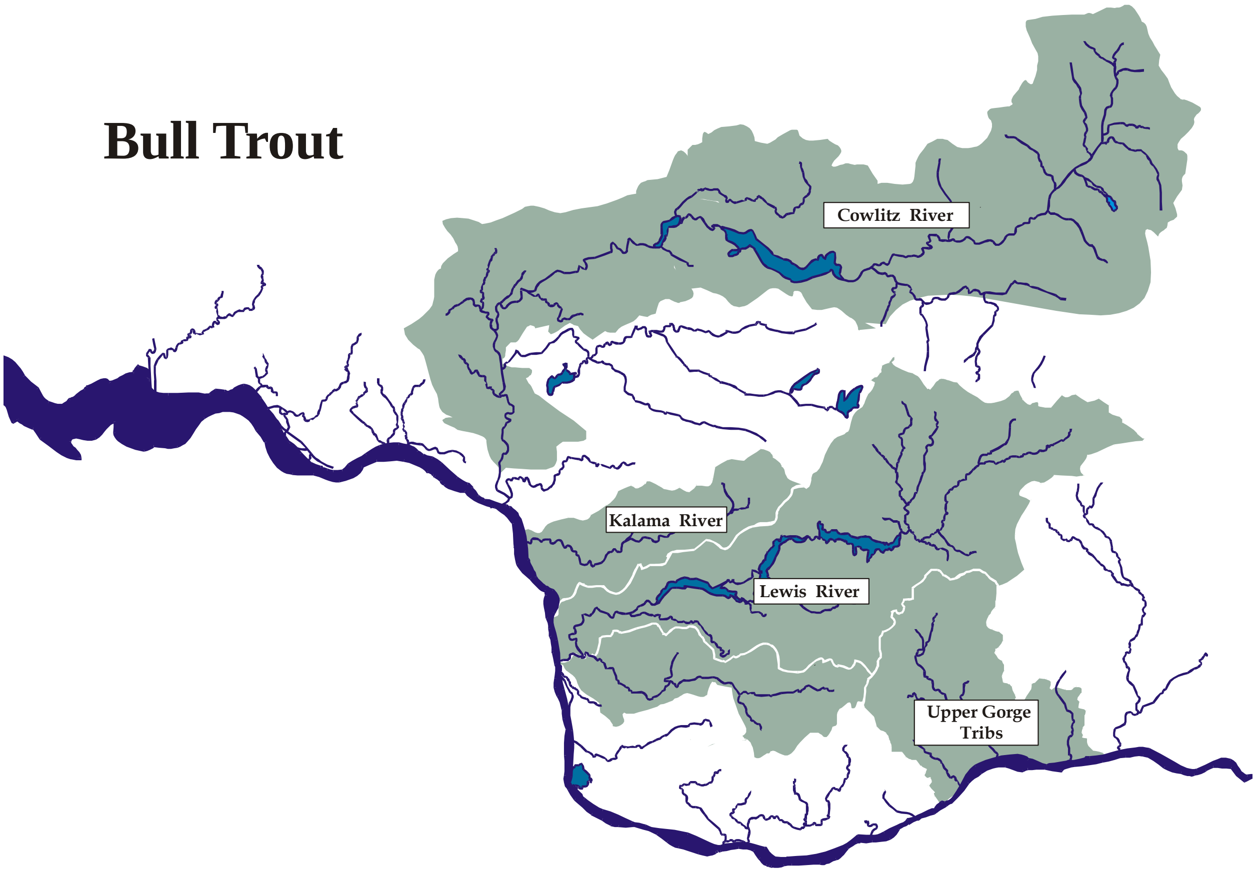

Figure 2-26. Distribution of historical bull trout populations among lower Columbia River subbasins.

2.5.3 Genetic Diversity

Genetic variability within populations is low, but genetic differences among populations are often marked. This suggests that many small populations have undergone genetic bottlenecks (McPhail and Baxter 1996). Genetic samples were taken from bull trout captured in Lake Merwin, Yale Lake, and Swift Reservoir in 1995 and 1996. Analysis showed that Lewis River basin bull trout were genetically similar to the Columbia River population (Spruell et al. 1998). Spruell et al. (2003) conducted microsatellite analysis and concluded that 'coastal' bull trout (west of the John Day River) were genetically distinguished from Snake River and Upper Columbia groups. Within the coastal population, however, some genetic variation was observed, primarily between drainages. Spruell et al. (1998) found that the Swift population was found to be significantly different from that in Yale Lake and Lake Merwin. This implies that there may have been biological separation of the upper and lower basin stocks prior to completion of Swift Dam in 1958.

2.5.4 ESU Definition

Because of widespread distribution, isolated populations, and variations in life history, bull trout populations are grouped by distinct population segments (DPS) rather than ESU. Bull trout are also grouped by recovery units, which serve as subsets of the distinct population segments. By examining distinct population segments, bull trout in most need of Federal protection become a listing priority. On June 10, 1998, the USFWS issued a final rule announcing the listing of bull trout in the Columbia and Klamath river basins as threatened under the ESA (Federal Register, Vol. 634, No. 111).

Within the Columbia River Basin distinct population segment of bull trout, the Lower Columbia River Recovery Unit includes the Lewis River and Klickitat River core areas in Washington. The Lewis River Core Area consists of the mainstem Lewis River and tributaries downstream to the confluence with the Columbia River, with the exclusion of the East Fork of the Lewis River. The Klickitat River Core Area includes the Klickitat River and all tributaries downstream to the confluence with the Columbia River.

2.5.5 Life History Diversity

Bull trout exhibit resident, freshwater migratory, and anadromous life history patterns (Rieman and McIntyre 1993). Resident and migratory forms are known to coexist in the same subbasin or even in the same stream. While it is unknown whether resident and migratory forms of bull trout can produce progeny exhibiting the alternate life history behavior, multiple life history forms of other char are known to give rise to one another (Rieman and McIntyre 1993).

In the lower Columbia River, bull trout may exhibit resident or freshwater migratory life history patterns; anadromous bull trout have not been observed. Confirming the existence of anadromous bull trout populations is difficult because of the geographic overlap with Dolly Varden and the difficulty in discerning between the two species.

2.5.6 Abundance

Status of bull trout is difficult to ascertain because of the lack of commercial harvest, hatchery production, and scarcity of data. The Lewis River bull trout population was classified as depressed because of chronically low numbers (WDFW 1998). Adfluvial populations exist in Yale and Swift reservoirs in the Lewis River system. No fish passage is in place at the dams impounding these reservoirs; bull trout are thought to move downstream during spill events. Swift Reservoir bull trout spawn in Rush and Pine creeks. Cougar Creek is the only known spawning location for bull trout in Yale Reservoir; however, there may be potential for spawning in Ole Creek if flow is augmented. Bull trout in Merwin Reservoir are thought to be present due to spill from Yale Reservoir; however, there is no spawning population in Merwin Reservoir (WDFW 1998). WDFW and PacifiCorp have engaged in a program to relocate bull trout from the Yale tailrace back to Yale Reservoir (Table 2-4).

Table 2-4. Bull trout collected from the Yale tailrace (Lake Merwin) and transferred to the mouth of Cougar Creek (Yale Reservoir) or released back into Yale Reservoir (1995–2000).

Year	No. Collected in Yale Tailrace	No. Transferred to Mouth of Cougar Creek	No. Released Back into Yale Reservoir
1995	15	9	6
1996	15	13	2
1997	10	10	0
1998	6	6	0
1999	6	0	6
2000	7	7	0
Total	59	45	14

* not including recaptures

Historical information describing the abundance and distribution of bull trout in the Lewis River basin is limited. However, the number of bull trout spawners utilizing Cougar Creek has been documented annually since 1979. During this period, the number of adult spawners in Cougar Creek (based on annual peak counts) has ranged from 40 in 1979 to 0 in 1981 and 1982 (Figure 2-27). The low number of spawners observed in the early 1980s may be related to impacts associated with the 1980 eruption of Mt. St. Helens.

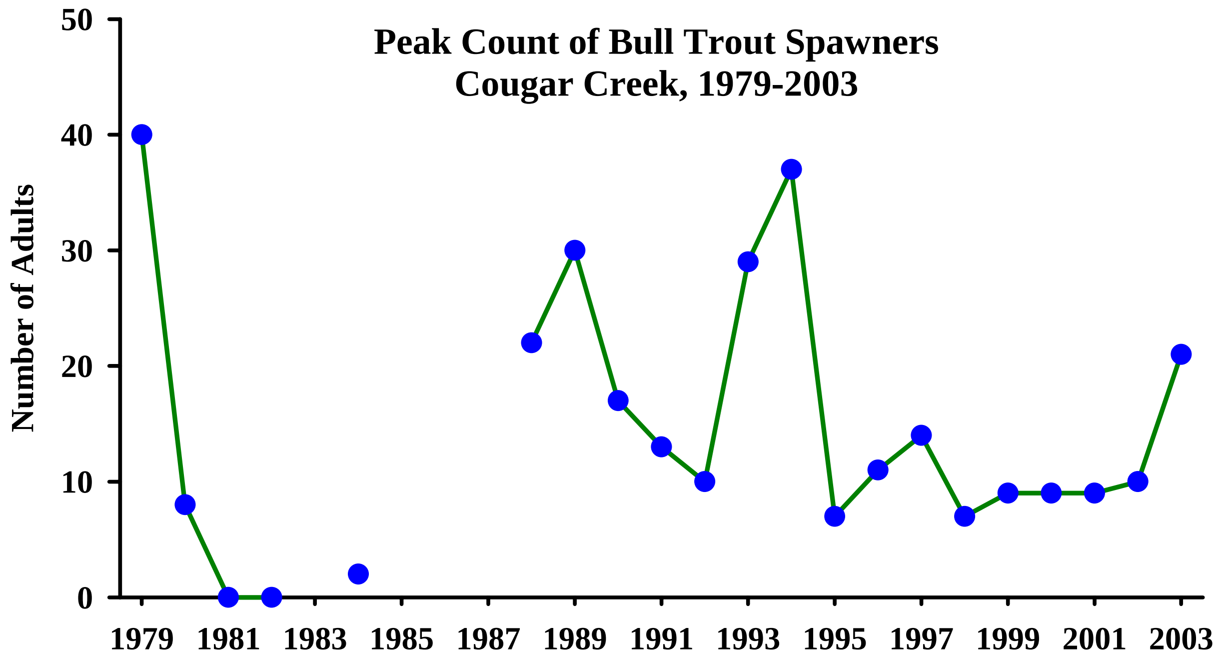

Figure 2-27. Annual peak counts of bull trout spawners observed in Cougar Creek, 1979–2003.

In addition to the survey work conducted in Cougar Creek, the U.S. Forest Service (USFS), WDFW, and PacifiCorp have collected distribution and abundance information on bull trout since the late 1980s. Bull trout collected at the head of Swift Reservoir have been marked with Floy (anchor) tags every spring since 1989 to facilitate mark and recapture counts in Rush and Pine creeks (i.e. the primary spawning tributaries for the Swift bull trout population; Lesko 2001). Between 1994 and 2003, the annual spawner population in Swift Reservoir has ranged from 101 to 911 fish (Figure 2-28; Lesko 2001; personal communication, Dan Rawding and J. Weinheimer, WDFW, 2000).

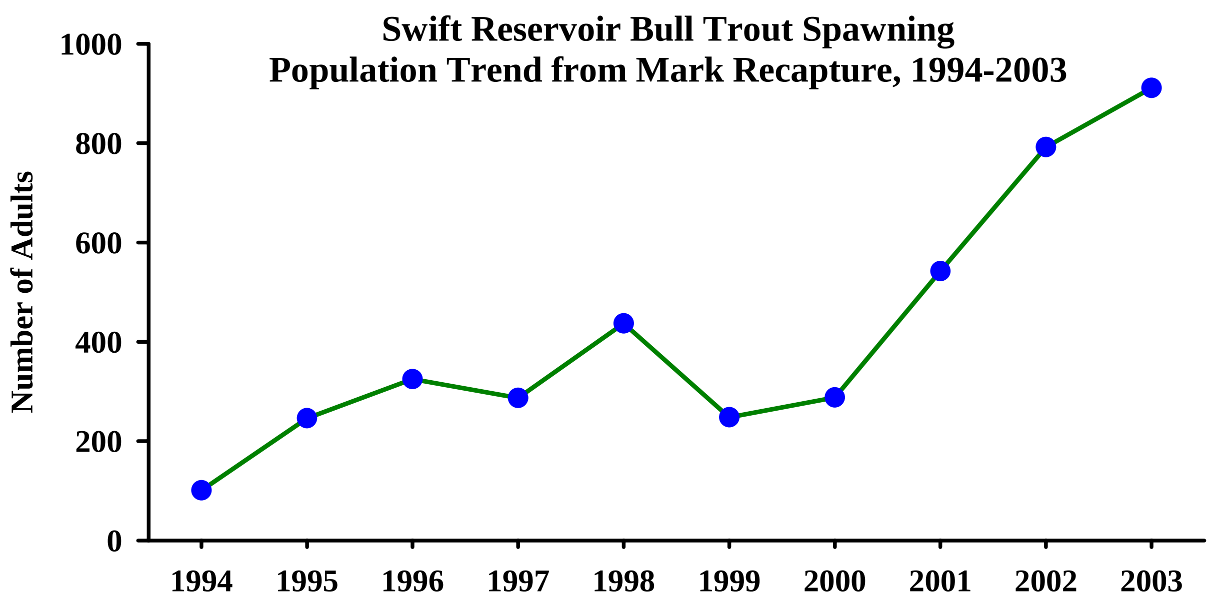

Figure 2-28. Spawning population estimate of bull trout in Swift Reservoir, 1994–2003 (source: Dan Rawding and John Weinheimer, WDFW).

2.5.7 Productivity

The two Lewis basin bull trout populations appear to maintain low but fairly stable production levels in the limited habitat available. The Yale Reservoir production is less certain on an annual basis because of dependence on only one known stream for spawning; thus, a catastrophic event to Cougar Creek would significantly change the productivity of Yale Reservoir bull trout. Additionally, the Yale Reservoir production could be reduced if a significant number are entrained at Yale Dam and displaced to Merwin Reservoir. The Swift bull trout production appears to be more stable, with both Pine and Rush Creek supplying spawning and rearing habitat.

2.5.8 Listing Status

According to WDFW (1998), the bull trout populations in the Lewis River basin are considered at moderate risk of extinction. The bull trout in the coterminous United States was listed as threatened under the ESA on November 1, 1999 (64 FR 58910). Earlier rulemakings had listed distinct population segments of bull trout as threatened in the Columbia River, Klamath River, and Jarbidge River basins (63 FR 31647, 63 FR 42757, 64 FR 17110).

For listing purposes the range of bull trout was broken into distinct population segments. Bull trout occur in widespread, but fragmented habitats and have several life history patterns. In addition, threats are diverse and the population status and trends vary considerably throughout the range. By examining distinct population segments, bull trout in most need of Federal protection become a listing priority. Many of the actions intended to protect other declining salmonids may also help bull trout. Stream and habitat protection and restoration, reduction of siltation from roads and other erosion sites, and modification of land management practices to improve water quality and temperature are all important.

The Bull Trout Recovery Team has developed a draft recovery plan providing a framework for implementing recovery actions in the coterminous United States. Because bull trout are widely distributed over a large area and have differing threats, the U.S. Fish and Wildlife Service identified 27 recovery units based on large river basins and generally following existing boundaries of conservation units for other fish species described in state plans, where possible. Each recovery unit has its own individualized recovery strategy.

Bull trout in the Lower Columbia River Recovery Unit are included in the Columbia River Basin distinct population segment of bull trout. In the two core areas, local populations of bull trout exist in the Cougar, Pine, and Rush creeks (tributaries of the Lewis River) and the West Fork of the Klickitat River. No local populations have been identified in the White Salmon River, but that area contains core habitat and after migratory obstructions are addressed, could support bull trout that migrate from the Columbia River. Additional research is needed to determine if the Cowlitz and Kalama rivers are important for bull trout recovery.

2.6 Cutthroat Trout *(Oncorhynchus clarki clarki)*

The life history of the coastal cutthroat subspecies *(Oncorhynchus clarki clarki)* is probably the most complex and flexible of any Pacific salmonid (Northcote 1997). Cutthroat trout are generalists—they exhibit several life histories and exist in many small streams not suitable for other salmonids. Cutthroat trout are widely distributed in Washington lower Columbia River tributary systems and are not federally listed. Because most individuals are resident or fluvial, cutthroat trout are more affected by local habitat conditions than by mainstem Columbia River and estuary effects.

The subspecies, sea-run cutthroat trout in the Southwest Washington/Northwest Oregon area, was a candidate for listing as threatened, but the USFWS found that a listing was not warranted. Cutthroat have been documented in over 1,300 locations within the lower Columbia distinct population segment. Because coastal cutthroat trout are not a listed species, historical populations and recovery criteria have not been identified by the Willamette/Lower Columbia Technical Recovery Team.

2.6.1 Life History and Requirements

The flexibility of coastal cutthroat subspecies allows the expression of many life history patterns (Figure 2-29). They can rear to maturity in salt or fresh water, migrate large distances, remain in their natal area throughout their life, or exhibit any combination of these behaviors. The following terms define the potential life histories expressed by coastal cutthroat:

- anadromous—fish that migrate to sea during their life,
- fluvial—fish that migrate but remain within a stream until maturity,
- adfluvial—fish that migrate to rear in a reservoir or lake to maturity, or
- resident—fish that rear to maturity near their natal area.

Their diverse life history strategies have enabled the coastal cutthroat subspecies to persist where other salmonid species have not. Isolated above migration barriers in most coastal streams, coastal cutthroat trout are the only salmonid species present, and in small streams, they often are the only species of fish (Connolly 1997, Heggenes et al. 1991b, Glova 1987).

Multiple life history forms frequently coexist in the same watershed and even in the same stream (June 1981, Johnston 1982, Heggenes et al. 1991a, Johnson et al. 1999). Where multiple forms coexist, it is possible for temporal and spatial differences in reproductive behavior to promote genetic differentiation (Zimmerman 1995). It is also possible for some subbasins within a drainage to contain entirely anadromous or entirely freshwater forms (Zimmerman 1995, Johnson et al. 1999). Observed migration patterns suggest that life history patterns may vary within as well as among subpopulations.

There is evidence to suggest that life history patterns may be flexible. For example, research has shown that some sea-run cutthroat may spawn before their first saltwater migration (Giger 1972, Tomasson 1978, Fuss 1982, Johnson et al. 1999). It is evident that not all individuals within a population behave similarly, even if they exhibit the same given life history pattern. Individuals in a cohort may respond to environmental factors differently at any point along the migratory pathway. This suggests that individuals may possess some degree of adaptive flexibility (Dill 1983, Johnson et al. 1999).

Northcote (1997) reviewed the diversity of life-history strategies of coastal cutthroat trout, often in the same basin, and concluded that:

> *"[T]he coastal cutthroat trout has responded to pressures of environmental variability and unpredictability by partitioning its populations into a broad migratory/residency spectrum, 'bet-hedging' its long-term continuity . . ."*

This 'bet-hedging' represents a strength of the species that has enabled it to persist in a large number of streams throughout its range where other salmonids are absent, or where migration has been blocked.

The observed complexity of life history forms of coastal cutthroat trout and the intermingling of various forms within populations, along with the plasticity of individuals within any given life history pattern, make identification of discrete life history types challenging for any single individual or population.

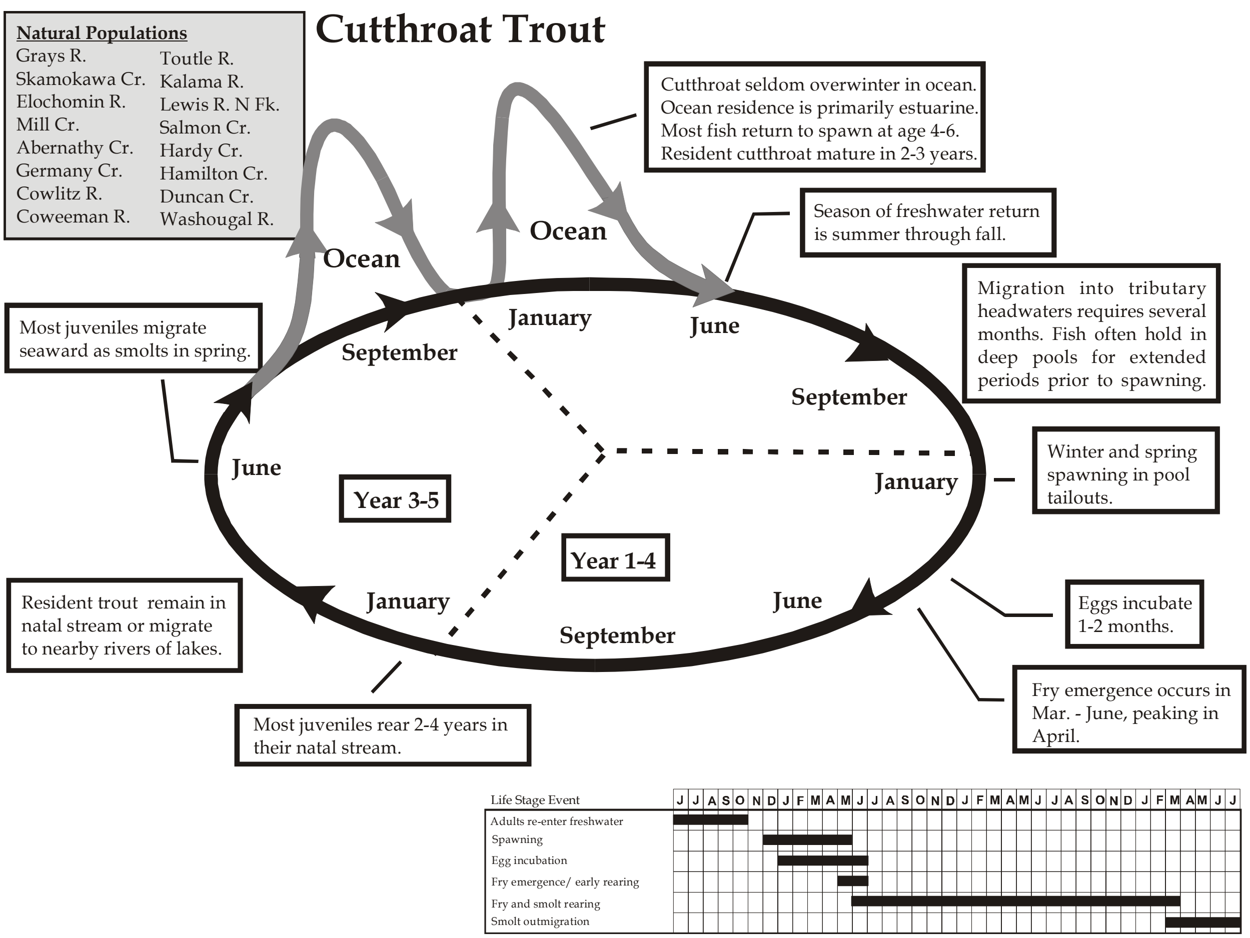

Figure 2-29. Cutthroat trout life history.

2.6.1.1 Upstream Migration Timing

If coastal cutthroat trout exhibit any of the migratory life histories, migration timing to spawning areas is quite variable among populations. Sea-run cutthroat trout may return to spawn from late June through the following April. Re-entry timing is relatively consistent across years within a stream, but varies widely between streams (Giger 1972).

Peak upstream migration of sea-run cutthroat in Oregon coastal rivers is typically in the fall (mid-September through October (Giger 1972)), and they do not overwinter in the ocean. Pearcy (1997) reported that the highest number of sea-run cutthroat appeared in ocean purse seine catches from May to early August. The fall run timing of sea-run fish is confirmed by the peak in late summer in sport catch in Oregon estuaries of sea-run cutthroat. Giger (1972; p.12) concluded from the timing of sport catches in the Siuslaw, Alsea, and Nestucca rivers, "*dates of occurrence of peak catches on the study streams were quite similar, indicating that groups of fish were entering all streams at approximately the same time*." Further, Giger (1972; p.12) who seined only in the Alsea, noted that peak catches by seine and by the sport fishery showed "*reasonably good agreement*." Trotter (1989) reviewed coastal cutthroat life histories along the entire coast of North America, and found that upstream migration in large streams typically begins in July or August and peaks in September to October.

The timing of departure from the estuary on the upstream journey appeared to be related to stream temperatures. Giger (1972) reported that in both 1969 and 1970, the movement of sea-runs out of the Alsea estuary coincided with September's first sharp drop in stream temperatures. Based on observations in three study streams (Siuslaw, Alsea, and Nestucca), Giger (1972; p.19) concluded, "*It was believed to be high temperature that primarily restricted use of the upper estuary sections during July and August*." Thus, sea-run cutthroat appeared to remain in the estuary into the fall until after stream temperatures dropped. Giger (1972) reported that the cutthroat fishery was shortest in the Nestucca estuary and lasted longest into the fall in the Siuslaw estuary. Stream temperatures entering those estuaries were lowest and highest respectively for the three estuaries he studied.

2.6.1.2 Spawning

Coastal cutthroat trout typically spawn from December through June, with peak spawning in February (Trotter 1989). Redds are made mainly in small streams (less than 10 cfs base flow) with low gradients, typically in pool tailouts with water depth of 6-18 in (15-45 cm) (Trotter 1997). Redds are constructed in substrate where particle size ranges from 0.2-2 in (5-50 mm) in diameter (Trotter 1997, Cramer 1940). Coastal cutthroat pairs engage in spawning activity during both day and night, and the ritual can extend up to 3 days (Trotter 1997, Scott and Crossman 1973).

Cutthroat spawn in small headwater streams, generally upstream of all other anadromous salmonids, although some overlap may occur (Johnston 1982). Edie (1975) surveyed tributary and mainstem areas throughout the basin of Washington's Clearwater River in both 1973 and 1974, and found that chinook predominated in the lowermost stream reaches, coho predominated in the intermediate stream reaches, and coastal cutthroat were the most abundant species in the headwaters where gradient is high (2-6%) and channels are small (1-10 feet [0.3-3 m]). Similarly, Johnston (1982) reported that cutthroat spawn in small headwater stream areas upstream from the more dominant salmonids, and that these streams usually average less than 5 cfs during summer. Montgomery et al. (1999) classified channels in Pacific Northwest streams into three categories of gradient: <1%, 1-3%, and >3%; in most streams, stream gradients >3% correlated with the cutthroat-only zone (no chinook, coho or steelhead).

Behnke (1992) noted, "*Where coastal cutthroat and rainbow trout coexist, they are ecologically separated at spawning by the preference of cutthroat trout for smaller tributary streams and of rainbow trout for main river channels*." Moring and Youker (1979) reviewed available data on cutthroat trout in the Willamette basin, and found that most spawning takes place in small streams with flows as low as 0.5-1.0 cfs. DeWitt (1954) surveyed cutthroat trout in streams of the northern California coast, and found, "*Fish-of-the-year were taken only in the very smallest tributaries, usually in those with summer flows less than one cubic foot per second. Most of the cutthroat brood streams examined were too small to be named*."

Magee et al. (1996), investigating west slope cutthroat trout spawning, found in a large Montana river basin that, although suitable spawning gravels were located throughout the basin, most redds were clustered in two areas within first and second order tributaries. Spawning streams were only 3-10 ft wide (1-3 m) with low to moderate gradients (0.5-3.8%). Sumner (1962) found from comprehensive sampling of a small watershed on the Oregon coast that cutthroat spawned in the smallest tributaries available.

Recent studies of west slope cutthroat trout in Montana have indicated that, even though egg survival was greatly depressed by sedimentation, rearing habitat rather than egg survival was the limiting factor to population size. Magee et al. (1996) examined the distribution and particle composition of cutthroat trout redds in the Taylor Fork of the Gallatin River, a sediment-rich basin. Substrate embeddedness in the areas where cutthroat built redds was high, averaging 50%. The cutthroat trout in this study were resident, and average spawner FL was only 7.4 in (189 mm) for males and 7.5 in (191 mm) for females. In spite of the high fine-sediment levels in the redds, and the low survival of eggs expected with so much fines, sedimentation did not appear to limit recruitment. Magee et al. (1996) reached the following conclusion, "*Our results support the theory that resident salmonid populations are typically not limited by reduced spawning success and that recruitment is probably limited by available rearing habitat*."

Coastal cutthroat trout are iteroparous, with high incidences of repeat spawning (Trotter 1997). While some fish spawn each year for at least 5 years, not all fish spawn every year. Some may not return to salt water but rather remain in fresh water for a year or more (Giger 1972, Tomasson 1978). Research documenting significantly larger sizes for spawning females than males—along with anecdotal evidence of large females spawning with smaller, cryptically colored males—suggests that males may be able to exhibit precocious maturation (Johnson et al. 1999).

2.6.1.3 Incubation and Emergence

Incubation period expressed in degree-days ranges from 362-500—usually 6–7 weeks (Trotter 1997, Johnson et al. 1999). A literature review by Bell (1986) reported optimum temperature for egg incubation from 39.9-54.9°F (4.4-12.7°C).

After hatching, developing embryos require 100-350 degree-days before emergence from the gravel (Trotter 1997, Johnson et al. 1999). Emergence occurs between March and June depending on location and time of spawning (Trotter 1997); peak emergence takes place in April (Giger 1972, Scott and Crossman 1973, Johnson et al. 1999). Total length of newly hatched fry is about 1.0 in (25 mm) (Trotter 1997). Fry move to stream margins and backwaters quickly after emergence, where they remain throughout summer (Glova and Mason 1976, Moore and Gregory 1988, Johnson et al. 1999).

Cutthroat fry typically emerge later and at smaller size that other salmonids (Johnston 1982, Griffith 1988). Thus, spatial separations during spawning may be an important evolutionary adaptation to reduce competition for suitable sites and to reduce interaction of young-of-year (YOY) cutthroat with behaviorally-dominant salmonids (Johnson et al. 1999). The adaptation to spawning in small streams also may provide refuge for age 0 cutthroat from competition or predation by older fish of the same species, because several studies have shown that older year classes of cutthroat out-compete younger ones for rearing space. Bisson et al. (1988) found a sharp difference in habitat preference of age 0 cutthroat from that of older age groups that appeared to result from competition. Age 0 cutthroat preferred backwater pools and glides, but avoided pools in the main channel, while age 1 and older cutthroat strongly preferred pools in the main channel. Connolly (1996) found that densities of age 0 cutthroat in pools of 16 coastal streams was negatively correlated to abundance of age 2 and older fish (age 0 abundance increased as age 2 decreased). Connolly (1997) argued that competition for space between age classes—not spawning success—regulated the abundance of cutthroat in coastal streams.

2.6.1.4 Juvenile Rearing

Juvenile rearing of cutthroat trout usually progresses downstream as age increases, except with resident forms that may move only a few meters during their life. Cutthroat fry typically rear for their first year in the small headwater streams where adults spawn, higher in the basin than other salmonids except for bull trout (Sumner 1962, Lowry 1965, Sumner 1972, Edie 1975, Magee et al. 1996).

House (1995) showed large annual fluctuations in cutthroat abundance in a stream of the Molalla River basin that retained stable habitat and no major perturbation events over 11 years, particularly among YOY trout. Abundance of age 2 cutthroat fluctuated 6-fold, while ages 3 and 4 fish fluctuated only 2-fold between years. House (1995) concluded, "*Models that obtain data by separating habitat types and that consider only older age-classes of trout may be the most accurate in predicting changes in population levels*." This finding is consistent with the deduction that space for rearing of age >2 cutthroat was the limiting factor to cutthroat production, because regardless of high or low abundance of age 0-2 fish, abundance of age >2 fish changed relatively less.

Connolly (1996) used a different study format to arrive at conclusions similar to those of House (1995). Connolly (1996) sampled resident cutthroat in 16 coastal streams above barriers, and found habitat units (e.g. pool or riffle) with adult cutthroat present usually had only one or two adults. Further, "*pools with adult cutthroat trout often lacked young-of-year cutthroat trout, and pools without adults often had numerous young-of-year*." Connolly (1996) repeated sampling between years in two streams, and found that abundance of age 2+ cutthroat dropped following the 1992 drought, but abundance of age 0+ fish increased. Connolly (1996) concluded that pool vacancy created a strong opportunity of YOY. Thus, competition between age classes affected their abundance in different habitat types. Similarly, Reeves et al. (1997) report that cutthroat populations were reset to predominantly age 0 in Needle Branch Creek following summers of drought with near zero flow in 1988 and 1992.

As further evidence that pool habitat is a limiting factor to cutthroat production, habitat restoration projects that create new pools have been found to increase production of age 1+ and older cutthroat. Solazzi et al. (1997) created about 10 new pools per km in two test streams on the Oregon coast; he found that production of cutthroat smolts increased 2- to 4-fold relative to control streams. House (1996) examined the effects that placement of instream habitat structures in the East Fork of Lobster Creek, Oregon, had on salmonid production. These treatments significantly increased surface area of pool and low-gradient riffle habitats, and "*treated areas supported significantly more juvenile coho salmon and cutthroat trout and had higher overall salmonid biomass than control areas, whereas age-0 trout (cutthroat trout plus steelhead) and juvenile steelhead showed no increases*."

Other research indicates that habitat for age 0 fish and for overwintering of age 1 and older fish are generally not limiting factors. Solazzi et al. (1997) sampled cutthroat during summer and winter in numerous coastal streams of Oregon, and found no strong preference during winter for any habitat type. This finding suggests there is not a specific habitat in short supply during winter. Chapman and Knudsen (1980) found in paired test and control stream sections that biomass of cutthroat trout per surface area of stream was reduced in test sections (channelized or grazed) for age 1 and older fish, but not for age 0 fish. Thus, the limitation came at age 1 and older rather than age 0. However, Tschaplinski (2000) found in Carnation Creek that scarcity of cobbles large enough to provide winter refuge for yearling steelhead caused steelhead to seek cover in rootwads and pools. These are not the typical winter cover for age 1+ steelhead. Given that steelhead and cutthroat are a minor part of the fish fauna in Carnation Creek, Tschaplinski (2000) reasoned that lack of cobble substrate was a limiting factor responsible for the low steelhead abundance in Carnation Creek. Because cutthroat and steelhead show similar winter habitat preferences, we assume that cutthroat likewise would be limited if cobbles were scarce.

2.6.1.5 Juvenile Migration

Age 1+ parr of migratory types emigrate from their natal stream to rear downstream, primarily in pools (Sumner 1962, Lowry 1965, Sumner 1972, Giger 1972, Edie 1975, Fuss 1982, Bisson et al. 1988, Trotter 1989, Dambacher 1991, Magee et al. 1996). In the fall, some of these fish will find overwinter habitat where they are, and others will migrate back upstream toward their natal area (Sumner 1962, Lowry 1965, Sumner 1972, Giger 1972). The migrating juveniles are referred to as parr, rather than smolts, because they do not undergo the physiological change that prepares them for adaptation to living in salt water. The second spring, age 2+ juveniles again migrate downstream to larger water, some traveling as far as the estuary (Sumner 1962, Lowry 1965, Sumner 1972, Giger 1972, Trotter 1989). These migration patterns by parr are similar for fluvial, adfluvial, and anadromous life histories of cutthroat trout.

Many studies have found a progressive downstream movement of cutthroat each year as they increase in age and size. Sumner (1962) sampled cutthroat trout extensively in the Sand Creek basin on the northern Oregon coast and found,

> *"The downstream movement of initial migrants was by stages. Fingerlings marked in a small tributary of Sand Creek above the traps apparently left their natal stream at the age of 1 year in the usual downstream-migration period and spend a year in the main stream above the rack, some of them passing down through the trap the following spring."*

Similarly, Lowry (1965) studied cutthroat movements in three streams in the Alsea River basin and found that age 1 fish moved downstream from their natal stream throughout the spring and summer to rear in large channels downstream. Trotter (1989), in his compendium of cutthroat life histories reported,

> *"Juvenile cutthroat trout that survive their first winter range more widely than young-of-the-year fish (Giger 1972). Sometimes as early as the winter of their first year,*

but more generally in the spring, many begin downstream movement to the main stem. The onset of winter freshets triggers an upstream movement that often takes the fish back into the tributaries."

Downstream movement of age 1+ and older cutthroat during spring, followed by upstream movement during fall, is typical of both the fluvial and sea-run life histories. Giger (1972) fished a full weir on Crooked Creek, a major tributary of the Alsea River (at RM 54 [86.9 km]), and found that both parr and smolt cutthroat outmigrated in spring. Outmigration past the trap peaked in April, followed by outmigration of smolts from the estuary in mid-May, as determined by seining. Giger (1972) reported, "*Essentially all ocean-destined migrants had left the estuary by the end of May*." Also of note, the last fish to migrate downstream in the spring were the parr, and they were more numerous than smolts throughout April and May. The consistency of this behavior between years indicates that the stimulus to migrate downstream in spring is an inherited trait that affects both parr and smolts. In 1968, 500 of these parr were tagged at the Crooked Creek trap, with many tagged fish captured by anglers in the Alsea River and estuary through the summer. Giger (1972) found that "*One-year-olds were notably absent from the samples taken at Crooked Creek or in the estuary*." Age 1 parr did not move downstream, and nearly all of the migrant parr were age 2 fish greater than 4.3 in (110 mm) long. (Crooked Creek is a large tributary, and cutthroat spawning areas likely would be far upstream from the trap in tributaries or headwater areas.)

Migratory and nonmigratory cutthroat do rear together in the same reaches of a stream, and they can be distinguished only by whether or not they migrate. In one coastal stream where both resident and anadromous cutthroat were studied together, Heggenes et al. (1991a) found that resident cutthroat selected considerably deeper habitats than actively migrating cutthroat. Heggenes et al. (1991a) found that 48% of resident cutthroat moved less than 10 ft (3 m) from their home site in this stream during the 9 months studied (winter to late summer). While most of the population was static, a small fraction was mobile and was recovered at the farthest sampling station over 984 ft (300 m) from their starting point. Observed timing of presence in fresh water indicated "*a substantial proportion of the mobile fraction of the population consisted of residents*" and that others were likely anadromous. Heggenes et al. (1991a) concluded, "*We were unable to tell which fish were anadromous, although the larger fish are likely to be*." These observations underscore that different life-history types of coastal cutthroat do rear in the same streams and are difficult to distinguish.

Smoltification in cutthroat trout and other salmonids involves various morphological, physiological, and behavioral changes. Visual evidence of smoltification includes loss of parr marks and the development of a silvery appearance. Earliest and latest recorded ages of smoltification in sea-run coastal cutthroat are 1 and 6, respectively (Trotter 1997).

Studies of sea-run cutthroat consistently show that size and age at smolting is related to growth rate, which is influenced by habitat suitability, light, nutrients, and temperature. Most sea-run juveniles smolt between ages 2+ through 4+, with faster-growing fish tending to smolt at a younger age (Sumner 1962, Bulkley 1966, Giger 1972, Sumner 1972, Fuss 1982). This consistent finding across many studies indicates that slower-growing individuals must survive more years before smolting (Sumner 1962, Lowry 1965, Sumner 1972, Giger 1972, Fuss 1982, Frissell 1992, Harvey and Nakamoto 1997). This also indicates that slower growth would lead to older age at maturity, which is likely true of either migratory or non-migratory cutthroat trout. Generally, cutthroat that reached 6 in (145 mm) by their second annulus smolted in the spring of that year (Sumner 1972, Giger 1972, Trotter 1997). Fish that reached that size at their first annulus, were generally about 50% longer at age 2 than fish smolting at age 3+ and 4+. Thus, a 50% increase in growth rate would likely reduce average age at smolting by 1 year.

Several lines of evidence indicate that survival is greater for faster-growing individuals. First, faster growth clearly results in younger age at maturity, and since survival is a function of time, those fish maturing in less time face lower mortality. Second, Fuss (1982) noted that back-calculated size-at-age of surviving adult cutthroat (based on scales) typically showed that adult survivors were larger than average at each age compared to juveniles sampled in the stream. Finally, experiments of hatchery sea-run cutthroat trout demonstrate that survival to adulthood is positively correlated to size-at-release. Tipping (1986) differentially tagged cutthroat in 0.4 in (1 cm) length intervals from 6-10 in (16-25 cm) for release in 1982 and again over the range of 7-9 in (18-23 cm) in 1983. Combined returns of these fish to the hatchery and sport fishery showed an exponential increase in survival as smolt length increased up to 8.7 in (22 cm) in 1982 and up to 9 in (23 cm) in 1983. There was no increase in survival as smolt length increased from 8.7-9.8 in (22-25 cm) during 1982, but cutthroat in a natural setting likely would smolt before reaching such lengths.

Giger (1972) noted that parr were recruited into the estuary during spring "*following which individuals became remarkably sedentary*," during summer. Giger (1972) deduced, "*The spring downstream shifting or progression of non-smolting juvenile cutthroat is a logical feature for this species which spends from two to five years in fresh water before migrating to sea.*" Parr captured in the estuary were about 55% age 2 and 40% age 3. Mean length of parr was 6 in (146 mm) while the mean length of smolts was 9 in (231 mm). Because many cutthroat are migratory, the densities of juvenile cutthroat in any given stream section during summer do not reflect production of fish born in that area. Headwater streams will be depopulated during summer following spring emigration of parr, while the larger channels will be populated by fish migrating from upstream.

2.6.1.6 Adults in Freshwater

Several studies have shown that densities of age >2 cutthroat consistently differ between channel unit types, and that stratification of cutthroat densities by channel unit type is a useful starting point for classifying habitat capacity. Cramer (1998) found from snorkeling 90 reaches of small-channel streams distributed throughout the Umpqua basin that age 2 and older cutthroat (> 8 in [> 203 mm]) were predominantly in pools, occasionally in riffles, and rarely in glides. Sleeper (1994) found about 85% of cutthroat were in pools during six snorkel surveys in Cummins Creek, Oregon, where flow ranged from 3.5-475 cfs during the 18 months studied. Sleeper (1994) concluded, "*For the most part, salmonid abundance was directly related to pool size.*"

The dependence of cutthroat production in small streams on pools was further confirmed by an analysis of cutthroat presence or absence in streams above barrier falls in the Umpqua basin. Cramer (1998) found a linear correlation between the percentage of 110 streams above barrier falls that had cutthroat present, and the number of pools available upstream of the barrier. Every stream with more than 52 pools present above the barrier (there were 22 of these) still had fish present, even though these small stream segments had been isolated most likely for thousands of years. With reference to length of stream, every stream with more than 3 miles (5 km) of stream habitat above the barrier (there were 10 of these) had fish present (Cramer 1998). The difference in these measures of habitat availability resulted from differences in gradient and geomorphology between streams (Cramer 1998). The percentage of surface area composed by pools was negatively correlated to stream gradient, and presence of cutthroat above barrier falls also was negatively correlated to stream gradient. All 10 streams with gradients of 2% or less had fish present, and none of the nine streams with 18% gradient or more had fish present. Cramer (1998) found this same gradient limit (about 18%) was operative as well for cutthroat presence in streams without migration barriers.

ODFW extensively sampled rearing densities of anadromous salmonids in various habitat types of Oregon coastal streams during 1985-92, with the intention of developing a habitat capacity model for steelhead and cutthroat (Johnson et al. 1991, 1993). They used multiple pass electrofishing and mark-recapture techniques to estimate fish densities. In order to reduce bias from lack of seeding (spawner densities in each stream were unknown) they excluded streams from their calculation if average fish densities were less than 0.1 parr/m^2 in main-channel pools. From their samples in 30 qualifying streams, densities of cutthroat > 3.5 in (90 mm) long in the main-channel averaged about 0.19 fish/yd^2 (0.16 fish/m^2) in pools, 0.012 fish/yd^2 (0.01 fish/m^2) in riffles, 0.024 fish/yd^2 (0.02 fish/m^2) in rapids, and 0.05 fish/yd^2 (0.04 fish/m^2) in glides (Johnson et al. 1993).

Cramer (1998) found that the proportion of pools with cutthroat > 8 in (20 cm) present increased as maximum pool depth increased. To express this quantitatively, Cramer grouped fish observations for pools in half-foot increments in depth for each subbasin (North, South, and Main Umpqua) and calculated the percentage of pools in that increment with cutthroat > 8 in (20 cm) present. In all three subbasins, Cramer (1998) found no cutthroat > 8 in (20 cm) in pools < 1 foot deep (0.3 m) and the highest frequency of cutthroat in pools > 3 feet deep (0.9 m). Cramer (1998) indicated that the use of pools by cutthroat increases by a factor of about 2.5 for each foot of increased depth.

Similar findings have been reported by Bisson et al. (1988) for third and fourth order streams in western Washington and by Heggenes et al. (1991b) for a coastal stream in British Columbia. Bisson et al. (1988) found that both age 1+ and 2+ cutthroat trout were most abundant in deep pools. Bisson et al. (1988) showed a positive correlation of cutthroat use with pool depth and a similar relationship for age 1+ steelhead. The relationship with age >1 cutthroat showed that the depth preference crossed from negative to positive as depths exceeded about 9.8 in (25 cm). Conversely, age 0+ cutthroat tended to use shallow pools and showed a negative correlation of pool use to pool depth. Further, Bisson et al. (1988) found, "*Average depth did not strongly affect habitat use by underyearlings of any species, but age-1+ steelhead and age-1+ and age-2+ cutthroat trout preferentially used deep pools with ample cover*." Edie (1975) also found that habitat preferences changed sharply between age 0 and age 1 for juvenile steelhead and cutthroat. "*The younger trout make heavy use of the riffle areas of the stream and the older fish utilize the pools*." Edie (1975) distinguished cutthroat and steelhead at age 1 and found the preference for pools was greater among cutthroat than among steelhead, and that most of this difference came from less use of riffles by cutthroat.

Cramer (1998) found that the percentage of riffles with cutthroat > 8 in (20 cm) present was also related to water depth; deeper riffles had a higher probability of cutthroat being present. In the lower Umpqua subbasin, only two of 235 riffles (0.9%) under 0.75 feet deep (0.2 m) had an adult cutthroat present, whereas eight of 50 riffles (16.3%) that were 1 foot (0.3 m) or more deep had adult cutthroat present. In the North Umpqua, where there was more variation in riffle depth, no cutthroat were found in the 80 riffles < 1 foot deep, two (4%) were found in the 70 riffles 1-2 feet deep, and two (20%) were found in 10 riffles that were greater than 2 feet deep. In the South Umpqua, only two of 129 riffles surveyed had cutthroat present and both of those riffles were 1 foot deep. As in the other subbasins, about half of all riffles in the South Umpqua were less than 1 foot deep. Studies in other streams have also shown that cutthroat use of riffles is dependent on depth. Sleeper (1994) conducted snorkel surveys in Cummins Creek, and found that riffles were too shallow for use during summer, but received heavy use where they were sufficiently deep.

Cramer (1998) found that characteristics of each pool and riffle, other than depth, showed no correlation to cutthroat presence. Stream temperature did not help explain variation in the numbers of fish present in streams of the Umpqua basin, and Cramer (1998) observed cutthroat in streams where temperature was recorded at 80°F (27° C) (Calapooyia Creek), although fish were in deep pools that may have been thermally stratified.

Research has shown that inherent differences between watersheds with surface rocks predominantly of basalt (relatively hard) and of sandstone (relatively soft) result in general differences in use of those watersheds by cutthroat trout. Reeves et al. (1997) found that watershed disturbance in basalt watersheds had less deleterious effect on cutthroat trout than in sandstone watersheds. Reeves et al. (1997) cite several studies showing little effect of logging in basalt basins of the Cascade range, but substantial reductions in cutthroat production following logging in sandstone streams of the coast range. They suggest that temperature and sedimentation rates were less in the basalt watersheds so fish could benefit from increased light penetration after shading was reduced. They also point out that basalt streams have more boulders than sandstone streams (i.e. basalt is harder to break down, so particles are larger and slower to transport).

Hicks (1989) found in sandstone streams that substrate was commonly bedrock, while in basalt streams, bedrock was only typical as a lateral boundary of channel units. Basalt is harder than sandstone and likely to result in a greater depth to width ratio than sandstone, thus riffles and glides for a given flow would likely be deeper and narrower in a basalt stream than a sandstone stream. Since cutthroat show a strong preference for greater depth, the combination of more boulders and greater depth may explain the stronger tendency of cutthroat to disperse out of pools in basalt than sandstone streams. Hicks (1989) concluded from his comparison of basalt and sandstone streams:

> *"Streams in basalt had relatively high channel gradients (mean 2.6%) with channel morphology more conducive to production of steelhead, resident rainbow trout, and cutthroat trout than to production of coho salmon. Such streams had large substrate inherently provided considerable habitat stability and complexity."*

2.6.1.7 Estuary and Ocean

Sea-run coastal cutthroat stay close inshore while in salt water, typically within 30 miles (50 km) of the shore (Trotter 1997). Residence time in salt water is typically short. Studies in Oregon and Alaska indicate that coastal cutthroat trout remained at sea an average of only 91 days, with a range of 5–158 days (Giger 1972, Jones 1973, 1974, 1975, Johnson et al. 1999). In these studies, ocean migration patterns appeared similar from year to year, with individuals remaining close to shorelines and rarely crossing open bodies of water wider than 5 miles (8 km).

In some populations, ocean residence periods are spent primarily or entirely in estuarine and tidewater environments (Northcote 1997, Trotter 1997). On the Oregon coast (Giger 1972, Sumner 1972, Tomasson 1978) and in the Cowlitz River (Tipping 1981), adult cutthroat will remain in the estuary throughout the summer, returning to fresh water in the fall. Tomasson

(1978) reported that coastal cutthroat trout in the Rogue River system did not enter the open sea, but remained in the estuary throughout the summer. He posited that sea-run coastal cutthroat trout in the Rogue River may remain in the estuary to avoid predation by half-pounder steelhead that reside in the nearshore ocean in the summer. Even in populations that venture into the nearshore ocean, reports indicate that sea-run cutthroat prefer ocean environments of relatively low salinity and high freshwater influences, such as the nearshore plumes of large rivers (Trotter 1997). The highest incidences of sea-run cutthroat catches in the ocean off the Washington and Oregon coasts have been in areas where temperature averages 56° F (13.4 °C) (Pearcy 1997, Trotter 1997).

Marine survival of sea-run cutthroat trout can be higher than that of other salmonids by as much as 40%, with predation the main cause of mortality in the ocean environment (Giger 1972, Trotter 1997).

2.6.2 Distribution

The coastal cutthroat trout is the most widely distributed and abundant of all cutthroat subspecies. It is native to the coastal belt from Prince William Sound in southeastern Alaska to the Eel River in northern California. In Washington, *O. c. clarki* exists in both sea-run and resident forms, and is found inland to the Cascade Crest (Trotter 1989). Their geographic distribution coincides with the coastal temperate forest belt (Johnson et al. 1999). The subspecies seems to be highly adapted to this region, and even where there is access beyond the coastal forests, they do not penetrate far inland (Sumner 1972, Trotter 1989, Johnson et al. 1999). Anadromous, fluvial, and resident life history forms distribute themselves throughout lower Columbia tributary watersheds (Figure 2-30). Resident forms have been observed throughout the subbasins from the Grays River to Bonneville Dam (WDFW 2000). Anadromous forms exist in all Washington tributaries of the lower Columbia and access to spawning habitat is available in most watersheds. However, upstream migrants are excluded from upper tributary reaches in each subbasin, where steep gradients and high flows can limit passage (WDFW 2000).

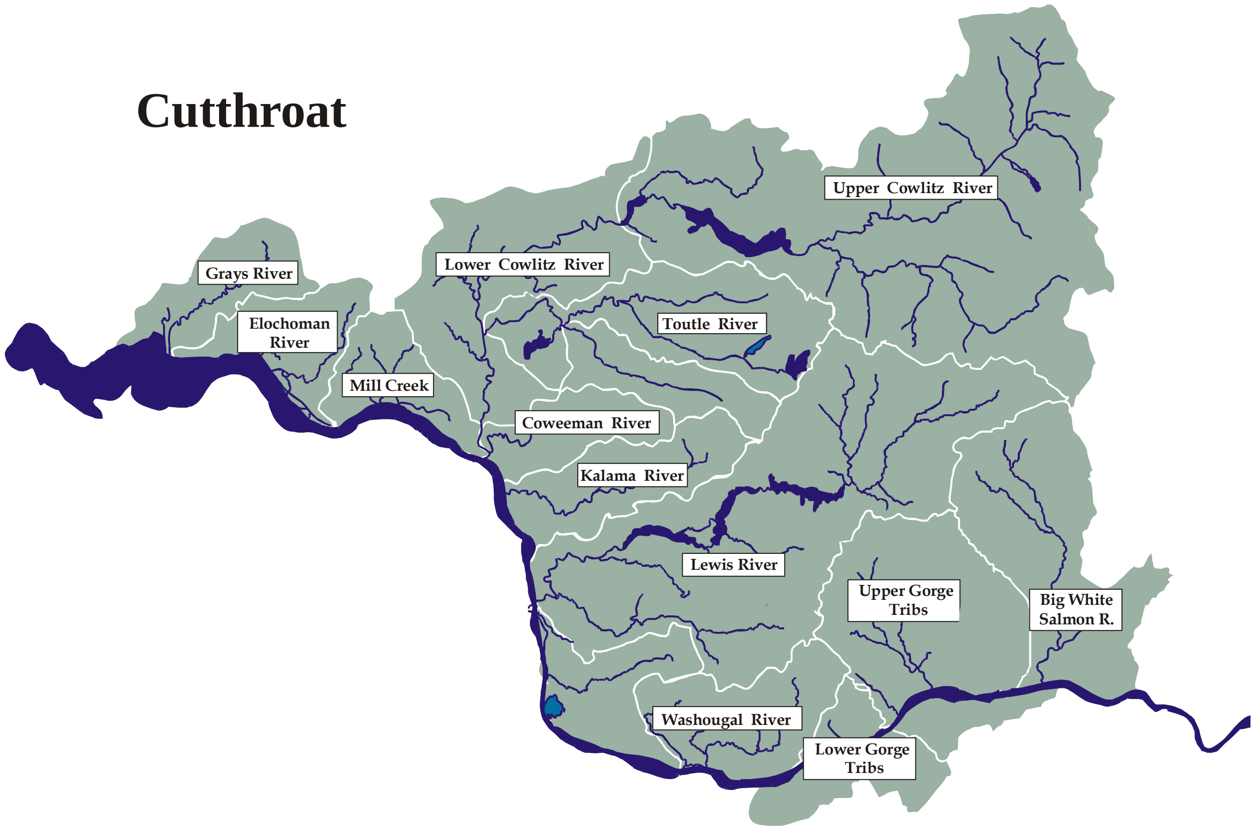

Figure 2-30. Distribution of historical cutthroat trout populations among lower Columbia River subbasins.

2.6.3 Genetic Diversity

Behnke (1992) posits that cutthroat trout are native to western North America and, along with rainbow trout, broke off from a common trout ancestral form in the Snake and Columbia River basins approximately 2 million years ago. About 1 million years ago, the coastal cutthroat subspecies diverged and has remained intact, colonizing rivers from northern California to Alaska (Behnke 1997, Johnson et al. 1999).

Coastal cutthroat trout differ from other anadromous Pacific salmonids in that they exhibit a greater level of genetic diversity among local populations (Johnson et al. 1999). This may be a result of greater reproductive isolation between groups, higher levels of genetic drift in smaller populations, or a combination of the two. Though different populations of coastal cutthroat trout may consist primarily of one life history type or another, genetic research suggests that different life history forms do not represent different evolutionary lineages (Johnson et al. 1999).

Hybridization between coastal cutthroat trout and rainbow/steelhead is widespread (Johnson et al. 1999). About one-third of coastal cutthroat trout samples collected by researchers in British Columbia, Washington, Oregon, and California contained hybrids (Johnson et al. 1999). Cutthroat-rainbow hybrids tend to have intermediate morphological and behavioral characteristics, and may have lower fitness as a result (Johnson et al. 1999). Most hybrids detected with molecular methods are 0+ and 1+ fish, and adult hybrids are not often observed. Still, the presence of introgressed individuals in some populations suggests that at least some hybrids can mature and reproduce successfully.

2.6.4 ESU Definition

The Southwestern WA/Columbia River coastal cutthroat ESU includes naturally spawning populations (and their progeny) below natural barriers in the Columbia River and its tributaries downstream from Klickitat River (WA) and Fifteenmile Creek (OR), inclusive, including Willamette River downstream from Willamette Falls, and in coastal drainages between Columbia River and Grays Harbor (WA), inclusive.

2.6.5 Life History Diversity

The life history of the coastal cutthroat is probably the most complex and flexible of any Pacific salmonid. Cutthroat trout are generalists—they exhibit several life histories and exist in many small streams not suitable for other salmonids. Isolated above migration barriers in most coastal streams, coastal cutthroat trout are the only salmonid species present, and in small streams, they often are the only species of fish (Connolly 1997, Heggenes et al. 1991b, Glova 1987).

Multiple life history forms frequently coexist in the same watershed and even in the same stream (June 1981, Johnston 1982, Heggenes et al. 1991a, Johnson et al. 1999). Where multiple forms coexist, it is possible for temporal and spatial differences in reproductive behavior to promote genetic differentiation (Zimmerman 1995). It is also possible for some subbasins within a drainage to contain entirely anadromous or entirely freshwater forms (Zimmerman 1995, Johnson et al. 1999).

There is evidence to suggest that life history patterns may be flexible. It is evident that not all individuals within a population behave similarly, even if they exhibit the same given life history pattern. Individuals in a cohort may respond to environmental factors differently at any point along the migratory pathway. This suggests that individuals may possess some degree of adaptive flexibility (Dill 1983, Johnson et al. 1999).

The observed complexity of life history forms of coastal cutthroat trout and the intermingling of various forms within populations, along with the plasticity of individuals within any given life history pattern, make identification of discrete life history types challenging for any single individual or population.

2.6.6 Abundance

The total abundance of coastal cutthroat trout in the lower Columbia basin is difficult to estimate because of their wide range of life history types, the lack of commercial harvest, and low hatchery production figures. Coastal cutthroat are reported in all lower Columbia River drainages, and anadromous individuals are either documented or believed to be present in all Washington tributaries feeding the Columbia downstream of Bonneville Dam.

Because only distribution and not abundance data exist for resident cutthroat trout in lower Columbia tributaries, the status of this life history form cannot be determined (WDFW 2000). At present, WDFW describes cutthroat as depressed in all rivers entering the Columbia from its mouth to the Kalama River, citing either long-term negative trends or short-term severe declines (WDFW 2000). The abundance of coastal cutthroat trout in the Lewis River and in tributaries entering the Columbia between the Lewis River and Bonneville Dam is unknown, because of lack of data.

Data used to document sport harvest of coastal cutthroat was collected during a 1971–95 salmon and steelhead study where data on cutthroat also was recorded. No distinctions between life history forms were made, but most cutthroat caught were presumably anadromous or fluvial. Sampling protocols exhibit some inconsistencies, especially in the earlier years, and some sampling was incomplete. Changes in angling regulations during the study may have affected cutthroat catch, but the extent of any reduction cannot be determined. Therefore, the quality of coastal cutthroat sport catch data in the lower Columbia is only fair. Total sport harvest from 1971–95 shows a sharp decline in sport catch after 1985, when more restrictive angling regulations were implemented (Figure 2-31).

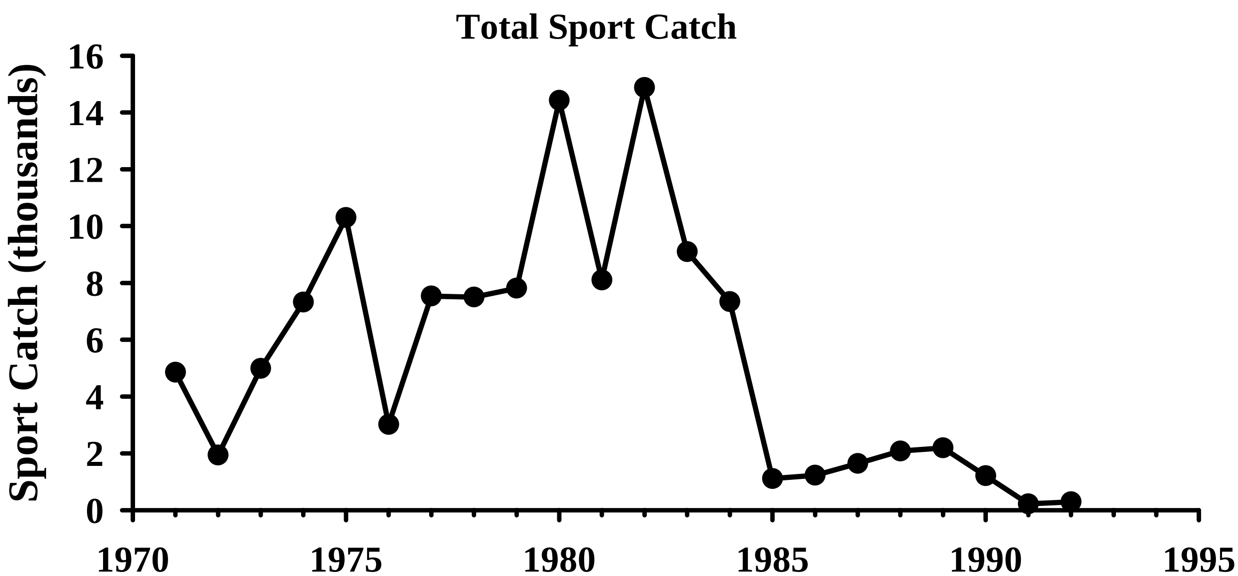

Figure 2-31. Total sport catch of coastal cutthroat trout in lower Columbia Washington tributaries. Data used to generate this figure are incidental to data collected during salmon and steelhead studies. No data for the Lewis River or tributaries from the mouth of the Lewis to Bonneville Dam is available.

Escapement measured at a weir trap in the Elochoman River evidences a decline after 1976, though the data are missing from 1981–87. Cowlitz River trap counts fluctuate around a mean of about 2,000 fish, and do not exhibit a downward trend over the 24 years from 1971–94, with a high of 6,103 fish in 1982 and a low of 383 fish in 1988 (Figure 2-32).

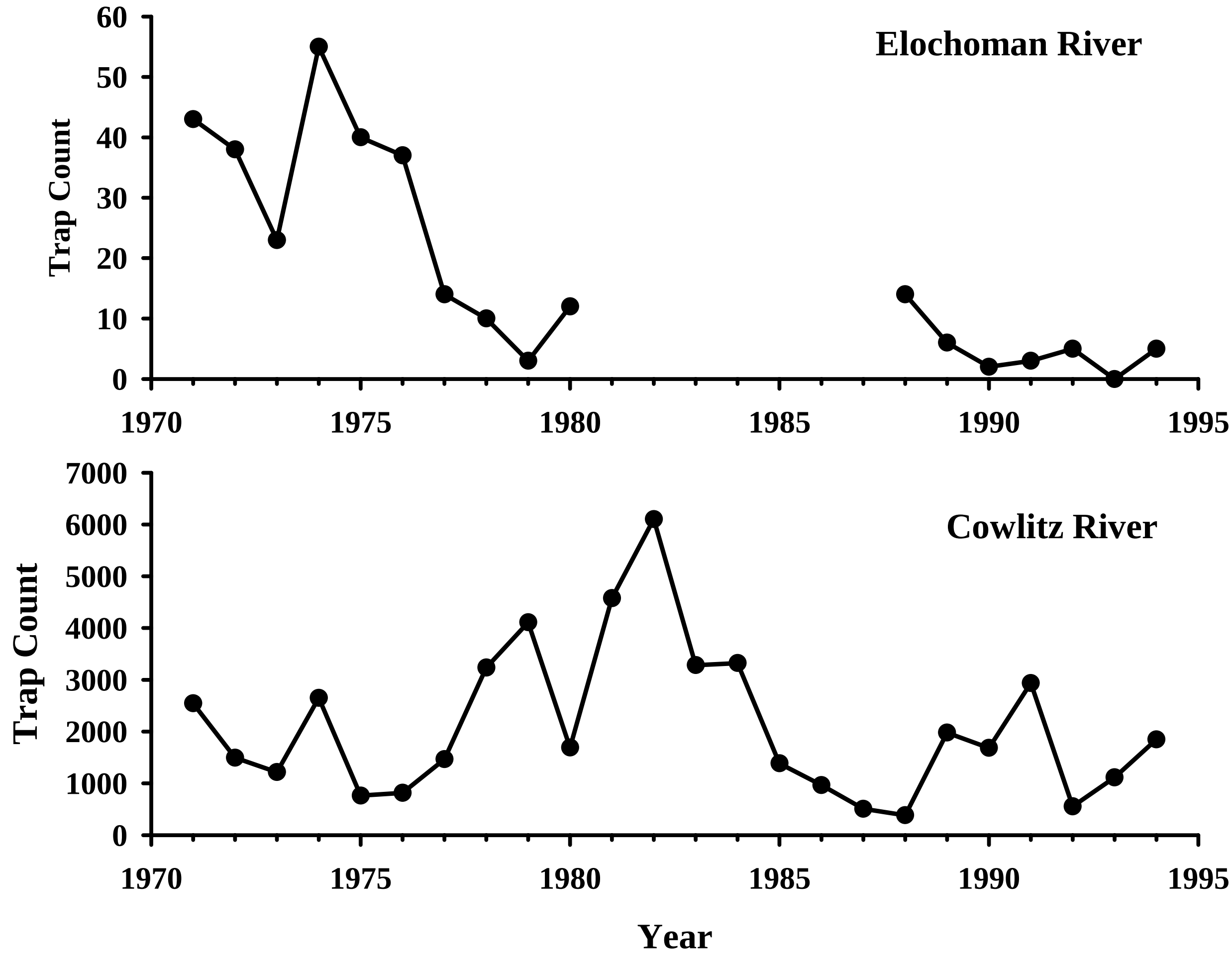

Figure 2-32. Coastal cutthroat counts at Elochoman River and Cowlitz River weir traps, 1971–94.

2.6.7 Productivity

A NMFS status report on sea-run cutthroat trout in the lower Columbia River indicates that returns of both naturally spawned and hatchery-produced fish have declined in almost all lower river tributaries over the past 10–15 years. The potential reduction in life-history diversity is a key concern. In many streams, freshwater forms are well distributed with relatively high abundance in comparison to anadromous forms in the same streams. NMFS concluded that habitat degradation and poor ocean and estuarine conditions are the likely causes of the severe depletion of anadromous forms of cutthroat trout.

2.6.8 Listing Status

To date, no investigation has been published regarding the persistence probability of southwest Washington cutthroat trout populations. The subspecies, sea-run cutthroat trout in the Southwest Washington/Northwest Oregon area, was a candidate for listing as threatened, but the USFWS found that a listing was not warranted.

In April 1999, NMFS and the USFWS issued a joint proposed rule for the listing of southwestern Washington/Columbia River sea-run cutthroat trout. The ESU includes populations of coastal cutthroat trout in the Columbia River and its tributaries downstream from the Klickitat River in Washington and Fifteenmile Creek in Oregon (inclusive) and the Willamette River and its tributaries downstream form Willamette Falls. Cutthroat trout found in the Lewis River are included in this ESU, although the status of Lewis River coastal cutthroat trout is currently unknown because of "*insufficient quantitative information to identify a trend in abundance or survival*" (WDFW 2000).

On April 26, 2000, the coastal cutthroat trout of the Umpqua River ESU (i.e. naturally spawning populations in mainstem and tributaries) was removed from the list of endangered and threatened species because NMFS determined that the Umpqua River population was part of a larger distinct population segment that was not previously, nor currently warrants listing as threatened or endangered under ESA (Federal Register, Vol. 65, No. 81).

On July 5, 2002, the USFWS issued a withdrawal of the Proposed Rule to List the Southwestern Washington/Columbia River Distinct Population Segment of the Coastal Cutthroat Trout as Threatened because of "*the latest information indicating relatively healthy-sized total populations in a large portion of the DPS, and our improved understanding of the ability of freshwater forms to produce anadromous progeny, lead us to conclude that this DPS does not meet the definition of a threatened species (in danger of becoming endangered in the foreseeable future) at this time*." (Federal Register, Vol. 67, No. 129)

Volume I, Chapter 3
Limiting Factors

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3.0 Limiting Factors

This chapter is focused on the specific limiting factors of fishing, hatcheries, and subbasin, mainstem Columbia River, estuarine, and ocean habitat and environmental conditions, that affect fishery resources of the Lower Columbia Planning Area. The intent of this chapter is not to prescribe recovery measures (those will be forthcoming in the subsequent management plan), but to explain the types of conditions which limit lower Columbia fish populations. This chapter should be regarded as an overview of the types of limiting factors that will be considered when recovery strategies are developed in the management plan and to provide a frame of reference for the types of factors considered in life cycle modeling and recovery analyses. Throughout this chapter under each major heading, broad topics are introduced in the initial sections while subsequent sections expand on these broad topics and discuss many of the intricate details.

3.1 Fishing

This section provides an overview of fisheries and fishery regulatory processes that would be considered when analyzing potential fishery impacts to focal fish species of the lower Columbia River. It is intended to illustrate the complexities in fishery management involving salmon and steelhead which travel through various freshwater and ocean jurisdictions during their life cycle and are subject to numerous catch allocation agreements, conservation requirements, and legal mandates. The section explains the different types of fishery impacts, the types of fisheries and areas in which fisheries occur, and the multitude of jurisdictions and processes these fish are subjected to. This section also provides perspective on historic and current harvest impacts for each species, including an estimate of change in hatchery and wild harvest rates from the 1930s to date, and an illustration of current harvest distribution (who is catching the fish) between ocean and freshwater fisheries. The section also displays several examples of management criteria, including ESA mandates, which drive the harvest of individual species in the various fisheries to which they contribute. Catch and effort numbers illustrate the magnitude of targeted or incidental catches as the majority of present-day effort is focused on harvestable hatchery fish and healthy wild fish.

In the early part of the 20th century, nearly all commercial fisheries in this region operated in freshwater, where they harvested only mature salmon. Ocean fisheries became more important in the late 1950s as more restrictions were imposed on freshwater and coastal estuary fisheries. Ocean harvest of salmon peaked in the 1970s and 1980s. In recent years, ocean commercial and recreational harvest of salmon has generally been reduced as a result of international treaties, fisheries conservation acts, regional conservation goals, the Endangered Species Act, and state and tribal management agreements.

Analysis of fisheries questions may consider a variety of direct and indirect effects. Direct effects include mortality in fisheries that are managed to specifically harvest target stocks. Indirect effects include incidental mortality of fish that are caught and released, encounter fishing gear but are not landed, or are harvested incidentally to the target species or stock. Indirect effects also might include genetic, growth, or reproductive changes when fishing rates are high and selective by size, age, or run timing. The emphasis of weak stock management has changed over the last 25 years, as ocean and freshwater fisheries have been widely reduced and refocused on hatchery-origin or healthy wild fish using a combination of time, area, and mark-selective regulations: Although direct harvest of weak stocks or populations, including many of those of

Washington's lower Columbia River, has never been a desirable management practice, incidental fishery impacts have now become much more important in managing weak stocks than directed harvest. On the other hand, limits intended to protect weak stocks in mixed stock fisheries reduce access to healthy wild or hatchery runs. Relatively small numbers or proportions of a protected stock may be impacted in a mixed stock fishery, but the regulatory consequences of those small impact allowances can result in significant reduction in harvest opportunity in mixed stock fisheries.

Fishery impact analyses may be conducted at population or fishery-specific levels. Population-specific analyses would treat impacts by all fisheries in aggregate. Fishery-specific analyses would consider fine-scale impacts. By nature of their wide ranging travels, anadromous salmonids can be exposed to a wide variety of fisheries from their lower Columbia watershed of origin all the way to Canada and Alaska (Figure 3-1). This broad distribution can substantially complicate analysis and attempts to limit impacts on specific stocks.

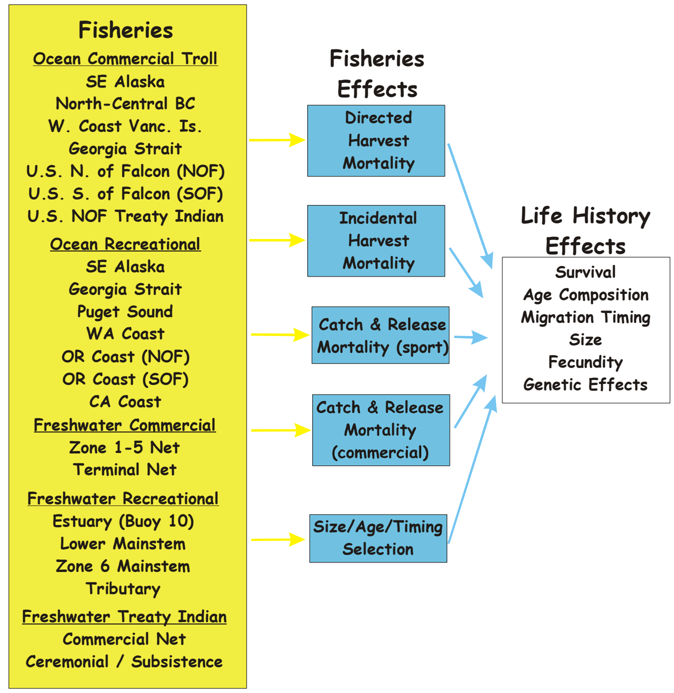

Figure 3-1. Fisheries, fisheries effects, and life history effects.

Analysis of fishing and harvest is also complicated by the need to consider fisheries impacts at both the species impact and population goal levels. Fishing mortality can be considered an impact that interacts with other factors to affect salmon productivity and viability and thus needs to be addressed as part of recovery planning and actions. However, directed harvest or increased accessibility to other populations in mixed stock fisheries are also key elements of broad recovery goals, because recovery objectives include sustaining healthy, harvestable populations.

3.1.1 Sources of Fishing Effects

3.1.1.1 Directed Harvest Mortality

Harvest mortality occurs in fisheries directed at a particular species or stock; this harvest can occur in single (terminal) or mixed (intercept) stock fisheries. The most effective method for targeting a specific stock is the prosecution of single stock fisheries. Single stock fisheries most commonly occur in terminal harvest areas where one stock is known to be present through the use of stock identification techniques, historical run timing data, or escapement survey methods.

In mixed stock fisheries, the management challenge is to harvest from mixed populations having various available surpluses, sometimes including populations with no surplus, as the populations move through the fishery area at various rates and abundances. Harvest of a specific stock in the mix can be achieved by management decisions (e.g. fishery openings when the targeted stock is abundant relative to other stocks), fishery adaptations (e.g. gear designed to target specific stock/species), or fishery regulations (e.g. prohibitions of retaining certain species). Stock identification techniques are constantly being improved to assist managers in making informed and timely fishery decisions. For example, scale pattern analysis, CWT analysis, and genetic stock identification techniques have been applied in-season to determine the stocks present in a fishery, providing managers with timely stock composition data. Time and area fishery openings are also effective in targeting specific stocks and reducing impact to other stocks when information is available about the migration timing and migration route of a specific stock. In many cases where the targeted stock is a distinct size relative to other stocks in the fishery, gear modifications, such as specific mesh size requirements, can be effective in harvesting certain size fish while allowing other fish to escape the fishery. In the Columbia River, certain fisheries are focused on harvesting adipose fin-clipped hatchery-reared fish only by targeting marked hatchery fish while utilizing gear modifications to allow protected stocks to escape. Regulations prohibiting harvest of wild fish (i.e. nonadipose fin-clipped fish) have been relatively successful. However, the occurrence of delayed mortality as a result of releasing wild fish captured in commercial fisheries is not completely understood.

3.1.1.2 Incidental Harvest Mortality

Salmonid migration timing and routes are dynamic and considerable variation can occur from year to year. Thus, despite the various methods discussed above to target a specific stock and minimize effects on weak stocks, incidental harvest of non-targeted stocks still occurs. Most fisheries have specific reporting requirements and limits for incidental bycatch that are intended to lessen the harvest impacts to non-targeted stocks. In the case of the Columbia River, specific incidental harvest percentages are set for protected stocks; fisheries are managed so as not to exceed these harvest limits of protected stocks.

Access to strong stocks in Columbia River and ocean fisheries is regulated by impact limits on weak populations mixed with the strong. Each fishery is controlled by a series of regulating factors. Many regulating factors that affect harvest impacts on Columbia River stocks are associated with laws, policies, or guidelines established to manage other individual or combined stocks, but indirectly control impacts of Columbia River fish as well. Harvest managers configure fisheries to optimize harvest of strong stocks within the series of constraints for weak stock protection. ESA-listed fish generally comprise a small percentage of the total fish caught by any fishery. Every harvested ESA-listed fish may correspond to tens, hundreds, or even thousands of other fish in the total catch. As a result of weak stock fishery constraints, strong hatchery and wild runs may go unharvested. Small reductions in fishing rates on ESA-listed populations can translate to larger reductions in catch of other stocks, with substantial economic consequences.

3.1.1.3 Catch and Release Mortality

Catch and release regulations have been used for years to manage sport fisheries. Generally, catch and release restrictions allowed resident fish to grow older and larger, thereby creating improved angling opportunities. More recently, catch and release has been employed in anadromous fish management practices to enable retention of hatchery salmon and steelhead and release of wild fish in mixed-stock fisheries. Because of the wide range of knowledge among sport anglers regarding proper fish handling techniques and the different degrees of how fish species react to handling stress, mortality occurs as a result of catch and release. Although sport fishing catch and release mortality varies widely among fisheries, it is believed to be low compared to other harvest related mortality.

Catch and release has been employed in the Columbia River commercial fishery since 1950 for non-legal size sturgeon and since 1975 for steelhead. Catch and release is a relatively new concept for commercial salmon fishing, and has recently become a significant part of managing Columbia River spring chinook stocks. Recent recovery efforts in the Columbia Basin have focused on maintaining and rebuilding native wild stocks. The hatchery practice of marking released fish with an adipose fin clip has allowed fishery managers to implement fisheries which harvest only hatchery fish while requiring the release of protected wild stocks. Significant gear modifications are continually being evaluated and utilized to reduce any handling mortality that can occur as a result of being caught and released by the commercial fishery. Delayed catch and release mortality of wild fish in these hatchery-selective fisheries is not completely understood and is presently being evaluated.

3.1.1.4 Gear or Fishery Selectivity

Commercial fishing gear can be size-selective, depending on the type of gear (i.e. gill net vs. seine) or the size of gear (i.e. mesh size). As mentioned in the mixed stock fishery discussion, size selectivity can be a desired result if the gear is designed to harvest a specific size stock or species. However, commercial fishing gear size selectivity can also be undesirable. For example, if a fishery disproportionately harvests the larger individuals in a population, the remaining smaller individuals comprise the effective population (i.e. those individuals that spawn in any given year). If this process is repeated annually, the effect on the adult population is a decreased average size at maturity or potentially a modified age composition.

Fisheries may also be selective for a particular timing or segment of the run, depending on management practices. For example, a fishery may disproportionately harvest the early portion of a run because of market- or industry-driven needs. Because run timing is heritable (Garrison and Rosentreter 1981), fisheries may alter run timing traits due to systematic temporal removals from populations over time. Although there is evidence that run timing alterations have occurred in certain stocks, it is not a forgone outcome for all stocks exposed to fisheries. In the Columbia River, hatchery coho-targeted fisheries, in conjunction with hatchery practices, have altered run timing (Cramer and Cramer 1994). Hatchery coho brood stock was often obtained from the early part of the run, which generally resulted in early run timing for hatchery adults. Effort in fisheries targeting hatchery fish is concentrated during the time of hatchery fish abundance. Consequently, consistent harvest of wild fish with the early run trait can also occur, thereby reducing this early run trait in the spawning population and altering run timing of the wild stock. Effects of selective fisheries are most likely to occur if harvest rates are high; lower harvest rates will likely mitigate for selectivity.

3.1.2 Effects of Fisheries on Population Biology and Structure

Fishing has direct and indirect effects on salmon populations, especially if harvest rates are high and/or prolonged. Harvest can influence the number, biomass, age, size, and fecundity of spawners, as well as the genetic characteristics and population structure. In many lower

Columbia salmon populations, as well as others, the biological characteristics of contemporary populations have been shaped by continued harvest patterns.

3.1.2.1 Abundance

Following other mortality causes in each returning cohort, harvest clearly determines the number of adult salmon remaining to perpetuate the population. Much of the future discussion about recovery and sustainability will be focused on a new paradigm for determining the number of salmon required to fill the habitat to capacity (Schoonmaker et al. 2003).

In addition to the important function of salmon spawning escapement for supplying eggs for subsequent generations, recent scientific evidence has shown that adult salmon carcasses provide a significant source of nutrients delivered from marine to freshwater ecosystems (Kline et al. 1993, Bilby et al. 1996, Cederholm et al. 1999). Not only do the carcasses form the basis of a nutrient pathway via primary production, but flesh and eggs are directly consumed by aquatic insects (Wipfli et al. 1999) and by rearing fish (Bilby et al. 1996). This biological feedback loop benefits future salmon production. The chronic depression of salmon biomass to freshwater ecosystems may be contributing to reduced carrying capacity for salmon (Cederholm et al. 1999, Knudsen 2002). Probably the most important implication for Pacific salmon is that the production relationship (returning adults per spawner) is influenced not only by the number of eggs deposited in the gravel, but also by the amount of biomass delivered and retained in the watershed (Cederholm et al. 1999). The carrying capacity for freshwater production depends on both the physical space available and the amount of nutrients provided to the system. This varies, depending on the freshwater life history of the species and the nutrient interdependence among species but, in any case, there is a feedback mechanism relating the number of adults allowed to escape harvest directly to the productivity of the system. This biological control factor must be considered in contemporary productivity analyses.

3.1.2.2 Age, Size, Sex, Fecundity

Selective fishing (as described above) affects salmon population age, size, sex, and fecundity structure directly by influencing certain characteristics in the targeted populations or indirectly by gradually influencing the population’s heritable characteristics (discussed below). Gear or run timing selectivity may influence the annual productivity of the population by removing the older, larger individuals, too many of one sex, or the larger females carrying the most eggs. Fishing-influenced changes in the average sizes and ages of salmon populations have been well documented (Ricker 1981). For example, body size is related to redd digging success (Beacham and Murray 1987) and or fecundity -- larger fish usually carry more eggs (Sandercock 1991). When too many individuals with the most reproductive potential are removed, the population’s productivity is reduced.

3.1.2.3 Genetics

As fisheries are continually prosecuted, the genetics of the target populations can be gradually changed, especially if there is selection for certain sizes of fish or portions of the run timing (Reisenbichler 1997). Because of their tendency to home to their natal streams, Pacific salmon have evolved a diversity of genetic and phenotypic population characteristics (Waples 1991a). Every spawning population is potentially a unique genotype (Healey and Prince 1995). There is even evidence of genetically controlled divergence within a single, relatively small spawning area (Woody et. al. 2000). Examples of apparently heritable ecological strategies for success include variations in body size correlated with differences in stream flows (Beacham and Murray 1987), run timing for spawning and incubation survival (Smoker et al. 1998), duration of egg incubation (Woody 1998), and a variety of freshwater rearing strategies (e.g., Wood et al. 1987, Bisson et al. 1997). Lastly, as numbers are reduced by harvest, especially in small populations, all the attributes controlled by genetic diversity are threatened by inbreeding and/or genetic drift (Reisenbichler 1997).

3.1.2.4 Population Structure and Diversity

Reduced abundance also affects the structure and biodiversity of populations. Salmon populations are generally structured hierarchically with genetic relatedness usually corresponding to geographical distance (Allendorf and Waples 1995). Independent populations are defined as a group of the same species that spawns in a particular location and season and which, for the most part, do not interbreed with other spawning groups (Myers et al. 2003). Each independent population evolves characteristics of productivity, body size, run timing, fecundity, etc. that correspond with the habitat features it experiences throughout its life history. The combination of these features across populations constitutes the biodiversity of a group of populations, commonly referred to as a stock when mixed together for harvest management purposes. As harvest usually occurs at the stock level, a similar harvest rate is applied to the mixture of populations, some having higher production potential than others. Heavy harvest rates, especially when combined with habitat problems and natural variation, can therefore drive the weaker populations to low levels, even to extinction (e.g., Walters and Cahoon 1985). As weaker populations are diminished or eliminated, the total biodiversity and genetic variation within and between the hierarchical populations is reduced (Riddell 1993). Setting harvest rates to maximize use of high productivity hatchery populations is particularly troublesome for intermingled wild populations that cannot withstand the hatchery harvest rate (NRC 1996, Knudsen 2002). The use of selective fisheries for marked hatchery fish is expected to ameliorate this effect on lower Columbia spring need to decide chinook, coho, and steelhead.

3.1.3 Fisheries Management Structure

Because of their exposure to fisheries across large geographic regions of the West Coast, Pacific salmon and steelhead management is governed by numerous regional organizations. Fisheries of the Columbia River are established within the guidelines and constraints of the Pacific Salmon Treaty (PST), the Columbia River Fish Management Plan (CRFMP), the Endangered Species Act (ESA) administered by NOAA Fisheries, The Pacific Fishery Management Council (PFMC), the states of Oregon and Washington, the Columbia River Compact, and management agreements negotiated between the parties to *US v. Oregon*.

3.1.3.1 Pacific Salmon Commission

Management of Pacific salmon has long been a matter of common concern to the United States and Canada. After many years of negotiation, the PST was signed in 1985 to set long-term goals for the benefit of the salmon and the two countries. The principal goals of the treaty are to enable both countries, through better conservation and enhancement, to increase production of salmon and to ensure that the benefits resulting from each country's efforts accrue to that country.

The Pacific Salmon Commission (PSC) is the body formed by the governments of Canada and the United States to implement the treaty. The Commission itself does not regulate the salmon fisheries but provides regulatory advice and recommendations to the two countries. It has responsibility for all salmon originating in the waters of one country which are 1) subject to interception by the other, 2) affect management of the other country's salmon, or 3) biologically affect the stocks of the other country. In addition, the PSC is charged with taking into account the conservation of steelhead trout while fulfilling its other functions.

The Commission has a dual role; to conserve Pacific salmon in order to achieve optimum production, and to divide the harvests so that each country reaps the benefits of its investment in salmon management. The Commission has a variety of tools at hand to achieve its mandate. It may recommend that the countries implement harvest limitations, time and area closures, gear restrictions, or other measures to control harvests. In addition, the Commission may recommend use of enhancement techniques to strengthen weak runs, mitigate for damage done by logging, mining or dam building, or for other purposes. The PSC gives both countries a forum through which to resolve the difficult problems surrounding salmon harvest management.

PSC members represent the interests of commercial and recreational fisheries as well as federal, state, and tribal governments. Each country has one vote; the agreement of both is required for any recommendation or decision. Four regional panels (Southern, Northern, Transboundary, and Fraser River) provide technical and regulatory advice; panel membership reflects a range of governmental and fishing interests.

3.1.3.2 Pacific Fishery Management Council

The Magnuson-Stevens Fishery Conservation and Management Act of 1976 is the principal law governing marine fisheries in the United States. The Act was adopted for the purposes of managing fisheries 3-200 miles offshore of the US coastline, phasing out foreign fishing activity within this zone, recovering overfished stocks, and conserving and managing fishery resources. In 1996, Congress passed the Sustainable Fisheries Act, which revised the Magnuson Act and reauthorized it through 1999; later reauthorization bills have been presented but have not been enacted. The Pacific Fishery Management Council (PFMC) is one of eight regional fishery management councils established by the Magnuson Act. The PFMC is responsible for fisheries off the coasts of California, Oregon, and Washington. Thus, the Council is responsible for all ocean fisheries, including salmon, groundfish, pelagic fish, etc., and does not focus solely on salmonids.

Chinook and coho salmon are the main salmon species managed by the PFMC in waters extending from the Canadian border to Mexico, and 3-200 nautical miles offshore (Table 3-1). In odd-numbered years, the Council may also manage special fisheries near the Canadian border for pink salmon. Sockeye, chum, and steelhead are rarely caught in the Council's ocean fisheries. The Council's Salmon Fishery Management Plan (SFMP) describes the goals and methods for salmon management. Central parts of the plan are annual spawner escapement goals for the major salmon stocks and an allocation of the harvest among different fisheries or locations (i.e. allocations are set for ocean or inland commercial, recreational, or tribal fisheries as well as for specific ports). The Council uses management tools such as season length, quotas, bag limits, and gear restrictions to achieve fishery management goals.

Annually, a preseason process of meetings and public hearings is used to develop recommendations for management of the ocean fisheries. Past harvest data and preseason salmon abundance forecasts are the primary basis for management decisions concerning season structure and harvest quotas. Final recommendations are adopted annually in April and implemented by NOAA Fisheries beginning in May. The Salmon Technical Team (STT) provides technical information and data analysis to the Council; the team is comprised of eight representatives from state, federal, and tribal fisheries management agencies. The Salmon Advisory Subpanel (SAS) has 17 members who represent commercial, recreational, and tribal interests, as well as a public representative and a conservation representative.

Impacts to each species vary widely, depending on many complicated factors which include annual salmon abundance and ESA restrictions. The PFMC evaluates ESA consultation standards each year and provides guidance for the upcoming ocean fishing season. The standards for 2003 are presented for those ESUs with potential connections to lower Columbia River salmonids (Table 3-1). Further ESA restrictions apply to specific inside Columbia River fisheries and are discussed in the species-specific sections to follow.

3.1.3.3 North of Falcon

Folded into the PFMC management process is a parallel public process referred to as North of Falcon (NOF). The NOF process integrates management of ocean fisheries between Cape Falcon (on the north Oregon coast) and the Canadian border with inland area fisheries. Columbia River fisheries are a significant part of the NOF process. Coordination and shaping of the ocean and freshwater fisheries occurs to assure that fish conservation objectives are met and there is reasonable sharing of the conservation burden between the fisheries and various user groups. In this process there are allocation agreements reached between Oregon and Washington ocean and freshwater commercial and sport fisheries as well as mandated allocation agreements between the states and treaty Indian tribes. Conditions for incidental take permits concerning ESA-listed Columbia River populations are often developed during the NOF process.

3.1.3.4 State Fishery Regulations

Regulations for lower Columbia tributary sport fisheries are developed through state public process and adopted into law by the respective Fish and Wildlife Commissions of Washington and Oregon for their jurisdictional waters. Mainstem Columbia joint waters are coordinated for consistency in the Compact forum (see below) but are adopted into law by the respective states. The state regulatory process includes adoption of permanent rules as well as emergency regulations to enable quicker adjustments of fisheries when needed to meet conservation objectives or provide additional harvest opportunity. The state regulations are made consistent with management strategies reached in the NOF process.

3.1.3.5 US v. Oregon

In 1968, the US District Court ruled that Columbia River treaty Indians were entitled to an equitable share of the upper Columbia River fish returns, in a court case known as *US v. Oregon*. After 20 years of legal tests and negotiations, the CRFMP was adopted by District Court order in 1988 and agreed to by the parties: the United States; the states of Oregon, Washington, and Idaho; and the four treaty Indian tribes. The purpose of the CRFMP as defined by the court was to:

. . . provide a framework within which the Parties may exercise their sovereign powers in a coordinated and systematic manner in order to protect, rebuild, and enhance upper Columbia

River fish runs while providing harvests for both treaty Indian and non-Indian fisheries. In order to achieve the goals of the CRFMP, the Parties intend to use habitat protection authorities, enhancement efforts, artificial production techniques, and harvest management to ensure that Columbia River fish runs continue to provide a broad range of benefits in perpetuity.

In 1996, the parties to *US v. Oregon* negotiated three-year (1996–98) management agreements: one each for upper Columbia fall chinook and uppe Columbia spring chinook, summer chinook, and sockeye. The agreements were a result of a 1995 court settlement where the parties agreed to discuss the possibility of amending the CRFMP. The 1996–1998 management agreements formed the basis for recent agreements, and included escapement goals, production measures and harvest allocations. Annual agreements have occurred for fall chinook, coho, and summer steelhead during 1999-2003. A 5-year agreement for harvest was reached for spring chinook, summer chinook, and sockeye for the period 2001–2005. The CRFMP is currently being negotiated for a longer-term agreement for all species to be in place by 2004.

Table 3-1. List of species managed by the PFMC with potential impacts on lower Columbia River salmonids.

ESU	Stock Representation in Salmon FMP	ESA Consultation Standard	Council Guidance for 2003
Lower Columbia River chinook—threatened	Sandy, Cowlitz, Kalama, Lewis spring	No specific requirements	Meet hatchery escapement goals
	Sandy, Cowlitz, Kalama fall	Brood year adult equivalent exploitation rate on Coweeman tule fall chinook ≤ 49%	Same as consultation standard
	North Fork Lewis fall	5,700 MSY level adult spawning escapement	Same as consultation standard
Upper Willamette chinook— threatened	Upper Willamette River spring	No specific requirements; rare occurrence in Council fisheries	Same as consultation standard
Upper Columbia River spring chinook—endangered	Upper Columbia River spring	No specific requirements; rare occurrence in Council fisheries	No additional constraints; Ocean fishery impacts minor
Snake River fall chinook— threatened	Snake River fall	30% reduction from the 1988–93 average adult (age 3 & 4) exploitation rate for all ocean fisheries	Same as consultation standard
Snake River spring/summer chinook— threatened	Snake River spring/ summer	No specific requirements; rare occurrence in Council fisheries	Same as consultation standard
Oregon Coast coho—threatened	S. central OR coast N. central OR coast N. OR coast	15% (in 2003) combined marine/ freshwater exploitation rate	Same as consultation standard
Lower Columbia River/Southwest Washington coho—candidate	Sandy and Clackamas River	No specific requirements	≤ 20% ocean exploitation rate

3.1.3.6 Columbia River Compact

In 1918, the US Congress ratified a compact between Oregon and Washington covering concurrent jurisdiction of Columbia River fisheries. The Columbia River Compact comprises the Washington Fish and Wildlife Commission (WFWC) and the Oregon Fish and Wildlife Commission (OFWC). In recent years, the commissions have delegated decision-making authority to the state fish and wildlife agency's director or designee. Periodic hearings to adopt or review seasonal commercial regulations are held just before major fishing seasons to consider current information and establish season dates and gear restrictions. Additional hearings are held in-season when updated information concerning run size, attainment of escapement goals, or catch guidelines indicates a need to adjust the season.

The Compact jurisdiction includes the Columbia River from the mouth to just upstream of McNary Dam. The Compact sets fishing seasons in the non-Indian commercial Zones 1-5 (Mouth to Bonneville Dam) and in the treaty Indian commercial area Zone 6 (Bonneville Dam to McNary Dam) (Figure 3-2).

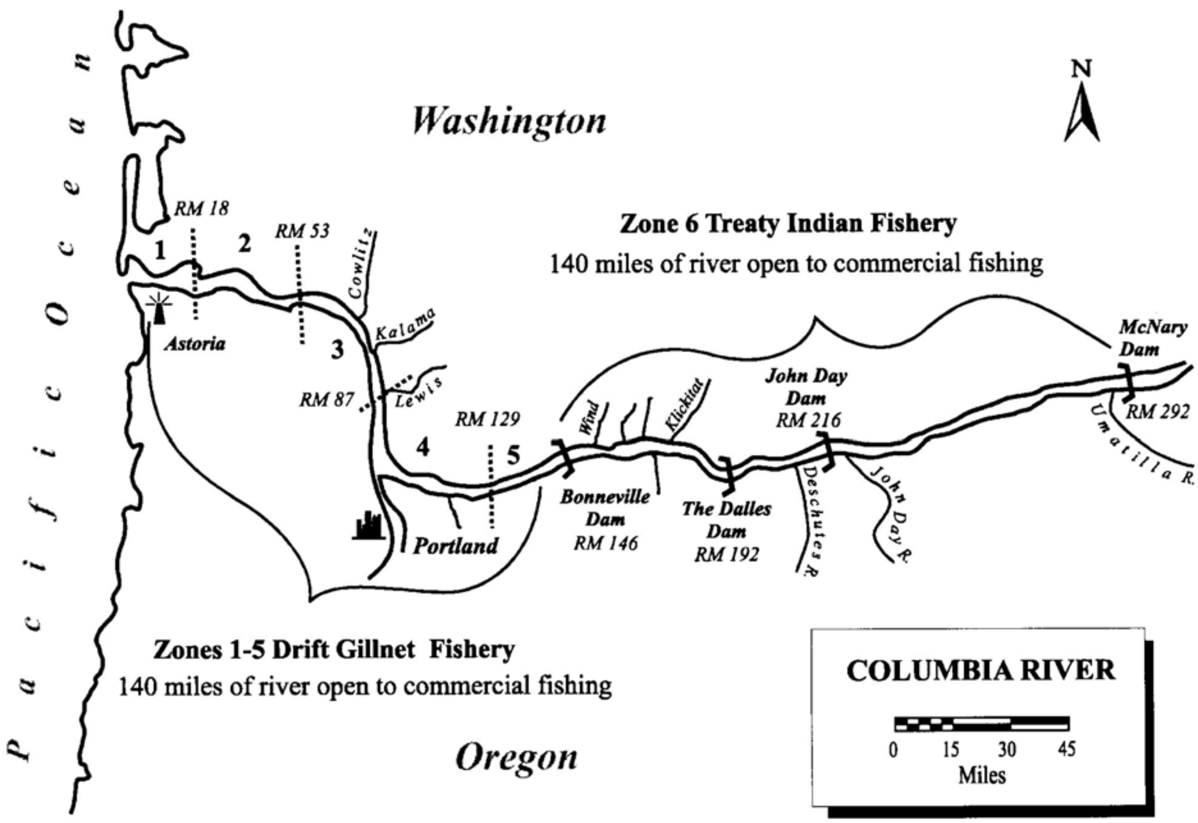

Figure 3-2. Columbia River commercial fishing zones.

3.1.3.7 Endangered Species Act (ESA)

Throughout the 1990s, 12 Columbia River basin Evolutionarily Significant Units (ESUs) were listed as threatened (T) or endangered (E):

- Snake River fall chinook (T—April 1992)
- Snake River spring/summer chinook (T—April 1992)
- lower Columbia River chinook (T—March 1999)
- upper Willamette River chinook (T—March 1999)
- upper Columbia River spring chinook (T—March 1999)
- Columbia River chum (T—March 1999)
- Snake River sockeye (E—November 1991)
- upper Columbia River steelhead (E—August 1997)
- Snake River steelhead (T—August 1997)
- lower Columbia River steelhead (T—March 1998)
- upper Willamette River steelhead (T—March 1999)
- middle Columbia River steelhead (T—March 1999)

An additional ESU (lower Columbia/SW Washington coho) was designated as a candidate species in July 1995. NOAA Fisheries also reviewed the status of this ESU and its boundary designations in 2001-2003, but has not published findings on the review. In addition, numerous other listed or candidate ESUs along the California, Oregon, and Washington coasts affect ocean fisheries targeted on harvesting Columbia River salmonids. Because of the ESA status of many Columbia River salmonids, harvest managers must consult annually with NOAA Fisheries to assure fishers are regulated to meet no-jeopardy standards established for ESA-listed species. NOAA Fisheries issues incidental take permits to regulatory agencies and Tribes for fisheries that have satisfied ESA regulatory requirements.

3.1.4 Fisheries Types and Areas

By nature of their wide-ranging migrations, anadromous salmonids can be exposed to a variety of fisheries from their basin of origin all the way to Canada and Alaska. Lower Columbia River salmonids are harvested in commercial, sport, and tribal fisheries throughout the West Coast of the United States and Canada. The following sections are a brief description of different regional fisheries.

3.1.4.1 Canada/Alaska Ocean

Numerous fisheries in Canada and Southeast Alaska harvest far-north migrating chinook stocks from the lower Columbia River basin. Some Columbia River coho salmon are also harvested in many Canadian fisheries. Canadian marine fisheries include commercial troll and net fisheries as well as recreational sport fisheries in northern BC, Central BC, West Coast of Vancouver Island, Strait of Georgia, and Strait of Juan de Fuca. In Southeast Alaska, treaty (i.e. US/Canada agreement described below) chinook marine fisheries include commercial troll and net fisheries as well as recreational sport fisheries. In recent years, chinook harvest in terminal fisheries and harvest of Alaska hatchery production has increased, although these harvests are not subject to PST limitations.

In June 1999, under the PST, Canada and the US agreed on a framework for chinook fishing regimes for 1999–2008 wherein Southeast Alaska (all gear), northern BC (troll and recreational), and West Coast Vancouver Island (troll and outside recreational) fisheries are to be regulated under aggregate abundance-based management (AABM) regimes. These fishery regimes establish catch ceilings derived from estimates of total aggregate abundance of all stocks contributing to specific components of the fisheries and target fisheries harvest rates. Eventually, the US and Canada plan to incorporate management regimes for AABM fisheries based on total mortality rather than catch. For fisheries not driven by AABM regimes, the 1999 agreement established conservation obligations to reduce harvest rates on depressed chinook stocks by 36.5% for Canadian fisheries and 40% for US fisheries, relative to levels observed during 1979-1982.

The June 1999 agreement included commitments to develop abundance-based regimes for fisheries along the Washington-British Columbia border. The purpose is to conserve natural coho production units from Washington, Oregon, and southern BC by establishing exploitation rate constraints based on projected resource status. These regimes are still under development.

3.1.4.2 United States West Coast Ocean

Ocean fisheries along the U.S. West Coast are separated into four major management areas (Figure 3-3):

1. US/Canada border to Cape Falcon, Oregon
2. Cape Falcon, Oregon to Humbug Mountain, Oregon
3. Humbug Mountain, Oregon, to Horse Mountain, California
4. Horse Mountain, California to the US/Mexico border.

These management areas are further divided into subareas depending on the type of fishery. Numerous treaty Indian commercial troll, non-Indian commercial troll, and recreational marine fisheries exist along the West Coast (Figure 3-4 and Figure 3-5).

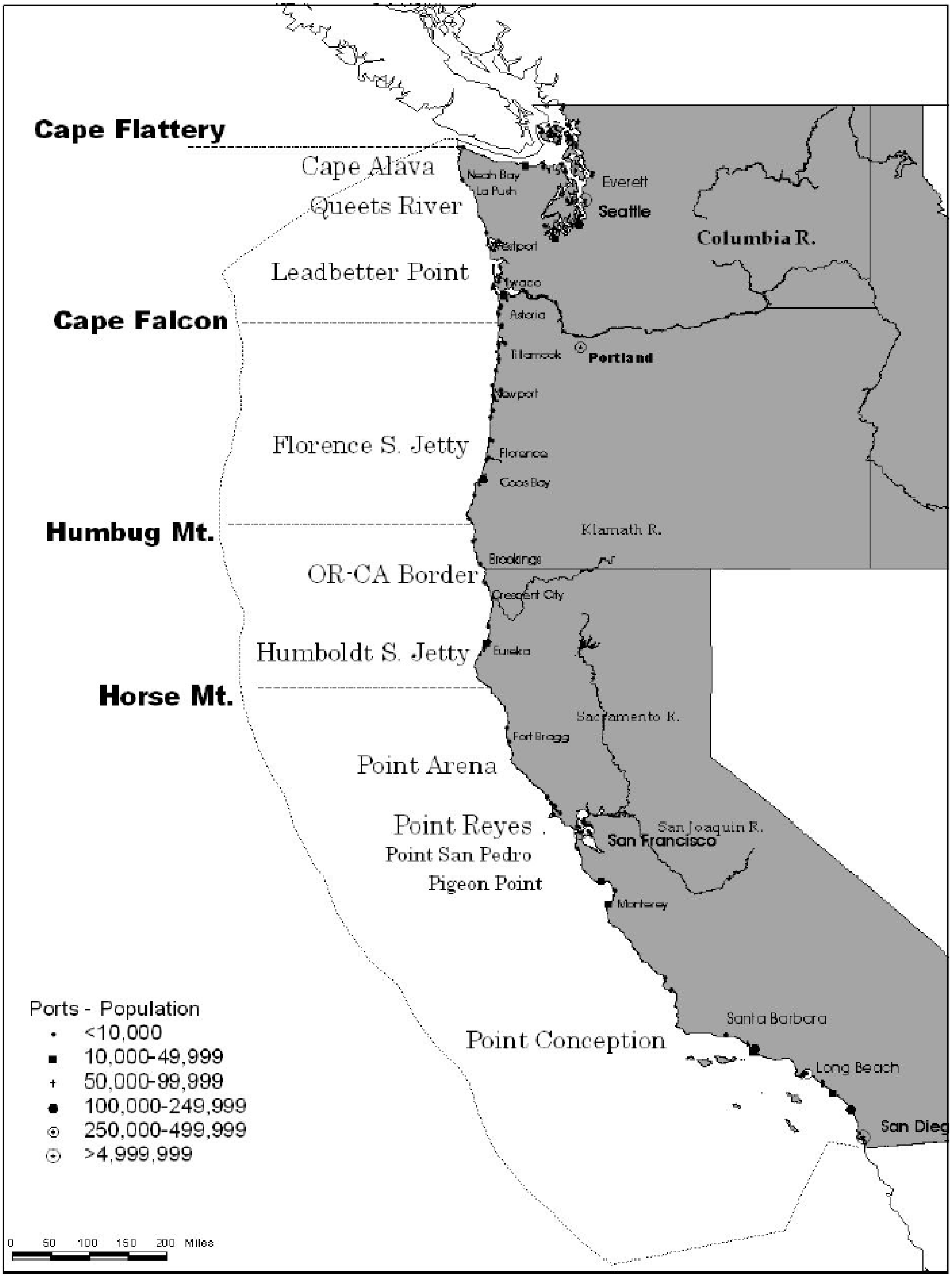

Figure 3-3. Major management areas in US West Coast ocean fisheries.

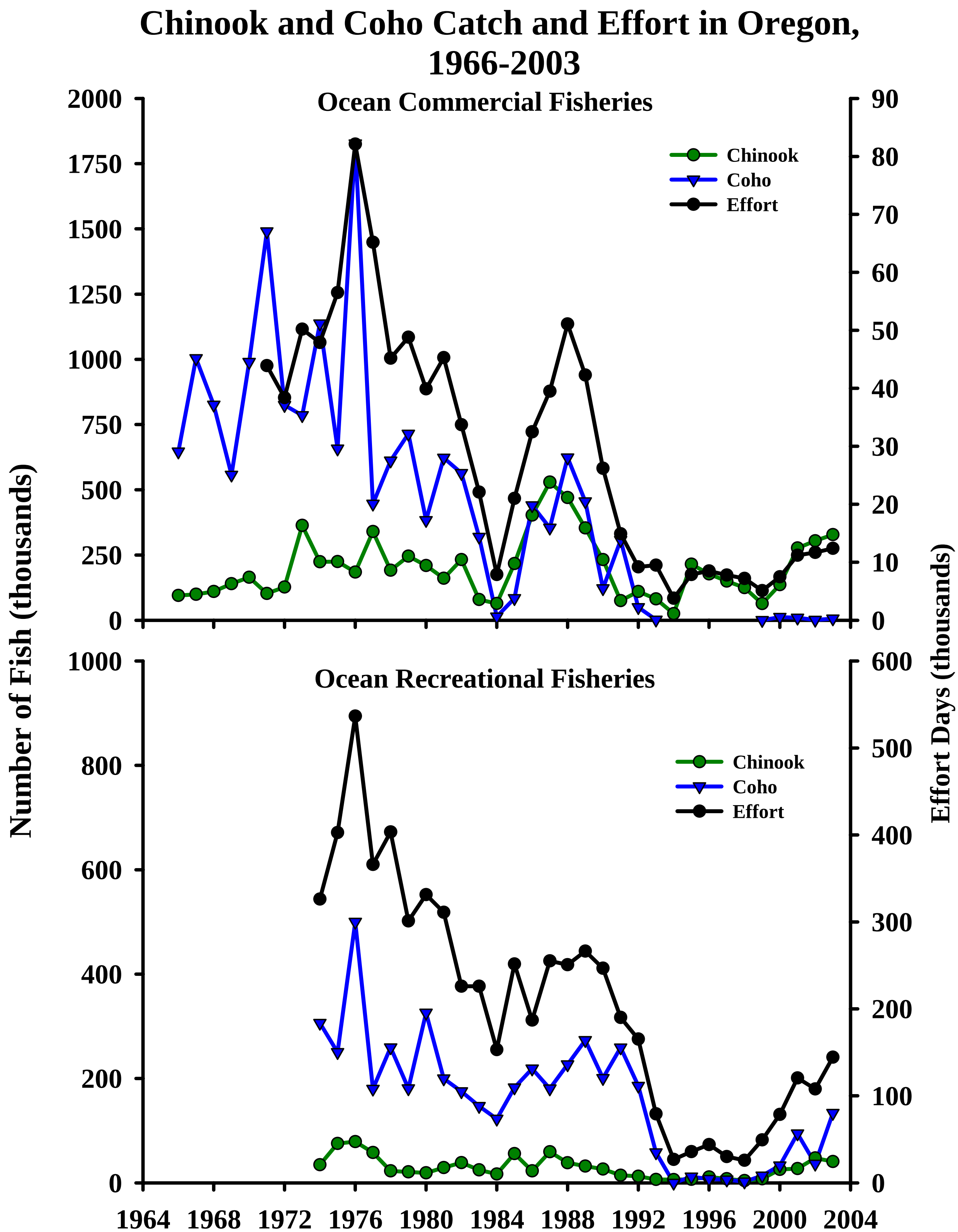

Figure 3-4. Commercial and recreational ocean catch and effort for chinook and coho in Oregon, 1966–2003.

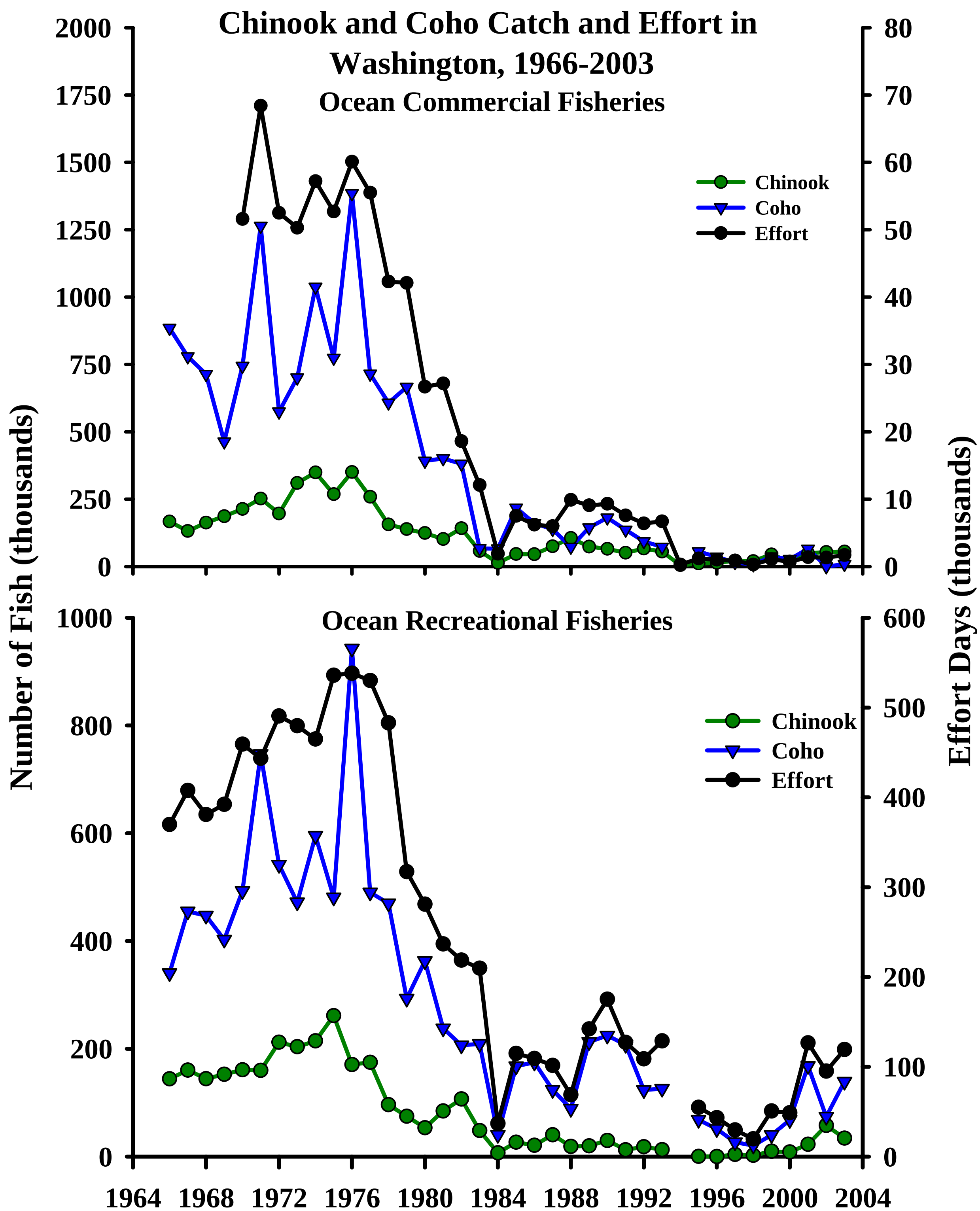

Figure 3-5. Commercial and recreational ocean catch and effort of chinook and coho in Washington, 1966–2003.

3.1.4.3 Lower Columbia River Commercial

Europeans began using Pacific salmon about 1830 and, by 1861, commercial fisheries became important. In 1866, salmon canning began in the Northwest and the non-Indian commercial fishery grew rapidly. Salmon and steelhead landings exceeded 40 million pounds annually several times between 1883 and 1925 (Figure 3-6). Since 1938, landings have ranged from a high of 31.6 million pounds (2,122,500 fish) to a low of 0.9 million pounds in 1995 and 1999 (around 68,000 fish).

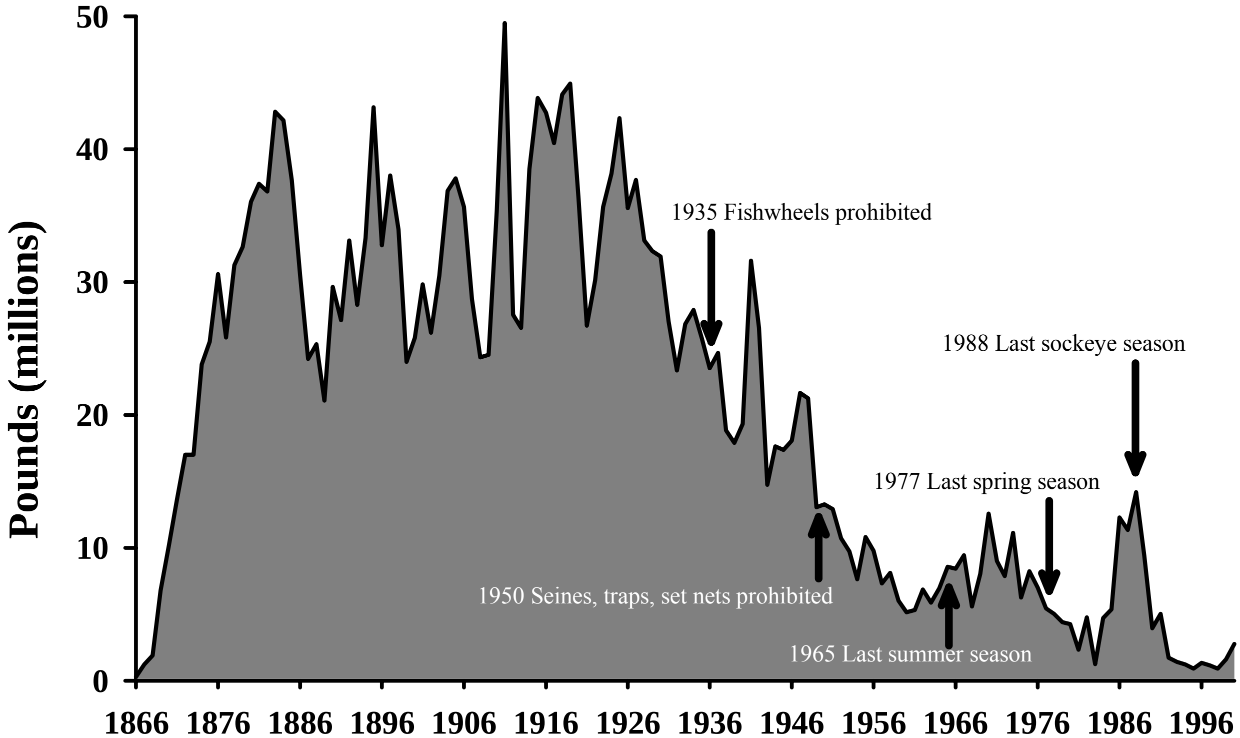

Figure 3-6. Commercial landings of salmon and steelhead from the Columbia River in pounds, 1866–1999 (ODFW and WDFW 2000).

Since the early 1940s, Columbia River commercial landings of salmon and steelhead have steadily declined, reflecting changes in fisheries in response to declines in salmonid abundance. Recent annual commercial harvests have fluctuated for each species, primarily depending on variable abundance of hatchery production (Figure 3-7). In the late 1950s, non-Indian commercial harvest comprised almost 100% of the Columbia River commercial fisheries landings; the percentage steadily declined to about 25% in 1995. The non-Indian percentage of commercial landings has increased to about 50% in recent years (Figure 3-8). Treaty Indian commercial landings became a larger portion of the total Columbia River commercial landings following a 1968 federal court ruling regarding equitable Indian and non-Indian harvest sharing (Figure 3-8).

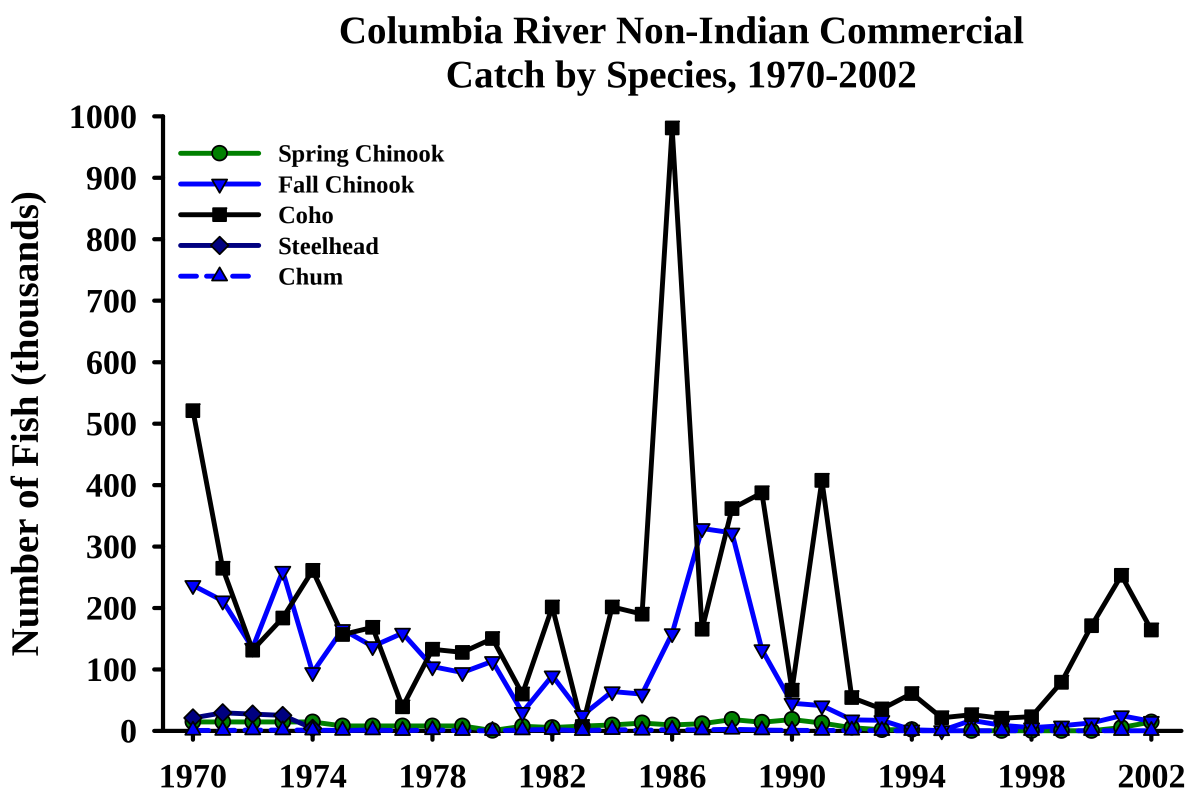

Figure 3-7. Non-Indian commercial fishery catch in the Columbia River, 1970–2002.

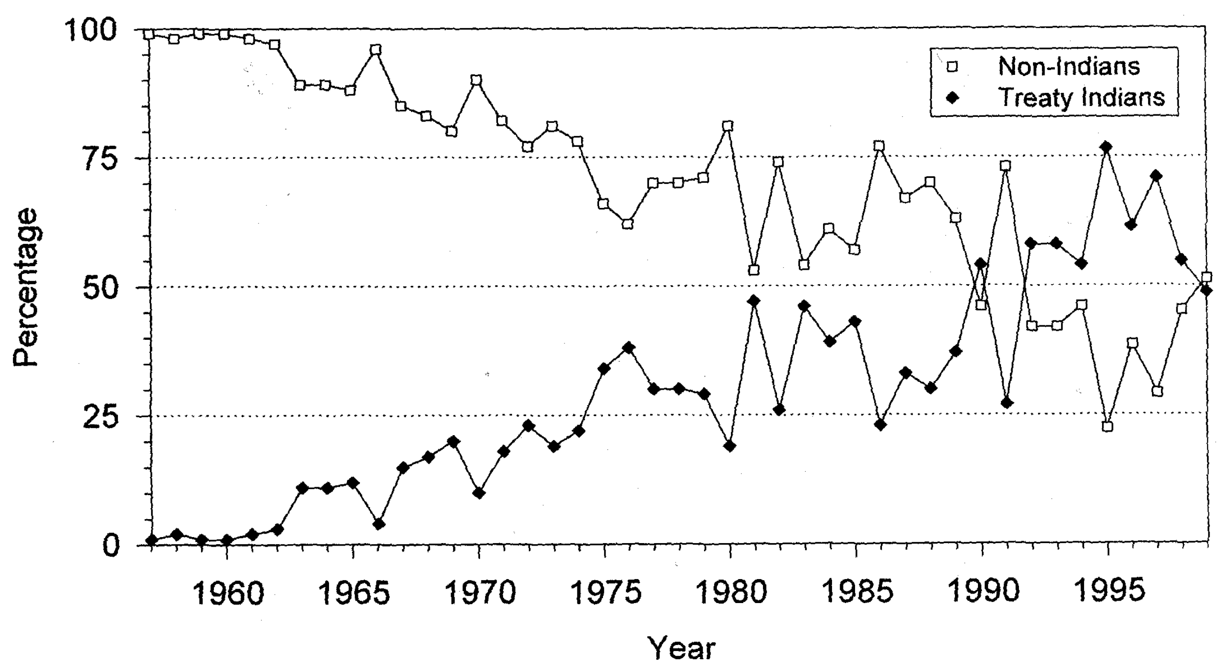

Figure 3-8. Percentage of Columbia River commercial landings of salmon and steelhead in pounds made by non-Indians and treaty Indians, 1957–02 (ODFW and WDFW 2004).

Lower Columbia River non-Indian commercial fisheries occur below Bonneville Dam in the mainstem (statistical Zones 1-5) or in select off-channel fishing areas (statistical Zones 7, 71, 74, and 80). Commercial fishing seasons in the mainstem Columbia River are established by the Columbia River Compact, while Select Area seasons are established by the state in which the fishery occurs. Zone 6 (from above Bonneville Dam to McNary Dam) was open to non-Indian commercial fishing until 1956; gill nets, set lines, and seines were used, although seines were finally prohibited in 1950. In 1957, Zone 6 was closed to non-Indian commercial fishing (see further discussion under Treaty Indian Fishery below).

The number of drift gill net licenses in the commercial fishery declined after 1938, with a low of 597 in 1969, but increased to a high of 1,524 in 1979. In 1980, a limited entry vessel permit moratorium went into effect. In the mid-1980s, 288 licenses were purchased and permanently retired; 135 licenses were bought back by Washington in 1995–96. In 1999, Columbia River commercial licenses totaled 591.

The number of seasons and fishing days allowed for the commercial mainstem fishery has declined dramatically since 1938. Initially, fishing seasons were closed only in March and April and from August 25–September 10. There has been no summer fishing season since 1964 and no spring season since 1977. Throughout the 1980s and 1990s, August and September seasons have been limited by time, area, and harvest quotas. Before 1943, over 270 fishing days were allowed annually. From 1977 through the 1980s an average of 38 fishing days were allowed annually and, in the 1990s, 29 average annual fishing days were allowed.

Commercial fishing in Columbia River off-channel areas was initiated in 1962 with the adoption of salmon seasons for Youngs Bay, Oregon. Initially, openings were concurrent with the late fall mainstem gill net seasons; however, seasons have been separate since 1977. Recent declines in mainstem fishing opportunities prompted Bonneville Power Administration (BPA) to fund a research project to expand net-pen programs to select off-channel fishing areas. The result of this effort was the Select Area Fishery Enhancement (SAFE) project, which has expanded to Tongue Point/South Channel and Blind/Knappa Slough in Oregon and Deep River and Steamboat Slough in Washington. These fisheries primarily target hatchery coho returning to the release sites; Select Area bright fall chinook also are targeted in the Youngs Bay fishery.

3.1.4.4 Lower Columbia River Recreational

Before 1975, lower Columbia River recreational fisheries primarily targeted salmon and steelhead. Season closures for spring and summer chinook and declines of other salmonids transitioned much of the effort to sturgeon (Figure 3-9). Recent-year improvements in salmonid returns and selective fishery opportunities in the recreational fishery have resulted in a rebound in salmonid angler effort, and catch of certain salmonids has also increased in the mainstem Columbia (Figure 3-10).

The lower Columbia River mainstem below Bonneville Dam is separated into two main areas for recreational harvest; Buoy 10 (ocean/in-river boundary) to the Rocky Point/Tongue Point line, and the Rocky Point/Tongue Point line to Bonneville Dam. Recreational harvest does occur in Zone 6 above Bonneville Dam, but catch is very low compared to the fisheries below Bonneville.

The Buoy 10 fishery is extremely popular, especially with small boat anglers. Chinook and coho are the targeted species, although other salmonids are harvested. The main harvest and effort time is mid-August to Labor Day and effort can be substantial, especially in years of high salmon abundance. During 1986-2000, effort in the Buoy 10 fishery ranged from 9,300 angler trips in 1994 to 186,000 angler trips in 1988.

Before 1975, recreational fisheries in the lower Columbia mainstem primarily focused on salmon and steelhead. During 1975-1983 fishery closures for spring chinook and summer steelhead severely reduced salmonid angling opportunities. During 1984–1993, improved upriver summer steelhead, upriver fall chinook, and lower river spring chinook runs provided greater salmonid angling opportunities. Poor returns in the mid- to late 1990s again limited recreation salmon fishing opportunities. Since 2001, improved spring chinook runs and selective fishery implementation has increased angler effort by approximately 100,000 trips, increasing the lower Columbia salmon and steelhead sport fishing effort to about 250,000 trips per year. Since 1986, lower Columbia sturgeon angler effort has ranged from approximately 140,000 to 200,000 trips per year.

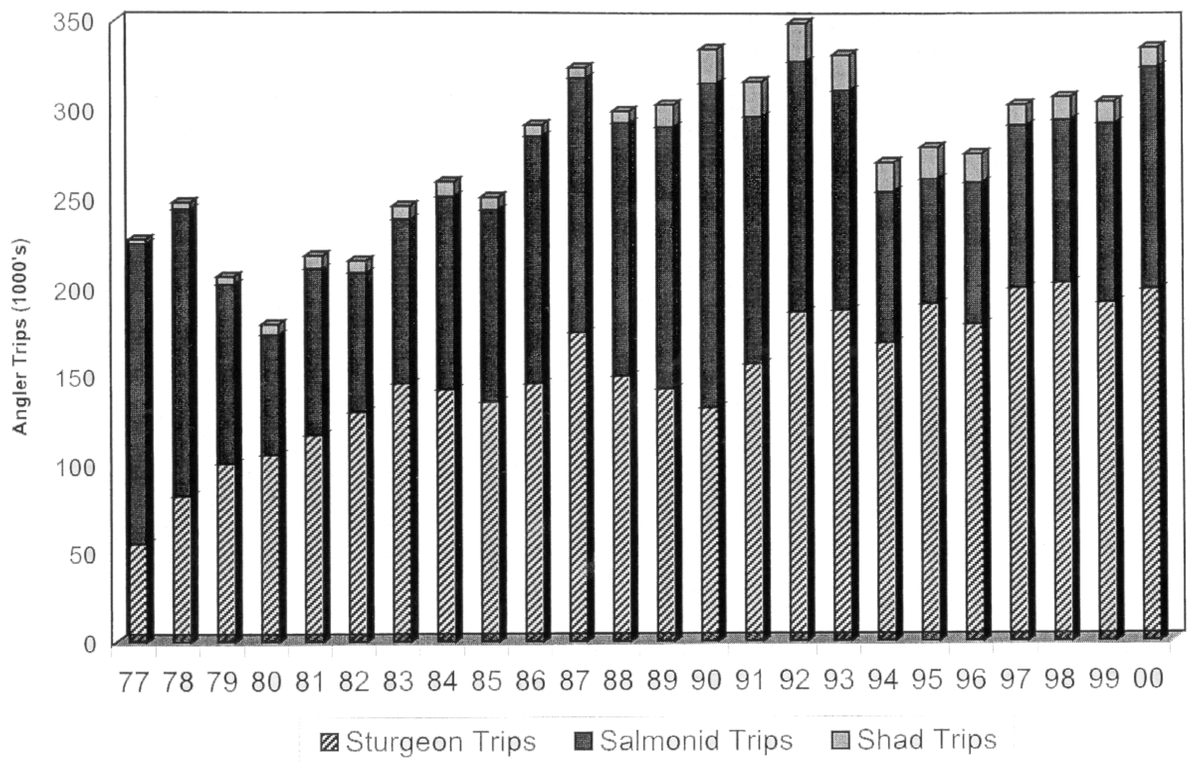

Figure 3-9. Angler effort by species on the lower Columbia River, 1977–2000.

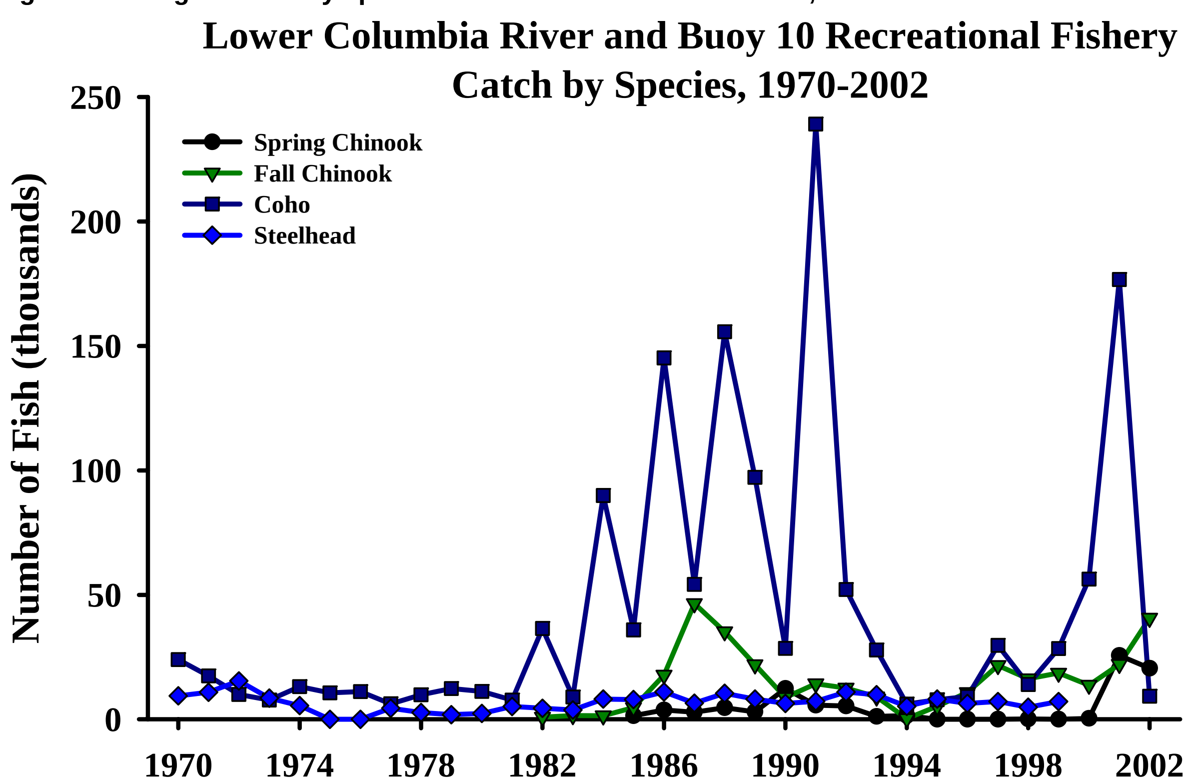

Figure 3-10. Recreational fishery catch in the lower Columbia River, 1970–2002.

3.1.4.5 Lower Columbia Tributary Recreational

Salmon and steelhead sport fishing occurs in most Washington lower Columbia River tributaries. Tributary harvest is managed consistent with objectives of the WDFW wild salmonid policy. They are principally managed to meet wild salmon and steelhead escapement objectives and to meet the objectives of the artificial propagation programs (WDFW FMEP, 2003). Fishing seasons are established based on forecasts of salmon and steelhead returning to the tributaries. In years when returns are forecasted below escapement requirements, harvest is reduced or eliminated. Harvest reductions are made by time and area closures, gear restrictions, or changes in bag limits.

Most of the tributary harvest is focused on hatchery-produced returns of steelhead, chinook, and coho. An exception is in the North Lewis River where tributary harvest of the healthy, wild fall chinook return is allowed in most years. Hatchery-produced winter and summer steelhead, spring chinook, and coho are marked as juveniles with an adipose fin-clip, which enables tributary sport anglers to identify hatchery fish for retention and release unmarked wild fish. Hatchery-produced fall chinook are not all marked, so fall chinook fisheries retain both wild and hatchery fish. However, fishing for fall chinook is prohibited in the Coweeman and East Fork Lewis rivers, where no hatchery fish are released. Trout fisheries in the streams are regulated to minimize impacts to anadromous salmonids. The general season commences June 1, after salmon smolts have migrated, and minimum size limits and gear restrictions also offer protection for juvenile salmonids.

