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Viability Criteria for Steelhead within the Puget Sound Distinct Population Segment

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Dedication

Robert August Hayman
27 February 1952–20 December 2011

In the midst of developing and drafting this document, the Puget Sound Steelhead Technical Recovery Team was saddened by the death of one of its members, Bob Hayman. His contributions to the TRT went beyond his extensive knowledge of the Skagit River basin; it was his determination that the TRT process be logical, consistent, and transparent that ensured its work would meet the highest standards. Bob's good-natured and humble manner made him likeable even when he was challenging your thinking. He was a tireless worker, and the determination he displayed in both his professional endeavors and battling cancer was inspiring. We all miss him.

Executive Summary

Introduction

Under the U.S. Endangered Species Act (ESA), the National Oceanographic and Atmospheric Administration's National Marine Fisheries Service (NMFS) is required to identify measurable and objective delisting criteria as part of recovery planning. These delisting criteria must describe the conditions under which a listed species or distinct population segment (DPS) is no longer in danger of extinction (endangered) or likely to become so in the foreseeable future (threatened). We define a viable DPS as one that is unlikely to be at risk of extinction in the foreseeable future; for this purpose, we adopted the viable salmonid population (VSP) criterion of a 100-year timeline (McElhany et al. 2000, Viable Salmon Populations and the Recovery of Evolutionarily Significant Units, NOAA Tech. Memo. NMFS-NWFSC-42) to evaluate risk of extinction. Ultimately, the identification of delisting criteria requires the consideration of technical analyses relating to viability, which are contained in this report, and policy decisions such as acceptable levels of risk, which are not. It presents the biological viability criteria recommended by the Puget Sound Steelhead Technical Recovery Team (PSS TRT). The framework and the analyses it supports do not set targets for delisting, nor do they explicitly identify specific populations or groups of populations for recovery priority. Rather, the framework and associated analyses are meant to provide a technical foundation for those charged with recovery of listed steelhead (*Oncorhynchus mykiss*) in Puget Sound, from which they can develop effective recovery plans at the watershed scale (and higher) that are based on biologically meaningful criteria.

This technical memorandum develops viability criteria for the Puget Sound Steelhead DPS as identified in the NMFS status review (Hard et al. 2007, Status Review of Puget Sound Steelhead, NOAA Tech. Memo. NMFS-NWFSC-81). The DPS was listed as a threatened species under the ESA in May 2007. Under the ESA, a threatened species is "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range" and an endangered species is "any species which is in danger of extinction throughout all or a significant portion of its range." The viability of a threatened species is therefore at some risk, and in an attempt to quantify that risk, the current document was developed by the PSS TRT, which was composed of scientists from federal, state, tribal, and local government agencies with expertise in steelhead biology and management.

The primary purpose of this document is to recommend objective, measurable, biological criteria for assessing the recovery and progress toward recovery of the Puget Sound Steelhead DPS. In doing so, we applied these criteria to an assessment of the current biological status of the DPS. We do not provide recommendations for criteria for delisting of the DPS under the ESA, nor do we evaluate whether the DPS should or should not be listed. In addition to evaluating the biological requirements presented here, listing and delisting decisions require evaluations regarding particular listing factors and conservation measures, which are beyond the scope of this report. Thus the criteria presented here provide a necessary but not sufficient set of criteria to determine the ESA listing status of the DPS.

Approach to Developing the Criteria

We developed viability criteria based on the considerations laid out in the VSP document (McElhany et al. 2000, Viable Salmon Populations and the Recovery of Evolutionarily Significant Units, NOAA Tech. Memo. NMFS-NWFSC-42, online at http://www.nwfsc.noaa.gov/assets/25/6190_06162004_143739_tm42.pdf), which identified four key population parameters that influence the persistence of populations.

- Size—the abundance of all life stages of the species (often only measured as adults);
- Growth rate (productivity)—production over the entire life cycle, often measured as recruits or returns per spawner, or as long-term population growth rate (λ);
- Spatial structure—distribution of individuals among spawning and rearing habitat areas, and connectivity among those areas; and
- Diversity—variation in traits (phenotypic and genotypic) among individuals within and among populations in the DPS.

The goal of recovery is not merely to meet a set of defined criteria, but rather to restore or repair ecological processes that lead to long-term sustainability of the resource. Our approach views the DPS as a complex structure with important processes operating at scales ranging from individual breeding aggregations to the entire DPS. Along this continuum, we identified three hierarchical units for defining attributes related to biological processes that characterize DPS status, from smallest to largest: demographically independent populations (DIPs), major population groups (MPGs), and the entire DPS. We developed VSP-based criteria that pertain to each of the 32 constituent DIPs within the three MPGs (Northern Cascades, Central and South Puget Sound, and Hood Canal and Strait of Juan de Fuca) that compose the Puget Sound DPS. These criteria were developed using the Puget Sound Steelhead DPS population structure the TRT identified in its previous publication (Myers et al. 2015, Identifying Historical Populations of Steelhead within the Puget Sound Distinct Population Segment, NOAA Tech. Memo. NMFS-NWFSC-128).

Analysis of Viability

To achieve full recovery, steelhead populations in the Puget Sound DPS need to be robust enough to withstand natural environmental variation and even some catastrophic events, and they should be resilient enough to support harvest and habitat loss due to human population growth. Recovery therefore requires sufficient abundance and productivity of the DPS and diversity among and within its constituent populations distributed across the DPS's range. Achieving ESA goals requires biological sustainability into the foreseeable future (as well as the absence of threats, which are outside the scope of this report). DPS sustainability implies that the number and distribution of sustainable populations are sufficient and that DPS-level diversity is conserved. Avoiding endangerment requires biological persistence, that is, assurance that all significant parts of the DPS have a high likelihood of persisting over a VSP time horizon, which is 100 years. Achieving viability at each level implicitly requires meeting all criteria for the lower levels.

Our analysis of steelhead viability rests on two primary components: one component focused on evaluating the quantitative abundance information available for individual populations, and the other component aimed at assessing viability at multiple hierarchical scales (DIP→MPG→DPS) by incorporating qualitative and quantitative information pertaining to all four VSP criteria. For the first component—a quantitative analysis of individual population data—we applied two distinct population viability analyses (PVAs) to the DIPs that have sufficient information on abundance and productivity to support them. The first of these PVAs, based loosely on a conventional spawner-recruit analysis, estimates extinction risk from combinations of abundance and productivity across the parameter space. The second of these PVAs, based on an autoregressive state-space analysis of abundance time series, uses estimates of demographic stochasticity to predict future abundance and extinction risk.

For the second component, we developed a knowledge-based decision support system to incorporate these PVAs into a framework to characterize viability that can 1) accommodate quantitative and qualitative information, including habitat-based (rather than strictly biological) metrics, and 2) estimate viability at multiple hierarchical scales (i.e., DIPs, MPGs, and the entire DPS). This framework is based on a Bayesian network (BN, aka Bayesian belief network or Bayes net) of probabilities that links the key aspects of viability—abundance, productivity, diversity, and spatial structure—across the DPS and all its constituent populations. The network framework we designed links criteria at a variety of scales and aggregates them from population-level criteria, through MPG-level criteria, to criteria for the entire DPS.

Identification and Application of Viability Criteria

For each DIP, the components of population-level diversity are: 1) VSP abundance—adult and juvenile abundance relative to estimated capacity, and the probability that abundance would reach a specified quasi-extinction threshold (an abundance considered to be tantamount to functional extinction) within 100 years; 2) VSP productivity—the number of smolts per spawner and the number of adults per smolt (both of which influence long-term population growth rate), and the frequency of repeat spawners; 3) VSP diversity—the degree of hatchery influence (using a simple propagule-pressure analysis), variation in adult run timing, the fraction of smolts from resident fish, and current age structure relative to the historic distribution; and 4) VSP spatial structure—the fraction of intrinsic potential rearing and spawning habitat occupied, where intrinsic potential is the area of habitat suitable for steelhead rearing or spawning, at least under historical conditions.

A BN constructed for each DIP estimated DIP viability from each of these VSP parameters using Bayesian probabilities. The BNs for each DIP were then combined to produce a composite BN for estimating the viability of each MPG, with weights provided to consider DIP representation of each major life history type (summer run versus winter run). The composite BNs for each MPG were then combined to estimate the viability of the DPS as a whole, with the contribution of each MPG weighted equally. In combination with the requirement that all MPGs within a viable DPS must be viable, this approach ensured that for an DPS to be viable, a sufficient fraction of its constituent DIPs spread across all its MPGs, representing each major life history type, must be viable.

In defining viability criteria and metrics, we used this hierarchical decision support system to provide an assessment of the current status of the DPS by applying the metrics to data available through 2011. This analysis is provided for illustration to demonstrate how the criteria might be used in the future to assess progress toward recovery. The assessment framework evaluates viability using objective measures of spawner abundance, productivity, hatchery influence, spawner and juvenile distribution, and several measures of population diversity (e.g., resident fish contribution, spawn timing, and relative effective population size). Some of these measures were estimated from physical habitat data, in particular metrics extracted from Geographic Information System (GIS) data layers that measure intrinsic potential steelhead production. (The algorithm used by the TRT to estimate intrinsic potential spawning or rearing area and potential steelhead production is described in Appendix C of this report). Fully quantifiable criteria are not available for several metrics, so we used TRT members' judgment to evaluate those. In evaluating DIP-level viability, we gave the combined diversity and spatial structure metrics equal weight of the more quantifiable abundance and productivity metrics. We then used recent observations of population performance (primarily trends in abundance) to evaluate how certain we can be that the Puget Sound Steelhead DPS is sustainable under current conditions.

The PSS TRT considered two issues specific to steelhead viability for which little information is available in Puget Sound populations: the demographic contributions of a resident life history and the degree of iteroparity (repeat spawning). In determining the viability of steelhead DIPs in the Puget Sound DPS, the TRT considered the potential influence of co-occurring resident *O. mykiss* on anadromous steelhead demographics. The TRT concluded that in basins where anadromous *O. mykiss* abundance is below the quasi-extinction threshold, the risk of extinction is not necessarily 100% if resident *O. mykiss* are present below natural, long-standing migration barriers. Inclusion of resident fish in the viability criteria is dependent, however, on several conditions. First, the abundance of resident fish must be large enough to be self-sustaining (this will be largely dependent on the numbers of resident females). Second, there must be some evidence of interbreeding between the anadromous and resident forms (this will most likely be established by genetic analysis of the relationship between the resident fish and the most proximate anadromous population). To accommodate these considerations, the TRT included in the BN for DIP viability a diversity node describing the proportional contribution of smolts from resident fish.

Iteroparity is another primary characteristic of *O. mykiss*, with potentially substantial demographic consequences. Simulation analyses of the demographic consequences of iteroparity were conducted to develop a suite of age-structured matrix population projection models with annual time steps that incorporate density dependence, varying levels of repeat spawning, and varying levels of density-independent stochasticity and fishing mortality. The stochastic models were used to evaluate the influence of varying levels of repeat spawning on population abundance and age structure under varying levels of fishing.

The modeling indicated that the average proportion of repeat spawners in an adult steelhead population is relevant to population abundance and stream capacity in two ways. First, relative to a population with no repeat spawners, for a given average capacity of juveniles, a population with repeat spawners will have a larger average adult spawning population. Second, a given average number of adult spawners can be sustained by fewer juveniles when repeat spawners are present than when they are not. It is therefore likely that viable populations (DIPs) of steelhead can be sustained in smaller stream basins than is the case for semelparous Pacific salmon (*Oncorhynchus* spp.). The model results also indicated that repeat spawning provides increased levels of resilience compared to populations without repeat spawning. From a population rebuilding and recovery perspective, it appears that for small population sizes such as the ones considered in this modeling exercise, the value of specific levels of repeat spawning to population resilience (and repeat spawning) are most likely to be realized under a no-fishing scenario, regardless of the level of environmental variation (at least for the range and kind of variation employed in the models).

While these analyses are exploratory, they and the Integral Projection Model analysis applied to a wild Alaskan steelhead population in Appendix G of this technical memorandum tend to reinforce the TRT's conviction that iteroparity is an important consideration in a comprehensive evaluation of viability for anadromous coastal steelhead. The team therefore included in the BN for DIP viability a productivity node describing the influence of iteroparity on aspects of viability.

Viability Criteria for Puget Sound Steelhead

The listed unit under the ESA for *O. mykiss*, including anadromous steelhead, is the DPS, and this is the unit that must be considered for delisting. Thus delisting criteria must ultimately address the extinction risk of the DPS. In approaching the development of criteria for assessing viability (and subsequent delisting) of DPSs, we have relied on the language in the ESA, information described in the listing decision, concepts outlined in the VSP document, and published research describing salmon populations and their past or potential responses to environmental changes. The ESA in Section 4.2.1 lists five potential factors for decline that must be considered in species listing decisions:

1. The present or threatened destruction, modification, or curtailment of its habitat or range;
2. Overutilization for commercial, recreational, scientific, or educational purposes;
3. Disease or predation;
4. The inadequacy of existing regulatory mechanisms; and
5. Other natural or man-made factors affecting its continued existence.

NMFS identified all five factors as contributing to the endangerment of Pacific salmonids, including steelhead. In considering how viability criteria might inform population delisting requirements, the TRT also attempted to consider these factors for decline.

For the Puget Sound Steelhead DPS, the TRT developed in this report criteria for recovery of steelhead that share some similarities:

1. The viability (as reflected in abundance, productivity, diversity, and spatial structure) of a majority of steelhead populations in each of the MPGs across the DPS is detectably higher than currently, using conventional PVA.

2. At least 40% of steelhead populations in each of the three MPGs within the DPS achieve viability, depending on the historical biological characteristics and acceptable risk levels for populations within each region. (The threshold of 40% corresponds to an average probability of DIP viability of 67% if the DIPs that are not viable have probabilities of viability <55%.)
3. At least 40% of populations from each major life history type (summer run, winter run) historically present within each of the MPGs are viable using these criteria.
4. Natural production of steelhead from tributaries to Puget Sound not identified as primary spawning or rearing habitat for any of the 32 identified populations is sustained to provide ecological diversity and productivity sufficient to support DPS-wide recovery.

From these considerations, the PSS TRT developed the following constructs for developing viability criteria at the three hierarchical levels of DIP, MPG, and DPS.

DIP Viability

1. Using a simple three-bin classification (not viable, intermediate, viable), a DIP is scored 1, 2, or 3, depending on whether the probability of viability as computed by its viability BN is <40%, 40–85%, or >85%, respectively. For a DIP to be considered viable, its probability of viability must be at least 85%; the TRT considered this estimate to be sufficiently high to insure persistence over 100 years. To estimate this value, the mean abundance and productivity criteria for the candidate DIP are each double weighted; this produces DIP viability scores ranging from 6 to 18 points. DIPs with scores less than 11 are considered not viable, those with scores between 11 and 14 are considered to have intermediate viability, and those with scores greater than 14 (>85%, rounded up) are considered viable.

MPG Viability

2. An MPG is considered viable if 40% (rounded up) of its DIPs (including extinct as well as extant historical populations) are viable and mean DIP viability exceeds a threshold for viability.
    a. DIPs exhibiting distinct life history strategies (i.e., summer run versus winter run) will be considered separate components of the MPG. Therefore, a minimum of 40% of summer-run and 40% of winter-run populations within an MPG must be viable to achieve MPG viability.
    b. DIPs containing both winter-run and summer-run subpopulations predominantly exhibit the winter-run life history strategy in Puget Sound and will be considered winter-run for the purpose of estimating 2a. This approach should be reevaluated as data on summer-run populations becomes available.
    c. A viable MPG must, in addition to the criterion outlined in 2a and 2b, have a geometric mean (averaged over all its DIPs) score of at least 2.2 to be considered viable, using the 1–3 scale for individual DIPs described under the DIP Viability heading above.

DPS Viability

3. A DPS is considered viable only if all its component MPGs are viable.

The TRT applied the criteria to the 32 DIPs in the Puget Sound Steelhead DPS to provide a status assessment of the current viability of these units. This assessment indicates low viability for many DIPs, all MPGs, and the DPS as a whole. Nearly all DIPs have insufficient current abundance and productivity scores to be considered viable. Most DIPs also have low scores for diversity and spatial structure, largely because of extensive hatchery influence, low breeding population sizes, and freshwater habitat fragmentation or loss.

The Puget Sound Steelhead DPS is not considered to be viable by the TRT. The TRT concluded that the DPS is currently at very low viability; most of the 32 DIPs, all three MPGs, and the DPS as a whole are at low viability. Nearly all DIPs in both the Central and South Puget Sound MPG and the Hood Canal and Strait of Juan de Fuca MPG are not viable. Most of the DIPs score low for all four VSP criteria, and nearly all DIPs have insufficient current abundance and productivity scores to be considered viable. Most DIPs also have low scores for diversity and intermediate scores for spatial structure. The pattern of low viability is widespread throughout Puget Sound, across all three MPGs, and includes both summer-run and winter-run populations. The populations with highest viability—and those with highest abundance and diversity—are in northeastern Puget Sound (Northern Cascades MPG).

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We are grateful to the National Marine Fisheries Service's West Coast Region for its support to the Puget Sound Steelhead Technical Recovery Team (PSS TRT) during the preparation of this report. Members of the Puget Sound Domain Team, including Elizabeth Babcock, Steve Leider, Amilee Wilson, Alison Agness, Samantha Brooke, and Tim Tynan, were especially helpful to the PSS TRT's deliberations. We also acknowledge the advice and contributions of Susan Bishop, Bruce Crawford, Bob Donnelly, and Steve Stone, West Coast Region; Tristan Peter-Contesse and Jason Mulvihill-Kuntz, Puget Sound Partnership; and Lloyd Moody and Sara Laborde, Governor's Salmon Recovery Office.

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Abbreviations

BN	Bayesian network
BRT	Biological Review Team
CPT	conditional probability table
DIP	demographically independent population
DPS	distinct population segment
DSS	decision support system
ESA	U.S. Endangered Species Act
ESU	evolutionarily significant unit
IP	intrinsic potential
IPM	integral projection model
MARSS	multivariate autoregressive state-space
MPE	most probable explanation
MPG	major population group
NMFS	National Marine Fisheries Service
PSS TRT	Puget Sound Steelhead Technical Recovery Team
PVA	population viability analysis
QET	quasi-extinction threshold
SAS	smolt-to-adult survival
SaSI	Salmonid Stock Inventory
SPAZ	Salmon Population AnalyZer
SSH	summer-run steelhead
VSP	viable salmonid population
WDFW	Washington Department of Fish and Wildlife
WSH	winter-run steelhead

Introduction

The goal of recovery and restoration is to restore or repair ecological processes that lead to long-term sustainability of natural resources. Like other coastal steelhead (*Oncorhynchus mykiss*) in the southern portion of their range in the western United States, that is, California (Boughton et al. 2007) and Oregon (Chilcote 1997), steelhead in Puget Sound have shown declines in abundance for decades (Hard et al. 2007). The Puget Sound Steelhead Distinct Population Segment (DPS) was listed as a threatened species under the U.S. Endangered Species Act (ESA) in May 2007. The Puget Sound Steelhead Technical Recovery Team (PSS TRT), which convened in March 2008, is charged with developing criteria for viability of steelhead in Puget Sound that contribute to the long-term sustainability of these fish. Puget Sound steelhead have a complex, hierarchical structure affected by ecological processes operating at scales ranging from individual spawning aggregations up to the entire DPS. Along this continuum, the TRT members identified three scales as important for defining attributes related to biological processes that define status: demographically independent populations (DIPs), major population groups (MPGs), and the entire DPS. We developed viability criteria at each scale. The TRT also developed attribute sets for each scale to assess viability status.

The framework described in this report and the analyses it supports do not set biologically based targets for delisting or recovery of steelhead, nor do they explicitly identify specific populations or groups of populations for recovery priority. Rather, the framework and associated analyses are meant to provide a technical foundation for those charged with recovery of listed steelhead in Puget Sound, from which they can develop effective recovery plans at the watershed scale (and higher) that are based on biologically meaningful criteria.

Viable Salmonid Population Criteria

Our approach to assessing viability of Puget Sound steelhead adhered to the viable salmonid population (VSP) criteria originally described by McElhany et al. (2000) to address conservation and recovery of Pacific salmonids under the ESA. According to McElhany et al. (2000), four parameters form the key to evaluating viability status at the population scale. We slightly modify their description of these parameters here:

- Population size (abundance)—the abundance of all life stages of the species;
- Population growth rate (productivity)—production over the entire life cycle, often measured as recruits or returns per spawner, or as long-term population growth rate (λ);
- Diversity—phenotypic and genotypic variation in traits among individuals in a population and among populations in the conservation unit; and
- Spatial structure—distribution of individuals among spawning and rearing habitat areas, and connectivity among those areas.

In considering recovery of salmon and steelhead units (evolutionarily significant units [ESUs] in the case of Pacific salmon [*O.* spp.] and DPSs in the case of steelhead and Atlantic salmon [*Salmo salar*]), the National Marine Fisheries Service (NMFS) focuses on these parameters for three reasons. First, they are reasonable predictors of avoiding extinction risk, which reflects the ability of a population to persist into the future (i.e., its viability). We adopt here the VSP approach of considering a conservation unit's viability primarily as its ability to persist over a 100-year time frame. Second, they reflect general processes that are important to all populations of all species. Third, the parameters are measurable.

Population size, or abundance, is recognized as an important parameter because, all else being equal, small populations are at greater risk of extinction than large populations, primarily because several processes that affect population dynamics operate differently in small populations than they do in large populations. These processes are deterministic density effects, environmental variation, genetic processes, demographic stochasticity, ecological feedback, and catastrophes. McElhany et al. (2000) provided guidelines relating minimum abundance to each of these processes at both the "viable" and "critical" levels, where a critical level implies a high risk of extinction over a short time period.

Population growth rate (i.e., productivity over the entire life cycle) and factors that affect population growth rate provide information on how well a population is "performing" in the natural habitats it occupies during the life cycle. Estimates of population growth rate that indicate a population is consistently failing to replace itself demonstrate increased extinction risk. Although the overall focus is on population growth rate over the entire life cycle, estimates of stage-specific productivity—particularly productivity during freshwater life history stages—are also important to comprehensive evaluation of population viability. Other measures of population productivity, such as intrinsic productivity and the intensity of density dependence, may provide important additional information for assessing a population's viability. The McElhany et al. (2000) guidelines for population growth rate are closely linked with those for abundance.

Several steelhead traits exhibit considerable diversity within and among populations, and this variation can have important effects on population viability (Boughton et al. 2007). In a spatially and temporally varying environment, there are three general reasons why biological diversity is important for population (and DPS) viability. First, it allows a population to use a wider array of environments than they could without it. Second, diversity protects against short-term spatial and temporal changes in the environment. Third, genetic diversity provides the raw material for adapting to long-term environmental change.

Finally, when evaluating population viability, it is important to take spatial structure into account for two main reasons: 1) because there is a time lag between changes in spatial structure and species (DPS)–level effects, overall extinction risk at the 100-year time scale may be affected in ways not readily apparent from short-term observations of abundance and productivity, and 2) population spatial structure affects evolutionary processes and may therefore alter a population's ability to respond to environmental change. Spatially structured populations in which subpopulations occupy habitat patches connected by some low to moderate migration rates are often generically referred to as metapopulations. A metapopulation's spatial structure depends fundamentally on habitat quality, spatial configuration, and demographic dynamics, as well as the dispersal characteristics of individuals in subpopulations. Pacific salmonids such as steelhead are generally recognized to have some degree of metapopulation structure. Our criteria for steelhead population spatial structure attempt to account for this structure.

Population viability is typically estimated as some correlate of the probability that a low-abundance threshold, known as a quasi-extinction threshold (QET), will be reached within 100 years. With sufficient quantitative data on abundance or productivity, this can be estimated using a variety of statistical approaches that involve analysis of time series. Incorporating information on population diversity and spatial structure is also important in assessing viability in a larger, holistic sense, but combining this information with abundance and productivity data in a statistical analysis of extinction risk remains a major challenge in evaluating viability in a conservation framework.

However, the ESA is not concerned with the viability of populations per se, but rather with the extinction risk faced by an entire conservation unit (i.e., an ESU or DPS). A key question is how the viabilities of a suite of constituent populations and major population groups are related to the sustainability of the entire unit. Three factors need to be considered when relating VSP parameters to viable conservation units: 1) catastrophic events, 2) long-term demographic processes, and 3) long-term evolutionary potential. McElhany et al. (2000) provided several guidelines related to these factors with an emphasis on risks from catastrophic events:

1. **Conservation units should contain multiple populations.** If an ESU (or DPS) is made up of multiple populations, it is less likely that a single catastrophic event will cause it to become extinct. Also, these conservation units may function as metapopulations over the long term and the existence of multiple populations would be necessary for the operation of sustainable population-level extinction/recolonization processes. In addition, multiple populations within a unit increase the likelihood that a diversity of phenotypic and genotypic characteristics will be maintained, thus allowing natural evolutionary processes to operate and increasing the unit's viability in the long term. Obviously, this guideline does not apply to units that appear to contain a single population. In units containing a single population, Guideline 6 below becomes more important.
2. **Some populations in a conservation unit should be geographically widespread.** Spatially correlated environmental catastrophes are less likely to drive a widespread unit to extinction. This guideline also directly relates to the ESA mandate of protecting a species in a "significant portion of its range."
3. **Some populations should be geographically close to each other.** On long temporal scales, conservation units may function as metapopulations and having populations geographically close to one another facilitates connectivity among existing populations. Thus a viable unit composed of multiple populations requires both widespread (Guideline 2) and spatially close populations.
4. **Populations should not all share common catastrophic risks.** A conservation unit containing populations that do not share common catastrophic risks is less likely to be driven to extinction by correlated environmental catastrophes. Maintaining geographically widespread populations is one way to reduce risk associated with

correlated catastrophes (Guideline 2), but spatial proximity is not the only reason why multiple populations could experience a correlated catastrophic risk.

5. **Populations that display diverse life histories and phenotypes should be maintained.** When a conservation unit's populations have a fair degree of life history diversity (or other phenotypic diversity), the unit is less likely to go extinct as a result of correlated environmental catastrophes or changes in environmental conditions that occur too rapidly for an evolutionary response. In addition, assuming phenotypic diversity is caused at least in part by genetic diversity, maintaining diversity allows natural evolutionary processes to operate within a unit.
6. **Some populations should exceed VSP viability criteria.** In other words, some constituent populations should have a high probability of persistence for longer than the criteria required for viability of an entire conservation unit. Larger and more productive (resilient) populations may be able to recover from a catastrophic event that would cause the extinction of a smaller population. A conservation unit that contains some populations in excess of VSP threshold criteria for abundance and population growth rate is less likely to go extinct in response to a single catastrophic event that affects all populations. It is important to note that the abundance guidelines do not take catastrophes into account. If a unit consists of a single population or a small number of populations, the viability criteria for them should be stronger than the viability criteria for populations within a unit composed of many populations.
7. **Evaluations of a conservation unit's status should take into account uncertainty about unit-level processes.** Our understanding of unit-level spatial and temporal process is very limited. Conservation units are believed to have been historically self-sustaining and the historical number and distribution of populations serves as a useful default goal in maintaining viable units.

Approaches to Assessing Viability

We considered several approaches to assessing the viability of the Puget Sound Steelhead DPS. Population viability analysis (PVA) (Beissinger and McCullough 2002) is a set of quantitative or semiquantitative methods used to estimate the probability that a population or collection of populations will persist for a defined period of time in a particular environment. The seminal papers by Dennis et al. (1991) and Thompson (1991) described some of these methods. PVA can assist with evaluating viability at the population level, but it can be challenging to extend this analysis to evaluate viability at higher spatial scales involving multiple populations (yet see, e.g., Ward et al. 2010). We therefore explored additional tools, including decision support systems (DSSs), to conduct a more comprehensive and hierarchical assessment of Puget Sound steelhead viability (see the Decision Support Systems as Tools for Assessing Viability section of this technical memorandum).

In recent status review updates for Pacific salmon and steelhead (e.g., Good et al. 2005), biological review teams (BRTs) have adopted a risk assessment method that has been used for Pacific salmon recovery planning and is outlined in the VSP report (McElhany et al. 2000). In this approach, risk assessment is addressed first at the population level, then at the overall ESU or DPS level. In this approach, individual populations are assessed according to the four VSP criteria: abundance, growth rate/productivity, spatial structure, and diversity. The condition of individual populations is then summarized at the ESU or DPS level, and the patterns of abundance, distribution, and diversity of populations are considered in evaluating the status of the ESU or DPS as a whole. These larger scale patterns include the total number of viable populations, geographic distribution of these populations (to ensure inclusion of major life history types and to buffer the effects of regional catastrophes), and connectivity among these populations (to ensure appropriate levels of gene flow and recolonization potential in case of local extirpations). The considerations are reviewed in McElhany et al. (2000).

The revised risk matrix integrates the four major population VSP criteria directly into the risk assessment process. After reviewing all relevant biological information for the ESU, each evaluator assigns a risk score (see below) to each of the four criteria. The scores are tallied and reviewed by the evaluation team before making its overall risk assessment. Although this process helps to integrate and quantify a large amount of diverse information, there is no simple way to translate the risk matrix scores directly into an assessment of overall risk. For example, simply averaging the values of the various risk factors would not be appropriate; an ESU or DPS at high risk for low abundance would be at high risk even if there were no other risk factors.

Scoring population viability criteria: Risks for each population viability factor are ranked on a scale of 1 (very low risk) to 5 (very high risk):

1. **Very low risk.** Unlikely that this factor contributes significantly to risk of extinction throughout all or a significant portion of the range, either by itself or in combination with other factors.
2. **Low risk.** Unlikely that this factor contributes significantly to risk of extinction throughout all or a significant portion of the range by itself, but some concern that it may, in combination with other factors.
3. **Moderate risk.** This factor contributes significantly to long-term risk of extinction throughout all or a significant portion of the range, but does not in itself constitute a danger of extinction in the near future.
4. **High risk.** This factor contributes significantly to long-term risk of extinction throughout all or a significant portion of the range and is likely to contribute to short-term risk of extinction in the foreseeable future.
5. **Very high risk.** This factor by itself indicates danger of extinction throughout all or a significant portion of the range in the near future.

A “recent events” category considers events that have predictable consequences for ESU or DPS status in the future, but have occurred too recently to be reflected in the population data. Examples include a climatic regime shift or El Niño event that may be anticipated to result in increased or decreased marine productivity in subsequent years. It is scored as follows:

++ (double plus)	expect a strong improvement in status of the ESU or DPS,
+ (single plus)	expect some improvement in status,
0	neutral effect on status,
– (single minus)	expect some decline in status,
– – (double minus)	expect a strong decline in status.

This analysis of overall risk to the ESU or DPS throughout all or a significant portion of its range has used categories that correspond to definitions in the ESA: in danger of extinction, likely to become endangered in the foreseeable future, or neither. These evaluations do not consider protective efforts (conservation measures) and therefore are not recommendations regarding listing status. The overall risk assessment reflects professional judgment by each evaluation team member and thus is necessarily somewhat subjective. This assessment is guided by the results of the risk matrix analysis as well as expectations about likely interactions among factors. For example, a single factor with a high risk score might be sufficient to result in an overall score of "in danger of extinction throughout all or a significant portion of the range," but a combination of several factors with more moderate risk scores could also lead to the same conclusion.

To allow for uncertainty in judging the actual risk facing the ESU or DPS, the evaluation team has often adopted a "likelihood point" method, often referred to as the FEMAT method because it is a variation of a method used by scientific teams evaluating options under former President Clinton's Forest Plan (Forest Ecosystem Management: An Ecological, Economic, and Social Assessment Report of the Forest Ecosystem Management Assessment Team, or FEMAT). In this approach, each team member distributes 10 likelihood points among the three ESU/DPS risk categories, reflecting one's opinion of how likely that category correctly reflects the true ESU/DPS status. Thus if a member were certain the ESU or DPS was in the "not at risk" category, all 10 points could be assigned to that category. A reviewer with less certainty about ESU/DPS status could split the points among two or even three categories. This method has been used in all status review updates for anadromous Pacific salmonids since 1999; although not strictly scientific, it does capture a range of scientific judgment and opinion about extant risk.

Assessing a population's viability in a comprehensive way requires consideration of all four VSP criteria (see also Boughton et al. 2007). That being said, relating the contribution of lost diversity or eroded spatial structure quantitatively to extinction risk is no trivial matter. We therefore adopted methods that can accommodate both qualitative and quantitative information to assess viability and develop viability criteria. These methods are explained in detail below. But first we describe our approaches to evaluating current viability at the population level using quantitative criteria. For this effort we relied on two different methods: viability curve analysis and evaluation of abundance data time series.

Viability Assessment Methods

The PSS TRT adopted methods described by Good et al. (2005) to evaluate data that affect the four VSP parameters in the Puget Sound Steelhead DPS; these methods are described briefly below. State and tribal comanagers provided data on abundance, the fraction of hatchery origin spawners (where available), harvest, age structure, and hatchery releases to the TRT. Data on adult returns were obtained from a variety of sources, including time series of freshwater spawner surveys, redd counts, and historical catch data. Time series of estimated abundance were assembled and analyzed for each population that had sufficient data. Estimates of harvest were provided for several populations.

Recent Abundance

Recent abundance of natural spawners is reported as the geometric mean (and range) of the most recent data to be consistent with previous coast-wide status reviews of steelhead. Geometric means were calculated to represent the recent abundance of natural spawners for each DIP within the DPS. Geometric means were calculated for the most recent 5 years; this time frame was selected to correspond with modal age at maturity. Zero values in the data set were replaced with a value of 1 and missing data values within a multiple-year range were excluded from geometric mean calculations. The geometric mean is the *n*th root of the product of the *n* data

$$\overline{X_G} = \sqrt[n]{N_1 N_2 N_3 \dots N_n}$$

where $N_{t\,=\,1\dots n}$ is the abundance of natural spawners in year t. Arithmetic means (and ranges) were also calculated for the most recent abundance data

$$\overline{X_A} = \left(\sum_{t=1}^{n} N_t\right) / n$$

where N_t is the abundance of natural spawners in year t.

Trends in Abundance

Short-term and long-term trends were calculated from time series of the total number of adult spawners. Short-term trends were calculated using data from 1995 to the most recent year (2010). Long-term trends were calculated using all the data in a time series. Trend was calculated as the slope of the regression of the number of natural spawners (log-transformed) over the time series. To mediate for zero values, 1 was added to natural spawners before transforming the data. Trend was reported in the original units as exponentiated slope, such that a value greater than 1 indicates a population trending upward and a value less than 1 indicates a population trending downward. The regression was calculated as

$$\ln(N + 1) = \beta_0 + \beta_1 X + \varepsilon$$

where N is the natural spawner abundance, β_0 is the intercept, β_1 is the slope of the equation, and ε is the random error term.

Confidence intervals (95%) for the slope, in their original units of abundance, were calculated as

$$\exp(\ln(b_1) - t_{0.05(2),df} s_{b_1}) \leq \beta_1 \leq \exp(\ln(b_1) + t_{0.05(2),df} s_{b_1})$$

where b_1 is the estimate of the true slope, β_1, $t_{0.05(2),df}$ is the two-sided t-value for a confidence level of 0.95, df is equal to $n - 2$, n is the number of data points in the time series, and s_{b_1} is the standard error of the estimate of the slope, b_1.

Population Growth Rate

In addition to analyses of trends in natural spawners, the median short-term population growth rate (λ) of natural-origin spawners was calculated where possible as a measure for comparative risk analysis. Lambda incorporates overlapping generations and calculates running sums of cohorts. Its interpretation can be difficult because there is no adjustment for density dependence, which is known to be a factor affecting productivity during the freshwater phase of the life cycle in steelhead. Nevertheless, it is widely applied in viability assessment, as most population extinctions are the result of steady declines, $\lambda < 1$. It has been developed for data sets with high sampling error and age-structure cycles (Holmes 2001). These methods have been extensively tested using simulations for both threatened and endangered populations, as well as for stocks widely believed to be at low risk (Holmes 2004), and cross-validated with time-series data (Holmes and Fagan 2002).

Ideally, the λ of natural-origin spawners would be calculated on the basis of natural production alone. Nearly all Puget Sound steelhead abundance data series are based on redd counts to which an average number of fish per redd is applied. Counts or proportions of hatchery fish contributing to natural production typically were not available. Thus it is likely that our computation of λ includes a mixture of hatchery and natural-origin spawners, which could bias the estimates of productivity of wild fish. A multistep process based on methods developed by Holmes (2001) and Holmes and Fagan (2002) and described in McClure et al. (2003) was used to calculate estimates for λ, its 95% confidence intervals, and its probability of decline, $P(\lambda < 1)$. The first step was calculating 4-year running sums (R) for natural-origin spawners as

$$R_t = \sum_{i=1}^{4} N_{t-i+1}$$

where N_t is the number of natural-origin spawners in year t. A 4-year running sum window was used, as the analysis by McClure et al. (2003) indicates this is an appropriate window for a diverse range of salmonid life histories.

Next an estimate of μ, the rate at which the median of R changes over time (Holmes 2001), was calculated as

$$\hat{\mu} = mean\left(ln\left(\frac{R_{t+1}}{R_t}\right)\right)$$

the mean of the natural log-transformed running sums of natural-origin spawners. The point estimate for λ was then calculated as the median annual population growth rate

$$\hat{\lambda} = e^{\hat{\mu}}$$

Confidence intervals (95%) were calculated for $\hat{\lambda}$ to provide a measure of the uncertainty associated with the growth rate point estimate. First an estimate of variability for each population was determined by calculating an estimate for using the slope method (Holmes 2001). The slope method formula is

$$\hat{\sigma}^2_{pop} = \text{slope of the regression of } var\left(ln\left(\frac{R_{t+\tau}}{R_t}\right)\right)\text{vs. } \tau$$

where τ is a temporal lag in the time series of running sums.

Individual population variance estimates were highly uncertain, so a more robust variance estimate, $\hat{\sigma}^2_{avg}$, was obtained by averaging the $\hat{\sigma}^2_{pop}$ estimates from all the populations in the DPS. This average variance estimate was then applied as the variance for every population in the DPS. The degrees of freedom associated with the average variance estimate are obtained by summing the degrees of freedom for each of the individual population variance estimates. The degrees of freedom for the individual population estimates were determined using the method of Holmes and Fagan (2002), which identifies the adjusted degrees of freedom associated with slope method variance estimates. The calculation for the adjusted degrees of freedom is

$$d = 0.212n - 1.215$$

where *n* is the length of the time series. Using the average variance estimate and the summed degrees of freedom, the 95% confidence intervals for λ were calculated as

$$\exp(\hat{\mu} \pm t_{0.05(2),df})\sqrt{\hat{\sigma}^2_{slp}/(n-4)}$$

Recruitment

Recruits, or spawners in the next generation, from a given broodyear were calculated as

$$C_t = \sum_{i=1}^{MaxAge} N_{t+i}A(i)_{t+i}$$

where C_t is the number of recruits from broodyear *t*, N_t is the number of natural-origin spawners in year *t*, and $A(i)_t$ is the fraction of age *i* spawners in year *t*. The estimate of preharvest recruits is similarly

$$C(preHarvest)_t = \sum_{i=1}^{MaxAge} P_{t+i}A(i)_{t+i}$$

where $C(preHarvest)_t$ is the number of preharvest recruits in year t, P_t is the number of natural-origin spawners that would have returned and appeared on the spawning grounds in year t had there not been a harvest, and $A(i)_t$ is the fraction of age i spawners in year t had there not been a harvest. (Because P_t is in terms of the number of fish, it can be quite difficult to estimate; thus simplifying assumptions are often made. See Hard et al. 2007.)

Demographic Data and Analyses

The data considered in this report include estimates of steelhead natural escapement, as calculated from index redd count statistics obtained from the Washington Department of Fish and Wildlife (WDFW). These data are for winter-run steelhead primarily (the sole summer-run exception is from the Tolt River) and date from as early as 1977. The 20 populations under analysis are considered by the TRT to be potential DIPs. However, sufficient quantitative abundance data were not available for other populations among the 32 total candidate DIPs. The steelhead DIPs for which insufficient quantitative data were available for analysis are: Drayton Harbor Winter Run, Nooksack River Winter Run, South Fork Nooksack River Summer Run, Nookachamps Creek Winter Run, Baker River Summer Run and Winter Run, Sauk River Summer Run and Winter Run, Canyon Creek Summer Run, Deer Creek Summer Run, North Fork Skykomish River Summer Run, North Lake Washington Tributaries Winter Run, South Puget Sound Tributaries Winter Run, and East Kitsap Peninsula Winter Run. Basic analyses of natural escapement data are in Table 1 through Table 3; these analyses focus on data from the entire time series, data since 1995, and data from the most recent 5 years.

Data from the Entire Series

Since 1977, Puget Sound steelhead abundance has shown a widespread declining trend over much of the DPS (Table 1). None of the 21 populations evaluated exhibit estimates of long-term population growth rate ($\lambda = R_0 = e^r$, where R_0 is the net birth rate and r is the intrinsic geometric growth rate) that are significantly positive; only two of these has a point estimate of $\lambda > 1$ (indicating positive population growth): Samish River winter run and Snohomish/ Skykomish Rivers Winter Run. In fact, only one estimate of population growth rate was significantly different from 1, indicating little evidence for population trend, but statistical power is limited by missing data and high variability. In general, the highest growth rates over the entire series were estimated in the Northern Cascades MPG, and the lowest rates were estimated in the Central and South Puget Sound MPG. Data from redd counts in the Nooksack River in 2010 and 2011 indicated abundances of 1901 and 1774 spawners, respectively, but no data series was available to estimate trend.

Data Since 1995

Puget Sound winter-run steelhead abundance since 1995 has also shown a widespread declining trend over much of the DPS (Table 2). Only one estimate of population growth rate was significantly different from 1, indicating little evidence for population trend, but the statistical power of those estimates was even lower in this shorter data set. For most populations, the point estimates of population growth are even more negative in this recent period. Only one of the 20 populations evaluated exhibit a point estimate of growth rate that was significantly positive (West Hood Canal Tributaries Winter Run).

Table 1. Estimates of exponential trend in the natural logarithm (ln) of natural spawners (λ) for DIPs of steelhead in the Puget Sound DPS over the entire data series. The Tahuya River winter run is part of the South Hood Canal Tributaries Winter-Run DIP and the Morse Creek winter run is part of the Strait of Juan de Fuca Tributaries DIP. Where necessary, data were interpolated between adjacent years (max. n = 3) for some populations to estimate the 95% CIs. NC = not calculated.

Population	**Exp. trend ln(nat. spawners) (95% CI)**
Northern Cascades (South Salish Sea) MPG	
Drayton Harbor Tributaries Winter Run	NC
Nooksack River Winter Run	NC
South Fork Nooksack River Summer Run	NC
Samish River winter run[a] (1979–2011)	1.031 (0.915–1.160)
Nookachamps Creek Winter Run	NC
Baker River Summer Run and Winter Run	NC
Sauk River Summer Run and Winter Run	NC
Skagit R. Summer Run and Winter Run (1978–2011)	0.997 (0.921–1.079)
Canyon Creek Summer Run	NC
Pilchuck River Winter Run (1981–2011)	0.984 (0.879–1.101)
Stillaguamish River Winter Run (1985–2011)	0.963 (0.879–1.055)
Deer Creek Summer Run	NC
Snohomish/Skykomish Rivers Winter Run (1981–2011)	1.012 (0.887–1.156)
North Fork Skykomish River Summer Run	NC
Snoqualmie River Winter Run (1981–2011)	0.967 (0.902–1.037)
Tolt River Summer Run (1985–2011)	0.973 (0.801–1.182)
Central and South Puget Sound MPG	
Cedar River Winter Run (1981–2011)	0.774 (0.592–1.010)
N. Lake Washington and Lake Sammamish Winter Run	NC
Green River Winter Run (1978–2011)	0.975 (0.885–1.074)
Puyallup/Carbon Rivers Winter Run (1983–2011)	0.939 (0.860–1.026)
White River Winter Run (1983–2011)	0.974 (0.888–1.068)
Nisqually River Winter Run (1980–2011)	0.938 (0.853–1.031)
South Puget Sound Tributaries Winter Run	NC
East Kitsap Peninsula Tributaries Winter Run	NC
Hood Canal and Strait of Juan de Fuca MPG	
South Hood Canal Tributaries Winter Run (1998–2011)	0.904 (0.431–1.896)
Tahuya River winter run[a] (1981–2011)	0.991 (0.877–1.121)
East Hood Canal Tributaries Winter Run (1981–2011)	0.985 (NC)
Skokomish River Winter Run (1982–2011)	0.975 (0.867–1.097)
West Hood Canal Tributaries Winter Run (1997–2011)	1.237 (1.083–1.393)
Dungeness R. Summer Run and Winter Run (1988–96)	0.935 (NC)
Strait of Juan de Fuca Tribs. Winter Run (1998–2010)	0.921 (0.501–1.692)
Morse Creek winter run[a] (1984–2010)	0.972 (0.907–1.042)
Elwha River Winter Run[b] (1986–1997)	0.864 (NC)

[a] One component of a DIP

[b] Historically, a summer run may have been present in the Elwha River; however, in the aftermath of dam construction in the early 1900s, a native summer run in the Elwha River basin may no longer be present. Further work is needed to distinguish whether existing feral summer-run steelhead are derived from introduced Skamania Hatchery (Columbia River) summer-run steehead.

Table 2. Estimates of exponential trend in the natural logarithm (ln) of natural spawners (λ) for DIPs of steelhead in the Puget Sound DPS since 1995. The Tahuya River winter run is part of the South Hood Canal Tributaries Winter-Run DIP and the Morse Creek winter run is part of the Strait of Juan de Fuca Tributaries Winter-Run DIP. Where necessary, data were interpolated between adjacent years (max. n = 3) for some populations to estimate the 95% CIs. NC = not calculated.

Population	**Exp. trend ln(nat. spawners) (95% CI)**
Northern Cascades (South Salish Sea) MPG	
Drayton Harbor Tributaries Winter Run	NC
Nooksack River Winter Run	NC
South Fork Nooksack River Summer Run	NC
Samish River winter run[a]	0.993 (0.592–1.666)
Nookachamps Creek Winter Run	NC
Baker River Summer Run and Winter Run	NC
Sauk River Summer Run and Winter Run	NC
Skagit River Summer Run and Winter Run	0.966 (0.494–1.891)
Canyon Creek Summer Run	NC
Pilchuck River Winter Run	0.928 (0.413–2.085)
Stillaguamish River winter-run	0.895 (0.729–1.098)
Deer Creek Summer Run	NC
Snohomish/Skykomish Rivers Winter Run	0.953 (0.220–4.123)
North Fork Skykomish River Summer Run	NC
Central and South Puget Sound MPG	
Cedar River Winter Run	0.678 (0.434–1.060)
N. Lake Washington and Lake Sammamish Winter Run	NC
Green River Winter Run	0.902 (0.703–1.157)
Puyallup/Carbon Rivers Winter Run	0.928 (0.700–1.229)
White River Winter Run	0.987 (0.794–1.228)
Nisqually River Winter Run	0.965 (0.615–1.514)
South Puget Sound Tributaries Winter Run	NC
East Kitsap Peninsula Tributaries Winter Run	NC
Hood Canal and Strait of Juan de Fuca MPG	
South Hood Canal Tributaries Winter Run	0.904 (0.431–1.896)
Tahuya River winter run[a]	0.974 (0.743–1.276)
East Hood Canal Tributaries Winter Run	0.985 (NC)
Skokomish River Winter Run	1.007 (0.720–1.408)
West Hood Canal Tributaries Winter Run	1.237 (1.083–1.393)
Sequim/Discovery Bays Tributaries Winter Run	0.887 (0.709–1.111)
Dungeness River Summer Run and Winter Run	NC
Strait of Juan de Fuca Tributaries Winter Run	0.921 (0.501–1.692)
Morse Creek winter run[a]	0.948 (0.830–1.083)
Elwha River Winter Run[b]	NC

[a] One component of a DIP.

[b] Historically, a summer run may have been present in the Elwha River; however, in the aftermath of dam construction in the early 1900s, a native summer run in the Elwha River basin may no longer be present. Further work is needed to distinguish whether existing feral summer-run steelhead are derived from introduced Skamania Hatchery (Columbia River) summer-run steehead.

Table 3. Geometric means of natural spawners for DIPs of steelhead in the Puget Sound DPS over the most recent 5 years of available data. The Tahuya River winter run is part of the South Hood Canal Tributaries Winter-Run DIP and the Morse Creek winter run is part of the Strait of Juan de Fuca Tributaries Winter-Run DIP. Where necessary, data were interpolated between adjacent years (max. missing n = 1) for some populations to estimate the 95% CIs. NC = not calculated.

Population	**Geometric mean (95% CI)**
Northern Cascades (South Salish Sea) MPG	
Drayton Harbor Tributaries Winter Run	NC
Nooksack River Winter Run (2010–2011)	1,820.8 (NC)
South Fork Nooksack River Summer Run	NC
Samish River winter run[a] (2007–2011)	581.3 (467.7–694.9)
Nookachamps Creek Winter Run	NC
Baker River Summer Run and Winter Run	NC
Sauk River Summer Run and Winter Run	NC
Skagit River Summer Run and Winter Run (2007–2011)	4,078.0 (3,578.2–4,577.8)
Canyon Creek Summer Run	NC
Pilchuck River Winter Run (2007–2011)	512.0 (390.0–634.1)
Stillaguamish River Winter Run (2007–2011)	301.3 (240.5–362.1)
Deer Creek Summer Run	NC
Snohomish/Skykomish Rivers Winter Run (2007–2011)	917.5 (785.3–1,049.7)
North Fork Skykomish River Summer Run	NC
Snoqualmie River Winter Run (2009–2011)	591.9 (NC)
Tolt River Summer Run (2007–2011)	70.7 (58.4–82.9)
Central and South Puget Sound MPG	
Cedar River Winter Run (2007–2011)	1.9 (0.6–3.2)
North Lake Washington and Lake Sammamish Winter Run	NC
Green River Winter Run (2007–2011)	660.3 (456.6–864.1)
Puyallup/Carbon Rivers Winter Run (2007–2011)	394.2 (340.9–447.6)
White River Winter Run (2007–2011)	366.2 (275.9–456.5)
Nisqually River Winter Run (2007–2011)	367.8 (280.4–455.2)
South Puget Sound Tributaries Winter Run	NC
East Kitsap Peninsula Tributaries Winter Run	NC
Hood Canal and Strait of Juan de Fuca MPG	
South Hood Canal Tributaries Winter Run (2007–2011)	101.8 (74.8–128.9)
Tahuya River winter run[a] (2007–2011)	84.3 (58.3–110.4)
East Hood Canal Tributaries Winter Run (2007–2011)	30.1 (15.5–44.7)
Skokomish River Winter Run (2007–2011)	407.9 (359.5–456.3)
West Hood Canal Tributaries Winter Run (2007–2011)	181.7 (141.4–222.0)
Sequim/Discovery Bays Tributaries Winter Run (2007–2011)	17.1 (11.4–22.9)
Dungeness River Summer Run and Winter Run (1992–1996)[b]	304.0 (289.4–318.6)
Strait of Juan de Fuca Tributaries Winter Run (2006–2010)	176.4 (133.8–219.0)
Morse Creek winter run[a] (2007–2011)	56.9 (NC)
Elwha River Winter Run (1993–1997)[b]	125.1 (82.0–168.1)

[a] One component of a DIP.

[b] These are the most recent 5 years of data.

Data from the Most Recent 5 Years

Over the most recent 5 years where data were available, Puget Sound winter-run steelhead abundance has been low over much of the DPS, with a geometric mean less than 250 fish annually for eight of the 20 populations evaluated (Table 3). One of these is in the Northern Cascades MPG (Tolt River Summer Run), one is in Central and South Puget Sound MPG (Cedar River Winter Run), and six winter-run DIPS are in the Hood Canal and Strait of Juan de Fuca MPG (South Hood Canal Tributaries, East Hood Canal Tributaries, West Hood Canal Tributaries, Sequim/Discovery Bays Tributaries, Strait of Juan de Fuca Tributaries, and Elwha River). Only seven populations had a geometric mean greater than 500 fish—Nooksack River, Samish River, Skagit River, Pilchuck River, Snohomish River/Skykomish Rivers, Snoqualmie River, and Green River winter runs—and all but one of these are in the Northern Cascades MPG. The Cedar River Winter-Run population is extremely low in abundance. Eleven populations have no quantitative abundance estimates, and an additional two—Dungeness River Summer Run and Winter Run and Elwha River Winter Run—have no recent quantitative estimates.

Historically, a summer run may have been present in the Elwha River; however, it is possible that the run was extirpated or residualized when the Elwha and Glines Canyon dams were constructed in the early 1900s at RKM 7.9 and RKM 21.6, respectively. Although summer-run steelhead have been observed in the pools below the Elwha Dam in recent years, it is most probable that these fish are the product of nonnative Skamania Hatchery summer-run steelhead releases into the Elwha River. The historical distribution of Elwha River summer-run steelhead is unknown. Further study is required to establish whether there is any legacy of a native summer run above or below the recently removed dams.

Collectively, these data indicate that several populations are at low abundance (13 of 20 populations with fewer than 500 spawners annually) and suggest that nearly all populations exhibit declining trends (low power and high variability precludes detection of clear trends in many cases using these simple measures). Natural escapement of winter-run steelhead throughout Puget Sound is currently far below historical estimates, particularly in southern Puget Sound and on the Olympic Peninsula.

Population Viability Analysis

The abundance trends and numbers described above alone do not always give a clear picture of population viability. Small salmonid populations can be viable if not trending downward. In addition, estimates of population growth rate and annual trend for steelhead, based largely on redd counts of a mixture of wild and feral hatchery fish, may be biased because of the extended freshwater life history of these fish, and due to density-dependent ecological interactions between hatchery and wild fish on spawning or rearing grounds. Finally, these data address only two of the four VSP criteria. Evaluating the viability of populations or aggregates of populations that are perceived to be at high risk of extinction—usually the case with threatened and endangered species—is typically challenging. Such populations are often characterized by meager data on abundance or productivity. Ideally, when such data do exist, a PVA, complete with estimates of extinction risk and minimum viable population sizes, will be conducted. But even if a detailed PVA is not done, an attempt to describe the population's likely future trajectory is crucial to identifying population status and recovery options. Identifying a population or conservation unit's current state and likely future state is valuable in informing managers about how much improvement in abundance and productivity is necessary to arrest a declining trend (Holmes 2001). Such an exercise can be instrumental in helping to identify priorities among different populations with different trends and to quantify the short-term risks until better information is acquired. A now widely used estimator for population extinction was proposed by Dennis et al. (1991), in which treating a time series of abundance as a diffusion process can yield estimates of extinction risk. A key advantage of this approach is that it requires only two parameters to estimate the state of the population into the future based on past censuses, namely, the arithmetic mean and variance of the log population growth rate; this variance accounts for sources of variability of abundance, including environmental and demographic stochasticity and observation error (Dennis et al. 1991). The approach makes analysis of time series tractable, in part because it treats a population's trajectory as though it is at stochastic equilibrium, subject primarily to random variation, with a state at a particular time little influenced by previous states (especially as the interval increases). However, this method has difficulty capturing the ability of populations at high risk of extinction to recover from very low levels.

Evaluating the viability of DIPs of steelhead was made challenging by the lack of quantitative abundance and productivity data for many of them. Additionally, some data sets describe multiple DIPs within a single basin. For DIPs that had quantitative or semiquantitative abundance estimates, we employed a combination of viability curves developed from stock-recruitment analysis and autoregressive state-space models to analyze population viability. Viability curves such as those depicted in Figure 1 are useful in illustrating the current extinction

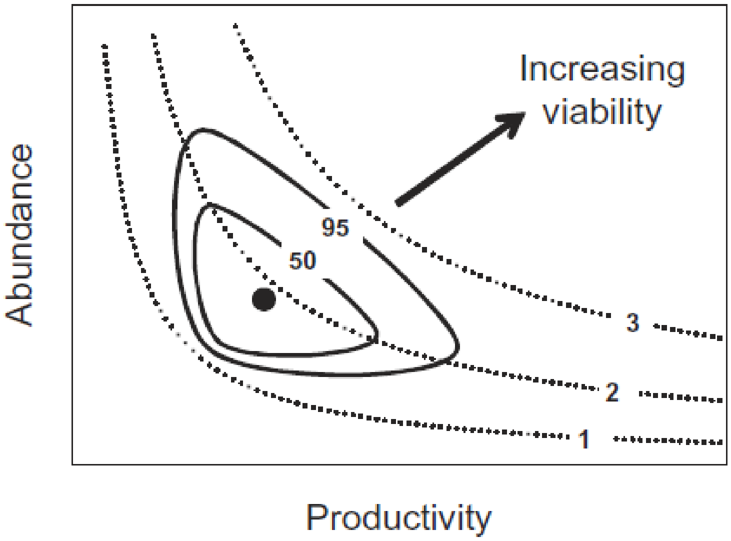


Figure 1. Viability as a function of abundance and productivity. The dotted curves depict isopleths of equivalent extinction risk along different combinations of abundance and productivity and corresponding to different viability categories (1, 2, or 3). For a given abundance, viability increases with increasing productivity; conversely, for a given productivity, viability increases with increasing abundance. The black dot and solid lines estimate the current risk of extinction and its 50% and 95% confidence ellipses for a hypothetical population. (Adapted from McElhany et al. 2007 and Waples et al. 2010.)

or quasi-extinction risk of a population under recent conditions of abundance and productivity. They require a demographic model to forecast future population viability from available trends in abundance and productivity.

For 20 of the 32 steelhead DIPs, we used viability curves to characterize the relationship between population abundance, productivity, and quasi-extinction risk (or conversely, the probability of persistence; see Table 4 and Table 5). DIPs for which we could not conduct such analyses included Drayton Harbor Tributaries Winter Run, Nooksack River Winter Run, South Fork Nooksack River Summer Run, Sauk River Summer Run and Winter Run, Baker River Summer Run and Winter Run, Canyon Creek Summer Run, Nookachamps Creek Winter Run, North Fork Skykomish River Summer Run, Deer Creek Summer Run, South Puget Sound Tributaries Winter Run, and East Kitsap Peninsula Tributaries Winter Run. The viability curve approach was developed in an effort to establish recovery criteria for threatened salmon and steelhead populations and was first described in McElhany et al. (2003). The points that describe a viability curve reflect combinations of abundance and productivity that generate the same probability of reaching or exceeding a QET (Figure 2). The QET is established for each

Table 4. Population persistence categories (McElhany et al. 2006).

Population category	Probability of persistence	Probability of extinction	Description of extinction risk in 100 years
0	0–40%	60–100%	Either extinct or very high risk
1	40–75%	25–60%	Relatively high risk
2	75–95%	5–25%	Moderate risk
3	95–99%	1–5%	Low (“negligible”) risk (VSP)
4	>99%	<1%	Very low risk

Table 5. Population abundance (number of spawners) relative to persistence category and population size category (after McElhany et al. 2007). The Chinook salmon (*Oncorhynchus tshawytscha*) data were used to construct the viability curves in Figure 2.

Species	Size category	Persistence category 0	1	2	3	4
Chum salmon	Small	<250	250–300	300–500	500–1,000	>1,000
(*O. keta*)	Medium	<500	500–600	600–700	700–1,000	>1,000
	Large	<700	700–850	850–1,000	1,000–1,200	>1,200
Chinook salmon	Small	<100	100–200	200–500	500–1,000	>1,000
	Medium	<350	350–450	450–600	600–1,000	>1,000
	Large	<600	600–750	750–1,000	1,000–1,300	>1,300
Coho salmon	Small	<500	500–700	700–1,000	1,000–1,300	>1,300
(*O. kisutch*)	Medium	<1,000	1,000–1,400	1,400–2,000	2,000–2,400	>2,400
	Large	<1,600	1,600–2,000	2,000–3,000	3,000–3,600	>3,600
Steelhead	Small	<100	100–200	200–500	500–750	>750
	Medium	<200	200–250	250–500	500–1,000	>1,000
	Large	<400	400–450	450–500	500–1,000	>1,000

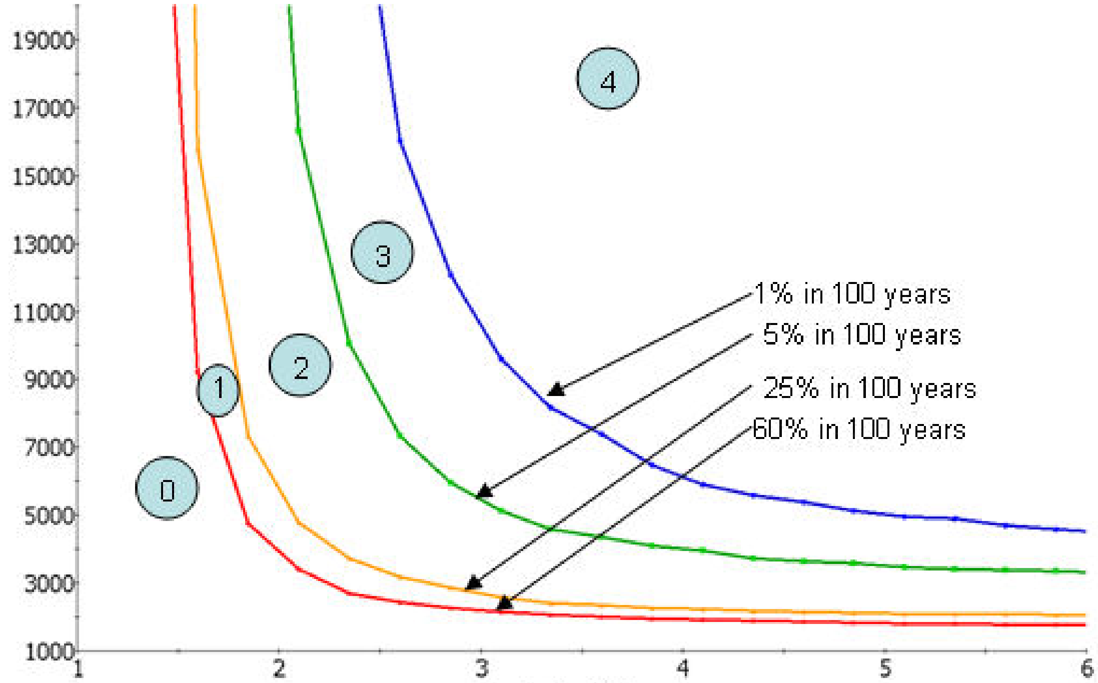


Figure 2. Hypothetical viability curves showing the relationship between levels of extinction risk and population persistence categories. The example is based on the Chinook salmon values in Table 5. Each of the curves indicates a different risk level corresponding to a combination of abundance (y-axis) and productivity (x-axis, here as recruits per spawner). The numbers in the circles are the persistence categories associated with each region of the plot (i.e., the area between adjacent curves). For example, a population with a risk category of 0 is described as a population that is nearly extinct and a population with a risk category of 3 or 4 is described as viable (Table 4).

population by first considering the potential of the freshwater habitat to support steelhead rearing and spawning, then applying a minimal marine survival rate to estimates of smolt production that would produce an abundance of spawners considered to be too low for sustained viability. Populations with productivity and abundance combinations above (to the right of) the curve have a lower extinction risk than those along the curve, while those below (to the left of) the curve have a higher risk.

Relating abundance, productivity, and extinction risk is accomplished using a simulation model with a stochastic recruitment function having terms for productivity, carrying capacity, recruitment variability, age structure, future harvest rate, and a reproductive failure threshold. To estimate extinction risk for any particular set of input parameters, the model is run thousands of times to determine the fraction of simulations that drop below a critical risk threshold. A viability curve is drawn by determining combinations of productivity and capacity (abundance) that are associated with a given level of risk. Drawing the curve for any particular group of fish requires appropriate estimates of recruitment variability, age structure, future harvest rate, and reproductive failure threshold. Note that the curve is not calculated from empirical estimates of productivity and abundance—instead, a range of hypothetical abundances and capacities (abundances) are explored to produce a curve. The viability curve can be thought of as a target for population abundance and productivity to achieve a certain level of extinction risk; it is not a comprehensive evaluation of population status (McElhany et al. 2003). We apply it to Puget Sound steelhead to evaluate status relative to desirable levels of abundance and productivity and compare these results to other methods to assess population viability (described in the Population Viability Analysis with Autoregressive State-Space Models subsection below).

In order to evaluate a particular population's status relative to a viability curve estimate, the population's abundance and productivity must be estimated. We used the MeanRS method (McElhany et al. 2006) to estimate these parameters. Productivity is a measure of a population's resilience or tendency to return to higher abundance if the population declines to low abundance. Using the MeanRS method, this tendency is estimated as the geometric mean recruits per spawner for the broodyears with the lowest half of spawner abundances. The abundance is estimated as the geometric mean recruitment over the time series. The characteristics of the MeanRS method compared to other possible approaches are described in McElhany et al. (2006). The MeanRS method is solidly based on the empirical data because it does not depend on extrapolation outside the observed ranges of recruitment and abundance.

Estimating a population's abundance and productivity requires input data on population spawner abundance, the fraction of hatchery origin spawners, harvest rates, and the population age structure. All of these parameters are estimated with error, sometimes considerable error. This error is incorporated into the analysis by using a Monte Carlo approach of simulating many equally plausible data sets based on our understanding of the measurement errors, then calculating the MeanRS output for each simulated data set. This gives a distribution of possible abundance and productivity combinations for the current state of the population, which can be depicted in the form of probability contours. We used the Salmon Population AnalyZer (SPAZ) computer program, version 1.3.4, to generate viability curves and current status probability contours (McElhany and Payne 2006).

Stochastic Population Viability Analysis

We used a stochastic population viability model incorporated into SPAZ to characterize viability of populations for which we had temporal estimates of relative abundance (typically, number of spawners estimated from redd counts), using estimates of population abundance and productivity under different demographic models. SPAZ can estimate extinction risk, minimum viable population size, and population change criteria. The program estimates stock-recruitment parameters from spawner-recruit data and incorporates environmental stochasticity, in the form of random uncorrelated environmental variation, and parameter uncertainty, in the form of random variation in parameter estimates. These estimates can be obtained in SPAZ using any of a variety of stock-recruitment models, including constant recruitment, random walk (with or without drift), stochastic exponential (with ceiling), stochastic hockey stick, running sum growth, stochastic (logistic) Ricker, and Beverton-Holt models (Myers et al. 1994, Barrowman and Myers 2000).

The SPAZ program has the capability of incorporating observation error and process error into the analyses. Observation error, assumed to be normally distributed in SPAZ, is measurement error that contributes to uncertainty in estimating abundance at any given time, but it does not accumulate or contribute to future uncertainty. Process error, on the other hand, is random error that contributes to temporal variation in population dynamics, but is not directly accounted for by the model. It can result from a number of sources, including genetic or demographic stochasticity (sampling effects) or environmental stochasticity (random environmental variation). It can accumulate and contribute to future uncertainty. In SPAZ it is treated with a lognormal distribution and is considered temporally autocorrelated.

We used SPAZ to generate estimates of relative risk of a population reaching a low-abundance threshold, termed a QET, under simple stock-recruitment models such as the random walk with drift and the stochastic hockey stick. Using these risk estimates, we generated viability curves for steelhead populations that depict the risk of these populations reaching a particular QET within 100 years, given a range of particular combinations of current abundance and population growth rate. An example of the results of this type of PVA is in Figure 3, showing how the joint estimate of population abundance and productivity, depicted by the colored probability contours at lower left, relates to a series of four viability curves representing different levels of abundance and productivity that correspond to the projections of the population reaching a QET (e.g., QET = 2, 20, 50, or 100 spawners in a given year) within 100 years with 5% probability.

Of the 32 DIPs of Puget Sound steelhead the PSS TRT members identified (Table 1), we applied PVA to 16 of them for which we had sufficient abundance time series (Figure 4 through Figure 25). For each of these PVAs we made the following assumptions:

1. We based these PVAs on available natural escapement data, which were available for 20 of the 32 DIPs for most years from 1977 to 2011 (with some notable exceptions), as too few populations provided total run size (harvest plus escapement) information. Such PVAs will tend to underestimate extinction risk if temporal declines in total run size are steeper than those for escapements. Additional characteristics of the escapement data are briefly summarized in Table 6.
2. We had empirical age structure data available for only three populations in the Puget Sound DPS. We used the two data sets that were longest to determine a fixed average age structure for each Puget Sound DIP and apply it across its abundance time series. For DIPs in northern and southern Puget Sound and Hood Canal drainages, we used the estimated average age structure from the Green River Winter-Run DIP, available from a WDFW 28-year time series between 1978 and 2005. The age structure applied was 0.1% age-2, 5.8% age-3, 49.8% age-4, 45.3% age-5, 4.0% age-6, and 0.3% age-7 adults. For DIPs in Strait of Juan de Fuca drainages, we used the estimated average age structure from the Snow Creek winter-run steelhead population, available from a WDFW 34-year time series between 1977 and 2011. This age structure was similar, except that more age-4 and fewer age-5 fish are represented (0.2% age-2, 6.3% age-3, 70.2% age-4, 21.9% age-5, 1.4% age-6, and <0.1% age-7 adults). We applied normally distributed random error around these estimates.
3. For most of the DIPs, we applied a running sum growth model (McElhany and Payne 2006) to initiate the analysis of abundance and productivity, which relied on 4-year running sums of spawner counts. We fitted a hockey stick recruitment model (Barrowman and Myers 2000) to these data. The estimation of both the growth parameter (μ) and the process error (σ^2) using the slope method (Holmes 2001) was done

on the (corrected) running sums. A MeanRS bootstrap method was then used to calculate recruits per spawner, a growth curve was fit to the data, and population projections were used to estimate extinction risk for a set of four fixed QETs. For Dungeness River Summer Run and Winter Run, Strait of Juan de Fuca Tributaries Winter Run, and Elwha River Winter Run, we relied on a simpler, random-walk-with-trend model to estimate minimum viable population sizes from their shorter time series of abundance.

4. We did not account for differences in reproductive fitness between hatchery and natural spawners (see discussion in Araki et al. 2008), but simply assumed that all adults identified as natural spawners in the data were of wild origin.
5. We ran 100,000 bootstrap replicates using estimates of intrinsic growth rate, maximum number of recruits, and process error. These were drawn from uniform distributions with ranges of 0.5–4.0, 1–30, and 0.2–4.0, respectively.
6. We ran 1,000 risk projections for several QET values that always included 1 and a habitat-based estimate of QET that we derived from our intrinsic potential (IP) metrics (with 5% probability of reaching QET over a 100-year time horizon).
7. We did not account for nonanadromous (resident) recruits of steelhead spawners. For example, precocious male offspring of steelhead are known to participate in steelhead spawning and sire anadromous offspring. Most of the empirical escapement data we used are based on redd counts multiplied by a uniform spawners per redd number, which may or may not adequately account for resident spawners. Not accounting for resident recruits may underestimate productivity and potentially lead to overestimates of extinction risk.

The results of the SPAZ analyses indicate that two of the seven populations that could be evaluated in the Northern Cascades MPG are at moderate to high levels of risk of reaching QET. The two populations at moderate to high quasi-extinction risk in this MPG—Stillaguamish River Winter Run and Tolt River Summer Run—are at risk primarily because of high variance in productivity. For the Central and South Puget Sound MPG, the SPAZ analyses indicate that a much higher fraction of populations are at high risk: four of the five populations that could be evaluated are at high levels of risk of reaching QET. The four winter-run populations at high quasi-extinction risk in this MPG—Cedar River, Puyallup/Carbon Rivers, White River, and Nisqually River—are at risk primarily because of very low productivity (the Cedar River and Lake Washington populations are also at very low abundance). For the Hood Canal and Strait of Juan de Fuca MPG, the SPAZ analyses indicate that seven of the eight populations that could be evaluated are at moderate to high levels of risk of reaching QET. The five winter-run populations clearly at high quasi-extinction risk in this MPG—South Hood Canal Tributaries, East Hood Canal Tributaries, West Hood Canal Tributaries, Skokomish River, and Sequim/Discovery Bays Tributaries—are at risk primarily because of low productivity. Two others—Dungeness River Summer Run and Winter Run and Elwha River Winter Run—are also at low or unknown recent abundance. Because the abundance data used in the SPAZ analyses are generally indexes of natural escapement rather than reasonably precise estimates of natural run size, these analyses are likely to underestimate risk to viability from low abundance or productivity. The conclusions are further tempered by the rather restrictive assumptions described above, which may not apply to all these populations except in broad terms.

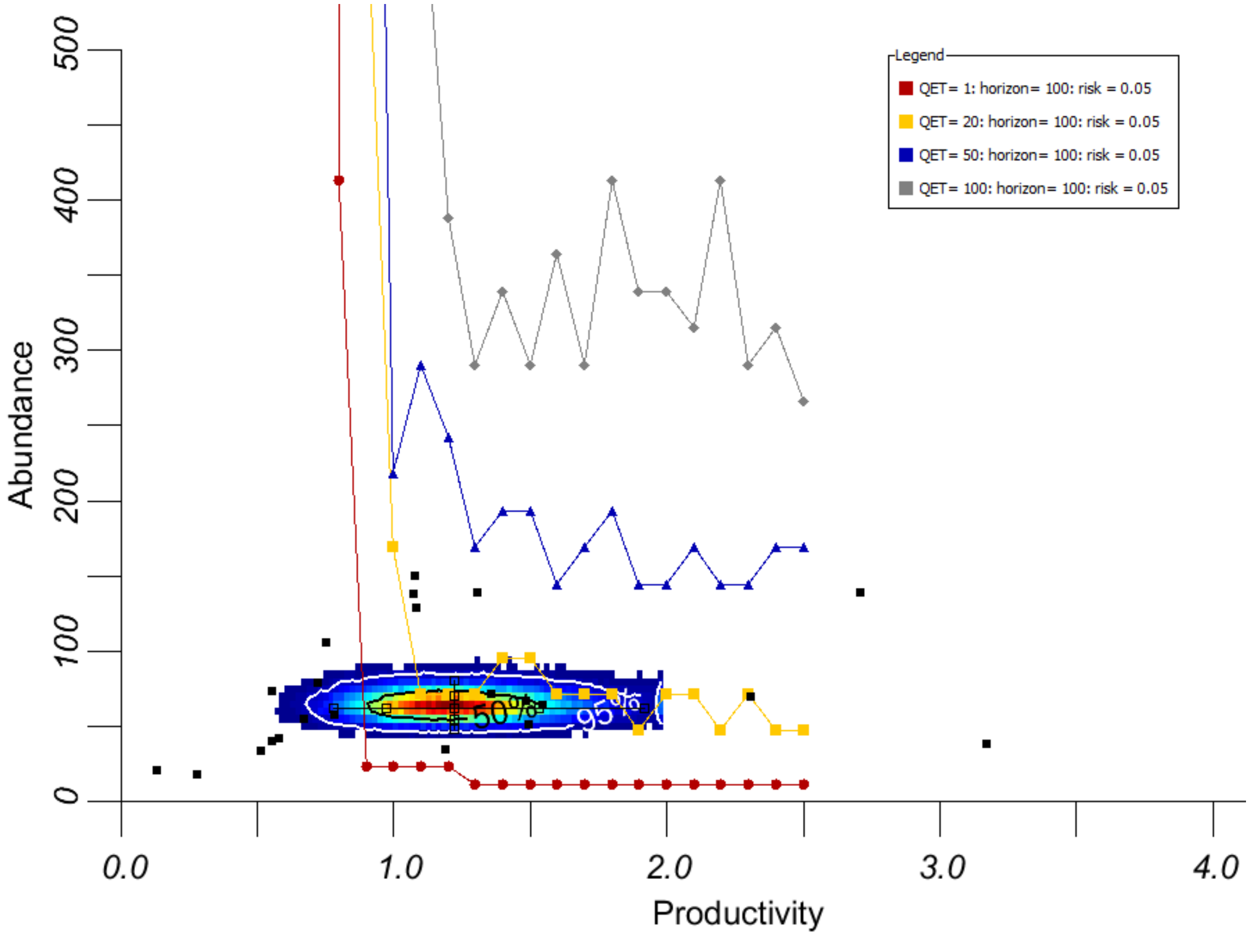


Figure 3. An example of the results of a PVA conducted with the program SPAZ (McElhany and Payne 2006). The plot depicts a joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and estimated productivity data are given by the black squares (for the wild Snow Creek winter-run steelhead population in Washington State, 1978–2011). The joint estimates and the data are portrayed in relation to four viability curves representing different combinations of abundance and productivity that correspond to the projections of the population reaching a QET (equal to 2, 20, 50, or 100 spawners in a given year) within 100 years with 5% probability. Assumptions of this PVA include no harvest and stationarity of conditions into the future. The PVA relies on a hockey stick recruitment model and uses the geometric mean of recruits per spawner to estimate productivity over the observed data. The bootstrapped estimates incorporate a range of uniform priors for population growth rate, number of recruits, and process error. Comparison with the values in Table 5 and Figure 2 suggest that this population is at a high risk of quasi-extinction over the next 100 years, and that this risk is posed by both low abundance and high variance in productivity.

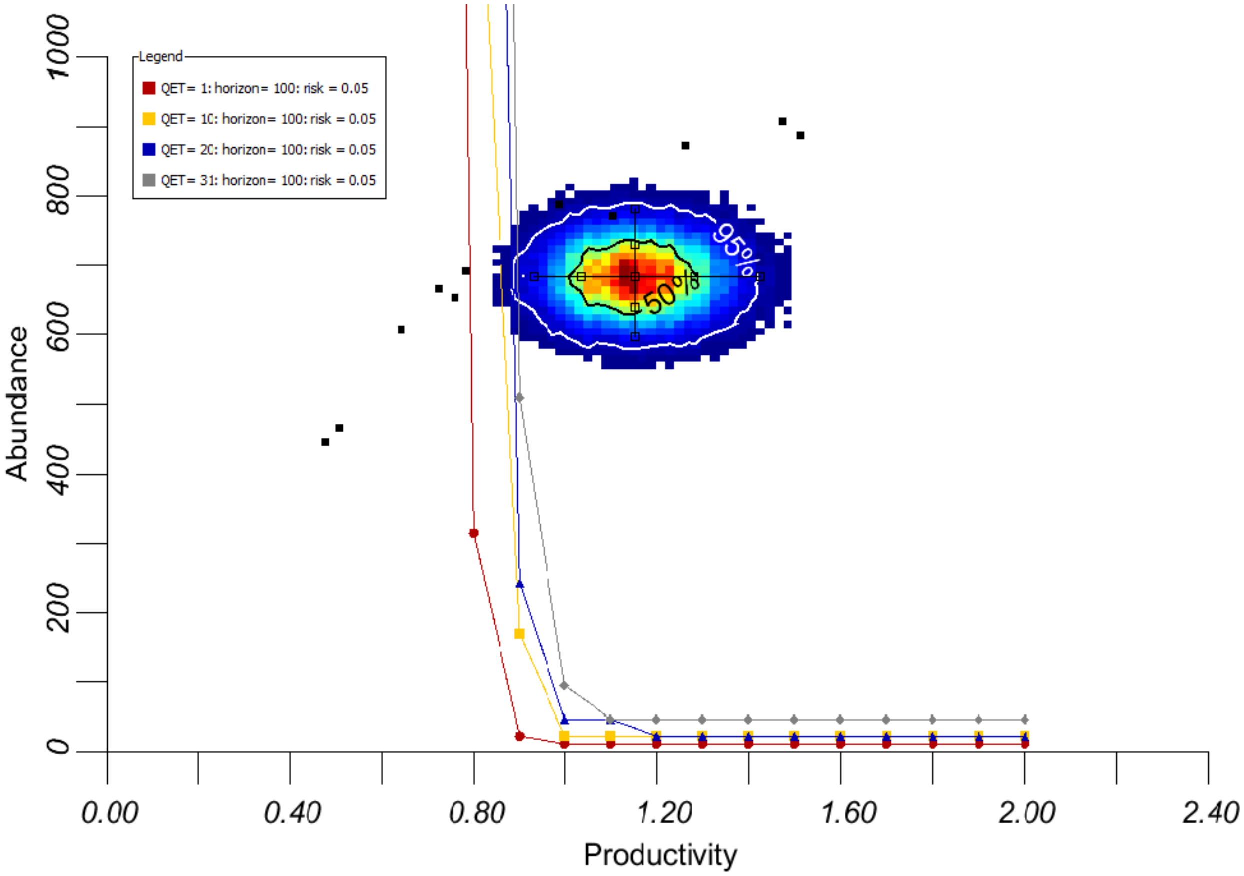


Figure 4. PVA for the Samish River and Bellingham Bay Tributaries Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto four viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, 20, or 31 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a low risk of quasi-extinction over the next 100 years.

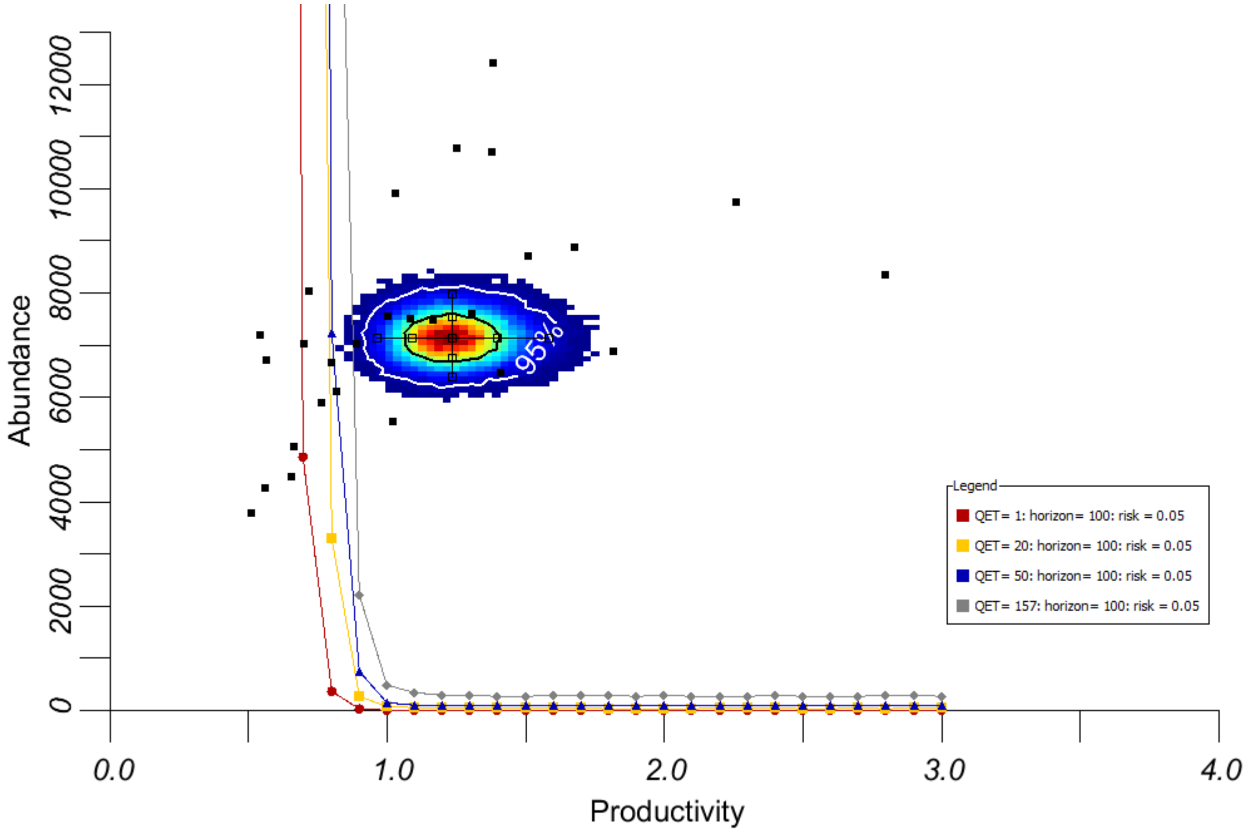


Figure 5. PVA for the Skagit River Summer Run and Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto four viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 20, 50, or 157 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a low risk of quasi-extinction over the next 100 years.

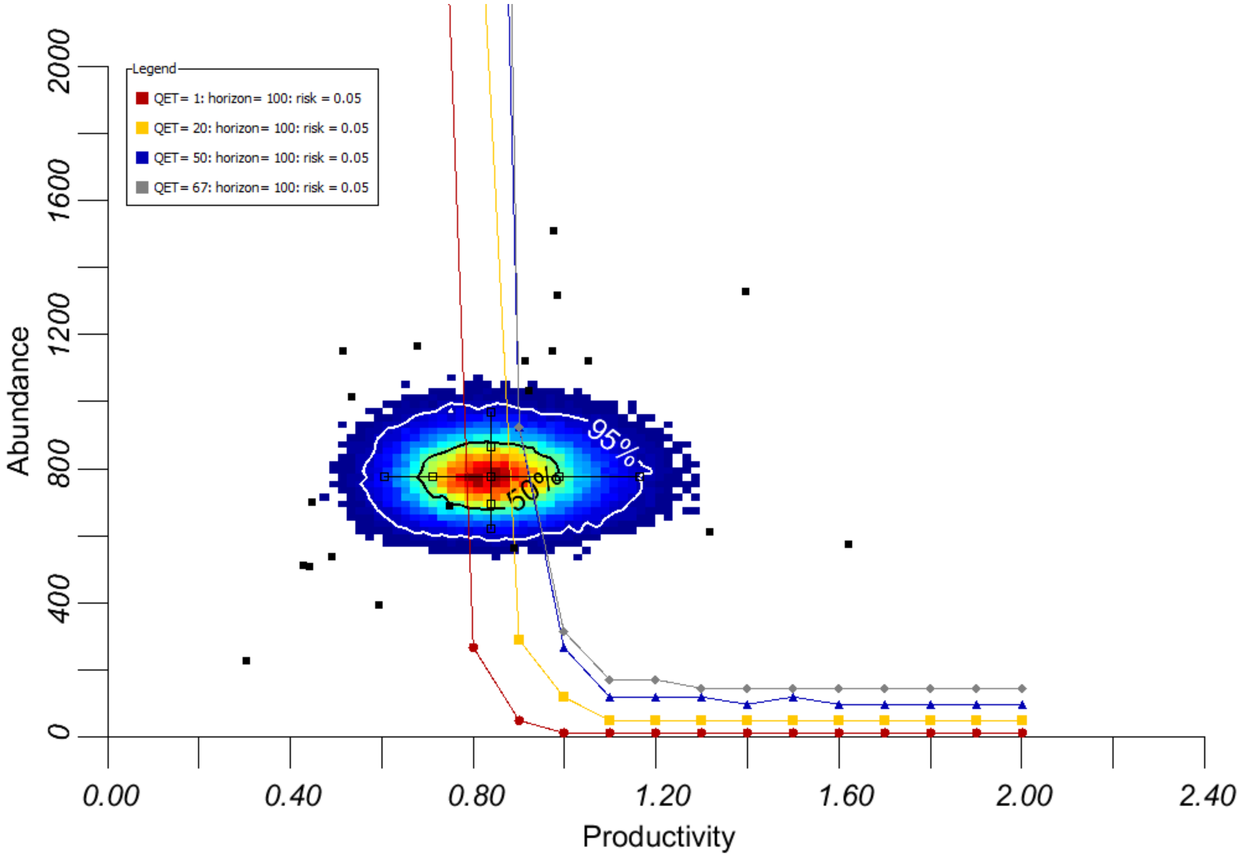


Figure 6. PVA for the Stillaguamish River Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto four viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 20, 50, or 67 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a very high risk of quasi-extinction over the next 100 years.

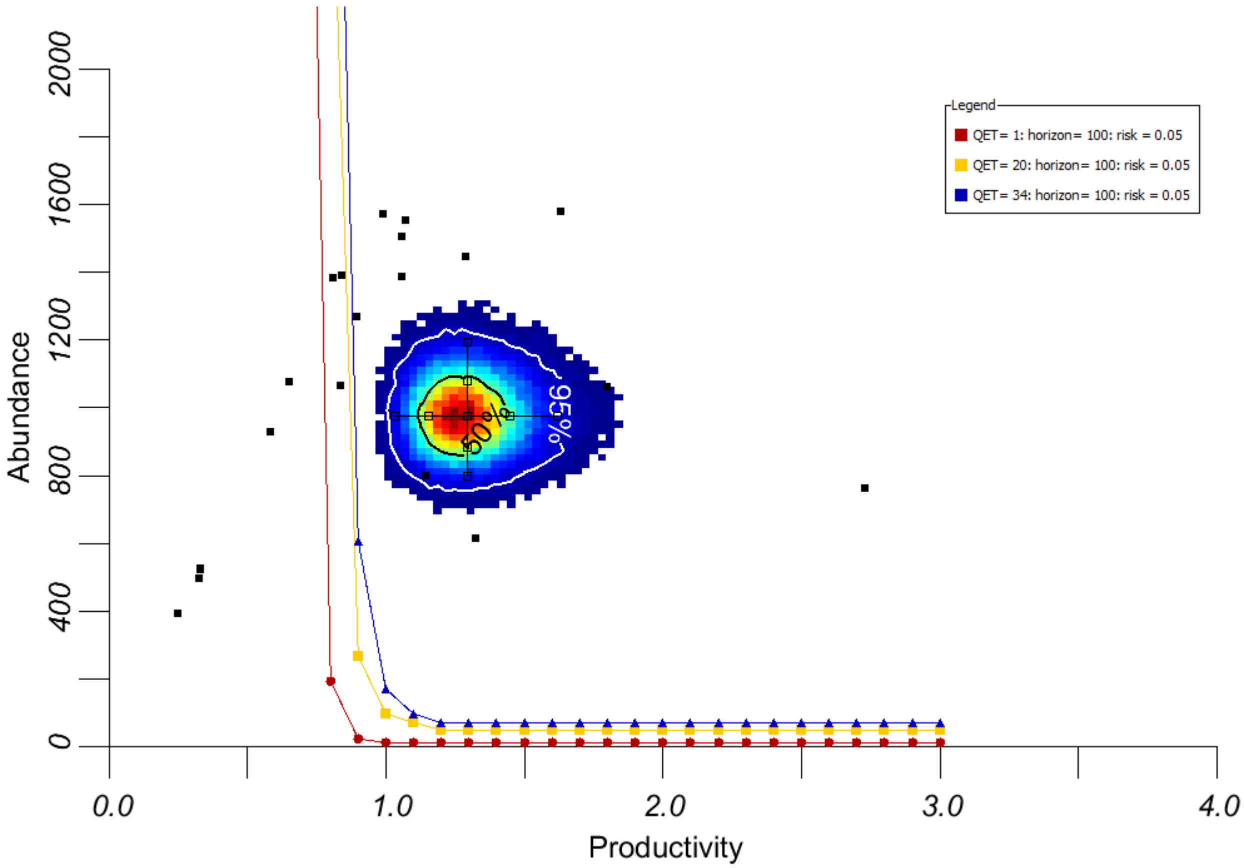


Figure 7. PVA for the Pilchuck River Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto three viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 20, or 34 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a very low risk of quasi-extinction over the next 100 years.

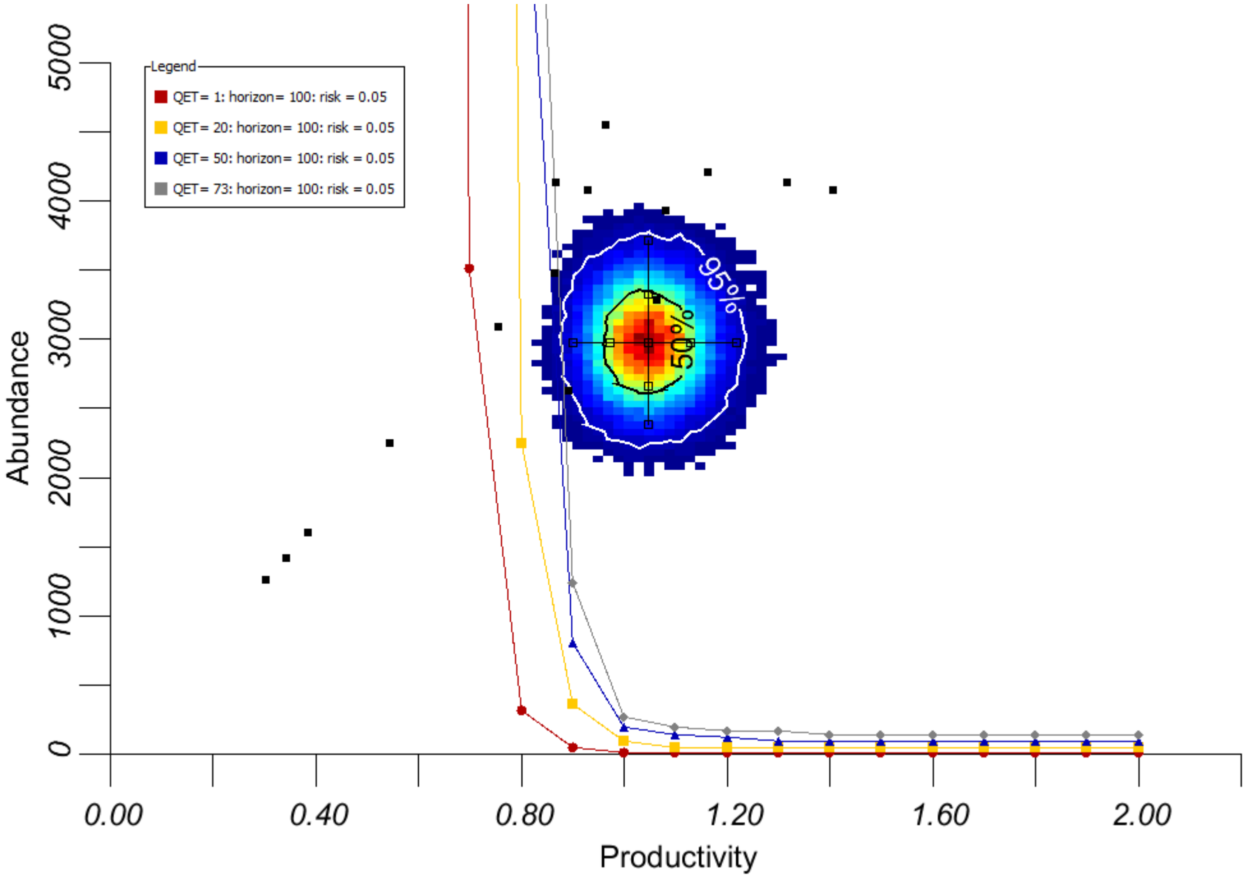


Figure 8. PVA for the Snohomish/Skykomish Rivers Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto four viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 20, 50, or 73 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a low risk of quasi-extinction over the next 100 years.

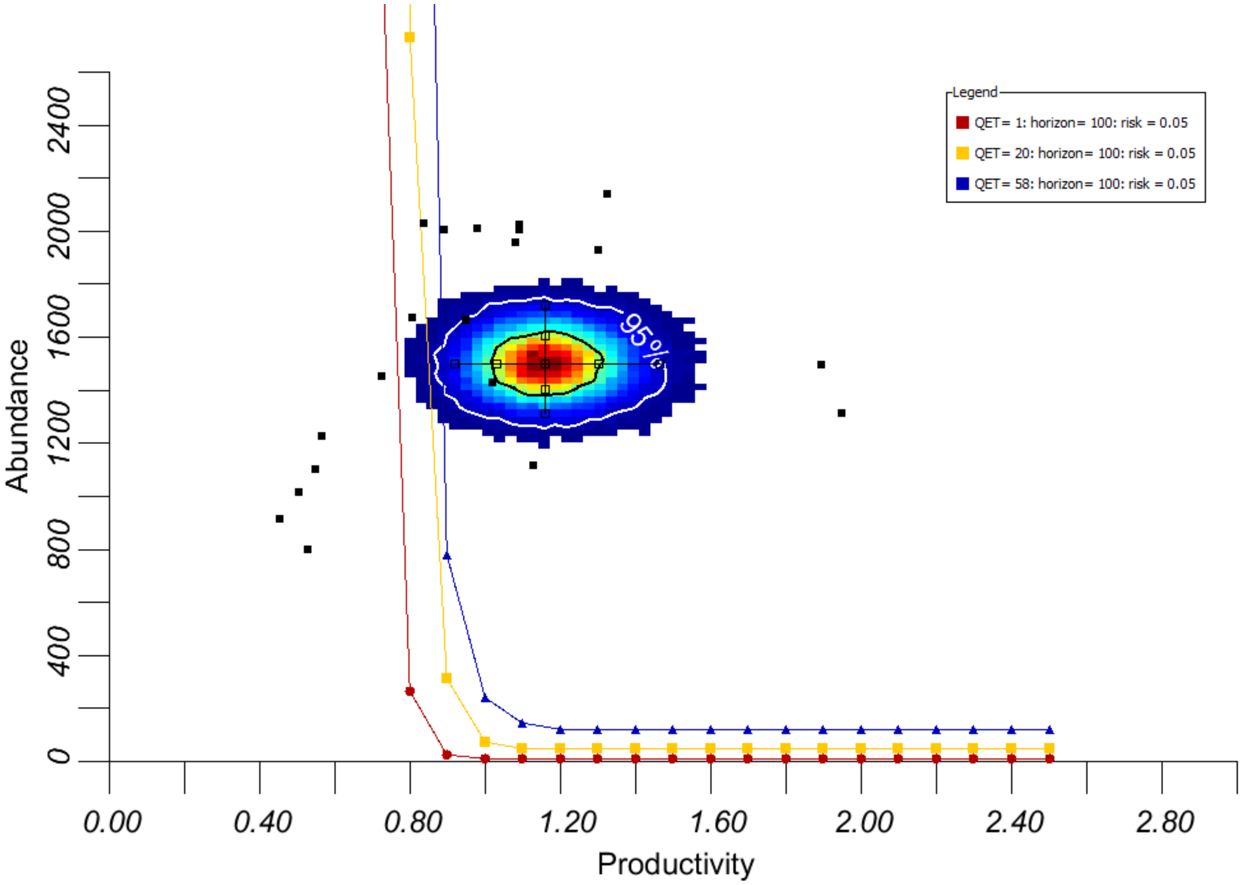


Figure 9. PVA for the Snoqualmie River Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto three viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 20, or 58 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a moderate risk of quasi-extinction over the next 100 years.

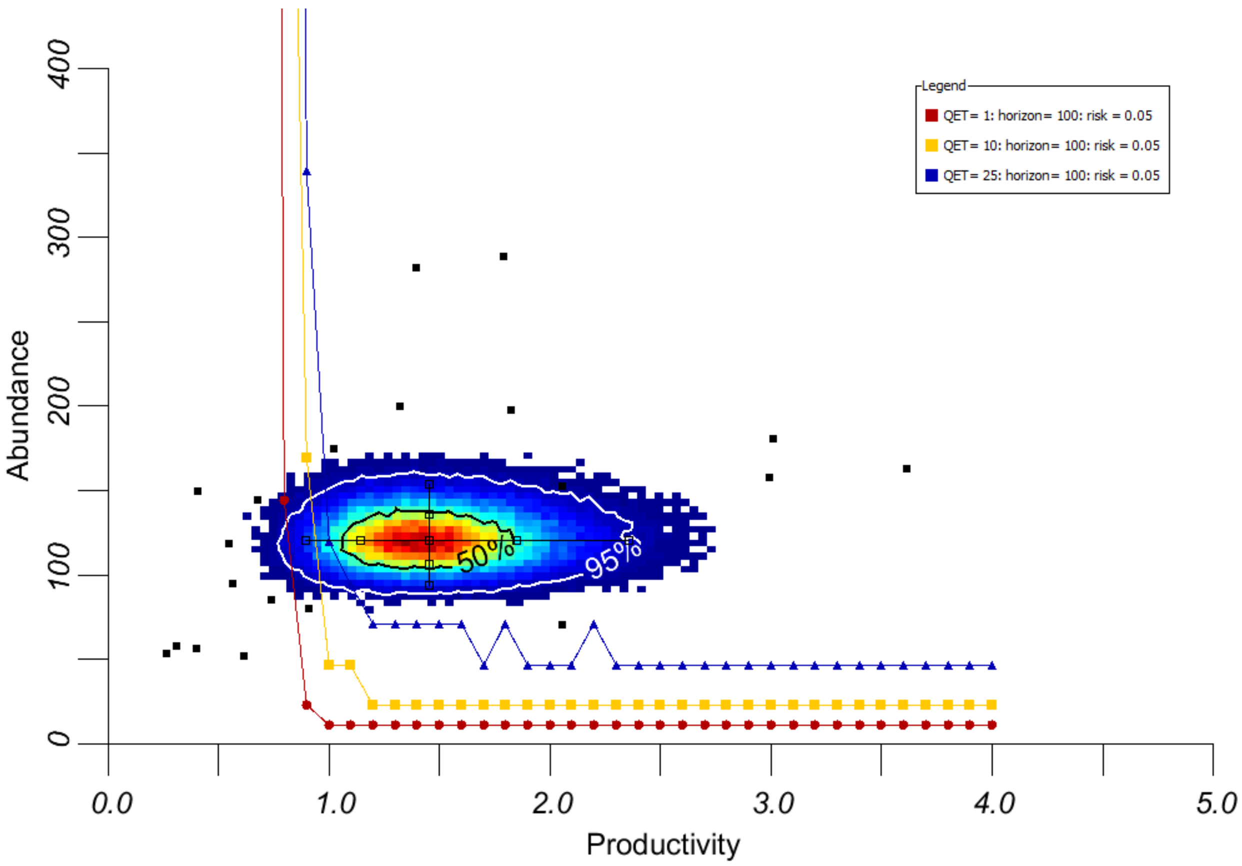


Figure 10. PVA for the Tolt River Summer Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto three viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, or 25 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a high risk of quasi-extinction over the next 100 years.

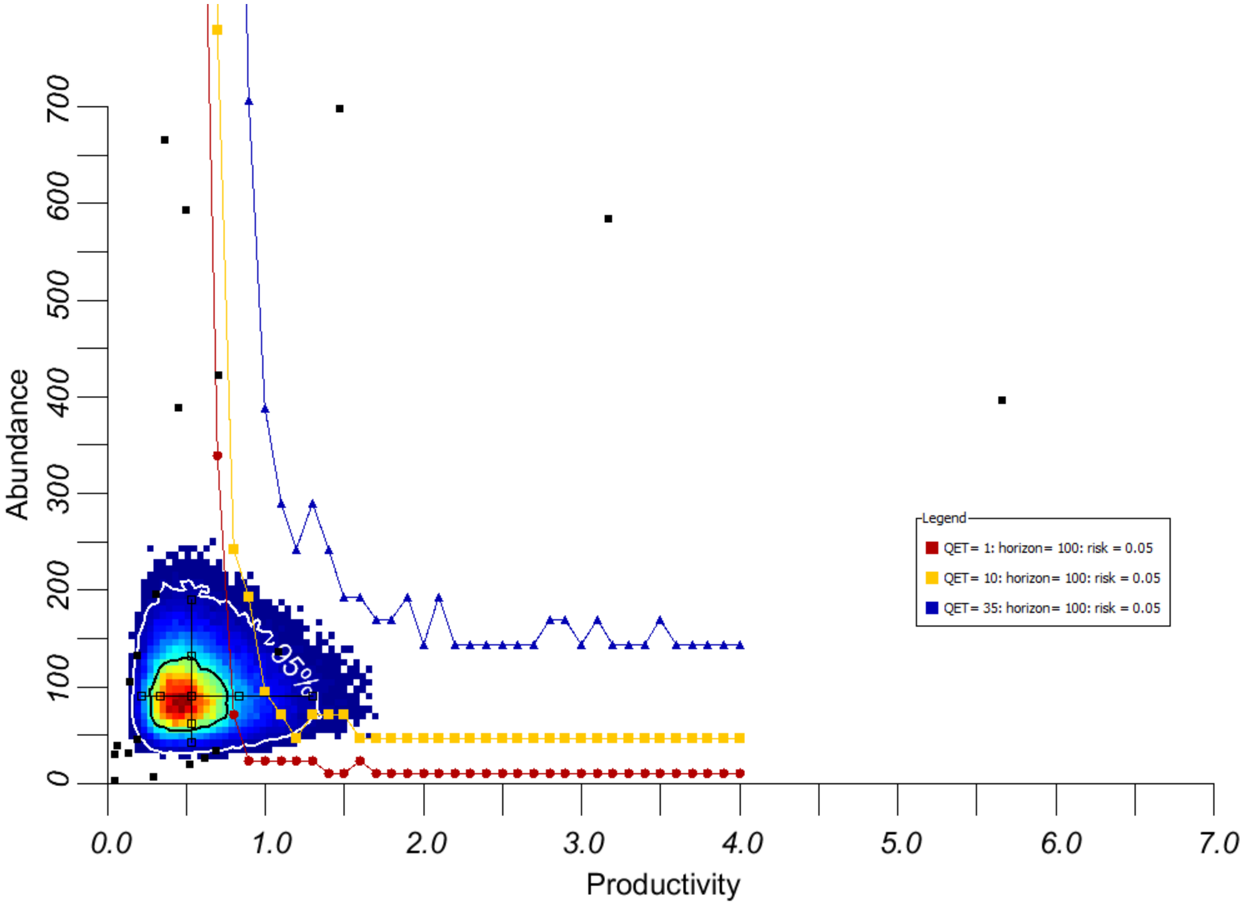


Figure 11. PVA for the Cedar River Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto three viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, or 35 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a very high risk of quasi-extinction over the next 100 years.

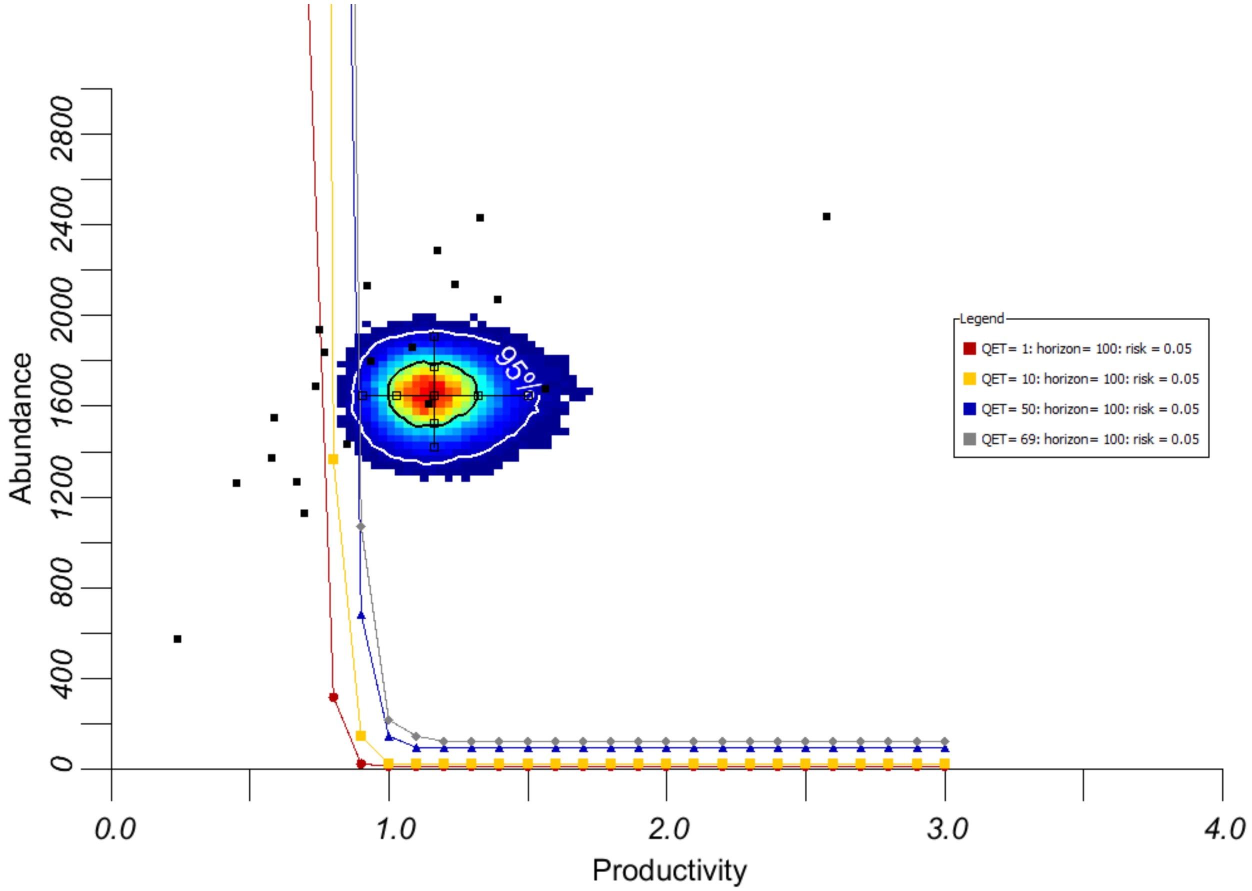


Figure 12. PVA for the Green River Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto four viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, 50, or 69 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a moderate risk of quasi-extinction over the next 100 years.

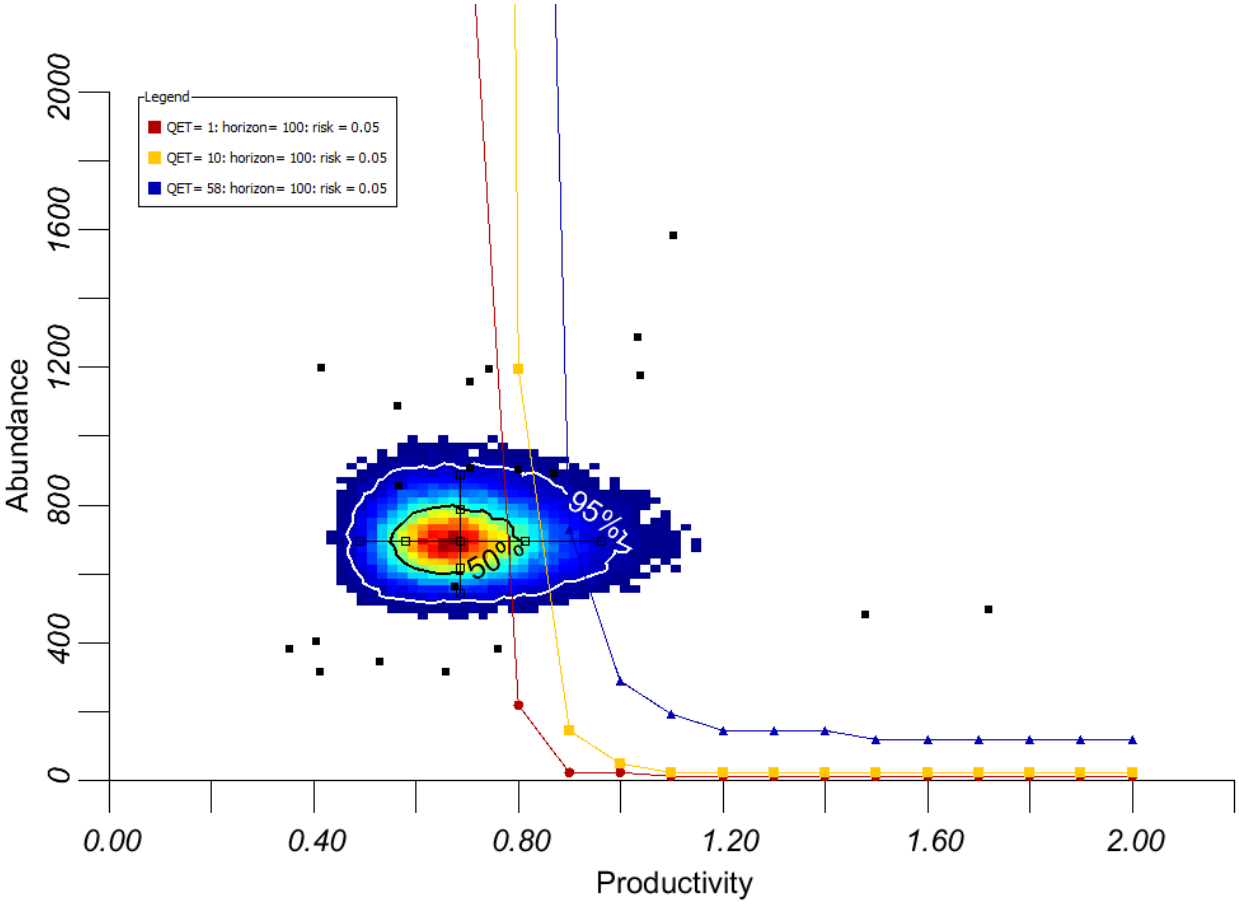


Figure 13. PVA for the Puyallup/Carbon Rivers Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto three viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, or 58 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a very high risk of quasi-extinction over the next 100 years.

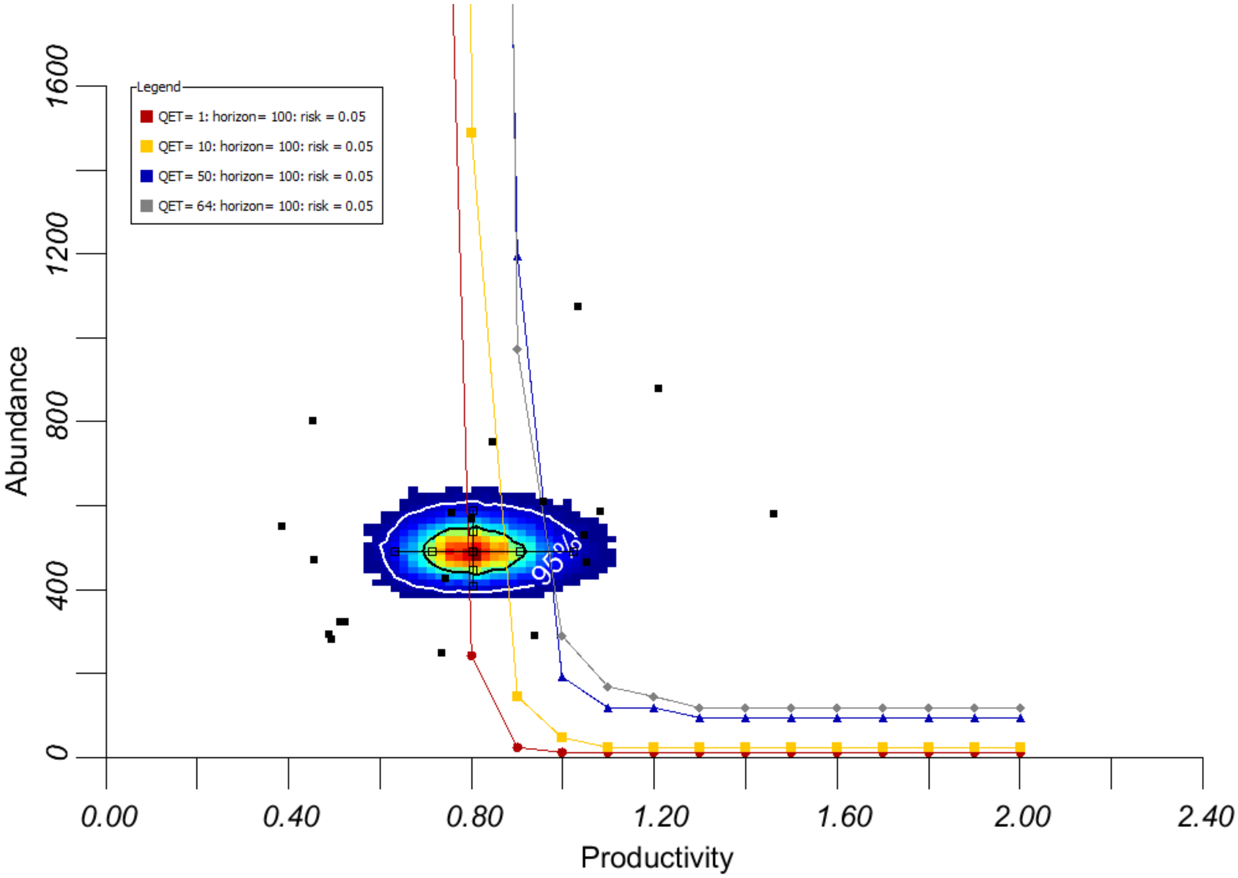


Figure 14. PVA for the White River Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto four viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, 50, or 64 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a very high risk of quasi-extinction over the next 100 years.

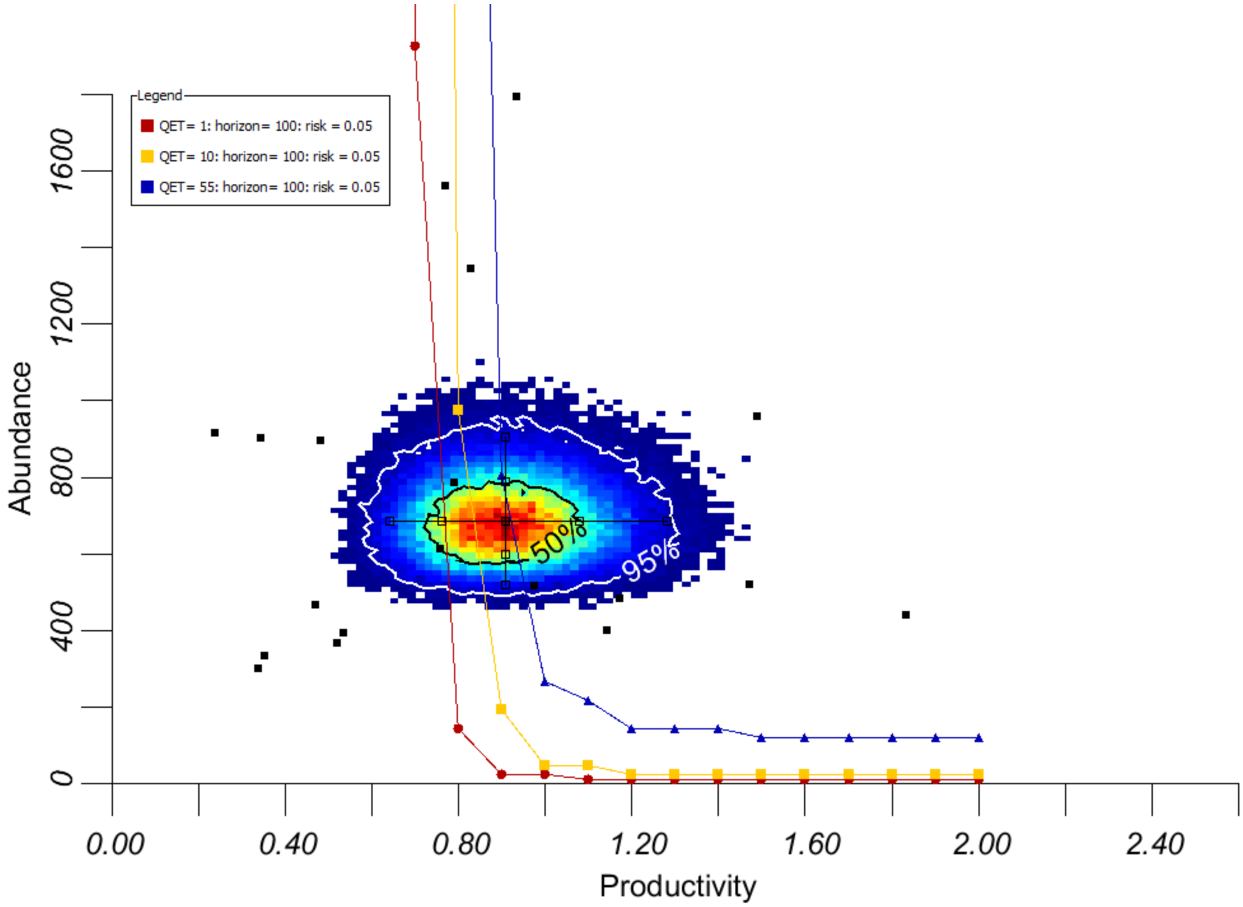


Figure 15. PVA for the Nisqually River Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto three viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, or 55 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a very high risk of quasi-extinction over the next 100 years.

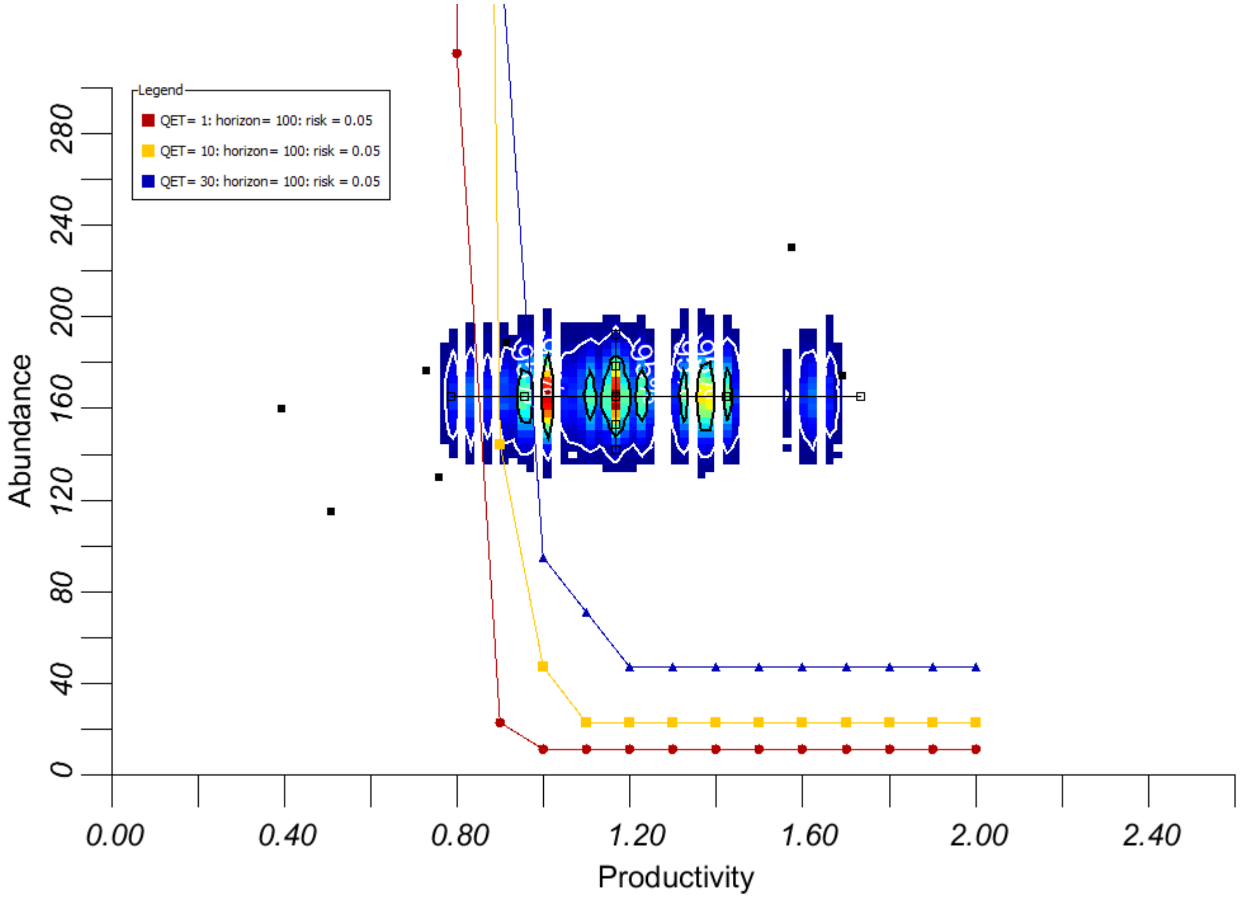


Figure 16. PVA for the South Hood Canal Tributaries Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto three viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, or 30 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a high risk of quasi-extinction over the next 100 years.

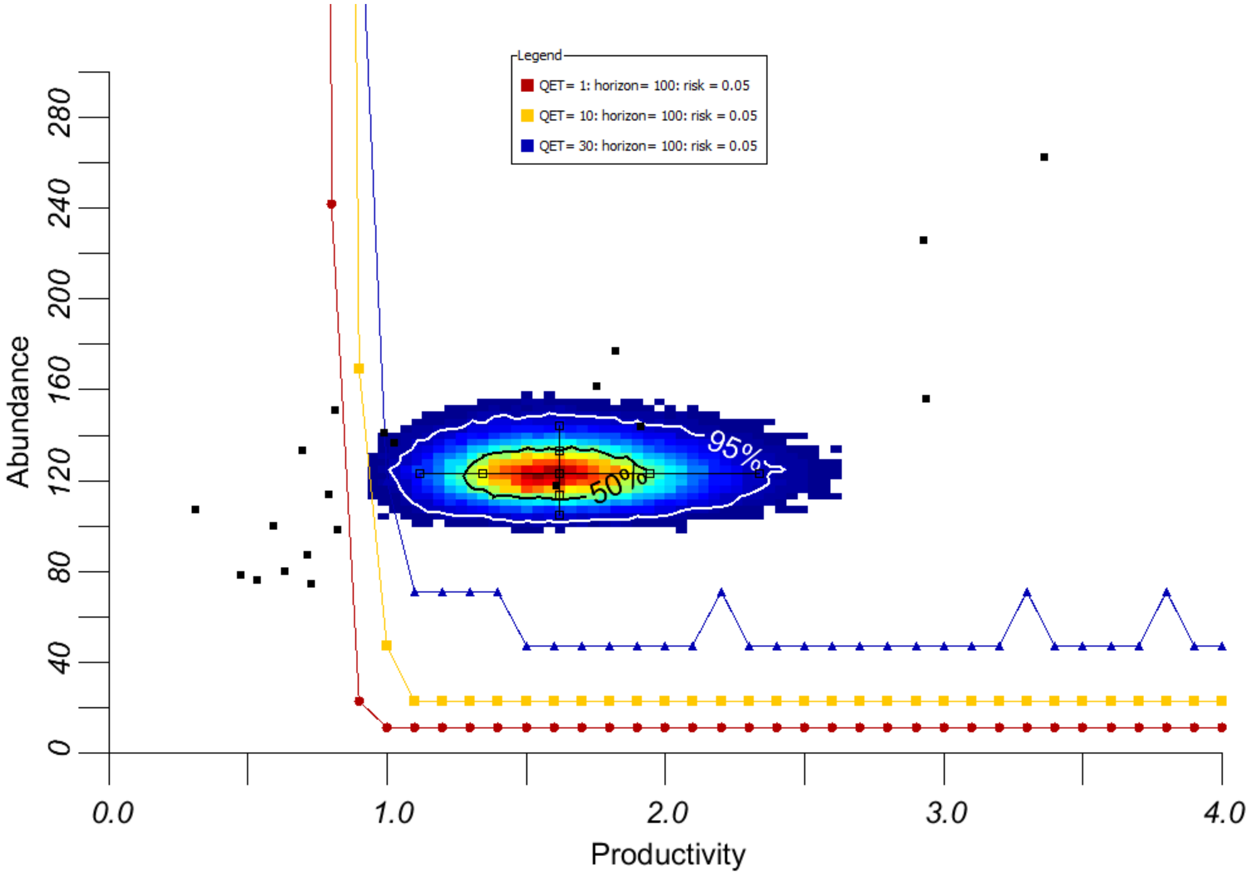


Figure 17. PVA for the Tahuya River winter-run steelhead, part of the South Hood Canal Tributaries Winter-Run DIP, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto three viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, or 30 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a low risk of quasi-extinction over the next 100 years.

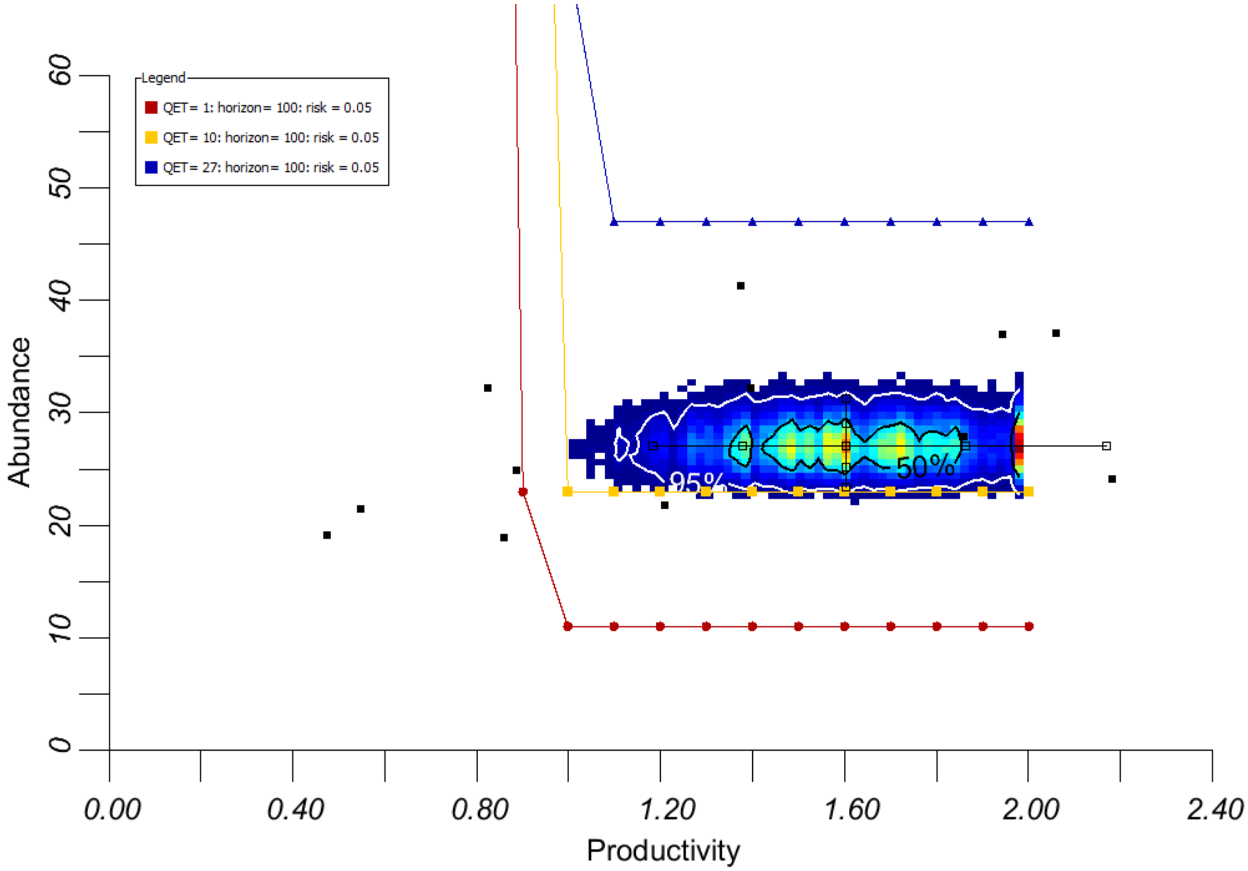


Figure 18. PVA for the East Hood Canal Tributaries Winter Run, conducted with SPAZ (McElhany and Payne 2006). Only Dewatto River population data are available for this DIP. The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto three viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, or 27 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a high risk of quasi-extinction over the next 100 years.

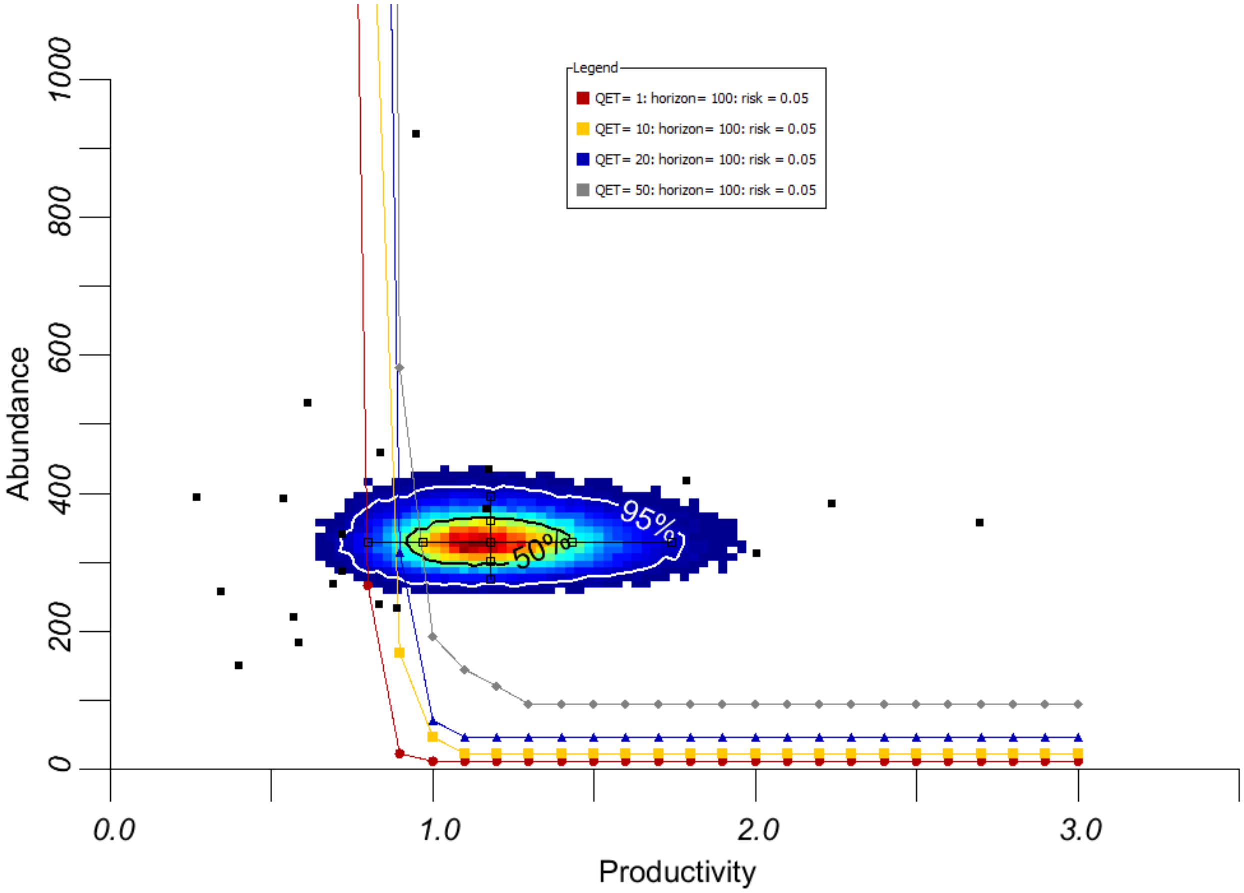


Figure 19. PVA for the Skokomish River Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto four viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, 20, or 50 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a high risk of quasi-extinction over the next 100 years.

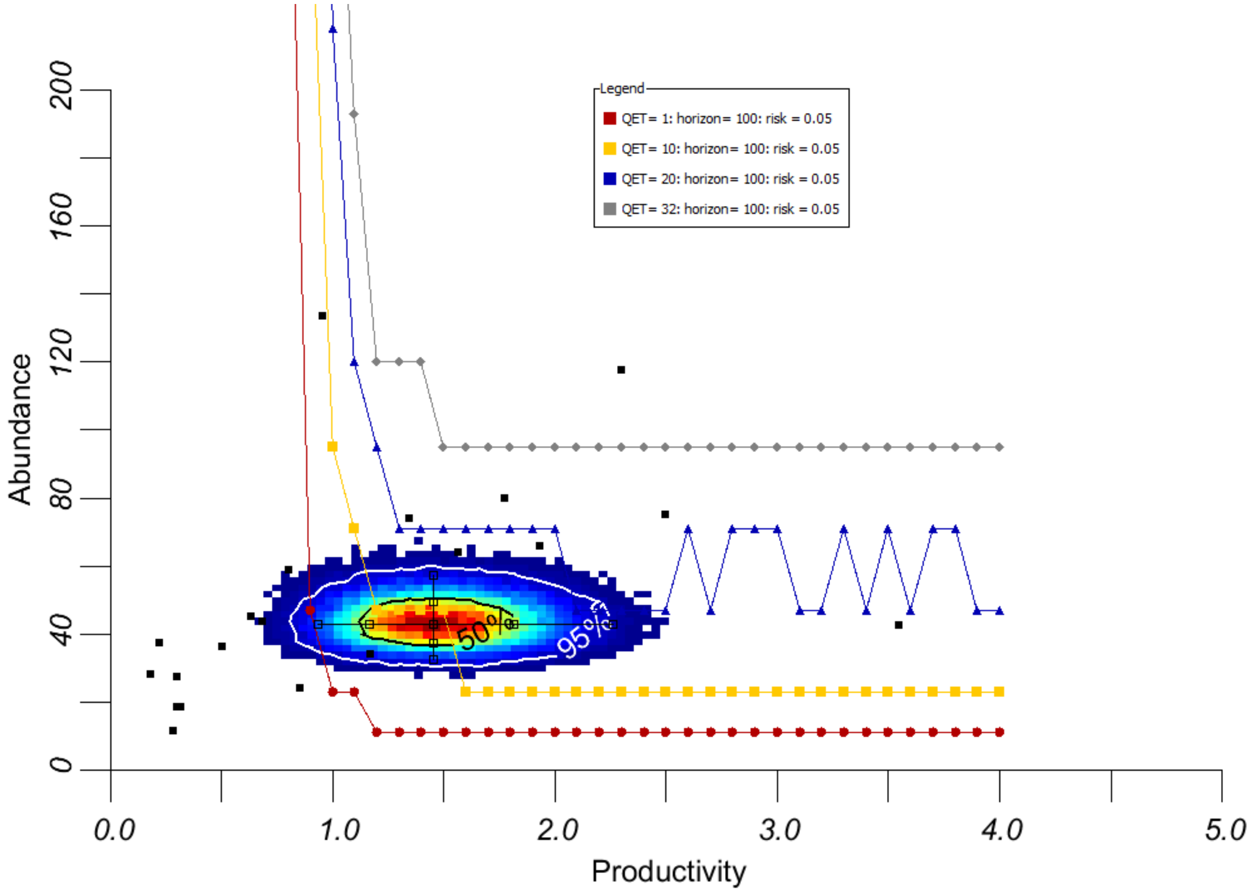


Figure 20. PVA for the West Hood Canal Tributaries Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto four viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, 20, or 32 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a very high risk of quasi-extinction over the next 100 years.

