Recovery Goal Review and Updated Guidance for the Hood Canal Summer Chum Salmon ESU

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Prepared by

Larry Lestelle
Biostream Environmental

Norma Sands
Norma Sands VRAP Consulting

Thom Johnson
Point No Point Treaty Council (retired)

Mark Downen
Washington Department of Fish and Wildlife

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Executive Summary

This report reviews the existing recovery goals for the Hood Canal Summer Chum evolutionarily significant unit (ESU) and provides guidance for updating the goals from those contained in the recovery plan for the ESU. Various analyses presented herein are based on data collected through 2016, though some additional insights are given based on preliminary data from 2017.

During the 1970s and 1980s the ESU as a whole, its populations, and subpopulations suffered sharp declines in abundance. Factors contributing to the decline were identified to be habitat degradation (freshwater, estuarine, and nearshore marine), harvest rates greater than were sustainable, and changes in climate/ocean conditions. 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. In the 1970s, harvest rates rapidly increased on these runs, so much that the rates sometimes approached, or exceeded, 80%, averaging close to 60% in the 1980s. Many subpopulations within the ESU were extirpated during this period. In effect, all of these factors converged in the 1980s—as the productive resiliency of the populations for sustaining themselves was being exceeded.

The ESU was listed as threatened with extinction in 1999 under the U.S. Endangered Species Act (ESA) (NMFS 1999). In 2000, tribal and state co-managers distributed the Summer Chum Salmon Conservation Initiative, 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, which was followed by a federal supplement to that plan in 2007. The federal supplement incorporated results of a viability analysis, issued as a stand-alone document in 2009 (Sands et al. 2009). All of these documents serve as the foundational documents for recovering the ESU.

The ESU is composed of two independent populations, called the Hood Canal and the Strait of Juan de Fuca (SJDF) populations, which are produced in streams and rivers that enter the marine waters bearing those names. The major spawning aggregations, identified as subpopulations, occur in independent watersheds that drain to the waters of either the eastern SJDF or Hood Canal. Eight or nine of the subpopulations were classified as extant and at least that many were considered by the Puget Sound Technical Recovery Team (PSTRT) to have been extirpated.

Over the past decade, remarkable progress seems to have been made in moving towards recovery of the ESU. The abundances of both populations that comprise the ESU have grown steadily and now exceed estimated thresholds that define low risk of extinction (lower ends of ranges). The abundance of the SJDF population has exceeded the low risk threshold in eight out of the past eight years, while the Hood Canal population has exceeded the threshold in five of the past five years based on thresholds given in Sands et al. (2009). Progress is also evident in strengthening the measures of biological diversity and spatial structure, which is also required for recovery; two subpopulations formerly classified as extirpated (Chimacum Creek and Skokomish River) have strongly rebounded. We conclude that there is a reasonable potential for delisting of the ESU from the protections of the ESA within the next decade.

While very significant recovery actions have been implemented over the past 30+ years, which have served to improve population performance, questions remain about the role that climate related factors have had in the rebound. Evidence exists that the performance of the populations and subpopulations

jumped up substantially corresponding with the regime shift that occurred in the Pacific Decadal Oscillation (PDO) in about 1999. The causal mechanisms for the correspondence with the PDO are not known. Marine survival conditions appear to have been largely favorable for the ESU since then. Beginning in early 2014, however, the PDO index turned sharply higher (to a warm phase) with monthly index values remaining high since then—potentially signaling that marine survival conditions have deteriorated over this period. In fact, ocean productivity indicators along the coasts of Oregon, Washington, and British Columbia during these years have declined sharply and remain at low levels currently.

Poor ocean survival conditions can mask improvements that have been made in freshwater and estuarine conditions from restoration efforts. Therefore, caution is advised in interpreting the upswing in the performance of the ESU and moving toward delisting prematurely. The final section of this report addresses this matter and provides decision criteria (in the form of a decision pathway) that can be used to evaluate the potential for delisting.

The report presents both background information and results of updated analyses to enable decision makers and scientists to evaluate topics of relevance to potential delisting over the next six or seven years. Highlights of these topics are summarized below.

Aspects of Recovery Goals

A viable ESU is naturally self-sustaining with a negligible risk of extinction (<5%) over a 100-year time period. Viability criteria given by the National Marine Fisheries Service (NMFS) consist of four characteristics of the ESU's performance: abundance and productivity among the populations that comprise the ESU, biological diversity within and among the populations, and spatial structure (distribution pattern) over the geographic area of the ESU. All four of these criteria need to be sufficiently met to achieve a negligible risk of extinction and, hence, a viable ESU.

Regarding broad-sense goals, when the recovery plan was prepared for the summer chum ESU in 2005, explicit broad-sense goals had not yet been defined by the state and tribal co-managers. To date, broad-sense goals still remain to be defined. The clearest goal statement for broad-sense recovery was given in the recovery plan (HCCC 2005), which stated the goal to be:

> *"To protect, restore and enhance the productivity, production and diversity of Hood Canal summer chum salmon and their ecosystems to provide surplus production sufficient to allow future directed and incidental harvests of summer chum salmon."*

Aspects of broad-sense goals are expected to be further clarified by the co-managers.

The ESU and Existing Criteria for Recovery

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). 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, together with depressed abundances of populations that may historically have served as source populations, have fragmented the ESU's structure, resulting in an increased risk of population extinction.

The PSTRT used the Viability and Risk Assessment Procedure (VRAP), a stochastic simulation model, to identify abundance and intrinsic productivity criteria to achieve a negligible risk of extinction for each of the two populations in the ESU. Modeling results were used to identify levels of population performance needed to achieve these criteria, expressed as the average spawning escapements under various exploitation rates. The results were also expressed as viability curves defining relationships between abundance and intrinsic productivity. These relationships are used to identify how existing population performance metrics defined by spawner-recruit parameters compare to the viability curves, enabling us to assess a population's level of extinction risk (relative solely to abundance and intrinsic productivity).

With regard to biological diversity and spatial structure viability criteria, the PSTRT concluded that all of the extant subpopulations within the ESU need to remain persistent, due to the significant loss in biological diversity that has already occurred in the ESU. They also concluded that one or more subpopulations/spawning aggregations need to be persistent within each of the ecological diversity groups within each population, where these groups are defined by geographic areas within the boundaries that define the ESU, except the Dungeness and Toandos areas were excluded for this purpose. Those areas were excluded due to uncertainties about whether robust spawning aggregations existed in those areas historically. The PSTRT also stated that some extirpated subpopulations would need to be restored to achieve the needed spatial distribution of spawning aggregations for viability.

Updated Viability Assessment for Abundance

We employed the same methods used in Sands et al. (2009) to update the viability assessment for population abundance by incorporating more recent data on population abundance. A previous update was reported in Lestelle et al. (2014). The analysis presented here utilizes 11 and 6 more years of brood year data than were available to Sands et al. (2009) and Lestelle et al. (2014) respectively.

This assessment reported herein is based on data through brood year 2012, incorporating adult returns through 2016. The updated assessment produces viability thresholds for the SJDF population moderately higher than those given in Sands et al. (2009). In contrast, the 5% risk threshold was lowered substantially for the Hood Canal population in this assessment compared to both of the earlier analyses. Changes in the 5% risk thresholds were associated with changes in the amount of year-to-year variation in adult recruitment around the predicted stock-recruitment curve for each population. The changes in the viability thresholds are due mainly to longer data sets used in the analysis, which produced more precise estimates of variation.

Results considering all of the years in the data series show that the Hood Canal population is at negligible risk of extinction (<5% risk) with current biological performance, provided that the exploitation rate is held to a very low level. In contrast, the analysis shows 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%.

We also present viability curves for each of the subpopulations that were considered to be extant by the PSTRT and plot the relevant metrics for each subpopulation to demonstrate how performance varies within each population by subpopulation. The results suggest that all of the subpopulations are at relatively high risk of extinction when considering all of the years in the data series, except for the Union and Quilcene subpopulations, which are comparatively more productive. These results indicate that the low risk of extinction for the Hood Canal population is due largely to the disproportionate contribution of a few subpopulations to the overall abundance of this population, primarily because of the Quilcene and Union subpopulations.

Incorporating Effects of the PDO

Conditions related to salmon survival within the Northeast Pacific Ocean and connecting marine waters have been linked to the PDO, a phenomenon of Pacific climate variability that includes changes in air and sea temperatures, winds, and precipitation. These conditions can affect food webs related to the marine survival of salmon.

The PDO has been generally believed to create climate regimes that can last 20 to 40 years. 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. States in climate that persist on a decadal scale are called regimes and a sudden change to a new regime has been referred to as a regime shift. The PDO index is derived from monthly sea surface temperatures (SST) in the North Pacific Ocean. Extended periods when the annual deviations in the index are greater than the overall average are referred to as a warm phase of the PDO; a cool phase occurs when deviations for extended periods are less than the overall average.

The PDO is thought to have been in its cool phase from about 1890 to 1925 and from 1945 to 1977. It was in its warm phase from 1925 to 1945 and from 1977 to the late 1990s. It shifted back to its cool phase beginning about 1998.

The marine survival of some salmon species and populations has been linked to patterns of the PDO index. In particular, there is a strong correspondence between marine survival and the PDO index for some Puget Sound populations, and particularly for both populations of the Hood Canal Summer Chum ESU.

We present data for both populations in the ESU, as well as their subpopulations, to demonstrate patterns of correspondence between performance and the PDO index. Coincidentally, both populations demonstrated substantial improvement in performance soon after the ESU was ESA-listed – corresponding with the shift in the PDO index to the cool phase. Improved performance has continued to the present time, though there is a notable difference between the SJDF and Hood Canal populations. The available data demonstrate a time lag between the PDO index for the Hood Canal population (up to 2-year lag) but essentially no lag for the SJDF population.

We speculate that different time lags are caused by differences in water circulation and mixing patterns within the various parts of Puget Sound. Such differences may affect localized plankton blooms and food webs at variable times, perhaps differing by many months, as the effects of a PDO shift are transferred slowly into the various branches of Puget Sound. We have hypothesized that the correspondence of the PDO and summer chum performance is due to some kind of climate system forcing on the Puget Sound ecosystem related to the PDO. Causal mechanisms have not been identified. We suggest that the mechanisms may be related to how nutrients from oceanic waters are moved into the inland sea, which then affect food webs of importance to outmigrating summer chum fry.

An opportunity has availed itself for advancing our understanding about the effect of the PDO and a possible lag of the effect into Puget Sound. The PDO index turned strongly positive (warm phase) in January 2014 and has remained in the positive state to the present time. This means that if indeed an effect of the PDO is occurring on the summer chum populations, then fry produced from brood year 2013 (fry year 2014) could have experienced a downturn in survival. We projected a downturn in SJDF summer chum adult returns beginning in 2016, which we expect will continue for at least several years. We also project a downturn in Hood Canal adult returns beginning in 2018, taking into account a two-year lag in effect. We now know that the SJDF population did not demonstrate a downturn in 2016, but abundance dropped sharply in 2017. In contrast, the Hood Canal population continued its upward trajectory in both 2016 and 2017.

Despite the large downturn in the SJDF population in 2017, we cannot conclude that it was due to PDO-related effects. Other possible explanations exist. Drawing on recently published information (Weinheimer et al. 2017), we show that stream flow effects on fry production within at least some components of the SJDF population were also responsible for the downturn in the SJDF population in 2017. Flow-related effects on fry production are expected to continue with reduced adult returns in 2018. We expect that different combinations of flow-related and PDO-related conditions over the next several years and between the two populations will provide results to identify the relative role of the PDO to the rebound of the ESU since 2000.

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 severity of a 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 functional habitats to exist to buffer and offset poor marine survival.

Updated Assessment for Spatial Structure and Diversity

A forum for invited scientists with expertise in the performance of the Hood Canal Summer Chum ESU, conservation biology, and salmon recovery planning was held in April 2017 to address the recovery criteria related to biological diversity and spatial structure. The purpose of the forum was to review the conclusions reached by the PSTRT for these criteria as documented in Sands et al. (2009), review the most recent available information on summer chum performance, and to formulate recommendations for updating the biological diversity and spatial structure criteria for recovery if deemed needed.

Based on information that came from the forum, we concluded that all of the requirements for biological diversity and spatial structure are currently being met in the ESU with the exception of the condition of spawning aggregates on the Kitsap Peninsula. There is currently only one robust subpopulation (Union River) within this region. Based on newer information that was not available to the PSTRT (i.e., the analysis presented in Sands et al. 2009), we conclude that at least one additional robust subpopulation is needed to meet the requirements for spatial structure in this region. The re-establishment of the Dewatto River subpopulation is seen at this time as being the highest priority for the region because it is approximately halfway between the northern end of the region and Union River, located in the south-eastern corner of the region. The habitat in Dewatto River is also believed to be relatively intact. We discuss the possibilities for other subpopulations also (most notably for Big Beef Creek) that were considered to have been extirpated by the PSTRT.

Harvest Considerations

We present information to facilitate discussions among the co-managers to help address harvest issues prior to and following delisting based on analyses contained in this report. We provide several examples for how harvest-related actions might be shaped to aid in recovery, while at the same time fostering updated harvest provisions consistent with the performance of the ESU's populations and subpopulations. We recognize that there is a need for a comprehensive review of the existing Base Conservation Regime (BCR) that considers the information presented in this report.

Summary of Recommendations and a Decision Pathway for Recovery

The last section of the report provides a set of recommendations to address ten issues for the purpose of expediting progress in planning for possible delisting. These issues will require attention by the parties having recovery planning authority and responsibilities—namely by HCCC as the regional recovery organization, WDFW and relevant tribal co-managers, and NMFS. The ten issues are the following:

1. Viability criteria;
2. Harvest regimes;
3. Broad sense recovery goals;
4. Lilliwaup Creek supplementation;
5. Dungeness River reintroduction;
6. Status of threats to the ESU;
7. Climate change planning;
8. Need for continuing restoration efforts;
9. Monitoring, evaluation, and research; and
10. Accounting for PDO-related effects – a decision pathway.

Issue number ten, which incorporates our final recommendation at the end of the section, provides a decision pathway that addresses the largest critical uncertainty potentially affecting a delisting decision, that is, the extent that marine survival conditions (i.e., PDO-related factors) have driven the rebound since 2000. If marine survival drops sharply, which could occur based on the phase shift in the PDO index seen in 2014, the question then becomes how the populations will perform with the combination of reduced survival and the benefits of restoration actions that have occurred. The set of circumstances that has occurred since the PDO phase shift, together with recent flow-related effects on at least two subpopulations, are expected to be highly informative about the current performance characteristics of the populations relative to recovery criteria.

We project that sufficient information should be available no later than the end of 2022 or 2023 to make an informed decision about delisting. A decision pathway is explained based on the record of run sizes that occur in 2017 to 2022.

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Recovery Goal Review and Updated Guidance for the Hood Canal Summer Chum Salmon ESU

1.0 Introduction

This report reviews the existing recovery goals for the Hood Canal Summer Chum evolutionarily significant unit (ESU) and provides guidance for updating the goals. It is an updated version of a document submitted to the Hood Canal Coordinating Council (HCCC) in 2014, which gave recommendations for updating recovery goals for the ESU (Lestelle et al. 2014).

The Hood Canal Summer Chum ESU was listed as threatened with extinction in 1999 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 (SJDF) populations, which are produced in streams and rivers that enter the marine waters bearing those names (Sands et al. 2009) (Figure 1).[1] 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 SJDF or Hood Canal (Figure 2). Eight or nine of the subpopulations were classified as extant and at least that many were considered to be extirpated.[2]

In 2000, tribal and state co-managers distributed the Summer Chum Salmon Conservation Initiative (SCSCI) (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). The federal supplement incorporated results of a viability analysis; that analysis was issued as a stand-alone document in 2009 (Sands et al. 2009). All of these documents serve as the foundational documents for recovering the ESU.

The 2014 guidance report reviewed all information then available on the status of the populations and their subpopulations and updated the viability analysis with newer information than was used by the Puget Sound Technical Recovery Team (PSTRT) (Sands et al. 2009). That guidance report incorporated five additional brood years of data to the analysis done by the PSTRT. This updated report incorporates another six additional years of data. Hence this newest analysis presented herein adds 11 years of brood year data to the original viability work done by the PSTRT. The analysis incorporates data up to and including population run sizes returning to their spawning streams in 2016. However, during the period when this report was being prepared, returning runs were being assessed in late summer/fall of 2017 and we have been able to review some of this information, albeit in preliminary form. Where relevant to this report, we make note of information available from 2017.

This report also incorporates information obtained from two science forums held in the spring of 2017 to address aspects of the recovery goals related to climate factors and to biological diversity and spatial

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[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] / Uncertainty existed on how to classify the Dungeness subpopulation.

distribution within the ESU. In addition, we incorporated input obtained from meetings between HCCC staff with National Marine Fisheries Service (NMFS)[3], Washington Department of Fish and Wildlife (WDFW), and relevant tribes to review the results and recommendations contained within the 2014 guidance report (Lestelle et al. 2014) and to learn of any related issues in need of attention.

This report has been prepared to be a stand-alone document, and as such it is not a supplement to our earlier report. Parts of this current version remain similar to or unchanged from the 2014 report, but significant parts have been added or updated. We elected to include material in this version that was given in the earlier report so that the reader has all of the key substance to follow the entire flow of presentation. We condensed or eliminated some sections where we thought it would be helpful to the current version.

As seen in this document, remarkable progress seems to have been made over the past decade in moving towards recovery of the ESU. The abundances of both populations that comprise the ESU have grown steadily and now exceed estimated thresholds that define low risk of extinction (lower ends of ranges). Abundance of one of the two populations (Hood Canal population) has exceeded the low risk threshold in five out of the past five years and in the other population (Strait of Juan de Fuca population) in eight of the past eight years based on thresholds as given in Sands et al. (2009). Progress is also evident in strengthening the measures of biological diversity and spatial structure, which is also required for recovery; two subpopulations formerly classified as extirpated have strongly rebounded.

While very significant recovery actions have been implemented over the past 30+ years, which no doubt have served to improve population performance, questions remain about the role that climate related factors have had in the rebound. Evidence exists that the performance of the populations and subpopulations jumped up substantially corresponding with the regime shift that occurred in the Pacific Decadal Oscillation (PDO) in about 1999.[4] The causal mechanisms for the correspondence with the PDO are not known. Marine survival conditions appear to have been largely favorable for the ESU since then. Beginning in early 2014, however, the PDO index turned sharply higher (to a warm phase) with monthly index values remaining high since then—potentially signaling that marine survival conditions have deteriorated over this period. In fact, ocean productivity indicators along the coasts of Oregon, Washington, and British Columbia during these years have declined sharply (Peterson et al. 2015) and remain at low levels currently (Werner et al. 2017).

As discussed in this report, poor ocean survival conditions can mask improvements that have been made in freshwater and estuarine conditions from restoration efforts. Therefore, caution is advised in interpreting the upswing in the performance of the ESU and moving toward delisting prematurely. The final section of this report addresses this matter and provides decision criteria (in the form of a decision pathway) that can be used to evaluate the potential for delisting.

This report is organized into the following sections:

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[3] / The National Marine Fisheries Service (NMFS), informally known known as NOAA Fisheries , is an office of the National Oceanic and Atmospheric Administration. This report uses the name NMFS to refer to that federal agency unless a specific citation refers to NOAA Fisheries.

[4] / The PDO is a recurring long-lived (i.e., extending over multiple years) El Niño-like pattern of Pacific climate variability. Major changes in northeast Pacific marine ecosystems have been correlated with phase changes in the PDO.

1. Introduction;
2. Aspects of Recovery Goals;
3. History of Recovery Planning for the ESU;
4. The ESU and Existing Criteria for Recovery;
5. Updated Viability Assessment for Abundance;
6. Incorporating Effects of Climate Factors;
7. Updated Assessment for Spatial Structure and Diversity;
8. Harvest Considerations;
9. Summary of Recommendations and a Decision Pathway for Recovery.

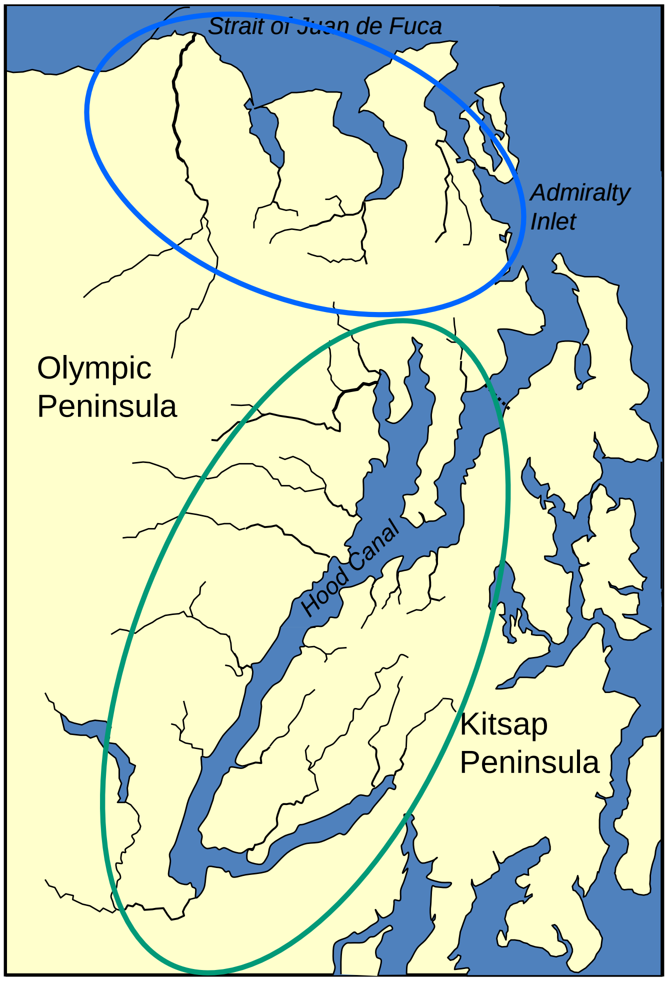

Figure 1. The two geographic regions where the natal streams of the two independent populations that comprise the Hood Canal Summer Chum ESU are located. The blue oval encompasses the geographic domain of the natal streams of the Strait of Juan de Fuca population. The green oval encompasses the natal stream domain of the Hood Canal population.

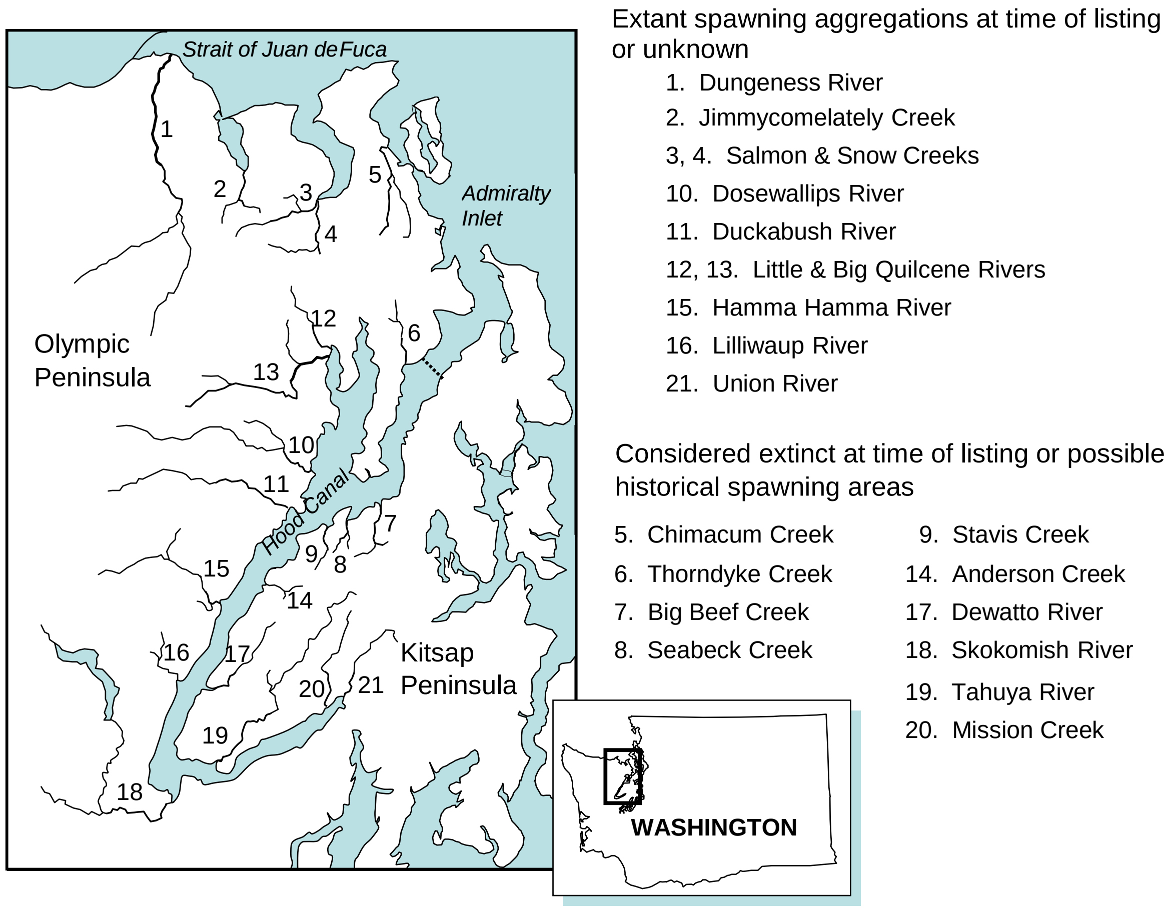

Figure 2. Extant and historical spawning distribution of the Hood Canal Summer Chum ESU as given in Sands et al. (2009). It should be noted that the Dungeness aggregation is misclassified, as its historic status remains uncertain. Taken from Sands et al. (2009).

2.0 Aspects of Recovery Goals

A salmon recovery goal typically includes two aspects needed for 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 for ecosystem services, such as harvest (McElhany et al. 2000; NMFS 2000).

NMFS is responsible for determining whether a salmon ESU warrants delisting from the federal ESA. Determinations for listing and delisting under the ESA are to be based solely on the best scientific and commercial data available after conducting a review of the status of the species and taking into account efforts being made to protect the species. A viable status for an ESU means that there is less than a 5% risk of extinction over 100 years. Broad-sense goals also need to be considered in determining whether delisting is warranted.

Recovery is described by NMFS as follows (McElhany et al. 2000):

> "NMFS and the U.S. Fish and Wildlife Service define recovery under the Act to be "improvement in the status of a listed species to the point at which listing is no longer appropriate" under the ESA (50 CFR S402.02). This indicates that there is a strong connection between listing and delisting criteria. Thus, the biological criteria used to make listing determinations should also be used to evaluate recovery. However, there are several reasons why listing and delisting criteria values should not be identical. First, if simple threshold values were used for setting criteria, a species that fluctuates around the critical value might require repeated listing and delisting actions even though its status had changed relatively little. Second, listing and delisting criteria require that population trends (and other factors) are considered in addition to abundance, and trends are expected to differ between declining and recovering populations. Finally, delisting should occur only in conjunction with an approved, comprehensive recovery plan that lays out conservation measures that address the factors that led to the initial decline and those that impede recovery. The preceding discussion applies to recovery as the ESA defines it. In addition, the NMFS, states, tribes, and many other stakeholders have an interest in recovering salmon populations to the point at which they can support sustainable harvest or other "broad sense" recovery goals (e.g., to produce fully functional ecosystems). Recognizing this, NOAA has made the following commitment (Garcia 1998):
>
>> *It is our policy that the recovery of salmonid populations must achieve two goals: 1) Restore salmonid populations to the point where they no longer require the protection of the ESA, and 2) restore salmonid populations to a level that allows meaningful exercise of tribal fishing rights. We see no conflict between the statutory goals of the ESA and the federal trust responsibilities to Indian tribes. Rather, the two federal responsibilities complement one another."*

A viable ESU is naturally self-sustaining with a negligible risk of extinction (<5%) over a 100-year time period. Viability criteria given by NMFS consist of four characteristics of the ESU's performance: abundance and productivity among the populations that comprise the ESU, biological diversity within and among the populations, and spatial structure (distribution pattern) over the geographic area of the ESU (McElhany et al. 2000). All four of these criteria need to be sufficiently met to achieve a negligible risk of extinction and, hence, a viable ESU.

Regarding broad-sense goals, it bears noting that since the recovery plan was prepared for the summer chum ESU in 2005, explicit broad-sense goals have not been defined by the state and tribal co-managers. However, it is understood that full recovery must provide for harvest. The clearest goal statement for broad-sense recovery for the ESU was given in the recovery plan (HCCC 2005), which stated the goal to be:

> *"To protect, restore and enhance the productivity, production and diversity of Hood Canal summer chum salmon and their ecosystems to provide surplus production sufficient to allow future directed and incidental harvests of summer chum salmon."*

This report provides information that can be used to refine this goal statement if needed.

3.0 History of Recovery Planning for the ESU

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

3.1 Events Leading to ESA Listing

Beginning in the late 1970s, summer chum returning to the Hood Canal and 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%, and averaged close to 60% 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 recent range of 3 to 15% for the Hood Canal population and to less than 2% for the SJDF population (PNPTT and WDFW 2014) (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[5] 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.

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[5] / Supplementation is defined as "the use of artificial propagation in the attempt to maintain or increase natural production while maintaining the long term fitness of the target population, and keeping ecological and genetic impacts on non-target populations within specified biological limits" (RASP 1992). Outplanting of salmon fry for the purpose of reintroducing a species where an extirpation has occurred is included in this definition—see RASP (1992).

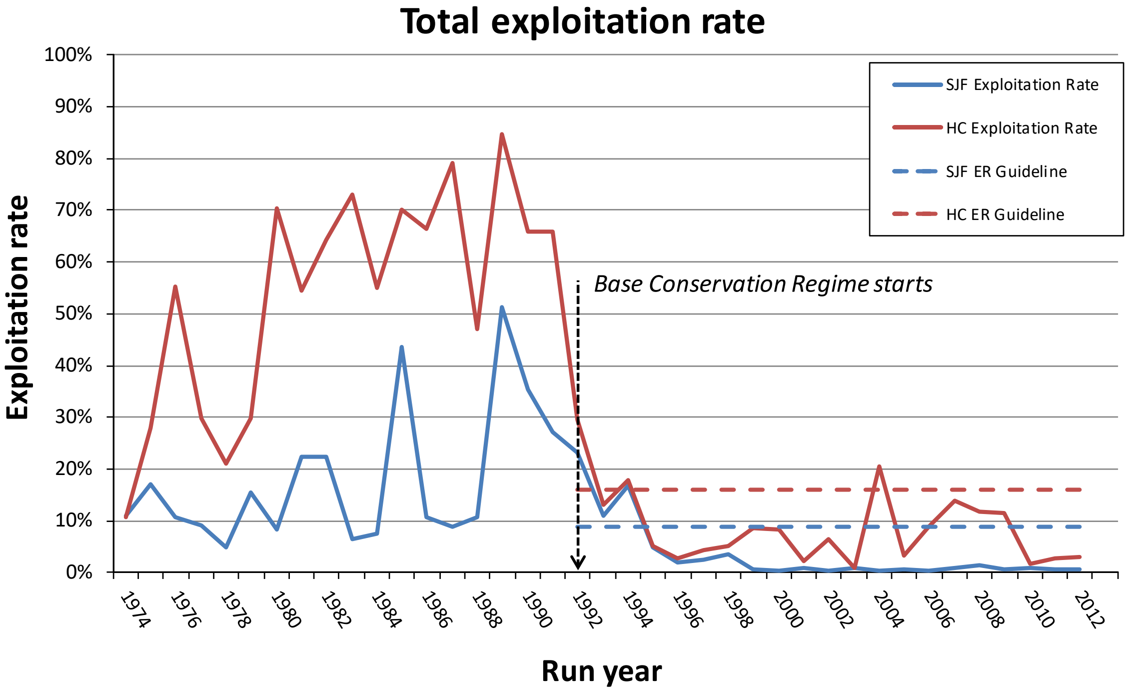

Figure 3. Estimated total exploitation rates on SJDF and Hood Canal summer chum in 1974 to 2012 (PNPTT and WDFW 2014).

3.2 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 PSTRT was charged with identifying appropriate conservation units (i.e., independent populations) within the ESU and for developing recovery criteria consistent with 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 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 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). As noted earlier, the viability analysis done by the PSTRT and reported on in Sands et al. (2009) is considered part of the federal supplement.

3.3 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 SCSCI[6]) 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. Thus, the goals also 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.

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[6] / 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.

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. 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 4 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 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 (i.e., 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.

4.0 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 recovery criteria for the ESU as presented in Sands et al. (2009).

4.1 Spatial Structure and Diversity of the ESU

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 Viable Salmonid Population (VSP) parameters that define viability (McElhany et al. 2000).[7]

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:

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[7] / The four VSP parameters are abundance, productivity, spatial structure, and diversity.

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 Strait of Juan de Fuca;
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 eastern 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.[8]

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.[9] Similarity relationships, as seen in the dendogram in Figure 4, show a “chaining” pattern for summer chum comprising the ESU, which is a pattern where the dendogram forms successive additions of branches associated with individual spawning aggregations rather than as distinct clusters.[10] 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.

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[8] / 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.

[9] / An isolation-by-distance pattern was subsequently supported by additional genetic analyses reported in Small et al. (2009) and Small et al. (2013).

[10] / A dendogram is a tree diagram that serves as a visual representation of the relationships between populations or subpopulations within a species.

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).

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 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.

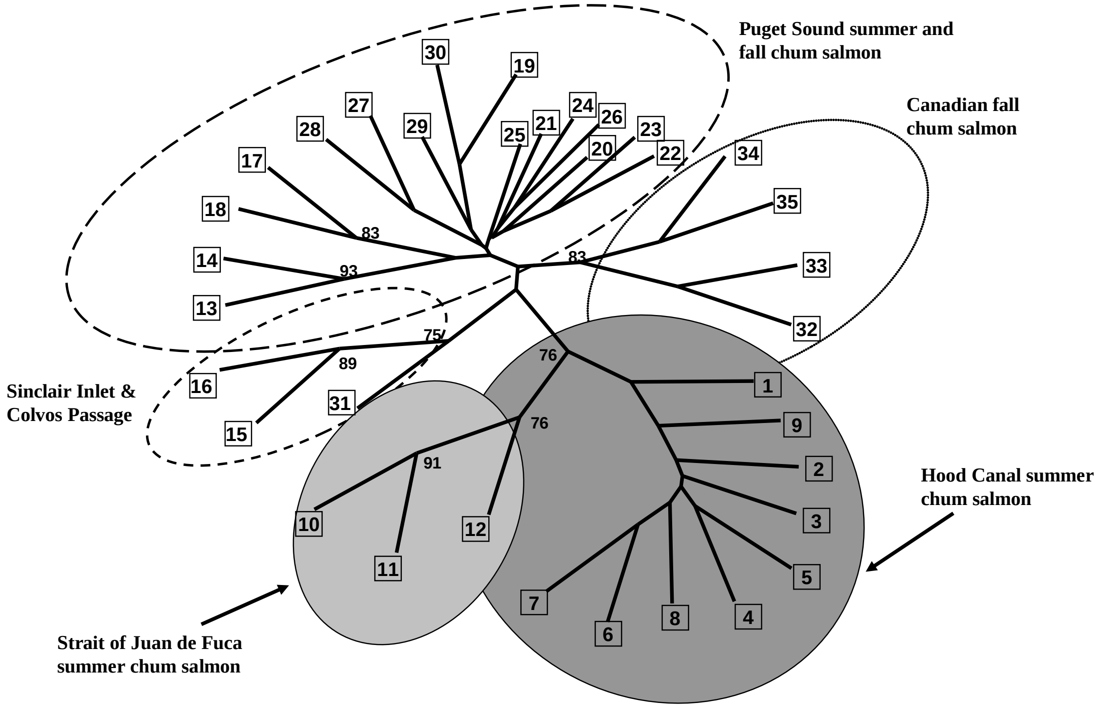

Figure 4. Relationship of 35 summer and fall chum salmon populations using Cavalli-Sforza and Edwards (1967) chord distance and neighbor-joining clustering (modified from Kassler and Shaklee 2003). Dendrogram shows chaining of Hood Canal summer chum salmon samples. Key to summer chum salmon samples: 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).

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. Moreover, some subpopulations were likely very large in some years and were thus capable of functioning as source populations for smaller systems within the ESU through both density-dependent and density-independent processes. 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 remained at the time the PSTRT did its work, with a ninth possible aggregation in the Dungeness River. There was uncertainty about whether the Dungeness River represented a distinct subpopulation or whether it more aptly should be considered a minor spawning aggregation within the SJDF population (Sands et al. 2009). Based on the data available to the PSTRT in the mid-2000s, at least ten spawning aggregations were believed to 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, has been 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 was estimated to be intermediate between timing patterns for eastside Hood Canal and SJDF streams. These 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. More recent efforts to trap and enumerate outmigrant summer chum in several streams (Hamma Hamma River, Duckabush River, and Salmon Creek) have generally supported these findings (Weinheimer 2013; Weinheimer et al. 2017), though a full analysis of timing differences has not been completed.

  It is likely that the 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.

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 1 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, together with depressed abundances of populations that may historically have served as source populations, have fragmented the ESU’s structure, resulting in an increased risk of population extinction.

Table 1. 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*

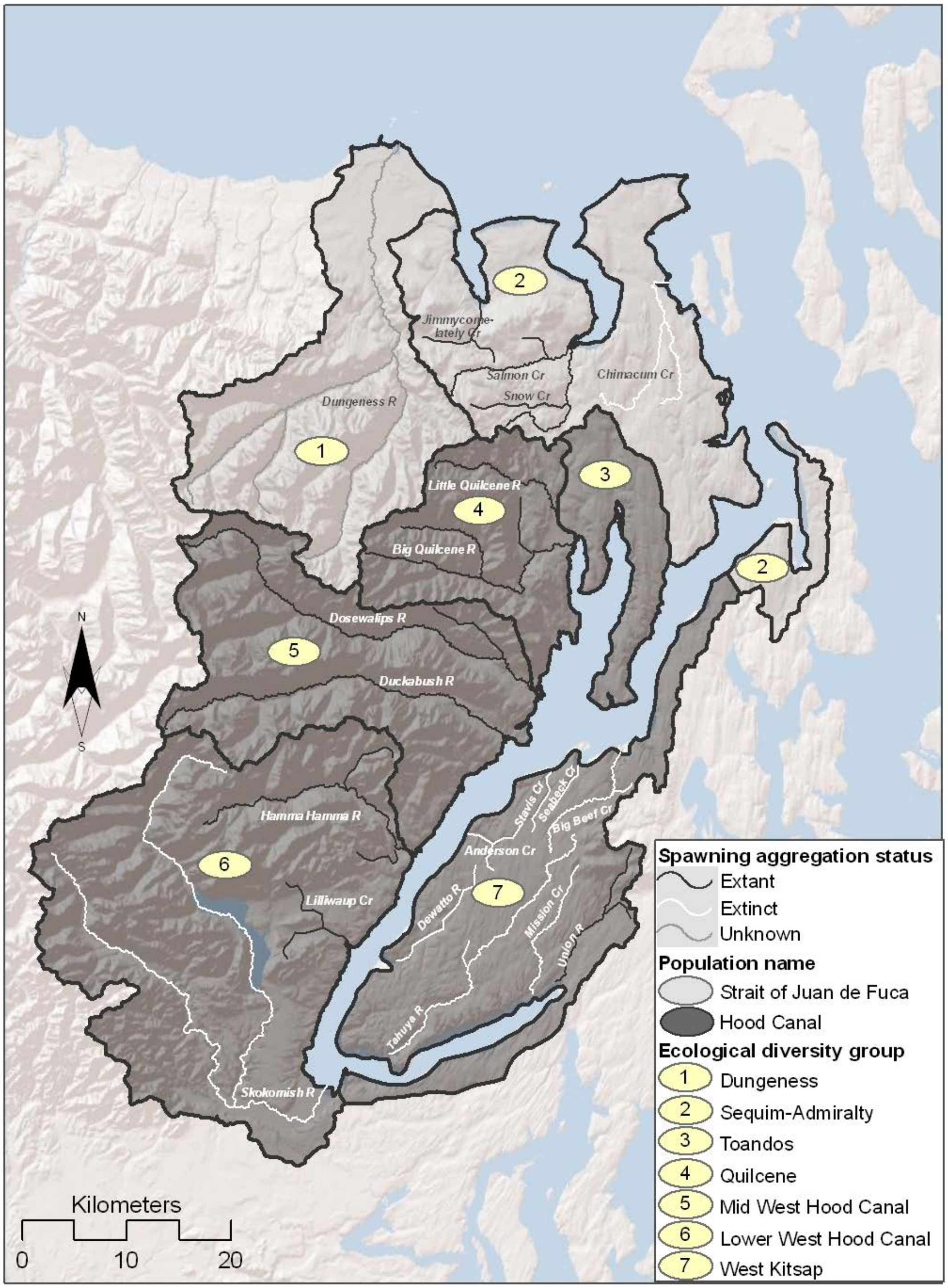

Figure 5. The seven ecological diversity groups delineated by the PSTRT for the Hood Canal summer chum ESU. Taken from Sands et al. (2009).

4.2 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 returning in 1974-2016, 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 the eight subpopulations considered as extant at the time the PSTRT did its work, i.e., up to the issuance of Sands et al. (2009) on the right side of the figure. The number of adult recruits in a run year (i.e., the year of return to the spawning grounds) is the sum of the number of spawners and the number of fish harvested belonging to the same subpopulation (or population).[11] 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 (including reintroduction) programs.

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 both populations has generally produced greater abundances relative to the 1970s.

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 2016.

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 extirpation of the indigenous fish. A supplementation program, begun on Salmon Creek with brood year 1992 and ended with brood year 2003, 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. Only native Salmon Creek stock was used in the program. Supplementation fish were never directly released into Snow Creek, though strays from the Salmon Creek program have spawned in Snow Creek. The reintroduction program in Chimacum Creek was initiated with brood year 1996 and ended with brood year 2003; returning summer chum to Chimacum Creek are considered a range extension of the Salmon-Snow Creek subpopulation. A supplementation program on Jimmycomelately Creek, using indigenous stock, was carried out for brood years 1999 through 2010. 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

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[11] / The number of recruits produced in each run year is estimated through a process called “run reconstruction”, which is completed by the co-manager biologists.

start (in 1999) of a supplementation program and the first age 3 returns from the program in 2002. It should be noted that substantial habitat restoration work was done in both Jimmycomelately and Salmon creeks in the early to mid-2000s with some work continuing in Salmon Creek after that time. A relatively minor amount of restoration work has been done in Snow Creek.

- 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 is similar to numbers in Jimmycomelately Creek in recent years. It should be noted that routine spawner counts began in Chimacum Creek in the 1990s.

It bears noting that the pattern of increased run sizes of natural-origin fish for the Salmon-Snow Creek subpopulation since the downturn seen in the 1990s has been much stronger in Salmon Creek than Snow Creek, as seen below:

Years	Salmon Cr average run size	Snow Cr average run size
1974 - 1988	1,118	512
1989 - 2000	537	42
2001 - 2016	2,954	444

We attribute the much stronger increase in Salmon Creek to both the supplementation program and a much larger habitat restoration effort than in Snow Creek.

Figures 8-9 show numbers of adult recruits for the six Hood Canal subpopulations considered extant when the PSTRT did its work (charts arranged from north to south). Supplementation using native brood stock from the respective streams has occurred for four of the subpopulations. Only Duckabush and Dosewallips have had no direct supplementation programs, though supplementation-origin strays have spawned in both streams (Figure 8). Due to continued low spawner returns combined with lack of focused habitat protection or restoration efforts, supplementation has been on-going in Lilliwaup Creek up through brood year 2017. Of the six natal streams and stream-mouth estuaries for these subpopulations, habitat restoration work has been most focused in Big and Little Quilcene, Dosewallips, and Union rivers. Little or no restoration work has occurred in Duckabush and Hamma Hamma rivers and Lilliwaup Creek.

Figures 8-9 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. Supplementation has continued to the present time in Lilliwaup Creek due to the extremely low run sizes that occurred.

- 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 the two Hood Canal subpopulations considered to have been extirpated by the PSTRT where reintroductions have occurred. 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 and ended in 2014. Both Big Beef Creek and Tahuya River have undergone substantial changes in channel characteristics since the 1970s due to watershed alterations. Limited restoration work has occurred in both streams, though several substantial restoration projects were completed in lower Big Beef Creek over the past five years. Additional work in lower Big Beef Creek is being planned. Figure 10 shows the following:

- Both of these 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 subpopulations generally mirrored those seen in the other subpopulations except in Union River—though for these two, production has not rebounded except for small responses likely as a result of the reintroduction efforts. We note that adult returns to Tahuya River, including natural-origin fish, exhibited a comparatively substantial increase in 2016.[12] Returns from the last brood year of supplementation in Tahuya River will largely end with the adult summer chum returning in 2018.
- 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). Although the reintroduction programs have resulted in some increases in abundance in both streams, natural-origin production does not appear to being sustained. This indicates that the quality of available habitats is generally poor and that more habitat protection and restoration actions are needed. We note that the recent restoration projects in lower Big Beef Creek were done well after the reintroduction program was terminated. Another attempt at reintroduction may soon be appropriate because of the recent restoration work done there—this is further discussed in Section 7.0.

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[12] / Preliminary data for 2017, not shown in Figure 10, show continued improvement in adult returns to Tahuya River in 2017.

