

**Guidance for Updating Recovery Goals for the
Hood Canal and Strait of Juan de Fuca
Summer Chum Salmon Populations**

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Executive Summary

This report reviews the status of the existing recovery goals of the Hood Canal Summer Chum evolutionarily significant unit (ESU) and provides new analyses for updating the goals and assessing the gaps between current population performance and those goals. Seven recommendations are offered with respect to updating the goals, prioritizing future habitat restoration and protection actions, addressing harvest goals, continuing reintroduction efforts, and maintaining monitoring and evaluation efforts.

The Hood Canal Summer Chum ESU is comprised of two independent populations: the Hood Canal population and the Strait of Juan de Fuca population. The ESU was listed as threatened under the U.S. Endangered Species Act (ESA) in 1999. In 2000, tribal and state co-managers distributed the Summer Chum Salmon Conservation Initiative (WDFW and PNPTT 2000), a comprehensive plan for the implementation of summer chum salmon recovery that integrated habitat, harvest, and hatchery actions. The formal recovery plan for the ESU was prepared by the Hood Canal Coordinating Council (HCCC) in 2005 (HCCC 2005), which was followed by a federal supplement to that plan (NMFS 2007) and a viability analysis (Sands et al. 2009). These documents (including the initiative of 2000) serve as the foundational documents for recovering the ESU.

The Puget Sound Technical Recovery Team (PSTRT) published its set of recovery criteria for the ESU in 2009, providing details for abundance and productivity, spatial structure, and diversity (Sands et al. 2009), consistent with guidance given in McElhany et al. (2000). The quantitative analysis of abundance and productivity was based on observed run sizes of the two populations for brood years 1974 to 2001 (hence effectively covering returning run sizes through about 2005).

This report considers new information of relevance to the viability recovery goals, namely the inclusion of five more brood years in the analysis along with additional information about climate/ocean regime shifts that has bearing on evaluating recovery. Also, consideration is given to how to address expected effects of climate change in the recovery goals. Trends in climate change are projected to likely affect summer chum performance and NMFS encourages recovery planners to account for this in their planning (Ford et al. 2011).

A second part of the report assesses the performance of the major spawning aggregations that comprise the ESU's two populations and sets forth recovery targets for each consistent with the overall recovery goals for the populations. Recovery targets for the extant subpopulations are identified to help guide habitat restoration and protection efforts. The targets take into account climate/ocean regime shifts as well as some amount of future climate change effects.

The assessment of subpopulation performance was made for several baseline time periods, providing a means to consider changes in performance resulting from on-going and projected habitat losses associated with watershed development (e.g., increases in impervious surfaces), as well as from habitat restoration. The current baseline (called the 2014 baseline), which takes into account expected maturation of habitat restoration and protection actions implemented since 2000, as well as current projections of watershed development, is used to identify the gaps between expected future performance associated with current baseline conditions and viability thresholds. Based on these gaps, and on the geographic pattern of subpopulation performance seen over the ESU, recommendations for prioritizing future recovery actions are provided.

Seven recommendations are offered with respect to updating recovery goals, prioritizing future habitat restoration and protection actions, addressing harvest goals, continuing reintroduction efforts, and continuing monitoring and evaluation for the Hood Canal Summer Chum ESU. The recommendations are presented in a way for the reader to follow a progressive logic, such that after recommendation one is given, subsequent recommendations then build on preceding recommendations. A summary of the recommendations follows:

1. We recommend that the quantitative recovery goals for productivity and abundance for the Strait of Juan de Fuca and Hood Canal populations presented in the NMFS supplement to the summer chum recovery plan (NMFS 2007) be updated with results of the viability analysis presented in this paper. It is recognized that recovery goals and population viability criteria are to be an adaptively managed part of the recovery plan and that as new data and modeling results become available, the recovery goals and population viability criteria would be refined over time (WDFW and PNPTT 2000, PNPTT and WDFW 2003, HCCC 2005, NMFS 2007).
2. We recommend that the recovery thresholds for abundance (or capacity and productivity) viability take into account the impacts of climate and ocean regimes on the performance and survival of summer chum in the Hood Canal ESU. The recovery thresholds should be based on performance during the warm phase regime of the Pacific Decadal Oscillation (PDO) when summer chum performance is adversely impacted and risk of extinction is increased. The PDO is currently in the cool phase and it may remain in this state for several more years.
3. We recommend that the viability recovery goals should take into account the impacts of climate change on the performance and survival of summer chum in the Hood Canal ESU. Recovery goals should account for at least a 5 percent increase in the variation of summer chum population performance. We assume here that summer chum population performance variation is approximately equal to the percent increase in environmental variation resulting from climate change. How population performance is correlated with environmental variation is uncertain and will need some level of assessment and monitoring. As time proceeds and estimates of variation improve, this recommended value should be reviewed and updated.
4. We recommend that habitat protection and restoration actions be done strategically and distributed among the areas affecting the Strait of Juan de Fuca and Hood Canal populations and their subpopulations. This strategy should balance the need to reduce the performance gaps for subpopulations projected to be below viability thresholds, while also addressing the continuing need to strengthen performance in the core subpopulations for the sake of bolstering overall population abundance.
5. We recommend that the co-managers consider strategic ways of addressing summer chum harvest goals by applying information contained in this report. Given the current performance of the co-managers' Base Conservation Regime (BCR), we recommend that it be retained as the primary harvest management tool toward recovery. It is particularly well suited to address fishery risk when the summer chum populations and subpopulations are at low levels, as they had been, in the vicinity of their critical abundance thresholds identified in the co-managers' SCSCI and HCCC recovery plan. In addition, the BCR appears to provide adequate protection during the cool and warm phases of the PDO and to the weaker subpopulations. On the other hand, the co-managers should continue their development of the basic provisions and criteria

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6. We conclude that successful reintroductions into watersheds where spawning aggregations have been extirpated are likely to be critically important in light of expected climate change effects combined with expected downturns in production that will occur during warm phases of the PDO. Long-term viability of the summer chum populations will likely depend on recovering at least some of the lost spatial structure and diversity that existed prior to the recent extirpations.
7. We recommend that the regular and comprehensive sampling and monitoring of the SJDF and Hood Canal summer chum populations that is being done by the State and Tribal co-managers be continued. The co-managers annually collect and evaluate information on spawner escapement, harvest, run size, age composition, natural-origin vs. supplementation-origin composition, and genetics for summer chum populations and subpopulations throughout the summer chum ESU (WDFW and PNPTT 2007, PNPTT and WDFW 2014). The analysis in the recovery plan (HCCC 2005, NMFS 2007) and in this paper was only possible because this comprehensive monitoring and evaluation was done.

The information on summer chum population and subpopulation performance presented in this document is based on observed variability in the returns of salmon related to the escapement. This estimate of variability will likely change over time as environmental conditions change and recovery actions are taken. As such, viability goals should be re-evaluated on a regular basis with new analyses and methods incorporated as they become available.

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Guidance for Updating Recovery Goals for the Hood Canal and Strait of Juan de Fuca Summer Chum Salmon Populations

1.0 Introduction

In 1999, summer chum salmon (*Oncorhynchus keta*) comprising the Hood Canal Summer Chum evolutionarily significant unit (ESU) were listed as threatened with extinction under the U.S. Endangered Species Act (ESA) (NMFS 1999). The ESU is composed of two independent populations, called the Hood Canal and the Strait of Juan de Fuca populations, which are produced in streams and rivers that enter the marine waters bearing those names (Sands et al. 2009) (Figure 1).¹ In 2000, tribal and state co-managers distributed the Summer Chum Salmon Conservation Initiative (WDFW and PNPTT 2000), a comprehensive plan for the implementation of summer chum salmon recovery that integrated habitat, harvest, and hatchery actions. The formal recovery plan for the ESU was prepared by the Hood Canal Coordinating Council (HCCC) in 2005 (HCCC 2005), which was followed by a federal supplement to that plan (NMFS 2007) and a viability analysis (Sands et al. 2009). These documents (including the initiative of 2000) serve as the foundational documents for recovering the ESU.

In this paper, we review the status of the existing recovery goals, as presented in the aforementioned reports, for the two populations within the ESU and consider new information for updating the goals. A salmon recovery goal typically includes two aspects of recovery: ESA recovery, which deals with the statutory requirements under the federal ESA for meeting viability criteria for populations and the ESU as a whole, and a broader view of recovery (or broad-sense recovery) that reflects various societal goals and needs for ecosystem services, such as harvest (McElhany et al. 2000; NMFS 2000). A viable ESU is defined as one that is naturally self-sustaining with a negligible risk of extinction (<5%) over a 100-year time period. The viability criteria define four characteristics of the ESU's performance needed to achieve viability: abundance and productivity among the populations that comprise the ESU, diversity within and among the populations, and spatial structure over the geographic area of the ESU (McElhany et al. 2000).

The viability analysis prepared by Sands et al. (2009) presented viability criteria that defined population performance thresholds for achieving delisting within the context of providing for various levels of harvest. From the time of that analysis and continuing to the present time, explicit broad-sense goals have not been defined by state and tribal co-managers, though it is understood that full recovery must provide for harvest, as stated in NMFS (2007).²

¹ / Both the ESU as a whole and one of its two constituent populations are named "Hood Canal", which can lead to confusion unless a distinction is clearly made in the text. For clarity, all references in this paper meant to apply to the ESU as a whole are referred to explicitly as the Hood Canal Summer Chum ESU, or simply as the Hood Canal ESU (wherever naming is appropriate). The population by the same name is referred to as the Hood Canal summer chum population, or simply as Hood Canal summer chum.

² / Harvest of Hood Canal Summer Chum ESU fish was dramatically cut in the early 1990s, as it remains to the present time. As part of the Summer Chum Salmon Conservation Initiative (WDFW and PNPTT 2000), tribal and state co-managers formally established a harvest Base Conservation Regime (BCR), but have not yet determined a long-term harvest goal for the ESU while ensuring that viability standards are being met.

Here, we consider new information for updating the viability criteria and present the analysis, as Sands et al. (2009) did, within a context for meeting different harvest levels. Of particular relevance in this paper is how shifts in climate/ocean regimes and their effects on summer chum production should be considered in setting recovery goals. In addition, trends in climate change are projected to likely affect future summer chum performance, and the National Oceanographic and Atmospheric Administration's (NOAA) National Marine Fisheries Service (NMFS) encourages recovery planners to account for this in their planning (Ford ed. 2011). This paper provides guidance for updating the recovery goals to address these issues.

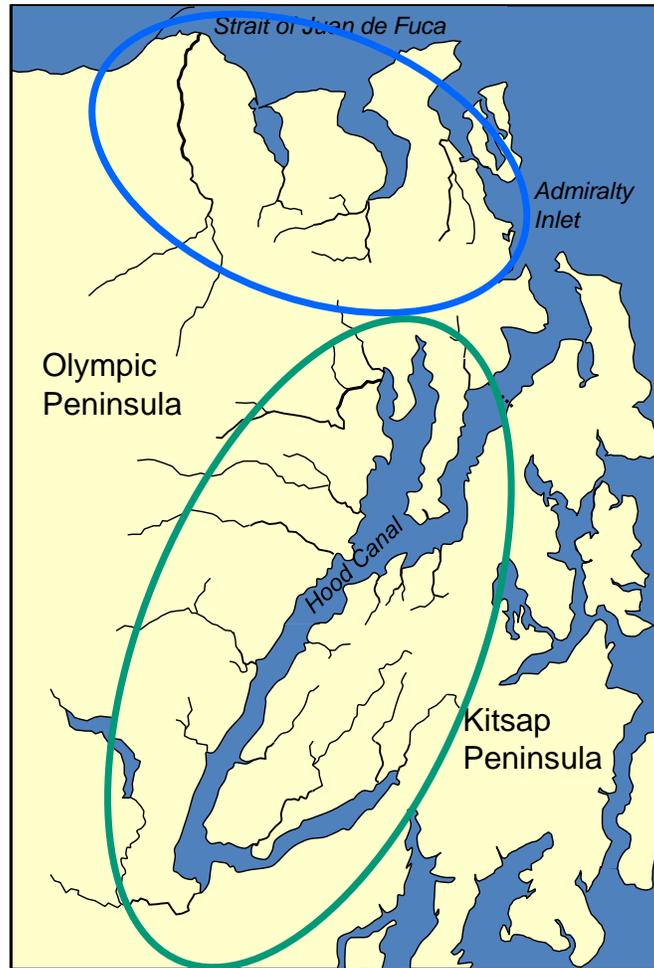


Figure 1. The two geographic regions where the natal streams of the two independent populations that comprise the Hood Canal Summer Chum ESU are located.

A second, related part of this paper assesses the performance of the major spawning aggregations that comprise the ESU's two populations and sets forth recovery targets for each consistent with the overall recovery goals for the populations. The major spawning aggregations, identified to be subpopulations by Sands et al. (2009), occur in independent watersheds that drain to the waters of either the eastern Strait of Juan de Fuca or Hood Canal (Figure 2). Eight or nine of the subpopulations are classified as extant and at least that many are considered to be extirpated. Recovery targets for the extant subpopulations are identified herein to help guide habitat restoration and protection efforts—we see them as waypoints in

charting a course to population recovery and measuring progress to that end. The new targets would take into account climate/ocean regime shifts as well as some amount of future climate change effects.

The assessment of subpopulation performance was made for several baseline time periods, enabling us to consider changes in performance resulting from on-going and projected habitat losses associated with watershed development (e.g., increases in impervious surfaces), as well as from habitat restoration. The current baseline (called the 2014 baseline), which takes into account expected maturation of habitat protection and restoration actions implemented since 2000, as well as current projections of watershed development, is used to identify the gaps between expected future performance associated with current baseline conditions and viability thresholds. Based on these gaps, and on the geographic pattern of subpopulation performance seen over the ESU, we provide recommendations for prioritizing future recovery actions.

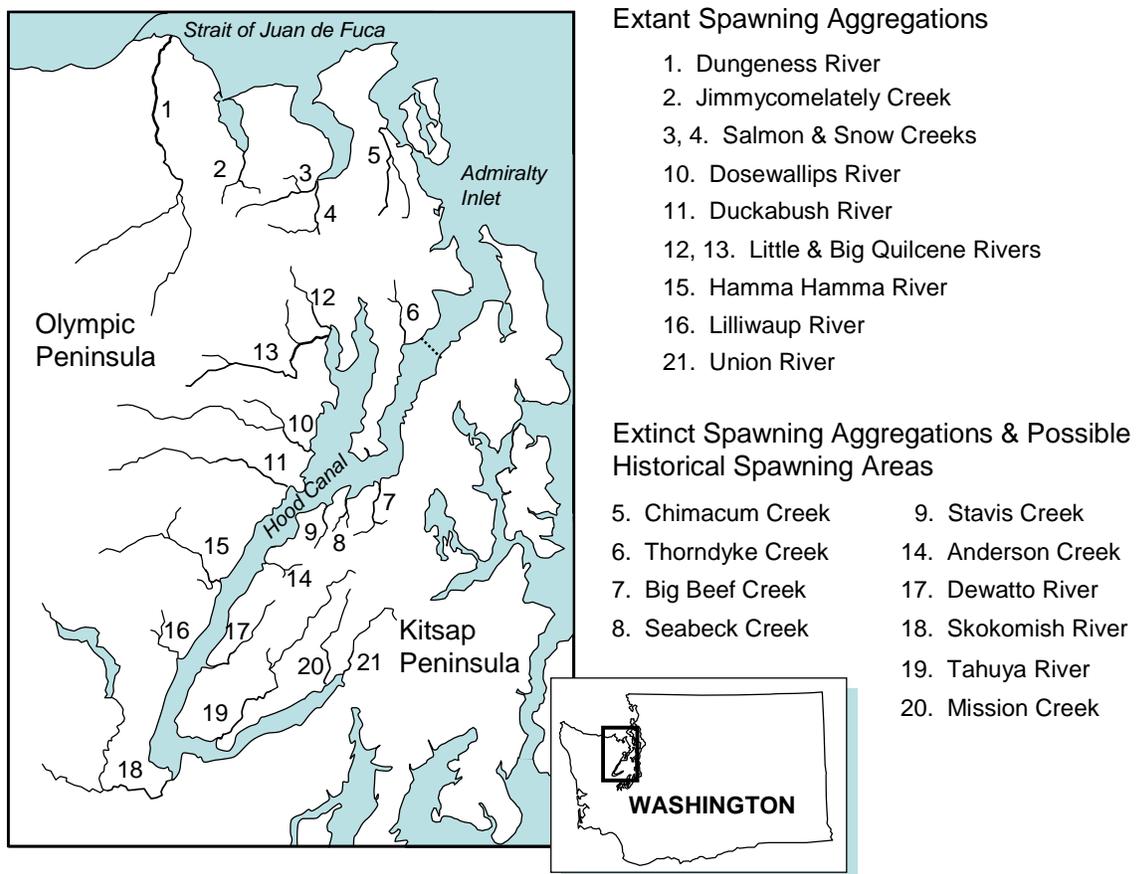


Figure 2. Extant and historical spawning distribution of the Hood Canal Summer Chum ESU as given in Sands et al. (2009). It should be noted that the Dungeness aggregation is misclassified, as its status (including its historic status) was—and remains—uncertain. Taken from Sands et al. (2009).

The objectives for this paper are as follows:

1. Review the status of the existing recovery goals for the Hood Canal summer chum ESU, assess and consider new information (recent run sizes, climate/ocean regime shifts, and climate change effects) for updating the goals, and provide specific recommendations for updated goals;

2. Identify the gaps between population and subpopulation performance associated with current baseline habitat conditions and appropriate updated recovery goals and targets;
3. Provide guidance for achieving population recovery in relation to the individual subpopulations, i.e., where should restoration work be focused and how much remains to be done there.

This report is organized into the following sections:

1. Introduction;
2. History of recovery planning for the ESU;
3. The ESU and existing criteria for recovery;
4. Updated quantitative viability assessment;
5. Shifts in climate and ocean regimes;
6. Considerations for effects of climate change;
7. Relative performance of subpopulations;
8. Recommendations.

2.0 History of Recovery Planning for the ESU

This section provides a brief history of recovery planning for the ESU.

2.1 Events Leading to ESA Listing

Beginning in the late 1970s, summer chum returning to the Hood Canal and the Strait of Juan de Fuca (SJDF) regions of Puget Sound showed significant downturns in abundance (Tynan 1992; Cook-Tabor 1994). The declines were initially most evident in streams in the Hood Canal region. Typical spawning escapements there fell from combined escapements of tens of thousands of fish for all streams in the 1970s to average annual returns by the early 1990s of under one thousand fish. In the SJDF region, where run sizes are typically smaller than those in the Hood Canal region, average spawning escapement fell from about 2,000 in 1968-1978 to a few hundred fish by the mid-1990s.

Factors contributing to the declines were identified to be habitat degradation (freshwater, estuarine, and nearshore marine), harvest rates greater than were sustainable, and changes in climate/ocean conditions contributing to reduced marine survival (WDFW and PNPTT 2000). It was evident that losses in habitat quantity and quality had accrued over many years as watersheds, their estuaries, and adjacent nearshore areas were logged and/or modified through development. Then, in the 1970s, harvest rates began to rapidly increase on these runs (Figure 3). Harvest rates on returning Hood Canal fish, for example, were found to have escalated so much that they sometimes approached, or exceeded, 80 percent, and averaged close to 60 percent in the 1980s. Canadian interceptions had also increased sharply in the 1980s, contributing to the high harvests. In addition, it was increasingly evident that natural survival in the ocean had undergone a sharp downturn (WDFW and PNPTT 2000). Several subpopulations associated with individual streams in the two regions were extirpated during this period (Johnson et al. 1997). In effect, all of these factors converged in the 1980s—as the productive resiliency of the populations for sustaining themselves was being exceeded.

By 1992, state and tribal co-managers had recognized the unfolding situation—and what it portended. They acted to provide greater protection for the remaining runs and to intervene for the sake of recovery. They adopted a harvest Base Conservation Regime (BCR), which reduced total harvest rates to

the current range of 3 to 15 percent for the Hood Canal population and to less than 2 percent for the SJDF population (Figure 3). Since 1992, Hood Canal and SJDF summer chum have only been caught incidentally in fisheries targeting other populations and species. Other conservation measures were applied, including the development and implementation of short-term hatchery supplementation and reintroduction actions aimed at boosting abundance while safeguarding genetic resources. Habitat degradation was recognized as occurring in all of the natal watersheds, as well as along the marine shorelines used by emigrating summer chum fry; therefore, the co-managers saw that habitat restoration and protection measures would need to be the centerpiece in a recovery initiative. In 2000, tribal and state co-managers distributed the Summer Chum Salmon Conservation Initiative (WDFW and PNPTT 2000), a comprehensive plan for the implementation of summer chum salmon recovery that integrated habitat, harvest, and hatchery actions.

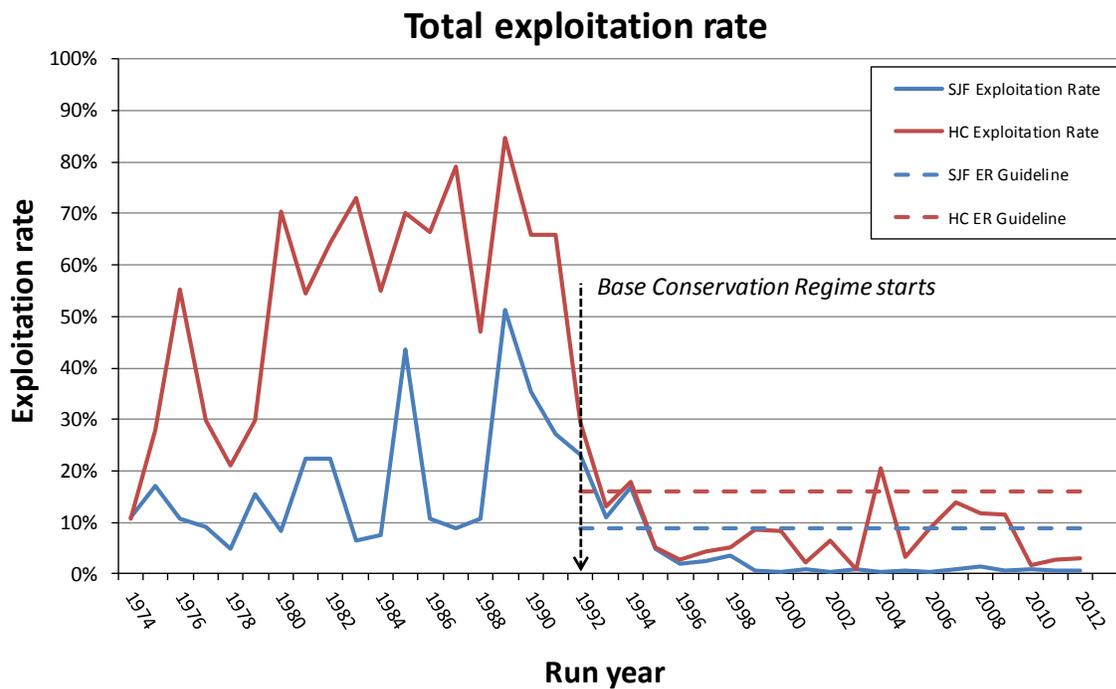


Figure 3. Estimated total exploitation rates on SJDF and Hood Canal summer chum in 1974 to 2012.

2.2 ESA Listing

In 1994, NMFS determined that the summer chum originating from the Hood Canal and the eastern SJDF regions constituted an ESU (Federal Register 1994). Subsequently, in March 1999, NMFS listed the ESU as threatened under the ESA. To help guide the ESA recovery effort, the Puget Sound Technical Recovery Team (PSTRT) was charged with identifying appropriate conservation units (i.e., independent populations) within the ESU and for developing recovery criteria consistent population viability as defined in McElhany et al. (2000).

The harvest and artificial production (hatchery) components of the co-mangers' Summer Chum Salmon Conservation Initiative (SCSCI) were subsequently approved by the National Marine Fisheries Service (NMFS) under Limits 6 and 5, respectively, of the Endangered Species Act 4(d) rule (NMFS 2001, 2002).

The Washington state legislature in 2005 designated the Hood Canal Coordinating Council (HCCC) to be the regional recovery organization for the Hood Canal Summer Chum ESU. The HCCC completed the Summer Chum Recovery Plan in November 2005 (HCCC 2005), which incorporated the SCSCI's harvest and artificial production management provisions. The HCCC recovery plan was subsequently adopted by the state of Washington in 2006 and by NMFS in 2007. Upon adoption by NMFS, that agency issued the federal supplement to the HCCC recovery plan. These documents together serve as the formal plan for recovery (NMFS 2007).

2.3 The Recovery Plan and Its Implementation

As noted above, the recovery plan for the Hood Canal summer chum ESU is comprised of two primary documents: the HCCC plan (HCCC 2005) and the federal supplement issued in 2007 (NMFS 2007). The HCCC plan focuses primarily on site-specific management actions aimed at protecting and restoring habitat needed to achieve population performance measures consistent with recovery. The HCCC plan also fully adopted and incorporated the co-managers' harvest and hatchery supplementation actions given in the SCSCI. In its review of the HCCC plan, NMFS concluded that the habitat actions combined with the harvest and hatchery elements would put both the Hood Canal and SJDF populations on trajectories toward recovery (NMFS 2007). NMFS noted that it expected further work would be done to address recovery priorities as new information becomes available. Since about 2000, major habitat actions in both regions have been implemented.

At the time when the HCCC plan was prepared, viability recovery goals had not yet been identified by the PSTRT. Therefore, the HCCC plan adopted interim recovery goals that had been developed by the co-managers under the SCSCI (PNPTT and WDFW 2003). The plan stated that the goals "are designed to provide numeric targets of summer chum salmon abundance and escapement for the purposes of recovery planning", and further, that "When realized, the recovery goals are expected to provide, on average, sufficient surplus abundance to allow for directed and incidental harvests of summer chum salmon." As such, the interim goals reflected to some extent broad-sense goals, though how much of the goals were thought to be needed for viability was not made clear. It also bears noting that the interim goals were simplified as they did not address all of the various aspects of population performance needed for recovery.

The interim recovery goals developed through the SCSCI were meant to represent the abundances of natural-origin adult summer chum that would be produced on average for each subpopulation (called stocks in the SCSCI³) in the ESU, if the quality and quantity of habitats were restored to conditions that produced the runs prior to the downturns in the late part of the 20th century. The premise of the approach was that the subpopulations were relatively healthy in the period shortly before the sharp downturns. Thus the interim goals were not meant to reflect historic production potential or even the full production potential that might be feasible through effective habitat and fisheries management actions. The authors of the interim goals stated: "However, these goals do point to recovery, at least at (subpopulation) levels that existed before recent (subpopulation) declines, and they can be modified when new information and assessments become available."

The interim goals were expressed as both an abundance of adult recruits (adults present prior to harvest) and the number of spawners needed to produce them (Table 1). Thus the goals also

³ / All references to "stock" in the SCSCI reports have been changed herein to "subpopulation" for consistency with how they are referred to in Sands et al. (2009) and within this paper.

incorporated an aspect of productivity, i.e., how effective, on average, each spawner needed to be in producing the next generation of adult recruits.

It is important to note that the co-managers considered it necessary to identify subpopulation-specific numeric recovery goals. They saw these goals as essential for measuring the success of recovery efforts at the natal watershed scale. Watershed-specific habitat restoration measures, harvest management actions, and hatchery supplementation programs could then be linked directly to the performance of each subpopulation and thereby to recovery progress in a manner considered most informative.

NMFS (2007), in issuing its supplement to the HCCC plan, drew on the findings of the PSTRT from its just-released draft report detailing the population structure of the ESU and related characteristics of viability for the ESU (Sands et al. 2009). NMFS accepted the PSTRT's ESU-level and population-level viability criteria as the appropriate biological component of the delisting criteria to be used for the ESU. The PSTRT formulated its numeric recovery goal for each population's abundance as the number of fish at the spawner stage (Table 1). Other aspects of the viability criteria, e.g., spatial structure and diversity, considered the number, distribution, and genetic diversity of distinct spawning aggregations (i.e., subpopulations) needed for recovery. Section 3 of this report describes in greater detail the various components of the recovery goals adopted by NMFS in its supplement (NMFS 2007).

NMFS (2007) accepted the co-managers' interim recovery goals for the eight extant subpopulations (Table 1) as appropriate short-term targets. The subpopulation-specific targets were seen as providing a logical intermediate step toward achieving the PSTRT's viability criteria for each population within the ESU. NMFS stated that it would use the long-term viability criteria (such as the population-level numeric goal) for its eventual delisting determination, but also acknowledged that the criteria may be refined as new information becomes available.

In 2009, the PSTRT issued its final report on the population structure of the ESU along with related viability criteria for achieving recovery (Sands et al. 2009). Portions of that report are summarized in the following section.

Table 1. Top – Projected range of minimum spawning escapement thresholds needed for recovery of the SJDF and Hood Canal summer chum populations with exploitation rates between 0 to 10% associated with expected ranges of capacity and productivity (includes all subpopulations within each population). The ranges encompass values derived using two models employed by the PSTRT (see Sands et al. 2009). Bottom – Co-managers’ interim subpopulation-specific recovery goals formulated by WDFW and PNPTCT (2003) and adopted by NMFS (2007) as intermediate goals (only subpopulations classified as extant at the time when the co-managers formulated the goals are shown).

NMFS (2007) population-level minimum recovery goals

Population	Minimum spawners	
	Low range	High range
SJDF	4,500	5,600
Hood Canal	17,900	24,700

Co-managers' interim goals by extant subpopulation

Population	Sub-population	Adult recruits	Spawners
SJDF	JCL	520	330
	Salm-Snow	1,560	970
	Total	2,080	1,300
Hood Canal	Quilcene	4,570	2,860
	Dosewallips	3,080	1,930
	Duckabush	3,290	2,060
	Hamma	6,060	3,790
	Lilliwaup	3,130	1,960
	Union	550	340
	Total	20,680	12,940

3.0 The ESU and Existing Criteria for Recovery

This section describes the population structure of the ESU and its associated diversity, abundance patterns since the early 1970s, and the different recovery criteria for the ESU as presented in Sands et al. (2009).

3.1 ESU and Population Structure and Diversity

The spatial structure of salmon populations and their biological diversity are closely associated (Waples et al. 2001; LCWTRT 2003); both are described here for the ESU. The ESU’s spatial structure refers both to the spatial distribution of breeding units within the ESU and the processes that generate that distribution (McElhany et al. 2000). Spatial structure of the spawning aggregations depends fundamentally on the spatial configuration of habitat, i.e., its distribution, quality, and quantity, and to demographic and genetic connectivity (strays) among the spawning aggregations. Biological diversity within the ESU consists of the diversity in genetic composition and phenotypic traits (such as life history

characteristics), the latter being controlled by genetics, environment, or interactions of genetic and environmental factors. Diversity in spatial structure promotes genetic variation (Waples et al. 2001; LCWTRT 2003), and is often reflected in life history variation (Waples et al. 2001; Hanski et al. 2004; Olsen et al. 2008). Spatial structure and biological diversity are two of the four VSP parameters that define viability (McElhany et al. 2000).⁴

The Hood Canal Summer Chum ESU is one of four genetically distinct lineages of chum salmon in the Pacific Northwest (Johnson et al. 1997). These lineages encompass all chum salmon populations produced in watersheds from Oregon to southern British Columbia. Each lineage is designated as an ESU on the basis of distinctive genetic and life history traits; they are identified as follows:

1. Pacific Coast ESU, which includes all natural populations from the Pacific coasts of California, Oregon, and Washington, west of the Elwha River on the SJDF;
2. Columbia River ESU, which includes all chum produced in streams of the Columbia River basin;
3. Puget Sound/Strait of Georgia ESU, which includes all chum populations from Puget Sound, the Strait of Georgia, and the SJDF up to and including the Elwha River, with the exception of summer chum from Hood Canal and the SJDF; and
4. Hood Canal Summer Chum ESU, which includes all summer chum produced in Hood Canal and the SJDF.

In 2009, the PSTRT published its conclusions about the population structure of the Hood Canal Summer Chum ESU, together with its assessment of performance characteristics needed for viability (Sands et al. 2009). Based on multiple lines of evidence, the PSTRT determined that the ESU consists of two independent populations: a SJDF population spawning in streams of the eastern SJDF and Admiralty Inlet, and a Hood Canal population spawning in rivers and streams that flow directly to Hood Canal. Lacking data on long-term migration rates between different spawning aggregations on which to identify populations and subpopulations, the PSTRT drew on various types of information to serve as proxies to help understand reproductive isolation of the aggregations. The information included allozyme and microsatellite DNA variation, straying patterns, historical and current geographical distributions, and life history and ecological variations.⁵

The PSTRT concluded that each of the two populations historically was spatially structured by the many independent stream drainages used by spawning aggregations spread through the geographic area encompassing the ESU (Figure 2). These aggregations occurred in the largest streams in the region (such as the Dosewallips and Skokomish rivers), as well as in some of the smallest streams (such as Salmon and Big Beef creeks).

In reaching this conclusion, the PSTRT found that an isolation-by-distance pattern best explains the genetic structure of the ESU based on genetic similarity analyses. Similarity relationships, as seen in the dendrogram in Figure 4, show a “chaining” pattern for summer chum comprising the ESU, which is a pattern where the dendrogram forms successive additions of branches associated with individual

⁴ / The four VSP parameters are abundance, productivity, spatial structure, and diversity.

⁵ / The PSTRT (Sands et al. 2009) referred to a collection of spawning aggregations as an independent population based on its extent of genetic and demographic isolation from other spawning aggregations, such that exchanges of individuals among the spawning aggregations do not substantially affect population dynamics or extinction risk of the different aggregations (i.e., independent populations) over a 100-year time frame. A subpopulation referred to a unique spawning aggregation that may be independent for periods less than 100 years or whose likelihood of persistence depends on limited exchanges of individuals with other such groups within the population.

spawning aggregations rather than as distinct clusters.⁶ We would note that the Hood Canal Summer Chum ESU does form a distinct cluster apart from the clustering that forms the Puget Sound/Strait of Georgia chum ESU. There is some evidence of separate clustering between the SJDF summer chum and Hood Canal summer chum, but the PSTRT thought that this may have been at least partly due to extirpations of some spawning aggregations.

When subpopulations occur geographically in line, as they generally do in this ESU, a one-dimensional stepping-stone pattern of straying can occur, where most genetic exchange is between neighboring subpopulations. The pattern can lead to significant genetic differences among subpopulations that are several subpopulations apart, even though there may be few or no observable differences between some adjacent subpopulations. The stepping-stone process of migration leads to patterns of genetic isolation by distance that can be detected with empirical data (Wright 1943).

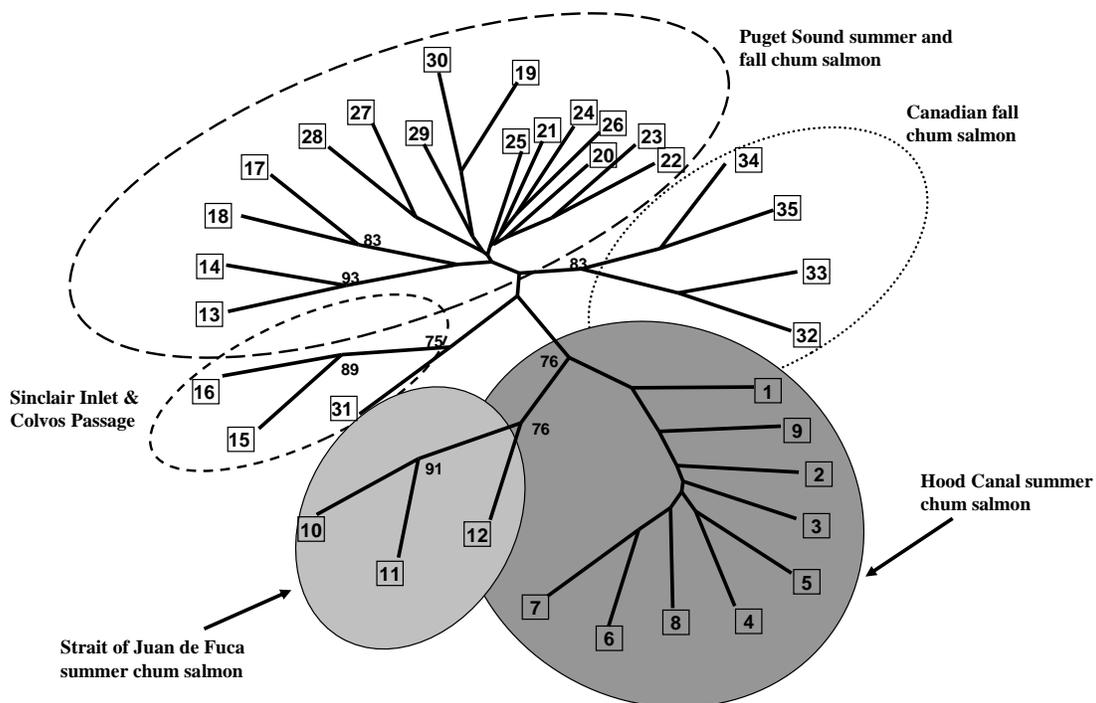


Figure 4. Relationship of 35 summer and fall chum salmon populations using Cavalli-Sforza and Edwards (1967) chord distance and neighbor-joining clustering (modified from Kassler and Shaklee 2003). Dendrogram shows chaining of Hood Canal summer chum salmon samples. Key to summer chum salmon samples: 1, Union River; 2, Lilliwaup Creek; 3, Hamma Hamma River; 4, Duckabush River; 5, Dosewallips River; 6, Big Beef Creek (introduced from Quilcene National Fish Hatchery); 7, Quilcene Bay; 8, Quilcene National Fish Hatchery (introduced from Quilcene Bay); 9, Little Quilcene River; 10, Snow Creek; 11, Salmon Creek; 12, Jimmycomelately Creek. Taken from Sands et al. (2009).

The PSTRT (Sands et al. 2009) found that it was likely that the extirpations of some subpopulations, such as those along the northeastern Olympic Peninsula and the northwestern Kitsap Peninsula, have

⁶ / A dendrogram is a tree diagram that serves as a visual representation of the relationships between populations or subpopulations within a species.

increased geographical isolation of the SJDF subpopulations from central Hood Canal and Union River subpopulations. Greater isolation has likely increased genetic drift and promoted greater allele frequency differences among these aggregations than occurred historically. Consequently, the observation that SJDF, central Hood Canal, Lilliwaup Creek, and Union River aggregations are evolving more or less independently (as suggested by the genetic analyses) may be an artifact of recent extirpations rather than a reflection of the long-term, viable population structure of the ESU.

Historically, the ESU was likely more connected by straying among subpopulations than occurs today because more subpopulations existed and they were more closely spaced. Extirpation of some subpopulations, for example, has resulted in the Union River now being more isolated both demographically and genetically than it was historically. Greater connectivity through straying would have limited the extent of genetic differentiation between neighboring subpopulations more than what is found today in genetic sampling of the extant subpopulations.

The PSTRT concluded, based on all of the genetic analyses and other lines of evidence as noted earlier, that the many subpopulations that existed historically comprised two independent populations and functioned as a single metapopulation. Straying among the subpopulations and populations served to maintain sufficient demographic and genetic connectivity among the spawning aggregations to help maintain the long-term viability of the ESU. Eight recognized subpopulations remain today, with a ninth possible aggregation in the Dungeness River. There is uncertainty about whether the Dungeness River represents a distinct subpopulation or whether it more aptly should be considered a minor spawning aggregation within the SJDF population (Sands et al. 2009). At least ten spawning aggregations have been extirpated.

The fragmentation of spatial structure in the past several decades has increased the risk of still more localized extirpations, which in turn has increased overall risk to each of the two populations and to the whole ESU. Small subpopulations within a fragmented spatial structure have increased vulnerability to the combined effects of deterministic factors (habitat degradation, overexploitation) and stochastic factors (demographic, environmental, genetic and catastrophic) (Morris and Doak 2002; Frankham 2005). Moreover, with loss of the contribution of strays from nearby subpopulations, small and more isolated subpopulations have a greater probability of inbreeding depression and loss of genetic diversity, thereby likely reducing fitness of individuals (Allendorf and Luikart 2007) and constraining the adaptability of subpopulations to future environmental change (Lande and Shannon 1996).

In considering the geographic distribution of the subpopulations, patterns of habitat characteristics across the ESU, and the patterns of genetic differentiation, the PSTRT concluded that all of the remaining extant subpopulations, as well as some that have been extirpated, are needed for the long-term viability of both populations and the ESU (Sands et al. 2009). The existing spatial structure and associated biological diversity were seen as critical to maintain and, in addition, the PSTRT concluded that several other extirpated subpopulations should be restored through reintroductions (as initiated by the co-managers with the SCSCI).

The importance of biological diversity within populations is well established. Greater diversity stabilizes population processes and provides increased resiliency to environmental change (MacArthur 1955; McElhany et al. 2000; Hooper et al. 2005). The benefits of biological diversity to long-term performance and sustainability of salmon populations are well supported in recent research (Hilborn et al. 2003; Moore et al. 2010; Schindler et al. 2010; Thorson et al. 2013).

Particularly noteworthy to this paper is the work of Hilborn et al. (2003) demonstrating the importance of biological diversity to salmon performance. They presented evidence that the sockeye population complex in Bristol Bay, which consists of many spawning aggregations, affords stability over long periods of climatic variation due to how various stock components respond differently to different climatic conditions. They concluded that different geographic and life history components that are minor producers during one climatic regime can be dominant producers during others, emphasizing that biological diversity (and accompanying spatial structure) is critical for maintaining resilience to environmental change. As will be seen later in this paper, evidence exists that a similar pattern, though on a smaller scale than in Bristol Bay, has existed—and likely still exists—for the Hood Canal Summer Chum ESU.

It is important to recognize here other characteristics of biological diversity within the ESU besides the genetic aspects described above. Some variations in life history and phenotypic characteristics among the subpopulations have been identified, though the PSTRT noted that relatively little information on this matter exists (Sands et al. 2009). Many life history traits of the ESU that have been assessed are similar among subpopulations, which may reflect a common adaptation to spawning in streams at a time when they are at or near their annual low flows and the early migration of juveniles from freshwater to estuarine and nearshore areas. Some traits, however, differ markedly and are relevant to this paper:

- Entry timing of adults from the marine areas into the natal spawning streams is earliest for the Union subpopulation, and is followed by the Quilcene subpopulation, which on average occurs about a week later than in the Union River, and then is followed by the other subpopulations, which occurs between about 10-14 days later than in the Union River (WDFW and PNPTT 2000). The latest entry timing occurs on average into Salmon and Snow Creeks. Average entry timing into Lilliwaup Creek, which is the subpopulation closest to the Union subpopulation, is nearly as late as in Salmon and Snow Creeks.
- Fry emergence timing from the spawning beds, which is virtually the same timing as fry entry into the nearshore environment, is estimated to be earliest for the eastside Hood Canal subpopulations and latest for the SJDF subpopulations, with peak emergence about three weeks later on average for the SJDF summer chum (Tynan 1997). Emergence timing for westside Hood Canal subpopulations is estimated to be intermediate between timing patterns for eastside Hood Canal and SJDF streams. Estimates of fry emergence timing were made through application of relationships developed on the amount of thermal units required from egg deposition to emergence (from research on Big Beef Creek by Koski 1975) and data on water temperature regimes for representative streams. Recent efforts to trap and enumerate outmigrant summer chum in several streams (e.g., Hamma Hamma River, Duckabush River, and Salmon Creek) also generally support these findings (personal communications, T. Johnson, PNPTC and M. Downen and J. Weinheimer, WDFW).

It is likely that these differences in spawning and fry emergence timing patterns among the subpopulations are adaptive and are important to the long-term performance of each subpopulation. Spawning timing is believed to be keyed primarily to water temperature regimes and other environmental factors that prevail during the incubation period in order to maximize survival of emergent fry under the prevailing natural conditions that exist at emergence (Miller and Brannon 1981; Healey 1982; Brannon 1987; Quinn et al. 2002). The timing of spawning migration and spawning is largely genetically controlled and therefore both can be highly selected for. Spawning date is the primary factor that controls when fry emerge from the gravel, thereby determining the environmental conditions that will be encountered by newly emerged

fry. Brannon (1974) reasoned that fry emergence timing and estuarine entry for pink and chum fry represents the most favorable balance of forces affecting survival, to the extent that compensatory mechanisms have evolved both in spawning time and rate of development to assure its timing repetition (as described in Tynan 1997). Fry emergence timing, therefore, tends to be stabilized so that fry find, on average, optimal conditions for survival (Miller and Brannon 1981). The potential relevance of these timing patterns is discussed further when we consider how the conditions for food abundance encountered by young summer chum fry in the nearshore and marine environments can differ both spatially and temporally in Hood Canal and the SJDF.