Tributary spring chinook fisheries generally occur from February to August with a peak in April-May. Fall chinook fisheries occur from August to January, with a tule peak in late August-mid September and a Lewis bright peak in mid September-mid October. Coho fisheries occur during August-January with two peaks; early coho catch peaks in September and late coho in October. Fisheries targeting winter steelhead are concentrated from December through February and close by March 15, except the Cowlitz, Kalama, Lewis, and Washougal extend to May 31. Summer steelhead enter tributary fisheries from March through October with most of the catch occurring from late May through August (WDFW, 2003).

Tributary sport harvest of hatchery salmon and steelhead can be significant (see species sections below). Steelhead tributary fisheries harvest 30-70% of the returning hatchery adults. Steelhead returning to hatcheries are often recycled downstream to provide an additional sport catch opportunity. Harvest of hatchery spring chinook can also be substantial if forecasts indicate a strong return. Harvest rates are typically 20-40%, but can range as high as 70% in the Lewis River if there are no regulatory restrictions. Fall chinook and coho tributary harvest rates typically range from 5 to 25%, but the total numbers of fish harvested can be substantial in many years, due to large numbers of adult coho and fall chinook returning to the rivers.

3.1.4.6 Treaty Indian

Treaty Indian harvest includes commercial, and ceremonial and subsistence (C&S) fisheries. The treaty Indian set net fishery above Bonneville Dam (statistical Zone 6) involves members of the four Columbia River treaty Indian tribes: Yakama Nation, Nez Perce Tribe, Confederated Tribes of the Umatilla Indian Reservation, Confederated Tribes of the Warm Springs Reservation. The tribal C&S fisheries are of highest priority and generally occur before tribal commercial fishing. The Columbia River treaty tribes regulate treaty Indian C&S fisheries in Zone 6.

Indian and non-Indian commercial harvest was permitted in Zone 6 until 1956. The boundaries of Zone 6 were from Bonneville Dam upstream to the mouth of the Deschutes River during this period. In 1957, joint action by Oregon and Washington closed Zone 6 to commercial fishing, but treaty Indian fisheries were permitted during 1957-1968 through tribal ordinances. In 1968, the states reestablished commercial fishing in Zone 6 exclusively for treaty Indian harvest. In 1969, the upstream boundary of the zone was extended to the mouth of the Umatilla River, river mouth closure and dam sanctuary areas were established, and gear restrictions were set. The fishery is conducted primarily with set gill nets, although some dip netting still occurs primarily at Cascade Locks, the Lone Pine site, and below John Day Dam.

Similar to the non-Indian commercial fishery, the number of seasons and fishing days allowed for the treaty Indian commercial fishery has declined dramatically. Despite the decline in fishing opportunity, the percentage of Columbia River commercial fishery landings made by treaty Indians has steadily increased since the late 1950s (Figure 3-8). In 1999, 59 commercial fishing days were allowed in the treaty Indian fishery, although most of those days were in February and March during the targeted sturgeon fishery. Fishing effort targeting fall salmonids occurs in late August and September. Fall chinook harvest increased substantially in 2001 and 2002 as a result of significant increases in fall chinook returns. As with non-Indian harvest, treaty Indian harvest of salmon increased in 2001 and 2002 as a result of a significant increase in Columbia River salmon abundance (Figure 3-11)

C&S fisheries are usually open year-round; ceremonial fishing is conducted with gill nets via tribal permit while subsistence fishing is conducted by individuals primarily using dip nets, hook and line, or gill nets. Some tribal permits allow subsistence fishing with gill nets when commercial fisheries are closed. Spring chinook salmon are the most important ceremonial fish for the Columbia River treaty tribes. Significant tribal commercial harvest of spring chinook occurred in 2001 for the first time since 1977 as a result of a substantial increase in upper Columbia spring chinook returns (Figure 3-11), and a Columbia River management agreement which establishes ESA fishery impact limits based on and abundance-based management strategy.

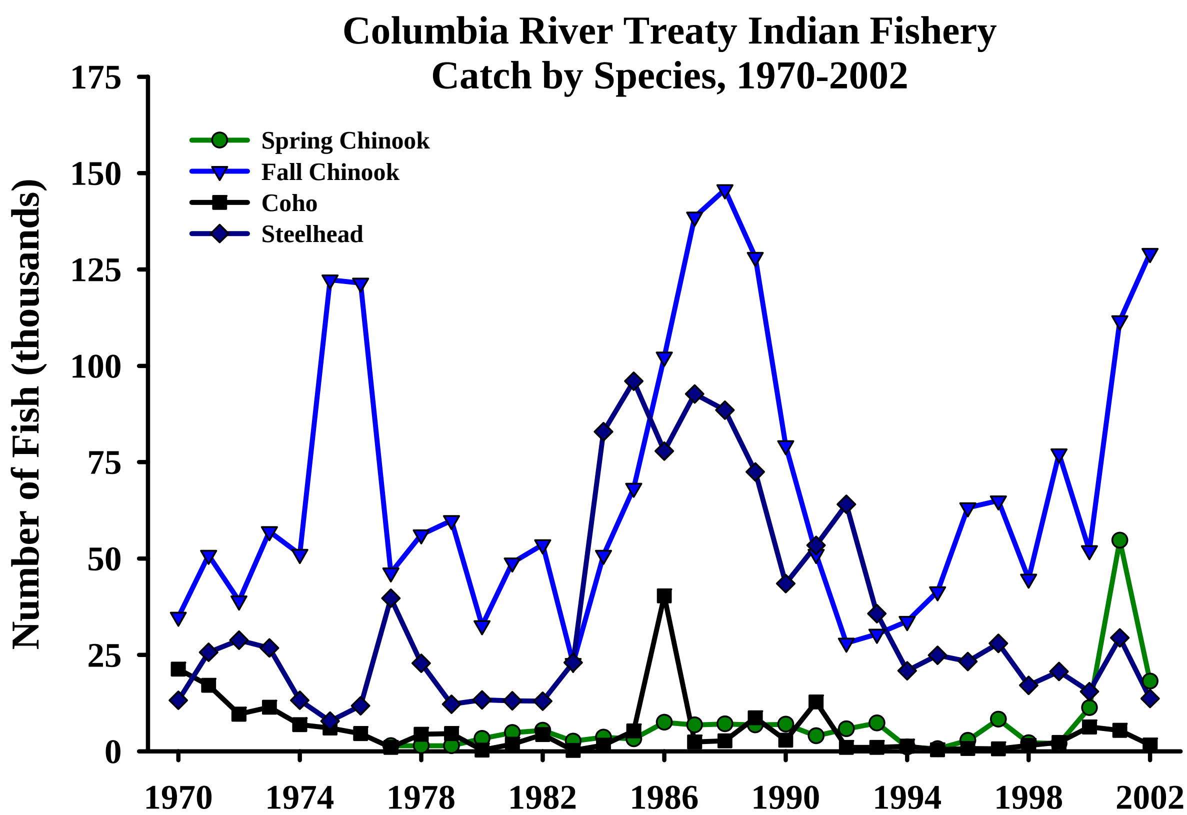

Figure 3-11. Treaty Indian fishery catch, 1970–2002.

3.1.5 Spring Chinook Fishery

Before 1976, over 50% of the mainstem Columbia River spring chinook run was harvested, primarily in April and May. After 1977, target fisheries for upriver spring chinook were eliminated and, as a result, lower Columbia River commercial fisheries ended by early March and sport fisheries closed before April. Consequently, harvest rates were reduced substantially. No lower Columbia fisheries during the April/May peak of the runs occurred again until 2001 when adipose fin-clipped hatchery adults returned, enabling fisheries to selectively retain marked hatchery fish and release unmarked wild fish. Commercial fisheries began using live capture methods in 2001, with gear changed from gillnet to tangle net web combined with on-board fish recovery boxes. These selective fishery capabilities in the lower Columbia spring chinook fisheries have increased hatchery harvest opportunity substantially while minimizing harvest mortality on wild spring chinook.

Lower Columbia River commercial harvest of spring chinook ranged from 0 to 18,300 fish during 1985–2002; Washington-origin lower Columbia spring chinook provided a small portion of the catch during the same period (harvest ranged from 0 to 2,200 for lower river stocks other than Willamette; Figure 3-12).

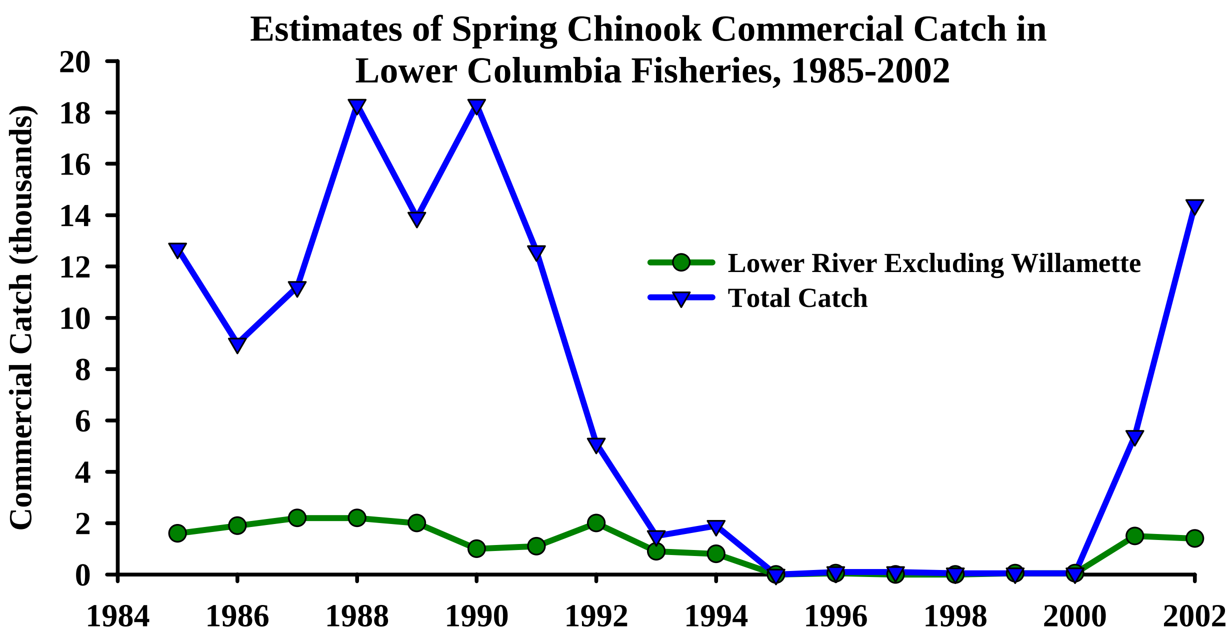

Figure 3-12. Total commercial catch (excluding the Willamette River) of spring chinook in the lower Columbia fisheries.

The 1985–2002 lower Columbia total harvest of spring chinook ranged from zero in 1995 to 32,800 in 2002. Fisheries harvest bottomed out during 1994–2000 when Columbia spring chinook runs crashed, but increased in 2001 when runs improved and again in 2002 when runs continued to improve and selective fisheries were implemented. The mainstem Columbia sport harvest of spring chinook has exceeded the commercial harvest in the two most recent years (Figure 3-13).

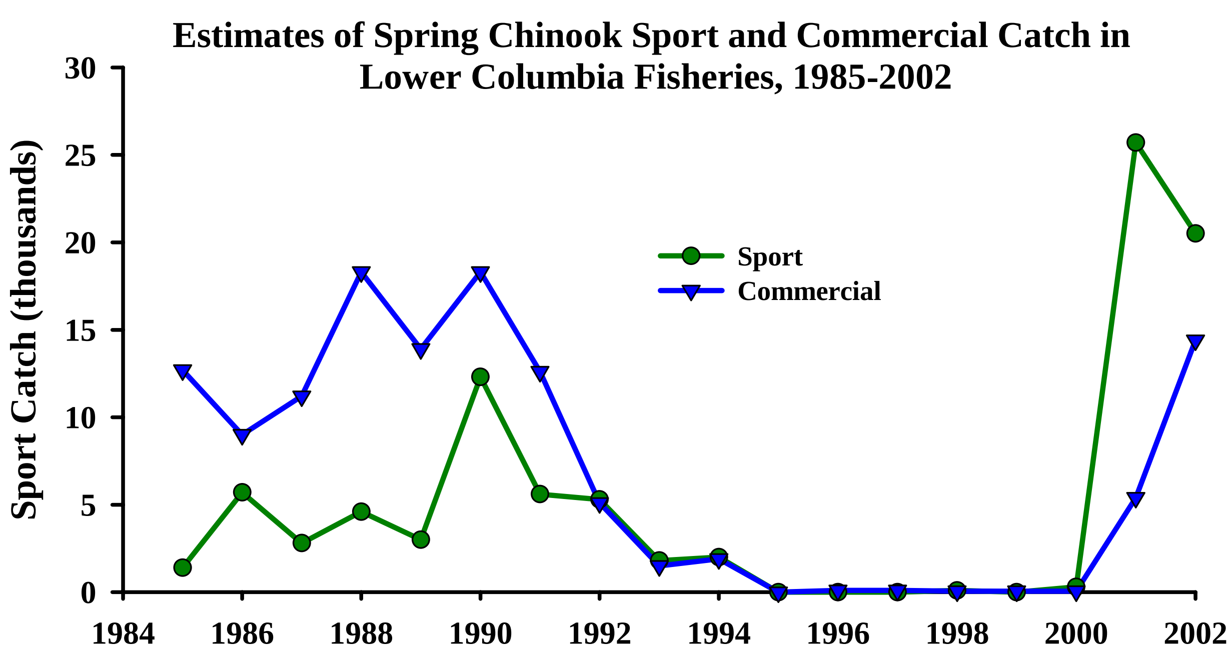

Figure 3-13. Harvest (sport and commercial) of spring chinook in the lower Columbia fisheries.

3.1.5.1 Spring Chinook Harvest Over Time

Historically, commercial seasons for spring chinook occurred in the lower Columbia River in winter and spring. The seasonal structure from 1909 to 1942 was fairly constant, with commercial fishing open 270 days each year. Before 1942, spring chinook harvest rates typically were 50% or greater. However, lower Columbia stocks were harvested at a lower rate than upper river stocks because March and most of April—peak time for lower Columbia spring chinook—was closed to fishing. Reductions in the commercial season began in 1943. The commercial spring season (late April–May) was first reduced and then in 1975 completely eliminated to protect depressed stocks of upper Columbia River wild spring chinook. From 1975 to 2001, commercial fishing was closed by early March. In 2002, full fleet selective commercial fisheries were implemented in late February to late March enabling increased harvest of hatchery spring chinook.

Sport harvest in the mainstem Columbia River was generally concentrated in April until 1975, when the spring sport fishery was closed. The sport fishery closed by mid- to late March until the coming of selective fisheries in 2001. During 2001–2003, the selective April–May sport fishery was significant for harvest of hatchery spring chinook. As the mainstem Columbia fishery has been restricted, the tributary fisheries have increased in importance. Most harvest of lower Columbia spring chinook now occurs in the tributary sport fisheries, chiefly in April and May.

Ocean harvest of spring chinook was far less than the Columbia River harvest until the 1950s, when the ocean commercial fishery grew rapidly in response to reduced commercial opportunity in the coastal rivers and estuaries. The ocean harvest of spring chinook peaked in the 1970s and, by the 1990s, was significantly reduced. Total harvest of wild spring chinook significantly reduced after selective fisheries wer implemented (Figure 3-14).

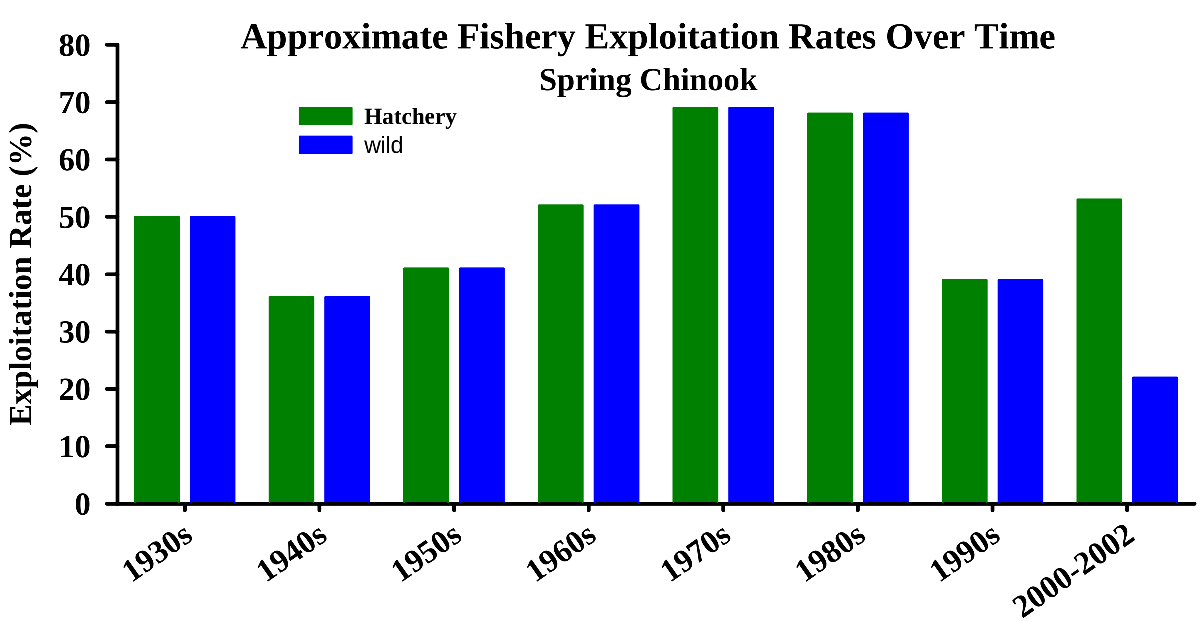

Figure 3-14. Spring chinook fishery exploitation over time. Harvest dominated by Columbia River commercial fisheries until 1950s. Ocean harvest significant 1960–1990. Sport harvest increased in 1960s. Tributary sport harvest more significant after 1975. Selective harvest in Columbia River beginning in 2001.

3.1.5.2 Current Spring Chinook Harvest Distribution

Ocean Fisheries

Current harvest impacts to wild lower Columbia spring chinook are reduced from historical impacts. The majority of harvest-related mortality of wild spring chinook now occurs in ocean fisheries because they are not selective for hatchery marked fish (whereas most Columbia River fisheries are currently selective for hatchery fish as describe in the next section). Historically, most ocean harvest occurred in Canadian fisheries although Canadian chinook fisheries have been substantially reduced in recent years. CWT recoveries from 1989-1994 brood year Cowlitz River Hatchery spring chinook determined the following distribution: Cowlitz River sport (29%), British Columbia (29%), Washington coast (22%), Columbia River (6%), Oregon coast (5%) and Alaska (3%). In the same period, Lewis River Hatchery spring chinook were distributed to: Lewis sport (69%), Alaska (11%), British Columbia (10%), Washington coast (5%), Columbia River (4%), and Oregon coast (1%). CWT data suggests that upriver spring chinook are impacted far less by ocean fisheries than are other Columbia River chinook stocks.

While lower Columbia spring and fall chinook are both harvested in Pacific Ocean fisheries, spring chinook are less subject to ocean fisheries harvest than are falls because of the differences in the patterns and timing of their migration. Although mature fish comprise the majority of the fall chinook catch in the ocean, a significant portion of the spring chinook catch can be immature fish. The impacts of the Washington ocean harvest typically depend on the abundance levels of Columbia fall chinook; these drive Washington ocean chinook quota levels. Additional details are located in Fall Chinook, PSC Fisheries, and PFMC Ocean Fisheries sections.

Future ocean harvest likely will remain similar to levels of recent years (~18%) because of PST abundance-based management agreements and the anticipation of further development of selective fisheries for chinook (Table 3-2). It is noted, however, that lower Columbia spring chinook are not included directly as a stock to be considered in abundance-based management agreements with Canada. Harvest impacts in ocean fisheries could be higher than 18% in years when chinook abundance is high for key Canadian or US fall chinook stocks. Ocean harvest could potentially be reduced if selective chinook fisheries were implemented through the PSC and PFMC processes but there are significant technical complexities in implementing selective ocean chinook fisheries.

Table 3-2. Example of lower Columbia spring chinook harvest exploitation rates under current management.

Fishery	H*	W**	Comment
Alaska	4%	4%	PSC guidelines for chinook
Canada	9%	9%	PSC abundance-based management
Washington/Oregon/California ocean	5%	5%	Quotas based on fall chinook abundance
Columbia River	15%	2%	Selective commercial and sport fisheries
Tributary	20%	2%	Selective sport fisheries
Total	53%	22%	Total lower Columbia stocks (Cowlitz, Kalama, Lewis)
			Wind and Little White Salmon are upriver stock; ocean harvest is negligible, but total harvest may be similar to lower Columbia hatchery spring chinook because Columbia harvest includes treaty Indian fishery upstream of Bonneville Dam

* H denotes hatchery fish exploitation rate. Columbia River fisheries managed for commercial/sport allocation and hatchery escapement.

** W denotes wild fish exploitation rate. Columbia River fisheries managed to meet ESA standards for wild Willamette and upriver spring chinook.

In-river Commercial

In the Columbia River, spring chinook are harvested in non-Indian winter commercial gillnet fisheries. From 1938 to 1973, approximately 55% of upriver spring chinook runs were harvested in directed Columbia River commercial and sport fisheries. During 1975-2000 (excluding 1977), no lower river fisheries targeted upriver stocks and commercial fisheries focused on Willamette spring chinook. Recent conservation measures to protect Willamette River spring chinook required the release of wild Willamette spring chinook in all freshwater fisheries. Additionally, since 2001 Columbia River sport and commercial fisheries have been able to retain adipose fin-clipped hatchery fish only and must release unmarked wild fish. As a result, a new tangle net commercial fishery was developed in Zones 1-5 that was selective for adipose fin-clipped hatchery spring chinook. Multiple gear and education requirements were mandatory for all fishery participants. The new regulations were adopted to improve the survival rates of wild fish captured and released during the fishery. (Lower Columbia fishery impacts on wild spring chinook now come primarily from catch and release handling mortality.) Although upriver wild spring chinook are retained in treaty Indian fisheries, total impacts to upriver spring chinook are constrained by ESA impact limits.

A 2001 management agreement negotiated between the *US v. Oregon* parties (states, tribes, federal agencies) and NOAA Fisheries concerning limitations on ESA-listed upriver and Snake River wild spring chinook allowed for a 17% total impact rate on ESA-listed upriver spring chinook and 2% of this impact was allocated to non-Indian fisheries. The 2% non-Indian allocation was further allocated among commercial and sport fisheries in the lower Columbia; at 1.02% for sport, and 0.68% for commercial. The remaining 0.3% was reserved for upper Columbia and Snake River non-Indian fisheries. Spring chinook are harvested in Zone 6 Indian winter commercial fisheries although sturgeon are the primary target species for the winter fishery. Spring chinook are harvested annually in both tribal commercial gillnet and C&S Zone 6 spring fisheries. The focus for tribal spring fisheries is to attain at least 10,000 spring chinook for ceremonial needs. Since 2001, increased Upper Columbia spring chinook abundance has enabled significant tribal ceremonial and subsistance harvest as well as commercial harvest (Figure 3-15).

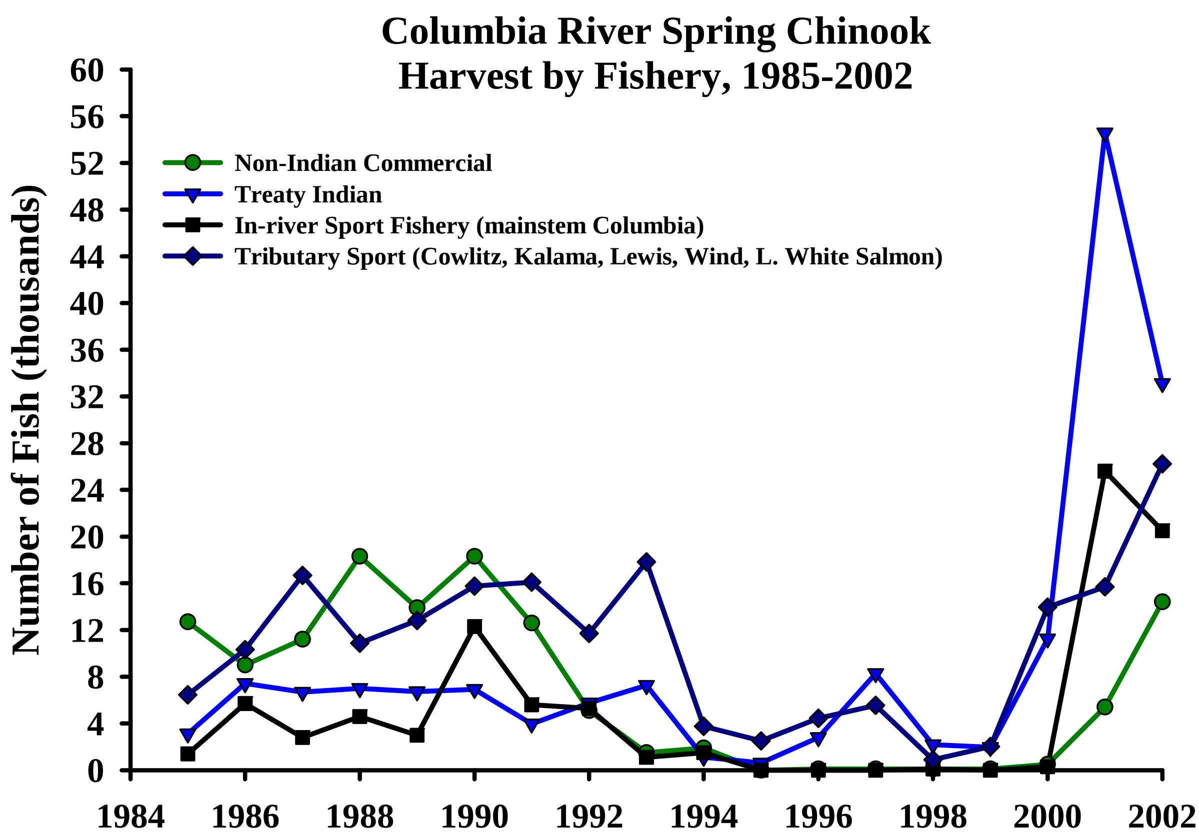

Figure 3-15. Harvest of spring chinook in the Columbia River mainstem from 1985–2002.

In-river Sport

Spring chinook are the focus of considerable recreational fishing effort in Columbia River estuary, mainstem, select area, and tributary fisheries. In recent years, harvest has been selective for adipose fin-clipped hatchery fish. The selective fishery strategy has enabled the mainstem sport fishery to extend into April and May for the first time since 1977. The major hatchery populations in the lower Columbia River contributing to these fisheries include Cowlitz, Kalama, Lewis, Wind, and Little White Salmon spring chinook. The Wind and Little White Salmon tributary sport fisheries are not yet selective, but are expected to become selective in 2005 when all returning hatchery adults will be adipose fin-clipped.

Substantial spring chinook sport fisheries have existed in some lower Columbia subbasins. Average annual spring chinook sport harvest during the late 1970s and early 1980s was 6,410 in the Cowlitz River, 1,149 in the Kalama basin, and 5,504 in the Lewis River. Total annual sport harvest in the Cowlitz, Kalama, and Lewis rivers combined was about 6,000 to 15,000 for the years 1980–93, but has dropped to 3,200 or less since 1994 (Figure 3-16). The reduction in sport harvest corresponds to reduction in spring chinook runs to these rivers beginning in the mid-1990s.

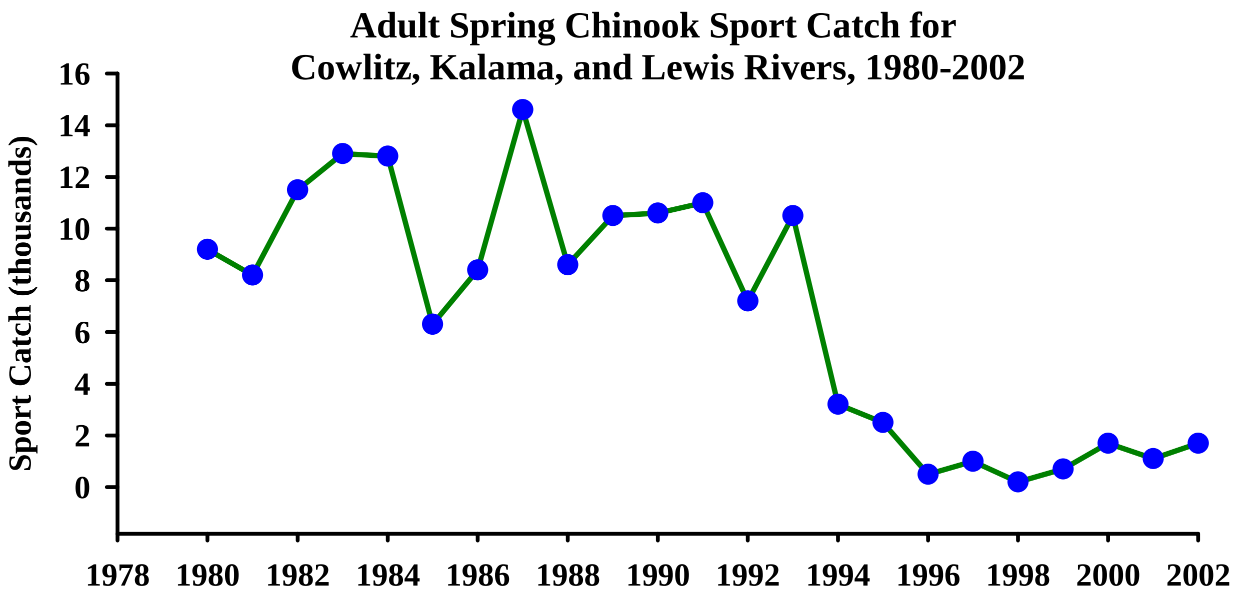

Figure 3-16. Total sport harvest of adult spring chinook in the Cowlitz, Kalama, and Lewis rivers.

Sport harvest is substantial in the Wind and Little White Salmon (Drano Lake) and much larger than the Lewis, Cowlitz, or Kalama in recent years, with some years' sport harvest exceeding 10,000 fish. Harvest in the Wind and Little White Salmon is shared between the sport fishery and subsistence and commercial harvest by the Yakama Nation.

3.1.5.3 Spring Chinook In-River Harvest Management Details

Annual spring chinook fisheries in the mainstem Columbia are planned consistent with a 2001-2005 agreement between the state, federal, and tribal parties to the *US v. Oregon* federal court case. The agreement establishes the total harvest impact limits for ESA-listed upriver origin wild spring chinook and treaty Indian and non-Indian harvest sharing. The lower Columbia fisheries are also regulated consistent with ESA limits on Willamette wild spring chinook and sport and commercial allocation of Willamette hatchery spring chinook. Regulations are being developed to establish ESA limitations on lower Columbia River wild spring chinook, however, this regulation development process has lagged behind similar processes that established ESA limitations on upriver and Willamette wild spring chinook.

When entering the Columbia River, spring chinook have unique migratory characteristics specific to their stocks. Upper and lower Columbia spring chinook stocks enter the Columbia River at different times. Harvest managers make use of these differences to set different seasons for different stocks so that harvest rates can be adjusted. In both the mainstem Columbia sport and commercial fisheries, as well as the tributary sport fisheries, current Columbia River management employs selective fishing for marked hatchery spring chinook.

Lower Columbia River spring chinook stocks can be separated into two groups for in-river fisheries management; lower river spring chinook (Cowlitz, Kalama, and Lewis iver populations in Washington and Willamette River in Oregon), and upriver spring chinook (Wind and Little White Salmon River populations).

Mainstem Columbia River harvest impacts on Willamette wild spring chinook average 4.3%, while the Snake River wild limits for lower Columbia fisheries are 1.7% (ODFW and

WDFW, 2001). The Willamette spring chinook migration through the lower Columbia is earlier than lower Columbia River spring chinook; Snake River spring chinook are later timed. Therefore, a mid-range impact of approximately 3% is a reasonable expectation for lower Columbia River wild spring chinook stocks in mainstem Columbia fisheries.

Select Area fisheries for spring chinook were developed in the mid-1990s along the Oregon shore of the Columbia River, primarily in Youngs Bay. Spring chinook smolts are released in off-channel areas outside of the normal migration corridor for populations of wild and hatchery spring chinook and are harvested in subsequent years near the release sites. One site on the Washington side of the Columbia River (Deep River) has had limited success for spring chinook select area fisheries. The existing Select Area fisheries likely harvest few spring chinook destined for Washington tributaries of the lower Columbia River basin.

The *US v. Oregon* agreement for spring chinook management establishes a sliding scale of harvest impact limits for ESA-listed upriver origin wild spring chinook based on the abundance of wild Snake River spring chinook. The agreement also establishes treaty Indian and non-Indian harvest sharing (Table 3-3). Fisheries that selectively harvest hatchery fish have dramatically reduced the impacts of the non-Indian fishery on wild fish. (treaty Indian fisheries are not limited to hatchery fish.) The lower Columbia fisheries are also regulated consistent with ESA limits on Willamette wild spring chinook, 20% for 2001 and 15% for 2002 and beyond.

Table 3-3. Sliding scale* of harvest impacts on wild upriver spring chinook based on Snake River wild spring chinook run size (adapted from the 2001-05 Interim Management Agreement).

Columbia River Mouth Run Size	Snake River Run Size**	Proposed Tribal Harvest Rate	Non-Indian Harvest Rate[§]	Total Harvest Rate	Non-Indian Wild Limited Rate
<25,000	<2,500	5%	<0.5%	<5.5%	<0.5%
25,000	2,500	5%	0.5%	5.5%	0.5%
30,000	3,000	5%	1%	6%	0.5%
40,000	4,000	6%	1%	7%	0.5%
50,000	5,000	7%	1.5%	8.5%	1%
75,000	7,500	7%	2%	9%	1.5%
100,000	10,000	8%	2%	10%	
130,000	*13,000*	*9%*	*2%*	*11%*	
200,000	20,000	10%	2%	12%	
250,000	25,000	11%	2%	13%	
300,000	30,000	12%	2%	14%	
350,000	35,000	13%	2%	15%	
400,000	40,000	14%	2%	16%	
450,000	45,000	15%	2%	17%	

Italics indicate 2003 preseason projections; the spring chinook run forecast at the river mouth is 145,400.

* This scale is applied if the Snake River wild spring chinook run is ≥7.5% of the total run. The limited harvest rate would be used if the Snake River wild forecast is less than 7.5% of the total run.

**If the Snake River wild spring chinook forecast is less than 10,000, the total harvest rate is restricted to 9% or less. When wild fish harvest rate is restricted to 9% or less, non-Indian fisheries transfer 0.5% harvest rate to treaty Indian fisheries, however, non-Indian fisheries would never go below a 0.5% harvest rate.

[§] If the total forecast is <25,000 or the Snake River forecast is <2,500, the non-Indian harvest rate would be maintained as close to zero as possible while maintaining minimal fisheries targeting other harvestable species.

Non-Indian sport and commercial allocation is based on abundance of upriver wild spring chinook as well as Willamette hatchery spring chinook (Table 3-4). The 2003 mainstem Columbia River spring chinook allocation for non-Indian fisheries was guided by five major principles: 1) meet conservation requirements for wild spring chinook, including ESA-listed species, 2) manage spring chinook harvest within the provisions of the *US v. Oregon* management agreement, 3) meet hatchery escapement goals, 4) implement selective fisheries to focus sport and commercial harvest on hatchery fish, and 5) allocate 15% of the non-Indian upriver spring chinook impacts to sport and non-treaty Indian fisheries upstream of McNary Dam and provide for a lower river fisheries management buffer.

Table 3-4. Allocation of non-Indian upriver wild spring chinook impacts based on Willamette hatchery and upriver wild spring chinook abundance.

		Willamette Hatchery Fish Run Size		
		<40,000	**40-75,000**	**>75,000**
Upriver Fish Run Size* (Impacts)	30-<50,000 (0.85%)	Comm—10% (0.08)** Sport—90% (0.77) §	Comm—30% (0.25) Sport—70% (0.60)	Comm—25% (0.21) Sport—75% (0.64)
	50-<75,000 (1.25%)	Comm—40% (0.5) Sport—60% (0.75)	Comm—35% (0.44) Sport—65% (0.81)	Comm—30% (0.37) Sport—70% (0.88)
	>75,000 (1.7%)	Comm—50% (0.85) Sport—50% (0.85)	Comm—40% (0.68) Sport—60% (1.02)	***Comm—35% (0.59)*** ***Sport—65% (1.11)***

Italics indicate the 2003 estimated run sizes and allocation among non-Indian commercial and sport fisheries

* An upriver run size update along with an assessment of upriver impact needs and Willamette allocation will be conducted after mid-April.

** If the sport fishery impact allocation will be used before May 15 and the commercial fishery does not need its entire upriver impact allocation to attain the Willamette allocation or an equitable catch share, commercial impacts may be transferred to the sport fishery.

§ If the sport fishery does not need their entire upriver spring chinook allocation to continue the fishery through May 15, the remaining sport impacts may be transferred to the commercial fishery for late spring commercial fishing opportunity.

Every year, after annual run size forecasts are available and public input has been received, the Columbia River Compact sets the structure of sport and commercial fisheries to meet allocation policies and fisheries objectives. Initial fishery planning is based on preseason run forecasts, but seasons are adjusted for smaller or larger runs based on dam counts and information about fishery catch rates. Fish run sizes and catches are monitored in-season so that catch does not exceed allowed guidelines.

Commercial harvest constraints resulting from low abundance of wild fish and ESA limitations to protect listed stocks provided much of the motivation for the development of a new fishery. Meanwhile, the recent hatchery practice of marking all hatchery releases with an adipose fin clip gave the fisheries the capability of selecting hatchery fish. Starting in 2000, modifications to gillnet gear (e.g. reducing mesh size to a maximum of 4½ inches) were tested to evaluate their effectiveness: could hatchery fish be retained and wild fish be released and survive? Gear testing indicated that, while the small mesh gill nets could not gill chinook salmon, they could retain live chinook salmon by tangling. This meant fish could be retained or released after determining whether they were of wild or hatchery origin.

A 2002 winter season demonstration involved a non-Indian commercial tangle net fishery using 5½ inch maximum mesh size and targeting hatchery spring chinook salmon. Salmon catches increased throughout the duration of the fishery; chinook adipose fin mark rate ranged from 42 to 72% and averaged 50% for the season. Chinook catches and impact rates are presented in Table 3-5. The steelhead:chinook ratio decreased over the period of the fishery. Early on, the ratio averaged 2.5:1; during the middle part of the fishery, the ratio averaged 0.9:1; and at the end of the fishery, the ratio averaged 0.4:1. Steelhead mark rate fluctuated between 20 and 50%; season average steelhead mark rate was 40%. A total of 21,600 steelhead were handled and it is possible that some steelhead were handled more than once. Immediate mortality rate for steelhead was estimated at 2%; Most of the steelhead (84%) handled were released in condition 1 (vigorous, not bleeding). Some steelhead handled may have been summer steelhead, rather than winter steelhead.

Table 3-5. Spring chinook catch and released during the 2002 non-Indian commercial tangle net fishery in the lower Columbia River.

	Spring Chinook Kept				Spring Chinook Released			
Fishing Period	Upriver	Willamette River	Other Lower River	Total	Upriver	Other Lower River	Total	Upriver Impacts[a]
1/7–2/15	19	115	20	154	25	29	54	0.007%
2/25–3/1	175	311	52	538	317	97	414	0.015%
3/4–3/8	302	386	76	764	426	132	558	0.022%
3/10–3/15	1,037	897	205	2,139	1,690	475	2,165	0.082%
3/17–3/22	3,417	1,824	384	5,625	4,967	741	5,708	0.251%
3/24–3/25	1,489	955	190	2,634	2,623	422	3,045	0.123%
3/26–3/27	2,051	744	148	2,943	2,779	253	3,031	0.145%
Season Totals	*8,490*	*5,232*	*1,075*	*14,797*	*12,827*	*2,149*	*14,975*	*0.645%*

a Upriver impacts were derived directly from WDFW fishery monitoring data; impacts are calculated based on the percent of upriver spring chinook handled during the fishery, total spring chinook catch for the fishery, upriver spring chinook run size, and a long-term catch and release mortality factor.

After analysis of this 2002 fishery, objectives for the 2003 tangle net fishery were identified as:

1. provide commercial fishers with an opportunity to harvest their allocation of surplus Willamette hatchery spring chinook,
2. manage the fishery to remain within ESA-related impact limits for listed upriver and Willamette River wild spring chinook stocks,
3. improve steelhead condition at capture and reduce steelhead handling and mortality, and
4. maintain adequate spring chinook catch rate to limit total fishing time.

The selective fishery management for spring chinook commercial fisheries has increased opporunity and harvest volume compared to recent recent past The 2002 commercial spring chinook fishery ex-vessel (value), increased from an average of $686,000 during 1988-1997 to $1,462,000 in 2002 (Table 3-6).

Table 3-6. Ex-vessel value (in thousands of dollars expressed in 2002 dollars) of in-river commercial harvest of Columbia River spring chinook, 1988–2002.

		Oregon		Washington	
		Non-Indian Gill Net	Treaty Indian	Non-Indian Gill Net	Treaty Indian
1988–97	Price per Pound	3.87	3.38	4.43	4.20
	Ex-V. Value	433	2	245	6
1998	Price per Pound	2.75	0	0	4.29
	Ex-V. Value	98	0	0	*
1999	Price per Pound	2.97	0	2.98	4.23
	Ex-V. Value	84	0	*	*
2000	Price per Pound	2.79	2.91	5.01	1.97
	Ex-V. Value	236	2	16	52
2001	Price per Pound	2.67	1.39	3.84	1.28
	Ex-V. Value	594	34	135	283
2002	Price per Pound	2.95	1.21	4.23	1.18
	Ex-V. Value	932	17	295	218

* Less than $500.

Treaty Indian spring chinook fisheries occur in the Wind River and in Drano Lake (Little White Salmon) following annual agreement with WDFW regarding sport and Indian catch allocation. The Yakama Nation sets regulations for subsistence fisheries in the Wind River and commercial fisheries in Drano Lake. Washington sets commercial regulations consistent with the tribal regulations. In recent years, the Columbia River Compact has adopted rules allowing Yakama tribal members to sell Drano Lake commercially-caught spring chinook in Oregon. Yakama Tribes also collect surplus spring chinook at Carson and Little White Salmon hatcheries for ceremonial and subsistence purposes. The tribal harvest and surplus distribution in these tributaries has increased in recent years in response to larger returns (Figure 3-17).

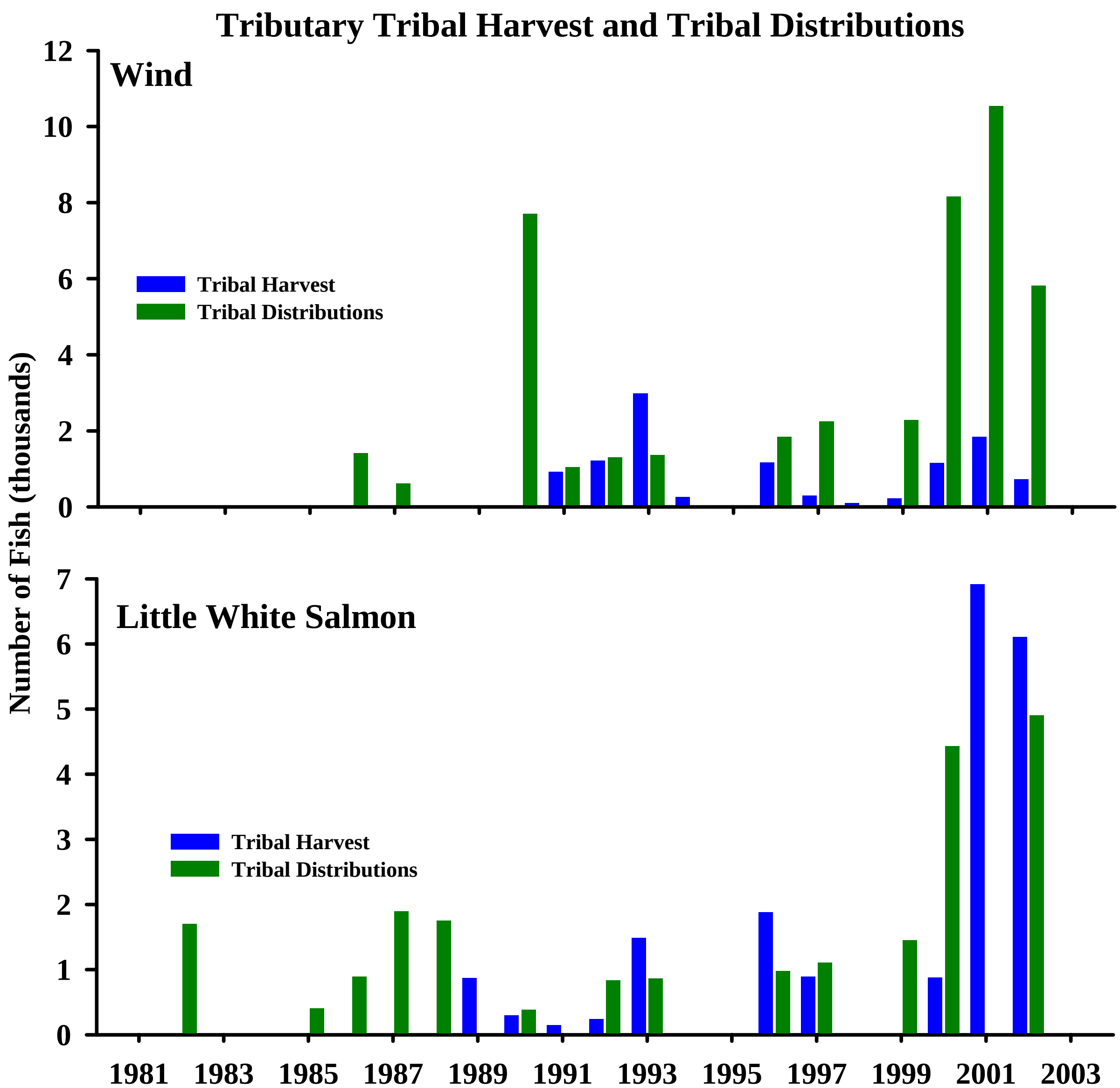

Figure 3-17. Tributary tribal harvest and tribal distributions of spring chinook in the Wind and Little White Salmon rivers, 1982–2002.

Significant spring chinook sport fisheries have existed in the lower mainstem and many lower Columbia basins. Sport seasons are set by the Washington Fish and Wildlife Commission and managed and monitored in-season by WDFW. Sport harvest is substantial in the Wind and Little White Salmon (Drano Lake) rivers and is much larger than the Lewis, Cowlitz, or Kalama in recent years, with total sport harvest recently exceeding 10,000 fish (Figure 3-18). Harvest in the Wind and Little White Salmon is shared between the sport fishery and subsistence and commercial harvest by the Yakama Nation.

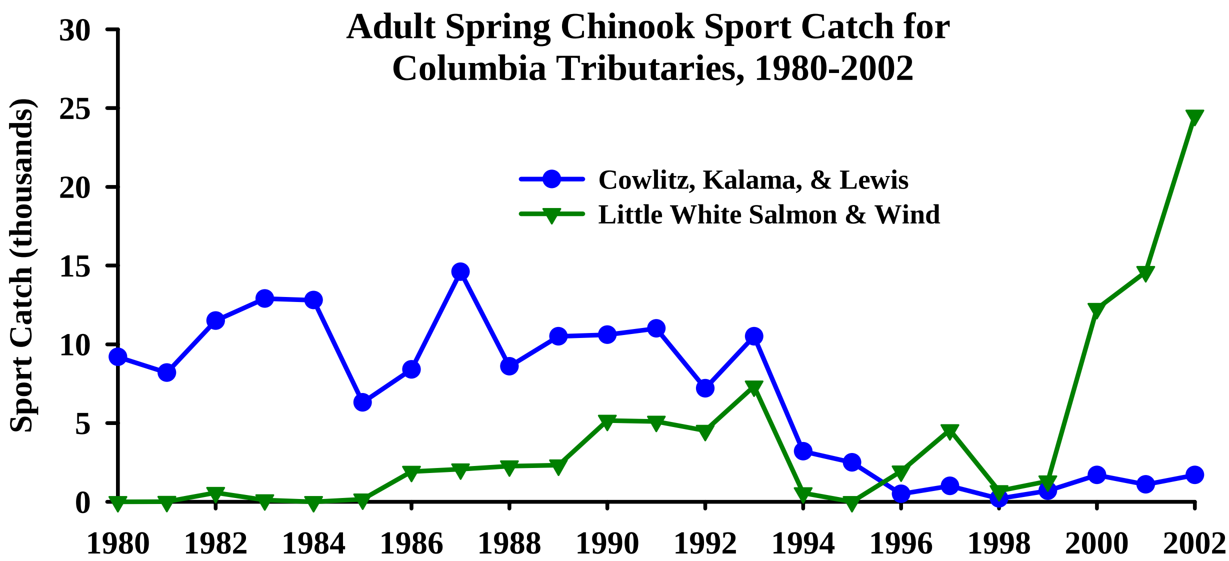

Figure 3-18. Total sport harvest of adult spring chinook in lower Columbia tributaries and Bonneville area tributaries fisheries.

Significant angler effort is expended during Columbia River recreational fisheries, creating significant economic impacts. Recreational fishing effort and angler satisfaction have increased in recent years compared to the 1990s because of hatchery-selective fishing opportunity. There is significant spring chinook fishing effort in the mainstem Columbia below Bonneville Dam, and the Willamette, Cowlitz, Kalama, Lewis, Wind, and Little White Salmon rivers all support significant tributary sport fisheries.

3.1.6 Fall Chinook Fishery

Columbia River fall chinook are harvested in ocean commercial and recreational fisheries from Oregon to Alaska, as well as the Columbia River commercial gill net and sport fisheries. Lower Columbia tule fall chinook are an important contributor to Washington ocean troll and sport fisheries as well as the Columbia River estuary sport fishery. In the past, harvest rates on fall-run stocks have been moderately high, with an average total exploitation rate of 65% (1982–1989 brood years) (PSMFC 1994). The average ocean exploitation rate for this period was 46%, while the freshwater harvest rate on the fall run has averaged 20%, ranging from 30% in 1991 to 2.4% in 1994.

Currently, ocean and mainstem Columbia River fisheries are managed for Snake and Coweeman River wild fall chinook ESA harvest rate limits, consequently limiting harvest of other co-mingled Columbia River fall chinook stocks. Unlike spring chinook, hatchery fall chinook are not marked so total harvest rate is the same for hatchery and wild fish. Ocean and mainstem Columbia River fisheries on tule stocks are limited to a 49% harvest rate because of the ESA harvest limits on Coweeman fall chinook. Columbia River harvest of Snake River fall chinook is limited to 31.29%, of which 8.25% is non-Indian harvest, and 23.04% is treaty Indian harvest. These ESA harvest limits on Snake River and Coweeman fall chinook were established in consultation processes between state and tribal governments and NOAA Fisheries. Coweeman fall chinook were selected to represent lower Columbia tule fall chinook stocks because they have not been influenced by hatchery production, and are considered a genetic legacy. These are maximum harvest rates and actual harvest is often less. Annual harvest varies depending on management response to annual chinook abundance determined in PSC, PFMC, and Columbia River Compact forums. Considerable basin-specific data are available to address harvest effects on, and specific distribution of, distinct Columbia fall chinook stocks.

Harvest of lower Columbia fall chinook is managed within four separate, broad stock units:

- Lower River Hatchery (LRH) stock are an earlier spawning component and contain both hatchery and naturally produced fish returning to most of the Washington lower Columbia tributaries.
- Lower River Wild (LRW) stock is primarily produced from the Lewis River and is all naturally produced.
- Upriver Bright (URB) stock is primarily produced in the Columbia Basin upstream of the lower Columbia area, but there are non-listed URB natural spawners present in the mainstem Columbia immediately below Bonneville Dam and in the lower Wind River.
- Bonneville Pool Hatchery (BPH) stock are an earlier spawning hatchery component released at Spring Creek Hatchery upstream of Bonneville Dam with some natural spawning components in tributaries between Bonneville and The Dalles dams.

These three stocks have different migratory characteristics and there are management criteria specific to each stock. Columbia River fisheries are managed based on annual forecasts of abundance for each stock in aggregate. Tributary fisheries are managed based on the annual abundance of returns to the specific tributaries. The harvest of fall chinook in the Columbia River is subject to *US v. Oregon* Fall Management Agreements regarding Indian and non-Indian allocation, as well as agreements on the allocation of sport and commercial fishing and ESA requirements for listed fall chinook. Additionally, annual agreements for allocation of harvest between sport and commercial and ocean and Columbia River fisheries are made during the North of Falcon process, a public process aimed at balancing harvest and fishery escapement objectives between ocean and freshwater users. The Columbia River Compact (Oregon and Washington joint regulatory forum) sets specific Columbia River commercial and sport seasons that meet the intent of the annual agreements.

Annual ocean harvest of Columbia River fall chinook is developed through provisions of the Pacific Salmon Treaty and the Pacific Fishery Management Council process for fisheries off the coasts of Washington, Oregon, and California. Lower Columbia fall chinook ocean harvest occurs primarily off the coasts of British Columbia and Washington.

Columbia River fall chinook are an important contributor to ocean fisheries from Oregon to Alaska. The LRH component is the most southerly distributed and the abundance of this stock is a major consideration when setting chinook harvest levels off the Washington coast. The LRH fish also contribute significantly to Canadian fisheries. LRW and URB components are more northerly distributed in the ocean.

The modern day commercial harvest of lower Columbia fall chinook peaked during 1987–88 when record fall chinook numbers returned to the Columbia River. Harvest of lower river hatchery stock (tules) was almost 180,000 adults and lower river wild stock was nearly 19,000 adults (Figure 3-19). The commercial harvest of lower river fall chinook reduced significantly after 1989 and remained low through the 1990s.

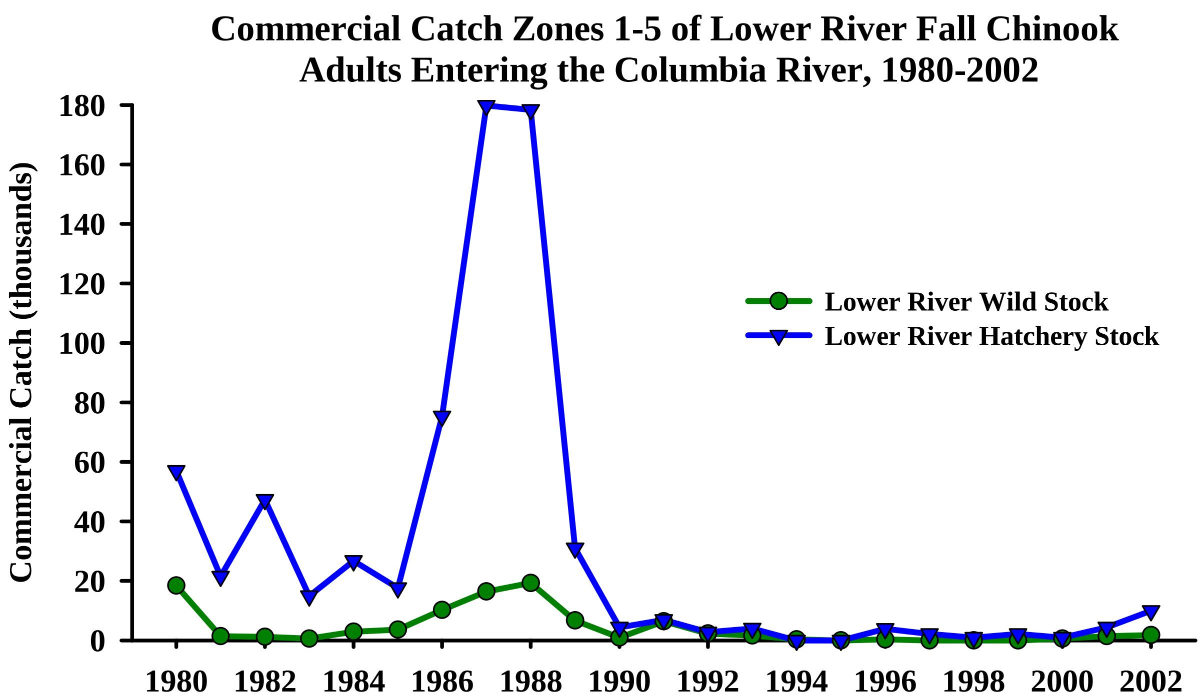

Figure 3-19. Commercial harvest of lower river wild and lower river hatchery stock fall chinook in the Columbia River.

Columbia sport harvest of lower river chinook peaked in 1987–89, with lower river hatchery harvest nearly 33,000 in 1987 and lower river wild harvest nearly 5,000 in 1989 (Figure 3-20).

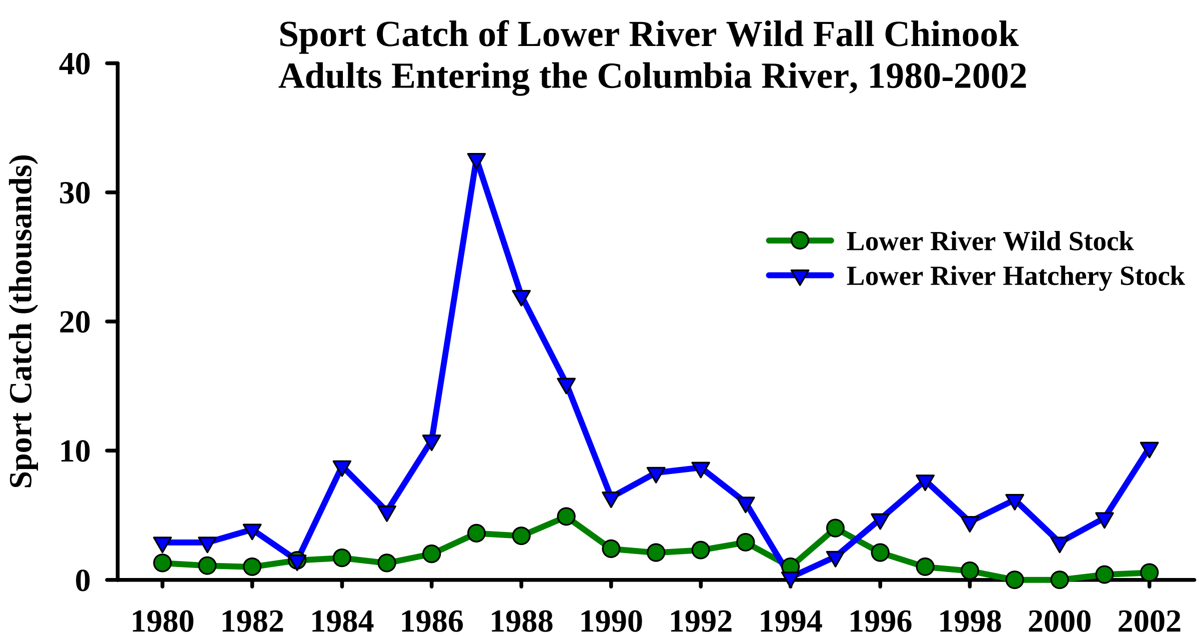

Figure 3-20. Sport harvest of lower river wild and lower river hatchery fall chinook stocks in the Columbia River.

3.1.6.1 Fall Chinook Harvest Over Time

Lower Columbia fall chinook were historically harvested in Columbia River fall fisheries from August to October. Before 1949, Columbia River commercial seasons were open daily during the fall, except for a closed period from August 25 to September 10. Most harvest was from Columbia River commercial fishing until the 1950s, when ocean fisheries increased in response to reduced Columbia River and coastal estuary commercial fisheries. Ocean harvest peaked in the 1970s, but in the 1990s reduced significantly in response to declines in the abundance of Columbia River tule chinook. Columbia River mainstem sport fisheries for fall chinook began increasing in the 1980s, and now the annual mainstem sport harvest of fall chinook is similar to the commercial fishery. Fall chinook tributary fisheries advanced in popularity in the 1960s. Most tribal chinook harvest occurred in a dip net fishery at Celilo Falls, with tribal commercial landings of salmon ranging from 0.8 to 3.5 million pounds annually during 1938–1956. The Celilo fishery ended in 1957 with the inundation of the falls by The Dalles Dam. Commercial fishing in Zone 6 (Bonneville to McNary dams) was closed by state law during 1957–1967. It reopened exclusively for treaty Indian commercial fishing in 1968 following federal court decisions regarding treaty Indian fishing rights. Since 1980, URB fall chinook have been the primary fall chinook harvested in the Columbia River, however the harvest of LRH stock has also been very large in some years. The largest harvest of fall chinook occurred in 1987 (Figure 3-21), when a record 872,000 fall chinook adults returned to the Columbia River.

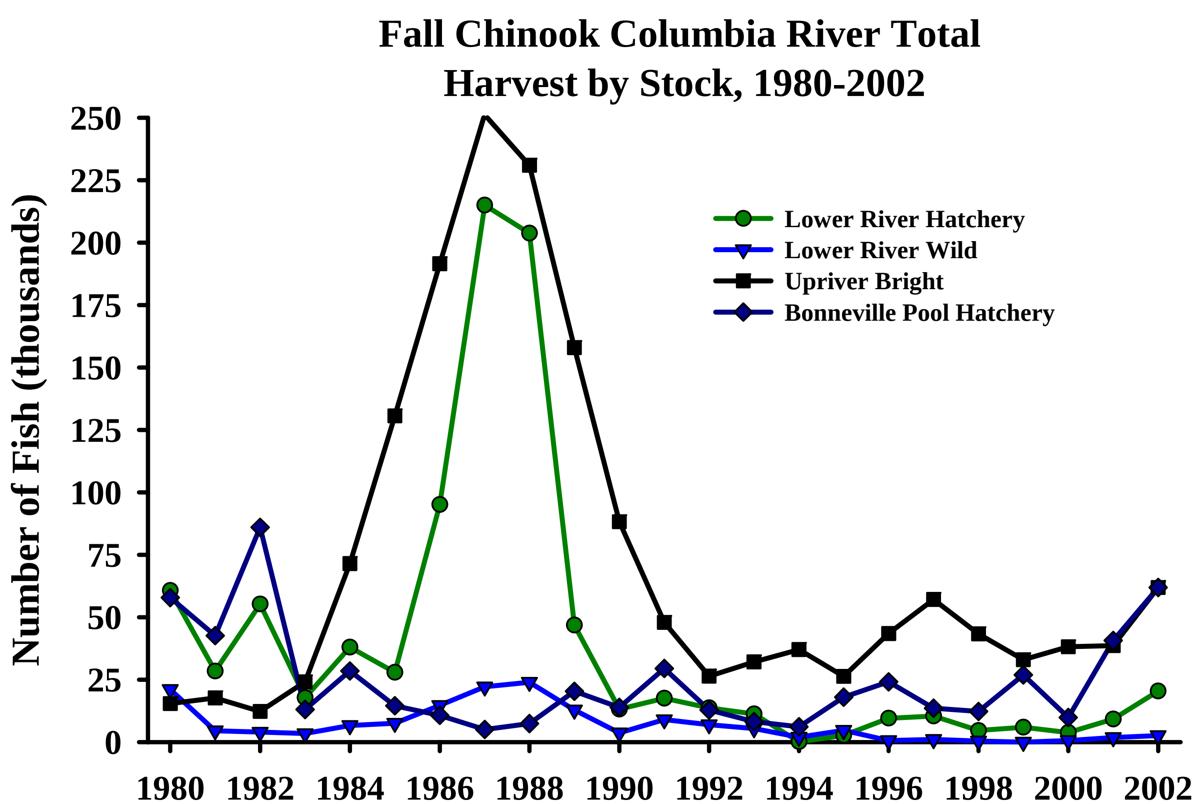

Figure 3-21. Total harvest of fall chinook in the Columbia River from 1980–98.

In general, the approximate fall chinook fishery exploitation rate over time held steady around 70-80% until the 1990s when fisheries were reduced as a result of ESA-driven management changes (Figure 3-22).

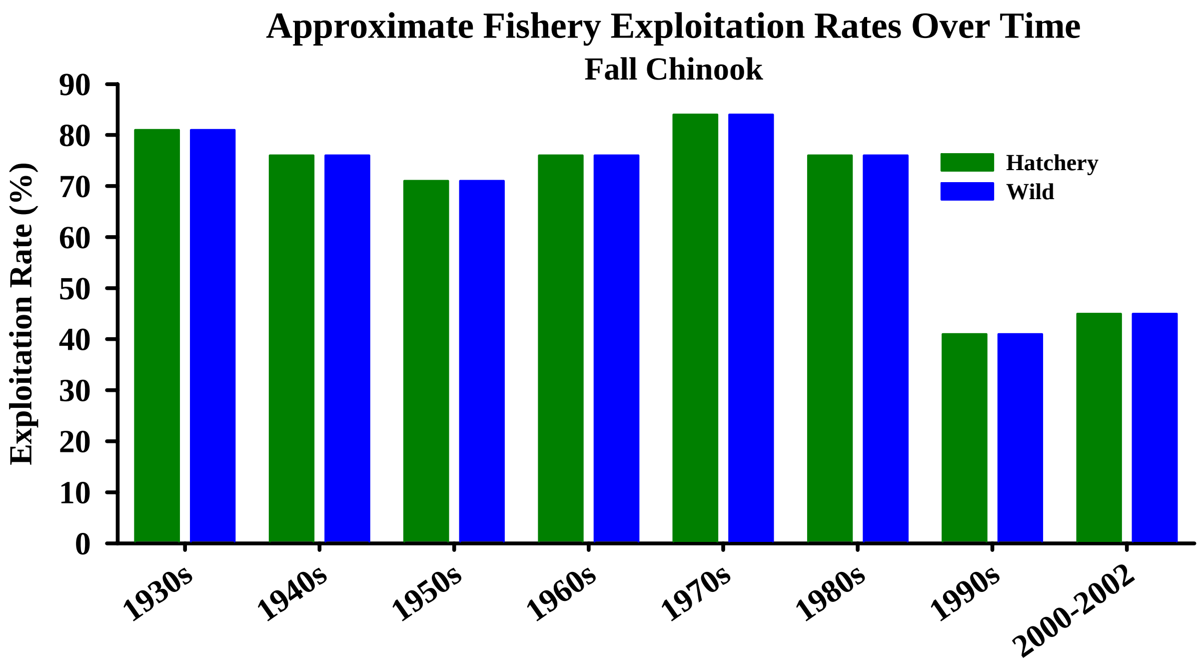

Figure 3-22. Approximate fall chinook fishery exploitation rate over time. Primarily Columbia River commercial harvest until ocean fishery expansion in 1950s. Northern migration with Canada and Alaskan interception significant in some years. Commercial harvest primarily in September. LRH component important to Washington ocean fisheries. Mainstem Columbia sport harvest increased in 1990s. Tributary sport harvest focus is September.

3.1.6.2 Current Fall Chinook Harvest Rates and Distribution

The current harvest of lower Columbia fall chinook is significantly reduced from past harvest levels. Reductions in the Columbia River harvest actually began by the 1950s, but coincided with increased ocean harvest, resulting in relatively high total harvest rates until the 1990s. The current harvest levels average about 45% for the three fall chinook stocks present in the lower Columbia.

Because of their northerly migration patterns, fall chinook are harvested in Canada and Alaska fisheries more than other salmonids. For example, the majority of fishery CWT recoveries of 1989-94 brood Cowlitz Hatchery fall chinook were distributed between Washington ocean (30%), British Columbia (21%), Alaska (15%), Cowlitz River (11%), and Columbia River (8%) sampling areas. Also, CWT recoveries of Kalama fall chinook 1992-1994 brood indicate the majority of the harvest occurred in British Columbia (36%), Alaska (38%), Washington ocean (6%), and Columbia River (14%) fisheries. Upriver bright fall chinook stocks also have a northerly migration. CWT data analysis of the 1989-1994 brood years suggests that the majority of the URB fall chinook harvest occurred in Alaska (24%), British Columbia (23%), and mainstem Columbia River (42%) fisheries. However, tule fall chinook stocks originating from the Bonneville Pool are more southerly distributed. CWT data analysis of the 1971-1972 brood years from Spring Creek Hatchery indicates that the majority of Bonneville Pool Hatchery fall chinook stock harvest occurred in British Columbia (28%) and Washington (38%) ocean commercial and recreational fisheries. Canadian interception of Columbia River fall chinook was reduced beginning in the mid-1990s because of management concerns for depressed Canadian chinook stocks. Current Canadian harvest is limited by the recent abundance-based management agreement negotiated through the PST process.

In Washington coastal and Columbia River fisheries, the harvest of fall chinook was reduced in the 1990s because of the reduced abundance of fall chinook and ESA limitations (Figure 3-22). While LRH stock fall chinook have rebounded in abundance in recent years, fall chinook harvest is limited by ESA constraints on LRH natural spawners (Coweeman index) and on Snake River Wild (SRW) fall chinook. The ESA limits total harvest (combined ocean, Columbia River, and tributary) of Coweeman natural fall chinook to 49% or less (Table 3-7). The ESA restricts southern US ocean harvest of SRW chinook (a component of the URB stock) to a 30% reduction from the 1989–1993 average harvest rate. The ESA restricts Columbia River harvest of SRW chinook to 31.29%, allocated at 23.04% for treaty Indian fisheries and 8.25% to non-Indian fisheries.[1] Ocean, Columbia River, and tributary fisheries are managed to attain a minimum of 5,700 LRW natural spawners to the North Lewis River. Although hatchery fall chinook are not mass marked, and wild harvest rates are likely similar to hatchery harvest rates, differential harvest can be achieved between fall chinook stocks depending on management strategies implemented in a given year.

Table 3-7. Example of lower Columbia fall chinook current harvest exploitation and distribution under current management.

Fishery	Tule*	LRW**	URB[§]	Comments
Alaska	3%	10%	10%	PSC abundance-based management
Canada	12%	9%	15%	PSC abundance-based management
Washington/Oregon/California ocean	15%	3%	2%	PSC, ESA, allocation constraints
Columbia River	10%	8%	20%	ESA, allocation, *US v. Oregon* constraint
Tributary	5%	10%	1%	ESA, escapement goal driven
Total	45%	40%	48%	Wild and hatchery fish rates

*Lower river tule harvest driven by 49% limit for Coweeman fall chinook

**Lower river wild harvest driven by 5,700 minimum natural escapement to North Lewis

[§]Upriver harvest driven by Snake River wild ESA constraint and *US v. Oregon* Indian /non-Indian allocation agreement

[1] *US v. Oregon* Management Agreement

3.1.6.3 Fall Chinook Harvest Management Details

PSC Fisheries

In southeast Alaska, chinook salmon are harvested in ocean commercial troll, commercial net, and recreational fisheries. Total southeast Alaska chinook catch (in numbers of fish) from 1987–2002 has ranged from 155,700 in 1996 to 373,900 in 2002 (Figure 3-23).

The spring troll fisheries are designed to increase the harvest of chinook salmon produced by Alaskan hatcheries by allowing trolling in the small nearshore areas close to the hatcheries where fish concentrate. Although there is no ceiling on the number of chinook salmon harvested in the spring fisheries, the take of PST-governed chinook salmon is limited according to the percentage of the Alaskan hatchery fish taken.

Summer and winter troll fisheries primarily harvest PST-governed chinook salmon and these fish are counted toward the Alaska fisheries allocation. Southeast Alaska commercial net fisheries target fish other than chinook salmon, but chinook are harvested incidentally in these fisheries. In the recreational fisheries of southeast Alaska, the harvest of chinook salmon can be substantial: the recreational fishery harvest in 2002 was 85,200 chinook salmon, with 27,000 from Alaska hatcheries.

Directed chinook harvest occurs in numerous fisheries through Canadian PSC-managed waters; chinook also are incidentally harvested in sockeye-directed fisheries (Figure 3-23). Canadian chinook fisheries are managed through either abundance-based management agreements (AABM) or Individual Stock Base Management (ISBM) limits (Table 3-8). Management of each fishery is directed by the abundance of the stocks of concern. Selective fishery practices are used to protect stocks, and these include gear requirements such as single barbless hooks and on-board revival tanks for resuscitating salmon for release.

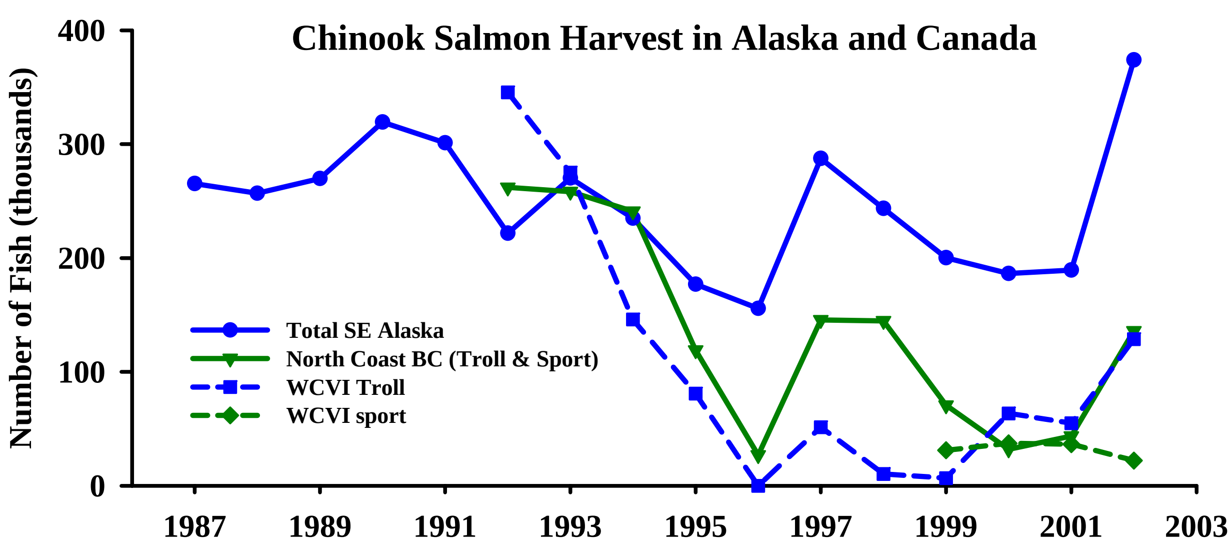

Figure 3-23. Chinook salmon harvest in PSC fisheries in Alaska and Canada, by area, 1987–2002.

Table 3-8. Management regime for Canadian chinook salmon fisheries affected by the PST.

Fishery	Management Regime (AABM or ISBM)
North Coast BC commercial troll	AABM
Queen Charlotte Islands sport	AABM
North and Central BC (including commercial net, marine sport along mainland coast, freshwater sport, Native fisheries in both marine and freshwater)	ISBM
West Coast Vancouver Island ({WCVI} including commercial troll and outside sport)	AABM
Southern BC (including commercial net fisheries in Johnstone Strait, Strait of Juan de Fuca, Strait of Georgia, and the Fraser River, commercial troll fishery in Strait of Georgia, sport fisheries in the "inside" of WCVI, marine and freshwater sport fisheries, and both marine and freshwater Native fisheries)	ISBM

PFMC Fisheries

Chinook salmon are one of two primary target species in Pacific Coast salmon fisheries in PFMC-managed waters (i.e. Canadian border to Mexico, 3-200 nautical miles offshore). PFMC management focuses on five major stocks of Columbia River Basin fall chinook: lower river hatchery tule stock (LRH) and lower river wild bright stock (LRW), Spring Creek Hatchery tule stock (SCH), all of which are part of the ESA-listed lower Columbia River ESU; upriver bright stock (URB), which includes the ESA-listed Snake River fall chinook ESU; and mid-Columbia bright stock (MCB).

The PFMC STT annually publishes stock-specific preseason run forecasts that shape fishery management planning and harvest targets for the coming year. Forecasts are prepared by WDFW, ODFW, and the Columbia River Technical Advisory Committee (TAC) and presented annually in the PFMC *Preseason Report 1.* Since 1964, age-specific estimates of escapement and in-river fishery catches have been used to establish age-specific linear regression relationships of cohort returns in previous run years. Therefore, the relationship of cohort returns from past years can be used as a predictor of the coming year return; for example, abundance of age 3 chinook in 2002 can be applied to a linear relationship of age 3 and age 4 chinook to estimate the age 4 chinook return in 2003. Total run- or stock-specific forecasts are calculated by adding the estimated age-specific returns of all age classes represented in the run.

Ocean fisheries planning for the area North of Cape Falcon is coordinated between the PFMC and PSC. These fisheries are subject to the chinook ISBM obligations contained within the 1999 Letter of Agreement. Management objectives for the chinook fisheries in the North of Falcon area are to satisfy standards for ESA-listed stocks and, to the extent possible, provide for viable ocean and in-river fisheries while protecting depressed Columbia River natural stocks and the needs of hatcheries for fall chinook brood stock. Lower Columbia River and Bonneville Pool hatchery fall chinook historically have been the major stocks contributing to ocean fishery catches in the North of Cape Falcon area, and typically drive annual fishery quota levels. Federal ESA standards and the need to limit impacts on Puget Sound and lower Columbia chinook stocks guide fishery management decisions; harvest is generally constrained by chinook harvest quotas

Table 3-9). Fisheries in the North of Cape Falcon area are divided into outside (ocean) and inside (Puget Sound and in-river) fisheries; treaty troll, non-treaty troll, and numerous recreational fisheries occur in this area.

Table 3-9. PFMC pre-season adopted chinook catch quotas (in thousands of fish) for ocean fisheries north of Cape Falcon and critical stocks driving management, 1983–2001.

Year	Critical Stocks	Treaty Troll	Non-Indian Troll	Sport
1983	Columbia River hatchery and depressed upriver stocks	—	114.0	88.0
1984	LRH and SCH	8.3	16.7	10.3
1985	SCH	10.5	47.5*	37.2
1986	SCH	12.5	51.0	37.1
1987	SCH	15.8	58.2**	44.6
1988	Columbia River upriver stocks	60.0	73.7	29.8
1989	Columbia River upriver stocks	32.0	47.5	47.5
1990	LRH	31.2	37.5	37.5
1991	LRH	33.0	40.0	40.0
1992	Columbia River tules and Snake River falls	33.0	47.0	33.0
1993	Columbia River tules and Snake River falls	33.0	35.0	25.0
1994	LRH and Snake River falls	16.4	0	0
1995	LRH and Snake River falls	12.0	0	0
1996	LRH and Snake River falls	11.0	0	0
1997	Snake River falls	15.0	11.5	5.2
1998	LRH	15.0	6.5	3.5
1999	LRW (Lewis River)	30.0	28.5	21.5
2000	Columbia River tules and LRW (Lewis River)	25.5	12.5	12.5
2001[§]	Columbia River tules	37.0	30.0	30.0

*Plus 7,430 hooking mortality for pink fishery.
** Plus 3,250 hooking mortality for pink fishery.
[§] Sharing of impacts on ESA-listed Puget Sound chinook also affected the shaping of ocean and inside fisheries.

Ocean chinook harvest in PFMC-managed waters occurs throughout the year. California ocean commercial troll fisheries occur from April to October, although most of the landings occur May–July (Figure 3-24). Oregon ocean commercial troll fisheries generally occur from May to November, although in recent years, fisheries have occurred in April and, in 2002, the fishery opened in March for the first time since 1976 (Figure 3-24). The largest harvests historically occurred in July and August; 2002 harvest was greatest in June, September, and October. Washington ocean non-Indian troll fisheries occur from May to September; most of the harvest typically occurs in May and June (Figure 3-24). Treaty Indian commercial ocean troll fisheries occur throughout the year; the majority of harvest occurs from May to August (Figure 3-24). The ex-vessel value and the price per pound of troll-caught chinook in California, Oregon, and Washington ocean fisheries has declined since the 1980s (Figure 3-25). Ex-vessel values have increased slightly in recent years compared to the 1990s, potentially because of increased harvest as a result of higher ocean productivity and salmon abundance.

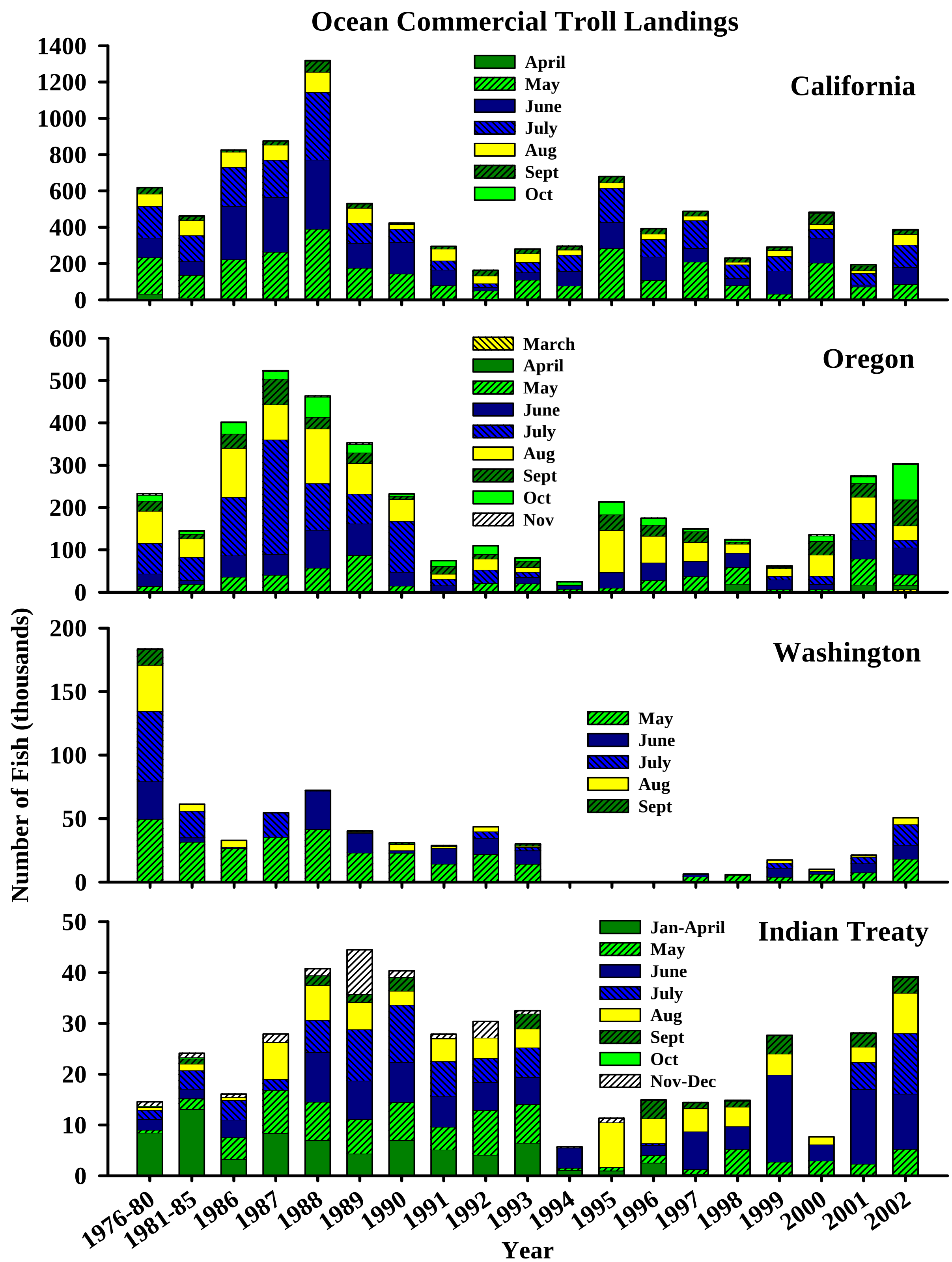

Figure 3-24. California, Oregon, Washington, and treaty Indian ocean commercial troll landings (in thousands of fish) by month, 1976–2002.

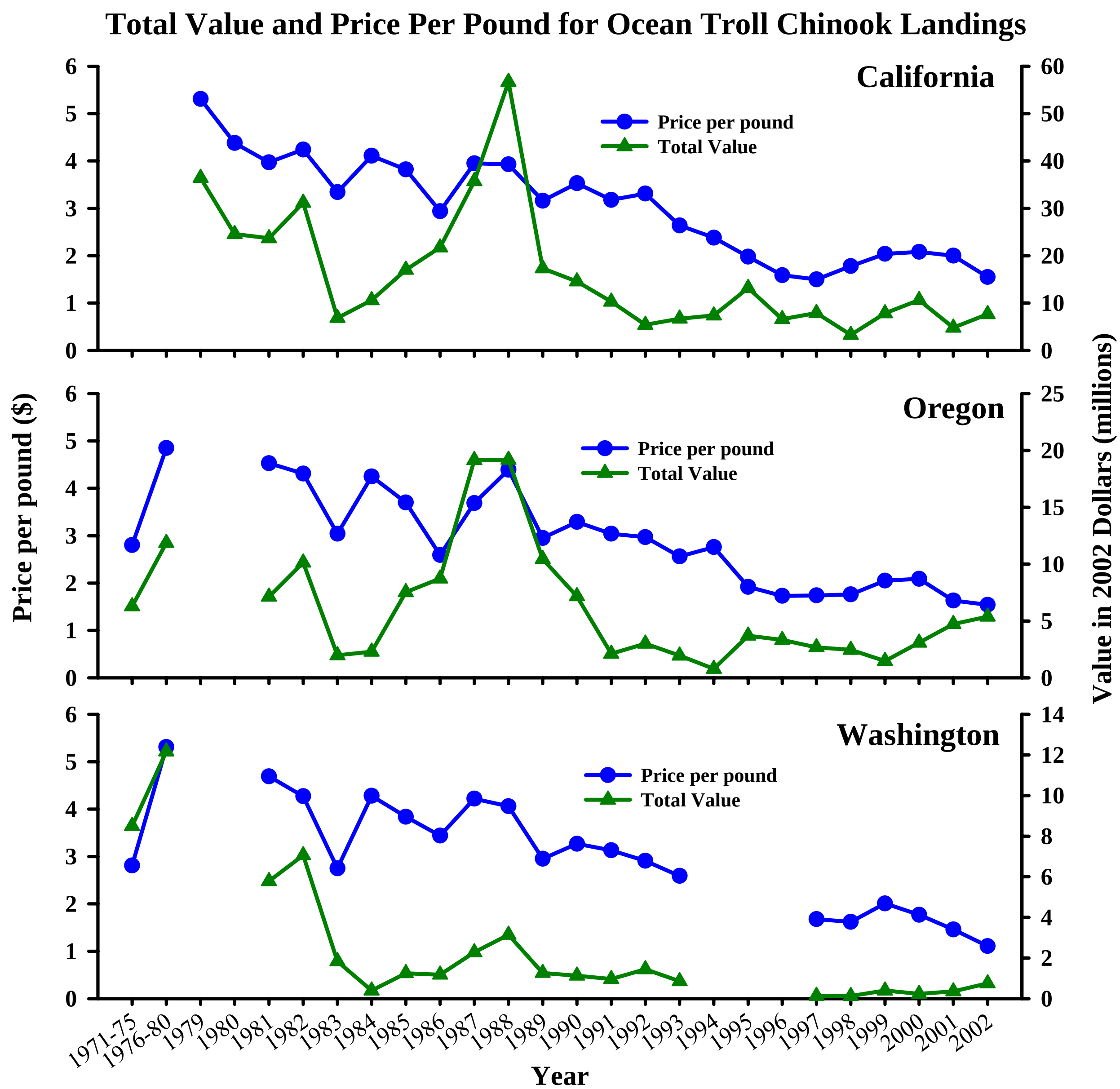

Figure 3-25. Total value and price per pound (in 2002 dollars) for ocean troll chinook landings in California, Oregon, and Washington 1979–2002.

The recreational ocean harvest of chinook in California is generally greater in the charter boat sector than the private sector, although in recent years, private boat landings have exceeded charter boat landings (Figure 3-26). In Oregon, the recreational ocean harvest of chinook is dominated by private boats although, compared to the 1990s, the charter boat catch has increased in recent years (Figure 3-26). In Oregon, the ocean recreational harvest occurs from April to November; most landings occur in July and August. In Washington, charter boat landings historically exceeded private boat landings; after years of no harvest in the mid 1990s, catch of the two boat types have increased similarly in recent years (Figure 3-26).

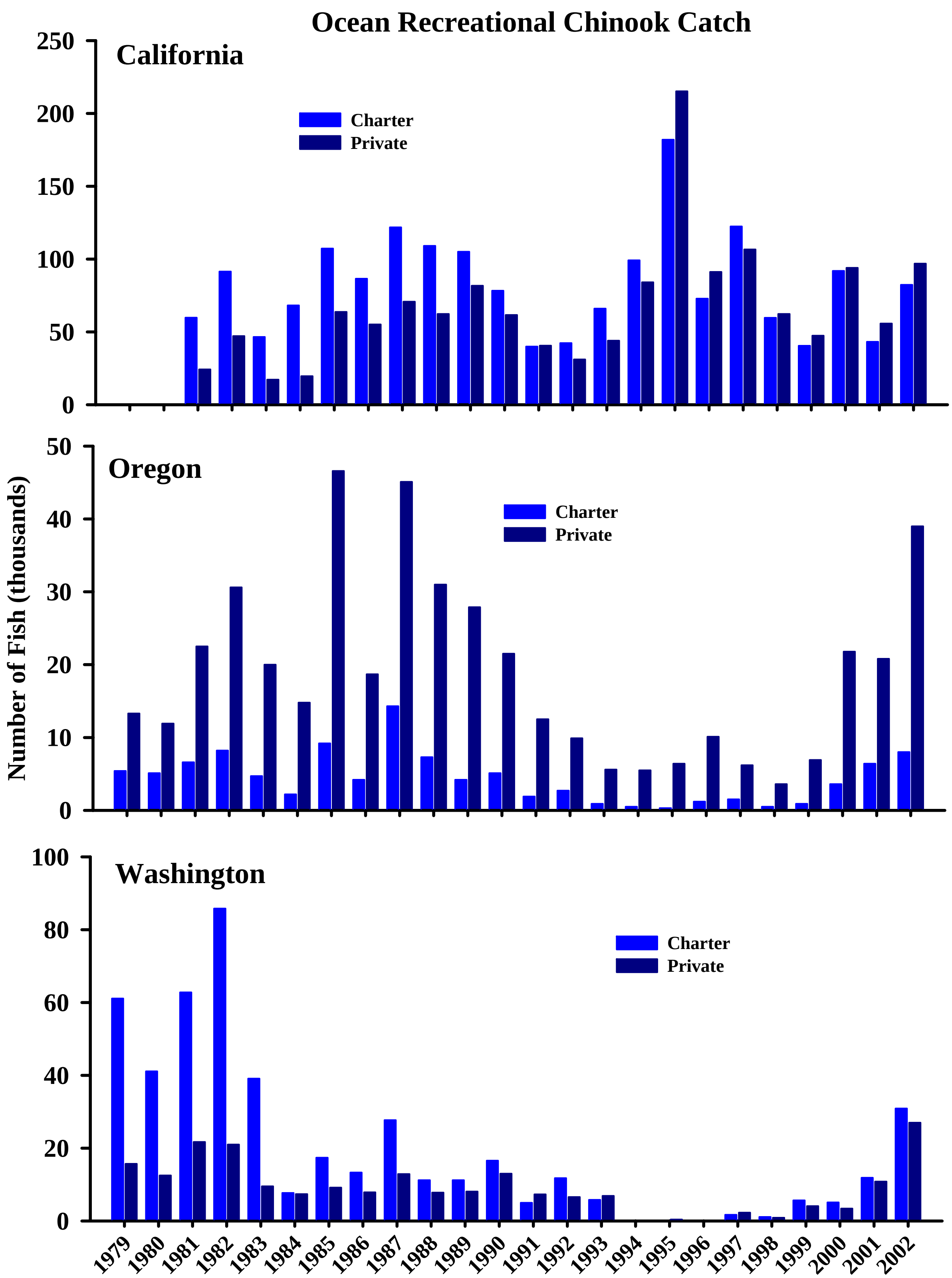

Figure 3-26. California, Oregon, and Washington ocean recreational salmon effort (in thousands of angler trips) by boat type, 1979–2002.

Ocean fisheries management in PFMC-managed waters for the 2003 seasons was constrained by:

1. endangered Sacramento River winter chinook south of Point Arena,
2. threatened California Coastal chinook south of Cape Falcon,
3. Klamath River fall chinook south of Cape Falcon,
4. threatened lower Columbia River natural tule chinook north of Cape Falcon, and
5. management goals for naturally produced coho stocks over the entire PFMC management area, including threatened Oregon and California coastal stocks.

Specific management criteria for each West Coast stock were established for the 2003 season to achieve desired escapement objectives and manage the allowable ocean harvest (Table 3-10 and Table 3-11).

Table 3-10. Management criteria and projected key stock escapements (in thousands of fish) for chinook salmon in PFMC-adopted ocean salmon fisheries, 2003*.

Key Stock/Criteria	Projected Ocean Escapement** or Other Criteria	Spawner Objective or Other Standard
Columbia Upriver Brights	253.2	57.3; minimum ocean escapement to obtain 43.5 adults over McNary Dam, with normal distribution and no mainstem harvest
Mid-Columbia Brights	93.6	16.6; minimum ocean escapement to attain 5.75 adults for Bonneville Hatchery and 2.0 for Little White Salmon Hatchery egg-take, assuming average conversion and no mainstem harvest
Lower Columbia River Hatchery Tules	116.9	23.4; minimum ocean escapement 14.3 adults for hatchery egg-take, with average conversion and no lower river mainstem or tributary harvest
Lower Columbia River Natural Tules	47%	≤49%; ESA guidance met by a total adult equivalent fishery exploitation rate on Coweeman tules (NOAA Fisheries ESA consultation standard)
Lower Columbia River Wild (threatened) [§]	23.4	5.7; MSY spawner goal for North Lewis River fall chinook (NOAA Fisheries ESA consultation standard)
Spring Creek Hatchery Tules	101.9	11.1; minimum ocean escapement to attain 7.0 adults for Spring Creek Hatchery egg-take, assuming average conversion and no mainstem harvest
Snake River Fall (threatened)	67%	≤70% of 1988–93 average age 3 and 4 AEQ exploitation rate for all ocean fisheries (NOAA Fisheries ESA consultation standard)
Klamath River Fall	35.0	≥35.0 adult spawners to natural spawning areas
Age 4 ocean harvest rate	16%	<16%; NOAA Fisheries ESA consultation standard for threatened California Coastal chinook
Federally recognized tribal fishery	50%	50% share of adult harvest; equates to 41.4 adult fish for the Yurok and Hoopa tribal fisheries
KMZ recreational fishery	14.8%	Share of adult ocean harvest (none specified for 2003)
CA/OR commercial fishery	51%/49%	Share of adult commercial ocean harvest for the States of California/Oregon (none specified for 2003)
Klamath River recreational fishery	26.1%	>15% share of nontribal adult harvest specified by California Fish and Game Commission; equates to 10.8 adult fish
Sacramento River Winter (endangered)	Yes	Duration and timing of commercial and recreational seasons south of Point Arena not to differ substantially relative to those of 2000 and 2001 (NOAA Fisheries ESA consultation standard)
Sacramento River Fall	517.0	122.0-180.0; Sacramento River fall natural and hatchery adult spawners

* Projections assume a SE Alaska TAC of 366.7 chinook per PST agreement. For Canadian chinook fisheries, assumed TACs were 112.5 for WCVI and a 1.4 in the Strait of Georgia troll fishery. All other Canadian troll and sport fisheries were assumed to have the same impact rates as in 2002

** Ocean escapement is the number of salmon escaping ocean fisheries and entering fresh water

[§] Includes minor contributions from EF Lewis and Sandy Rivers

Table 3-11. Tentatively adopted 2003 fishery management measures for PFMC fisheries to mitigate potential impacts on ESA-listed ESUs of lower Columbia River salmonids.

ESU	Stock Representation in Salmon FMP	ESA Consultation Standard	2003 Council Guidance
Lower Columbia River chinook—threatened	• Cowlitz, Kalama, Lewis spring • Lower River Hatchery fall • NF Lewis fall	• No specific requirements • Brood year adult equivalent exploitation rate on Coweeman tule fall chinook < 49% • 5,700 MSY level adult spawning escapement	• Meet hatchery escapement goals • 47% total ocean and freshwater AEQ exploitation rate • 23,400 adults to Columbia River mouth
Upper Willamette chinook— threatened	Upper Willamette River spring	No specific requirements; rare occurrence in PFMC fisheries	Troll fisheries N of Cape Falcon do not begin before 5/1
Upper Columbia River spring chinook—endangered	Upper Columbia River spring	No specific requirements; rare occurrence in Council fisheries	Troll fisheries N of Cape Falcon do not begin before 5/1
Snake River fall chinook—threatened	Snake River fall	≥30% reduction from the 1988-93 average adult (age 3 & 4) exploitation rate for all ocean fisheries	33% reduction from 1988-93 average (age 3 & 4) AEQ ocean exploitation rate
Snake River spring/summer chinook—threatened	Snake River spring/summer	No specific requirements; rare occurrence in PFMC fisheries	Troll fisheries N of Cape Falcon do not begin before 5/1

Columbia River Fisheries

Columbia River fall fishing seasons are set by the Columbia River Compact, which is charged by Congressional and statutory authority to establish Columbia River Indian and non-Indian fishing seasons in joint waters bordering Washington and Oregon. The Compact considers annual abundance forecasts (produced by state biologists and endorsed by federal and tribal biologists) for each fall chinook management stock in order to assure seasons set by the Compact are consistent with Ocean and In-River Management Agreements, treaty Indian and non-Indian allocation mandates, conservation measures of the ESA, as well as *US v. Oregon* and state established escapement goals. The Compact considers agency, tribal, and public testimony in public hearings prior to taking regulatory action.

2002 Columbia River Salmon Management Guidelines

The CRFMP expired on July 31, 1999. A Management Agreement for upper Columbia River fall chinook, steelhead, and coho has been reached by all parties for fall fisheries occurring in 2002. The following guidelines will be in place for the 2002 fall fishery management period.

- Allowable SRW fall chinook impacts in combined non-Indian and treaty Indian mainstem fisheries below the confluence of the Snake River for 2002 result in a 30% reduction from base period harvest rates. The corresponding impact rate is 31.29% of the aggregate URB run.
- The freshwater URB impact rate of 31.29% will be allocated 23.04% for treaty Indian fisheries and 8.25% for non-Indian fisheries.
- Treaty Indian fall fisheries will be managed to limit impacts on wild Group B index steelhead to no greater than 15%. All non-Indian fisheries outside the Snake River basin will be managed for an upriver wild steelhead impact rate to not exceed 2% on wild Group B index steelhead.
- Upriver fall chinook escapement goals include 7,000 adult fall chinook (4,000 females) to Spring Creek Hatchery and 43,500 adult fall chinook (natural and hatchery included) for spawning escapement above McNary Dam.
- Ocean and lower river fisheries will be managed to provide for Bonneville Dam escapement of at least 50% of the upriver coho salmon return.
- Non-Indian fisheries will be managed for an impact rate of less than 5% for Columbia River chum salmon.
- Combined ocean and freshwater fisheries will be managed to limit impacts on wild coho destined for Oregon tributaries to no more than 14% based on the 2002 Incidental Take Permit issued by the OFWC.

Columbia River fall chinook runs are divided for stock-specific management of Columbia River fisheries; the six major fall chinook stock components are LRH, LRW, URB, BPH, mid Columbia River Brights (MCB; includes hatchery production of URB stock downstream of McNary Dam), and Select Area Brights (SAB; includes bright stock of Rogue River origin released from net pens in Youngs Bay, OR). Each stock varies in annual abundance and therefore the stock mix in fisheries is different in any given year (Table 3-12). The *US v. Oregon* TAC accounts for specific stock abundances to make a pre-season projection of harvest of each stock by fishery, time, and area. The pre-season forecasts are used to establish harvest agreements between Indian and non-Indian fisheries, sport and commercial fisheries, and ocean and Columbia River fisheries. State biologists monitor actual fish runs and fishery harvest by stock (Table 3-13) to assure fisheries are adjusted in-season to meet management requirements. Several emergency Compact hearings are held during the course of each fall season to close or add fisheries in response to in-season updates.

Table 3-12. Stock accountability of fall chinook returning to Columbia River, 1980–2002.

Return Year	Total Return	URB	BPH	MCB*	LRH	LRW	SAB
1980	320,000	76,800	97,800	0	105,600	38,800	
1981	278,900	66,600	86,300	4,400	94,900	25,000	
1982	363,100	79,000	120,700	8,800	139,500	13,000	
1983	237,600	86,100	28,900	14,400	88,100	16,800	
1984	309,400	131,400	47,500	11,800	102,400	13,300	
1985	362,800	196,400	33,200	5,700	111,000	13,300	1,600
1986	494,800	281,600	16,600	17,400	154,800	24,500	2,000
1987	871,000	420,700	9,100	57,000	344,100	37,900	2,300
1988	784,700	339,900	12,000	78,000	309,900	41,700	3,200
1989	552,000	261,300	26,800	93,100	130,900	38,600	1,200
1990	312,900	153,600	18,900	59,000	60,000	20,300	1,100
1991	275,500	103,300	52,400	35,400	62,700	19,800	2,000
1992	219,000	81,000	29,500	31,100	62,600	12,500	2,300
1993	214,900	102,900	16,800	27,400	52,300	13,300	2,100
1994	254,000	132,800	18,500	33,700	53,600	12,200	3,200
1995	242,800	106,500	33,800	34,100	46,400	16,000	6,000
1996	330,800	143,200	33,100	59,700	75,500	14,600	4,700
1997	321,500	161,700	27,400	58,900	57,400	12,300	3,800
1998	255,400	142,300	20,200	36,800	45,300	7,300	3,500
1999	309,500	166,100	50,500	50,600	40,000	3,300	2,900
2000	253,300	155,700	20,500	36,900	27,000	10,200	4,900
2001	548,800	232,600	125,000	76,400	94,300	15,700	5,000
2002	733,100	276,900	160,800	108,400	156,400	24,900	5,700

***** URB stock below The Dalles Dam

Table 3-13. Stock composition of adult fall chinook landed in mainstem Columbia River fisheries, 2001.

	Stock						
	LRH	LRW	BPH	URB	MCB	Other*	Total
Non-Indian Fisheries							
Recreational	4,845	356	3,159	11,146	7,483	1,590	28,579
Early August commercial	528	112	673	394	25	161	1,893
Late Aug/Sept commercial	2,654	985	2,815	6,596	4,186	258	17,494
October commercial	53	285	88	338	2,812	9	3,585
Select area commercial	1,193	0	117	823	0	2,040	4,203
Subtotal	9,273	1,738	6,852	19,297	14,506	4,088	55,754
Treaty Indian Fisheries							
Sales to licensed buyers	0	0	33,808	18,520	7,800	110	60,238
C&S and other non-ticketed catch	0	0	18,528	16,295	8,770	0	43,593
Subtotal	0	0	52,336	34,815	16,570	110	103,831
Total	*9,273*	*1,738*	*59,188*	*54,112*	*31,076*	*4,198*	*159,585*

* includes select area brights, spring chinook, and non-Columbia chinook

The lower river run (i.e. below Bonneville Dam) is composed of LRH, LRW, and MCB stocks, as well as minor stock components of LRB and SAB fall chinook. MCB stocks are also produced in basins above Bonneville Dam, such as the Wind and Little White Salmon River basins. Columbia River salmon management guidelines for the 2002 fall fisheries were driven by the following restrictions on fall chinook:

- Allowable Snake River wild (SRW) (part of URB group) fall chinook impacts in combined non-Indian and treaty Indian mainstem fisheries below the confluence of the Snake River result in a 30% reduction from base period harvest rates (31.29% impact rate of the aggregate URB run),
- Freshwater URB impact of 31.29% will be allocated 23.04% for treaty Indian harvest and 8.25% for non-Indian fisheries,
- Above Bonneville Dam fall chinook escapement goals include 7,000 BPH adults to Spring Creek Hatchery and 43,500 URB adults past McNary Dam,
- Lower river hatchery (LRH) escapement goal of 14,700 adult chinook and Coweeman wild combined ocean and Columbia River exploitation rate of less than 49%, and
- Lewis River wild (LRW) chinook escapement goal of 5,700 adults.

Early fall seasons target fall chinook, particularly the non-Indian commercial openings in Zones 4 and 5, as well as the treaty Indian commercial harvest in Zone 6 (Figure 3-27).

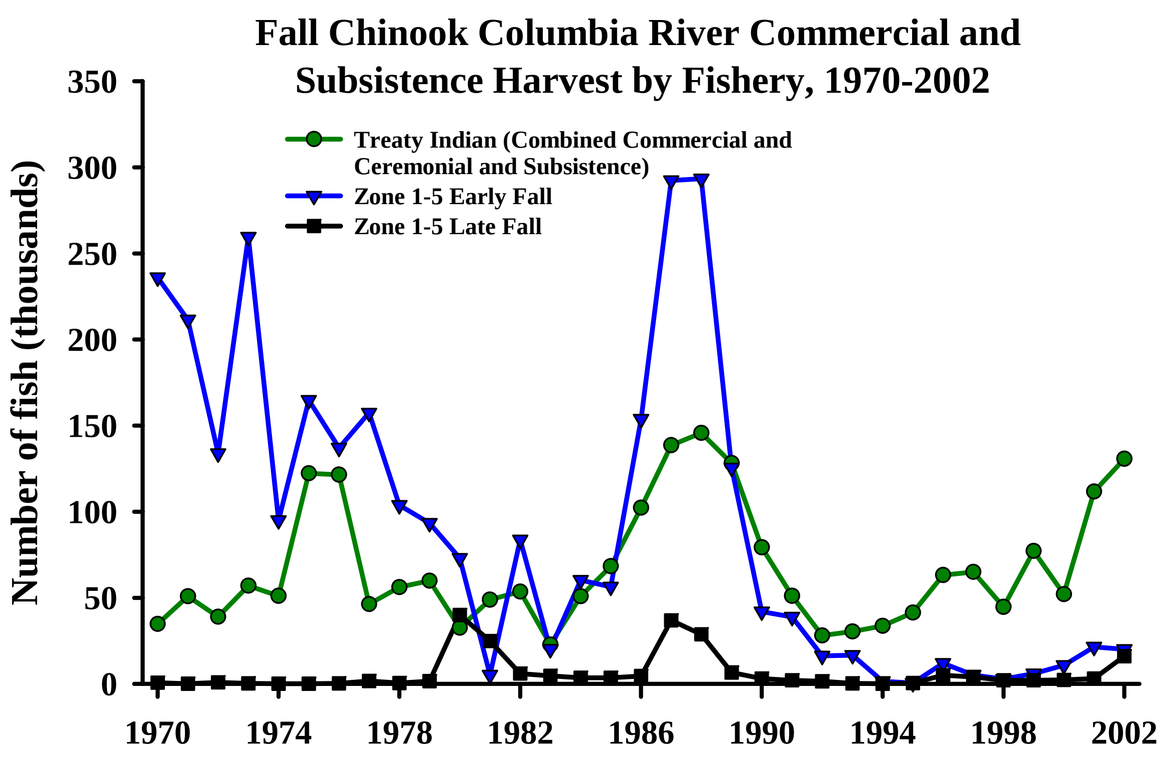

Figure 3-27. Commercial and subsistence harvest of fall chinook in the Columbia River from 1970–2001.

Since 2000, an agreement between non-Indian commercial and sport fishing interests has been established to address allocation of fall chinook between non-Indian fisheries and general season structure. These agreements have been negotiated annually during the NOF pre-season planning process. Sharing of non-Indian SRW ESA impacts is addressed as well as equitable sharing of total chinook harvest between commercial, mainstem Columbia sport, and tributary sport fisheries. The 2002 non-Indian fall chinook management agreement included the following elements:

2002 Non-Indian Columbia River Fall Fishery Chinook Allocation Agreement

- Expected total catch of fall chinook in the mainstem Columbia River downstream of the Snake River and in lower Columbia River tributaries is 85,400 of which 45,300 (53%) are expected to be harvested by the sport fishery and 40,100 (47%) by the commercial fishery.
- This agreement is limited by the non-Indian allocation of URB fall chinook impacts of 8.25% as per the 2002 U. S. *v. Oregon* Fall Management Agreement. Non-Indian catch estimates are based on pre-season abundance forecasts referenced in Model Run "2002 MR-6" .
- URB fall chinook impacts in fisheries downstream of the Snake River are allocated preseason at 4.36% to the sport fishery and 3.89% to the commercial fishery. The Columbia River Compact will use this URB impact allocation as guidance for making in-season management decisions concerning the Columbia River sport and commercial fisheries. Actual URB impacts in the fisheries may differ from pre-season estimates based on actual fishery catches, stock composition, and run-size updates. The U. S. *v. Oregon* TAC will update the URB run-size beginning in mid-September.
- The Buoy 10 sport fishery is modeled at 90% of the chinook catch estimated for a full fishery to the end of the year (with a two fish daily limit) which is expected to deliver enough chinook to continue the fishery through Labor Day. URB impacts with this fishery are projected to be 1.70%; or 39% of the total sport impacts of 4.36%.
- The mainstem sport fishery below McNary Dam is modeled at 95% of the chinook catch estimated for a full fishery to the end of the year (with a two fish daily limit), which is expected to provide enough chinook to continue the fishery through September, unless the mid-September URB run size and fishery updates indicate this fishery cannot continue past mid-September. URB impacts associated with this fishery are 2.66%; or 61% of the total sport impacts of 4.36%. For 2003 fall fishery discussions, the mainstern sport fishery will begin at 100%.
- Expectations for the commercial fishery include:
- An early August salmon fishery up to four nights during the first week of August with potential for fishing during the early part of the second week of August in Zones 2 and 3 only. During the first week of August, the open area will include Zone 1 upstream to Longview Bridge and an 8-in minimum mesh restriction. Projected catch is 16,800 salmon. Chinook[URB impacts not used in this fishery will transfer to August Zone 4-5 fishery.
- Late August Zone 4-5 fishery during the last two week of August. Fishing is expected to occur 2-3 nights per each week with breaks in between fishing days. This fishery will not occur past August 29. Mesh size is 9-in minimum. Chinook/URB impacts not used in this fishery will transfer to September fisheries. Expected catch is 8,300 chinook plus any transfers from the early August commercial fishery.
- Late fall fishery to begin the week of September 15. Fishery to occur in as much of Zones 1-5 as possible and will target coho or chinook as determined by remaining impacts and in-season run strength. The late September chinook harvest will be determined by the mid-September URB run size update and the actual URB impacts remaining that can be used by the commercial fishery.

No sturgeon retention will be allowed in the August fisheries. Directed sturgeon fishing may occur during September or October to meet commercial allocation.

Columbia River fall commercial fisheries are set by time, area, and gear type to correspond to timing differences between different fall chinook stocks and other species to focus harvest on particular stocks and species at rates consistent with management intent. The commercial fishing areas are divided into zones with landings recorded by individual landing zone. Zones 1-5 are located downstream of Bonneville Dam. Zone 6 is located between Bonneville and McNary dams and is an exclusive tribal commercial fishing area. Non-Indian sport fisheries can occur in Zones 1-6 (Figure 3-2).

Recent year commercial fisheries have been set in August primarily in Zones 4-5 to access URB, BPH, and MCB fish. The peak of the fall chinook abundance in the lower river areas (Zones 1-2) occurs in late August and early September. Commercial fisheries have not been set in the lower area during this peak time to avoid over-harvest (Figure 3-28 and Figure 3-29). Commercial fisheries in the lower zones are typically set after mid-September to access the lower river and upriver chinook stocks and coho after the peak of the chinook runs has cleared the mainstem Columbia. Treaty Indian commercial fisheries are focused in September to harvest fall chinook and summer steelhead (Figure 3-28 and Figure 3-29).

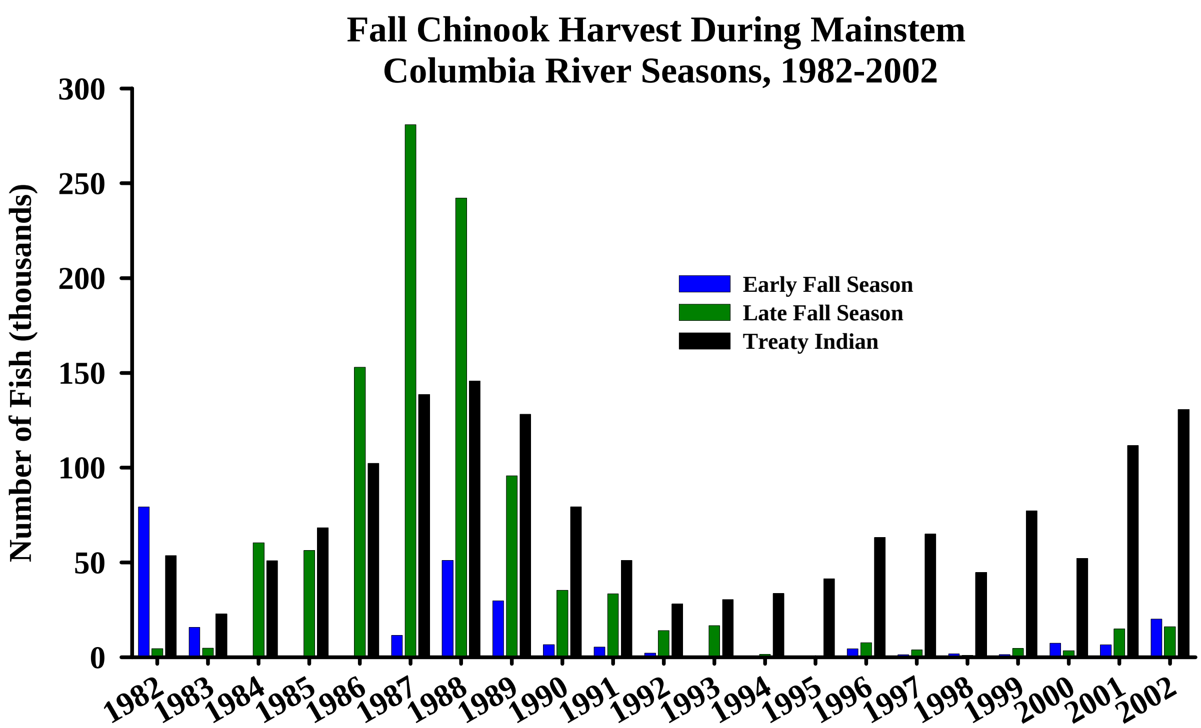

Figure 3-28. Number of adult chinook landed during early fall, late fall, and treaty Indian mainstem Columbia River seasons.

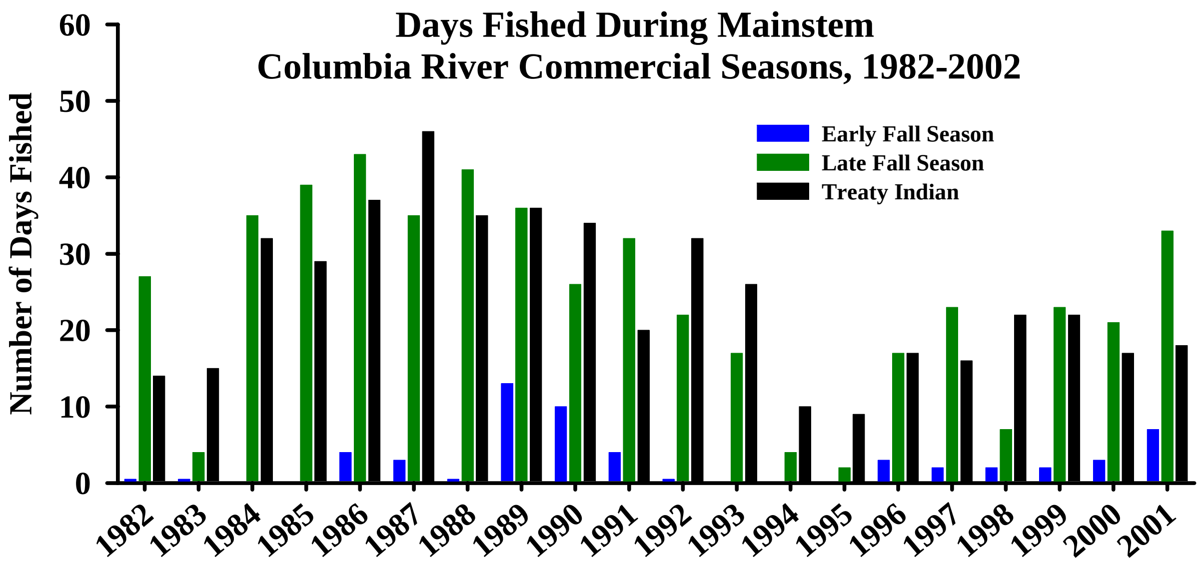

Figure 3-29. Number of days fished during early fall, late fall, and treaty Indian mainstem Columbia River commercial seasons, 1982–2001.

Columbia River sport fisheries typically open in August. The Buoy 10 (estuary) fishery is managed under a total catch guideline to assure chinook limits are not exceeded. In some years, there are emergency closures in late August and early September. The mainstem sport fishery upstream of the estuary area (upstream of Grays Point) is intended to remain open for the entire fall season, but on occasion it has closed early to avoid exceeding agreed chinook harvest levels. Chinook harvest in both of these sport fisheries can be significant (Figure 3-30).

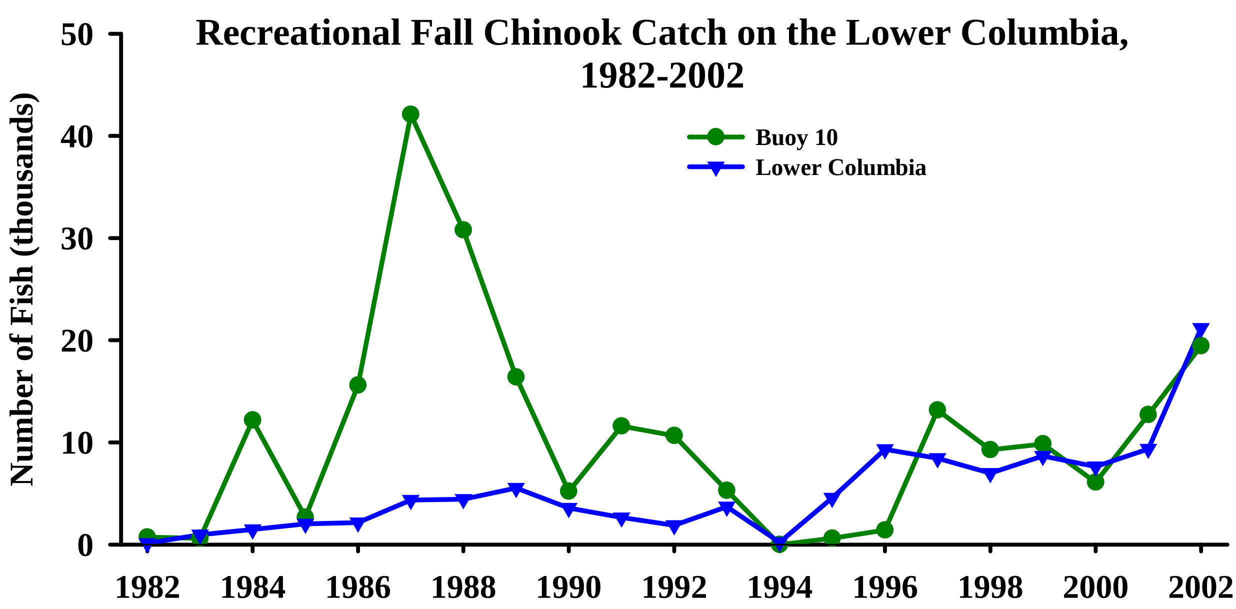

Figure 3-30. Buoy 10 fishery recreational catch and combined Oregon and Washington angler catch of chinook on the lower Columbia River, 1982–2000.

Tributary fall chinook sport fisheries for LRH tules occur principally in the Washougal, Cowlitz, Kalama, Grays, and Elochoman Rivers with most harvest occurring from late August through September. Annual harvest rates within each basin vary depending on abundance (Figure 3-31 and Figure 3-32). In large run years, harvest rates in individual tributaries can exceed 20%. In low run years, tributaries may be closed if needed to meet hatchery escapement needs. Fall chinook fishing is closed in the Coweeman and EF Lewis Rivers and in Abernathy Creek to protect natural spawning chinook.

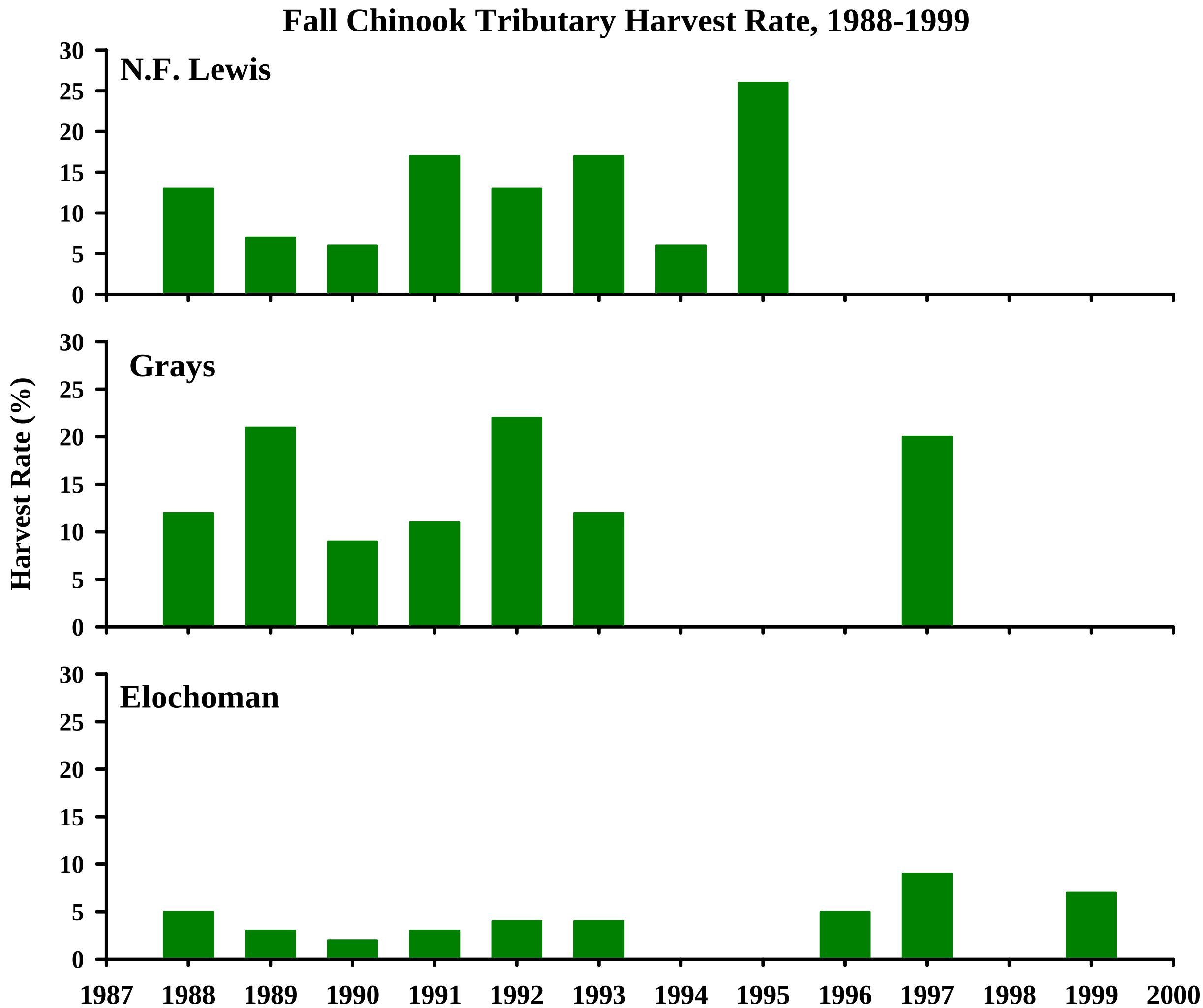

Figure 3-31. Fall chinook tributary harvest rate in the NF Lewis, Grays, and Elochoman Rivers, 1988–99. Harvest rate equals sport catch divided by run size at tributary mouth.

North Lewis River fall chinook sport fishing occurs from late August into October as Lewis River fish spawn later than tule stocks. Fishing is open when LRW fall chinook projections indicate there are sufficient returns to harvest chinook and meet the natural spawning escapement goal of 5,700 natural spawners. The fishery was open annually except for 1996–2000, when run forecasts indicated low returns. The LRW stock rebounded in 2001 and the sport fishery was reopened. The North Lewis River fall chinook sport fishery is the only lower Columbia tributary fishery which targets healthy natural produced fall chinook, as hatchery fall chinook are not produced from the Lewis River.

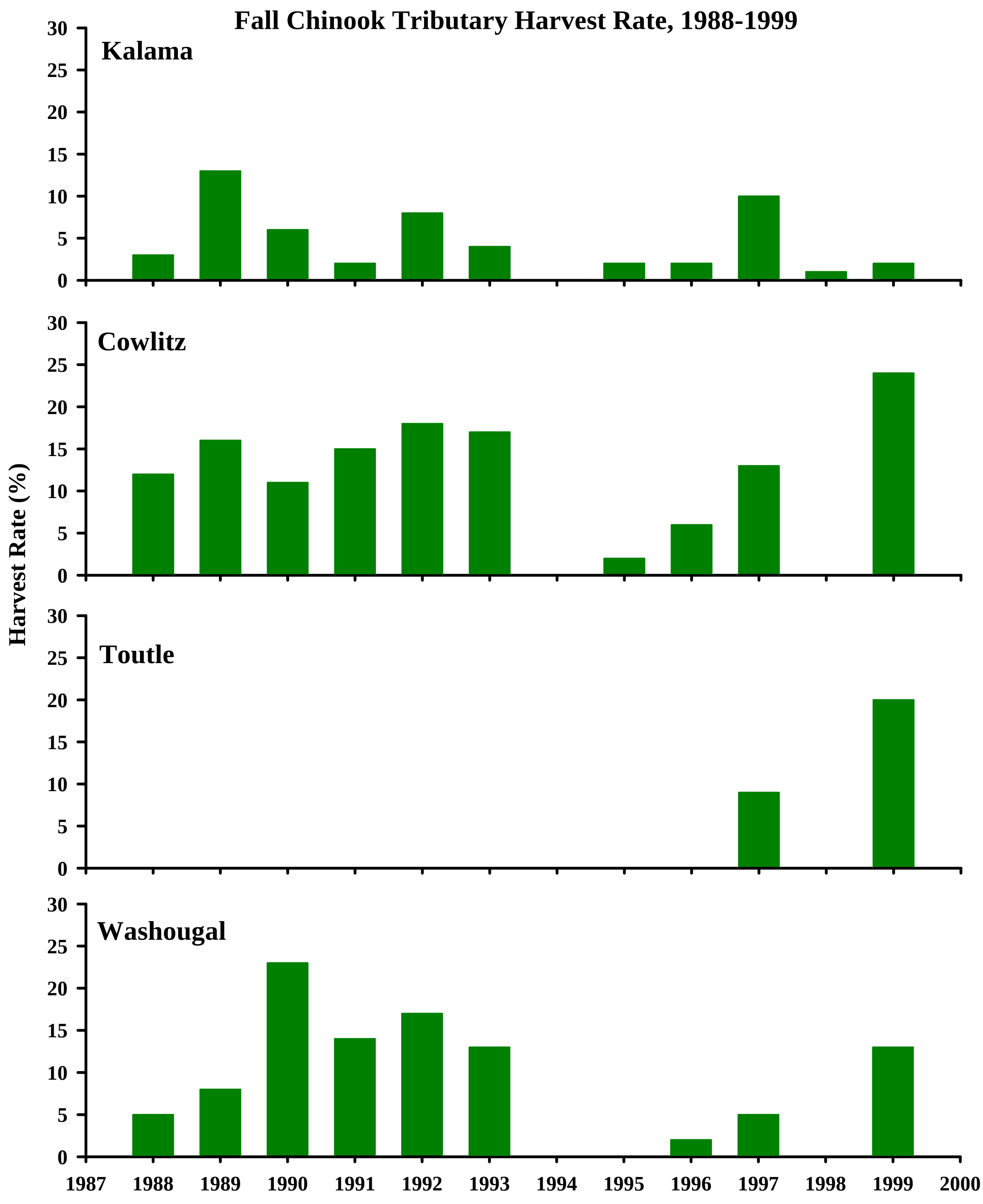

Figure 3-32. Fall chinook tributary harvest rate in the Kalama, Cowlitz, Toutle, and Washougal Rivers, 1988–99. Harvest rate equals sport catch divided by run size at tributary mouth.

3.1.7 *Coho Fishery*

Impacts to lower Columbia River coho salmon are harvested in ocean commercial, sport, and tribal fisheries; in Columbia River sport, commercial, and treaty Indian fisheries; and in tributary sport fisheries. These fisheries, and their management structure, are briefly discussed in the harvest overview section of this report. Like other salmon stocks in the Columbia River, integrating the management of coho ocean and Columbia River fisheries is essential to meeting conservation requirements for ESA-listed or critical stocks and to promote fishery opportunity on healthy hatchery and wild populations. Inside the Columbia River, early and late stock coho are managed separately; differences in the timing of fish runs enable managers to structure seasons to meet separate harvest objectives for the stocks.

3.1.7.1 Coho Harvest Over Time

Coho salmon received significant harvest pressure beginning in the late 1800s, particularly on the lower Columbia River. Peak commercial catches of wild coho in the Columbia River occurred in 1925 (Lichatowich and Mobrand 1995); since the 1960s, Columbia River commercial catch has consisted primarily of hatchery-produced coho. Commercial landings of coho salmon in Washington, Oregon, and California from 1882 to 1982 have been estimated by Shepard et al. (1985). These estimates show relatively constant landings since 1895, ranging mainly between 1.0 and 2.5 million fish, with a low of 390,000 fish (1920) and a high of 4.1 million fish (1971). Columbia River coho became an important marine, as well as freshwater, harvest species in the 1960s.

Ocean harvest of coho in the Oregon Production Index (OPI) peaked in the 1970s and early 1980s (Figure 3-33) and resulted in high coho exploitation when combined with freshwater fisheries aimed at harvesting large hatchery production (Figure 3-34). For example, ocean and Columbia River combined harvest rates of Columbia River-produced coho ranged from 70 to 90% during 1970-1983. During this time, naturally produced coho were managed like hatchery stocks and were subject to similar harvest rates. In the mid-1980s, ocean fisheries harvest was reduced to protect several Puget Sound and Washington coastal wild coho stocks. Beginning in the early 1990s, Columbia River coho commercial seasons were closed before November to reduce harvest of late Clackamas River wild coho. Coho in the Oregon Coast ESU were listed as threatened under the ESA in 1998; subsequent harvest restrictions to protect Oregon Coast coho likely also benefited naturally produced lower Columbia River coho.

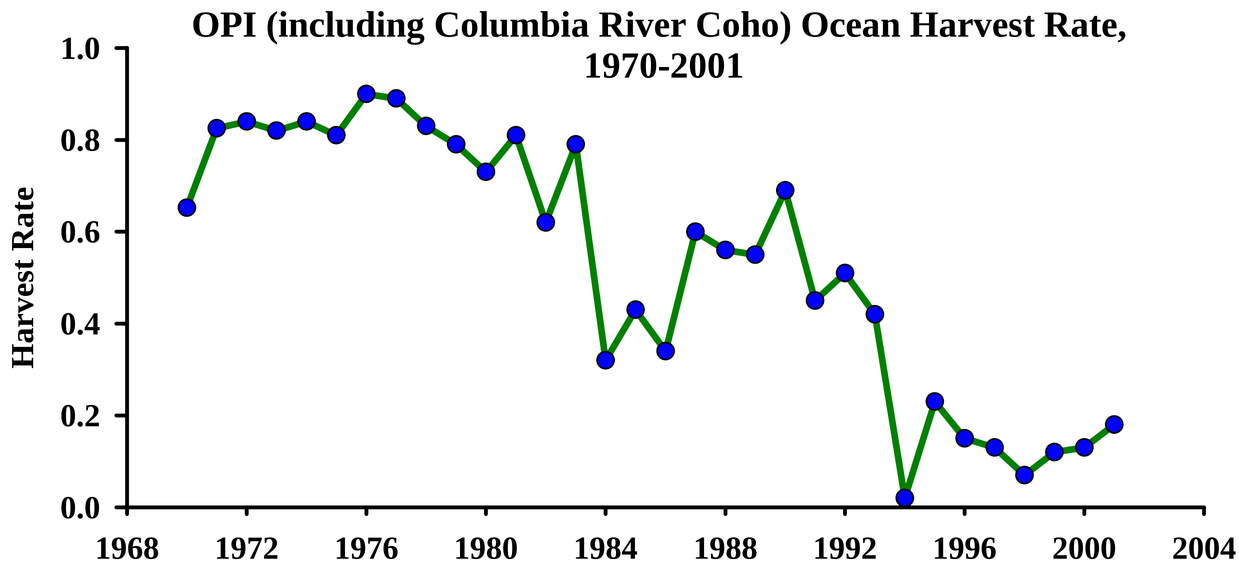

Figure 3-33. Coho ocean harvest rate based on Oregon Production Index.