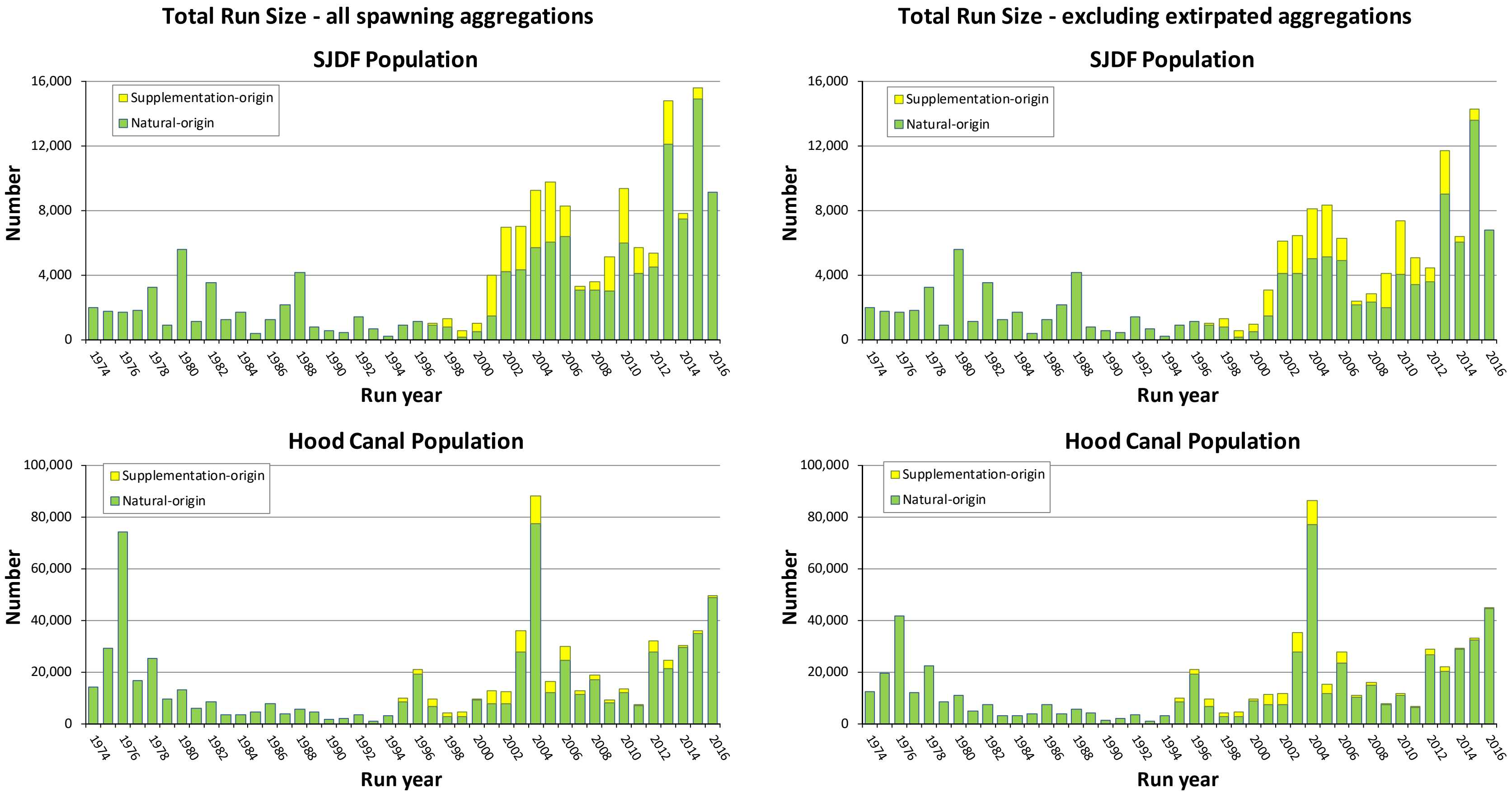

Figure 6. Left side – Estimated numbers of adult recruit summer chum for the two populations within the Hood Canal Summer Chum ESU returning by year in 1974 to 2016, incorporating all spawning aggregations for which data exist, extirpated and extant. Right side – Estimated numbers of adult recruits incorporating only data for extant subpopulations. Recruits of naturally spawned (natural-origin) and hatchery spawned (supplementation-origin) fish are shown. See Figure 2 for a list of subpopulations considered as either extant or extirpated at the time of the PSTRT's evaluation (Sands et al. 2009). Note that the y-axis scale differs between charts.

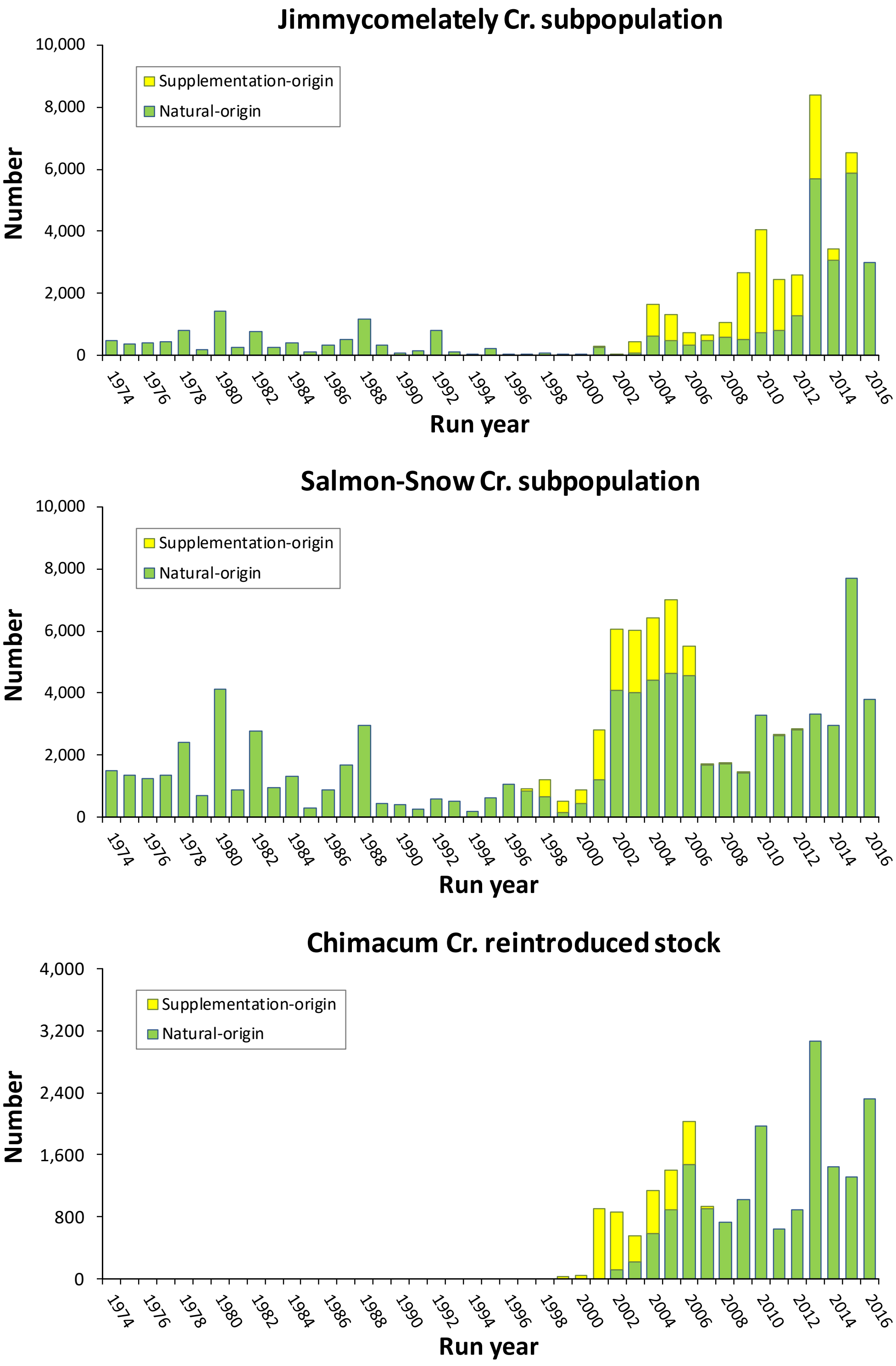

Figure 7. Estimated numbers of adult recruit summer chum comprising the Jimmycomelately Creek and Salmon-Snow Creek subpopulations of the SJDF population returning by year in 1974 to 2016, and estimated adult recruits resulting from reintroduction efforts in Chimacum Creek. The number of adult recruits represents fish returning to spawning streams plus estimated harvest in the return year. Recruits of naturally spawned (natural-origin) and hatchery spawned (supplementation-origin) fish are shown. Note that the y-axis scale differs among charts.

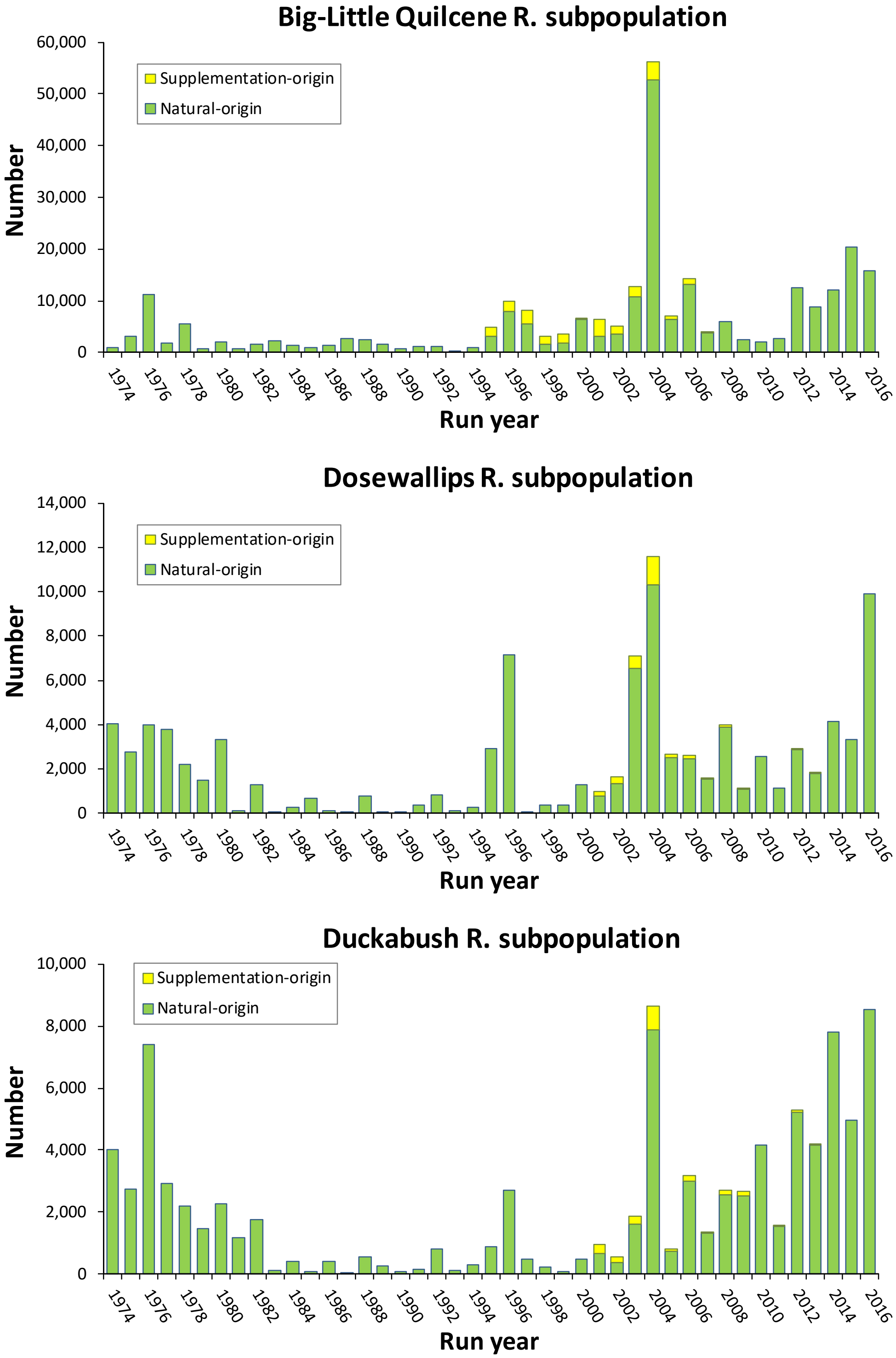

Figure 8. Estimated numbers of adult recruit summer chum comprising the Big-Little Quilcene, Dosewallips, and Duckabush rivers subpopulations of the Hood Canal population returning by year in 1974 to 2016. The number of adult recruits represents fish returning to spawning streams plus estimated harvest in the return year. Recruits of naturally spawned (natural-origin) and hatchery spawned (supplementation-origin) fish are shown. Note that the y-axis scale differs among charts.

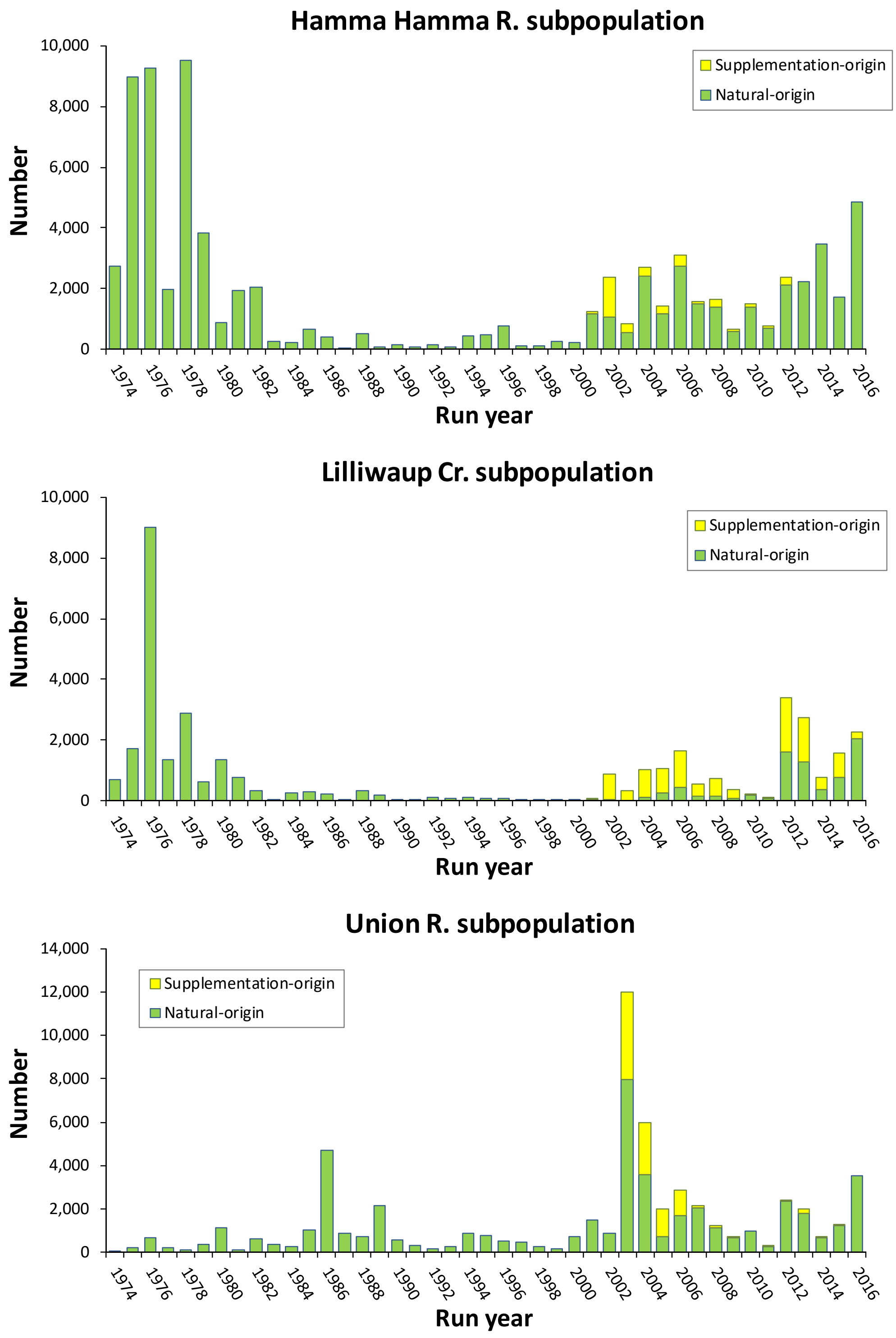

Figure 9. Estimated numbers of adult recruit summer chum comprising the Hamma Hamma River, Lilliwaup Creek, and Union River subpopulations of the Hood Canal population returning by year in 1974 to 2016. The number of adult recruits represents fish returning to spawning streams plus estimated harvest in the return year. Recruits of naturally spawned (natural-origin) and hatchery spawned (supplementation-origin) fish are shown. Note that the y-axis scale differs among charts.

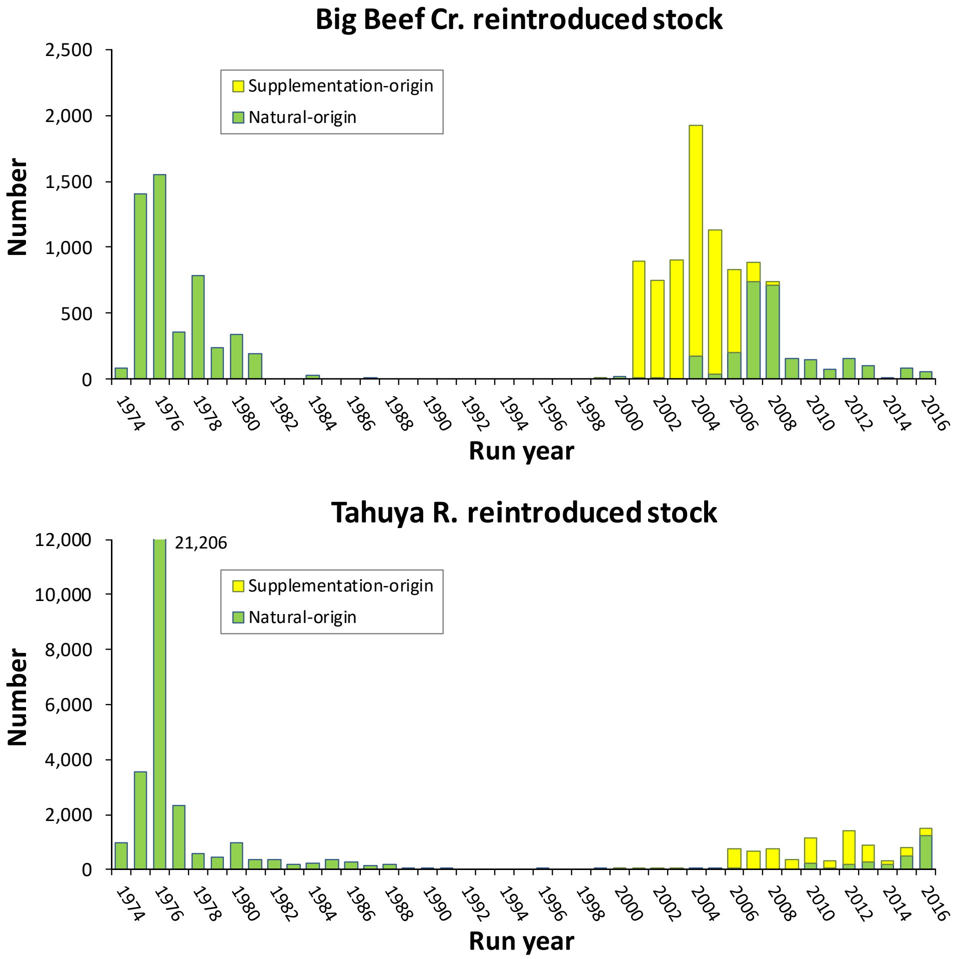

Figure 10. Estimated numbers of adult recruit summer chum resulting in Big Beef Creek and Tahuya River subpopulations returning by year in 1974 to 2016. Results of reintroduction efforts in both streams are also shown. The number of adult recruits represents fish returning to spawning streams plus estimated harvest in the return year. Recruits of naturally spawned (natural-origin) and hatchery spawned (supplementation-origin) fish are shown. Note that the y-axis scale differs between charts.

Figure 11 shows adult recruits for three of the Hood Canal spawning aggregations determined by the PSTRT to have been extirpated at the time of their analysis and where no reintroduction efforts have occurred. Two of the aggregations, Dewatto River and Skokomish River, were considered to be subpopulations, while the third, Anderson Creek, was probably a minor spawning aggregation due to the smaller size of its watershed. It is important to recognize that no spawning escapement data exist for the Skokomish River prior to 2001. 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.

During the past 12 years, restoration work in the river-mouth estuary of the Skokomish River has been extensive (SIT and WDFW 2017). Roads and dikes have been removed or breached, fill has been removed, large amounts of sediment have been removed or flushed out to Hood Canal, tidal channels have been opened or reformed, and estuarine marsh and wetlands have been restored.

No restoration work has occurred in Dewatto River or Anderson Creek. It bears noting that Dewatto River and its river-mouth estuary are relatively intact compared to habitats in lower Big Beef Creek and Tahuya River.

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 spawning aggregations mirrored those seen in the other subpopulations except for Union River.
- The escapement of summer chum in the Dewatto River remained less than 30 fish from 2000 through 2011 but increased to about 200 fish annually from 2012 through 2016. The number of spawners was estimated to be 300 fish in 2016, the largest number since 1978. Since 1995, when the first returns of supplementation fish returned to a river in Hood Canal (Big Quilcene River), the estimated percentage of stray supplementation-origin spawners in Dewatto River has averaged 23% based on otolith analysis. In 2016, only 3% of the spawners in Dewatto River were estimated to be stray supplementation-origin fish—the spawning escapement that year was estimated to be about 240 fish. The Anderson Creek spawning aggregation has demonstrated little or no increase in abundance since about 2000.
- The Skokomish River has demonstrated a dramatic increase in summer chum production in recent years. This upsurge may be partly due to supplementation-origin fish straying from programs in Lilliwaup Creek and Tahuya River nearby (PNPTT and WDFW 2014) and some fish straying from the Union River. It is clearly evident, however, that most of the summer chum being observed in the Skokomish River are being produced there based on otolith analysis and the very large numbers of fish seen recently. From 2003 to 2011, the estimated number of summer chum spawning in the Skokomish River remained low, averaging 30 fish. Between 2012 and 2016, the number of spawners increased from about 500 fish to nearly 2,800 fish (with about 260 to over 2,600 natural-origin fish over this period). The preliminary estimate of spawner abundance in 2017 is approximately 4,400 fish, the large majority of which is believed to have been produced in the Skokomish River. The Skokomish spawners are concentrated in the lower Skokomish River, but they also spawn in Vance Creek. Only small numbers of summer chum have been seen in lower North Fork. It bears noting that changes in the harvest management regime in the Skokomish River since 2015 may also have contributed to the increased spawner escapements in recent years (SIT and WDFW 2017). We believe that the extensive restoration effort in the river estuary has given a major benefit to the run.

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 subpopulations (those classified by the PSTRT as extant), continuing to the present time, though the response in Lilliwaup Creek has been slow. Two subpopulations, Skokomish and Dewatto, both classified as extirpated by the PSTRT, have also rebounded, most dramatically in the Skokomish River.[13] The rebound to date has been small on the Dewatto River. Neither of these two rivers has had a supplementation or re-introduction program.
- Some subpopulations that exhibited relatively strong adult production prior to about 1980 showed precipitous declines shortly afterwards and have not rebounded; these had been considered to have been extirpated, though small numbers of fish are periodically observed, perhaps due to straying.
- Of the three streams where reintroductions have been made, only Chimacum Creek has clearly sustained a strong positive response. The 10-year average of the number of natural-origin adults produced ending in 2016 was approximately 1,300 fish with a maximum of about 3,100 fish in 2013. The initial positive response in Big Beef Creek seen in 2007 and 2008 subsequently faltered after supplementation ended. The average number of natural-origin adults produced from 2009 to 2016 was less than 100 fish. The total number of summer chum adults that returned to Big Beef Creek in 2017 was seven fish. Of the three reintroduction efforts, the response of natural-origin production in the Tahuya River has been slowest to materialize, though it has increased recently. In 2016, an estimated 1,484 total spawners returned to the Tahuya River of which 1,212 fish were estimated to be natural-origin fish (82%). The preliminary estimate of total spawners that returned in 2017 is approximately 1,900 fish, suggesting that more than 1,300 may have been natural-origin fish based on the recent average composition of natural and supplementation-origin fish.

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[13] / Very small numbers of summer chum spawners were periodically observed in both the Skokomish and Dewatto rivers during the 1990s although both subpopulations had been classified by the PSTRT as extirpated.

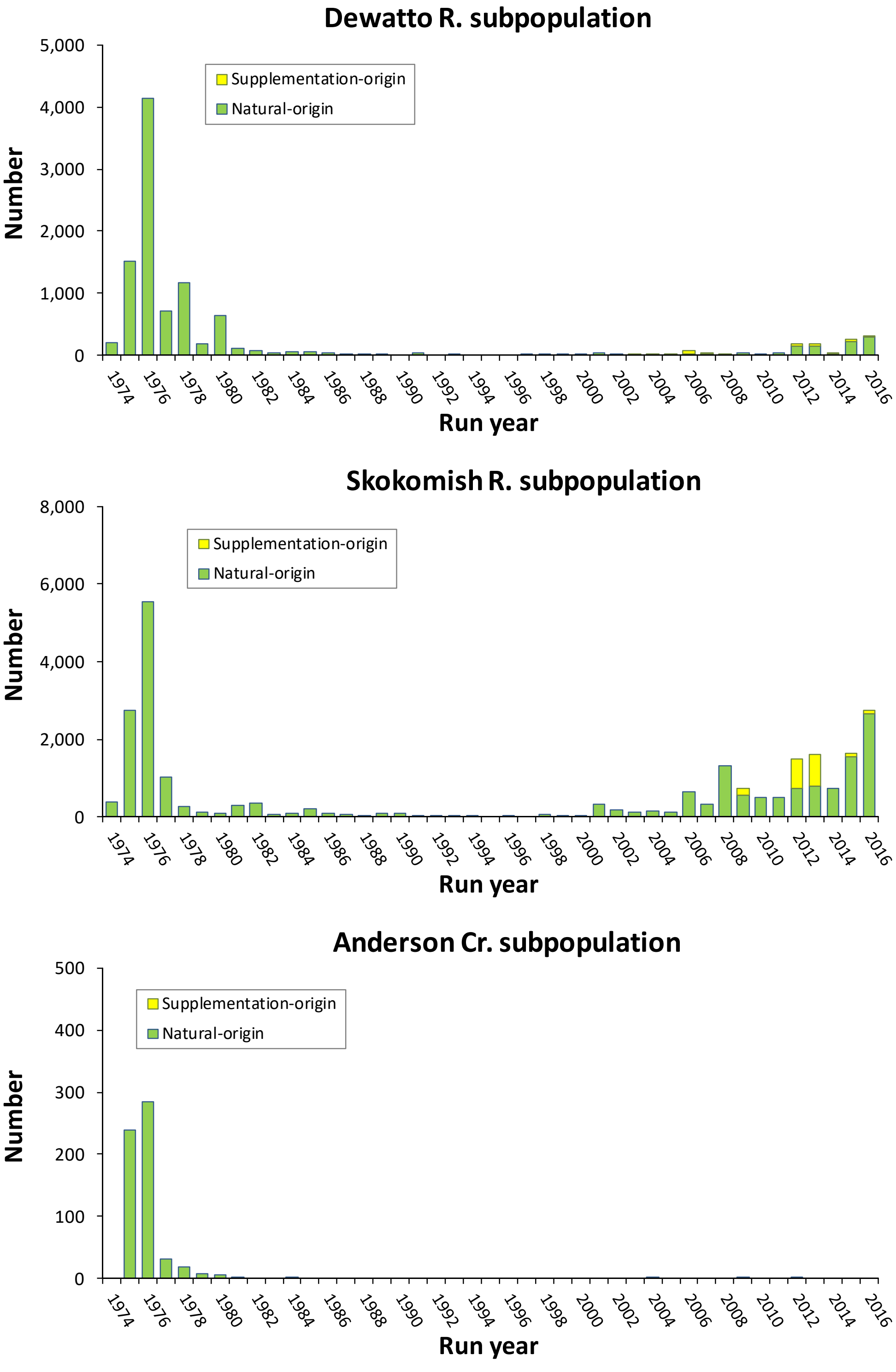

Figure 11. Estimated numbers of adult recruit summer chum comprising the Dewatto River, Skokomish River, and Anderson Creek spawning aggregations of the Hood Canal population returning by year in 1974 to 2016. The number of adult recruits represents fish returning to spawning streams plus estimated harvest in the return year. Recruits of naturally spawned (natural-origin) and hatchery spawned (supplementation-origin) fish are shown. Note that the y-axis scale differs among charts.

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. Run sizes in the most recent year prior to this report (2016) were strong.[14] 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 (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).

It bears noting here that these patterns of increasing abundance within this ESU over the past decade or longer are generally not evident for fall-run chum within the Puget Sound region as a whole, or in regions to the north. Of interest to this report are the conclusions of Malick and Cox (2016) in an analysis of productivity trends of chum salmon stocks in Washington, British Columbia, and Alaska. They concluded that widespread declines in productivity were evident throughout Washington and British Columbia with 81% of stocks showing recent declines in productivity, although the exact forms of declines varied by region. Particularly notable for our analysis is that three stocks produced from Washington streams, South Puget Sound winter chum (Nisqually), South Puget Sound fall chum, and Hood Canal fall chum demonstrated increasing productivity trends in recent years, in contrast to all other chum stocks assessed except for a few in Alaska. Their analysis did not include the summer chum populations within the Hood Canal Summer Chum ESU.

Malick and Cox (2016) suggested two possible reasons to explain the positive productivity trends for the three stock groups originating in the southern branches of Puget Sound: (1) differences in physical and biological conditions in different parts of Puget Sound favoring survival for these stocks compared to elsewhere or (2) restoration efforts having positive effects on these stocks. We believe it is likely that the positive productivity trend is due primarily to the first reason, as discussed in Section 6.0 of this document for summer chum.

4.3 Existing Recovery Criteria (from Sands et al. 2009)

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

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[14] / We note that the pattern of continuing strength in the rebound did not continue to 2017. The assessment of spawning escapements was completed by the time this report was written. Of particular interest is that the Hood Canal population continued its trajectory of increasing abundance into 2017 but the SJDF population did not. The SJDF population exhibited a sharp drop-off in abundance for each of the three subpopulations. These divergent patterns between the two populations are relevant to Sections 6.0 and 9.0 of this report.

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, the PSTRT identified abundance and productivity criteria to achieve a negligible risk of extinction for each of the two populations (Sands et al. 2009). These values were incorporated into the recovery plan supplement as issued by NMFS (NMFS 2007). 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 5,600 spawners for the SJDF population, and between 17,900 and 24,700 spawners for the Hood Canal population (Table 2).[15] Results shown in Table 2 are the extremes of the ranges using two analytical models, one based on an assumption that density-dependence is not occurring for summer chum (i.e., all survival is random or stochastic) and another model (VRAP) that assumes density-dependent survival. We presented in our earlier report (Lestelle et al. 2014) evidence for a strong density-dependent effect and concluded that the stochastic model is not appropriate to be applied. Therefore, all analyses used in our earlier report and in this one, employ the VRAP model. Using only the results from the VRAP model, the ranges represented in Table 2 would be revised to 4,500-5,400 for the SJDF population and 17,900-21,500 for the Hood Canal population. Further details about these quantitative viability criteria are described in Section 5.0.

Figure 12 displays the patterns of estimated spawning escapement for each population from 1974 to 2016 together with the minimum spawning escapement thresholds to achieve viability (<5% risk of extinction) based on abundance and productivity using the ranges in Table 2 (using two model approach). Recent spawning escapements for both populations generally exceed the upper ends of the threshold ranges (Table 2) to meet the 5% risk level.

We would note that the recovery criteria for abundance and productivity given in Sands et al. (2009) are to be applied solely at the population level—not at the ESU level. Moreover, no specific criterion for abundance is meant to be applied at the subpopulation level. Whether or not the abundance criterion is met for delisting is to be determined for each population.

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 4.1 (Table 1 and Figure 5).

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[15] / See Tables 5, 7, and 8 in Sands et al. (2009). Spawning escapements would need to be higher at higher exploitation rates. Table 5 is based on a density-independent survival model. Tables 7 and 8 employ VRAP, a density-dependent survival model.

Table 2. Top – Projected range of minimum spawning escapement thresholds needed to achieve <5% risk of extinction over a 100 year period for 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 PNPTC (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

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 (not going extinct), persistence does not carry the rigor of viability with <5% risk of extinction over a 100-year time frame 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.[16]

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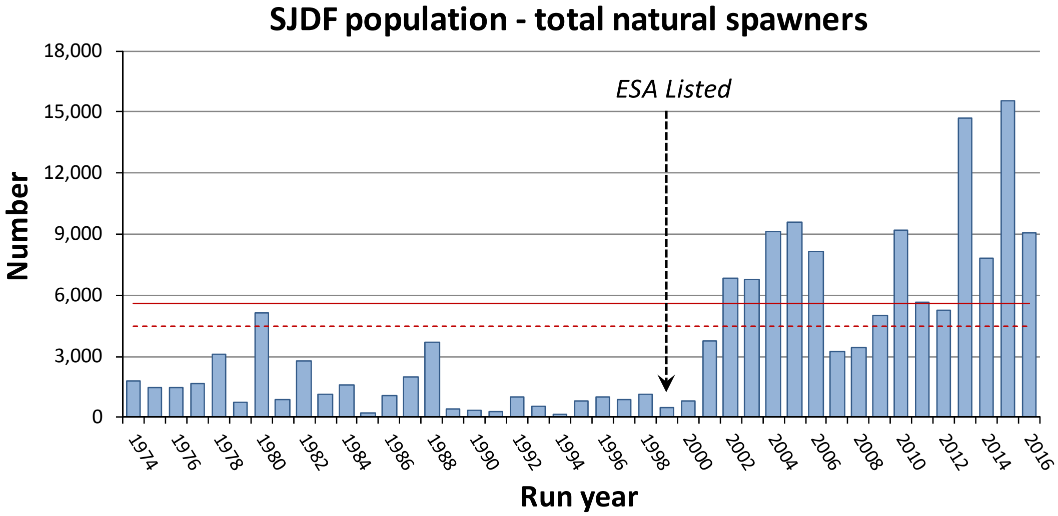
[16] / Fausch et al. (2006) provide some useful insights for understanding "persistence" within the context of this paper: "Often the terms "persistence" and "viability" are used interchangeably, but here we recognize viability as a larger conservation objective. The basic impetus for conservation planning is not simply to guarantee persistence



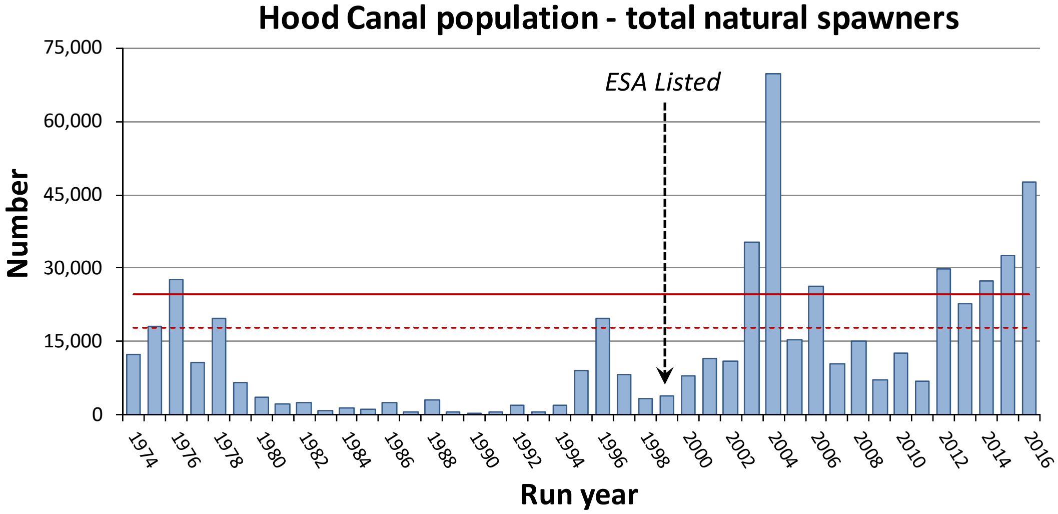


Figure 12. Estimated numbers of naturally spawning summer chum in the SJDF and Hood Canal populations in 1974 to 2016. The upper (solid red line) and lower (dashed red line) ends of the minimum spawning thresholds needed to achieve <5% risk of extinction as shown in Table 2 are displayed; those ranges are based on analyses in Sands et al. (2009). Note that the y-axis scale differs between charts.

For a subpopulation to be robust, therefore, we infer that it should demonstrate some level of vigor and health—which should be reflected in periodic positive growth rates for the subpopulation and maintenance of at least some average minimal abundance. This matter is further discussed in Sections

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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.”

5.1 and 7.1 of the report—we define robust for use in considering when the spatial structure viability criterion are met in Section 7.1.

The criteria for spatial structure and diversity given in Sands et al. (2009) were the following:

- 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;
- 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 was 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, that is, the subpopulations need to remain sufficiently robust to provide beneficial contribution at the population level.

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, i.e., <5% risk of extinction over 100 years (Sands et al. 2009).

5.0 Updated Viability Assessment for Abundance

This section provides an updated quantitative viability assessment from those presented in Sands et al. (2009) and Lestelle et al. (2014). The analysis presented here utilizes 11 and 6 more years of brood year data than were available to Sands et al. (2009) and Lestelle et al. (2014) respectively.

5.1 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, analysis of the combination of abundance and productivity is an effective means of evaluating risk separate from the other two population characteristics (diversity and spatial structure).

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 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 productivity rates at 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 captures aspects of both abundance and productivity.

We note here that the definition and use of the term "productivity" can differ in the salmon recovery literature. It is important to recognize the distinction in two uses of the term to understand its use here. 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. For the remainder of this paper, the term productivity will be used in the sense of intrinsic productivity, as it is usually 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 13A). 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, vary (on average) in relation to the size of the reproducing parent population (Ricker 1954; Beverton and Holt 1957; Ricker 1975). In Figure 13A, 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).

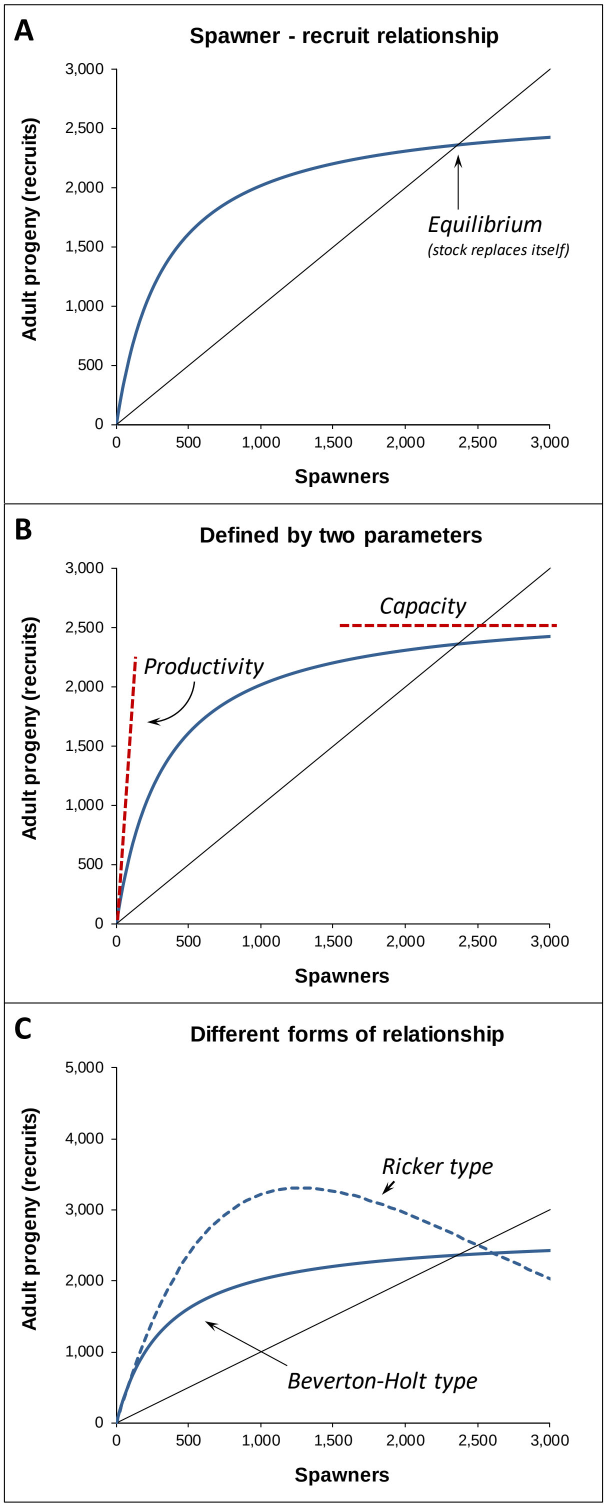

Figure 13. A - Relationship between spawner abundance and adult progeny (recruits) (A) with equilibrium abundance identified. B – The relationship is defined by two parameters: productivity and capacity. C – Two common forms of a spawner-recruit relationship, the Beverton-Holt, which rises to an asymptotic capacity, and the Ricker, which is dome-shaped.

The equilibrium abundance would be what would tend to be observed on the average over some period of years, if habitat conditions remain relatively constant and there is no harvest (Ricker 1975).[17] 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 defined by the two parameters capacity and productivity (Figure 13B) (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.[18] 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 13B. 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). Our interest here is 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; Mobrand 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

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[17] / This average under such conditions would be the geometric mean of observed run sizes. The arithmetic mean would 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).

[18] / 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. It is noted 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 13C). 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.

Another aspect of performance that affects viability is variation in production that occurs as a result of variability in natural processes among years. Relatively wide variation in production is typically seen in spawner and recruit abundance data sets (Figure 14). 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 (i.e., random) environmental and biological effects. The result is that empirical data can have a large amount of 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).

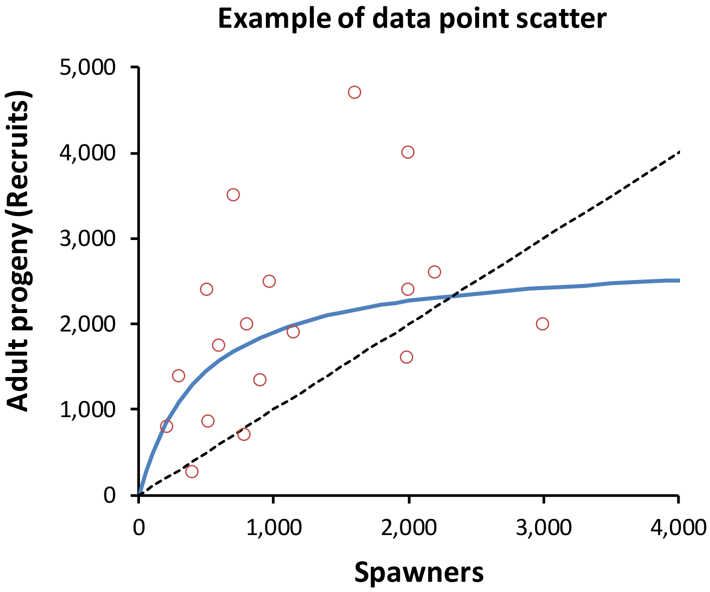

Figure 14. Hypothetical example of variation in recruit production around the underlying spawner-recruit relationship.

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 15). 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.

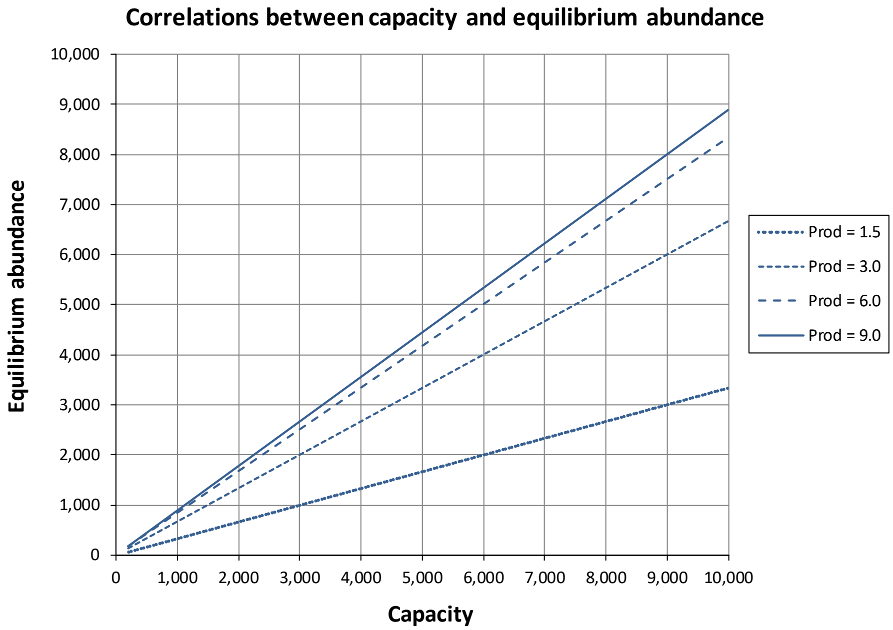

Figure 15. Correlation lines between capacity and equilibrium abundance at different levels of intrinsic productivity (Prod) with the Beverton-Holt form of the S-R curve.