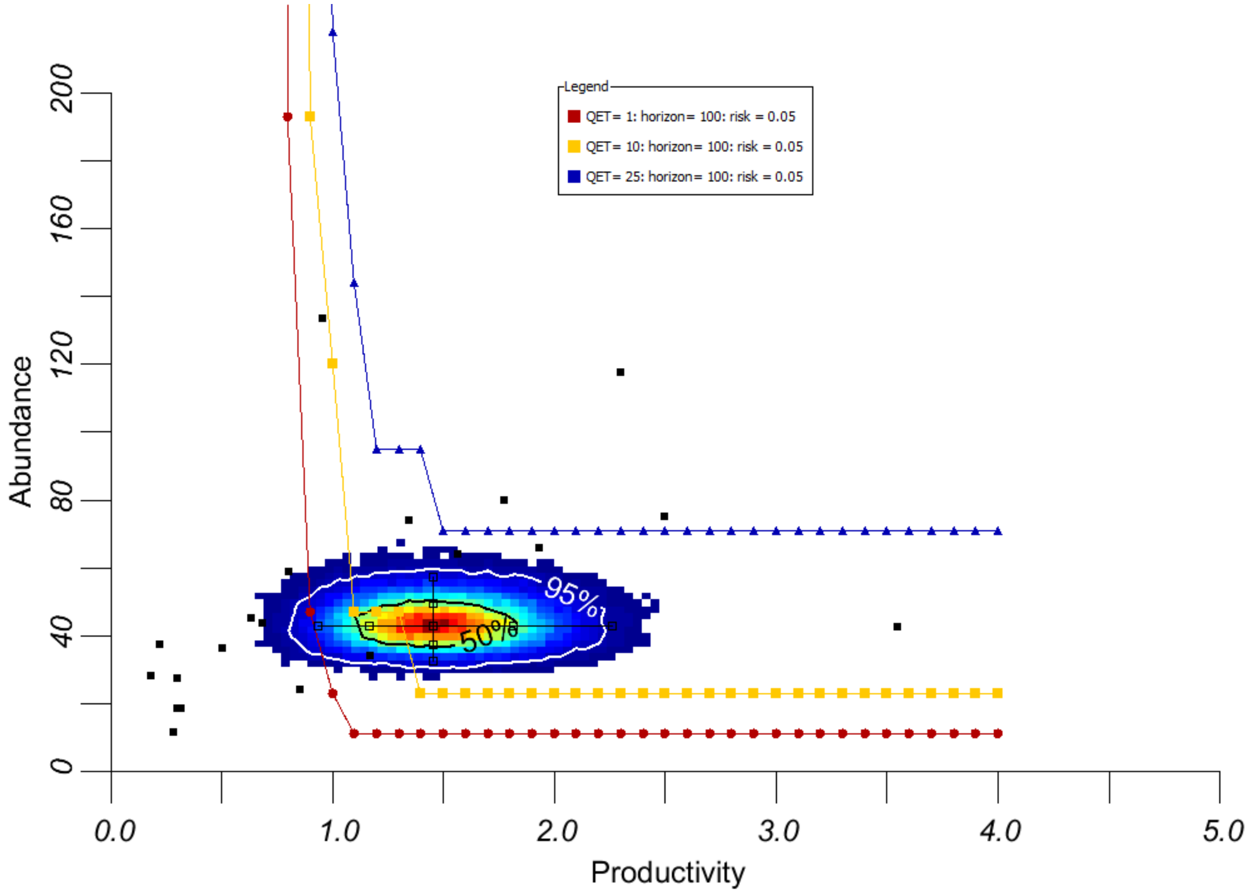


Figure 21. PVA for the Sequim/Discovery Bays Tributaries Winter Run, conducted with SPAZ (McElhany and Payne 2006). Only Snow Creek population data are available for this DIP. The plot depicts the joint estimate of population abundance (y-axis) and productivity (recruits per spawner, x-axis), represented by the connected open squares (showing the approximate 50% and 95% probability centroids of projected abundance and productivity) overlain by colored probability contours. Actual abundance and productivity data are given by the solid black squares. The joint estimate and the data are overlaid onto three viability curves representing different combinations of abundance and productivity that correspond to a 5% risk of reaching a QET of 1, 10, or 25 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a hockey stick recruitment model and uses the geometric mean of running sums of recruits per spawner to estimate productivity over the observed data. Comparison with the values in Table 5 and Figure 2 suggests that this population is at a high risk of quasi-extinction over the next 100 years.

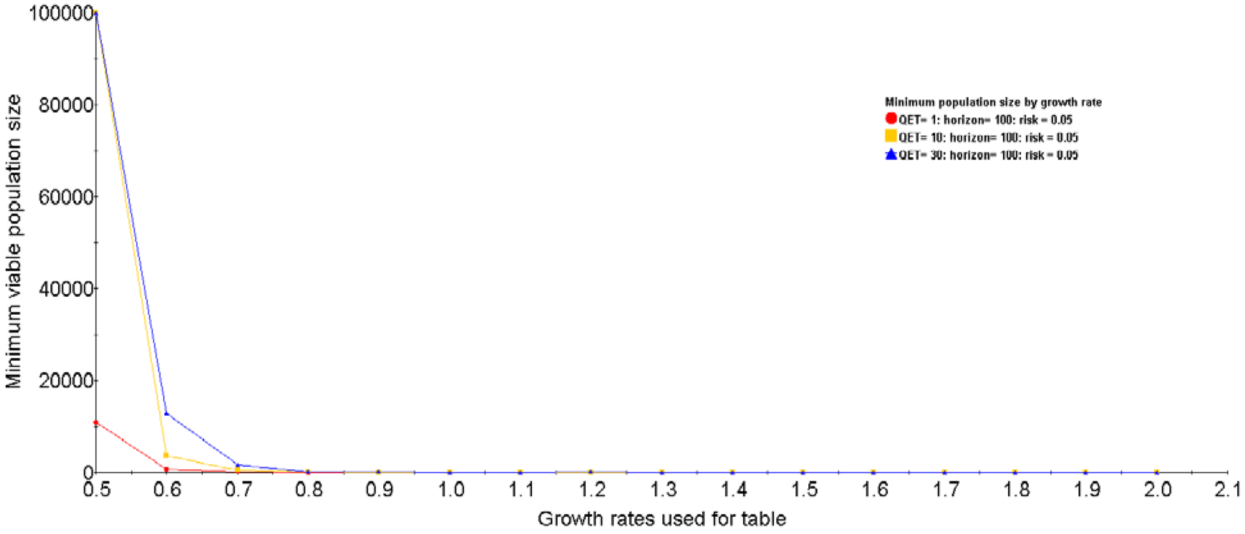


Figure 22. PVA for the Dungeness River Summer Run and Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the estimates of minimum viable population size (y-axis) as a function of population growth rate (x-axis). The estimates correspond to a 5% risk of reaching a QET of 1, 10, or 30 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a random walk with trend model to estimate productivity over the observed data. Comparison with the available abundance data (from the 1980s and 1990s) and the values in Table 5 and Figure 2 suggests that this population is at a very high risk of quasi-extinction over the next 100 years.

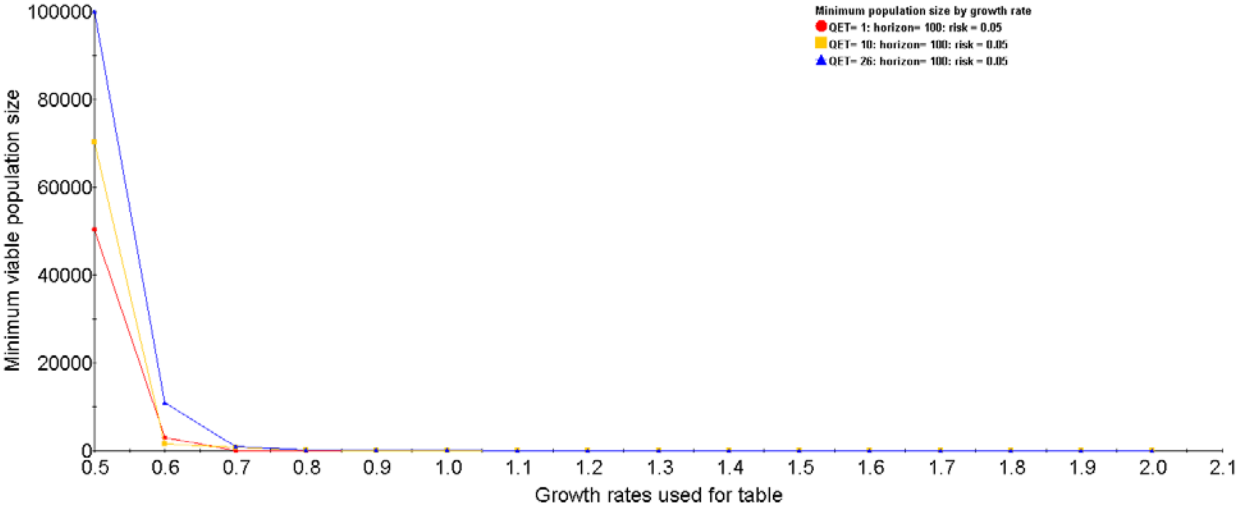


Figure 23. PVA for the Strait of Juan de Fuca Tributaries Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the estimates of minimum viable population size (y-axis) as a function of population growth rate (x-axis). The estimates correspond to a 5% risk of reaching a QET of 1, 10, or 26 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a random walk with trend model to estimate productivity over the observed data. Comparison with the available abundance data (from the 1980s and 1990s) and the values in Table 5 and Figure 2 suggests that this population is at a very high risk of quasi-extinction over the next 100 years.

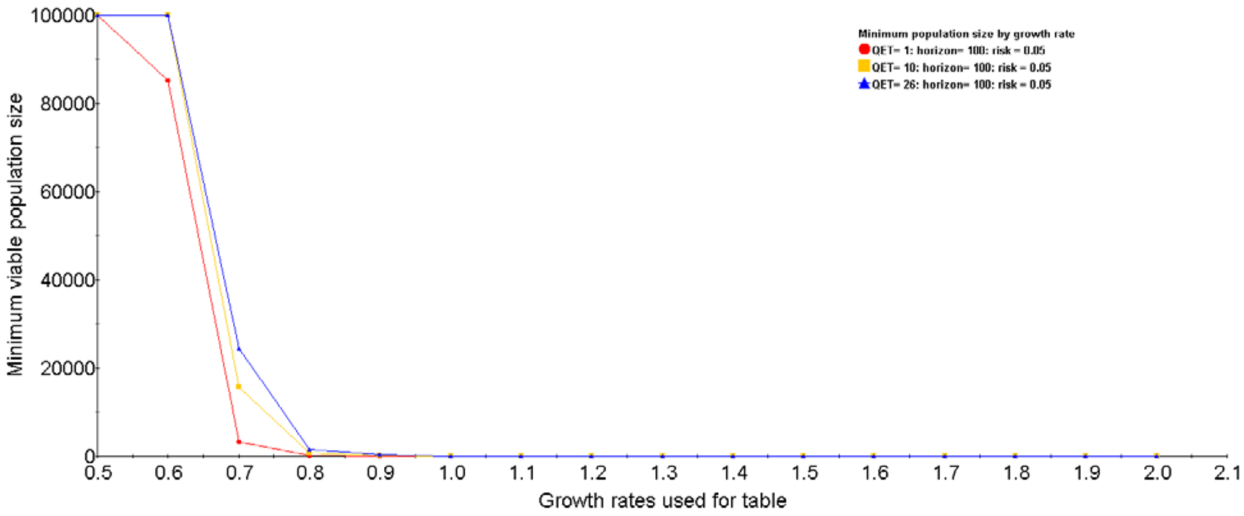


Figure 24. PVA for the Morse Creek winter-run steelhead, part of the Strait of Juan de Fuca Tributaries Winter-Run DIP, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the estimates of minimum viable population size (y-axis) as a function of population growth rate (x-axis). The estimates correspond to a 5% risk of reaching a QET of 1, 10, or 26 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a random walk with trend model to estimate productivity over the observed data. Comparison with the available abundance data (from the 1980s and 1990s) and the values in Table 5 and Figure 2 suggests that this population is at a very high risk of quasi-extinction over the next 100 years.

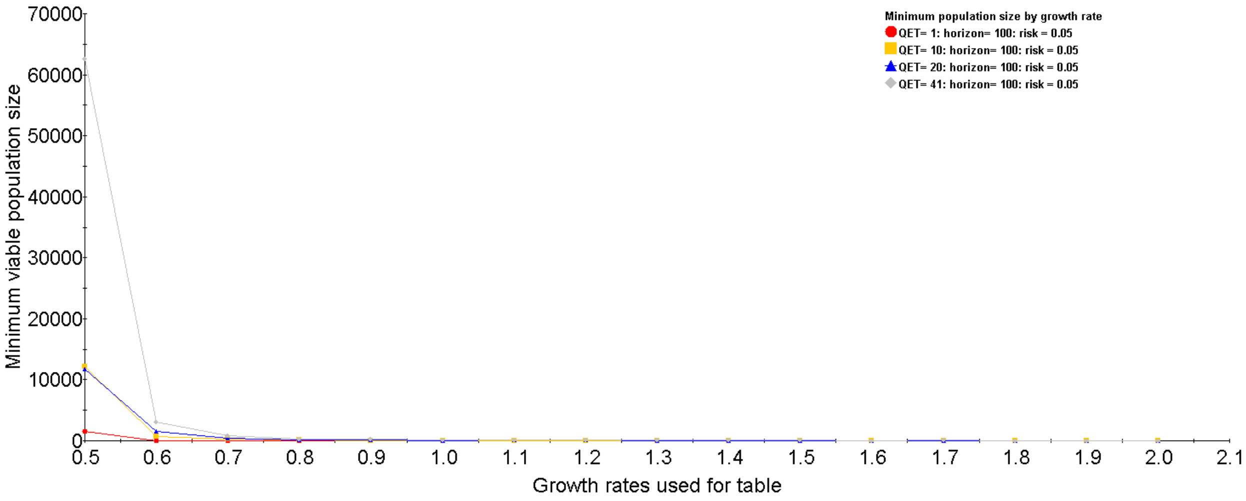


Figure 25. PVA for the Elwha River Winter Run, conducted with SPAZ (McElhany and Payne 2006). The plot depicts the estimates of minimum viable population size (y-axis) as a function of population growth rate (x-axis). The estimates correspond to a 5% risk of reaching a QET of 1, 10, 20, or 41 spawners (the habitat-based estimate of QET for this population) in a given year over a 100-year time horizon. The PVA relies on a random walk with trend model to estimate productivity over the observed data. Comparison with the available abundance data (from the 1980s and 1990s) and the values in Table 5 and Figure 2 suggests that this population is at a very high risk of quasi-extinction over the next 100 years.

Table 6. Description of WDFW natural escapement data available for Puget Sound steelhead populations and their relationship to the TRT-designated DIPs. WSH = winter-run steelhead; SSH = summer-run steelhead, and SaSI = Salmonid Stock Inventory (a WDFW database accessible through its "Salmonscape" Web site at http://wdfw.wa.gov/mapping/salmonscape).

DIP name	Drainages included in DIP	Drainages represented in escapement data	Important contrasts between DIP and escapement data	Escapement data type
Nooksack River WSH	Mainstem Nooksack River and all major tributaries and many side channels	Mainstem, north fork, middle fork, south fork, tributaries, and side channels	Escapement data are from 2010 and 2011 only, and 2011 data were made by expanding off 2010 data	Escapement estimates for redds in mainstem index areas and associated tributaries based on aerial counts and Area Under the Curve estimation
Samish River WSH	Samish River and four creeks entering Bellingham Bay	Mainstem Samish and Samish tributary Friday Creek	—	Total escapement estimates based on cumulative redd counts
Skagit River SSH/WSH	Mainstem Skagit River and all tributaries except Baker River, Sauk River, and Nookachamps Creek	Mainstem Skagit River (RM 22.5 to 94.1), Alder, Diobsud, Rocky, O'Toole, Cumberland, Day, Sorenson, Hansen and Jones creeks, and WSH in Sauk and Cascade rivers	SaSI data include Sauk River escapement, and Sauk River SSH/WSH is a separate designated DIP	Total escapement estimates based on cumulative redd counts
Stillaguamish River WSH	Entire Stillaguamish Basin except Deer and Canyon creeks	North Fork Stillaguamish and tributaries upstream of Deer Creek	South Fork Stillagaumish escapements are not included in SaSI data	"Spawner counts"
Snohomish/ Skykomish WSH	Mainstem Snohomish, Skykomish Basin	Mainstem Snohomish to RM 51.5 on South Fork Skykomish, Wallace River RM 0.0 to 5.8, Sultan River RM 0.0 to 15.0, east and west forks of Woods Creek, Proctor, Elwell/Young's, Olney, Lewis, and Salmon creeks	—	Total escapement estimates based on redd counts
Pilchuck River WSH	Pilchuck River	Mainstem Pilchuck River from RM 0.0 to 15.3, and Worthy, Dubuque, and Little Pilchuck creeks	—	Total escapement estimates based on redd counts

Table 6 continued. Description of WDFW natural escapement data available for Puget Sound steelhead populations and their relationship to the TRT-designated DIPs. WSH = winter-run steelhead; SSH = summer-run steelhead, and SaSI = Salmonid Stock Inventory (a WDFW database accessible through its “Salmonscape” Web site at http://wdfw.wa.gov/mapping/salmonscape).

DIP name	Drainages included in DIP	Drainages represented in escapement data	Important contrasts between DIP and escapement data	Escapement data type
Snoqualmie River WSH	Mainstem Snoqualmie River and its tributaries	Mainstem Snoqualmie upstream to Snoqualmie Falls (RM 40.5), mainstem Tolt River, north and south forks of Tolt, Raging River and Tokul, Cherry, Harris, Griffin, Patterson, Canyon, and Deep creeks	—	Total escapement estimates based on redd counts
Tolt River SSH	North and south forks of Tolt River	South Fork Tolt River, RM 3.3 to 7.8	No estimates for North Fork Tolt in SaSI data	Total escapement estimates based on redd counts February 15 through May. (Cumulative redd counts are multiplied by 0.81 to account for multiple redd construction by females, then by 2 to represent the spawning pair [female and male])
Cedar River WSH	Cedar River	Cedar River and Issaquah and Bear creeks	Cedar and Lake Washington data combined	Total escapement estimates based on redd counts
North Lake Washington WSH	Bear, Issaquah, and Swamp creeks and other tributaries	Data are not available separately from Cedar River	—	—
Green River WSR	Green River	Green River main stem spawning areas and index reaches in Soos and Newaukum creeks totaling 12 miles	—	Total escapement estimates based on cumulative redd counts
Puyallup/Carbon WSH	Puyallup (excluding White River) and Carbon rivers	Mainstem Puyallup River and tributaries, excluding White River, Carbon River	Puyallup and Carbon are separate SaSI stocks; data were combined to form the TRT data set	Total escapement estimates based on redd counts in all suitable spawning habitat of Puyallup River and tributaries, and of Carbon River

Table 6 continued. Description of WDFW natural escapement data available for Puget Sound steelhead populations and their relationship to the TRT-designated DIPs. WSH = winter-run steelhead; SSH = summer-run steelhead, and SaSI = Salmonid Stock Inventory (a WDFW database accessible through its "Salmonscape" Web site at http://wdfw.wa.gov/mapping/salmonscape).

DIP name	Drainages included in DIP	Drainages represented in escapement data	Important contrasts between DIP and escapement data	Escapement data type
White River WSH	White River	Important note: data are from fish management staff, which combines SaSI data (steelhead counts at Buckley Dam trap) and escapement (redd counts) below the dam in mainstem White River and Boise Creek.	2009 to 2011 counts at the Buckley Dam trap included returning adults from the wild broodstock hatchery program; these fish are passed upstream so are potential spawners	Counts of steelhead at the Buckley Dam trap (RM 24.3) and escapement estimates below dam in White River and Boise Creek
Nisqually WSH	Nisqually River	Nisqually River and some tributaries such as Mashel River are now surveyed	—	Total escapement estimates based on redd counts
East Hood Canal Tributaries WSH	Dewatto River, Big Beef and Anderson creeks, other small west Kitsap peninsula streams	Dewatto River only	SaSI data for Dewatto River only	Index escapement estimates based on redd counts from RM 0.0 to 3.75
South Hood Canal Tributaries WSH	Tahuya and Union rivers, Mission Creek, other small South Hood Canal streams	Tahuya and Union rivers. (Also, Tahuya data provided separately as a longer data series that may be useful for PVA-type analyses)	SaSI data for Tahuya and Union only, Tahuya and Union are separate SaSI stocks, data were combined to form the TRT data set	Index escapement estimates based on redd counts from Tahuya RM 1.0 to 11.0, and from Union RM 0.3 to 4.0
Skokomish WSH	Skokomish River	Mainstem Skokomish, North Fork Skokomish, and South Fork Skokomish	—	Total escapement estimates based on redd counts in index areas in mainstem Skokomish RM 0.0 to 9.0, N. F. Skokomish RM 9.0 to 13.0, and S. F. Skokomish RM 0.0 to 21.4

Table 6 continued. Description of WDFW natural escapement data available for Puget Sound steelhead populations and their relationship to the TRT-designated DIPs. WSH = winter-run steelhead; SSH = summer-run steelhead, and SaSI = Salmonid Stock Inventory (a WDFW database accessible through its "Salmonscape" Web site at http://wdfw.wa.gov/mapping/salmonscape).

DIP name	Drainages included in DIP	Drainages represented in escapement data	Important contrasts between DIP and escapement data	Escapement data type
West Hood Canal Tributaries WSH	Hamma Hamma, Duckabush, Dosewallips, Big and Little Quilcene rivers, Tarboo Creek	Hamma Hamma, Duckabush, Dosewallips, and Little Quilcene rivers	SaSI data do not include Big Quilcene River or Tarboo Creek	Index escapement estimates based on redd counts in Hamma Hamma RM 0.3 to 1.8, in Duckabush RM 0.0 to 2.6, in Dosewallips RM 0.2 to 12.0, and in Little Quilcene RM 0.2 to 5.3
Sequim/ Discovery Bays Tributaries WSH	Snow, Salmon, and Jimmycomelately creeks and other streams draining to Sequim and Discovery bays	Snow Creek (Discovery Bay stream)	SaSI data only for Snow Creek population	Total escapement estimates based on Snow Creek trap counts and on redd counts downstream of trap
Dungeness WSH	Dungeness River	Dungeness River (no recent data series available)	—	Index escapements based on redd counts in index areas. Escapement estimates are often not made because of difficult survey conditions, such as high flows in May
Strait of Juan de Fuca Tributaries WSH	Ennis, White, Morse, Siebert, and McDonald creeks	Morse and McDonald creeks. (Also, Morse data provided separately as a longer data series that may be useful for PVA-type analyses)	SaSI data only for Morse and McDonald creeks	Total escapement estimates based on redd counts in Morse Creek from RM 0.0 to 4.7, and in index areas of McDonald Creek from RM 0.0 to 5.4
Elwha River WSH	Elwha River	Elwha River within 4.9 miles downstream of (former) Elwha Dam. No recent SaSI data series available	—	Total escapement estimates based on hatchery/wild ratios in commercial harvest and return to the Elwha Hatchery

If we accept the assumptions underlying these analyses as reasonable and conclude that estimates based on index redd counts for these steelhead populations are tightly correlated with total natural spawning abundance (and that for about the last 10 years spawning abundance and total run size have been similar), the analyses indicate that the majority (considerably more than half) of steelhead populations in the Puget Sound DPS are at high levels of quasi-extinction risk. These analyses have been applied to populations for which semiquantitative estimates of abundance and productivity exist (or can be roughly inferred). For populations for which we have no quantitative information, we hypothesize that they would likely show a similar pattern, because the few data available for these populations show them either to be very small or to have declined precipitously from historic sizes. The risk of quasi-extinction of steelhead in Puget Sound appears to be highest in the Central and South Puget Sound MPG and the Hood Canal and Strait of Juan de Fuca MPG. In the Northern Cascades MPG, two of the seven populations that could be evaluated are at high levels of risk of reaching QET. One of these is a summer-run population (Tolt River). For the Central and South Puget Sound MPG, four of the five populations that could be evaluated are at high levels of risk of reaching QET. The populations at high quasi-extinction risk are at risk primarily because of very low productivity (the Lake Washington population is also at very low abundance). For the Hood Canal and Strait of Juan de Fuca MPG, all eight populations appear to be at high levels of risk of reaching QET. These risk assessments may underestimate true risk levels because of the nature of the data available. It is also important to recognize that these analyses do not explicitly incorporate uncertainty into the risk assessment; they do not fully describe a risk "envelope."

The populations at high QET risk are spread geographically across the DPS. Very little information exists for many populations in the DPS, in particular in Central and South Puget Sound DIPs where steelhead once occurred in abundance in smaller streams and rivers, based largely on punch card data collected by the Washington Department of Game in the first half of the twentieth century. Collectively, these analyses provide strong evidence that most steelhead populations in the Puget Sound DPS are at low levels of viability in terms of abundance and productivity.

Population Viability Analysis with Autoregressive State-Space Models

The above analyses make some assumptions about population demography that may be too restrictive in evaluating extinction risk. In particular, they assume that the population census data are accurate, and they apply a particular stock-recruitment model and an average estimate of adult age distribution that may not apply to all populations or reflect temporal changes in age structure. In addition, as described above, the steelhead data themselves are typically based on indexes of redd counts rather than precise estimates of total run size. Consequently, we also evaluated population viability by applying approaches based on maximum-likelihood estimation to analyze the time series reflecting indices of abundance and infer underlying population dynamics without making such restrictive assumptions. We used the state-space framework described by Hinrichsen and Holmes (2009) and Ward et al. (2010), and implemented in the Multivariate Autoregressive State-Space (MARSS) Program, version 2.3, (Holmes and Ward 2011) in the R statistical environment (R Development Core Team 2012) to estimate parameters describing population dynamics over time from basic abundance data. The state-space framework has a distinct advantage in evaluating ecological applications such as time series of abundance, because they can accommodate missing data and can account for both intrinsic biological factors (process error) and measurement error (nonprocess error) in terms of their contributions to the variability observed in the time series.

State-space models have some other advantages. They explicitly incorporate consideration of uncertainty in forecasts of future population trends, assuming that current conditions persist. They do not require an assumption of a specific underlying demographic structure (e.g., a specific spawner-recruit relationship). The MARSS models used here are fit iteratively to the data via maximum likelihood, using a Kalman-filtered expectation-maximization algorithm. This algorithm is especially well-suited to dynamic systems where hidden random variables occur in the model. The Kalman filter, which is widely applied to the analysis of time series, uses diffusion approximation methods to solve for the expected values of the hidden states (of the multivariate autoregressive processes), conditioned on the data over the entire time series. This approach is appropriate for steelhead abundance data for Puget Sound because these data primarily include observed redd counts, often from index stream reaches, which makes estimation of total abundance challenging.

State-space models have been applied frequently to time series of ecological data because they are able to separate these sources of error, while requiring less information than some other methods (such as estimates of age structure) in the absence of estimates of observation error or replicated observations. Widely used in ecology using both maximum-likelihood and Bayesian frameworks, state-space models have focused primarily on analysis of single time series (e.g., Staples et al. 2004, Dennis et al. 2006), but are capable of handling higher dimension data as well. The basic form of a univariate state-space model with Gaussian (normally distributed) errors is

$$x_t = x_{t-1} + \mu + e_t$$

which is closely related to a stochastic, discrete-time Gompertz model (Ives et al. 2003, Dennis et al. 2006). In these equations x_t is abundance in year *t*, x_{t-1} is abundance in the previous year, μ is a parameter estimating population growth rate (e.g., intrinsic rate of increase), and e_t is the process error in year *t* (e_t is distributed multivariate normal with mean = 0 and variance = σ^2). The term x_{t-1} can be multiplied by a parameter to reflect the strength of density dependence. The multivariate version of the MARSS model, which can accommodate *n* multiple populations, takes the form (Holmes and Ward 2011)

$$\boldsymbol{x}_t = \boldsymbol{x}_{t-1} + \boldsymbol{\mu} + \boldsymbol{e_t}$$

$$\boldsymbol{y}_t = \boldsymbol{a} + \boldsymbol{Z}\boldsymbol{x}_t + \boldsymbol{\varepsilon_t}$$

The first of these equations models the population process and the second the observation process. In these equations, $\boldsymbol{x_t}$ and $\boldsymbol{x}_{t-1}$ are $n \times 1$ vectors of true population sizes in years *t* and $t - 1$, $\boldsymbol{\mu}$ is an *n*-element vector of population growth rates, $\boldsymbol{e_t}$ is an *n*-element vector of process errors in year *t* ($\boldsymbol{e_t}$ is distributed multivariate normal with mean = 0 and covariance matrix $\boldsymbol{Q}$), $\boldsymbol{y_t}$ is an *n*-element vector of observations in year *t*, **a** is an *n*-element vector that represents bias in observation errors, $\boldsymbol{Z}$ is an $n \times n$ matrix of zeros and ones that translates the *n* sizes in year *t* to *n* observations in year *t*, and $\boldsymbol{\varepsilon_t}$ is an *n*-element vector of observation errors in year *t* ($\boldsymbol{\varepsilon_t}$ is distributed multivariate normal with mean = 0 and covariance matrix $\boldsymbol{R}$).

The state-space-based PVAs were computed from the same data used for the SPAZ analyses: estimates of natural escapement (rather than run size) for most of the Puget Sound steelhead populations. The PVAs provide estimates of process and measurement error and use these to compute probabilities of extinction risk and associated confidence intervals. Process error is particularly significant in this context because it reflects variability resulting from intrinsic natural processes, such as demographic stochasticity. The PVAs estimated by MARSS do not account for density dependent effects on productivity and abundance, but this is a frequent assumption of PVA when applied to small or declining populations. If habitat capacity is changing or if Allee effects expressed at low abundance are important influences on population trends, they are not detected by these methods. Although missing data are not strictly limiting to the approach (so long as sufficient data are present in the time series), the PVAs do assume that a population is stationary through time; that is, trends are constant and environmental conditions affecting mortality and production (including harvest) persist. Because it is a state-space approach, a MARSS analysis can provide more precision in estimates of trend because observation error is explicitly included in the analysis (ignoring observation error tends to lead to inflated estimates of process error). The state-space framework partitions the total variance into process and observation variance, which can yield more constrained, realistic estimates of process error and, as a result, more precise estimates of viability metrics. The model also allows estimation of a risk "envelope" that directly incorporates uncertainty resulting from the variability inherent in the time series into the risk projection.

Some concern has been expressed with this approach (and with the slope method), centering on potential bias in overestimating the precision of population growth rate from the analysis of trend in abundance data. However, this concern is based primarily on an assumption that an analysis is estimating the observation variance, not the process variance. The slope and state-space (e.g., MARSS) methods separate process and observation errors and estimate the process variance, so that extinction risk can be forecast from a model of population variability that is free of observation variance.

An example of the results of a PVA using MARSS applied to a time series of population abundance data is shown in Figure 26. The key features and assumptions of this analysis include the following:

- The state-space model uses a diffusion approximation approach,
- No specific demographic structure (e.g., age structure) is assumed,
- Process error due to demographic stochasticity etc. is partitioned from observation error and estimated,
- Density independence of the population's trajectory and stationarity (meaning that an underlying trend that is a function solely of time can be removed, leaving a stationary process) is assumed, and
- Dispersions around estimates are obtained from a Hessian matrix (a numerically estimated square matrix of second-order partial derivatives of the function describing the population trajectory).

In this example, the abundance data are plotted over time with the fitted state-space estimate in the top left graph; MARSS estimated the annual change in abundance (μ_{est}) at −0.057, reflecting a decline of 5.7% per year and corresponding to a growth rate ($\lambda = e^{\mu est}$) of 0.944. In

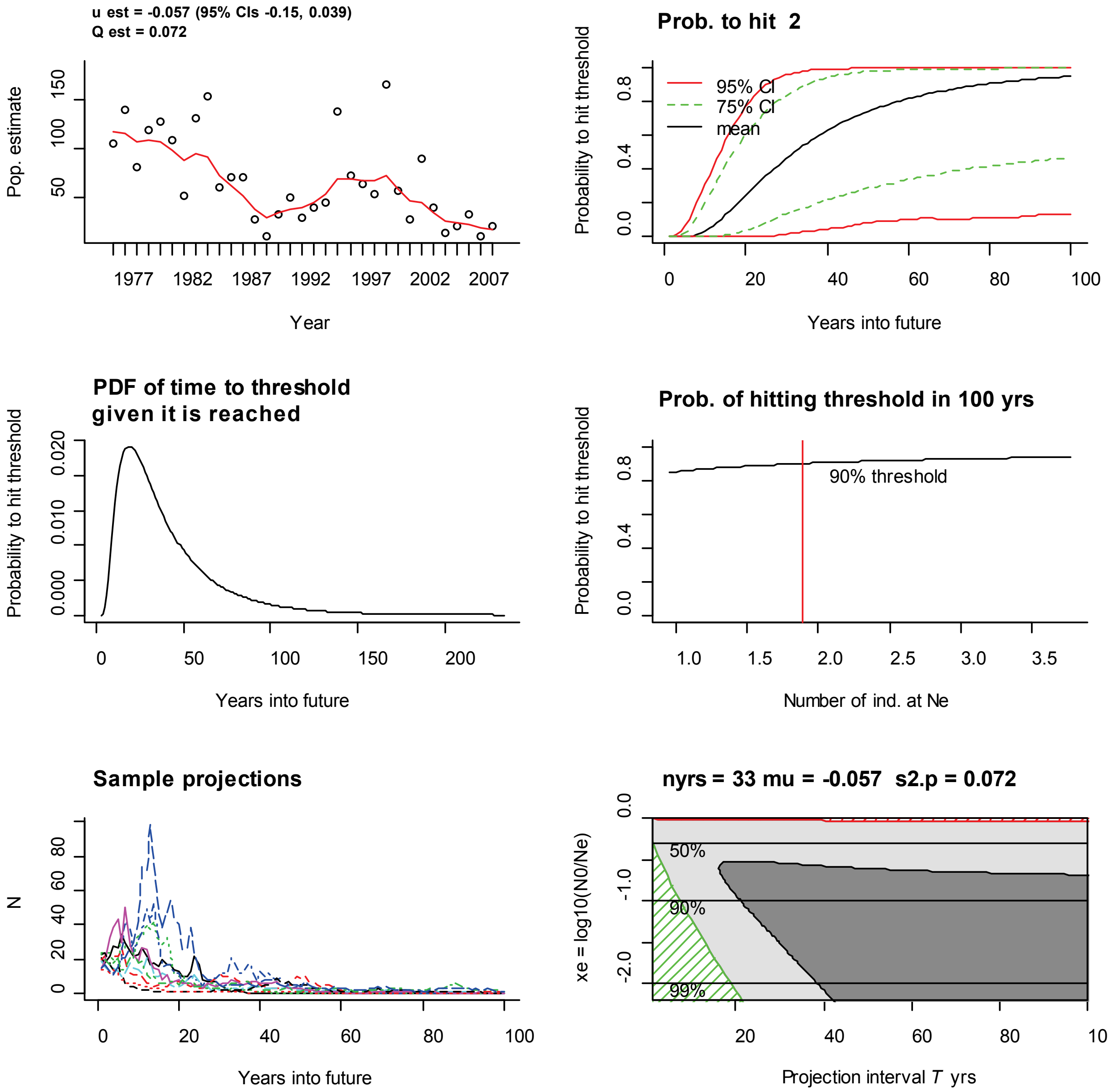


Figure 26. PVA output from MARSS (Holmes and Ward 2011) as applied to a time series of wild steelhead population abundance (Snow Creek winter-run steelhead on the Olympic Peninsula). Top left, time series of abundance data showing estimated annual change in abundance (μ_{est}, or $\ln(\lambda)$) and process error (Q_{est} or s_p^2). Top right, estimated probability of reaching a QET of 10% of the current abundance (21 in 2009 and 10 in 2010). Middle left, plot of the probability density function of the time to reach QET, given that it is reached. Middle right, plot of the probability of reaching QET within 100 years as a function of variation in current abundance N_e (the red vertical line shows the actual data). Lower left, sample projections for the population as a function of years into the future. Lower right, plot of the risk envelope (Ellner and Holmes 2008) showing the expected rate of decline in abundance as a function of time projection in years. The red and green hatched areas represent parameter spaces where rates of population decline over specific time periods are estimated with 95% or higher confidence, with the green area reflecting minimum time horizons for specified rates of decline and the red area reflecting maximum time horizons for those rates. The gray areas encompass the uncertainty envelope for estimating extinction risk ($P < 0.95$).

the top right graph, MARSS estimated that the probability of reaching a QET of 2 over time was 80% by about year 60. The lower left graph plots several sample projections for the population as a function of years into the future, showing low variability among the projections. In the lower right graph of Figure 26, MARSS has plotted a risk envelope, along with the estimates of annual change in abundance (μ_{est}) and process error (Q_{est} or s_p^2 [s2.p in the figure], a reflection of variability resulting from intrinsic natural processes, such as demographic stochasticity). In this plot, the y-axis can be considered to represent the expected rate of decline in abundance from current (N_e) to that at the end of the projection (N_0), plotted against the time projection in years on the x-axis. The gray areas encompass the uncertainty in estimating probabilities of reaching the specified QET. So, for example, in this plot one could conclude with 95% confidence that an expected rate of decline of 50% in this population ($y = -1.0$) is not likely to occur within 100 years; a rate of decline of 90% is not likely before 8–10 years (and a decline of 99% is not likely before 18–20 years). However, beyond the near term, the precise level of extinction risk is uncertain and highly uncertain after about 20 years (for a specified decline of 90%).

We applied autoregressive state-space models to abundance time series for 16 Puget Sound steelhead populations identified as DIPs. The PVA results for each DIP are described in Appendix A, where a full set of MARSS plots is provided. Summaries of the MARSS-based PVA analyses—which evaluate the trends in estimated wild abundance for putative DIPs of Puget Sound steelhead over the entire estimated escapement data series, project population trends 100 years into the future, and where possible evaluate these projections against specified viability criteria—are provided in the following list.

1. For the Samish River winter-run population, see the top two plots of Figure A-1.
2. For the Skagit River Summer Run and Winter Run, see the top two plots of Figure A-2.
3. For the Stillaguamish River Winter Run, see the top two plots of Figure A-3.
4. For the Pilchuck River Winter Run, see the top two plots of Figure A-4.
5. For the Snohomish/Skykomish Rivers Winter Run, see the top two plots of Figure A-5.
6. For the Snoqualmie River Winter Run, see the top two plots of Figure A-6.
7. For the Tolt River Summer Run, see Figure A-7.
8. For the Cedar River Winter Run, see Figure A-8.
9. For the Green River Winter Run, see the top two plots of Figure A-9.
10. For the Puyallup River winter-run population, see the top two plots of Figure A-10.
11. For the White River Winter Run, see the top two plots of Figure A-11.
12. For the Nisqually River Winter Run, see the top two plots of Figure A-12.
13. For the South Hood Canal Tributaries Winter Run, see Figure A-13.
14. For the Tahuya River winter-run population, see Figure A-14.
15. For the East Hood Canal Tributaries Winter Run, see Figure A-15.
16. For the Skokomish River Winter Run, see the top two plots of Figure A-16.
17. For the West Hood Canal Tributaries Winter Run, see the top two plots of Figure A-17.

18. For the Sequim/Discovery Bays Tributaries Winter Run, see Figure A-18.
19. For the Dungeness River Summer Run and Winter Run, see the top two plots of Figure A-19.
20. For the Strait of Juan de Fuca Tributaries Winter Run, see the top two plots of Figure A-20.
21. For the Morse Creek winter-run population, see the top two plots of Figure A-21.
22. For the Elwha River Winter Run, see Figure A-22.

The QETs applied here (Table 7, see also Appendix B) are based on a low average of 24 spawners for Snow Creek winter-run steelhead measured over 4 consecutive years, then scaling by the ratio of the estimate of IP for the watershed supporting the candidate DIP (IP is the area of habitat suitable for steelhead rearing or spawning, at least under historical conditions) to that of Snow Creek. The Snow Creek winter-run steelhead population was chosen because it is a natural anadromous population with sustained wild production in a relatively stable watershed, and it provides accurate estimates of adult escapement, smolt production, and IP to serve as a basis for estimating QETs throughout the Puget Sound Steelhead DPS.

In summary, as did the SPAZ analyses, the MARSS PVAs indicate that the majority of steelhead populations in the Puget Sound DPS are at moderate to high levels of quasi-extinction risk. The risk appears to be high throughout the DPS, but especially in the Central and South Puget Sound MPG and in tributaries to the Strait of Juan de Fuca. Although the variability in the data meant that nearly all the estimates of population growth did not differ significantly from 1 (i.e., a stable trend; the one exception in these analyses was the Cedar River Winter-Run DIP), collectively the trends in abundance point to declining abundance and low productivity. In the Northern Cascades MPG, all but one of the seven populations that could be evaluated show evidence of long-term negative population growth, based on the point estimates ($\lambda < 1$), and three of these populations exhibit probabilities of reaching their specified QET within 100 years that exceed 50%. In this MPG, the Stillaguamish River Winter-Run, Snoqualmie River Winter-Run, and Tolt River Summer-Run populations had the lowest productivities and steepest declines in abundance. Only the Samish River winter-run population showed evidence of positive population growth and a low QET risk. For the Central and South Puget Sound MPG, all of the five populations that could be evaluated show evidence of long-term negative population growth, and four of these exhibit probabilities of reaching their specified QET within 100 years that exceed 50% (three of these are >90%). In this MPG, the Cedar River Winter-Run, Puyallup/Carbon Rivers Winter-Run, and Nisqually River Winter-Run populations had the lowest productivities and steepest declines in abundance. Only the White River Winter-Run population showed evidence of nearly neutral population growth ($\lambda \approx 1$) and a low QET risk (≈40% in 100 years). For the Hood Canal and Strait of Juan de Fuca MPG, six of the eight populations show evidence of long-term negative population growth and these also exhibit probabilities of reaching their specified QET within 100 years that exceed 50%. In this MPG, the South Hood Canal Tributaries Winter Run, Strait of Juan de Fuca Tributaries Winter Run, Dungeness River Summer Run and Winter Run, and Elwha River Winter Run had the lowest productivities and steepest declines in abundance. Only the East Hood Canal Tributaries Winter Run and West Hood Canal Tributaries Winter Run showed evidence of neutral or positive population growth ($\lambda \geq 1$) and a low QET risk (≤40% in 100 years).

Table 7. Estimated QETs for adult steelhead in Puget Sound. SSH = summer-run steelhead and WSH = winter-run steelhead. Intrinsic potential (IP) is total area across habitat categories suitable for steelhead use.

Puget Sound steelhead DIP	**IP (m^2)**	**Estimated QET**
Drayton Harbor Tributaries WSH	597,409	26
Nooksack River WSH	11,119,563	73
South Fork Nooksack River SSH	795,382	27
Samish R. and Bellingham Bay Tribs. WSH	1,616,020	31
Nookachamps River WSH	870,466	27
Skagit River SSH/WSH	30,038,382	157
Baker River SSH/WSH	2,850,284	36
Sauk River SSH/WSH	17,894,638	103
Stillaguamish River WSH	9,814,400	67
Deer Creek SSH	1,575,597	31
Canyon Creek SSH	91,697	24
Snohomish/Skykomish Rivers WSH	11,175,208	73
Pilchuck River WSH	2,446,486	34
Snoqualmie River WSH	7,738,534	58
North Fork Skykomish River SSH	414,682	25
Tolt River SSH	231,883	25
N. Lake Wash. and Lake Sammamish WSH	2,840,214	36
Cedar River WSH	2,545,580	35
Green River WSH	10,170,477	69
Puyallup/Carbon Rivers WSH	7,779,880	58
White River WSH	9,070,004	64
Nisqually River WSH	7,069,169	55
South Puget Sound Tributaries WSH	4,253,116	42
East Kitsap Peninsula Tributaries WSH	256,617	25
East Hood Canal Tributaries WSH	676,924	27
South Hood Canal Tributaries WSH	1,496,762	30
Skokomish River WSH	6,041,453	50
West Hood Canal Tributaries WSH	1,815,949	32
Sequim/Discovery Bays Tributaries WSH	228,335	25
Dungeness River SSH/WSH	1,449,576	30
Strait of Juan de Fuca Tributaries WSH	627,758	26
Elwha River WSH	3,915,221	41

Examination of the uncertainty surrounding the risk projections in the MARSS plots in Appendix A indicates that precise characterization of viability is not possible for most populations after a couple of decades. Nevertheless, the overarching trends are clear and viability appears generally to be low throughout most of the DPS. Because these abundance data in most cases are indexes of natural escapement rather than reasonably precise estimates of natural run size, we emphasize that these estimates of viability for the DIPs are likely to overestimate their true viabilities in most cases.

The populations at high QET risk are spread geographically across the DPS. Collectively, these PVAs conducted with SPAZ and MARSS provide strong evidence that most steelhead populations in all three MPGs within the Puget Sound DPS are at low levels of viability with respect to abundance and productivity, and many populations—probably at least a quarter of them—are at high risk of reaching specified quasi-extinction thresholds.

Analyses at the Major Population Group Level

Figure 27 through Figure 32 examine the trends in estimated natural escapement for Puget Sound steelhead over the entire data series (1985–2011), including their residuals (Figure 28, Figure 30, and Figure 32), for 20 DIPs combined into three putative MPGs in the DPS: Northern Cascades, Central and South Puget Sound, and Hood Canal and Strait of Juan de Fuca. In each case, the graphs plot the maximum-likelihood estimate of log(total number of natural steelhead) for the candidate populations in the MPG against the observed data, assuming that 1) each population time series follows a single MPG trajectory and is simply scaled up or down relative to it, and 2) variances in the observation errors for each time series are multivariate normal but allowed to be unique for each population. The approximate CIs were computed either using a numerically estimated Hessian matrix (a square matrix of second-order partial derivatives of the function) or via parametric bootstrapping (see Holmes and Ward 2011). The relatively tight CIs arise because the estimate of composite process error is small and because all the time-series data are fit to a single “population” trajectory. The total MPG estimate accounts for the bias estimated for the first population time series.

The Northern Cascades MPG shows a clearly declining trend in natural spawner abundance (Figure 27). The average long-term MPG growth rate (μ_{est}, equivalent to ln(λ); see Table 1 and Table 2) is estimated from the slope of the regression. This growth rate is negative (−0.039), corresponding to an estimated loss in abundance of 3.9% per year and a λ of 0.962. The process error (Q_{est}), which is the temporal variability in population growth rate arising from demographic stochasticity, estimated from the variance of residuals around the regression line (Figure 28), is 0.024. The Central and South Puget Sound MPG also shows a clearly declining trend in wild abundance (Figure 29 and Figure 30). Its estimated long-term MPG growth rate is negative, with a loss of 6.9% per year ($\lambda = 0.933$), and its estimated process error is <0.001. The Hood Canal and Strait of Juan de Fuca MPG shows a negative long-term population growth rate of 1.3% per year ($\lambda = 0.987$), with an estimated process error of 0.096 (Figure 31 and 32). All three MPGs contain populations with trending residuals in abundance (Figure 28, Figure 30, and Figure 32), which indicates that the population dynamics of these populations are following different demographic processes or trajectories, suggesting that each MPG is not functioning demographically as a single, cohesive metapopulation, but rather contains distinctly different populations that reflect the DIP identifications. This type of diversity is natural for a complex unit like an MPG and we do not further investigate its demographic substructure here.

In summary, all three steelhead MPGs exhibit declining trends in abundance and estimated composite long-term growth rates are negative, ranging from about 1 to 7% annually. For all but a few of the DIPs of steelhead in these MPGs, estimates of mean population growth rates obtained from observed spawner or redd counts are declining—typically 3 to 10% annually—and extinction risk within 100 years for most populations in the DPS is estimated to be moderate to high, especially for most populations in the Central and South Puget Sound MPG and the Hood Canal and Strait of Juan de Fuca MPG. Collectively, these analyses indicate that steelhead in the Puget Sound DPS remain at risk of extinction throughout all or a significant portion of their range in the foreseeable future, but are not currently in danger of imminent extinction.

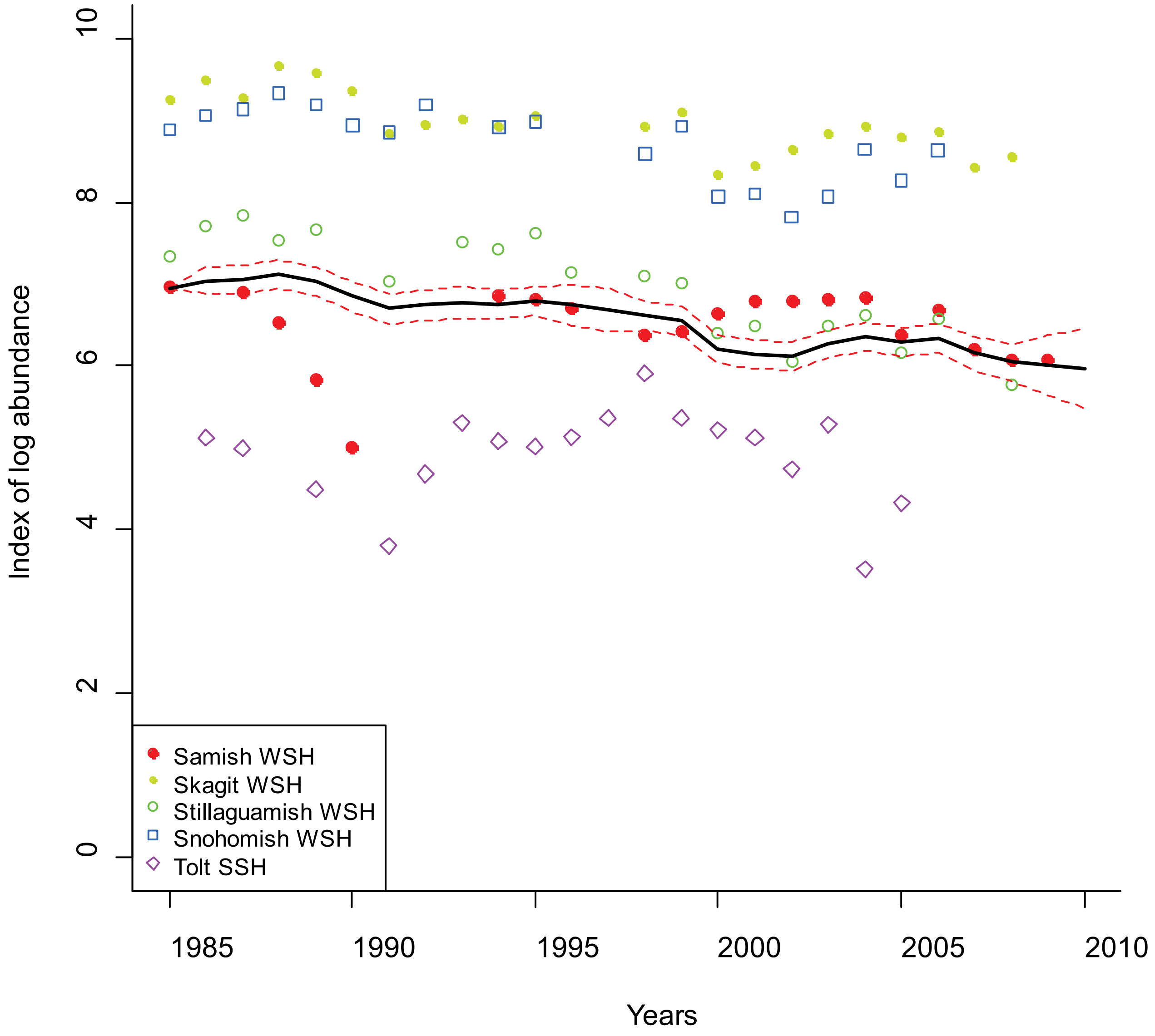


Figure 27. Graph of the trend in estimated total Puget Sound winter-run and one summer-run (Tolt River) steelhead population for a putative Northern Cascades MPG. The graph plots the maximum-likelihood estimate of log(total no. steelhead) in the MPG against the observed data, assuming a single population model for the MPG. The estimate of the log(total MPG count) (solid black line) has been scaled relative to the Samish River population. The 95% CIs around the total MPG estimate are given by the red dashed lines. (Note that these are not the CIs around the observed data, which are expected to fall outside the CI, depending on population-specific nonprocess error, but are instead around the composite estimate per Holmes and Ward 2011.)

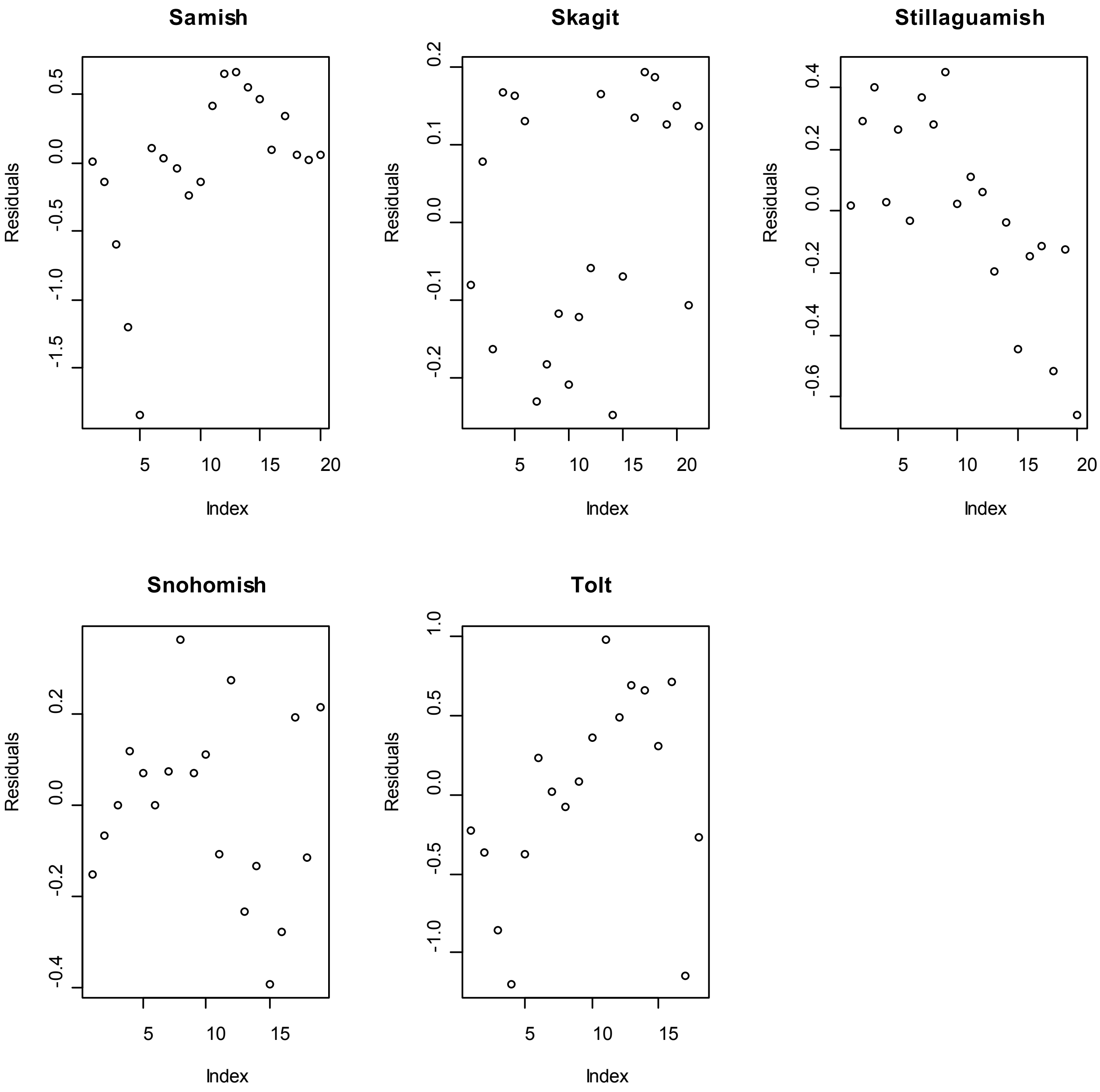


Figure 28. Plots of the residuals from regression of steelhead counts over time for five populations in the Northern Cascades MPG. Note the increasing trend for Samish River winter-run steelhead and the declining trend for Stillaguamish River Winter-Run steelhead.

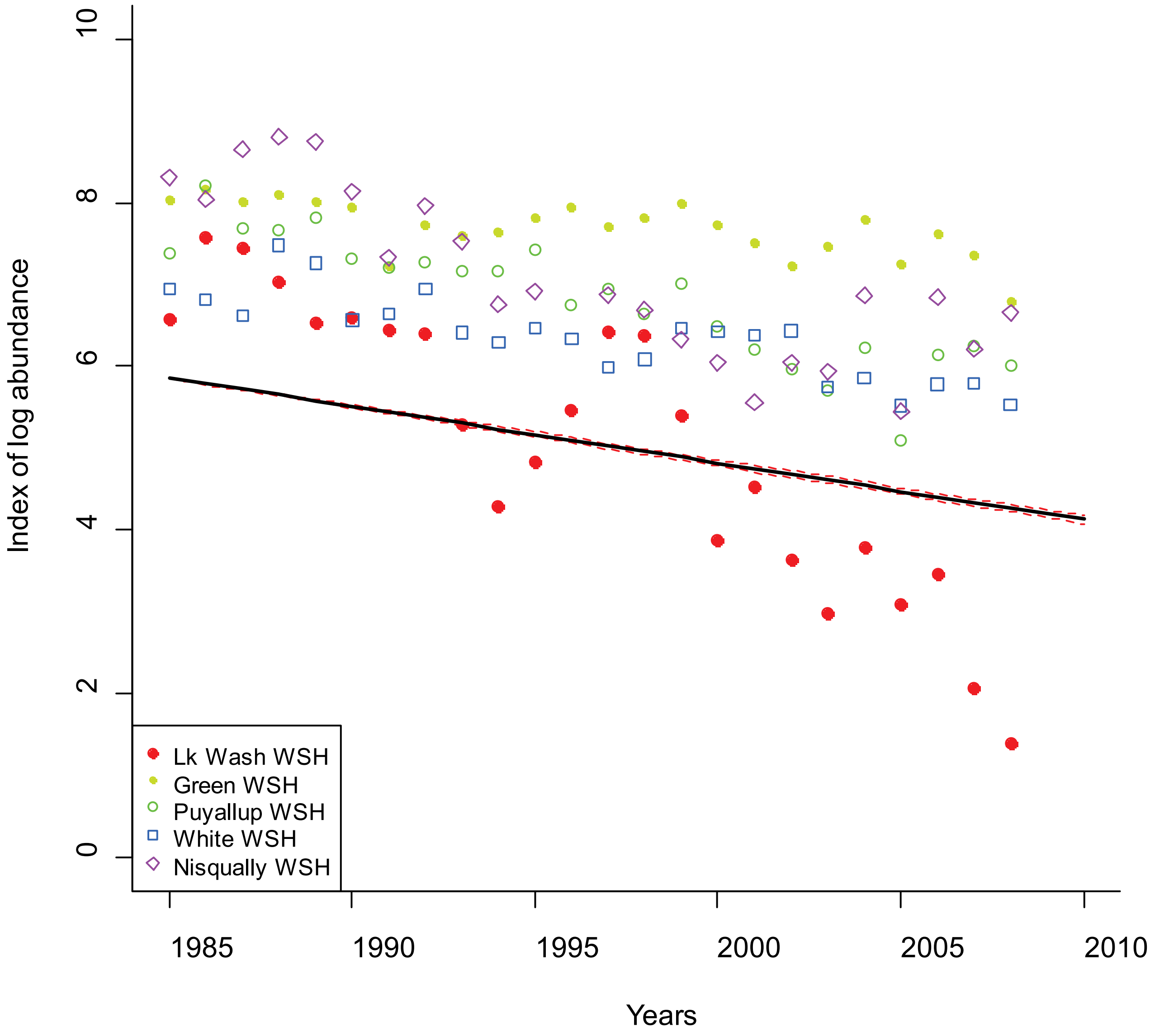


Figure 29. Plot of the trend in estimated total Puget Sound winter-run steelhead for a putative Central and South Puget Sound MPG. The graph plots the maximum-likelihood estimate of log(total no. steelhead) in the MPG against the observed data. The estimate of the log(total MPG count) (solid black line) has been scaled relative to the Lake Washington population. The 95% CIs around the total MPG estimate are given by the red dashed lines. (Note that these are not the confidence intervals around the observed data, which are expected to fall outside the CI, depending on population-specific nonprocess error, but are instead around the composite estimate per Holmes and Ward 2011.)

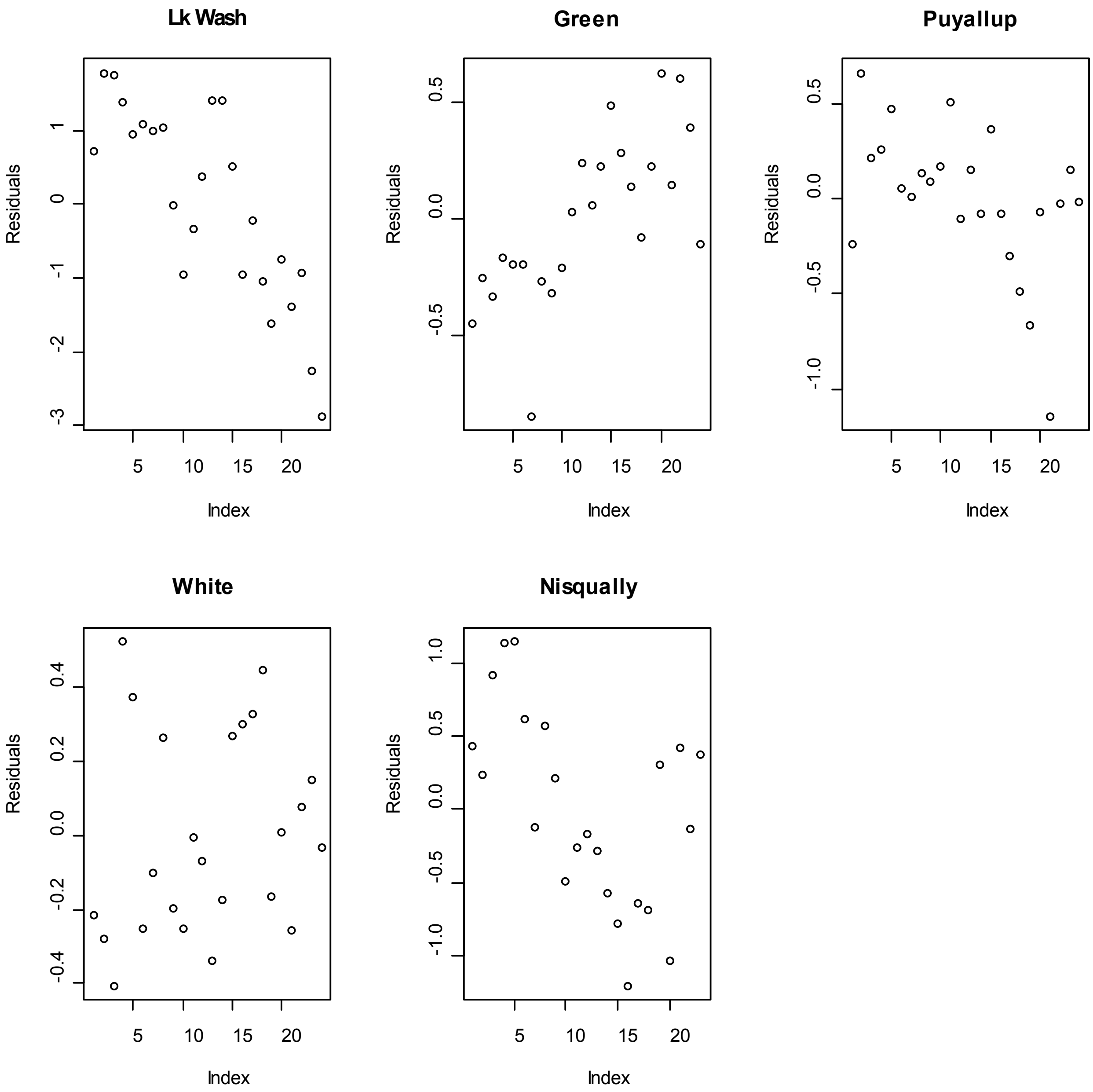


Figure 30. Plots of the residuals from regression of steelhead counts over time for five winter-run populations in the Central and South Puget Sound MPG. Note the declining trends for Lake Washington, Puyallup River, and Nisqually River winter-run steelhead and the increasing trend for the Green River Winter Run.

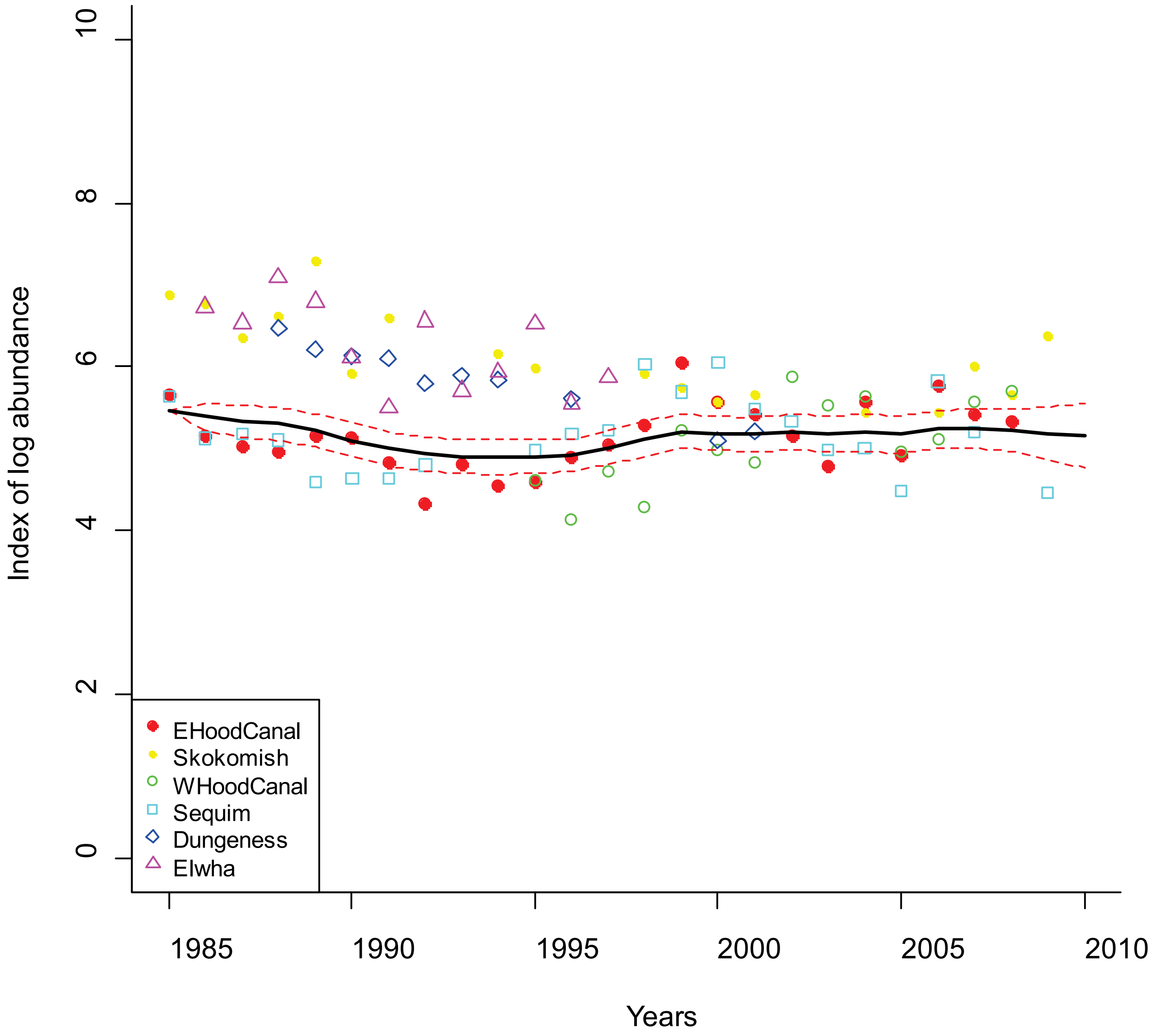


Figure 31. Graph of the trend in estimated total Puget Sound summer-run and winter-run steelhead for a putative Hood Canal and Strait of Juan de Fuca MPG. The graph plots the maximum-likelihood estimate of log(total no. steelhead) in the MPG against the observed data. The estimate of the log(total MPG count) (solid black line) has been scaled relative to the Elwha River Winter-Run population. The 95% CIs around the total MPG estimate are given by the red dashed lines. (Note that these are not the confidence intervals around the observed data, which are expected to fall outside the CI, depending on population-specific nonprocess error, but are instead around the composite estimate per Holmes and Ward 2011.)

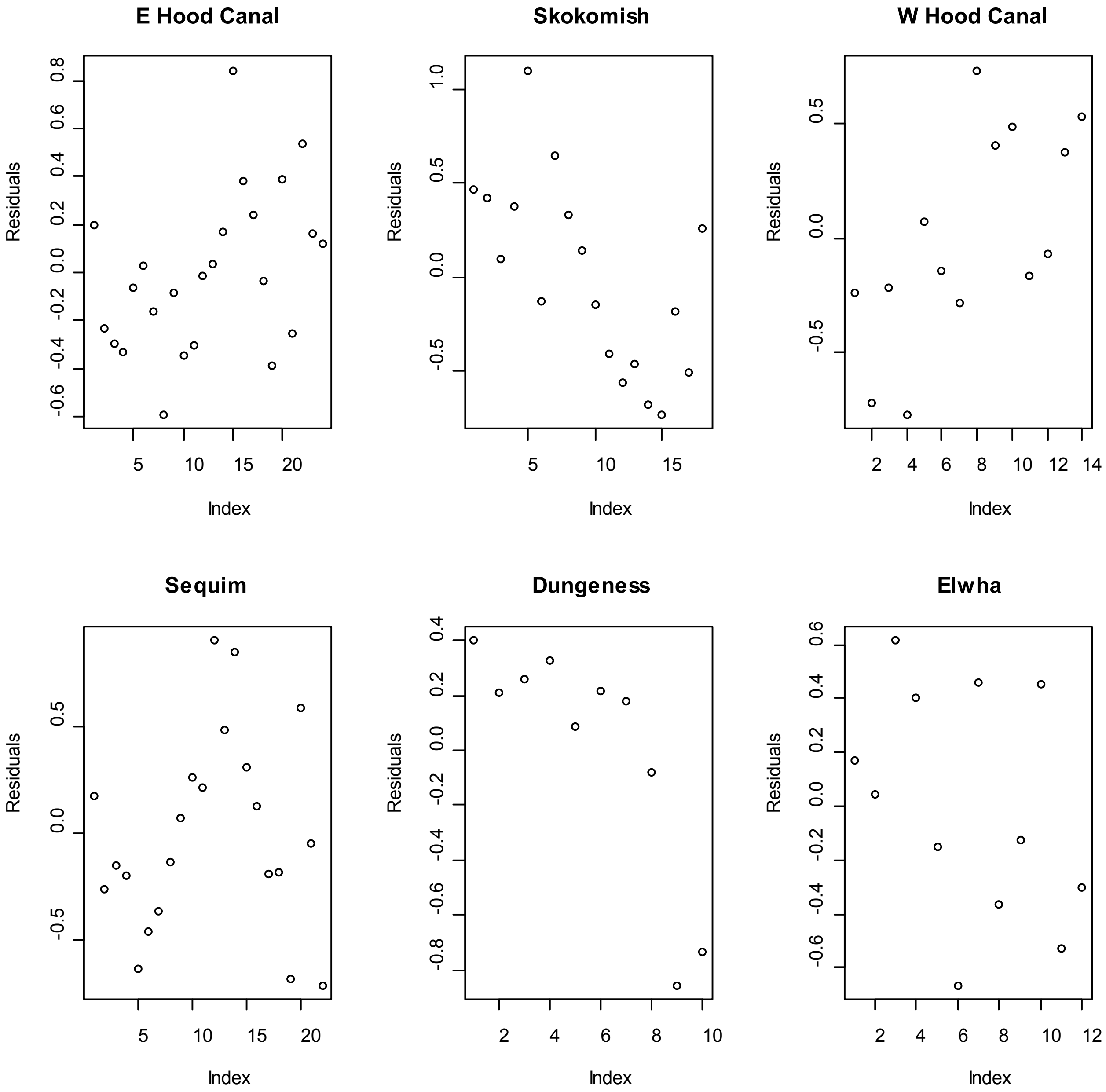


Figure 32. Plots of the residuals from regression of steelhead counts over time for six summer-run and winter-run populations in the Hood Canal and Strait of Juan de Fuca MPG. Note the declining trends for Skokomish River Winter-Run and Dungeness River Summer-Run and Winter-Run steelhead, and the increasing trend for West Hood Canal Tributaries Winter-Run steelhead.

DPS Viability Assessment Methods

The viability of a DPS is dependent on the viability of all of its component MPGs, which in turn are evaluated based on the combined status of their component DIPs. The PSS TRT adopted methods described by Good et al. (2005) to evaluate data that affect the four VSP parameters for each DIP in the Puget Sound Steelhead DPS. State and tribal comanagers provided data on abundance, harvest, age structure, and hatchery releases. Data on adult returns were obtained from a time series of freshwater spawner surveys and redd and trap counts. Where possible, time series were assembled and analyzed for each population that had sufficient data.

In addition, a comprehensive assessment of viability at each level depends on consideration of diversity and spatial structure as well as abundance and productivity, but these factors are typically much more difficult to quantify. In this subsection, we review the four VSP components and describe the metrics the TRT considered most useful for each component. The utility of a metric is based on its biological relevance and its availability.

Abundance

Population size or abundance is recognized as an important parameter because, all else being equal, small populations are at greater risk of extinction than large populations, primarily because several processes that affect population dynamics operate differently in small populations than they do in large populations. These processes are deterministic density effects (e.g., depensation at low density), environmental variation, genetic processes (e.g., genetic drift), demographic stochasticity, ecological feedback, and susceptibility to catastrophe. McElhany et al. (2000) provided guidelines relating minimum abundance to each of these processes at both the “viable” and “critical” levels, where a critical level implies a high risk of extinction over a short time period.

Metrics: adult spawners, expanded from redd counts; weir, trap, or dam counts of adult fish.

Productivity

Population growth rate (i.e., productivity over the entire life cycle) and factors that affect population growth rate provide information on how well a population is “performing” in the habitats it occupies during the life cycle. Estimates of population growth rate that indicate a population is consistently failing to replace itself are an indicator of increased extinction risk. Although the overall focus is on population growth rate over the entire life cycle, estimates of stage-specific productivity—particularly productivity during freshwater life history stages—are also important to comprehensive evaluation of population viability. Other measures of population productivity, such as intrinsic productivity and the intensity of density dependence, may provide important information for assessing a population’s viability. The McElhany et al. (2000) guidelines for population growth rate are closely linked with those for abundance.

Metrics: lambda, recruit/spawner estimates, smolt-or-parr-to-adult survival estimates, trend analysis; freshwater habitat-based IP (parr production) estimates.

Diversity

Several salmonid traits exhibit considerable diversity within and among populations, and this variation has important effects on population viability. In a spatially and temporally varying environment, there are three general reasons why diversity is important for species and population viability. First, diversity allows a species to use a wider array of environments than it could without diversity. Second, diversity protects a species against short-term spatial and temporal changes in the environment. Third, genetic diversity provides the raw material for surviving long-term environmental change. In order to conserve the adaptive diversity of salmonid populations, it is essential to 1) conserve the environment to which they are adapted, 2) allow natural process of regeneration and disturbance to occur, and 3) limit or remove human-caused selection or straying that weakens the adaptive fit between a salmonid population and its environment or limits a population's ability to respond to natural selection.

Metrics: age structure, repeat spawning rate, spawn timing range (within run types), and contribution of resident *O. mykiss* to anadromous production; hatchery introgression (in contrast to competition).

Spatial Structure

When evaluating population viability, it is important to take within-population spatial structure into account for two main reasons: 1) because there is a time lag between changes in spatial structure and species-level effects, overall extinction risk at the 100-year time scale may be affected in ways not readily apparent from short-term observations of abundance and productivity, and 2) population structure affects evolutionary processes and may therefore alter a population's ability to respond to environmental change. Spatially structured populations in which subpopulations occupy habitat patches connected by low to moderate stray rates are often generically referred to as metapopulations. A metapopulation's spatial structure depends fundamentally on habitat quality, spatial configuration, and dynamics, as well as the dispersal characteristics of individuals in the population. Pacific salmonids including steelhead are generally recognized as having metapopulation structure and the guidelines for spatial structure describe general rules of thumb regarding metapopulation persistence.

Metrics: accessible habitat (present vs. historical), proportion of spawning or rearing habitat occupancy.

Applying Viability Analyses to Multiple Spatial Scales

Assessing the viability of a DPS is rooted in the evaluation of DIP VSP parameters. From a recovery planning standpoint, recommended biological delisting criteria for Puget Sound steelhead are more easily understood from a DPS to DIP perspective. The PSS TRT has recommended a number of biological viability criteria for the DPS; individual DIP viability criteria are less fully developed, although the general assessment approach has been outlined. In this subsection, we present the DPS- and MPG-based viability criteria. These criteria were developed using the Puget Sound Steelhead DPS population structure the TRT identified in its previous publication, Identifying Historical Populations of Steelhead within the Puget Sound Distinct Population Segment (Myers et al. 2015). In that technical memorandum, the TRT identified three MPGs containing a total of 32 historically present steelhead DIPs (Figure 33). The existing configuration of the Puget Sound Steelhead DPS is based on information compiled to date, and it is possible that future analyses of new data will suggest additional changes. It is unlikely that these changes will affect the number or configuration of the MPGs, but most probably would result in identification of additional DIPs. The following DPS viability criteria would be unaffected by changes in the number of DIPs.

DPS Viability

1. The DPS is considered viable only if all of its component MPGs are considered viable.

MPG Viability

2. An MPG is considered viable if at least 40% (rounded up) of its component DIPs are considered viable.
    a. DIPs exhibiting major life history strategies (e.g., summer run vs. winter run) will be considered separate components within an MPG. Therefore, 40% of summer-run populations and 40% of winter-run populations within an MPG that contains both life history types must be viable.
    b. DIPs containing winter-run and summer-run subpopulations predominantly exhibit the winter-run life history strategy and will be considered winter run for the purpose of 2a.
3. Within an MPG, those DIPs that do not meet the VSP criteria for viability must be maintained at a level such that the probability of reaching a specified QET within 100 years is no greater than 0.25 (the geometric mean of all populations in the MPG).
    a. Where a DIP is below QET, but a self-sustaining resident population of *O. mykiss* is present in the accessible anadromous zone, the QET risk cannot be greater than 0.90.
4. Viability criteria for DIPs within an MPG cannot be allowed to degrade to the point where an MPG is no longer viable.

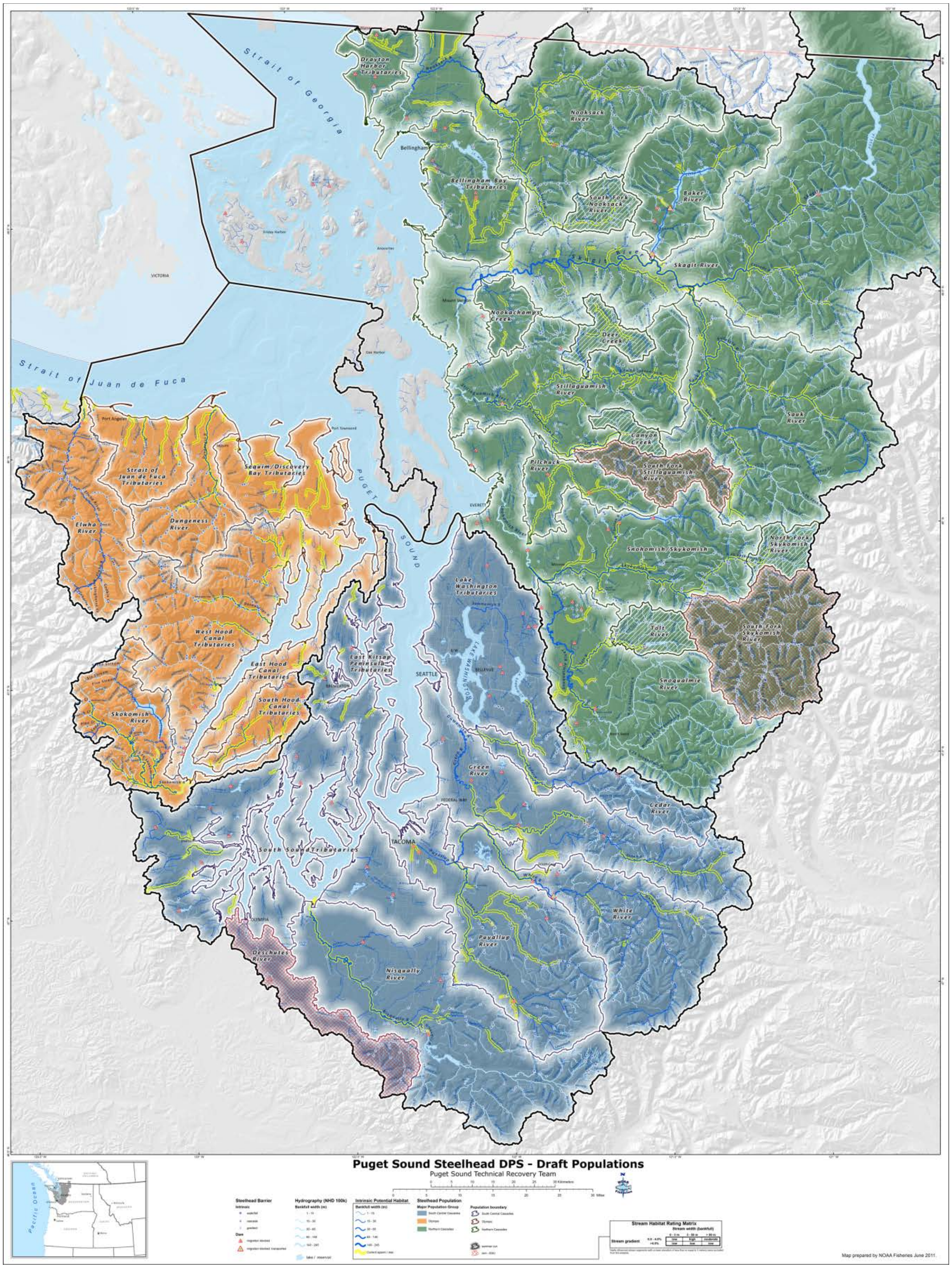


Figure 33. Three MPGs for the Puget Sound Steelhead DPS: Northern Cascades (green), Central and South Puget Sound (blue), and Hood Canal and Strait of Juan de Fuca (orange). Areas currently occupied by out-of-DPS steelhead introduced into historically inaccessible areas are in brown or purple (e.g., Chamber's Creek winter-run steelhead introduced (via fish ladder) above Tumwater Falls in the Deschutes River); crosshatched DIPs identify summer-run populations.

Decision Support Systems as Tools for Assessing Viability

Evaluating the viability of Puget Sound steelhead at the level of MPGs and the entire DPS requires methods that can accommodate not only qualitative and quantitative information, but also information of varying quality. In the following subsections, we describe some of these methods.

DSSs are increasingly being employed to assess status and viability when uncertainty about variables influencing viability is considered extensive (McCann et al. 2006, Newton et al. 2007, Newton 2010). A complete assessment of the biological condition of a complex entity such as a DPS is necessarily multifaceted, including a variety of interrelated criteria, with varying data quality. These criteria relate to biological processes at a variety of time and space scales, with processes varying from individual stream reaches to the entire range of the DPS. To track this large suite of data and criteria in a transparent and logically consistent framework, we used a knowledge-based DSS.

A DSS is a computer-based tool that can analyze and compare numerous types of data, producing results that assist managers in making a decision (Turban and Aronson 2001). DSSs, which include decision trees and approximate logic models, allow decision makers to perform complex evaluations quickly, present a consistent assessment that draws from a variety data sources, and track large sets of information accurately, thus improving the choices made without overriding human judgment (Rauscher 1999). A system evaluating ecological conditions may contain substantial uncertainty about the precise conditions that are optimal for the target organisms because of gaps in information and the lack of perfect knowledge about the interrelationships among relevant factors (Reynolds et al. 2000).

One type of DSS incorporates approximate logic (Zadeh 1965, Reynolds et al. 2000), popularly known as fuzzy logic, to assist evaluation of conditions and functional relationships based on imprecise information. These DSSs typically rely on relationships between influence variables and response variables referred to as truth membership functions. These functions can take any form—from linear, to a threshold or sigmoidal relationship, to nonlinear—so long as one is confident that the available information supports that characterization.

Wainwright et al. (2008) used this type of DSS to identify viability criteria for threatened Oregon coast coho salmon (*O. kisutch*). Their DSS used a fuzzy logic network framework to link criteria at a variety of scales and aggregate them from fine-scale watershed-level criteria, through population-level criteria and MPG-level criteria, to criteria for the entire coho salmon ESU. We attempted to adapt this tool for Puget Sound steelhead viability but ultimately did not employ it, because the lack of quantitative information available for Puget Sound steelhead made it difficult to develop robust truth membership functions to parameterize the model.

The PSS TRT opted instead to develop a novel tool for this purpose, another DSS model based on Bayesian networks (BNs) (Marcot et al. 2001, Newton et al. 2007). BNs represent a decision tool that provides a transparent, graphical framework for characterizing relationships among a diverse set of variables. They are increasingly being used to evaluate viability and project future status of organisms of conservation concern (e.g., polar bears [*Ursus maritimus*], Amstrup et al. 2008, 2010; marbled murrelets [*Brachyamphus marmoratus*], Steventon et al. 2003) and have been applied to salmonids (Rieman et al. 2001). Because these networks usually incorporate likelihoods (conditional probabilities) for each of the input variables, typically by using discrete uniform distributions to treat all values of a finite set of possible values as equally probable, they are effective models of reasoning under uncertainty (Jensen 2001). They also have the advantage that they can readily incorporate new information as it becomes available to reduce uncertainty in the decision process.

Bayesian Networks for Evaluation of Viability

Evaluating viability of Puget Sound steelhead at the level of DIP, MPG, and DPS requires an approach that can explicitly incorporate uncertainty into the evaluation and deal with available information of variable quality for all four VSP criteria. We sought a method that could accommodate these types of information in a transparent, consistent way. We elected to construct a series of BNs (aka Bayesian belief networks or Bayes nets per Newton et al. 2007) to assess viability at these multiple scales and ascertain the influence of factors on viability. A BN is a probabilistic graphical model (e.g., Lee and Rieman 1997, Marcot et al. 2001, Conroy et al. 2008, Peterson et al. 2008) designed to efficiently represent a probability distribution underlying a set of variables that influence each other, but that is too complex to be represented using a series of tables and equations. A main feature of a BN is that there is only one probability distribution that satisfies the structure of the BN (as reflected in the constraints among variables in the form of probabilistic conditional dependencies), and this distribution can be estimated by testing the BN by locking values of the variables at particular probabilities (a process known as instantiation). The objective of a BN is to characterize the dependencies among its constituent variables (depicted by nodes) given the influences between them (depicted by edges).

A BN is composed of nodes and edges. Each node represents a variable that can take any of several values, either as discrete states or as represented by a continuous function. Each edge connecting two nodes represents probabilistic dependencies among these nodes (the arrow indicates the direction of influence). In a BN, the conditional probabilities that reflect these dependencies are estimated from Bayes' theorem. In its simplest form, Bayes' theorem states that the probability of event b occurring given that event a has occurred, $P(b|a)$, is

$$P(b|a) = \frac{P(a|b) \text{ x } P(b)}{P(a)}$$

where $P(a)$ is the probability of a, $P(b)$ is the probability of b, and $P(a|b)$ is the probability of a given that b has occurred (Bayes and Price 1763, Gelman et al. 1995). We can demonstrate this relationship with a simple example. Suppose that we know from previous studies that slowly growing (event b) juvenile steelhead (i.e., progeny of steelhead parents) adopt a freshwater resident life history (event a), $P(a|b)$, about 10% of the time. Suppose we also know that the average probability of a growth rate sufficiently low to induce residency in this population, $P(b)$, is 0.3, and the probability of sampling a resident *O. mykiss* in this population which has anadromous access to the ocean is 1 in 5 (so, $P(a)$ is 0.2). We want to know how likely it is that a juvenile *O. mykiss* born of steelhead parents will become resident as a result of slow growth—that is, how likely is it that depressed growth (*b*) will result in residency (*a*) in this population? We can use Bayes' theorem to compute the probability that residency results from depressed growth, given this information

$$P(b|a) = \frac{0.1\ x\ 0.3}{0.2} = 0.15$$

Under these conditions, we would predict that if a fish born of anadromous parents becomes resident, then the likelihood that this is caused by depressed growth is 15%.

Another way of expressing Bayes' theorem is one that includes a hypothesis (*H*), evidence (*E*), and prior knowledge or past experience (*K*), so that the probability that a hypothesis is true given the evidence and past experience is

$$P(H|E,K) = \frac{P(H|K)\ \mathrm{x}\ P(E|H,K)}{P(E|K)}$$

where $P(H|K)$ is the probability of the hypothesis given prior knowledge, $P(E|K)$ is the probability of the evidence given prior knowledge, and $P(E|H,K)$ is the probability of the evidence given the hypothesis is true and prior knowledge. The term $P(H|E,K)$ is the posterior probability, or the probability of hypothesis *H* after considering the effect of the evidence *E* on prior knowledge *K*. The term $P(H|K)$ is the prior probability *H* given *K* alone. The term $P(E|H,K)$ is the likelihood and gives the probability of *E* given that both *H* and *K* are true. The term $P(E|K)$ is independent of *H* and is usually regarded as a normalizing or scaling constant (Gelman et al. 1995).

A BN is intended to directly incorporate estimates of uncertainty into evaluating relationships between variables. As a DSS, it can be as simple or as complex as one thinks is supportable. It has the advantage that its components—its criteria—can easily be constructed through simple verbal guidelines, actual data, or estimated functional relationships. A BN has some features that can favor its use.

- BNs can lay out parameters and how they interact transparently in a clear decision framework;
- BNs explicitly depict uncertainty as prior probabilities, which lend themselves well to risk analysis; and
- BNs can be used to help identify key factors that influence the outcome of interest or to help prioritize research, monitoring, or other actions.

BNs are not intended as a substitute for human judgment or as a means of making decisions, but rather to support decision-making processes by integrating available information and human judgment into a sensible conceptual framework.

Figure 34 depicts an example of a simple BN to evaluate viability of a population based on current abundance and estimated population growth rate. For this example and other analyses using BNs in this report, we used the program Netica v. 5.09 (Norsys Software Corp.,

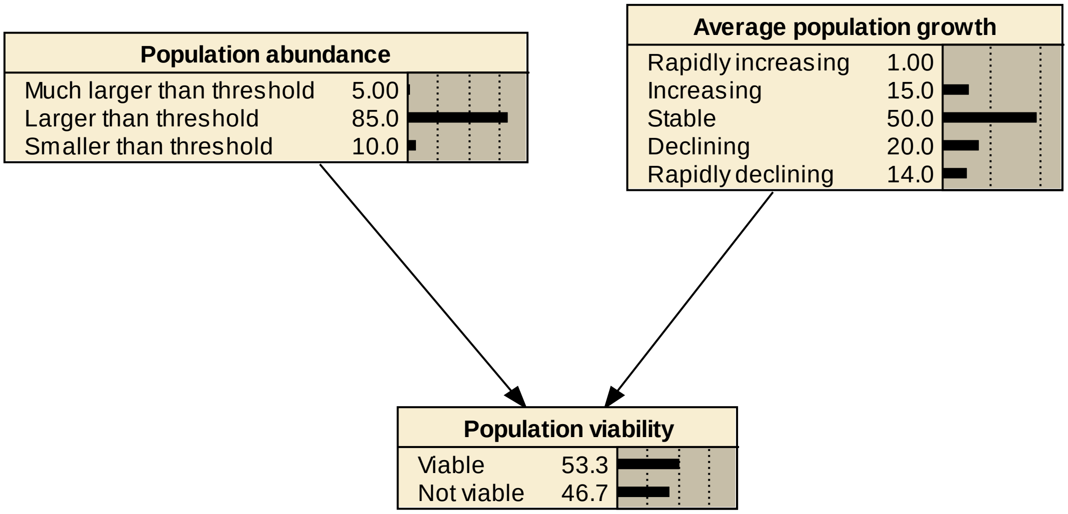


Figure 34. A simple BN depicting the influence of three discrete states of population abundance and five discrete states of average population growth on a dichotomous estimate of population viability. The bars and values in the two parental nodes (Population abundance and Average population growth) show estimates of the probabilities of each state for these factors; the bars and values in the child or outcome (decision-aiding) node (Population viability) depict resulting estimates of the probabilities of each viability state. The joint probabilities of all possible combinations of these states of Population abundance and Average population growth, which are inputs from the user, are given in Table 8.

Vancouver, British Columbia) to construct and evaluate the networks, following the general guidelines recommended by Marcot et al. (2006). In this BN, one parent node, “Population abundance,” summarizes an assessment of the current abundance of the population relative to a low-abundance threshold that would trigger concern (or remedial action); the other parent node, “Average population growth,” summarizes an assessment of the population’s growth rate (e.g., an estimate of the annual percent change in abundance). In this simple BN, we assume that these two factors alone determine “Population viability,” the child node of interest, and they do so independently.

For each parent node in the BN, the black bars and corresponding numbers depict prior probabilities of each state. In this example, we are using discrete states to describe each variable in the BN. For example, the assessment of current population abundance reflected in Figure 34 is that current abundance is larger than the population’s low-abundance threshold with a probability of 85%. However, because this estimate is likely to be obtained from sampling, it incorporates some uncertainty. To capture this uncertainty, in this case we have estimated that the probability that the population is much larger than this threshold (say, 10 × the threshold abundance) is 5%, and the probability that the population is smaller than this threshold is 10%. The assessment of the population’s growth (based, say, on an estimate of linear trend in abundance) is that the population is stable, but with a probability of only 50%. We think that there is some likelihood that the population is increasing (e.g., 1–2% annually) or rapidly increasing (>5% annually), here estimated at 16%, and we also think that there is some likelihood that the population is declining, here estimated at 34%. Note that the probabilities of each state must sum to 1.

The Population viability node is an output node; it represents the results of a decision-aiding model. In Figure 34 we have depicted the probability that the population is viable or not, given the probability distributions for population abundance and growth rate described above. The Bayesian calculations underlying the BN, parameterized with the probabilities given in the parent nodes for Population abundance and Average population growth and with a set of conditional probabilities that represent all the combinations of states for abundance and growth (Table 8), result in the probabilities estimated for viability (53.3%) or nonviability (46.7%) given in the child node for Population viability. The prior information that the population is thought to be somewhat larger than the low-abundance threshold and stable, albeit with some uncertainty, leads to a BN outcome of uncertainty that the population is viable, with probability 53.3%.

What if we are more certain about the population's state, for example, because of new information? One can instantiate the BN to correspond to particular states of the variables and examine how viability is affected by this additional information. Instantiation, the term used to indicate that node states are set at particular values, is important in conducting sensitivity analyses for a BN. For example, if additional information led us to conclude that the population is in fact larger than the low-abundance threshold (P = 100%) and its growth is positive (e.g., $\lambda > 1$, P = 100%), the BN estimates that the probability the population is viable is now 70% (Figure 35, top panel). It turns out that if we are certain that the population is much larger than the threshold and its growth is positive, the probability that the population is viable rises to 95%. If, on the other hand, additional information led us to conclude that the population is in fact smaller than the low-abundance threshold (P = 100%) and its growth is negative (e.g., $\lambda < 1$, P = 100%), the BN estimates that the probability the population is viable is now only 5% (Figure 35, bottom panel). From these results it is easy to see that the estimates directly reflect the corresponding values in the conditional probability table (CPT) (Table 8).

Table 8. CPT underlying the outcome node of population viability for the BN in Figure 34, representing the joint probabilities of all possible combinations of states of population abundance and population growth described in the BN's parent nodes.

Population abundance	**Average population growth**	**Viable**	**Not viable**
Much larger than threshold	Rapidly increasing	0.99	0.01
	Increasing	0.95	0.05
	Stable	0.90	0.10
	Declining	0.75	0.25
	Rapidly declining	0.50	0.50
Larger than threshold	Rapidly increasing	0.80	0.20
	Increasing	0.70	0.30
	Stable	0.60	0.40
	Declining	0.45	0.55
	Rapidly declining	0.35	0.65
Smaller than threshold	Rapidly increasing	0.50	0.50
	Increasing	0.40	0.60
	Stable	0.30	0.70
	Declining	0.05	0.95
	Rapidly declining	0.01	0.99

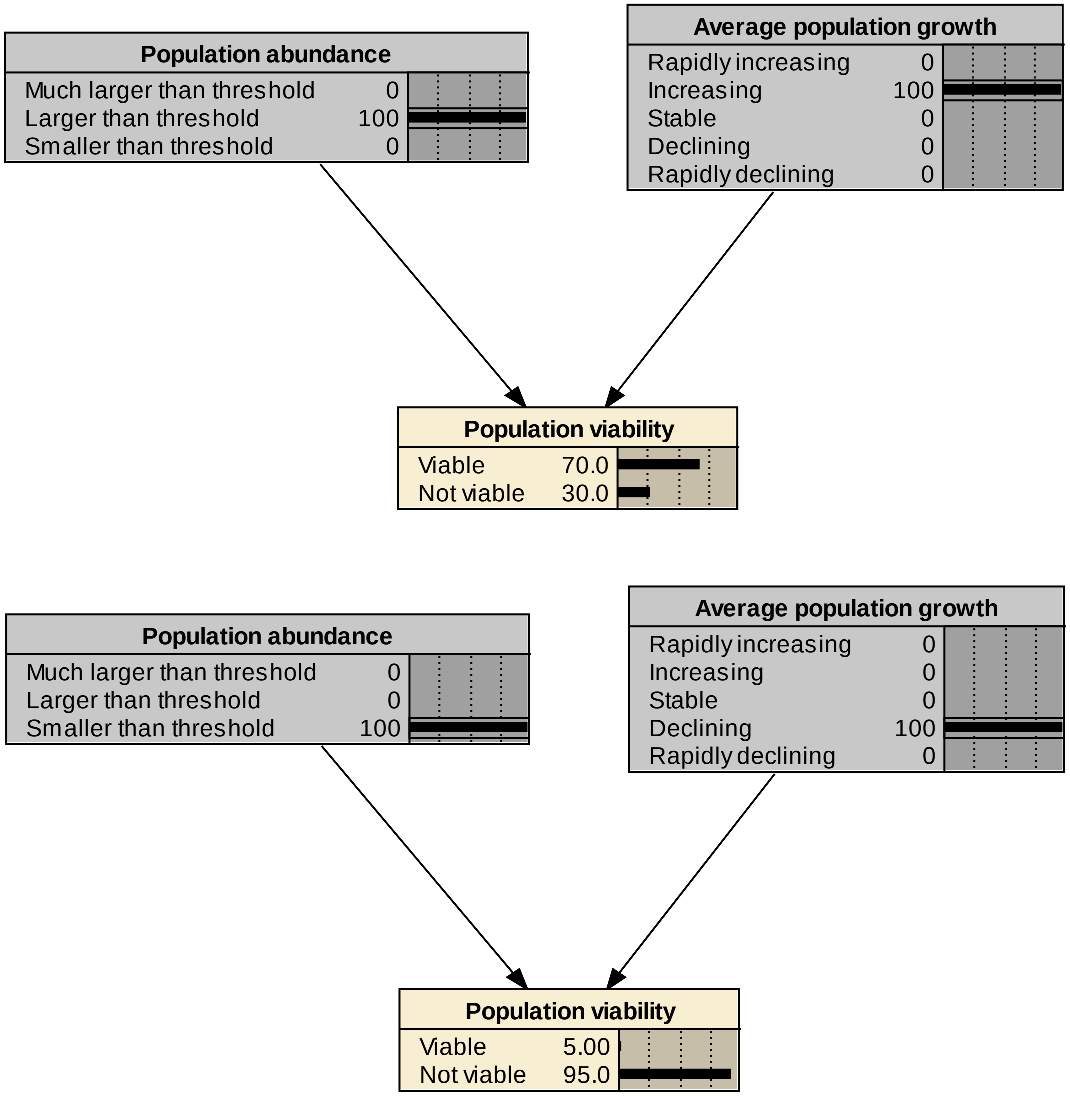


Figure 35. Top panel, the BN in Figure 34 instantiated at particular states of population abundance and growth, depicting certainty that the population is larger than the low-abundance threshold and increasing in size. Note how this additional knowledge has altered the assessment of viability, given these prior probabilities and the CPT in Table 8. Bottom panel, the BN in Figure 34 instantiated at different states for population abundance and growth, depicting certainty that the population is smaller than the low-abundance threshold and declining in size.

To determine the most likely configuration of probabilities in the network given the evidence, we can estimate the most probable explanation (MPE) for this network. The MPE is a complete variable instantiation across the entire network that gives the highest probability, given the current evidence (distributions of conditional probabilities). The MPE is equivalent to the maximum-a-posteriori probability, which asks the question, Given the values at some nodes, what are the most likely findings across the network? In a Bayesian sense, it can be considered an estimate of the most plausible explanation of the data. It is computed as follows. The probability of a complete variable instantiation $\boldsymbol{x}$ is

$$\Pr(\boldsymbol{x}) = \prod_{\mathfrak{x}\boldsymbol{u}\sim\boldsymbol{x}} \theta_{\mathfrak{x}|\boldsymbol{u}}$$

where ~ is the compatibility among instantiations (i.e., $\mathfrak{x}\boldsymbol{u}$ is compatible with $\boldsymbol{x}$). An MPE is a complete variable instantiation given evidence $\boldsymbol{e}$ that has the highest probability

$$\begin{aligned} \mathrm{MPE}(\boldsymbol{e}) &\overset{\text{def}}{=} \arg^{\max}_{\mathrm{x}\sim\mathrm{e}} \Pr(\boldsymbol{x}) \\ &= \arg^{\max}_{\mathrm{x}\sim\mathrm{e}} \prod_{\mathfrak{x}\boldsymbol{u}\sim\boldsymbol{x}} \theta_{\mathfrak{x}|\boldsymbol{u}} \end{aligned}$$

Note that there may be more than one MPE for a network (i.e., more than one complete instantiations with the same probability).

For the network in Figure 34, the MPE is shown in Figure 36. It indicates that, given the data, our best guess is that the population abundance is larger than the threshold, that its dynamics are stable (growth rate near zero), and that the population is viable. However, although the MPE indicates that the population is most likely viable (i.e., that probability is instantiated at 100%), the estimated probability that the population is not viable is still rather high (66.7%) given the MPE, reflecting the considerable uncertainty about the population's status in both size and, especially, growth rate. Note that under MPE the "probabilities" given within a node in a network are not additive. The reason for this is that in computing MPE across the network, Netica software represents the MPE probability within each node as a full bar (i.e., *P* = 100%; the actual probability for that state given the evidence is given by the uninstantiated network without computing MPE. Figure 34 indicates that this value is 53.3%. Figure 36 indicates that the MPEs for abundance, growth, and viability are Larger than threshold, Stable, and Viable, respectively. These states do correspond to the largest probabilities for these respective nodes (85%, 50%, and 53.3%, respectively) in the uninstantiated network in Figure

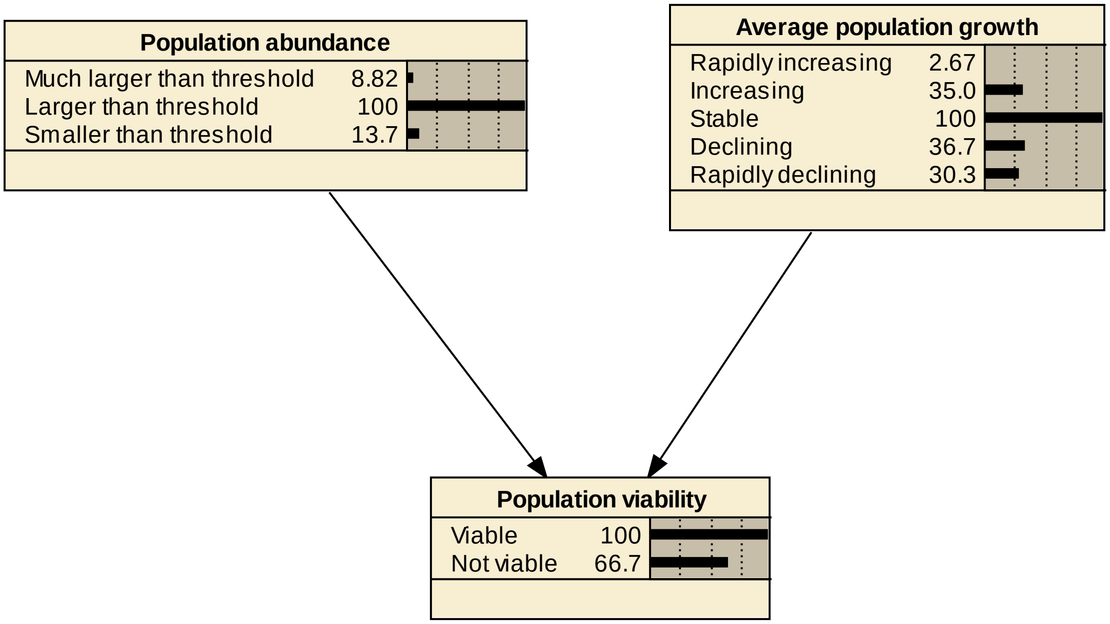


Figure 36. MPE for the BN in Figure 34.

34. In computing the MPE for the network, Netica software simply estimates the probabilities of the alternate states at each node, given that the other nodes are in their most probable configurations (and scaling by the same factor that was used to bring the MPE bar to 100%. Since MPE is computed and probabilities are adjusted at each node, complex networks can be difficult to characterize simply. This is the primary value of MPE in a BN—it identifies the most likely states of each variable given the evidence and quantifies their likelihoods relative to those of the other states.

As expected, the viability outcome directly depends on the set of probabilities used to parameterize the BN. It is important to remember that these probabilities, including those in the CPT, can represent estimates obtained from best judgement (informed opinion or belief), actual data, or a combination of these sources of information, and they can be revised when additional information becomes available (Marcot et al. 2006, Newton et al. 2007).

We can conduct a sensitivity analysis of the BN to determine how sensitive the population viability results are to changes in the states of population abundance and growth. This sensitivity is often measured by estimating the entropy reduction (or reduction in mutual information) or belief variance (expected reduction in real variance) resulting from the degree to which findings at a node are influenced by the findings at another node, given the findings that are entered into the network. In information theory, entropy is a measure of the uncertainty of a random variable(s). Mutual information corresponds to the reduction in the entropy of a variable achieved by learning the state of another variable. In a BN, entropy reduction (in the case of discrete or categorical variable states) or variance reduction (in the case of continuous variables) is the expected reduction in variation at an output node due to a finding at an input node (Marcot et al. 2006). It is calculated by

$$I = \mathrm{H(Q)} - H(Q|F) = \sum_{\mathrm{q}} \sum_{\mathrm{f}} \frac{\mathrm{P(q,f)log_2[P(q,f)]}}{\mathrm{P(q)P(f)}}$$

where I is the entropy reduction, H(Q) is the entropy of output node Q before any new findings from input node F, H(Q|F) is the entropy of Q after new findings from F, P(q,f) is the probability of finding q at node Q given finding f at node F, and P(q) and P(f) are the probabilities of findings q and f, respectively (Appendix B in Marcot et al. 2006). In structuring a BN, one seeks to maximize entropy of the entire network, which corresponds to maximizing the mutual information among the variables. For example, an entropy reduction of 5% at node Q resulting from a new finding at node F reflects a reduction in mutual information of 5%; this means that the uncertainty in the state of Q is reduced by 5% by knowing the state of F.

The results of a sensitivity analysis (Table 9) show that the estimate of viability is sensitive to variation in both population abundance and growth rate, but slightly more sensitive to changes in abundance for this particular BN; the entropy reduction for sensitivity of viability to change in abundance was 4.27%, whereas that for sensitivity to change in productivity was 4.07%. This is largely because the population’s viability changes considerably when its size drops below the low-abundance threshold, as depicted by the probabilities in the CPT (Table 8).

Table 9. Results of a sensitivity analysis of the BN in Figure 34, showing the sensitivity of population viability to changes in the states of population abundance and growth rate.

	Probability ranges			**Root mean**
Viability	Minimum	Current	Maximum	**square change**
Sensitivity to findings at abundance[a]				
Viable	0.2264	0.533	0.8224	0.1179
Not viable	0.1776	0.467	0.7736	0.1179
Sensitivity to findings at growth rate[b]				
Viable	0.3235	0.533	0.7795	0.1174
Not viable	0.2205	0.467	0.6765	0.1174

[a] Entropy reduction = 0.0426 (4.27%) and belief variance = 0.01389 (5.58%).
[b] Entropy reduction = 0.0406 (4.07%) and belief variance = 0.01379 (5.54%).

Use of Bayesian Networks to Incorporate Viable Salmonid Population Criteria into a Viability Assessment Framework

We constructed a BN to characterize the influence of a number of features of coastal steelhead biology and estimates of the four VSP parameters—abundance, productivity, diversity, and spatial structure—for individual populations on viability at the level of the DIP, the MPG, and the entire DPS. In essence, a BN is a set of variables (nodes) linked by probabilities that represent their interdependence. Our objective in building the BN model was to evaluate the net effects of several features of steelhead populations related to VSP parameters and their habitat on the viability of populations across the entire Puget Sound DPS. We did so by synthesizing available information and expert knowledge on abundance, productivity, diversity, and spatial structure of Puget Sound steelhead. The BN that determines DPS viability is composed of three subnetworks that determine the viability of each MPG. Each of these MPG subnetworks is composed of a series of subnetworks that determine the viability of each DIP in the constituent MPG. A representative BN to characterize the viability of a typical winter-run steelhead population in Puget Sound is depicted in Figure 37.

Our goal was to construct each such network to minimize its complexity while still retaining the capacity to evaluate the influence of all four VSP parameters on viability. We aimed to keep the number of nodes small but include key variables for which we either had available information or else thought information might become available later that could assist with viability assessment and future recovery planning. Consequently, some nodes can be considered placeholders with underlying conditional probabilities that reflect considerable uncertainty. As indicated by Figure 37, the effect of DIP abundance on DIP viability incorporates influences of adult and juvenile abundance relative to capacity for production, and quasi-extinction risk (as estimated from a PVA, if available). The effect of DIP productivity on DIP viability incorporates influences of population growth rate and frequency of repeat spawning; in the network, population growth rate is influenced by both freshwater survival (smolts per spawner) and marine survival (adults per smolt). Next, the influence of DIP diversity on DIP viability incorporates influences of the distribution of run timing, influence of hatchery fish on natural diversity, the adult age distribution, and the proportion of migrant smolts

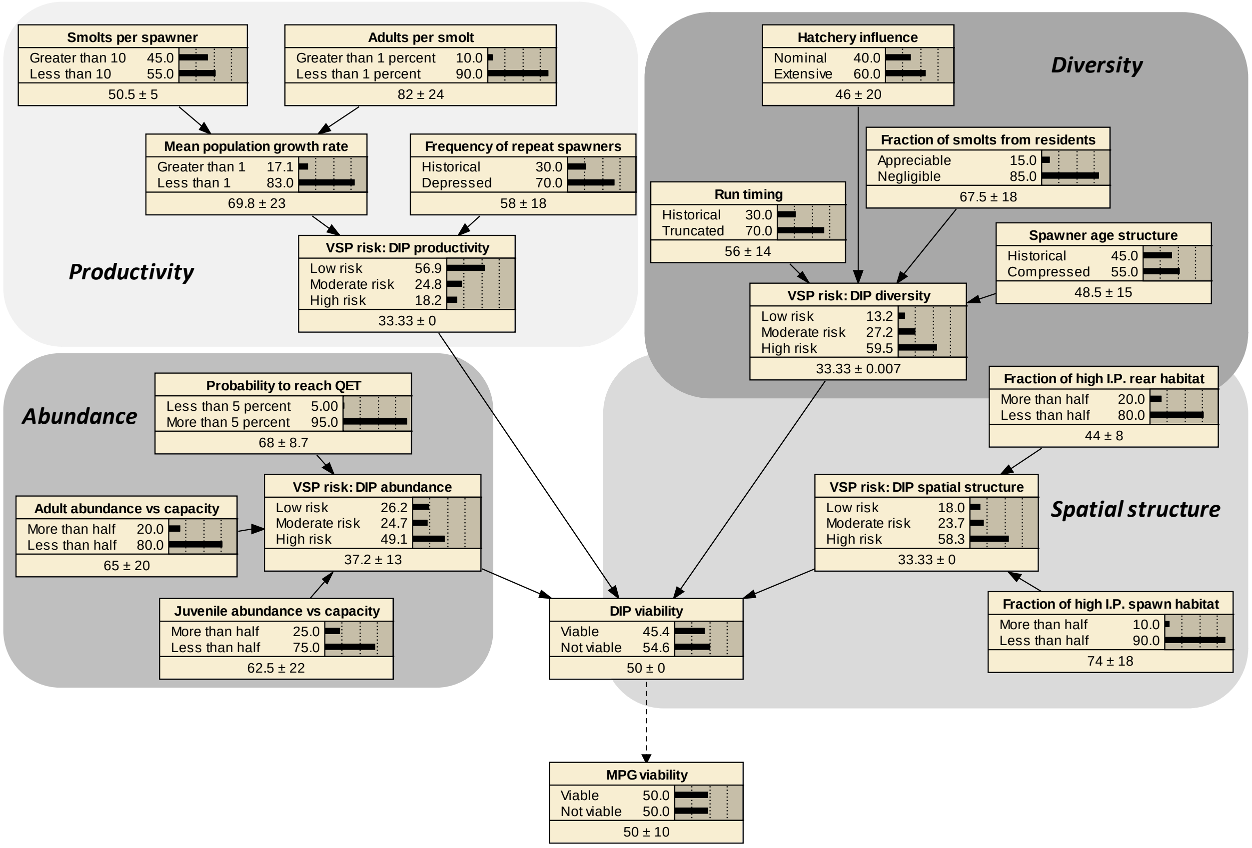


Figure 37. A BN to characterize the viability of a hypothetical DIP of winter-run steelhead in the Puget Sound DPS. The network is intended to be minimally complex while estimating the influence of all four VSP parameters (abundance, productivity, diversity, and spatial structure) on population viability. The influence of the DIP's abundance on its viability is represented by the "VSP risk: DIP abundance" node (lower left) that incorporates influences of adult abundance, juvenile abundance, and quasi-extinction risk (as estimated from a PVA, if possible) on DIP viability. The influence of the DIP's productivity on its viability is represented by the "VSP risk: DIP productivity" node (upper left) that incorporates influences of population growth rate and frequency of repeat spawning on DIP viability; the node for population growth rate is itself influenced by freshwater survival (smolts per spawner) and marine survival (adults per smolt). The influence of the DIP's diversity on its viability is represented by the "VSP risk: DIP diversity" node (upper right) that incorporates influences of the distribution of run timing, influence of hatchery fish on natural diversity, the adult age distribution, and the proportion of migrant smolts produced by resident adults. The influence of the DIP's spatial structure on its viability is represented by the "VSP risk: DIP spatial structure" node (lower right) that incorporates influences of the fraction of IP habitat occupied by rearing juveniles and the fraction of IP habitat occupied by spawning adults. This network is a subnetwork that then determines the viability of the corresponding MPG, as indicated by the dashed arrow at bottom; similar subnetworks for the viability of each DIP in the MPG combine to influence the MPG's overall viability. A description of the nodes in the network is in Table 10. The underlying CPTs for this subnetwork are in Table 11 and Table 12.

produced by resident adults. Finally, the influence of DIP spatial structure on DIP viability incorporates influences of the fraction of IP habitat occupied by rearing juveniles and the fraction of IP habitat occupied by spawning adults.

In this network, the values for the input nodes and the conditional probabilities underlying each node were assigned by TRT members after considering available data, opinions and comments from other steelhead biologists, reviews of other models and analyses, reports, and similar approaches in the scientific literature (see References section and Appendix A). In most cases, the intermediate and output conditional probabilities pertain to steelhead demographic and life history responses that are poorly understood; in these cases, we attempted to account for this uncertainty by assigning moderate to high levels of spread in the conditional probabilities. In general, probability values were chosen to produce a linear relationship between the score for a VSP category and the DIP viability (with a consequent distribution of probability of being viable ranging from 0 to 100%), with all VSP criteria weighted equally in the network. A description of the nodes in the network for DIP viability in Figure 37 and Figure 38 is in Table 10. The probabilities that underlie the network are in Table 11 and Table 12.

We initially weighted abundance and productivity criteria equally with diversity and spatial structure criteria in our network CPTs, even though the evidence was that DIP viability for Puget Sound steelhead was somewhat more dependent on how current abundance compared with historical estimates, based on preliminary networks constructed before weighting (Figure 39). Comparisons of estimates of relative viability for Puget Sound steelhead DIPs between current (“depressed”) and historical estimates of the four VSP parameters (abundance, productivity, diversity, and spatial structure) point to the primary importance of abundance to DIP viability. Therefore, after constructing each network, in assessing final categories of viability for each DIP we weighted mean abundance and productivity criteria more heavily than mean diversity and spatial structure criteria for this reason and because more quantitative demographic information was available (see Viability Criteria for Puget Sound Steelhead section of this technical memorandum).

Our approach to parameterizing the BNs for each DIP was as follows. For the abundance criterion, we first computed each DIP’s geometric mean abundance over the most recent 5 years from available abundance data. For DIPs that have no recent abundance data, we used the mean geometric mean abundance for all DIPs within the MPG, scaled by the estimate of total IP area for that DIP, as the input variable. (For the Northern Cascades MPG, we excluded the Skagit River basin for these calculations for DIPs with no abundance data because of its uniquely large size among drainages within the MPG.) We computed the probability that the DIP would reach its estimated QET abundance within 100 years; the QET for each DIP was based on the DIP habitat’s estimate of IP, the empirical Puget Sound steelhead parr densities computed by Gibbons et al. (1985), using an estimate of 0.0754 parr/m^2, and the Chapman (1981) estimate of parr-to-smolt survival of 0.30 to estimate smolt productivity at 0.023 smolts/m^2. For each DIP, we calculated the probability of reaching QET using MARSS algorithms applied to the abundance data (as described in the earlier Viability Assessment Methods section under the Population Viability Analysis With Autoregressive State-Space Models subsection). For those DIPs for which we could not calculate QETs, we used the mean QET computed for the associated MPG as the input variable.

Table 10. Titles and descriptions of the input, intermediate, and decision nodes in the BN describing population viability of steelhead. See the immediately previous text for a description of how the input probabilities were calculated for these nodes, and Table 11 and Table 12 for the final input probabilities used.

Node title	Node description	States
Input nodes: VSP abundance		
Adult abundance vs. capacity	The probability that the fraction of estimated spawner carrying capacity represented by the current geometric mean adult abundance in the DIP is <25% or >25%	Greater than 25% Less than 25%
Juvenile abundance vs. capacity	The probability that the fraction of estimated rearing capacity represented by the current mean parr or smolt abundance in the DIP is <25% or >25%	Greater than 25% Less than 25%
Probability to reach QET	The probability that the QET estimated for this DIP will be reached within 100 years (from a PVA, if available)	Greater than 20% Less than 20%
Intermediate node: VSP abundance	The probability that the DIP's abundance poses a risk to DIP viability	Low risk Moderate risk High risk
Input nodes: VSP productivity		
Mean population growth rate	The probability that the mean population growth rate estimated for this DIP is <1 (declining population size) or >1 (increasing population size). This metric is influenced by two major components: smolts per spawner and adults per smolt (below)	Greater than 1 Less than 1
Smolts per spawner	A measure of productivity in freshwater (and a component of population growth rate), it is the probability that the number of smolts produced per spawner in the DIP is <50 or >50	Greater than 50 Less than 50
Adults per smolt	A measure of productivity in the ocean (and a component of population growth rate), it is the probability that the smolt to adult survival rate in the DIP is <2% or >2%	Greater than 2% Less than 2%
Frequency of repeat spawners	The probability that the frequency of repeat spawning in this DIP is historical (e.g., 10–20%) or compressed (<10%)	Historical Depressed
Intermediate node: VSP productivity	The probability that the DIP's productivity poses a risk to DIP viability	Low risk Moderate risk High risk

Table 10 continued. Titles and descriptions of the input, intermediate, and decision nodes in the BN describing population viability of steelhead. See the immediately previous text for a description of how the input probabilities were calculated for these nodes, and Table 11 and Table 12 for the final input probabilities used.

Node title	Node description	States
Input nodes: VSP diversity		
Run timing	The probability that the current distribution of spawn timing (within a run type) in the DIP is historical vs. truncated or altered	Historical Altered
Hatchery influence	The probability that hatchery influence from Chambers Creek WSH or Skamania SSH stock in the DIP is nominal or extensive (e.g., hatchery plants have occurred for >5 years or involved more than 500,000 fish)	Nominal Extensive
Fraction of smolts from residents	The probability that the fraction of smolts produced by residents in the DIP is appreciable or negligible (e.g., the fraction of smolts produced by residents is >10% or <10%)	Appreciable Negligible
Input Nodes: VSP spatial structure		
Fraction of IP spawning habitat	The fraction of estimated available steelhead spawning habitat area in the DIP (IP) under historical conditions utilized by the current distribution of spawners. In the network's current form, this is the estimated fraction (%) of accessible spawning habitat that is occupied	Greater than 20% Less than 20%
Fraction of IP rearing habitat	The fraction of estimated available steelhead rearing habitat area in the DIP (IP) under historical conditions utilized by the current distribution of rearing juveniles. In the network's current form, this is the estimated fraction (%) of accessible rearing habitat that is occupied	Greater than 20% Less than 20%
Intermediate node: VSP spatial structure	The probability that the DIP's spatial structure poses a risk to DIP viability	Low risk Moderate risk High risk
Decision node: DIP viability	The probability that the DIP is viable (based on the collective influence of the four VSP parameters on DIP viability)	Viable Not viable
Decision node: MPG viability	The probability that the MPG is viable (based on the collective influence of the viabilities of each of its composite DIPs)	Viable Not viable

Table 11. CPT for the input nodes describing risk for each VSP parameter for the BNs describing viability of Puget Sound steelhead populations (DIPs). The probabilities input for each DIP were estimated from data, PVA results, and expert opinion.

Abundance						**Productivity**					**Diversity**							**Spatial structure**				
			Risk					**Risk**							**Risk**					**Risk**		
Adult abund. vs. capacity	**Juvenile abund. vs. capacity**	***P*(QET)**	Low	Moderate	High	**Mean growth rate[a]**	**Frequency repeats**	Low	Moderate	High	**Run timing**	**Fraction smolts from residents**	**Age structure**	**Hatchery influence**	Low	Moderate	High	**Fraction IP rearing habitat**	**Fraction IP spawn habitat**	Low	Moderate	High
>25	>25	<10	90	9	1	>1	Hist[b]	90	9	1	Hist	App[b]	Hist	Nom[b]	90	9	1	>20	>20	70	25	5
>25	>25	>10	75	15	10	>1	Dep[b]	70	20	10	Hist	App	Hist	Ext[b]	69	20	11	>20	<20	50	20	30
>25	<25	<10	60	15	25	<1	Hist	40	40	20	Hist	App	Com[b]	Nom	60	30	10	<20	>20	50	20	30
>25	<25	>10	40	20	40	<1	Dep	1	9	90	Hist	App	Com	Ext	54	25	21	<20	<20	5	25	70
<25	>25	<10	15	25	60						Hist	Neg[b]	Hist	Nom	40	40	20					
<25	>25	>10	10	20	70						Hist	Neg	Hist	Ext	39	30	31					
<25	<25	<10	15	20	65						Hist	Neg	Com	Nom	20	50	30					
<25	<25	>10	1	9	90						Hist	Neg	Com	Ext	19	40	41					
											Alt[b]	App	Hist	Nom	10	50	40					
											Alt	App	Hist	Ext	9	40	51					
											Alt	App	Com	Nom	10	40	50					
											Alt	App	Com	Ext	9	30	61					
											Alt	Neg	Hist	Nom	5	35	60					
											Alt	Neg	Hist	Ext	4	25	71					
											Alt	Neg	Com	Nom	5	20	75					
											Alt	Neg	Com	Ext	1	9	90					

[a] For calculating mean growth rate values (>1 and <1): **if** the adults per smolt is >2% and the smolts per spawner is >50%, **then** the probability that the growth rate is >1 is 95% and the probability that the growth rate is <1 is 5%; **if** the adults per smolt is >2% and the smolts per spawner is <50%, **then** the probability that the growth rate is >1 is 40% and the probability that the growth rate is <1 is 60%; **if** the adults per smolt is <2% and the smolts per spawner is >50%, **then** the probability that the growth rate is >1 is 20% and the probability that the growth rate is <1 is 80%; and **if** the adults per smolt is <2% and the smolts per spawner is <50%, **then** the probability that the growth rate is >1 is 5% and the probability that the growth rate is <1 is 95%.

[b] Key: Hist = historical, Dep = depressed, Alt = altered, App =appreciable, Neg = negligible, Com = compressed, Nom = nominal, and Ext = extensive.

Table 12. CPT describing the influence of each of the four VSP parameters on DIP viability for the BNs describing viability of Puget Sound steelhead. Note that the values in the first four columns are risk values (e.g., a value of “Low” for DIP abundance implies a low risk to DIP viability posed by abundance in the DIP). The values were chosen to produce a linear relationship between VSP category and DIP viability (with a consequent distribution of probability of being viable ranging from 0 to 100%), with all VSP criteria weighted equally.

VSP risk				**DIP viability**	
DIP abundance	DIP productivity	DIP diversity	DIP spatial structure	Viable	Not viable
Low	Low	Low	Low	100.0	0.0
Low	Low	Low	Moderate	91.7	8.3
Low	Low	Low	High	83.3	16.7
Low	Low	Moderate	Low	91.7	8.3
Low	Low	Moderate	Moderate	83.3	16.7
Low	Low	Moderate	High	75.0	25.0
Low	Low	High	Low	83.3	16.7
Low	Low	High	Moderate	75.0	25.0
Low	Low	High	High	66.7	33.3
Low	Moderate	Low	Low	83.3	16.7
Low	Moderate	Low	Moderate	75.0	25.0
Low	Moderate	Low	High	66.7	33.3
Low	Moderate	Moderate	Low	75.0	25.0
Low	Moderate	Moderate	Moderate	66.7	33.3
Low	Moderate	Moderate	High	58.3	41.7
Low	Moderate	High	Low	66.7	33.3
Low	Moderate	High	Moderate	58.3	41.7
Low	Moderate	High	High	50.0	50.0
Low	High	Low	Low	66.7	33.3
Low	High	Low	Moderate	58.3	41.7
Low	High	Low	High	50.0	50.0
Low	High	Moderate	Low	58.3	41.7
Low	High	Moderate	Moderate	50.0	50.0
Low	High	Moderate	High	41.7	58.3
Low	High	High	Low	50.0	50.0
Low	High	High	Moderate	41.7	58.3
Low	High	High	High	33.3	66.7
Moderate	Low	Low	Low	83.3	16.7
Moderate	Low	Low	Moderate	75.0	25.0
Moderate	Low	Low	High	66.7	33.3
Moderate	Low	Moderate	Low	75.0	25.0
Moderate	Low	Moderate	Moderate	66.7	33.3
Moderate	Low	Moderate	High	58.3	41.7
Moderate	Low	High	Low	66.7	33.3
Moderate	Low	High	Moderate	58.3	41.7
Moderate	Low	High	High	50.0	50.0
Moderate	Moderate	Low	Low	66.7	33.3
Moderate	Moderate	Low	Moderate	58.3	41.7
Moderate	Moderate	Low	High	50.0	50.0
Moderate	Moderate	Moderate	Low	58.3	41.7
Moderate	Moderate	Moderate	Moderate	50.0	50.0

Table 12 continued. CPT describing the influence of each of the four VSP parameters on DIP viability for the BNs describing viability of Puget Sound steelhead. Note that the values in the first four columns are risk values (e.g., a value of "Low" for DIP abundance implies a low risk to DIP viability posed by abundance in the DIP). The values were chosen to produce a linear relationship between VSP category and DIP viability (with a consequent distribution of probability of being viable ranging from 0 to 100%), with all VSP criteria weighted equally.

VSP risk				**DIP viability**	
DIP abundance	DIP productivity	DIP diversity	DIP spatial structure	Viable	Not viable
Moderate	Moderate	Moderate	High	41.7	58.3
Moderate	Moderate	High	Low	50.0	50.0
Moderate	Moderate	High	Moderate	41.7	58.3
Moderate	Moderate	High	High	33.3	66.7
Moderate	High	Low	Low	50.0	50.0
Moderate	High	Low	Moderate	41.7	58.3
Moderate	High	Low	High	33.3	66.7
Moderate	High	Moderate	Low	41.7	58.3
Moderate	High	Moderate	Moderate	33.3	66.7
Moderate	High	Moderate	High	25.0	75.0
Moderate	High	High	Low	33.3	66.7
Moderate	High	High	Moderate	25.0	75.0
Moderate	High	High	High	16.7	83.3
High	Low	Low	Low	66.7	33.3
High	Low	Low	Moderate	58.3	41.7
High	Low	Low	High	50.0	50.0
High	Low	Moderate	Low	58.3	41.7
High	Low	Moderate	Moderate	50.0	50.0
High	Low	Moderate	High	41.7	58.3
High	Low	High	Low	50.0	50.0
High	Low	High	Moderate	41.7	58.3
High	Low	High	High	33.3	66.7
High	Moderate	Low	Low	50.0	50.0
High	Moderate	Low	Moderate	41.7	58.3
High	Moderate	Low	High	33.3	66.7
High	Moderate	Moderate	Low	41.7	58.3
High	Moderate	Moderate	Moderate	33.3	66.7
High	Moderate	Moderate	High	25.0	75.0
High	Moderate	High	Low	33.3	66.7
High	Moderate	High	Moderate	25.0	75.0
High	Moderate	High	High	16.7	83.3
High	High	Low	Low	33.3	66.7
High	High	Low	Moderate	25.0	75.0
High	High	Low	High	16.7	83.3
High	High	Moderate	Low	25.0	75.0
High	High	Moderate	Moderate	16.7	83.3
High	High	Moderate	High	8.3	91.7
High	High	High	Low	16.7	83.3
High	High	High	Moderate	8.3	91.7
High	High	High	High	0.0	100.0

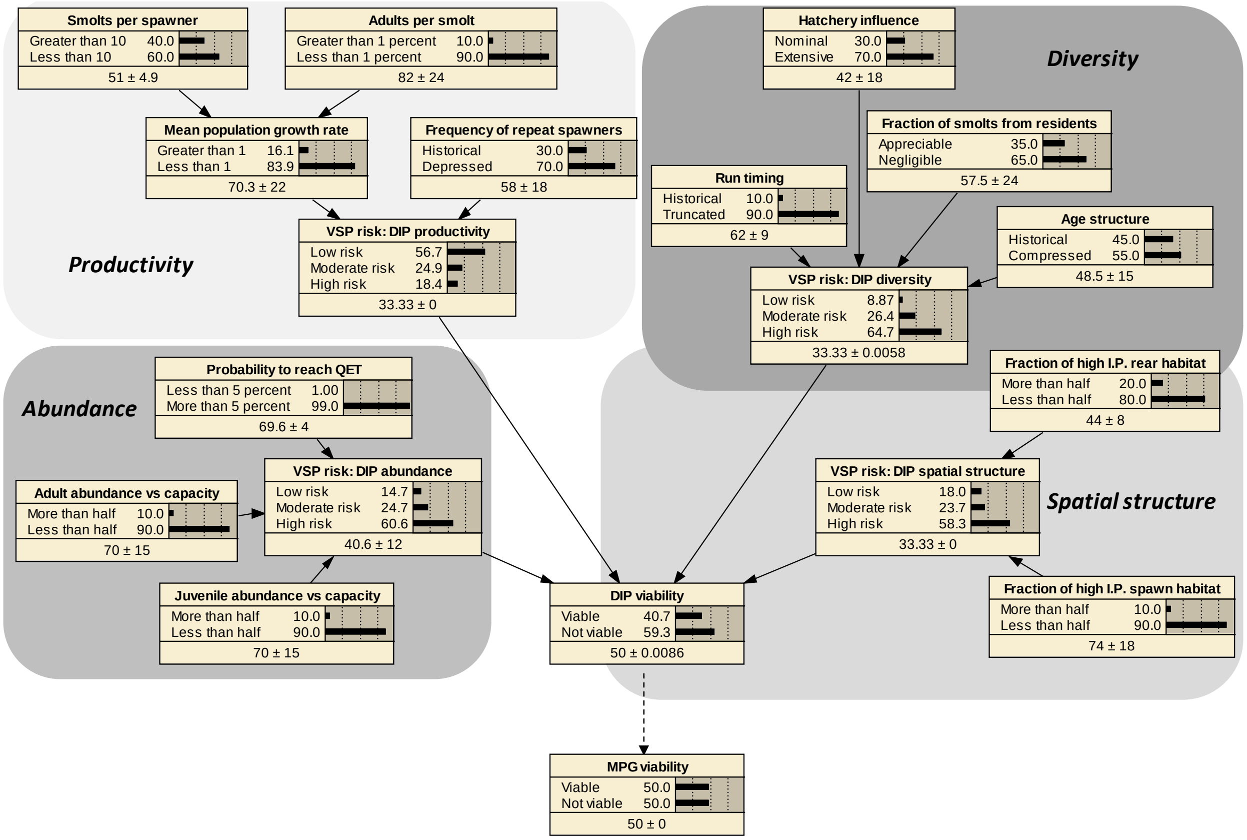


Figure 38. A BN to characterize the viability of a hypothetical DIP of summer-run steelhead in the Puget Sound DPS. The primary differences between this BN and that for winter-run steelhead are in the conditional probabilities underlying abundance, productivity, and diversity (especially the run timing and hatchery influence nodes). See the Figure 37 caption for details.

We used the IP metrics (i.e., the sum of low, medium, and high IP areas in reaches accessible by steelhead) and the juvenile steelhead densities estimated by Gibbons et al. (1985) and Chapman (1981) for each DIP to calculate parr and smolt densities and spawner abundances, based on 1%, 5%, and 20% marine survival rates; we considered these minimum, viable, and capacity abundances, respectively (see Appendix C). We then parameterized the influence of abundance on probabilities of viability in the BNs for each DIP using a compound rule: if the estimated geometric mean abundance was less than 25% of the adult abundance calculated from occupied IP area at a density of at least 1 fish/km at a marine survival rate less than 1%, the probability of viability was 20%; if the abundance was less than that for 1 fish/km at a marine survival rate less than 5%, the probability was 40%; if the abundance was less than that for 1 fish/km at a marine survival of 20%, the probability was 80%; otherwise, the probability was 90%. Using this rule, the probabilities of DIP viability based on abundance alone ranged from 20 to 90%. In addition, we used the output from the PVAs for available DIPs to estimate the probability that a population was near its QET abundance. We did so in this way: if the ratio of the recent (last 5 years) geometric mean abundance to the QET was greater than 10, the probability that abundance was near QET was 10%; if this ratio was greater than 8, the probability was 20%; if this ratio was greater than 6, the probability was 40%; if this ratio was greater than 4, the probability was 60%; if this ratio was greater than 2, the probability was 80%;

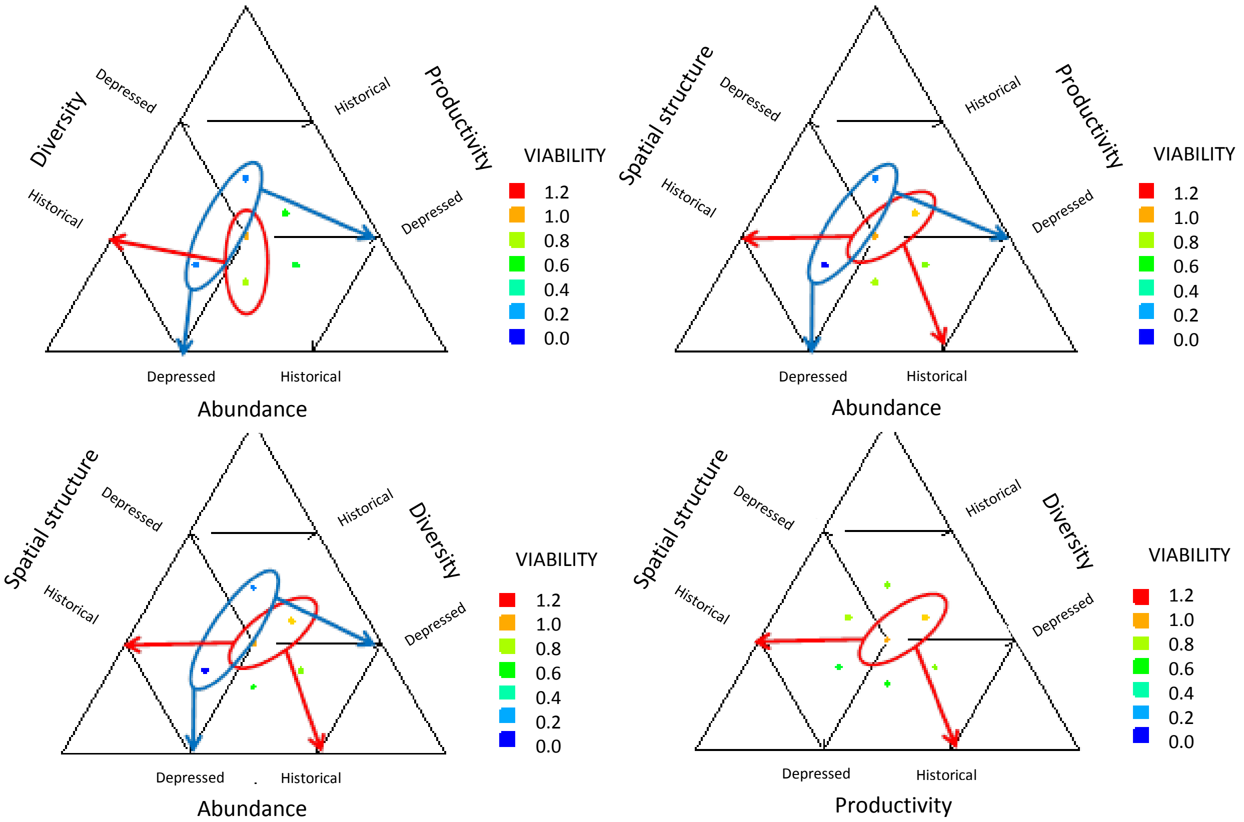


Figure 39. Comparisons of estimates of relative viability for Puget Sound steelhead DIPs between current (“depressed”) and historical estimates of VSP parameters (abundance, productivity, diversity, and spatial structure). The red ovals encompass the highest estimates of DIP viability; the blue ovals encompass the lowest estimates. Note that the blue ovals tend to be shifted to the left of the red ovals, indicating the important contribution of abundance to DIP viability.

otherwise, the probability was 90%. We applied these probabilities to both adult and juvenile abundances, because we had almost no information on historical or current juvenile abundances. These probabilities were then entered into each BN’s CPT for the abundance criterion and combined with the other probabilities in Table 11 to determine the overall contribution of abundance to DIP viability.

For the productivity criterion, we relied primarily on the long-term population growth rate (λ) estimated by the PVAs from MARSS. For DIPs that we could calculate this metric for, we used the mean growth rate computed for all DIPs within the MPG. We parameterized the influence of productivity on probabilities of viability in the BNs for each DIP in this way: if the point estimate of growth rate was greater than 1.1, the probability of viability was 90%; if the growth rate was between 1.0 and 1.1, the probability of viability was 80%; if the growth rate was between 0.99 and 1.0, the probability was 70%; if the growth rate was between 0.95 and 0.99, the probability was 50%; if the growth rate was between 0.90 and 0.95, the probability was 30%; if the growth rate was between 0.85 and 0.9, the probability was 20%; otherwise, the probability was 10%. We selected values of smolts per spawner and adults per smolt to approximate the specified growth rates estimated from the PVAs, so that freshwater and marine influences on productivity could be accounted for separately in the network. Probabilities for the influence of frequency of iteroparity on viability through productivity were determined by consensus among PSS TRT members. All these probabilities were then entered into the CPTs in Table 11 to determine the overall contribution of productivity to DIP viability.