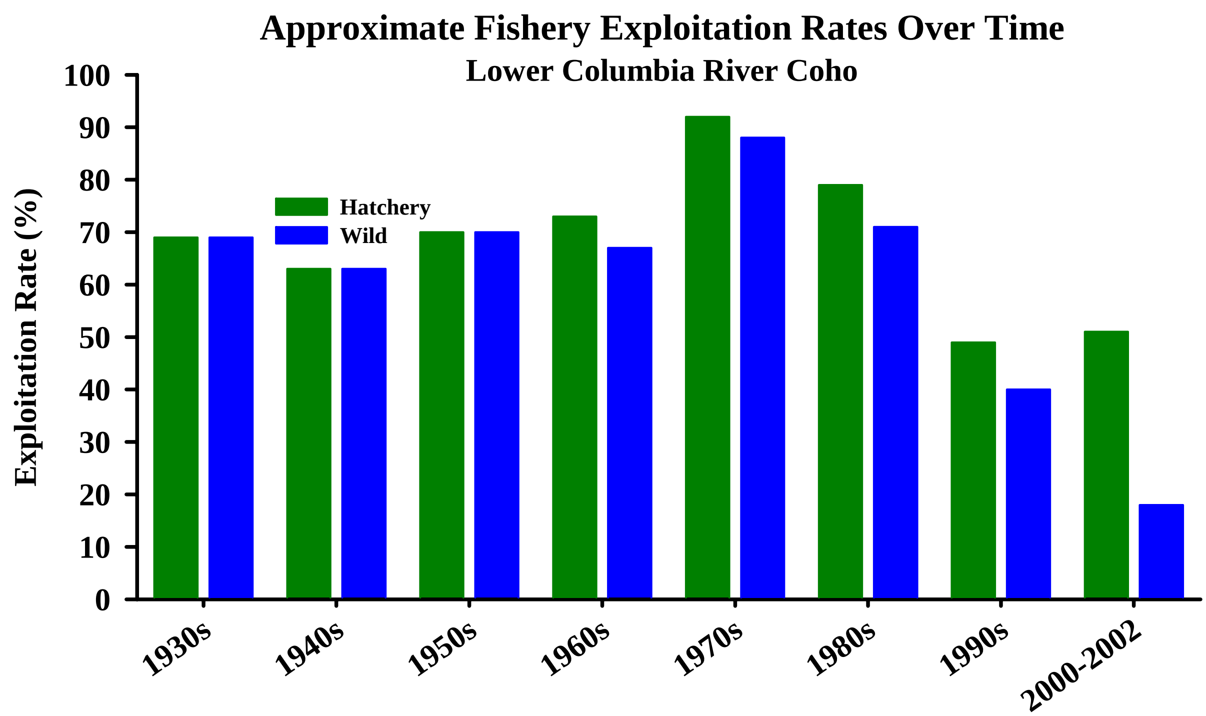

Figure 3-34. Approximate coho fishery exploitation rates over time. Primarily Columbia River harvest until 1950s. Ocean harvest peaked 1970s–80s. Coho remain an ocean sport fishery focus. Sport harvest in Lower Columbia estuary began to be significant in 1980s. Columbia commercial harvest focused on late September–October. Differential harvest of wild fish commenced in 1960s when late fall fisheries were reduced. Selective harvest in ocean and Columbia began in 1998 and provided greater differences in wild and hatchery harvest rates.

Beginning with the 1995 brood, most Columbia River hatcheries mass marked hatchery-released fish with an adipose fin clip. Since marked fish began returning as adults in 1998, fisheries managers have been able to prosecute selective sport fisheries for marked hatchery coho where all unmarked fish were required to be released. In addition, because there are run timing differences between some hatchery and wild stocks, Columbia River commercial fisheries have employed select area and time strategies to target hatchery fish to reduce impacts on wild coho. As a result of these selective management strategies employed during 1998–2002, combined fisheries harvest of ESA-listed coho was less than 15% annually, while harvest of Columbia River hatchery coho was maintained near 50 percent.

Recent harvest management practices have resulted in greater commercial harvest of late hatchery coho compared to early coho (Figure 3-35). Peak migration time for early coho in the Columbia River is September; harvest of early coho is currently restricted because of harvest constraints on fall chinook and Sandy River wild coho which also migrate during September. Columbia River commercial coho harvest effort is concentrated in October during the peak migration of late hatchery coho; there are no concurrent harvest restrictions for other salmonids during this period.

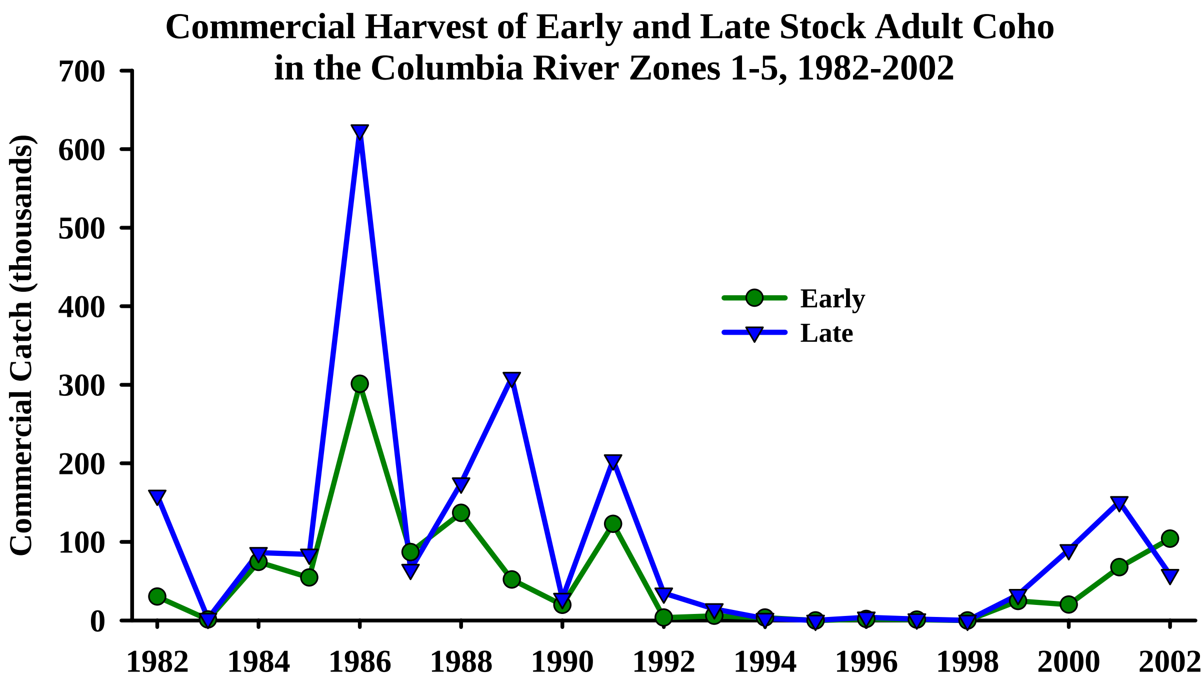

Figure 3-35. Columbia River Zone 1-5 commercial harvest of early and late stock coho.

3.1.7.2 Current Coho Harvest Distribution

Lower Columbia wild coho returning to Oregon tributaries were placed on the Oregon State Endangered Species List in 1999. Impacts to Oregon state listed coho have been managed under an abundance-based management plan similar to OCN coho.

CWT data analysis of hatchery coho from the mid-to late 1990s brood years consistently show greater harvest percent age of late coho with less percent age accounted for in the escapement compared to early coho. For example, CWT analysis of Fallert Creek (lower Kalama) early coho from the 1995–1997 brood years indicated that 30% were captured in a fishery and 70% were accounted for in escapement. However, 76% of Kalama Falls late coho from the 1995–1997 brood years were captured in a fishery and 24% were accounted for in escapement. In the Cowlitz basin, 34% of Toutle Hatchery early coho from the 1995–1997 brood years were captured in a fishery while 66% were accounted for in escapement. Meanwhile, 64% of Cowlitz Hatchery late coho from the 1994 and 1997 brood years were captured in a fishery while 36% were accounted for in escapement.

CWT data also provide some insight into the general distribution of fish. CWT hatchery coho from the 1995–1997 brood years, regardless of whether they were early or late coho, were primarily (50-60%) recovered in the Columbia River sampling area, 20-40% were recovered in the Washington ocean sampling area, and 10-20% of coho CWT recoveries were reported in the Oregon ocean sampling area. The one notable exception to this pattern is early coho from the Lewis River Hatchery; 58% CWT were reported in the Washington ocean, 21% in the Columbia River, and 21% in the Oregon ocean. In general, lower Columbia River hatchery and wild coho are harvested in West Coast ocean or Columbia River sport or commercial fisheries (Table 3-14).

Columbia River coho do not migrate as far north as Columbia River chinook; consequently, few Columbia River coho are harvested in Alaska or Canadian fisheries. Commercial ocean troll fisheries typically focus on chinook, but Indian and non-Indian ocean troll coho harvest can be significant in years of large hatchery abundance. Selective fisheries for adipose fin-marked hatchery coho have been implemented in most PFMC area ocean fisheries since 1998.

Table 3-14. Example of lower Columbia coho harvest exploitation and distribution under current management (combined early and late stock).

Fishery	H*	W*	Comment
Alaska	0%	0%	Do not typically migrate far north
Canada	<1%	<1%	Constrained by PSC and Thompson coho management
WA/OR/CA/Ocean	30%	9%	Selective sport and troll fisheries
Columbia River	15%	8%	Sport selective, commercial time and area restricted
WA Tributaries	6%	1%	Sport selective
Total Exploitation	51%	18%	Late stock hatchery harvest rate higher than early stock hatchery harvest rate. Wild stock that enter the Columbia November and later have a lower harvest rate.

* H=Hatchery, W=Wild

The Oregon production index (OPI) area coho stocks include all Washington, Oregon, and Northern California natural and hatchery stocks from streams south of Leadbetter Point, Washington. Historically, OPI stocks contributed primarily to Oregon and northern California ocean fisheries and, to a lesser extent, ocean fisheries off Washington and British Columbia. In recent years, more of the coho harvest has shifted to southern Washington coastal fisheries as a result of management actions aimed at reducing impacts to Oregon coastal natural coho stocks. The largest naturally produced component of the OPI area coho stock is Oregon coast natural (OCN) coho, which is managed as an aggregate stock with four identified components from Oregon systems south of the Columbia River. There are three threatened ESUs within the naturally produced OPI coho stocks; central California coast (CCC) coho (1996), southern Oregon/northern California (SONC) coho (1997), and Oregon coast (OC) coho (1998). OPI area coho harvest is driven by harvest restrictions on these listed stocks. No directed coho fisheries are allowed in any commercial and recreational fisheries off the California coast to protect threatened CCC coho. Marine fishery impacts on threatened CCC and SONC coho must be no more than 13% based on projected impacts on Rogue/Klamath hatchery coho. Combined marine and freshwater impacts on OCN coho should not exceed levels in the abundance-based fishery management plan (15% in recent years).

Ocean commercial troll and recreational fisheries south of Cape Falcon (near Cannon Beach) have been closed to unmarked coho retention since 1993. Selective commercial troll fisheries for marked coho have occurred since 2000 in areas from Cape Falcon to the Queets River in Washington; all directed coho recreational fisheries in the OPI area have been selective for hatchery coho since 1998. Terminal recreational harvest in Oregon coastal systems is limited to areas where surplus hatchery returns are expected. Improved hatchery coho populations in the OPI area have expanded hatchery coho ocean commercial and recreational harvest opportunities in recent years (Figure 3-36).

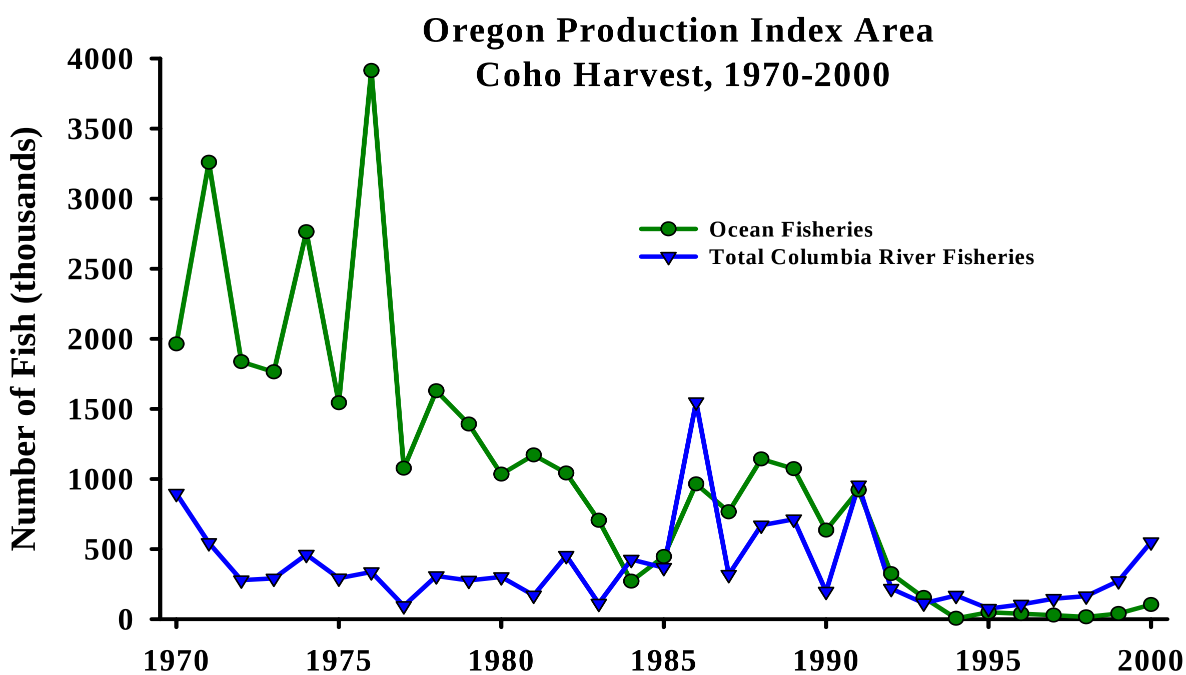

Figure 3-36. Harvest of Oregon Production Index Coho in the ocean and Columbia River, 1970–2000.

In the Columbia River, numerous commercial coho fisheries still exist, including non-Indian commercial harvest in the lower river as well as treaty Indian commercial harvest in Zone 6 above Bonneville Dam (Figure 3-37). Columbia River commercial coho fisheries are limited by chinook constraints in the early fall season, which results in limited early stock coho harvest. Most commercial coho harvest is focused in late September and October when late stock hatchery coho abundance is highest (Figure 3-35). Late fall seasons, primarily in Zone 3, target coho in the lower river below the mouth of the Lewis River. Columbia River commercial fisheries are closed before November to avoid harvest of late wild Clackamas coho, chum, and winter steelhead. Columbia River commercial fisheries retain all coho, but are managed by time and area to reduce impacts to wild fish. Columbia River commercial harvest of coho was low during the 1990s but has increased in recent years because of improved hatchery coho populations; coho harvest in treaty Indian fisheries in the Columbia River has generally been low (Figure 3-37).

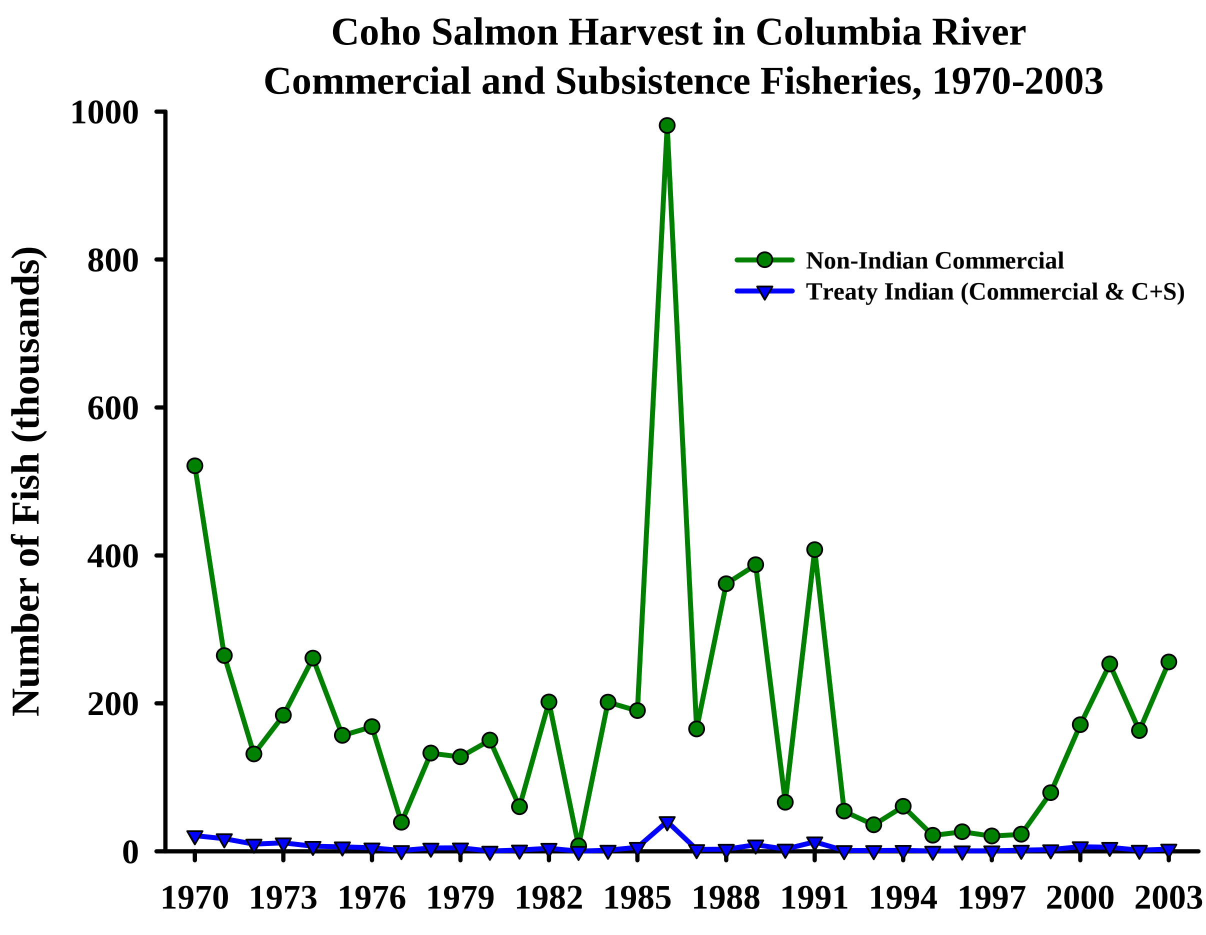

Figure 3-37. Commercial and subsistence harvest of coho salmon in the Columbia River from 1970–2001.

Columbia River hatchery coho are very important to the Lower Columbia estuary (Buoy 10), mainstem, and tributary sport fisheries (Figure 3-38). Selective fisheries for adipose-marked hatchery coho have been implemented in Columbia River and tributary sport fisheries since 1998, except in fisheries above Bonneville Dam.

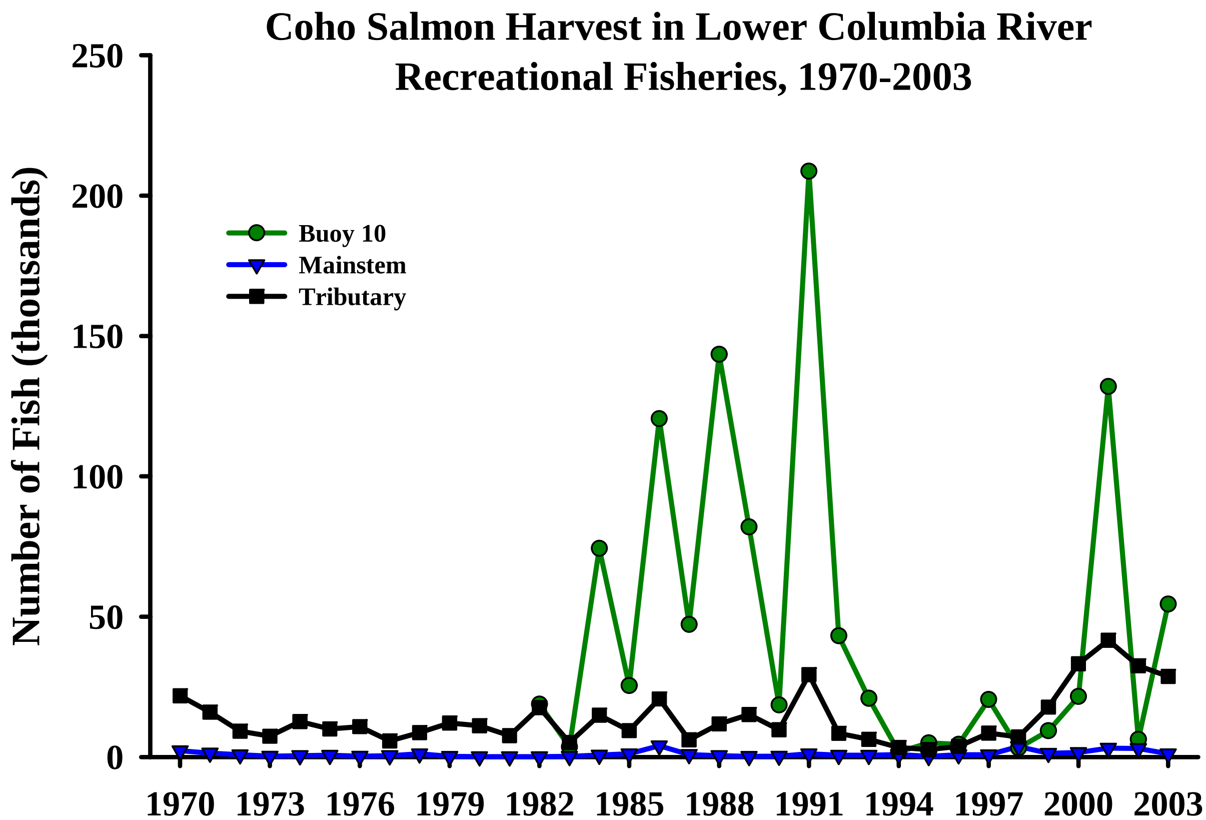

Figure 3-38. Recreational harvest of coho salmon in the Lower Columbia River for 1970–2001.

A substantial Columbia River estuary sport fishery exists between Buoy 10 and the Astoria-Megler Bridge; harvest is primarily early run coho, however, harvest of late coho can also be substantial (Figure 3-39). Angler trips to the Buoy 10 fishery in years of high hatchery coho abundance can exceed 150,000 during August and September.

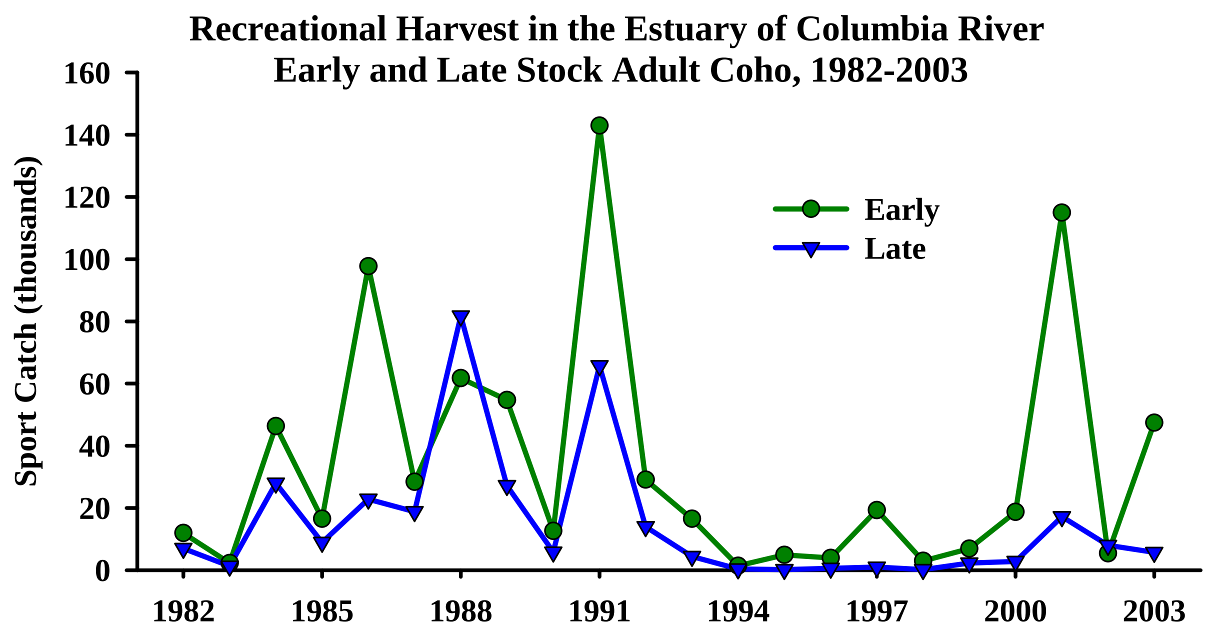

Figure 3-39. Recreational harvest of Columbia River early/late stock adult coho in Columbia River estuary.

Tributary sport fisheries for coho also occur in many basins throughout the lower Columbia (Figure 3-40). Data from the late 1980s indicate that average annual harvest was over 1,000 coho in some tributaries (e.g. 1,183 in the Elochoman [1981–88], 1,494 in the Cowlitz [1986–90], 1,272 in the Kalama [1979–86], and 3,500 in the NF Lewis [1980–98]).

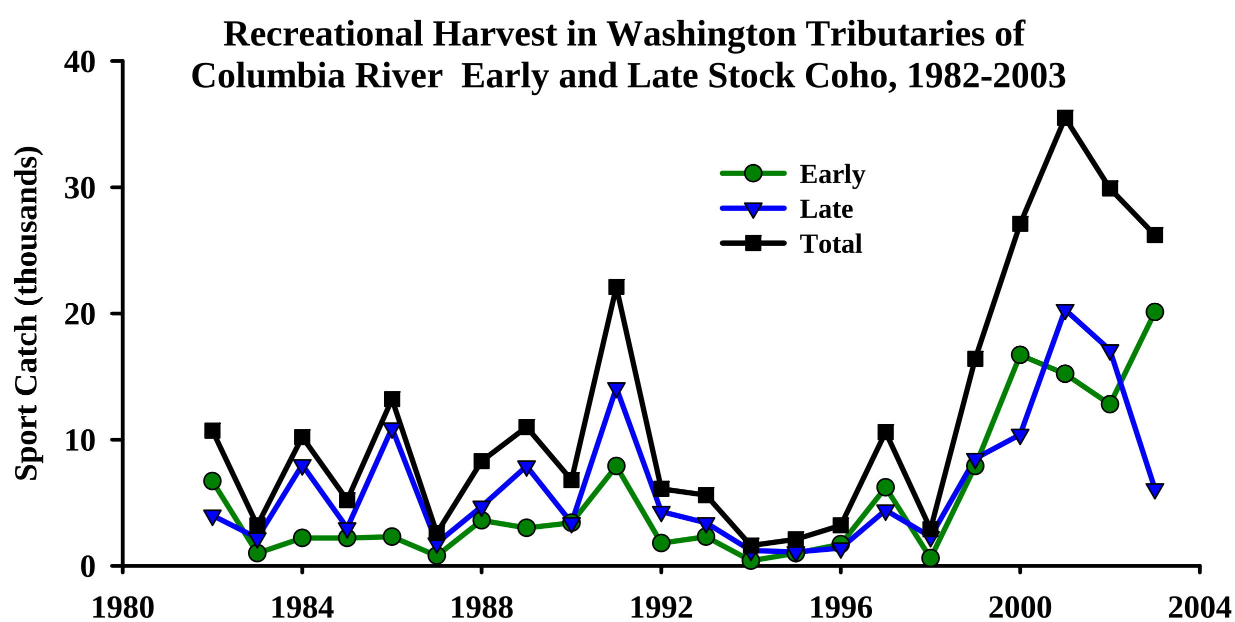

Figure 3-40. Recreational harvest of Columbia River early/late stock coho in Washington tributaries of Columbia River.

3.1.7.3 Coho Harvest Management Details

PSC Fisheries

Coho salmon management and harvest in Alaska and Canada are governed by the 1999 Letter of Agreement (LOA) negotiated as part of the PST. The LOA specifies provisions for in-season conservation and information sharing for northern boundary coho salmon. The LOA specifies catch-per-unit-of-effort (CPUE) levels in Southeast Alaska commercial troll fisheries that trigger conservation measures. The LOA also contains total commercial harvest levels in July that trigger early region-wide troll closures. Targeted coho fisheries in Canada are currently limited to southern British Columbia; the 2002 management objective for all Canadian fisheries was to limit the total exploitation rate on Thompson River coho (Canadian population) to 3% (Table 3-14).

PFMC Ocean Fisheries

Coho and chinook are the two primary salmonid species harvested in Pacific Coast ocean fisheries occurring in PFMC managed waters, extending from the Canadian border to Mexico, and 3-200 nautical miles offshore. The PFMC STT annually publishes stock-specific preseason run forecasts provided by state and tribal biologists. These forecasts shape fishery management planning and harvest targets for the coming year; forecasts are presented annually in the PFMC Preseason Report 1 in February. The majority of coho harvested in US ocean fisheries originate from rivers within the Oregon Production Index (OPI) area; stocks include hatchery and natural production from the Columbia River, Oregon coast, and northern California. The individual stock components originating in the OPI area in which abundance is estimated annually include: 1) public hatchery (OPIH), 2) Oregon coastal natural river (OCNR), 3) Oregon coastal natural lake (OCNL), 4) private hatchery (PRIH), and hatchery smolt production from the Oregon coastal Salmon Trout Enhancement Program (STEP).

The OPI area public hatchery stock is composed primarily of production from Columbia River facilities and net pens, with lesser contribution from facilities in Oregon coastal rivers and the Klamath River basin. OPIH forecasts are generated using multiple linear regression methods and a relationship established between coho jacks and the subsequent year's returning adults for the major stock components (i.e. Columbia River, Oregon coastal, Klamath River). The OPIH stock predictor is partitioned into Columbia River early and late stocks, and coastal stocks north and south of Cape Blanco, Oregon.

Integrated management of ocean and Columbia River coho fisheries is essential to the conservation and recovery of federal and state ESA-listed coho stocks. Therefore, fishery planning and management actions by PSC, PFMC, and the Columbia River Compact must be consistent and complementary. Federal ESA-listed coho stocks driving management and harvest constraints include the Oregon coast (OCN) ESU, southern Oregon/ northern California coasts (SONC) ESU, and central California coast (CCC) ESU; Oregon state-listed lower Columbia River wild coho (LCN, a federal candidate species) limits coho fisheries harvest. OCN and LCN coho are assumed to have similar temporal and spatial distributions in ocean fisheries. Harvest limits on OCN coho therefore benefit LCN coho.

In 1997, PFMC approved an amendment to the Fishery Management Plan that changed the basis for coho fisheries management from spawner escapement objectives to exploitation rate limits. The maximum allowable exploitation rates for OCN vary in response to changes in observed brood year-specific parental spawner abundance and marine survival conditions (Table

3-15). Similar exploitation rate matrices were developed for ocean fisheries mortality of LCN coho (Table 3-16). Because the exploitation rate matrices incorporate the same marine survival index, and OCN and LCN coho likely experience the same ocean conditions, managers must be mindful of situations where improved parental spawner abundance of OCN coho allows for ocean exploitation levels that make it impossible to achieve the fishery exploitation rates for LCN.

Table 3-15. Harvest management matrix identifying allowable fishery impacts and ranges of resulting recruitment based on parental spawner abundance and marine survival (OCN work group revisions to original PFMC matrix).

	Marine Survival Index (based on return of jacks per hatchery smolt)			
Parental Spawner Status*	**Extremely Low (<0.0008)**	**Low (0.0008-0.0014)**	**Medium (>0.0014-0.0040)**	**High (>0.0040)**
High	E	J	O	T
>75% of full seeding	≤8%	≤15%	≤30%	≤45%
Medium	D	I	N	S
>50% to ≤75% of full seeding	≤8%	≤15%	≤20%	≤38%
Low	C	H	M	R
>19% to ≤50% of full seeding	≤8%	≤15%	≤15%	≤25%
Very Low	B	G	L	Q
>4 fish/mile to ≤19% of full seeding	≤8%	≤11%	≤11%	≤11%
Critical**	A	F	K	P
≤4 fish/mile	0-8%	0-8%	0-8%	0-8%

Sub-aggregate and Basin-specific Spawner Criteria Data

Sub-aggregate	Miles of Available Spawning Habitat	100% of Full Seeding	Critical: 4 fish/mile	Critical: 12% of full seeding	Spawner Status Intervals: 19% of full seeding	Spawner Status Intervals: 50% of full seeding	Spawner Status Intervals: 75% of full seeding
Northern	899	21,700	3,596	NA	4,123	10,850	16,275
North-Central	1,163	55,000	4,652	NA	10,450	27,500	41,250
South-Central	1,685	50,000	6,740	NA	9,500	25,000	37,500
Southern	450	5,400	NA	648	1,026	2,700	4,050
Total	*4,197*	*132,000*		*15,636*	*25,099*	*66,050*	*99,075*

* Parental spawner abundance status for the OCN aggregate assumes the status of the weakest sub-aggregate.

** Critical parental status is defined as ≤4 fish per mi for the Northern, North-Central, and South-Central sub-aggregates; because of high quality spawning habitat in the Rogue River basin, critical status for the Rogue River (Southern sub-aggregate) is defined as 12% of full seeding of high quality habitat.

Table 3-16. Harvest management matrix for Lower Columbia Natural (LCN) coho with maximum allowable ocean fishery mortality rates.

Parental Escapement*	Marine Survival Index (based on return of jacks per hatchery smolt) Critical (<0.0008)	Low (<0.0015)	Medium (<0.0040)	High (>0.0040)
High >75% of full seeding	<8%	<15%	<30%	<45%
Medium >50% to ≤75% of full seeding	<8%	<15%	<20%	<38%
Low >20% to ≤50% of full seeding	<8%	<15%	<15%	<25%
Very Low >10% to ≤20% of full seeding	<8%	<11%	<11%	<11%
Critical ≤10% of full seeding	0-8%	0-8%	0-8%	0-8%

* Full Seeding: Clackamas River = 3,800; Sandy River = 1,340.

Fisheries off the Oregon and Washington coasts are developed through the PFMC and NOF processes and are subject to agreements of the PST. The NOF process integrates ocean and river management objectives, constraints, and agreements to formulate a coordinated management plan. For example, coho fisheries in 2002 were structured to address standards for ESA-listed stocks (especially OCN coho) and PST obligations regarding Thompson River coho (BC stock). US fisheries, including those in Puget Sound, were limited to a total exploitation rate under 10% on Thompson River coho. Low expected abundance levels of lower Columbia River hatchery coho reduced coho quotas off the Washington and Oregon coasts compared to 2001.

In establishing ocean salmon fisheries that impact OPI area coho stocks, PFMC is guided by the NMFS 1999 Supplemental Biological Opinion and Incidental Take Statement for the three ESA-listed coho stocks in the OPI area. To protect threatened CCC coho, no directed coho fisheries or retention of coho is allowed in commercial and recreational fisheries off the California coast. Marine fishery impacts on threatened CCC and SONC coho must be no more than 13% based on projected impacts on Rogue/Klamath hatchery coho. Marine and freshwater impacts on OCN coho should not exceed levels in the abundance-based fishery management plan (15% in recent years).

The PFMC management process includes evaluating proposed fishing seasons and quotas by assessing their ability to meet management criteria for key coho stocks present in West Coast fisheries. Table 3-17 displays the management criteria and projected results of ocean salmon seasons adopted by the PFMC for 2003.

A recent important management tool in the PFMC fishery management process is the use of selective fisheries for hatchery-marked adipose fin-clipped fish. For planning purposes, the STT estimates the rate of marked fish expected to be caught in particular fisheries to anticipate potential effects on wild fish. The 2003 expected mark rates for selective coho ocean fisheries are presented in Table 3-18.

Table 3-17. Management criteria and projected key stock escapements (in thousands of fish) for coho salmon in PFMC-adopted ocean salmon fisheries, 2003.

Key Stock/Criteria	Projected Ocean Escapement* or Other Criteria (Council Area Fisheries)	Spawner Objective or Other Standard
Columbia River		
Upper Columbia	52%	50%; minimum % of the run to Bonneville Dam.
Columbia River Hatchery Early	246.4	38.7; minimum ocean escapement to attain hatchery egg-take goal of 19.6 early adult coho, assuming average conversion and no mainstem or tributary harvest
Columbia River Hatchery Late	145.9	19.4; minimum ocean escapement to attain hatchery egg-take goal of 15.2 late adult coho, with average conversion and no lower river mainstem or tributary harvest
Coastal Natural		
Quillayute Fall	21.2	6.3-15.8; MSY adult spawner range (not annual target); annual management objectives may be different and are subject to agreement between WDFW and the treaty tribes
Hoh	10.4	2.0-5.0; MSY adult spawner range (not annual target); annual management objectives may be different and are subject to agreement between WDFW and the treaty tribes
Queets Wild	19.6	5.8-14.5; MSY adult spawner range (not annual target); annual management objectives may be different and are subject to agreement between WDFW and the treaty tribes
Queets Supplemental	1.1	
Grays Harbor	52.3	35.4; MSP level of adult spawners; annual management objectives may be different and are subject to agreement between WDFW and the treaty tribes
Oregon Coastal Natural (threatened)	14.4%	≤15%; marine and freshwater fishery exploitation rate
Northern California (threatened)	9.6%	≤13%; marine fishery exploitation rate for R/K hatchery coho (NOAA Fisheries ESA consultation standard)
Puget Sound		
Skagit	37% (5.4%) 97.9	≤60%; 2003 total exploitation rate ceiling based on comanager comprehensive coho management plan**; 30.0 MSP level of adult spawners identified in FMP
Stillaguamish	37% (7.8%) 27.7	≤50%; 2003 total exploitation rate ceiling based on comanager comprehensive coho management plan[b]; 17.0 MSP level of adult spawners identified in FMP
Snohomish	33% (7.8%) 147.6	≤60%; 2003 total exploitation rate ceiling based on comanager comprehensive coho management plan[b]; 70.0 MSP level of adult spawners identified in FMP
Hood Canal	41% (5.9%) 25.8	≤45%; 2003 total exploitation rate ceiling based on comanager comprehensive coho management plan[b]; 21.5 MSP level of adult spawners identified in FMP
Strait of Juan de Fuca	14% (5.8%) 18.0	≤40%; 2003 total exploitation rate ceiling based on comanager comprehensive coho management plan[b]; 12.8 MSP level of adult spawners identified in FMP
Canada		
Interior Fraser (Thompson River)	8.3%	≤10%; total exploitation rate for all US fisheries south of the US/Canada border

* Ocean escapement is the number of salmon escaping ocean fisheries and entering fresh water.

** Annual management objectives may be different than FMP goals and are subject to agreement between WDFW and the treaty tribes under US District Court orders. Total exploitation rate includes all fisheries and is calculated as total fishing mortality divided by total fishing mortality plus spawning escapement.

Table 3-18. Expected mark rates for Council-adopted ocean salmon fisheries with selective coho retention, 2003.

Area	Fishery	June	July	August	September	2002 Observed
North of Cape Falcon						
Neah Bay (Area 4)	Recreational	39%	57%	45%	52%	39%
	Non-Indian troll	—	47%	47%	52%	NA
La Push (Area 3)	Recreational	64%	54%	64%	18%	28%
	Non-Indian troll	—	55%	50%	71%	NA
Westport (Area 2)	Recreational	75%	74%	72%	74%	56%
	Non-Indian troll	—	60%	70%	50%	NA
Columbia River (Area 1)	Recreational	89%	87%	83%	83%	58%
	Non-Indian troll	—	77%	78%	77%	NA
Buoy 10	Recreational	—	—	81%	81%	74%
South of Cape Falcon						
Cape Falcon to Humbug Mt.	Recreational	—	—	—	—	56%
Tillamook	Recreational	80%	75%	67%	—	—
Newport	Recreational	77%	75%	68%	—	—
Coos Bay	Recreational	74%	71%	58%	—	—

Coho production in the OPI area has exceeded 4.3 million fish (1976) and been as low as 216,400 fish in 1995, (Figure 3-41). Production was consistently low throughout the 1990s but since 2000 has increased and is similar to the average production since 1970 (1.5 million coho). The highest ocean escapement to the Columbia River was over 1.5 million fish in 1986; the lowest Columbia River escapement was 75,200 coho in 1995. Columbia River escapements since 2000 have exceeded the 1970-2002 escapement average (407,200 coho). Historically, most of the production was harvested in ocean fisheries; ocean fisheries accounted for almost 90% of the OPI production in some years, while ocean escapement to the Columbia River was less than 10% of OPI production during these years (Figure 3-42). In recent years, ocean fisheries account for about 10-15% of the total OPI coho production, while Columbia River escapement has been approximately 70% of the total OPI coho production. The remaining coho include escapement to the Oregon Coast and California OPI areas.

Commercial troll fisheries have been closed to coho retention south of Cape Falcon since 1993. In 2000 and 2001, commercial troll selective fisheries for marked hatchery coho occurred from Cape Falcon, OR to the Queets River, WA. In 2002, commercial troll selective fisheries for marked hatchery coho occurred from Cape Falcon to Leadbetter Point, WA. Limitations on chinook harvest (such as fishery closures in July and a 4-spread requirement on gear) have also been used to reduce impacts to OCN coho.

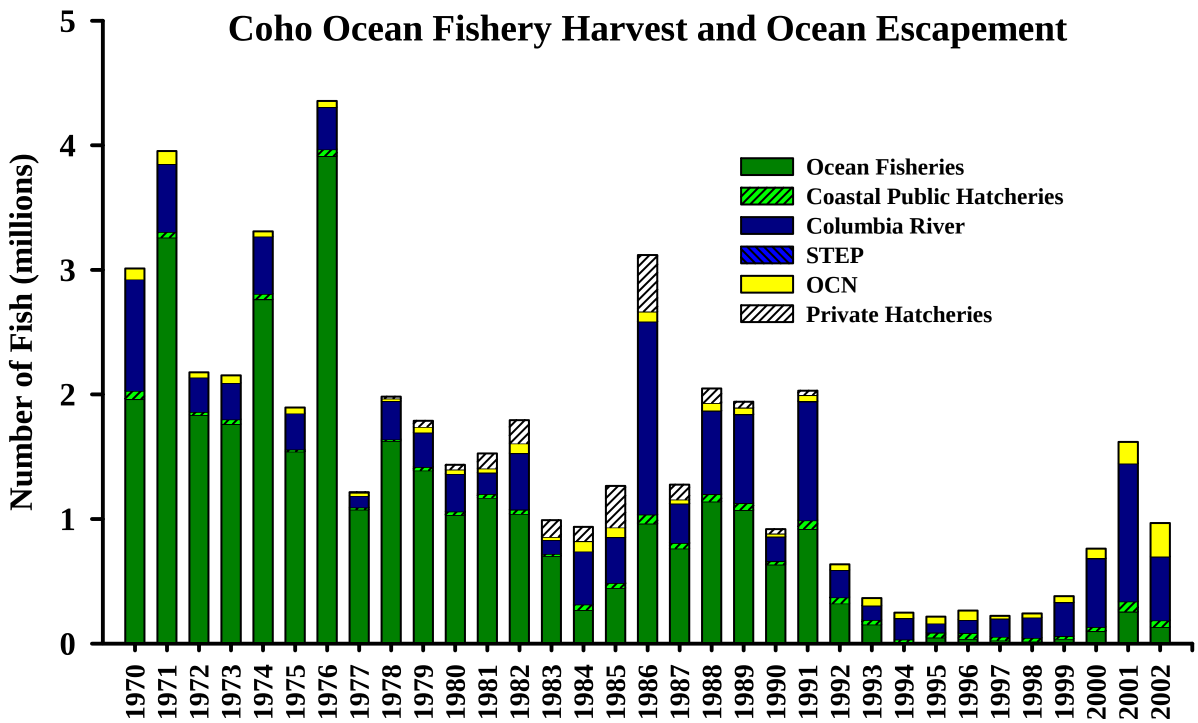

Figure 3-41. Coho salmon ocean fisheries harvest and ocean escapement of the primary coho stock components within the OPI area, 1970–2002.

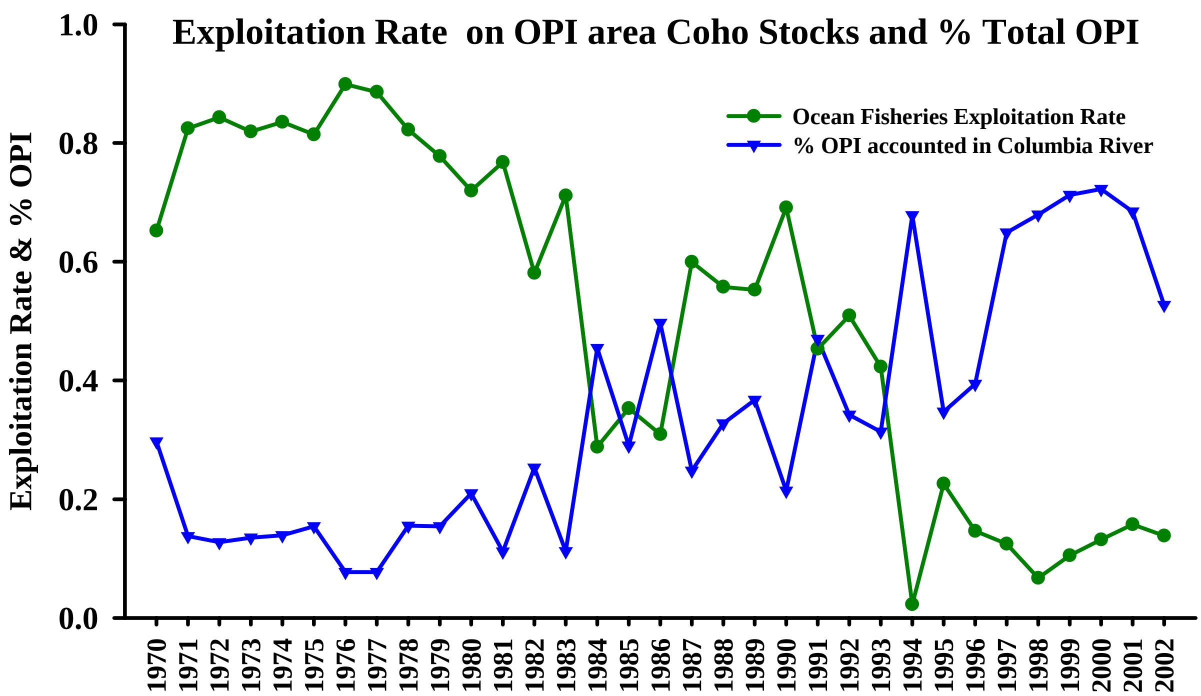

Figure 3-42. Exploitation rate of ocean fisheries on OPI area coho stocks and percent of total OPI area production accounted for by coho ocean escapement to the Columbia River.

Ocean coho harvest in PFMC-managed waters generally occurred from May–October. California ocean commercial troll fisheries occur from May–October, although most landings are in June and July (Figure 3-43). Oregon ocean commercial troll fisheries generally are from June–October, with the largest harvests in July and significant harvest in August (Figure 3-43). Washington ocean non-Indian troll fisheries are from May–September and most of the harvest occurs in July and August (Figure 3-43). Washington treaty Indian commercial ocean troll fisheries occur throughout the year, with the majority of harvest in July–August, although in recent years, the September harvest has been substantial (Figure 3-43).

The ex-vessel value and the price per pound of troll-caught coho in California, Oregon, and Washington ocean fisheries has declined since the 1980s (Figure 3-44). Minimal fishing occurred in all areas throughout the 1990s; recent year hatchery selective fisheries have occurred in Oregon and Washington. The total ex-vessel value of these fisheries in recent years has been a fraction of their historical value. Price per pound also has generally been low, except for the Washington 2002 fisheries when the price was comparable to some historical years.

Retention of coho in ocean recreational fisheries has been restricted since 1993. Since 1998, coho-directed recreational fisheries in the OPI area have been selective for adipose fin-clipped hatchery-marked fish. Improving hatchery coho populations in the OPI area in recent years have allowed increasing opportunities for a hatchery-marked coho fishery. Recreational ocean harvest of coho in California is generally greater in the private sector than by charter boats; harvest has been minimal since the 1994 season for either boat type (Figure 3-45). In Oregon, recreational ocean harvest of coho is dominated by private boats (Figure 3-45). In Washington, coho landings by charter boats historically exceeded private boat landings, but the private boat harvest has been greater in recent years (Figure 3-45).

Angler effort in California has remained relatively steady over the past 20 years due to stable hatchery chinook runs, primarily from the Sacramento River. Beginning in the early 1980s, angler effort was reduced significantly in Washington ocean fisheries in response to constraints on chinook and coho, and angler effort in Oregon lessened due to constraints on Oregon coastal wild coho. Angler effort in both Washington and Oregon has rebounded recently because of improved chinook and coho abundance and implementation of selective fisheries (Figure 3-46).

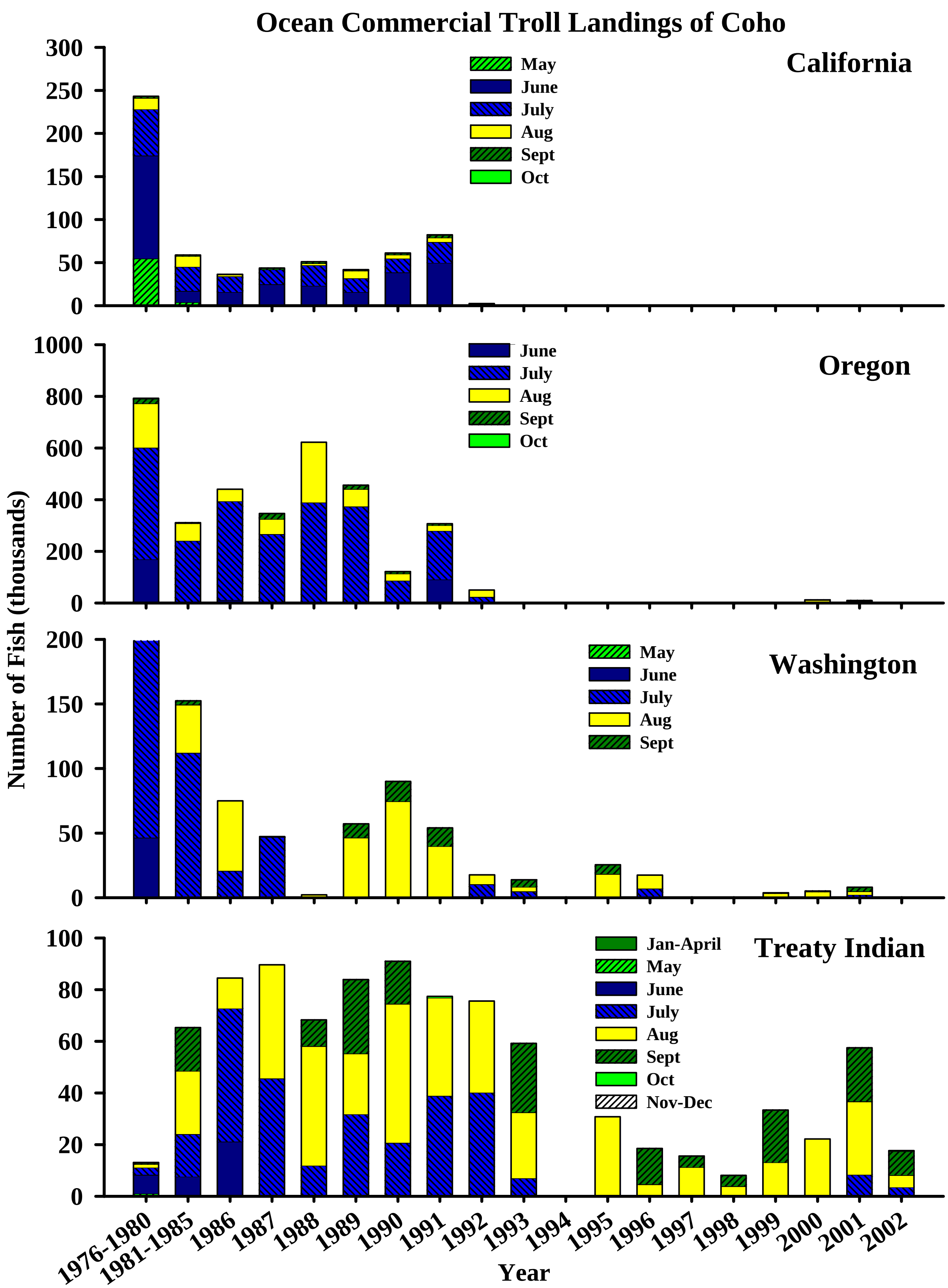

Figure 3-43. California, Oregon, Washington, and treaty Indian ocean commercial troll landings (in thousands of fish) by month, 1976–2002.

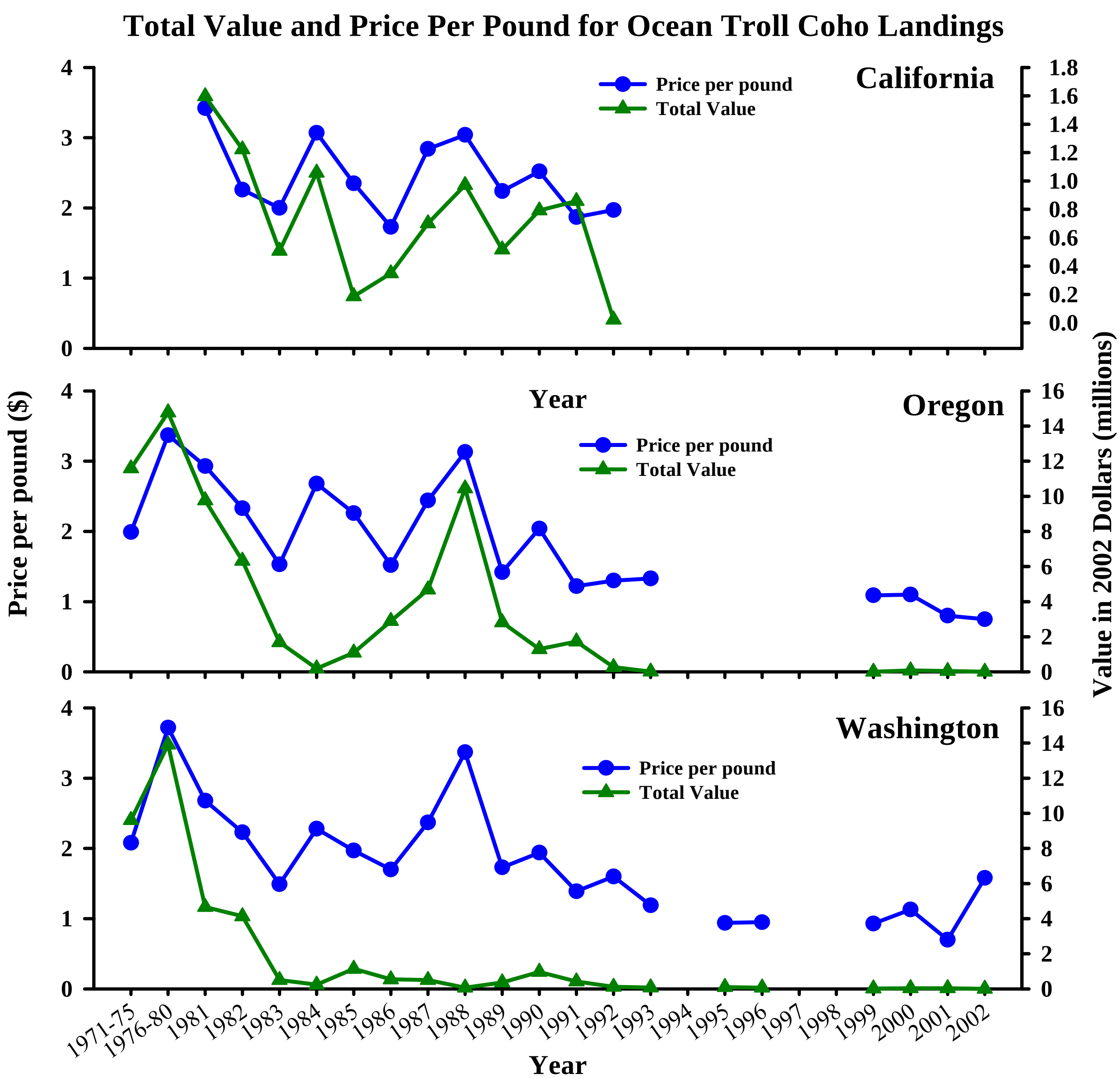

Figure 3-44. Total value and price per pound (in 2002 dollars) for ocean troll coho landings in California, Oregon, and Washington, 1971–2002.

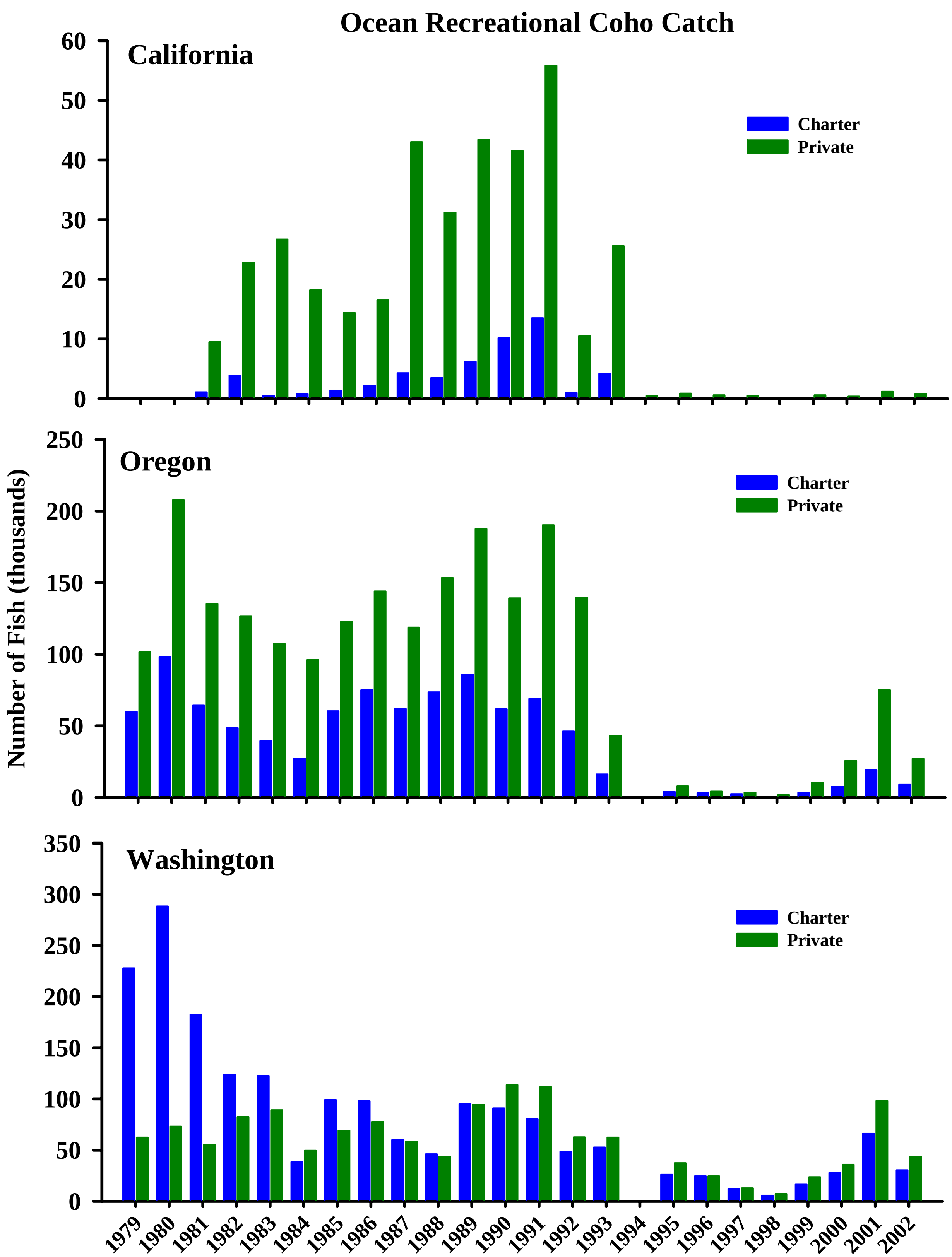

Figure 3-45. Ocean recreational coho catch 1979–2002.

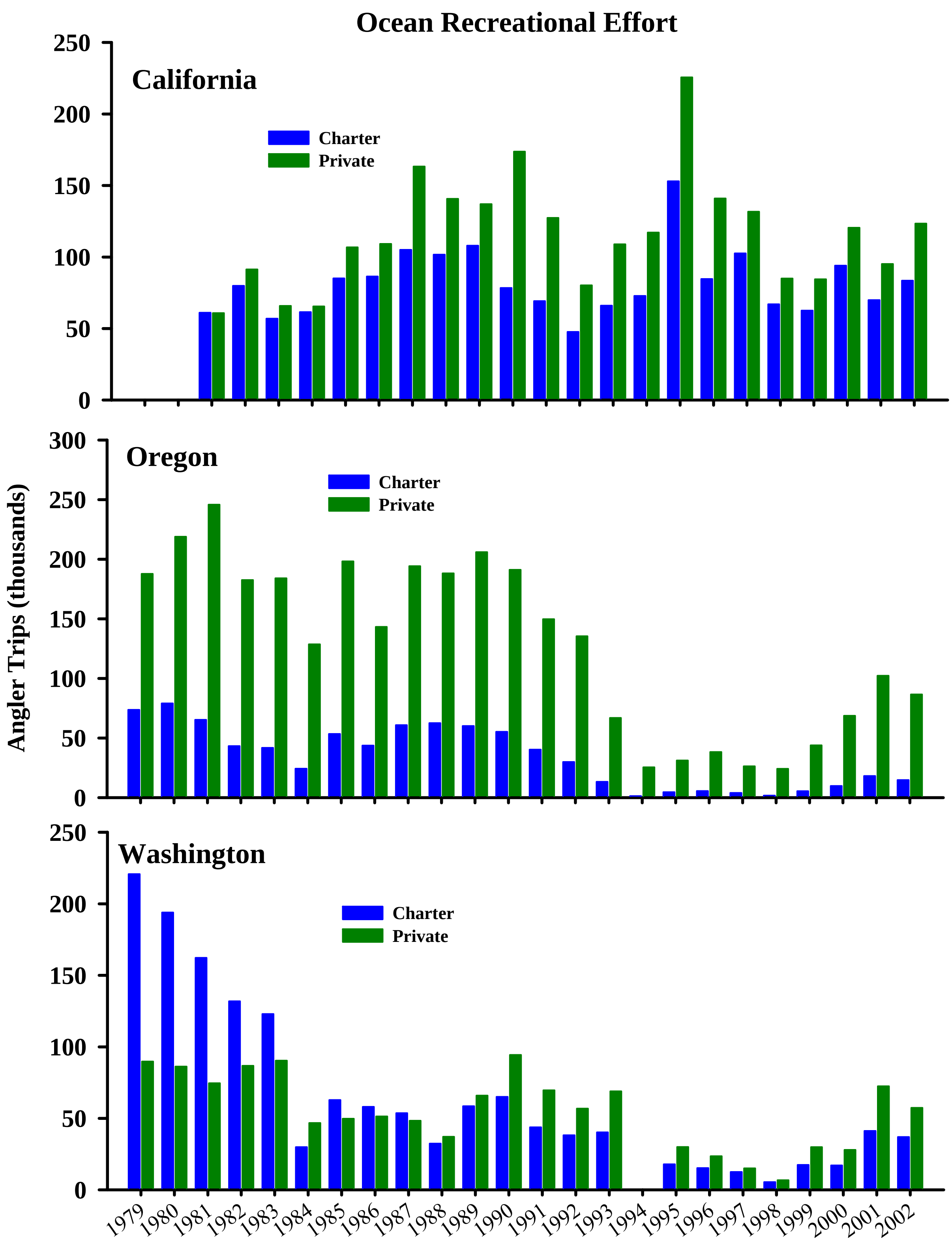

Figure 3-46. California, Oregon, and Washington ocean recreational salmon effort (in thousands of angler trips) by boat type, 1979–2002.

Columbia River Fisheries

Coho are harvested in Columbia River mainstem and Select Area commercial fisheries, as well as in Buoy 10, mainstem Columbia, and tributary sport fisheries. Coho also are harvested in treaty Indian fisheries in the Columbia River and tributaries upstream of Bonneville Dam. The Columbia River Compact manages coho fisheries under the requirements of a *US v. Oregon* Fall Management Agreement for upper Columbia coho and Oregon ESA limitations for Lower Columbia Natural coho. The resulting management requirements are:

- pass a minimum of 50% of upper Columbia coho through ocean and lower Columbia River fisheries to escape over Bonneville Dam, and
- fishery impacts to lower Columbia natural coho not to exceed management matrix levels as adopted by the OFWC (14% in 2002).

Maximum allowable freshwater impacts were developed for OCN coho (Table 3-19) to guide Columbia River coho fisheries. These rates were adopted by OFWC and are implemented by ODFW and WDFW through the Columbia River Compact.

Table 3-19. Harvest management matrix for Lower Columbia Natural (LCN) coho with maximum allowable freshwater fishery mortality rates.

	Marine Survival Index (based on return of jacks per hatchery smolt)			
Parental Escapement*	**Critical (<0.0008)**	**Low (<0.0015)**	**Medium (<0.0040)**	**High (>0.0040)**
High >75% of full seeding	<4%	<7.5%	<15%	<22.5%
Medium >50% to ≤75% of full seeding	<4%	<7.5%	<11.5%	<19%
Low >20% to ≤50% of full seeding	<4%	<7.5%	<9%	<12.5%
Very Low >10% to ≤20% of full seeding	<4%	<6%	<8%	<10%
Critical ≤10% of full seeding	0-4%	0-4%	0-4%	0-4%

* Full Seeding: Clackamas River = 3,800; Sandy River = 1,340.

Combined total harvest rates, including ocean and Columbia River, also were adopted to ensure that total exploitation rates are consistent with state coho management requirements (Table 3-20). Established for Oregon state-listed coho, these fishing rates also provide harvest protection for wild Washington coho.

Table 3-20. Cumulative exploitation rates for LCN coho under the combined management protocols proposed for setting ocean and in-river fishery harvest rates.

	Marine Survival Index (based on return of jacks per hatchery smolt)			
Parental Escapement*	**Critical (<0.0008)**	**Low (<0.0015)**	**Medium (<0.0040)**	**High (>0.0040)**
High >75% of full seeding	<11.7%	<21.4%	<40.5%	<57.4%
Medium >50% to ≤75% of full seeding	<11.7%	<21.4%	<29.2%	<49.8%
Low >20% to ≤50% of full seeding	<11.7%	<21.4%	<22.7%	<34.4%
Very Low >10% to ≤20% of full seeding	<11.7%	<16.3%	<18.1%	<19.9%
Critical ≤10% of full seeding	0-11.7%	0-11.7%	0-11.7%	0-11.7%

* Full Seeding: Clackamas River = 3,800; Sandy River = 1,340.

The Buoy 10 area at the mouth of the Columbia River provides the most popular and productive sport coho fishing for Columbia River stocks. Buoy 10 angler trips exceed 100,000 in years of high coho abundance, and the combined Oregon and Washington economic impact has been as high as 9 million (Table 3-21). The coho harvest in the Buoy 10 sport fishery has exceeded 100,000 fish four times since 1986, and exceeded 200,000 fish in 1992 (Figure 3-38). Coho salmon are actively feeding when entering the Columbia estuary and fishing can be quite successful during mid-August to mid-September. Sport harvest of coho is less productive in the mainstem Columbia upstream of the estuary area.

Table 3-21. Angler trips and economic impact (in 2002 dollars) of the Buoy 10 recreational fishery, 1982–2002.

Year	Angler Trips (000s)	Economic Impact (000s) Oregon	Economic Impact (000s) Washington
1982	17.3	NA	NA
1983	7.1	NA	NA
1984	67.4	NA	NA
1985	32.2	NA	NA
1986	102.2	NA	NA
1987	125	$2,169	$3,928
1988	183	$3,075	$6,212
1989	156	$2,346	$5,148
1990	80	$1,264	$2,386
1991	172	$2,672	$5,544
1992	115	$1,762	$3,638
1993	76	$1,179	$2,182
1994	9	$189	$230
1995	25	$491	$615
1996	18	$373	$420
1997	56	$910	$1,728
1998	30	$507	$860
1999	50	$907	$1,276
2000	73	$1,335	$2,000
2001	126	$2,636	$2,940
2002	84	$1,803	$1,849

WDFW statewide rules declare that salmon fisheries are closed unless otherwise specified in Special Rules. Depending on the strength of adult salmon returns, WDFW promulgates regulations allowing spring chinook, fall chinook, and coho salmon fisheries in lower Columbia River tributaries. Coho fisheries typically overlap fall-run chinook fisheries in the Washington tributaries. Salmon-directed fisheries will vary from year to year and from stream to stream depending on the health of salmonid populations and sizes of runs forecast for each particular stream. Fisheries for adipose fin-clipped hatchery coho salmon destined for the Grays, Elochoman, Cowlitz, Toutle, Kalama, Lewis, Washougal, and Little White Salmon rivers occur from August through January in most years. Anglers experience good success rates for coho in the tributary fisheries (Figure 3-38). Selective fishery regulations have been in place for all lower Columbia River sport fisheries since 1998.

Fall commercial fisheries before late-September primarily harvest early coho and fall chinook. Commercial fisheries after early October primarily harvest late hatchery coho stocks and sturgeon; fisheries between these two time periods harvest both early and late coho stocks. Late fall seasons in October primarily target hatchery coho in the lower river below the mouth of the Lewis River.

Commercial fishing in Columbia River off-channel areas (i.e. Select Area fisheries) commenced in 1962 when salmon seasons were adopted for Youngs Bay, OR. Initially, openings were concurrent with the late fall mainstem gill net seasons but seasons have been separate since 1977. Recent declines in mainstem fishing opportunities prompted BPA to fund a research project to expand net-pen programs to select off-channel fishing areas. The result of this effort was the Select Area Fishery Enhancement (SAFE) Project, which has expanded to Tongue Point/South Channel and Blind/Knappa Slough in Oregon and Deep River and Steamboat Slough in Washington. Coho fisheries occur in all five Select Areas; these fisheries primarily target hatchery coho returning to specific release sites. Coho-targeted Select Area fisheries occur from August through October; most harvest occurs in September and October. The 2001 fall Select Area fisheries harvest totaled 33,687 coho salmon.

Coho salmon are the target species for late fall lower Columbia River commercial fisheries. Late fall coho seasons end before November to avoid impacts to late returning wild Clackamas coho, chum, and winter steelhead. Late returning wild Washington coho also benefit from the November season closure. Coho are also incidentally harvested in early fall commercial fisheries targeting fall chinook. Coho salmon are also harvested in treaty Indian commercial and subsistence fisheries in Zone 6 above Bonneville Dam (Figure 3-37). No prohibitions are in place on wild coho retention for the treaty Indian fisheries, but coho harvest in the treaty Indian commercial fishery is usually minor because of constraints to protect wild steelhead.

The PFMC uses a model to estimate catch, mortality, and escapement of early and late Columbia River coho; the model also partitions the fish into lower and upriver coho. Results of the 2002 model run are summarized in Table 3-22; coho salmon exploitation rates can be inferred from the model. Note the change in the ratios of marked and unmarked coho in fisheries as marked coho are removed from the population prior to the fish entering the next fishery.

Table 3-22. Estimated catch, mortality, and escapement of marked and unmarked Columbia River basin coho salmon, 2002.

	Marked (Hatchery)		Unmarked (Wild)	
Harvest & Interim Abundance	**No. of Fish**	**Exploitation Rate**	**No. of Fish**	**Exploitation Rate**
Ocean abundance	326,649		49,234	
Alaska & Canada harvest	120	0.04%	24	0.05%
US v. Oregon area ocean abundance	326,529		49,210	
US v. Oregon area catch and mortality (w/o treaty troll)	123,761	38%	5,173	11%
Ocean natural mortality	65,344		9,910	
Columbia River mouth abundance	137,424		34,127	
Buoy 10 catch and mortality	19,074	6%	1,115	2%
Mainstem recreational catch and mortality	977	0.3%	23	0.05%
August commercial catch	74	0.02%	26	0.05%
September commercial catch	7,975	2%	2,025	4%
October commercial catch	8,429	3%	1,571	3%
Tributary escapement	100,895		29,367	
*Total exploitation**		*49.36%*		*20.15%*

* Does not include treaty Indian ocean troll fisheries or tributary recreational harvest.

3.1.8 *Chum Fishery*

3.1.8.1 Chum Harvest Over Time

Chum salmon once were very abundant in the Columbia River Basin with commercial landings ranging from 1 to 8 million pounds (80,000 to 650,000 fish) in most years before the early 1940s. Chum salmon were harvested in significant numbers in mainstem Columbia River commercial fisheries until their decline in the early 1950s. Chum were harvested in late fall with most caught in November. Corresponding with the decline in salmon returns, late fall commercial fisheries were reduced. December has been closed to commercial salmon fishing since 1949 and November commercial fisheries have been closed or minimized since 1959. Commercial chum landings gradually diminished during the 1940s and 1950s to less than 50,000 pounds annually by 1959 (Figure 3-47). Now there are neither recreational nor commercial fisheries for chum salmon in the Columbia River (ODFW and WDFW 1995). Some chum are taken incidentally in the gill net fisheries for coho and chinook salmon, but commercial landings have been 500 pounds or less since 1993 (Figure 3-47).

NOAA Fisheries' biological opinions limit the incidental impact of Columbia River fisheries targeting other species to 5% of the annual return of chum listed under the Endangered Species Act (ESA). Since Columbia River chum salmon were listed in 1999, fisheries impacts have remained below the ESA limit.

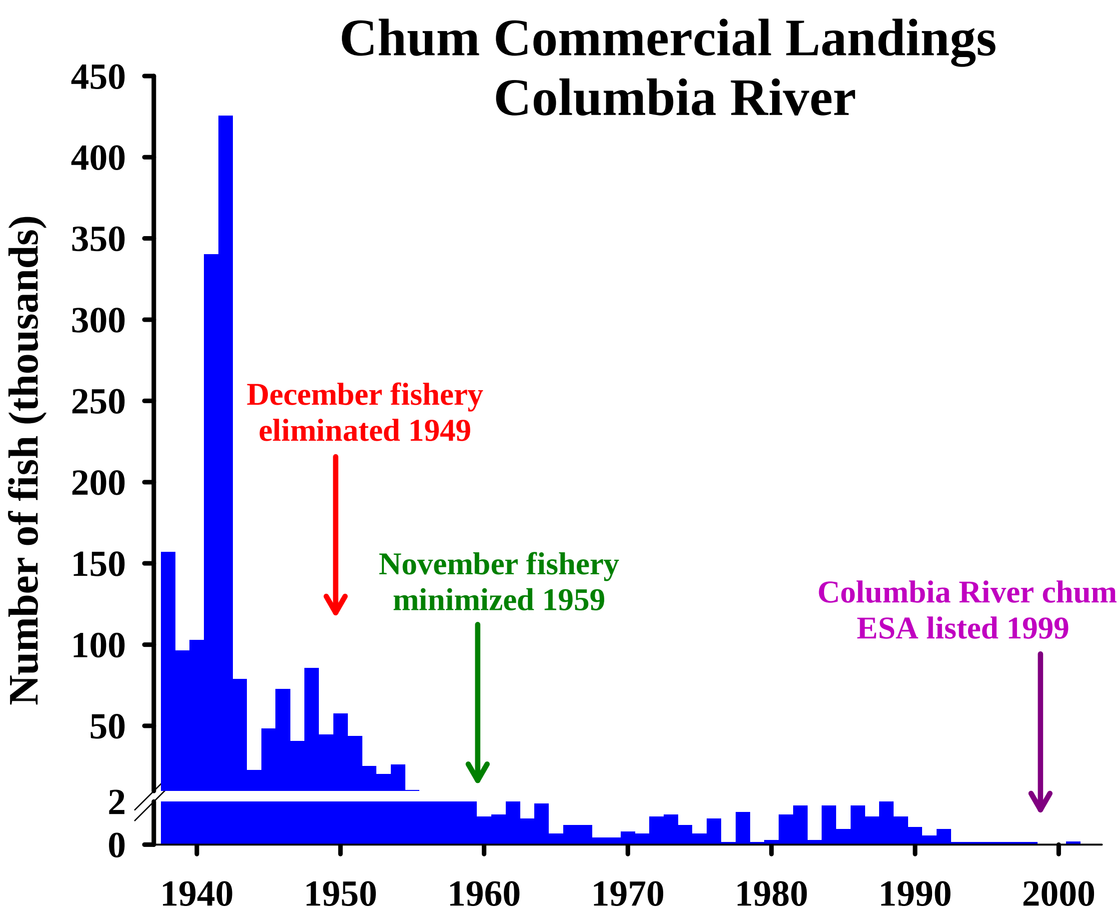

Figure 3-47. Commercial landings of chum salmon in the Columbia River from 1938–2002.

Few chum are landed in ocean fisheries south of the Strait of Juan De Fuca in Washington. Most landings occur in Canada and Alaska ocean fisheries. There is no specific information on the ocean distribution of Columbia River chum, but it is suspected they migrate similar to Puget Sound stocks, moving to the high seas of the Pacific until they mature and then migrate directly back to the Columbia River. The mature salmon would be present along the coasts of Oregon and Southern Washington in the fall after the ocean seasons have closed, and not present in the chum fisheries farther north.

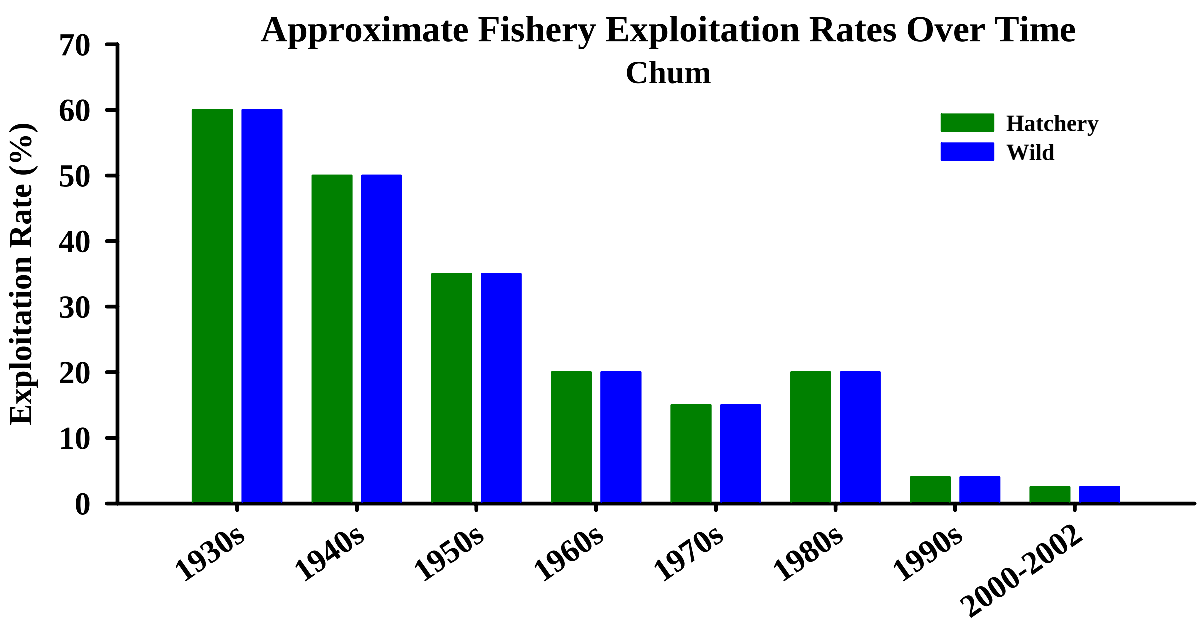

Figure 3-48. Fishery exploitation rates over time. Significant Columbia commercial harvest until 1950s. Ocean interception is rare. Columbia commercial harvest steadily decreased since 1950s. Currently, no target commercial or sport fisheries for chum.

3.1.8.2 Current Chum Harvest

Columbia River chum are harvested incidental to coho, chinook, and sturgeon during the late fall commercial season (Table 3-23). Recent commercial landings have been small, ranging from 0 to 128 chum since 1993, when management measures were implemented to protect late wild coho and chum. According to ESA management limits, Columbia River fall fisheries salmon management requires an incidental impact rate of less than 5% for Columbia River chum salmon; the season structure of the commercial fishery has resulted in less than ESA limits for the annual harvest since Columbia River chum salmon were listed as threatened in 1999. Directed chum salmon ocean commercial fisheries are limited to southern British Columbia and Washington and are managed under agreements resulting from the Pacific Salmon Treaty. Contribution of lower Columbia River chum stocks to these fisheries is expected to be minimal.

Generally, most mainstem commercial fisheries are closed before the primary chum salmon migration time. Mainstem Columbia and Washington tributary sport fishing regulations require release of all chum caught while fishing for other species. Chum are not normally encountered in treaty Indian fall fisheries upstream of Bonneville Dam.

Table 3-23. Example of current chum harvest.

Fishery	Harvest	Comment
Ocean	<1%	High seas migration and direct return to Columbia likely avoids Northern chum fisheries
Columbia River	1.5%	Incidental to commercial coho fisheries
Tributary	1%	Incidental to steelhead salmon fisheries
Total	*2.5%*	*No directed Columbia Basin fisheries*

3.1.8.3 Chum Harvest Management Details

Directed chum salmon ocean commercial fisheries occur in Alaska, British Columbia, and Washington (Table 3-24) and are managed under agreements resulting from the Pacific Salmon Treaty (PST). Although there is very little specific information on the ocean distribution of Columbia River chum salmon, given the timing and distant location of the ocean fisheries that target chum, the contribution of stocks of lower Columbia River chum to these fisheries is expected to be minimal.

Table 3-24. Preliminary 2002 chum salmon harvest in ocean fisheries managed under PST.

Fishery	Total 2002 Harvest
ALASKA	
SE Alaska District 104 purse seine	75,218
SE Alaska District 101 drift gill net	144,920
SE Alaska District 106 drift gill net	112,541
SE Alaska District 108 drift gill net	2,017
SE Alaska District 111 drift gill net	231,966
CANADA	
Johnstone Strait	648,000
Strait of Georgia	225,000
Fraser River	100,530
West coast Vancouver Island	554,000
WASHINGTON	
Strait of Juan de Fuca Treaty Indian	1,303
San Juan Islands/Point Roberts Treaty Indian	59,314
San Juan Islands/Point Roberts non-Indian	49,952
Total PST Harvest	*2,204,761*

Late fall commercial fisheries are regulated by the Columbia River Compact and are focused on harvest of late stock hatchery-produced coho destined for Washington lower Columbia River facilities, and on harvest of white sturgeon remaining on the annual commercial allocation. The Compact exercises time, area, and gear regulations to target hatchery coho and sturgeon while minimizing impacts to chum and late wild Clackamas coho. Clackamas coho are a later-timed late coho which begin entry into the Columbia in late October and early November, similar to Columbia River chum entry time. The commercial season typically closes by late October, unless significant late hatchery coho remain in the river and/or sturgeon harvest allocation is not yet attained. Commercial fisheries extending into the end of October or early November are restricted to larger mesh size to catch sturgeon and avoid chum and coho, or closed in the very lowest part of the lower Columbia where the chum and Clackamas coho presence would be highest. The Compact staff monitors the incidental catch of chum throughout October and recommends preventive regulations earlier in the season, if chum begin to be intercepted earlier than normal. The management strategies employed since 1993 have enabled access to coho, chinook, and sturgeon while minimizing chum harvest (Figure 3-49). Although the most significant reduction in chum harvest occurred in the 1950s, another significant reduction occurred in the 1990s resulting from late fall commercial management changes (Figure 3-50).

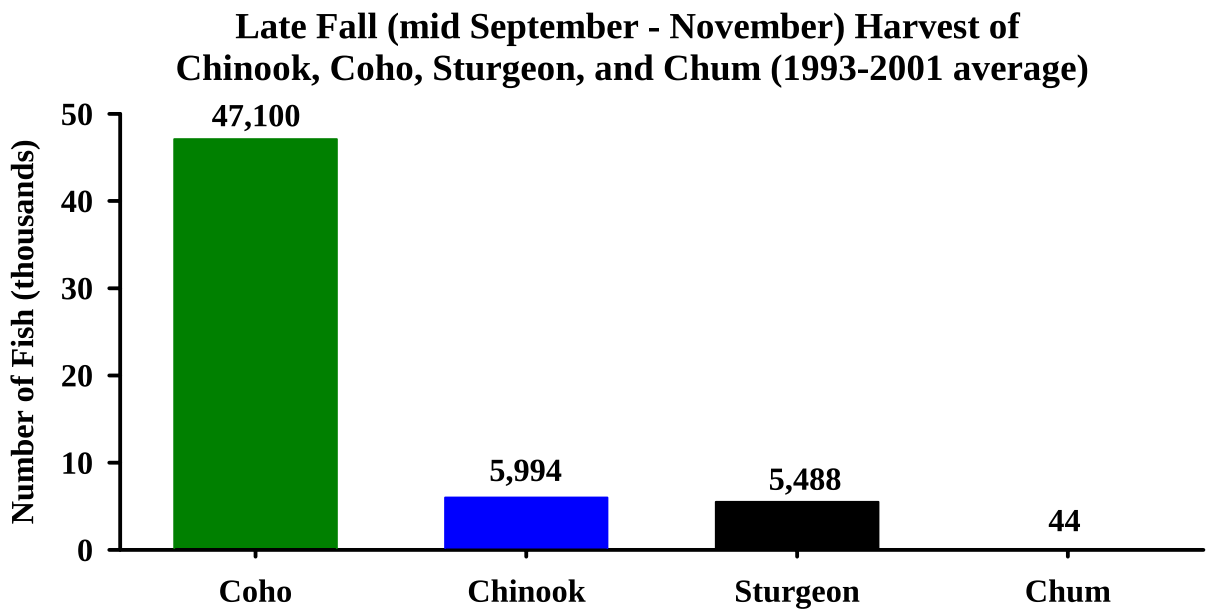

Figure 3-49. Average harvest of chinook, coho, sturgeon, and chum in late fall (mid September to November), 1993-2001.

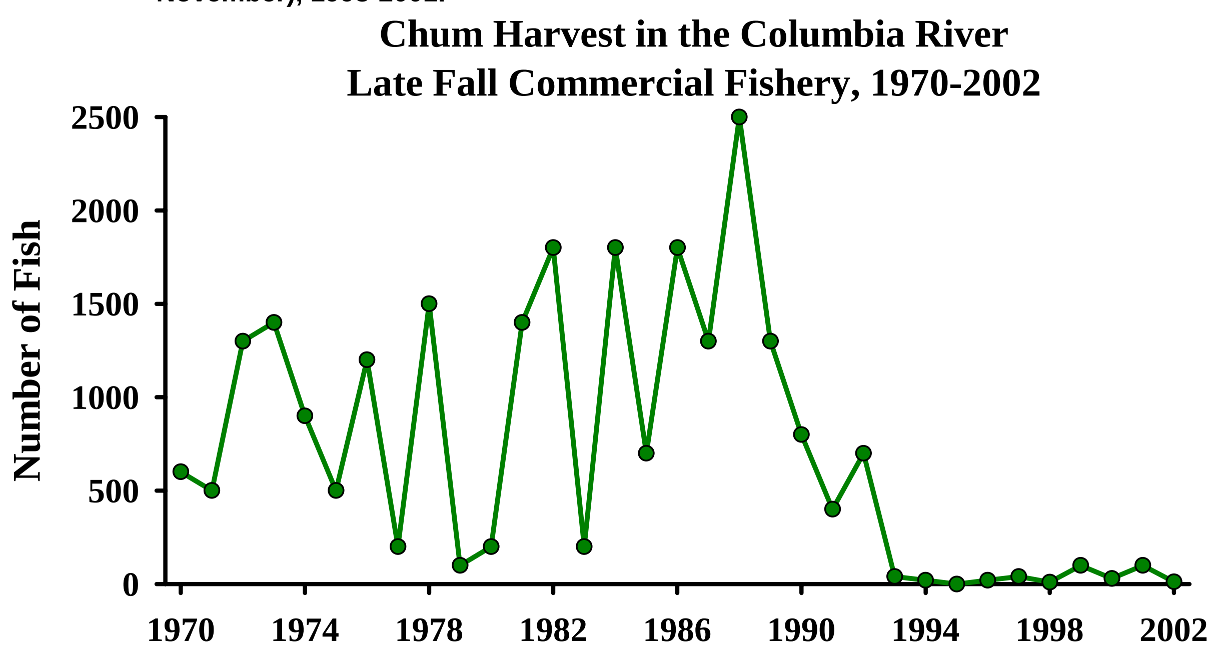

Figure 3-50. Late fall commercial harvest of chum salmon in the Columbia River, 1970–2002.

Recreational harvest impacts of chum salmon in the lower Columbia River is minimal. Targeted salmon fisheries in the Grays River were estimated to harvest about 5-10% of the wild chum run prior to 1995. WDFW's salmon catch record system was originally designed to track chinook and coho harvest; pink, sockeye, and chum salmon were combined in one category so direct chum salmon catch estimates are unavailable. Retention of chum salmon in the mainstem Columbia River and the tributaries has been prohibited since 1992 in Oregon and since 1995 in Washington; Washington tributaries are closed to chum salmon fishing. Current chum salmon interception rates in Washington tributary recreational fisheries are estimated to be less than 5% with a hooking mortality estimate of 8.6%; these estimates result in a tributary sport fishing mortality rate of less than 1% from 1995 to the present (WDFW 2003).

In a biological assessment of incidental impacts of 2002 Columbia River Fall Fisheries on ESA-listed salmon and steelhead, the *US v. Oregon* TAC estimated a Columbia River chum run size of 2,400 fish and a mainstem Columbia harvest of 38 chum. A WDFW estimate of 1 % mortality rate in Washington tributary fisheries results in a combined Columbia basin harvest impact estimate of 2.58 % of the total chum return in 2002 (Table 3-25).

Table 3-25. Harvest related mortality estimates for ESA-listed chum salmon in Columbia River basin fisheries during August–December, 2002.

Fishery	Columbia River Chum
Mainstem salmonid sport fishery	0
Mainstem commercial salmon/sturgeon fishery	35
Select Area fall commercial fisheries	3
Total mainstem harvest	*38*
Tributary sport fisheries	24
Run size at Columbia River mouth	2,400
Harvest/mortality	**2.58%**

3.1.9 Summer Steelhead Fishery

3.1.9.1 Summer Steelhead Harvest Over Time

Historically, steelhead were harvested in Columbia non-Indian fall commercial gillnet fisheries along with salmon. From 1938-74, steelhead catch ranged from 4,000 to 239,800 (Figure 3-53). Non-Indian commercial steelhead harvest has been prohibited since 1975.

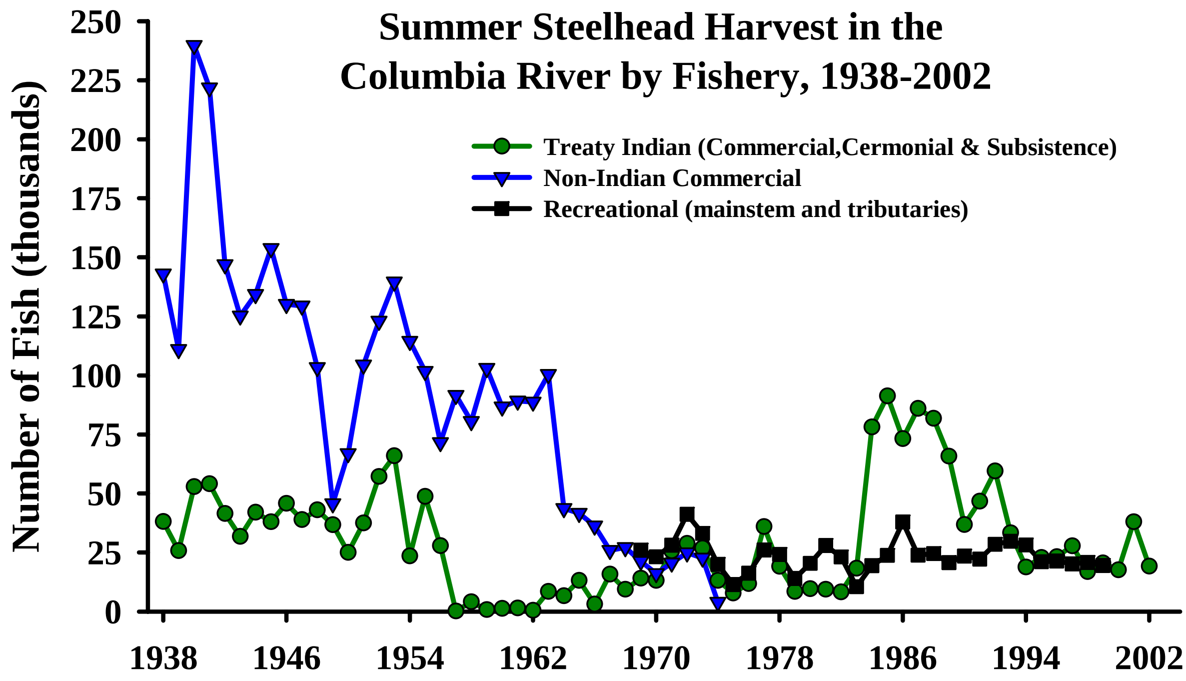

Figure 3-51. Harvest of summer steelhead in the Columbia River from 1938–2002.

Commercial harvest rates were highest when the Columbia River was open 270 days per year (pre-1943). Summer steelhead commercial harvest was reduced beginning in 1965 when the summer commercial seasons (June and July) were closed to protect depressed summer chinook populations. The Columbia and tributary recreational fisheries began increasing in effort and total harvest in the 1960s. After 1975, when non-Indian commercial take of steelhead was prohibited, the harvest impacts of lower Columbia steelhead were almost entirely from recreational fisheries (although incidental catch and release mortality of summer steelhead can occur in lower Columbia River fall gill net fisheries). The treaty Indian commercial fishery became more significant after 1968 following federal court decisions clarifying treaty Indian fishing rights. Most treaty Indian steelhead harvest occurs in September during the fall salmon season. The sport harvest of summer steelhead in the lower Columbia tributaries can be significant in years of high production (Figure 3-52). Release of wild steelhead in the mainstem Columbia and Washington tributaries was implemented in 1984.

Lower Columbia River steelhead were listed as threatened under the federal ESA in 1998. This ESU includes all naturally spawned summer and winter steelhead in the Columbia River basin and tributaries between the Cowlitz and Wind rivers (inclusive) in Washington and the Willamette and Hood rivers (inclusive) in Oregon, excluding steelhead in the upper Willamette River above Willamette Falls.

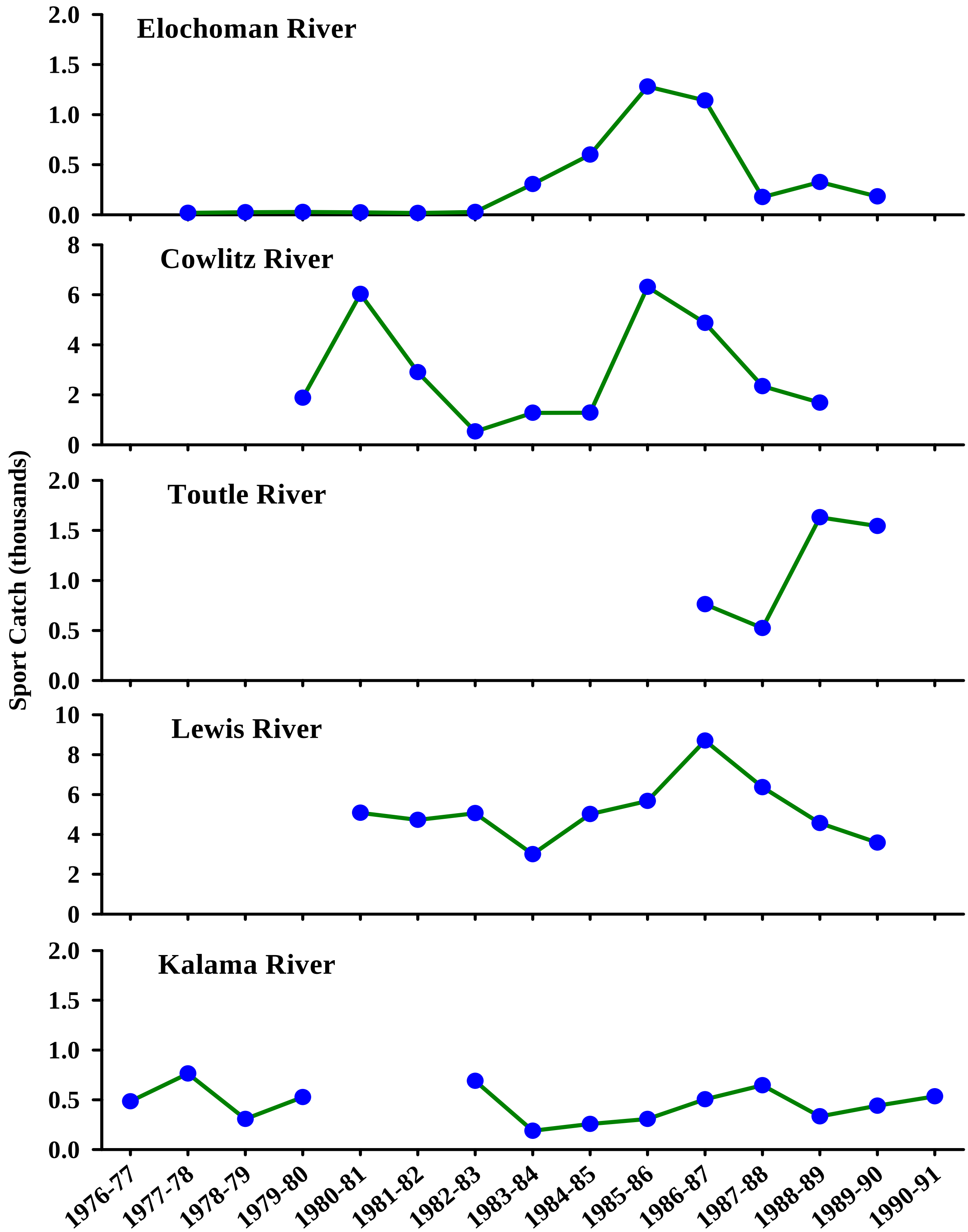

Figure 3-52. Summer steelhead sport catch harvest (combined wild and hatchery) in Washington lower Columbia basin tributaries.

3.1.9.2 Current Summer Steelhead Harvest

The Columbia River summer steelhead run is comprised of populations from lower and upper river tributaries. The lower river component of the run is primarily hatchery-produced, derived from Skamania stock, and tends to be earlier timed (May-June) than upriver stocks. The upriver summer steelhead run has historically been separated into A and B groups based on run timing. Group A steelhead include early-returning Skamania stock which pass Bonneville Dam prior to July and are primarily destined for Bonneville Pool tributaries. Group A also includes non-Skamania stock that pass Bonneville Dam from late June through late August on their way to tributaries throughout the Columbia and Snake River basins. Group B steelhead return to the Clearwater and Salmon rivers in Idaho and pass Bonneville Dam from late August through October. In recent years, Group A and B steelhead have not shown the bimodal migration timing peaks. To alleviate fisheries management problems that occur with overlapping runs, the *US v. Oregon* TAC developed a new method in 1999 to assess the returns of Group A and B steelhead. The new index method defined three index stocks: Skamania Index (all fish counted at Bonneville Dam from April 1 to June 30), Group A Index (fish passing Bonneville Dam from July 1 to October 31 that are less than 30 in [78 cm] FL), and Group B Index (fish passing Bonneville Dam from July 1 to October 31 that are greater than or equal to 30 in [78 cm] FL).

Treaty Indian commercial and C&S fisheries in Zone 6 target summer steelhead. Since 1984, the commercial catch has been sampled to determine the percentage of hatchery and wild/natural fish for both Group A and B Index components. Harvest of wild fish in the treaty Indian commercial fishery is compared to the number of wild fish passing Bonneville Dam to determine and manage treaty Indian harvest impacts.

The majority of summer steelhead sport harvest occurs in the tributaries. Tributary harvest is limited to hatchery-marked steelhead only; the date at which this regulation became effective varies by tributary. Hatchery-only harvest restrictions on mainstem Columbia River sport fisheries have been in effect since 1984 to protect wild summer steelhead.

The non-Indian commercial handling of summer steelhead is limited by time and gear restriction. Large mesh gear (minimum of 8 in) is used to harvest chinook and sturgeon while minimizing the capture of steelhead. Prior to 2002, large mesh gill nets were used to target spring chinook and sturgeon while minimizing steelhead handling. In 2002, a live capture spring chinook commercial tangle net fishery was established and resulted in significant steelhead handling in smaller 5.5-in mesh gill net gear. Mesh size was further reduced to avoid capture of steelhead by gilling (instead tangling the fish) and improve survival of released steelhead. Treaty Indian fishery impacts to wild summer steelhead are limited to a maximum of 15% according to a *US v. Oregon* Fall Management Agreement and ESA requirements.

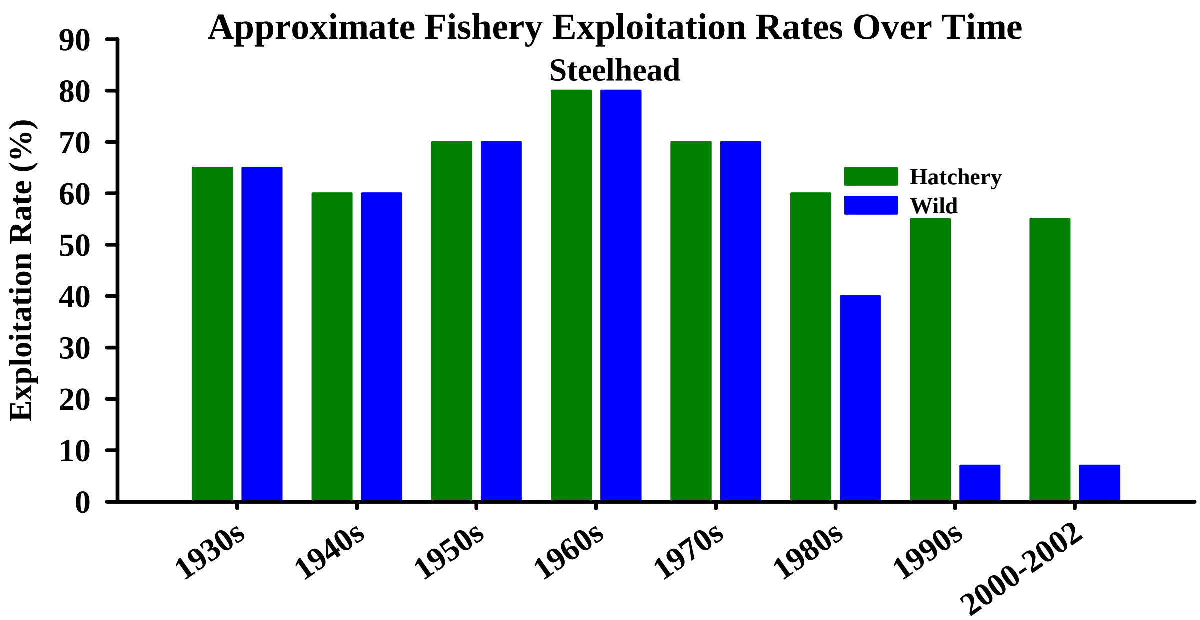

Figure 3-53. Steelhead fishery exploitation rate over time. Columbia commercial harvest significant until prohibited in 1975. Popular sport fish in mainstem Columbia and tributaries with significant catch since 1950s. Selective fisheries implemented for summer steelhead beginning in 1984.

Impacts from mainstem Columbia sport harvest occur primarily during summer months. The tributary sport fishery harvest rates of hatchery summer steelhead are variable, generally ranging from 30-70 %, with the highest rates in the tributaries where there is the most hatchery production (Cowlitz, Lewis, Kalama). The wild steelhead impacts also vary by tributary (3-6%) with the highest impacts in the tributaries with the most hatchery production. Distribution and estimated total harvest exploitation of hatchery and wild steelhead is illustrated in Table 3-26.

Table 3-26. Example of lower Columbia steelhead harvest exploitation and distribution under current management.

Fishery	H	W	Comment
Ocean	< 1%	< 1%	High seas migration results in negligible ocean harvest
Columbia River	15%	2%	Sport and incidental commercial impacts
Tributary	55%	5%	Harvest rate varies by tributary
Total Exploitation	70%	7%	Wind River wild approx. 12%, including treaty Indian harvest

Treaty Indian commercial and subsistence harvest of Wind River steelhead occurs in the Bonneville Pool, with most harvest occurring during the fall commercial seasons that target fall chinook. The treaty Indian fisheries are limited to a 15% harvest rate on wild Group B Index steelhead headed to the Snake River, which imposes fishery regulations that result in harvest limitations on other wild steelhead populations, including Wind River steelhead. The harvest of Wind River steelhead by the treaty Indian fall commercial fishery is likely lower than the wild steelhead stocks which pass through the entire treaty Indian fishery from Bonneville Dam to McNary Dam.

Generally, steelhead are not caught in commercial or recreational fisheries in the ocean. Although mark and tag data indicate that high seas steelhead distribution and drift net fisheries overlap, ocean harvest is minimal because the ocean migration pattern of most steelhead is beyond the typical ocean fisheries.

Current harvest impacts for wild steelhead populations below Bonneville Dam are associated with release handling mortality in non-Indian shad, sockeye, and fall salmon commercial fisheries that target salmon and mainstem Columbia, and tributary sport fisheries that target hatchery steelhead and salmon. Wind River steelhead harvest impacts include retained harvest in the treaty Indian fishery above Bonneville Dam. Steelhead incidental capture and handling is minimized through time, area, and gear restrictions. In 1999, an estimated 100 steelhead non-retention mortalities occurred in the fall commercial fisheries.

3.1.9.3 Summer Steelhead Harvest Management Details

Annual fishery management planning relies on run forecasts to set annual harvest quotas and predict harvest impacts on ESA-listed stocks. Managers utilize numerous forecast methods to estimate annual steelhead runs; different methods are often appropriate for different components of the run and the individual run components can be added to obtain the total run estimate. For example, with the 2003-2004 upriver summer steelhead run, the 1-salt return was predicted using the recent 5-year average, while the 2-salt return was predicted using a regression relationship between 1-salt and 2-salt returns. Independent estimates were made for the Group A and the Group B Index, and the wild and hatchery components of each.

Columbia River 2002 fall fisheries salmon management was guided by the following restrictions on steelhead harvest:

- Treaty Indian fall fisheries would be managed to limit impacts on wild Group B Index steelhead to 15% or less
- All non-Indian fisheries outside the Snake River basin will be managed for an upriver wild steelhead impact rate not to exceed 2% on Group B index steelhead
- Lower Columbia wild steelhead impacts are limited to 2%.

Summer steelhead sport fisheries exist on the mainstem Columbia River and within the tributaries. Mainstem harvest usually occurs between Tenasillahe Island and Bonneville Dam and few steelhead are caught below Tenasillahe Island. Summer steelhead enter mainstem Columbia River fisheries from March through October, but most of the catch occurs from late May through August. Generally, Group A summer steelhead comprise most of the mainstem Columbia River sport harvest annually (Figure 3-54). Steelhead are also handled during warm water recreational fisheries in Columbia River pools from the mouth of the River to Priest Rapids Dam, although impact to steelhead is minor. Creel survey data from 1993-1996 in the area between Bonneville and McNary dams and in 1994 between McNary and Priest Rapids dams indicated only 1% of steelhead were caught by non-salmonid anglers.

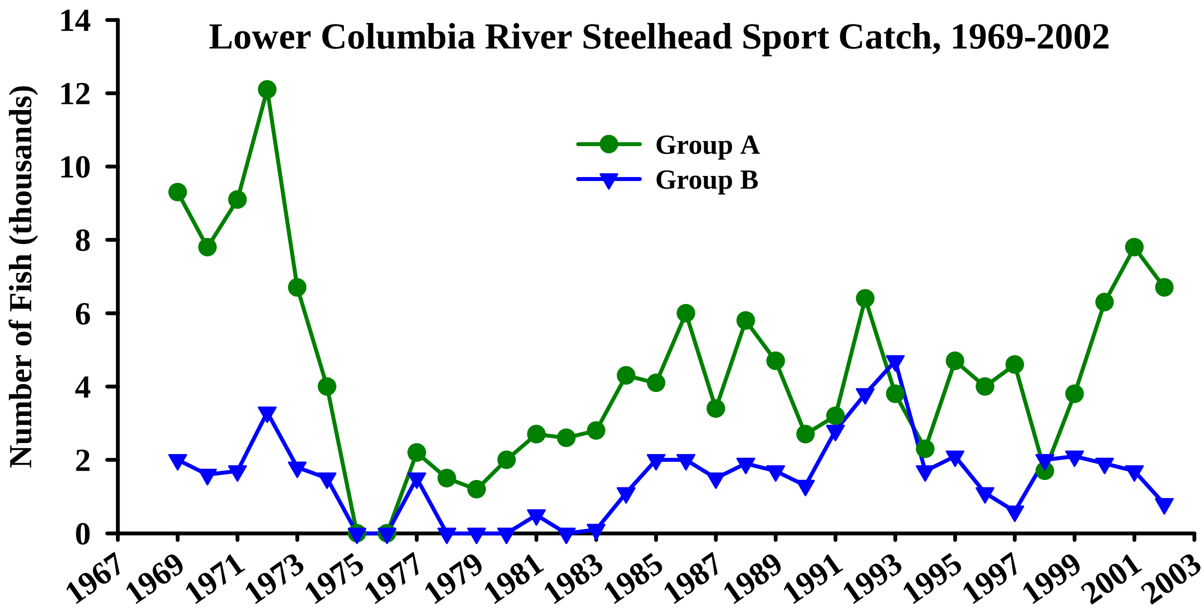

Figure 3-54. Lower Columbia River sport catch by steelhead index group, 1969–2002.

The majority of lower Columbia-origin summer steelhead sport harvest occurs in the tributaries and most of the tributary harvest occurs on the Washington side of the lower Columbia (Figure 3-55).

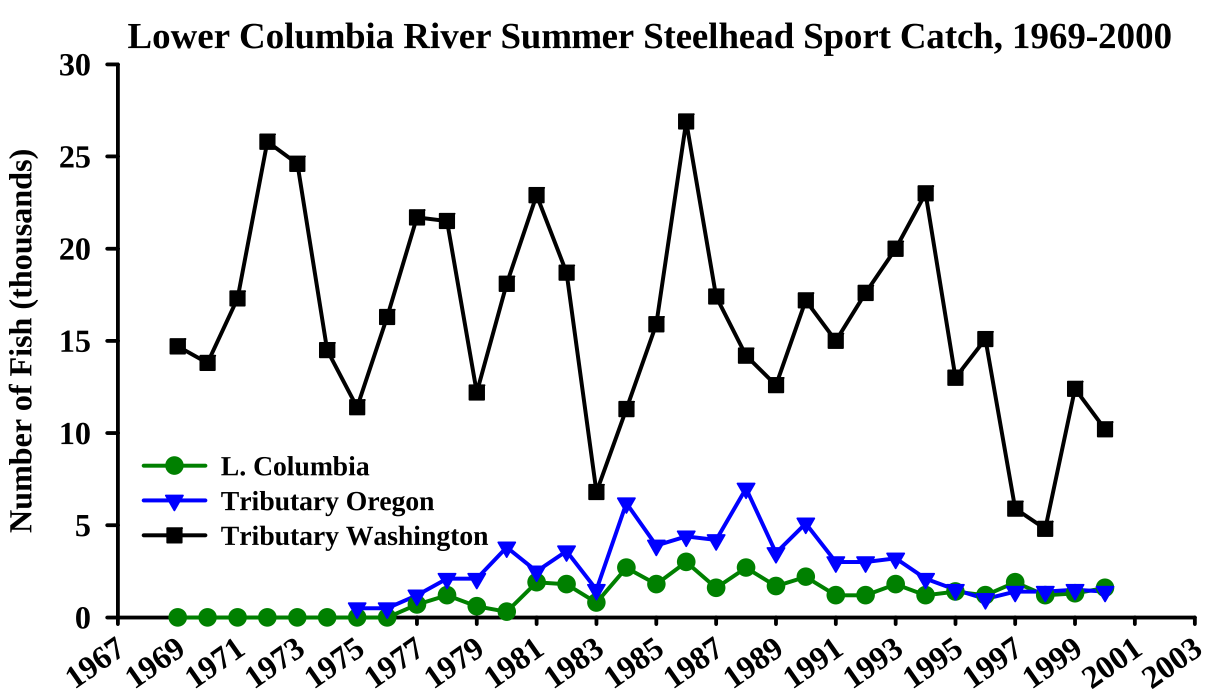

Figure 3-55. Lower Columbia River sport catch by area, 1969-2000.

Summer steelhead are native to the Kalama, Lewis, Washougal, and Wind basins. Hatchery smolts are released in the Elochoman, Cowlitz, Toutle, Kalama, Lewis, Washougal, and Little White Salmon basins for fisheries opportunity. All summer steelhead streams in Washington have substantial sanctuary water which is closed to fishing; these areas are located in the upper watersheds where an estimated 90% of the wild summer steelhead spawning occurs. Summer steelhead can also be taken incidentally in fall chinook targeted fisheries; however, the interception rate for non-targeted species is expected to be 1% or less (WDFW 2001). WDFW recreational steelhead selective fisheries are managed to achieve a maximum 10% steelhead mortality for summer steelhead populations below Bonneville Dam. WDFW manages harvest impacts for Wind River summer steelhead to 4% or less because of adverse effects on productivity caused by the operation of Bonneville Dam, fisheries research activities, and mainstem harvest impacts.

Tributary harvest rates for Kalama River wild steelhead have been made by WDFW since 1976. The Kalama River is assumed to be representative of changes in wild steelhead harvest rates after the adoption of wild steelhead release regulations. Harvest rates for both summer and winter steelhead declined from more than 60% during harvest fisheries to less than 10% in the current wild steelhead release fisheries (Figure 3-56).

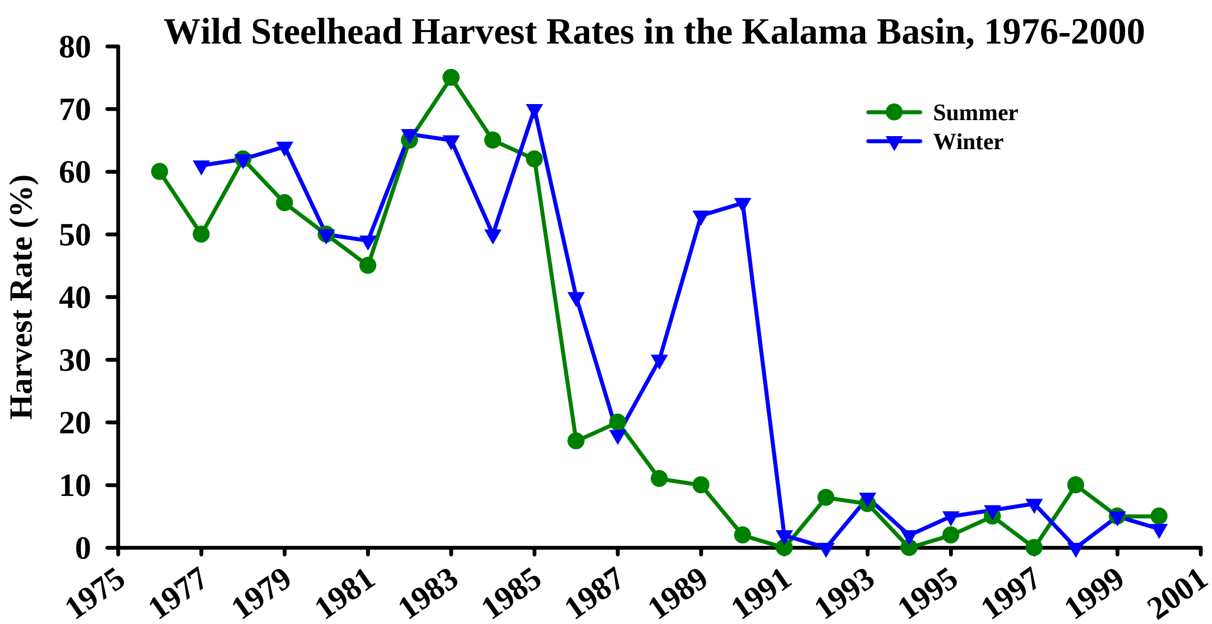

Figure 3-56. Wild steelhead harvest rates for summer and winter steelhead in the Kalama basin, 1976–2000. (Harvest for summer steelhead after 1984 and winter steelhead after 1991 is adult mortality as a result of hooking mortality in the wild steelhead release fisheries.)

WDFW (FMEP 2003) estimated tributary sport fishery encounter rates and mortality rates for wild summer steelhead in Washington tributary recreational fisheries affecting ESA-listed summer steelhead populations (Table 3-27). These estimates include all types of recreational fisheries (for all species) in the Kalama, Lewis, Washougal, and Wind River watersheds.

Table 3-27. Estimated take of ESA-listed steelhead in Washington tributary recreational fisheries.

Affected Stock	Anticipated Encounters*	Expected Mortality**
Kalama River summer steelhead	60%	5%
EF Lewis River summer steelhead	40%	3%
Washougal River summer steelhead (mainstem)	40%	3%
Wind River summer steelhead	<10%	1%

* Anticipated encounters are catch and released fish; the numbers represent the percentage of fish from a stock anticipated to be incidentally encountered by anglers of a particular fishery.

** Expected mortality is the hooking mortality of incidentally caught fish; expected mortalities are included in the anticipated encounters in terms of take.

Since 1984, returns of Group A and Group B summer steelhead have been enumerated at Bonneville Dam and sampled for wild and hatchery percentage. Group A total return (hatchery and wild) has ranged from 115,600 to 515,100 and the percentage of the run that is wild has ranged from 14% to 45%. The Group B total return (hatchery and wild) has ranged from 13,200 to 129,900 and the percentage of the run that is wild has ranged from 8% to 32%. The largest returns were recent, Group A in 2001 and Group B in 2002 (Table 3-28).

Table 3-28. Wild and hatchery contribution to Group A and Group B Index summer steelhead returns to Bonneville Dam, 1984–2003.

Year	Group A Index			Group B Index		
	% Wild	% Hatchery	Total Return	% Wild	% Hatchery	Total Return
1984	27	73	195,700	14	86	98,000
1985	18	82	281,500	32	68	40,900
1986	20	80	287,500	16	84	64,000
1987	45	55	238,300	31	69	45,000
1988	37	63	173,100	22	78	81,600
1989	30	70	193,100	16	84	77,600
1990	23	77	115,600	17	83	47,200
1991	26	74	234,100	22	78	28,300
1992	18	82	241,500	22	78	57,400
1993	21	79	136,700	12	88	36,200
1994	18	82	121,100	20	80	27,500
1995	14	86	180,000	14	86	13,200
1996	15	85	174,300	21	79	18,800
1997	15	85	208,300	11	89	36,600
1998	26	74	134,700	8	92	40,200
1999	32	68	176,400	17	83	22,100
2000	29	71	216,700	21	79	40,900
2001	27	73	515,100	14	86	86,400
2002*	27	73	323,100	25	75	129,900
2003**	25	75	279,600	18	82	64,700

* Preliminary.

** Projected.

Winter and spring treaty Indian commercial and C&S fisheries in Zone 6 targeting sturgeon can also harvest summer steelhead, but the majority of the treaty Indian summer steelhead harvest occurs during fall fisheries (Figure 3-57). In some years, the Treaty Indian Tribes have instituted an 8 in (20 cm) minimum mesh size restriction to reduce the handle of steelhead in the fall fishery and maintain impacts to wild Group B Index steelhead below the ESA limit of 15%. Also, tribal harvest generally focuses on the peak of the fall chinook run, thereby reducing the number of days needed to fish and minimizing potential impacts to steelhead. In 2001, fall treaty Indian fisheries harvested 29,200 steelhead, which is the largest harvest since 1992.

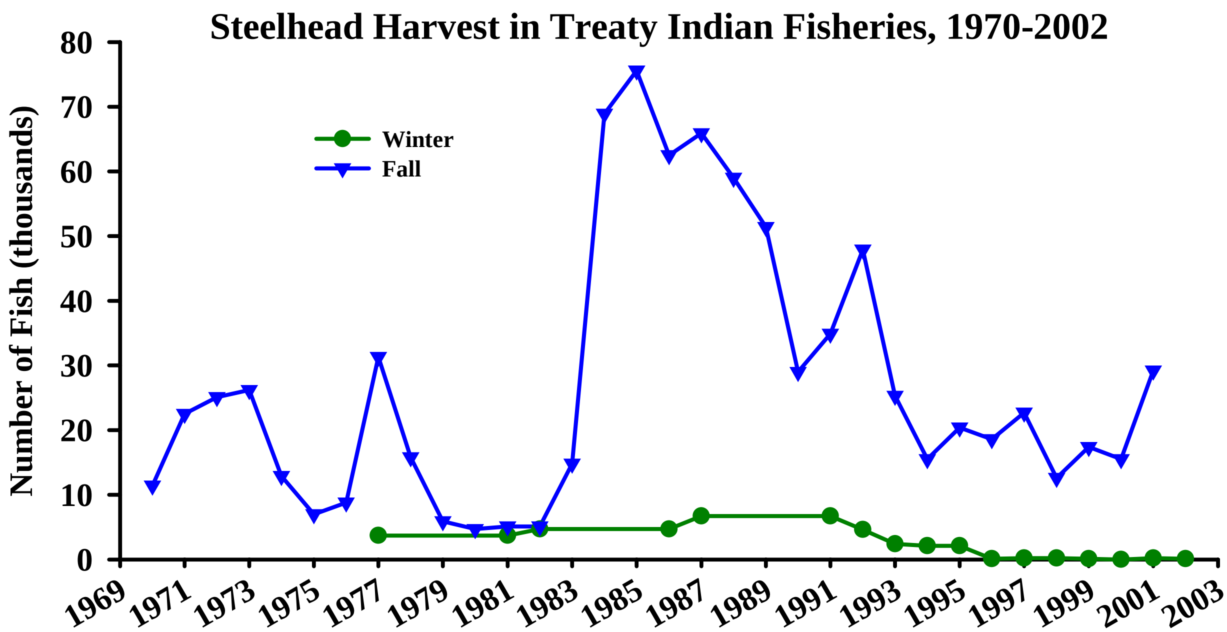

Figure 3-57. Steelhead harvest in treaty Indian fisheries by season, 1970–2002.

Since 1985, the commercial catch has been sampled to determine the percentage of hatchery and wild/natural fish for both Group A and B Index components. Harvest of wild fish in the treaty Indian commercial fishery is compared to the number of wild fish passing Bonneville Dam (Table 3-29) to determine the percentage of the wild runs that are harvested. These data are used to regulate treaty Indian harvest of wild steelhead. Since 1985 the treaty Indian harvest rate has ranged from 2%-21% for Group A wild steelhead and from 11%-37% for Group B wild steelhead. However, recent year harvest rates have been less than 10% for wild Group A steelhead and less than 15% for wild Group B steelhead.

Table 3-29. Wild steelhead catch (in thousands of fish) in treaty Indian fisheries by index group, 1985–2001.

Year	Group A Index		Group B Index	
	Number	**% of Wild Run**	**Number**	**% of Wild Run**
1985	10.8	20.7	4.0	31.0
1986	7.8	13.8	2.7	26.7
1987	16.8	15.7	5.2	37.2
1988	11.0	17.1	4.2	23.4
1989	9.0	15.9	4.3	35.0
1990	4.3	16.0	1.9	21.5
1991	8.8	14.6	1.9	30.0
1992	7.2	16.2	3.3	26.3
1993	4.4	15.2	0.8	19.1
1994	2.2	10.3	1.0	18.6
1995	2.7	10.4	0.3	18.6
1996	2.3	8.9	1.4	34.8
1997	3.2	10.4	0.6	14.3
1998	3.1	8.8	0.5	15.6
1999	4.3	7.6	0.5	12.6
2000	2.3	3.7	1.0	11.4
2001	5.5	4.0	1.4	11.4
2002	2.4	2.0	1.1	3.4

The estimated harvest related mortality of listed steelhead ESUs for 2002 non-Indian fall fisheries is summarized in Table 3-28 and for 2002 treaty Indian fisheries in Table 3-31. This information is developed by the *U.S. v. Oregon* Technical Advisory Committee in an annual Biological Assessment and submitted to NOAA Fisheries for reference when considering fisheries in their Biological Opinion. If fisheries are determined to meet ESA harvest limits they are authorized with an Incidental Take Permit which is delivered to the State and Tribes prior to fisheries being set.

Table 3-30. Harvest related mortality estimates for listed steelhead ESUs in proposed Columbia River basin fisheries[*] during August—December 2002.

Fishery	Upper Columbia Hatchery	Upper Columbia Wild	Snake River Wild	Lower Columbia Wild	Mid Columbia Wild	Total Wild	Total Listed
Mainstem salmonid sport fishery (below Bonneville)	763	15	204	20	179	419	1,182
Mainstem salmonid sport fishery (above Bonneville)	1,899	60	382	72	451	965	2,864
Mainstem commercial salmon/sturgeon fishery	28	3	126	0	35	164	192
Select Areas fall commercial fisheries	0	0	0	0	1	1	1
Wanapum tribal subsistence fishery	17	3	0	0	0	3	17
TOTAL Harvest/Mortality	2,704	81	713	92	665	1,552	4,256
Run Size at Columbia River Mouth	21,771	5,496	75,400	31,068	65,716	177,680	199,451
Harvest/Mortality Rate (%)	*12.4*	*1.5*	*0.9*	*0.3*	*1.0*	*0.9*	*2.1*

[1] Includes only those fisheries that have mortality of listed steelhead.

Table 3-31. Estimated total harvest of steelhead in the 2002 proposed treaty Indian fall fisheries and incidental harvest by ESU.

ESU	Treaty Indian Fall Season Fisheries: Zone 6	Tributaries	Hanford Reach	Total
TOTAL HARVEST	29,150	1,710	100	30,960
Lower Columbia River Steelhead	27	3	0	30
Harvest Rate				**0.96%**
Mid Columbia River Steelhead	1,880	157	0	2,037
Harvest Rate				**3.94%**
Upper Columbia River Steelhead	1,725	0	100	1,825
Harvest Rate				**7.51%**
Snake River Steelhead	5,751	0	0	5,751
Harvest Rate				**7.81%**

3.1.10 *Winter Steelhead Fishery*

Winter steelhead are an important recreational fishery throughout their range. The vast majority of the harvest of winter steelhead occurs in the tributaries of the lower Columbia. In most areas, provisions for separating hatchery and wild fish were not in place until 1987. Since 1987, hatchery steelhead have been marked with an adipose fin clip. Regulations mandating the release of wild fish are in place.

Mainstem Columbia River harvest is typically small and incidental to spring chinook fisheries. Since 2001 spring chinook salmon fisheries in the lower Columbia have been extended due to implementation of selective fishing for spring salmon. The extended commercial and sport fisheries have increased handling of wild winter steelhead compared to the previous 25 years, but the total impact of lower Columbia steelhead listed under the ESA is limited to 2% or less of the annual return.

Generally, steelhead are not caught in commercial or recreational fisheries in the ocean. Although mark and tag data indicate that high seas steelhead distribution and drift net fisheries overlap, ocean harvest is minimal because the ocean migration pattern of most steelhead is seaward of the ocean salmon fisheries. Non-Indian commercial harvest of steelhead in the Columbia River has been prohibited since 1975. Mainstem Columbia sport fisheries have been regulated for selective harvest of adipose fin-marked hatchery fish and have required the release of wild steelhead since 1984. Some Washington tributary winter steelhead recreational fisheries were restricted to wild steelhead release in 1986. The remaining tributary winter steelhead recreational fisheries were restricted to wild steelhead release in 1992, with the exception of the South Fork Toutle, which began wild release regulations in 1994.

Current harvest impacts for wild steelhead populations below Bonneville Dam are associated with release handling mortality in non-Indian commercial fisheries that target salmon and mainstem Columbia and tributary sport fisheries that target hatchery steelhead and salmon. Wind River steelhead harvest impacts include retained harvest in the treaty Indian fishery above Bonneville Dam.

3.1.10.1 Winter Steelhead Harvest Over Time

Historically, steelhead were harvested in Columbia non-Indian winter commercial gillnet fisheries. From 1953-74, steelhead catch ranged from 2,400 to 23,400 (Figure 3-58). Non-Indian commercial steelhead harvest has been prohibited since 1975. Commercial harvest rates were highest when the Columbia River was open 270 days per year (pre-1943). Commercial harvest of wild winter steelhead was lower than summer steelhead, because beginning in 1909, the season was closed from early March to late April. The Columbia and tributary recreational fisheries began increasing in effort and total harvest in the 1960s. After 1975, when non-Indian commercial take of steelhead was prohibited, the harvest impacts of lower Columbia steelhead were almost entirely from recreational fisheries. The treaty Indian commercial fishery became more significant after 1968 following federal court decisions clarifying treaty Indian fishing rights. Most treaty Indian steelhead harvest occurs in September during the fall salmon season.

Steelhead incidental capture and handling occurs in sturgeon and winter salmon fisheries; capture of steelhead is minimized through time, area, and gear restrictions. For example, the lower Columbia winter commercial gill net fishery was restricted to a 7 1/4 in minimum mesh size in 1970 to reduce steelhead handle. In 1975, the minimum mesh size restriction was increased to 8 in, concurrent with the prohibition of non-Indian commercial steelhead harvest. From 1975-90, a seasonal average of less than 500 steelhead were handled annually as a result of incidental capture in winter fisheries. Monitoring data for the same period indicates that steelhead immediate mortality from handling was about 17%. Monitoring in the 1990s by the Marine Mammal Observer Program suggests that steelhead immediate mortality from handling may be lower than 17%.

Limited numbers of winter steelhead are harvested annually in the treaty Indian winter commercial fishery (Figure 3-58). Most harvest likely occurs in Bonneville Pool. The winter treaty Indian fishery targets sturgeon. The 2002 winter commercial gill net landings totaled 78 steelhead; all steelhead were caught in the Bonneville Pool.

Winter steelhead annual recreational harvest in the lower Columbia River and tributaries has exceeded commercial harvest since the 1950s (Figure 3-58).

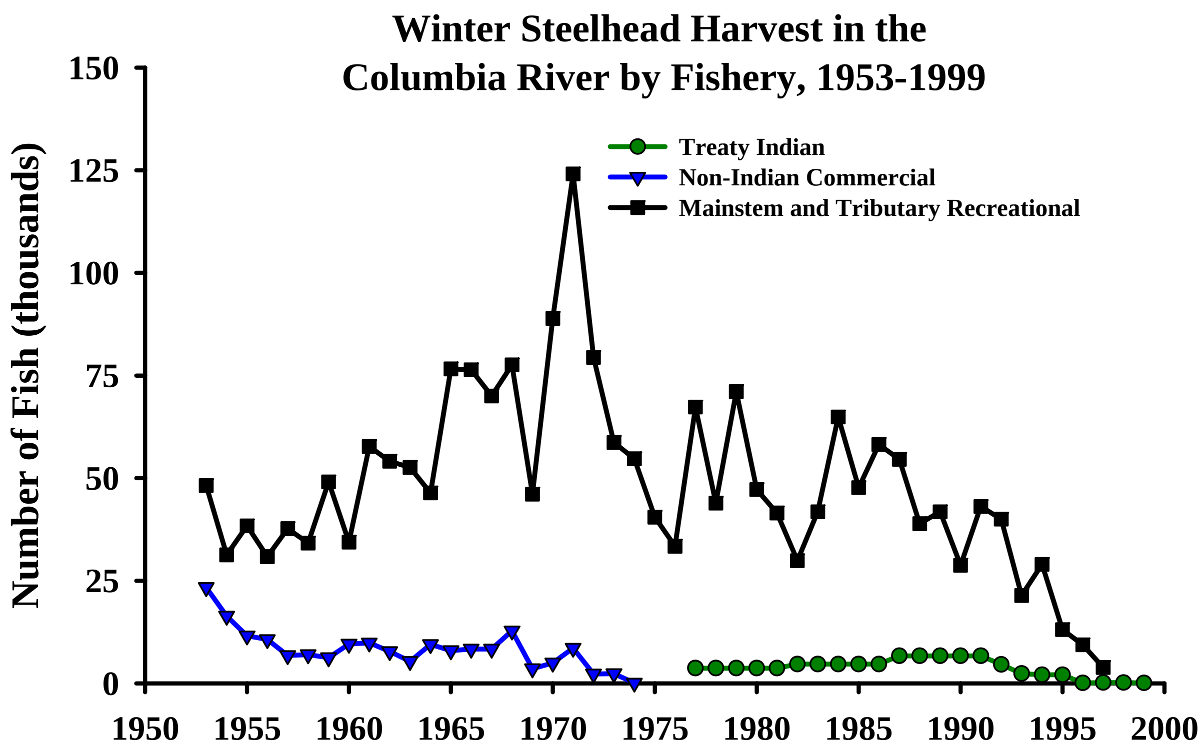

Figure 3-58. Harvest of winter steelhead in the Columbia River from 1953–99.

Steelhead incidental capture and handling occurs in sturgeon and winter salmon fisheries; capture of steelhead is minimized through time, area, and gear restrictions. From 1975-1990, a seasonal average of less than 500 steelhead annually were handled as a result of incidental capture in winter fisheries. Monitoring data for the same period indicates that steelhead immediate mortality from handling was about 17%. Monitoring in the 1990s by the Marine Mammal Observer Program suggests that steelhead immediate mortality from handling may be lower than 17%. Recent year (2002-2003) handling of winter steelhead has increased in the winter/spring commercial fishery due to gear and seasonal structure changes associated with selective spring chinook salmon fisheries.