An example of a viability curve is seen in Figure 16. Viability curves are generated using quantitative modeling incorporating performance characteristics associated with the population of interest. Figure 17A-B portrays two different population conditions, one in which the population would be expected to have a low risk of extinction (Figure 17A) and one with a high risk of extinction (Figure 17B). 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 LCWTRT (2003). Sands et al. (2009) expressed their viability curves using capacity.[19] Use of the capacity parameter is retained here for defining viability curves to remain consistent with Sands et al. (2009).

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[19] / 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.

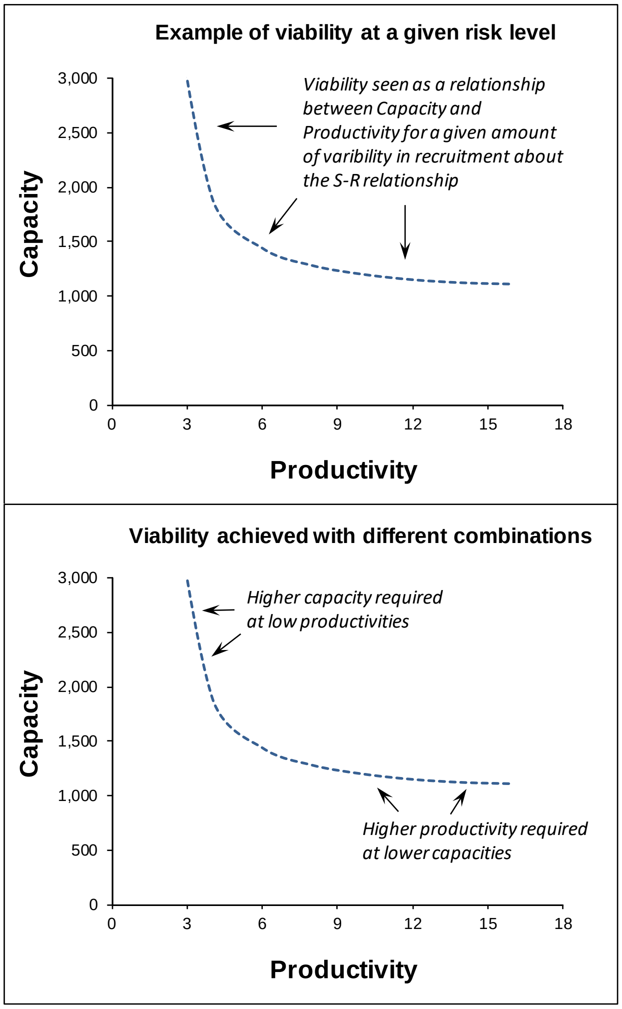

Figure 16. Example of a viability curve showing a relationship between capacity and productivity that defines a performance threshold for a salmon population associated with given level of extinction risk. At relatively low productivities, a higher capacity is required to maintain low extinction risk, whereas at higher productivities a lower capacity is required for the same level of extinction risk.

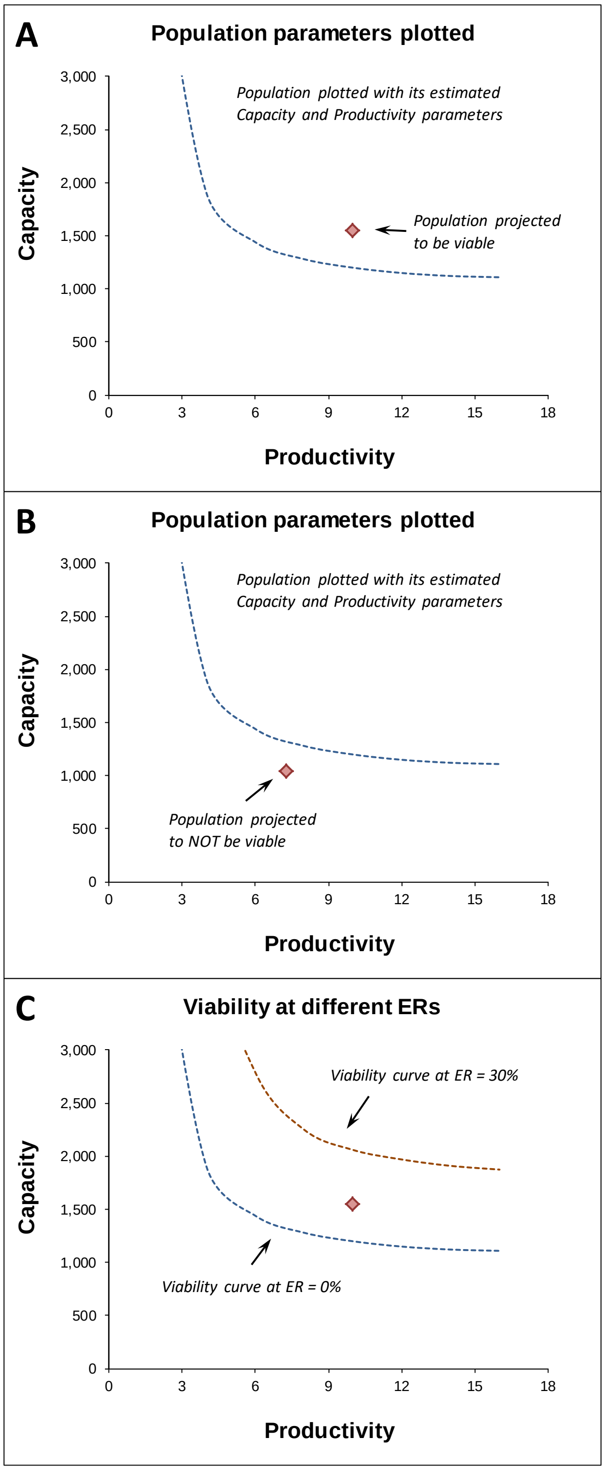

Figure 17. Hypothetical population performance plotted in relation to viability curves. A – The population is at low risk of extinction relative to the viability curve. B – The population is at high risk relative to the viability curve. C – The population is at low risk relative to the viability curve with an exploitation rate (ER) of 0 % but at high risk with an exploitation rate of 30%.

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 (ER). Figure 17C shows that the hypothetical population represented in the figure would have a negligible risk of extinction with an exploitation rate of 0%, whereas it would be at high risk of extinction under a harvest regime with a 30% 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%).

Viability curves are derived using quantitative modeling to perform a population viability analysis (PVA). A PVA uses quantitative methods to predict the likelihood that a population of concern will be above some minimum size or a risk level of extinction over some future period of years. 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.

Viability thresholds have been expressed in different forms or with different reference levels in conservation biology—it is useful for our application to mention several here. These reference levels are helpful to compare to the viability thresholds presented in sections of this report that follow. One rule-of-thumb that has been used to characterize viability is referred to as the 50/500 rule introduced by Franklin (1980). He suggested that as a general rule-of-thumb, in the short-term the effective population size N_e should not drop below 50 animals and in the long-term that it should not be less than 500.[20] The 50/500 rule has been widely debated by conservation biologists through the years (e.g., Flather et al. 2011; Jamieson and Allendorf 2012; Frankham et al. 2014)—Allendorf and Luikart (2007) provide helpful insights:

> "Nevertheless, we believe that the 50/500 rule is a useful guideline for the management of populations. Its function is analogous to a warning light on a dashboard of a car. If the N_e of an isolated population is less than 50, we should be concerned about a possible increased probability of extinction because of genetic effects…These numbers should not, however, be used as targets. When the low fuel light comes on in your car, you do not stop filling the fuel tank once the light goes off. It is also important to remember that 50/500 is based only on genetic considerations. Some populations may face substantial risk of extinction because of demographic stochasticity before they are likely to be threatened by genetic concerns (Lande 1988; Pimm et al. 1988)."

Recently, Frankham et al. (2014) advocated for increasing the two lower limits for the rule to 100/1,000. Franklin et al. (2014) argued that the 50/500 rule was still valid, though they acknowledged that problems exist with the "use of simple rules in a complicated world"—but recommended against changing the general rule-of-thumb. We note that we apply aspects of the 50/500 rule to define robust subpopulations in Section 7.1 as a way of determining when the spatial structure viability criterion are being met.

In the PVA modeling described in the following section, Sands et al. (2009) applied effective population sizes (N_e) of about 250 and 200 to the Hood Canal and SJDF populations respectively. These values were

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[20] / The term "effective population size" (N_e) is a central, theoretical concept in conservation biology—it defines the number of breeders in an "ideal" population that would have the same amount of inbreeding or genetic drift as in the actual population of interest when adverse effects of inbreeding would be absent (Allendorf and Luikart 2007). Thus an actual population smaller than the effective population size would be expected to be adversely affected by inbreeding effects.

used to estimate what is referred to as the quasi-extinction threshold (QET) for the modeling; the estimates were 350 and 300 spawners for each population respectively, as explained in Sands et al. (2009). The QET values are estimates of thresholds meant to represent levels where the populations might be expected to irreversibly decline to complete extinction. The reader needs to keep in mind that these values are for each population—they are not meant to reflect thresholds for the subpopulations. We also applied values for effective size (N_e) for each subpopulation in modeling done in 2014 and 2017 of 50 spawners, as also described in Sands et al. (2009).

One final note here about basic concepts is helpful in considering the uncertainty that exists about estimates of viability thresholds. Morris and Doak (2002), in their comprehensive book on population viability analysis, advised care in how results of viability analysis are viewed—and applied. They recommended that PVA results be viewed more qualitatively and not as absolute gauges of population status or probabilities of extinction. Uncertainties exist in the data and in the modeling techniques. They stated:

> “The last point we wish to make is that constructing a PVA is like building a house—the work is never really completed. Additional data are constantly being accumulated, in part thanks to basic monitoring…As a result, the quality of previously performed viability assessments is likely to improve as additional data are collected and incorporated into the model. More data allow us to improve the accuracy of the estimates for parameters currently in the model, to reconsider basic assumptions we made in constructing the PVA, and to incorporate more factors into the model.”

5.2 Previous Assessment (Sands et al. 2009)

Sands et al. (2009) employed a PVA model called the Viability and Risk Assessment Procedure (VRAP) to assess viability risk for both populations in the ESU.[21] 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, a range of target exploitation rates, variability around the S-R relationship, and a quasi-extinction threshold.[22]

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. 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[23] 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)[24], which is easier to understand in comparing variability between populations and subpopulations.

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[21] / Sands et al. (2009) also used a second PVA model to investigate viability risk. This alternative approach assumed density-independent mortality only, an unreasonable assumption given the clear effect of density-dependence operating on the summer chum population; see discussion in Lestelle et al. (2014). Results using this alternative model are not shown in this report.

[22] The quasi-extinction threshold is 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).

[23] / A type of statistical probability distribution.

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

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.

The VRAP model was run to assess the rate of extinction for a population over 100 years under different combinations of intrinsic productivity and capacity using the estimates of variability and a range of exploitation rates. The output from VRAP specifies the minimum capacity values needed to keep the risk of extinction to 5% 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 17.

VRAP output also provides for each combination of capacity, productivity, and exploitation rate values the associated average 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. (It is important to note here that Sands et al. reported their findings for spawning escapement as the arithmetic average escapement.)

The reader should refer to Sands et al. (2009) and Lestelle et al. (2014) for further details on the methods used with VRAP.

An abbreviated summary of results from the VRAP analysis in Sands et al. (2009) is given in Table 3. These results utilize population data for brood years 1974-2001. The results 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.

Table 3. Minimum abundance viability thresholds (5% risk of extinction over 100 years) for the SJDF and Hood Canal populations of summer chum as given in Sands et al. (2009) derived with VRAP modeling. The results are shown as a range, based on different values for productivity (P) that bracket a reasonable range of values for each population. The results are shown with two exploitation rates (ER): 0% and 10%. Data for brood years 1974-2001 were used in the modeling.

Population	ER	Range of average escapements		Capacity range	
		Low	High	Low	High
		P=6	P=3	P=6	P=3
SJDF	0%	4,700	5,100	3,300	4,300
	10%	4,600	5,400	3,700	5,300
		P=9	P=5	P=9	P=5
Hood Canal	0%	17,900	20,600	13,000	17,000
	10%	18,600	21,500	15,500	20,500

It is important to note that VRAP identifies viability as a spawner-recruit function defined by productivity and capacity parameters. Both parameters together define the S-R relationship—and both parameters are the result of habitat conditions. The modeling is used to assess ranges of combinations of parameter values that would be consistent with exceeding thresholds for low risk of extinction. For example, the capacity threshold for the SJDF population of 3,300 in Table 3 corresponds to a productivity of 6. If the productivity was only 3, then the larger capacity of 4,300 is needed to be viable.

The results for spawning escapement in Table 3 require careful consideration and explanation to make sense. Focusing on SJDF population with an ER of 0%, the capacity (C) is 3,300 at a productivity (P) of 6. One can calculate the equilibrium abundance (Neq) with these two parameters as

$$\text{Neq} = C - C/P = 2{,}750$$

This result raises the question: How could the average expected escapement associated with this productivity and capacity be 4,700 as seen in the table, which is larger than both the Neq value and the capacity value?

If the model was entirely deterministic (i.e., no stochastic elements) with no variability, then the spawning escapement would be equal to the Neq value or 2,750. However, the model incorporates lognormal variability, which as noted earlier will skew average recruitment high. The escapements given in Table 3 are the arithmetic average (not the geometric mean, see footnote 19) 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.[25] 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 report.

As one adds harvest into the analysis, 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%. 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 spawning escapement modeled using VRAP with variation included would drop slightly to 4,600.

No attempt was made by Sands et al. (2009) to quantify viability thresholds for individual subpopulations because recovery as evaluated by the abundance criterion is to only be determined at the population level.

5.3 Updated Assessment

The viability analysis was updated using data through brood year 2012 and the same methods employed by Sands et al. (2009) and Lestelle et al. (2014).

A comparison of the estimates of process error variation (CVs) between the earlier assessments and this update is given in Table 4. Changes in the CVs in the two populations were evident again in this update from the previous one in similar magnitudes as seen between the analyses in Sands et al. (2009) and Lestelle et al. (2014). The CV increased slightly from the 2014 analysis to this update for the SJDF population, but dropped substantially for the Hood Canal population. Similar amounts of change were seen between the 2009 and 2014 analyses.

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[25] / This point may be confusing to the reader not familiar with the modeling concepts described here. VRAP as it is currently structured incorporates lognormal variability but only reports the arithmetic mean of expected spawning escapements. An updated version of the model would also report the geometric mean.

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 1974-2012 brood year data.

Population	Assessment	BY	CV
SJDF	Sands et al. 2009	74-01	107%
	Lestelle et al. 2014	74-06	111%
	2017 update	74-12	113%
Hood Canal	Sands et al. 2009	74-01	134%
	Lestelle et al. 2014	74-06	120%
	2017 update	74-12	105%

The changes in CVs between the two earlier analyses 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 three assessments and is consistent with empirically derived estimates of the parameter for the two populations. The updated assessment produces viability thresholds for the SJDF population moderately higher 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 in this updated analysis compared to both of the earlier analyses. For comparison to the escapements shown in Table 5, average spawning escapements of natural-origin spawners (excluding supplementation-origin fish) observed for the two populations during three time periods are given in Table 6.

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 (2017 update). ER is exploitation rate and P is intrinsic productivity. Escapement values are arithmetic means[26] as in Sands et al. (2009). Results from Lestelle et al. (2014) are also shown for comparison.

Population	ER	Assessment	Range of average escapements		Capacity range	
			Low	High	Low	High
			P=6	P=4	P=6	P=4
SJDF	0%	Sands et al. 2009	4,700	4,800	3,300	3,700
		Lestelle et al. 2014	5,700	6,200	5,100	6,300
		2017 update	6,300	6,600	4,600	5,400
	10%	Sands et al. 2009	4,600	5,100	3,700	4,500
		Lestelle et al. 2014	5,600	6,100	5,800	7,100
		2017 update	6,600	7,000	5,400	6,400
			P=8	P=6	P=8	P=6
Hood Canal	0%	Sands et al. 2009	18,300	19,100	13,500	15,000
		Lestelle et al. 2014	8,700	9,100	7,000	7,800
		2017 update	4,800	4,900	3,600	3,900
	10%	Sands et al. 2009	18,300	20,400	15,500	18,500
		Lestelle et al. 2014	8,700	9,600	8,000	9,300
		2017 update	5,000	5,100	4,200	4,500

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[26]/ The arithmetic mean is skewed high (by up to 40% or more) due to the lognormal distribution of observed escapements compared to the geometric mean, which is equivalent to what this report refers to as equilibrium abundance.

Table 6. Average numbers of natural-origin spawners for the SJDF and Hood Canal populations for three time periods. Averages are shown for both the arithmetic mean (AM) and geometric mean (GM).

Population	Years	AM	GM
SJDF	1990-1999	660	563
	2000-2009	5,674	4,723
	2010-2016	9,612	8,912
Hood Canal	1990-1999	5,027	2,662
	2000-2009	21,000	16,089
	2010-2016	25,697	22,029

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 both the 2014 and 2017 updates were more within the usual range of data, which has dampened the CV. Such an extreme outlier brood year has not occurred within the data record for the SJDF population, and therefore the amount of change in the CV between the different analyses has been much less. (In Section 6.1.2.1, Figure 22 shows that a continued recent increase in adult recruitment relative to the mean is also causing a gradual increase in the CV for the SJDF population as seen in Table 4.)

It bears noting that the much lower CV for the Hood Canal population in the most recent analysis compared to the CV for the SJDF population is likely due in part to the number of subpopulations that comprise the population. Ten different spawning aggregations contribute to the population size in Hood Canal whereas only three contribute to the size of the SJDF population (under current conditions). The larger number of contributing spawning aggregations in Hood Canal tends to buffer interannual variation in population size (a portfolio effect). Lacking the same extent of such an effect, the fewer contributing aggregations in the SJDF population results in greater variation in the data used to define the S-R relationships for this population, which is the basis for calculating the CV.

In reviewing the results shown in Table 5, note that the estimates of spawning escapement shown are meant to only reflect expected numbers of spawners for a given level of viability risk. The values and the thresholds do not mean that the populations would spiral toward extinction if population performances were to drop below those levels. The values shown also do not reflect management goals—they are not aspirational spawning escapement goals that might be targeted to yield desired harvest levels. Tables 5 and 6 considered together provide a gauge to help evaluate the status of the populations relative to risk levels.

Viability curves for the two populations using the updated assessment, shown with exploitation rates of 0 and 30%, are provided in Figure 18. Our estimates of productivity and capacity for the populations using all data for brood years 1974 to 2012 are also shown plotted (filled diamonds). 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%. For return years 2000 through 2016, exploitation rates averaged about 7.4% for Hood Canal summer chum and 0.6% for SJDF summer chum (Scott Bass, PNPTC, personal communications; see also Table 10 in this report).

Viability curves (5% risk) for the eight subpopulations classified as extant by the PSTRT, shown with exploitation rates of 0 and 30%, are provided in Figure 19. Estimates of productivity and capacity for the subpopulations using all data for brood years 1974 to 2012 are also plotted on the graphs (filled diamonds). These graphs and data are provided to illustrate the amount of differences that are seen in productivity and capacity among the subpopulations. The reader needs to keep in mind that the standard for viability for the ESU is measured at the population scale, not at the subpopulation level.

The results suggest that all of the subpopulations are at relatively high risk of extinction when considering all of the years in the data series, except for the Union and Quilcene subpopulations. Results for several of the subpopulations suggest that they are very close to the 5% risk level. 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. The Quilcene subpopulation, comprised of fish produced in both the Big and Little Quilcene rivers, has a relatively high capacity compared to other streams in the ESU; this subpopulation may generally act as a core subpopulation to the Hood Canal population.

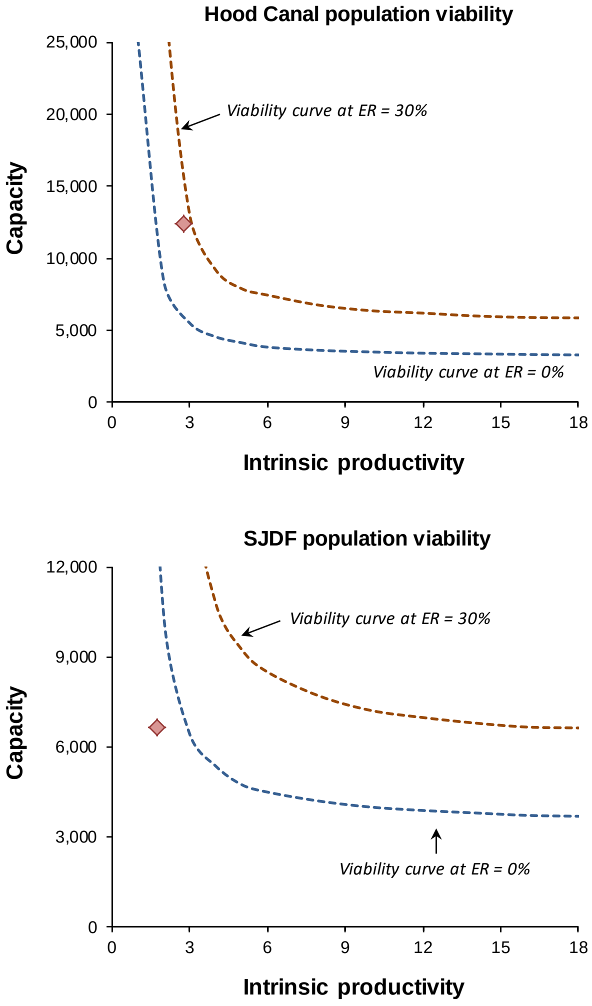

Figure 18. Viability curves with a 5% extinction risk over 100 years for the Hood Canal and SJDF summer chum populations with associated exploitation rates of 0 and 30%. The estimate of intrinsic productivity versus capacity is plotted for each population (calculated over brood years 1974 to 2012) to compare population performance with viability curves.

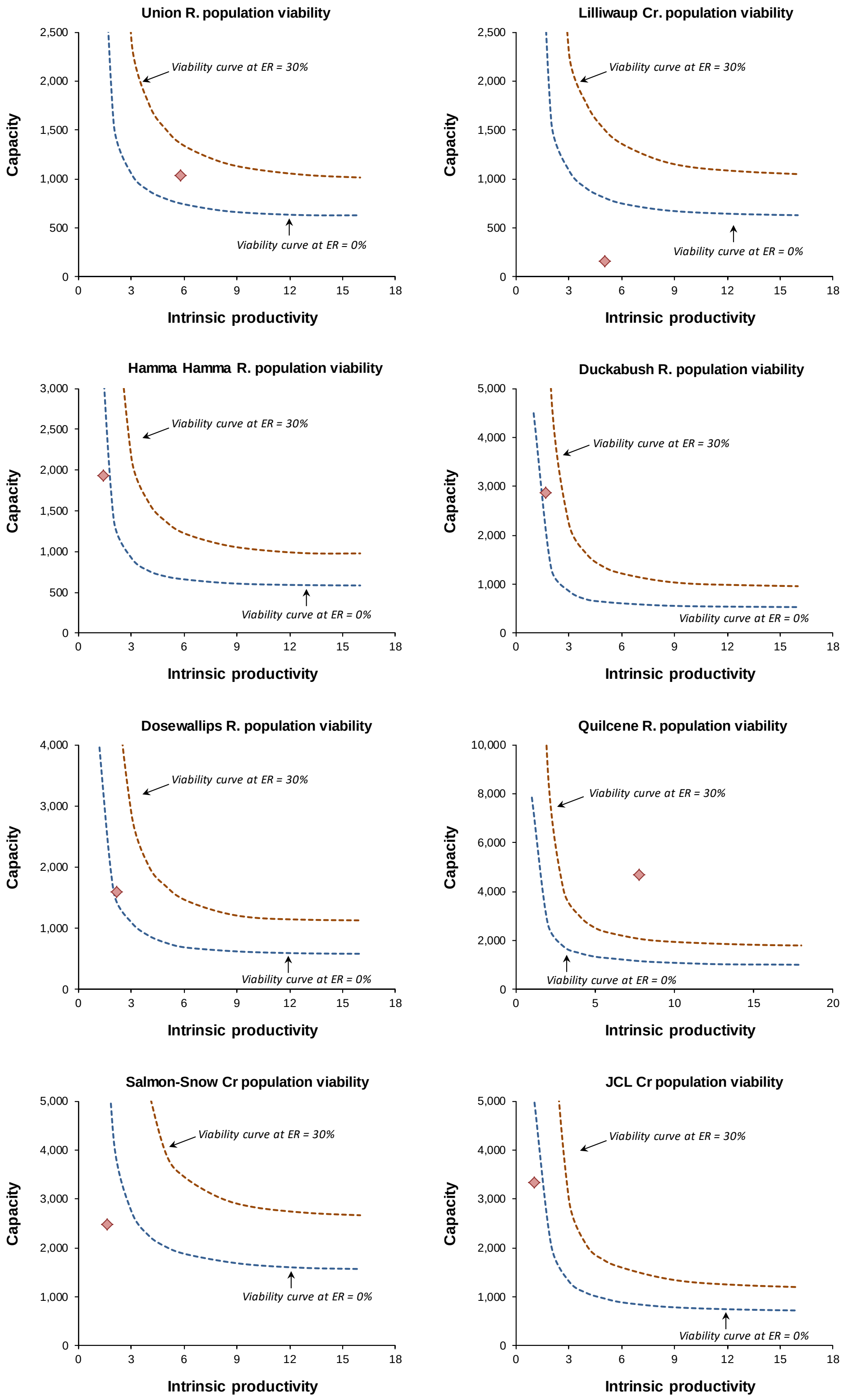

Figure 19. Viability curves with a 5% extinction risk over 100 years for the extant subpopulations belonging to the Hood Canal and SJDF summer chum populations with associated exploitation rates of 0 and 30%. The estimates of intrinsic productivity versus capacity are plotted for each subpopulation (calculated over brood years 1974 to 2012) to compare population performance with viability curves.

6.0 Incorporating Effects of Climate Factors

This section considers the effects of climate-related phenomena on summer chum performance and provides guidance for incorporating these effects into recovery planning. The potential effects of the Pacific Decadal Oscillation (PDO) as well as longer-term climate change effects are addressed.

The earth's climate system is not isolated to the atmosphere—it includes the oceans and the continents. The interactions of these climate system components produce what we experience as climate, but they can also have profound effects on short-term and longer-term characteristics of associated ecosystems. In recent years, evidence has grown that the PDO, for example, is having a strong effect on the performance of the Hood Canal Summer Chum ESU, which potentially can mask the importance of freshwater and sub-estuarine habitat conditions. Recovery of the ESU needs to account for the role of climate-related factors.

In March 2017, the HCCC held a forum of invited scientists to review the results of analyses on summer chum performance related to climate factors and to solicit their input on how the results should be considered in recovery planning. Participants in the forum are listed in Appendix A. Comments received through the forum have been integrated into the material presented here.

6.1 PDO Effects

Conditions related to salmon survival within the Northeast Pacific Ocean and connecting marine waters have been linked to two climate processes: the PDO and the El Nino-Southern Oscillation (ENSO). Both phenomena 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 can affect food webs related to the marine survival of salmon.

In our earlier report (Lestelle et al. 2014), we hypothesized that the correspondence of the PDO and summer chum performance is due to some kind of climate system forcing on the Puget Sound ecosystem related to the PDO.[27] We hypothesized that conditions within Puget Sound being influenced by large-scale climate related forcing factors were affecting summer chum fry survival. Causal mechanisms were not identified. We suggested that the mechanisms may be related to circulation patterns within the branches of Puget Sound and how nutrients from oceanic waters are moved into the inland sea, which then affect food webs of importance to outmigrating summer chum fry. It bears noting that summer chum fry emerge from spawning gravels much earlier than normal-timed fall chum and generally emigrate to the marine environment earlier than other wild salmon fry in the region (Tynan 1997; Lestelle et al. 2005).[28] Therefore, their emigration through the river mouth estuaries and the

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[27] / Factors external to an environmental system, such as an ecosystem, that can drive or force the system to a different state or condition, usually for some extended period, are referred to as forcing factors. Large-scale climate factors, for example, can operate to force a change in zooplankton dynamics within a region (Batchelder et al. 2013).

[28] / Chinook fry produced from natural spawners in the Skokomish River also emerge and emigrate to the estuary very early. However, these fish are derived from George Adams Hatchery stock and their timing has been significantly advanced compared to the original indigenous Chinook population in that river. Survival of these fry in nature is poor. Domestication and adaptation to the hatchery environment has been suggested as the cause of the timing advancement for these fish. See SIT and WDFW (2017).

nearshore environment is often earlier than the major zooplankton blooms in the region (Tynan 1997; Lestelle et al. 2005).

Evidence for climate system forcing by large-scale phenomena like the ENSO and PDO on non-biological and biological conditions in different parts of the Salish Sea has been documented by Moore et al. (2008) and Li et al. (2013). Zimmerman et al. (2015) reported that marine survival of coho can differ for fish originating from Strait of Georgia and Puget Sound, the major basins of the Salish Sea, as well as among the subbasins within Puget Sound. The authors suggested that the differences may be related to differential effects of large-scale climate forcing on oceanographic properties of these geographic areas, in addition to the effects of more localized factors. Li et al. (2013) provided evidence of large-scale climate forcing on zooplankton patterns in the Strait of Georgia.

6.1.1 Description of the PDO and Relevance

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 has been described as a recurrent pattern of inter-decadal climate variability characterized by persistent winter North Pacific atmospheric and oceanic circulation patterns (Mantua et al. 1997). It 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.

Causes for the PDO are not currently known (website for the University of Washington’s Joint Institute for the Study of the Atmosphere and Ocean or JISAO). Likewise, the potential predictability for this climate oscillation is not known.

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 has been referred to as a regime shift (Hare and Francis 1995; Hare and Mantua 2001).

The PDO has been generally 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 1950 to mid-2017, derived from monthly sea surface temperatures (SST) in the North Pacific Ocean, poleward of 20°N, as reported by JISAO (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.

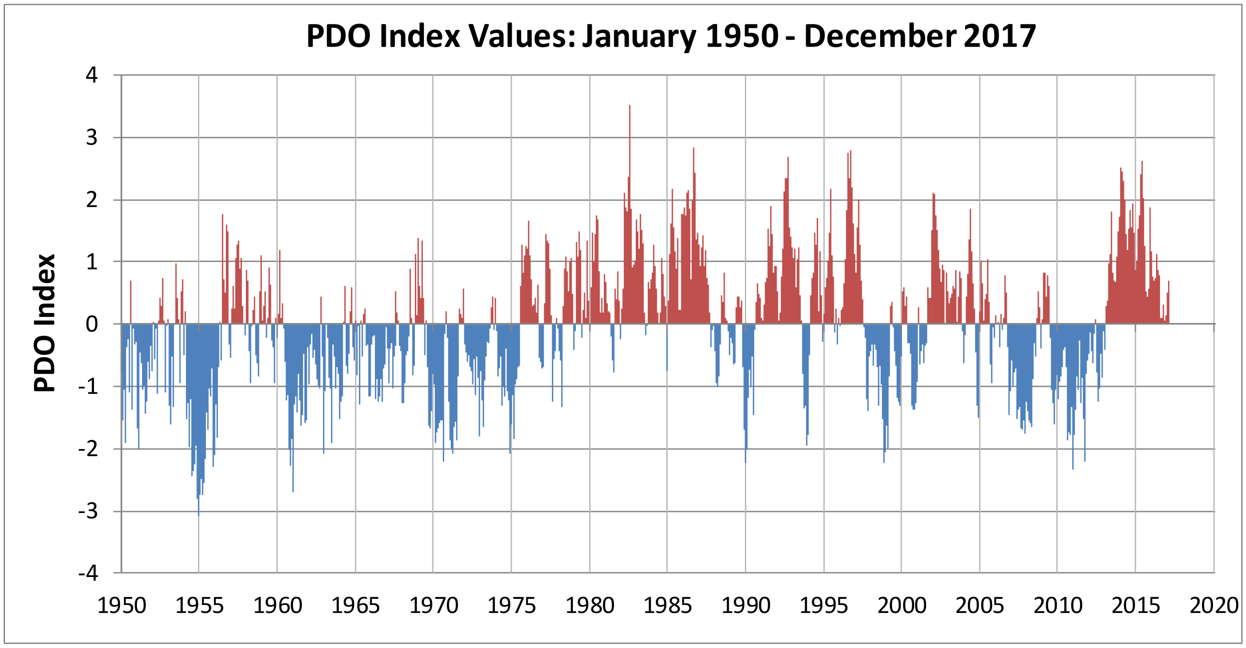

Figure 20. Annual deviations from the long-term annual average PDO index for 1900 to 2017 (only 1950-2017 are shown), derived from monthly sea surface temperatures (SST) in the North Pacific Ocean, poleward of 20°N, as reported by the Joint Institute for the Study of the Atmosphere and Ocean at the University of Washington (http://jisao.washington.edu/pdo/PDO.latest).

The PDO is thought to have been in its cool phase from about 1890 to 1925 and from 1945 to 1977. 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 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 as part of a cyclic pattern. The 1989 shift was not a reversal within a cycle 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 remained intact through the end of 2013, when it swung back to the warm phase. The PDO index has been in the warm phase since the start of 2014 through the present time (at least to the end of 2017).

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 coho 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 (Thomson 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 remained evident until at least 2004 (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).

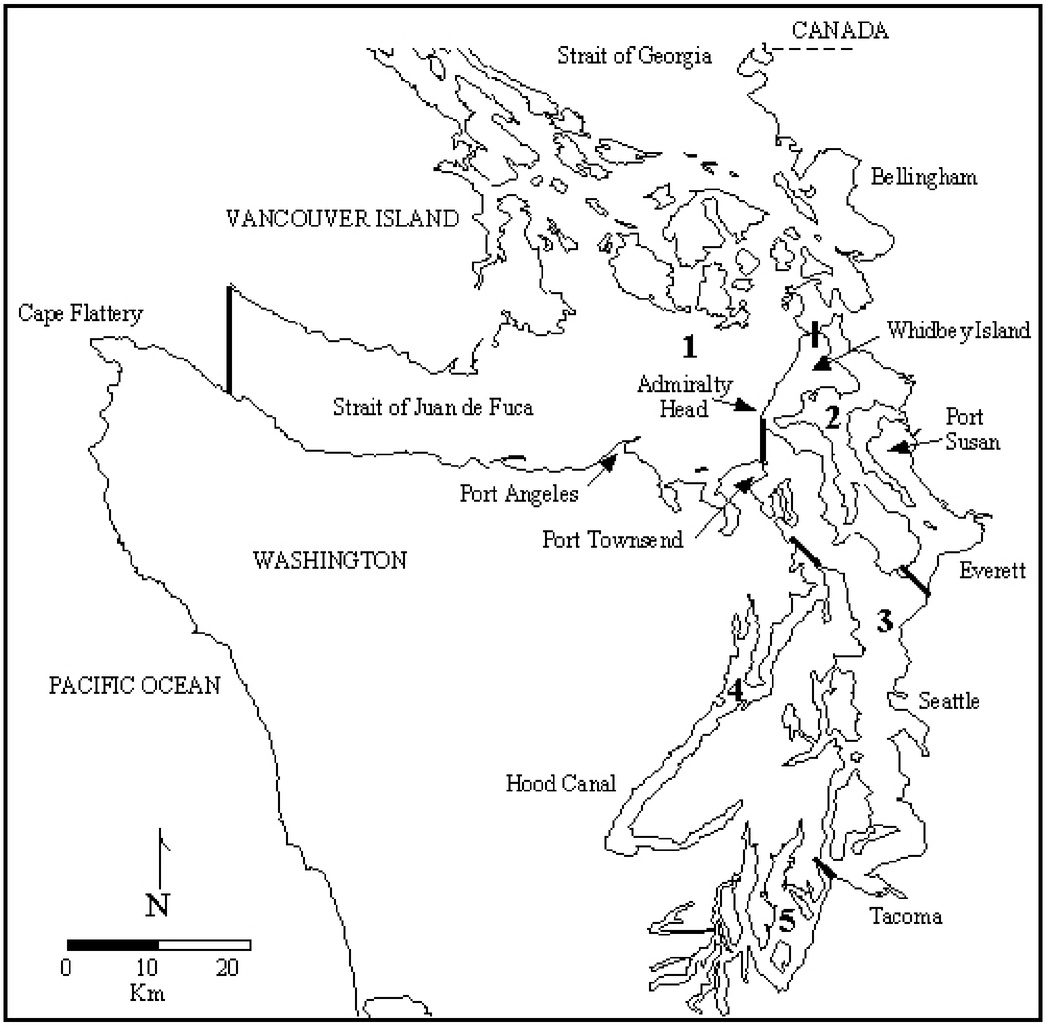

Figure 21. Puget Sound subbasins (called basins here): 1) Northern Basin, 2) Whidbey Basin, 3) Main Basin, 4) Hood Canal, and 5) Southern Basin. Dark bars show basin delineations. Taken from Gustafson et al. (2000).

Strickland (1983) and Nelson (undated) reported that relatively shallow sills at various locations in Puget Sound, including in Hood Canal, act to essentially slow 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 into Puget Sound, 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 Hood Canal is highly stratified, except under strong wind forcing events, 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 whose performance has been shown to be strongly tied to the estuarine experience (Salo 1991; Johnson et al. 1997).

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. Because of these different patterns among the populations, the authors hypothesized that the effect of the PDO occurs very close to where the natal rivers drain to the marine environment, and therefore when the fry are very young, and not in common rearing areas such as in the SJDF or the Pacific Ocean when the juveniles are older.

6.1.2 Evidence for Effects of Regime Shifts on Summer Chum Performance

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.

6.1.2.1 Strait of Juan de Fuca Population

Figure 22 is a composite graphic for the SJDF population enabling the reader to inspect the temporal patterns of total spawners (natural spawning) and resulting adult recruits from those 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 analysis extends through brood year 2012, which in effect includes return years up to 2016.[29] The reader will note that the PDO index pattern (as anomalies) shown in Figure 22 appears to roughly correspond with the anomalies for brood year recruitment. In Section 6.1.2.3, we discuss possible lags in how the effect of the PDO may be manifested in the pattern seen for 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 (note that the y-intercepts in the lower right chart differ with productivity for brood years 1999-2012 being higher).

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[29] / The reader should note that for brood year 2012 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 (return year 2016).

SJDF population

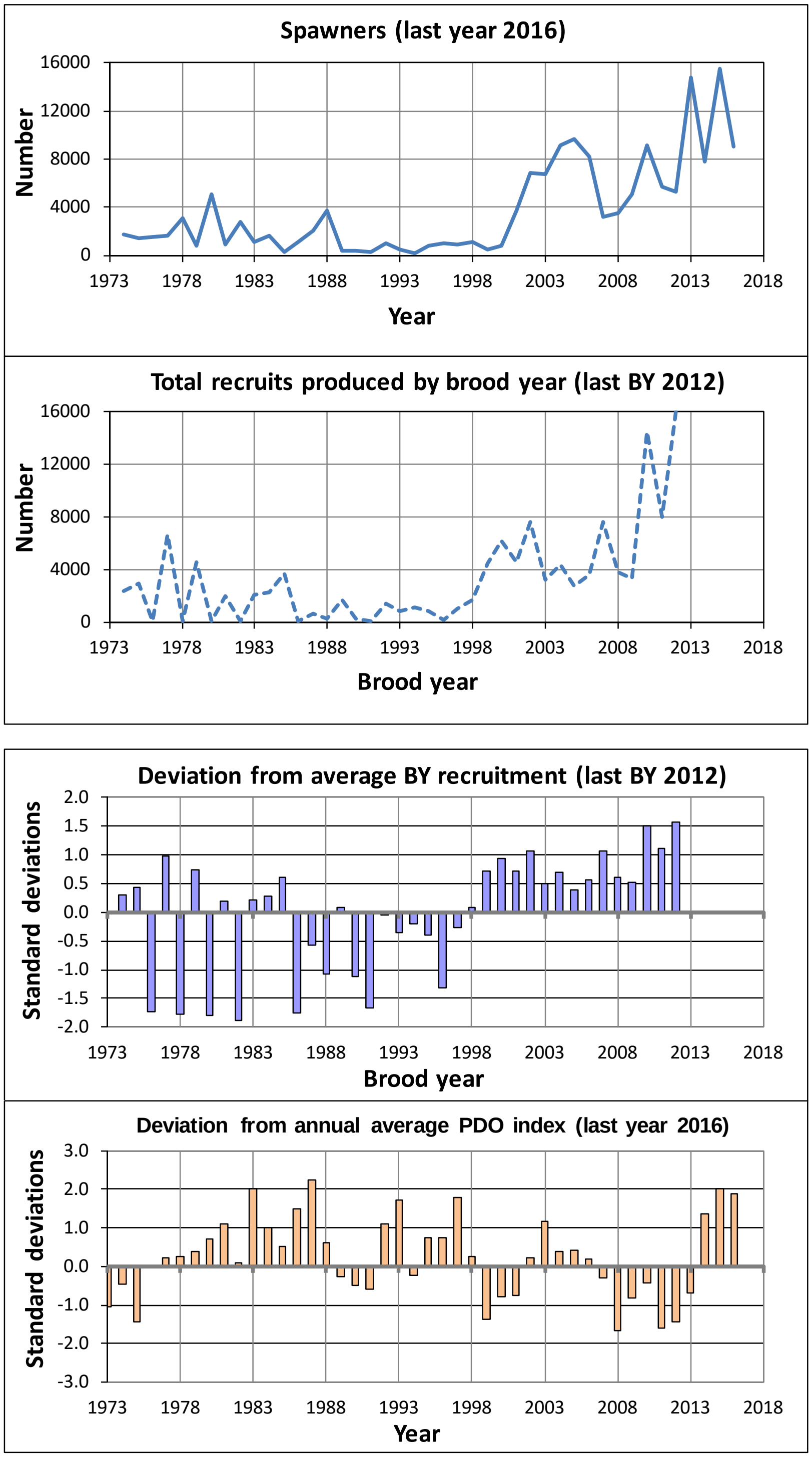

Figure 22. For the SJDF population from top to bottom: Patterns of spawner abundance by year, resulting total recruits by brood year, deviations from average recruitment by brood year, and deviations from the average PDO index by year. The year shown in the top graph becomes the brood year in the second graph. The report narrative explains how to consider when the PDO affects brood year recruits.

Linearizing Spawners and Recruits by PDO regime
SJDF Population Brood Years 1979 - 2012

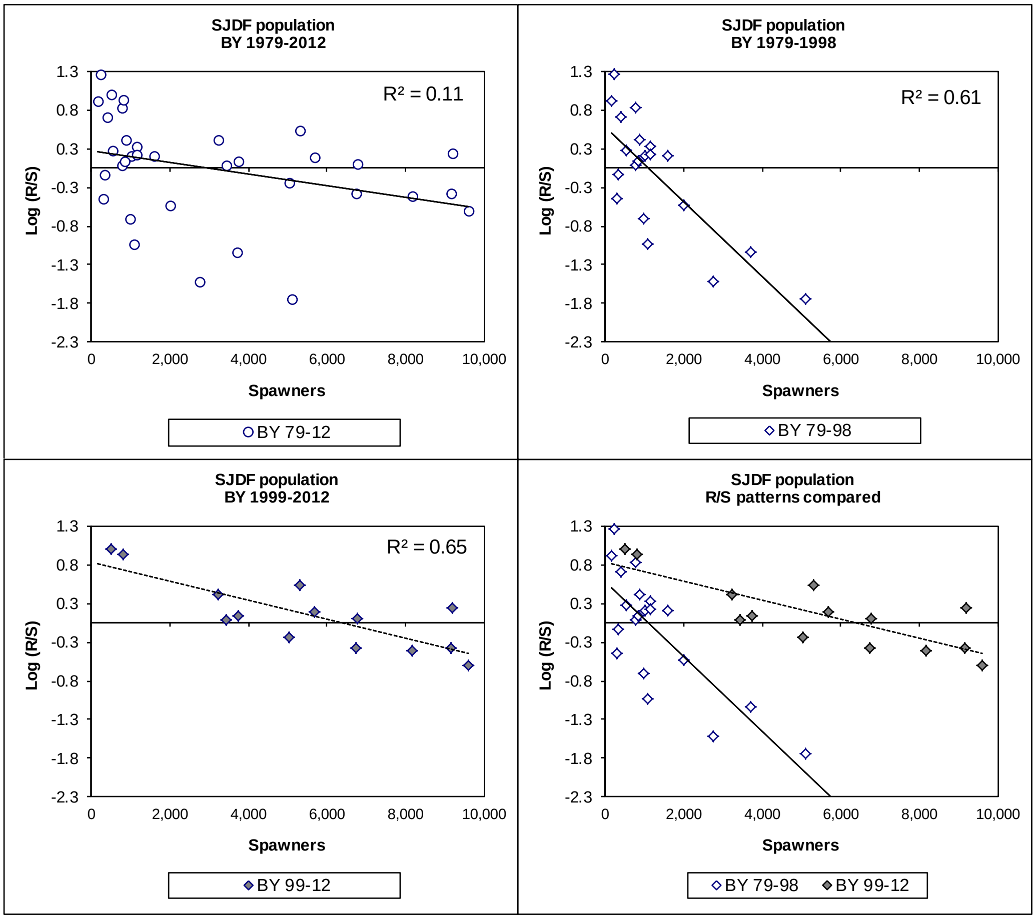

Figure 23. Spawner (S) and recruitment (R) plots grouped for different time periods for the SJDF summer chum population. Brood years 1979 to 2012 are used. The y-axis for each chart is the log of R/S.