For the diversity criterion, we relied primarily on two considerations: 1) the potential influence of hatchery-produced steelhead, most of which are either highly domesticated (Chambers Creek winter run) or out-of-basin source stocks (Skamania River summer run), on wild fish; and 2) evidence for an alteration in natural spawn timing from historical patterns. Iteroparity was under consideration as a factor influencing diversity because of its importance as a temporal risk-spreading strategy, but the TRT decided to consider iteroparity under productivity because of its strong influence on population growth rate. To assess hatchery influence, for each DIP we looked at the number of years of hatchery plants and the cumulative number of hatchery fish planted, based on historical hatchery records. We parameterized the influence of this diversity element on probabilities of viability in the BNs for each DIP in this way: if the number of years of hatchery plants was fewer than 5, the probability of extensive hatchery influence was 10%; if there were no hatchery plants, the probability was 5%. However, if the number of years of hatchery plants was 5 or more, we calculated the average annual number of hatchery fish planted by dividing the cumulative number of hatchery fish planted by the number of years planted. This number was then divided by the estimated IP area and multiplied by 1,000. If this metric was greater than 60, the probability of extensive hatchery influence from this diversity criterion was 90%; if it was between 40 and 60, the probability was 80%; if it was between 30 and 40, the probability was 60%; if it was between 20 and 30, the probability was 40%; if it was between 10 and 20, the probability was 20%; otherwise, the probability was 10%.

To assess whether spawn timing was altered, we examined current spawn timing distributions relative to available documentation for historical distributions from unpublished reports and the published literature (in addition to the References section, see Appendix D). We parameterized the influence of this diversity element on probabilities of viability in the BNs for each DIP in this way: if there was evidence that the current distribution had changed (e.g., shifted earlier or become compressed) relative to the historical distribution for a particular DIP, the probability of viability was 60%; if there was no evidence for change, the probability was 95%. Probabilities for other diversity factors affecting viability, fraction of smolts from resident fish and age structure, were determined by consensus among TRT members. There are no available data on the first factor for Puget Sound steelhead, but the TRT considered it important to include because resident fish that are sympatric with anadromous fish are known from studies of several steelhead populations to be capable of producing smolts, which could provide a demographic buffer, especially for small populations. There are some age data available for selected Puget Sound steelhead populations. All these probabilities, including those for effects of hatchery fish production, were then combined with those in Table 11 to determine the overall contribution of diversity to DIP viability.

Finally, for the spatial structure criterion, we first examined how steelhead DIPs tended to be related according to habitat characteristics, estimating a Gower similarity coefficient (Figure 40) that incorporated maximum elevation, current spawnable area, mean bankfull width, mean stream gradient, maximum mean temperature, and presence of permanent snowpack in the

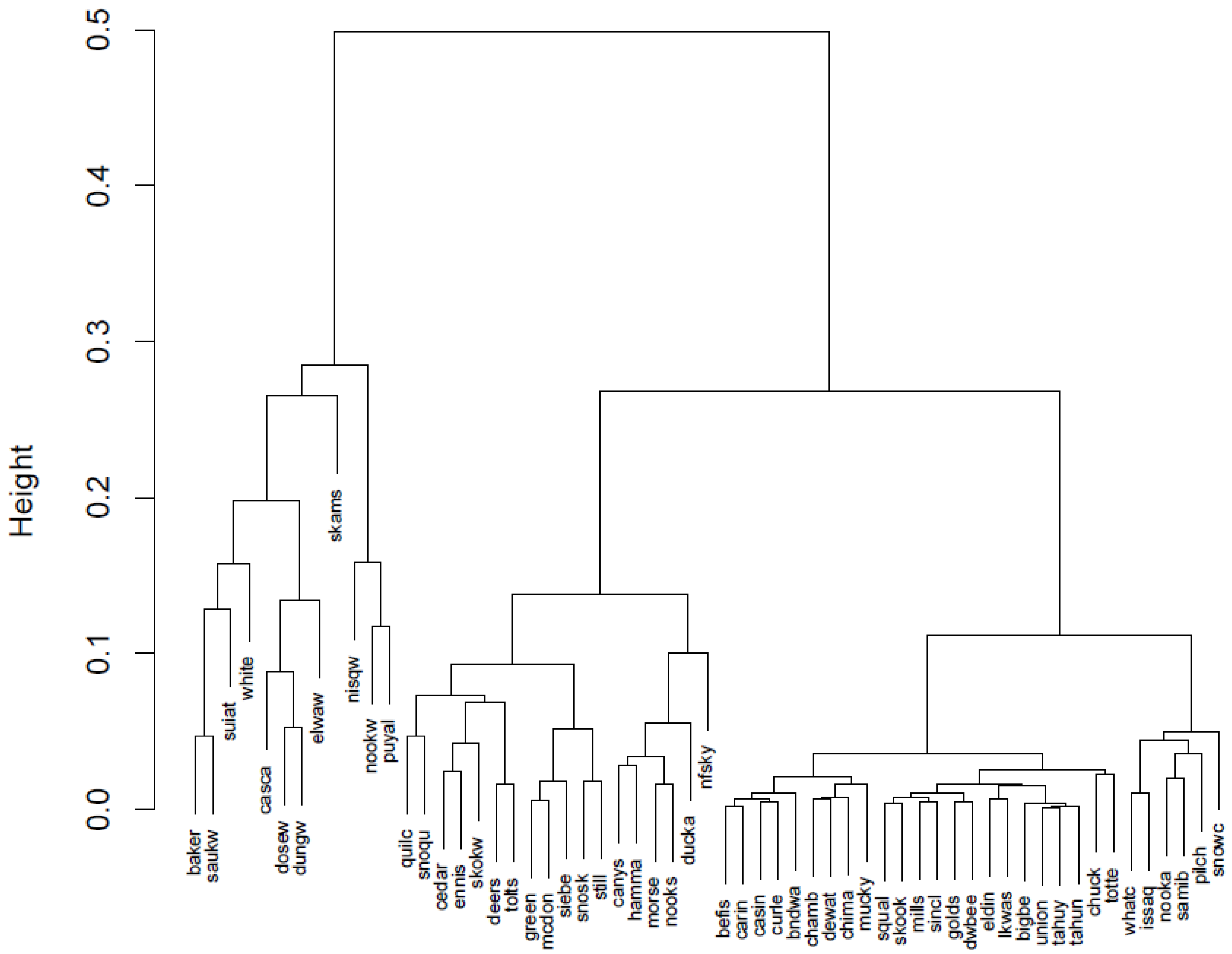


Figure 40. Dendrogram of Gower distances. Relationships among steelhead DIPs with the Puget Sound DPS based on a Gower (1971) similarity index. Three primary clusters are evident in the dendrogram: one for high elevation areas of moderate to high gradient and influenced by snowpack (left), one for low gradient areas with little or no snowpack influence (right), and one for streams with intermediate characteristics (middle).

watersheds harboring the DIPs. The Gower index clustered the DIPs with an agglomerative coefficient of 0.93. We determined probabilities of viability in the BNs for each DIP by first determining occupancy of adult steelhead in reaches within distinct habitat classes incorporating steelhead IP area as described in Appendix C. We further categorized stream reaches by whether they were main stem or tributary, and whether the hydrograph was influenced by snowpack or not (intersection of the reach with a 0°C contour in January). We then performed a series of regressions and multivariate analyses (using principal component analysis) to examine the relationships between observed steelhead productivity and habitat occupancy. These analyses revealed weak but significant relationships between drainage-wide productivity, as measured by population growth rate (λ) or exponential trend in abundance, and steelhead occupancy of smaller mainstem and larger tributary reaches influenced by snowpack. Linear regressions of λ or exponential trend in abundance on occupancy of particular habitat classes incorporating steelhead IP area revealed that these productivity metrics increased most strongly with occupancy of tributary reaches influenced by snowpack; the relationships were relatively weak (r^2 = 0.1–0.2), but this reflected primarily a rise in productivity when a small fraction of the total IP represented by that habitat class was occupied (10–15% or more).

For example, a principal component analysis (with varimax rotation of principal components) of spawner occupancy (fraction of total IP area occupied by spawning steelhead) and population growth rate across various habitat categories (main stem vs. tributary, three stream width classes, three stream gradient classes, and snowpack vs. no snowpack influence) produced eight eigenvectors with eigenvalues greater than 1.0, explaining 85.8% of the variation. Although the relationships were complex, many of the higher loadings on the first 2–3 principal components were associated with snowpack (more positive values) vs. no snowpack (more negative values), with higher productivity also associated with occupancy of stream reaches of particular widths and gradients.

A principal component analysis of a subset of the data focusing on smaller mainstem and larger tributary reaches (3–20 m) of moderate to high gradient (>0.25%) revealed two principal components that explained 73.5% of the variation and showed that population growth rate was most closely associated with steelhead occupancy of reaches influenced by snowpack. A principal component analysis of tributary reaches alone revealed two components that explained 77.9% of the variation and showed that population growth rate was highly associated with steelhead occupancy of reaches with snowpack. Finally, a principal component analysis of tributary reaches influenced by snowpack revealed two components that explained 84.6% of the variation and showed that population growth rate was highly associated with steelhead occupancy of larger reaches with moderate to high gradient.

We used this information to parameterize the influence of the spatial structure element on probabilities of viability in the BNs for each DIP in a simple way: if the proportion of occupied IP habitat area in mainstem reaches less than 20 m bankfull width and influenced by snowpack and in tributary reaches greater than 3 m bankfull width and influenced by snowpack was less than 20%, the probability of viability was 40%; if the proportion was 20% or more, the probability of viability was 80%. We used 20% of historical area as our baseline, because the regressions of population growth rate on occupancy rate indicated a positive change in productivity of steelhead if occupancy of these habitat areas exceeded this rate. We applied these probabilities to both spawning and rearing occupancies, because we had almost no information on rearing occupancy. These probabilities were then combined with those in Table 11 to determine the overall contribution of spatial structure to DIP viability.

We determined the overall viability of each DIP from each DIP-level BN using the conditional probabilities shown in Table 12. Supplementary data describing each DIP are summarized in Appendix E. For many DIPs, some specific viability assessments are based on little or no quantitative information. Although some critics might argue that the criteria these assessments are assigned to can be considered “placeholders” and should be left out of a Bayesian network framework until better information becomes available, we include them for three reasons:

- Eliminating such factors from the network, when they are thought to be important to an overall assessment, defeats the purpose of creating an informative network.

- Bayesian networks are designed to accommodate both factors that can be quantified and those that are evaluated based on collective judgement in the absence of more definitive information, which can be updated when more information becomes available.
- Eliminating such factors tends to place higher importance on factors with better information, regardless of whether those factors are more critical to viability or not.

DIP Viability

The conditional probabilities underlying the BN for the hypothetical winter-run steelhead population in Puget Sound (Figure 37, Table 11, and Table 12) lead to the conclusion that the viability of this DIP is relatively low (45.4%); the probability that the DIP is viable should be on the order of 90% or higher to be confident of viability. If additional information led us to conclude that the population is in fact larger than the low-abundance threshold (adult abundance, P = 100%) and its growth is positive (i.e., $\lambda > 1$, P = 100%), the BN estimates that the probability that the population is viable rises to 60.4%. On the other hand, if additional information led us to conclude that the population is in fact smaller than the low-abundance threshold (P = 100%) and its growth is negative ($\lambda < 1$, P = 100%), the BN estimates that the probability that the population is viable drops to 41.9%. Certainty that hatchery influence is nominal and that resident fish produce an appreciable fraction of smolts, with no instantiation at any other node in the network, prompts the estimate of probability that the population is viable to increase to 48.9%. Certainty that more than half the high IP spawning habitat is at capacity for adults, with no instantiation at any other node, prompts the probability that the population is viable to rise to 51.2%. Meeting most of the criteria for the positive contribution of VSP parameters in the network to DIP viability goes a long way to increasing probability that the DIP is viable; for example, if all of the lowest-level parent nodes in the network except for iteroparity and resident smolt production are instantiated at the highest levels, the probability that the DIP is viable rises to 84.2%. If all such criteria are satisfied, the probability that the DIP is viable is 92.5%. The reason it is no higher is because of uncertainty in the overall contribution of risk for each VSP parameter, reflected by the distribution of probabilities there. This uncertainty is not zero because not all factors that can influence risk for each VSP parameter are considered in the network.

Similarly, the conditional probabilities underlying the BN for the hypothetical summer-run steelhead population in Puget Sound (Figure 38, Table 11, and Table 12) lead to the conclusion that the viability of this DIP is low (40.7%). Instantiating the network at the adult low-abundance threshold (P = 100%) and positive population growth ($\lambda > 1$, P = 100%) causes the probability that the population is viable to rise to 58.8%. Instantiating at the adult low-abundance threshold (P = 100%) and negative population growth ($\lambda < 1$, P = 100%) causes the probability the population is viable to drop to 38.3%. As for the winter-run steelhead BN, meeting most of the criteria for the positive contribution of VSP parameters in the network to DIP viability goes a long way to increasing probability that the DIP is viable; if all of the lowest-level parent nodes in the network except for iteroparity and resident smolt production are instantiated at the highest levels, the probability that the DIP is viable rises to 85.1%. (cf. 84.2% for the winter-run case). If all such criteria are satisfied, the probability that the DIP is viable is again 92.5%. The results of the sensitivity analyses for this BN are essentially identical to those for the winter-run case (Table 13).

Table 13. Results of a sensitivity analysis of the influence of three components of DIP abundance to the VSP risk posed by DIP abundance in the BN in Figure 38.

	Probability ranges			Root mean square change
	Minimum	Current	Maximum	
Sensitivity to findings at juvenile abundance[a]				
Low risk	0.1565	0.262	0.5785	0.182700
Moderate risk	0.2349	0.247	0.2510	0.006972
High risk	0.1866	0.491	0.5925	0.175800
Mean of real value	27.95	37.20	40.29	5.343
Sensitivity to findings at adult abundance[b]				
Low risk	0.1806	0.262	0.5875	0.16280
Moderate risk	0.1949	0.247	0.2600	0.02605
High risk	0.2176	0.491	0.5594	0.13670
Mean of real value	28.48	37.20	39.38	4.361
Sensitivity to findings at *P*(QET)[c]				
Low risk	0.2550	0.262	0.3950	0.03051
Moderate risk	0.2400	0.247	0.3795	0.03040
High risk	0.2255	0.491	0.5050	0.06092
Mean of real value	30.56	37.20	37.55	1.523

[a] Variance reduction = 28.54 (16.7%), entropy reduction = 0.1335 (8.85%), and belief variance = 0.0247 (6.03%).
[b] Variance reduction = 19.02 (11.1%), entropy reduction = 0.0945 (6.26%), and belief variance = 0.01889 (4.61%).
[c] Variance reduction = 2.321 (1.36%), entropy reduction = 0.01133 (0.751%), and belief variance = 0.001594 (0.389%).

The MPE for a BN is an estimate of the most likely network, given the current evidence (i.e., the distributions of conditional probabilities). In other words, it is the most plausible explanation of the data given the evidence. The MPEs for these BNs indicate that neither of these DIPs is viable. The MPEs also indicate that for each network, the risk of DIP abundance to viability is high, the risk of DIP productivity to viability is low, the risk of DIP diversity to viability is high, and the risk of DIP spatial structure to viability is high. These MPE estimates are consistent with the probability distributions for these nodes depicted in Figure 37 and Figure 38. Under MPE, the uncertainty that each DIP is not viable is relatively high; the probability that each DIP is viable is 49.9%.

Figure 41 shows how the four different VSP criteria combine to affect the probability that a representative steelhead DIP in Puget Sound is viable. In this example, abundance and productivity have the greatest influences on viability, approximately twice those of diversity and spatial structure. The contributions of the four VSP criteria to the probability of viability for all 32 candidate DIPs, as estimated from the BNs, are summarized in Figure 42.

The BNs for viability for each of the 32 candidate DIPs the TRT identified for the Puget Sound DPS indicate that population viabilities vary appreciably both within and among MPGs, but these viabilities are all low (each of the 32 BNs is in Appendix F). Over the entire DPS, the probabilities that a DIP was viable are estimated from the BNs to range from 33.4 to 60.2%, with

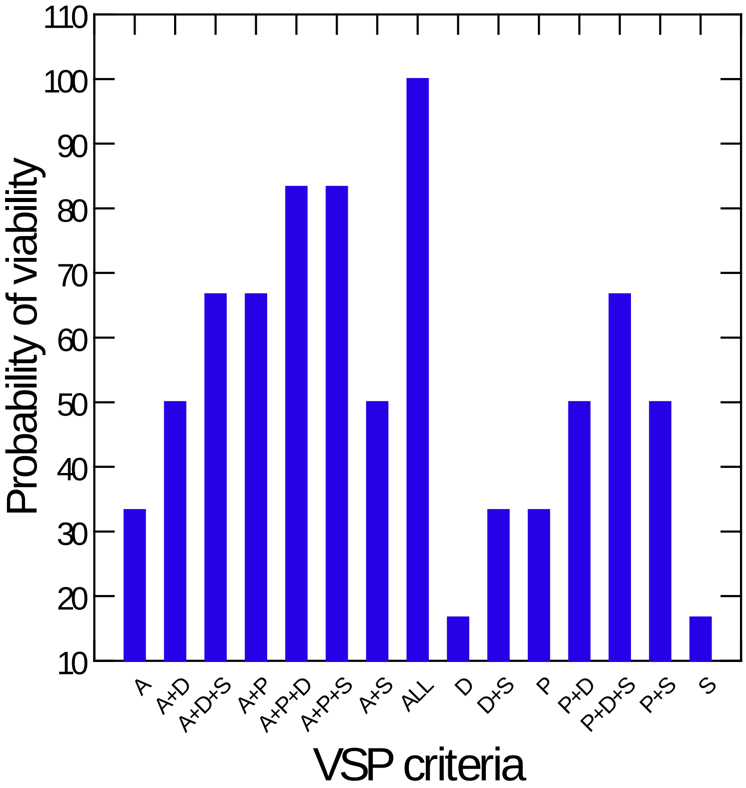


Figure 41. Bar plot showing how the probabilities of viability for a representative DIP of Puget Sound steelhead vary with the contribution to risk from different combinations of VSP criteria. Probabilities were derived from the BN model of steelhead population viability. In this example, abundance and productivity criteria are weighted twice those for diversity and spatial structure. A = abundance, P = productivity, D = diversity, and S = spatial structure.

an average of 44.1%. Average DIP viability is 47.9% in the Northern Cascades MPG, 39.9% in the Central and South Puget Sound MPG, and 40.6% in the Hood Canal and Strait of Juan de Fuca MPG. Average DIP viability for winter-run populations (or those that include both winter-run and summer-run fish) across the DPS is 42.7%; average viability for summer-run populations across the DPS is 52.7%. In most cases, these viabilities reflect the substantial influence of all four VSP criteria, but especially of abundance and diversity, as components of the BN models.

To illustrate how viability of the 32 candidate steelhead DIPs varies with respect to the influence of each of the four VSP criteria, we plotted estimated DIP viability as a function of the influence of pairs of these criteria. Figure 41 depicts the influence of the values of abundance and productivity on the estimate of viability. Higher values of the VSP components indicate higher influence on viability. Viabilities are computed from the DIP-level BNs in Appendix F.

The plots reveal interesting geographical patterns in population viability and how it varies with the VSP criteria. With respect to the combination of VSP abundance and productivity, viability is generally highest for DIPs in the Northern Cascades MPG (Figure 43). The BN models indicate that viability with respect to abundance and productivity is highest for a few Northern Cascades DIPs (e.g., Canyon Creek Summer Run and North Fork Skykomish River

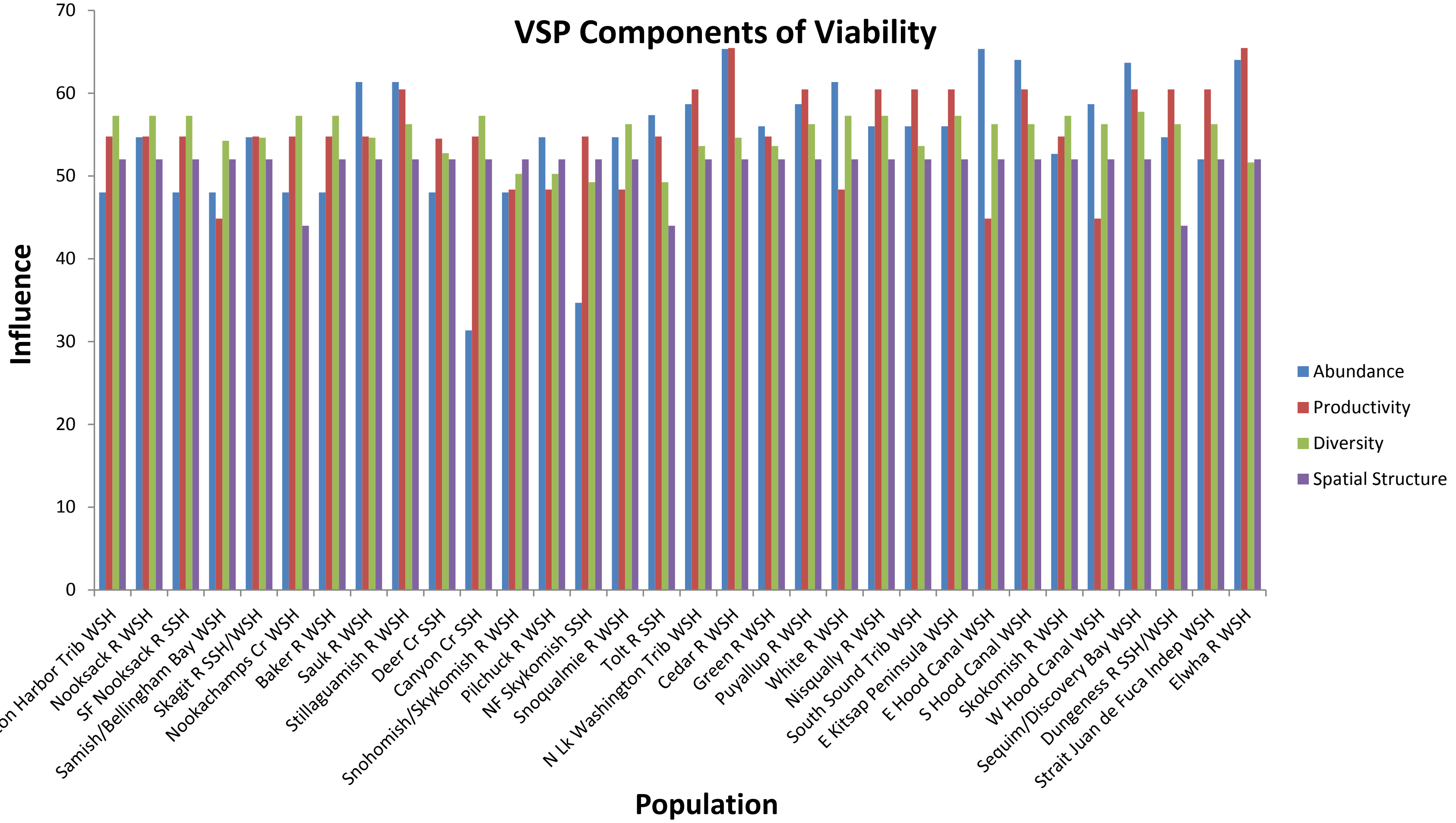


Figure 42. Bar plot of the relative influences of the four VSP parameter estimates (abundance, productivity, diversity, and spatial structure) applied to the viabilities of each of the 32 candidate DIPs of steelhead in the Puget Sound DPS. These values are estimated from the respective BNs. SSH = summer-run steelhead and WSH = winter-run steelhead.

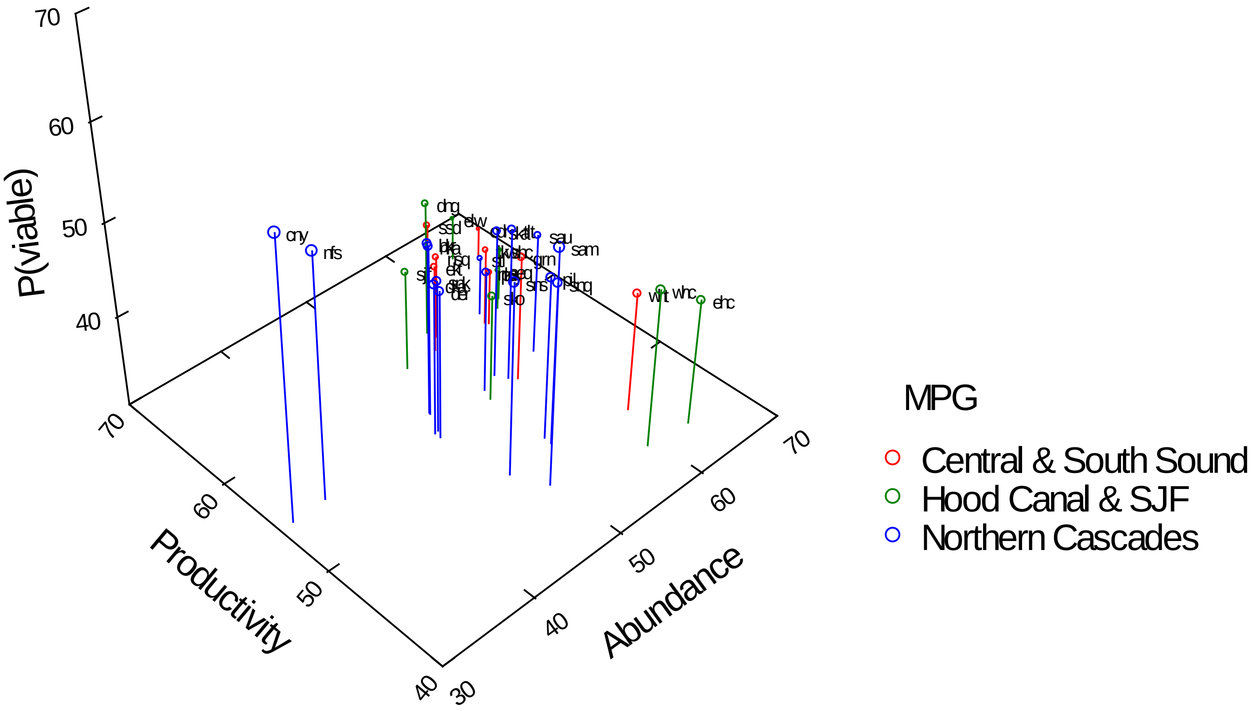


Figure 43. Scatter plot of the viabilities of each of the 32 candidate DIPs of steelhead in the Puget Sound DPS as a function of their VSP parameter estimates of influence of abundance and productivity. Viabilities are computed from the DIP-level BNs in Appendix F. Abundance and productivity scores are estimated from intermediate metrics computed by the DIP-level BNs. Three-letter DIP codes (SSH = summer-run steelhead and WSH = winter-run steelhead) are: dra = Drayton Harbor Tributaries WSH, nks = Nooksack River WSH, sns = South Fork Nooksack River SSH, sam = Samish River and Bellingham Bay Tributaries WSH, ska = Skagit River SSH and WSH, nka = Nookachamps Creek WSH, bkr = Baker River SSH and WSH, sau = Sauk River SSH and WSH, stl = Stillaguamish River WSH, der = Deer Creek SSH, cny = Canyon Creek SSH, snk = Snohomish/Skykomish Rivers WSH, pil = Pilchuck River WSH, nfs = North Fork Skykomish River SSH, snq = Snoqualmie River WSH, tlt = Tolt River SSH, cdr = Cedar River WSH, lkw = North Lake Washington and Lake Sammamish WSH, grn = Green River WSH, puy = Puyallup/ Carbon Rivers WSH, wht = White River WSH, nsq = Nisqually River WSH, ssd = South Puget Sound Tributaries WSH, ekt = East Kitsap Peninsula Tributaries WSH, ehc = East Hood Canal Tributaries WSH, shc = South Hood Canal Tributaries WSH, sko = Skokomish River WSH, whc = West Hood Canal Tributaries WSH, seq = Sequim/Discovery Bays Tributaries WSH, dng = Dungeness River SSH and WSH, sjf = Strait of Juan de Fuca Tributaries WSH, and elw = Elwha River WSH.

Summer Run, probabilities of viability = 55–60%). The viabilities of these DIPs are influenced more by productivity than by abundance. Most other populations in this MPG with lower viabilities are influenced more by low abundance. Viabilities are lower in the Central and South Puget Sound MPG and the Hood Canal and the Strait of Juan de Fuca MPG, in general, and were lowest in the Central and South Puget Sound MPG, where most populations are at low abundance (e.g., Cedar River Winter Run and Puyallup/Carbon Rivers Winter Run, probabilities of viability 30–40%). In the latter MPG, only the White River Winter-Run DIP shows modest viability. The decline in DIP viability with declining abundance and productivity is clearly nonlinear, and viability seems to be somewhat more sensitive to productivity than to abundance. Most DIPs in the DPS exhibit low probabilities of viability with respect to abundance and productivity (40–50%).

With respect to the combination of VSP abundance and diversity, there is less variation evident for the influence of diversity on DIP viability (Figure 44). For DIPs in both the Central and South Puget Sound MPG and the Hood Canal and Strait of Juan de Fuca MPG, where viabilities are lower, populations are influenced by both low abundance and low diversity. DIPs in the Northern Cascades MPG tend to show a wider range of viabilities and, on average, viabilities are higher with respect to these criteria. The change in DIP viability with abundance and diversity is highly variable. As for abundance and productivity, most DIPs in the DPS exhibit low probabilities of viability with respect to abundance and diversity (40–50%).

Regarding the combination of VSP abundance and spatial structure, the two summer-run DIPs with highest viabilities (Canyon Creek and North Fork Skykomish River) show a strong influence of spatial structure (Figure 45). Only the Nookachamps Creek Winter Run and the Tolt River Summer Run in the Northern Cascades MPG and the Dungeness River Summer Run and Winter Run in the Hood Canal and Strait of Juan de Fuca MPG show a relatively modest influence of spatial structure on viability. Most DIPs throughout the DPS show an appreciable influence of both abundance and spatial structure on viability.

The influences of VSP productivity and diversity on DIP viability are more variable than those for the previous combinations of VSP parameters (Figure 46). The DIPs with the lowest viabilities (e.g., Sequim/Discovery Bays Tributaries Winter Run, Elwha River Winter Run, Cedar River Winter Run) indicate the very high influence of low productivity. Those with the highest viabilities (e.g., Canyon Creek Summer Run and North Fork Skykomish River Summer Run) show a more consistent influence of productivity than of diversity. Viability appears to be most sensitive to low productivity.

When considered with productivity, spatial structure has a modest influence on population viability; for DIPs with a high influence of spatial structure on viability, the DIPs with lowest viabilities were influenced heavily by low productivity (Figure 47). In general, DIP viability declines steeply when productivity is low. For the few DIPs that have a modest influence of spatial structure on viability (Nookachamps Creek Winter Run, Tolt River Summer Run, and Dungeness River Summer Run and Winter Run), viability tends to be moderate. The pattern of viability with respect to the combination of VSP diversity and spatial structure is similar to the pattern with respect to productivity and spatial structure (Figure 48).

According to the BN models, a typical DIP's viability is lowest whenever abundance and productivity are limiting (probabilities of viability < about 30%). As a representative example, for the Samish River and Bellingham Bay Tributaries Winter-Run DIP in the Northern Cascades MPG, the sensitivity of viability to DIP abundance (entropy reduction 6.7%), DIP productivity (entropy reduction 5.2%), DIP diversity (entropy reduction 1.3%, primarily a result of hatchery fish influence), and DIP spatial structure (entropy reduction 1.6%) indicated that Samish River steelhead viability is limited primarily by both abundance and productivity. For the abundance criterion, the most important contributing factor was adult abundance (entropy reduction 2.1%), followed by juvenile abundance (entropy reduction 0.1%) and by the probability of reaching the

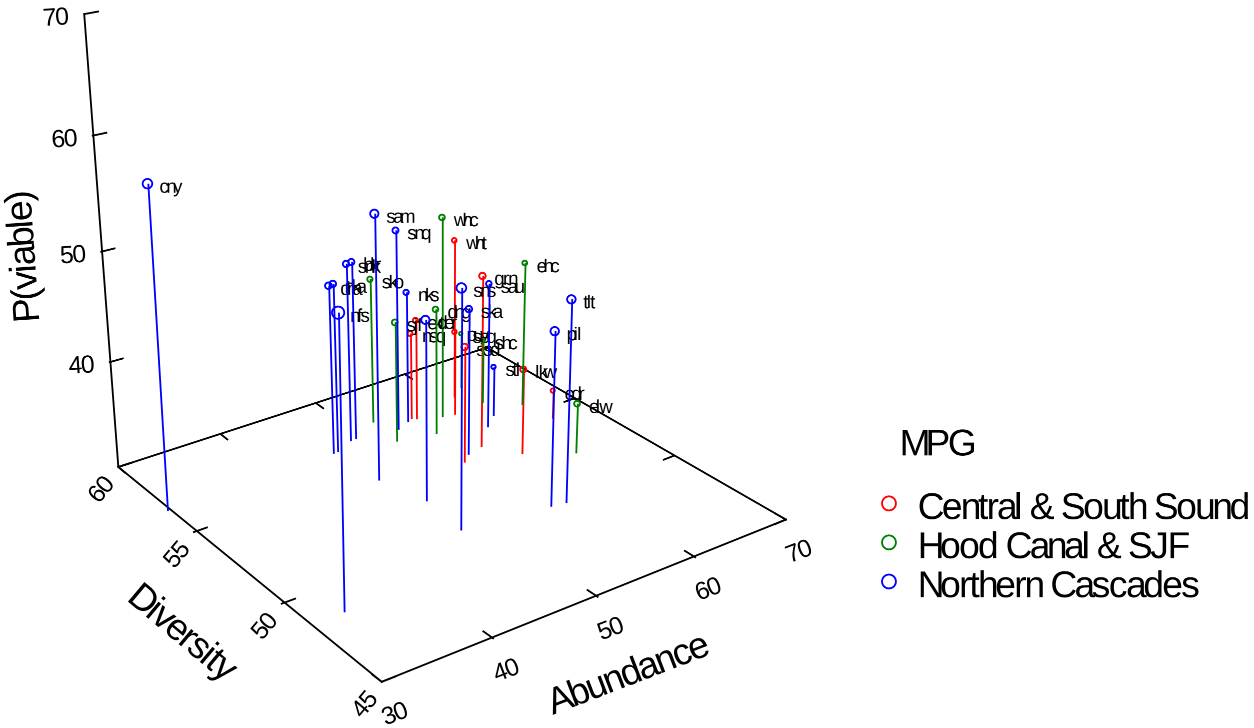


Figure 44. Scatter plot of the viabilities of each of the 32 candidate DIPs of steelhead in the Puget Sound DPS as a function of their VSP parameter estimates of influence of abundance and diversity. Viabilities are computed from the DIP-level BNs in Appendix F. Abundance and diversity scores are estimated from intermediate metrics computed by the DIP-level BNs. Three-letter DIP codes are as in Figure 43.

specified QET (entropy reduction <0.1%). For the productivity criterion, the most important contributing factor was population growth rate (entropy reduction 2.0%), followed by iteroparity (entropy reduction 0.3%). Other factors contributing to DIP viability included marine survival rate (entropy reduction 1.2%), spatial structure criteria (total entropy reduction for spawning and rearing area occupied, 0.5%), and altered spawn timing (entropy reduction 0.3%). Hatchery influence and alteration of age structure had minor influences on viability. The MPE for the viability of this DIP is that it is not viable, but there is considerable uncertainty around this MPE: the probability that this DIP could be viable is estimated at 69.2%.

By comparison, for the Nisqually River Winter-Run DIP in the Central and South Puget Sound MPG, the sensitivity of viability to DIP abundance (entropy reduction 6.0%), DIP productivity (entropy reduction 6.9%), DIP diversity (entropy reduction 1.3%), and DIP spatial structure (entropy reduction 1.7%) indicated that Nisqually River steelhead viability is limited more by productivity. For the abundance criterion, the most important contributing factor was adult abundance (entropy reduction 1.2%), followed by the probability of reaching the specified QET (entropy reduction 0.2%) and by juvenile abundance (entropy reduction <0.1%). For the productivity criterion, the most important contributing factor was population growth rate (entropy reduction 2.8%), followed by iteroparity (entropy reduction 1.4%). Other factors contributing to viability included marine survival rate (entropy reduction 1.5%), spawn timing (entropy reduction 0.3%), and spatial structure criteria (total entropy reduction for spawning and

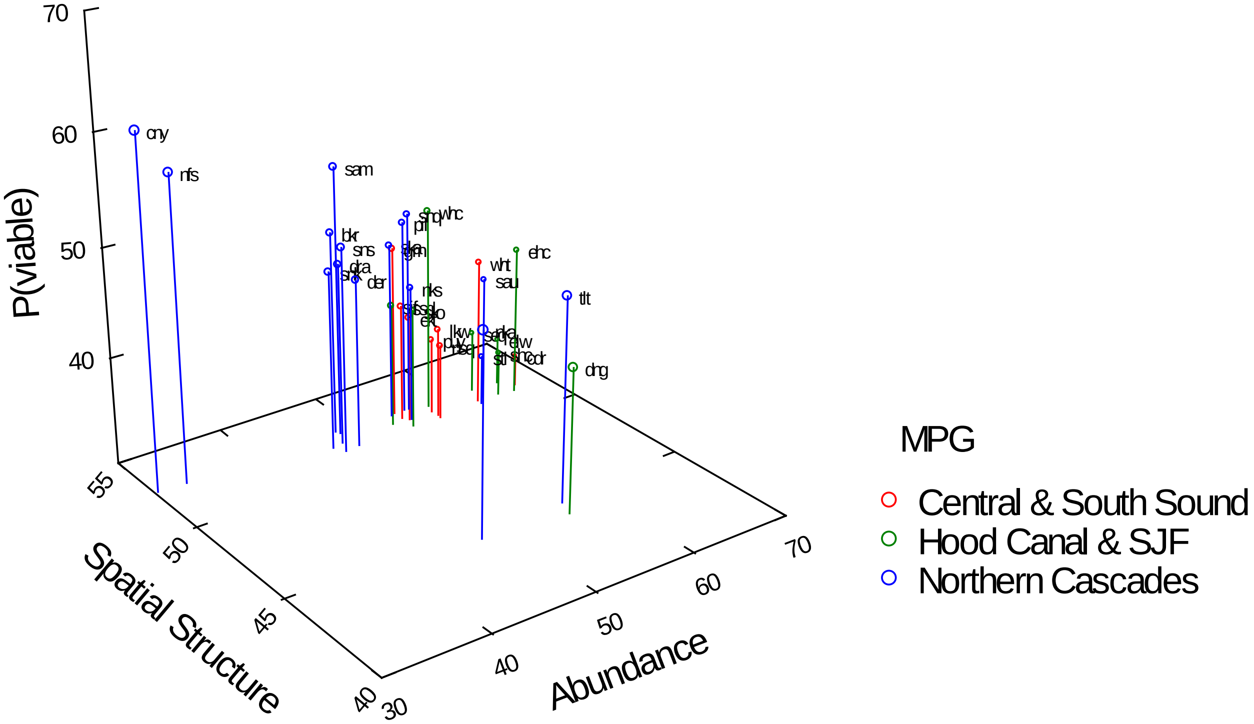


Figure 45. Scatter plot of the viabilities of each of the 32 candidate DIPs of steelhead in the Puget Sound DPS as a function of their VSP parameter estimates of influence of abundance and spatial structure. Viabilities are computed from the DIP-level BNs in Appendix F. Abundance and spatial structure scores are estimated from intermediate metrics computed by the DIP-level BNs. Three-letter DIP codes are as in Figure 43.

rearing area occupied, 0.5%). Hatchery influence and alteration of age structure had minor influences on viability. The MPE for the viability of this DIP is that it is not viable, and there is little uncertainty around the MPE, because the probability that it could be viable is estimated at only 11.9%.

As an example for a summer-run population, for the Tolt River Summer-Run DIP in the Northern Cascades MPG, the sensitivity of viability to DIP abundance (entropy reduction 6.2%), DIP productivity (entropy reduction 6.8%), DIP diversity (entropy reduction 1.3%), and DIP spatial structure (entropy reduction 1.1%) indicated that Tolt River steelhead viability is limited by both abundance and productivity. For the abundance criterion, the most important contributing factor was adult abundance (entropy reduction 2.1%), followed by juvenile abundance (entropy reduction 0.3%) and the probability of reaching the specified QET (entropy reduction 0.1%). For the productivity criterion, the most important contributing factor was population growth rate (entropy reduction 3.2%), followed by iteroparity (entropy reduction 0.9%). Other factors contributing to viability included marine survival rate (entropy reduction 1.8%) and spawn timing (entropy reduction 0.3%); hatchery influence, age structure, and adult and juvenile occupancy of IP areas had minor influences on viability. The MPE for the viability of this DIP is that it is not viable, but this MPE has a high level of uncertainty: the probability that it could be viable is estimated to be as high as 62.8%.

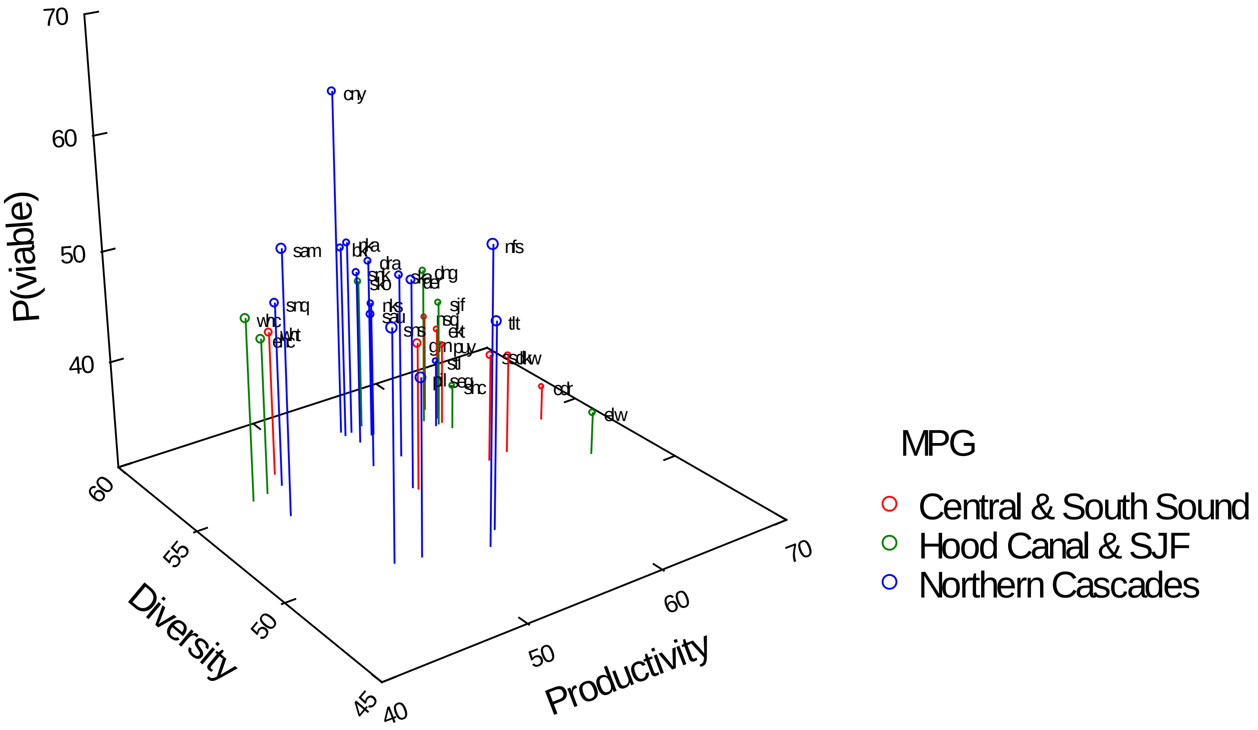


Figure 46. Scatter plot of the viabilities of each of the 32 candidate DIPs of steelhead in the Puget Sound DPS as a function of their VSP parameter estimates of influence of productivity and diversity. Viabilities are computed from the DIP-level BNs in Appendix F. Productivity and diversity scores are estimated from intermediate metrics computed by the DIP-level BNs. Three-letter DIP codes are as in Figure 43.

The BNs for the remaining 29 steelhead DIPs yield similar conclusions for DIP viability, as well as the range of influence of VSP parameters on DIP viability. Throughout the DPS, the probability that a DIP is viable does not exceed about 60%. For the DIP with the highest estimated viability, the Canyon Creek Summer Run in the Northern Cascades MPG, while the MPE is that this DIP is viable, there is considerable uncertainty around the MPE: the probability that this DIP could be nonviable is 85%.

MPG Viability

We evaluated the viability of each of the three Puget Sound steelhead MPGs by combining the DIP-level BNs for each MPG into an encompassing network with an edge connecting each node determining DIP viability to a single node for MPG viability. Figure 49 depicts such a network for one of the smaller MPGs—the Hood Canal and Strait of Juan de Fuca MPG. The CPT describing the influence of DIP viability on MPG viability for this BN is in Table 14 (similar CPTs, not shown, underlie the BNs for the Northern Cascades MPG and the Central and South Puget Sound MPG). In this table of probabilities and the corresponding values for other MPGs, the MPG is considered viable only if 40% or more of the constituent DIPs are considered viable as an outcome of the DIP viability BN analyses and, if summer-run DIPs are present, at least 40% of these populations are also considered viable.

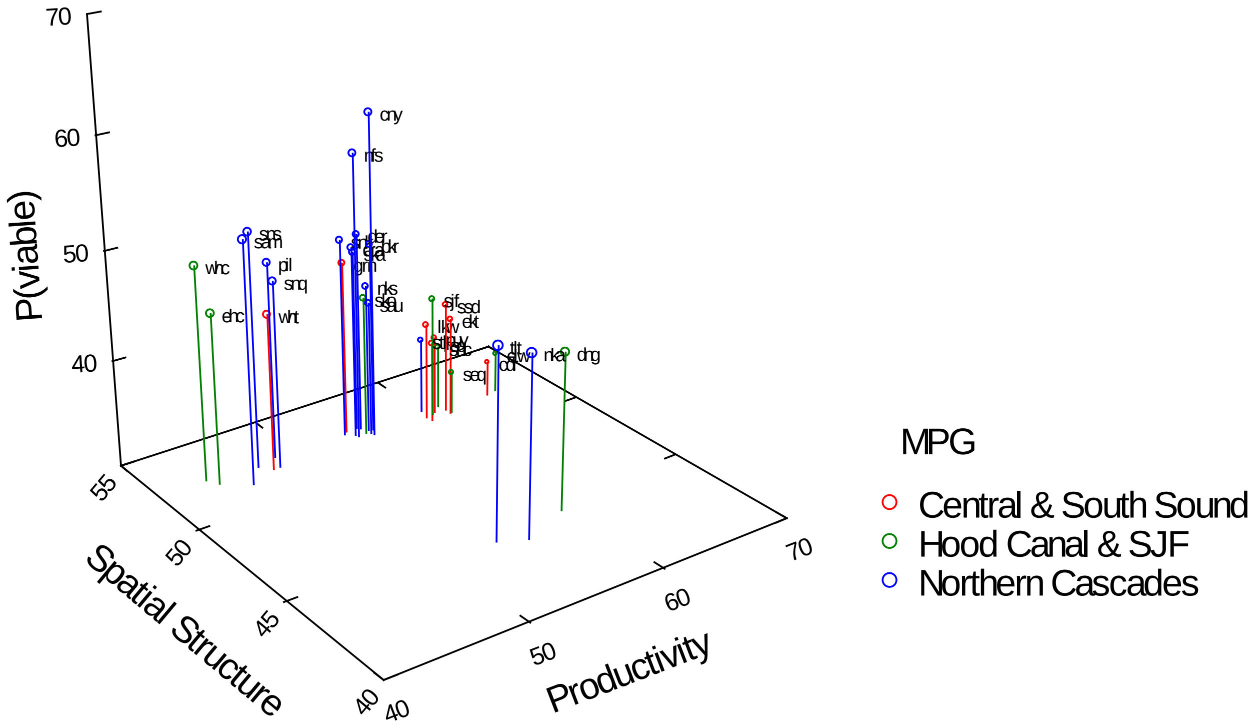


Figure 47. Scatter plot of the viabilities of each of the 32 candidate DIPs of steelhead in the Puget Sound DPS as a function of their VSP parameter estimates of influence of productivity and spatial structure. Viabilities are computed from the DIP-level BNs in Appendix F. Productivity and spatial structure scores are estimated from intermediate metrics computed by the DIP-level BNs. Three-letter DIP codes are as in Figure 43.

Under these conditions, a BN analysis computes that the probability of viability for the Hood Canal and Strait of Juan de Fuca MPG as a whole is only 16.4% (Figure 49). Average DIP probability of viability is 40.6% in this MPG. The BN for the Hood Canal and Strait of Juan de Fuca MPG indicates that the probability that this MPG is viable would rise to 100% if at least one DIP from each of the four major regions of the MPG (i.e., Strait of Juan de Fuca [4 DIPs], East Hood Canal [1 DIP], South Hood Canal [2 DIPs], and West Hood Canal [1 DIP]) were viable with certainty (i.e., by instantiating certainty of viability for these DIPs). MPG viability would rise to 89.5% if all four DIPs on the Strait of Juan de Fuca were instantiated at fully viable. A sensitivity analysis (output not shown) indicates that the mutual information provided by the BN model for viability of this MPG was 0.64, which can be interpreted as meaning that the model reduced the uncertainty surrounding the MPG's viability by about 64%. The entropy reduction contributed by the viability of its constituent DIPs varied from 5.2% (Sequim/ Discovery Bays Tributaries Winter Run) to 10.5% (Dungeness River Summer Run and Winter Run). The MPE for the viability of this MPG is that it is not viable and there is little uncertainty around this MPE: the probability that the MPG could be viable is 1.3%.

A BN analysis for the Central and South Puget Sound MPG shows that the probability of its viability is also low: 17.2%. Average DIP probability of viability in this MPG is 39.9%. The BN for the MPG indicates that the probability that this MPG is viable would rise to at least 87.7% if at least one DIP from each of the four major regions of the MPG (i.e., Central Puget

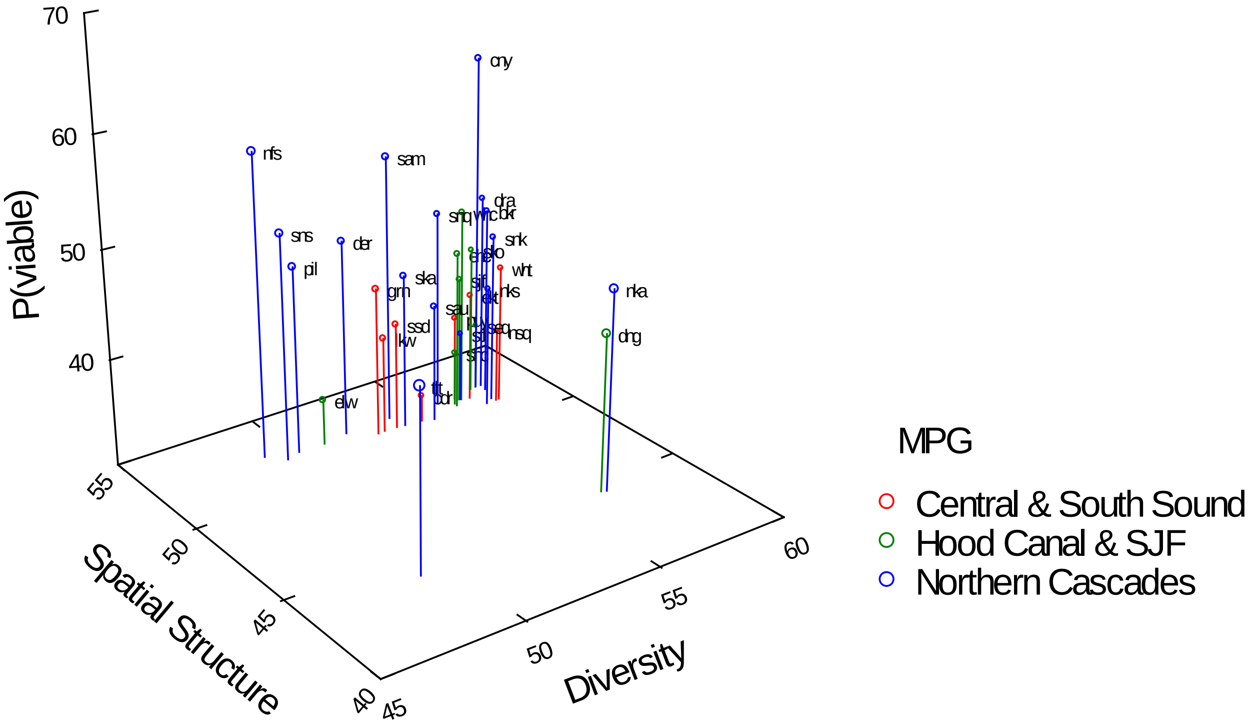


Figure 48. Scatter plot of the viabilities of each of the 32 candidate DIPs of steelhead in the Puget Sound DPS as a function of their VSP parameter estimates of influence of diversity and spatial structure. Viabilities are computed from the DIP-level BNs in Appendix F. Diversity and spatial structure scores are estimated from intermediate metrics computed by the DIP-level BNs. Three-letter DIP codes are as in Figure 43.

Sound [3 DIPs], Southeast Puget Sound [3 DIPs], South Puget Sound Tributaries [1 DIP], and East Kitsap Peninsula Tributaries [1 DIP]) were viable. A sensitivity analysis (output not shown) indicates that the mutual information provided by the BN model for viability of this MPG was 0.66. The entropy reduction contributed by the viability of its constituent DIPs showed little variation, ranging from 6.6% (Cedar River Winter Run) to 6.7% (North Lake Washington and Lake Sammamish Winter Run). The MPE for the viability of this MPG is that it is not viable, and there almost no uncertainty around this MPE: the probability that the MPG could be viable is < 0.1%.

A BN analysis for the Northern Cascades MPG indicates a probability of viability of 36.3%. Average DIP probability of viability in this MPG is the highest of the three MPGs at 47.9%. Although this MPG contains twice as many DIPs as the other two in the DPS (16 versus 8 for both the Hood Canal and Strait of Juan de Fuca MPG and the Central and South Puget Sound MPG), its viability is also low because not enough of its constituent DIPs within each major life history type are at reasonable levels of viability. The BN for this MPG indicates that its viability would rise to at least 88.7% if at least half of the DIPs in it were viable. Given the conditions specified by the underlying CPT, the BN also indicates that the probability that this MPG is viable would rise to 87.9% if at least half (6 of 11) of the winter-run DIPs and at least half (3 of 5) of the summer-run DIPs in the MPG were viable (spread geographically across the MPG, and including the Skagit River Summer Run and Winter Run). It indicates as well that the

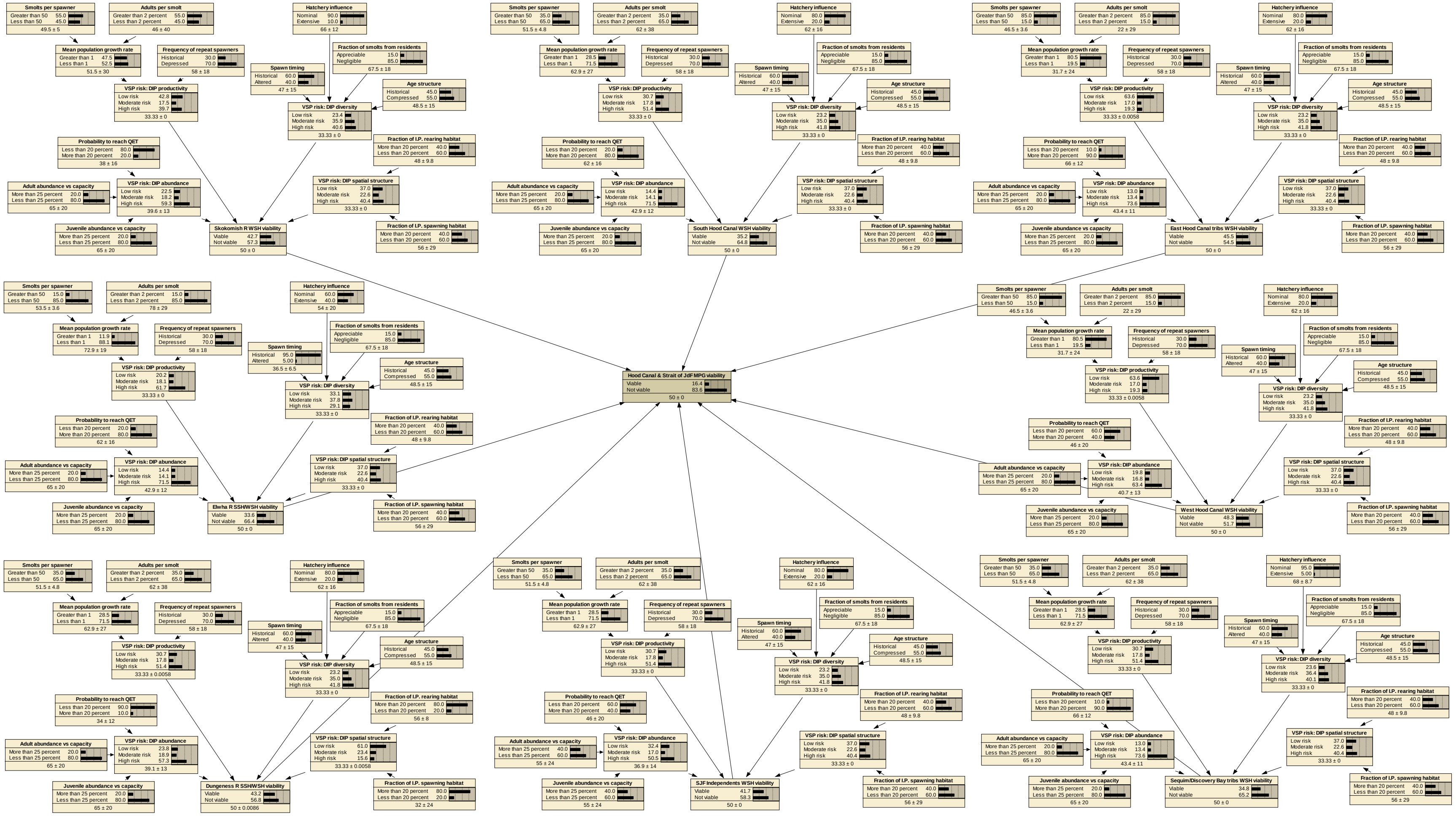


Figure 49. BN used to characterize the viability of the Hood Canal and Strait of Juan de Fuca steelhead MPG. The CPT describing the influence of DIP viability on MPG viability is in Table 14.

Table 14. CPT describing the influence of DIP viability on MPG viability for the BN in Figure 49. Similar CPTs were used to parameterize the Northern Cascades MPG and the Central and South Puget Sound MPG, which contain 16 and 8 steelhead DIPs, respectively. The Northern Cascades MPG contains twice as many DIPs as the other MPGs, but we used an analogous CPT to characterize the influence of DIP viability on MPG viability, which explicitly requires greater than 40% viable DIPs of each distinct life history type (summer run [SSH], winter run [WSH], or combined summer run and winter run) for MPG viability.

DIP viability								**MPG viability**	
E. Hood Canal Tribs. WSH	S. Hood Canal Tribs. WSH	Skokomish River WSH	W. Hood Canal Tribs. WSH	Sequim/Disc. Bays WSH	Dungeness R. SSH & WSH	Strait of J. de F. Tribs. WSH	Elwha River WSH	Viable	Not viable
V	V	V	V	V	V	V	V	100	0
V	V	V	V	V	V	V	NV	100	0
V	V	V	V	V	V	NV	V	100	0
V	V	V	V	V	V	NV	NV	100	0
V	V	V	V	V	NV	V	V	100	0
V	V	V	V	V	NV	V	NV	100	0
V	V	V	V	V	NV	NV	V	100	0
V	V	V	V	V	NV	NV	NV	100	0
V	V	V	V	NV	V	V	V	100	0
V	V	V	V	NV	V	V	NV	100	0
V	V	V	V	NV	V	NV	V	100	0
V	V	V	V	NV	V	NV	NV	100	0
V	V	V	V	NV	NV	V	V	100	0
V	V	V	V	NV	NV	V	NV	100	0
V	V	V	V	NV	NV	NV	V	100	0
V	V	V	V	NV	NV	NV	NV	100	0
V	V	V	NV	V	V	V	V	100	0
V	V	V	NV	V	V	V	NV	100	0
V	V	V	NV	V	V	NV	V	100	0
V	V	V	NV	V	V	NV	NV	100	0
V	V	V	NV	V	NV	V	V	100	0
V	V	V	NV	V	NV	V	NV	100	0
V	V	V	NV	V	NV	NV	V	100	0
V	V	V	NV	V	NV	NV	NV	100	0
V	V	V	NV	NV	V	V	V	100	0
V	V	V	NV	NV	V	V	NV	100	0
V	V	V	NV	NV	V	NV	V	100	0
V	V	V	NV	NV	V	NV	NV	100	0
V	V	V	NV	NV	NV	V	V	100	0
V	V	V	NV	NV	NV	V	NV	100	0
V	V	V	NV	NV	NV	NV	V	100	0
V	V	V	NV	NV	NV	NV	NV	0	100
V	V	NV	V	V	V	V	V	100	0
V	V	NV	V	V	V	V	NV	100	0
V	V	NV	V	V	V	NV	V	100	0

Table 14 continued. CPT describing the influence of DIP viability on MPG viability for the BN in Figure 49. Similar CPTs were used to parameterize the Northern Cascades MPG and the Central and South Puget Sound MPG, which contain 16 and 8 steelhead DIPs, respectively. The Northern Cascades MPG contains twice as many DIPs as the other MPGs, but we used an analogous CPT to characterize the influence of DIP viability on MPG viability, which explicitly requires greater than 40% viable DIPs of each distinct life history type (summer run [SSH], winter run [WSH], or combined summer run and winter run) for MPG viability.

DIP viability								**MPG viability**	
E. Hood Canal Tribs. WSH	S. Hood Canal Tribs. WSH	Skokomish River WSH	W. Hood Canal Tribs. WSH	Sequim/Disc. Bays WSH	Dungeness R. SSH & WSH	Strait of J. de F. Tribs. WSH	Elwha River WSH	Viable	Not viable
V	V	NV	V	V	V	NV	NV	100	0
V	V	NV	V	V	NV	V	V	100	0
V	V	NV	V	V	NV	V	NV	100	0
V	V	NV	V	V	NV	NV	V	100	0
V	V	NV	V	V	NV	NV	NV	100	0
V	V	NV	V	NV	V	V	V	100	0
V	V	NV	V	NV	V	V	NV	100	0
V	V	NV	V	NV	V	NV	V	100	0
V	V	NV	V	NV	V	NV	NV	100	0
V	V	NV	V	NV	NV	V	V	100	0
V	V	NV	V	NV	NV	V	NV	100	0
V	V	NV	V	NV	NV	NV	V	100	0
V	V	NV	V	NV	NV	NV	NV	0	100
V	V	NV	NV	V	V	V	V	100	0
V	V	NV	NV	V	V	V	NV	100	0
V	V	NV	NV	V	V	NV	V	100	0
V	V	NV	NV	V	V	NV	NV	100	0
V	V	NV	NV	V	NV	V	V	100	0
V	V	NV	NV	V	NV	V	NV	100	0
V	V	NV	NV	V	NV	NV	V	100	0
V	V	NV	NV	V	NV	NV	NV	0	100
V	V	NV	NV	NV	V	V	V	100	0
V	V	NV	NV	NV	V	V	NV	100	0
V	V	NV	NV	NV	V	NV	V	100	0
V	V	NV	NV	NV	V	NV	NV	0	100
V	V	NV	NV	NV	NV	V	V	100	0
V	V	NV	NV	NV	NV	V	NV	0	100
V	V	NV	NV	NV	NV	NV	V	0	100
V	V	NV	NV	NV	NV	NV	NV	0	100
V	NV	V	V	V	V	V	V	100	0
V	NV	V	V	V	V	V	NV	100	0
V	NV	V	V	V	V	NV	V	100	0
V	NV	V	V	V	V	NV	NV	100	0
V	NV	V	V	V	NV	V	V	100	0
V	NV	V	V	V	NV	V	NV	100	0

Table 14 continued. CPT describing the influence of DIP viability on MPG viability for the BN in Figure 49. Similar CPTs were used to parameterize the Northern Cascades MPG and the Central and South Puget Sound MPG, which contain 16 and 8 steelhead DIPs, respectively. The Northern Cascades MPG contains twice as many DIPs as the other MPGs, but we used an analogous CPT to characterize the influence of DIP viability on MPG viability, which explicitly requires greater than 40% viable DIPs of each distinct life history type (summer run [SSH], winter run [WSH], or combined summer run and winter run) for MPG viability.

DIP viability								MPG viability	
E. Hood Canal Tribs. WSH	S. Hood Canal Tribs. WSH	Skokomish River WSH	W. Hood Canal Tribs. WSH	Sequim/Disc. Bays WSH	Dungeness R. SSH & WSH	Strait of J. de F. Tribs. WSH	Elwha River WSH	Viable	Not viable
V	NV	V	V	V	NV	NV	V	100	0
V	NV	V	V	V	NV	NV	NV	100	0
V	NV	V	V	NV	V	V	V	100	0
V	NV	V	V	NV	V	V	NV	100	0
V	NV	V	V	NV	V	NV	V	100	0
V	NV	V	V	NV	V	NV	NV	100	0
V	NV	V	V	NV	NV	V	V	100	0
V	NV	V	V	NV	NV	V	NV	100	0
V	NV	V	V	NV	NV	NV	V	100	0
V	NV	V	V	NV	NV	NV	NV	0	100
V	NV	V	NV	V	V	V	V	100	0
V	NV	V	NV	V	V	V	NV	100	0
V	NV	V	NV	V	V	NV	V	100	0
V	NV	V	NV	V	V	NV	NV	100	0
V	NV	V	NV	V	NV	V	V	100	0
V	NV	V	NV	V	NV	V	NV	100	0
V	NV	V	NV	V	NV	NV	V	100	0
V	NV	V	NV	V	NV	NV	NV	0	100
V	NV	V	NV	NV	V	V	V	100	0
V	NV	V	NV	NV	V	V	NV	100	0
V	NV	V	NV	NV	V	NV	V	100	0
V	NV	V	NV	NV	V	NV	NV	0	100
V	NV	V	NV	NV	NV	V	V	100	0
V	NV	V	NV	NV	NV	V	NV	0	100
V	NV	V	NV	NV	NV	NV	V	0	100
V	NV	V	NV	NV	NV	NV	NV	0	100
V	NV	NV	V	V	V	V	V	100	0
V	NV	NV	V	V	V	V	NV	100	0
V	NV	NV	V	V	V	NV	V	100	0
V	NV	NV	V	V	V	NV	NV	100	0
V	NV	NV	V	V	NV	V	V	100	0
V	NV	NV	V	V	NV	V	NV	100	0
V	NV	NV	V	V	NV	NV	V	100	0
V	NV	NV	V	V	NV	NV	NV	0	100
V	NV	NV	V	NV	V	V	V	100	0

Table 14 continued. CPT describing the influence of DIP viability on MPG viability for the BN in Figure 49. Similar CPTs were used to parameterize the Northern Cascades MPG and the Central and South Puget Sound MPG, which contain 16 and 8 steelhead DIPs, respectively. The Northern Cascades MPG contains twice as many DIPs as the other MPGs, but we used an analogous CPT to characterize the influence of DIP viability on MPG viability, which explicitly requires greater than 40% viable DIPs of each distinct life history type (summer run [SSH], winter run [WSH], or combined summer run and winter run) for MPG viability.

DIP viability								**MPG viability**	
E. Hood Canal Tribs. WSH	S. Hood Canal Tribs. WSH	Skokomish River WSH	W. Hood Canal Tribs. WSH	Sequim/Disc. Bays WSH	Dungeness R. SSH & WSH	Strait of J. de F. Tribs. WSH	Elwha River WSH	Viable	Not viable
V	NV	NV	V	NV	V	V	NV	100	0
V	NV	NV	V	NV	V	NV	V	100	0
V	NV	NV	V	NV	V	NV	NV	0	100
V	NV	NV	V	NV	NV	V	V	100	0
V	NV	NV	V	NV	NV	V	NV	0	100
V	NV	NV	V	NV	NV	NV	V	0	100
V	NV	NV	V	NV	NV	NV	NV	0	100
V	NV	NV	NV	V	V	V	V	100	0
V	NV	NV	NV	V	V	V	NV	100	0
V	NV	NV	NV	V	V	NV	V	100	0
V	NV	NV	NV	V	V	NV	NV	0	100
V	NV	NV	NV	V	NV	V	V	100	0
V	NV	NV	NV	V	NV	V	NV	100	0
V	NV	NV	NV	V	NV	NV	V	0	100
V	NV	NV	NV	V	NV	NV	NV	0	100
V	NV	NV	NV	NV	V	V	V	100	0
V	NV	NV	NV	NV	V	V	NV	0	100
V	NV	NV	NV	NV	V	NV	V	0	100
V	NV	NV	NV	NV	V	NV	NV	0	100
V	NV	NV	NV	NV	NV	V	V	0	100
V	NV	NV	NV	NV	NV	V	NV	0	100
V	NV	NV	NV	NV	NV	NV	V	0	100
V	NV	NV	NV	NV	NV	NV	NV	0	100
NV	V	V	V	V	V	V	V	100	0
NV	V	V	V	V	V	V	NV	100	0
NV	V	V	V	V	V	NV	V	100	0
NV	V	V	V	V	V	NV	NV	100	0
NV	V	V	V	V	NV	V	V	100	0
NV	V	V	V	V	NV	V	NV	100	0
NV	V	V	V	V	NV	NV	V	100	0
NV	V	V	V	V	NV	NV	NV	100	0
NV	V	V	V	NV	V	V	V	100	0
NV	V	V	V	NV	V	V	NV	100	0
NV	V	V	V	NV	V	NV	V	100	0
NV	V	V	V	NV	V	NV	NV	100	0

Table 14 continued. CPT describing the influence of DIP viability on MPG viability for the BN in Figure 49. Similar CPTs were used to parameterize the Northern Cascades MPG and the Central and South Puget Sound MPG, which contain 16 and 8 steelhead DIPs, respectively. The Northern Cascades MPG contains twice as many DIPs as the other MPGs, but we used an analogous CPT to characterize the influence of DIP viability on MPG viability, which explicitly requires greater than 40% viable DIPs of each distinct life history type (summer run [SSH], winter run [WSH], or combined summer run and winter run) for MPG viability.

DIP viability								**MPG viability**	
E. Hood Canal Tribs. WSH	S. Hood Canal Tribs. WSH	Skokomish River WSH	W. Hood Canal Tribs. WSH	Sequim/Disc. Bays WSH	Dungeness R. SSH & WSH	Strait of J. de F. Tribs. WSH	Elwha River WSH	Viable	Not viable
NV	V	V	V	NV	NV	V	V	100	0
NV	V	V	V	NV	NV	V	NV	100	0
NV	V	V	V	NV	NV	NV	V	100	0
NV	V	V	V	NV	NV	NV	NV	0	100
NV	V	V	NV	V	V	V	V	100	0
NV	V	V	NV	V	V	V	NV	100	0
NV	V	V	NV	V	V	NV	V	100	0
NV	V	V	NV	V	V	NV	NV	100	0
NV	V	V	NV	V	NV	V	V	100	0
NV	V	V	NV	V	NV	V	NV	100	0
NV	V	V	NV	V	NV	NV	V	100	0
NV	V	V	NV	V	NV	NV	NV	0	100
NV	V	V	NV	NV	V	V	V	100	0
NV	V	V	NV	NV	V	V	NV	100	0
NV	V	V	NV	NV	V	NV	V	100	0
NV	V	V	NV	NV	V	NV	NV	0	100
NV	V	V	NV	NV	NV	V	V	100	0
NV	V	V	NV	NV	NV	V	NV	0	100
NV	V	V	NV	NV	NV	NV	V	0	100
NV	V	V	NV	NV	NV	NV	NV	0	100
NV	V	NV	V	V	V	V	V	100	0
NV	V	NV	V	V	V	V	NV	100	0
NV	V	NV	V	V	V	NV	V	100	0
NV	V	NV	V	V	V	NV	NV	100	0
NV	V	NV	V	V	NV	V	V	100	0
NV	V	NV	V	V	NV	V	NV	100	0
NV	V	NV	V	V	NV	NV	V	100	0
NV	V	NV	V	V	NV	NV	NV	0	100
NV	V	NV	V	NV	V	V	V	100	0
NV	V	NV	V	NV	V	V	NV	100	0
NV	V	NV	V	NV	V	NV	V	100	0
NV	V	NV	V	NV	V	NV	NV	0	100
NV	V	NV	V	NV	NV	V	V	100	0
NV	V	NV	V	NV	NV	V	NV	0	100
NV	V	NV	V	NV	NV	NV	V	0	100

Table 14 continued. CPT describing the influence of DIP viability on MPG viability for the BN in Figure 49. Similar CPTs were used to parameterize the Northern Cascades MPG and the Central and South Puget Sound MPG, which contain 16 and 8 steelhead DIPs, respectively. The Northern Cascades MPG contains twice as many DIPs as the other MPGs, but we used an analogous CPT to characterize the influence of DIP viability on MPG viability, which explicitly requires greater than 40% viable DIPs of each distinct life history type (summer run [SSH], winter run [WSH], or combined summer run and winter run) for MPG viability.

DIP viability								**MPG viability**	
E. Hood Canal Tribs. WSH	S. Hood Canal Tribs. WSH	Skokomish River WSH	W. Hood Canal Tribs. WSH	Sequim/Disc. Bays WSH	Dungeness R. SSH & WSH	Strait of J. de F. Tribs. WSH	Elwha River WSH	Viable	Not viable
NV	V	NV	V	NV	NV	NV	NV	0	100
NV	V	NV	NV	V	V	V	V	100	0
NV	V	NV	NV	V	V	V	NV	100	0
NV	V	NV	NV	V	V	NV	V	100	0
NV	V	NV	NV	V	V	NV	NV	100	0
NV	V	NV	NV	V	NV	V	V	100	0
NV	V	NV	NV	V	NV	V	NV	0	100
NV	V	NV	NV	V	NV	NV	V	0	100
NV	V	NV	NV	V	NV	NV	NV	0	100
NV	V	NV	NV	NV	V	V	V	100	0
NV	V	NV	NV	NV	V	V	NV	0	100
NV	V	NV	NV	NV	V	NV	V	0	100
NV	V	NV	NV	NV	V	NV	NV	0	100
NV	V	NV	NV	NV	NV	V	V	0	100
NV	V	NV	NV	NV	NV	V	NV	0	100
NV	V	NV	NV	NV	NV	NV	V	0	100
NV	V	NV	NV	NV	NV	NV	NV	0	100
NV	NV	V	V	V	V	V	V	100	0
NV	NV	V	V	V	V	V	NV	100	0
NV	NV	V	V	V	V	NV	V	100	0
NV	NV	V	V	V	V	NV	NV	100	0
NV	NV	V	V	V	NV	V	V	100	0
NV	NV	V	V	V	NV	V	NV	100	0
NV	NV	V	V	V	NV	NV	V	100	0
NV	NV	V	V	V	NV	NV	NV	0	100
NV	NV	V	V	NV	V	V	V	100	0
NV	NV	V	V	NV	V	V	NV	100	0
NV	NV	V	V	NV	V	NV	V	100	0
NV	NV	V	V	NV	V	NV	NV	0	100
NV	NV	V	V	NV	NV	V	V	100	0
NV	NV	V	V	NV	NV	V	NV	0	100
NV	NV	V	V	NV	NV	NV	V	0	100
NV	NV	V	V	NV	NV	NV	NV	0	100
NV	NV	V	NV	V	V	V	V	100	0
NV	NV	V	NV	V	V	V	NV	100	0

Table 14 continued. CPT describing the influence of DIP viability on MPG viability for the BN in Figure 49. Similar CPTs were used to parameterize the Northern Cascades MPG and the Central and South Puget Sound MPG, which contain 16 and 8 steelhead DIPs, respectively. The Northern Cascades MPG contains twice as many DIPs as the other MPGs, but we used an analogous CPT to characterize the influence of DIP viability on MPG viability, which explicitly requires greater than 40% viable DIPs of each distinct life history type (summer run [SSH], winter run [WSH], or combined summer run and winter run) for MPG viability.

DIP viability								**MPG viability**	
E. Hood Canal Tribs. WSH	S. Hood Canal Tribs. WSH	Skokomish River WSH	W. Hood Canal Tribs. WSH	Sequim/Disc. Bays WSH	Dungeness R. SSH & WSH	Strait of J. de F. Tribs. WSH	Elwha River WSH	Viable	Not viable
NV	NV	V	NV	V	V	NV	V	100	0
NV	NV	V	NV	V	V	NV	NV	0	100
NV	NV	V	NV	V	NV	V	V	100	0
NV	NV	V	NV	V	NV	V	NV	0	100
NV	NV	V	NV	V	NV	NV	V	0	100
NV	NV	V	NV	V	NV	NV	NV	0	100
NV	NV	V	NV	NV	V	V	V	100	0
NV	NV	V	NV	NV	V	V	NV	0	100
NV	NV	V	NV	NV	V	NV	V	0	100
NV	NV	V	NV	NV	V	NV	NV	0	100
NV	NV	V	NV	NV	NV	V	V	0	100
NV	NV	V	NV	NV	NV	V	NV	0	100
NV	NV	V	NV	NV	NV	NV	V	0	100
NV	NV	V	NV	NV	NV	NV	NV	0	100
NV	NV	NV	V	V	V	V	V	100	0
NV	NV	NV	V	V	V	V	NV	100	0
NV	NV	NV	V	V	V	NV	V	100	0
NV	NV	NV	V	V	V	NV	NV	100	0
NV	NV	NV	V	V	NV	V	V	100	0
NV	NV	NV	V	V	NV	V	NV	0	100
NV	NV	NV	V	V	NV	NV	V	0	100
NV	NV	NV	V	V	NV	NV	NV	0	100
NV	NV	NV	V	NV	V	V	V	100	0
NV	NV	NV	V	NV	V	V	NV	0	100
NV	NV	NV	V	NV	V	NV	V	0	100
NV	NV	NV	V	NV	V	NV	NV	0	100
NV	NV	NV	V	NV	NV	V	V	0	100
NV	NV	NV	V	NV	NV	V	NV	0	100
NV	NV	NV	V	NV	NV	NV	V	0	100
NV	NV	NV	V	NV	NV	NV	NV	0	100
NV	NV	NV	NV	V	V	V	V	100	0
NV	NV	NV	NV	V	V	V	NV	0	100
NV	NV	NV	NV	V	V	NV	V	0	100
NV	NV	NV	NV	V	V	NV	NV	0	100
NV	NV	NV	NV	V	NV	V	V	0	100

Table 14 continued. CPT describing the influence of DIP viability on MPG viability for the BN in Figure 49. Similar CPTs were used to parameterize the Northern Cascades MPG and the Central and South Puget Sound MPG, which contain 16 and 8 steelhead DIPs, respectively. The Northern Cascades MPG contains twice as many DIPs as the other MPGs, but we used an analogous CPT to characterize the influence of DIP viability on MPG viability, which explicitly requires greater than 40% viable DIPs of each distinct life history type (summer run [SSH], winter run [WSH], or combined summer run and winter run) for MPG viability.

DIP viability								**MPG viability**	
E. Hood Canal Tribs. WSH	S. Hood Canal Tribs. WSH	Skokomish River WSH	W. Hood Canal Tribs. WSH	Sequim/Disc. Bays WSH	Dungeness R. SSH & WSH	Strait of J. de F. Tribs. WSH	Elwha River WSH	Viable	Not viable
NV	NV	NV	NV	V	NV	V	NV	0	100
NV	NV	NV	NV	V	NV	NV	V	0	100
NV	NV	NV	NV	V	NV	NV	NV	0	100
NV	NV	NV	NV	NV	V	V	V	0	100
NV	NV	NV	NV	NV	V	V	NV	0	100
NV	NV	NV	NV	NV	V	NV	V	0	100
NV	NV	NV	NV	NV	V	NV	NV	0	100
NV	NV	NV	NV	NV	NV	V	V	0	100
NV	NV	NV	NV	NV	NV	V	NV	0	100
NV	NV	NV	NV	NV	NV	NV	V	0	100
NV	NV	NV	NV	NV	NV	NV	NV	0	100

probability that this MPG is viable would rise to 85.9% if all five summer-run DIPs and the Skagit River Summer-Run and Winter-Run DIP in this MPG were viable (100% if all five summer-run and three winter-run DIPs were viable). A sensitivity analysis (output not shown) indicates that the mutual information provided by the BN model for viability of this MPG was 0.94. The entropy reduction contributed by the viability of its constituent DIPs showed considerable variation, ranging from 0.6% (Stillaguamish River Winter Run) to 7.6% (Snoqualmie River Winter Run and Tolt River Summer Run). The MPE for the viability of this MPG is that it is not viable and there little uncertainty around this MPE: the probability that the MPG could be viable is only 4.0%.

Despite the low uncertainty about viability for all three MPGs, we also assessed the reliability of the BNs to predict MPG viability by simulating 1,000 random networks in each case. We used two metrics to assess the quality of the information provided by the networks: an error rate measured by a “confusion matrix,” which assesses how often the opinions or beliefs produced by the network fail to line up with the actual assignments to node categories based on the conditional probabilities; and a corresponding “quality of test,” which estimates the fraction of correctly predicted cases given a specified cutoff probability. In all cases we used the recommended value for the cutoff probability (40% for the Northern Cascades MPG, 20% for the Central and South Puget Sound MPG, and 20% for the Hood Canal and Strait of Juan de Fuca MPG). For the Northern Cascades MPG, the analysis indicated that the BN was a mediocre predictor of MPG viability from the data. The confusion matrix indicated an error rate of 36.3% and the ability of the test to predict correct viability results was only 63.7% at a cutoff probability of 40% or higher (100% at a cutoff probability of 0%). For the Central and South Puget Sound MPG, the BN was a better predictor of MPG viability; the confusion matrix indicated an error rate of 17.2% and the ability of the test to predict correct viability results was 82.8% at a cutoff probability of 20% or higher. In the case of the Hood Canal and Strait of Juan de Fuca MPG, the BN's predictive power for viability was similar to that for the Central and South Puget Sound MPG. The confusion matrix indicated an error rate of 16.4% and the ability of the test to predict correct viability results was 83.6% at a cutoff probability of 20% or higher. These results suggest that the networks for the two smaller MPGs provide reasonable frameworks that capture most of the variation contributing to MPG viability across these complex groups of constituent populations. The larger size and complexity of the Northern Cascades MPG contributes to the diminished capability of its network to predict how variation in its VSP parameters contributes to this MPG's viability.

Collectively, these analyses indicate that all three steelhead MPGs are at very low viability. One implication of these analyses is that increasing the abundance and productivity of DIPs of both major life history types (summer run and winter run) and spread geographically across each MPG is essential to bringing all MPGs to viable condition. Maintaining life history diversity and the spatial distribution of steelhead in IP stream reaches across DIPs within the MPGs would also be essential to elevating viability in each of them.

DPS Viability

As previously described, the viability of the DPS as a whole critically depends on the combined viability of all its constituent MPGs; a DPS with even a single nonviable MPG cannot be viable. The BN depicted in Figure 50 shows how the viability of the entire DPS depends on MPG viability.

Combining the BNs for each MPG to create a network for the entire DPS (as outlined in Figure 50) and following the conditional probabilities in Table 15 indicates that the DPS is not viable: the probability of DPS viability computed by the network is 1.0%. A sensitivity analysis (output not shown) indicates that the mutual information provided by the BN model for viability of the DPS is only 0.082. The entropy reduction contributed by the viability of the Northern Cascades MPG is 18.3%, the entropy reduction contributed by the viability of the Central and South Puget Sound MPG is 32.0%, and that for the Hood Canal and Strait of Juan de Fuca MPG is 32.9%. We assessed the ability of these networks to predict DPS viability by simulating 1,000 random networks in each case. The ability of the combined network to predict DPS viability was very high; the confusion matrix indicated an error rate of 1.0% and the ability of the test to predict correct viability results was greater than 98.9% at a cutoff probability of 2% or more. This result is not unexpected, as there is little uncertainty about the overall condition of each steelhead MPG constituting the Puget Sound DPS using these networks, and the DPS-level viability requires that all MPGs be viable.

We also used a simpler modification of these combined networks to evaluate the viability of the DIPs, MPGs, and DPS relying on only abundance and productivity criteria. The basis for this exercise was to address the question, What if we rely primarily on quantitative demographic

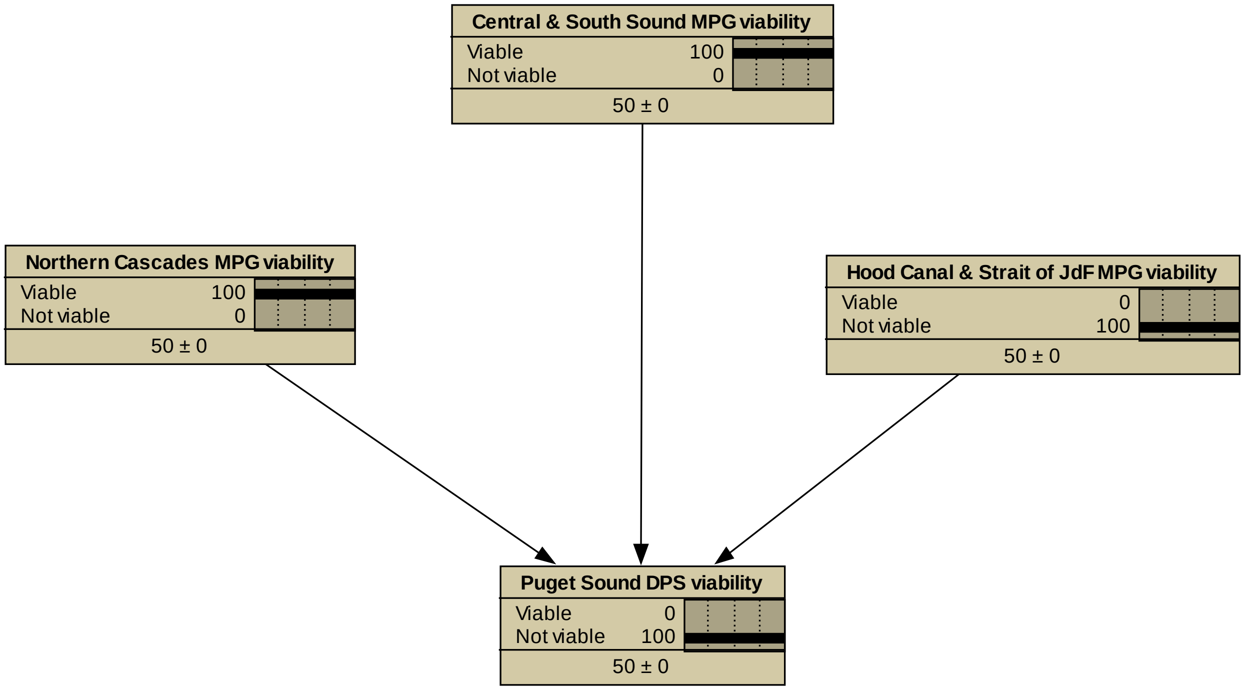


Figure 50. Highest-level BN used to characterize the viability of the Puget Sound Steelhead DPS. DPS viability requires that each MPG be viable (see Table 15); in this case, instantiation shows that one MPG that is not viable (e.g., Hood Canal and Strait of Juan de Fuca) leads to the situation where the DPS is also not viable. The viability of each MPG is estimated using BNs like that in Figure 49.

Table 15. CPT describing the influence of MPG viability on DPS viability for the BN in Figure 50.

				DPS state	
North Cascades MPG	**Central & South Puget Sound MPG**	**Hood C. & Strait of J. de Fuca MPG**	**Puget Sound DPS**	Viable	Not viable
Viable	Viable	Viable	Viable	100	0
Viable	Viable	Not viable	Not viable	0	100
Viable	Not viable	Viable	Not viable	0	100
Viable	Not viable	Not viable	Not viable	0	100
Not viable	Viable	Viable	Not viable	0	100
Not viable	Viable	Not viable	Not viable	0	100
Not viable	Not viable	Viable	Not viable	0	100
Not viable	Not viable	Not viable	Not viable	0	100

information to assess steelhead viability? To do so, we created subsets of each of the DIP BNs that included only the abundance and productivity nodes, and eliminated the iteroparity criterion from the productivity node. We used the same underlying CPTs for the abundance and productivity criteria as in the more comprehensive BNs and, as before, we used MPG-wide averages for abundance and productivity where these data were not available for individual DIPs. This analysis substantially increased the probabilities of viability at each level. Probabilities of viability for DIPs varied from a low of 64.9% for the Stillaguamish River Winter Run to a high of 86.8% for the Canyon Creek Summer Run in the Northern Cascades MPG. The MPE for the Stillaguamish River Winter Run was that it is viable, but there was a 75% probability that this DIP is not viable; the MPE for the Canyon Creek Summer Run was that it is viable, and there was only a 5.5% probability that this DIP is not viable. For the Central and South Puget Sound MPG, probabilities of DIP viability varied from a low of 61.3% for Cedar River Winter Run to a high of 69.5% for White River Winter Run. The MPE for the Cedar River Winter Run was that it is viable, but there was a 75% probability that this DIP is not viable; the MPE for the White River Winter Run was that it is viable, but the probability that this DIP is not viable was estimated at 49.9%. For the Hood Canal and Strait of Juan de Fuca MPG, probabilities of DIP viability varied from a low of 61.8% for the Elwha River Winter Run to a high of 72.0% for West Hood Canal Tributaries Winter Run. The MPE for the Elwha River Winter Run was that it is viable, but there was a 75% probability that this DIP is not viable; the MPE for the West Hood Canal Tributaries Winter Run was that it is viable, but the probability that this DIP is not viable was estimated at 49.9%.

Under this analysis, the probabilities of viability for the three MPGs were 89.9% for the Northern Cascades MPG, 74.6% for the Central and South Puget Sound MPG, and 70.5% for the Hood Canal and Strait of Juan de Fuca MPG. The MPEs for MPG viability were that all three MPGs are viable, with probabilities that each MPG is not viable of 12.4%, 31.6%, and 56.2%, respectively. Finally, the probability of viability for the entire DPS was 42.3%; the MPE was that the DPS is viable, with a probability that the DPS is not viable equal to 56.2%.

This exercise illustrates how the structure of a BN can strongly influence the inference about viability at nodes throughout the network. If we rely on the more quantifiable and readily available of the VSP criteria alone, estimates of steelhead viability are higher throughout the DPS for two main reasons: the PSS TRT's largely qualitative assessment of diversity and spatial distribution of steelhead is that the widespread state of these factors in Puget Sound steelhead poses risk to sustainability, and the TRT's more quantitative assessment of demography (abundance and productivity) is conservative with respect to its influence on risk to persistence. If one were to accept this analysis (and its underlying probability theory) as a reasonable evaluation of the status of Puget Sound steelhead, one would still have to conclude that there is substantial uncertainty in the status of at least two of the MPGs and the entire DPS. Furthermore, the main problem with this analysis is that it is at odds with the threatened status of the DPS and with concerns about continued risks to persistence and sustainability of many of its populations. It also indicates that the existing BNs may be too benign with respect to risk of quasi-extinction; the viabilities for several DIPs and all the MPGs are at odds with the outcomes of several of the PVAs, in part because of the MPG-wide averaging for DIPs with no quantitative information. It is also probably due in part to the fact that this analysis does not consider the effects of factors that are known to be risks to wild Puget Sound steelhead, including limited availability of productive rearing and spawning habitat, the widespread use of Chambers Creek and Skamania River hatchery fish throughout the DPS, and perceived changes in aspects of diversity for several populations.

The TRT considers the networks that encompass all four VSP criteria to more accurately reflect the current status of Puget Sound steelhead. The vast majority of steelhead populations throughout Puget Sound do not appear to be viable, regardless of where one looks; most are declining steeply from historical levels and many are very small. Despite the lack of demographic information for many of the populations and the lack of clarity regarding the precise states of diversity and spatial structure for nearly all of them, these BNs appear to provide a framework that assesses the broad, overall view of steelhead viability with reasonable confidence. The TRT considers these networks to represent a reasonable starting point for identifying the major criteria for effective recovery of Puget Sound steelhead, and believes that they help to identify key gaps in knowledge that must be addressed if further declines in steelhead viability are to be arrested.

Diversity Considerations for Steelhead Viability: Resident Life History and Iteroparity

In this section, we briefly consider how the presence of freshwater resident *Oncorhynchus mykiss* and the degree of iteroparity may affect viability of steelhead. Both factors may mitigate extinction risk, but few analyses are available in the published literature to guide their consideration in a viability analysis of anadromous fish. We briefly describe NMFS's previous considerations of these factors and outline an approach and justification for including them in our overall assessment of viability.

Resident Fish as a Factor Influencing Viability

O. mykiss exhibits varying degrees of anadromy. Nonanadromous forms are usually called rainbow trout. Although the anadromous and nonanadromous forms have long been taxonomically classified within the same species, in any given area the exact relationship between the forms is not well-understood.

The 2003 Biological Review Team (BRT) (Good et al. 2005) had to consider in more general terms how to conduct an overall risk assessment for a DPS that includes both resident and anadromous populations, particularly when the resident fish may outnumber the anadromous ones but their biological relationship is unclear or unknown. Some guidance is found in Waples (1991), which outlines the scientific basis for the NMFS ESU policy. That paper suggested that an ESU (or equivalently, a DPS) that contains both forms could be listed based on a threat to only one of the life history traits "if the trait were genetically based and loss of the trait would compromise the 'distinctiveness' of the population" (p. 16). That is, if anadromy were considered important in defining the distinctiveness of the ESU, loss of that trait would be a serious ESA concern. In discussing this issue, the NMFS ESU policy (NMFS 1991) affirmed the importance of considering the genetic basis of life history traits such as anadromy and recognized the relevance of a question one commenter posed: What is the likelihood of the nonanadromous form giving rise to the anadromous form after the latter has gone locally extinct? To focus the issue, Good et al. (2005) considered a hypothetical scenario that has varying degrees of relevance to individual steelhead ESUs/DPSs. In this scenario, the once-abundant and widespread anadromous life history is extinct, or nearly so, but relatively healthy native populations of resident fish remain in many geographic areas. The question the BRT (Good et al. 2005) had to consider was: Under what circumstances would one conclude that such a DPS was not in danger of extinction or likely to become endangered? The BRT identified the required conditions as:

- The resident forms are capable of maintaining connectivity among populations to the extent that the DPS's historical evolutionary processes are not seriously disrupted.

- The anadromous life history is not permanently lost from the DPS but can be regenerated from the resident forms.

The task faced by the PSS TRT ideally involves a finer scale evaluation of the contribution of resident fish to the viability of individual anadromous DIPs. While Good et al. (2005) developed generalized guidelines to evaluate the relationship between the two life history forms, despite differences in that relationship for both coastal and interior steelhead DPSs, the TRT was able to focus on coastal *O. mykiss* in Puget Sound. Additionally, the TRT benefited from a number of recent studies on the interactions between Puget Sound resident and anadromous *O. mykiss*. In general, there appeared to be a relatively close relationship between sympatric resident and anadromous forms below long-standing natural barriers. This may be due in part to the relatively short geologic time period since the Pleistocene glaciations. It may also be that, below impassable barriers, truly resident populations do not exist. Rather, the degree of anadromy in an *O. mykiss* DIP may be somewhat plastic, with environmental and ecological cues influencing the relative rate of anadromy. For example, in the Lake Washington basin (including the Cedar River) the anadromous populations of *O. mykiss* and cutthroat trout (*O. clarkii*) have dwindled to near zero levels, yet resident fish of both species are widely abundant. This is thought to be due in part to changes (improvements) in the productivity of Lake Washington and Lake Sammamish.

In determining the viability of steelhead DIPs in the Puget Sound DPS, the TRT considered the potential influence of co-occurring resident *O. mykiss* on anadromous steelhead demographics. Based on recent studies (see References section, e.g., Zimmerman and Reeves 2000, Thrower et al. 2004, Pearse et al. 2009), it is clear that there is some degree of interaction between resident and anadromous fish. This interaction can be both genetic and ecological (competition, predation, etc.) and has the potential to be a positive or negative influence on steelhead viability. Interactions between resident and anadromous fish can be especially beneficial when the abundance of anadromous fish is especially low, near QET. Thus resident fish may be most important not in bringing a DIP to full viability, but in preventing a DIP from being extirpated. Any reduction in risk of extinction would be directly linked to the genetic relationship between resident and anadromous *O. mykiss* and the sustainability of the resident population. Resident *O. mykiss* could be the result of nonnative introductions from other basins within Puget Sound or from outside of the DPS (most prominently in northern California). Alternatively, resident fish may be emigrating into anadromous-accessible areas from upstream of stream structures impassable to upsteam migrants. In either case, where substantial genetic differences exist between resident and anadromous fish, it is unlikely that interbreeding between these two life history forms will be beneficial to the fitness of steelhead. Where genetic differences between the two forms are more moderate, the two life history forms may have recently diverged or may coexist with continued gene exchange.

If resident *O. mykiss* are a potential reservoir of the anadromous genetic legacy in a steelhead population, they can play an important role in the viability of that population. This is especially important in areas that historically contained anadromous *O. mykiss*, but have been rendered inaccessible to upstream migration post-European contact. The TRT concluded that in basins where anadromous *O. mykiss* abundance is below the QET threshold, the risk of extinction is not necessarily 100% if resident *O. mykiss* are present below long-standing migration barriers. Inclusion of resident fish in the viability criteria is dependent, however, on several conditions. First, the abundance of resident fish must be large enough to be self-sustaining (this will be largely dependent on the numbers of resident females). Secondly, there must be some evidence of interbreeding between the anadromous and resident forms (this will most likely be established by genetic analysis of the relationship between the resident fish and the most proximate anadromous population). Identifying resident fish that contain the genetic legacy of their historical anadromous population is an important step in potentially reestablishing anadromous *O. mykiss* to some basins and subbasins.

A comprehensive risk assessment must consider the effect of resident *O. mykiss* that have anadromous access on the viability of the entire DPS. This task is especially difficult because little or no information is available about the abundance and distribution of resident fish, or about the extent and nature of their interactions with anadromous populations. The 2003 BRT incorporated information about rainbow trout populations into their analyses of the four VSP criteria and their assessments of extinction risk for *O. mykiss* ESUs (Good et al. 2005). In several ESUs, Good et al. (2005) concluded the presence of relatively numerous rainbow trout populations reduced risks to ESU abundance. However, there is considerable scientific uncertainty regarding the potential of the resident form to contribute to the productivity, spatial structure/connectivity, and diversity of steelhead ESUs (Varanasi 2004).

Good et al. (2005) underscored the importance of the anadromous life history form in reducing risks to these latter three VSP parameters, and thus in contributing to a viable *O. mykiss* ESU or DPS in total. Although there is the potential for rainbow trout populations to generate steelhead migrants, it may be short-lived if the reproductive success of steelhead offspring is low. Finally, the BRT concluded if the anadromous life history form in an ESU is extirpated or critically depressed, it is unlikely the resident life history form alone is capable of maintaining the productivity, connectivity, and diversity necessary for a viable *O. mykiss* ESU (Good et al. 2005).

Subsequent to the conclusions of the 2003 BRT, NMFS solicited opinions from two expert panels to review the issue of viability in listing units that contain both rainbow trout and steelhead (note that the language below refers to ESUs, although listing units for *O. mykiss* are now considered DPSs). The independent Recovery Science Review Panel (RSRP) identified anadromy as “an evolutionarily significant component of *O. mykiss* diversity” (RSRP 2004). In its review of available information, the panel concluded “resident populations by themselves should not be relied upon to maintain long-term viability of an ESU.” Similarly, the Independent Scientific Advisory Board (ISAB) found the long-term consequences of the extirpation of a major life history form would have deleterious consequences on the entire ESU (ISAB 2005).

To be viable, an ESU or DPS needs more than simple persistence over time; it needs to be sustained in an ecologically and evolutionarily functional state. Evaluation of ESU/DPS viability should not only rest on the numbers of component populations or on the abundance and productivity of those individual populations, but also should be based on the integration of population dynamics within the ecosystem as a whole. This concept of ESU/DPS viability does not accommodate the loss of populations or the anadromous or resident life history form from any given ESU/DPS, because that loss would represent a loss in diversity for the ESU/DPS that would put its long-term viability at risk.

Where both life history forms are present, the ISAB considered that the resident forms contribute to the overall abundance and diversity of an ESU/DPS, but was unsure of the contribution by resident fish to connectivity and spatial structure. Overall, the presence of both resident and anadromous life history forms is "critical for conserving the diversity of steelhead/rainbow trout populations and, therefore, the overall viability of ESUs."

In a 2004 review of currently listed steelhead ESUs, the Northwest Fisheries Science Center concluded that "None of these ESUs is likely to persist in total into the foreseeable future, because substantial parts of the ESUs are at risk of extinction" (Varanasi 2004). That review supported the 2003 BRT conclusions that the ESUs were at risk of extinction, now or in the foreseeable future, because the anadromous life history represented a "significant portion of the species 'range,' such that its loss is a direct threat to the ESUs" (Varanasi 2004).

Courter et al. (2010, 2013) conducted deterministic simulation modeling of population viability for an interior population of *O. mykiss* with both resident and anadromous life history types (Upper Yakima River) and concluded that extinction risk was low when production of steelhead from both resident and anadromous fish was accounted for. Viability was highly sensitive to the quasi-extinction risk thresholds for both resident spawners and anadromous steelhead, but Courter et al. (2010, 2013) found that natural variation in stage-specific survival rarely caused steelhead abundance to drop below 25 spawners over 4 consecutive years, yet almost always caused abundance to drop below 100 spawners. The modeling indicated that after 1,000 iterations, steelhead abundance dropped below a QET of 50 spawners in about a quarter of the simulated scenarios. Courter et al. (2010) stated that "our modeling as well as historic fish counts reveal that a reduction in steelhead below 50 spawners across 4 or more consecutive years does not lead to extinction when resident rainbow trout are abundant in the population." They concluded that their data highlights the sensitivity of steelhead abundance to marine survival and demonstrates that when resident fish are highly abundant and contribute sufficient smolts, steelhead can be maintained in the population at low abundance.

These results are consistent with the evidence for plastic and genetic influences on migratory life history in this species. In at least some coastal systems, smolt production from resident fish has been documented, although the two forms can sometimes be genetically differentiated. Pearse et al. (2009) found evidence for rapid evolution of a resident life history in a coastal California steelhead population above a natural migration barrier in response to selection against seaward migration. In cases where anadromous steelhead abundance is very low in a population that also harbors a resident life history, smolt production from resident fish may contribute substantially to steelhead viability. In their controlled breeding study of *O. mykiss* from Sashin Creek, Alaska, Thrower et al. (2004) found that resident fish there can produce substantial numbers of smolts, albeit with lower marine survival (see also Thrower and Hard 2009). They concluded that "the judicious use of freshwater sequestration merits consideration as a temporary component of a comprehensive strategy for the maintenance of endangered anadromous populations of *O. mykiss* in cases where the likelihood of rapid restoration of freshwater habitats for anadromous fish is low."

An *O. mykiss* population expressing a combination of migratory strategies (freshwater resident, anadromous) and a heritable propensity to produce both types of progeny means that residents can serve as a buffer when anadromous productivity is low—extinction risk is lower when residents are relatively abundant. But it is important to remember that while a population of residents may indeed provide a genetic reservoir to produce anadromous migrants, these fish may have reduced adaptive potential to cope with the marine environment and smolt production may entail a cost in fitness. This is an active area of research that merits immediate attention to help guide development of conservation strategies for steelhead. For our analysis, we simply assumed that the presence of locally adapted, resident fish with an opportunity to interact with steelhead in Puget Sound (e.g., in the Cedar River watershed) limited extinction risk for steelhead to a level not exceeding 90% in 100 years.

Iteroparity as a Factor Influencing Viability

Steelhead are iteroparous anadromous salmonids. It is widely thought that this ability to reproduce repeatedly over the lifespan can contribute significantly to population productivity and provide a temporal buffer in demographic and environmental stochasticity. In the Pacific Northwest, the current frequency of repeat spawners appears to be low, generally less than 10% and frequently much lower, and rates appear to be declining in several populations in recent decades (e.g., Figure 51). A systematic analysis of the contribution of repeat spawning to population dynamics has not, to our knowledge, been undertaken for coastal steelhead. Nevertheless, a comprehensive evaluation of population viability must consider this contribution,

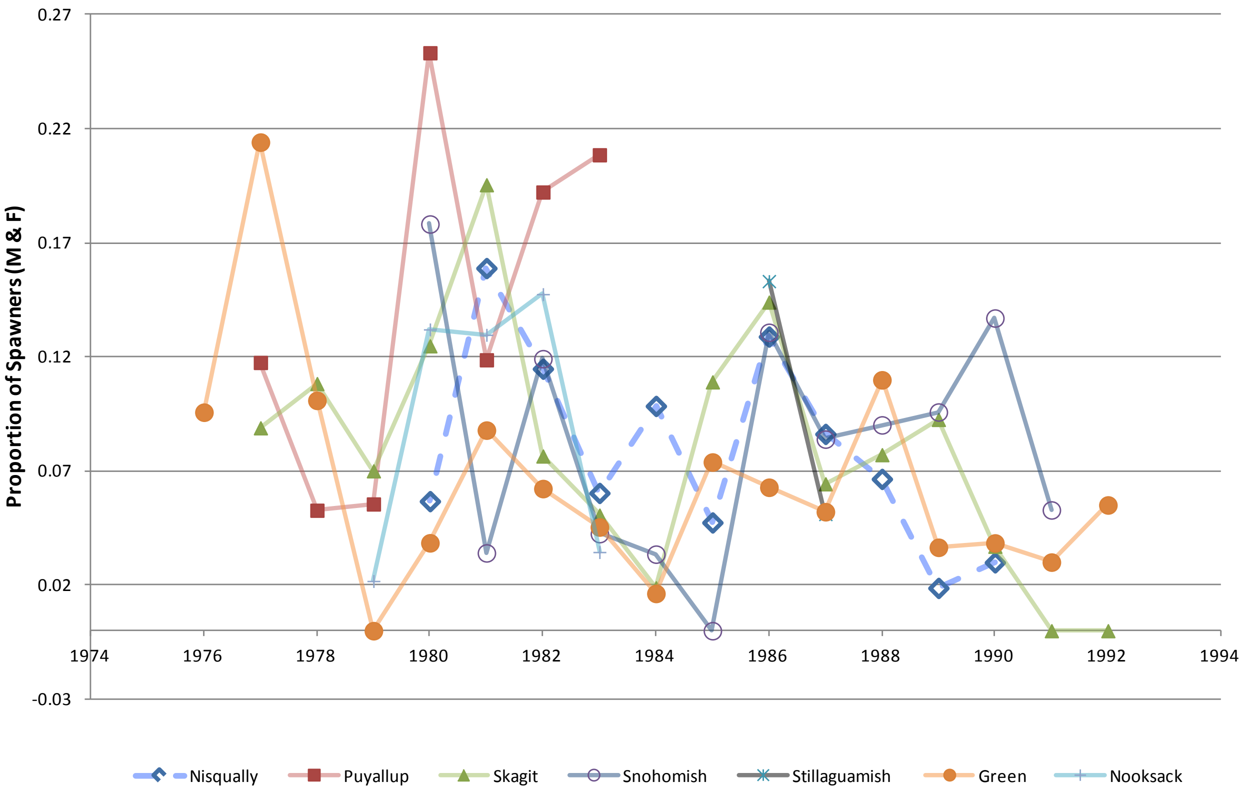


Figure 51. Temporal changes between the 1970s and 1990s in the estimates of repeat spawning rates for seven winter-run steelhead populations from the Northern Cascades MPG and the Central and South Puget Sound MPG.

and that is a primary reason why iteroparity is a component of the BNs for viability developed in the previous section of this report.

In this subsection of the report, we summarize some preliminary analyses that incorporate the proportion of repeat spawners in adult populations as a viability parameter influencing productivity. These unpublished analyses, conducted by Nick Gayeski of the Wild Fish Conservancy in coordination with and reviewed by the PSS TRT, rely on an approach that develops a suite of age-structured matrix population projection models with annual time steps that incorporate density dependence, varying levels of repeat spawning, and varying levels of density-independent stochasticity and harvest mortality. The models were used to evaluate the influence of varying levels of repeat spawning on population abundance and age structure under varying levels of harvest.

The models incorporated iteroparity under deterministic and stochastic conditions, with and without harvest mortality. Gayeski's models simulated a winter-run steelhead population with six age-classes and three ages at maturation, with age of first maturation at age 4. The entire population was assumed to be anadromous and had no interactions with resident fish. Smolts were assumed to be all age 2, and repeat spawners were limited to one repeat spawning event in the year following the year of first reproduction. Therefore, repeat spawners were either 5 or 6 years old. All harvest mortality occurred on mature fish following river entry and prior to spawning. Sex ratio was 0.5.

Female fecundity scaled with size and Kamchatka wild steelhead data were used to determine individual fecundities. First-time spawners were assumed to deposit more eggs than repeat spawners of the same age, reflecting the assumption that repeat spawners pay a cost in fecundity due to the costs of first spawning and to having less time available for growth in the ocean after spawning. This fecundity/growth cost is in addition to a survival cost that is paid for having matured and spawned at a younger age rather than remaining in the ocean and maturing 1 or 2 years later.

Density-dependence was modeled as a Beverton-Holt functional response. There is a scarcity of data on age-specific and stage-specific survival or mortality rates for adult and juvenile steelhead. The model employed values from a few of the steelhead studies available and theoretical considerations based on life history theory and allometry to produce estimates for both juvenile and adult survival rates (Ward and Slaney 1993, McGurk 1996, Quinn 2005).

The effect of stochasticity was evaluated by modeling random variation in the smolt-to-age-3 transition rate as a random variable drawn from a beta distribution with specific mean and coefficient of variation. Two ocean survival scenarios were evaluated that span reasonable levels of favorable and unfavorable variation in this transition rate.

Figure 52 summarizes some of the deterministic model results under a 20% harvest rate, assuming smolt-to-adult survival rates of 15% and a variety of repeat spawning rates. The median abundance declines from 330 for a population with 25% repeat spawners (which is more than 100 adults lower than without harvest) to just above 250 with no repeat spawners.

From his modeling, Gayeski concluded that:

1. The average proportion of repeat spawners in an adult steelhead population is relevant to population abundance and stream capacity in two ways: first, relative to a population with no repeat spawners, for a given average capacity of juveniles, a population with repeat spawners will have a larger average adult spawning population; second, a given average number of adult spawners can be sustained by fewer juveniles when repeat spawners are present than when they are not. It is therefore likely that viable populations (DIPs) of steelhead can be sustained in smaller stream basins than is the case for Pacific salmon.
2. When population resilience to environmental variation or harvest mortality is measured in terms of either the probability of declining below specific levels of annual spawner abundance or the expected frequency of the spawning population declining below some threshold of concern (quasi-extinction), repeat spawning provides increased levels of resilience compared to populations without repeat spawning. However, the impact of harvest mortality on the average proportion of repeat spawning confounds understanding of precisely how much resilience a given mean level of repeat spawning (measured under deterministic conditions) affords a population, especially at low levels of total abundance. The significance of repeat spawning may be weakened (or underestimated) when small population sizes are considered.
3. When both harvest mortality and environmental variation in smolt survival are present, repeat spawning increases population resilience. The precise extent to which resilience is increased is sensitive to the harvest rate and to both the magnitude of the average value of life-stage survival rates subject to environmental variation and the amount of variation in the rate as measured by the coefficient of variation.
4. From a population rebuilding and recovery perspective, for small population sizes such as the ones considered in this modeling exercise, it appears that the value of specific levels of repeat spawning to population resilience (and repeat spawning) are most likely to be realized under a zero harvest scenario, regardless of the level of environmental variation (at least for the range and kind of variation employed in the models).

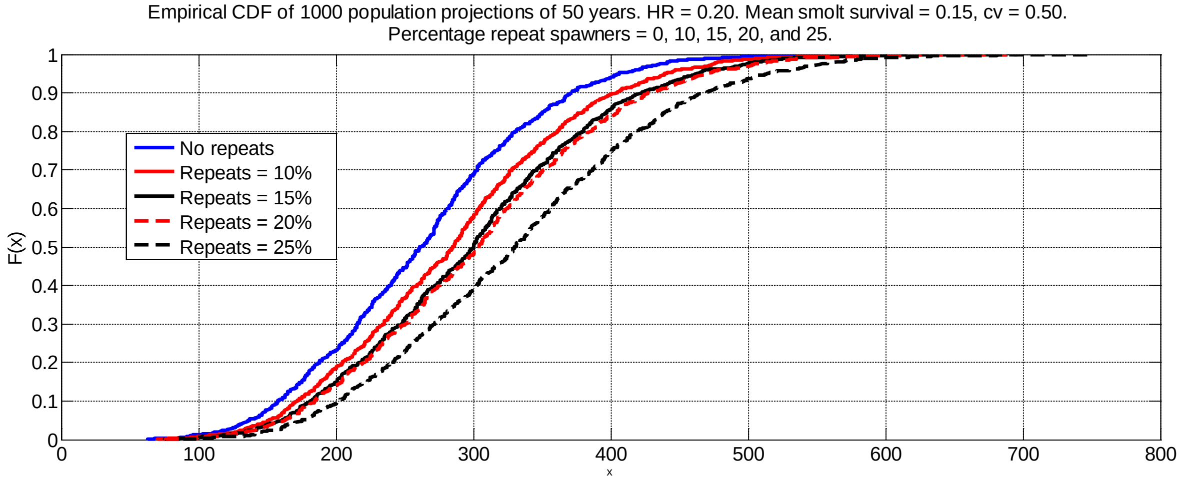


Figure 52. Empirical cumulative densities (CDF) of 1,000 deterministic population projections over 50 years for steelhead exposed to a harvest rate of 20% under a mean smolt survival of 15% with a coefficient of variation of 50%. The different curves represent different repeat spawning rates ranging from 0 to 25%.

These exercises are simulations with assumptions that may not be realistic for many natural steelhead populations, and demonstrating the effect of specific levels of repeat spawning on population resilience would realistically require a no-harvest regime considering the extent of natural environmental variation influencing survival, growth, and iteroparity. A comprehensive parentage would be required to fully address this question and provide empirical evidence to support these conclusions. That said, these analyses reinforce the TRT's conviction that iteroparity is an important consideration in a comprehensive evaluation of viability for steelhead. Iteroparity is also arguably an important factor for diversity (and also for population persistence through temporal risk spreading), but the TRT did not consider this issue quantitatively. The TRT thinks that the degree of iteroparity is likely to be especially influential on viability in small populations during periods when marine mortality varies widely (see also the Integral Projection Model analysis of a small wild steelhead population described in Appendix G).

Viability Criteria for Puget Sound Steelhead

Under the ESA, NMFS is required to identify measurable and objective delisting criteria as part of recovery planning. These delisting criteria must describe the conditions under which a listed species or DPS is no longer in danger of extinction (endangered) or likely to become so in the foreseeable future (threatened). We define a viable DPS as one that is unlikely (with less than an estimated 5% probability) to be at risk of extinction in the foreseeable future; for this purpose, we adopt the VSP criteria of a specified QET and a 100-year timeline (McElhany et al. 2000) to evaluate risk of extinction. Ultimately, the identification of delisting criteria requires the consideration of technical analyses relating to viability, which are contained in this steelhead document, and policy decisions such as acceptable levels of risk, which are not. This document presents the PSS TRT's recommended biological viability criteria.

The listed unit under the ESA for *Oncorhynchus mykiss* including anadromous steelhead is the DPS, and this is the unit that must be considered for delisting. Thus delisting criteria must ultimately address the extinction risk of the DPS. In approaching the development of criteria for assessing viability of (and, subsequently, delisting) DPSs, we have relied on the language in the ESA, information described in the listing decision, concepts outlined in the McElhany et al. (2000) VSP document, and published research describing salmon populations and their past or potential responses to environmental changes. The ESA lists five potential factors for decline that must be considered in species listing decisions (ESA Section 4.2.1):

1. The present or threatened destruction, modification, or curtailment of its habitat or range;
2. Overutilization for commercial, recreational, scientific, or educational purposes;
3. Disease or predation;
4. The inadequacy of existing regulatory mechanisms; and
5. Other natural or man-made factors affecting its continued existence.

NMFS identified all five factors as contributing to the endangerment of Pacific salmonids, including steelhead. In considering how viability criteria might inform biologically based population delisting requirements, the TRT also attempted to consider these factors for decline. The NMFS (2010) population recovery approach for threatened Puget Sound Chinook salmon (see also Ruckelshaus et al. 2006) identified six delisting criteria for this ESU.

1. The viability status of all populations in the ESU is improved from current conditions.
2. At least two and up to four Chinook salmon populations in each of five biogeographical regions (i.e., MPGs) within the ESU achieve viability, depending on the historical biological characteristics and acceptable risk levels for populations within each region.
3. At least one population from each major genetic and life history group historically present within each of the five biogeographical regions is viable.

4. Tributaries to Puget Sound not identified as primary freshwater habitat for any of the 22 identified populations are functioning in a manner that is sufficient to support an ESU-wide recovery scenario.
5. Production of Chinook salmon from tributaries to Puget Sound not identified as primary freshwater habitat for any of the 22 identified populations occurs in a manner consistent with an ESU recovery.
6. Populations that do not meet the viability criteria for all VSP parameters (i.e., abundance, productivity, diversity, and spatial structure) are sustained to provide ecological functions and preserve options for ESU recovery.

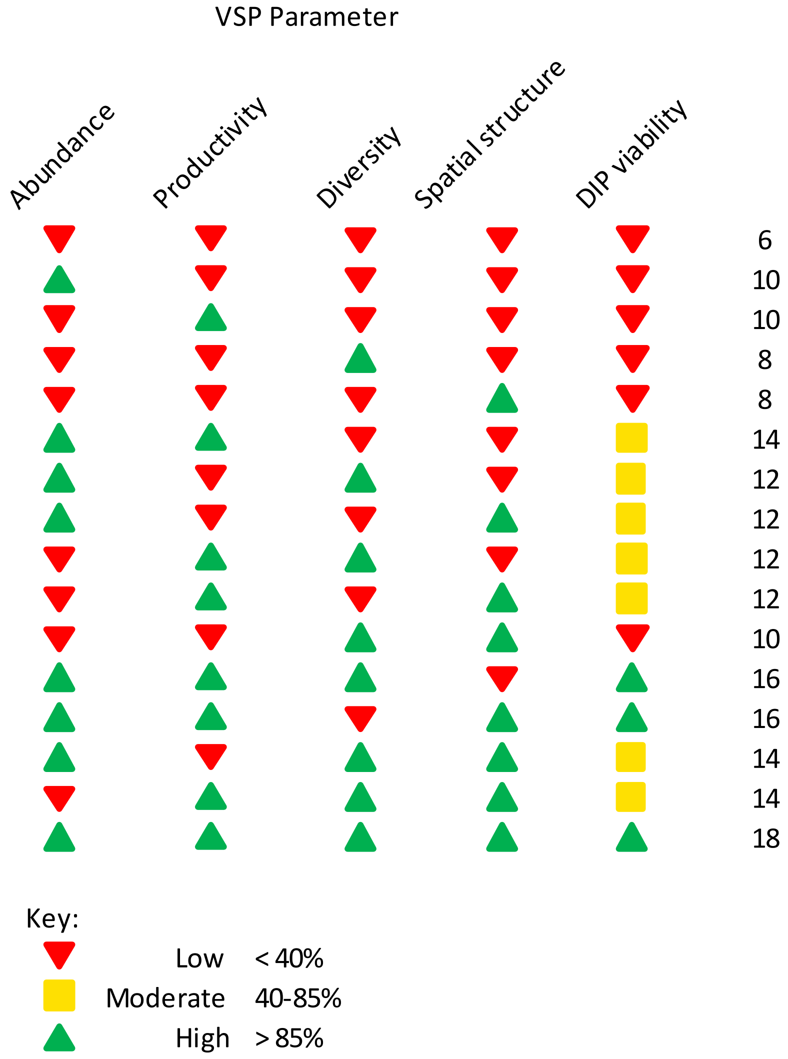
The TRT approach evaluated the extinction risks facing the DPS by assessing the viability of the individual populations (DIPs) and MPGs within that DPS. The TRT identified population-level viability criteria based on a combination of the four parameters indicated in the VSP document and information about the habitat requirements of the listed fish. From the analyses in this report and building from these population-level criteria, the TRT identified viability criteria for the Puget Sound steelhead DPS that take into account these key VSP parameters and incorporate hierarchical viability criteria for MPGs and for the entire listed unit. Some of these criteria for steelhead share similarities to NMFS population recovery approach criteria for Puget Sound Chinook salmon listed above.

1. The viability (as reflected in abundance, productivity, diversity, and spatial structure) of a majority of steelhead populations in each of the MPGs across the DPS is detectably higher than currently, using an accepted form of population viability analysis and additional tools like those described in this report.
2. At least 40% of steelhead populations in each of the three MPGs within the DPS achieve viability, depending on historical biological characteristics and acceptable levels for risk to population persistence within each region. To be viable, an MPG's constituent DIPs must have a mean viability above a specified threshold (described in the DIP Viability subsection below).
3. A minimum of 40% of summer-run and 40% of winter-run populations historically present within each of the MPGs must be viable using the VSP-based criteria described in this report.
4. Natural production of steelhead from tributaries to Puget Sound not identified as primary spawning or rearing habitat in any of the 32 identified populations is sustained to provide sufficient ecological diversity and productivity to support DPS-wide recovery.

The viability analyses described earlier (in the Viability Assessment Methods section and the Decision Support Systems as Tools for Assessing Viability section) indicate that sufficient abundance and productivity from each major life history type in a sufficient fraction of populations spread spatially across each MPG are critical to the viability of each MPG, which DPS viability depends on. That said, diversity and spatial structure—which appear to be limited throughout the DPS, based on limited data—make essential contributions to MPG (and therefore DPS) viability as well.

From these considerations, the PSS TRT developed the simple viability criteria framework outlined below. The TRT developed these criteria using the Puget Sound Steelhead DPS population structure identified in its previous publication, Identifying Historical Populations of Steelhead within the Puget Sound Distinct Population Segment (Myers et al. 2015) and implemented here: a DPS composed of 3 MPGs containing a total of 32 historically present DIPs. The configuration of the Puget Sound Steelhead DPS adopted in this report is based on information compiled to date and it is possible, perhaps likely, that additional data will suggest future changes. These changes would most likely involve the number of DIPs identified rather than the number or configuration of the MPGs.

The criteria for viability for each unit within the DPS follow basic “traffic light” rule sets (summarized in Figure 53 through Figure 55). Using a three-bin classification (low = not viable, moderate = intermediate, and high = viable), the DIPs in an MPG are scored at each of the four VSP criteria with 1 (red inverted triangle), 2 (yellow square), or 3 points (green upright triangle), corresponding to the contributions of these scores to DIP viability. These scores reflect



VSP Parameter

Abundance	Productivity	Diversity	Spatial structure	DIP viability	
▼	▼	▼	▼	▼	6
▲	▼	▼	▼	▼	10
▼	▲	▼	▼	▼	10
▼	▼	▲	▼	▼	8
▼	▼	▼	▲	▼	8
▲	▲	▼	▼	■	14
▲	▼	▲	▼	■	12
▲	▼	▼	▲	■	12
▼	▲	▲	▼	■	12
▼	▲	▼	▲	■	12
▼	▼	▲	▲	▼	10
▲	▲	▲	▼	▲	16
▲	▲	▼	▲	▲	16
▲	▼	▲	▲	■	14
▼	▲	▲	▲	■	14
▲	▲	▲	▲	▲	18

Key:

▼	Low	< 40%
■	Moderate	40-85%
▲	High	> 85%

Figure 53. Viability criteria framework for DIPs of Puget Sound steelhead.

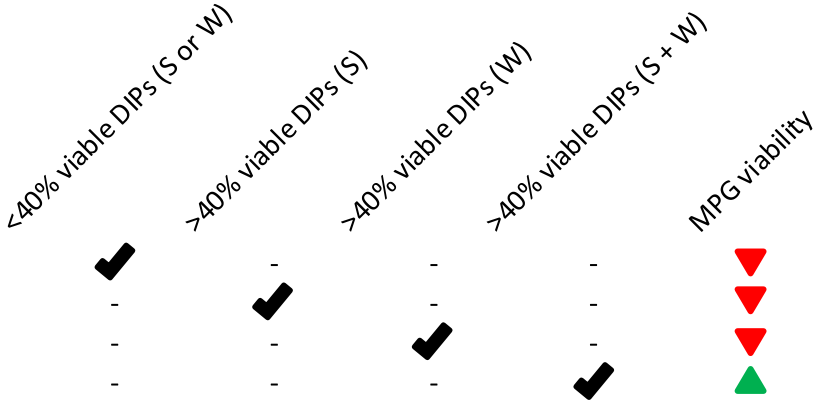


Figure 54. Viability criteria framework for MPGs of Puget Sound steelhead based on the viability of component DIPs containing distinct life history types (S = summer run and W = winter run). Key is in Figure 53.

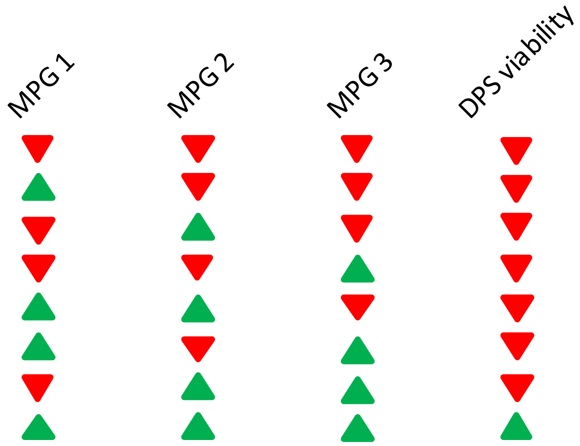


Figure 55. Viability criteria framework for the entire Puget Sound Steelhead DPS. Key is in Figure 53.

probabilities that DIP viability is influenced by the VSP criteria, ranging from less than 40% to 40–85% and greater than 85%, respectively (Figure 53).

MPG viability depends on two criteria. First, as indicated in Figure 54, a minimum of 40% of the DIPs in the MPG that exhibit each of the two distinct life history strategies (summer run vs. winter run) must be viable using the same “traffic light” rule set (i.e., score = 3). (The threshold of 40% corresponds to an average probability of DIP viability of 67% if the DIPs that are not viable have probabilities of viability of 50%.) Second, the DIPs in the MPG must have a geometric mean score of at least 2.2, a value the TRT determined based on consideration of the first criterion and examining how MPG viability under the traffic light rule set varies. Following these two criteria, to be viable an MPG with 8 DIPs must have at least 4 viable DIPs (score = 3) and no more than 1 DIP that is considered not viable (score = 1). An MPG with 11 winter-run and 5 summer-run DIPs must have at least 5 viable winter-run and 2 viable summer-run DIPs and no more than 1 DIP of each life history type that is considered not viable.

Finally, DPS viability depends only on one criterion: that each of its component MPGs is considered viable (Figure 55).

The viability criteria for each level within the DPS are summarized below.

DIP Viability

1. Using a simple three-bin classification (not viable, intermediate, viable), a DIP is scored 1, 2, or 3, depending on whether the probability of viability as computed by its viability BN is less than 40%, 40–85%, or greater than 85%, respectively. For a DIP to be considered viable, its probability of viability must be at least 85%, as calculated by the BN (see Appendix F). To estimate this value, the mean abundance and productivity criteria for the candidate DIP are each double-weighted; this produces DIP viability scores ranging from 6 to 18 points. DIPs with viability scores less than 11 are considered not viable, those with viability scores between 11 and 14 are considered to have intermediate viability, and those with viability scores greater than 14 (>85% rounded up) are considered viable.

MPG Viability

2. An MPG is considered viable if 40% (rounded up) of its DIPs (including extinct as well as extant historical populations) are viable and mean DIP viability exceeds the specified threshold for viability (Table 16).
    a. DIPs exhibiting distinct life history strategies (summer run vs. winter run) will be considered separate components of the MPG. Therefore, a minimum of 40% of summer-run and 40% of winter-run populations within an MPG must be viable to achieve MPG viability.
    b. DIPs containing both winter-run and summer-run subpopulations predominantly exhibit the winter-run life history strategy in Puget Sound and will be considered winter run for the purpose of estimating 2a.
    c. A viable MPG must, in addition to the criterion outlined in 2a and 2b, have a geometric mean score of at least 2.2 to be considered viable. This score is the minimum achieved across a range of MPG sizes (with number of DIPs ranging from 8 to 16) with a sufficient fraction of DIPs having viabilities of greater than 85%.

DPS Viability

3. A DPS is considered viable only if all its component MPGs are viable.

A chart of representative viability criteria applied to the DPS to depict their relationships is in Figure 56. This figure emphasizes the hierarchical nature of the criteria across the levels of the components of the DPS. The framework partitions criteria at the DIP level between persistence and sustainability factors related to VSP components. For example, the framework considers spawner abundance, productivity, occupancy and density of suitable habitat by adults and juveniles, frequency of iteroparity, and sources of human-induced mortality as factors that

Table 16. Number of viable DIPs required for MPG viability in each of the Puget Sound steelhead MPGs with a requirement of 40% viable DIPs of each major life history type. Some winter-run DIPs may include some summer-run fish as well, which is most common in the Northern Cascades MPG and along the Strait of Juan de Fuca.

MPG	Life history type	No. of DIPs	No. viable
Northern Cascades	Summer run	5	2
	Winter run	11	5
Central and South Puget Sound	Summer run	0	0
	Winter run	8	4
Hood Canal and Strait of Juan de Fuca	Summer run	0	0
	Winter run	8	4

influence demography primarily and, therefore, population persistence. It considers effective population size, influence of hatchery fish (both genetic and ecological impacts), age variation in spawners, and variation in spawn timing as factors that influence diversity primarily and, therefore, population sustainability. The framework also conveys the importance of having populations distributed across major habitat types, having connectivity between resident and anadromous fish where they occurred historically, and minimizing risk to multiple DIPs due to catastrophic risk. It attempts to include factors considered important to viability, even if current information is inadequate to characterize their influence on viability estimates accurately. Finally, it considers harvest mortality as a factor in the context of other human-induced risk to steelhead such as freshwater habitat loss or climate change.

The TRT applied the criteria to the 32 DIPs in the Puget Sound Steelhead DPS to provide a status assessment of the current viability of these units in Figure 57. The hierarchical BNs produced the viability values that were used to generate the scores. Because this figure categorizes the viability scores for each criterion and for each DIP as a whole into 3 bins as described above, one of its primary purposes is to provide a readily visualized summary of patterns of viability for DIPs across the DPS. Another purpose of this figure is to highlight viability criteria that lack sufficient information, a key first step in outlining a monitoring program for recovery. That assessment, summarized in Figure 57, clearly indicates low viability for more than half of the DIPs, all MPGs, and the DPS as a whole. Nearly all DIPs have insufficient current abundance and productivity scores to be considered viable. Most also have low scores for diversity and spatial structure, largely because of extensive hatchery influence, low breeding population sizes, and freshwater habitat fragmentation or loss. In the Central and South Puget Sound MPG and the Hood Canal and Strait of Juan de Fuca MPG, nearly all DIPs are not viable. The mean score for the 16 DIPs in the Northern Cascades MPG is 11.9, just above the threshold for intermediate viability; however, no summer-run populations are viable using the assessment, so the MPG is not considered to be viable. The mean score for the 8 DIPs in the Central and South Puget Sound MPG is 9.5, and that for the 8 DIPs in the Hood Canal and Strait of Juan de Fuca MPG is 9.9; both scores are below the threshold for intermediate viability.

Under an alternative assessment in which all viability criteria for which data are insufficient are given an intermediate score (2), the outcome of viability changes surprisingly little, even at the level of individual DIPs (Figure 58). Most DIPs still have insufficient current

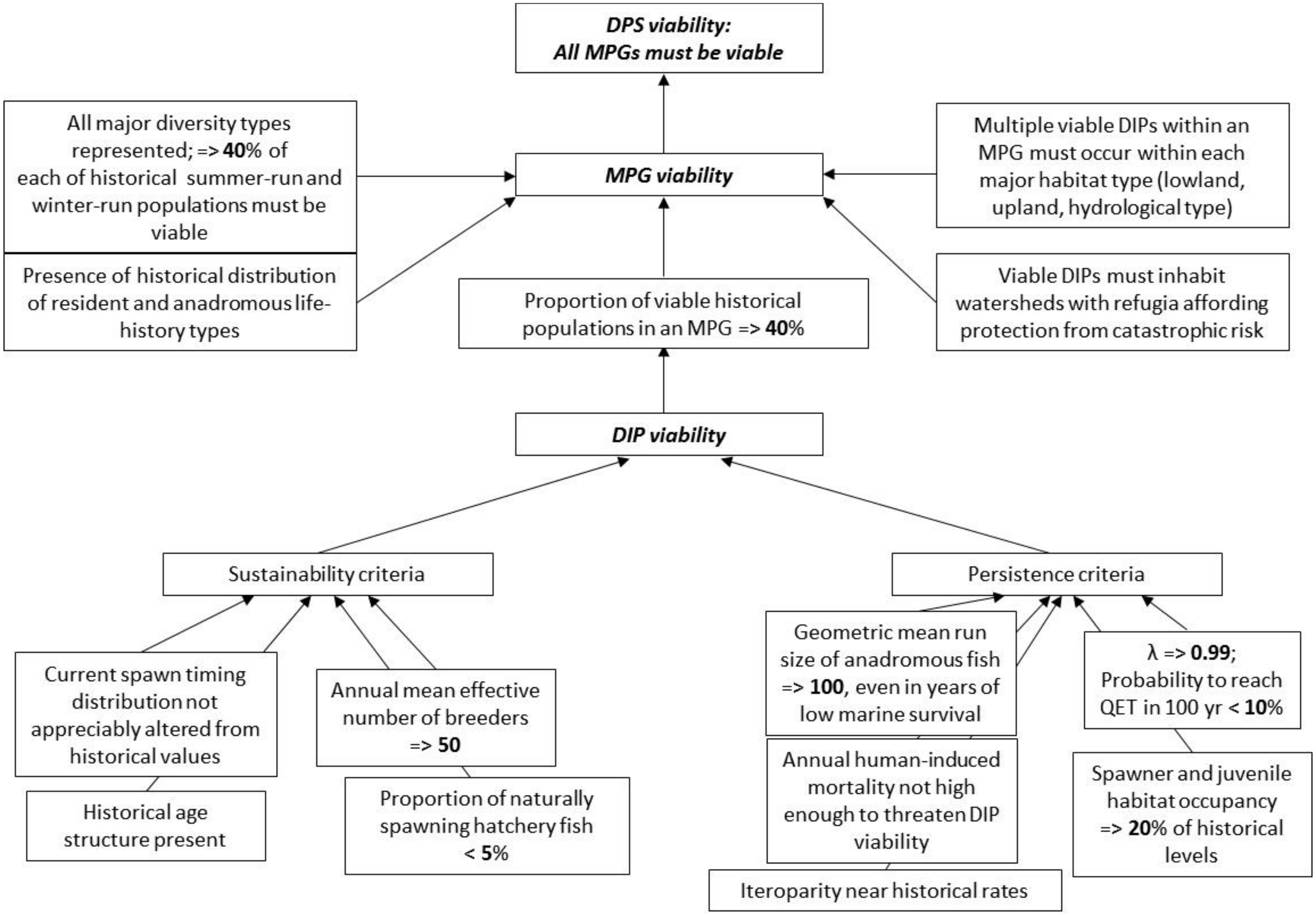
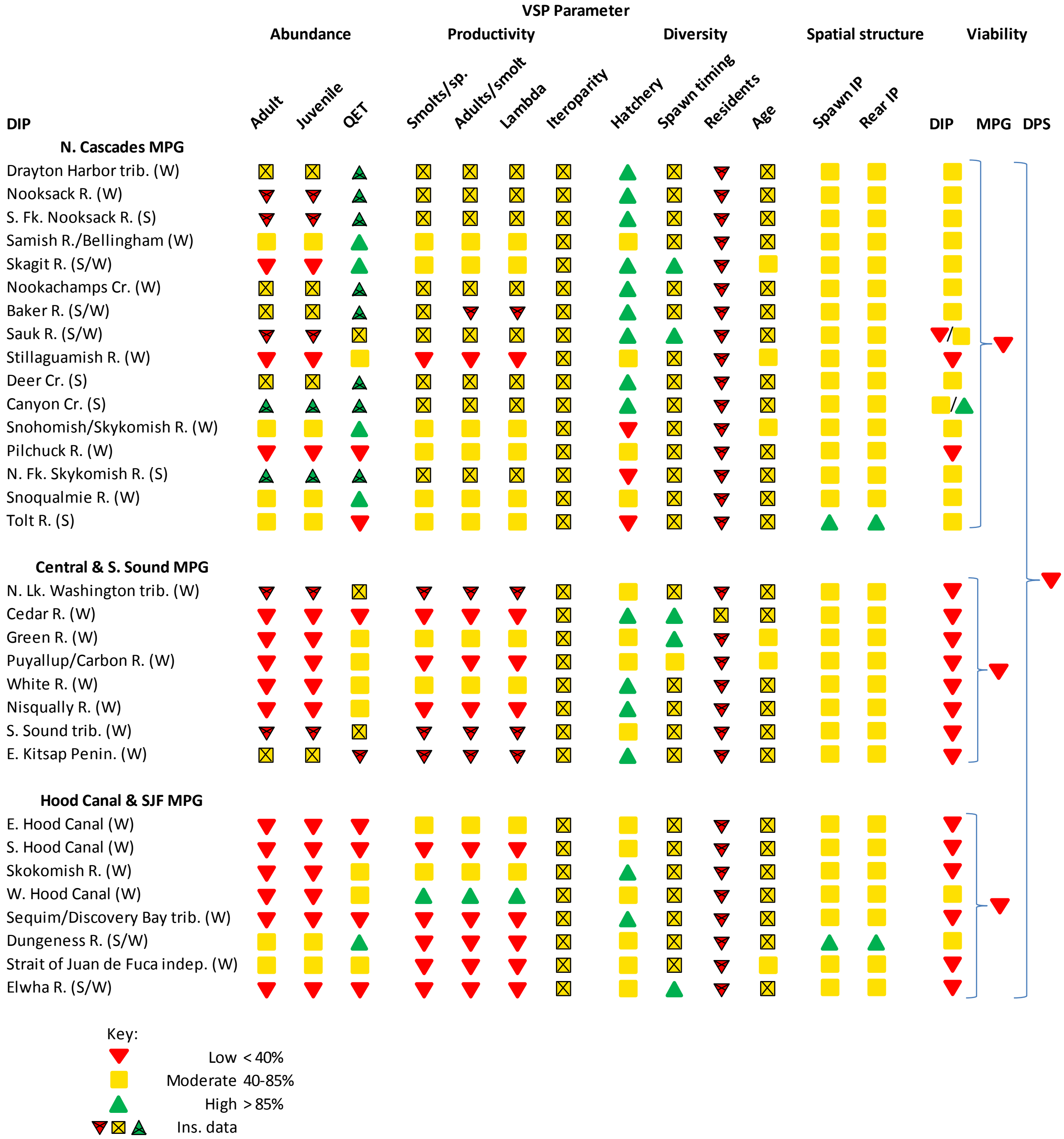


Figure 56. A set of recommended viability criteria for a DPS of steelhead. The chart shows how viability at the scale of the DIP, MPG, and the entire DPS are related. For DIP viability, criteria are divided into those contributing to persistence (i.e., abundance and productivity, extinction risk, and effects of human-induced mortality) and those contributing to sustainability (i.e., diversity and spatial structure, influence of hatchery fish, age structure, and spawn timing). Human-induced mortality includes mortality from fishing, pollution, and habitat loss.