- Another variation in life history among the subpopulations is the relatively large proportion of the Jimmycomelately subpopulation that returns to spawn as small 2-year-old fish in some years (WDFW and PNPTT 2000, 2003, 2007, 2014; WDFW and PNPTC 2006). The PSTRT suggested that this distinct life history trait may be related to the significantly different genetic characteristics of Jimmycomelately summer chum. Other subpopulations in the ESU do not exhibit this trait as much.

In characterizing biological diversity of Pacific salmon species, Waples et al. (2001) found ecological diversity to be an informative indicator. Ecological diversity refers to the spatial and temporal patterns of abiotic and biotic conditions that comprise the ecosystems experienced by the fish over the course of their life histories. These conditions include such elements as streamflow regime, freshwater physical habitat characteristics, stream channel gradient, vegetative cover, geology, nearshore marine environmental attributes, and climate. These conditions characterize different selective regimes that salmon experience that promote adaptive genetic differentiation and life history diversity (Quinn et al. 2000; Waples et al. 2001; Beechie et al. 2006). Waples et al. (2001) divided the Pacific Northwest and California into 12 ecological regions, using ecoregions as defined by the Environmental Protection Agency (EPA) with some modification. Both life history and genetic diversity showed a strong, positive correlation with the extent of ecological diversity associated with the ecological regions.

The PSTRT applied a similar approach as that of Waples et al. (2001) to the Hood Canal Summer Chum ESU to formulate an indicator of how summer chum biological diversity may be structured within the ESU (Sands et al. 2009). The PSTRT used Level IV ecoregional units and sixth level hydrologic units (EPA 2004) and also considered distinct subregions within the marine environment into which the hydrological units flow. The resultant geographic groupings were called ecological diversity groups – seven groups were identified as given in Table 2 and mapped in Figure 5. The PSTRT used these ecological diversity groups for help in defining recovery criteria for spatial structure and diversity.

The foregoing description of the ESU's structure leads us to an important conclusion. Substantial structure and diversity have been lost, although there remains significant spatial structure and diversity in the ESU, as evidenced by its geographic pattern and distribution, genetic structure, and life history characteristics. Extirpations of many subpopulations have fragmented the ESU's structure, resulting in an increased risk of population extinction.

Table 2. Seven ecological diversity groups as proposed by the PSTRT for the Hood Canal Summer Chum ESU by geographic region and associated spawning aggregation. From Sands et al. (2009).

Geographic region (population)	Proposed ecological diversity groups (names)	Freshwater ecoregions headwaters	Freshwater ecoregions lower reaches	Spawning aggregations: Extant* and extinct**
Eastern Strait of Juan de Fuca	Dungeness	High Olympics Low Olympics Coast Range Volcanics	Olympic Rainshadow	Dungeness R (unknown)
	Sequim-Admiralty	Olympic Rainshadow	Olympic Rainshadow	Jimmycomelately Cr* Salmon Cr* Snow Cr* Chimacum Cr**
		Central Puget Sound Lowlands	Central Puget Sound Lowlands	Unknown
Hood Canal	Toandos	Olympic Rainshadow	Olympic Rainshadow	Unknown
	Quilcene	High Olympics Low Olympics Coast Range Volcanics	Coast Range Volcanics	Big Quilcene R* Little Quilcene R*
	Mid West Hood Canal	High Olympics Low Olympics Coast Range Volcanics	Coast Range Volcanics	Dosewallips R* Duckabush R*
	West Kitsap	Central Puget Sound Lowlands	Central Puget Sound Lowlands	Big Beef Cr** Seabeck Cr** Stavis Cr** Anderson Cr** Dewatto R** Tahuya R** Mission Cr** Union R*
	Lower West Hood Canal	High Olympics Low Olympics Coast Range Volcanics	Central Puget Sound Lowlands	Hamma Hamma R* Lilliwaup Cr* Skokomish R*

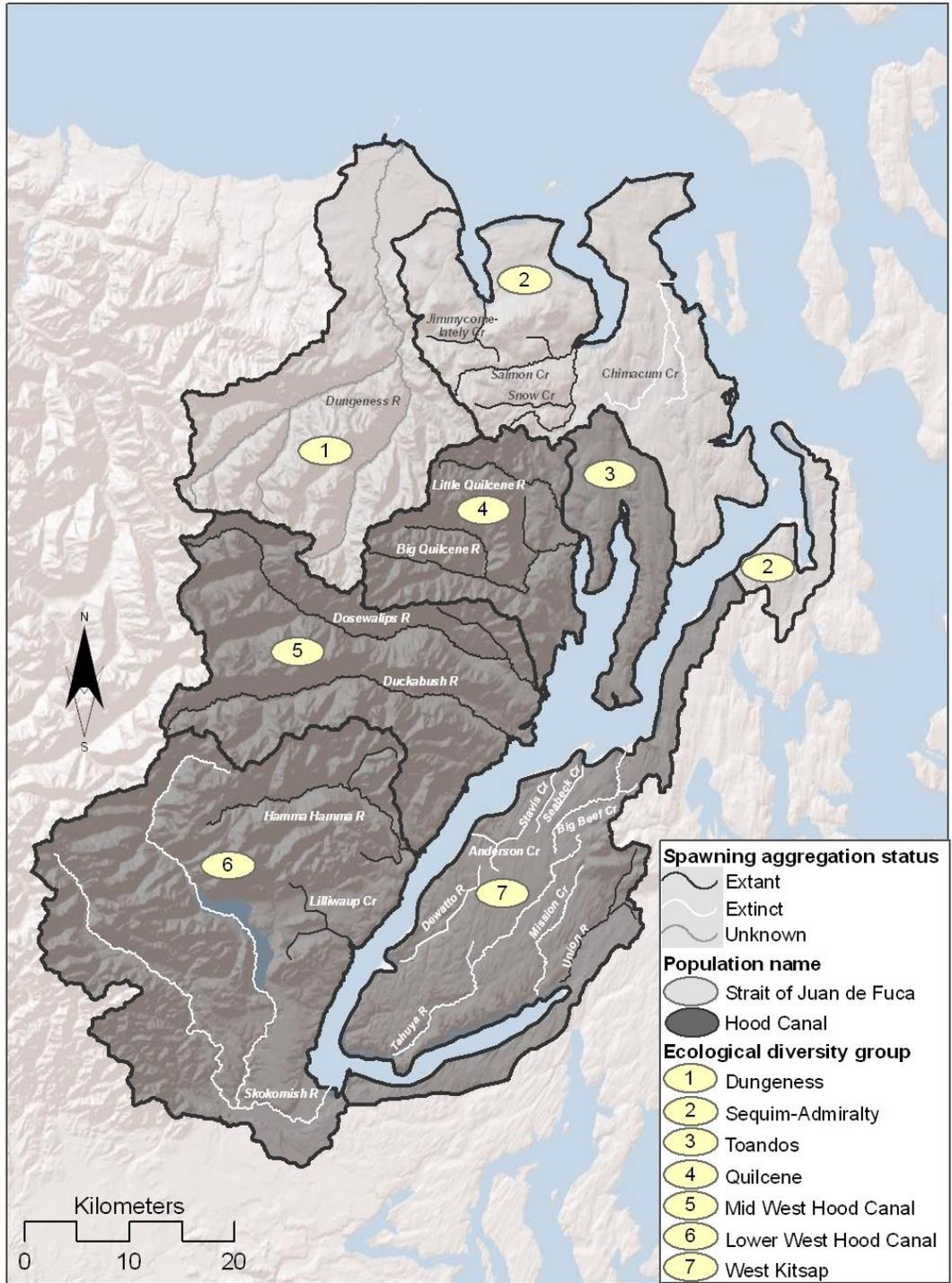


Figure 5. The seven ecological diversity groups delineated by the PSTRT for the Hood Canal summer chum ESU. Taken from Sands et al. (2009).

3.2 Abundance Patterns

Since 1974, the Hood Canal and SJDF populations and their subpopulations have exhibited similarities in abundance patterns, though striking differences exist as well. For each population, Figure 6 shows the

estimated total numbers of adult recruits (pre-harvest), incorporating data for all spawning aggregations for which data exist (extant and extirpated) on the left side of the figure and with data from only extant spawning aggregations on the right side. The number of adult recruits is the sum of the number of spawners and the number of fish harvested (see Section 4.3 for a description of how recruits have been estimated). The figure incorporates both natural-origin and hatchery supplementation-origin fish (only natural-origin fish would count in comparing to a recovery goal). No other summer chum hatchery fish exist in the ESU besides those that have been produced in supplementation (or reintroduction programs). Also shown on the figure are several reference points (displayed as horizontal lines). Both the low and high range values adopted by NMFS for minimum spawner abundance are shown, as well as the total abundance of recruits for each population identified by the co-managers as interim goals based on the sum of the recruits for only extant subpopulations (see Table 1).

The right side of Figure 6 best illustrates the temporal patterns for adult recruits by using only streams where spawning fish have been observed and counted in all years since 1974. Some of the data incorporated into the left side of the figure are from streams where counts did not exist in all years. The patterns for the two populations show the following with respect to natural-origin recruits:

- Relatively high abundances existed in the 1970s, followed by subsequent downturns that extended through the late 1990s, with a return to relatively high abundances after about 2000.
- The downturn for the SJDF population occurred somewhat later than it did in Hood Canal.
- The return to high abundances after 2000 for the SJDF population produced greater abundances relative to the 1970s for this population than in Hood Canal, with the exception of one year after 2000 in Hood Canal that had an exceptionally high abundance.

For each subpopulation, Figures 7-11 show the estimated total numbers of adult recruits returning (pre-harvest) to their streams of origin for 1974 to 2012. For the extant subpopulations, the figures also show the abundance of adult recruits identified by the co-managers as interim goals (see Table 1).

Figure 7 shows abundance patterns for the two extant SJDF subpopulations together with data that exist for Chimacum Creek, which is a reintroduced stock due to the extinction of the indigenous fish. A supplementation program, begun on Salmon Creek in 1992, was conceived with the objectives to rebuild and stabilize the Salmon-Snow Creek subpopulation and to allow for the transfer of surplus eggs or fry to reintroduce summer chum to Chimacum Creek. The reintroduction program in Chimacum Creek was initiated in 1996 and ended in 2003 and the summer chum are considered a range extension of the Salmon-Snow Creek subpopulation. The figure shows the following:

- The temporal patterns for the two extant subpopulations are similar though the upturn since 2000 in abundance occurred earlier in Salmon-Snow Creek and was stronger than in Jimmycomelately Creek. The delay in Jimmycomelately Creek may have been at least partly due to the extremely low abundances that occurred in this stream in the late 1990s and the later start (in 1999) of a supplementation program and the first age 3 returns from the program in 2002.
- Summer chum have shown a strong response to the reintroduction program and the availability of suitable natural habitats in Chimacum Creek, in part due to the extensive habitat protection and restoration actions that have been implemented. The abundance of returning natural-origin fish being similar to numbers in Jimmycomelately Creek in recent years. It should be noted that routine spawner counts began in Chimacum Creek in the 1990s.

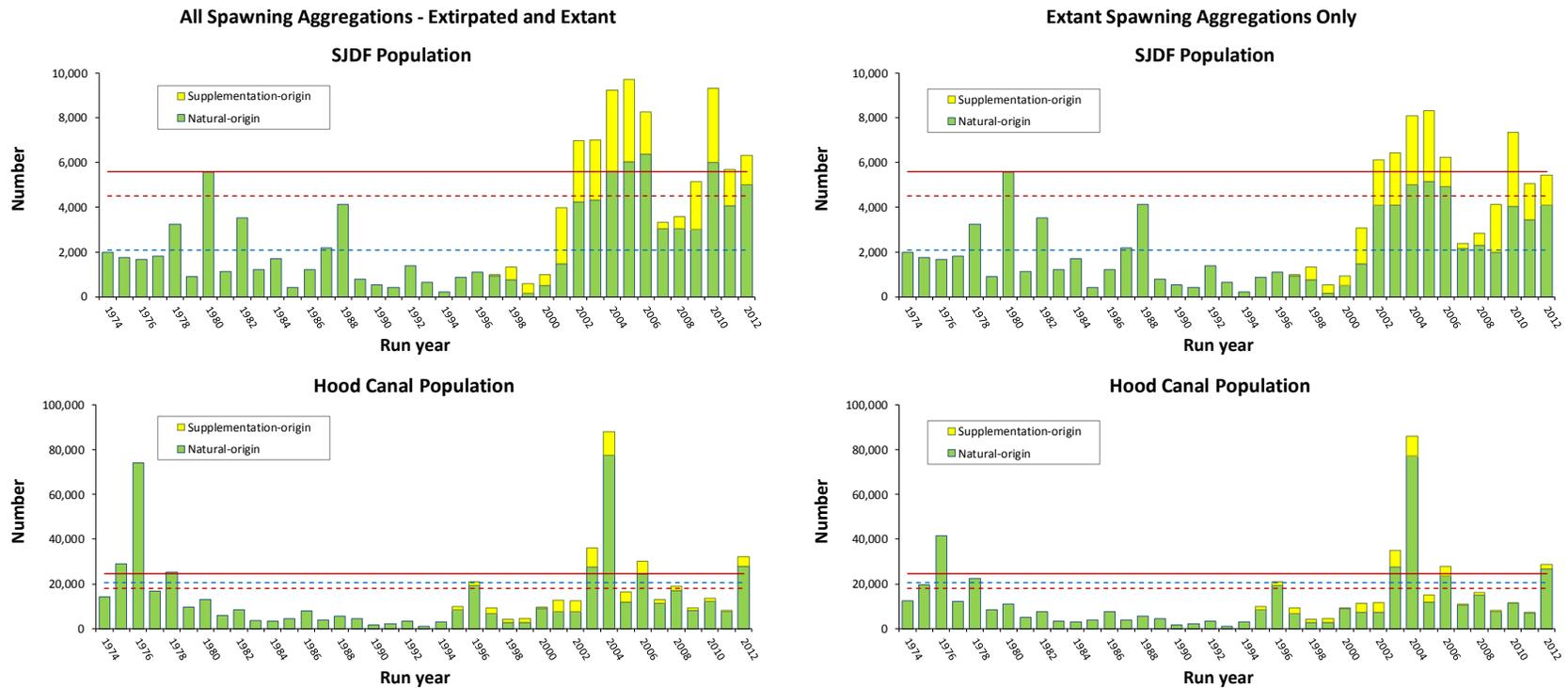


Figure 6. Left side – Estimated numbers of adult recruit summer chum for the two populations within the Hood Canal Summer Chum ESU returning by year in 1974 to 2012, incorporating all spawning aggregations for which data exist, extirpated and extant. Right side – Estimated numbers of adult recruits incorporating only data for extant subpopulations. The solid red line is the high range estimate for the minimum natural-origin spawner abundance threshold for recovery, the dashed red line is the low range estimate, and the dashed blue line is the aggregate co-managers’ interim abundance goal (see Table 1). Recruits of naturally spawned (natural-origin) and hatchery spawned (supplementation-origin) fish are shown.

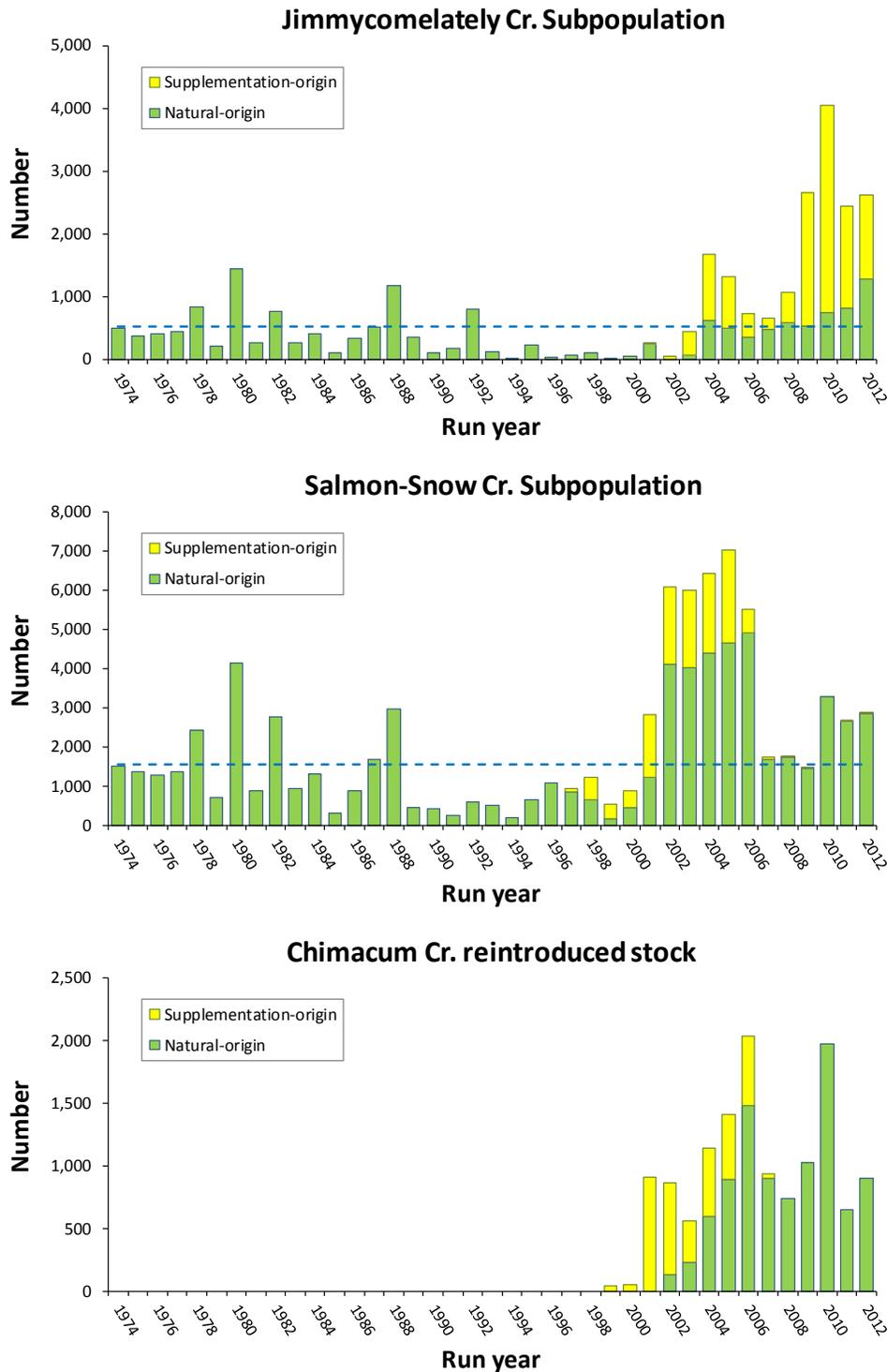


Figure 7. Estimated numbers of adult recruit summer chum comprising the Jimmycomelately Creek and Salmon-Snow Creek subpopulations of the SJDF population returning by year in 1974 to 2012, and estimated adult recruits resulting from reintroduction efforts in Chimacum Creek. The number of adult recruits represents fish returning to spawning streams plus estimated harvest in the return year. The dashed blue line is the co-managers' interim natural-origin adult abundance goal for the subpopulation (see Table 1). Recruits of naturally spawned (natural-origin) and hatchery spawned (supplementation-origin) fish are shown.

Figures 8-9 show numbers of adult recruits for the six extant Hood Canal subpopulations. The charts for the subpopulations are arranged from north to south. The figures show the following:

- Abundances for each of the extant subpopulations were relatively low during the 1980s and 1990s with some deviations from this pattern in a few years for three of the subpopulations:
 - Union River showed an upturn for a few years in the mid to late 1980s; and
 - The Duckabush and Dosewallips rivers showed notable increases for two years in the mid-1990s.
- All of the extant subpopulations have demonstrated clear increases in production since about 2000 compared to the 1980s and 1990s, with exceptions as noted above, although the upturn in Lilliwaup Creek has been modest but it appears to be building. It is noted that spawning escapements in Lilliwaup Creek were extremely small in the late 1990s and the delayed upturn is likely at least partly related to that condition.
- Four of the six subpopulations, i.e., not including the most northern subpopulation or the most southern subpopulation, produced relatively strong runs in the 1970s, generally stronger or equivalent to production levels that occurred after about 2000. In the 1970s, production from the most northern subpopulation—the Quilcene—was generally moderate in size relative to most years since 2000. In contrast to all of the other subpopulations, production in the Union River in the 1970s was low, generally lower than other years after 1980.

Figure 10 shows adult recruits for two extirpated subpopulations where reintroductions have occurred belonging to the Hood Canal population. The reintroduction program on Big Beef Creek was initiated in 1996 using Quilcene stock and the program ended in 2004. The reintroduction program on Tahuya River was initiated in 2003 using Union River stock; this program is still underway and is scheduled to end in 2014. Figure 10 shows the following:

- Both of these extirpated subpopulations had high levels of production prior to 1980, similar to the patterns seen for all of the extant subpopulations in the southern half of Hood Canal except for Union River. The downturns in production in both of these subpopulations generally mirrored those seen in the other subpopulations except in Union River—though for these two, production has not rebounded except for a small response as a result of the reintroduction effort in Tahuya River.
- The responses of natural production in Big Beef Creek and Tahuya River to the reintroduction efforts have been small, in stark contrast to the response in Chimacum Creek (Figure 7). The reintroduction programs contributed to increases in summer chum abundance in both Big Beef Creek and Tahuya River, but natural-origin production has not been sustained. This may indicate that the availability of suitable natural habitats is limited and that more habitat protection and restoration actions are needed.

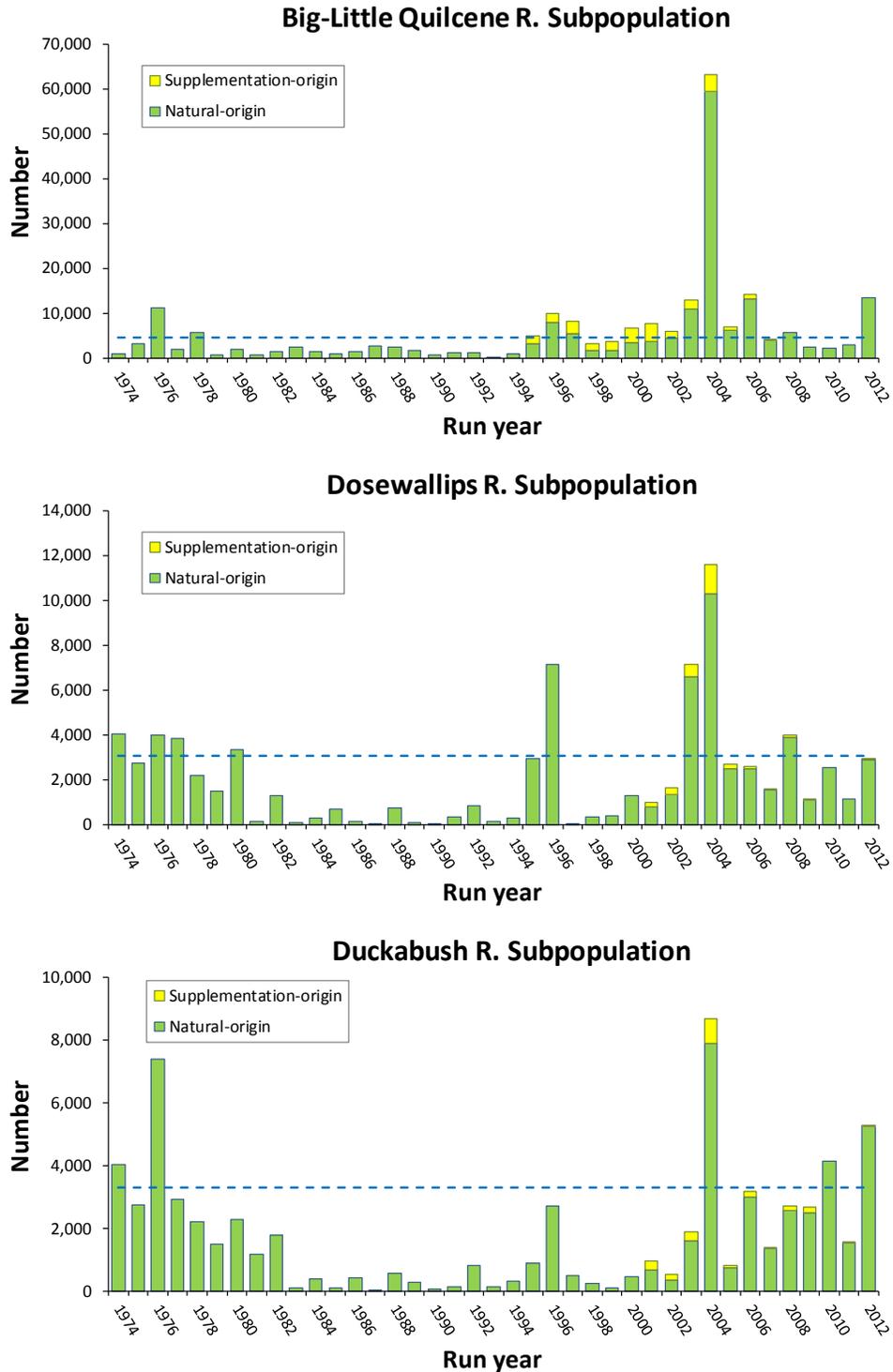


Figure 8. Estimated numbers of adult recruit summer chum comprising the Big-Little Quilcene, Dosewallips, and Duckabush rivers subpopulations of the Hood Canal population returning by year in 1974 to 2012. The number of adult recruits represents fish returning to spawning streams plus estimated harvest in the return year. The dashed blue line is the co-managers’ interim natural-origin adult abundance goal for the subpopulation (see Table 1). Recruits of naturally spawned (natural-origin) and hatchery spawned (supplementation-origin) fish are shown.

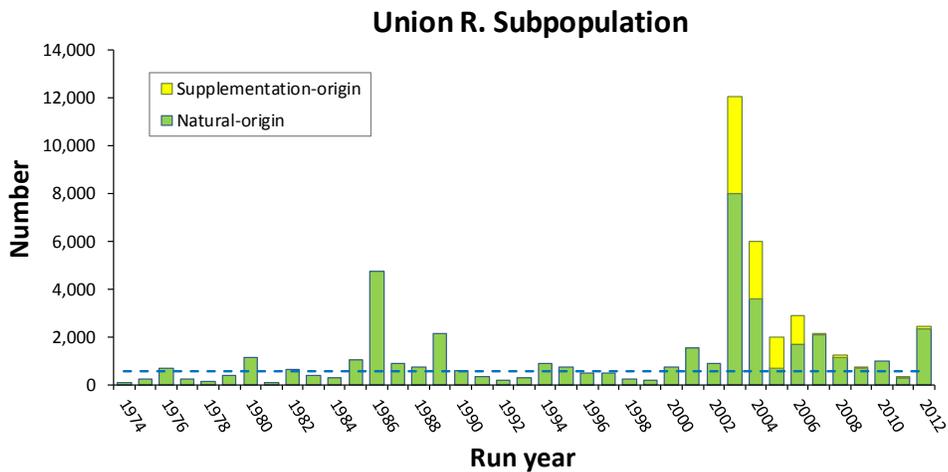
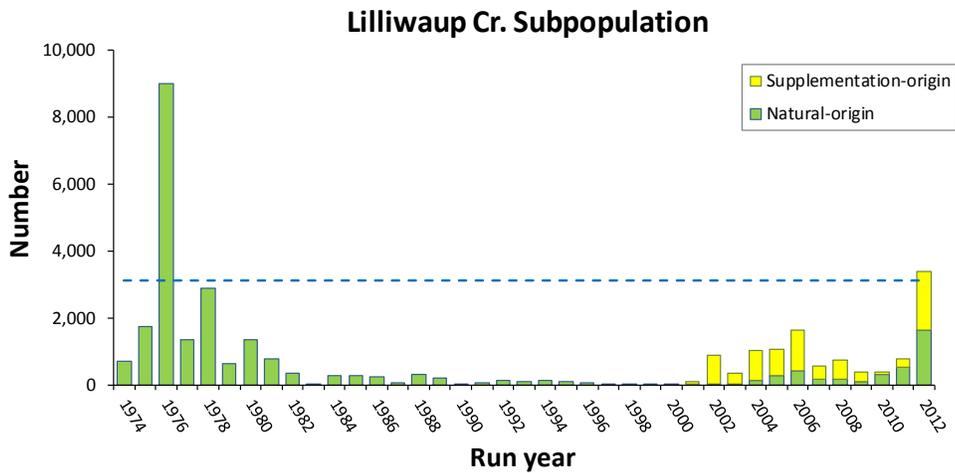
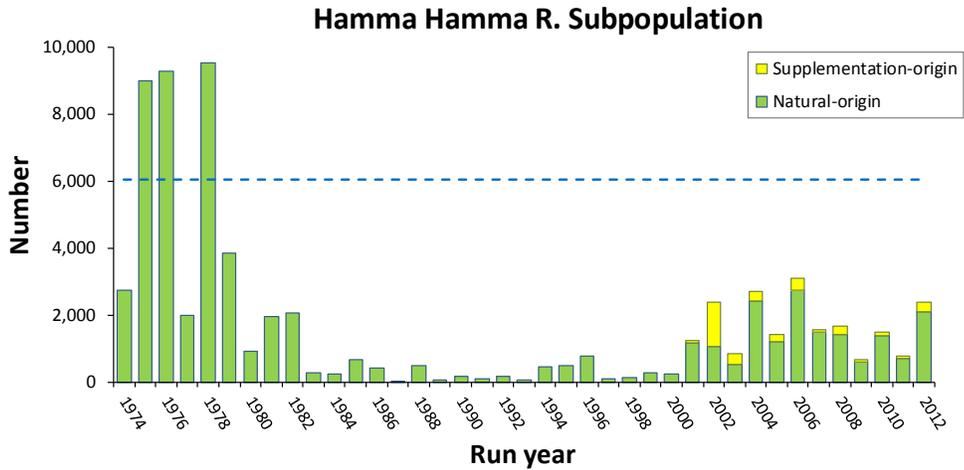


Figure 9. Estimated numbers of adult recruit summer chum comprising the Hamma Hamma River, Lilliwaup Creek, and Union River subpopulations of the Hood Canal population returning by year in 1974 to 2012. The number of adult recruits represents fish returning to spawning streams plus estimated harvest in the return year. The dashed blue line is the co-managers’ interim natural-origin adult abundance goal for the subpopulation (see Table 1). Recruits of naturally spawned (natural-origin) and hatchery spawned (supplementation-origin) fish are shown.

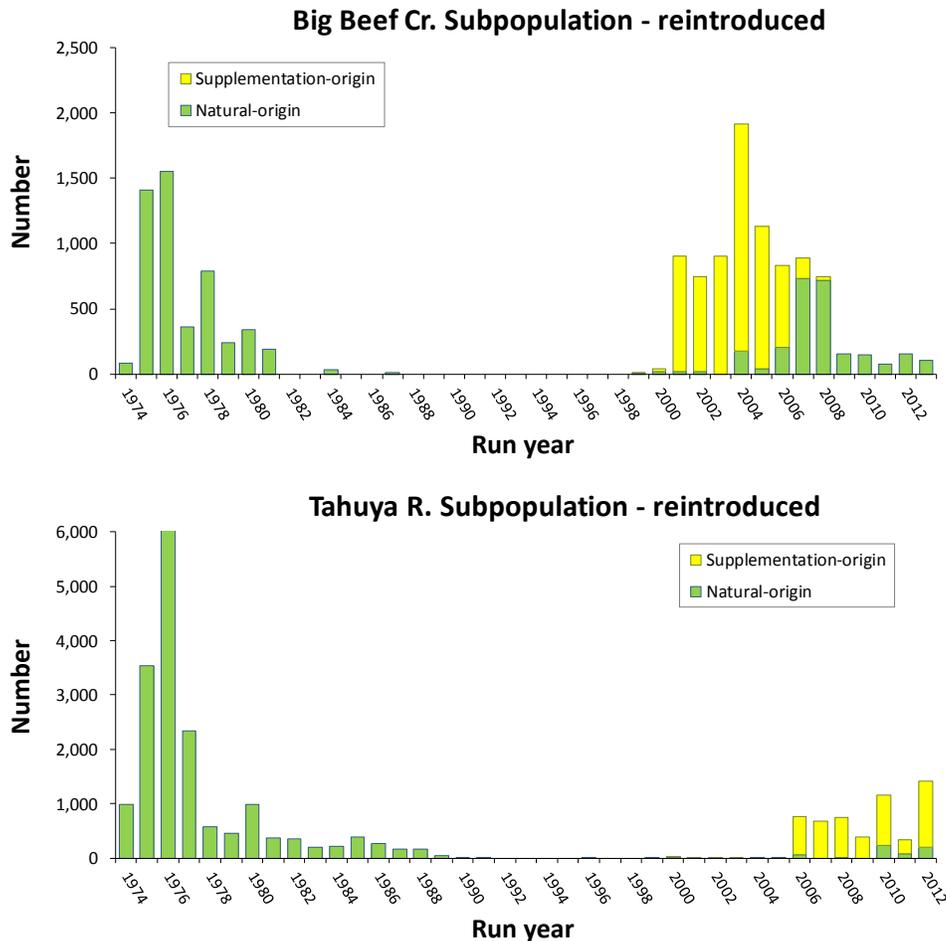


Figure 10. Estimated numbers of adult recruit summer chum resulting from reintroduction efforts in Big Beef Creek and estimated recruits comprising the Tahuya and Dewatto River subpopulations returning by year in 1974 to 2012. Results of reintroduction efforts in Tahuya River are also shown. All three subpopulations have been determined to be extinct. Recruits of naturally spawned (natural-origin) and hatchery spawned (supplementation-origin) fish are shown.

Figure 11 shows adult recruits for three of the extirpated spawning aggregations belonging to the Hood Canal population where no reintroduction efforts have occurred. Two of the aggregations, Dewatto River and Skokomish River, would be considered subpopulations, while the third, Anderson Creek, was probably a minor spawning aggregation due to its smaller size. It is important to recognize that no spawning escapement data exist for the Skokomish River prior to about 2000. The numbers of recruits estimated for all prior years for Skokomish River are based entirely on catch estimates using run reconstruction but with escapements set to zero; hence estimated total recruits for the 1970s and 1980s (i.e., years before major fishery cutbacks) are underestimated, likely by a substantial amount. Figure 11 shows the following:

- Each of these three spawning aggregations had high levels of production prior to 1980, similar to the patterns seen for all of the other subpopulations in Hood Canal except for Union River and to some extent for Big and Little Quilcene River. The downturns in production in these three aggregations mirrored those seen in the other subpopulations except for Union River.
- Since about 2000, the Dewatto River and Anderson Creek spawning aggregations have demonstrated little or no increase in abundance. Skokomish River is showing a marked increase

in production, in part due to supplementation-origin fish straying from programs on Lilliwaup and Tahuya rivers nearby (PNPTT and WDFW 2014).

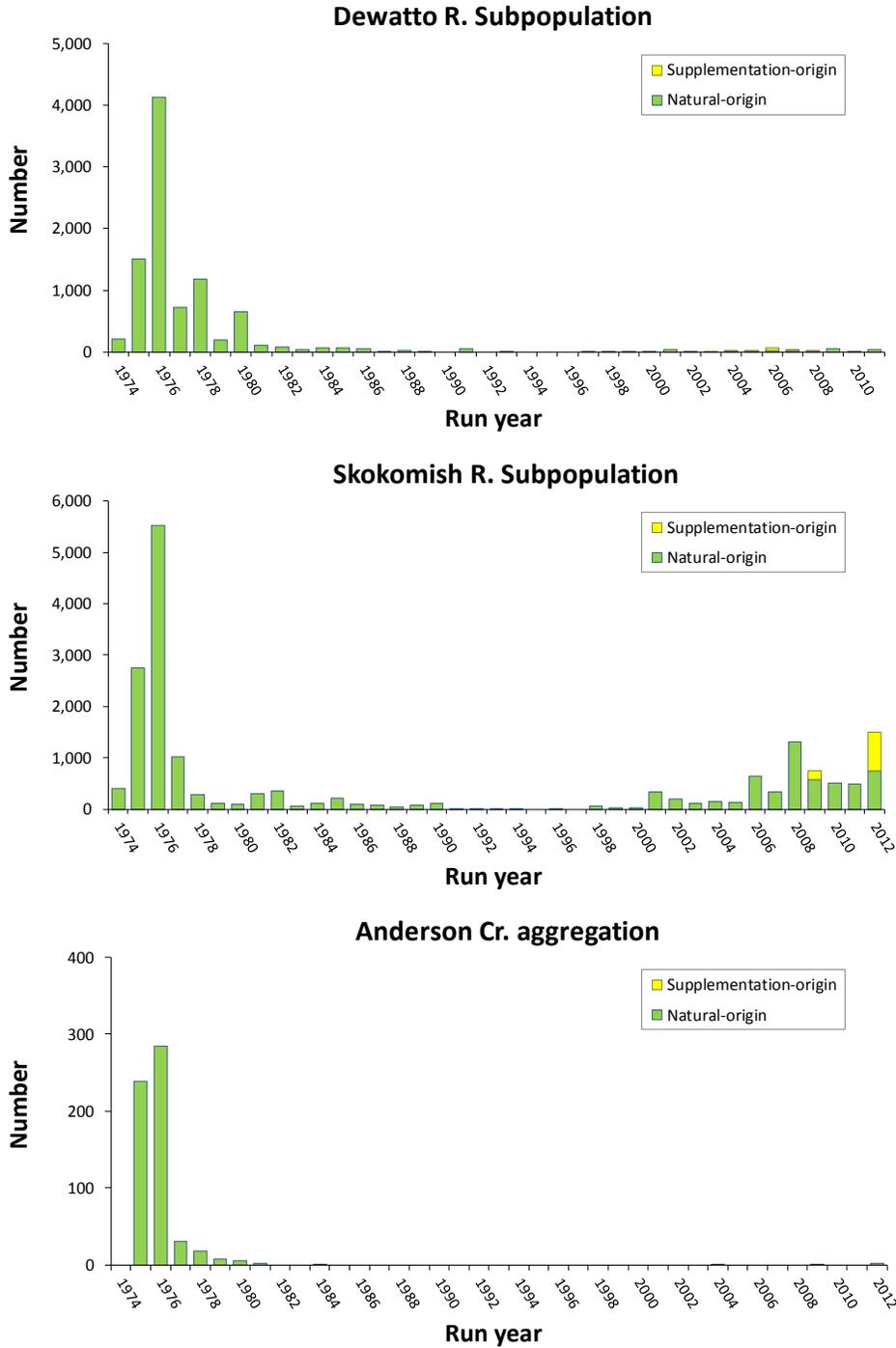


Figure 11. Estimated numbers of adult recruit summer chum comprising the Dewatto River, Skokomish River, and Anderson Creek subpopulations of the Hood Canal population returning by year in 1974 to 2012. The number of adult recruits represents fish returning to spawning streams plus estimated harvest in the return year. Recruits of naturally spawned (natural-origin) and hatchery spawned (supplementation-origin) fish are shown.

In summary, the abundance patterns for the populations and subpopulations show the following:

- Adult production was relatively strong for both populations and their subpopulations in the early years of the data record, with the notable exception of Union River, lasting to about 1980 in Hood Canal and into the late 1980s in the SJDF.
- Following these time periods, adult production generally declined sharply for both populations and their subpopulations, continuing through the 1990s, though notably the Union, Duckabush, and Dosewallips rivers exhibited brief periods when abundance jumped significantly higher relative to other years during this time.
- Since the early 2000s, production of natural-origin adults has rebounded sharply for both populations and their extant subpopulations, continuing to the present time, though the response in Lilliwaup Creek has been slow. It appears that Skokomish River, considered to be extirpated and having no reintroduction program, is also showing some amount of rebound.
- Some subpopulations that exhibited relatively strong adult production prior to about 1980 showed precipitous declines shortly afterwards and have not rebounded; these are considered to have been extirpated. The situation for the Skokomish River is unclear due to some apparent amount of rebound.
- Of the three streams where reintroductions have been made, only Chimacum Creek has demonstrated a strong positive response; responses in Big Beef Creek and Tahuya River have thus far been small.

It is noteworthy that the subpopulations demonstrate both synchrony and asynchrony over the period of record. The general pattern exhibited by most subpopulations is reflected in the patterns for the populations: high production during the 1970s, followed by declines that bottomed in the early to late 1990s, followed by a rebound that continues to the present time. However, some deviations from this pattern are notable and are relevant to material presented later in this paper. Both the Union and Quilcene subpopulations exhibited relatively low production in the 1970s, particularly in the Union River, demonstrating some level of asynchrony among the subpopulations. Also, the Union, Duckabush, and Dosewallips subpopulations showed substantial jumps in production at different times during the period of general decline for the populations, further demonstrating periodic asynchrony.

Asynchrony among spawning aggregations within a population complex suggest local scale differences in environmental factors affecting performance of different aggregations (e.g., Rogers and Schindler 2008; Schindler et al. 2010). Asynchronous performance responses are evidence of biological diversity, and such responses have been referred to as a portfolio effect, analogous to the effects of asset diversity on the stability of financial portfolios (Schindler et al. 2010).

3.3 Recovery Criteria

The PSTRT presented viability criteria for both populations, as well as the ESU as a whole, based on guidance given in McElhany et al. (2000) (Sands et al. 2009). Each of the four criteria evaluating viability—abundance, productivity, spatial structure, and diversity—were addressed. NMFS defines a viable population as one that is self-sustaining with a negligible risk (<5%) of extinction over a 100-year time frame. The abundance and productivity criteria were evaluated using quantitative modeling; spatial structure was evaluated using a quantitative measure but the diversity criterion was expressed qualitatively.

Based on population modeling, using methods described in Section 4.0 of this report, the PSTRT identified abundance and productivity criteria to achieve a negligible risk of extinction for each of the two populations (Sands et al. 2009). It was estimated that at levels of population performance needed to achieve these criteria, average spawning escapements with exploitation rates between 0 and 10 percent would need to be between 4,500 and 5,400 spawners for the SJDF population, and between 17,900 and 21,500 spawners for the Hood Canal population.⁷ Further details about these quantitative viability criteria are described in Section 4.0.

The PSTRT determined that the spatial structure of the extant subpopulations and their biological diversity were critical to maintain—and restore to some extent through reintroductions in streams where extirpations had occurred—to achieve a negligible risk of extinction for the ESU. Applying principles of conservation biology that link population structure and diversity to patterns of ecological diversity across a landscape (e.g., Healey and Prince 1995; McElhany et al. 2000; Waples et al. 2001), the PSTRT delineated ecoregions within the boundaries of the ESU and an associated seven ecological diversity groups, as described in Section 3.1 (Table 2 and Figure 5).

The geographic areas associated with these groups were recognized as having distinct ecological conditions that would be experienced by summer chum produced within them. The premise was that the amount of spatial structure and biological diversity needed for population viability could be met if persistent subpopulations/spawning aggregations exist within all of these ecological diversity groups. Sufficient biological diversity should, therefore, be manifested over time to provide for population stability and resilience with significant environmental variation (i.e., provide a portfolio effect). It is important to note that while subpopulations/spawning aggregations need to be persistent (and that means not going extinct), persistence does not carry the rigor of viability (i.e., <5 percent risk of extinction over a 100-year time frame) that is used at the population level. We reason, however, that persistence in this context implies that the subpopulation/spawning aggregation must contribute positively to overall population health and viability and not merely be persisting near the edge of extinction. This suggests that the long-term performance of the subpopulation/spawning aggregation should be sufficiently robust to provide beneficial contribution at the population level.⁸

The criteria for spatial structure and diversity were established to be the following, respectively:

- Most spawning aggregations need to be within 20 km of adjacent aggregations, and the major subpopulations (larger river systems) need to be less than approximately 40 km apart;

⁷ / See Tables 7 and 8 in Sands et al. (2009). Spawning escapements would need to be higher at higher exploitation rates.

⁸ / Fausch et al. (2006) provide some useful insights for understanding “persistence” within the context of this paper: “Often the terms “persistence” and “viability” are used interchangeably, but here we recognize viability as a larger conservation objective. The basic impetus for conservation planning is not simply to guarantee persistence of a species, but to ensure that natural ecological and evolutionary processes are allowed to continue and perhaps change through time. For a single species, this broader view of maintaining process, not just persistence, is referred to here as “viability” (see McElhany et al. 2000). In the prioritization process, this may equate to conservation of both evolutionary and ecological values simultaneously. For example, evolutionary values can be associated with genetically pure but isolated populations that may persist for some time, but if those populations cannot evolve and adapt with changing environments they may not be viable in the long term. Populations that are likely to persist and remain viable represent a higher overall value and logically a higher priority in any assessment of risk. In short, persistence is generally viewed as a necessary but not sufficient objective for attaining full conservation of a species.”

- One or more subpopulations/spawning aggregations need to be persistent within each of the ecological diversity groups within each population (Table 2).

It is noted that the PSTRT determined that uncertainty still exists about the nature of any spawning aggregations within the Dungeness and Toandos ecological diversity groups (Sands et al. 2009); no firm determination has been made about the need for recovering spawning aggregations within these groups.