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 subpopulations considered to be extant by the PSTRT at the time of its analysis (Sands et al. 2009) are provided in Appendix B. Both subpopulations show a strong effect of the 1998 regime shift to productivity, whereas only the Salmon-Snow subpopulation shows a strong positive response to capacity. The capacity of the Jimmycomelately subpopulation appears to have had less response to the regime shift though it still appears to be evident. It is important to recognize that substantial restoration work was done in both Salmon Creek and Jimmycomelately Creek after the PDO regime shift in 1998.

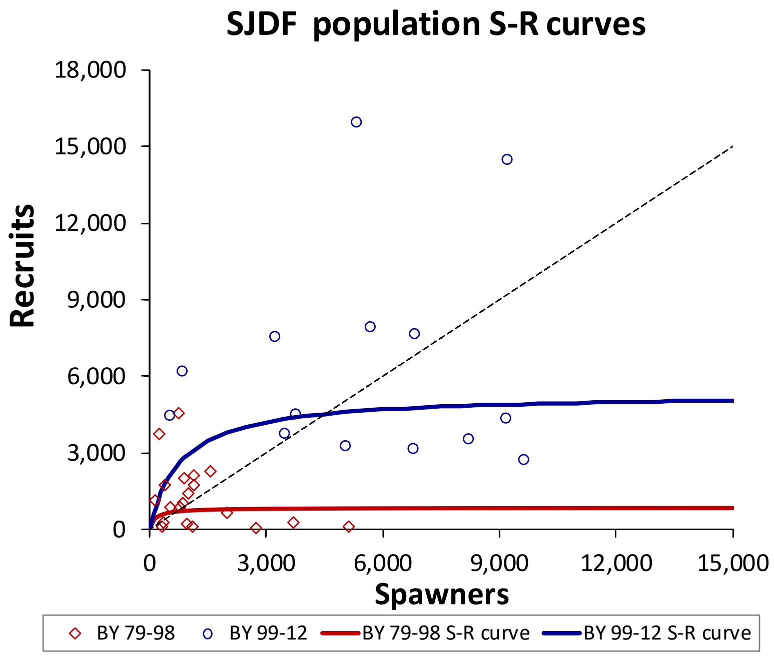

Figure 24. Spawner-recruit (S-R) plots for brood years 1979-1998 and 1999-2012 for the SJDF summer chum population.

6.1.2.2 Hood Canal Population

Figure 25 is a composite graphic for the Hood Canal population enabling the reader to inspect the temporal patterns of total spawners (natural spawning) and resulting adult recruits from those 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 analysis extends through brood year 2012, which in effect includes return years up to 2016 (see footnote 30). The reader will note that the PDO index pattern (as anomalies) shown in Figure 22 appears to roughly correspond with the anomalies for brood year recruitment. In Section 6.1.2.3, we discuss possible lags in how the effect of the PDO may be manifested in the pattern seen for 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 1999-2012 separately, and finally for the two separate groups on the same plot. The two brood year periods clearly show different patterns but it is not evident that productivity differs between the two periods (y-intercepts are approximately the same).

Figure 27 shows the S-R data for the Hood Canal 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 subpopulations considered to be extant by the PSTRT at the time of its analysis are provided in Appendix B. The effect of regime shifts on performance varies among the subpopulations. Our preliminary conclusions about how productivities and capacities have been affected by PDO-related effects based on visual inspections of the patterns are summarized below. We infer that a change in productivity has occurred if the patterns and intercepts in the linear regressions are markedly different between the two time periods; a change in capacity is inferred if the asymptotic limit on the S-R graphs is markedly different. We find an effect to be likely (by yes), unlikely (by no), or that it is unclear whether an effect has occurred (by uncertain):

Subpopulation (south to north)	Productivity	Capacity
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 patterns 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

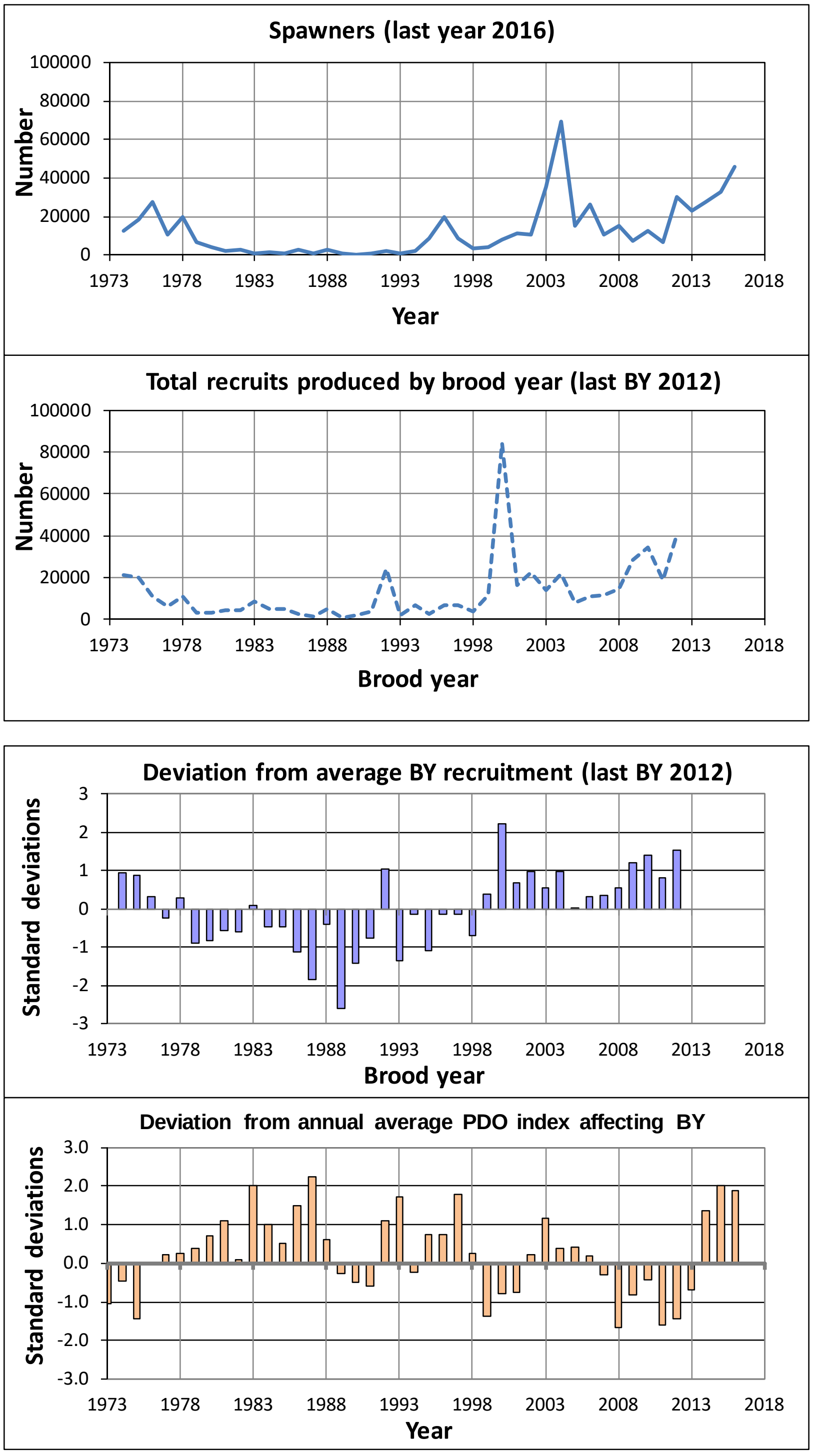

Figure 25. For the Hood Canal population from top to bottom: Patterns of spawner abundance by year, resulting total recruits by brood year, deviations from average recruitment by brood year, and deviations from the average PDO index by year. The year shown in the top graph becomes the brood year in the second graph. The report narrative explains how to consider when the PDO affects brood year recruits..

Linearizing Spawners and Recruits by PDO regime
Hood Canal Population Brood Years 1979 - 2012

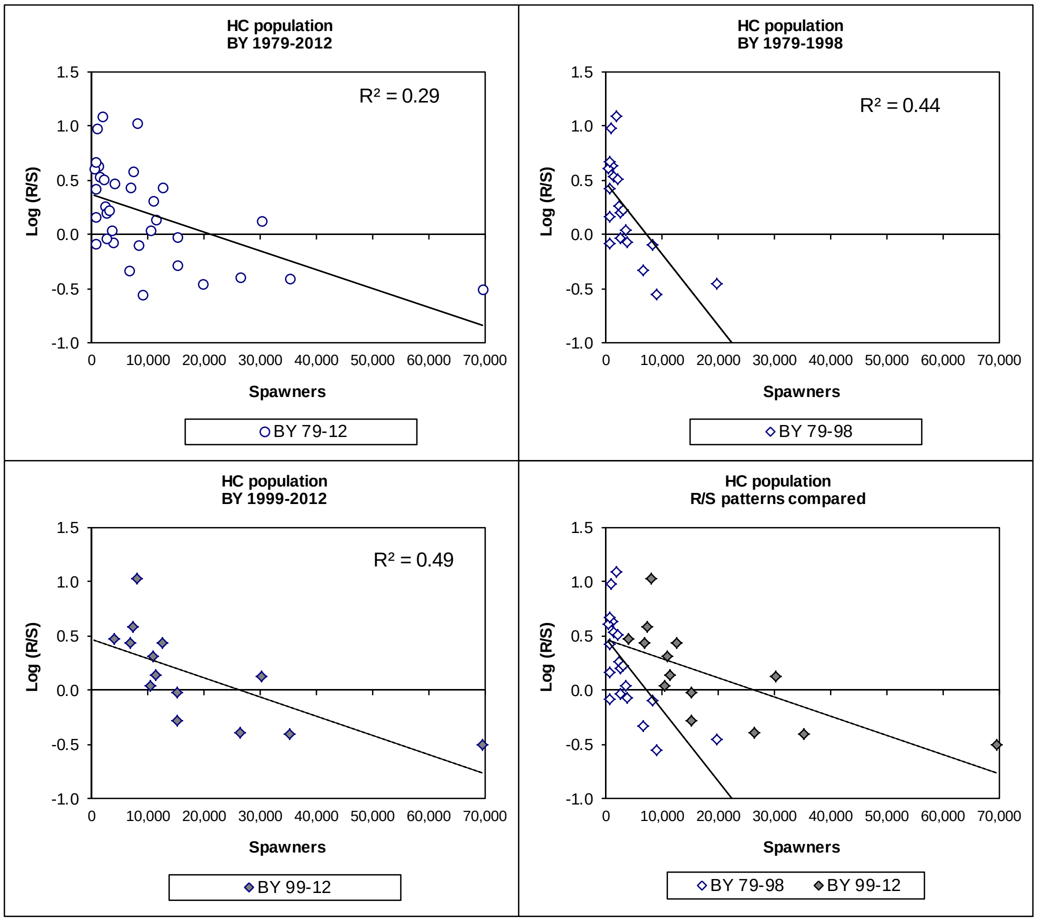

Figure 26. Spawner (S) and recruitment (R) plots grouped for different time periods for the Hood Canal summer chum population. Brood years 1979 to 2012 are used. The y-axis for each chart is the log of R/S.

6.1.2.3 Effect of Time Lags from the PDO Index Year

Slight dissimilarities in how the anomalies for recruitments by brood year align with the PDO index anomalies between the two populations (comparing the bottom two graphs in Figures 22 and 25) caused us to suspect differences in the timing of the effect of the PDO on the two populations. Because we have hypothesized that the effect is somehow being experienced by summer chum fry within the marine environment primarily near their natal streams, we suspected that different time lags in the effects may exist between the populations. We speculate that different time lags could be caused by differences in water circulation and mixing patterns within the various parts of Puget Sound. Such differences may affect localized plankton blooms and food webs at variable times, perhaps even differing by many months, as the effects of a PDO shift are transferred slowly into the various branches of Puget Sound.

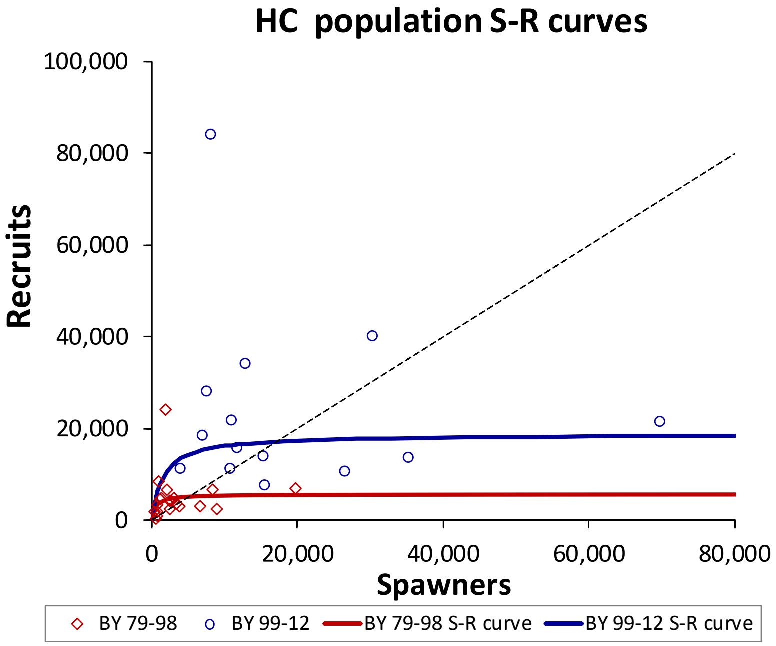

Figure 27. Spawner-recruit (S-R) plots for brood years 1979-1998 and 1999-2012 for the Hood Canal summer chum population.

To examine this possibility, we performed correlation analysis between the number of adult recruits produced by brood year and the PDO index with different time lags in number of years. Table 6 provides the correlation coefficients r for seven correlations for each population using different time lags, where the lag times are stepped by one year (either lagged from the brood year or fry year, both are shown in the table). We used natural log transformed adult recruits because of the log normal distribution of salmon recruitment data (Hilborn and Walters 1992).

The results are consistent with our hypothesis. There appears to be no lag in an effect for the SJDF population (using fry outmigrant year) but a lag appears to be evident for the Hood Canal population. The strongest correlations for the SJDF and HC populations apply lags of fry year – 0 and fry year – 2 respectively. Figure 28 shows the correlation for the Hood Canal population using a lag of fry year – 2. We note, however, that statistically significant correlations exist for Hood Canal over a broader range of lags, beginning with a lag of one year from the fry year (fry year – 1) (Table 7). This may be due to the extended period that exists for whether the index is positive or negative that occurs in the record.[30] Moreover, given the amount of variability that is evident in Figure 28, the length of delay in how the PDO effects are transferred into the branches of Puget Sound likely varies among years. We suggest the variability around the correlation line is partly due to interannual differences in how the lag effect influences population performance.

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[30] / A more refined, powerful statistical approach would be to remove density-dependent effects on adult recruitment rates and use a time-varying estimate of the productivity parameter with a Kalman filter procedure as applied recently by Malick and Cox (2016) for analyzing chum and pink productivities. The procedure was not available to be used in this report.

Table 7. Summary of correlation analysis between PDO index values (annual average of monthly average values) and natural log transformed adult recruits for the SJDF and Hood Canal populations using different time lags. Lags are shown for both brood year and fry year. * indicates the relationship is significant at the 0.05 level and ** indicates it is significant at the 0.01 level.

Lag (offset) to PDO index year		Correlation coefficient r			
Brood year (BY)	Fry year (FY)	SJDF		HC	
BY - 3	FY - 4	0.261		0.458	**
BY - 2	FY - 3	0.237		0.528	**
BY - 1	FY - 2	0.270		0.591	**
BY - 0	FY - 1	0.348	*	0.471	**
BY + 1	FY - 0	0.525	**	0.298	
BY + 2	FY + 1	0.187		0.257	
BY + 3	FY + 2	0.000		0.084	

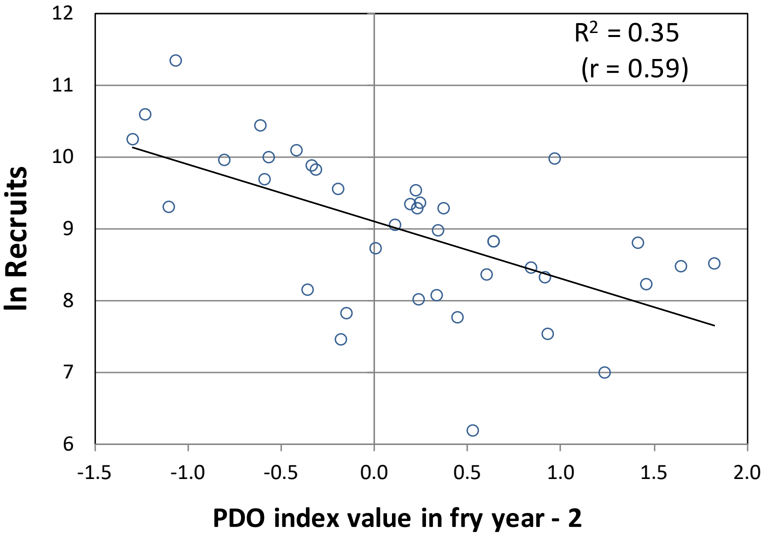

Figure 28. Relationship between the PDO index value (average of monthly average values) and natural log transformed adult recruits for the Hood Canal summer chum population using a lag of fry year - 2, brood years 1974-2012 (*P* << 0.001).

An opportunity has availed itself for advancing our understanding about the effect of the PDO and a possible lag of the effect into Puget Sound. The PDO index turned strongly positive (warm phase) in January 2014 and has remained in the positive state to the present time (Figure 20). This means that if indeed an effect of the PDO is occurring on the summer chum populations, then fry produced from brood year 2013 (fry year 2014) could have experienced a downturn in survival. Summer chum return to spawn mostly at age-3 and age-4 (about equally split); therefore, the returns from brood year 2013 would have occurred primarily in 2016 and again in 2017.

Our expectations for how the populations should respond to the observed changes in the PDO index beginning in 2014 are summarized in Table 8. Using results of the correlation analysis, we assumed for the sake of the projections in the table that the SJDF and Hood Canal populations would have lags of fry year – 0 and fry year – 2 respectively. From this, we expected that we should have seen some evidence for a downturn in abundance for the SJDF population in 2016, and further, an even greater downturn in 2017 (when both age-4 fish would be returning from brood year 2013 and age-3 fish from brood year 2014). We expect to see the first evidence of a downturn in recruitment for the Hood Canal population in 2018 (age-3 fish from brood year 2015) and a greater downturn in 2019 (age-4 fish from brood year 2015 and age-3 fish from brood year 2016).

Table 8. Projected exposure of fry migrants from the SJDF and Hood Canal populations to either the warm or cool PDO phase with time lags based on correlation analysis (Table 7). Return years subjected to the warm phase are highlighted in yellow. Return years 2016 to 2022 are shown. Projections for the SJDF population assume the time lag is fry year – 0; those for the Hood Canal population assume the lag is fry year–2.

Population	Spawn year	Fry out	Focus PDO year	PDO index fry year - lag	Age 3 returns	Age 4 returns
SJDF	2013	2014	2014	Warm	2016	2017
	2014	2015	2015	Warm	2017	2018
	2015	2016	2016	Warm	2018	2019
	2016	2017	2017	Warm	2019	2020
	2017	2018	2018	?	2020	2021
	2017	2019	2019	?	2021	2022
HC	2013	2014	2012	Cool	2016	2017
	2014	2015	2013	Cool	2017	2018
	2015	2016	2014	Warm	2018	2019
	2016	2017	2015	Warm	2019	2020
	2017	2018	2016	Warm	2020	2021
	2017	2018	2016	Warm	2021	2022

We now know that there was no evidence of a downturn in the abundance of the SJDF population in 2016 (see Figures 6 and 7). Age-3 spawners, which Table 8 predicts should have been adversely affected in return year 2016, returned in substantial numbers. However, preliminary data for return year 2017 (not presented in this report) showed a large decline in abundance for the SJDF population (both for age-3 and age-4 fish), while the Hood Canal population continued on its upward trajectory in abundance. These preliminary results are consistent with our projections shown in Table 8 with the exception of the SJDF return in 2016.

Despite the large downturn in the SJDF population in 2017, we cannot conclude that it was due to PDO-related effects. Other possible explanations exist. Weinheimer et al. (2017) presented findings from the Salmon Creek summer chum component of the Salmon-Snow Creek subpopulation showing a large effect of peak winter flow on survival from egg deposition to fry outmigration (Figure 29). WDFW traps both adult summer chum returns and fry outmigrants in lower Salmon Creek to assess summer chum freshwater performance in the stream. Over the range of peak flows estimated for Salmon Creek from brood years 2007 to 2016 seen in Figure 29, survival from egg to fry outmigration has ranged from 1% to about 45%.

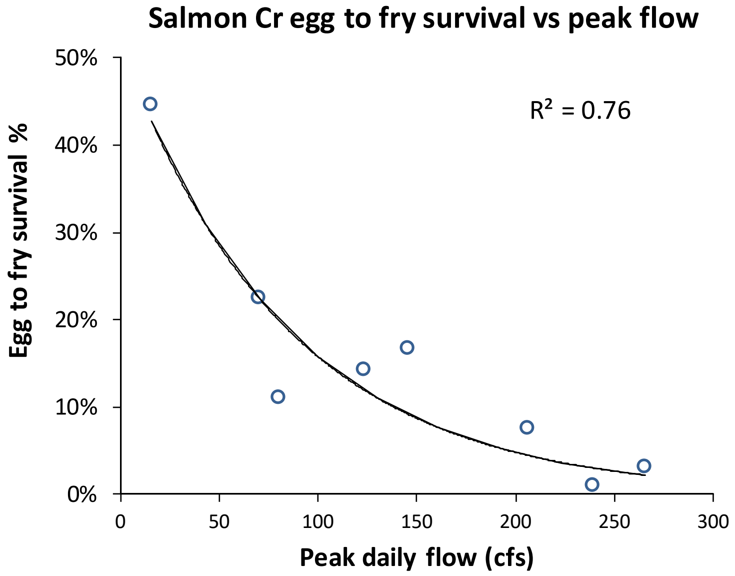

Figure 29. Relationship between the peak daily flow and percent egg to fry survival for summer chum in Salmon Creek for brood years (BY) 2007 to 2016 (*P* < 0.01). Data points are not given for brood years 2011 and 2012 because the stream gauge was inoperative during egg incubation period. See Table 9 for data. The figure is updated from one presented in Weinheimer et al. (2017).

Table 9 summarizes associated metrics for the Salmon Creek subpopulation component (metrics have been updated and expanded through brood year 2016 from those given in Weinheimer et al. (2017)). It should be noted that the Salmon Creek streamflow gauge was inoperative for large portions of the 2011-12 and 2012-13 winters so no estimates of peak flow are listed for those years. The lowest egg to fry survivals occurred in consecutive years—for the cohorts of brood years 2014 and 2015. The resulting numbers of fry outmigrants from those brood years were substantially less than all other brood years in the data record and particularly compared to the two brood years before 2014 (Figure 30). The two years with the largest numbers of fry outmigrants also had the highest egg to fry survivals. For the brood years listed in Table 9, six years of estimates of the number of resulting adult recruits are available; these data suggest a linear relationship between the number of fry outmigrants and subsequent adult recruitment (Figure 31). We note, however, that brood year 2012 is the last brood year represented in Figure 31. The shift in the PDO to the warm phase occurred in early 2014 and therefore, any PDO-related effects would not yet have occurred for any of the years seen in Figure 31.

Some preliminary observations about adult returns to Salmon Creek from brood years 2013 and 2014 are relevant and warrant comment. Brood year 2013 produced the second highest fry outmigration in the data record (Table 9). Adult recruitment from this brood year occurred primarily in 2016 (age-3 adults) and 2017 (age-4 fish). The abundances of age-3 fish returning in 2016 to Salmon Creek and to SJDF streams in general were large, showing no downturn from the prior pattern of large run sizes. In contrast, the returns of age-4 summer chum in 2017 to SJDF streams were greatly reduced, diverging from the prior pattern for the cool PDO phase.[31] Moreover, brood year 2014 produced the second lowest fry outmigration in the data record (Table 9), sharply reduced from the number of fry outmigrants in the previous year. Preliminary estimates of brood year 2014 age-3 adult returns in 2017

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[31] / It bears noting that the difference in response of the two age classes from the same brood year cohort suggest a possible effect of the PDO in the Pacific Ocean, not within the SJDF, i.e., after the period of fry outmigration.

show low abundance and likely reflect the small number of fry outmigrants together with any PDO-related effect on the fry within SJDF waters. The adult return for 2018 will be produced from the two consecutive years of exceptionally low numbers of fry outmigrants in combination with any effects of the warm PDO phase—this suggests very low adult abundance in 2018.

Table 9. Salmon Creek summer chum performance measures for brood years (BY) 2007 to 2016: spawners, potential egg deposition (PED), fry outmigrants, percent egg to fry survival, and adult recruits produced from each brood year. The peak daily average flow (cfs) during the egg incubation period for November through January is also shown (data from Washington Department of Ecology stream flow monitoring website). Data for PED and fry outmigrants are updated from those presented in Weinheimer et al. (2017). See Weinheimer et al. (2017) for methods.

BY	Spawners	PED	Fry	Egg to fry %	Adult recruits	Peak flow (cfs)
2007	1,274	1,307,500	186,983	14.3%	4,036	124
2008	1,568	1,672,500	376,025	22.5%	1,910	70
2009	1,237	1,400,000	105,611	7.5%	1,025	206
2010	2,740	3,017,500	333,349	11.0%	3,738	80
2011	2,279	2,367,500	635,632	26.8%	3,228	NA
2012	2,306	2,492,500	1,216,263	48.8%	7,989	NA
2013	2,746	2,592,500	1,153,949	44.5%		16
2014	2,460	2,730,600	87,191	3.2%		266
2015	6,841	7,593,510	75,140	1.0%		239
2016	3,154	3,500,940	582,792	16.6%		146

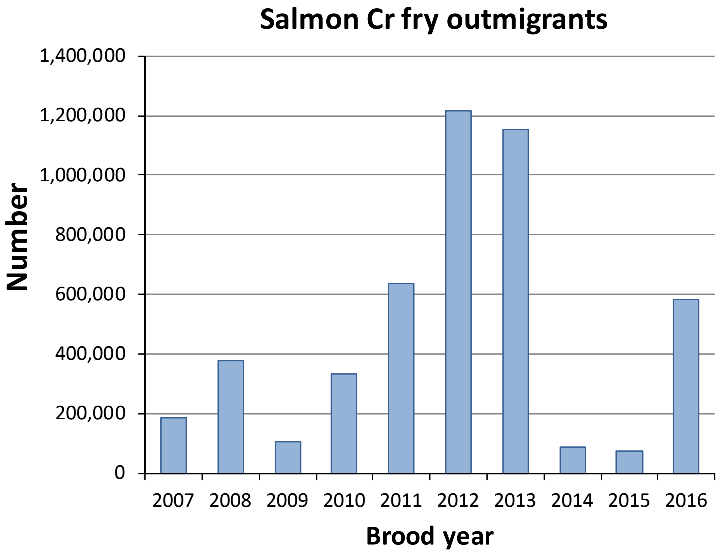

Figure 30. Estimated number of summer chum fry outmigrants in Salmon Creek, brood years 2007 to 2016.

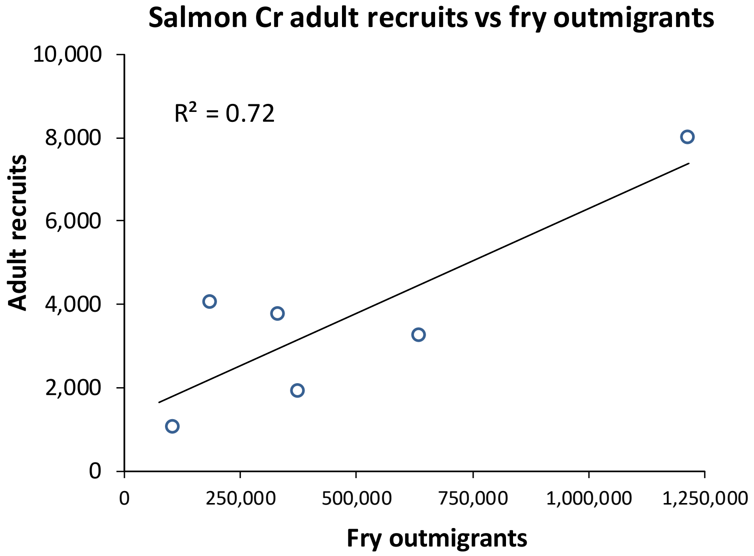

Figure 31. Relationship between the estimated number of fry outmigrants in Salmon Creek and estimated adult recruitment associated with those fry, brood years 2007 to 2012. The estimate of adult recruitment in 2016 is preliminary and subject to revision.

WDFW also currently estimates the abundance of summer chum fry outmigrants for the Duckabush River subpopulation within the Hood Canal population. Fewer years of data are available than exists for Salmon Creek but results to date provide additional information to help understand the effects of different factors on the performance of this population. Table 10 summarizes the relevant metrics to this discussion for brood years 2010 to 2016. The two smallest fry abundances occurred from brood year 2014 and 2015 (Figure 32), the same years that produced the smallest fry outmigrations in Salmon Creek. Fry abundances for the two streams are generally correlated over the seven years having estimates from both streams (Figure 33A) ($P < 0.05$ with log transformed data). The relationship between peak flow during the incubation period and egg to fry survival is not statistically significant but there is a general pattern of higher flows producing lower survivals (Figure 33B). Only three of the brood years with fry estimates have results for adult recruits and these show a wide range of recruitment over a small range of fry estimates (Figure 33C).

We note that there is poor correlation between peak flows during the egg incubation period (November-January) in the two streams for the years having data in both streams when one extreme data point is excluded (Figure 34). A statistically significant correlation exists when that one data point is included. Overall, however, we conclude that there is a generally poor correspondence between peak flows in the two streams, illustrating some degree of different response to winter weather patterns between the regions.

It is important to recognize that age-3 adult returns to the Duckabush River in 2018 will have been produced from the smallest fry outmigration in the data set (Figure 32), which coincidentally matches with a forecasted downturn in age-3 adults associated with the PDO phase change (Table 8). However, we note that there was no evidence of a downturn in adult abundance in 2017 for either the Duckabush River or the Hood Canal population as a whole from the second lowest fry outmigration in the data set (Figure 32). Age-3 adult returns in 2017 were produced from brood year 2014 fry.

Table 10. Duckabush River summer chum performance measures for brood years (BY) 2010 to 2016: spawners, fry outmigrants, percent egg to fry survival, and adult recruits produced from each brood year. The peak daily average flow (cfs) during the egg incubation period for November through January is also shown (data from USGS stream flow monitoring website). Data for fry outmigrants and egg to fry survival are from Weinheimer (2018).

BY	Spawners	Fry	Egg to fry %	Adult recruits	Peak flow (cfs)
2010	4,110	347,597	7.9%	8,285	4,280
2011	1,529	209,891	17.8%	3,274	2,770
2012	5,241	285,468	5.1%	6,469	3,290
2013	4,129	480,202	11.4%		1,110
2014	7,741	130,126	1.6%		4,880
2015	4,920	47,479	0.9%		3,500
2016	8,470	200,712	2.2%		2,450

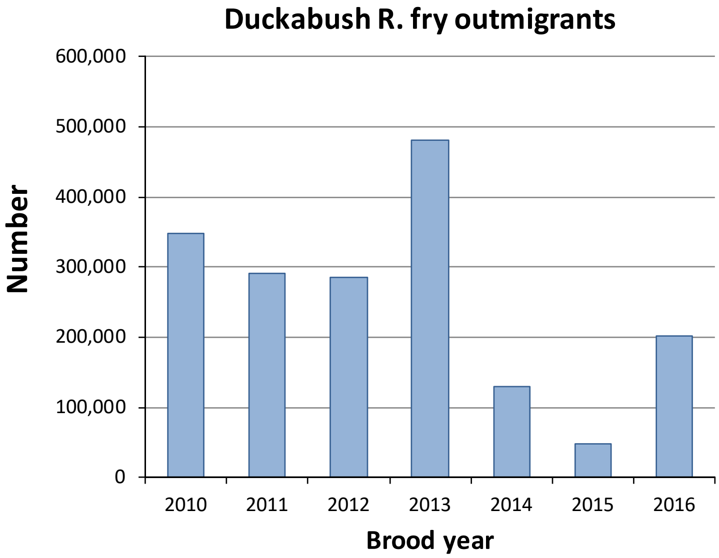

Figure 32. Estimated number of summer chum fry outmigrants in Duckabush River, brood years 2010 to 2016.

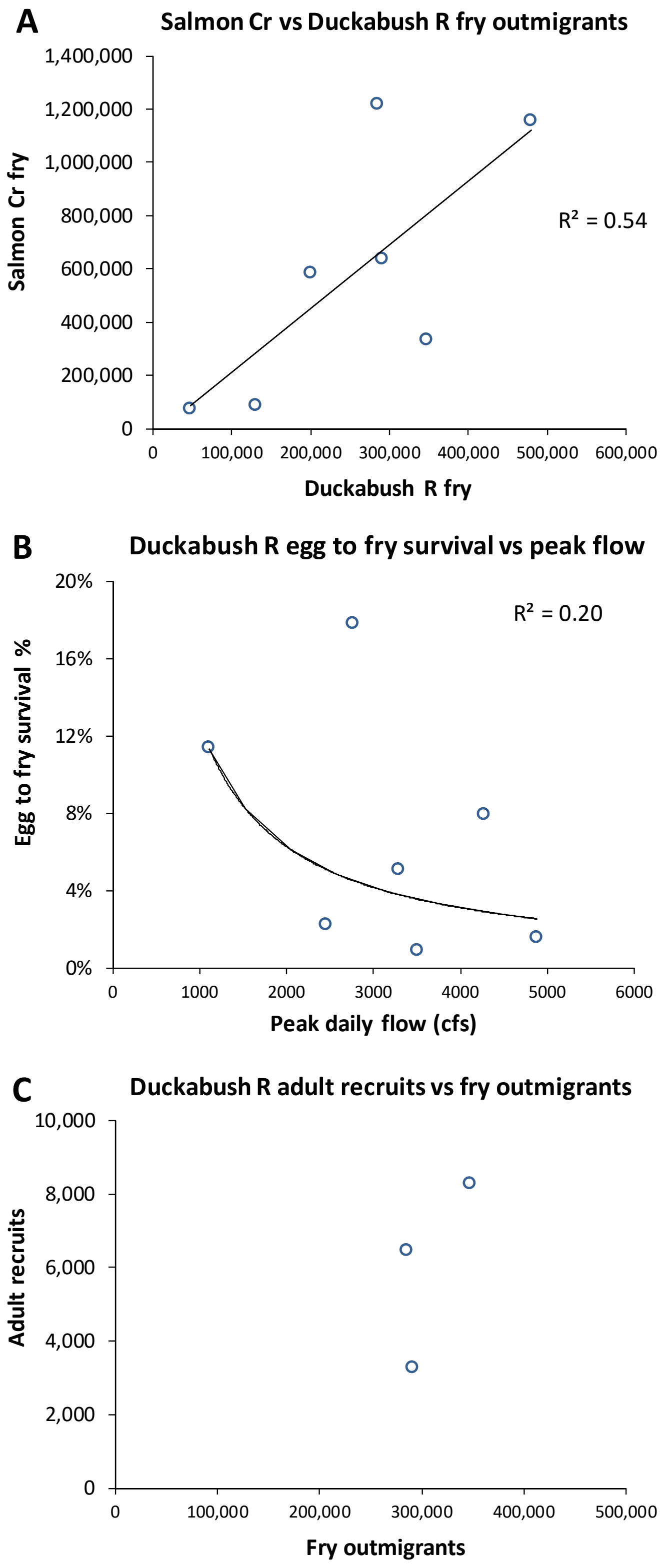

Figure 33. Duckabush River summer chum performance relationships: A – Correlation between Salmon Cr. fry outmigrant abundance and Duckabush R. fry outmigrant abundance. B – Egg to fry survival vs. peak flow during incubation period. C – Adult recruits vs. fry outmigrants.

Text

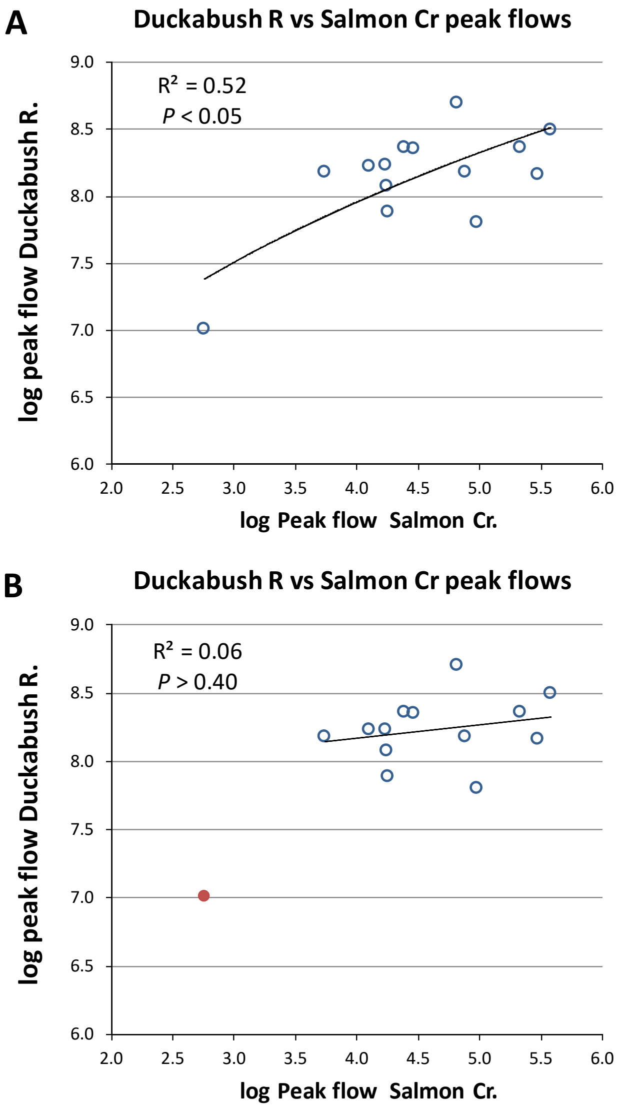

Figure 34. Correlation of peak flows in Duckabush R. and Salmon Cr. during the summer chum egg incubation period (November – January, as defined by Weinheimer et al. (2017). All flow data are log transformed. A – Correlation with all data for Water Years 2003-2018 (without 2012 and 2013 due to missing data). B – Correlation excluding the extreme low data point (shown as the red filled circle).

6.1.3 Effects of Ocean/Climate Shifts on Viability

We consider here the effects of the PDO on viability risk. 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 35). The results show that viability is very strongly affected by the ocean/climate regime for the brood years analyzed.

The performance of the Hood Canal population exceeded 5% threshold risk curves during both the warm and cool PDO regimes with a 0% exploitation rate. After the 1998 regime shift, the Hood Canal population even exceeds the threshold associated with a 30% exploitation rate by a large margin, while the SJDF population exceeds the threshold with a 0% exploitation rate by a relatively substantial amount but does not exceed the threshold associated with a 30% 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 36 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 one exception: Lilliwaup Creek. This subpopulation remained at high risk 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 particularly noteworthy that the Big and Little Quilcene subpopulation is well above even the viability curve with a 30% exploitation rate.

It is also 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 landslides 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. Figure 9 also shows a rebuilding pattern in Lilliwaup Creek in recent years.

It is important to recognize that all of the results shown in Figures 35 and 36 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.

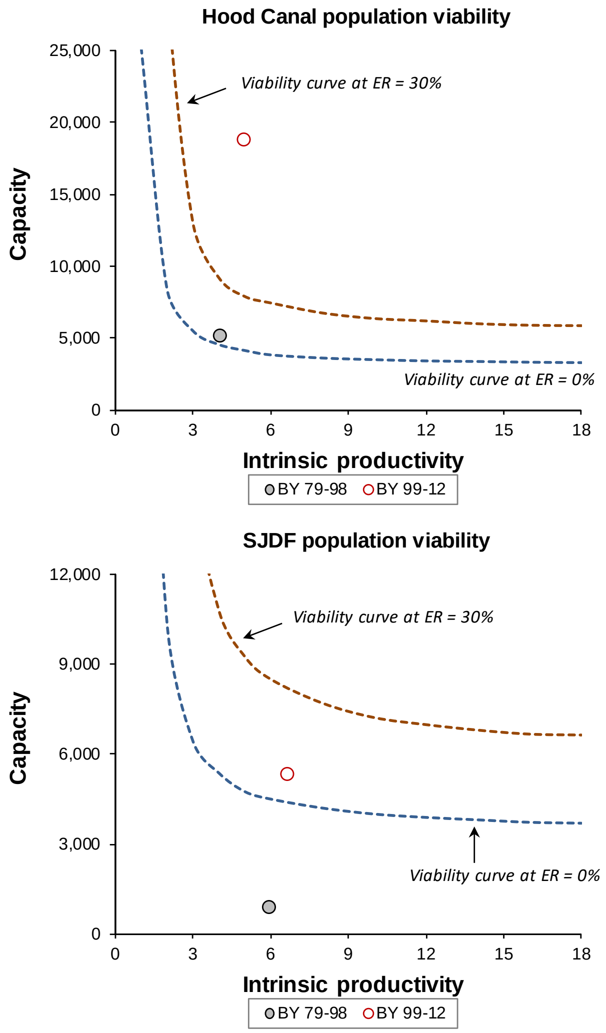

Figure 35. Population performance parameters for brood years (BY) 1979 to 1998 (warm PDO) and 1999 to 2012 (cool PDO) plotted relative to viability curves (5% extinction risk) for the SJDF and Hood Canal summer chum populations. Viability curves associated with both 0 and 30% exploitation rates are shown.

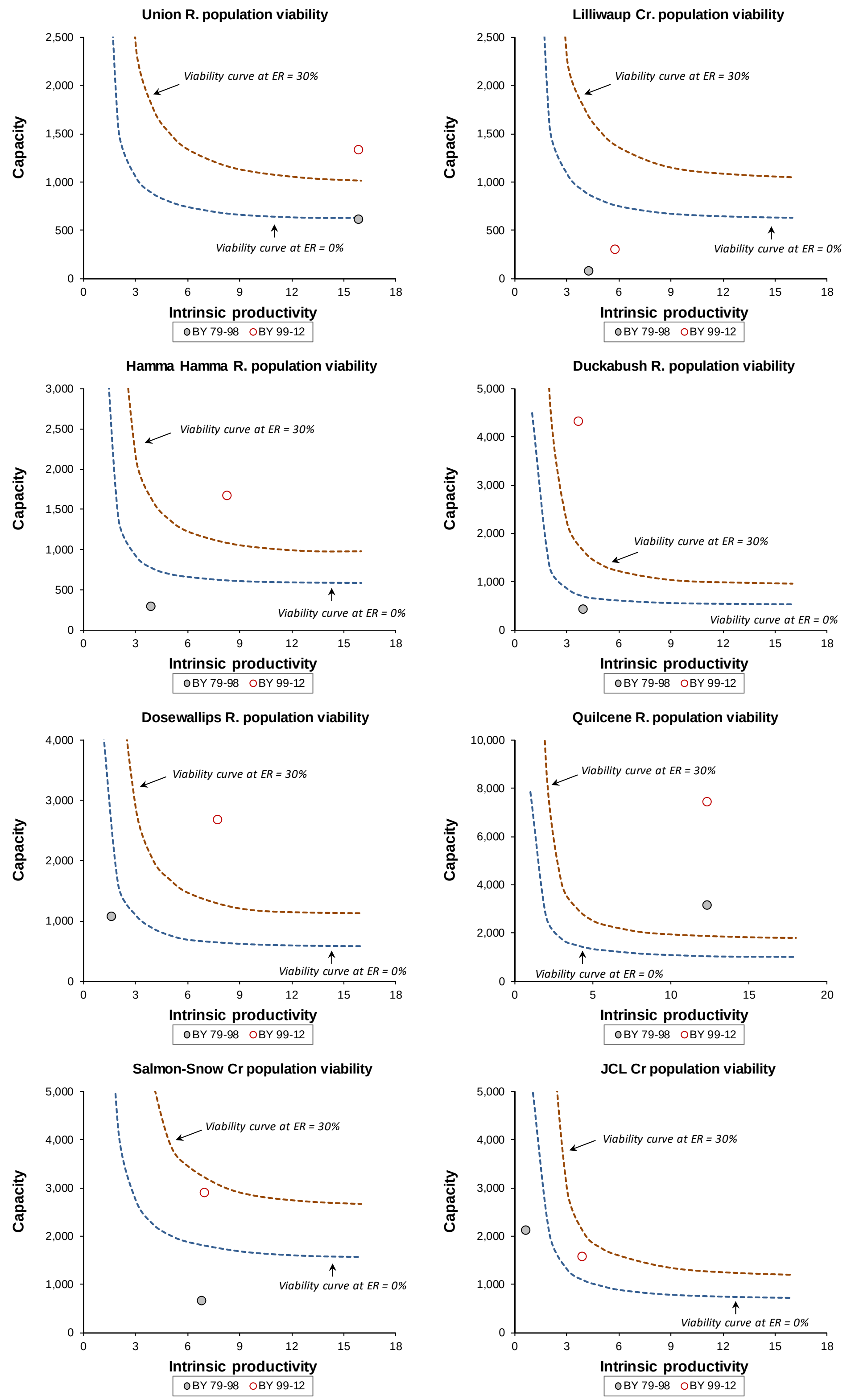

Figure 36. Performance parameters for brood years (BY) 1979 to 1998 (warm PDO) and 1999 to 2012 (cool PDO) plotted relative to viability curves (5% extinction risk) for extant subpopulations belonging to the SJDF and Hood Canal summer chum populations. Viability curves associated with both 0 and 30% exploitation rates are shown.

