INDEPENDENT SCIENTIFIC ADVISORY BOARD

Review of Spring Chinook Salmon in the Upper Columbia River

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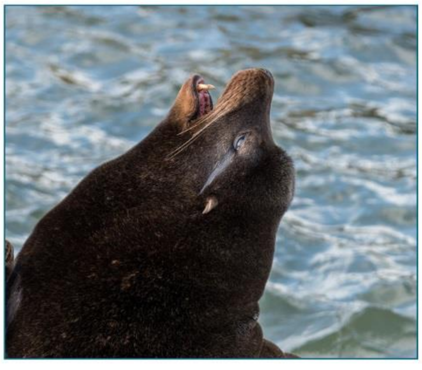

Cover design by Eric Schrepel

Cover photos: Spring Chinook salmon, Spring Creek, Methow River, Washington courtesy of U.S. Fish and Wildlife Service. Wanapum Dam Juvenile Fish Bypass and California sea lion by Tony Grover. ISAB Upper Columbia tour of Dillwater instream wood project site, Entiat River, Washington by Erik Merrill.

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Executive Summary

In response to a request from the Independent Scientific Advisory Board's (ISAB) Administrative Oversight Panel (April 2017 memorandum), the ISAB reviewed habitat assessment, research and monitoring, and prioritization and coordination of recovery actions for spring Chinook salmon in the Wenatchee, Entiat, and Methow basins of the Upper Columbia River. The review was requested because spring Chinook populations in the Upper Columbia River remain at high risk of extinction despite a decade of habitat restoration actions guided by the *2007 Upper Columbia Spring Chinook Salmon and Steelhead Recovery Plan*.

The Upper Columbia Salmon Recovery Board, tribes, state and federal managers, public utility districts (PUDs), and local groups assisted the ISAB in its review. Leaders in these groups presented valuable information to aid our review, including a field review of the Wenatchee and Entiat subbasins and more than 25 presentations. The recovery program in the Upper Columbia Basin is one of the better examples of an explicit strategy to guide local recovery actions, monitoring, and adaptive management.

The ISAB developed the following key findings and recommendations to address the Oversight Panel's four major questions (see section 1.1 of the Report for the specific questions):

1. **Is the identification of limiting factors for Upper Columbia River spring Chinook based on sound scientific principles and methods?**

The Upper Columbia Salmon Recovery Board (UCSRB) has refined its analysis of limiting factors substantially since the first assessments in the late 1990s. The scientific principles for identifying factors limiting the recovery of Upper Columbia spring Chinook salmon are generally sound. Limiting factors are defined in the 2007 Upper Columbia Recovery Plan as environmental conditions that negatively affect the abundance, productivity, spatial structure, and diversity of salmon populations. Analysis of limiting factors based on current habitat conditions is useful for prioritizing restoration actions. The UCSRB recently refined their traditional habitat-based approaches by weighting limiting factors based on anticipated survival benefits and geographic extent. Assessments of limiting factors are used to prioritize recovery actions, and the recent history of restoration is relatively consistent with the rankings of limiting factors.

Limiting factors analyses of the headwater basins do not incorporate the full life history of the fish and their geographic range from egg to adult spawner. Limiting factors for spring Chinook salmon in the Upper Columbia have been assessed through traditional freshwater habitat assessments, analysis of density dependence, and life cycle models. Each have important applications in management of the four Hs, but to date there has been no integration of all three approaches.

Empirical density dependence data and life-cycle modeling conducted by regional scientists in the Upper Columbia provide more holistic analyses of limiting factors over the full life cycle of spring Chinook salmon than traditional assessments of habitat-related limiting factors in freshwater. These traditional habitat-based approaches can be used in planning and prioritizing restoration actions, but more explicit integration of these approaches would strengthen future efforts.

The goal of the UCSRB program is to identify limiting factors within four categories of human impacts (threats), including habitat, harvest, hydrosystem, and hatcheries (four Hs), both within and outside the Upper Columbia watershed. While the Recovery Plan recognizes the influence of all Hs, analysis of limiting factors has not integrated the four Hs to determine which have the greatest influence on spring Chinook populations. Life-cycle models for these subbasins are beginning to provide evidence that addresses all Hs, but they are in early stages of development. Limiting factors are considered to be working hypotheses that can be tested, and monitoring and adaptive management are critical for understanding and addressing them. Key uncertainties such as ocean productivity and global climate change are identified.

Recommendation: The ISAB recommends integrating the results of the different approaches—limiting factors analysis, density dependence analysis, life-cycle modeling—for identifying limiting factors to guide future revision of the Biological Strategy of the UCSRB's Regional Technical Team and the Recovery Plan for spring Chinook salmon. This integration will require a collaborative process that includes significant participation by experts, practitioners, and management teams from all Hs.

1a. Are Snake River spring Chinook doing better than Upper Columbia spring Chinook, in terms of abundance, diversity, spatial structure, and productivity?

Differences in geography and biology result in differences in the spatial structure and diversity of the two evolutionarily significant units (ESUs). Nonetheless, most measures of population-level abundance and productivity for spring Chinook and assessments of habitat are similar in the two ESUs. In-river survival of spring Chinook smolts migrating in the Upper Columbia as compared to Snake River spring/summer Chinook smolts does not appear to differ. Smolt-to-adult survival (SAR) of wild Upper Columbia and Snake River Chinook also does not appear to differ.

Because the Snake River ESU has many more populations and larger absolute abundance than the Upper Columbia ESU, the consequences of the differences in relative proportions may be worse for Upper Columbia spring Chinook than for Snake River spring/summer Chinook. If in-river survivals are equal for the two ESUs, adverse events create greater risks for the Upper

Columbia populations because they are less buffered by adjacent populations. For example, if the proportion of losses to pinniped predation continues to increase but are the same for the two ESUs, the reduction of Upper Columbia spring Chinook spawners, which have smaller numbers, might reduce their ability to find mates on the spawning grounds. Thus, the ISAB believes that the Upper Columbia spring Chinook ESU may be exposed to greater risks than the Snake River spring/summer Chinook ESU. The same concern was expressed by NOAA Fisheries when the Upper Columbia ESU was originally listed as endangered and the Snake River ESU as threatened, listings that were not changed in the most recent reviews.

Recommendation: The ISAB recommends continued comparison of Chinook recovery in both ESUs to determine which restoration actions are most effective. Rigorous RME programs are essential to understand the trends and factors that influence them in these two basins.

1b. Differences between summer Chinook and spring Chinook in the Upper Columbia

Differences between spring and summer Chinook in the Upper Columbia could identify limiting factors for spring Chinook. Summer Chinook have greater juvenile life history diversity than spring Chinook and therefore may have greater intrinsic resilience to limiting factors than spring Chinook. Summer Chinook enter the estuary as subyearlings and yearlings over a much longer period than spring Chinook. Later out-migration timing of summer Chinook subyearlings coincides with the increased spill regime that was established to reduce dam related mortalities. Their outmigration timing is also better synchronized with increased flows and higher turbidity due to spring run-off. Smaller size at emigration may reduce vulnerability of summer Chinook to avian predators that prefer larger prey (e.g., Caspian terns). Summer Chinook typically rear for shorter periods in tributaries than spring Chinook and are less likely to experience capacity limitations or survival bottlenecks in subbasins. Contrary to previous understanding, some Upper Columbia spring Chinook juveniles rear in mainstem reservoirs for prolonged periods. The lower 235 river kilometers between Bonneville Dam and the ocean was identified as a feeding area for juvenile salmonids where competition for resources with other species and age classes may occur.

Recommendation: The ISAB recommends continued investigations of the effects of summer Chinook on spring Chinook in Upper Columbia River subbasins, including effects of hatchery practices, redd superimposition, competition, outmigration behavior, relative rates of survival and behavior in the mainstem Columbia, and the relative effects of pinnipeds and harvest in the lower river and estuary. Management actions should be developed to lessen constraints on spring Chinook abundance as information becomes available.

Analysis of demographic trends in the Upper Columbia may reveal relationships between summer and spring Chinook more effectively if adult returns are grouped by broodyear. Juveniles are counted by cohort or year class, and adults should be counted on a similar basis.

1c. Are pinnipeds potentially a significant source of mortality for Upper Columbia spring Chinook? Can the effect of pinniped predation of Upper Columbia spring Chinook be quantified?

Pinnipeds are potentially a significant source of mortality for Upper Columbia spring Chinook adults. However, population-specific estimates of predation impacts on Upper Columbia spring Chinook were not available to the ISAB at the time of this review. The estimated consumption of all Chinook salmon populations combined by pinnipeds in the Columbia River increased sharply over the past decade, likely exceeding mortality by fisheries. Potential impacts varied by pinniped species, salmon life stage, and run timing. In 2017, the reduced salmonid runs and persistently high numbers of pinnipeds later in the migration period suggested the total impact by pinnipeds on this year's salmonid run may be large.

Efforts to quantify the effects of pinniped predation on Upper Columbia spring Chinook continue to make progress. However, additional data and evaluation of uncertainties in the estimates and model structure are needed to improve estimates of pinniped predation on Upper Columbia spring Chinook salmon. In-river mortality of upriver spring/summer Chinook salmon adults peaked in 2014 and 2015 (~100,000 fish) and decreased in 2016 and 2017 to levels (~22,000 fish) similar to those in 2010-2013 (~31,000 fish). Estimates of adult Chinook mortality are highest in the estuary and Bonneville tailrace. A recent modeling study reported that additional sea lion predation on Columbia River Chinook salmon in the ocean (ocean age-1 salmon) may be similar to in-river consumption rates of adult Chinook.

Recommendation: The ISAB reiterates its recommendations from past reviews (see 3.4.2), and recommends proceeding with the pinniped recommendations listed in NOAA Fisheries 2016 Five-Year Upper Columbia Status Report: "(1) expand pinniped monitoring efforts to assess interactions between pinnipeds and listed species, (2) maintain predatory pinniped management actions at Bonneville Dam to reduce the loss of upriver listed salmon and steelhead stocks, (3) complete life-cycle/extinction risk modeling to quantify predation rates by predatory pinnipeds on listed salmon and steelhead stocks in the Columbia River and Willamette River, and (4) expand research efforts in the Columbia River estuary on survival and run timing for adult salmonids migrating through the lower Columbia River to Bonneville Dam." The second recommendation is a necessary precautionary measure while better data are collected.

In addition, the ISAB recommends identifying and investigating other potentially significant sources of mortality of Upper Columbia spring Chinook smolts and adults in the Columbia

River plume/ocean shelf habitats, estuary, and lower mainstem and tributaries. New information from NOAA's tagging and modeling efforts revealed important data gaps, including lack of population-specific survival estimates for Upper Columbia spring Chinook.

The ISAB recommends use of a variety of approaches to quantify pinniped predation impacts, such as the ongoing tagging studies and coast-wide bioenergetics/life-cycle modeling. Comparison of multiple models could reduce structural uncertainty (e.g., comparing a bioenergetics approach to individual-based models or time series models).

2a. Are habitat recovery actions being prioritized and sequenced strategically? How should habitat projects be prioritized?

The ISAB found the UCSRB's system for solicitation, review, and project design to be scientifically sound with regard to habitat conditions and potential effects of hatcheries and the hydrosystem. Current methods of prioritization (e.g., Ecosystem Diagnosis and Treatment (EDT) Model, Habitat Suitability Index (HSI), and Regional Technical Team's (RTT) Biological Strategy) are useful for planning and prioritizing ecological concerns and related habitat restoration actions.

The UCSRB focuses most attention on determining which kinds of actions will be biologically beneficial toward recovery of Upper Columbia spring Chinook and assesses fish abundance, productivity, and risk of extinction. The procedure for characterizing cost effectiveness is not quantitative and does not provide a rigorous basis for prioritizing actions. If funds are unlimited, there would be no need to prioritize actions, but that clearly is not the case. The criteria used by the Regional Technical Team and the Citizens Advisory Committee are vague, and the results are weighted so they have little influence on project priorities. Cost effectiveness is considered by the UCSRB and the Salmon Recovery Board, but there is no explicit analysis and the evaluations are not documented.

The existing prioritization process could be strengthened by incorporating explicit analysis of performance, time, and cost in a cost-effectiveness assessment. Studies have shown that doing so can improve outcomes by an order of magnitude. However, this will require empirical or other ways of estimating biological benefits as well as estimating project costs. Effectiveness in terms of smolts per adult (i.e., freshwater productivity) is difficult to estimate, as are the effects over time, but developing a clear method would better highlight knowledge gaps. In the interim, eliciting estimates from a set of experts could provide an objective starting point for assessing relative cost effectiveness of projects. We recommend developing a standard estimation of cost-effectiveness ratios for evaluating each proposed project and describe a simplified approach as a starting point (section 4.1.2).

Recommendation: The ISAB recommends applying a transparent cost-effectiveness analysis of each proposed project, perhaps by using the approach in the simple example we describe in this Report's Box 4.1 as a starting point. The lack of rigorous cost effectiveness analysis and its minor influence on prioritization of restoration actions for all Hs limit the USCRB and participants in their recovery efforts. This is a common deficiency throughout the Columbia Basin, but such analyses would allow the program to use its limited resources more effectively.

2b. Is there evidence that past projects have improved habitat for this ESU? What types of habitat projects should be prioritized in the future?

There is evidence that some types of projects have improved habitat more than others. The ISAB found that there is sufficient evidence that protecting habitat, removing barriers to restore connectivity, and reconnecting side channels and floodplains have positive effects on anadromous salmonids, including spring Chinook salmon. Projects at different scales within the Upper Columbia provide strong evidence that structures that increase pools and habitat complexity can increase fish production, survival, and abundance. Effects of log and boulder structures should be measured to understand effects of specific types of structures in particular watersheds.

Empirical data and modeling from the Upper Columbia and other locations within and beyond the Columbia River Basin support ranking habitat protection as a high priority, followed by removing barriers, and reconnecting floodplains and side channels. Increasing habitat complexity using log and boulder structures is a useful short-term approach, but a long-term strategy is needed to restore processes that maintain channel complexity and supply and retain large wood in rivers. Less information was available on projects that increase instream flows or address water quality, although these can also be effective.

Recommendation: Projects that restore and sustain key fluvial and ecological processes should be considered high priority, given predictions for future climate and building on the successes of the projects completed so far. A key goal will be to provide habitats that are resilient to changing conditions and extreme events, and ones that provide connected habitats needed to sustain the full range of life history diversity among spring Chinook in the Upper Columbia.

The ISAB recommends designing rigorous experiments and continued careful monitoring to measure the effectiveness of habitat restoration practices in the upper Columbia subbasins across a hierarchy of biological responses, including use of habitats by fish, and their abundance, growth, survival, and productivity. Viable Salmonid Population (VSP) measurements should be compared against model predictions to verify and improve modeling approaches.

2c. How well are actions in other management sectors (all H's, i.e., habitat and hydrosystem, hatcheries, and harvest) aligned with recovery efforts?

The UCSRB has developed a useful process for prioritizing restoration projects and coordinating recovery actions. The regional recovery plan, limiting factors assessment, life-cycle models, and monitoring provide critical information for recovery actions. However, a continued challenge is coordinating groups in the three subbasins responsible for the four Hs. More than 16 independent coordinating committees and several other major working groups make critical decisions on recovery actions. Currently, there is no process for integrating their separate efforts into a coordinated action plan across the three subbasins.

Coordination of habitat actions and harvest management and hatchery operations also could be improved. Continued management to reduce effects of hatcheries on fitness of spring Chinook in the three subbasins need to be coordinated with prioritizing and implementing habitat restoration projects to improve recovery efforts. Collaborative discussions of the UCSRB and harvest co-managers about influences of adult return rates on spring Chinook recovery and potential harvest recommendations would strengthen recovery efforts of the UCSRB.

Recommendation: The ISAB encourages the UCSRB and its participants to develop a systematic, collective process for coordination of the actions, monitoring efforts, and decisions across the numerous working groups and coordinating committees in the three subbasins.

If return of adult spawners or recruitment substantially limit recovery in the upper Columbia, then discussions of the effects of harvest on escapement between co-managers and participants in the UCSRB could improve recovery efforts. More dialogue between the Regional Technical Team and harvest co-managers under U.S. v. Oregon could align habitat restoration actions with returns of adult spawners needed for recovery.

Hatchery supplementation has not increased spring Chinook abundance or productivity and genetic diversity has decreased compared to historical diversity. Coordination between habitat restoration actions and hatchery operations and studies of the effects of hatchery supplementation should remain critical components of spring Chinook salmon recovery in the Upper Columbia.

3. **Is a research, monitoring, and evaluation (RME) framework in place that can adequately address the questions in #2 above? Can this RME framework provide suitable data to test and validate hypotheses, inform management decisions, and confirm that limiting factors were correctly identified and are being addressed effectively?**

The RME program is funded largely through the responsibilities of the PUDs under licensing agreements. As a result, it is largely focused on assessing hatchery practices and the effects of hatcheries on spring Chinook salmon populations. While this is a critical aspect of recovery in the upper Columbia, it does not address all actions of the recovery program. Currently, there is no RME Plan that encompasses all Hs and their related working groups, and there is no process to coordinate monitoring efforts across the subbasins and address information needs related to all Hs.

Approaches and methods of the Regional Technical Team, PUDs, and regional fisheries agencies are generally appropriate and can be used to answer questions about effects of hatcheries and the hydrosystem, but analyses could be improved. Upper Columbia RME planning efforts among the PUDs, WDFW, UCSRB resulted in a thoughtful process to identify reference populations that could be used to help assess how hatchery supplementation efforts were influencing total spawner abundance, natural origin spawner abundance, and productivity (recruits per spawner) in supplemented streams.

In many instances in the Upper Columbia, the RME program compares a supplemented population to three or more reference populations, but the results of the individual assessments are not aggregated. A more sophisticated analytical approach that simultaneously examines data from the supplemented population and all reference populations would improve the precision of the estimates and increase the power to detect effects.

The RME efforts track temporal changes in abundance, productivity, spatial structure, and genetic diversity (VSP parameters) relative to control or reference populations. However, many factors (e.g., environmental conditions experienced under artificial culture, genetic changes due to domestication, environmental conditions experienced by adults and their offspring in natural streams) can also alter VSP parameters. Influences of these factors on VSP parameters must be disentangled before long-term effects of hatchery practices on fitness can be estimated. Life-cycle models are a promising way to evaluate the relative effects of ecological concerns and human actions to better design and prioritize information needs and potential effectiveness of restoration actions.

Assessing potential impacts of early fish cultural practices is problematic because they rely on historical accounts and speculations about possible consequences. Recent genetic studies of Columbia River Chinook indicate hatchery practices and effects of mainstem and tributary dams have reduced genetic diversity of Upper Columbia spring Chinook populations. Genetic analyses offer increasing potential to quantify the influences of past practices on the fitness of anadromous salmon and steelhead populations.

Recommendation: The ISAB recommends developing an integrated RME Plan that encompasses all Hs and their related working groups to coordinate monitoring efforts related to all Hs across the three subbasins.

The ISAB recommends that (a) biological criteria should be given more weight when selecting reference populations in comparing supplementation streams with reference streams, and (b) the treatment population should be compared to all of its reference populations simultaneously rather than one-at-a-time. These will lead to reduced uncertainty about the effects of supplementation and increase the power to detect effects.

Consideration should also be given to using Bayesian Analysis which allows for incorporation of prior beliefs on the value of supplementation and integration of multiple performance measures, and Bayesian Analysis can more easily address different sources of variability.

3a. To what extent has the fitness of the Upper Columbia spring Chinook ESU been negatively or positively affected by historical and current hatchery programs in this ESU?

Contemporary populations of Upper Columbia River Chinook salmon exhibit significantly lowered genetic diversity compared with historical stocks (i.e., before European colonization), which included fish from areas above Chief Joseph Dam. Losses of genetic diversity occur when population size diminishes and geographic range contracts over time. Snake River Chinook salmon populations do not appear to have experienced the same degree of loss of genetic diversity over time because comparative changes in abundance and distribution are not as pronounced as those in the Upper Columbia River (Johnson et al. 2018).

Genetic monitoring indicates that past hatchery programs have contributed to further erosion of genetic diversity in Upper Columbia River salmon through founder effects, reduced fitness of hatchery-origin fish, and high variance in family size of brood stock. Some practices in Upper Columbia hatcheries have increased straying rates, which have caused genetic differences among spring Chinook populations returning to the same subbasins to decrease.

Lowered genetic diversity in Upper Columbia populations means fewer populations with local adaptations and less ability of existing populations to adapt to changes in climate and other factors. Preservation of existing genetic diversity in Upper Columbia populations should thus remain a key goal and guiding principle for hatchery operations, stocking and supplementation, resident fish mitigation, and other management activities. A recently updated and comprehensive monitoring and evaluation plan is in place to provide guidance on best practices for hatchery operations. Basinwide genetic monitoring of natural- and hatchery-origin fish is also essential to inform adaptive management aimed at preserving the remaining genetic diversity in Upper Columbia Basin Chinook salmon.

The extensive history of past transplant failures casts doubt on whether the Grand Coulee Fish Maintenance Project (GCFMP) successfully preserved the genetic heritage of the salmonid stocks located above Grand Coulee Dam. Small extant populations of salmonids existed in Upper Columbia prior to the GCFMP. It is possible that these fish had fully seeded the poor juvenile habitat then available. Additionally, spring and summer Chinook captured at Rock Island Dam and used as hatchery broodstock or natural spawners, originated from multiple upstream spawning areas, which may perform poorly relative to those originating from populations closer to their transplant locations. Evidence indicates some hatchery programs have altered genetic diversity and fitness of Columbia River Chinook salmon.

Recommendation: In view of the lack of response to supplementation programs in the Upper Columbia, the ISAB recommends continued improvement of their hatchery practices and RME program and additional studies to determine why spring Chinook have not responded to supplementation. Additional investigations of genetic diversity and comparison of historical samples with contemporary samples of spring Chinook from the Upper Columbia are also needed to better understand the extent of loss of genetic diversity and likely causes.

Further investigations that explore the factors responsible for early maturation in male spring Chinook salmon are also encouraged. Conservation benefits and risks associated with releasing precocious parr are affected by the factors that drive early maturation in hatchery stocks. Consequently, a thorough understanding of what enhances precocious maturation in hatchery stocks would help determine if anything should be done to control the release of early maturing male parr.

3b. To what extent have contemporary supplementation programs provided a demographic benefit to the natural populations?

Current hatchery operations have not increased overall abundance or the abundance of natural-origin spring Chinook. Studies of supplemented and non-supplemented reaches reported no differences in either total or natural origin returns. Recent trends in recovery of spring Chinook in the Wenatchee (supplemented population) and Entiat (non-supplemented populations) do not differ. Productivity of these populations appears to be stable.

In the Methow subbasin, three supplemented spring Chinook populations on the Twisp, Chewuch, and Methow rivers did not have increased overall abundance or productivity relative to reference populations. Natural-origin abundance did not change in the Chewuch and Methow and decreased in the Twisp. In the Wenatchee subbasin, spring Chinook hatchery programs include two supplementation programs (Chiwawa River and Nason Creek) and a segregated harvest augmentation (Leavenworth Hatchery). The supplementation program in the Chiwawa did not change overall abundance, natural-origin abundance, or productivity.

Straying rates in some hatchery programs are quite high (e.g., ≥3 5% for the Twisp and Chewuch programs in the Methow and Chiwawa hatchery in the Wenatchee), with most fish moving to other locations within the same watershed and some fish straying into other Upper Columbia River subbasins. Genetic diversity of the Chewuch population is decreasing and becoming similar to the Methow, which could be caused by straying. Some degree of straying can expand the spatial diversity of a population, but high straying rates can erode stock specific adaptations and lower productivity.

Recommendation: Continued development and validation of the life-cycle model being developed for Wenatchee River spring Chinook is encouraged. The model is designed to evaluate the effects of hatchery domestication, climate change, pinniped predation, ocean conditions, and freshwater habitat on the population dynamics of Wenatchee spring Chinook. At present, many of the model's parameter values are fixed, with only a few being estimated by an ad hoc calibration method. Given its many assumptions and fixed parameters, its current outputs can only be used qualitatively (ISAB 2017-1). The model, however, appears to have the flexibility to incorporate new data and efforts to refine it are underway. When completed, it has the potential to be a valuable tool for management.

Understanding the genetic legacy from Chinook stocks of the Columbia River above the Grand Coulee Dam and effects of past management is critical for managing spring Chinook in the Upper Columbia. Historical samples (e.g., museum specimens, scale samples from the early collections and hatchery operations) should be compared with contemporary spring Chinook to measure changes in genetic diversity. It is also important to continually refine RME activities that are focused on trends in contemporary genetic diversity of Upper Columbia River Chinook salmon. To be maximally effective, RME activities should be designed and coordinated for the entire Upper Columbia watershed.

4. Are the life-cycle and habitat models in development for the Upper Columbia ESU useful for informing the identification, prioritization, and evaluation of restoration actions?