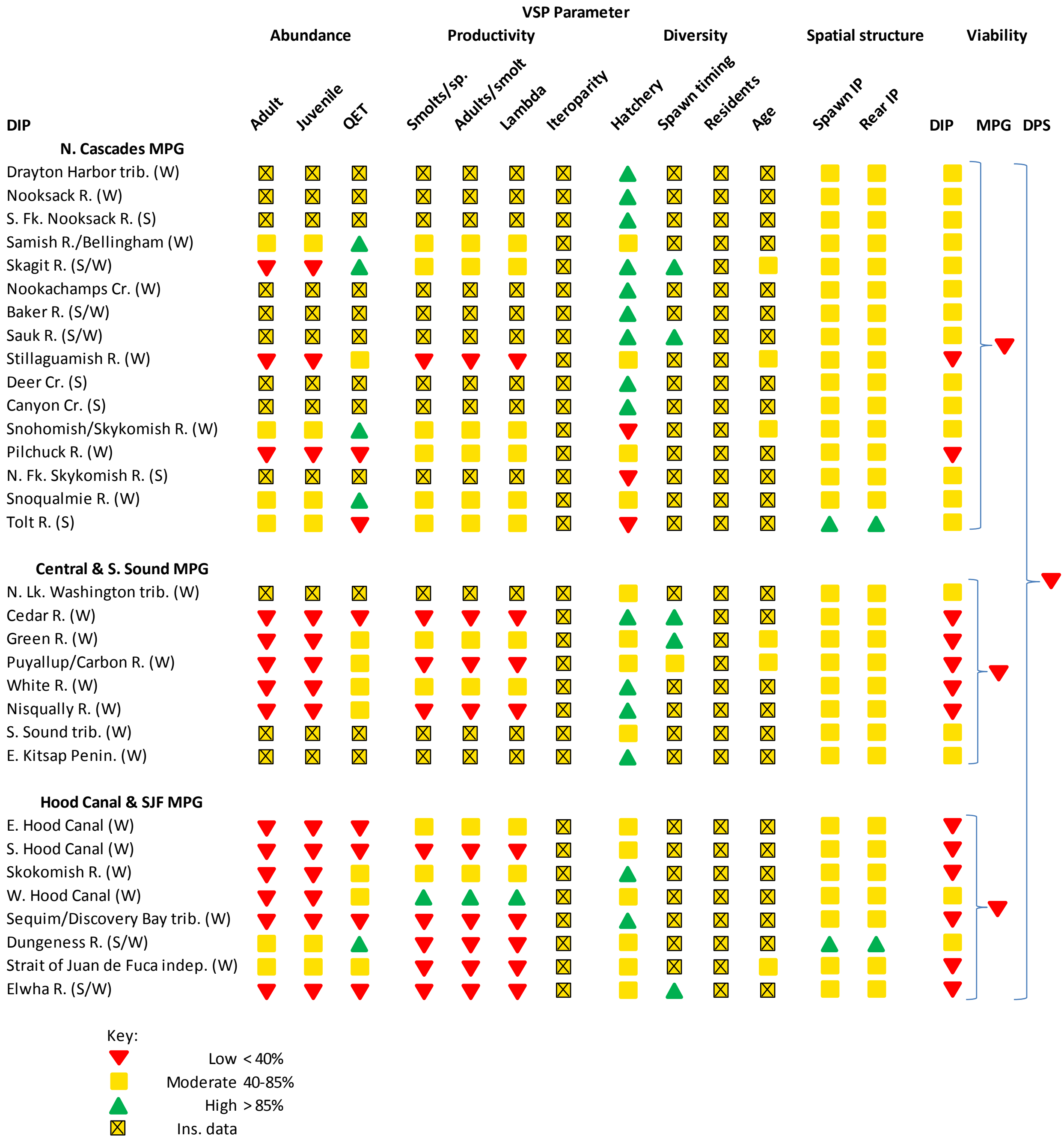


	VSP Parameter															
	Abundance			Productivity				Diversity				Spatial structure		Viability		
DIP	Adult	Juvenile	QET	Smolts/sp.	Adults/smolt	Lambda	Iteroparity	Hatchery	Spawn timing	Residents	Age	Spawn IP	Rear IP	DIP	MPG	DPS
N. Cascades MPG																
Drayton Harbor trib. (W)	☒	☒	△	☒	☒	☒	☒	▲	☒	▽	☒	■	■	■		
Nooksack R. (W)	▽	▽	△	☒	☒	☒	☒	▲	☒	▽	☒	■	■	■		
S. Fk. Nooksack R. (S)	▽	▽	△	☒	☒	☒	☒	▲	☒	▽	☒	■	■	■		
Samish R./Bellingham (W)	■	■	▲	■	■	■	☒	■	☒	▽	☒	■	■	■		
Skagit R. (S/W)	▼	▼	▲	■	■	■	☒	▲	▲	▽	■	■	■	■		
Nookachamps Cr. (W)	☒	☒	△	☒	☒	☒	☒	▲	☒	▽	☒	■	■	■		
Baker R. (S/W)	☒	☒	△	☒	▽	▽	☒	▲	☒	▽	☒	■	■	■		
Sauk R. (S/W)	▽	▽	☒	☒	☒	☒	☒	▲	▲	▽	☒	■	■	▼/■	▼	
Stillaguamish R. (W)	▼	▼	■	▼	▼	▼	☒	■	☒	▽	■	■	■	▼		
Deer Cr. (S)	☒	☒	△	☒	☒	☒	☒	▲	☒	▽	☒	■	■	■		
Canyon Cr. (S)	△	△	△	☒	☒	☒	☒	▲	☒	▽	☒	■	■	■/▲		
Snohomish/Skykomish R. (W)	■	■	▲	■	■	■	☒	▼	☒	▽	■	■	■	■		
Pilchuck R. (W)	▼	▼	▼	■	■	■	☒	■	☒	▽	☒	■	■	▼		
N. Fk. Skykomish R. (S)	△	△	△	☒	☒	☒	☒	▼	☒	▽	☒	■	■	■		
Snoqualmie R. (W)	■	■	▲	■	■	■	☒	■	☒	▽	☒	■	■	■		
Tolt R. (S)	■	■	▼	■	■	■	☒	▼	☒	▽	☒	▲	▲	■		
Central & S. Sound MPG																
N. Lk. Washington trib. (W)	▽	▽	☒	▽	▽	▽	☒	■	☒	▽	☒	■	■	▼		▼
Cedar R. (W)	▼	▼	▼	▼	▼	▼	☒	▲	▲	☒	☒	■	■	▼		
Green R. (W)	▼	▼	■	■	■	■	☒	■	▲	▽	■	■	■	▼		
Puyallup/Carbon R. (W)	▼	▼	■	▼	▼	▼	☒	■	■	▽	■	■	■	▼	▼	
White R. (W)	▼	▼	■	■	■	■	☒	▲	☒	▽	☒	■	■	▼		
Nisqually R. (W)	▼	▼	■	▼	▼	▼	☒	▲	☒	▽	☒	■	■	▼		
S. Sound trib. (W)	▽	▽	☒	▽	▽	▽	☒	■	☒	▽	☒	■	■	▼		
E. Kitsap Penin. (W)	☒	☒	▽	▽	▽	▽	☒	▲	☒	▽	☒	■	■	▼		
Hood Canal & SJF MPG																
E. Hood Canal (W)	▼	▼	▼	■	■	■	☒	■	☒	▽	☒	■	■	▼		
S. Hood Canal (W)	▼	▼	▼	▼	▼	▼	☒	■	☒	▽	☒	■	■	▼		
Skokomish R. (W)	▼	▼	■	■	■	■	☒	▲	☒	▽	☒	■	■	▼		
W. Hood Canal (W)	▼	▼	■	▲	▲	▲	☒	■	☒	▽	☒	■	■	■		
Sequim/Discovery Bay trib. (W)	▼	▼	▼	▼	▼	▼	☒	▲	☒	▽	☒	■	■	▼	▼	
Dungeness R. (S/W)	■	■	▲	▼	▼	▼	☒	■	☒	▽	☒	▲	▲	■		
Strait of Juan de Fuca indep. (W)	■	■	■	▼	▼	▼	☒	■	☒	▽	■	■	■	▼		
Elwha R. (S/W)	▼	▼	▼	▼	▼	▼	☒	■	▲	▽	☒	■	■	▼		

Key:

▼	Low < 40%
■	Moderate 40-85%
▲	High > 85%
▽ ☒ △	Ins. data

Figure 57. Estimates of current viability (low = not viable, moderate = intermediate, high = viable) for the 32 DIPs of Puget Sound steelhead (S = summer run and W = winter run) using the VSP framework. Note that many criteria are supported by insufficient data and in most (but not all) of those cases they were given an intermediate value with respect to influence on viability.



VSP Parameter

DIP	Abundance			Productivity				Diversity				Spatial structure		Viability		
	Adult	Juvenile	QET	Smolts/sp.	Adults/smolt	Lambda	Iteroparity	Hatchery	Spawn timing	Residents	Age	Spawn IP	Rear IP	DIP	MPG	DPS
N. Cascades MPG																
Drayton Harbor trib. (W)	☒	☒	☒	☒	☒	☒	☒	▲	☒	☒	☒	🟨	🟨	🟨	▼	▼
Nooksack R. (W)	☒	☒	☒	☒	☒	☒	☒	▲	☒	☒	☒	🟨	🟨	🟨		
S. Fk. Nooksack R. (S)	☒	☒	☒	☒	☒	☒	☒	▲	☒	☒	☒	🟨	🟨	🟨		
Samish R./Bellingham (W)	🟨	🟨	▲	🟨	🟨	🟨	☒	🟨	☒	☒	☒	🟨	🟨	🟨		
Skagit R. (S/W)	▼	▼	▲	🟨	🟨	🟨	☒	▲	▲	☒	🟨	🟨	🟨	🟨		
Nookachamps Cr. (W)	☒	☒	☒	☒	☒	☒	☒	▲	☒	☒	☒	🟨	🟨	🟨		
Baker R. (S/W)	☒	☒	☒	☒	☒	☒	☒	▲	☒	☒	☒	🟨	🟨	🟨		
Sauk R. (S/W)	☒	☒	☒	☒	☒	☒	☒	▲	▲	☒	☒	🟨	🟨	🟨		
Stillaguamish R. (W)	▼	▼	🟨	▼	▼	▼	☒	🟨	☒	☒	🟨	🟨	🟨	▼		
Deer Cr. (S)	☒	☒	☒	☒	☒	☒	☒	▲	☒	☒	☒	🟨	🟨	🟨		
Canyon Cr. (S)	☒	☒	☒	☒	☒	☒	☒	▲	☒	☒	☒	🟨	🟨	🟨		
Snohomish/Skykomish R. (W)	🟨	🟨	▲	🟨	🟨	🟨	☒	▼	☒	☒	🟨	🟨	🟨	🟨		
Pilchuck R. (W)	▼	▼	▼	🟨	🟨	🟨	☒	🟨	☒	☒	☒	🟨	🟨	▼		
N. Fk. Skykomish R. (S)	☒	☒	☒	☒	☒	☒	☒	▼	☒	☒	☒	🟨	🟨	🟨		
Snoqualmie R. (W)	🟨	🟨	▲	🟨	🟨	🟨	☒	🟨	☒	☒	☒	🟨	🟨	🟨		
Tolt R. (S)	🟨	🟨	▼	🟨	🟨	🟨	☒	▼	☒	☒	☒	▲	▲	🟨		
Central & S. Sound MPG																
N. Lk. Washington trib. (W)	☒	☒	☒	☒	☒	☒	☒	🟨	☒	☒	☒	🟨	🟨	🟨	▼	
Cedar R. (W)	▼	▼	▼	▼	▼	▼	☒	▲	▲	☒	☒	🟨	🟨	▼		
Green R. (W)	▼	▼	🟨	🟨	🟨	🟨	☒	🟨	▲	☒	🟨	🟨	🟨	▼		
Puyallup/Carbon R. (W)	▼	▼	🟨	▼	▼	▼	☒	🟨	🟨	☒	🟨	🟨	🟨	▼		
White R. (W)	▼	▼	🟨	🟨	🟨	🟨	☒	▲	☒	☒	☒	🟨	🟨	▼		
Nisqually R. (W)	▼	▼	🟨	▼	▼	▼	☒	▲	☒	☒	☒	🟨	🟨	▼		
S. Sound trib. (W)	☒	☒	☒	☒	☒	☒	☒	🟨	☒	☒	☒	🟨	🟨	🟨		
E. Kitsap Penin. (W)	☒	☒	☒	☒	☒	☒	☒	▲	☒	☒	☒	🟨	🟨	🟨		
Hood Canal & SJF MPG																
E. Hood Canal (W)	▼	▼	▼	🟨	🟨	🟨	☒	🟨	☒	☒	☒	🟨	🟨	▼	▼	
S. Hood Canal (W)	▼	▼	▼	▼	▼	▼	☒	🟨	☒	☒	☒	🟨	🟨	▼		
Skokomish R. (W)	▼	▼	🟨	🟨	🟨	🟨	☒	▲	☒	☒	☒	🟨	🟨	▼		
W. Hood Canal (W)	▼	▼	🟨	▲	▲	▲	☒	🟨	☒	☒	☒	🟨	🟨	🟨		
Sequim/Discovery Bay trib. (W)	▼	▼	▼	▼	▼	▼	☒	▲	☒	☒	☒	🟨	🟨	▼		
Dungeness R. (S/W)	🟨	🟨	▲	▼	▼	▼	☒	🟨	☒	☒	☒	▲	▲	🟨		
Strait of Juan de Fuca indep. (W)	🟨	🟨	🟨	▼	▼	▼	☒	🟨	☒	☒	🟨	🟨	🟨	▼		
Elwha R. (S/W)	▼	▼	▼	▼	▼	▼	☒	🟨	▲	☒	☒	🟨	🟨	▼		

Key:

▼	Low < 40%
🟨	Moderate 40-85%
▲	High > 85%
☒	Ins. data

Figure 58. Estimates of current viability (low = not viable, moderate = intermediate, high = viable) for the 32 DIPs of Puget Sound steelhead (S = summer run and W = winter run) using the VSP framework, but assuming that all viability criteria without quantifiable information are given a score of 2 (intermediate).

abundance and productivity scores to be considered viable, especially in the Central and South Puget Sound MPG and the Hood Canal and Strait of Juan de Fuca MPG. Most have intermediate scores for diversity and spatial structure. In the Central and South Puget Sound MPG and the Hood Canal and Strait of Juan de Fuca MPG, most (11 of 16) DIPs are not viable. The mean score for the 16 DIPs in the Northern Cascades MPG is 11.9, just above the threshold for intermediate viability; however, no summer-run populations are viable using the assessment, so the MPG is still not considered to be viable. The mean score for the 8 DIPs in the Central and South Puget Sound MPG is 10.7, and that for the 8 DIPs in the Hood Canal and Strait of Juan de Fuca MPG is 10.1; both scores are still below the threshold for intermediate viability. Twelve of the 32 DIPs lack sufficient information on the abundance and productivity criteria to be certain about their influences on viability, and all DIPs throughout the DPS have insufficient information on each of the diversity and spatial structure criteria.

Conclusions

Based on a combination of quantitative PVAs, collective judgment of qualitative information, and use of habitat-based metrics to estimate IP, the PSS TRT has developed and applied a framework to evaluate the current biological status of the Puget Sound Steelhead DPS. This framework is based on a series of hierarchical BNs intended to integrate the various sources of information that influence viability at the levels of the DIP, the biogeographic stratum or MPG, and the entire listed DPS. This information is biological (e.g., juvenile and adult abundance, population growth rate, freshwater and marine survival, iteroparity, spawn timing), related to habitat characteristics (e.g., habitat use and selection with respect to hydrograph, reach size and gradient, elevation), and related to management (e.g., hatchery influence, harvest effects). The TRT applied the framework to the Puget Sound Steelhead DPS to evaluate its status with the intent to assist in guiding recovery actions. Responses to reviewer comments on an earlier draft of this technical memorandum are in Appendix H.

The framework and the analyses it supports do not set recovery targets at any of the three levels, nor do they explicitly identify specific populations or groups of populations for recovery priority. Rather, the framework and associated analyses are meant to provide a technical foundation for those charged with recovery of listed steelhead in Puget Sound, from which they can develop effective recovery plans at the watershed scale (and higher) that are based on biologically meaningful criteria. They are also meant to highlight the data that are needed in future monitoring and evaluation studies to improve these efforts. It should be evident from this report that identifying the best way forward in recovery planning for steelhead would be enhanced considerably by additional information on the demography, diversity, and distribution of steelhead in watersheds throughout Puget Sound.

Consistent with its status as a threatened species under the ESA, the Puget Sound Steelhead DPS is not considered to be viable by the TRT. Using a comprehensive set of BNs that incorporate factors influencing all four VSP criteria (abundance, productivity, diversity, and spatial structure), nearly all of its 32 constituent DIPs and all 3 of its constituent MPGs are at low viability. Nearly all DIPs in the Central and South Puget Sound MPG and the Hood Canal and Strait of Juan de Fuca MPG are not viable. Most of the DIPs score low for all four VSP criteria and nearly all DIPs have insufficient current abundance and productivity scores to be considered viable. Most DIPs also have low scores for diversity and intermediate scores for spatial structure. The pattern of low viability is widespread throughout Puget Sound, across all three MPGs, and includes both summer-run and winter-run populations. The populations with highest viability—and those with highest abundance and diversity—are in northeastern Puget Sound (Northern Cascades MPG).

An analysis incorporating a simpler set of BNs that rely only on abundance and productivity criteria produced estimates of steelhead viability that are considerably higher throughout the DPS, but even under this analysis there is substantial uncertainty in the status of at least two of the MPGs and of the entire DPS. Despite the lack of quantitative information for several aspects of diversity and spatial structure, the TRT considers these VSP criteria to be essential for DPS viability into the future and recommends the use of the more comprehensive networks that encompass all four VSP criteria to evaluate the viability of Puget Sound steelhead.

The TRT's BN framework provides a means of evaluating steelhead viability with explicit reference to estimates of historical abundance, productivity, diversity, and spatial distribution of steelhead in this region. This framework provides a systematic and transparent mechanism to evaluate viability of the DPS and its component MPGs and DIPs, and lends itself to evaluating alternative recovery scenarios and the effects of specific or alternative recovery actions, especially those operating at the watershed scale. The TRT's companionate technical memorandum (Myers et al. 2015) describes the identification of the DIPs in this DPS that inform the framework for developing these viability criteria.

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Appendix A: State-Space Analyses of Viability of Puget Sound Steelhead DIPs

This appendix contains Figure A-1 through Figure A-22. Several of the figures have only the first two panels described below. For each demographically independent population (DIP), the results of the multivariate autoregressive state-space (MARSS) population viability analyses (PVAs) are summarized in up to six subplots in each figure. In each group of plots, the top left panel plots the observed counts and the fitted state-space estimate against year, giving the MARSS maximum-likelihood estimate of fit to the abundance data (red curve), the estimated long-term population growth rate (μ_{est}, equivalent to $\ln(\lambda)$), and the estimate of process error (Q_{est}). The top right panel plots the probability that the population will reach a particular quasi-extinction threshold (QET) abundance within the next 100 years (with approximate 95% CIs). The QETs for each population applied here (Table 7) are based on an estimate derived from the intrinsic potential metrics described in the main text (the Use of Bayesian Networks to Incorporate Viable Salmonid Populations Criteria into a Viability Assessment Framework subsection of the Decision Support Systems as Tools for Assessing Viability section).

The middle left panel plots the probability density function (PDF) of the time in years to reach QET, given that it is reached within 100 years, and the middle right panel depicts the probability of reaching QET in 100 years, given as a function of the number of individuals at the end of the projection. The bottom left panel plots several of the sample population projections estimated by MARSS.

Finally, the bottom right panel depicts the regions of high certainty and uncertainty surrounding the population projections (an extinction risk "envelope," see Ellner and Holmes 2008). The green region is where the upper 95% CIs of the projections do not exceed $P = 0.05$ (i.e., where the probability of the specified population decline is <5%). The red region is where the lower 95% CIs of the projections exceed $P = 0.95$ (i.e., where the probability of the specified population decline is >95%). The grey regions define less certain areas of parameter space between these extremes, with the dark grey region representing the region of highest uncertainty. Note that not all plots and corresponding estimates could be constructed for each population. For example, we were not able to calculate PVA estimates for putative winter-run steelhead (*Oncorhynchus mykiss*) DIPs in the Nooksack River or in South Puget Sound tributaries, nor were we able to do so for any summer-run steelhead populations in the Puget Sound DPS, except for that in the Tolt River.

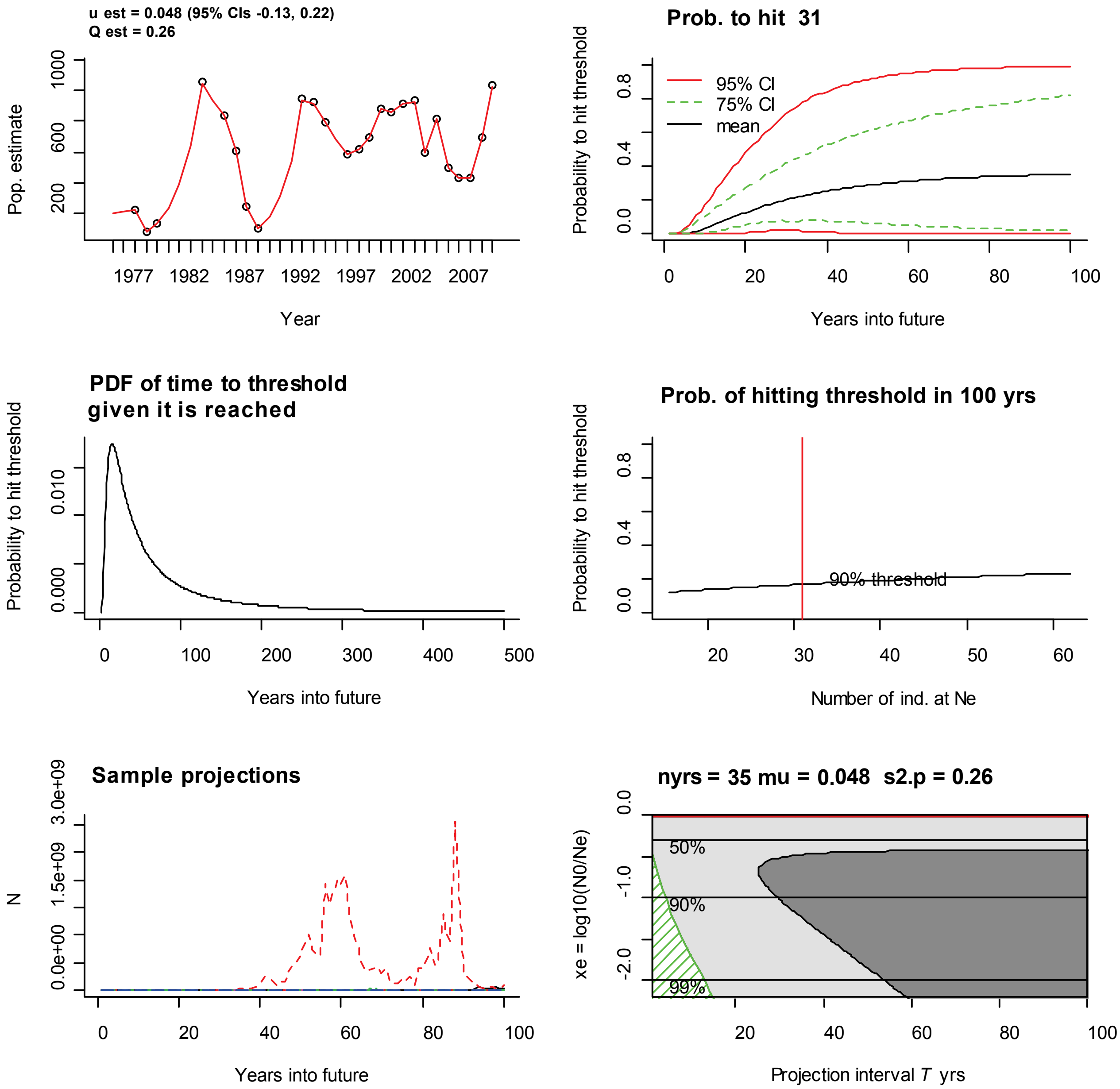

Figure A-1. MARSS-based (Holmes and Ward 2011) PVA for the Samish River winter-run population. Steelhead counts in the Samish River have varied considerably with a decline over the past decade, but have generally declined. The estimated mean population growth rate (μ_{est}) is 0.048 (λ = 1.049) and process error (Q_{est}) is 0.260, with no significant evidence for population trend. Assuming that these counts are a reasonable reflection of spawner abundance, the estimated probability that this population would decline to a QET of 31 fish is relatively low—about 30% within 100 years. We can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 5–8 years and that a 99% decline will not occur within the next 15 years. However, beyond the very near term (i.e., after about 20 years), we are uncertain about the precise level of extinction risk.

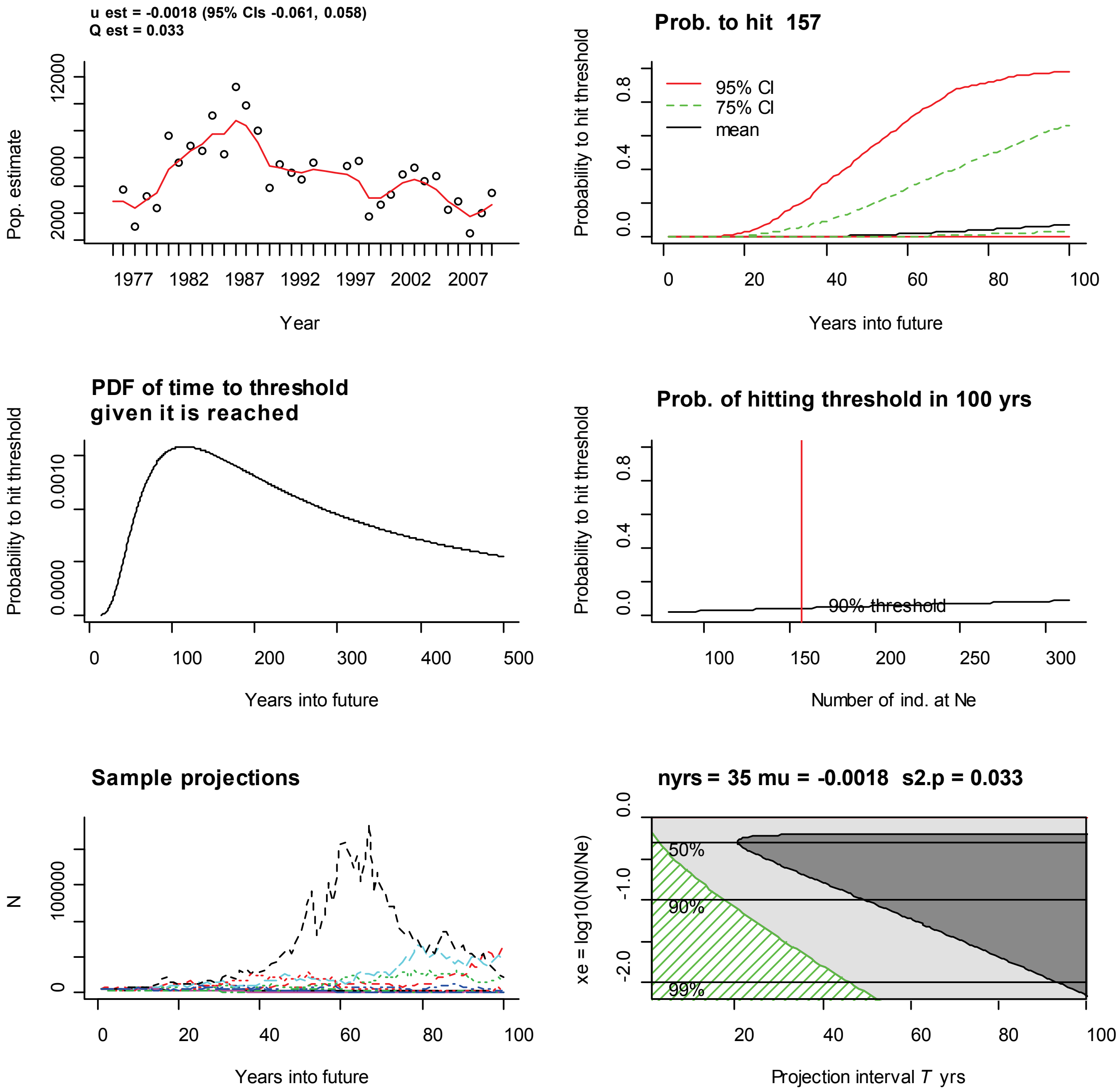

Figure A-2. MARSS-based (Holmes and Ward 2011) PVA for the Skagit River Summer Run and Winter Run. Steelhead counts in the Skagit River have been highly variable, but have declined since the early 1980s. The estimated mean population growth rate is −0.002 ($\lambda = 0.998$) and process error is 0.033, with no significant evidence for population trend. The estimated probability that this population would decline to a QET of 157 fish is very low—less than 10% within 100 years. We can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 20 years and that a 99% decline will not occur within the next 45 years. However, beyond the near term (after just a few decades), we are uncertain about the precise level of extinction risk.

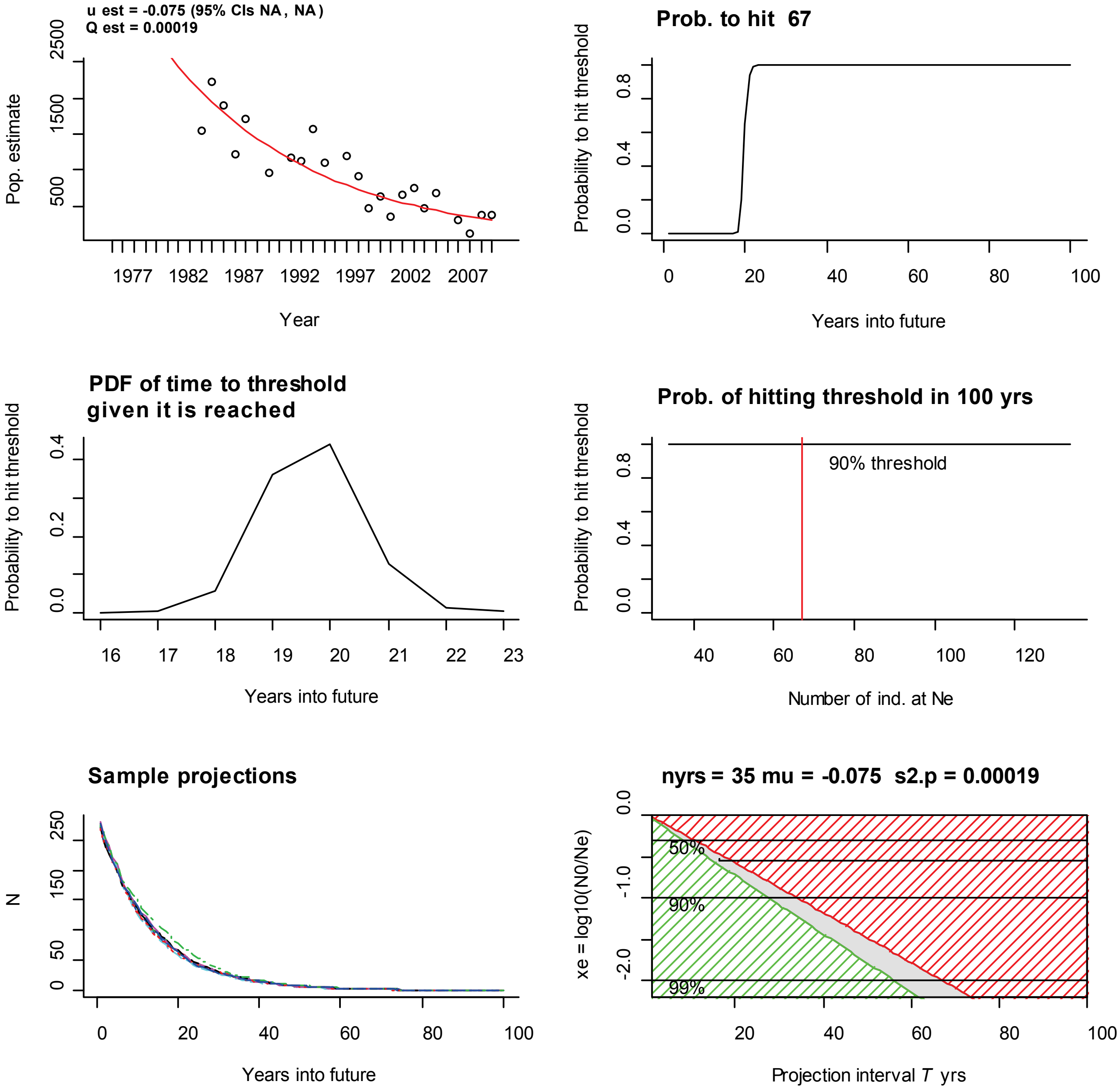

Figure A-3. MARSS-based (Holmes and Ward 2011) PVA for the Stillaguamish River Winter Run. Steelhead counts in the Stillaguamish River (represented by north fork counts only) have declined steadily since the early 1980s. The estimated probability that this population would decline to a QET of 67 fish is high—about 90% within 25 years. With an estimated mean population growth rate of −0.075 ($\lambda = 0.928$) and process error of <0.001, we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 30 years and that a 99% decline will not occur within the next 55 years. However, a 50% decline is highly likely within 10 years and a 90% decline within 35 years. There is little uncertainty about a decline in this population if current conditions continue.

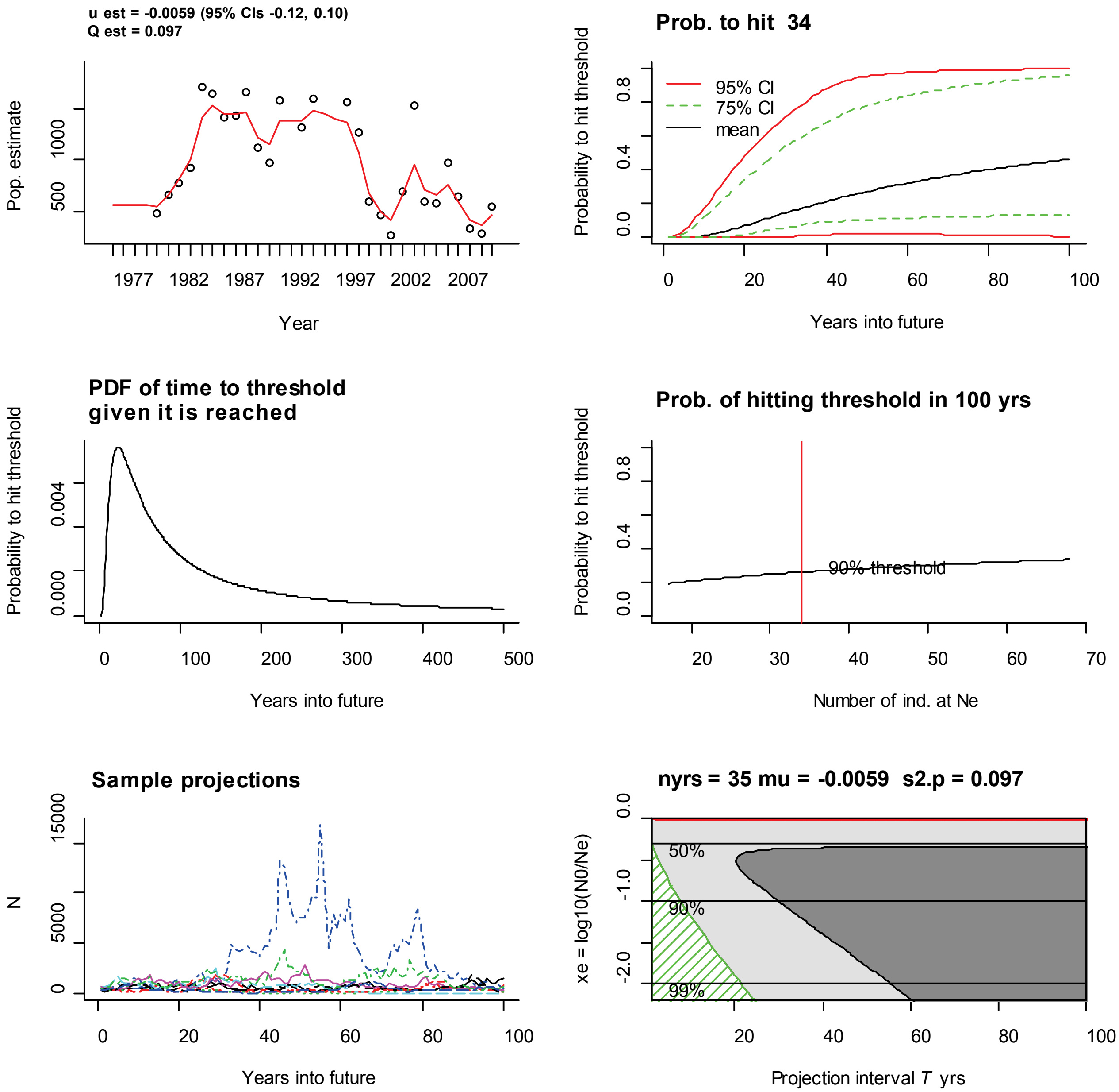

Figure A-4. MARSS-based (Holmes and Ward 2011) PVA for the Pilchuck River Winter Run. Steelhead counts in the Pilchuck River have declined detectably from levels during the late 1980s and early 1990s and been relatively low in most years since 2000. The estimated probability that this population would decline to a QET of 34 fish is relatively low—about 40% within 100 years. With an estimated mean population growth rate of −0.006 ($\lambda = 0.994$) and process error of 0.097, we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 6–8 years and that a 99% decline will not occur within the next 20 years. However, beyond the very near term (i.e., after about 20 years), we are uncertain about the precise level of extinction risk.

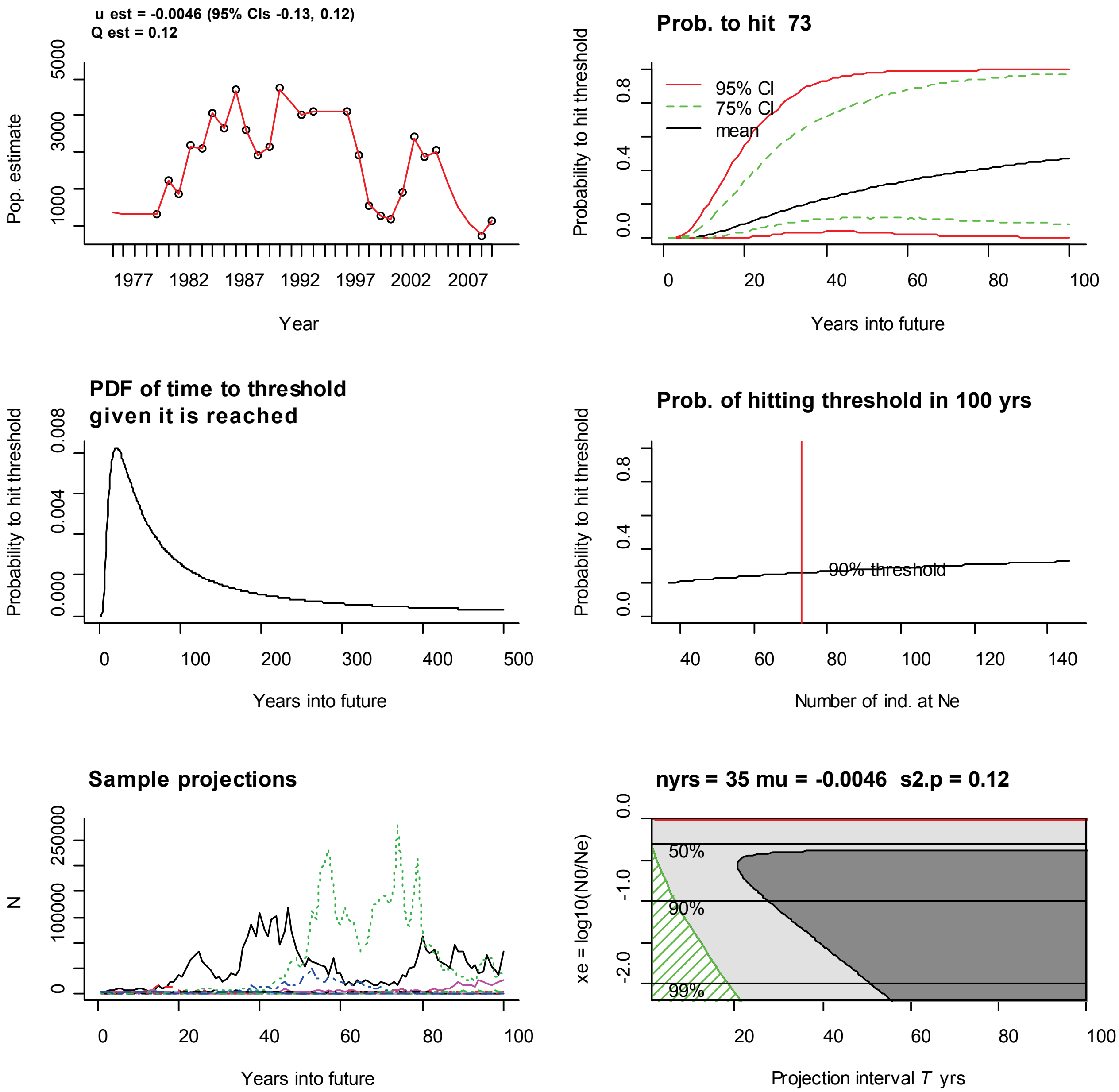

Figure A-5. MARSS-based (Holmes and Ward 2011) PVA for the Snohomish/Skykomish Rivers Winter Run. Steelhead counts in the Snohomish River have generally declined since the early 1990s, but have varied widely. The estimated probability that this population would decline to a QET of 73 fish is low—about 40% within 100 years. With an estimated mean population growth rate of −0.005 ($\lambda = 0.995$) and process error of 0.120, we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 8–10 years and that a 99% decline will not occur within the next 20 years. However, beyond the very near term (i.e., after about 20 years), we are uncertain about the precise level of extinction risk.

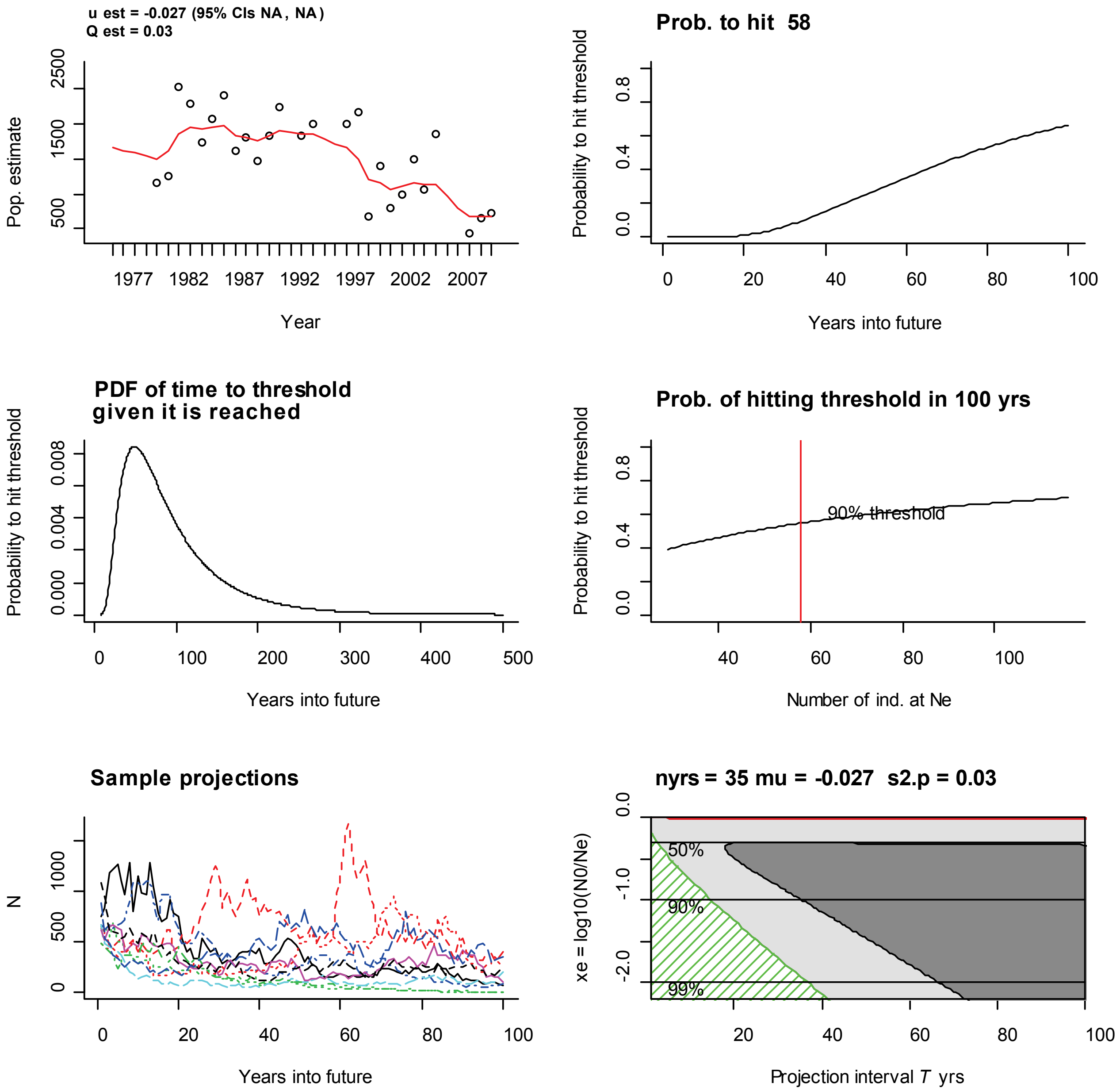

Figure A-6. MARSS-based (Holmes and Ward 2011) PVA for the Snoqualmie River Winter Run. Steelhead counts in the Snoqualmie River have declined since the early 1990s. The estimated probability that this population would decline to a QET of 58 fish is relatively high—nearly 70% within 100 years. With an estimated mean population growth rate of −0.027 ($\lambda = 0.973$) and process error of 0.030, we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 15 years and that a 99% decline will not occur within the next 35 years. However, beyond the next 30–40 years, we are uncertain about the precise level of extinction risk.

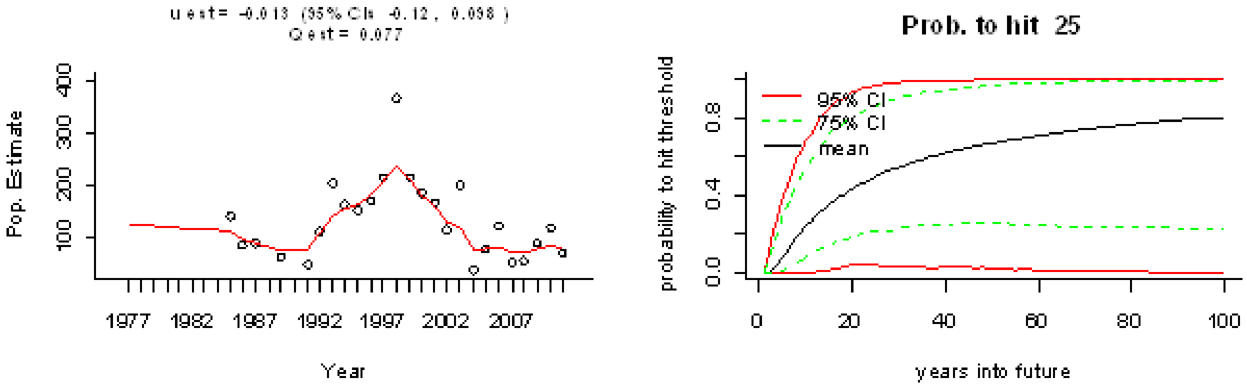

Figure A-7. MARSS-based (Holmes and Ward 2011) PVA for the Tolt River Summer Run. Steelhead counts in the Tolt River have varied since the 1980s, declining from a modest high in the late 1990s. The estimated probability that this population would decline to a QET of 25 fish is high—about 80% within 100 years. The estimated mean population growth rate is −0.013 (λ = 0.987) and the process error is 0.077. We are highly uncertain about the precise level of extinction risk for this population, but it is likely to be high because abundance is typically low yet highly variable.

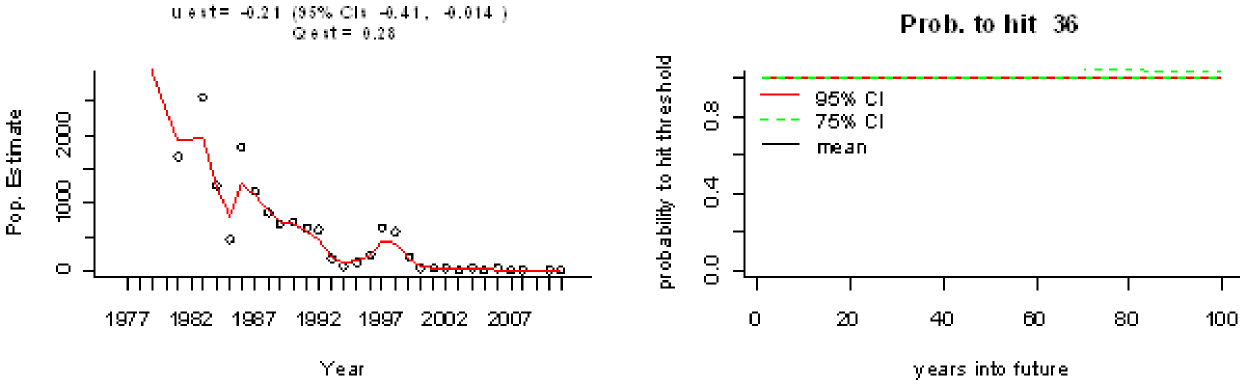

Figure A-8. MARSS-based (Holmes and Ward 2011) PVA for the Cedar River Winter Run. Steelhead counts in the Lake Washington watershed, including the Cedar River, have declined sharply since the early 1980s and have been very low since the early 1990s. The estimated probability that this population would decline to a QET of 36 fish is high—at least 90% within the next few years. The estimated mean population growth rate is −0.210 (λ = 0.811) and process error is 0.280. We are uncertain about the precise level of extinction risk for this population, but it is clearly alarmingly high due to very low abundance.

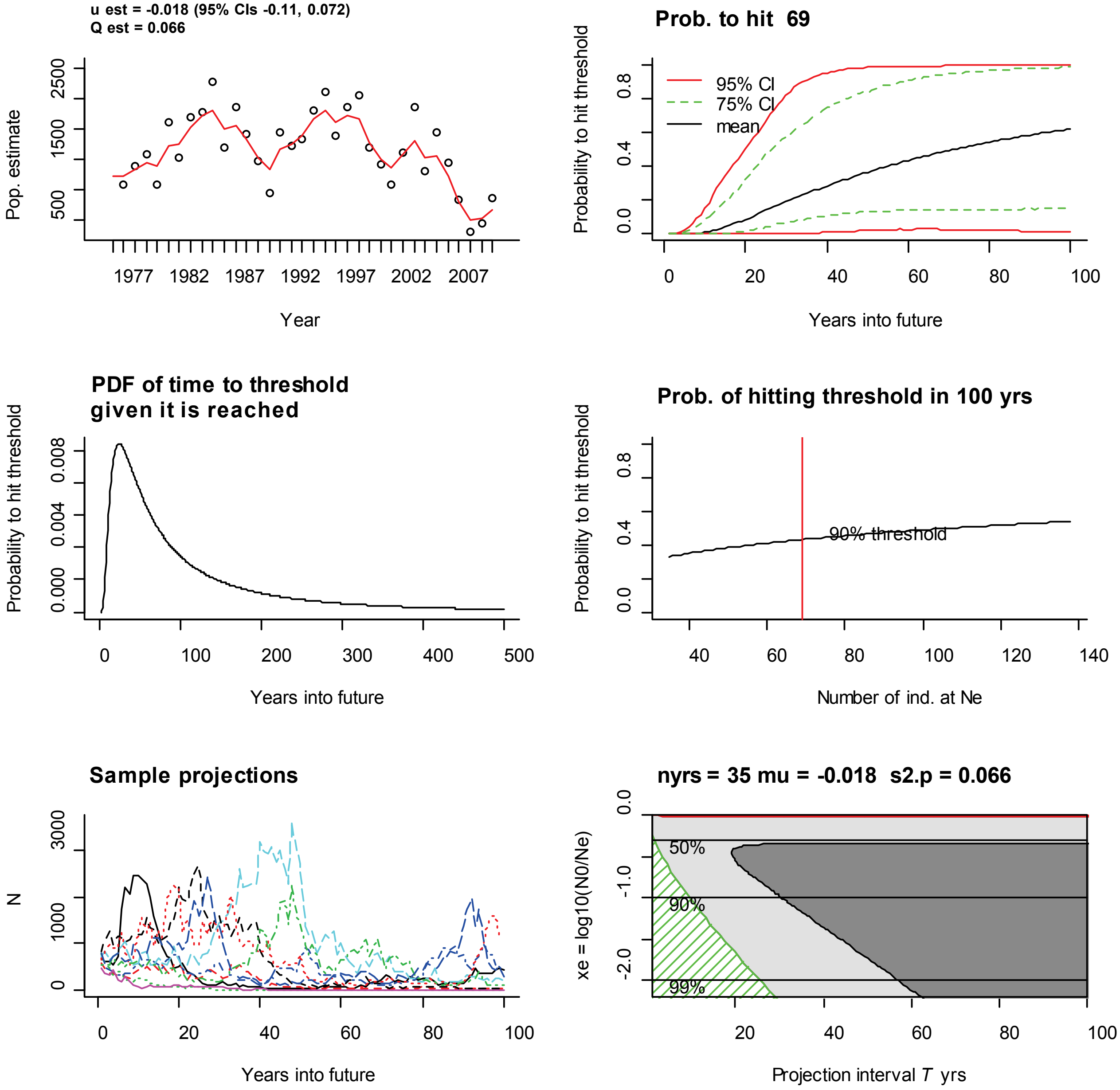

Figure A-9. MARSS-based (Holmes and Ward 2011) PVA for the Green River Winter Run. Steelhead counts in the Green River have been variable, but have exhibited a clear decline in recent years. The estimated probability that this population would decline to a QET of 69 fish is moderately high over a VSP time frame—about 50% within 100 years. With an estimated mean population growth rate of −0.018 ($\lambda = 0.982$) and process error of 0.066, we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 10 years and that a 99% decline will not occur within the next 25 years. However, beyond the near term (i.e., after about 30 years), we are uncertain about the precise level of extinction risk.

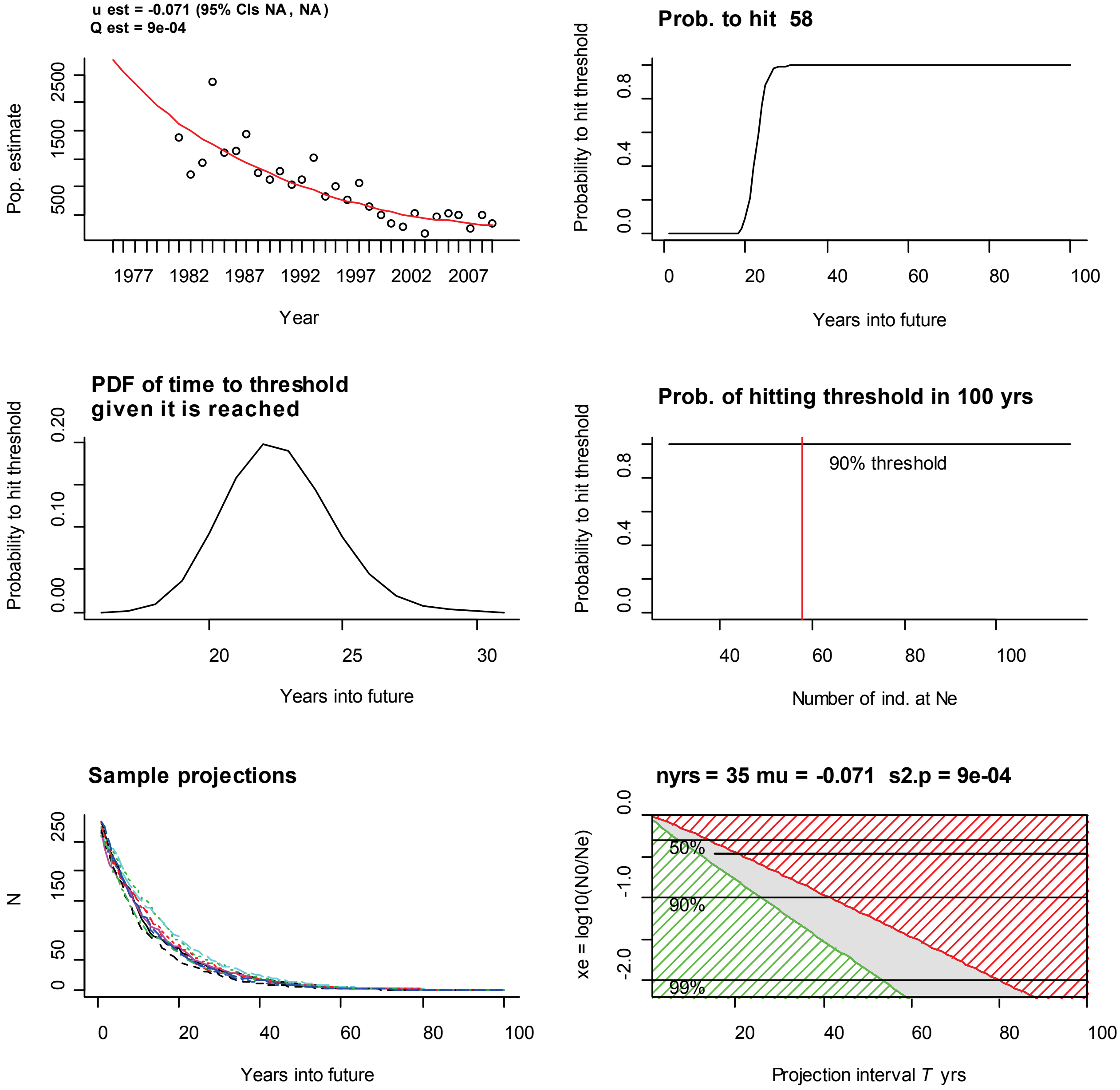

Figure A-10. MARSS-based (Holmes and Ward 2011) PVA analysis for the Puyallup River winter-run population. Steelhead counts in the Puyallup River have declined steadily since the 1980s. The estimated probability that this population would decline to a QET of 58 fish is high—about 90% within 30 years. With an estimated mean population growth rate of −0.071 ($\lambda = 0.931$) and process error of <0.001, we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 5–10 years and that a 99% decline will not occur within the next 25 years. However, a 50% decline is highly likely within 15 years and a 90% decline within 40 years. There is little uncertainty that the population will continue to decline within the foreseeable future if current conditions persist.

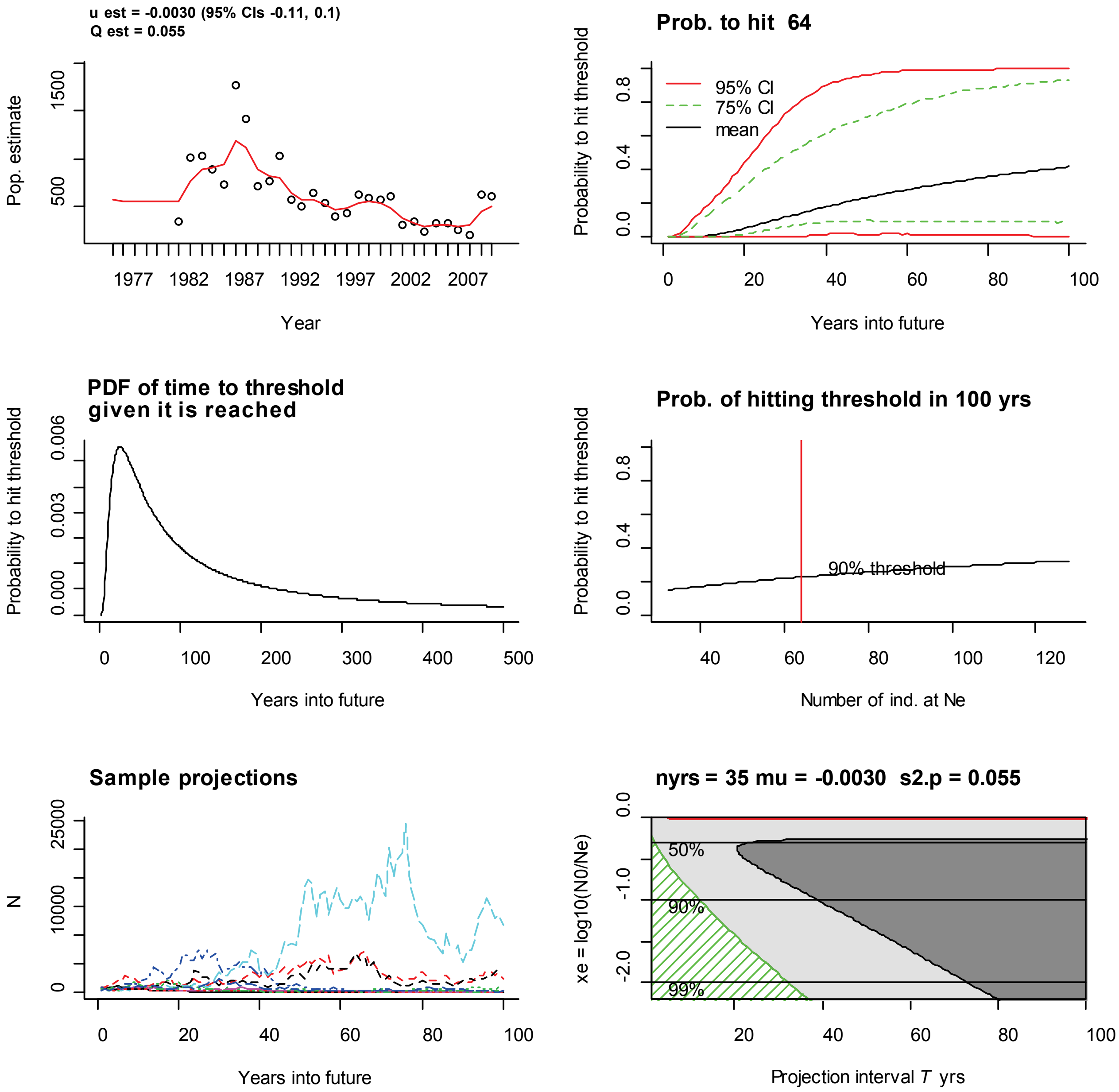

Figure A-11. MARSS-based (Holmes and Ward 2011) PVA for the White River Winter Run. Steelhead counts in the White River have declined steadily since the 1980s. The estimated probability that this population would decline to a QET of 64 fish is relatively low—about 40% within 100 years. With an estimated mean population growth rate of −0.003 ($\lambda = 0.997$) and process error of 0.055, we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 20–15 years and that a 99% decline will not occur within the next 30–35 years. However, beyond the next few decades, we are uncertain about the precise level of extinction risk.

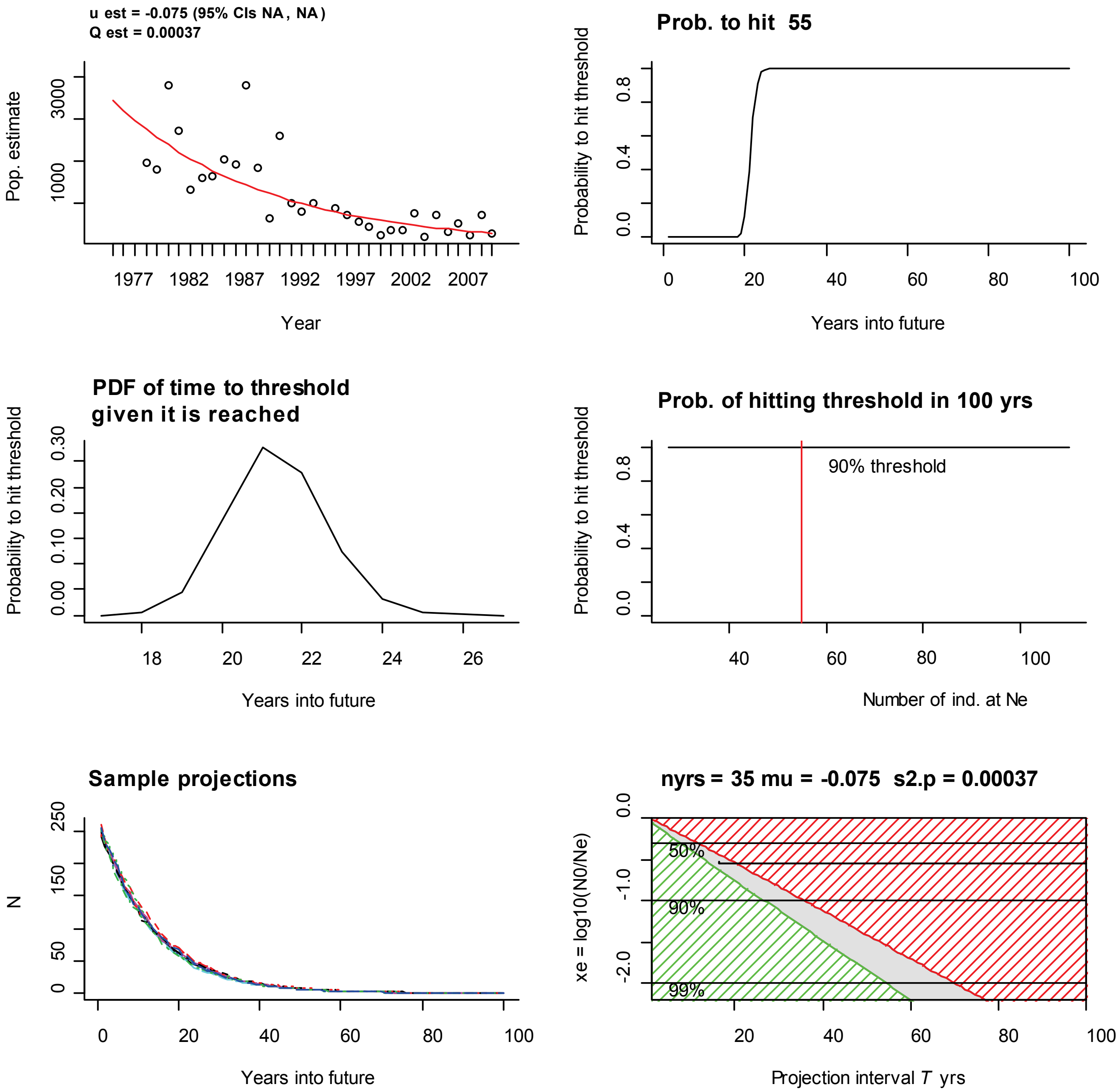

Figure A-12. MARSS-based (Holmes and Ward 2011) PVA for the Nisqually River Winter Run. Steelhead counts in the Nisqually River declined steadily since about 1990 and have remained low since then. The estimated probability that this population would decline to a QET of 55 fish is high—about 90% within 25 years. With an estimated mean population growth rate of −0.075 ($\lambda = 0.928$) and process error of <0.001, we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 25–30 years and that a 99% decline will not occur within the next 55 years. However, a 50% decline is highly likely within 10 years and a 90% decline within 40 years. There is little uncertainty that the population will continue to decline if current conditions persist.

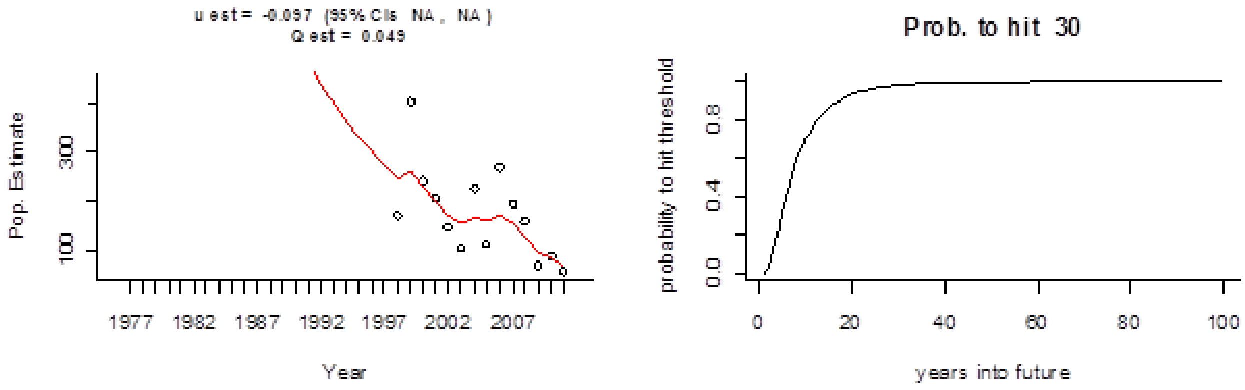

Figure A-13. MARSS-based (Holmes and Ward 2011) PVA for the South Hood Canal Tributaries Winter Run (including the Dewatto and Tahuya rivers). Steelhead counts in South Hood Canal declined steadily since the late 1990s. The estimated probability that this population would decline to a QET of 30 fish is high—about 90% within 20 years. The estimated mean population growth rate is −0.097 (λ = 0.908) and process error is 0.049. We are uncertain about the precise level of extinction risk for this population, but it is clearly high because of low and steeply declining abundance.

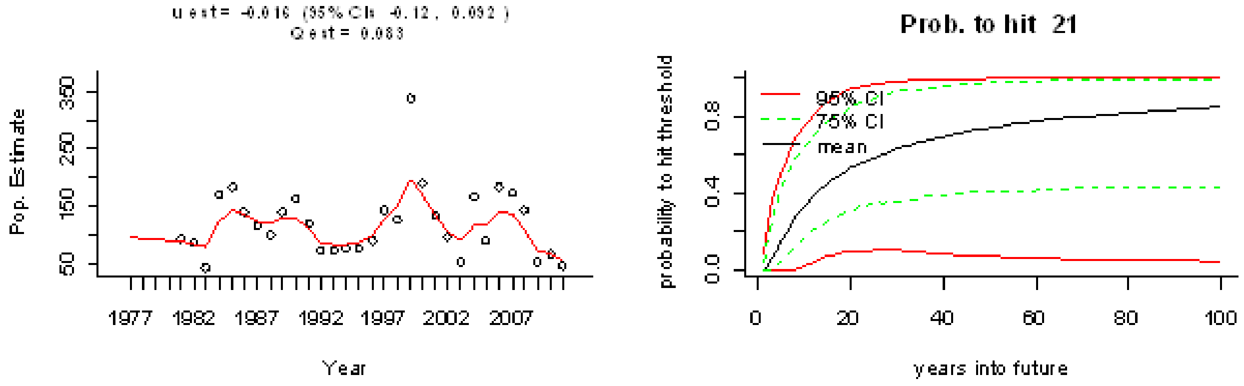

Figure A-14. MARSS-based (Holmes and Ward 2011) PVA for the Tahuya River winter-run population, which we analyzed because of longer time series available compared to Tahuya and Union rivers combined data. Steelhead counts in the Tahuya River have varied and often been quite low, but there has been no distinct trend since the 1980s. The estimated probability that this population would decline to a QET of 21 fish is relatively high—over 80% within 100 years. The estimated mean population growth rate is −0.016 (λ = 0.984) and process error is 0.083. We are uncertain about the precise level of extinction risk for this population, but it is clearly relatively high because of low abundance.

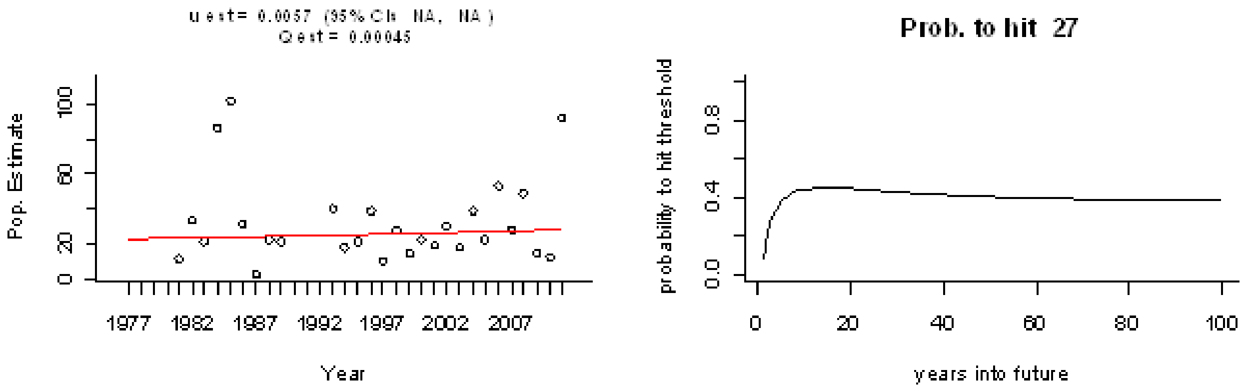

Figure A-15. MARSS-based (Holmes and Ward 2011) PVA for the East Hood Canal Tributaries Winter Run. Steelhead counts in East Hood Canal have varied and often been quite low, but there has been no distinct trend since the 1980s. The estimated probability that this population would decline to a QET of 27 fish is relatively low—about 40% within 100 years. The estimated mean population growth rate is 0.006 ($\lambda = 1.006$) and process error is <0.001. We are uncertain about the precise level of extinction risk for this population, but it is clearly neither safely low (because of the stable trend) nor very high (because of the very low abundance).

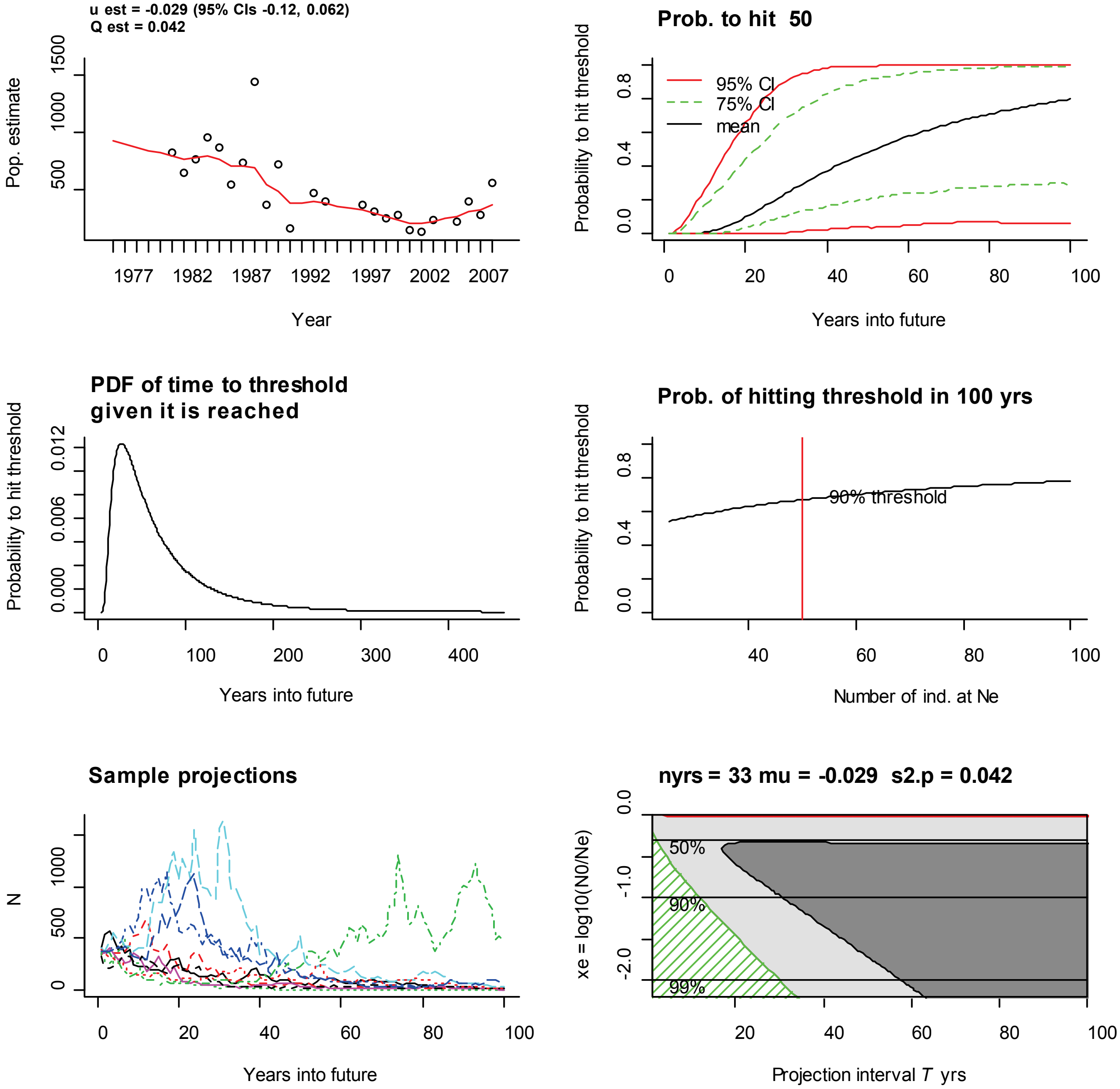

Figure A-16. MARSS-based (Holmes and Ward 2011) PVA for the Skokomish River Winter Run. Steelhead counts in the Skokomish River have declined since the 1980s. The estimated probability that this population would decline to a QET of 50 fish is relatively high—over 70% within 100 years. With an estimated mean population growth rate of −0.029 ($\lambda = 0.971$) and process error of 0.042, we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 10 years and that a 99% decline will not occur within the next 30 years. However, beyond the next two or three decades, we are uncertain about the precise level of extinction risk.

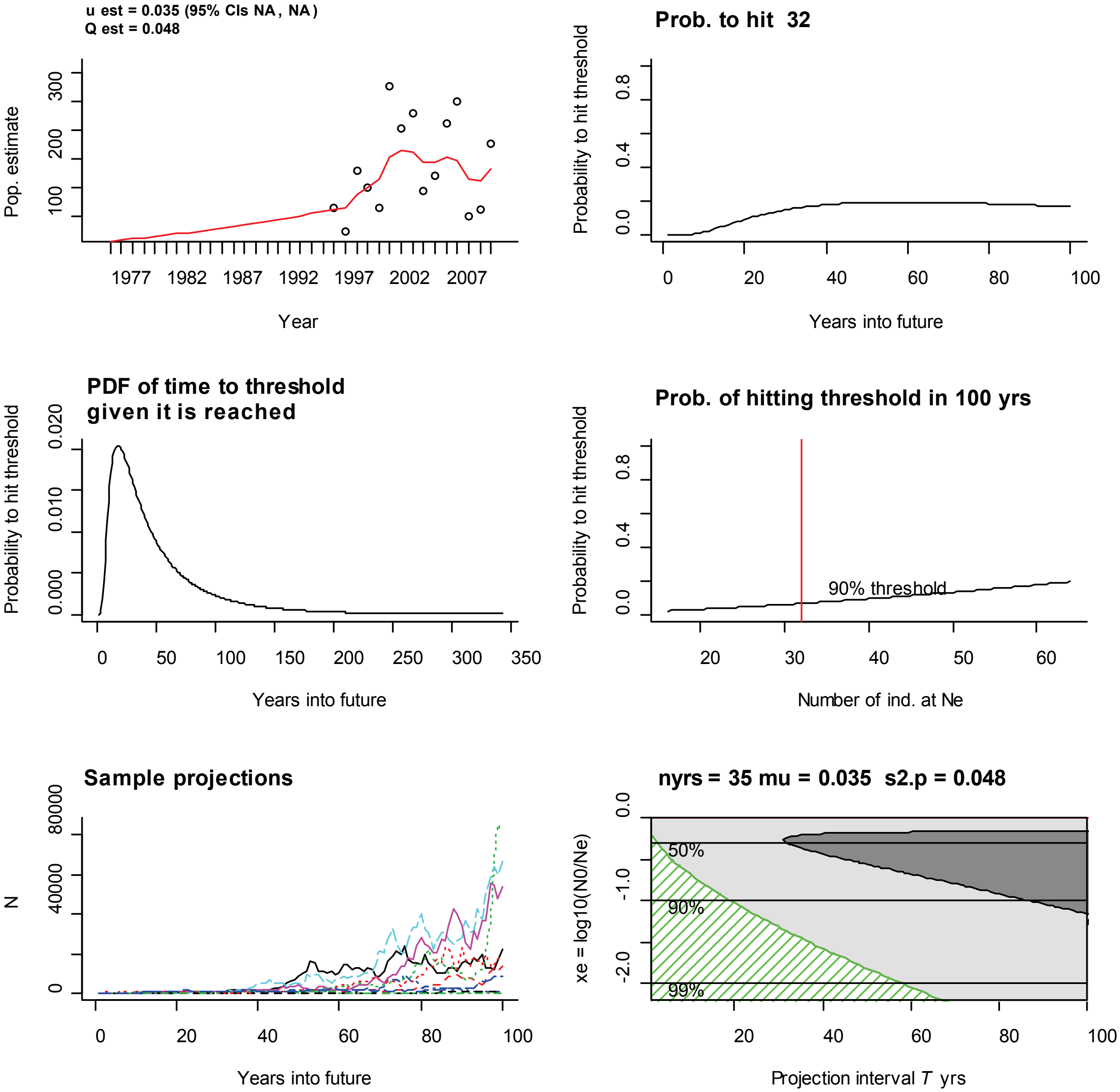

Figure A-17. MARSS-based (Holmes and Ward 2011) PVA for the West Hood Canal Tributaries Winter Run. Steelhead counts in West Hood Canal have tended to increase since the mid 1990s, but variability in abundance has been high. The estimated probability that this population would decline to a QET of 32 fish is low—less than 20% within 100 years. With an estimated (positive) mean population growth rate of 0.035 ($\lambda = 1.096$) and process error of 0.048, we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 20 years and that a 99% decline will not occur within the next 60 years. However, beyond the next few decades, we are uncertain about the precise level of extinction risk. One issue with this analysis is the inclusion of supplementation fish in the returns for the Hamma Hamma River, where the contribution of hatchery fish to overall abundance is out of proportion to the stream size.

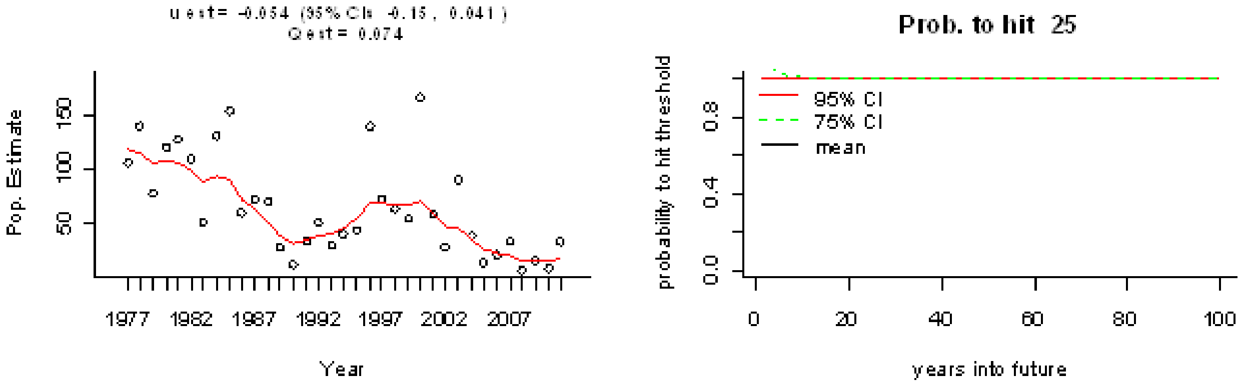

Figure A-18. MARSS-based (Holmes and Ward 2011) PVA for the Sequim/Discovery Bays Tributaries Winter Run. Steelhead counts in this area have varied considerably since the 1980s, but generally have declined and recently have been very low. The estimated probability that this population would decline to a QET of 25 fish is high—about 90% within 100 years. The estimated mean population growth rate is −0.054 (λ = 0.947) and process error is 0.074. We are uncertain about the precise level of extinction risk for this population, but it is clearly high due to low abundance.

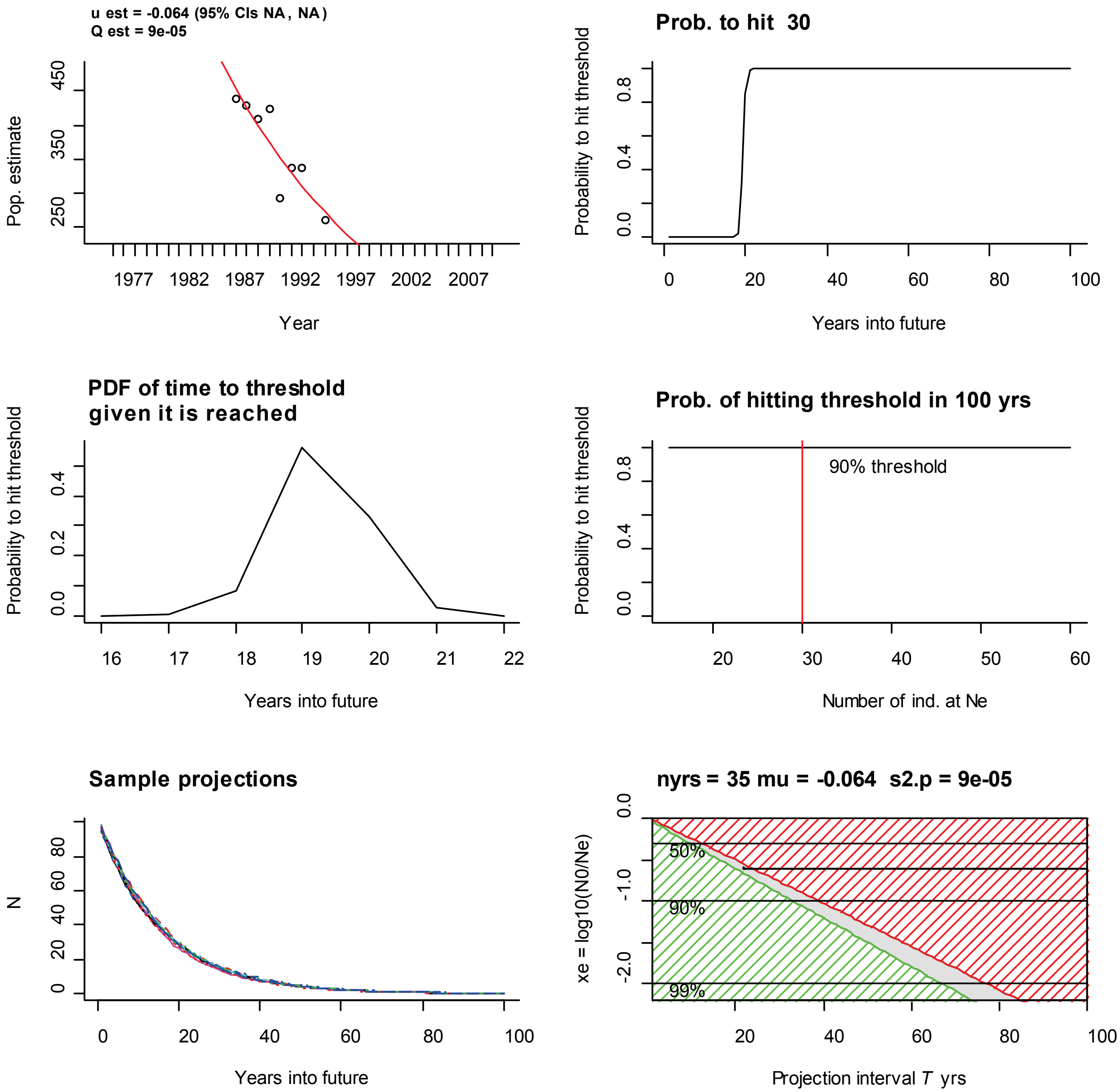

Figure A-19. MARSS-based (Holmes and Ward 2011) PVA for the Dungeness River Summer Run and Winter Run. Steelhead counts in the Dungeness River declined steeply between the late 1980s and the late 1990s. Accurate counts in more recent years are not available. The estimated probability that this population would decline to a QET of 30 fish is high—about 90% within 20 years. With an estimated mean population growth rate of −0.064 ($\lambda = 0.938$) and process error of <0.001, this analysis indicates that we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 35 years and that a 99% decline will not occur within the next 65 years. However, a 50% decline is highly likely within 12 years and a 90% decline within 40 years. The analysis indicates that there is little uncertainty that the population would continue to decline if current conditions observed in the 1990s persisted into the future. However, the lack of data in recent years means that the population's viability is far from certain.

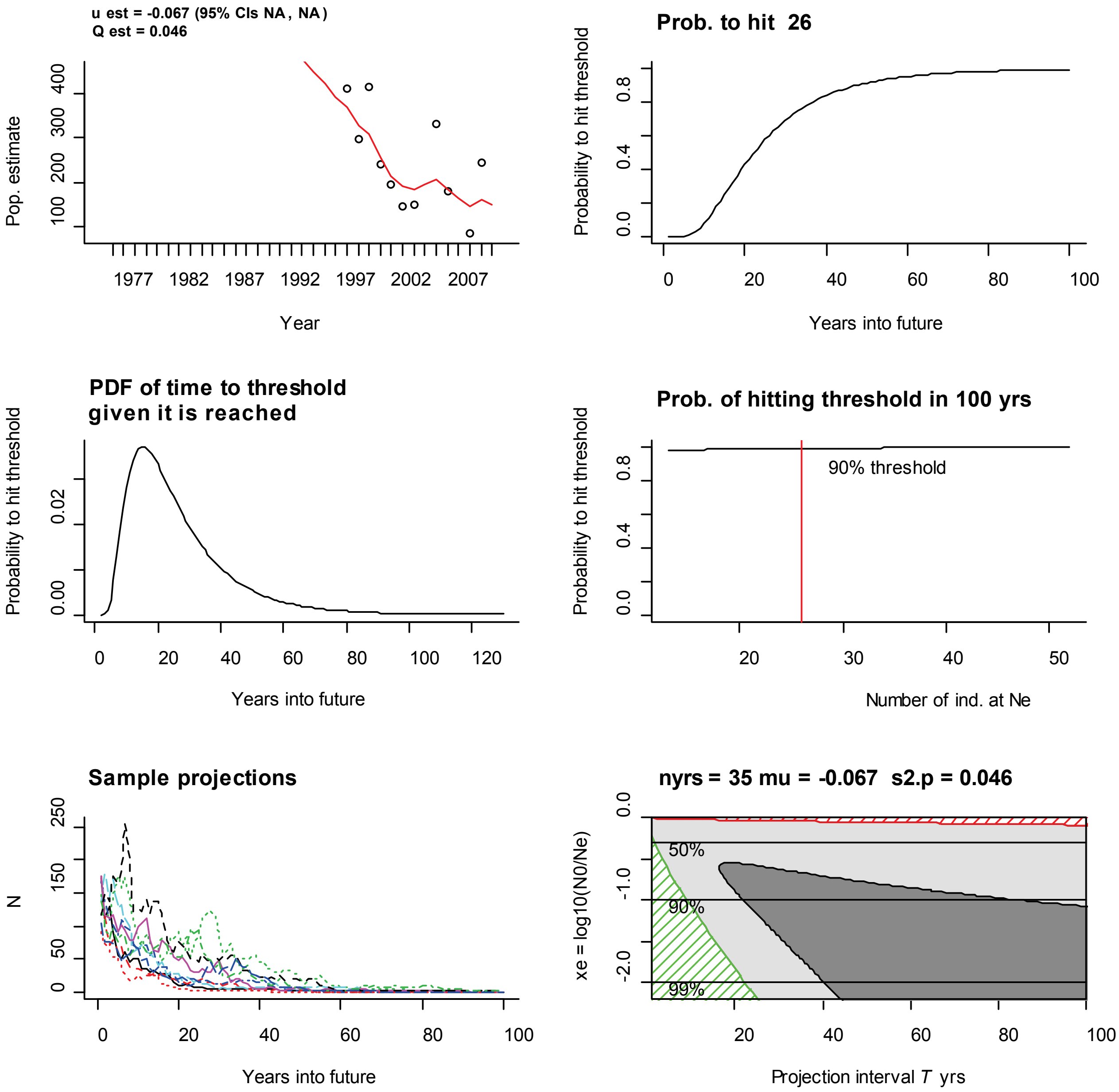

Figure A-20. MARSS-based (Holmes and Ward 2011) PVA for the Strait of Juan de Fuca Tributaries Winter Run (represented by Morse and McDonald creeks data). Steelhead counts in this area have declined steeply since the late 1990s. Based on the limited count data, the estimated probability that this population would decline to a QET of 26 fish is high—about 90% within 60 years. With an estimated mean population growth rate of −0.067 (λ = 0.935) and process error of 0.046, we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 10 years and that a 99% decline will not occur within the next 25 years. However, we can be confident of a modest decline (perhaps 20%) within 100 years.

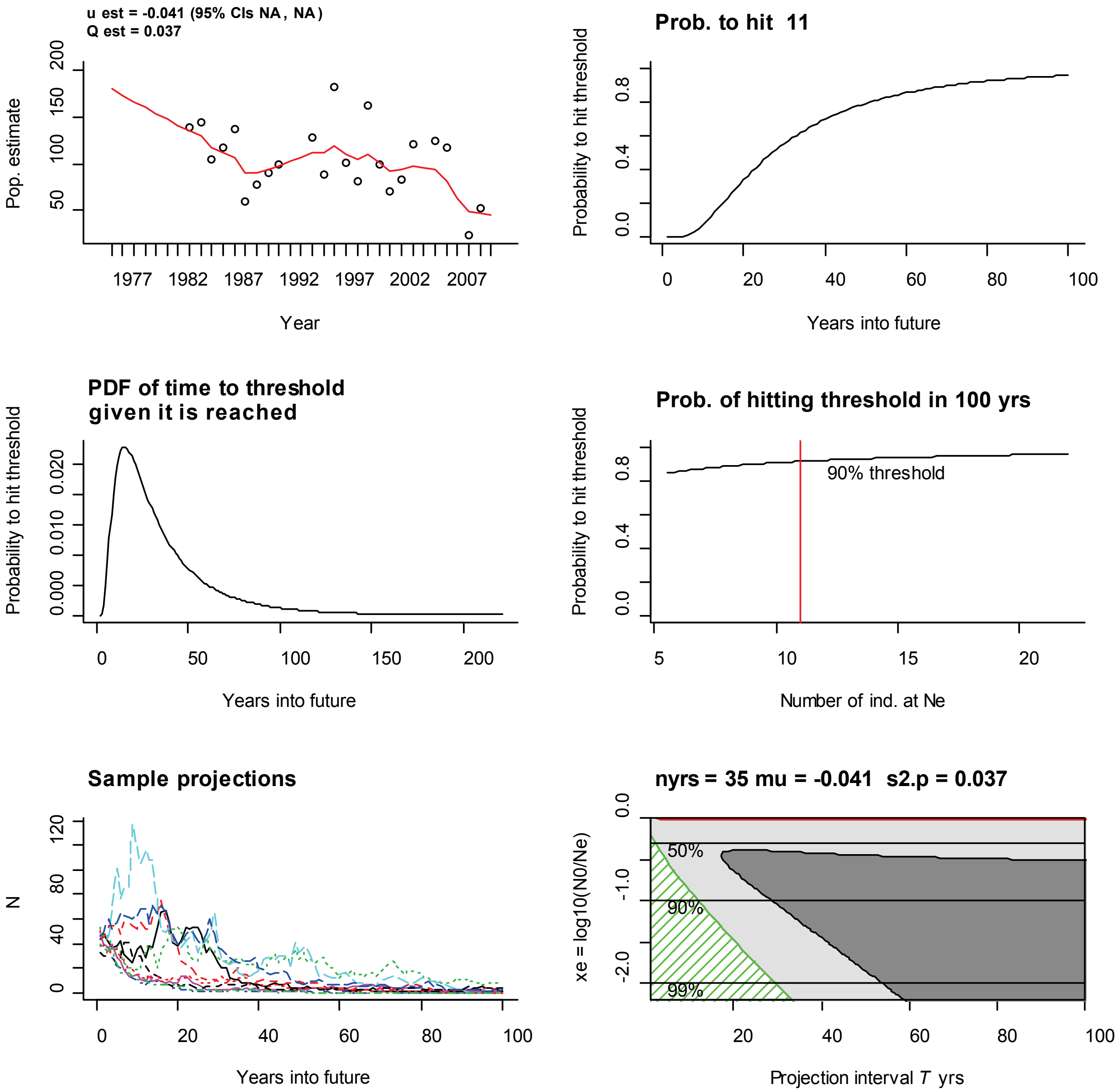

Figure A-21. MARSS-based (Holmes and Ward 2011) PVA for the Morse Creek winter-run population, part of the Strait of Juan de Fuca Tributaries Winter-Run DIP, which we analyzed because of the longer time series available compared to Morse and McDonald creeks combined data. Steelhead counts in Morse Creek have declined, albeit at a slower rate than Strait of Juan de Fuca Tributaries steelhead as a whole (Figure A-20), since the mid 1980s. The estimated probability that this population would decline to a QET of 11 fish is high—almost 90% within 100 years. With an estimated mean population growth rate of −0.041 ($\lambda = 0.960$) and process error of 0.037, we can be highly confident ($P < 0.05$) that a 90% decline in this population will not occur within the next 12 years and that a 99% decline will not occur within the next 30 years. However, beyond the next 30 years, we are uncertain about the precise level of extinction risk.

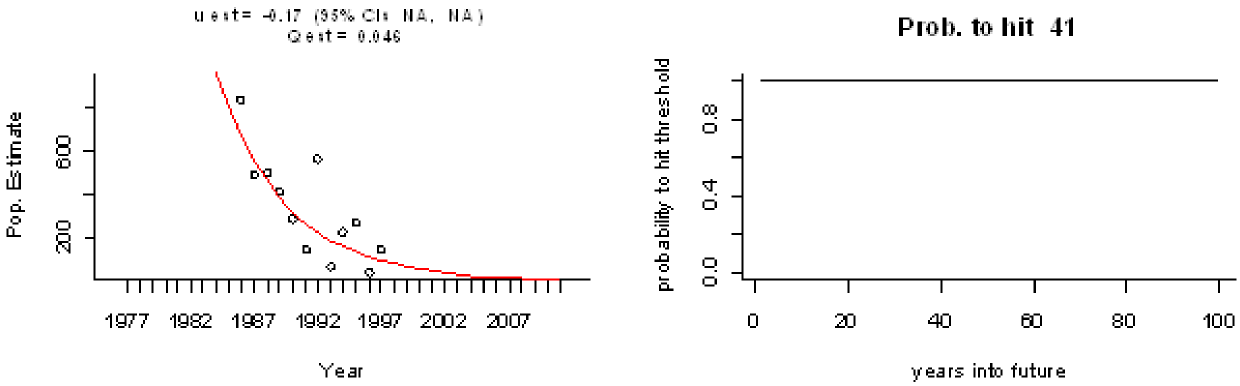

Figure A-22. MARSS-based (Holmes and Ward 2011) PVA for the Elwha River Winter Run. Like Dungeness River steelhead counts, counts in the Elwha River declined steeply between the 1980s and late 1990s, after which data are not available. Based on the limited count data, the estimated probability that this population would decline to a QET of 41 fish is high—at least 90% currently. The analysis indicates that the estimated mean population growth rate is −0.17 ($\lambda = 0.844$) and the process error is 0.046. The analysis indicates that there is little uncertainty that the population would continue to decline if current conditions observed in the 1990s persisted into the future. However, the lack of data in recent years means that the population’s viability, while low, is far from precisely known.

Appendix B: Interim Abundance-Based Viability Guidelines for Puget Sound Steelhead

Assessment of distinct population segment (DPS) viability is rooted in the evaluation of demographically independent population (DIP) viable salmonid population (VSP) parameters. From a recovery planning standpoint, however, viability criteria for Puget Sound steelhead (*Oncorhynchus mykiss*) are more easily understood from a top-down DPS to DIP perspective. In part this reflects the nature of the challenge the Puget Sound Steelhead Technical Recovery Team (PSS TRT) faced in identifying population-specific viability criteria for these steelhead. In contrast to Chinook salmon (*O. tshawytscha*), the quality and quantity of data on steelhead abundance and life history are much more limited. Although there have been a number of recent efforts by the comanaging agencies to consolidate existing databases and expand sampling (especially genetic sampling), the PSS TRT was only able to assess VSP status for roughly 50–70% of the 32 DIPs in the DPS. In developing DIP-level viability criteria, the TRT had to compare the metrics that are most biologically relevant to viability with those that are available or may become available in the foreseeable future. While this process continues, there is a desire by public, governmental, and tribal entities to begin recovery efforts. As an interim measure, the TRT recommends the use of DIP abundance goals to support initial recovery planning. Existing DPS and major population group (MPG)–level viability criteria (described below) provide the framework for developing recovery scenarios. Interim abundance criteria were estimated using the intrinsic potential model and correspond to different levels of sustainability (Appendix C).

1. Those DIPs selected for achieving “viable” status should maintain a 4-year average spawner abundance specified by a 5% smolt-to-adult survival (SAS) rate (Table B-1).
2. Of the remaining populations in each MPG, at least 75% of the populations should maintain 4-year average spawner abundances specified by a 1% SAS rate (Table B-1).
3. Finally, the remaining 25% of the populations in each MPG should not be allowed to degrade below their current level of abundance.
4. Spawner counts will consider “naturally produced” adults only, not including those naturally produced fish that are known to be descended from out-of-DPS lineages.

In addition to these abundance thresholds, the TRT recommends the following actions to address information needs while recovery planning is pursued.

1. Juvenile and adult monitoring efforts should be expanded into DIPs with little or no current monitoring.
2. Genetics samples should be acquired in those DIPs that are currently not represented.
3. Selected DIPs (or subbasins within DIPs) should be intensively monitored for life stage survivals.
4. Selected DIPs (or subbasins within DIPs) should be monitored to estimate the contribution of resident *O. mykiss* to anadromous escapement.

5. Selected DIPs (or subbasins within DIPs) should be monitored to estimate the influence of hatchery introductions.

Given the current status of most DIPs in the Puget Sound Steelhead DPS, these abundance levels are well above current estimates. It falls upon the recovery planning entities to determine which populations are to be recovered to full viability. Under any potential scenario, it is likely that considerable time and effort will be required to reach the viability criteria. Under these circumstances, there appears to be little risk to establishing interim abundance goals.

Table B-1. Interim DIP abundance goals for steelhead in Puget Sound, based on a 4-year average. Abundance goals for summer-run fish (two creeks and three rivers in italics) are still under review. QET = quasi-extinction threshold, SAS = smolt-to-adult survival, low (minimum) abundance = 100, and viable = 250.

Population basin					**Abundance**		
Population name	Area (km^2)	Mean elev. (m)	Total stream length (m)	**QET**	Low (1% SAS)	Viable (5% SAS)	Capacity (20% SAS)
Baker River	771	999	421,859	36	503	2,514	10,056
Canyon Creek	*100*	*864*	*47,716*	*24*	*100 (12)*	*250 (60)*	*243*
Cedar River	650	461	402,349	35	595	2,975	11,899
Deer Creek	*180*	*761*	*105,313*	*31*	*157*	*786*	*3,144*
Drayton Harbor Tributaries	223	37	206,057	26	243	1,213	4,852
Dungeness River	564	978	306,740	30	246	1,232	4,930
East Hood Canal Tributaries	342	99	174,736	27	127	635	2,540
East Kitsap Peninsula Tributaries	703	75	259,413	25	156	779	3,115
Elwha River	833	1,021	472,871	41	712	3,558	14,231
Green River	1,444	463	834,472	69	1,977	9,884	39,537
Nisqually River	1,991	524	1,030,771	55	1,533	7,665	30,660
Nookachamps Creek	183	252	159,503	27	123	616	2,462
Nooksack River	1,982	619	1,257,480	73	2,205	11,023	44,091
North Fork Skykomish River	*156*	*1,195*	*117,602*	*25*	*100 (66)*	*331*	*1,325*
N. Lake Washington Tributaries	978	119	441,887	36	527	2,634	10,536
Pilchuck River	356	253	242,383	34	519	2,597	10,386
Puyallup River	1,395	672	803,817	58	1,472	7,358	29,432
Samish R./Bellingham Bay Tribs.	661	203	453,694	31	319	1,596	6,386
Sauk River	1,897	1,132	1,079,263	103	2,323	11,615	46,460
Sequim/Discovery Bays Tribs.	557	197	234,042	25	100 (51)	256	1,024
Skagit River	5,543	1,098	2,815,113	157	6,478	32,388	129,551
Skokomish River	634	570	411,699	50	1,003	5,015	20,060
Snohomish/Skykomish River	1,595	420	1,021,690	73	2,139	10,695	42,779
Snoqualmie River	1,615	620	1,134,038	58	1,674	8,370	33,479
South Fork Nooksack River	*172*	*926*	*99,347*	*27*	*114*	*568*	*2,273*
South Hood Canal Tributaries	295	126	216,935	30	299	1,493	5,970
South Puget Sound Tributaries	1,860	84	582,451	42	985	4,927	19,709
Stillaguamish River	1,230	398	927,234	67	1,912	9,559	38,236
Strait of Juan de Fuca Tributaries	403	611	246,441	26	100 (73)	364	1,456
Tolt River	*182*	*784*	*117,732*	*25*	*100 (32)*	*250 (160)*	*641*
West Hood Canal Tributaries	1,433	715	842,382	32	361	1,804	7,217
White River	1,285	1,061	863,251	64	1,749	8,745	34,981
			Totals:	1,462	30,449	153,194	613,662

Appendix C: Calculation of Intrinsic Potential Estimates for Puget Sound Steelhead DIPs

The sustainability of any demographically independent population (DIP) is based in large part on the maintenance of a minimum numbers of spawners over an extended period. In a simplified life cycle model of steelhead (*Oncorhynchus mykiss*), adult abundance is based on freshwater productivity and marine survival. Freshwater productivity in turn is based on habitat quality and stream characteristics. In order to estimate freshwater productivity, the Puget Sound Steelhead Technical Recovery Team (PSS TRT) generated estimates of stream length, stream area (wetted bankfull area), and stream gradient using GIS-based data layers obtained from the U.S. Geological Survey's National Hydrography Dataset (at a scale of 1:100,000) for 200 m stream reaches. These data were integrated into an intrinsic potential (IP) model to estimate productivity, adapted from data in Gibbons et al. (1985) and the Interior Columbia TRT's steelhead IP model, and based primarily on stream size, gradient (using the U.S. Geological Survey 10 m resolution digital elevation model), and hydrograph for reaches below impassable barriers. For Puget Sound steelhead, we adopted a model incorporating three stream widths and three stream gradient classes. Stream habitat was initially classified as having low, moderate, and high productivity (Table C-1). We then further categorized stream reaches by whether they were main stem or tributary, and whether the hydrograph was influenced by snowpack or not (intersection of the reach with a 0°C contour in January).

Adult steelhead occupancy of stream reaches was determined from the Washington Department of Fish and Wildlife's Salmonscape database (online at http://wdfw.wa.gov/mapping/salmonscape), steelhead spawning surveys where redd locations were documented, and subbasin reports produced by the Washington Conservation Commission.

Steelhead freshwater productivity has been estimated in a number of ways. Chapman (1981) estimated production under pristine conditions at 0.0877 parr/m^2 (equivalent to 0.0263 smolts/m^2). Gibbons et al. (1985) developed a more complex productivity model, based on

Table C-1. Stream habitat rating matrix (below natural barriers) for streams for Puget Sound steelhead. Stream size and gradient categories were assigned by PSS TRT members based on consideration of the Interior Columbia TRT's IP model and on expert opinion. The PSS TRT used these basin characteristics to calculate the IP of Puget Sound steelhead basins in order to establish whether a large enough population could be sustained into the foreseeable future.

	Stream bankfull width		
Stream gradient	0–3 m	3–20 m	>20 m
0.00–0.25%	High	Moderate	Low
0.25–4.00%	Moderate	High	Moderate
>4.00%	Low	Low	Low

observed parr densities categorized according to stream gradient and size, with parr productivity for Puget Sound streams varying from 0.05 to 0.12 parr/m^2. Gibbons et al. (1985) found that small independent tributaries had some of the highest productivities. On average, western Washington stream productivity was 0.0717 parr/m^2 with 0.0265 spawners/parr (Gibbons et al. 1985). Similarly, the U.S. Army Corps of Engineers (1988) estimated potential steelhead freshwater productivity at 0.067 parr/m^2 for streams and 0.041 parr/m^2 for rivers. We combined an average estimate for parr productivity of 0.0754 parr/m^2 with the Chapman (1981) parr-to-smolt survival of 0.30, to establish a 0.023 smolts/m^2 level of productivity. Low productivity areas were not included in the estimate of potential parr numbers. This is similar to estimates for the Keogh River, 0.032 smolts/m^2 (Tautz et al. 1992). Smolt-to-adult survival (SAS) was estimated using Keogh River studies (Ward and Wightman 1989), with a 20% survival rate selected to estimate average pre-European contact productivity.

Using the 20% SAS rate as an estimate of capacity provided a frame of reference for estimating two other abundance thresholds. First, a viability abundance threshold, where a population is sustainable over longer periods, specifically meets the 5% risk of reaching quasi-extinction threshold over a 100-year period. For planning purposes, IP calculations using a 5% SAS provided a reasonable estimate of the abundance threshold for a viable population. Alternatively, a low abundance threshold provides demographic sustainability over periods less than 100 years and is large enough to prevent the loss of genetic diversity over a similarly short period of time. Using a 1% SAS rate in calculating the IP capacity provided an estimate of this low abundance threshold. Review of the IP calculations suggested that abundance thresholds for populations with higher gradient headwaters were too low; this was especially the case for summer-run populations. For all populations, the low abundance threshold was set at a minimum of 100 adults/year (4-year average), while the minimum viable abundance threshold was set at 250 adults/year (4-year average).

Appendix D: Selected Steelhead Documents

In addition to the documents cited in this technical memorandum and listed in the References section, the following is a bibliography of documents we found useful for various research aspects, such as general steelhead biology or information about particular watersheds.

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Appendix E: Description of Puget Sound Steelhead DIPs

This appendix presents in a condensed form the viable salmonid population (VSP) attributes (McElhany et al. 2000) of each of the steelhead (*Oncorhynchus mykiss*) demographically independent populations (DIPs) that existed historically in the Puget Sound Steelhead Distinct Population Segment (DPS), as well as hydrology data. Much of the fish data were obtained from the Washington State Salmon and Steelhead Stock Inventory (WDF et al. 1993), the Washington State Salmonid Stock Inventory (WDFW 2002), the Salmon and Steelhead Hatchery Assessment Group report (SSHAG 2003), and the Pacific States Marine Fisheries Commission's StreamNet database (StreamNet no date).

The Four VSP Attributes

Abundance and Productivity

Productivity and related trends in abundance reflect conditions that drive a population's dynamics and thus determine its abundance. Changes in environmental conditions, including ecological interactions, can influence a population's intrinsic productivity or the environment's capacity to support a population or both. Such changes may result from random environmental variation over a wide range of temporal scales (environmental stochasticity). In this section, however, we are most concerned with measures of population growth and related parameters that reflect systematic changes in a population's dynamics (McElhany et al. 2000).

Abundance estimates for each DIP are generally based on expansions of redd counts from index areas. In a few cases, fish passage facilities allow relatively precise counts of returning adults. In other areas, systematic surveys are not available and anecdotal reports alone establish the presence of steelhead. In these cases, it may be necessary to use adjacent DIPs as proxies to estimate abundance; however, except where accessibility is limited, it is likely that the absence of survey activity is related to low steelhead abundance.

Productivity is generally derived from abundance data series, either as lambda, trend, or spawner/recruit calculations. Given the relatively high uncertainty in abundance estimates (particularly expansions of index redd counts), productivity estimates include considerable uncertainty.

Several historical measures of abundance are included to provide context for present-day abundance estimates. These include a summary of salmon streams conducted in 1930 (WDG 1931–1940). Populations were categorized as being large, medium, scarce, very scarce, and absent, with no numerical equivalents given. Punch card estimates are derived from Washington Department of Game estimates. These estimates are "corrected" by a factor of 0.60 to account for reporting bias. Peak catch estimates were selected from those return years that were not supplemented, based on data provided in WDG (no date). Punch card estimates do not include any adjustment for commercial or tribal catch. Finally, habitat based intrinsic potential (IP) estimates were developed to approximate historical production. It should be noted that this system was developed to establish whether or not the basin could support a DIP; it is not considered a precise estimate and likely underestimates potential productivity.

Diversity

In their VSP report, McElhany et al. (2000) identify three general reasons why diversity is important for species and population viability. First, diversity allows a species to use a wider array of environments than it could without it. For example, varying adult run and spawn timing allows several salmonid species to use a greater variety of spawning habitats than would be possible without this diversity. Second, diversity protects a species against short-term spatial and temporal changes in the environment. Fish with different characteristics have different likelihoods of persisting—depending on local environmental conditions. Therefore, the more diverse a population is, the more likely it is that some individuals would survive and reproduce in the face of environmental variation. Third, genetic diversity provides the raw material for surviving long-term environmental changes. Salmonids regularly face cyclic or directional changes in their freshwater, estuarine, and ocean environments due to natural and human causes, and genetic diversity allows them to adapt to these changes.

Two major life history types of steelhead were historically and are presently found in Puget Sound: the summer run and winter run. The life histories of summer-run and winter-run steelhead have considerable overlap. Both rear in freshwater for 1–4 years prior to smoltification, select similar habitats for freshwater rearing, and spend 1–4 years in the ocean. However, substantial differences separate these races at the time of adult freshwater entry, degree of sexual maturity at entry, and spawning time.

Each year, the majority of naturally produced Puget Sound summer-run steelhead enters freshwater between May and October. These fish are still in the process of sexually maturing upon return to their natal streams. The fish subsequently spawn between January and June, with peak spawning between late February and early April. In contrast, wild winter-run steelhead enter freshwater as sexually mature fish between December and May. Spawning occurs between February and June, with peak spawning time in late April and early May, almost 2 months later than wild summer-run steelhead.

On average, there is a 2-month difference in peak spawning time between winter-run and summer-run steelhead, although there is some temporal overlap in the spawning distribution (Busby et al. 1996). Within the same watershed, winter-run and summer-run steelhead maintain a high degree of reproductive isolation by spawning in geographically distinct areas; these areas are generally separated by temporal migration windows created by falls or cascades. In a few cases, winter-run and summer-run steelhead spawn in the same stream reach, but at slightly different times. Hatchery introductions, especially with nonnative steelhead, and modifications to barrier falls are a potential source for the breaking down of historical reproductive barriers and the erosion of locally adapted genotypes.

Diversity status is difficult to quantify due to the near absence of information on historical life history characteristics. Any actions that affect basic demographic and evolutionary processes (e.g., patterns of mutation, selection, drift, recombination, migration, and population turnover) have the potential to alter a species' diversity (McElhany et al. 2000). At a minimum, information on hatchery introductions is available for most DIPs. In some cases, genetic information can substantiate the influence of hatchery fish on the naturally spawning population. Additionally, habitat diversity (the basis of life history diversity) has been assessed for most populations and there are estimates for the proportion of lost habitat diversity.

Regarding artificial propagation, hatchery releases for each DIP have been compiled for the most recent 15 years (3 to 4 generations). Releases of small *O. mykiss* fry (250 fish per lb, or ≈2 g each) have been excluded. The contribution of these fish to steelhead abundance is thought to be minimal. Additionally, many fry releases were into lakes and not necessarily intended to enhance the anadromous population. Hatchery broodstock names have been retained, but where Chambers Creek or Skamania River hatchery-derived stocks are used, a notation of "CC" or "SK" has been added. Hatchery broodstocks derived from either of these sources are considered as being out-of-DPS stocks.

Spatial Structure

Spatial structure is defined by McElhany et al. (2000) as the geographic distribution of individuals in a population and the processes that create that distribution (i.e., habitat structure or migrational barriers). From a demographic perspective, spatial structure within a DIP prevents the stochastic extirpation of smaller breeding units through migration between units and the reduction in the effects of catastrophic events.

Steelhead within a DIP utilize a variety of habitats for spawning, rearing, and adult holding. Ecological differences among primary streams and their tributaries likely influence the expression of distinct life history strategies. Spatial structure can be viewed on an absolute scale: How much of the historically accessible habitat is still available or usable? Alternatively, the distribution of accessible habitat may also affect spatial structure.

Guidance

In evaluating the four VSP attributes, some overlap is to be expected; however, one should be mindful to keep "double scoring" effects to a minimum. Each attribute should be considered for its contribution to population viability or conversely to its contribution to extinction risk. The quantity and quality of the information available will determine, in turn, the accuracy of the estimate of extinction risk. Uncertainty due to data quality should be reflected in VSP scores.

Summaries of Puget Sound Steelhead DIPs

Northern Cascades Steelhead Major Population Group

1. Drayton Harbor Tributaries Winter Run

Overview—This population (Figure E-1) includes steelhead that spawn in tributaries from the Canadian border to Sandy Point, primarily in Dakota and California creeks (Smith 2002). Although steelhead redds are still reported in this DIP, abundance information is very limited. Spawning habitat is limited.

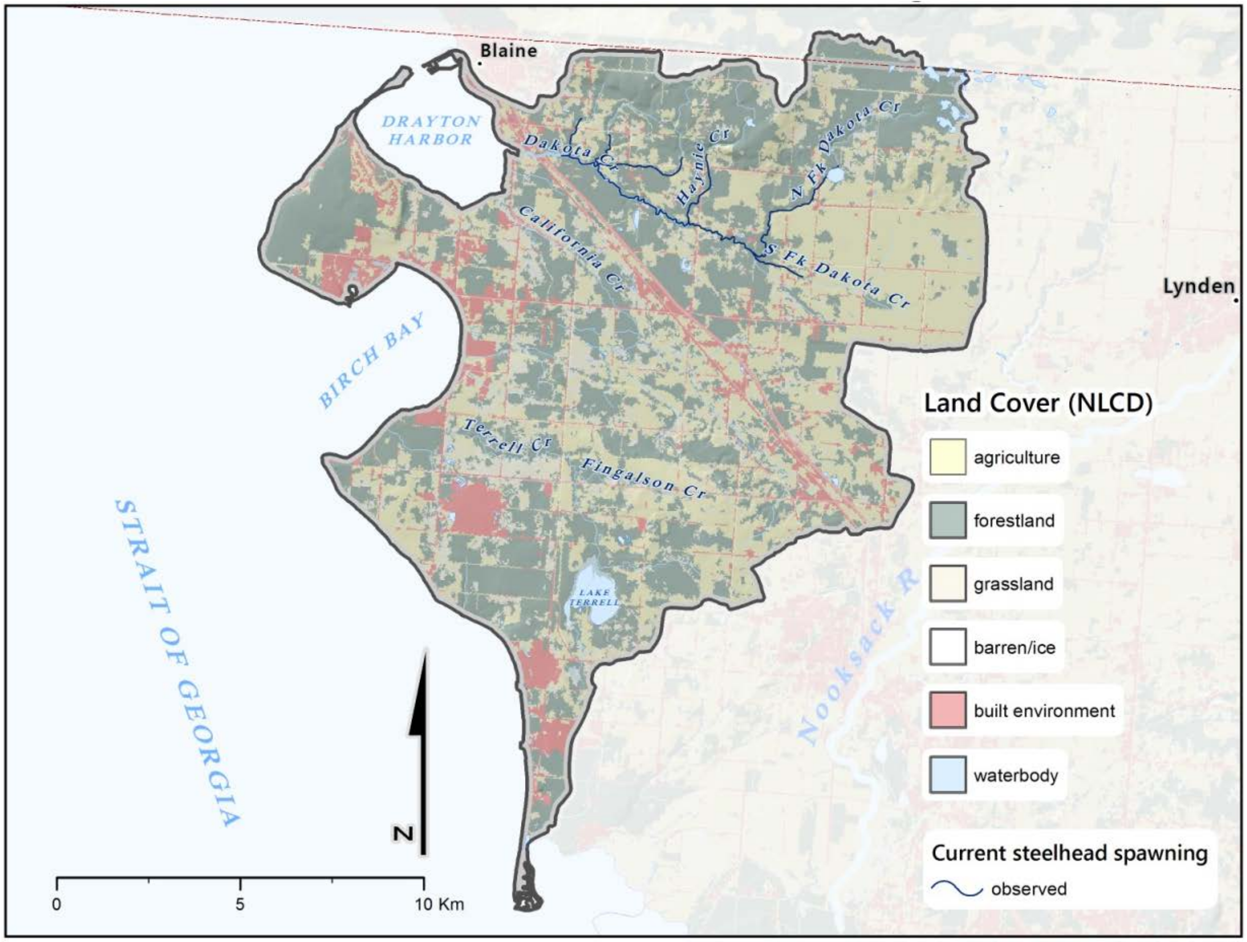

Figure E-1. Map of Drayton Harbor Tributaries Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate*	
Abundance	Medium-sized	67	2,426	4,852
Year	1930	1957	—	

* The IP estimate for this drainage may overestimate capacity for the system, to an unknown degree, because the geology of this drainage appears to limit spawning habitat for salmonids here.

Diversity—Anecdotal reports suggest the steelhead returning to this DIP are slightly smaller in size and spawn earlier than other nearby populations. There are no known Drayton Harbor Tributaries Winter Run hatchery releases (zero within and zero outside the DPS) since 1995.

Spatial structure and hydrology—Figure E-2 and Figure E-3.

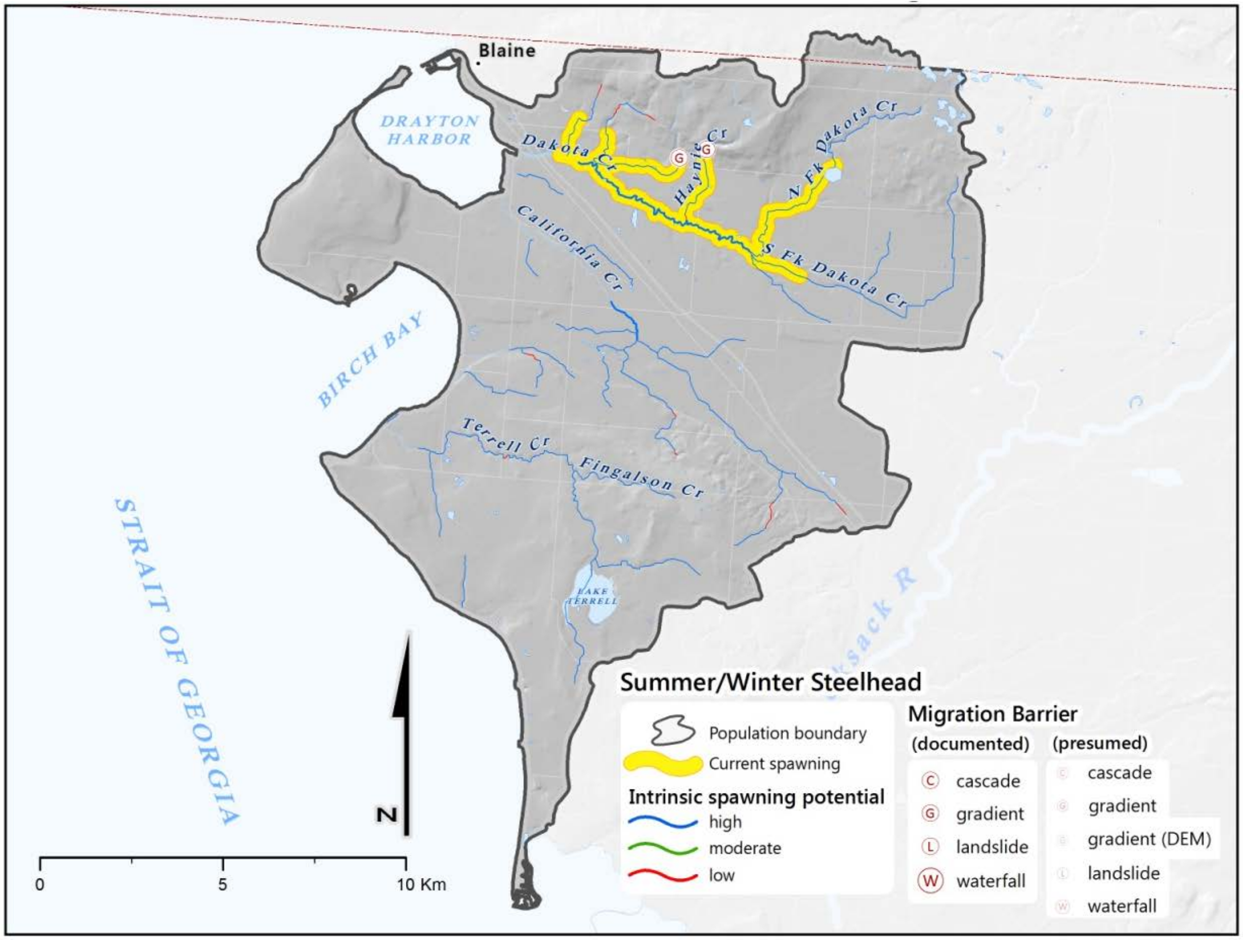

Figure E-2. Map of Drayton Harbor Tributaries Winter-Run DIP spatial structure, including migration barriers and spawning potential.

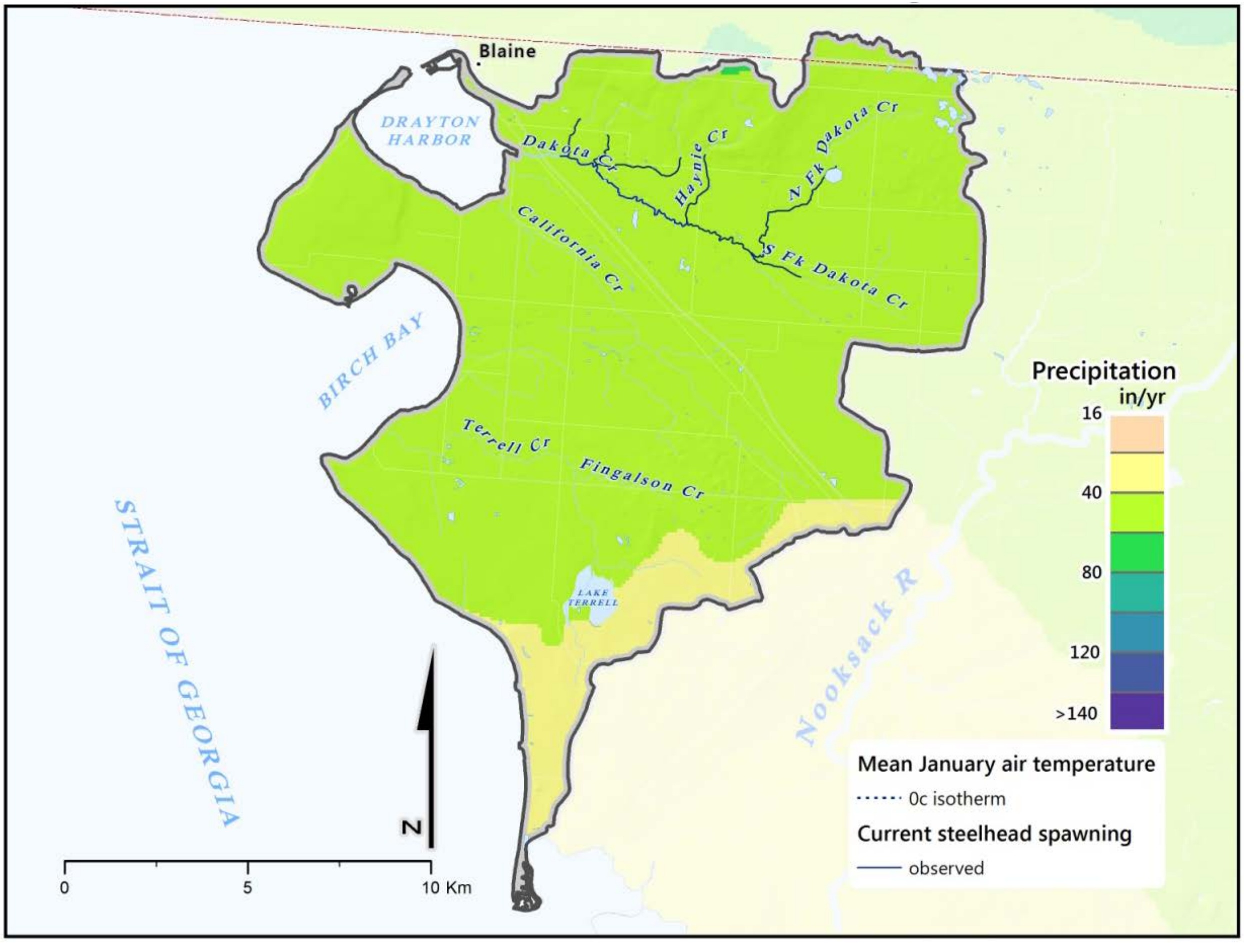

Figure E-3. Map of Drayton Harbor Tributaries Winter-Run DIP area hydrology.

2. Nooksack River Winter Run

Overview—This population (Figure E-4) includes winter-run steelhead in the north fork, middle fork, and south fork of the Nooksack River. The Salmonid Stock Inventory (WDFW 2002) suggests that the Middle Fork Nooksack River may have supported a summer run of steelhead prior to the construction of the impassable diversion dam at RKM 11. Glacial conditions limit spawner surveys in the North Fork Nooksack River. Lowland tributaries (Fishtrap and Bertram creeks) presently contribute significantly to overall abundance.

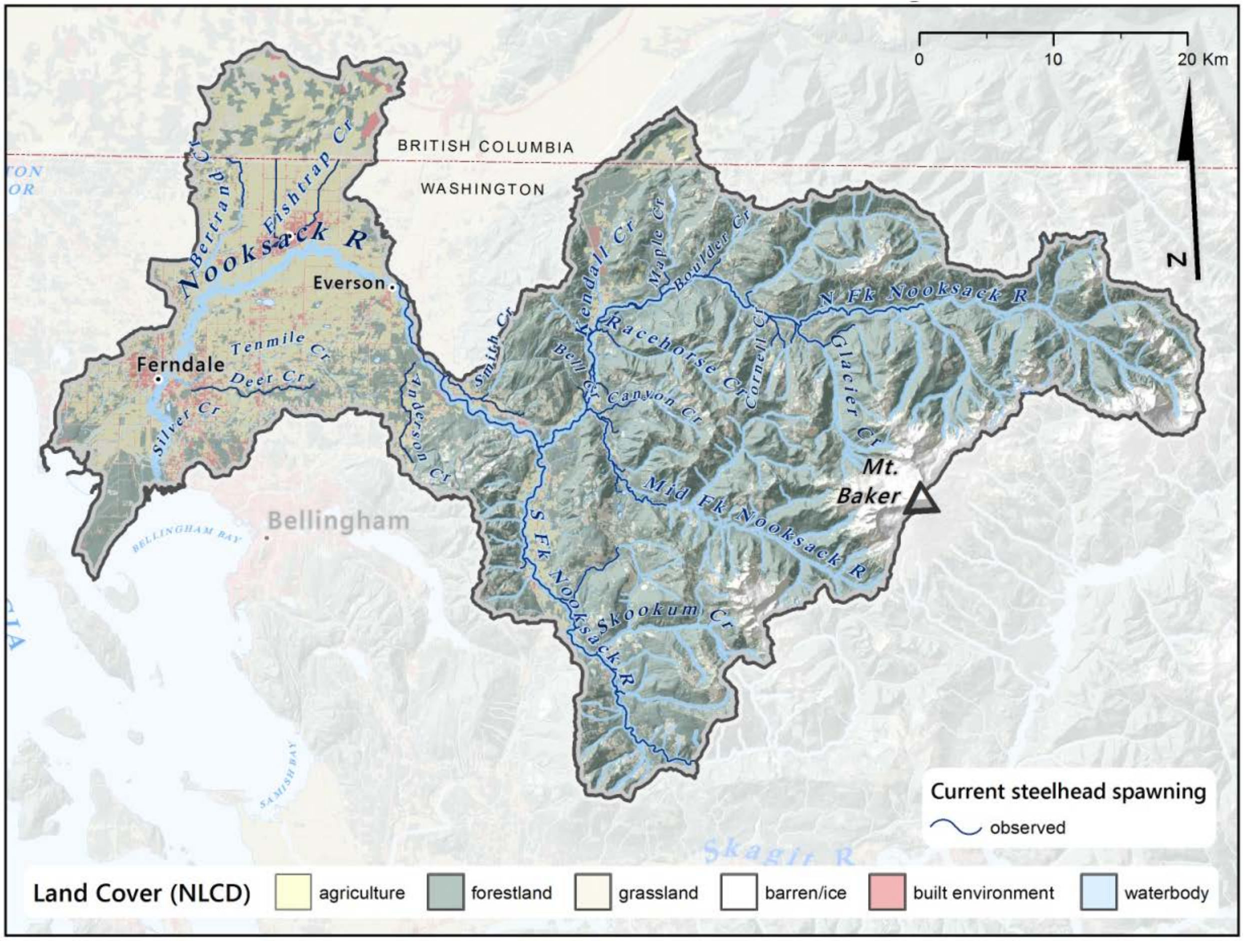

Figure E-4. Map of Nooksack River Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate
Abundance	Multiple large & medium	2,114	22,045 44,091
Year	1930	1953	—

Tributary	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Mainstem north fork	—	—	—	—	1,574	—	—	—	—	—	1,308	1,069
South fork	—	—	—	—	—	—	—	—	—	—	524	425
Middle fork	—	—	—	—	—	—	—	—	—	—	69	104

Diversity—

Nooksack River Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Kendall Cr.	1996–2011	14	Tokul/Bogachiel (CC)	0	1,413,238
M.F. Nooksack	1995–1998	4	McKinnon Pd (CC)	0	268,215
			Total:	0	1,681,453

Spatial structure and hydrology—Figure E-5 and Figure E-6.

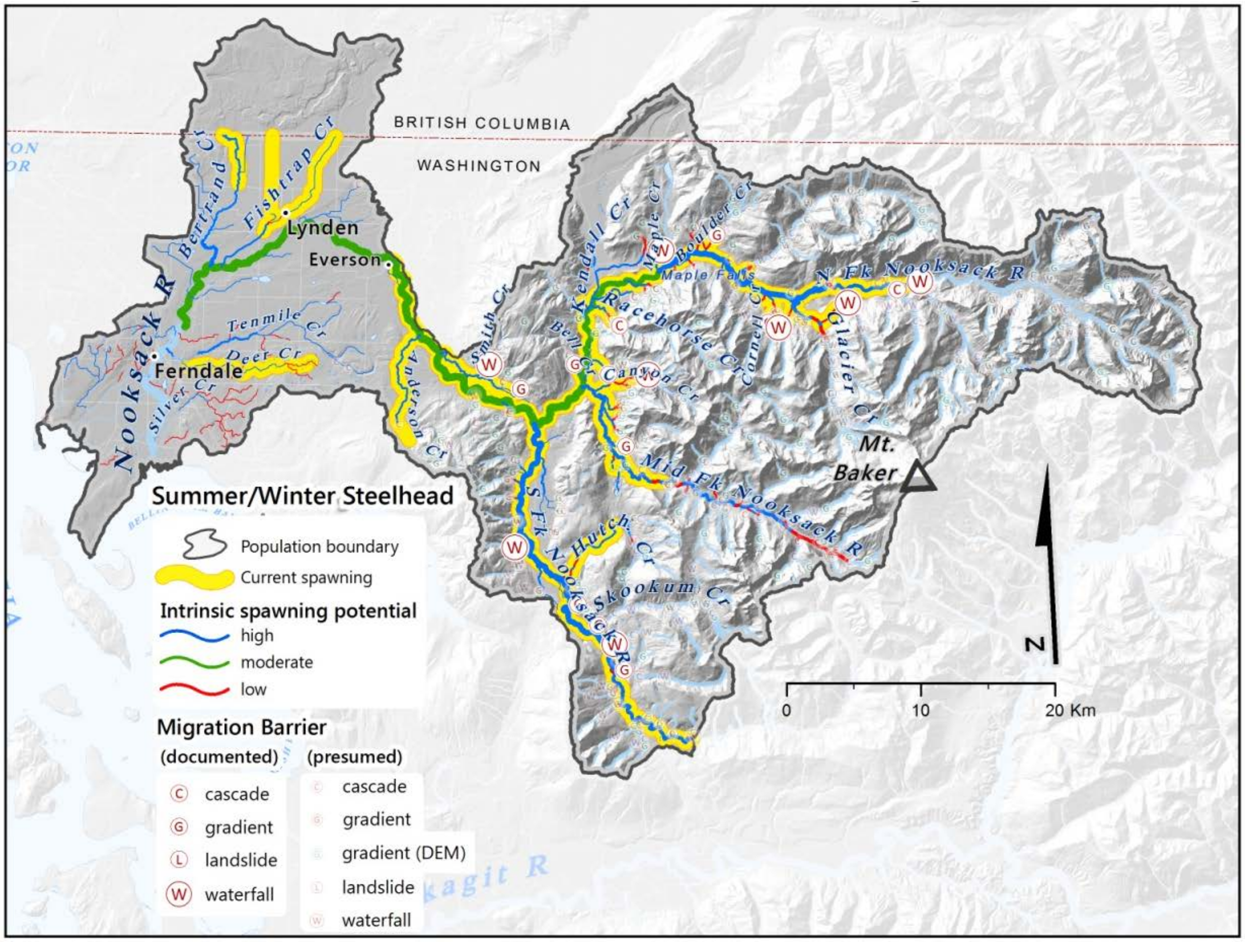

Figure E-5. Map of Nooksack River Winter-Run DIP spatial structure, including migration barriers and spawning potential.