6.1.4 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 years (i.e., in the cool phase of the PDO and with relatively good summer chum population performance) and concluded that recovery has been achieved (for both populations). It is evident that during the cool phase of the PDO that marine conditions can so override the contributions of freshwater and sub-estuarine 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 37a) and oceanic cycles (Figure 37b), which can be created by first characterizing these two factors separately, then in combination, producing a pattern of expected ocean escapement (or recruits) (Figure 37c). 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 (2008) 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 severity 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 functional 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 protection and restoration and/or without periodic supplementation or recolonization by stronger, dispersing populations.

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).

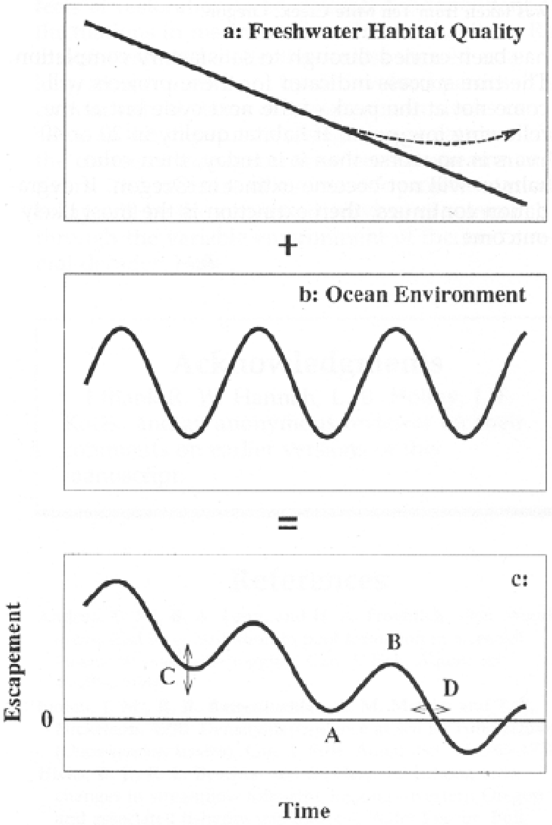

Figure 37. Conceptual model presented by Lawson (1993) of the effects of declining habitat quality and cyclic changes in ocean productivity on the abundance of a salmon population. a – Trajectory over time of habitat condition. Dotted line represents possible of habitat restoration projects. b – Generalized 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.

6.2 Considerations for Climate Change Effects

NMFS urges salmon recovery planners to consider the effects of climate change patterns on future recovery (Ford 2011). 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 have not been experienced with regularity for some much longer period of time (for example, hundreds or thousands of years). Climate change effects will likely persist for some extended period of time into the future.

In our earlier report (Lestelle et al. 2014), we examined the possible effects of climate change on summer chum viability by considering how climate change patterns may affect interannual variability of abundance. Climate change patterns are exhibiting greater environmental variation in various parts of the world—evidence shows that extreme events in some areas are becoming more extreme (e.g., Alverson et al. 2001; Furniss et al. 2010). We reasoned that if these same kinds of patterns are exhibited within the Puget Sound region, then variability in population performance should increase. We used the

VRAP model to examine the effects on viability by assuming several increments of increase in interannual variability (by increasing the coefficient of variability used in modeling, see Table 4 in this report). We found that the viability curves (as seen in Figures 35 and 36 in this report) were shifted up and to the right, setting higher thresholds for viability. The results illustrated that the importance of restoration and protection actions will increase if climate change increases population variability.

Based on the VRAP analysis, Lestelle et al. (2014) advised that some allowance be incorporated into the viability thresholds to account for the likelihood that some environmental factors will increase in their variability, thereby effectively raising and shifting viability curves for a given level of risk. Weinheimer et al. (2017) suggested that increased variability in egg to fry survival was likely to occur if peak winter flows increase as has been predicted for some streams in Western Washington (Mauger et al. 2015).

Whether an increase in summer chum population variability can be expected was a discussion topic at the HCCC climate forum held in March 2017. Several invited scientists expressed uncertainty about whether population variability will increase, stating that it would be premature to draw such a conclusion. They suggested that while certain conditions may periodically be more extreme, other conditions may also change, acting to buffer or ameliorate effects of those extremes. While the participants in the forum generally concluded that overall population performance will likely decline as a result of climate change, there was no general agreement that this would be accompanied by increased population variability.

We have concluded, as a result of the climate forum, that climate change should be addressed through adaptive management based on results of monitoring and the accrual of greater knowledge about climate-related factors affecting summer chum. We suggest, however, that the co-managers be proactive in addressing possible climate change effects. We recommend that the strategy to mitigate or offset future climate change effects be to continue restoration and protection work once delisting occurs.

This proactive strategy would embrace a principle of smart investing for the future—start now and don't wait until the need is too large to address. One key to enabling summer chum to cope with adverse climate change effects will be to improve and protect characteristics of habitat quality within freshwater, river mouth estuarine, and nearshore habitats. Also, re-establishing subpopulations recognized to have been extirpated would add capacity to a population, as well as providing greater biological diversity and spatial structure, all of which would increase a population's resilience to climate change effects. We note that there may be a large untapped potential to increase the quantity of available habitat (capacity) for the SJDF population if a reintroduction effort in the Dungeness River were to be successful (discussed in Section 7.0).

7.0 Updated Assessment for Spatial Structure and Diversity

This section presents the results of an updated assessment of the status of biological diversity and spatial structure characteristics of the ESU based on the outcome of a forum of invited scientists who reviewed the most-up-to-date information. The forum also provided a set of recommendations on what remains needed to achieve delisting.

7.1 Diversity and Spatial Structure Forum

A forum for invited scientists with expertise in the performance of the Hood Canal Summer Chum ESU, conservation biology, or salmon recovery planning was held in April 2017 to address the recovery criteria related to biological diversity and spatial structure. The purpose of the forum was to review the conclusions reached by the PSTRT for these criteria as documented in Sands et al. (2009), review the most recent available information on summer chum performance, including the findings presented earlier in this report, and to formulate recommendations for updating the biological diversity and spatial structure criteria for recovery if needed.

The forum was held over two days. Two participants (Ken Currens and Kit Rawson) had been members of the PSTRT when the recovery criteria were originally formulated. The lead author for the section of the Sands et al. (2009) report that addressed biological diversity and spatial structure was Ken Currens. Three of the participants are population geneticists (Ken Currens, Adrian Spidel, and Maureen Small). One NOAA Fisheries scientist participated (Tim Tynan). The full list of participants is given in Appendix C.

Six questions were initially used to facilitate discussion among the participants:

1. *How should recovery efforts be balanced between abundance/productivity and diversity/spatial structure?*
2. *From a practical standpoint, are biological diversity and spatial structure simply two terms that mean the same thing with regard to recovery?*
3. *What targets for biological diversity and spatial structure need to be met to delist the ESU?*
4. *What targets/characteristics need to be achieved for delisting with respect to abundance/productivity for each subpopulation?*
5. *Is there a need to give special considerations to address the Dungeness River, Skokomish River, and West Kitsap streams because of either their size and locations (Dungeness and Skokomish) or the pattern of extirpated subpopulations that exists in the surrounding geographic area (West Kitsap streams)?*
6. *What should the priorities be for reintroduction efforts?*

To answer the questions required that forum participants generally agree on a criterion for what constitutes “persistence” of a subpopulation. How abundant does a spawning aggregation need to be to be considered as persisting in a watershed to satisfy the spatial structure criterion for recovery? The participants recognized that persistence in this case is neither the presence of very few spawners in the stream nor equivalent to a viability threshold as defined for the entire population. All of the participants agreed that some functional form of “robustness” needs to be achieved as qualitatively described earlier in this document. In Section 4.3, we described a robust spawning aggregation/subpopulation as one that contributes positively to overall population health and viability and is not merely 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. Forum participants agreed with this general definition.

Following the forum, we (authors of this report) further reviewed various aspects of persistence and viability to give greater clarity to the definition of robust for the sake of application in this document.

We found that robustness as applied here can be inferred from different measures of abundance used in conservation biology, two of which were described earlier in this report. These two measures are the following:

- At the population level, the 50/500 rule is a useful guideline to avoid adverse genetic issues affecting viability, that is, in the short-term the effective population size (N_e) should not drop below 50 spawners and over a longer time period should not be less than 500 spawners (see Section 5.1 for the basis of this).
- To address demographic stochasticity concerns at the population level (such as depensation factors), minimum abundances of 350 and 300 spawners were estimated for the quasi-extinction thresholds (QET) for the Hood Canal and SJDF summer chum populations respectively for viability modeling (Sands et al. 2009). These values are within the range of the 50/500 rule.

For application at the subpopulation level, we apply the measures defined above to define robust in a way that sets what we consider to be high estimates of abundance, thereby providing a buffer against misclassification. The values given above are meant to be applied to the entire population—by applying them in the manner given here at the subpopulation level should act as a safety net. We define a robust subpopulation as one that over the previous two generations (eight years) has an average spawner abundance of at least 300 fish and does not fall below 50 fish in any year within that 8-year span. We suggest use of the arithmetic mean to calculate the previous 8-year average. Use of this average value will normally result in some years within the 8-year period being larger than 500 fish.

7.2 Findings

Each of the six questions above is addressed briefly by summarizing the key points from the discussions. Following this summary, the key conclusions that we (i.e., the authors of this report) have reached from those discussions, as well as our recommendations are provided.

1. *How should recovery efforts be balanced between abundance/productivity and diversity/spatial structure?*

The consensus view of the forum participants on this question was that the criteria for both sets of VSP parameters need to be met for recovery; balance is not relevant.

2. *From a practical standpoint, are biological diversity and spatial structure simply two terms that mean the same thing with regard to recovery?*

The consensus view of the forum participants on this question was that these parameters are not the same—they both provide aspects of population performance important to long-term viability. However, because biological diversity is not a population characteristic that commonly changes within relatively short time periods and it is usually difficult to monitor, spatial structure can be used as a surrogate measure for biological diversity within a time period relevant to recovery. As noted in Section 4.1 of this document, the spatial distribution of ecological diversity within the geographic domain of the ESU is believed to be a useful indicator of the biological diversity of the ESU (Waples et al. 2001). This assumption is the basis for how the PSTRT delineated the ecological diversity groups within the ESU (Figure 5). For practical reasons, therefore, spatial structure can be used to satisfy both parameters for recovery and delisting purposes.

3. *What targets for biological diversity and spatial structure need to be met to delist the ESU?*

The consensus view of the forum participants was that the conceptual spatial framework for the ecological diversity groups as developed in Sands et al. (2009) is appropriate for identifying where subpopulations need to exist for ESU viability. At least one robust subpopulation (as defined earlier) needs to be present in each ecological diversity group but with notable exceptions in the Dungeness and Toandos ecological groups. High uncertainty exists about the historic presence of robust subpopulations in both of those groups and therefore a requirement for a robust subpopulation is seen as unjustified.

The participants reviewed the geographic delineations for the ecological diversity groups (Figure 5) and the basis of the delineations. Ken Currens clarified that the delineations were based solely on non-genetic information due to lack of sufficient genetic data to determine population spatial structure. The premise on which the PSTRT based their conclusions is that population structure and biological diversity are closely linked to ecological diversity. Ecological diversity is largely determined by differences in habitat characteristics across an area of the scale and composition of that encompassing the Hood Canal and eastern Strait of Juan de Fuca basins.

Forum participants expressed divergent views regarding the delineations of the Mid West Hood Canal group and the Lower West Hood Canal ecological diversity groups. It was noted that the delineations had changed from those that were originally defined in the recovery plan for the ESU. The Hamma Hamma River was originally placed in the Mid West Hood Canal group in the recovery plan, but the PSTRT decided to include this watershed with Lilliwaup Creek and Skokomish River as the Lower West Hood Canal group based on their analysis.[32] Forum participants noted that the genetics of Hamma Hamma fish have been found to essentially be indistinguishable from Dosewallips and Duckabush fish (Sands et al. 2009, Small et al. 2009 and 2013). It was also noted that distinct differences exist in the flow regime patterns among the streams. The Hamma Hamma River generally has a bimodal flow pattern, showing both a mid-winter peak and a modest late spring snowmelt peak, having some similarity to the Dosewallips and Duckabush hydrographs. Lilliwaup Creek and the South Fork Skokomish River Skokomish River, however, are largely rainfall driven with peak flows occurring in early to mid-winter with little or no spring-time snowmelt pulse. The forum participants reached no consensus conclusion during their deliberation for altering the boundaries of the Lower West Hood Canal group as determined by the PSTRT. We (authors of this report) conclude that the delineations of these two ecological diversity groups should be reconsidered at a future date to enable additional evaluation.

It was noted that the delineations for two of the ecological diversity groups, the West Kitsap group and the Lower West Hood Canal group were altered over the course of the work done by the PSTRT. Their final conclusion is seen in how the units were delineated in Figure 5. Discussion among the forum participants focused on these two units.

The forum participants concluded that there is a need to update the delineation of the West Kitsap group and its application to the spatial structure criterion. Two reasons for the needed update were highlighted. The first reason is that new genetics data since the work of Sands et al. (2009) provides improved understanding about the pattern of straying among the Hood Canal subpopulations and the relatedness of subpopulations on opposite shorelines of Hood Canal. Small et al. (2013) expanded on

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[32] / The recovery plan applied a spatial structure that conceptually was similar to the one developed by the PSTRT, but some differences existed in how the geographic areas were delineated. The geographic areas were called Conservation Units in the recovery plan.

previous genetic analysis by incorporating samples collected in the 1970s from Big Beef Creek; summer chum in this stream were extirpated a few years after these samples had been collected and results of this new analysis therefore brought improved understanding to the work that had been used by the PSTRT.

The new analysis showed that the earlier conclusion in Sands et al. (2009) about straying patterns was still fundamentally sound, i.e., a stepping-stone pattern of straying is evident where most genetic exchange is between neighboring subpopulations. But whereas the PSTRT assumed that fish stray to subpopulations on opposite shorelines because of their proximity by distance, the new data showed that straying occurs between neighboring subpopulations along the same shoreline. In other words, the genetic data do not demonstrate that strays cross Hood Canal and enter streams on the opposite shoreline. The manner in which the PSTRT had delineated the West Kitsap group assumed that fish cross to the opposite shoreline in straying.

The second reason for concern about how the West Kitsap group had been delineated is the pattern of where subpopulations have been extirpated within the Hood Canal population (Figure 38); the participants reviewed this pattern, which helped clarify the issue of concern. The pattern of extirpations shows that subpopulations on the Kitsap Peninsula have been particularly susceptible to extirpation—only the Union River subpopulation can be considered robust and it is located in the extreme corner of the West Kitsap ecological group. Subpopulations on the west side of Hood Canal, particularly those north of Lilliwaup Creek are strong. Given the pattern of straying (hence potential recolonization) that occurs among subpopulations, that is along the same shoreline, there appears to be a low probability for natural recolonization of streams on the east side of Hood Canal within a near-term time frame (<20 years).

The forum participants found that due to the pattern of extirpations that have occurred within Hood Canal, and the manner in which straying and recolonization occurs, that the likelihood for recovery would be substantially improved if at least one additional subpopulation along the eastern shoreline was to be re-established, preferably in the mid-part of the shoreline. To facilitate this, it was recommended that the West Kitsap group be subdivided into two geographic units so it is clear that at least one robust subpopulation is needed within each of the new ecological groups. The two new groups would be defined as the East Hood Canal group and the South Hood Canal group. The delineation between the two groups would be at Rendsland Creek, which flows into Hood Canal near the south west tip of the Kitsap Peninsula. The distance along the shoreline of the Kitsap Peninsula between Big Beef Creek in the north and Union River in the south is approximately 50 miles (80 km) and Rendsland Creek is approximately equidistant from these two streams.

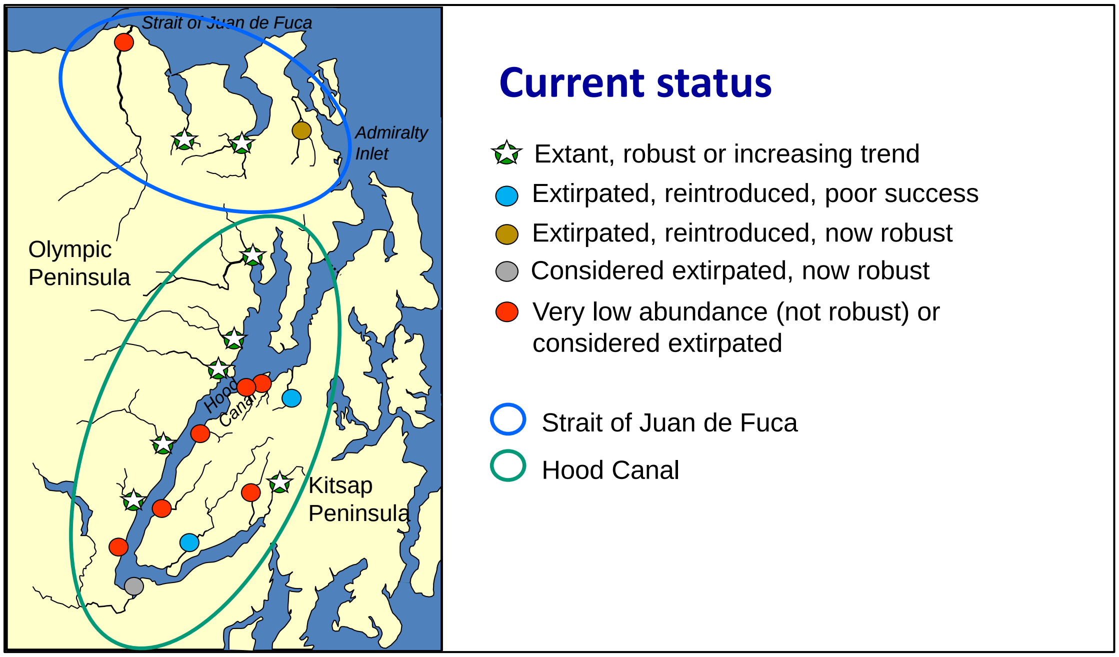

Figure 38. Current status of subpopulations identified as part of the Hood Canal Summer Chum ESU.

4. *What targets/characteristics need to be achieved for delisting with respect to abundance/productivity for each subpopulation?*

As presented in Section 4.3, abundance and productivity parameters for the subpopulations do not have to achieve a standard for low risk of extinction as do the populations. The forum participants found, however, that at least one robust subpopulation needs to be present in each ecological diversity group, where a robust subpopulation is one that demonstrates a level of vigor and health reflected in periodic positive growth rates for the subpopulation. A quantitative measure of robustness was presented above. In contrast, if only a few spawners are present from year to year in a given stream, summer chum may persist in the stream but not at a level considered robust.

As noted above, the West Kitsap should be subdivided into two new groups defined as the East Hood Canal group and the South Hood Canal group.

A description of the status of each of the subpopulations in the ESU where summer chum spawners are found is provided later in this section.

5. *Is there a need to give special considerations to address the Dungeness River, Skokomish River, and West Kitsap streams because of either their size and locations (Dungeness and Skokomish) or the pattern of extirpated subpopulations that exists in the surrounding geographic area (West Kitsap streams)?*

Summer chum continue to persist in the Dungeness River likely due to strays[33] from other subpopulations—but due to the small numbers present the spawning aggregation is not considered to be robust. The consensus view of the forum's participants is that no special considerations need to be given to boost the abundance of the spawning aggregation because of uncertainty about its historic performance. However, an experimental re-introduction effort could yield useful information to better understand the potential for establishing a robust spawning aggregation in the river. Habitat protection and restoration work is on-going in the Dungeness River watershed. It is noted that if a robust spawning aggregation could be established in the Dungeness River that it may offer the largest potential for increasing the capacity of the SJDF summer chum population due to the size of the river.

The summer chum subpopulation in the Skokomish River, considered by the PSTRT to have been extirpated, has shown a strong rebound in abundance over the past 20 years. The consensus view of the forum participants was that no special recovery efforts are warranted to be directed specifically at this subpopulation. The subpopulation now meets the criterion for being considered robust as we have defined it for this assessment.[34] It is recognized that the large restoration effort called the Skokomish River Basin Ecosystem Restoration Project, led by the U.S. Army Corps of Engineers and authorized for federal funding, will provide significant habitat benefits to the summer chum subpopulation (USACE 2015; SIT and WDFW 2017). In addition, a recently re-developed fisheries harvest plan for southern Hood Canal and the Skokomish River aimed at Chinook conservation by the Skokomish Indian Tribe and WDFW will also benefit Skokomish River summer chum (SIT and WDFW 2017).

Special consideration to be given to the West Kitsap ecological group on the Kitsap Peninsula was described under question three above. The forum's participants recommended that the West Kitsap ecological diversity group be subdivided to form two ecological groups and that at least one robust subpopulation be established in the new East Hood Canal ecological group. Furthermore, if only one robust subpopulation is established there, then it would need to be in the Dewatto River, thereby applying the spacing criterion determined by the PSTRT in Sands et al. (2009) for major spawning aggregations. The forum participants believe that the Dewatto River currently has the best potential for being re-colonized by summer chum because the watershed is considered to be relatively intact with respect to natural ecological processes. However, recent restoration efforts in the lower reaches of Big Beef Creek cannot be discounted – substantial restoration work has been done in lower Big Beef Creek over the past five years, well after the reintroduction program was terminated. It was expressed that a re-establishment of subpopulations in both Dewatto and Big Beef Creek, if possible, would be additionally beneficial to the population.

6. What should the priorities be for reintroduction efforts?

Given the importance that the forum's participants placed on re-establishing at least one robust subpopulation in the new East Hood Canal ecological diversity group, and particularly in Dewatto River, the priority for a reintroduction effort is Dewatto River. Based on available information, habitat

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[33] / Summer chum observed in the lower Dungeness River may also be "dip-ins", that is, fish that are actually in the process of migrating to their natal stream but that wander into the lower Dungeness River before moving back out to the SJDF and continuing on their way to their home stream.

[34] / The average number of spawners estimated for brood years 2009-2016 (eight years) was 722 fish (arithmetic mean) with a minimum number in this period of 25 fish (see Table 11). By including the preliminary data for 2017 (not included in this report), the minimum number of spawners has been 61 fish over an eight year period. With 2017 data, the eight year average is also increased.

conditions within this river are believed to be suitable for re-establishing a robust spawning aggregation. It is recognized, however, that major uncertainties exist about the likelihood for a successful re-introduction if undertaken. Small numbers of summer chum continue to persist in the river. It is unknown why abundance has remained so low given the strong rebounds in other Hood Canal streams, including in the Skokomish River where the natal subpopulation was considered to have been extirpated. A reintroduction effort using supplementation methods in Dewatto River could boost the abundance of spawners sufficiently to overcome a depensatory factor if such a factor is impeding a rebound.[35]

The forum's participants recognized that if the Dewatto River habitat is sufficiently intact, and depensation is not impeding a rebound, then natural recolonization processes could enable the subpopulation to recover without intervention. The issue may simply be allowing enough time for natural processes to produce the needed response. The participants also recognized that if a downturn in marine survival occurs due to the PDO phase change, then there could be a significant delay in recovery of a Dewatto subpopulation without supplementation. It would be helpful to have an assessment done of habitat conditions in Dewatto River and its estuary to reduce uncertainties about why a natural recolonization has not already occurred. Meanwhile, protection of suitable and productive habitat in Dewatto River should be identified as a priority action to stem any further potential habitat loss.

As noted under question four, an experimental re-introduction effort could be useful to learn whether the Dungeness River will support a robust subpopulation of summer chum.

We list below the conclusions of the forum's participants on the status of the subpopulations or spawning aggregations (i.e., with respect to being robust, see Section 7.1 above). Table 11 summarizes the average abundances of natural-origin spawners for each spawning aggregation over the past eight years (2009-2016).

- For the SJDF population, the various spawning aggregations are performing as follows:
    - The Jimmycomelately subpopulation is robust.
    - The Salmon-Snow subpopulation is robust and each of the constituent spawning aggregations appears to also be robust.
    - The Chimacum spawning aggregation is robust and the reintroduction effort was successful. Chimacum summer chum are considered to be a range extension of the Salmon-Snow subpopulation and further bolsters it.
    - Very few summer chum continue to be present in the Dungeness River and they are probably strays; this provides evidence, however, that some level of persistence continues to exist in the river—this is desired and meets a need for persistence, albeit at a very low level, in this river. Uncertainty about the historic status of a subpopulation in the river remains.
- For the Hood Canal population, the spawning aggregations are performing as follows:

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[35] / Depensatory factors can keep an animal population from rebounding at very low densities. An example would be predation on the population if the number killed by predators is largely independent of the abundance of the population (Roemer et al. 2002; Hilborn et al. 2014).

- The Big-Little Quilcene, Dosewallips, Duckabush, Hamma Hamma, and Union subpopulations are robust.
- The Lilliwaup subpopulation continues to be supplemented and the abundance of natural-origin spawners now exceeds our definition of robust. We note, however, that this is occurring while supplementation has been on-going..
- The Skokomish subpopulation, considered to have been extirpated by the PSTRT, is trending strongly upwards without having had direct aid by a reintroduction effort. The spawning escapement exceeded 2,000 spawners in 2016. The preliminary estimate for 2017 exceeds 4,300 spawners. Also, by including 2017 preliminary data, the minimum spawning escapement over the 8-year period from 2009 to 2017 becomes 61 fish. We now consider the subpopulation to be robust.
- The Tahuya and Big Beef reintroductions, which have been completed, have not produced strong responses in natural production. Relatively small numbers of summer chum continue to persist in these streams and the spawning aggregations are not considered robust. Returns to Big Beef Creek have averaged less than 100 fish annually with only seven spawners returning in 2017. Returns to the Tahuya River have been higher, averaging just over 1,000 spawners including supplementation fish with proportions of natural-origin spawners ranging from 15% in 2012 to 82% in 2016. Degraded habitat conditions are the likely cause of the poor response to the reintroduction efforts. A contributing factor in the poor response in Big Beef Creek may have been the use of Quilcene stock in the reintroduction effort. Quilcene stock are now seen as being less suited to Big Beef Creek than Union River stock based on the findings of Small et al. (2013). We note that substantial restoration work has been done in lower Big Beef Creek in the past five years, well after the reintroduction program was terminated. We suggest that re-initiating the reintroduction effort using Union River stock or Lilliwaup Creek stock (or a combination) may now be appropriate.
- The Dewatto subpopulation is persisting at a low level of production. The average number of adult recruits over the past five years has been less than 200 fish. The subpopulation is not robust.
- The Anderson spawning aggregation, believed to have been a minor aggregation historically, shows no indication of a rebound. Chronic low-water conditions and extensive beaver dam activity have resulted in years where no summer chum are observed. Since 2010 there have only been two years with summer chum escapements above zero, two spawners in 2012 and seven in 2017. This spawning aggregation is not robust.
- No summer chum are produced in streams within the Toandos ecological diversity group (see Figure 5 in this report) and it is assumed for the sake of recovery that none were produced there historically.

Table 11. Summary of average abundances of natural-origin summer chum spawners in subpopulations/spawning aggregations within the SJDF and Hood Canal populations over the past eight years (2009-2016). Metrics shown are arithmetic mean (AM), geometric mean (GM), minimum spawners (Min), and maximum spawners (Max).

Population	Subpopulation	AM	GM	Min	Max
SJDF	Jimmycomelately	2,653	1,636	202	6,532
	Salmon-Snow	3,480	3,154	1,437	7,688
	Chimacum	1,581	1,407	640	3,066
Hood Canal	Big/Little Quilcene	8,395	6,043	1,490	17,366
	Dosewallips	3,321	2,589	1,094	9,899
	Duckabush	4,780	4,212	1,515	8,470
	Hamma Hamma	2,130	1,721	597	4,845
	Lilliwaup	787	421	60	1,912
	Skokomish	722	295	25	2,674
	Union	1,393	1,070	285	3,512
	Tahuya	299	158	8	1,212
	Dewatto	116	68	9	290
	Anderson	< 5	< 5	0	< 5
	Big Beef	97	80	15	156

7.3 Conclusions and Recommendations

Our conclusions and associated recommendation related to biological diversity and spatial structure are given below.

- The recovery requirements for biological diversity will be satisfied by meeting the requirements for spatial structure within the ESU.
- Requirements for spatial structure are currently being met across the ESU except on the Kitsap Peninsula.
- We recommend that the West Kitsap ecological diversity group be subdivided at Rendsland Creek into an East Hood Canal group and a South Hood Canal group.
- The robust status of the Union River subpopulation meets the minimum need for spatial structure in the South Hood Canal group.
- We recommend that at least one subpopulation be re-established in the new East Hood Canal group, either through natural recolonization processes or a reintroduction using supplementation methods. A new reintroduction effort within this geographic area, if it were to be successful, would likely accelerate the re-establishment of a subpopulation within this area.
- The re-establishment of the Dewatto River subpopulation is seen at this time as being the highest priority for the new East Hood Canal group because it is approximately halfway between the most northern subpopulation in the West Kitsap group as it was originally defined and Union River, located in the eastern corner of the West Kitsap group. The habitat in Dewatto River is also believed to be relatively intact. We recommend that an assessment of habitat conditions be

made in the Dewatto River and its estuary to help resolve uncertainties about the river's current conditions. Re-establishment of a Dewatto subpopulation may produce a source subpopulation to help facilitate natural recolonization of other streams on the Kitsap Peninsula. A robust Dewatto spawning aggregation would satisfy the PSTRT's original intent of 25 miles (40 km) between major spawning aggregations.

- If a new reintroduction effort is undertaken within the new East Hood Canal group—one directed at the Dewatto River—a parallel re-introduction could also be considered for Big Beef Creek using the same donor stock source. Such an approach could be helpful in understanding why the previous re-introduction effort failed in Big Beef Creek and may provide information on why Kitsap streams north of Union River have generally not responded like other streams in Hood Canal to favorable PDO conditions. We also recommend that if a new reintroduction effort is to be undertaken that consideration should be given to using a combination of Union River and Lilliwaup donor stocks. The previous attempt at a reintroduction in Big Beef Creek employed Quilcene stock. The findings of Small et al. (2013) strongly suggest that the historic Big Beef Creek stock was more closely related to Union River and Lilliwaup Creek fish than to those subpopulations directly across Hood Canal from Big Beef Creek. Further discussion among the co-managers and their geneticists would be required on this matter if a reintroduction is undertaken.

8.0 Harvest Considerations

This section is intended to facilitate discussions among the co-managers to help address harvest issues prior to and following delisting based on information in this report. We provide here several examples for how harvest-related actions might be shaped to aid in recovery, while at the same time fostering updated harvest provisions consistent with the performance of the ESU's populations and subpopulations. We recognize that there is a need for a comprehensive review of the existing Base Conservation Regime (BCR) that considers the information presented in this report.

Beginning in the early 1990s, the co-managers implemented a harvest management regime to significantly curtail fisheries impacts on the Hood Canal Summer Chum ESU to begin a rebuilding process (WDFW and PNPTT 2000). Some of the measures under the regime were also adopted and applied in relevant Canadian fisheries to reduce their mixed-stock fishery impacts on the Hood Canal ESU.

The fisheries regime that was developed, which became part of the Summer Chum Salmon Conservation Initiative (SCSCI) and was adopted in the HCCC summer chum recovery plan (HCCC 2005), resulted in sharp reductions in harvest impacts on the ESU (Figure 3). This harvest regime, called the BCR, has remained in place with little change to the present time. The goal of the BCR was to protect and begin restoring the ESU in the short-term while contributing to restoring it in the long-term to a desired productive state that could sustain both directed and incidental harvests of the ESU's summer chum.

The BCR allows for no fisheries specifically directed at the Hood Canal ESU's populations—all harvest of these fish occurs incidentally in fisheries directed at other species or populations. The development and implementation of the regime, therefore, has required rigorous attention to shaping and regulation of fisheries directed at other target species or populations.

The co-managers recognized when developing the BCR that the harvest strategies employed would “continue to evolve and adapt as additional information is collected, analyzed, and incorporated” (WDFW and PNPTT 2000). They stated:

> “At present, because of the lack of sufficient information on summer chum productivity, it is not possible to construct a regime based on more sophisticated biologically-based objectives such as maximum sustained yield (MSY). The combination of specific management actions and fishery specific exploitation rates comprising the Base Conservation Regime is based on a conservative integration of the existing data and management experience.”

It was intended for the regime to be responsive to feedback mechanisms in order to provide for adaptive management towards meeting the goal of protecting the ESU, while maintaining harvest opportunities on other species (WDFW and PNPTT 2000).

We review several aspects of the BCR in this section, propose that several modifications to the regime be considered based on current understanding of the ESU, and propose that two additional harvest regimes be formulated, one to be used during cool phases of the PDO when population performance is strong but prior to delisting and one to be used after delisting. We suggest that developing a draft recovered harvest regime now (before delisting) would incentivize the parties to continue working hard to reach recovery and also inform other planning processes about future expectations. The two new regimes proposed here are called a Recovering Conservation Regime (RCR) and a Recovered Harvest Regime (RHR).

It is important to note that we have not attempted to do a complete review of the BCR in light of the information presented in this report. Such a review can only be done by co-managers and their appropriate personnel. We emphasize here that we have only done an initial and fairly cursory review for the sake of providing examples of elements that we think will merit consideration.

The elements we propose for consideration in updating the BCR and for developing the new regimes are based on these key observations about the status and performance of the ESU’s populations:

- The ESU’s populations, while not yet recovered, demonstrate clear, significant improvements in performance compared to their status when the BCR was initially developed;
- The strong performance of the ESU in recent years is the result of both improved marine survivals and the benefits of restoration actions, which include harvest actions;
- The populations and their subpopulations demonstrate much stronger performance during cool phases of the PDO and greater capability to provide harvest benefits during those times; and
- The Big and Little Quilcene subpopulation demonstrates stronger performance than all other subpopulations in the ESU and a stronger resistance to viability risk (Figure 36).

8.1 Base Conservation Regime

The BCR was intended to provide the needed level of protection from adverse harvest impacts necessary to begin the process of recovery for the ESU. It is important to note that the developers of the regime assumed that recovery was far enough in the future and information sufficiently lacking that only interim management objectives for the regime could be formulated at the time of its development (Section 3.5.6 in WDFW and PNPTT 2000). These objectives guided the development of specific parts of the regime—but it was noted that “these shall be modified in response to new information as it becomes available.”

The BCR is comprised of a conservative four-way set of controls:

1. A base set of fishery-specific management actions for fisheries in pre-terminal (both Canadian and U.S.), Washington terminal, and Washington extreme terminal areas (Section 3.5.6.1, Tables 3.29-3.34 in WDFW and PNPTT 2000);
2. Management unit and stock abundance and escapement thresholds that trigger review of and possible adjustment of the management actions (Section 1.7.3, Appendix Report 1.5 in WDFW and PNPTT 2000);
3. Expected fishery specific exploitation rate targets and ranges based on the application of the BCR on the ESU’s summer chum management units (Section 3.5.6.1 in in WDFW and PNPTT 2000); and
4. Overall management performance standards based on natural production against which to assess success of the regime and make necessary adjustments (Section 3.5.6.3 in WDFW and PNPTT 2000). The actions required depend both on the status of the management unit and the stocks within them, with the most conservative controls prevailing.

These controls were expected to initiate rebuilding the populations by providing incremental increases in spawning escapement over time, while allowing a limited opportunity to harvest other species. In total, the actions were expected to result in, on the average, a 10.9% total (range = 3.3-15.3%) exploitation rate on the Hood Canal management units and an 8.8% (range=2.8-11.8%) exploitation rate on Strait of Juan de Fuca management units. The Quilcene/Dabob Bay Management Unit would be managed for a stepped fishing schedule based on pre-season and/or in-season assessment of spawning escapements returning to the Big and Little Quilcene rivers (PNPTT and WDFW 2014). It is noted that management in the Quilcene/Dabob Bay Management Unit posed particular challenges to harvest managers due to the overlapping run timing of summer chum returning to the Big and Little Quilcene rivers and large numbers of hatchery coho returning to the Quilcene National Fish Hatchery on the Big Quilcene River and net pens in Quilcene Bay.

Since 2000 the BCR has accomplished its objective of controlling and reducing incidental catches of summer chum in fisheries targeting other species. Table 12 summarizes exploitation rates on the ESU’s two populations for 2000 to 2016, as well as spawning escapements to the Big and Little Quilcene rivers. Results show that exploitation rates (assessed post-season) were well below the BCR targets for Canadian fisheries, U.S. pre-terminal fisheries, and the Hood Canal terminal area fisheries. In Canadian fisheries, the lower than predicted level of exploitation has been the result of the absence of Canadian commercial fisheries for sockeye and pink salmon in most years (PNPTT and WDFW 2014). The same management considerations have also acted to reduce the U.S. pre-terminal exploitation to lower than anticipated levels. Terminal area interceptions are normally expected in the Hood Canal fisheries (Strait of Juan de Fuca has no applicable terminal fishing areas). However, again because of other factors, such as fishery restrictions to protect Chinook and a reduction in fishing effort for coho, exploitation rates have been lower than expected (PNPTT and WDFW 2014).

The co-managers concluded in their most recent 5-year review of the SCSCI that the BCR’S performance has far exceeded their expectations for how well it would limit summer chum impacts (PNPTT and WDFW 2014). The BCR was designed to be particularly conservative due to uncertainties about condition of the populations when the BCR was developed—it is clear that the plan's authors accomplished that purpose.

After reviewing the results of the BCR in light of new understanding about the performance of the ESU, we suggest that two broad changes could be considered for the BCR without altering the intent of the BCR or its original limits to harvest: (1) the BCR be applied when population abundance is weak, such as during warm phases of the PDO or when there is evidence of weakness based on pre-season forecasts and/or in-season updates but not during periods when performance is strong, as seen in recent years, even though delisting has not yet occurred; and (2) where opportunities exist, provide for some liberalization of fisheries not targeting summer chum so that overall resulting exploitation rates are closer to the upper exploitation rate limits specified in the BCR.

Prior to delisting and during periods of strong population performance by the ESU, we suggest that fisheries be regulated according to a new harvest regime called a Recovering Conservation Regime (RCR). The ESU's populations, and particularly the Hood Canal population, have sufficient productivity and capacity buffer during periods of strong performance to enable some relaxation of harvest restrictions. This could provide the co-managers greater flexibility to conduct their fisheries where opportunities exist, such as in the Quilcene/Dabob Bay Management Unit.

We also suggest that consideration be given to liberalizing restrictions specified in the BCR to allow some additional flexibility in regulating fisheries so that total resulting exploitation rates are closer to the stated BCR limits than has occurred (see Table 12). We suggest that a full review of the current restrictions would be useful to determine if some additional flexibility can be accommodated (Tables 3.29 to 3.34 in WDFW and PNPTT 2000). We are not suggesting that the overall exploitation rate limits on the populations be relaxed when the BCR is in effect—simply that greater flexibility be provided if possible under the existing limits.

Table 12. Exploitation rate limits and estimated actual resulting exploitation rates associated with the Base Conservation Regime (BCR) for years 2000 to 2016. Values in red font and italics indicate that the BCR exploitation rate limit or minimum escapement objective was not met. From Scott Bass (PNPTC, personal communications).

Fishery	BCR Limits	Actual Exploitation Rates by Fishery																	Average
	(Range)	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	
Canada	6.3% (2.3% - 8.3%)	0.3%	0.4%	0.2%	0.1%	0.1%	0.2%	0.1%	0.5%	0.6%	0.5%	0.2%	0.3%	0.3%	0.1%	0.2%	0.2%	0.1%	0.3%
U.S. Preterminal Fisheries																			
Juan de Fuca	2.5%	0.2%	0.4%	0.2%	0.8%	0.1%	0.3%	0.3%	0.5%	0.8%	0.2%	0.6%	0.2%	0.3%	0.4%	0.1%	0.2%	0.0%	0.3%
Hood Canal	2.5% (0.5% - 3.5%)	0.2%	0.4%	0.2%	0.8%	0.1%	0.3%	0.3%	0.5%	0.8%	0.2%	0.6%	0.2%	0.3%	0.4%	0.1%	0.2%	0.0%	0.3%
Hood Canal Mixed Terminal Fisheries	2.1% (0.5% - 3.5%)	0.4%	0.9%	1.1%	0.0%	0.1%	0.2%	0.9%	*4.0%*	0.1%	0.2%	0.8%	0.3%	0.6%	1.9%	0.5%	0.4%	0.5%	0.8%
Quilcene Extreme Terminal[1]																			
Exploitation Rate	n/a	7.4%	0.6%	4.8%	0.0%	20.2%	2.6%	7.8%	8.8%	10.9%	10.7%	0.0%	1.9%	2.0%	2.4%	7.9%	8.2%	8.3%	6.1%
Quilcene escapement	1,500 (min.)	5,898	6,373	4,487	12,733	38,153	6,672	11,876	2,526	3,861	*1,490*	2,073	2,580	11,739	7,950	9,685	17,366	14,284	9,397
Regional Totals																			
Juan de Fuca	8.8% (2.8% - 11.8%)	0.4%	0.7%	0.4%	0.8%	0.2%	0.5%	0.4%	0.9%	1.4%	0.6%	0.8%	0.5%	0.5%	0.5%	0.3%	0.4%	0.1%	0.6%
Hood Canal	15.9% (8.3% - 20.3%)	8.2%	2.1%	6.2%	0.8%	*20.5%*	3.3%	9.1%	13.7%	12.5%	11.5%	1.6%	2.7%	3.1%	4.7%	8.7%	8.9%	8.8%	7.4%
ESU	- - -	7.5%	3.8%	4.1%	0.8%	18.7%	2.3%	7.6%	12.9%	10.7%	12.6%	1.3%	1.8%	3.0%	4.1%	7.4%	6.5%	7.6%	6.6%

[1] No fisheryspecific exploitation rate is defined for this fishery. Instead, management relies on a stepped fishing schedule based on an inseason assessment of natural escapement. Up to 2 days of gillnet fishing are allowed per week as expected escapement increases; a 1 day per week gillnet fishery is expected to add 5% to the Hood Canal population exploitation rate (see SCSCI Table 3.35) and a higher exploitation rate is expected for a 2 day per week gillnet fishery.

In some cases, restrictions under the BCR may be outdated, for example the imposition of an area restriction on the Skokomish River at the Highway 106 Bridge. The updated Skokomish Chinook recovery plan (SIT and WDFW 2017) allows for fisheries to occur in the river upstream of the mouth during July and August and for the river to close to harvest during September—no restriction is placed on the river above or below the Highway 106 Bridge. The September closure is intended to aid in recovering Skokomish Chinook, but it will also provide harvest protection to summer chum during their primary period of river entry. We suggest that a complete review of each of the BCR restrictions would be useful to provide for greater consistency in how ESA-listed species are managed between plans and to give greater flexibility in shaping fisheries where possible and where current status would support it.

We evaluated the patterns of population abundance to determine logical breakpoints for delineating between weak and strong population run sizes (see Figure 6). We used a frequency histogram of total run size of natural-origin recruits (NORs) to examine separation of weak from strong runs (Figure 36); the results are consistent with what we would generally conclude from simply separating run sizes based on their association with the warm or cool phases of the PDO. Figure 39 provides frequency histograms of run sizes for years 1983 to 2016, the return years that encompass brood years 1979 to 2012 used elsewhere in this report.

Based on the patterns seen in both Figures 6 and 39, we find that run sizes of natural-origin fish can be effectively separated into weak and strong categories using 10,500 and 5,000 for the Hood Canal and SJDF populations, respectively. These delineations identify run sizes grouped broadly into two categories for each population: a strong run category and a weak run category. The strong run category generally consists of runs produced during the cool PDO phase. Over the entire period of record (43 years beginning in 1974), 44.2% and 23.3% of the returning runs for the Hood Canal and SJDF populations respectively returned as strong runs using these criteria (Table 13). Over the past 20 years, 60% and 45% of the runs exceeded the strong run delineations for the Hood Canal and SJDF populations, respectively (Table 13).