Lower Columbia River steelhead were listed as threatened under the federal ESA in 1998. This ESU includes all naturally spawned summer and winter steelhead in the Columbia River basin and tributaries between the Cowlitz and Wind rivers (inclusive) in Washington and the Willamette and Hood rivers (inclusive) in Oregon, excluding steelhead in the upper Willamette River above Willamette Falls.

3.1.10.2 Current Winter Steelhead Harvest

Winter steelhead enter the Columbia River from November to May; the hatchery run peaks from December to January and the wild run peaks from March to April. Winter steelhead are destined primarily for tributaries below Bonneville Dam; a few Bonneville Pool tributaries support winter steelhead runs.

Winter steelhead sport fisheries occur primarily in Columbia River tributaries. Hatchery-only harvest restrictions on mainstem Columbia River sport fisheries have been in effect since 1984 to protect wild steelhead. Release of all wild steelhead in recreational fisheries is now required basin-wide.

Limited numbers of winter steelhead are harvested annually in the treaty Indian winter commercial fishery. Most harvest likely occurs in Bonneville Pool. The winter treaty Indian fishery targets sturgeon.

Non-Indian commercial handling of wild winter steelhead occurs during winter and spring salmon seasons. Prior to 2002, large mesh gill nets were used to target spring chinook and sturgeon while minimizing steelhead handle. In 2002, a live capture spring chinook commercial tangle net fishery was established and resulted in a significant steelhead handle in smaller 5.5-in mesh gill-net gear. WDFW and ODFW are continuing to experiment with season structure and gear to avoid excessive impacts to winter steelhead during live capture spring salmon seasons. treaty Indian fishery impacts to wild steelhead are limited to a maximum of 15 % according to a *US v. Oregon* Fall Management Agreement and ESA requirements.

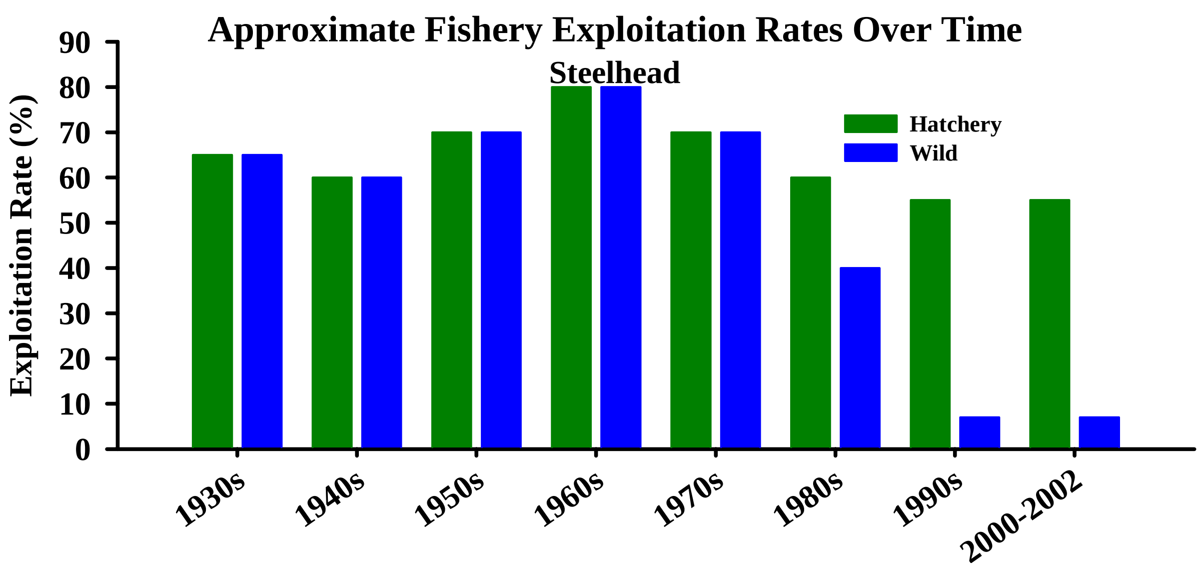

Figure 3-59. Steelhead fishery exploitation rate over time. Columbia commercial harvest significant until prohibited in 1975. Popular sport fish in mainstem Columbia and tributaries with significant catch since 1950s. Selective fisheries implemented for summer steelhead1984-92.

3.1.10.3 Winter Steelhead Harvest Management Details

Winter steelhead enter the Columbia River from November to May; the hatchery run peaks from December to January and the wild run peaks from March to April. Winter steelhead are destined primarily for tributaries below Bonneville Dam; a few Bonneville Pool tributaries support winter steelhead runs.

Recent year (2002-2003) handling of winter steelhead has increased in the winter/spring commercial fishery because of gear and seasonal structure changes associated with selective spring chinook salmon fisheries. The developing tangle net commercial fishery on the lower Columbia River has successfully targeted hatchery spring chinook while releasing wild chinook, with an estimated 10% catch and release mortality on wild spring chinook. Because the impact to wild spring chinook has been substantially reduced as a result of the reduction in gill net mesh size to 5.5 in maximum, the fishery remains open longer, creating more potential opportunities for encounters with wild winter steelhead. The 2002 fishery was open much later in the year (i.e. into late March) than recent commercial seasons and the fishery timing coincided with the early part of the peak of the wild winter steelhead run. While the reduction in mesh size allowed for the release of chinook, steelhead may be more susceptible to mortality resulting from injuries sustained during gill net entanglement.

Preliminary data from the 2002 winter season indicate that the steelhead catch greatly exceeded the preseason catch expectations because of an extremely large 2002 winter steelhead run, fishery timing, and gear employed. A total of 21,600 steelhead were handled during the 2002 fishery, of which 8,640 (40%) were marked and 12,960 (60%) were unmarked. It is possible that some steelhead were handled more than once. Preliminary monitoring results indicate that the immediate mortality rate for steelhead was 2.0%, which results in an immediate mortality estimate of 260 unmarked steelhead. A gear components study was conducted using test fishery and tangle net fishery data from 2000—02 to assess the effect of different mesh sizes on steelhead and chinook condition and mortality after gear entanglement. Immediate and total mortality of steelhead was lowest with 5- or 6-in mesh, although, these mesh sizes also had the lowest sample size in the study (Table 3-32). The 5.5-in mesh gill net had the lowest percentage of steelhead categorized as condition 1 (vigorous, not bleeding) after capture; the highest percentage of condition 1 steelhead after capture occurred with the 5-in mesh gill net (Table 3-32).

For the 2003 winter season, the Columbia River Compact considered regulations requiring large mesh size early in the season to limit steelhead handle and small mesh size (less then 4.25-in) later in the season to promote high survival rates of released steelhead. Other considerations to reduce steelhead handle are reduced fishery effort during peak abundance time of winter steelhead and the use of large (>12 in) mesh excluder panels on the top 5-10 feet of the net.

Table 3-32. Summary of catch rate, condition, and survival for steelhead captured in various spring fisheries, 2000–02[a].

Mesh Size	3.5[b]	4.0[c]	5.0[d]	5.5[c]	6.0[d]	SE[d]
Sample Size	105	93	7	45	13	9
CPUE[e]	0.44	0.52	0.34	0.75	0.56	0.40
Immediate Mortality[f]	15 (14.3%)	12 (12.9%)	0 (0%)	9 (20.0%)	1 (7.8%)	0 (0%)
Total Mortality[g]	NA	4 of 14 (28.6%)	0 (0%)	6 of 15 (40.0%)	2 of 13 (15.4%)	2 of 9 (22.2%)
Capture Condition[h]	1: 63% 2: 3% 3: 16% 4: 0% 5: 18%	1: 53% 2: 3% 3: 23% 4: 4% 5: 18%	1: 100% 2: 0% 3: 0% 4: 0% 5: 0%	1: 39% 2: 14% 3: 20% 4: 7% 5: 20%	1: 50% 2: 0% 3: 33% 4: 8% 5: 8%	1: 40% 2: 20% 3: 30% 4: 0% 5: 10%
Method of Capture[i]	Tooth tangle to max	Max to opercle	Max to opercle	Opercle to wedge	Wedge	Opercle to wedge

[a] The information used in this table is pooled from various test and experimental fisheries conducted over three years. Many factors varied among the studies, including study protocol, personnel, and data collected.
[b] CPUE and immediate mortality data from 2000 & 2001 test fishing and 2001 permit fishery.
[c] CPUE and immediate mortality data from 2001 permit fishery and 2002 test fishing; total mortality from 2002 test fishing.
[d] All data from 2002 test fishery.
[e] CPUE standardized to 150-fathom net length; depth was not standardized and drift times and methods vary among studies.
[f] Defined as fish that could not be recovered thus died on-board a vessel. (Note that data for 3.5- and 4.5-in mesh includes 3:1 hang ratios, which appears to cause excessive tangling and increased mortality.)
[g] Data from 2002 test fishery; defined as total mortality after 48 hours and includes immediate mortality.
[h] Standard condition ranking scale. Data for 3.5-in mesh from 2001 permit fishery and 2001 test fishery and 4.5-and 5.5-in mesh from 2001 permit fishery and 2002 test fishery.
[i] Data from 2002 test fishery and general observations.

Winter steelhead are native to all major and most minor basins in the lower Columbia River. Steelhead in tributaries downstream of the Cowlitz River are considered part of the SW Washington ESU and are not listed under the ESA. Winter steelhead sport fisheries occur mostly in Columbia River tributaries; fisheries are primarily in the Grays, Elochoman (both in the SW Washington ESU), Cowlitz, Toutle, Coweeman, Kalama, Lewis, Salmon, and Washougal basins.

Fisheries targeting winter steelhead are concentrated from December to February and close by March 15, except in the Cowlitz, Kalama, Lewis, and Washougal basins where winter steelhead fisheries extend through May 31. The closed periods in the tributaries are set to protect wild spawning steelhead. Winter steelhead are also taken incidentally in spring chinook targeted fisheries from February—May; however, the interception rate for non-targeted species is expected to be 1% or less (WDFW 2001). Hatchery-only harvest restrictions on mainstem Columbia River sport fisheries have been in effect since 1984 to protect wild steelhead. In Washington, some tributary winter steelhead fisheries adopted wild steelhead release regulations in 1986; the remaining tributary winter steelhead fisheries adopted wild steelhead release regulations in 1992, except for the South Fork Toutle which went to wild steelhead release in 1994. Release of all wild steelhead in recreational fisheries is now required basin-wide. WDFW recreational steelhead selective fisheries are managed to achieve a maximum 10% steelhead mortality for winter steelhead populations both above and below Bonneville Dam.

The estimated encounter and take of wild winter steelhead in Washington tributary recreational fisheries is summarized in Table 3-33 (WDFW 2003).

Table 3-33. Estimated take of ESA-listed steelhead in Washington tributary recreational fisheries.

Affected Stock	Anticipated Encounters[a]	Expected Mortality[b]
Coweeman River winter steelhead	30%	1%
SF Toutle River winter steelhead	38%	2%
Cowlitz River winter steelhead	70%	4%
Kalama River winter steelhead	70%	4%
Mainstem/NF Lewis River winter steelhead	70%	4%
EF Lewis River winter steelhead	40%	2%
Washougal River winter steelhead	40%	2%
Wind River winter steelhead	30%	1%
Salmon Creek winter steelhead	30%	1%
Other tributaries winter steelhead	30%	1%

a Anticipated encounters are catch and released fish; the numbers represent the percentage of fish from a stock anticipated to be incidentally encountered by anglers of a particular fishery.

b Expected mortality is the hooking mortality of incidentally caught fish; expected mortalities are included in the anticipated encounters in terms of take.

3.1.11 Bull Trout Fishery

Sport fishing for bull trout was eliminated in the Lewis and White Salmon drainages in 1992. Hooking mortality may occur from catch and release of bull trout in fisheries targeting other fish, particularly the coho and kokanee fisheries in Merwin and Yale reservoirs (WDFW 1998). Incidental catch of bull trout is thought to be low, however. In the Lewis River system, incidental take of bull trout is thought to be higher above Swift Reservoir (WDFW 1998). WDFW has actively set fishery regulations to protect bull trout in reservoirs and tributaries in the Lewis River basin.

3.1.12 Sea Run Cutthroat Fishery

There is no direct commercial harvest of coastal cutthroat trout, and gill-net mesh size is too large for much incidental handle of cutthroat in the fishery. Angler harvest of coastal cutthroat trout has declined significantly since the implementation of more restrictive sport regulations in 1985 aimed at protecting wild anadromous salmonids (Figure 3-60). Tributaries in all subbasins in the lower Columbia region are closed to retention of wild (unmarked) cutthroat. Open fishing periods differ from subbasin to subbasin but many have spring closures to protect spawning cutthroat and steelhead. Sport catch of cutthroat trout is open year-round in some reservoirs. Hooking mortality does occur, particularly during steelhead/salmon seasons, but the extent of wild cutthroat mortality from hooking and illegal harvest is believed to be low (WDFW 2000).

In 1985, the daily bag limit on the Columbia River was reduced from 8 to 2 trout with a 12 in (30 cm) minimum size (subsequently raised to 14 in [36 cm]). The change was aimed at allowing most female cutthroat to spawn at least once before harvest. In 1992, wild cutthroat release regulations were implemented basin-wide.

Columbia River anadromous coastal cutthroat are not harvested in Pacific Ocean or Columbia River commercial fisheries, and incidental impacts from these fisheries is negligible. Directed harvest only occurs in sport fisheries, which selectively harvest hatchery sea-run cutthroat produced from the Cowlitz Trout Hatchery. The sport fishery harvest of hatchery cutthroat occurs primarily in mainstem Columbia and Cowlitz River bank fisheries. Hatchery cutthroat programs at Beaver Creek in the Elochoman River, Merwin Hatchery in the Lewis River, and Skamania Hatchery in the Washougal River were terminated in recent years due to budget shortfalls and poor return rates of the hatchery produced cutthroat trout.

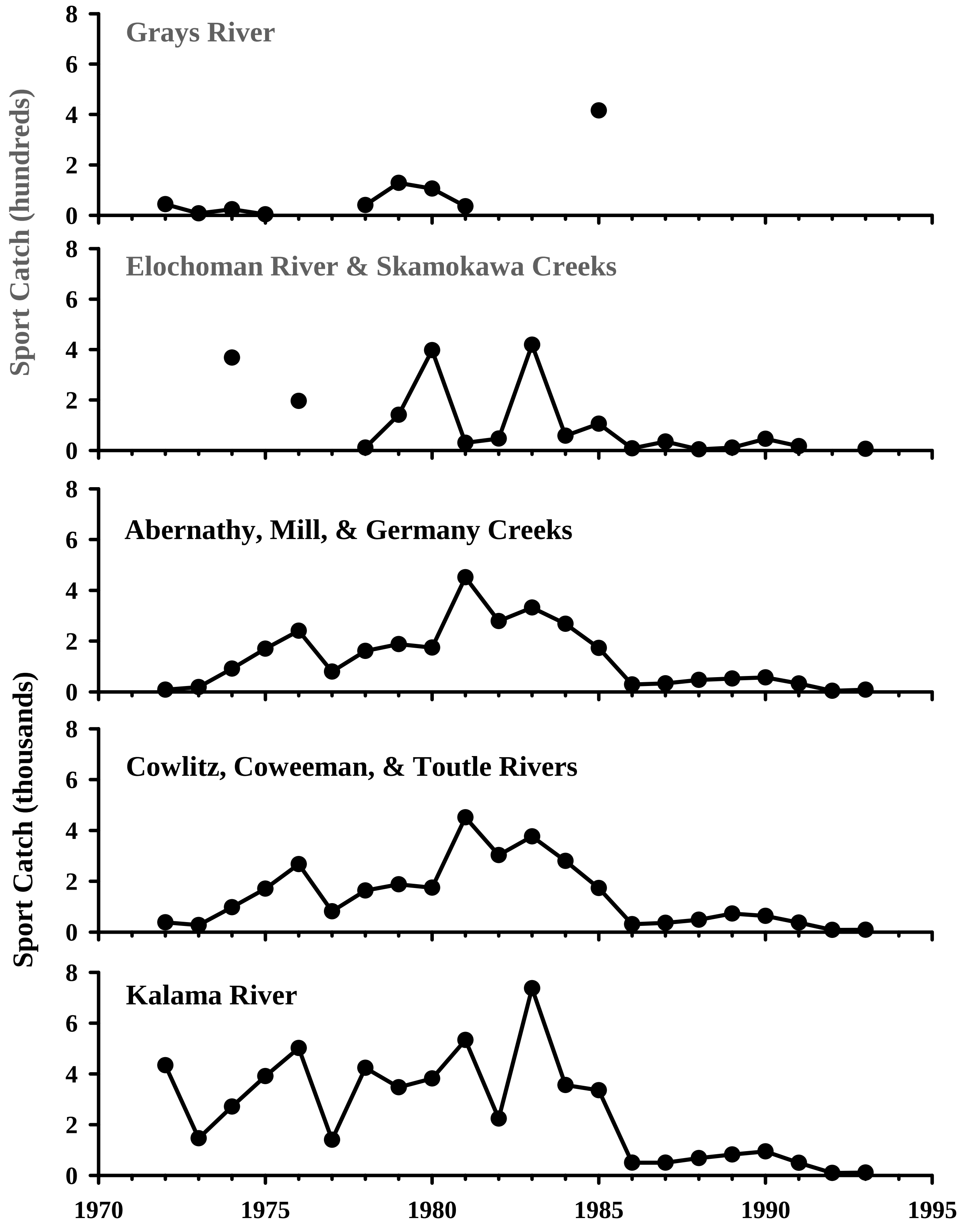

Figure 3-60. Sport angler harvest of coastal cutthroat trout.

3.1.12.1 Cutthroat Trout Harvest Over Time

Harvest of cutthroat trout in lower Columbia Washington tributaries averaged about 7,000 per year until 1985 when more restrictive regulations were enacted to protect wild trout and juvenile salmonids. After 1985, tributary harvest was reduced to only a few hundred per year (Figure 3-61).

Columbia River mainstem sport catch of cutthroat in the 1970s was significant, with estimated harvest typically in the range of 4-10,000 per year. Harvest steadily decreased in the 1980s, and the annual catch dropped to a low of 500 fish in 1987 (Figure 3-62). Wild cutthroat release regulations were enacted in the Columbia River in 1994.

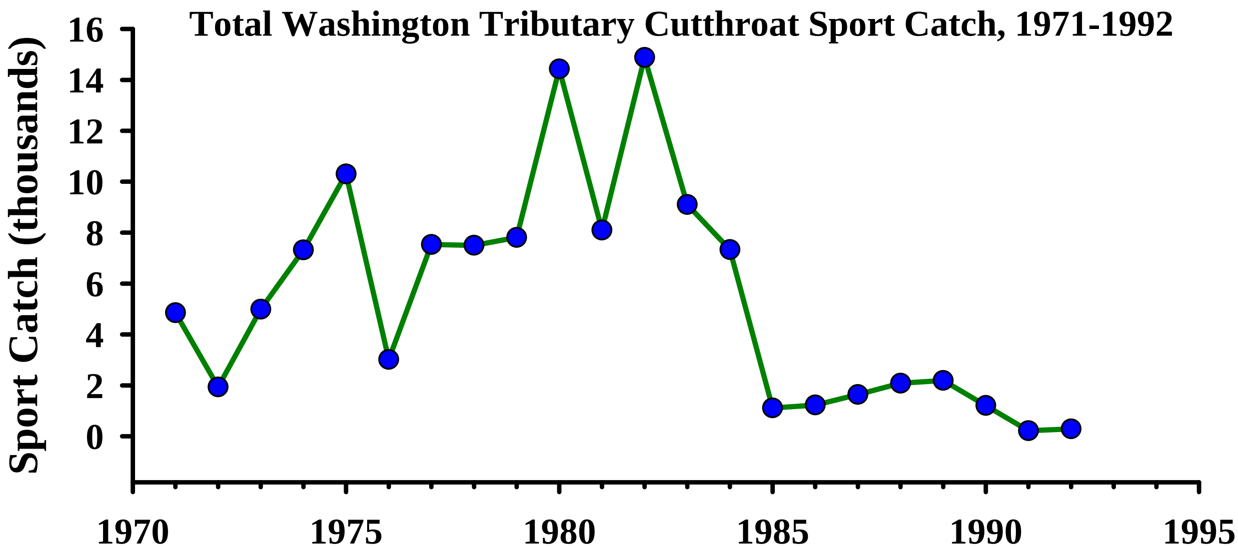

Figure 3-61. Total sport catch of coastal cutthroat trout in lower Columbia Washington tributaries. Data used to generate this figure are incidental to data collected during salmon and steelhead studies. No data for the Lewis River or tributaries from the mouth of the Lewis to Bonneville Dam is available.

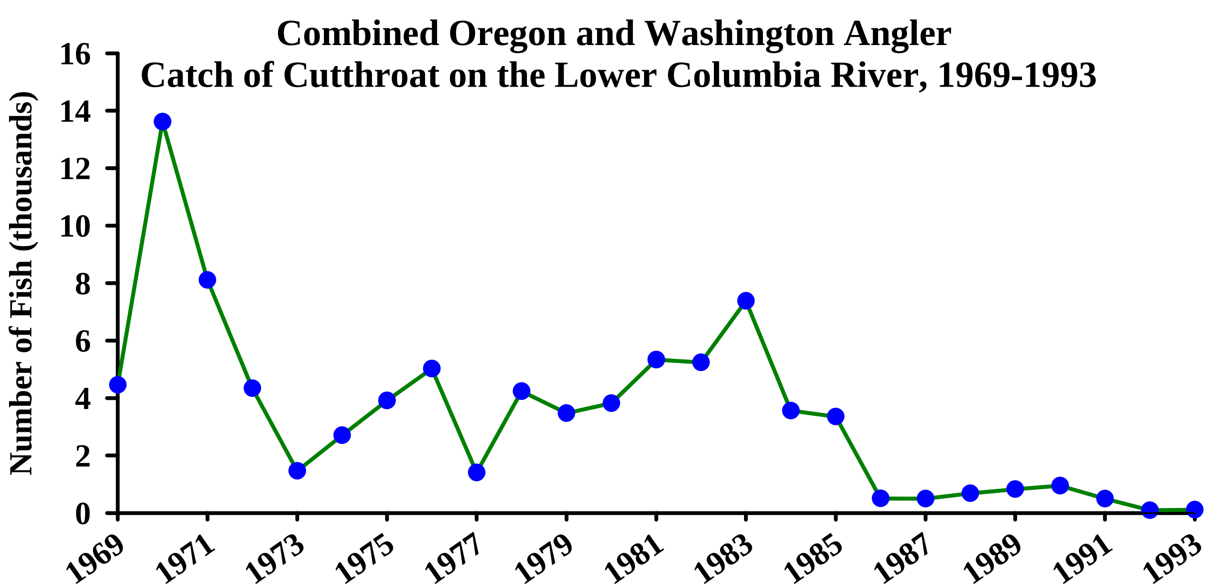

Figure 3-62. Catch of cutthroat trout on the Lower Columbia River, 1969–93.

3.1.12.2 Current Cutthroat Trout Harvest

Hatchery cutthroat trout continue to be harvested in the mainstem Columbia River and Washington (Cowlitz) tributary sport fisheries. Wild release and minimum size regulations have reduced impacts to wild cutthroat trout significantly. The Columbia River harvest of cutthroat has averaged less than 100 fish annually since wild cutthroat release regulations were implemented in 1994 (Figure 3-63).

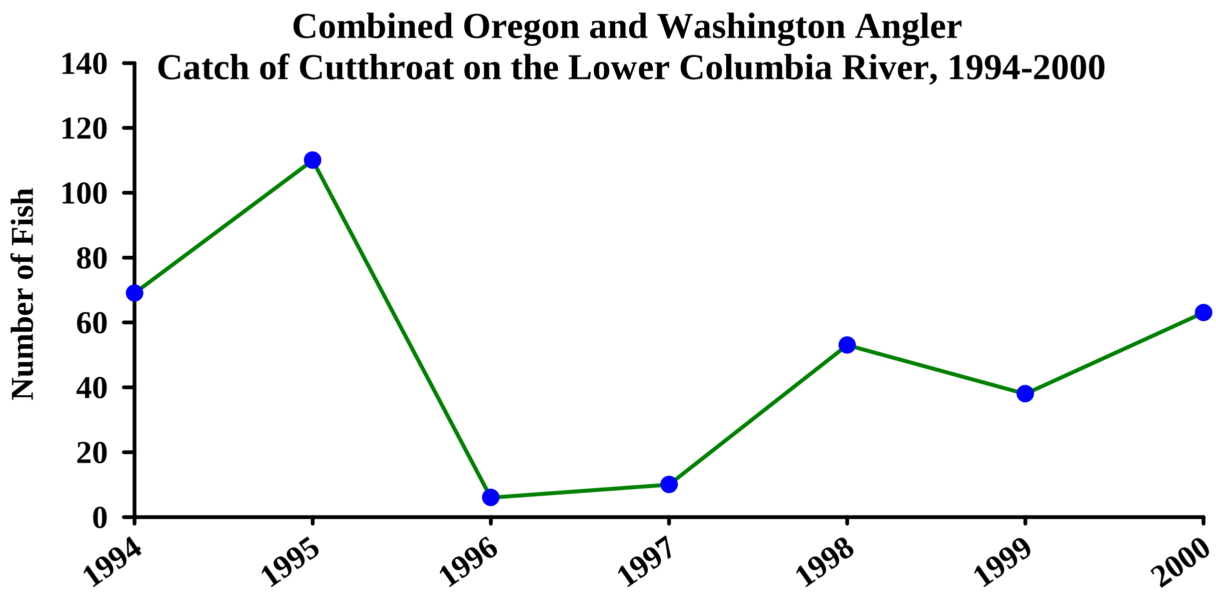

Figure 3-63. Catch of cutthroat trout on the lower Columbia River, 1994—2000.

3.1.12.3 Cutthroat Trout Harvest Management Details

Washington biologists introduced a fishery management strategy to meet conservation needs of resident and anadromous trout populations in 1985. Minimum size and bag limits were made different for lakes and streams to provide maximum angler access to trout planted from hatcheries into standing waters and in reservoirs above dams, and increase protection of wild trout and juvenile salmonids in streams.

Trout fisheries have the potential to impact juvenile salmonids and anadromous cutthroat. However, WDFW has implemented time and area, bag limit, and size restrictions to reduce impacts. The general trout season is June 1–October 31, which maintains a closed season during the peak smolt out-migration period in the spring. The minimum legal size in small streams is generally 8 inches (20 mm) to protect smolts and 12-14 inches (30-35 mm) in larger streams to protect anadromous cutthroat trout and larger steelhead smolts. The bag limit is two legal-sized trout. An important management objective for sea-run cutthroat is to allow most to spawn at least once prior to being subjected to harvest. Most first-year spawning cutthroat are less than 12 inches (30 mm), and virtually all first-year spawners are less than 14 inches (35 mm).

During 1992-94, wild cutthroat release regulations were enacted in all Washington tributaries where anadromous cutthroat are present, and in the mainstem Columbia River. The minimum size of 12-14 inches (30-35 mm) is still important for wild sea run cutthroat protection even though all wild fish must be released. The minimum size rule assures that smaller cutthroat are still protected from retention by anglers that have not accurately identified the species.

Selective gear rules are imposed in some areas, usually upper watershed streams, to promote catch and release fisheries where fish populations are depressed. These restrictions allow only the use of unscented artificial flies or lures with one single barbless hook, and prohibit the use of bait. Most areas where anadromous cutthroat are present allow bait to be used. Hooking mortality is presumed to be higher with use of bait and was estimated at 6-8% for sea-run cutthroat caught with worm-baited hooks in the Samish River. Trout hooked with bait and released must be counted towards the daily bag limit in Washington.

As an illustration of rules to protect sea run cutthroat, Table 3-34 represents the trout regulations in most Washington streams in the lower Columbia. This table represents only open seasons, minimum size, bag limit, and special rules for 2003-04. It does not represent all of the specific regulatory requirements for fishing in these streams, including specific open and closed areas. Complete fishing regulation information is contained in the Washington Sport Fishing Rules Pamphlet, 2003/2004.

Table 3-34. Trout fishing regulations in lower Columbia tributaries in Washington.

Stream	Season	Min. Size	Bag Limit	Special Rules
Deep River	Year-round	14"	2	Release wild cutthroat
West Fork Grays	June 1–Aug. 31	8"	2	None
East Fork Grays	June 1–Oct. 31	14"	2	Release wild cutthroat Selective gear rules
Elochoman River	June 1–Mar. 15	14"	2	Release wild cutthroat
Mill Creek Abernathy Creek Germany Creek	June 1–Aug. 31 Nov. 1–Mar. 15	14"	2	Release wild cutthroat
Coal Creek	June 1–Aug 31 Nov.1–Feb 29	14"	2	Release wild cutthroat
Coweeman River	June 1–Mar. 15	12"	2	Release wild cutthroat
Lower Cowlitz	June 1–Mar. 31	12"	5 (2 > 20")	Release wild cutthroat
Upper Cowlitz (Clear Fork & Muddy Fork)	June 1–Oct. 31	8"	2	Release cutthroat
Tilton River	June 1–Mar. 31	8"	5 (1 > 12")	None
EF, NF, SF, WF Tilton River	June 1–Oct. 31	12"	2	Selective gear rules
Cispus River	June 1–Oct. 31	8"	2	Release cutthroat
NF Cispus River	June 1–Oct. 31	8"	2 (1 > 12")	Release cutthroat
Lower Kalama	Year-round	20"	2	Release wild cutthroat
Mid Kalama	Year-round	14"	2	Release wild cutthroat
Upper Kalama	June 1–Mar. 31	14"	2	Release wild cutthroat
EF Lewis	June 1–Mar. 15	—	—	Catch and release
Lower NF Lewis	Year-round	20"	2	Release wild cutthroat
Upper NF Lewis	June 1–Oct. 31	—	—	Catch and release
Cedar Creek	June 1–Mar. 15	12"	2	Release wild cutthroat
Cougar Creek	June 1–Aug. 31	8"	2	--
Salmon Creek	June 1–Mar. 15	12"	2	Release wild cutthroat
Washougal River	June 1–Mar. 15	—	—	Catch and release
Hamilton Creek	June 1–Oct. 31	12"	2	Release wild cutthroat
Wind River	July 1–Mar. 15	14"	2	None
Little White Salmon River	June 1–Oct. 31	8"	2	None

3.2 Hatcheries

There are 20 salmon and steelhead production hatcheries in the lower Columbia basin (Figure 3-64) as well as a number of associated rearing facilities and acclimation sites. These hatcheries have played a major role in producing salmon for harvest. Fisheries managers and the public are attempting to find the balance between hatchery facilities that can; 1) produce excess fish for harvest, 2) augment natural production, 3) help to rebuild depleted wild populations, and/or 4) serve as conservation banks for severely reduced populations, all while minimizing impacts on natural production. The long history of hatcheries in the lower Columbia, and their associated effects on wild fish, cannot be erased simply by closing all hatcheries. To do so would eliminate important hatchery-based fisheries and some key natural production, especially tule fall chinook and coho, now largely supported by hatchery augmentation. Rather, modifying hatchery programs so they support an integrated, comprehensive approach to rehabilitating depleted populations, and providing fish for harvest while minimizing impacts to wild fish, should be the goal for hatcheries into the future (NRC 1996).

To set the stage for a discussion of hatcheries and their role in past, present, and future lower Columbia salmon production and restoration, requires some basic definitions of the various types of hatchery programs. These range on a continuum from major production facilities to small genetic conservation programs and can be organized according to the programs' history and purpose. Multiple programs with different or complimentary purposes may be found at a single facility.

Production hatcheries are used primarily to rear and release large numbers of fish that support fisheries. These are usually characterized by large physical plants and may incorporate satellite rearing and acclimation facilities. Many production hatcheries were originally constructed to mitigate for lost habitat upstream of dams.

Augmentation programs are usually more closely tied to local natural production but are primarily oriented to producing fish for harvest (Kapuscinski 1997). In most cases, the differences between the hatchery and natural fish are difficult to discern and natural reproduction is largely supported by hatchery fish. These programs are often associated with large production hatcheries and incorporate satellite rearing and acclimation facilities.

Supplementation programs use artificial propagation in an attempt to maintain or increase natural production, while maintaining the long-term fitness of the target population and keeping the ecological and genetic impacts on non-target populations within specified biological limits (RASP 1992).

Conservation hatcheries use artificial propagation techniques to maintain populations when they are at critically low numbers. They may include the use of captive broodstock but ultimately are aimed at rebuilding wild populations through supplementation strategies (Waples et al. 1991). There are currently no true conservation hatchery programs in the lower Columbia planning area.

This hatchery section first describes detrimental effects that hatchery programs can potentially have on natural fish populations. This section is intended to illustrate the types of potential risks associated with hatchery operations in general and describe the specific lower Columbia basin hatchery programs in the context of those risk factors, including magnitude and time of hatchery fish released by species, adult returns of hatchery and natural fish, genetics, hatchery/wild interaction potential, the effects of water quality and diseases, passage problems, mixed harvest potential, and programs to supplement wild fish. The section is not intended, however, to quantify the risks to natural fish populations nor reach conclusions concerning presence or absence of risk factors in particular hatchery programs in the lower Columbia basin. Rather, it provides perspective on the overall importance of hatcheries in the lower Columbia as well as specific details on individual programs that can be used, during development of the Management Plan, in formulating risk assessments for impacted natural fish populations and the risks to fisheries and fisheries agreements as a result of potential adjustments to present hatchery programs.

3.2.1 Lower Columbia Basin Hatchery Operations

Throughout the twentieth century, the primary purpose for construction of lower Columbia basin production hatcheries was to enhance fisheries and to mitigate for reduced ability of the habitat to produce natural fish at historical levels (Lichatowich 1999). Almost all hatchery program production of salmon and steelhead in the lower Columbia basin is funded by federal monies as mitigation for fishery losses associated with the development of mainstem Columbia River federal dams, or from licensed operators of the tributary dams in the Cowlitz and Lewis rivers (Radtke and Davis 2000). As efforts move forward to restore those same natural populations that the hatchery programs were intended to replace, hatchery programs will continue to be evaluated for compatibility with natural populations (ISAB 2003). As wild population rebuilding unfolds, however, the objective to maintain adequate salmon and steelhead hatchery production to support fisheries in the lower Columbia should not be dismissed.

The balance of hatchery and natural fish is currently dominated by hatchery fish as was expected when the hatchery mitigation programs were developed. For perspective on the role of Columbia River hatchery fish, by 1987, hatchery-origin fish dominated returns: 95% of coho, 70% of spring chinook, 80% of summer chinook, 50% of fall chinook, and 70% of steelhead were produced by hatcheries (CBFWA 1990, cited in NRC 1996). As natural population recovery is implemented, the fish balance should begin to swing back towards natural production over time, although the rate and magnitude of the swing will depend on the relative success in rebuilding natural populations, with consideration given to total adult production and the public's demand for harvest opportunities, now principally provided by hatchery production.

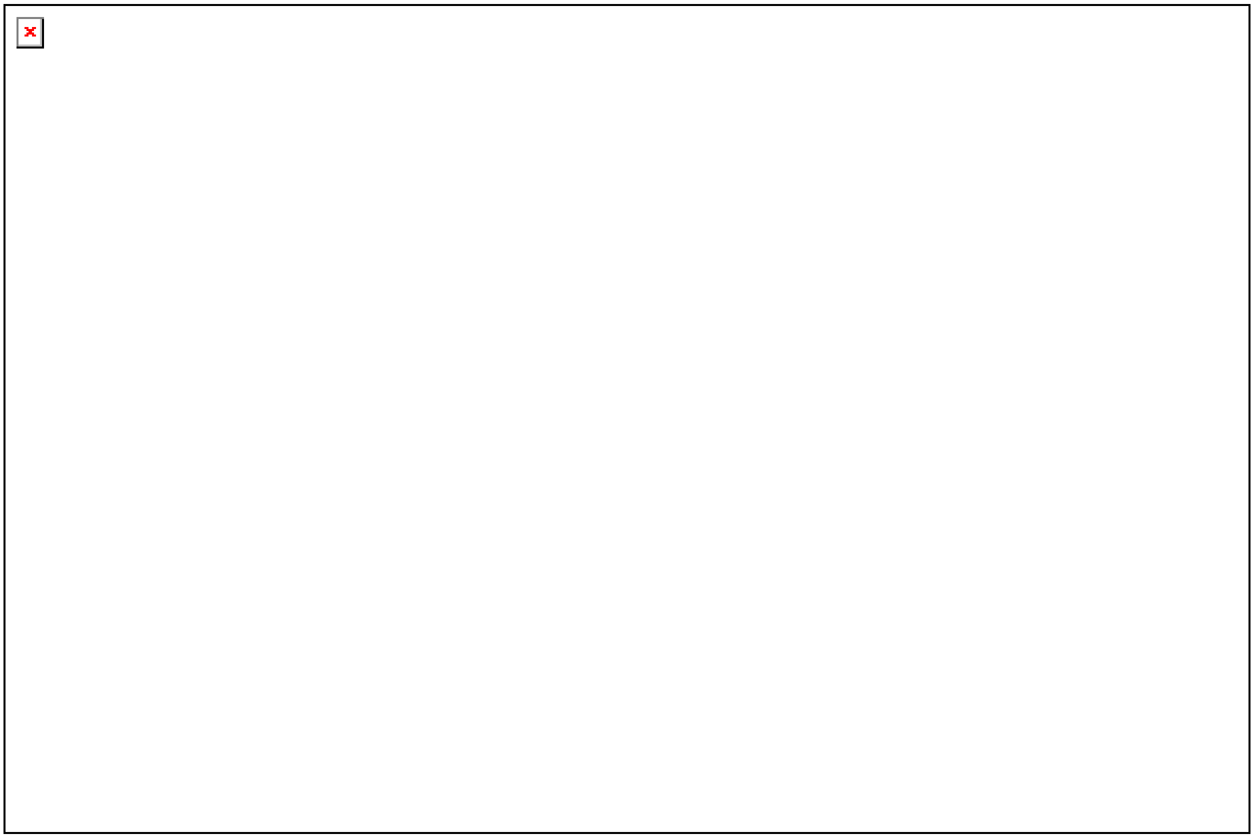

Figure 3-64. Lower Columbia production fish hatcheries and beginning dates of operation.

Hatchery production in the lower Columbia River watershed began in the late 1800s. The first Washington hatchery was built on Baker’s Bay near the mouth of the Columbia River in 1894 (Wahle and Smith 1979). Soon after, state and federal hatchery operations began to enhance commercial fisheries; by the 1890s, many hatchery and egg-take stations were operating between the Chinook River (near the Columbia River mouth) and the Little Spokane River (upper basin).

In 1895, the first state-operated hatchery in Washington was built on the Lower Kalama River and is still in operation. The first federal chinook salmon hatchery on the lower Columbia River was built on the Little White Salmon River in 1897 (Nelson and Bodle 1990). Hatchery production exploded during the early 1900s. By 1905, approximately 62 million fry were released annually.

Throughout the 1900s, the negative effects of agricultural development, timber activities, and other land use practices, and the development of the Columbia River dam complex increased the need to mitigate for reduced natural production. Artificial production appeared to be the only means available to fishery managers to compensate for fish losses and the resulting decline in fish available for harvest.

The first half of the twentieth century witnessed an explosive increase of hatcheries and hatchery production. From 1913 to 1930, about 320 million chinook salmon fry were released into the lower Columbia River by Washington state hatcheries alone; similar production numbers are estimated for Oregon and federal hatchery efforts. Hatchery operations dropped during the

Great Depression and were temporarily interrupted during World War II, and production declined to one-tenth of that seen during pre-war years at Washington state hatcheries.

In response to the construction of Bonneville and Grand Coulee dams, Congress passed the Mitchell Act in 1938, which required the construction of hatcheries to compensate for fish losses caused by the dams as well as by logging and pollution. A 1946 amendment to the Mitchell Act led to the development of the Lower Columbia River Fishery Development Plan, which initiated the major phase of hatchery construction in the Columbia River basin. The plan was later expanded to include the upper Columbia River and the Snake River.

Although most of the lost natural salmonid production was located in the upper Columbia and Snake River basins, only four of the 39 propagation facilities authorized by the Mitchell Act were constructed above The Dalles Dam in the mid-Columbia River. Facilities were not constructed in the upper basin because of concerns with the ability of fish to bypass dams in the upper watershed and because the primary goal of the program was to provide fish for harvest in the ocean and lower river fisheries (Myers et al. 1998).

In 1990, total annual hatchery juvenile production (202.5 million) plus estimated wild production (about 145.2 million) equaled about 347.7 million juveniles in the Columbia River, while historical wild juvenile abundance equaled about 264.5 million (Kaczynski and Palmisano 1992). However, the number of juveniles effectively migrating to the lower Columbia and successfully reaching the estuary is likely still less than historical numbers after adjusting for modern-day passage mortality through dam structures and post-release mortality suffered by the hatchery fish.

Hatchery programs in the lower Columbia basin have included all salmonids native to the region. (Species-specific hatchery program information is presented in the Program section below.) Salmonids often have been transferred among watersheds, regions, states, and countries, either to initiate or maintain hatchery populations or naturally spawning populations. The transfer of non-native fish into some areas has shifted the genetic profiles of some hatchery and natural populations so that the affected population is genetically more similar to distant hatchery populations than to local populations (Howell et al. 1985, Kostow 1995, Marshall et al. 1995). Until recently, the transfer of hatchery salmon between distant watersheds and facilities was a common practice (Matthews and Waples 1991, WDF et al. 1993, Kostow 1995). However, agencies recently have initiated policies to reduce the exchange of non-indigenous genetic material among watersheds. For example, Washington chinook salmon managers adopted a statewide plan in 1991 to reduce the number of out-of-basin hatchery-to-hatchery transfers. However, the plan did not explicitly prohibit introductions of non-native salmonids into natural populations; rather, the plan included genetic guidelines specifying which transfers between areas were acceptable.

3.2.2 Hatchery Effects

Hatchery programs provide one of the few alternatives for mitigating the large losses of salmon populations, for example, in instances where dams completely block access to salmon spawning areas. However, poorly designed hatchery programs often are detrimental to wild salmon production (Cone and Ridlington 1996, Walters et al. 1988, NRC 1996, Lichatowich 1999). Comprehensive analyses of the impacts of hatcheries on wild salmon involve investigating a variety of effects, many poorly understood.

Hatchery effects on wild fish can be positive and/or negative. Hatchery managers have numerous operational choices (left panel, Figure 3-65) that affect the biology and productivity (center panel, Figure 3-65), and thereby influence the life cycle, of both the hatchery fish and the wild fish with which they interact (right panel, Figure 3-65). Direct and indirect effects and hatchery releases can impact natural stocks in a number of possible ways. The following sections present more detailed information on how hatchery practices can result in life cycle effects on wild populations; the magnitude and actual occurrence of these effects vary among hatcheries and depend on specific operational procedures at individual facilities.

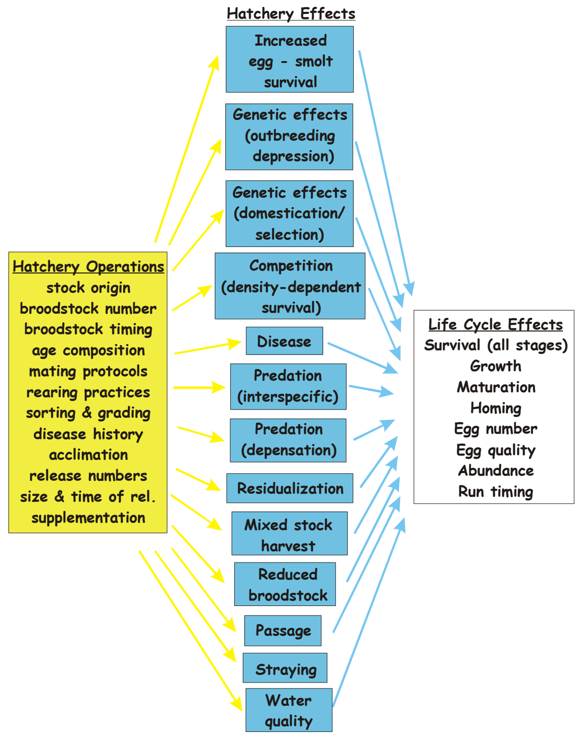

Figure 3-65. Potential links between hatchery operating procedures and effects on salmonids.

3.2.2.1 Increased Egg to Smolt Survival

Hatcheries substantially increase net productivity by increasing egg-to-smolt survival; because hatcheries are able to control the incubation and rearing environment, they usually can achieve higher egg-to-smolt survival than is realized in the natural environment. Because hatcheries allow greater than normal survival, individuals that would have died in the natural environment survive to increase competition, predation, genetic effects, disease proliferation, and mixed stock fisheries effects among each other and their wild counterparts. Hatchery fish have also exhibited reduced fitness and survival per individual compared to wild fish (NRC 1996, Reisenbichler 1997). When hatchery fish stray and spawn in the wild, the fitness of natural offspring populations can likewise be reduced (Waples 1991, Reisenbichler 1997).

On the other hand, because of their ability to produce many offspring from relatively few adults, hatchery programs have been widely considered for supplementation of weak natural runs (Cuenco et al. 1993), although this approach remains controversial (NRC 1996). Conservation hatchery programs are a key component in ongoing attempts to preserve and rebuild several listed Columbia basin salmon stocks (Waples and Do 1994).

3.2.2.2 Genetic Effects

Genetic effects of hatchery practices can influence wild fish populations because hatchery fish become genetically different from local wild fish within a few generations (Resisenbichler 1997). In general, the genetic effects of hatcheries and hatchery fish can be grouped into three major categories (Waples 1991, Krueger and May 1991): 1) the genetic effects of artificial propagation on the hatchery fish, 2) the direct genetic effects of hatchery fish spawning with wild fish in the natural habitat, and 3) the indirect genetic effects of hatchery fish on wild populations due to competition, predation, disease transfer, changes in fishing mortality, or any other factor that affects the abundance or effective population size of the wild population (Campton 1995). Here we will discuss direct genetic effects; the third point is addressed under subsequent headings.

Genetic differences in hatchery fish. –The reasons for genetic differences in hatchery fish are attributable to:

- Taking broodstock from a non-local population,
- Random effects (genetic drift or founder effects) of small hatchery population size,
- Artificial selection by hatchery personnel,
- Increased survival of individuals poorly suited to natural habitat (relaxed selection), and
- Natural selection of fish that are well adapted to hatchery survival (domestication selection) (Reisenbichler 1997).

Loss of genetic variation within a population generally occurs through either genetic drift or selection as listed above. *Genetic drift* is most commonly identified by the loss of infrequent alleles and a resulting increase in homozygosity in small populations. The rate of genetic drift is governed by the effective population size (i.e., the number of spawners that effectively contribute gametes to the next generation), rather than the simple number of fish in the population. The artificial reduction in effective population size is caused by two common hatchery practices: 1) males are often used to fertilize multiple females, and 2) incubation trays that contain the entire spawn of several females, rather than a sampling of all matings. Simon et al. (1986) found that survival from smolt to age 2+ was significantly and positively correlated ($P<0.01$) to effective population size. Waples and Teel (1990) found effective population sizes of chinook salmon in some hatcheries to be less than 100 even when returns were greater than 1,000 fish. The loss of genetic variability to genetic drift has been documented for salmonids (Allendorf and Phelps 1980, Ryman and Stahl 1980; Waples and Teel 1990) and is commonly discussed in hatchery manuals regarding spawner numbers and sex ratios (Hershberger and Iwamoto 1983, Kapuscinski and Jacobsen 1987). New guidelines for hatchery practices issued by state and federal agencies on the West Coast have been designed to eliminate artificial reductions in effective population size.

Selection can be either purposeful or inadvertent, but its consequences are the same in either case. Genetic variability is lost when only a segment of the population, not representative of the whole, is selected for broodstock. This effect appears to have been manifested to some degree in nearly every hatchery population of salmonids in the Columbia basin (e.g., see Cramer et al. 1991 regarding coho hatcheries). Most commonly, it results from the practice of taking eggs from the first fish arriving at the hatchery and then ceasing the egg take once the egg-incubation capacity of the hatchery is reached. Furthermore, because we cannot predict how the entire gene complex of a population will be affected by selection for a specific trait, selection should be avoided where enhancement of natural populations is desired (Krueger et al. 1981). Several studies have demonstrated that selective breeding in hatcheries has reduced viability as a result of the loss of genetic variability (Ryman 1970, Kincaid 1976, Allendorf and Utter 1979, Allendorf and Phelps 1980, Ryman and Stahl 1980).

Domestication selection results from unintentional selection for survival in a hatchery environment (Resienbichler 1997). This selection may result from culling the slow growing fish, from disease treatments, or from the effects of growth differences in the hatchery on survival to maturity. A particular type of domestication selection that is difficult to eliminate relates to how hatchery practices can provide selective advantages to fish that spawn during a specific time of the spawning season. For example, the earliest spawning fish in a hatchery also produce the earliest emergent fry and therefore the largest smolts at release. Numerous studies have demonstrated with every salmonid species that survival to adulthood increases as smolt size at a given time increases. Thus, when a hatchery eliminates the environmental perils of early spawning, a new selective advantage is provided to early spawning fish. Hatchery practices can minimize this selectivity scenario by taking eggs throughout the spawning period and may also control growth by regulating water temperature.

Genetic influence of hatchery fish on natural spawners. − Spawning of hatchery salmonids in the wild with naturally produced fish has the potential to adversely affect genetic characteristics of natural populations (Campton 1995, Reisenbichler 1997). For hatchery fish to have a genetic impact on naturally spawning fish, two conditions must be true: 1) the hatchery fish must be genetically different from the natural fish, and 2) the hatchery and natural fish (or their descendants) must interbreed. The magnitude of genetic impact will depend on the extent to which these two conditions are true (see discussion on straying below).

Three types of genetic risks have been identified which may impact the long-term productivity of wild populations, including:

- Loss of between-population identity or variation,
- Decreases in within-population genetic variation, and
- Decreased fitness (Campton 1995).

The loss of between-population variation or diversity is a primary genetic risk of introducing non-indigenous fish to wild populations. When populations having different genetic profiles interbreed, they may tend toward homogeneity (Campton 1995). For example, populations of wild steelhead on the northwest coast of Washington, where nonnative hatchery steelhead had been extensively stocked since the 1940s, were genetically more homogenous than wild, unstocked steelhead in British Columbia (Reisenbichler and Phelps 1989). Lower Columbia River wild coho salmon are now genetically indistinguishable from hatchery fish stocked for a number of years in large numbers (Flagg et al. 1995). In the long run, this potential loss of diversity weakens the biological resiliency essential to the variable structure required for a healthy salmon ESU.

The loss of within-population variation results when hatchery populations with reduced genetic variation, as described above, spawn naturally with local populations (genetic swamping). The genetic variation of the local populations is subsequently reduced, especially when the number of hatchery fish is large (high stray rates or widespread dispersal of hatchery juveniles). For example, an introduced stock of coho salmon that is substantially different from the native stock might survive at roughly 20% the rate of the native stock, while a similar stock introduced from a nearby stream might survive at roughly 80% of the rate of the native stock (Reisenbichler 1986).

Loss of fitness, as expressed by reduced reproductive success and survival, occurs from the interbreeding of two genetically diverged populations, such as hatchery fish and wild fish, and is referred to as outbreeding depression (Campton 1995). A number of studies have revealed that feral hatchery fish spawning in the wild, either with each other or with wild fish, clearly have reduced reproductive success, lower juvenile growth and survival, and lower marine survival than their wild counterparts (Reisenbichler and McIntyre 1977, Nickelson et al. 1986, Leider et al. 1990). In particular, naturally spawning Skamania stock steelhead introduced into the Kalama River (1- to 2-month differences in time of spawning) were only 28% as successful at producing smolt offspring as the native fish (Chilcote et al. 1986). Survival of wild Kalama steelhead was reduced to 43% of normal when a wild fish mated with a Skamania stock hatchery steelhead (Chilcote et al. 1986). Also, studies with hatchery releases have indicated hatchery fish derived from local populations perform much better in their native environment than do hatchery fish from other populations (Bams 1976, Altukhov and Salmenkova 1986).

3.2.2.3 Population Mixing

Populations can be mixed, and result in genetic and life history effects, through a number of management activities. Obviously, massive releases of smolts from hatcheries and widespread outplanting from production hatcheries have the single most dramatic effect. Hatchery transfers, intentional augmentation and supplementation of natural production, and straying from hatchery programs all contribute to negative impacts on wild populations. The ISAB (2003) concluded that hatchery programs based on hatchery broodstock lines, and which allow the hatchery products to interact intensively with natural populations, almost certainly impose a large cost on the affected natural populations.

Hatchery Transfers

Most hatchery populations have been affected to some degree by transfers between hatcheries to fill egg-take goals years of low return. Examples within the Columbia basin of hatchery populations that have undergone substantial transfers are early-type coho (Cramer et al. 1991) and tule fall chinook. Many hatcheries have been founded with broodstock from other hatcheries. As examples, Skamania steelhead, Carson spring chinook, and Cowlitz coho have been used at a number of hatcheries.

Populations are also mixed when brood fish are taken at a dam where more than one population must pass. For example, the Bonneville upriver bright stock of fall chinook was developed at Bonneville Hatchery by taking their broodstock from bright fall chinook trapped out of the fish ladder at Bonneville Dam. These fish were a mixture of fall chinook that originally spawned throughout the Columbia basin above Bonneville Dam. Similarly, Carson stock spring chinook were developed at Carson National Fish Hatchery by trapping spring chinook at Bonneville Dam as broodstock.

Supplementation

Although the original purpose of most Northwest hatcheries was to provide harvest opportunities in the face of declining salmonid abundance, augmentation and supplementation of natural production have become the focus of some recent salmonid recovery efforts (RASP 1992, Cuenco 1993, ISAB 2003). Augmentation and supplementation are generally aimed at either enhancing existing stocks of anadromous fish or reintroducing stocks formerly present in particular subbasins. Hatchery programs designed to supplement endangered or exploited salmonid populations, like more traditional hatchery programs, can reduce population fitness because the animals are reared under low-mortality conditions that can favor maladaptive traits. The scale of hatchery operations and practices employed in smaller supplementation programs can often be considerably less than those at hatcheries designed to provide for harvest opportunities. However, supplementation programs have similar concerns regarding genetic and ecological effects as other hatchery programs (ISAB 2003). In the extreme case of continual, large-scale augmentation, where the hatchery and natural populations are integrated, the empirical basis is inadequate for determining the cost to the natural population (ISAB 2003). The ISAB (2003) recognized that Columbia Basin supplementation occurs at a number of intentional and unintentional levels:

> "*Most of the hatchery programs are not integrated with natural production because they rely extensively on fish of hatchery origin for their broodstock. Nevertheless, the hatchery productions from these programs are present in large numbers on the breeding grounds of many natural spawning stocks. In some cases this is deliberate, in others it is inadvertent. Either way, this constitutes a supplementation action.*"

Developing and improving supplementation, as well as standard, hatchery programs will continue to be a key component in ongoing attempts to preserve and rebuild listed Columbia basin salmon stocks.

Straying

For hatchery and wild fish to interbreed, they must spawn in the same place at the same time. The degree of genetic mixing and the effects on life history that occurs when hatchery fish are released in a wild population varies dramatically, depending on the ability of the hatchery fish to survive to maturity and on temporal isolation mechanisms. Leider et al. (1986) found that 36% of all wild summer steelhead in the Kalama River mated with hatchery fish, even though spawning by hatchery fish peaked one month earlier than wild fish. The high rate of interbreeding in the Kalama River resulted from the much greater abundance of hatchery fish than wild fish.

Hatchery or fish management practices that lead to straying of hatchery fish at the time of return are key factors governing the risk of reduced diversity and fitness in locally adapted populations. Evidence indicates that straying is more likely among some races of salmon than others. Chapman et al (1991) reviewed the evidence on straying of spring and summer chinook throughout the Columbia basin and found high homing fidelity to nearly every hatchery. However, straying of spring chinook from Lookingglass Hatchery in the Grand Ronde basin into other tributaries of the Grand Ronde River was significant in the 1980s. Quinn and Fresh (1984) documented evidence from Cowlitz River spring chinook that social interaction aids in homing; straying rates increase as spawner abundance declines. To reduce the potential for straying, hatchery programs routinely release hatchery salmon from acclimation ponds to improve homing fidelity. Research by ODFW (2002) on coastal steelhead populations showed that direct stream releases did not increase the potential straying relative to acclimation.

Management practices which may increase the straying rate are:

- Broodstock transfers, mixed broodstock origins

- Releasing hatchery fish close to the mouth of the stream to which adults are intended to return
- off-station releases of fish
- Not acclimating fish prior to releases
- Rearing juveniles in other basins/water sources prior to release

Environmental conditions affecting straying rates include protracted periods of low flow and high water temperatures at the time and place adult fish are targeted to return.

3.2.2.4 Competition

The potential for intra- and inter-specific competition for food or space between hatchery and wild stocks depends on the degree of spatial and temporal overlap in resource demand and supply (Steward and Bjornn 1990, McMichael et al. 2000). The capacity for hatchery fish to significantly alter the behavior and survival of wild fish via competition remains a controversial subject (Steward and Bjornn 1990). There are five areas where competition and crowding may occur between hatchery and natural fish in the migration corridor: in rearing streams, during downstersam migration, in mainstem reservoirs, in the estuary, and in the ocean.

- *Rearing Stream—* Streams in which juvenile salmonids rear have a limited amount of the resources necessary for survival and growth. When hatchery fish are released into streams where wild fish are present, there can be competition for food and space (McMichael et al. 2000). Competition between wild and hatchery individuals is most likely to occur if the fish are of the same species and they share the same habitat and diet. Juvenile salmon establish and defend foraging territories through aggressive contests (Nielsen 1992). When hatchery fish are released into streams where there are wild fish, hatchery fish may be more aggressive, disrupting natural social interactions (Nielsen 1994). Often hatchery-reared individuals may be larger than wild fish in the same stream, and occupy the best feeding territories, placing their wild counterparts at a disadvantage and reducing the number of wild fish in the natural habitat (McMichael et al. 1997). Because carrying capacity of many streams and watersheds has been degraded by contamination, development, logging, and other causes, the effects of competition on wild salmonids may be further exacerbated.
- *Downstream Migration—* Few studies have directly addressed the possibility of density dependent competition during juvenile emigration (Hard 1994). Since salmonid smolts actively feed during their downstream migration (Becker 1973; Muir and Emmett 1988, Sagar and Glova 1988), it is reasonable to conjecture that increased density from hatchery releases could increase competition for wild smolts.
- *Reservoirs—*Salmonid smolts actively feed during normal downstream migration (Becker 1973, Muir and Emmelt 1988, Sagar and Glova 1988). Muir and Coley (1994) hypothesized that smolts passing through reservoirs were negatively affected by starvation and that increased hatchery production could further deplete food resources. From 1987-91, empty stomachs were observed in 26% to 38% of the yearling chinook salmon smolts sampled at Lower Granite Dam and in 1991, this compared to less than 55 empty stomachs at McNary and Bonneville dams (Muir and Coley 1994). This data suggests that, in some reservoir areas or portions of reservoir areas, food availability is limited and that increased hatchery production could compound the problem. The areas where food is limited and the effect of

reduced feeding success on smolt performance and survival are unknown (Muir and Coley 1994). Neither Chapman et al. (1994) nor Witty et al. (1995) found documentation of density-related interaction in Snake and Columbia River reservoirs. Ultimate impacts on adult fish production would vary greatly in any one year as a result of multiple additional influences on smolt-to-adult survival, including flow-related passage time through the reservoirs and on to the estuary.

- *Estuarine Conditions*—The estuary is clearly an important rearing area for juvenile anadromous salmonids of all species and sizes as they move toward the ocean (Healey 1991). Extensive hatchery production programs may have at times exceeded the carrying capacity of the Columbia River estuary, resulting in competition between natural and hatchery fish. Furthermore, the productivity of the Columbia River estuary likely has decreased over time as a result of habitat degradation, which would increase the likelihood for competition in the estuary. Simenstad and Wissmar (1984) cautioned that estuary conditions may limit rearing production of juvenile chinook, and many other studies have demonstrated the importance of the estuary to survival and population fitness (Miller et al. 2003).

  The intensity and magnitude of competition in estuaries depends partially on the residence time of hatchery and natural juvenile salmonids. Duration of estuary use probably depends partially upon fish size at arrival (Chapman et al. 1994). Chapman et al. (1994) concluded that the survival of juveniles transported to below Bonneville Dam at a size too small to ensure high initial marine survival may depend upon growth in the estuary for successful ocean entry. Some workers (Reimers 1973, Neilson et al. 1985) have suggested that the amount of time spent in estuaries may relate to competition for food; that estuarine residence time increases with increased competition, because fish take longer to reach the threshold size needed for successful ocean entry. Thus, if large numbers of hatchery fish are present in the estuary, growth and survival of wild fish could be reduced (Chapman et al. 1994). In contrast, Levings et al. (1986) reported that the presence of hatchery chinook salmon did not affect residency times and growth rates of wild juveniles in a British Columbia estuary and that hatchery fish used the estuary for about half the time that wild fry were present (40-50 days).

  Natural populations of salmon and steelhead migrate from natal streams over an extended period (Neeley et al. 1993, Neeley et al. 1994); consequently, they also enter the estuary over an extended period (Raymond 1979). Hatchery fish are generally—but not always—released over a shorter period, resulting in a mass emigration into natural environments. In recent years, managed releases of water, commonly called water budgets, have been used to aid mass and fast migration of hatchery and wild smolts through the migration corridor. Decisions regarding the mode of travel in the migration corridor (i.e., in-river migration or collection/transportation) are made by managers to expedite movement of smolts to the estuary (Williams et al. 1998). Water budget management, combined with large releases of hatchery fish, result in large numbers of juvenile salmon and steelhead in the estuary during spring months when the estuary productivity is low. Fish that arrive in the estuary later in the season may benefit from increased food supplies. Chapman et al. (1994) notes that subyearling chinook released later in the summer returned at significantly higher rates than subyearlings released early in the summer.

- *Ocean Conditions*—There has been a general consensus that most density-dependent mechanisms at sea, if they occur, take place very early; probably within the first few weeks after smolts enter the ocean (Gunsolus 1978, Peterman 1982, 1987, Fisher and Pearcy 1988, Beamish et al. 2004). Factors which may contribute to competition in the ocean include: hatchery-reared fish that successfully forage upon reaching the ocean (Paszkowski and Olla 1985a, 1985b), food production in the ocean varies in time and space (Healey and Groot 1987), migratory salmonids remain in fairly cohesive groups (Pearcy 1984), and migration routes of different stocks and species may overlap (Steward and Bjornn 1990). Therefore, competition is possible between hatchery and wild fish in the ocean, particularly in nearshore areas (Peterman and Routledge 1983, Peterman 1989, and Emlen et al. 1990) and especially during periods of low ocean productivity (Steward and Bjornn 1990). McCarl and Rettig (1983) found evidence for density-dependent mortality in the area referred to as the Oregon Production Index Area (OPIA) which includes the Pacific coastal water bounded on the north by Leadbetter Point, Washington, south to Monterey Bay, California. They suggested that variability in smolt survival increased with the number of smolts, and hatchery smolts should be limited if the stability of fisheries was an important goal. However, Nickelson (1986) challenged these claims, suggesting that wild and hatchery fish do not occur together at sea and that there is no evidence supporting density-dependent mortality at sea. Witty et al. (1995) suggest that nearshore density-dependent mortality may occur when large numbers of hatchery juveniles are present during years of low ocean productivity.

Density interactions also may occur at sea away from nearshore areas. Several researchers have reported indications that oceanic carrying capacity can be taxed, with feed-back density effects in salmon populations (Chapman and Witty 1993). Adult size tends to decline in large populations of Fraser River pink salmon (Peterman 1987) noted that the average weight of pink salmon was less during years of larger hatchery populations. Chum salmon culture programs in Japan suggested the presence of density-dependent production limitations, expressed in mean size of adult fish produced as mass enhancement efforts proceeded (Kaeriyama 1989). Eggers et al. (1983) found that mean length of sockeye in Bristol Bay related inversely to magnitude of the return. Eggers et al. (1983) noted that the effect of density-dependent growth was reduced in years of higher ocean temperatures, suggesting that temperature effects moderated depression of growth in years of high fish density. Peterman (1987) reported that density-dependent processes, associated with available food during early ocean rearing, can reduce fish size. Taken together, these studies indicate a strong potential for oceanic competition between hatchery and wild salmon.

3.2.2.5 Disease

Hatchery programs often succeed or fail depending upon success in controlling pathogens. Types, abundance, and virulence (epidemiology) of pathogens and parasites in hatchery fish are generally known, but less is known about diseases and parasites in natural fishes of the Columbia River basin or the vectors and amounts of disease transmitted from hatchery to wild fish (Steward and Bjornn 1990). Hatchery fish are always confined to some degree, which creates opportunities for epizootic outbreaks. Often, but not always, hatchery fish are infected by pathogens in the hatchery water supply or by natural fish entering the hatchery. Regardless of control measures, hatcheries release some fish infected with pathogens and parasites although every attempt is made by hatchery managers and biologists to minimize release of impaired fish to the natural environment.

Disease is thought to result in significant post-release mortality among hatchery fish, being either directly responsible for mortality or predisposing fish to mortality from other causes

(Steward and Bjornn 1990). Steward and Bjornn (1990) found little evidence to suggest that the transmission of disease from infected hatchery fish to wild salmonids is widespread. However, there has been little research on this subject, and since most disease-related losses probably go undetected, researchers have concluded that the full impact of disease on stocks is probably underestimated (Goede 1986, Steward and Bjornn 1990). Increasing fish abundance through the release of large numbers of hatchery fish could alter normal population mechanisms and trigger outbreaks of pathogens in natural fish, both in tributary rearing areas and in mainstem migration corridors. McMichael et al. (2000) reported that disease incidence in cohabiting hatchery and wild fish increased with temperature and was likely influenced by the stress of interaction. Disease management practices as outlined by IHOT and the Pacific Northwest Fish Health Protection Committee have reduced the abundance and virulence of pathogens in hatchery populations.

3.2.2.6 Predation

The two primary predator-prey relationships that can result from hatchery and wild fish interactions include predation by hatchery fish on natural fish and the functional response of non-salmonid fish preying on natural fish as a result of increased numbers of hatchery and natural salmonids. Predator-prey interactions between hatchery steelhead and naturally produced salmon has been identified as a concern (Chapman and Witty 1993). Hatchery chinook salmon predation on wild chinook salmon has been reported by Sholes and Hallock (1979). Fresh (1997) cited several studies that indicated hatchery coho, steelhead, and chinook preyed on wild fry of conspecifics as well as pink and chum fry.

Residualism of hatchery salmon and steelhead is common (McMichael et al. 2000). Cannamela (1992) assumed total residualization rates of 10-25% based on Partridge (1985, 1986) and Chrisp and Bjornn (1978). Residual steelhead commonly exceed 10 in (250 mm) TL in Columbia River basin migration corridors, a threshold size at which piscivorous behavior of steelhead or rainbow trout increases markedly (Ginetz and Larkin 1976, Parkinson et al. 1989, Horner 1978, Partridge 1985,1986, Beauchamp 1990). However, most residual steelhead observed are in poor condition and likely do not survive long enough to become piscivorous (Petit, Idaho Department of Fish and Game, personal communication). This hypothesis is consistent with findings by Mauser (1991, unpublished), Partridge (1986), and Schuck (1991, unpublished) as described by Cannamela (1992). Recent hatchery management practices to address residualism concerns include targeting the size at release for steelhead to a range of 185-220 mm. Constructing dams and associated fish handling facilities and hatcheries have established places in the migration corridor where hatchery and wild smolts concentrate, thus greatly increasing the opportunity for predation. Creating reservoirs has increased the area of the river’s cross-section and decreased the velocity and turbidity of the flow, thus enhancing the efficiency of the predators (Junge and Oakley 1966).

Large concentrations of hatchery fish may adversely affect wild juveniles by stimulating functional responses from bird and non-salmonid fish predators (Steward and Bjornn 1990). In the Columbia basin migration corridor, this response is likely to occur at the head of reservoirs, at the face of dams, and at turbine spillway and bypass discharge areas. There is evidence that prey availability immediately below mainstem dams on the Columbia River affects predation rates by northern pikeminnow on juvenile salmonids (Petersen and DeAngelis 1992). Below McNary Dam, Vigg (1988) demonstrated that the predation rate of northern pikeminnow on juvenile salmonids increased with increased salmonid density to an asymptote at higher salmonid densities. Conversely, Cada et al. (1994) note that the importance of predation by northern pikeminnow and other predators at the Columbia River hydroelectric projects may be lessened by the possibility that many fish being consumed are hatchery smolts; they speculate that hatchery fish are more vulnerable than wild fish. Large numbers of hatchery fish may provide a swamping effect and reduce the predation on naturally produced salmonids.

3.2.2.7 Mixed Stock Harvest

Because hatchery and naturally produced salmon and steelhead are often captured in the same ocean and river fisheries, when hatchery production stimulates harvest effort, the catch of naturally produced fish can be increased as well. Since hatcheries provide an environment where the survival rate to smolting is much greater than in the wild, the proportion of returning adults needed to support the population is much less and, therefore, the targeted harvest rate has been at times much greater than the commingled wild populations can sustain. Thus, stimulating harvest has been a notable impact of hatchery programs on natural production (Hilborn 1992). Harvest managers have grappled with the challenge of regulating the fisheries so that surplus hatchery fish can be harvested without over-harvesting the wild fish that are intermixed in the same fishery.

Harvest management strategies focused on hatchery fish harvest were common practice for several species in the lower Columbia for many years (Flagg et al. 1995). Fishery strategies which maximized harvest of surplus hatchery fish were consistent with the mitigation objectives which established the hatchery programs. Current harvest management strategies have transitioned to minimize harvest of weak wild stocks to meet conservation objectives under ESA (see previous section on Fishing). Seasons are structured and regulated in an attempt to provide reasonable opportunity to harvest hatchery and healthy wild stocks within the limits of the weak stock management focus.

Selective harvest of adipose fin-clipped hatchery steelhead, coho, and spring chinook, and release of unclipped wild fish, is now required in all lower Columbia and tributary sport fisheries. Hatchery-origin fall chinook are not currently adipose fin-clipped for selective harvest and selective regulations are not in place for fall chinook fisheries. Wild fish harvest rates are also controlled by annual structure of fishing seasons (see previous section on Fishing). The lower Columbia commercial fishery now uses tangle-net gear and on-board fish recovery boxes to enable release of wild spring chinook and retention of adipose fin-clipped hatchery spring chinook. The commercial fishery is also regulated by time and area restrictions to focus harvest on hatchery coho while minimizing impacts on wild coho (see previous section on Fishing).

3.2.2.8 Passage

Hatchery collection facilities use weirs, ladders, and screens to block fish passage, capture fish for the collection of broodstock, and regulate numbers, stocks, and species of fish entering and passing above hatchery facilities. All weirs cause some degree of migration delay. Most weirs cannot accommodate upstream passage of large fish unless they are staffed to provide passage. Weirs often cannot be operated as desired or according to protocol because of physical and biological constraints such as high water, cold or warm water teperatures, low flow, and/or staffing problems (Witty et al. 1995). Weirs operated to block fish passage for the purpose of collecting hatchery broodstock, or to implement supplementation programs, usually have specific operating criteria that vary facility-to-facility and year-to-year. Estimated production potential above weirs is usually known, and escapement is designed accordingly. Operating weirs to meet escapement and hatchery production goals is often a challenge (Witty et al. 1995).

Hatchery fish ladders have the potential to block or delay natural fish passage. These impacts can vary from very significant to insignificant depending on:

- numbers or proportion of the run affected,
- quantity and quality of habitat above the ladder, and
- impacts on life history characteristics (Witty et al. 1995).

Problems with inadequate screening at hatcheries can be divided into two categories: screen systems that fail to keep natural fish out of hatchery facilities and screen systems that fail to keep hatchery fish out of natural environments. The impacts of natural fish entering hatchery facilities are:

- removing natural fish from their natural environment,
- exposing natural fish to disease and predation in hatchery environments,
- introducing disease from natural fish to the hatchery environment,
- natural fish in environments unsuited for their survival, and
- releasing natural fish in environments which will result in changing biological balance, changing genetics of endemic stocks, or otherwise upsetting management objectives.

Some possible impacts of hatchery fish escaping into natural environments are:

- introduction of non-endemic species or stocks,
- changing biological balance, changing genetics, or upsetting management objectives,
- exposing natural fish to disease, competition or predation from hatchery fish, and
- failing to meet hatchery program objectives.

The degree of impact may or may not be directly related to numbers of fish entering or leaving hatchery facilities, but potential impacts are related to fish numbers (i.e. when all hatchery fish escape as compared to a small number of hatchery fish escaping (Witty et al. 1995).

3.2.2.9 Water Quality

General water quality effects resulting from the operation of hatchery facilities include potential impacts from water withdrawal and hatchery effluent. All hatcheries are required to comply with NPDES standards for clean water prescribed by WDOE. Many facilities have incorporated settling ponds that improve water quality discharges.

Many fish hatcheries and satellite facilities divert natural stream flows upstream of hatchery facilities and return the water downstream of the hatchery. The volume of water removed varies according to fish production profiles in the hatchery. Withdrawal of natural stream flows results in a stream channel with reduced flow, no flow, or unnatural flow patterns. When evaluating impacts of water withdrawal on natural fish and their environments, one should consider whether:

- fish passage or homing is affected, and/or
- fish production is significantly affected.

Making these evaluations requires knowledge of life history characteristics and population dynamics of affected natural fish and comparing this information to:

- measured area affected by water withdrawal,
- time of year when water is withdrawn,
- percent of flow withdrawn, and
- location where water is returned.

The impact of hatchery water withdrawal requires an examination of past, present, and proposed operations at each hatchery (Witty et al. 1995).

Hatchery effluent may contain organic waste, chemicals, fish pathogens, and warmer or cooler water. The main forms of wastes in hatchery effluent are suspended solids and dissolved nutrients; especially nitrogen and phosphorus (Pillay 1992). Measuring the impacts of effluent one should consider (Witty et al. 1995):

- pounds of fish produced,
- effluent treatment facilities,
- rate of dilution in the recipient waters,
- quality of water entering the hatchery, and
- water quality standards set by state and federal regulations.

The nature and extent of chemical use in hatcheries depends on the locality, species of fish reared, nature and intensity of culture operations, and the frequency of disease occurrence (Pillay 1992). There is a potential for harmful effect of chemicals in natural environments. If chemicals used in hatcheries are deemed safe by the Food and Drug Administration, their dispersal into natural environments should be considered safe. The level of impact from discharged hatchery effluent on fish survival is unknown, but is presumed to be small and localized at outfall areas, as effluent is diluted downstream (NMFS 1995). Hatchery facilities that rear greater than 20,000 lbs annually must obtain state and federal pollution discharge (NPDES) permits that set limits on the release of effluent from the facilities.

Hatchery effluent may increase populations and virulence of indigenous pathogens. Virulent pathogens are usually associated with epizootics in natural populations, whereas facultative pathogens tend to emerge as causes of epizootics in cultured populations (Pillay 1992). Despite the absence of conclusive evidence of major infections of wild stocks from aquaculture, very little research has been done to define the role of aquaculture in the outbreak of diseases in natural fish (Pillay 1992). Agencies use guidelines outlined by the Pacific Northwest Fish Health Protection Committee (PNFHPC) to control fish pathogens in hatchery effluent.

Some hatcheries heat or cool water to control embryo development, although the amount of water treated usually is not great. If the water temperature in the natural environment is changed, adverse impacts on natural fish could occur (Witty et al. 1995).

3.2.3 Lower Columbia Hatchery Production

3.2.3.1 Spring Chinook

Spring chinook salmon populations in the lower Columbia River have been heavily influenced by hatchery programs, which were developed to mitigate for lost spring chinook production associated with dam construction and other habitat degradation (see Habitat section below). Present spring chinook salmon populations in the lower Columbia River are primarily produced by hatchery programs. Total releases have changed over the years by basin. In the Lewis and Cowlitz River basins, annual releases have generally been less than 1.5 million smolts. Spring chinook releases into the Wind River basin are primarily from the Carson National Fish Hatchery (NFH), while releases into the Little White Salmon River basin are primarily from the Little White Salmon NFH, although some releases from the Carson NFH have been made. The Carson NFH has often produced more spring chinook adults than the Little White Salmon NFH, although production numbers have been similar in recent years.

The current (2003 brood) release goal of yearling spring chinook in the lower Columbia Washington tributaries totals *5,137,000* (Table 3-35).

Table 3-35. Current (2003 brood) annual release goals of spring chinook salmon juveniles (subyearling and yearling) into lower Columbia basins.

Basin	Hatchery	Release Goal	
		Yearling	Subyearling
L. Cowlitz	Cowlitz Salmon Hatchery	967,000	
U. Cowlitz	Cowlitz Trout Hatchery		300,000
Kalama	Kalama Falls/Fallert Creek Hatchery	500,000	
Lewis	Lewis/Speelyai Hatcheries	1,050,000	
Deep River	Cowlitz/Lewis Hatcheries	200,000	
Little White Salmon	LWS/Carson Hatcheries	1,000,000	
Wind	Carson/LWS Hatcheries	1,420,000	
Lower Columbia Total		*5,137,000*	*300,000*

The Cowlitz River spring chinook stock has received only limited transfers of non-native stocks, but is strongly influenced by hatchery-derived fish (WDF et al. 1993). Stocks on the Lewis and Kalama rivers are a composite of the Cowlitz River spring chinook stock and other lower Columbia and Willamette River spring chinook salmon stocks (WDF et al. 1993). Numerically, most of the spring chinook spawning naturally in lower Columbia River tributaries on the Washington side are now hatchery strays (Marshall et al. 1995). All Washington populations of spring chinook salmon in the lower Columbia River are currently managed as populations of mixed origin (WDF et al. 1993).

Adult returns to the hatcheries below Bonneville Dam (Cowlitz-type spring chinook) has ranged from a few hundred to nearly 25,000 during 1950–2000. The hatchery returns of upriver spring chinook stock to the Little White Salmon and Carson hatcheries on the Wind and Little White Salmon rivers ranged from a few hundred to over 20,000 during 1950–2000. Since 1995, returns to the Little White Salmon and Carson hatcheries have exceeded the hatchery returns to Cowlitz, Kalama and Lewis hatcheries (Figure 3-66).

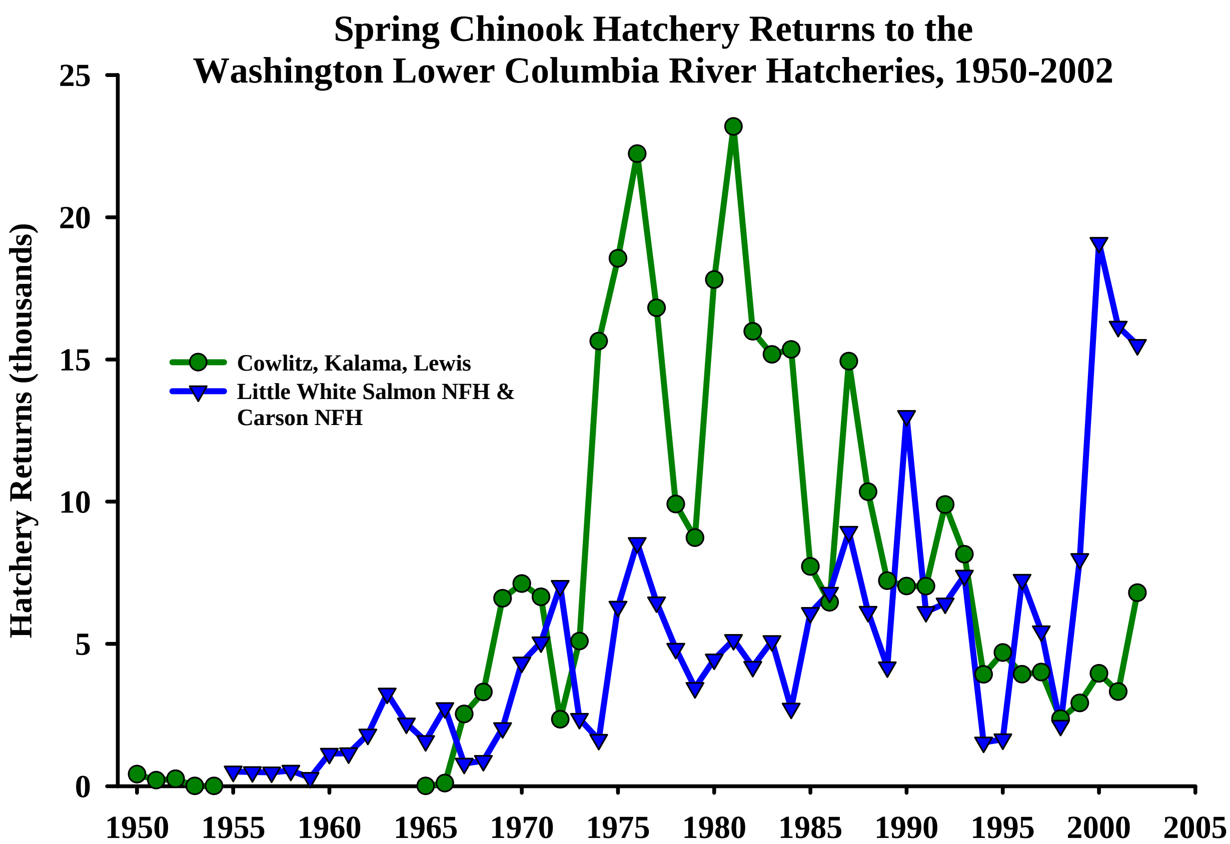

Figure 3-66. Hatchery returns of spring chinook to the Washington lower Columbia River hatcheries.

Hatchery-produced spring chinook provide significant harvest opportunity in the mainstem Columbia and in Cowlitz, Kalama, Lewis, Wind, and Little White Salmon tributary sport fisheries. Hatchery-produced spring chinook are now adipose fin-clipped to provide selective harvest opportunity in the mainstem Columbia and in Cowlitz, Kalama, Lewis rivers and in the future in Wind and Little White salmon tributary sport fisheries. Total adult spring chinook returns to the tributaries below Bonneville Dam (Cowlitz-type spring chinook) have ranged from 3,100 to 36,900 during 1980-2002 (Figure 3-67). The adult returns of Carson-stock spring chinook to the Wind and Little White Salmon rivers have ranged from about 1,200 to over 46,900 during 1980-2002 (Figure 3-67). The adult returns to these five tributaries are believed to be nearly 100% from hatchery-produced smolts.

Cowlitz River spring chinook are the largest component of the lower Columbia River hatchery spring chinook stocks. Historically, total (i.e. adults and jacks) spring chinook hatchery returns to the Cowlitz normally have been greater than 10,000 fish, with a peak return in 1987 of nearly 37,000. However, in recent years, hatchery returns to the Cowlitz have declined to a magnitude similar to spring chinook returns in the Kalama and Lewis River basins. Meanwhile, hatchery returns of spring chinook to the Wind and Little White Salmon rivers have increased in recent years. The adult production from the Little White Salmon NFH and Carson NFH has exceeded the adult production from Cowlitz, Kalama and Lewis hatcheries since 1995 (Figure 3-67).

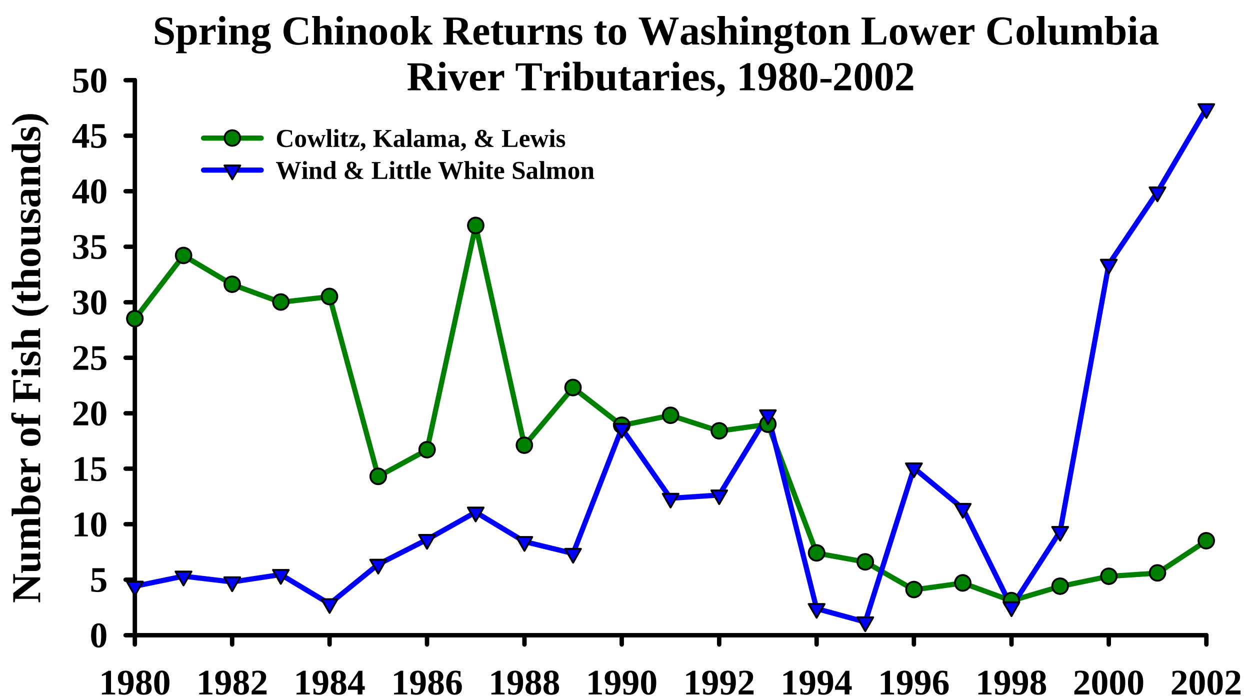

Figure 3-67. Returns of adult spring chinook (escapement and harvest) to Washington lower Columbia River tributaries, 1980–2000.

3.2.3.2 Fall Chinook

Currently, there are 10 hatcheries (WDFW, ODFW, and USFWS) that release fall chinook salmon into the lower Columbia River ESU in Washington. The current (2003 brood) release goals for Washington hatcheries in the lower Columbia ESU total 35.7 million juveniles (Table 3-36).

Table 3-36. Current (2003 brood) annual release goals of fall chinook salmon juveniles (subyearling and yearling) into Washington lower Columbia basins.

Basin	Brood Source	Annual Release Goal	
		Tule	URB
Little White Salmon	LWS/Priest Rapids Hatcheries		2,000,000
Washougal	Washougal Hatchery	4,000,000	
Kalama	Kalama Hatchery	5,000,000	
Toutle/Green	NF Toutle Hatchery	2,500,000	
Cowlitz	Cowlitz Salmon Hatchery	5,000,000	
Abernathy	Abernathy Hatchery	Program discontinued	
Elochoman	Elochoman Hatchery	2,000,000	
Grays	Grays River	Program discontinued	
Chinook	Sea Resources Hatchery	107,500	
Columbia (Bonneville Pool)	Spring Creek Hatchery	15,100,000	
Total		*33,707,000*	*2,000,000*

Within the ESU, however there are differences in degree of hatchery influence on the local stocks. The Cowlitz fall chinook returns are predominately produced from the Cowlitz Salmon Hatchery, however there have been few transfers of outside stocks into the Cowlitz. The Kalama Hatchery stock has also generally maintained eggs solely from Kalama-origin fish. The North Lewis has maintained a healthy wild component with minimal hatchery influence and Lewis River fall chinook hatchery production was discontinued after 1985. The EF Lewis and Coweeman fall chinook populations are at low levels but are not influenced by hatchery production.

Historical releases of most fall chinook hatchery programs peaked in the late 1970s and 1980s, with over 10 million chinook released annually in the Grays, Cowlitz, Kalama, Washougal, and Little White Salmon River basins (Figure 3-68 and Figure 3-69). The highest annual release of fall chinook in a lower Columbia basin was over 30 million chinook in the Little White Salmon River in 1978.

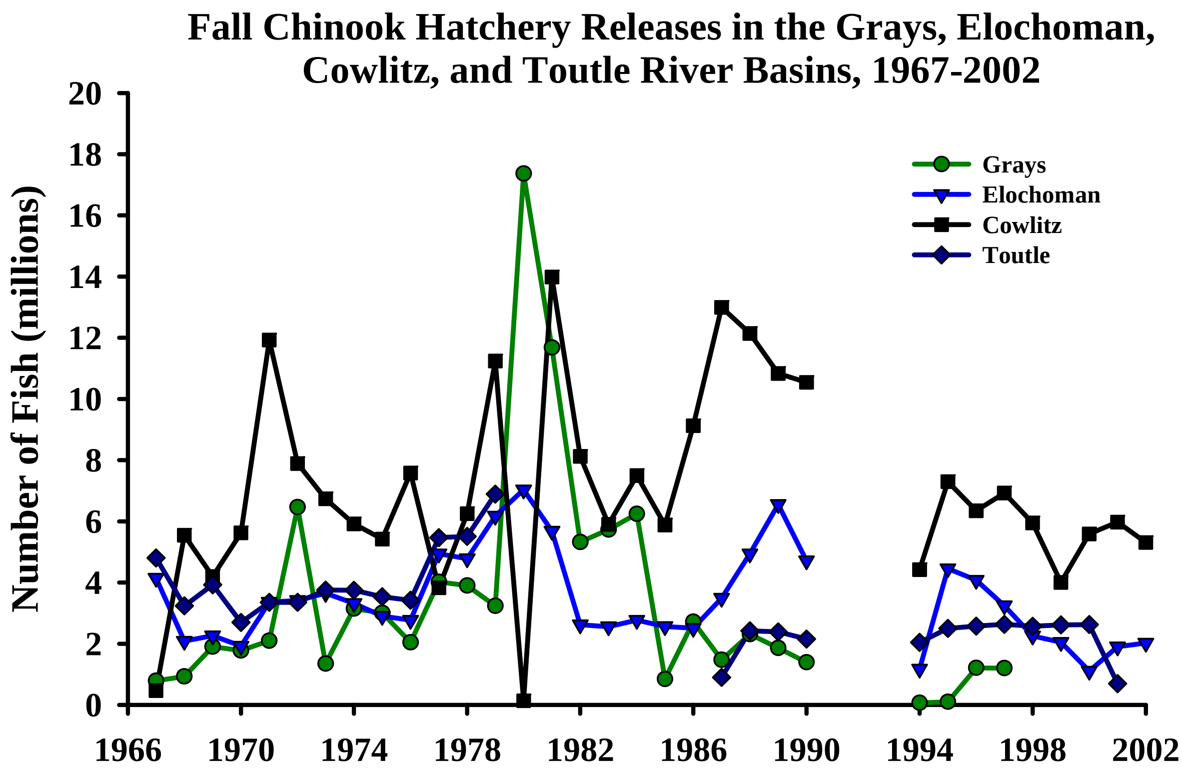

Figure 3-68. Hatchery releases of fall chinook to the Grays, Elochoman, Cowlitz, and Toutle River basins, 1967–2002.

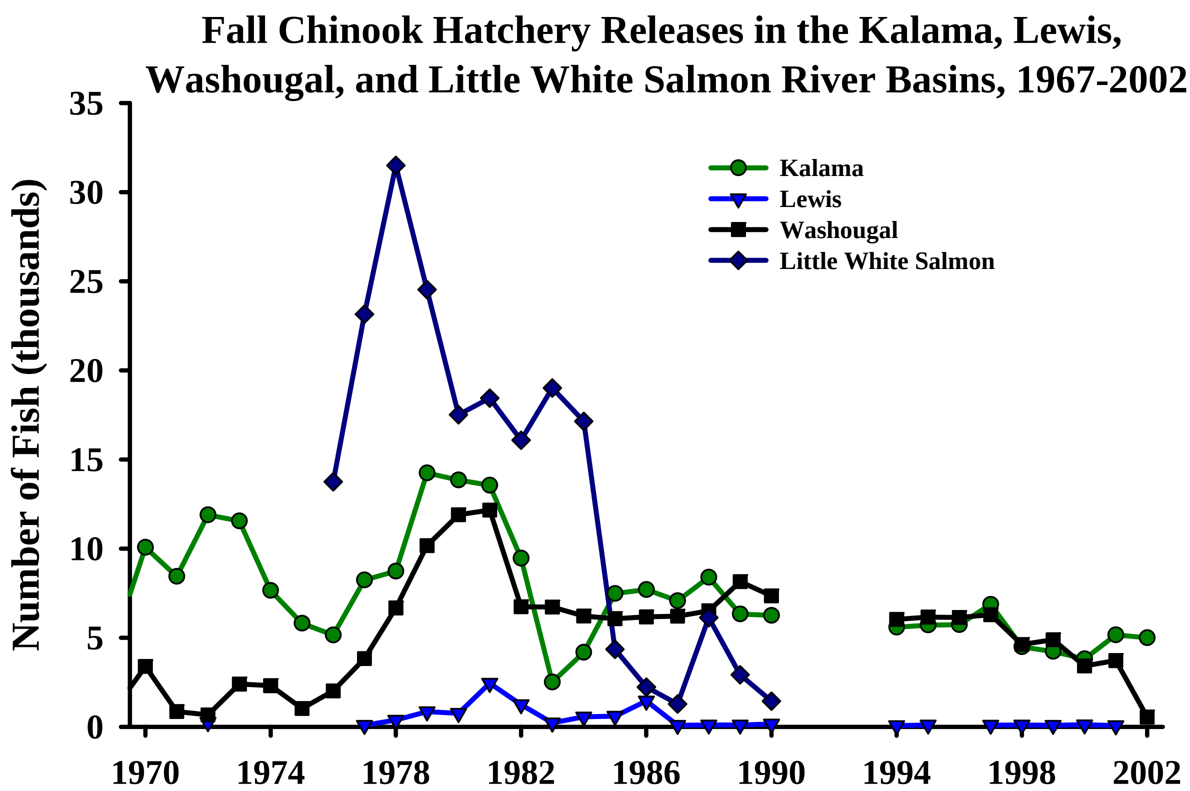

Figure 3-69. Hatchery releases of fall chinook to the Kalama, Lewis, Washougal, and Little White Salmon River basins, 1967–2002.

Throughout the range of fall chinook salmon, stocks have often been transferred among watersheds, regions, states, and countries, either to initiate or maintain hatchery populations or naturally spawning populations. The transfer of non-native fish into some areas has shifted the genetic profiles of some hatchery and natural populations so that the affected population is genetically more similar to distant hatchery populations than to local populations (Howell et al. 1985, Kostow 1995, Marshall et al. 1995). However, most fall chinook salmon releases into the Lower Columbia River ESU originated from stocks within the ESU, although some upriver stocks were propagated as described earlier. Because of extensive mixing of hatchery and wild populations, it is often difficult to determine the proportion of native and non-native hatchery fish released into a given watershed. Transplanted hatchery fish routinely acquire the name of the river system into which they have been released. The majority of fall run chinook salmon populations in Washington tributaries of the lower Columbia are thought to be essentially one stock, widely mixed as a result of adult straying and egg transfers between hatcheries (Howell et al. 1985, Utter et al. 1989, WDF et al. 1993, Marshall et al. 1995).

The majority of natural spawners in the Grays, Elochoman, Cowlitz, Kalama, and Washougal rivers has been of hatchery origin and strays from several lower Columbia River hatcheries are found in these basins (WDF et al. 1993, Marshall et al. 1995). As well, strays from Oregon's Rogue River fall run chinook program have been observed in the Elochoman River and Abernathy Creek (WDF et al. 1993, Marshall et al. 1995). However, the release location of this stock has been changed to address this problem. These select area brights are uniquely marked for monitoring and removal at hatchery traps.

Large numbers of upriver bright fall chinook strays from the Little White Salmon NFH and Bonneville Hatchery programs have been found naturally spawning above Bonneville Dam in the Wind, White Salmon, and Klickitat rivers (WDF et al. 1993). Broodstock for this program was collected by intercepting various upriver bright stocks headed for spawning sites above The Dalles Dam.

Lower Columbia River fall chinook salmon hatchery stocks continue to comprise the majority of all chinook salmon in the Lower Columbia River ESU. However, influence of hatchery fish on natural spawning populations in the North Lewis, East Lewis, and Coweeman rivers is thought to be negligible. Returns to lower Columbia River hatchery facilities in Washington typically range from 10,000 to 40,000 adults (Figure 3-70).

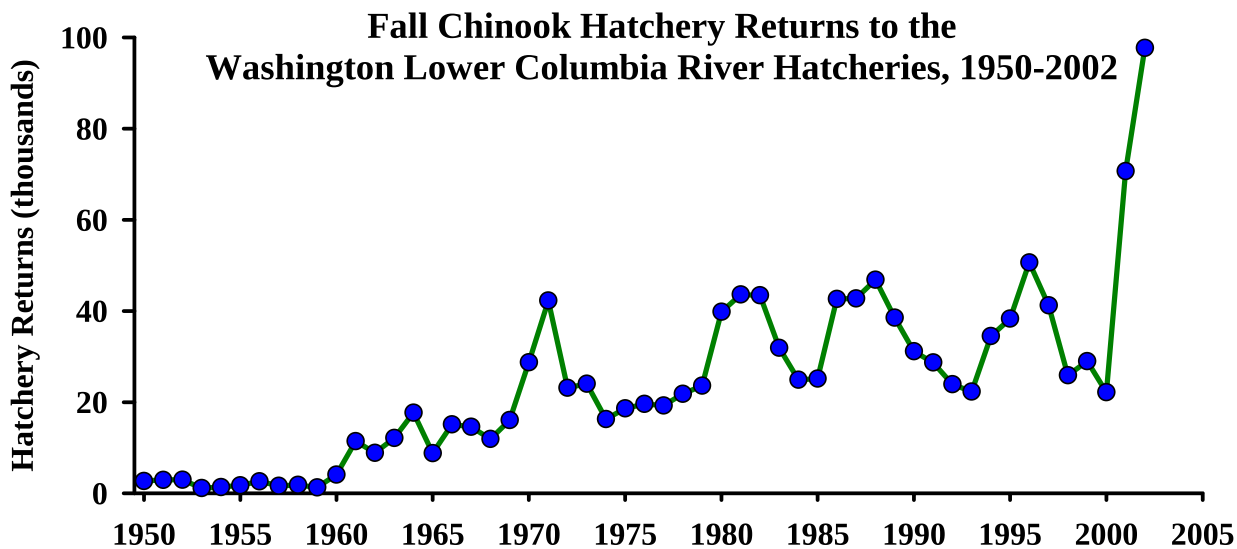

Figure 3-70. Hatchery returns of fall chinook to Washington lower Columbia River hatcheries.

Adult hatchery returns of lower Columbia hatchery fall chinook vary annually and this number is affected by numerous factors, including hatchery juvenile releases, smolt-to-adult survival, ocean survival, and harvest rates.

- Cowlitz tule fall chinook are the largest individual hatchery run with annual returns usually around 5,000 fish and many years with escapement over 10,000 fish (Figure 3-71). Tule fall chinook hatchery returns in the Cowlitz River basin peaked in the early 1970s and again in the late 1980s.
- Kalama and Washougal tule fall chinook are the next largest hatchery runs in the lower Columbia River (Figure 3-71). Kalama tule fall chinook returns peaked in the early 1970s and the late 1990s, while Washougal tule fall chinook returns peaked in the late 1980s and late 1990s.
- The Elochoman River has a substantial hatchery return of tule fall chinook; returns in the Elochoman peaked in the late 1980s and late 1990s (Figure 3-71).
- In the Lewis River basin, tule fall chinook hatchery returns have been relatively low and constant over time (Figure 3-71).

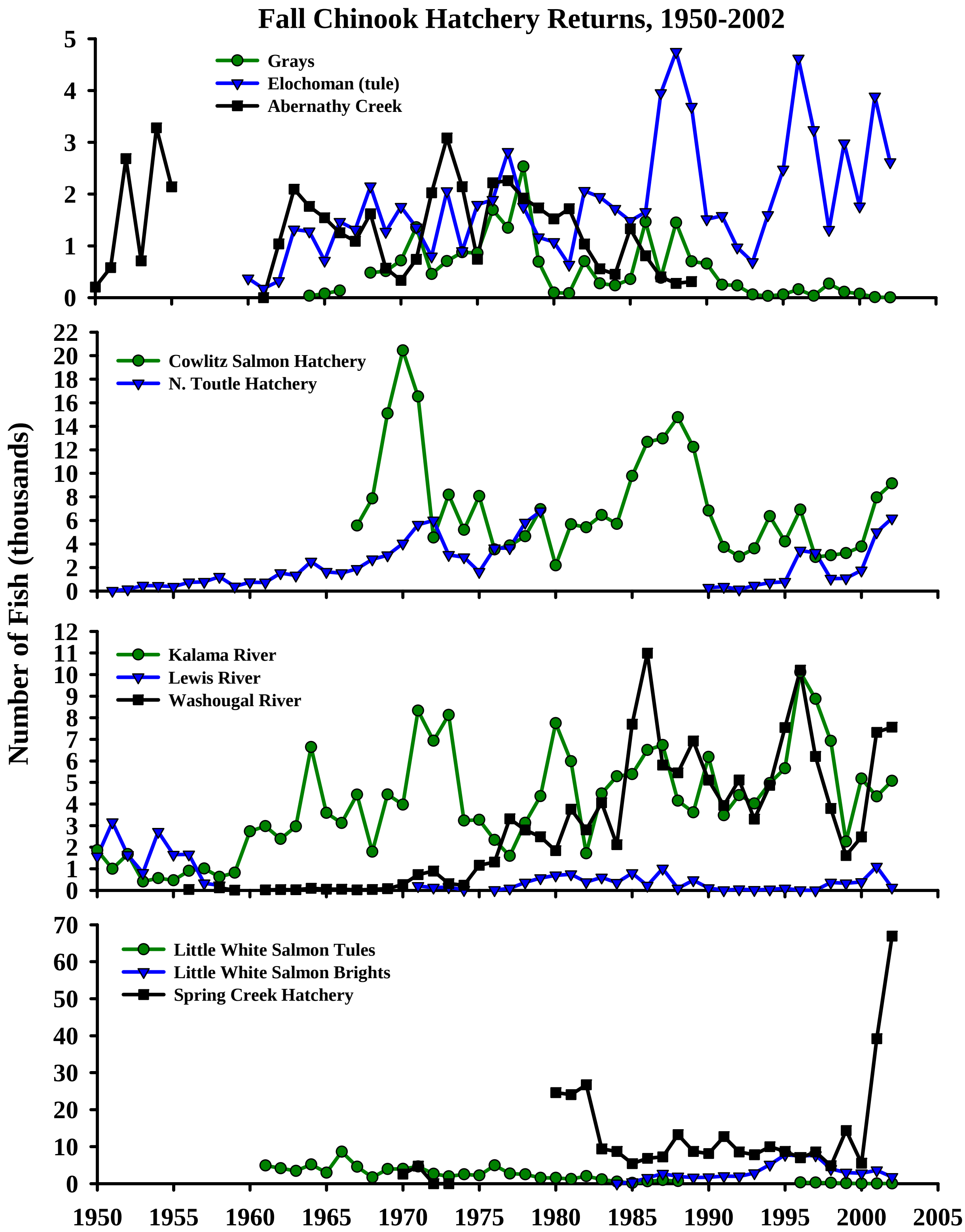

Figure 3-71. Hatchery returns of fall chinook to Columbia River subbasin, 1950–2002.

3.2.3.3 Coho Salmon

In *past* years, hatchery production of coho salmon in the lower Columbia River/southwest Washington coast ESU has far exceeded that of any other area with respect to the number of hatcheries and quantities of fish produced. Many hatcheries within this ESU released 1-3 million smolts annually, with the two largest hatcheries, Cowlitz and Lewis, releasing an average of 6-7 million smolts (Table 3-37).

Table 3-37. Average annual releases of coho salmon juveniles (fry and smolts) from Washington lower Columbia hatchery facilities during release years 1987–91 (NMFS 1995).

Hatchery	5-Year Average
Washougal	3,885,612
Lewis	6,180,000
Kalama Falls	990,000
Fallert Creek (Lower Kalama)	831,605
Toutle	478,090
Cowlitz	7,956,089
Elochoman	2,013,032
Grays River	744,655
Sea Resources	125,500
Total	*23,204,583*

However, in recent years, Washington lower Columbia hatchery programs have reduced coho production, either as a result of reduced funding, reprogramming federal-funded production to facilities above Bonneville Dam, or mitigation adjustments. The current annual release goal for 2003 brood totals 9.7 million yearling smolts (Table 3-38).

Table 3-38. Current (2003 brood) annual release goals of coho salmon smolts from Washington lower Columbia hatchery facilities.

Basin	Brood Source	Early (Type S)	Late (Type N)	Total
Little White Salmon	LWS Hatchery	1,000,000		1,000,000`
Washougal	Washougal Hatchery		500,000	500,000
NF Lewis	Lewis River	880,000	815,000	1,695,000
Kalama	Fallert Creek(S)/Kalama Falls(N)	350,000	350,000	700,000
NF Toutle (Green R.)	NF Toutle Hatchery	800,000		800,000
L. Cowlitz	Cowlitz Salmon Hatchery		3,200,000	3,200,000
Elochoman	Elochoman Hatchery	418,000	512,000	930,000
Columbia (Steamboat slough)	Grays River Hatchery	200,000		200,000
Grays	Grays River Hatchery	150,000	`	150,000
Deep River	Grays River Hatchery	400,000		400,000
Chinook	Sea Resources Hatchery	52,500		52,500
Lower Columbia TOTALS		4,250,500	5,377,000	9,627,500

Extensive stock transfers have occurred within the Lower Columbia River/ Southwest Washington Coast Coho ESU. Most transfers of coho salmon have used stocks from within the ESU, although transfers from outside the ESU have also occurred, including those from the Oregon coast, Olympic Peninsula, and Puget Sound/Strait of Georgia ESUs. Outplanting records show a similar pattern to transfers between hatcheries, with extensive use of within-ESU stocks, in addition to less frequent use of stocks from the other three ESUs. Most movement of coho salmon, either as hatchery transfers or off-station releases, has occurred within each of the three areas of this ESU (Oregon-side Columbia River, Washington-side Columbia River, and southwest Washington coast), with little movement of fish among the three areas (NMFS 1995). There has been liberal exchange of early and late stock coho among hatcheries on the Washington side, with the exception of the Cowlitz Hatchery which has maintained the original late stock without transfers into the program. On the other hand, Cowlitz Type N stock coho have been used widely in several Washington lower Columbia tributary hatchery programs.

Because of past hatchery practices, many coho stocks in Washington-side tributaries of the lower Columbia are now considered mixed and of composite production. In the once productive Cowlitz River basin, for example, DeVore (1987) accounted for the 1982-brood hatchery release and concluded wild/natural production was minor. Of the 4,635 naturally spawning coho in the Cowlitz in 1985, an estimated 91% were from hatchery smolt releases. Hatchery coho have been planted in the Cowlitz since at least 1915, when the Tilton River Hatchery operated downstream of Morton until 1921. Stock mixing probably began in 1915 (DeVore 1987). Since 1968, the Cowlitz Salmon Hatchery has been producing coho salmon. The mitigation goal is to maintain annual returns of 25,500 coho adults to the hatchery.

Many streams, especially those downstream of the Cowlitz River, have had years of hatchery production with an earlier-timed stock than historically spawned in each river. For example, the Grays and Elochoman hatcheries have maintained an early Type S hatchery program with fish primarily originating from Toutle River stock, although the natural returns to these rivers were principally fish that spawned from late November to March. The effect of introducing an early coho stock to a basin that historically supported a late coho stock (or vice versa) is not completely understood; there may be little or no interaction between early hatchery stock and any remnant late stock because of the temporal segregation of the runs.

There may be some benefit to the way late stock production has historically been managed. Hatchery programs have been operated to take eggs prior to mid-December to assure adults produced would be accessible to most of the freshwater fisheries. Because of this practice to maximize exposure to the fisheries, there could be a late spawning remnant of natural production which has not been significantly mixed with hatchery fish nor subjected to the same harvest pressure as the earlier-timed fish. This appears to be the case in the Cowlitz basin as there is documentation of a “late” late coho run. These fish could be extremely important to future recovery efforts. Currently, as part of the reintroduction program above Cowlitz Falls Dam, adult coho are being released into the upper Cowlitz River basin.

Historical juvenile hatchery coho salmon releases by basin have generally ranged from 1-5 million annually (Figure 3-72). In recent years, releases into the Cowlitz River have exceeded releases in other basins and have averaged about 5 million juveniles annually, including coho reintroduced to the Upper Cowlitz above the dams. Grays River Hatchery coho production has declined substantially from 1970s and 1980s production levels.

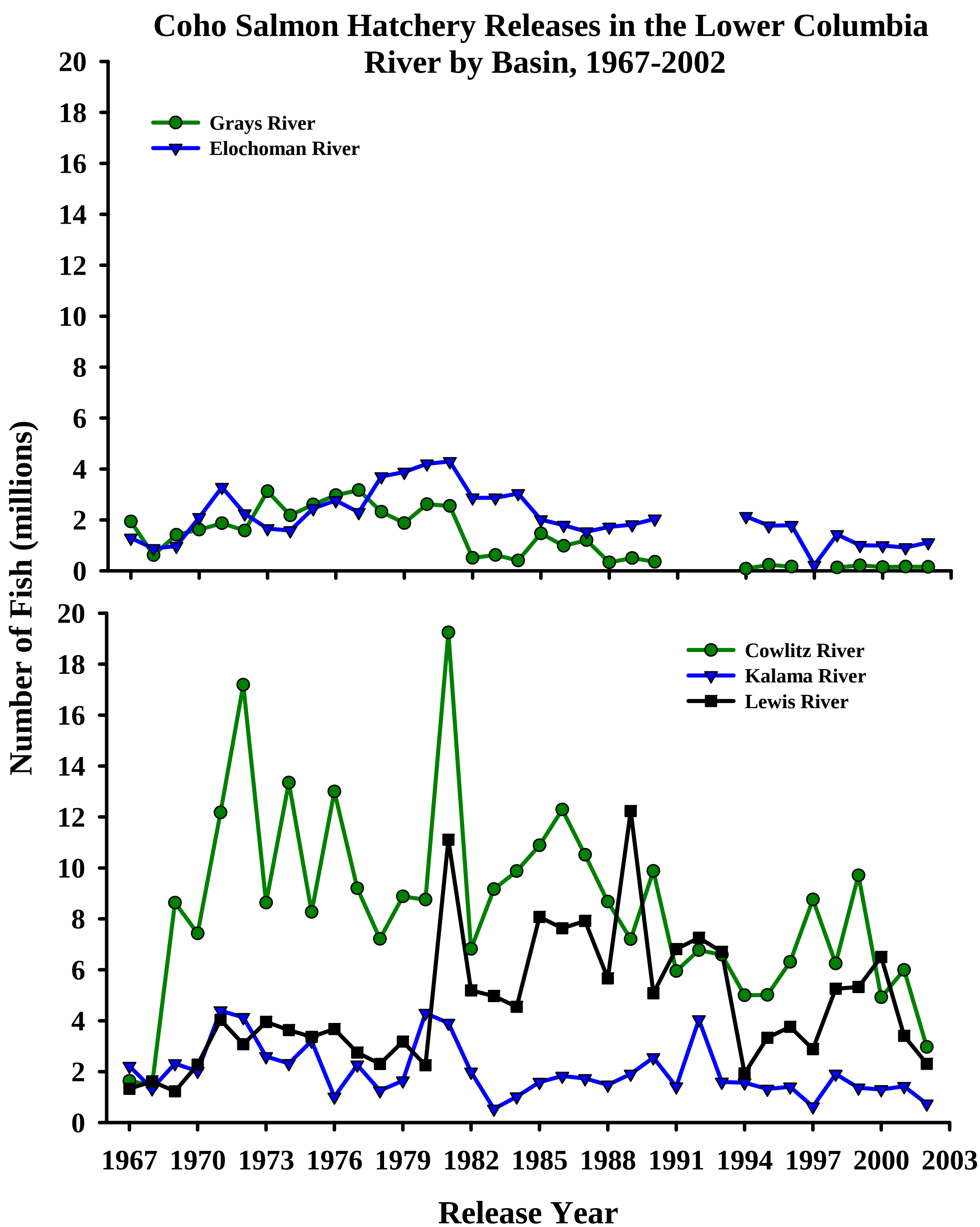

Figure 3-72. Hatchery releases of coho salmon in the lower Columbia River by basin, 1967–2002.

During 1978–2002, hatchery releases at the Little White Salmon NFH peaked at approximately 3.7 million in 1984. The lowest number of releases from this hatchery was approximately 750,000 in 1993 (Figure 3-73).

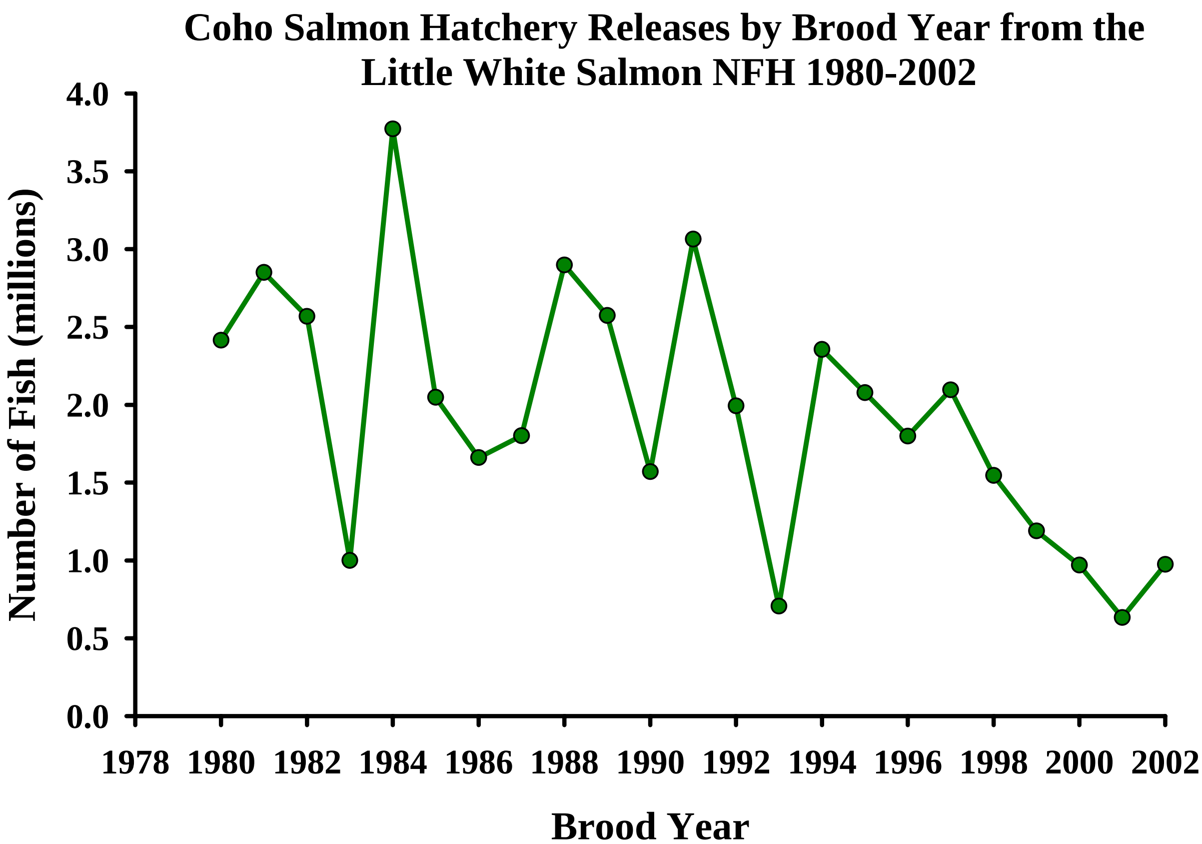

Figure 3-73. Hatchery releases of coho salmon by brood year from the Little White Salmon NFH, 1980–2002.

Since 1982, adult coho returns to Washington hatcheries below Bonneville Dam have ranged from 4,759 (1987) to 91,407 (2001) for early stock, and from 11,776 (1995) to 177,941 (2001) for late stock (Figure 3-74).

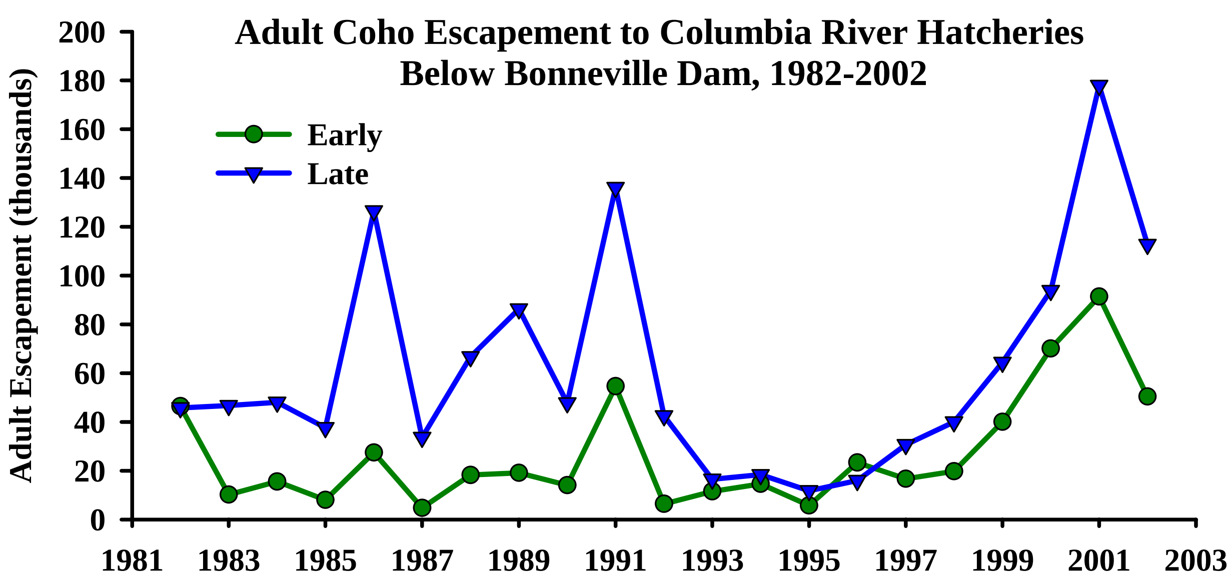

Figure 3-74. Columbia River adult coho escapement to hatcheries.

From 1983 to 1992, the average annual hatchery escapement to the Cowlitz River was 28,572, and to the Cowlitz Salmon Hatchery was over 75,000 coho in 2001 and 2002 (Figure 3-75). The Cowlitz Salmon Hatchery produces Type N (late) coho while the North Toutle Hatchery produces Type S (early) coho. In 1980, some North Toutle Type S coho strayed to Cowlitz Hatchery as a result of the eruption of Mt. St. Helens.

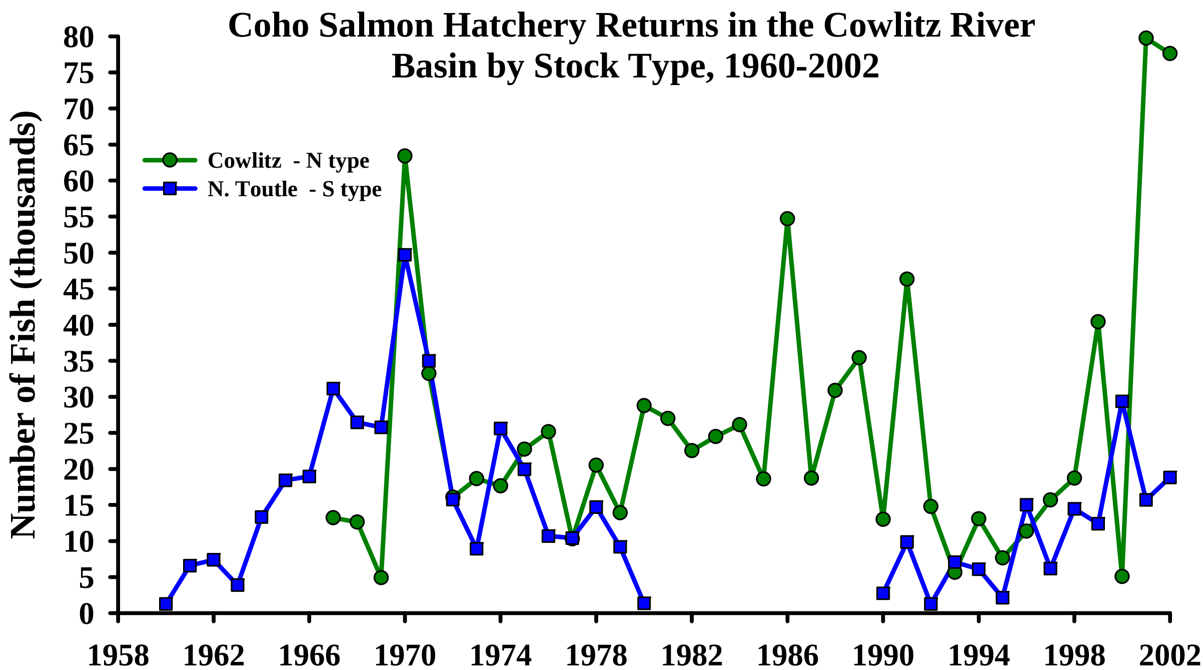

Figure 3-75. Hatchery returns of adult coho salmon in the Cowlitz River basin by hatchery and stock type, 1960-2002.

Significant coho production also has occurred in the Lewis, Kalama, and Washougal River basins.

- The Lewis River Hatchery produces both Type S and Type N coho while the Speelyai Hatchery produces Type S coho (Figure 3-76). The largest annual hatchery coho return to the Lewis River basin was over 95,000 adult fish in 1999.
- In the Kalama basin, the Kalama Falls Hatchery was the primary producer of coho salmon with the largest annual return of over 40,000 adult coho in 1966 (Figure 3-77). In recent years, similar-sized hatchery returns of Type S coho have been documented at the Kalama Falls and Fallert Creek hatcheries.
- In the Washougal basin, historical production at the Washougal Hatchery was Type S coho; the largest annual hatchery return was approximately 45,000 coho in 1968 (Figure 3-78). In recent years, Washougal Hatchery production has shifted to Type N coho and annual adult returns have averaged about 5,000 fish.

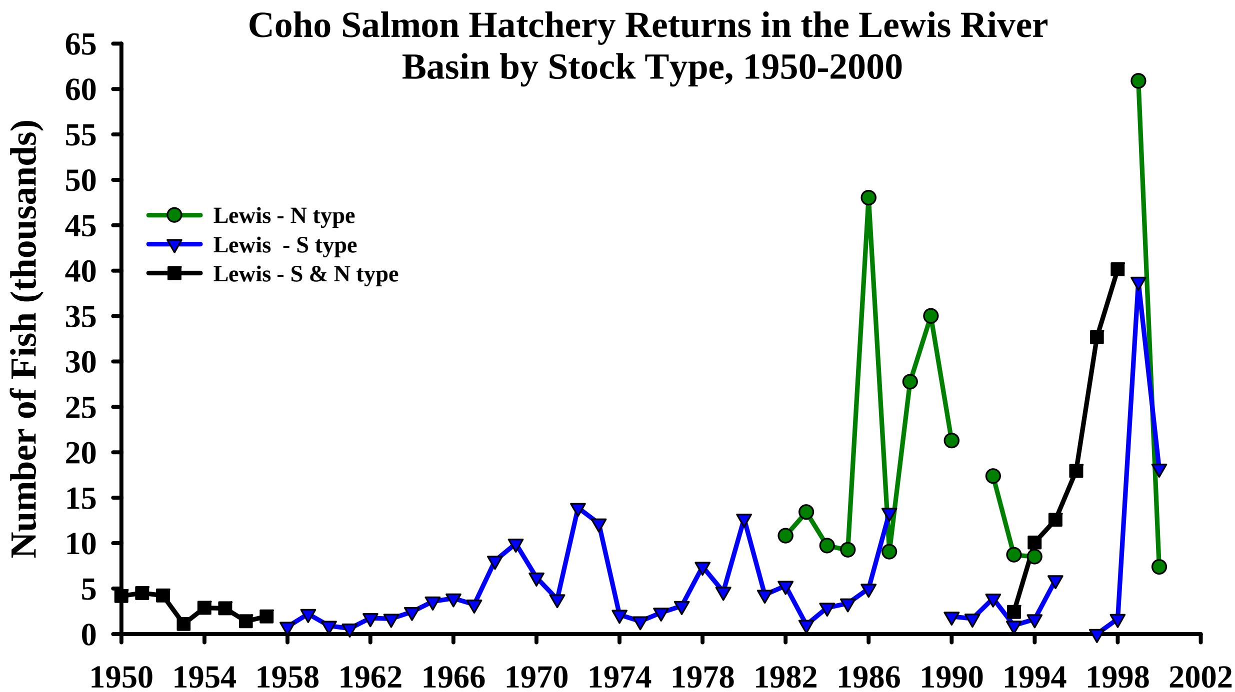

Figure 3-76. Hatchery returns of adult coho salmon in the Lewis River basin by stock type, 1950–2002.

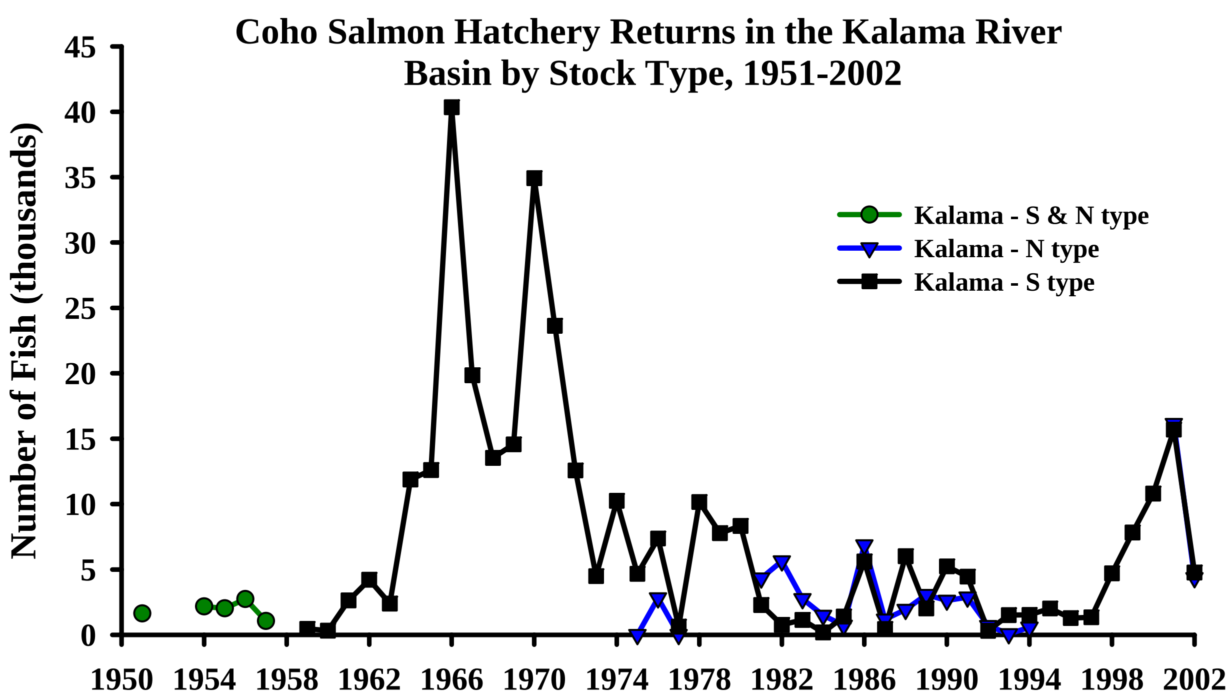

Figure 3-77. Hatchery returns of adult coho salmon in the Kalama River basin by stock type, 1951–2002.

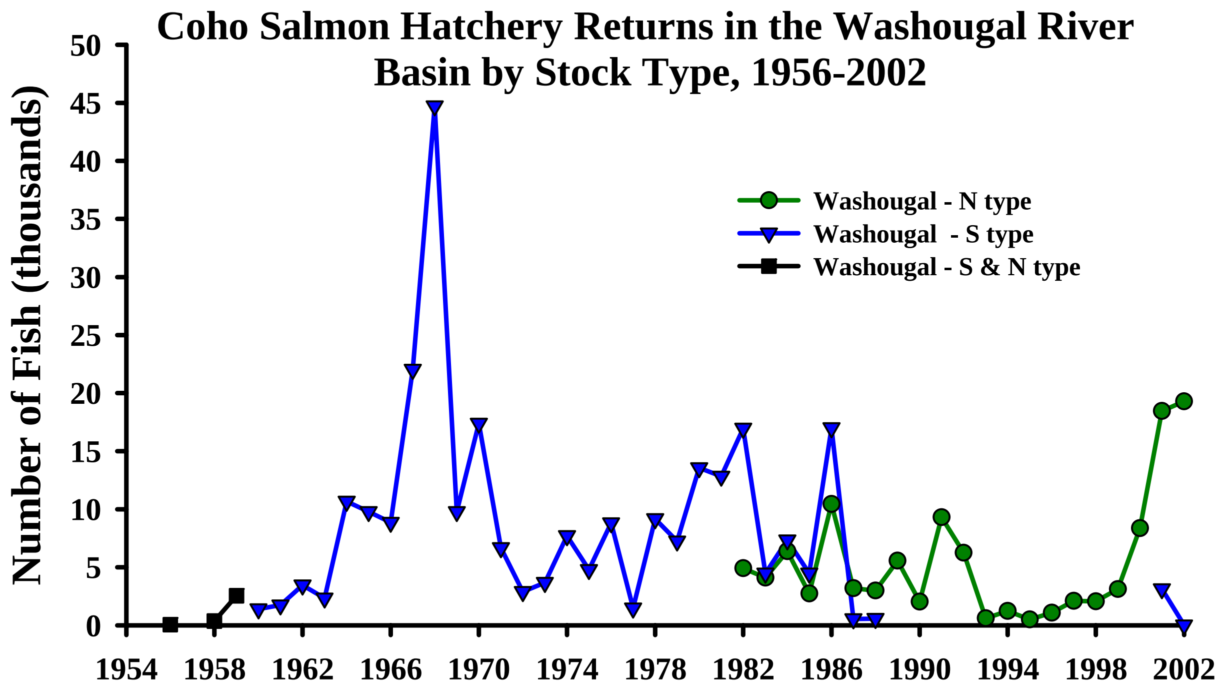

Figure 3-78. Hatchery returns of adult coho salmon in the Washougal River basin by stock type, 1956–2002.

No consistent pattern is apparent in historical hatchery coho returns to the Grays or Elochoman rivers (Figure 3-79 and Figure 3-80); generally, hatchery returns in the 1960s and 1970s were higher than those in the 1980s and 1990s.

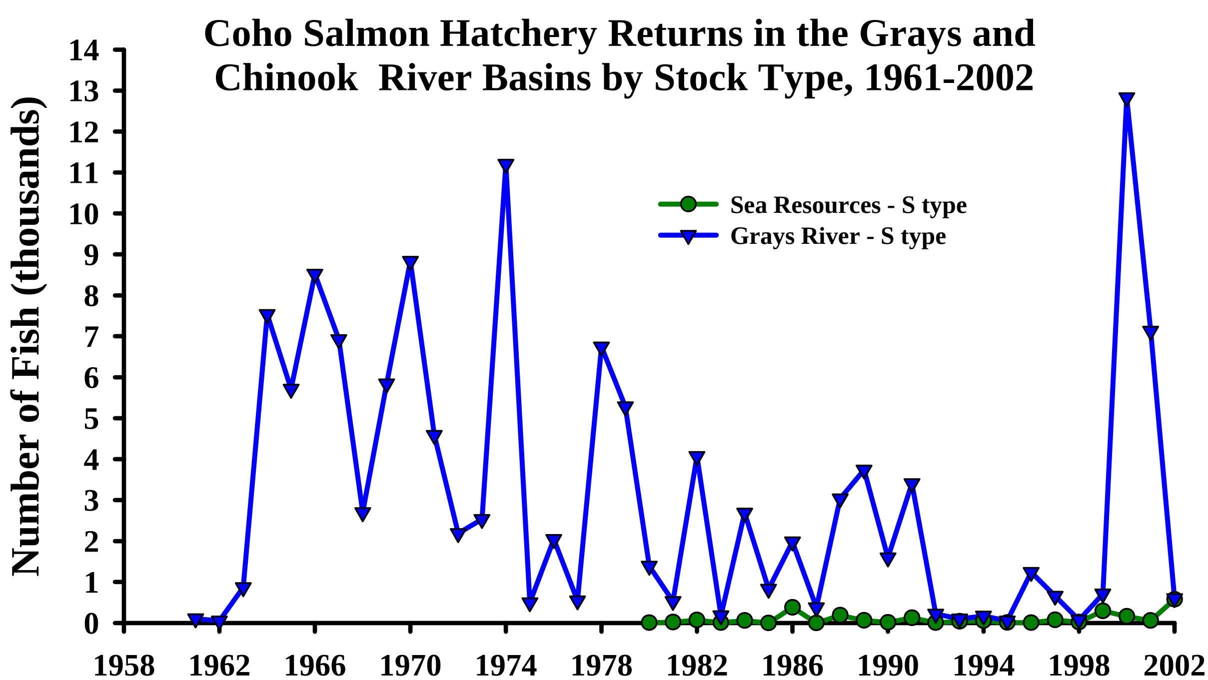

Figure 3-79. Hatchery returns of adult coho salmon in the Grays and Chinook River basins by stock type, 1961–2002.