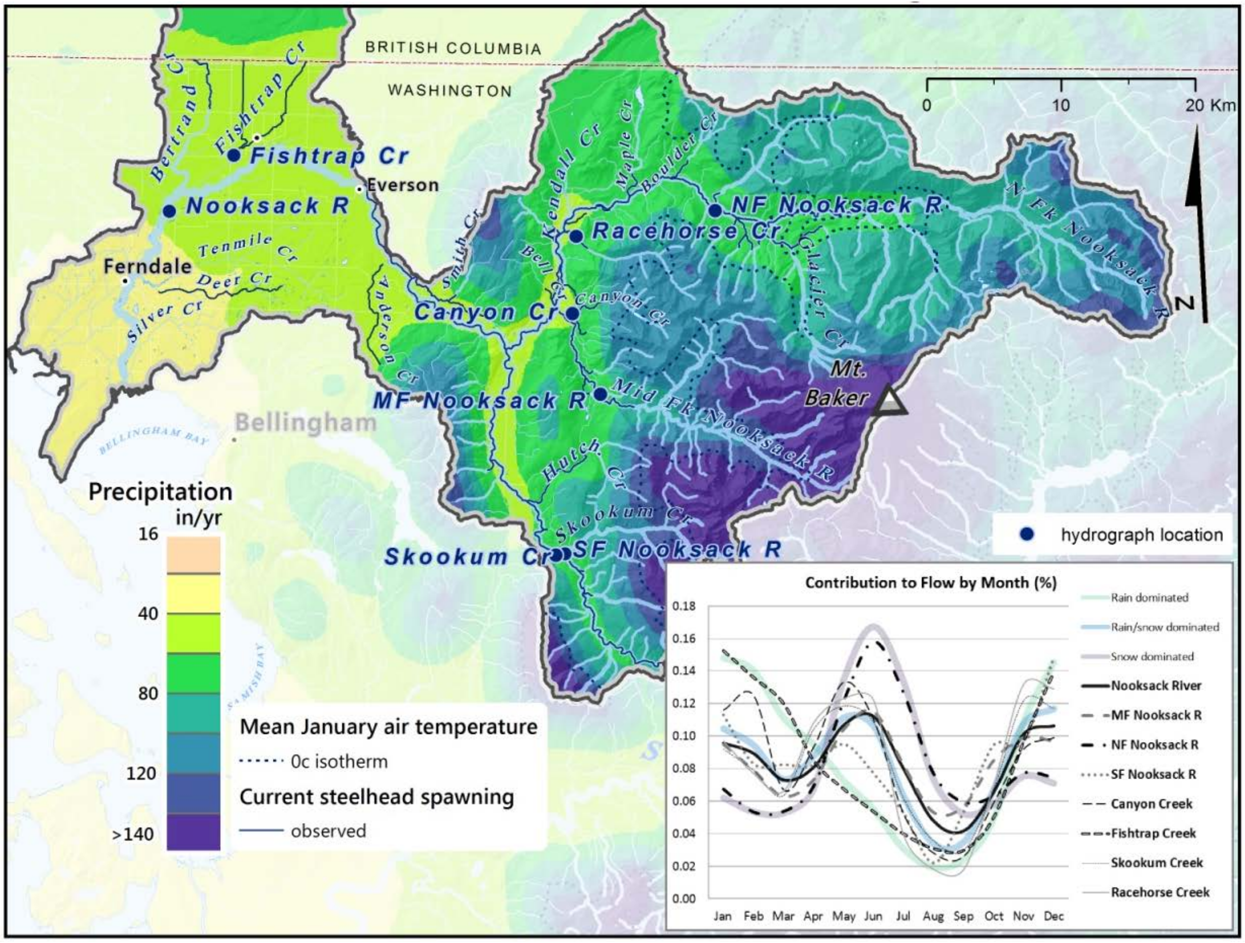

Figure E-6. Map of Nooksack River Winter-Run DIP area hydrology.

3. South Fork Nooksack River Summer Run

Overview—This population (Figure E-7) includes steelhead that spawn in tributaries from the headwaters to Sandy Point, primarily in Dakota and California creeks (Smith 2002). Although steelhead redds are still reported in this DIP, abundance information is very limited.

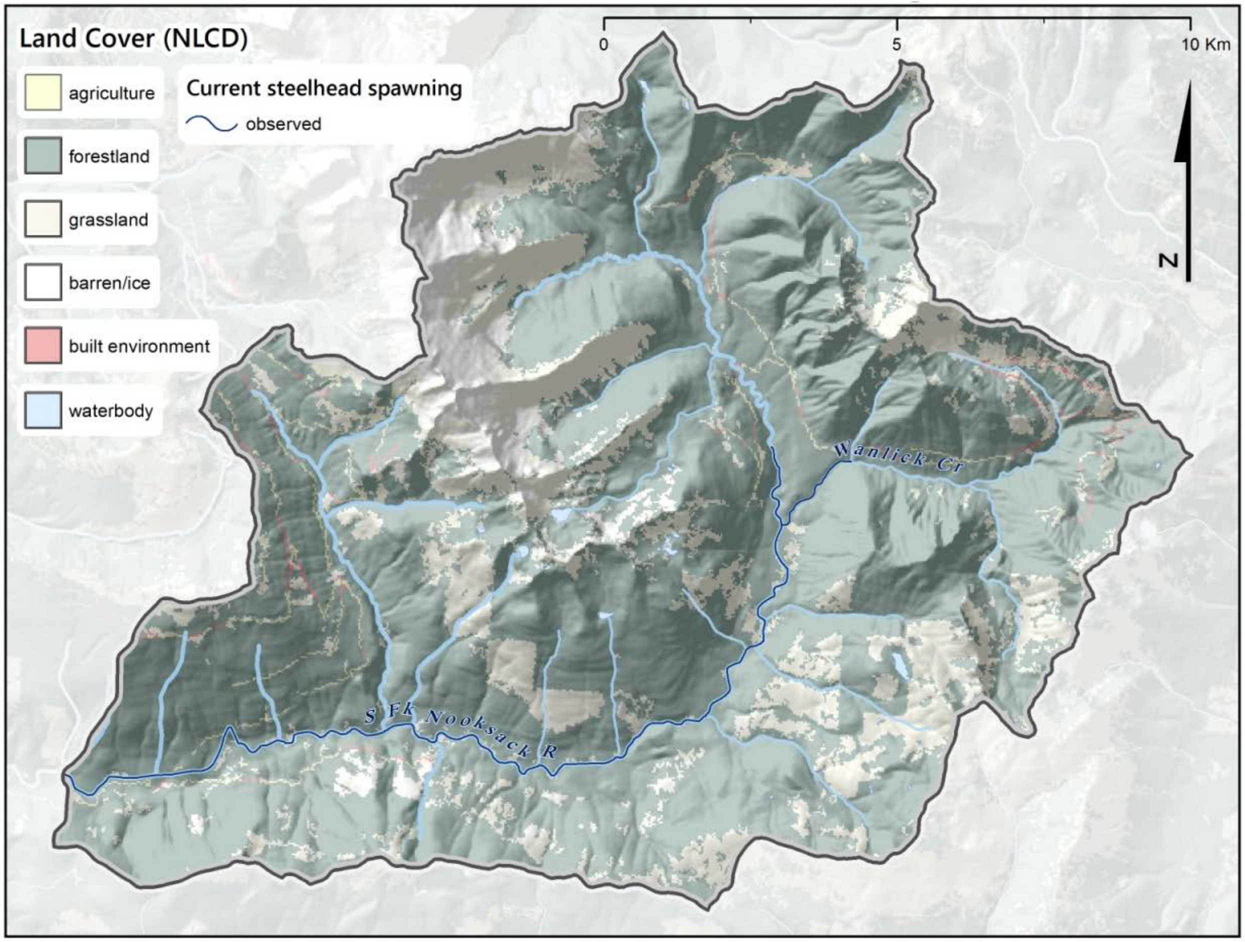

Figure E-7. Map of South Fork Nooksack River Summer-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate	
Abundance	Medium run	31	1,137	2,273
Year	1930	1964	—	

Diversity—Hatchery releases for the Nooksack River Summer Run are zero within the DPS and zero outside the DPS.

Spatial structure and hydrology—Figure E-8 and Figure E-9.

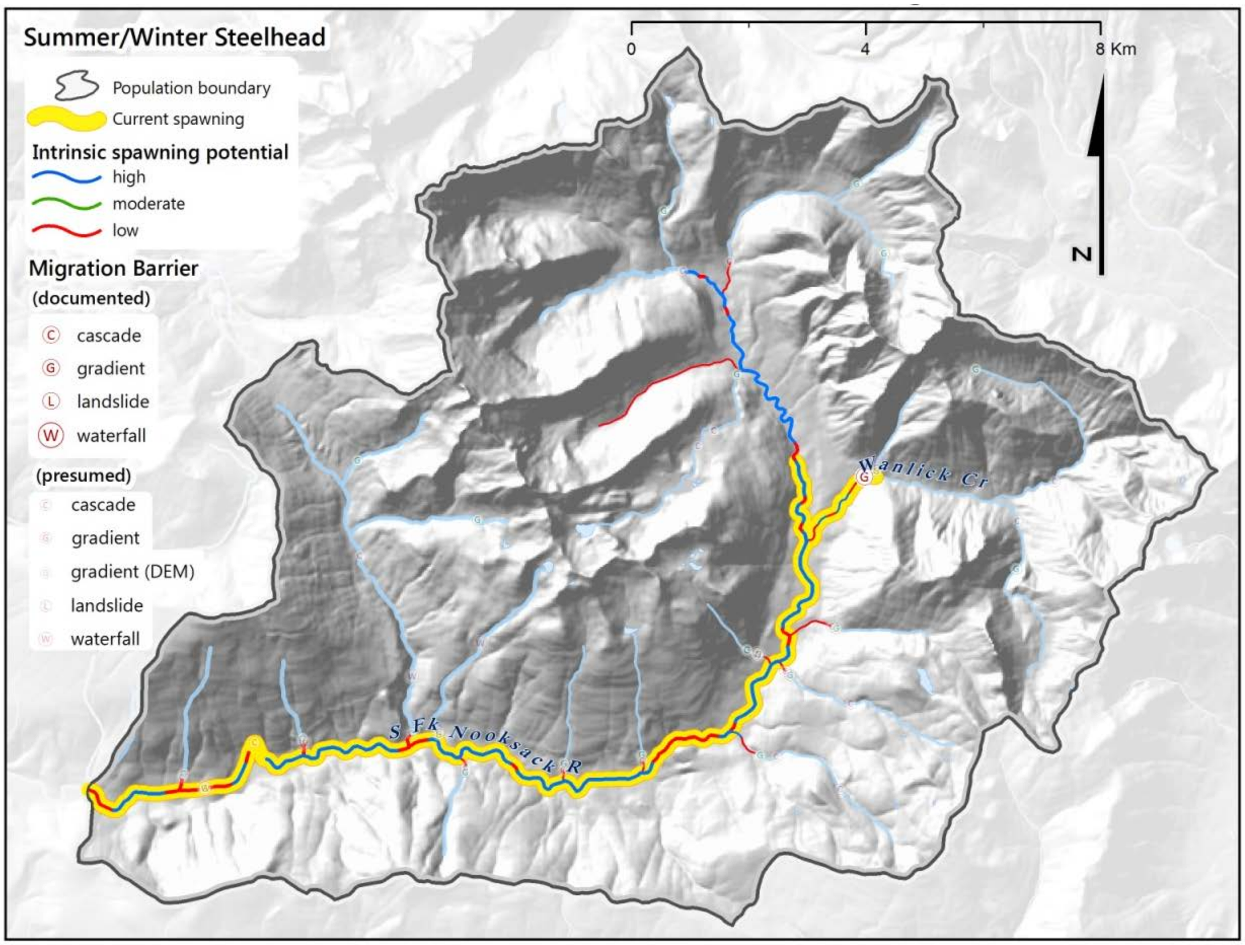

Figure E-8. Map of South Fork Nooksack River Summer-Run DIP spatial structure, including migration barriers and spawning potential.

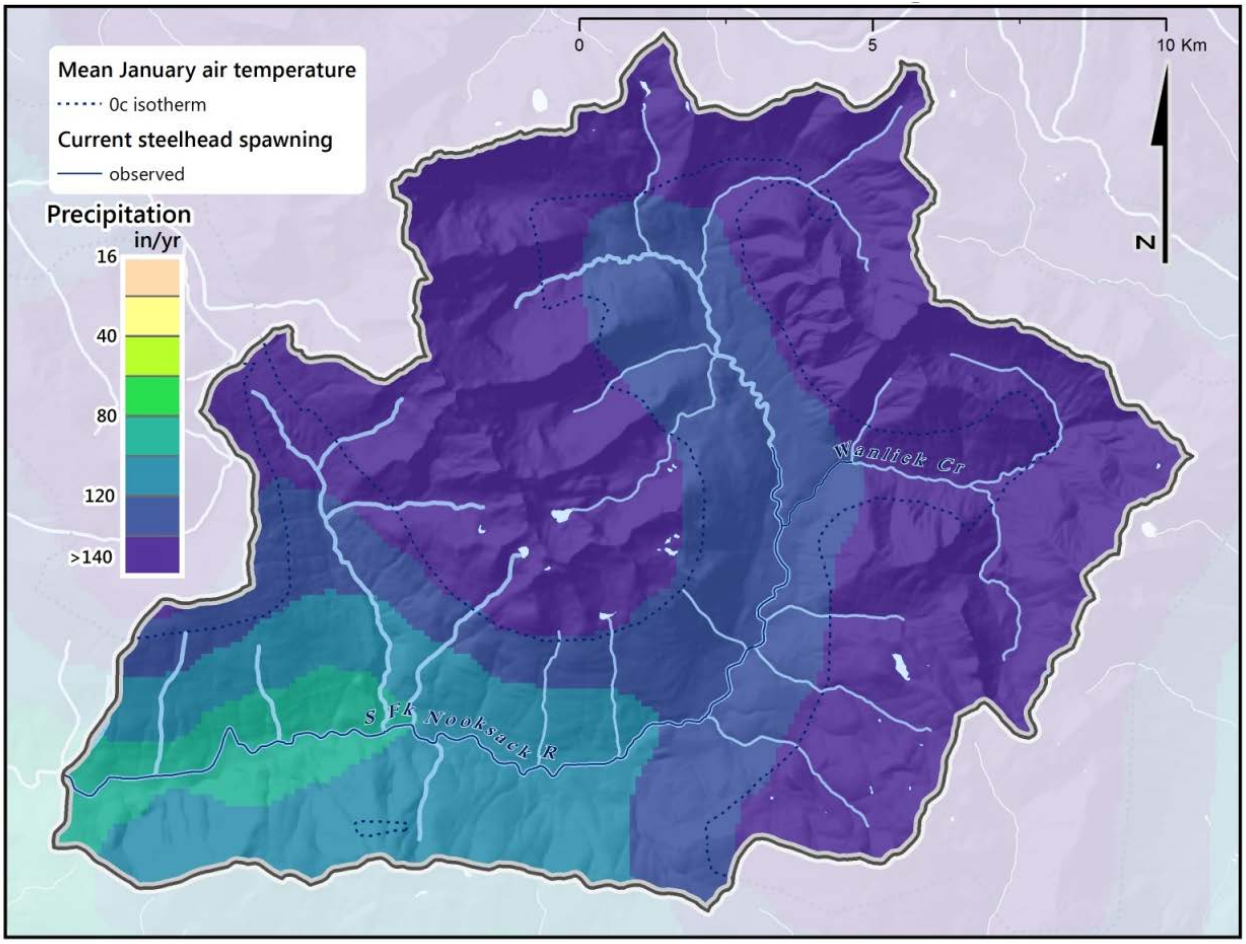

Figure E-9. Map of South Fork Nooksack River Summer-Run DIP area hydrology.

4. Samish River and Bellingham Bay Tributaries Winter Run

Overview—This population of steelhead (Figure E-10) exists in an independent tributary to Puget Sound. The Samish River drains into Samish and Bellingham bays, whereas the headwaters are more closely associated with the Skagit River basin. The majority of historical and present production is in the Samish Basin, with lesser contribution from the Bellingham Bay tributaries.

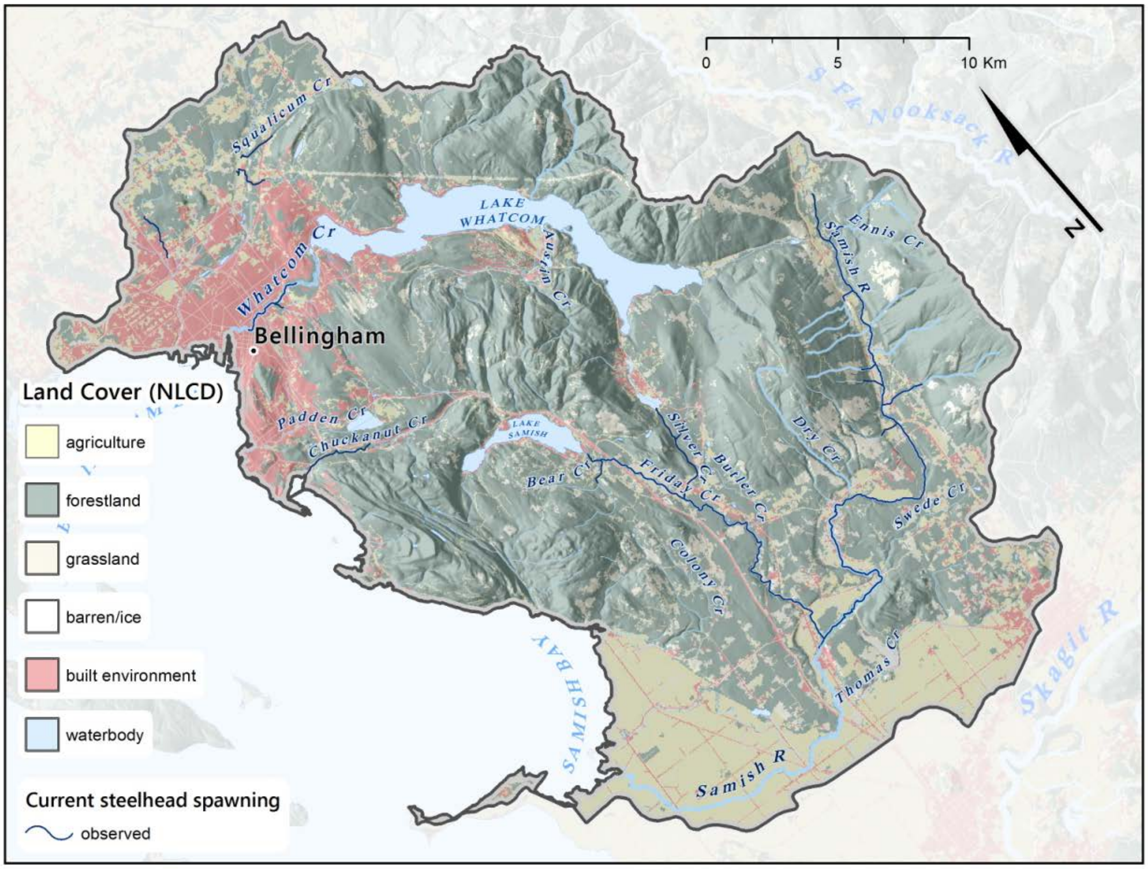

Figure E-10. Map of Samish River and Bellingham Bay Tributaries Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Not surveyed	1,934	534 (389–732)	3,193	6,386
Year	1930	1951	2005–2009	—	

The Samish River and Bellingham Bay Tributaries Winter-Run DIP's exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 1.008 (0.972–1.045).

For a state-space analysis of population viability, see the top two plots of Figure A-1 in Appendix A.

Diversity—

Samish River and Bellingham Bay Tributaries Winter Run				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Samish River	1995–2008	10	Whatcom R.Tokul (CC)	0	324,073
Whatcom Cr.	1995–2009	12	Whatcom H. (CC)	0	170,508
			Total:	0	494,581

Spatial structure and hydrology—Figure E-11 and Figure E-12.

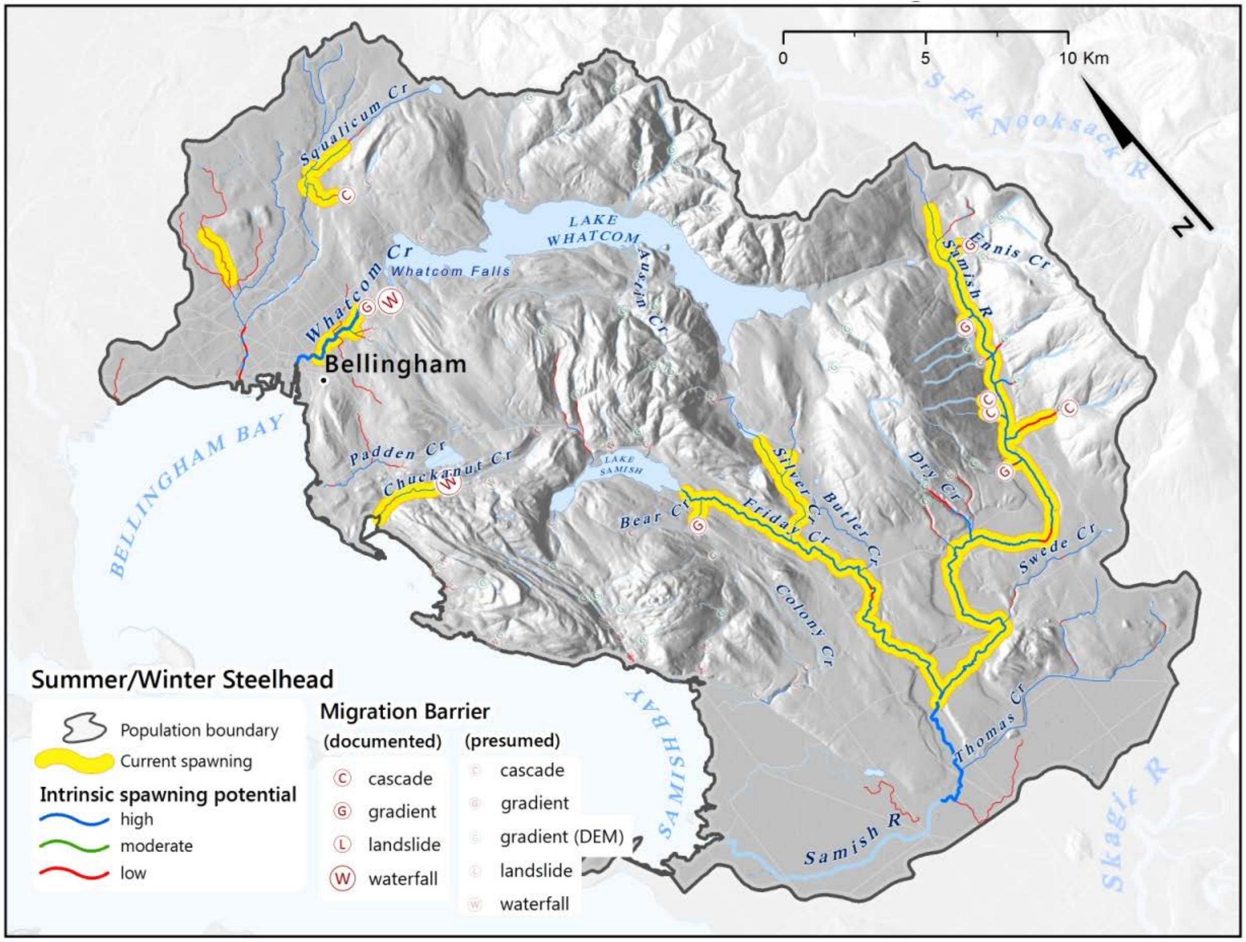

Figure E-11. Map of Samish River and Bellingham Bay Tributaries Winter-Run DIP spatial structure, including migration barriers and spawning potential.

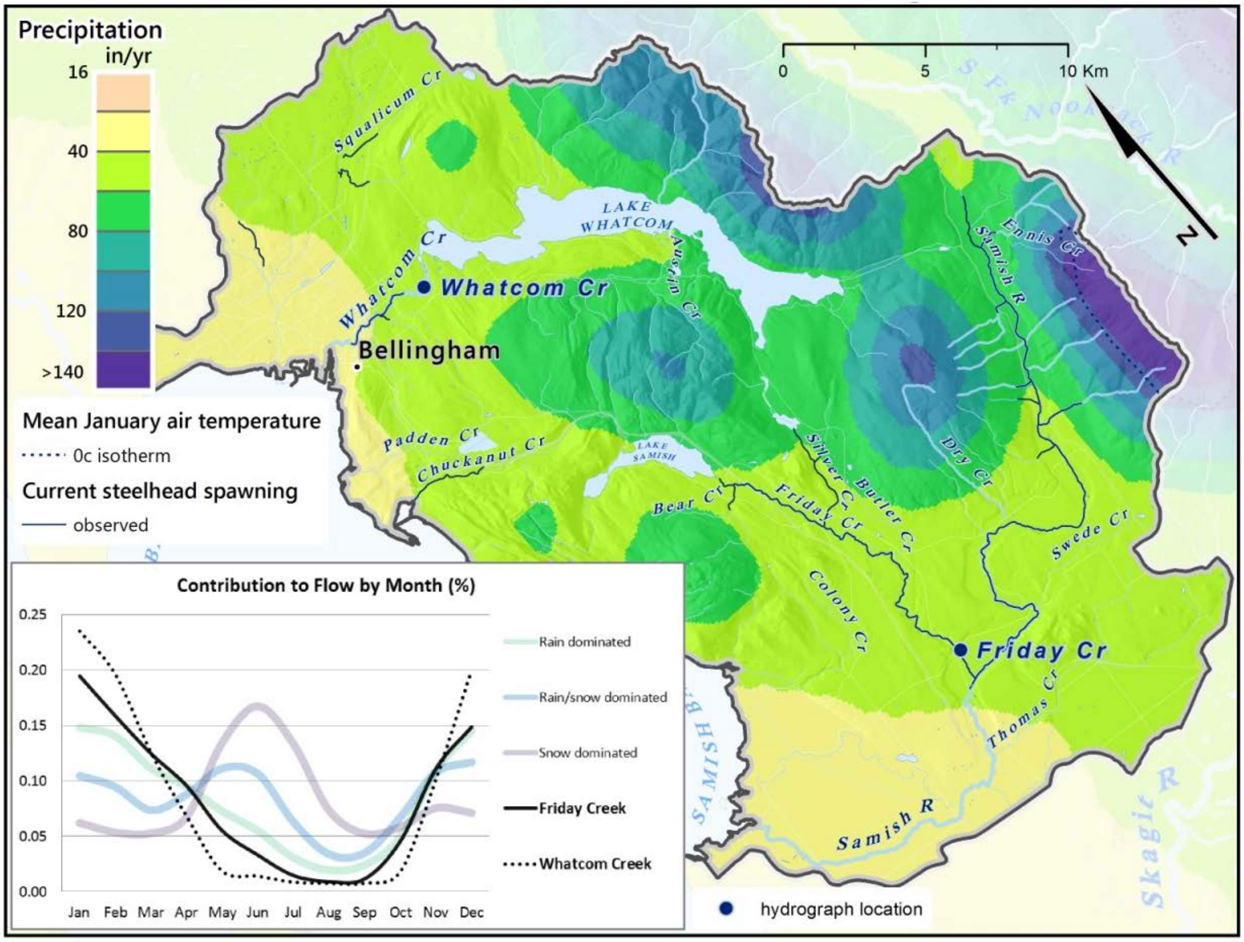

Figure E-12. Map of Samish River and Bellingham Bay Tributaries Winter-Run DIP area hydrology.

5. Skagit River Summer Run and Winter Run

Overview—The population (Figure E-13) includes the mainstem Skagit and numerous tributaries. Major tributaries include: Cascade River, Finney Creek, Day Creek, Bacon Creek, and Illabot Creek. Historically, this DIP was likely one of the primary producers of summer-run and winter-run steelhead in Puget Sound.

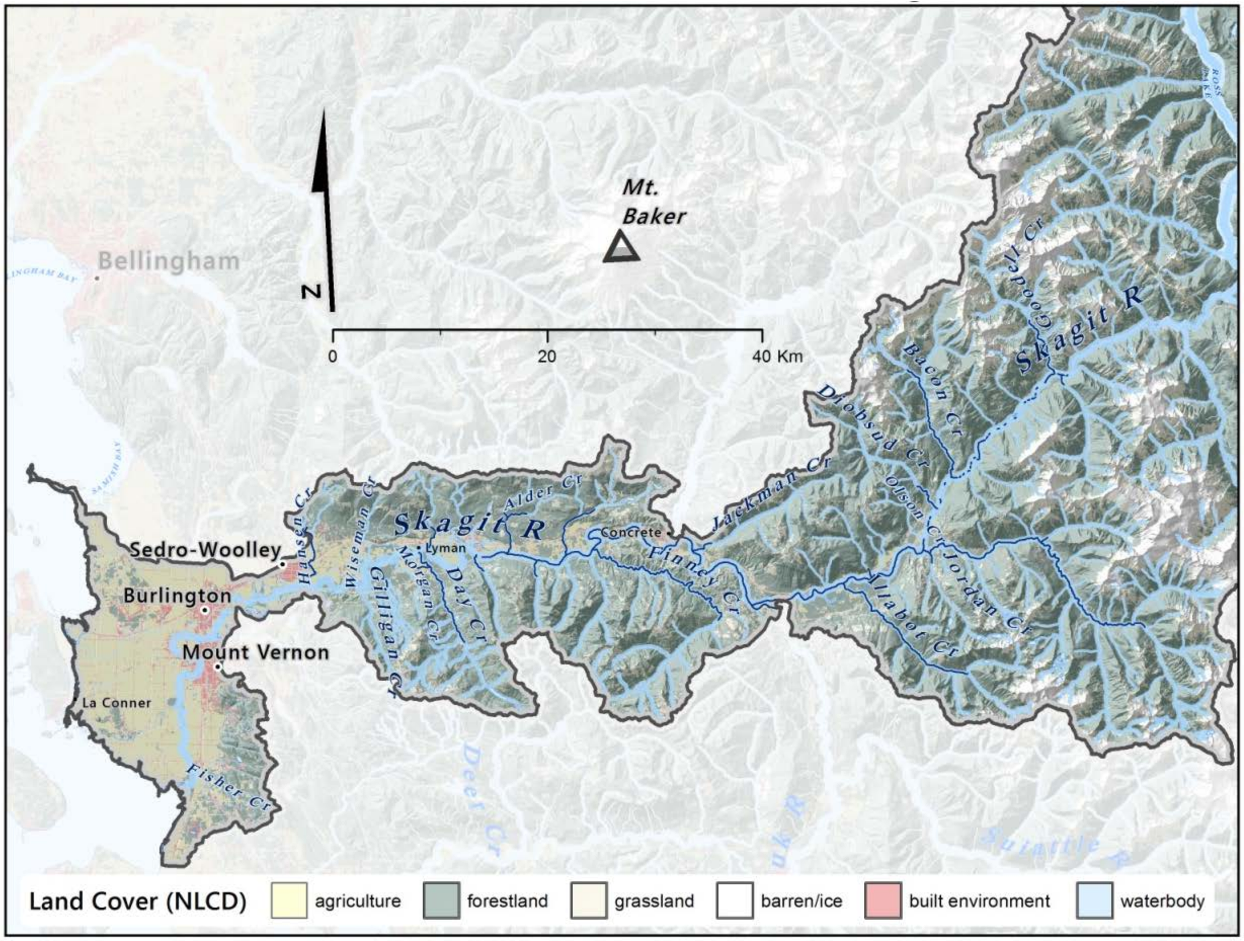

Figure E-13. Map of Skagit River Summer-Run and Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate
Abundance	Multiple large and medium	8,022	4,648 (2,827–7,642)	64,775 129,551
Year	1930	1950	2005–2009	—

The Skagit River Summer-Run and Winter-Run DIP’s exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 0.969 (0.954–0.985).

For a state-space analysis of population viability, see the top two plots of Figure A-2 in Appendix A.

Diversity—

Skagit River Summer Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Skagit River	1995–1998	3	Stillaguamish H.	0	71,256
			Total:	0	71,256

Skagit River Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Hamilton Slough	1997–1999	2	Barnaby Slough (CC)	0	38,350
Barnaby Slough	1996–2008	12	Barnaby Slough (CC)	0	1,911,179
Skagit River	1995–2001	5	Bogachiel/Skagit (CC)	0	747,392
Cascade River	1995–2010	16	Marblemount H. (CC)	0	2,576,846
Grandy Creek	1996–2002	6	Barnaby/Marblemount	0	324,461
			Total:	0	5,598,228

Spatial structure and hydrology—Figure E-14 and Figure E-15.

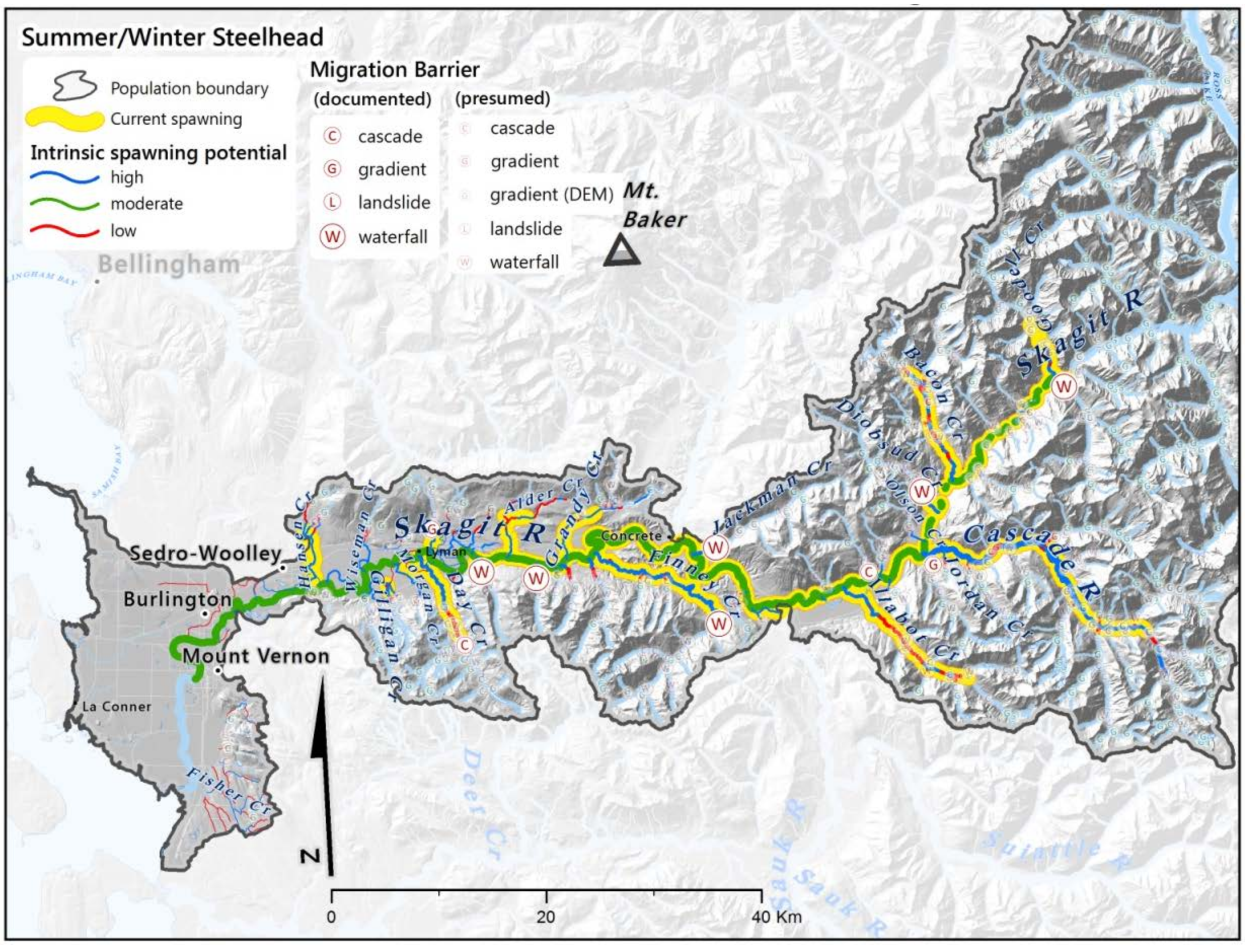

Figure E-14. Map of Skagit River Summer-Run and Winter-Run DIP spatial structure, including migration barriers and spawning potential.

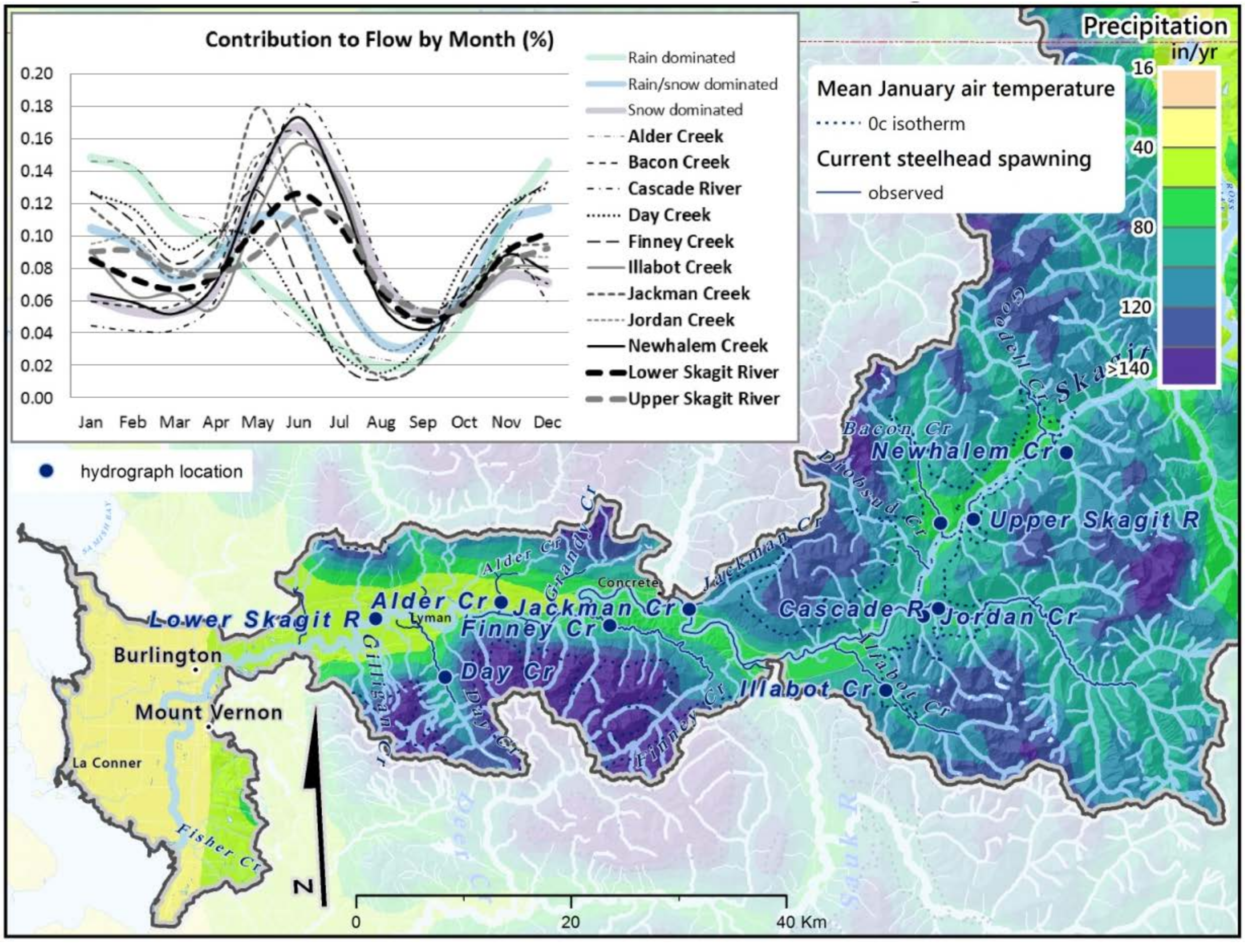

Figure E-15. Map of Skagit River Summer-Run and Winter-Run DIP area hydrology.

6. Nookachamps Creek Winter Run

Overview—Nookachamps Creek (Figure E-16) was identified as a potential DIP for winter steelhead. This basin met the criteria for basin size and IP production. Very little information is available for this basin other than anecdotal reports of steelhead abundance. Spawning and rearing habitat is primarily in the forks, not the main stem.

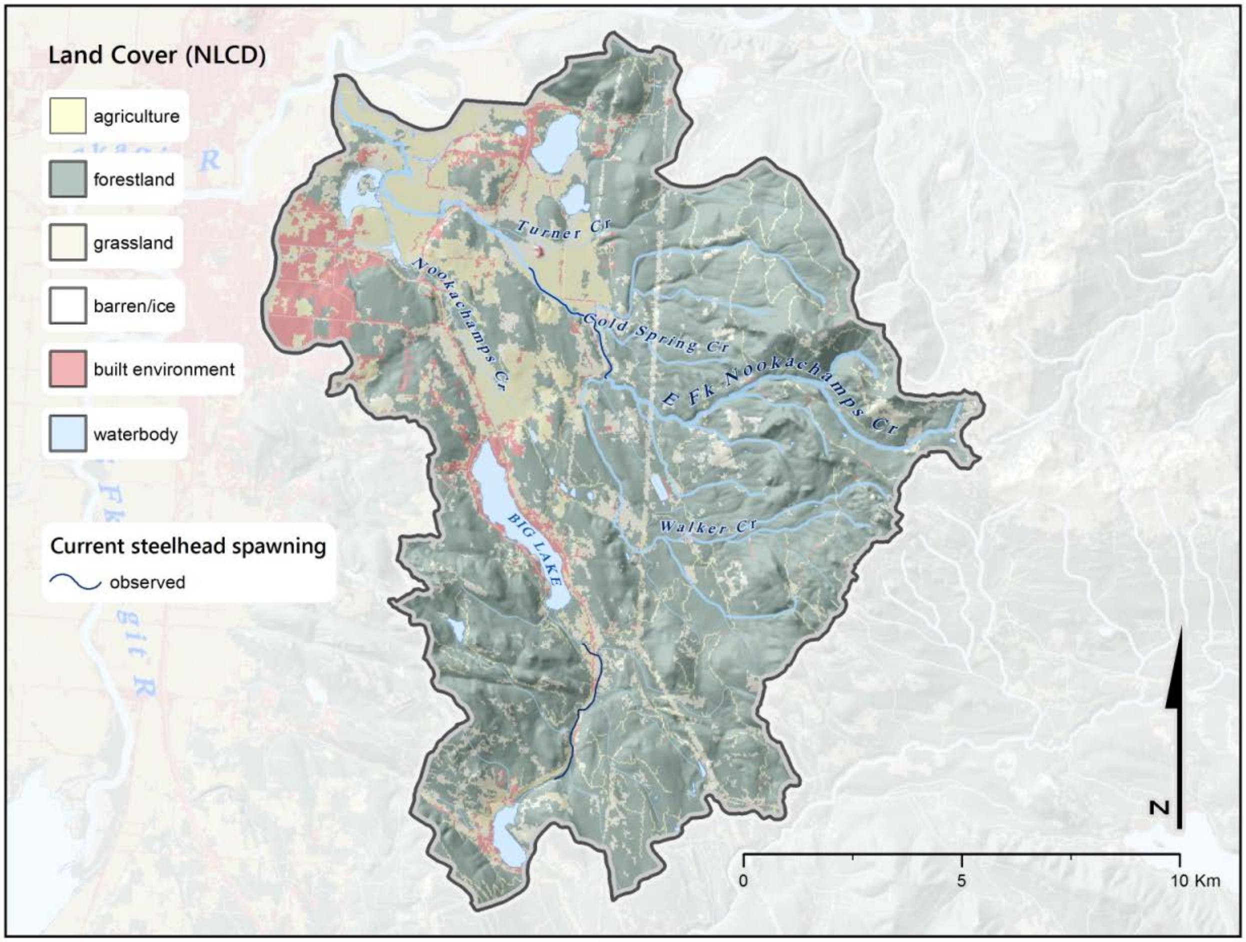

Figure E-16. Map of Nookachamps Creek Winter-Run DIP land cover and spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate	
Abundance	Scarce	NA	1,231	2,462
Year	1930	NA	—	

Diversity—Hatchery releases for the Nookachamps Creek Winter Run are zero within the DPS and zero outside the DPS.

Spatial structure and hydrology—Figure E-17 and Figure E-18.

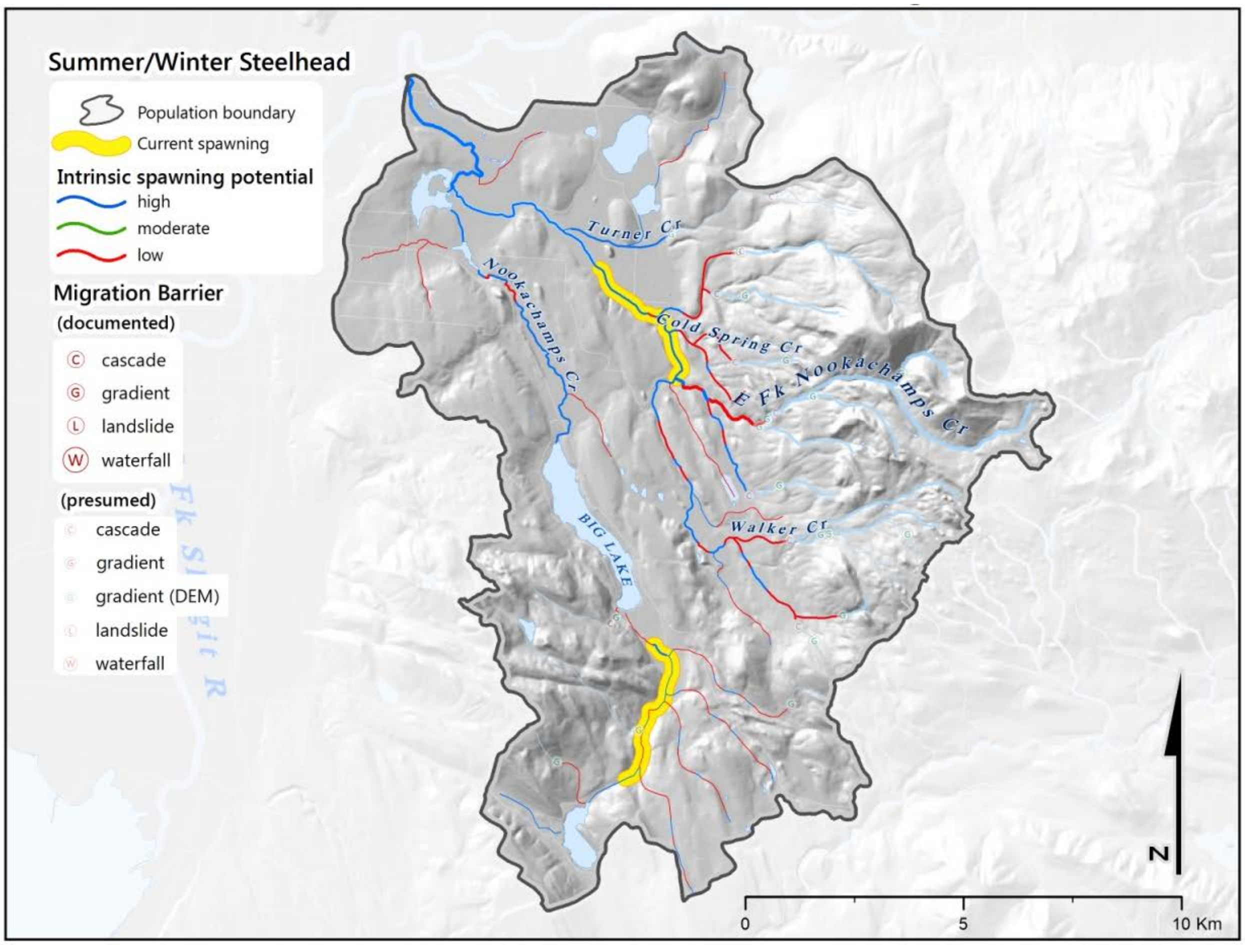

Figure E-17. Map of Nookachamps Creek Winter-Run DIP spatial structure, including migration barriers and spawning potential.

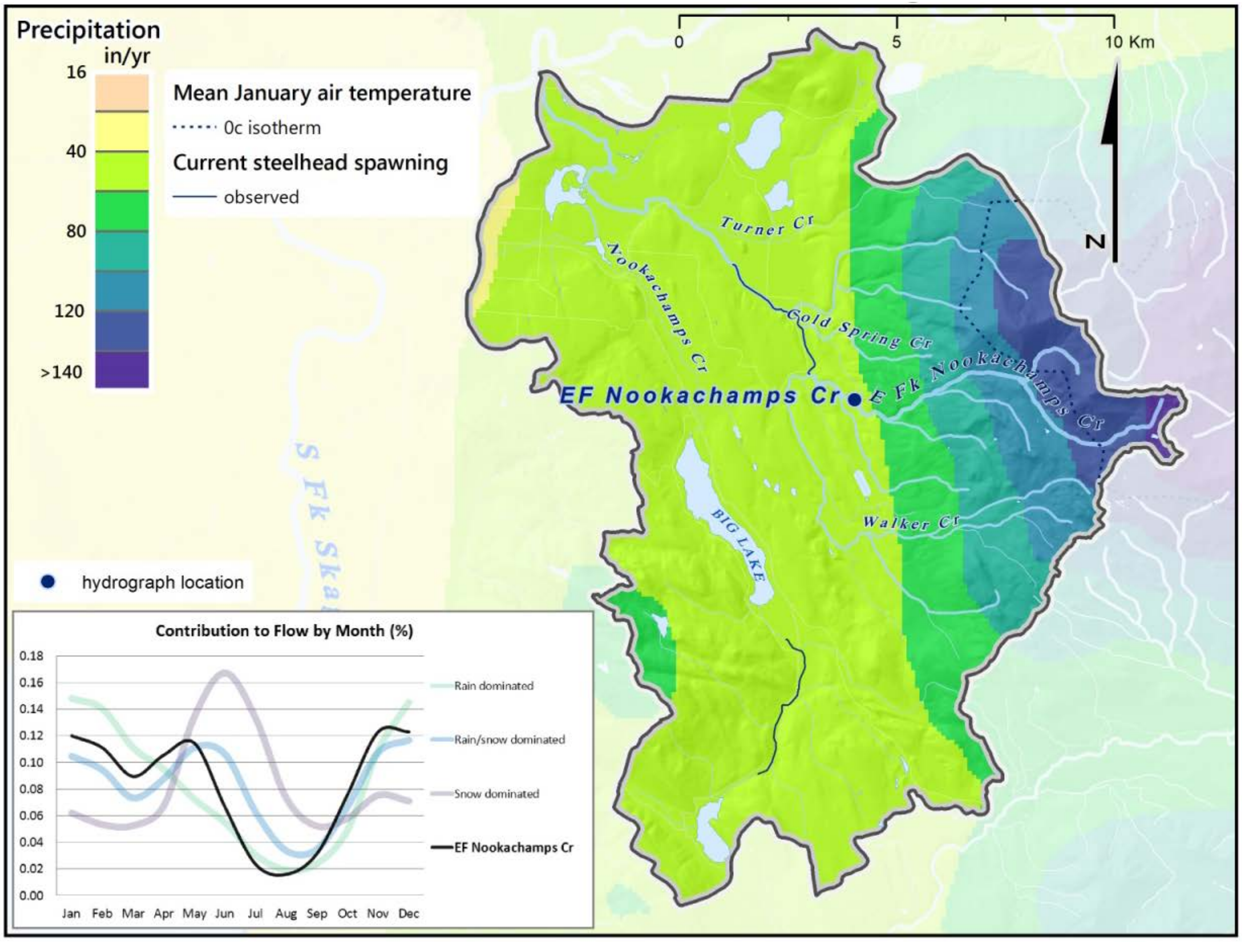

Figure E-18. Map of Nookachamps Creek Winter-Run DIP area hydrology.

7. Baker River Summer Run and Winter Run

Overview—Historically, Baker River (Figure E-19) was likely a major contributor to Skagit River steelhead runs. Baker River is the second largest tributary to the Skagit River, with a basin size of 771 km^2. Access to historical spawning grounds is blocked by the upper and lower Baker dams. Trap and haul operations currently provide the only access to headwater regions.

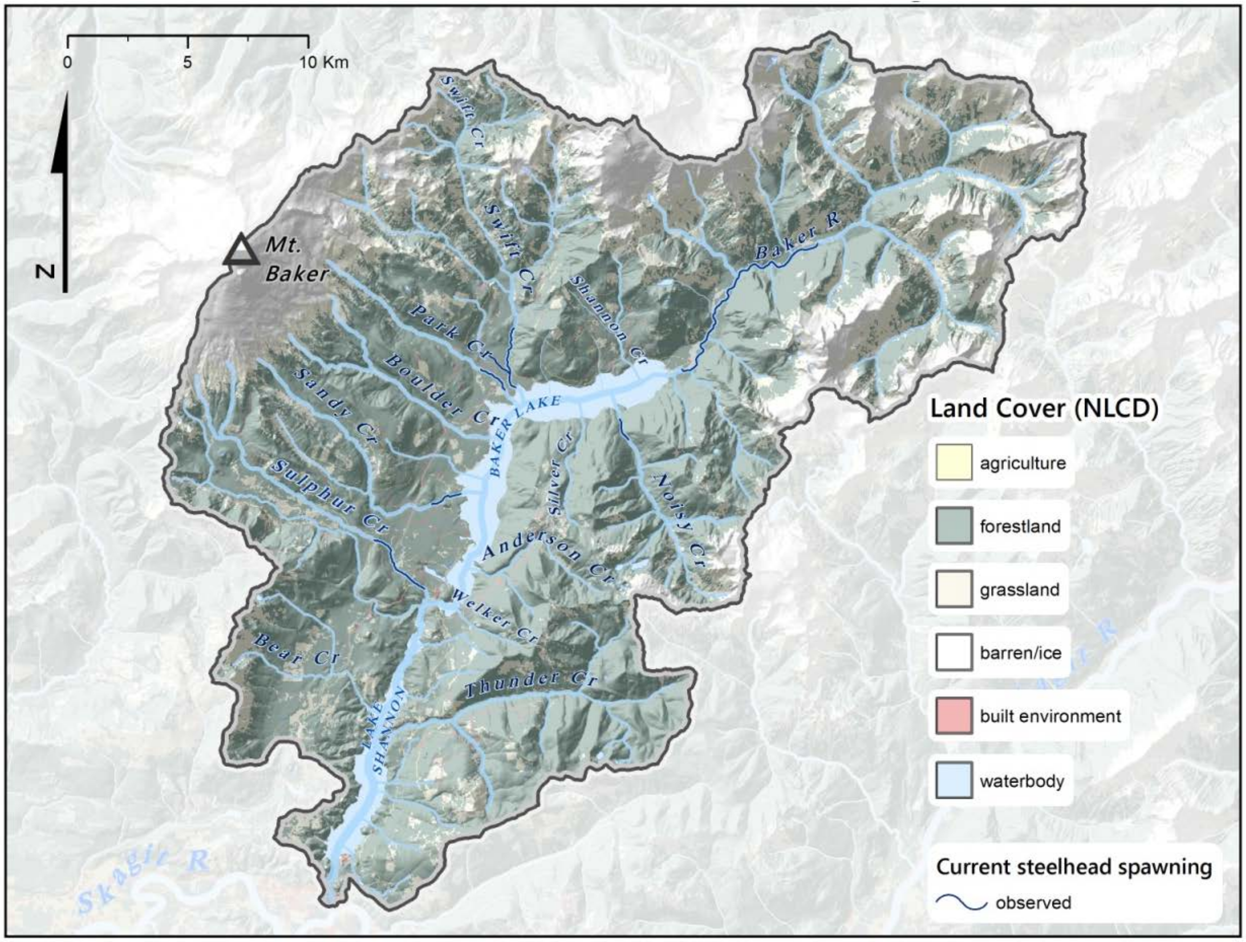

Figure E-19. Map of Baker River Summer-Run and Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate	
Abundance	Medium	NA	5,028	10,056
Year	1930	—	—	

Diversity—

Baker River Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Baker River	1995–2010	14	Chambers/Baker/Skagit	0	679,322
			Total:	0	679,322

Spatial structure and hydrology—Figure E-20 and Figure E-21.

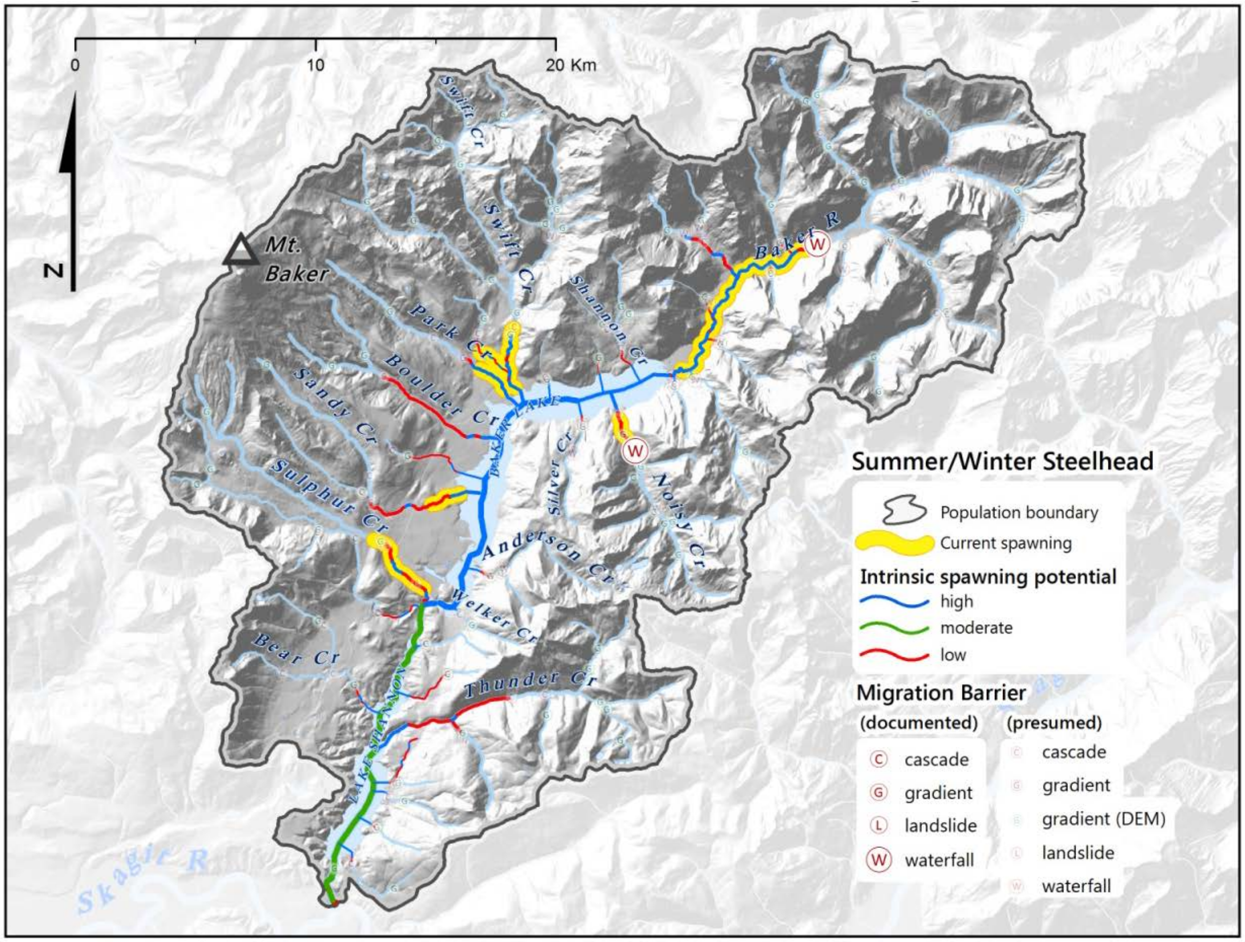

Figure E-20. Map of Baker River Summer-Run and Winter-Run DIP spatial structure, including migration barriers and spawning potential.

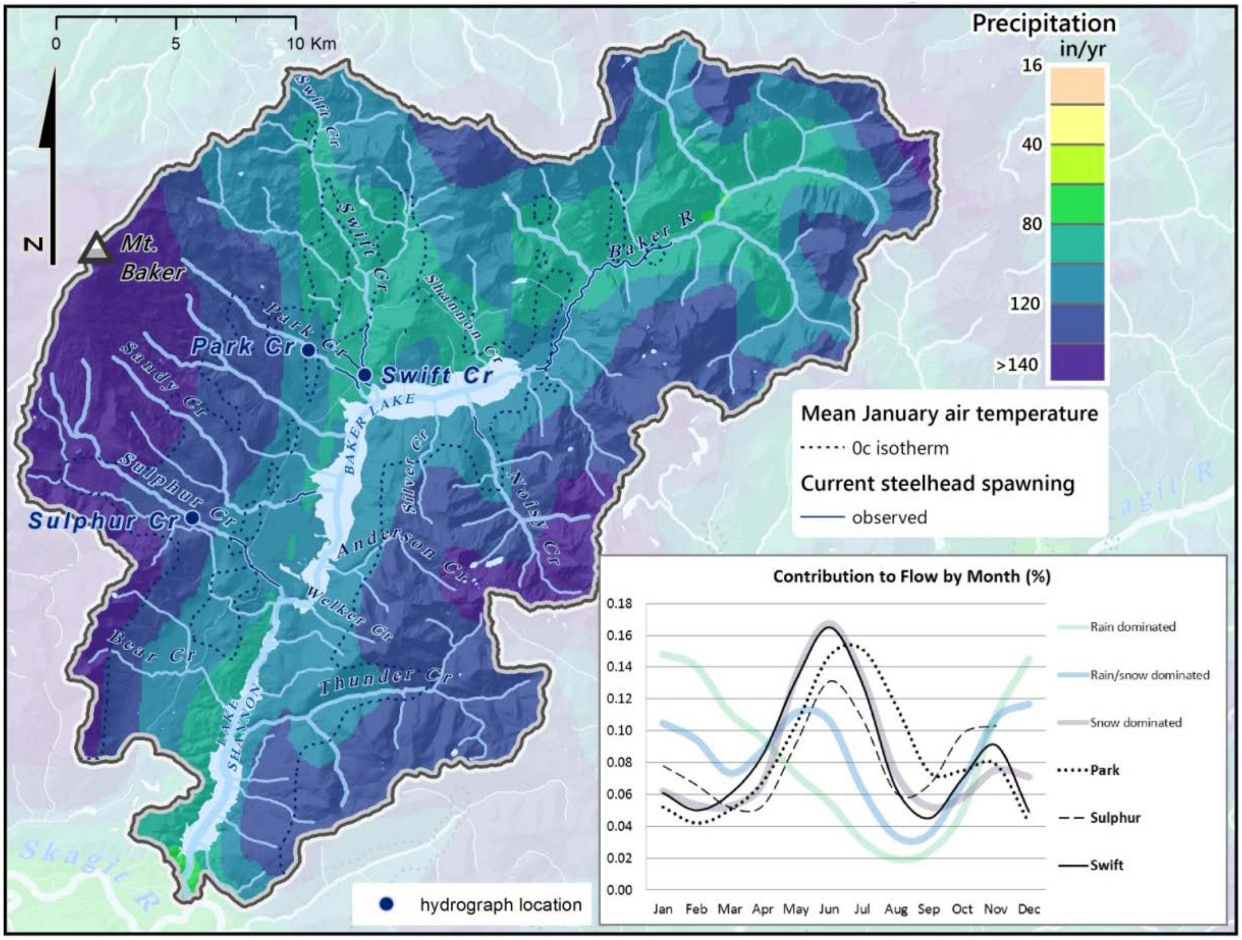

Figure E-21. Map of Baker River Summer-Run and Winter-Run DIP area hydrology.

8. Sauk River Summer Run and Winter Run

Overview—This population (Figure E-22) includes both winter-run and summer-run steelhead in the Sauk River and its tributaries. Much of the basin consists of higher gradient reaches and some tributaries are glacially influenced. Historically important, recent returns have been relatively poor.

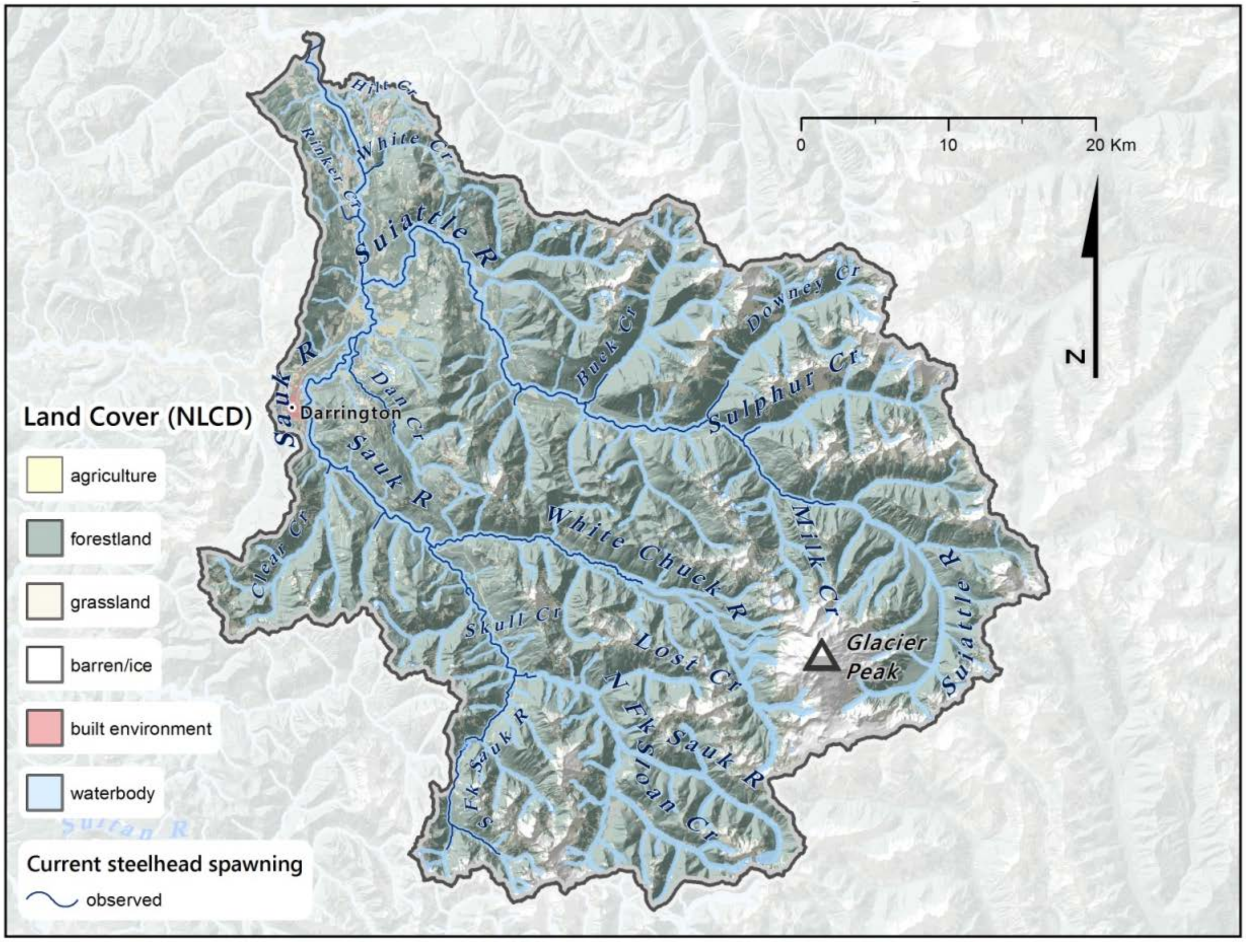

Figure E-22. Map of Sauk River Summer-Run and Winter-Run DIP land cover and spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Multiple medium	679	2,541	23,230	46,460
Year	1930	1951	2004–2006	—	

Tributary	1997	1998	1999	2000	2001	2002	2003	2004	2005	2011
Sauk (summer)	—	—	—	—	—	—	—	—	—	—
Sauk (winter)	—	—	—	—	—	—	—	2,726	1,962	3,068

Diversity—

Sauk River Summer Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
—	—	—	—	—	—
			Total:	0	0

Sauk River Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Sauk River	1995–2008	14	Skagit/Stillaguamish (CC)	0	323,918
			Total:	0	323,918

Spatial structure and hydrology—Figure E-23 and Figure E-24.

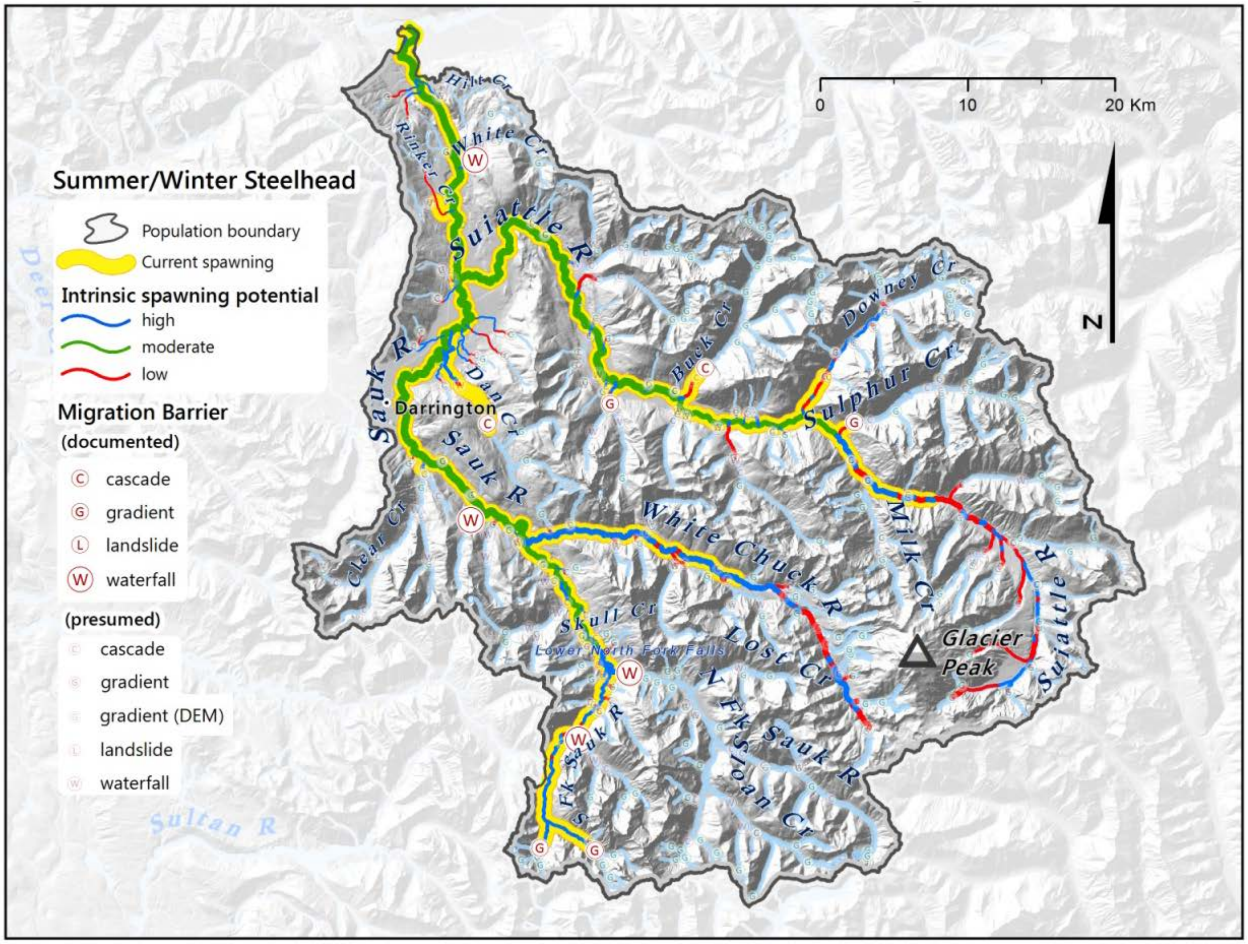

Figure E-23. Map of Sauk River Summer-Run and Winter-Run DIP spatial structure, including migration barriers and spawning potential.

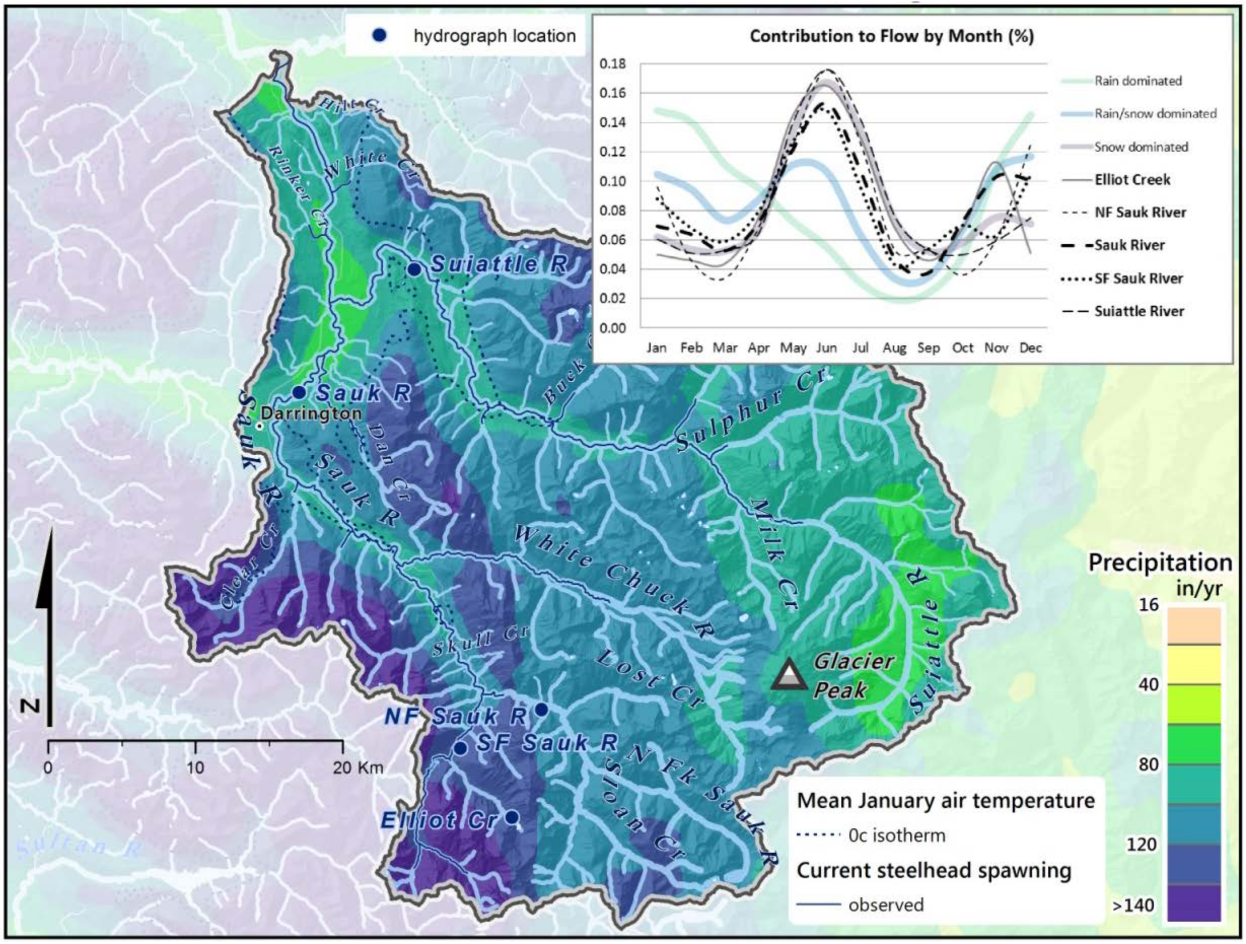

Figure E-24. Map of Sauk River Summer-Run and Winter-Run DIP area hydrology.

9. Stillaguamish River Winter Run

Overview—Winter-run steelhead (Figure E-25) spawn in the mainstem north fork and mainstem south fork of the Stillaguamish River and in numerous tributaries. Winter-run steelhead are considered distinct from the summer-run steelhead that spawn in Deer Creek and Canyon Creek above flow barriers (e.g., above RKM 0.8 in Canyon Creek) because of the likely geographic and temporal separation of spawners.

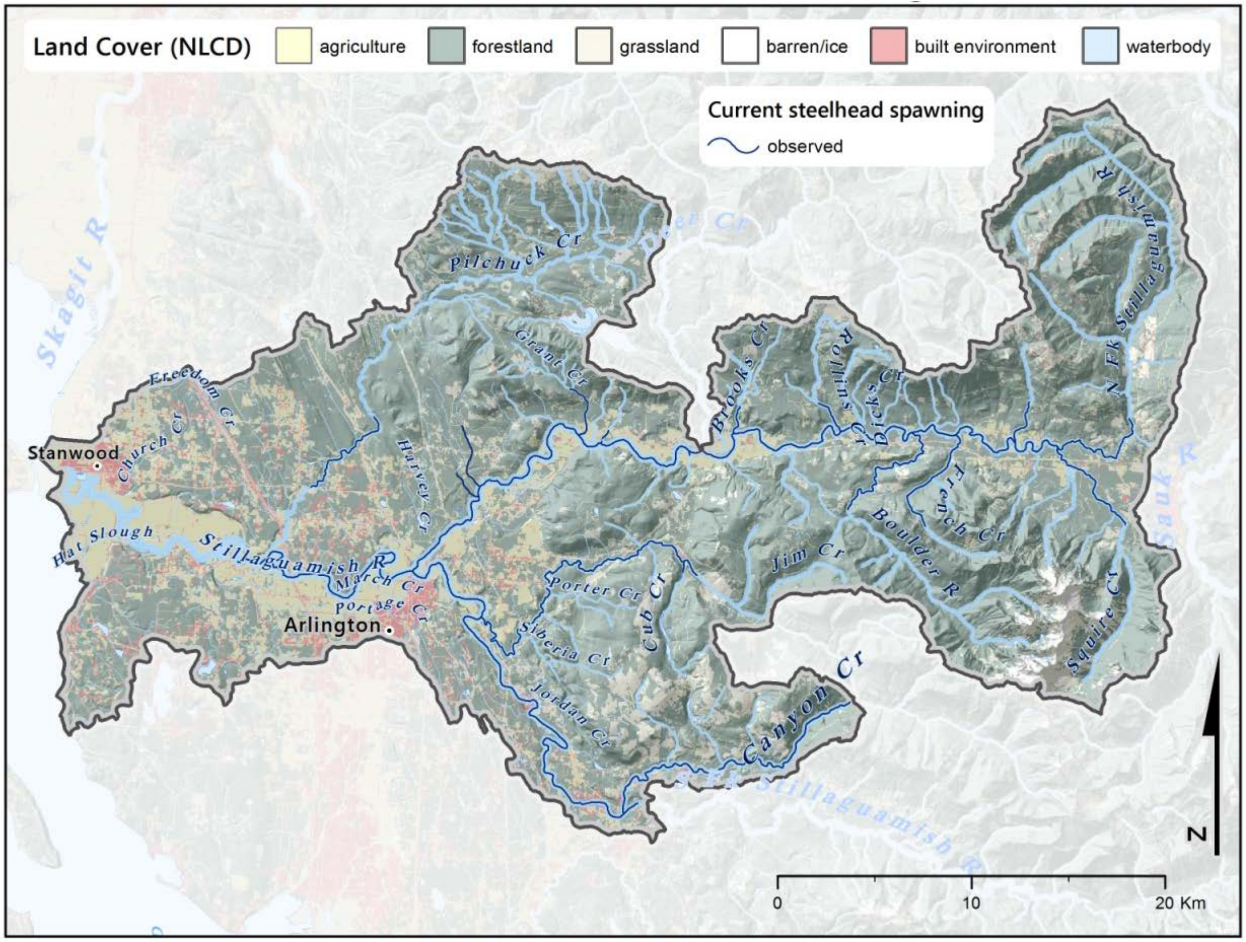

Figure E-25. Map of Stillaguamish River Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Large & medium	1,564	327 (100–1,067)	19,118	38,236
Year	1930	1954	2005–2009	—	

The Stillaguamish River Winter-Run DIP's exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 0.910 (0.887–0.934).

For a state-space analysis of population viability, see the top two plots of Figure A-3 in Appendix A.

Diversity—

Stillaguamish River summer run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Stillaguamish River	1998–2005	5	Snohomish Hatchery (SK)	0	178,558
North Fork Stillaguamish R.	1995–2011	16	Stillaguamish/ Snohomish H. (SK)	0	1,077,547
			Total:	0	1,256,105

Stillaguamish River Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Canyon Creek	1995–2003	9	Chambers/Tokul/Sky (CC)	0	132,365
Johnson Creek	1996–2001	3	Chambers/ Stillaguamish	0	35,267
Pilchuck Creek	1995–2009	12	Chambers/Snohomish (CC)	0	117,045
Stillaguamish River	1995–2011	17	Chambers/Bogachiel (CC)	0	2,048,991
			Total:	0	2,333,668

Spatial structure and hydrology—Figure E-26 and Figure E-27.

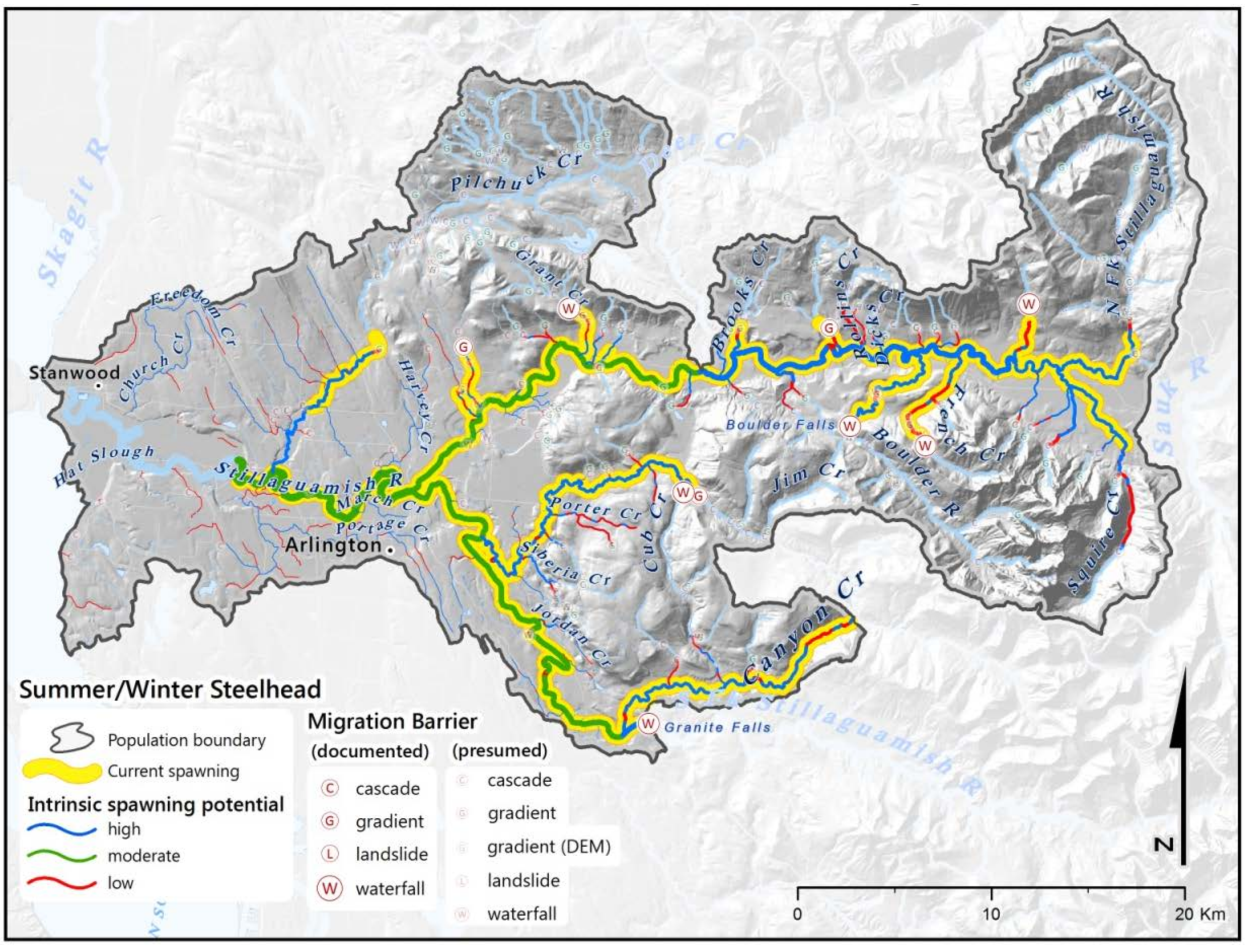

Figure E-26. Map of Stillaguamish River Winter-Run DIP spatial structure, including migration barriers and spawning potential.

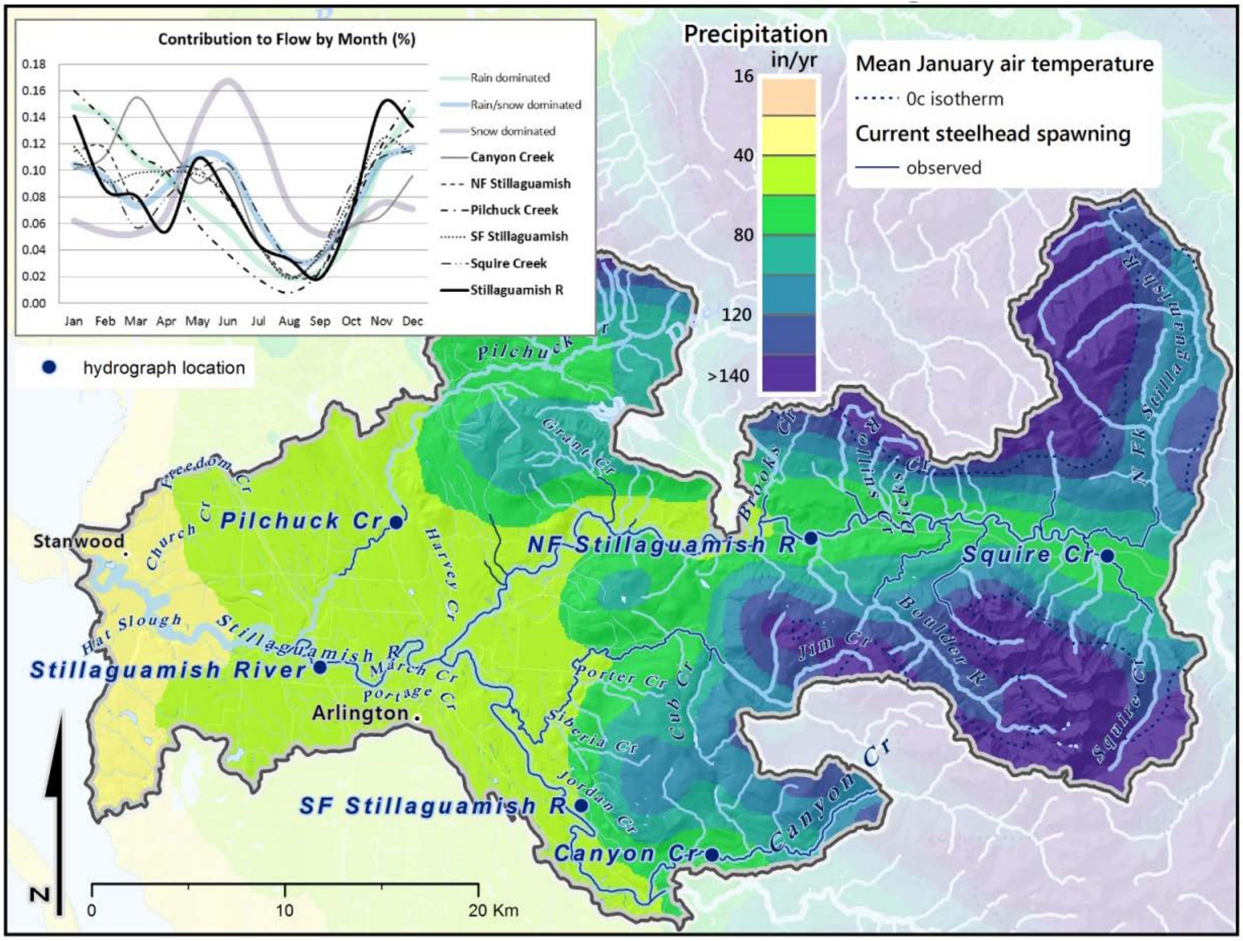

Figure E-27. Map of Stillaguamish River Winter-Run DIP area hydrology.

10. Deer Creek Summer Run

Overview—This steelhead population (Figure E-28) spawns and rears in the upper portion of Deer Creek. Steep canyons and cascades from RKM 2.5 to 8 may present a temporal barrier to returning winter-run fish, although Deer Creek is accessible to summer steelhead up to approximately RKM 32. Habitat below these canyons and cascades, below RKM 4, is considered winter-run steelhead habitat. Even under pristine conditions, the steelhead run into Deer Creek may not have been very large, potentially 1,000 to 2,000 adults (WSCC 1999), although the 1929 survey classified Deer Creek as a large population (WDFG 1932).

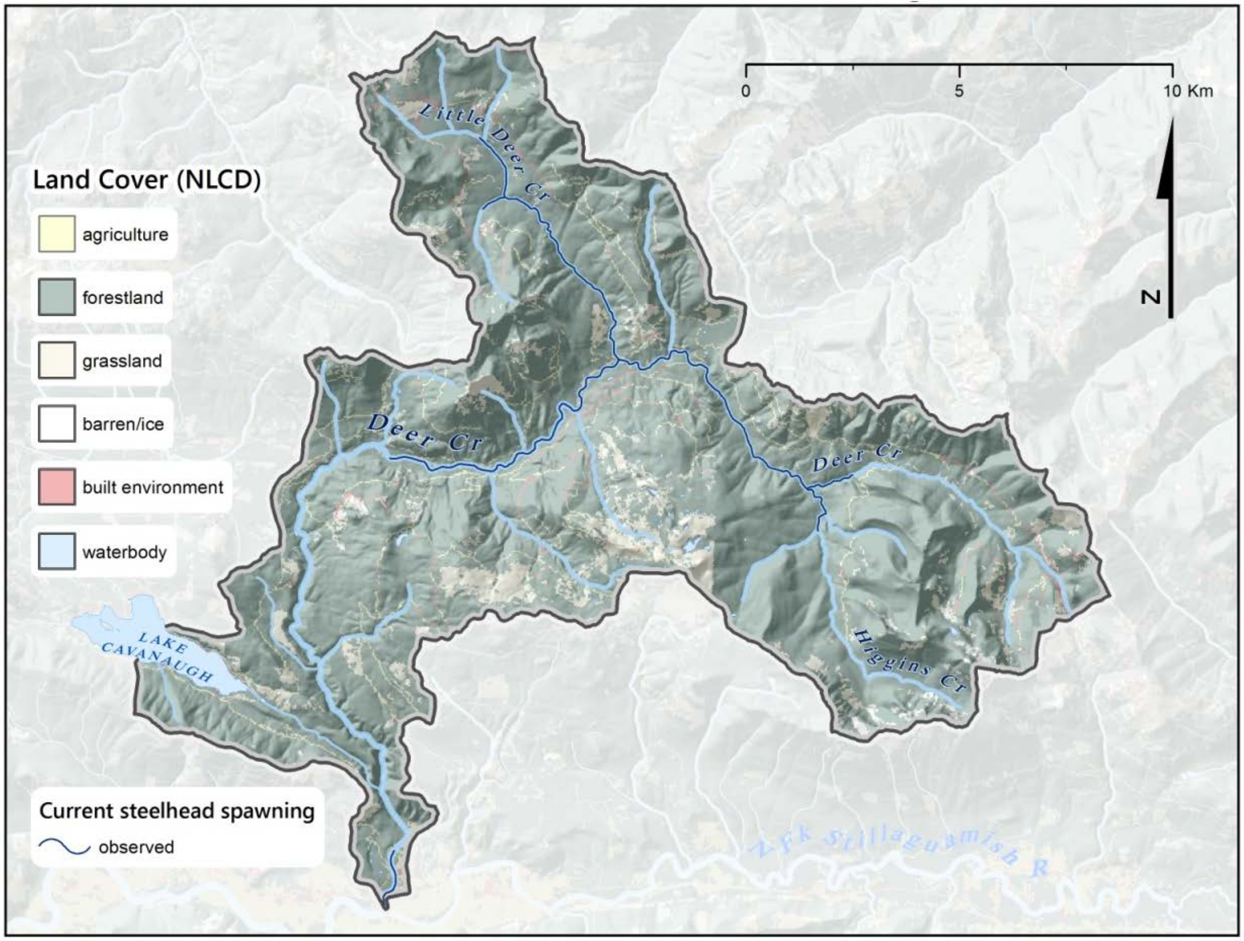

Figure E-28. Map of Deer Creek Summer-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Escapement	IP estimate	
Abundance	Large	NA	460	1,572	3,144
Year	1930	—	1994	—	

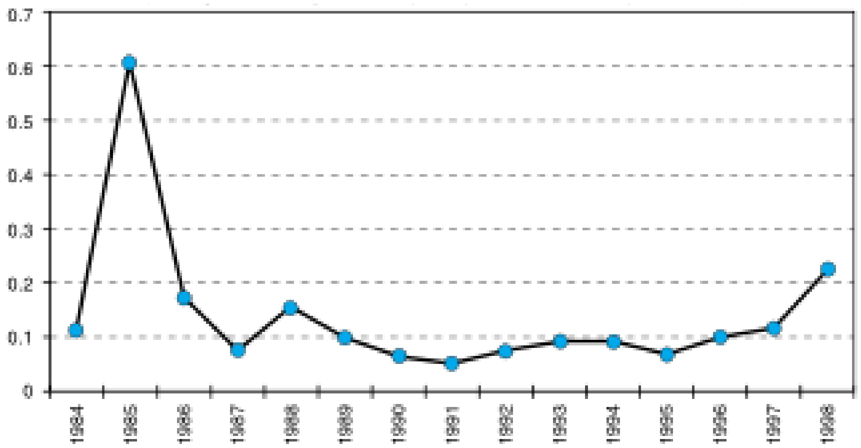

Figure E-29. Deer Creek steelhead parr density changes over time. Average annual age 1+ parr per square meter density for 14 sites. (Reprinted from Kraemer 1999.)

Diversity—Some Skamania Hatchery summer-run steelhead were released in the 1950s to 1970s, although there was thought to be little influence on the population. Hatchery releases for 1995 to the present are zero within the DPS and zero outside the DPS.

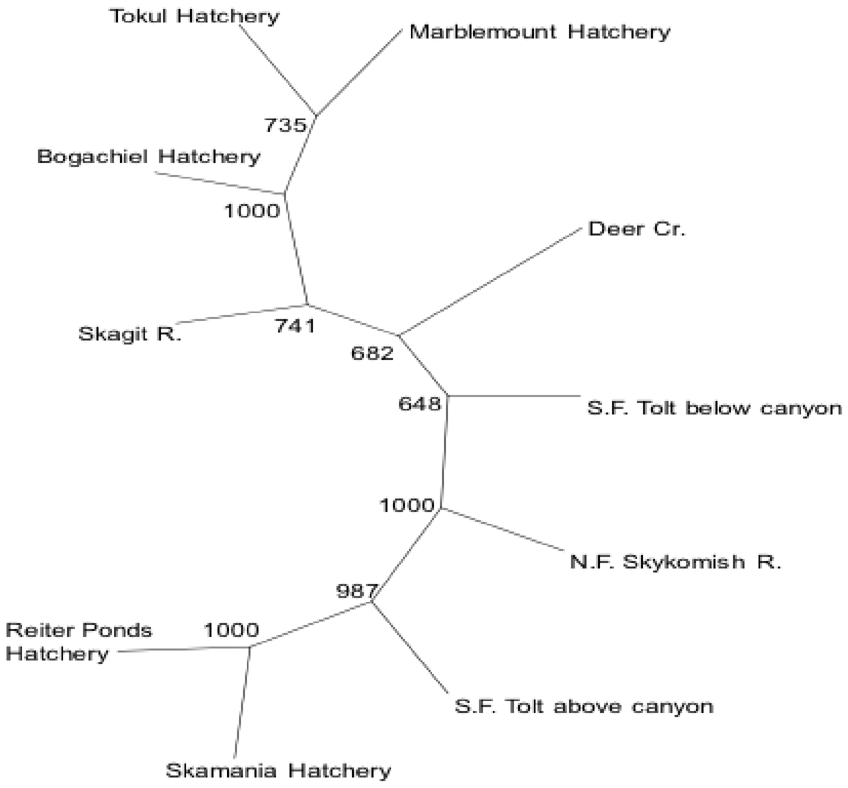

Figure E-30. Genetic relationship of steelhead collections from the Snohomish River basin, Puget Sound, and the Skamania Hatchery using Cavalli-Sforza and Edwards (1967) chord distance. Bootstrap values are shown at each node. (Reprinted from Kassler and Bell 2011.)

Spatial structure and hydrology—Figure E-31 and Figure E-32.

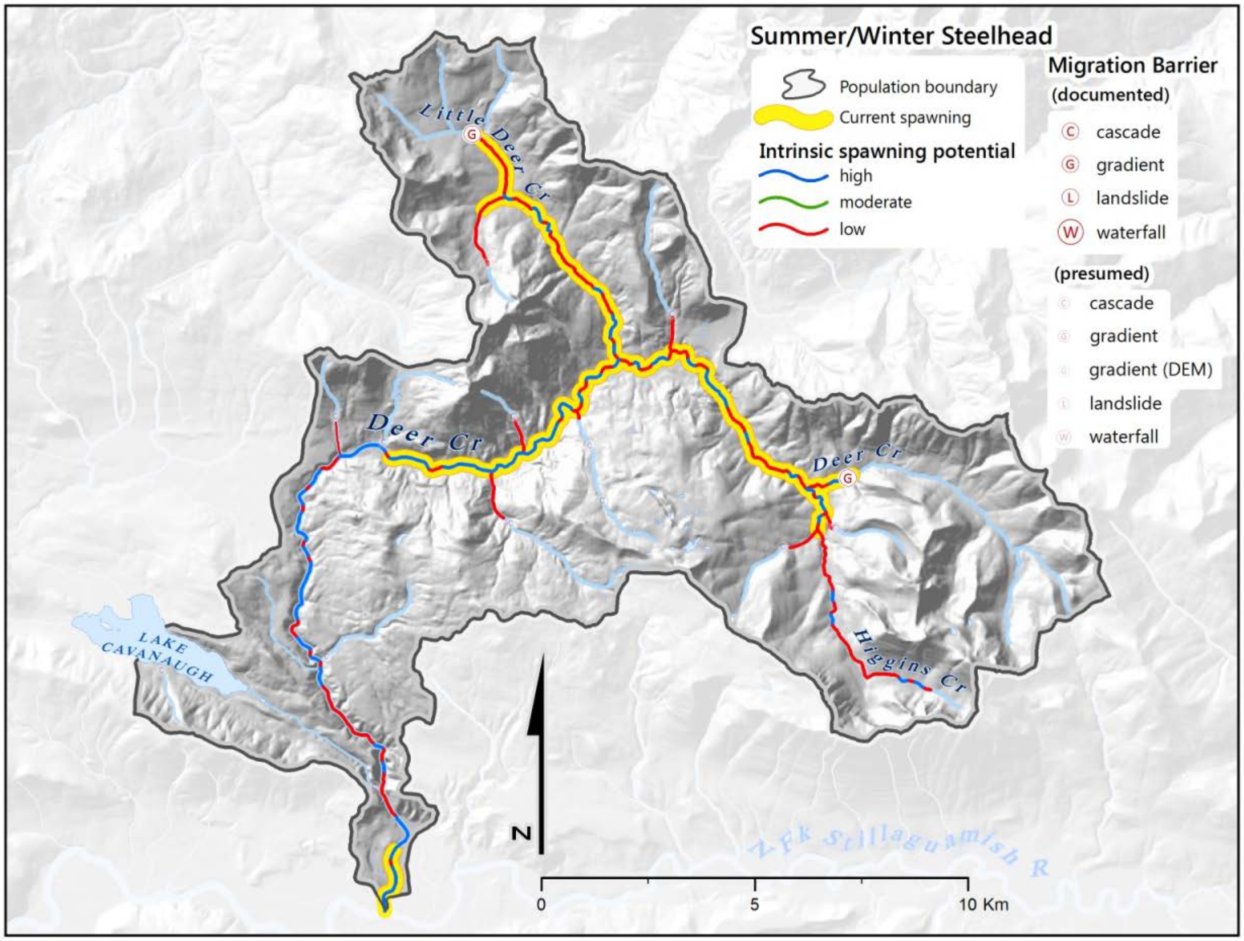

Figure E-31. Map of Deer Creek Summer-Run DIP spatial structure, including migration barriers and spawning potential.

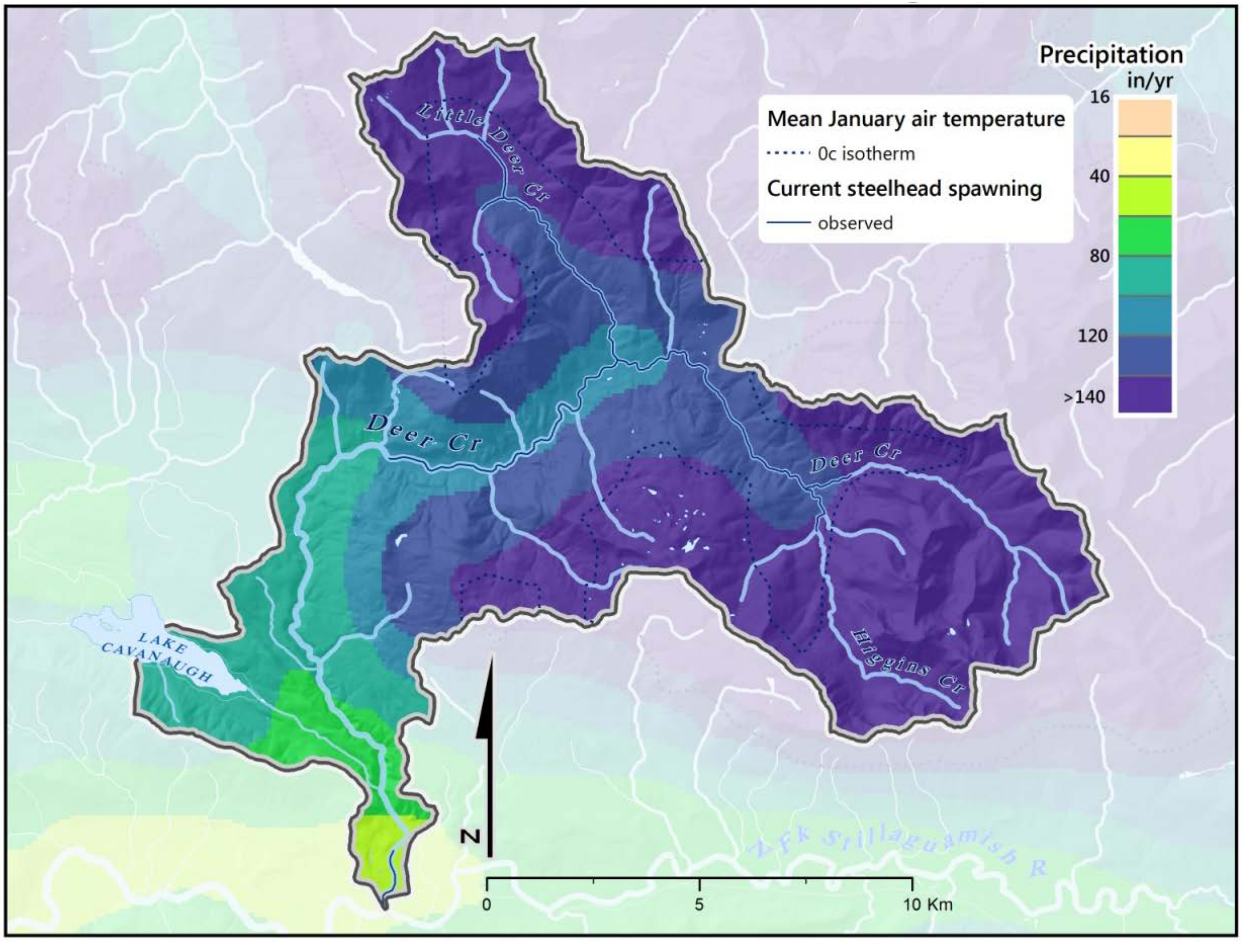

Figure E-32. Map of Deer Creek Summer-Run DIP area hydrology.

11. Canyon Creek Summer Run

Overview—There is relatively little information available on the existing summer run of steelhead in the Canyon Creek basin (Figure E-33). Information provided by local biologists indicates that a summer run is still present in the basin. Historically, Canyon Creek was identified as having a relatively good-sized run of steelhead.

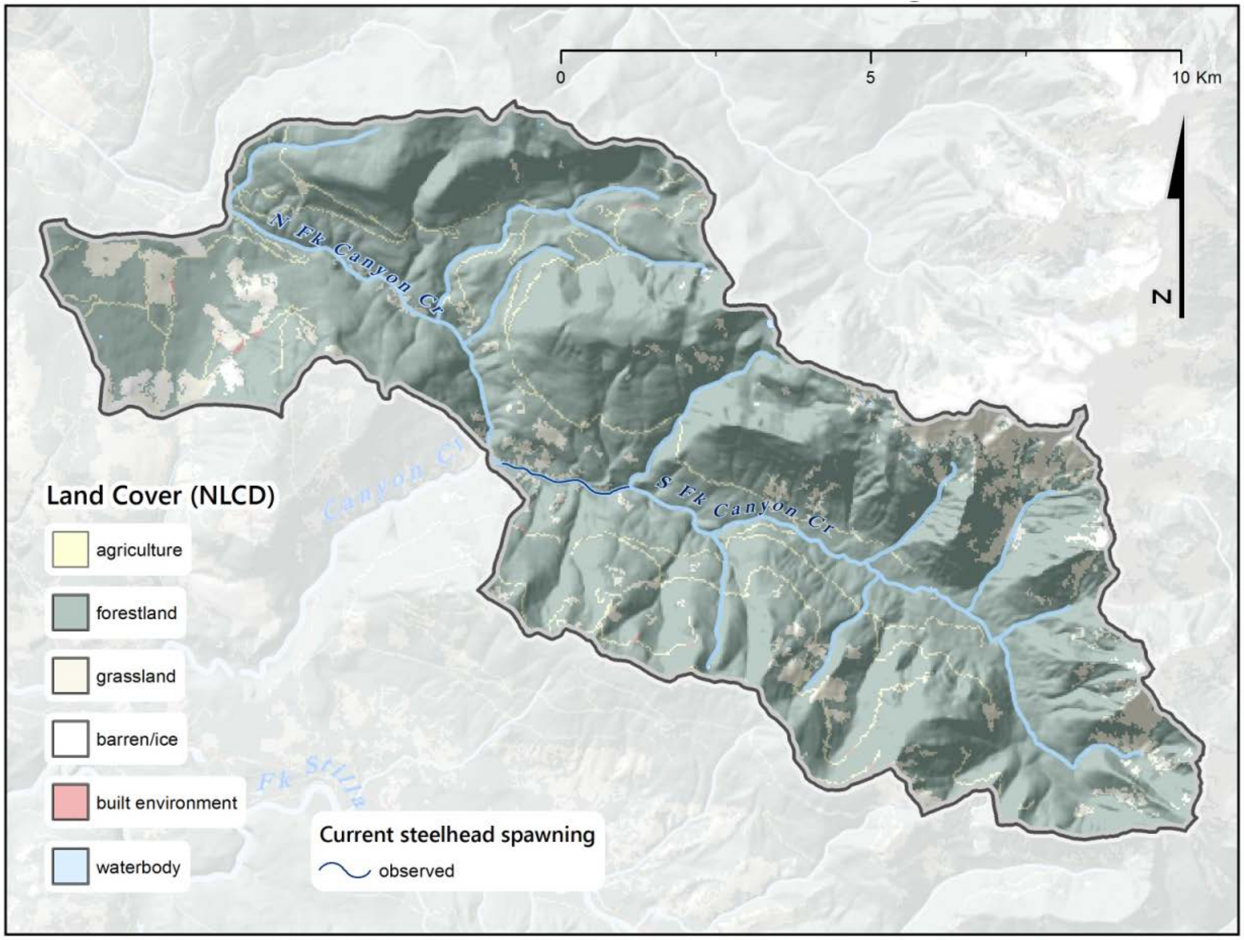

Figure E-33. Map of Canyon Creek Summer-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate	
Abundance	Large	283*	121	243
Year	1930	1951	—	

* Punch card–based catch estimates did not differentiate between run times prior to 1960.

Diversity—

Canyon Creek Summer Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Canyon Creek	1995–2008	5	Skykomish/Snohomish Hatchery (SK)	0	40,596
			Total:	0	40,596

Canyon Creek winter run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
—	—	—	—	—	—
			Total:	0	0

Spatial structure and hydrology—Figure E-34 and Figure E-35.

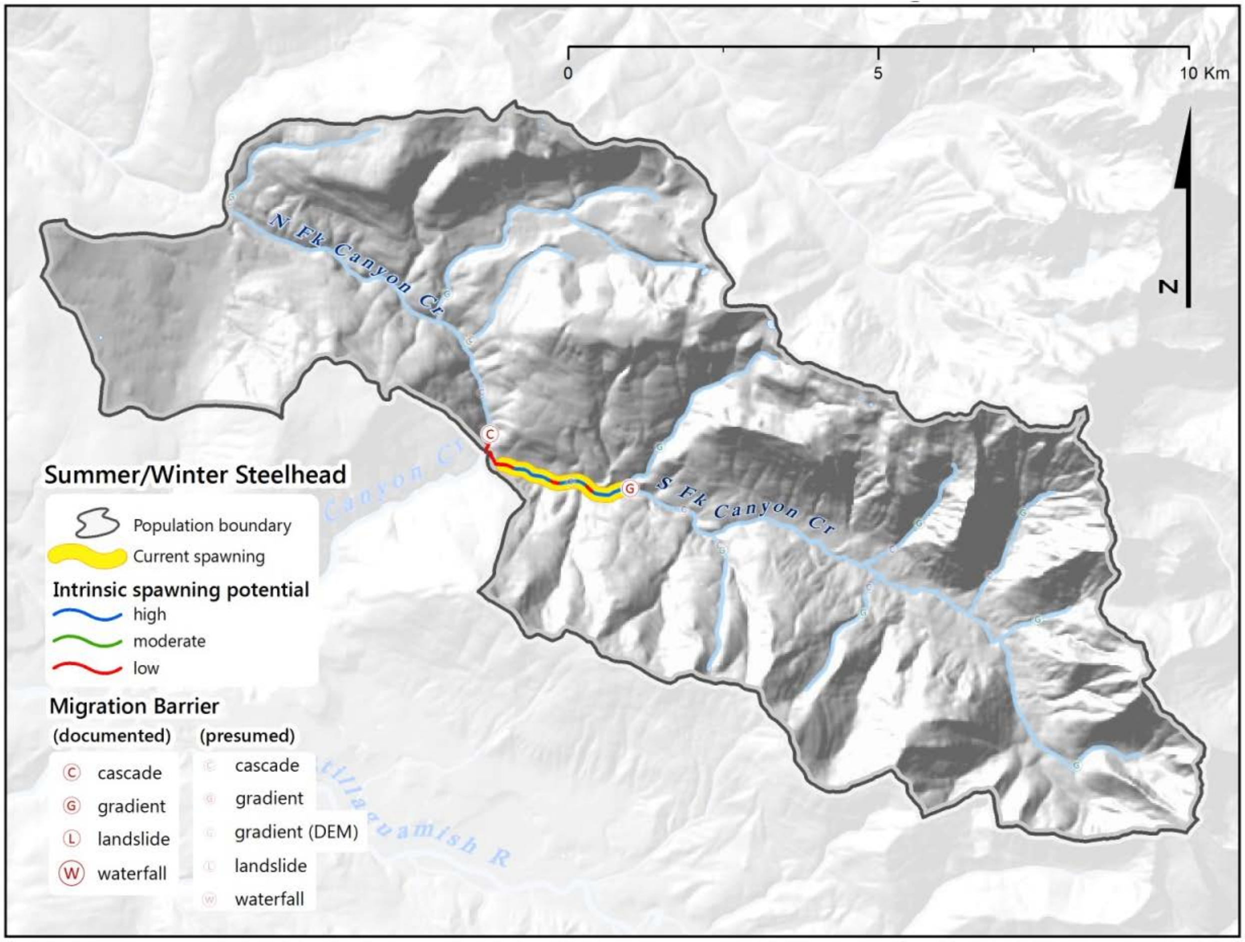

Figure E-34. Map of Canyon Creek Summer-Run DIP spatial structure, including migration barriers and spawning potential.

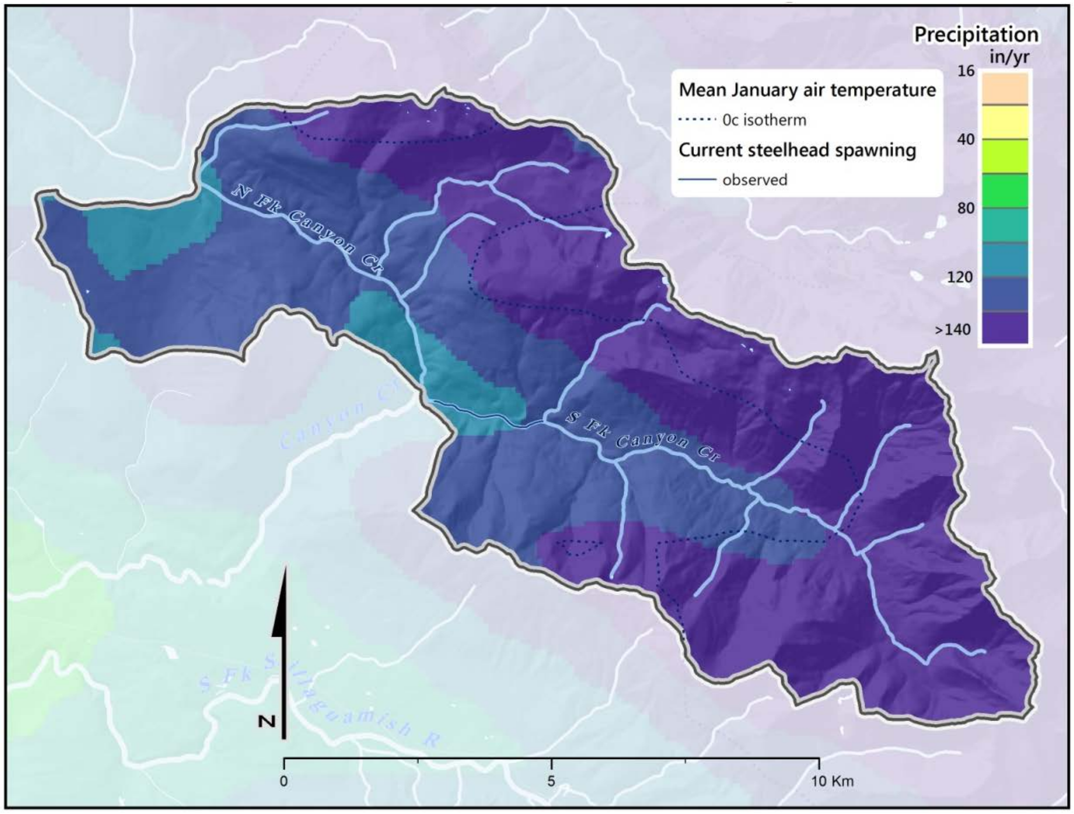

Figure E-35. Map of Canyon Creek Summer-Run DIP area hydrology.

12. Snohomish/Skykomish Rivers Winter Run

Overview—This population includes winter-run steelhead in the mainstem Snohomish, Skykomish, and Wallace rivers, the North Fork Skykomish River below Bear Creek Falls, the Sultan River, and several tributaries (Figure E-36). WDFW (2003) identified three winter-run populations in the Snohomish Basin based on geographic discreteness. This basin includes lowland and higher elevation tributaries. Historically, it was one of the primary steelhead producers in Puget Sound.

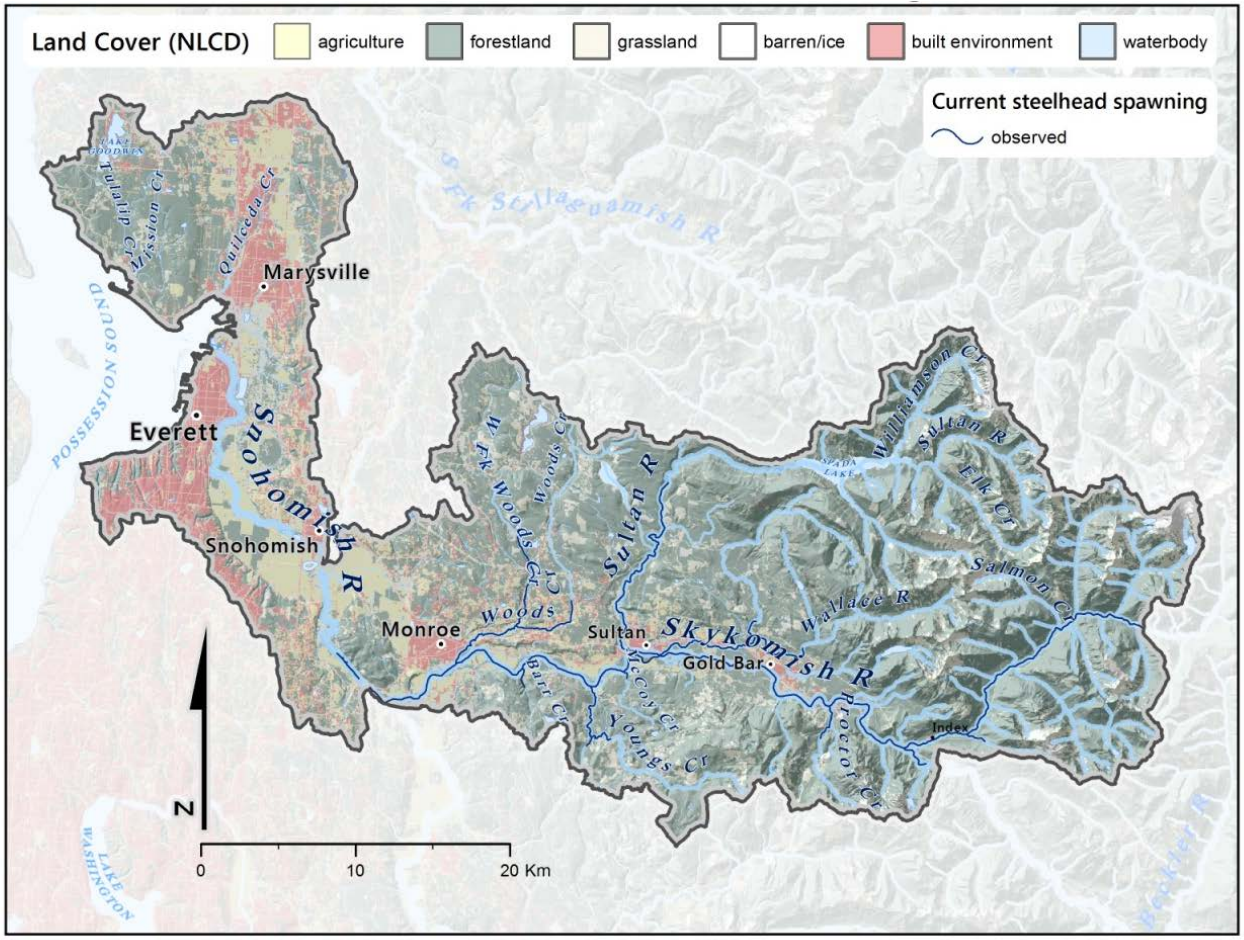

Figure E-36. Map of Snohomish/Skykomish Rivers Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Multiple large and medium	4,801	4,573 (500–41,865)	21,389	42,779
Year	1930	1951	2005–2009	—	

The Snohomish/Skykomish Rivers Winter-Run DIP's exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 0.963 (0.941–0.985).

For a state-space analysis of Snohomish/Skykomish Rivers Winter-Run DIP viability, see the top two plots of Figure A-5 in Appendix A.

Diversity—

Snohomish/Skykomish summer run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Silver Creek	1998–1999	2	Snohomish H. (SK)	0	19,334
Index Creek	2000	1	Snohomish H. (SK)	0	16,300
Skykomish R.	1995–2011	16	Sky/ Snohomish H. (SK)	0	2,400,055
Sultan River	1995–2009	15	Sky/ Snohomish H. (SK)	0	266,980
			Total:	0	2,702,669

Snohomish/Skykomish Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Howard Creek	1997–2001	6	Tokul/Snohomish (CC)	0	173,317
Skykomish R.	1995–2011	17	Chambers/Tokul (CC)	0	2,464,540
Sultan River	1995–2009	15	Chambers/Tokul (CC)	0	344,781
Wallace River	1995–2011	17	Chambers/Tokul (CC)	0	309,863
			Total:	0	3,292,501

Spatial structure and hydrology—Figure E-37 and Figure E-38.

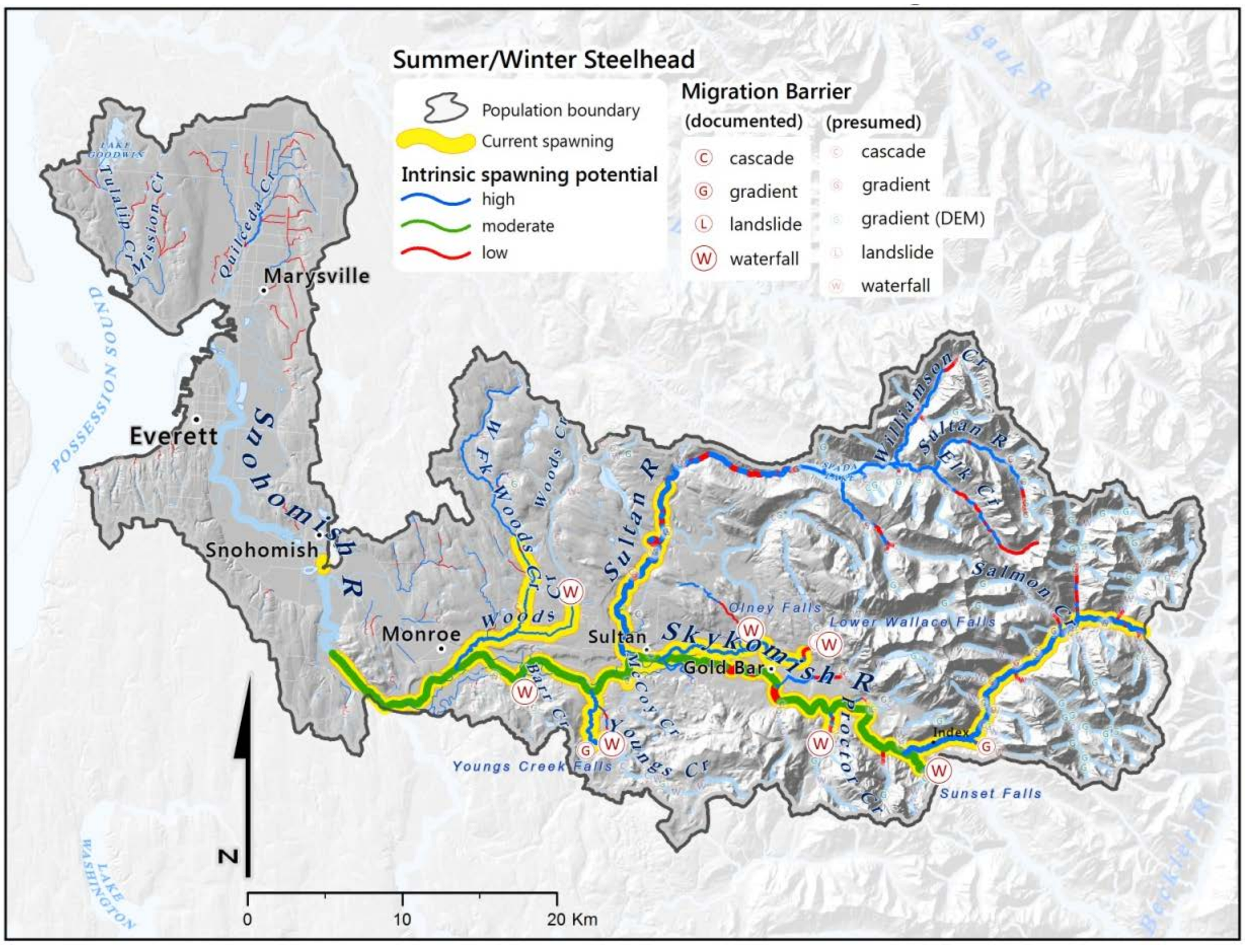

Figure E-37. Map of Snohomish/Skykomish Rivers Winter-Run DIP spatial structure, including migration barriers and spawning potential.

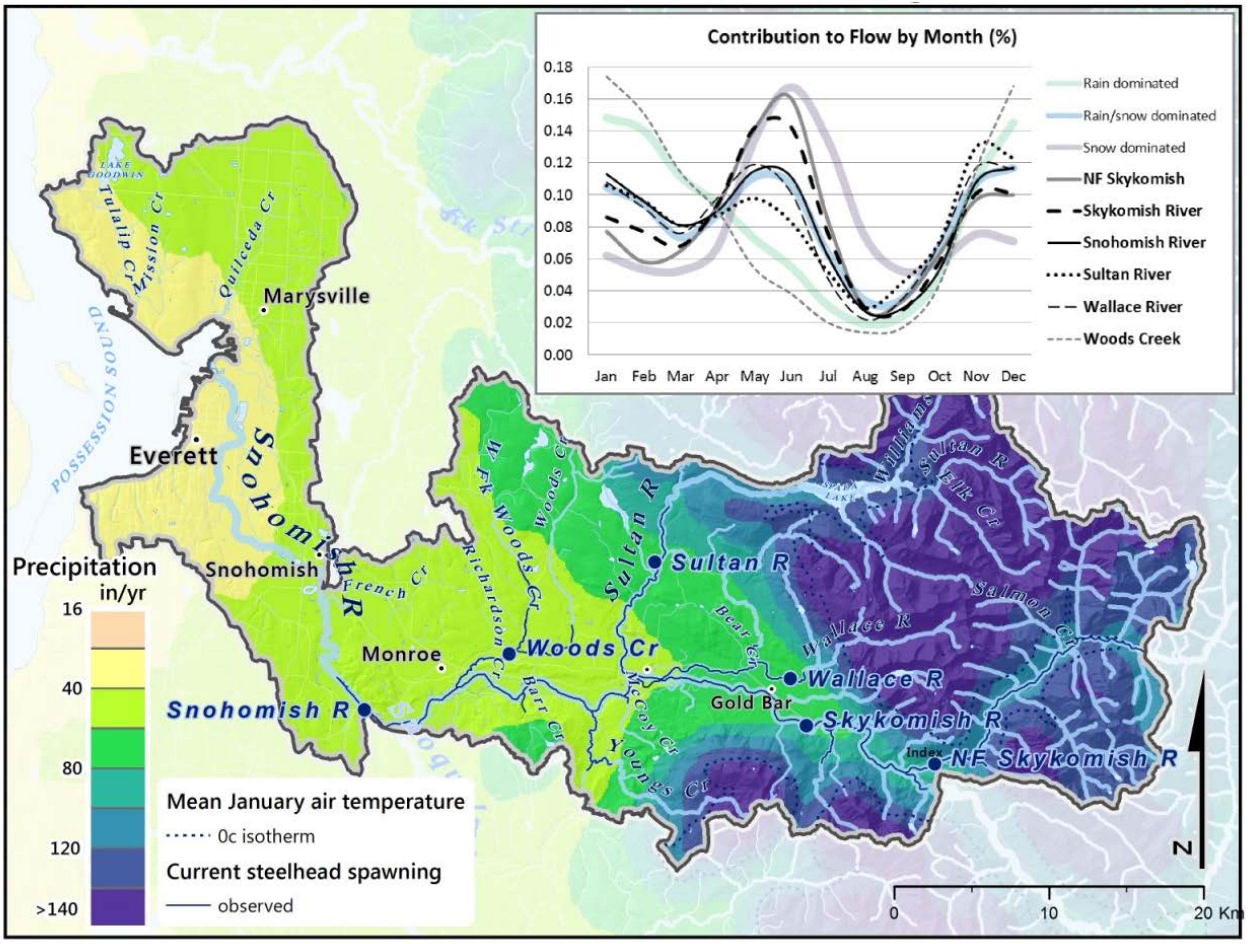

Figure E-38. Map of Snohomish/Skykomish Rivers Winter-Run DIP area hydrology.

13. Pilchuck River Winter Run

Overview—The Pilchuck River (Figure E-39) flows through the Northern Cascades and Puget Lowlands ecoregions. The basin is relatively low gradient and low altitude and exhibits a rainfall-dominated flow pattern. There appears to be sufficient habitat (366 km^2) to support a sustainable population. The Puget Sound Steelhead Technical Recovery Team noted that run timing for this DIP was significantly earlier than other populations in the greater Snohomish Basin, providing further confirmation of its DIP status.

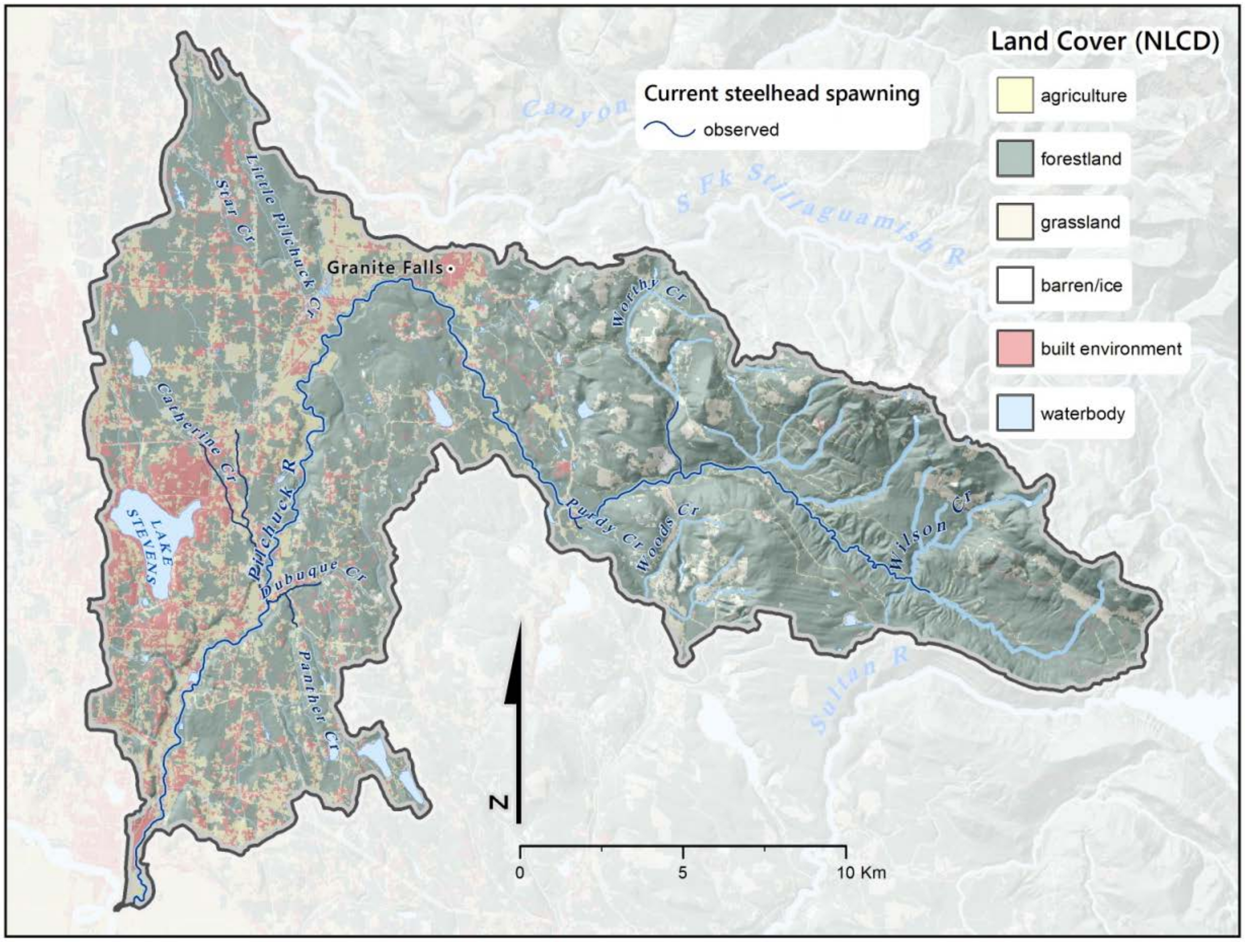

Figure E-39. Map of Pilchuck River Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Large	1,583	469.81	5,193	10,386
Year	1929	1952	2006–2010	—	

Tributary	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
Pilchuck	462	279	696	1,522	604	580	—	646	344	294

For a state-space analysis of population viability, see the top two plots of Figure A-4 in Appendix A.

Diversity—

Pilchuck River Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Pilchuck River	1995–2009	15	Tokul/Stillaguamish/ Skagit (CC)	0	390,193
			Total:	0	390,193

Spatial structure and hydrology—Figure E-40 and Figure E-41.

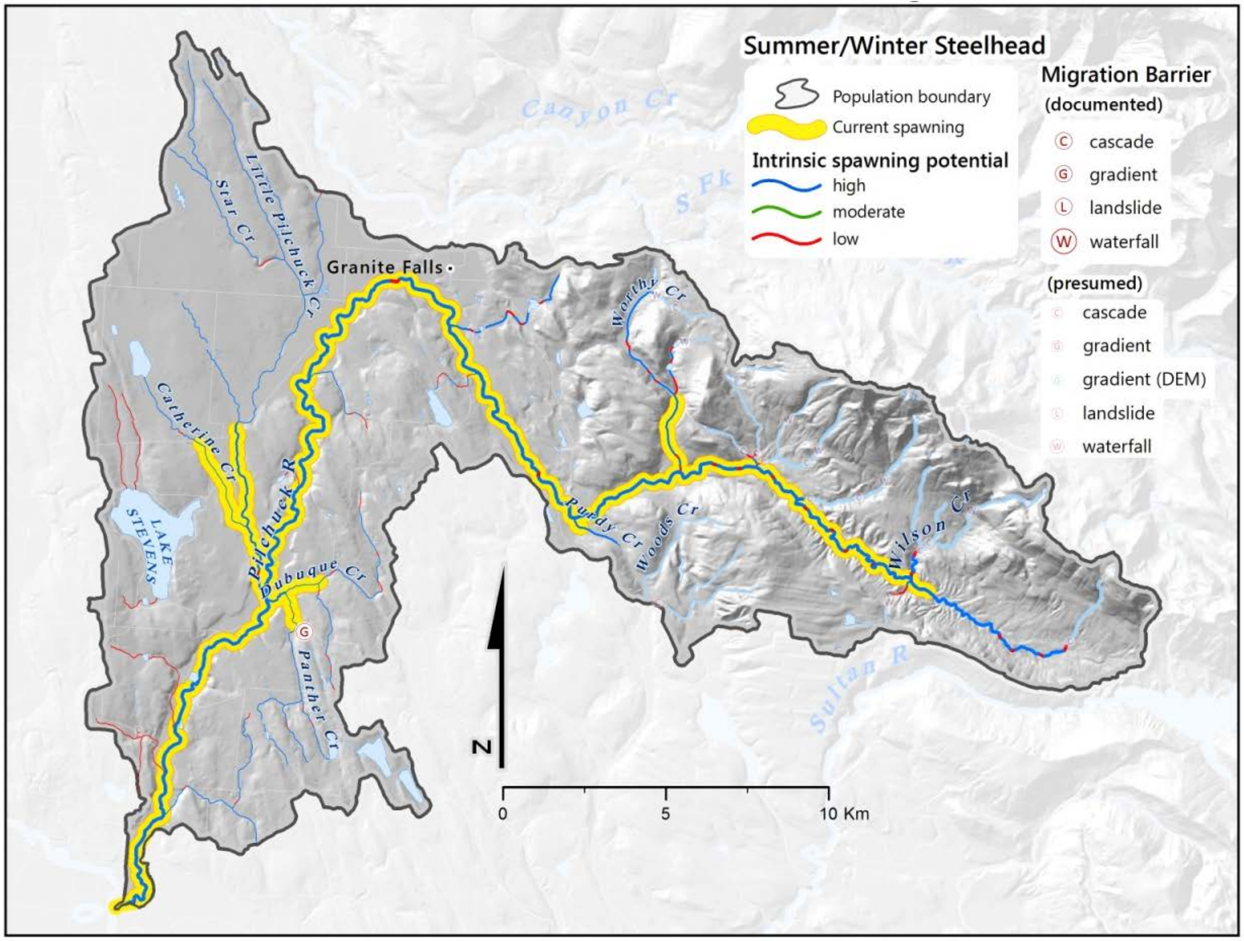

Figure E-40. Map of Pilchuck River Winter-Run DIP spatial structure, including migration barriers and spawning potential.