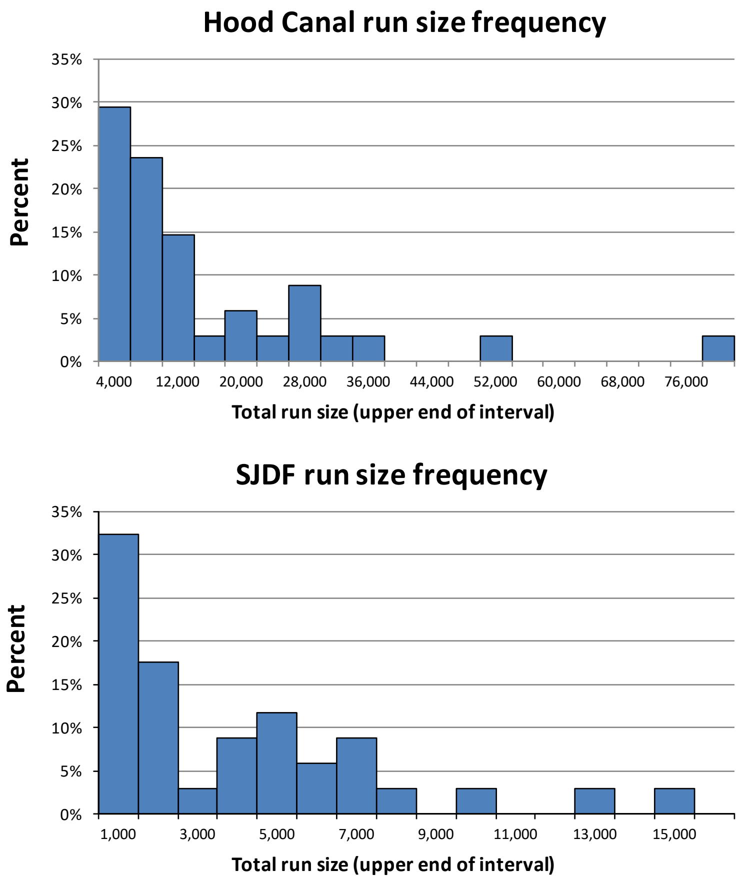

Figure 39. Frequency histograms for population abundance (natural origin recruits) for return years 1983 to 2016 for the Hood Canal and SJDF summer chum populations.

Table 13. Number and percentage of years when the Hood Canal and SJDF summer chum populations exceeded suggested abundance levels to be used in designating strong performance run sizes.

Period	No. of years	No. years >10,500	%
Hood Canal population			
All years in record (1974 - 2016)	43	19	44.2%
Past two PDO regimes (1983 - 2016)	34	13	38.2%
Previous 20 years (1997 - 2016)	20	12	60.0%
SJDF population			
Period	**No. of years**	**No. years >5,000**	**%**
All years in record (1974 - 2016)	43	10	23.3%
Past two PDO regimes (1983 - 2016)	34	9	26.5%
Previous 20 years (1997 - 2016)	20	9	45.0%

We propose that these delineations between weak and strong run size categories be considered for in-season management of the relevant fisheries using both pre-season and in-season estimators. The earliest in-season estimator that could be applied would be available on about August 10 using a still preliminary model being developed by Scott Bass (PNPTC, personal communications). This estimator uses a combination of Canadian test fisheries and PDO index values. Additional work is still needed to refine and improve in-season estimation. If the run size of natural-origin fish is predicted to be less than 5,000 for the SJDF population or 10,500 fish for the Hood Canal population, then the BCR could be in effect for the population not meeting that threshold. If larger natural-origin run sizes are predicted, a RCR could be in effect.

8.2 Recovering Conservation Regime

We propose that a RCR be used during years prior to delisting when the ESU's populations demonstrate what we are calling a strong abundance status, as outlined in the previous section. The strong abundance status would exist in years when the Hood Canal and SJDF populations exceed 10,500 and 5,000 natural-origin fish for the two populations, respectively. The status of each population would be determined separately and their status could differ between populations. It appears that marine survival conditions for the two populations can differ due to a lag effect of the PDO for the Hood Canal population, or regional differences in peak winter streamflow conditions (see Section 6.1.2.3).

The purpose of a RCR would be to provide the co-managers greater flexibility in conducting their fisheries where additional opportunities for harvest of other target species might exist with a relaxation of constraints imposed by the BCR. This report demonstrates that performance of both populations is strongly affected by the marine survival regimes that correspond with PDO phases. When marine survival conditions are good, corresponding with the cool PDO phase, the ESU's populations, and particularly the Hood Canal population, have sufficient productivity and capacity buffer to justify some relaxation of harvest constraints.

To help inform the development of a RCR and RHR, it is helpful to consider population performance reference points, such as MSY and the unfished equilibrium run sizes based on spawner-recruit (S-R) analysis (Caddy and Mahon 1995; Walters and Martell 2004). Reference points can be used to inform the co-managers about setting conservative limits on exploitation rates and spawning escapement thresholds for the ESU's populations. We are not suggesting that MSY be a management objective because of the difficulties that it can cause in management, both theoretically and practically (Walters and Martell 2004). Reference points based on the concept, however, can be used to inform the development of conservative, safer ranges for exploitation rates (Cooney 1984; Walters and Martell 2004). We also propose that in developing a RCR and RHR that consideration be given for differences in performance characteristics among subpopulations, i.e., by providing greater protection to the weakest subpopulations and more flexibility for impacts on the strongest subpopulations.

Table 14 identifies spawning escapement reference points for the SJDF and Hood Canal summer chum populations, along with those for the Big and Little Quilcene subpopulation, based on up-to-date S-R analysis. The reference points are theoretical MSY spawner abundances (i.e., estimated spawners corresponding with MSY) and the unfished equilibrium run size abundance (i.e., the estimated geometric mean run sizes that would be expected in the absence of any fishing). The analysis was done for three time periods: all brood years within the data record (1974-2012), brood years corresponding to the warm PDO regime (1979-1998), and brood years associated with the cool PDO regime beginning in 1999. In addition, we show the results for both the Beverton-Holt and Ricker S-R curves (see Figure 13).

Results based on the Ricker S-R model are considered to be more conservative than those using the Beverton-Holt form (see Walters and Martell 2004).

Table 15 lists estimated MSY exploitation rates for the populations, as well as for each of subpopulation using the Beverton-Holt, Ricker, and Hockey Stick forms of the S-R relationships. Estimates are provided for each of the three relevant time periods of interest as listed above. These estimates of MSY exploitation rates will generally be most conservative using the Hockey-Stick form of a S-R relationship, followed by the Ricker, then the Beverton-Holt, though the Ricker form sometimes provides the most conservative estimate. These estimates of MSY exploitation rate are another set of reference points that can help inform the development of reasonable, safe harvest regimes.

We suggest that a RCR could employ exploitation rate-type management, along with spawning escapement floors (Cooney 1984), for each of the two populations designed to protect the populations while liberalizing exploitation compared to the BCR to give greater flexibility to the co-managers. More specifically we suggest a RCR to include the following:

- Determination of whether population performance for a given year is expected to be weak or strong (Table 13) based on both pre-season and/or in-season estimators;
- Use of intermediate exploitation rate levels that would be greater than, or perhaps overlap with, the upper limits range defined by the BCR (see Table 12) but less than levels to be used in a RHR;
- Implementation in fisheries that have been developed to harvest other species, yet because a RCR would employ allowable exploitation rates on summer chum there would be no restriction based on ratios of species composition in the catch—this could enable the full execution of the fisheries provided the summer chum allowable exploitation rate is not exceeded. This provision could be particularly helpful for fisheries in the Quilcene/Dabob Bay Management Unit.

While a more complete analysis of exploitation rates will be needed to establish a RHR than what we describe in the next section, we propose that the rates would be well less than those identified by the theoretical MSY rates seen in Table 15. We suggest that appropriate rates may be in the neighborhood of 40% on the strongest subpopulations and much less on the weakest subpopulations. Assuming that rate is reasonable for the sake of this report, we suggest that RCR rates might be in the neighborhood of 20-25% at the upper limit.

8.3 Recovered Harvest Regime

The purpose of a Recovered Harvest Regime (RHR), once it is developed, would be to describe the allowable limits to harvest and associated exploitations on the delisted ESU. We suggest that the co-managers develop a draft regime in order to better understand the possibilities for harvest with delisting. Sufficient data currently exists to begin development of the regime.

Table 14. Estimated MSY spawners and unfished equilibrium abundance levels for the SJDF and Hood Canal summer chum populations and the Big and Little Quilcene subpopulation during three time periods of brood years. The reference points are shown as estimated with the Beverton-Holt (BH) and Ricker spawner-recruit relationships. Blanks in the table indicate that an estimate could not be made.

Population	Subpopulation	Brood years	MSY spawners		Unfished equilibrium abundance	
			BH	Ricker	BH	Ricker
SJDF	All	1974-2012	576	1,062	1,720	2,241
		1979-1998		455		1,097
		1999-2012		2,389		6,403
Hood Canal	All	1974-2012	4,710	8,555	13,377	19,319
		1979-1998	1,491	3,050	5,026	7,196
		1999-2012		11,316		26,719
	Quilcene	1974-2012	765	4,393	4,683	11,479
		1979-1998	354	1,633	2,575	5,042
		1999-2012		5,440		13,173

As noted in the previous section, the reference points provided in Tables 14 and 15 are informative for how a RHR could be designed. We suggest, for example, that the RHR might employ exploitation rate-type management, along with spawning escapement floors (Cooney 1984), though other conceptual approaches exist. Based on the estimates of MSY exploitation rates in Table 15, upper limits to exploitation rates might be in the neighborhood of 40%. This rate is less than estimates of the MSY rates for each population. Under such an approach, we would envision that the rates would only be applied fully during years of strong run sizes, thereby providing a safety margin for the populations. We think that it might be necessary to have two conditions for a RHR, one that would be more liberalized for use during strong abundance years (when the 40%-type rate would be used) and one during weak abundance years. During weak abundance years, rates in the neighborhood of those to be used for a RCR or BCR may be most appropriate.

Table 15. Estimated MSY exploitation rates for the SJDF and Hood Canal summer chum populations and their subpopulations during three time periods of brood years. The estimates are shown as estimated with the Beverton-Holt (BH), Hockey-Stick (HS), and Ricker spawner-recruit relationships. Blanks in the table indicate that an estimate could not be made. It is important to recognize that this report is not suggesting use of these reference points as management objectives—see text.

Population	Subpopulation	Brood years	BH	HS	Ricker	Range	
						Low	High
SJDF	All	1974-2012	0.24	0.11	0.19	0.11	0.24
		1979-1998		0.59	0.42	0.42	0.59
		1999-2012		0.61	0.59	0.59	0.61
	JCL	1974-2012	0.02			0.02	0.02
		1979-1998					
		1999-2012	0.49	0.56	0.21	0.21	0.56
	Salmon-Snow	1974-2012	0.42	0.22	0.27	0.22	0.42
		1979-1998		0.62	0.54	0.54	0.62
		1999-2012		0.62	0.55	0.55	0.62
Hood Canal	All	1974-2012	0.53	0.40	0.42	0.40	0.53
		1979-1998	0.70	0.50	0.55	0.50	0.70
		1999-2012		0.55	0.47	0.47	0.55
	Quilcene	1974-2012	0.81	0.72	0.73	0.72	0.81
		1979-1998	0.85	0.80	0.81	0.80	0.85
		1999-2012		0.56	0.64	0.56	0.64
	Dosewallips	1974-2012	0.40	0.32	0.28	0.28	0.40
		1979-1998	0.39	0.22	0.28	0.22	0.39
		1999-2012		0.64	0.39	0.39	0.64
	Duckabush	1974-2012	0.36	0.23	0.27	0.23	0.36
		1979-1998	0.70	0.50	0.53	0.50	0.70
		1999-2012	0.67	0.48	0.54	0.48	0.67
	Hamma Hamma	1974-2012	0.34	0.16	0.22	0.16	0.34
		1979-1998	0.73	0.49	0.46	0.46	0.73
		1999-2012		0.65	0.79	0.65	0.79
	Lilliwaup	1974-2012	0.60	0.70	0.30	0.30	0.70
		1979-1998		0.52	0.05	0.05	0.52
		1999-2012	0.85	0.58	0.73	0.58	0.85
	Union	1974-2012	0.73	0.59	0.49	0.49	0.73
		1979-1998		0.75	0.86	0.75	0.86
		1999-2012	0.80	0.38	0.23	0.23	0.80

9.0 Summary of Recommendations and a Decision Pathway for Recovery

The performance of the Hood Canal Summer Chum ESU since about 2000 has strongly rebounded compared to what it was during most of the prior 15 years. The rebound can be attributed to two causes: recovery actions and beneficial marine survival factors positively affecting adult abundance. Based on analyses given in this document, we conclude that there is a reasonable potential for delisting of the ESU from the protections of the ESA within the next decade.

We provide here our recommendations for addressing ten issues to expedite progress in planning for possible delisting. These issues will require attention by the parties having recovery planning authority and responsibilities—namely by HCCC as the regional recovery organization, WDFW and relevant tribal co-managers, and NMFS. The ten issues are the following:

1. Viability criteria;
2. Harvest regimes;
3. Broad sense recovery goals;
4. Lilliwaup Creek supplementation;
5. Dungeness River reintroduction;
6. Status of threats to the ESU;
7. Climate change planning;
8. Need for continuing restoration efforts;
9. Monitoring, evaluation, and research; and
10. Accounting for PDO-related effects – a decision pathway.

Issue number ten, which incorporates our final recommendation near the end of this section, provides a decision pathway that addresses the largest critical uncertainty potentially affecting a delisting decision, that is, the extent that marine survival conditions (i.e., PDO-related factors) have driven the rebound since 2000. If marine survival drops sharply, which could occur based on the phase shift in the PDO index seen in 2014, the question then becomes how the populations will perform with the combination of reduced survival and the benefits of restoration actions that have occurred.

Each issue is described briefly followed by our recommendations.

1. **<u>Viability criteria:</u>** This report provides updated recovery criteria for VSP parameters based on the most up-to-date information available.
    - <u>Recommendation</u>: We recommend that the criteria presented in Section 5.3, and specifically Table 5, be used to set updated recovery criteria for the capacity and productivity metrics and expected minimum spawning escapements. Once harvest regimes are defined, Table 5 along with supporting information used in its formulation, would be applied to finalize the capacity and productivity criteria along with expected spawning escapements so they are consistent with adopted harvest regimes.

- Recommendation: Based on conclusions reached in the diversity forum, we recommend that spatial structure be used as the single metric to represent both biological diversity and spatial structure as described in Section 7.0. We recommend that the spatial structure criterion be considered satisfied for the sake of delisting provided each of the following provisions is met.
    - All spawning aggregations currently considered to be robust (as defined in this report) are maintained in this condition; these aggregations are Jimmycomelately, Salmon-Snow, Chimacum, Big-Little Quilcene, Dosewallips, Duckabush, Hamma Hamma, Lilliwaup, Skokomish, and Union.
    - A robust Dewatto spawning aggregation is re-established either by re-introduction or by natural re-colonization unless there is a rebound seen in both Big Beef Creek and Tahuya River (to meet the spacing criterion of subpopulations)—either one of these events combined with the other provisions under this recommendation would be considered as satisfying the spatial structure criterion. (We consider that a re-introduction effort into Dewatto River would be the fastest way of re-establishing a robust spawning aggregation, particularly if a downturn occurs in marine survival due to PDO effects.)
    - Summer chum continue to persist in the Dungeness River either through straying, persistence of the existing stock, or a re-introduction effort.

2. **Harvest regimes:** The Base Conservation Regime (BCR) was formulated in the 1990s to minimize incidental harvests of the ESU's populations during the recovery period. More specifically, it was developed at a time when the populations were at critically low levels and strongly affected by the warm phase of the PDO. The co-managers recognized when developing the BCR that the harvest strategies employed would "continue to evolve and adapt as additional information is collected, analyzed, and incorporated" (WDFW and PNPTT 2000).

    - Recommendation: We recommend that three harvest regimes be considered for use under different circumstances. The regimes could incorporate elements like those presented in Section 8.0. The three regimes are listed below.
        - An updated BCR be applied prior to delisting and when population sizes are particularly low due to low marine survival regimes (i.e., during identified warm phases of the PDO). Updated elements could recognize, for example, how harvest management has been revamped in the Skokomish River as part of the Skokomish Chinook Recovery Plan (SIT and WDFW 2017). Provisions within the BCR as currently exist could be loosened where opportunities exist to result in overall exploitation rates closer (but not to exceed) to the upper guideline limits than has typically occurred in past years.
        - A Recovering Conservation Regime (RCR) be applied prior to delisting and when population sizes are strong due to high marine survival phases (i.e., during cool phases of the PDO). The overall exploitation rates to be applied could be higher than those used in the updated BCR but lower than those in a RHR – we suggest that they would be between the two levels. Further analysis will be needed to investigate the appropriate level of harvest during cool phases of the PDOs. It would be prudent to consider different levels of harvest in the various geographic areas to strive to place higher harvest impacts on the strong subpopulations and less impacts on weaker subpopulations. Consideration will be needed for the appropriate harvest levels in

mixed subpopulation marine areas. These considerations would help protect the spatial structure of the ESU.

- o A Recovered Harvest Regime (RHR) be applied after delisting. Developing a draft of the provisions for this harvest regime now (before delisting) would incentivize the parties to continue working hard to reach recovery and also inform other planning processes about future expectations. We find that there is sufficient information available now to develop a RHR. It also would be prudent to consider different levels of harvest in the various geographic areas to place higher harvest impacts on the strong subpopulations and less impacts on weaker subpopulations, especially in mixed subpopulation areas. This would help protect the spatial structure of the ESU.

3. **Broad-sense recovery goals:** Broad-sense goals have not been defined for the ESU other than as a broadly-worded goal to meet both viability standards and provide for ecosystem services, such as harvest. The recovery plan (HCCC 2005) provides initial draft language for this goal. Two questions are important to be addressed by the co-managers: (1) Should delisting be considered if broad-sense goals have not been defined, or if they have been defined, can delisting occur if those broad-sense goals are not being achieved? (2) If broad-sense goals need to be achieved for delisting, what specific goals must be met beyond simply meeting the minimum thresholds for viability?

   - Recommendation: We recommend that broad-sense goals be defined as exceeding viability targets by a sufficient margin to ensure to the extent possible that either a RCR or a RHR can be safely implemented on a consistent basis. We present this as two options, though we recognize there are other options for defining broad-sense goals. Discussions among the co-managers are needed to address the two questions above.

4. **Lilliwaup Creek supplementation:** Supplementation actions have been on-going in Lilliwaup Creek since their initiation in 1992 due to prolonged weakness of the subpopulation until recently (see Figure 9). Eggs were again procured from native broodstock in 2017 for fry releases in 2018.

   - Recommendation: We recommend that the effort be ended with the 2017 egg take to determine the response of the subpopulation without this intervention. The effort can be restarted if it is determined to be needed following a period of returns without supplementation fish. We note, however, that further consideration of some continuing supplementation in Lilliwaup Creek may be needed if a new reintroduction effort is undertaken on the Kitsap Peninsula (for example, in Dewatto River). In that case, Lilliwaup stock may need to be considered for use in such an effort.

5. **Dungeness River re-introduction:** Uncertainty remains about the historic status of summer chum in the Dungeness River. This uncertainty was recognized by the PSTRT in their analysis of recovery needs based on biological diversity and spatial structure criteria (Sands et al. 2009; Ken Currens, NWIFC, *personal communications*). To achieve delisting, the PSTRT as well as the participants in the diversity forum described in this document (Section 7.1) did not find that a reintroduction of summer chum into the Dungeness River was necessary. Instead, the conclusion was that it is important that summer chum persist in that river, even at very small numbers, either through straying, persistence of the existing stock, or a reintroduction effort. However, we note that if the Dungeness River is capable of producing large numbers of summer chum, particularly given the protection and restoration work that is currently on-going there, it likely represents the largest

potential of any subpopulation to increase capacity of the SJDF population. Figure 28 suggests that the SJDF population's resistance to both climate change and harvest would be increased with greater capacity.

- Recommendation: We conclude that no specific actions need to be taken in the Dungeness River at this time for the sake of delisting unless it is found that summer chum are no longer present in the river. We recommend, however, that consideration be given to conducting an experimental re-introduction effort to expedite learning about factors affecting performance of the species in that geographic area. Knowledge gained from such an effort could be valuable for future planning to help address climate change factors or to add capacity to the SJDF population to strengthen it or to increase harvest potential. The co-managers may want to consider whether demonstrable improvements have been made in the habitat conditions of the lower river in determining if an experimental reintroduction should go forward.

6. **Status of threats to the ESU:** To achieve delisting of the ESU requires that the population viability criteria be met as well as certainty that the threats that originally caused the need for listing have been adequately abated. Threats are defined as those human activities or naturally induced actions that produce the limiting factors that adversely affect salmon performance (e.g. riparian vegetation removal). Assessing the status of threats also requires that the linkages between threats and changes in the salmon population performance are sufficiently understood (NMFS 2007). NMFS recognized that some perceived threats may become insignificant because of changes in the natural environment or changes in the way threats affect the life cycle of the species.

   - Recommendation: HCCC should facilitate a review of the threats and limiting factors to the ESU to determine their status and need for additional attention. Chapter 6 of the recovery plan (HCCC 2005) describes the threats and limiting factors as understood at the time the plan was prepared.

7. **Climate change planning:** Climate change effects relevant to summer chum are expected to gradually increase over time. Though there is uncertainty about whether any of these effects may benefit summer chum performance (e.g., by benefiting food webs within Puget Sound relevant to summer chum), the general consensus at the climate forum was that performance will likely be adversely affected. Discussion on this topic in Weinheimer et al. (2017) supported this view. Lestelle et al. (2014) recommended that some allowance be incorporated into the viability thresholds to account for the likelihood that some environmental factors will increase in their variability, thereby increasing variability in summer chum production and productivity, and effectively raising viability curves for a given level of risk. While the participants in the forum generally concluded that overall population performance will likely decline as a result of climate change, there was no general agreement that this would be accompanied by increased population variability

   - Recommendation: We recommend that climate change be addressed through adaptive management based on results of monitoring and the accrual of greater knowledge about climate-related factors affecting summer chum.
   - Recommendation: We recommend that the strategy to mitigate or offset future climate change effects be to continue restoration and protection work once delisting occurs. The strategy would embrace a principle of smart investing for the future—start now and don't wait until the need is too large to address. In general, the key to enabling summer chum to cope

with adverse climate change effects will be to improve and protect characteristics of habitat quality within freshwater, river mouth estuarine, and nearshore habitats. As noted above, however, there may be a large untapped potential to increase the quantity of available habitat (capacity) for the SJDF population if a reintroduction effort in the Dungeness River were to be successful. We also note that population capacity of the Hood Canal population would be increased if one or more subpopulations can be re-established on the Kitsap Peninsula.

8. **Need for continuing restoration efforts:** Until delisting is assured, there remains a need to aggressively move forward with restoration and protection actions within the relevant geographic areas that support the ESU. One test of real progress made in habitat restoration in the region will be how well the populations perform during a downturn in marine survival associated with the warm phase of the PDO.

    - Recommendation: We recommend that habitat restoration and protection actions continue to be aggressively advanced within the geographic boundaries of the ESU. The actions should be done strategically with consideration for both populations in the ESU. The actions 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 Union for the Hood Canal population. We consider the core subpopulation in the SJDF population to be Salmon-Snow Creek. 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.

9. **Monitoring, evaluation, and research:** Without monitoring, there is no scientifically valid way of assessing the status of the populations, their responses to PDO factors and climate change, whether delisting is appropriate, and whether benefits from restoration work are occurring. Some level of systematic monitoring and evaluation is essential to address each of these matters.

    In addition, important questions exist about the apparent effect of PDO-related factors on the performance of the ESU. If these effects are real, as illustrated in the patterns of performance presented in this report, then what mechanisms are producing these patterns? We hypothesize that there is some kind of transfer of PDO-related factors from the ocean to the inner recesses of Puget Sound, such as into Hood Canal, that affect localized food webs of importance to young summer chum fry (see Sections 6.1.1 and 6.1.2). Answering such questions has an important bearing on managing the ESU's two populations. The question also has bearing on understanding performance patterns for Puget Sound salmon runs in general.

    There is currently much shared interest from entities in British Columbia and the United States to determine the primary factors affecting the survival of juvenile salmon and steelhead in the Salish Sea. Marine survivals in general have declined significantly for many populations that originate in rivers that drain to this area (e.g., Zimmerman et al. 2015; Malick and Cox 2016). Only three of the many chum and pink populations analyzed by Malick and Cox (2016) that originate in Washington and British Columbia demonstrate increasing productivity in recent years—all others show marked long-term declines in productivity. Two of those three populations (all fall or winter chum) originate in South Puget Sound and the third one in Hood Canal. Summer chum from the Hood Canal ESU were not included in the analysis. Due to the high interest to better understand the causes of declining productivity for many Salish Sea salmon populations, a large international, multifaceted research project, the Salish Sea Marine Survival Project, is underway to determine the important survival factors (http://marinesurvivalproject.com/the-project/). We find that the questions raised in this report, particularly related to the performance of the Hood Canal ESU, are highly relevant to the research objectives of that project.

It is important to note regarding the Salish Sea Project that annual sampling aimed at understanding spatial-temporal patterns of plankton composition and abundance within Hood Canal generally begins in April according to researchers who participated in the HCCC climate forum. Richard Beamish, retired Fisheries and Oceans Canada, who participated in the climate forum, recommended that sampling to understand summer chum performance patterns needs to occur throughout the winter and spring and should focus attention on Dabob and Quilcene bays to understand why Big-Little Quilcene summer chum are so productive. Another climate forum participant, Brian Beckman of NOAA Fisheries, suggested that otoliths that have been collected from different subpopulation spawners can be analyzed to compare early marine life growth patterns among subpopulations and between warm and cool PDO phases—this would be helpful since plankton sampling was not rigorously done in the past.

- Recommendation: It is vital that the on-going monitoring and evaluation of the SJDF and Hood Canal summer chum populations being done by WDFW and tribal co-managers be continued and fully funded. The co-managers annually collect and evaluate a variety of information. Basic assessments include spawner escapements, harvest, total run sizes, age composition, and natural-origin vs. supplementation-origin composition (when and where reintroductions or supplementation occur)—these estimates are made for all of the spawning aggregations. Reintroduction/supplementation has recently occurred in Tahuya River and Lilliwaup Creek so a need continues in these areas to assess whether returning adults are natural-origin or supplementation-origin fish. Future reintroduction efforts may be considered for Dewatto River and Big Beef Creek as discussed earlier in this document. In addition, estimates of outmigrant fry and egg to fry survival are made in Salmon Creek and Duckabush River—these efforts should be continued. Where these fry assessments are made, it is essential that stock composition be determined with genetic analysis, i.e., to assess percentages of summer- or fall-run chum in the fry outmigration (e.g., see Weinheimer 2018). We further recommend that fry outmigration be assessed in Union River to provide the same level of monitoring being done in Salmon Creek and Duckabush River. These three spawning aggregations belong to different ecological diversity groups (Figure 5) and experience very different hydrological conditions within the natal streams and different estuarine characteristics near the natal streams. Monitoring fry output, egg to fry survival, and fry to adult marine survival for these three distinctive ecological groups will provide information critical for improving understanding about effects of climate change and PDO-related factors. This information would also be directly applicable in run size forecasting, thereby improving overall fisheries management for the ESU.
- Recommendation: We recommend that communications between HCCC and the Salish Sea Project occur in order to promote an expansion of the investigation to address questions and hypotheses raised in this report. This will need to be done very soon to determine if it is possible to expand sampling starting winter 2018-19. We also recommend that HCCC or one of its partners move forward with a study proposal to analyze summer chum otoliths from relevant sites and years aimed at assessing spatial-temporal patterns in the growth of summer chum fry. The otolith study would be particularly helpful to understand whether performance patterns

observed over time for the subpopulations are correlated with early marine growth patterns seen in the otoliths.

10. **<u>Accounting for PDO-related effects – a decision pathway</u>:** As noted in Section 6.1.2.3, an opportunity has availed itself to advance our understanding about the effect of the PDO and whether a lag occurs in the effect being transferred into Hood Canal. The PDO index turned strongly positive (warm phase) in January 2014 and has remained in the positive state to the present time (through 2017). This means that if an effect of the PDO continues to occur on the summer chum populations, then fry produced from brood year 2013 (fry year 2014) could have experienced a downturn in survival, suggesting that fewer adults would have returned to spawn in 2016 and 2017 (as age-3 and age-4 spawners). Adverse effects of the PDO phase shift would be expected to continue for some period into the future.

    Based on our analysis (Table 8), we projected that we would see a downturn in returning SJDF adult summer chum first in 2016, then continuing into 2017 and at least into the next several years beyond that. And based on our hypothesis of a lagged effect for the Hood Canal population, we projected the first downturn in returning adults in 2018 and then continuing at least into the next several years. Data presented in this report show that no downturn occurred for the SJDF population in 2016 for age-3 adults—large numbers returned to all of the natal streams. However, preliminary results for 2017 (data not presented in this report) showed a major downturn in the SJDF spawner abundance in 2017 (both age-3 and age-4 fish) while the Hood Canal population continued to surge higher. These results for 2017 are consistent with our projections.

    Other factors besides PDO-related effects are also operative, which include those occurring in freshwater such as flow-related effects. In the two streams being monitored for outmigrant fry—Salmon Creek in the SJDF region and Duckabush River in the Hood Canal region—the lowest abundance of fry migrants within the data series occurred with the brood year 2014 and 2015 cohorts (Figures 30 and 32). We expect that the downturn in the returns to SJDF streams will be particularly severe in 2018 due to the combination of PDO-related effects and low abundance of fry migrants. Similarly, we project a downturn in the abundance of Hood Canal spawners in 2018 due to a combination of these factors. It bears noting that the abundances of fry outmigrants from brood year 2016 (Figures 30 and 32) and brood year 2017 (Mark Downen, WDFW, personal communications) were much higher than those in the two prior brood years.

    The schedule of returning spawners at age-3 and age -4 over the next six or seven return years, i.e., 2017 to 2022 or 2023 (Table 7), provides a basis for assessing progress toward recovery during a warm PDO phase. All of these return years will include adult summer chum that should have experienced conditions as emigrant fry within Hood Canal and the SJDF affected by the warm PDO phase. If marine survival drops sharply, which could occur based on the phase shift in the PDO index in 2014, the question in focus will become: How low will the abundance of returning runs be and will all subpopulations respond the same? The answer to these questions will provide the means for assessing how well the populations perform with the combination of reduced marine survival and the benefits of restoration actions that have occurred to that time.

    These circumstances give a rational basis for designing a decision pathway by which a delisting decision can be made (Figure 40). We project that sufficient information should be available no later than the end of 2022 or 2023 to make a highly informed decision about delisting. The decision pathway is explained in the following based on the record of run sizes for the years 2017 to 2022. Two pathways will be possible based on how the ESU performs during those years:

1. Path 1, shown as the left branch in Figure 40, would be evident if the returning run sizes for these years remain strong.
    a. This path would likely signify that a PDO-related effect as described in this report either did not exist, i.e., the correlations and patterns of population performance were coincidental, or that a PDO effect had waned. If this occurs, on-going research by NMFS should be informative to help assess what was happening with the PDO and other large-scale climate system phenomena at that time.
    b. Provided that population performance remains higher than viability thresholds, it should be evident that the populations are at low risk of extinction (<5% risk).
    c. The co-managers and NMFS would review all information available on population performance, including spatial structure (e.g., status of a Dewatto spawning aggregation), broad sense goals, status of threats (including climate change), and other concerns and determine whether delisting is appropriate.
2. Path 2, the right branch in Figure 40, would be apparent if there is a significant downturn in run sizes compared to recent years.
    a. This path would strengthen the evidence for a PDO-related effect as described in this report. If this occurs, on-going research by NMFS should be informative to help assess what was happening with the PDO and other large-scale climate system phenomena at that time.
    b. The key question then comes into focus: How weak or vulnerable have the populations become during the period? Three paths become possible.
        i. If the run sizes clearly drop below viability thresholds, then ESA-listing would likely remain in place (left branch).
        ii. If the run sizes drop slightly below or remain slightly above viability thresholds, then a review of the circumstances and other conditions may be necessary but the ESA-listing would likely still be necessary (middle branch).
        iii. Despite the downturn in abundances, if run sizes remain clearly above viability thresholds, then the co-managers and NMFS would review all information available on population performance, including spatial structure (e.g., status of a Dewatto spawning aggregation), broad sense goals, status of threats (including climate change), and other concerns and determine whether delisting is appropriate (right branch).

We recognize that events over the next six years may not turn out as cleanly as portrayed in the figure. One possibility is that one population may demonstrate it is on one side of the decision pathway while the other population may demonstrate it is on the opposite side. Since both the SJDF and HC populations need to be viable for the ESU to be viable, delisting would not occur in this case.

- <u>Recommendation</u>: We recommend that the decision pathway in Figure 40 be used to guide delisting considerations over the next six years.

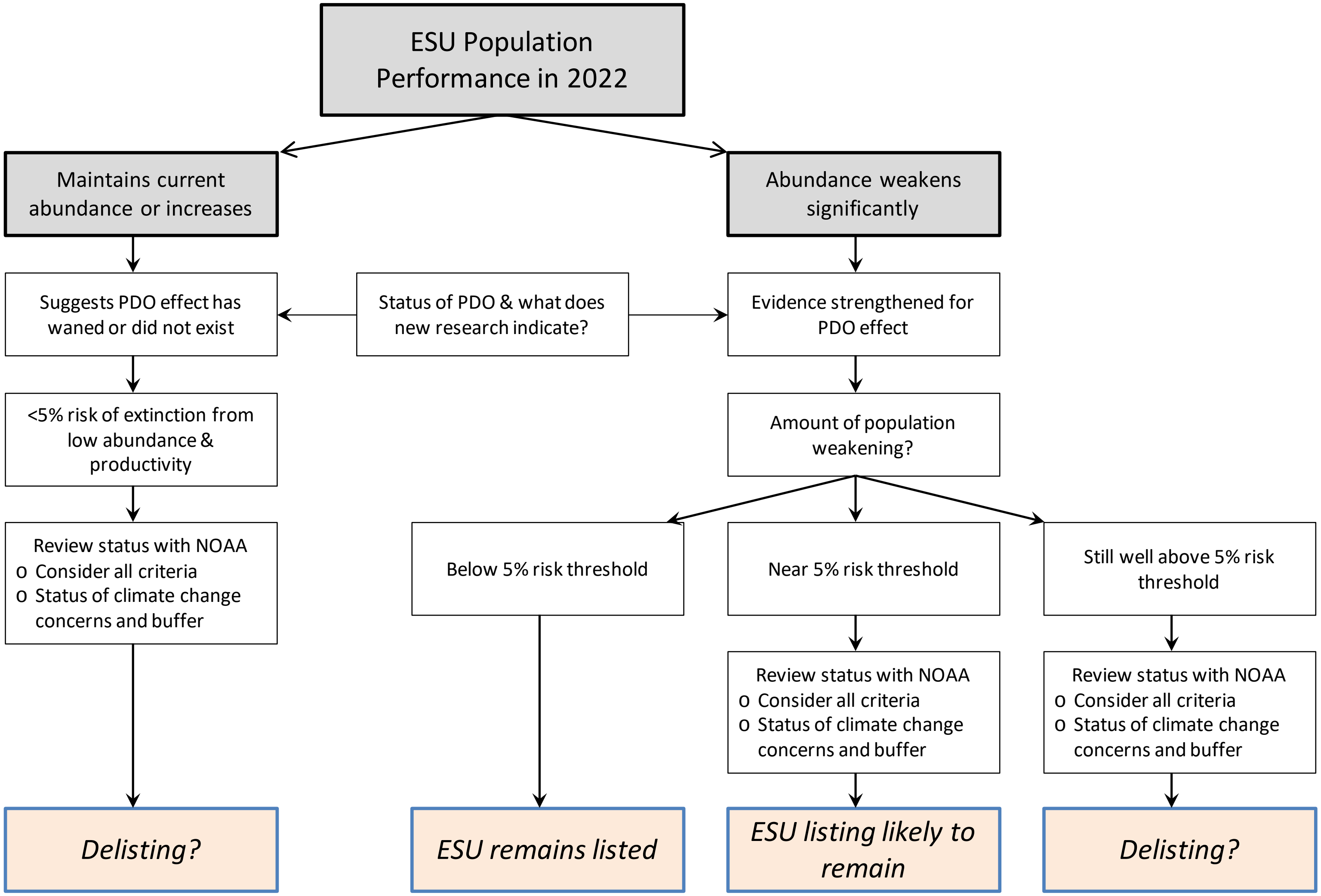

Figure 40. Decision pathway for delisting considerations for the Hood Canal summer chum ESU in 2022.

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Appendix A – Participants in Climate Forum

HOOD CANAL COORDINATING COUNCIL
Hood Canal Summer Chum Climate Forum
March 9, 2017
Kitsap Conference Center, Bremerton, WA

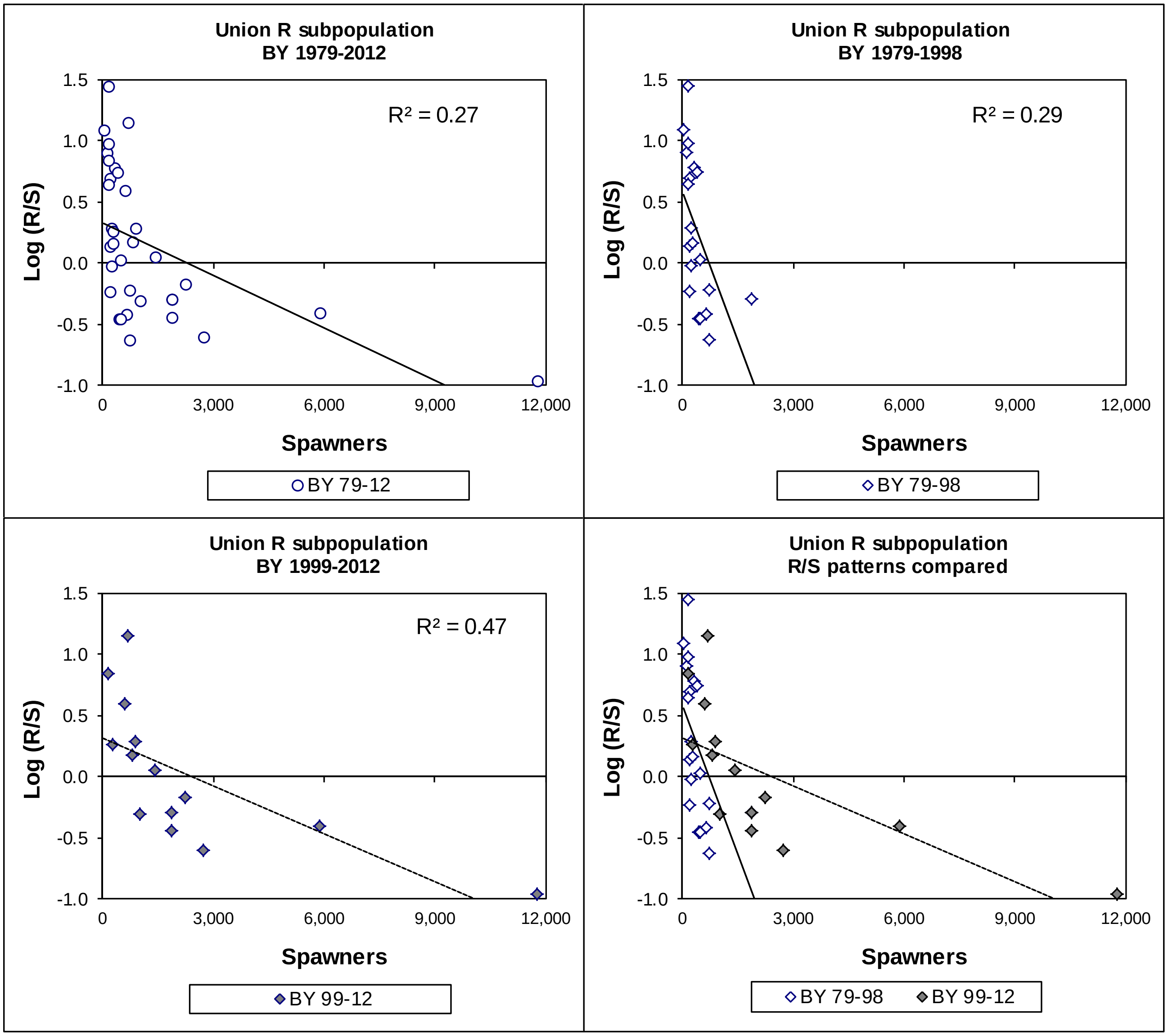
Invited scientists who participated in the climate forum:

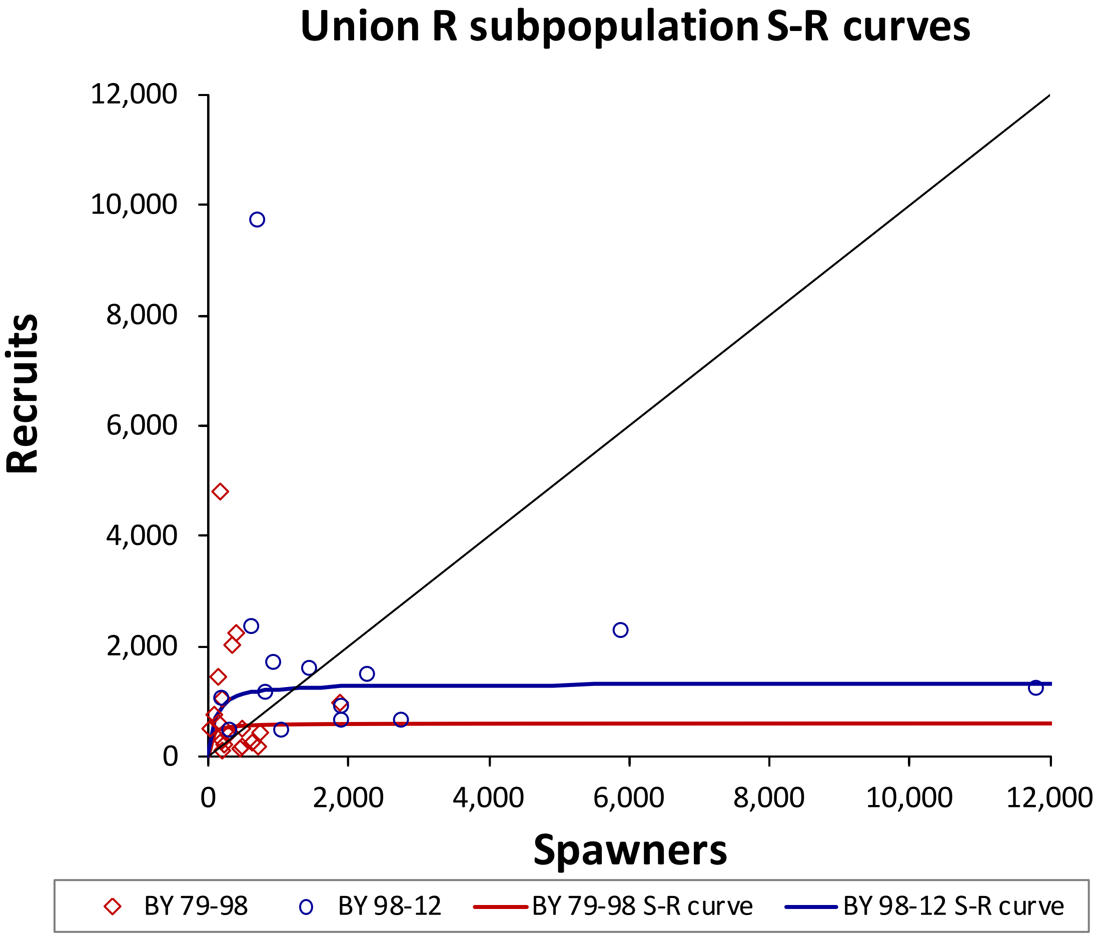
- Anderson, Joe, Washington Department of Fish and Wildlife
- Beamish, Richard, Department of Fisheries and Oceans Canada (retired)
- Beauchamp, Dave, U.S. Geological Services
- Beckman, Brian, National Marine Fisheries Service (NOAA)
- Bond, Nick, Joint Institute for the Study of the Atmosphere and Ocean, University of Washington
- Crozier, Lisa, National Marine Fisheries Service (NOAA)
- Fresh, Kurt, National Marine Fisheries Service (NOAA)
- Mauger, Guillaume, Climate Impacts Group, University of Washington
- McClure, Michelle, National Marine Fisheries Service (NOAA)
- Ramirez, Mary, School of Aquatic and Fishery Sciences, University of Washington

Other attendees who participated in the forum are not listed above.