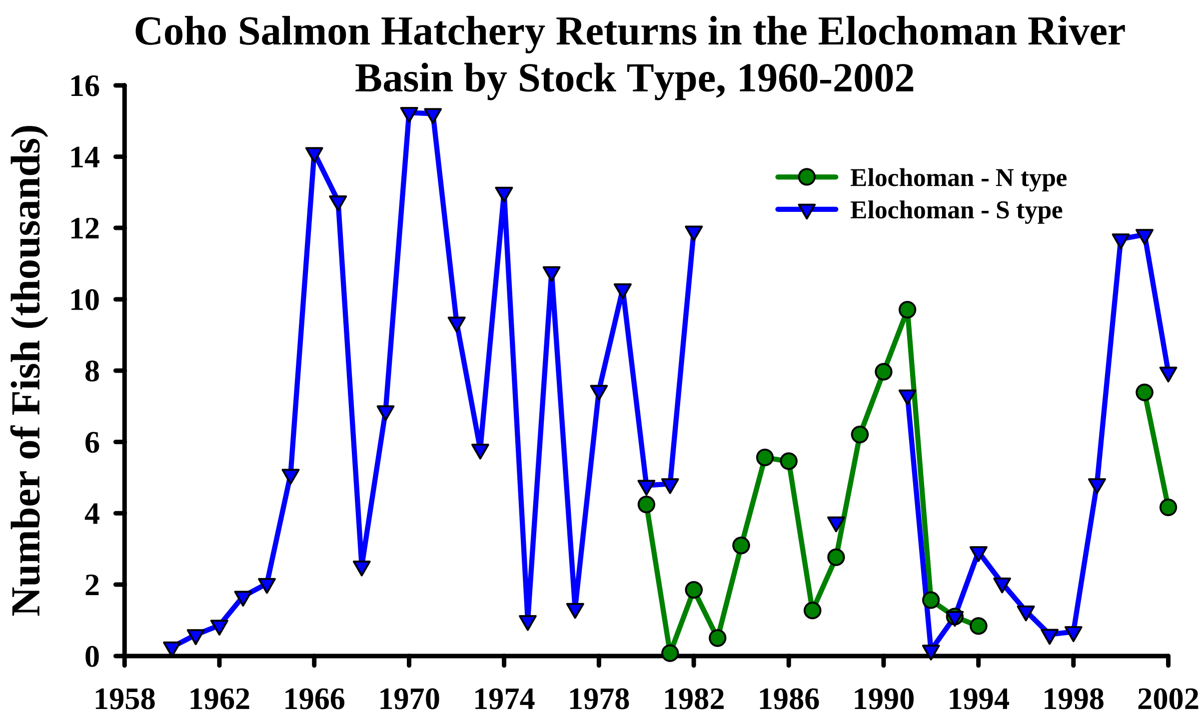

Figure 3-80. Hatchery returns of adult coho salmon in the Elochoman River basin by stock type, 1954–2002.

3.2.3.4 Chum Salmon

The number of hatchery chum salmon produced in Washington is generally small relative to the number naturally produced, and very small compared to the number of hatchery chum salmon produced annually in other areas such as Japan (2 billion) or Alaska (over 450 million; Salo 1991, McNair 1996). In the early 1900s, hatchery managers in the lower Columbia River made little effort to collect chum salmon as the stock declined, primarily because of their low market value in the commercial fishery. The majority of eggs were collected at the Lewis River Hatchery (up to 750 females being spawned in any one year). However, transfers of hatchery chum from outside the Columbia River basin were substantial. During 1913-18, some 30 million chum fry (predominantly from the Chehalis River) were released throughout the Columbia River, including sites above Celilo Falls, the Methow and Walla Walla rivers. At that time, hatchery practices emphasized releasing unfed fry and the success of many of these transfers, especially those far upriver, is doubtful.

Later introductions of non-native chum to lower Columbia tributaries also were generally unsuccessful. For instance, eyed chum eggs of non-local origin were introduced into Spring Channel, a tributary to Hamilton Creek, in the 1970s with no apparent increase in adult production. Several attempts also were made to augment natural chum production in Grays River with releases from the Grays River Hatchery. Releases from 1982 to 1991 included juveniles resulting from small numbers of adults trapped at Grays River Hatchery and chum of Hood Canal and Japanese origin. Hatchery releases have failed to produce significant adult returns (WDF et al. 1993). The average total number of released juvenile hatchery chum salmon is well below the releases of other salmonid species (Figure 3-81).

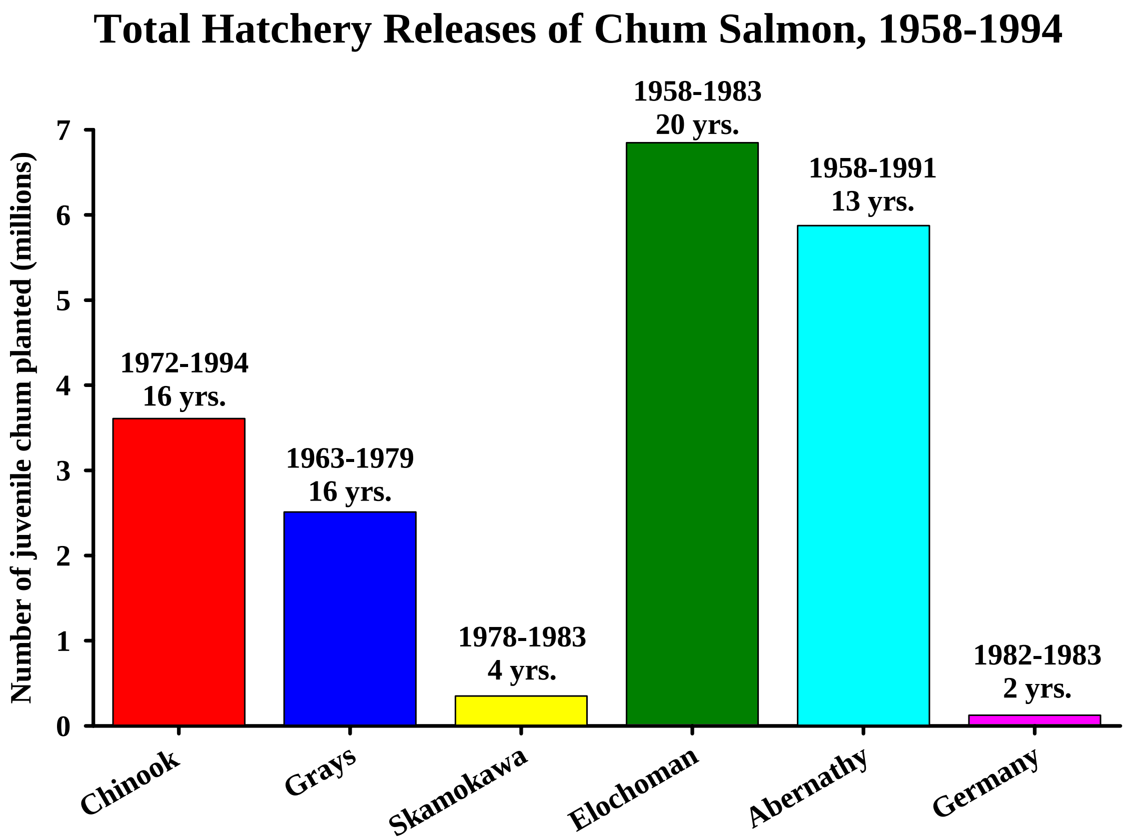

Figure 3-81. Total historical hatchery releases of chum salmon into the lower Columbia basins.

There are currently three hatchery programs in operation in the lower Columbia (Table 3-39). Releases of chum fry from the Grays River Hatchery into the Grays River continues at a level of 300,000 per year. In 2002, natural spawn returns to the Grays River basin increased substantially to about 10,000 adults. The hatchery fish were otolith marked, so it will be possible to estimate the proportion of hatchery fish in the return based on samples collected annually on the spawning grounds. Sea Resources Hatchery currently releases 147,500 Grays River stock chum per year into the Chinook River. A new chum hatchery program has commenced at the Washougal hatchery and is part of a multi-faceted lower Gorge natural spawning chum maintenance and enhancement project. The project is aimed at restoration of natural chum in Duncan Creek and also utilizes the Washougal hatchery to support chum production in the mainstem Columbia near Ives Island during years when mainstem Columbia flows are not adequate to fully support spawners, and in Hamilton and Hardy creeks in years when mainstem and/or tributary flows may compromise adult entry into the streams or passage to the most productive spawning areas.

Table 3-39. Current (2003 brood) chum juvenile hatchery release goals for Washington lower Columbia tributaries.

Basin	Brood Source	Release Goal
Columbia & Tribs near Bonneville Dam	Washougal Hatchery	100,000
Grays	Grays River Hatchery	300,000
Chinook	Grays River Hatcher	147,500
Lower Columbia Total		547,500

Historical hatchery returns of chum salmon in the lower Columbia River have generally been below 500 fish per hatchery, except for 3 years in the early 1990s at the Sea Resources Hatchery and in 1999 at the Grays River Hatchery (Figure 3-82).

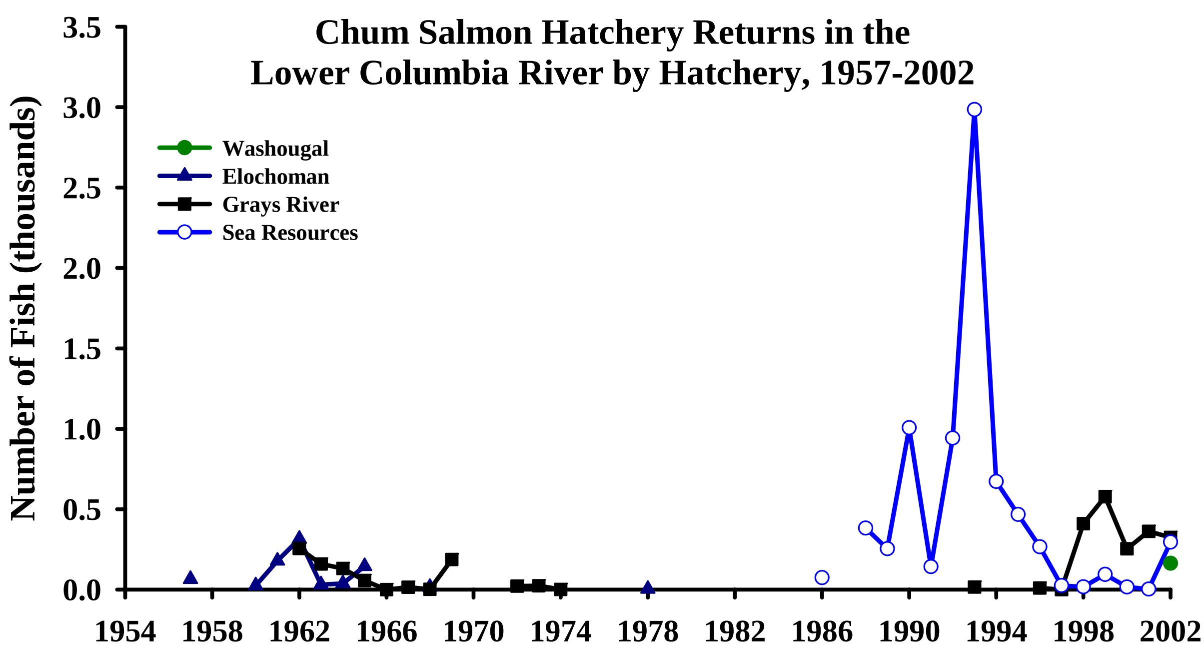

Figure 3-82. Chum salmon hatchery returns in the lower Columbia River by hatchery, 1957–2002.

Chum salmon have been the primary species raised by the Sea Resources Hatchery, on the Chinook River. This hatchery had a return of 3,000 fish in 1993, the largest hatchery chum salmon return ever documented in the lower Columbia River. Until recently, the Sea Resources Hatchery raised chum using Bear River (Willapa Bay) stock. This non-native stock has now been replaced with local stocks from nearby Grays River (Keller 1999). The Sea Resources Hatchery final returns of the Bear River stock in 1997 and 1998 were 11 and 17 chum, respectively (Keller 1999). In 1999, 60,000 Grays River-stock chum fry were released into the Chinook River from the Sea Resources Facility resulting in 600 three-year-old adults that returned in 2002.

3.2.3.5 Summer Steelhead

Hatchery releases of summer steelhead occur in the Elochoman, Cowlitz, NF Toutle (released at mouth of Green River), SF Toutle, Kalama, NF Lewis, EF Lewis, and Washougal rivers. Approximately 1 million summer steelhead smolts are released annually within the lower Columbia River ESU (Table 3-40).

Table 3-40. Current (2003 brood) summer steelhead smolt release goals.

Basin	Brood Source	Release Goal
Washougal	Skamania Hatchery	60,000
NF Lewis	Merwin Hatchery	175,000
NF Lewis	Skamania Hatchery	50,000
EF Lewis	Skamania Hatchery	25,000
Kalama	Skamania Hatchery	30,000
Kalama	Kalama Wild	60,000
NF Toutle	Skamania Hatchery	25,000
SF Toutle	Skamania Hatchery	25,000
L Cowlitz	Cowlitz Trout Hatchery	500,000
Elochoman	Merwin Hatchery	30,000
Lower Columbia Total		980,000

The NMFS status review of West Coast steelhead identified several steelhead broodstocks that have been widely used and have the greatest potential to affect native steelhead populations because of their broad distribution and extensive incorporation in artificial propagation programs (Busby et al. 1996). Among these broodstocks is the Skamania summer steelhead stock. This stock was developed in the late 1950s at the Skamania Hatchery from Washougal and Klickitat river summer steelhead. The Skamania Hatchery is located about one mile from the mouth of the WF Washougal River. Skamania summer steelhead stock has been released throughout Washington, Idaho, Oregon, California, Indiana, Rhode Island, and North Carolina (Crawford 1979, CDFG 1994). In many cases, Skamania summer steelhead have been introduced to provide angling opportunities where summer steelhead did not naturally exist. However, in the Columbia River, Skamania summer steelhead have been released in basins having endemic summer steelhead populations.

In a recent report of steelhead in the lower Columbia River ESU (WLC-TRT unpublished), the fraction of hatchery fish in the escapement over the last 4 years was calculated for some lower Columbia River basins. For the entire lower Columbia ESU (including Oregon basins), the hatchery fraction of spawners was 24%. For Washington basins, the highest hatchery fractions were observed in the Kalama River summer steelhead (35%) and the Wind River summer steelhead (21%). The hatchery fractions were not calculated for NF Lewis, EF Lewis, and Washougal summer steelhead because of a lack of data.

Most summer steelhead programs (i.e. Kalama, Lewis, Washougal, and Wind River basins) have released fewer than 300,000 juveniles annually during the past 20 years (Figure 3-83). The Cowlitz summer steelhead hatchery program released 500,000 or more juveniles and, when including fingerling plants in lakes to provide recreational fishery opportunity, the 2002 releases totaled almost 1.5 million.

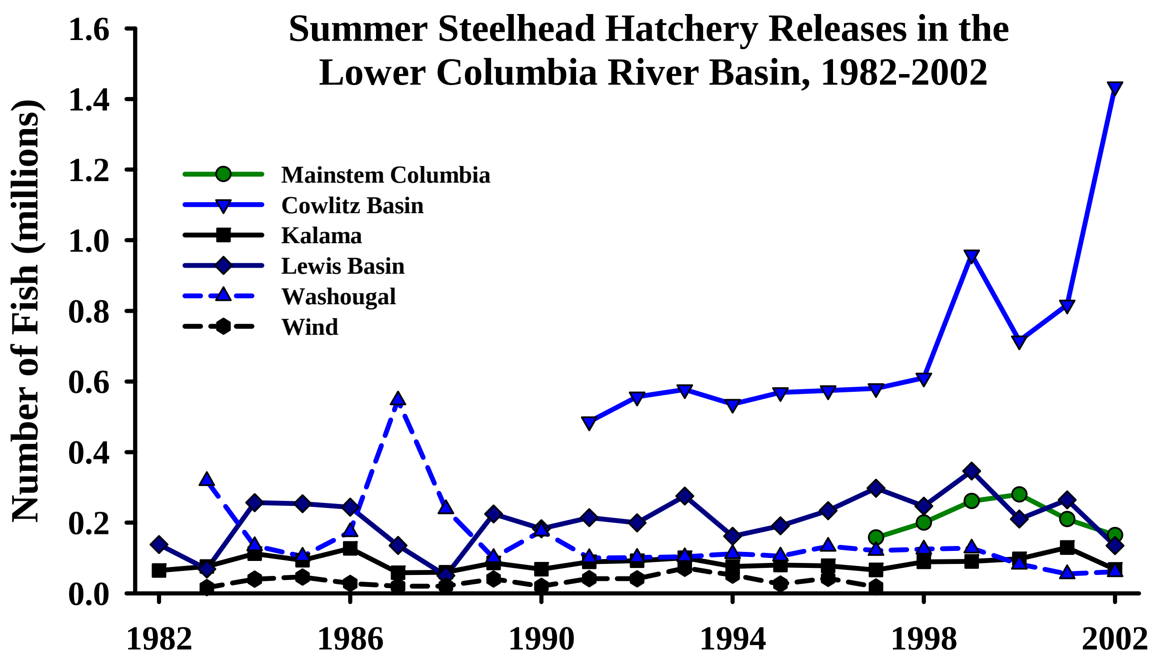

Figure 3-83. Hatchery releases of summer steelhead in the lower Columbia River by basin, 1982–2002.

Hatchery summer steelhead are widespread and escape to spawn naturally throughout the region. Though hatchery-origin fish contribute substantially to natural production, wild summer steelhead are purported to be reproductively isolated from hatchery fish by spatial and temporal differences. There is overlap, however, between summer and winter steelhead spawn time (WDFW 1993) and some stocks appear to have had substantial hatchery contribution to wild spawning (e.g., Kalama winter and summer steelhead). Nehlsen et al. (1991) identified several stocks from the Lower Columbia and Southwest Washington ESUs as a special concern because of hatchery influence. The impacts of hatchery fish on wild stocks have been studied in the Kalama basin. Skamania hatchery summer steelhead have in the past comprised around 75% of total summer spawning escapement in the Kalama system (WDFW 1993). Even though 40% of returning naturally produced adults are estimated to have at least one parent of hatchery origin, the wild stock has retained genetic traits of considerable adaptive value relative to hatchery stock (Leider et al. 1995, WDFW 1993).

Adult summer steelhead returns to the lower Columbia hatchery facilities are highly variable and are dependent on variable smolt to adult survival as well as variable sport fishery harvest rates. The returns to the hatchery racks only represent a part of the hatchery returns, as the tributary sport fisheries are significant for hatchery steelhead, typically harvesting 50% or more of the fish returning to the rivers (WDFW 2003).

- Hatchery summer steelhead returns to the Cowlitz Salmon Hatchery have been variable, ranging from about 250 to 3,500 and returns to the Cowlitz Trout Hatchery have ranged from about 1,000 to 3,000 (Figure 3-84). Although the period for the return data set is short, the trend in returns through the latter part of the 1990s appears to be decreasing. Summer steelhead returns to the North Toutle Hatchery are low, primarily due to the program being developed to rely on releases from other stations as well as an effective tributary sport harvest.

- Hatchery summer steelhead returns to the Kalama River were variable in the latter part of the 1990s (Figure 3-84). Although the period for the return data set is short, the trend in returns appears to be decreasing.

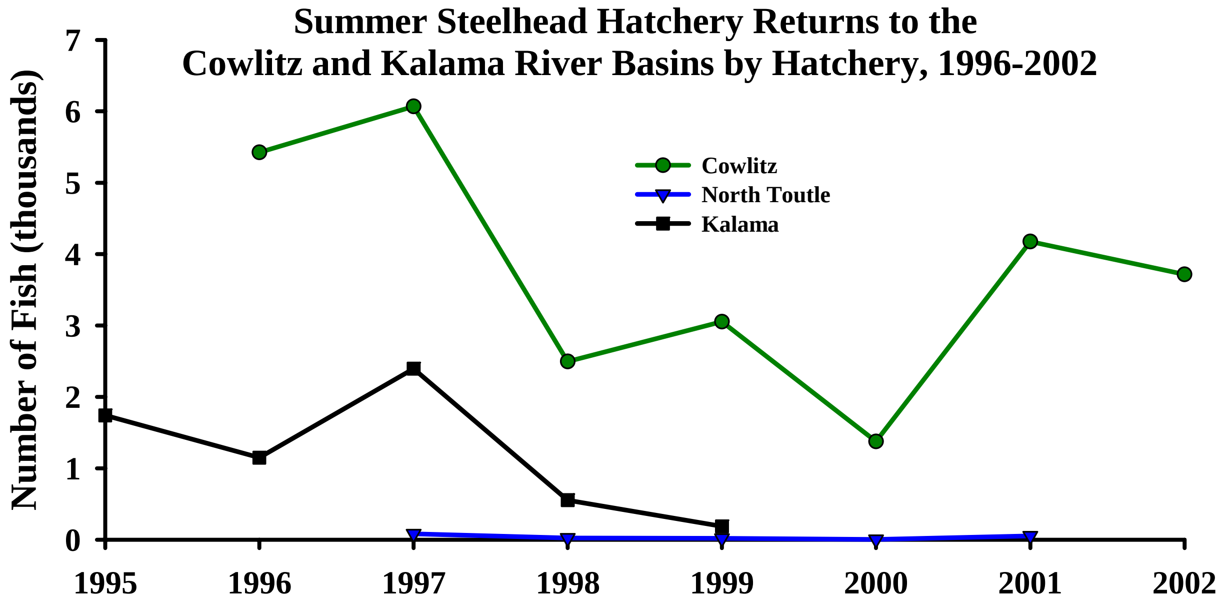

Figure 3-84. Hatchery returns of summer steelhead to the Cowlitz and Kalama River basins by hatchery, 1996–2002.

- In the Lewis basin, hatchery returns also have been variable in recent years. Summer steelhead hatchery returns to the Lewis River basin have typically ranged between 1,000 and 3,000 fish (Figure 3-85).
- In the Washougal basin, returns to the Skamania Hatchery are usually between 500 and 3,000 summer steelhead, though there is little consistency in returns from year to year.

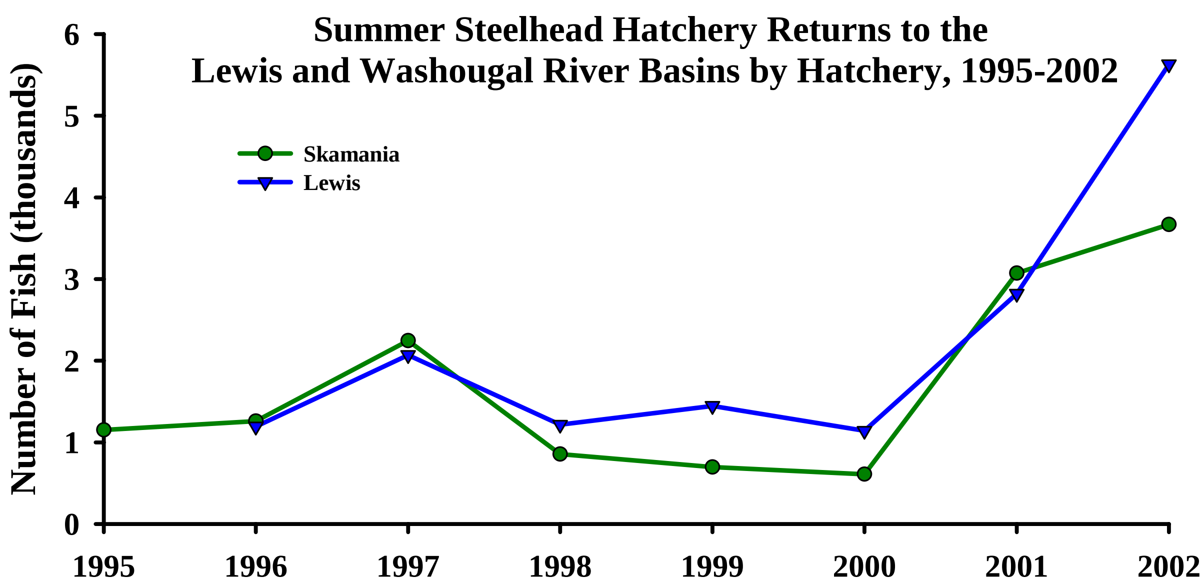

Figure 3-85. Hatchery returns of summer steelhead to the Lewis and Washougal River basins by hatchery, 1995–2002.

3.2.3.6 Winter Steelhead

Hatchery releases of winter steelhead occur in the Grays, Elochoman, Cowlitz, Tilton, Coweeman, Kalama, NF Lewis, EF Lewis and Washougal rivers, and in Salmon Creek. The current (2003 brood) goal is to release 1.2 million winter steelhead smolts and 350,000 subyearling winter steelhead into lower Columbia River tributaries (Table 3-41).

Table 3-41. Current (2003 brood) Winter steelhead smolt release goals.

Basin	Brood Source	Release	
		Yearling	Subyearling
Washougal	Skamania Hatchery	60,000	
Salmon Creek	Skamania Hatchery	20,000	
NF Lewis	Merwin Hatchery	100,000	
EF Lewis	Skamania Hatchery	90,000	
Kalama	Kalama Falls Hatchery	45,000	
Kalama	Kalama Late Wild	45,000	
Coweeman	Elochoman Hatchery	20,000	
L. Cowlitz	Cowlitz Trout Hatchery	300,000	
L. Cowlitz	Cowlitz Trout Hatchery (late winter)	352,500	
Upper Cowlitz	Cowlitz Trout Hatchery (late winter)	37,500	250,000
Tilton	Cowlitz Trout Hatchery (late winter)		100,000
Elochoman	Elochoman Hatchery	60,000	
Elochoman	Elochoman (late wild)	30,000	
Grays	Elochoman Hatchery	40,000	
Lower Columbia Total		1,200,000	350,000

Hatchery winter steelhead are widespread and escape to spawn naturally throughout the region. Though hatchery-origin fish contribute substantially to natural production, wild winter steelhead are purported to be reproductively isolated from hatchery fish by spatial and temporal differences. The listed populations of winter steelhead in the ESU are generally late-run; returning and spawning later than the early returning Chambers Creek stock (a Puget Sound stock), commonly used for many of the hatchery programs. (The Chambers Creek Hatchery broodstock originated from Chambers Creek (Tacoma, Washington) in the 1920s. Hatchery winter steelhead from Chambers Creek have been released throughout the lower Columbia River.) There is also overlap, however, between summer and winter steelhead spawn time (WDFW 1993) and some stocks appear to have had substantial hatchery contribution to wild spawning (i.e., Kalama winter and summer steelhead).

The Beaver Creek Hatchery (on a tributary of the Elochoman River) formerly produced approximately 400,000-500,000 winter steelhead smolts annually. The hatchery has utilized broodstock from the Elochoman and Cowlitz rivers and Chambers Creek. Smolts from the Beaver Creek Hatchery have been planted throughout the lower Columbia River. Beaver Creek Hatchery was closed in 1999 due to Mitchell Act funding shortfalls.

In a recent status report of steelhead in the lower Columbia River ESU (WLC-TRT unpublished), the fraction of hatchery fish in the escapement over the last 4 years was calculated for some lower Columbia River basins. For the entire lower Columbia ESU (including Oregon basins), the hatchery fraction of spawners was 24%. For Washington basins, the highest hatchery fraction was observed in the Coweeman River winter steelhead (50%). The winter steelhead hatchery fraction was estimated as <2% for a number of stocks: SF Toutle River (2%), NF Toutle River (0%), Kalama River (0%), EF Lewis (0%), and Washougal River (0%).

- In the Elochoman basin, approximately 100,000 winter steelhead have been released annually, although the 2000 releases totaled approximately 350,000 juveniles (Figure 3-86). The current Elochoman program includes Elochoman Hatchery (Beaver Creek) origin winter steelhead smolt releases as well as a Elochoman late wild winter steelhead smolt releases. Annual winter steelhead releases to the Grays River, Skamokawa Creek, and other lower Columbia tributaries (Mill, Germany and Abernathy creeks) have generally been less then 50,000 juveniles annually. Winter steelhead are no longer released into Skamokawa, Mill, Germany and Abernathy creeks.

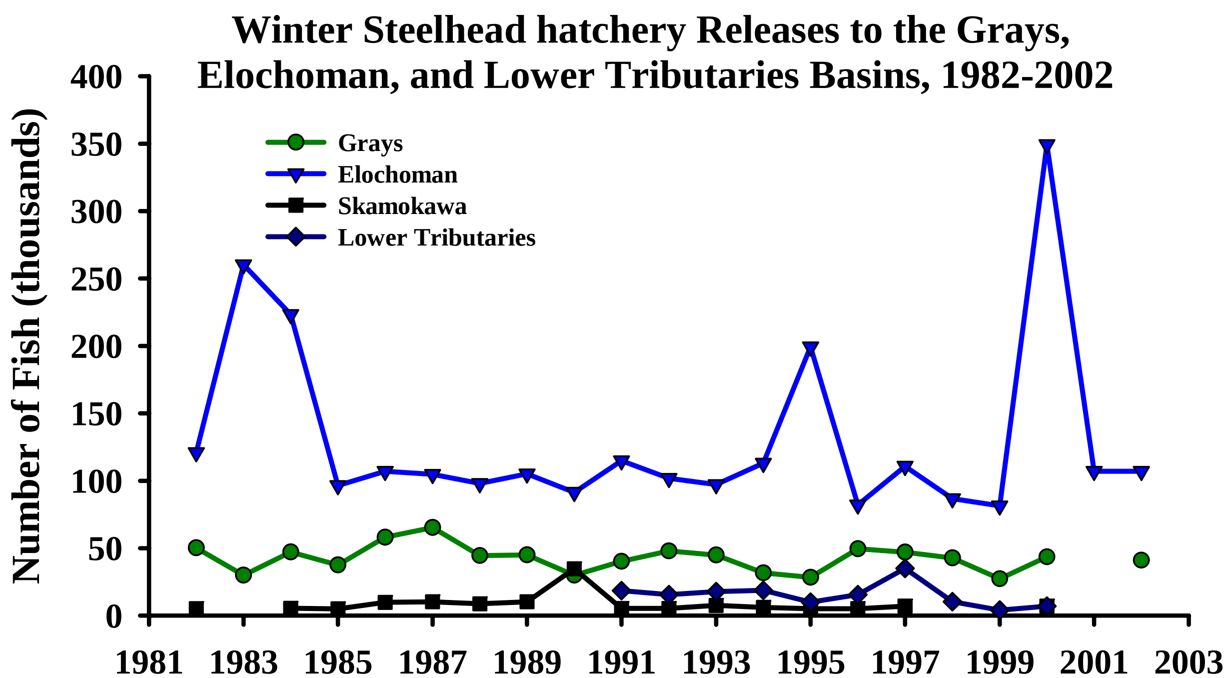

Figure 3-86. Hatchery releases of winter steelhead to the Grays, Elochoman, and lower tributaries basins, 1982–2002.

- The Cowlitz Trout Hatchery, located on the mainstem Cowlitz at RM 42, has two programs producing winter steelhead: an early winter stock derived from Cowlitz River and Chambers Creek stock and a late winter Cowlitz stock used to reintroduce natural production in the upper Cowlitz. Smolts from the Cowlitz Trout Hatchery have been planted throughout the region. In some cases, the influence of hatchery winter steelhead is pronounced. For example, Cowlitz River "wild" winter steelhead are almost all the progeny of Cowlitz Hatchery winter steelhead (WDF 1993). Total winter steelhead annual releases to the Cowlitz River have considerably exceeded releases to other lower Columbia River basins. Releases into the Cowlitz basin were generally over 1 million annually throughout the 1990s (Figure 3-87).

- In the Kalama basin, Gobar Pond (four miles up Gobar Creek at RM 19.5) has been utilized as an acclimation site for hatchery steelhead before their release. Yearling hatchery winter steelhead from the Cowlitz or Beaver Creek hatcheries have been released into Gobar Pond for subsequent release to the Kalama basin. Approximately 100,000 hatchery winter steelhead smolts are released in the Kalama River basin annually, except for a release of approximately 300,000 smolts in 2000 (Figure 3-87). The Kalama Falls Hatchery is continuing a research program to investigate the effectiveness of using naturally produced late-run winter steelhead for broodstock to replace non-listed, early-run winter steelhead from Beaver Creek.

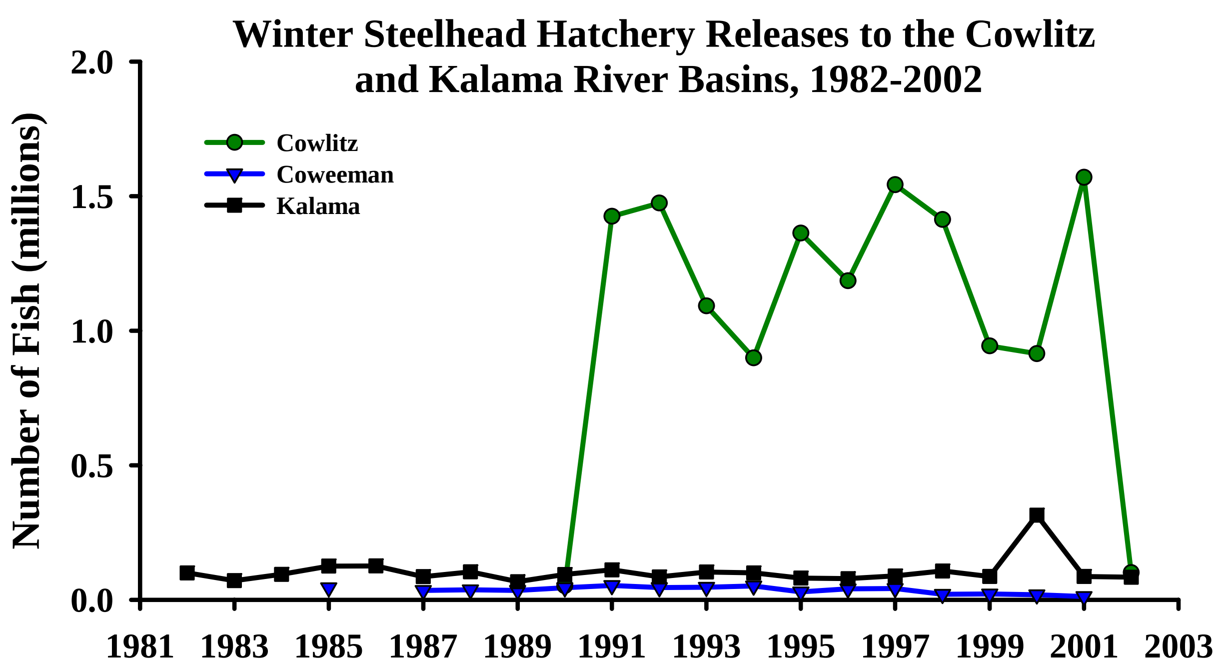

Figure 3-87. Hatchery releases of winter steelhead to the Cowlitz and Kalama River basins, 1982–2002.

- In the Lewis basin, a net pen system has operated on Merwin Reservoir since 1979; annual winter steelhead smolt production has averaged 35,000 fish. The source of the broodstock is from the Merwin Dam trap. Merwin Hatchery (just downstream from Merwin Dam) has produced winter steelhead since the early 1990s and hatchery fish are released primarily within the Lewis River basin. Releases in the EF Lewis have averaged about 100,000 juveniles annually, while releases to the NF Lewis have been slightly higher (Figure 3-88). The 2003 releases were approximately 80,000 smolts into the EF Lewis and 100,000 into the mainstem Lewis from the Island Boat Launch in the lower river. An additional 90,000 Skamania stock winter steelhead were released into the EF Lewis River in 2003.
- Approximately 20,000 winter steelhead also are released annually into Salmon Creek near Vancouver.
- In the Washougal River basin, winter steelhead annual release numbers have been variable, with up to about 180,000 fish released in the late 1980s and approximately 60,000 fish released in recent years.

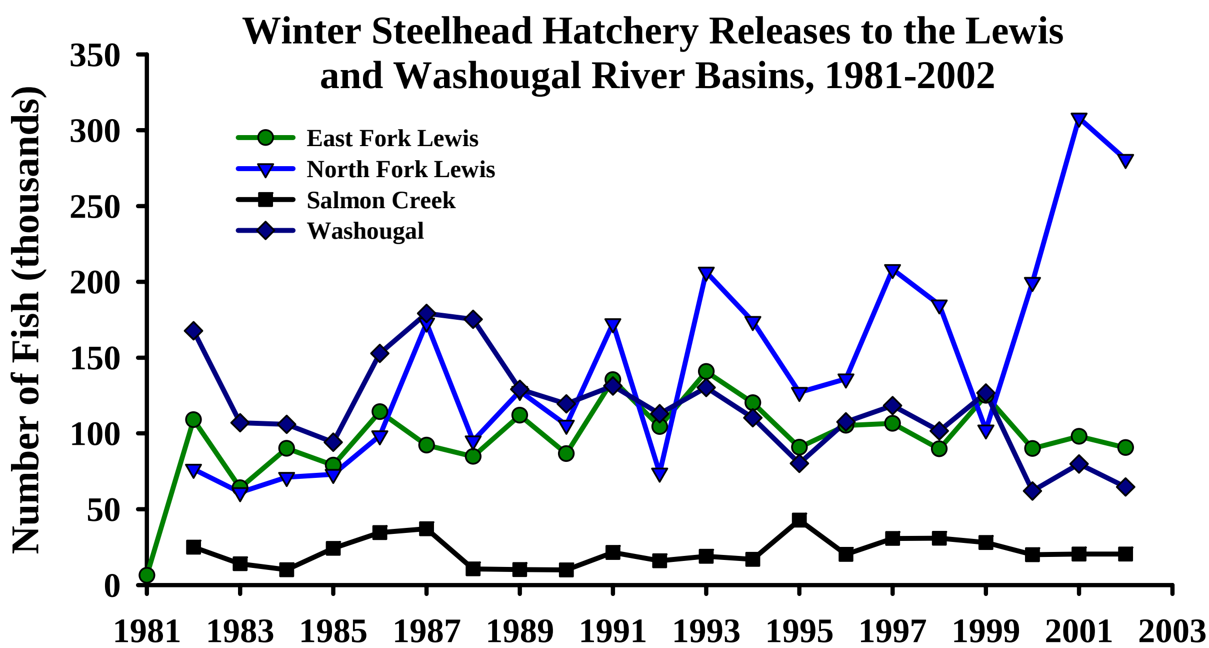

Figure 3-88. Hatchery releases winter steelhead to the Lewis and Washougal River basins, 1981–2002.

In recent years, winter steelhead hatchery returns to most Washington hatcheries in the lower Columbia basin have been below 1,000 fish (Figure 3-89). Notable exceptions include the Cowlitz Salmon and Cowlitz Trout Hatcheries where hatchery returns were >5,000 fish during 1996 and 1997. For the four hatcheries with returns below the 1,000 fish level, hatchery returns seemed to mirror one another (i.e. experienced increased or decreased production during same years). The two Cowlitz basin hatchery returns also tracked annually with each other, but did not mirror the returns from the other lower Columbia River hatcheries.

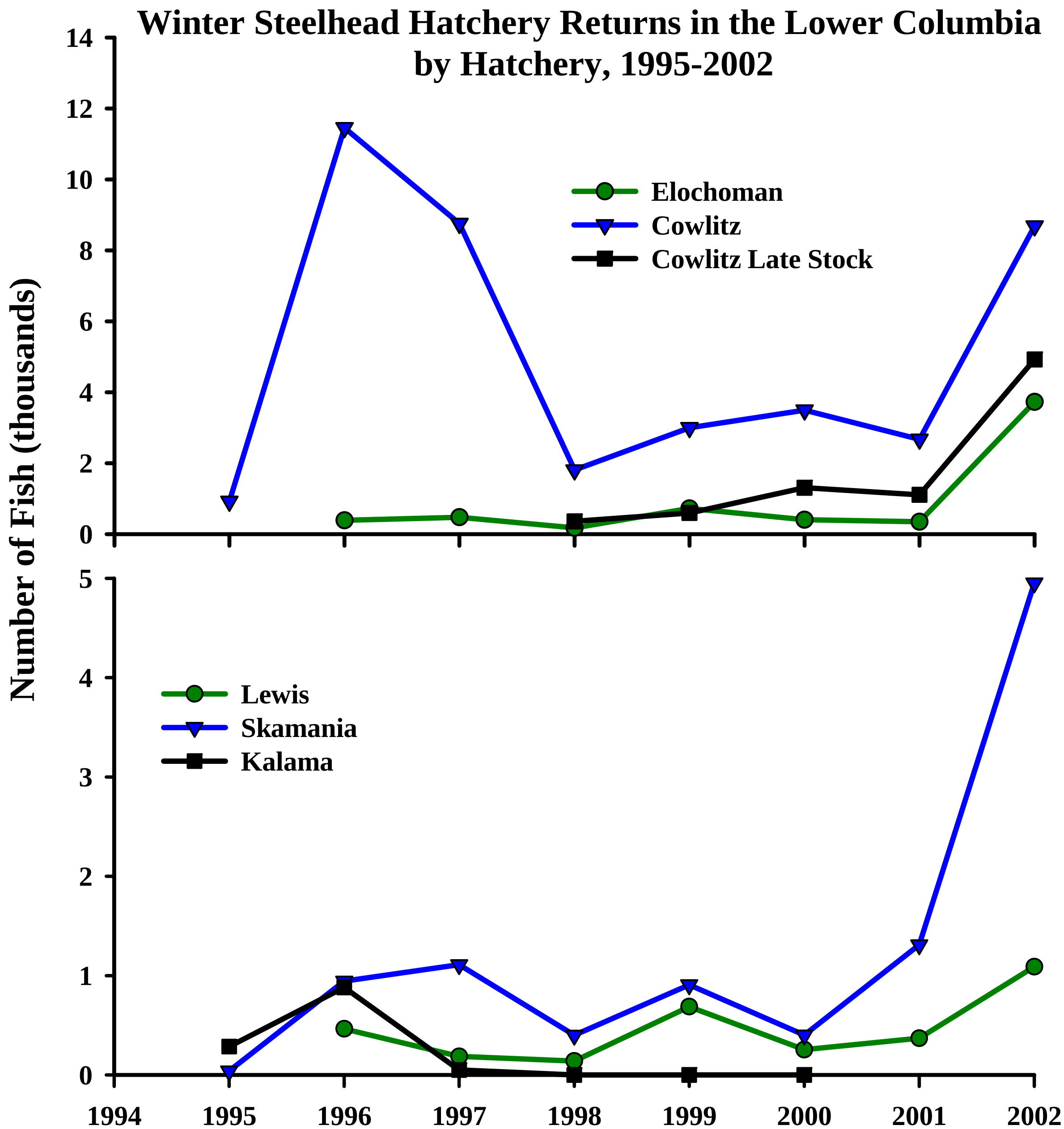

Figure 3-89. Hatchery returns winter steelhead in the lower Columbia River by hatchery, 1995–2002.

3.3 Subbasin Fish Habitat Conditions

Properly functioning stream habitats are critical for recovering and sustaining healthy populations of salmon and trout in the Lower Columbia region. Many essential habitat features have been altered or degraded by human activities such as dams, logging, agriculture, urban development, road building, gravel mining, channelization, and water withdrawals. This section is focused on identifying the habitat factors that generally preclude salmonid populations from attaining their full production potential. The topics are addressed from the viewpoint of the fishes' needs for free access to spawning and rearing habitats, adequate streamflows, good water quality, critical habitats, proper substrates, large woody debris, channel stability, healthy riparian areas, and functional floodplains.

Each of the following limiting factor categories include a general overview of how fish are impacted by the limiting factor and how the factor is influenced by biophysical processes and land-use. A synopsis of current conditions throughout the region is also presented for each category. This broad-scale view of current conditions represents the aggregate of the subbasin-scale information presented in Volume II of this Technical Foundation. The primary information sources include fish habitat modeling (Ecosystem Diagnosis & Treatment) and watershed process modeling (Integrated Watershed Assessment), which are presented in detail in Vol. II. Information was also obtained from Washington State Conservation Commission Limiting Factors Analyses, which were conducted in Water Resource Inventory Areas (WRIAs) 24 - 29 from 1999 through 2002. A map of the WRIA boundaries in the LCFRB planning area can be found in (Figure 3-90). Additional sources of information include various agency reports and databases including USFS Watershed Analyses, the WDOE 303(d) list of impaired water bodies, and the WDFW Salmon and Steelhead Habitat Inventory and Assessment Program (SSHIAP) fish barrier database.

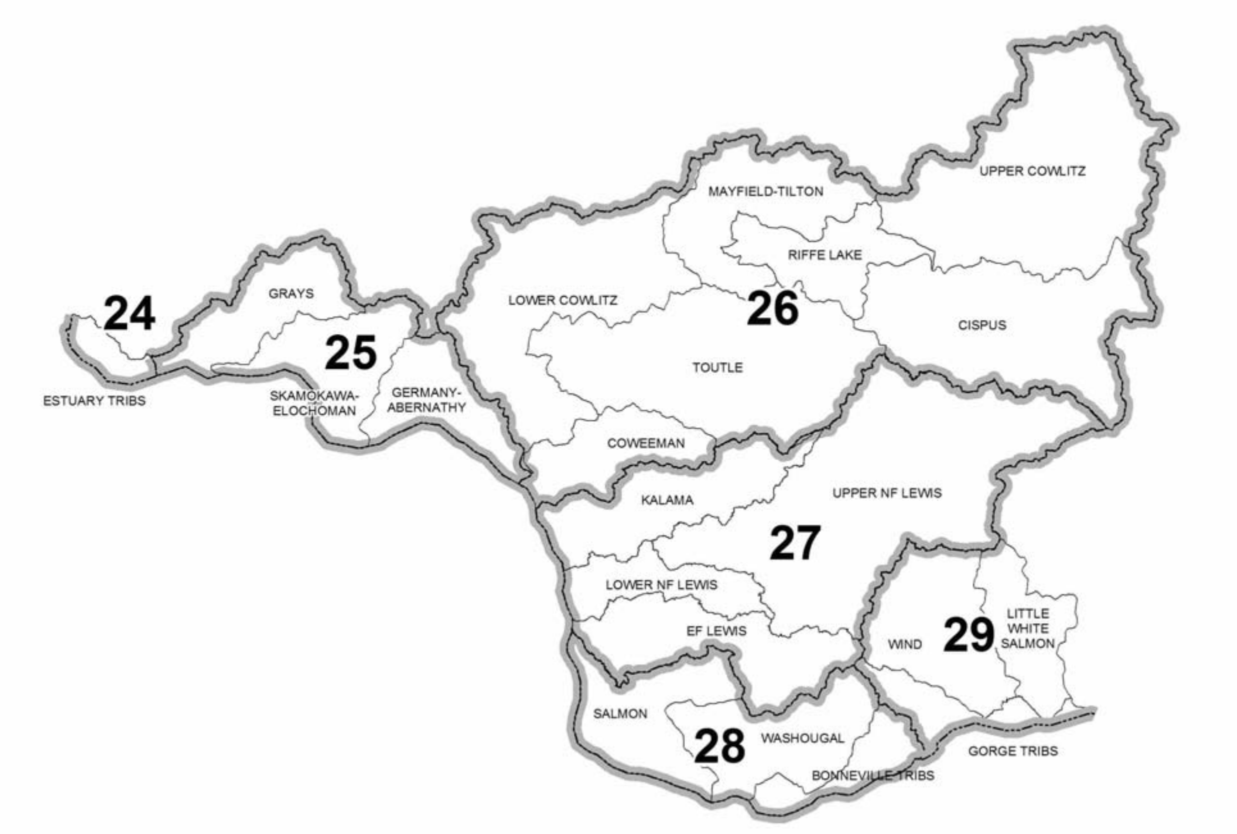

Figure 3-90. Water Resource Inventory Area (WRIA) map encompassing LCFRB planning area.

3.3.1 Passage Obstructions

3.3.1.1 Processes and Effects

Fish passage barriers that limit access to spawning and rearing habitats are a significant factor affecting salmon populations in many Northwest watersheds. Barriers in lower Columbia watersheds primarily include culverts and dams with occasional other barriers, such as irrigation diversion structures, fish weirs, beaver dams, road crossings, tide gates, channel alterations, and localized temperature increases. Passage barriers effectively remove habitat from the subbasin, thereby reducing habitat capacity. In situations where a substantial amount of historic spawning or rearing habitat has been blocked, such as in the Cowlitz or Lewis River subbasins, production potential of salmonid populations have been severely reduced. To some degree, depending on the species, formerly unused downstream habitats may compensate for the lost upstream habitat. For example, chinook or chum salmon may be able to adapt to spawning/rearing in subbasin mainstem habitats below barriers while coho salmon and steelhead are less likely to utilize mainstem habitats because they are more commonly found spawning in headwater portions within the subbasin. However, the degree to which downstream habitats may be utilized after the construction of passage barriers is limited by the downstream effects of those barriers, such as alterations of flow and temperature as a results of hydropower or flood control dam operations.

As early as 1881, Washington enacted legislation to protect fish access to habitat by disallowing the installation of barriers or providing for their removal. Recent efforts include an appropriation by the 1998 state legislature of $5.75 million to inventory and repair barriers throughout the state. Despite these efforts, barriers continue to be a problem in the lower Columbia region.

Although dams are responsible for the greatest share of blocked habitat, inadequate culverts make up approximately 86% of all barriers (WDFW SSHIAP data). Estimates made from culvert surveys throughout the state indicate that approximately half of culvert problems are related to private and public logging roads (State of Washington 1999). The 1950s saw the beginning of extensive road building associated with increased logging activities. Many early logging roads were not outfitted with properly-sized culverts, and despite recent efforts to upgrade critical road crossings, an extensive backlog of passage restoration needs remains.

3.3.1.2 Current Conditions

The major hydropower systems on the Cowlitz and Lewis rivers are responsible for the greatest share of blocked habitat. Culverts and other barriers are also a concern throughout the region. A region-wide view of barriers to anadromous fish and the extent of upstream blocked habitat are depicted in Figure 3-91.

- In the Lewis River basin alone, the 240-foot high Merwin Dam has blocked 80% of the available steelhead habitat since 1931 (WDF/WDW 1993). The dam blocked the majority of the spring chinook habitat as well.
- In the Cowlitz basin, the three mainstem dams inundated a total of 48 miles of historical steelhead, chinook, and coho habitat.
- The Sediment Retention Structure (SRS) on the North Fork Toutle River is a total barrier to salmonids. The Toutle Trap just below the SRS, which is the trapping facility for all salmonids returning to the upper N.F. Toutle River, has been difficult to operate in recent years due to increasing amounts of debris and sediment coming down from the SRS.
- Throughout the region, as many as 800 culverts have been identified that block passage of salmonids. The bulk of these are associated with private and public logging roads.

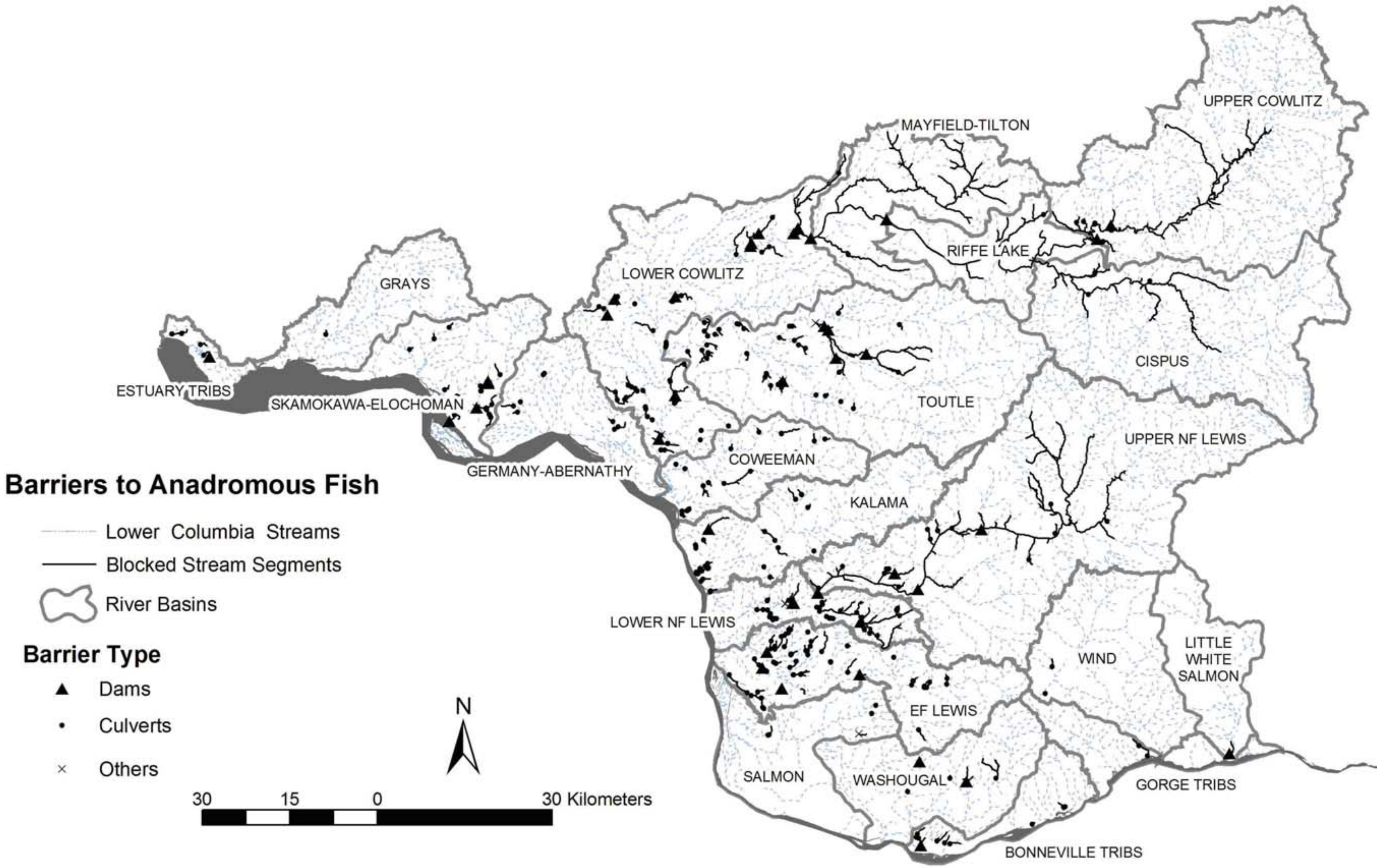

Figure 3-91. Regional map depicting blockages to anadromous fish and the extent of potentially accessible stream segments above blockages. Blockages and potential stream segments are included if passage for any anadromous species is obstructed. The primary source for these data is the Salmon and Steelhead Habitat Inventory and Assessment Project (SSHIAP).

3.3.2 *Stream Flow*

3.3.2.1 Processes and Effects

Stream flow patterns are controlled by local climate, geology, basin topography, land cover, and ocean climate patterns. Two annual stream flow patterns dominate in the Lower Columbia region. High elevation basins typically experience a flow regime dominated by snowmelt, with peak flows occurring during spring melt conditions, whereas lower elevation basins experience winter peak flows as a result of winter rain storms.

Aquatic organisms have adapted to the range of habitat conditions that are created and maintained by natural streamflow regimes (Poff et al. 1997) and a range of streamflows are necessary for creating habitat diversity (Bisson et al. 1997). Streamflows in excess of natural conditions, however, can increase hillslope sediment delivery and alter channel morphology through bed and bank erosion, with subsequent impacts on aquatic habitats (Chamberlain et al. 1991). Alterations to winter and spring flows can affect incubation and emigration survival by increasing the likelihood of scouring eggs and alevins from the gravel or displacing juveniles from rearing habitats (e.g., Pearsons et al. 1992, Montgomery et al. 1996). Decreased summer low flow volumes can impact aquatic habitats through loss of available habitat area and increased risk of elevated stream temperatures. Alterations to summer and fall flows may impact spawner distributions and juvenile rearing success.

Characteristics of catchment land cover influence the rate, duration, and magnitude of water runoff in a basin. In the Pacific Northwest, alterations of land cover affect runoff by decreasing soil infiltration rates, interrupting subsurface flow, and increasing snow accumulation and melt rates.

Although western Washington is characterized as having abundant rainfall, a significant portion of annual precipitation is lost as evapo-transpiration due to the dense forest cover. Precipitation that is not lost to evapo-transpiration or deep groundwater storage enters streams via three primary methods:

- surface flow (rapid),
- shallow subsurface flow (slow), and
- groundwater flow (very slow).

In undisturbed basins in the Pacific Northwest, shallow subsurface flow accounts for nearly all of the runoff entering stream channels, except during periods of low flow when groundwater sources dominate (Ziemer and Lisle 1998). The lack of surface runoff in an undisturbed basin is due to the rate of infiltration exceeding precipitation. If the infiltration rate is changed, then precipitation that normally transmits slowly to stream channels as subsurface flow or that contributes to groundwater storage is instead rapidly transported as surface flow. This can decrease the amount of groundwater available to supply flow to streams in dry periods and can increase the magnitude and rate of peak flows during storm events. These conditions are especially prevalent in urbanizing basins, where native vegetation has been converted to impervious surfaces such as pavement, rooftops, and lawns (Leopold 1968, Fresh and Luchetti 2000). The drainage network in the form of gutters, drains, and storm sewers further increases the magnitude and rate of delivery of storm flows to downstream channels. Previous studies have indicated that 10-20% impervious area in a basin can alter stormflow volumes (Hollis 1975) and severely impact aquatic systems (Booth and Jackson 1997).

Infiltration rates are also decreased due to timber harvest operations, forest road building, and conversion of forest land to agriculture. Interception of subsurface flow due to forest road cuts is another major source of runoff manipulation. Excavation of road cuts on hillslopes penetrates the soil mantle, redirecting shallow subsurface flow into road ditches, which accelerates the delivery of water to stream channels.

Streamflow volumes may also be increased due to forest practices that increase snow accumulation and melt rates. Forest canopies naturally intercept snowfall, much of which melts in the canopy and reaches the forest floor as wet snow or meltwater (Ziemer and Lisle 1998). Removal of canopy cover increases the amount of snow that accumulates. In addition, melt rates may be increased due to the convective transfer of heat to the snow surface during storm events. In this way, the water available for runoff may be increased during rain-on-snow events (Coffin and Harr 1992).

3.3.2.2 Current Conditions

Stream flow impairment is difficult to assess without a sufficiently long time series of flow records, and even with such information, it is often difficult to distinquish true flow alterations from natural fluctuations. For this reason, land cover conditions that are known to influence the timing, rate, magnitude, and duration of stream flows are often used as indicators of potential stream flow impairment. These generally include one or more of the following metrics: forest seral-stage, percentage watershed imperviousness, and road density.

- The Integrated Watershed Assessment (IWA) identified hydrologic (runoff) impairments across the study area according to landscape characteristics including impervious surfaces, vegetation cover, and road densities (see Vol. II for presentation of subbasin-level results). IWA hydrology impairment results are depicted for the entire region in Figure 3-92. The greatest impairments are located in lower elevation portions of the basins, which are dominated by private timber lands. Functional conditions are most prevalent in upper watersheds in public land.
- Fish habitat modeling suggests that stream flow impairments are limiting fish production in many basins. The most impacted reaches are located in middle and upper basin areas within or downstream of areas with intensive timber harvest and road building activities.
- The Vancouver metropolitan area, along with the cities of Camas and Washougal, comprise the largest urban area in Southwest Washington and are located primarily in the Lake River/Salmon Creek and Washougal River basins in WRIA 28. Of land area in WRIA 28, 13% is urban land, with 20% in agricultural uses (WDOE WRIA data). These areas have high degrees of imperviousness with a substantial loss of native forests and wetlands. Urban development plays a relatively minor role throughout the remainder of the region. WRIAs 25 (Grays/Elochoman), 26 (Cowlitz), 27 (Lewis), and 29 (Wind) each have less than 2% of the land area in urban uses.
- Forest lands have received significant alteration, particularly those in the western portion of the region and those in lower elevation areas that are in private commercial timber land ownership. In WRIA 25, 79% of land area is forest land, and 83% of the land is private. This WRIA has received intensive timber harvests over the past 50 years. On the whole, WRIAs

26, 27, and 29 have received less alteration to forest lands, attributable to more than 40% of their land area in federal ownership.

- Many forest stands have been clearcut and are in early seral stages, with over 20 (or 3.5%) of 567 7th field HUCs having over 20% of forest cover in early seral stages, and a few of these have over 40% in early seral stage conditions.
- The preponderance of roads in the region is another major influence on runoff conditions. There are approximately 24,000 miles of roads in the region, and the region has an average road density of 4.15 mi/sq mi. In many basins the forest road density exceeds 7 mi/sq mi.
- Analyses by the USFS on national forest lands in many upper basins indicate a risk of increased peak flows for moderate return interval flows (i.e. 2-year flow), attributed primarily to forest practices activities.
- Peak flow reductions created by the Cowlitz and Lewis River hydropower systems limit the potential for scour of salmon redds in downstream channels, however, these flow alterations may also limit the occurance of channel-forming flows that may be important for the maintenance of key habitat types.
- Instream flow assessments, including primarily the Toe-Width method, were applied to many lower Columbia streams in the fall of 1998 (Caldwell et al. 1999).[2] Most of these analyses indicated sub-optimal flows for both spawning and rearing life stages.

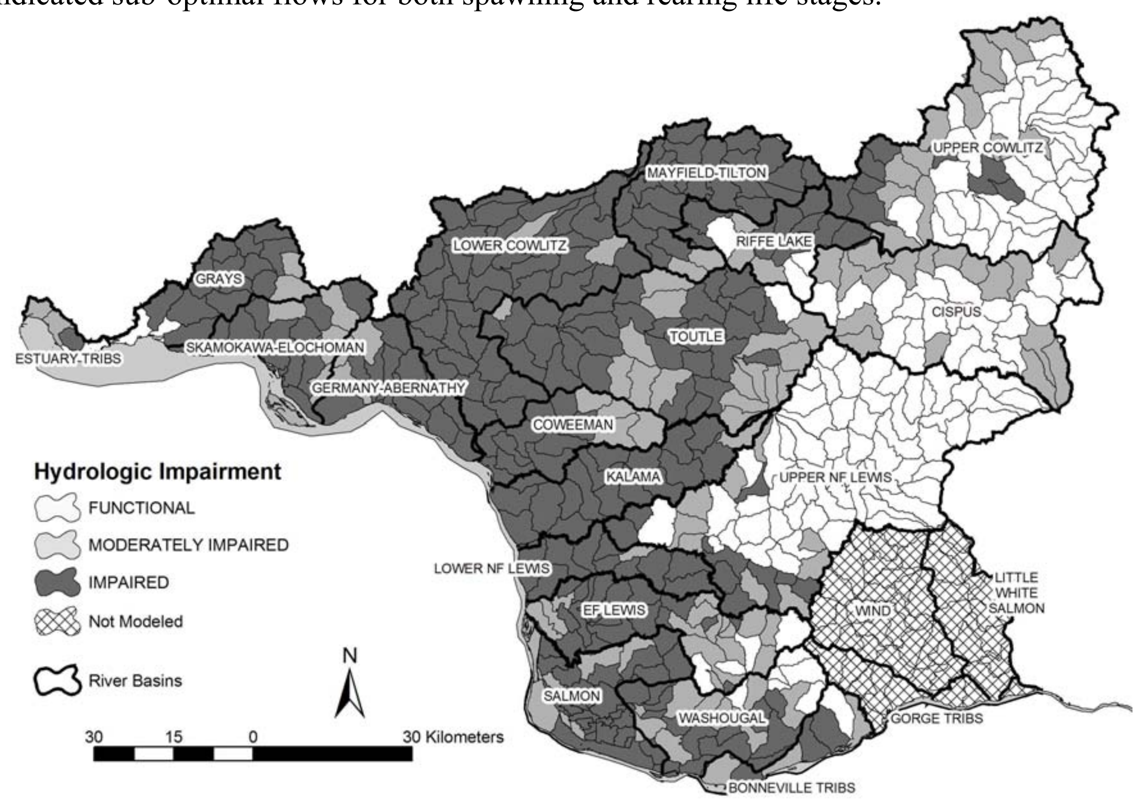

Figure 3-92. Map of hydrologic impairments across the lower Columbia region. Impairment categories were calculated as part of the Integrated Watershed Assessment (IWA). (see Vol. II for presentation of subbasin-level results). These impairment ratings represent local hydrology (runoff) conditions, not including upstream effects.

[2] The Toe-Width is the distance from the toe of one streambank to the toe of the other streambank across the stream channel. This width of the stream is used in a power function equation to derive the flow needed for spawning and rearing salmon and steelhead.

3.3.3 *Water Quality*

3.3.3.1 Processes and Effects

Clean, cool, and clear water is essential to salmonids. The health of aquatic habitats declines as temperature, turbidity, nutrients, and other parameters exceed natural ranges and if chemical and biological contaminants are found in significant quantities. Stream temperature is of particular concern in the Northwest due to its importance to fish and its response to land use activities. Brett (1952) found that juvenile Pacific salmonid species generally preferred temperatures in the range of 54-57°F (12°-14°C). Upper lethal limits have been found to be in the 75-81°F (24-27°C) range depending on species and acclimation temperatures (Brett 1952, Hynes 1970, Sullivan et al. 2000).

Stream temperature is readily altered by removing the riparian canopy cover and increasing the channel width. Both canopy cover and channel width are impacted by a variety of land uses. Temperature also has a negative correlation with dissolved oxygen although interactive effects of photosynthesis and groundwater inputs can alter this relationship (Hynes 1970). Current Washington State temperature standards are less than 64°F (18°C) for class A ("excellent") streams and 61°F (16°C) for class AA ("extraordinary") streams. In the lower Columbia region, most streams lying within national forest land are class AA, while most lower basin streams are designated class A. Streams that are monitored according to DOE protocols and regularly exceed the standards are included on the state's 303(d) list for impaired water bodies.

Turbidity is also a major concern in the Northwest, as it is readily increased by land use practices that produce and deliver fine sediment to stream channels. Turbidity has a strong impact on salmonid feeding success, egg incubation, respiration, and physiological stress.

Changes in nutrient dynamics can impact stream productivity. Forestry activities in riparian areas contribute organic debris and increase light availability, which increases primary production and can increase fish productivity. However, these benefits are often offset by detrimental impacts of logging to physical habitat. Increased nutrification also occurs due to agriculture where fertilizers and animal wastes increase the delivery of inorganic and organic compounds. Detrimental impacts from these inputs is seen most in slow-moving river and lake waters where algal blooms result in depleted dissolved oxygen, and anaerobic respiration can pollute waters.

Fecal coliform bacteria is also a concern in many lower Columbia basins and is usually related to livestock wastes and failing septic systems. Other pollutants occur to a lesser degree in lower Columbia basins and are related to mining wastes, urban runoff, and industry.

3.3.3.2 Current Conditions

The Washington State Department of Ecology 303(d) list of threatened and impaired water bodies represents the most comprehensive and uniform documentation of water quality impairments throughout the region. Water quality-impaired stream segments included on the 303(d) list include streams monitored by the WDOE or documented impairments submitted to WDOE by other entities. There are many impairments that are documented by various other organizations that do not appear on to the 303(d) list for a number of reasons. The 303(d) list therefore does not reflect all of the potential water quality concerns in lower Columbia streams.

The streams listed on the draft 2002/2004 303(d) list are displayed in Figure 3-93. Only selected parameters are shown. There are also stream segments listed for a variety of other water quality parameters, including DDT, arsenic, lead, sediment bioassay, and others, but they comprise only a small portion of the listed streams.

- The most common water quality concern in the region regards water temperature. Over 150 streams in the lower Columbia region have one or more segments on the 303(d) list for temperature problems (Figure 3-93). However, many streams with temperature problems are not included on the 303(d) list. Most temperature exceedances have been attributed to reduction in riparian tree canopy cover, increased stream widths, and decreased low flow volumes during the summer. Temperature problems are scattered throughout the forested and developed areas of the region. Dissolved oxygen levels are a related problem and are of most concern in WRIA 28, although most of the listed stream segments are within the Vancouver metropolitan area and are not in significant salmon and steelhead streams.
- Fish habitat modeling indicates that high summer stream temperatures are a major limiting factor for steelhead and coho in many basins (habitat modeling results are presented for each subbasin in Vol. II of this Technical Foundation).
- The presence of fecal coliform bacteria is also considered a problem in the region, with over 30 stream segments on the 303(d) list (Figure 3-93). Most of the listed segments are within the urban and rural residential areas in WRIA 28 and are likely the result of failing septic systems. Runoff from livestock grazing also has been identified as a contributor to the bacteria problem in many areas.
- There are few sediment-related problems in the lower Columbia region that are on the 303(d) list. Chronic suspended sediment problems (measured by turbidity) are generally not a concern except for portions of the Toutle and Lewis basins that drain Mount St. Helens. Excessive delivery of fine sediment to stream channels during runoff events, however, is a concern throughout the region. This issue is discussed in detail in the Substrate and Sediment section.

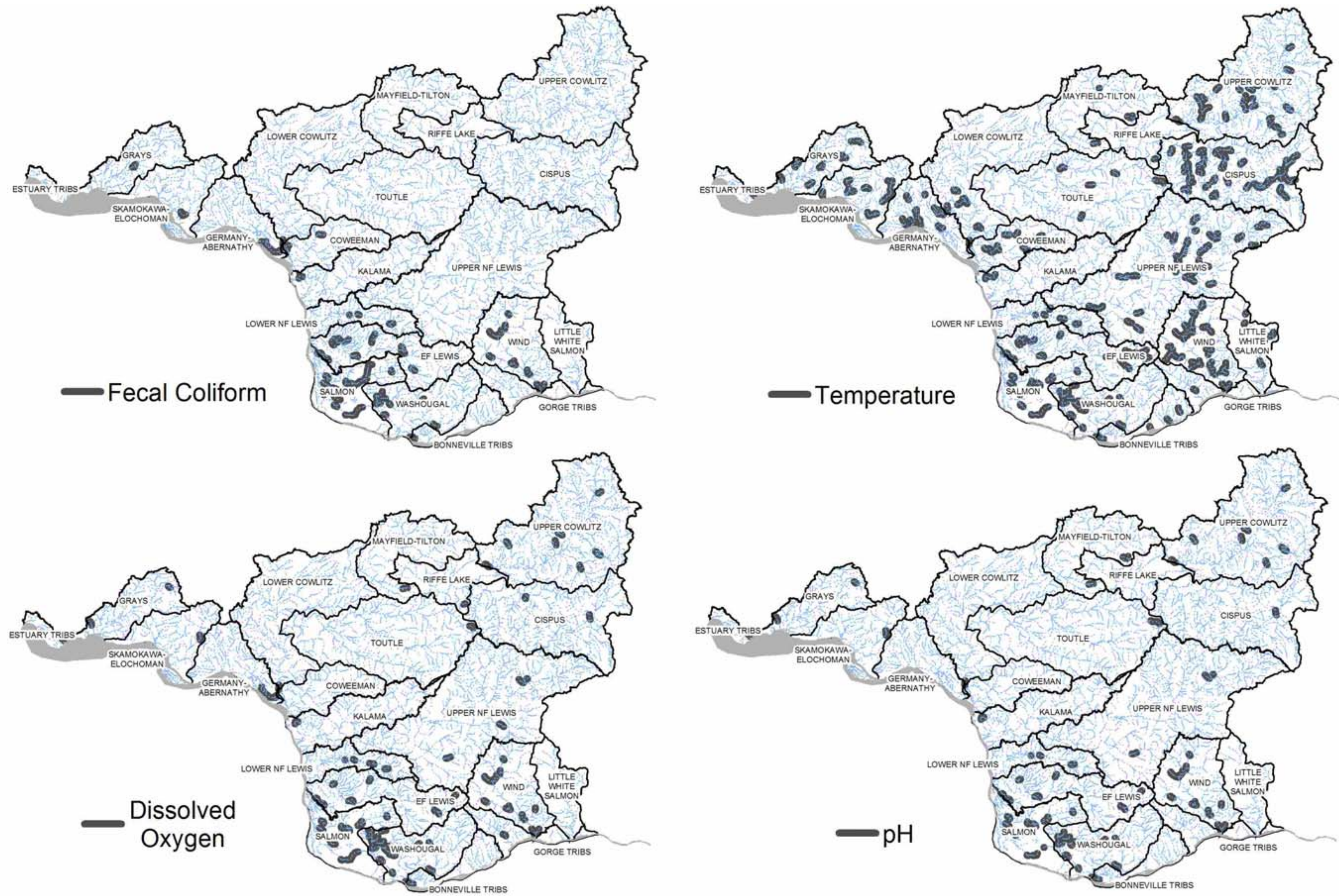

Figure 3-93. Map of stream segments on the 2002/2004 303(d) list for selected parameters. The selected parameters are the most widespread water quality impairments in the region.

3.3.4 Critical Habitats

3.3.4.1 Processes and Effects

The distribution, dimensions, and quality of stream channel habitat units greatly affects the health of fish populations (Bjornn and Reiser 1991). Although fish use a variety of habitat types to different degrees depending on lifestage, pools and backwater habitats are often regarded as the most crucial. For example, spawning often occurs at the downstream end of pools, where the right combinations of substrate and flow conditions are found. Pools also provide important cover and food resources for juvenile fish. Backwater and side channel habitat are especially important for some species, because they are often the site of upwelling, providing cool water in the summer as well as nutrient-rich water important for growth. They also provide refuge from flood flows. For these reasons, pool and side channel availability are commonly used as metrics to assess overall stream habitat condition. Functional connectivity between the various critical habitats for each life history stage is also critical (Mobrand et al. 1997).

The creation and maintenance of stream channel habitats is a function of the interaction between the underlying geology and the dynamics of flow, sediment, and large woody debris. Disrupting these physical processes may result in habitat unit types that are outside of natural ranges of quality and quantity. In the lower Columbia region, processes that drive channel conditions have been altered to various degrees by land management activities. The greatest impacts on stream habitat units have been practices that have directly altered stream channels such as splash dam logging, diking, channelization, stream clean-outs, gravel mining, and dam building. Upland and riparian land use practices that alter flow, sediment, and wood recruitment are less direct, but equally important, impacts.

3.3.4.2 Current Conditions

In many lower Columbia streams, habitat surveys provide information on pool and side channel availability. In other areas, local experts have provided information as part of the limiting factors analysis process, as described in each subbasin chapter in Volume II. Still, there is little information regarding specific stream channel conditions in many areas. In general, the evidence shows an overall decrease in side channel and pool habitats.

- The greatest loss of stream habitat has resulted from the Cowlitz and Lewis River hydropower systems, where many miles of stream channel lie beneath a series of reservoirs, and additional miles are blocked from access.
- The other major loss of habitat is in the lower reaches of stream systems that have been diked and channelized for agricultural, industrial, and residential uses. Coastal basins have been especially affected; historically, these systems had extensive networks of estuarine side channels that are now isolated or filled. Chum spawning habitat and coho winter rearing habitat have been particularly impacted by loss of off-channel and side channel areas.
- Upper basin stream systems have suffered less pool and side channel degradation, though the impacts to some fish populations may be greater because of the concentration of quality spawning and rearing habitat. As in the lower basins, side channels have been lost due primarily to erosion control diking and riprap. Some channels are impacted by stream channel incision that has persisted since past splash-damming and riparian timber harvest.
- The loss of pool habitat as a result of decreased large wood quantities and degraded riparian areas is also a concern. In most upper forested basins in the region, the quantity of pool habitat is in the low end of the range considered adequate for salmonids.

- The presence of good side channel and pool habitats has been identified in some areas. These are most often associated with woody debris. An assessment in the upper Cowlitz basin indicated that streams containing LWD had 15 times the number of pools as streams without large wood (EA 1998 as cited in Wade 2000).

3.3.5 Substrate and Sediment

3.3.5.1 Processes and Effects

Proper substrate and sediment conditions are necessary for spawning, egg incubation, and early rearing of salmonids. Substrate and sediment are delivered to spawning and rearing areas during natural disturbance events, mediated by LWD and existing habitat complexity (Bisson et al. 1997). However, excessive fine sediment delivered to channels can suffocate salmonid eggs, inhibit emergence of fry from gravels, decrease feeding success, increase physiological stress, and through adsorption, may facilitate the transport and persistence of chemical contaminants (Welch et al. 1998). The size of substrate preferred by spawning salmon ranges from less than 0.4 in (1 cm) to over 4.7 in (12 cm) in diameter, depending on the species and size of the fish (Bjornn and Reiser 1991, Schuett-Hames et al. 2000). During redd construction, spawning substrates are cleared of fine sediments; however, during the incubation period, redds are susceptible to accumulation of fines.[3] Sediment accumulation can impede intergravel flow necessary to supply embryos with oxygen and carry away wastes. Embryo survival declines as percentage fines increases (Bjornn and Reiser 1991). Fine sediment may also limit the ability of alevins to move around and to ultimately emerge from the gravels. Studies have shown that alevins have trouble emerging when percent fines exceed 30-40% (Bjornn and Reiser 1991). Substrate conditions also are important for juvenile salmonid rearing. Substrates provide cover, protection from high flows, and macroinvertebrate production. Juvenile production and densities have been shown to decrease with increased gravel embeddedness (Crouse et al. 1981, Bjornn et al. 1977 [from Bjornn and Reiser 1991]). Embedded substrates may also reduce the availability of macroinvertebrate food resources (Bjornn et al. 1977, Hawkins et al. 1983).

Many factors can affect substrate conditions, including streambed scour, reduced gravel recruitment, substrate extraction, and increased fine sediment delivery.

- Scouring of substrates may result from increased flood flows, alterations to channel geometry, loss of channel stability, splash dam logging, and debris flows.
- Gravel recruitment is reduced by dams, bank armoring, and channel alterations.
- Direct extraction of substrates has occurred in some areas due to gravel mining operations.

Increased sediment transport and delivery due to upslope land use has a major impact on in-stream habitats. Sediment is contributed to stream channels through surface erosion, gully erosion, and mass wasting (Ward and Elliot 1995). The amount of erosion resulting from these processes is related to climate, soil, slope, and vegetation conditions.

- Surface erosion primarily occurs as sheet and rill erosion on agricultural, urban, and range lands, but it also may occur on forest road surfaces or areas disturbed during timber harvest. Surface erosion can be extremely high in developing urban areas that are under construction, where erosion may increase from 2 to 40,000 times the preconstruction rate (McCuen 1998).

[3] Fines are typically defined as sediment sizes less than 0.85 mm (0.033 inches) diameter, and percentage fines greater than about 17% are considered not properly functioning according to NMFS (NMFS 1996).

- Gully erosion results from concentrated flow and commonly generates sediment volumes an order of magnitude greater than sheet and rill erosion. Gullies are often associated with forest road ditches, where ditch and culvert design and/or maintenance are inadequate to effectively convey runoff volumes.
- Mass wasting, in the form of landslides and debris flows, can deliver huge amounts of sediment to stream channels. Landslides may be rapid or slow (slumps) and can occur on shallow or steep slopes. Water saturation, vegetation removal, and human-induced flow concentration (i.e. roads) are often responsible for landslides in forested areas. Debris flows are caused by similar disturbances, though generally involve higher water content, initiate on steeper slopes, and travel farther than landslides. Debris flows are common in steep headwater or tributary channels and can contribute large amounts of sediment and woody debris to salmonid streams.

3.3.5.2 Current Conditions

Substrate conditions across the lower Columbia region vary with respect to channel types, position within the watershed, and natural and anthropogenic disturbances.

- Fish habitat modeling indicates that fine sediment is one of the primary factors limiting fish production for most salmonid populations in the lower Columbia region.
- Many stream reaches suffer from a lack of adequate spawning gravels and high concentrations of fines. Spawning gravels are often embedded with fines—a particular problem in coastal basins that have sedimentary geology and a high occurrence of mass wasting. Historical chum and chinook spawning sites on lower river segments are especially susceptible to accumulations of fines. Accumulations of fines near the mouths of streams entering the Columbia River upstream of Bonneville Dam have increased since dam construction.
- High rates of sediment delivery have been a continual problem in the Toutle River watershed and other streams impacted by the Mt. St. Helens eruption, although conditions have been improving. Conditions have improved more quickly in the SF Toutle and Green River than in the NF Toutle, which received the greatest impact.
- The Sediment Retention Structure (SRS) on the mainstem NF Toutle contributes to sediment impairment in the Toutle River. The SRS was constructed after the 1980 Mt. St. Helens eruption in an effort to reduce downstream sediment aggradation and thus improve conveyance of flood waters in the lower Toutle and Cowlitz rivers. The structure has since been overtopped with sediment and has become a chronic source of fine sediment to downstream areas. The SRS is believed to be preventing the recovery of the system (Wade 2000).
- Past and current land use has created upslope land cover conditions that are susceptible to increased sediment production and delivery to streams. The IWA identified sediment supply problems across the study area according to landscape characteristics including topographical slope, soil erodability, and unsurfaced road densities. IWA sediment impairment results are depicted for the entire region in Figure 3-94 (see Vol. II for a presentation of subbasin-level results).

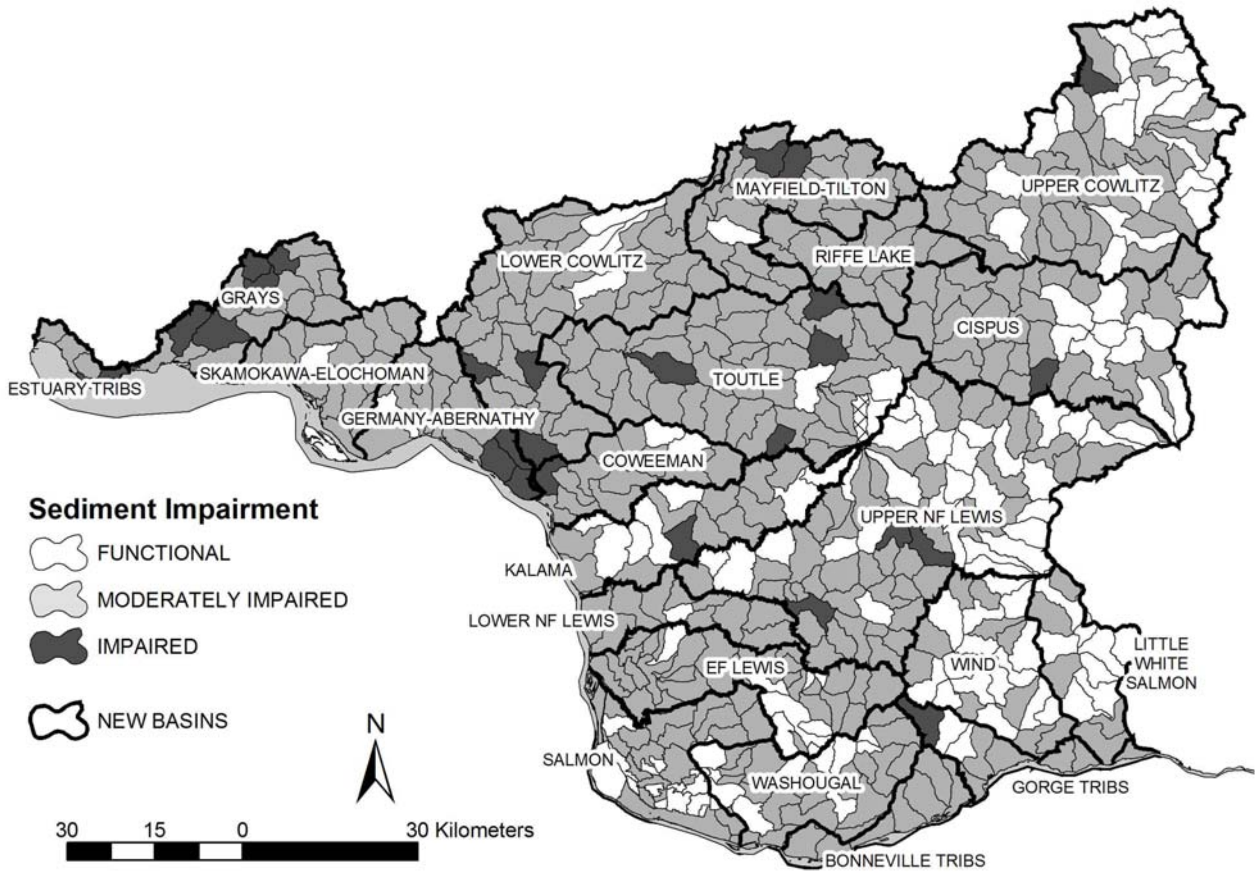

Figure 3-94. Map of sediment supply problems across the lower Columbia region. Impairment categories were calculated as part of the Integrated Watershed Assessment. (see Vol. II for presentation of subbasin-level results). These impairment ratings represent local sediment supply conditions, not including upstream effects.

3.3.6 Woody Debris

3.3.6.1 Processes and Effects

Woody debris is an important component of stream ecosystems. Removal of riparian vegetation can decrease wood recruitment as well as reduce bank stability (Beechie et al. 2000). Reduced bank stability increases sedimentation of pools and increases width to depth ratios, thus reducing the quality and quantity of pool habitat. Juvenile and adult salmonids rely directly on LWD for shade, protection from disturbance, and protection from predation (Bisson et al. 1988, Solazzi et al. 2000). Studies have shown that fish production is positively correlated with presence of large organic debris (Bjornn and Reiser 1991). Woody debris also retains organic matter, provides sites for macroinvertebrate colonization, and can trap salmon carcasses (Murphy and Meehan 1991, Cederholm et al. 1989). An indirect benefit of LWD to salmonids is its influence on stream channel morphology and habitat complexity. LWD tends to be stationary in small streams, where it affects local bank stability and creates patches of scour and deposition. In large streams, LWD moves more readily and often forms jams. Accumulations of LWD affect bank stability, scour, bar formation, and may also induce rapid channel adjustments (Keller and Swanson 1979). In some streams, LWD may also be important for the establishment of floodplain and riparian habitats (Abbe and Montgomery 1996).

Another significant attribute of LWD is the role it plays in pool formation. Stable woody debris traps sediments and can form steps in otherwise uniform channels. In some cases, LWD can create depositional areas in channels that would otherwise be composed of bedrock (Montgomery et al. 1996). Abundance of LWD has been positively correlated with pool area, pool volume, and pool frequency (Carlson et al. 1990, Beechie et al. 2000).

LWD is recruited to stream channels through bank erosion, mass wasting, blowdown, and debris torrents. Removal of riparian timber decreases the potential for future LWD recruitment. Although timber harvest may increase short-term wood loading in some instances, long-term recruitment and persistence of wood in streams is highest in older forest types (Bilby and Ward 1991, Beechie et al. 2000). LWD is removed from stream channels through fluvial transport or by direct removal. Direct removal of LWD was a common practice in the 1970s and 1980s when log jams were believed to impede fish passage. Wood removal has occurred in other locations in order to reduce flood potential (Shields and Nunnally 1984). As expected, the removal of LWD has been shown to alter channel morphology and decrease habitat complexity (Smith et al. 1993).

3.3.6.2 Current Conditions

The various agencies conducting stream surveys in the lower Columbia region define LWD differently. In general, minimum diameter to be considered for LWD ranges from about 4-14 inches (10-36 cm), while minimum lengths range from 6.5-49 ft (2.13-15 m). The definition of what constitutes poor conditions also varies, but is generally fewer than 80 pieces/mi or fewer than 0.2 pieces per channel width (NMFS 1996, Schuett-Hames et al. 2000, Wade 2000).

- LWD conditions are considered poor across much of the lower Columbia region. Only a handful of surveyed streams have good conditions.
- The amount of LWD affects the EDT habitat attribute ‘habitat diversity’. For many lower Columbia stream systems, EDT modeling indicates that habitat diversity is the habitat factor that is serving to depress population performance to the greatest extent.
- In many areas where LWD is adequate, it is concentrated in large jams, although many of the large jams that existed historically on low-gradient large systems such as the Cowlitz, are no longer present (Mobrand Biometrics 1999).
- Low LWD abundance in many upper basins is attributed to past timber harvest and scour from splash dam logging. In other areas, poor conditions are attributed to past fires that have reduced recruitment. USFS and other crews removed instream wood in some streams during the 1980s because it was believed to impede fish passage while in other streams, local residents have removed LWD due to flooding and erosion concerns.
- In general, it is believed that LWD recruitment potential is increasing in most basins due to re-growth of riparian forests. Current riparian buffer regulations prevent significant harvest along most streams, which will eventually serve to restore instream LWD levels (WFPB 2000). Restoration projects that involve the re-introduction of wood into stream systems have and will continue to increase instream LWD.

3.3.7 Channel Stability

3.3.7.1 Processes and Effects

Excessive sediment delivered from unstable stream banks can suffocate salmonid eggs, inhibit emergence of fry from gravels, decrease feeding success, and increase physiological stress. Unstable banks also increase mass wasting and have subsequent effects on channel morphology. Bank stability processes vary depending on location in a catchment. In steep headwater systems, channels are typified by stable substrates (i.e. bedrock, boulders) and thus have greater resistance to erosion. With the exception of debris flows, sediment entering these channels is predominantly from upslope sources. Channels lower in the catchment, on the other hand, tend to have higher rates of bank erosion, with, in many instances, channel sources contributing far more sediment than upslope sources. It is in these channels that the impact of unstable streambanks is greatest on salmonids.

Patterns of erosion and deposition within stream channels have a strong influence on channel form, including meander formation and floodplain development. The distribution and dimensions of aquatic habitats, such as pools and riffles, are therefore governed in part by bank stability. A study on Salmon Creek, a lower Columbia tributary, found that landslides increased the amount of sediment stored in channel bars at the expense of pools (Perkins 1989 as cited in Montgomery and Buffington 1998). Factors that control bank stability include bank material composition, flow properties, channel geometry, and vegetation (Knighton 1998). While vegetation may not have the greatest controlling influence on stability, it is readily altered by land use, and therefore of particular concern. Root systems increase resistance to the erosive forces of flowing water and denser vegetation generally results in narrower and deeper channels. The woody roots of trees are particularly useful in providing long-term channel stability (Beschta 1991).

Land use activities that modify vegetation conditions and channel geometry can reduce bank stability. Timber harvesting and conversion of riparian forests to agriculture, residential, and other developed uses reduce vegetative cover on stream banks. These practices have been widespread in the lower Columbia region over the past century. Livestock grazing increases bank erosion through direct trampling and removal of vegetation (Trimble and Mendel 1995). Stream channelization may also increase channel erosion by increasing water depth, which increases shear stress (product of depth and slope) and therefore scour potential on the channel bed. Channel straightening increases stream gradient, which also increases scour potential and transport capacity (Knighton 1998). Increased runoff volumes due to upland land uses can increase stream power which can increase erosive forces. Increased streamflows due to urbanization can alter channels dramatically through widening and incision (Booth 1990). Alternatively, streambank reinforcement for erosion control, such as riprap, reduces habitat complexity and can result in diminished salmonid abundance (Knudsen and Dilley 1987).

3.3.7.2 Current Conditions

Bank stability problems have been identified in most basins throughout the lower Columbia region. Loss of bank stability is attributed to a number of factors. These include most land use activities mentioned above, namely timber harvest, land use conversion, straightening and channelization, livestock grazing, and flow alterations. In some cases, the natural geology exacerbates instability. This is the case in areas underlain by sedimentary rock in coastal basins, mudflow deposits around Mt. St. Helens (Toutle and Lewis basins), and Bretz Flood deposits in lower portions of Columbia Gorge basins. Bank stability has been reduced in many lower catchment channels by riparian and floodplain development that has resulted in straightened and channelized streams. In some areas, natural channel movement is perceived as a bank stability problem when developed or agricultural property within the channel migration zone is threatened. There are bank stability concerns across the region.

- The stream channel has rapidly adjusted due to avulsions into gravel mining pits on Salmon Creek and the lower EF Lewis River. The impact of these avulsions on aquatic habitat may be minor in some cases.
- Livestock grazing has impacted streambanks. Efforts to exclude cattle with fences have reduced this impact.
- Timber harvests and road building have increased runoff and sediment supply to channels. Sediment inputs can increase in-channel sediment aggradation, resulting in high width-to-depth ratios and an elevated rate of channel movement. New forest practices rules that regulate road building, timber harvests on steep slopes, and riparian timber harvest should alleviate channel instability problems.

Despite these problem areas, the limiting factors analyses noted generally good bank stability conditions in the Jim Crow, Skamokawa, Elochoman, lower Cowlitz, Kalama, and Washougal basins. Other areas of good bank stability are a result of erosion control projects which may present their own impacts on fish, as noted above.

3.3.8 Riparian Function

3.3.8.1 Processes and Effects

Riparian areas are an important interface between upland and aquatic systems (Gregory et al. 1991). Riparian vegetation directly and indirectly affects fish habitat suitability through influences on water temperature, habitat diversity, sedimentation, wood recruitment, and bank stability (Beschta 1991). Reaches with less canopy cover tend to exhibit higher maximum temperatures and larger diurnal temperature fluctuations than reaches with more canopy cover (Beschta et al. 1987, Sullivan et al. 1990). Shading from riparian canopy cover tends to be most important in summer due to high sun angles, reduced cloud cover, and longer days. In winter, canopy cover can inhibit the re-radiation of heat away from the stream, reducing the occurrence of extreme low temperatures (Beschta et al. 1987). Riparian cover also may be important for reducing wind velocities that contribute to convective heat loss (Sinokrot and Stefan 1993) and may have an important influence on the stream microclimate (Adams and Sullivan 1989, Rutherford et al. 1997), though these effects are not well understood. Canopy cover has a greater affect on small streams than large streams since wider streams are less likely to be shaded.

Riparian canopy cover provides other benefits in addition to moderating stream temperatures. Riparian canopies are an important source of allochthonous inputs (e.g. litterfall) of carbon and nitrogen to the stream system (Gregory et al. 1991, Beschta 1997a). Attenuation of light by tree canopies also may be an important factor affecting macroinvertebrate distribution and abundance. Meehan (1996) found a significant difference in macroinvertebrate abundance in shaded versus non-shaded reaches. Shade has also been shown to affect drift of benthic invertebrates. Algal growth and benthic productivity are affected by shade (Hynes 1970).

In addition to the benefits realized by adequate canopy cover, intact riparian forests also provide a source of LWD recruitment to stream channels. In small streams, fallen trees often remain where they fall and have a dramatic influence on habitat complexity. Wood has greater mobility in larger streams, where it more readily accumulates in jams. In-stream wood, as well as floodplain forests, provides roughness elements that increase flow resistance and reduces downstream flood effects. Trees also provide bank stability through erosion resistance created by roots. (See the Woody Debris section above for additional information on the importance of LWD to salmonids.)

3.3.8.2 Current Conditions

Riparian conditions are generally considered poor across the lower Columbia region. The IWA riparian assessment (Figure 3-95), which modeled riparian impairment across the region using vegetative cover characteristics, indicates that most of the region suffers from moderately impaired riparian conditions. The most intact riparian areas are located in the upper elevations of the upper Cowlitz and upper Lewis basins, while the greatest impairments are located in the lowest elevations, especially around the urbanized Vancouver, WA metropolitan area.

- Many lower elevation riparian zones that historically had forest cover have been converted to land uses such as agriculture, residential development, or transportation corridors.
- Cattle access to streambanks is an ongoing problem in many areas.
- Middle and upper basin riparian areas suffer from young forest stands and/or a predominance of deciduous vegetation due to past timber harvests. These conditions are expected to improve on forest lands with the relatively recent regulations (WAC 2000) that govern forest practices in riparian areas.

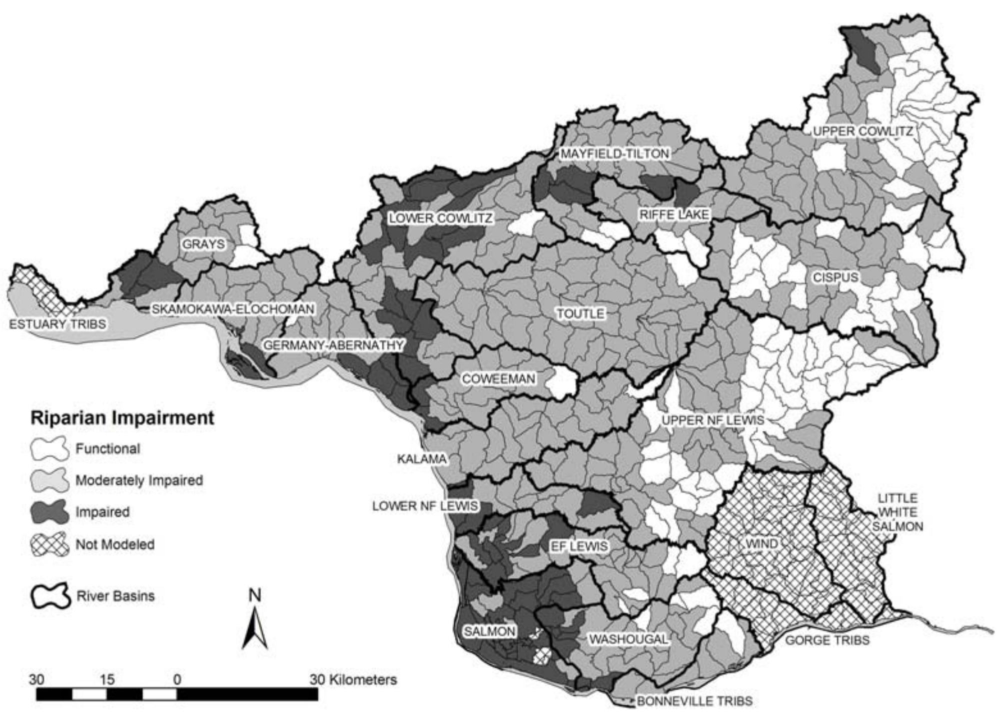

Figure 3-95. Map of riparian impairments across the lower Columbia region. Impairment categories were calculated as part of the Integrated Watershed Assessment. (see Vol. II for presentation of subbasin-level results).

3.3.9 Floodplain Function

3.3.9.1 Processes and Effects

The interaction of rivers with their floodplains is important for flood flow dampening, nutrient exchange, and maintenance of stream and off-channel habitats. For example, several researchers have demonstrated the importance of off-channel floodplain habitats for juvenile coho salmon rearing (Cederholm et al. 1988, Nickelson et al. 1992). As a stream accesses its floodplain, the increase in cross-sectional area decreases the flow velocity, reducing downstream flow volumes and limiting erosivity. If a stream is isolated from its floodplain, either through channel incision, diking, or floodplain filling, then the potential for downstream flooding and channel instability may be increased (Wyzga 1993, as cited in Knighton 1998). Floodplains also are important for nutrient exchanges between the stream and terrestrial vegetation. The stream hyporheic zones are especially important for maintenance of water quality, nutrient processing, and biological diversity (Edwards 1998). Hyporheic zones underlie most floodplain forests and are easily disrupted by activities that isolate floodplains or disrupt subsurface flow patterns.

Floodplains are isolated from rivers by human activities in a number of ways. Diking and channelization serve to fix the stream in a specific location, preventing overbank flows and meander migrations. This practice often occurs in combination with filling of floodplain sloughs, oxbow lakes, and side channels in order to facilitate development or create crop or pasture land. Floodplains can also be isolated from rivers through channel dredging intended to increase flow conveyance. As a result, flow magnitudes that historically would have inundated the floodplain are confined within the channel. Diking, dredging, and floodplain filling projects are often combined with channel straightening, which can increase stream gradients and in turn increase channel erosion potential. Road crossings of streams can limit floodplain function by forcing the stream into a particular location (e.g. at a bridge), preventing natural flooding and meander patterns.

3.3.9.2 Current Conditions

Floodplain function in the lower Columbia region has been altered by diking, channelization, channel incision, filling of side channels, and mining.

- Diking has occurred extensively within tidally influenced areas near the mouths of many streams. The effects on aquatic biota have been especially severe on coast range basins such as the chinook and Grays rivers where a large percentage of off-channel estuary habitat has been isolated from the river. Dikes were constructed and floodplain channels were filled to create cropland. Recent strides have been taken to restore estuary habitat by breaching dikes and removing tide-gates.
- The lower reaches of many stream systems have been diked extensively for residential, commercial, and agricultural purposes. The most affected stream segments are the lower Cowlitz and lower North Fork Lewis rivers, where channelization projects have isolated large amounts of historically available habitats.
- Transportation corridors are a ubiquitous cause of floodplain constriction on many streams, as roads tend to follow stream valley bottoms. Many streams have been artificially straightened to accommodate roadways.

3.4 Mainstem Conditions

The mainstem Columbia River serves lower Columbia region salmon primarily as a corridor for juvenile and adult anadromous salmon migrating between the tributary rivers and streams where they spawn and rear, and the ocean where they grow and mature. Margins and backwater areas are also important for juvenile salmonid rearing. The lower Columbia River mainstem has also recently become an important spawning area for chum salmon and fall chinook; in particular, spawning has been observed for about the last 5 years in areas near Pierce and Ives islands, near the mouths of multiple creeks in the lower Columbia Gorge area, and along the Washington shore near the I-205 bridge. The number of chum salmon spawners in the mainstem Columbia has been substantial; mainstem spawners represent from ¼ to ½ of all spawners in the Lower Columbia River Chum Salmon ESU. Preservation of spawning habitat and incubation flows in the mainstem will be vital to continued spawning success.

Environmental conditions in the mainstem Columbia River have been dramatically altered by extensive dam construction for hydropower generation, flood control, navigation, and irrigation. Virtually all major tributaries contain at least one large dam and reservoir. The mainstem Snake and Columbia rivers are now a staircase of dams from Canada, Montana, and Idaho all the way to Bonneville Dam at RM 146, as illustrated by Figure 3-96. Hydropower regulation throughout the system influences water levels in Bonneville Pool and flow releases from Bonneville Dam, which affects lower Columbia ESU salmon populations. The hydropower infrastructure and flow regulation has implications for adult migration, juvenile migration, mainstem spawning success, estuarine rearing, water temperature, water clarity, gas supersaturation, predation, and stranding caused by large ship wakes.

Figure 3-96. Some major dams of the Columbia River basin.

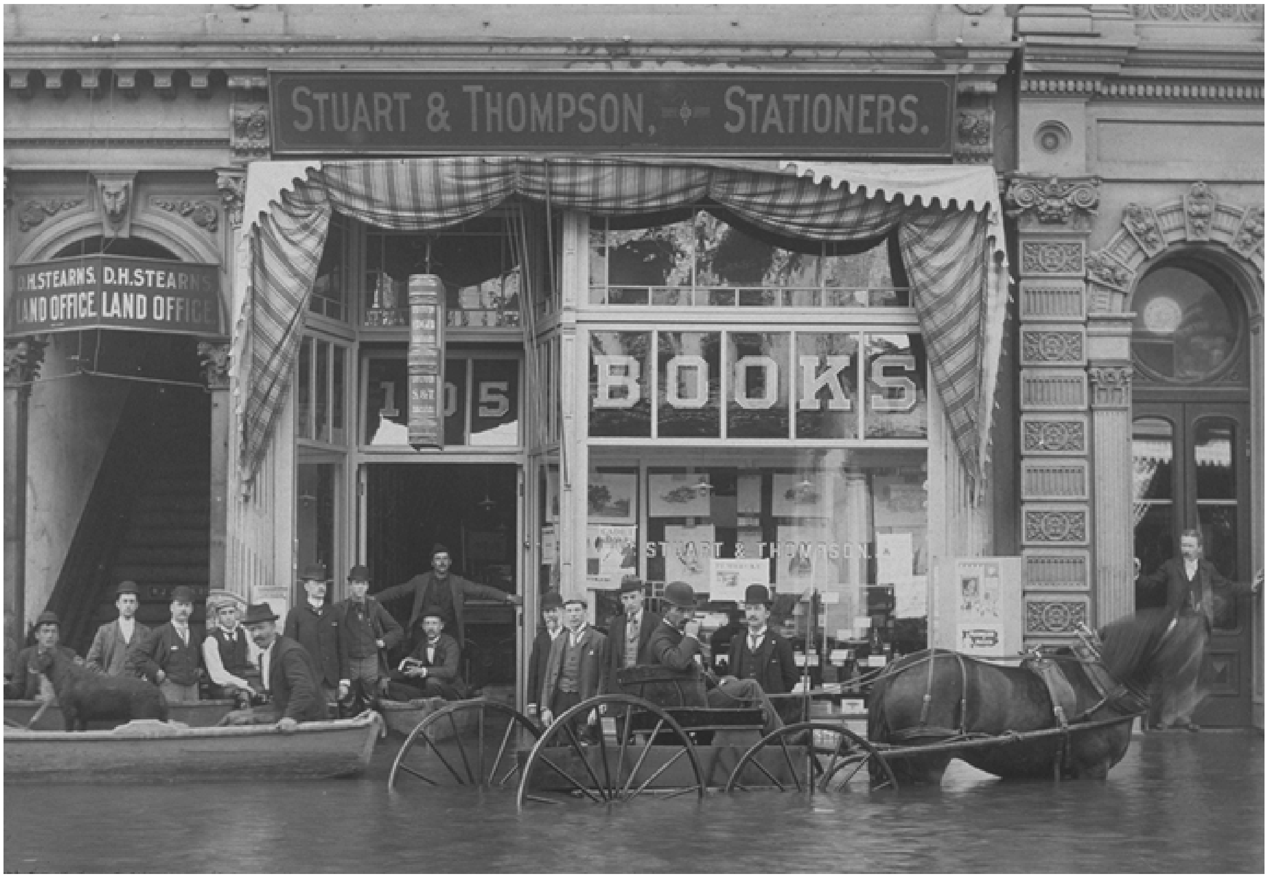

Figure 3-97. Photo of flooded Portland street before completion of Columbia basin flood control system (Oregon Historical Society Photo).

3.4.1 Flow

Before the development of the hydrosystem, Columbia River flows were characterized by high spring runoff from snowmelt and regular winter and spring floods (Figure 3-97). Dam construction and operation have altered Columbia River flow patterns substantially throughout its basin. Spring flows are greatly reduced from historical levels as water is stored for power generation and irrigation, while summer and winter flows have increased (Figure 3-98). Flood control operations have reduced flood volume and frequency. Hydrosystem operations change to accommodate daily fluctuations in power demand and can result in significant daily flow variation downstream from some hydropower facilities.

Changes in flow patterns can affect salmon migration and survival through both direct and indirect effects. Juvenile and adult migration behavior and travel rates are closely related to river flow. Greater flows increase velocity, which increases juvenile and decreases adult travel rates. Extensive study has detailed the relationship between juvenile migration travel times and flow volume. The relationship is particularly strong at low to moderate flow volumes. Flow regulation and reservoir construction has increased smolt travel times through the Columbia and Snake mainstems many-fold, although the significance of this relationship to juvenile survival remains a subject of considerable controversy. The potential delay of emigrants reaching the estuary during a critical physiological window for smoltification or for ocean dispersion is a significant concern, especially for upriver salmon stocks, where delays are compounded across long migration distances. Moreover, increased travel times also increase exposure to Columbia

River predation. For lower basin stocks, however, the mainstem journey is relatively short and only fish originating in the Wind, Big White Salmon, Little White Salmon, and Columbia Gorge tributaries are directly affected by passage through one mainstem dam (Bonneville).

Interactions of flow and dam passage can be particularly problematic for migrating salmon. General passage issues have been discussed in the subbasin habitat section above, but higher flows generally increase the survival of juveniles as they pass through the dams, because more fish can pass over the spillways, where mortality is low, than through the powerhouses, where turbine passage mortality can be significant. The increased spill typically associated with high flows also reduces travel time by avoiding fish delays in dam forebays. For this reason, many fish and hydrosystem managers implement a water budget of prescribed flows to facilitate fish migration rates and dam passage. In contrast, increased flow and spill can increase mortality and delay upstream passage of adults at dams as fish have a more difficult time locating the entrances to fishways and also are more likely to fall back after exiting the fish ladder (Reischel and Bjornn 2003).

Flow also affects habitat availability for mainstem spawning and rearing stocks. Significant numbers of chum and fall chinook spawn and rear in the mainstem and side channels of the Columbia downstream from Bonneville Dam. Flow patterns determine the amount of habitat available and can also dewater redds or strand juveniles (NMFS 2000c).

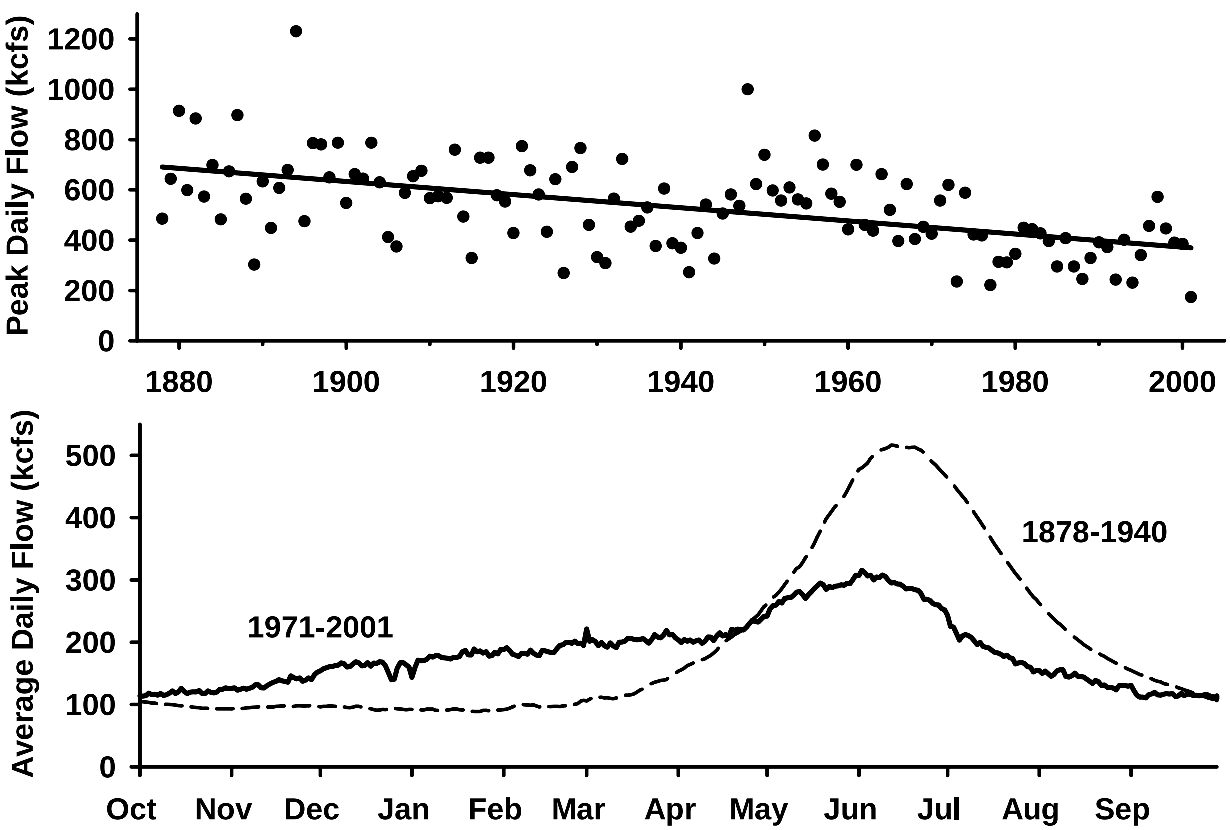

Figure 3-98. Historical changes in average daily flow patterns and flood frequency in the Columbia River at The Dalles.

3.4.2 Water Temperature and Clarity

Flow regulation and reservoir construction have increased average water temperature in the Columbia River mainstem as illustrated in Figure 3-99. Summer water temperatures now regularly exceed optimums for salmon (NMFS 2000a). Water temperatures in fish ladders can be higher than ambient river temperatures, which compounds this problem.

High water temperatures can cause migrating adult salmon to stop their migrations or seek cooler water that may not be in the direct migration route to their spawning grounds (NMFS 2000a). In the lower Columbia, many summer and fall migrating adults typically pull into the cooler Cowlitz, Lewis, and Wind River mouths before continuing up the Columbia. Warm temperatures can increase the fishes' susceptibility to disease, but the overall effects of delay in migration rate due to high water temperature are unknown. Since the early 1990s, some upper basin dams have been operated to provide cold water for downstream temperature control to benefit migrating juvenile and adult salmon.

Flow regulation and reservoir construction also have increased water clarity. Increased water clarity can affect salmon through food availability and susceptibility to predation.

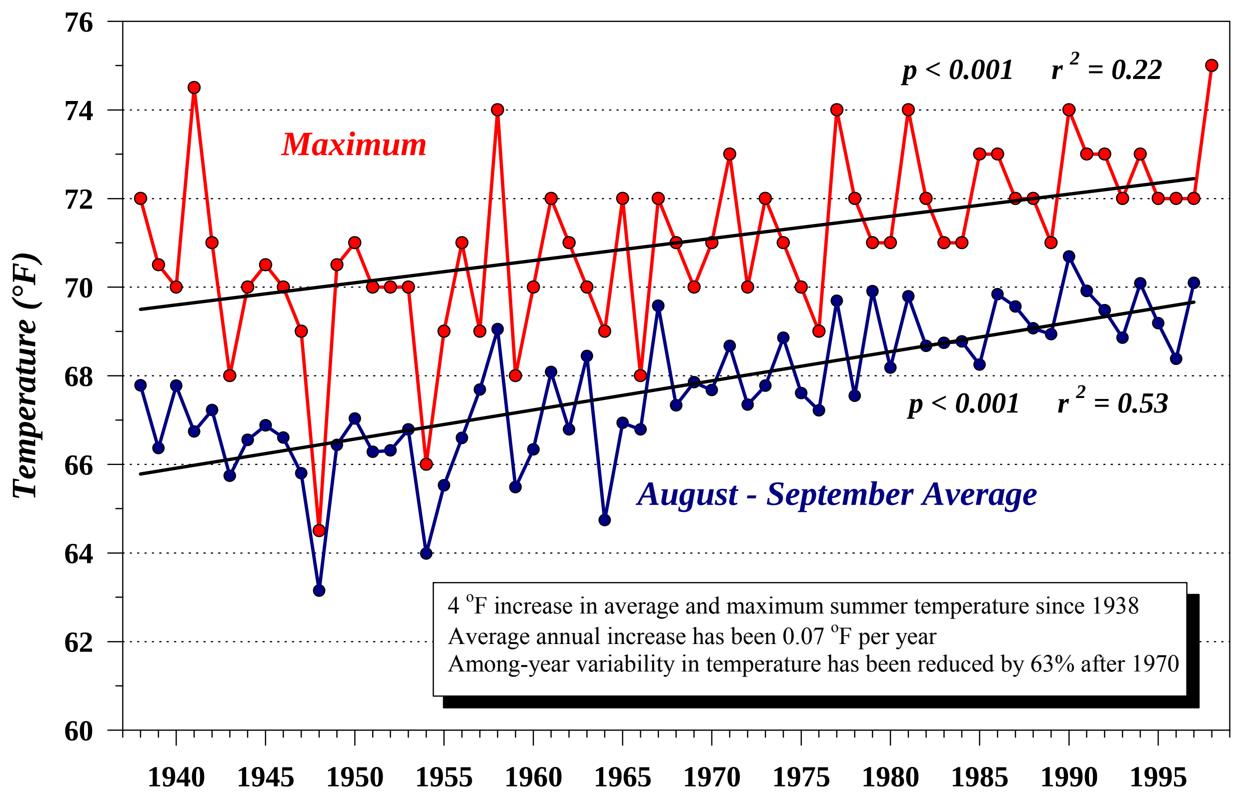

Figure 3-99. Historical changes in summer water temperatures at Bonneville Dam.

3.4.3 Gas Supersaturation

There are important trade-offs between fish passage and gas saturation to be considered when formulating spillway operation policies at lower Columbia River dams. Supersaturating water with atmospheric gases, primarily nitrogen, can occur when water is spilled over high dams. These high concentrations of gases are absorbed into the fishes' bloodstream during respiration. When the gas comes out of solution, bubbles may form and subject the fish to gas bubble disease as in the bends suffered by human divers. The severity of gas bubble disease varies depending on species, life stage, body size, duration of exposure, water temperature, swimming depth, and total dissolved gas (Ebel et al. 1975, Fidler and Miller 1993).

High dissolved gas levels associated with dam operations have resulted in significant salmon mortality—especially before the problem was identified and measures taken to reduce its incidence (Ebel 1969). Measures implemented over the last 40 years include increasing headwater storage during spring, installing additional turbines, and installing flip-lip flow deflectors to reduce plunging and air entrainment of spilled water (Smith 1974). Monitoring shows that salmonid mortality continues to be associated with exceptionally high river flows (NMFS 2000). For instance, Bonneville Dam turbines exceeded 130% capacity for 24 days in 1997. During that time, daily prevalence of gas bubble disease was high in sockeye (14-100% for 3 weeks) but lower for chinook (0-6.5% prevalence).

Gas supersaturation poses the greatest risk for Washington lower Columbia basin salmon stocks that must pass Bonneville Dam or are destined for areas downstream. Gas levels equilibrate slowly; thus, gas levels at Bonneville Dam that are high enough to have impacts on fish may extend for long distances downstream. Dissolved gas saturations levels below lethal levels may still have chronic effects, such as increased susceptibility to disease or predation; these effects are poorly understood. The issue of gas supersaturation has been discussed in detail in the Total Maximum Daily Load report developed jointly by the Oregon Department of Environmental Quality and the Washington Department of Ecology for dissolved gas levels in the lower Columbia River (Pickett and Harding 2002).

3.4.4 Predation

Significant numbers of salmon are lost to fish, bird, and marine mammal predators during migration through the mainstem Columbia River. Predation likely has always been a significant source of mortality but has been exacerbated by habitat changes. Piscivorous birds congregate near dams and in the estuary around man-made islands and consume large numbers of emigrating juvenile salmon and steelhead (Roby et al. 1998). Caspian terns, cormorants, and gull species are the major avian predators (NMFS 2000a). While some predation occurs at dam tailraces and juvenile bypass outfalls, by far the greatest numbers of juveniles are consumed as they migrate through the Columbia River estuary, as discussed in section 3.5.3.1. Native fishes, particularly northern pikeminnow, prey on juvenile salmonids. Marine mammals prey on adult salmon, but the significance is unclear.

Fishes—including northern pikeminnow, walleye, smallmouth bass, and salmonids—prey on juvenile salmonids. Pikeminnow have been estimated to consume millions of juveniles per year in the lower Columbia, as outlined in Table 3-42.

Table 3-42. Projected abundance of northern pikeminnow, salmonid consumption rates, and estimated losses of juvenile salmonids to predation*

Location	Length (km)	Number of pikeminnow	Consumption Rate (smolts/predator day)	Estimated Losses (millions/year)
Estuary to Bonneville Dam	224	734,000	0.09	9.7
Bonneville Reservoir	74	208,000	0.03	1.0

* From NMFS (2000b).

Pikeminnow numbers likely have increased as favorable slack-water habitats have been created by impoundment and flow regulation. In unaltered systems, pikeminnow predation is limited by smolt migratory behavior; the smolts are suspended in the water column away from the bottom and shoreline habitats preferred by pikeminnow. However, dam passage has disrupted juvenile migratory behavior and provided low velocity refuges below dams where pikeminnow gather and feed on smolts (Friesen and Ward 1999). The diet of the large numbers of pikeminnow observed in the forebay and tailrace of Bonneville Dam is composed almost entirely of smolts. Pikeminnow also concentrate at dam bypass outfalls and hatchery release sites to prey on injured or disoriented fish, and pikeminnow eat many healthy smolts as well. Predation rates on salmonids are often much lower in areas away from the dams, although large numbers of predators in those areas can still impose significant mortality.

In 1990, responding to observed predation problems, a pikeminnow management program was instituted that pays rewards to anglers for each pikeminnow caught and retained over a prescribed size. Through 2001, over 1.7 million pikeminnow had been harvested, primarily in a sport reward fishery. Modeling results project that potential predation on juvenile salmonids by northern pikeminnow has decreased 25% since fishery implementation (Friesen and Ward 1999, NMFS 2000a). By paying only for pikeminnow over a certain size, the program takes advantage of their population characteristics—they are relatively long-lived and only the large individuals are fish predators. Relatively low exploitation rates of only 10-20% per year compound over time to substantially reduce pikeminnow survival to large predaceous sizes.

Walleye are voracious predators of fishes, including juvenile salmonids. On a fish-per-fish basis, walleye are as damaging as pikeminnow, but walleye are considerably less abundant and consume fewer juvenile salmonids (e.g. Rieman et al. 1991). Originally introduced into the upper Columbia basin, walleye since the 1970s gradually have spread downstream throughout the lower mainstem. Significant numbers of walleye have become established in Bonneville Reservoir and between Bonneville Dam and the estuary. Walleye population sizes are quite variable and driven by periodic large year classes that occur during warm, low flow springs. Walleye are subject to a small, directed sport fishery but were not included in the sport reward fishery because projected exploitation effects on salmonids were low. Unlike pikeminnow, most walleye predation occurs in smaller individuals not readily caught by anglers and unaffected by the compounding effects of annual exploitation.

Other introduced fishes—including smallmouth bass and channel catfish—also have been found to consume significant numbers of juvenile salmonids. However, these species are more significant problems in upstream areas than in the lower river where their abundance is low.

Piscivorous birds congregate near dams and in the estuary around man-made islands where they consume large numbers of outmigrating juvenile salmon and steelhead (Roby et al. 1998). Caspian terns, cormorants, and gull species are the major avian predators (NMFS 2000a).

While some predation occurs at dam tailraces and juvenile bypass outfalls, by far the greatest numbers of juveniles are consumed as they migrate through the Columbia River estuary. Ruggerone (1986) estimated that gulls consumed 2% of the juvenile salmon and steelhead passing Wanapum Dam but comparable estimates have not been made for Bonneville Dam. Roby et al. (1998) estimated that avian predators consumed 10-30% of the total estuarine salmonid smolt population in 1997. (Additional discussion of bird predation in the estuary is included in section 3.5.3.1.)

Marine mammals prey on adult salmon, but the significance is unclear. Seals and sea lions are common in and immediately upstream of the Columbia River estuary and are regularly observed up to Bonneville Dam. Seals and sea lions are regularly reported to prey on adult salmon and steelhead, although diet studies indicate that other fish comprise the majority of their food. Large numbers of pinnipeds might translate into significant salmon mortality despite this occasional use. However, it is difficult to interpret the significance of this mortality factor for salmon, considering that large pinniped populations have always been present in the Columbia River. However, current marine mammal predation may be proportionally more significant, since all sources of mortality on depressed stocks become more important. Their numbers were reduced by hunting (including bounty hunters) and harassment from the late 1800s until the Federal Marine Mammal Protection Act (FMMPA) was adopted in 1972. Their numbers have significantly increased since the adoption of FMMPA. Fishers historically viewed seals and sea lions as competitors and the old Fish Commission of Oregon funded a control program. These mammals can be troublesome to sport and commercial fishers by taking hooked or net-caught fish before they can be landed.

3.4.5 Adult Dam Passage

Fish ladders provide for upstream dam passage of adult salmon but are not 100% effective. Salmon may have difficulty locating ladder entrances and fish also may fall back over the dam after exiting from the fish ladder (Reichel and Bjornn 2003). These problems can result in significant upstream passage losses at dams. Average per dam survival rates in the lower Columbia River mainstem have been estimated at approximately 89% for spring chinook, 94% for fall chinook, and 95% for steelhead based on fish counts at successive dams, fallbacks after dam passage, harvest, and tributary escapements (*US v. Oregon* Technical Advisory Committee, unpublished data).

Fallback of adult salmon and steelhead after dam passage can be substantial; high levels of fallback are typically associated with periods of high flow and spill (Bjornn and Peary 1992). Keefer and Bjornn (1999) estimated recent fallback rates at Bonneville Dam of 12-15% for chinook (1996–98), 4-13% for sockeye (1997), and 5-10% for steelhead (1996–97). Fallback was substantially greater at the Bradford Island ladder exit at Bonneville Dam than the Washington shore ladder (Bjornn et al. 1998); 14-21% of sockeye and chinook salmon fell back over the dam (Reichel and Bjornn 2003). Adult salmonids that fallback over dams do not translate into a total loss as some fish may re-enter the fish ladder, successfully pass the dam, and continue upstream migration.

Passage delays in dam tailraces result from dynamic and complex flow patterns and the relatively small volume of water comprising ladder attraction flows. Fish may require a few hours or a few days to locate ladders once they reach the tailrace (Table 3-43). The delay is generally longer when flows are high and when large amounts of water are being spilled (NMFS

2000). Ladder systems at Columbia and Snake River dams are operated to produce hydraulic conditions that maximize fish attraction and minimize delay. Operations are based on criteria developed by NMFS, ACOE, and state and tribal fishery managers. The criteria relate to such factors as water depth and head on the gate entrances, collection channels, ladder flows and ladder exits (NMFS 2000).

Table 3-43. Median entry times in days into Bonneville Dam fish bypass system by upstream chinook and steelhead migrants, 1996–97.

Species	1996	1997
Chinook	2.0	2.2
Steelhead	1.9	0.3

From NMFS (2000a)

Passage delays at dams are at least partially offset by more rapid movement of fish through slackwater reservoirs, so the net effect of dam and reservoir construction on upstream travel time for adults is unclear. The OFC (1960) found that, prior to impoundments in the Snake River, chinook migration rates averaged 11-15 mi/day (17.7-24.1 km/day). Chinook salmon migration rates through the Snake River reservoirs in 1991-93 ranged from 19.3 to 40.4 mi/day (31-65 km/day), while migration rates through free-flowing river sections above Lower Granite Dam ranged were generally less than 6.8 mi/day (11 km/day) (Bjornn 1998). Bjornn et al. (1999) estimated that median travel time for salmon to pass the four dams and reservoirs in the lower Snake River in 1993 was the same or less with the dams as without the dams. Quinn et al. (1997) found that travel time between Bonneville and McNary dams over the last 40 years has decreased.

3.4.6 Juvenile Dam Passage

Delay and mortality of juvenile salmon at mainstem dams has proved to be one of the most difficult and contentious problems associated with hydropower development. Smolts typically migrate near mid-channel in the upper water column where water velocities are greatest. Delay results as juveniles stack up in dam forebays during daylight, when they are reluctant to sound to enter turbine or spillway intakes. Juveniles may experience substantially different mortality rates depending on whether passage occurs via turbines, spill, or a fish bypass system. Fish passage at Bonneville Dam is particularly complex, with two passage routes at each of the two powerhouses, plus an unattached spillway.

The turbines are the most hazardous passage route. Mortality results from abrupt pressure changes in the turbines and from mechanical injury. Iwamoto and Williams (1993) reviewed fish survival data through the Columbia River system and concluded that turbine survival, taken as a whole, averaged 90% per dam. Balloon tag tests conducted by Normandeau Associates Inc. indicated survival rates in the mid-90% range (Normandeau Associates Inc. et al. 1995, 1996, 1999).

Spillways are a much safer passage route than turbines (Whitney et al. 1997). Holmes (1952) reported that spillway survival at Bonneville Dam was 97% using pooled data and 96% using weighted averages. Improvements to spillway and tailrace configurations have been implemented since Holmes' study, and more recent research at other Columbia and Snake River projects have estimated typical spill survival to be around 98-100% (NMFS 2000). Historical operations attempted to minimize spill in order to maximize power generation. Current practices provide dedicated spill to facilitate dam passage by juveniles.

Juvenile bypass systems to divert fish from turbine intakes are now in place at most mainstem dams in the Columbia River system, including Bonneville Dam. Most systems involve submersible traveling screens that project downward into the intakes of turbines and deflect fish upward from the turbine intake into the gatewell. Fish guidance efficiency (FGE) measures the proportion of fish entering turbine intakes that is guided into the bypass system (Brege et al. 1988). FGE varies by species, stock, fish condition, time of day, dam, turbine unit, season, environmental conditions, and project operation (NMFS 2000). Typical values for Bonneville Dam range from 16 to 44% (Table 3-44). Bypass mortality rates are typically quite low (<1%). The Bonneville second powerhouse bypass has been a conspicuous exception; past survival problems have recently been ameliorated by modifying the collection channel to improve hydraulic conditions and a new conveyance pipe and outfall have been installed to reduce predation problems (Gilbreath and Prentice 1999).

Table 3-44. Average juvenile fish guidance efficiencies (NMFS 2000) and 1988–97 bypass mortality rates (Martinson et al. 1998) at Bonneville Dam.

	Fish Guidance Efficiency (%)		Bypass Mortality	
Species	**Powerhouse 1**	**Powerhouse 2**	**Powerhouse 1**	**Powerhouse 2**
Yearling Chinook	38	44	0.1%	1.5%
Subyearling Chinook	16	18	0.4%	1.4%
Steelhead	41	48	0.1%	1.1%
Coho	—	—	0.1%	0.9%
Sockeye	—	—	0.4%	7.9%

3.4.7 Other Factors

Other factors have the potential to affect salmon in transit through the Columbia River mainstem. Contaminants are one example, although their significance to migrating salmon is likely to be much less than for resident fish. Significant levels of dioxins/furans, DDT, and metals have been identified in lower Columbia River fish and sediment samples. The direct and indirect effects of channel dredging are also a source of significant speculation and limited data exists, although the potential for dredging effects is limited by in-water work periods selected outside salmon migration timeframes. Stranding of juvenile salmonids on shorelines by ship wakes also has been documented, but estimates of its magnitude are not available.

3.5 Estuary and Lower Columbia River

The lower Columbia River mainstem and estuary subbasins are treated in detail in Volume II, Chapter I of this Technical Foundation. This section is intended to briefly and generally describe the limiting factors in the estuary and lower mainstem as they relate to salmonid survival, production, and life history diversity, as well as the abundance and status trends of other focal species populations in the estuary and lower mainstem.

3.5.1 Function of the Estuary and Lower Mainstem

Estuaries have important impacts on juvenile salmonid survival. Estuaries provide juvenile salmonids an opportunity to achieve the critical growth necessary to survive in the ocean (Neilson and Geen 1986, Wissmar and Simenstad 1988 as cited in Nez Perce et al. 1995, Aitkin 1998 as cited in USACE 2001, Miller and Sadro 2003). Juvenile chinook salmon growth in estuaries is often superior to river-based growth (Rich 1920a, Reimers 1971, Schluchter and Lichatowich 1977). Estuarine habitats provide young salmonids with a productive feeding area, free of marine pelagic predators, where smolts can undergo physiological changes necessary to acclimate to the saltwater environment. Studies conducted by Emmett and Schiewe (1997) in the early 1980s have shown that favorable estuarine conditions translate into higher salmonid survival. These findings are consistent with the results of Kareiva et al. (2001, as cited in Fresh et al. 2003); they demonstrated that improvement of juvenile salmon survival during the estuarine and early ocean stage would significantly improve salmon population growth rates.

Juxtaposition of high-energy areas with ample food availability and sufficient refuge habitat is a key habitat structure necessary for salmonid growth and survival in the estuary. In particular, tidal marsh habitats, tidal creeks, and associated complex dendritic channel networks may be especially important to subyearlings as areas of both high insect prey density, and as potential refuge from predators afforded by sinuous channels, overhanging vegetation, and undercut banks (McIvor and Odum 1988). Furthermore, areas of adjacent habitat types distributed across the estuarine salinity gradient may be necessary to support annual migrations of juvenile salmonids (Simenstad et al. *in press*, as cited in Bottom et al. 2001). For example, as subyearlings grow, they move across a spectrum of salinities, depths, and water velocities. For species like chum and ocean type chinook salmon that rear in the estuary for extended time periods, a broad range of habitat types in the proper proximities to one another may be necessary to satisfy feeding and refuge requirements within each salinity zone. Additionally, the connectedness of these habitats likely determines whether juvenile salmonids are able to access the full spectrum of habitats they require (Bottom et al. 1998).

Juvenile salmonids must continually adjust their habitat distribution in relation to twice-daily tidal fluctuations as well as seasonal and anthropogenic variations in river flow. Juveniles have been observed to move from low-tide refuge areas in deeper channels to salt marsh habitats at high tide and back again (Healey 1982). These patterns of movement reinforce the belief that access to suitable low-tide refuge near marsh habitat is an important factor in production and survival of salmonid juveniles in the Columbia River estuary.

The importance of proximally available feeding and refuge areas may hold true even for species that move more quickly through the estuary. For example, Dawley (1989) found prey items in the majority of stomachs of salmon smolts known to migrate through the Columbia estuary quickly (i.e., days), indicating that these smolts are utilizing estuarine resources.

Additionally, radio-tagged coho in Grays Harbor estuary moved alternatively from low velocity holding habitats to strong current passive downstream movement areas (Moser et al. 1991). Further, Fresh et al. (2003) reported that both small and large chinook salmon (i.e., ocean- and stream-type chinook from upper and lower basin populations) utilized peripheral marsh and forested wetland habitat in the Columbia River estuary. Consistent with these observations, Dittman et al. (1996) suggest that habitat sequences at the landscape level may be important even for species and life history types that move quickly through the estuary during the important smoltification process, as salmon gather the olfactory cues needed for successful homing and these cues may depend on the environmental gradients experienced during migrations.

3.5.2 Habitat Change

Anthropogenic factors have substantially influenced current habitat conditions in the lower Columbia River mainstem and estuary. The primary anthropogenic factors that have determined estuary and lower mainstem habitat conditions include hydrosystem construction and operation (i.e., water regulation), channel confinement (primarily diking), channel manipulation (primarily dredging), and floodplain development and water withdrawal for urbanization and agriculture. Generally, these anthropogenic factors have influenced estuary and lower mainstem habitat conditions by altering hydrologic conditions, sediment transport mechanisms, and/or salinity and nutrient circulation processes. Often, there are no simple connections between a single factor and a single response, as many of the factors and responses are interrelated. Further, evaluation of anthropogenic factors are difficult to separate from concurrent natural variation.

3.5.2.1 River Flow

Flow effects from upstream dam construction and operation, irrigation withdrawals, shoreline anchoring, channel dredging, and channelization have significantly modified estuarine habitats and have resulted in changes to estuarine circulation, deposition of sediments, and biological processes (ISAB 2000, Bottom et al. 2001, USACE 2001, Johnson et al. 2003b). Flow regulation in the Columbia River basin has been a major contributor to the changes that have occurred in the estuary from historic conditions. The predevelopment flow cycle of the Columbia River has been modified by hydropower water regulation and irrigation withdrawal (Thomas 1983, Sherwood et al. 1990 as cited in Nez Perce et al. 1995, Weitkamp 1994, NMFS 2000c, Williams et al. 2000, Bottom et al. 2001, USACE 2001).