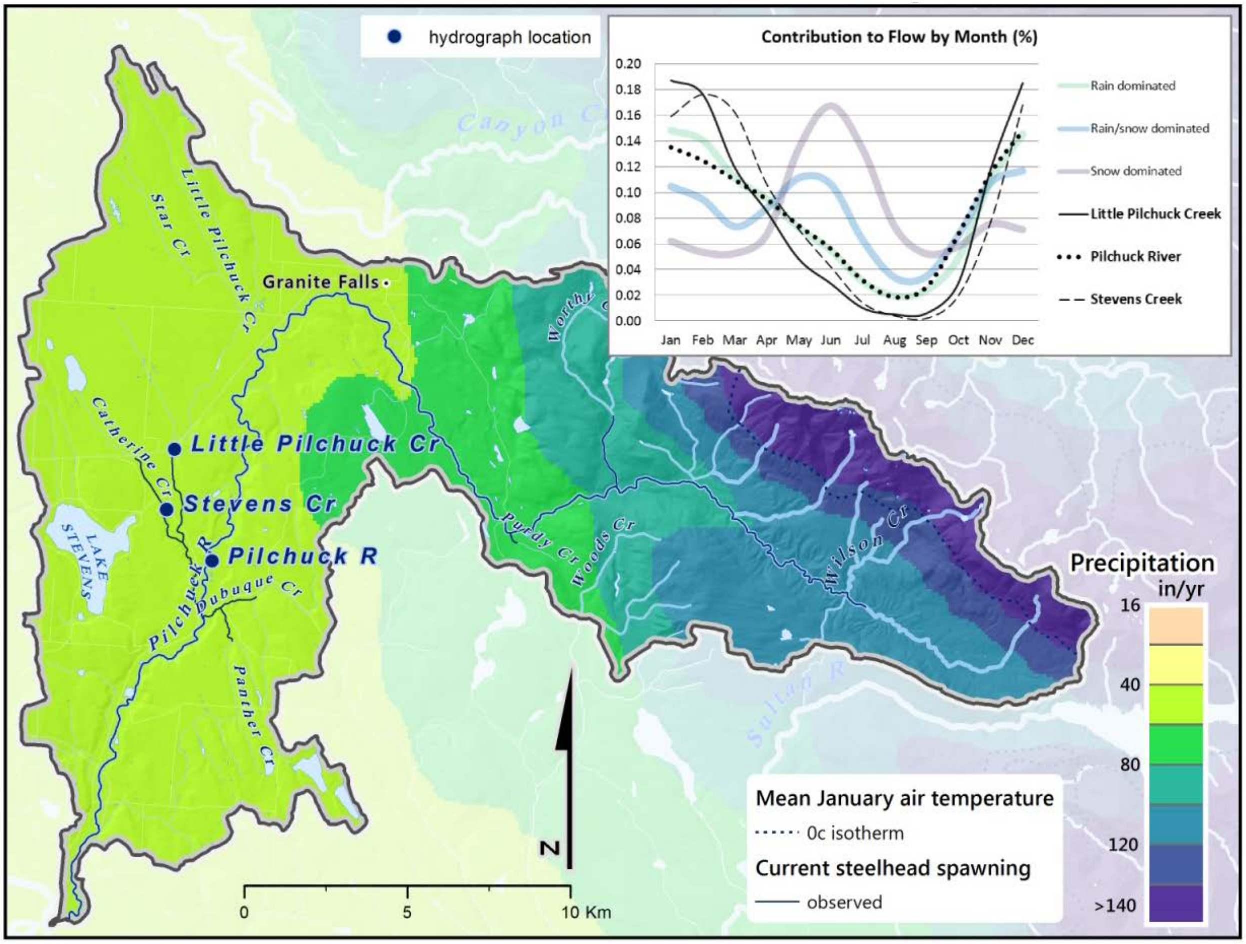

Figure E-41. Map of Pilchuck River Winter-Run DIP area hydrology.

14. North Fork Skykomish River Summer Run

Overview—Summer-run steelhead in the North Fork Skykomish River (Figure E-42) spawn primarily above Bear Creek Falls (RKM 21) (WDFW 2002). Spawning habitat is limited above these falls and accessible habitat may terminate at RKM 31 (Williams et al. 1975). Falls and cascades may provide some level of reproductive isolation from winter-run steelhead in the Skykomish River, but probably also limit population abundance.

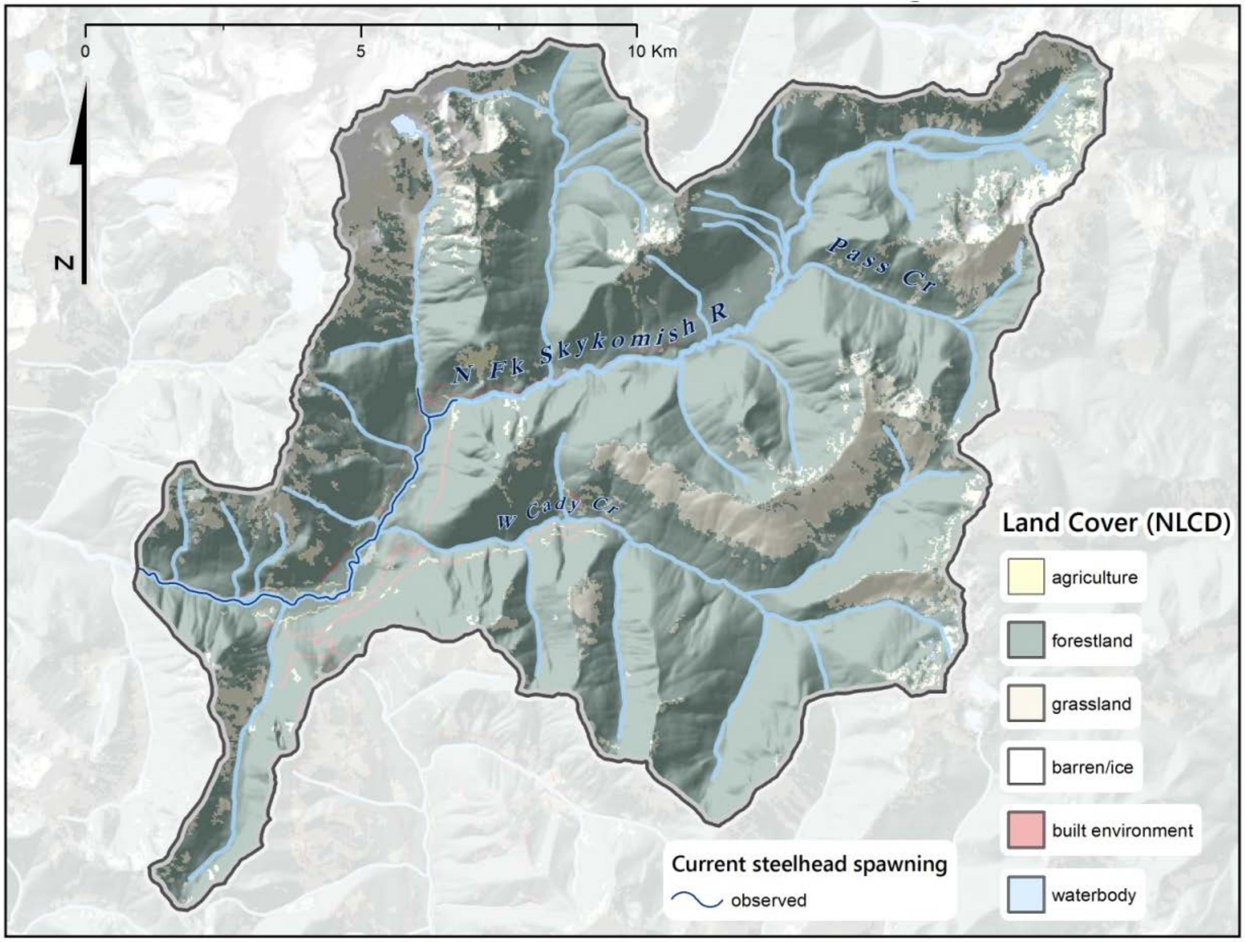

Figure E-42. Map of North Fork Skykomish River Summer-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate	
Abundance	Medium	229*	663	1,325
Year	1930	1963	—	

* Includes only summer-run fish caught in the north fork and not those caught in the main stem.

Diversity—

North Fork Skykomish River Summer Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
North Fork Skykomish R.	1996–2006	11	Skykomish/ Snohomish H. (SK)	0	306,641
			Total:	0	306,641

Spatial structure and hydrology—Figure E-43 and Figure E-44.

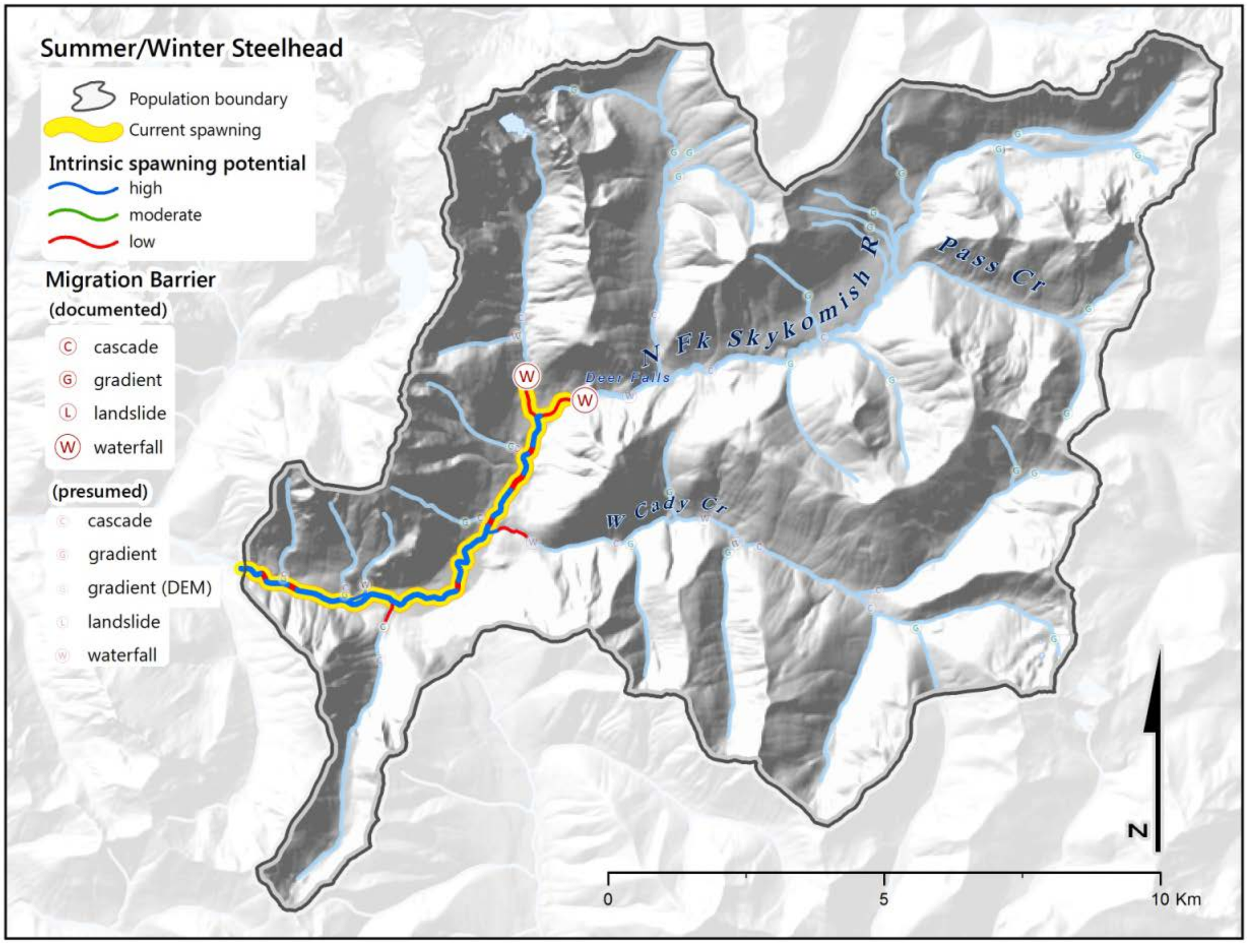

Figure E-43. Map of North Fork Skykomish River Summer-Run DIP spatial structure, including migration barriers and spawning potential.

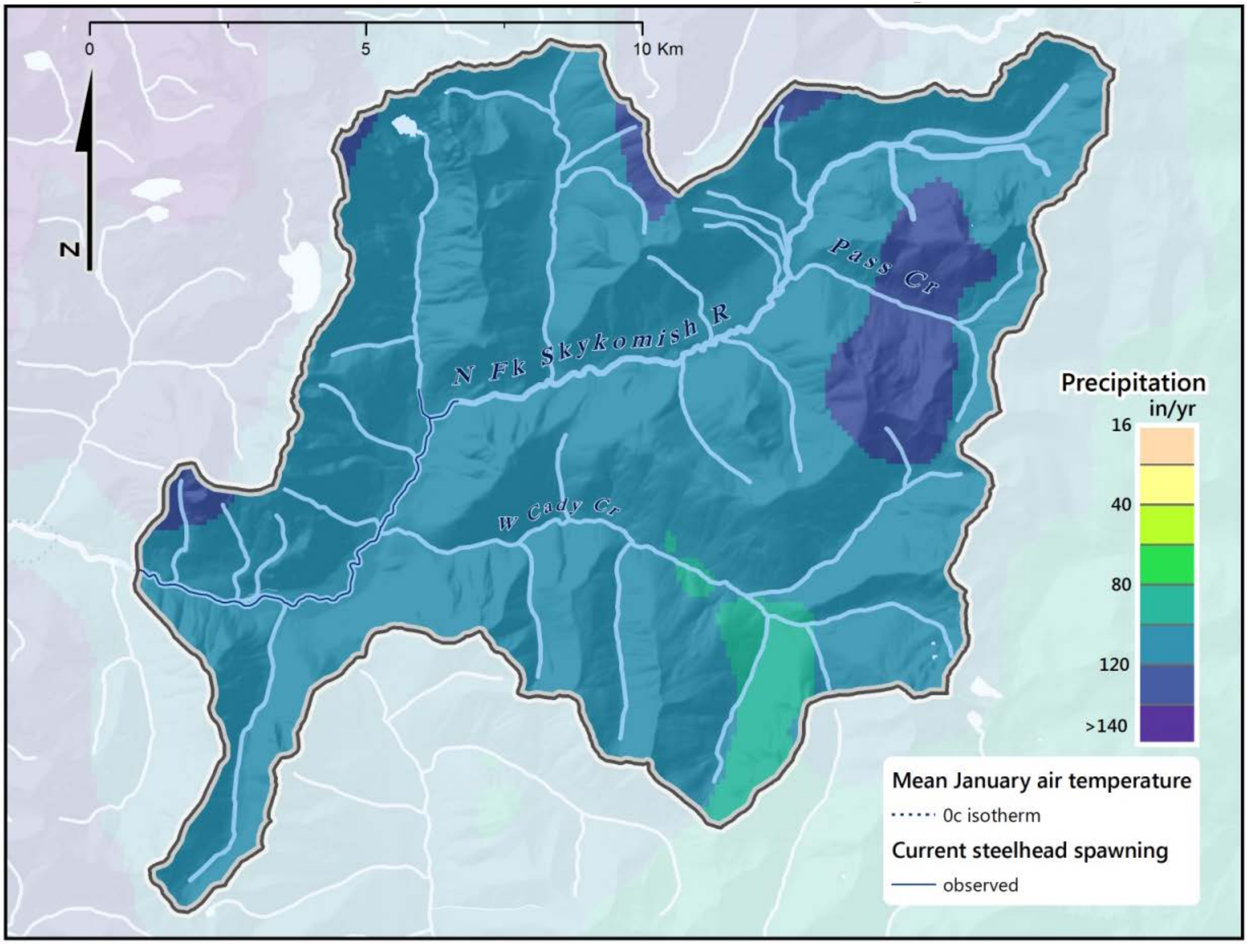

Figure E-44. Map of North Fork Skykomish River Summer-Run DIP area hydrology.

15. Snoqualmie River Winter Run

Overview—This DIP (Figure E-45) includes steelhead in the mainstem Snoqualmie River and those in major tributaries: Tolt, Raging, and Tokul creeks. There are numerous historical references indicating that this basin sustained large runs of steelhead.

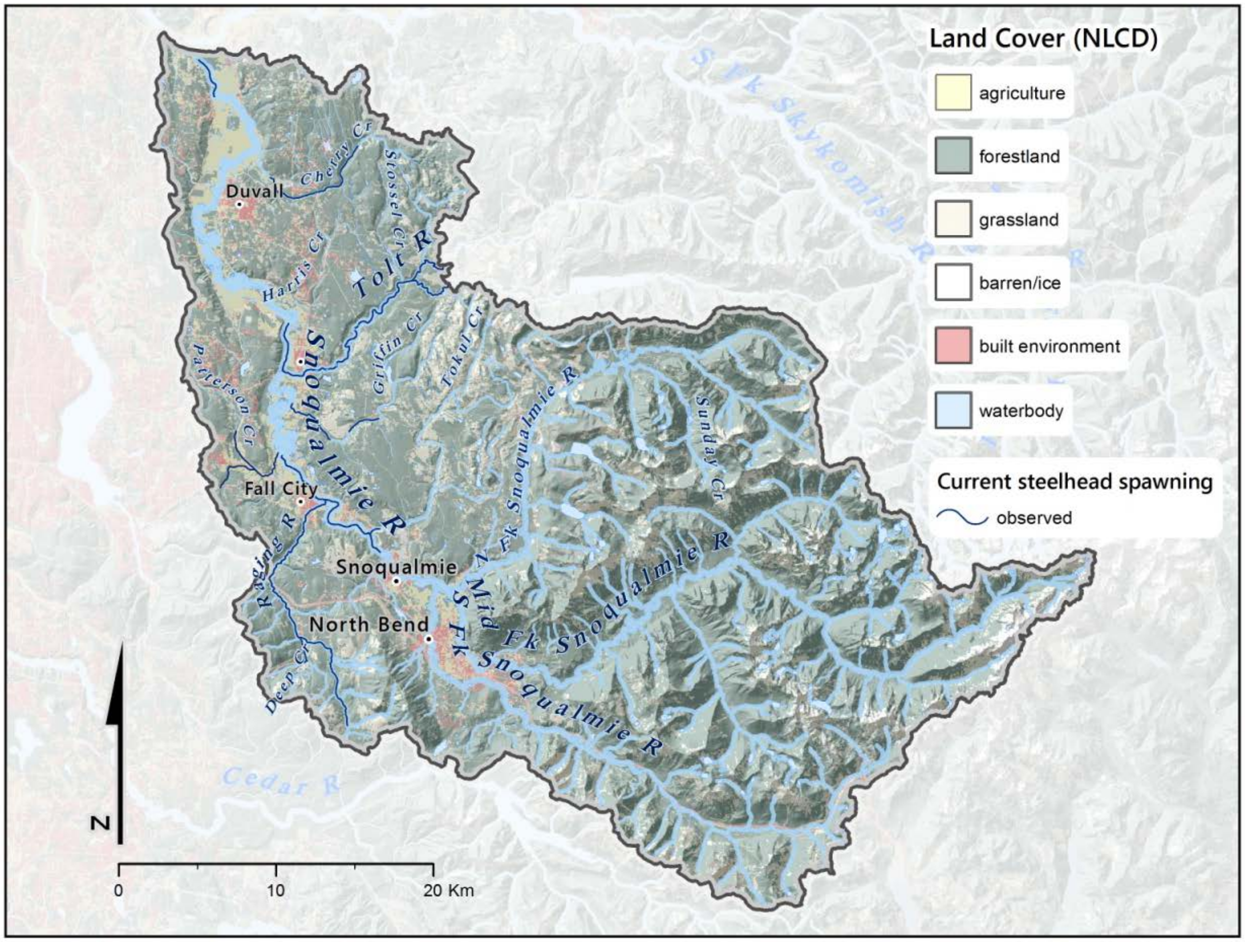

Figure E-45. Map of Snoqualmie River Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Large and medium	2,791	1,092.06	16,740	33,479
Year	1930	1951	2005–2010	—	

Tributary	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
Pilchuck	1,395	789	988	1,506	1,060	1,856	—	—	—	662

For a state-space analysis of Snoqualmie River Winter-Run DIP viability, see the top two plots of Figure A-6 in Appendix A.

Diversity—

Snoqualmie River summer run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Snoqualmie R.	1995–2008	12	Skykomish/ Snohomish H. (SK)	0	494,660
Tokul Creek	2001	1	Snohomish H. (SK)	0	28,330
Raging River	1999–2007	6	Snohomish H. (SK)	0	177,626
			Total:	0	640,616

Snoqualmie River Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Raging River	1995–2008	14	Tokul Creek H. (CC)	0	206,934
Snoqualmie R.	1995–2009	15	Tokul Creek H. (CC)	0	2,290,994
Tokul Creek	2009–2011	3	Tokul Creek H. (CC)	0	329,638
Tolt River	1995–2008	11	Tokul Creek Hatchery/Reiter(CC)	0	246,776
			Total:	0	3,074,342

Spatial structure and hydrology—Figure E-46 and Figure E-47.

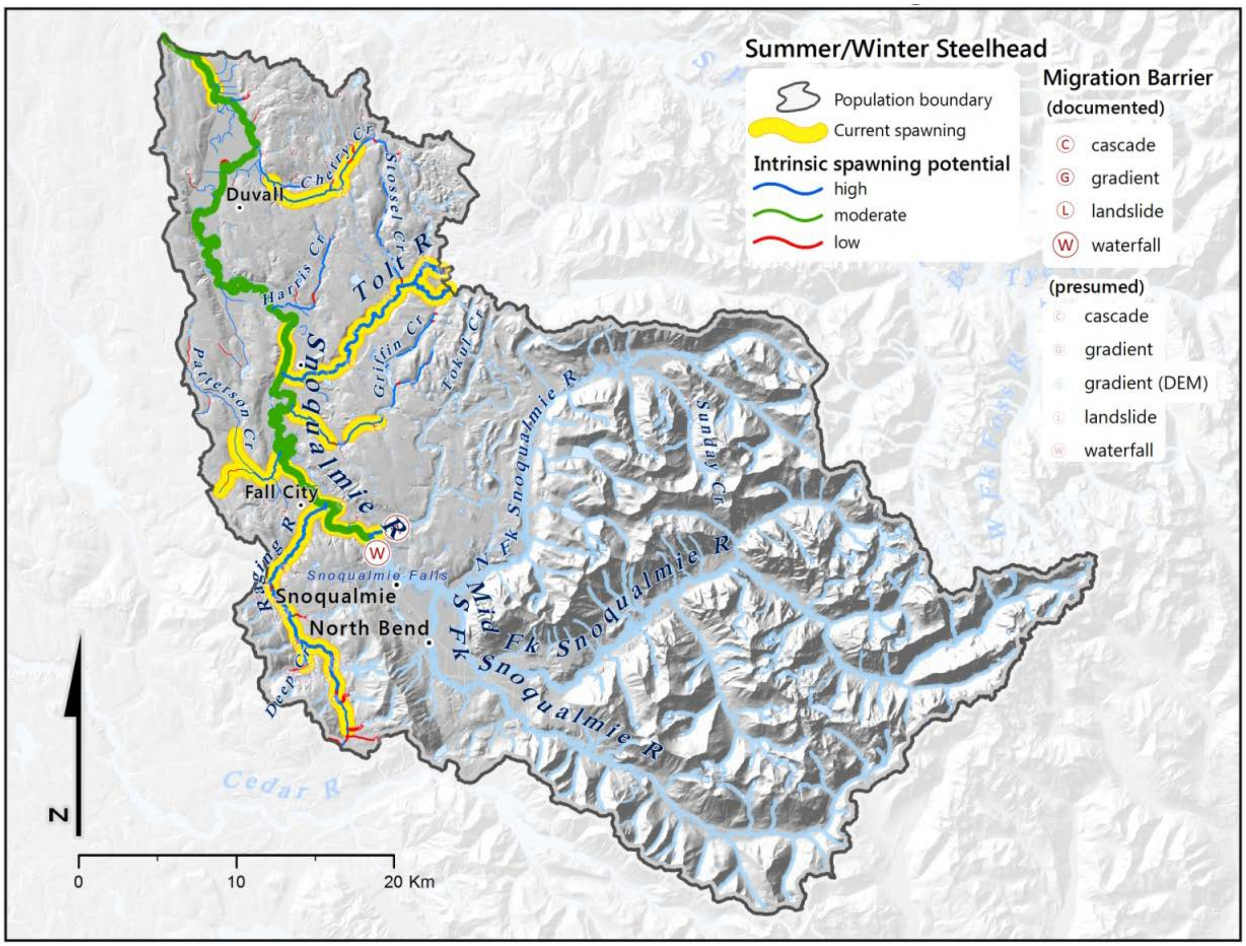

Figure E-46. Map of Snoqualmie River Winter-Run DIP spatial structure, including migration barriers and spawning potential.

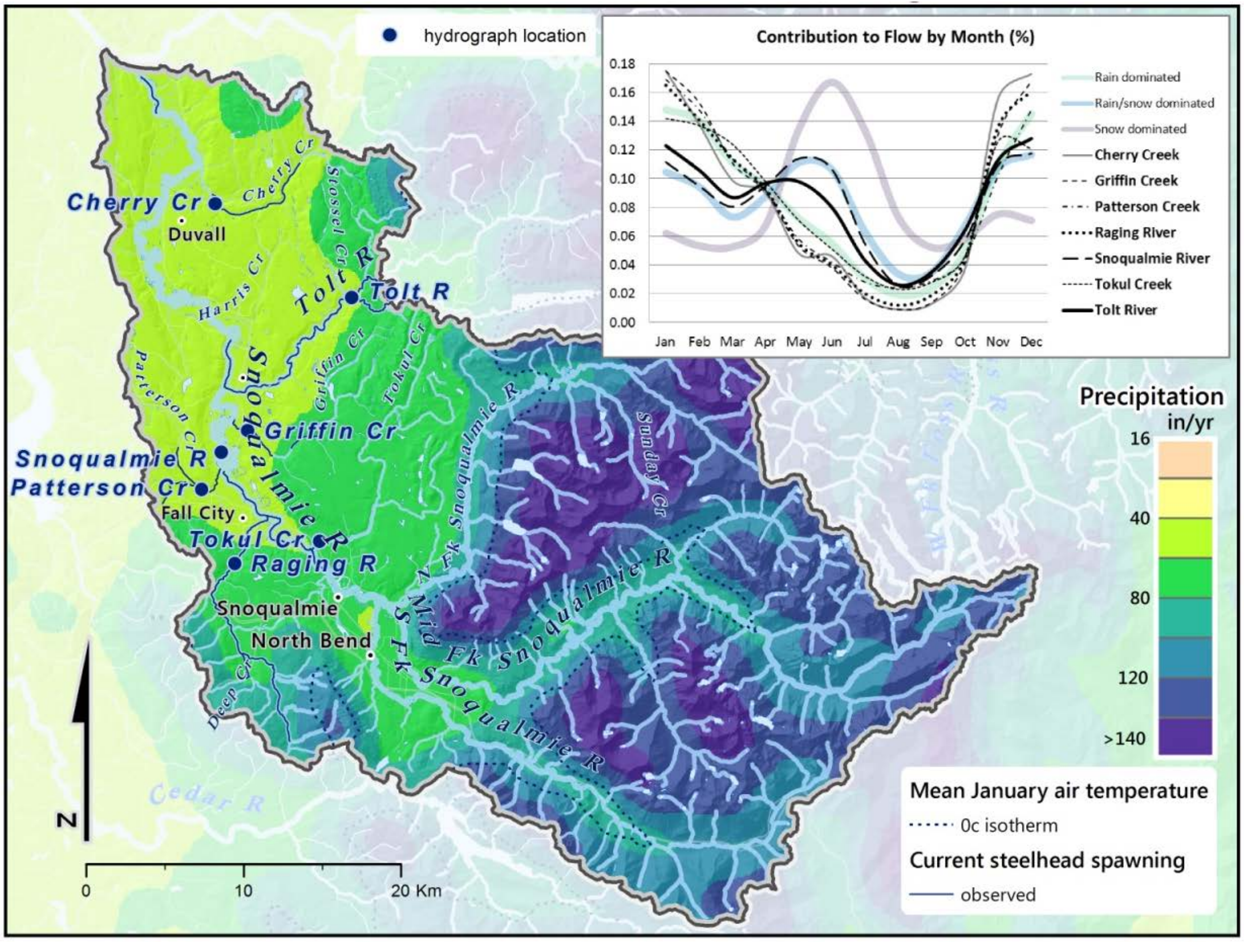

Figure E-47. Map of Snoqualmie River Winter-Run DIP area hydrology.

16. Tolt River Summer Run

Overview—Summer-run steelhead are found in the North Fork and South Fork Tolt River basins (Figure E-48). Both forks are typical of summer-run steelhead habitat and contain a number of falls and cascades, although the north fork is higher gradient with steeply sloped canyon walls (Williams et al. 1975). Genetically, Tolt River steelhead are similar to other Snohomish Basin steelhead (Phelps et al. 1997).

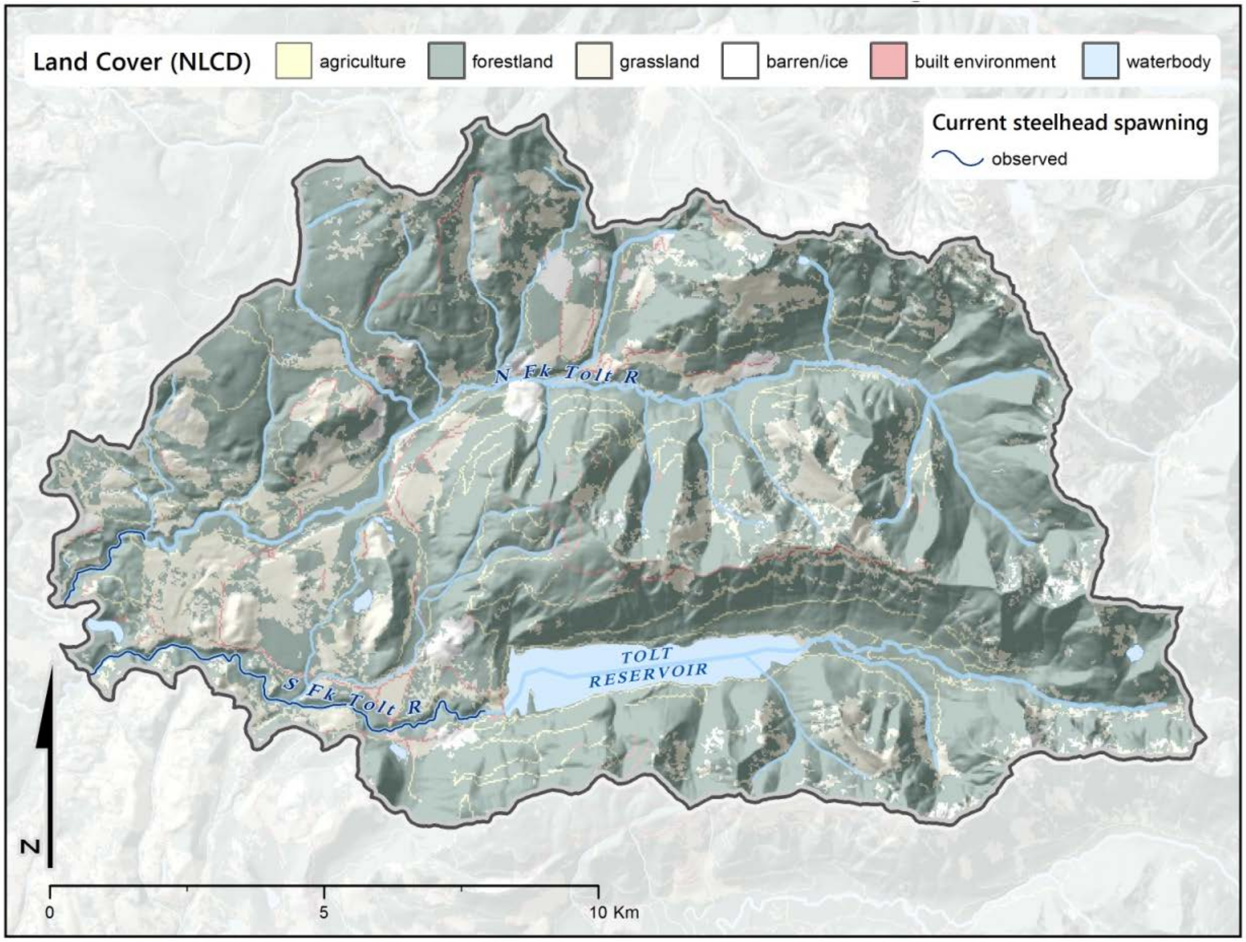

Figure E-48. Map of Tolt River Summer-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Large and medium	304*	78.64	321	641
Year	1930	1964	2005–2010	—	

* Includes only summer-run fish caught in the Tolt River and not those caught in the mainstem Snoqualmie or Snohomish rivers.

For a state-space analysis of population viability, see Figure A-7 in Appendix A.

Diversity—Hatchery releases for Tolt River Summer Run are zero within the DPS and zero outside the DPS.

Spatial structure and hydrology—Figure E-49 and Figure E-50.

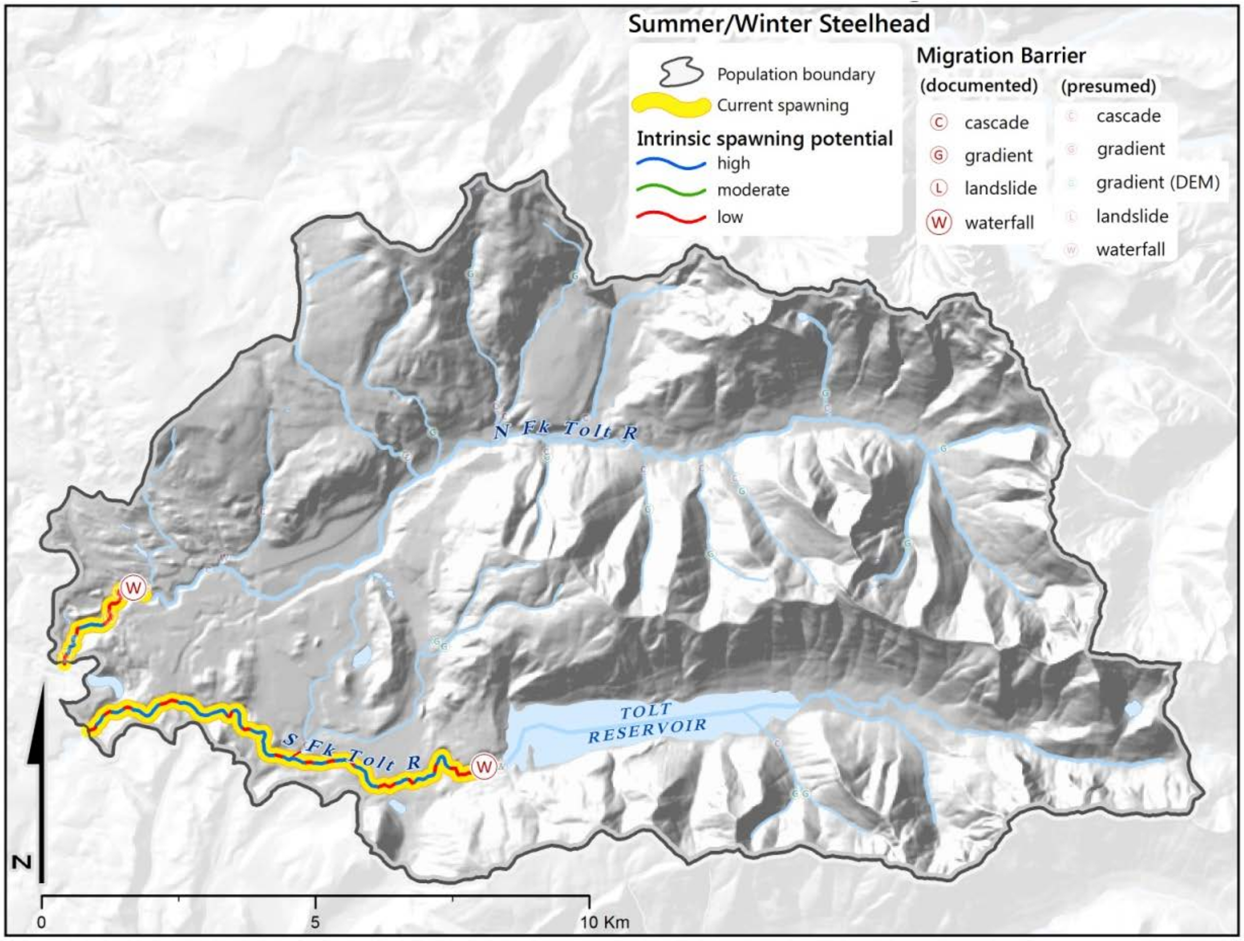

Figure E-49. Map of Tolt River Summer-Run DIP spatial structure, including migration barriers and spawning potential.

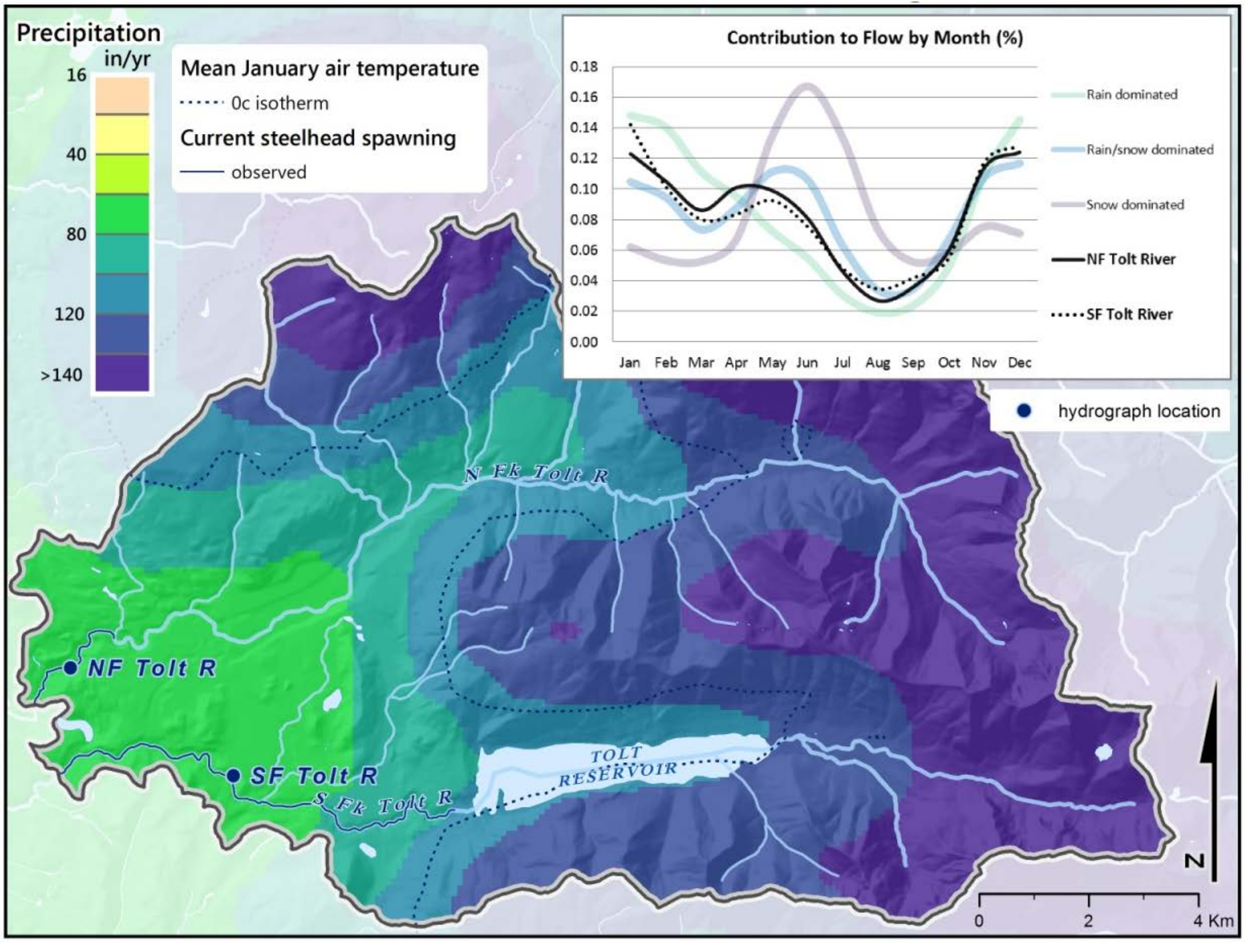

Figure E-50. Map of Tolt River Summer-Run DIP area hydrology.

Central and South Puget Sound Major Population Group

17. Cedar River Winter Run

Overview—This population includes steelhead in the Cedar River and tributaries to the southern end of Lake Washington, primarily May and Coal creeks (Figure E-51). Diversion of the Cedar River from its confluence with the Black River to Lake Washington in the early 1900s may have impacted steelhead life history and abundance. The hydrology of the Cedar River (rain/snow) is distinct from other tributaries in the Lake Washington system. Also, until recently much of the upper watershed was blocked to anadromous access.

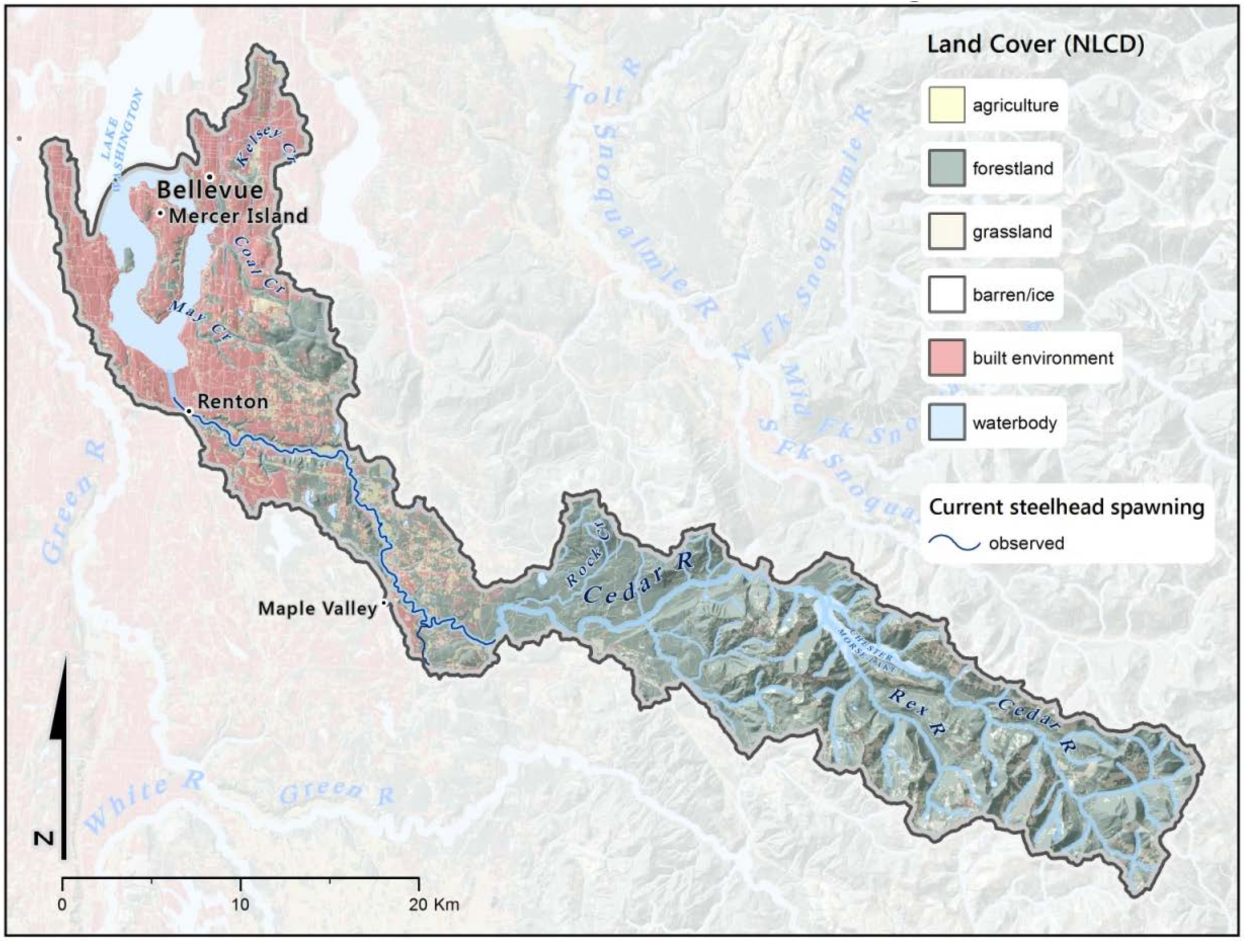

Figure E-51. Map of Cedar River Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate	
Abundance	Scarce	353	5,949	11,899
Year	1930	1954	—	

For a state-space analysis of population viability, see Figure A-8 in Appendix A.

Diversity— Hatchery releases for Cedar River Winter Run are zero within the DPS and zero outside the DPS.

Spatial structure and hydrology—Figure E-52 and Figure E-53.

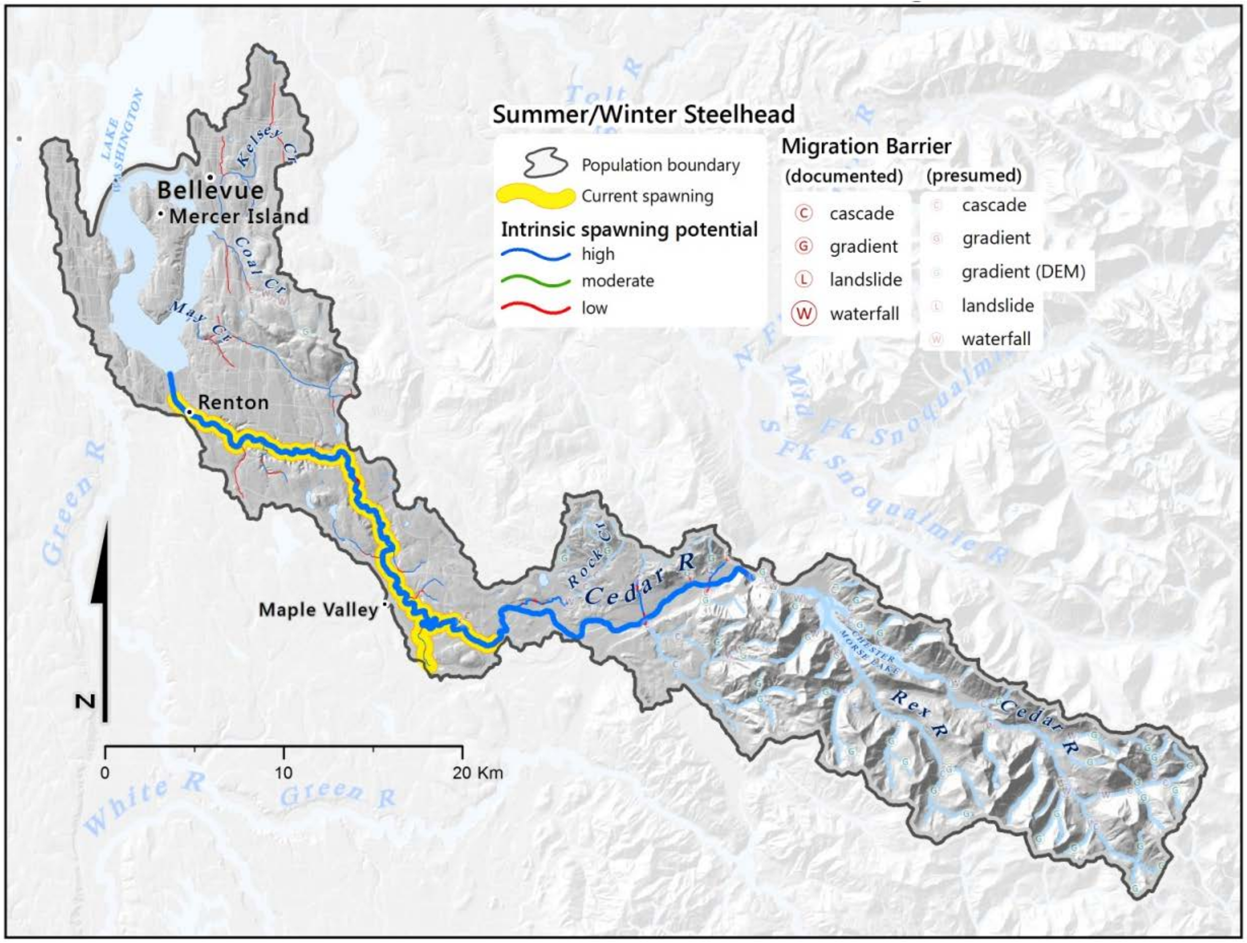

Figure E-52. Map of Cedar River Winter-Run DIP spatial structure, including migration barriers and spawning potential.

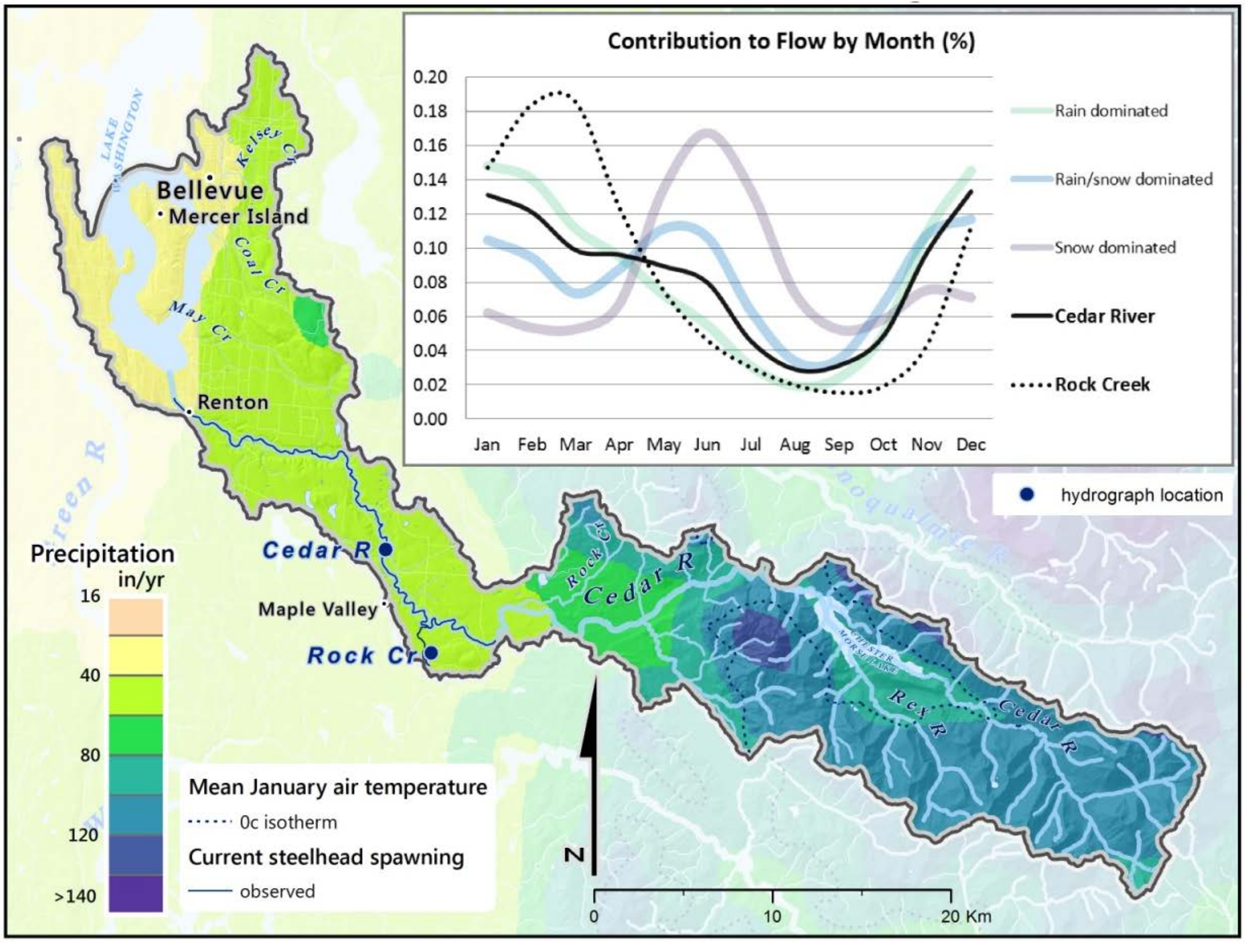

Figure E-53. Map of Cedar River Winter-Run DIP area hydrology.

18. North Lake Washington and Lake Sammamish Winter Run

Overview—This population includes steelhead that spawn in tributaries to North Lake Washington, the Sammamish River, and Lake Sammamish (Figure E-54). The majority of the tributaries drain lowland habitat, with the exception of Issaquah Creek. Given the major modifications in the Lake Washington system, primarily the rerouting of the outlet from the Black River to the Lake Washington Ship Canal and widespread land development, it is difficult to infer historical distribution from current abundance.

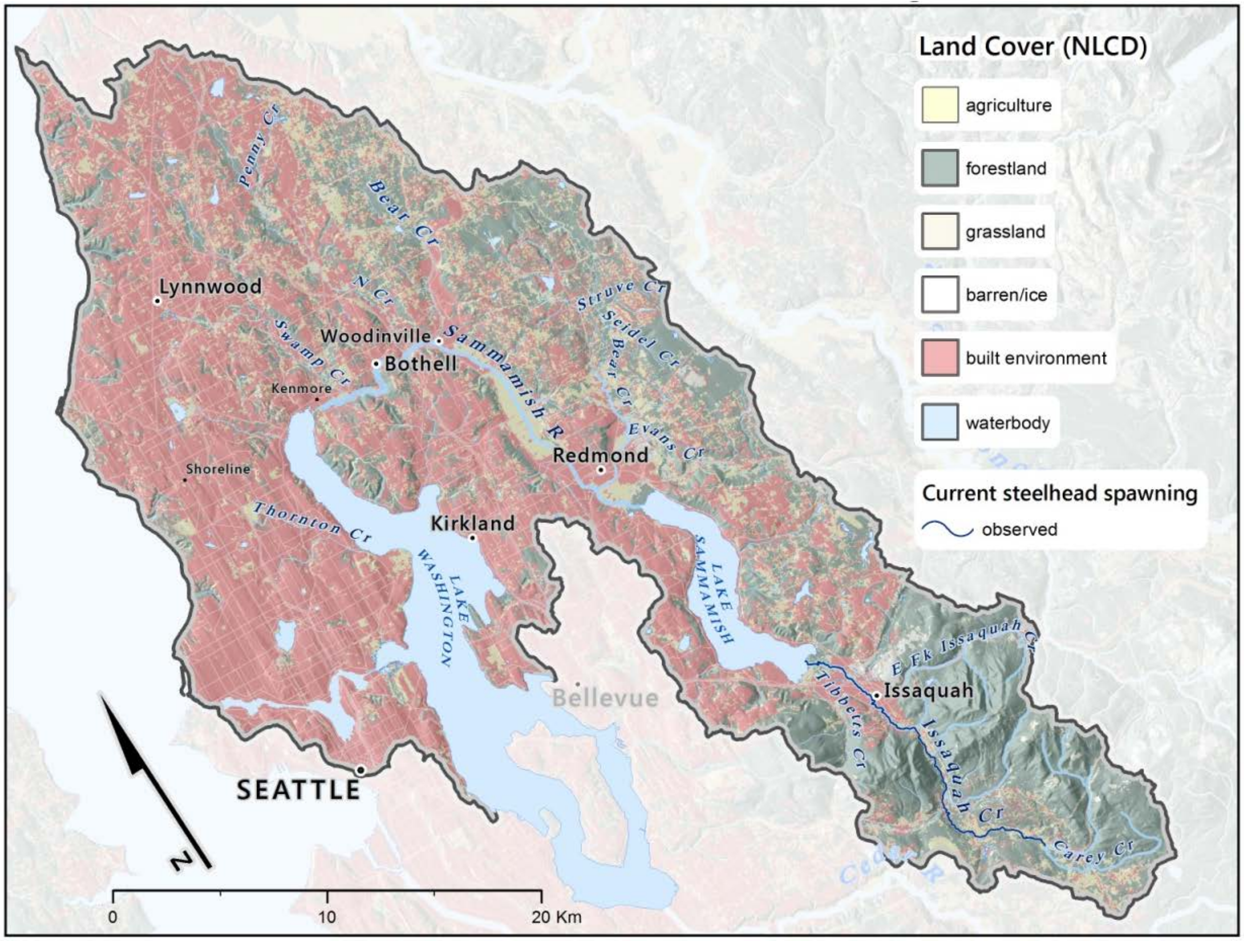

Figure E-54. Map of North Lake Washington and Lake Sammamish Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—The North Lake Washington and Lake Sammamish Winter Run's exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 0.807 (0.770–0.845).

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Absent[a]	748[b]	12 (3–55)	5,268	10,536
Year	1930	1952	2005–2009	—	

[a] Surveys were not conducted in Swamp Creek, North Creek, or Issaquah Creek.

[b] Includes catch from the Lake Washington Ship Canal (which may have been destined for the Cedar River).

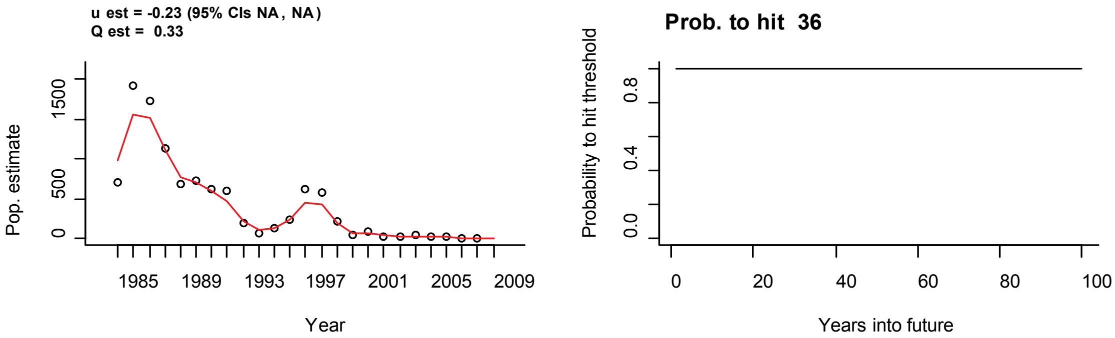

Figure E-55. State-space analysis of population viability. The left panel plots the observed counts and the fitted state-space estimate against year, giving the MARSS maximum-likelihood estimate of fit to the abundance data (red curve), the estimated long-term population growth rate (μ_{est}, equivalent to ln(λ)), and the estimate of process error (Q_{est}). The right panel plots the probability that the population will reach a particular quasi-extinction threshold abundance within the next 100 years (with approximate 95% CIs).

Diversity—

North Lake Washington Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Issaquah Creek	1998–2000	3	Lake Washington	39,516	0
			Total:	39,516	0

Spatial structure and hydrology—Figure E-56 and Figure E-57.

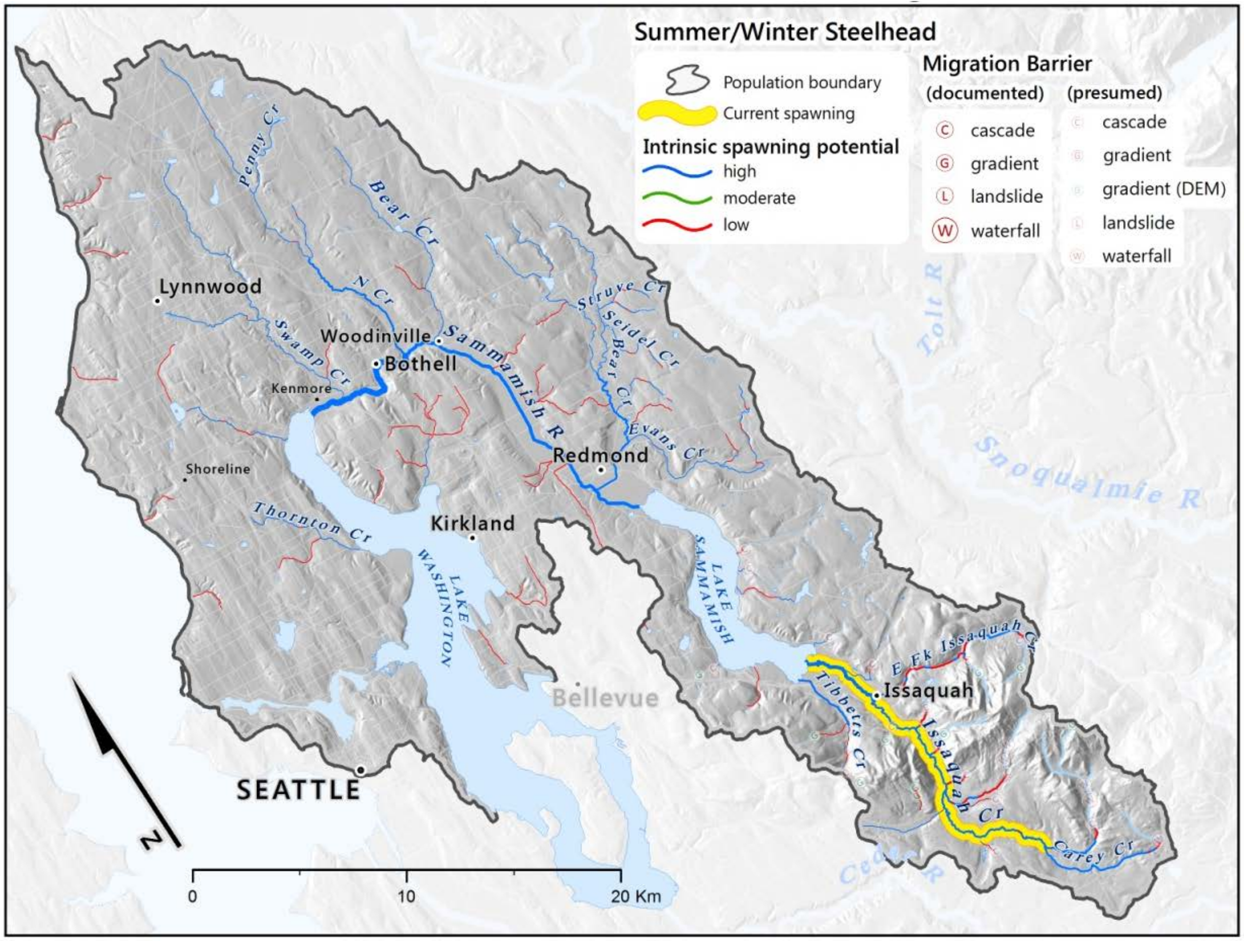

Figure E-56. Map of North Lake Washington and Lake Sammamish Winter-Run DIP spatial structure, including migration barriers and spawning potential.

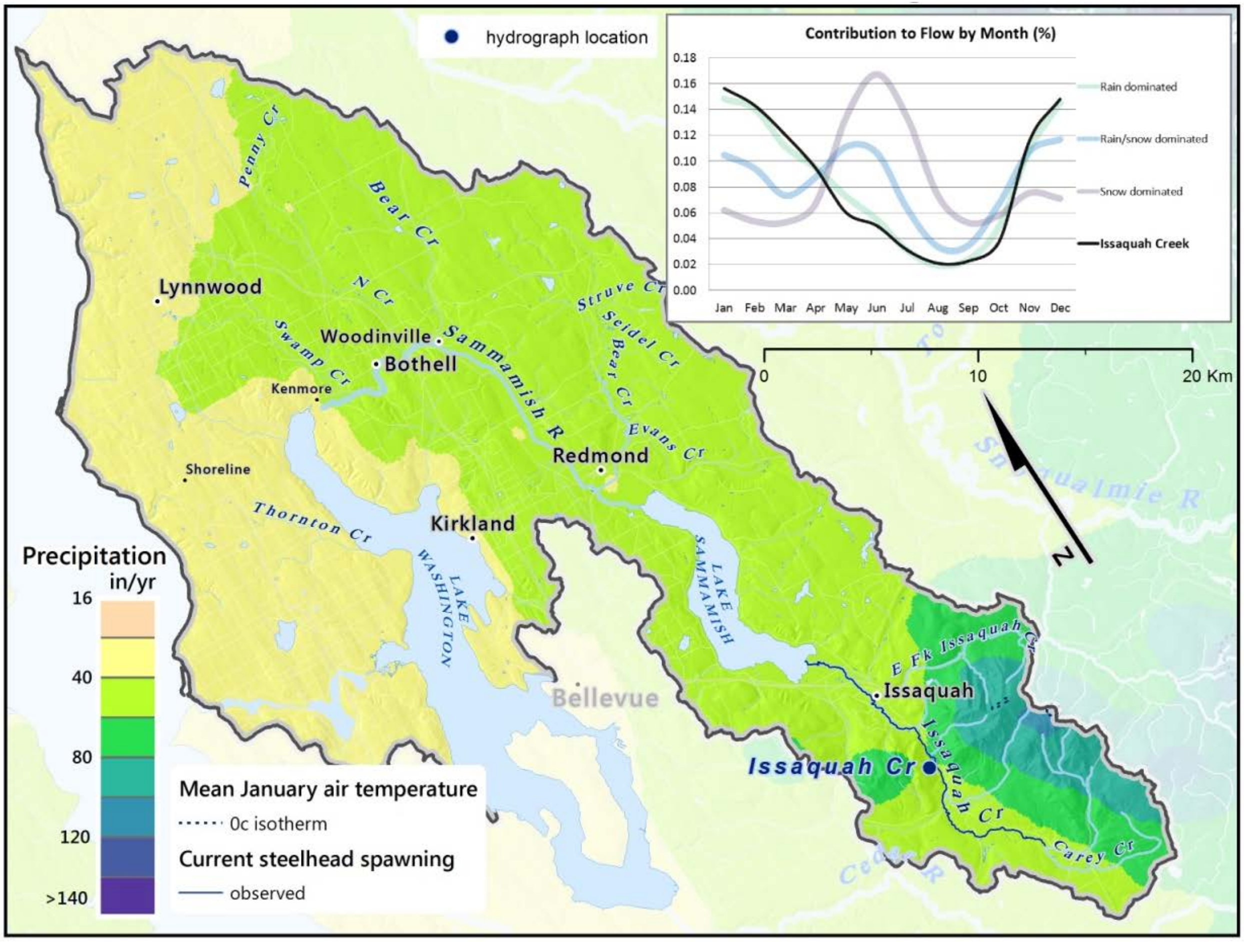

Figure E-57. Map of North Lake Washington and Lake Sammamish Winter-Run DIP area hydrology.

19. Green River Winter Run

Overview—This population includes steelhead that spawn in the Green River and its tributaries including the Duwamish River (Figure E-58). In the last 150 years, the Green River basin has undergone a number of considerable changes. Connectivity with the Black, Cedar, and White rivers has been lost (although on a geologic time scale these connections were likely transitory), as has access to much of the headwater areas of the Green River.

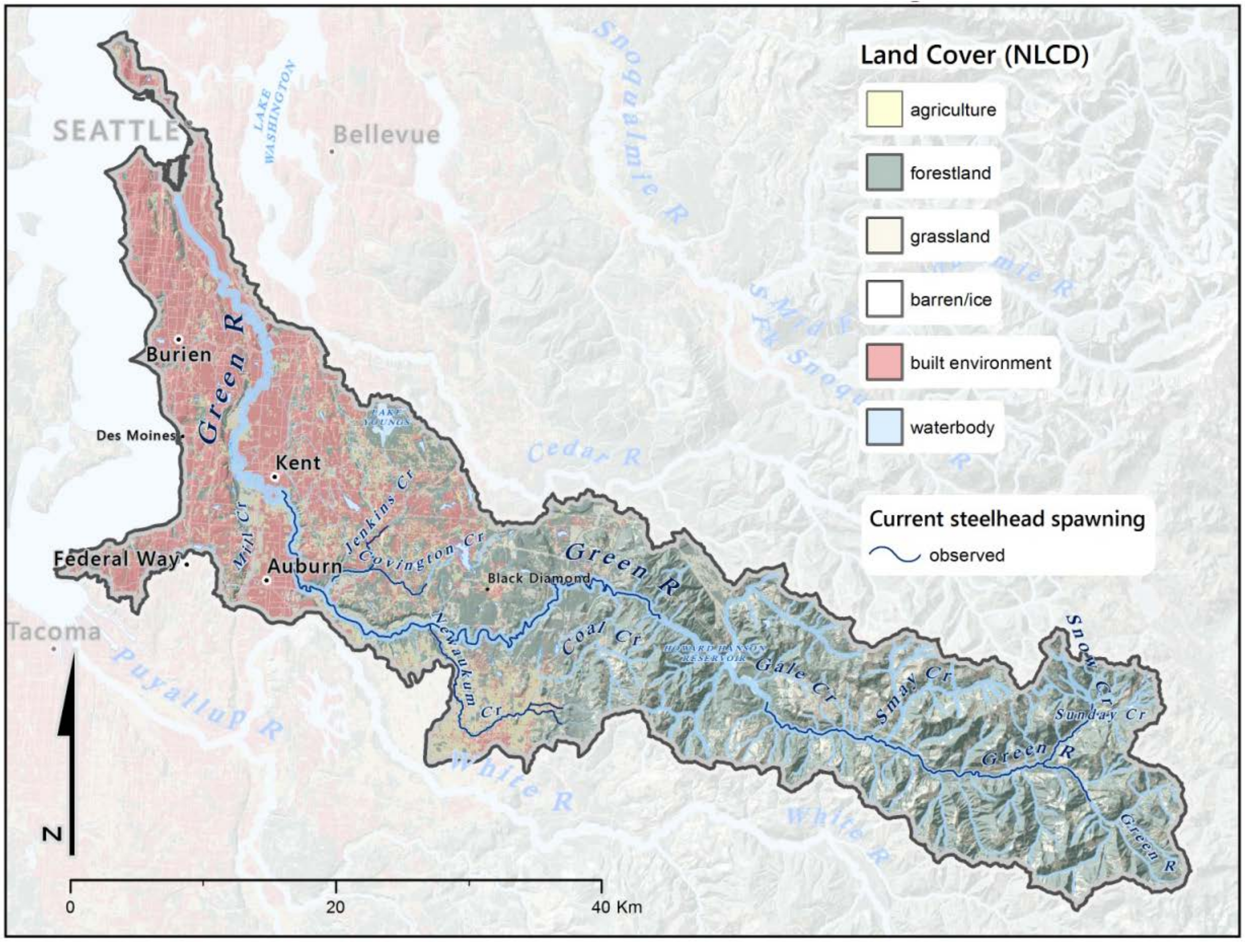

Figure E-58. Map of Green River Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate
Abundance	Large	4,242	986 (401–2,428)	19,768 39,537
Year	1930	1950	2005–2009	—

The Green River Winter Run's exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 0.992 (0.969–1.016).

For a state-space analysis of population viability, see the top two plots of Figure A-9 in Appendix A.

Diversity—

Green River summer run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Big Soos Cr.	2002–2011	10	Green River (SK)	0	355,017
Green River	1995–2010	16	Green River (SK)	0	1,012,567
Icy Creek	2010–2011	2	Green River (SK)	0	46,600
			Total:	0	1,414,184

Green River Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Big Soos Cr.	2011	1	Green River (CC)	0	55,033
Big Soos Cr.	2003–2011	9	Green River native	461,849	0
Crisp Creek	1999–2002	4	Tokul Creek H. (CC)	0	257,000
Crisp Creek	1998–2004	3	Green River native	134,300	0
Duwamish R.	2000–2001	2	Green River native	95,269	0
Friday Creek	1996–1997	2	Green River native	2,262	0
Green River	1995–2011	15	Bogachiel/Tokul (CC)	0	1,805,397
Green River	1995–2009	13	Green River native	1,005,555	0
Icy Creek	2011	1	Icy Creek (CC)	0	25,000
Icy Creek	2010–2011	2	Green River native	28,964	0
Sunday Creek	1995–2000	4	Green River native	46,585	0
			Total:	1,774,784	2,142,430

Spatial structure and hydrology—Figure E-59 and Figure E-60.

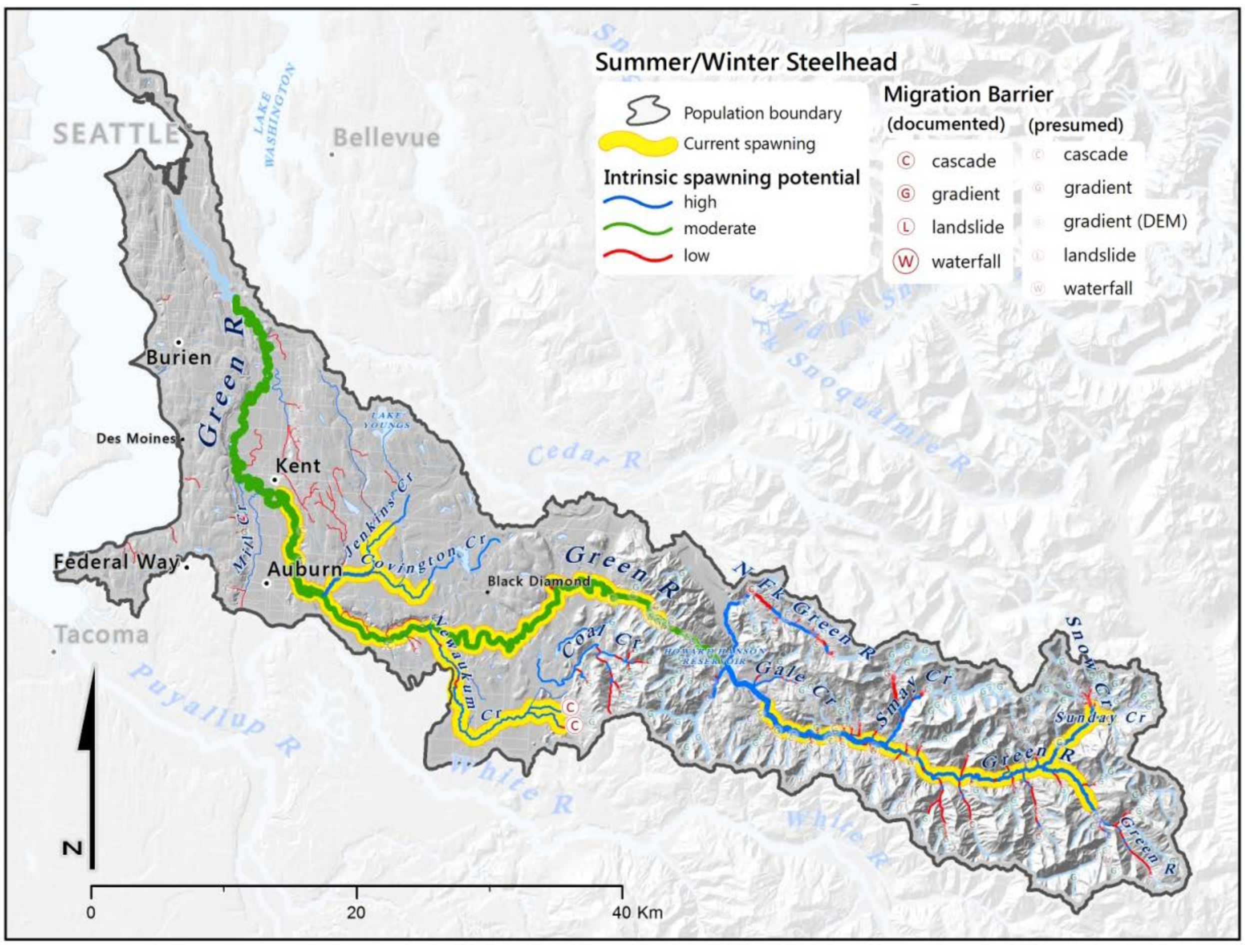

Figure E-59. Map of Green River Winter-Run DIP spatial structure, including migration barriers and spawning potential.

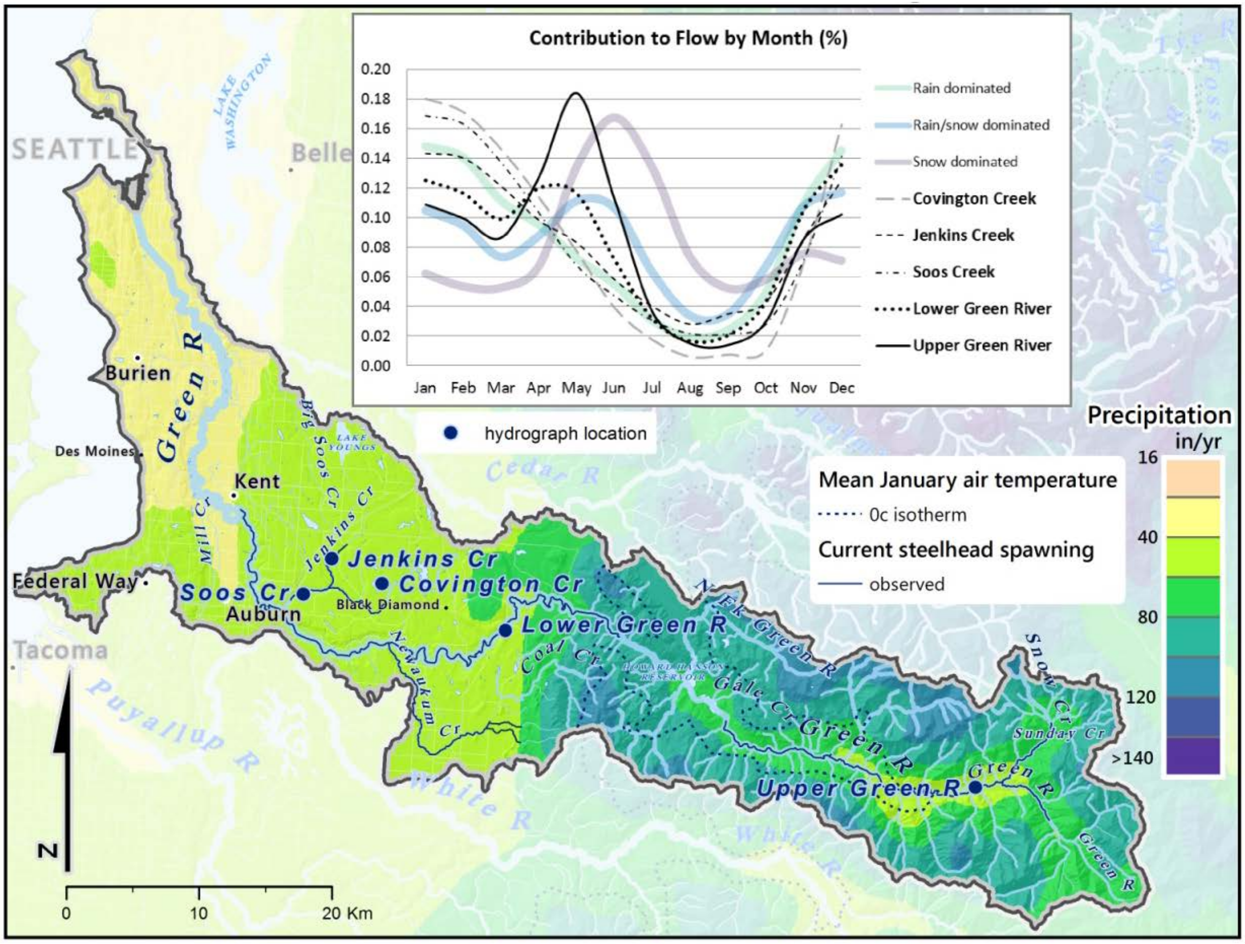

Figure E-60. Map of Green River Winter-Run DIP area hydrology.

20. Puyallup/Carbon Rivers Winter Run

Overview—This population includes two Salmonid Stock Inventory stocks (WDFW 2002), the Puyallup and Carbon rivers (Figure E-61). The Puget Sound Steelhead Technical Recovery Team determined that the mainstem Puyallup below the confluence of the Puyallup River and White River was more closely associated with the Carbon River than the White River.

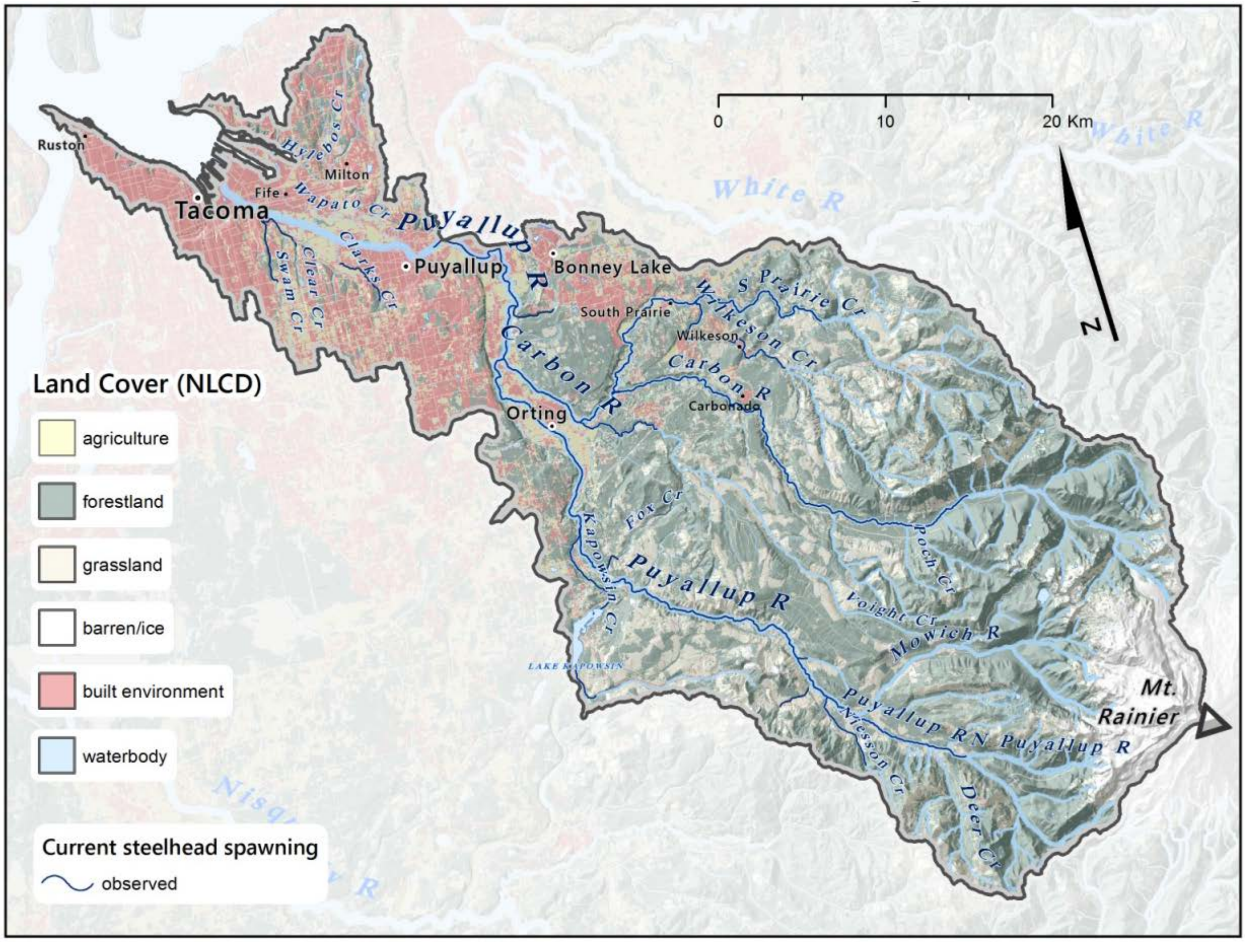

Figure E-61. Map of Puyallup/Carbon Rivers Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Multiple large and medium	9,190*	326 (178–596)	14,716	29,432
Year	1930	1950	2005–2009	—	

* Catch from the mainstem Puyallup River and Carbon River, which may include White River steelhead.

The Puyallup/Carbon Rivers Winter Run’s exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 0.919 (0.899–0.938).

Tributary	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
Mainstem Puyallup	119	78	52	91	64	139	91	133	51	74
Carbon River	358	248	235	410	98	323	418	367	190	398

For a state-space analysis of population viability, see the top two plots of Figure A-10 in Appendix A.

Diversity—

Puyallup/Carbon Rivers Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Clarks Creek	1996–2009	6	Puyallup H. (CC)	0	67,550
Diru Creek	1995–2000	6	Puyallup Tribal H. (CC)	0	592,091
Puyallup River	1995–1998	4	Tokul Creek H. (CC)	0	369,761
Viola Creek	2009	1	Voight Creek H. (CC)	0	187,550
Voight Creek	1995–2009	15	Voight Creek H. (CC)	0	2,214,396
			Total:	0	3,431,348

Spatial structure and hydrology—Figure E-62 and Figure E-63.

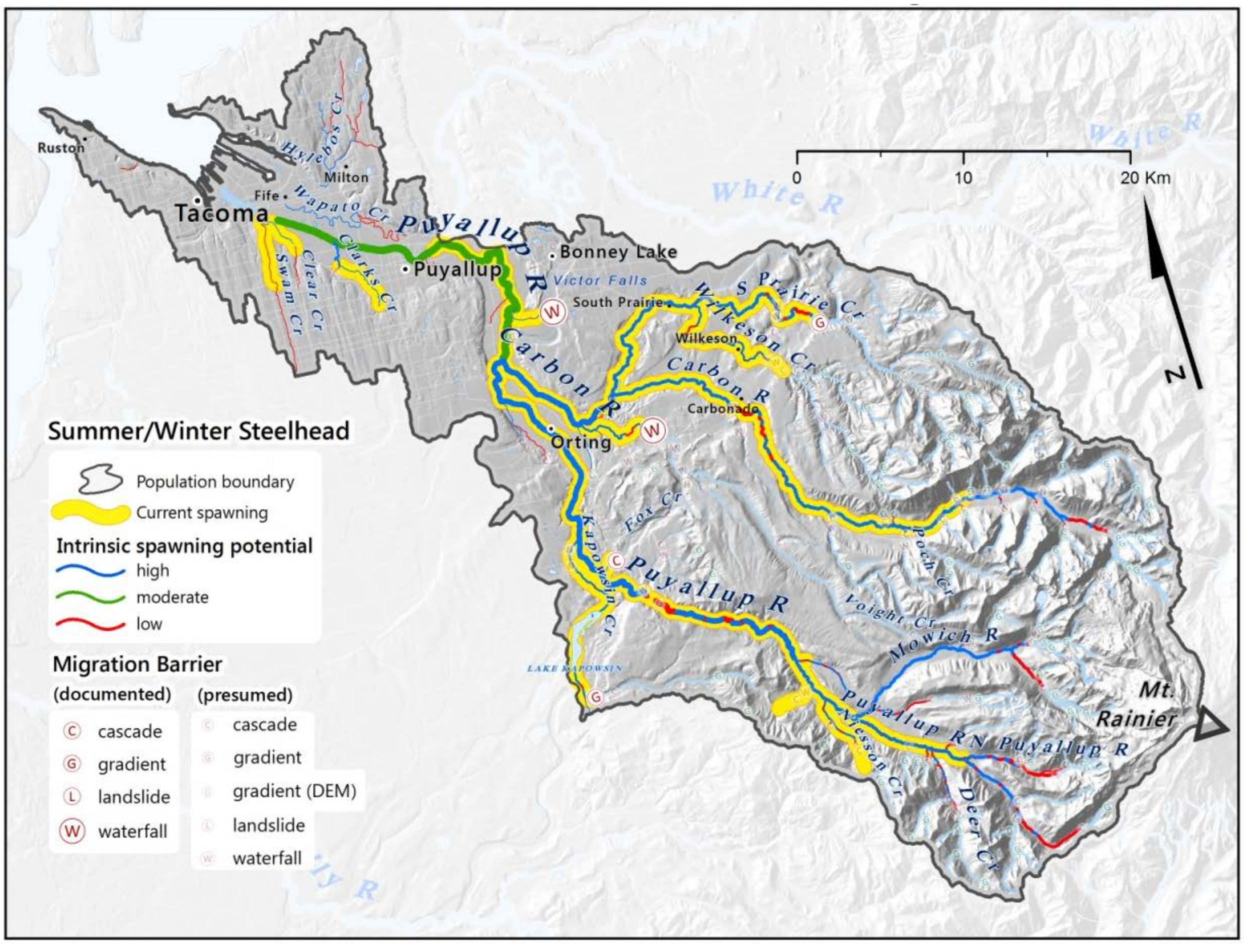

Figure E-62. Map of Puyallup/Carbon Rivers Winter-Run DIP spatial structure, including migration barriers and spawning potential.

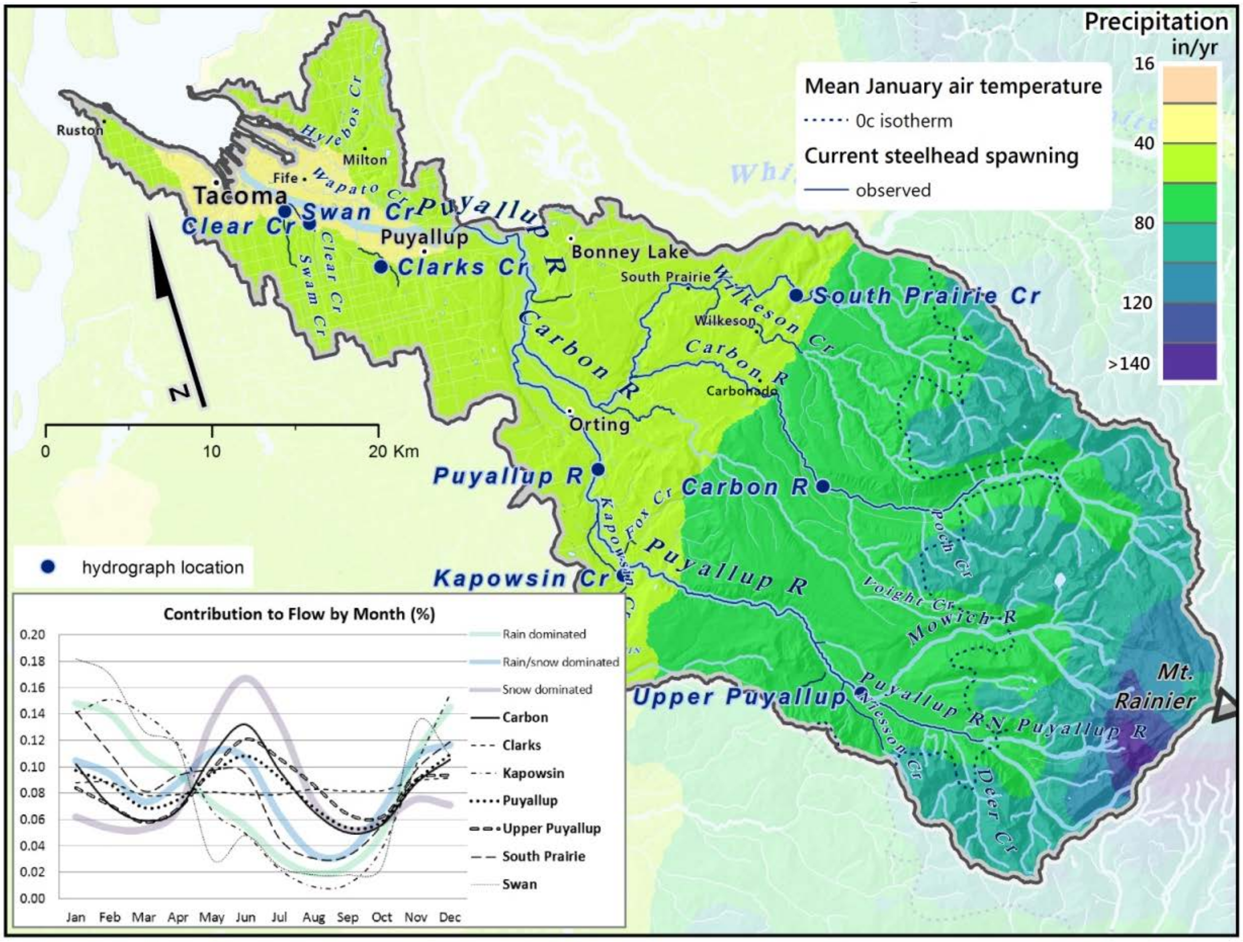

Figure E-63. Map of Puyallup/Carbon Rivers Winter-Run DIP area hydrology.

21. White River Winter Run

Overview—This population is one Salmonid Stock Inventory stock (WDFW 2002), the White River (Figure E-64). The Technical Recovery Team determined this population begins at the confluence of the White and Puyallup rivers. This DIP includes a number of glacially influenced tributaries.

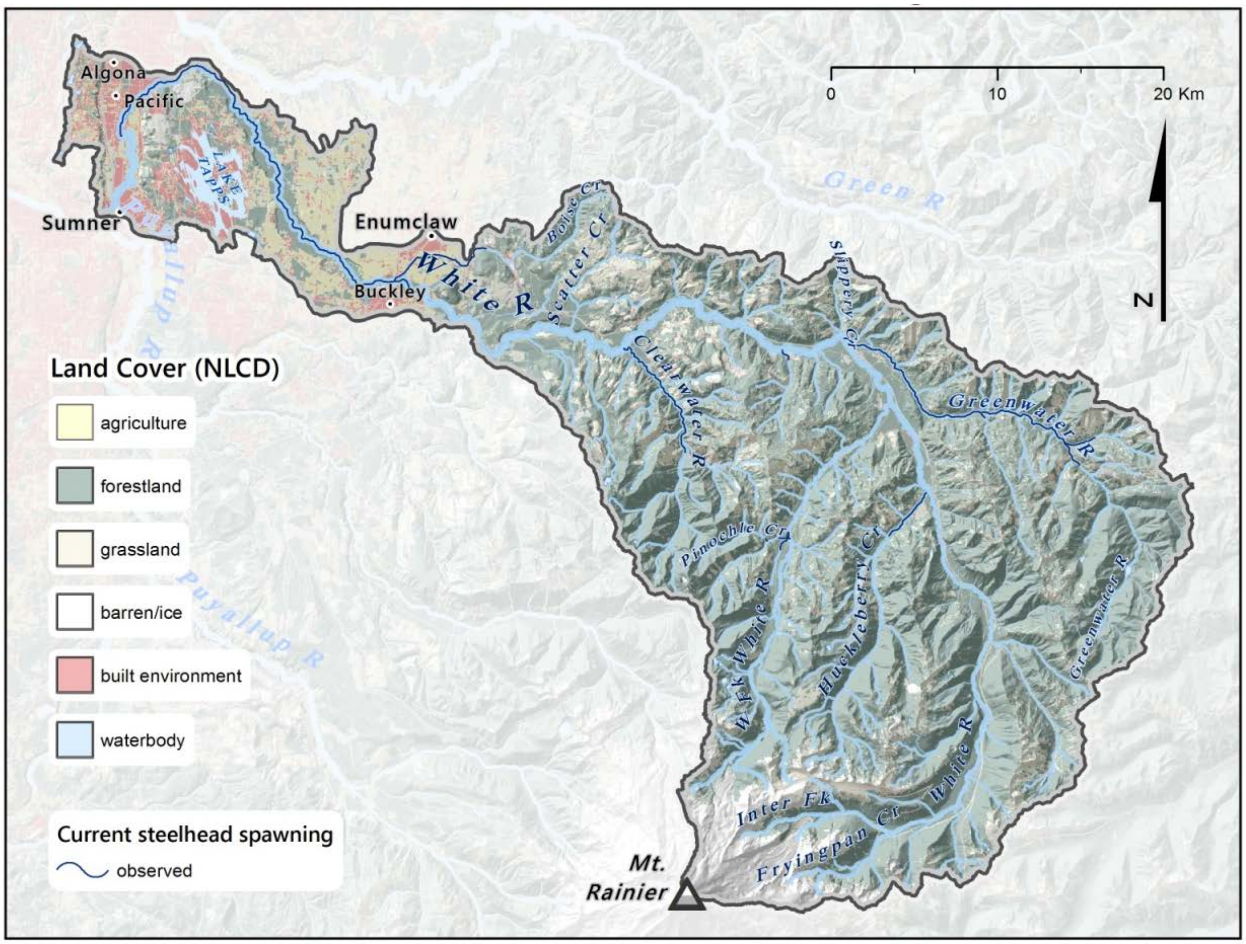

Figure E-64. Map of White River Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Medium	827*	265 (206–342)	17,490	34,981
Year	1930	1952	2005–2009	—	

* Does not include White River steelhead caught in the mainstem Puyallup River.

The White River Winter Run's exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 0.938 (0.923–0.952).

For a state-space analysis of population viability, see the top two plots of Figure A-11 in Appendix A.

Diversity—

White River Winter Run hatchery releases					Total releases	
Watershed	Duration	Years	Source		Within DPS	Outside DPS
White River	1995–2010	9	Puyallup H. (CC)		0	205,979
				Total:	0	205,979

Spatial structure and hydrology—Figure E-65 and Figure E-66.

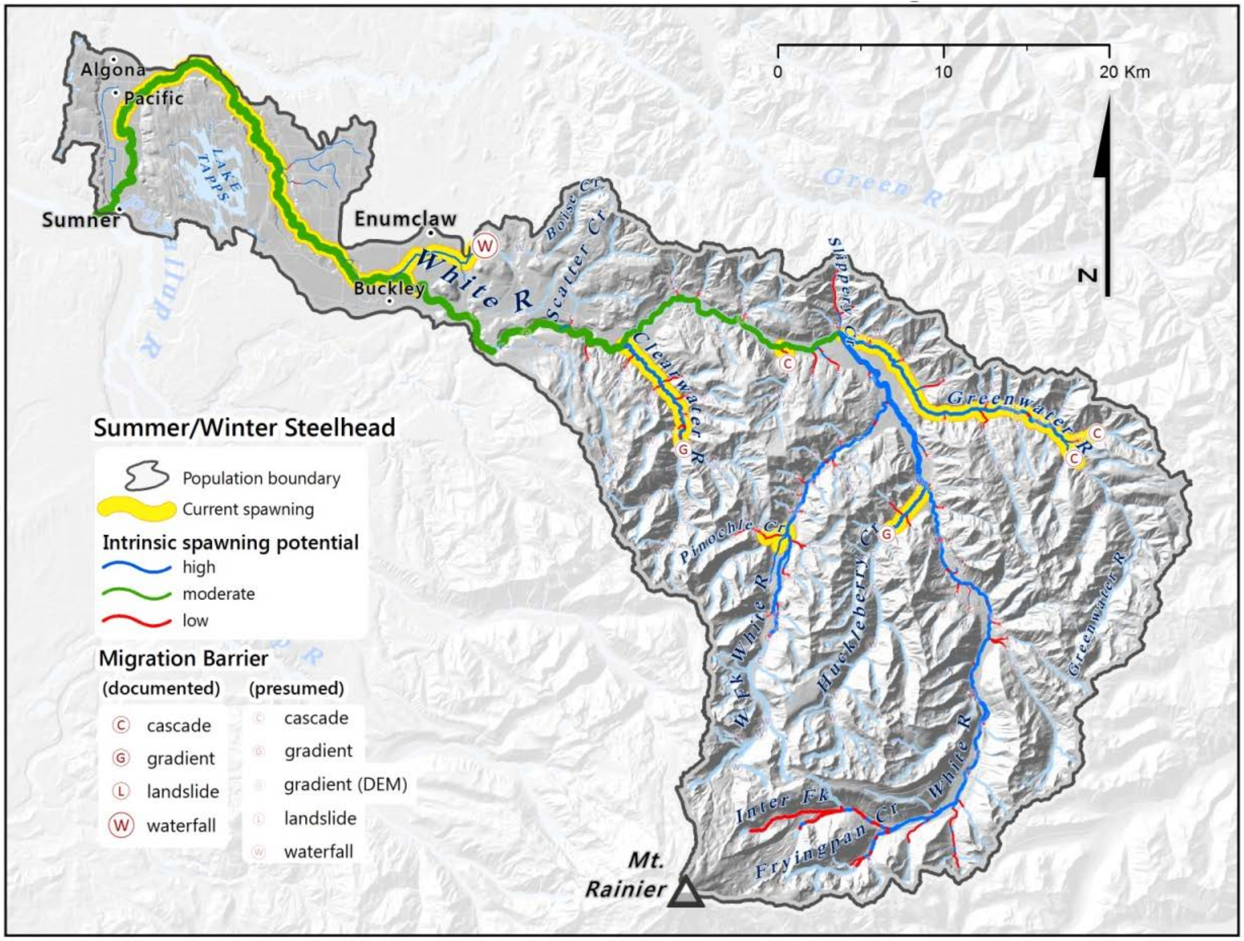

Figure E-65. Map of White River Winter-Run DIP spatial structure, including migration barriers and spawning potential.

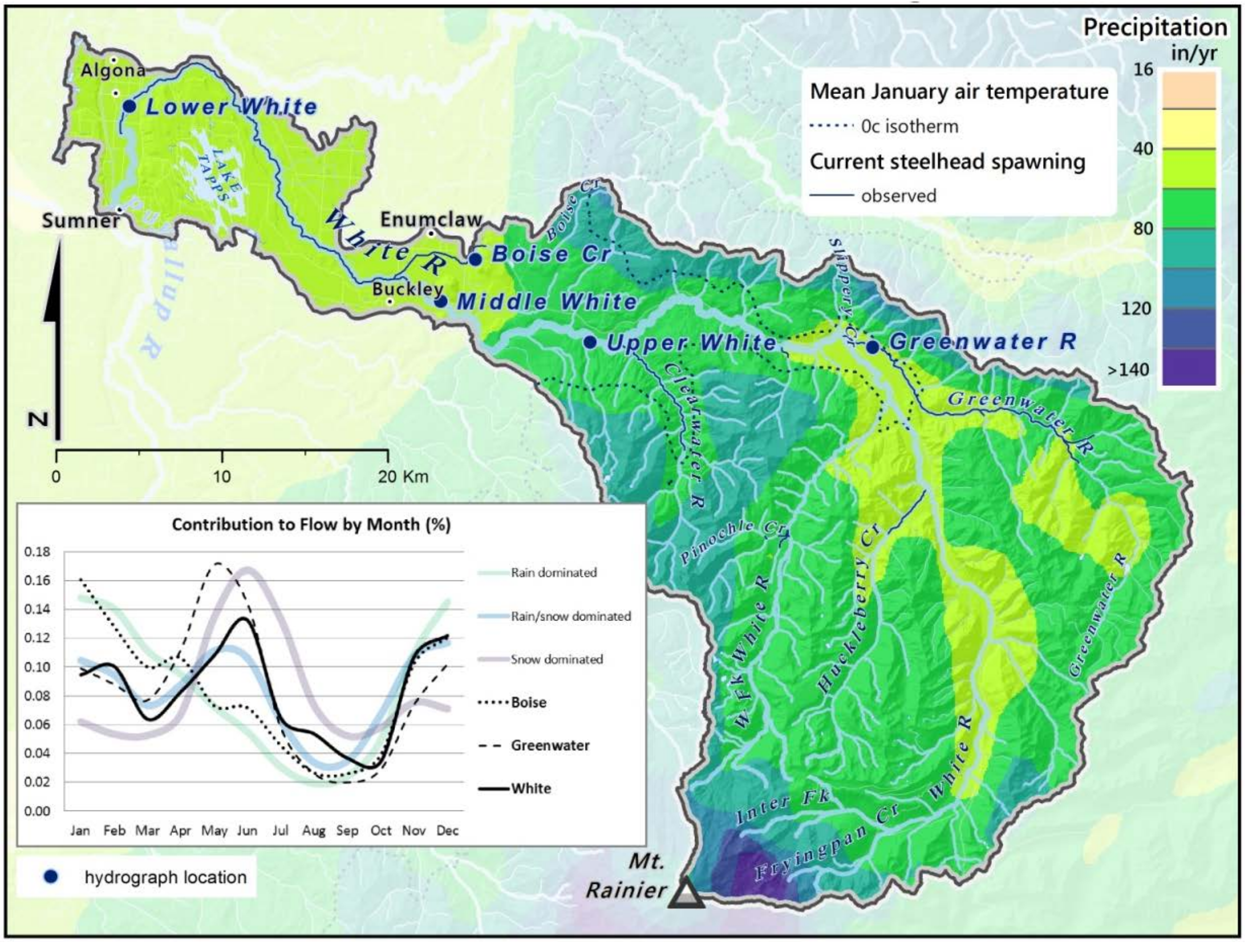

Figure E-66. Map of White River Winter-Run DIP area hydrology.

22. Nisqually River Winter Run

Overview—In contrast to the rest of the southern Puget Sound region, which is predominantly lowland stream habitat, the Nisqually River (Figure E-67) drains the glacial slopes of Mt. Rainier. The Technical Recovery Team considered that ecological differences between the Nisqually River and surrounding South Puget Sound tributaries are sufficient to result in life history and reproductive differentiation. Anecdotal information suggests that the Nisqually River steelhead population was quite large.

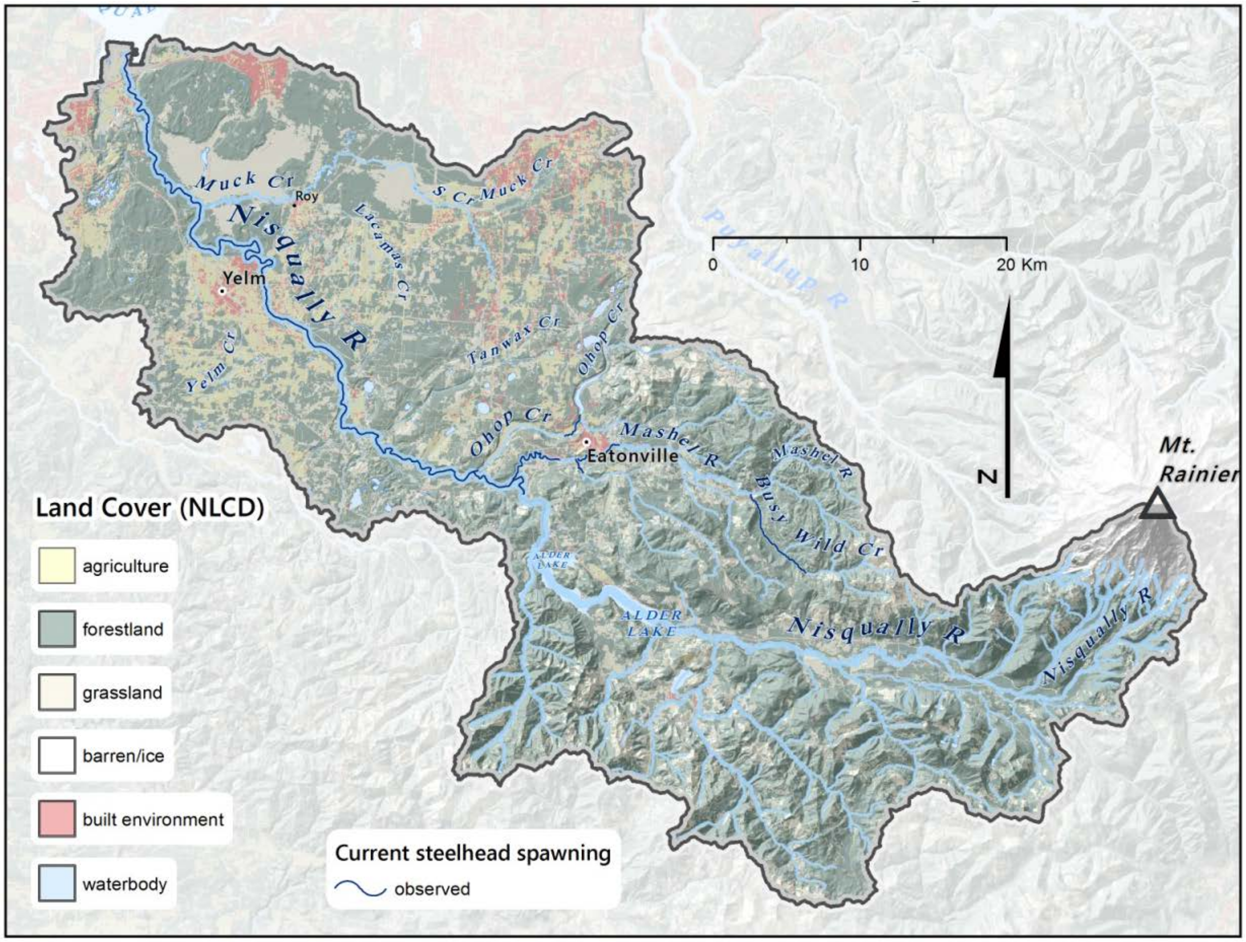

Figure E-67. Map of Nisqually River Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Multiple medium	2,769	402 (178–908)	15,330	30,660
Year	1930	1952	2005–2009	—	

The Nisqually River Winter Run's exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 0.914 (0.890–0.940).

For a state-space analysis of population viability, see the top two plots of Figure A-12 in Appendix A.

Diversity—

Nisqually River Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Lake St. Clair	1998	1	Skykomish Hatchery	0	11,494
			Total:	0	11,494

Spatial structure and hydrology—Figure E-68 and Figure E-69.

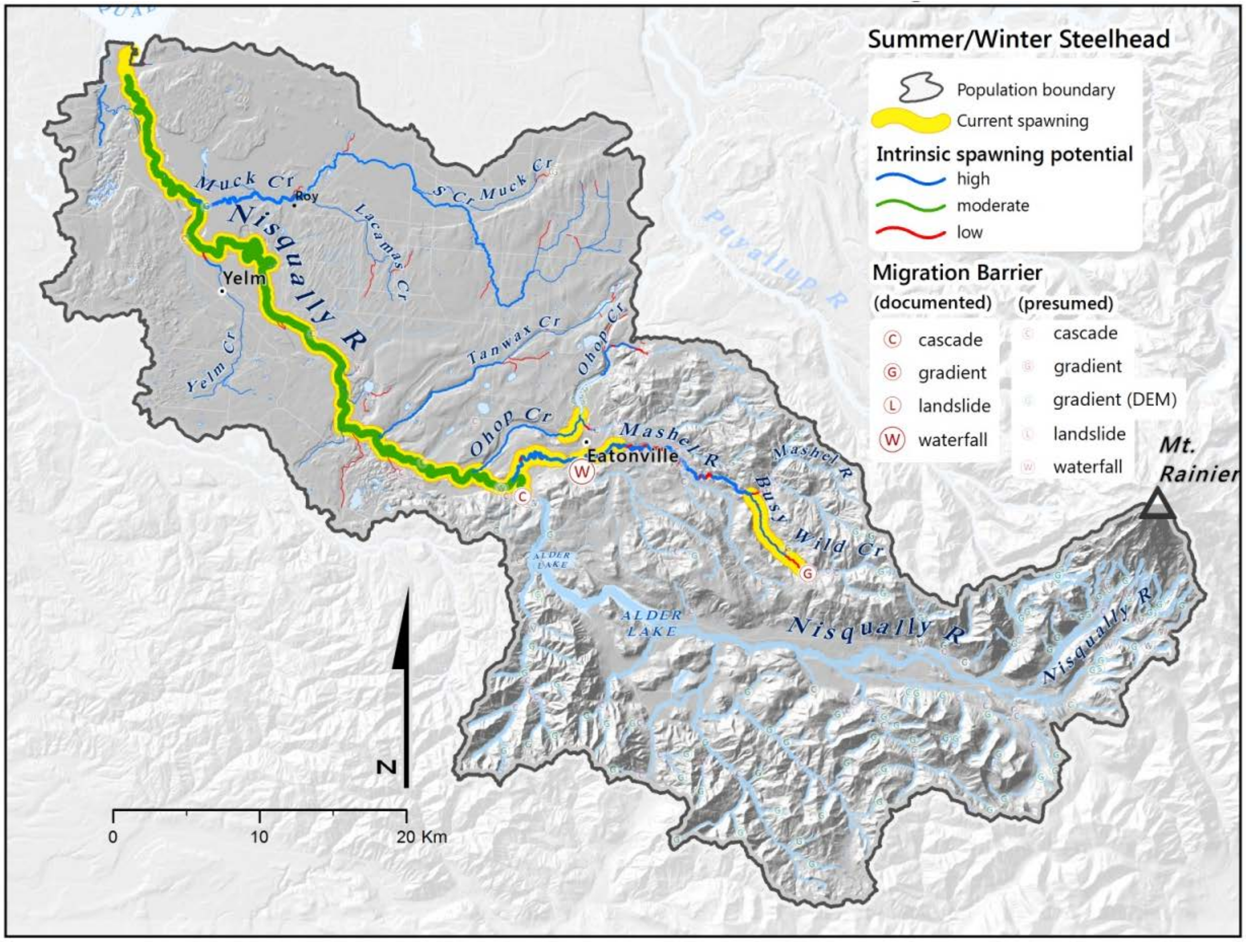

Figure E-68. Map of Nisqually River Winter-Run DIP spatial structure, including migration barriers and spawning potential. Historically, a series of cascades near the present site of the La Grande and Alder dams may have been a seasonal barrier, but they also could have posed a complete barrier to fish passage.

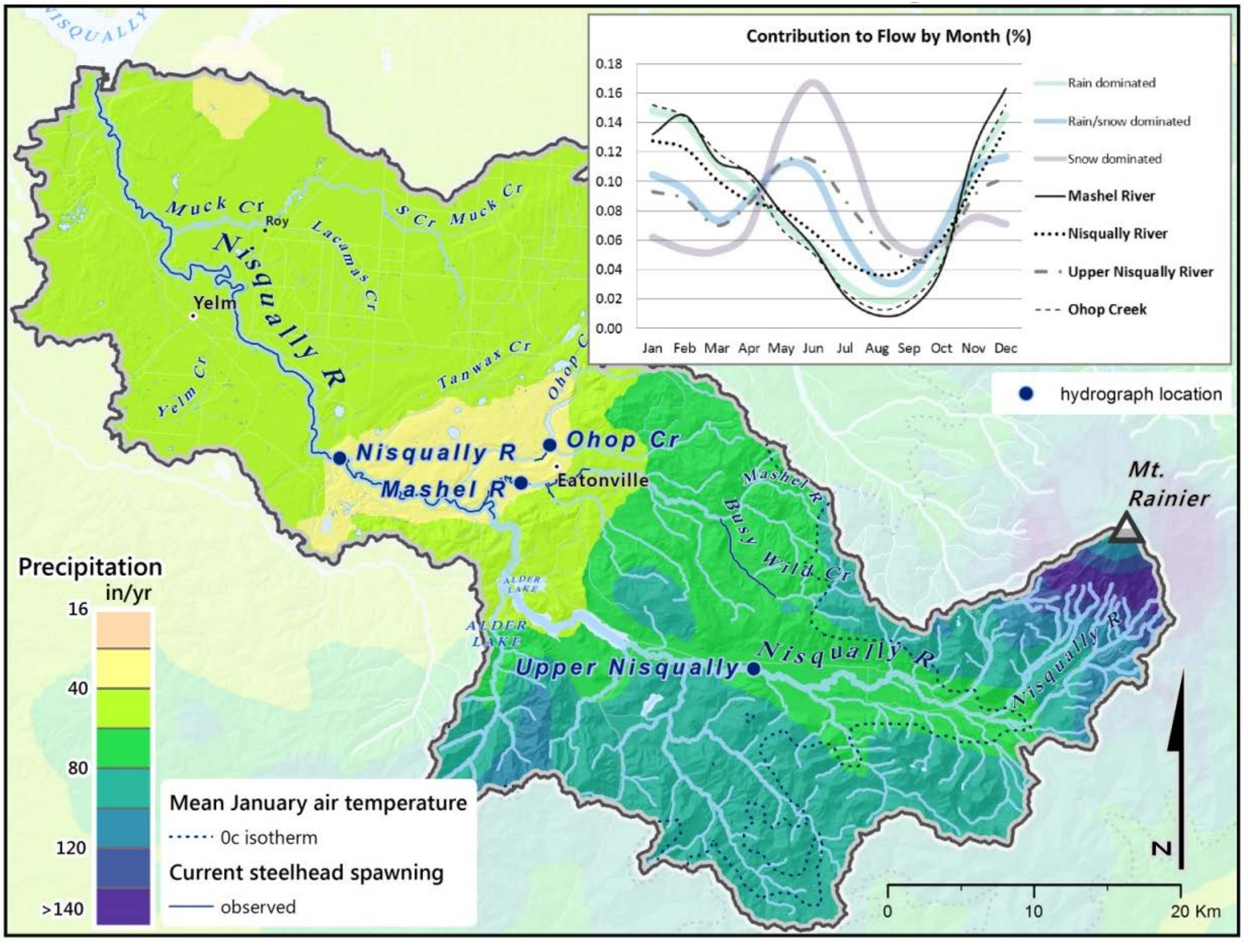

Figure E-69. Map of Nisqually River Winter-Run DIP area hydrology.

23. South Puget Sound Tributaries Winter Run

Overview—This population includes four Salmonid Stock Inventory stocks (WDFW 2002): Chambers Creek, Eld Inlet, Totten Inlet, and Hammersley Inlet and Case/Carr Inlet—effectively all of the lowland tributaries entering into South Puget Sound (Figure E-70). Little current or historical information is available on abundance or life history characteristics for this population.

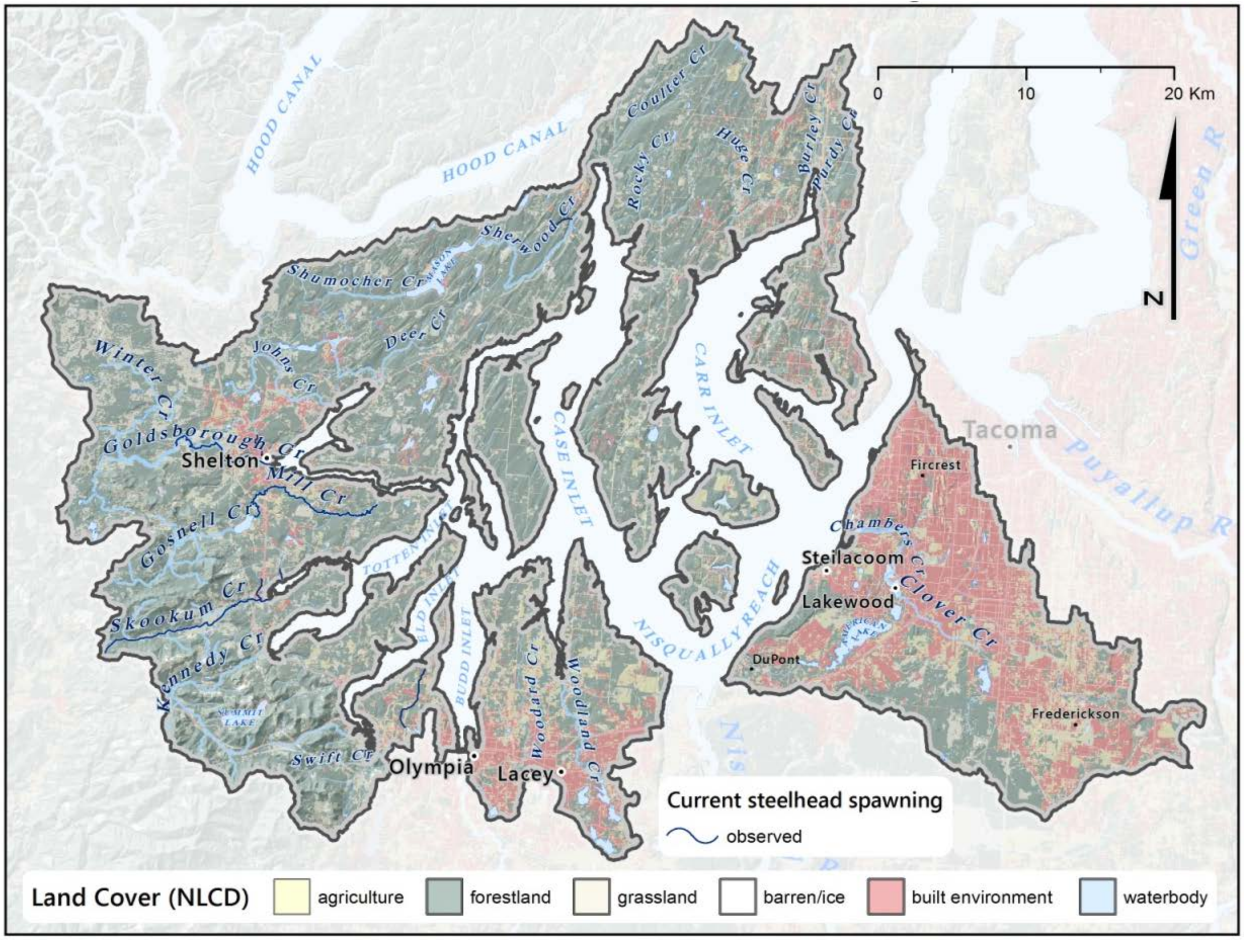

Figure E-70. Map of South Puget Sound Tributaries Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate	
Abundance	Not available	85*	9,854	19,709
Year	—	—	—	

* Compiled from a number of different streams draining to South Puget Sound: Coulter Creek, Goldsborough Creek, Kennedy Creek, Mill Creek, Percival Creek, and Sherwood Creek.

Diversity—

South Puget Sound Tributaries Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Kennedy Creek	1996	1	Bogachiel H. (CC)	0	9,996
Deschutes R.	1996–2005	9	Bogachiel/Tokul (CC)	0	290,654
Chambers Cr.	1995–1996	2	Chambers Creek H.	0	90,887
			Total:	0	391,537

Spatial structure and hydrology—Figure E-71 and Figure E-72.

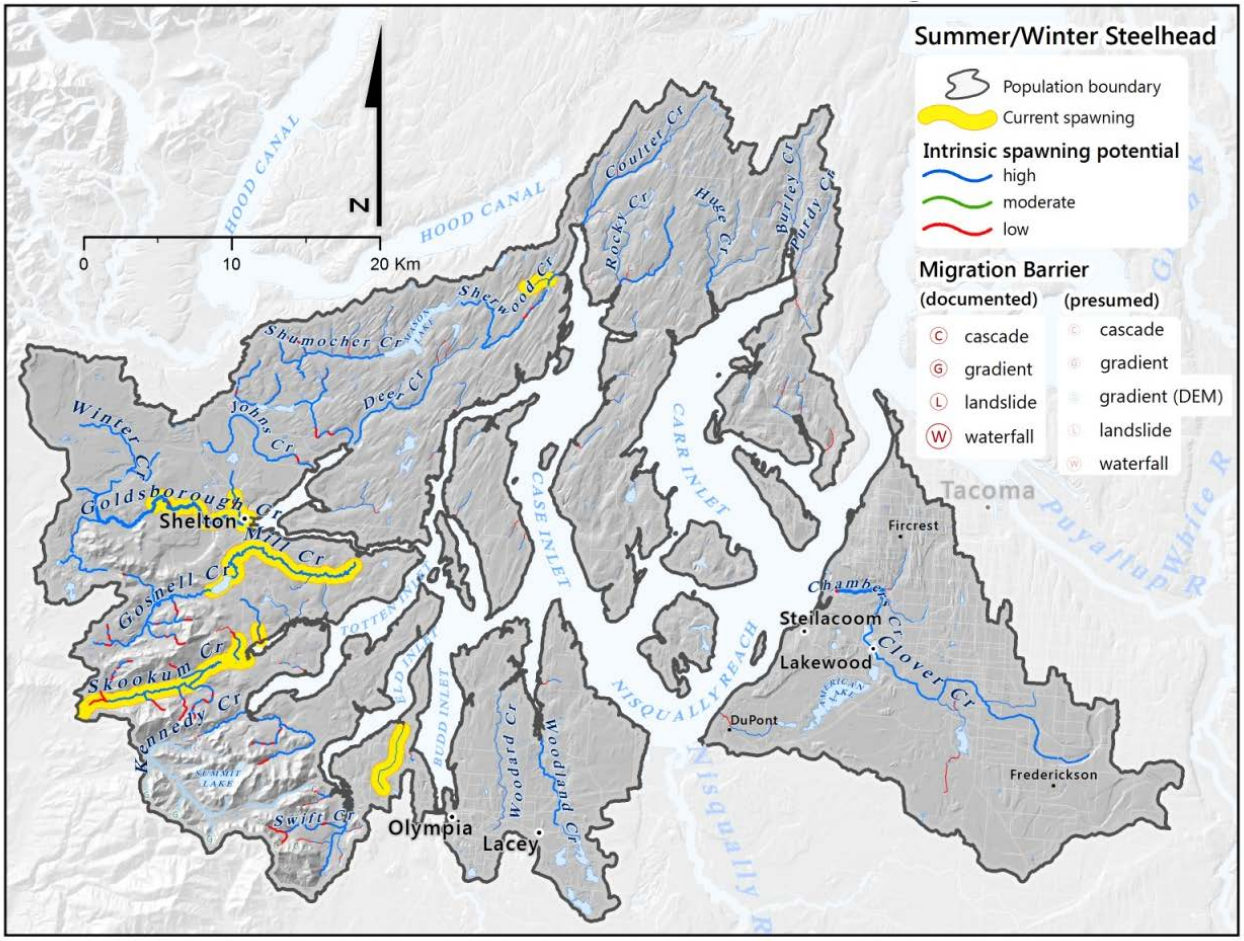

Figure E-71. Map of South Puget Sound Tributaries Winter-Run DIP spatial structure, including migration barriers and spawning potential.

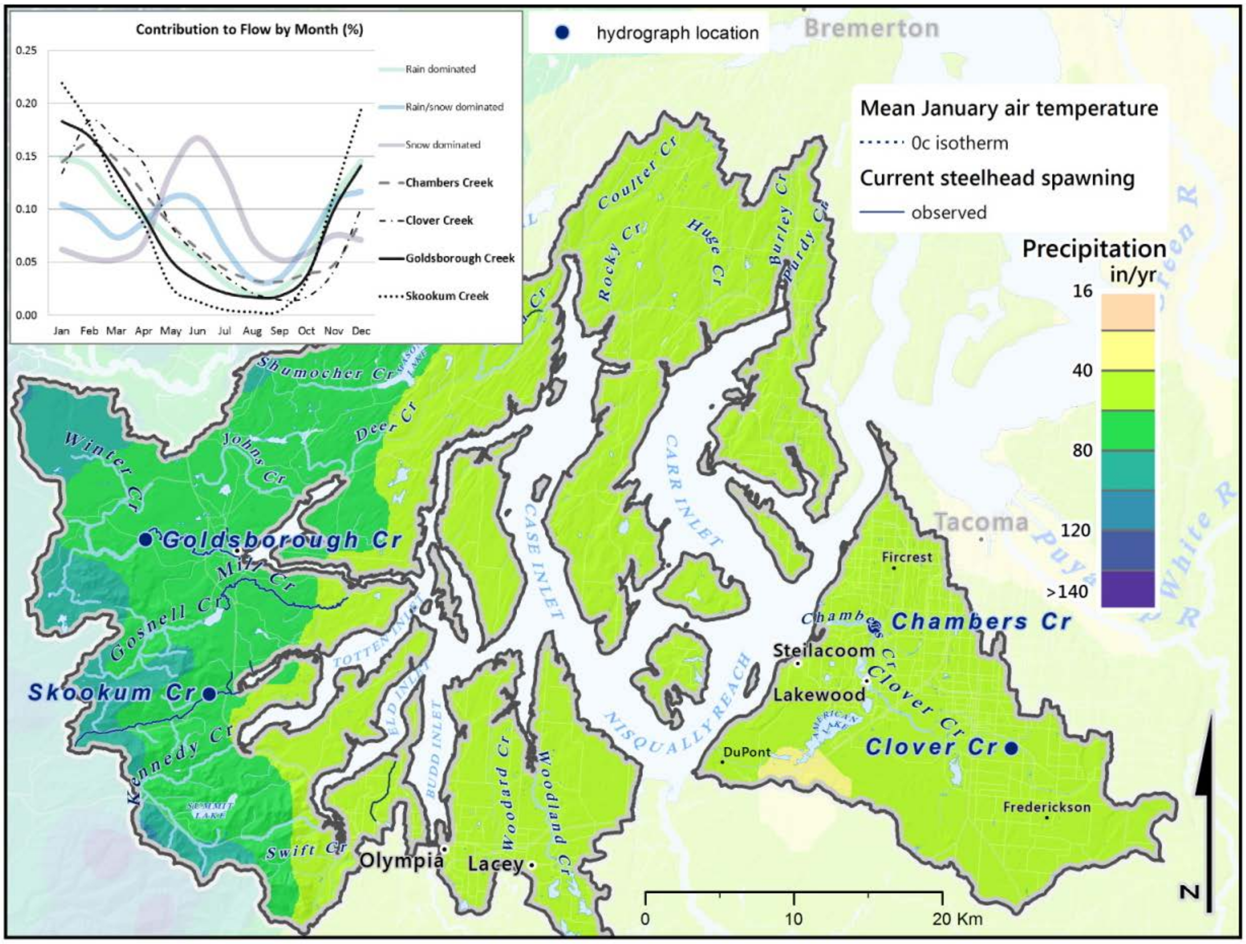

Figure E-72. Map of South Puget Sound Tributaries Winter-Run DIP area hydrology.

24. East Kitsap Peninsula Tributaries Winter Run

Overview—This population includes small lowland independent tributaries on the east side of the Kitsap Peninsula (Figure E-73). There is limited information, other than presence, for the east side of the Kitsap Peninsula, with the exception of Curley Creek.

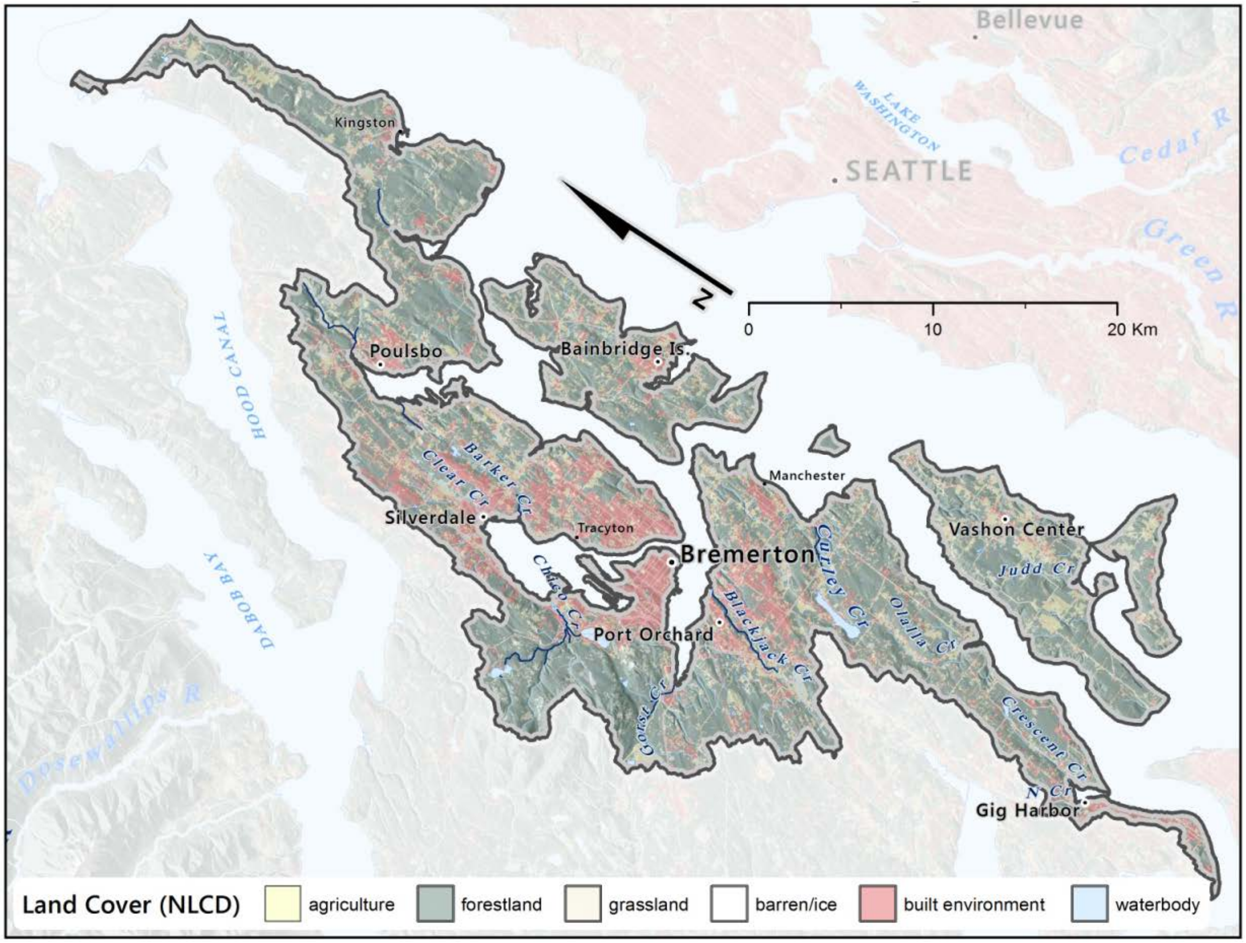

Figure E-73. Map of East Kitsap Peninsula Tributaries Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate	
Abundance	Not available	53	1,557	3,115
Year	—	1963	—	

Diversity—Hatchery releases for the East Kitsap Peninsula Tributaries Winter Run are zero within the DPS and zero outside the DPS.

Spatial structure and hydrology—Figure E-74 and Figure E-75.

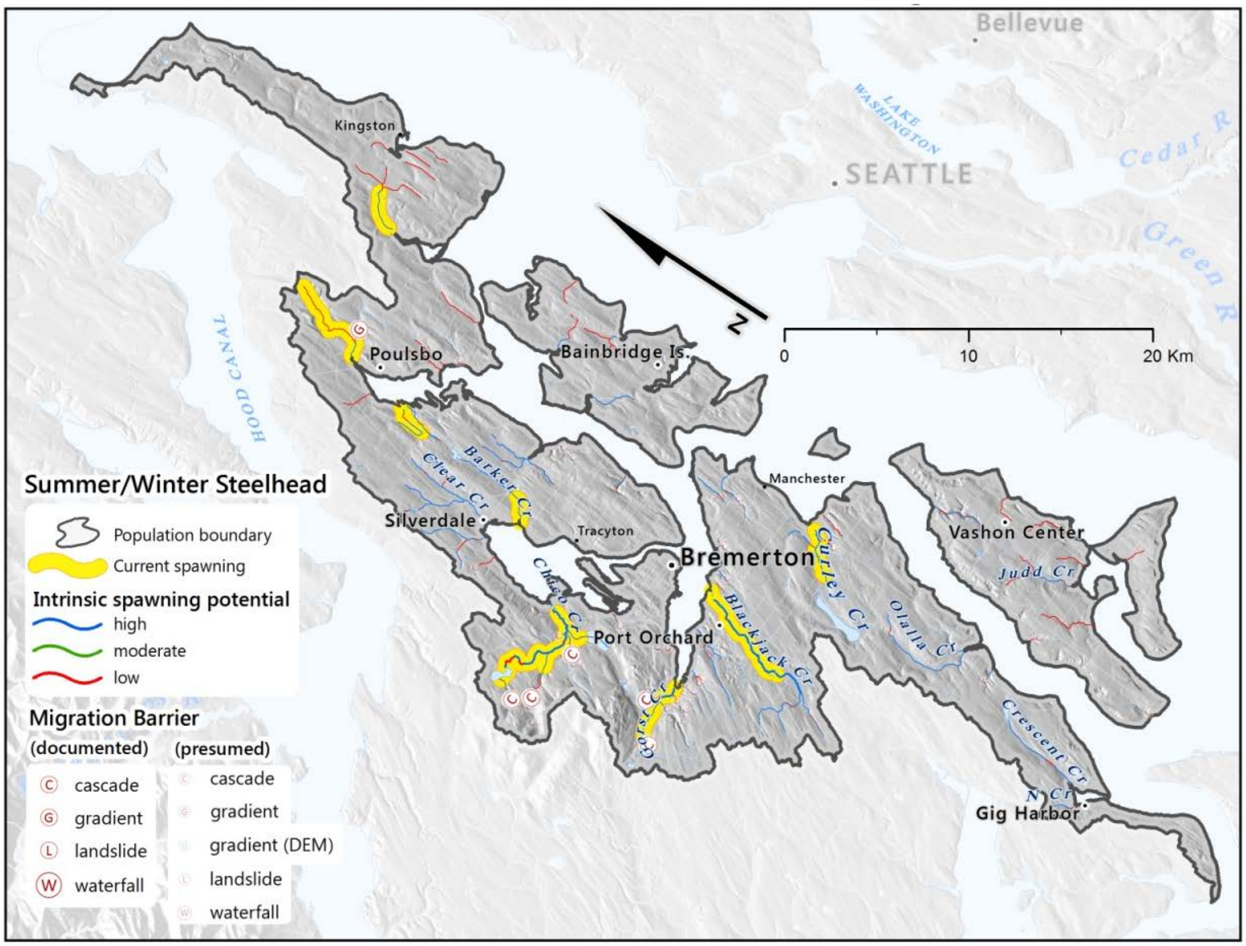

Figure E-74. Map of East Kitsap Peninsula Tributaries Winter-Run DIP spatial structure, including migration barriers and spawning potential.

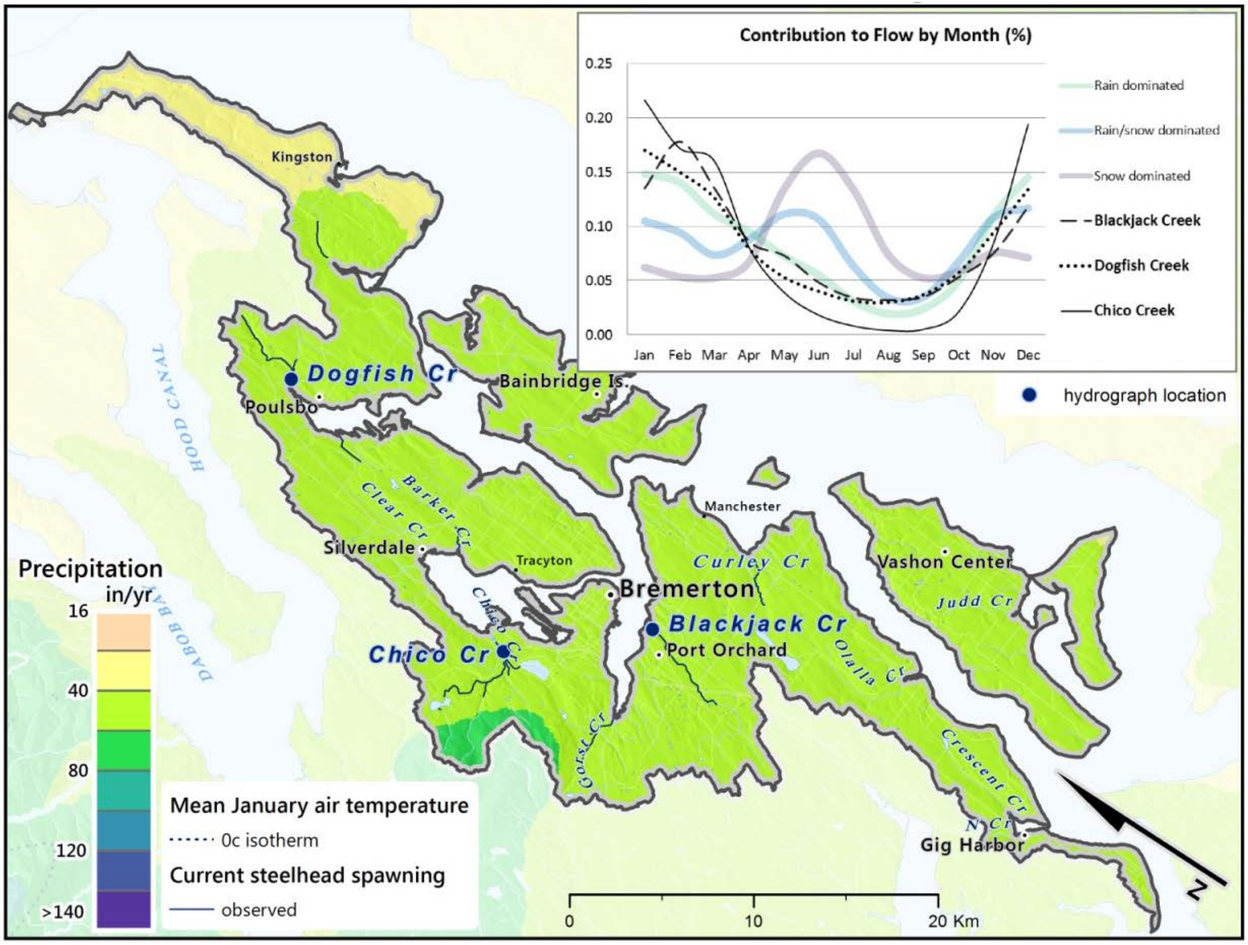

Figure E-75. Map of East Kitsap Peninsula Tributaries Winter-Run DIP area hydrology.