In general, the life-cycle models being developed will be useful to investigate the relative impacts of restoration actions. The ISAB recently reviewed life-cycle models, including those of the Wenatchee, Entiat, and Methow rivers (ISAB 2017-1). Although these models are still relatively early in their development, they have analyzed potential limiting factors for spring Chinook salmon in the Wenatchee and Entiat basins and provided a food web model for the Methow River floodplain.

Empirical studies of the benefits of management actions (e.g., fish-in and fish-out studies of habitat improvements) are limited to short time and small spatial scales. They are difficult to scale up and evaluate at larger system levels. Life-cycle models can be used to scale up management actions to larger spatial (e.g., entire Columbia River, estuary and ocean) and temporal scales (e.g., entire life cycles), with appropriate attention to model limitations.

Current models rarely model non-linearities and feedback mechanisms (e.g., density dependence, interactions within and among species) in more than one life-cycle stage. At this point, the models are useful for ranking the relative benefit of management actions at the population level but may not perform well when predicting the exact numerical responses in salmon populations. These models are also useful for identifying which life-cycle stages are most sensitive and examining potential scenarios for improvements. They can also be used to generate hypotheses for experiments and management actions. Life-cycle models strengthen limiting factors analysis, prioritization, and RME programs by identifying data that are missing and needed. The next steps in modelling are to better calibrate the models to actual life-cycle data both at fine and large temporal and spatial scales and to include more complex relationships in life-cycle stages where the models are most sensitive.

Recommendation: The ISAB recommends continued development of the life-cycle models, incorporation of more recent information on fish habitat relationships, and development of scenarios that more completely represent the restoration actions and factors that are likely to influence recovery. The life-cycle models should be continually refined and improved. We recommend using the life-cycle models to rank proposed restoration actions and incorporate their results in analysis of cost effectiveness.

Some restoration actions are river specific, but other actions are common across the models. It would be helpful to develop a set of scenarios for these actions (e.g., incorporate recent restoration project types, use Comparative Survival Study predicted results for in-river survival from changes in spill/flow to evaluate overall impacts of changes in river survival).

Sensitivity analyses should be performed on all models to identify which limiting factors are most important. These sensitivity analyses should use a standardized set of options. The models should be calibrated to earlier life-cycle stages. For example, the food web models should be calibrated to actual data on smolts produced rather than trying to calibrate for the entire life-cycle. This will improve confidence in the direct benefits of some restoration actions.

We recommend leveraging the experience gained in applying the EDT models in the Okanogan and Methow subbasins if EDT models are developed for the other subbasins. The species habitat rules in the EDT model should be evaluated closely if the model is used.

Where possible, multiple models can be compared to better understand and quantify uncertainties and relationships between limiting factors and responses in the basin.

1. Introduction

1.1. Review Charge

In an April 2017 memorandum, the Independent Scientific Advisory Board (ISAB) Administrative Oversight Panel[1] asked the ISAB to conduct a review to inform Upper Columbia River spring Chinook recovery and research efforts. The Upper Columbia River spring Chinook evolutionary significant unit (ESU) was listed as endangered in 1999 and includes three extant populations for the Wenatchee, Entiat, and Methow subbasins as well as one extinct population for the Okanogan subbasin. The Oversight Panel's review request describes that despite a decade of habitat restoration actions guided by the 2007 Upper Columbia Spring Chinook Salmon and Steelhead Recovery Plan, Upper Columbia River spring Chinook populations remain at high risk of extinction (NOAA 2016). The Oversight Panel asked for the ISAB's high-level evaluation of available information to inform the Council and recovery practitioners about aspects of their programs, plans, and projects that would benefit from further refinement and consideration of alternatives.

The Oversight Panel specifically asked:

1. Is the identification of limiting factors for Upper Columbia River spring Chinook based on sound scientific principles and methods? Are the most important survival bottlenecks or factors limiting this ESU's recovery identified? Where and when do the most important limiting factors occur? Is density dependence considered? Are the necessary data available to identify the limiting factors? Are assumptions, data gaps, and key uncertainties identified?

    a) Based on recent status reviews and other relevant assessments, are Snake River spring Chinook doing better than Upper Columbia spring Chinook, in terms of abundance, diversity, spatial structure, and productivity? If so, do we know why? Do limiting factors and life histories differ between Snake River and Upper Columbia spring Chinook? For example, are there key limiting factors for Upper Columbia spring Chinook upstream of Priest Rapids dam?

    b) Pinniped predation appears to be increasing rapidly in the lower Columbia River. Are pinnipeds potentially a significant source of mortality for Upper Columbia

[1] The ISAB Administrative Oversight Panel consists of the Northwest Power and Conservation Council's chair, the executive director of the Columbia River Inter-Tribal Fish Commission, and the science director of NOAA-Fisheries' Northwest Fisheries Science Center.

spring Chinook? Can the effect of this predation on Upper Columbia spring Chinook be quantified?

2. Are habitat recovery actions being prioritized and sequenced strategically, given existing knowledge and data gaps? Is there evidence that past projects have improved habitat for this ESU? How should habitat projects be prioritized and what types of habitat projects should be prioritized in the future? Why? How well are actions in other management sectors (all H's, i.e., habitat and hydrosystem, hatcheries, and harvest) aligned with recovery efforts? Specific input to inform development and refinement of the Upper Columbia's proposed prioritization framework for projects would be much appreciated.

3. Is a research, monitoring, and evaluation (RME) framework in place that can adequately address the questions in #2 above? Can this RME framework provide suitable data to test and validate hypotheses, inform management decisions, and confirm that limiting factors were correctly identified and are being addressed effectively? If not, what changes need to be made to the RME Framework and what critical uncertainties (ISAB/ISRP 2016-1; draft Research Plan) and hypotheses should be investigated to provide the answers? Do we know how to test these hypotheses?

   Specific questions associated with uncertainties regarding hatchery fish interactions and research in the Upper Columbia include:

   a) To what extent has the fitness of the Upper Columbia spring Chinook ESU been negatively or positively affected by historical and current hatchery programs in this ESU?

   b) To what extent have contemporary supplementation programs provided a demographic benefit to the natural populations?

   c) Is the current methodology in the PUD hatchery monitoring and evaluation program (see Appendix C) sufficient to answer the questions above (a and b)?

4. Are the life-cycle and habitat models in development for the Upper Columbia ESU useful for informing the identification, prioritization, and evaluation of restoration actions? At what resolution scale can this guidance be applied, for example, watershed, population, or reach scale? Are there other approaches that would be useful?

The Oversight Panel encouraged the ISAB to organize briefings and site visits with researchers and restoration practitioners involved with Upper Columbia spring Chinook recovery, many of whom provided input on the review questions assigned to the ISAB. These entities include

NOAA Fisheries; the Upper Columbia Salmon Recovery Board (UCSRB); Washington Department of Fish and Wildlife; the Tribes including the Colville Tribes, the Yakama Nation, and the Upper Columbia United Tribes; and Grant, Chelan, and Douglas County Public Utility Districts. Finally, the Oversight Panel recognized that several recent ISRP and ISAB reviews were related to restoration and monitoring efforts in the Upper Columbia and should be considered in the ISAB's review, including the ISRP's review of the UCSRB's umbrella habitat restoration project ISRP 2017-2 and the ISAB's review NOAA's Life-cycle Model (ISAB 2017-1).

1.2. Introduction to Upper Columbia Spring Chinook Salmon and Their Recovery

The Upper Columbia River (UCR) supports important species of fish and wildlife, water resources, forests, and productive working lands for the Columbia River Basin. The four major river basins—the Wenatchee, Entiat, Methow, and Okanogan—comprise 4.5% of the Columbia Basin area and currently contribute approximately 3% of the total mean flow of the Columbia River after domestic and agricultural water withdrawals (Figure 1.1). The Upper Columbia River provides critical habitats for spring and summer Chinook salmon, sockeye salmon, coho salmon, steelhead, and Pacific lamprey. Development of the Columbia River hydrosystem has strongly impacted anadromous fish populations, and Chief Joseph Dam completely blocks upstream fish passage on the mainstem Columbia River. Of the 19 evolutionarily significant units (ESU) of salmon and steelhead in the Columbia Basin, 13 are listed as threatened or endangered under the Endangered Species Act (ESA).

The questions addressed by the ISAB in this report focus on the ESU for spring Chinook salmon *Oncorhynchus tshawytscha* in the Upper Columbia River, which includes three populations in the Wenatchee, Entiat, and Methow subbasins (see distribution maps in Figures 1.2, 1.3, 1.4). We did not include the Okanogan River in this review because spring Chinook salmon were extirpated from the basin in the 1930s by hydropower development, overfishing, and habitat degradation (see Box 1.1 below).

Figure 1.1. Map of mainstem Columbia River and location of the Wenatchee, Entiat, and Methow basins (source: www.digitalarchives.wa.gov/GovernorLocke/gsro/regions/upper.htm). Boundary of Columbia Basin is outlined in white and the three basins in this report are highlighted.

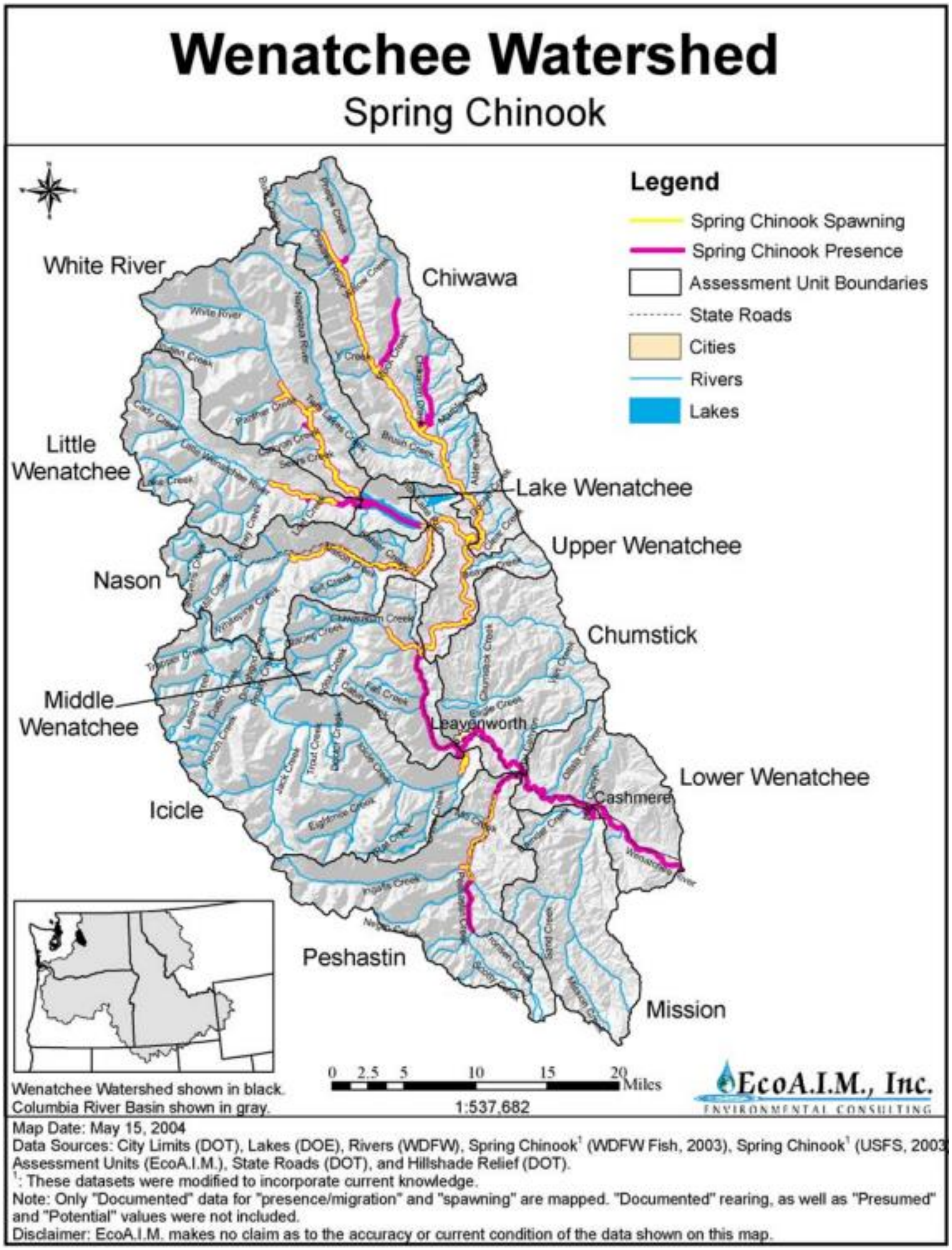


Figure 1.2. Map of spring Chinook salmon distributions in the Wenatchee River basin (Wenatchee Subbasin Plan 2004)

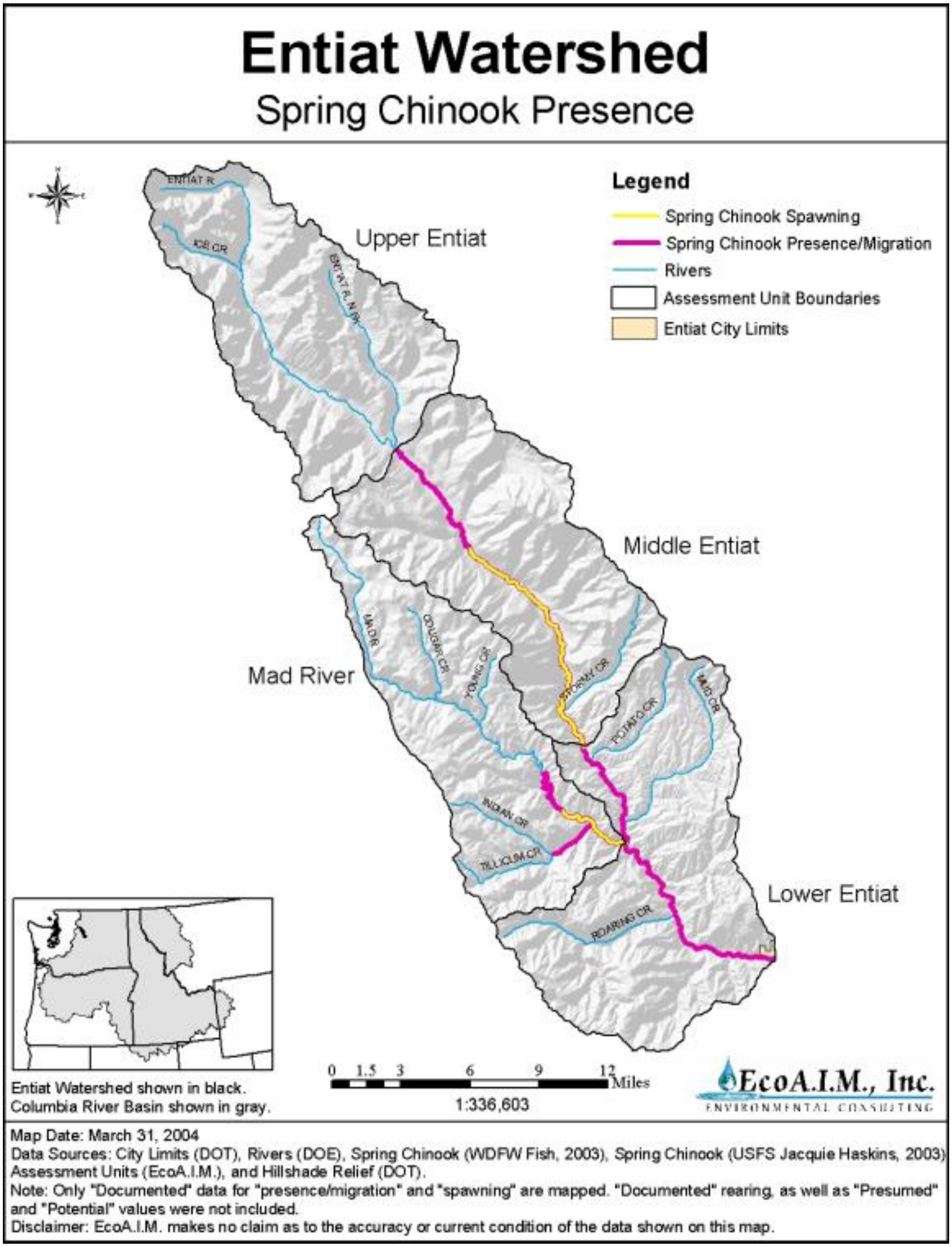


Figure 1.3. Map of spring Chinook salmon distributions in the Entiat River basin (Entiat Subbasin Plan 2004)

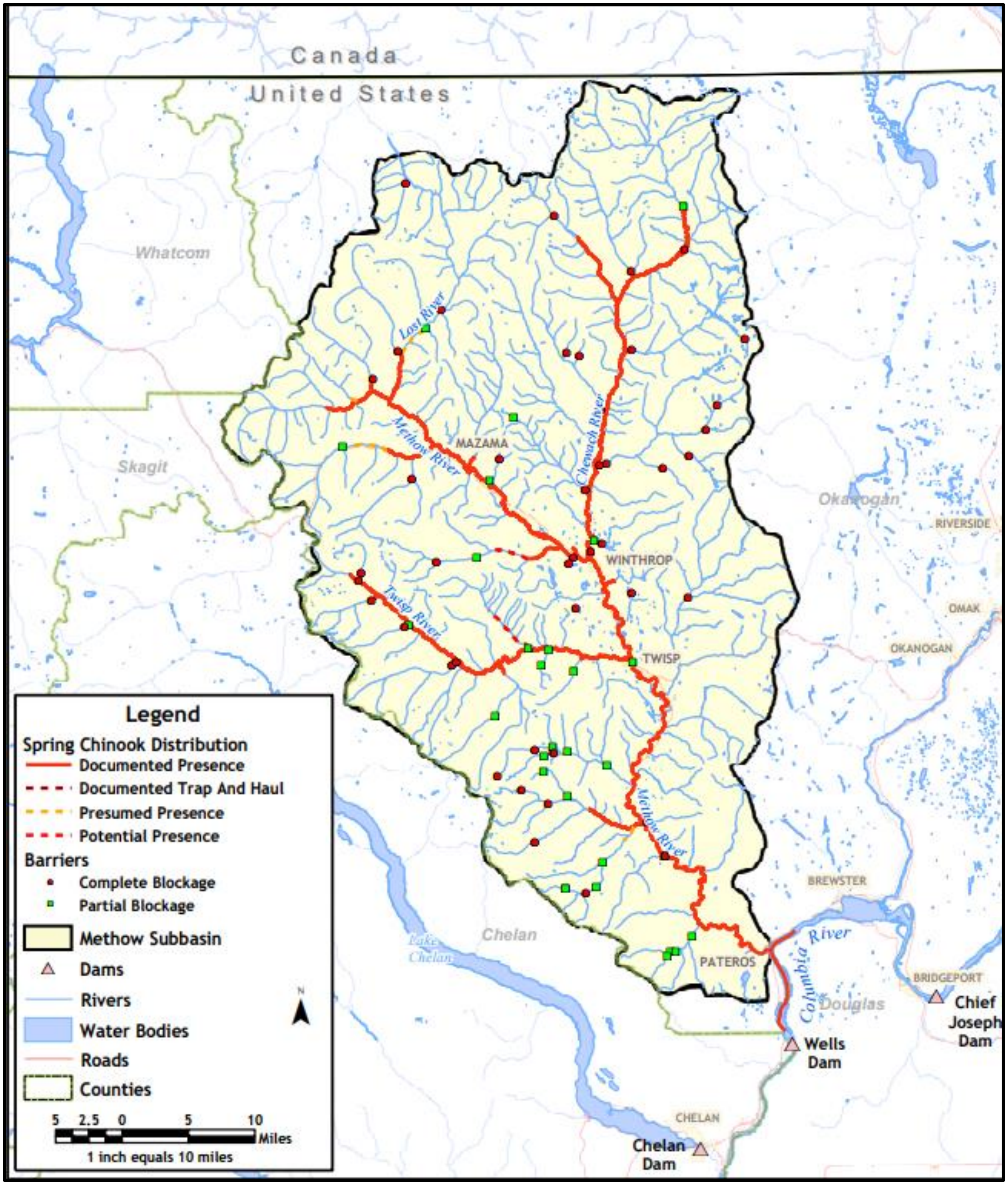


Figure 1.4. Map of spring Chinook salmon distributions in the Methow River basin (Methow Subbasin Plan 2004)

Fifty miles downstream of Grand Coulee Dam and 12 miles upstream of the Okanogan River confluence, Chief Joseph Dam construction began in 1949, and it now creates the upstream limit of anadromous salmon and steelhead in the mainstem Columbia River. The Colville Tribe, Canadian government, and Upper Columbia River communities are exploring options to provide passage above these dams to historical natal rivers (see Box 1.1). Though this report does not include the Okanogan Basin, we review the Ecosystem Diagnosis and Treatment (EDT) model used to prioritize habitat projects in the Okanogan River, which also is being developed in the Methow Basin.

NOAA Fisheries listed the Upper Columbia River spring Chinook ESU as endangered under the ESA in 1999. The Upper Columbia Salmon Recovery Board (UCSRB) and National Marine Fisheries Service (NOAA Fisheries) developed the Upper Columbia Spring Chinook Salmon and Steelhead Recovery Plan (2007) to guide management and restoration actions for recovery of these populations. The overall goal of the recovery plan was “to secure long-term persistence of viable populations of naturally produced spring Chinook and steelhead distributed across their native range.” The plan included more specific objectives to reclassify (downlist) these populations from endangered to threatened and then ultimately to delist the populations. Recovery for naturally produced UCR spring Chinook populations ultimately will require viable levels of abundance, low risk of extinction, spatial distributions throughout previously occupied areas, and natural patterns of genetic and phenotypic diversity. The Recovery Plan for the UCR requires populations of spring Chinook salmon to meet recovery criteria of less than a 5% extinction risk over a 100- year period in each of three basins (Wenatchee, Entiat, and Methow) (ICTRT 2007a, b). The Recovery Plan developed both short-term and long-term actions to reduce impacts of harvest, hatcheries, the hydrosystem, and habitat degradation in each of the three river basins. The actions were based on analysis of potential limiting factors identified in previous landscape assessments of the Entiat (Andonaegui 1999), Wenatchee (Andonaegui 2001), and Methow (NPCC 2004).

NOAA Fisheries (NMFS 2016) concluded that spring Chinook salmon in the UCR have not recovered substantially and remain at high risk of extinction after more than nine years of management actions and more than 300 habitat projects in the three river basins (Figure 1.5). Given the long history of multiple impacts on UCR spring Chinook, time required to achieve restoration objectives, and the ESU’s relatively long generation time (3-5 years), major recovery in less than 10 years would be unlikely. Nonetheless, the apparent slow rate of recovery in UCR spring Chinook populations has raised questions about the reasons for the lack of improvement in their status.

Box 1.1

Okanogan and Blocked Areas Spring Chinook Reintroduction Programs

The Upper Columbia spring Chinook salmon ESU currently contains three distinct populations in the Wenatchee, Entiat, and Methow subbasins. Spring Chinook populations once existed in the Okanogan and in multiple areas above the Grand Coulee Dam. Two reintroduction programs are attempting to expand the spatial distribution, genetic diversity, productivity, and abundance of spring Chinook in the Upper Columbia.

The Confederated Tribes of the Colville Reservation (CCT) worked with NOAA Fisheries to establish an experimental non-essential population of spring Chinook in the Okanogan subbasin. In 2014, The CTT obtained spring Chinook parr and eyed eggs from Methow-Chewuch River. The parr were placed into an acclimation pond in the Okanogan subbasin and reared for five months before being released in mid-April, 2015. The eyed eggs were incubated at the newly constructed Chief Joseph Hatchery and were held and reared at an acclimation site in the Okanogan prior to being released. Eggs from Methow-Chewuch spring Chinook continue to be imported into the Chief Joseph Hatchery and used in the Tribes' three-phased reintroduction program. The first phase is an attempt to colonize Omak Creek with spring Chinook; phase two focuses on local adaption, and phase three concentrates on spring Chinook conservation. The initial results are promising as adults from the first release of fish made in 2015 returned to the Okanogan in May 2017.

The second effort, led by the Spokane Tribe of Indians and other Upper Columbia Tribal Nations and their partners, is designed to reintroduce spring Chinook and other salmonid species into areas blocked by the Grand Coulee Dam. Planning efforts include multiple tributary and mainstem habitat assessments, estimates of the rearing capacity of Lake Roosevelt, life cycle modeling, an evaluation of risks associated with reintroduction, appraisals of possible donor stocks, and a review of fish passage technologies at high head dams. Future reintroductions depend on results of these assessments. Reestablishing spring Chinook in currently blocked areas would not only be of tremendous cultural value but it would also expand existing VSP parameters for Upper Columbia spring Chinook.