Notwithstanding the criterion above, the PSTRT stated that all of the extant subpopulations need to remain persistent, due to the significant loss in biological diversity that has already occurred (Sands et al. 2009). We interpret persistent to mean here as explained above.

Regarding the need for reintroductions in streams where subpopulations have been extirpated, the PSTRT stated: “Although it may not be necessary to reestablish spawning aggregations in all rivers and streams where they historically occurred, meeting spatial structure population viability criteria in the Hood Canal and Strait of Juan de Fuca populations will require reestablishing spawning aggregations in many of the major rivers and the smaller streams and creeks where they have been extirpated.”

For the ESU to meet all of the viability criteria, the PSTRT determined that both populations need to achieve a negligible risk of extinction (Sands et al. 2009).

4.0 Updated Quantitative Viability Assessment

This section provides an updated quantitative viability assessment from the one presented in Sands et al. (2009). We also provide a general description of the methods used in performing the analysis.

4.1 Basic Concepts

Basic concepts used in performing the viability assessment are described here. A general understanding of the concepts and associated terms is needed for the reader to follow much of the presentation through the remainder of this paper.

Risk of extinction of a population can be directly related to the combination of two characteristics (or parameters) of a population: abundance and productivity. While all four viability criteria identified by McElhany et al. (2000) are important and informative to risk assessment, an assessment of the combination of abundance and productivity provides a convenient means of evaluating risk separate from the other two population characteristics (diversity and spatial structure). (Still, a full risk assessment requires an evaluation of all four population characteristics.)

The VSP guidelines developed by McElhany et al. (2000) present the rationale for considering these two parameters in combination—if productivity increases, the required abundance for viability generally decreases. The other two parameters can usually be defined independently from abundance and productivity (though the argument can be made that if diversity is low, abundance needs to be higher). The guidelines recommend that a viable population should:

- Be large enough to have a high probability of surviving environmental variation observed in the past and expected in the future;

- Be resilient to environmental and anthropogenic disturbances and support/provide ecosystem functions;
- Demonstrate sufficient productivity to support a net replacement rate of 1:1 or higher at abundance levels established as long-term targets;
- Demonstrate productivity rates at relatively low numbers of spawners that, on the average, are sufficiently greater than 1.0 to allow the population to rapidly return to abundance target levels after perturbations.

Each of these guidelines capture aspects of both abundance and productivity.

We would note here that the definition and use of the term “productivity” can differ somewhat in the salmon recovery literature; it is important to recognize the distinction in two uses of the term to understand its use in this paper. Often the term is used to mean the population’s growth rate from one generation to the next; in this sense it is the number of adult progeny produced per parent spawner (or recruits per spawner) measured for each generation. The term is also used to refer to what is called intrinsic productivity, which McElhany et al. (2000) defines as the maximum population growth rate when free of density-dependent limitations. Population growth rate for salmon populations, expressed simply as recruits per spawner, is highly density dependent for populations that fluctuate widely.⁹ For the remainder of this paper, the term productivity will be used in the sense of intrinsic productivity, as it is often applied in population dynamics literature (e.g., Hilborn and Walters 1992).

The two parameters, productivity and abundance, are conceptually seen in a spawner-recruit (S-R) relationship (Figure 12A). S-R relationships are widely used in fisheries science to conceptualize and define the basic underlying performance characteristics of a fish population. The relationship is a theoretical depiction of how adult progeny, or recruits, varies (on average) in relation to the size of the reproducing parent population (Ricker 1954; Beverton and Holt 1957; Ricker 1975). In Figure 12A, the diagonal straight line (where recruits equal parent spawners) is called the replacement line. Over some period of years, during which environmental conditions might be more or less constant, even accounting for year to year variation, the population would tend toward some equilibrium abundance in the absence of fishing, which is where the replacement line and the S-R relationship intersect. At the equilibrium point, the spawner population just replaces itself in the next generation. Here, a population’s growth rate has a value of 1 (recruits per spawner = 1).

⁹/ The effect of spawner density on the value of recruits per spawner is especially evident in populations that are subjected to supplementation actions, whereby the number of naturally spawning fish is boosted by hatchery intervention.

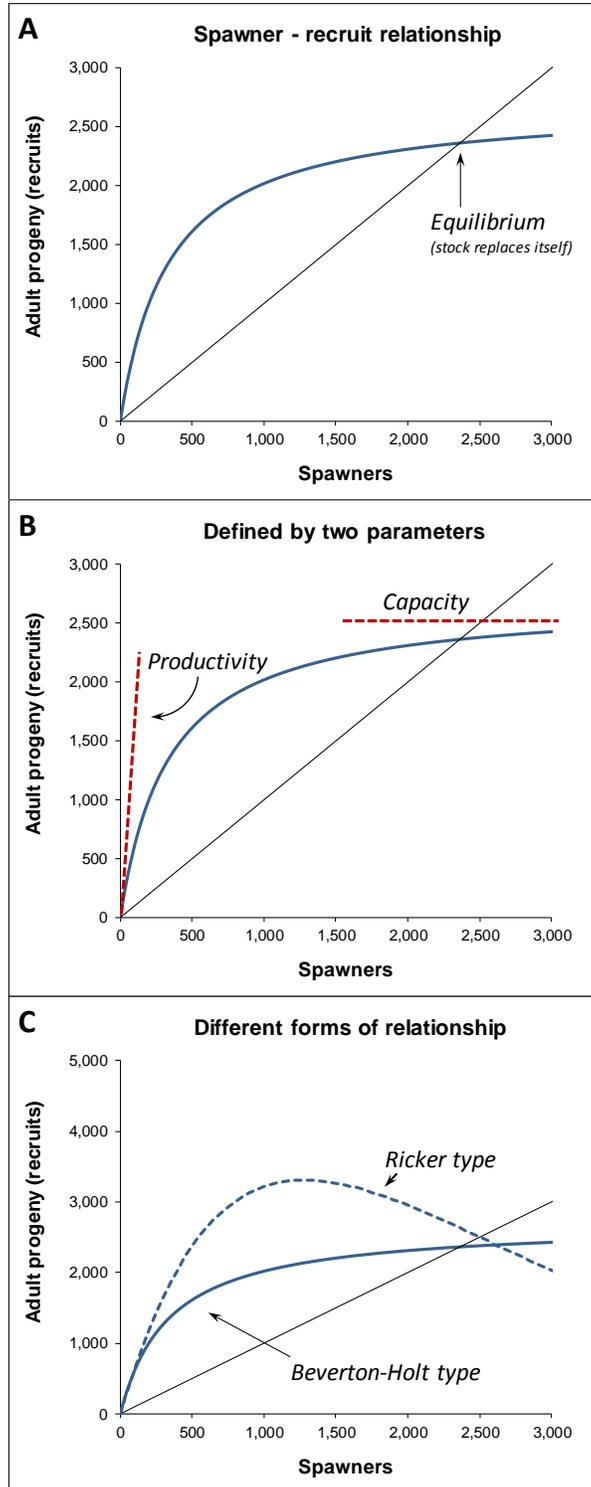


Figure 12. A - Relationship between spawner abundance and adult progeny (recruits) (A) with equilibrium abundance identified. B – The relationship is defined by two parameters: productivity and capacity. C – Two common forms of a spawner-recruit relationship, the Beverton-Holt, which rises to an asymptotic capacity, and the Ricker, which is dome-shaped.

The equilibrium abundance would be what we would tend to observe on the average over some period of years, if habitat conditions remain relatively constant and there is no harvest (Ricker 1975).¹⁰ Equilibrium abundance can also be determined for different fixed rates of harvest (Ricker 1975). At equilibrium abundance, a salmon population has maximized, on average, its use of the available habitat conditions to the extent it can, given all of the mortality pressures acting on the population. The equilibrium abundance in this sense provides a useful way to compare population performance under different scenarios.

The S-R relationship is normally defined by the two parameters capacity and productivity (Figure 12B) (Hilborn and Walters 1992). Equilibrium abundance, therefore, can be defined by these two parameters (Ricker 1975). Capacity, in the conventional population dynamics sense, regulates potential abundance, since the environment has a finite amount of habitat and food that can be utilized by the population. As a population grows, competition for resources among individuals increases, ultimately placing a limit on how large the population can grow. Intrinsic productivity, in contrast, defines performance at very low abundance, when competition for resources is assumed to be negligible. Defined exactly, it is the theoretical maximum number of recruits that would be produced per spawner (on average) in the absence of any competition, density-dependence, or depensation.¹¹ The productivity value for the population is given by the slope of the S-R relationship at (or very near) the curve's origin on the x-y axis seen in Figure 12B. The capacity value in the same figure is the asymptote of the curve in the upper right.

The two performance parameters—capacity and productivity—are determined both by biological and habitat-related factors. Biological characteristics of a population include life history traits, genetic diversity and fitness, fecundity, and sex ratio, among others, all of which are affected by and operate within the template of habitat (Southwood 1977; Begon and Mortimer 1986). We focus here on how habitat characteristics affect the parameters because recovery decisions regarding habitat restoration and protection are a primary means for closing the gaps between current performance levels and recovery goals for summer chum. We note, however, that reintroductions to restore extirpated subpopulations would add capacity, as well as increasing the potential for a portfolio effect over a wide range of environmental conditions.

The two basic characteristics of habitat that relate to these two performance parameters are habitat quality and habitat quantity. Productivity is determined by the quality of the habitat (i.e., by characteristics that are not competed for by members of the population). Capacity is determined by the combination of both the quantity and quality of the habitat (Moussalli and Hilborn 1986; Moberg et al. 1997). Changes in either the quality or quantity of habitat in which the population is produced will alter the shape of the S-R relationship, either causing a decline in performance (as would occur by habitat

¹⁰ / This average under such conditions would be the geometric mean of observed run sizes. The arithmetic mean would almost always be larger, perhaps substantially, because the distribution of recruits at given levels of spawners is usually lognormal (Ricker 1975). A lognormal distribution will occasionally show very large recruitment, having a long tail toward the upward end. Also, the amount of variation at a given level of spawners will be proportional to the average recruitment, so we expect to see lower variability at small recruitments and higher variability at large recruitments (Hilborn and Walters 1992).

¹¹ / The description of the S-R relationship given here does not incorporate the concept of depensation, which is believed to occur at extremely low population densities such that recruits produced per spawner can actually decrease (e.g., Liermann and Hilborn 2001). Depensation might occur, for example, as a result of spawners being less successful at finding mates at extremely low abundance, thereby reducing reproductive success per available spawner. Depensation is taken into account in our dynamic modeling for this paper, see methods below.

degradation) or an improvement in performance (as would occur by habitat restoration). Alterations in the underlying S-R relationship result in a change to equilibrium abundance of the population over some period of years. We would note here that climate change can affect both habitat quality and quantity, which could then alter the S-R relationship and associated equilibrium abundance.

Different forms of the S-R relationship are recognized, such as the Beverton-Holt, characterized by the number of recruits approaching an asymptotic limit at high spawner abundance, and the Ricker, which is dome-shaped over a range of spawner abundances (Figure 12C). A third form sometimes applied in recovery planning is a rectilinear form called the hockey stick, since it is shaped like a hockey stick (straight rising limb on the left to a sharp break when the capacity is reached). The hockey stick is essentially a variant of the Beverton-Holt but without the curved rising limb. These different forms are mentioned later in this paper.

Another aspect of performance that affects viability is variation in production that occurs as a result of variability in natural processes between years. Relatively wide variation in production is typically seen in spawner and recruit abundance data sets (Figure 13). Whereas the underlying S-R relationship is expressed by a deterministic function, the actual number of recruits produced at any level of spawners reflects many interacting stochastic environmental and biological effects. The result is that empirical data can have considerable scatter around any underlying S-R relationship. The amount of variation in performance around the underlying production relationship has a critical role in the viability assessment (McElhany et al. 2000; ICTRT 2007). Any quantitative assessment of viability needs to incorporate this type of variation in performance (Morris and Doak 2002).

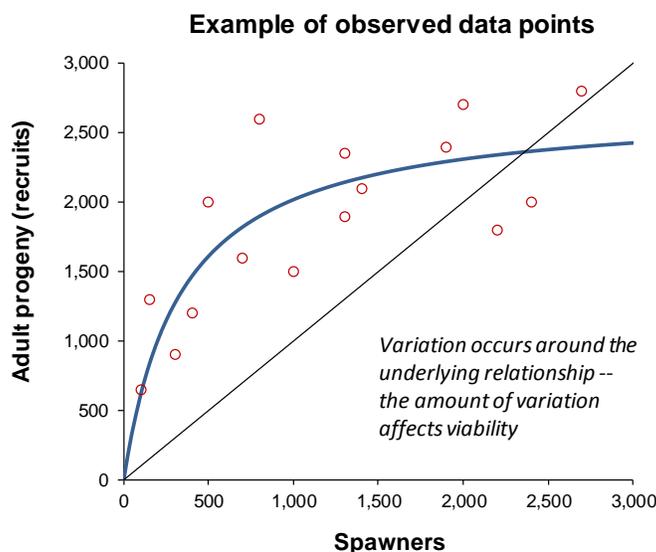


Figure 13. Hypothetical example of variation in recruit production around the underlying spawner-recruit relationship.

The capacity and productivity parameters that define the S-R relationship are linked to each other relative to extinction risks associated with the amount of short-term environmental variation (ICTRT 2007). Modeling exercises show that at a particular productivity level, populations with higher levels of capacity are more resilient in the face of year to year variability than those with smaller capacities. In contrast, populations with relatively high productivities are more robust at a given level of capacity

relative to populations with lower productivity. These patterns have served as the basis for defining “viability curves” as a metric for evaluating performance in relation to capacity and productivity (e.g., LCWTRT 2003; ICTRT 2007; Sands et al. 2009).

It is important to clarify here the relation between capacity and equilibrium abundance associated with the S-R relationship. While they each mean something different with regard to the S-R curve, they are perfectly correlated ($r^2 = 1.0$) for any given productivity value > 1 ; the slope of the correlation line is set by the productivity value (Figure 14). This is true for both the Beverton-Holt and Ricker forms of the S-R curve. At any given value of productivity > 1 , an increase in the capacity value directly increases the equilibrium abundance value. The relevance of this is seen below with respect to viability curves.

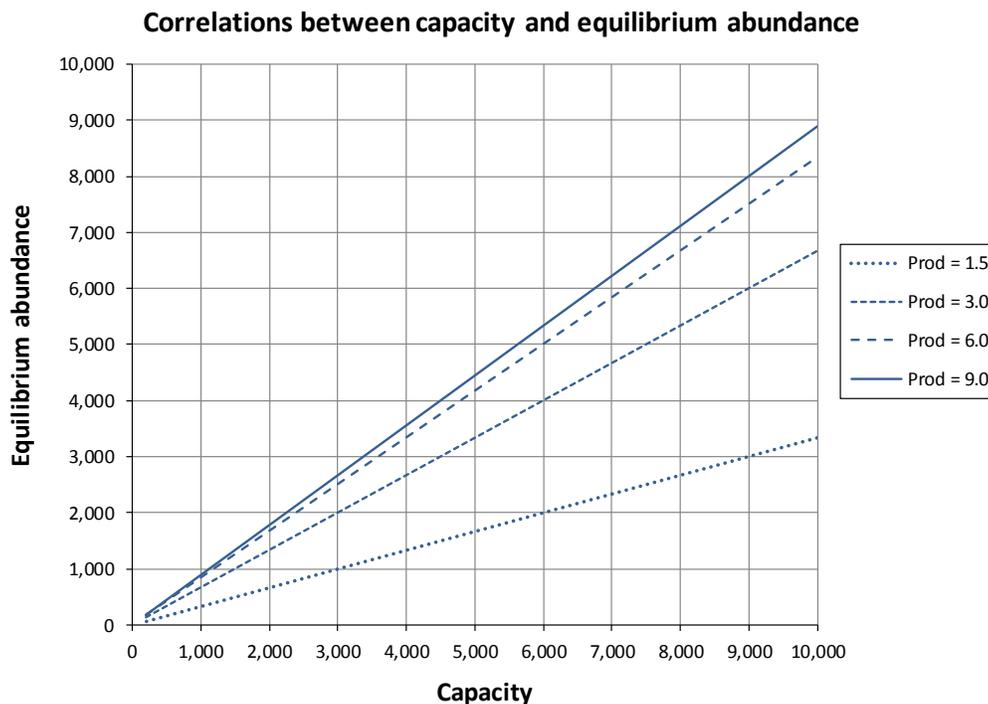


Figure 14. Correlation lines between capacity and equilibrium abundance at different levels of intrinsic productivity (Prod) with the Beverton-Holt form of the S-R curve.

An example of a viability curve is seen in Figure 15. Viability curves are generated using quantitative modeling incorporating performance characteristics associated with the population of interest. Figure 16A-B portrays two different population conditions, one in which the population would be expected to have a low risk of extinction (Figure 16A) and one with a high risk of extinction (Figure 16B). A data point plotted in each graph represents a population characterized by its capacity and productivity parameters, which provides the basis for plotting performance relative to the viability curve.

The viability curves can be shown just as easily with equilibrium abundance plotted instead of capacity (converting the y-axis to equilibrium abundance), as done by in LCWTRT (2003). Sands et al. (2009)

expressed their viability curves using capacity.¹² We retain the use of capacity herein to define viability curves to remain consistent with Sands et al. (2009), though we do employ the use of equilibrium abundance values for one portion of the presentation where we think it is more appropriate.

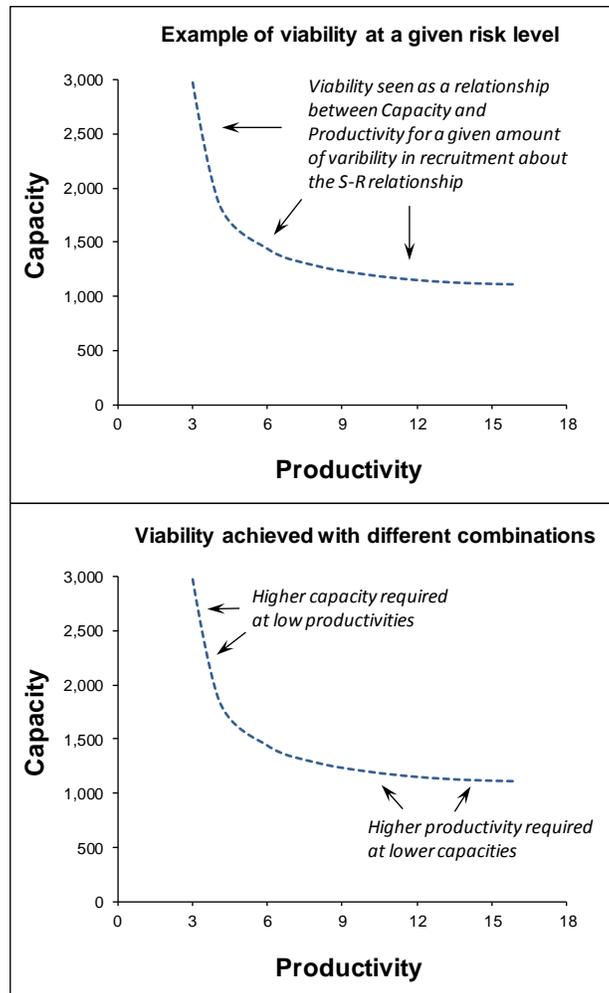


Figure 15. Example of a viability curve showing a relationship between capacity and productivity that defines a performance threshold for a salmon population associated with given level of extinction risk. At relatively low productivities, a higher capacity is required to maintain low extinction risk, whereas at higher productivities a lower capacity is required for the same level of extinction risk.

¹² / Note that Figures 15 and 16 in Sands et al. (2009) label the y-axis as abundance but the figure captions clarify that capacity is actually being quantified. Also, Tables 7 and 8 in Sands et al. (2009) clarify that capacity is being used to define the viability curves.

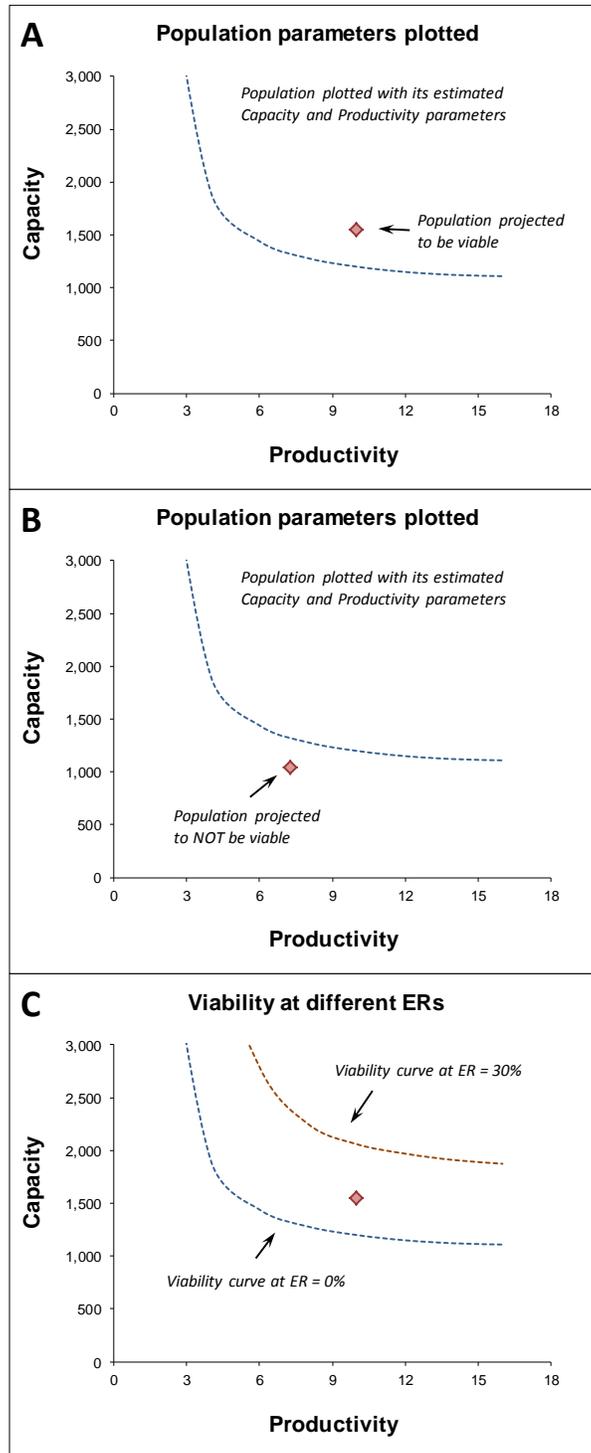


Figure 16. Hypothetical population performance plotted in relation to viability curves. A – The population is at low risk of extinction relative to the viability curve. B – The population is at high risk relative to the viability curve. C – The population is at low risk relative to the viability curve with an exploitation rate (ER) of 0 percent but at high risk with an exploitation rate of 30 percent.

It is important to recognize that viability curves can be derived to show viability at a given level of risk of extinction under different fishery exploitation rates. Figure 16C shows that the population represented in the figure would have a negligible risk of extinction with an exploitation rate of 0 percent, whereas it would be at high risk of extinction under a harvest regime with a 30 percent exploitation rate. Thus the population shown in the figure could be expected to have a low risk of extinction with relatively low exploitation rates (i.e., with ERs less than about 15 percent).

Viability curves are derived using quantitative modeling to perform a population viability analysis (PVA). Various approaches exist to perform a PVA, which can vary widely in detail and quantification (Morris and Doak 2002). All PVA applications include some way of assessing the risk of reaching a specified threshold in performance over time.

4.2 Previous Assessment

Sands et al. (2009) used two different quantitative PVA approaches to assess viability thresholds for the two populations belonging to the Hood Canal ESU. One approach employed a density-independent model, assuming that the population time series approximates a Brownian motion (Dennis et al. 1991). Under this model, there is no underlying relationship between spawners and recruits as seen in Figure 12; production is assumed in this case to be driven entirely by random processes. The computer program SimSalmon was used to model this approach.

The second approach assumes that some form of a density-dependent underlying relationship exists between spawners and recruits, such as seen in Figure 12. For this approach, the Viability and Risk Assessment Procedure (VRAP) was employed (Sands 2009; Sands, in prep).

The assessment was made using spawner and adult recruit data for brood years 1974 to 2001 for both the SJDF and Hood Canal populations. No attempt was made to quantify viability thresholds for individual subpopulations because recovery as evaluated by the abundance criterion is to only be determined at the population level.

Sands et al. (2009) presented numeric recovery goals for abundance (using capacity) and productivity with both modeling approaches. They did not recommend one approach over the other, suggesting that additional data was needed to arrive at a conclusion about the most appropriate type of assessment. An abbreviated summary of results is shown in Table 3 for both populations under each modeling approach. The results using the VRAP model are given as a range in capacity (incorporating a reasonable range of productivities) and a range in expected spawning escapement associated with a specific pair of capacity and productivity values.

It is important to note that VRAP identifies viability as a spawner-recruit function defined by productivity and capacity parameters. For example, the capacity threshold for the SJDF population of 3,300 in Table 3 corresponds to a productivity of 6. If the productivity were only 3, then the larger capacity of 4,300 is needed to be viable. For the case of productivity (P) = 6, capacity = 3,300, and zero harvest, the equilibrium abundance recruitment would be 2,750 fish while the expected observed escapement, given the variability of the population, is 4,500 fish. If the model were entirely deterministic, this would also be the associated expected spawning escapement (with no harvest). However, the model incorporates lognormal variability, which as noted earlier will skew average recruitment high. The escapements presented in Table 2 are the arithmetic average (not the geometric mean, see footnote 10) of the last year of a 100 year model run over 3,000 repetitions; hence the average spawning escapements for a

capacity and productivity pair are shown to be larger than even the capacity values.¹³ The viability target is not the escapement, but it is the combination of the productivity and capacity parameters. When the population reaches that viability condition, one would expect to see escapements averaging the given corresponding escapement levels in Table 3 (arithmetic mean). The importance of this point will be seen in subsequent sections of this paper.

As one adds harvest to the situation, the capacity for a given productivity level needs to increase to maintain viability. Using the SJDF example as in the paragraph above, at P = 6, capacity needs to be 3,700 to maintain viability with an exploitation rate of 10 percent. While the expected resulting equilibrium escapement computed deterministically would remain approximately the same (it drops slightly) as in the case without any harvest, the arithmetic mean modeled using VRAP with variation included would drop slightly to 4,600.

Table 3. Minimum abundance viability thresholds for the SJDF and Hood Canal populations of summer chum as given in Sands et al. (2009) derived with two modeling approaches. The density-independent model (SimSalmon) did not explicitly incorporate exploitation rate (ER), whereas an ER was incorporated explicitly in the density-dependent model (VRAP). The results from VRAP are shown as a range, based on different values for productivity that bracket a reasonable range of values for each population.¹⁴

Population	Model	ER	Escapement range		Capacity range	
			Low	High	Low	High
SJDF	Density-independent	0%	5,600			
			P=6	P=3	P=6	P=3
	Density-dependent	0%	4,700	5,100	3,300	4,300
		10%	4,600	5,400	3,700	5,300
Hood Canal	Density-independent	0%	24,700			
			P=9	P=5	P=9	P=5
	Density-dependent	0%	17,900	20,600	13,000	17,000
		10%	18,600	21,500	15,500	20,500

4.3 Methods for Updating the Assessment

We updated the analysis presented in Sands et al. (2009) by using the most up-to-date data available on spawner and adult recruitment abundance. This data set encompassed brood years 1974 to 2006, giving us five more brood years (2002-2006) than used in the earlier analysis. Also, some of the escapement data and harvest data from 2001 and later had been updated, as well as some older data. Age data were also revised for all years and some earlier data became available for the SJDF population. See Sands (2007, 2013) for details on estimating age structure and related data elements used in the analysis.

¹³ / This point may be confusing to the reader not familiar with the modeling concepts described here. VRAP as it is configured incorporates lognormal variability but only reports the arithmetic mean of expected spawning escapements. The model should report the results using geometric mean; see also footnote 10.

¹⁴ / For SimSalmon modeling, only results are shown with model parameterization as recommended by the PSTRT; a much larger set of results under different conditions were presented in Sands et al. (2009). The range of intrinsic productivity was 3 to 6 for SJDF and 5 to 9 for Hood Canal.

Data for natural spawning escapement estimates, broodstock take, and catch per fishing area were provided by the co-managers (see WDFW and PNPTT 2000 for escapement estimation methods). Total recruitment includes, for the purposes of these analyses, the number of adult summer chum caught in Canadian and U.S. fisheries, fish that spawn naturally, and fish removed for broodstock take. Observed predation on adult chum by marine mammals was not included since observations were sporadic. Recruitment is calculated from run reconstruction analyses after the method used by the co-managers (see WDFW and PNPTT 2000 for run reconstruction methods).

Some of the naturally spawning summer chum in some streams and years have been returning hatchery-origin fish (resulting from supplementation actions). The supplementation program has always employed native-run fish returning to each of the target streams for broodstock (except for initiation of reintroduction programs, where an adjacent subpopulation was used). Naturally spawning hatchery-origin fish were included as part of the natural spawner escapement in our assessment. However, hatchery-origin spawners were not counted as recruits, so only naturally produced recruitment was included for each brood year. It bears noting that Small et al. (2009) found no effect of the supplementation program on the genetic structure of the populations and their subpopulations. Small et al. (2013) reported no change in the genetic diversity of wild-born fish in the supplemented subpopulations. Also, Berejikian et al. (2009) found no significant differences in reproductive success between summer chum supplementation-origin natural spawners and natural-origin natural spawners.

Our updated assessment employed only VRAP for the viability analysis, since it is clear that the populations exhibit obvious patterns of density-dependence, as seen in Figure 17. The figure plots natural-origin recruits per spawner against spawner abundance for each brood year for all of the data for each population. The obvious curvilinear patterns seen for each population demonstrate very strong density effects on recruitment. Use of a density-independent model for a viability assessment is inappropriate in this case (Morris and Doak 2002).

We applied the same procedures in using VRAP as described in Sands et al. (2009). We provide a brief description here to aid the reader; refer to the earlier document for a more detailed description.

VRAP is a stochastic simulation model that projects recruits and spawners over a period of years based on a S-R relationship, a given starting population size, and a target exploitation rate. In VRAP, harvest mortality may be estimated by age for two fishery types: mixed maturity stock fishery and mature stock fishery. VRAP is also used for doing viability analysis of Chinook, which are subject to fisheries that harvest both immature and mature fish. For summer chum, all harvest is assumed to be taken on fish that are maturing in a given year and that are returning to the spawning grounds. This simplifies the use of VRAP for summer chum compared to how the model must be operated for analyzing Chinook populations.

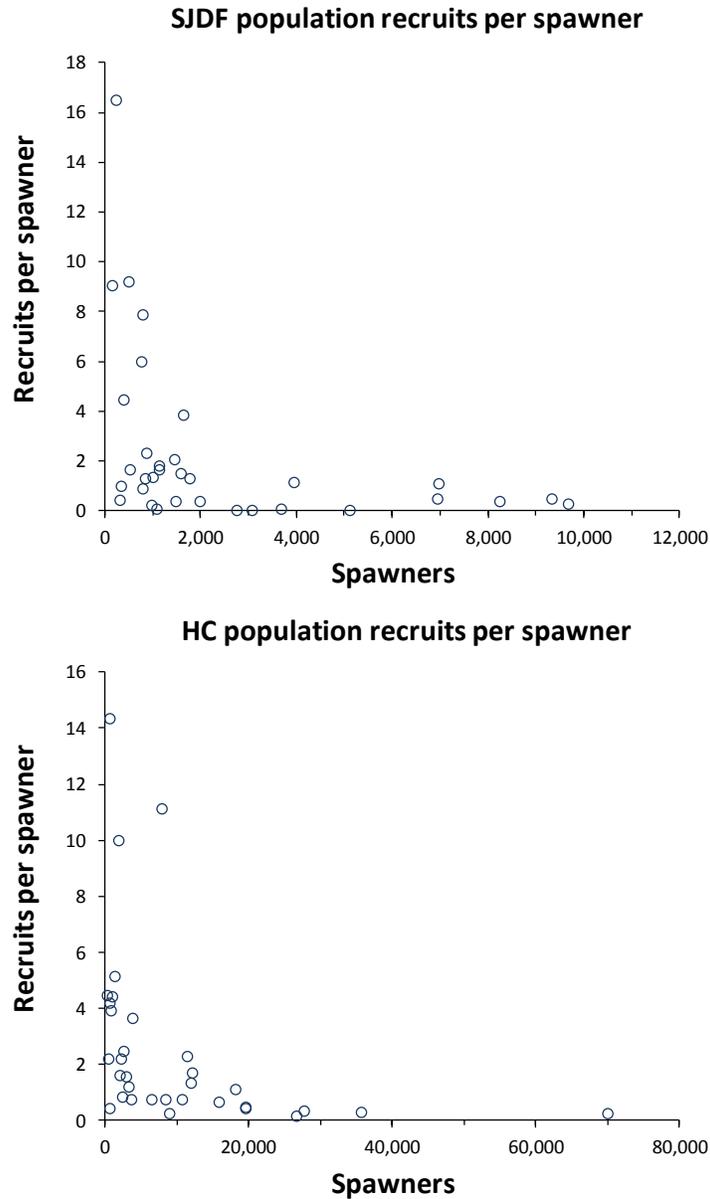


Figure 17. Plots of natural-origin recruits per spawner versus spawner abundance for SJDF and Hood Canal summer chum, brood years 1974 to 2006.

The model is operated by assuming a Beverton-Holt, a Ricker, or a hockey-stick type S-R function. Each type was employed in our assessment enabling us to compare results. Prior to running VRAP, the population data were analyzed with another model—called the Dynamic Model—to assess the best-fit S-R relationship for the data set (see also Sands et al. 2009). This step also produced the estimate of variability to be used in VRAP, where variability is expressed by the difference between the observed recruits and predicted recruits. The variability is assumed to have a gamma distribution and the two gamma parameters are estimated from the error data and are used as input to VRAP. The error can also

be described as the coefficient of variation (CV)¹⁵, which is easier to understand in comparing variability between populations and subpopulations. The Dynamic Model differs somewhat in how it fits the S-R relationship compared to most conventional approaches to fitting stock and recruit data by minimizing the error term for spawning escapement instead of for recruitment. This procedure was developed because it is believed that greater error exists in the estimates of recruits than for spawners, since the error for recruits includes the error inherent in escapement plus the error introduced from catch estimates and age estimates. It is noted, however, that we compared estimates of capacity and productivity for each population and subpopulation derived using various methods for purposes of plotting performance parameters on graphs with viability curves.

In using VRAP to determine the S-R curve, i.e., intrinsic productivity and capacity, that allows for <5% rate of extinction over 100 years, we use the error parameters determined by the Dynamic Model plus the maturation rates and fishing rates by age used by the Dynamic Model (estimated using cohort run reconstruction). The VRAP model is run for a series of productivity values, each model run determining the capacity needed for that productivity for the population to be viable at a level of risk and for a given exploitation rate. We ran the model for exploitation rates of 0, 10, 20, and 30 percent for this paper.

Another input parameter important for determining viability is the quasi-extinction threshold (QET), defined as "...the minimum number of individuals below which the population is likely to be critically and immediately imperiled." (Ginsburg et al. 1982; Morris and Doak 2002). The population QETs were taken from Sands et al. (2009) and were 300 fish for the SJDF and 350 fish for Hood Canal. For individual spawning aggregations we used a QET of 63 fish and we used a QET of 126 fish for the Quilcene and Salmon/Snow subpopulations, which have two spawning aggregations each. For determining returns, for spawning at or less than the QET the return was zero. Since most summer chum return at age 3 or 4, when spawning is less than the QET four years in a row, extinction is assumed to occur. This method introduces depensation into the modeling.

The output from VRAP specifies the minimum capacity values needed to keep the risk of extinction to 5 percent or less at each productivity value over a range of productivities relevant to the analysis. These results are then easily plotted to form the viability curves as illustrated earlier in Figure 14. VRAP output also provides for each combination of capacity, productivity, and exploitation rate values of the associated spawning escapement level that would be expected. This spawner escapement level is the average escapement that would result from the specified exploitation rate associated with a single pair of capacity and productivity values. But as noted at the end of Section 4.2, this calculated average escapement is substantially higher than the equilibrium value on the S-R curve for the reasons described earlier.

We also applied the VRAP viability analysis to each of the eight extant subpopulations (two in SJDF and six in Hood Canal) for brood years 1974-2006. We found, however, that in using the Dynamic Model for each of the subpopulation data sets that the patterns of variation (represented by CVs) differed to an extent between subpopulations that we lacked confidence in applying them. Tom Cooney, a NOAA Fisheries scientist who is a member of the Interior Columbia TRT, advised us that it would be more reasonable to apply the overall population CV to each of the subpopulations. He had seen similar differences in CVs among spawning aggregations for other populations that could not be resolved with

¹⁵ / Coefficient of variation = 1/square-root (gamma a), where gamma a is the first parameter of the gamma distribution.

existing knowledge. The pattern of CV for the overall population should be sufficiently applicable to each of the major spawning aggregations of a population to serve our purpose.

4.4 Results of the Updated Assessment

Analysis of populations and subpopulations using the Dynamic Model generally resulted in best fit S-R relationships using the Beverton-Holt function. Sands et al. (2009) also applied the Beverton-Holt form in that earlier analysis. We applied this form of the S-R relationship for all data sets for the sake of consistency between populations and subpopulations in producing our viability results. Also, because several data sets for subpopulations were especially difficult to fit with a S-R relationship, we used multiple approaches and compared results to derive what we considered to be the most likely S-R parameters, which we then used in plotting with viability curves to assess performance against a viability threshold. Generally, we found that it was more difficult to estimate productivity for some data sets than to derive what we believe are reasonable estimates of capacity. The Beverton-Holt function appeared to provide the most robust means of estimating capacity. It bears noting that the shape of the viability curves, which tend to flatten out at relatively high productivities, suggest that estimation of capacity may generally be more critical than for productivity.

A comparison of the estimates of process error variation (CVs) between the earlier assessment and this one is given in Table 4. While the CV increased modestly from the earlier assessment for the SJDF population (from 107% to 111%), it declined by a relatively large amount for the Hood Canal population (from 134% to 120%). These changes directly affect the resulting viability thresholds derived with VRAP. (We would note that near the end of the preparation of this report, we recomputed CVs again after including preliminary data for brood years 2007 to 2009 combined with all of the earlier data. These new computations also included all of the newest updates (though preliminary) in age data. For SJDF, CVs for brood year periods 1974 to 2001, 1974 to 2006, and 1974 to 2009 were computed to be 100 percent, 112 percent, and 114 percent, respectively. For Hood Canal, CVs for the same three brood year periods were computed to be 136 percent, 115 percent, and 110 percent, respectively.)

Table 4. The coefficient of variation (CV) related to process error for the SJDF and Hood Canal populations of summer chum for the analysis based on 1974-2005 data (brood years 1974-2001); Sands et al. 2009) and the analysis based on 1974-2010 data (brood years 1974-2006; 2013 update).

Population	Assessment	BY	CV
SJDF	Sands et al. 2009	74-01	107%
	2013 update	74-06	111%
Hood Canal	Sands et al. 2009	74-01	134%
	2013 update	74-06	120%

The changes in CVs between the analysis reported in Sands et al. (2009) and this one directly affect the viability thresholds derived from VRAP (Table 5). The range of productivities used in obtaining these results is similar between the two assessments. The updated assessment produces viability thresholds for the SJDF population moderately higher (i.e., by approximately 20 percent) than those given in Sands et al. (2009). A larger CV (greater variation) produced an increase in the viability threshold capacity

value. For the Hood Canal population, the thresholds were lowered by a significant amount—by approximately 50 percent of those reported earlier—due to a substantially reduced CV.

Table 5. Minimum abundance viability thresholds for the SJDF and Hood Canal populations of summer chum as given in Sands et al. (2009) derived using the VRAP model and as updated in the current analysis (2013 update). ER is exploitation rate and P is intrinsic productivity. Escapement values are arithmetic means¹⁶ as in Sands et al. (2009).

Population	ER	Assessment	Escapement range		Capacity range	
			Low	High	Low	High
SJDF	0%	Sands et al. 2009	P=6 4,700	P=4 4,800	P=6 3,300	P=4 3,700
		2013 update	5,700	6,200	5,100	6,300
	10%	Sands et al. 2009	4,600	5,100	3,700	4,500
		2013 update	5,600	6,100	5,800	7,100
Hood Canal	0%	Sands et al. 2009	P=8 18,300	P=6 19,100	P=8 13,500	P=6 15,000
		2013 update	8,700	9,100	7,000	7,800
	10%	Sands et al. 2009	18,300	20,400	15,500	18,500
		2013 update	8,700	9,600	8,000	9,300

The changes in the viability thresholds are due mainly to longer data sets used in the analysis, which produced more precise estimates of CV. This is most evident for the Hood Canal population. For the analysis reported in Sands et al. (2009), high variability in the data set (CV=134%, Table 4) for this population was largely due to the extremely high return from the 2000 brood year (3.5 times as high as the next highest return). The new data added to the data set for the 2013 update were within the usual range of data. A sensitivity test on the combined data set for this population was made by reducing the 2000 recruits to a more normal return and calculating the CVs for both time periods; there was only a very slight drop in the CV levels from those shown in Table 4.

Viability curves for the two populations using the updated assessment, shown with exploitation rates of 0 and 30 percent, are provided in Figure 18. Our estimates of productivity and capacity for the populations using all data for brood years 1974 to 2006 are also shown plotted. These results signal that the Hood Canal population would be considered to be at negligible risk of extinction with current biological performance, provided that the exploitation rate is held to a very low level. In contrast, the analysis signals that the SJDF population has a much higher risk of extinction compared to the Hood Canal population when seen over the entirety of the data series, even with the exploitation rate set to 0 percent. For return years 2000 through 2012, exploitation rates average about 7.6 percent for Hood Canal and about 0.6 percent for SJDF (PNPTT and WDFW 2014).

Viability curves (5 percent risk) for the eight extant subpopulations, shown with exploitation rates of 0 and 30 percent, are provided in Figure 19. Estimates of productivity and capacity for the subpopulations using all data for brood years 1974 to 2006 are also plotted on the graphs. The results suggest that all of

¹⁶/ The arithmetic mean is skewed high (by approximately 35 to 40 percent) due to the lognormal distribution of observed escapements compared to the geometric mean, which is equivalent to what this paper refers to as equilibrium abundance. This paper will recommend that geometric mean be used as a measure of whether viability thresholds have been achieved.

the subpopulations are at relatively high risk of extinction when considering all of the years in the data series together, except for the Dosewallips and Quilcene subpopulations. Results for the Dosewallips subpopulation suggest that it is very close to being viable at the 5 percent risk threshold without any harvest. In contrast to all of the other subpopulations, performance of the Quilcene subpopulation is much higher than its viability thresholds, even if a relatively high harvest regime (e.g., ER >30%) was to be in place. These results suggest that starkly different performance characteristics exist among the subpopulations. It is also clear that the reason why performance for the Hood Canal population, measured by aggregate abundance, was shown to exceed the viability threshold is due largely to the Quilcene subpopulation, and secondarily to the Dosewallips subpopulation. These two streams, despite the extent of watershed development and management, support relatively high capacities compared to other streams in the ESU; they appear to be the strongest subpopulations (see Section 5.2.3) and may generally act as core subpopulations to the Hood Canal population.

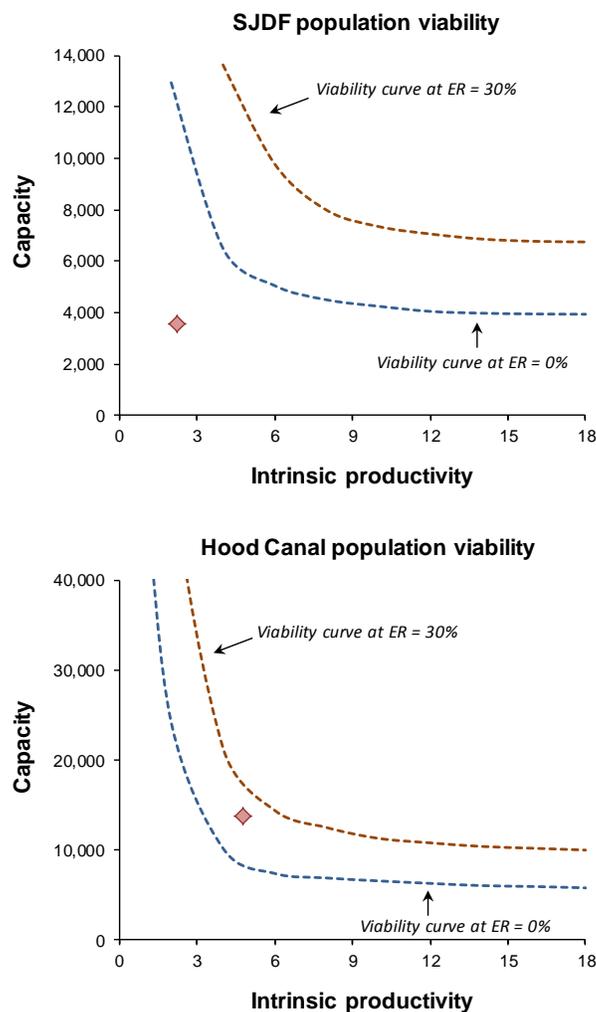


Figure 18. Updated viability curves with a 5 percent extinction risk over 100 years for the Hood Canal and SJDF summer chum populations with associated exploitation rates of 0 and 30%. Estimates of intrinsic productivity and capacity for each population are plotted for brood years 1974 to 2006.