Hood Canal and Strait of Juan de Fuca Major Population Group

25. East Hood Canal Tributaries Winter Run

Overview—This population includes winter steelhead spawning in small lowland independent tributaries on the west side of the Kitsap Peninsula (eastern shore of Hood Canal) from Point No Point to the southern end of Hood Canal (Alderbrook and Twanoh creeks). The primary streams in this DIP include Big Beef Creek, Anderson Creek, and the Dewatto River (Figure E-76).

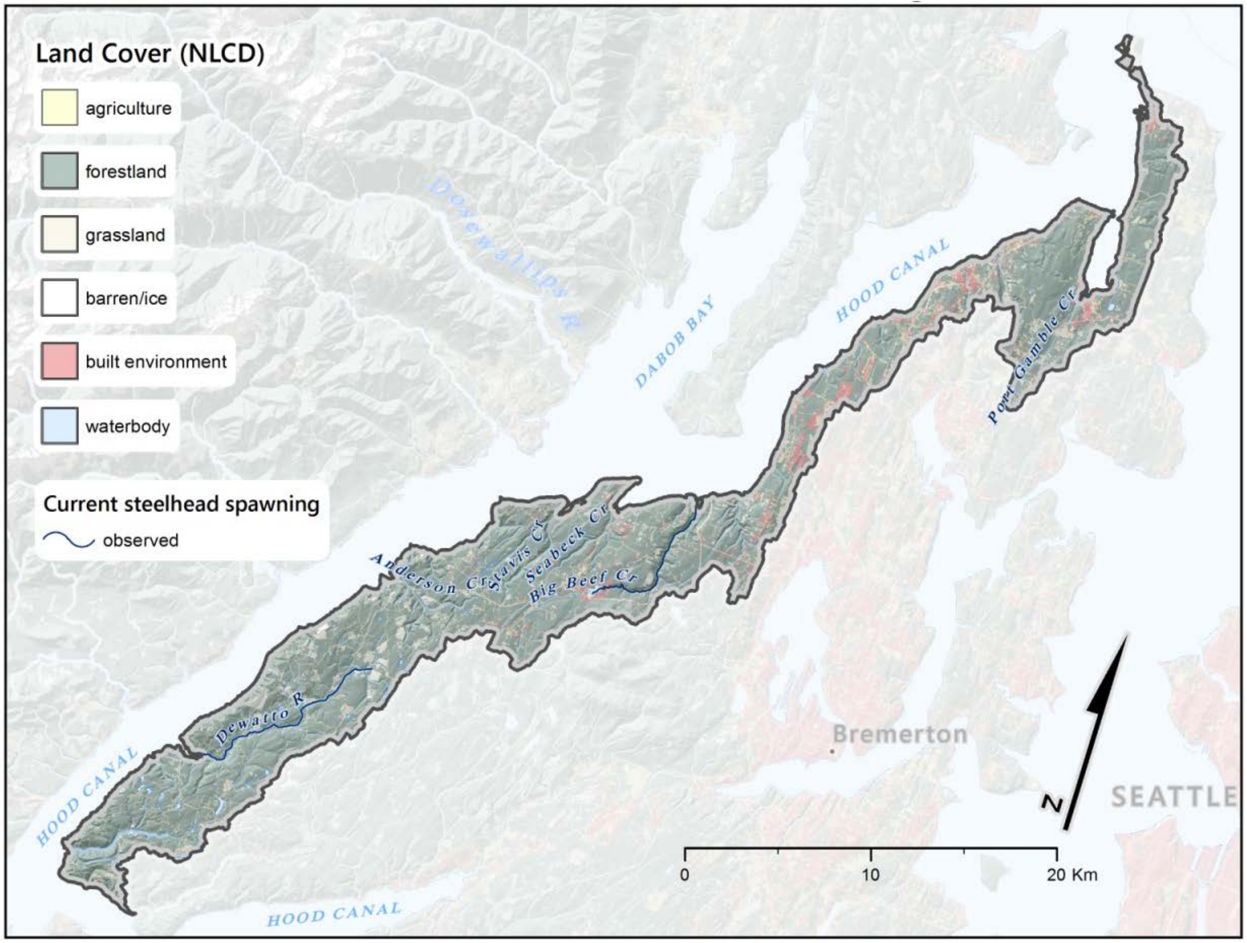

Figure E-76. Map of East Hood Canal Tributaries Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Not available	327*	213 (122–372)	1,270	2,540
Year	—	1958	2005–2009	—	

* Includes estimates from Big Beef Creek and Dewatto River.

The East Hood Canal Tributaries Winter Run's exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 1.022 (0.997–1.048).

Tributary	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Dewatto River	30	18	39	23	53	28	49	15	13	92

For a state-space analysis of population viability, see Figure A-15 in Appendix A.

Diversity—

East Hood Canal Tributaries Winter Run hatchery releases					Total releases	
Watershed	Duration	Years	Source		Within DPS	Outside DPS
Dewatto River	2009–2010	2	Native		14,183	0
				Total:	14,183	0

Spatial structure and hydrology—Figure E-77 and Figure E-78.

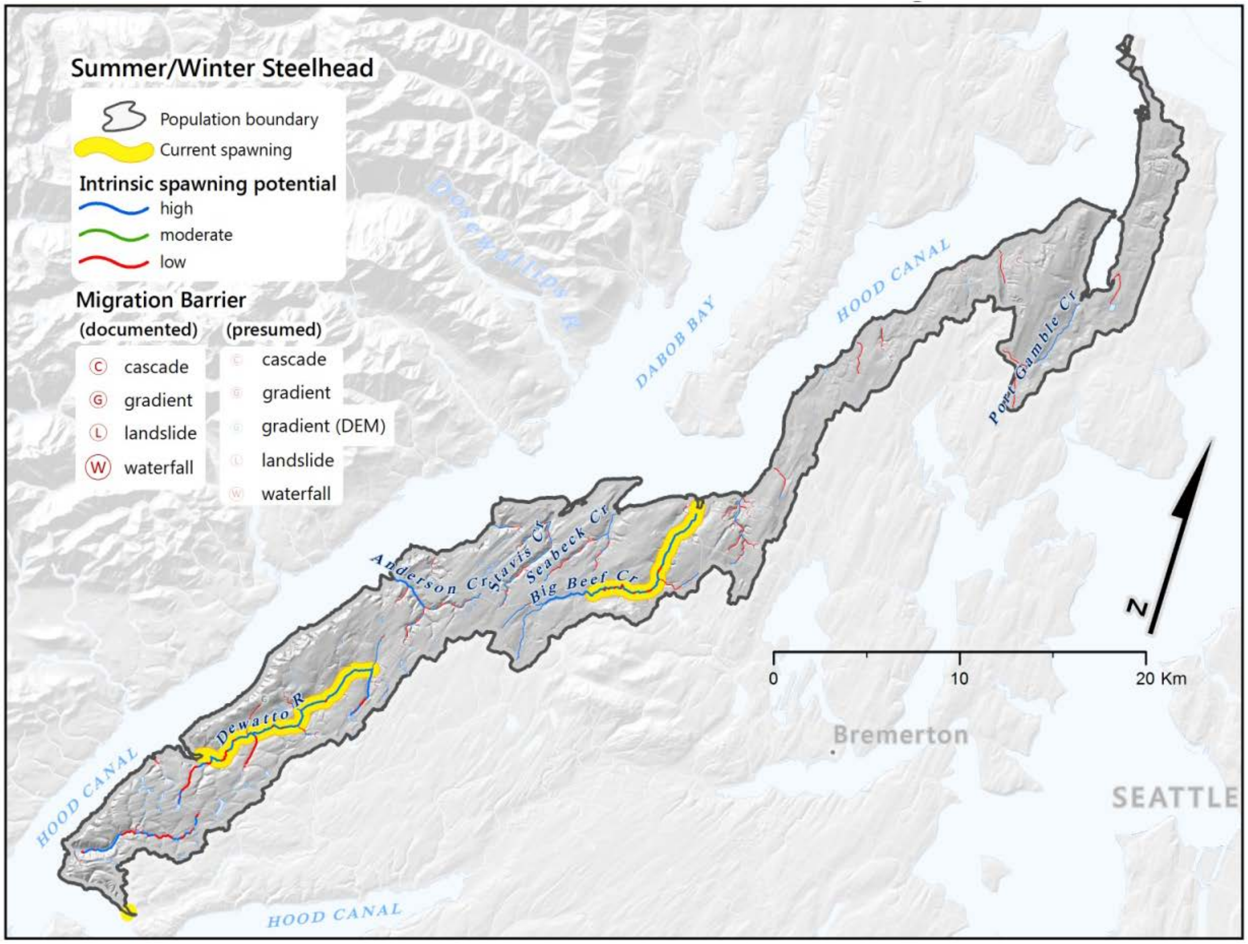

Figure E-77. Map of East Hood Canal Tributaries Winter-Run DIP spatial structure, including migration barriers and spawning potential.

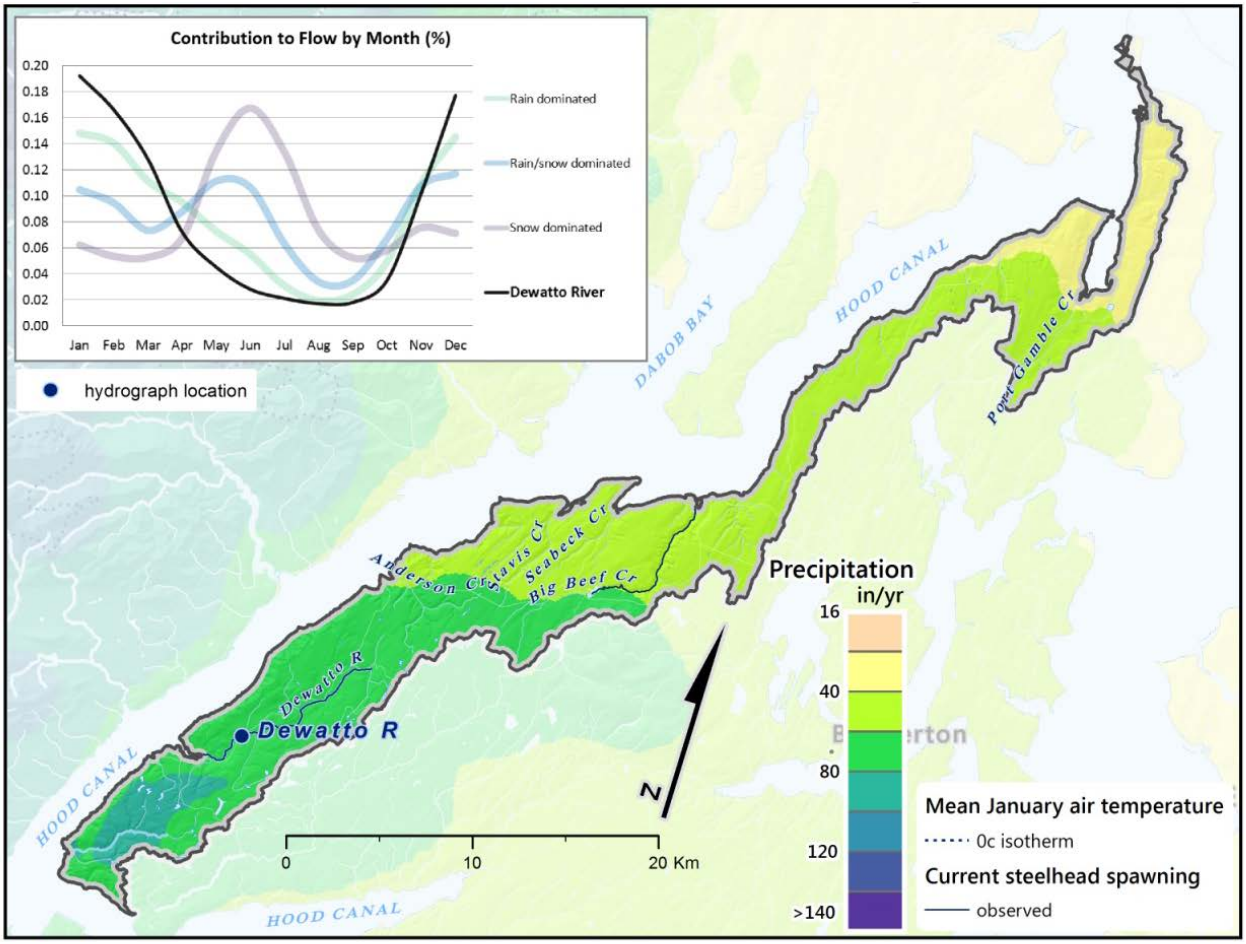

Figure E-78. Map of East Hood Canal Tributaries Winter-Run DIP area hydrology.

26. South Hood Canal Tributaries Winter Run

Overview—This population includes winter steelhead spawning in independent tributaries on the southwest side of the Kitsap Peninsula (eastern shore of Hood Canal), including the Tahuya and Union rivers to the southern end of Hood Canal (Alderbrook and Twanoh creeks). The primary streams in this DIP include the Tahuya and Union rivers (Figure E-79). Stream surveys conducted in 1932 give very general estimates of abundance with larger runs of steelhead in the Tahuya and Union rivers (WDG 1931–1940).

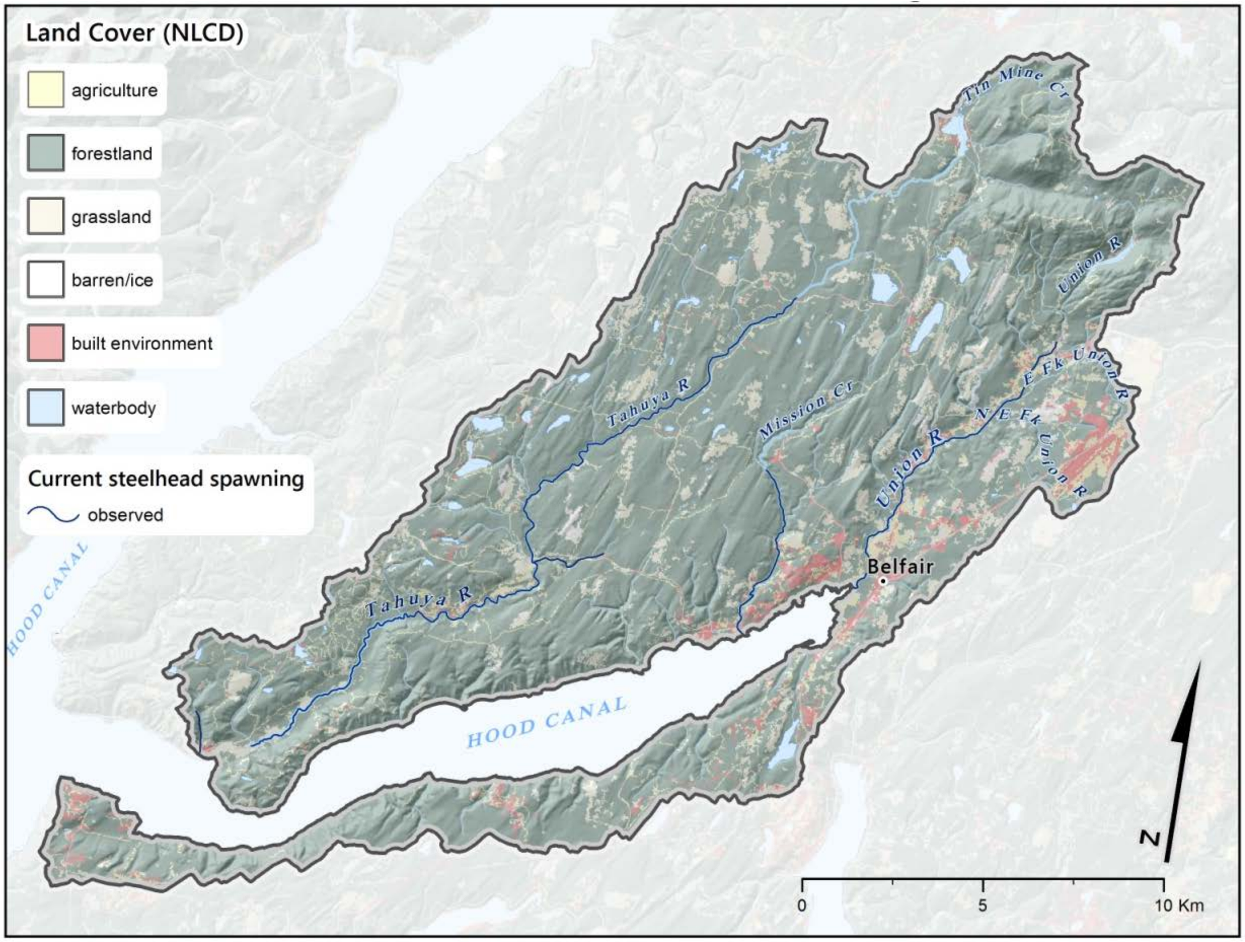

Figure E-79. Map of South Hood Canal Tributaries Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate	
Abundance	Not available	640	2,985	5,970
Year	—	1952	—	

Tributary	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Tahuya River	97	53	168	91	183	175	144	53	68	47
Union River	49	50	58	23	86	21	15	15	21	11

For a state-space analysis of South Hood Canal Tributaries Winter Run viability, see Figure A-13 in Appendix A. For a state-space analysis of Tahuya River winter-run population viability, see Figure A-14 in Appendix A.

Diversity—Hatchery releases for the South Hood Canal Tributaries Winter Run are zero within the DPS and zero outside the DPS.

Spatial structure and hydrology—Figure E-80 and Figure E-81.

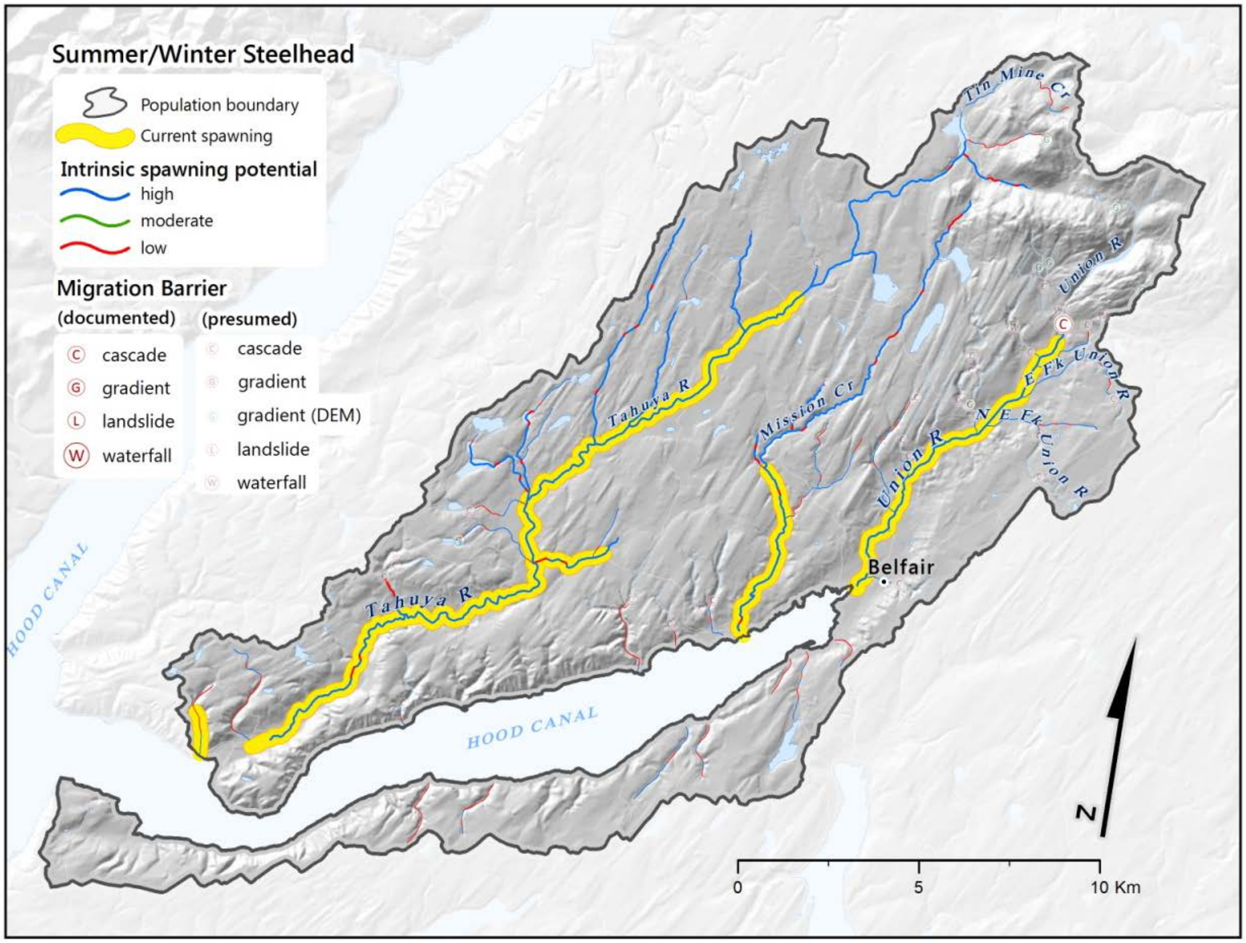

Figure E-80. Map of South Hood Canal Tributaries Winter-Run DIP spatial structure, including migration barriers and spawning potential.

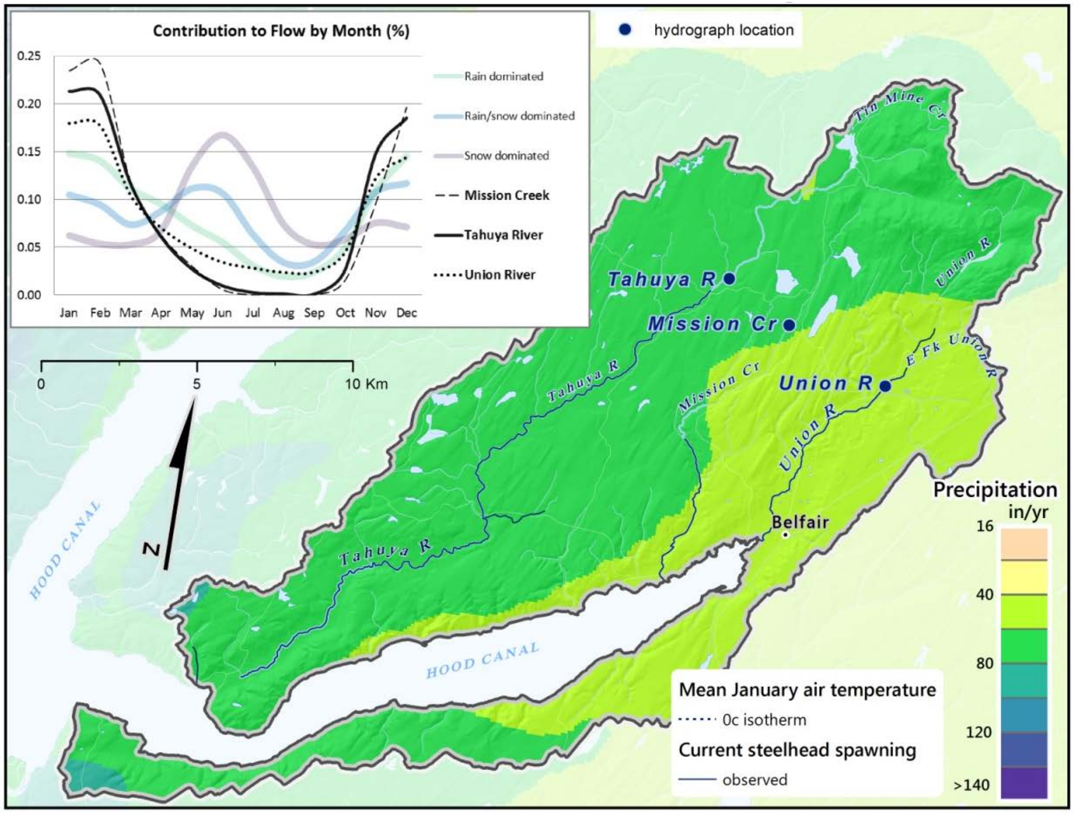

Figure E-81. Map of South Hood Canal Tributaries Winter-Run DIP area hydrology.

27. Skokomish River Winter Run

Overview—This population contains native winter-run steelhead in the north and south forks of the Skokomish River (Figure E-82). Much of the North Fork Skokomish River is currently inaccessible beyond Cushman Dam No. 2 (RKM 27.8). Steelhead currently utilize both lowland tributaries (for example Vance Creek) and headwater areas of the South Fork Skokomish River. There has been considerable debate as to whether winter-run steelhead had access beyond the series of falls in the lower North Fork Skokomish River; steelhead may have had access at least to the Staircase Rapids at RKM 48.1 (Williams et al. 1975).

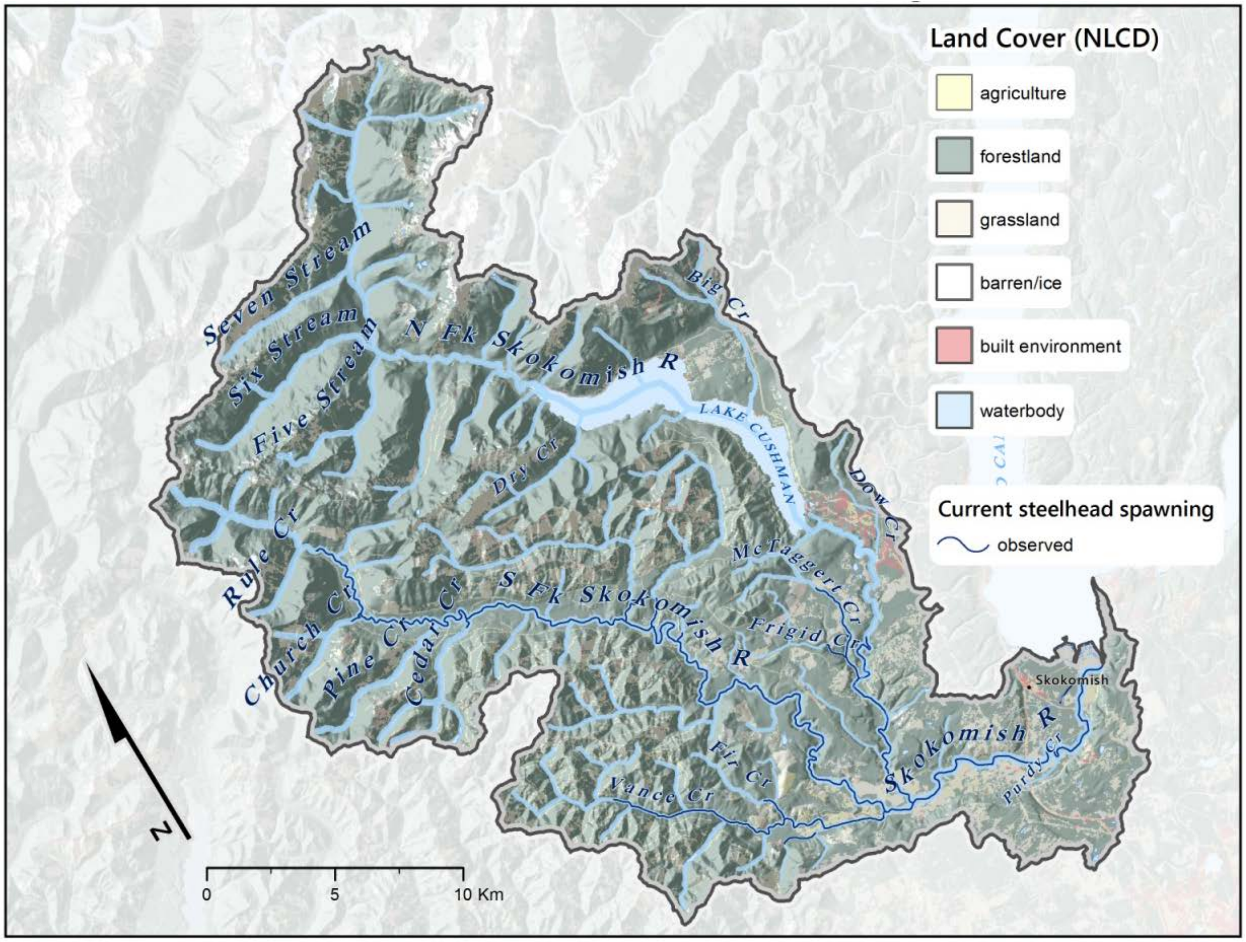

Figure E-82. Map of Skokomish River Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Not available	693	355 (183–686)	10,030	20,060
Year	—	1952	2005–2009	—	

The Skokomish River Winter Run's exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 0.956 (0.932–0.979).

For a state-space analysis of population viability, see the top two plots of Figure A-16 in Appendix A.

Diversity—

Skokomish River Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
South Fork Skokomish R.	1995–2011	11	Eells Springs (CC)	0	450,198
Hunter Creek	1996–1997	2	Bogachiel H. (CC)	0	43,351
			Total:	0	493,549

Spatial structure and hydrology—Figure E-83 and Figure E-84.

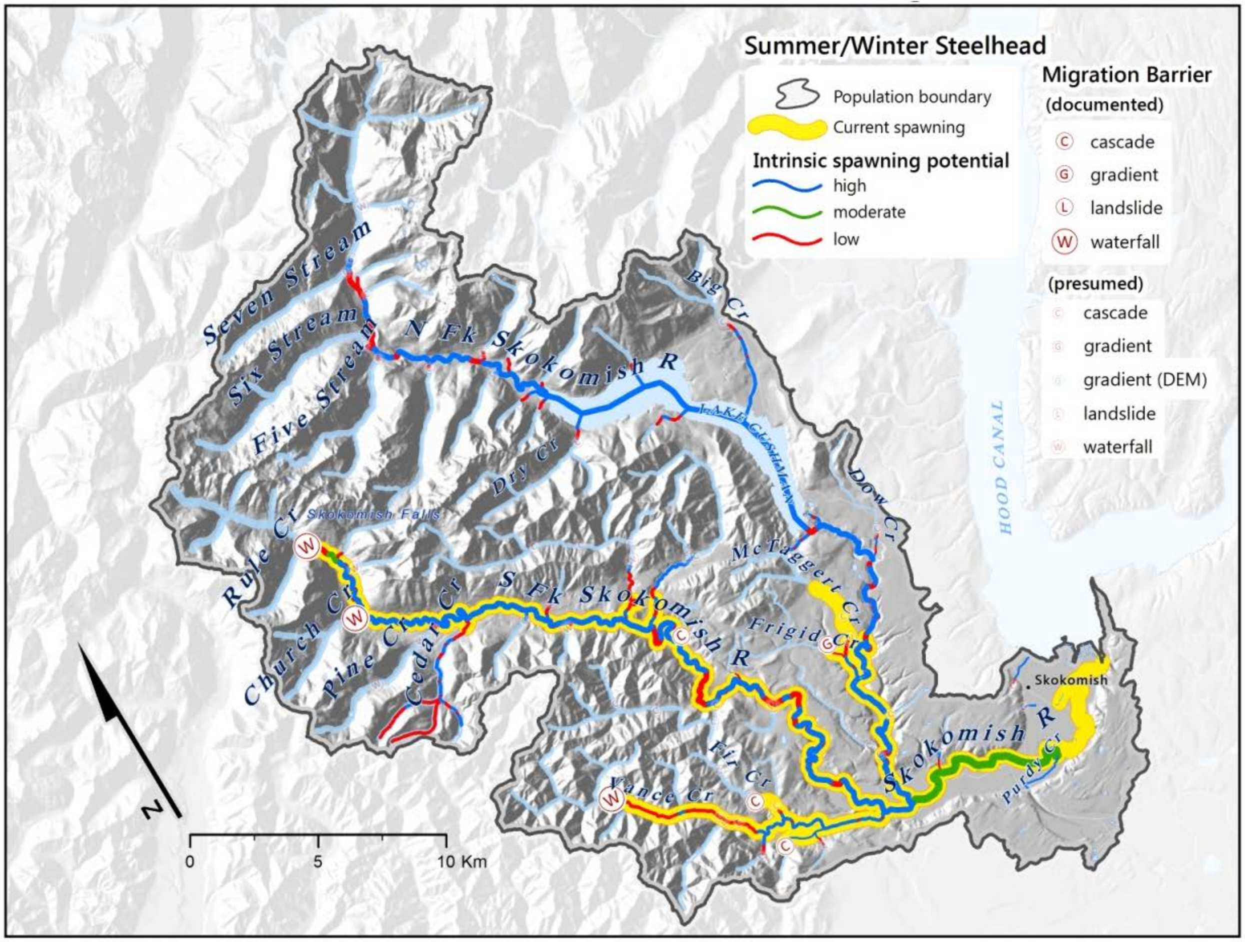

Figure E-83. Map of Skokomish River Winter-Run DIP spatial structure, including migration barriers and spawning potential.

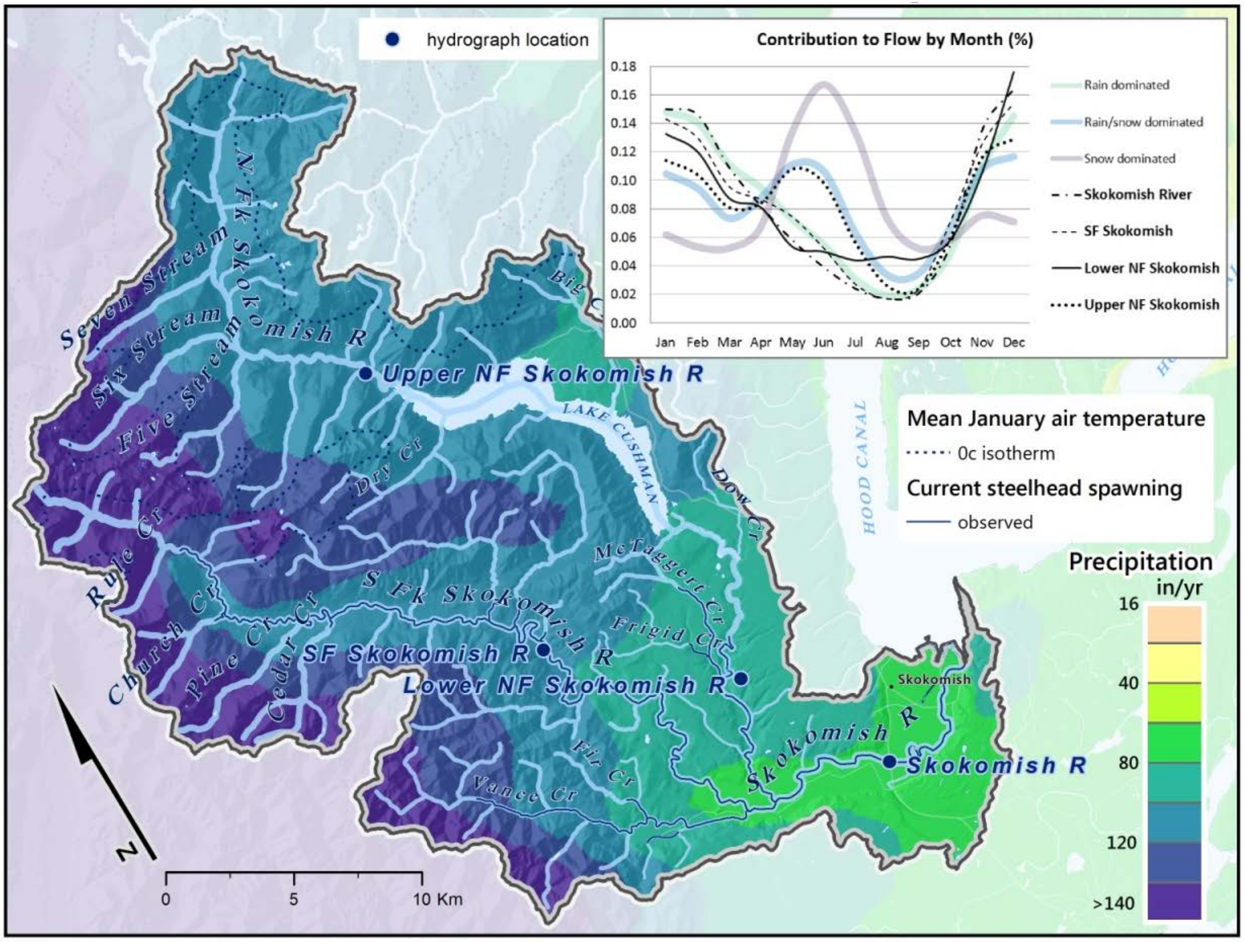

Figure E-84. Map of Skokomish River Winter-Run DIP area hydrology.

28. West Hood Canal Tributaries Winter Run

Overview—This population combines winter-run steelhead from four Salmonid Stock Inventory stocks (WDFW 2002): Hamma Hamma, Duckabush, and Dosewallips rivers, and Quilcene/Dabob Bay (Figure E-85). WDFW (2002) identified these as distinct stocks based on their geographic separation; however, genetically, resident, parr, and smolt *O. mykiss* from the Duckabush and Dosewallips cluster together relative to steelhead populations on the east side of Hood Canal. All of these rivers drain the east slope of the Olympic Mountains, with falls or cascades blocking access to most of the higher elevation habitat.

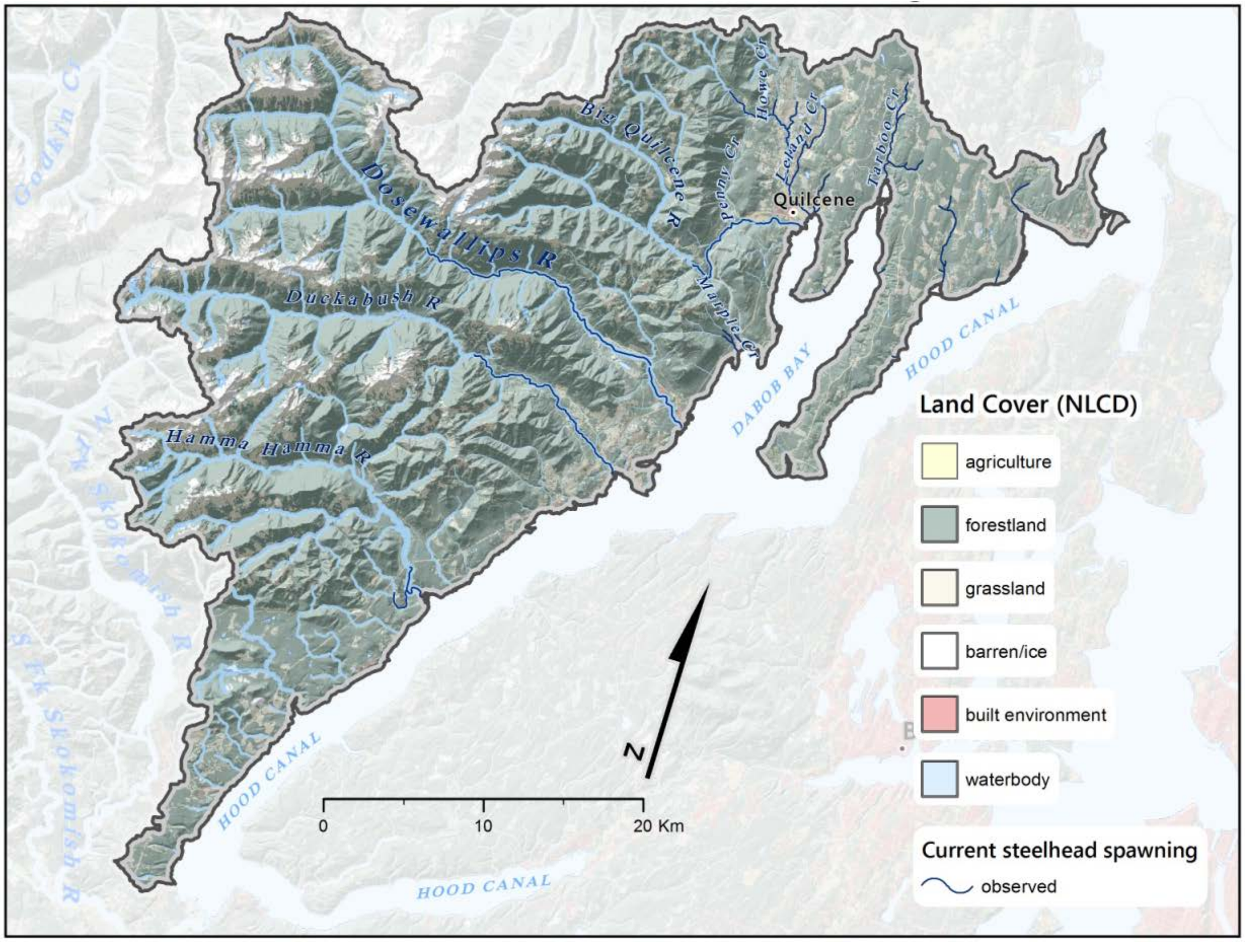

Figure E-85. Map of West Hood Canal Tributaries Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Large	982*	208 (118–366)	3,608	7,217
Year	1932	1952	2005–2009	—	

* Hamma Hamma, Duckabush, Dosewallips, and Quilcene rivers; this year includes some hatchery returns.

The West Hood Canal Tributaries Winter Run's exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 1.022 (0.997–1.048).

Tributary	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Hamma Hamma R.	230	134	214	123	73	193	198	81	42	45
Duckabush River	16	8	29	10	21	16	18	12	29	120
Dosewallips River	52	96	—	—	—	15	42	—	—	31
Quilcene River	30	16	36	9	76	39	41	6	41	31

For a state-space analysis of population viability, see the top two plots of Figure A-17 in Appendix A.

Diversity—

West Hood Canal Tributaries Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Dosewallips River	1995–2003	7	Bogachiel Hatchery	0	80,142
Duckabush River	1995–2003	7	Bogachiel Hatchery	0	65,860
Duckabush River	2009–2010	2	Native	6,595	0
Hamma Hamma R.	2000–2005	5	Native	7,306	0
John Creek (HH)	2000–2005	4	Native	2,837	0
			Total:	16,738	146,002

Spatial structure and hydrology—Figure E-86 and Figure E-87.

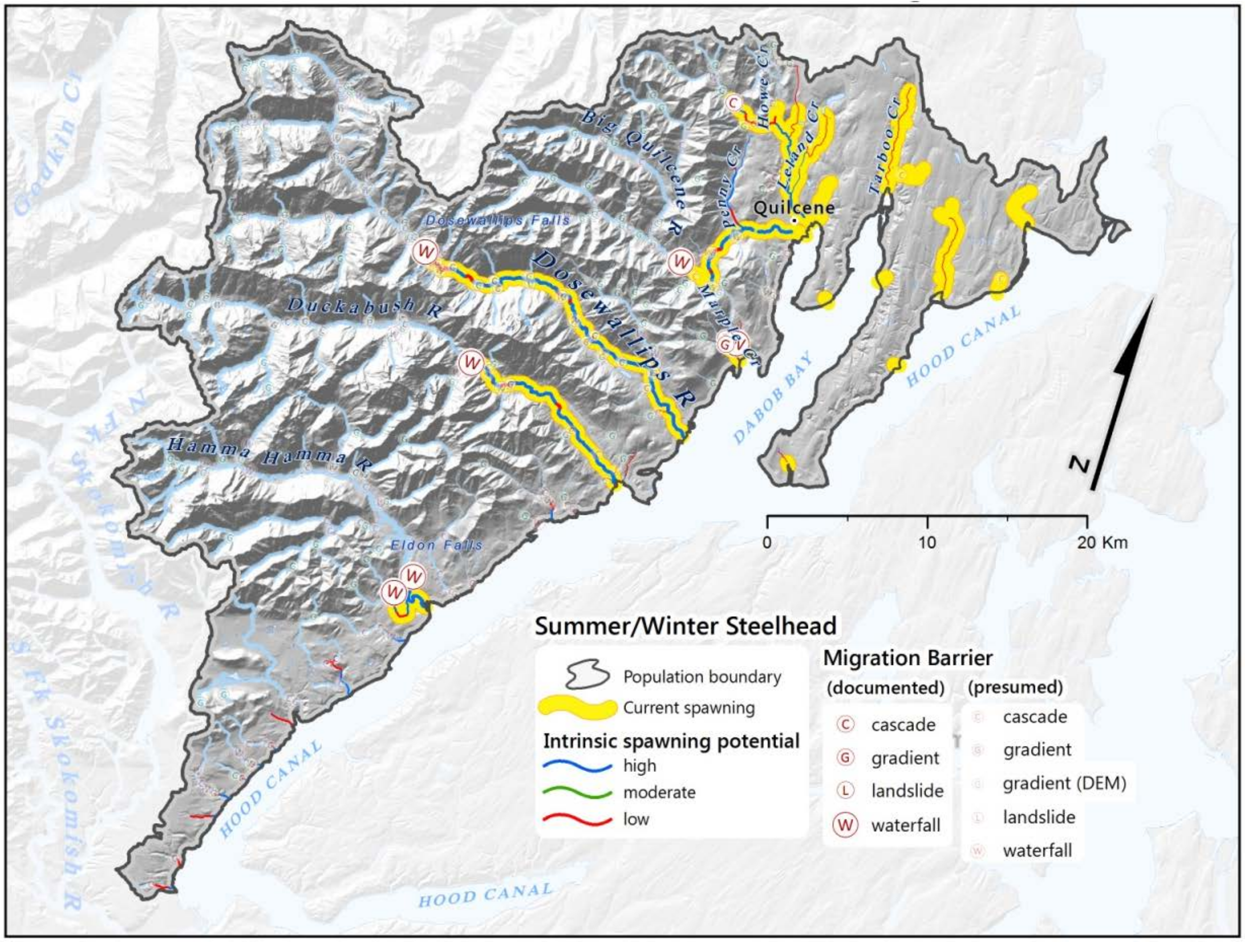

Figure E-86. Map of West Hood Canal Tributaries Winter-Run DIP spatial structure, including migration barriers and spawning potential.

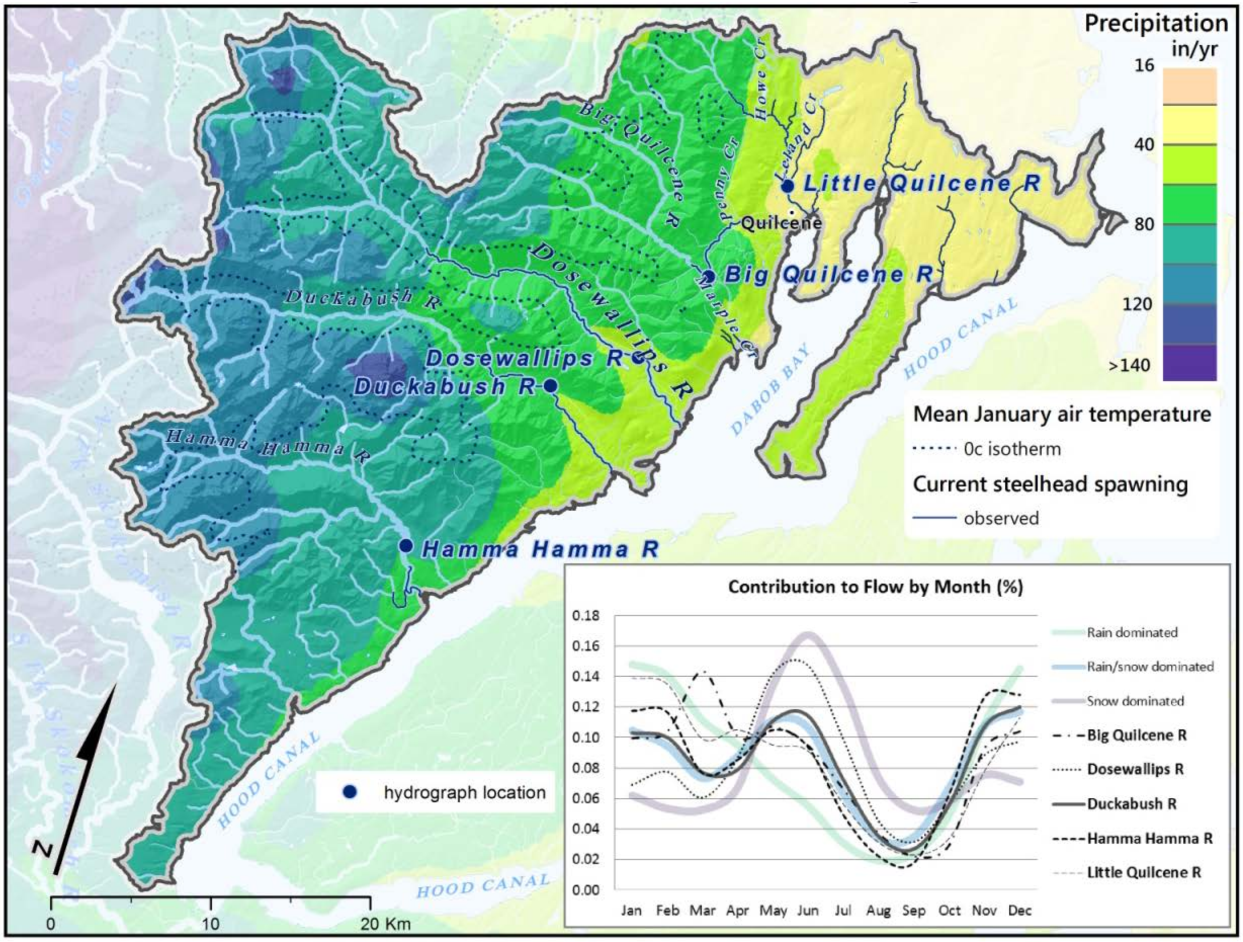

Figure E-87. Map of West Hood Canal Tributaries Winter-Run DIP area hydrology.

29. Sequim/Discovery Bays Tributaries Winter Run

Overview—This population combines two Salmonid Stock Inventory stocks, Sequim Bay and Discovery Bay (Figure E-88), and includes winter-run steelhead that occupy streams in the Quimper Peninsula (Pt. Townsend) that were not included in the WDFW (2002) stock list. The entire population is located within the Puget Lowland Ecoregion and stream flows are rain dominated, with many streams lacking surface flow during summer. Although the 802 km^2 basin size for this DIP is well above the minimum, the majority of the area contains relatively small, independent streams.

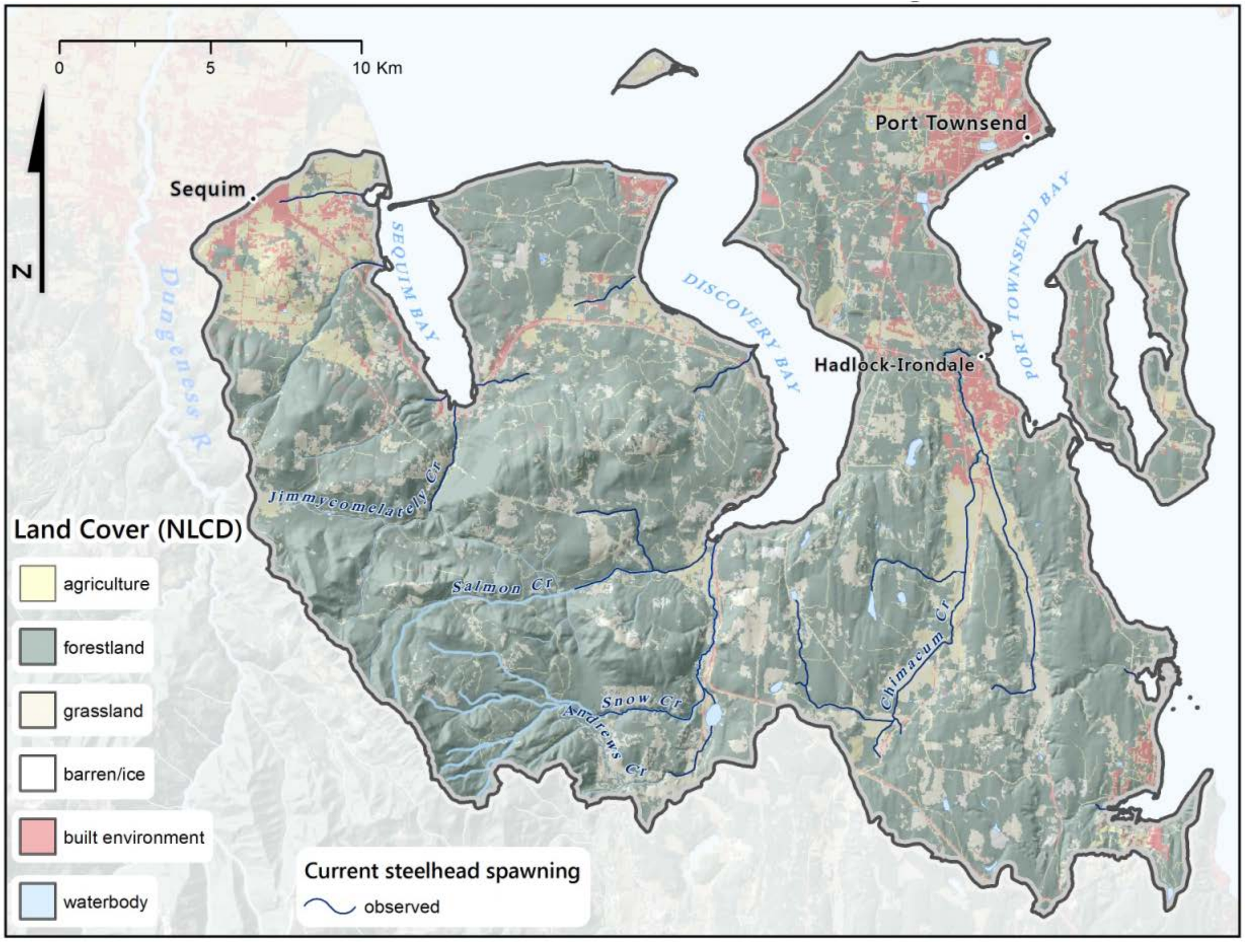

Figure E-88. Map of Sequim/Discovery Bays Tributaries Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate	
Abundance	Not available	200	512	1,024
Year	—	1962	—	

For a state-space analysis of population viability, see Figure A-18 in Appendix A.

Diversity—Hatchery releases for the Sequim/Discovery Bays Tributaries Winter Run are zero within the DPS and zero outside the DPS.

Spatial structure and hydrology—Figure E-89 and Figure E-90.

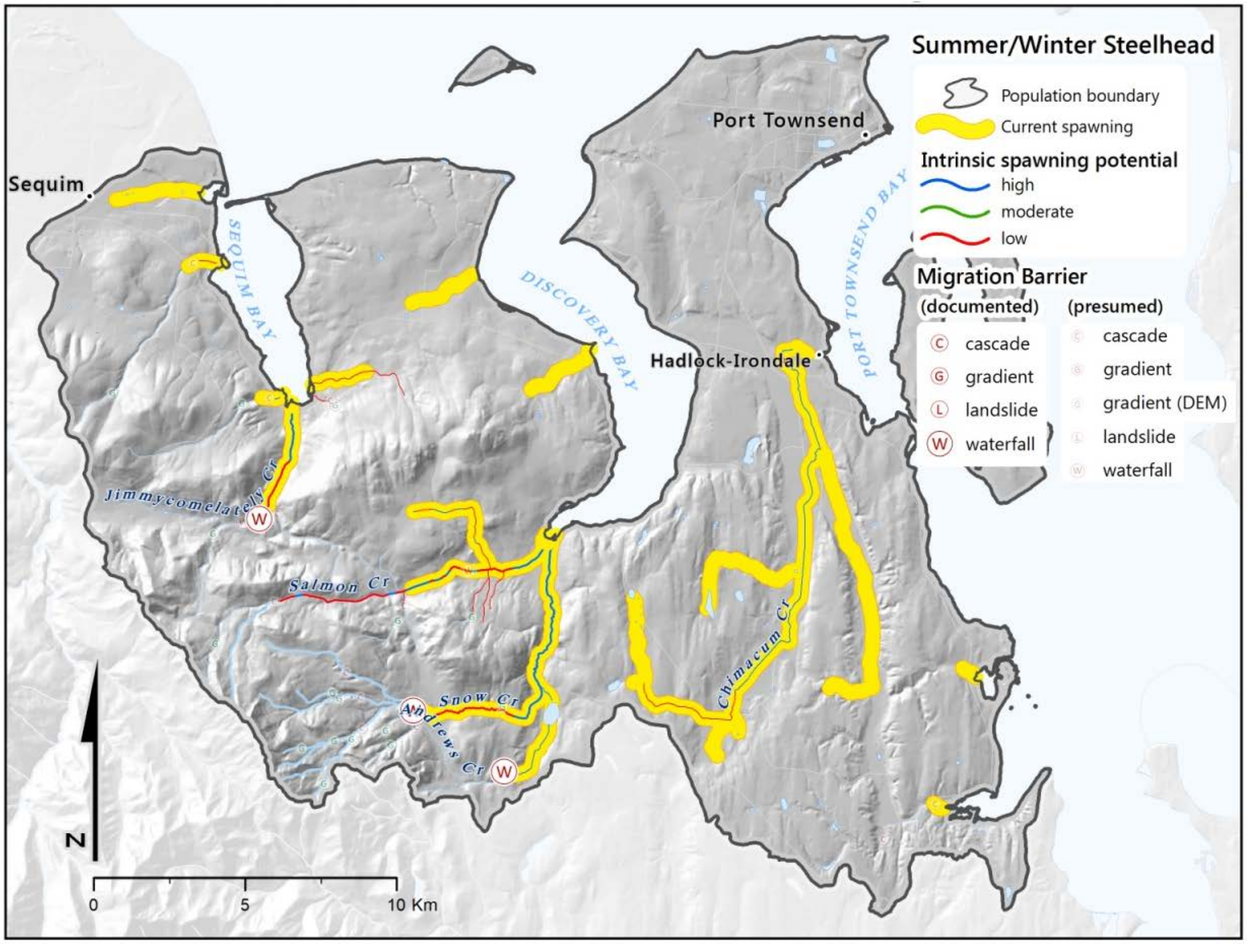

Figure E-89. Map of Sequim/Discovery Bays Tributaries Winter-Run DIP spatial structure, including migration barriers and spawning potential.

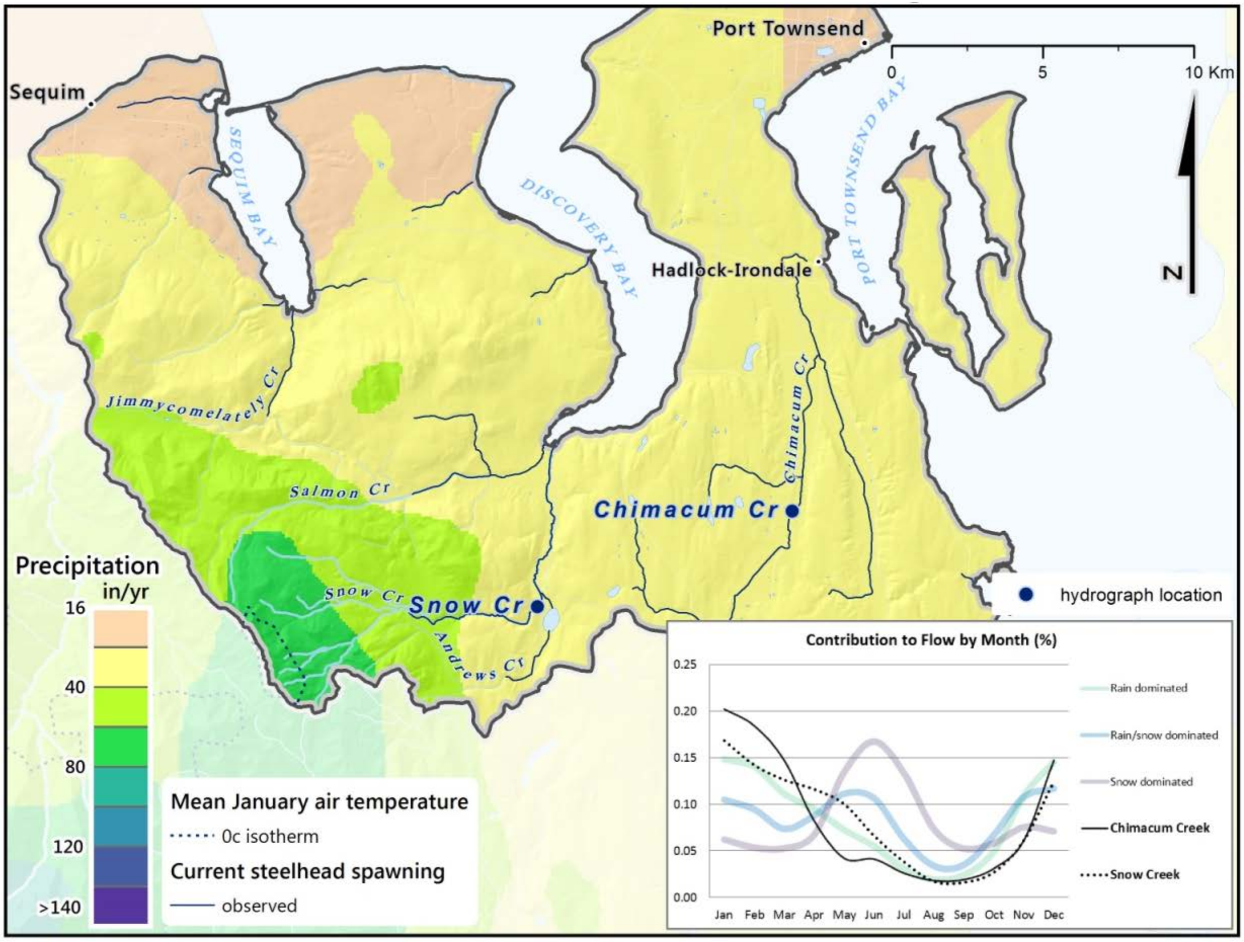

Figure E-90. Map of Sequim/Discovery Bays Tributaries Winter-Run DIP area hydrology.

30. Dungeness River Summer Run and Winter Run

Overview—This population includes steelhead spawning in the mainstem Dungeness and Grey Wolf rivers (Figure E-91). Much of the mainstem and upper headwaters are glacially influenced. Anecdotal information indicates that this population was historically quite abundant.

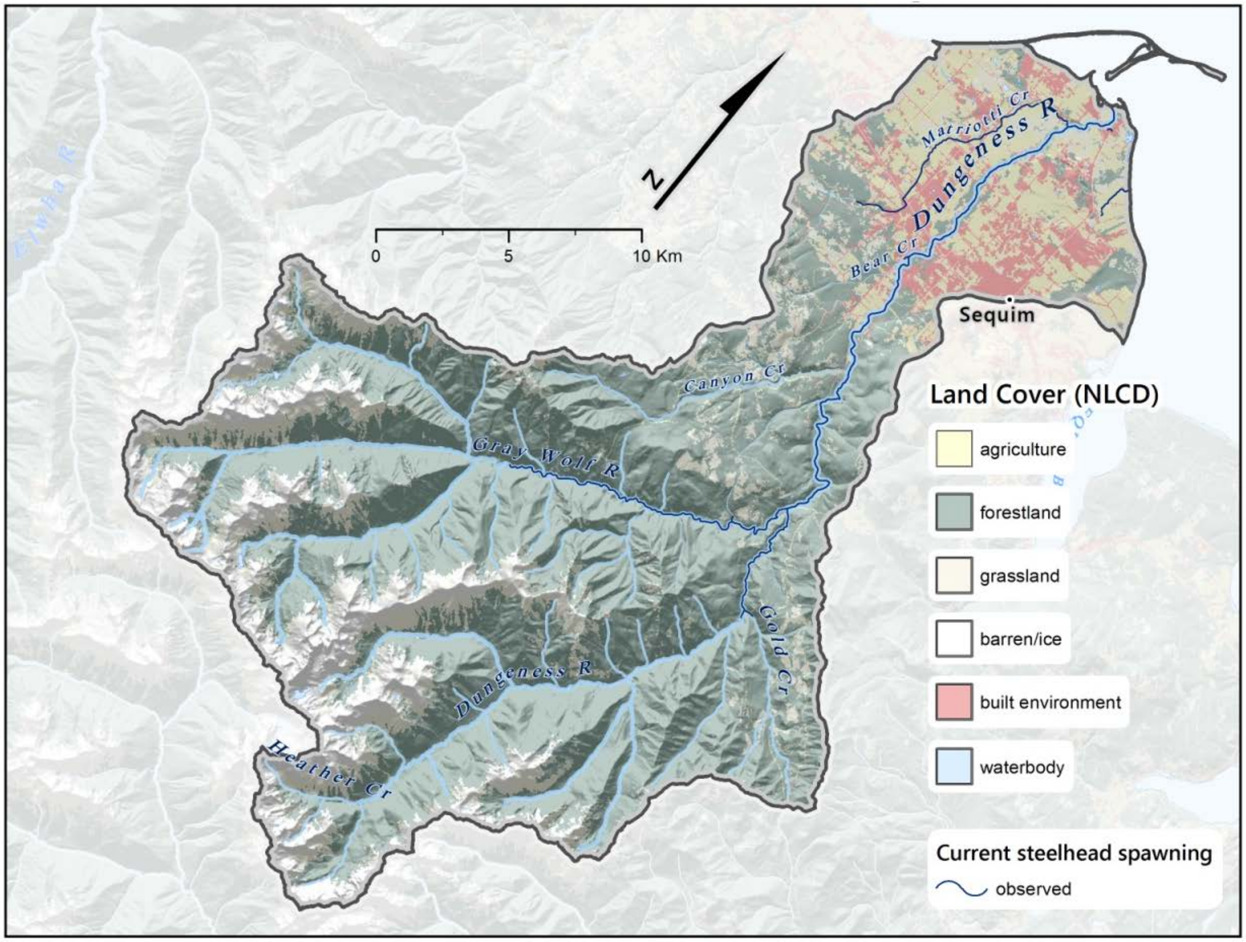

Figure E-91. Map of Dungeness River Summer-Run and Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate	
Abundance	Not available	348	2,039	4,078
Year	—	1946	—	

The Dungeness River Summer-Run and Winter-Run DIP's exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 0.926 (0.909–0.943).

For a state-space analysis of population viability, see the top two plots of Figure A-19 in Appendix A.

Diversity—

Dungeness R. Summer Run and Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Dungeness	1995–2010	16	Bogachiel/Dungeness (CC)	0	158,427
			Total:	0	158,427

Spatial structure and hydrology—Figure E-92 and Figure E-93.

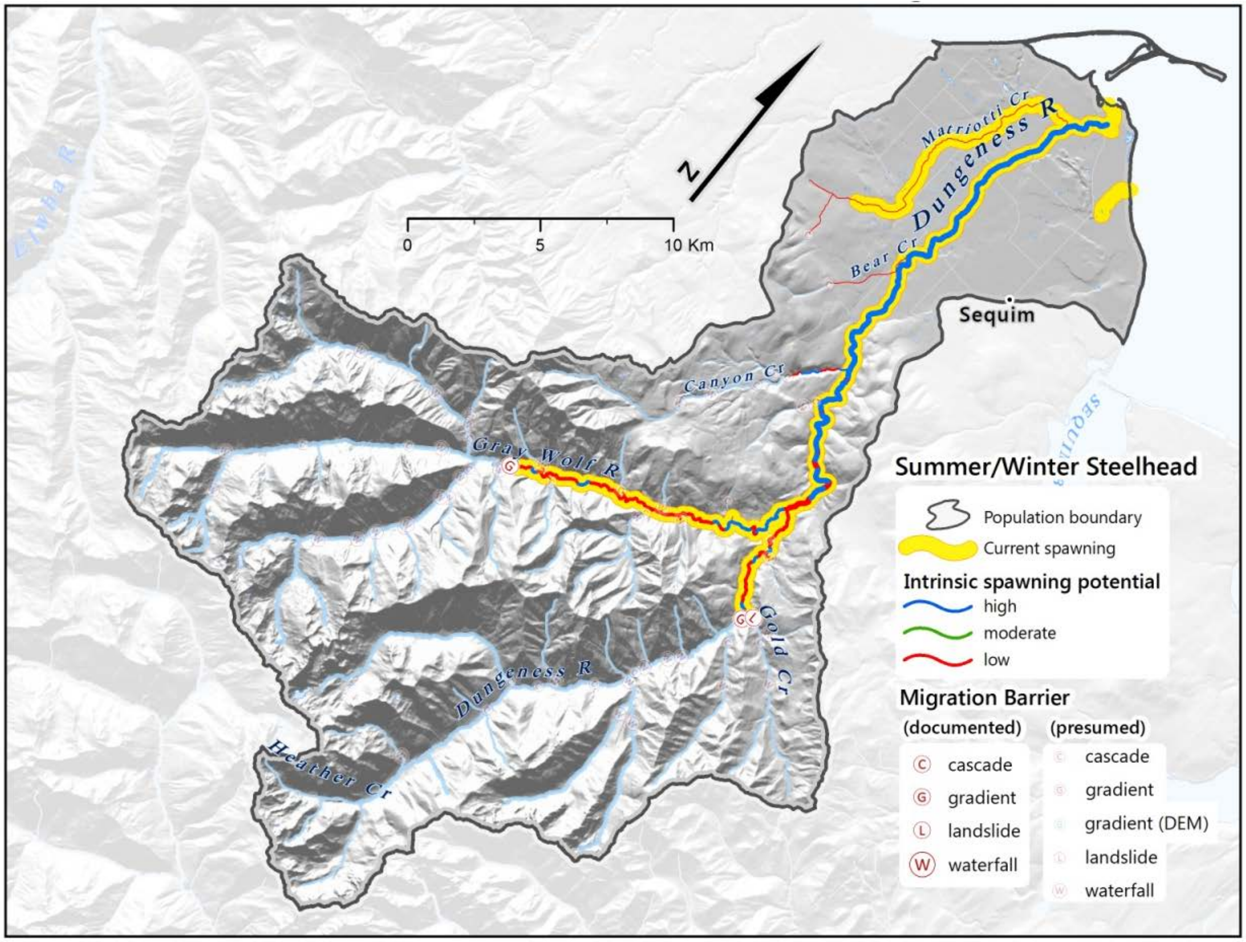

Figure E-92. Map of Dungeness River Summer-Run and Winter-Run DIP spatial structure, including migration barriers and spawning potential.

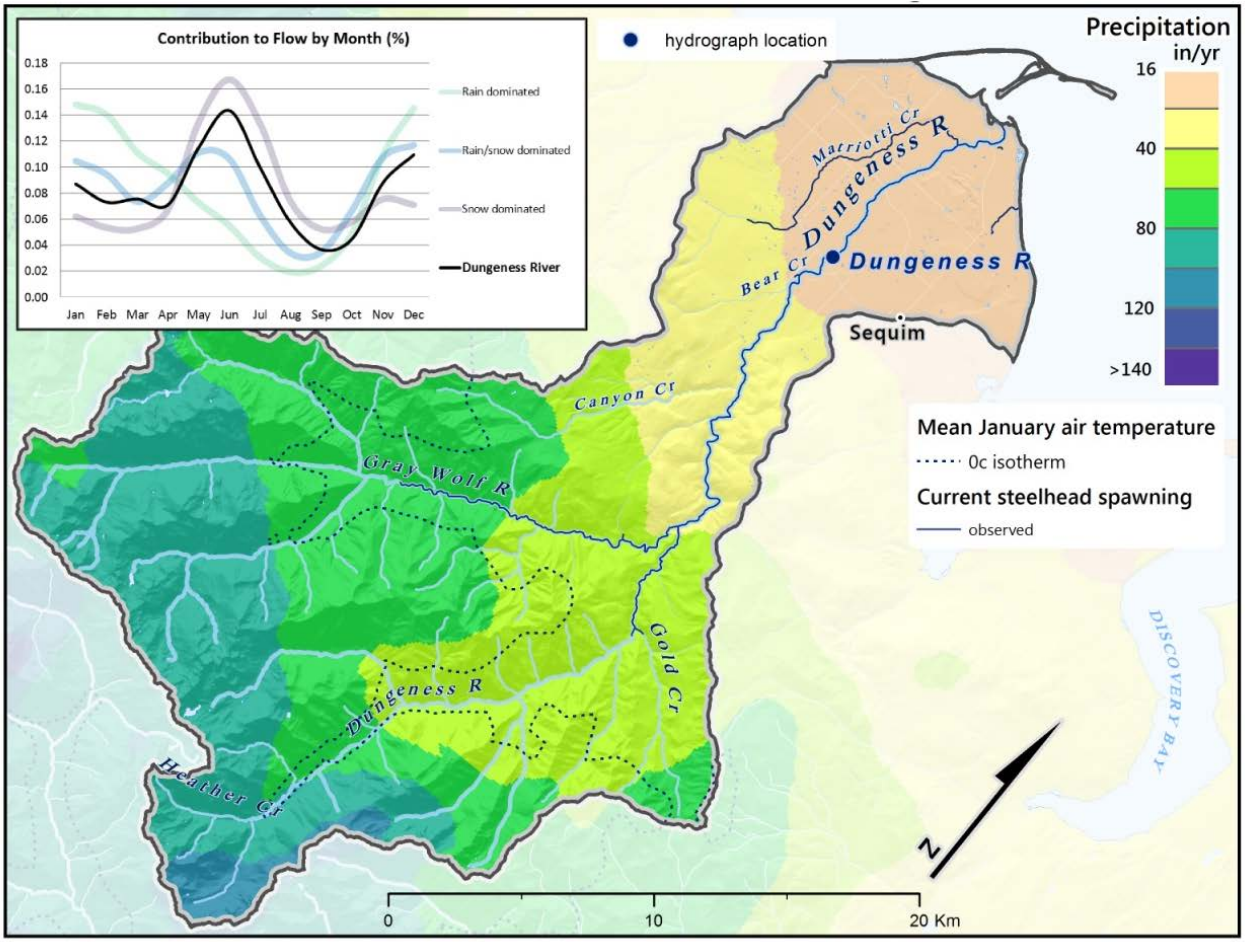

Figure E-93. Map of Dungeness River Summer-Run and Winter-Run DIP area hydrology.

31. Strait of Juan de Fuca Tributaries Winter Run

Overview—This population consists of steelhead spawning in small independent tributaries to the Strait of Juan de Fuca between the Dungeness and Elwha rivers, including Ennis, White, Morse, Siebert, and McDonald creeks (Figure E-94). While each of the tributaries is relatively small, collectively, the creeks cover a 410 km^2 watershed.

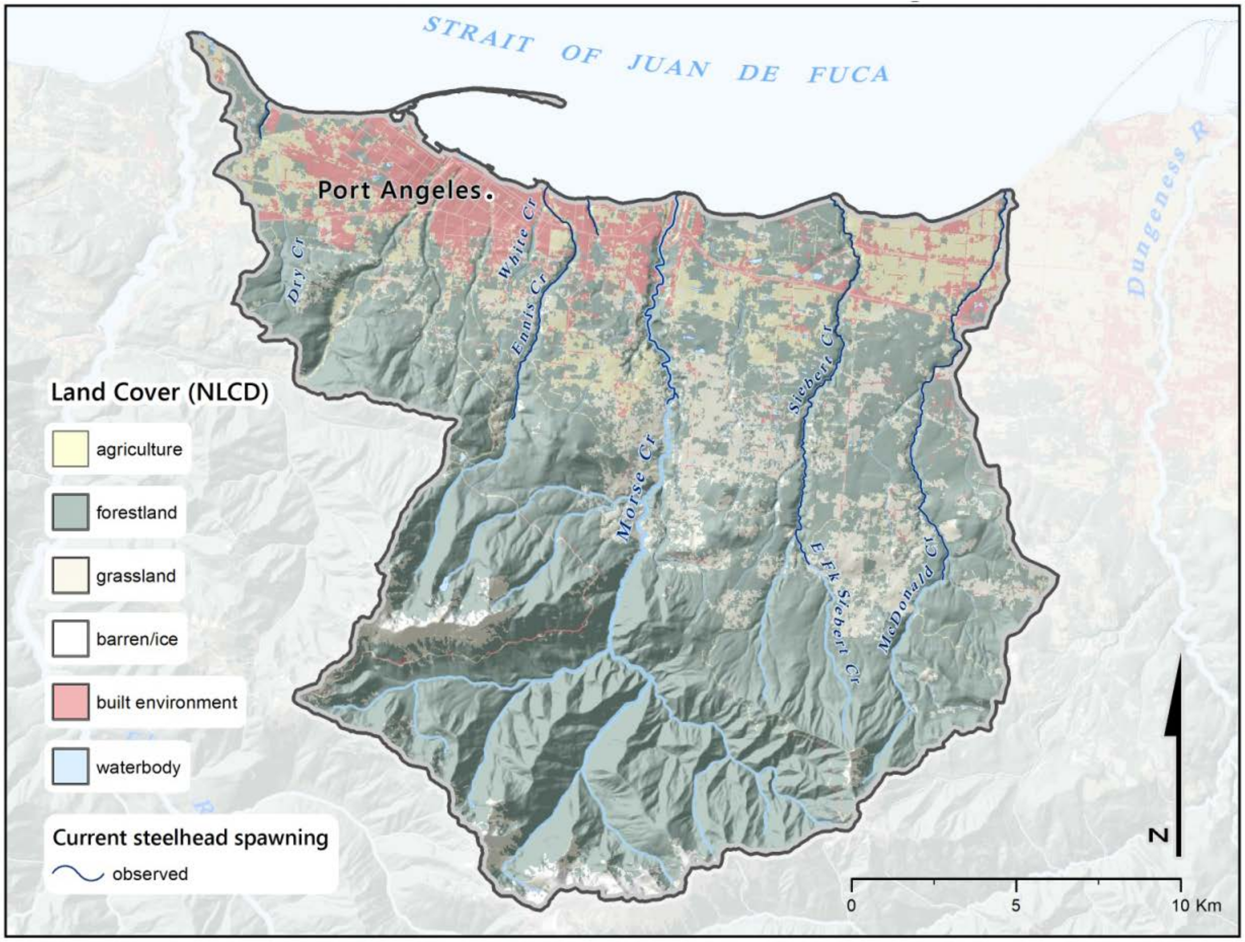

Figure E-94. Map of Strait of Juan de Fuca Tributaries Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	Geometric mean	IP estimate	
Abundance	Not available	258*	147 (53–405)	728	1,456
Year	—	1958	2005–2009	—	

* Includes Morse and McDonald creeks only.

The Strait of Juan de Fuca Tributaries Winter Run's exponential trend in the natural logarithm of natural spawners (95% CI) is 0.964 (0.899–1.031).

Tributary	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Morse Creek	71	84	121	—	124	118	—	24	—	—
McDonald Creek	125	63	29	89	206	63	49	63	193	—

For a state-space analysis of the DIP's viability, see the top two plots of Figure A-20 in Appendix A. For a state-space analysis of Morse Creek winter-run population viability, see the top two plots of Figure A-21.

Diversity—

Strait of Juan de Fuca Tributaries Winter Run hatchery releases				Total releases	
Watershed	Duration	Years	Source	Within DPS	Outside DPS
Morse Creek	1995–2004	10	Bogachiel Hatchery	0	60,115
			Total:	0	60,115

Spatial structure and hydrology—Figure E-95 and Figure E-96.

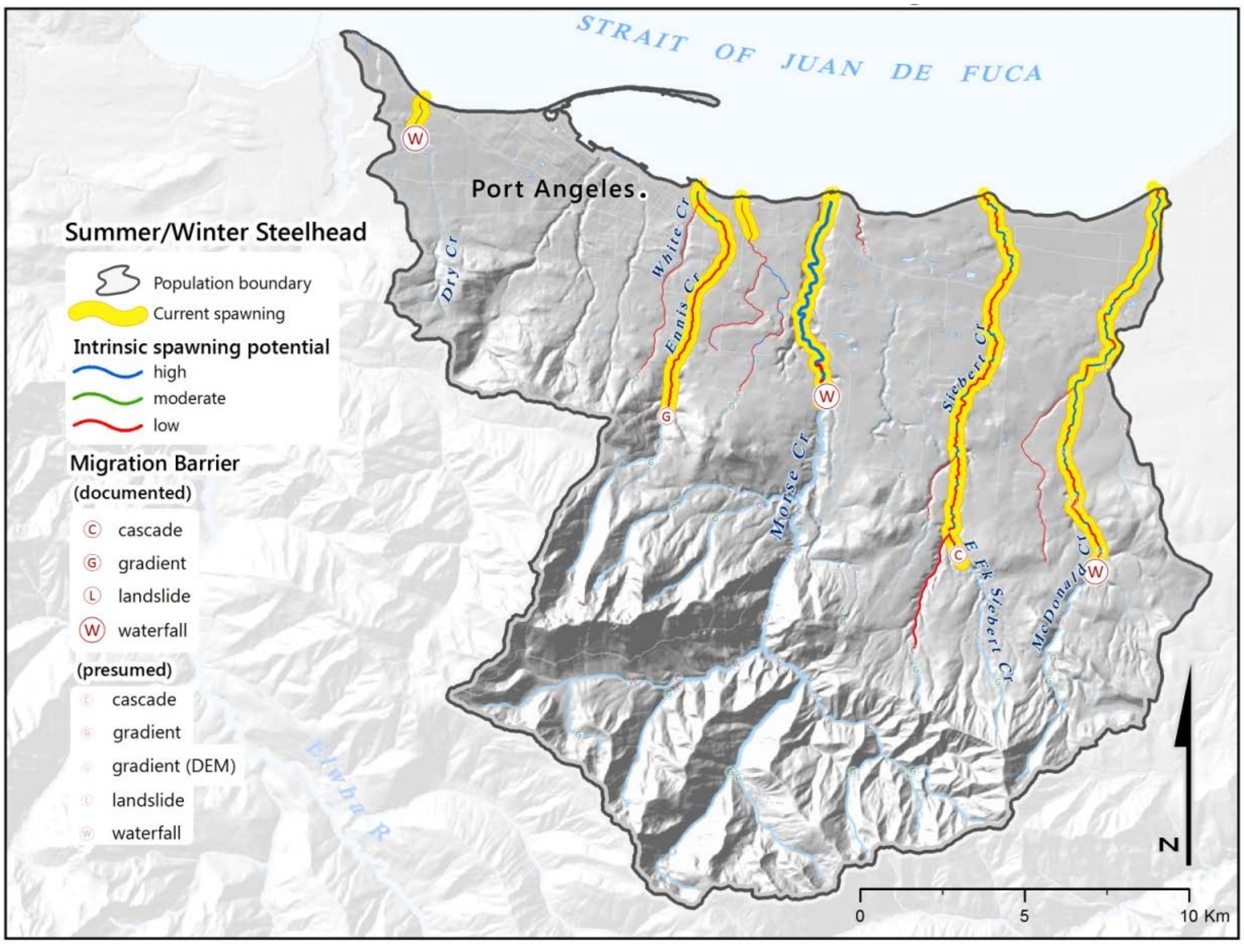

Figure E-95. Map of Strait of Juan de Fuca Tributaries Winter-Run DIP spatial structure, including migration barriers and spawning potential.

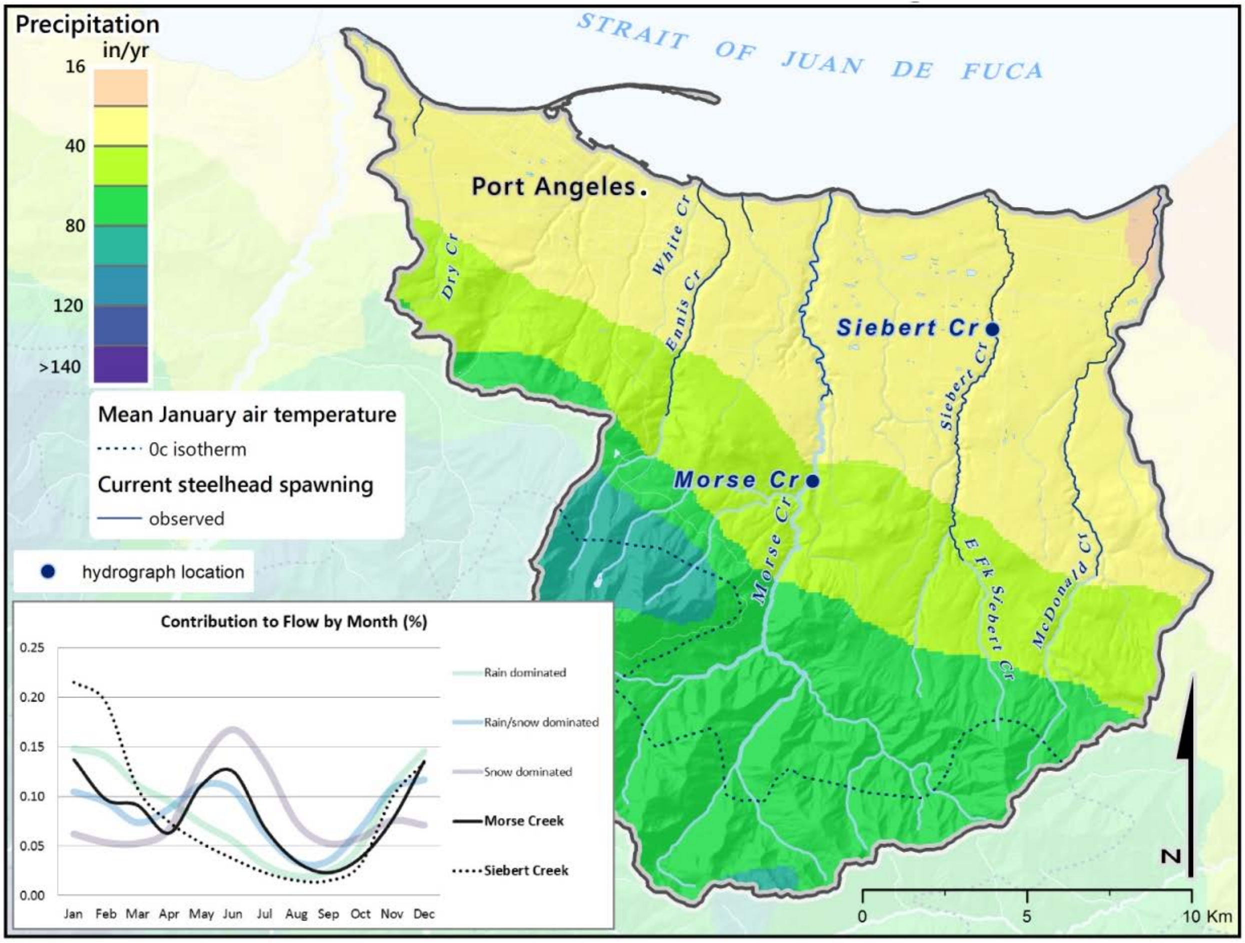

Figure E-96. Map of Strait of Juan de Fuca Tributaries Winter-Run DIP area hydrology.

32. Elwha River Winter Run

Overview—Winter-run steelhead were historically present in the Elwha River basin (Figure E-97), although little is known of their distribution of life history diversity prior to the construction of the two Elwha River dams in the early 1900s. Until recently, only 8 km of lowland mainstem habitat was accessible in the lower Elwha River. Ongoing restoration activities after dam removal will allow access to the entire basin in the near future.

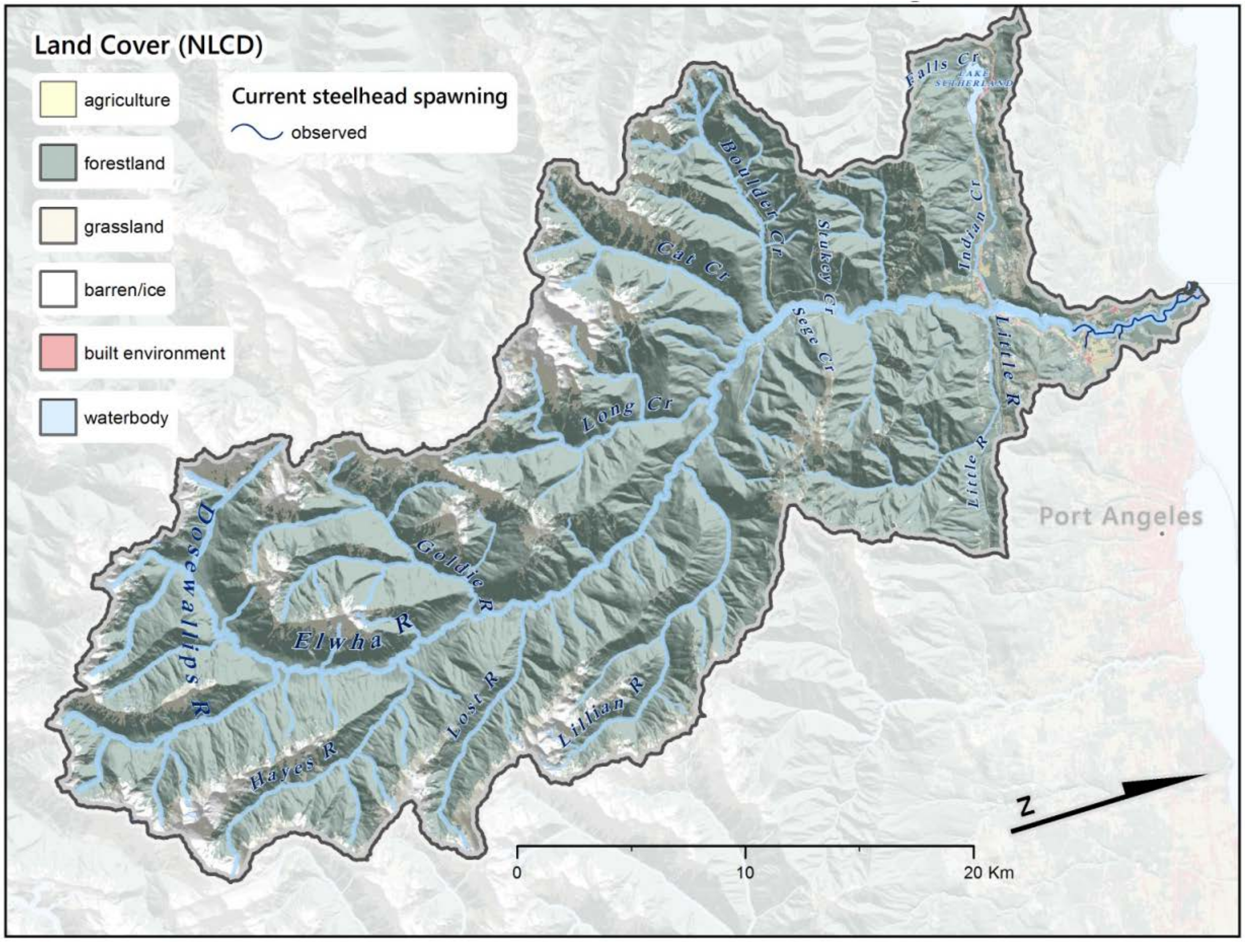

Figure E-97. Map of Elwha River Winter-Run DIP land cover and observed spawning area.

Abundance and productivity—

	WDF survey	Punch card max.	IP estimate	
Abundance	Not available	746	7,116	14,231
Year	—	1952	—	

The Elwha River Winter Run’s exponential trend in the natural logarithm of natural spawners (95% CI) from 1985 to 2009 is 0.840 (0.749–0.943).

For a state-space analysis of population viability, see Figure A-22 in Appendix A.

Diversity—

Elwha River summer run hatchery releases					Total releases	
Watershed	Duration	Years	Source		Within DPS	Outside DPS
Elwha River	1995–2010	16	Bogachiel (SK)		0	85,445
				Total:	0	85,445

Elwha River Winter Run hatchery releases					Total releases	
Watershed	Duration	Years	Source		Within DPS	Outside DPS
Elwha River	1995–2010	16	Elwha/Bogachiel (CC)		0	2,036,243
				Total:	0	2,036,243

Spatial structure and hydrology—Figure E-98 and Figure E-99.

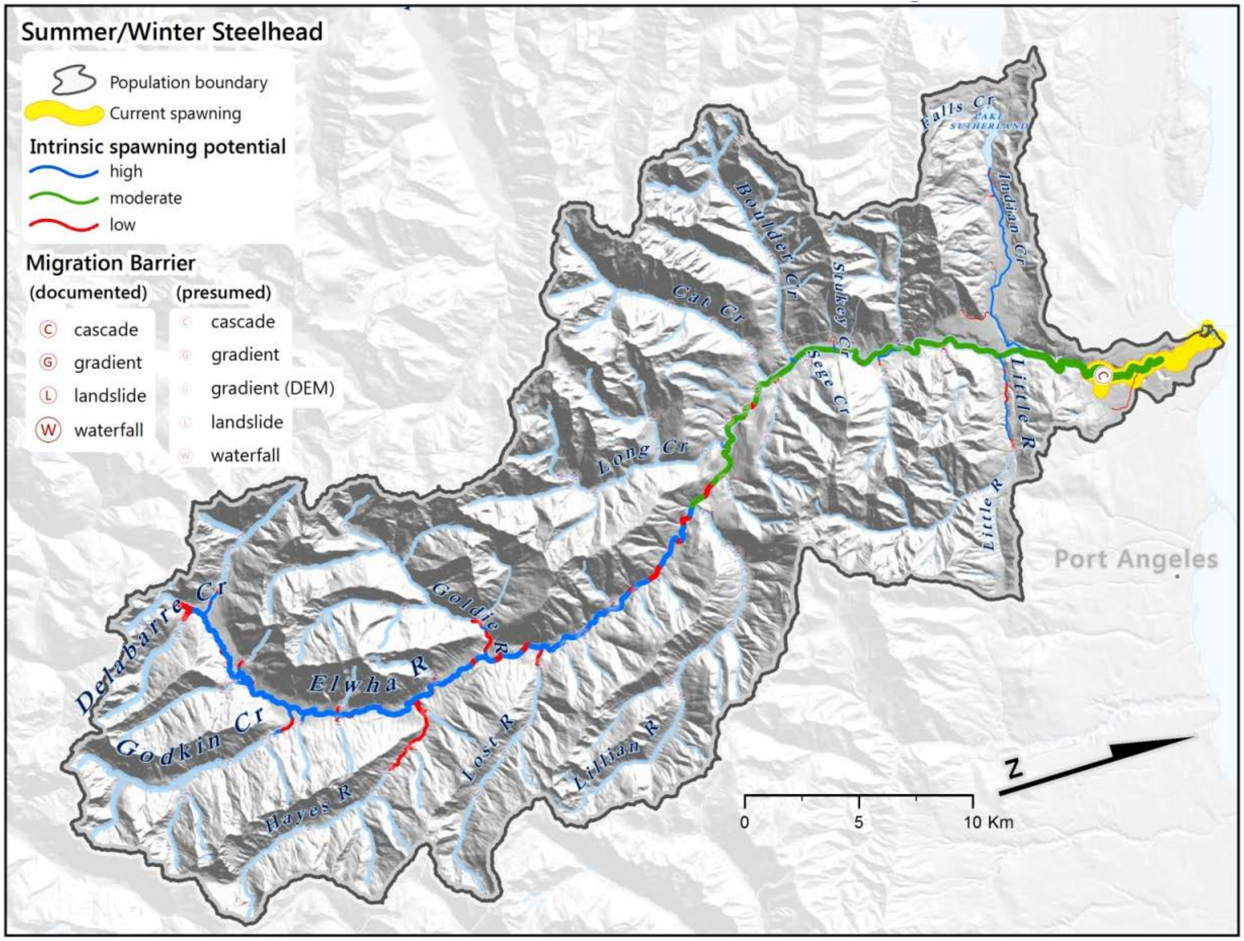

Figure E-98. Map of Elwha River Winter-Run DIP spatial structure, including migration barriers and spawning potential. The two dams constructed in the early 1900s at RKM 7.9 and RKM 21.6 were recently removed.

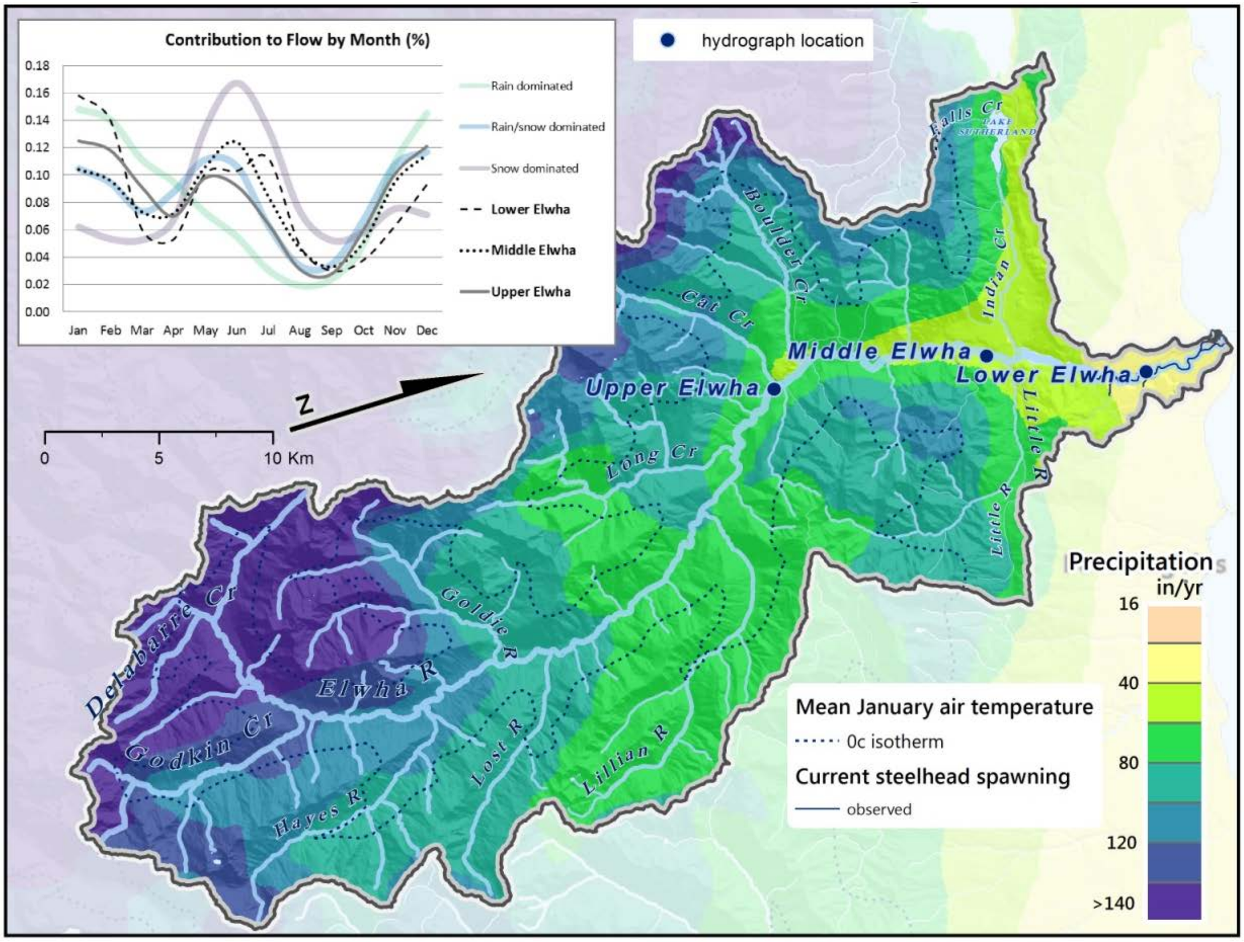

Figure E-99. Map of Elwha River Winter-Run DIP area hydrology.

Appendix F: Bayesian Networks Characterizing Viability of Puget Sound Steelhead DIPs

Appendix F contains Figure F-1 through Figure F-32. They show the Bayesian networks (BNs) for viability for each of the 32 candidate demographically independent populations (DIPs) of steelhead (*Oncorhynchus mykiss*) identified by the Puget Sound Steelhead Technical Recovery Team for the Puget Sound Steelhead Distinct Population Segment. In structuring a BN, one seeks to maximize entropy of the entire network. For a DIP to be considered viable, its probability of viability must be at least 85%, as calculated by the BN for viability. The figures indicate that population viabilities vary appreciably both within and among major population groups (MPGs), but these viabilities are all low. In most cases, these viabilities reflect the substantial influence of all four viable salmonid population criteria (i.e., abundance, growth rate, diversity, and spatial structure), but especially of abundance and diversity, as components of the BN models. A more detailed description of BNs is in the body of this technical memorandum.

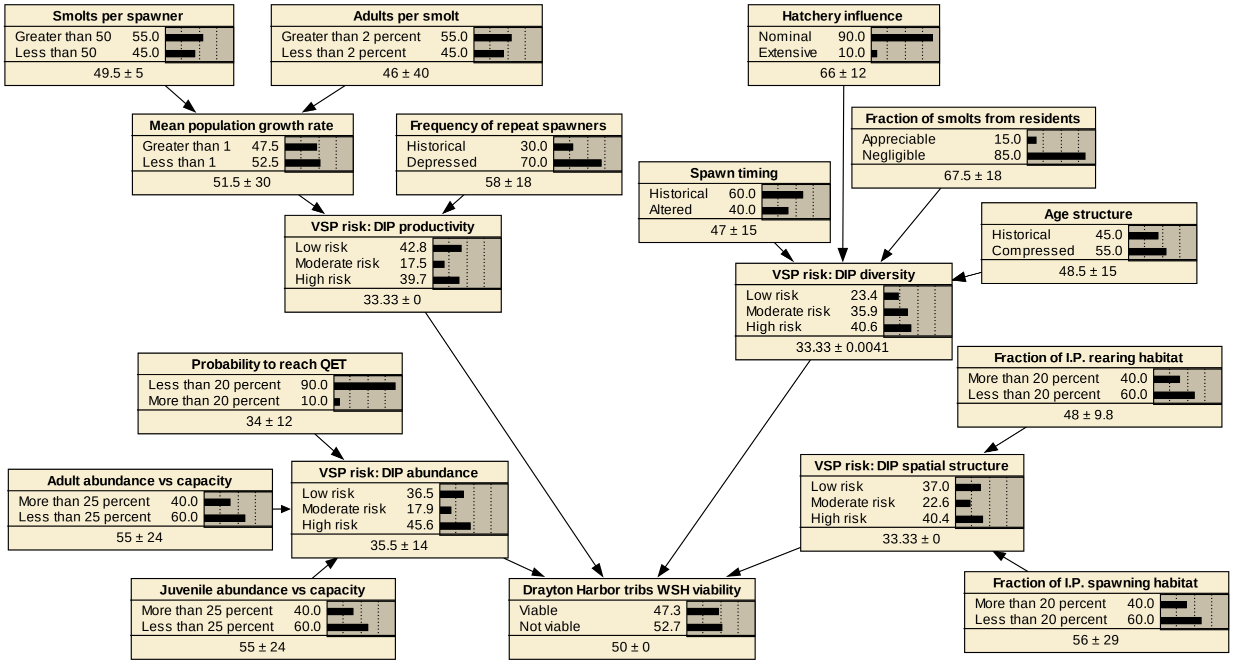

Figure F-1. BN for viability of the Drayton Harbor Tributaries Winter-Run DIP in the Northern Cascades MPG.

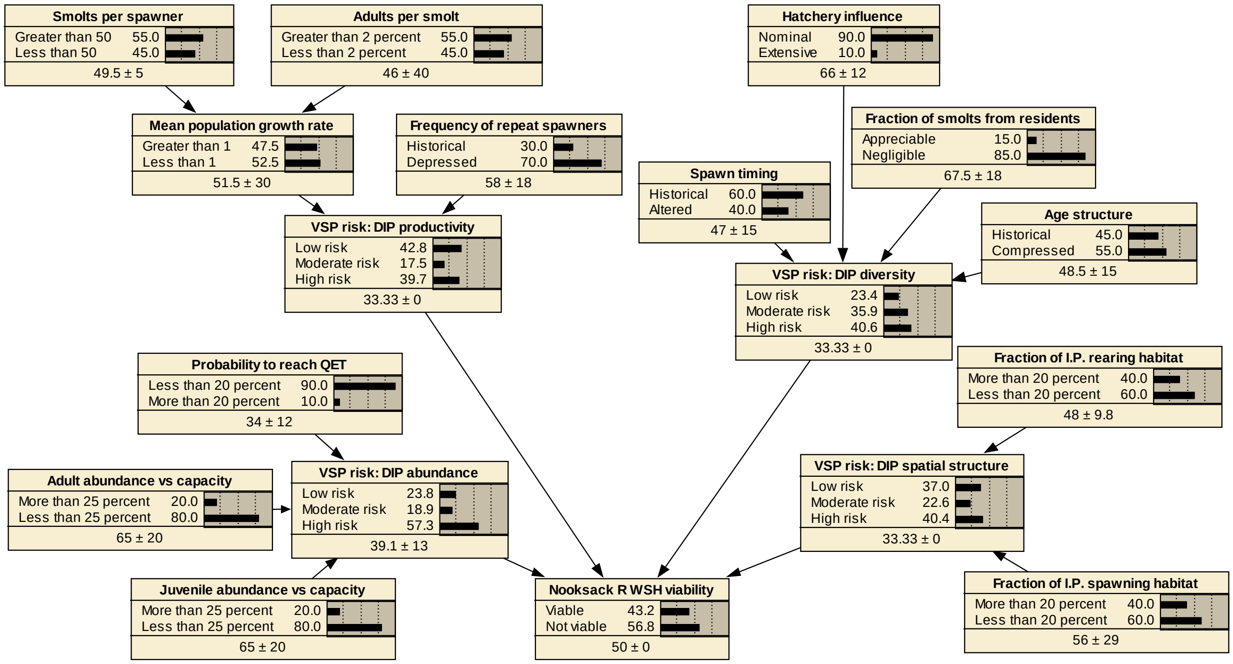

Figure F-2. BN for viability of the Nooksack River Winter-Run DIP in the Northern Cascades MPG.

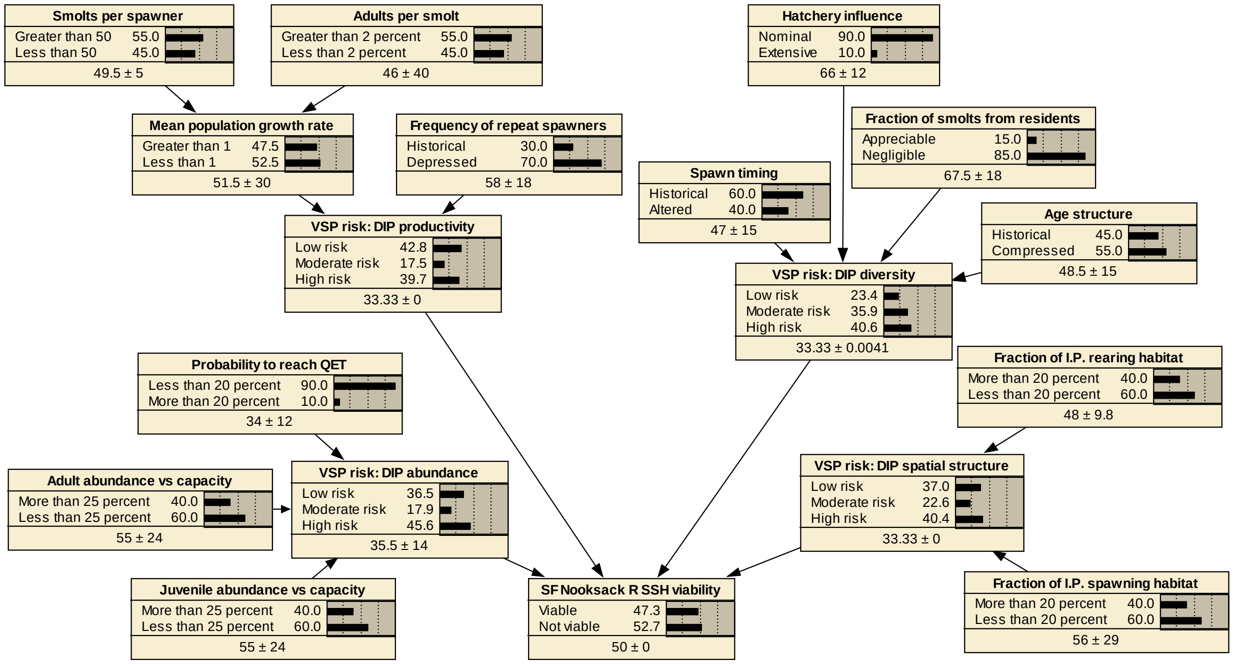

Figure F-3. BN for viability of the South Fork Nooksack River Summer-Run DIP in the Northern Cascades MPG.

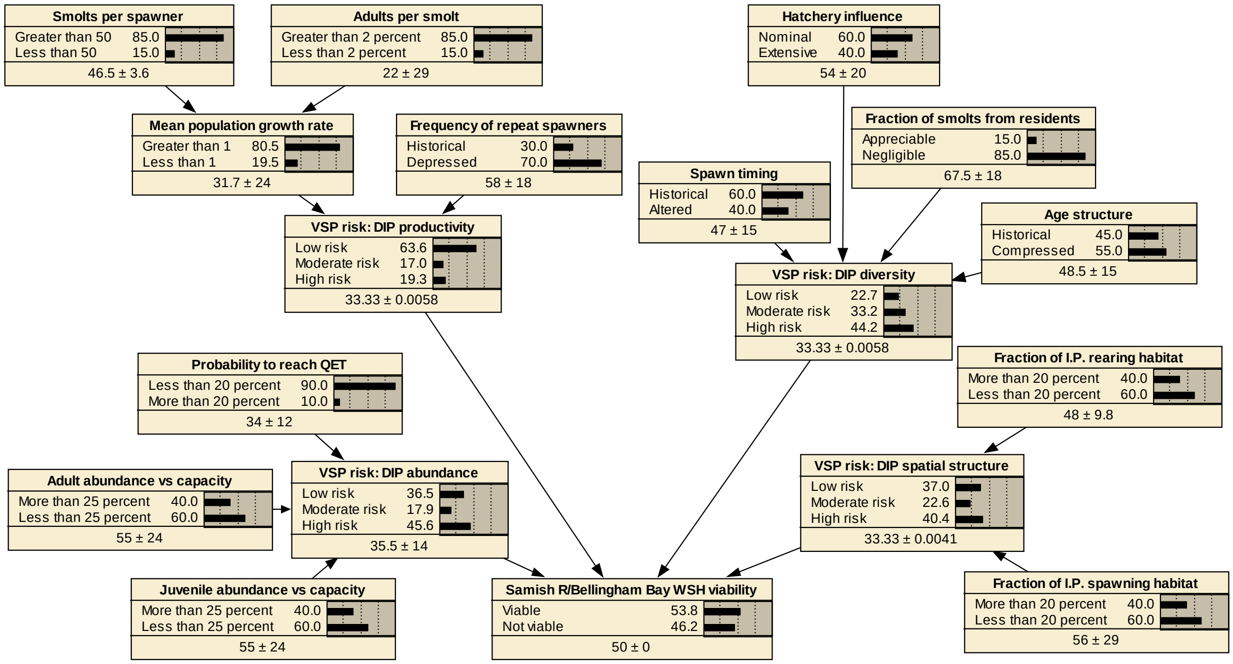

Figure F-4. BN for viability of the Samish River and Bellingham Bay Tributaries Winter-Run DIP in the Northern Cascades MPG.

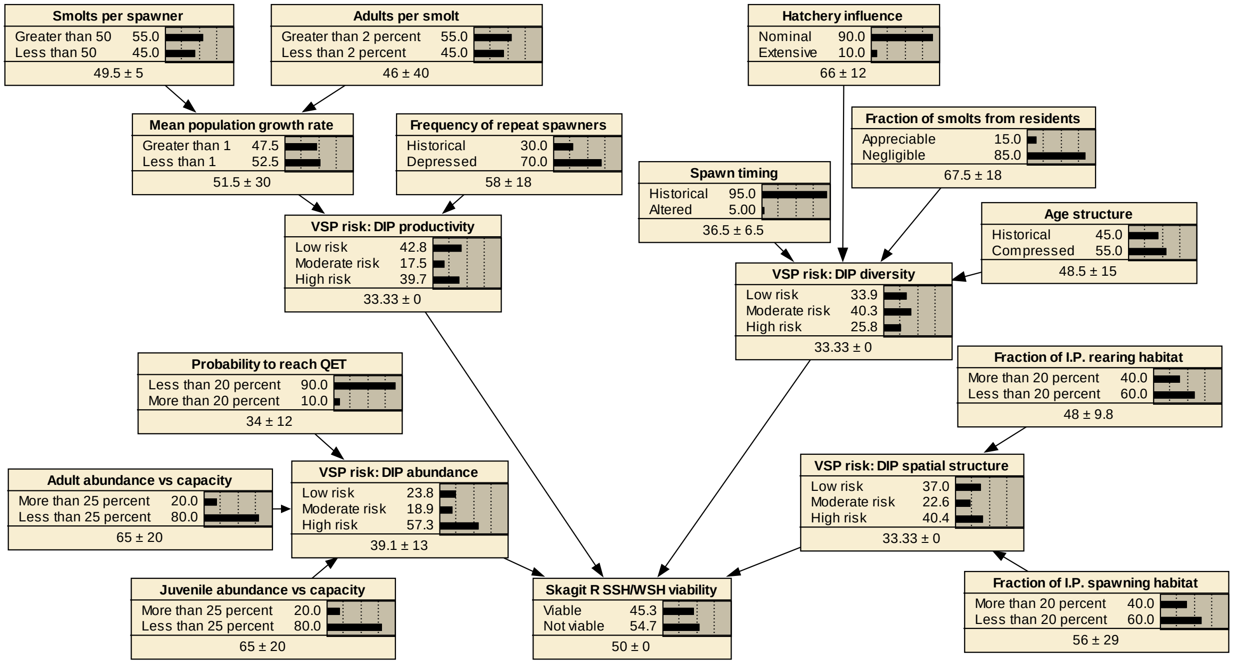

Figure F-5. BN for viability of the Skagit River Summer-Run and Winter-Run DIP in the Northern Cascades MPG.

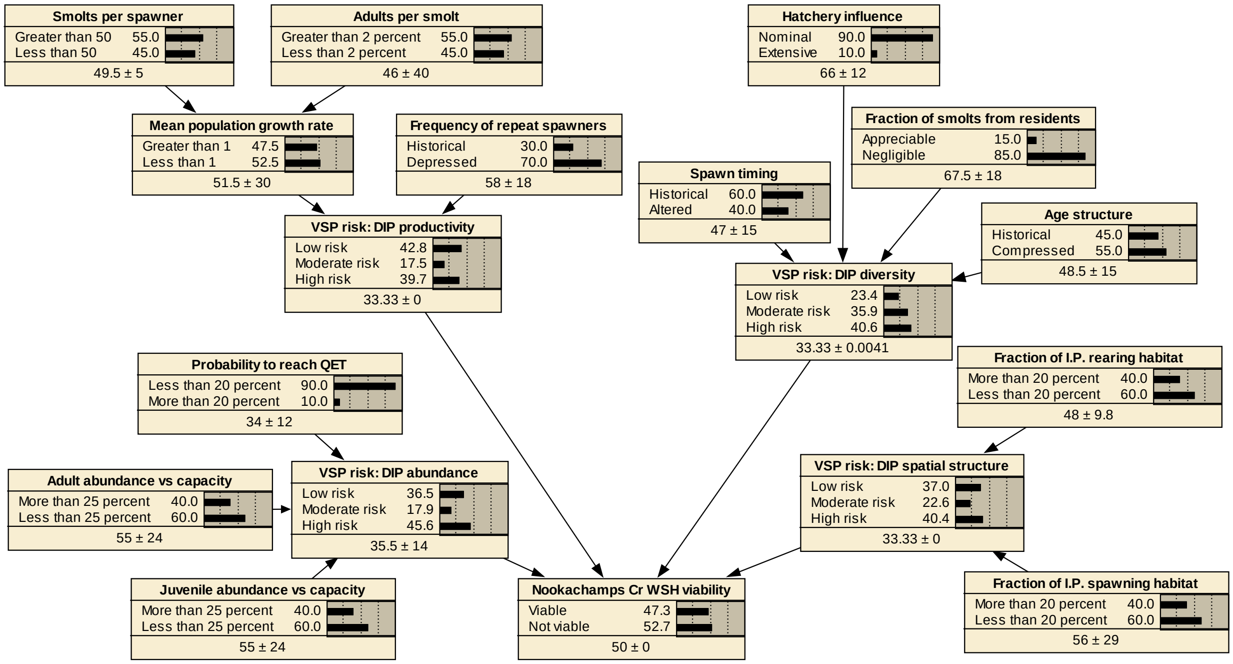

Figure F-6. BN for viability of the Nookachamps Creek Winter-Run DIP in the Northern Cascades MPG.

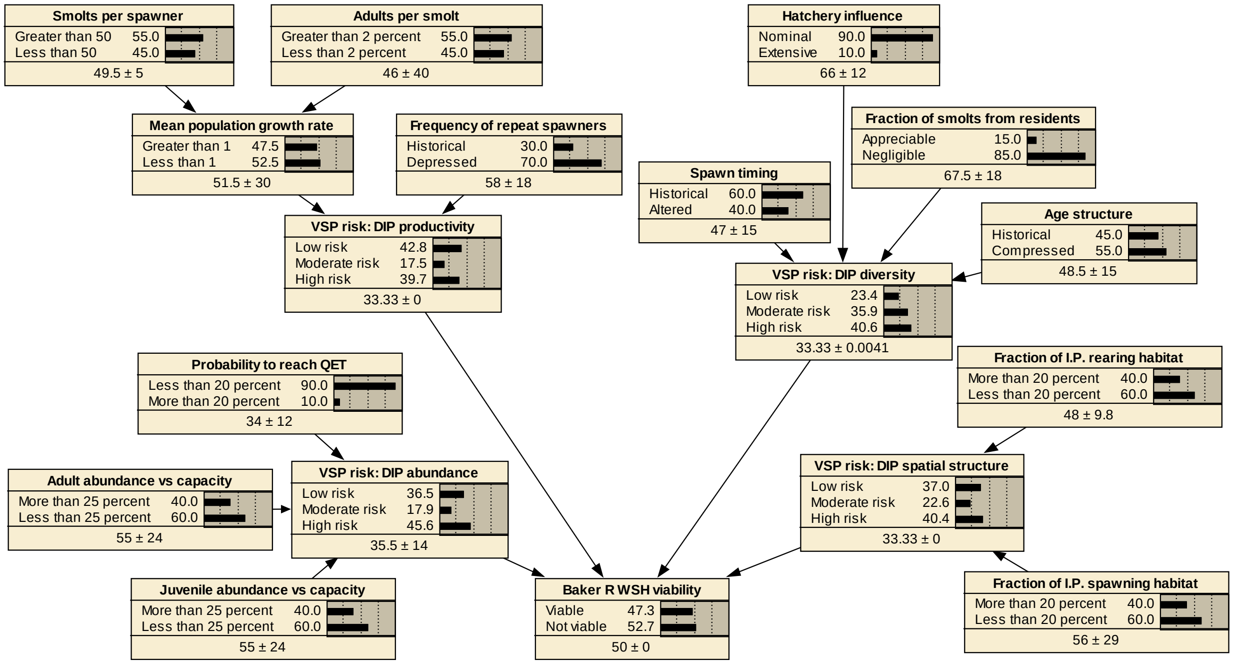

Figure F-7. BN for viability of the Baker River Summer-Run and Winter-Run DIP in the Northern Cascades MPG.

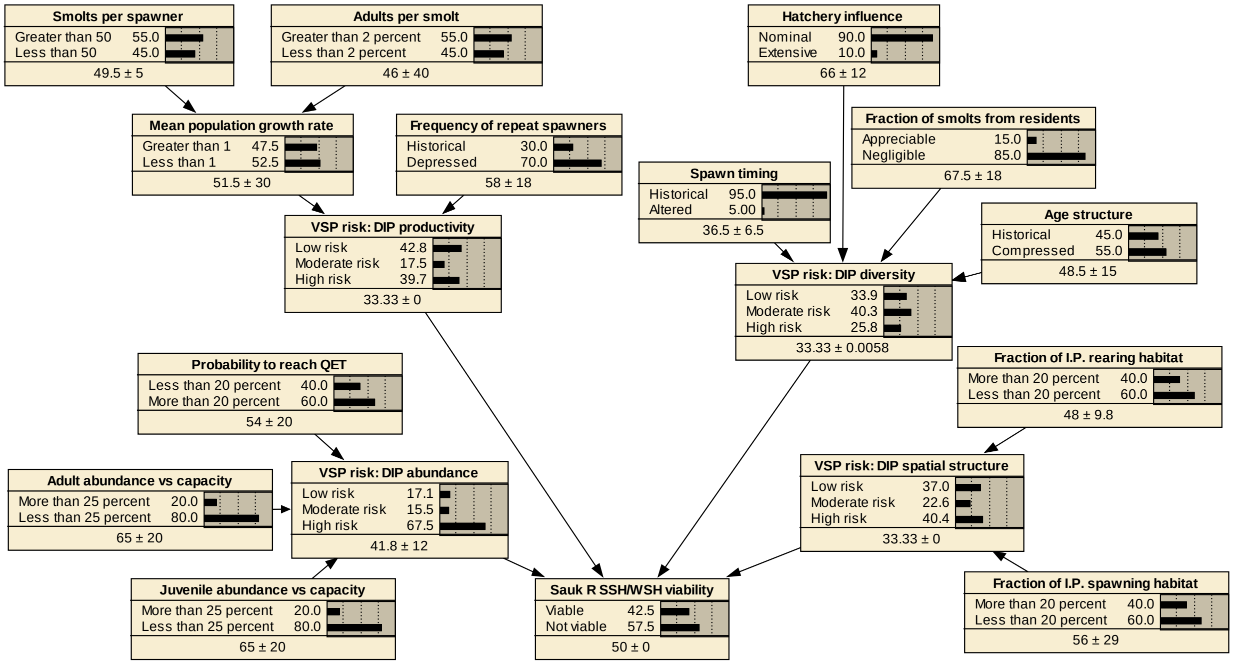

Figure F-8. BN for viability of the Sauk River Summer-Run and Winter-Run DIP in the Northern Cascades MPG.

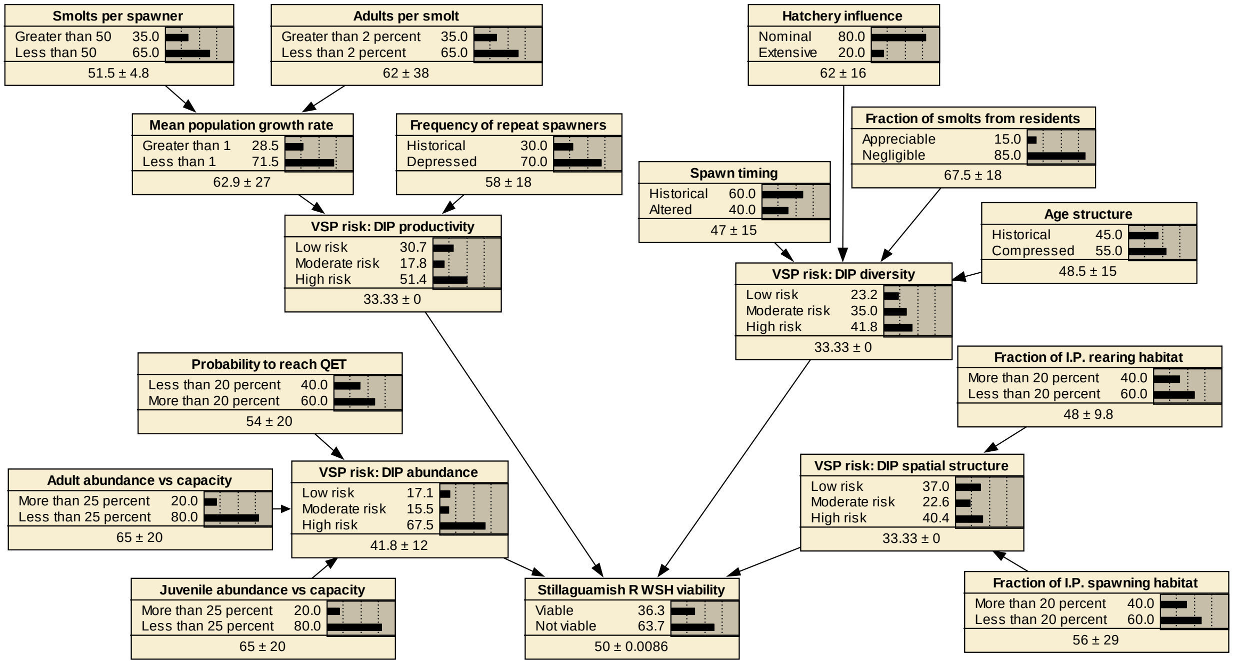

Figure F-9. BN for viability of the Stillaguamish River Winter-Run DIP in the Northern Cascades MPG.

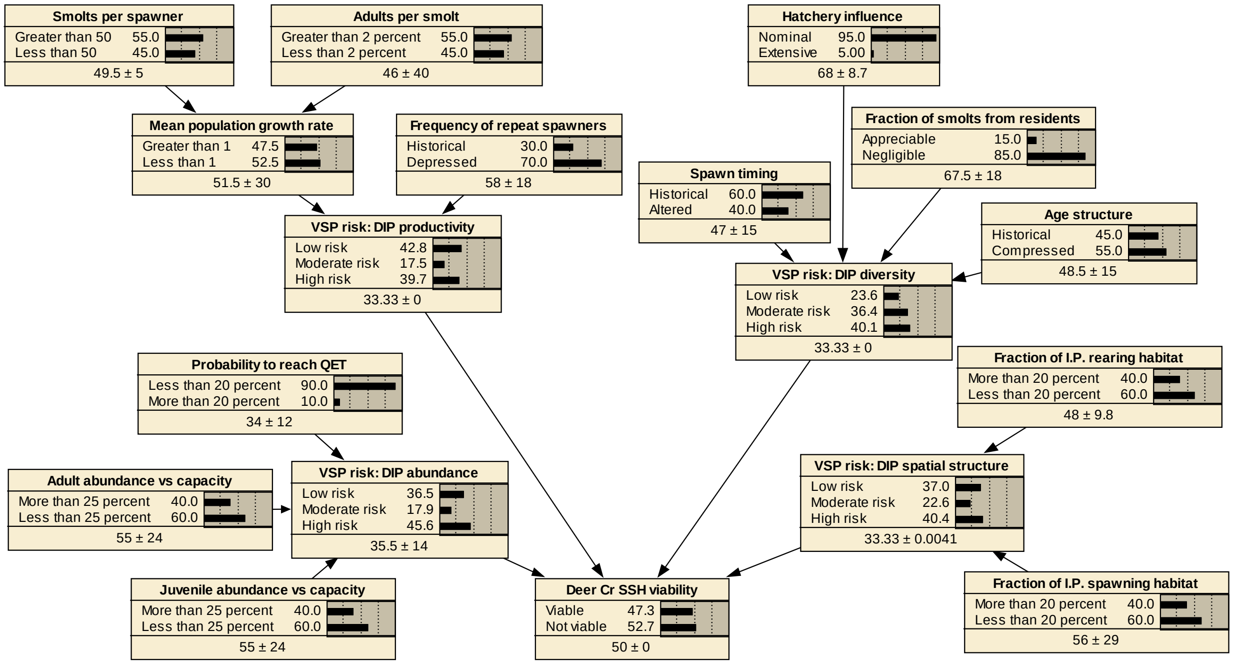

Figure F-10. BN for viability of the Deer Creek Summer-Run DIP in the Northern Cascades MPG.

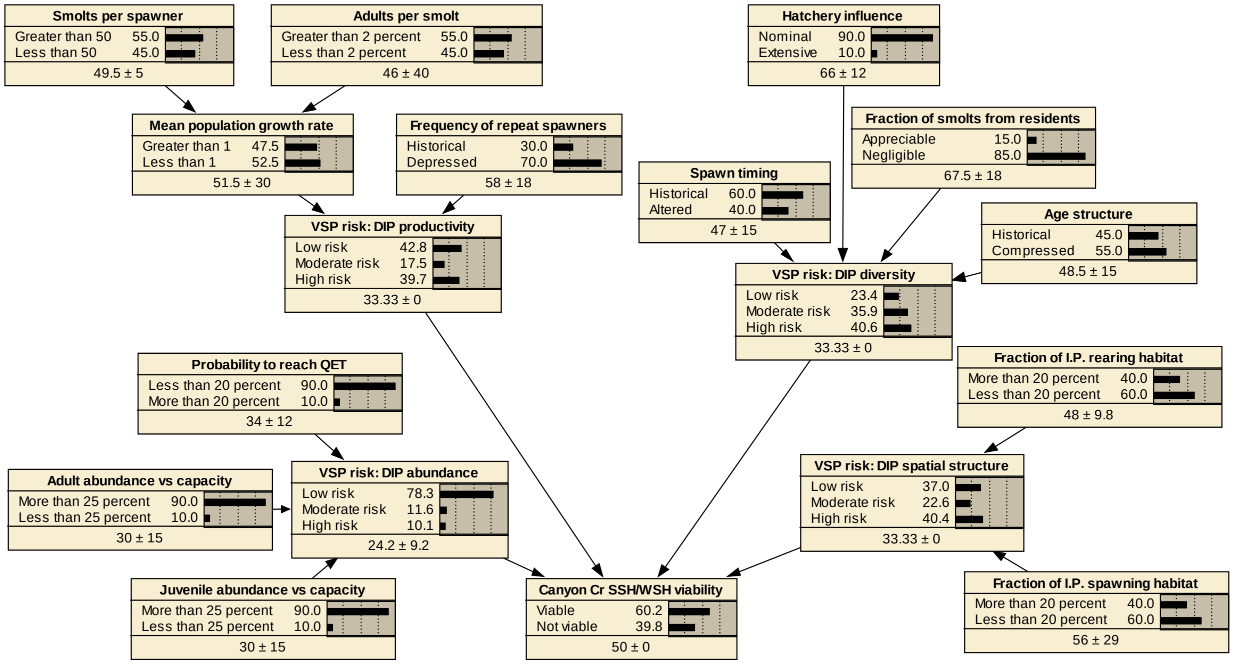

Figure F-11. BN for viability of the Canyon Creek Summer-Run DIP in the Northern Cascades MPG.

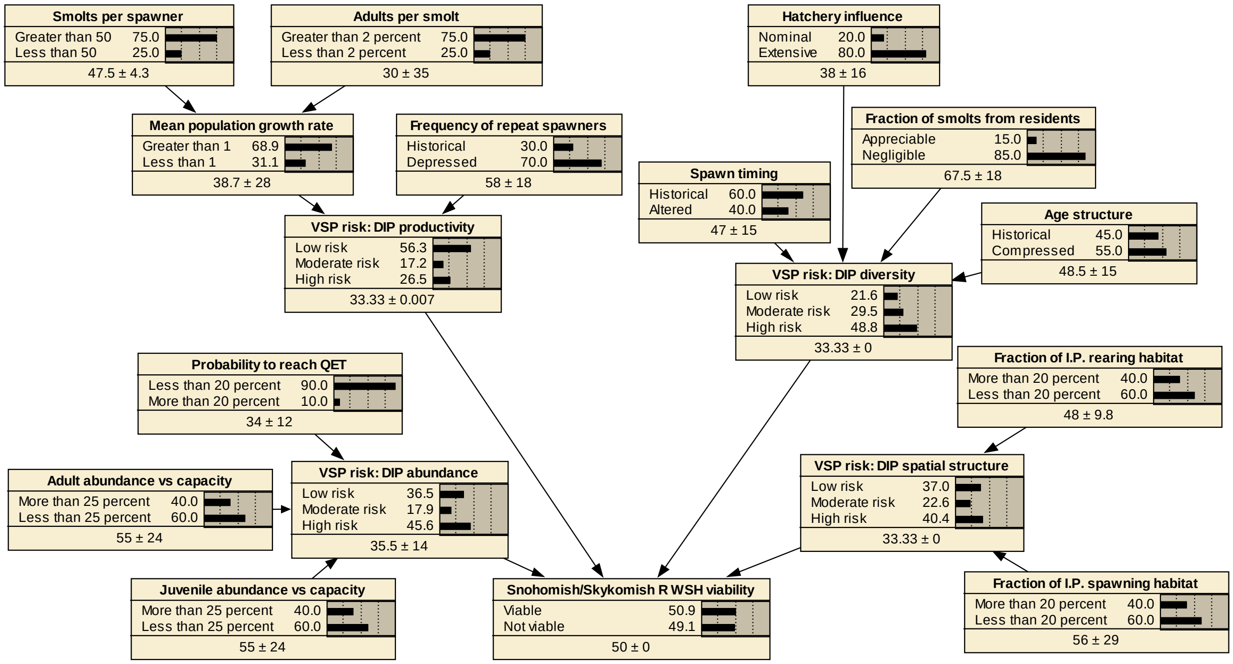

Figure F-12. BN for viability of the Snohomish/Skykomish Rivers Winter-Run DIP in the Northern Cascades MPG.

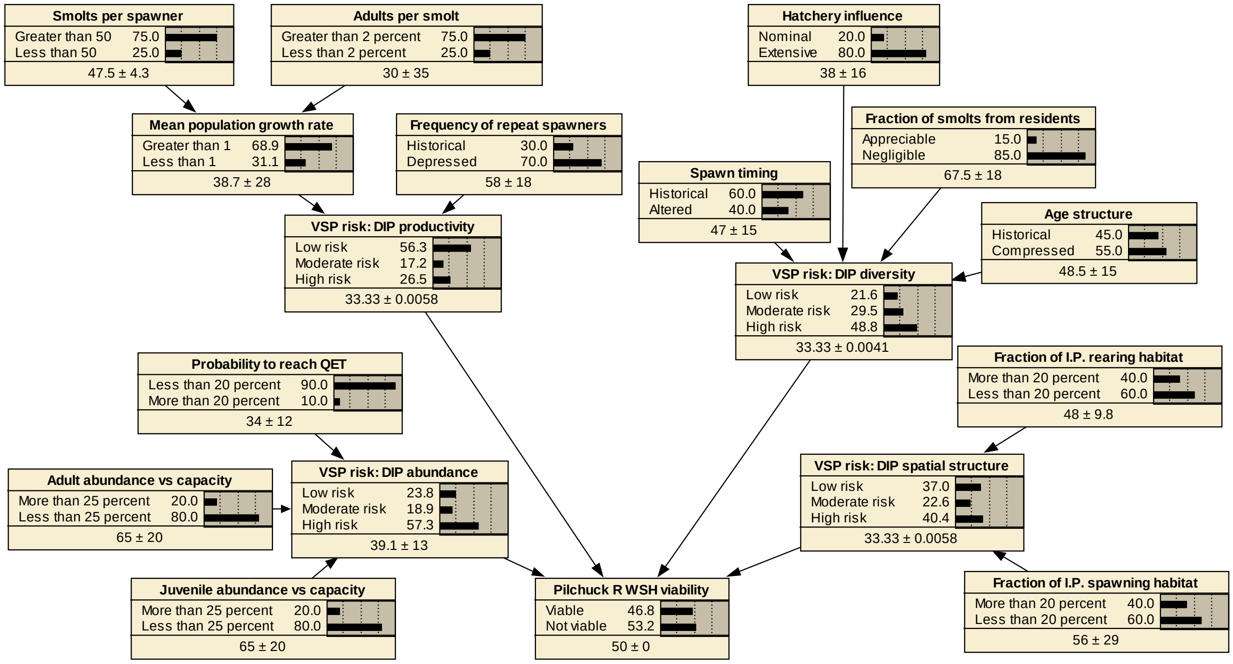

Figure F-13. BN for viability of the Pilchuck River Winter-Run DIP in the Northern Cascades MPG.

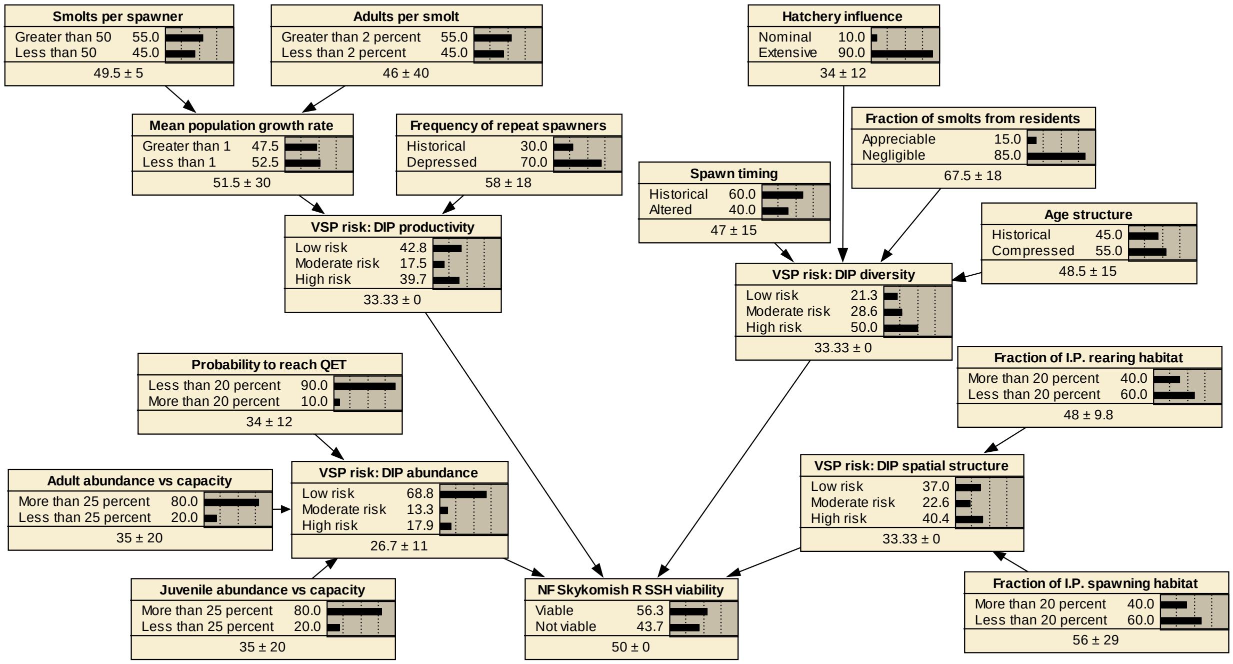

Figure F-14. BN for viability of the North Fork Skykomish River Summer-Run DIP in the Northern Cascades MPG.