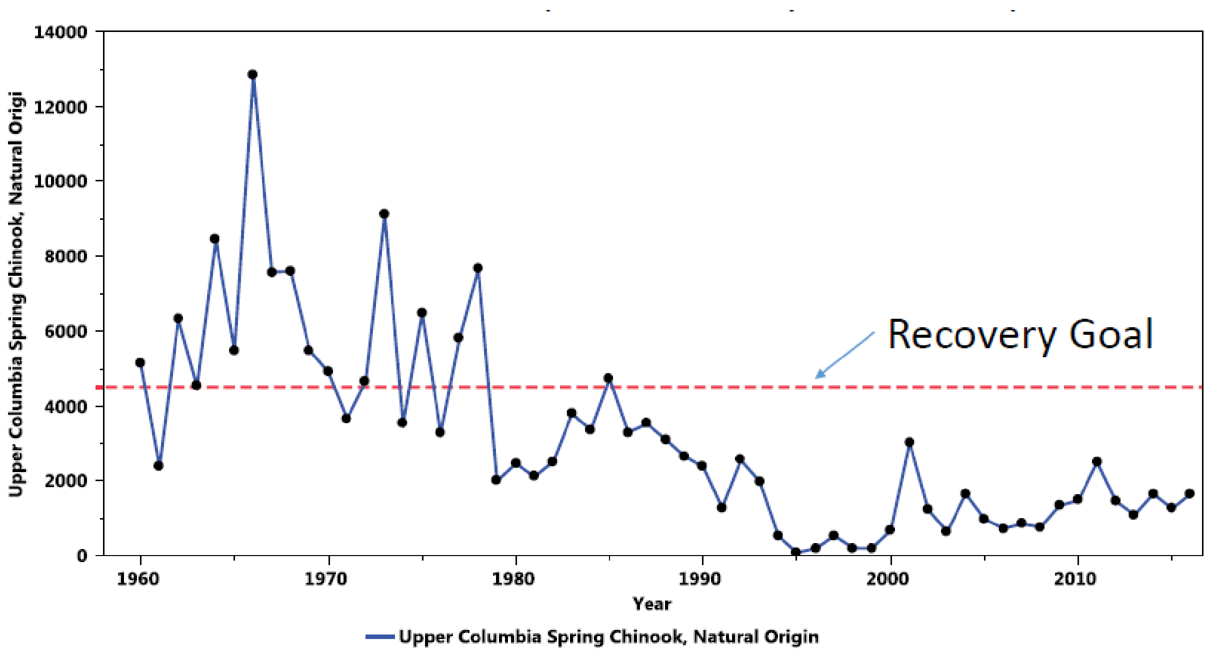


Figure 1.5. Abundance of natural origin spring Chinook salmon in the Upper Columbia River from 1960-2016 (Todd Pearsons, presentation to ISAB)

The Upper Columbia River historically supported major runs of salmon and steelhead, but overfishing, mining, logging, grazing, agriculture, water withdrawal, and growth of towns along the river caused major declines in their abundance in the early 20th century (Fish and Hanavan 1948; Mullan et al. 1992). Even before construction of major dams on the Columbia River, salmon and steelhead populations in the UCR had declined greatly:

> "Their runs, however had been virtually decimated during the past thirty or forty years largely through the construction of impassable mill and power dams and by numerous unscreened irrigation diversions." (Fish and Hanavan1948)

Commercial harvest of UCR salmon and steelhead exceeded 85% of returning adults in the 1930s and 1940s (Mullan 1987). Harvest remained high (40-60%) through 1974, decreasing to 5-18% from 1977-2013 (NOAA Salmon Population Summary database). Managers introduced millions of spring Chinook fry raised in lower Columbia River hatcheries into UCR tributaries from 1899-1931, potentially weakening local adaptations to the upper river habitats and environmental conditions. Development of agriculture and towns in these subbasins removed substantial quantities of water. More than 75% of the surface water of the Wenatchee is allocated for irrigation, the majority of which occurs during the summer low-flow period (Washington Department of Ecology 1995).

Development of the Columbia River hydrosystem in the 1930s had major impacts on all anadromous fish in the Upper Columbia River. Starting in 1939, Grand Coulee Dam has compressed 1,140 miles of river used for anadromous migration to less than 677 miles. From 1939 through 1943, fisheries managers intercepted all adult salmon and steelhead at Rock Island Dam (at river mile 453) and fish were mixed together and held for redistribution as part of the Grand Coulee Fish Maintenance Project (GCFMP) (Fish and Hanavan 1948). During the initial period from 1939 to 1943, the GCFMP stocked spring Chinook and summer steelhead in Nason Creek in the Wenatchee River basin and summer Chinook were released in the mainstem Wenatchee and Entiat rivers (Fish and Hanavan 1948). During this period, construction of the Leavenworth, Entiat, and Winthrop National Fish Hatcheries was completed and adult fish were first held in the Leavenworth National Fish Hatchery in 1940. In-river racks blocked adult salmon released into Nason Creek and the Entiat River from leaving the tributaries. Fisheries managers also collected some brood stock from the lower Columbia River (Mullan 1987). The upriver stocks were mixed together in the GCFMP hatchery program, and all UCR spring Chinook stocks have been affected by the outplanting of hatchery fish from a variety of in- and out-of-basin stocks. The history of human decimation of salmon populations, mixing of stocks, and hatchery practices likely weakened or homogenized the prehistoric adaptations of Chinook salmon to the local physical, environmental, and biological characteristics of the UCR basins. Recent analysis of prehistoric and contemporary mitochondrial DNA variation in Chinook salmon found reduced genetic diversity from the ancient to contemporary period and greater loss of genetic diversity in the Upper Columbia than in the Snake River (Johnson et al. 2018). The hydrosystem now creates significant migratory barriers with seven major dams between the mouth of the Columbia River and the mouth of the Wenatchee River, eight before the Entiat, and nine before the Methow and Okanogan. Currently, there is no anadromous fish passage above Chief Joseph Dam, located on the Columbia River 12 miles upstream of the Okanogan River confluence.

The biology, ecology, and migratory behavior of UCR spring Chinook salmon present challenges for their survival, responses to restoration actions, and rate of recovery (see detailed discussion in sections III.2 and III.3). UCR spring Chinook are stream-type Chinook, spending one year in freshwater before migrating to the ocean. Adult UCR spring Chinook return to their natal streams to spawn in August and September. This makes them vulnerable to low water, warm temperatures, and harvest in the mainstem Columbia River and UCR tributaries. Young fish emerge from the gravel in spring (April-May), and they either rear in their natal stream or migrate to lower reaches or mainstem Columbia River. Water withdrawal, mining, logging, agriculture, road building, and residential and urban development in the subbasins has affected habitat quality. Cold temperatures and low food productivity of headwater streams in these subbasins further limits survival and growth. Juvenile Chinook in these basins spend their first summer and winter in the natal streams or lower river and migrate to the ocean through the hydrosystem during their second spring, though some may migrate to the Columbia River as subyearlings in their first summer or fall. Yearlings migrate rapidly downstream through the hydrosystem, lower river and estuary (channel habitats), and northward to summer rearing habitats (mid-shelf) in the northern Gulf of Alaska. We know much less about subsequent oceanic distributions and migration patterns. UCR spring Chinook and similar fish (e.g., Carson stock releases from Leavenworth Hatchery) are coded-wire tagged and are usually not recovered in ocean salmon fisheries. Precocious spring Chinook (mostly males, referred to as jacks) spend only one year in the ocean, returning to spawn at age 3. Adult salmon spend 2 to 3 years in the ocean and return to the Columbia River in March or April, primarily at ages 4 and 5. Ocean conditions have been correlated to variation in abundance of returning adult salmon. Early entry into the Columbia River potentially makes spring Chinook more vulnerable to pinniped predators in the lower river than later arriving stocks. The long migration route from and returning to the UCR decreases the likelihood of survival for both juvenile and adult spring Chinook salmon.

1.3. Perspectives of Recovery

The Oversight Panel asked the ISAB to examine the alignment of management actions in all sectors (all H's) with recovery efforts for UCR spring Chinook salmon. The term "recovery" is used frequently to express different and often confusing perspectives. Concepts of recovery are important contexts for evaluating responses of threatened or endangered salmon and steelhead to management and restoration actions (Liss et al. 2006 in Return to the River, Lichatowich et al. 2017). Treaty obligations to U.S. Indian tribes establish criteria for providing access to traditional fishing locations and adequate abundances to meet their traditional uses (Fryer et al. 2016). Fisheries managers of the Columbia River Basin also have legal responsibilities to implement actions to recover viable salmon populations under the Endangered Species Act and are required to mitigate for resource values lost (Upper Columbia River Umbrella Program ISRP 2017-2).

Though ESA-related requirements and Tribal obligations are the most immediate concerns of recovery, other perspectives of recovery also are relevant in management decisions. Abundance targets for delisting all salmon and steelhead in the Columbia Basin alone (delisting requires achieving other criteria as well) amount to 234,000 adult spawners of natural origin (compiled from www.nwcouncil.org/ext/maps/AFObjPrograms/). Even if these targets for ESA-listed stocks are achieved, abundances of anadromous salmonids might still be substantially less than historical numbers of approximately 5 to 9 million fish[2] (p. 53 in ISAB 2015-1) and the Northwest Power and Conservation Council's goal of 5 million fish (NPCC 2014). The UCSRB Recovery Plan (2007) is designed to "improve chances of meeting recovery goals and objectives

[2]The Fish and Wildlife Program includes an upper bound of 10 to 16 million returning adult salmon and steelhead for historical abundances (NPCC 1986). Higher estimates are based on summing the peak five-year average annual catches for individual species rather than the peak five-year average annual catches for all species combined.

while achieving sustainable, harvestable sport, commercial and cultural fisheries," a goal broader than simply delisting endangered salmon or meeting obligations for tribal fisheries.

Different interests and perspectives create a complicated and sometimes divisive discussion of regional management actions for recovery (Williams et al. 2006, UCSRB Recovery Plan 2007, NPCC 2014, Lackey 2017). Laws and management programs have evolved in an imperfect attempt to reflect society's preferences and values, but they also reflect the tradeoffs and conflicts that arise between competing interests. Resource managers are responsible for making decisions on behalf of society to achieve those outcomes. In evaluating recovery efforts and analyses for UCR spring Chinook salmon, we recognize not only the legal requirements of delisting, mitigation, and tribal obligations, but also the broader recovery goals of the partners working to restore salmon populations in the UCR.

2. Review Process

2.1. Sources of Information

The methods used by the ISAB to provide the requested independent scientific advice and recommendations followed the ISAB's formal review procedures. Materials reviewed by the ISAB primarily included published scientific journal articles and unpublished reports (grey literature) by government agencies and other entities working in the Basin. ISAB members were familiar with the foundational scientific literature on spring Chinook salmon in the Columbia Basin. The ISAB also used computer search engines to find additional relevant sources of information from academic journals and the internet.

The Fish and Wildlife Program is based on recommendations from the region's federal and state fish and wildlife agencies, appropriate Indian tribes, and other interested parties aimed at protecting, mitigating, and enhancing fish and wildlife species and their habitat that are affected by the hydrosystem. Many Upper Columbia fish and wildlife management and monitoring projects and analyses are supported through the Fish and Wildlife Program and reviewed by the ISAB and ISRP for their scientific merit. These efforts include the Umbrella Habitat Restoration Projects, Wildlife Habitat Mitigation Programs, Comparative Survival Studies (CSS) of the Fish Passage Center, Life-Cycle Monitoring Program (LCMP), and regional habitat restoration effectiveness monitoring programs (CHaMP, ISEMP, AEM). Recent reviews of programs in the UCR include the Upper Columbia River Umbrella Program (ISRP 2017-2), Upper Columbia River Wildlife Habitat Mitigation Program, life-cycle models for the Wenatchee and Entiat Rivers (ISAB 2017-1), and Comparative Survival Study's draft annual report (ISAB 2017-2). In addition, several recent science reviews examined major issues related to salmon and steelhead management in the UCR, including Climate Change (ISAB 2007-2), Human Population Impacts (2007-3), Food Webs (ISAB 2011-1), Review of the 2009 FWP (ISAB 2013-1), Density Dependence (ISAB 2015-1), Critical Uncertainties (ISAB/ISRP 2016-1), and Predation (ISAB 2016-1). This review draws on these findings in assessing the effectiveness of management actions to restore spring Chinook populations in the UCR.

2.2. Field Tours and Presentations at ISAB Meetings

The ISAB was briefed on the current status of UCR spring Chinook salmon populations, recent findings from research, monitoring, and evaluation (RME), and management actions in the three subbasins. The ISAB conducted a field tour of the Wenatchee and Entiat river basins, led by local biologists and program leaders on July 19-21. The ISAB followed the field review with a series of three 1-day meetings in Portland, Oregon on September 15, October 27, and December 8, 2017 (see Appendix A for a list of meetings and links to presentations). This report's Acknowledgments section identifies the many individuals and organizations who provided materials, briefings, and site tours to inform this review.

3. Assessment of Upper Columbia River Spring Chinook

Questions submitted to ISAB:

Is the identification of limiting factors for Upper Columbia River spring Chinook based on sound scientific principles and methods? Are the most important survival bottlenecks or factors limiting this ESU's recovery identified? Where and when do the most important limiting factors occur? Is density dependence considered? Are the necessary data available to identify the limiting factors? Are assumptions, data gaps, and key uncertainties identified?

a. Based on recent status reviews and other relevant assessments, are Snake River spring Chinook doing better than Upper Columbia spring Chinook, in terms of abundance, diversity, spatial structure, and productivity? If so, do we know why? Do limiting factors and life histories differ between Snake River and Upper Columbia spring Chinook? For example, are there key limiting factors for Upper Columbia spring Chinook upstream of Priest Rapids dam?

b. Pinniped predation appears to be increasing rapidly in the lower Columbia River. Are pinnipeds potentially a significant source of mortality for Upper Columbia spring Chinook? Can the effect of this predation on Upper Columbia spring Chinook be quantified?

3.1. Identification of Limiting Factors

One of the first steps in development of a recovery plan for a population is identification of potential factors that limit the population (Booth et al. 2016). The ISAB considers limiting factors to be environmental characteristics, ecological processes, or anthropogenic structures or actions that constrain a population's diversity, productivity, and abundance. Limiting factors can be assessed for specific life stages, but from a recovery perspective, limiting factors are those that influence the full life cycle and number of reproductive adults. Recent authors and assessments also have distinguished ecological concerns from limiting factors (Hamm 2012). Ecological concerns include abiotic features (e.g., temperature, oxygen, physical habitat) or biotic interactions (e.g., competition, predation, disease, prey availability). Ecological concerns may affect productivity and capacity, but they do not necessarily limit abundance. The distinction is valid but blurred in practice because many factors listed in limiting factors analyses are ecological concerns but they are called limiting factors. Most limiting factors analyses identify important habitat characteristics (Reeves et al. 1989, Mobrand Biometrics 1997) and attempt to determine how much positive change is possible in habitat factors (or characteristics) that are limiting. Traditional identification of limiting factors integrates a diverse array of habitat data, information on fish populations and habitat relationships, and professional opinion. More recent habitat assessments have focused at larger scale processes that shape local habitat conditions (Beechie et al. 2010, Booth et al. 2016) or weight the limiting factors by their geographic extent and influence on fish survival (UCSRB 2014b). Evidence related to adult returns and population dynamics often is scarce, and the analysis of habitat is restricted to the freshwater portion of their range, often only their natal stream basins. Recent studies of density dependence and life-cycle modeling provide more robust methods for quantitatively and rigorously defining limiting factors.

3.1.1. Potential Limiting Factors and Threats in UCR Tributary Basins

Subbasin Plans

Early analyses of limiting factors primarily identified environmental or local habitat conditions that potentially influence fish abundance by juvenile and adult fish in the freshwater environment. The Washington State Conservation Commission conducted analyses of habitat limiting factors for UCR spring Chinook in the Entiat (Andonaegui 1999) and Wenatchee basins (Andonaegui 2001), providing the basis for the Subbasin Plans for the Entiat and Wenatchee basins. The Methow Subbasin Plan (2004) relied on an unpublished report of a habitat limiting factors analysis by Williams (2000). All of these assessments identified six major categories of limiting factors:

1. Access to spawning and rearing habitat
2. Riparian Condition
3. Channel Conditions/Dynamics
4. Habitat Elements
5. Water Quality
6. Water Quantity

Sixteen additional attributes of habitat were assessed within these broader characteristics of limiting factors, including factors such as riparian and streambank conditions, floodplain connectivity, channel morphology and substrates, large wood, water temperature, fine sediments, and changes in flow regimes. Habitat attributes were assessed for smaller watersheds within the three subbasins, and the major potential limiting factors considered to be “Not Properly Functioning” differed by watershed. The major limiting factors identified in the Subbasin Plans were:

- Elevated Temperature
- Sediment
- Riparian/Floodplain Function
- Habitat Diversity
- Obstructions and Barriers
- Channel Stability
- Flow
- Competition (hatchery fish and non-native fish)

Basin planners identified objectives and management actions for each subbasin and developed methods for monitoring and evaluating the program. However, the actions to address limiting factors in the Subbasin Plans were largely limited to in-basin habitat and did not prioritize the actions. In the limiting factors analysis for the Wenatchee River, the Technical Advisory Group for the Wenatchee Subbasin Plan ranked their recommendations for habitat actions:

1. Maintain highly functional habitat in Wenatchee subbasin watersheds.
2. Maintain and restore habitat on the mainstem Wenatchee River
3. Restore ecosystem functions and connectivity within the Wenatchee subbasin
4. Evaluate the relationship between stream flows and water use in the subbasin
5. Increase instream low-flows negatively impacted by human activities

Weighted Analysis of Limiting Factors

After development of the UCR subbasin plans, several efforts attempted to improve the use of habitat information and assessments of limiting factors by weighting factors based on survival benefits for Chinook salmon and steelhead and geographic extent of the habitat condition. Action Agencies (AAs)—U.S. Army Corps of Engineers, Bonneville Power Administration, and Bureau of Reclamation—of the Federal Columbia River Power System (FCRPS) formed regional expert panels after the court-ordered remand of the 2008 BiOp to estimate changes in habitat quality improvements (HQIs) to address limiting factors for salmon and steelhead. The panels examined past habitat improvements (look back analysis) and anticipated future improvements (look forward analysis).

The U.S. Bureau of Reclamation attempted to assess the relative contribution of limiting factors across the Wenatchee, Entiat, and Methow basins by accounting for the geographic extent and spring Chinook populations in the three basins, based largely on professional judgment. The Remand Habitat Workgroup developed a method for estimating habitat quality and egg-to-smolt survival benefits based on professional judgment of local biologists (see www.usbr.gov/pn/fcrps/habitat/panels/reference/1B-CA-AppC.pdf for a description of the methods and analyses). Within each subbasin, regional experts identified limiting factors for assessment units, which were weighted for their proportional contribution to the survival of spring Chinook (Figure 3.1).[3]

Major factors related to habitat impairment in all three basins included riparian vegetation, wetlands, floodplains, and instream structural complexity, which have been major limiting factors in previous analyses. Sediment quantity was a substantial factor in the Methow and Entiat basins, and decreased water quantity was a factor in the Methow. Such approaches to integrating assessments of potential habitat limiting factors may be useful for future planning in the UCR, but the analysis may be biased by the reference contexts (e.g., current landscapes, past landscapes, plausible future landscapes) and reliance on collective professional judgment rather than empirical analysis. In spite of these limitations, the assessment of the relative importance of habitat conditions by the Remand Habitat Workgroup was consistent with previous assessments of limiting factors in these basins.

[3] Habitat quality was calculated by summing the limiting factor scores multiplied by the respective limiting factor weights. Habitat quality at the population level was estimated as the sum of the scores for the assessment unit multiplied by the weighting factor for the assessment unit. Habitat impairment was estimated as one minus the habitat quality score for the limiting factor. The sum of the fractional habitat impairment for all limiting factors in all three basins adds up to a value of one.

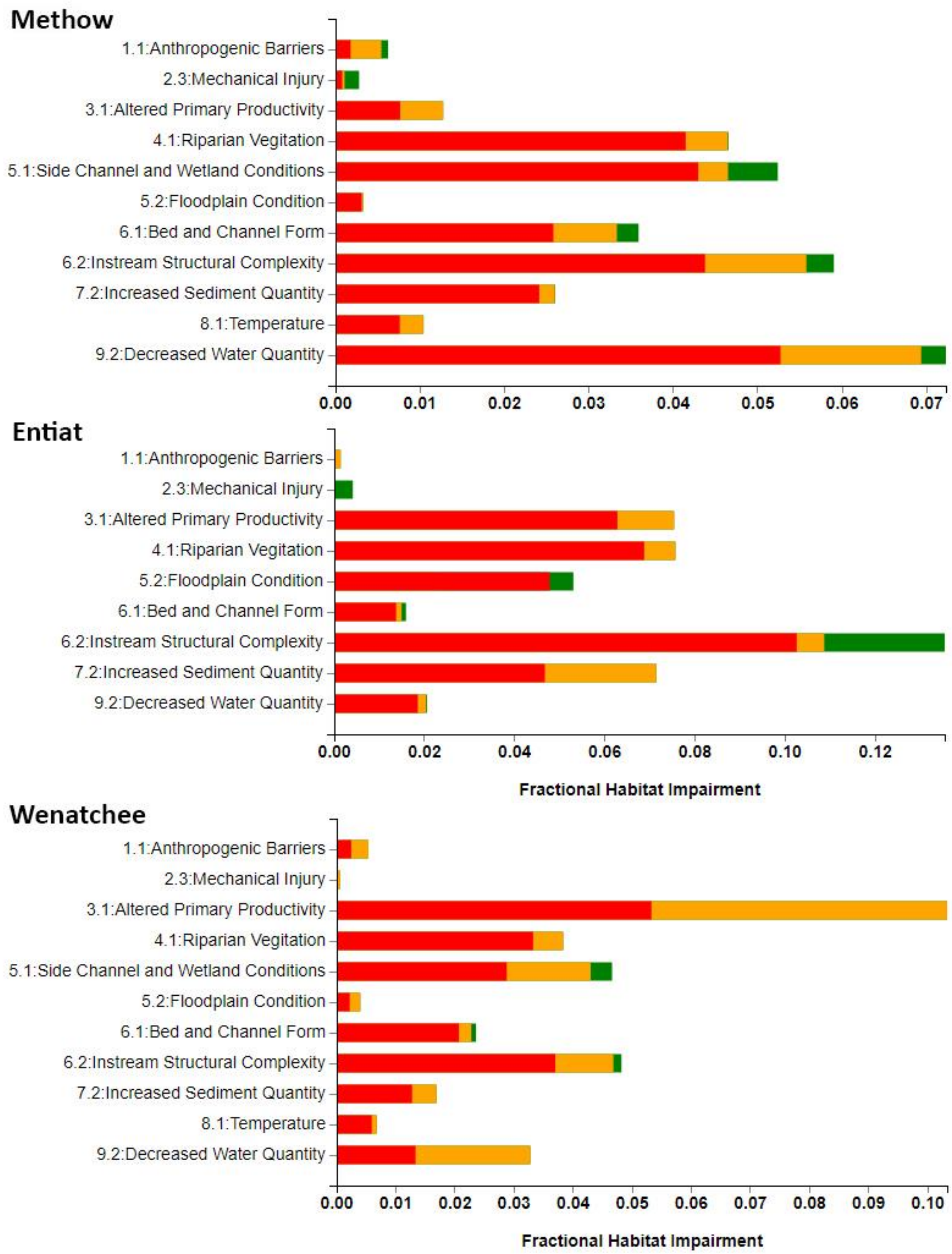


Figure 3.1. Estimates of relative contribution to habitat impairment as a limiting factor for the Upper Columbia River spring Chinook salmon (U.S. Bureau of Reclamation; illustrated in www.onefishtwofish.net/viz/HabitatLimitingFactors1d.html). Yellow represents the estimated improvements of restoration actions from 2009-2016, green represents anticipated improvement through 2018, and red represents remaining impairment expected after 2018. Note that the categories for floodplains and side channels and wetlands can be combined to represent overall floodplain features.

The UCSRB created a working group of the members of the 2012 FRCPS Expert Panel to refine the habitat analyses of limiting factors for the Upper Columbia (UCSRB 2014b). The working group ranked the ecological concerns (limiting factors weighted by assessment unit and expected survival benefit) for each basin in the Upper Columbia (Table 3.1). Instream structural complexity, riparian condition, bed and channel form, and increased sediment were ranked highest for the four basins (including the Okanogan basin). These rankings were generally consistent with the original limiting factors identified in the subbasin plans, but the approach to weight limiting factors for their perceived survival benefit and assessment unit geographic extent provided a more informative basis for designing recovery actions.

Interestingly, floodplain condition was ranked as a low ecological concern in contrast to other assessment and recent restoration priorities in the UCR. This discrepancy may reflect the evolution of geomorphic perspectives and habitat focus of regional scientists and practitioners. In the late 1990s, attention was focused on instream habitat structure, with little emphasis on the processes that shape instream habitat and the importance of floodplains as both determinants of geomorphic structure and critical habitat and sources of food resources. The recent assessments by the Expert Panels included ecological concern categories for floodplain as well as side channels and wetlands, but the combination of these features more completely represents floodplains in the landscape. The USCRB's more recent emphasis on floodplains as areas of higher ecological concern reflects the gradual recognition of the role of floodplains for aquatic ecosystems over the last 30 years.