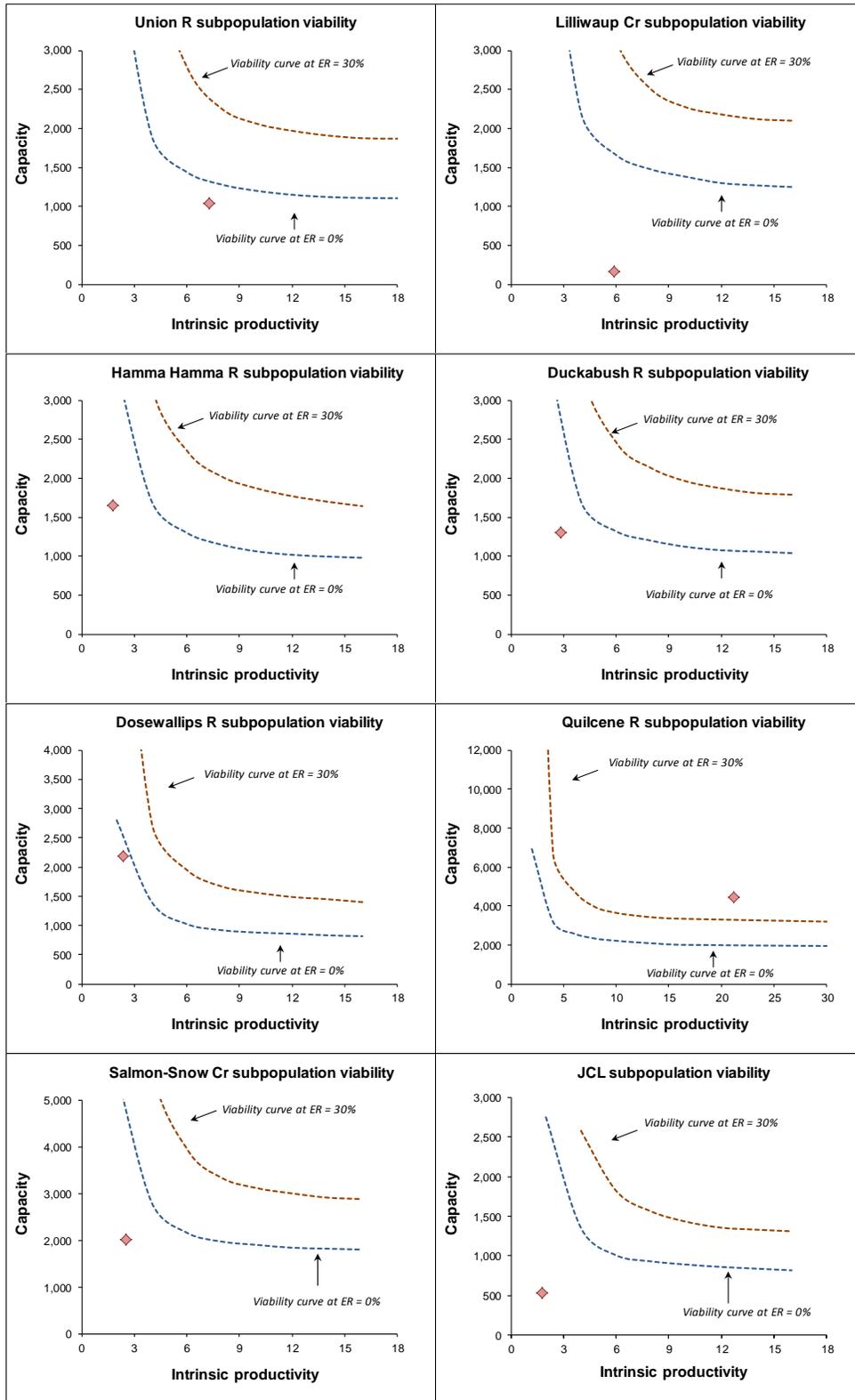


Figure 19. Viability curves with a 5 percent extinction risk over 100 years for the extant subpopulations belonging to the Hood Canal and SJDF summer chum populations with associated exploitation rates of 0 and 30% Estimates of intrinsic productivity and capacity for each subpopulation are plotted for brood years 1974 to 2006.

5.0 Shifts in Climate and Ocean Regimes

This section examines the potential role of shifts in decadal-scale climate and ocean regimes to summer chum performance and considers the implications of such shifts to recovery. The SCSCI scientists (WDFW and PNPTC 2000) recognized that regime shifts might have contributed to the decline of summer chum beginning in the late 1970s. Based mainly on how fall chum appeared to perform during the 1980s and 1990s, they suggested that it was unlikely that ocean conditions were a significant contributor to the decline. However, they concluded that a contributing role of the ocean in abundance patterns could not be ruled out. It bears noting that the SCSCI report was being prepared at the time of a significant change that was occurring in ocean conditions and the authors had no way of knowing what those changes might bring.

By the mid-2000s, it had become evident that a shift in climate/ocean regimes had occurred. Beamish et al. (2004b), as well as others, presented evidence for a shift in the late 1990s that may have affected the performance of at least some chum and pink salmon populations in the Pacific Northwest. The assessment performed by the HCCC in preparation of the recovery plan using EDT modeling incorporated climate/ocean shifts as a significant factor affecting summer chum performance (Lestelle et al. 2005a and b).

5.1 Climate and Ocean Regimes

Conditions related to salmon survival within the Northeast Pacific Ocean and connecting marine waters are driven by two climate processes: the Pacific Decadal Oscillation (PDO) and the El Niño-Southern Oscillation (ENSO). Both ENSO and PDO are patterns of Pacific climate variability that include changes in air and sea temperatures, winds, and precipitation (Mantua et al. 1997; Mantua and Mote 2001). These conditions affect food webs related to the marine survival of salmon.

ENSO is Earth's dominant source of year-to-year climate variation (Rasmussen and Wallace 1983); it influences interannual variation in climate, ocean circulation, and sea surface temperature. In contrast, the PDO is a recurrent pattern of interdecadal climate variability characterized by persistent winter North Pacific atmospheric and oceanic circulation patterns (Mantua et al. 1997). The PDO has been described as a long-lived ENSO-like pattern of Pacific climate variability (Zhang et al. 1997, cited by Hare and Mantua 2001). The spatial patterns between the two are very similar: both favor anomalously warm sea surface temperatures near the equator and along the coast of North America, and anomalously cool sea surface temperatures in the central North Pacific.

Climatologists now know that climate patterns like the PDO can suddenly shift, resulting in abrupt changes in the characteristics of related natural phenomena, such as sea temperature, ocean currents, and biological processes (Hare and Mantua 2000). States in climate that persist on a decadal scale are called regimes and a sudden change to a new regime is referred to as a regime shift (Hare and Francis 1995; Hare and Mantua 2001).

The PDO is believed to create climate regimes that can last 20 to 40 years. In contrast, ENSO events are measured in months to several years. Figure 20 displays annual deviations from the long-term annual average PDO index for 1900 to 2012, derived from monthly sea surface temperatures (SST) in the North Pacific Ocean, poleward of 20°N, as reported by the Joint Institute for the Study of the Atmosphere and Ocean at the University of Washington (<http://jisao.washington.edu/pdo/PDO.latest>). Extended periods

when the annual deviations are greater than the overall average are referred to as a warm phase of the PDO; a cool phase is when deviations for extended periods are less than the overall average.

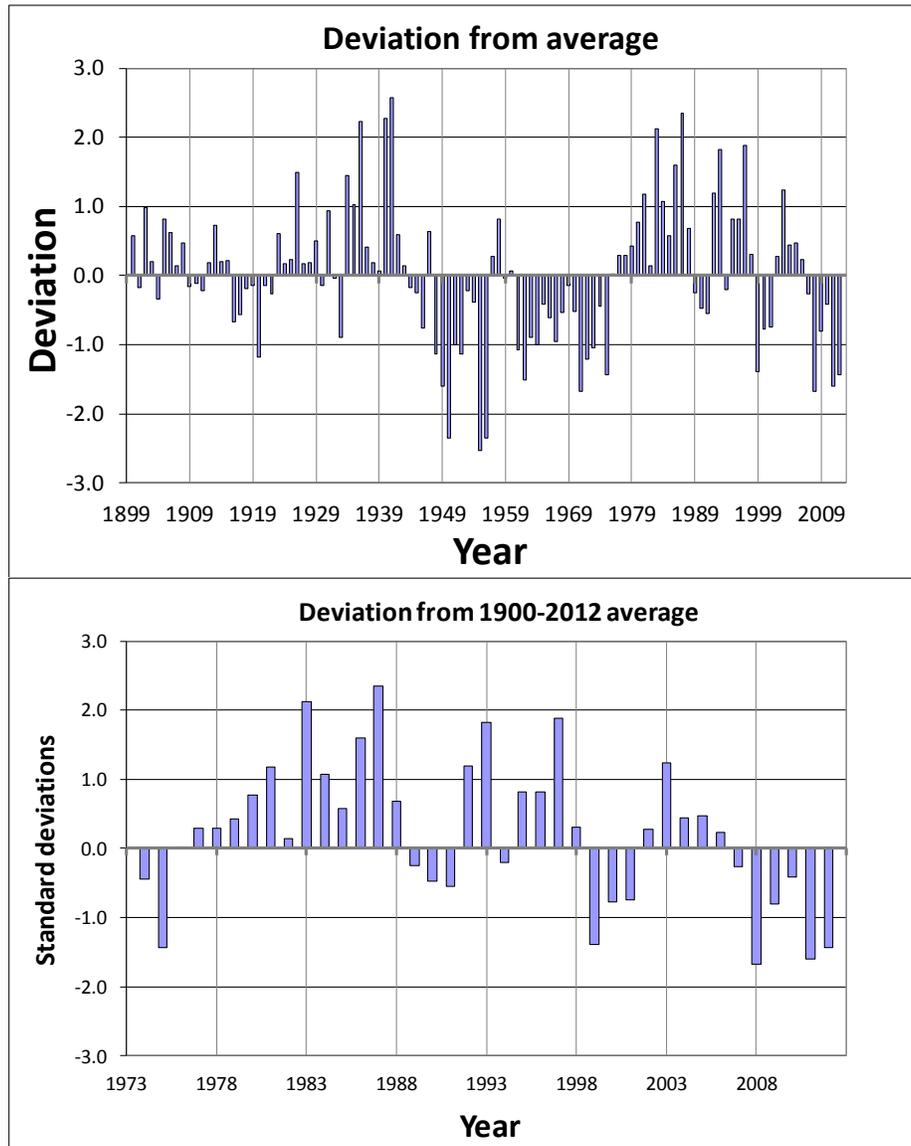


Figure 20. Annual deviations from the long-term annual average PDO index for 1900 to 2012, derived from monthly sea surface temperatures (SST) in the North Pacific Ocean, poleward of 20°N, as reported by the Joint Institute for the Study of the Atmosphere and Ocean at the University of Washington (<http://jisao.washington.edu/pdo/PDO.latest>). The bottom panel shows just the portion of the top panel relevant to the brood years analyzed in this paper.

The PDO is thought to have been in its cool phase from about 1890 to 1925 and from 1945 to 1977.¹⁷ It was in its warm phase from 1925 to 1945 and from 1977 to the late 1990s (Mantua and Mote 2001). The PDO shifted back to its cool phase beginning about 1998. The pattern of regimes is complicated by what

¹⁷ / Figure 20 is not clear about the PDO phase in the early 1900s. The reader should refer to original references cited in the text about why the regimes were delineated the way they were, such as by Mantua and Mote (2001).

appear to be mini-regimes or shifts associated with interacting climate oscillations. One of these oscillations appears to have a longer periodicity than the other, creating uncertainty about what the overall pattern and periodicity might be. For example, the 1977-97 regime is recognized by some to have been comprised of two regimes, the second beginning in 1989 (Hare and Mantua 2001; Beamish et al. 2000).

Beamish et al. (2000) examined some of the effects of the 1989 regime shift. They concluded that a regime shift does not need to be an oscillation or cyclic, but can simply be a change to a different state rather than a reversal of conditions. The 1989 shift was not a reversal but it was an abrupt change to another state.

Figure 20 suggests the 1998 shift was interrupted for several years in the mid-2000s when the index swung back to the warmer phase for several years before swinging back to the cool phase. The cool phase has remained intact since then.

The marine survival of salmon has been linked to these climate phenomena. Hare et al. (1999) identified an "inverse production regime" driven by the PDO, where the warm phase of the PDO is beneficial to Alaska stocks and detrimental to some Washington, Oregon, and California (WOC) stocks. British Columbia stocks have shown a mixed response. The cool phase of the PDO has the opposite effect on Alaska and some WOC stocks.

How a species responds to a PDO shift is not necessarily consistent within a region (Beamish et al. 2000)—this appears to be related to how conditions within different water bodies in a region are influenced by the shift. This is clearly seen with coho salmon produced in rivers of Washington State. Coho populations produced in Washington coastal streams (i.e., on the western side of the Olympic Peninsula) experienced unfavorable and reduced marine survival during the warm phase of the PDO beginning in 1977. In contrast, Puget Sound coho experienced favorable and improved conditions for marine survival at the same time. These opposite responses appear to be due to how conditions within Puget Sound are influenced differently by the PDO shift than those in the open ocean off Washington (Pinnix 1999; Pinnix and Francis undated). Beamish et al. (2000) reported that Strait of Georgia coho responded to the 1977 regime shift similarly to Washington coastal coho—hence biological responses to regime shifts can differ dramatically between Strait of Georgia and Puget Sound. However, the performance of populations from the three areas—Strait of Georgia, Puget Sound, and Washington coast—similarly turned down following the 1989 shift.

The effects of climate patterns within a large estuarine system like the Puget Sound-Georgia Basin appear to be strongly affected by estuarine circulation. The SJDF is the primary conduit for water exchange between the Pacific Ocean and the Puget Sound-Georgia Basin system. Flow through SJDF reflects river-influenced outflow at the surface and ocean-influenced inflow at depth (Thompson 1994). Variation in water properties within SJDF occurs as a result of either offshore oceanic variability (e.g., coastal ocean upwelling-downwelling) or to variability in river flows (Newton et al. 2003) or both. Climate events affect both. The effects of these factors across the Puget Sound-Georgia Basin system are variable and complex. Locations and sizes of rivers, tides, wind patterns, and bathymetric features like submerged sills and water depth create different suites of conditions that influence the responses of biological communities across the basin (Strickland 1983).

Beamish et al. (2004a) demonstrated that the climate regime shift in 1998 was linked to significant physical and biological changes observed in the Strait of Georgia that subsequently occurred. The shift

dramatically improved biological productivity in that area for four species of salmon. This improved productivity remains evident, indicating that the regime beginning in 1998 is still intact (Sweeting et al. 2004).

Estuarine circulation patterns in Puget Sound are more variable and biological conditions less directly affected by inflow from the SJDF than in the Strait of Georgia. This is due to a more complex arrangement of topographic features and proportionately less freshwater input for the sizes of the separate subbasins (based on data contained in Gustafson et al. 2000, Harrison et al. 1983, and Nelson undated). Figure 21 identifies the principal subbasins of the Puget Sound complex as delineated by Gustafson et al. (2000).

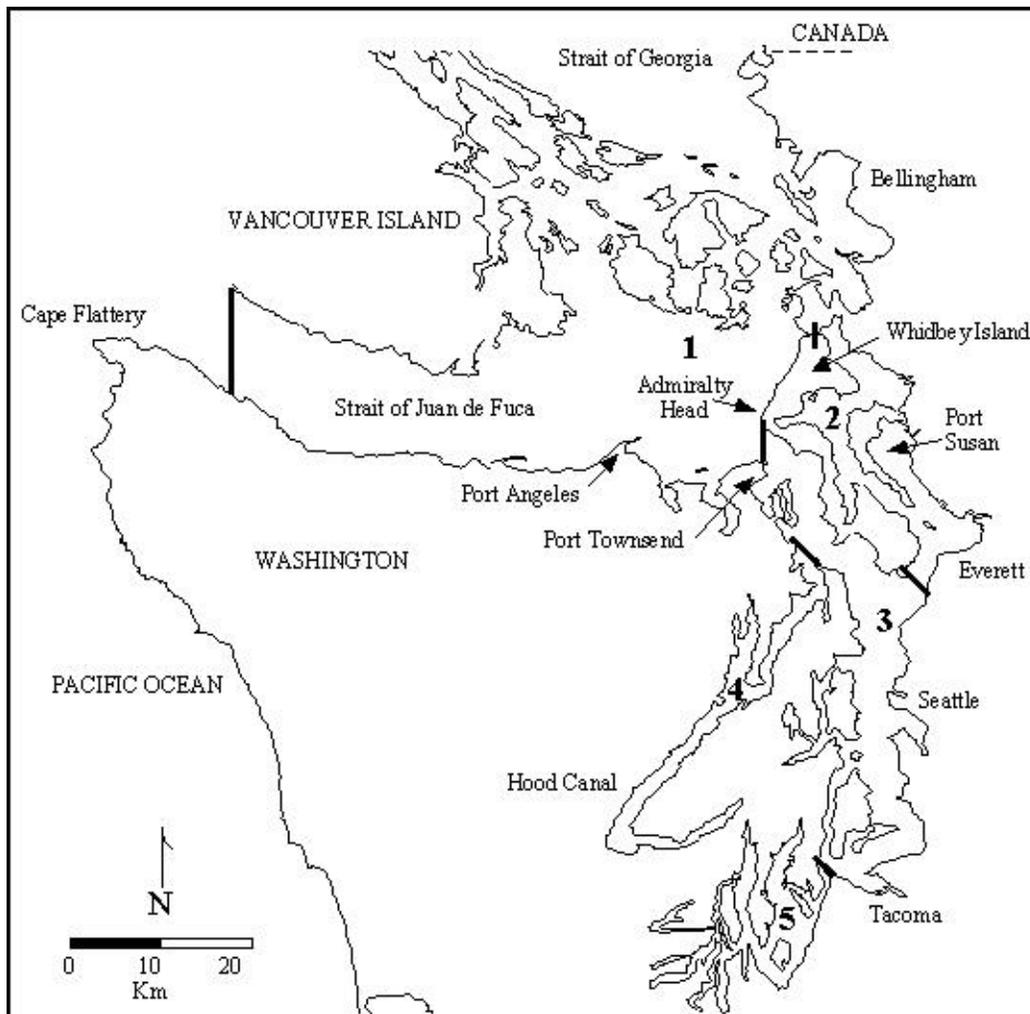


Figure 21. Puget Sound subbasins (called basins here): 1) Northern Basin, 2) Whidbey Basin, 3) Main Basin, 4) Hood Canal, and 5) Southern Basin. Dark bars show basin delineations. Taken from Gustafson et al. (2000).

Strickland (1983) and Nelson (undated) reported that relatively shallow sills at various locations in Puget Sound, including in Hood Canal, act to essentially limit the transfer of nutrient rich water farther into the Sound. As the deep inflowing layer of salty water upwells over the various sills that constrict flow at or near the entrances to most subbasins, it forces mixing with the surface outgoing layer of fresher water. The nature of this mixing, and the transfer of deep SJDF water farther landward, appears to be critical to

how climate regime shifts can affect local conditions within Puget Sound. The extent that a regime shift influences a subbasin should be related to its distance from Admiralty Inlet and the number of sills restricting SJDF inflow. This may explain why Puget Sound coho did not suffer a downturn in survival following the 1977 regime shift while Strait of Georgia coho did. The 1989 regime shift was associated with an even stronger influence across a much broader region and any buffering provided by Puget Sound was apparently overwhelmed with respect to coho performance.

Hood Canal branches off Admiralty Inlet just south of its second (most southern) sill. A relatively shallow sill (50 m deep), located near the north end of Hood Canal, constricts the passage of a deep salty layer at that point. Due to its topography, the water column in the Canal is highly stratified, except under strong wind forcing, with a shallow lens of fresh to brackish water at the surface overlaying waters of near-ocean salinity (citations given in Simenstad 2000). Water exchange is limited and residence time long, especially in the southern reaches of Hood Canal and in Dabob Bay. Nutrient rich water from the SJDF intrudes only in late summer.

The effect of these different circulation patterns within each subbasin on plankton production is varied. Strickland (1983) described it as “a game of ecological poker, in which each arm of the Sound is dealt a different hand from the same physiographic deck, with its biological behavior determined accordingly.” Circulation patterns—and associated mixing of water layers and flushing rates—within various areas of the Sound create widely different sets of conditions that influence plankton productivity. Hot spots for phytoplankton production, or particularly early blooms, can occur where certain unique features interact (Strickland 1983). Such changes are of a magnitude and duration that should affect the survival of a species like chum salmon whose performance has been shown to be strongly tied to the estuarine experience.

A group of Canadian scientists, seeking to understand possible mechanisms further, performed a meta-analysis on 120 wild stocks of sockeye, pink, and chum salmon from rivers in Puget Sound to Norton Sound in Alaska, a distance of more than 3000 km (Pyper et al. 2001; Mueter et al. 2002a; Mueter et al. 2002b; Pyper et al. 2002). The scientists analyzed correlations between spawner-recruit data and three coastal environmental variables—upwelling index, surface sea temperature, and surface sea salinity; their datasets spanned the period 1948 to 1996, with differing numbers of years available for different stocks. They were looking for and comparing spatial scales of correlation in the marine variables with salmon survival, hoping to learn at what scale survivals varied among stocks similarly and how survivals correlated with the marine variables.

Pyper et al. (2002) concluded on the basis of these analyses that the key biological or physical environmental processes influencing year-to-year variation in chum survival operate primarily at local or regional spatial scales as opposed to the scale of the entire northeastern Pacific Ocean. Variability covaried on a scale of up to about 1000 km with the strongest association occurring within approximately 550 km. Pyper et al. (2002) further concluded that it appears that mechanisms causing chum survival to covary similarly between populations were primarily operating in the early marine life phase, meaning that populations located within Washington State and southern British Columbia appear to be affected similarly by marine conditions. Populations originating further north appear to be responding to marine conditions localized to those areas. Pyper et al. (2001) reported similar patterns of covariance by pink salmon; they concluded that pinks were being affected at a slightly smaller scale—i.e., by marine conditions somewhat closer to natal streams. These findings suggest that chum and pink fry originating in Puget Sound are strongly affected by marine survival conditions localized to this region, encompassing at least the SJDF and the area south of the western edge of Vancouver Island. Pyper et al.

(2001, 2002) offered no explanations about how PDO and ENSO processes may be affecting localized marine conditions.

While uncertainty remains about the role of the PDO on marine survival of chum salmon, recent returns of both summer and fall chum to Puget Sound strongly suggest that they are being positively affected by the cool phase of the PDO. A similar pattern is evident for pink salmon.

Regime shifts operating on Fraser River pink salmon were analyzed by Beamish et al. (2004b) using stock-recruit methods in a way to clearly illustrate effects of shifts in marine survival. The traditional way of removing the effect of density for examining the population data is by plotting the log of adult recruits (R) divided by spawners (S), i.e., $\log(R/S)$, against spawners (Hilborn and Walters 1992)—this is done by applying a Ricker form of the S-R model. (This procedure linearizes the plots and enables them to be visually inspected for patterns.) This is useful for distinguishing different patterns—or shifts—in productivity that might correspond with regime shifts. The linear regression y-intercept of the plot gives an estimate of stock productivity as would be obtained with Ricker fits to the data.

Beamish et al. (2004b) found clear shifts in production relationships that correspond well with climate regime shifts. Linear regressions revealed how productivity was affected by regime shifts. Prior to the documented 1998 shift, the most productive regime in the data set was 1978-88. The 1990-98 regime was extremely unproductive (Beamish included 1998 here because the shift occurred in late 1998, hence 1998 conditions applied to brood year 1997 juveniles). Beamish et al. (2004a) reported that pink salmon early marine survival, growth, and feeding increased significantly beginning with the 1999 brood year in the Strait of Georgia.

Lestelle et al. (2006) applied the procedures used by Beamish et al. (2004b) to pink salmon populations in Puget Sound. They concluded that while there was clear evidence of effects of regime shifts, the patterns differed somewhat from those reported for Fraser River pinks. Certain Puget Sound populations (e.g., Skagit) demonstrated no response to any regime shift for the period of record examined. Other populations (e.g., Stillaguamish-Snohomish and Hood Canal) showed a strong response to the regime shifts. Lestelle et al. (2006) suggested that the level of effect of regime shifts appears to be related to water circulation patterns and the amount of direct influence by the SJDF inflow. Populations produced in rivers emptying directly to the SJDF and in those rivers feeding the main basin of Puget Sound closest to Admiralty Inlet showed the greatest response to the 1998 shift. Moreover, the effect on population productivity appears to occur within the very early phase of exposure to the marine environment.

5.2 Evidence for Effects of Regime Shifts on Summer Chum

We applied the basic procedure described by Beamish et al. (2004b) to the SJDF and Hood Canal summer chum populations and to each of their extant subpopulations. As noted above, the procedure removes the effect of density by plotting the log of adult recruits (R) divided by spawners (S), i.e., $\log(R/S)$, against spawners (Hilborn and Walters 1992). (This procedure linearizes the plots and enables them to be visually inspected for patterns.) The linear regression y-intercept of the plot gives an estimate of population productivity as would be obtained with Ricker fits to the data and parameter estimation techniques.

In performing the analysis, it became evident that certain spawning aggregations appeared to show little or no effect of regime shifts on intrinsic productivity, whereas in certain of those cases it appeared that there was good evidence of an effect on capacity.

5.2.1 Strait of Juan de Fuca Population

Figure 22 is a composite graphic for the SJDF population enabling the reader to inspect the pattern of adult recruits, presented by brood year, and brood year spawners. Also shown are the patterns of brood year recruitment presented as standard deviations (or anomalies) from the average brood year recruitment for the time period of interest, as well as the deviations for the annual average PDO index for the same time period. The reader is reminded that our analysis here only extends through brood year 2006, which in effect includes return years up to 2010.¹⁸ The PDO index pattern shown in Figure 20, which also covers 2012, clearly shows that the 1998 regime shift is still in effect. Similarities in the patterns of deviations seen in the bottom part of Figure 22 show a strong correspondence between marine regimes and brood year recruitment.

Figure 23 presents the plots of the log of recruits per spawner on spawner abundance, showing the data plotted for all brood years together, then for brood years 1979-1998 and brood years 1999-2006 separately, and finally for the two separate groups on the same plot. The figure clearly illustrates a strong effect of the 1998 regime shift on productivity for the SJDF population.

Figure 24 shows the S-R data for the population plotted along with the S-R curves (using the Beverton-Holt form) for each of the two time periods. The figure illustrates that the 1998 regime shift had a very large effect on population capacity.

Similar graphics for each of the two extant subpopulations of the SJDF summer chum population are provided in Appendix A. Both subpopulations show a strong effect of the 1998 regime shift on productivity, whereas only the Salmon-Snow subpopulation shows a positive response to capacity. The capacity of the Jimmycomelately subpopulation appears to have no substantial response to the regime shift.

¹⁸ / The reader should note that for brood year 2006 the recruitment of 4 year old fish was estimated based on a recent year average because at the time the analysis was performed the empirical estimate for this age class was not yet available.

SJDF population

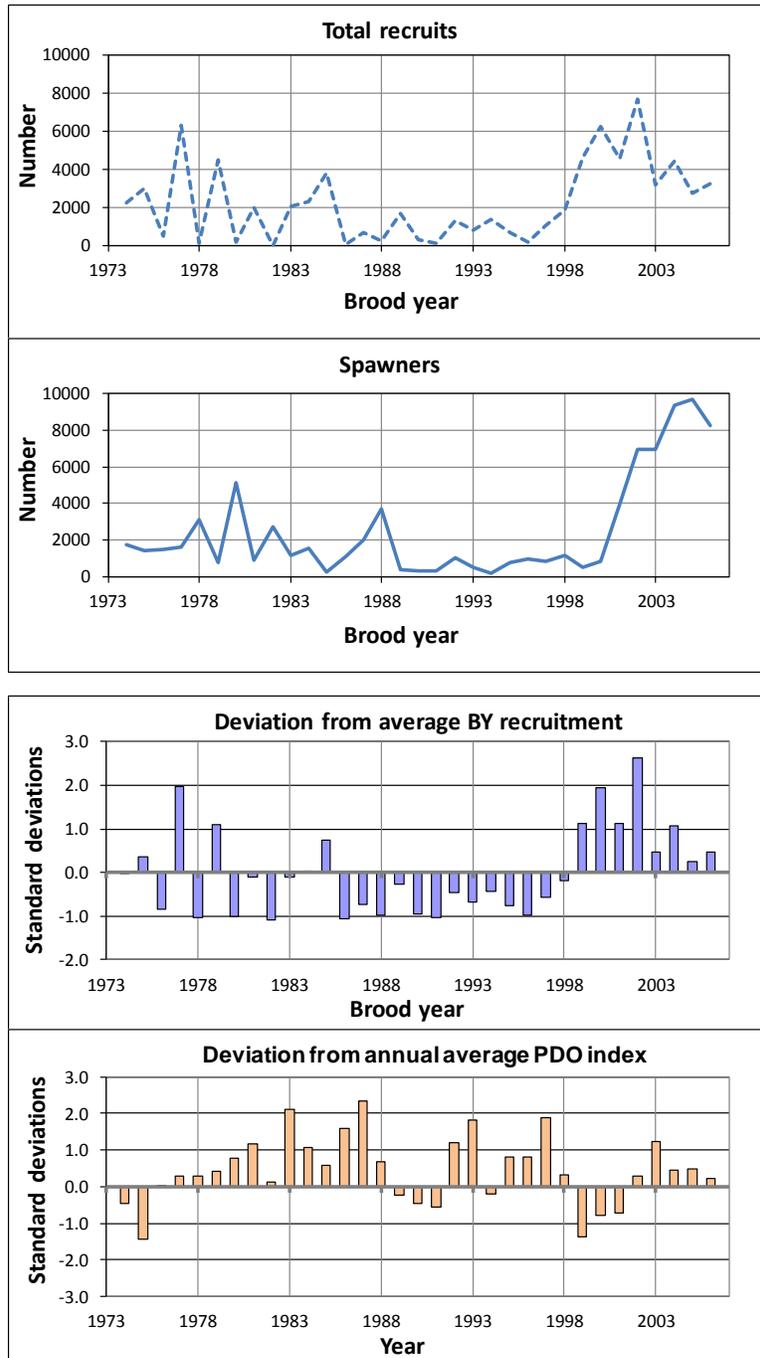


Figure 22. Patterns of total recruits for the SJDF summer chum population by brood year, spawner abundance by brood year, deviations from average recruitment, and deviations from the average PDO index.

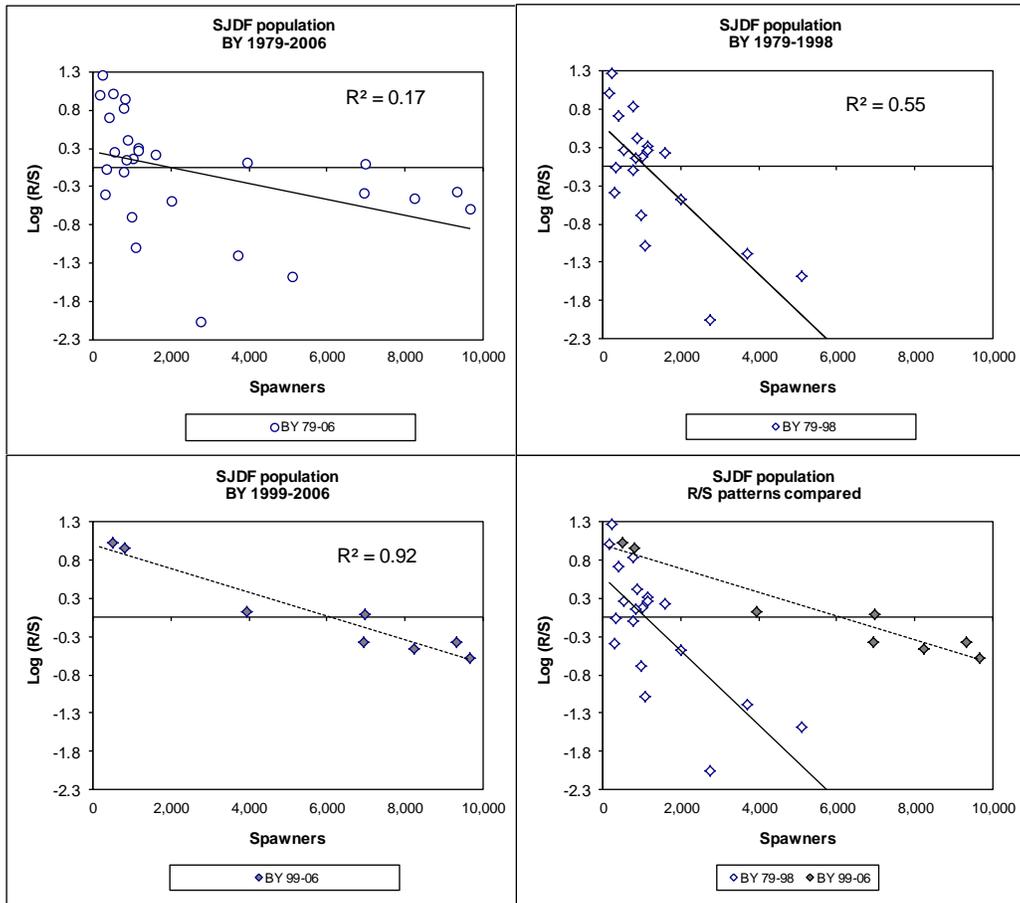


Figure 23. Spawner (S) and recruitment (R) plots grouped for different time periods for the SJDF summer chum population. Brood years 1979 to 2006 are used.

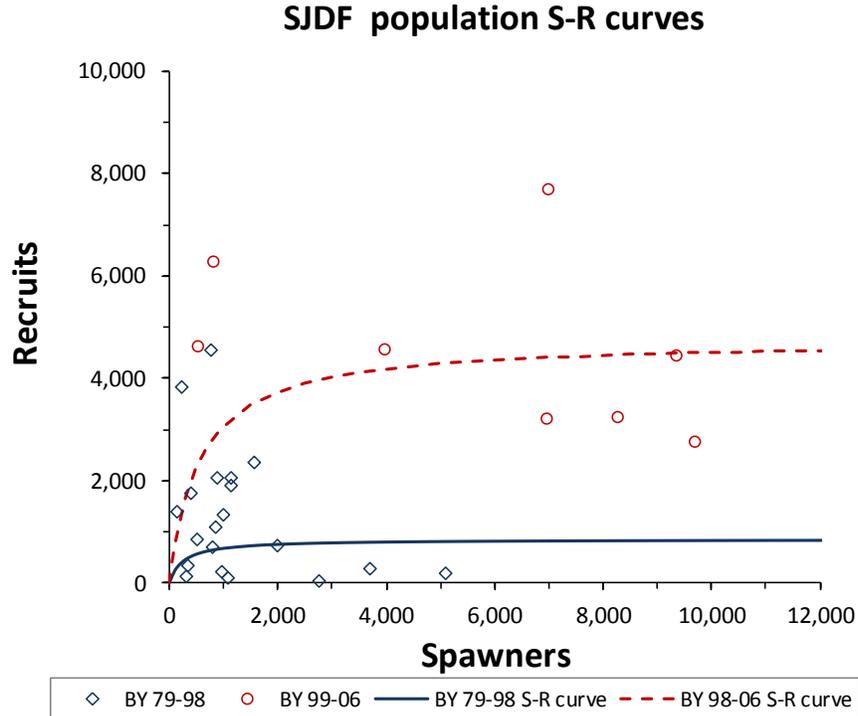


Figure 24. Spawner-recruit plots for brood years 1979-1998 and 1999-2006 for the SJDF summer chum population.

5.2.2 Hood Canal Population

Figure 25 is a composite graphic for the Hood Canal pattern of adult recruits, presented by brood year, brood year spawners, and the patterns of brood year recruitment presented as standard deviations (or anomalies), as well as the deviations for the annual average PDO index for the same time period. The reader is reminded that our analysis here only extends through brood year 2006, which in effect includes return years up to 2010 (see footnote 18). The PDO index pattern shown in Figure 20, which also covers 2012, clearly shows that the 1998 regime shift is still in effect. Similarities in the patterns of deviations seen in the bottom part of Figure 25 show a good correspondence between marine regimes and brood year recruitment.

Figure 26 presents the plots of the log of recruits per spawner on spawner abundance, showing the data plotted for all brood years together, then for brood years 1979-1998 and brood years 1999-2006 separately, and finally for the two separate groups on the same plot. We also show plotted separately brood years 1999-2006 with and without 2004 used for comparison. Brood year 2004 had an exceptionally large number of spawners, and without doubt there would have been an extremely strong drop-off in reproductive success due to heavy superimposition, which would have affected the slope of the associated regression thereby potentially giving a false signal for productivity. Whether or not brood year 2004 is included in the regression plots affects how one can interpret the results. Excluding brood year 2004, we would conclude that the regime shift had a large effect on productivity.

Figure 27 shows the S-R data for the population plotted along with the S-R curves (using the Beverton-Holt form) for each of the two time periods. The figure illustrates that the 1998 regime shift appears to have had a large effect on population capacity.

Similar graphics for each of the six extant subpopulations of the Hood Canal summer chum population are provided in Appendix A. The effect of regime shifts on performance varies among the subpopulations. Our preliminary conclusions based on visual inspections of the patterns are summarized below, indicating that we find an effect to be likely (by yes), unlikely (by no), or that it is unclear whether an effect has occurred (by uncertain):

<u>Subpopulation (south to north)</u>	<u>Productivity</u>	<u>Capacity</u>
Union	no	yes
Lilliwaup	yes	yes
Hamma Hamma	yes	yes
Duckabush	yes	yes
Dosewallips	yes	yes
Quilcene	uncertain	yes

We note, however, that we have greater reservation about whether there has been an effect on capacity for the Union and Lilliwaup subpopulations than for the other subpopulations. The pattern of data points for these two subpopulations make it less clear about the effect, though the analysis indicate that a positive effect did occur with the 1998 regime shift.

Hood Canal population

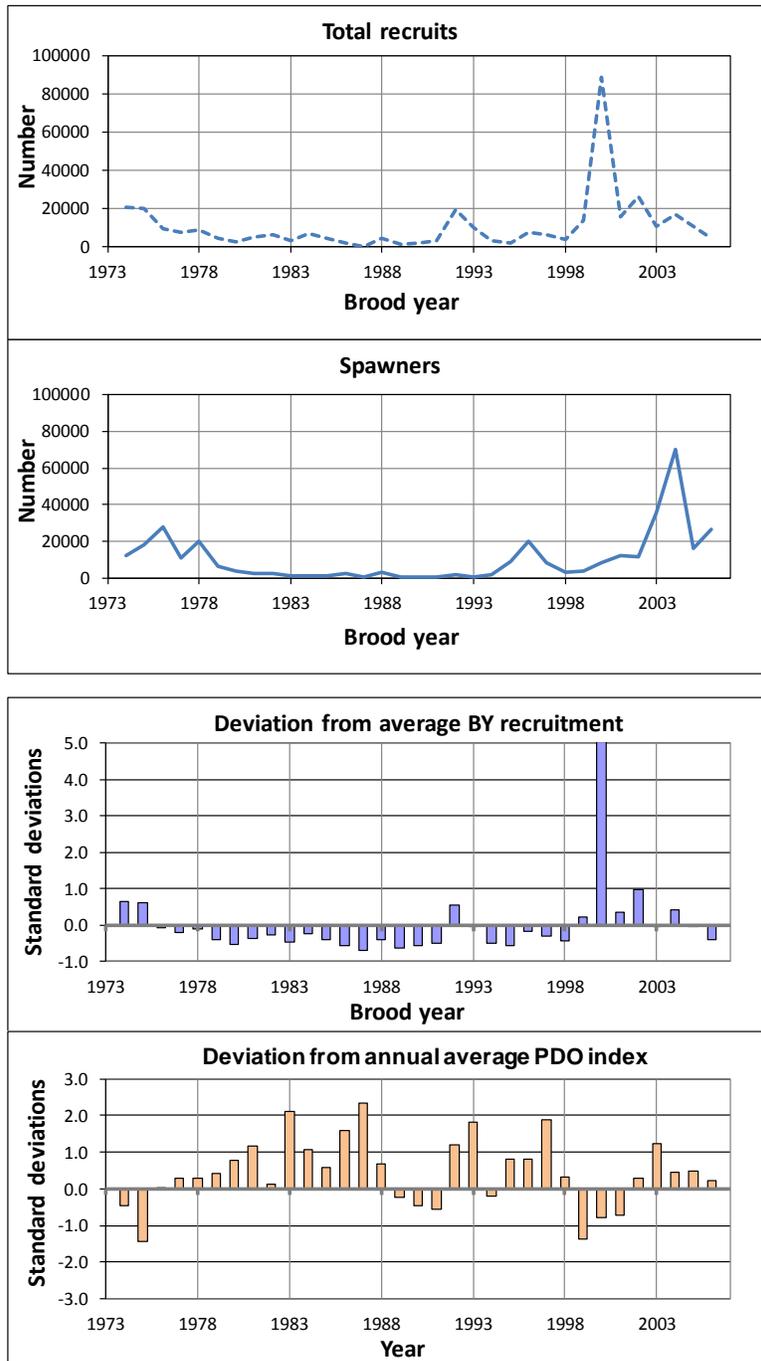


Figure 25. Patterns of total recruits for the Hood Canal summer chum population by brood year, spawner abundance by brood year, deviations from average recruitment, and deviations from the average PDO index.

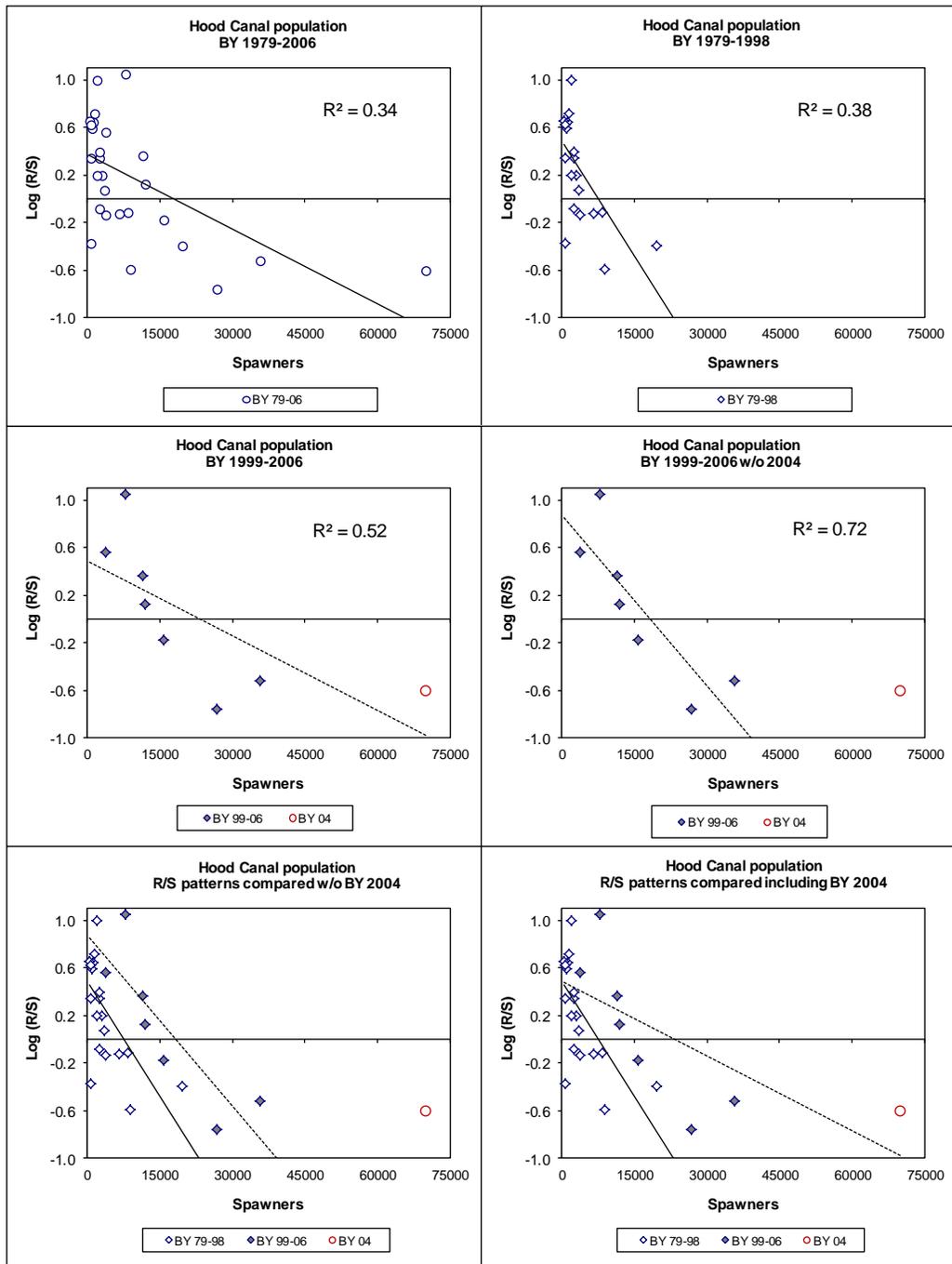


Figure 26. Spawner (S) and recruitment (R) plots grouped for different time periods for the Hood Canal summer chum population. Brood years 1979 to 2006 are used.