Winter drawdown of reservoirs and filling of reservoirs during the spring runoff season has decreased spring freshet magnitude and increased flows over the rest of the year. The historic flow records at The Dalles, Bonneville Dam, and Beaver, Oregon, demonstrate that spring freshet flows have been reduced by about 50% and winter flows have increased about 30% (Figure 3-98). Most of the spring freshet flow reduction is attributed to flow reduction, about 20% is a result of irrigation withdrawals, and only a small portion (5%) is connected to climatic change (Bottom et al. 2001).

Reduction of maximum flow levels, dredged material deposition, and diking have all but eliminated overbank flows in the Columbia River (Bottom et al. 2001), resulting in reduced large woody debris recruitment and riverine sediment transport to the estuary. Overbank flows were historically a vital source of new habitats. Moreover, historic springtime overbank flows greatly increased habitat opportunity into areas that at other times are forested swamps or other seasonal wetlands. Historic bankfull flow levels were common prior to 1975 but are rare today. Further, the season when overbank flow is most likely to occur today has shifted from spring to winter, as western subbasin winter floods (not interior subbasin spring freshets) are now the major source of peak flows (Bottom et al. 2001, Jay and Naik 2002).

3.5.2.2 Channel Alterations

Thomas (1983) suggested that channel confinement (i.e. diking) is particularly detrimental to estuary habitat capacity because it entirely removes habitat from the estuarine system, while other anthropogenic factors change estuary habitats from one type to another. The lower mainstem and estuary habitat in the Columbia River has, for the most part, been reduced to a single channel where floodplains have been reduced in size, off-channel habitat has been lost or disconnected from the main channel, and the amount of large woody debris has been reduced (NMFS 2000c). Dikes prevent overbank flow and affect the connectivity of the river and floodplain (Tetra Tech 1996); thus, the diked floodplain is higher than the historic floodplain and inundation of floodplain habitats only occurs during times of extremely high river discharge (Kukulka and Jay 2003). It is estimated that the historical estuary had 75 percent more tidal swamps than the current estuary partially because tidal and flood waters could reach floodplain areas that are now diked or otherwise disconnected from the main channel (USACE 2001, Johnson et al. 2003b).

Development and maintenance of the shipping channel has greatly affected the morphology of the estuary. The extensive use of jetties and pile dikes to maintain the shipping channel has impacted natural flow patterns and large volumes of sediments are dredged annually. Dredged materials are disposed of in the ocean, in the flow adjacent to the shipping channel, along shorelines, or on upland sites. Annual maintenance dredging since 1976 has averaged 3.5 million cubic yards per year in the estuary. By concentrating flow in one deeper main channel, the development of the navigation channel has reduced flow to side channels and peripheral bays.

3.5.2.3 Sediment Budget

Sediments in the estuary may be marine- or freshwater-derived and are transported via suspension in the water column or bed load movement. Riverine sediments available for transport has decreased as a result of dam construction; reservoirs restrict bedload movement and trap upstream supply of sediments. Sand sediments are vital to natural habitat formation and maintenance in the estuary; dredging and disposal of sand and gravel have been among the major causes of estuarine habitat loss over the last century (Bottom et al. 2001).

Sediment transport is non-linearly related to flow; thus, it is difficult to accurately apportion causes of sediment transport reductions into climate change, water withdrawal, or flow regulation (Jay and Naik 2002). However, the largest single factor in reduced sediment transport appears to be the reduction of spring freshet flow as a result of water regulation and irrigation withdrawal. Recent analyses indicate a two-thirds reduction in sediment-transport capacity of the Columbia River relative to the pre-dam period (Sherwood et al. 1990, Gelfenbaum et al. 1999). Therefore, flow reductions affect estuary habitat formation and maintenance by reducing sediment transport (Bottom et al. 2001, USACE 2001). The reduction in sand and gravel transport has been higher (>70% reduction compared to predevelopment flow) than for silt and clay transport (Bottom et al. 2001), which has important implications for habitat formation and food web dynamics.

Construction of the north and south jetties at the Columbia River mouth significantly increased sediment accretion in nearby marine littoral areas. Ocean currents that formerly transported sediments alongshore were disrupted and accretion, particularly in areas adjacent to the river mouth (i.e. Long Beach, Clatsop Spit), increased significantly in the late 1800s and early 1900s. Sediment accumulation rates have slowed since 1950, potentially as a result of reduced sediment supply from adjacent deltas or the Columbia River (Kaminsky et al. 1999). Because of the decreased sediment supply from the Columbia River and ebb-tidal deltas, recent modeling results indicate that the shorelines immediately north of the historic sediment source areas at the entrance to the Columbia River are susceptible to erosion in the future (Kaminsky et al. 2000).

3.5.2.4 Climate

Evaluation of anthropogenic factors are complicated by climatic effects. Variations in climate-driven Columbia River discharge occur in time scales from years to centuries (Chatters and Hoover 1986, 1992 as cited in Bottom et al. 2001). The Columbia Basin's response to climatic cycles is governed by the basin's latitudinal position; climate in the region displays a strong response to both the Pacific Decadal Oscillation (PDO) and El Niño Southern Oscillation Index (ENSO) cycles (Mantua et al. 1997 as cited in Bottom et al. 2001). The El Niño weather pattern produces warm ocean temperatures and warm, dry conditions throughout the Pacific Northwest. The La Niña weather pattern is typified by cool ocean temperatures and cool/wet weather patterns on land. Climate directly affects river flow and observed changes to flow are often substantial. Further, El Niño patterns result in poor ocean productivity in the Pacific Northwest and California, as was observed in the mid 1990s (Hare et al. 1999). The effects of poor estuary and mainstem habitats are exaggerated during periods of low ocean productivity.

Current climate projections predict gradual warming of the region, potentially with higher precipitation, particularly in winter (Hamlet and Lettenmaier 1999). The predicted future climate conditions will possibly reduce the likelihood of spring freshets caused by heavy spring rain on late snowpack because warmer temperatures will not allow the accumulation of snow late into the spring. This rain on snow freshet pattern has historically produced the most substantial peaks in river discharge (Bottom et al. 2001). However, despite our ability to measure changes in climate, Bottom et al. (2001) discussed the difficulty in separating climate versus anthropogenic effects on river discharge and the habitat-forming processes it governs.

3.5.2.5 Measured Changes in Estuary Habitats

Thomas (1983) documented substantial changes to estuary habitats from historic to current conditions as summarized below. The area covered by Thomas (1983) encompassed RM 0-46.5; similar analyses of habitat changes for the lower mainstem up to Bonneville Dam (RM 146) are not available because little historical data exist upstream of RM 102 (Johnson et al. 2003b). Estuary-wide tidal marsh and tidal swamp acreage has decreased 43% and 77%, respectively, from 1870 to 1983, primarily as a result of dikes and levees that have disconnected the main channel from these floodplain habitats and also from water regulation that has decreased historic peak flows that previously provided water to these habitats. Losses of tidal marsh habitat have been most extensive in Youngs Bay, where a loss of over 6,000 acres was documented. Extensive tidal swamp habitat has been lost in all estuary areas where this habitat was historically present. Losses of medium- and deep-water habitat acreage have been less severe (25% and 7%, respectively). Acreage of medium-depth water habitat was lost in all areas of the estuary except the upper estuary, where a slight increase in acreage was observed; acreage loss was greatest in the entrance, Cathlamet Bay, and Baker Bay areas of the estuary. Similarly, deep-water habitat acreage was lost in most areas of the estuary; losses were highest in the Baker Bay and upper estuary areas. Only shallows/flats estuary habitat realized a net increase 10% in acreage from 1870 to 1983. This increase in acreage was primarily a result of water regulation that has decreased historic peak erosive flows and decreased erosion following construction of the jetties at the river mouth. In total, 36,970 acres (23.7%) of the estuarine habitat acreage has been lost from 1870 to 1983. During this period, lost estuarine habitats were converted to the following non-estuarine habitats: developed floodplain (23,950 acres), natural and filled uplands (5,660 acres), non-estuarine swamp (3,320 acres), non-estuarine marsh (3,130 acres), and non-estuarine water (910 acres).

3.5.3 Ecological Interactions

3.5.3.1 Predation

Significant numbers of salmon are lost to fish, bird, and marine mammal predators during migration through the mainstem Columbia River. Predation likely has always been a significant source of mortality but has been exacerbated by habitat changes. Piscivorous birds congregate near dams and in the estuary around man-made islands and consume large numbers of emigrating juvenile salmon and steelhead (Roby et al. 1998). Caspian terns, cormorants, and gull species are the major avian predators (NMFS 2000a). While some predation occurs at dam tailraces and juvenile bypass outfalls, by far the greatest numbers of juveniles are consumed as they migrate through the Columbia River estuary. Marine mammals prey on adult salmon, but the significance is unclear (marine mammal predation is discussed in section 3.4.4.

Caspian terns are native to the region but were not historically present in the lower Columbia River mainstem and estuary; they have recently made extensive use of dredge spoil habitat and are a major predator of juvenile salmonids in the estuary. The terns are a migratory species whose nesting season coincides with salmonid outmigration timing. Since 1900, the tern population has shifted from small colonies nesting in interior California and southern Oregon to large colonies nesting on dredge spoil islands in the Columbia River and elsewhere (NMFS 2000c). Many of these Columbia River dredge spoil islands were created as a result of dredging the navigational channel after the eruption of Mt. St. Helens in 1980 although Rice Island was initially constructed from dredge spoils around 1962 (Geoffrey Dorsey, USACE, personal communication). Caspian terns did not nest in the estuary until 1984 when about 1,000 pairs apparently moved from Willapa Bay to nest on East Sand Island. Those birds (and others) moved to Rice Island in 1987 and the colony expanded to 10,000 pairs. Diet analysis has shown that juvenile salmonids make up 75% of food consumed by Caspian terns on Rice Island. Roby et al. (1998) estimated Rice Island terns consumed between 6.6 and 24.7 million salmonid smolts in the estuary in 1997, and that avian predators consumed 10-30% of the total estuarine salmonid smolt population in that year. However, there are no data to compare historical and modern predation rates or predator populations. Further, current predation studies are limited because of the unknown effects hatchery rearing and release programs have had on salmon migration behavior and predator consumption. Nevertheless, evidence suggests that current predator populations could be a substantial limiting factor on juvenile salmon survival (Bottom et al. 2001). Ryan et al. (2003) estimated species-specific predation by Caspian terns from 1988-2000; predation by Caspian terns was consistently highest on steelhead (9.4-12.7%) and consistently lowest on yearling chinook salmon (1.6-2.9%) while predation on coho salmon was intermediate (3.6-4.1%).

Recent management actions have been successful in discouraging Caspian tern breeding on Rice Island while encouraging breeding on East Sand Island, which may decrease predation on juvenile salmonids. However, estimates of potential decreases in salmonid mortality from reduced tern predation assume that there is no compensatory mortality later in the life cycle (Fresh et al. 2003). This assumption may not be realistic, as Roby et al. (2003) hypothesized that tern predation was 50% additive. Thus, actual improvements in juvenile salmonid survival resulting from management actions that reduce tern predation would likely be lower than current estimates (Fresh et al. 2003).

Northern pikeminnow are also a significant predator on salmonid smolts in the lower Columbia River as discussed above in section 3.4.4. However, pikeminnow abundance in the estuary is likely low because of salinity; thus, pikeminnow predation is not likely to be an important limiting factor on juvenile salmonids in the estuary.

3.5.3.2 Competition

American shad (*Alosa sapidissima*) populations have grown substantially since introduction into the Columbia River system in 1885 (Welander 1940, Lampman 1946). In recent years, 2-4 million adults have been counted annually at Bonneville Dam. The transition of the estuarine food web from a macrodetritus to microdetritus base (i.e. increased importation of plankton from upstream reservoirs) has benefited zooplanktivores, including American shad (Sherwood et al. 1990). Because of the abundance of American shad in the Columbia River system, studies have been launched to investigate species interactions between shad, salmonids, and other fish species such as northern pikeminnow, smallmouth bass, and walleye (Petersen et al. In press). A pattern is slowly emerging that suggests the existence of American shad is changing trophic relationships within the Columbia River. Because of their abundance, consumption rates, and consumption patterns, American shad may have modified the estuarine food web. One study found that in the Columbia River estuary and lower mainstem (up to RKm 62) shad diet overlapped with subyearling salmonid diets, which may indicate competition for food. Juvenile shad and subyearling salmonids also utilize similar heavily vegetated backwater habitats (McCabe et al. 1983). Another study examined shad abundance as prey contributing to faster growth rates of northern pikeminnow, which in turn are significant predators of juvenile salmonids (Petersen et al. In press). Commercial harvest has been considered as a means to reduce the abundance of American shad in the Columbia River, but harvest has been restricted because the shad spawning run coincides with the timing of depressed runs of summer and spring chinook, sockeye, and summer steelhead (WDFW and ODFW 2002).

Estuaries may be “overgrazed” when large numbers of ocean-type juveniles enter the estuary *en masse* (Reimers 1973, Healey 1991). Food availability may be negatively affected by the temporal and spatial overlap of juvenile salmonids from different locations; competition for prey may also develop when large releases of hatchery salmonids enter the estuary (Bisbal and McConnaha 1998), although this issue remains unresolved (Lichatowich 1993 as cited in Williams et al. 2000). Reimer (1971) suggested that density-dependence affects growth rate and hypothesized that fall chinook growth in the Sixes River was poor from June to August because of greater juvenile densities in the estuary but that increased growth rate in the fall resulted from smaller population size and a better utilization of the whole estuary.

The potential exists for large-scale hatchery releases of fry and fingerling ocean-type chinook salmon to overwhelm the production capacity of estuaries (Lichatowich and McIntyre 1987). However, Witty et al. (1995) could not find any papers or studies that evaluated specific competition factors between hatchery and wild fish in the Columbia River estuary. Simenstad and Wissmar (1984) cautioned that the estuary condition may limit rearing production of juvenile chinook, and many other studies have demonstrated the importance of the estuary to early marine survival and population fitness. However, rivers such as the Columbia, with well-developed estuaries, are able to sustain larger ocean-type populations than those without (Levy and Northcote 1982).

3.5.4 Effects of Ecosystem Changes on Salmonids

Natural and anthropogenic factors have negatively altered the habitat-forming processes, available habitat types, and the estuarine food web, resulting in decreased salmonid survival and production. The most significant habitat effects have resulted from modified river flow and channel manipulations. River flow changes have occurred as a result of hydrosystem operations, water withdrawals for agriculture and urban development, and decreased precipitation from climate changes. Channel manipulations encompass a suite of factors, but primarily refer to dikes that disconnect the river and floodplain or dredging that alters the river's bathymetry. Subsequently, estuary and lower mainstem habitat changes have facilitated the increase of important juvenile salmonid predators (specifically, Caspian terns and northern pikeminnow), thereby decreasing juvenile salmonid survival and abundance through the lower mainstem and estuary.

In a recent analysis of limiting factors, Fresh et al. (2003) evaluated the effects of river flow, habitat quality/availability, contaminant toxicity, and Caspian tern predation on juvenile salmonid abundance, life history diversity, and viable salmon population criteria. They concluded that the most important limiting factors are flow and habitat changes and the primary effects are on shallow water habitats and the salmonid life history strategies that depend on these habitats. Thus, habitat losses that have occurred in the estuary and lower mainstem (namely shallow water, peripheral habitats such as wetlands and side channels) are more limiting on subyearling life history strategies (commonly ocean-type life history) than yearling life history strategies (stream-type salmonids) that are not critically associated with these habitat types (Fresh et al. 2003). They further evaluated the effects of each limiting factor on viable salmon population criteria (abundance, population growth rate, spatial structure, and diversity; McElhany et al. 2000) and concluded that flow and habitat substantially limit all viability criteria for ocean-type salmonids.

3.5.4.1 Decreased Habitat Diversity and Productivity

Historically, floodwaters of the Columbia River inundated the margins and floodplains along the estuary, allowing juvenile salmon access to a wide expanse of low-velocity marshland and tidal channel habitats (Bottom et al. 2001). Flooding occurred frequently and was important to habitat diversity and complexity. Historical flooding also allowed more flow to off-channel habitats (i.e. side channels and bays) and deposited more large woody debris into the ecosystem. Historically, seasonal flooding increased the potential for salmonid feeding and resting areas in the estuary during the spring/summer freshet season by creating significant tidal marsh vegetation and wetland areas throughout the floodplain (Bottom et al. 2001). These conditions rarely exist today because of hydropower system water regulation.

Salmonid production in estuaries is supported by detrital food chains (Healey 1979, 1982). Therefore, habitats that produce and/or retain detritus, such as emergent vegetation, eelgrass beds, macro algae beds, and epibenthic algae beds, are particularly important (Sherwood et al. 1990). Diking and filling activities in the estuary have likely reduced the rearing capacity for juvenile salmonids by decreasing the tidal prism and eliminating emergent and forested wetlands and floodplain habitats adjacent to shore (Bottom et al. 2001, NMFS 2000c). Dikes throughout the lower Columbia River and estuary have disconnected the main channel from a significant portion of the wetland and floodplain habitats. Further, filling activities (i.e. for agriculture, development, or dredge material disposal) have eliminated many wetland and floodplain habitats. Thus, diking and filling activities have eliminated the emergent and forested wetlands and floodplain habitats that many juvenile salmonids rely on for food and refugia, as well as eliminating the primary recruitment source of large woody debris that served as the base of the historic food chain. The current estuary food web is microdetritus based, primarily in the form of imported phytoplankton production from upriver reservoirs that dies upon exposure to salinity in the estuary (Bottom and Jones 1990 as cited in Nez Perce et al. 1995, Bottom et al. 2001, USACE 2001). The historic macrodetritus-based food web was distributed throughout the lower river and estuary, but the modern microdetritus-based food web is focused on the spatially confined ETM region of the estuary (Bottom et al. 2001). This current food web is primarily available to pelagic feeders and is a disadvantage to epibenthic feeders, such as salmonids (Bottom and Jones 1990 as cited in Nez Perce et al. 1995, Bottom et al. 2001, USACE 2001).

Columbia River mainstem reservoirs trap sediments and nutrients, as well as reduce sediment bedload movement, thereby reducing sediment and nutrient supply to the lower Columbia River. The volume and type of sediment transported by the mainstem Columbia River has profound impacts on estuarine habitat formation, food webs, and species interactions. For example, organic matter associated with the fine sediment supply maintains the majority of estuarine secondary productivity (Simenstad et al. 1990, 1995 as cited in Bottom et al. 2001). Also, turbidity (as determined by suspended sediments) regulates light penetration needed for primary production and decreases predator efficiency on juvenile salmonids. Further, the type of sediment transported has profound effects on habitat formation. Sand and gravel substrates are important components of preferred salmonid habitat in the estuary, but sand and gravel transport has been reduced more (>70% reduction compared to predevelopment flow) than silt and clay transport (Bottom et al. 2001).

Additionally, the decreased habitat diversity and modified food web has decreased the ability of the lower Columbia River mainstem and estuary to support the historic diversity of salmonid life history types that used streams, rivers, the estuary, and perhaps the Columbia River plume as potential rearing areas. Bottom et al. (2001) identified several forms of ocean-type chinook life histories, based on the scale pattern, length, and time of capture data collected by Rich (1920). Wissmar and Simenstad (1998) and Bottom et al. (2001) suggest there may be as many as 35 potential ocean-type chinook salmon life history strategies. Bottom et al. (2001) suggested that human effects on the environment have caused chinook life history patterns to be more constrained and homogenized than historic data show. Most modern ocean-type chinook fit into one of three groups: subyearling migrants that rear in natal streams, subyearling migrants that rear in larger rivers and/or the estuary, or yearling migrants. Abundance patterns of juvenile chinook in the estuary may have shifted somewhat toward more yearling juveniles because of hatchery management practices.

3.5.4.2 Altered Migration Patterns

Hydrologic effects of the Columbia River dams include water level fluctuations, altered seasonal and daily flow regimes, reduced water velocities, and reduced discharge volume. Altered flow regimes can affect the migratory behavior of juvenile and adult salmonids. For example, water level fluctuations associated with hydropower peak operations may reduce habitat availability, inhibit the establishment of aquatic macrophytes that provide cover for fish, and strand juveniles during the downstream migration. Reservoir drawdowns reduce available habitat which concentrates organisms, potentially increasing predation and disease transmission (Spence et al. 1996 as cited in NMFS 2000c).

Water regulation, as part of hydropower system operations, has drastically reduced historic spring freshet flows and altered juvenile salmon emigration behavior. Often, historic lower Columbia River spring freshet flows were approximately four times the winter low flow levels. Today, spring freshet flows are only about twice the winter low flow level, which is now generally increased during reservoir drawdown in winter. Spring freshets are very important to the emigration of juvenile salmonids; freshet flows stimulate salmon downstream migration and provide a mechanism for rapid migrations.

3.5.5 Effects of Ecosystem Changes on Other Species

Despite substantial changes to the Columbia River estuary and lower mainstem ecosystem, many focal species have stable or increasing abundance trends. Regardless of their current abundance trend, implementation of an ecosystem-based approach to recovery of ESA-listed species indicates that an evaluation of effects of each recovery action on other species is warranted. The status and abundance trends of the non-salmonid focal species in the Columbia River estuary and lower mainstem is summarized below. These and other species are fully treated in Volume III.

- The lower Columbia white sturgeon population is among the largest and most productive in the world. The deep water habitats in which sturgeon are commonly associated remain available throughout the lower mainstem and estuary. Hydrosystem development and operation has artificially created what functionally amounts to white sturgeon spawning channels downstream from Bonneville Dam, resulting in reliable annual recruitment (L. Beckman USGS (retired), G. McCabe Jr. NMFS (retired), M. Parsley, USGS, Cook Washington, personal communication). Further, sturgeon have demonstrated substantial variability in feeding locations; white sturgeon have potentially benefited from changes to the estuarine food web.

- NOAA Fisheries completed a status review for green sturgeon in 2003 and determined that listing under the Endangered Species Act was not warranted but are a candiate species. Green sturgeon spend most of their life in nearshore marine and estuarine waters from Mexico to southeast Alaska (Houston 1988; Moyle et al. 1995). While green sturgeon do not spawn in the Columbia Basin, significant populations of subadults and adults are present in the estuary during summer and early fall. Green sturgeon are occasionally observed as far upriver as Bonneville Dam. These fish may be seeking warmer, summer river waters in the northern part of their range.

- The northern pikeminnow population has flourished with habitat changes in the mainstem Columbia River and its tributaries. The highest density of northern pikeminnow in the mainstem Columbia River below the Snake River confluence is found in the lower mainstem from The Dalles to the estuary. A pikeminnow management program has been implemented in the Columbia and Snake rivers in an attempt to reduce predation mortality of juvenile salmonids by reducing numbers of large, old pikeminnow that account for most of the losses. A bounty fishery program for recreational anglers is aimed at balancing pikeminnow numbers rather than eliminating the species and has also stimulated development of a popular fishery.

- Eulachon (smelt) numbers and run patterns can be quite variable; low runs during the 1990s raised considerable concern by fishery agencies. Current patterns show a substantial increase in run size compared to the 1990s. The low returns in the 1990s are suspected to be primarily a result of low ocean productivity. Eulachon support a popular sport and commercial dip net fishery in the tributaries, as well as a commercial gill-net and small trawl fishery in the Columbia. They are used for food and are also favored as sturgeon bait. Nevertheless, hydropower development on the Columbia River has decreased the available spawning habitat for eulachon. Prior to the completion of Bonneville Dam, eulachon were reported as far upstream as Hood River, Oregon (Smith and Saalfeld 1955). Additionally, dredging has the potential to impact adult and juvenile eulachon (Larson and Moehl 1990); dredging operations in the lower Columbia River have made local substrate unstable for the incubation of eulachon eggs. Thus, future dredging operations should be scheduled to avoid eulachon spawning areas during peak spawning times (Romano et al. 2002).

- Field observations and trapper data indicate the river otter population abundance in the lower Columbia River mainstem and estuary was relatively low in the early 1980s (Howerton et al. 1984); low abundance may be the normal equilibrium level for river otters in this region. River otters are concentrated in shallow water tidal sloughs and creeks associated with willow-dogwood and Sitka spruce habitats located primarily in the Cathlamet Bay area. Although dikes throughout the estuary have disconnected substantial amounts of side channel and floodplain habitats from the mainstem, the Cathlamet Bay area remains as one of the most intact and productive tidal marsh and swamp habitat throughout the entire estuary. Further, because river otters are capable of traveling over land, it is not understood how the loss of habitat connectivity of side channel and floodplain habitat has affected species' behaviors such as foraging, resting, mating, and rearing. Contaminants in river otter tissue may have adverse physiological effects, however, data suggests that the effects may be temporary (Tetra Tech 1996).

- Habitat conversion, losses, and isolation coupled with low population productivity are currently the most important threats to Columbian white-tailed deer population viability. Nevertheless, the Columbian white-tailed deer population appears stable at low numbers and shows initial indicators of increasing abundance and productivity. In 1999, the USFWS proposed to delist the Columbian white-tailed deer throughout the entire range, but public concern over delisting motivated USFWS to withdraw the delisting proposal.

Columbian white-tailed deer are present in low-lying mainland areas and islands in the Columbia River upper estuary and along the river corridor. They are most closely associated with Westside oak/dry Douglas fir forest within 200m of a stream or river; acreage of this habitat type has decreased substantially from historic to current conditions. Restoration of contiguous preferred habitat is vital to population recovery.

- The Caspian tern breeding population in the estuary has increased significantly from historic to current conditions because of the formation of mid-channel islands, primarily from dredge spoil disposal. The largest breeding colony of Caspian terns in North America is currently located in the Columbia River estuary, a location where terns historically did not breed. Terns are a conservation concern because very few breeding colonies exist; thus, terns are susceptible to catastrophic events, disease, or other factors that may affect terns during the breeding season.
- The Washington and Oregon bald eagle populations were listed as endangered under the ESA in 1978. In 1995, the USFWS reclassified the listing to threatened. In 1999, the USFWS proposed to delist the bald eagle throughout its range, however, this delisting has not been finalized. Bald eagle populations in the Columbia River estuary and lower mainstem have suffered from low reproductive success because of contaminants in the ecosystem that caused eggshell thinning. Despite this, the population has been slowly increasing, presumably because of adult recruitment from adjacent populations. Bald eagles are strongly associated with large trees during nesting, perching, and roosting; thus, the loss of mature forest habitats in the Columbia River estuary and lower mainstem has likely decreased potential eagle territories.
- The osprey population along the lower Columbia River mainstem has increased slightly in recent years. Although forest habitats used for nesting have likely decreased, osprey have adapted to nesting on man-made structures. Contaminant levels in osprey tissue are high enough to result in decreased egg thickness, but the increasing population in recent years suggests that young production is not a limiting factor.
- The lower Columbia River mainstem and estuary is not a historic breeding or overwintering area for sandhill cranes. Sandhill cranes currently do not breed in the area, but agricultural development throughout the lower Columbia River floodplain has attracted overwintering sandhill cranes. All cranes observed wintering at Ridgefield NWR and Sauvie Island Wildlife Area, Oregon, in late November 2001 and February 2002 were Canadian sandhills, and based on observations of marked birds, wintering cranes regularly move back and forth between these areas (Ivey et al. in prep.). Though not known to be a historical wintering area, an average of a few hundred, but up to 1,000, cranes have wintered in the area during the last 7 or 8 years (J. Engler, personal communication). Reclamation of agricultural land for habitat restoration projects may discourage overwintering by sandhill cranes, although future development of herbaceous wetlands may provide adequate winter habitat for sandhill cranes currently using the region.
- Within Washington, yellow warblers are apparently secure and are not of conservation concern; likewise, the red-eyed vireo is common, more widespread in northeastern and southeastern Washington, and not a conservation concern. The yellow warbler and red-eyed vireo are both riparian obligate species; warblers prefer shrub-dominated habitats

and vireos prefer dense, closed canopy forests. Habitat alterations along the lower Columbia River corridor have likely been more damaging to the possible presence of red-eyed vireos as opposed to yellow warblers because dense riparian forests along the lower Columbia River are likely less abundant than shrub-dominated wetland habitat. However, there are no data to compared historic and current breeding populations in the Columbia River estuary and lower mainstem.

- The only non-salmonid focal species population currently experiencing a decreasing trend is that of Pacific lamprey. However, Columbia River estuary and lower mainstem altered habitat conditions are not expected to be the primary factor in declining Pacific lamprey populations. The principle problems affecting Pacific lamprey populations include passage at mainstem dams (both upstream blockage of adults and high mortality of juveniles migrating downstream) and poor tributary habitat conditions, particularly low flow, degraded riparian conditions, and high water temperature (Close 2000).

3.6 Ocean Conditions

Just 7 years after record low returns that many feared were the last gasps of endangered salmon and steelhead populations, record high numbers of salmon and steelhead were counted at Bonneville Dam.[4] Although dominated by hatchery fish, the 868,000 chinook, 260,000 coho, 115,000 sockeye, and 630,000 steelhead counted at Bonneville Dam in 2001 represent 5- to 25-fold increases from recent low counts of 189,000 chinook, 10,000 coho, 9,000 sockeye, and 162,000 steelhead.

Have fears of salmon extinction been overblown? Are the increases in response to two decades of costly protection and restoration? Have salmon recovered and is ESA listing no longer warranted? At least partial answers to these questions can be found by examining ocean productivity patterns and their effects on salmon survival.

Biologists have only recently come to understand the importance of the ocean in the variation of salmon and steelhead numbers. Salmon management traditionally assumed relatively constant—or at least randomly variable—ocean conditions. After all, how could a water body so vast change from year to year? Anadromy was a tremendously successful life history pattern that traded high mortality over the long migration from freshwater to salt and back, against the large size and fecundity that could be gained in productive ocean pastures.

However, large fluctuations in smolt-to-adult survival over the last three decades have demonstrated that ocean conditions are much more dynamic than previously thought. We now understand that the ocean is subject to annual and longer-term climate cycles just as the land is subject to periodic droughts and floods. Land and ocean weather patterns are related and their combination drives natural variation in salmon survival and productivity as those seen in recent years (Hartman et al. 2000).

3.6.1 Ocean Climate Patterns

Fluctuating ocean conditions and regional weather follow large-scale atmospheric pressure gradients and circulation patterns. The El Niño weather pattern produces warm ocean temperatures and warm, dry conditions throughout the Pacific Northwest. The La Niña weather

[4] 403,000 in 1994 and 411,000 in 1995; 1.9 million in 2001 and 1.4 million in 2002.

patterns is typified by cool ocean temperatures and cool/wet weather patterns on land. Of the several indices that describe ocean conditions, the most widely known is the ENSO. It is based on sea surface temperatures in the Pacific Ocean off the coast of South America. The PDO is a similar index based on conditions in the north Pacific. The PDO often, but not always, tracks with the ENSO. ENSO episodes can have substantial short-term impacts on salmonid production, while the PDO has long term (decadal length) effects (Hare et al 1999).

Annual weather patterns tend to occur in successive years rather than randomly. Thus, warm dry years tend to occur in close association with a higher than average frequency and cool, wet years also tend to co-occur. Periods of warm, dry or cool, wet conditions are called regimes; transition periods are called regime shifts. Recent history is dominated by a high frequency of warm dry years, along with some of the largest El Niños on record—particularly in 1982-83 and 1997-98, as illustrated by Figure 3-100. In contrast, the 1960s and early 1970s were dominated by a cool, wet regime. A close examination of the historical record reveals a long, irregular series of periodic regime shifts in ocean conditions. Many climatologists suspect that the conditions observed since 1998 may herald a return to the cool wet regime that prevailed during the 1960s and early 1970s.

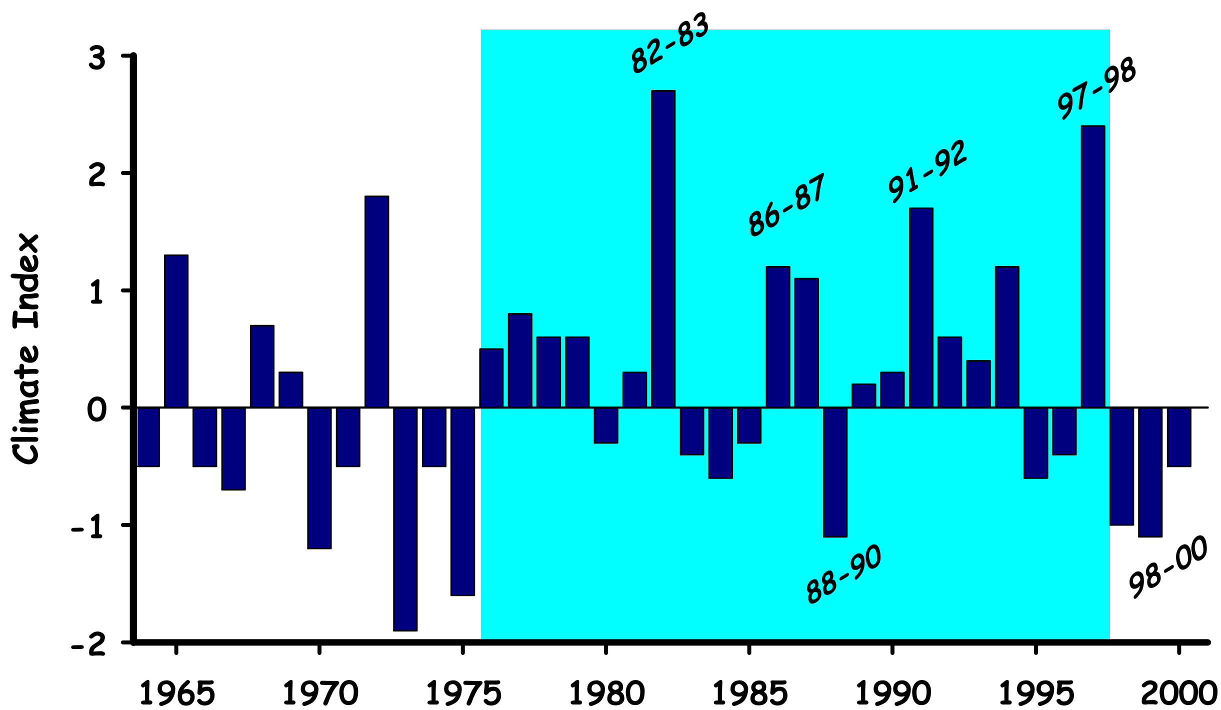

Figure 3-100. Annual variation in the multivariate El Niño southern oscillation (ENSO) index for December. Recent strong El Niño (positive values) and La Niña (negative values) years are labeled.

3.6.2 Climate and Ocean Productivity

Significant changes in oceanographic conditions are associated with El Niño/La Niña patterns. During El Niño, deep, warm, nutrient-poor layers of water push northward along the Oregon and Washington coasts. These layers block upwelling of cool nutrient-laden subsurface waters, which in turn reduces primary productivity by phytoplankton and secondary productivity by zooplankton. Juvenile salmon reaching the ocean find limited food resources and this reduces their growth and survival. Unproductive El Niño conditions also affect bird and pinniped survival and productivity. For instance, Welch et al. (1997) noted widespread mortality of northern fulmars (an offshore seabird) from Oregon to Vancouver Island with substantial numbers of starving birds washing ashore in the winters of 1994 and 1995. In addition, warm waters bring large numbers of predaceous mackerel, tuna, and even marlin into Northwest waters to further reduce salmon survival prospects. In contrast, La Niña conditions are associated with strong upwelling of cool nutrient-rich water, high productivity along the Oregon and Washington coasts, and good growth and survival of Northwest salmon stocks.

El Niño produces the opposite effect on productivity in the North Pacific off Canada and Alaska. Northern salmon stocks in Alaska generally appear to benefit from improved ocean productivity and increased smolt-to-adult survival rates during warm, dry periods (Downton and Miller 1998, Hare et al. 1999). Physical and biological domains in the North Pacific are divided by a transition zone called the Subarctic Front (Figure 3-101). Shifts in the location and structure of this front associated with ocean climate patterns drive differences in salmonid predator abundance and food resources in the North and Far North Pacific (NMFS 1996, Pearcy 1992). High atmospheric pressure along the Pacific Northwest coast during El Niño years is associated with low pressure off the Aleutian Island chain that increases upwelling in the Gulf of Alaska and provides very productive conditions for Alaska salmon. Pacific Northwest and Alaska salmon survival is thus inversely correlated: when ocean conditions are good in the Pacific Northwest, they tend to be poor in Alaska. When Alaska salmon returns are high, Pacific Northwest salmon returns are typically low.

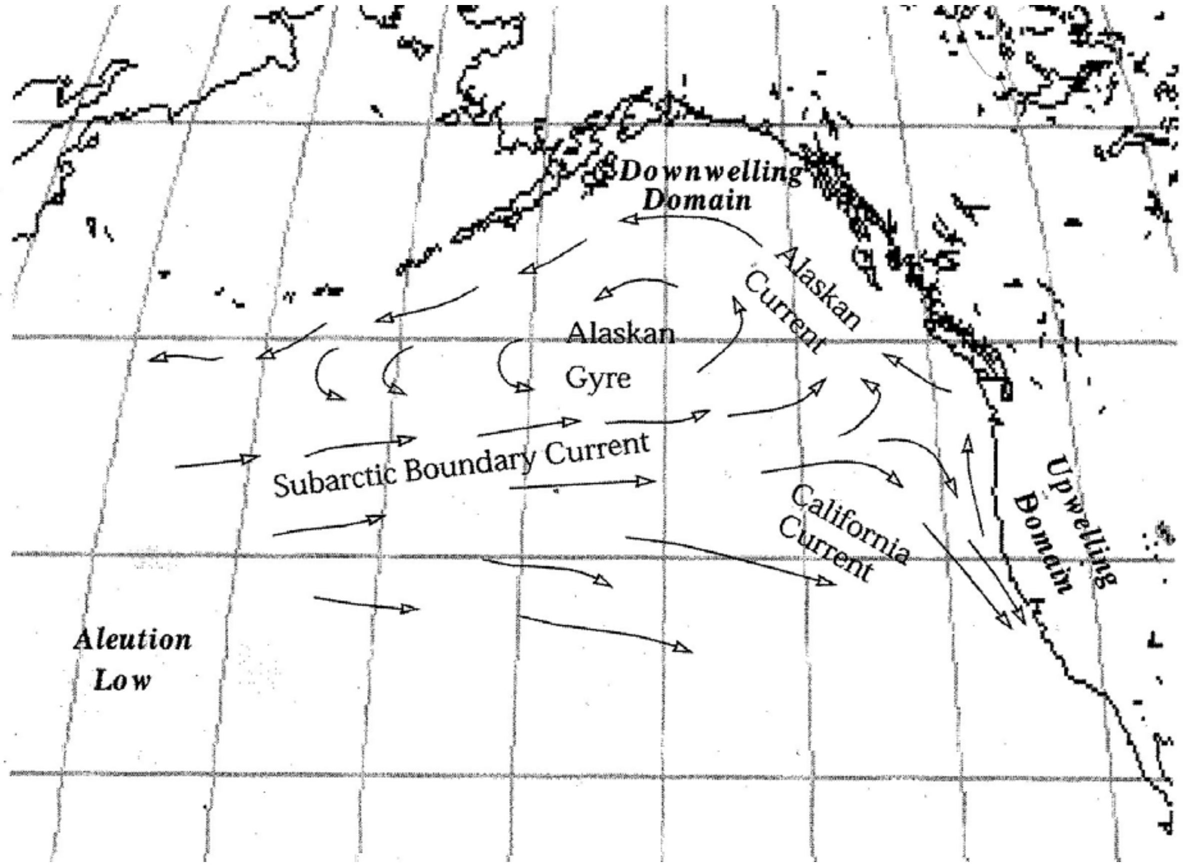

Figure 3-101. North Pacific currents and production domains. Years with an intense winter Aleutian low shift (warm dry in the Pacific Northwest) the subarctic current northward, strengthen the Alaskan current and increase the downwelling domain production. Years with a weak Aleutian low (cool wet in the Pacific Northwest) shift the subarctic current southward and strengthen the California current and the upwelling domain production (Anderson 2000).

Climate effects on ocean productivity can be compounded by parallel effects in freshwater. In the Pacific Northwest, cool, wet patterns that improve ocean survival and growth also increase precipitation, increase streamflow, and reduce temperature. Increased stream flow and cooler temperatures increase stream habitat quantity and quality for rearing salmonids. These changes also improve migration survival conditions for both juveniles and adults. Conversely, salmon productivity is reduced by low flows and warm temperatures during drought years that are often associated with El Niño. El Niños thus produce compound impacts by reducing both freshwater and saltwater survival conditions.

The PDO is a decadal or longer pattern of climate and oceanic conditions in the North Pacific Ocean associated with the Aleutian low pressure system. The PDO causes shifts in sea surface temperatures and plankton abundance on a decadal time scale (WDFW and PNPTT 2000, Mantua et al. 1997). The most recent shift occurred in 1977 (Ebbesmeyer et al. 1991), resulting in warmer coastal sea surface temperatures, cooler temperatures in the central Pacific Ocean, and more abundant plankton. While ocean conditions are affected by the PDO, the phenomenon also influences freshwater environments as well, as precipitation and temperature patterns on land are also affected by the PDO. The PDO regimes have been related to abundance patterns in zooplankton, and subsequent production of Alaskan pink and sockeye salmon (Hare and Francis 1977). The most recent PDO shift has been related to increases in production of pink, chum, and sockeye salmon in the North Pacific Ocean (Beamish and Bouillon 1993). It is possible that PDO effects on salmonid production can be more important than the shorter term ENSO-driven variation.

3.6.3 Effects on Fish Abundance and Survival Patterns

The regime shift to predominantly warm dry conditions from 1975 to 1998 produced widespread effects on salmon and other ocean fishes throughout the North Pacific (Beamish and Bouillon 1993, McKinnell et al. 2001, Pyper et al. 2001). Abrupt declines in salmon populations coincided with the regime change throughout the Pacific Northwest (Hare et al 1999).

Although trends in ocean conditions are a major driving force in the survival and abundance patterns of Pacific salmon and steelhead, the degree of effect varies among species and populations within species. Migration patterns in the ocean may differ dramatically and expose different stocks to different conditions in different parts of the ocean. Some species have broad, offshore migration patterns that may extend as far as the Gulf of Alaska (steelhead, chum, some chinook). Others have migration patterns along the Washington, British Columbia, Oregon and California coasts (chinook, coho, cutthroat). Thus, ocean conditions do not have coincident effects on survival across species or populations.

Oregon and Washington coho stocks are particularly sensitive to El Niño effects because of their local ocean distribution pattern. Coronado and Hilborn (1998) estimated ocean survival rates for CWT marked coho from Pacific Northwest hatcheries during 1971–1990. Trends changed in 1983 toward decreasing survival south of mid-British Columbia and increasing survival north of mid-British Columbia. They noted similar survival trends between hatchery, net pen, and wild coho and concluded that; “the dominant factor affecting coho salmon survival since the 1970s is ocean conditions.” Tschaplinski (2000) found that marine survival of coho smolts from Carnation Creek, British Columbia, varied up to 6-fold between years (0.05 to 0.30). Holtby et al. (1990) found that variation in survival was significantly correlated to early ocean growth rates and sea-surface salinities related to upwelling of nutrient-rich water.

Widespread changes in ocean conditions have had similar dramatic effects on ocean survival of steelhead (Table 3-45 and Figure 3-102). Cooper and Johnson (1992) showed that variation in steelhead run sizes and smolt-to-adult survival was highly correlated between runs up and down the West Coast. Smolt-to-adult survival rates generally varied 10-fold between good and bad years. Ocean survival rates for three West Coast steelhead populations where good annual index data were available showed high variability and a generally declining trend since the late 1970s (Table 3-45 and Figure 3-102).

Similar survival patterns have been documented for other Pacific salmon species including sockeye (Farley and Murphy 1997, Kruse 1998, Peterman et al. 1998, McKinnell et al. 2001) and pink salmon (Pyper et al. 2001).

Warm dry regimes result in generally lower survival rates and abundance, and they also increase variability in survival and wide swings in salmon abundance. Some of the largest Columbia River fish runs in recorded history occurred during 1985–1987 and 2001–2002 after strong El Niño conditions in 1982–83 and 1997–98 were followed by several years of cool wet conditions.

Table 3-45. Annual means of smolt-to-adult survival rates for several runs of steelhead where smolt and adult numbers are monitored*.

Smolt Year	Eagle Creek Hatchery	Kalama Hatchery Winter	Kalama Hatchery Summer	Snow Creek Wild	Keogh River Wild
1974			3.1%		
1975		0.5%	6.0%		
1976		2.4%	5.4%		
1977		0.5%	4.0%		15.2%
1978		1.3%	18.1%	6.5%	7.4%
1979		1.4%	16.0%	10.7%	15.2%
1980		0.8%	9.6%	5.6%	8.4%
1981		0.6%	2.8%	2.2%	25.4%
1982		1.7%	4.9%	6.1%	26.1%
1983	2.4%	1.5%	8.0%	10.5%	15.5%
1984	2.3%	3.0%	12.3%	4.8%	18.3%
1985	1.2%	1.2%	8.0%	3.5%	25.3%
1986	0.8%	1.6%	6.2%	7.1%	9.8%
1987	0.6%	2.0%	7.8%	1.3%	13.2%
1988	1.2%	1.3%	6.1%	1.7%	6.7%
1989	0.9%	1.8%	4.9%	1.6%	15.2%
1990	1.7%	2.4%	13.7%	3.0%	6.2%
1991	1.0%	1.2%	6.2%	2.1%	3.6%
1992	0.7%	0.4%	3.6%	1.6%	2.3%
1993	0.7%	0.5%	1.6%	2.8%	4.6%
1994	0.2%	2.0%		6.6%	2.5%

* Data from C. Sharpe, WDFW, Kalama; D. Dysart, USFWS, Eagle Creek National Fish Hatchery; D. Rawding, WDFW, White Salmon; and B. Ward, University of British Columbia, Vancouver, BC.

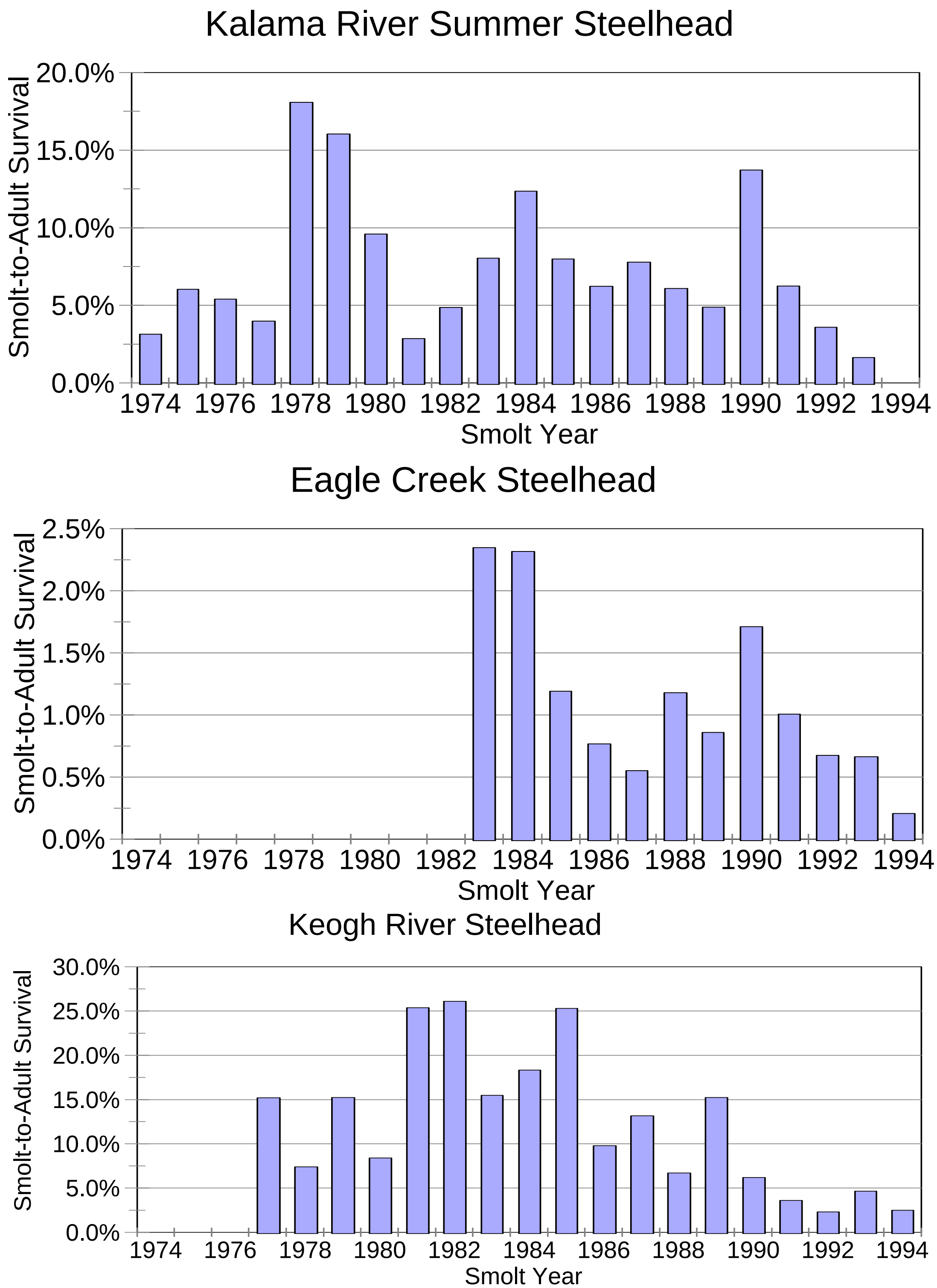

Figure 3-102. Annual means of smolt-to-adult survival rate of winter and summer steelhead from Kalama River Hatchery, winter steelhead from Eagle Creek National Fish Hatchery, and wild winter steelhead from the Keogh River, British Columbia.

3.6.4 Implications for Salmon Protection and Recovery

The reduced productivity that accompanied an extended series of warm dry conditions after 1975 has, together with numerous anthropogenic impacts, brought many weak Pacific Northwest salmon stocks to the brink of extinction and precipitated widespread ESA listings. Salmon numbers naturally ebb and flow as ocean conditions vary. Healthy salmon populations are productive enough to withstand these natural fluctuations. Weak salmon populations may be severely stressed during periods of poor ocean survival. Weak populations may disappear or lose the genetic diversity needed to withstand the next cycle of low ocean productivity (Lawson 1993).

Looked at over decades, ocean productivity patterns confound our ability to recognize and measure risk factors and the benefits of protection and restoration actions implemented to date. For instance, a favorable climate regime counteracted the detrimental impacts of Columbia River basin hydrosystem development after 1945, while an unfavorable climate regime negated the beneficial effects of salmon mitigation efforts after 1977 (Anderson 2000). Similarly, productive ocean conditions during the 1960s and early 1970s masked declines in wild fish numbers and inflated expectations for increasing hatchery coho production.

Fluctuations in fish run size and studies of ocean conditions over the last 20 years have greatly increased our understanding of the influence of inter-decadal climate patterns on salmon population dynamics, but do not fundamentally alter recent assessments of status and extinction risks. Extinction is most likely during extended periods of poor ocean conditions like those coincident with the ESA listing of many West Coast salmon and steelhead during the 1990s. Large salmon returns in the last few years are a temporary response to improved ocean conditions following the 1997–98 El Niño; they are not likely to represent the average future condition.

Recent improvements in ocean survival may portend a regime shift to generally more favorable conditions for salmon. The large spike in recent runs and a cool, wet climate would provide a respite for many salmon populations driven to critical low levels by recent conditions. The respite provides us with the opportunity to continue protection and restoration to forestall extinction when the ocean again turns sour—as it inevitably will. The risk is that temporary increases in survival and abundance may erode the sense of urgency for salmon recovery efforts.

The Natural Research Council (1996) concluded:

> *Any favorable changes in ocean conditions—which could occur and could increase the productivity of some salmon populations for a time—should be regarded as opportunities for improving management techniques. They should not be regarded as reasons to abandon or reduce rehabilitation efforts, because conditions will change again.*

Volume I, Chapter 4
Conceptual Framework and Recovery Standards

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4.0 Conceptual Framework and Recovery Standards

Chapter 4 describes the conceptual framework for analyzing recovery planning questions, restoring healthy salmon populations, and defining specific recovery goals and objectives. This framework provides a context for development of a recovery plan based on information in this technical foundation. The following sections identify key elements that must be addressed in an effective recovery planning process.

4.1 Conceptual Framework for Recovery Planning Analysis

Developing an effective recovery plan will require systematic analysis of questions related to goals, status, strategies, and proposed actions based on the best available scientific methods and data. Analytical approaches that systematically relate fish status to underlying causal factors and actions can be extremely powerful tools for evaluating recovery goals and actions. Systematic analyses based on the scientific method facilitate the study, description, and prediction for complex systems and promote good decision-making (Grant 1986).

Various analytical approaches and tools are available for evaluating fish recovery planning questions and complex habitat, harvest, hydro, and hatchery relationships. The acronyms for several of these tools are in wide usage (e.g. EDT, VSP, PVA, PCC) but the thicket of technical details pertaining to their capabilities, information bases, and weaknesses can be difficult to penetrate. The selection of an analytical approach is further complicated by the realities that no single tool comprehensively addresses recovery planning, many relationships are not fully understood, and the data needed to drive the tools are inconsistent across the planning area.

4.1.1 Planning Questions

Effective fish recovery planning depends on our ability to answer the five fundamental questions: 1) where are we now; 2) how did we get here; 3) where do we want to go; 4) how do we get to where we want to go; and 5) how do we know when we get there? *Where we are now* clarifies key subjects of interest and their current state. *How did we get here* helps us to know what needs fixing. *Where do we want to go* articulates our goals. *How we get there* identifies changes needed to close the gap between current status and our goals. *How we know when we're there* recognizes the continuing need for monitoring and adaptation in the implementation of any plan.

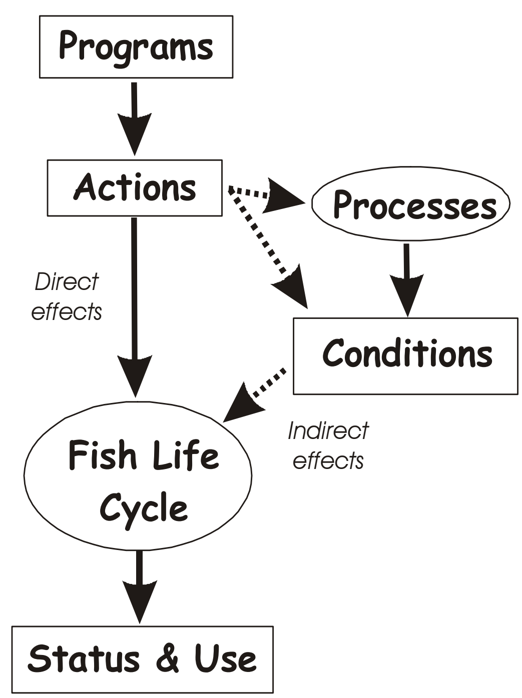

Figure 4-1. Direct & indirect effects.

While general planning questions can be simply stated, answers can be difficult and complicated. Fish are affected by a complex array of factors and our understanding of the relationships among these factors is highly indistinct. Efforts are complicated further by the need to consider multiple species, a large and diverse area, and a patchwork of overlapping jurisdictions and constituencies.

Fundamental questions also need to be answered at several different levels in terms of fish populations and ESU status, fish life cycle parameters such as mortality rates, factors for decline, and programs by which actions may be affected. The planning process must describe how harvest, hatchery, hydropower, and habitat factors have influenced key fish species in the past and their current impact. In addition, the planning process must project the current trajectory of these influences. Specific questions also exist within each area of impact. For instance, with reference to habitat, it is important to understand the relationships between land use practices, watershed processes, stream habitat conditions, and their particular effects on the life stages of each species. The recovery planning process must weigh human-induced effects on mortality at different life stages throughout the life cycle, identify how mortality can be reduced overall, and determine how the allocation may be changed to meet delisting and other social goals.

4.1.2 Viability & Use Recovery Goals

Recovery planning analyses must address both population viability goals related to Endangered Species Act (ESA) objectives and requirements, and broad goals related to a desire to support opportunities for other fish uses such as fishing (Figure 4-2). Population viability goals generally represent minimum standards for fish restoration. These standards signify a level where unique groups of populations are no longer in danger of extinction or threatened with declines to a level where extinction looms. Broad sense goals generally correspond to higher levels of fish restoration that maintain population viability while providing additional fish for other uses.

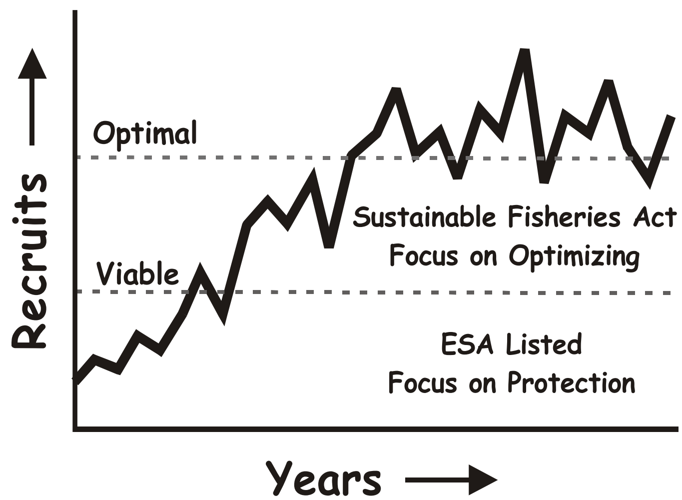

Figure 4-2. Viability and use opportunities.

Population viability guidelines for fish abundance, productivity, diversity, and spatial structure have been identified by NOAA Fisheries based on Viable Salmonid Population (VSP) concepts (McElhany et al. 2000). The VSP concept provides a general framework for analysis of population viability but does not prescribe specific measures or criteria. Specific recovery criteria corresponding to VSP guidelines have been identified by a Technical Recovery Team established by NOAA Fisheries for listed Lower Columbia Basin salmon in Oregon and Washington. Recovery goals and analyses in the LCFRB planning process must address VSP guidelines and corresponding Willamette/Lower Columbia Technical Recovery Team (TRT) criteria for consistency with federal Endangered Species Act (ESA) requirements. The LCFRB process for identifying goals is described in the Recovery Standards section below and the detailed methods are described in Chapter 5.

The basis for identifying broad recovery goals is much less defined than the basis for population viability goals. Many recovery planning efforts that focus solely on minimum persistence standards do not address use goals other than to limit incidental fishing impacts to achieve desired population levels. Another approach has been to assume achievement of broad goals where population viability goals are met. Systematic analyses can help clarify the relationships between viability and broader recovery goals, and define bounds for realistic restoration levels.

4.1.3 All Fish Limiting Factors Addressed

Recovery planning analyses must equitably address all factors that limit fish status and have contributed to fish declines. These include the spectrum of human-induced mortality factors that affect the fish throughout their life cycles. These factors are sometimes referred to as the four Hs (hatcheries, harvest, hydropower, and habitat). Reference to this convenient characterization highlights the need to treat all factors limiting recovery in a similar and comprehensive fashion, although the 4-H label oversimplifies the complex of direct and indirect relationships and the relative impacts of the different factors that affect fish.

A comprehensive treatment of fish factors limiting recovery also warrants careful consideration of other influences that are beyond our control. For instance, the recent fluctuation in fish survival highlights the critical importance of environmental conditions including ocean and climate cycles. Variation in ocean productivity and survival related to recent El Niño/La Niña patterns has produced dramatic swings in salmon returns during the last 10 years from some of the lowest to highest escapements on record. The effects of human-caused mortality and restoration measures must be considered in the context of these highly variable survival rates. For instance, it would be inappropriate to assume that fish were headed for extinction based solely on abundance and productivity trends during a periodic down-cycle in ocean conditions. Similarly, a conclusion that recovery had been achieved following bumper returns in good ocean years would be equally fallacious. Recovery planning analyses must consider variable ocean conditions as an indifferent backdrop to the effects of human activities on salmon. Periodic downturns challenge the persistence and health of impaired salmon populations and can precipitate irreversible consequences where fish have been heavily impacted by human-induced factors.

A comprehensive analysis of all human and environmental factors limiting recovery is needed to ensure the effectiveness of related salmon recovery efforts. For instance, it would do little good to implement aggressive land use limitations to improve tributary stream habitats if benefits are offset by an excessive harvest response. Conversely, reductions in fisheries to improve spawning escapement will not restore fish populations if productivity is continually eroded by declines in freshwater survival.

Finally, the comprehensive analysis of all factors limiting recovery helps ensure equitability in balancing the costs of salmon recovery among different stakeholders. Without a systematic analytical approach for assessing impacts, discussions of site and action-specific recovery actions are easily confounded by counterproductive finger-pointing.

4.1.4 Life Cycle Focus

A fish life cycle focus provides a systematic means of effectively relating fish-specific recovery goals to factors limiting recovery and potential restoration actions (Figure 4-3). A life cycle focus identifies life stage-specific numbers, birth rates, and death rates that describe the biological processes regulating fish status. Stage-specific numbers and rates provide a consistent way to estimate fish effects from the impacts of a variety of stage-, time-, and area-specific factors that limit recovery. The life cycle approach also provides the means of distinguishing wild and hatchery fish and explicitly evaluating the effects of their interactions.

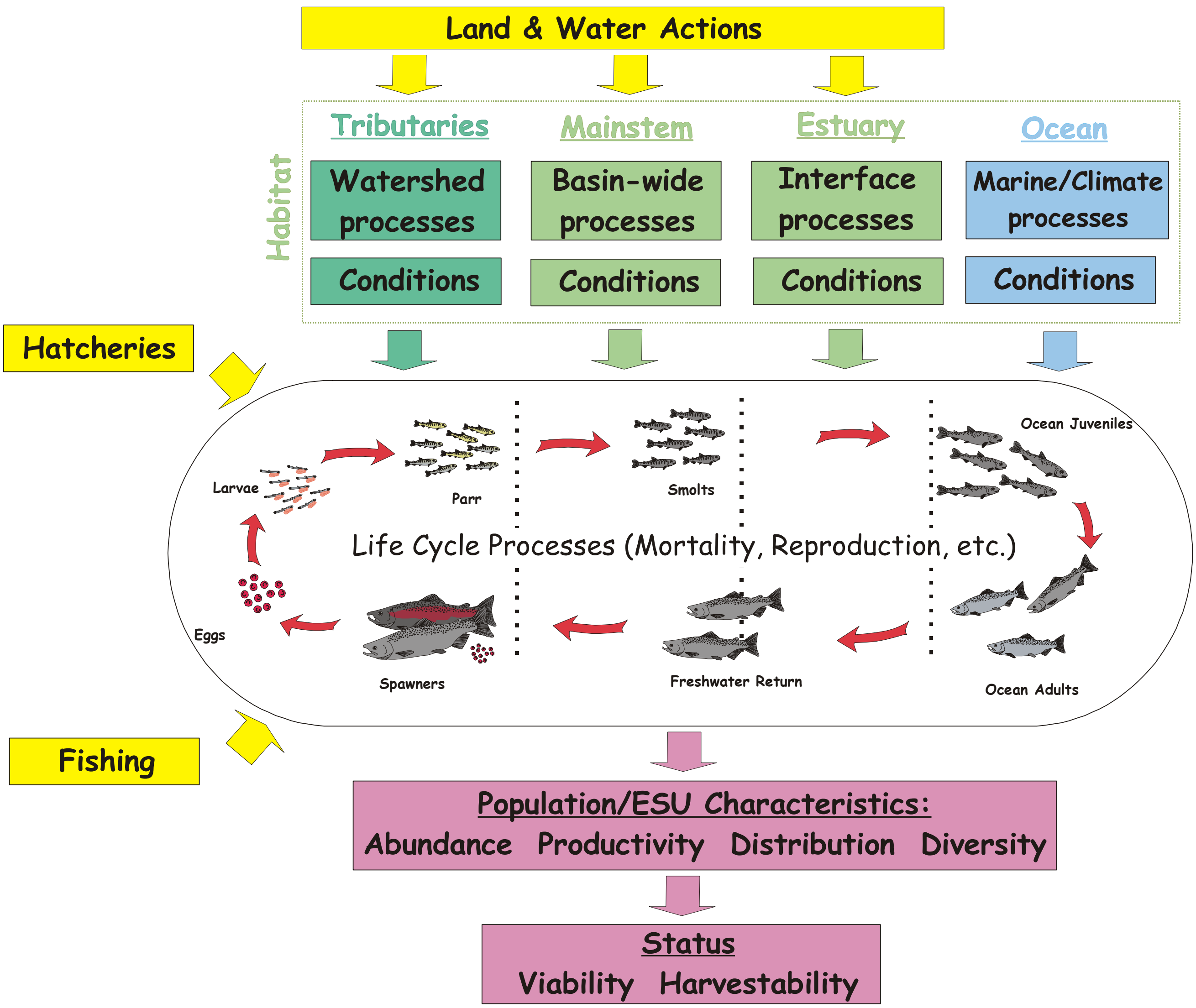

Figure 4-3. Conceptual depiction of the relationships between fish and factors limiting recovery.

4.1.5 Action-Specific

Recovery planning analyses must relate fish goals and status to specific actions, areas, and time periods. Analyses can identify the relative contributions of habitat, hatchery, and harvest impacts but should also relate necessary changes to specific activities that can produce the desired effect. Specific programs and activities need to be identified because that is the level at which changes will be implemented. Analyses that are not specific will fail to provide a clear blueprint for recovery implementation and risk failure to ensure accountability.

Specificity in time and space are also important. Viability risks are extremely sensitive to implementation schedules, especially where small population numbers increase exposure to chance extinction events. Thus, fishing strategies that reduce impacts in low run years but increase catch in large run years might substantially reduce the risks of a fixed fishing rate strategy while optimizing use benefits. Similar suites of measures can also produce substantially different outcomes if implemented in different areas. For instance, concentrating aggressive habitat restoration actions in high quality habitats where fish production is already significant may provide relatively little benefit. These areas might be high priorities for protection but low priorities for restoration. Within marginal areas, systematic analyses can help distinguish smaller subareas for priority restoration where modest investments can restore significant fish production, from severely degraded sites where similar investments would be relatively ineffective.

4.1.6 Flexibility

Analyses with the flexibility to incorporate a variety of information, approaches, and tools will provide the most robust assessments. All analyses necessarily rely on abstractions of the immeasurable complexity of ecosystems. Each approach emphasizes selected components of the system in an attempt to capture key factors and relationships of interest. Every alternative has its strengths and weaknesses and no single approach is ideal. Data constraints and limitations in our understanding of key processes and relationships also result in an uneven understanding of different components of the system that make it difficult to conduct analyses of all questions with a uniform degree of confidence. Flexibility, incorporating a variety of approaches including different alternatives for some questions, will maximize the information base for difficult recovery planning considerations. Similar results from different analytical approaches to the same problem can provide strong corroboration for conclusions and substantially reduce uncertainty. Contrasting results can highlight critical uncertainties that should be addressed with safety factors, contingencies, and future investigations.

Analytical alternatives vary in the breadth of the factors and relationships included. By design, different tools may reflect smaller or larger pieces of the puzzle. For instance, a stream temperature model that estimates reach-specific temperatures based on aspect, riparian conditions, and flows describes very specific results on a very narrow portion of the system of habitat effects on salmon. In contrast, the EDT approach that relates salmon population characteristics to several dozen riparian and stream habitat conditions incorporates a relatively broad portion of the system. Data demands often limit the breadth of analysis by any given tool. Attempts to develop analytical approaches across very broad swaths of a system are almost invariably constrained by information availability. The broader the analysis, the more likely it is to be confounded by unknown interactions in relationships.

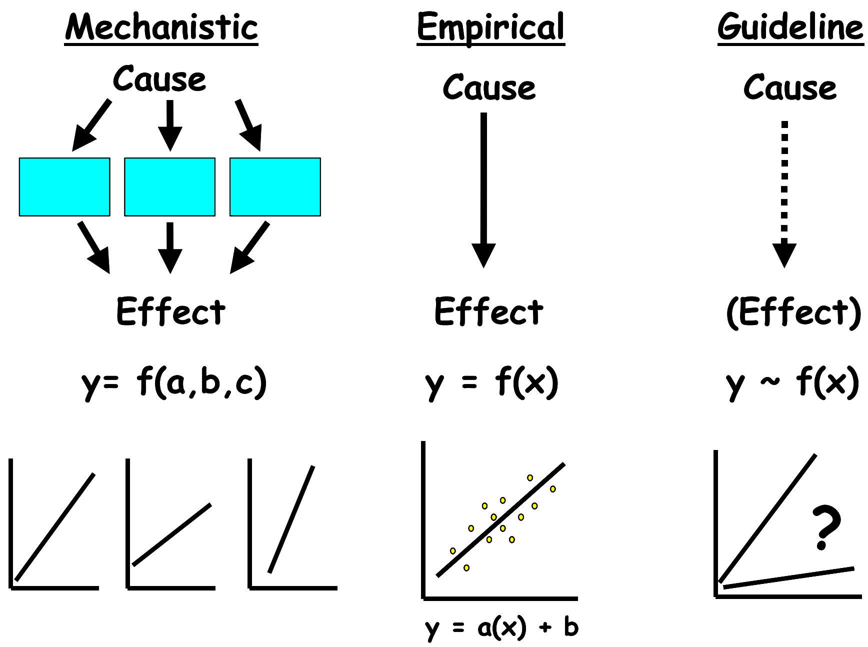

Figure 4-4. Different approaches to analytical specificity.

Analytical alternatives also vary in the specificity or detail with which different components of the system are described (Figure 4-4). *Mechanistic analyses* that explicitly detail explanations of all the links between cause and effect are one extreme. For instance, a mechanistic analysis of how road construction affects stream sedimentation might include specific functional relationships, and describe erosion, runoff, and landslides. Mechanistic approaches explicitly relate effects to specific actions and can provide a high degree of confidence in predictions where pertinent relationships are clearly understood. However, mechanistic approaches require large amounts of data, produce misleading results where key elements are overlooked, and, by virtue of their complexity, can reduce transparency in underlying assumptions.

Empirical analyses represent an intermediate level of specificity. Empirical analyses (also called predictive or correlative) relate to cause without an explicit explanation of the detailed mechanisms that tell us why the effect occurs. For instance, an empirical analysis of road effects on sediment might be described based on stream sediment measurements in a series of subbasins with different road densities. Empirical approaches can provide very high predictive capability with a limited amount of data and may be intuitively transparent because they can be fundamentally simple. However, a faulty understanding of underlying mechanisms can lead to spurious (chance) correlations with no predictive power. Note that complex mechanistic analyses are often based on a series of empirical relationships nested in a hierarchical relationship to even more fine-scale underlying processes.

Guidelines represent a relatively low level of specificity. Guidelines provide general recommendations based on assumed cause and effect relationships but do not explicitly estimate corresponding effects. For instance, a guideline might take the form of a direction to build fewer roads in critical salmon areas. Numerical guidelines can also provide benchmarks by which effects or progress might be qualified (e.g., salmon production is not substantially affected when road density is less than x miles/acre). Guidelines can provide relatively robust general direction but do a poor job of representing specific actions, benefits, and tradeoffs.

Finally, analytical alternatives also vary widely in the precision by which specific elements are quantified. Some alternatives—such as guidelines—may be entirely non-quantitative (e.g. salmon production can be greatest in wilderness areas where road disturbances are minimal). Other alternatives explicitly quantify characteristics and relationships (e.g. every x% change in road density produces a y% reduction in salmon productivity). Among quantitative alternatives, a deterministic approach produces one result for one set of inputs and contains no random variables. In contrast, a quantitative stochastic approach includes random variables and produces many results for one set of inputs. Deterministic approaches effectively represent average conditions such as the average benefit of stream improvement on smolt production. Stochastic approaches more effectively represent the chances of rare events such as fish extinction probabilities in response to variable ocean survival.

Available analytical tools represent the spectrum of breadth, specificity, and precision. In the world of perfect recovery planning, we would aspire to use a very detailed and data-rich mechanistic description of our system to maximize the specificity with which we could explore actions. However, data limitations often require us to work with less specific and less quantitative approaches to provide defensible results. The Recovery Standards section below and Chapter 5 describe the approaches selected, with the foremost analytical considerations in mind, for establishing recovery goals, and identifying remedial actions required.

4.1.7 Measuring and Managing Uncertainty

Expectations for analysis must be tempered by our imperfect understanding of the complex interaction of fish, limiting factors, and human activities, and the incomplete nature of the available data. All models and analytical approaches are abstractions of reality subject with varying degrees of uncertainty. Systematic scientific analyses will reduce but not eliminate uncertainty. Clear paths for action will be provided by some analyses where relationships are well understood and data are substantial. Analyses in the gray areas may provide only partial answers and general compass directions. A gap will remain between what we know and what we wish we knew. The analyses will provide insight for decisions, as well as the most important data gaps and/or weaknesses, but the conundrum of decisions without full information will continue. Thus, science provides a firm footing for recovery planning but will not supplant the need to make difficult policy decisions with less than complete information.

Complex analyses are almost universally based on some type of summary or derived value. Uncertainty associated with these derived values can be described in terms of measurement error and process error. Measurement error has to do with whether field observations really represent actual conditions or missed the mark because of poor measurement techniques. For instance, 337 fish might actually be 542, if rainy weather makes them difficult to see. This measurement uncertainty is typically expressed in terms of statistical confidence intervals. Process error has to do with whether the underlying assumptions that account for natural variation are appropriately accounted for in the experimental design or statistical model. For instance, a 542-fish population estimate based on daily fish ladder counts could be biased severely if each daily estimate was variously and unaccountably influenced by flow through the fish ladder. Process uncertainty can be particularly difficult to quantify and becomes increasingly important as analytical models become more complicated.

These inherent uncertainties complicate the process of deriving, deducing, inferring, or interpolating estimates needed to characterize and evaluate fish status and limiting factors. To a large degree, detailed descriptions of data quality and uncertainty will be a product of analyses. Significant analysis may be required to quantify the degree of uncertainty and its significance to other inferences. Analyses thus ultimately provide a context for interpretation of data quality and uncertainty.

The key to effective analysis in an uncertain world is to frame an approach that recognizes that uncertainties will always remain in specific data, analyses, and assumptions. In this recovery planning process, uncertainties will be addressed by the following.

- Explicitly identifying uncertainties and transparently communicating methods, strengths, and limitations of each analysis.
- Incorporating known uncertainties into the risk-based population viability modeling framework for integrated fish life cycle analyses. . For instance, expressing ocean survival as a random variable that affects extinction risk incorporates uncertainty in ocean conditions. Uncertainty in any population process or limiting factor can be captured similarly.
- Incorporating corroborative analyses to validate key conclusions independently.
- Using analyses to identify the risks associated with key uncertainties. Sensitivities of results to critical assumptions and uncertainties will be described for each analysis in the form of testable hypotheses that may be addressed with future monitoring and evaluation through adaptive management.
- Identifying conclusions based on the weight of all evidence, rather than any specific analytical result, and with appropriate safety margins to buffer risks.

4.2 Biological Basis for Recovery Standards: Extinction, Viability, and Use

In order to affect salmon recovery, it is particularly helpful to understand why fish go extinct in the first place. Extinction results from the interaction of fish population processes and external factors to reduce population size to critical low levels that are no longer self-sustaining. Population processes regulate how salmon respond to factors for decline. Characterization of population processes can provide useful recovery standards when used in conjunction with fish population size. Recovery standards may also consider factors for decline. External factors for decline can be partitioned into human factors that may potentially be modified and natural factors that must be also be considered. This section describes extinction processes and factors in further detail.

4.2.1 Definitions

Recovery efforts can aspire to restore salmon to various degrees across a continuum of status levels (Figure 4-5). Biological reference points provide convenient terms for describing various levels. *Extinction* is the obvious low bound on population status. Extinction typically refers to the irreversible disappearance of a species or, in the case of Pacific salmon, an Evolutionarily Significant Unit (ESU), as described in Chapter One. Local extinctions of subpopulations are sometimes referred to as *extirpation.* Functional extinction typically occurs at population sizes greater than zero when numbers fall to critical low levels from which they cannot recover. This functional extinction level is often referred to as *quasi-extinction.* Because it is often unclear where this functional extinction level occurs, quasi-extinction is defined as a low abundance that does not guarantee extinction but from which recovery cannot be assured.

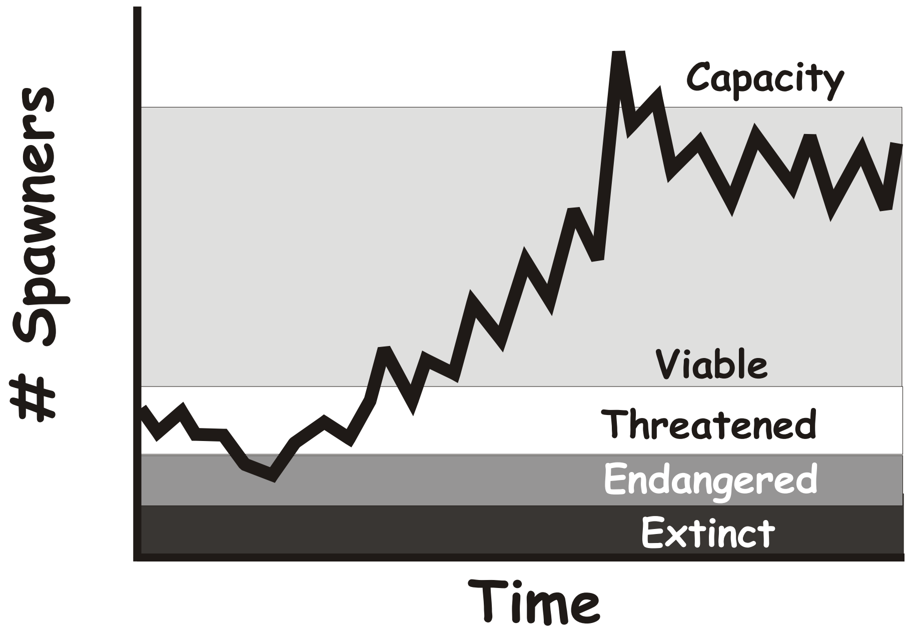

Figure 4-5. Continuum of abundance levels corresponding to potential fish recovery standards.

A species that is not at risk of extinction is typically referred to as *viable*. Viability is also equivalent to having a high likelihood of long-term persistence. The Federal Endangered Species Act (ESA) qualifies non-viability at two levels: *endangered* with extinction and *threatened* with becoming endangered with extinction.

Capacity is at the opposite end of the status spectrum from extinction. Capacity is the maximum number of individuals that available resources can support. Capacity is expressed through density-dependent population limits that reduce survival, growth, or reproduction via competition. Capacity may change as habitat quantity or quality increase. Thus, we can distinguish between realized capacity based on existing habitat conditions and potential capacity if habitat conditions were improved. Capacity and viability are not necessarily mutually exclusive. A population may not be viable where numbers are constrained by low capacity of a small area with poor quality habitat. Abundance may also be less than the hypothetical habitat capacity as a result of mortality factors.

Fish *recovery* refers to the restoration of fish status to some level at or above viability. Specific recovery goals may be defined anywhere within the range between minimum viability and the hypothetical capacity of a fully restored habitat. From an ESA standpoint, recovery refers to the abundance required for an ESU to not be threatened or endangered with extinction (i.e. the minimum level consistent with viability). However, healthy, harvestable populations require recovery to levels greater than the minimum viability standard. Furthermore, restoration of specific ecosystem functions might require recovery to even higher levels near system capacity. Population viability constraints place a hard floor on recovery goal discussions but other values will drive the selection of higher specific recovery goals intended to balance tradeoffs between increased population sizes, direct fishing impacts on the fish, and indirect effects of our activities on habitat use.

4.2.2 Variation Among Populations within Species

While many salmon populations are in danger of extinction, others are healthy and can readily sustain fisheries that provide fish for the supermarket. Healthy populations are common in Canada and Alaska in cases where habitat quality remains high and harvest is managed well. A large portion of the wild salmon available in stores and restaurants originates from Canadian and Alaskan fisheries. Farm-raised salmon from Washington, Canada, Chile, and Norway have assumed an increasing share of the market in recent years. Chinook and coho from Columbia River hatcheries and remaining healthy wild populations from Washington and Oregon continue to provide harvestable numbers, especially in years of high survival. However, healthy populations are the exception rather than the rule in the Columbia River basin where salmon have been depleted by a variety of insults. Some Columbia Basin salmon populations have already disappeared (Nehlsen et al. 1991). Others are at or near the brink of extinction (see ESA-listing information in the Species Overviews).

Even within a salmon species, not all populations are the same. Each salmon species is comprised of many related but different populations, each of which is specifically adapted to the unique local conditions of their natal watersheds. Thus, a population of wild coho salmon from the Columbia River cannot be replaced with wild coho salmon transplanted from Alaska. Differences among populations in adjacent watersheds may be small where habitat conditions are similar but differences typically increase with distance (Riddell 1993). Salmon that stray or are transplanted among widely separated watersheds do not fare as well as the native stock. Once lost, the unique features of each population may be gone forever.

Population differences within a species result from local adaptations that have been naturally selected over hundreds of generations to optimize success under the prevailing conditions. Adaptations may be expressed in a variety of forms such as run timing that returns adults to streams exactly when spawning conditions are optimal or that allows smolts to arrive at the estuary during the critical physiological window for transition from fresh to salt water. Local adaptation is made possible by the homing of salmon across thousands of miles of ocean and river to spawn in the same river or stream where they were born. Recent studies have shown that homing may be so exact that many salmon even spawn in the same river bend or riffle where they originated. Local adaptation and homing go hand in hand to give each salmon the best chance for reproductive success by returning to the exact conditions to which they are best suited. The degree of difference among populations can often, but not always, be identified by genetic analysis.

Preservation of unique groups of salmon populations is a central tenet in the development of recovery standards. Salmon populations are often organized into groups for various management purposes. Populations within a species that have similar life histories are often referred to as "races" (e.g. winter steelhead, spring chinook, early run "tule" fall chinook). Populations within races that are grouped together for harvest management purposes are referred to as "stocks". When salmon or trout species are listed as threatened or endangered under ESA, populations within a region are grouped into ESUs, which become the organizational groups to which recovery standards are applied.

4.2.3 Minimum Viable Populations

Fish go extinct when population size falls to critically low levels from which they cannot recover. However, underlying population processes including abundance, productivity, diversity, and spatial distribution are intimately related with abundance and can be the ultimate determinants of whether populations are viable or doomed. NOAA fisheries has incorporated each of these parameters into a Viable Salmonid Population Concept (McElhany et al. 2000) that provides a useful framework for analysis of population viability.

4.2.3.1 Abundance

Ideally, two determined fish of the opposite sex could forestall extinction but in practice many more are needed to ensure population persistence and provide the raw material for recovery. Small population sizes are subject to a variety of limiting factors from which it may be difficult to recover (Lande and Barrowclough 1987, Nelson and Soulé 1987, Lynch 1996). Small numbers risk genetic bottlenecks that reduce diversity. The genetic diversity of salmon populations maximizes population persistence and productivity by allowing the salmon to capitalize on a wide range of habitats and environmental conditions. Small numbers increase chances of inbreeding, possibly resulting in severe genetic side effects (e.g. expression of deleterious recessive genes). Small numbers also increase demographic risks where scattered fish are unable to find mates, sex ratios are skewed by chance, or numbers are too few to escape predators (Hilborn and Walters 1992, Courchamp et al. 1999). Small numbers may also increase risks of extinction from natural downturns in survival conditions or catastrophes (e.g., poor ocean conditions, volcanoes, floods, chemical spills, dam failures, etc.) (Lawson 1993).

Reduced productivity at low densities is often referred to as depensation (also termed “Allee effects” or “inverse density dependence”) (Figure 4-6). McElhany et al. (2000) noted that depensation is a destabilizing influence at very low abundance and can result in a spiraling slide toward extinction. This downward spiral is sometimes referred to as an “extinction vortex.” Small population sizes that can lead to this downward spiral are called “quasi-extinction.”

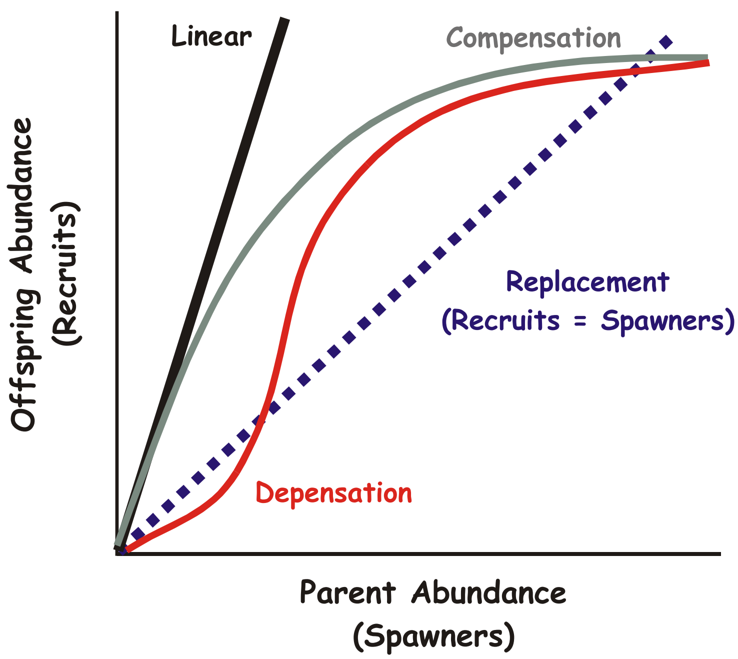

Figure 4-6. Contrast of reduced productivity at low population sizes (depensation) that results from small population processes and at high population sizes that results from competition for limited resources (compensation).

Thus, species can be predisposed to extinction by low population sizes that reduce population resilience well before extinction actually occurs. Cumulative effects of periodic poor spawning escapements may increase chances of future extinction even where numbers temporarily rebound (in good ocean years for instance) (Lawson 1993). Population sizes needed for recovery to levels that will ensure long-term population persistence and viability are established based on the size of buffer needed to avoid low critical population sizes in the face of normal environmental variation.

McElhany et al. (2000) identified key characteristics of viable and critical population abundance guidelines. Viable population size guidelines are reached when a population is large enough to: 1) survive normal environmental variation, 2) allow compensatory processes to provide resilience to perturbation, 3) maintain genetic diversity, 4) provide important ecological functions, and 5) not risk effects of uncertainty in status evaluations. Critical population size guidelines are reached if a population is low enough to be subject to risks from: 1) depensatory processes, 2) genetic effects of inbreeding depression or fixation of deleterious mutations, 3) demographic stochasticity, and 4) uncertainty in status evaluations.

Although biologists generally agree that extinction risks become increasingly acute as numbers decrease, there is little agreement on where functional extinction occurs and what population level is viable. Various viability and critical population size guidelines have been identified based on largely theoretical considerations for genetic and demographic risk. For instance, numbers needed to minimize genetic risks typically range from 30 to several thousand individuals based on theoretical models of genetic characteristics, effective spawner population sizes, and genetic diversity (Franklin 1980, Soule 1980, Allendorf and Ryman 1987, Lynch 1990, Waples 1990, Thompson 1991, Gabriel and Burger 1992, IUCN 1994, Lande 1995, NMFS 1995, Allendorf et al. 1997, McElhany et al. 2000). Thompson (1991) identified a 50/500 “rule of thumb” where 50 fish is a short term-effective population size which limits inbreeding and 500 is a long-term effective population size which maintains genetic variability. Recent viability analyses by NOAA Fisheries generally assume a 50-fish quasi-extinction threshold and produce minimum population viability levels of at least 500 fish to ensure that critically low numbers do not result from normal variation associated with environmental variation (McElhany et al. 2003). Uncertainties in actual minimum viable population sizes will require definition of recovery standards that incorporate appropriate safety factors.

4.2.3.2 Productivity

Productivity refers to a population’s ability to replace itself and reflects a population’s resiliency or the ability to rebound from a low level to the equilibrium population level. Highly productive populations produce larger numbers of juveniles or recruits per parent and can more readily rebound from low levels following perturbation. Less productive populations produce smaller numbers of offspring or recruits per parent and rebound more slowly or not at all. Extinction risks depend on the combination of abundance and productivity. For instance, risks might be much less for a highly productive population even at low spawning escapements than for a larger population where productivity is low. Productivity or population growth rate guidelines are reached when a population’s productivity is such that: 1) abundance can be maintained above the viable level, 2) viability is independent of hatchery subsidy, 3) viability is maintained even during extended sequences of poor environmental conditions, 4) declines in abundance are not sustained, 5) life history traits are not in flux, and 6) conclusions are independent of uncertainty in parameter estimates (McElhany et al. 2000).

4.2.3.3 Diversity

Diversity refers to the variability among individuals and populations in life history, behavior, and physiology. Diversity traits include some that are completely genetically based and others that vary as a result of a combination of genetic and environmental factors. Correlations between diversity and population productivity have been observed in many populations (NRC 1996). Diversity is related to population viability because it allows a species to use a wider array of environments, protects species against short-term spatial and temporal changes in the environment, and provides the raw material for surviving long-term environmental changes (McElhany et al. 2000). Diversity can be especially important in maintaining stock productivity in the face of variable environmental conditions such as those accompanying regime shifts in ocean productivity. Some population segments are better adapted to certain conditions than others and different population segments may fare better during different climate conditions. For example, a north-migrating component of a population might fare better during warm years when the ocean is productive but a south-migrating component might be favored during cold years when the North Pacific is less favorable. Similarly, an early-spawning population segment may thrive in warm years when feeding and rearing conditions are favorable at emergence but the same population segment may suffer in cold years when they emerge too early for the prevailing conditions. According to McElhany et al. (2000), diversity guidelines are reached when: 1) variation in life history, morphological, and genetic traits is maintained, 2) natural dispersal processes are maintained, 3) ecological variation is maintained, and 4) effects of uncertainty are considered.

4.2.3.4 Spatial Structure

Spatial structure refers to the amount of habitat available, the organization and connectivity of habitat patches, and the relatedness and exchange rates of adjacent populations. Large habitat patches or a connected series of smaller patches are generally associated with increased population viability. Thus, spatial structure guidelines are reached when: 1) the number of habitat patches is stable or increasing; 2) stray rates are stable; 3) marginally suitable habitat patches are preserved; 4) refuge source populations are preserved, and 5) uncertainty is taken into account (McElhany et al. 2000).

4.2.4 Ocean and Climatic Variability

Large fluctuations in salmon numbers during the last few years have highlighted the importance of ocean conditions in regulating salmon survival and abundance, although different species and populations respond differently to fluctuations in ocean conditions. Twenty years ago fish scientists generally regarded the ocean as a vast and consistently productive environment for salmonids. However, frequent El Niño circulation patterns over the last 20 years have demonstrated that environmental conditions are much more dynamic than previously thought. Ocean conditions are not randomly sorted – poor years tend to occur in groups as do good years. Transitions between good and poor regimes occur unpredictably and are obvious only in hindsight. Low salmon survival during El Niño years results in population declines and critically low numbers. Abnormally good salmon survival in cool, wet years following large El Niños results in temporary population increases and record returns like those seen in 2001–03.

Periodic poor ocean cycles are the stressor that bottoms out populations compromised by habitat degradation and overuse (Lawson 1993). Healthy populations are able to ride out the declines without lingering effects. Ocean conditions have always varied and always will. Just because salmon numbers decline during poor survival periods should not mean fish are threatened or endangered with extinction. Alternatively, high numbers returning in good ocean years does not mean that threatened or endangered fish are recovered. Recent large salmon runs suggest that we may have entered a period of better-than-average ocean survival conditions. Rather than relaxing the need for salmon recovery, this pattern provides an opportunity to implement substantive changes for population rebuilding needed to withstand the next down cycle. Habitat and demographic improvements require time to become effective and may come too late if the next period of decline is the one from which the population cannot recover.

4.2.5 The Difference Between Wild and Hatchery Fish

By both design and happenstance, fish produced in hatcheries sometimes intermingle with wild fish in spawning areas and contribute to natural production. Hatchery contributions to wild populations vary widely among species and populations depending on hatchery proximity and practices. Some natural spawning populations include large fractions of hatchery fish. Other populations are largely free of hatchery influence.

Effects of natural spawning by hatchery fish have been extremely controversial (see Hatchery Section in Chapter 3). One issue has been the potential for reduced fitness and viability of some wild populations as a result of the introduction of domesticated or non-local hatchery fish that are ill-suited to local conditions. A second issue is the difficulty of accurately measuring numbers and productivity of wild populations where hatchery influence is significant. These concerns can be at odds with the fishery mitigation and conservation values of hatcheries. Conservation values include preserving genetic stocks where habitat is gone, reintroducing fish in areas where habitat has been restored, and bolstering survival to offset survival bottlenecks. It can be especially difficult to distinguish situations where hatchery contributions to natural spawning reduce wild population productivity because of fitness effects or supplement wild population productivity because of high hatchery survival rates.

Definition of recovery standards is particularly concerned with the issue of accurate accounting of wild fish. Where defined in terms of population viability, recovery will depend on sustainable long-term production of wild fish in natural habitats. Populations maintained through a continuing influx of hatchery fish are not considered sustainable if they might become extinct whenever the subsidy is removed. Hatchery subsidies have not proven to be a viable long-term alternative to wild production because of gradual fitness loss over successive hatchery generations and the vagaries of future hatchery program commitments and funding. Thus, definition of recovery standards and assessments of status currently distinguish hatchery contributions to spawner numbers and subsequent generations of offspring. This does not mean that hatchery fish should be never be allowed to spawn in the wild. It does mean that accounting practices need to make the necessary adjustments to accurately represent the wild population component independent of significant hatchery fish effects, thus providing an accurate assessment of the ability of the habitat conditions to support wild populations.

4.2.6 Biological and Social Values

Considerations of both biological and social values are implicit in the definition of recovery standards. The line between biological and social considerations can sometimes be difficult to distinguish, especially because social values can often be expressed in biological terms. For instance, where the predominant social value derives from fishery benefits, a biological standard equivalent to maximum or optimum sustainable yield might be considered. Where the predominant social value derives from water use rather than fishery benefits, a biological standard equivalent to minimum population viability might be considered. Where ecological, intrinsic, or cultural fish values predominate, a biological standard equivalent to pre-development capacity might be considered.

Definition of appropriate recovery standards will require difficult decisions by policy makers to balance a complex of competing biological and social values. Biological constraints provide the sideboards for policy decisions but social values will ultimately drive where within these sideboards we aim. Social rather than biological values increasingly drive definitions of recovery standards as population numbers increase above the minimum viability threshold necessary to safely conserve the species and meet the legal requirements of the ESA. Above the viability threshold, various arguments can be made for and against the intrinsic biological and ecological value of one standard or another. The fish have a natural tendency to expand their populations, but do not "care" whether they are recovered to viable, optimum yield, or capacity levels. However, people care about which standards are specified because each alternative has large implications to different combinations of social, economic, and cultural costs and benefits.

Of course, not everyone will agree that different standards above the viability minimum are driven primarily by social rather than biological values. This counter argument is probably strongest with respect to ecological values of fish to other parts of the ecosystem. For instance, salmon provide food for bears and eagles, and marine-derived nutrients that substantially affect plant and animal productivity, and even subsequent salmon production, in many watersheds. The real pitfall occurs when the biological and social tradeoffs implicit in various standards are not clearly articulated and/or distinguished. These pitfalls can lead to unrecognized conflicts of interest, especially when social values are represented in purely biological terms.

4.3 Recovery Standards

This section of the Technical Foundation defines standards by which recovery will be described and measured for focal fish species in the Washington subbasins of the Lower Columbia River. Standards identify where we need to go relative to where we are now. This gap between current and desired conditions determines the nature, magnitude, and costs of actions required to achieve fish recovery. Where the gap is large, effective recovery will require substantive actions across a broad area involving potentially costly changes in a variety of human activities. This information will provide a context for definition in the next phase of recovery planning: the setting of specific recovery goals by the Lower Columbia Fish Recovery Board consistent with the Washington legislative mandate of this recovery planning effort and Federal ESA requirements.

4.3.1 Definitions

Recovery standards can be expressed in a hierarchy of increasingly specific levels. *Recovery Goals* capture the biological and social purpose or vision for fish restoration efforts. The overarching goal of this recovery planning effort is *to restore viable fish populations under selected levels of utilization.* This broad recovery goal addresses Washington's legislated mandate for "healthy and harvestable" salmon populations. This goal also addresses Federal requirements under the ESA to protect populations from extinction and under the Magnuson Act to manage for optimum sustainable harvests. Viability is shorthand for ensuring the continued existence of a species (i.e. avoiding extinction). Utilization is broadly defined to include direct effects such as fishing and indirect effects such as water use and watershed development. Specific recovery goal might also be extended to incorporate the concept of ecological health.

Criteria are biological expressions of objectives consistent with Recovery Goals. For instance, general criteria have been defined by the Willamette/Lower Columbia Technical Recovery Team and endorsed by the Executive Committee to describe fish numbers, population processes, and conditions consistent with viability. *ESU and Strata Level Criteria* describe acceptable probabilities of persistence or risk for groups of populations needed to assure that acceptable probabilities are achieved. *Population Level Criteria* include factors related to individual population status and viability including abundance, productivity, diversity, and distribution. Population level criteria may also include related factors such as habitat quantity and quality. Changes in salmon numbers, productivity, and distribution can be directly related to specific habitat parameters such as temperature, substrate, channel type, etc. Fish habitat metrics are particularly effective when used in conjunction with population size and population quality metrics to identify key limiting factors that can be targeted by specific recovery actions.

Recovery Targets are biologically based numerical expressions of recovery goals that reflect both biological and social factors.

4.3.2 Biological Viability Criteria

The Willamette/Lower Columbia Technical Recovery Team (TRT) has identified a series of criteria consistent with a viability recovery goal (Figure 4-7). Criteria were based on the Viable Salmonid Population (VSP) Concept developed by NOAA fisheries and incorporate species, population size, population quality, and habitat metrics (McElhaney et. al. 2000).

TRT guidelines include ESU, Strata, and Population Persistence Criteria. ESU criteria prescribe that every life history and ecological zone stratum should be preserved (Box 1). Strata criteria prescribe preservation of multiple populations within each stratum at levels sufficient to maintain normal species processes. Population persistence criteria ensure that no populations are sacrificed until viability is assured and identify aggressive recovery efforts that recognize the uncertain outcome of population-specific efforts. An ESU is viable only where ESU and Strata criteria are all met.

Each ESU consists of one or more strata that represent different life history and ecological zone combinations. For instance, the Lower Columbia River Chinook Salmon ESU includes Washington and Oregon populations from the Columbia River mouth to the Big White Salmon River in Washington and the Hood River in Oregon. Distinct ecological zones in this range include Coast, Cascades, and Gorge watersheds. Chinook life history types include stream-type spring run, ocean-type fall run (tules), and ocean-type late fall run (brights). Thus, chinook salmon strata include Coast fall, Cascade fall, Cascade late fall, Gorge spring, etc.

TRT population criteria are based on population size, population quality, and fish habitat metrics (Box 2). Population size

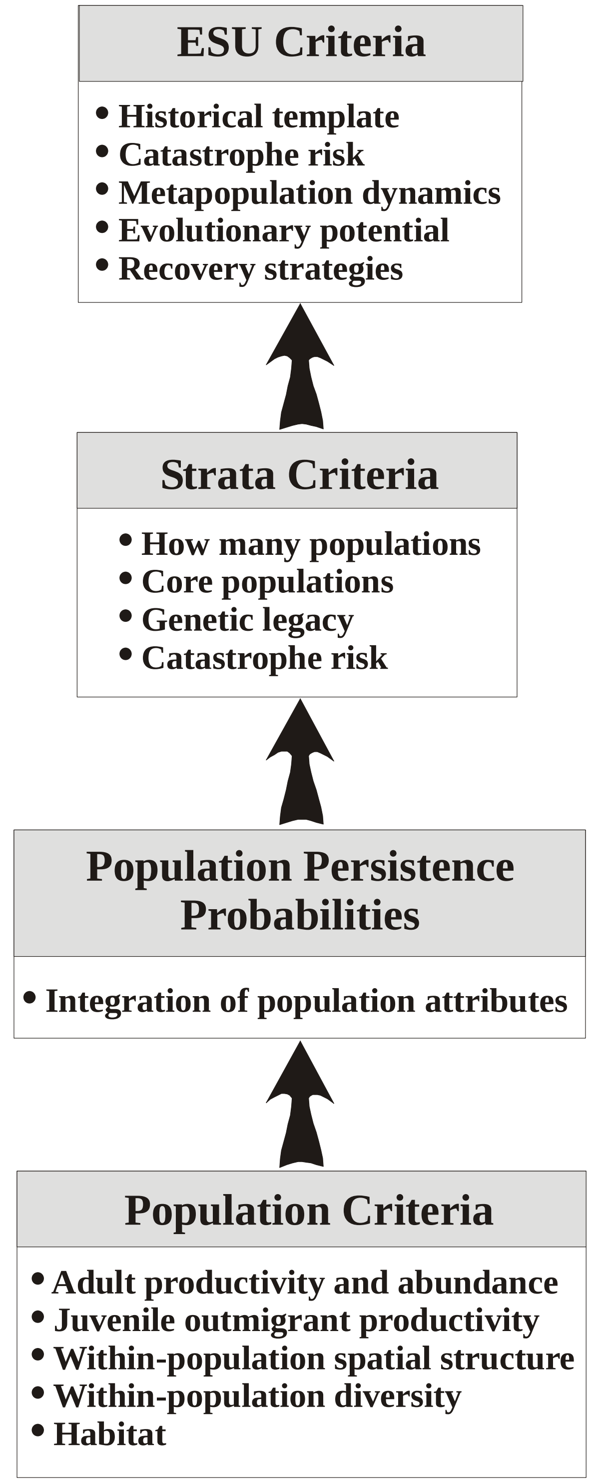

Figure 4-7. Willamette/Lower Columbia approach to viability criteria.

criteria are the same as the recovery planning target ranges discussed previously. Adult productivity criteria are based on annual rate of population increase and recruits per spawner. Juvenile emigrant productivity criteria are based on an increasing trend in emigrant numbers. Spatial structure and diversity criteria provide sufficient diversity to support desired levels of productivity, abundance, and diversity. Habitat criteria prescribe stable or increasing trends in quantity and quality. Abundance, productivity, diversity, and distribution are highly interrelated to each other. Thus, high diversity often results in high productivity and a wider distribution because different components of the population are adapted to a wider array of habitat and environmental conditions. Conversely, high productivity results in larger population sizes that are more likely to avoid abundance bottlenecks that reduce diversity and often increases population spatial structure as returning fish spread across wider areas.

Box 1. ESU and Strata Level Recovery Guidelines from the Willamette/L. Columbia Technical Recovery Team

ESU-Level Recovery Strategy Criteria Guidelines

1. Every stratum (life history and ecological zone combination) that historically existed should have a high probability of persistence.
2. Until all ESU viability criteria have been achieved, no population should be allowed to deteriorate in its probability of persistence.
3. High levels of recovery should be attempted in more populations than identified in the strata viability criteria because not all attempts will be successful.

Strata Criteria Guidelines

1. Individual populations within a stratum should have persistence probabilities consistent with a high probability of strata persistence.
2. Within a stratum, the populations restored/maintained at viable status or above should be selected to:
   a. Allow for normative metapopulation processes, including the viability of “core” populations, which are defined as the historically most productive populations
   b. Allow for normative evolutionary processes, including the retention of the genetic diversity represented in relatively unmodified historic gene pools.
   c. Minimize susceptibility to catastrophic events.

TRT standards also provide an integrated scoring system to project population persistence probabilities from population criteria and to estimate ESU viability based on the status of individual populations in the various strata (McElhany et al. 2003). Each population attribute is evaluated separately on a 0-4 scale where 0 is either extinct or at very high risk of extinction, 1 is at relatively high risk, 2 is at moderate risk, 3 is at low risk (i.e. viable), and 4 is at very low risk. Attribute scores are averaged across each attribute to identify net population persistence level. Attributes are weighted equally except that the abundance and productivity attribute is weighted twice because the TRT concluded this metric provided the most direct and objective measure of population viability. Where juvenile data are unavailable for a population, juvenile criteria do not contribute to the average. The TRT also defined a high probability of ESU persistence (i.e. recovery) equivalent to an average of at least medium-high probability of persistence in all historic populations within the ESU with at least 2 individual populations in each stratum at a high probability of persistence. Table 4-1 illustrates an example application of the integrated scoring system on a simplified ESU.

Box 2. Population level viability guidelines from the Technical Recovery Team

Adult Population Productivity and Abundance

1. In general, viable populations should demonstrate a combination of population growth rate, productivity, and abundance that produces an acceptable probability of population persistence. Various approaches for evaluating population productivity and abundance combinations may be acceptable, but must meet reasonable standards of statistical rigor.
2. A population with non-negative growth rate and an average abundance approximately equivalent to estimated historic average abundance should be considered to be in the highest persistence category. The estimate of historic abundance should be credible, the estimate of current abundance should be averaged over several generations, and the growth rate should be estimated with adequate statistical confidence. This criterion takes precedence over criterion 1.

Juvenile Migrant Production

1. The abundance of naturally produced juvenile migrants should be stable or increasing as measured by observing a median annual growth rate or trend with an acceptable level of confidence.

Within-Population Spatial Structure

1. The spatial structure of a population must support the population at the desired productivity, abundance, and diversity levels through short-term environmental perturbations, longer-term environmental oscillations, and natural patterns of disturbance regimes. The metrics and benchmarks for evaluating the adequacy of a population's spatial structure should specifically address:
    a. Quantity: Spatial structure should be large enough to support growth and abundance, and diversity criteria.
    b. Quality: Underlying habitat spatial structure should be within specified habitat quality limits for life-history activities (spawning, rearing, migration, or a combination) taking place within the patches.
    c. Connectivity: spatial structure should have permanent or appropriate seasonal connectivity to allow adequate migration between spawning, rearing, and migration patches.
    d. Dynamics: The spatial structure should not deteriorate in its ability to support the population. The processes creating spatial structure are dynamic, so structure will be created and destroyed, but the rate of flux should not exceed the rate of creation over time.
    e. Catastrophic Risk: the spatial structure should be geographically distributed in such a way as to minimize the probability of a significant portion of the structure being lost due to a single catastrophic event, either anthropogenic or natural.

Within-Population Diversity

1. Sufficient life-history diversity must exist to sustain a population through short-term environmental perturbations and to provide for long-term evolutionary processes. The metrics and benchmarks for evaluating the diversity of a population should be evaluated over multiple generations and should include:
    a. a substantial proportion of the diversity of a life-history trait(s) that existed historically,
    b. Gene flow and genetic diversity should be similar to historic (natural) levels and origins,
    c. Successful utilization of habitats throughout the habitat, and
    d. Resilience and adaptation to environmental fluctuations

General Habitat

1. The spatial distribution and productive capacity of freshwater, estuarine, and marine habitats should be sufficient to maintain viable populations identified for recovery.
2. The diversity of habitats for recovered populations should resemble historic conditions given expected natural disturbance regimes (e.g. wildfire, flood, volcanic eruptions, etc.). Historic conditions represent a reasonable template for a viable population; the closer the habitat resembles the historic diversity, the greater the confidence in its ability to support viable populations.
3. At a large scale, habitats should be protected and restored, with a trend toward an appropriate range of attributes for salmonid viability. Freshwater, estuarine, and marine habitat attributes should be maintained in a non-deteriorating state.

Table 4-1. Example calculation of integrated population risk assessment based on interim recovery criteria.

Stratum	Population	Attributes: Prod. & Abund.	Juveniles	Spatial Structure	Diversity	Habitat	Population Persistence Category	Persistence Assessment
A	1	0	0	0	0	0	0	Extinct
	2	1	1	2	1	2	1.3	Low
	3	3	3	3	4	3	3.2	High
	Average						***1.5***	***Low***[1]
B	1	0	0	0	0	0	0	Extinct
	2	3	3	3	4	3	3.2	High
	3	3	3	3	4	3	3.2	High
	Average						2.1	***Moderate***[2]
C	1	1	1	2	1	2	1.3	Low
	2	3	3	3	4	3	3.2	High
	3	3	3	3	4	3	3.2	High
	Average						2.6	***High***[3]

1 Low stratum rating equivalent to a 0-75% probability of persistence and an average strata score of less than 2.0.

2 Moderate stratum equivalent to a 75-95% probability of persistence and an average strata score of greater than 2.0 with at least 2 populations greater than 3.0.

3 High strata rating equivalent to 95% or greater probability of persistence and an average strata score of at least 2.25 with at least 2 populations greater than 3.0.

TRT viability criteria provide guidelines for recovery planning but do not anticipate every combination of cases that might be encountered. Criteria also do not prescribe specific rules for evaluating all information that might inform assessments for each population attribute related to persistence. Thus, the interim criteria provided by the TRT provide strong guidelines for recovery and biological sideboards on how integrated status assessments should be made but also allow for some flexibility in application consistent with the spirit of the criteria.

4.3.3 Planning Ranges: Balancing Biological and Social Goals

In this Technical Foundation, recovery targets are expressed as a planning range. The lower bound is a minimum consistent with population viability. The upper bound is the realistic maximum based on the capacity of a restored system. Methods for setting target planning ranges for abundance were developed by the LCFRB and are based on guidelines identified by the Willamette/Lower Columbia Technical Recovery Team (see Chapter 5). Planning ranges are intended to be used in close conjunction with TRT population attributes.

Planning ranges are species and population specific. Planning ranges vary among individual populations as a result of subbasin differences in habitat quantity, habitat quality, fish distribution, population productivity, etc. Threatened or endangered ESUs typically include some populations where current numbers fall within the target planning range but a majority of populations fall below the planning range (Figure 4-8). Recovery will require moving future abundance into the planning target range for a significant number of all historic populations for each species and ESU. The exact numbers that constitute a "significant proportion" were identified in criteria prescribed by the Willamette/Lower Columbia Technical Recovery Team.

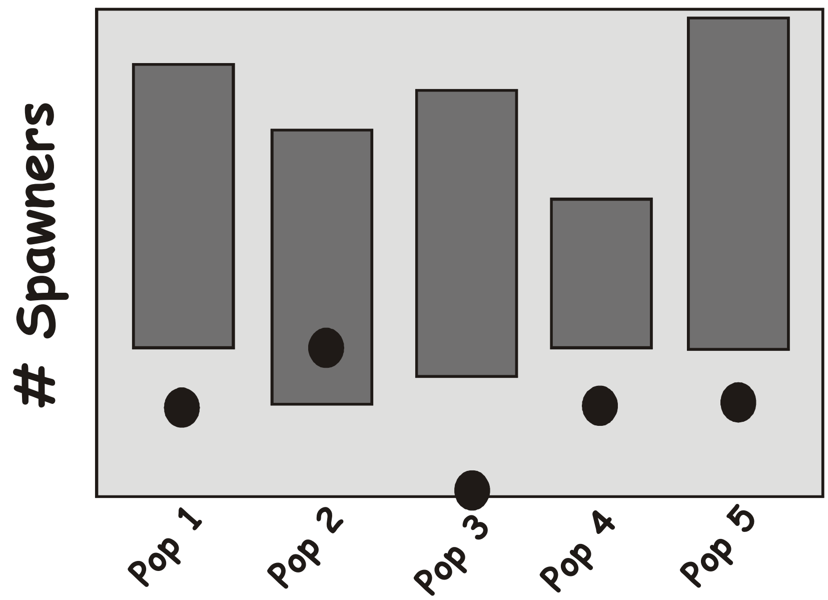

Figure 4-8. Example comparison of current status with planning target ranges for Washington lower Columbia River salmon recovery (Note: Top and bottom of bars are the maximum and minimum planning ranges; points are current status).

Planning range targets may be defined in terms of either abundance or productivity. Because extinction is fundamentally a numbers issue, abundance is a direct estimate of the key parameter of concern. If we knew absolutely nothing else about the fish, estimates or indices of annual spawner abundance would tell us where problems are acute and when trends have been reversed. Fish numbers can be measured at various life stages and different stages provide slightly different information on status. For instance, numbers of adult spawners can be compared with minimums required to maintain an effective genetic population size to preserve diversity. Numbers reaching adulthood in the ocean would identify preharvest spawning potential. Numbers of juvenile migrants would identify patterns in freshwater productivity and habitat conditions.

Specific goals within this planning range will be selected through a collaborative process led by the Lower Columbia River Fish Recovery Board during subsequent phases of the recovery planning process. Target planning ranges provide the biological sideboards for policy decisions by the lower Columbia Fish Recovery Board while considering the desired balance among human impacts. Specific recovery goals will be defined within the target planning range. The lower end of the range is the minimum conservation standard that must be achieved in a critical number of populations. The upper end of the range sets a realistic maximum for expectations. Average population sizes need only to exceed the low bound of the target planning range within a specified time period to ensure long-term population viability and meet Federal ESU delisting requirements. Higher average numbers will be required to meet State and Federal requirements for healthy harvestable populations. Exactly how high will depend on the desired balance of fisheries, other human impacts, and ecosystem values. Numbers near the low end of the planning range would provide only limited opportunity for harvest but would require smaller changes in water or land use. Conversely, numbers near the upper end of the planning range would provide optimum harvest opportunity but would require greater habitat changes.

Volume I, Chapter 5
Assessments of Current Status & Limiting Factors

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5.0 Assessments of Current Status and Limiting Factors

Other sections of this Technical Foundation summarize the available information on fish status, limiting factors, and recovery standards. This chapter includes assessments of current population status relative to potential recovery benchmarks for each focal fish species. This chapter also describes analyses, based on a synthesis of the best available scientific information, of the relative significance of six actors for decline: fishing, hatcheries, stream habitat, mainstem and estuary habitat, dams, and predation. (Only factors within the realm of human management were included.) These evaluations provide a road map of possible avenues for recovery and a basis for more detailed assessments of recovery scenarios and strategies in the next phase of the recovery planning process. The assessment approach is an adaptation of alternatives previously identified in the Lower Columbia Fish Recovery Analytical Framework (TWC and SPCA 2003).

For effective interpretation by both highly technical scientific professionals and an informed lay audience, descriptions of current status and factors for decline must be technically defensible, based on the best available data, as well as intuitively easy to interpret. A sound technical approach was needed to provide effective guidance and to withstand intense scientific scrutiny. "Best available data" is the standard for evaluating Endangered Species assessments. In many cases, the "best available" may be less than ideal but scientific information can support informed decisions, provide direction, reduce uncertainty, and generate testable hypotheses even where the data is not definitive. Finally, descriptions need to be intuitively easy to understand by a mix of technical and non-technical people who will be called upon to make scientific and policy decisions based on this data.

Specific assessments for each species include: 1) estimates of current viability for each population, 2) comparisons of current fish numbers with recovery planning ranges, 3) descriptions of the biological significance of each population, 4) indices of the relative effects of each limiting factor for each fish population, and 5) subjective summaries of the recovery prospects for each focal fish species. Estimates of current viability provide an systematic representation of current status. Planning ranges will help identify biological objectives for recovery planning relative to the healthy and harvestable goal identified by the LCFRB. Biological significance will provide a useful index for sorting populations in future considerations of alternative recovery scenarios. Indices of limiting factor effects will help inventory threats to viability and potential avenues for recovery. Summaries will help highlight potentially effective recovery strategies.

5.1 Approach

5.1.1 Current Viability

The first step towards recovery is understanding current population viability, the long-term prospects for preservation of a naturally self-sustaining population. A population is viable where persistence probabilities are high. High persistence probabilities correspond to low extinction risks and constitute recovery for key species units (Evolutionarily Significant Units) under the Federal Endangered Species Act. Minimum component population levels required to ensure that ESUs do not go extinct constitute the low end of recovery planning targets identified by the Lower Columbia River Fish Recovery Board.

We evaluated viability based on standards developed by the Willamette/Lower Columbia Technical Recovery Team (TRT), consisting of a committee of scientists convened by NOAA Fisheries to provide technical guidance in fish recovery. As detailed in the previous chapter on Recovery Standards, TRT viability guidelines are based on scores assigned to attributes related to the viability of each individual fish population within an ESU. Attributes include spawner abundance, productivity, juvenile outmigrant numbers, diversity, spatial structure, and habitat conditions (McElhany et al. 2003). Each population is rated for each attribute on a 0-4 scale based on the available information. Individual attribute ratings are averaged for each population. The rating scale corresponds to 100-year persistence probabilities: 0=0-40%, 1 = 40-75%, 2 = 75-95%, 3 = 95-99%, 4 > 99%. Population scores can then be counted and averaged across a geographic strata for each species for comparison with recovery benchmarks established by the TRT. The lower Columbia region includes Coast, Cascade, and Gorge strata identified by the TRT to capture within-ESU differences in population characteristics related to differences in geographical and environmental conditions in different ecological zones. These benchmarks include a strata average persistence probability greater than 2.25 with at least two populations at high persistence probabilities (≥3.0). Because this viability approach is a building block for population significance, it is described in more detail below.

Population status was scored independently by the TRT and by Washington or Oregon fish biologists with specific knowledge and expertise on lower Columbia River salmon populations. TRT and State scores were averaged for the purposes of this evaluation. Independent estimates in Washington were completed by LCFRB scientific consultants (Ray Beamesderfer and Guy Norman) and WDFW staff (Dan Rawding). Table 4-1 includes more detailed explanations of criteria applied to Washington scores. Population-specific rationales for LCFRB Washington scores may be found in technical appendices of Volume VI of this Technical Foundation. Oregon estimates were completed by Oregon Department of Fish and Wildlife Staff. Most of the Technical Foundation has been focused on lower Columbia River salmon populations in the Washington jurisdiction of this recovery planning effort. However, assessments of ESU viability also require information on Oregon populations. Recovery criteria address ESU-wide status and prospects for recovery. We therefore included summary information on Oregon stock status in this assessment to provide a context for Washington planning considerations.

Population trends and extinction risks are also reported based on analyses of population time series data by NOAA fisheries. TRT scores and time series analyses are alternative but related approaches to assessing population viability that can be used for cross-corroboration. In the NOAA time series analyses, abundance trends were described with median annual growth rates

(λ) based on slopes fit to 4-year running sums of abundance (Holmes 2000). Values less than and greater than 1.0 indicate decreasing and increasing trends, respectively, over the period of record. Extinction risks were based on two different models that make slightly different assumptions about future patterns from recent abundance time series data. The first model estimates the probability of extinction using the Dennis-Holmes method based on the risk that a population starting with the most recent four year sum will decline to less than 50 spawners given the population growth rate (λ) and observed variation in abundance. The second model uses population growth rate and variance derived from time series data with different statistical assumptions and also incorporates a nonlinear stock-recruitment population function (McElhany et al. 2003).

Current population sizes were also compared with historical "template" numbers to provide a perspective on differences that have contributed to current viability. Historical numbers were available from EDT analyses based on assumed habitat conditions. For comparison, historical numbers were also independently estimated by NOAA Fisheries based on a simple "back-of-envelope" (BOE) calculation – these estimates were only presented in our tables for comparison and were not used in the final summaries of this Technical Foundation. The BOE calculations extrapolated an assumed historical abundance of each ESU from literature sources and partitioned the total into populations based on respective fractions of accessible stream miles. The BOE was likely confounded by an assumption that all accessible streams supported similar densities of fish and relied on an assumed historical Columbia River run size. On the other hand, EDT estimated different stream-specific densities based on assumed differences in habitat conditions and relationships between habitat conditions and fish numbers.

Table 4-1. Population persistence categories used to score fish status relative to recovery criteria guidelines (Descriptions from McElhany et al. 2003, applications identified by WDFW & LCFRB staff).

Category	Description	Application[1]
	Population Persistence	
0	Either extinct or very high risk of extinction	Very low (0-40%) probability of persistence for 100 years
1	Relatively high risk of extinction	Low (40-75%) probability of persistence for 100 years
2	Moderate risk of extinction	Medium (75-95%) probability of persistence for 100 years
3	Low (negligible) risk of extinction	High (95-99%) probability of persistence for 100 years
4	Very low risk of extinction	Very High (>99%) probability of persistence for 100 years
	Adult Abundance and Productivity	
0	Numbers and productivity consistent with either functional extinction or very high risk of extinction	Extinction risk analysis estimates 0-40% persistence probability.
1	Numbers and productivity consistent with relatively high risk of extinction	Extinction risk analysis estimates 40-75% persistence probability.
2	Numbers and productivity consistent with moderate risk of extinction	Extinction risk analysis estimates 75-95% persistence probability.
3	Numbers and productivity consistent with low (negligible) risk of extinction	Extinction risk analysis estimates 95-99% persistence probability.
4	Numbers and productivity consistent with very low risk of extinction	Extinction risk analysis estimates >99% persistence probability.
	Juvenile Out-Emigrants	Evaluated based on the *occurrence* of natural production, whether natural production was *self sustaining* or supplemented by hatchery fish, *trends* in numbers, and *variability* in numbers.
0	Consistent with either functional extinction or very high risk of extinction[3]	No significant juvenile production either because no natural spawning occurs or because natural spawning by wild or hatchery fish occurs but is unproductive.
1	Consistent with relatively high risk of extinction[3]	Long term trend in wild natural production is strongly negative. Also includes the case where significant natural production occurs in many years but originates primarily from hatchery fish.
2	Consistent with moderate risk of extinction[3]	Sample data indicates that significant natural production occurs in most years and originates primarily from naturally-produced fish. No trend in numbers may be apparent but numbers are highly variable with only a small portion of the variability related to spawning escapement.
3	Consistent with low risk of extinction[3]	Sample data indicates significant natural production by wild fish occurs in all years. No long term decreasing trend in numbers is apparent. Juvenile numbers may be variable but at least some of this variability is related to fluctuations in spawning escapement.
4	Consistent with very low risk of extinction[3]	Sample data indicates significant natural production by wild fish occurs in all years. Trend is stable or increasing over extended time period. Variability in juvenile production is low or a large share of the observed variability is correlated with spawning escapement.

Category	Description	Application[1]
	Within-Population Spatial Structure	
0	Spatial structure is inadequate in quantity, quality[2], and connectivity to support a population at all.	*Quantity* was based on whether all areas that were historically used remain accessible. *Connectivity* based on whether all accessible areas of historical use remain in use. *Catastrophic risk* based on whether key use areas are dispersed among multiple reaches or tributaries. Spatial scores of 0 were typically assigned to populations that were functionally extirpated by passage blockages.
1	Spatial structure is adequate in quantity, quality[2], and connectivity to support a population far below viable size	The majority of the historical range is no longer accessible and fish are currently concentrated in a small portion of the accessible area.
2	Spatial structure is adequate in quantity, quality[2], and connectivity to support a population of moderate but less than viable size.	The majority of the historical range is accessible but fish are currently concentrated in a small portion of the accessible area.
3	Spatial structure is adequate in quantity, quality[2], and connectivity to support population of viable size, but subcriteria for dynamics and/or catastrophic risk are not met	Areas may have been blocked or are no long used but fish continue to be broadly distributed among multiple reaches and tributaries. Also includes populations where all historical areas remain accessible and are used but key use areas are not broadly distributed.
4	Spatial structure is adequate to quantity, quality, connectivity, dynamics, and catastrophic risk to support viable population.	All areas that were historically used remain accessible, all accessible areas remain in use, and key use areas are broadly distributed among multiple reaches or tributaries.
	Within-Population Diversity	
0	All four diversity elements (life history diversity, gene flow and genetic diversity, utilization of diverse habitats2, and resilience and adaptation to environmental fluctuations) are well below predicted historical levels, extirpated populations, or remnant populations of unknown lineage	*Life history diversity* was based on comparison of adult and juvenile migration timing and age composition. *Genetic diversity* was based on the occurrence of small population bottlenecks in historical spawning escapement and degree of hatchery influence especially by non local stocks. *Resiliency* was based on observed rebounds from periodic small escapement. Diversity scores of 0 were typically assigned to populations that were functionally extirpated or consisted primarily of stray hatchery fish.
1	At least two diversity elements are well below historical levels. Population may not have adequate diversity to buffer the population against relatively minor environmental changes or utilize diverse habitats. Loss of major presumed life history phenotypes is evident; genetic estimates indicate major loss in genetic variation and/or small effective population size. Factors that severely limit the potential for local adaptation are present.	Natural spawning populations have been affected by large fractions of non-local hatchery stocks, substantial shifts in life history have been documented, and wild populations have experienced very low escapements over multiple years.
2	At least one diversity element is well below predicted historical levels; population diversity may not be adequate to buffer strong environmental variation and/or utilize available diverse habitats. Loss of life history phenotypes, especially among important life history traits, and/or reduction in genetic variation is evident. Factors that limit the potential for local adaptation are present.	Hatchery influence has been significant and potentially detrimental or populations have experienced periods of critical low escapement.

Category	Description	Application[1]
3	Diversity elements are not at predicted historical levels, but are at levels able to maintain a population. Minor shifts in proportions of historical life-history variants, and/or genetic estimates, indicate some loss in variation (e.g. number of alleles and heterozygosity), and conditions for local adaptation processes are present.	Wild stock is subject to limited hatchery influence but life history patterns are stable. Extended intervals of critical low escapements have not occurred and population rapidly rebounded from periodic declines in numbers.
4	All four diversity elements are similar to predicted historical levels. A suite of life-history variants, appropriate levels of genetic variation, and conditions for local adaptation processes are present.	Stable life history patterns, minimal hatchery influence, no extended interval of critical low escapements, and rapid rebounds from periodic declines in numbers.
	Habitat	
0	Habitat is incapable of supporting fish or is likely to be incapable of supporting fish in the foreseeable future	*Unsuitable habitat.* Quality is not suitable for salmon production. Includes only areas that are currently accessible. Inaccessible portions of the historical range are addressed by spatial structure criteria[2].
1	Habitat exhibits a combination of impairment and likely future conditions such that population is at high risk of extinction	*Highly impaired habitat.* Quality is substantially less than needed to sustain a viable population size (e.g. low bound in target planning range). Significant natural production may occur in only in favorable years.
2	Habitat exhibits a combination of current impairment and likely future condition such that the population is at moderate risk of extinction	*Moderately impaired habitat.* Significant degradation in habitat quality associated with reduced population productivity.
3	Habitat in unimpaired and likely future conditions will support a viable salmon population	*Intact habitat.* Some degradation in habitat quality has occurred but habitat is sufficient to produce significant numbers of fish. (Equivalent to low bound in abundance target planning range.)
4	Habitat conditions and likely future conditions support a population with an extinction risk lower than that defined by a viable salmon population. Habitat conditions consistent with this category are likely comparable to those that historically existed.	*Favorable habitat.* Quality is near or at optimums for salmon. Includes properly functioning through pristine historical conditions.

[1] *Rules applied for each TRT criteria and category to develop integrated status assessments for example purposes of this technical foundation. Application rules were derived by project staff working in close association with WDFW staff. Application rules do not represent assessment by the Technical Recovery Team.*

[2] *Because recovery criteria are closely related, draft category descriptions developed by the Technical Recovery Team often incorporate similar metrics among multiple criteria. For instance, habitat-based factors have been defined for diversity, spatial structure, and habitat standards. To avoid double counting the same information, streamline the scoring process, and provide for a systematic and repeatable scoring system this application of the criteria used specific metrics only in the criteria where most applicable. This footnote denotes these items.*

[3] *This is a modification of the interim JOM criteria identified by the TRT. JOM scores consistent with persistence probabilities for other criteria. Consistent with an attempt to avoid double counting similar information in different criteria, data quality considerations were not included in the revised JOM criteria descriptions because they are scored separately for all criteria. This modification removes confounding effects of cases where no JOM data is available and provides*

5.1.2 Recovery Planning Ranges

5.1.2.1 <u>Definition</u>

Recovery planning ranges provide approximate benchmarks for describing the biological objectives of recovery. Planning ranges are fish numbers for each population at: 1) minimum averages needed to ensure population viability (i.e. avoid extinction) and 2) realistic maximums that might be achieved by widespread restoration of favorable habitat conditions for salmon. The low bound of the planning range thus represents potential delisting goals for ESA populations. The high bound represents a limit to potential expectations rather than a goal.

Planning ranges were described both in terms of spawner numbers and population productivity. Greater fish numbers generally correspond to greater population productivity and increased population viability. Each alternative for describing status lends itself to different applications and analyses. Fish numbers can be measured directly and provide an intuitively easy-to-understand description of how well a population is doing. Productivity (replacement rate) provides a more direct description of the dynamics that determine status and viability. Viability level reflects persistence probabilities and extinction risks that are a particular concern for conservation and preservation of sensitive populations including those listed under the ESA.

Comparisons of current numbers and planning ranges provide an index of the difference between current, viable, and potential values (Figure 5-1). The low bound of the planning range is equivalent to a high level of viability as described by the Willamette/Lower Columbia Technical Recovery Team. Very high levels of viability are assumed to occur at population levels less than the potential reflected by the high bound on the planning range.

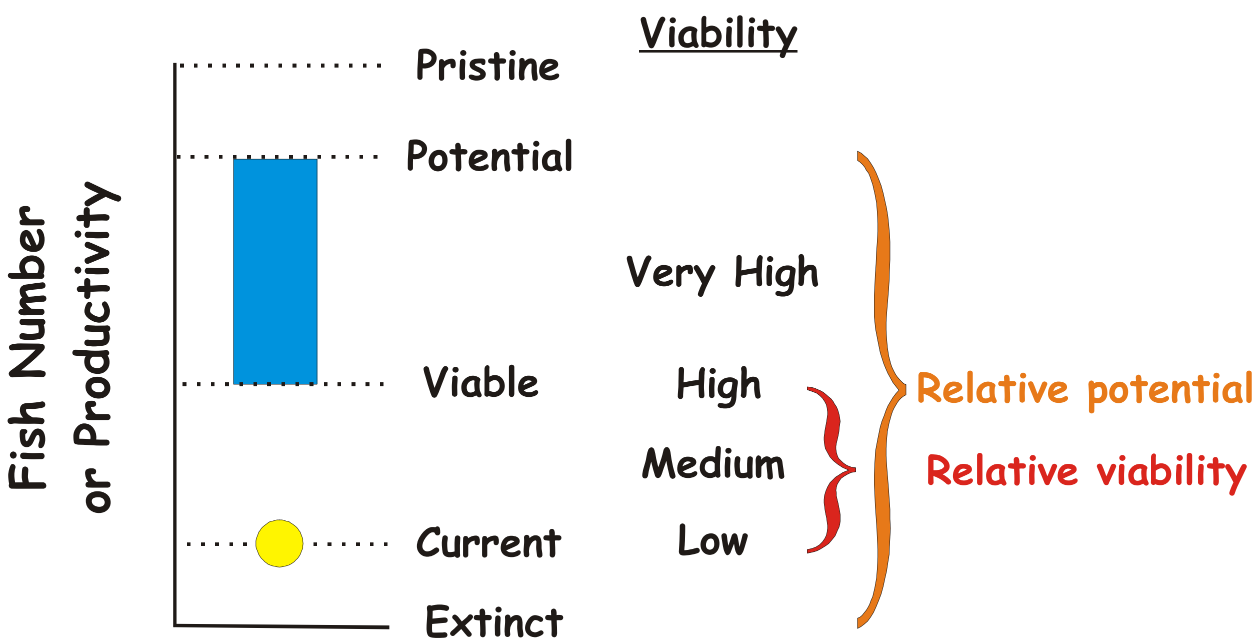

Figure 5-1. Depiction of generic recovery planning ranges relative to viability levels identified by the Willamette/Lower Columbia Technical Recovery Team.

5.1.2.2 Derivation

The low bound of the planning range was generally based on Population Change Criteria (PCC) developed by NOAA Fisheries. PCC determines the population growth rate and average abundance after 20 years needed to minimize risks of falling below critical low population sizes over 100 years. Estimates were based on recent 4-year average spawning escapement of naturally produced fish for each population and annual variation in escapement of each species (McElhany et al. 2003).

Planning range abundance values at viability were expressed as 4-year average spawner numbers. Default PCC values of 600, 1,100, and 1,400 spawners were used for steelhead, chum, and chinook, respectively, where either spawning escapement data were not available, numbers were thought to average less than 150 spawners per year, or estimated PCC values were less than default values. In populations where the available assessments indicate that extinction risks are not significant (i.e. less than 5% within 100 years), current abundance (recent 4-year natural spawning escapements) values were used as the low bound rather than the PCC values. (PCC derivation is based on assumption of an at-risk population. Where the population is not at risk, PCC numbers are undefined. Spawner numbers rather than EDT-derived population estimates (Neq) were used for comparability with PCC units.) Where PCC numbers exceed potential habitat capacity under properly functioning conditions estimated using EDT, the PFC+ EDT value was used as a minimum and no upper bound was specified. (This situation most commonly results from the apparent presence of large numbers of naturally-produced spawners from hatchery-origin spawners in preceding generations.)

Planning range productivity values at viability were expressed as median annual population growth rates (λ). Current estimates were derived by NOAA Fisheries from escapement time series data analyses (Holmes 2000). Population productivity values needed to achieve PCC growth rates require proportionately larger increases where λ is less than 1.0 (McElhany, personal communication). Thus, viable median annual population growth rates were: $(1 + \Delta\lambda)$ where $\Delta\lambda$ = population change criteria for productivity derived by McElhany et al. (2003). Default PCC values ($\Delta\lambda$) of 9%, 14%, and 15% population growth per year were used for steelhead, chum, and chinook, respectively, where population-specific PCC estimates were not available.

The upper end of the planning range represents the theoretical capacity if currently-accessible habitat was restored to good, albeit not pristine, conditions represented by the "properly functioning habitat conditions" identified by NOAA Fisheries. Abundance and productivity at PFC was estimated using the Ecosystem Diagnosis and Treatment Model as describe in Volumes II and VI of this Technical Foundation. PFC describes stream conditions suitable for salmon throughout the accessible range. In this application, the upper end of the planning range also assumed no removal of existing dams, no fishing, and the estuary at historical productivity levels. PFC stream habitat conditions and historical estuary productivity levels are typically referenced as PFC+ to distinguish from PFC stream habitat conditions with current estuary productivity levels.