Table 3.1. Rankings of ecological concerns across the four basins of the Upper Columbia River. Average numerical scores of the panel members were categorized as high, moderate, low, and minimal (UCSRB 2014b)

Ecological Concern	Total	Wenatchee	Entiat	Methow	Okanogan
Instream Structural Complexity	High	Moderate	Low	High	High
Riparian Condition	High	Moderate	Low	High	High
Bed and Channel Form	High	Low	Low	Moderate	High
Increased Sediment Quantity	High	Low	Low	Moderate	High
Anthropogenic Barriers	Moderate	Moderate	Low	Moderate	Moderate
Decreased Water Quantity	Moderate	Low	Low	Moderate	High
Temperature	Moderate	Low	Minimal	Low	High
Side Channel and Wetland	Low	Low	Low	Moderate	Low
Food-Competition	Low	Minimal	Minimal	Minimal	High
Altered Primary Productivity	Low	Low	Low	Moderate	Minimal
Mechanical Injury	Low	Low	Minimal	Low	Moderate
Predation	Low	Minimal	Minimal	Minimal	Minimal
Floodplain Condition	Low	Low	Low	Low	Low

Rankings of ecological concerns were compared to habitat improvement actions from 1996 to 2012 for the four UCR basins to evaluate the programmatic alignment between recovery actions and assessment of major ecological concerns (UCSRB 2014b). Instream structure and floodplain restoration accounted for the greatest number of projects and is consistent with the ranking of ecological concerns (Fig 3.2). Riparian conditions were ranked as one of the highest ecological concerns but were less numerous than several other ecological concerns. The attempt of the UCSRB to align their recovery actions with their analysis of limiting factors is useful, but it can be misleading because the different units of measure for different actions (e.g., miles, acres, number of structures added or removed) make comparison difficult (Figure 3.3).

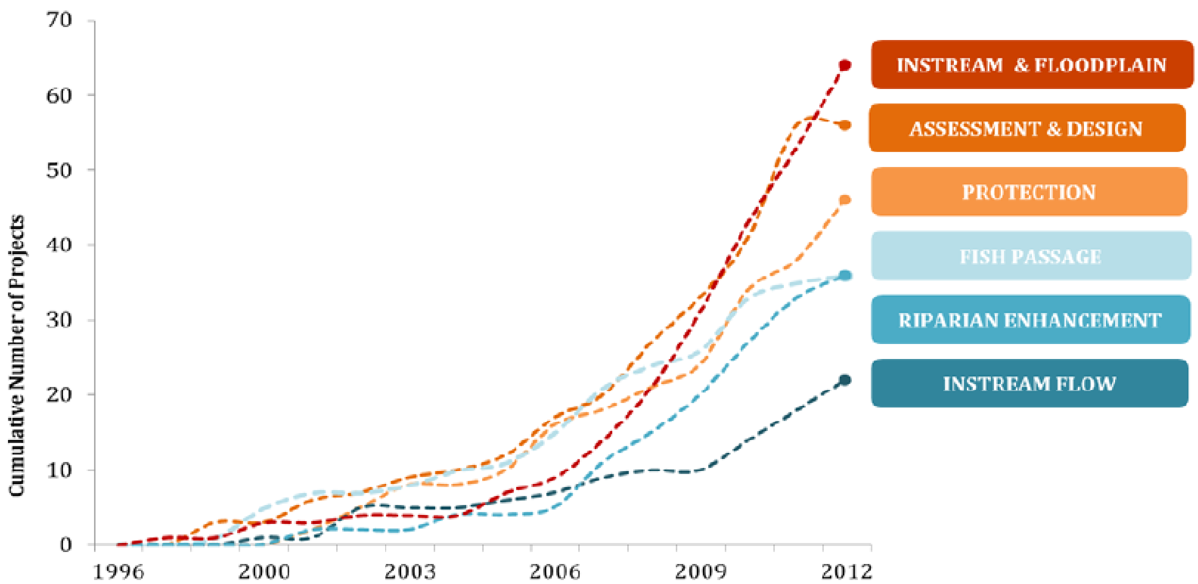


Figure 3.2. Cumulative number of habitat projects implemented in the Upper Columbia region between 1996-2012 by project type (UCSRB 2014b)

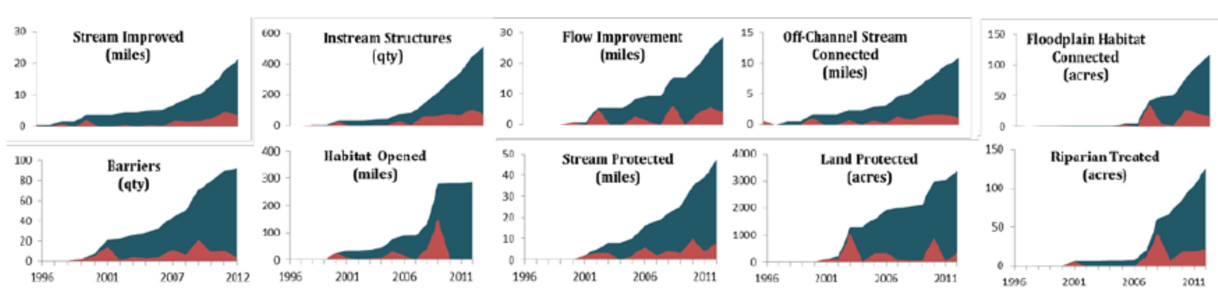


Figure 3.3. Assessment of the quantity (miles, acres, numbers) of recovery actions in the UCR between 1996-2012 by year (red) and cumulatively (blue) (UCSRB 2014b)

Process-based Analysis

Limiting factors analyses most often identify local habitat conditions and rank their potential relative influence on fish populations, based primarily on professional judgment. The narrow focus on local in-channel characteristics overlooks fundamental landscape processes that shape freshwater habitats (ISAB 2011-4). A recent alternative approach focuses on landscape characteristics at larger scales that are responsible to local habitat conditions (Beechie et al. 2010, Booth et al. 2016, Roni et al. 2017). One advantage of process-based approaches is that they identify causal factors that create habitat features rather than the outcomes of habitat change. As a result, they potentially inform the design and prioritization of habitat restoration action more effectively than assessments of local habitat conditions. A second advantage is they rely on larger scale information and can use remotely sensed information rather than detailed field measurements, making them more cost effective. The process-based approach has been initiated in a small section of the Methow Basin, but the assessment is not completed (Tim Beechie, personal communication).

Integration with Limiting Factors Analysis for Other Fish Species

One of the challenges of synthesizing past reports on limiting factors is the diverse and non-standardized names and groupings of limiting factors used by different agencies and assessment groups. The Subbasins/Species Dashboard of the NPCC organized the diverse limiting factors assessments from the subbasin plans into 22 impairment categories that are further condensed into 7 major categories of limiting factors. This approach is similar to NOAA's 34 impairment categories grouped into 10 major categories of ecological concerns. The Dashboard graphically illustrates the top six major limiting factors for the three basins:

- Entiat
- Methow
- Wenatchee

Assessments of habitat in the three UCR subbasin plans have identified relatively similar limiting factors. Limiting factors related to instantaneous mortality (e.g., migration impediments, hatchery effects, harvest, predation, non-native species) were frequently identified in all three basins (Figure 3.4). Habitat quantity and quality were the most numerous type of limiting factor in the Wenatchee and Entiat basins, but this limiting factor was relatively minor in the Methow basin. Sediment conditions were a greater concern in the Methow basin than in the other two basins. While aggregation of different types of limiting factors makes the subbasin assessments consistent with NOAA limiting factors and provides a simpler basis for comparing basins, it has the disadvantage of obscuring details of limiting factors for individual species and types of ecological concerns.

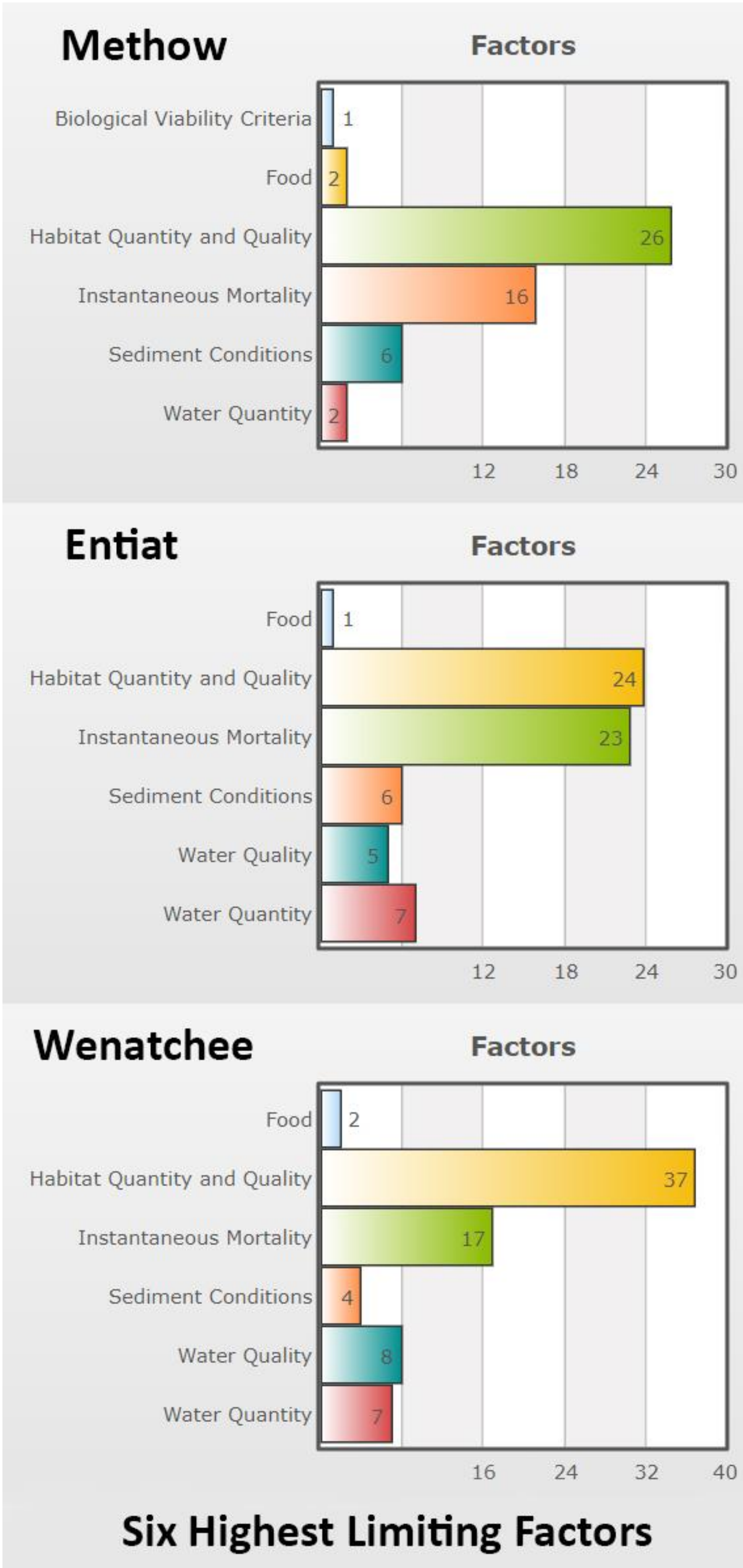


Figure 3.4. Number of limiting factors identified for all fish species of concern in the Wenatchee, Entiat, and Methow Subbasin Plans (www.nwcouncil.org/ext/dashboard)

3.1.2. Consideration of Density Dependence in Analysis of Limiting Factors

The ISAB recently reviewed evidence for density dependence in salmon and steelhead populations in the Columbia Basin (ISAB 2015-1). Density dependence is related to the influence of limiting factors because it represents the dynamics of a population in which changes in fish density affect vital rates (e.g., birth rate, death rate, immigration, or emigration) and result in changes in the growth rate of a population. Density dependence also can alter the behavior, growth, body size and age at maturation, survival, or fecundity of individual fish. If density dependence is occurring in a population, physical, environmental, or ecological factors may limit population abundance by reducing the productive capacity of the habitat.

Analysis of density dependence in spring Chinook salmon populations revealed differing degrees of density dependence in the Wenatchee, Entiat, and Methow Basins (Zabel and Cooney 2013, Murdoch et al. 2011, Hillman et al. 2017). Productivity (adult returns per spawner) for Chinook salmon in the three rivers was lower at moderate spawning abundances (Figure 3.5). The relationship between spawners and adult returns was not significant for the Wenatchee ($p=0.18$). Analysis of productivity for the Chiwawa River of the Wenatchee showed density dependence for spring Chinook (presentation to ISAB by Pearsons and Graf). Though the relationship was statistically significant for the Methow, the relationship is strongly influenced by a single observation at very high numbers of adult returns. Evidence for density dependence in the Methow is less compelling without this single observation. Of the three major basins, only the Entiat exhibited strong and unambiguous density dependence. Within these basins, Chinook populations in smaller watersheds, such as the Chiwawa and Twisp, exhibited density dependence.

Recent analysis of the relationship between outmigrating smolts and adult returns in the Methow subbasin revealed significant density dependence in spring Chinook in the Twisp River but not in the Methow subbasin as a whole (U.S. Bureau of Reclamation Unpublished Data). This analysis of smolts per spawner suggests that freshwater productivity for spring Chinook may be limited by rearing habitat or food in some tributaries but not in the Methow subbasin overall. This analysis of smolts per spawner focuses on the productivity of the juvenile portion of their life cycle in their natal streams. In contrast, the analysis by Zabel and Cooney (2013) was based on adult returns per spawner, reflecting the full life cycle and influence of factors within and beyond the Methow subbasin. In combination, both analyses increase our understanding of the relative influences of the freshwater conditions within the subbasin and possible limitations in the mainstem Columbia, estuary, and ocean.

Further analysis of limiting factors is warranted for UCR spring Chinook salmon, but simply listing potential limiting factors and eliciting professional opinions will not provide an accurate or even relative basis for designing and ranking restoration actions in a recovery plan.

Moreover, a limiting factor at one life stage may not limit the abundance of the population at a subsequent life stage. Analysis must include the full life cycle of the population and assess the effects of physical, environmental, ecological, and anthropogenic factors on adult spawners across multiple generations. This highlights the importance of developing better life-cycle models to assess limiting factors and guide research, monitoring, and evaluation (RME), management decisions, and policy development.

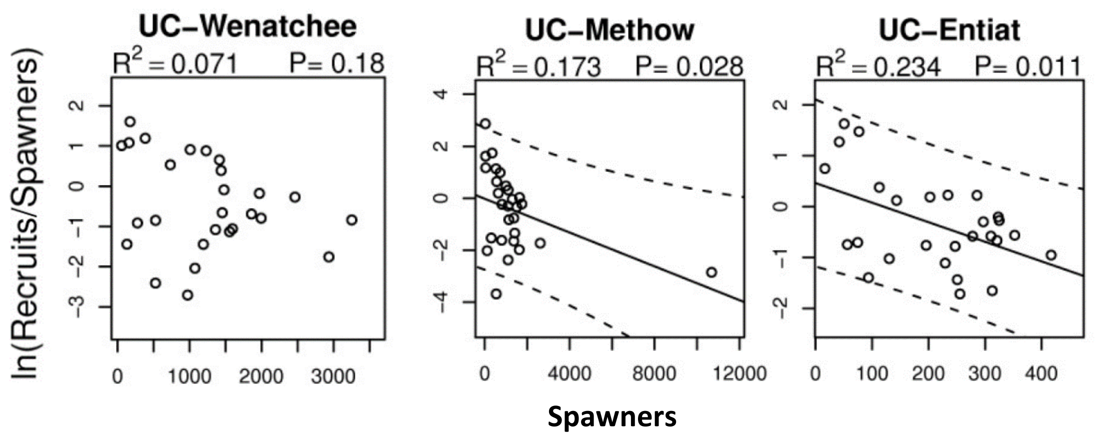


Figure 3.5. Evidence for density dependence in Upper Columbia River spring and summer Chinook populations, brood years 1980 to ~2005 (from Zabel and Cooney 2013). Relationships based on the linearized form of the Ricker model. Recruitment includes ocean and in-river harvests. Dashed lines represent 95% prediction intervals for a specified number of spawners when regression was statistically significant ($P < 0.05$). Values less than log[R/S] < 0 indicate recruits are not replacing spawners.

3.1.3. Use of Life-Cycle Models to Identify Limiting Factors

Several life-cycle models have been developed for salmon and steelhead in the Columbia River Basin. These models provide an approach for assessing limiting factors for the complete life cycle of salmon and steelhead based on empirical data from research and monitoring, published literature, and best available science. Figure 3.6 illustrates an example of the overall structure of the life-cycle models. The ISAB recently reviewed life-cycle models, including those of the Wenatchee, Entiat, and Methow rivers (ISAB 2017-1). Though these models are still relatively early in their development, they demonstrate several critical outcomes that rigorously identify limiting factors for UCR spring Chinook salmon. Life-cycle models also provide the ability to examine the sensitivity of the model outputs to specific relationships or parameters within the models. In particular, the life-cycle models for the Wenatchee and Entiat river basins provide analyses of potential limiting factors for spring Chinook salmon and the potential responses to restoration actions.

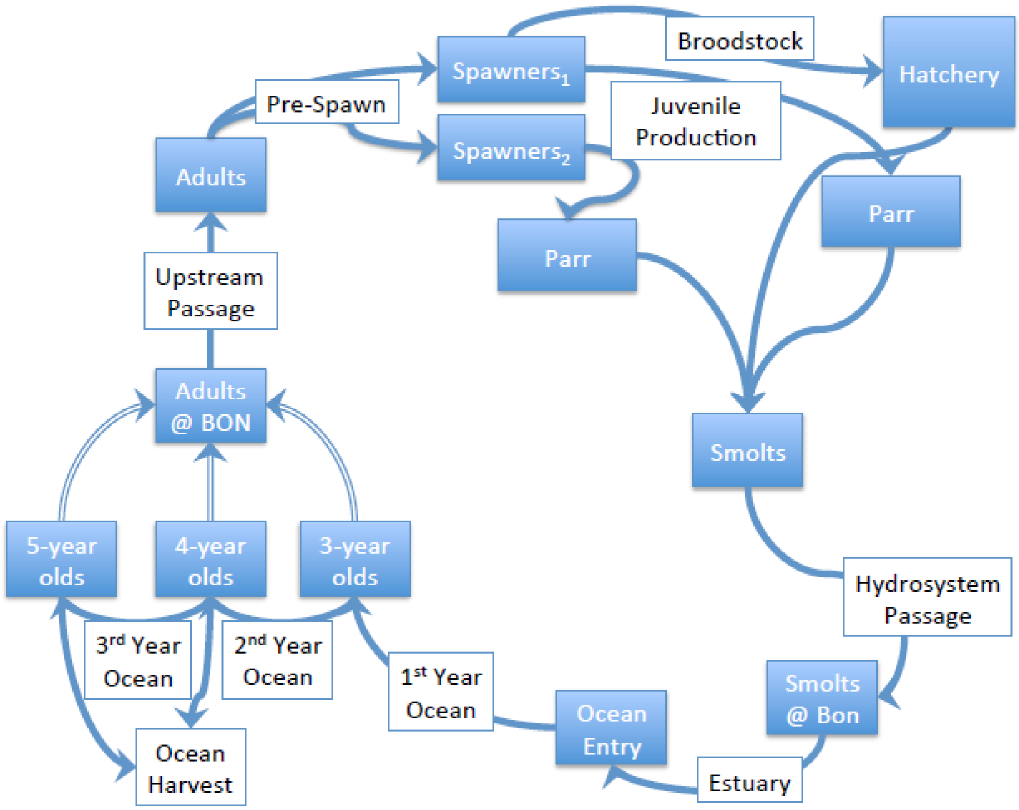


Figure 3.6. Diagram of the life-cycle model for spring Chinook salmon in the Wenatchee subbasin. Life-cycle models for spring Chinook in the Upper Columbia River Basin are structured similarly to represent the different life-cycle stages and freshwater, estuary, and ocean habitats (Jorgensen et al. 2017)

Wenatchee

NOAA Fisheries has developed a full life-cycle model for the Wenatchee population of spring Chinook (Jorgensen et al. 2017). The model evaluated relative effects of hatchery operations, hydrosystem operations, harvest, freshwater rearing habitat, pinniped predation, ocean conditions, and climate change. Scenarios were examined for current conditions for these major factors and compared with scenarios for improved habitat, reduced harvest, reduced hatchery effects, increased spill in the hydrosystem, lower pinniped predation, and bad ocean conditions. In general, hatchery practices, pinniped predation, and ocean conditions had the greatest influences on quasi-extinction probabilities (Figure 3.7). In the Wenatchee Life Cycle Model, outputs were most sensitive to ocean survival after the first year, parr-to-smolt survival, and adult-upstream survival through the hydrosystem. These analyses allow fisheries managers to identify critical life history stages and functional relationships for future RME and model refinement.

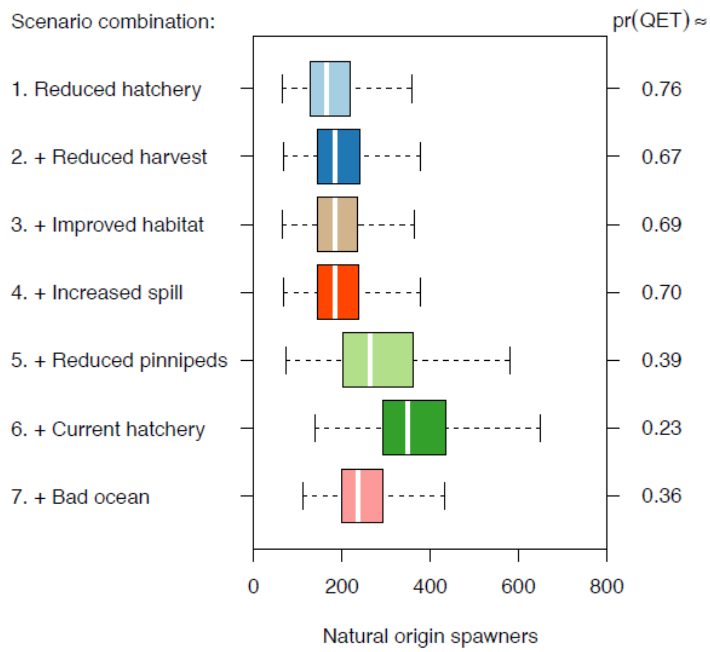


Figure 3.7. Population responses to cumulative effects of scenarios of management actions, measured by median spawner abundance and by estimates of extinction risk (Jorgensen et al. 2017). QET represents Quasi-Extinction Threshold, an operational definition of abundance that represents extinction.

A full life-cycle model also has been used to examine the potential influences of habitat limiting factors in the Wenatchee Basin related to the 2007 UCR Recovery Plan (Honea et al. 2009). The model indicated that restoration actions (Table 3.2) potentially would increase numbers of spawners, but the increases would be relatively small compared to numbers of spawners projected for the scenario of historical conditions (Figure 3.8). Abundance of spawners decreased without restoration actions, but spawner numbers decreased far more with accelerated habitat degradation relative to current conditions. The model also provided estimates of the sensitivity of the model outputs to different habitat components. Numbers of smolts and spawners were most sensitive to fine sediments in the gravel and temperature during incubation (Table 3.2). The model outputs were less sensitive to fry and adult capacity, water temperature during other life stages, and coarse sediments. These findings indicate that restoration actions represented in the scenario may have relatively small influences on populations of UCR spring Chinook. However, the scenario in the model did not include the full range of habitat restoration actions being implemented in the Wenatchee Basin.

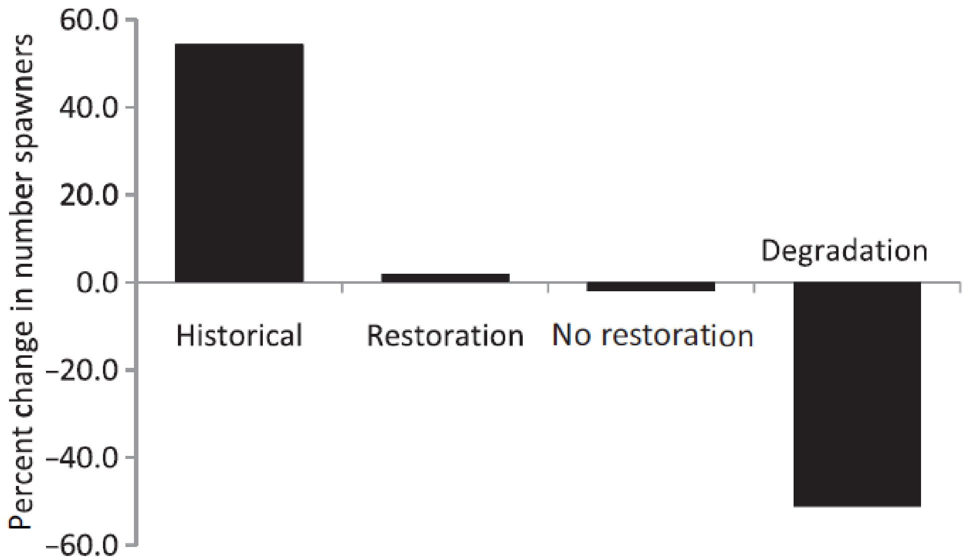


Figure 3.8. Percent change in annual spawners relative to current conditions for scenarios of historical conditions, restoration actions, no restoration, and increased habitat degradation in the Wenatchee Basin (Honea et al. 2009)

Table 3.2. Change in the number of smolts and spawners relative to current scenario from improvement and degradation of individual habitat variables with other variables held at estimated current values (Honea et al. 2009)

	Improved (%)		Degraded (%)	
Habitat variable	Smolts	Spawners	Smolts	Spawners
Fine sediment	161*	75*	–29*	–12*
Incubation water temperature	1	0	–17*	–10*
Spawner capacity	2	5*	–1	–1
Fry capacity	3	1	0	–1
Spawner water temperature	0	0	–2	–3
Fry water temperature	0	–1	–1	–1
Cobble and boulder in pools	0	0	–1	–1

*Significantly different from result of current scenario with t-tests, $\alpha = 0.05$ and Bonferroni adjustment for multiple comparisons.