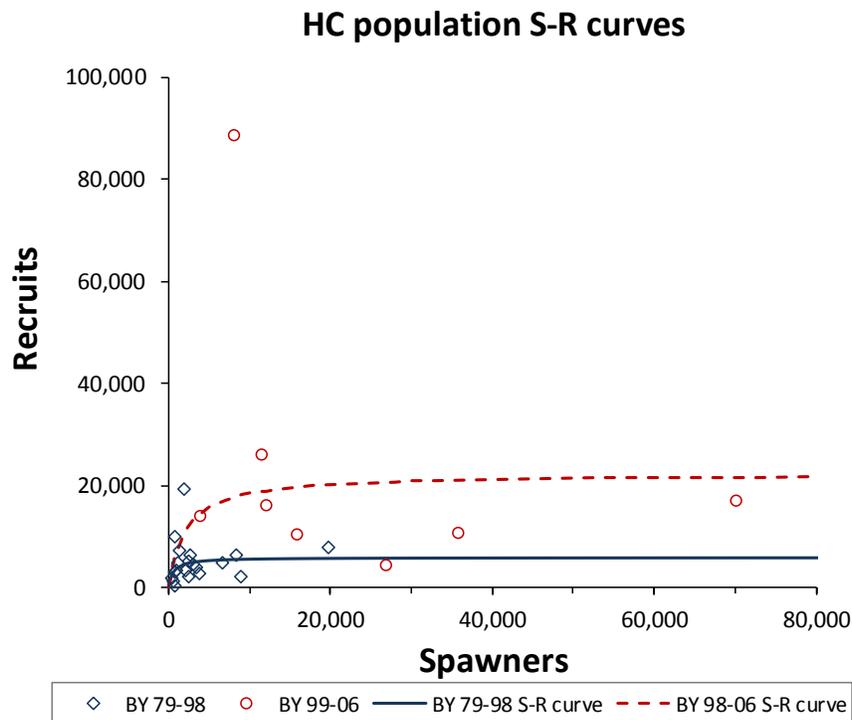


Figure 27. Spawner-recruit plots for brood years 1979-1998 and 1999-2006 for the Hood Canal summer chum population.

5.2.3 Spatial Patterns for Effects of Regime Shifts

The results from the previous section show that the regime shift of 1998 had different levels of effect on performance depending on location of the natal stream of a subpopulation. To consider further the spatial pattern of effects, we formulated what we call a standardized capacity index to compare the relative role of the marine environment in producing adult recruits for each subpopulation associated with each marine regime. The capacity index was defined to be the adult recruit capacity estimated for each group of brood years associated with a marine regime (i.e., brood years 1979-1998 and 1999-2006) divided by the surface area of the stream reaches utilized by summer chum spawners for each subpopulation.¹⁹

Figure 28 shows that a clear pattern exists during the marine regimes associated with the shifts of 1977/1978 and 1998: the contribution of the marine environment to potential production of adult recruits for a subpopulation is greatest for subpopulations located closest to the SJDF during both regimes. There is a clear south to north increase in production potential associated with the marine environment during both marine regimes shown.

¹⁹ / The surface area of the freshwater stream reaches was obtained from the EDT data set for each subpopulation. The surface areas were computed for the month of September, when peak spawning typically occurs for the subpopulations.

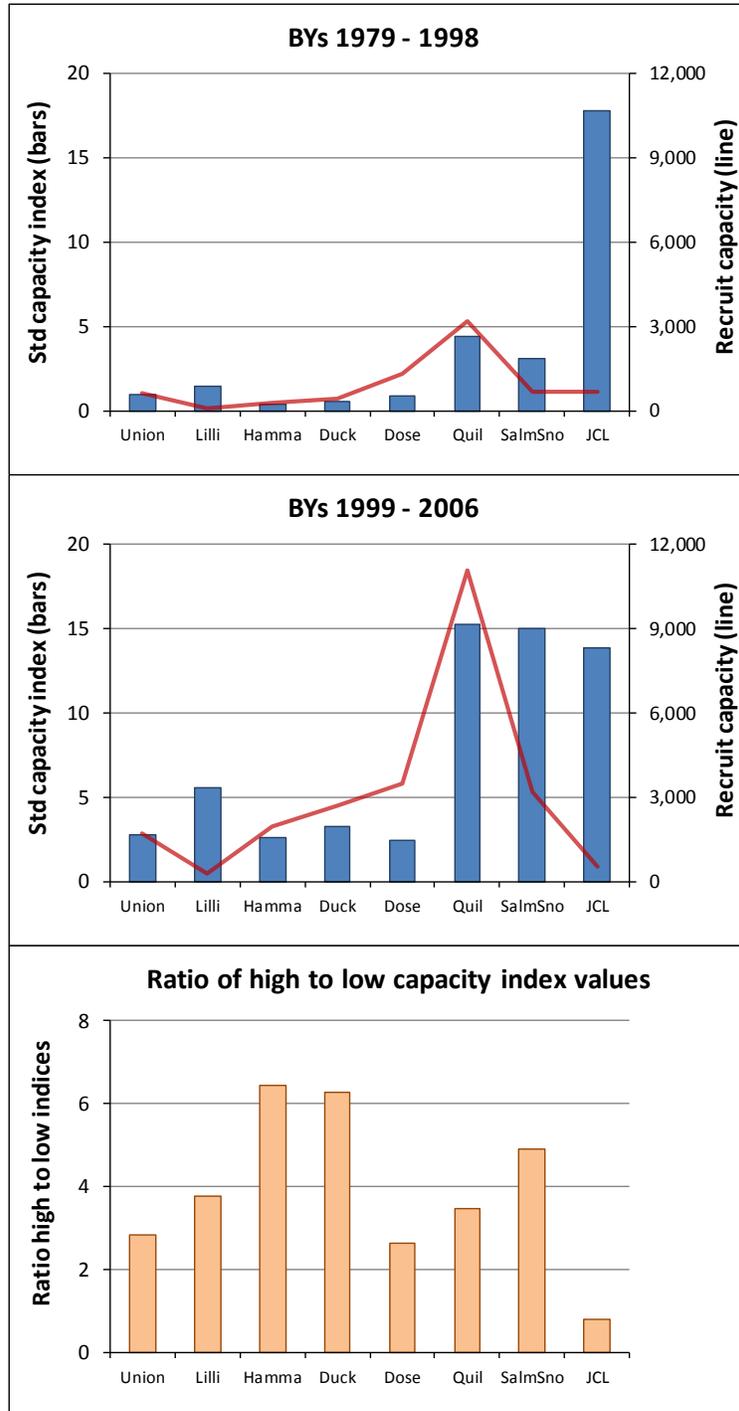


Figure 28. Spatial patterns from south to north of production potential for the eight extant subpopulations within the Hood Canal summer chum ESU using a capacity index (bars) to show the contribution of the marine environment to adult recruit production. The line in the top two charts is the estimated capacity (as total recruits) of the population.

The standardized capacity index values for all subpopulations are substantially greater after the regime shift of 1998 except for Jimmycomelately (JCL). A comparison between regimes for this subpopulation

suggests that the contribution of the marine environment to production potential is always high, regardless of regime, which may be due to the spawning stream being in closest proximity to the SJDF compared to the other subpopulations. The three subpopulations closest to the SJDF exhibit much higher capacity index values than for the more southerly subpopulations after the 1998 shift.

The estimated capacities for each subpopulation during both regimes are also plotted in the two upper panels of Figure 28 (solid red lines). The patterns indicate that in recent decades the core production units in Hood Canal are the Quilcene and Dosewallips subpopulations during both regimes. For the SJDF population, the core production unit is the Salmon-Snow Creek subpopulation. The largest capacity production units are assumed to be the core subpopulations.

Whether these three subpopulations are consistently the largest capacity production units in the ESU extending over multiple regimes (i.e., including regimes prior to the 1978 shift) is less certain. The abundance patterns shown in Figures 8-11 suggest that in Hood Canal the largest production units may have been south of Dabob Bay, i.e., south of the Quilcene ecological diversity group (Figure 5) though not including the Union subpopulation. If this is true, it would suggest that different subpopulations could be more dominant during some regimes, while others are more dominant during others, as Hilborn et al. (2003) reported to occur within the Bristol Bay population complex. Changes in which subpopulations are most dominant during a particular PDO period might occur as a result of variable influence of regime shifts and how marine food webs are more or less affected within a geographic area.

The bottom panel in Figure 28 shows the ratio of the capacity index values after the regime shift to values before the shift. The ratio shows the relative increase in the capacity index due to the 1998 shift. The greatest increases occurred for two adjacent subpopulations (Hamma Hamma and Duckabush) in the mid-section of Hood Canal south of Dabob Bay.

5.2.4 Effects of Ocean/Climate Shifts on Viability

The effect of the 1998 regime shift on the viability of each population is seen by plotting estimates for productivity and capacity for each population unit with their viability curves for the two regimes beginning with brood year 1979 (Figure 29). The results show that viability is very strongly affected by the ocean/climate regime for the brood years analyzed.

Neither population is shown to exceed the 5 percent risk threshold curve with a 0 percent exploitation rate during the regime associated with brood years 1979-1998, though the Hood Canal population is only slightly below the threshold. After the 1998 regime shift, the Hood Canal population exceeds even the threshold associated with a 30 percent exploitation rate by a large margin, while the SJDF population is only slightly above the threshold with a 0 percent exploitation rate. The SJDF population is shown to have been at very high risk of extinction during brood years 1979-1998.

Similar graphics are provided in Figure 30 for each of the extant subpopulations. It is evident that the state of the ocean/climate regime has a very strong effect on the viability of each subpopulation with two exceptions: Lilliwaup Creek and Jimmycomelately. Both of these subpopulations remained at high risk of extinction after the 1998 shift. In all other cases, performance increased substantially after the 1998 shift to exceed the minimum viability curve by a relatively large amount.

It is noteworthy, as discussed earlier in the report, that spawning escapements reached extremely low levels in Lilliwaup Creek in the 1990s, such that depensation effects might have become operative. The

slow rebound of this subpopulation following the 1998 regime may have been affected by such low spawner abundances (Figure 9). Also, large slides in the lower drainage that occurred over a several year period in the 2000s likely hampered a rebound. The abundance pattern seen in Figure 9 (e.g., 1976) suggests that Lilliwaup Creek has produced much larger numbers of fish in at least some years during the cool phase of the PDO.

It is important to recognize that all of the results shown in Figures 29 and 30 represent performance characteristics of the production units prior to substantial contributions of restoration actions, which began in earnest in the mid-2000s in some watersheds.

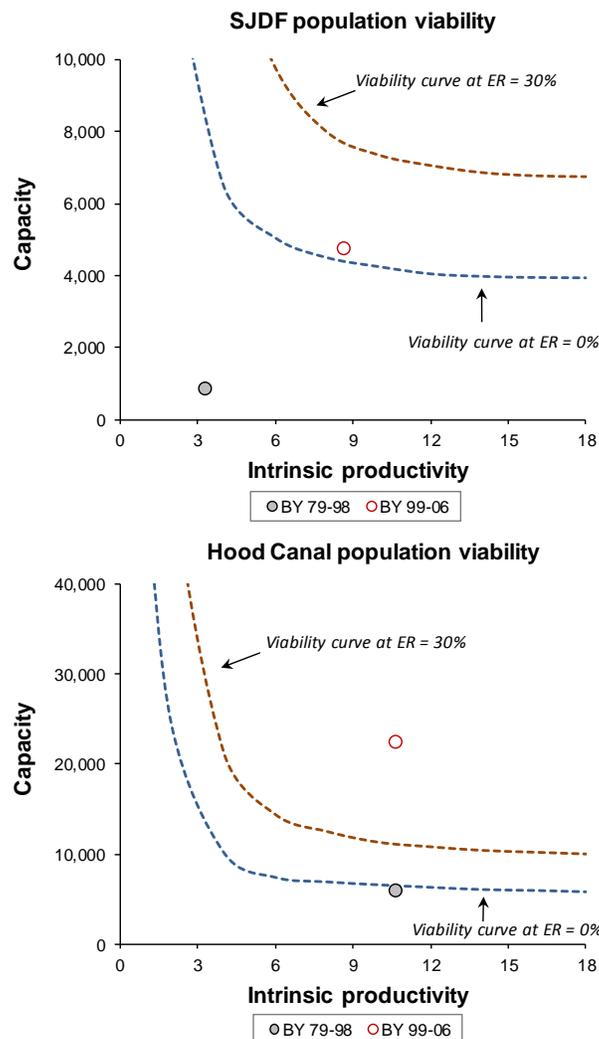


Figure 29. Population performance parameters for brood years (BY) 1979 to 1998 (warm PDO) and 1999 to 2006 (cool PDO) plotted relative to viability curves (5 percent extinction risk) for the SJDF and Hood Canal summer chum populations. Viability curves associated with both 0 and 30% exploitation rates are shown.

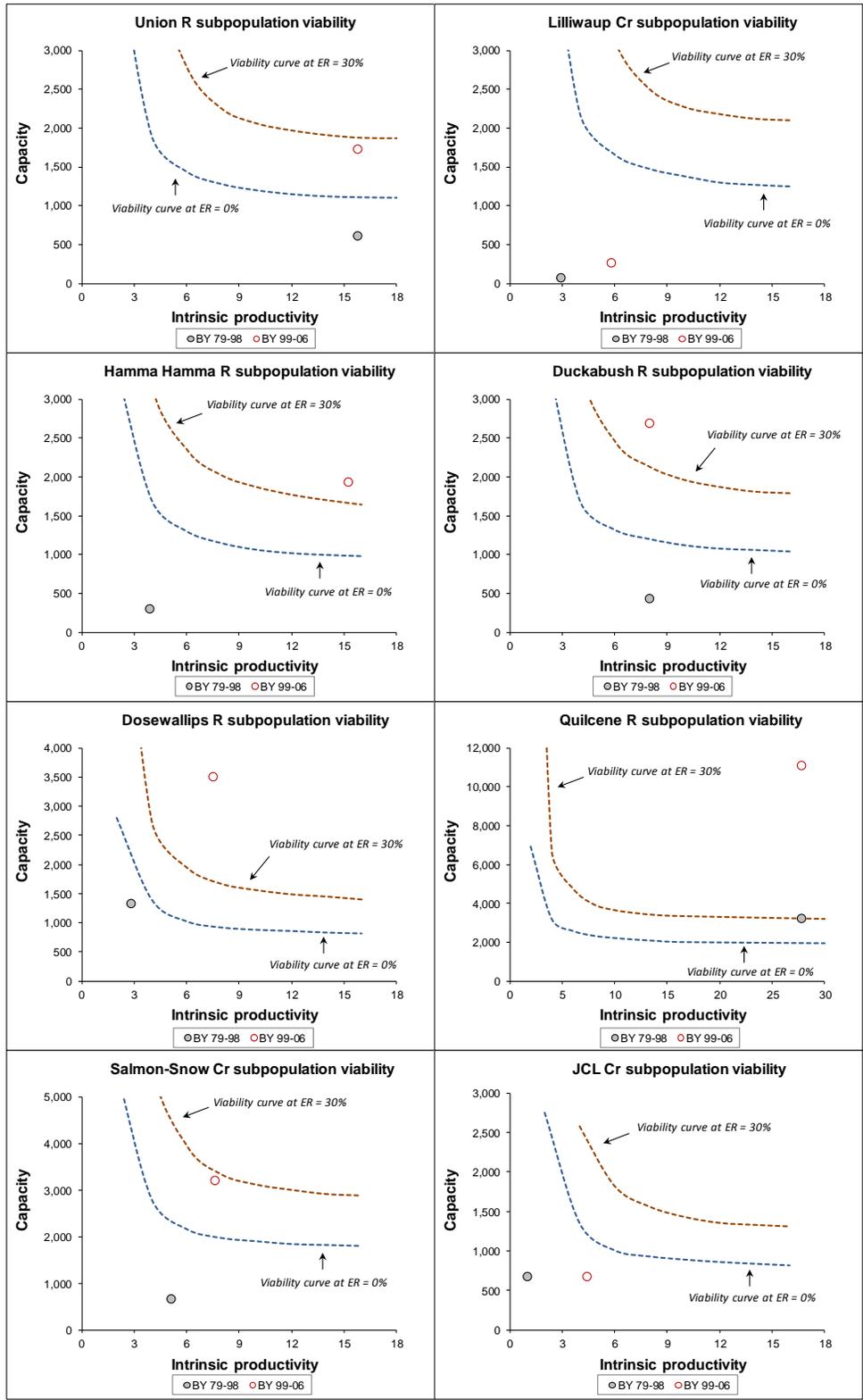


Figure 30. Performance parameters for brood years (BY) 1979 to 1998 (warm PDO) and 1999 to 2006 (cool PDO) plotted relative to viability curves (5 percent extinction risk) for extant subpopulations belonging to the SJDF and Hood Canal summer chum populations. Viability curves associated with both 0 and 30% exploitation rates are shown.

5.3 Implications for Recovery Goals

Differences in how the populations and their subpopulations have performed relative to viability curves demonstrate stark contrasts in extinction risks between ocean/climate regimes. We would be wrong if we looked only at recent year performances and concluded that recovery has been achieved (for both populations and most subpopulations). It is evident that during the cool phase of the PDO that marine conditions can so override the contributions of freshwater and subestuarine habitats that one might conclude those habitats are largely inconsequential.

Lawson (1993) considered the effects of decadal-scale cycles in marine survival to the viability of Oregon coastal coho. He proposed a model of the combined effects of freshwater habitat degradation (Figure 31a) and oceanic cycles (Figure 31b), which can be created by first characterizing these two factors separately, then in combination, producing a pattern of expected ocean escapement (or recruits) (Figure 31c). Periods of high marine survival mask the long-term downward trend driven by declining freshwater habitat conditions. Lawson stated that during a period when marine survival is high that managers and politicians will naturally have a tendency to relax restoration efforts and claim success for their projects. He concluded that the true measure of success for salmon recovery will be when populations perform at a level needed to survive through episodes of low marine survival and reduced abundance. Similarly, NMFS (2010) reviewed the PDO index pattern with regard to salmon survival and concluded: “The survival and recovery of these species will depend on their ability to persist through periods of unfavorable hydrologic and oceanographic conditions.”

We propose that summer chum recovery can only be truly evaluated during a period when it is evident that a warm phase of the PDO (and relatively poor summer chum population performance) has been in place for some number of years. Population and subpopulation performance must be sufficient to survive the entirety of a future warm phase of the PDO. Thus it becomes evident that oceanic effects do not reduce the importance of the condition of freshwater and subestuarine habitats. Rather, oceanic effects heighten the importance of freshwater and river-mouth estuarine habitats during periods of low marine survival. To maintain viability through the duration of a warm PDO phase, which can last at least 20 years, will require relatively good habitats to exist to buffer and offset poor marine survival. We note that both Big Beef Creek and Tahuya River exhibited poor to modest response to reintroduction efforts during the cool phase of the PDO. This may indicate that current habitat quality is too poor for naturally spawning fish to sustain a spawning aggregation without habitat restoration and/or without periodic supplementation.

The benefits of habitat restoration actions should also enhance abundance levels during a cool PDO phase, boosting abundances over levels that would be achieved without restoration, which should have carry-over benefits through periods of poor marine conditions. It bears noting that Lawson (1993) suggested that during such periods of elevated abundances associated with improved oceanic conditions that natural straying from core subpopulations to extirpated subpopulations would be increased, thereby aiding recovery of a metapopulation. In fact, more summer chum ‘strays’ from supplementation programs were observed in adjacent streams during years of relatively high escapements compared to years with lower escapements (WDFW and PNPTT 2007).

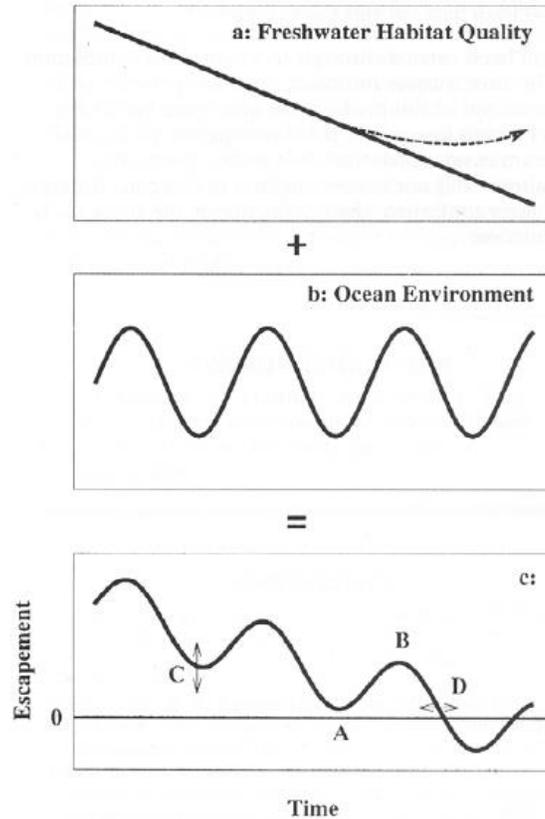


Figure 31. Conceptual model presented by Lawson et al. (1993) of the effects of declining habitat quality and cyclic changes in ocean productivity on the abundance of a salmon population. a – Trajectory over time of habitat condition. Dotted line represents possible of habitat restoration projects. b – Generalized time series of ocean productivity. c – Sum of the two top panels where letters represent the following: A = current condition, B = situation in the future, C = change in escapement from increasing or decreasing harvest, and D = change in time of extinction from increasing or decreasing harvest.

6.0 Considerations for Effects of Climate Change

NMFS urges salmon recovery planners to consider the effects of climate change patterns on future recovery (Ford ed. 2011). This section provides an approach that enables us to consider how viability curves might reasonably be shifted as a result of climate change. In considering these effects, it enables us to assess what additional level of habitat restoration might be needed to mitigate for climate change.

6.1 Necessity for Considering Climate Change Effects

Climate change is not the same as climate regime shifts described in Section 5. Past patterns of regime shifts suggest that they are generally cyclical, shifting to a new type of equilibrium condition, then shifting back again to a former condition. In contrast, climate change implies a directional, gradual change to a new set of conditions that has not been experienced for some much longer period of time. Climate change would likely persist for some extended period of time into the future.

Climate change patterns seen in many areas of the world are signaling environmental variation is increasing—major climate events are occurring with greater frequency, and often they are more extreme when they occur (e.g., Alverson et al. 2001; Furniss et al. 2010). Examples of increased environmental variation in the Pacific Northwest in recent decades include an increase in frequency of heat waves, a northward shift of Pacific storm tracks with intensification of storms, and increased interannual variability in precipitation (meaning wetter wet years and drier dry years) (Furniss et al. 2010). Frequency and intensity of flooding, as well as late summer low flows, in streams of the Puget Sound region are projected to increase this century (Mantua et al. 2010; Mote and Salathe 2010; Elsner et al. 2010), which, in turn, can also be expected to alter circulation patterns within the Puget Sound complex.

There is uncertainty about how these various factors may affect environmental characteristics of importance to summer chum. In some cases, average conditions are likely to worsen for summer chum, negatively affecting survival, while other survival factors, such as those affected by marine conditions might tend to improve survival.²⁰ It is unknown, for example, how climate change might affect patterns of the PDO as it pertains to summer chum survival. Marine survival might tend to worsen somewhat, or it might tend to improve. Because of the uncertainty about how average survival conditions might change in various life stages of summer chum, we assumed that only variation in performance will change. We would note that implications of possible changes in survival conditions could be explored with life cycle modeling, such as with the EDT model used in this paper.

Theoretical models generally predict that increasing environmental variation will increase the probability of extinction (Lande et al. 2003; Drake and Lodge 2004). To consider how increased environmental variation associated with climate change might affect the viability of summer chum, we incorporated greater variation into the modeling used to estimate viability thresholds.

6.2 Projections for Effects of Climate Change

VRAP was used to model viability curves under climate change scenarios for each of the populations and their extant subpopulations. We assumed that variation in performance for each population will increase by 5, 10, or 15 percent with climate change over the next several decades. Percentage increases in variation were applied to the current condition variation levels (CVs)²¹ given in Table 3. These percentage increases were inferred from observed changes in environmental variation seen in weather patterns in North America in recent years. We recognize that there is considerable uncertainty about how much environmental variation might increase in the Puget Sound region; our approach provides a first step in examining this issue, which can be expanded upon at a future date.

²⁰ / Survival conditions might worsen, for example, from reduced late summer stream flows, which would tend to force summer chum to use less preferred spawning habitat (channel thwalweg rather than channel margins, see Lestelle et al. 2005c)—such areas can be more prone to greater scour during winter floods, which could become worse due to more intense winter storms. In contrast, early marine survival conditions could improve if a slightly warmer climate produces earlier phytoplankton and zooplankton blooms that could favor summer chum survival; this race tends to emigrate from Hood Canal on the leading edge of when plankton blooms occur, generally resulting in lower average marine survival than fall chum experience—see discussion in Lestelle et al. (2005a).

²¹ / VRAP needs two input parameters for determining variability. We found that there was a relationship between the two parameters, so we could change gamma a as described and then determine the corresponding gamma b. $\text{Gamma b} = 1,368 * (\text{gamma a})^{1.305}$.

Figure 32 provides the new viability curves with 5 and 10 percent climate change effects for each population, shown with performance parameters plotted separately for brood years 1979-1998 and 1999-2006. The viability curves are shifted up and to the right, setting a higher threshold in each case for viability to be achieved. During the warm phase of the PDO (brood years 1979-1998), neither population would be viable with a 5 percent increase in variation. The results illustrate that the beneficial effects of restoration and protection actions will become more important to achieve recovery with climate change.

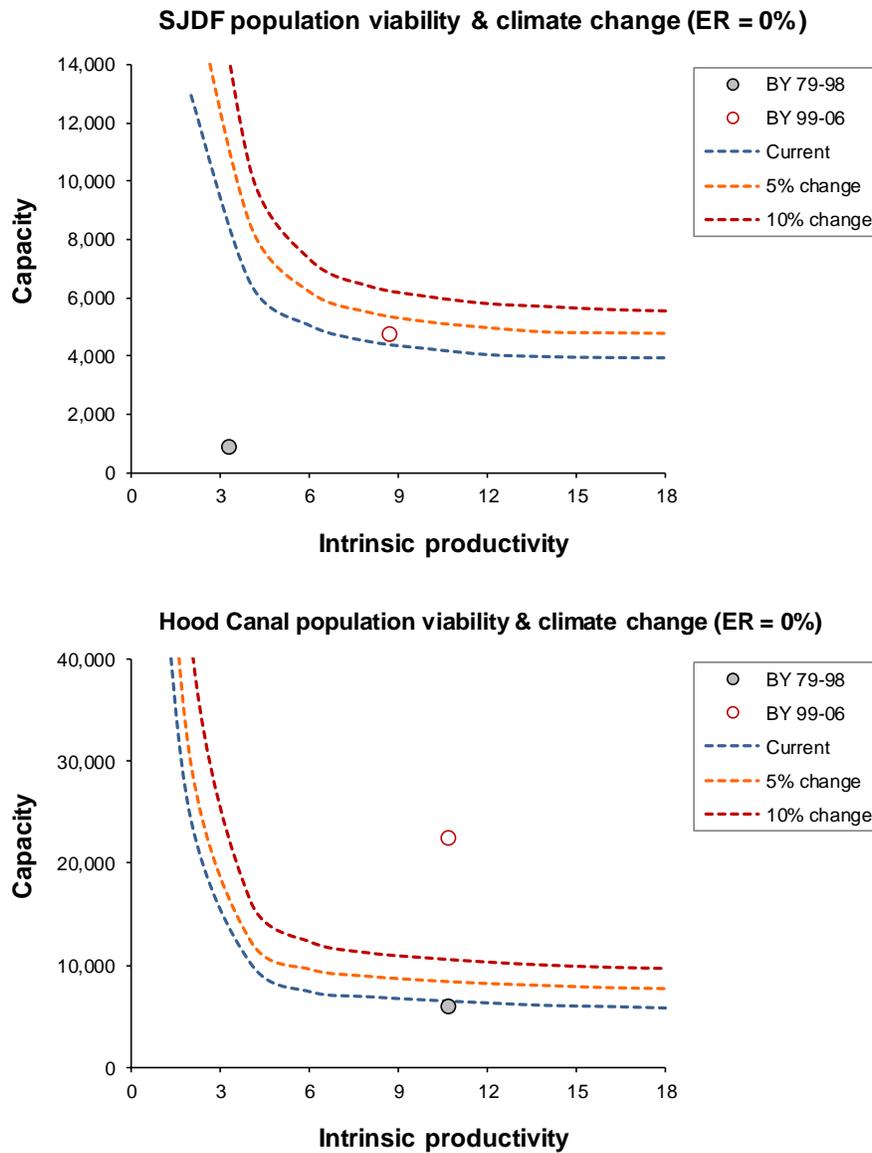


Figure 32. Population performance parameters for brood years (BY) 1979 to 1998 (warm PDO) and 1999 to 2006 (cool PDO) plotted relative to viability curves (5 percent extinction risk) for the SJDF and Hood Canal summer chum populations with variation increased by 5 and 10 percent to reflect future climate change.

7.0 Projected Effects of Habitat Actions

This section provides an assessment of population and subpopulation performance and associated habitat conditions for relevant baseline time periods based on habitat modeling. The baselines serve as reference conditions for comparison to viability thresholds, giving us a way to identify gaps in performance relative to a viability target.

We remind the reader that the viability standards using abundance and productivity need only to be achieved at the population scale for the sake of delisting (NMFS 2007; Sands et al. 2009). Here, our assessment of performance gaps at the subpopulation level is meant to help guide strategic restoration and recovery planning at that scale. While it is not a requirement for delisting, recovery planning, to the extent feasible, should aim to achieve viability thresholds as the minimum target for each subpopulation, recognizing that limitations and constraints will prevent reaching those targets on some subpopulations. As noted in Section 3.3 (Recovery Criteria), true recovery can likely only be gained if enough subpopulations are sufficiently robust to provide beneficial contributions to the population as a whole.

Modeling the effects of changes to habitat conditions, either associated with future watershed development that further degrades habitat or with restoration actions that improve habitat, is the only way of evaluating potential changes to salmon performance to guide recovery planning now. Recovery planning cannot wait until actions have been fully implemented and their outcomes fully realized to assess the need for additional actions and plan accordingly. Actual outcomes from habitat actions will be realized after many decades. Effectiveness and validation monitoring activities will need to consider actual on-the-ground outcomes over time, which will include how the salmon populations, in fact, respond. In the meantime, planning can benefit by using modeling projections to assess expected outcomes.

7.1 Assessment of Baseline Conditions

We identified four sets of baseline conditions, representing different time periods with regard to habitat condition. These baselines serve to identify how much adverse change to habitat has taken place in the past—giving a reference condition for how much potentially can be regained, how much adverse change might still occur with future watershed development, and how much habitat improvement is projected to occur from actions that have already been implemented (or soon to be from actions already funded). We refer to these baselines as follows:

- Historic condition;
- Year 2001, which marked the time when most significant habitat protection and restoration actions began;
- Year 2001 with projected future watershed buildout; and
- Year 2014, which is actually a projection into the future of what conditions will result from actions implemented between 2001 and 2014, including effects of future buildout.

The 2014 baseline is the reference condition for assessing the gap that remains to be met between it and recovery by future actions, which as yet remain unfunded or to be identified.

We assessed each of these baselines for each of the eight extant subpopulations—extirpated subpopulations have not been modeled. We then rolled up the results to the population level to formulate the assessment for the two populations, recognizing that contributions from reintroductions

were not taken into account. The Ecosystem Diagnosis and Treatment (EDT) model was used to assess habitat characteristics in the natal watersheds, their subestuaries, and for the nearshore environment within Hood Canal and adjacent areas of the Puget Sound complex (Lestelle et al. (2005a and b). Buildout effects, which represent projected watershed development to approximately the year 2025 (described in Peterson 2006 and applied in Lestelle et al. 2005b), and habitat restoration and protection actions were modeled as described in those documents, as well as in Thompson et al. (2009). Similar action modeling, using comparable techniques as used here, performed by the lead author to this paper has also been described in Carmichael and Taylor (2009) and NMFS (2012).

A brief description of the EDT model is helpful here. The EDT model is a habitat-based model developed to estimate salmonid population performance measures as determined by characteristics of the aquatic habitat (Figure 33) (Mobernd et al. 1997; Blair et al. 2009). It was developed to aid in evaluating potential actions and to help in decision making about which actions to move forward with. The model incorporates a wide range of environmental attributes, including both abiotic and biotic elements, which are used to characterize the environment. While the model considers conditions at all life stages—producing population parameters for the entire life cycle—it was originally built to address factors that affect freshwater survival. It was expanded to model marine habitats for chum (Lestelle et al. 2005a and b) and pink (Lestelle et al. 2006) salmon and was used to help develop the HCCC summer chum recovery plan.

Conceptual model

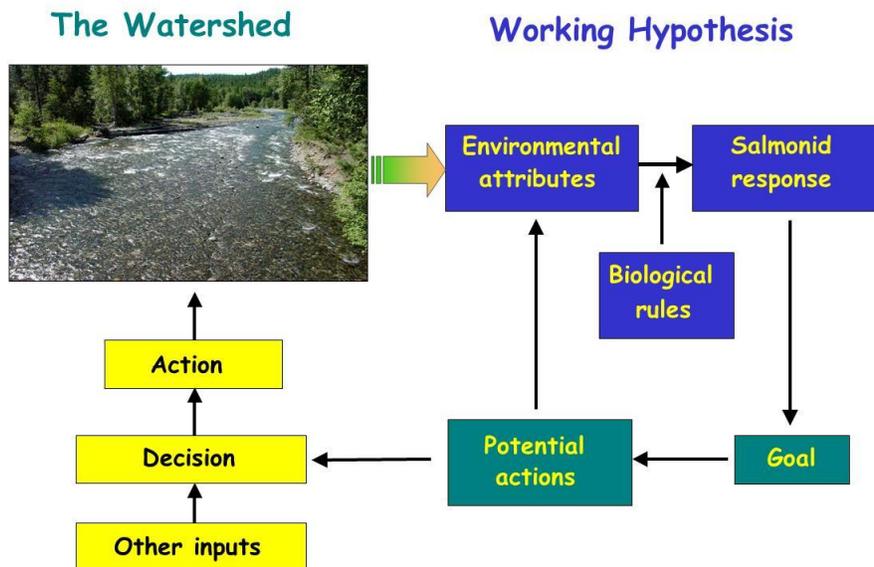


Figure 33. Simplified conceptual framework of the EDT model and how it is used in watershed and recovery planning. The left side of the chart is meant to represent the actual watershed (or environment) of interest, together with the decision making process that selects actions and implements them. The right side of the chart represents the modeling process of the watershed’s habitats to project the performance of a salmonid population in response to the habitat condition. Modeling is used to evaluate different habitat scenarios and to compare their projected outcomes to goals.

The model applies sets of biological rules (Figure 33), derived from extensive literature review and analyses, to project population performance for the species of interest. The standard output consists of Beverton-Holt population parameters for juvenile salmon yield (leaving the natal watershed during outmigration), total adult recruits (in the absence of fishing), and spawners (successfully spawning). The parameters produced by the model are intrinsic productivity, capacity, and equilibrium abundance—all of which are estimated for each scenario being modeled. Output is also provided in the form of a quantitative limiting factors analysis for diagnostic purposes and to assess the overall effectiveness of restoration actions.

The model was designed for analyzing effects of habitat actions on population performance. As part of its setup, it requires characterizations of both historic and existing habitat conditions for all stream reaches used by the population of interest (see Lichatowich et al. 1995). Similar characterizations are done for estuarine and marine segments for summer chum. These characterizations provide a simple, logical framework for considering the effectiveness of possible habitat actions (Figure 33). The model is then used to explicitly define the extent that an action might be expected to move reach-specific habitat characteristics back toward the pre-development state (Thompson et al. 2009). It also allows for defining actions that represent future watershed development, as would occur over time with human population growth. In this case, the model is used to identify the extent that reach characteristics move further away from the pre-development state and toward a fully developed one. Actions intended to give protection against further habitat loss can thereby be analyzed.

To analyze actions for the Hood Canal ESU, historic and more recent freshwater habitat characteristics, approximately representing those that existed in 2001, were identified by a technical team assembled by HCCC in 2005. The approach used to characterize estuarine and nearshore habitat conditions is described in Lestelle et al. (2005b). Outputs from the EDT model were used to formulate a diagnosis for each of the subpopulations and subsequently to help develop restoration and protection actions for different areas relevant to summer chum.

More recently, HCCC staff assembled detailed information on all of the restoration actions that have been implemented, or soon to be, in each of the natal watersheds and adjacent estuaries. The information included type of action (e.g., levee removal, placement of logjams, riparian plantings, channel reconfiguration, sediment removal) and the scale and intensity of the action treatment. This information was then used to model the expected outcomes for the actions. The modeling procedures essentially provide a set of hypotheses about action effectiveness, as well as expected outcomes represented as performances of the summer chum subpopulations. To model the 2014 baseline, all of the relevant actions to each subpopulation were modeled together. The model produced the projected subpopulation performance that would be expected 100 years into the future (after all action aspects have fully matured).

The reader should note here that the modeled 2014 baseline outcome is not what is expected in 2014 in terms of subpopulation performance, but essentially what would be expected in 2114 if no other actions or watershed development occurs from what was applied in the model.

Modeling outcomes were formulated to represent what would be expected under both the warm and cool phases of the PDO. These results were used to compare to viability curves for each subpopulation under a no climate change scenario and 5 and 10 percent increases in variability associated with climate change.

It should be noted that all of the modeling results have been scaled so they can be directly compared to empirical data on population and subpopulation production levels and estimation of S-R parameters (as outlined in Carmichael and Taylor 2009 and NMFS 2012).

7.2 Identification of Performance Gaps

Modeling results for the two populations and their extant subpopulations are presented in Tables 6-8. The tables provide estimates of intrinsic productivity and equilibrium abundance for each of the four baselines described above, together with viability abundance thresholds (with and without climate change) to achieve negligible risk of extinction. We use the equilibrium abundance parameter here instead of capacity so that we can sum up the values for the subpopulations to get the total equilibrium abundance for each population, recognizing that these totals do not include fish returning to watersheds where reintroductions have occurred. It is incorrect to simply sum up subpopulation capacity values to derive population level capacities. Also, reporting the results here with equilibrium abundance instead of capacity provides a simpler, less abstract metric for planners to use in comparing modeling results to empirical data on observed run sizes. (We remind the reader that the equilibrium abundance value is equal to the geometric mean of expected abundance.²²) The productivity values shown for the populations are weighted averages from the subpopulations, weighted by equilibrium abundance. Results in the tables were formulated for an exploitation rate of 0 percent; thresholds would be somewhat higher when harvest exploitation rate is considered.

We estimated equilibrium abundance for both the warm and cool phases of the PDO for the two populations and for the eight extant subpopulations. A comparison of the equilibrium abundance (NEQ) during the warm or cool PDO phase for each scenario (2001 Base, 2001BaseBO, and 2014 BaseBO) versus the viability abundance threshold provides a measure of the estimated gap between the current performance during the warm or cool PDO phase and viability for a population or subpopulation.

Table 6 provides results for the Hood Canal and SJDF populations. Under the warm phase of the PDO, the 2001 baseline with buildout for the Hood Canal population (NEQ = 3,677) was projected to be substantially below the viability threshold for all three climate change conditions (NEQ = 5,817, 7,566, or 9,487). However, the 2014 baseline with buildout (NEQ = 8,012) was projected to be higher than the threshold for both no climate change (NEQ = 5,478) and a 5 percent climate change condition (NEQ = 7,272). This scenario does not achieve the viability threshold with a 10 percent climate change condition (NEQ = 9,137).

The Hood Canal summer chum population is performing much better under the cool phase versus the warm phase of the PDO. During the cool phase of the PDO, equilibrium abundance (NEQ) for each of the four scenarios in Table 6 exceeds the viability thresholds for the Hood Canal population. The 2014 baseline with buildout (NEQ = 28,204) was projected to be substantially higher than the viability threshold for each of the climate change conditions (NEQ = 5,478; 7,272; and 9,137).

²² / For the existing data sets for the Hood Canal and SJDF populations, the arithmetic means of total recruitment or spawning escapement are approximately 35 to 40 percent higher than the geometric means.

Table 6. Modeled results for four baselines described in the text for performance of the Hood Canal and SJDF summer chum populations. Prod is the estimated intrinsic productivity and NEQ is equilibrium abundance. Thresholds expressed as abundance associated with the given productivity level for negligible risk (<5%) of extinction under three climate conditions are also shown.

Population	Scenario	Prod in PDO phase		NEQ in PDO phase		Viability abundance threshold with climate change		
		Warm	Cool	Warm	Cool	0% chg	5% chg	10% chg
Hood Canal	Historic	28.3	30.0	17,693	62,476	5,478	7,272	9,137
	2001 Base	15.4	16.3	5,152	19,682	5,591	7,350	9,204
	2001 BaseBO	11.4	12.4	3,677	14,892	5,817	7,556	9,487
	2014 BaseBO	19.8	20.5	8,012	29,291	5,478	7,272	9,137
SJDF	Historic	21.2	30.0	4,386	17,632	3,721	4,514	5,242
	2001 Base	4.8	8.7	775	3,398	4,609	5,892	7,121
	2001 BaseBO	3.2	6.0	401	2,560	4,353	5,429	6,469
	2014 BaseBO	17.6	24.0	2,644	10,168	3,721	4,514	5,242

For the SJDF population and with the warm phase of the PDO in effect, the gap between the 2001 baseline with buildout is much greater than it is for the Hood Canal population under all three climate conditions. The 2001 baseline with buildout for the SJDF population (NEQ = 401) was projected to be substantially below the viability threshold for all three climate change conditions (NEQ = 4,353; 5,429; or 6,469). The 2014 baseline with buildout (NEQ = 2,644) is improved, but was also projected to be lower than the viability thresholds (Table 6). We would note that if natural-origin summer chum that are now spawning in Chimacum Creek after being reintroduced there are incorporated into the numbers (see Figure 7), the SJDF population would more closely approach the viability thresholds.

The SJDF summer chum population is also performing better under the cool phase versus the warm phase of the PDO. During the cool phase of the PDO, for the 2014 baseline with buildout scenario, equilibrium abundance (NEQ = 10,029) was projected to be substantially higher than the viability threshold for each of the climate change conditions (NEQ = 3,721; 4,514, and 5,242) (Table 6). As a result of the habitat protection and restoration actions that have taken place in SJDF watersheds, there has been a marked improvement over the 2001 baseline (NEQ = 2,711) or 2001 baseline with buildout (NEQ = 2,420) scenarios which did not achieve the viability thresholds.

To further illustrate how the modeling results for each of the four scenarios compare to the viability thresholds, we approximated capacity for each population (using results in Tables 6 and 7 and methods described in Section 4.1) and plotted the population parameters (intrinsic productivity and capacity) with the viability curves for the three climate change conditions (Figures 34 and 35). During the warm PDO phase, the Hood Canal population under the 2014 baseline with buildout is projected to be slightly above the viability curve with a 5 percent climate change in effect, while the performance of the SJDF population is seen to be substantially below the current climate viability curve. When performance from reintroduced subpopulations is included, the situation would be improved for both populations, but particularly for the SJDF population where the abundance of summer chum in Chimacum is contributing substantially. During the cool PDO phase, both populations were projected to be well above the 10 percent climate change viability curves under the 2014 baseline with buildout. The reader should note that the results shown in the figures are with no fishery exploitation. The thresholds would be moved up and to the right when accounting for fishery effects.