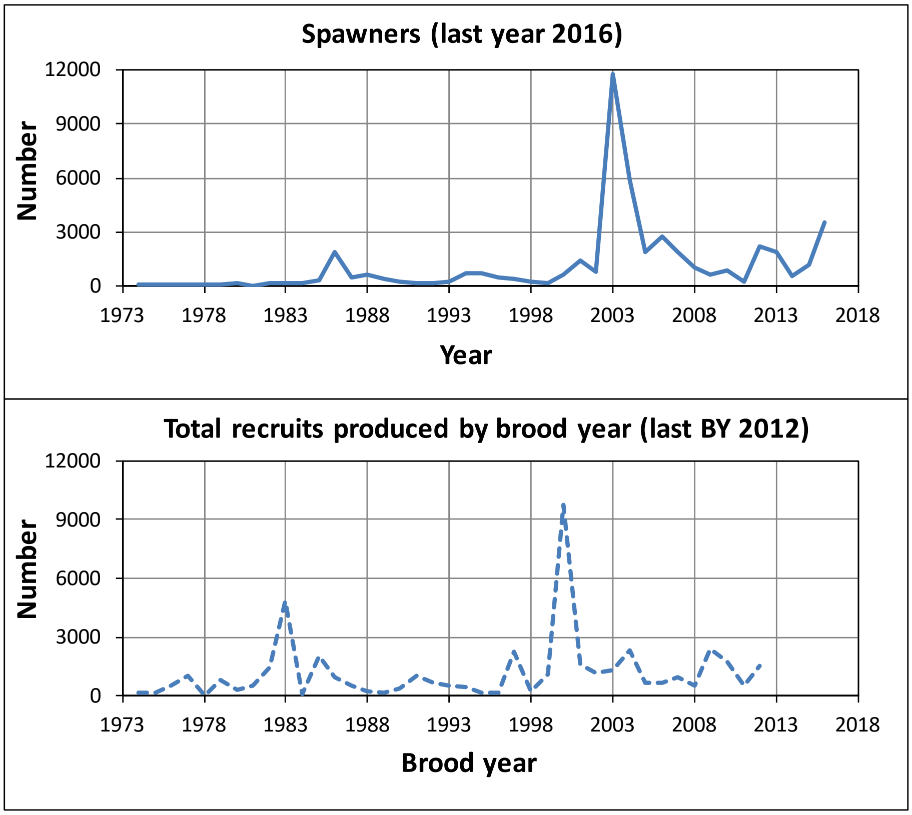
Appendix B – Patterns of Effects of Regime Shifts on Subpopulations

Union River Log (S/R) Plots



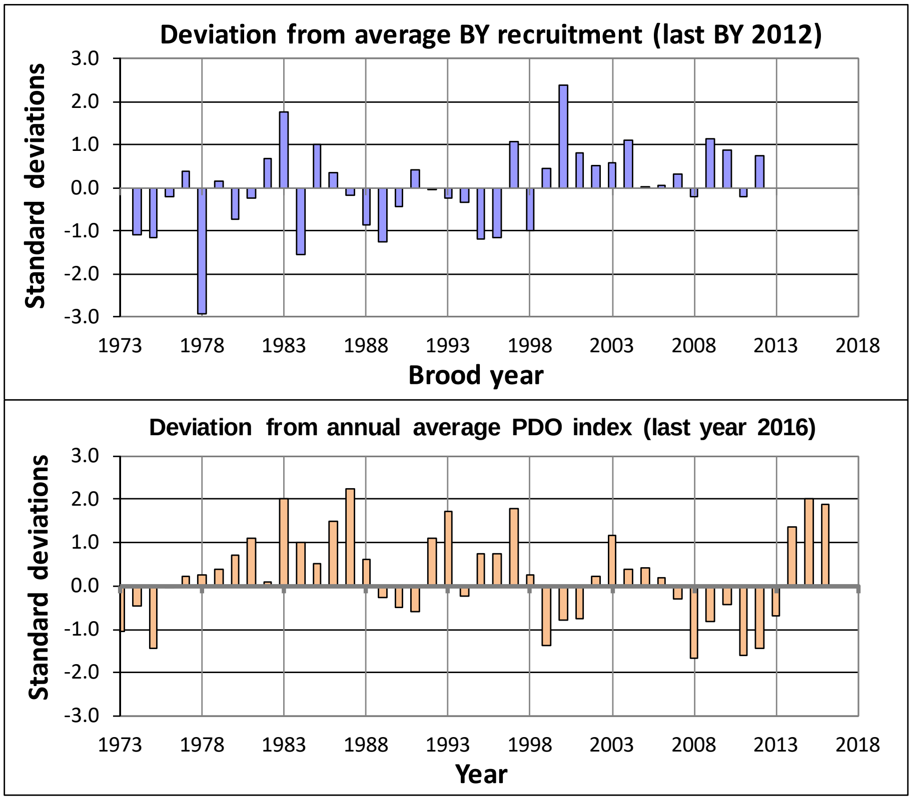

Union R subpopulation S-R curves
Recruits
12,000
10,000
8,000
6,000
4,000
2,000
0
0
2,000
4,000
6,000
8,000
10,000
12,000
Spawners
◇ BY 79-98
○ BY 98-12
BY 79-98 S-R curve
BY 98-12 S-R curve

Union R subpopulation recruit pattern



Spawners (last year 2016)

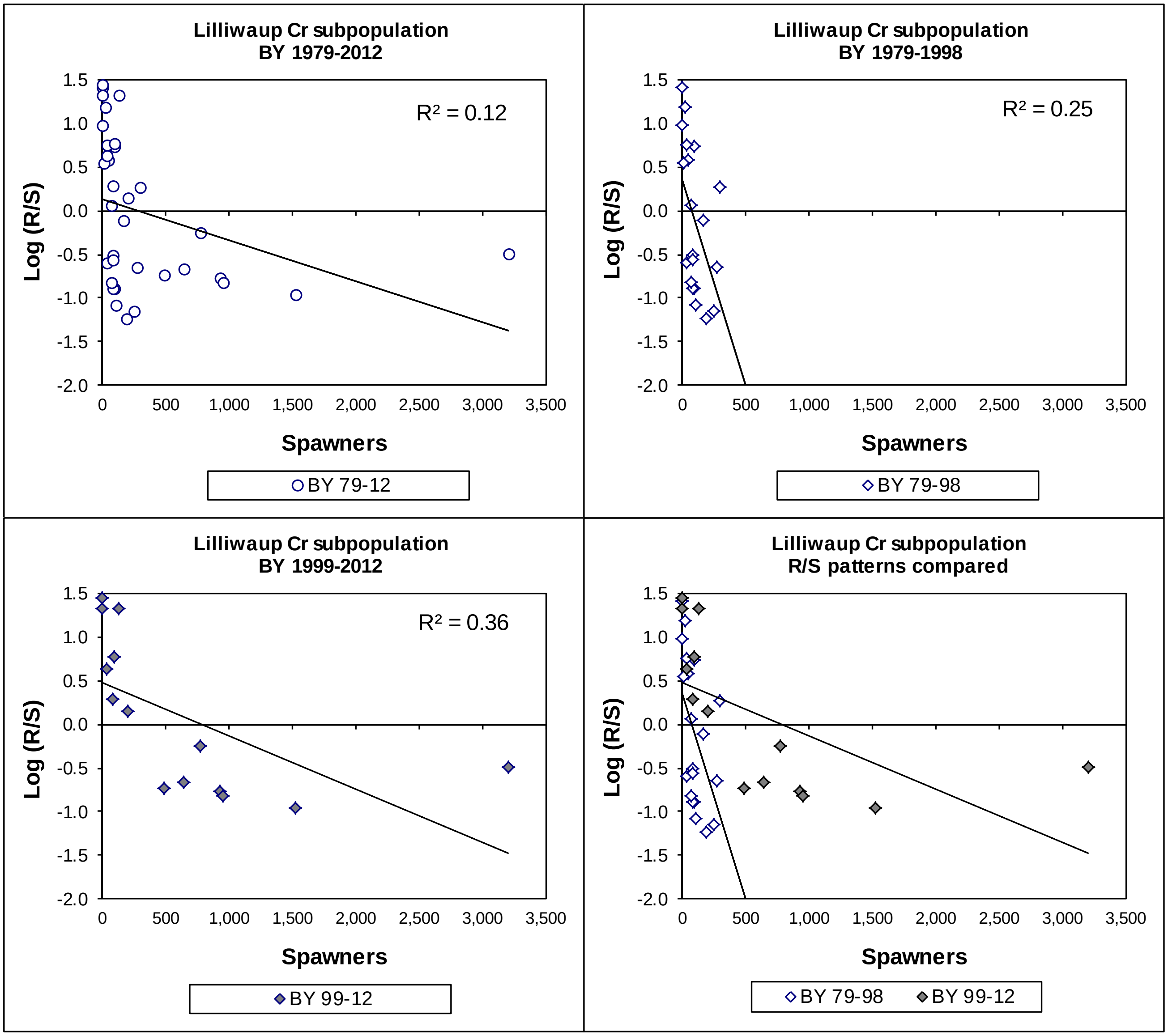
Total recruits produced by brood year (last BY 2012)

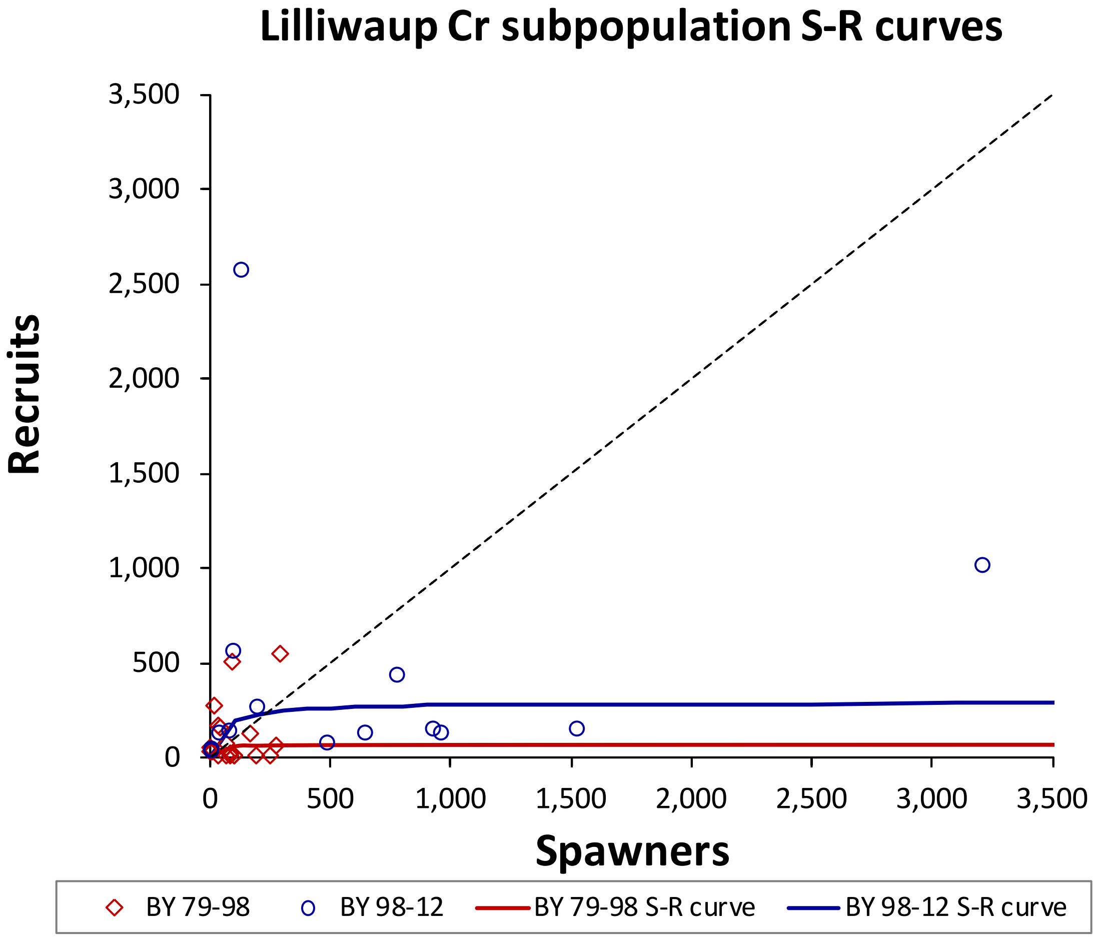


Deviation from average BY recruitment (last BY 2012)

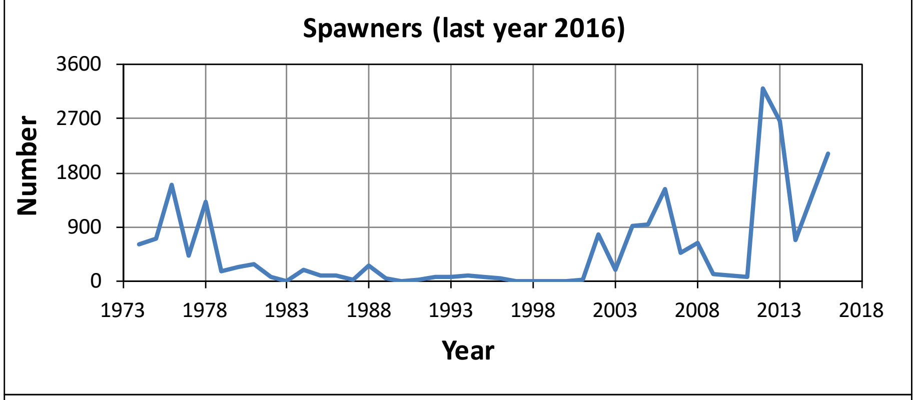
Deviation from annual average PDO index (last year 2016)

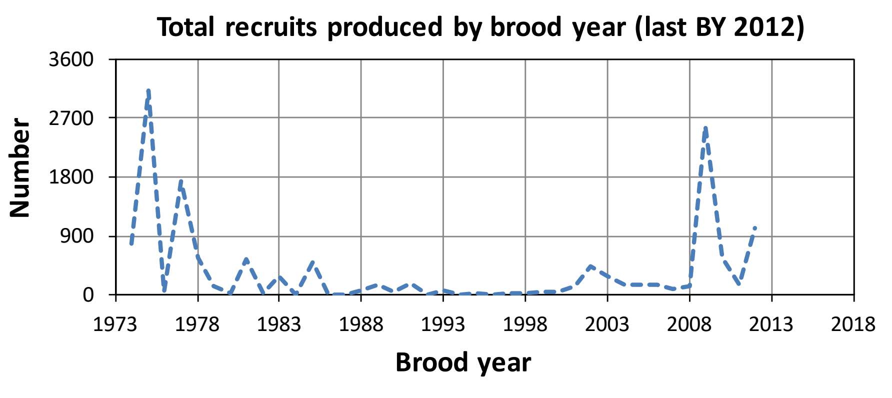
Lilliwaup Creek Log (S/R) Plots

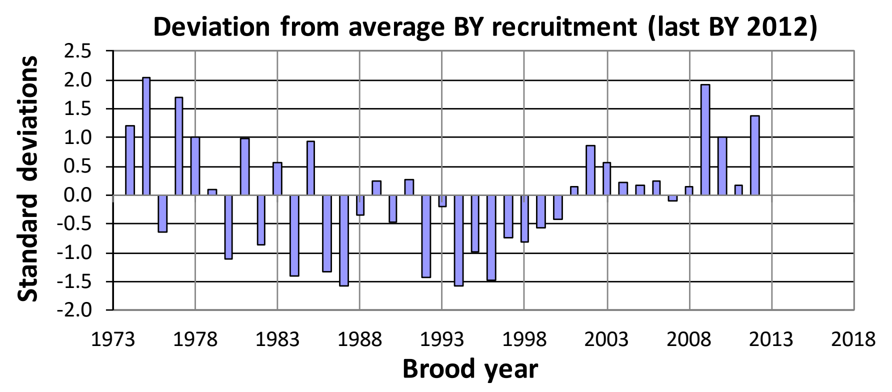


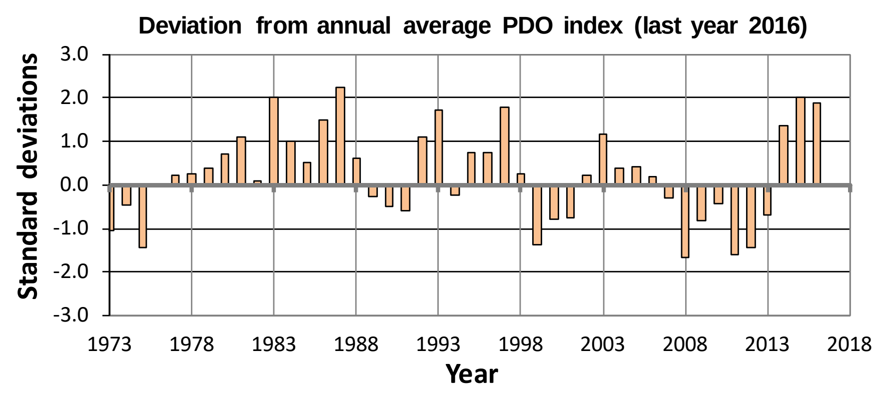

Lilliwaup Cr subpopulation S-R curves
Recruits
3,500
3,000
2,500
2,000
1,500
1,000
500
0
Spawners
0
500
1,000
1,500
2,000
2,500
3,000
3,500
◇ BY 79-98
○ BY 98-12
BY 79-98 S-R curve
BY 98-12 S-R curve

Lilliwaup Cr subpopulation recruit pattern

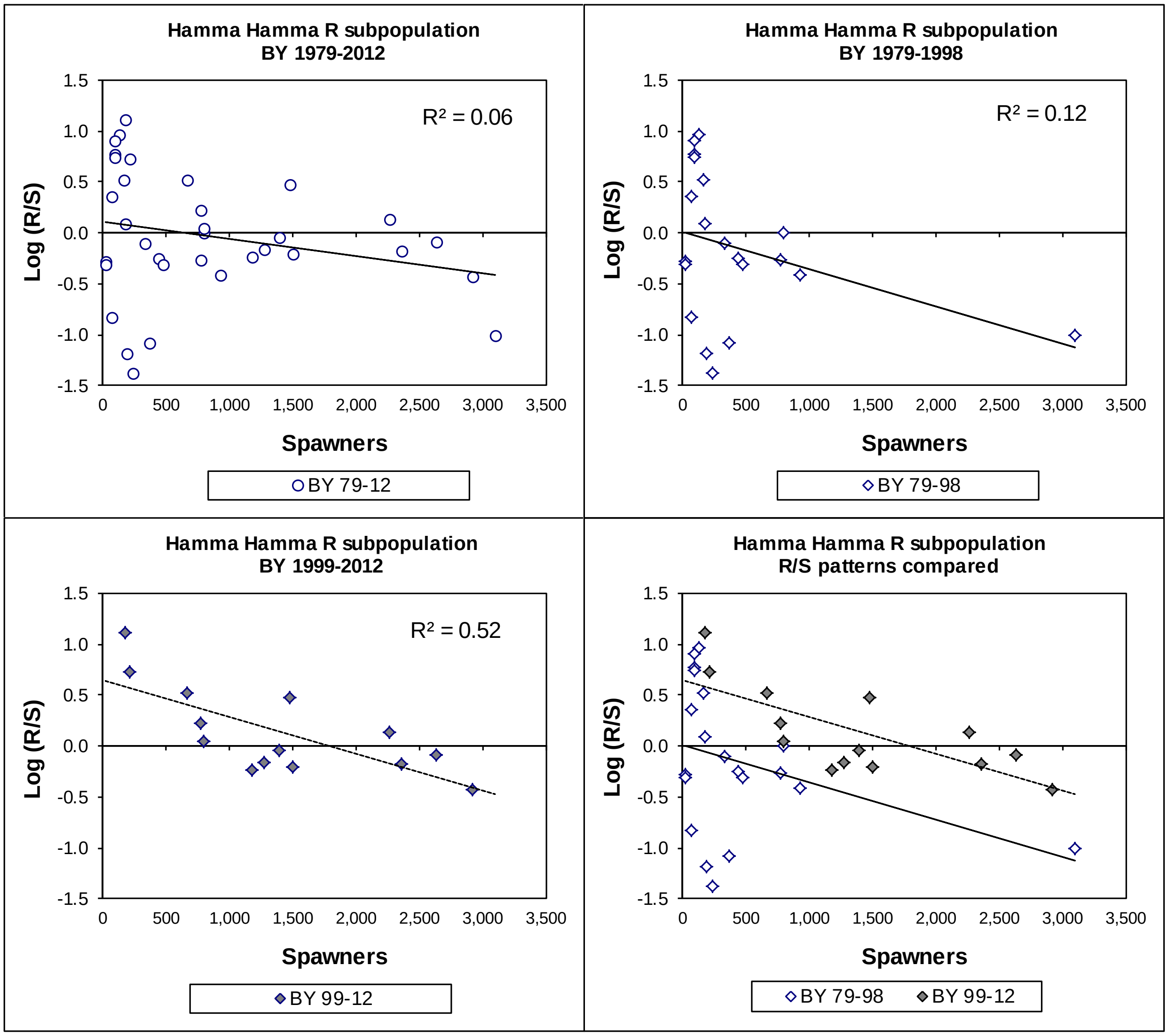


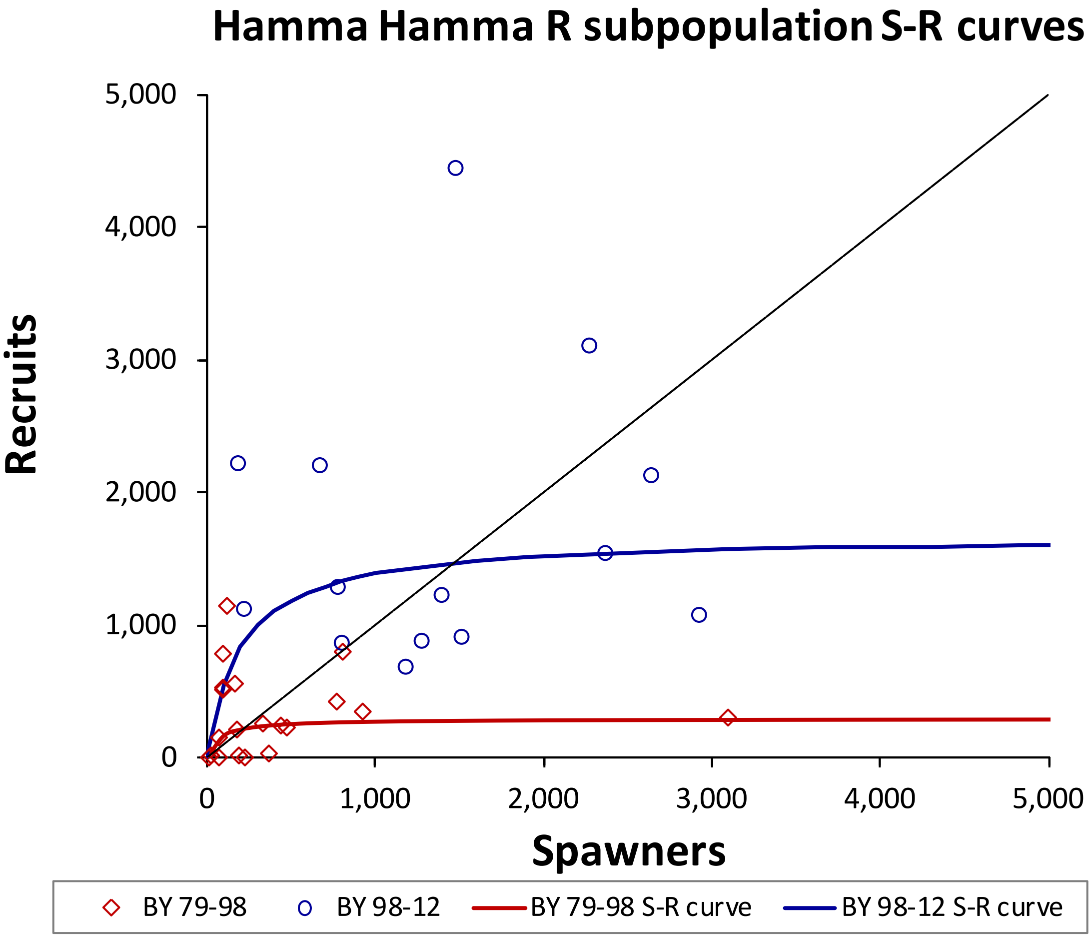




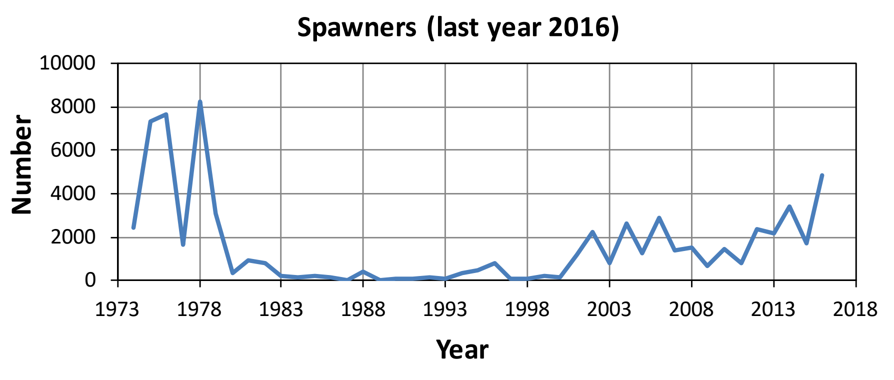


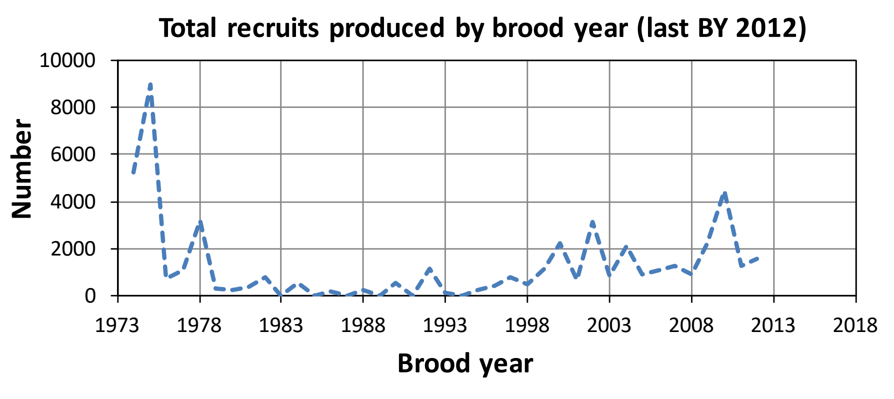
Hamma Hamma River Log (S/R) Plots

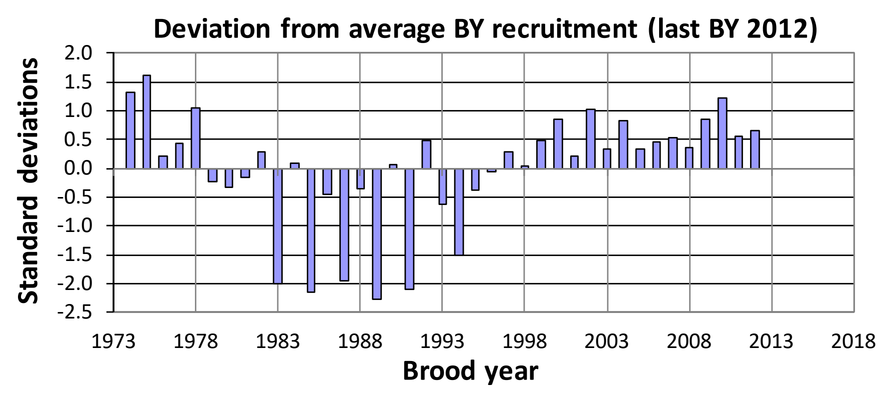


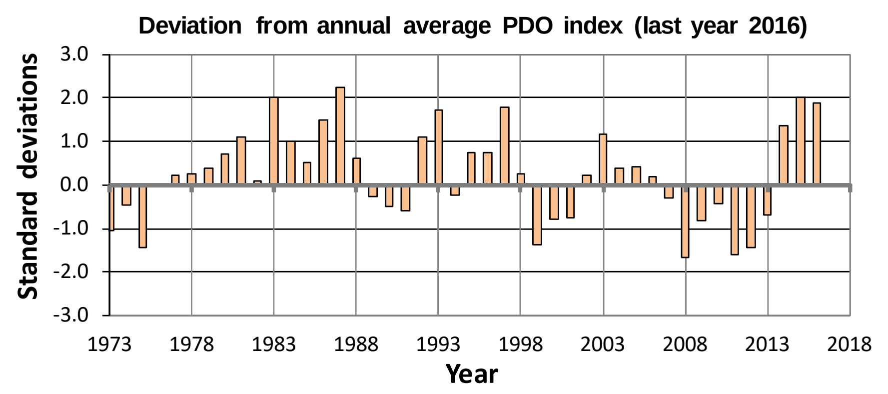

Hamma Hamma R subpopulation S-R curves
Recruits
5,000
4,000
3,000
2,000
1,000
0
0
1,000
2,000
3,000
4,000
5,000
Spawners
◇ BY 79-98
○ BY 98-12
BY 79-98 S-R curve
BY 98-12 S-R curve

Hamma Hamma R subpopulation recruit pattern

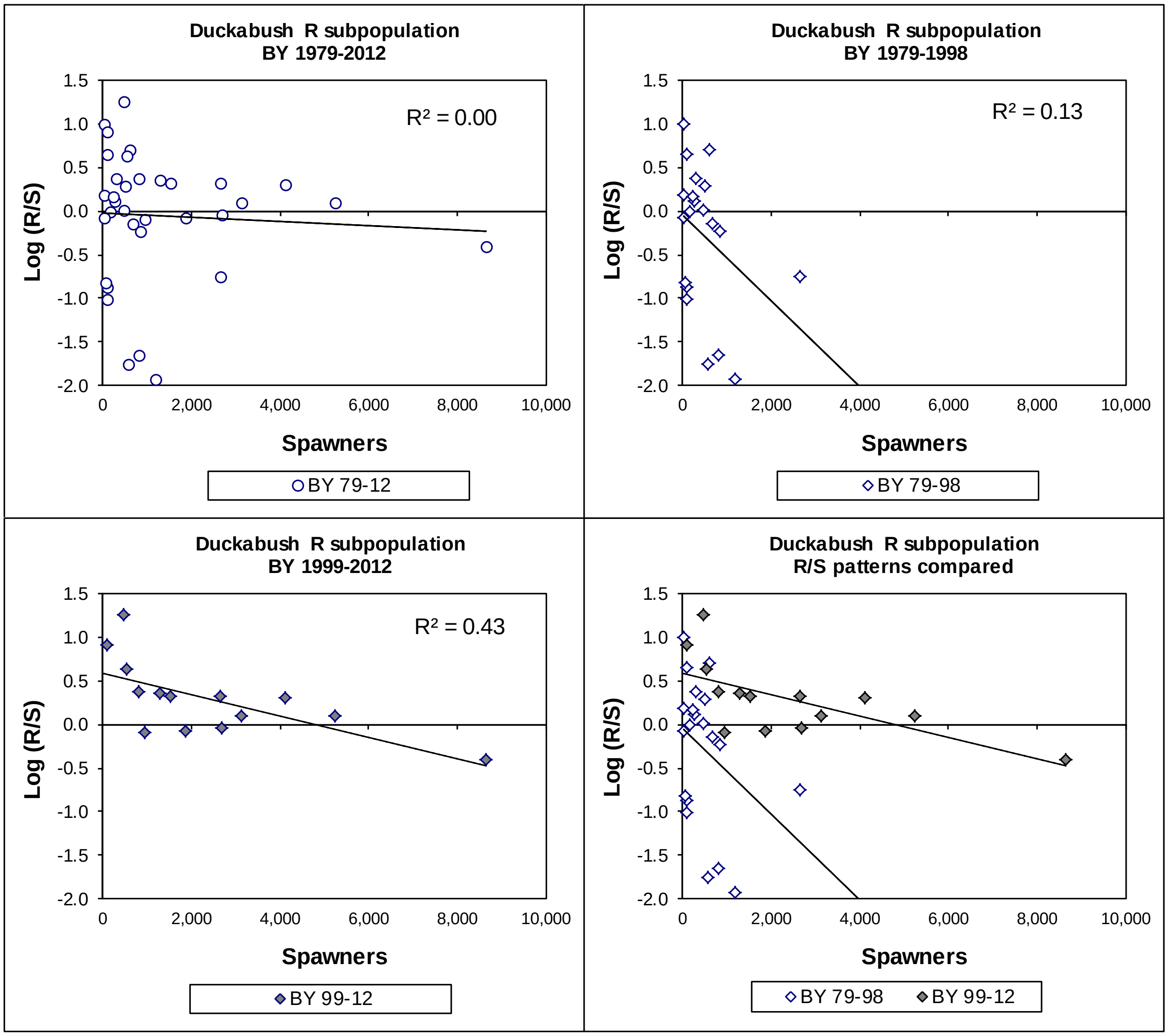

Spawners (last year 2016)


Total recruits produced by brood year (last BY 2012)


Deviation from average BY recruitment (last BY 2012)


Deviation from annual average PDO index (last year 2016)

Duckabush River Log (S/R) Plots

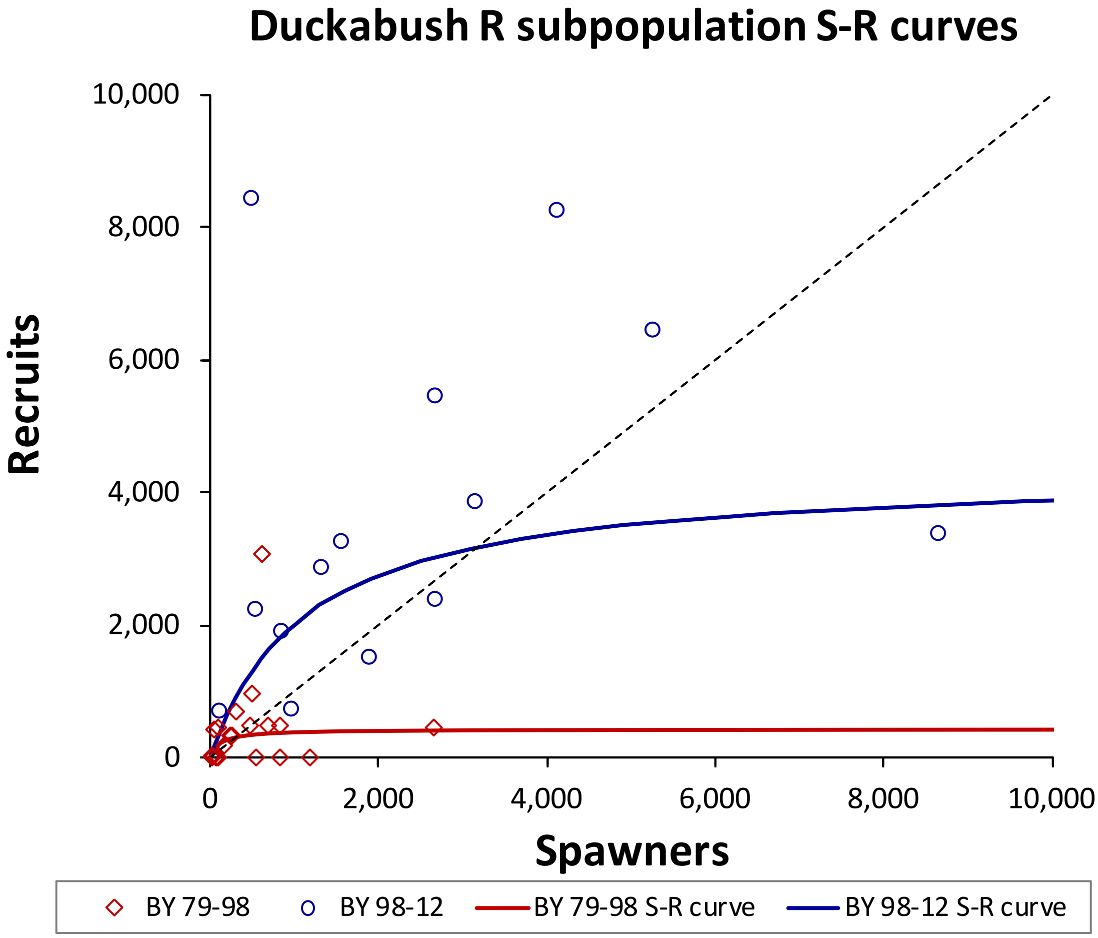


Duckabush R subpopulation BY 1979-2012

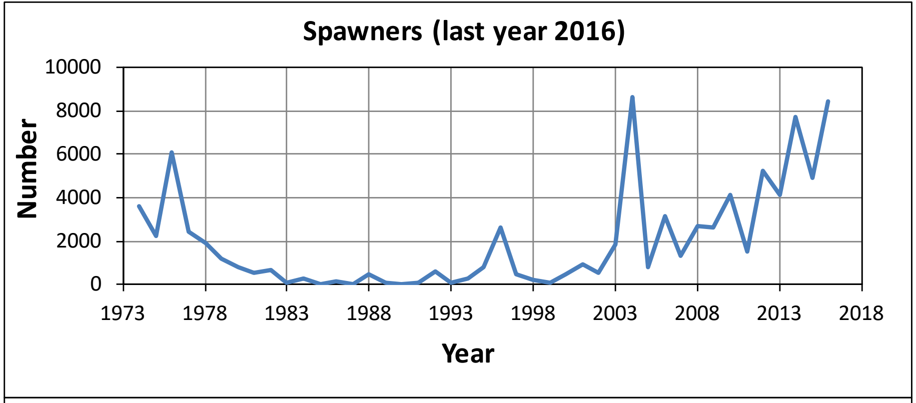
Duckabush R subpopulation BY 1979-1998

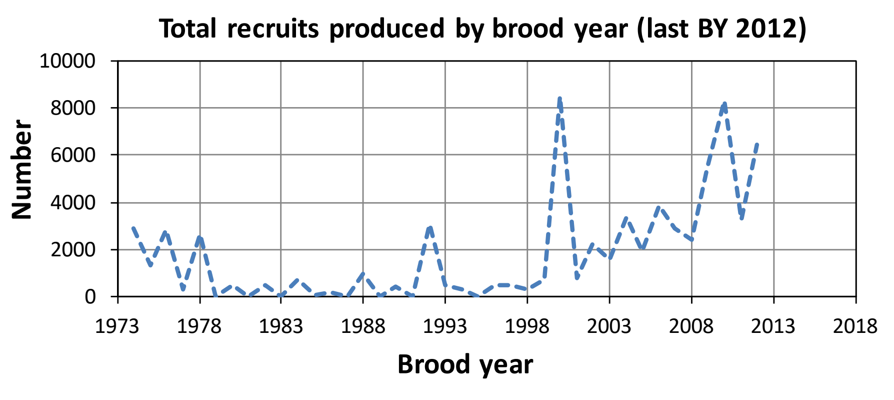
Duckabush R subpopulation BY 1999-2012

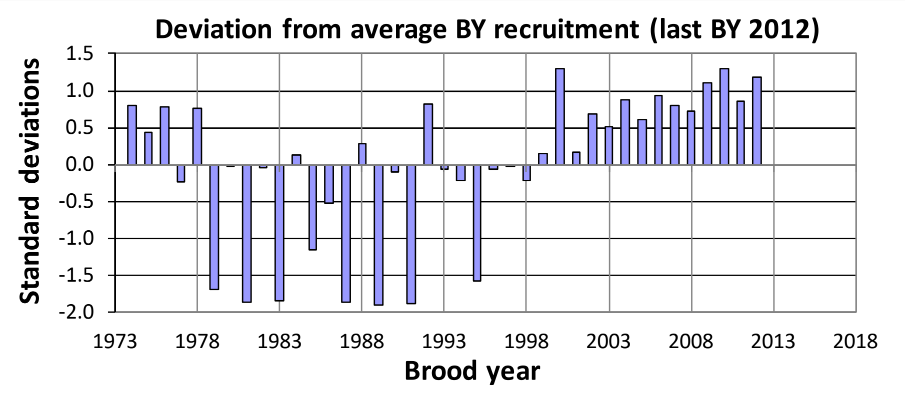
Duckabush R subpopulation R/S patterns compared

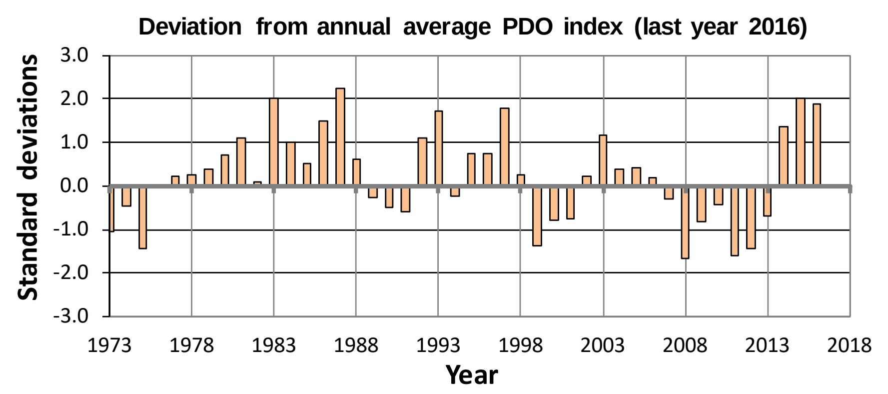

Duckabush R subpopulation S-R curves
Recruits
10,000
8,000
6,000
4,000
2,000
0
0
2,000
4,000
6,000
8,000
10,000
Spawners
◇ BY 79-98
○ BY 98-12
BY 79-98 S-R curve
BY 98-12 S-R curve

Duckabush R subpopulation recruit pattern

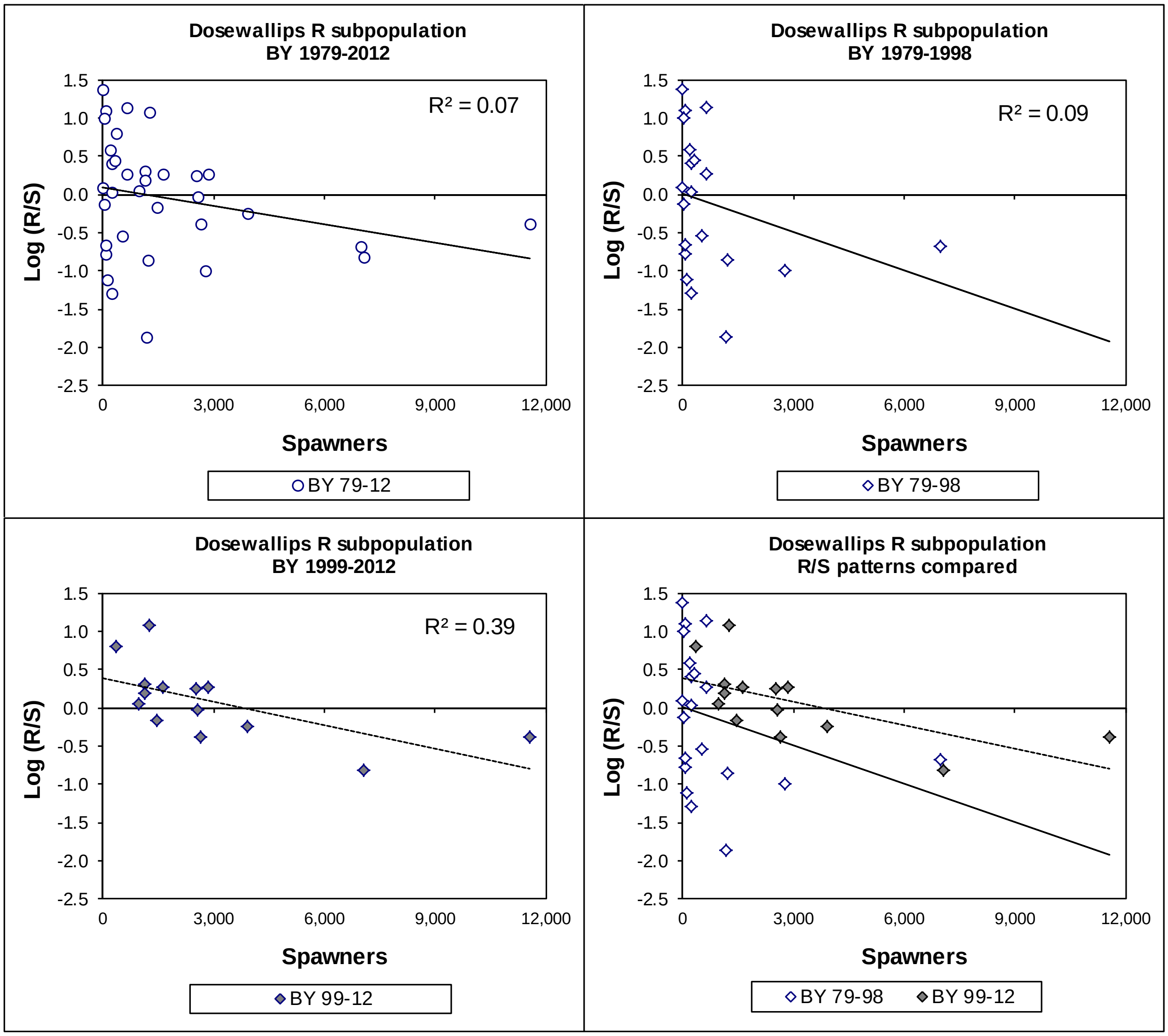

Spawners (last year 2016)


Total recruits produced by brood year (last BY 2012)


Deviation from average BY recruitment (last BY 2012)


Deviation from annual average PDO index (last year 2016)