The life-cycle model for the Wenatchee also was used to explore the influence of climate change on spring Chinook salmon and their responses to habitat variables (Honea at al. 2016). The model estimated that spawner abundances would decrease by 4 to 7% in response to regional warming estimated by global climate models. Projected spawner abundances were most sensitive to temperatures during spawning (August and September). Increased streambed scour related to increases in flooding also had a significant negative effect on spawner abundances. The model indicated that fine sediments had the greatest influence on spring Chinook populations, but potential regional warming and changes in stream discharge may also influence their recovery.

Entiat

NOAA Fisheries developed a life-cycle model for the Entiat River basin as well (Saunders et al. 2017). The model was used to examine scenarios of 1) current conditions, 2) current and proposed habitat improvement actions (side-channel habitat creation and addition of large wood and boulders), and 3) habitat improvement and 2% increase in juvenile Chinook salmon survival (based on influence of habitat structures on overwinter survival). The model projected that habitat improvement would only modestly increase spawner abundance compared to current conditions, averaging only 7% greater numbers. Habitat improvements plus additional survival related to over-wintering habitat increased spawner abundances slightly more, but none of the scenarios met recovery goals for spawners over a 100-year period and all scenarios predicted a high risk of extinction for UCR spring Chinook salmon.

The life-cycle models provide powerful tools for evaluating the potential success of the Upper Columbia River Recovery Plan and the restoration actions designed to recover spring Chinook salmon. All models for UCR spring Chinook projected only small increases in spawner abundances in response to management actions, though the full range of habitat-related recovery actions has not been included in the models. These projections are consistent with the lack of change in the risk of extinction over the last 10 years (NMFS 2016). Habitat degradation, hatchery influences, hydrosystem modifications, and harvest in the lower Columbia River have affected these populations for more than 150 years. The slight improvements in adult spawner populations projected by the life-cycle models are consistent with the lack of recovery of UCR spring Chinook salmon observed since the Recovery Plan was developed in 2007.

3.1.4. Influence of Factors Other Than UCR Habitat

The Upper Columbia Spring Chinook Salmon and Steelhead Recovery Plan (2007) developed by the Upper Columbia Salmon Recovery Board (UCSRB) and National Marine Fisheries Service (NOAA Fisheries) expanded the scope of potential limiting factors and prioritized potential restoration actions. Rather than focusing primarily on in-river habitat, identification of limiting factors in the Recovery Plan was based on factors related to criteria for Viable Salmonid Populations (McElhaney et al. 2000), including harvest, hydrosystem effects, hatchery effects, and habitat degradation. While the Plan recognizes the influence of all Hs, analysis of limiting factors has not integrated the four Hs to determine which have the greatest influence on spring Chinook populations in the Upper Columbia. Life-cycle models for these subbasins are beginning to provide evidence that addresses all Hs, but they are in early stages of development.

Hatchery practices have had major influences on UCR spring Chinook salmon (see Chapter 5.4.2.2). Though hatchery management practices have improved greatly and the Entiat and Leavenworth National Fish Hatcheries are operated as "segregated" programs (i.e., producing spring Chinook that are not part of the ESU), out-of-basin strays have introgressed with local populations and may have replaced some locally derived spring Chinook salmon populations (McElhaney et al. 2000). After 2008, the Entiat Hatchery no longer produced spring Chinook and now produces summer Chinook as part of a segregated harvest augmentation program. NOAA Fisheries concluded that state hatchery operations may not be fully coordinated and pose risks to natural origin fish.

3.1.4.1. Influence of the Hydrosystem on UCR spring Chinook Salmon

Hydrosystem development has had major impacts on the abundance, distribution, productivity, and diversity of UCR spring Chinook salmon populations, affecting both juvenile and adult life stages. The threats appear to be greatest on the juvenile and smolt life stages and are related to dam-related mortality and predation influenced by the reservoir environment.

The three populations of spring Chinook (Wenatchee, Entiat, Methow) migrate out and return via the mainstem Columbia River. The Wenatchee stock must pass through seven dams; the Entiat stock must pass eight dams; and the Methow stock must pass nine dams (Figure 1.1). Generally, the spring Chinook stocks return as spawners between May and August with a peak in May and June at Priest Rapids Dam. Adults hold in tributaries before spawning in August/September.

Fry emerge in early spring and the majority of juveniles spend a full year in tributaries. In spring, they emigrate as yearlings to the Columbia River and out to sea. Some juveniles spend an additional year in freshwater before emigration (NMFS 2016; summary taken from UCSRB integrated recovery report). Recent information presented to the ISAB by Desgroseillier indicated that a substantial proportion (about 50%) of Entiat spring Chinook juveniles emigrate as sub-yearlings and overwinter in the mainstem Columbia River. Nevertheless, most Upper Columbia spring Chinook are primarily exposed to the hydrosystem as yearlings emigrating to sea and as adults returning to spawn.

The Comparative Survival Study report (CSS 2017) presents a comprehensive analysis of the experience of many stocks as they migrate through the hydrosystem. About 30,000 juvenile fish/year (mixture of steelhead/Chinook and wild and hatchery) are tagged in the Entiat, Methow, Chiwawa, and Wenatchee tributaries. An additional 4000 spring Chinook juveniles are captured, PIT-tagged, and released at Rock Island Dam as part of the Smolt Monitoring Program, but the origins of these Chinook are not known.

The CSS (2017) examined travel times of Upper Columbia yearling Chinook (all stocks combined) as they traveled downstream from Rock Island Dam (RIS) to McNary Dam (MCN) and then to Bonneville Dam (BON). A regression model for fish travel time (FTT) was constructed using several environmental covariates (see Figure 3.9). Yearling spring Chinook took about 15 days to migrate from RIS to MCN and then another 15 days to migrate from MCN to BON. Julian day, water transit time (WTT), and temperature are important predictors of fish travel time for the RIS-MCN leg, but spill percentages and number of dams with spillway weirs was not. All these variables were important as predictors of fish travel time for the MCN-BON leg.

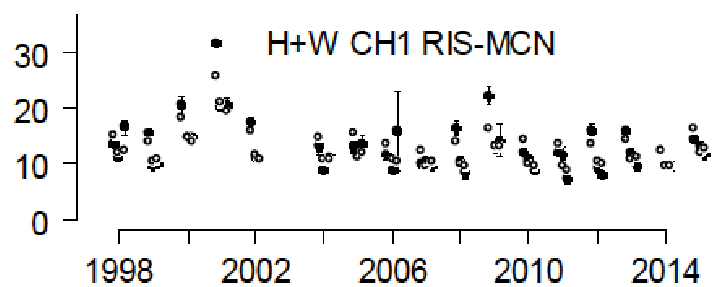
Similarly, models were developed for instantaneous mortality (Figure 3.10) and survival (Figure 3.11 and 3.12). The survival probability from RIS-MCN is about 0.60 and from MCN-BON is about 0.70 for an overall in-river survival of about 0.40 (RIS-BON). However, the models for fish travel time, survival, and instantaneous mortality for yearling hatchery/wild Chinook for the RIS-MCN segment (Figure 3.15) had the worst fit compared to the fit of the regression models for a) the Snake River Lower Granite Dam to MCN (LGR-MCN) for wild spring/summer Chinook, b) LGR-MCN for hatchery Chinook, or c) MCN-BON for hatchery/wild groups. This indicates considerable variation for the RIS-MCN segment has not been explained by the variables considered in their modelling exercises. More modelling work is needed to identify explanatory variables for this segment that can improve predictions, so they have similar predictive ability compared to other release groups and river segments.

Estimates of SARs for UCR stocks likely are overstated because dams upstream of MCN in the Upper Columbia do not have full detection capabilities for PIT-tagged juveniles, so mortality of UCR Chinook smolts that occurs upstream of MCN is not accounted for. CSS (2017) reported SARs both with and without jacks (1-salt male Chinook), and estimated SARs for juveniles detected at MCN that return as adults to Bonneville (MCN–BON) for wild spring Chinook from the Entiat/Methow River (2006–2014) and the Wenatchee River (2007–2014). Estimated SARs (MCN-BON including jacks) for Upper Columbia wild spring Chinook ranged from 0.54% to 3.26% in 2006-2014. However, estimated SARs based on estimated smolts at Rocky Reach Dam (RRE) were only about 58% of the SAR values estimated for MCN-BON, indicating substantial mortality between RRE and MCN.

The CSS (2017) also used smolts PIT-tagged at RIS by the Smolt Monitoring Program (SMP) to estimate SARs farther upriver. The SMP estimated survival from RIS to MCN for smolts captured, PIT-tagged and released at RIS. Survival was estimated in 2-week periods across several migration years and consistently indicated that a large mortality occurs from RIS to MCN (geometric mean survival is about 0.60). Estimated SARs from RIS to BON (including jacks) were consequently reduced by a similar factor as shown in Figure 3.13. Both analyses above indicate that a substantial portion of mortality occurs in the mainstem above MCN, but there is limited analysis in CSS (2017) to investigate why.

The CSS (2017) has not computed estimates for first year estuary survival (S.oa) or first year adult survival (S.o1) for Upper Columbia spring Chinook populations (p. 128 of CSS 2017). When completed, such analyses would be useful to determine if there are differential survival rates in the ocean for these stocks and better inform the life-cycle models. There also appears to be high-interregional correlation of SARs among wild/hatchery and spring/summer Chinook populations indicating common environmental factors are influencing survival rates from outmigration to the estuary and ocean environments.

The CSS (2017) started a new analysis on returning adult success rates, but no analysis has been done for Upper Columbia stocks. Consequently, it is difficult to understand where the major sources of mortality for adults occur. Analyses of survival for the first year in the estuary, first-year adult survival, and survival of adult spawners from the Upper Columbia stocks should be completed as soon as possible.



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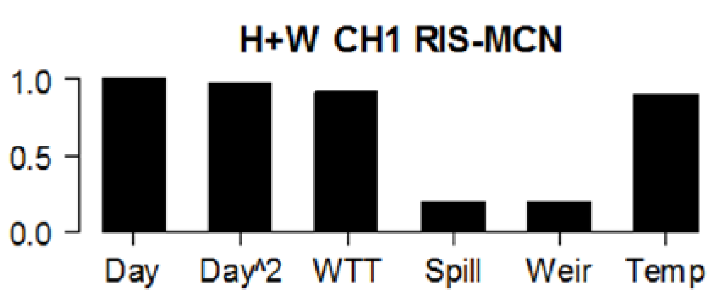


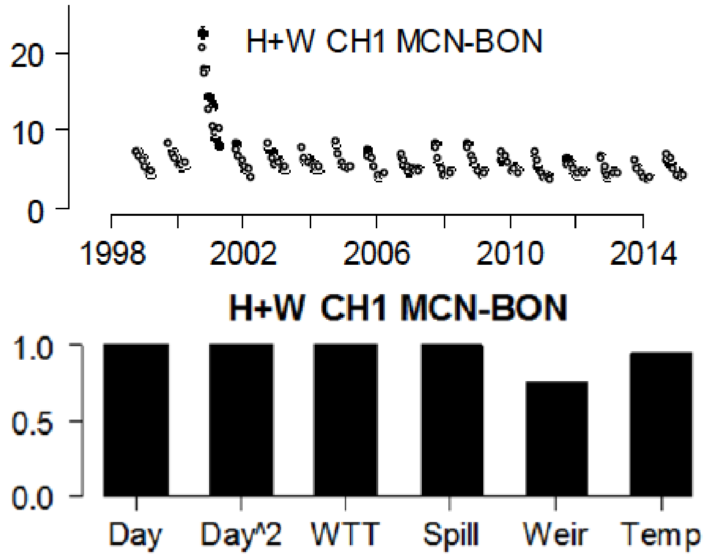
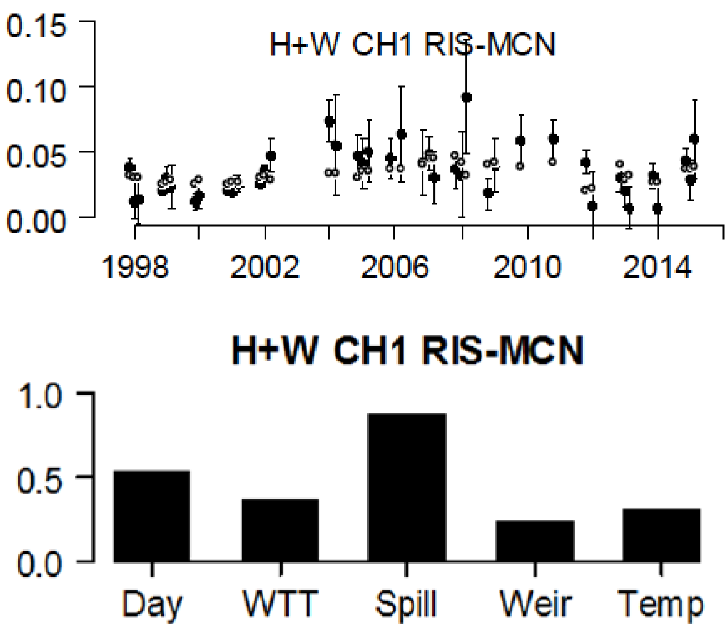


Figure 3.9. Actual (open circles) and predicted (filled circles) mean Fish Travel Time (FTT, days) for yearling Chinook between RIS -> MCN and MCN -> BON (upper graph) and relative variable importance for predictive model of FTT from six covariates (lower graph). Taken from Figure 3.2 and 3.5 of CSS 2017. Error bars are ±1 SE. Model covariates included: Julian day of cohort release (Day), the quadratic effect of Julian day of cohort release (Day^2), water transit time (WTT), average spill proportion (Spill), the number of dams with spillway weirs (Weir), and water temperature (Temp).



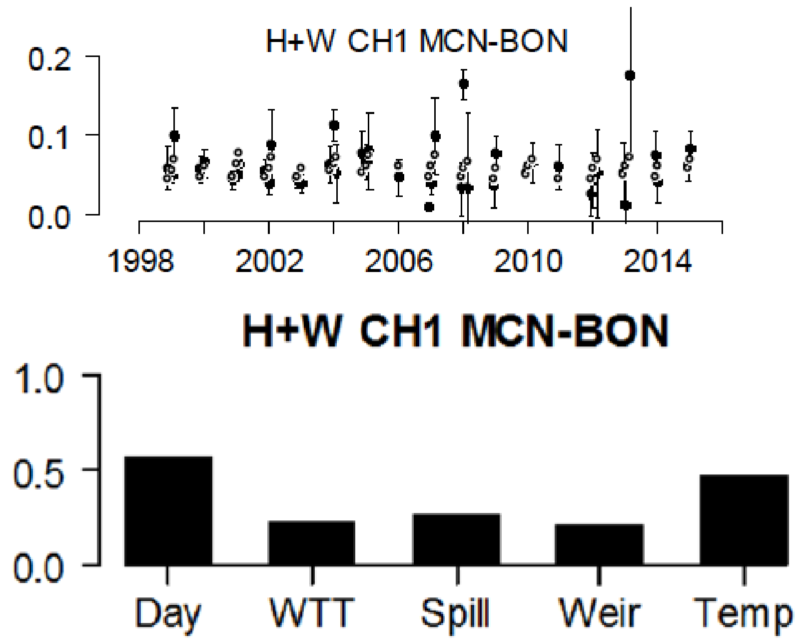


Figure 3.10. Actual (open circles) and predicted (filled circles) for Instantaneous Mortality for yearling Chinook between RIS -> MCN and MCN -> BON (upper graph of pair) and variable importance (lower graph of pair). Taken from Figures 3.3 and 3.6 of CSS 2017. Error bars are ±1 SE. Model variables included: Julian day of cohort release (Day), water transit time (WTT), average spill proportion (Spill), the number of dams with spillway weirs (Weir), and water temperature (Temp).

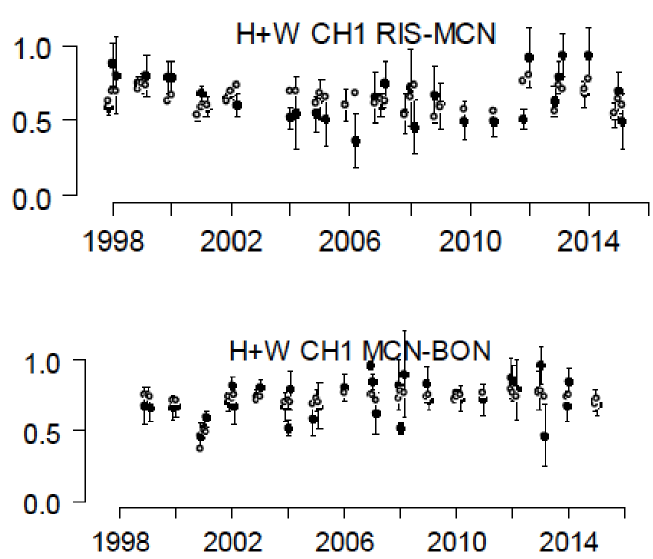


Figure 3.11. Actual (open circles) and predicted (filled circles) of in-river survival for yearling Chinook between RIS -> MCN and MCN -> BON. Take from Figure 3.4 of CSS 2017. Error bars are ±1 SE. Variable importance plots not given in CSS 2017. In-river survival was estimated by developing models for travel time (FTT) and for instantaneous mortality (Z) and then combining the estimates using S = exp(- Z * FTT). Consequently, variable importance plots are not applicable.

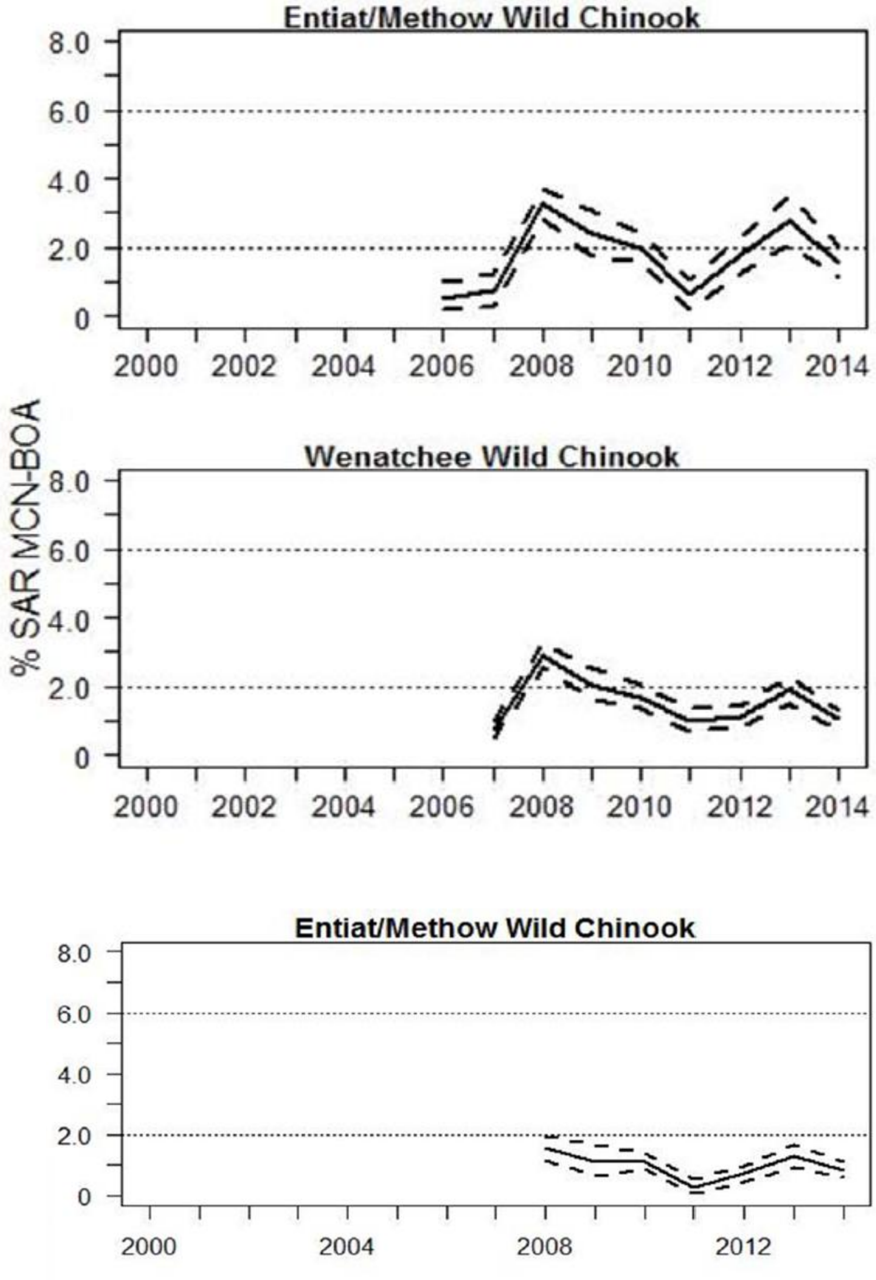
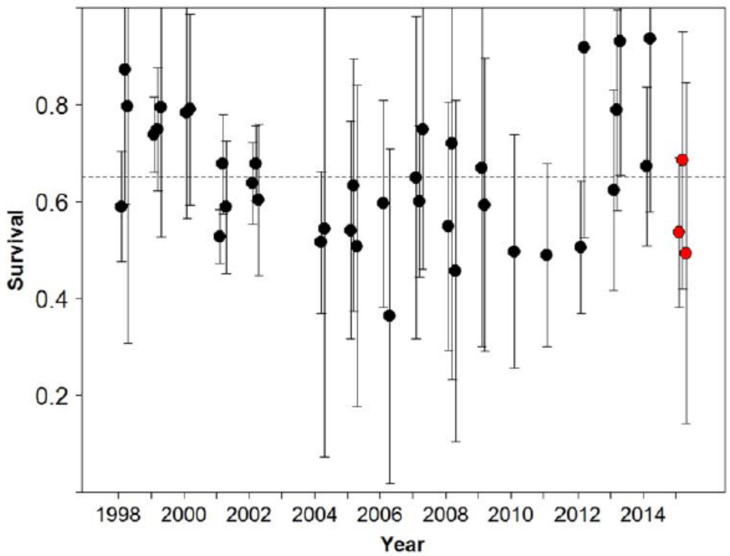


Figure 3.12. SARs (including jacks) for MCN-BON (upper graph) and RRE-BON (lower graph) for Upper Columbia spring Chinook with confidence intervals computed using bootstrapping. Taken from Figure 4.14 and 4.15 of CSS 2017. Migration year 2014 is complete through 2-salt returns only.



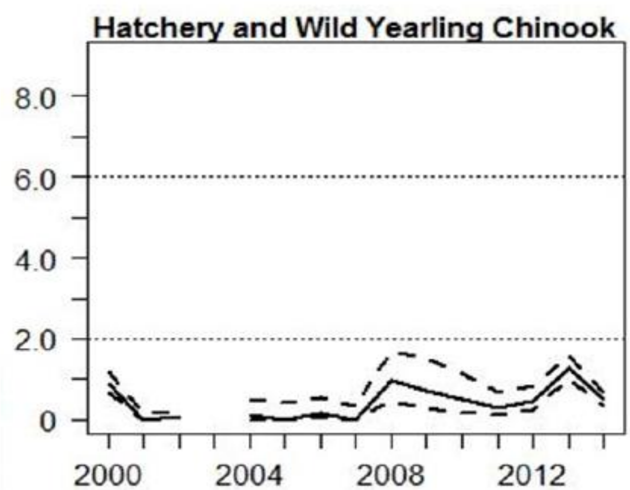


Figure 3.13. Spring out-migrants' yearling spring Chinook juvenile survival from RIS to MCN (top graph). These are 2-week Cormack-Jolly-Seber estimates for smolts captured, PIT-tagged, and released at RIS as part of the SMP project (FPC 2015 annual report). The confidence interval plotted is 95%. The geometric means (through 2014) noted by the horizontal dashed line are 0.61 and 0.59 for Chinook. Taken from Figure 4.17 of CSS 2017. Lower graph is overall SARs (including jack, RIS-BOS). Taken from Figure 4.18 of CSS 2017.