Table 7 provides modeling results for the six extant subpopulations belonging to the Hood Canal population; results are presented graphically in Figures 36-41. Under the warm phase of the PDO, the 2001 baseline with buildout for the Big and Little Quilcene rivers subpopulation was projected to be essentially right at the viability threshold with no climate change but lower than the threshold for the 5 and 10 percent climate change conditions. The 2014 baseline with buildout was projected for the Big and Little Quilcene rivers subpopulation to be substantially higher than the threshold for all climate change conditions. Each of the other five extant subpopulations was projected to be below the threshold in the 2001 baseline with buildout and the 2014 baseline with buildout. Of these five subpopulations, the Dosewallips subpopulation was projected to perform the best compared to its viability threshold (smallest gap). The other four subpopulations were projected to perform poorly (largest gaps) during the warm phase of the PDO.

The Hood Canal summer chum subpopulations are performing much better under the cool phase versus the warm phase of the PDO. Under the cool phase of the PDO, each of the subpopulations, except Lilliwaup, achieves their viability abundance threshold under the 0 percent climate change condition. The Union subpopulation achieves the threshold under the 5 percent climate change condition and the Duckabush, Dosewallips, and Quilcene subpopulations achieve their thresholds under the 5 and 10 percent climate change conditions (Table 7, Figures 36-41).

It is noteworthy that only the Dosewallips and Quilcene, the two northernmost subpopulations in Hood Canal, achieve their abundance viability thresholds under historic conditions (i.e., pristine habitat) during the warm phase of the PDO (Table 7, Figures 36-41). The Lilliwaup Creek subpopulation is seen to not achieve the abundance threshold even during the cool PDO phase. These results suggest that there are carryover benefits to maintaining abundance from the cool to the warm phase of the PDO, thereby ameliorating some portion of the downturn experienced during the warm phase. The results also suggest that spatial structure and diversity may be particularly important to the Hood Canal population, whereby asynchrony among the subpopulations during the warm phase is important to the overall population's health (as discussed at the end of Section 3.2 Abundance Patterns). We would also note that the capacity (and equilibrium abundance) parameter for Lilliwaup Creek is likely underestimated for the cool PDO phase; the data used were for brood years 1999 to 2006, which likely were strongly affected by extremely low escapements during the prior years. Summer chum production in Lilliwaup Creek was much higher in the 1970s prior to the warm phase of the PDO than it was after about year 2000 (Figure 9), demonstrating that the stream is capable of producing much larger run sizes than suggested by Table 7.

Table 7. Modeled results for four baselines described in the text for performance of the six extant subpopulations of the Hood Canal summer chum population. Prod is the estimated intrinsic productivity and NEQ is equilibrium abundance. Thresholds expressed as abundance associated with the given productivity level for negligible risk (<5%) of extinction under three climate conditions are also shown.

Sub-population	Scenario	Prod in PDO phase		NEQ in PDO phase		Viability abundance threshold with climate change		
		Warm	Cool	Warm	Cool	0% chg	5% chg	10% chg
Union	Historic	30.0	30.0	961	2,710	1,047	1,350	1,625
	2001 Base	15.8	15.8	571	1,611	1,047	1,349	1,627
	2001 BaseBO	12.5	12.5	497	1,400	1,071	1,355	1,704
	2013 BaseBO	13.1	13.1	512	1,443	1,059	1,336	1,653
Lilliwaup	Historic	19.4	30.0	127	492	1,252	1,484	1,885
	2001 Base	3.1	6.2	66	305	2,376	3,004	4,349
	2001 BaseBO	2.0	3.9	44	253	2,615	3,325	5,015
	2013 BaseBO	1.2	2.3	14	214	2,615	3,325	5,015
Hamma	Historic	17.0	30.0	417	2,776	926	1,256	1,419
	2001 Base	3.9	8.6	209	1,590	1,186	1,511	2,033
	2001 BaseBO	3.8	8.3	203	1,557	1,245	1,603	2,157
	2013 BaseBO	3.8	8.3	203	1,557	1,245	1,603	2,157
Duck	Historic	30.0	30.0	625	3,924	994	1,359	1,500
	2001 Base	14.4	15.3	398	2,507	1,009	1,431	1,505
	2001 BaseBO	12.7	13.5	368	2,316	995	1,333	1,544
	2013 BaseBO	13.0	13.8	373	2,349	1,000	1,366	1,533
Dose	Historic	16.3	30.0	1,707	4,689	767	1,003	1,118
	2001 Base	2.8	10.0	864	3,147	1,373	1,732	2,304
	2001 BaseBO	2.2	7.9	724	3,001	1,396	1,779	2,391
	2013 BaseBO	3.8	13.4	984	3,244	1,053	1,290	1,660
Quil	Historic	30.0	30.0	13,855	47,885	1,887	2,367	2,788
	2001 Base	20.0	20.0	3,044	10,522	1,887	2,367	2,788
	2001 BaseBO	15.5	15.5	1,842	6,365	1,897	2,356	2,840
	2013 BaseBO	24.0	24.0	5,926	20,483	1,887	2,367	2,788

Table 8 provides modeling results for the two extant subpopulations belonging to the SJDF population; results are presented graphically in Figures 42-43. Under the warm phase of the PDO, the 2001 baseline with buildout for both subpopulations was projected to be substantially below the viability thresholds for all three climate change conditions. The Jimmycomelately subpopulation was projected to be extirpated (NEQ = 0) in the 2001 baseline with buildout for all climate change conditions. With the 2014 baseline with buildout (NEQ = 785), the Jimmycomelately subpopulation was projected to be slightly below the threshold for no climate change (NEQ = 842)—and the gap increased under the 5 and 10 percent climate change conditions. The Salmon and Snow creeks subpopulation for the 2014 baseline with buildout (NEQ = 1,859) was projected to be above the threshold with no climate change (NEQ = 1,688) and slightly below the threshold (NEQ = 1,943) with a 5 percent climate change condition. The gap was more substantial with the 10 percent climate change condition (NEQ = 2,364).

The SJDF summer chum subpopulations are performing much better under the cool phase than during the warm phase of the PDO. Under the cool phase of the PDO, the Salmon and Snow creeks subpopulation exceeds the abundance thresholds at 0 percent climate change condition for each of the

four scenarios and at 5 and 10 percent climate change conditions for the 2014 baseline with buildout. The Jimmycomelately subpopulation exceeds the abundance threshold for the 2014 baseline with buildout at 0 percent, but not at 5 or 10 percent climate change conditions (Table 8, Figures 42 and 43).

It is noteworthy that both the Salmon/Snow and Jimmycomelately subpopulations in SJDF achieve their abundance viability thresholds under historic conditions (i.e., pristine habitat) during the warm phase of the PDO (Table 8, Figures 42-43).

It bears noting that when equilibrium abundance does not exceed a risk threshold for viability that it is not a foregone conclusion that the production unit will be extirpated. The threshold means that negligible risk exists and, when the threshold is not exceeded, risk is elevated. Also, the relationship between performance during the warm and cool phases is uncertain. There may be a carryover effect from the cool phase of the PDO when abundance is bolstered to the warm phase of the PDO when abundance drops; if a carryover effect occurs, it may provide some measure of resilience during the warm phase.

Table 8. Modeled results for four baselines described in the text for performance of the two extant subpopulations of the SJDF summer chum population. Prod is the estimated intrinsic productivity and NEQ is equilibrium abundance. Thresholds expressed as abundance associated with the given productivity level for negligible risk (<5%) of extinction under three climate conditions are also shown.

Sub-population	Scenario	Prod in PDO phase		NEQ in PDO phase		Viability abundance threshold with climate change		
		Warm	Cool	Warm	Cool	0% chg	5% chg	10% chg
SalmSnow	Historic	24.0	30.0	3,312	16,485	1,688	1,943	2,364
	2001 Base	6.3	9.4	550	2,863	1,803	2,144	2,639
	2001 BaseBO	5.6	8.3	401	2,106	1,852	2,221	2,726
	2013 BaseBO	22.4	24.0	1,859	9,264	1,688	1,943	2,364
JCL	Historic	12.5	30.0	1,074	1,148	814	946	1,114
	2001 Base	1.0	4.9	225	535	932	1,250	1,580
	2001 BaseBO	0.8	3.7	0	455	932	1,250	1,580
	2013 BaseBO	6.2	24.0	785	904	842	1,090	1,236

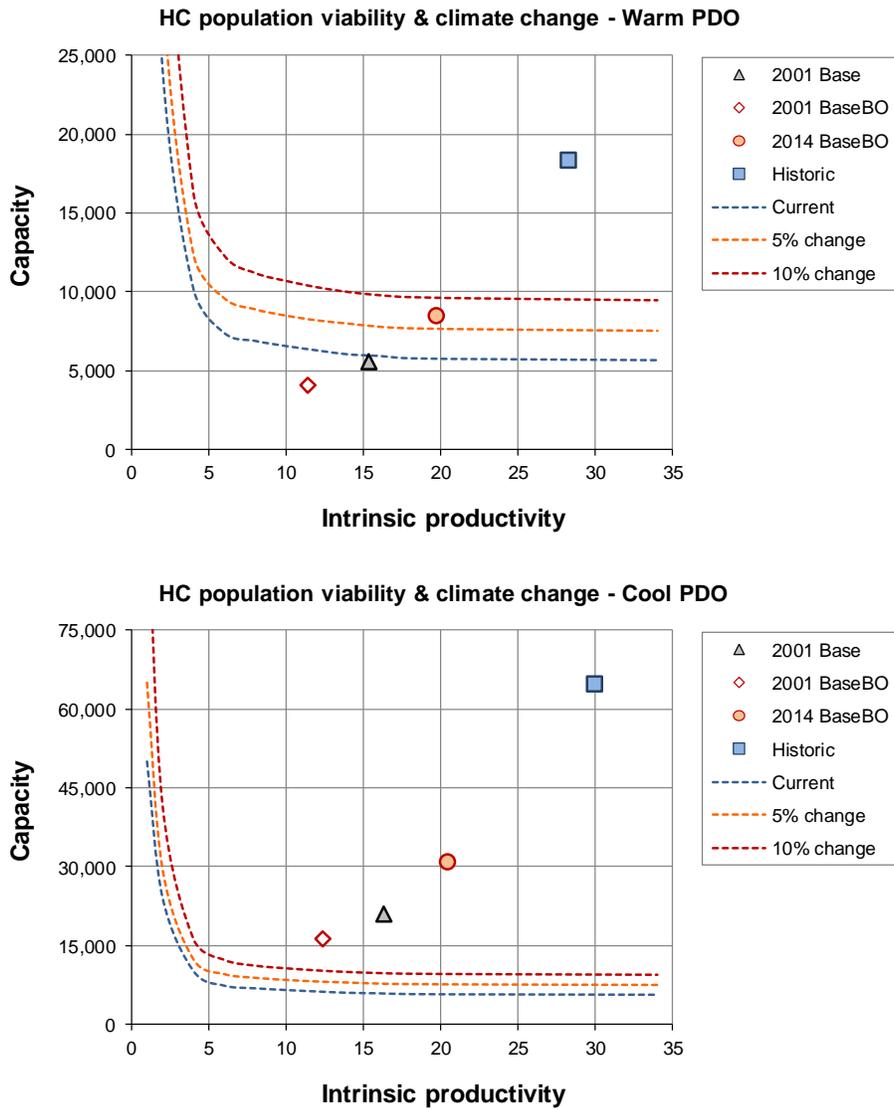


Figure 34. Modeled results for four baseline scenarios (described in the text) showing population performance parameters (intrinsic productivity and capacity) relative to viability curves (5 percent extinction risk) for the Hood Canal summer chum population with variation increased by 5 and 10 percent to reflect future climate change under the warm (top) and cool (bottom) phases of the PDO.

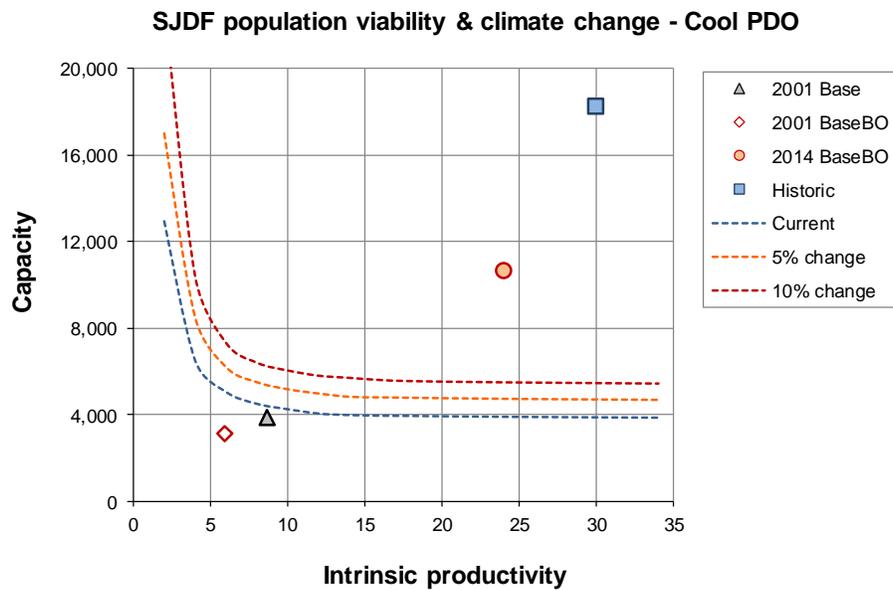
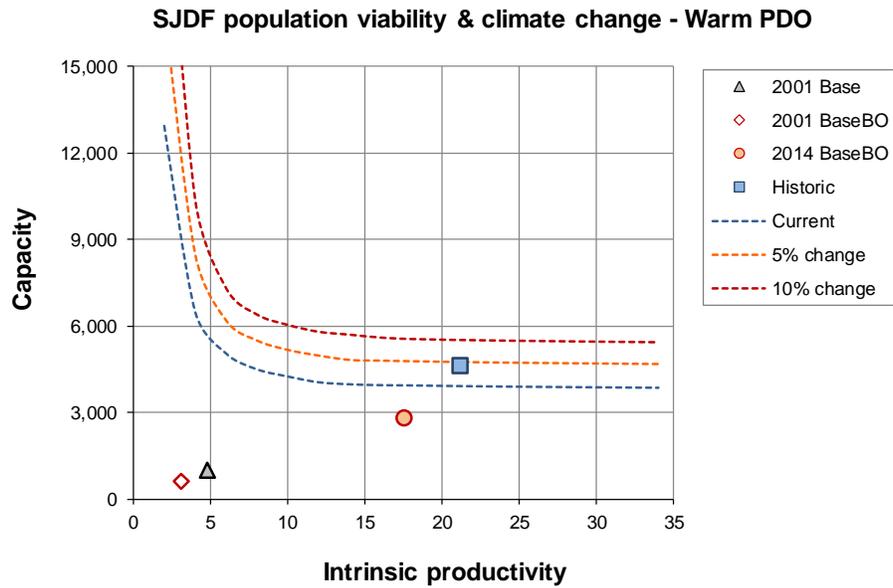


Figure 35. Modeled results for four baseline scenarios (described in the text) showing population performance parameters (intrinsic productivity and capacity) relative to viability curves (5 percent extinction risk) for the SJDF summer chum population with variation increased by 5 and 10 percent to reflect future climate change under the warm (top) and cool (bottom) phases of the PDO.

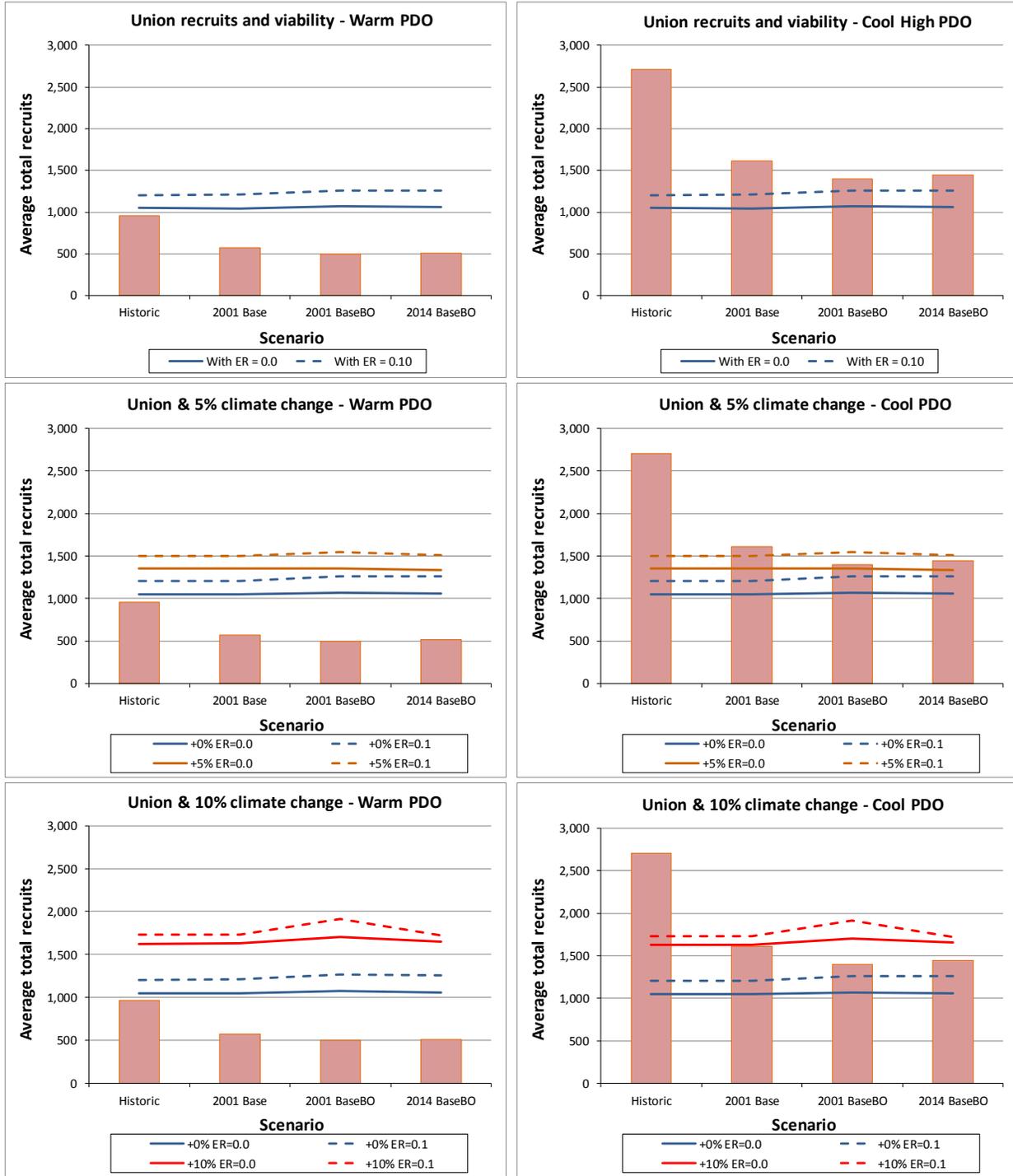


Figure 36. Population performance as measured by projected equilibrium abundance for Union River summer chum for baseline reference conditions: historic, 2001, 2001 with future buildout, and 2014 with future buildout. Viability thresholds for abundance are also plotted by assuming the reference condition productivity and an associated capacity. The viability thresholds are shown with ERs of 0 and 10%.

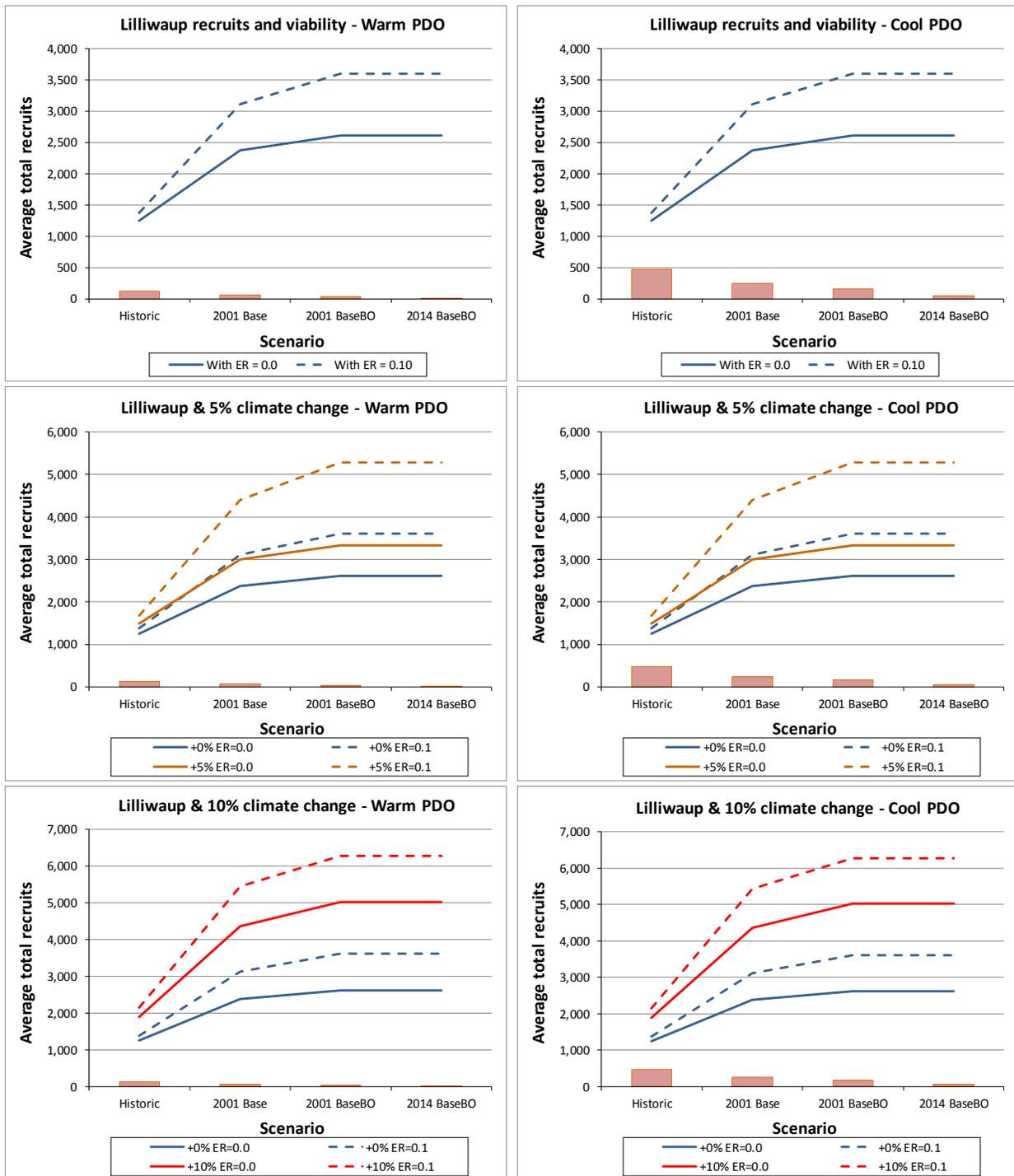


Figure 37. Population performance as measured by projected equilibrium abundance for Lilliwaup Creek summer chum for baseline reference conditions: historic, 2001, 2001 with future buildout, and 2014 with future buildout. Viability thresholds for abundance are also plotted by assuming the reference condition productivity and an associated capacity. The viability thresholds are shown with ERs of 0 and 10%.

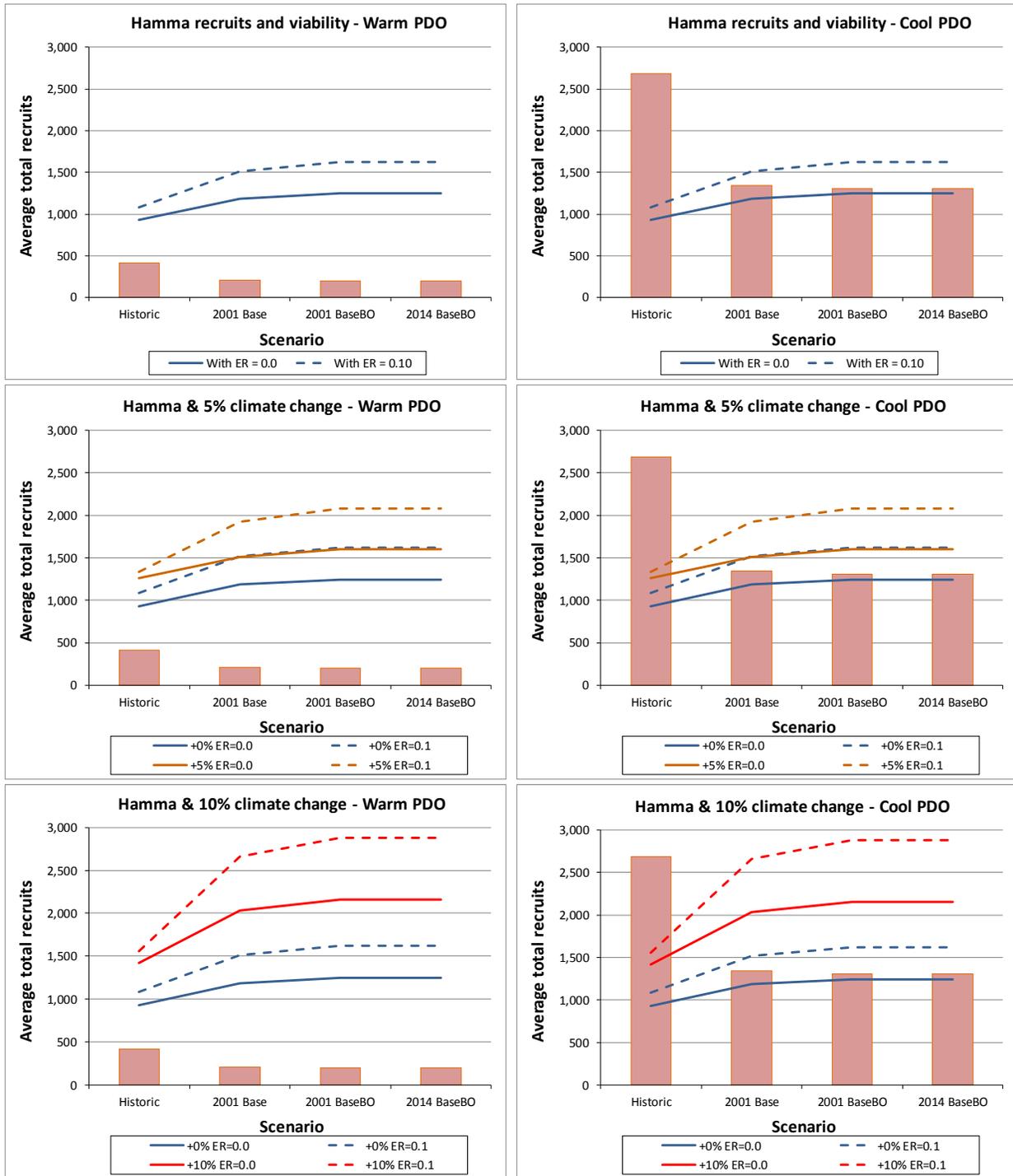


Figure 38. Population performance as measured by projected equilibrium abundance for Hamma Hamma River summer chum for baseline reference conditions: historic, 2001, 2001 with future buildout, and 2014 with future buildout. Viability thresholds for abundance are also plotted by assuming the reference condition productivity and an associated capacity. The viability thresholds are shown with ERs of 0 and 10%.

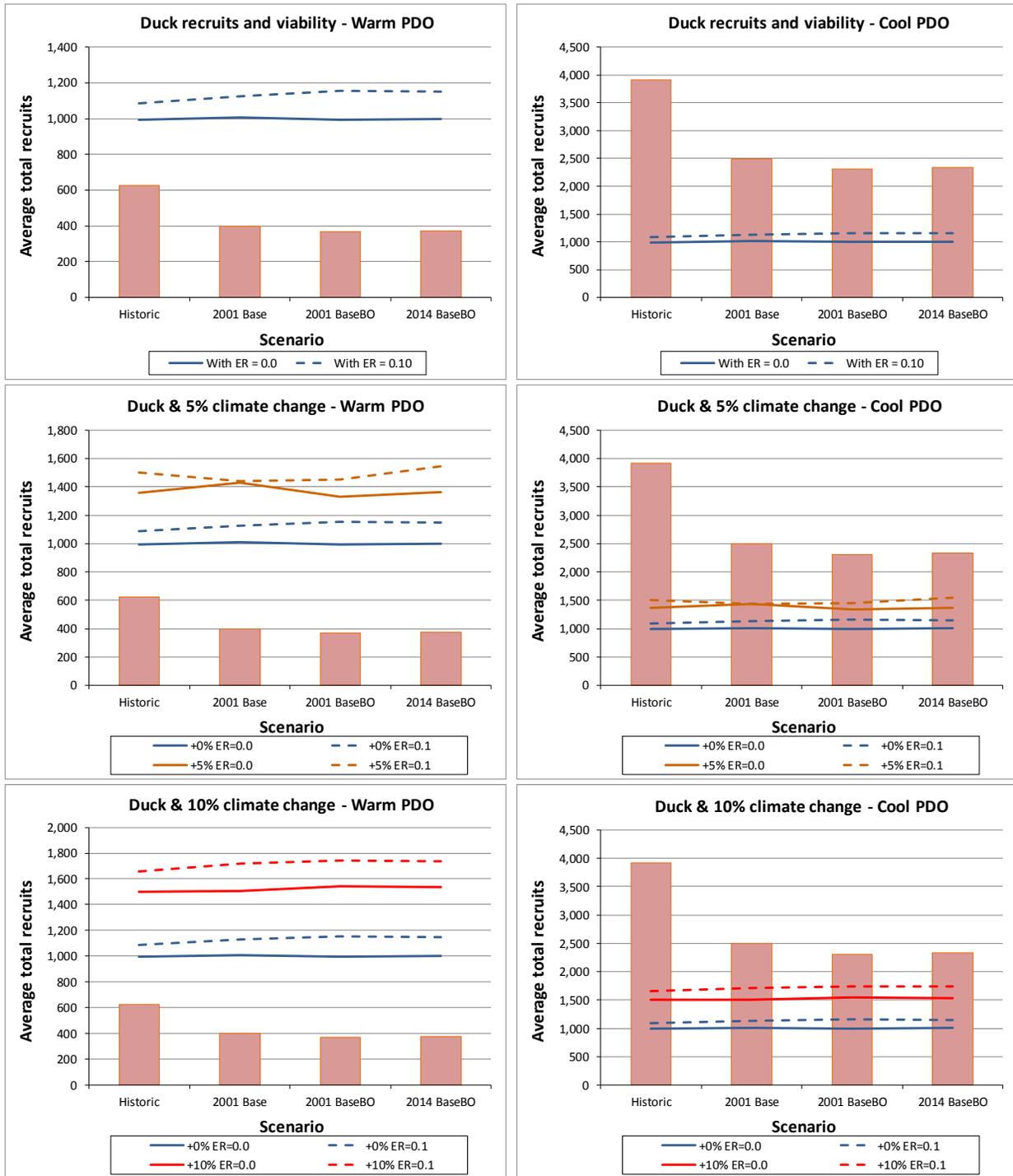


Figure 39. Population performance as measured by projected equilibrium abundance for Duckabush River summer chum for baseline reference conditions: historic, 2001, 2001 with future buildout, and 2014 with future buildout. Viability thresholds for abundance are also plotted by assuming the reference condition productivity and an associated capacity. The viability thresholds are shown with ERs of 0 and 10%.

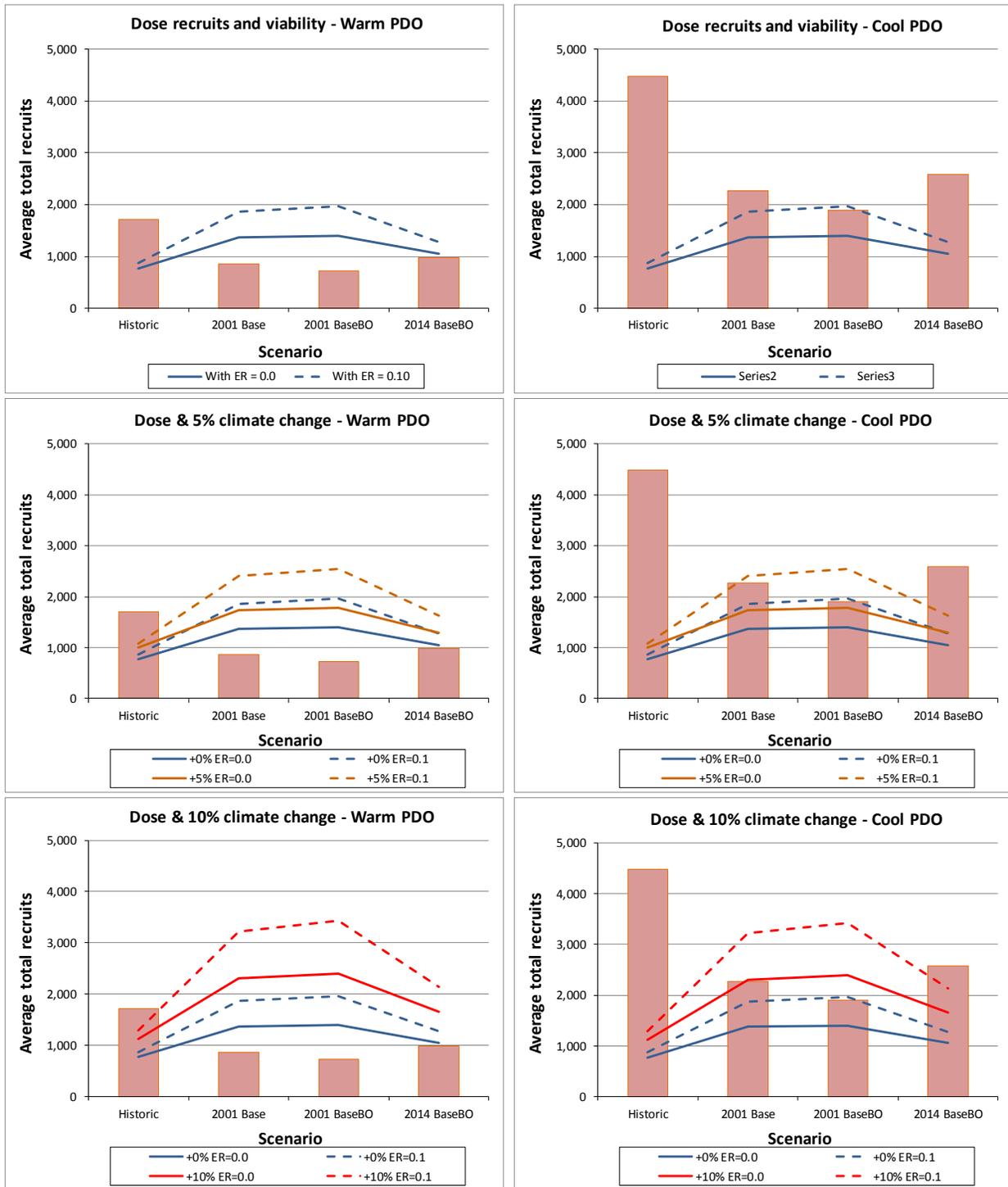


Figure 40. Population performance as measured by projected equilibrium abundance for Dosewallips River summer chum for baseline reference conditions: historic, 2001, 2001 with future buildout, and 2014 with future buildout. Viability thresholds for abundance are also plotted by assuming the reference condition productivity and an associated capacity. The viability thresholds are shown with ERs of 0 and 10%.

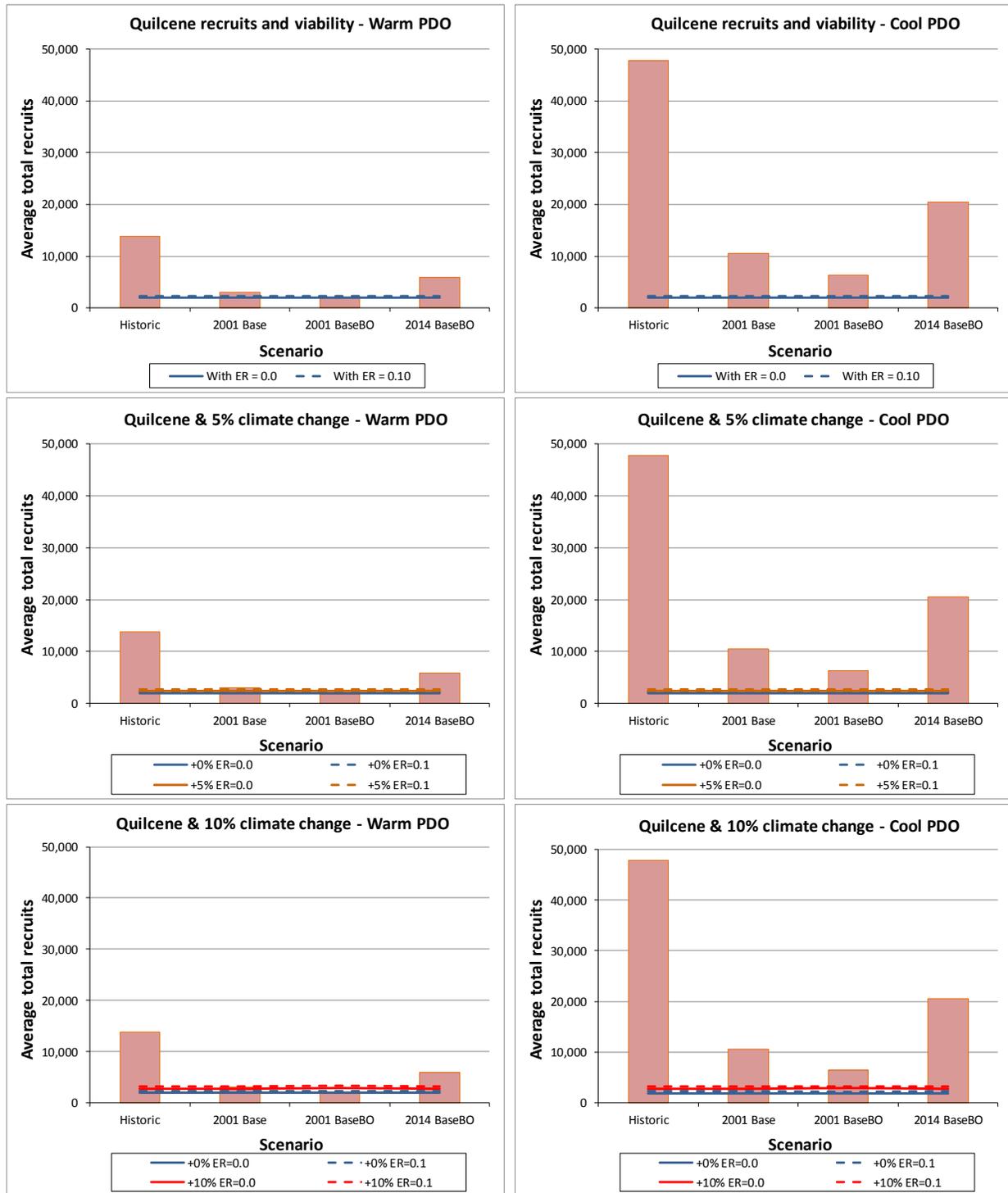


Figure 41. Population performance as measured by projected equilibrium abundance for Big and Little Quilcene rivers (combined) summer chum for baseline reference conditions: historic, 2001, 2001 with future buildout, and 2014 with future buildout. Viability thresholds for abundance are also plotted by assuming the reference condition productivity and an associated capacity. The viability thresholds are shown with ERs of 0 and 10%.

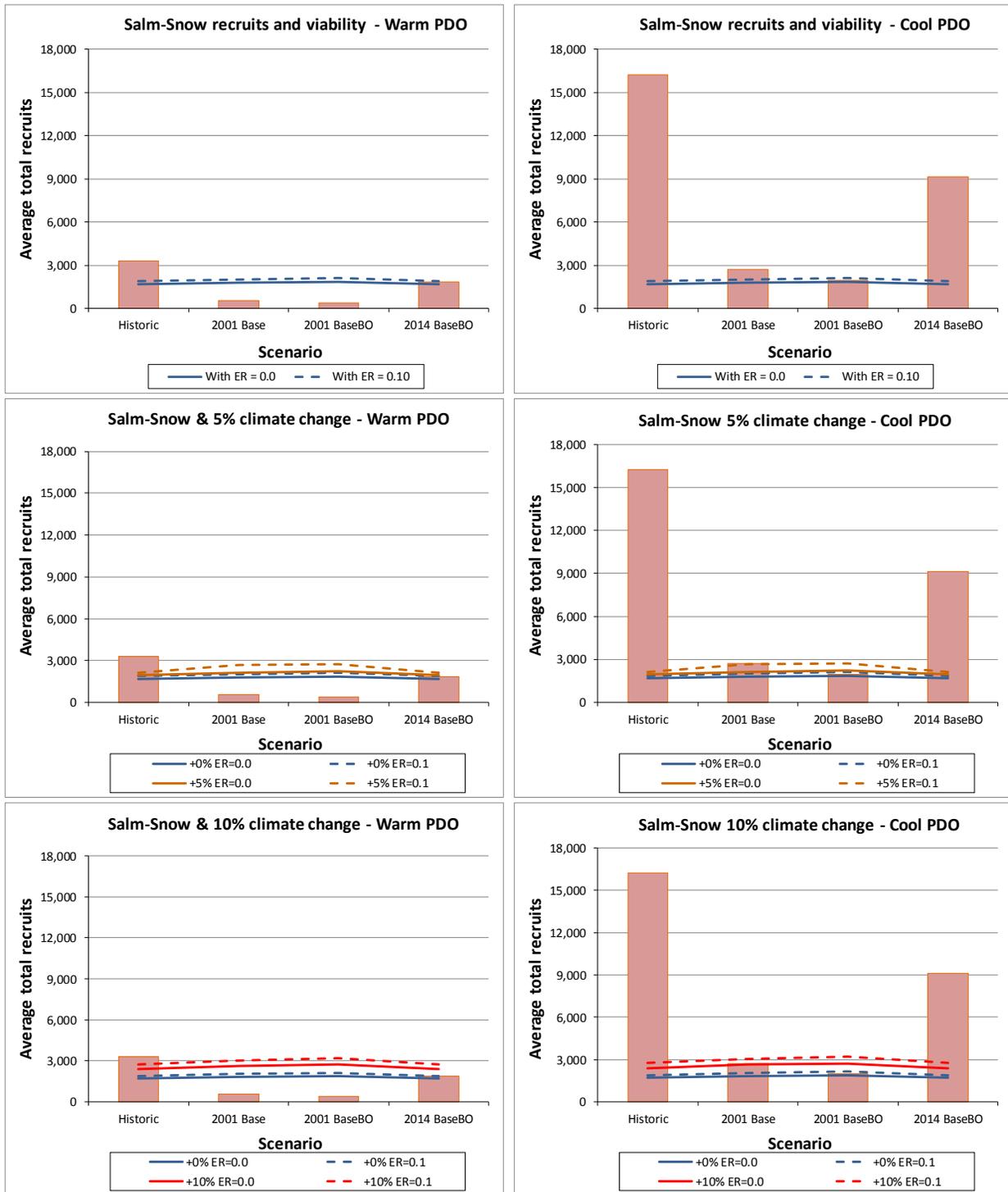


Figure 42. Population performance as measured by projected equilibrium abundance for Salmon and Snow creeks (combined) coho for baseline reference conditions: historic, 2001, 2001 with future buildout, and 2014 with future buildout. Viability thresholds for abundance are also plotted by assuming the reference condition productivity and an associated capacity. The viability thresholds are shown with ERs of 0 and 10%.