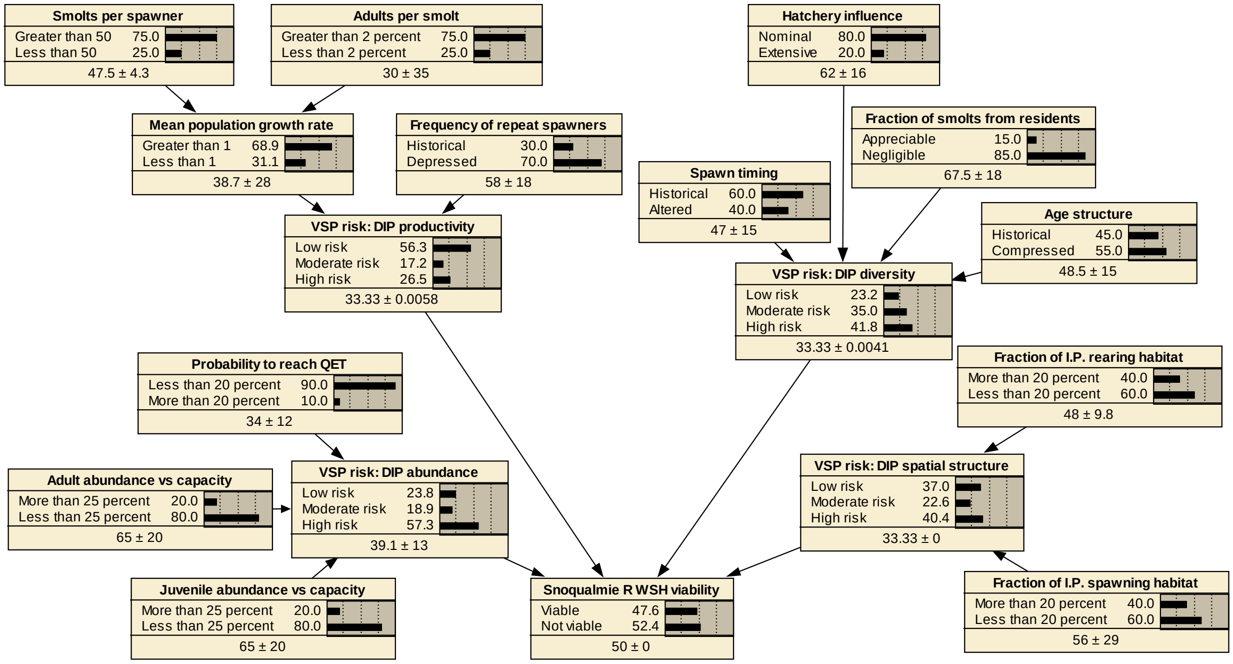

Figure F-15. BN for viability of the Snoqualmie River Winter-Run DIP in the Northern Cascades MPG.

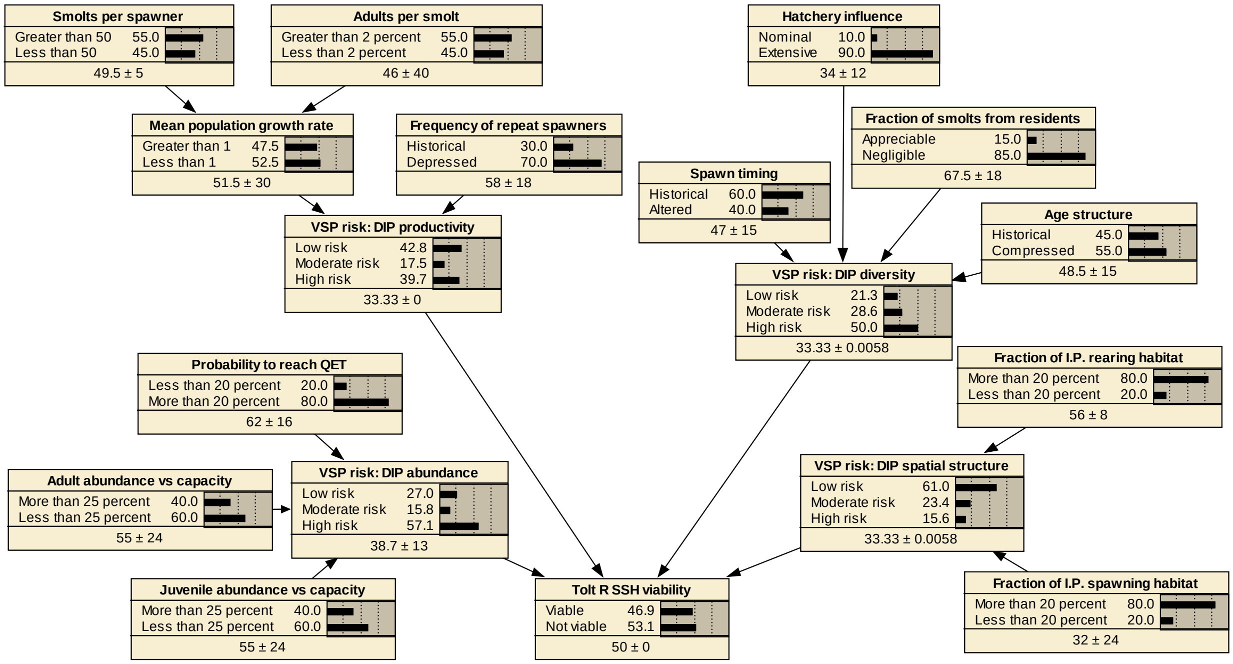

Figure F-16. BN for viability of the Tolt River Summer-Run DIP in the Northern Cascades MPG.

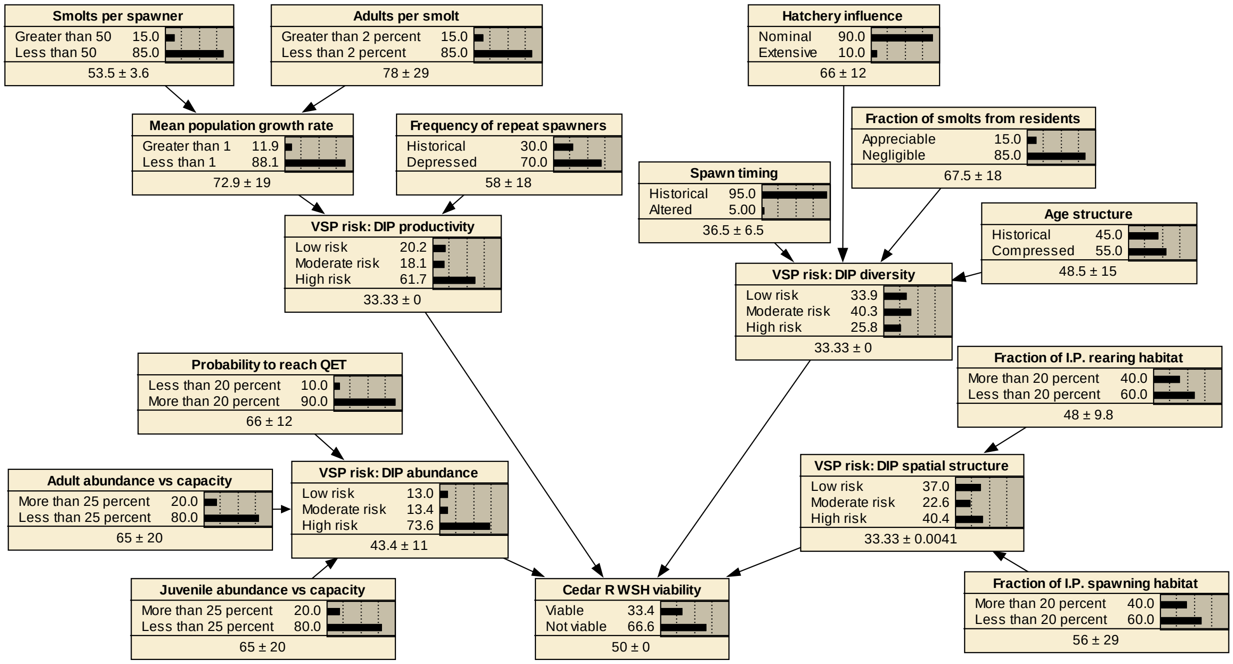

Figure F-17. BN for viability of the Cedar River Winter-Run DIP in the Central and South Puget Sound MPG.

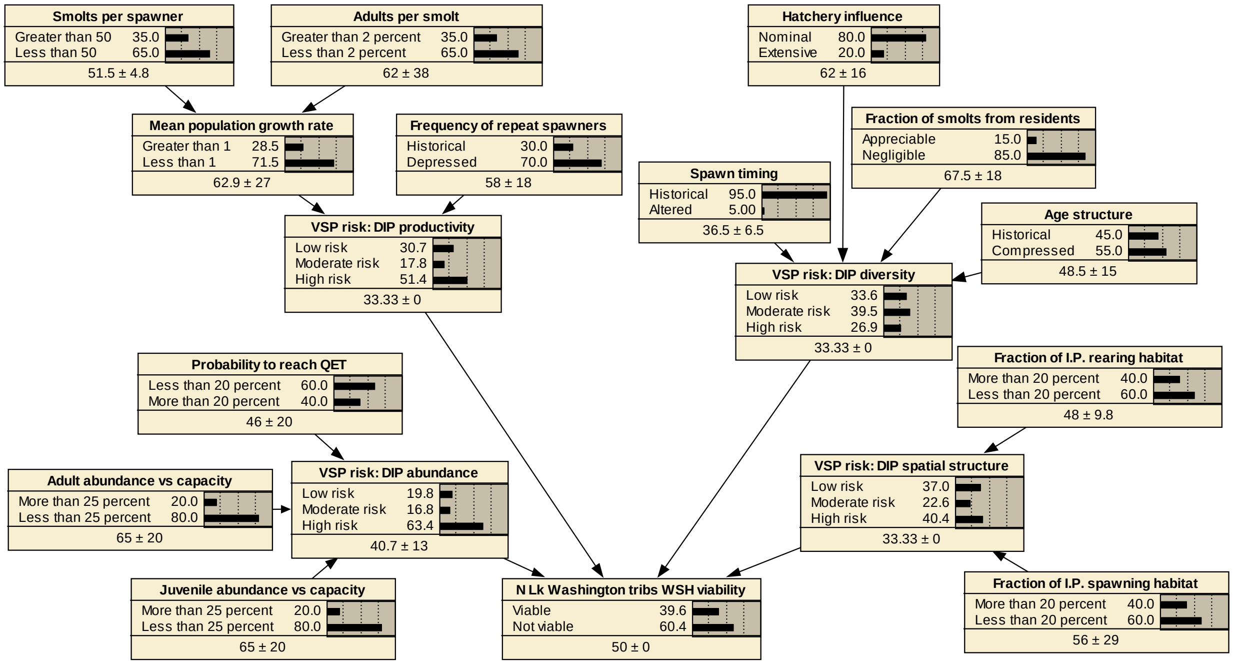

Figure F-18. BN for viability of the North Lake Washington and Lake Sammamish Winter-Run DIP in the Central and South Puget Sound MPG.

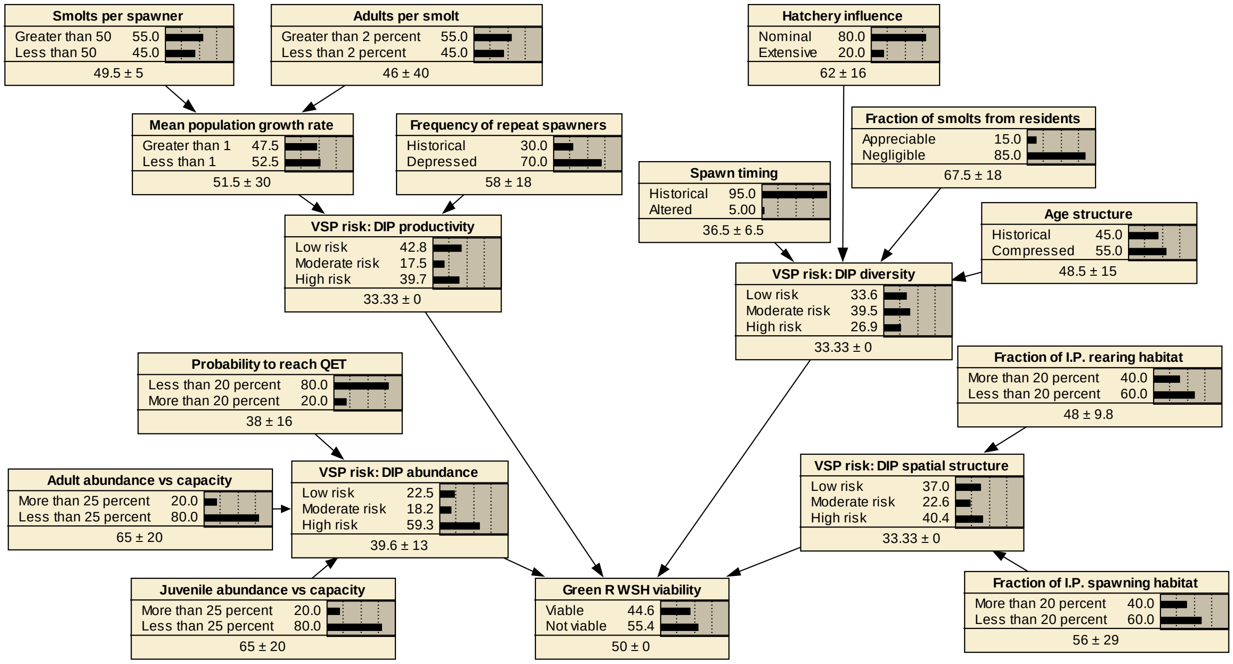

Figure F-19. BN for viability of the Green River Winter-Run DIP in the Central and South Puget Sound MPG.

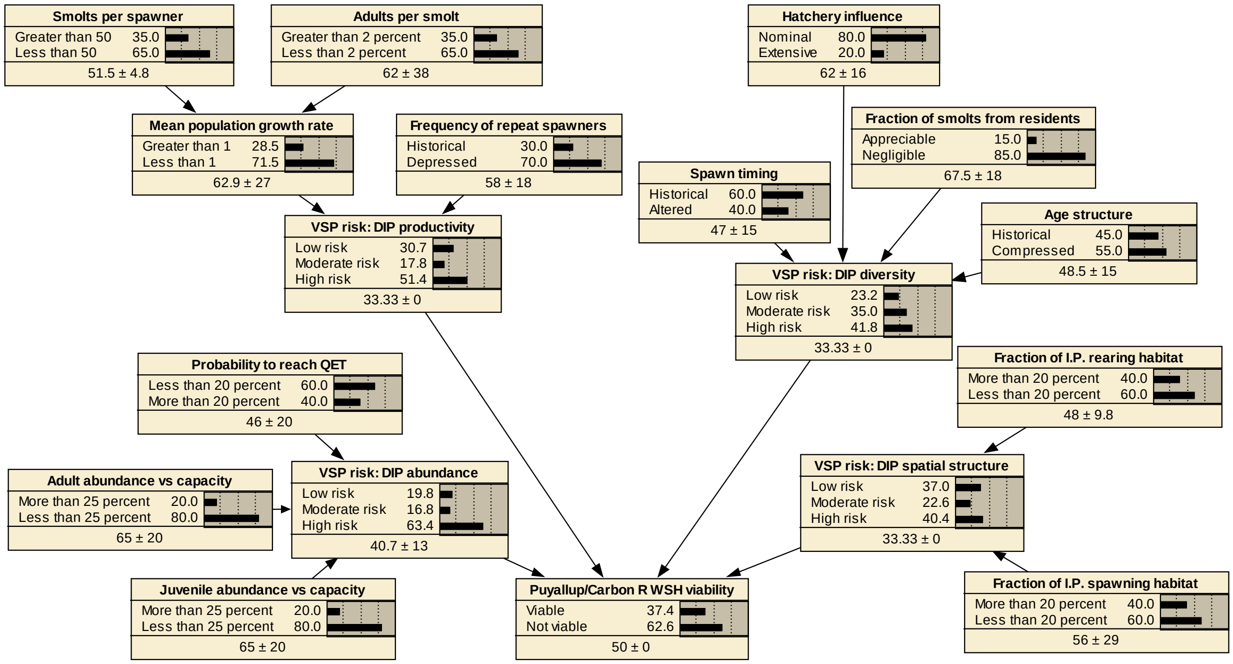

Figure F-20. BN for viability of the Puyallup/Carbon Rivers Winter-Run DIP in the Central and South Puget Sound MPG.

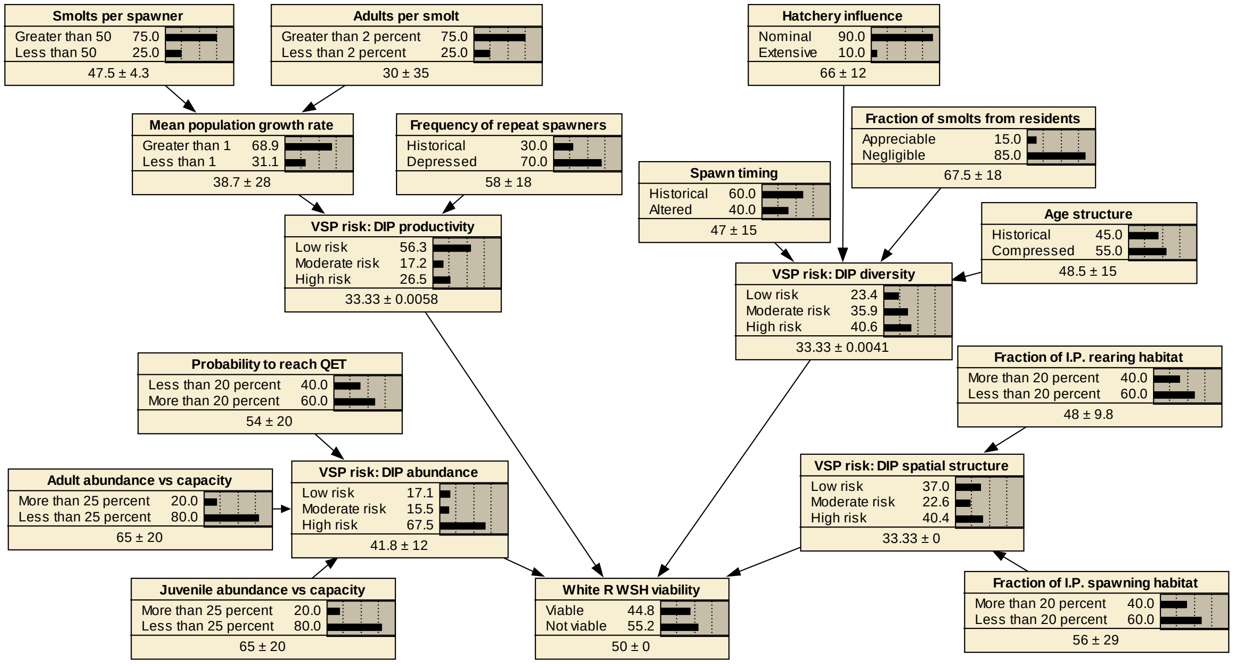

Figure F-21. BN for viability of the White River Winter-Run DIP in the Central and South Puget Sound MPG.

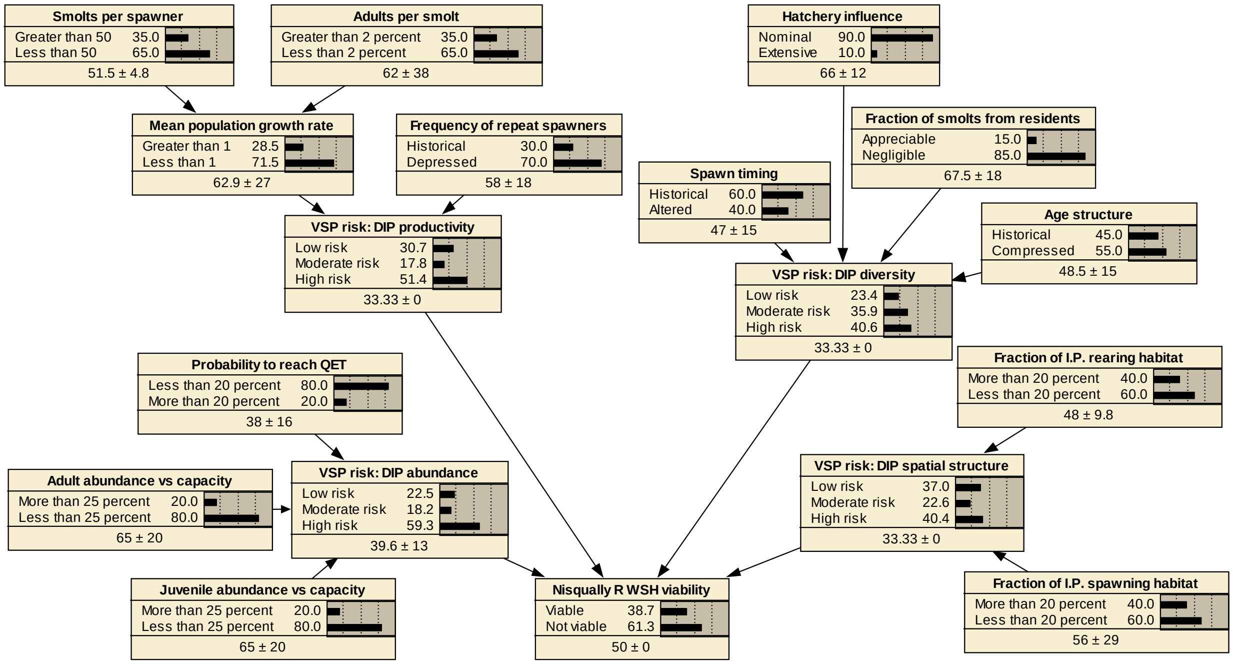

Figure F-22. BN for viability of the Nisqually River Winter-Run DIP in the Central and South Puget Sound MPG.

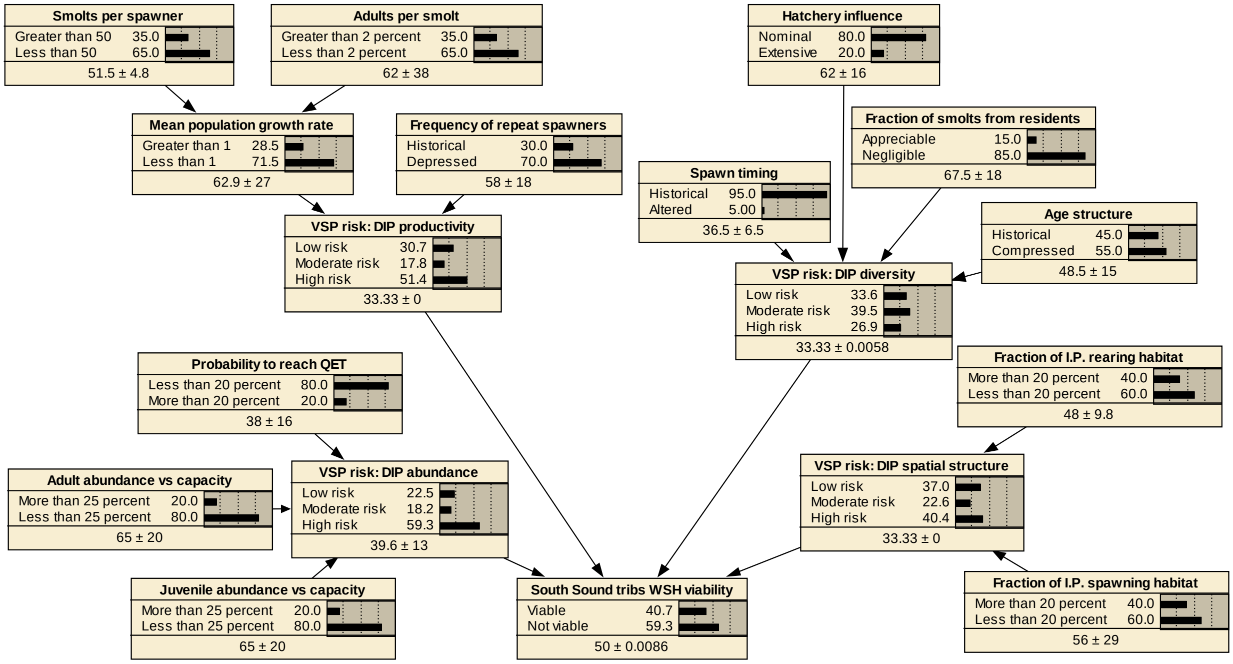

Figure F-23. BN for viability of the South Puget Sound Tributaries Winter-Run DIP in the Central and South Puget Sound MPG.

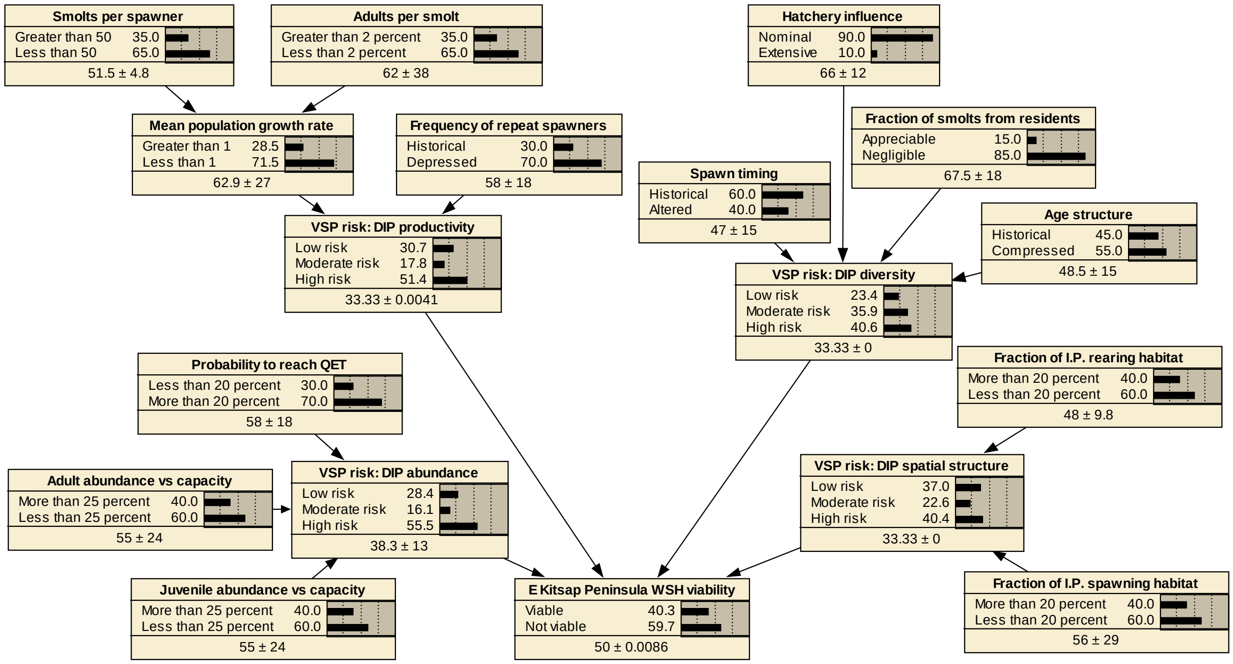

Figure F-24. BN for viability of the East Kitsap Peninsula Tributaries Winter-Run DIP in the Central and South Puget Sound MPG.

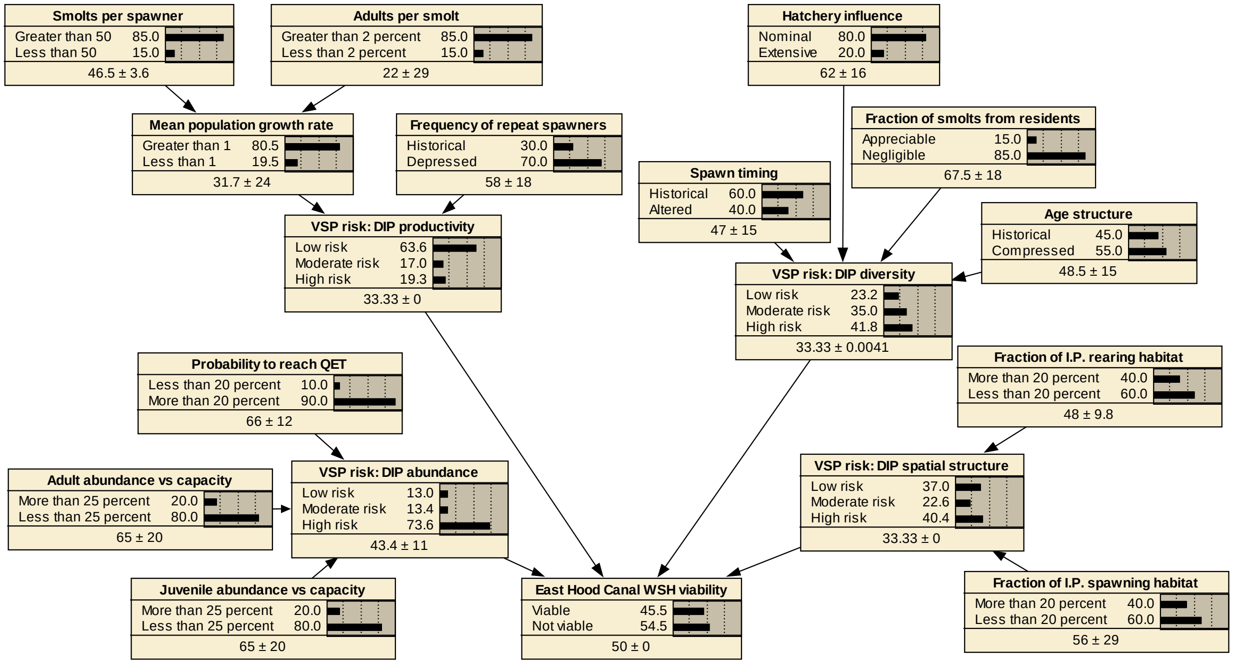

Figure F-25. BN for viability of the East Hood Canal Tributaries Winter-Run DIP in the Hood Canal and Strait of Juan de Fuca MPG.

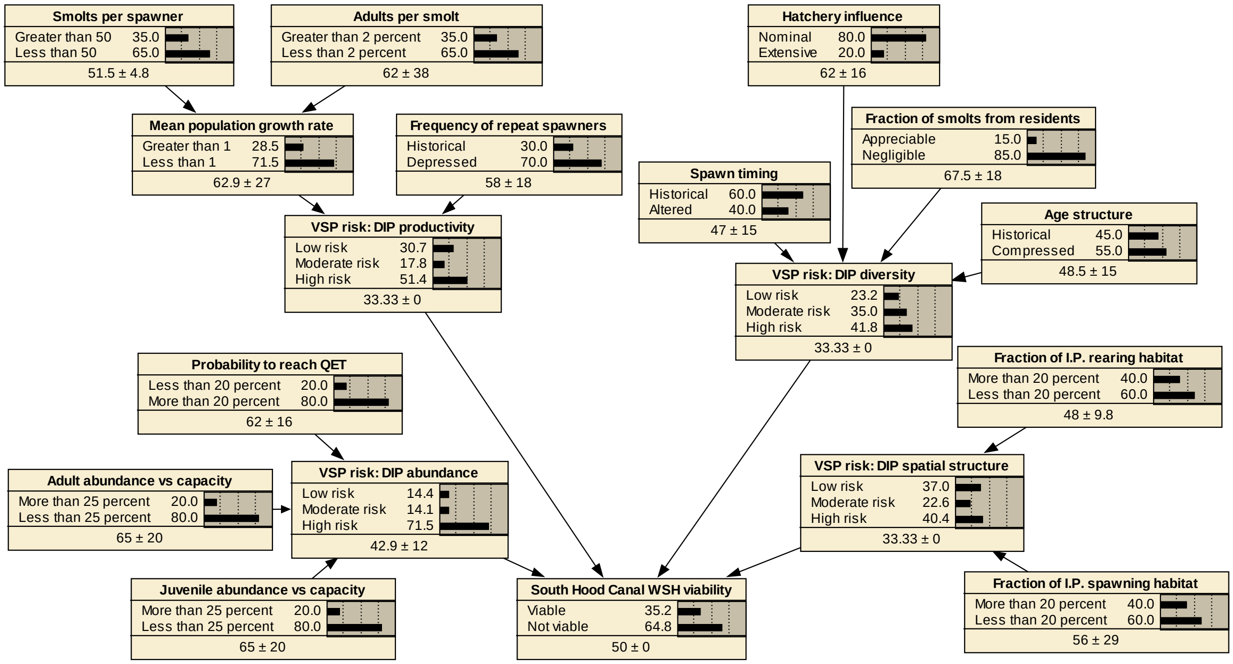

Figure F-26. BN for viability of the South Hood Canal Tributaries Winter-Run DIP in the Hood Canal and Strait of Juan de Fuca MPG.

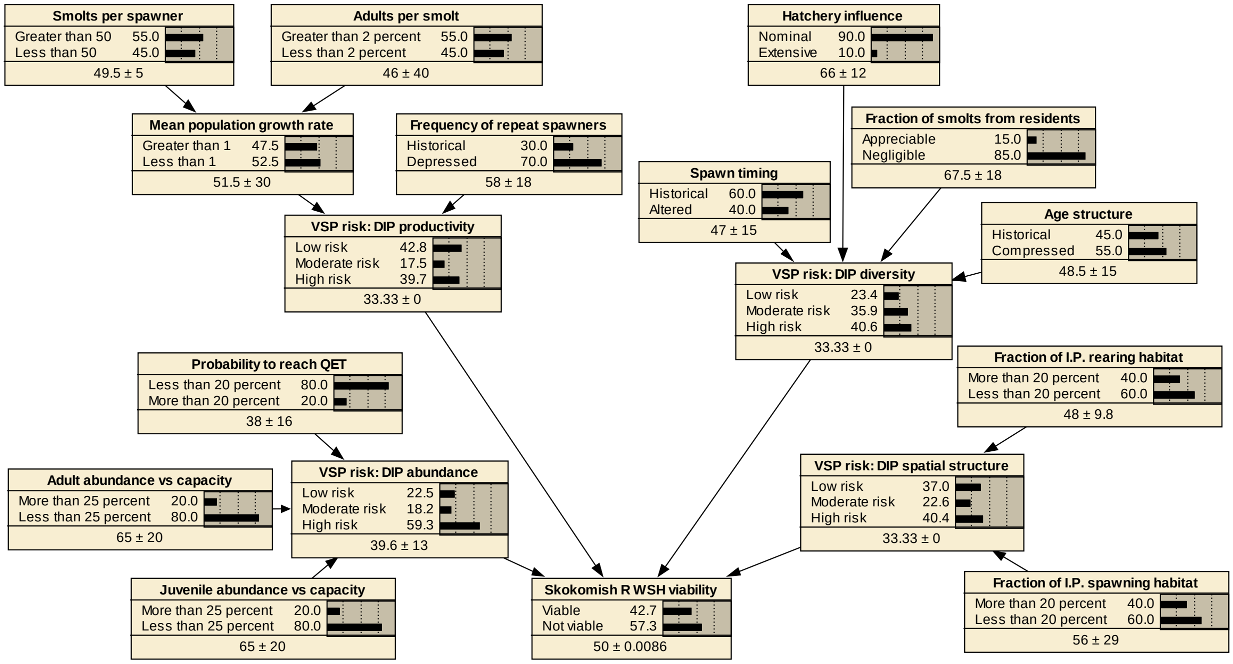

Figure F-27. BN for viability of the Skokomish River Winter-Run DIP in the Hood Canal and Strait of Juan de Fuca MPG.

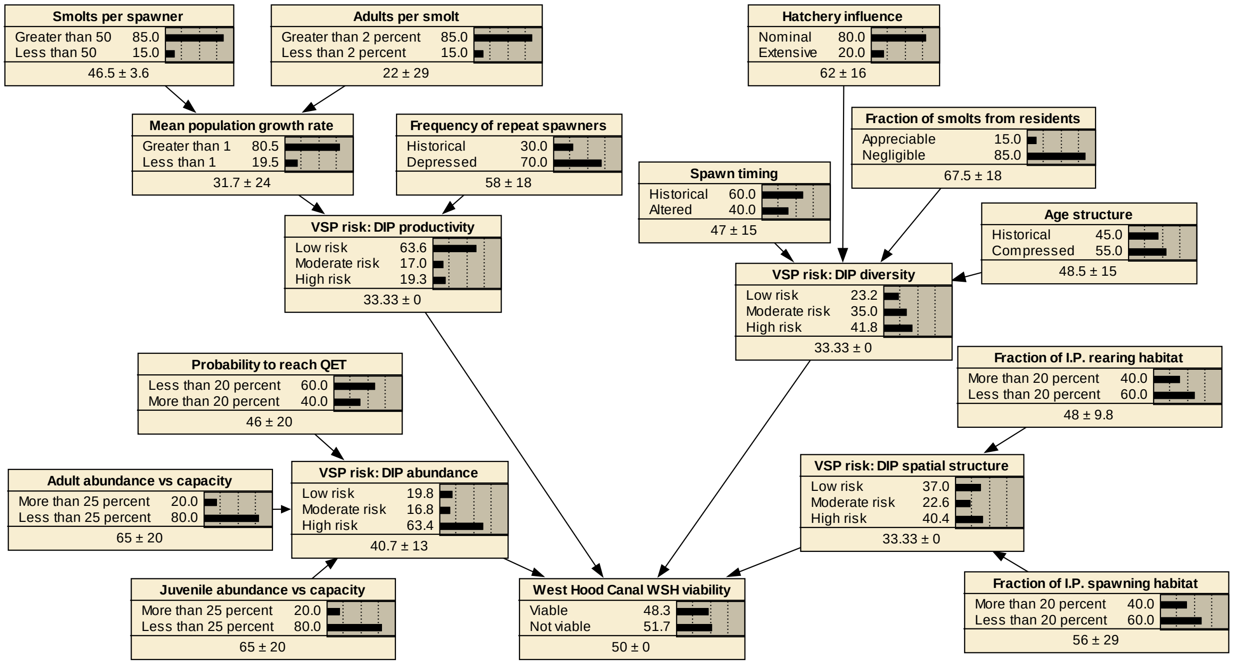

Figure F-28. BN for viability of the West Hood Canal Tributaries Winter-Run DIP in the Hood Canal and Strait of Juan de Fuca MPG.

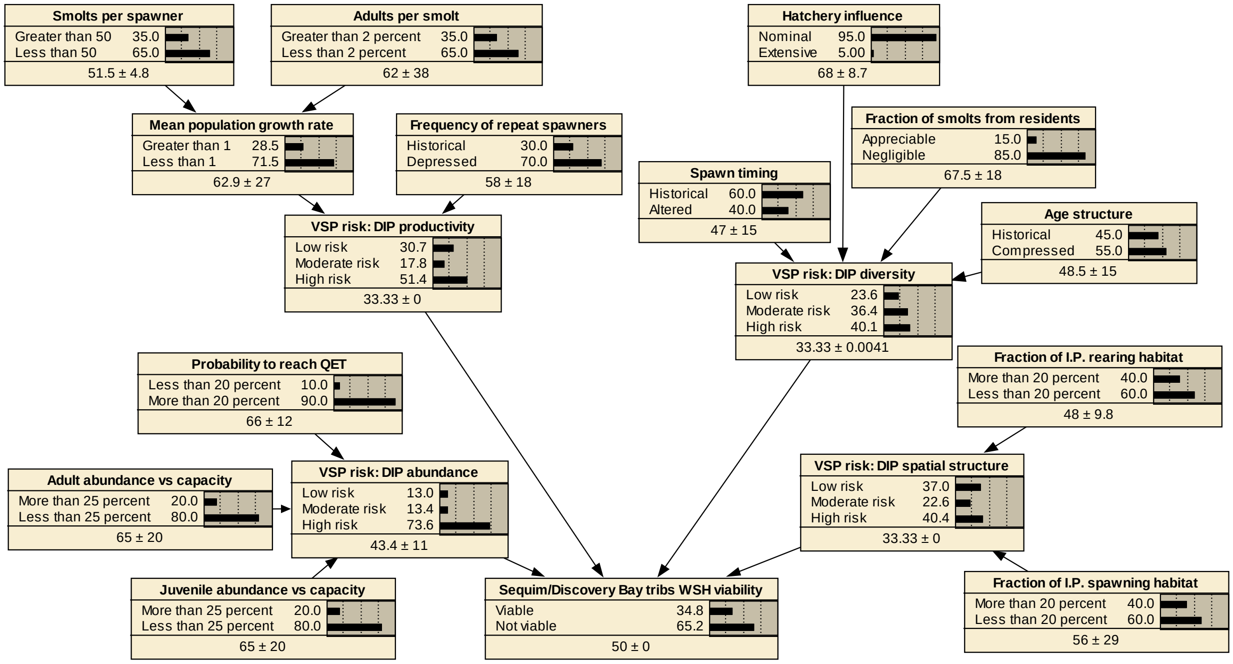

Figure F-29. BN for viability of the Sequim/Discovery Bays Tributaries Winter-Run DIP in the Hood Canal and Strait of Juan de Fuca MPG.

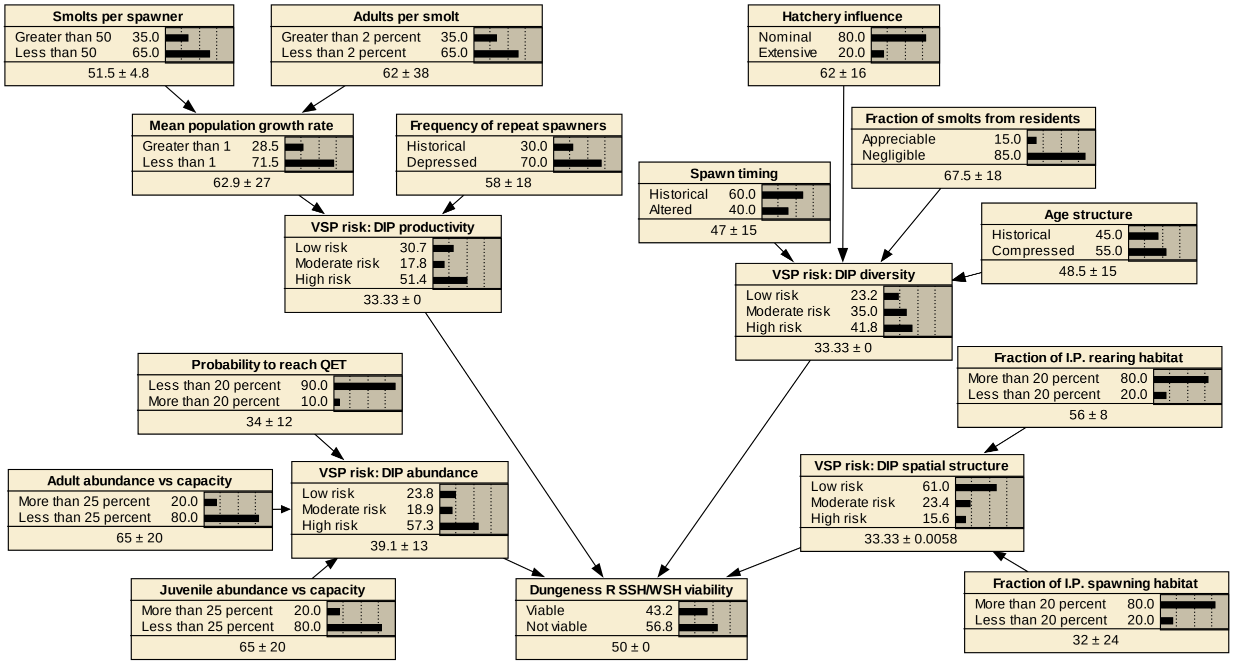

Figure F-30. BN for viability of the Dungeness River Summer-Run and Winter-Run DIP in the Hood Canal and Strait of Juan de Fuca MPG.

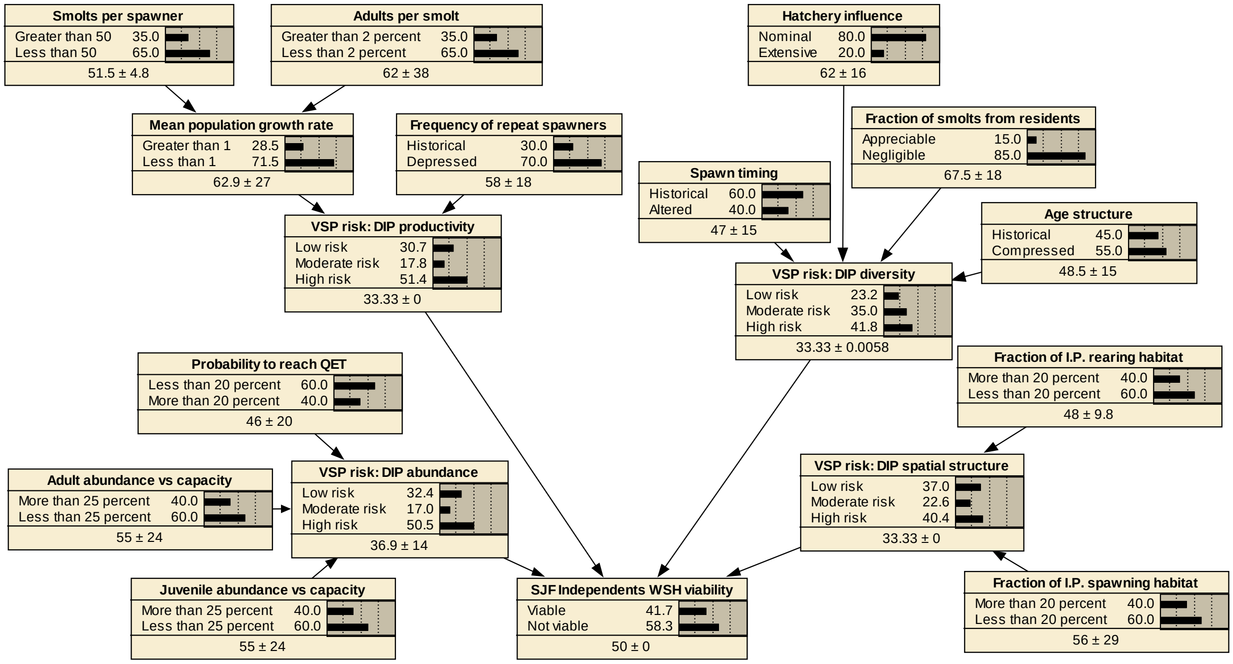

Figure F-31. BN for viability of the Strait of Juan de Fuca Tributaries Winter-Run DIP in the Hood Canal and Strait of Juan de Fuca MPG.

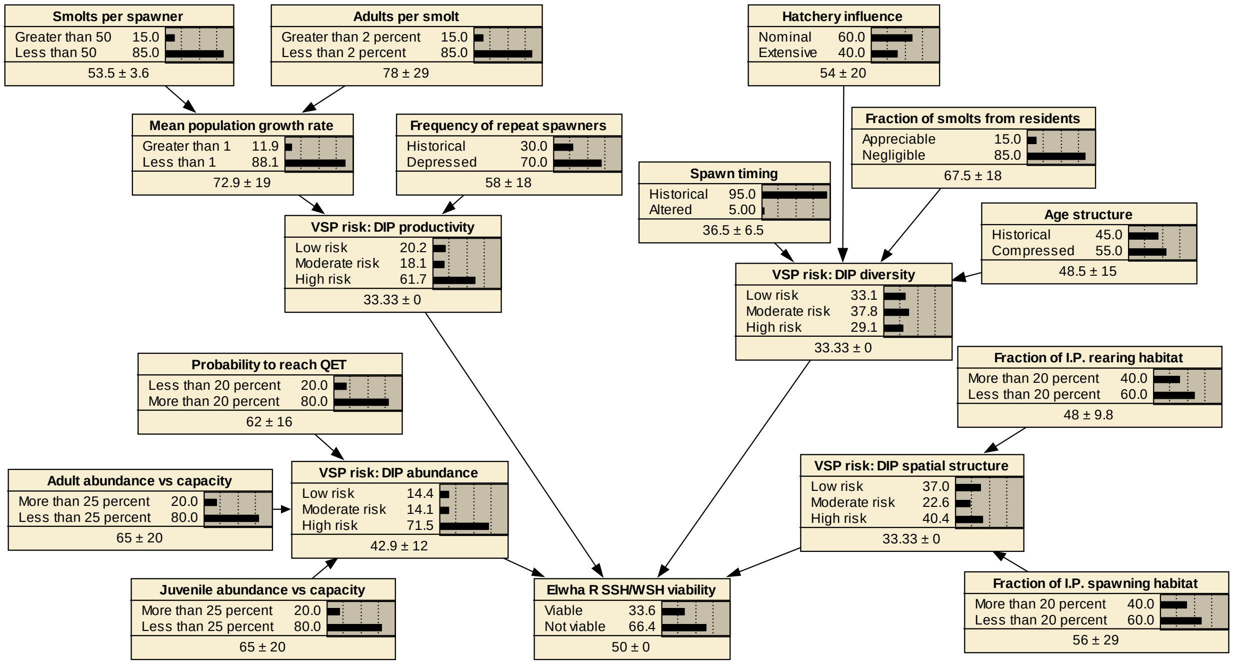

Figure F-32. BN for viability of the Elwha River Winter-Run DIP in the Hood Canal and Strait of Juan de Fuca MPG.

Appendix G: Identifying Key Factors Influencing the Population Dynamics and Productivity of Wild Steelhead

The intrinsic and extrinsic factors that influence the abundance and demographic dynamics of coastal steelhead (*Oncorhynchus mykiss*) populations, which are often small and variable, are typically difficult to identify. Fish survival, growth, age, and fertility all affect individual fitness and, consequently, population productivity, but it is usually difficult to determine which of these factors is more important than another for a particular population. Although demographic trends can be estimated from time series of periodic (e.g., annual) census data, determining the mechanisms underlying these trends requires at a minimum estimating the basic vital rates of survival, growth, and fertility. Ascertaining these factors can help to illuminate particular aspects of the life history that are contributing to population decline or increasing the risk to viability. Determining whether a population is stable or declining is a primary objective of conservation, but uncovering the mechanisms underlying these states is critical to identifying sustainable conservation and recovery strategies.

In an attempt to provide some insight into the factors that might constrain viability or enhance productivity in coastal steelhead populations, in this section we apply a population viability analysis (PVA) to a population of wild steelhead to assess trend, dynamics, and extinction risk. We follow this with an analysis of these dynamics in combination with information on individual survival, size and estimated growth, and fertility. Recent theoretical advances in understanding how demography and adaptation affect population dynamics, combined with powerful genetic approaches to identify relationships among individuals, provide a much more powerful set of tools for achieving these aims than has previously been available. When coupled with careful, regular monitoring of population abundance and key characteristics of potential breeders, such as size, age, fecundity, and timing of breeding, the ability to effectively reconstruct molecular pedigrees for wild populations using DNA markers allows unprecedented opportunities to dissect the components of population dynamics.

To this end, we analyzed demographic and genetic data collected from a small wild population of steelhead from Sashin Creek, Baranof Island, Alaska (Thrower and Joyce 2004, Thrower and Hard 2009). The population is enumerated annually as the individuals pass through a monitoring weir that permits collection of all migrating adults and juveniles. The data collected include size (length in millimeters), adult sex, and stream entry and exit timing. We estimated age and fecundity (number of returning adult offspring) from a molecular pedigree we reconstructed for the anadromous adults in the creek during this period from a set of 13 polymorphic DNA microsatellite markers using FRANz 0.99 software (Riester et al. 2009). FRANz uses a full Bayesian probability model to assign parentage; for these data we set the threshold for parentage assignment at 95%.

The temporal trend in the size of the anadromous population between 1986 and 2011 is shown in the upper left panel of Figure G-1. A PVA applied to this time series with the Multivariate Autoregressive State-Space (MARSS) Program, version 2.3, (Holmes and Ward 2011) in the R statistical environment (R Development Core Team 2012) indicates that, over this period, the population is declining at a rate of 3.4% annually, corresponding to a population growth rate of 0.967; the process variance (variance thought to arise from genetic or demographic stochasticity or from random environmental variation), σ_p^2, is estimated at 0.00019. The MARSS model, described more fully in the main text of this technical memorandum, uses a state-space framework to project the population's dynamics 100 years into the future. What this PVA indicates is that, based on the observed trend, the population is likely to reach a minimum threshold size of 10 adults within 20–25 years (Figure G-1, upper right and lower left panels); over the next 17–27 years, the population is expected to decline by up to 50% if current conditions persist (Figure G-1, lower right panel).

What the PVA alone does not—and cannot—tell us is what is driving these dynamics. To aid this effort, we need additional information and another approach. One approach is to examine how particular phenotypes contribute directly to population productivity. Coulson et al. (2006) and Pelletier et al. (2007) developed a method they called "de-lifing," which essentially estimates how population growth rate varies when individuals are removed sequentially from the population with replacement. The method is akin to the jackknifing procedure common in

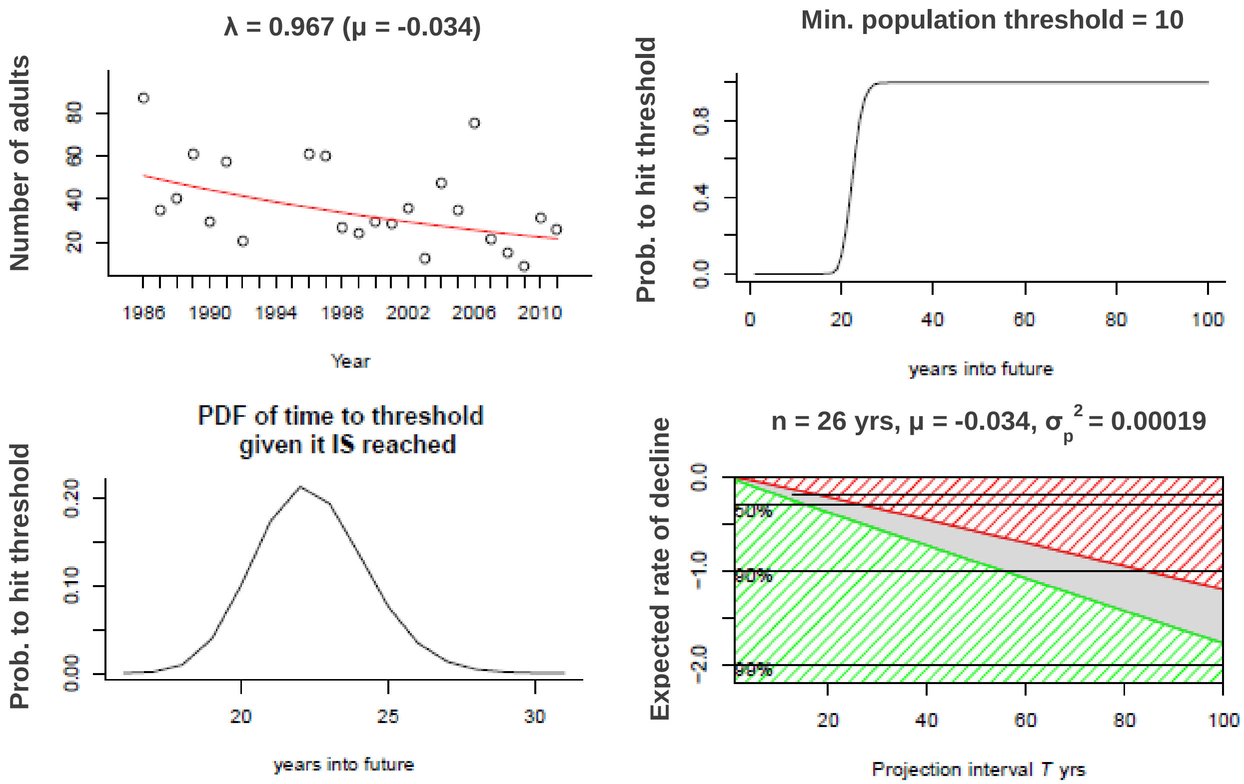

Figure G-1. PVA of wild Sashin Creek steelhead, based on a time series of adult escapement between 1986 and 2011.

statistical practice. De-lifing fits linear models that estimate the relationships between fitness, measured here as lifetime reproductive success, and phenotypic variation, then examines how population productivity changes with these variables. The analysis indicates that males residing longer in the stream make significantly higher contributions to population productivity ($P < 0.05$); there is a tendency for larger females to contribute more to productivity, but these results are not significant ($P > 0.05$). The analysis is summarized graphically in Figure G-2. In this figure, contour plots of relative fitness for males and females sampled from the pedigree suggest that large females tend to have higher fitness; stream entry and residence time information obtained from the molecular pedigree suggests further that larger males that remain in the stream longer have higher fitness, while the large females that spawn and then return to the ocean within 2–3 weeks tend to have higher fitness.

Another approach that is gaining traction in the ecological literature, called an integral projection model (IPM), is appealing because it can help to determine the direct influences of individual survival, growth, and reproduction on population productivity. An IPM is an innovative analytical tool designed to estimate the temporal dynamics of a population when the fates of its members depend on characteristics that are continuous or reflect a combination of discrete and continuous characteristics. These characteristics or states include traits such as size (length, weight), age, survival, fecundity, and distribution of offspring sizes. In much the same way that an analysis of function-valued traits related to an analysis of discrete traits (Kingsolver

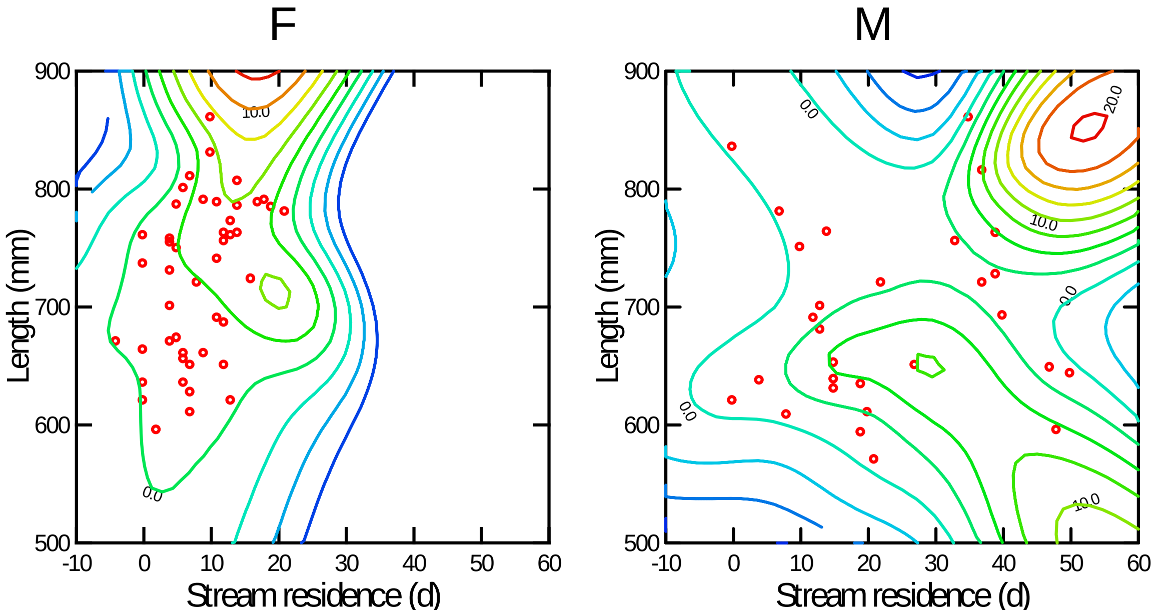

Figure G-2. Contour plots summarizing for adult females (left) and males (right) the relationship between relative fitness, estimated as individual contribution to population growth rate (z-axis, i.e., the contours), size in length (y-axis), and stream residence time in days (x-axis). The data are given by the open red circles and the contour lines represent contours of individual fitness (× 1,000). The highest fitness values are represented by the red peaks on the plots, with blues lines representing the valleys of lowest values of fitness and green lines slopes of intermediate values. The contributions to population productivity were estimated according to the formulas in Coulson et al. 2006.

et al. 2001), an IPM is analogous to a matrix population model (Leslie 1945, 1948, Lefkovitch 1965, Caswell 2001), the classical approach to estimating vital rates that affect population productivity and persistence from demographic data on stage or age transitions. A key difference between an IPM and a matrix population model is that the IPM does not discretize state variables a priori, and its approach is pertinent to the study of population dynamics of species where an individual's state (e.g., sexual maturity or reproductive lifespan) depends directly on its growth, size, or age. This is important because assuming that continuous underlying states are discrete can lead to bias in estimating population dynamic parameters (Metcalf et al. 2013). An IPM evaluates the distribution of individuals across these state variables between census times (typically annually) by projecting from models that estimate the underlying vital rates as a function of the continuous or quasi-continuous state variables (Easterling et al. 2000, Ellner and Rees 2006). When combined with pedigree or marking methods that identify individuals and their relatives, an IPM can provide this information.

Why should we consider such approaches as the IPM? The answer is simple: When we cannot conduct robust, replicated experiments to ascertain how aspects of life history affect population dynamics, the best alternative is to apply appropriate models to the available information, then analyze these models with suitable quantitative methods that can identify these relationships. Indeed, if one could simply rely on correlations between demographic trends and changes in key life history parameters to confirm hypotheses about the factors that limit viability in natural populations, then methods such as IPMs and evolutionary models would be unnecessary in inferring causality. Unfortunately, this fact and the inability to conduct the relevant experiments that might answer these questions for natural populations in particular environments make such methods essential. The IPM is one rigorous and systematic way to try to uncover the relationships between these key demographic and evolutionary patterns. In essence, it is a formal framework for analyzing the dynamics of a system, such as a natural population, that is structured by a mix of discrete and continuous characters that can be measured. The study of the dynamics of these character distributions has the potential to afford insight into ecological dynamics across a range of levels of organization, including communities as well as populations (Coulson et al. 2010, 2011, Coulson 2012).

The data we considered for the demographic analysis constituted a subset of 144 individuals from five consecutive broods and representing two complete generations (parents and their adult offspring) collected between 1997 and 2008. We applied the IPM to these data using IPMpack software, version 1.6 (Metcalf et al. 2013) in R version 2.15.2 (R Development Core Team 2012). We built a survival function for the IPM around a polynomial logistic regression relating size in year t to survival from year t to year t +1, and we built a growth function around a polynomial regression relating size in year t to survival from year t to year t +1.

An IPM is characterized by a kernel or function $\boldsymbol{K}$ that represents the probabilities of growth and survival between stages and the production of offspring and their recruitment

$$n(y, t+1) = \int_L^U \boldsymbol{K}(y, x) n(x, t) dx$$

where $n(y, t + 1)$ is the distribution across size y of n individuals at census $t + 1$, $n(x,t)$ is the distribution across size x of n individuals at census t, and L is the lower and U the upper size limit represented in the IPM (Metcalf et al. 2013).

K is itself composed of one kernel that defines survival and growth (the ***P*** kernel) and another kernel that defines reproduction (the ***F*** kernel)

$$\int_L^U \boldsymbol{K}(y,x)n(x,t)dx = \int_L^U [\boldsymbol{P}(y,x) + \boldsymbol{F}(y,x)]n(x,t)dx$$

where the ***P*** kernel is given by

$$\int_L^U \boldsymbol{P}(y,x)n(x,t)dx = \int_L^U survival(x)growth(y,x)dx$$

and the ***F*** kernel is given by

$$\int_L^U \boldsymbol{F}(y,x)n(x,t)dx = \int_L^U c_1c_2c_z \dots fecundity_1(x)fecundity_2(x) \dots fecundity_z(y,x)\, dx$$

where c_i are constants for the functions relating size and fecundity and *z* indicates the stages or years (Metcalf et al. 2013).

Using the average and SD of adult length estimated from the data (706.54 and 67.68 mm, respectively), we first constructed survival and growth elements of the IPM. We did not have longitudinal size estimates for most (123) of the individuals, only estimates of average growth in length between years estimated from the pedigree. We fitted a polynomial regression to the longitudinal size data available for 22 adults to estimate the size in year *t* + 1 (e.g., a 6-year-old adult) from the size in year *t* (e.g., a 5-year-old adult) for all adults, based on a mean of approximately 9.3% annual growth in length and with normally distributed random error based on the variation observed in size in year *t*. The fitted regression was $y_{t+1} = 533.2 - 0.0872\, y_t + 0.0006\, y_t^2$ ($r^2 = 0.778$). Mean generation time was 6 years. We defined the survival component of the ***P*** kernel as a polynomial logistic regression relating size and size^2 in year *t* to survival from year *t* to year *t* + 1 (y_{surv}), and the corresponding growth component as a polynomial regression relating size and size^2 in year *t* to size in year *t* + 1 (y_{growth}). The models provided the following fits:

$$y_{surv} = -13.52 + 0.00297 * size - 0.00001633 * size^2$$

$$y_{growth} = 0.005988 - 0.02961 * size + 0.0007527 * size^2$$

These data are plotted in Figure G-3, with curves representing expected values (from the two equations immediately above) fitted to the points. The relationship between annual growth and initial size was nonlinear. There was no evidence that adults returning earlier or later than average had lower or greater growth or survival. The plot of survival on size suggests that the increase in annual survival with size is appreciable, perhaps 30–40% for fish greater than 800 mm vs. 10% for smaller fish.

Diagnostics applied to the estimated ***P*** matrix, using a starting size of 550 mm and a target size of 750 mm, showed no evidence of outliers or that the size range or the number of size bins selected was insufficient (Figure G-4). These diagnostics do not indicate any serious problems in fitting vital rate functions to the data or estimating IPM components.

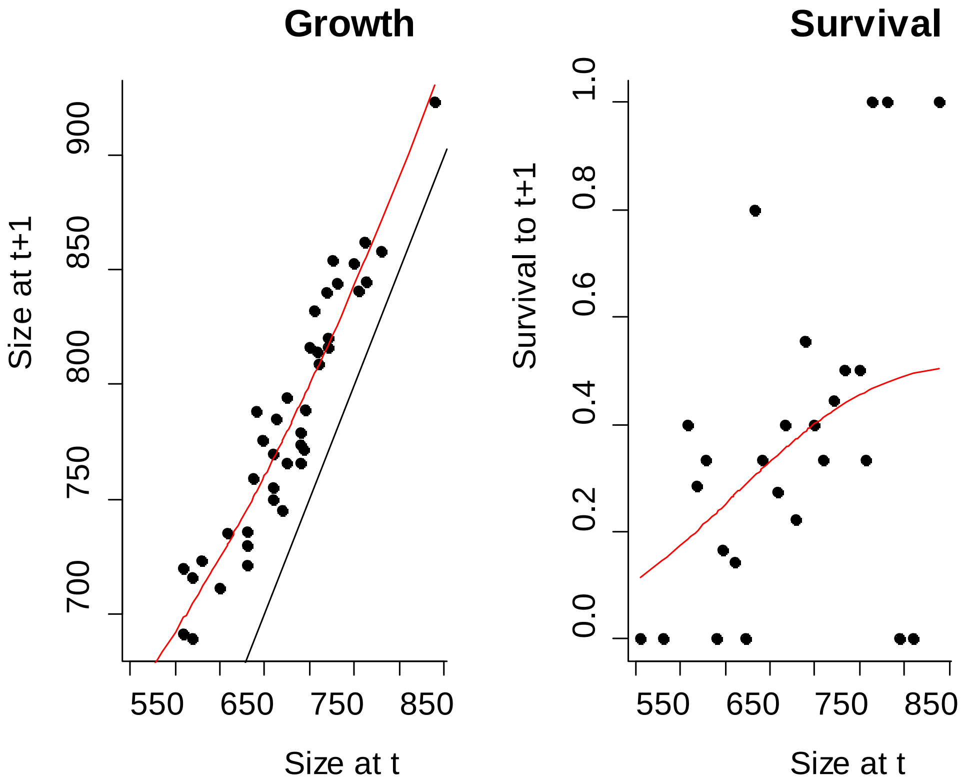

Figure G-3. Estimated growth and survival functions for Sashin Creek steelhead. The corresponding red curves are the fitted expected values. The black line indicates the 1:1 relationship of sizes in subsequent years.

We then defined a normally distributed fertility kernel ***F*** representing adult offspring production based on parental size in year *t* (y_{fec}), using the observed data to define the parameters of that distribution. The general linear model estimate was

$$y_{fec} = 0.224535 + 0.001198 * size$$

Figure G-5 indicates no consistent pattern in adult progeny produced as a function of adult size. There is also no consistent pattern with adult run timing (data not shown). The maximum number of adult offspring produced by a spawning adult was four over the time series, produced by adults as small as 610 mm and as large as 750 mm.

Figure G-6 depicts the survival-growth transition kernel ***P*** that accounts for survival and growth transitions only (i.e., no covariates), between size in year *t* and size in year *t* + 1, using 70 mesh points. The transition kernel estimates a weak curvilinear fit of survival and growth transitions to the size data, with the highest (and much higher) probability of surviving and growing to return the following year for the largest adults. From this transition kernel, the estimates of mean life expectancy and time to reach the target size across the range of mesh points in the survival-growth transition kernel ***P*** are shown in Figure G-7. The left plot predicts a survival advantage for larger adults, as expected from the relationship observed in Figure G-3; the predicted life expectancy rises from about 1 year for adults less than 600 mm long to about 1.75 years for fish 750 mm long. The right plot of passage time estimates that surviving

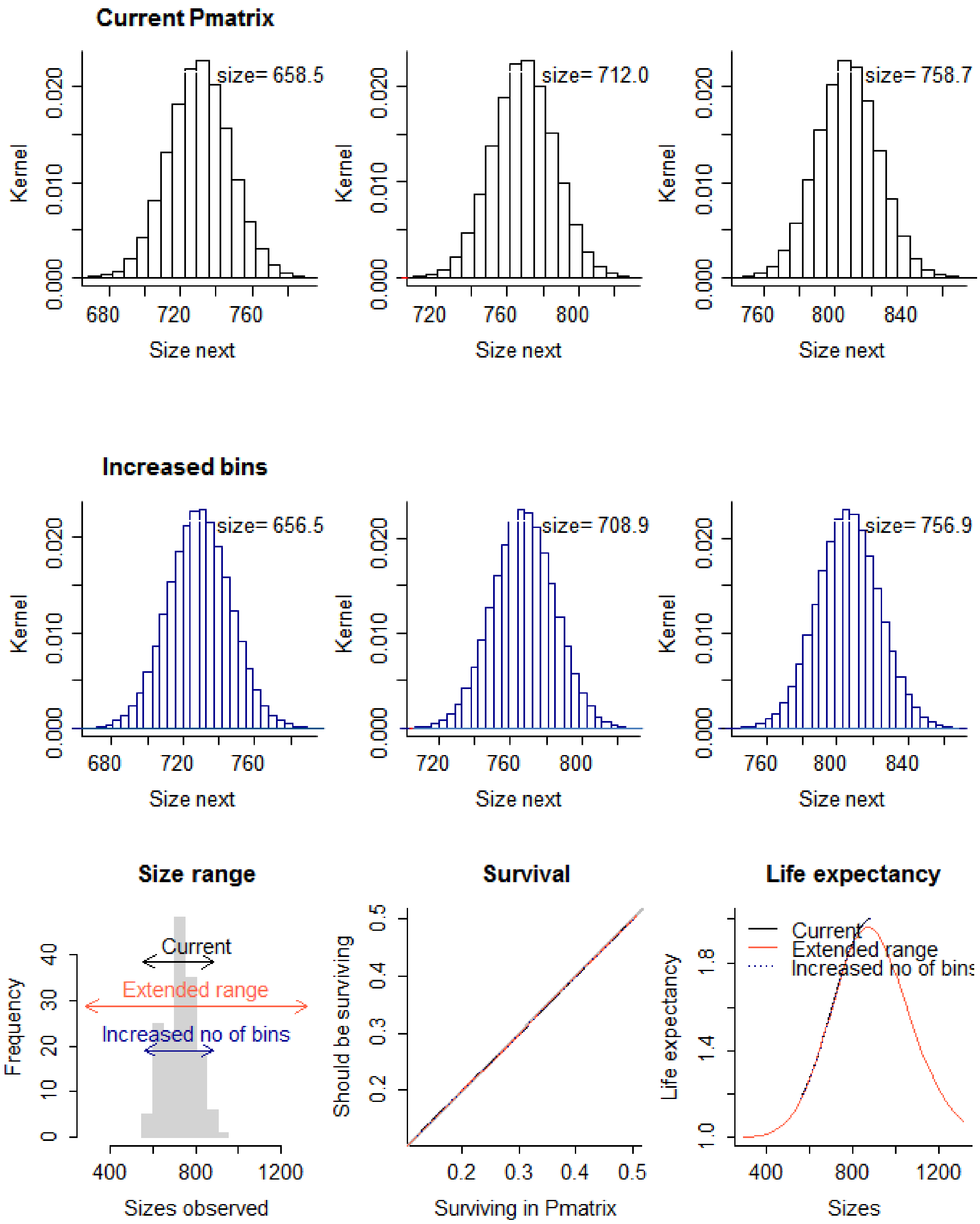

Figure G-4. IPM diagnostics applied to the steelhead survival and growth data, indicating an appropriate size range and resolution in number of size bins to estimate IPM matrices. There is no mismatch between the observed sizes and the range fitted in the ***P*** matrix. The red lines and blue bars indicate the results of increasing the size range and bin resolution, respectively, on the model fitting. The correspondence of the black and red lines in the Survival and Life expectancy plots indicate that the predictions of the discretized matrix do not depart appreciably from the fitted survival model over the range of observed values.

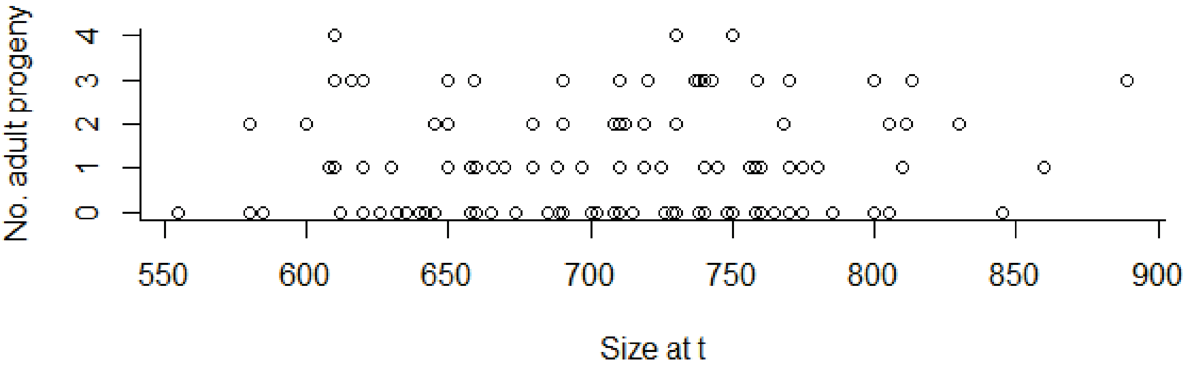

Figure G-5. Estimated number of adult offspring as a function of adult size in Sashin Creek steelhead.

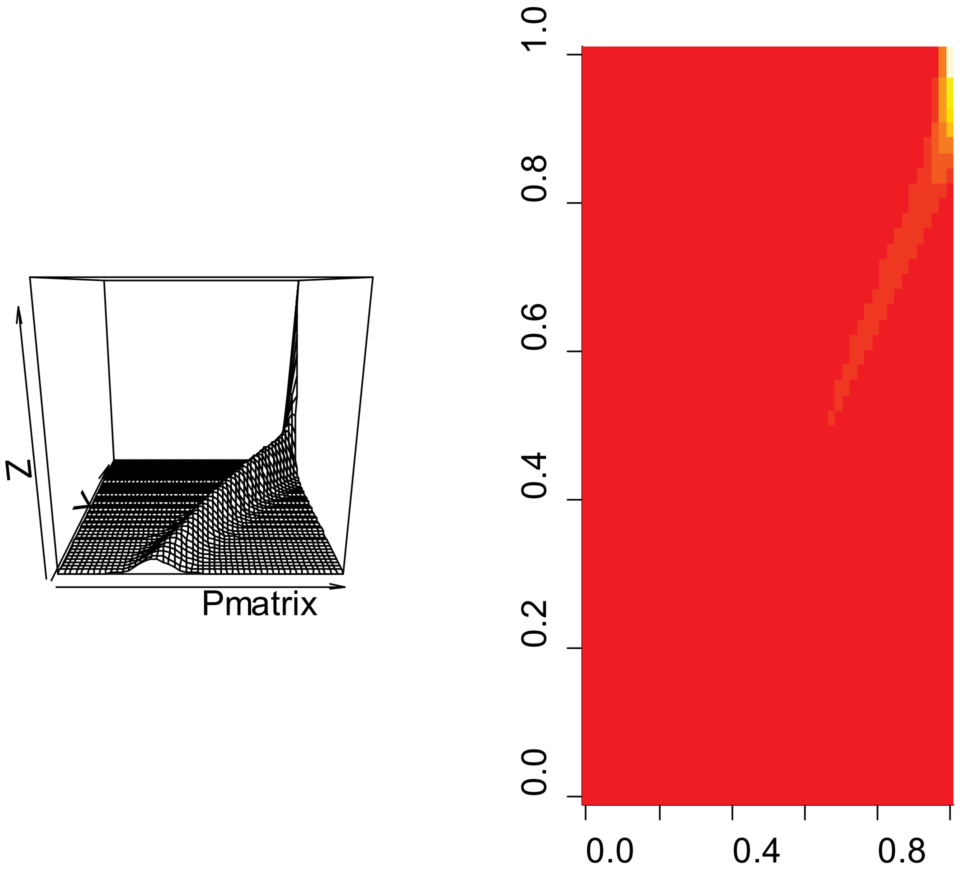

Figure G-6. Left, a graphical depiction of the survival-growth transition kernel ***P*** indicating the probability of a survival and growth transition (z-axis) as a function of sizes in adjacent annual censuses, showing a weak curvilinear fit to the size data. Right, a view of the kernel from above, with the red regions indicating low probabilities and yellow regions (upper right quadrant) higher probabilities of a transition.

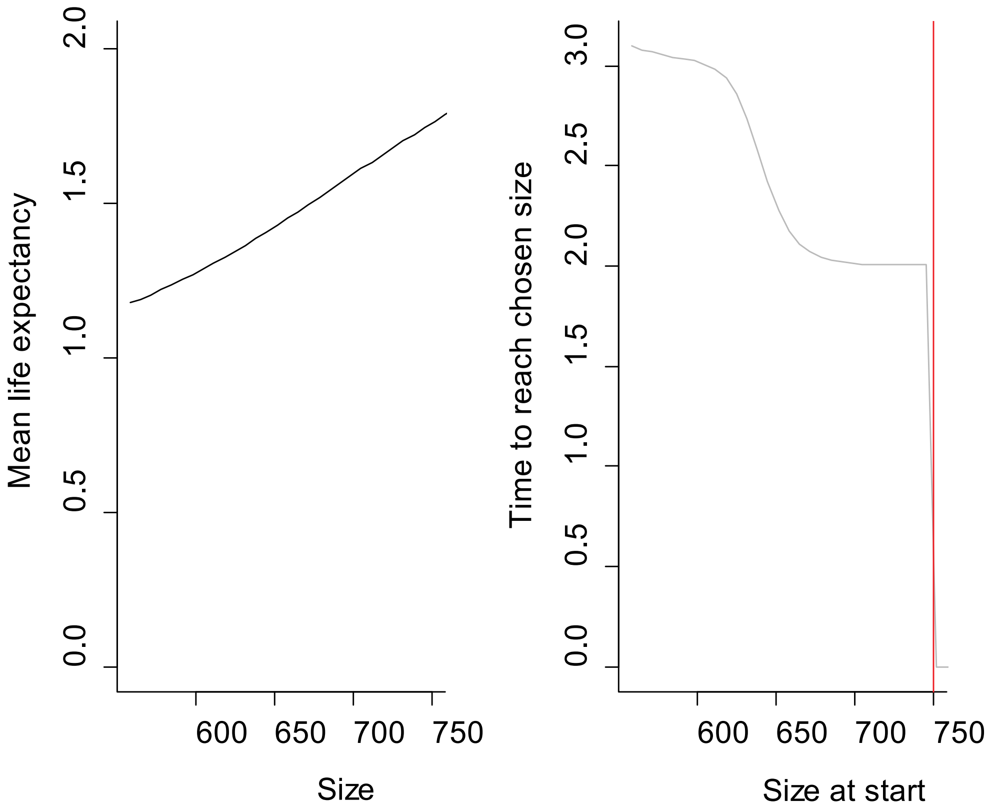

Figure G-7. Estimated mean life expectancy and passage time to reach a target size of 750 mm as a function of size in year *t*.

steelhead take 2 years to grow to a target size of 750 mm from a length of 650 and 3 years if smaller than that. Growth at sea between successive spawning events has a clear influence on future survival and fertility.

An IPM constructed from the data, assuming a target size of 750 mm and using a logistic model of survival and a linear model of growth with no covariates except for size, produced the following results for survival, growth, life expectancy, and passage time as functions of size (Figure G-8). Given the estimates of survival and growth, the IPM predicts that mean life expectancy increases from 1.0 ±0.5 year for adults less than 550 mm to nearly 2.0 ±1.0 year for fish greater than 850 mm long. The IPM predicts passage times to 750 mm of 2 years for adults greater than 600 mm long and up to 4 years for adults smaller than 450 mm. The ***P*** kernel at lower left indicates the probabilities of a survival-growth transition as a function of sizes at successive censuses (year *t* on the x-axis and year *t* + 1 on the y-axis). The ***F*** kernel at lower right indicates the probabilities of a reproduction transition as a function of those sizes. The probabilities of individual survival and growth transitions are relatively closely associated with size, but they are high only for very large fish (ca. 850 mm long). The probabilities of reproduction for surviving adults (production of adult offspring) are not closely linked to size and indeed can be high for fish of various sizes, but these probabilities tend to be lower for fish that grow less between spawning events.

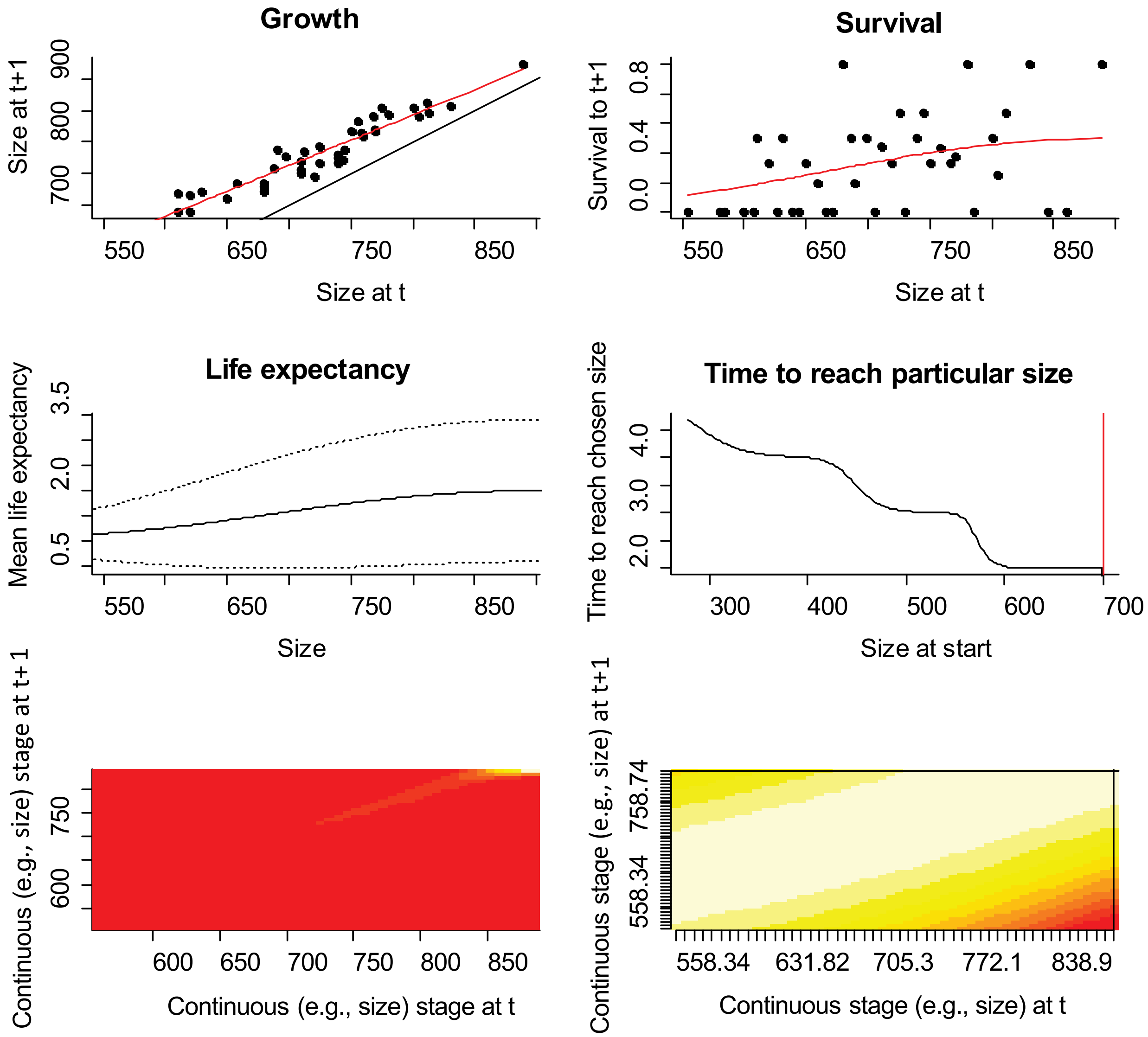

Figure G-8. Summary of some key results from IPM incorporating a simple logistic function describing survival and a simple polynomial function describing growth, with no covariates. The top four plots indicate predicted patterns of growth, survival, life expectancy, and passage time as functions of size. Fitted functions are given by the red curves for the first two plots. Ninety-five percent CIs for life expectancy bracket the predicted values along the solid curve. The bottom left plot illustrates the probability of a survival-growth transition (on the z-axis, color contours) and sizes at two successive censuses (year *t* on the x-axis and year *t* + 1 on the y-axis). The bottom right plot illustrates the probability of a reproductive transition (on the z-axis, color contours) and sizes at two successive censuses (year *t* on the x-axis and year *t* + 1 on the y-axis). In both of these plots, low values of transition probability are represented by red regions and high values by yellow regions. The predicted transition for individual survival and growth is more closely linked to size; that for reproduction (production of adult offspring) is not closely linked to size.

Combining the ***P*** and ***F*** kernels to construct a full IPM that integrates growth, survival, and fecundity and is fitted iteratively to the 1997–2008 data estimates the population's net reproductive rate per generation, R_0, at 1.80. This represents the mean number of adult females produced by a female over her lifetime. With a mean generation time of 6 years, this estimate of R_0 corresponds to an estimate of the per capita growth rate or finite rate of increase, λ, of 1.10, and an estimate of the instantaneous rate of change, r, of 9.8% increase per year. The population is on average showing positive population growth over the 1997–2008 period. The overall weak declining trend of the population between 1986 and 2011 (Figure G-1) is driven largely by a population decline between the 1980s and the late 1990s.

Figure G-9 summarizes some projections of the IPM described in Figure G-8. The results indicate that population growth rate is sensitive to both growth and survival, but depends heavily on the reproductive success of larger adults, especially those larger than 750 mm. Sensitivity analysis measures how changes in population growth rate are affected by small changes in survival-growth and reproduction transitions. Proportional changes in population growth rate relative to proportional changes in these transitions are examined through elasticity analysis, which standardizes sensitivities to minimize bias due to trait-scale effects. Figure G-10 supports the pattern of Figure G-9, showing how sensitive population growth rate is to the nonlinear component of the relationship between survival and growth.

Taken together, the results of this analysis quantify an expected pattern: that survival, growth, and lifetime reproductive success of relatively large breeding adults are critical to higher productivity of this wild steelhead population. It appears that sufficiently high interannual survival of steelhead in the marine environment is essential to high productivity, implying the importance of iteroparity. It is also evident that rapid marine growth of adults can have substantial benefits for the survival as well as the fecundity of first-time spawners and repeat spawners.

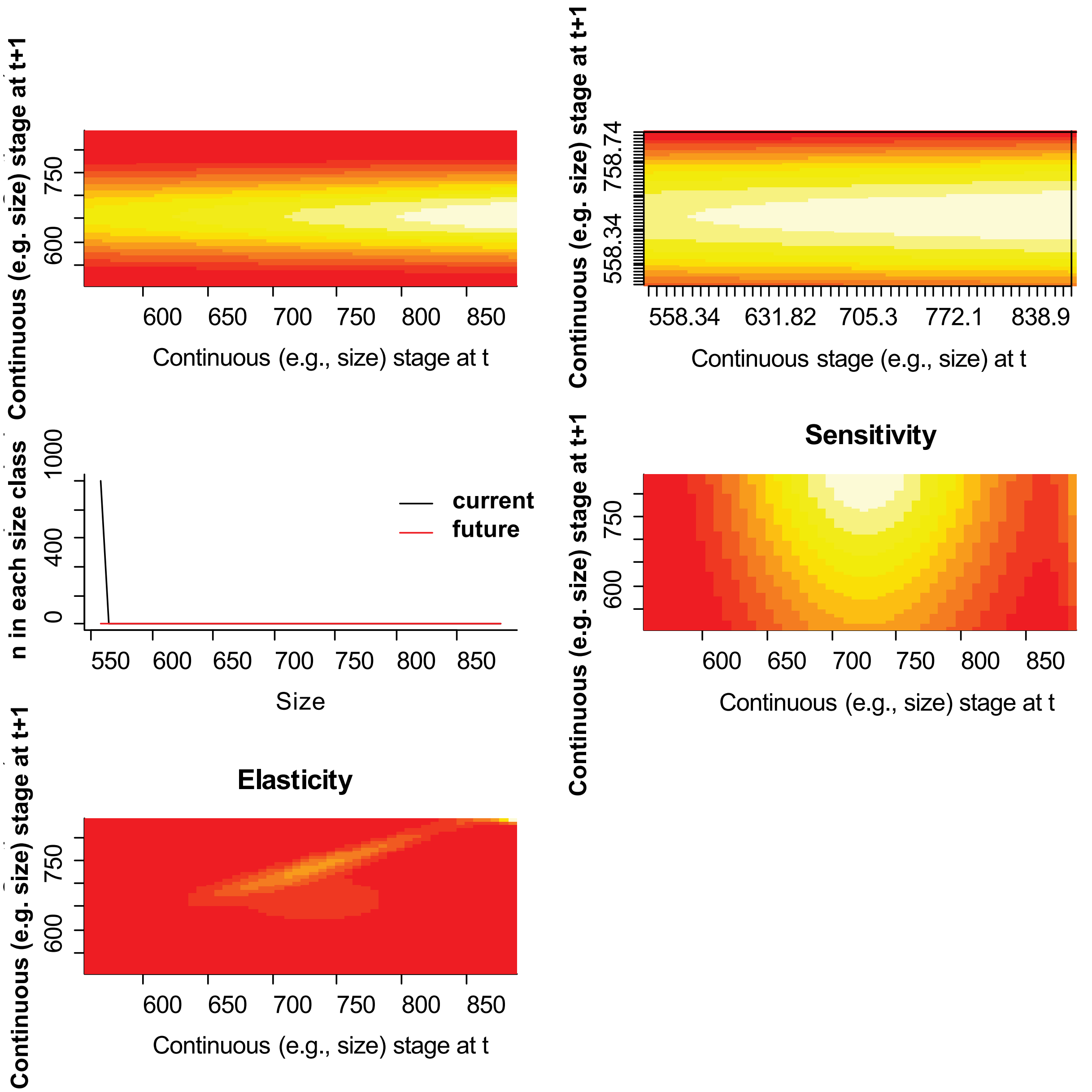

Figure G-9. A few projections from the IPM described in Figure G-8. The top two plots show the projected distributions of individual survival and growth (left) and fecundity (right) in the population over 10 years against the mesh points of size in the ***P*** kernel. The middle left plot projects that a size structure dominated by smaller (≈550 mm) adults will produce few larger adults in the future. The sensitivity plot (middle right) illustrates the predicted relationship between population growth rate (on the z-axis, color contours) and sizes at two successive censuses (year *t* on the x-axis and year *t* + 1 on the y-axis), showing that it is most sensitive to the reproductive success of adults initially between about 700 and 750 mm and subsequently greater than 750 mm, suggesting that population growth rate is sensitive to annual growth of breeders as well as iteroparity. The corresponding elasticity plot (lower right) suggests that population growth rate is most influenced by the reproductive success of adults between 700 and 750 mm. In these two latter plots, low sensitivities and elasticities are represented by red regions and high values by yellow regions.

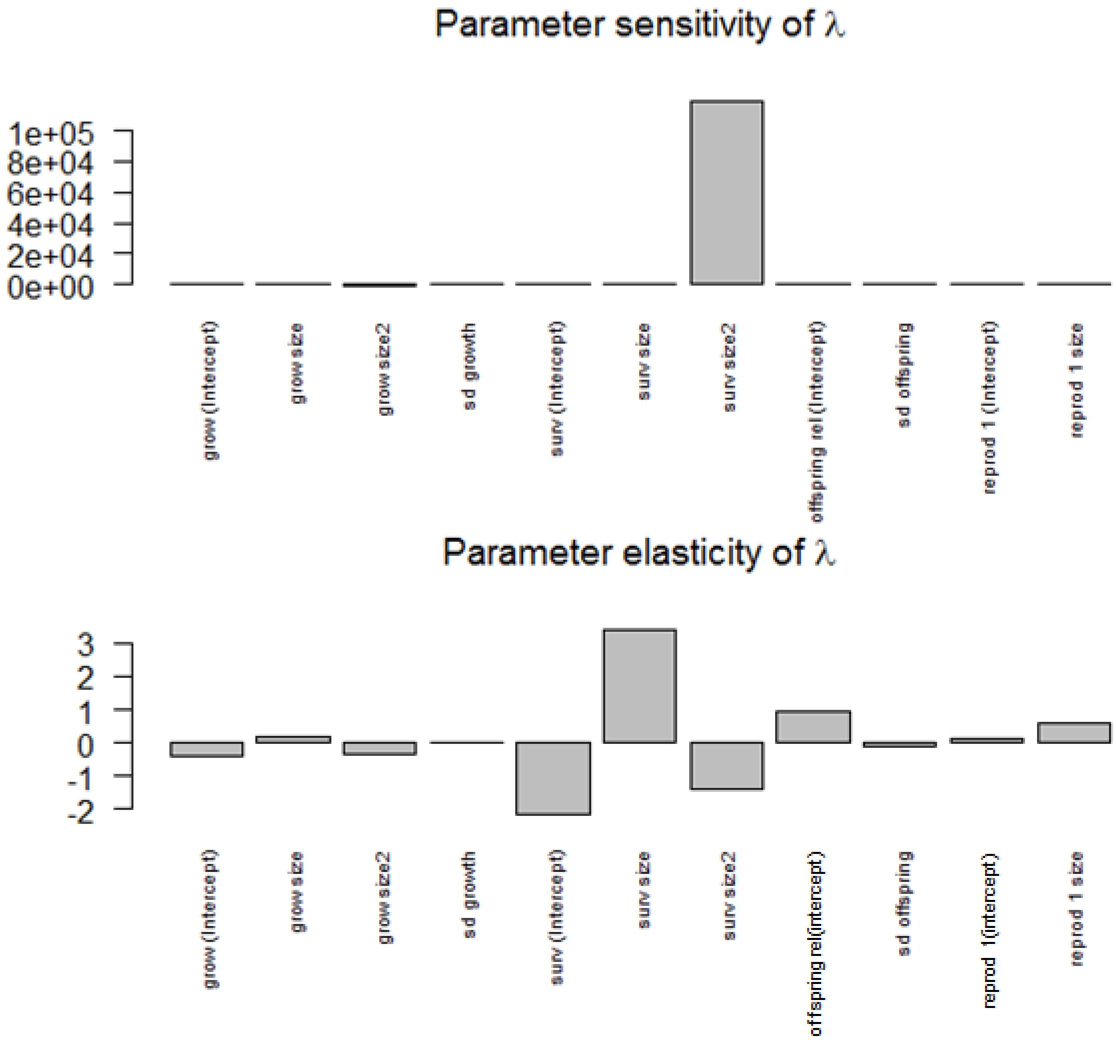

Figure G-10. Graphical summary of the sensitivity and elasticity analyses of the estimate of finite rate of population increase (λ). The sensitivity plot indicates the large effect that the relationship between survival and the nonlinear growth component has on λ, suggesting that the survival of fast-growing fish is key to higher productivity. The corresponding elasticity plot indicates that the most influential factors for high productivity are the linear component of the relationship between survival and growth and the average production of adult offspring. The factors that tend to depress population productivity are low survival between years and the nonlinear component of the relationship between survival and growth that indicates disproportionately higher survival of larger adults, implying that poor marine survival and limited annual growth in some years could threaten population viability.

Appendix H: Responses to Reviewer Comments

In September 2013, we asked several scientists with expertise in steelhead (*Oncorhynchus mykiss*) biology and viability analysis to review the draft of this Viability Criteria for Steelhead within the Puget Sound Distinct Population Segment document that was completed in July 2013. We received comments from three peer reviewers and respond to them in this appendix. Where appropriate, we have incorporated these responses into this technical memorandum, representing the final review. Here we have summarized and organized the reviewers' comments into categories relevant to issues raised by the Puget Sound Steelhead Technical Recovery Team (PSS TRT). The reviewers are not specifically identified in order to preserve their anonymity. We also include and respond to comments provided by the Washington Department of Fish and Wildlife (WDFW) and Washington tribal comanagers.

In general, the reviewers supported the conclusions of the TRT. However, some of the reviewers questioned aspects of the analytical methods the TRT employed to assess the status of populations within the Puget Sound Steelhead Distinct Population Segment (DPS), and provided additional information on specific steelhead populations that will be useful for recovery planning.

Substantive Scientific Comments

Development of Viability Criteria

Review

Several reviewers had concerns about the approach the TRT used to assess viability of Puget Sound steelhead, which focused on the lack of empirical support for parameters influencing VSP viability. These comments are summarized below.

In general, while we agree with the initial approach taken here to quantify viability (e.g., decision support system–Bayesian network analyses of viability) for each demographically independent population (DIP) within the Puget Sound DPS, we have significant concerns with the lack of empirical data support for many of the parameters listed under each viable salmonid population (VSP) criterion used in the final determination of viability. Specifically, while some stocks within Puget Sound have sufficient time-series data to assess contributions of abundance and productivity to overall viability, empirical data on parameters within the diversity and spatial structure are lacking for most if not all DIPs. For parameters within specific VSP criteria where the TRT has used a combination of limited data and expert knowledge to assign values, we recommend that these values be excluded from the current assessment of viability and remain as place holders until they can be informed by sufficient empirical data. However, we do not dispute the potential importance many of these data-limited parameters to the continued viability of Puget Sound steelhead.

While results from this study provide a reference point for technical and policy discussion regarding setting interim abundance and productivity goals for each TRT-proposed DIP, much of the data utilized in the assessment of each DIP is not supported by sound empirical studies, but rather is a combination of temporally sparse information and expert opinion from the TRT panel. Specifically, the TRT states that of the four VSP parameters (abundance, productivity, diversity, and spatial structure), abundance and productivity are the most supported by empirical data, whereas diversity and spatial structure have very little empirical support. Even so, sufficient time-series data to effectively assess temporal trends in abundance and productivity for many of the DIPs are lacking (page 32, lines 1182–1187 of the review draft). Furthermore, the authors specifically state they identified and used as placeholders parameters to quantify viability for each VSP category for which currently little is available is, but anticipate more may become available later through future studies (page 108, lines 1608–1613 of the draft). While we do not dispute that all four VSP criteria (and the many parameters used to assess viability for each category) are important to the future viability of the Puget Sound DPS, we have significant concerns about the use of parameters in the viability analyses for which little empirical data are available. We ask that these potentially important parameters for which empirical support is lacking be excluded from current viability results for specific DIPS, but still utilize parameters as place holders within the conceptual model framework for future inclusion in the viability analyses as they are informed by new studies.

The primary recommendation is that parameters for which almost quantitative information was available should be excluded from the viability assessment, and from the Bayesian network models for viability. In most cases, these parameters would especially include those associated with diversity and spatial structure.

Response

The TRT discussed this issue at length and took two actions to address it in the revision of this document. First, the TRT unanimously felt that it would be inappropriate to exclude these parameters from a decision-theoretic model like the Bayesian network (BN). The TRT’s charge was to evaluate viability of DIPs, major population groups (MPGs), and the DPS for all four VSP criteria—abundance, productivity, diversity, and spatial structure—with the best information available. We therefore included metrics for these criteria for three reasons (see page 121 of the draft):

- Eliminating such factors from the network when they are thought to be important to an overall assessment defeats the purpose of creating an informative network;
- Bayesian networks are designed to accommodate both factors that can be quantified and those that are evaluated based on collective judgement in the absence of more definitive information, which can be updated when more information becomes available; and
- Eliminating such factors tends to place higher importance on factors with available information, regardless of whether those factors are more critical to viability or not.

The TRT also investigated some scenarios that did what the reviewers asked for (e.g., page 149 et seq.) to evaluate this issue further. In general, these scenarios resulted in substantial improvements in viability at each level. Probabilities of viability for DIPs throughout the DPS varied from 64.9% to 86.8%. Probabilities of viability for the three MPGs were 89.9% for the

Northern Cascades MPG, 74.6% for the Central and South Puget Sound MPG, and 70.5% for the Hood Canal and Strait of Juan de Fuca MPG. Probability of viability for the entire DPS was 42.3%; the most probable explanation was that the DPS is viable, with a probability that the DPS is not viable equal to 56.2%.

This exercise illustrates how the structure of a BN can strongly influence the inference about viability at nodes throughout the network. If we rely on the more quantifiable and readily available of the VSP criteria alone, estimates of steelhead viability are higher throughout the DPS for two main reasons: the TRT's largely qualitative assessment of diversity and spatial distribution of steelhead is that the widespread state of these factors in Puget Sound steelhead poses risk to sustainability, and the TRT's more quantitative assessment of demography (abundance and productivity) is conservative with respect its influence on risk to persistence. The primary problem with this analysis is that it is at odds with the threatened status of the DPS and with concerns about continued risks to persistence and sustainability of many of its populations. It also indicates that the existing BNs may be too benign with respect to risk of quasi-extinction; the viabilities for several DIPs and all the MPGs are at odds with the outcomes of several of the population viability analyses (PVAs), in part because of the MPG-wide averaging for DIPs with no quantitative information. It is also probably due in part to the fact that this analysis does not consider the effects of factors that are known to be risks to wild Puget Sound steelhead, including limited availability of productive rearing and spawning habitat, the widespread use of Chambers Creek and Skamania River hatchery fish throughout the DPS, and perceived changes in aspects of diversity for several populations.

The TRT considers the networks that encompass all four VSP criteria to more accurately reflect the current status of Puget Sound steelhead. The vast majority of steelhead populations throughout Puget Sound do not appear to be viable, regardless of where one looks; most are declining steeply from historical levels and many are very small. Despite the lack of demographic information for many of the populations and the lack of clarity regarding the precise states of diversity and spatial structure for nearly all of them, the TRT considers these networks to represent a reasonable starting point for identifying the major criteria for effective recovery of Puget Sound steelhead, and believes that they help to identify key gaps in knowledge that must be addressed if further declines in steelhead viability are to be arrested. The TRT therefore did not consider these alternative scenarios further in developing the viability criteria.

Review

Several reviewers provided comments on individual parameters within each VSP criterion regarding whether the parameter should be excluded from the current viability analyses due to lack of empirical data or whether we think that these parameters can be informed by studies currently being conducted in the Skagit Basin that are either at or near completion. They went on to say: It would be very helpful if the TRT outlined the data completeness and gaps, caveats, and assumptions for each individual DIP included in the viability analysis.

Comments on productivity were as follows.

- Smolts per spawner: This information is available for many DIPs and is estimated annually using redd counts and smolt traps.

- Adults per smolt: Can be estimated using estimates of annual escapement and outmigrant smolt abundance.
- Mean population growth rate: Informed from the recent status of smolts per spawner and adults per smolt parameters.
- Frequency of repeat spawners: We agree that this parameter would have a direct influence on productivity by increasing recruitment. However, this parameter could also fit under the diversity node. Because the frequency of repeat spawners represents a relatively minor contribution to the overall spawning stock (see Figure 71 of the draft), we would view this life history characteristic more as an index of diversity vs. productivity. Additionally, how did the TRT make the comparison between historical vs. current? Based on the inclusion of this parameter under productivity, it appears the authors are using frequency of repeat spawners as an index of productivity for each DPS. Given how little information is currently available regarding the proper assessment of the importance of this parameter to overall viability, we suggest that this parameter be used as a placeholder and not be assigned a probability for inclusion in the viability analyses for each DIP at this point in time. To properly assess the contribution of repeat spawners to overall smolt production (which would then really be a productivity parameter rather than diversity parameter), a genetic-based parentage study would need to be conducted.

  Comments on abundance were as follows.

- Probability to reach quasi-extinction threshold: Estimates from multivariate autoregressive state-space (MARSS) modeling of viability in previous section of study.
- Adult abundance vs. capacity: GIS-based assessment of abundance and habitat availability.
- Juvenile abundance vs. capacity: GIS-based assessment of abundance and habitat availability.

For any estimate of capacity, we have enough historical information from the Skagit to quantitatively assess current capacity by assessing spawner/recruit curves. Specifically, we should not rely on potentially out-of-date information from Gibbons et al. (1985) for estimates of optimal densities and for specific values of marine survival or coarse-scaled measurements of habitat availability (e.g., channel width and gradient) to understand adult or juvenile abundance vs. capacity. Rather, in the Skagit River system, we have an abundance of much finer scale habitat data that enables us to quantify habitat availability at the channel unit scale. In addition, the Skagit Yearling Salmonid Study is fairly close to being finished and will likely help inform current estimates of rearing habitat use vs. rearing capacity.

Comments on diversity were as follows.

- Hatchery influence: Poorly understood. For the Skagit, the Saltonstall-Kennedy study will provide Skagit-specific results for the degree of hatchery genetic influence on the wild/natural populations. We suggest using these results rather than using an assumed value of hatchery influence based on categories of smolt release size within the watershed of specific DIPs. Please mark as a placeholder until we can support this parameter with available data.

- Fraction of smolts from residents: Unless some new genetic studies have taken place within each DIP that we are unaware of, this information is contrived or produced from the literature. Either way, there is likely considerable variation in this parameter between DIPs that should be accurately accounted for. Given that our current understanding of this cannot accurately account for the likely significant variation in this parameter between DIPs, this should only be included as a place holder and not incorporated into the viability analysis until more information comes available.
- Run timing: Comparison between current and historical run timing is likely not possible for many DIPs. Therefore, please provide a clear explanation for how this comparison was made for individual DIPs. Otherwise, we need to come up with a more sound method to better assess this parameter as it relates to viability.
- Spawner age structure: Comparison between current age structure and historical is not possible for many DIPs. Similar to run timing, please provide a clear explanation for how this comparison was made for individual DIPs.

For parameters in which there is a probability split between a comparison of current vs. historical values for the specific parameter, we are unclear of the time frame that is used for this comparison. In fact, it would be good for the authors to include a discussion regarding what (if any) historical data were used in the historical vs. current status assessments for each of these parameters.

Comments on spatial structure were as follows.

- Fraction of intrinsic potential rearing habitat occupied by juveniles.
- Fraction of intrinsic potential spawning habitat occupied by adults.

These parameters do not appear to satisfy the spatial structure criteria. These help address the carrying capacity question for DIPs but do not help in the assessment of spatial structure. How are these different than those parameters included in the abundance category (e.g., adult abundance vs. capacity)? Another way to ask this question is to first ask what is the current diversity of habitat available to juveniles and adult steelhead within each basin (e.g., temperature/gradient combination)? Second, what percentage of each habitat type is currently occupied by juveniles and adults? Finally, we cannot assume that the proportion of available habitat occupied varies directly with abundance.

These appear to be quantified using GIS. This method can only provide ballpark estimates of these parameters. Additionally, viability was estimated based on the relationship between smolt abundance, densities (fish/km), and marine survival rates quantified for each DIP in Gibbons et al. (1985). We suspect that these relationships are entirely variable through time, based on changes in rearing and spawning habitat availability. Again, juvenile occupancy can be informed by the nearly completed Skagit Yearling Salmonid Study.

Response

See the TRT's previous response. Again, the TRT unanimously felt that it would be inappropriate to exclude these parameters from a decision-theoretic model like the BN, because it defeats the purpose of such a network, because these models can easily and transparently accommodate new or additional information, and because eliminating parameters tends to put greater weight on factors with available information, regardless of whether those factors are more critical to viability or not.

As stated earlier, the TRT investigated scenarios that did what the reviewers asked for to evaluate this issue further. In general, these scenarios resulted in improvements in viability at each level, which illustrated how the structure of a BN can strongly influence the inference about viability at nodes throughout the network. Under these alternative scenarios, estimates of steelhead viability are higher throughout the DPS. The primary problem with this analysis is that it is at odds with the threatened status of the DPS and with concerns about continued risks to persistence and sustainability of many of its populations. It also indicates that the existing BNs may be too benign with respect to risk of quasi-extinction. Therefore, the TRT considers the networks that encompass all four VSP criteria to more accurately reflect the current status of Puget Sound steelhead.

In the revised report, we responded directly to the reviewer's comments about the spatial structure criteria; as described in the main text, we developed a spatially explicit method of determining spawning steelhead presence/absence from stream reaches suitable for the species (based on the TRT's intrinsic potential criteria) and developed spatial structure scores from these data to make them more independent from the abundance criteria. We applied these scores to the rearing criterion due to the lack of quantitative juvenile data.

Finally, we note that the studies the reviewer referred to are incomplete and the TRT did not have access to those data to incorporate in this report.

Review

Reviewers had concerns about the use of quantitative thresholds to help prioritize populations (e.g., 15% of viability abundance or growth rates of >1), because they may not result in DIPs that meet these criteria across MPGs. They suggested a qualitative approach to characterize DIP viability criteria (e.g., increasing or stable growth rate, highest population growth rate over a given time period).

Response

The TRT adopted quantitative viability criteria for DIP abundance and productivity that could be 1) used to conduct PVAs at the DIP level and 2) discretized for use in the BN models for DIP viability. The TRT felt that this would more accurately reflect the patterns of viability across distinct MPGs and would at least partially address the reviewers' concerns without essentially rescaling relative viabilities to a more ambiguous scale.

Methods of Population Viability Analysis

Review

A WDFW reviewer commented that the TRT should have used Dennis et al. (1991) methods for its PVAs instead of the "slope" methods based on Holmes (2001) and Holmes and Fagan (2002) to evaluate viability of individual steelhead populations. Detailed comments on this were provided in a memorandum from WDFW, which contained a tabular comparison of the Dennis et al. (1991) and Holmes (2001) methods. The primary thrust of these comments was: Based on analysis conducted by WDFW biometricians, variances and subsequent width of confidence intervals of median term rates of population change are underestimated, giving a false sense of precision in estimates of lambda (population growth), and the amount of information contained in the data.

Response

After review of the comments and consulting Dr. Holmes, the TRT elected to retain the multivariate autoregressive state-space analysis methods (Holmes and Ward 2011) for PVAs of individual steelhead populations in the DPS. (These methods were also used to evaluate trends in abundance and productivity by the National Marine Fisheries Service in the recent status review updates of all listed West Coast salmonids [Good et al. 2005, Ford 2011].) The TRT made this decision for two reasons.

First, there seems to be some confusion over what the slope method does; the reviewer subscribes to the notion that this method underestimates the combined process and observation variance. The point of the slope method is that it tries to estimate the process variance in a way that it can forecast extinction risk using a model of population dynamics that is free of observation variance. In doing so it accounts for observation variance, something that the Dennis et al. (1991) method does not do. The slope method is an extension of Dennis et al. (1991) that accounts for both process and nonprocess error. In fact, the Dennis et al. (1991) method actually tends to overestimate the process variance. There appears to be considerable controversy over the value of the variance estimates obtained by the Dennis et al. (1991) method for estimating extinction risk. Dr. Holmes provided a list of several recent publications that point toward a statistical consensus to avoid the Dennis et al. (1991) method for this purpose.

Second, the TRT incorporated these PVA results provided by MARSS modeling for individual populations and some population groups, after discretizing them into broad classes of abundance and productivity as they affect viability, into a hierarchical BN model to evaluate viability at the levels of a) DIP), b) major population groups (MPGs, or “strata”), and c) the DPS (the listed unit) as a whole. This model is described in detail in the main body of this technical memorandum. As a consequence, most of any specific differences between the methods would have little bearing on these broader results. The WDFW reviewers did not comment on these approaches or results.

The TRT amended the text in the description of the PVA methods in the body of the report to reflect these changes.

Review

A reviewer commented that the use of lambda can be problematic in assessing productivity for steelhead because of their extended freshwater life history and expression of population dynamics that are so strongly density dependent.

Response

The TRT agrees with this concern, so it also used adult recruits per spawner as a metric in assessing productivity in steelhead populations. However, alternative metrics like adult recruits per spawner can be more sensitive to changes in life history that might be occurring during the longer time series, such as declining age at maturation as a result of increasing abundance of hatchery fish on spawning grounds, and therefore lambda was used as the primary metric. It is a conventional metric that can be applied to a wide variety of taxa with distinct life histories, and is appropriate when applying state-space models to time series of count data. Since the TRT discretized the lambda metric in applying its BN model approach to viability at the DIP scale, the TRT considers the use of this altered metric is suitable for this purpose.

Intrinsic Potential Model

Review

A WDFW reviewer proposed that the TRT did not apply results from the Gibbons et al. (1985) report correctly to estimate intrinsic potential (IP) for steelhead. The reviewer's primary concern was the TRT's assignment of moderate to high values for IP to very low-gradient reaches, especially for reaches less than 20 m in width, and low values to higher gradient reaches regardless of width. The reviewer's comments, with minor editorial changes, are presented below.

The Gibbons et al. (1985) work suggests that lumping 0–4% large river and tributary "mainstem" gradients together is not appropriate. Very low-gradient, large main stems have very low production potential. Gradients in the 0.25–3.0 range have greatly increased densities. In general, Gibbons et al. (1985) had much higher tributary parr densities compared to mainstem parr densities. If the TRT evaluated results from other parr density studies and choose parameters for Table C-1 based on "averaging" among studies, including Gibbons et al. (1985), that should be clearly described.

The Table C-1 matrix as it stands does not produce reliable numbers. Personal observations in some of the low-gradient streams (i.e., Nookachamps Creek, Drayton Harbor streams) have production potentials that are unrealistically high. On the flip side, some high-gradient streams (i.e., Deer Creek, Nooksack River) are way too low. Even with current poor ocean survival rates and degraded habitat, both are probably producing more steelhead than the model would predict.

The reviewer suggested an alternative to Table C-1 that might better reflect the Gibbons et al. (1985) parr density estimates, giving higher IP ratings to higher gradient stream habitats and lower IP ratings to lower gradient stream habitats.

Stream habitat rating matrix (below natural barriers)

	Stream bankfull width		
Stream gradient	0–3 m	3–20 m	>20 m
0.0–1.0%	High	Moderate	Low
>1.0%	Moderate	High	Moderate

Response

The information on density of *O. mykiss* parr contained in the Gibbons et al. (1985) report was only part of the information the TRT used to develop its IP model. For example, the TRT states in Appendix C that this matrix was developed from GIS data used to develop an IP model that was adapted from data in Gibbons et al. (1985) and from the Interior Columbia TRT's steelhead IP model; it incorporates information from both works. In addition, the TRT thought that some of the concerns expressed by the reviewer would not necessarily apply generally to steelhead-supporting systems throughout the Puget Sound DPS. After consideration of these comments, the TRT modified the habitat rating matrix it used in Table C-1 to reflect this input as follows.

Stream habitat rating matrix (below natural barriers)

	Stream bankfull width		
Stream gradient	0–3 m	3–20 m	>20 m
0.00–0.25%	High	Moderate	Low
0.25–4.00%	Moderate	High	Moderate
>4.00%	Low	Low	Low

This change largely implements the suggested changes for lower stream gradient reaches, but retains low IP for very high-gradient reaches. In part this reflects, in addition to Gibbons et al. (1985) and the above comments, the historical evidence the TRT examined for widespread production of steelhead in small, low-gradient streams, and the lack of evidence for production in very high-gradient reaches, where other species such as bull trout (*Salvelinus confluentus*) might be more common.

Review

A reviewer commented that it is critically important to account for naturally spawning hatchery fish to evaluate their impact on natural productivity. The reviewer was especially concerned that the simple use of redd counts does not directly lead to this accounting and could seriously affect estimates of productivity and, therefore, viability. This issue is particularly important because of the widespread reliance in the past on hatchery steelhead production in Puget Sound, employing broodstocks that are either highly selected for particular traits (Chambers Creek winter-run fish) or sourced from out-of-basin (Skamania River summer-run fish). Not only are some of the redds in the data considered here undoubtedly built by feral hatchery fish, but more importantly, the progeny of early spawning hatchery fish that survive to interact ecologically with progeny of later spawning wild fish are likely to have impacts on productivity over time.

Response

The TRT agrees with the reviewer about the need to account for naturally spawning hatchery fish to accurately assess the effect of feral hatchery fish on natural productivity, and agrees that the use of redd count data alone is not likely to fully account for this effect. The TRT is not sure how to deal with this issue using the available data. The ongoing studies of hatchery introgression in the Skagit River by WDFW should provide some insight into the magnitude of this problem. Given the widespread published evidence for reduced fitness of naturally spawning steelhead in the presence of feral hatchery fish, this is of considerable concern. However, the TRT considers the estimates of productivity to be conservative with respect to this factor. Moreover, a strong majority of the TRT thought that the metric it employed to characterize the influence of feral hatchery fish on viability (through reduction in diversity) was a reasonable proxy for this effect, using a straightforward "propagule pressure" approach. The TRT thought it was a rather conservative approach, in part because it did not influence productivity directly. An altered BN model that attempted to capture this influence would undoubtedly have produced an even grimmer assessment of steelhead population viability.

Minor Scientific Comments

Review

A reviewer commented that the fact that each MPG is weighted equally in assessing the viability of the DPS in total may bias the results.

Response

We explain why each MPG must be weighted equally in assessing DPS viability at several points in the document. The reason for this is that each MPG is considered an essential component of the DPS in all regards of VSP criteria such that the loss or sufficient degradation of any MPG means that the DPS no longer retains its essential biological characteristics and therefore cannot remain viable as a unit.

Review

A reviewer indicated that the TRT identified required conditions for resident life history to contribute to anadromous life history viability. These conditions need to state whether this information is or can even be quantified.

Response

In the Resident Fish as a Factor Influencing Viability subsection of the Diversity Considerations for Steelhead Viability section of this technical memorandum, the TRT makes it clear that this is an unresolved issue for most *O. mykiss* populations. This issue is an active area of research that merits immediate attention to help guide development of conservation strategies for steelhead. Therefore, this information is not quantified. However, it could be with a wild pedigree of known residents and anadromous fish and an appropriate demographic analysis of the anadromous population (e.g., with an integral projection model approach, as described in Appendix G).

Review

A reviewer indicated concern that the dependence of resilience of small populations on specific levels of repeat spawning can only be demonstrated under a no-harvest scenario, regardless of environmental variation. Furthermore, a comprehensive genetic study on parentage would be required to correctly answer this question and therefore would provide empirical evidence to support such broad conclusions.

Response

We amended the text in this section on iteroparity to reflect this concern.

Review

A reviewer asked how the TRT came up with these threshold numbers to assign whether a DIP is viable or not.

Response

The TRT developed these criteria collectively after considering the results of the individual PVAs, the BN model outputs, information from the published literature, and discussion of alternative values. In short, the criteria reflect a mixture of quantitative and qualitative information and collective judgment by TRT members. These criteria could obviously be altered with new or improved information to reassess viability under different scenarios.

Review

A reviewer commented that the TRT says it used comanager-provided data on abundance/productivity, fraction of hatchery-origin spawners, harvest, age structure, and hatchery releases in assessment of viability of individual DIPs. The reviewer asked to ensure that current/correct data were used (especially for Skagit) in these analyses.

Response

The TRT made repeated requests to comanagers for this information and was assured it received the most current and correct data through 2011 for this document, including that reflected in Salmonid Stock Inventory (SaSI) and Salmonscape databases. This information was updated frequently during the drafting of this report. It should be noted that at least two TRT members served as liaisons with state and tribal comanagers (their employers).

Review

A reviewer noted that the highest observed growth rates over entire series were in the Northern Cascades MPG, whereas the lowest were estimated in the Central and South Puget Sound MPG. The reviewer also noted that this is not a surprising result here, given variation in habitat and hatchery influence throughout the DPS.

Response

No response is required.

Review

A reviewer commented that the authors state many times throughout the document that accurately quantifying viability of many DIPs is a challenging endeavor due to the lack of quantitative abundance and productivity data for many of the DIPs. Given the nature, size, and complexity of this document, the reviewer recommended that all assumptions, caveats/uncertainties, data limitations, etc., be stated in an individual section at the front of the document. While acknowledging that the TRT does a good job of this throughout the document, the reviewer thought that a short summary of this for each DIP would be helpful as an appendix.

Response

The TRT reasoned that this point is, as the reviewer noted, emphasized at several points in the document. In combination with the information it provided in the main text as well as in Appendix C, the TRT decided that the document's information was sufficient to indicate where the uncertainties and limitations of the data at the DIP level are, and that an additional appendix was unnecessary.

Review

A reviewer acknowledged that the TRT did not account for nonanadromous recruits in their stochastic PVAs, but given that we have very little understanding of nonanadromous contributions to annual smolt production, the reviewer emphasized that these contributions must be quantified before its influence on viability can be assessed.

Response

The TRT does not disagree with this as an important step in quantitatively evaluating viability of an entire metapopulation, but argues in the document that, at a minimum, such nonanadromous recruits can serve to buffer steep declines in anadromous abundance during periods of low abundance and reduced marine survival.

Review

A reviewer commented that the language reflecting the use of index counts of natural escapement and not accounting for resident recruits in assessing productivity and extinction risk is contradictory to the statement about the potential overestimation of extinction risk from the exclusion of resident individuals to recruitment.

Response

The TRT was unclear about the meaning or intent of this comment and therefore made no changes to that section of the document.

Review

A reviewer indicated that the TRT utilizes SaSI for information on stock abundance and productivity. There needs to be a comparison between the SaSI tables and current comanager data to assess for discrepancies in data sources.

Response

See the response above about the information the TRT received from comanagers in this report. No further response is required.

Review

A reviewer indicated that the authors state that PVAs estimated by MARSS modeling do not account for the effects of density dependence on stock abundance and productivity. The reviewer noted that is a problem, given that we know that many populations for which the data are available are limited by density dependent effects. This assumes that viability is primarily related to abundance and not the ability of natal habitat to support adult spawning and juvenile rearing capacities required for persistence into the future.

Response

The TRT is aware of this limitation of the analyses, but there is not currently a way around it with this set of state-space models, which are among the most robust available to assess quasi-extinction risk from time-series data, such as that available. The TRT also wishes to point out that another likely influence of density dependence in many of these populations is probably depensatory at very low abundance (e.g., Allee effects), which could tend to depress the estimates of viability over those given in this report. Whether this depensation is more influential on the productivity of these depressed wild populations than the ecological effects of feral hatchery fish remains an open question.

Review

A reviewer noted that the assumption for the PVAs that population trends are linear and environmental conditions affecting mortality (e.g., marine and freshwater survival) and production are constant is not valid. The reviewer also noted that this is clearly not true for many DIPs within the Puget Sound DPS.

Response

The method does not assume that population trends are linear but, as the reviewer indicates, the assumption of stationarity is a property of the method.

Review

A reviewer noted that because a knowledge-based decision support system attempting to evaluate ecological conditions may lack precision due to data gaps and imperfect knowledge, it makes a clear statement of uncertainty due to the lack of data.

Response

The TRT concurs with this assessment, but thinks that this uncertainty is adequately captured in the decision support system (in this case, the BN model). Indeed, the calculations of most probable explanation and the use of the sensitivity analyses and confusion matrices provide for each model a quantitative estimate of this uncertainty over the entire framework.

Review

A reviewer noted that the viability outcome appears to be highly sensitive to how the parent nodes (e.g., VSP criteria) are parameterized. The main concern is that quality data representing significant time series for individual DIPs have significant limitations for quantifying the VSP criteria used in the decision support system viability analysis.

Response

The TRT understands the reviewer's point, but does not agree that the viability outcome is highly sensitive to how the parent nodes are parameterized. In most cases, the parameters for all VSP criteria would have to be shifted to substantially higher values or entire criteria would have to be removed for a DIP to become viable. In fact, the TRT was sensitive to this potential problem and chose fairly broad, discrete categories for many of the variables to help alleviate this concern. That said, the TRT agrees with the reviewer that data quality is a significant limitation to accurately quantifying the VSP criteria at the individual DIP level in the BN models.

Review

A reviewer commented that the TRT states that each network was constructed so as to minimize complexity without compromising the analytical ability to evaluate sensitivity of viability to the four VSP parameters; the reviewer was concerned that such oversimplification can bias the analysis, especially when these results can influence decisions that can significantly affect current conservation and management practices.

Response

The TRT agrees that finding a proper balance between model complexity and utility for informing conservation and management actions is difficult, but it is unclear to the TRT how additional model complexity without additional information on the key parameters would avoid this potential bias.

Review

A reviewer indicated that some of the nodes included in the BN models are placeholders and should be viewed as such unless new information comes available that allows the proper assessment of these parameters.

Response

This is true. In those cases, we tried to give values to the nodes that implied considerable uncertainty in influences on viability; the discretization of the parameters also helps to reduce this influence.

Review

A reviewer noted that estimates of DIP diversity incorporate influence of run timing distribution, hatchery fish on natural diversity, the adult age distribution, and the proportion of migrant smolts produced by resident adults. The reviewer argued that two of these parameters, hatchery influence and resident contribution to migrant smolts, lack empirical support from actual data. The authors need to make this very clear that data are lacking on these specific parameters and that they will be assessed more thoroughly/updated as empirical data to support estimates becomes available.

Response

The TRT is clear in the revised report about the methods used to derive these parameter values. In the case of hatchery influence, we used a systematic metric to provide a proxy for this effect, based on historical hatchery production (propagule pressure). In the case of resident contribution to migrant smolts, we agree that this lacks empirical support for Puget Sound *O. mykiss* populations (except potentially for the Cedar River). The TRT used broad categories of influence to estimate what that influence might be, and the TRT states repeatedly in the document that these data are lacking and the models should be updated as they become available.

Review

A reviewer pointed out that the TRT outlines the uncertainty in various parameters utilized in their analytical approach and how they dealt with this uncertainty in their analyses. For example, the TRT states that intermediate and output conditional probabilities pertain to demographic and life history responses that are poorly understood due to lack of empirical data. The TRT dealt with this significant uncertainty by double weighting probabilities of abundance and productivity (more well-known) compared to diversity and spatial structure (significant lack of empirical understanding), and by utilizing high levels of spread in conditional probabilities for parameters where empirical data are lacking. In addition, the reviewer felt that the contrived linear relationship between viability and VSP risk is perhaps an oversimplification of the influence of each VSP criterion on overall viability. The reviewer argued that, while this approach is valid in Bayesian methodology and for exploratory purposes, results from these analyses, especially with the inclusion of poorly understood parameters, should not influence firm "set-in-stone" policy decisions. Rather, the analyses should be meant to be adapted as new data becomes available (e.g., Skagit yearling and Saltonstall-Kennedy study).

Response

The TRT agrees with this statement in general, but is unsure what is meant by "these analyses…should not influence firm 'set-in-stone' policy decisions." Because these analyses can and should be adapted as new data become available, why shouldn't they influence policy decisions? Which policy decisions are immutable?

Review

A reviewer had a concern about the conditional probabilities used to describe risk for each VSP parameter. The TRT specifically states that probabilities for each parameter were estimated using a combination of empirical data, PVA results, and expert opinion. The conditional probabilities represent oversimplification of all the factors at play that could affect viability of individual DIPs. For example, under VSP parameter diversity, the categories for fraction smolts from residents include negligible vs. appreciable and categories for hatchery influence include nominal vs. extensive.

Response

The TRT thought that such simplification was necessary when data were insufficiently quantitative to provide greater resolution of influences on viability. Nevertheless, conditional probabilities are still required for the model and these were chosen based on that combination of information the reviewer describes.

Review

A reviewer argued that it is very unclear how the TRT estimated the influence of each parameter on viability of the Nookachamps Creek Winter-Run DIP, given that little or no historical data exist for this population.

Response

The TRT agrees with the reviewer that the Nookachamps Creek population is more problematic than some other populations to assess viability because of the lack of historical information. The TRT considered many of the parameters to likely be similar to those for other winter-run steelhead populations in low-elevation areas in the Northern Cascades MPG, such as the Samish River and Bellingham Bay Tributaries Winter Run.

Review

A reviewer commented that, in its inspection of the results of the BN analyses, the TRT implies that the results provide a complete explanation of fine-scale processes that drive population persistence. The reviewer went on to say that, "undoubtedly, increasing abundance and productivity of all stocks spread geographically across the DPS will increase the probability of persistence of the DPS. However, this does not take into account contributions of individual DIPs or entire MPGs to persistence of the DPS. I would imagine that given the high degree of variation in habitat quality between individual systems, some DIPs should have higher contributions to MPG and hence DPS persistence. However, I agree that both life history diversity and spreading risk across the entire geographic range of the DPS would increase the probability of persistence." In addition, the reviewer pointed out that the TRT's approach "demonstrates the importance of viability for each MPG to overall DPS viability…. I think this analysis is a bit coarse and does not take into account the scaled importance of individual DIPs to MPG viability and, similarly, the importance of individual MPGs to DPS viability."

Response

The TRT agrees with the reviewer's observation about variable contribution of DIPs to MPG and DPS viability, but did not consider this factor in the analyses. The TRT did not give higher weights to specific DIPs in terms of their influence on MPG and DPS viability, and this was deliberate. For DIPs of each major life history type (summer run vs. winter run), the influence of each DIP on MPG was considered to be equivalent (although each DIP had a different viability itself). As stated elsewhere in the document, DPS viability depended equally on the viability of each constituent MPG; this is considered an essential characteristic of an MPG.

Review

A reviewer noted that the conclusions depend heavily on the underlying conditional probabilities in the model and asked how they were derived.

Response

As the reviewer suspected, these probabilities were derived by a combination of data and expert opinion, and generally show simple, often linear relationships to viability. This protocol is described in detail in the report. More realistic relationships, informed by empirical data, could easily be implemented in the models.

Review

A reviewer asked which studies the TRT used to guide analysis of residence vs. anadromy.

Response

The TRT reviewed a variety of empirical studies and modeling investigations of both coastal and interior *O. mykiss* to guide its assessment of interactions between resident and anadromous forms of this species, including Zimmerman and Reeves (2000), Narum et al. (2004), Thrower et al. (2004), Pearse et al. (2009), Thrower and Hard (2009), and Courter et al. (2010, 2013). There is some debate over the extent of these interactions and their consequences, but some consensus seems to be emerging that these migratory forms are highly plastic, can be produced by parents of both types, and may have different fitnesses in different environments.

Review

A reviewer commented that the TRT made an assumption that in areas where there are known to be abundant populations of residents interacting with anadromous individuals, extinction risk was not allowed to exceed 90% in 100 years. The reviewer noted that this is likely to be a conservative estimate.

Response

The TRT adopted this value to reflect the likelihood that sufficient resident fish producing some smolts could ameliorate risk of extinction of an anadromous population in that drainage. The TRT concurs with the reviewer that this value would be case specific, and it could be higher or lower.

Review

A reviewer expressed concern that the simulation of the demographic effects of iteroparity on population persistence conducted by Nick Gayeski and summarized in this report might be too general to be applied to all DIPs within an MPG or to the entire DPS. The reviewer requested a copy of the Gayeski analyses to further review its validity. The reviewer also noted that, to validate these conclusions, there need to be complementary empirical studies assessing the importance of repeat spawners within individual DIPs (i.e., a comprehensive genetic study in which outmigrant smolts are successfully assigned to parents to understand the relative contribution of repeat spawners to smolt production).

Response

The summary of Gayeski's simulation modeling herein is the only report available, and it was reviewed and approved by the author. More comprehensive modeling work has not yet been published. The TRT agrees that the simulation results would not necessarily be applicable to individual DIPs in the Puget Sound DPS, and ultimately these results should be verified by empirical parentage studies. The TRT notes that Appendix G of this report is intended as a step in that direction, in describing the application of a demographic analysis to a molecular pedigree for a wild Alaskan steelhead population.

Review

A reviewer asked, in the TRT's description of viability criteria and scoring methodology for each DIP using "traffic light" rule sets, and in doing so determining whether an MPG could be considered viable, how was a geometric mean of 2.2 for viability scores for all DIPs within an MPG arrived at?

Response

Using the "traffic light" rule set, the TRT settled on a geometric mean viability of 2.2 based on an evaluation of viability scores for 8-DIP and 16-DIP MPGs, in which each DIP was given a rank score of 1, 2, or 3, corresponding to "not viable" (<40% probability of viability), "intermediate" (40–85% probability of viability), and "viable" (>85% probability of viability). Under a requirement of 40% viable DIPs in an MPG and no more than one DIP being not viable, the TRT adopted a minimum geometric mean viability score of 2.2 for the MPG to be considered viable.

Review

A WDFW reviewer commented on this statement in Appendix C: “Low productivity areas [those with gradients greater than 4%] were not included in the estimate of potential parr numbers.” The concern was that some of the highest parr densities observed by Gibbons et al. (1985) were in the higher gradient streams.

Response

We revised the estimates to consider the higher density reaches that had higher gradients and revised this statement accordingly.

Review

A WDFW reviewer thought that the IP estimate for the Drayton Harbor Tributaries Winter Run in Appendix E did not consider spawning habitat potential, indicating that this is the limiting factor for all salmonids in these drainages due to their geology.

Response

The TRT developed these estimates from the IP model in Appendix C and for consistency provided them in Appendix E. Because of the peculiar geology of this area, this IP estimate may overestimate the capacity of steelhead in this system, to be sure, but to an unknown degree. We added a statement to this effect to that appendix.

Review

A WDFW reviewer commented on the estimate of IP of steelhead in Nookachamps Creek given in Appendix E, stating that the estimate is “completely wrong.” The reviewer went on to say that the maps of spatial structure for several DIPs do not reflect reality, but did not provide specific information to show where they are inaccurate. The primary concern seems to be that some low gradient areas are given too high an estimate of IP, but spawning habitat is limiting in some of these areas.

Response

Without more detailed information to tailor the estimates to specific basins, it is difficult to address these comments specifically. We do not have specific punch card data for Nookachamps Creek, but for some systems we do have punch card data that indicate that steelhead were recovered from areas like these historically (i.e., in the first half of the twentieth century). The point is that current distribution should not be the primary guide to determining potential spawning or IP.

Review

A WDFW reviewer commented (on the spatial structure map in Appendix E) that in the Stillaguamish River, Canyon Creek above RKM 0.8 is summer-run steelhead territory, not winter-run steelhead territory.

Response

The distribution map in question considers both summer-run and winter-run steelhead. We added a comment to this effect to the text.

Review

A WDFW reviewer commented that summer-run steelhead spawning and rearing habitat in Deer Creek should not include the lower 2.5 miles, which are winter-run steelhead habitat.

Response

We amended the text in Appendix E to reflect this comment.

Review

A WDFW reviewer commented on the IP estimate for Canyon Creek in Appendix E, indicating that the estimate does not include the main stem of the creek down to RKM 0.8.

Response

The area in question is included in the Stillaguamish River estimate.

Review

A WDFW reviewer recommended that we clarify language regarding risk of extinction to read “no more than a 5% risk of extinction.”

Response

We made the suggested change where needed or altered the language to reflect a more general intention regarding extinction risk.

Editorial Comments

Review

Peer reviewers 1 and 2 both commented that the Executive Summary was too lengthy and should be more concise.

Response

We revised the Executive Summary substantially to meet this concern and incorporated many of the reviewers’ specific suggestions. In addition, we addressed each of the minor editorial comments made on the review draft by all reviewers and incorporated most of the suggested changes made by these reviewers throughout the entire document.

Policy-Related Comments

Review

A reviewer commented that delisting criteria should be clearly designated as recommended biological delisting criteria.

Response

We revised the document accordingly.

Review

A reviewer commented that the document differentiate the application of delisting criteria to extant vs. extinct populations.

Response

This technical memorandum does not address delisting criteria, but rather extinction risk and the criteria required for viability sufficient to minimize such risk. As the document focuses on DIPs that are all extant or their extirpation was recently documented (but may include extinct components), we focus herein on application of viability criteria to these (and more inclusive) units.

Review

A reviewer recommended that we further clarify the role of resident *O. mykiss* for steelhead viability.

Response

The revised document includes a complete section on the potential role of resident *O. mykiss* in viability of steelhead and its relevance to biological viability criteria.

Final Response to Reviewer Comments

The TRT's approach to assessing steelhead viability described in this technical memorandum enlists a variety of analytical tools to evaluate information available on steelhead ecology, population dynamics, phenotypic diversity, and habitat use—information that varies considerably in content and quality as well as quantity. Many of the comments and concerns of the reviewers have driven home this point, and their comments have focused on the utility of the TRT's hierarchical framework for assessing viability when empirical data are lacking, the validity of the analyses' underlying assumptions, and the approach the TRT has taken to evaluate the dependence of DPS viability on the viability of individual populations. However, none of the reviewers identified an alternative to the approach the TRT adopted. Clearly, some aspects of this approach have required a number of important assumptions, detailed in the body of this report, that are often clearly violated. With this in mind, the TRT has strived to implement methods that can accommodate and combine information of varying content and quality and are likely to be more robust to violation of key assumptions. The decision-theoretic approach we developed here, in the form of BN models, attempts to assess viability in a quasi-quantitative way across all three hierarchical levels within the Puget Sound DPS, from DIP to MPG to the DPS as a whole, and in a way that uses as much relevant information as possible while acknowledging the nature of that information. By discretizing the information in broad but distinct categories that describe its influence on viability, the models have allowed the TRT to assess current viability, examine the influence of different VSP parameters on viability, and establish a reasonably robust framework that can readily accept new or improved information on factors that influence viability. At a minimum, the work summarized in this report should serve to highlight the importance of 1) collecting the appropriate information from natural populations to help better evaluate viability and its components in Puget Sound steelhead, and 2) developing and implementing a rigorous, systematic monitoring program to detect changes in these populations. Figure 56 provides a simple sketch of some of the key data to collect and monitor for viability. Through the approach the TRT developed here, it should also prove useful to recovery planning by providing a transparent, systematic means of evaluating different recovery actions at the watershed scale.

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128 Myers, J.M., J.J. Hard, E.J. Connor, R.A. Hayman, R.G. Kope, G. Lucchetti, A.R. Marshall, G.R. Pess, and B.E. Thompson. 2015. Identifying historical populations of steelhead within the Puget Sound distinct population segment. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-128, 155 p. NTIS number PB2015-103741.

127 Roni, P., G.R. Pess, T.J. Beechie, and K.M. Hanson. 2014. Fish-habitat relationships and the effectiveness of habit restoration. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-127, 154 p. NTIS number PB2014-108836.

126 Russell, S., and M.S. Ruff. 2014. The U.S. whale watching industry of Greater Puget Sound: A description and baseline analysis. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-126, 171 p. NTIS number PB2014-105939.

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124 Anderson, L.E., and S.T. Lee. 2013. Washington and Oregon saltwater sportfishing surveys: Methodology and results. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-124, 61 p. NTIS number PB2014-101405.

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122 Frame, E., and K. Lefebvre. 2013. ELISA methods for domoic acid quantification in multiple marine mammal species and sample matrices. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-122, 20 p. NTIS number PB2013-109341.

121 Lian, C.E. 2012. West Coast limited entry groundfish cost earnings survey: Protocol and results for 2008. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-121, 62 p. NTIS number PB2013-104899.

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