3.1.4.2. Influence of Harvest on Upper Columbia River Spring Chinook Salmon

Commercial harvest of Upper Columbia River salmon and steelhead exceeded 85% of returning adults in the 1930s and 1940s (Mullan 1987). UCR spring Chinook are still harvested both in the ocean and lower river fisheries (NMFS 2016), though harvest of UCR spring Chinook in ocean fisheries appears to be low based on coded wire tag recovery. A non-selective tribal fisheries and selective sport harvest for spring Chinook occur in the lower Columbia River. A non-selective sport fishery for Chinook was opened above Priest Rapids Dam in 2002 but was modified to a selective fishery in 2014 to limit harvest of spring Chinook. The United States v. Oregon Management Agreement manages and coordinates ocean and in-river harvest in the Columbia River Basin under the jurisdiction of the federal court. Regional co-managers set annual harvest levels based on abundance of returning adult Chinook salmon (Table 3.3).

Table 3.3. Harvest rates for Chinook salmon in the Columbia River during the spring harvest season (presented to ISAB by Jeromy Jording, NOAA Fisheries).

Total Upriver Spring and Snake River Summer Chinook Run Size[6]	Snake River Natural Spring/Summer Chinook Run Size[1]	Treaty Zone 6 Total Harvest Rate[2,5]	Non-Treaty Natural Harvest Rate[3]	Total Natural Harvest Rate[4]	Non-Treaty Natural Limited Harvest Rate[4]
<27,000	<2,700	5.0%	<0.5%	<5.5%	0.5%
27,000	2,700	5.0%	0.5%	5.5%	0.5%
33,000	3,300	5.0%	1.0%	6.0%	0.5%
44,000	4,400	6.0%	1.0%	7.0%	0.5%
55,000	5,500	7.0%	1.5%	8.5%	1.0%
82,000	8,200	7.4%	1.6%	9.0%	1.5%
109,000	10,900	8.3%	1.7%	10.0%	
141,000	14,100	9.1%	1.9%	11.0%	
217,000	21,700	10.0%	2.0%	12.0%	
271,000	27,100	10.8%	2.2%	13.0%	
326,000	32,600	11.7%	2.3%	14.0%	
380,000	38,000	12.5%	2.5%	15.0%	
434,000	43,400	13.4%	2.6%	16.0%	
488,000	48,800	14.3%	2.7%	17.0%	

There have been no integrated assessments of limiting factors, including harvest, across the full range of the life history of spring Chinook salmon. Life-cycle models recently provided initial analyses of the relative effect of harvest on returning adult spring Chinook salmon. The life-cycle model for the Wenatchee basin evaluated a scenario of reduced harvest and found that harvest effects were less than those of estuarine adult survival and hatchery operations (Jorgensen et al. 2017). Recent modeling analysis of competing tradeoffs between marine mammal predation and fisheries harvest of Chinook salmon found that predators in the lower river and ocean affected survival more than fisheries harvest (Chasco et al. 2017).

3.1.5. Limiting Factors Conclusions

The Upper Columbia Salmon Recovery Board (UCSRB) has refined its analysis of limiting factors substantially since the first assessments in the late 1990s. The scientific principles for identifying factors limiting the recovery of Upper Columbia spring Chinook salmon are generally sound. Limiting factors are defined in the Recovery Plan of the Upper Columbia Salmon Recovery Board (UCSRB) as environmental conditions that negatively affect the abundance, productivity, spatial structure, and diversity of salmon populations. Analysis of limiting factors based on current habitat conditions is useful for prioritizing restoration actions. The UCSRB recently refined their traditional habitat-based approaches by weighting limiting factors based on anticipated survival benefits and geographic extent. Assessments of limiting factors are used to prioritize recovery actions, and the recent history of restoration is relatively consistent with the rankings of limiting factors.

Analysis of potential limiting factors based on current habitat conditions in the basins that contain natal streams is a useful context for planning and prioritizing restoration actions, but it does not incorporate:

- population dynamics of the fish and assessment of risk of extinction
- the full life history of the fish and their geographic range from egg to adult spawner
- temporal variation and climate change
- hatchery effects
- hydrosystem effects
- in-river and ocean harvest
- predation
- out-of-basin management actions

Limiting factors analysis based on current habitat conditions is useful for prioritizing restoration actions. The UCSRB recently refined their traditional habitat-based approaches by weighting limiting factors based on anticipated survival benefits and geographic extent. To date, analysis has focused on limiting factors related to habitat conditions. The limitations of four Hs on spring Chinook populations are considered in isolation. Limiting factors analyses of the headwater basins do not incorporate the full life history of the fish and their geographic range from egg to adult spawner. Limiting factors for spring Chinook salmon in the Upper Columbia have been assessed through traditional freshwater habitat assessments, analysis of density dependence, and life cycle models. These methods are complementary rather than mutually exclusive. Each have important applications in management of the four Hs, but to date there has been no integration of all three approaches.

Empirical density dependence data and life-cycle modeling conducted by regional scientists in the Upper Columbia provide more holistic analyses of limiting factors over the full life cycle of spring Chinook salmon than traditional assessments of habitat-related limiting factors in freshwater. These traditional habitat-based approaches can be used in planning and prioritizing restoration actions, but more explicit integration of these approaches would strengthen future efforts. As an example, monitoring data in the Okanogan basin is incorporated into an EDT model, which includes a life-cycle model.

The UCSRB program's goal is to identify limiting factors within four categories of human impacts (threats), including habitat, harvest, hydrosystem, and hatcheries (four Hs), both within and outside the Upper Columbia watershed. While the Recovery Plan recognizes the influence of all Hs, analysis of limiting factors has not integrated the four Hs to determine which have the greatest influence on spring Chinook populations. Life-cycle models for these subbasins are beginning to provide evidence that addresses all Hs, but they are in early stages of development. Limiting factors are considered to be working hypotheses that can be tested, and monitoring and adaptive management are critical for understanding and addressing them. Key uncertainties such as ocean productivity and global climate change are identified.

3.2. Comparison of Upper Columbia River Spring Chinook Recovery with Snake River Spring Chinook Recovery

The Oversight Panel asked the ISAB to compare the recovery of Upper Columbia River Spring Chinook (UCRSC) to that of Snake River Spring Chinook (SNRSC). The ISAB believes that despite geographic and biological differences in the two systems and the fact that only 10 years have passed since completion of the UCRSC recovery plan (Upper Columbia Salmon Recovery Board 2007), this comparison may provide insights into recovery of UCR spring Chinook. Here we describe the geographic and biological differences and similarities in the two ESUs and offer a comparison of the recovery of spring Chinook in the two systems.

3.2.2. Geography

3.2.1.1. Spatial Structure

Population spatial structure describes how populations are arranged geographically based on dispersal factors and quality of habitats (McElhany et al. 2000). There are three subbasins containing extant populations in the Upper Columbia ESU amounting to about 940,000 hectares (2.3 million acres), while the 26 subbasins in the Snake River ESU cover 11.5 million hectares (28.3 million acres) (estimates include both current distribution and functionally extirpated area). The ICTRT (2007) separated the populations into subbasin size categories and within population complexity based on:

> "A set of four population size categories (Basic, Intermediate, Large and Very Large) for Interior basin stream type chinook… The smallest populations were grouped into the Basic size category. Populations assigned to the Basic size categories tended to be simple in complexity, often with a relatively linear arrangement of spawning/rearing reaches. Median population size roughly doubled between size categories. Populations with significantly higher amounts of potential spawning habitat usually exhibited a higher degree of spatial diversity—e.g., multiple tributary branches. Contemporary redd survey results

> indicated that the distribution of spawners across sub-areas within a population was likely to be patchy. Relatively high spawning concentrations in particular sub-areas could be achieved in the larger, more complex population at lower overall spawning densities… We used two methods to characterize the relative within-population complexity of tributary spawning habitats—assigning each population to one of four general structural complexity categories… and estimating the number of relatively large, contiguous production areas within each population... We defined a branch as a river reach containing sufficient habitat to support 50 spawners. Major spawning areas (MaSAs) were defined as a system of one or more branches that contain sufficient habitat to support 500 spawners... We defined contiguous production areas capable of supporting between 50 and 500 spawners as minor spawning areas (MiSAs)." (ICTRT 2007, pages 15-16).

Two of the three subbasins (66%) in the UCR (Table 3.4) are very large compared to just two very large subbasins of 26 (8%) in the SNR (Table 3.5). The remaining SNR subbasins range in size from basic to large. By virtue of the larger number of subbasins, the SNR has a broader range of stream complexities (e.g., linear vs dendritic) than does the UCR; however, the UCR tends to have more major (MaSA) and minor (MiSA) spawning areas per subbasin than does the SNR (Tables 3.4 and 3.5).

Table 3.4. Intrinsic size and complexity ratings for historical populations within the Upper Columbia River Spring Chinook ESU. Organized by Major Population Groupings. Complexity categories: A = linear; B= dendritic; C= trellis pattern; D= core drainage plus adjacent but separate small tributaries (from ICTRT 2007a, page 20).

Major Population Group	Population	Weighted Area Category	Complexity	
			Category	# MaSA (# MiSA)
Eastern Cascades	Wenatchee	Very Large	B	5 (4)
	Methow	Very Large	B	4 (1)
	Entiat	Basic	A	1 (0)
	Okanogan River (ext) (US portion only)[1]	Intermediate	D	1 (3)

[1] Spring Chinook historically occupied tributary habitat in both the U.S. and Canada. Current ICTRT analyses are focused on the US portion, although additional MSAs or populations may exist in the Canadian portion.

Table 3.5. Intrinsic size and complexity ratings for extant Snake River Spring Chinook ESU populations organized by Major Population Groupings. Complexity categories: A = linear; B=dendritic; C= trellis pattern; D= core drainage plus adjacent but separate small tributaries. Underlined entries represent a change from the previous designation. Size categories in parentheses represent core tributary production areas (page 17, ICTRT, 2007a).

Major Population Group	Population	Weighted Area Category	Complexity	
			Category	#MaSA (#MiSA)
Lower Snake	Tucannon R	Intermediate	A	1 (0)
	Asotin R. (ext)	Basic	A	0 (1)
Grande Ronde/Imnaha R	Lostine/Wallowa R.	Large	B	3 (1)
	Upper Grande Ronde R.	Large	B	3 (2)
	Catherine Creek	Large	B	2 (2)
	Imnaha R. Mainstem	Intermediate	A	1 (1)
	Minam R.	Intermediate	A	2 (0)
	Wenaha R.	Intermediate	A	1 (0)
	Big Sheep Cr. (ext)	Basic	A	0 (1)
	Lookingglass Cr. (ext)	Basic	A	0 (1)
South Fork Salmon	South Fk Mainstem	Large	C	2 (2)
	Secesh R.	Intermediate	A	1 (1)
	East Fk/Johnson Cr.	Large	B	2 (0)
	Little Salmon R.	Inter. (Basic)	D	0 (3)

Middle Fork Salmon	Big Creek	Large	B	3 (0)
	Bear Valley	Intermediate	C	3 (0)
	Upper Mainstem MF	Intermediate	C	1 (2)
	Chamberlain Cr.	Inter. (Basic)	D	1 (3)
	Camas Creek	Basic	B	1 (1)
	Loon Creek	Basic	C	1 (0)
	Marsh Creek	Basic	C	1 (0)
	Lower Mainstem MF	Basic	A	0 (1)
	Sulphur Creek	Basic	A	1 (0)
Upper Salmon	Lemhi	Very Large	B	3 (2)
	Lower Mainstem	Very Large	C	3 (5)
	Pahsimeroi	Large	B	5 (0)
	Upper Salmon East Fk	Large	C	1 (0)
	Upper Salmon Mainstem	Large	C	3 (0)
	Valley Cr.	Basic	A	1 (0)
	Yankee Fork	Basic	C	1 (0)
	North Fork Salmon R.	Basic	D	1 (0)
	Panther Cr. (ext)	Intermediate	C	1 (2)

The more restricted geographic area of the UCR results in very homogeneous spawning elevations—ranging from about 1600 to 3000 feet—as opposed to the SNR where spawning areas are available up to 7000 feet (Figure 3.14). The UCR subbasins are located on the eastern slopes of the Cascade Mountains and receive between 25 to 55 inches of precipitation annually (Figure 3.15). The SNR is in the more arid eastern portion of the Columbia Basin, and SNR subbasins tend to receive less precipitation than those in the UCR (Figure 3.15). Hydrologic regimes in subbasins of both ESUs are a mix of rain, transition, and snow (Figure 3.16). The CHaMP assessment of the geomorphic condition is similar for the subbasins; however, UCR subbasins may contain a larger percent of intact and good areas than the SNR subbasins (Figure 3.17).

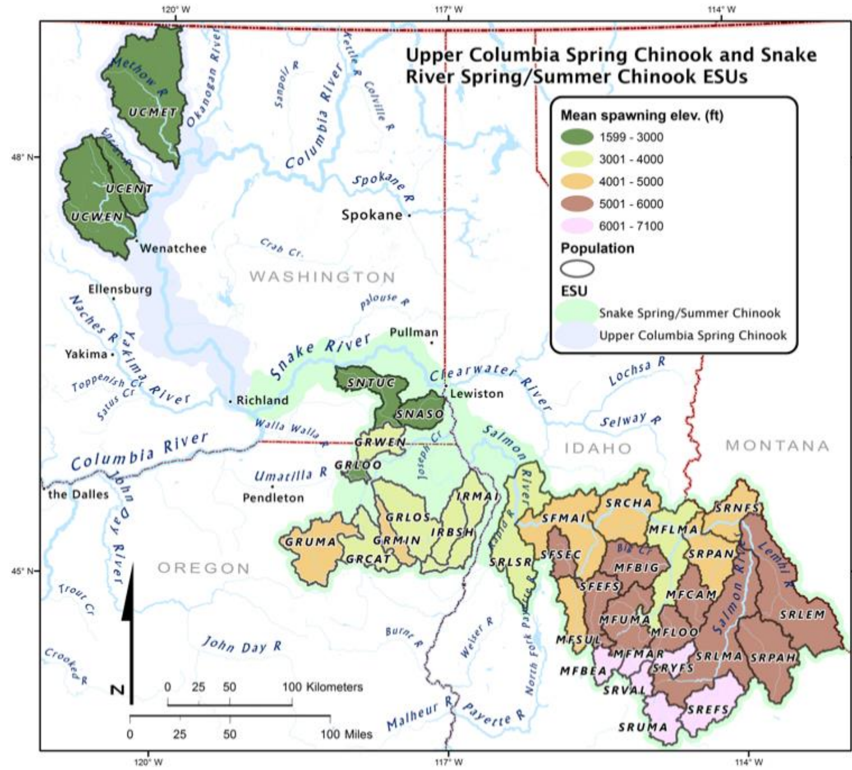


Figure 3.14. Mean spawning elevation available to spring Chinook salmon in the Upper Columbia River and spring/summer Chinook in the Snake River (from M. Ford 2017, NOAA Fisheries).

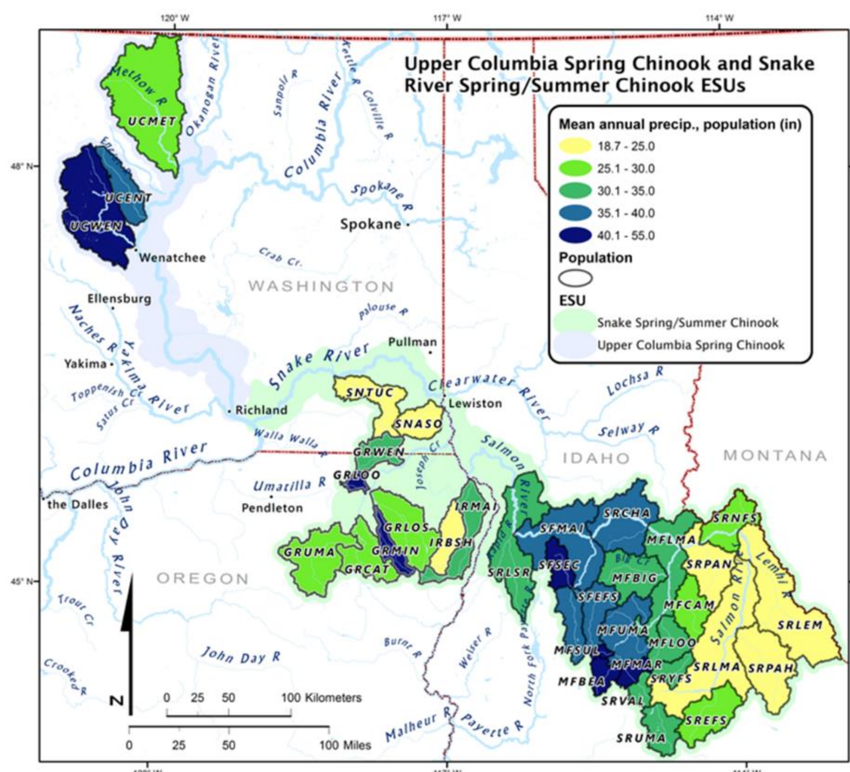


Figure 3.15. Mean annual precipitation in the Upper Columbia and Snake river spring Chinook ESU subbasins (from M. Ford 2017, NOAA Fisheries).

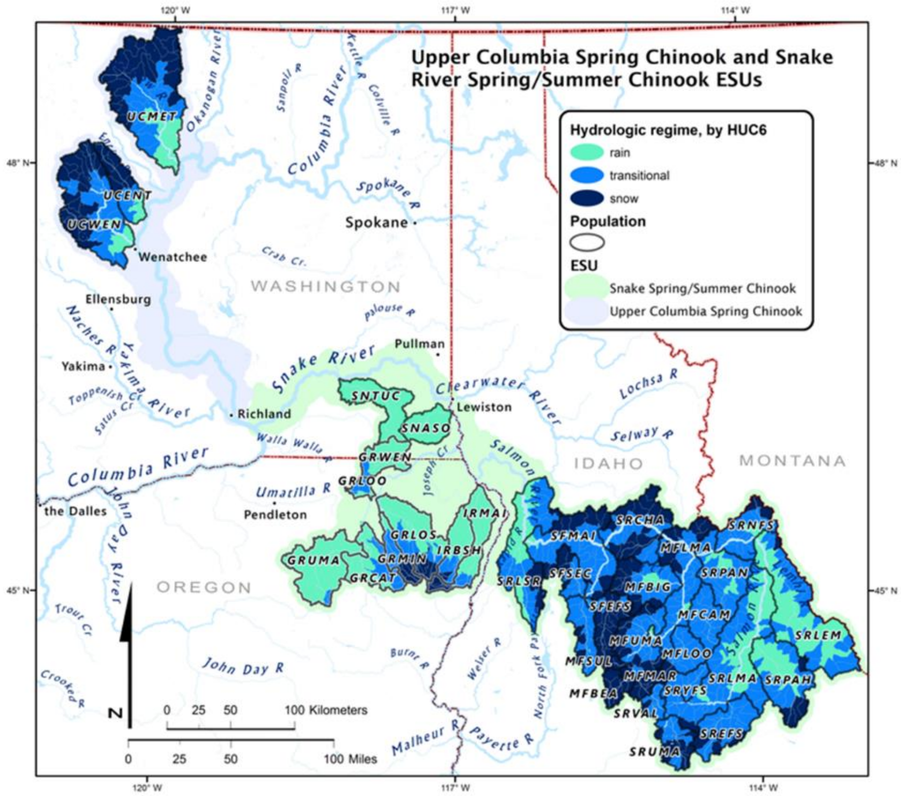


Figure 3.16. Hydrologic regime of Upper Columbia spring Chinook and Snake River spring/summer Chinook ESUs (from M. Ford 2017, NOAA Fisheries).

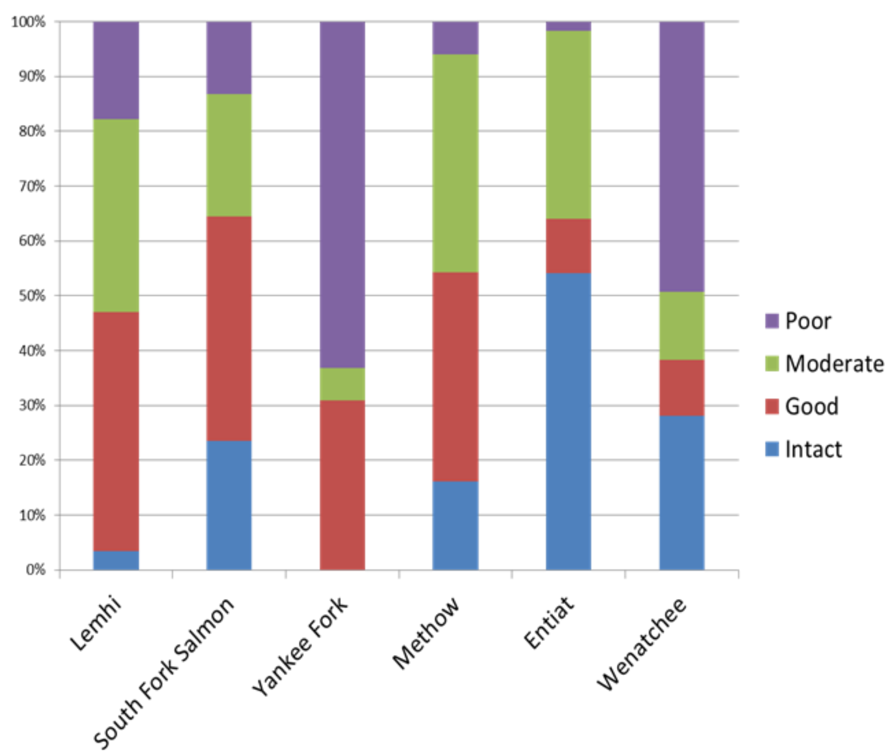


Figure 3.17. Geomorphic condition based on CHaMP analyses of subbasins of ESUs in the Upper Columbia River (three on the right) and representative subbasins in the Snake River (three on the left). Figure provided by M. Ford 2017, NOAA Fisheries with data from C. Jordan.

3.2.1 Comparison of Population Diversity

3.2.2.1 Life History and Genetic Diversity

Population diversity consists of the underlying genetic and life history characteristics that provide for population resilience and persistence across space and time (McElhany et al. 2000). Differences in the two ESUs may result in less population diversity in the UCR than the SNR. There are only three extant populations of spring Chinook in the Upper Columbia compared to 26 populations in the Snake River. Furthermore, the SNR ESU includes summer Chinook while summer Chinook in the UCR are not listed under the Endangered Species Act (ESA). Waples et al. (2004) reported that interior Columbia Basin Chinook salmon were closely related based on a yearling (stream-type) emigration life history. Additionally, yearling (stream-type) spring and summer Chinook have an ocean migration that differs from subyearling (ocean-type) emigrants that are readily harvested in the ocean (Myers et al. 1998). Genetic differences among Chinook populations are more strongly correlated to yearling life-history strategy and marine distribution than to spawning-run timing (Waples et al. 2004). Thus, the UCR summer Chinook are genetically more similar to UCR fall Chinook than UCR spring Chinook, while SNR summers are more genetically similar to springs than non-listed fall or summer Chinook in the mid and Upper Columbia River (Narum et al. 2010, Waples et al. 2004, NMFS 2015). At the time of ESA listing, the UCR spring Chinook were listed as endangered while spring Chinook in the SNR were listed as threatened (Ford et al. 2015). This suggests that the UCR spring Chinook populations were at a higher risk of extinction at the time of listing than were the SNR populations. These ESA designations continue today.

Genetic research published recently (Johnson et al. 2018) concludes that contemporary populations of Chinook salmon have less genetic diversity compared to ancient Chinook present in the Columbia Basin 300 to 7600 years ago, which included fish from areas above Chief Joseph Dam. Losses of genetic diversity occur when population size diminishes and geographic range size contracts over time. Snake River Chinook salmon populations do not appear to have experienced the same degree of loss of genetic diversity over time because comparative changes in abundance and distribution are not as pronounced as those in the UCR (Johnson et al. 2018). Based on genetic differences, the groups also differed demographically with the UCR Chinook having experienced a reduction in effective population size, while the SNR Chinook have not. The study included fall-, spring- and summer-run fish, and the authors suggest that their findings could be the result of differing genetic contributions of the runs and the cumulative effects of exploitation by native and non-native fishers both pre- and post-European immigration to North America (Johnson et al. 2018).

3.2.2.2. Abundance and Productivity

Abundance is the number of adult spawners measured over time based on life history, and productivity (population growth rate) is a measure of a population's ability to sustain itself over time (e.g., recruits per spawner; McElhany et al. 2000). To determine the abundance and productivity of salmon populations, the Interior Columbia Technical Recovery Team (ICTRT) considered the mean natural spawning abundance for a 5-year period (2010-2014) as compared to the previous 5-year period. All of the UCR spring Chinook populations had increased numbers of natural spawners (74% average increase), as did all but 2 of the 26 populations in the SNR (154% average increase; NMFS 2015). Despite the increases in natural spawners, the proportion of hatchery origin spawners (pHOS) also increased in populations that received hatchery supplementation. In the UCR, pHOS in the Wenatchee and Methow rivers in 2015 was 65 to 75% compared to about 15% for all UCR populations in the early 1990s. During this time, the Entiat population, which is not currently supplemented, increased to about 25% pHOS. In the SNR, 15 non-supplemented populations (out of 26) had pHOS of 7% or less (13 had 0% pHOS). The remaining populations experienced from 82 to 11% pHOS (mean =43%), but the mean pHOS for all 26 SNR populations was 19%. (All data were summarized from NMFS 2015).