The upper bound of the abundance planning range was defined in terms of equilibrium spawner numbers. Equilibrium numbers are long term averages that can be expected based on average marine survival patterns. For planning purposes, we conservatively assumed an upper bound of two times the lower abundance bound where EDT was not available.

The upper bound of the productivity planning range was based on EDT values which are expressed as the asymptotic Beverton-Holt recruit per spawner parameter (the slope at origin or β^{-1}: Ricker 1975). This parameter describes maximum adult spawner per spawner values which are realized at low spawner numbers. Spawner/spawner parameters were transformed into equivalent median annual population growth rates based on the following assumption:

$$\lambda_{pfc+} / \lambda_{current} = \mathrm{Ln}\ \beta^{-1}_{pfc+} / \mathrm{Ln}\ \beta^{-1}_{current}$$

Available estimates $\lambda_{current}$ (Holmes 2000), β^{-1}_{pfc+} (EDT), and $\beta^{-1}_{current}$ (EDT) were used to solve for $\lambda_{pfc+.}$

5.1.2.3 Improvement Increments

Recovery scenarios based on TRT guidelines prescribe biological objectives that target different recovery levels for different populations. Some populations need to be restored to high levels of viability. Other populations need to be improved to contribute to ESU viability but need not reach high levels of viability. Yet other populations need to reach very high levels of viability to compensate for recovery uncertainties and to provide opportunities for other uses such as harvest. Comparisons of current status with recovery planning ranges provide a means of estimating improvement increments necessary to reach any given population level. Increments based on productivity differences also provide a means for relating necessary improvements to manageable impact factors.

Proportional improvements in population productivity were estimated for recovery of populations from current status to contributing, high, and very high levels of population viability consistent with recovery scenarios. Improvements to reach high levels of viability were based on the difference between current and viable median annual population growth rates. Thus, proportional productivity improvements to reach viability (θ_{high}) are:

$$\theta_{high} = [(1 - \lambda) + \Delta\lambda] / \lambda$$

Contributing populations were arbitrarily assumed to increase half the distance between current and viable productivities:

$$\theta_{contributing} = \theta_{high} / 2$$

Populations at very high levels of productivity were arbitrarily assumed to increase to half the distance between viable and potential (e.g. the mid-point of the recovery planning range):

$$\theta_{very\ high} = \theta_{high} + (\theta_{potential} - \theta_{high}) / 2$$

where

$$\theta_{potential} = (\mathrm{Ln}\ \beta^{-1}_{pfc+} - \mathrm{Ln}\ \beta^{-1}_{current}) / \mathrm{Ln}\ \beta^{-1}_{current}$$

This alternative was chosen instead of using PFC+ for high viability under the presumption that persistence probability will approach 100% in many populations under conditions well below PFC+.

Average species values were used for θ_{high} where population-specific values were not available. We used whichever produced the greater increment: A) average of viable population productivities from populations with data or B) average of incremental improvements needed to move from current to viable in populations with direct estimates. Also note that in cases where λ

was greater than 1.0, we assumed that it was 1.0. These assumptions were needed to reconcile differences between λ estimates and TRT status score assignments. For instance, some population productivities already exceed the viability average yet were scored as not viable under TRT criteria. Otherwise we would be saying no improvement is needed to get to viable for populations that were scored to be less than viable.

Estimated productivity increments highlight order-of-magnitude improvements in productivity needed to reach recovery. Population-specific estimates should be considered with caution because of large uncertainties in assessments. Species averages and ranges provide general guidelines. These estimates build upon results of existing analytical frameworks (EDT & PCC) to make a first approximation of the scale of needed improvements. Both EDT and PCC relied on simplifying and sometimes differing assumptions. Our extrapolation of results is also beyond the immediate intended application of each method. Given the ultimate uncertainty in the effects of recovery actions and the need to implement an adaptive recovery plan, this approximation should be adequate for developing order-of-magnitude estimates to which recovery actions can be scaled consistent with the current best available science and data. However, the adaptive research and evaluation component of the recovery plan should include data collection and further analysis based on an integrated life cycle framework that meshes an age-structured density-dependent population model like EDT with a stochastic empirical approach like PCC to directly relate persistence probabilities to population productivity.

5.1.3 Population Significance

To facilitate future development of recovery scenarios consistent with biological guidelines for recovery, we developed a simple index to systematically rate the biological significance of each population based on the available data. Biological significance is one of several elements including feasibility, equity, and efficiency that will considered in the development of recovery scenarios. Biological significance will inform but not necessarily drive the selection of recovery scenarios. For instance, less "significant" populations or subbasins might be targeted for more intensive recovery efforts where feasibility is greater.

The biological significance of each fish population can be described in terms of current viability, potential production, and genetic character:

Current viability: likelihood that a population will not go extinct within a given time frame. The healthiest, most robust current populations are the most viable.

Core potential: number of fish that could be produced in a given area if favorable historical conditions could be at least partially restored.

Genetic character: current resemblance to historical characteristics that were intended to be preserved.

Specific guidelines related to each of these attributes are the basis for population viability criteria identified by the Willamette/Lower Columbia Technical Recovery Team (McElhany et al. 2003). For instance, current viability was defined by the TRT in terms of population persistence probability. (Current viability was based on the scoring approach described in the previous section). Potential production is related to the TRT core population designation. Core populations "represented the substantial portion of the ESU's abundance or contained life-history strategies that were specific to the ESU." Thus, core populations were typically the largest historical populations. Finally, the TRT designated genetic legacy populations as having "minimal influence from nonendemic fish due to artificial propagation activities, or the population may exhibit important life-history characteristics that are no longer found throughout much of their historical range in the ESU."

Biological significance ratings (B) were calculated for each population based on the following formula:

$$B = (V + C + G)/3$$

where

V = Current viability (Where are we now relative to the viability goal?)
C = Core potential (What is the potential of each population to produce fish?)
G = Genetic legacy (Which populations warrant extra consideration because they are most representative of the historical fish characteristics we are intent on preserving?)

The index is the simple arithmetic average of each of the three elements. Each factor was standardized to a scale of 0-1 so that each contributes equal weight in the calculation, unless there were compelling reasons for elevating any individual factor. Note that the TRT also identified criteria based on catastrophic risks that are not incorporated into this population index. Catastrophic risks are better considered later in the scenario development process where the net effect of population-specific risks on the strata risk can be controlled by the choice of specific combinations of populations (e.g. populations that are not next the same volcano.)

To facilitate qualitative consideration of biological significance in future development of recovery scenarios, populations were sorted in descending order and separated into up to 3 categories where values were similar. Categories were labeled A, B, and C. Splits were made based on incremental changes in the sequence within each strata. Categories represent rank relative to other populations within a species. Thus, each category may not be represented in every strata.

Current **population viability** (V) was calculated for each population based on the following formula:

$$V = P / 3.0$$

where

P = Population persistence category based on TRT criteria (see preceding section):

0 = very high risk of extinction (0-40% persistence probability in 100 years).
1 = high risk of extinction (40-75% persistence probability in 100 years).
2 = medium risk of extinction (75-95% persistence probability in 100 years).
3 = low risk of extinction (95-99% persistence probability in 100 years).
4 = very low risk of extinction (>99% persistence probability in 100 years).

Population persistence scores were based on fish population data for abundance, productivity, juvenile emigrant numbers, spatial structure, diversity, and habitat. According to TRT recovery guidelines, a population persistence score of 3 would correspond to a viable population (i.e. recovery under ESA). Thus, dividing the population persistence score by 3 normalized this element to a scale from 0 to 1.25 with a score of 1.0 denoting a viable population. A score of >1.0 would give extra credit for populations recovered to even greater levels although, because of the way TRT scores are defined, a score of greater than 3.0 is practically very difficult to achieve for any given population. Population persistence scores were standardized so that they would be equally weighted with potential production and genetic character scores that also contributed to the biological significance index.

Use of all TRT population persistence criteria (abundance, productivity, juvenile emigrant numbers spatial structure, diversity, and habitat) to index population viability will facilitate mapping population conditions back to specific TRT viability factors that can be addressed with specific recovery actions. Note that Population Change Criteria (PCC) thresholds identified by NOAA Fisheries are not used directly in this approach but are implicit in the population persistence scores. We did not use the PCC viability thresholds because current population sizes used in the derivation of those thresholds are 4-year averages, are confounded in many populations by natural offspring of hatchery fish spawning in the wild, and may not be representative of long-term wild fish numbers.

Core potential (C) was calculated for each population based on the following formula:

$$C = NEQ_{PFC+} / mNEQ_{PFC+}$$

where

NEQ_{PFC+} = Potential population size if favorable habitat conditions are restored throughout the subbasin of origin [realized habitat capacity (equilibrium population size or Neq) inferred with EDT model from habitat data with universal restoration of Properly Functioning Conditions identified by NOAA Fisheries plus estuary habitat improvements].

$mNEQ_{PFC+}$ = Maximum potential population size projected for any population of a given species and run type under favorable habitat conditions.

This approach addresses core population criteria of the TRT with EDT-based data. Core populations were designated by the TRT based on a qualitative review of the available information and expert opinion. However, EDT results represent the best available data on historical and potential size of each population. Core population designations by the TRT closely correspond with the core population potential estimates from EDT, but EDT estimates also provide for incremental scaling of core population potential rather than the all-or-nothing nature of the core designation. This data-driven approach thus provides for more fine-scale evaluations. Standardization of core population potential estimates versus the potential for the largest population in the species and run results in values being scaled from 0 to 1 where 1 is the largest potential population in each stratum. Values can then be compared among strata to flag the largest potential populations. Values are comparable among strata in an absolute scale.

Genetic Legacy (G) was scored directly from TRT designations:

$$G = \{1, 0\}$$

where

1 = Genetic legacy population according to TRT
0 = Not a genetic legacy population according to TRT

The all-or-nothing nature of the TRT designation flags key stocks but does not capture intermediate increments of genetic characteristics that might provide further guidance for scenario development. We examined data-driven approaches to quantifying the degree of genetic legacy for each population but suitable alternatives were limited by the available data. We often have good recent data on hatchery release numbers and broodstock origins as well as anecdotal historical information. For instance, the NOAA fisheries escapement dataset documents annual hatchery fraction in the natural escapement where data is available. Similarly, the recent NOAA Fisheries status review classified the divergence from the wild for current hatchery stocks throughout the basin (1-4 scale where 4 is large divergence from wild).

However, we lack similar information for the historical period when hatchery effects were substantially greater.

5.1.4 Current Limiting Factors

5.1.4.1 Net Effect of Manageable Factors

We evaluated factors currently limiting Washington lower Columbia River salmon and steelhead populations based on a simple index of potentially manageable impacts. The index incorporated human-caused increases in fish mortality, changes in habitat capacity, and other natural factors of interest (e.g. predation) that might be managed to affect salmon productivity and numbers. We refer to this index approach as the AEIOU Index (Adult Equivalent Impacts Occurring Unconditionally).

To inform the development of recovery scenarios and strategies by technical and policy groups, we needed to inventory key factors and place them in perspective relative to each other. The AEIOU Index is a simple screening device to help educate a diverse audience and to provide general guidance for recovery decisions. The relative importance of each factor will guide both technical decisions on what combinations of recovery measures can prove effective and policy decisions on where to focus efforts and how to balance the responsibilities and costs of the effort. In popular parlance, the factors for salmon declines have come to be known as the 4-H's: hydropower, habitat, harvest, and hatcheries.

This approach represents the relative order of magnitude of key limiting factors. It does not constitute a fine-scaled mechanistic analysis of limiting factors and dynamics of every listed population. The question was not whether a factor might be responsible for a 50% or 55% impact with a confidence interval of 5% or 50%. Rather, we needed to know whether a factor represented a 5% or 50% or 90% impact.

Only the subset of factors we can potentially manage were included in the AEIOU Index – natural mortality factors beyond our control (e.g. naturally occurring ocean mortality) are excluded (Figure 5-2). For instance, tributary habitat changes, estuary habitat changes, fishing, hydro and hatchery effects are all obviously human impacts. Natural mortality in freshwater, the estuary, and the ocean that occurs independent of human effects was factored out. Predation by fish, birds, and marine mammals was included in the analyses, although it can only minimally be managed by humans, because of the widespread public interest in the magnitude of the predation effect relative to human factors.

The index was calculated as:

$$I_x = F_x / \sum F_x$$

where

I_x = relative impact of factor x.
F_x = proportional reduction in fish numbers as a result of factor x.
$\sum F_x$ = sum total of all proportional reductions selected for inclusion.

For instance, if we were concerned only with tributary habitat availability (e.g. 50% reduction due to development) and harvest (e.g. 25% average harvest rate), impacts would be calculated:

$I_{\text{tributary habitat}} = 0.50 / (0.50 + 0.25) = 0.67$ of the impacts of concern
$I_{\text{harvest}} = 0.25 / (0.50 + 0.25) = 0.33$ of the impacts of concern

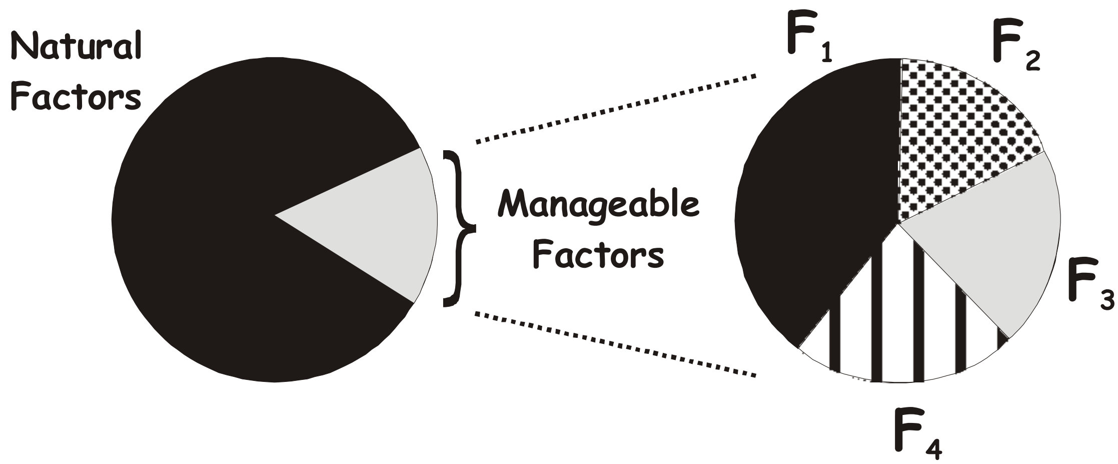

Figure 5-2. Manageable human factors affecting salmon mortality, productivity, and numbers represented as a portion of all factors and as their own pie.

With this index, the relative importance of any given factor decreases as additional factors are added. For instance, if we also included a 50% dam passage loss, the tributary habitat factor share of factors of concern is reduced from 0.67 to [0.50/ (0.50 + 0.25 + 0.50)] or 0.40. The factor effect is absolute (e.g. a 50% reduction) whereas the impact is relative to all factors of concern (0.67 becomes 0.40 when a new factor is added).

Factor level effects are most easily thought of as mortality rates. Our analyses include mortality associated with fishing, dam passage of juveniles and adult migrants, and predation by fish, birds, and marine mammals. Factor level effects also include other effects that reduce fish numbers and productivity including loss of tributary rearing capacity due to blockage and habitat degradation, reduced estuary survival due to habitat changes, and reduced natural population productivity due to interbreeding with less-fit hatchery fish.

The application of this index approach is limited to factors where we can reasonably quantify the effect. Other human-caused factors where data are sparse or effects are indirect may be overlooked or indistinguishable from natural productivity factors.

Factor level effects are described as unconditional adult equivalent effects that act independent of interactions with other factors. Unconditional factor effects are the proportional reduction in productivity or mortality of any given life stage. The reduction is relative to the potential number of that specific life stage rather than relative to numbers at an earlier or later life stage. Thus, the tributary habitat factor describes the reduction in smolt numbers relative to the number that would have been produced if habitat were unaffected, the harvest factor describes the reduction in adults relative to the number that would have survived in the absence of fishing, and so on.

Unconditional effects fairly represent factors that act on different parts of the life cycle. Each describes the proportional reduction associated with a given impact in the absence of the effects of other factors. Each factor level effect translates into an equivalent reduction in fish numbers or productivity (e.g. a 50% reduction in habitat quality reduces adult numbers by 50%

just as a 25% harvest mortality reduces adult numbers by 25%). Because factor effects are unconditional, the sum of all factor effects can be greater than 1.0 where many factors are included. However, the general absence of significant density-dependent mortality factors after the freshwater rearing stage makes this approach relatively robust:

$$N = B\ (1\text{-}M)\ (1\text{-}F_1)\ (1\text{-}F_2)\ (1\text{-}F_3) \ldots (1\text{-}F_n)$$

where

N = fish numbers
B = density dependent births (e.g. eggs produced by all natural spawners on average)
M = natural fish mortality throughout the life stage
$F_1, \ldots F_n$ = proportional reduction in fish numbers as a result of factor x for n factors.

In our special case where factor level effects may be considered density-independent, the net impact (Z) of a series of unconditional effects can be estimated:

$$Z = 1 - [(1\text{-}F_1)\ (1\text{-}F_2)\ (1\text{-}F_3) \ldots (1\text{-}F_n)]$$

Thus, the net impact (Z) represents the net impact of all factors considered. We compared net impacts of potentially manageable factors including human impacts among population to identify the proportional reduction in productivity and numbers (1-E) from a historical baseline that included no human impacts. In our simple example with a 50% habitat quality reduction and a 25% harvest mortality, the net impact would be 1-[(1-0.5)(1-0.25)] or a 62.5% reduction due to habitat and harvest impacts (only 37.5% of the historical number remains).

In developing descriptions of the relative impact of various factors for decline, we considered a variety of model-based quantitative approaches currently in application by the scientific community. However, most available alternatives were based on nuances that were difficult to grasp except by the quantitative scientists who developed them or were considerably more complicated than was necessary for our purposes. Examples included elasticity (Heppell 2000) or sensitivity analyses (Zabel 2003) based on matrix population models (Casell 2001). We also examined simple run reconstruction analyses based on juvenile or adult equivalents (e.g. LCFRB 2003). Both our simple index and more complicated life cycle modeling approaches are based on similar fish demographic data which is referenced in this report.

Estimated or assumed values for impact factors represent a reasonable first approximation and may be refined by more detailed evaluations of each individual factor. In many or most cases, we lack basin-specific fish population data. In some cases, current data are available but baseline historical data is almost invariably lacking. As a result, this exercise necessarily relied on a combination of inferences from other populations or areas, indirect analyses (EDT analysis of habitat data for instance), interpretations of our current scientific understanding of fish biology and system dynamics, or working hypotheses that are testable as part of recovery plan implementation. The diverse sources and nature of the information incorporated into this exercise makes it difficult to quantify the uncertainty in specific estimates. Clearly the uncertainty in specific point estimates is significant and caveats for their application are in order. Despite these limitations, these results are accurate representations of the available scientific information for the purpose of inventorying and generally describing the order-of-magnitude significance of potentially manageable factors for decline in a simple and intuitively understandable fashion.

The impact factors described in this assessment are a beginning rather than an end of the recovery scenario and strategy development process. As soon as the relative significance of various factors for decline is understood, the obvious next questions are: how big a change is needed to achieve recovery, what combinations of factor changes will be effective, and how difficult or costly will it be to affect each individual limiting factor by any given amount. A general sense of effective changes in any given factor can be gained by comparing specific impacts with increases in population growth rate or productivity identified by NOAA Fisheries. For instance, if population growth rates need to increase by 10% to reach desired population persistence probabilities, then we would need to decrease impact factors by an absolute value of 10% per year. More complex fish life cycle modeling approaches will be required to compound the effects of factors acting on different life stages, to estimate the net change in population productivity in response to combinations of recovery actions, and to relate changes to population viability. The basic mortality and productivity data incorporated into the simple AEIOU index provides some of the raw materials for these determinations.

5.1.4.2 Fisheries

Fishery assessments include estimates of total impacts on each population and the distribution of impacts among different fisheries. Impacts include direct harvest and catch-and-release mortality of all ocean and freshwater sport, commercial, and tribal fisheries. Extensive mortality data are available from Federal and State fishery regulatory agencies and Indian Tribes. These data are detailed for each species in earlier chapters of this Technical Foundation. Fisheries at the time of listing under the ESA are the basis for values used in the AEIOU Index of relative significance. Current rates are also reported where reductions have occurred. Population-specific estimates are typically inferred from species or stock-specific impacts rather than subbasin-specific estimates because pooled data provides more robust estimates. Historical trends in impacts are also summarized to illustrate past impacts that may have shaped current fish populations. Allocations of impacts among various ocean and freshwater fisheries will identify opportunities for considering the consequences of fishery-related recovery measures.

5.1.4.3 Hatcheries

To provide a conservative estimate of the potential for negative hatchery impacts on wild populations relative to other impact factors, this assessment evaluated: 1) intra-specific effects resulting from depression in wild population productivity that can result from interbreeding with less fit hatchery fish and 2) inter-specific effects resulting from predation of juvenile salmonids of other species. Fitness effects are among the most significant intra-specific hatchery risks and can also be realistically quantified based on hatchery fraction in the natural spawning population and assumed fitness of the hatchery fish relative to the native wild population. Predation is among the most significant inter-specific effects and can be estimated from hatchery release numbers by species. The index is:

$$F_{Hatchery} = F_{Intraspecific} + F_{Interspecific}$$

where

$F_{Intraspecific}$ = proportional reduction in natural productivity at equilibrium due to interbreeding of native and hatchery fish where hatchery fish are different.

$F_{Interspecific}$ = proportional reduction in natural productivity due to predation by larger hatchery smolts on smaller wild juveniles.

Intra-specific effects were estimated:

$$F_{Intraspecific} = p\ (1\text{-}f)$$

where

p = proportion of natural spawners that are of first generation hatchery origin.
f = relative productivity of native and hatchery fish (scale = 0-1).

This index assumed that equilibrium conditions have been reached for the hatchery fraction in the wild and for relative fitness of hatchery and wild fish. This simplifying assumption was necessary because more detailed information is lacking on how far the current situation is from equilibrium. In practice, actual differences in fitness of hatchery and natural fish at any given time depend on inherent differences in fitness and the degree and period of interaction (Lynch and O'Hely 2001). The index may thus over or underestimate the true current impact of hatchery spawners on wild fitness depending on past history. Current numbers of hatchery releases in each basin are also summarized to place associated risks in perspective.

The hatchery fitness index increases with the proportion of hatchery fish and decreases as hatchery fish are less productive than the wild fish (Figure 5-3). For instance, where hatchery fish comprise 50% of the natural spawners and fitness is 0, the hatchery impact index would be 0.50 (i.e. 50% reduction in productivity). Thus, in the case of random interbreeding of hatchery and wild fish, spawners would average 25% W:W, 50% W:H, and 25% H:H. The index results assumes 100% productivity of the W:W pairs, 50% productivity of the W:H pairs (average of 100% wild fitness and 0% hatchery fitness), and 0% productivity in the H:H pairs. In the alternative case where hatchery fish are equally fit with wild fish (f = 1.0), no hatchery fraction reduces wild productivity. Finally, where 50% of spawners are hatchery fish and hatchery fish fitness is only 50% of the wild fish, the hatchery impact index would be 0.25.

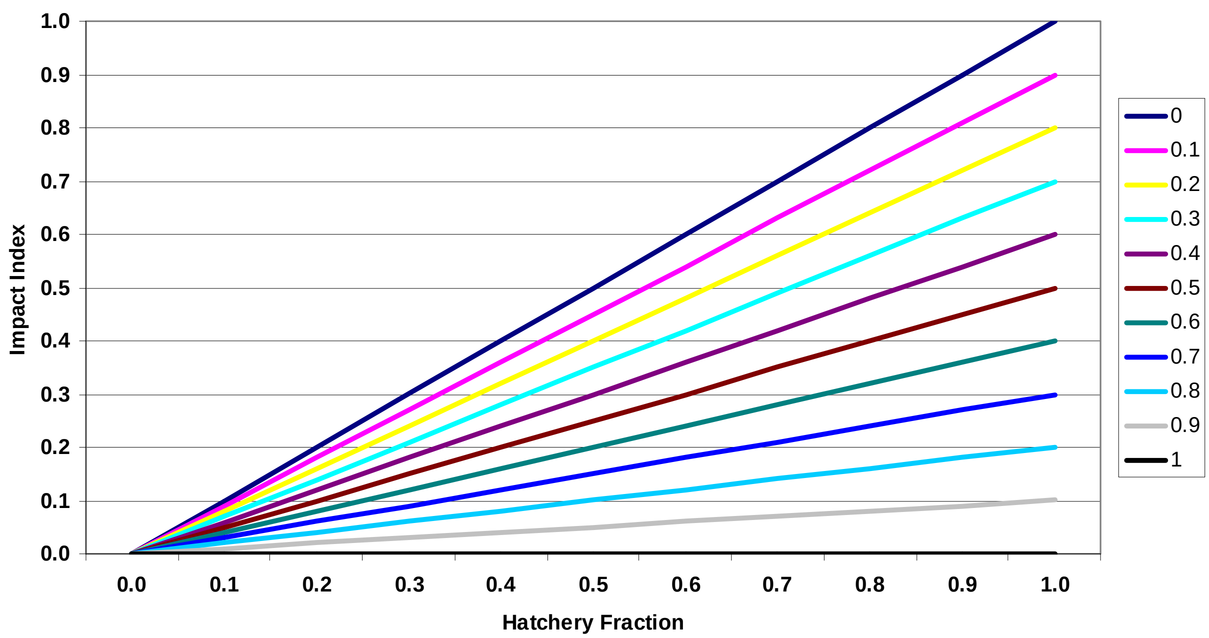

Figure 5-3. **Hypothetical effects of spawning by hatchery fish on wild population productivity relative to hatchery fraction and fitness of hatchery fish. Each line represents a different reduction in fitness (1-f) as depicted in the legend at right.**

Estimates of hatchery fraction were based on spawning ground survey data (typically CWT recoveries) where available. Where specific data were not available, approximate values are inferred from adjacent systems or available anecdotal information. Hatchery fractions are based on total hatchery and wild spawners that spawn *within the same period*. For instance, timing differences between hatchery and wild steelhead stocks often result in much less interbreeding than might be expected based on relative numbers of spawners (LCSCI 1998). These corrections were applied to steelhead populations where substantial differences in spawn timing occur but not chinook or chum where hatchery and wild spawn timing is similar.

Because population-specific fitness estimates are not available for most lower Columbia River populations, we applied hypothetical rates comparable to those reported in the literature and the nature of local hatchery program practices. Published information on relative fitness of hatchery and wild fish is limited (Berejikian and Ford 2003, TOAST 2004). Reisenbichler & McIntyre (1977) reported relative survival rates of Deschutes wild and Round Butte hatchery steelhead from egg to migration of 78% for H:H pairs, 80% for H:W pairs and 86% for W:W pairs. These differences are analogous to a 91% relative fitness of Round Butte hatchery fish which were only a few generations removed from the wild at the time of the study. In the Kalama River, Chilcote et al. (1986) reported a 28% relative fitness of Kalama wild summer and Skamania hatchery summer steelhead based on smolt production. This large reduction in fitness is likely driven by the high degree of domestication in the Skamania hatchery steelhead stock. Even larger differences become apparent where the hatchery stock is substantially different than the wild stock. For instance, a relative fitness of 0% was reported by Kostow et al. (2003) for a Skamania summer steelhead in Clackamas River relative to the native winter run. Finally, Oosterhout & Huntington (2003) assumed a 70% relative fitness for coastal Oregon hatchery and wild coho based on a recommended range of 0.5 to 0.9 by a technical scientific panel.

Increasing levels of domestication and interbasin transfers were assumed to reduce fitness consistent with hatchery categories identified by the salmon Biological Review Team based on historical data (Table 5-2). We generally assumed that hatchery fish are never as fit as the wild population even under the most enlightened hatchery practices. We described relative fitness values for each BRT category based on the literature review information above.

Interspecific hatchery effects were estimated:

$$F_{Interspecific} = (N_h)\ (r)$$

where

N_h = annual hatchery releases of salmon smolts with the potential to prey on the species of interest.

r = predation impact per hatchery fish released

For instance, intra-specific effects of 1 million potentially-predacious hatchery smolts would be 5% at an impact rate of 0.5% per 100,000 smolts.

Table 5-2. Fitness values assumed to correspond to hatchery categories reported by WCSBRT (2003).

Category	Description	Fitness
1	Hatchery population derived from native, local population; is released within range of the natural population from which it was derived; and has experienced only relatively minor changes from causes such as founder effects, domestication or non-local introgression.	0.9
2	Hatchery population was derived from local natural population, and is released within the range of the natural population from which it was derived, but is known or suspected to have experienced a moderate level of genetic change from causes such as founder effects, domestication or non-native introgression	0.7
3	The hatchery population was derived predominantly from other populations that are in the same ESU, but is substantially diverged from the local, natural populations(s) in the watershed in which it is released.	0.5
4	The hatchery population was predominantly derived from populations that are not part of the ESU in question; or there is substantial uncertainty about the origin and history of the hatchery population	0.3

Inter-specifies predation rates were assumed to be species-specific because of size and distribution differences. Natural fall chinook which rear in the lower portions of most subbasins are subject to predation by hatchery coho, winter steelhead, summer steelhead, and spring chinook that are typically reduced in lower to middle reaches (G. Norman, personal communication). Natural coho which rear in the lower and middle portions of most subbasins are also subject to predation by hatchery coho, winter steelhead, summer steelhead, and spring chinook. Chum salmon rear in lower subbasins and are subject to predation by winter steelhead, summer steelhead, and spring chinook which are released in March and April before juvenile chum have emigrated. Chum salmon are assumed not to be subject to significant predation by coho because coho are released in May after chum emigration. Inter-specific hatchery predation impacts on steelhead are not an issue because wild rearing areas of small juvenile steelhead are primarily in areas upstream of hatchery release sites. Impact rates were assumed to be 0.5% per 100,000 predators for fall chinook and chum, and 0.125% per 100,000 predators for coho (G. Norman, personal communication). Coho predation rates are less than those on the smaller fall chinook and chum. These rates provide reasonable magnitudes of predation impacts even in subbasins with large hatchery releases.

Fitness and predation effects of hatchery fish are two of a variety of potential positive and negative effects of hatchery and wild interactions. Because this exercise is primarily concerned with risks, the index did not consider the positive demographic benefits to natural spawner numbers from the additional hatchery fish and their progeny. Consideration of the numerical benefits of hatchery spawners to natural population numbers would substantially change the calculation, especially where wild and hatchery fitness are not substantially different. Nor does the index consider ecological interactions between hatchery and wild fish other than predation (e.g. competition, nutrient augmentation, or disease transfer). The net effect of direct and indirect ecological interactions may be either positive or negative and the occurrence and significance of each interaction is practically impossible to quantify.

5.1.4.4 Mainstem and Estuary Habitat

The effects of human-caused changes in mainstem and estuary habitat conditions on fish numbers are particularly difficult to quantify because of their complex and poorly understood nature. Salmon are affected during crucial smolt and adult migration stages. Mainstem and estuary areas also provide critical rearing habitats, particularly for spring chinook, fall chinook, and chum salmon which migrate to mainstem and estuary areas at pre-smolt life stages. Estimates of the impacts of human-caused changes in mainstem and estuary habitat conditions were generally based on changes in river flow, temperature, and predation as represented by EDT analyses for the NPCC Multispecies Framework Approach (Marcot et al. 2002).

In EDT analyses, estimates of the effects of human impacts on estuary habitat ($F_{\text{estuary habitat}}$) were represented as the difference in fish numbers between EDT results for Properly Functioning Conditions (NEQ_{PFC}) and Properly Functioning Conditions plus estuary restoration (NEQ_{PFC+}):

$$F_{\text{estuary habitat}} = (NEQ_{PFC+} - NEQ_{PFC}) / NEQ_{PFC+}$$

The hypothesized change in fish survival corresponding to estuary habitat changes was an explicit input of the EDT model calculations. EDT model results translate those changes into fish equivalents. This calculation is a reasonable approximation of the actual effect of estuary changes that could be more directly calculated with focused EDT analyses (L. Mobrand, personal communication 11/7/03).

Note that this definition potentially incorporates some indirect effects of dam construction and operation on fish habitat. Dam effects on fish productivity are evaluated separately where they can be distinguished from other factors.

5.1.4.5 Stream Habitat

Stream habitat assessments evaluate the effects of changes in subbasin watersheds and stream conditions on fish habitat quantity and quality. Analyses are based on analysis of stream habitat data using the Ecosystem Diagnosis and Treatment Model (EDT). EDT provides a systematic basis for inferring fish numbers from habitat conditions. Conditions for fish are described based on 46 habitat attributes. Habitat conditions are described for each homogenous stream reach used by the population of interest. The EDT model translates the 46 specific attributes into 17 “habitat survival factors” that represent hydrologic, stream corridor, water quality, and biological community characteristics related to habitat suitability and favorability for fish. Among other things, EDT then estimates average or equilibrium fish population sizes (N_{eq}) based on quantitative relationships between fish and limiting habitat factors distilled from an extensive literature review of salmon limiting factors.

EDT estimates are available in most subbasins for historical (template), current (patient), and “Properly Functioning” (PFC) habitat conditions. The historical/template condition is defined as pre-non-Native American/European influence and represents a hypothetical maximum. The current/patient condition represents the immediate past few years. PFC represents favorable habitat conditions for salmonids throughout the basin based on criteria identified in a general review of salmonid habitat requirements by NMFS (1996). The difference between historical and current conditions represents the degree of habitat degradation associated with subbasin development. The difference between current and PFC conditions represents the potential for improvement in fish numbers that might be achieved by restoring favorable habitat conditions throughout a given subbasin. PFC conditions are typically less than historical baseline. Current conditions are typically estimated from the available data including physical site surveys as well inferences from geospatial data, anecdotal evidence, and expert opinion. Detailed data on historical conditions are generally unavailable and so corresponding inputs are based on assumed conditions. The uncertainty of each EDT data input is also entered into the database that serves as an input for the model. Although data limitations frequently require significant assumptions in model inputs, our applications of results presumes that the model provides robust estimates of general habitat quantity and quality for fish, especially where results are used for relative comparisons of differences among areas or changes in conditions. More detailed descriptions and discussions of EDT methods, inputs, and results may be found in Technical Appendices (Volume VI) and Subbasin Chapters (Volume II).

Human impacts on stream habitat conditions were quantified based on differences in fish numbers between current and historical habitat conditions. The specific calculation also included corrections for estuary habitat effects that were contained in the historical EDT calculation:

$$F_{\text{tributary habitat}} = \{[NEQ_{\text{Historic}} * (1 - F_{\text{estuary habitat}})] - NEQ_{\text{Current}}\} / [NEQ_{\text{Historic}} * (1 - F_{\text{estuary habitat}})]$$

where

$F_{\text{tributary habitat}}$ = Proportional reduction in fish numbers as a result of human impacts on tributary habitat quantity and quality.

NEQ_{Historic} = Hypothetical average population size under pre-development habitat conditions in the subbasin and estuary.

NEQ_{Current} = Hypothetical average population size under current habitat conditions.

$F_{\text{estuary habitat}}$ = fish effects of human impacts on estuary habitat quality (see preceding section for definitions

The estuary correction was required because the difference between historical and current estimates produced by EDT is a function of both tributary and estuary habitat changes. However, we wanted to describe tributary and estuary changes separately because of the implications for recovery strategies and actions.

5.1.4.6 Dams

Dam impacts include access and passage effects. Access effects are the proportional reduction in available habitat where dams block passage. Access effects also include inundation of key spawning reaches in the lower portions of Bonneville Reservoir tributaries. Access impacts were based on historical EDT estimates of fish numbers produced from blocked areas versus the total produced in the subbasin. Access effects were included for the upper portions of the Lewis and Cowlitz basins. We also incorporated an assumed 20% reduction in productivity of chum salmon spawners in the mainstem below Bonneville Dam to account for flow effects during incubation. Loss of habitat availability because of dams was considered separate from other habitat impacts in tributaries.

Passage effects are loss rates of juveniles associated with attraction and collection efficiencies as well as direct mortality in all routes of passage. Passage effects were included for populations upstream from Bonneville Dam. Juvenile passage mortality rates at Bonneville Dam were assumed to average 10% for steelhead and chinook based on a review of the historical data in the technical foundation. Recent PIT tag studies suggest that average passage mortality rates may be less than 10% in some years. However, we hypothesize that fish from Washington tributaries in Bonneville Reservoir are more likely to pass via powerhouse 2 where guidance efficiencies and survival are less than the basin-wide average. Data were not sufficient to develop species or subbasin-specific estimates for spring chinook (yearling migrants) and fall chinook (subyearling migrants). We did not incorporate other dam-passage sources of mortality such as gas bubble disease and delayed passage effects. In the absence of specific data, chum salmon juvenile passage mortality was assumed to be twice that of steelhead and chinook because chum migrate at smaller, potentially more fragile sizes during early spring periods where spill measures to divert migrants from turbine passage are not in effect.

Adult passage mortality rates were assumed to be 5% for steelhead and 10% for spring and fall chinook based on conversion rate analyses for lower Columbia mainstem dams using dam counts, tributary escapement, and estimated harvest (*US v. Oregon* Technical Advisory Committee, unpublished data). Data are not available for chum salmon conversion rates but anecdotal information suggests rates are poor (G. Norman, personal communication). Consistent with this observation, we hypothesized a 50% adult upstream passage rate for chum.

Assumed dam passage mortality rates for juveniles and adults in this analysis are similar to those identified in the NPCC Multispecies Framework Approach (Marcot et al. 2002).

5.1.4.7 Predation

Predation impacts were based on approximate total mortality rates by northern pikeminnow, birds, and marine mammals. Detailed data on predation rates are limited, especially for marine mammals. However, anecdotal information is sufficient to generate order-of-magnitude estimates that place this impact in perspective relative to other impact factors.

Estimates of pikeminnow predation on juvenile salmonids are available for the Columbia River mainstem based on a series of studies by the Oregon Department of Fish and Wildlife and the Biological Resources Division of U.S. Geological Survey (see Limiting Factors Chapter of this Technical Foundation). Pikeminnow are of particular concern because they are among the most common salmonid predators among fish. Pikeminnow were estimated to consume approximately 9.7 million salmonids per year in the mainstem between Bonneville Dam and the estuary (Beamesderfer et al. 1996). Assuming approximately 200 million juvenile salmon and steelhead are available in the lower river per year, pikeminnow predation translates into a rate of 4.85%. Of this, approximately half occurs in the Bonneville Dam tailrace (Ward et al. 1995). The remainder was apportioned throughout the mainstem based on distance between the tributary mouth and the ocean. Bonneville Reservoir pikeminnow predation rate was calculated in a similar fashion (0.5%) with half assumed to occur in the Bonneville Forebay. The forebay rate was included for salmon populations originating in Bonneville Reservoir tributaries. Data were inadequate to estimate species differences in pikeminnow predation rates. Predation by other fishes, including walleye and hatchery salmonids, is not considered separate, hence, gets subsumed into estimated natural mortality. Walleye are substantially less abundant than pikeminnow. Data on predation rates by hatchery salmonids are not available. A pikeminnow sport reward fishery program has been implemented with a goal of reducing predation mortality on salmonids by 50%.

Tern predation on juvenile salmonids was based on Rice Island estimates by Roby et al. (1998) with corrections for recent translocation of the breeding colony to East Sand Island where salmonids are a less important diet item. We used a Rice Island predation mortality rate of 20% based on Roby et al.'s (1998) reported range of 10-30%. We estimated the corresponding East Sand Island predation rate at 9% by applying the difference in salmonid share of the diet at Rice (85%) and East Sand (40%) Islands (20% * 0.40/0.85). We hypothesize that tern predation accounts for the majority of the potentially manageable avian predation. Predation by other bird predators or birds in other areas is not addressed because of lack of data.

Estimates of marine mammal predation on adult salmonids were based on reported population sizes, literature values for daily ration, and reported diet shares of salmonids. Spring and fall predation mortality rates were estimated at 12% and 3% based on the following method. NMFS (2000) reported population sizes of about 2,000 in spring (1,700 harbor seals, 100-200+ sea lions). Fall population sizes were substantially less (1,000 total). Espenson (2003) quoted a daily ration equivalent to 1.2 – 2.0 salmon per day. To generate conservative minimum estimates we applied diet shares of 20% salmonids in spring NMFS (2000) and 50% salmonids in fall to an assumed daily ration equivalent to 1 salmon per day. Fall diet shares were assumed to be greater than spring because of fewer alternative foods and switching to more abundant salmon. This resulted in per predator consumption rates of 0.2 salmon per day in spring and 0.5 salmon per day in fall. Spring mortality rates were based on 24,000 salmon eaten versus average spring run sizes of 200,000 adult salmonids. Fall mortality rates were based on 30,000 salmon eaten versus average fall run sizes of 1,000,000 adult salmonids.

Because of the assumptions required by these calculations, our predation rates should be considered with caution. However, site-specific predation rates suggest that a 3-12% annual loss rate to marine mammals is reasonable. NMFS (2000) reported 250 salmon per year eaten by 10 sea lions at Willamette Falls based on direct observation. This translates into a 0.5% annual mortality rate based on a minimum Willamette Falls fish run of 50,000. Similarly, Espensen (2003) reported a 1.5% mortality rate by 100 sea lions in the Bonneville Dam tailrace.

5.1.5 Summary Assessments

For each species discussed below, we assessed the prospects and constraints for recovery by subjectively reviewing and synthesizing the results of the four methods described above. We did not finalize quantitative assessments; rather we chose to leave those final determinations for the recovery planning process. The information provided in the following sections sets the stage for recovery planners to determine the most appropriate summary method to support their decision-making.

5.2 Chinook

5.2.1 Current Viability

The Willamette/Lower Columbia Technical Recovery Team has identified 31 historical populations of chinook salmon in the Columbia River ESU. Washington accounts for seven of nine spring chinook (Figure 5-4), 13 of 20 early “tule” fall chinook (Figure 5-6), and 1 of 2 “bright” late fall chinook (Figure 5-5) populations in this ESU.

Current chinook population sizes and productivities are only a small fraction of historical numbers inferred with EDT from assumed pre-development habitat conditions (Table 5-3). EDT estimates of equilibrium numbers range from 100 to 8,900 for tule fall chinook and 0 to 3,000 for spring chinook under current equilibrium conditions. The Lewis bright chinook estimate is 9,400. Recent population estimates are typically less than EDT estimates, in part because of poor ocean survival periods. Historical chinook population sizes in Washington ranged from 300 to 38,300 based on EDT estimates. Back-of-envelope estimates by NOAA Fisheries yielded historical chinook population sizes in Washington of 6,200 to 48,400 based on presumed Columbia River run totals and subbasin habitat quantity. BOE estimates are typically greater than EDT estimates. We conservatively assume EDT estimates to be more accurate because they consider both habitat quantity and quality whereas the BOE estimates include only habitat quantity. EDT estimates are also independent of assumed total Columbia River run size and lower basin proportions upon which the BOEs are based.

Based on interim TRT population criteria, 100-year persistence probabilities for five populations are very low or already extinct (0-39%), 22 populations are low (40-74%), three populations are moderate (75-94%) , and only one population is relatively high (95-99%) (Table 5-4). All strata currently fall short of integrated TRT recovery criteria which specify an average persistence probability greater than 2.25 with at least two populations at high (>3.0) for each strata.

Population trends and extinction risks have been estimated for 12 chinook populations based on abundance time series data and two different models (NOAA Fisheries, unpublished data). Population trends were negative for 10 of 12 estimates (Table 5-4). Extinction risks averaged for both models were 90% or greater for 9 of 12 estimates. However, model-derived estimates appear overly pessimistic because of the limited time period of available data coincident with population declines following the ocean regime shift in the late 1970s as well as very large post-1983-84 El Niño returns which occur in the first half of most available time series. We assume that future estimates revised to consider cyclical patterns in ocean survival like those that have produced recent large returns will project much lower extinction risks. Differences between score-derived persistence probabilities and trend-derived extinction risks reflect different assumptions and uncertainties in these methods.

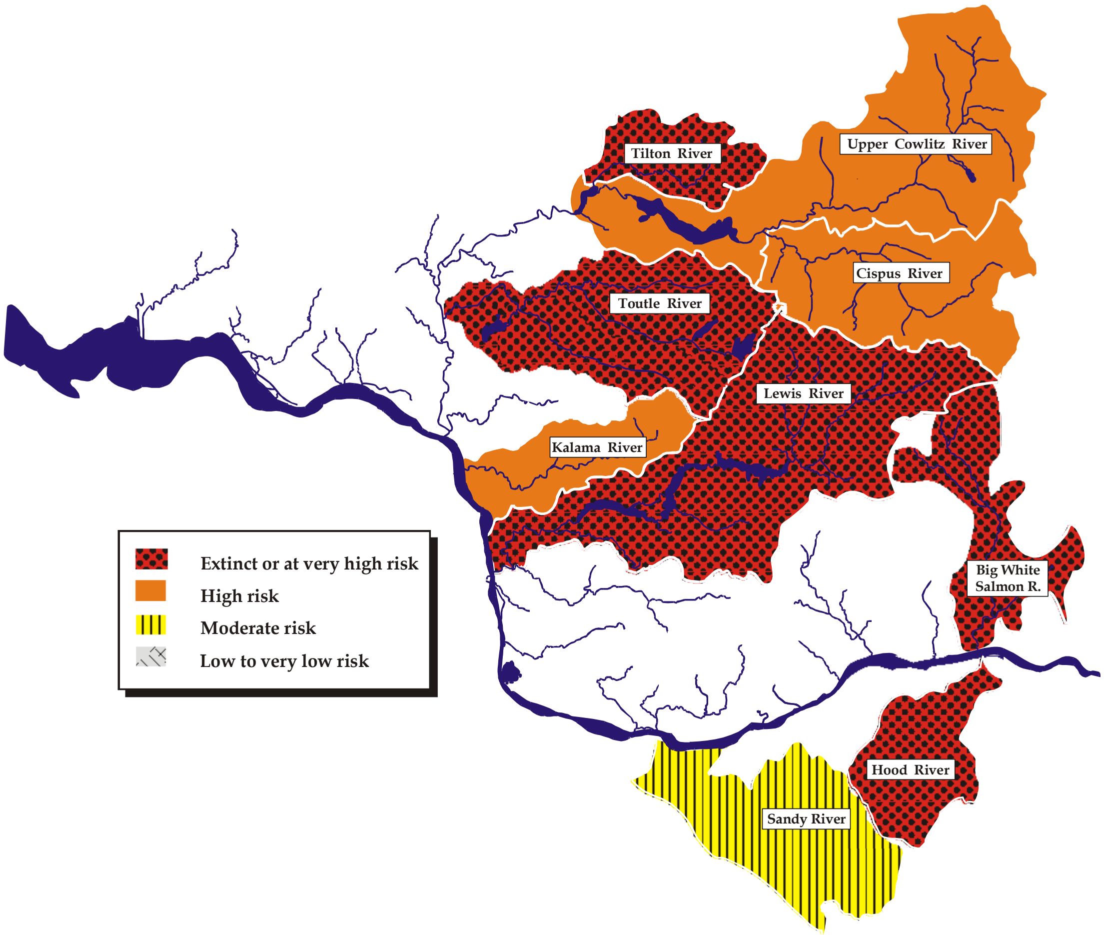

Figure 5-4. **Distribution of historical spring chinook populations among lower Columbia River subbasins.**

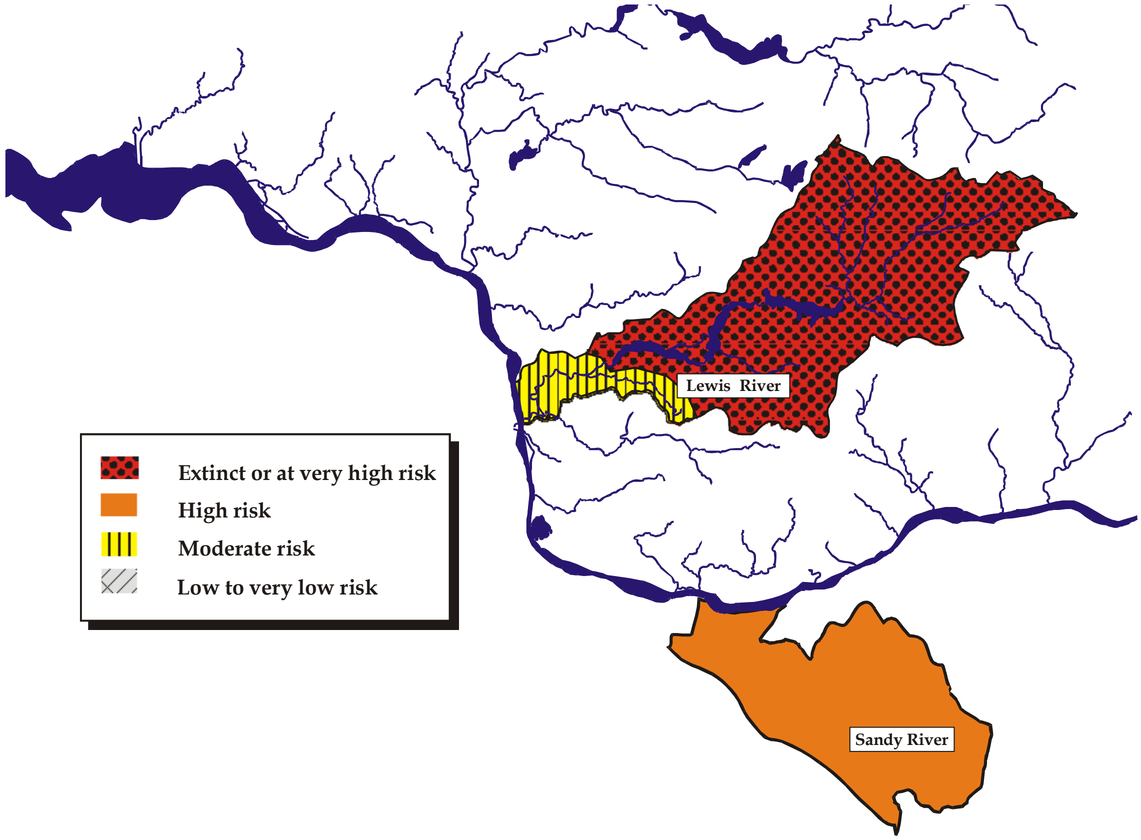

Figure 5-5. **Distribution of historic bright late fall chinook salmon populations among lower Columbia River subbasins. Extinction risks are based on viability scores.**

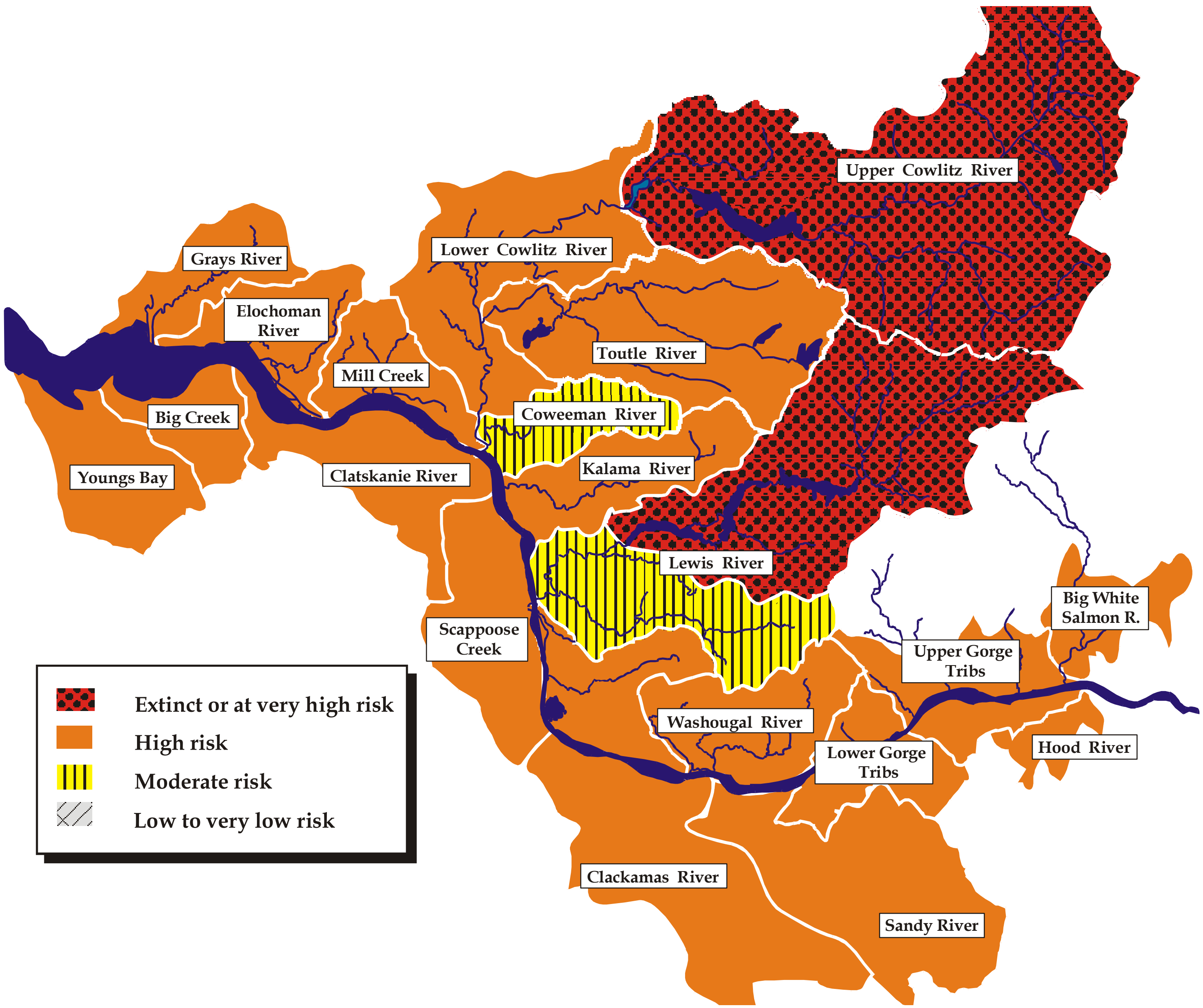
Upper Cowlitz River
Lower Cowlitz River
Grays River
Elochoman River
Toutle River
Mill Creek
Coweeman River
Big Creek
Kalama River
Youngs Bay
Clatskanie River
Lewis River
Big White Salmon R.
Scappoose Creek
Upper Gorge Tribs
Extinct or at very high risk
High risk
Moderate risk
Low to very low risk
Washougal River
Hood River
Lower Gorge Tribs
Clackamas River
Sandy River

Figure 5-6. **Distribution of historical tule fall chinook salmon populations among lower Columbia River subbasins. Extinction risks are based on viability scores.**

Table 5-3. Numbers and productivity for lower Columbia River chinook populations.

				EDT Equilibrium Population Size				BOE[8]	EDT Productivity			
Population	**Leg[1]**	**Core[2]**	**4-yr[3]**	**Current[4]**	**PFC[5]**	**PFC+[6]**	**Hist.[7]**	**Hist.**	**Current[4]**	**PFC[5]**	**PFC+[6]**	**Hist[7]**
Coast Fall												
Grays/Chinook			73	550	795	1,232	1,347	9,856	3.5	6.7	10.3	7.9
Eloch/Skam		1	140	2,060	2,934	4,547	4,564	10,834	3.4	7.0	10.9	11.8
Mill/Aber/Germ			250	1,365	2,072	3,211	4,855	7,526	3.4	6.3	9.8	11.9
Youngs Bay (OR)			--	--		--		14,446	--		--	--
Big Creek (OR)		1	--	--		--		8,458	--		--	--
Clatskanie (OR)			--	--		--		13,607	--		--	--
Scappoose (OR)			--	--		--		3,052	--		--	--
Cascade Fall												
Lower Cowlitz		1	602	8,873	20,865	33,210	38,767	29,847	5.9	11.0	17.6	14.5
Upper Cowlitz			0	5,056	11,046	17,851	28,015	24,325	2.3	3.7	4.9	8.6
Toutle		1	1,000	4,370	9,066	14,067	15,587	19,642	3.2	8.3	12.9	10.7
Coweeman	1		425	1,839	2,877	4,117	4,679	6,174	4.4	8.6	12.3	11.0
Kalama			1,192	1,581	2,367	3,230	3,766	6,640	3.3	6.9	9.5	8.7
Lewis/Salmon	1		235	1,472	2,637	3,903	4,639	30,057	3.4	6.9	10.2	8.7
Washougal			1,225	1,624	2,810	3,958	4,277	8,854	3.8	8.0	11.3	10.2
Clackamas (OR)		1	56	--		--		14,725	--		--	--
Sandy (OR)			208	--		--		15,145	--		--	--
Gorge Fall												
L Gorge			--	124	168	236	318	3,332	4.4	5.9	8.3	7.0
U. Gorge (Wind)		1	138	954	2,418	3,434	3,669	2,563	4.8	9.9	14.1	10.8
White Salmon		1	174			--		4,310	--		--	--
Hood (OR)			--	--		--		1,608	--		--	--
Cascade L Fall												
Lewis NF	1	1	6,493	9,388	10,134	16,612	18,359	19,460	11.2	12.3	20.1	14.7
Sandy (OR)	1	1	445	--		--		10,540	--		--	--
Cascade Spring												
Upper Cowlitz	1	1	365	3,019	6,426	8,117	21,750	38,318	2.5	4.5	5.6	15.8
Cispus		1	150	718	1,803	2,253	7,791	7,058	1.9	3.5	4.3	14.0
Tilton			150	869	3,176	3,897	5,436	13,321	1.9	7.2	8.9	15.1
Toutle			150	0	2,703	3,414	3,895	44,739	0.0	10.9	13.7	15.8
Kalama			105	413	756	945	6,077	15,125	1.8	3.1	3.8	17.2
Lewis NF		1	300	1,624	3,079	3,852	10,560	48,401	4.7	8.0	9.8	15.0
Sandy (OR)	1	1	2,649	--		--		28,605	--		--	--
Gorge Spring												
White Salmon				156	350	438	523		2.9	7.4	9.3	10.8
Hood (OR)		1	0	--		--		27,173	--		--	--

[1] *Genetic Legacy designation by the Technical Recovery Team. Genetic legacy populations represent unique life histories or are relatively unchanged by hatchery influences.*

[2] *Core population designation by Technical Recovery Team. Core populations were the largest historical populations and were key to metapopulation processes*

[3] *Recent 4-year average natural spawning escapements upon which PCC numbers are based (typically1997-2000 return years). Spawning escapements in 2002 and 2003 have generally been substantially greater than in the preceding years as these runs encountered much improved ocean survival conditions.*

[4] *Current number inferred with EDT from estimated and assumed habitat conditions.*

[5] *Estimate if habitat conditions are restored to "properly functioning" standards defined by NOAA Fisheries under current estuary conditions.*

[6] *Estimate if habitat conditions are restored to "properly functioning" standards defined by NOAA Fisheries and predevelopment estuary conditions are restored.*

[7] *Pre-development estimate inferred with EDT from assumed historical habitat conditions. Historical population sizes based on historical estuary and historical productivity based on current estuary.*

[8] *Back of envelope estimates of historical population sizes inferred from stream miles accessible and assumed total Columbia River run (NOAA Fisheries).*

Table 5-4. Estimated viability of lower Columbia River chinook.

Population	Leg[1]	Core[2]	Population Persistence Scores							Data		Extinction risk	
			A/P[3]	J[4]	S[5]	D[6]	H[7]	Net[8]	Prob.[9]	Years[10]	Trend[11]	Model 1[12]	Model 2[13]
Coast Fall													
Grays/Chinook			1	1	4	2	2	1.5	60%	1980-2000	0.86	1.00	1.00
Eloch/Skam		1	0.5	na	4	2.5	1.5	1.5	60%	1980-2000	0.86	1.00	0.98
Mill/Aber/Germ			1	na	3	2	2	1.4	50%	1980-2000	0.83	1.00	0.98
Youngs Bay (OR)			--	--	--	--	--	1.4	50%				
Big Creek (OR)		1	--	--	--	--	--	1.6	60%				
Clatskanie (OR)			--	--	--	--	--	1.6	60%				
Scappoose (OR)			--	--	--	--	--	1.4	50%				
Average								***1.45***	***60%***				
Cascade Fall													
Lower Cowlitz		1	1	na	4	2.5	1.5	1.6	60%	1980-2000	0.78	1.00	0.97
Upper Cowlitz			0	1	2	2	2	0.7	30%				
Toutle		1	1.5	na	3	2	1.75	1.3	50%				
Coweeman	1		2	na	4	3	2	2.3	80%	1980-2000	1.13	0.06	0.54
Kalama			1	na	4	2.5	2	1.9	70%	1980-2000	0.88	0.98	0.90
Lewis/Salmon	1		2	na	4	3	2	2.1	80%	1980-2000	0.97	0.97	0.80
Washougal			1	na	4	2	2	1.8	70%	1980-2000	0.89	0.99	0.90
Clackamas (OR)		1	--	--	--	--	--	1.4	50%	1967-1998	0.97	0.99	1.00
Sandy (OR)			--	--	--	--	--	1.7	60%				
Average								***1.64***	***60%***				
Gorge Fall													
L Gorge			1	1	3	2.5	2.5	1.2	50%				
U. Gorge (Wind)		1	1.5	1	2	2.5	2	1.3	50%				
White Salmon		1	*1.5*	*1*	*2*	*2.5*	*1.5*	1.3	50%	1980-2000	0.88	0.99	0.99
Hood (OR)			--	--	--	--	--	1.5	60%				
Average								***1.32***	***50%***				
Cascade L Fall													
Lewis NF	1	1	3	3	3	3.5	3	2.6	100%	1980-2000	0.95	0.68	0.60
Sandy (OR)	1	1	--	--	--	--	--	1.6	60%				
Average								***2.11***	***80%***				
Cascade Spring													
Upper Cowlitz	1	1	0.5	3	2	2	2	1.0	40%				
Lewis NF		1	0.5	na	2	2	2	0.9	40%				

Population	Leg[1]	Core[2]	Population Persistence Scores							Data	Trend[11]	Extinction risk	
			A/P[3]	J[4]	S[5]	D[6]	H[7]	Net[8]	Prob.[9]	Years[10]		Model 1[12]	Model 2[13]
Cispus		1	0.5	3	2	2	2	1.0	40%				
Kalama			0.5	na	4	1	1	1.1	40%				
Toutle			0	na	4	0	0	0.6	20%				
Tilton			0	na	0	0	0	0.1	0%				
Sandy (OR)	1	1	--	--	--	--	--	2.1	80%	1977-1998	1.09	0.00	0.03
Average								***0.98***	***40%***				
Gorge Spring													
White Salmon		1	0	na	0	0	0	0.0	0%				
Hood (OR)			--	--	--	--	--	0.5	20%				
Average								***0.2***	***10%***				

1 *Genetic Legacy designation by the Technical Recovery Team. Genetic legacy populations are relatively unchanged by hatchery influences or represent unique life histories.*
2 *Core population designation by Technical Recovery Team. Core populations were the largest historical populations and were key to metapopulation processes*
3 *Abundance and productivity rating by LCFRB biologists based on TRT criteria.*
4 *Juvenile emigration number rating by LCFRB biologists based on TRT criteria.*
5 *Spatial structure rating by LCFRB biologists based on TRT criteria.*
6 *Diversity rating by LCFRB biologists based on TRT criteria.*
7 *Habitat rating by LCFRB biologists based on TRT criteria.*
8 *Weighted average of population attribute scores. LCFRB and TRT scores are averaged.*
9 *Persistence probability corresponding to net population score (interpolated from corresponding persistence ranges).*
10 *Available abundance data time series upon which trend and extinction risk analyses by NOAA Fisheries were based.*
11 *Trend slope estimated by NOAA Fisheries based on abundance time series (median annual growth rate or λ).*
12 *Probability of extinction in 100 years (PE 100) estimated from abundance time series by NOAA Fisheries using Dennis-Holmes model.*
13 *Population projection interval extinction risks (PPI E) estimated from abundance time series by NOAA Fisheries using Population Change Criteria model.*

5.2.2 Recovery Planning Ranges

Planning ranges are presented in Table 5-5. Minimum abundance values vary among populations from 1,400 to 6,500 based primarily on PCC viability targets. Maximum planning numbers range from 1,400 to 33,200 based on subbasin potentials estimated with EDT for Properly Functioning Conditions.

Consistent with their current threatened population status, recent natural spawning escapements have almost universally averaged less than the lower viability bound of the planning range. Recent numbers have averaged fewer than 300 naturally-produced fish in 7 of 12 Washington tule fall chinook populations and 5 of 6 Washington spring chinook populations. Recent natural escapements of Washington lower Columbia chinook exceeded an average of 1,000 fish only in Toutle fall, Kalama fall, Washougal fall, and Lewis late fall populations. Recent average escapements (through 2000) were typically less than EDT equilibrium numbers based on current stream habitat conditions, primarily because of recent poor ocean survival cycles.

Substantial improvements in productivity are required in most populations to reach viable levels. Tule Fall chinook populations were estimated to require a 14% to 67% improvement in productivity to reach a level of high viability. Bright fall chinook were estimated to require a 6% improvement in productivity to reach a level of high viability. No estimates are available for spring chinook although the scale of limiting factors suggests that several-fold improvements in productivity will be required to reach viability.

5.2.3 Population Significance

The population significance index provides a simple sorting device to group populations in each strata based on current viability, core potential and genetic legacy (Table 5-6). Based on this index, no Coast Strata Washington tule chinook population is distinguishable from any other. The Elochoman population was designated as a core population by the TRT but production potential is not substantially greater than other strata populations. Current viability of all Coast strata tule populations is low and no unique genetic legacies have been identified. In the Cascade fall tule strata, Coweeman, Lewis/Salmon, and Lower Cowlitz sort to the top by virtue of their current viability, genetic legacy designations, or large historical population sizes. A low tier includes the Toutle, Kalama, Washougal, and Upper Cowlitz tule populations. No Gorge tule population is distinguished from the others by this index. Late fall bright chinook are represented by only one Cascade population each in Washington and Oregon.

Upper Cowlitz and Cispus spring chinook rank at the top of the Cascade strata by virtue of their genetic legacy designation and high historical core potential. A low tier includes Lewis, Toutle, Kalama, and Tilton populations.

Table 5-5. Population abundance and productivity planning ranges for lower Columbia River chinook populations.

Population	Recent Avg. no.	Abundance range Viable	Potential	Current viability	Current Prod.	Productivity range Viable	Potential	Productivity Improvement Increments Contrib	High	V high	Max
Coast Fall											
Grays/Chinook	73	1,400	1,400	**Low**	0.87	1.15	5.69	16%	33%	295%	558%
Eloch/Skam	140	1,400	4,500	**Low**	0.86	1.15	7.90	17%	35%	429%	824%
Mill/Aber/Germ	250	2,000	3,200	**Low**	0.83	1.15	5.76	19%	38%	314%	591%
Youngs Bay (OR)	--	1,400	2,800	**Low**	--	--	--	--	--	--	--
Big Creek (OR)	--	1,400	2,800	**Low**	--	--	--	--	--	--	--
Clatskanie (OR)	--	1,400	2,800	**Low**	--	--	--	--	--	--	--
Scappoose (OR)	--	1,400	2,800	**Low**	--	--	--	--	--	--	--
Cascade Fall											
Lower Cowlitz	602	3,900	33,200	**Low**	0.78	1.13	6.72	22%	45%	402%	760%
Upper Cowlitz	0	1,400	10,800	**V Low**	0.00	1.15	3.94	--	--	--	--
Toutle	1,000	1,400	14,100	**Low**	0.69	1.15	8.32	34%	67%	445%	824%
Coweeman	425	3,000	4,100	**Med**	1.13	1.14	5.51	7%	14%	201%	388%
Kalama	1,192	1,300	3,200	**Low**	0.88	1.12	6.49	14%	27%	332%	637%
Lewis/Salmon	235	1,900	3,900	**Med**	0.97	1.15	5.34	9%	19%	235%	451%
Washougal	1,225	5,800	5,800	**Low**	0.89	1.12	5.77	13%	26%	288%	550%
Clackamas (OR)	56	1,400	2,800	**Low**	--	--	--	--	--	--	--
Sandy (OR)	208	1,400	2,800	**Low**	--	--	--	--	--	--	--
Gorge Fall											
L Gorge (Ham.)	--	1,400	2,800	**Low**	0.92	1.15	5.14	13%	25%	248%	471%
U. Gorge (Wind)	138	1,400	2,400	**Low**	0.90	1.11	6.38	12%	24%	316%	608%
White Salmon	174	1,600	3,200	**Low**	--	--	--	--	--	--	--
Hood (OR)	--	1,400	2,800	**Low**	--	--	--	--	--	--	--
Cascade L Fall											
Lewis NF	6,493	6,500	16,600	**Med**	0.95	1.00	3.03	3%	6%	113%	220%
Sandy (OR)	445	5,100	10,200	**Low**	--	--	--	--	--	--	--
Cascade Spring											
Upper Cowlitz	365	2,800	8,100	**Low**	--	--	--	--	--	--	--
Cispus	150	1,400	2,300	**Low**	--	--	--	--	--	--	--
Tilton	150	1,400	2,800	**V Low**	--	--	--	--	--	--	--
Toutle	150	1,400	3,400	**V Low**	--	--	--	--	--	--	--
Kalama	105	1,400	1,400	**Low**	--	--	--	--	--	--	--
Lewis NF	300	2,200	3,900	**V Low**	--	--	--	--	--	--	--
Sandy (OR)	2,649	2,600	5,200	**Med**	--	--	--	--	--	--	--
Gorge Spring											
White Salmon	0	1,400	2,800	**V Low**	--	--	--	--	--	--	--

Hood (OR)	0	1,400	2,800	**V Low**	--	--	--	--	--	--	--

Notes

1. *Recent average numbers are observed 4-year averages or assumed natural spawning escapements. Data typically is through year 2000.*
2. *Abundance planning range refer to average equilibrium escapement numbers at viability as defined by NOAA's Population Change Criteria and potential as defined by WDFW's Ecosystem Diagnosis and Treatment assessments under properly functioning habitat and historical estuary conditions..*
3. *Current viability is based on Technical Recovery Team viability rating approach.*
4. *Current and planning range productivity values are expressed in terms of intrinsic rate of population increase. Estimates are available only where data exists to EDT and population trend assessments.*
5. *Productivity improvement increments indicate needed improvements to reach contributing, high, very high, and maximum levels of population viability or potential.*

Table 5-6. Biological significance categories of lower Columbia chinook populations based on current viability, core potential, and genetic legacy considerations.

	Raw ratings				Normalized values				
Population	Gen.[1]	Core[2]	Poten.[3]	Viab.[4]	Viab.[5]	Poten.[6]	Gen.[7]	Index[8]	Rank[9]
Coast Fall									
Eloch/Skam		1	4,500	1.5	0.49	0.14	0.00	0.21	C
Mill/Aber/Germ			3,200	1.4	0.47	0.10	0.00	0.19	C
Grays/Chinook			1,200	1.5	0.48	0.04	0.00	0.17	C
Clatskanie (OR)			2,800	1.6	0.53	0.08	0.00	0.21	--
Big Creek (OR)		1	2,800	1.6	0.52	0.08	0.00	0.20	--
Youngs Bay (OR)			2,800	1.4	0.45	0.08	0.00	0.18	--
Scappoose (OR)			2,800	1.4	0.45	0.08	0.00	0.18	--
Cascade Fall									
Coweeman	1		4,100	2.3	0.76	0.12	1.00	0.63	A
Lewis/Salmon	1		3,900	2.1	0.70	0.12	1.00	0.60	A
Lower Cowlitz		1	33,200	1.6	0.54	1.00	0.00	0.51	A
Toutle		1	14,100	1.3	0.45	0.42	0.00	0.29	C
Kalama			3,200	1.9	0.63	0.10	0.00	0.24	C
Washougal			4,000	1.8	0.61	0.12	0.00	0.24	C
Upper Cowlitz			10,800	0.7	0.23	0.33	0.00	0.18	C
Clackamas (OR)		1	2,800	1.4	0.45	0.08	0.00	0.18	--
Sandy (OR)			2,800	1.7	0.57	0.08	0.00	0.22	--
Gorge Fall									
White Salmon		1	3,200	1.3	0.43	0.10	0.00	0.17	C
U. Gorge (Wind)		1	2,400	1.3	0.44	0.07	0.00	0.17	C
L Gorge (Hamil.)			2,800	1.2	0.41	0.08	0.00	0.16	C
Hood (OR)			2,800	1.5	0.48	0.08	0.00	0.19	--
Cascade L Fall									
Lewis NF	1	1	16,600	2.6	0.88	1.00	1.00	0.96	A
Sandy (OR)	1	1	10,200	1.6	0.53	0.61	1.00	0.72	--
Cascade Spring									
Upper Cowlitz	1	1	8,100	1.0	0.34	1.00	1.00	0.78	A
Cispus	1	1	2,300	1.0	0.34	0.44	1.00	0.59	A
Lewis NF		1	3,900	0.9	0.31	0.48	0.00	0.26	C
Toutle			3,400	0.6	0.20	0.42	0.00	0.21	C
Kalama			900	1.1	0.38	0.11	0.00	0.16	C
Tilton			2,800	0.1	0.02	0.35	0.00	0.12	C
Sandy (OR)	1	1	5,200	2.1	0.70	0.64	1.00	0.78	--
Gorge Spring									
White Salmon		1	2,800	0.0	0.01	0.35	0.00	0.12	C
Hood (OR)			2,800	0.5	0.15	0.35	0.00	0.17	--

[1] *Genetic Legacy designation by the Technical Recovery Team. Genetic legacy populations are relatively unchanged by hatchery influences or represent unique life histories.*

[2] *Core population designation by Technical Recovery Team. Core populations were the largest historical populations and were key to metapopulation processes.*

[3] *Potential fish numbers based on upper end of planning range (typical value if accessible habitat restored to favorable, albeit not pristine, conditions based on EDT results for properly functioning conditions plus restored estuary).*

[4] *Provisional ratings by LCFRB consultants and WDFW staff based on TRT standards.*

[5] *Normalized population persistence score used in biological significance ranking.*

[6] *Normalized core population potential used in biological significance ranking.*

[7] *Genetic legacy score used in biological significance ranking.*

[8] *Average of now, potential and genetic scores.*

[9] *Strata ranking based on average population score.*

5.2.4 Current Limiting Factors

5.2.4.1 Net Effect of Manageable Factors

The net effects of quantifiable human impacts and potentially manageable predation on chinook salmon translates into an 85-100% reduction in productivity among Washington lower Columbia populations (Figure 5-7). Thus, current fish numbers are only 0-15% of what they would be if all manageable impacts were removed.

No single factor accounts for the majority of the reduction in fish numbers. Loss of habitat quantity and quality in the tributaries and the estuary account for significant shares of the impact. Dam construction constitutes the largest single impact for upper Cowlitz and Lewis populations of spring chinook and tule fall chinook. Dam construction is also a significant factor for Gorge chinook populations. Fishing is significant for fall chinook but less so for spring chinook. Hatchery effects vary among populations but are generally less than 20% of the total impact. Predation is among the lesser impacts we considered. Component chinook salmon impact factors and indices are shown in Table 5-7 and Table 5-8. The effects of each manageable limiting factor are discussed in the sections below.

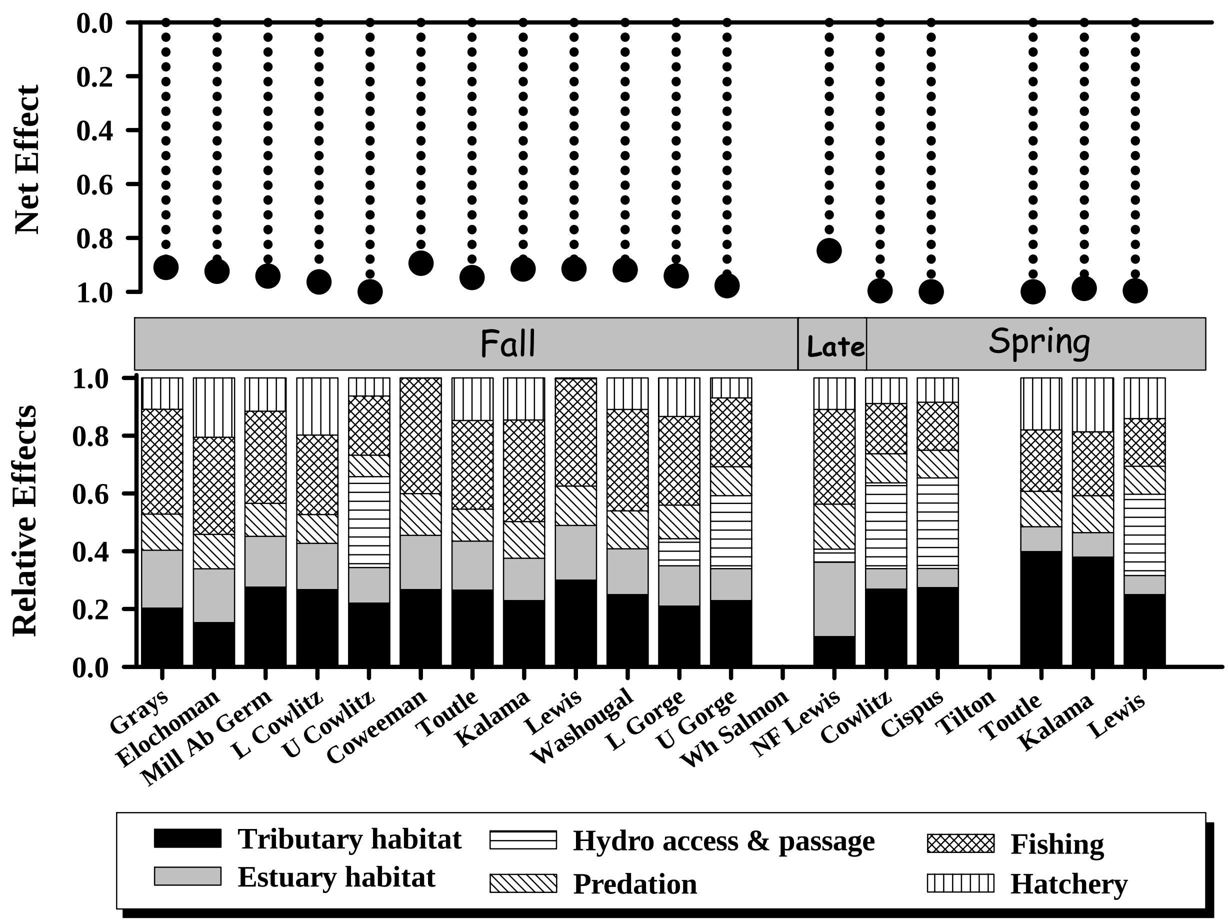

Figure 5-7. **Net effect and relative contribution of potentially manageable impact factors on chinook salmon in Washington lower Columbia River subbasins. Net effect is the approximate reduction from historical fish numbers as a result of manageable factors included in this analysis.**

Table 5-7 . Fall “tule” chinook salmon impact factors and index.

	Grays	Eloch	M/A/G	L Cowlitz	U Cowlitz	Coweem.	Toutle	Kal.	Lew/Sal	Wash.	L Gorge	U Gorge	Wh Sal
Inputs													
Neq Current	550	2,060	1,365	8,873	5,056	1,839	4,370	1,581	1,472	1,624	124	954	na
Neq PFC	795	2,934	2,072	20,865	11,046	2,877	9,066	2,367	2,637	2,810	168	2,418	na
Neq PFC+	1,232	4,547	3,211	33,210	17,851	4,117	14,067	3,230	3,903	3,958	236	3,434	na
Neq Historical	1,347	4,564	4,855	38,767	28,015	4,679	15,587	3,766	4,639	4,277	318	3,669	na
Hydro habitat loss	0.000	0.000	0.000	0.000	1.000	0.000	0.000	0.000	0.000	0.000	0.200	0.500	0.500
Dam passage mortality (juveniles)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.100	0.100
Dam passage mortality (adults)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.100	0.100
Predation mortality (juveniles)	0.200	0.206	0.209	0.211	0.211	0.211	0.211	0.212	0.215	0.220	0.223	0.251	0.251
Predation mortality (adults)	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030
Fishing	0.650	0.650	0.650	0.650	0.650	0.650	0.650	0.650	0.650	0.650	0.650	0.650	0.650
Hatchery fraction	0.370	0.690	0.470	0.670	0.670	0.000	0.900	0.670	0.000	0.570	0.950	0.170	0.220
Hatchery category	3	3	3	2	2	0	2	2	0	2	2	4	3
Hatchery fitness	0.5	0.5	0.5	0.7	0.7	0.0	0.7	0.7	0.0	0.7	0.7	0.3	0.5
Other hatchery species	190,000	1,050,000	0	5,319,500	0	20,000	850,000	1,380,000	115,000	620,000	0	1,420,000	0
Impacts (p reduction)													
Tributary habitat	0.367	0.300	0.564	0.636	0.708	0.437	0.565	0.427	0.530	0.465	0.451	0.631	na
Estuary habitat	0.355	0.355	0.355	0.372	0.381	0.301	0.355	0.267	0.324	0.290	0.291	0.296	na
Hydro habitat loss	0.000	0.000	0.000	0.000	1.000	0.000	0.000	0.000	0.000	0.000	0.200	0.500	0.500
Dam passage	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.190	0.190
Predation	0.224	0.230	0.233	0.235	0.235	0.235	0.235	0.236	0.239	0.243	0.246	0.273	0.273
Fishing	0.650	0.650	0.650	0.650	0.650	0.650	0.650	0.650	0.650	0.650	0.650	0.650	0.650
Hatchery fitness	0.185	0.345	0.235	0.201	0.201	0.000	0.270	0.201	0.000	0.171	0.285	0.119	0.110
Hatchery interspecies	0.010	0.053	0.000	0.266	0.000	0.001	0.043	0.069	0.006	0.031	0.000	0.071	0.000
Total (unconditional)	1.790	1.933	2.037	2.359	3.175	1.624	2.118	1.850	1.749	1.851	2.123	2.730	na
Impact index													
Tributary habitat	0.205	0.155	0.277	0.269	0.223	0.269	0.267	0.231	0.303	0.251	0.212	0.231	na
Estuary habitat	0.198	0.184	0.174	0.158	0.120	0.185	0.168	0.144	0.185	0.157	0.137	0.108	na
Hydro access/passage	0.000	0.000	0.000	0.000	0.315	0.000	0.000	0.000	0.000	0.000	0.094	0.253	na
Predation	0.125	0.119	0.114	0.099	0.074	0.144	0.111	0.127	0.136	0.132	0.116	0.100	na
Fishing	0.363	0.336	0.319	0.276	0.205	0.400	0.307	0.351	0.372	0.351	0.306	0.238	na
Hatchery	0.109	0.206	0.115	0.198	0.063	0.001	0.148	0.146	0.003	0.109	0.134	0.070	na

Table 5-8. Late Fall “bright” and spring chinook salmon impact factors and index.

	bright LF-Lewis	spring Cowlitz	Spring Cispus	spring Tilton	spring Toutle	spring Kalama	spring Lewis	spring Wh Salmon
Inputs								
Neq Current	9,388	3,019	718	--	0	413	1,624	--
Neq PFC	10,134	6,426	1,803	--	2,703	756	3,079	--
Neq PFC+	16,612	8,117	2,253	1,400	3,414	945	3,852	--
Neq Historical	18,359	21,750	7,791	--	3,895	6,077	10,560	--
Hydro habitat loss	0.069	0.900	1.000	1.000	0.000	0.000	0.902	0.900
Dam passage mortality (juveniles)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.100
Dam passage mortality (adults)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.100
Predation mortality (juveniles)	0.215	0.211	0.211	0.211	0.211	0.212	0.215	0.251
Predation mortality (adults)	0.030	0.120	0.120	0.120	0.120	0.120	0.120	0.120
Fishing	0.500	0.530	0.530	0.530	0.530	0.530	0.530	0.530
Hatchery fraction	0.130	0.90	0.90	0.90	0.90	0.90	0.90	1.000
Hatchery category	1	2	2	2	3	3	3	4
Hatchery fitness	0.9	0.7	0.7	0.7	0.5	0.5	0.5	0.3
Other hatchery species	3,070,000	0	0	0	0	0	0	0
Impacts (p reduction)								
Tributary habitat	0.162	0.825	0.885	--	1.000	0.915	0.808	--
Estuary habitat	0.390	0.208	0.200	1.000	0.208	0.200	0.201	--
Hydro habitat loss	0.069	0.900	1.000	1.000	0.000	0.000	0.900	0.900
Dam passage	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.190
Predation	0.239	0.306	0.306	0.306	0.306	0.307	0.309	0.341
Fishing	0.500	0.530	0.530	0.530	0.530	0.530	0.530	0.530
Hatchery fitness	0.013	0.270	0.270	0.270	0.450	0.450	0.450	0.700
Hatchery interspecies	0.154	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Total (unconditional)	1.525	3.039	3.190	--	2.494	2.402	3.197	--
Impact index								
Tributary habitat	0.106	0.271	0.277	--	0.401	0.381	0.253	--
Estuary habitat	0.256	0.069	0.063	--	0.084	0.083	0.063	--
Hydro access/passage	0.045	0.296	0.313	--	0.000	0.000	0.281	--
Predation	0.156	0.101	0.096	--	0.123	0.128	0.097	--
Fishing	0.328	0.174	0.166	--	0.213	0.221	0.166	--
Hatchery	0.109	0.089	0.085	--	0.180	0.187	0.141	--

5.2.4.2 Fisheries

Fishing impacts on lower Columbia wild spring chinook averaged 53% at listing and has since been reduced to 22%. Current mortality is incidental to target fisheries for fin-clipped Willamette, lower Columbia, and upper Columbia hatchery fish. Additional harvest of wild spring chinook occurs in the ocean incidental to target fisheries for Alaskan, Canadian, Columbia River Hatchery, and California Hatchery chinook stocks (Figure 5-8). The exploitation rate of spring chinook has fluctuated over time, ranging from 20 to 65%. The current exploitation of hatchery spring chinook is similar to historical exploitation rates, while wild spring chinook exploitation is considerably lower than historical rates.

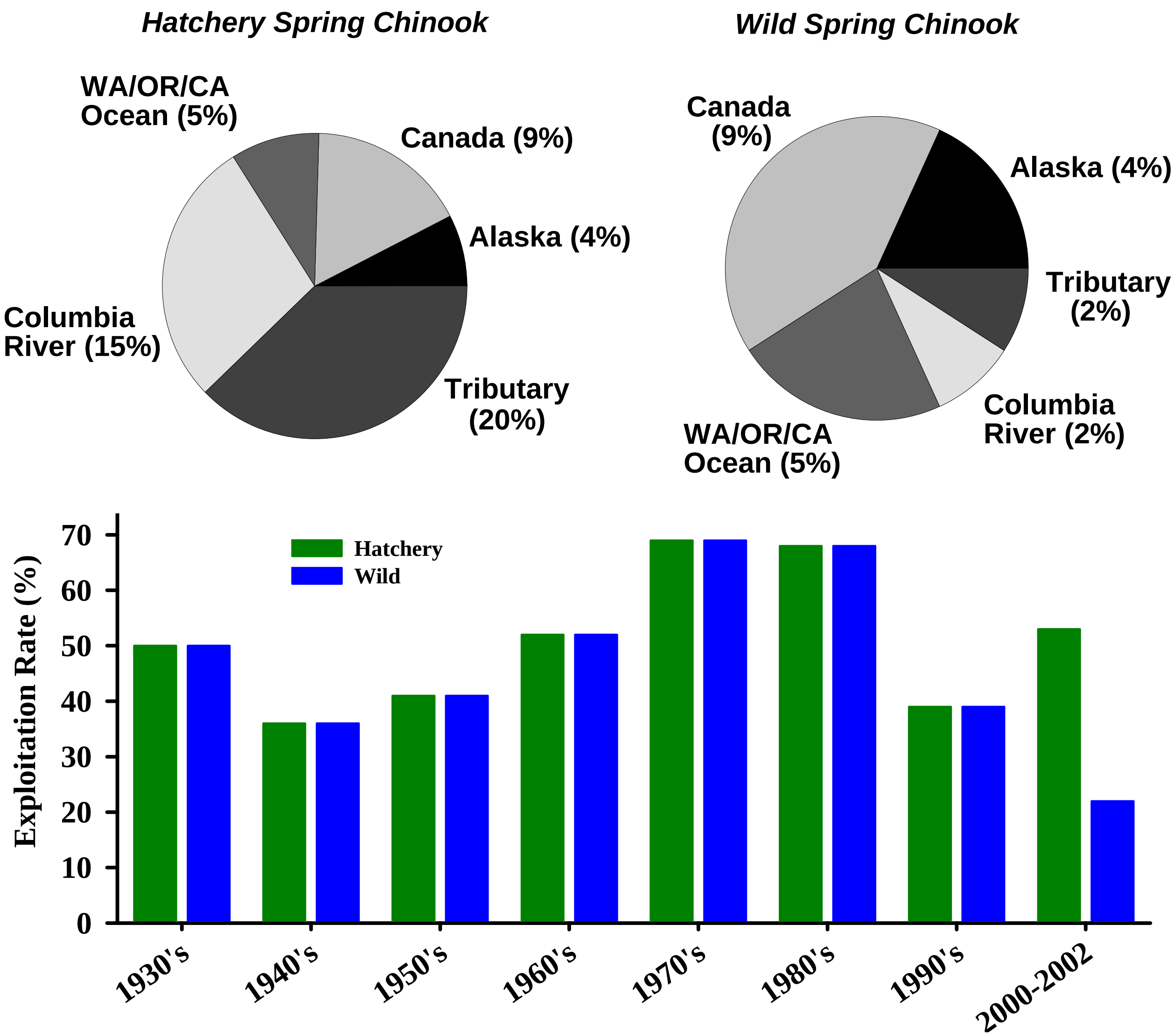

Figure 5-8. Approximate spring chinook fishery exploitation rates over time and allocation of current exploitation rates among fisheries

Fishing impact limits on lower Columbia fall chinook averaged 65% for tules and 50% for brights at listing. Recent rates have been less than 49% for tules and less than 40% on brights. Columbia basin fisheries targeting upriver bright and Columbia hatchery chinook account for a third to half of total fishing rates, with the remainder primarily distributed between Oregon, Washington, Canada, and Alaska ocean fisheries targeted on Alaska, Canadian, Columbia River hatchery, and California hatchery chinook (Figure 5-9). Current fishing rates on fall chinook are approximately half their historical average. For instance, chinook fishing rates remained fairly constant at 70-80% through the 1980s and 1990s, but have declined to approximately 40%.

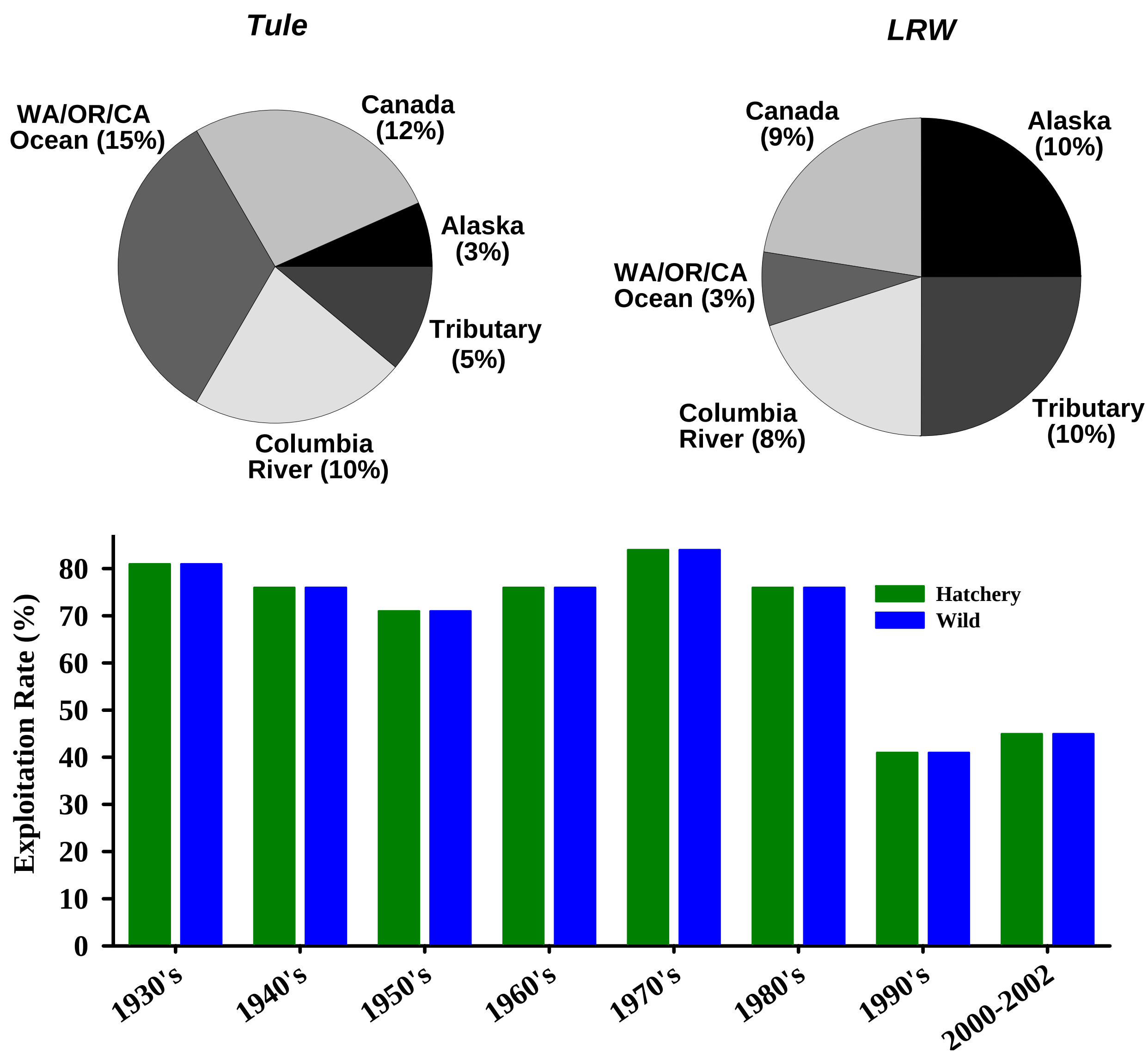

Figure 5-9. **Approximate fall chinook fishery exploitation rates over time and allocation of current exploitation rates among fisheries. Time series is for tule fall chinook.**

5.2.4.3 **Hatcheries**

Hatchery influence is significant for most Washington lower Columbia chinook populations (Table 5-9). Spring chinook populations are primarily naturally spawning hatchery fish. Hatchery releases of spring chinook smolts range from 0 to 1.267 million. Hatchery fractions of adults generally average 90% with reintroduction attempts in the upper Cowlitz basin relying entirely on hatchery stock. Current spring chinook hatchery broodstock are primarily derived from local populations and moderately affected by hatchery practices (category 2) or derived from other populations within the same ESU (category 3). The indexed potential for negative impacts of hatchery spawners on wild population fitness was estimated to range from 27 to 45%. However, the high incidence of hatchery spawners suggests that the fitness of natural and hatchery fish is now probably quite similar and natural populations could collapse without continued hatchery subsidy under current habitat conditions. Inter-specific hatchery predation impacts on juvenile fall chinook range from 0% in basins without significant releases of coho, steelhead or spring chinook to 15 and 27% in the Lewis and Cowlitz basins where large hatchery programs are underway.

For Tule fall chinook, potential fitness impacts of hatchery fish range from 0 to 34% (Table 5-9). Hatchery fish do not contribute to Coweeman and Lewis tule chinook populations, hence, their genetic legacy designation by the Technical Recovery Team. Hatchery fractions generally exceed 50% for other Cascade and Coast tule populations but are less among Gorge populations. Current hatchery releases of tule chinook smolts range from 0 to 5 million per year per subbasin although historical releases were greater. Hatchery broodstocks are primarily derived from local populations and moderately affected by hatchery practices (category 2) for Cascade populations. Many chinook salmon hatchery programs were developed initially from out-of-subbasin or multiple stocks but this practice has been largely discontinued. In most cases, brood stock mixing was limited to a few stocks and performed only during the initial years of establishing the hatchery program. After the hatchery program had been established, brood stock collection came from adults returning to the hatchery facility within the basin, aside from minimal outside brood stock usage during years of hatchery shortfalls. Most hatchery programs now use adults returning to the hatchery facility for brood stock; future genetic risks for outside brood stock usage appear minimal. In contrast, Coast and Gorge tule chinook hatchery programs primarily originated from other populations within the same ESU (category 3) because natural populations in those areas were typically small.

Bright fall chinook in the Lewis basin have included a small fraction of hatchery fish until recently when the program was discontinued. The potential for fitness impacts was estimated to be less than 1% because of the low incidence of hatchery fish and the native, local brood source (category 1).

Supplementation has not been the goal of most chinook hatchery programs; these are intended to mitigate for losses of chinook by providing fish for harvest opportunities. The exceptions are a fall chinook hatchery program at the Sea Resources Hatchery on the Chinook River, spring chinook, coho, and winter steelhead programs at the Cowlitz hatcheries for the upper Cowlitz River, and chum programs at the Grays River hatchery for the Grays River and at the Washougal Hatchery for lower Gorge chum. A spring chinook, coho, and winter steelhead supplementation program is expected to be developed at the Lewis River hatcheries for the upper Lewis.

Table 5-9. Presumed reductions in wild population fitness as a result of natural hatchery spawners and survival as a result of interactions with other hatchery species for Washington lower Columbia River chinook populations.

Population	Annual releases[a]	Hatchery fraction	Fitness category	Assumed fitness	Fitness impact	Interacting releases[j]	Interspecies impact
Coast Fall							
Chinook/ Grays	107,500[b]	0.37	3	0.5	0.18	190,000	0.01
Eloch/Skam	2,000,000	0.69	3	0.5	0.34	1,050,000	0.05
Mill/Aber/Germ	0[c]	0.47	3	0.5	0.24	0	0
Cascade Fall							
Lower Cowlitz	5,000,000	0.67	2	0.7	0.20	5,319,500	0.27
Upper Cowlitz	0	0.67	2	0.7	0.20	--	--
Toutle	2,500,000	0.90	2	0.7	0.27	850,000	0.04
Coweeman	0[d]	0.00	0	--	0.00	20,000	0
Kalama	5,000,000	0.67	2	0.7	0.20	1,380,000	0.07
Lewis/Salmon	0	0.00	0	--	0.00	115,000	0.01
Washougal	4,000,000	0.57	2	0.7	0.17	620,000	0.03
Gorge Fall							
L Gorge	0[e]	na	2	0.7	na	0	0
U. Gorge (Wind)	0[f]	0.17	4	0.3	0.12	1,420,000	0.07
White Salmon	0	0.22	3	0.5	0.11	0	0
Cascade L Fall							
Lewis NF	0[g]	0.13	1	0.9	0.01	3,070,000	0.15
Cascade Spring							
Cowlitz	1,267,000[h]	0.90	2	0.7	0.27	--	--
Cispus	--[h]	0.90	2	0.7	0.27	--	--
Tilton	0	0.90	2	0.7	0.27	--	--
Toutle	0	0.90	3	0.5	0.45	--	--
Kalama	500,000	0.90	3	0.5	0.45	--	--
Lewis NF	1,050,000[i]	0.90	3	0.5	0.45	--	--
Gorge Spring							
White Salmon	0[j]	1.00	4	0.3	0.70	--	--

[a] *Annual release goals.*

[b] *Number refers to fall chinook hatchery program underway to restore a naturally producing population in the Chinook River. The Grays River fall chinook hatchery program stopped releasing smolts in 1998; hatchery returns were expected to significantly diminish starting with the 2002 return.*

[c] *Abernathy hatchery stopped releasing fall chinook in 1995; hatchery returns were expected to significantly diminish starting with the 1999 return.*

[d] *Hatchery fall chinook have not been released in the Coweeman River basin since the early 1980s and tagged hatchery strays have not been recovered during spawning surveys since that time.*

[e] *There are no hatchery fall chinook programs in the lower gorge tributaries; fall chinook from the Spring Creek NFH were released in Hamilton in 1977.*

[f] *There are no hatchery fall chinook programs in the Wind River basin. Fall chinook were historically produced at the Carson NFH and released in the basin, however, production shifted to spring chinook in 1981.*

[g] *The Lewis River fall chinook hatchery program was discontinued in 1986. There is no hatchery fall chinook program in Salmon Creek.*

[h] *300,000 fingerling spring chinook from Cowlitz Trout Hatchery are released annually in an attempt to restore the upper Cowlitz population. An additional 967,000 yearlings are released in the lower Cowlitz from Cowlitz Salmon Hatchery.*

[i] *Current releases are in the lower Lewis. Reintroduction into the upper Lewis is also under consideration in the hydroelectric re-licensing process.*

[i] *No hatchery spring chinook are released into the White Salmon. However, 1,000,000 and 1,420,000 hatchery spring chinook are released in the Little White Salmon and Wind rivers, respectively.*

[j] *Includes steelhead, coho and spring chinook for fall chinook. Not applicable for spring chinook.*

5.2.4.4 Tributary Habitat

EDT analyses suggest that stream degradation has substantially reduced the habitat potential for chinook salmon in all Washington lower Columbia River subbasins where analyses have been completed (Figure 5-10). Declines in habitat quality and quantity for chinook salmon have reduced current productivity potential to 0-76% and current equilibrium fish numbers to 0-50% of the historical template. Substantial stream habitat improvements would be necessary to reach optimum conditions (i.e. PFC) for chinook salmon in most subbasins. Restoration of optimum habitat quality would be expected to increase habitat capacity by 100 to 24,000 adult chinook per subbasin.

Fall chinook salmon rely on the lower and middle mainstem reaches of large streams and rivers. Channel instability, low habitat diversity, and sedimentation consistently limit habitat suitability for fall chinook in these areas. Spring chinook use the upper basins of large river systems. Many of these upstream areas continue to provide suitable habitat for spring chinook but dams limit access in the Lewis and Cowlitz basins. More detailed descriptions of stream habitat conditions and effects on fish in each subbasin may be found in Volume II of the Technical Foundation.

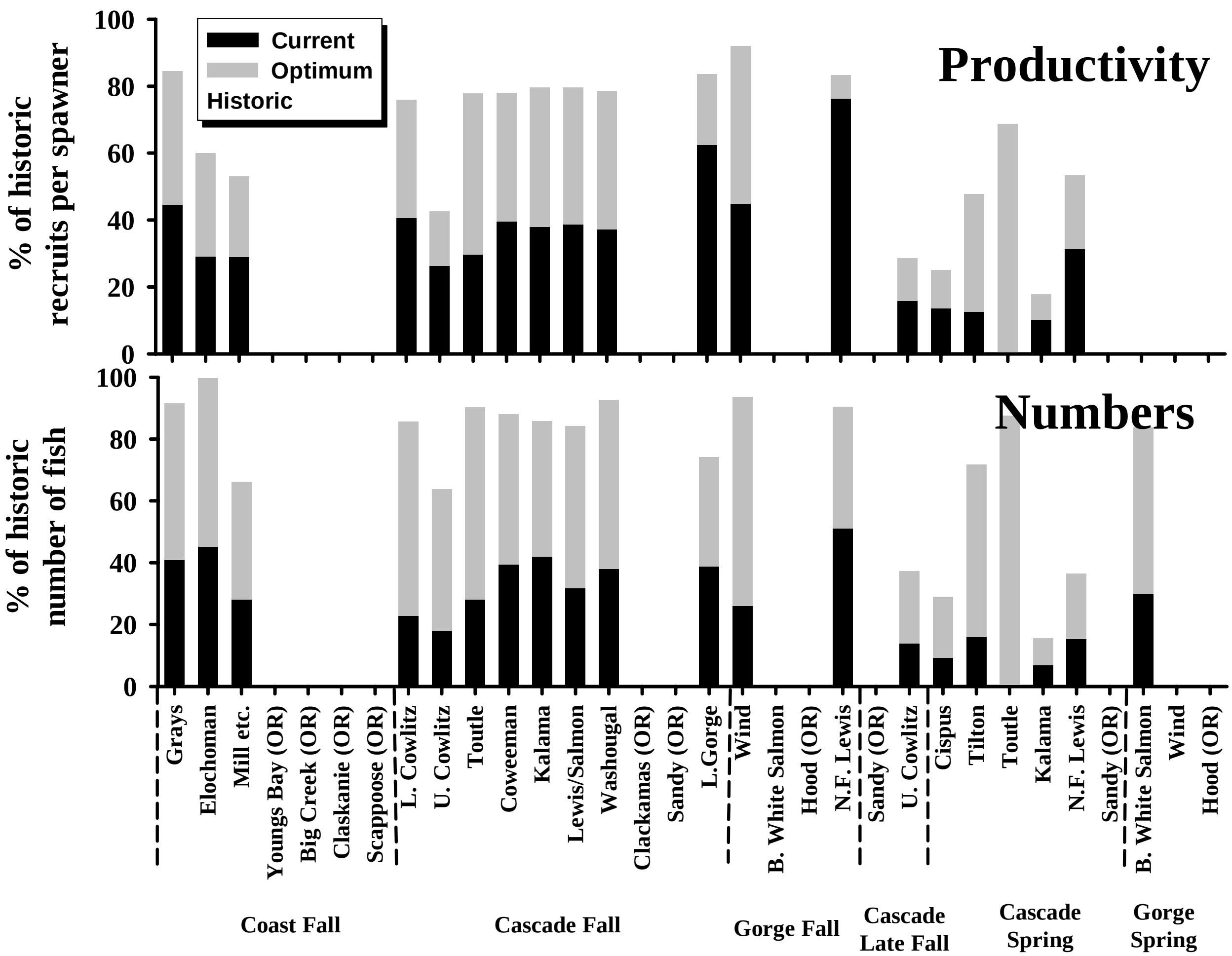

Figure 5-10. Current, optimal, and historical subbasin productivity and equilibrium numbers inferred for chinook salmon from stream reach habitat conditions using EDT.

5.2.4.5 Dams

Dam impacts on Washington lower Columbia chinook were estimated to range from 0 to 100% (Figure 5-11). Dams on the Cowlitz have inundated or blocked access to over 90% of the spring chinook and 45% of the fall chinook habitat based on EDT assessments. On the North Fork Lewis River, 90% of the spring chinook and 7% of the fall chinook habitat has been inundated or blocked. Passage mortality at Bonneville Dam was assumed to average 10% for juveniles and an additional 10% for adults based on a synthesis of the available literature. Bonneville Dam inundated approximately half of the fall chinook habitat on the Wind and White Salmon rivers. Assessments also included an assumed 20% reduction in fall chinook productivity in the Columbia River mainstem as a result of Bonneville Dam operations. Dam operations in the Cowlitz and Lewis River have similar potential to affect downstream habitat conditions for chinook but the significance of this impact is unknown.

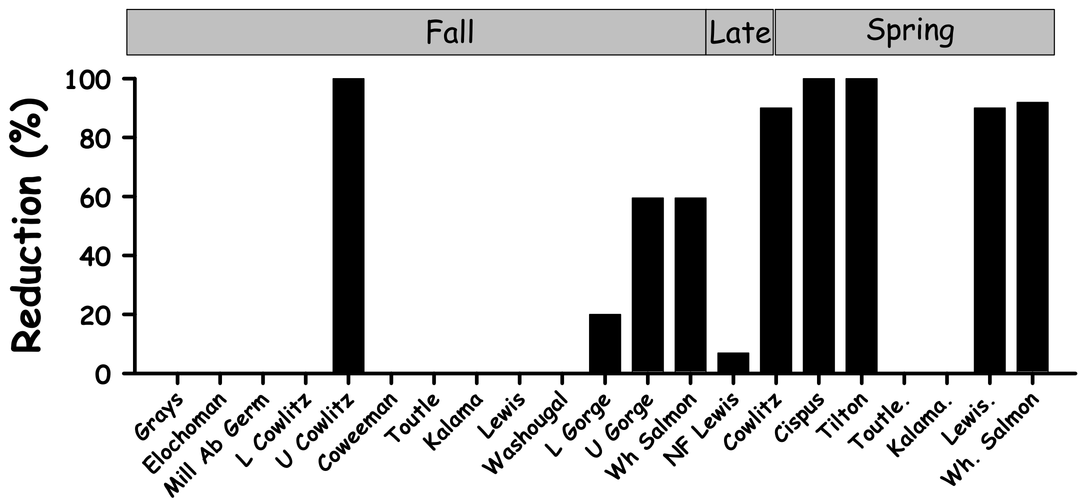

Figure 5-11. Assumed dam impacts on Washington lower Columbia chinook populations.

5.2.4.6 Mainstem and Estuary Habitat

Mainstem and estuary habitat impacts were estimated to account for approximately a 30-40% reduction in productivity of fall chinook which migrate as subyearlings and a 20% reduction for the primarily spring-migrating spring chinook.

5.2.4.7 Predation

Predation mortality rates of juvenile fall and spring chinook by pikeminnow and terns at the time of ESA listing was assumed to average 20% to 25% depending on travel distance from the subbasin to the ocean. Pikeminnow and tern management is projected to reduce salmonid predation by approximately 50%. Tern predation is almost entirely an artifact of recently established colonies on dredge spoil islands in the estuary but the current rate (9%) is less than half that observed prior to downstream translocation of the Rice Island colony (20%). Pikeminnow predation was greatest for populations that originate in Bonneville Reservoir tributaries, and must pass the pikeminnow gauntlet in Bonneville Dam forebay and tailrace, and then travel the entire 145 miles from Bonneville to the Estuary. Predation rates by seals and sea lions on adult chinook were assumed to be four times greater on spring migrants (12%) than on fall migrants (3%).

5.2.5 Summary Assessment

1. Human activities, including fishing, hatchery operation, alteration of stream, river, and estuary habitats, hydropower development and operation, and potentially manageable predation have collectively reduced productivity of spring and fall chinook populations to 0-15% of historical levels. Recovery efforts will require significant improvements in multiple areas because no single factor accounts for the majority of the reduction in fish numbers.

2. Current fishing impacts on spring chinook are modest and provide limited opportunities for increasing their numbers through additional regulation of fisheries. Higher fishing impacts on fall chinook salmon provide some opportunity for increasing numbers through additional fishery constraints. Impacts have been reduced since listing. Additional reductions would likely require changes in ocean and freshwater fisheries. Fall chinook impacts are widely distributed among U.S. ocean, Canada ocean, Alaska ocean, and Columbia River fisheries. Since Lower Columbia fall chinook comprise only a small portion of the catch in many fisheries, additional constraints for their protection will forgo harvest of larger numbers from healthy wild and especially hatchery populations. Intensive fishery management processes provide significant opportunities for limiting fishing risks by tailoring annual harvests to fish availability.

3. Reduced productivity of wild populations as a result of interbreeding with potentially less-fit hatchery fish is among the most significant of hatchery concerns for wild stock recovery although these negative effects are at least partially offset by the demographic benefits of additional spawners. Potential negative impacts increase with the proportion of hatchery spawners and the genetic and phenotypic disparity between wild and hatchery fish. Potential fitness impacts among Washington lower Columbia fall chinook populations range from 0 to 34%. Potential impacts are substantially greater among spring chinook populations (27-70%) where hatchery fish comprise the majority of many remaining runs. Inter-specific hatchery predation impacts on juvenile fall chinook range from 0% in basins without significant releases of coho, steelhead or spring chinook to 15 and 27% in the Lewis and Cowlitz basins where large hatchery programs are underway.

4. Stream habitat conditions significantly limit chinook salmon in all Washington lower Columbia River subbasins where EDT analyses have been completed. Substantial stream habitat improvements would be necessary to reach optimum conditions (i.e. PFC) in all subbasins. The significance of stream habitat suggests that recovery may not be feasible without substantial improvements in habitat quantity and quality.

5. Estuary and mainstem habitats are critical to chinook salmon life history with assumed habitat impacts greater for fall chinook (30-40% reduction) than spring chinook (20% reduction).

6. Hydropower development in tributaries is currently the most significant factor limiting spring chinook populations and effective recovery may not be feasible without effective passage measures at Cowlitz and Lewis dams. Hydro impacts on Gorge fall chinook populations are also significant but can be only partially addressed by passage improvements. For instance, inundation of limiting habitat for fall chinook in Bonneville tributaries may constrain restoration of large and productive natural spawning in those areas.

5.3 Chum Salmon

5.3.1 Current Viability

The Willamette/Lower Columbia Technical Recovery Team identified 16 historical populations of chum salmon in the Columbia River ESU (Figure 5-12). Eight occur only in Washington, six occur only in Oregon, and two are shared between states. Significant populations exist only in the Grays River and the lower Columbia River Gorge tributaries and mainstem.

Current chum population sizes and productivities are much less than historical numbers inferred with EDT from assumed pre-development habitat conditions (Table 5-10). EDT estimates of equilibrium numbers range from 400 to 7,900 under current conditions. Actual population estimates are typically much less than EDT estimates. Historical chum population sizes in Washington ranged from 6,600 to 479,800 based on EDT estimates. Back-of-envelope estimates by NOAA Fisheries yielded historical chinook population sizes in Washington of 15,000 to 295,000 based on presumed Columbia River run totals and subbasin habitat quantity. BOE estimates are typically greater than EDT estimates but relative differences among populations are similar.

TRT population criteria indicate that 100-year persistence probabilities are very low or already extinct (0-39%) for 12 populations, low (40-74%) for 3 populations, and moderate (75-94%) for 1 population. No chum population was judged to be currently at a high probability of persistence. All strata currently fall short of recovery criteria which specify an average persistence probability greater than 2.25 with at least 2 populations at high (>3.0) for each strata.

Population trends and extinction risks have been estimated for 2 chum populations based on abundance time series data and two different models (NOAA Fisheries, unpublished data). Population trends were negative for 1 of the 2 estimates (Table 5-11). Extinction risks averaged for both models were 50-60% per population. Differences between score-derived persistence probabilities and trend-derived extinction risks reflect different assumptions and uncertainties in these methods.

5.3.2 Recovery Planning Ranges

Planning ranges are available only for Washington populations (Table 5-12). Minimum values vary among populations from 1,100 to 4,300 according to Population Change Criteria numbers with larger current populations generally requiring greater minimum numbers to reach viable levels. Maximum planning range numbers range from 2,200 to 135,700 based on subbasin potentials estimated with EDT for Properly Functioning Conditions. Consistent with their current threatened population status, recent natural spawning escapements have universally averaged less than the low viability bound of the planning range. Recent numbers have averaged fewer than 300 naturally produced fish in 8 of 10 chum populations that occur in Washington. Recent poor ocean survival cycles have reduced recent average escapements to less than EDT equilibrium numbers based on current stream habitat conditions.

Substantial improvements in productivity are required in most populations to reach viable levels. Chum populations in the Grays River and lower Gorge were estimated to require an 8% to 12% improvement in productivity to reach a level of high viability. Other chum populations would require a 25% to 2000% increase in productivity to reach viable levels.

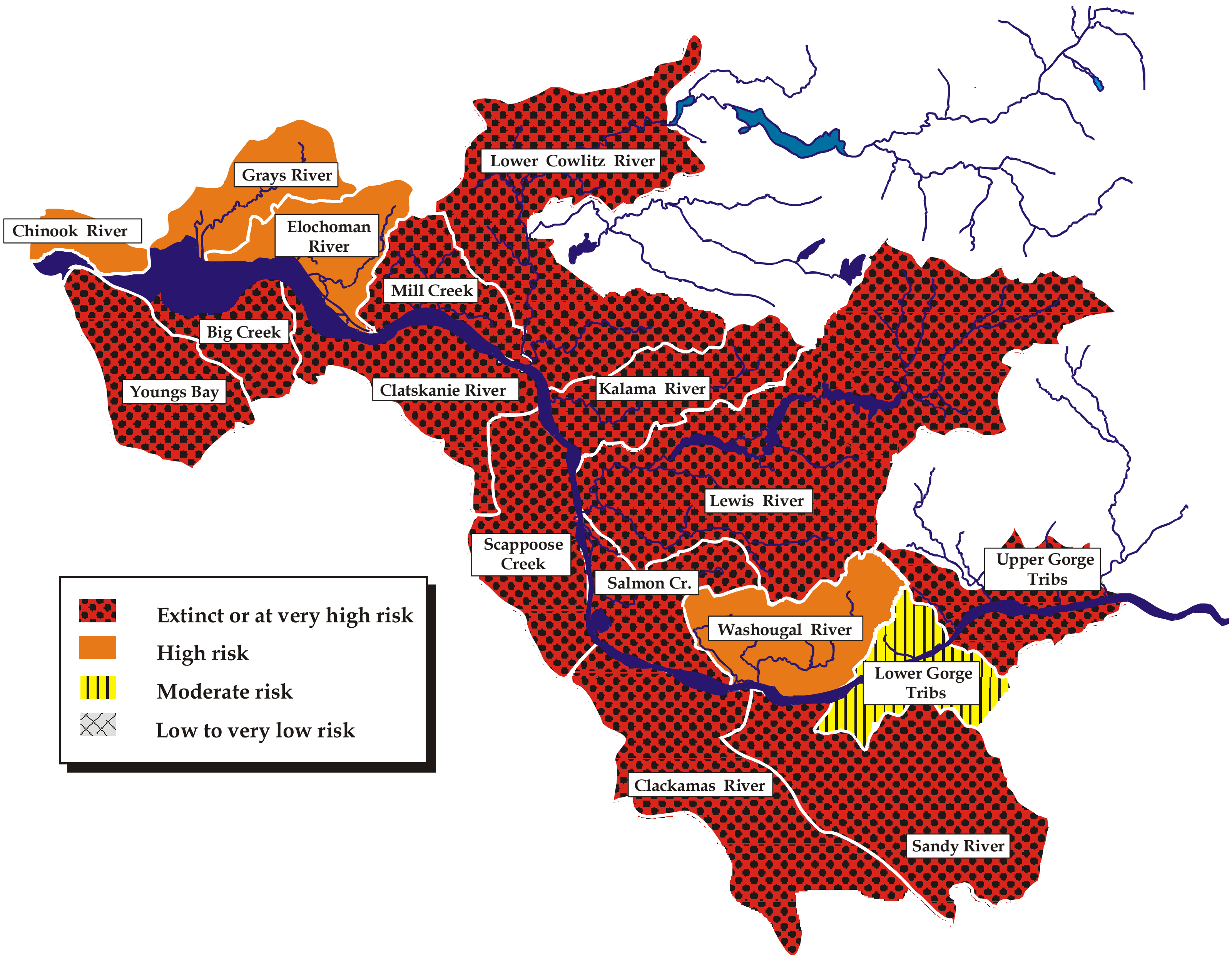

Figure 5-12. Distribution of historical chum salmon populations among lower Columbia River subbasins. Extinction risks are based on viability scores.

Table 5-10. Numbers and productivity of lower Columbia River chum populations.

Population	Leg[1]	Core[2]	4-yr[3]	EDT Equilibrium Population Size				BOE[8]	EDT Productivity			
				Current[4]	PFC[5]	PFC+[6]	Hist.[7]	Hist.	Current[4]	PFC[5]	PFC+[6]	Hist[7]
Coast												
Grays/Chinook	1	1	960	1,569	5,575	7,775	14,190	88,609	2.5	7.3	10.2	10.5
Eloch/Skam		1	≤150	1,640	5,888	8,212	16,320	53,408	2.1	6.1	8.6	9.3
Mill/Aber/Germ			≤150	603	2,129	2,969	6,587	41,674	1.9	5.6	7.8	8.8
Youngs Bay (OR)			≤150	--		--		123,405				
Big Creek (OR)			≤150	--		--		71,211				
Clatskanie (OR)			≤150	--		--		114,504				
Scappoose (OR)			≤150	--		--		31,559				
Cascade												
Cowlitz	1	1	≤150	7,892	55,258	135,721	479,781	294,553	1.9	6.8	16.7	9.9
Kalama			≤150	1,615	6,014	12,164	41,739	15,375	2.0	6.5	13.1	9.7
Lewis		1	≤150	9,070	30,051	71,006	294,363	121,382	2.4	6.6	15.6	9.7
Salmon			≤150	0	1,789	4,227	10,590	93,869	1.0	6.5	15.5	9.5
Washougal			≤150	699	3,971	9,350	42,553	25,086	1.6	7.1	16.6	10.5
Clackamas (OR)		1	≤150	--		--		117,336				
Sandy (OR)			≤150	--		--		40,461				
Gorge												
L Gorge	1	1	542	797	1,943	3,111	9,353	39,651	3.5	8.5	13.6	11.4
U. Gorge (Wind)			≤100	361	2,582	5,874	24,764	27,918	1.7	5.5	12.5	9.0

[1] *Genetic Legacy designation by the Technical Recovery Team. Genetic legacy populations represent unique life histories or are relatively unchanged by hatchery influences.*

[2] *Core population designation by Technical Recovery Team. Core populations were the largest historical populations and were key to metapopulation processes*

[3] *Recent 4-year average natural spawning escapements upon which PCC numbers are based (typically1997-2000 return years). Spawning escapements in 2002 and 2003 have generally been substantially greater than in the preceding years as these runs encountered much improved ocean survival conditions.*

[4] *Current number inferred with EDT from estimated and assumed habitat conditions.*

[5] *Estimate if habitat conditions are restored to "properly functioning" standards defined by NOAA Fisheries under current estuary conditions.*

[6] *Estimate if habitat conditions are restored to "properly functioning" standards defined by NOAA Fisheries and predevelopment estuary conditions are restored.*

[7] *Pre-development estimate inferred with EDT from assumed historical habitat conditions and current estuary conditions.*

[8] *Back of envelope estimates of historical population sizes inferred from stream miles accessible and assumed total Columbia River run (NOAA Fisheries).*

Table 5-11. Estimated viability of lower Columbia River chum salmon.

Population	Leg[1]	Core[2]	Population Persistence Scores							Data		Extinction risk	
			A/P[3]	J[4]	S[5]	D[6]	H[7]	Net[8]	Prob.[9]	Years[10]	Trend[11]	Model 1[12]	Model 2[13]
Coast													
Grays/Chinook	1	1	2	2	2	3	2	1.9	70%	1967-1998	1.000	1.043	0.006
Eloch/Skam		1	1	na	3	1	1	1.0	40%				
Mill/Aber/Germ			0.5	na	3	1	1	0.8	30%				
Youngs Bay (OR)			--	--	--	--	--	0.6	20%				
Big Creek (OR)			--	--	--	--	--	0.6	20%				
Clatskanie (OR)			--	--	--	--	--	0.6	20%				
Scappoose (OR)			--	--	--	--	--	0.6	20%				
Average								***0.85***	***30%***				
Cascade													
Cowlitz	1	1	0.5	na	3	1	1	0.8	30%				
Kalama			0.5	na	3	1	1	0.8	30%				
Lewis		1	0.5	na	3	1	1	0.8	30%				
Salmon			0	na	1.5	1	0	0.3	10%				
Washougal			1.5	na	3	2	2	1.3	50%				
Clackamas (OR)		1	--	--	--	--	--	0.4	10%				
Sandy (OR)			--	--	--	--	--	0.4	20%				
Average								***0.68***	***30%***				
Gorge													
L Gorge	1	1	3	2	3	4	2.5	2.1	80%	1944-2000	0.989	0.54	0.717
U. Gorge			1	na	1.5	1	1	0.6	30%				
Average								***1.39***	***50%***				

[1] Genetic Legacy designation by the Technical Recovery Team. Genetic legacy populations are relatively unchanged by hatchery influences or represent unique life histories.
[2]Core population designation by Technical Recovery Team. Core populations were the largest historical populations and were key to metapopulation processes
[3]Abundance and productivity rating by LCFRB biologists based on TRT criteria.
[4]Juvenile outmigration number rating by LCFRB biologists based on TRT criteria.
[5]Spatial structure rating by LCFRB biologists based on TRT criteria.
[6] Diversity rating by LCFRB biologists based on TRT criteria.
[7] Habitat rating by LCFRB biologists based on TRT criteria.
[8] Weighted average of population attribute scores. LCFRB and TRT scores are averaged.
[9] Persistence probability corresponding to net population score (interpolated from corresponding persistence ranges).
[10] Available abundance data time series upon which trend and extinction risk analyses by NOAA Fisheries were based.
[11] Trend slope estimated by NOAA Fisheries based on abundance time series (median annual growth rate or λ).
[12] Probability of extinction in 100 years (PE 100) estimated from abundance time series by NOAA Fisheries using Dennis-Holmes model.
[13] Population projection interval extinction risks (PPI E) estimated from abundance time series by NOAA Fisheries using Population Change Criteria model.

Table 5-12. Population abundance and productivity planning ranges for lower Columbia River chum populations.

	Recent	Abundance range		Current	Current	Productivity range		Productivity Improvement Increments			
Population	Avg. no.	Viable	Potential	viability	Prod.	Viable	Potential	Contrib	High	V high	Max
Coast											
Grays/Chinook	960	4,300	7,800	**Low**	1.00	1.08	2.78	4%	8%	93%	178%
Eloch/Skam	150	1,100	8,200	**Low**	0.74	1.14	2.48	27%	54%	144%	234%
Mill/Ab/Germ	150	1,100	3,000	**V Low**	0.70	1.14	2.38	32%	63%	152%	241%
Youngs (OR)	150	1,100	2,200	**V Low**	--	--	--	--	--	--	--
Big Creek (OR)	150	1,100	2,200	**V Low**	--	--	--	--	--	--	--
Clatskanie (OR)	150	1,100	2,200	**V Low**	--	--	--	--	--	--	--
Scappoose (OR)	150	1,100	2,200	**V Low**	--	--	--	--	--	--	--
Cascade											
Cowlitz	150	1,100	135,700	**V Low**	0.64	1.14	3.26	39%	78%	242%	407%
Kalama	150	1,100	12,200	**V Low**	0.76	1.14	2.97	25%	51%	172%	293%
Lewis	150	1,100	71,000	**V Low**	0.91	1.14	3.17	13%	25%	137%	248%
Salmon	150	1,100	4,200	**V Low**	0.00	1.14	3.17	--	--	--	--
Washougal	150	1,100	9,400	**Low**	0.43	1.00	2.85	66%	131%	345%	560%
Clackamas (OR)	150	1,100	2,200	**V Low**	--	--	--	--	--	--	--
Sandy (OR)	150	1,100	2,200	**V Low**	--	--	--	--	--	--	--
Gorge											
Lower Gorge	542	2,600	3,100	**Med**	0.99	1.11	2.71	6%	12%	93%	174%
Upper Gorge	100	1,100	5,900	**V Low**	0.06	1.14	2.92	963%	1927%	3512%	5097%

Notes

1. *Recent average numbers are observed 4-year averages or assumed natural spawning escapements. Data typically is through year 2000.*
2. *Abundance planning range refer to average equilibrium escapement numbers at viability as defined by NOAA's Population Change Criteria and potential as defined by WDFW's Ecosystem Diagnosis and Treatment assessments under properly functioning habitat and historical estuary conditions..*
3. *Current viability is based on Technical Recovery Team viability rating approach.*
4. *Current and planning range productivity values are expressed in terms of intrinsic rate of population increase. Estimates are available only where data exists to EDT and population trend assessments.*
5. *Productivity improvement increments indicate needed improvements to reach contributing, high, very high, and maximum levels of population viability or potential.*

5.3.3 Population Significance

The population significance index provides a simple sorting device to group populations in each strata based on current viability, core potential and genetic legacy (Table 5-13). In the Coast stratum, Grays chum sort to the top by virtue of their current viability and genetic legacy designations. The Elochoman population was designated as a core population by the TRT, but current numbers are not substantially greater than the Mill/Abernathy/Germany population. In the Cascade stratum, Cowlitz chum sort to the top by virtue of their current viability and genetic legacy designations. All other Cascade chum are grouped in a low tier. The Gorge chum populations are distinguished by core and legacy designations as well as current numbers for the lower Gorge population.

Table 5-13. Biological significance categories of lower Columbia chum populations based on current viability, core potential, and genetic legacy considerations.

	Raw ratings				Normalized values				
Population	**Gen[1]**	**Core[2]**	**Poten.[3]**	**Viab.[4]**	**Viab.[5]**	**Poten.[6]**	**Gen.[7]**	**Index[8]**	**Rank[9]**
Coast									
Grays/Chinook	1	1	7,800	1.9	0.63	0.06	1.00	0.56	A
Eloch/Skam		1	8,200	1.0	0.34	0.06	0.00	0.13	C
Mill/Ab/Germ			3,000	0.8	0.28	0.02	0.00	0.10	C
Youngs (OR)			2,200	0.6	0.18	0.02	0.00	0.07	--
Big Creek (OR)			2,200	0.6	0.18	0.02	0.00	0.07	--
Clatskanie (OR)			2,200	0.6	0.18	0.02	0.00	0.07	--
Scappoose (OR)			2,200	0.6	0.18	0.02	0.00	0.07	--
Cascade									
Cowlitz	1	1	135,700	0.8	0.27	1.00	1.00	0.76	A
Lewis		1	71,000	0.8	0.27	0.52	0.00	0.26	C
Washougal			9,400	1.3	0.43	0.07	0.00	0.17	C
Kalama			12,200	0.8	0.25	0.09	0.00	0.11	C
Salmon			4,200	0.3	0.11	0.03	0.00	0.05	C
Clackamas (OR)		1	2,200	0.4	0.12	0.02	0.00	0.04	--
Sandy (OR)			2,200	0.4	0.13	0.02	0.00	0.05	--
Gorge									
Lower Gorge	1	1	3,100	2.1	0.71	0.02	1.00	0.58	A
Upper Gorge			5,900	0.6	0.21	0.04	0.00	0.09	C

1 *Genetic Legacy designation by the Technical Recovery Team. Genetic legacy populations are relatively unchanged by hatchery influences or represent unique life histories.*
2 *Core population designation by Technical Recovery Team. Core populations were the largest historical populations and were key to metapopulation processes*
3 *Potential numbers based on top end of planning range (typical value if accessible habitat restored to favorable albeit not pristine conditions based on EDT results for properly functioning conditions plus restored estuary.*
4 *Provisional ratings by LCFRB consultants and WDFW staff based on TRT standards*
5 *Normalized population persistence score used in biological significance ranking.*
6 *Normalized core population potential used in biological significance ranking.*
7 *Genetic legacy score used in biological significance ranking.*
8 *Average of now, potential and genetic scores.*
9 *Strata ranking based on average population score.*

5.3.4 *Current Limiting Factors*

5.3.4.1 Net Effect of Manageable Factors

The net effects of quantifiable human impacts and potentially manageable predation on chum salmon translates into an 92-100% reduction in productivity among Washington lower Columbia populations (Figure 5-13). Thus, current fish numbers are only 0-8% of what they would be if all manageable impacts were removed.

Habitat degradation in spawning and rearing areas accounts for half or more of the manageable impacts in all populations except for the Gorge where direct hydropower impacts are also significant (Figure 5-13). Estuary habitat changes are also thought to be significant for chum salmon that emigrate from spawning areas at small sizes. Fishing and hatchery impacts are very small. Composite chum salmon habitat impact factors and indices are listed in Table 5-14.

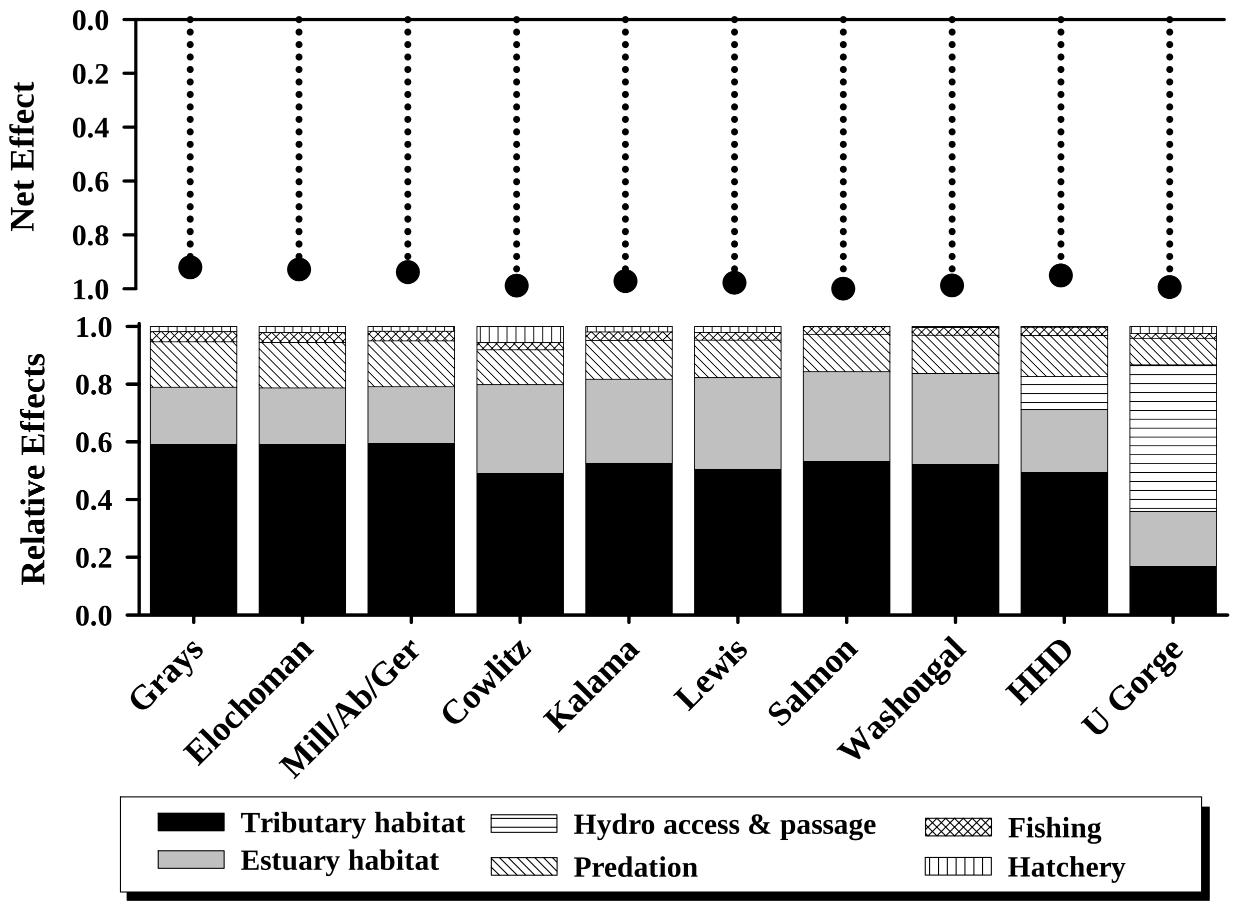

Figure 5-13. **Net effect and relative contribution of potentially manageable impact factors on chum salmon in Washington lower Columbia River subbasins. Net effect is the approximate reduction from historical fish numbers as a result of manageable factors included in this analysis.**

Table 5-14. Chum salmon impact factors and index.