Dosewallips River Log (S/R) Plots

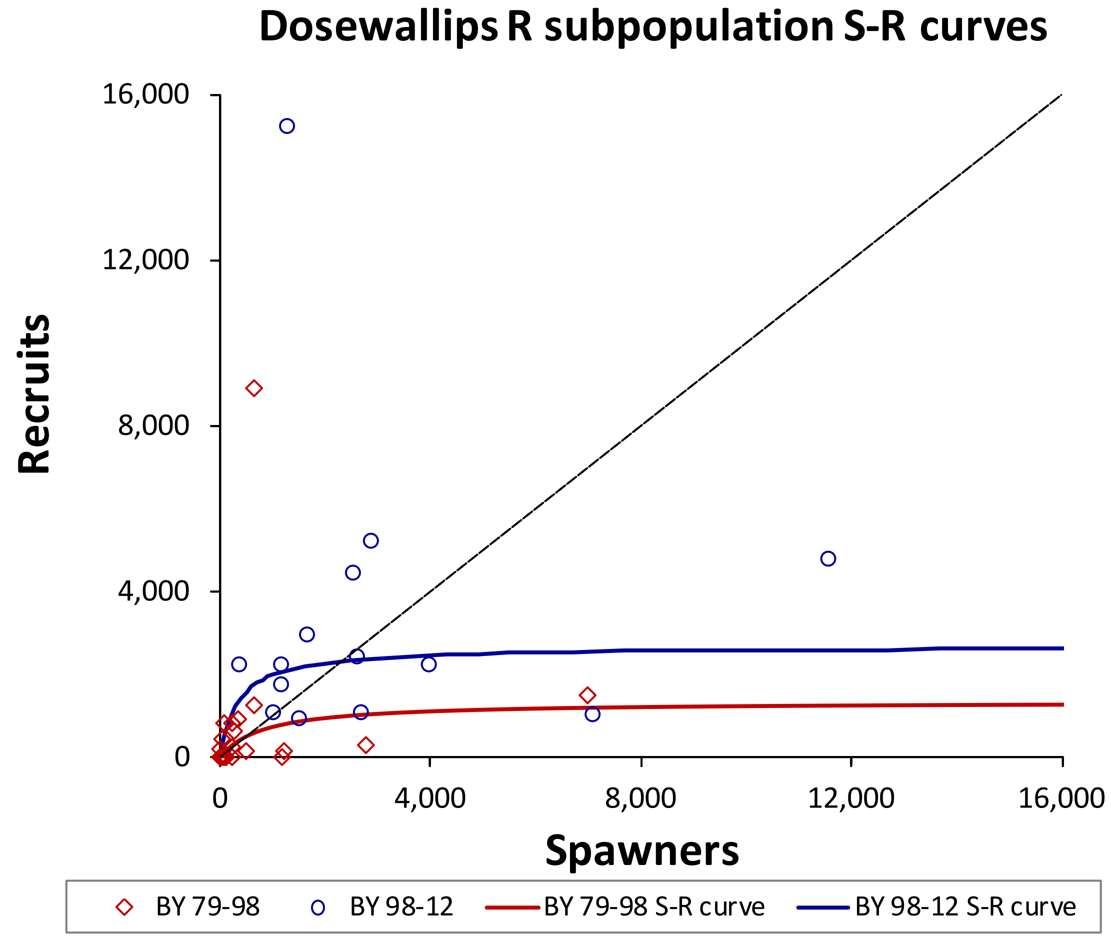


Dosewallips R subpopulation BY 1979-2012

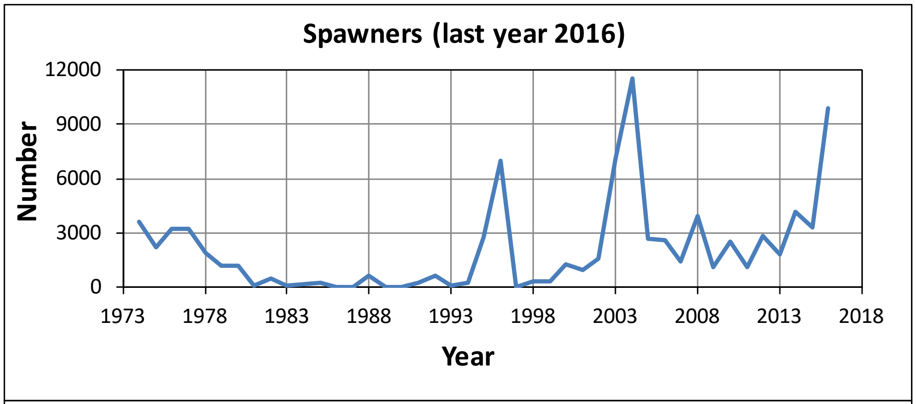
Dosewallips R subpopulation BY 1979-1998

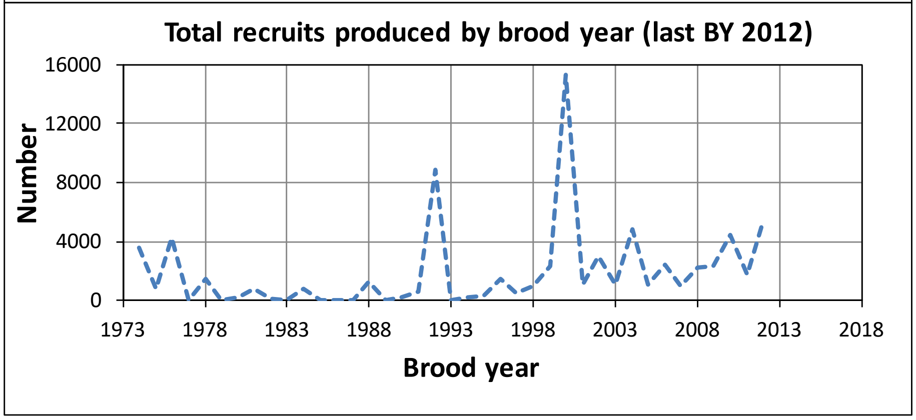
Dosewallips R subpopulation BY 1999-2012

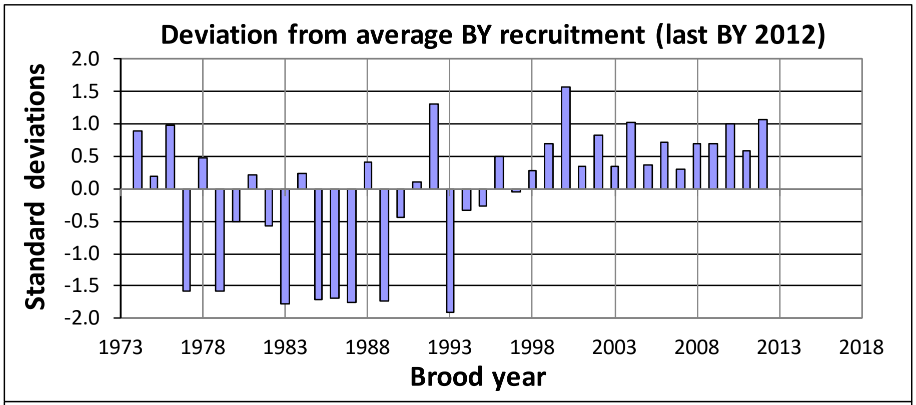
Dosewallips R subpopulation R/S patterns compared

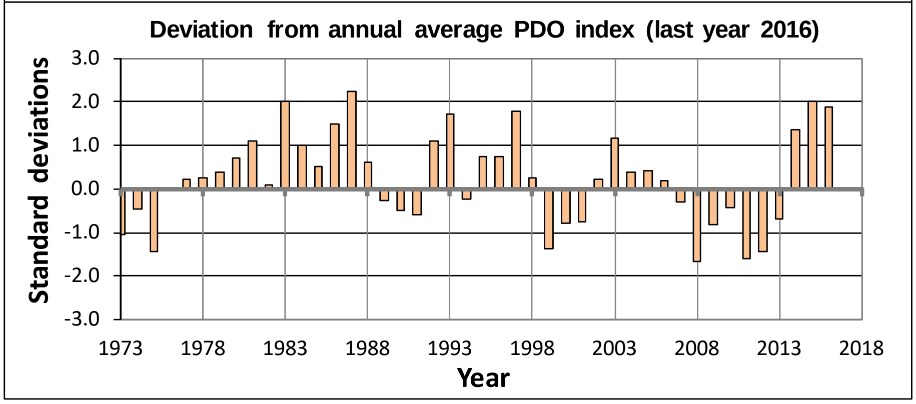

Dosewallips R subpopulation S-R curves
Recruits
16,000
12,000
8,000
4,000
0
0
4,000
8,000
12,000
16,000
Spawners
◇ BY 79-98
○ BY 98-12
BY 79-98 S-R curve
BY 98-12 S-R curve

Dosewallips R subpopulation recruit pattern

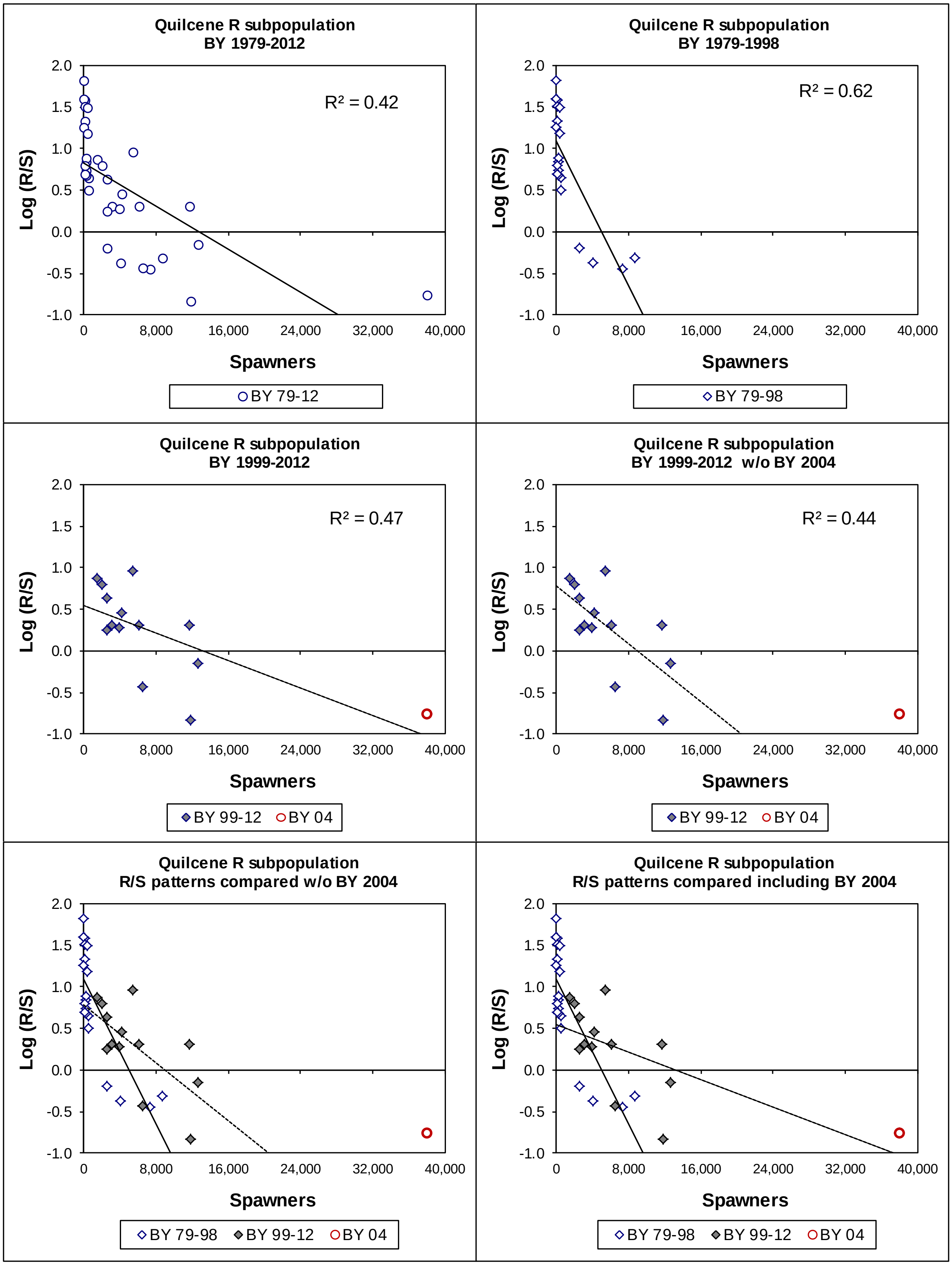


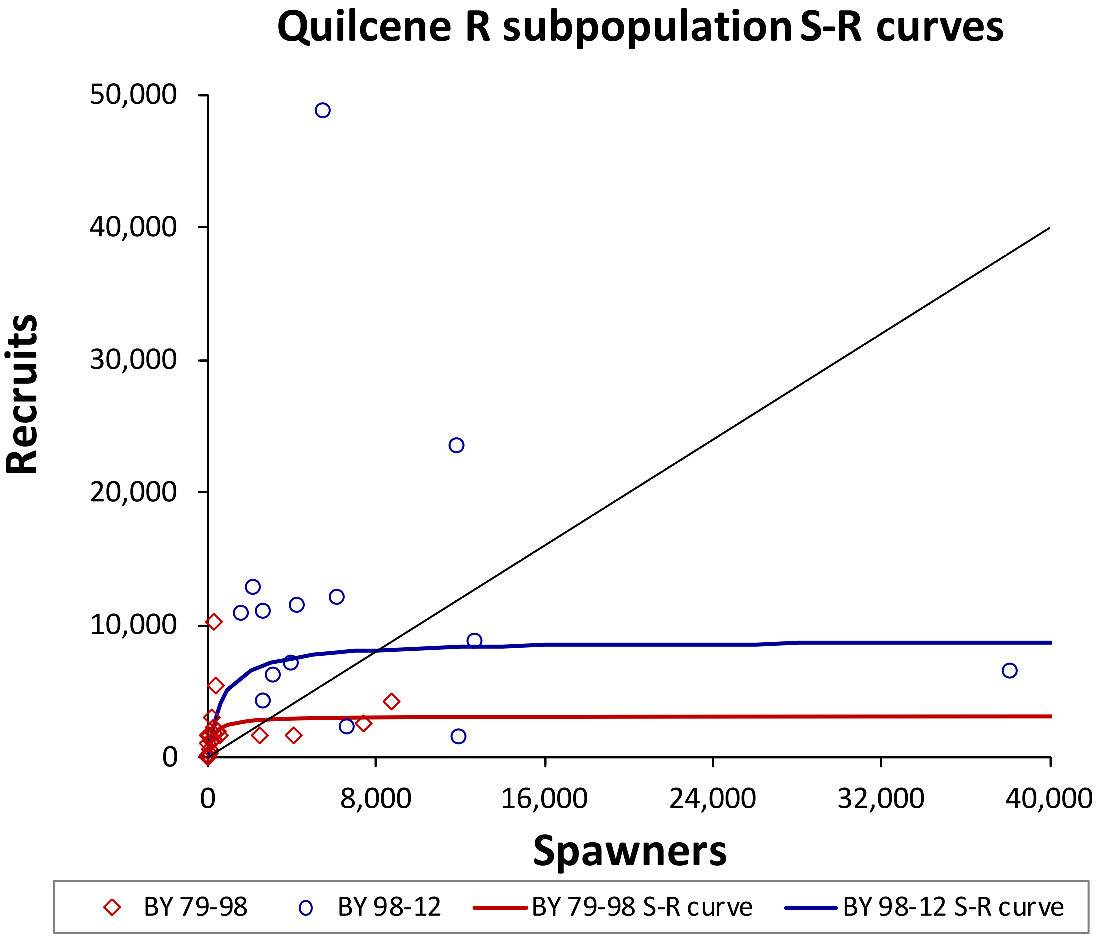




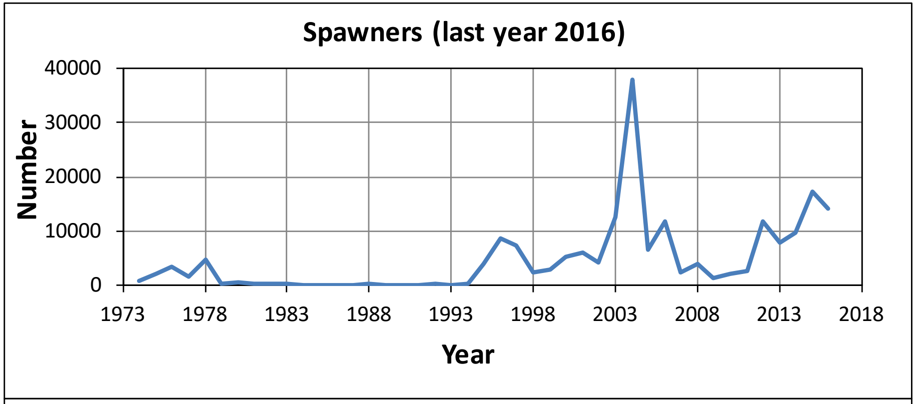


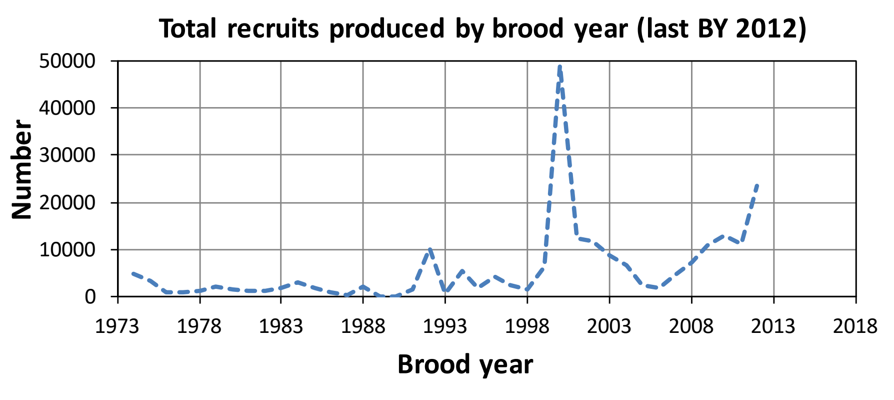
Big and Little Quilcene River Log (S/R) Plots

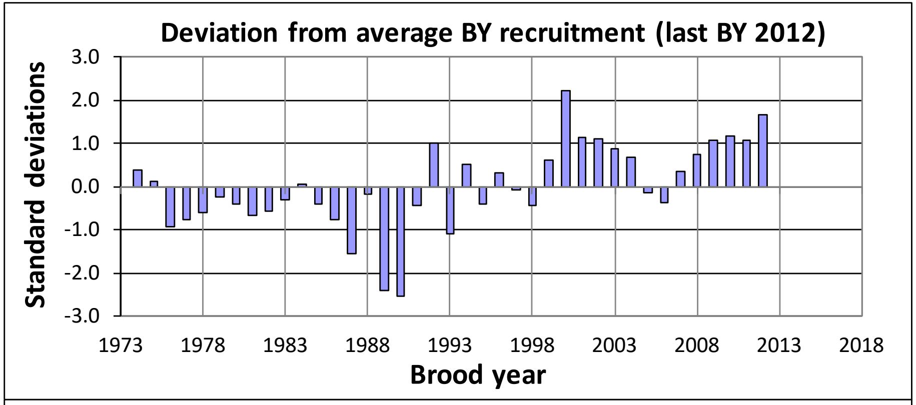


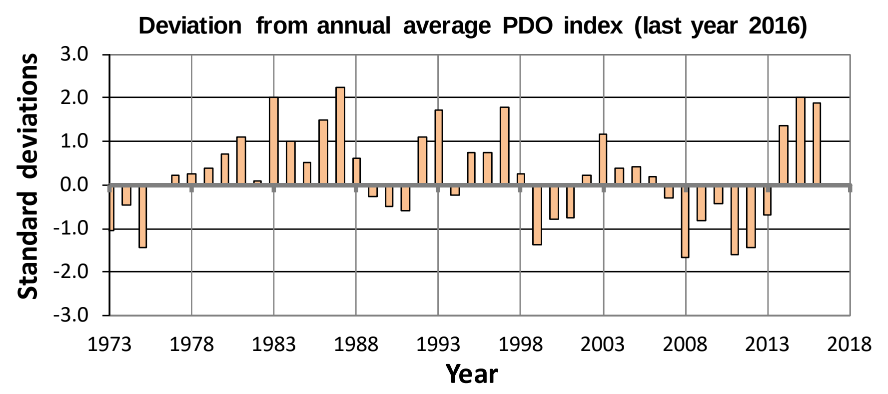

Quilcene R subpopulation S-R curves
Recruits
50,000
40,000
30,000
20,000
10,000
0
0
8,000
16,000
24,000
32,000
40,000
Spawners
BY 79-98
BY 98-12
BY 79-98 S-R curve
BY 98-12 S-R curve

Big and Little Quilcene R subpopulation recruit pattern

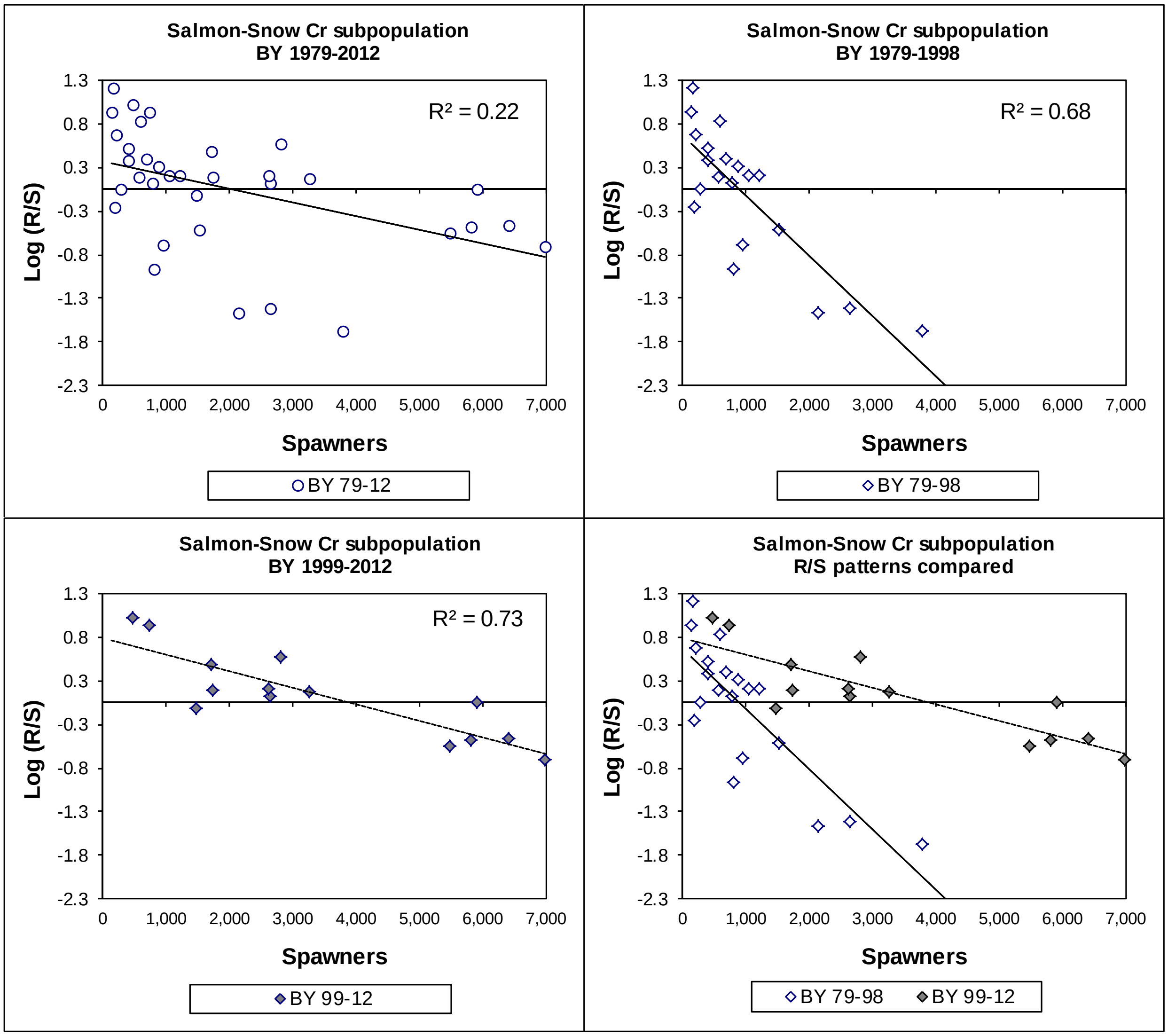


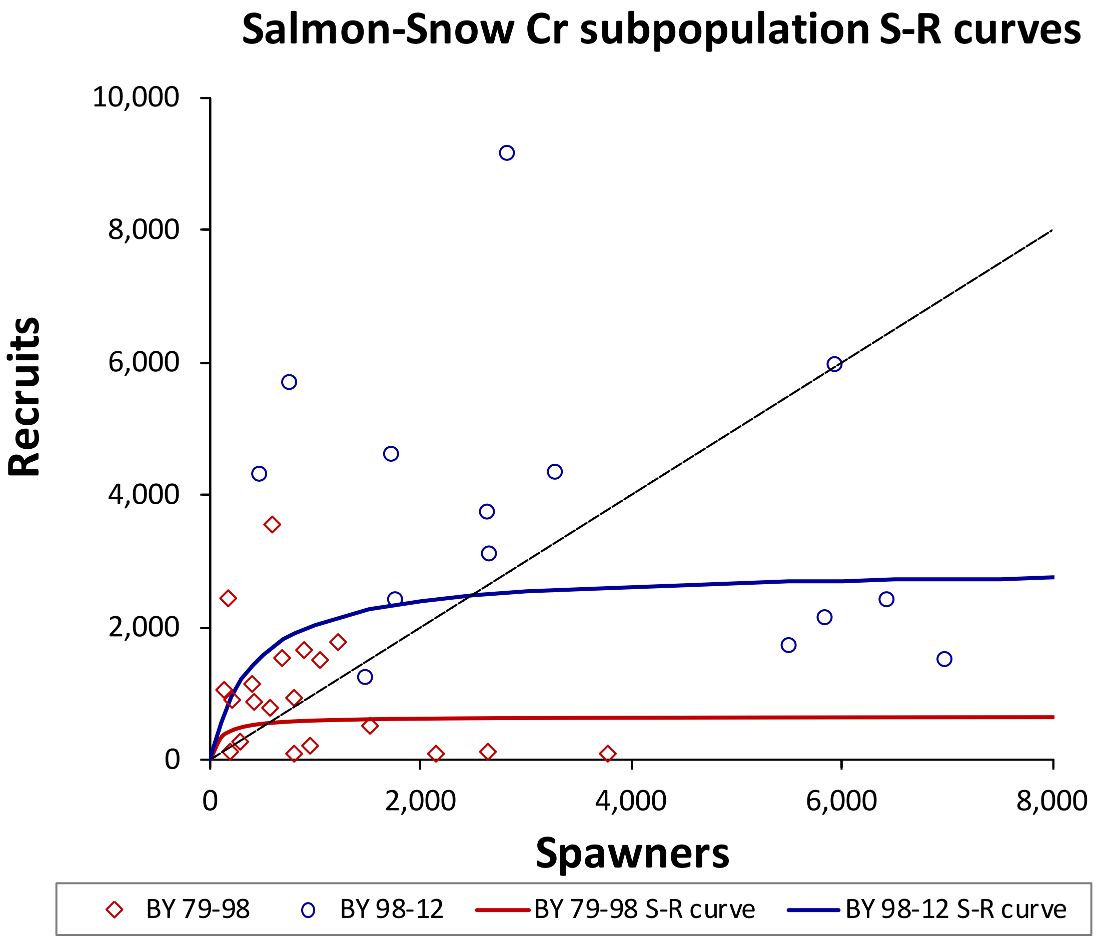




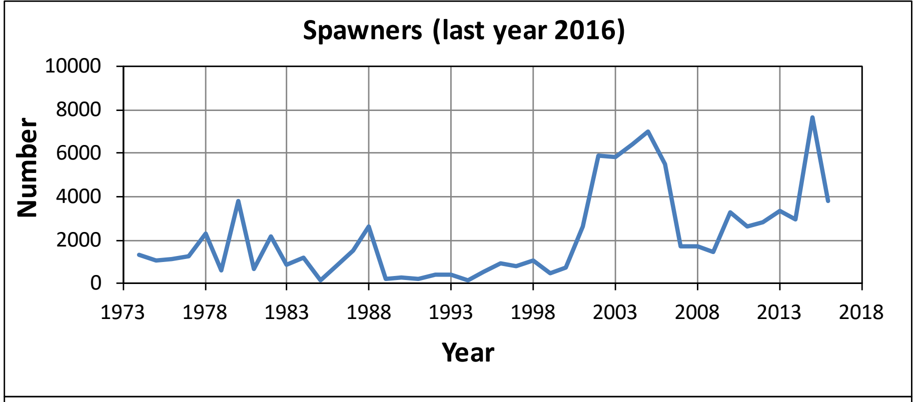


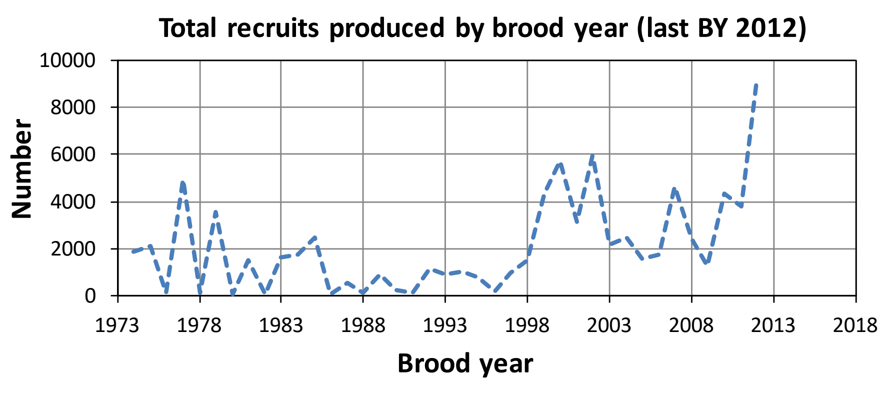
Salmon-Snow Creek Log (S/R) Plots

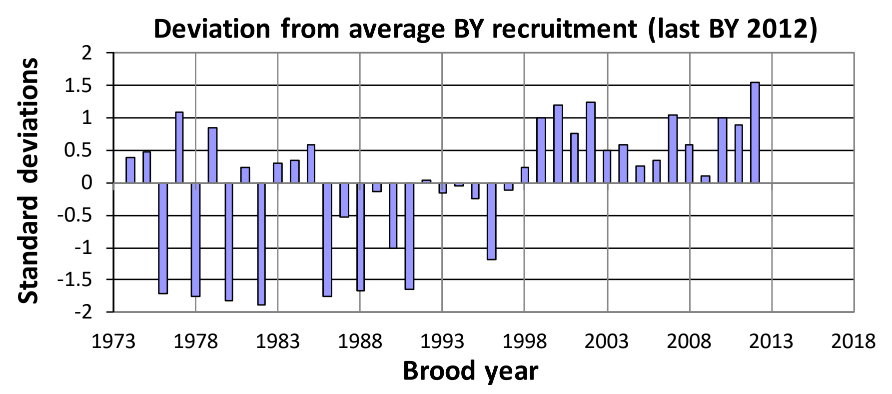


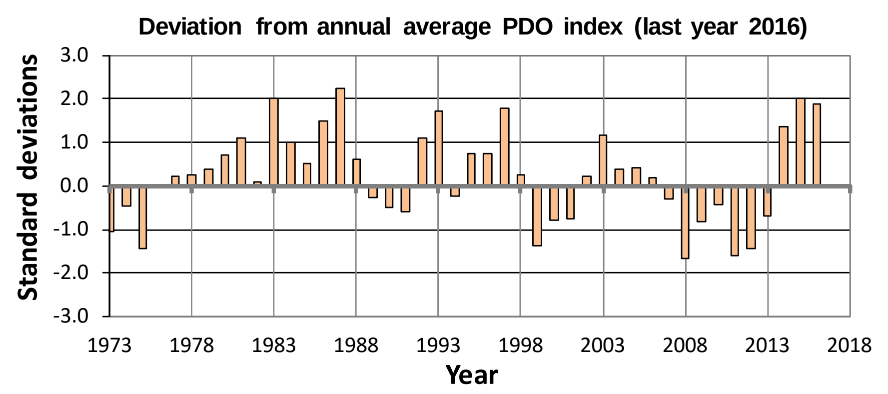

Salmon-Snow Cr subpopulation S-R curves
Recruits
10,000
8,000
6,000
4,000
2,000
0
0
2,000
4,000
6,000
8,000
Spawners
◇ BY 79-98
○ BY 98-12
BY 79-98 S-R curve
BY 98-12 S-R curve

Salmon-Snow Cr subpopulation recruit pattern

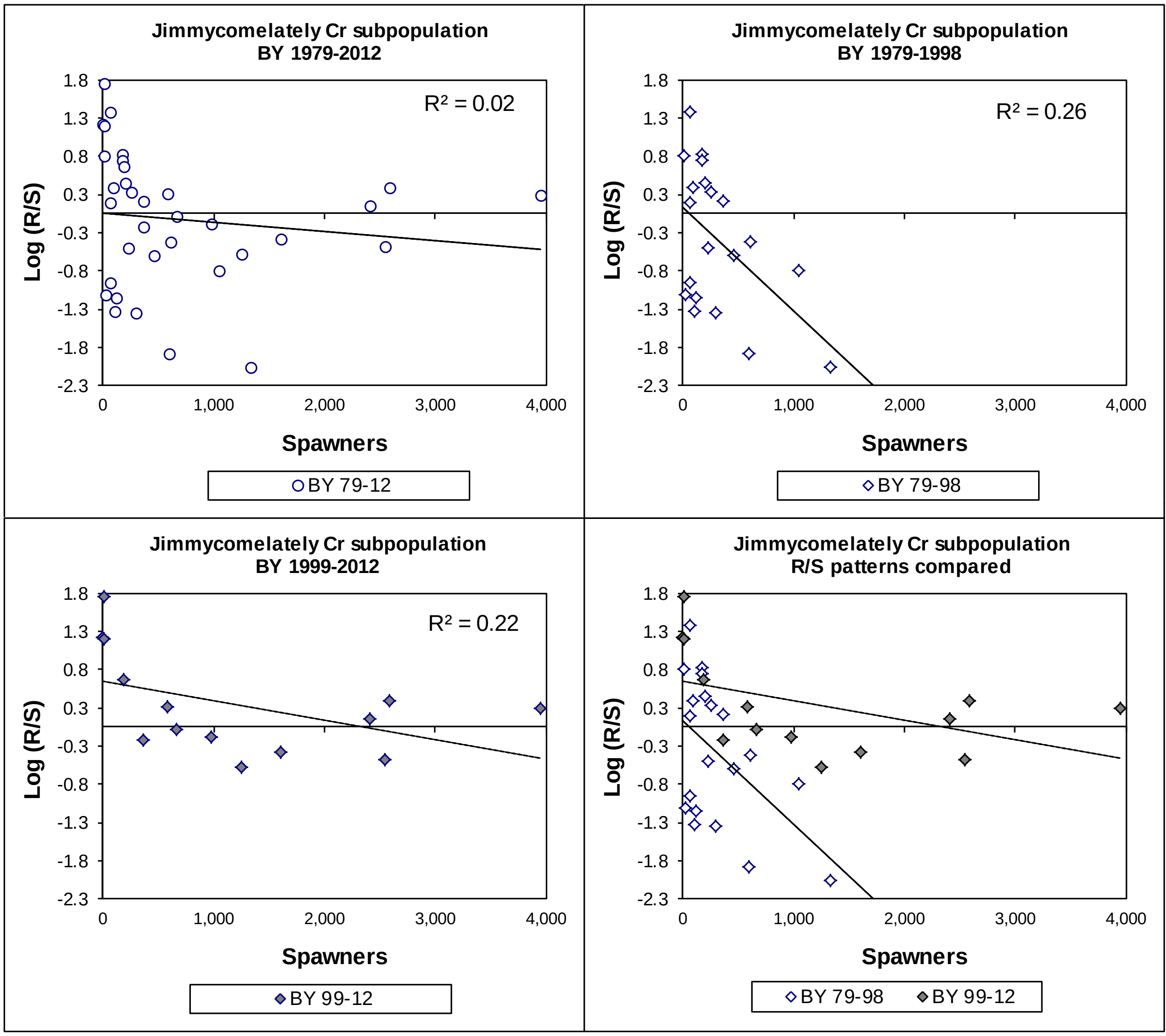

Spawners (last year 2016)

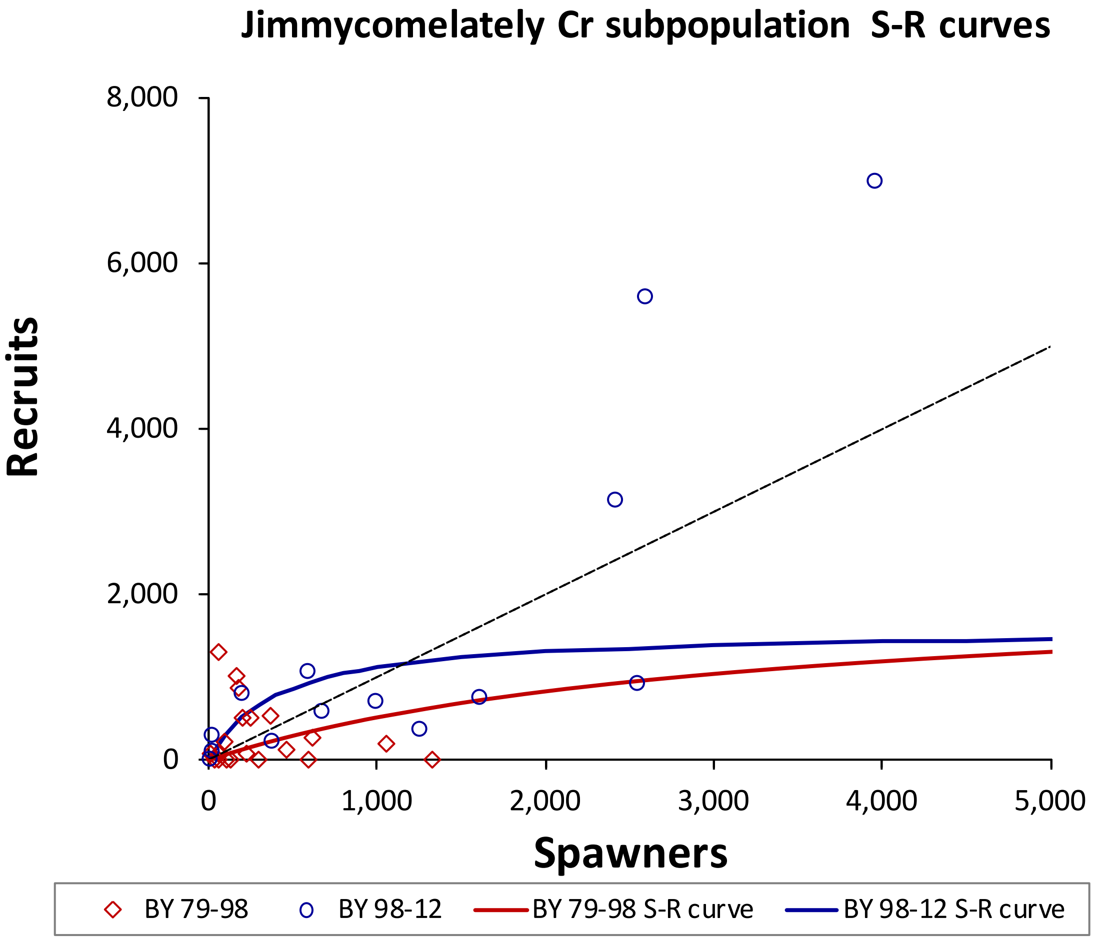

Total recruits produced by brood year (last BY 2012)


Deviation from average BY recruitment (last BY 2012)

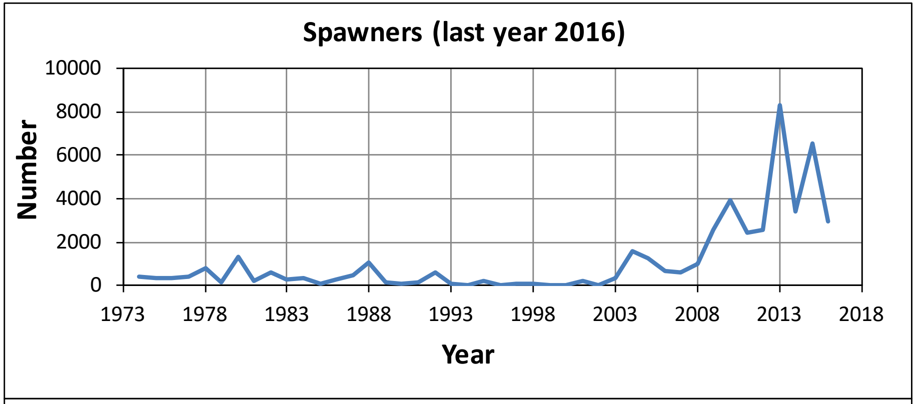

Deviation from annual average PDO index (last year 2016)

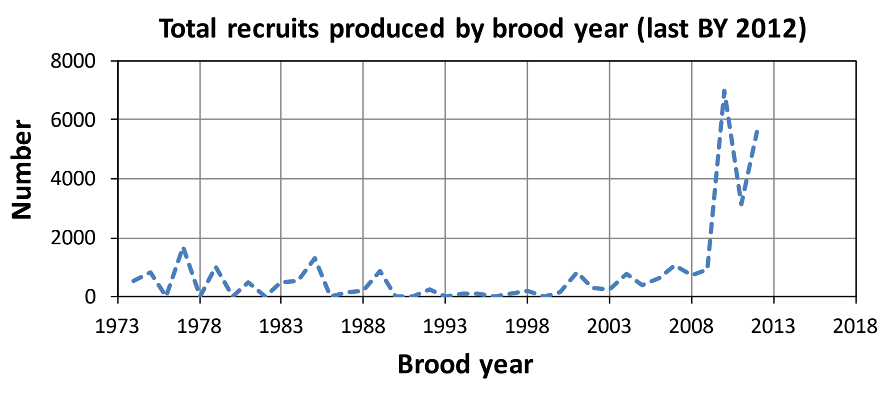
Jimmycomelately Creek Log (S/R) Plots

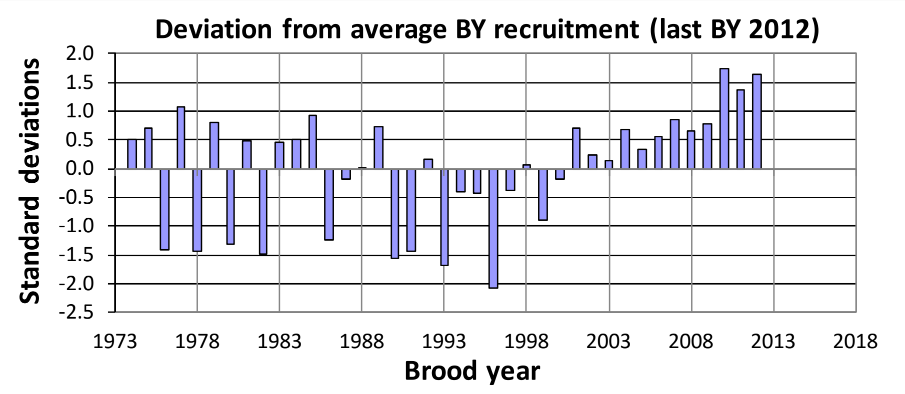


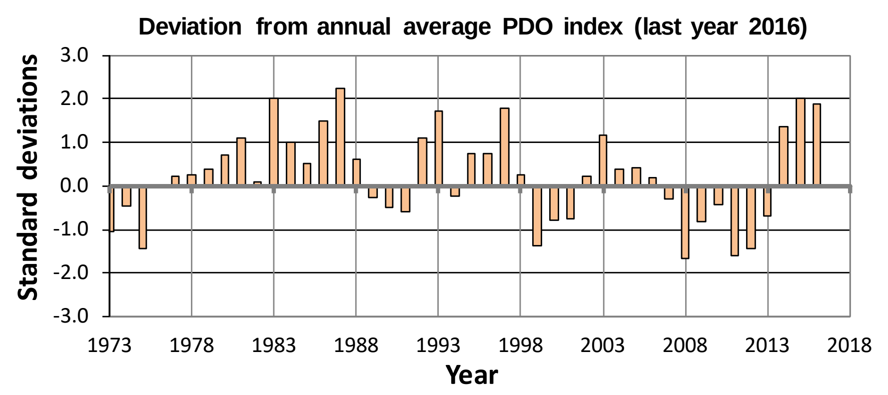

Jimmycomelately Cr subpopulation S-R curves
Recruits
8,000
6,000
4,000
2,000
0
0
1,000
2,000
3,000
4,000
5,000
Spawners
◇ BY 79-98
○ BY 98-12
BY 79-98 S-R curve
BY 98-12 S-R curve

Jimmycomelately Cr subpopulation recruit pattern









Appendix C – Participants in Biological Diversity Forum

Hood Canal Summer Chum Biological Diversity and Spatial Structure Forum
April 20 and 28, 2017
Hood Canal Coordinating Council Conference Room, Poulsbo, WA

Invited scientists who participated in the forum:

- Currens, Ken, Northwest Indian Fisheries Commission
- Downen, Mark, Washington Department of Fish and Wildlife
- Johnson, Thom, Point No Point Treaty Council
- Lestelle, Larry, Biostream Environmental
- Rawson, Kit, member of the Puget Sound Technical Recovery Team (PSTRT)
- Small, Maureen, Washington Department of Fish and Wildlife
- Spidle, Adrian, Northwest Indian Fisheries Commission
- Tynan, Tim, National Marine Fisheries Service (NOAA)

Appendix C – Participants in Biological Diversity Forum

Hood Canal Summer Chum Biological Diversity and Spatial Structure Forum
April 20 and 28, 2017
Hood Canal Coordinating Council Conference Room, Poulsbo, WA

Invited scientists who participated in the forum:

- Currens, Ken, Northwest Indian Fisheries Commission
- Downen, Mark, Washington Department of Fish and Wildlife
- Johnson, Thom, Point No Point Treaty Council
- Lestelle, Larry, Biostream Environmental
- Rawson, Kit, member of the Puget Sound Technical Recovery Team (PSTRT)
- Small, Maureen, Washington Department of Fish and Wildlife
- Spidle, Adrian, Northwest Indian Fisheries Commission
- Tynan, Tim, National Marine Fisheries Service (NOAA)