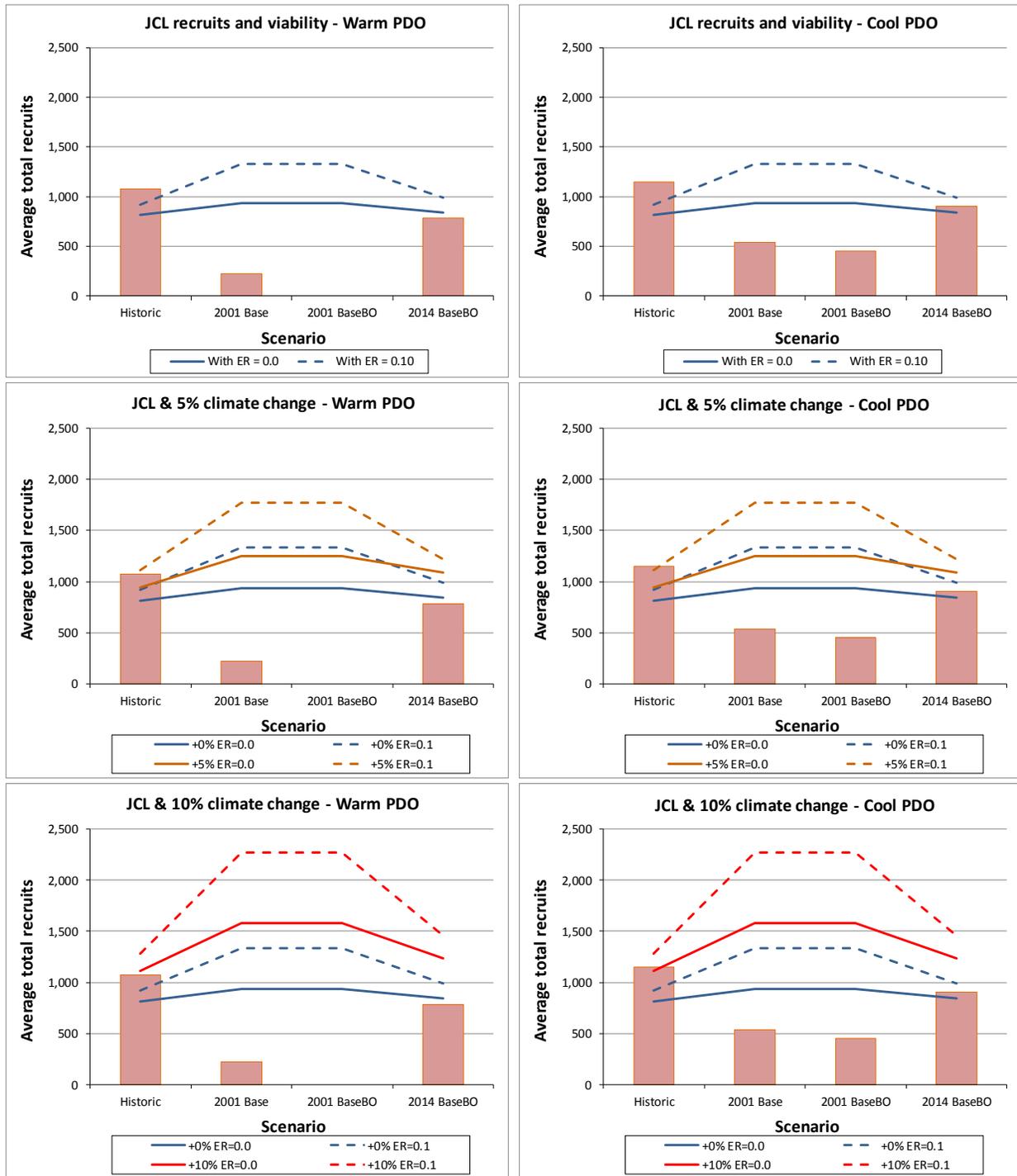


Figure 43. Population performance as measured by projected equilibrium abundance for Jimmycomelately Creek summer chum for baseline reference conditions: historic, 2001, 2001 with future buildout, and 2014 with future buildout. Viability thresholds for abundance are also plotted by assuming the reference condition productivity and an associated capacity. The viability thresholds are shown with ERs of 0 and 10%.

8.0 Recommendations

The following seven recommendations are offered with respect to updating recovery goals, prioritizing future habitat restoration and protection actions, addressing harvest goals, continuing reintroduction efforts, and continuing monitoring and evaluation for the Hood Canal Summer Chum ESU. The recommendations are presented in a way for the reader to follow a progressive logic, such that after recommendation one is given, subsequent recommendations then build on preceding recommendations.

1. We recommend that the quantitative recovery goals for productivity and abundance for both populations be updated with results of the viability analysis presented in this paper. It is recognized that recovery goals and population viability criteria are to be an adaptively managed part of the recovery plan and that as new data and modeling results become available, the recovery goals and population viability criteria would be refined over time (WDFW and PNPTT 2000; PNPTT and WDFW 2003; HCCC 2005; NMFS 2007).

The viability thresholds for negligible risk of extinction (i.e., <5% risk over a 100-year period) are most correctly portrayed as viability curves, representing a combination of capacity and productivity values (see, e.g., Figure 29). For each population, approximate values for capacity and a corresponding productivity associated with average spawning escapement viability thresholds are given in Table 9 at three exploitation rates (0, 10, and 20 percent). Two average spawning escapements are shown for each case, the arithmetic mean (AM), which is skewed high (by approximately 35 to 40 percent) due to the lognormal distribution of observed escapements, and the geometric mean (GM), which is equivalent to what this paper refers to as equilibrium abundance.²³

We recommend that the equilibrium abundance (i.e., geometric mean) values for minimum average spawning escapements be used to measure whether the revised viability thresholds (recovery goals) have been achieved for the Hood Canal and SJDF summer chum populations. One reasonable set of geometric mean escapements or viability thresholds are shown in Table 9 for an intrinsic productivity of 14 and the corresponding estimates of capacity for each population. Using other reasonable combinations of intrinsic productivity and capacity, though, would provide other reasonable estimates of equilibrium abundance and viability thresholds. For example, the arithmetic mean spawner escapements shown in Table 5 as viability thresholds for the revised analysis (2013 update) could be reduced by 35-40 percent to provide estimates of geometric mean spawning escapements for the range of intrinsic productivity and capacity values in Table 5.

We use the equilibrium abundance parameter so that we can sum up the values for the subpopulations to get the total equilibrium abundance for each population, recognizing that these totals do not include fish returning to watersheds where reintroductions have occurred. Also, reporting the results here with equilibrium abundance instead of capacity provides a simpler, less abstract metric for planners to use in comparing modeling results to empirical data on observed run sizes.

²³ / The arithmetic mean was used in Sands et al. (2009).

Table 9. Estimated values for capacity (Cap) associated with a productivity of 14 that define viability thresholds (5% risk) for three exploitation rates (0, 10, and 20%), and expected average spawning escapements that would be observed at those thresholds. All of the values shown are derived with the VRAP model as done in Sands et al. (2009). Minimum average spawning escapements are presented both as the arithmetic mean (AM) and the geometric mean (GM), which is equivalent to equilibrium abundance. The table lists capacity values at an intrinsic productivity of 14, which is consistent with a reasonable estimate of expected productivity that exceeds viability.²⁴

Population	ER = 0%			ER = 10%			ER = 20%		
	Cap	AM esc	GM esc	Cap	AM esc	GM esc	Cap	AM esc	GM esc
Hood Canal	6,100	8,100	5,700	7,500	8,900	6,200	8,500	8,800	6,200
SJDF	4,000	5,000	3,700	4,800	5,400	4,000	5,400	5,300	3,900

2. We recommend that the recovery thresholds for abundance (or capacity and productivity) viability take into account the impacts of climate and ocean regimes on the performance and survival of summer chum in the Hood Canal ESU. The recovery thresholds should be based on performance during the warm phase regime of the Pacific Decadal Oscillation (PDO) when summer chum performance is adversely impacted and risk of extinction is increased. The PDO is currently in the cool phase and it may remain in this state for several more years.

Table 6, presented earlier in the report, provided projected results based on EDT modeling for the two populations, identifying equilibrium abundance values for each of four baseline scenarios, including the 2014 baseline with buildout (see Section 7). A comparison of the equilibrium abundance (NEQ) during the warm or cool PDO phase for each scenario (2001 Base, 2001BaseBO, and 2014 BaseBO) versus the viability abundance threshold provides a measure of the estimated gap between the current performance of a population during the warm PDO or cool phase and viability (Figures 34 and 35). During the warm PDO phase, the Hood Canal population under the 2014 baseline with buildout is projected to be slightly above the viability curve with a 5 percent climate change in effect, while the performance of the SJDF population is seen to be substantially below the current climate viability curve. During the cool PDO phase, both populations were projected to be well above the 10 percent climate change viability curves under the 2014 baseline with buildout. When performance from reintroduced subpopulations is included, the situation would be improved for both populations, but particularly for the SJDF population where the abundance of summer chum in Chimacum is contributing substantially. The results shown in Table 6 were formulated for an exploitation rate of 0 percent; thresholds would be somewhat higher when exploitation is considered.

Performance after future warm phases have been experienced by the populations for at least 8 years (i.e., two summer chum generations) is advised as the minimum amount of time for evaluating population status as affected by restoration actions. As a surrogate for measuring performance during future warm phases of the PDO, EDT modeling, or some similar method, can be applied with procedures used to produce this paper. One value of modeling is that it enables projections to be made of future performance corresponding to when the restoration actions will be fully matured.

²⁴ / The capacity values generally flatten out in the viability curves with productivity values in the vicinity of 14 to 16, as seen in Figures 34 and 35, so larger values of productivity lead to very little change in the capacity threshold.

Ultimately, however, population performance that results from the actions will need to be validated through actual observations of returning adults.

3. We recommend that the viability recovery goals should take into account the impacts of climate change on the performance and survival of summer chum in the Hood Canal ESU. Recovery goals should account for at least a 5 percent increase in the variation of summer chum population performance. We assume here that summer chum population performance variation is approximately equal to the percent increase in environmental variation resulting from climate change. How population performance is correlated with environmental variation is uncertain and will need some level of assessment and monitoring. As time proceeds and estimates of variation improve, this recommended value should be reviewed and updated.

Three climate change conditions (no change and 5 and 10 percent increases in environmental variation) and the estimated viability thresholds for summer chum population performance that result from them are shown in Table 10. The 'no climate change' condition is the same as that presented in Table 9.

Table 10. The estimated impact of climate change on estimated values for capacity (Cap) of the Hood Canal and Strait of Juan de Fuca (SJDF) summer chum populations associated with a productivity of 14 that define viability thresholds (5% risk) for three exploitation rates (0, 10, and 20%), and expected average spawning escapements that would be observed at those thresholds. Results are shown for three climate conditions: no change and 5 and 10 percent increases in environmental variation. All of the values shown are derived with the VRAP model as done in Sands et al. (2009). Minimum average spawning escapements are presented both as the arithmetic mean (AM) and the geometric mean (GM), which is equivalent to equilibrium abundance. The table lists capacity values at an intrinsic productivity of 14, which is consistent with a reasonable estimate of expected productivity that exceeds viability.²⁵

No climate change

Population	ER = 0%			ER = 10%			ER = 20%		
	Cap	AM esc	GM esc	Cap	AM esc	GM esc	Cap	AM esc	GM esc
Hood Canal	6,100	8,100	5,700	7,500	8,900	6,200	8,500	8,800	6,200
SJDF	4,000	5,000	3,700	4,800	5,400	4,000	5,400	5,300	3,900

5% climate change

Population	ER = 0%			ER = 10%			ER = 20%		
	Cap	AM esc	GM esc	Cap	AM esc	GM esc	Cap	AM esc	GM esc
Hood Canal	8,000	10,700	7,400	8,500	10,100	7,000	9,300	9,800	6,800
SJDF	4,800	6,300	4,500	5,600	6,500	4,600	6,500	6,500	4,700

10% climate change

Population	ER = 0%			ER = 10%			ER = 20%		
	Cap	AM esc	GM esc	Cap	AM esc	GM esc	Cap	AM esc	GM esc
Hood Canal	10,000	13,300	9,300	11,700	14,300	9,700	13,900	14,900	11,500
SJDF	5,900	7,800	5,500	6,700	8,100	5,600	7,600	7,900	6,300

- We recommend that habitat protection and restoration actions be done strategically and distributed among the areas affecting the Hood Canal and SJDF populations and their subpopulations. This strategy should balance the need to reduce the performance gaps for subpopulations projected to be below viability thresholds, while also addressing the continuing need to strengthen performance in the core subpopulations for the sake of bolstering overall population abundance. These core subpopulations are the Quilcene and Dosewallips in the Hood Canal population and Salmon-Snow Creek in the SJDF population. This recommendation for balancing restoration actions reflects our understanding of the importance of biological diversity, spatial structure, and population abundance and productivity to long-term viability.

It is essential that all of the Hood Canal subpopulations currently classified as extant remain productive and abundant enough to contribute to the overall population's health over the long-term. For the Hood Canal population, there is a need to improve the performance of the weaker extant subpopulations to the extent feasible. Also, some of the weaker extant subpopulations

²⁵ / The capacity values generally flatten out in the viability curves with productivity values in the vicinity of 14 to 16, as seen in Figures 34 and 35, so larger values of productivity lead to very little change in the capacity threshold.

(Duckabush, Hamma Hamma, Lilliwaup, and Union) may prove to be the more abundant production units in some years. This need should be balanced, however, with an on-going need to strengthen the core subpopulations as they currently contribute significantly to the viability of the population (i.e., they are essentially ‘holding up’ the population). These core production units would also likely be critically important for abundance-related criteria (both capacity and productivity) especially if the effects on summer chum performance of the 5 percent climate change condition are as predicted or worse than assumed under Recommendation 3. Also, it should be recognized that the core subpopulations (or stronger subpopulations) may offer the greatest potential for the co-managers to realize harvest benefits—see Recommendation 5.

It is essential that all of the SJDF subpopulations currently classified as extant remain productive and abundant enough to contribute to the overall population’s health over the long-term. For the SJDF population, the greatest need is to secure and strengthen abundance (i.e., capacity) by protecting and restoring the Salmon and Snow creeks and Jimmycomelately Creek subpopulations. To improve spatial structure and diversity, as well as capacity, attention should also be given to help ensure that the reintroduced Chimacum subpopulation is both stabilized and strengthened. In addition, it is important to learn more about the potential for summer chum production in the Dungeness River; if this area can consistently support summer chum, then the greatest potential for increasing abundance (and viability) in the SJDF population would seem to exist in this river.

Tables 7 and 8, presented earlier in the report, provide modeling results for the eight extant subpopulations, identifying equilibrium abundance values for each of the four baseline scenarios described in Section 7. Also shown are estimated abundance values corresponding to combinations of productivity and capacity that are the thresholds to achieve negligible risk. The results shown were formulated for an exploitation rate of 0 percent; thresholds would be somewhat higher when exploitation is considered. For each subpopulation, a comparison of the equilibrium abundance (NEQ) during the warm or cool PDO phase for the 2014 baseline (2014 BaseBO) scenario versus the viability abundance threshold provides a measure of the estimated gap between the current performance of a subpopulation during the warm or cool PDO phase and viability.

5. We recommend that the co-managers consider strategic ways of addressing summer chum harvest goals by applying information contained in this report. Given the current performance of the co-managers’ Base Conservation Regime (BCR), we recommend that it be retained as the primary harvest management tool toward recovery. It is particularly well suited to address fishery risk when the summer chum populations and subpopulations are at low levels, as they had been, in the vicinity of the critical abundance thresholds identified in the co-managers’ SCSCI and HCCC recovery plan. In addition, the BCR appears to provide adequate protection during both the cool and warm phases of the PDO and to the weaker subpopulations. On the other hand, the co-managers should continue their development of the basic provisions and criteria for a “Recovering” regime as identified in the SCSCI. This new regime could be used when the status of summer chum, while not recovered, is sufficient to warrant departure from the strict application of the BCR in order to relieve some of the restrictions on fisheries for other stocks and species. For example, with careful consideration and monitoring, it may be possible to increase allowable fishery exploitation rates during cool phases of the PDO and to take advantage of differences in the production capabilities of the various subpopulations while providing adequate protection to the weakest subpopulations. An excellent summer chum DNA baseline exists and additional DNA sampling of summer chum caught in the fisheries could improve understanding about how the subpopulations distribute on their return migration. Such information would be important in modifying the harvest regime.

6. We conclude that successful reintroductions into watersheds where spawning aggregations have been extirpated are likely to be critically important in light of expected climate change effects combined with expected downturns in production that will occur during warm phases of the PDO. Long-term viability will likely depend on recovering at least some of the lost spatial structure and diversity that existed prior to the recent extirpations. Reintroduction efforts did occur in Big Beef Creek from 1996 through 2003 and reintroduction efforts have been ongoing in the Tahuya River since 2003 but are scheduled to be terminated there in 2014. To secure and strengthen the relatively small subpopulations in Big Beef Creek and Tahuya River, we recommend that habitat protection and restoration efforts should be targeted in those watersheds to provide more suitable and productive habitat. Once the habitats have been restored, additional reintroduction efforts could be considered, as appropriate. We also recommend that reintroduction efforts be expanded, particularly on the Kitsap Peninsula (West Kitsap ecological diversity group), to help ensure recovery of more of the spatial structure lost in this geographic area. Any expansion of reintroduction efforts needs to carefully consider the potential for conflicts with fisheries management in the surrounding areas; potential conflicts should be minimized to the extent possible.

The Skokomish River summer chum subpopulation, determined to be extirpated in past analyses (WDFW and PNPTT 2000; Sands et al. 2009), would seem to offer some benefits through all of the viability criteria. This production unit might have been a core subpopulation historically with regard to abundance while also likely having had an important role in spatial structure and diversity. However, it also poses perhaps the largest potential for fisheries conflicts, both for treaty commercial and non-treaty recreational fisheries within the immediate vicinity of the river. We suggest that a closer examination of available information be made to better understand the pattern seen in Figure 11 as it suggests that the subpopulation is rebounding. If this is true, there may exist opportunities to strengthen the subpopulation without increasing fishery conflicts. However, we would note that the stability of the lower Skokomish River habitat is likely to worsen before it improves due to activities aimed at scouring (or removing) extensive sediment deposits as identified in the Skokomish River Chinook recovery plan (SIT and WDFW 2010). We recommend that efforts be continued to monitor the situation with regard to summer chum production and we defer any further recommendation regarding the Skokomish subpopulation to a future re-evaluation of the ESU as a whole.

7. We recommend that the regular and comprehensive monitoring and evaluation of the SJDF and Hood Canal summer chum populations that is being done by the State and Tribal co-managers be continued. The co-managers annually collect and evaluate information on spawner escapement, harvest, run size, age composition, natural-origin vs. supplementation-origin composition, and genetics for summer chum populations and subpopulations throughout the ESU (WDFW and PNPTT 2007, PNPTT and WDFW 2014). The analysis in the recovery plan (HCCC 2005, NMFS 2007) and in this paper was only possible because this comprehensive monitoring and evaluation was done.

The information on summer chum population and subpopulation performance presented in this document is based on observed variability in the returns of salmon related to the escapement. This estimate of variability will likely change over time as environmental conditions change and recovery actions are taken. As such, viability goals should be re-evaluated on a regular basis with new analyses and methods incorporated as they become available.

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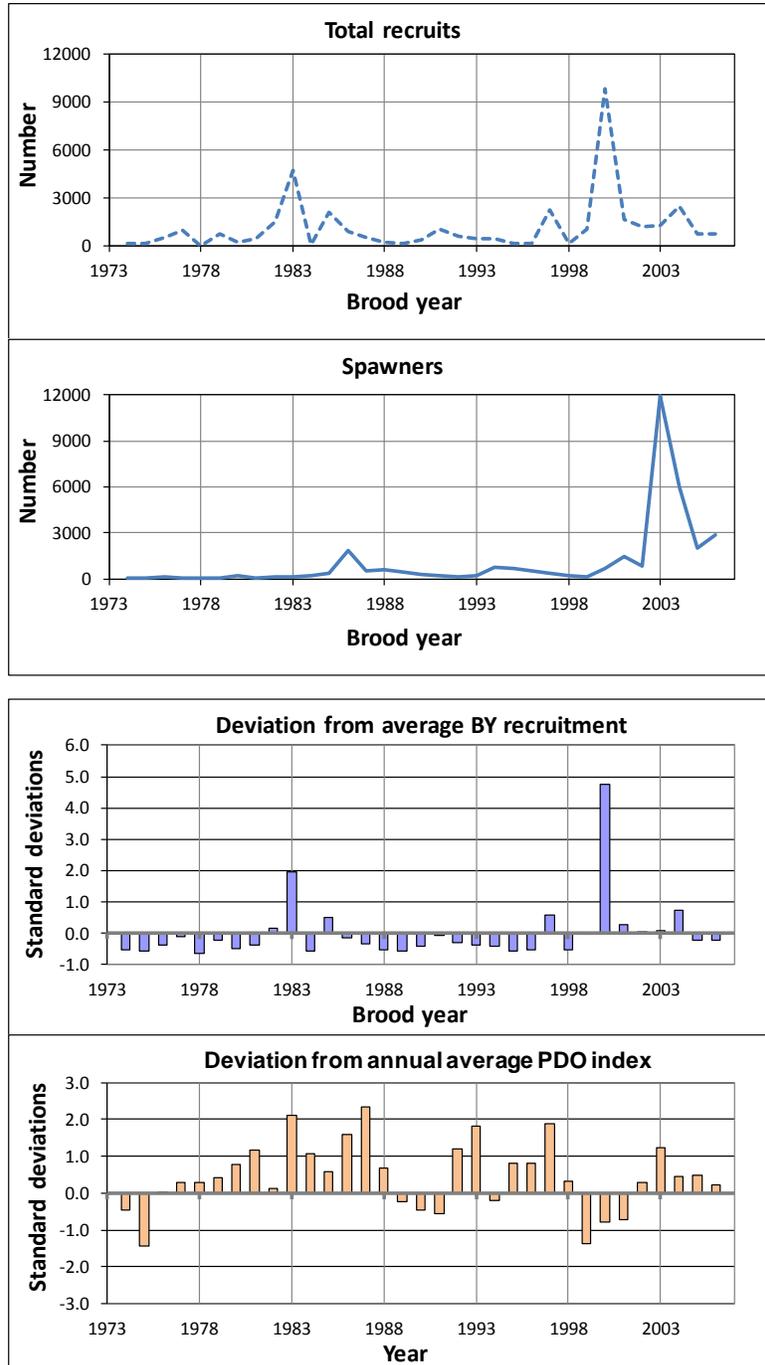
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Appendix A – Patterns of Effects of Regime Shifts on Subpopulations

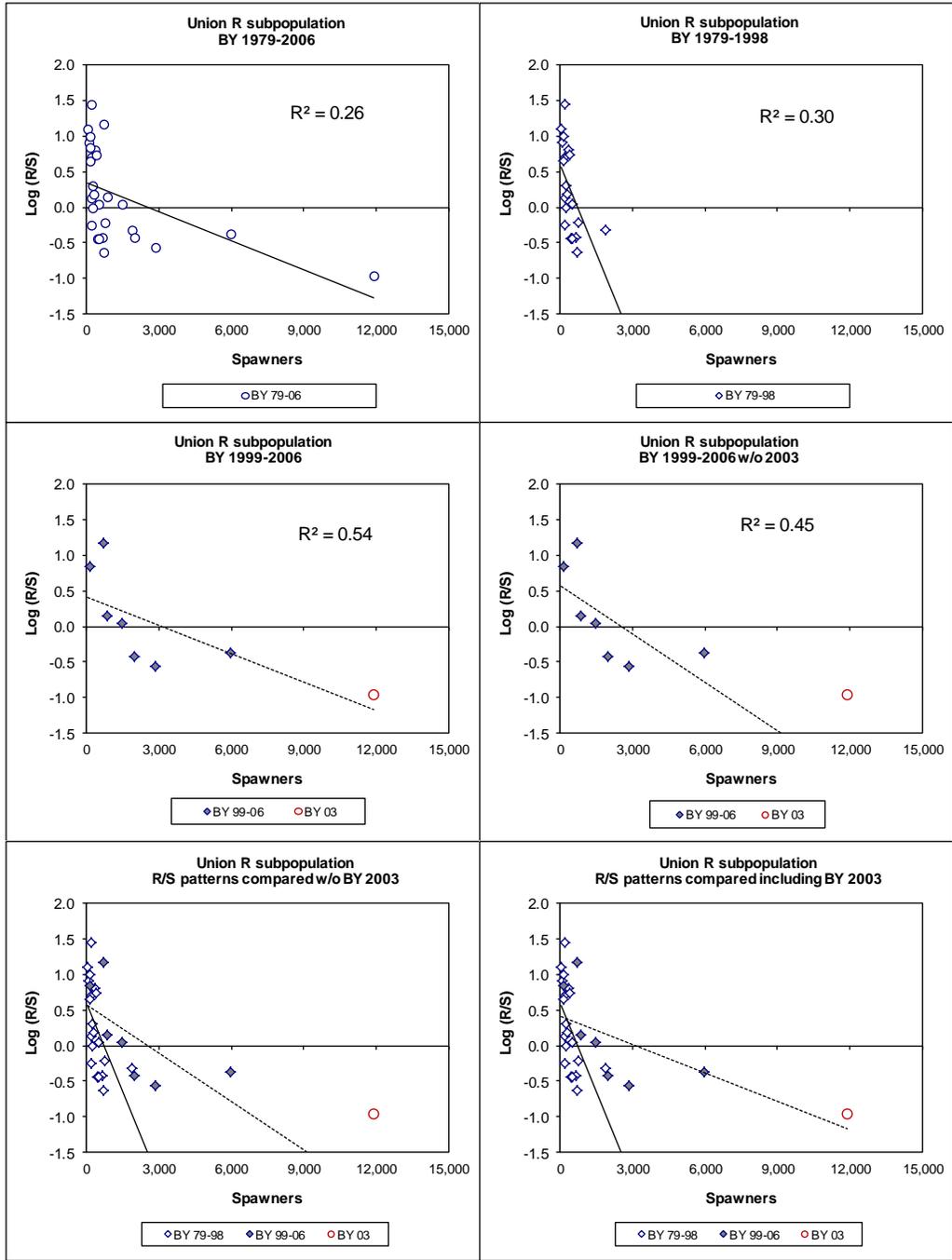
Union River Recruit Pattern

Union River subpopulation



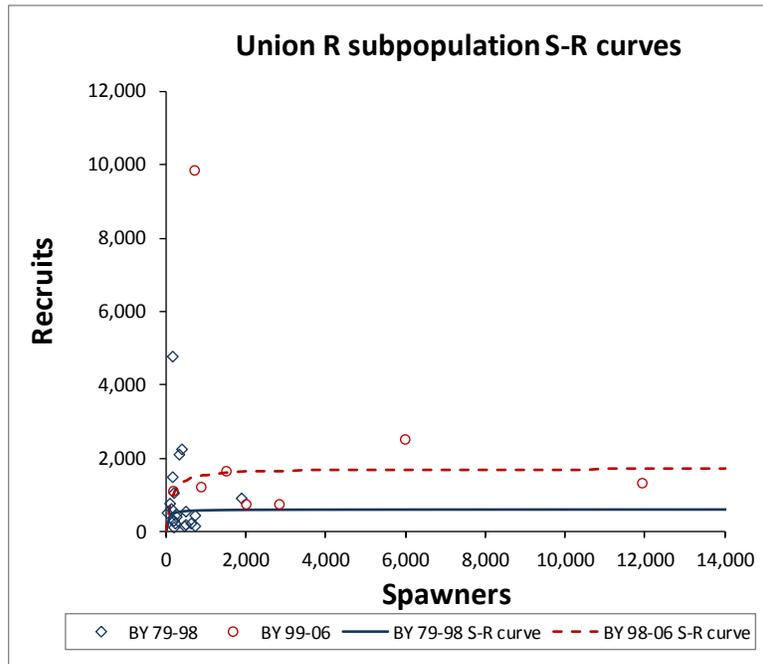
Appendix A - Figure 1. Patterns of total recruits for the Union River subpopulation by brood year, spawner abundance by brood year, deviations from average recruitment, and deviations from the average PDO index.

Union River Log (S/R) Plots



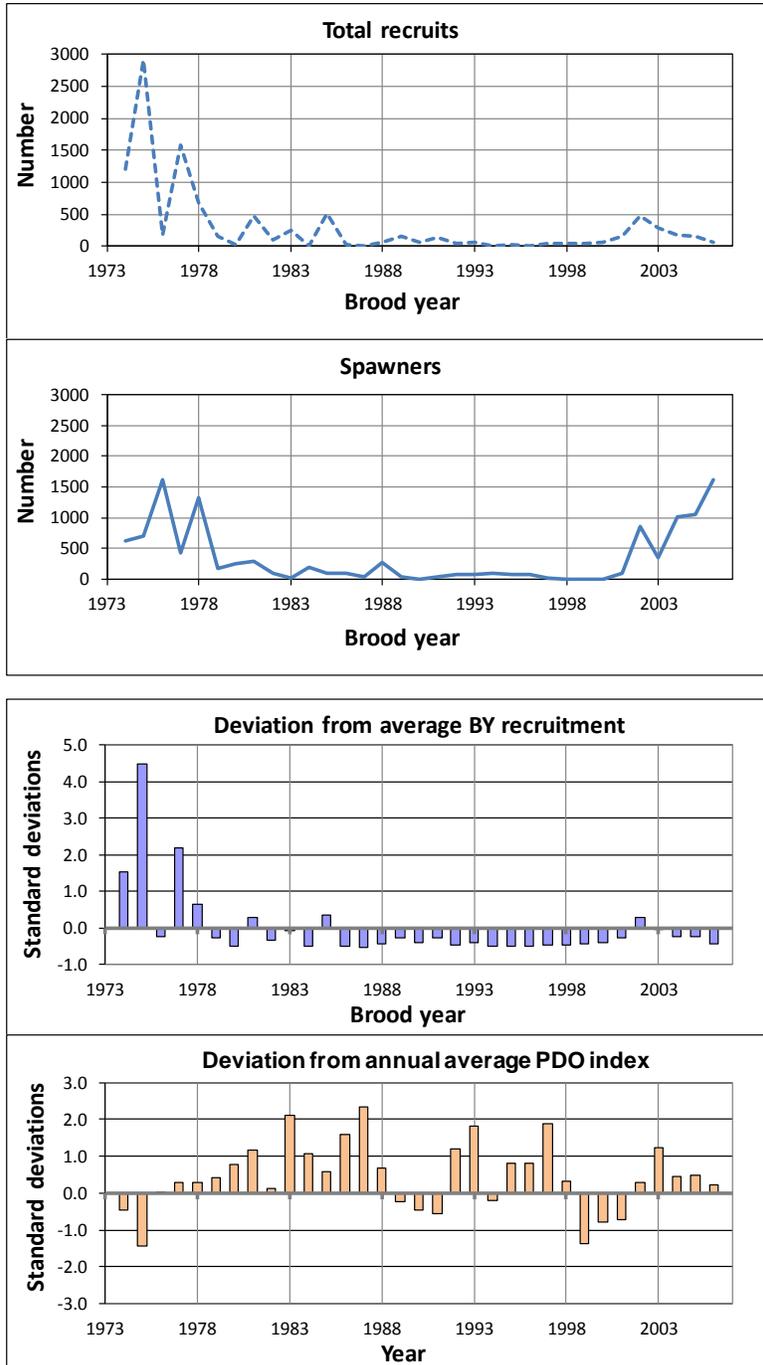
Appendix A - Figure 2. Spawner (S) and recruitment (R) plots grouped for different time periods for the Union River subpopulation. Brood years 1979 to 2006 are used.

Union River S-R curves



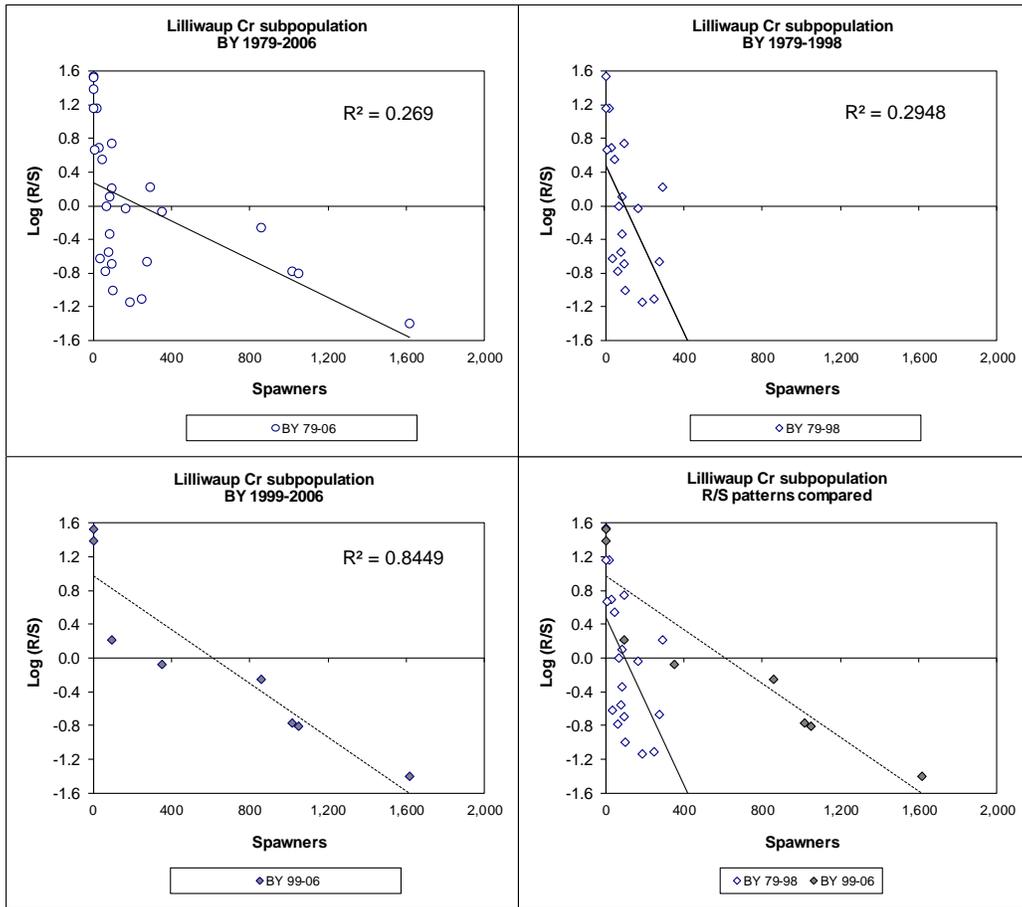
Lilliwaup Creek Recruit Pattern

Lilliwaup Creek subpopulation



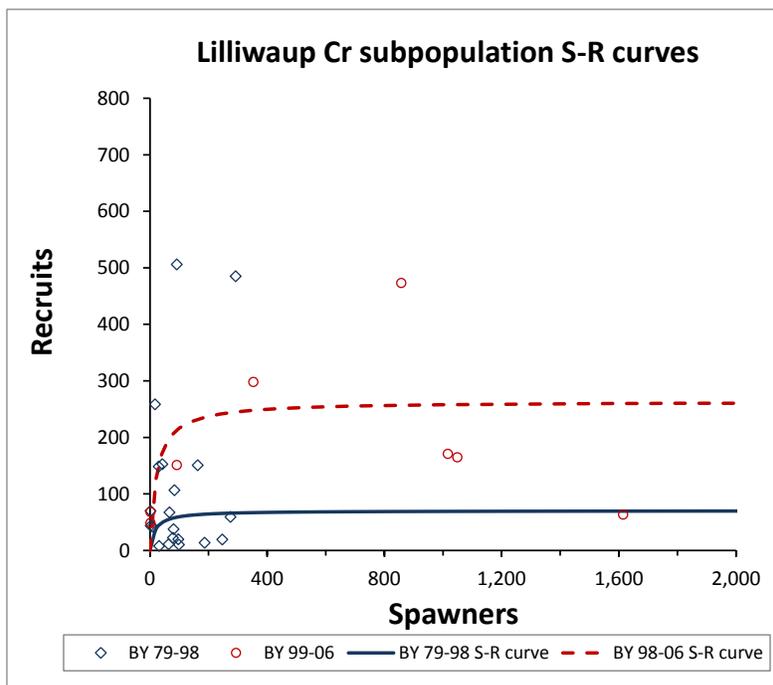
Appendix A - Figure 4. Patterns of total recruits for the Lilliwaup Creek subpopulation by brood year, spawner abundance by brood year, deviations from average recruitment, and deviations from the average PDO index.

Lilliwaup Creek Log (S/R) Plots



Appendix A - Figure 5. Spawner (S) and recruitment (R) plots grouped for different time periods for the Lilliwaup Creek subpopulation. Brood years 1979 to 2006 are used.

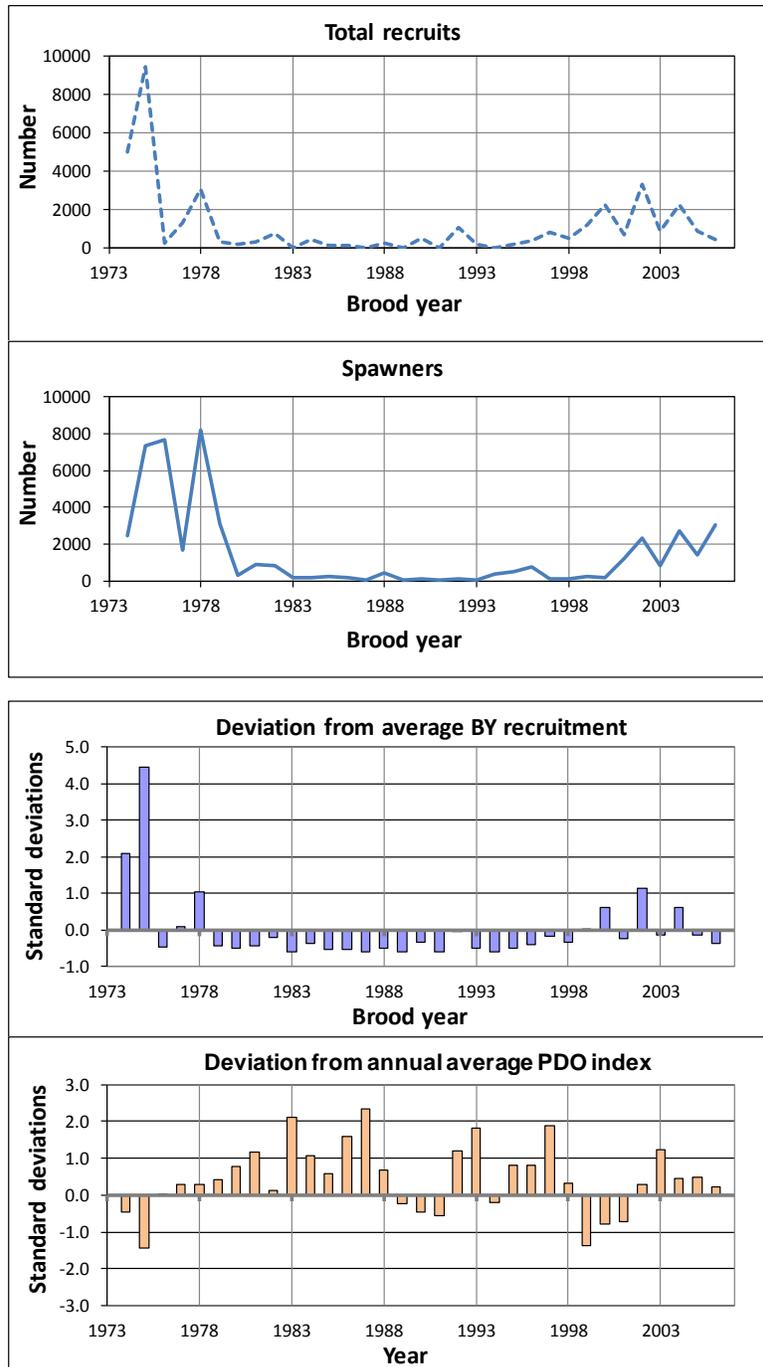
Lilliwaup Creek S-R curves



Appendix A - Figure 6. Spawner-recruit plots for brood years 1979-1998 and 1999-2006 for the Lilliwaup Creek subpopulation.

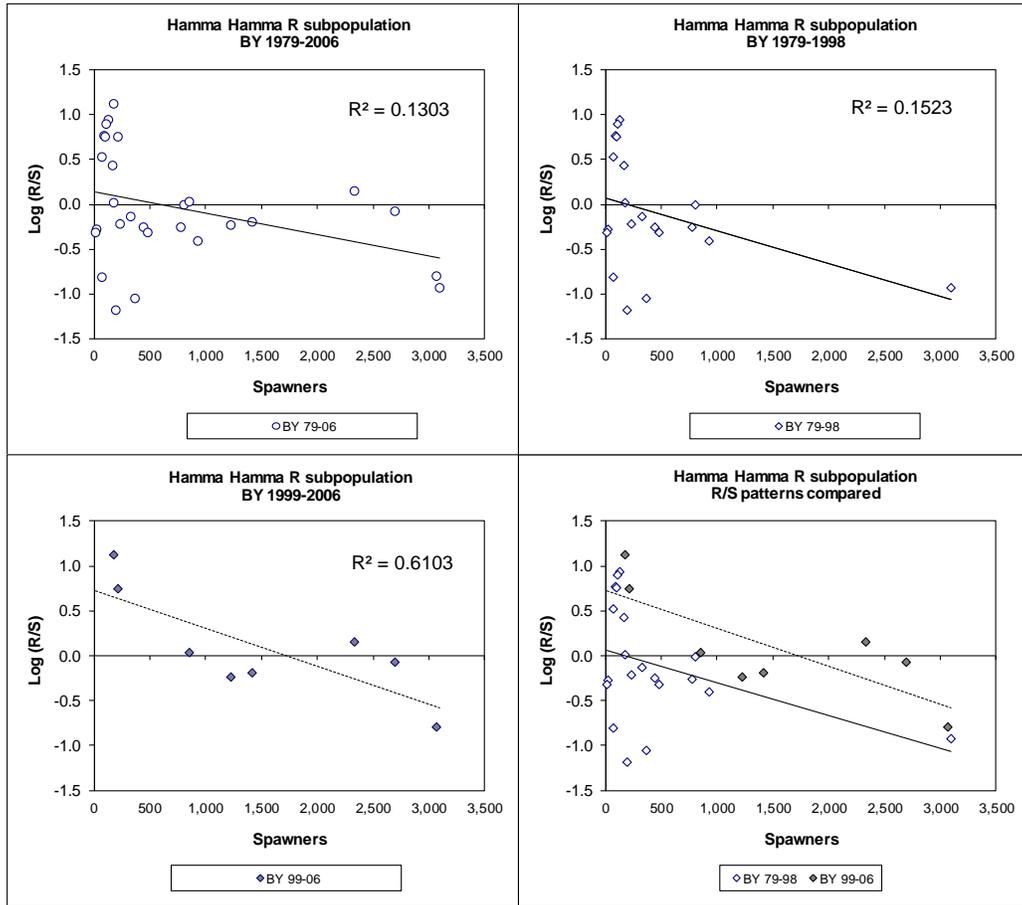
Hamma Hamma Recruit Pattern

Hamma Hamma River subpopulation



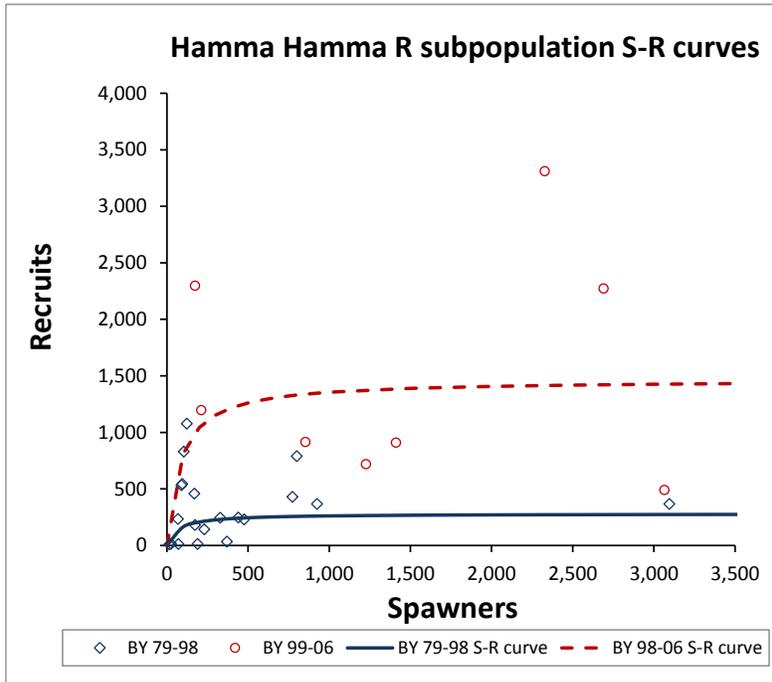
Appendix A - Figure 7. Patterns of total recruits for the Hamma Hamma River subpopulation by brood year, spawner abundance by brood year, deviations from average recruitment, and deviations from the average PDO index.

Hamma Hamma River Log (S/R) Plots



Appendix A - Figure 8. Spawner (S) and recruitment (R) plots grouped for different time periods for the Hamma Hamma River subpopulation. Brood years 1979 to 2006 are used.