The ICTRT identified recovery gaps, which is the minimum survival change needed to increase a population from its current status to its target viability (ICTRT 2007b). Gaps are quantifiable measures of difference in current status and the viability criteria for the abundance and productivity of a population. All of the UCR spring Chinook populations have high gaps (i.e., large differences between current status and viability criteria) while the SNR spring/summer Chinook populations vary from low to very high, with most populations in the high gap category similar to UCR spring Chinook (Figure 3.18).

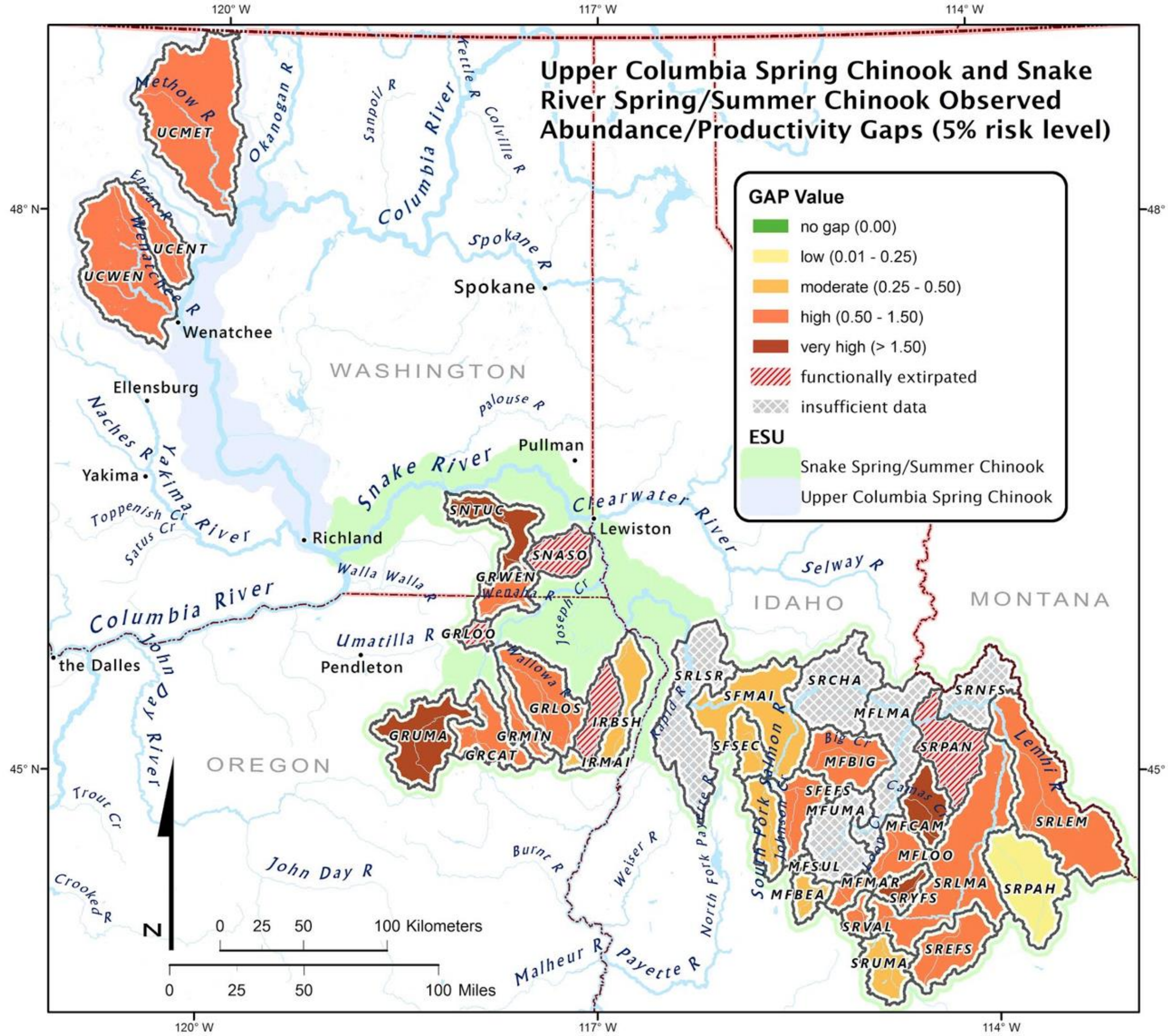


Figure 3.18. Abundance and productivity gaps for Snake River Spring/Summer Chinook ESU populations (map also includes Upper Columbia Spring Chinook ESU populations for comparison). Populations with insufficient data to generate gaps shaded in gray. Gaps are defined as relative improvement in productivity or limiting capacity required for a population to exceed its corresponding 5% risk viability curve (ICTRT 2007b). Figure is from NMFS (2015).

It appears that changes in abundance and productivity of the two ESUs are similar and the number of natural spawners have increased in both systems. However, on average, the increase in SNR populations in 2010-2014 was twice that of UCR populations. The proportion of hatchery origin spawners has also increased in hatchery supplemented populations in both systems, but since the SNR has a number of populations that are not supplemented, the average pHOS for all 26 SNR populations is less than half that of the average of the three UCR populations.

3.2.2.3. Smolt Migration and Survival through the Hydrosystem

Freshwater spawning and rearing habitats of SNR spring/summer Chinook populations are located upstream of Lower Granite Dam (LGR), and smolts must emigrate through four Snake River hydropower dams before reaching the mainstem Columbia River upstream of McNary Dam (MCN). The UCR spring Chinook populations emigrate into the Columbia mainstem and pass three, four, or five dams before reaching the free-flowing Hanford Reach and subsequently McNary Dam.

The Comparative Survival Study (CSS) tracks smolt migration and survival from Rock Island Dam (RIS) in the UCR and from LGR in the SNR to MCN, and from MCN to Bonneville Dam (BON). There are significant differences in the Chinook populations. The number of emigrants in the UCR is significantly less than in the SNR. To have adequate sample sizes and thus adequate statistical power, the CSS creates smolt cohorts of all fish passing a dam during a specific time period. In the UCR, the cohorts are composed of all wild and hatchery spring Chinook detected at RIS during a two-week period. In the SNR, wild and hatchery fish are in separate cohorts that pass LGR during a one-week period. With these caveats in mind, the probability of UCR spring Chinook smolts surviving (RIS to MCN) is about 60%, while SNR smolt probability of survival (LGR to MCN) is about 75 to 80% (Figure 3.19). It is important to note that the CSS reported that "within-year estimates of (survival) varied by up to 39 percentage points for both wild yearling Chinook and steelhead, and by up to 32 percentage points for hatchery yearling Chinook" (CSS 2017 Draft). As a result, they were not able to model the role of environmental factors on survival. Faulkner et al. (2017) calculated the probability of survival for smolts emigrating from hatcheries in the UCR and SNR to dams in the Columbia River beginning in the 1990s (Table 3.6). This study included mortalities from hatcheries in the tributaries to the dams. The results are very similar to those reported by CSS (2017) but also suggest the possibility that UCR smolts had better survival than the SNR smolts when the full emigration from hatchery to BON is considered (Faulkner et al. 2017).

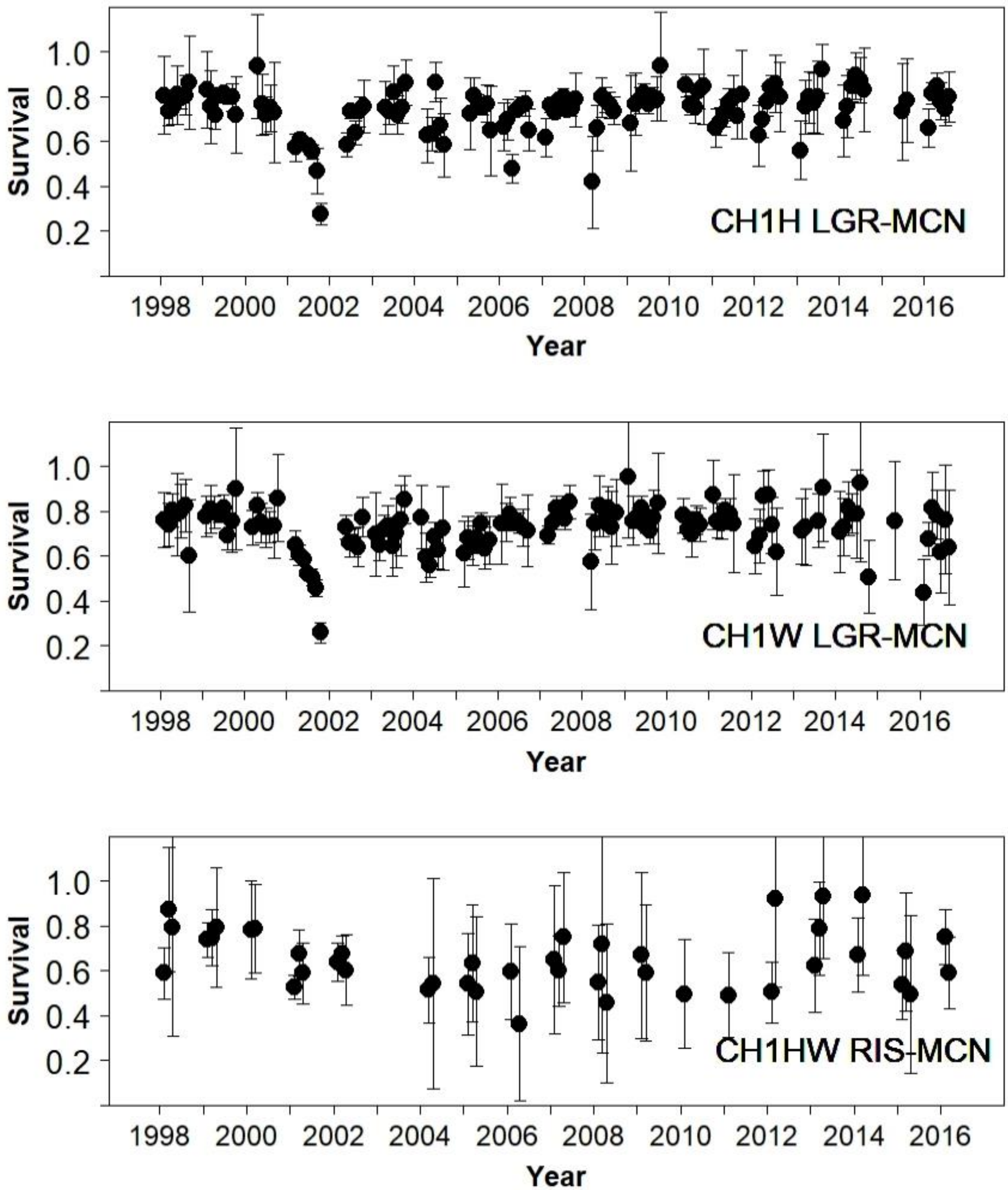


Figure 3.19. Estimates of in-river survival probability for release cohorts of hatchery (H) and wild (W) yearling Chinook salmon (CH1) migrating in the LGR–MCN and RIS–MCN reaches, 1998–2016. The error bars represent +/- 1 SE. Figure was provided by J. McCann, FPC and was derived from the CSS 2017 Draft.

Table 3.6. Probability of smolt survival when migrating from hatcheries in the Upper Columbia River (UCR) or the Snake River (SNR) to either Lower Granite Dam (LGR) or McNary Dam (MCN) and from MCN to Bonneville Dam (BON). Values are the means (SE) for the migration years listed. Data are from Faulkner et al. (2017). The ISAB derived values for Hatchery-BON by multiplying the mean values for the reaches.

Smolt Survival				
ESU	Reach	Years	Probability (SE)	Hatchery-BON
UCR	Hatchery-MCN	1999-2016	0.555 (0.012)	
	MCN-BON		0.809 (0.034)	0.449
SNR	Hatchery-LGR	1993-2016	0.629 (0.012)	
	LGR-MCN		0.736 (0.013)	
	MCN-BON		0.703 (0.021)	0.325

It is possible that avian and fish predation may be the cause of significant UCR Chinook mortality (McMichael et al. 2017). However, comparing data from different years (i.e., 2012 and 2014), it appears that that yearling Chinook migrating from RIS through the Hanford Reach did not experience any more avian predation, as a part of total mortality, than did yearling Chinook migrating in the lower Snake River (Table 3.7; Evans et al. 2015, 2016). Based on existing data, it is not possible to detect differences in smolt survival during their emigration from the UCR to BON versus SNR to BON.

Table 3.7. Reach-specific bird predation rates (percentage of tagged fish consumed, as means with SE) and total mortality of tagged yearling Chinook salmon released into the mid/Upper Columbia River at river kilometer 729 in 2014 (bottom) or into the lower Snake River at river kilometer 562 in 2012 (top). Five-weekly releases started on 30 April 2012 and 24 April 2014. Data from Evans et al. (2015).

	UCR (Rkm 729-545)		SNR (Rkm 562-525)		
	Bird Predation	Total Mortality	Bird Predation	Total Mortality	% Total
2012	NA		3.7% (1.7-7.2)	8.9% (8.7-8.9)	42.70%
2014	2.8% (1.0 - 5.9)	12.2% (11.3-12.6)	NA		29.50%

3.2.2.4. Smolt-to-Adult Returns

NMFS (2015) considered smolt-to-adult returns (SARs) from several ESA-listed salmonids in the Columbia River Basin including UCR natural origin spring Chinook and SNR spring/summer Chinook. These data were rescaled to three-year moving averages to reduce the impact of short-term climate variability and relied on a long data series of smolt and adult natural-origin spring Chinook in the Chiwawa River (Figure 3.20). All of the populations have similar patterns, with low SARs in 1990s and peaks in the early 2000s (NMFS 2015).

The CSS calculates SARs for UCR spring Chinook smolts passing Rocky Reach Dam (RRE) to their detection as adults at Bonneville Dam adult collection facility (BON-A). Snake River spring/summer Chinook SARs are calculated from LGR to BON-A. The CSS calculates SARs for individual hatcheries in both systems and also for the natural origin UCR spring Chinook (from the Entiat and Methow rivers) and SNR natural origin spring/summer Chinook. SARs for these groups of fish along with natural origin Wenatchee River spring Chinook are also calculated for smolts passing McNary Dam (MCN) and returning to BON-A. Comparing the natural origin Chinook groups is more appropriate than comparing the SARs from 3 UCR hatcheries with 10 SNR hatcheries. Furthermore, complete SNR data are from 2000 through 2014, as compared to UCR data from 2008 through 2014. Here we considered this overlapping 7-year period and found SARs for natural origin UCR spring Chinook did not differ from those of the natural origin SNR spring/summer Chinook (Table 3.8). Thus, since the UCR recovery plan has been in place (UCSRB 2007), there has been no difference in SARs of natural origin UCR spring Chinook compared to that of natural origin SNR spring/summer Chinook.

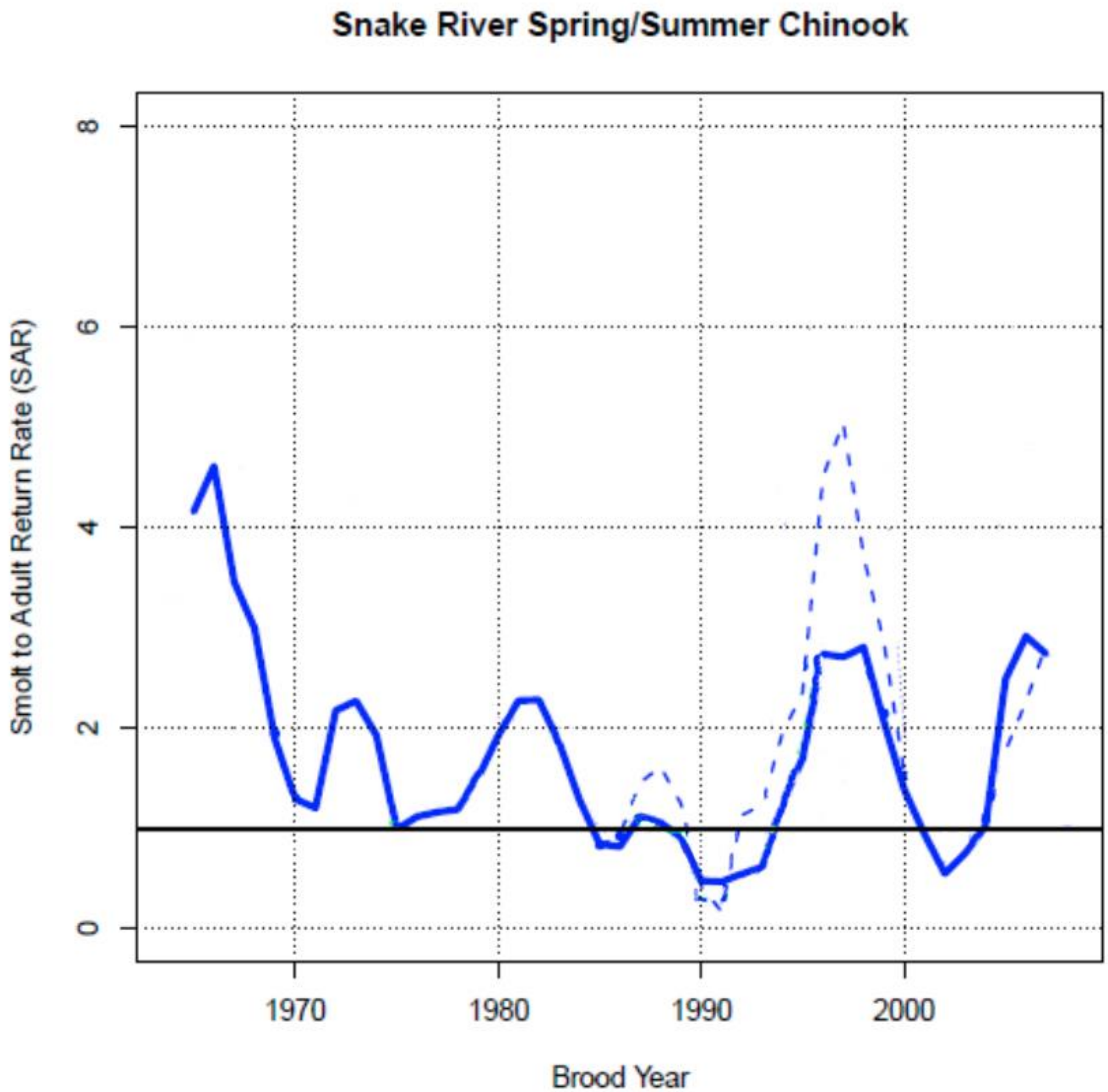


Figure 3.20. Snake River spring/summer Chinook salmon aggregate smolt to adult return rates (solid blue), Upper Columbia spring Chinook (blue dashed line). Each SAR series is rescaled by dividing annual values by the corresponding series mean to facilitate relative comparison. (Figure and caption derived from NMFS 2015).

Table 3.8. Smolt-to-adult return rate for Upper Columbia River (UCR) natural origin spring Chinook salmon (Sp Chin) and Snake River (SNR) natural origin spring/summer Chinook (Sp/Su Chin) that passed Lower Granite Dam (LGR), Rocky Reach Dam (RRE) or McNary Dam (MCN) and were subsequently detected at Bonneville Dam (BOA) as returning adults (jacks included). The UCR fish come from the Entiat (Ent), Methow (Met) or Wenatchee (Wen) river. P-value is based on a t-test with unequal variances; data for the Wenatchee were not included in statistical analysis. Data are summarized from CSS (2017 draft, Appendix B).

Smolt-to-Adult Returns					
	LGR-BOA	RRE-BOA	MCN - BOA		
Brood Year	SNR Sp/Su Chin	UCR Sp Chin (Ent + Met)	SNR Sp/Su Chin	UCR Sp Chin (Ent + Met)	UCR Sp Chin (Wen)
2008	4.13	1.72	4.07	3.26	2.89
2009	2.09	1.20	2.00	2.40	2.07
2010	1.16	1.21	1.28	1.97	1.70
2011	0.45	0.44	0.46	0.62	1.02
2012	1.84	1.03	2.32	1.77	1.14
2013	1.71	1.66	1.69	2.80	1.93
2014	0.68	1.07	0.43	1.87	1.11
Mean	1.72	1.19	1.75	2.10	1.69
SD	1.22	0.43	1.25	0.85	0.67
SE	0.17	0.06	0.18	0.12	0.10
P(t-test)	0.16		0.28		

3.2.2.5. Adult Mortalities in the Columbia River

3.2.2.5.1. Harvest

Exploitation rates (total harvest) for UCR spring Chinook are the same as those for the SNR spring/summer Chinook and have been averaging about 10% since 1980 (NMFS 2015). However, the rate for both ESUs appears have been increasing since 2000 (Figure 3.21).

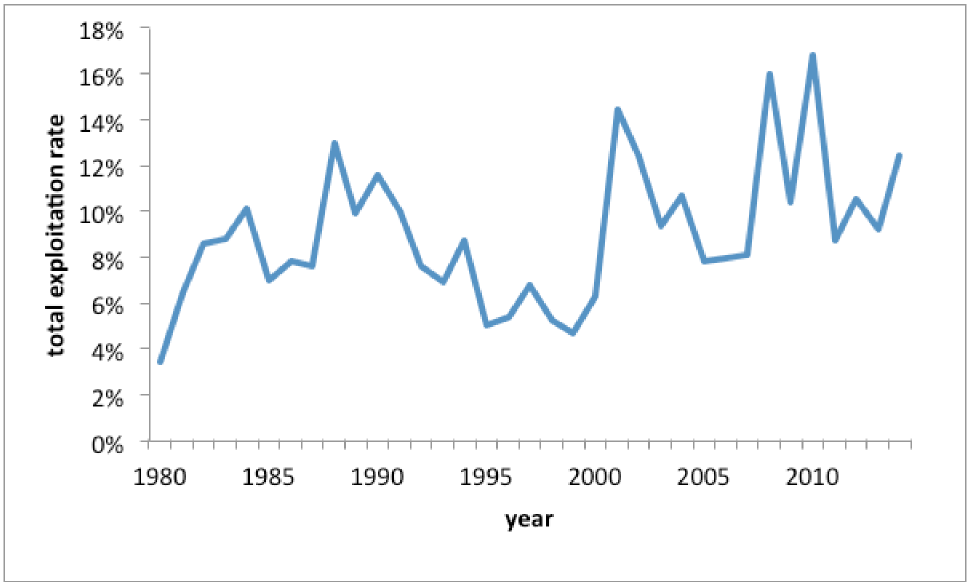


Figure 3.21. Exploitation rate (total harvest) for Upper Columbia River spring Chinook salmon. As reported in NMFS 2015, the exploitation rates for Snake River spring/summer Chinook were identical. Data are from Columbia River Technical Advisory Committee. Figure is from NMFS (2015).

3.2.2.5.2. Pinniped Predation

Researchers from NOAA-Fisheries have been investigating adult salmon migration timing and pinniped predation in the lower Columbia River. Sorel et al. (2017) cites Rub et al. (in preparation) as saying that survival of adult salmon migrating from Astoria to BON decreased between 2010 and 2015 and that survival is lowest in the fish migrating earliest in the season. These measures of survival coincide with increases in pinnipeds in the lower river. Sorel et al. (2017) documented the passage times of UCR and SNR adults at BON from 1998 through 2015 (Figure 3.22). There is significant overlap in the UCR and SNR spring-run Chinook passage. However, summer-run fish—which are part of the SNR ESU, but not the UCR ESU—arrive later and potentially would not be preyed upon to the extent of the spring-run Chinook.