	Grays	E/S	M/A/G	Cowlitz	Kalama	Lewis	Salmon	Wash.	HHD	U Gorge
Inputs										
Neq Current	1,569	1,640	603	7,892	1,615	9,070	0	699	797	361
Neq PFC	5,575	5,888	2,129	55,258	6,014	30,051	1,789	3,971	1,943	2,582
Neq PFC+	7,775	8,212	2,969	135,721	12,164	71,006	4,227	9,350	3,111	5,874
Neq Historical	14,190	16,320	6,989	479,781	41,739	294,363	10,590	42,553	9,353	24,764
Hydro habitat loss	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.200	0.900
Dam passage mortality (juveniles)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.200
Dam passage mortality (adults)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.500
Predation mortality (juveniles)	0.200	0.206	0.209	0.211	0.212	0.215	0.220	0.220	0.223	0.251
Predation mortality (adults)	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030
Fishing	0.050	0.050	0.050	0.050	0.050	0.050	0.050	0.050	0.050	0.050
Hatchery fraction	0.250	0.25	0.25	0.000	0.000	0.000	0.000	0.000	0.05	0.000
Hatchery category	1	1	1	0	0	0	0	0	1	1
Hatchery fitness	0.9	0.9	0.9	0.0	0.0	0.0	0.0	0.0	0.9	0.9
Other hatchery species	40,000	120,000	0	2,189,500	680,000	745,000	20,000	120,000	0	1,420,000
Impacts (p reduction)										
Tributary habitat	0.846	0.860	0.880	0.960	0.922	0.927	1.000	0.961	0.864	0.500
Estuary habitat	0.283	0.283	0.283	0.593	0.506	0.577	0.577	0.575	0.375	0.560
Hydro habitat loss	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.200	0.900
Dam passage	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.600
Predation	0.224	0.230	0.233	0.235	0.236	0.239	0.243	0.243	0.246	0.273
Fishing	0.050	0.050	0.050	0.050	0.050	0.050	0.050	0.050	0.050	0.050
Hatchery fitness	0.025	0.025	0.025	0.000	0.000	0.000	0.000	0.000	0.005	0.000
Hatchery interspecies	0.002	0.006	0.000	0.109	0.034	0.037	0.001	0.006	0.000	0.071
Total (unconditional)	1.430	1.454	1.470	1.947	1.747	1.830	1.871	1.836	1.740	2.955
Impact index										
Tributary habitat	0.592	0.592	0.598	0.493	0.528	0.507	0.534	0.524	0.496	0.169
Estuary habitat	0.198	0.195	0.192	0.305	0.289	0.315	0.308	0.313	0.216	0.190
Hydro access/passage	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.115	0.508
Predation	0.157	0.158	0.158	0.121	0.135	0.130	0.130	0.133	0.142	0.093
Fishing	0.035	0.034	0.034	0.026	0.029	0.027	0.027	0.027	0.029	0.017
Hatchery	0.019	0.021	0.017	0.056	0.019	0.020	0.001	0.003	0.003	0.024

5.3.4.2 Fisheries

Current fishing impacts on chum salmon are very low and provide no significant opportunity for increasing their numbers through additional regulation of fisheries. No sport or commercial fisheries target chum salmon. Impacts of 5% or less at listing and 3% or less at present are incidental to fisheries for other species (Figure 5-14). Historical fishing rates were much greater; these have been steadily reduced to current levels.

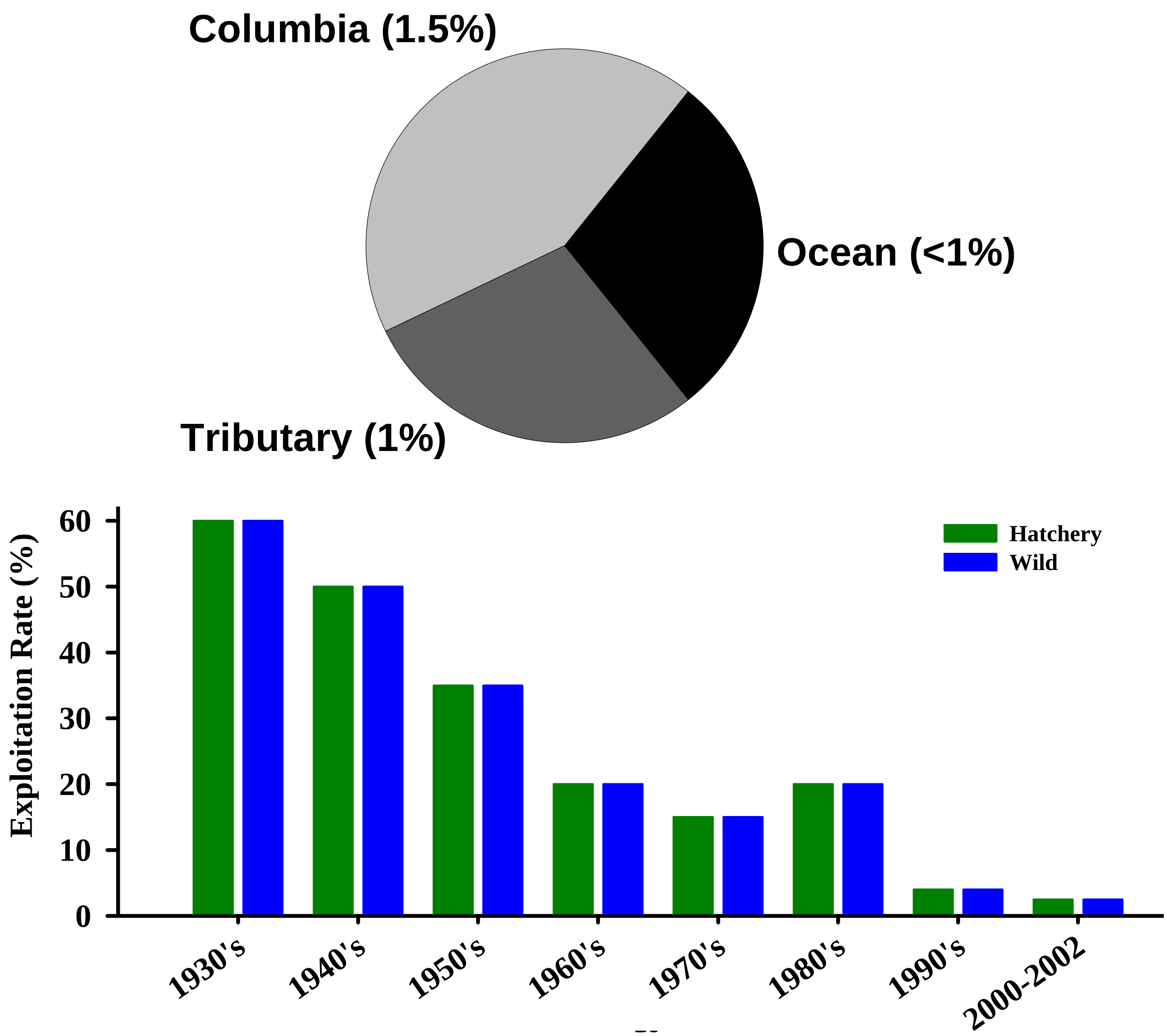

Figure 5-14. **Approximate chum fishery exploitation rates over time and allocation of current rates among fisheries.**

5.3.4.3 Hatcheries

Historical and current hatchery influences on chum are minimal. Hatchery chum salmon have been released into only 4 of 10 Washington populations (Table 5-15). Hatchery fish do not comprise a substantial fraction of any naturally spawning chum population and all originate from local wild populations (category 1 brood types). Current chum hatchery programs are focused on reintroduction (Chinook River) and conservation (Duncan Creek).

Inter-specific hatchery predation impacts on juvenile chum range from 0% in basins without significant releases of coho and spring chinook to a high of 11% in the Cowlitz basin where large hatchery programs are underway.

Table 5-15. Presumed reductions in wild population fitness as a result of natural hatchery spawners and survival as a result of interactions with other hatchery species for Washington lower Columbia River chum populations.

Population	Annual releases[a]	Hatchery fraction	Fitness category	Assumed fitness	Fitness impact	Interacting releases[g]	Interspecies impact
Coast							
Grays/Chinook[b]	447,500[b]	0.25	1	0.9	0.025	40,000	0.002
Eloch/Skam	0[c]	0.25	1	0.9	0.025	120,000	0.006
Mill/Ab/Germ	0[d]	0.25	1	0.9	0.025	0	0
Cascade							
Cowlitz	0[e]	0	--	--	0	2,189,500	0.109
Kalama	0[e]	0	--	--	0	680,000	0.034
Lewis	0[e]	0	--	--	0	745,000	0.037
Salmon	0[e]	0	--	--	0	0	0.001
Washougal	0[e]	0	--	--	0	120,000	0.006
Gorge							
Lower Gorge	100,000[f]	0.05		0.9	0.005	0	0
Upper Gorge	0[e]	0	--	--	0	1,420,000	0.071

[a] *Annual release goals.*

[b] *Releases include 300,000 in the Grays River to supplement natural production and 147,500 to restore a Chinook River population.*

[c] *Hatchery chum salmon have not been released in the basin since 1983.*

[d] *There is currently no chum salmon hatchery program in Mill, Abernathy, or Germany Creek; hatchery chum salmon have not been released in Abernathy Creek since 1991 or Germany Creek since 1983.*

[e] *There are no records of hatchery chum releases in the basin.*

[f] *A hatchery program recently began at the Washougal Hatchery utilizing Hardy Creek chum for brood stock; releases are planned for Duncan Creek to enhance current chum returns. Additional releases may occur in Hardy and Hamilton Creeks, and in the mainstem Columbia near Ives Island when low flows limit adult access to spawning areas.*

[g] *Includes steelhead and spring chinook.*

5.3.4.4 Tributary Habitat

EDT analyses suggest that stream degradation has substantially reduced the habitat potential for chum salmon in all Washington lower Columbia River subbasins where analyses have been completed (Figure 5-15). Declines in habitat quantity and quality for chum salmon have reduced current productivity potential to only 10-30% and current equilibrium numbers to only 0-11% of the historical template. Substantial stream habitat improvements would be necessary to reach optimum conditions (i.e. PFC) for chum salmon in all pertinent subbasins. Restoration of optimum habitat quality would be expected to increase habitat capacity by 1,000 to 47,000 adult chum per subbasin.

Chum salmon rely on the lower and middle mainstem stream reaches of large streams and rivers where channel instability, low habitat diversity, and sedimentation consistently limit habitat suitability. More detailed descriptions of stream habitat conditions and effects on fish in each subbasin may be found in Volume II of the Technical Foundation.

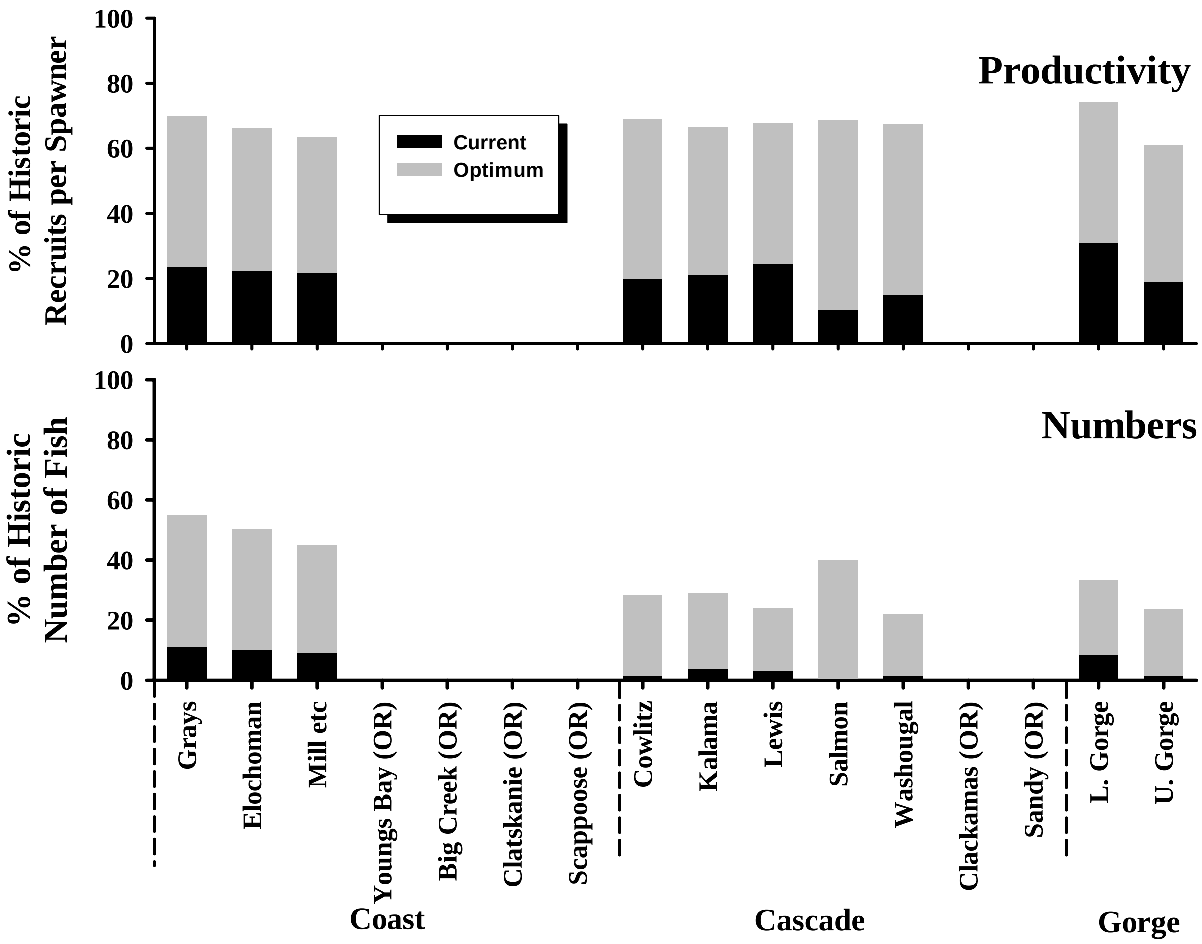

Figure 5-15. Current, optimal, and historical subbasin productivity and equilibrium population size inferred for chum salmon from stream reach habitat conditions using EDT.

5.3.4.5 Dams

Direct dam effects on Washington lower Columbia chum were limited to the Gorge populations. Assumed dam-related reductions were 20% in the lower Gorge population and 96% in the upper Gorge population. These impacts included assumed passage mortality at Bonneville Dam of 20% for juveniles and an additional 50% for adults. Bonneville Dam inundated an assumed 90% of the chum habitat in the Wind River. Lower Gorge assessments included an assumed 20% reduction in chum productivity in the Columbia River mainstem as a result of Bonneville Dam operations. Dam operations in the Cowlitz and Lewis River also have the potential to affect downstream habitat conditions for chum but the significance of this impact is unknown.

5.3.4.6 Mainstem and Estuary Habitat

Chum salmon rear and migrate through critical mainstem and estuary areas soon after emergence and emigration from tributary streams. Mainstem and estuary habitat impacts were assumed to account for approximately a 20-40% reduction in productivity of chum.

5.3.4.7 Predation

Potentially manageable predation mortality on chum salmon was assumed to average 20% to 25% depending on travel distance from the subbasin to the ocean. Chinook salmon rates were used in the absence of specific chum data. Pikeminnow and tern management is projected to reduce salmonid predation by approximately 50%. Tern predation is almost entirely an artifact of recently established colonies on dredge spoil islands in the estuary but the current rate (9%) is less than half that observed prior to downstream translocation of the Rice Island colony (20%). Pikeminnow predation was greatest for populations that originate in Bonneville Reservoir tributaries (5%), pass the pikeminnow gauntlet in Bonneville Dam forebay and tailrace, and travel the entire 145-mile length from Bonneville to the Estuary. Predation rates by seals and sea lions on adult chinook were assumed to average 3%.

5.3.5 Summary Assessment

1. Human activities including fishing, hatchery operation, alteration of stream, river, and estuary habitats, hydropower development and operation, and potentially manageable predation have collectively reduced productivity of chum populations to 0-8% of historical levels.
2. Current fishing impacts on chum salmon are very low and these provide limited opportunity for increasing their numbers through additional regulation of fisheries.
3. Existing chum hatchery programs pose no significant risk of reduced wild productivity as a result of interbreeding with potentially less-fit hatchery fish. Inter-specific hatchery predation impacts on juvenile chum range from 0% in basins without significant releases of coho and spring chinook to a high of 11% in the Cowlitz basin where large hatchery programs are underway.
4. Recovery efforts will require significant improvements in stream habitat quantity and quality. Stream habitat degradation accounts for large declines in chum salmon numbers in all populations.

5. Significant degradation has occurred in estuary and mainstem habitats assumed to be critical to chum salmon life history.
6. Hydropower impacts on chum salmon are poorly quantified but significant impacts result from operational effects on chum spawning habitat in the mainstem downstream from Bonneville Dam, passage mortality of adults and juveniles at Bonneville Dam, and the inundation by Bonneville Reservoir of lower tributary reaches in the Columbia River Gorge. Cowlitz and Lewis dams have not blocked significant amounts of chum habitat but flood control has altered habitat-forming processes in lower subbasin reaches favored by chum.

5.4 Coho

5.4.1 Current Viability

Because coho are not currently listed under the ESA, the Willamette/Lower Columbia Technical Recovery Team has not designated populations of coho in the Lower Columbia River. However, as part of the Status Review process for ESA-listed ESUs, the NOAA Fisheries Biological Review Team tentatively identified 25 historical LCR coho populations: 18 populations in Washington and 7 in Oregon (Figure 5-16). Designation of coho populations was based heavily on the WLCTRT's designation of population boundaries for LCR steelhead and chinook (Myers et al 2003).

Recent numbers have averaged fewer than 300 naturally produced fish in 16 of 18 Washington coho salmon populations and 3 of 7 Oregon coho populations. For those populations where no current spawning escapement estimate has been provided, the presence of wild coho in these basins is expected to be minimal (i.e. upper Cowlitz, Cispus, Tilton, and Salmon). Recent natural escapements of Washington lower Columbia coho exceeded an average of 1,000 fish only in the lower Cowlitz and NF Lewis basins. The recent average escapements have also been consistently less than EDT equilibrium numbers based on current stream habitat conditions in part because of poor ocean survival conditions. Minimum historical coho population sizes in Washington ranged from 300 to 41,900 based on EDT estimates (Table 5-16). EDT underestimated coho numbers because current analyses do not include many of the smaller streams used by coho. Back-of-envelope estimates by NOAA Fisheries yielded historical coho population sizes in Washington of 10,200 to 119,000 based on presumed Columbia River run totals and subbasin habitat quantity. For coho populations, BOE estimates are consistently greater than EDT historical abundance estimates.

Based on interim TRT population criteria, 100-year persistence probabilities are very low or already extinct (0-39%) for 17 populations, low (40-74%) for 7 populations, and moderate (75-94%) for only 1 population; no coho populations had a relatively high (95-99%) 100-year persistence probability (Table 5-17). All strata currently fall short of integrated TRT recovery criteria which specify an average persistence probability greater than 2.25 with at least 2 populations at high (>3.0) for each strata.

Population trends and extinction risks have been estimated for 2 coho populations (i.e. Clackamas and Sandy) based on abundance time series data and two different models (NOAA Fisheries, unpublished data). Population trends were positive for both populations; extinction risks averaged for both models were relatively low (16% for the Clackamas and 53% for the Sandy; Table 5-17). Model-derived estimates are fairly optimistic, considering that the time period of available data was coincident with population declines following the ocean regime shift in the late 1970s and that the front half of the available time series is affected by very large post 1983-84 El Niño returns. However, Clackamas and Sandy River coho populations are not representative of other Lower Columbia River coho salmon populations because these two systems represent the only subbasins with appreciable numbers of wild coho remaining. Differences between score-derived persistence probabilities and trend-derived extinction risks reflect different assumptions and uncertainties in these methods.

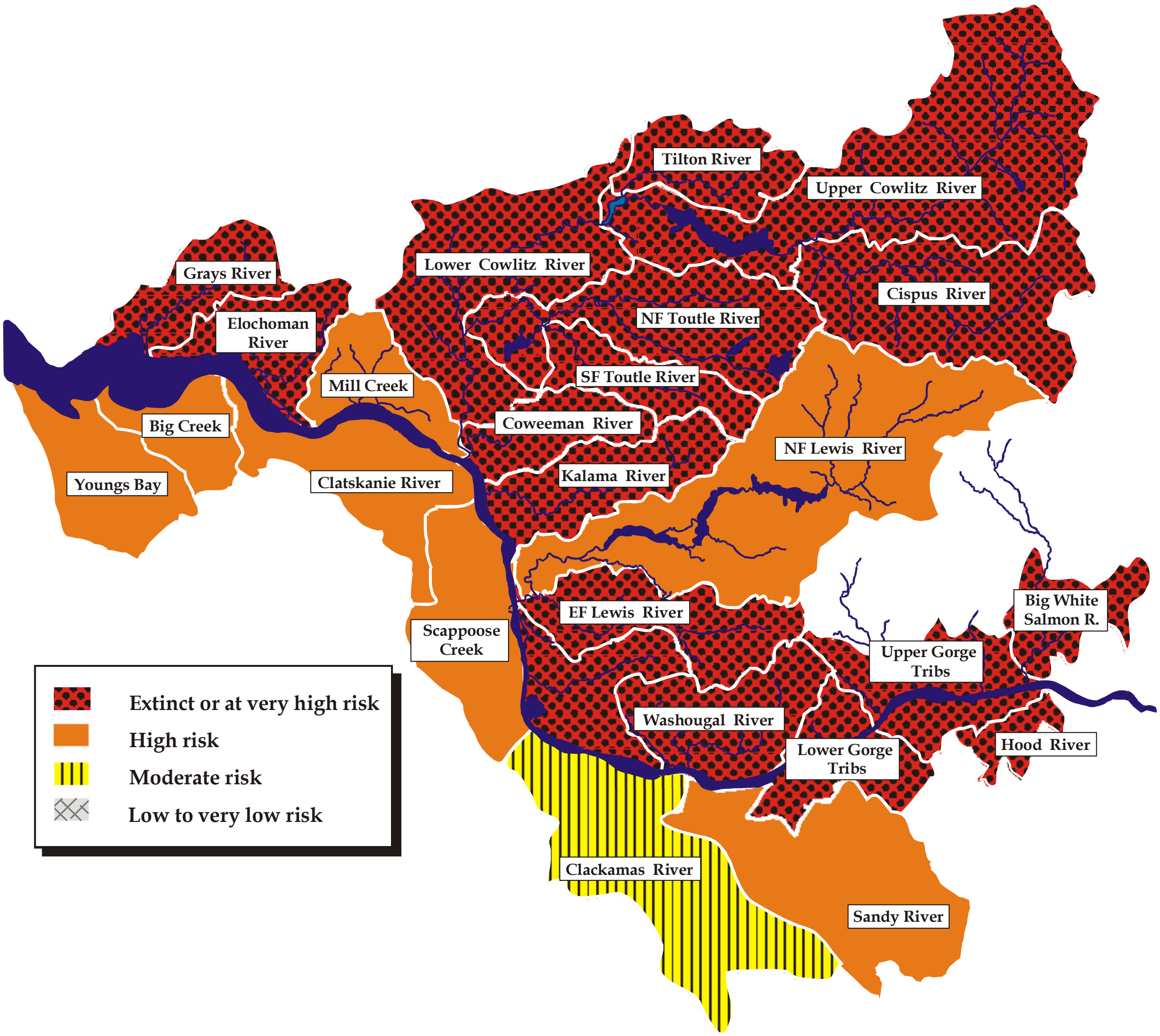

Figure 5-16. Tentative historical populations of Lower Columbia River coho salmon, based on TRT population designations for chinook and steelhead.

Table 5-16. Numbers and productivity for lower Columbia River coho populations.

					EDT Equilibrium Population Size				BOE[8]	EDT Productivity			
Population	**Leg[1]**	**Core[2]**	**4-yr[3]**	**HLFM[5]**	**Current[4]**	**PFC[5]**	**PFC+[6]**	**Hist.[7]**	**Hist.**	**Current[4]**	**PFC[5]**	**PFC+[6]**	**Hist[7]**
Coast													
Grays/Chinook			28	2,417	1,239	3,773	4,593	5,289	39,298	3.9	12.7	15.5	16.6
Eloch/Skam			32	--	2,396	5,787	7,045	13,885	43,200	4.3	9.6	11.7	21.5
Mill/Aber/Germ			24	--	2,045	3,010	3,664	10,621	30,007	4.7	8.0	9.7	19.7
Youngs Bay (OR)			403	--	--	--	--	--	57,599	--	--	--	--
Big Creek (OR)				--	--	--	--	--	33,724	--	--	--	--
Clatskanie (OR)			92	--	--	--	--	--	54,255	--	--	--	--
Scappoose (OR)			458	--	--	--	--	--	12,170	--	--	--	--
Cascade													
L Cowlitz	late		1,015	6,379	4,144	15,655	19,058	21,458	119,008	4.2	12.4	15.1	17.1
Coweeman	?		15	3,066	1,873	6,225	7,579	10,267	24,898	3.4	8.1	9.9	12.5
Toutle SF	early		44	--	3,860	27,027	32,901	41,912	16,537	2.2	9.1	11.1	13.1
Toutle NF	early		190	11,159	--		--	--	61,780	--	--	--	--
U Cowlitz	late		--	--	11,039	23,633	28,770	17,654	67,075	3.0	7.3	8.9	21.4
Cispus	late		--	--	3,752	5,351	6,612	8,029	12,356	4.0	7.5	9.2	22.1
Tilton	late		--	--	261	3,233	4,011	5,599	23,318	2.6	12.6	15.4	24.9
Kalama	both		18	1,674	484	1,033	1,282	1,620	26,477	3.8	8.7	10.8	12.5
Lewis NF	early		3,778	3,300	2,367	4,771	5,917	7,474	84,727	5.2	8.9	11.1	11.9
Lewis EF	late		43	888	1,066	3,306	4,101	5,309	41,899	2.6	8.8	11.0	12.6
Salmon	late		--	--	772	4,621	5,731	6,532	36,139	2.2	11.0	13.6	14.3
Washougal	late		14	1,554	824	3,362	4,170	4,860	35,303	2.2	7.6	9.4	10.5
Clackamas (OR)	late		1,684	--	--	--	--	--	58,714	--	--	--	--
Sandy (OR)	early		587	--	--	--	--	--	60,386	--	--	--	--
Gorge													
L Gorge			28	--	57	123	153	347	13,285	5.1	7.5	9.4	10.2
U. Gorge			233	--	418	898	1,114	1,174	10,219	2.9	4.8	5.9	5.4
White Salmon			129	--	--	--	--	--	17,187	--		--	--
Hood (OR)			<50	--	--	--	--	--	20,438	--		--	--

[1] *Genetic Legacy designation by the Technical Recovery Team. Genetic legacy populations represent unique life histories or are relatively unchanged by hatchery influences.*
[2] *Core population designation by Technical Recovery Team. Core populations were the largest historical populations and were key to metapopulation processes*
[3] *Recent 4-year average natural spawning escapements from TRT analysis were only available for the Clackamas and Sandy. Most spawning escapement estimates represent the relative abundance of each population based on recent WDFW or ODFW spawner survey data.*
[4] *Current number inferred with EDT from estimated and assumed habitat conditions.*
[5] *Estimate if habitat conditions are restored to "properly functioning" standards defined by NOAA Fisheries under current estuary conditions.*
[6] *Estimate if habitat conditions are restored to "properly functioning" standards defined by NOAA Fisheries and predevelopment estuary conditions are restored.*
[7] *Pre-development estimate inferred with EDT from assumed historical habitat conditions.*
[8] *Back of envelope estimates of historical population sizes inferred from stream miles accessible and assumed total Columbia River run.*
[9]*Estimated abundance based on Habitat Limiting Factors Model (Nickelson et al. 1992, Nickelson 1998) and assumed 4% marine survival.*

Table 5-17. Estimated viability of lower Columbia River coho salmon.

	Population Persistence Scores		Data		Extinction risk	
Population	**Net**[1]	**Prob.**[2]	**Years**[3]	**Trend**[4]	**Model 1**[5]	**Model 2**[6]
Coast						
Grays/Chinook	0.8	30%	--	--	--	--
Eloch/Skam	0.7	30%	--	--	--	--
Mill/Aber/Germ	1.0	40%	--	--	--	--
Youngs Bay (OR)	1.1	40%	--	--	--	--
Big Creek (OR)	1.1	40%	--	--	--	--
Clatskanie (OR)	1.3	50%	--	--	--	--
Scappoose (OR)	1.5	60%	--	--	--	--
Average	**1.06**	**40%**				
Cascade						
L Cowlitz	0.8	30%	--	--	--	--
Coweeman	0.8	30%	--	--	--	--
Toutle SF	0.8	30%	--	--	--	--
Toutle NF	0.7	30%	--	--	--	--
U Cowlitz	0.2	10%	--	--	--	--
Cispus	0.2	10%	--	--	--	--
Tilton	0.2	10%	--	--	--	--
Kalama	0.8	30%	--	--	--	--
Lewis NF	1.0	40%	--	--	--	--
Lewis EF	0.8	30%	--	--	--	--
Salmon	0.6	20%	--	--	--	--
Washougal	0.7	30%	--	--	--	--
Clackamas (OR)	2.0	80%	1973-1999	1.027	0.022	0.295
Sandy (OR)	1.9	70%	1977-1999	1.012	0.365	0.696
Average	**0.81**	**30%**				
Gorge						
L Gorge	0.7	30%	--	--	--	--
U. Gorge	0.7	30%	--	--	--	--
White Salmon	0.4	20%	--	--	--	--
Hood (OR)	0.7	30%	--	--	--	--
Average	**0.6**	**20%**				

[1] *Population persistence scores for Washington populations are based solely on TRT scores; LCFRB did not score coho. For Oregon tributaries, population persistence scores are the average of ODFW and TRT scores.*
[2] *Persistence probability corresponding to net population score (interpolated from corresponding persistence ranges).*
[3] *Available abundance data time series upon which trend and extinction risk analyses by NOAA Fisheries were based.*
[4] *Trend slope estimated by NOAA Fisheries based on abundance time series (median annual growth rate or λ).*
[5] *Probability of extinction in 100 years (PE 100) estimated from abundance time series by NOAA Fisheries using Dennis-Holmes model.*
[6] *Population projection interval extinction risks (PPI E) estimated from abundance time series by NOAA Fisheries using Population Change Criteria model.*

5.4.2 Recovery Planning Ranges

Planning ranges based on PCC are not available for Lower Columbia River coho populations. No estimates are available for coho although the scale of limiting factors suggests that several-fold improvements in productivity will be required to reach viability.

5.4.3 Population Significance

The population significance index provides a simple sorting device to group populations in each strata based on current viability, core potential, and genetic legacy considerations (Table 5-18). The WLCTRT has not designated "core" or "legacy" coho populations based on the abundance and genetic criteria utilized for ESA-listed salmonids (see section 5.1.3). An available surrogate for the "core" population designation is a relative index based on NOAA Fisheries BOE abundance estimates. The core potential population score was an index of how each population's BOE abundance related to the largest BOE–derived Columbia coho population (i.e. lower Cowlitz). There is no simple surrogate for the genetic legacy criteria utilized by the WLCTRT for other salmonids; thus, we had no basis for determining a genetic legacy population score. Since no genetic legacy score was calculated for any lower Columbia coho population, effects on the average population score and relative ranking were uniform across all coho populations.

Based on the population significance index, Washington coho salmon populations in the Coast strata are ranked in the same group (Table 5-15). Each of the Coast strata populations received similar scores for current viability and potential abundance. In the Cascade strata, the lower Cowlitz and NF Lewis sort to the top by virtue of their current viability and core potential designations. The second tier in the Cascade strata includes NF Toutle, upper Cowlitz, and EF Lewis populations; these populations had moderately large historical populations. A third Cascade tier includes Washougal, Kalama, Salmon, Coweeman, SF Toutle, Tilton, Cispus, populations; these populations were all relatively small and are all currently at low levels of viability. No Gorge coho population is distinguished from the others by this index.

Table 5-18. **Biological significance categories of lower Columbia coho populations based on current viability, core potential, and genetic legacy considerations.**

	Raw ratings		Normalized values				
Population	**Poten.[1]**	**Viab.[2]**	**Viab.[3]**	**Poten.[4]**	**Gen.[5]**	**Index[6]**	**Rank[7]**
Coast							
Grays/Chinook	4,600	0.8	0.28	0.33	0.00	0.20	B
Eloch/Skam	7,000	0.7	0.22	0.36	0.00	0.20	B
Mill/Ab/Germ	3,700	1.0	0.34	0.25	0.00	0.20	B
Youngs (OR)	1,200	1.1	0.37	0.48	0.00	0.28	--
Big Creek (OR)	1,200	1.1	0.37	0.28	0.00	0.22	--
Clatskanie (OR)	1,200	1.3	0.42	0.46	0.00	0.29	--
Scappoose (OR)	1,200	1.5	0.48	0.10	0.00	0.20	--
Cascade							
Lower Cowlitz	19,100	0.8	0.27	1.00	0.00	0.42	A
NF Lewis	5,900	1.0	0.34	0.71	0.00	0.35	A
N.F. Toutle	1,200	0.7	0.22	0.52	0.00	0.25	B
Upper Cowlitz	28,800	0.2	0.07	0.56	0.00	0.21	B
EF Lewis	4,100	0.8	0.26	0.35	0.00	0.20	B
Washougal	4,200	0.7	0.23	0.30	0.00	0.17	C
Kalama	1,300	0.8	0.27	0.22	0.00	0.16	C
Salmon	5,700	0.6	0.19	0.30	0.00	0.16	C
Coweeman	7,600	0.8	0.27	0.21	0.00	0.16	C
S.F. Toutle	32,900	0.8	0.26	0.14	0.00	0.13	C
Tilton	4,000	0.2	0.05	0.20	0.00	0.08	C
Cispus	6,600	0.2	0.07	0.10	0.00	0.06	C
Clackamas (OR)	1,200	2.0	0.67	0.49	0.00	0.39	--
Sandy (OR)	1,200	1.9	0.63	0.51	0.00	0.38	--
Gorge							
L Gorge (Ham.)	1,200	0.7	0.23	0.11	0.00	0.11	C
U Gorge (Wind)	1,100	0.7	0.23	0.09	0.00	0.11	C
White Salmon	1,200	0.4	0.13	0.14	0.00	0.09	C
Hood (OR)	1,200	0.7	0.22	0.17	0.00	0.13	--

[1]Potential fish numbers based on top end of planning range (based on twice the minimum viable population size for steelhead).

[2]Population viability scores for Washington populations are based solely on TRT scores; LCFRB did not score coho. For Oregon tributaries, population viability scores are the average of ODFW and TRT scores.

[3] Normalized population persistence score used in biological significance ranking.

[4] Normalized core population potential used in biological significance ranking. The TRT has not designated core populations for coho; the score is based on BOE abundance.

[5] Genetic legacy score used in biological significance ranking. The TRT has not assigned genetic legacy designations for coho; no surrogate is available for this metric.

[6] Average of now, potential and genetic scores.

[7] Strata ranking based on average population score.

5.4.4 *Current Limiting Factors*

5.4.4.1 Net Effects of Manageable Factors

The net effects of quantifiable human impacts and potentially manageable predation on coho salmon translates into a 92-100% reduction in productivity among Washington lower Columbia populations (Figure 5-17). Thus, current fish numbers are only 0-8% of what they would be if all manageable impacts were removed.

No single factor accounts for the majority of the reduction in fish numbers (Figure 5-17) Loss of tributary habitat quantity and quality generally account for significant shares of the impact, particularly in the NF Toutle population where tributary habitat loss accounts for over half of the total impact. Dam construction constitutes the largest single impact for upper Cowlitz, Cispus, Tilton, and NF Lewis populations but does not appear to be a primary limiting factor for other coho populations, including the upper Gorge. Fishing is a relatively low impact for most coho populations. Hatchery effects vary among populations but approach 30% of the total impact in some populations. Predation and estuary habitat conditions were among the lesser impacts we considered. Preliminary coho salmon impact factors and indices are listed in Table 5-19; the values in this table will be superceded by forthcoming coho-specific EDT analyses.

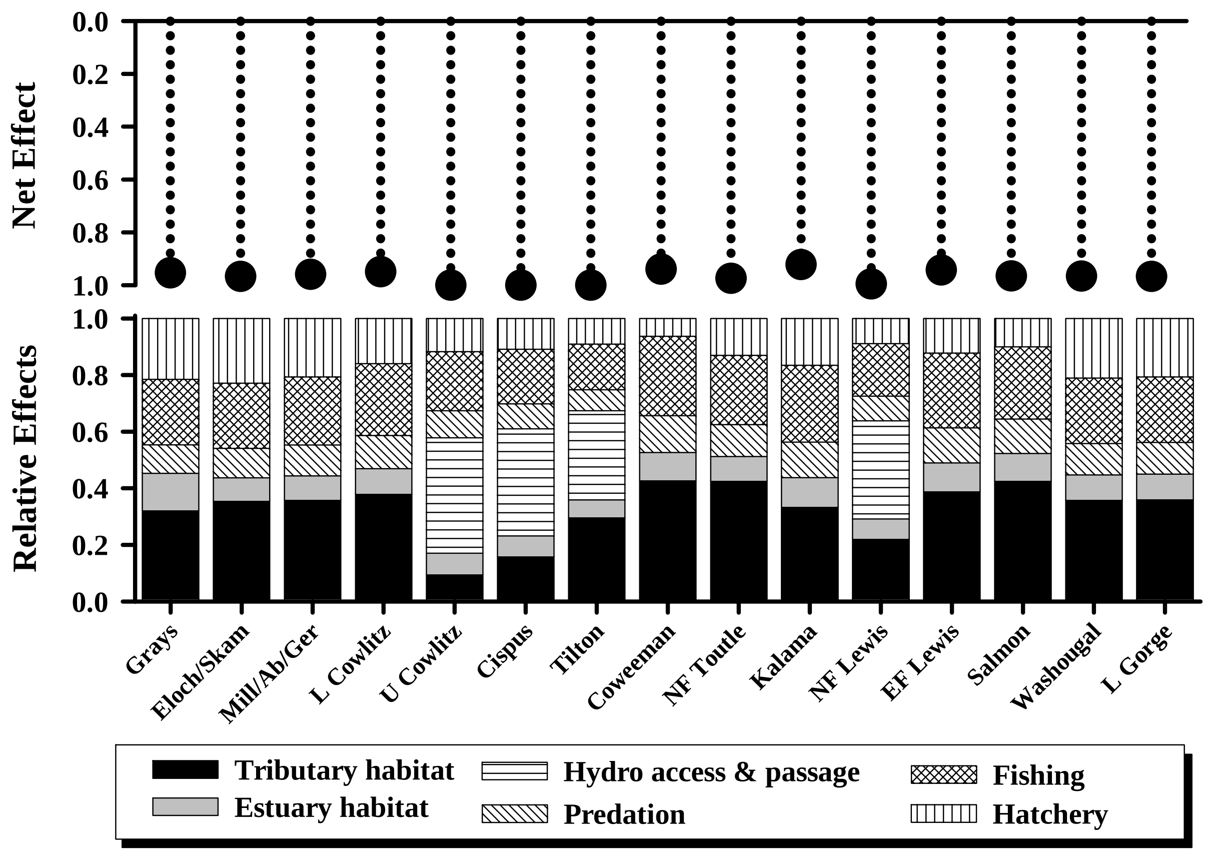

Figure 5-17. **Net effect and relative contribution of potentially manageable impact factors on coho salmon in Washington lower Columbia River subbasins. Net effect is the approximate reduction from historical fish numbers as a result of manageable factors included in this analysis.**

Table 5-19. Coho salmon impact factors and index.

	Grays	E/S	M/A/G	L Cowlitz	U Cowlitz	Cispus	Tilton	Cowee-man	NF Toutle	SF Toutle	Kalama	NF Lewis	EF Lewis	Salmon	Wash.	L Gorge	U Gorge
Inputs																	
Neq Current	1,239	2,396	2,045	4,144	11,039	3,752	261	1,873	3,860	3,860	484	2,367	1,066	772	824	57	418
Neq PFC	3,773	5,787	3,010	15,655	23,633	5,351	3,233	6,225	27,027	27,027	1,033	4,771	3,306	4,621	3,362	123	898
Neq PFC+	4,593	7,045	3,664	19,058	28,770	6,612	4,011	7,579	32,901	32,901	1,282	5,917	4,101	5,731	4,170	153	1,114
Neq Historical	5,289	13,885	10,621	21,458	17,654	8,029	5,599	10,267	41,912	41,912	1,620	7,474	5,309	6,532	4,860	347	1,174
Hydro habitat loss	0.000	0.000	0.000	0.000	1.000	1.000	1.000	0.000	0.000	0.000	0.000	0.952	0.000	0.000	0.000	0.000	0.010
Dam passage mort. (juv.)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.100
Dam passage mort. (ad.)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.050
Predation mort. (juv.)	0.200	0.206	0.209	0.211	0.211	0.211	0.211	0.211	0.211	0.211	0.212	0.215	0.215	0.220	0.220	0.223	0.251
Predation mort. (ad.)	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030
Fishing	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510
Hatchery fraction	0.95	0.99	0.88	0.85	0.96	0.96	0.96	0.38	0.86	0.87	0.98	0.69	0.78	0.67	0.91	0.91	0.86
Hatchery category	3	3	3	2	2	2	2	2	2	2	2	2	2	2	3	3	3
Hatchery fitness	0.5	0.5	0.5	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.5	0.5	0.5
Other hatchery species	190,000	1.1 mil	0	5.3mil	0	0	0	20,000	25,000	825,000	1.4 mil	3.0 mil	115,000	20,000	620,000	0	1.4 mil
Impacts (p reduction)																	
Tributary habitat	0.715	0.790	0.766	0.765	0.239	0.423	0.942	0.778	0.888	0.888	0.629	0.607	0.751	0.853	0.790	0.798	0.558
Estuary habitat	0.287	0.179	0.179	0.179	0.179	0.191	0.194	0.179	0.179	0.179	0.194	0.194	0.194	0.194	0.194	0.194	0.194
Hydro habitat loss	0.000	0.000	0.000	0.000	1.000	1.000	1.000	0.000	0.000	0.000	0.000	0.952	0.000	0.000	0.000	0.000	0.010
Dam passage	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.145
Predation	0.224	0.230	0.233	0.235	0.235	0.235	0.235	0.235	0.235	0.235	0.236	0.239	0.239	0.243	0.243	0.246	0.273
Fishing	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510	0.510
Hatchery fitness	0.475	0.495	0.440	0.255	0.288	0.288	0.288	0.114	0.258	0.261	0.294	0.207	0.234	0.201	0.455	0.455	0.430
Hatchery inter-species	0.002	0.013	0.000	0.066	0.000	0.000	0.000	0.000	0.000	0.010	0.017	0.038	0.001	0.000	0.008	0.000	0.018
Total (unconditional)	2.213	2.216	2.127	2.010	2.450	2.646	3.169	1.815	2.069	2.082	1.880	2.747	1.929	2.002	2.200	2.203	2.138
Impact index																	
Tributary habitat	0.323	0.356	0.360	0.381	0.097	0.160	0.297	0.429	0.429	0.426	0.335	0.221	0.389	0.426	0.359	0.362	0.261
Estuary habitat	0.130	0.081	0.084	0.089	0.073	0.072	0.061	0.098	0.086	0.086	0.103	0.071	0.100	0.097	0.088	0.088	0.091
Hydro access/passage	0.000	0.000	0.000	0.000	0.408	0.378	0.316	0.000	0.000	0.000	0.000	0.347	0.000	0.000	0.000	0.000	0.072
Predation	0.101	0.104	0.109	0.117	0.096	0.089	0.074	0.129	0.113	0.113	0.125	0.087	0.124	0.122	0.111	0.112	0.128
Fishing	0.230	0.230	0.240	0.254	0.208	0.193	0.161	0.281	0.246	0.245	0.271	0.186	0.264	0.255	0.232	0.232	0.239
Hatchery	0.216	0.229	0.207	0.160	0.118	0.109	0.091	0.063	0.125	0.130	0.166	0.089	0.122	0.101	0.210	0.207	0.209

5.4.4.2 Fisheries

Fishery impact rates on wild lower Columbia River coho averaged 53% at listing and have been reduced to 22% at present. The primary fisheries targeting Columbia River hatchery coho salmon occur in West Coast ocean and Columbia River mainstem fisheries (Figure 5-18). Hatchery-selective harvest regulations or time and area strategies have been widely implemented to limit impacts to wild coho. The exploitation rate of coho prior to the 1990s fluctuated from approximately 60 to 90%. Exploitation of wild and hatchery coho decreased significantly during the 1990s. The exploitation rate of wild coho has continued to decrease to current levels, while the exploitation of hatchery coho has remained similar to the 1990s rate.

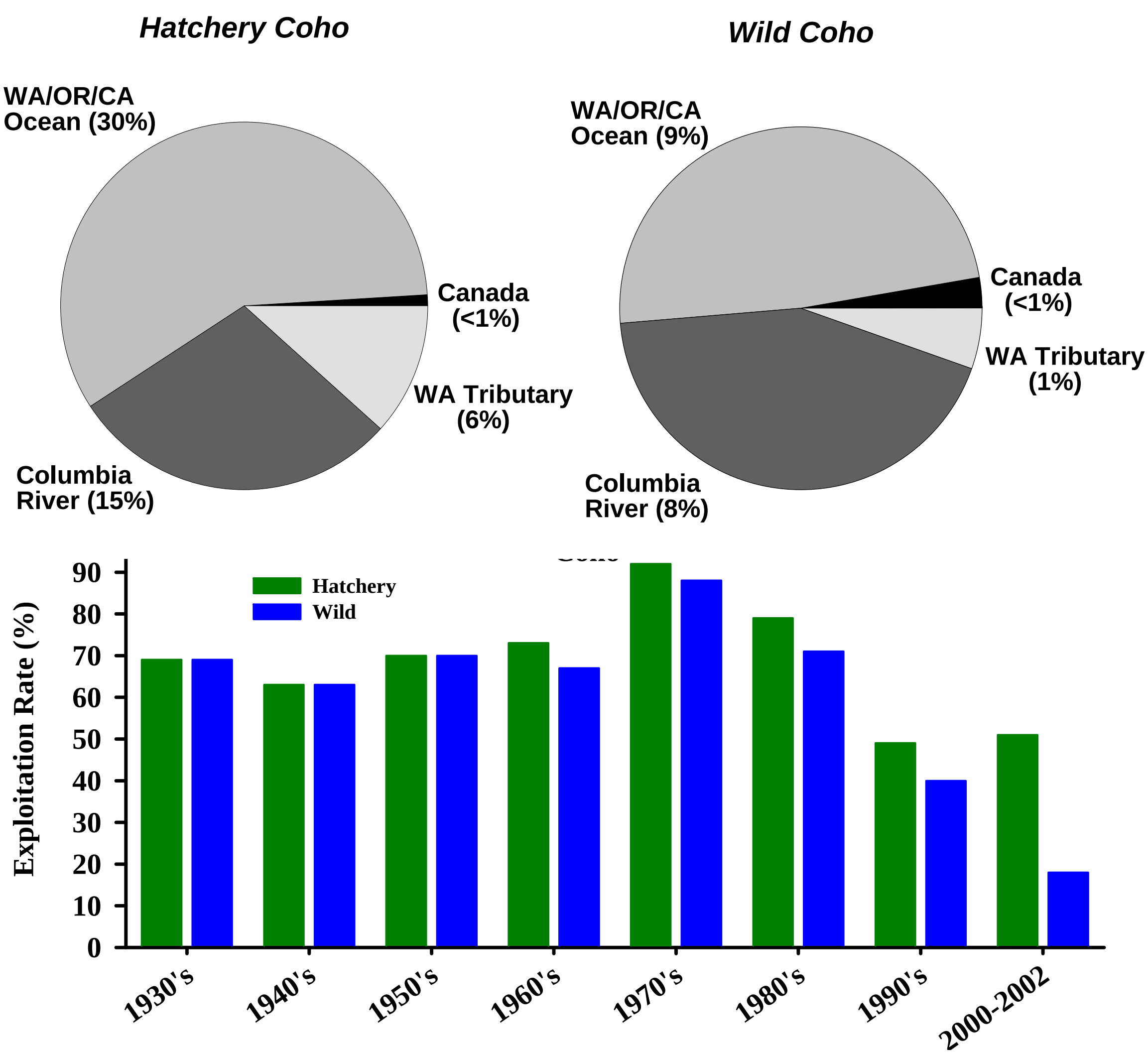

Figure 5-18. Approximate coho salmon fishery exploitation rates over time and allocation of current exploitation rates among fisheries.

5.4.4.3 Hatcheries

Hatchery influence continues to be significant for most Washington lower Columbia coho populations (Table 5-20). Most coho hatchery programs are intended to mitigate the loss of natural coho production by providing fish for harvest opportunity. Hatchery releases of coho salmon smolts range from 0 to 3.2 million in subbasins where wild coho populations occurred historically. The average adult hatchery fraction for all Washington lower Columbia coho populations was 84%. Average hatchery fraction varied slightly by strata: 90%, 82%, and 85% for the Coast, Cascade, and Gorge strata, respectively. Reintroduction attempts in the upper Cowlitz, Cispus, and Tilton basins have relied almost entirely on hatchery stock. Current coho salmon hatchery broodstock are primarily derived from local populations and moderately affected by hatchery practices (category 2) or derived from other populations within the same ESU (category 3).

The indexed potential for negative impacts of hatchery spawners on wild population fitness was estimated to range from 11-50%. However, the high incidence of hatchery spawners suggests that the fitness of natural and hatchery fish is now probably quite similar and natural populations could collapse without continued hatchery subsidy under current habitat conditions. In general, the highest potential impacts occur in basins where hatcheries have used broodstock from outside of the local basin; this practice has recently occurred in hatchery coho programs in the Coast and Gorge strata. The lowest potential impacts appear to occur in basins that do not maintain active coho salmon hatchery programs and the hatchery programs in adjacent basins utilized local populations for broodstock (e.g. Coweeman, EF Lewis, Salmon).

Inter-specific hatchery predation impacts on juvenile fall chinook range from 0% in basins without significant releases of coho, steelhead or spring chinook to a high of 7% in the Cowlitz basins where large hatchery programs are underway.

Table 5-20. Presumed reductions in wild population fitness as a result of natural hatchery spawners and survival as a result of interactions with other hatchery species for Washington lower Columbia River coho populations.

Population	Annual releases[b]	Hatchery fraction	Fitness category	Assumed fitness	Fitness impact	Interacting releases[m]	Interspecies impact
Coast Fall							
Chinook/ Grays	602,500[c]	0.95	3	0.5	0.48	190,000	0.00
Eloch/Skam	930,000[d]	0.99	3	0.5	0.50	1,050,000	0.01
Mill/Aber/Germ	0[e]	0.88	3	0.5	0.44	0	0
Cascade							
Lower Cowlitz	3,200,000[f]	0.85	2	0.7	0.26	5,319,500	0.07
Upper Cowlitz	0[h]	0.96	2	0.7	0.29	0	0
Coweeman	0[e]	0.38	2	0.7	0.11	20,000	0
SF Toutle	0[e]	0.87	2	0.7	0.26	25,000	0
NF Toutle	800,000[g]	0.86	2	0.7	0.26	825,000	0.01
Cispus	0[h]	0.96	2	0.7	0.29	0	0
Tilton	0[h]	0.96	2	0.7	0.29	0	0
Kalama	700,000[i]	0.98	2	0.7	0.29	1,380,000	0.02
NF Lewis	1,695,000[j]	0.69	2	0.7	0.21	3,070,000	0.04
EF Lewis	0[e]	0.78	2	0.7	0.23	115,000	0.00
Salmon	0[e]	0.67	2	0.7	0.20	20,000	0
Washougal	500,000[k]	0.91	3	0.5	0.46	620,000	0.01
Gorge							
L Gorge	0[e]	0.91	3	0.5	0.46	0	0
U. Gorge (LWS)	1,000,000[l]	0.86	3	0.5	0.43	1,420,000	0.02
White Salmon	0[e]	0.79	3	0.5	0.40	0	0

[a] *The TRT has not assigned genetic legacy designations to lower Columbia River coho populations.*

[b] *Annual release goals.*

[c] *Comprised of early coho (type S) released in the Grays, Deep, and Chinook Rivers from the Grays River and Sea Resources Hatcheries.*

[d] *Elokomin Hatchery goals include 418,000 early coho (type S) and 512,000 late coho (type N).*

[e] *Hatchery coho salmon are no longer released in the basin; hatchery fish in these basins appear to be strays from other programs.*

[f] *The Lower Cowlitz coho hatchery program is composed of late coho (type N). One goal of the late stock coho salmon hatchery program is to provide restocking of the upper Cowlitz basin. Reintroduction efforts have been challenged in passing juvenile production through the system.*

[g] *Comprised of early coho (type S) released in the NF Toutle and Green Rivers from the NF Toutle Hatchery.*

[h] *Hatchery coho (predominately late coho type N) fry and adults have been released since 1997 and 1998, respectively, into the upper Cowlitz and Cispus Rivers. Outmigrating juvenile coho are collected and transported around the Cowlitz Falls Dam; collection efficiencies have ranged from 17-45%. Recent efforts have also released adults into the Tilton River basin; any juveniles produced in the Tilton need to be collected at Mayfield Dam.*

[i] *The Fallert Creek Hatchery goal is 350,000 early coho (type S); the Kalama Falls Hatchery goal is 350,000 late coho (type N).*

[j] *Lewis River Hatchery goals include 880,000 early coho (type S) and 815,000 late coho (type N); fish are released in the lower Lewis River mainstem. Various possible salmonid reintroduction scenarios are currently being evaluated during the re-licensing process for the hydroelectric facilities on the Lewis River; the existing hatchery programs could become an integral part of any successful reintroduction program.*

[k] *The Washougal River Hatchery releases late coho salmon (type N); broodstock is normally derived from Washougal or Lewis River hatchery returns.*

[l] *The Little White Salmon hatchery goal is composed of early coho (type S).*

[m] *Includes steelhead, coho, and spring chinook.*

5.4.4.4 Stream Habitat

EDT analyses suggest that stream degradation has substantially reduced the habitat potential for coho in all Washington lower Columbia River subbasins where analyses have been completed (Figure 5-19). Declines in habitat quality and quantity for coho salmon have reduced current productivity and equilibrium population sizes to 10-60% of the historical template. Substantial stream habitat improvements would be necessary to reach optimum conditions (i.e. PFC) for coho salmon in any subbasin. Restoration of optimum habitat quality would be expected to increase habitat capacity by 1,000 to 23,000 adult coho per subbasin, based on preliminary planning ranges.

Coho salmon rely on the middle mainstem to upper stream reaches where a lack of habitat diversity, sedimentation, and flow consistently limit habitat suitability. More detailed descriptions of stream habitat conditions and effects on fish in each subbasin may be found in Volume II of the Technical Foundation.

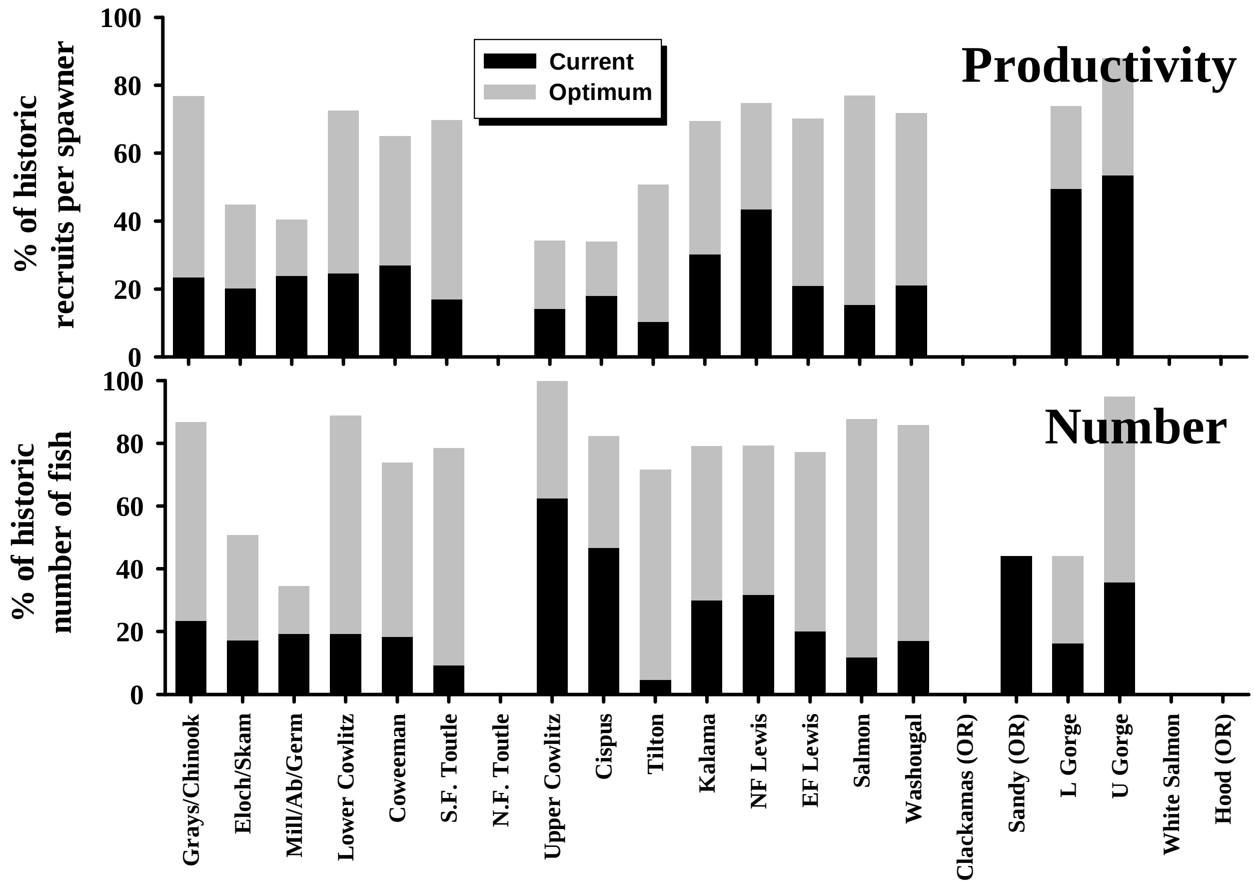

Figure 5-19. Current, optimal, and historical subbasin productivity and capacity inferred for coho salmon from stream reach habitat conditions using EDT.

5.4.4.5 Dams

At present, there are no EDT assessments that quantify the amount of historical coho habitat that has been blocked or inundated by dam construction within specific basins. Steelhead results give some idea of the scale of effect although coho utilize many more downstream tributary areas than do steelhead. Similarly, coho dam passage rate data are sparse. If similar to steelhead, passage mortality at Bonneville Dam would be assumed to average 10% for juveniles and an additional 5% for adults based on a synthesis of the available literature. Coho salmon generally spawn and rear in headwater and upper mainstem reaches of subbasins and are less subject to hydropower effects on downstream habitats than are chum and fall chinook.

5.4.4.6 Mainstem and Estuary Habitat

Mainstem and estuary habitat impacts were estimated to account for approximately a 10-20% reduction in productivity of coho salmon, if similar to steelhead. Coho salmon migrate through mainstem and estuary areas soon after emigration from tributary streams. Residence time in estuary and mainstem habitats is usually relatively brief, but smoltification and transition from fresh to salt water is a critical life stage.

5.4.4.7 Predation

Potentially manageable predation mortality was assumed to average 20% to 25% depending on travel distance from the subbasin to the ocean. Pikeminnow and tern management is projected to reduce salmonid predation by approximately 50%. Tern predation was almost entirely an artifact of recently established colonies on dredge spoil islands in the estuary but the current rate (9%) is less than half that observed prior to downstream translocation of part of the Rice Island tern colony (20%). Pikeminnow predation was greatest for populations that originate in Bonneville Reservoir tributaries (5%), pass the pikeminnow gauntlet in Bonneville Dam forebay and tailrace, and travel the entire 145-mile length from Bonneville to the Estuary. Predation rates by seals and sea lions on adult coho salmon added an assumed 3% mortality.

5.4.5 Summary Assessment

1. Human activities including fishing, hatchery operation, alteration of stream, river, and estuary habitats, hydropower development and operation, and potentially manageable predation have collectively reduced productivity of coho salmon populations to 0-8% of historical levels. Recovery efforts will require significant improvements in multiple areas because no single factor accounts for the majority of the reduction in fish numbers.

2. Implementation of selective fishery regulations in U.S. ocean and Columbia River fisheries has reduced impacts on wild coho salmon by over half. Additional reductions would require widespread changes in U.S. ocean and Columbia River fisheries. Because Lower Columbia wild coho salmon comprise only a small portion of the catch in many fisheries, additional constraints for their protection will forgo harvest of larger numbers from healthy wild and especially hatchery populations. Intensive fishery management processes provide significant opportunities for limiting fishing risks by tailoring annual harvests to fish availability.

3. Reduced productivity of wild populations as a result of interbreeding with potentially less-fit hatchery fish is among the most significant of hatchery concerns for wild stock recovery although these negative effects are at least partially offset by the demographic benefits of additional spawners. Potential negative impacts increase with the proportion of hatchery spawners and the genetic and phenotypic disparity between wild and hatchery fish. Potential fitness impacts among Washington lower Columbia coho salmon populations range from 11 to 50%. Potential impacts are greatest in the Coast and Gorge strata populations where out-of-basin stocks continue to be used for broodstock. Inter-specific hatchery predation impacts on juvenile fall chinook range from 0% in basins without significant releases of coho, steelhead or spring chinook to a high of 7% in the Cowlitz basins where large hatchery programs are underway.

4. The current conditions of stream habitats significantly limit coho salmon in all Washington lower Columbia River subbasins where EDT analyses have been completed. Substantial stream habitat improvements would be necessary to reach optimum conditions (i.e. PFC) in any subbasin. The significance of stream habitat suggests that recovery may not be feasible without substantial improvements in tributary habitat quantity and quality.

5. Estuary and mainstem habitats are important to coho salmon life history with assumed habitat impacts of 10-20%.

6. Hydropower development in the Cowlitz and Lewis have blocked 50-95% of the historical coho salmon habitat, based on data for steelhead. Mainstem dam passage affects upper Gorge populations although passage success for coho salmon may be as high as steelhead, which tends to be greater than other salmon species.

5.5 Steelhead

5.5.1 Current Viability

The Willamette/Lower Columbia Technical Recovery Team has identified 23 historical populations of steelhead in the Columbia River ESU (Figure 5-20, Figure 5-21). This ESU includes all populations from the Cowlitz River upstream to Hood River. Washington accounts for 5 of 6 summer and 14 of 17 winter run steelhead populations in this ESU. Three additional winter run populations of the unlisted Washington Coast ESU occur in lower Columbia subbasins included in this planning process.

Current steelhead population sizes and productivities are only a small fraction of historical numbers inferred with EDT from assumed pre-development habitat conditions (Table 5-21). EDT estimates of equilibrium numbers range from 60 to 2,300 under current conditions. Recent population estimates were typically much less than EDT estimates in part because of poor ocean survival conditions. Recent numbers have averaged fewer than 300 naturally produced fish in 6 of 9 Washington winter steelhead populations and 1 of 4 Washington summer steelhead populations where data is available. Recent natural escapements of Washington lower Columbia steelhead did not exceed an average of 1,000 fish in any basin. Recent average escapements have also been typically less than EDT equilibrium numbers based on current stream habitat conditions, primarily because of recent poor ocean survival cycles. Historical steelhead population sizes in Washington ranged from 300 to 7,400 based on EDT estimates. Back-of-envelope estimates by NOAA Fisheries yielded historical steelhead population sizes in Washington of 2,000 to 29,000 based on presumed Columbia River run totals and subbasin habitat quantity. BOE estimates are typically greater than EDT estimates. We conservatively assume EDT estimates to be more accurate because they consider both habitat quantity and quality whereas the BOE estimates include only habitat quantity. EDT estimates are also independent of assumed total Columbia River run size and lower basin proportions upon which the BOEs are based.

Based on interim TRT population criteria, 100-year persistence probabilities are very low or already extinct (0-39%) for 2 populations, low (40-74%) for 21 populations, and moderate (75-94%) for 3 populations (Table 5-21). All strata currently fall short of integrated TRT recovery criteria which specify an average persistence probability greater than 2.25 with at least 2 populations at high (>3.0) for each strata.

Population trends and/or extinction risks have been estimated for 12 steelhead populations based on abundance time series data and two different models (NOAA Fisheries, unpublished data). Population trends were negative for 7 of 12 populations (Table 5-22). Extinction risks averaged for both models were 80% or greater for 7 of 9 populations. Noteworthy exceptions include NF Toutle winter steelhead that are recovering from volcanic effects and Washougal summer steelhead. However, model-derived estimates appear overly pessimistic because of the limited time period of available data coincident with population declines following the ocean regime shift in the late 1970s as well as very large post 1983-84 El Niño returns which occur in the front half of most available time series. We assume that future estimates revised to consider cyclical patterns in ocean survival like those that have produced recent large returns will project much lower extinction risks consistent with persistence scores based on specific population attributes. Differences between score-derived persistence probabilities and trend-derived extinction risks reflect different assumptions and uncertainties in these methods.

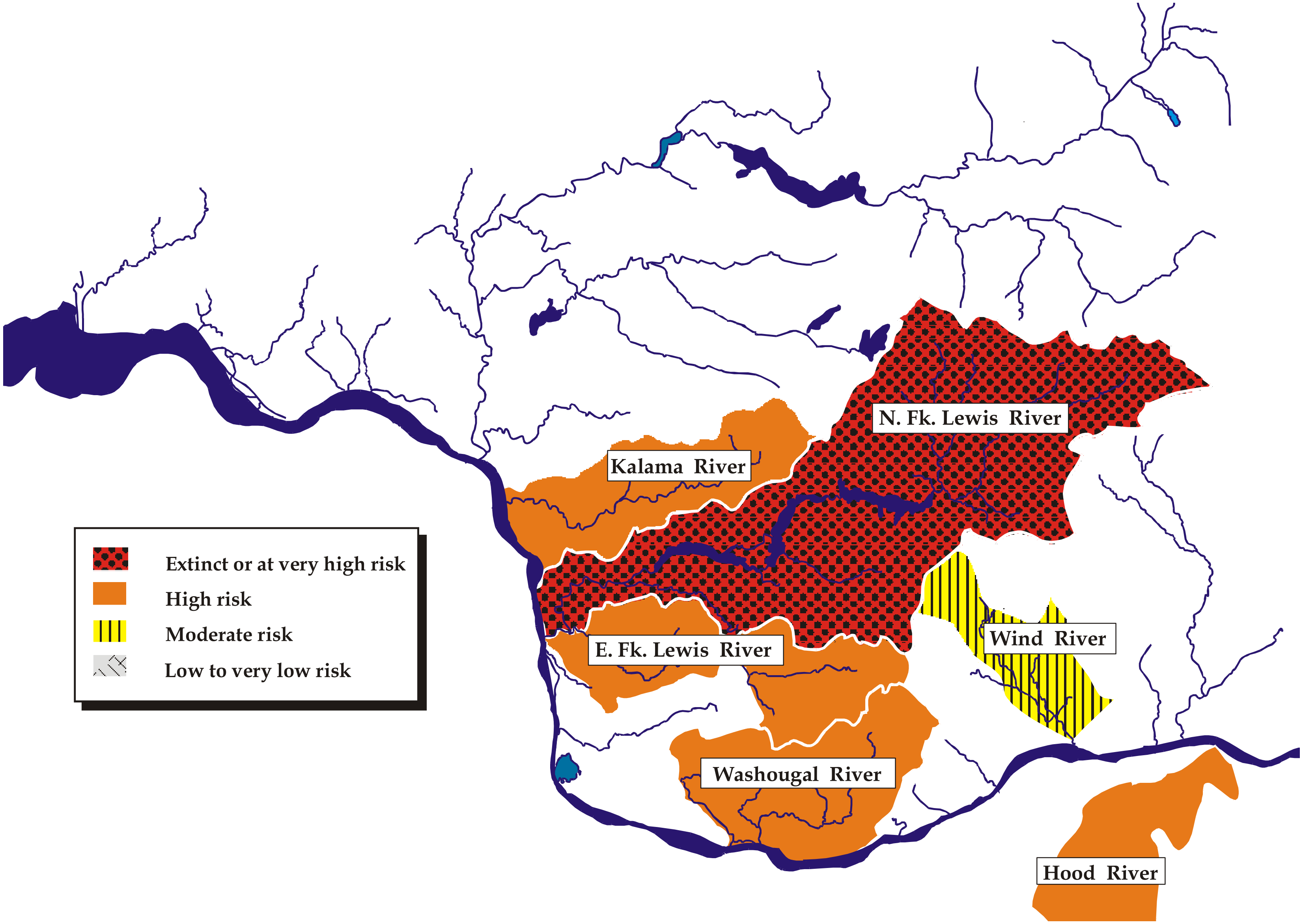

Figure 5-20. Distribution of historical summer steelhead populations among lower Columbia River subbasins. Extinction risks are based on viability scores rather than modeled risks.

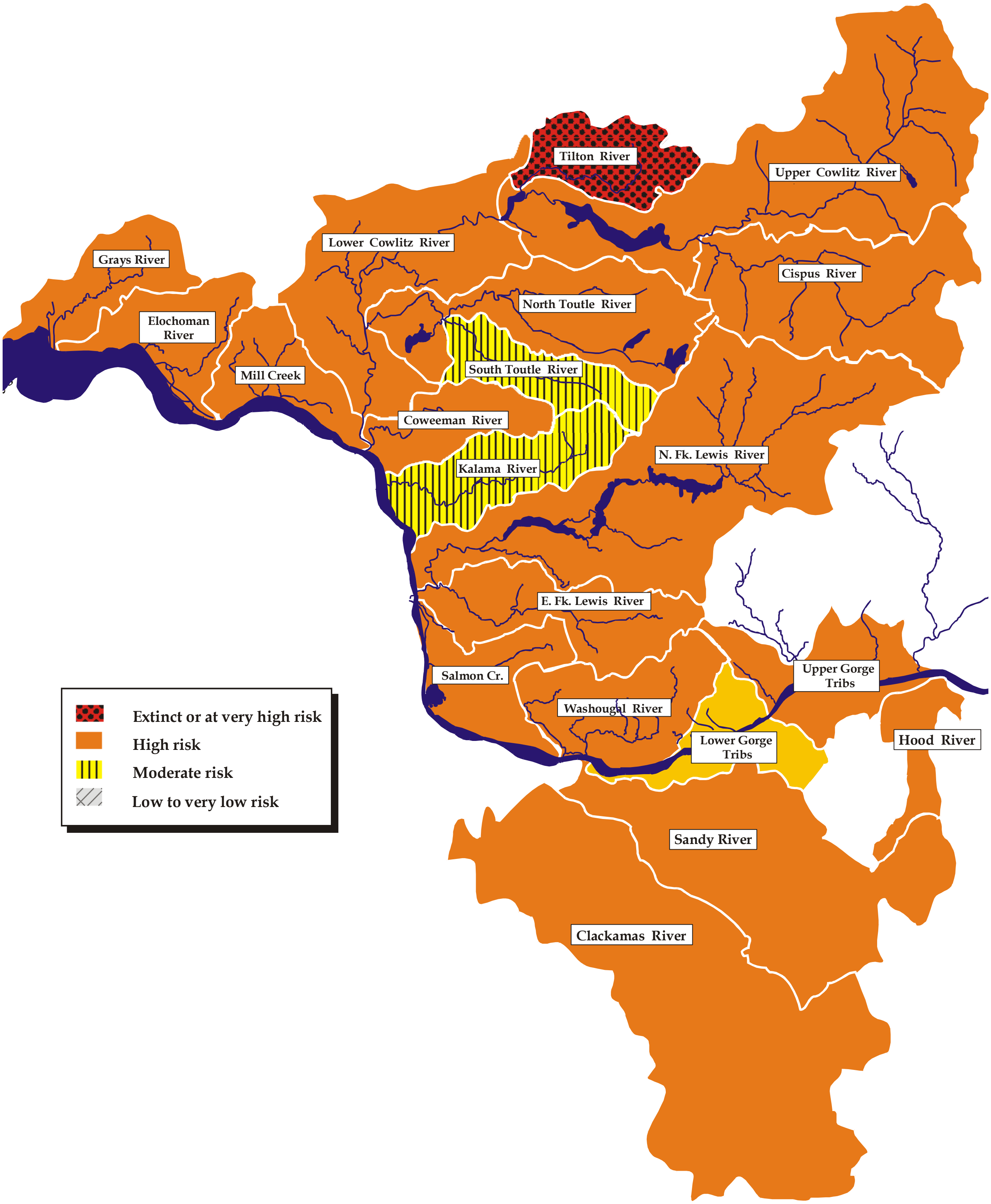

Figure 5-21. Distribution of historical winter steelhead populations among lower Columbia River subbasins (Myers et al. 2002). Extinction risks are based on viability scores rather than modeled risks.

Table 5-21. Numbers and productivity of lower Columbia River steelhead populations.

				EDT Equilibrium Population Size				BOE[8]	EDT Productivity			
Population	**Leg[1]**	**Core[2]**	**4-yr[3]**	**Current[4]**	**PFC[5]**	**PFC+[6]**	**Hist.[7]**	**Hist.**	**Current[4]**	**PFC[5]**	**PFC+[6]**	**Hist[7]**
Coast Winter												
Grays/Chinook				1,201	1,885	2,307	4,549		4.4	13.5	16.6	35.9
Eloch/Skam				541	842	1,031	1,365		4.3	10.5	12.9	20.1
Mill/Aber/Germ				897	1,191	1,458	1,966		5.2	9.3	11.4	19.3
Cascade Winter												
Lower Cowlitz				198	1,352	1,517	1,938	28,552	2.3	10.0	11.2	26.1
Coweeman			228	653	1,017	1,197	2,850	7,065	3.9	9.0	10.5	28.2
Toutle SF			453	670	1,673	1,884	4,192	4,521	3.3	12.0	13.5	34.7
Toutle NF		1	176	659	3,089	3,480	7,444	15,558	2.9	13.4	15.1	36.6
Upper Cowlitz	1	1	0	855	1,402	1,625	1,973	16,536	4.8	9.3	10.6	15.1
Cispus	1	1	0	624	1,001	1,159	1,504	2,805	4.2	7.4	8.5	13.1
Tilton			0	219	1,093	1,266	1,741	5,812	2.3	9.7	11.0	16.5
Kalama			541	445	614	703	1,014	7,769	4.0	9.2	10.6	17.2
Lewis NF		1		2,320	3,038	3,391	6,254	24,110	7.6	14.5	16.1	24.2
Lewis EF			77	631	1,109	1,278	2,901	10,431	3.7	10.4	11.9	29.9
Salmon				64	223	257	560	8,121	2.4	13.9	16.0	36.4
Washougal			421	500	909	1,037	2,223	9,530	3.8	12.6	14.4	33.8
Clackamas (OR)		1	277	--		--		29,352				
Sandy (OR)		1	589	--		--		18,219				
Gorge Winter												
L Gorge (Hardy				244	270	312	642	3,797	15.7	19.0	22.0	45.8
U. Gorge (Wind)				70	123	138	313	2,720	3.5	7.7	8.6	20.8
Hood (OR)	1	1	436	--		--		5,102				
Cascade Summer												
Kalama		1	291	788	953	996	1,264	6,711	4.5	8.2	8.5	13.2
Lewis NF								20,825				
Lewis EF	1		463	187	338	354	933	9,009	2.6	5.3	5.5	17.4
Washougal	1	1	136	639	876	921	2,289	8,232	4.3	6.7	7.1	20.5
Gorge Summer												
Wind		1	391	1,516	1,763	1,936	5,099	1,809	4.5	6.2	6.8	18.0
Hood (OR)			154	--		--		3,414				

[1] *Genetic Legacy designation by the Technical Recovery Team. Genetic legacy populations represent unique life histories or are relatively unchanged by hatchery influences.*
[2] *Core population designation by Technical Recovery Team. Core populations were the largest historical populations and were key to metapopulation processes*
[3] *Recent 4-year average natural spawning escapements upon which PCC numbers are based (typically1997-2000 return years). Spawning escapements in 2002 and 2003 have generally been substantially greater than in the preceding years as these runs encountered much improved ocean survival conditions.*
[4] *Current number inferred with EDT from estimated and assumed habitat conditions.*
[5] *Estimate if habitat conditions are restored to "properly functioning" standards defined by NOAA Fisheries under current estuary conditions.*
[6] *Estimate if habitat conditions are restored to "properly functioning" standards defined by NOAA Fisheries and predevelopment estuary conditions are restored.*
[7] *Pre-development estimate inferred with EDT from assumed historical habitat conditions.*
[8] *Back of envelope estimates of historical population sizes inferred from stream miles accessible and assumed total Columbia River run (NOAA Fisheries).*

Table 5-22. Estimated viability of lower Columbia River steelhead.

Population	Leg[1]	Core[2]	Population Persistence Scores A/P[3]	J[4]	S[5]	D[6]	H[7]	Net[8]	Prob.[9]	Data Years[10]	Trend[11]	Extinction risk Model 1[12]	Model 2[13]
Coast Winter													
Grays/Chinook			1.5	na	4	2.5	2	1.8	70%				
Eloch/Skam			1	na	4	2	2	1.5	60%				
Mill/Aber/Germ			1.5	2	4	2	2	1.7	60%				
Cascade Winter													
Lower Cowlitz			1	na	2	2	1.5	1.3	50%				
Coweeman			1.5	na	4	2.5	1.75	1.8	70%	1987-2002	0.82		
Toutle SF			2	na	4	3	2	2.1	80%	1984-2002	0.94	0.98	0.85
Toutle NF		1	2	na	3	3	1.75	1.8	70%	1989-2002	1.06	0.0	0.03
Upper Cowlitz	1	1	1	2	2	2	1.5	1.0	40%				
Cispus	1	1	1	2	2	2	1.5	1.0	40%				
Tilton			0.5	2	2	2	1.5	0.8	30%				
Kalama			3	2	4	3.5	2.5	2.3	90%	1977-2002	0.96	0.89	0.75
Lewis NF		1	1.5	2	2	2	2	1.3	50%				
Lewis SF			1.5	1	4	2.5	2	1.7	60%	1985-1994	0.84	1.00	0.97
Salmon			1	na	4	2	1	1.4	50%				
Washougal			1.5	na	4	2.5	2	1.6	60%	1991-2002	1.12		
Clackamas (OR)		1	--	--	--	--	--	1.6	60%	1958-1998	0.96	0.84	0.85
Sandy (OR)		1	--	--	--	--	--	1.7	60%	1978-1998	0.87	1.00	0.99
Gorge Winter													
L Gorge (Hardy only)			1.5	na	4	2.5	2	1.7	60%				
U. Gorge (Wind only data)			1.5	2	2	2.5	2	1.5	60%				
Hood (OR)	1	1	--	--	--	--	--	1.8	70%				
Cascade Summer													
Kalama		1	1.5	2	4	2.5	2.5	1.9	70%	1977-2003	1.00	1.00	0.99
Lewis NF			0	na	0	0	2	0.5	20%				
Lewis EF	1		1.5	1	4	2.5	2	1.8	70%	1996-2003	1.21		
Washougal	1	1	1.5	na	4	3	2	1.9	70%	1986-2003	1.00	0.48	0.72
Gorge Summer													
Wind		1	2	2.5	4	3	3	2.3	90%	1989-2003	0.96	0.99	0.78
Hood (OR)			--	--	--	--	--	1.4	50%				

[1] *Genetic Legacy designation by the Technical Recovery Team, relatively unchanged by hatchery influences or represent unique life histories).*
[2] *Core population designation by Technical Recovery Team, among the largest historical populations and key to metapopulation processes*

[3] *Abundance and productivity rating by LCFRB biologists based on TRT criteria.*
[4] *Juvenile outmigration number rating by LCFRB biologists based on TRT criteria.*
[5] *Spatial structure rating by LCFRB biologists based on TRT criteria.*
[6] *Diversity rating by LCFRB biologists based on TRT criteria.*
[7] *Habitat rating by LCFRB biologists based on TRT criteria.*
[8] *Weighted average of population attribute scores. LCFRB and TRT scores are averaged.*
[9] *Persistence probability corresponding to net population score (interpolated from corresponding persistence ranges).*
[10] *Available abundance data time series upon which trend and extinction risk analyses by NOAA Fisheries were based.*
[11] *Trend slope estimated by NOAA Fisheries based on abundance time series (median annual growth rate or λ).*
[12] *Probability of extinction in 100 years (PE 100) estimated from abundance time series by NOAA Fisheries using Dennis-Holmes model.*
[13] *Population projection interval extinction risks (PPI E) estimated from abundance time series by NOAA Fisheries using Population Change Criteria model.*

5.5.2 Recovery Planning Ranges

Minimum abundance planning range values vary among populations from 100 to 1,800. Populations with larger current numbers generally require greater minimum numbers to reach viable levels according to Population Change Criteria. Maximum planning range numbers range from 100 to 3,500 based on subbasin potentials estimated with EDT for Properly Functioning Conditions. Consistent with their current threatened population status, recent natural spawning escapements have averaged less than the low viability bound of the planning range for all populations except for East Fork Lewis summer steelhead.

Substantial improvements in productivity are required in most populations to reach viable levels. Existing steelhead populations were estimated to require a 5% to 33% improvement in productivity to reach a level of high viability.

5.5.3 Population Significance

The population significance index provides a simple sorting device to group populations in each strata based on current viability, core potential and genetic legacy considerations (Table 5-24).

Based on this index, Grays and Mill/Abernathy/Germany winter steelhead populations in the unlisted Coast strata may be categorized in a middle group with the Elochoman/Skamokawa populations slightly lower. In the Cascade stratum, Upper Cowlitz, Cispus, and North Toutle, populations sort to the top by virtue of their current viability, genetic legacy designations, or large historical potential. North Fork Lewis, South Toutle, Kalama, EF Lewis, and Coweeman rank in a middle tier. Lower Cowlitz, Washougal, Salmon, and Tilton populations sort to the bottom rank. The two Gorge stratum winter steelhead populations are similar in their significance.

Cascade summer steelhead population include the Washougal and East Fork Lewis in the top tier by virtue of their legacy status. Kalama summer steelhead fall in a middle tier distinguishable from North Fork Lewis in a third tier. Only one Gorge summer steelhead population occurs in Washington.

Table 5-23. Population abundance and productivity planning ranges for lower Columbia River steelhead populations.

	Recent	Abundance range		Current	Current	Productivity range		Productivity Improvement Increments			
Population	Avg. no.	Viable	Potential	Viability	Prod.	Viable	Potential	Contrib	High	V high	Max
Coast Winter											
Grays/Chinook	150	600	2,300	**Low**	0.93	1.09	2.04	8%	17%	67%	118%
Eloch/Skam	150	600	1,000	**Low**	0.93	1.09	1.94	8%	17%	62%	107%
Mill/Ab/Germ	150	600	1,500	**Low**	0.93	1.09	1.60	8%	17%	44%	72%
Cascade Winter											
Lower Cowlitz		600	1,500	**Low**	0.93	1.09	4.26	8%	17%	186%	356%
Coweeman	228	800	1,200	**Low**	0.82	1.09	1.86	17%	33%	80%	127%
S.F. Toutle	453	1,400	1,900	**Med**	0.94	1.07	2.26	7%	14%	78%	142%
N.F. Toutle	176	700	3,500	**Low**	1.06	1.09	3.06	5%	9%	99%	188%
Upper Cowlitz	0	600	1,600	**V Low**	0.00	1.09	2.30	--	--	--	--
Cispus	0	600	1,200	**V Low**	0.00	1.09	2.08	--	--	--	--
Tilton	0	600	1,300	**V Low**	0.00	1.09	2.33	--	--	--	--
Kalama	541	600	700	**Med**	0.96	1.00	1.88	2%	5%	50%	96%
NF Lewis		600	3,400	**Low**	0.93	1.09	38.57	8%	17%	2021%	4025%
EF Lewis	77	600	1,300	**Low**	0.84	1.09	2.74	15%	30%	128%	226%
Salmon		600	1,200	**Low**	0.00	1.09	5.17	8%	17%	235%	453%
Washougal	421	600	1,000	**Low**	1.12	1.09	3.85	4%	9%	127%	244%
Clackamas (OR)	277	1,000	2,000	**Low**	--	--	--	--	--	--	--
Sandy (OR)	589	1,800	3,600	**Low**	--	--	--	--	--	--	--
Gorge Winter											
L Gorge (HHD)		200	300	**Low**	0.00	1.09	1.17	8%	17%	21%	25%
U Gorge *(Wind)*		100	100	**Low**	0.00	1.09	2.12	8%	17%	72%	127%
Hood (OR)	436	1,400	2,800	**Low**	--	--	--	--	--	--	--
Cascade Summer											
Kalama	291	700	1,000	**Low**	1.00	1.08	1.65	4%	8%	36%	65%
N.F. Lewis		600	1,200	**V Low**	--	--	--	--	--	--	--
E.F. Lewis	463	200	400	**Low**	1.21	1.09	6.83	5%	9%	238%	467%
Washougal	136	500	900	**Low**	1.00	1.09	1.82	5%	9%	45%	82%
Gorge Summer											
Wind	391	1,200	1,900	**Med**	0.96	1.00	1.86	2%	4%	49%	94%
Hood (OR)	154	600	1,200	**Low**	--	--	--	--	--	--	--

1. *Recent average numbers are observed 4-year averages or assumed natural spawning escapements. Data typically is through year 2000.*
2. *Abundance planning range refer to average equilibrium escapement numbers at viability as defined by NOAA's Population Change Criteria and potential as defined by WDFW's Ecosystem Diagnosis and Treatment assessments under properly functioning habitat and historicalal estuary conditions..*
3. *Current viability is based on Technical Recovery Team viability rating approach.*
4. *Current and planning range productivity values are expressed in terms of intrinsic rate of population increase. Estimates are available only where data exists to EDT and population trend assessments.*
5. *Productivity improvement increments indicate needed improvements to reach contributing, high, very high, and maximum levels of population viability or potential.*

Table 5-24. Biological significance categories of lower Columbia steelhead populations based on current viability, core potential, and genetic legacy considerations.

	Raw ratings				Normalized values				
Population	Gen.[1]	Core[2]	Poten.[3]	Viab.[4]	Viab.[5]	Poten.[6]	Gen.[7]	Index[8]	Rank[9]
Coast Winter									
Grays/Chinook			2,300	1.8	0.59	0.64	0.00	0.41	B
Mill/Ab/Germ			1,500	1.7	0.56	0.42	0.00	0.33	B
Eloch/Skam			1,000	1.5	0.51	0.28	0.00	0.26	C
Cascade Winter									
Upper Cowlitz	1	1	1,600	1.0	0.33	0.44	1.00	0.59	A
Cispus	1	1	1,200	1.0	0.33	0.33	1.00	0.55	A
N.F. Toutle		1	3,500	1.8	0.61	0.97	0.00	0.53	A
NF Lewis		1	3,400	1.3	0.44	0.94	0.00	0.46	B
S.F. Toutle			1,900	2.1	0.70	0.53	0.00	0.41	B
Kalama			700	2.3	0.78	0.19	0.00	0.32	B
EF Lewis			1,300	1.7	0.57	0.36	0.00	0.31	B
Coweeman			1,200	1.8	0.59	0.33	0.00	0.31	B
Lower Cowlitz			1,500	1.3	0.44	0.42	0.00	0.29	C
Washougal			1,000	1.6	0.54	0.28	0.00	0.27	C
Salmon			1,200	1.4	0.45	0.33	0.00	0.26	C
Tilton			1,300	0.8	0.26	0.36	0.00	0.21	C
Clackamas (OR)		1	2,000	1.6	0.53	0.56	0.00	0.36	--
Sandy (OR)		1	3,600	1.7	0.55	1.00	0.00	0.52	--
Gorge Winter									
L Gorge (HHD)			300	1.7	0.56	0.08	0.00	0.21	C
U Gorge *(Wind)*			100	1.5	0.50	0.03	0.00	0.17	C
Hood (OR)	1	1	2,800	1.8	0.58	0.78	1.00	0.79	--
Cascade Summer									
Washougal	1	1	900	1.9	0.64	0.47	1.00	0.70	A
E.F. Lewis	1		400	1.8	0.59	0.21	1.00	0.60	A
Kalama		1	1,000	1.9	0.64	0.53	0.00	0.39	B
N.F. Lewis			1,200	0.5	0.17	0.63	0.00	0.27	C
Gorge Summer									
Wind		1	1,900	2.3	0.78	1.00	0.00	0.59	A
Hood (OR)			1,200	1.4	0.47	0.63	0.00	0.37	--

[1] *Genetic Legacy designation by the Technical Recovery Team. Genetic legacy populations are relatively unchanged by hatchery influences or represent unique life histories.*
[2] *Core population designation by Technical Recovery Team. Core populations were the largest historical populations and were key to metapopulation processes*
[3] *Potential fish numbers based on top end of planning range (typical value if accessible habitat restored to favorable albeit not pristine conditions based on EDT results for properly functioning conditions plus restored estuary.*
[4] *Provisional ratings by LCFRB consultants and WDFW staff based on TRT standards*
[5] *Normalized population persistence score used in biological significance ranking.*
[6] *Normalized core population potential used in biological significance ranking.*
[7] *Genetic legacy score used in biological significance ranking.*
[8] *Average of now, potential and genetic scores.*
[9] *Strata ranking based on average population score.*

5.5.4 Current Limiting Factors

5.5.4.1 <u>Net Effects of Manageable Factors</u>

The net effects of quantifiable human impacts and potentially manageable predation on steelhead translates into an 40-100% reduction in productivity among Washington lower Columbia populations (Figure 5-22). Thus, current fish numbers are only 0-60% of what they would be if all manageable impacts were removed.

No single factor consistently accounts for the majority of the reduction in fish numbers. Loss of tributary habitat quantity and quality is in many cases the most significant impact. Dam construction constitutes the largest single impact for upper Cowlitz and Lewis populations. Dam construction is also a factor for Gorge steelhead populations. Fishing is a minor impact, especially for winter steelhead. Hatchery effects vary among populations but are generally less than 20% of the total impact. Predation is among the lesser impacts we considered. Winter and summer steelhead impact factors and indices are shown in Table 5-25 and Table 5-26, respectively.

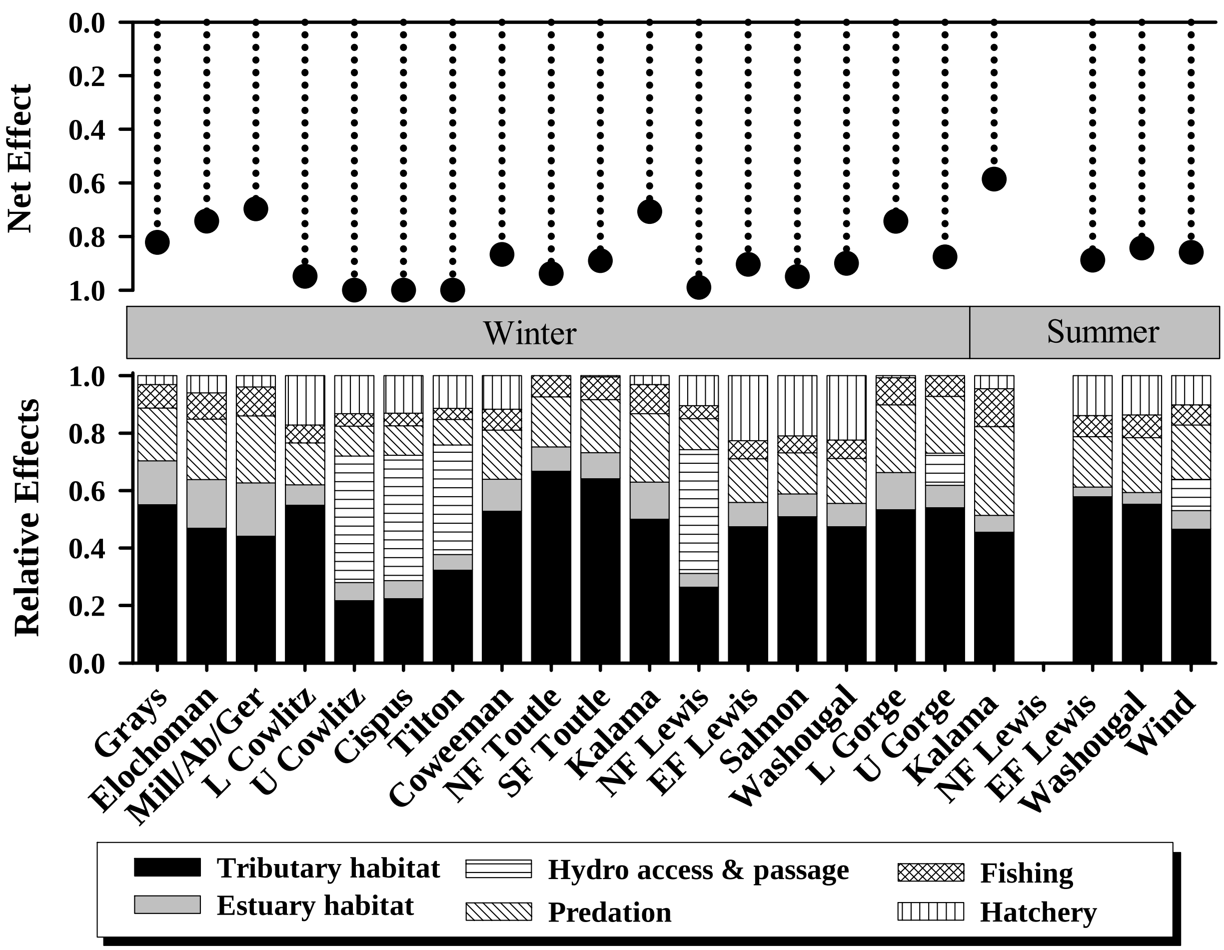

Figure 5-22. Net effect and relative contribution of potentially manageable impact factors on steelhead in Washington lower Columbia River subbasins.

Table 5-25. Winter steelhead impact factors and index.

	Grays	Elocho-man	Mill/Ab/Ger	L Cowlitz	U Cowlitz	Cispus	Tilton	Cowee-man	NF Toutle	SF Toutle	Kalama	NF Lewis	EF Lewis	Salmon	Washougal	L Gorge	U Gorge
Inputs																	
Neq Current	1,201	541	897	198	855	624	219	653	659	670	445	2,320	631	64	500	244	70
Neq PFC	1,885	842	1,191	1,352	1,402	1,001	1,093	1,017	3,089	1,673	614	3,038	1,109	223	909	270	123
Neq PFC+	2,307	1,031	1,458	1,517	1,625	1,159	1,266	1,197	3,480	1,884	703	3,391	1,278	257	1,037	312	138
Neq Historical	4,549	1,365	1,966	1,938	1,973	1,504	1,741	2,850	7,444	4,192	1,014	6,254	2,901	560	2,223	642	313
Hydro habitat loss	0.000	0.000	0.000	0.000	1.000	1.000	1.000	0.000	0.000	0.000	0.000	0.952	0.000	0.000	0.000	0.000	0.010
Dam pass mort. (juv)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.100
Dam pass mort. (ad.)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.050
Pred. mortality (juv.)	0.200	0.206	0.209	0.211	0.211	0.211	0.211	0.211	0.211	0.211	0.212	0.215	0.215	0.220	0.220	0.223	0.251
Pred. mortality (ad.)	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030	0.030
Fishing	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100
Hatchery fraction	0.05	0.09	0.06	0.92	1.00	1.00	1.00	0.23	0.00	0.02	0.31	0.77	0.51	0.51	0.50	0.01	0.00
Hatchery category	4	4	4	2	2	2	2	4	0	2	1	2	4	4	4	4	4
Hatchery fitness	0.3	0.3	0.3	0.7	0.7	0.7	0.7	0.3	0.0	0.7	0.9	0.7	0.3	0.3	0.3	0.3	0.3
Impacts (p reduction)																	
Tributary habitat	0.677	0.515	0.441	0.885	0.498	0.520	0.854	0.730	0.900	0.820	0.497	0.586	0.749	0.869	0.743	0.561	0.750
Estuary habitat	0.183	0.183	0.183	0.109	0.137	0.136	0.137	0.150	0.112	0.112	0.127	0.104	0.132	0.132	0.124	0.134	0.106
Hydro habitat loss	0.000	0.000	0.000	0.000	1.000	1.000	1.000	0.000	0.000	0.000	0.000	0.952	0.000	0.000	0.000	0.000	0.010
Dam passage	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.145
Predation	0.224	0.230	0.233	0.235	0.235	0.235	0.235	0.235	0.235	0.235	0.236	0.239	0.239	0.243	0.243	0.246	0.273
Fishing	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100	0.100
Hatchery	0.038	0.065	0.040	0.276	0.300	0.300	0.300	0.161	0.000	0.006	0.031	0.231	0.357	0.357	0.350	0.007	0.000
Total (unconditional)	1.222	1.093	0.997	1.605	2.270	2.291	2.626	1.376	1.347	1.273	0.991	2.212	1.577	1.702	1.561	1.048	1.385
Impact index																	
Tributary habitat	0.554	0.471	0.443	0.552	0.219	0.227	0.325	0.531	0.668	0.644	0.502	0.265	0.475	0.511	0.476	0.535	0.542
Estuary habitat	0.150	0.167	0.184	0.068	0.060	0.060	0.052	0.109	0.083	0.088	0.128	0.047	0.084	0.078	0.080	0.127	0.076
Hydro access/passage	0.000	0.000	0.000	0.000	0.441	0.437	0.381	0.000	0.000	0.000	0.000	0.431	0.000	0.000	0.000	0.000	0.112
Predation	0.183	0.210	0.233	0.146	0.103	0.102	0.089	0.171	0.174	0.184	0.238	0.108	0.151	0.143	0.156	0.235	0.197
Fishing	0.082	0.091	0.100	0.062	0.044	0.044	0.038	0.073	0.074	0.079	0.101	0.045	0.063	0.059	0.064	0.095	0.072
Hatchery	0.031	0.060	0.040	0.172	0.132	0.131	0.114	0.117	0.000	0.005	0.031	0.104	0.226	0.210	0.224	0.007	0.000

Table 5-26. **Summer steelhead impact factors and index.**

	Kalama	NF Lewis	EF Lewis	Washougal	Wind
Inputs					
Neq Current	788		187	639	1,516
Neq PFC	953		338	876	1,763
Neq PFC+	996		354	921	1,936
Neq Historical	1,264		933	2,289	5,099
Hydro habitat loss	0.000	0.500	0.000	0.000	0.010
Dam passage mortality (juveniles)	0.000	0.000	0.000	0.000	0.100
Dam passage mortality (adults)	0.000	0.000	0.000	0.000	0.050
Predation mortality (juveniles)	0.212	0.215	0.215	0.220	0.251
Predation mortality (adults)	0.030	0.030	0.030	0.030	0.030
Fishing	0.100	0.100	0.100	0.100	0.100
Hatchery fraction	0.35	0.93	0.27	0.25	0.21
Hatchery category	1	4	4	4	4
Hatchery fitness	0.9	0.3	0.3	0.3	0.3
Human impacts (p reduction)					
Tributary habitat	0.348	--	0.790	0.707	0.673
Estuary habitat	0.043	--	0.043	0.049	0.090
Hydro habitat loss	0.000	0.500	0.000	0.000	0.010
Dam passage	0.000	0.000	0.000	0.000	0.145
Predation	0.236	0.239	0.239	0.243	0.273
Fishing	0.100	0.100	0.100	0.100	0.100
Hatchery	0.035	0.651	0.189	0.175	0.147
Total (unconditional)	0.762	--	1.361	1.274	1.438
Human impact index					
Tributary habitat	0.457	--	0.581	0.555	0.468
Estuary habitat	0.057	--	0.032	0.038	0.062
Hydro access/passage	0.000	--	0.000	0.000	0.108
Predation	0.309	--	0.175	0.191	0.190
Fishing	0.131	--	0.073	0.078	0.070
Hatchery	0.046	--	0.139	0.137	0.102

5.5.4.2 Fisheries

Current fishing impacts on steelhead are low and provide limited opportunity for increasing their numbers through additional fishery regulation. The primary fisheries targeting steelhead occur in the Columbia River mainstem and tributaries (Figure 5-23); these fisheries harvest primarily hatchery fish and wild fish mortality is incidental. Fishing rates on wild steelhead have been reduced from their historical peaks in the 1960s by over 90% following prohibition of commercial steelhead harvest in the mainstem (1975), hatchery-only retention regulations in the mainstem starting in 1986, and hatchery-only retention regulations in the tributaries during the late 1980s and early 1990s. Interception of steelhead in ocean salmon fisheries is rare.

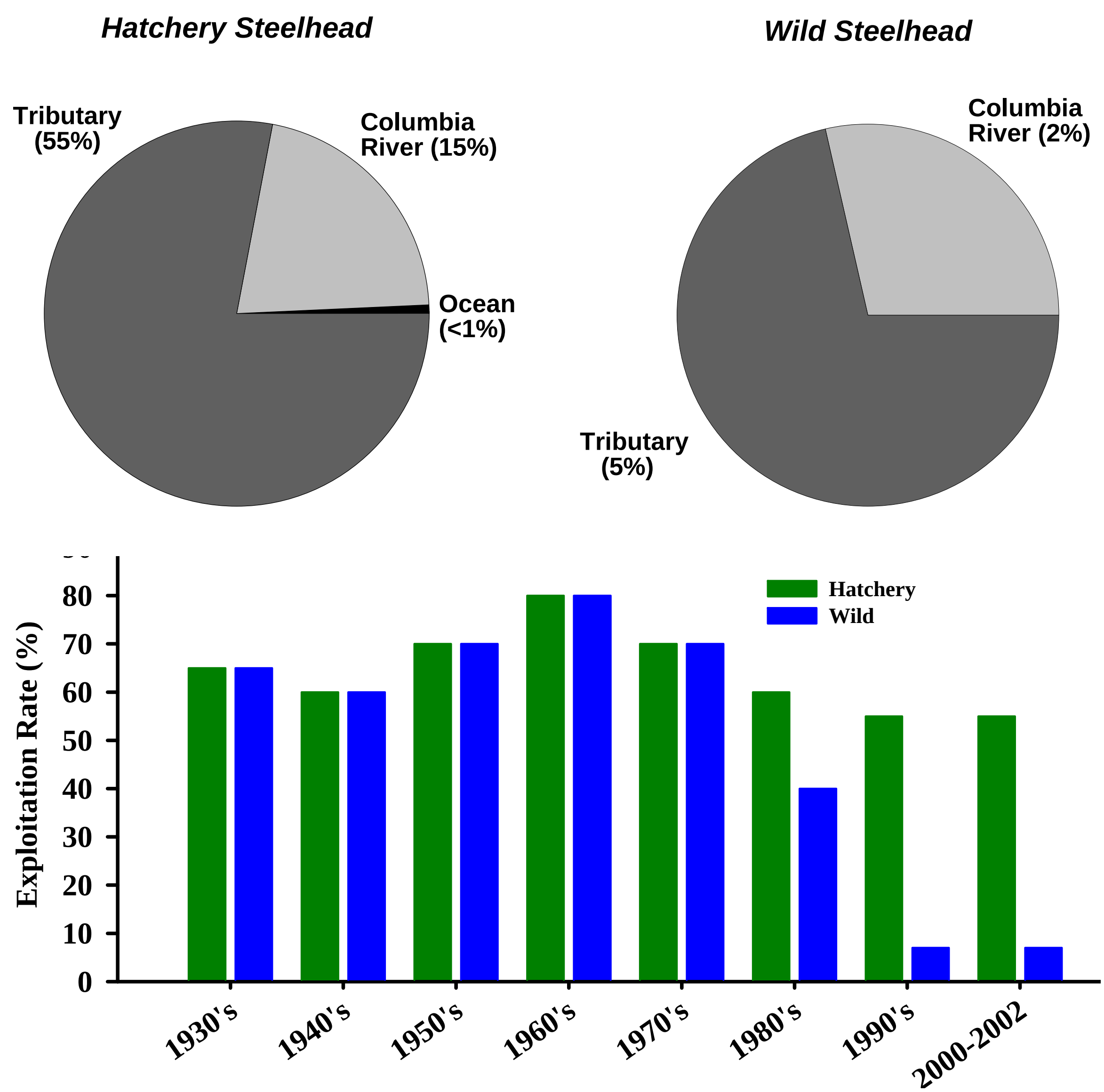

Figure 5-23. Approximate steelhead fishery exploitation rates over time and allocation of current exploitation rates among fisheries.

5.5.4.3 Hatcheries

With recent widespread changes in hatchery practices and substantial timing differences between many hatchery and wild stocks, hatchery influence is currently moderate to low for most Washington lower Columbia steelhead populations (Table 5-27). Most steelhead hatchery programs are intended to mitigate for the loss of natural steelhead production by providing fish for harvest opportunity. Many steelhead hatchery programs were developed from out-of-basin transfers or from multiple stocks; this practice continues today, primarily with Skamania summer and winter steelhead stocks released throughout the lower Columbia. In most cases, brood stock mixing was limited to a few stocks and performed only during the initial years of establishing the hatchery program. After the hatchery program had been established, brood stock collection came from returning adults, aside from minimal outside brood stock usage during years of hatchery shortfalls. Inter-specific hatchery predation impacts on steelhead are not an issue because wild rearing areas of small juvenile steelhead are primarily in areas upstream of hatchery release sites.

The indexed potential for negative impacts of hatchery spawners on wild population fitness of winter steelhead was estimated to range from 0 to 38%. Hatchery releases of winter steelhead currently range from 0 to 652,500 per subbasin. Hatchery fish continue to comprise 77-100% of the natural winter steelhead spawners in the lower Cowlitz and Lewis basins where large hatchery programs are operated to mitigate for lost access to upper basin spawning areas. Hatchery fractions on wild population spawning grounds during wild spawning periods are much lower in other subbasins, ranging from 0 to 23%. Reintroduction attempts in the upper Cowlitz basin rely entirely on hatchery stock that was originally derived from fish blocked at the dams. Current winter steelhead hatchery broodstock are derived from a variety of sources ranging from entirely natural fish (category 1) to highly domesticated stock (category 4). Hatchery fractions are generally low where broodstock of poor fitness are present whereas, more robust broodstock are present where hatchery fractions are high. In the Lewis and Cowlitz basins, the high incidence of hatchery spawners suggests that the fitness of natural and hatchery fish is now probably quite similar and natural populations could collapse without continued hatchery subsidy under current habitat conditions.

The indexed potential for negative impacts of hatchery spawners on wild summer steelhead population fitness was estimated to range from 4 to 65%. Hatchery releases of summer steelhead currently range from 0 to 225,000 per subbasin. Hatchery fish continue to comprise 93% of the natural summer steelhead spawners in the lower NF Lewis where a large hatchery program is operated to mitigate for lost access to upper basin spawning areas. Hatchery fractions on wild population spawning grounds during wild spawning periods are lower in other subbasins, ranging from 21 to 35%. Current winter steelhead hatchery broodstock are derived entirely from natural fish in the Kalama (category 1) but are highly domesticated elsewhere (category 4). In the NF Lewis, the high incidence of hatchery spawners suggests that the fitness of natural and hatchery fish is now probably quite similar and the natural population could collapse without continued hatchery subsidy under current habitat conditions.

Table 5-27. **Presumed reductions in wild population fitness as a result of natural hatchery spawners for Washington lower Columbia River steelhead populations.**

Population	Annual releases[a]	Hatchery fraction	Fitness category	Assumed Fitness	Potential impact	Interacting releases	Interspecies impact
Coast Winter							
Grays/Chinook	40,000	0.05	4	0.3	0.038	0	0
Eloch/Skam	90,000[b]	0.09	4	0.3	0.065	0	0
Mill/Aber/Germ	0[c]	0.06	4	0.3	0.040	0	0
Cascade Winter							
Lower Cowlitz	652,500d	0.92	2	0.7	0.276	0	0
Upper Cowlitz	287,500 [f]	1.00	2	0.7	0.300	0	0
Cispus	--[f]	1.00	2	0.7	0.300	0	0
Coweeman	20,000	0.23	4	0.3	0.161	0	0
S.F. Toutle	0[e]	0.02	2	0.7	0.006	0	0
N.F. Toutle	0[e]	0	2	0.7	0	0	0
Tilton	100,000 [g]	1.00	2	0.7	0.300	0	0
Kalama	90,000[h]	0	1	0.9	0	0	0
NF Lewis	100,000	0.77	2	0.7	0.231	0	0
EF Lewis	90,000	0	--	--	0	0	0
Salmon	20,000	na	na	na	na	0	0
Washougal	60,000	0	--	--	0	0	0
Gorge Winter							
L Gorge	0[i]	0.01	4	0.3	0.007	0	0
U Gorge	0[j]	0	--	--	0	0	0
Cascade Summer							
Kalama	90,000	0.35	1	0.9	0.035	0	0
N.F. Lewis	225,000	0.93	4	0.3	0.651	0	0
E.F. Lewis	25,000	0.27	4	0.3	0.189	0	0
Washougal	60,000	0.25	4	0.3	0.175	0	0
Gorge Summer							
Wind	0	0.21	4	0.3	0.147	0	0

[a] *Annual release goals.*

[b] *The Elochoman River winter steelhead hatchery program at the Beaver Creek Hatchery stopped releasing smolts in 1999; hatchery returns were expected to significantly diminish starting with the 2001 return. The Elokomin Salmon Hatchery started a 'wild' winter steelhead program in 2000 to replace the previous program with indigenous stock (30,000 smolts per year). An additional 60,000 hatchery fish are released per year for fisheries. An additional 30,000 summer steelhead are released each year.*

[c] *There are no steelhead hatchery programs in Mill, Abernathy, or Germany Creek. Sporadic small releases of winter steelhead have been made from the former Beaver Creek Hatchery program.*

[d] *Includes 300,000 hatchery stock and 352,500 late winter stock. An additional 500,000 summer steelhead are released per year.*

[e] *25,000 summer steelhead are also released in each of the North and South Toutle.*

[f] *Includes 37,500 yearlings and 250,000 subyearlings of late run stock intended to restore an upper Cowlitz basin population.*

[g] *Fingerling releases for reintroduction purposes.*

[h] *Includes 45,000 each of hatchery and late wild stocks. The winter steelhead program changed focus in 1998 1999; only wild steelhead are collected for brood stock.*

[i] *There are no hatchery steelhead programs in the lower gorge tributaries; winter steelhead from the Skamania and Beaver Creek Hatcheries were sporadically released in the basins since 1958.*

[j] *The Wind River winter and summer steelhead hatchery programs at the Carson NFH stopped releasing smolts in 1997; hatchery returns were expected to significantly diminish starting with the 1999 return.*

5.5.4.4 Stream Habitat

EDT analyses suggest that stream degradation has substantially reduced the habitat potential for steelhead in all Washington lower Columbia River subbasins where analyses have been completed (Figure 5-24). Declines in habitat quantity and quality for steelhead have reduced current productivity potential to 6-34% and equilibrium numbers to 10-60% of the historical template. Substantial stream habitat improvements would be necessary to reach optimum conditions (i.e. PFC) for steelhead in remaining subbasin. Restoration of optimum habitat quality would be expected to increase habitat capacity by 30 to 2,400 adult steelhead per subbasin.

Steelhead rely on the middle mainstem to upper stream reaches where a lack of habitat diversity, sedimentation, and flow consistently limit habitat suitability. More detailed descriptions of stream habitat conditions and effects on fish in each subbasin may be found in Volume II of the Technical Foundation.

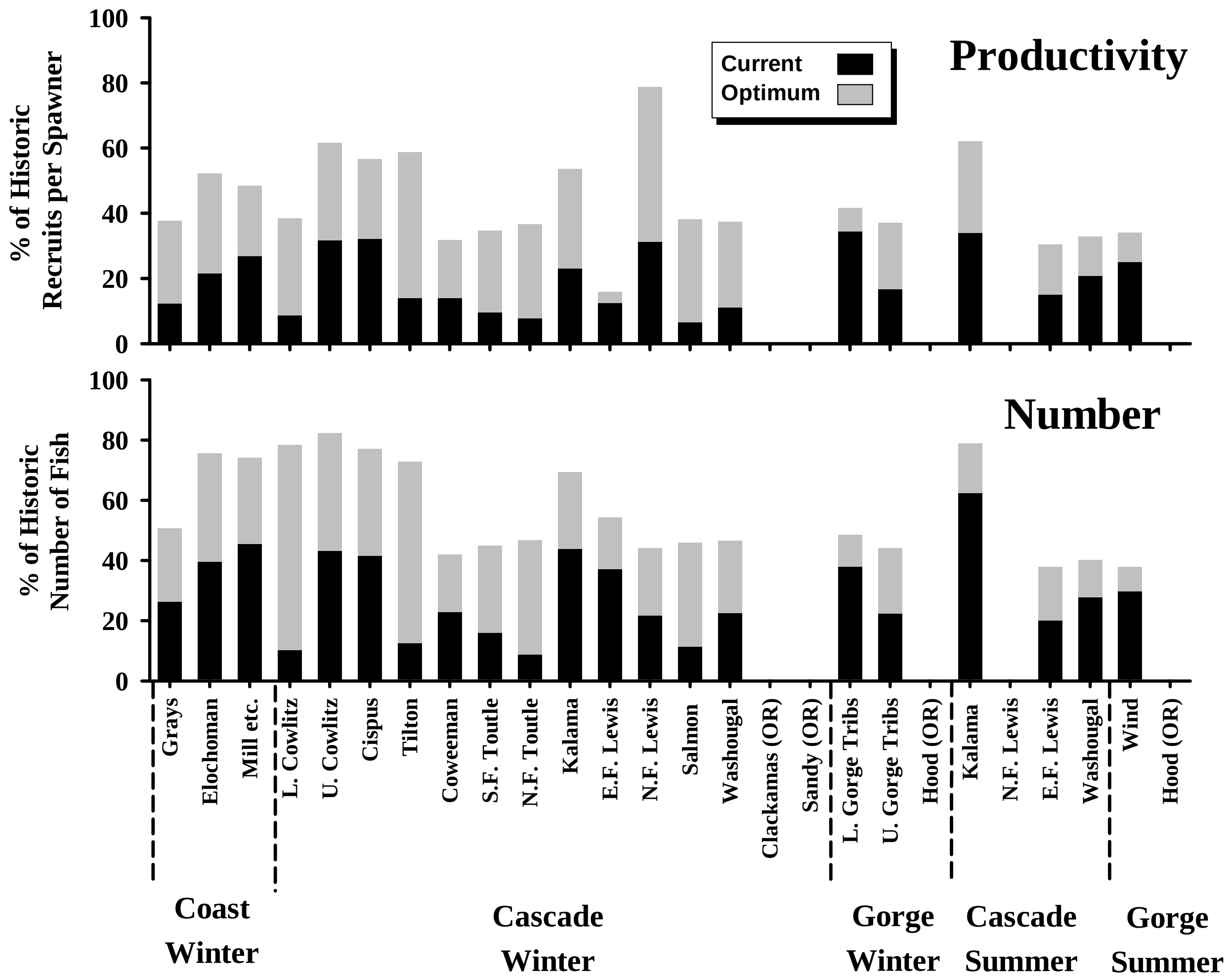

Figure 5-24. Current, optimal, and historical subbasin productivity and capacity inferred for steelhead from stream reach habitat conditions using EDT.

5.5.4.5 Dams

Dam impacts on Washington lower Columbia steelhead were estimated to range from 0 to 100% (Figure 5-25). Dams on the Cowlitz have inundated or blocked access to 100% of the winter steelhead habitat based on EDT assessments. In the North Fork Lewis, 95% of the winter steelhead habitat and approximately 50% of the summer steelhead habitat has been inundated or blocked. Passage mortality at Bonneville Dam was assumed to average 10% for juveniles and an additional 5% for adults based on a synthesis of the available literature. Steelhead generally spawn and rear in headwater and upper mainstem reaches of subbasins and are less subject to hydropower effects on downstream habitats than are chum and fall chinook.

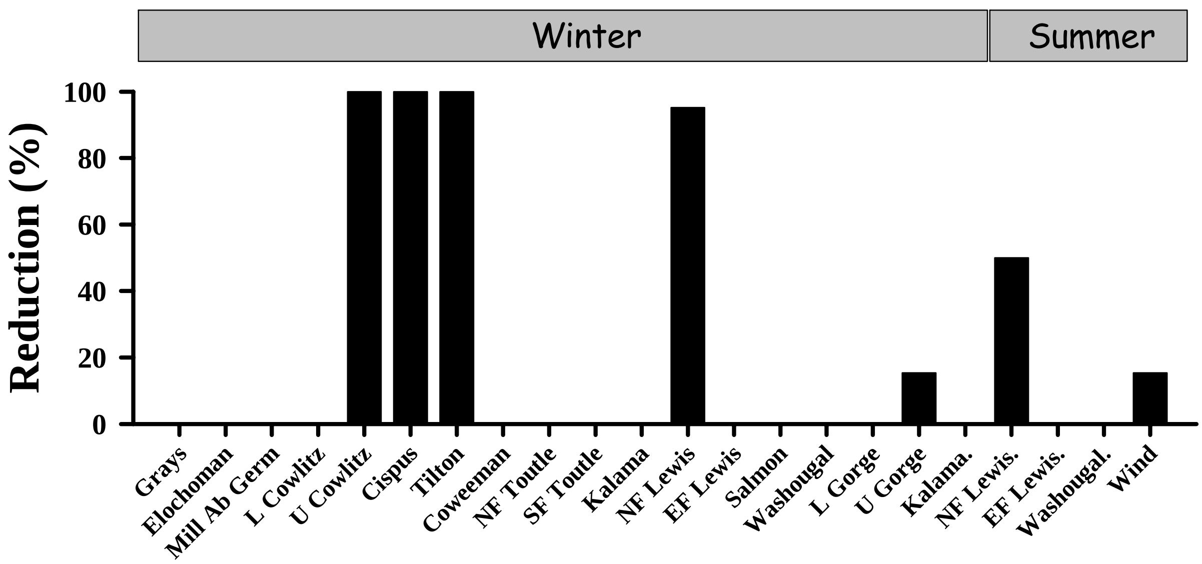

Figure 5-25. Assumed dam impacts on Washington lower Columbia steelhead populations.

5.5.4.6 Mainstem and Estuary Habitat

Mainstem and estuary habitat impacts were estimated to account for approximately a 10-20% reduction in productivity of winter and summer steelhead. Steelhead migrate through mainstem and estuary areas soon after emigration from tributary streams. Residence time in estuary and mainstem habitats is relatively brief, but smoltification and transition from fresh to salt water is a critical life stage.

5.5.4.7 Predation

Potentially manageable predation mortality was assumed to average 20% to 25% depending on travel distance from the subbasin to the ocean. Pikeminnow and tern management is projected to reduce salmonid predation by approximately 50%. Tern predation is almost entirely an artifact of recently established colonies on dredge spoil islands in the estuary but the current rate (9%) is less than half that observed prior to downstream translocation of the Rice Island colony (20%). Pikeminnow predation was greatest for populations originating in Bonneville Reservoir tributaries (5%), passing the pikeminnow gauntlet in Bonneville Dam forebay and tailrace, and traveling the entire 145 mile length from Bonneville to the Estuary. Predation rates by seals and sea lions on adult steelhead added an assumed 3% mortality.

5.5.5 Summary Assessment

1. Human activities including fishing, hatchery operation, alteration of stream, river, and estuary habitats, hydropower development and operation, and potentially manageable predation have collectively reduced productivity of winter and summer steelhead populations to 0-40% of historic levels. Recovery efforts will require significant improvements in multiple areas because no single factor accounts for the majority of the reduction in fish numbers.

2. Current fishing impacts on steelhead are relatively low and provide limited opportunities for increasing numbers through additional regulation of fisheries. Fishing impacts occur almost exclusively in Columbia basin sport fisheries. Selective fishery regulations were implemented for steelhead prior to listing.

3. Reduced productivity of wild populations as a result of interbreeding with potentially less-fit hatchery fish is among the most significant of hatchery concerns for wild stock recovery although these negative effects are at least partially offset by the demographic benefits of additional spawners. Potential negative impacts increase with the proportion of hatchery spawners and the disparity between wild and hatchery fish. Potential fitness impacts among Washington lower Columbia steelhead populations range from 0 to 65%. Potential impacts are greatest in the Cowlitz and Lewis basins where dams block most of the available steelhead habitat. Inter-specific hatchery predation impacts on steelhead are not an issue because wild rearing areas of small juvenile steelhead are primarily in areas upstream of hatchery release sites.

4. Stream habitat conditions s significantly limit steelhead in all Washington lower Columbia River subbasins where EDT analyses have been completed. Substantial stream habitat improvements would be necessary to reach optimum conditions (i.e. PFC) in most subbasins. The significance of stream habitat suggests that recovery may not be feasible without substantial improvements in habitat quantity and quality.

5. Estuary and mainstem habitats are important to steelhead life history with assumed habitat impacts of 10-20%.

6. Hydropower development in the Cowlitz and Lewis have blocked 50-95% of the summer and winter steelhead habitat. Mainstem dam passage affects upper Gorge populations although passage success for steelhead tends to be greater than among other salmon species.

5.6 Bull Trout

5.6.1 Current Viability

Bull trout were Federally listed as threatened in 1999. The USFWS has formulated a draft recovery plan, and identified 27 recovery units for bull trout. One of these is the Lower Columbia recovery unit, which has two core areas (the Lewis River and the Klickitat River). While no local populations have been identified within the White Salmon, the subbasin contains core habitat, and could support bull trout (USFWS 2002). Recent natural escapements in two upper Lewis River spawning areas currently average several hundred fish per year. The size of the Gorge population is unknown.

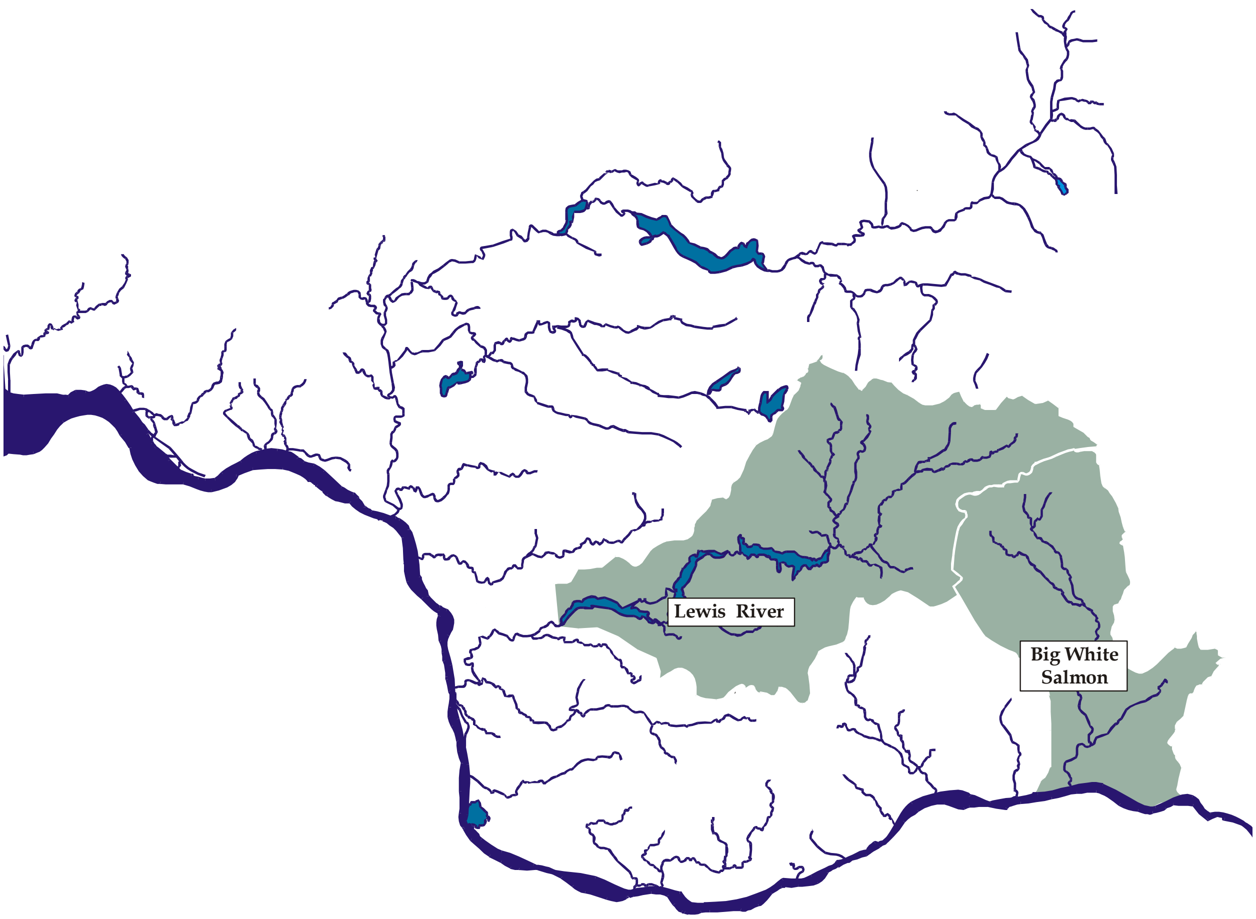

Figure 5-26. Distribution of historic bull trout populations among lower Columbia River subbasins.

5.6.2 Recovery Planning Ranges

At present, recovery standards for bull trout have only been partially identified. However, USFWS has compiled a list of research criteria to gather the data necessary to assess whether management actions are resulting in the recovery of bull trout in the Lower Columbia recovery unit. USFWS (2002) identified the following recovery standards and research needs:

1. Distribution of bull trout in the Lower Columbia recovery unit is unknown and considered a research need. Until additional information is obtained, at a minimum, the existing local populations in the recovery unit need to be maintained.
    a. USFWS (2002) states that "establishment of additional local populations . . . is essential for recovery." Potential sites which have or could support bull trout if restored should be evaluated for possible reintroduction.
    b. Factors that may limit potential for reintroduction should be identified
2. Estimated abundance of bull trout in the Lower Columbia recovery unit local populations is considered a research need.
    a. A complete set of data is not available from which to make a reliable estimate of bull trout abundances in any of the local populations.
    b. As more data is collected, population estimates will be conducted to more accurately reflect both migratory and resident life history forms.
3. Adult bull trout exhibit a stable or increasing trend for at least two generations at or above the identified abundance level (from criteria 2) within core areas.
    a. The development of a standardized monitoring and evaluation program to accurately describe trends in bull trout abundance has been identified as a priority research need.
4. Barriers to bull trout migration in the Lower Columbia recovery unit need to be addressed.
    a. Barriers that have been identified as primary impediments to recovery, and where connectivity must be reestablished are Swift 1 and 2 and Yale Dams on the Lewis River, and Condit Dam on the White Salmon River.

5.6.3 Summary Assessment

1. The historic distribution and abundance of bull trout in the lower Columbia region are unknown. Bull trout are known to exist in the Lewis drainage and some Gorge tributaries.
2. Hydropower development has negatively affected bull trout populations in the Lewis River system, where three hydroelectric dams block fish passage and eliminate connectivity of subpopulations.
3. The USFWS has recommended installing a means of fish passage at Condit Dam on the White Salmon River, although no bull trout are known to occupy that system now. Suitable habitat exists, and bull trout are believed to have existed in the White Salmon historically.
4. Fishing for bull trout is closed in Washington. Bycatch has been reported in the Lewis River watershed kokanee fishery but its impacts are believed to be very low.
5. There are no hatchery programs to produce bull trout. Interactions between bull trout and hatchery-produced salmonids have not been studied, and impacts are unknown.

5.7 Cutthroat

5.7.1 Current Viability

Lower Columbia River coastal cutthroat trout are not listed under the Federal ESA. The subspecies was a candidate for listing as "threatened," but the USFWS found on July 2002 that a listing was not warranted (50 CFR 17). Coastal cutthroat trout are widely distributed throughout suitable habitats of lower Columbia River subbasins (Figure 5-27) and historical distribution has not contracted appreciably (USFWS 2002). Cutthroat occur at over 1,300 documented locations within the lower Columbia distinct population segment.

The USFWS also found that, though there were few data to work with, populations in the Washington part of the distinct population segment under review "remained at levels comparable to healthy-sized populations, indicating that large-scale, long-term declines have not occurred at the landscape level." (USFWS 2002). Available density data for tributaries below Bonneville Dam were comparable to those from Olympic Peninsula and Puget Sound populations that were not considered to be in danger of extinction (50 CFR 17). While numbers of sea-run cutthroat appeared to have declined, the USFWS found that resident and anadromous forms were not segregated, and that because resident forms could give rise to anadromous progeny, the presence of healthy subpopulations of resident trout mitigated risks to anadromous forms to some degree.

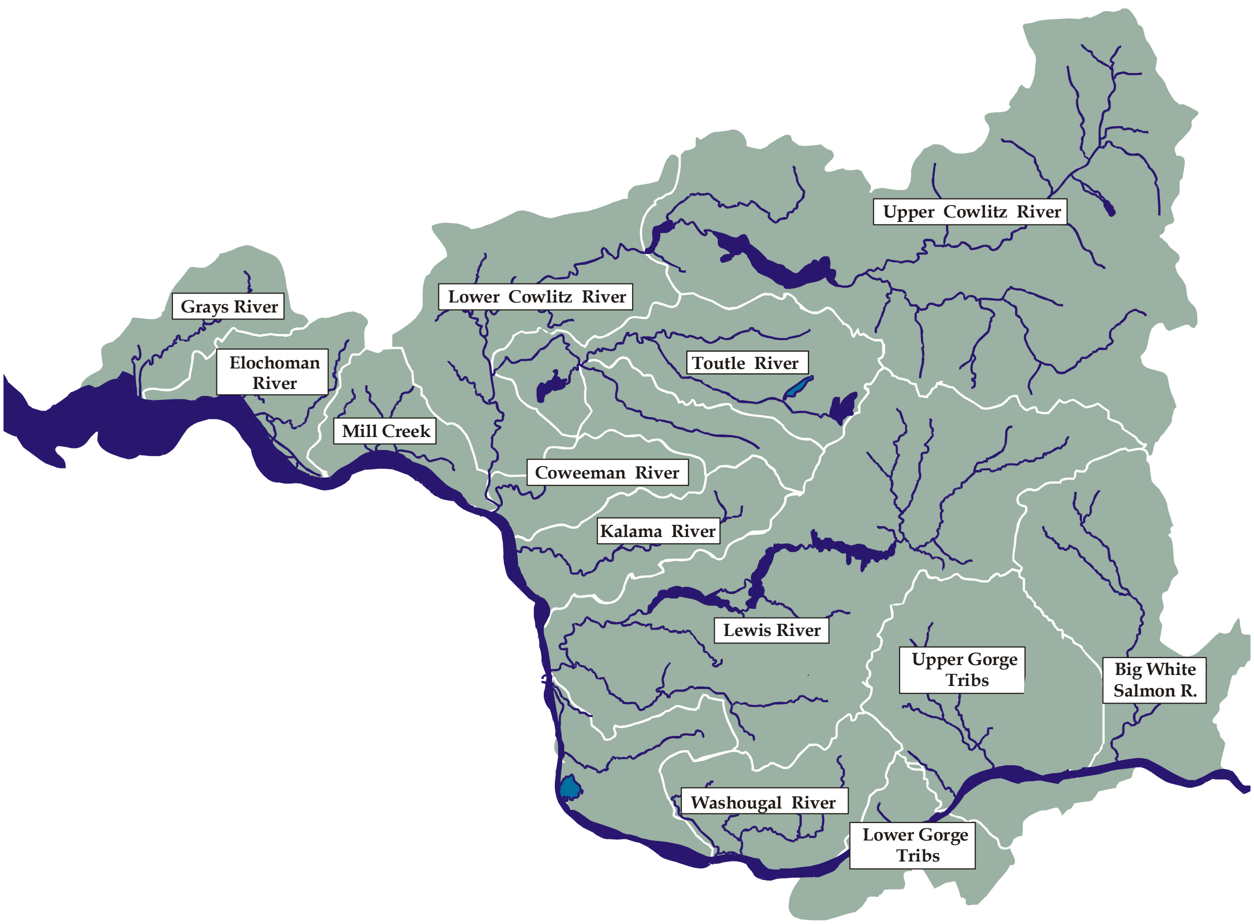

Figure 5-27. Distribution of historical cutthroat trout populations among lower Columbia River subbasins

5.7.2 Summary Assessment

1. Cutthroat trout are widely distributed in Washington lower Columbia River tributary systems and are not federally listed. Numbers of sea-run cutthroat appear to have declined but risks are ameliorated by the presence of healthy subpopulations of resident trout.
2. Current fishing impact is low and additional restrictions are not warranted given the current status of the species.
3. Some hatchery production of sea-run cutthroat occurs. Relative risks and benefits have not been quantified.
4. Cutthroat are a generalist species that exist in many small streams not suitable for other salmonids. Cutthroat are thus susceptible to habitat changes that do not directly affect anadromous species in Washington lower Columbia tributaries but suitable habitat conditions continue to be widely available.

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References

6.0 References

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