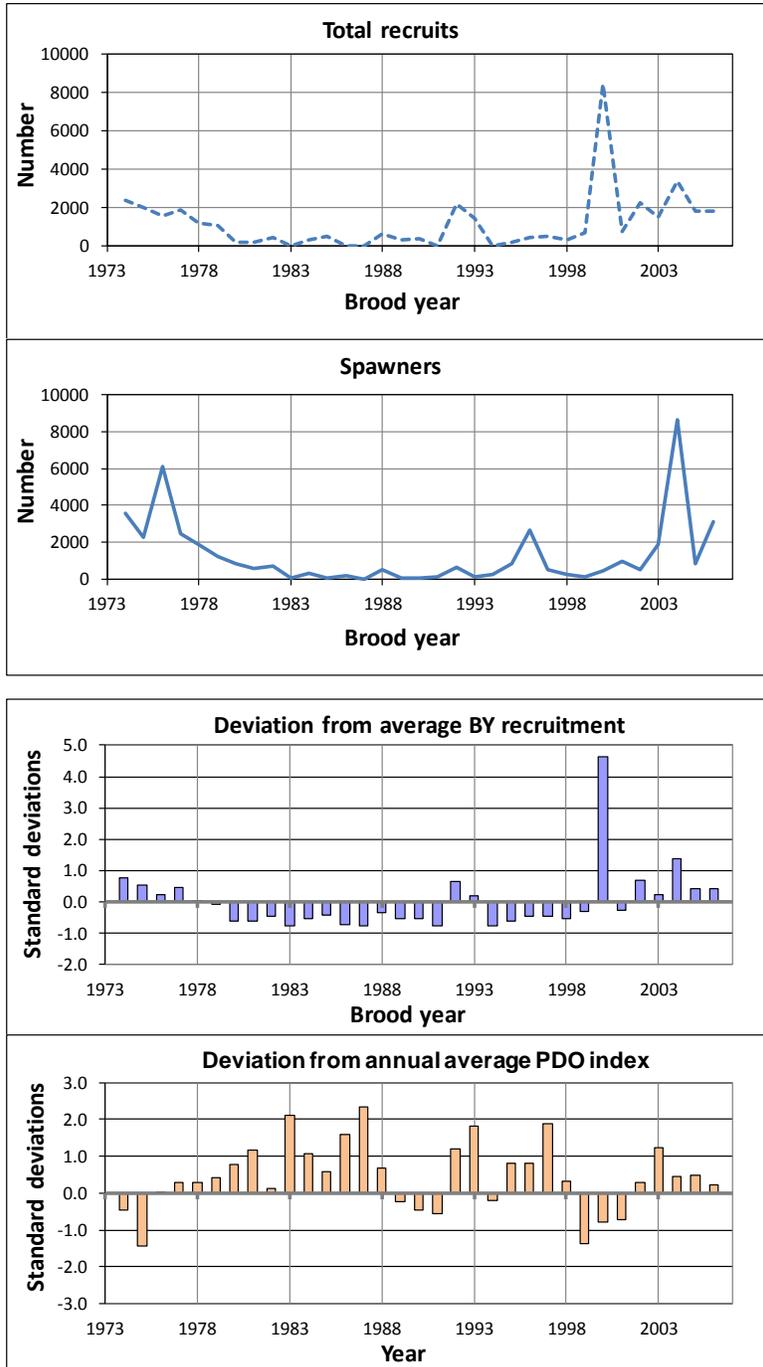
Hamma Hamma River S-R curves



Appendix A - Figure 9. Spawner-recruit plots for brood years 1979-1998 and 1999-2006 for the Hamma Hamma River subpopulation.

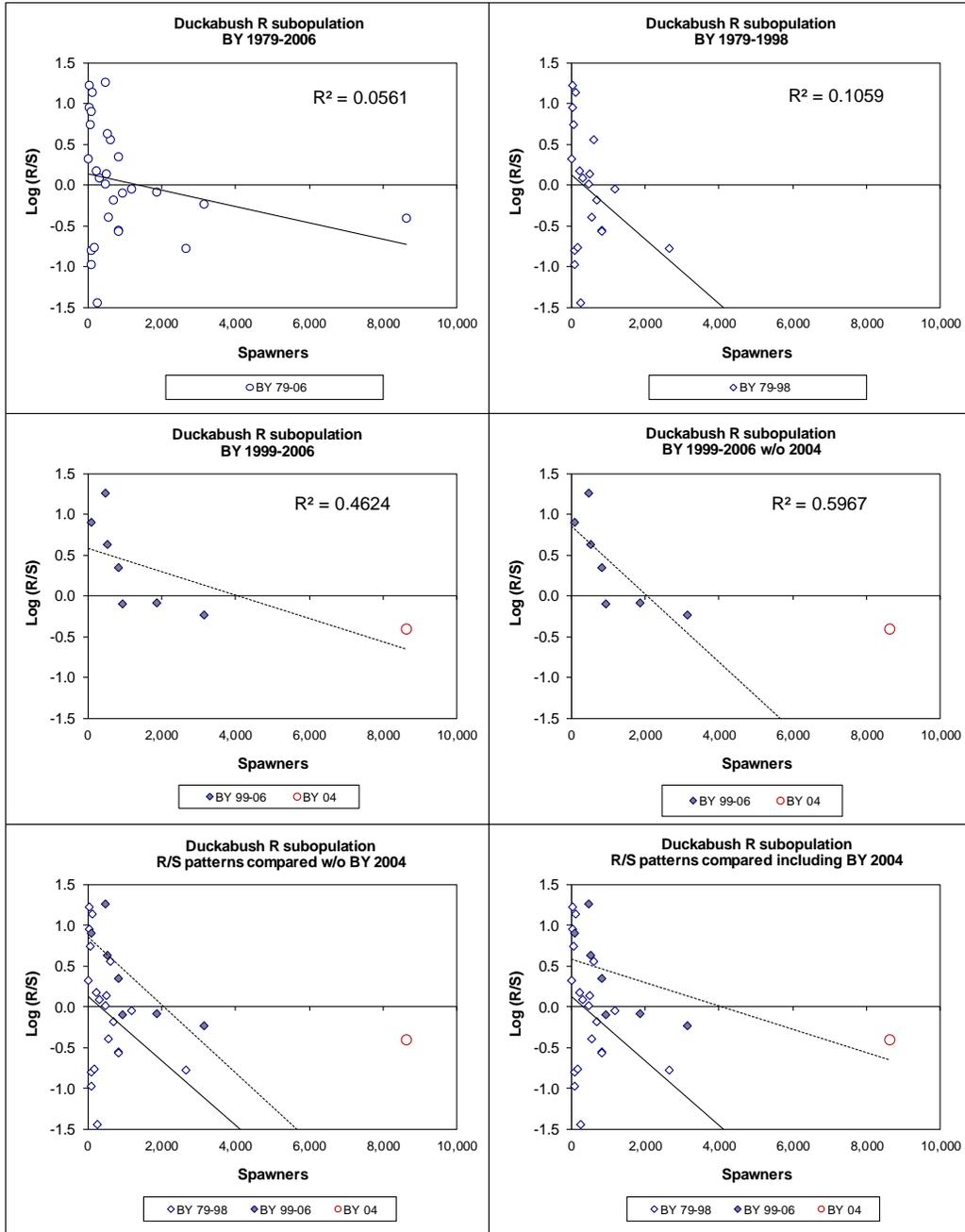
Duckabush River Recruit Pattern

Duckabush River subpopulation



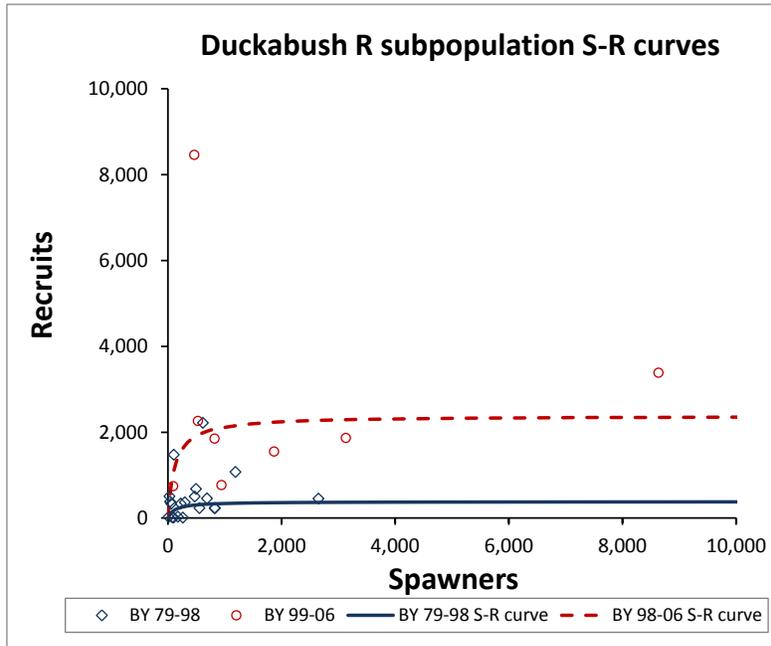
Appendix A - Figure 10. Patterns of total recruits for the Duckabush River subpopulation by brood year, spawner abundance by brood year, deviations from average recruitment, and deviations from the average PDO index.

Duckabush River Log (S/R) Plots



Appendix A - Figure 11. Spawner (S) and recruitment (R) plots grouped for different time periods for the Duckabush River subpopulation. Brood years 1979 to 2006 are used.

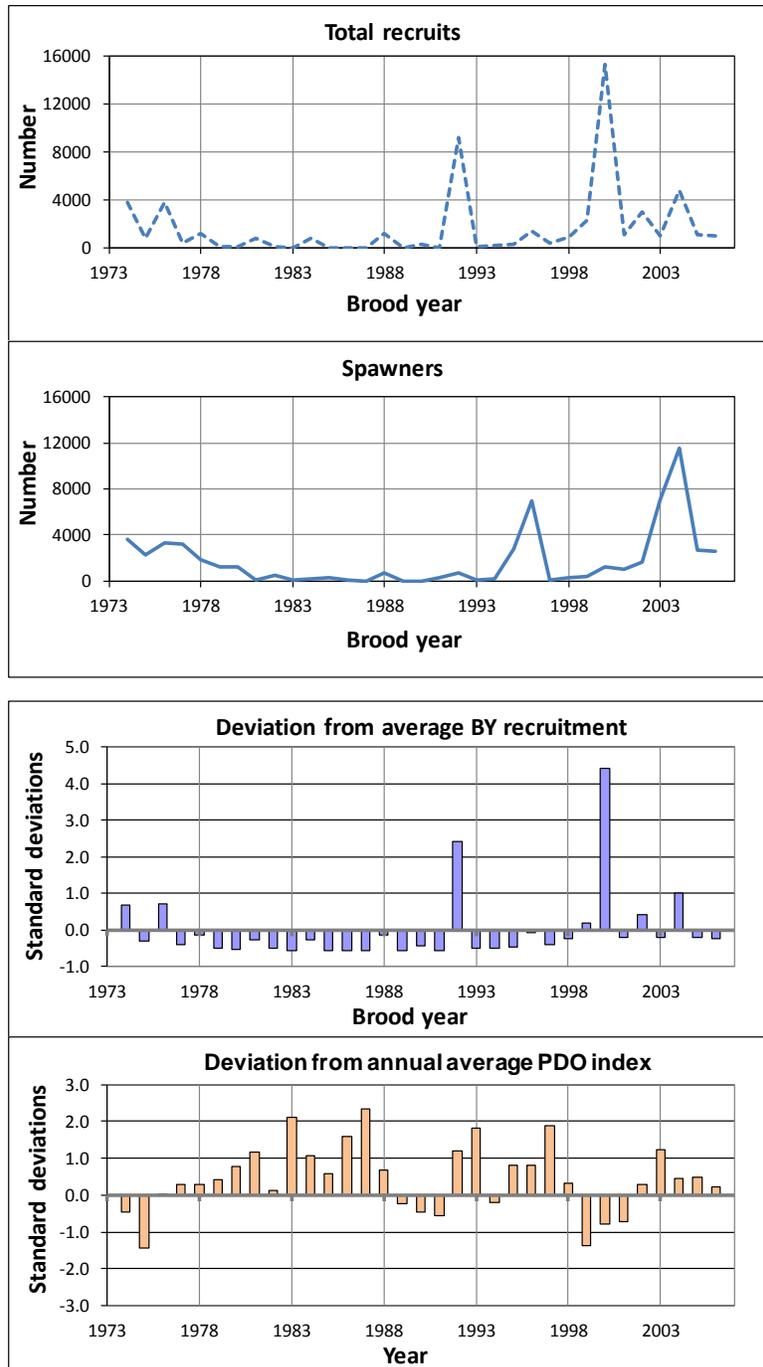
Duckabush River S-R curves



Appendix A - Figure 12. Spawner-recruit plots for brood years 1979-1998 and 1999-2006 for the Duckabush River subpopulation.

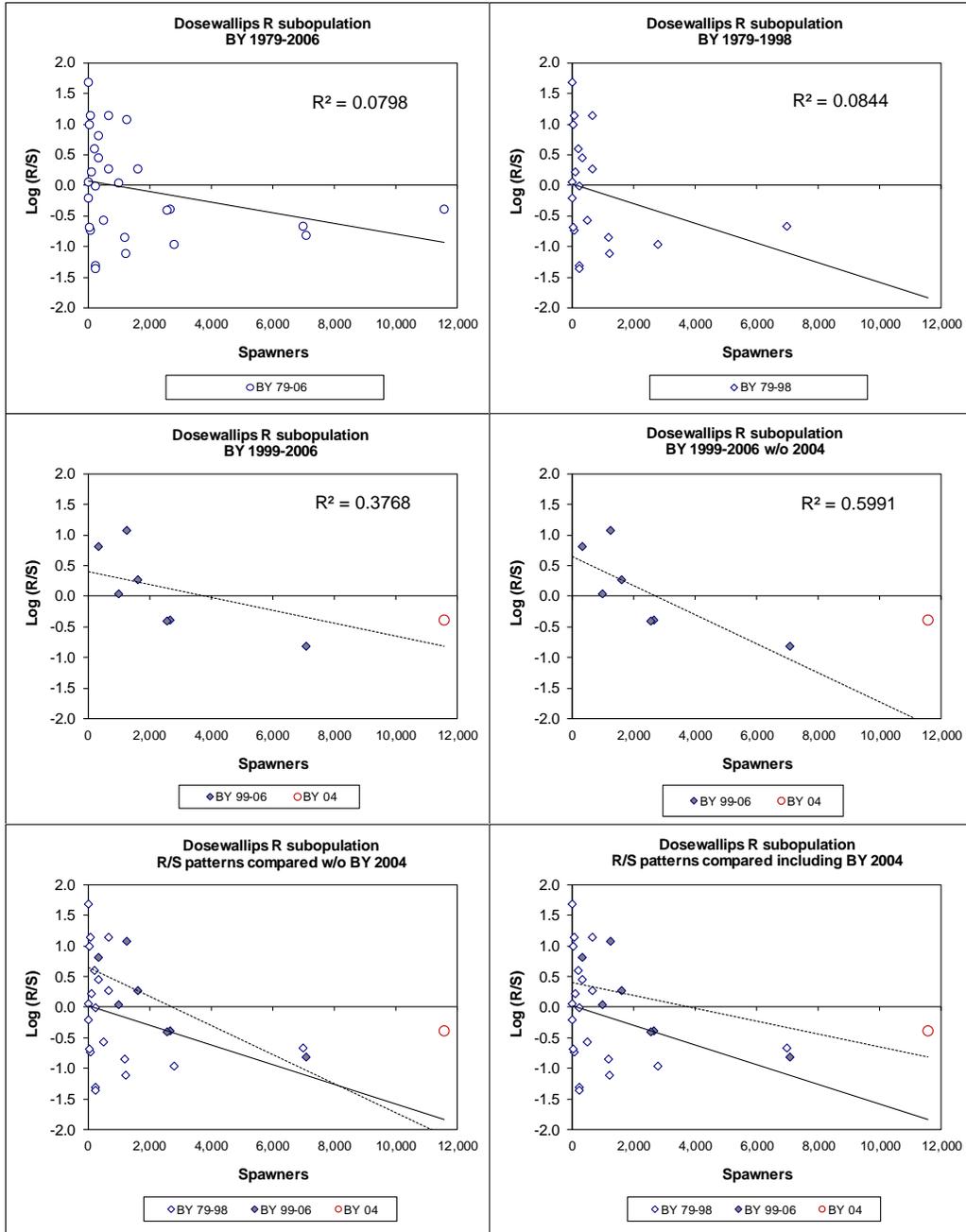
Dosewallips River Recruit Pattern

Dosewallips River subpopulation



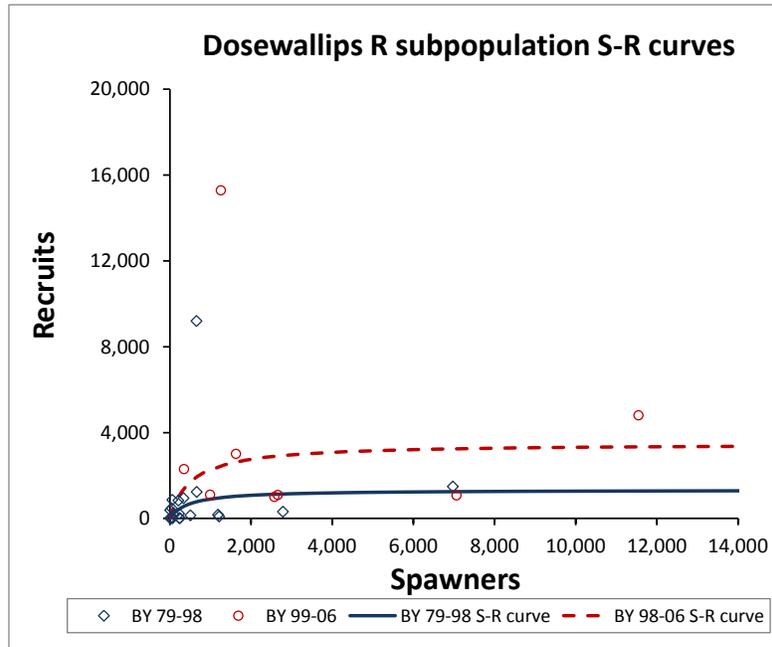
Appendix A - Figure 13. Patterns of total recruits for the Dosewallips River subpopulation by brood year, spawner abundance by brood year, deviations from average recruitment, and deviations from the average PDO index.

Dosewallips River Log (S/R) Plots



Appendix A - Figure 14. Spawner (S) and recruitment (R) plots grouped for different time periods for the Dosewallips River subpopulation. Brood years 1979 to 2006 are used.

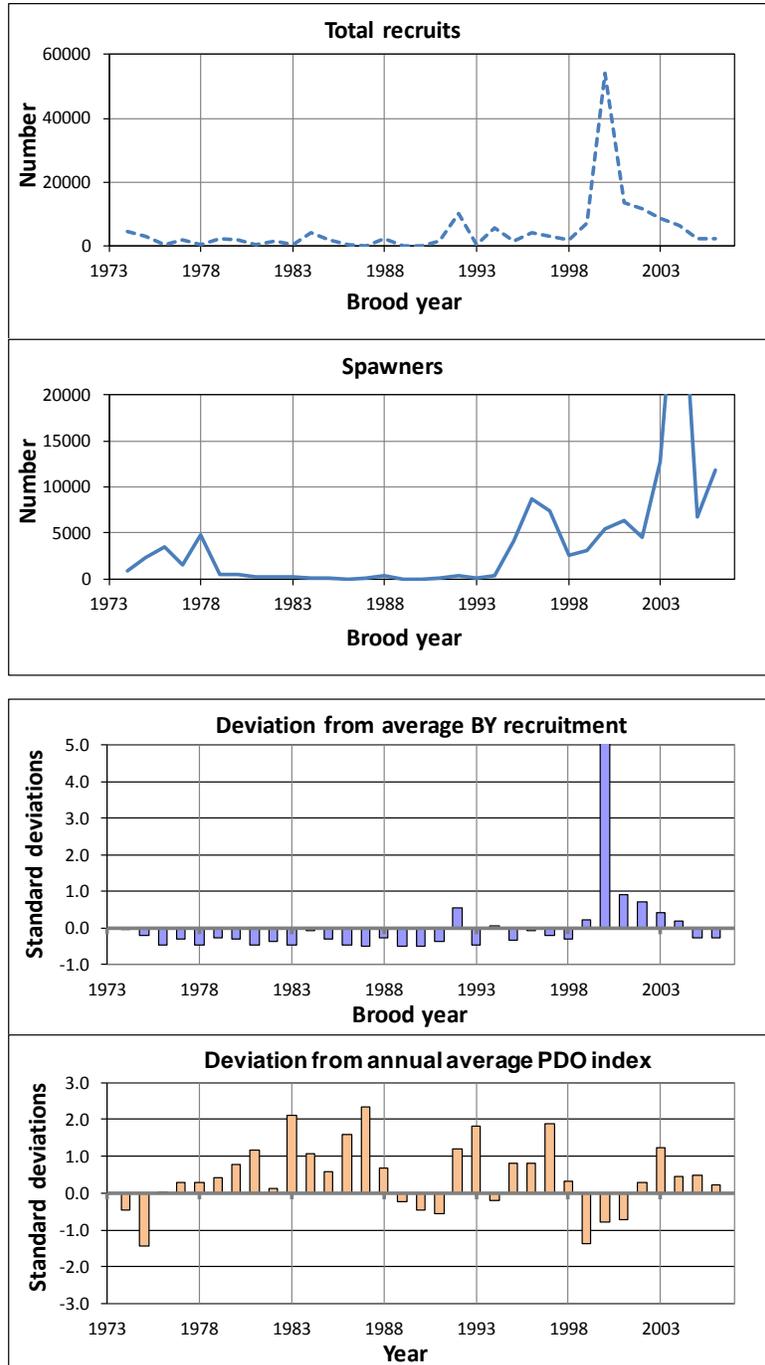
Dosewallips River S-R curves



Appendix A - Figure 15. Spawner-recruit plots for brood years 1979-1998 and 1999-2006 for the Dosewallips River subpopulation.

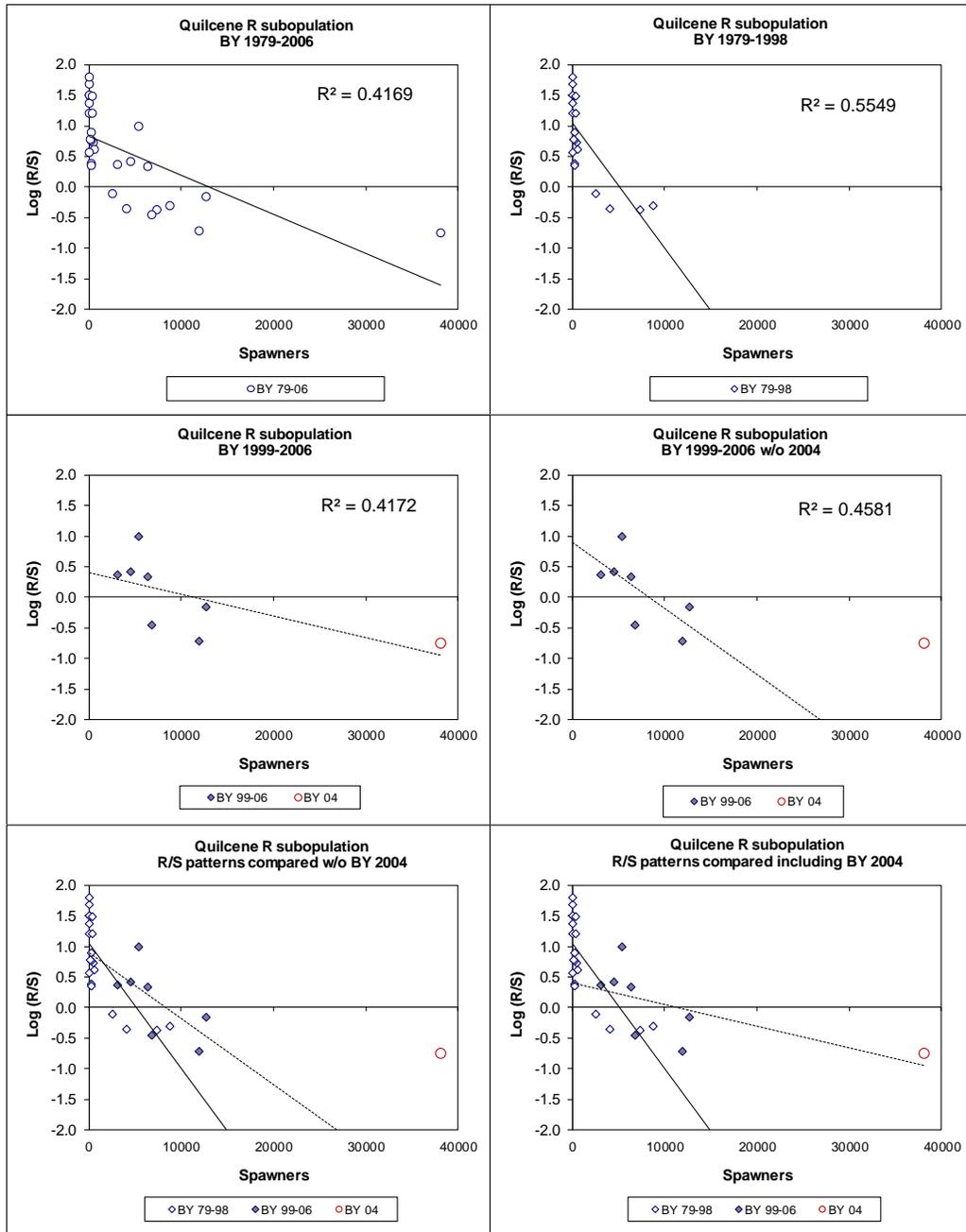
Big and Little Quilcene River Recruit Pattern

Quilcene River subpopulation



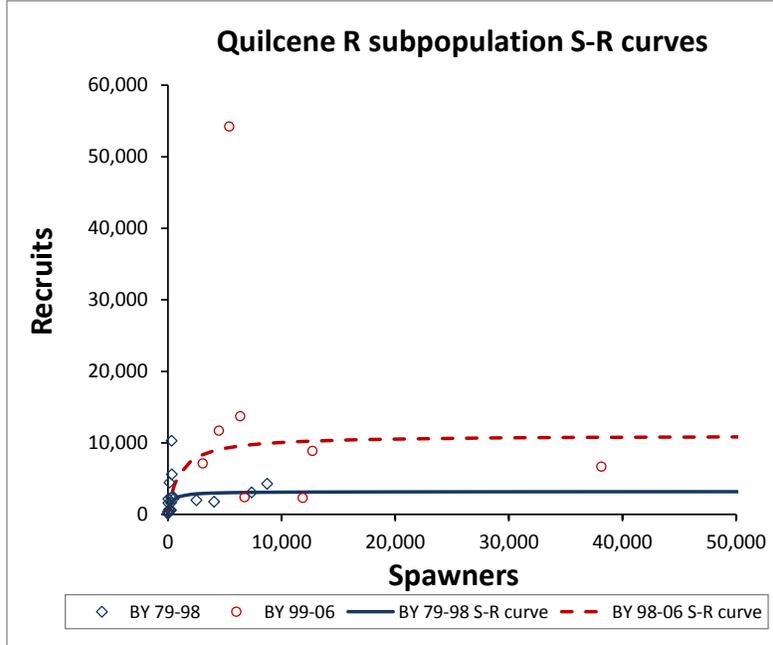
Appendix A - Figure 16. Patterns of total recruits for the Big and Little Quilcene River subpopulation by brood year, spawner abundance by brood year, deviations from average recruitment, and deviations from the average PDO index.

Big and Little Quilcene River Log (S/R) Plots



Appendix A - Figure 17. Spawner (S) and recruitment (R) plots grouped for different time periods for the Big and Little Quilcene River subpopulation. Brood years 1979 to 2006 are used.

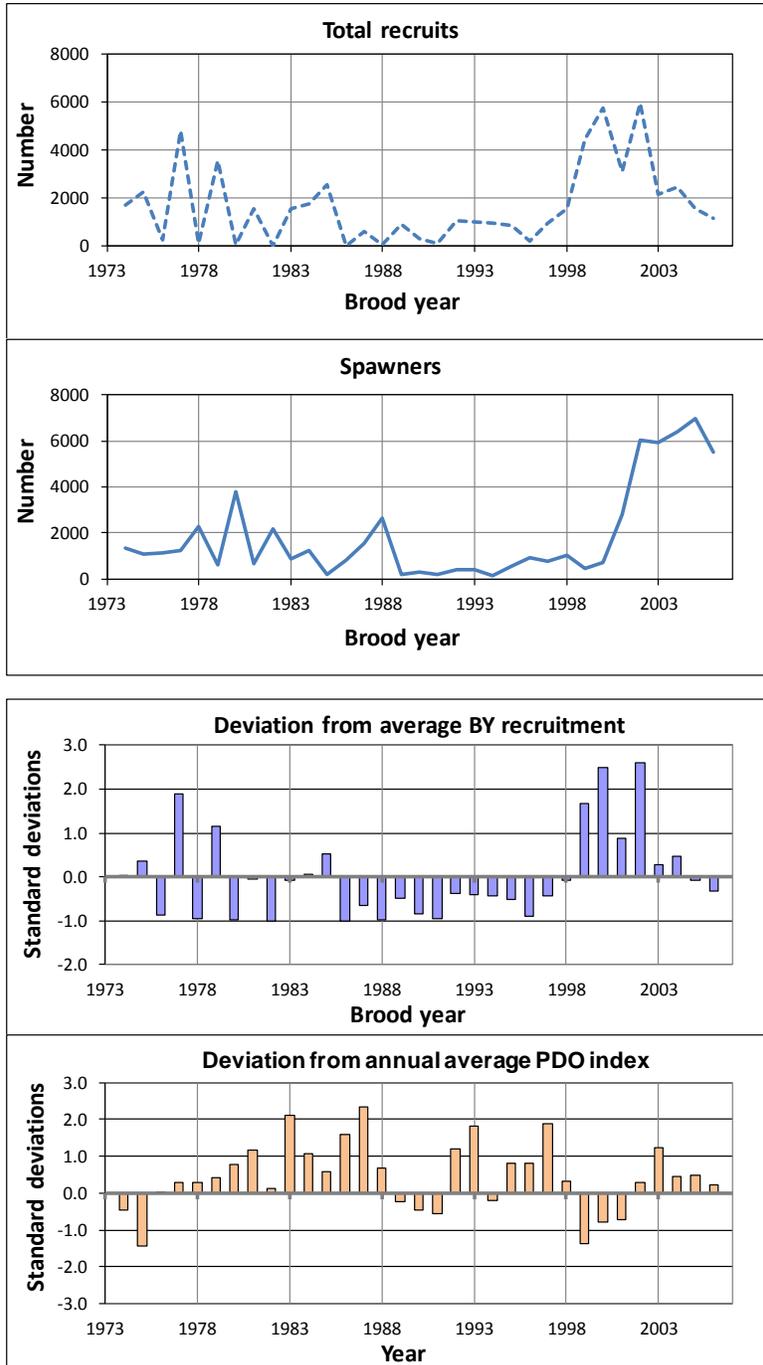
Big and Little Quilcene River River S-R curves



Appendix A - Figure 18. Spawner-recruit plots for brood years 1979-1998 and 1999-2006 for the Big and Little Quilcene River subpopulation.

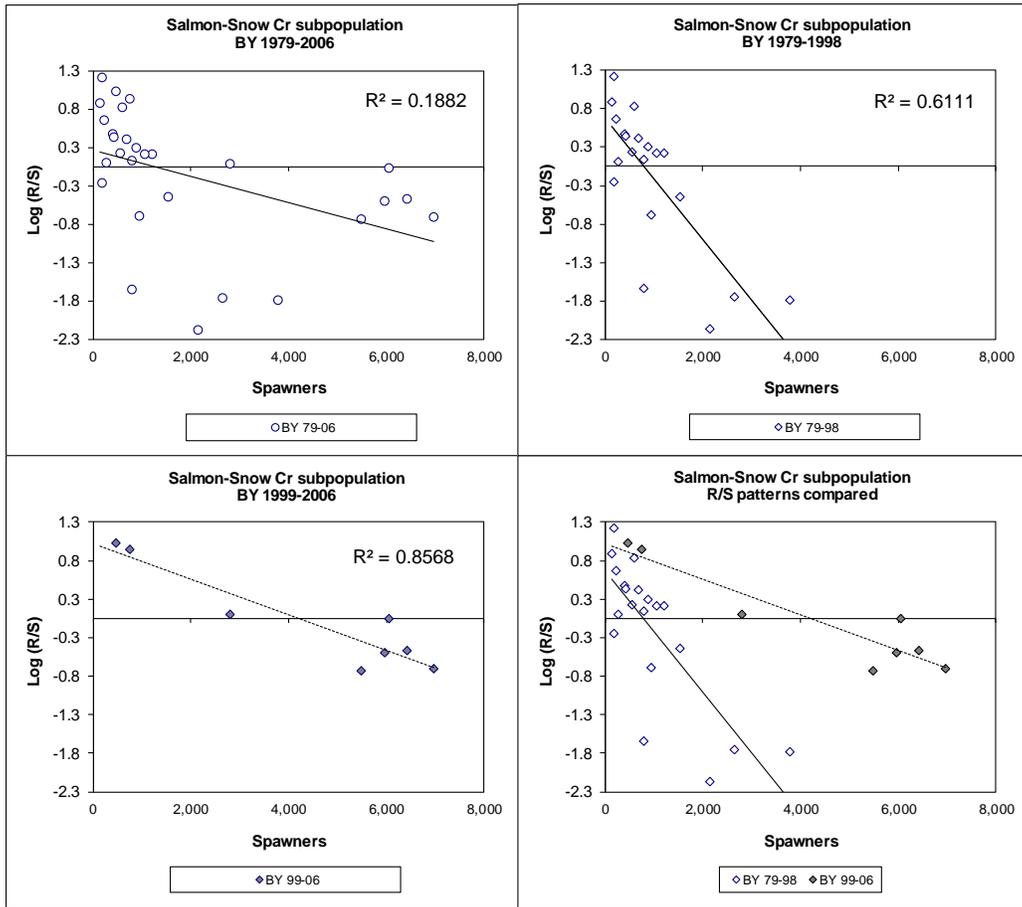
Salmon-Snow Creek Recruit Pattern

Salmon-Snow Creek subpopulation



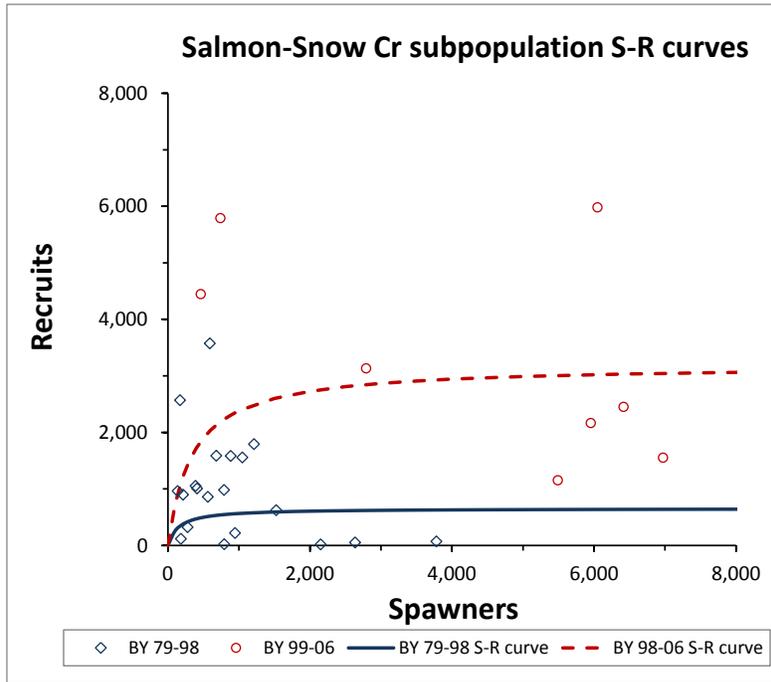
Appendix A - Figure 19. Patterns of total recruits for the Salmon and Snow Creek subpopulation by brood year, spawner abundance by brood year, deviations from average recruitment, and deviations from the average PDO index.

Salmon-Snow Creek Log (S/R) Plots



Appendix A - Figure 20. Spawner (S) and recruitment (R) plots grouped for different time periods for the Salmon and Snow Creek subpopulation. Brood years 1979 to 2006 are used.

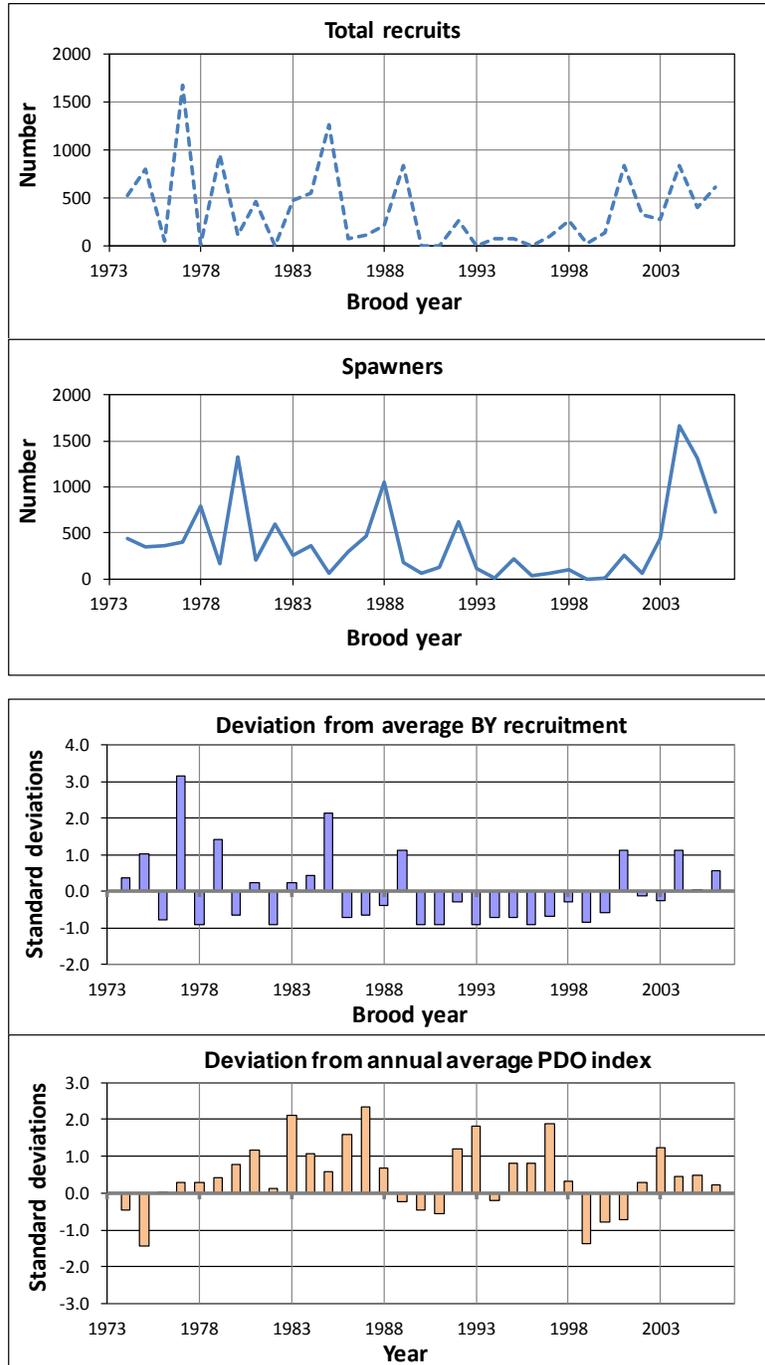
Salmon-Snow Creek S-R curves



Appendix A - Figure 21. Spawner-recruit plots for brood years 1979-1998 and 1999-2006 for the Salmon and Snow Creek subpopulation.

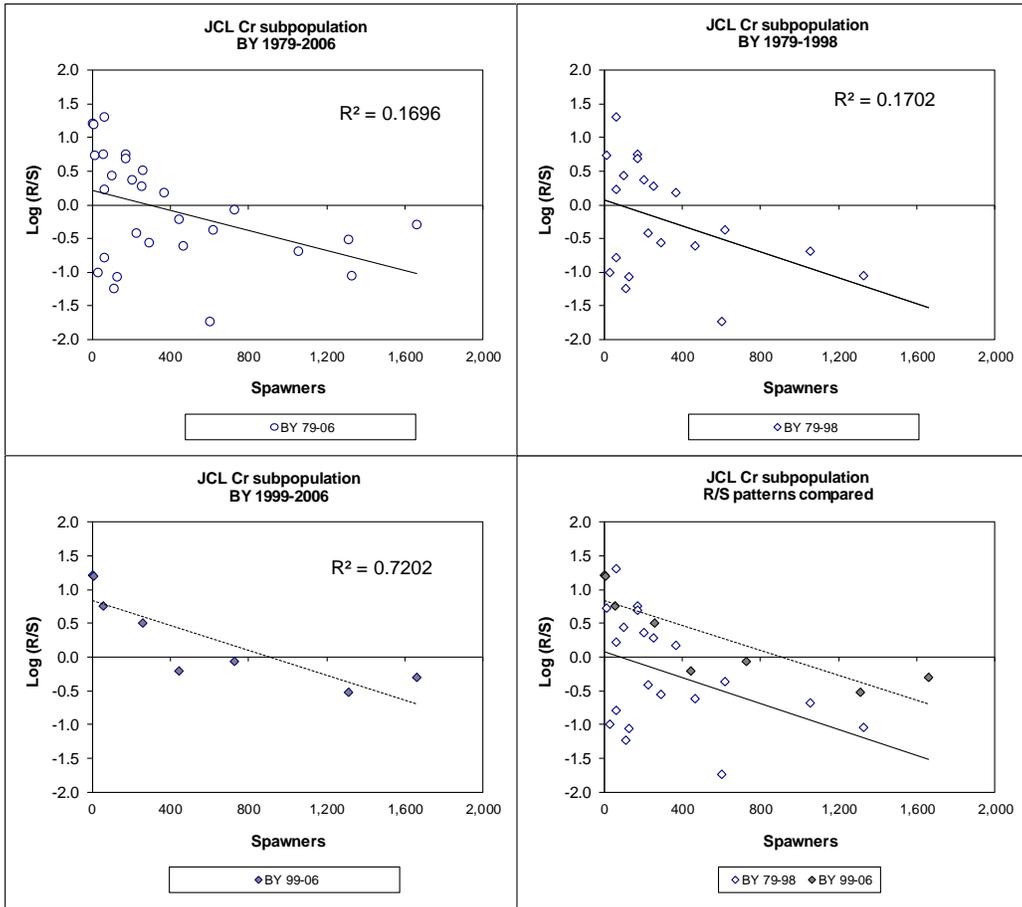
Jimmycomelately (JCL) Creek Recruit Pattern

Jimmycomelately Creek subpopulation



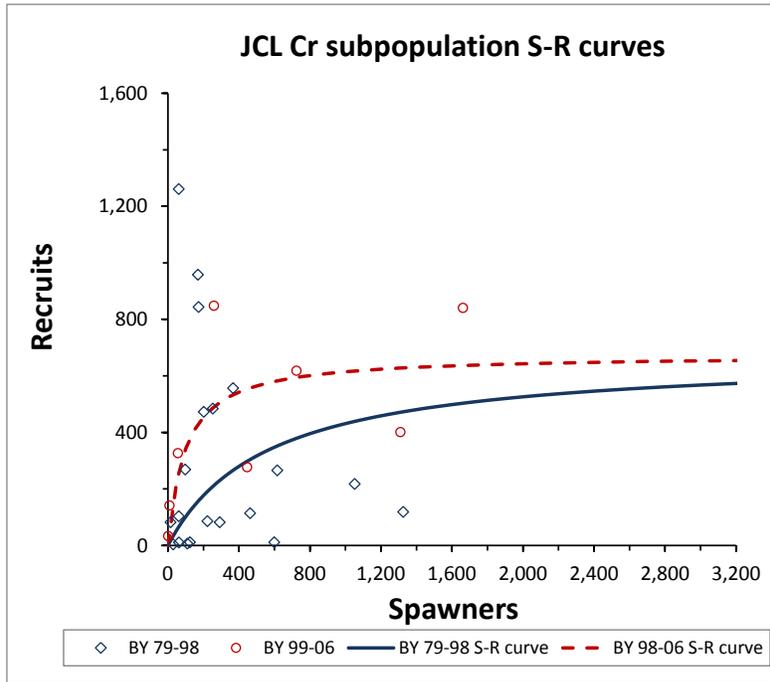
Appendix A - Figure 22. Patterns of total recruits for the Jimmycomelately Creek subpopulation by brood year, spawner abundance by brood year, deviations from average recruitment, and deviations from the average PDO index.

Jimmycomelately Creek Log (S/R) Plots



Appendix A - Figure 23. Spawner (S) and recruitment (R) plots grouped for different time periods for the Jimmycomelately Creek subpopulation. Brood years 1979 to 2006 are used.

Jimmycomelately Creek S-R curves



Appendix A - Figure 24. Spawner-recruit plots for brood years 1979-1998 and 1999-2006 for the Jimmycomelately Creek subpopulation.