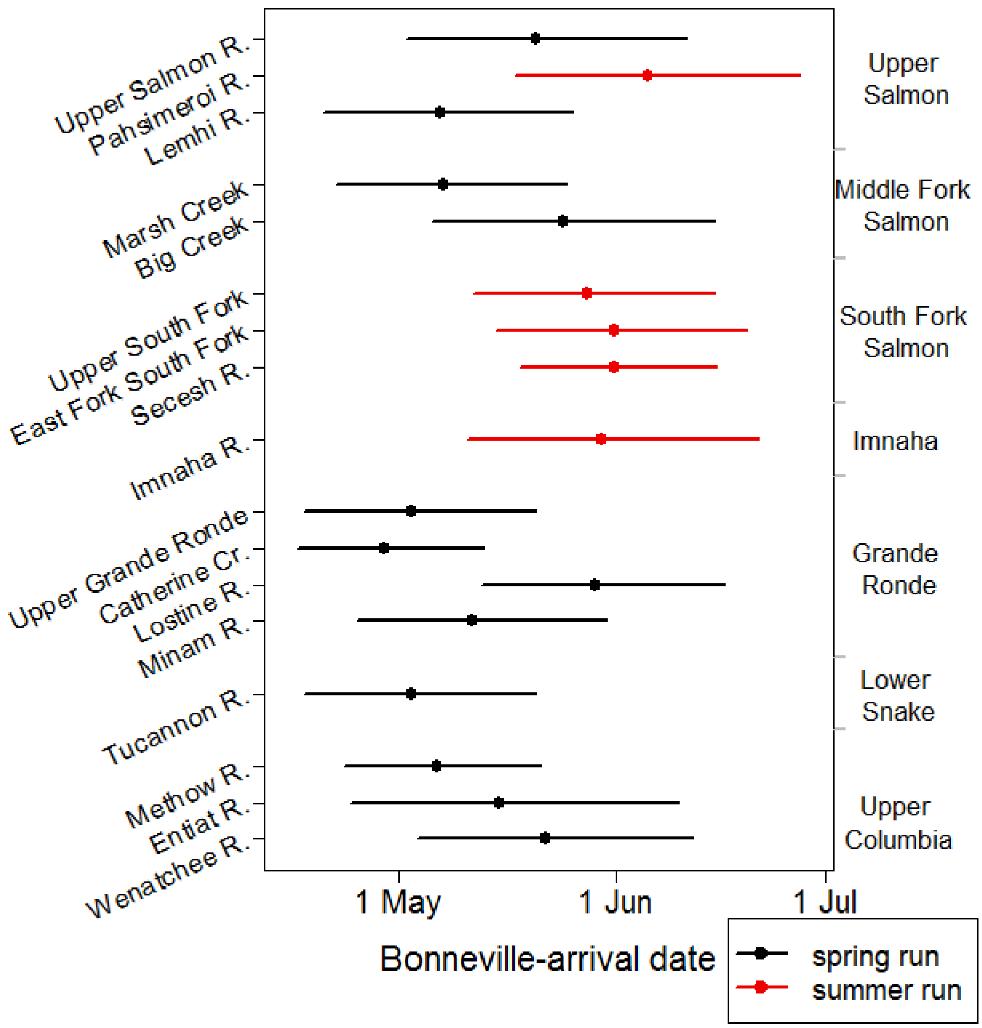


Figure 3.22. Average mean passage date (points) at Bonneville Dam for wild spring and summer Chinook salmon populations in 1998–2015. The bottom three points are from Upper Columbia River populations and all others are from the Snake River populations. The lines represent the ranges of dates in which the authors predicted that 75% of each population would pass Bonneville Dam in this hypothetical average year. There was considerable variation across populations, and earlier-migrating populations had lower survival than later-migrating populations (from Sorel et al. 2017). For comparison, Upper Columbia summer Chinook arrive at BON in July and August, i.e., later than the arrival dates shown in this figure.

The survival rates of the UCR and SNR adults in the lower river coincide with the increased number of pinnipeds and reflect the added jeopardy for early arriving fish.

> "The earliest-arriving of the spring-run populations examined—Lemhi River, Marsh Creek, Upper Grande Ronde, Catherine Creek, Tucannon River, and Methow River—had somewhat lower survival rates than other populations in 2010–2012, when annual medians ranged from 69% to 81%, and much lower

survival rates in 2013–2014, which had a 50–70% range in annual median survivals. Populations with intermediate run timing such as the Upper Salmon River, Big Creek, Minam River, Entiat River, and Wenatchee River had experienced survival that was somewhat lower in 2013–2015, with annual medians of 67–85%, than 2010–2012's 79% to 88% range. Late-arriving populations—Pahsimeroi River, Upper South Fork Salmon River, East Fork South Fork Salmon River, Secesh River, Imnaha River, and Lostine River—most of which are considered summer-run, had the highest survival rates with the least interannual variability" (Sorel et al. 2017, page 14).

Sorel et al. (2018) recently indicated that average survival in the lower Columbia River (i.e., below BON) in 2013-2015 was 13% lower than in 2001–2012 for the Methow River population. The Entiat River population experienced an 8% decline in survival between these periods, and survival of the Wenatchee River population was 4% lower (Sorel et al. 2018). The extent of this decreased survival correlates with the timing of arrival of these populations at BON (see Figure 3.11 in section 3.3.). The question of pinniped predation is also addressed in Section 3.4. Pinniped Predation (below).

3.2.2.5.3. Pre-spawning Mortality (PSM)

Over the past 16 years, an average of 46% of the UCR spring Chinook females died prior to spawning as compared to 15% of the UCR summer Chinook (Murdoch, 2017). A variety of sources suggest that annual pre-spawning mortality of spring/summer Chinook in the SNR is more like the UCR summer Chinook than the UCR springs. Cleary et al. (2014) reported PSM averaged 16% for natural origin Chinook spawners in the Lostine River between 2002-2012; Crump et al. (2017) reported the same 16% pre-spawning mortality for Chinook in the Lookingglass Creek from 2004-2016. The variance of the reported pre-spawning mortality in both SNR subbasins is high with the Lostine varying from 5% to 32% and Lookingglass ranging from 0 to 54%.

3.2.2.6. Ocean and Estuary

The ISAB reviewed relevant scientific information on smolt-to-adult estuary and ocean life history traits and population status indicators (see Appendix B), and concluded that UCR spring Chinook and SNR spring/summer Chinook ESUs exhibit similar estuary-ocean life histories and thus are likely to experience similar limiting factors in these habits. However, the ISAB found large gaps in scientific knowledge and understanding of estuary-ocean life histories and associated limiting factors for both ESUs. For example, there is a nearly complete absence of data on seasonal and spatial distribution, abundance, and survival both during and after the first winter at sea. Most of the available population-specific data for both ESUs are for hatchery-origin fish, which do not always exhibit life history traits and population dynamics of natural-origin conspecifics. Our review found that hatchery-origin fish differed between the two ESUs in key traits that could influence marine ecological interactions (e.g., predation, competition) and ocean growth and survival (e.g., smolt abundance and size at release, and ocean distance traveled before offshore dispersal). As we discussed in section 3.1, analysis of potential limiting factors based on current freshwater habitat conditions in natal basins is a useful context for planning and prioritizing habitat restoration actions in the UCR but does not account for the potential effects of highly variable out-of-basin factors, such as ocean conditions, on abundance, survival, diversity, and extinction risk of UCR spring Chinook salmon. The ISAB suggests that a practical approach to addressing and identifying estuary-ocean information gaps and limiting factors for UCR spring Chinook would be to coordinate and collaborate in this effort with the Council's Plume and Ocean Science Forum (assuming some form of that valuable forum continues to meet) and international treaty programs aimed at addressing these issues, such as the Pacific Salmon Commission, North Pacific Anadromous Fish Commission, North Pacific Fisheries Commission, and North Pacific Marine Sciences Organization (PICES).

The ISAB considers the NOAA "Module for the Ocean Environment" (NOAA 2014) to be a highly relevant document for the current review. The NOAA Ocean Module was developed to support integrated recovery plans for four SNR ESUs including the spring/summer Chinook ESU. The document summarizes recent (through 2014) scientific information on estuary-ocean life history and ecology of SNR spring/summer Chinook and anthropogenic risks to this ESU during the estuary-ocean life stages. In the context of recovery planning and implementation, NOAA (2014) provides compelling reasons for addressing potential limiting factors during estuary and ocean life stages. For example, the tidal freshwater/estuary/plume/ocean survival of yearling hatchery-origin SNR spring Chinook is highly correlated with the overall smolt-to-adult survival ratio (SAR) but not with in-river survival during the outmigration year (2000-2009). Thus, for example, the negative effects of poor estuary-ocean conditions on SARs might mask any positive effects of freshwater habitat restoration in the basin. The document also addresses key information needs, recovery strategies, adaptive recovery actions based on ocean conditions, and monitoring and research needs from the perspective of recovery planning (NOAA 2014). The ISAB recommends development and implementation of a similar ocean module for the UCR spring Chinook ESU.

3.2.2.7. Limiting Factors and Ecological Concerns

Comparing factors limiting the abundance of two listed but widely separated runs of salmon with similar life histories can provide insights for managers (Section 3.1.1). However, comparisons between UCR spring Chinook ESU and SNR spring/summer Chinook ESU should be regarded with a degree of skepticism because of physical, chemical, and biological differences between the watersheds, differences among tributaries within watersheds, and differences in the evolutionary history of the two runs.

To isolate the effects of limiting factors, we assume that limiting factors outside the two regions act in a similar fashion on both runs: both presumably experience similar ocean conditions, similar predation issues in the estuary, and similar problems passing big dams and reservoirs in the lower river, including predation by non-native fishes. Emigrating UCR spring Chinook must pass three to five dams and SNR spring Chinook pass four before reaching McNary Dam, so dams are roughly equivalent as sources of mortality. A complicating factor not adequately considered in this comparison is the potentially complex role of hatchery fish as a limiting factor. Hatcheries produce fish that can potentially compete for resources in both regions, but their relative impacts are not discussed here. We assume that hatchery impacts in the two basin are roughly equivalent, an assumption we acknowledge may be incorrect.

In the upstream areas in the two ESUs, most spawning and rearing takes place in tributary river systems, so the limiting factors discussed here are mainly habitat factors in the tributaries. The analysis of limiting factors in tributaries by the U.S. Bureau of Reclamation Remand Habitat Workgroup (2015) is the basis for this comparison.

The analysis examined 21 factors related to habitat and gave them weighted importance ratings for major tributaries/assessment units in each region. To provide a quick insight into similarities and differences in limiting factors in the two regions, we compared the number of tributary unit occurrences of significant ratings of impairment in habitat from the 21 factors in 2015 (n =30 tributary units for UCR spring Chinook and n=59 for SNRSC, Table 3.9). Factors important for both regions were (a) water quantity, (b) sedimentation, (c) amount of instream habitat complexity, (d) condition of riparian vegetation, and (e) effects of anthropogenic barriers. Factors important to UCRSC but not to SNRSC included (a) altered primary production, (b) condition of side channels and wetlands, (c) condition of floodplains, and (d) bed and channel form. Factors important mainly to SNRSC were water temperature and recruitment of large wood. The remaining 10 factors were not considered to be limiting for spring Chinook in most areas of either region. What this analysis does not reflect is the variation among tributary units in each region; each stream or site within a stream has its own particular challenges for spring-run Chinook. Overall, this analysis should be regarded mainly as an indicator of the relative importance of each factor for the purposes of comparison, with the numbers having little meaning on their own.

The difficulty of rating limiting factors on a local scale is indicated by this statement by NMFS (2016, page 47) that reflects the large-scale problems the SNRSC face:

"Both hydropower and land use activities have had significant impacts on habitat in the mainstem Snake River above Lower Granite Dam. Twelve dams have blocked and inundated habitat, impaired fish passage, altered flow and thermal regimes, and disrupted geomorphological processes in the mainstem Snake River. These impacts have affected juvenile and adult salmon and steelhead through loss of historical habitat, altered migration timing, elevated dissolved gas levels, juvenile fish stranding and entrapment, and increased susceptibility to predation. In addition, land use activities, including agriculture, grazing, resource extraction, and development have adversely affected water quality and diminished habitat quality throughout this reach."

Table 3.9. Comparison of limiting factors for Upper Columbia River spring Chinook salmon (UCRSC) with those for Snake River spring/summer Chinook salmon (SNRSC). Numbers of tributary units in which the factor is limiting (percent in parentheses) are based on ratings by experts for each factor, provided in www.onefishtwofish.net/viz/HabitatLimitingFactors1d. All of ratings in the top three (of five) categories were included in the total. For UCRSC, 30 tributary or management units were evaluated, while for SNRSC 59 tributary/management units were evaluated.

Factor	UCRSC # (%)	SNRSC # (%)
Anthropogenic barriers	19 (63)	38 (64)
Habitat competition	0(0)	5(8)
Predation	0(0)	0(0)
Mechanical injury	0(0)	9 (15)
Altered primary production	14(47)	1(2)
Food, competition	0(0)	0(0)
Altered prey composition	0(0)	0(0)
Riparian vegetation	29(97)	40(68)
Large wood recruitment	0(0)	22(37)
Side channel, wetland conditions	16(53)	15(25)
Floodplain condition	21(70)	13(22)
Bed, channel form	28(93)	20(34)
Instream structural complexity	25(83)	37(63)
Decreased sediment quality	0(0)	1(2)
Increased sedimentation	17(57)	51(86)
Temperature	8 (27)	36(61)

Dissolved oxygen	0(0)	13(22)
Turbidity	0(0)	2(3)
Contaminants	0(0)	2(3)
Increased water quantity	0(0)	1(2)
Decreased water quantity	19 (63)	37 (63)

NMFS (2014) identified four major interrelated factors that limited viability of SNR spring/summer Chinook salmon in the tributary systems: excess fine sediment, poor water quality (primarily warm temperatures), low water quantity (primarily low summer flows), and limited habitat quantity/diversity (primarily lack of deep pools and large wood). These factors also become apparent in the analysis in Table 3.7 and are similar in both ESUs. Sediment, for example, is a problem because runoff from surrounding lands is often laden with large quantities of fine sediment that buries or cements the coarser sediment (gravel, cobbles) needed for spawning by salmon. Excess sediment can also fill in pools and otherwise decrease habitat diversity needed at all life stages. When water is diverted from the rivers, especially in summer, capacity to flush or move sediment is reduced. Low flows also increase temperatures in many stream reaches, further reducing their capacity to support salmon at diverse life history stages.

In their five-year summary of spring Chinook status in the upper Columbia, NMFS (2016) stated,

> “Despite significant efforts to improve habitat conditions, much of the habitat in the range of UCR spring-run Chinook salmon...remains degraded. Restoring habitat to historic conditions may not be needed to attain viability, but considerable improvement is needed to restore habitat to levels that will support viable populations of...spring-run Chinook salmon. In particular, the poor status of the habitat is a major obstacle to achieving UCR spring-run Chinook salmon ESU recovery.”

Barnas et al. (2015) compared ecological concerns in ESUs throughout the Columbia River Basin using the data dictionary developed by Hamm (2012). Barnas et al. (2015) quantified ecological concerns, which are about the same in the two ESUs in that SNR has some spring/summer Chinook populations with fewer and some with more concerns than does UCR spring Chinook populations (Figure 3.23).

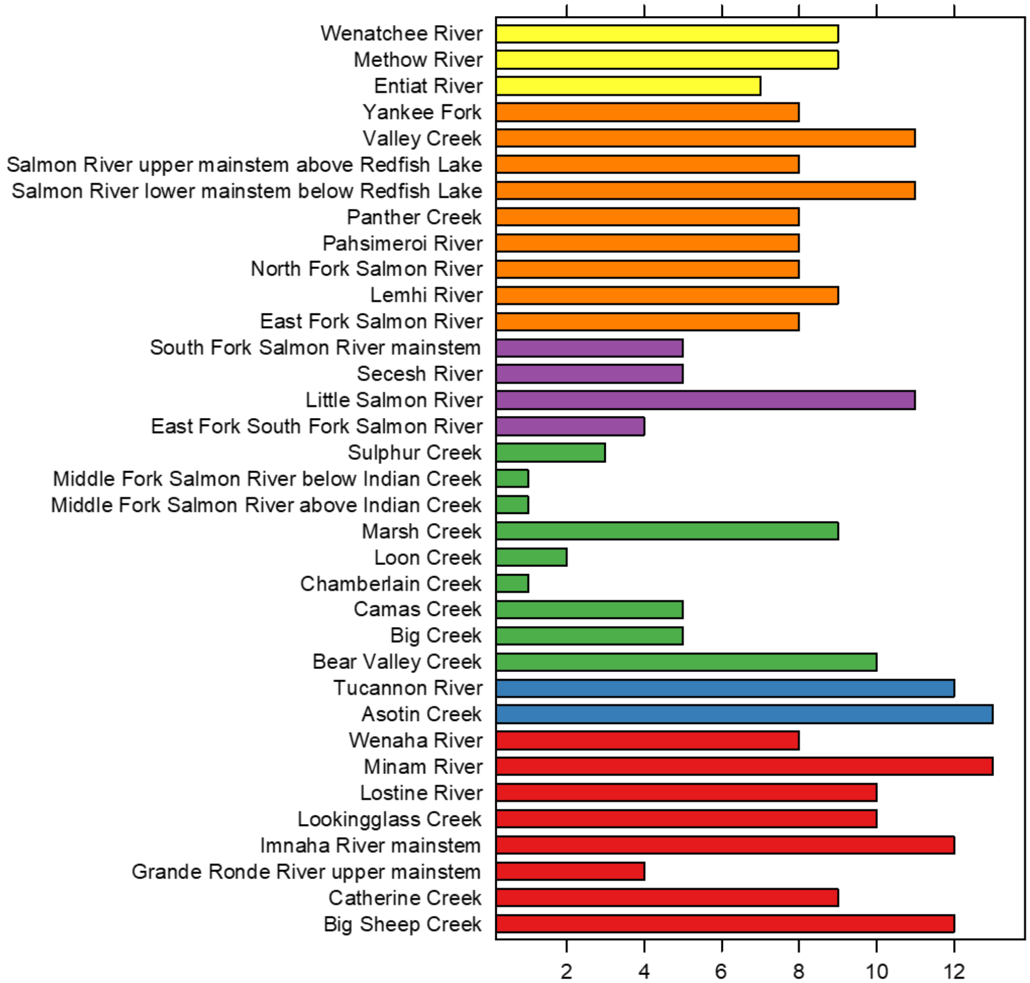


Figure 3.23. Number of ecological concerns (please see text for definition) for the three Upper Columbia spring Chinook populations (yellow bars) and Snake River spring/summer Chinook populations (all other bars). Longer bars indicate more ecological concerns. Figure is from M. Ford 2017, NOAA-Fisheries with data from Barnas et al. (2015).

3.2.3. Conclusion for Snake River and Upper Columbia Chinook Comparison

Differences in geography and biology result in differences in the spatial structure and diversity of the two evolutionarily significant units (ESUs). Nonetheless, most measures of population-level abundance and productivity for spring Chinook and assessments of habitat are similar in the two ESUs. In-river survival of spring Chinook smolts migrating in the Upper Columbia as compared to Snake River spring/summer Chinook smolts, and smolt-to-adult survival (SAR) of wild Upper Columbia and Snake River Chinook, do not appear to differ.

Because the Snake River ESU has many more populations and larger absolute abundance than the Upper Columbia ESU, the consequences of the differences in relative proportions may be worse for Upper Columbia spring Chinook than for Snake River spring/summer Chinook. If in-river survivals are equal for the two ESUs, adverse events create greater risks for the Upper Columbia populations because they are less buffered by adjacent populations. For example, if the proportion of losses to pinniped predation continues to increase but are the same for the two ESUs, the reduction of Upper Columbia spring Chinook spawners, which have smaller numbers, might reduce their ability to find mates on the spawning grounds. Thus, the ISAB believes that the Upper Columbia spring Chinook ESU may be exposed to greater risks than the Snake River spring/summer Chinook ESU. The same concern was expressed when the Upper Columbia ESU was originally listed as endangered and the Snake River ESU as threatened, listings that were not changed in the most recent reviews (NMFS 2016a; 2016b).

3.3. Comparing the Abundance, Migration Timing, and Life History Strategies of Upper Columbia River Summer and Spring Chinook

During our site visit, Upper Columbia researchers indicated that the number of natural origin summer Chinook returning to Upper Columbia subbasins was consistently greater than that observed for spring Chinook. The question of why this might be the case was raised. The fish are racial variants of the same species, therefore they are expected to be biologically similar. Studies of the alternative life-history strategies of each race may provide important insights on why Upper Columbia River summer Chinook are relatively successful while spring Chinook are less so. Such differences could identify limiting factors for spring Chinook in the UCR subbasins, and management actions could be developed to lessen their constraints on spring Chinook abundance.

Andrew Murdoch (Washington Department of Fish and Wildlife) synthesized available data on summer and spring Chinook returning to the Upper Columbia and presented this information to the ISAB (Murdoch 2017). Much of this material was produced by the monitoring and evaluation program funded by local PUDs (Chelan, Douglas, and Grant Counties). PUD scientists also contributed to the presentation. Due to the availability of data, the comparisons made were restricted to summer and spring Chinook returning to the Wenatchee, Methow, and Okanogan subbasins. Fortuitously, U.S. Fish and Wildlife Service (USFWS) researchers have been examining interactions between summer and spring Chinook in the Entiat River, which also were presented to the ISAB by Tom Desgroseillier (Desgroseillier 2017).

The Murdoch presentation contrasted the two races of Chinook at distinct life-cycle stages, based on differences observed for juveniles in freshwater, smolts before they enter the ocean, and adults after they enter the Columbia River estuary. The USFWS study used genetic tools to examine juvenile life history strategies of summer and spring Chinook salmon from the Entiat.

Life histories of UCR fish were compared to those observed in several other populations of Upper Columbia Chinook salmon. Additionally, superimposition by summer Chinook on spring Chinook redds and the possibility of hybridization between the two races in the Entiat subbasin were evaluated.

3.3.1. Comparisons between Adult Summer and Spring Chinook Salmon

3.3.1.1. General Abundance Information

From 1989 to 2016, an average of 1,241 natural origin (NOR) spring Chinook have collectively returned to the Wenatchee, Entiat, and Methow subbasins. During this time period no temporal trend in spring Chinook abundance was detected ($r^2 = 0.002$; p=0.83). During those same return years, an average of 12,572 NOR summer Chinook returned to the Wenatchee, Methow, and Okanogan subbasins. Even though summer Chinook exhibited a slight overall increase in abundance ($r^2 = 0.142$; p=0.048), numbers varied greatly (Figure 3.24). Additionally, no temporal trend appears to exist in the percentage of spring Chinook returning to upper

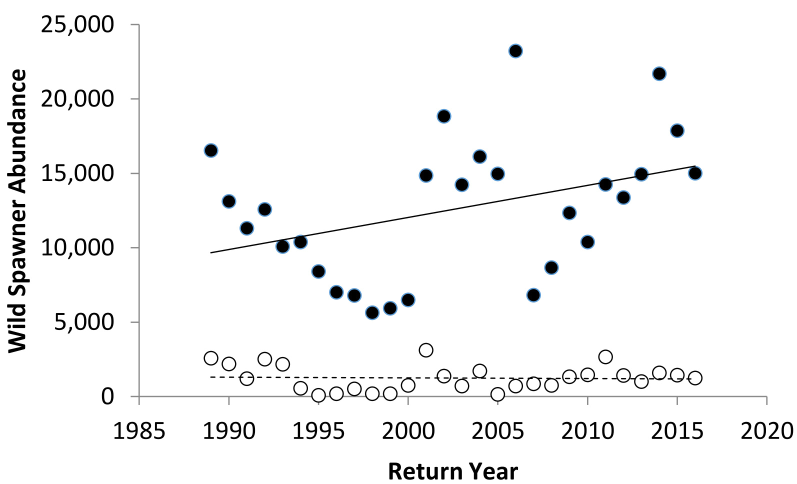


Figure 3.24. The abundance of natural origin adult summer (black circles) and spring Chinook (open circles) returning to Upper Columbia River subbasins. Overall trends in abundance from 1989 – 2016 are shown for summer (solid line) and spring (dashed line) Chinook. From Murdoch (2017).

Columbia tributaries. This percentage has averaged 8.6% but is quite variable, ranging from a low of 0.93% in 1995 when just 79 spring Chinook were counted to a high of 17.8% in 1993 when 2,179 returned.

Adult returns shown in Figure 3.24 do not include numbers of fish harvested in the Columbia River or in ocean fisheries. A sport fishery and non-selective tribal fisheries for spring Chinook occur in the lower river. In 2002, a non-selective sport fishery began for Chinook above Priest Rapids Dam. This upriver fishery was changed to a selective fishery in 2014 to limit the harvest of spring Chinook (Murdoch 2017). Addition of harvest numbers to the return values would disproportionately increase abundance of summer Chinook, making the disparity in the current abundance of naturally produced fish from these two races in the Upper Columbia even more evident.

Summer Chinook have not always been the predominate race of Chinook returning to the Upper Columbia. Counts of Upper River Columbia spring and summer Chinook began in 1933 after the Rock Island Dam was completed and have continued to the present day. The data we were able to find did not separate hatchery from natural origin fish as was done in Figure 3.24. Nevertheless, they show that the relative abundance of spring and summer Chinook in the Upper Columbia has varied over time (Figure 3.25 A&B). In some years spring Chinook were more abundant than summer Chinook. Perhaps the biggest change in relative abundance between the two Chinook races occurred in the late 1990's and early 2000's.

The patterns of yearly abundance of NOR spring and summer Chinook in Figure 3.25 led to three speculations. First, the substantial increase in NOR summer Chinook abundance in the late 1990's and early 2000's was linked to the summer spill program that was initiated under the 1995 BiOp. Second, the selective fishery started in 2014 has increased the abundance of NOR spring Chinook. And finally, the abundance of both summer and spring Chinook NORs are likely affected by some common factors. A weak but positive relationship ($r^2 = 0.20$; $p=0.017$) between the annual abundance levels of NOR spring and summer Chinook was detected when return values were correlated with one another.

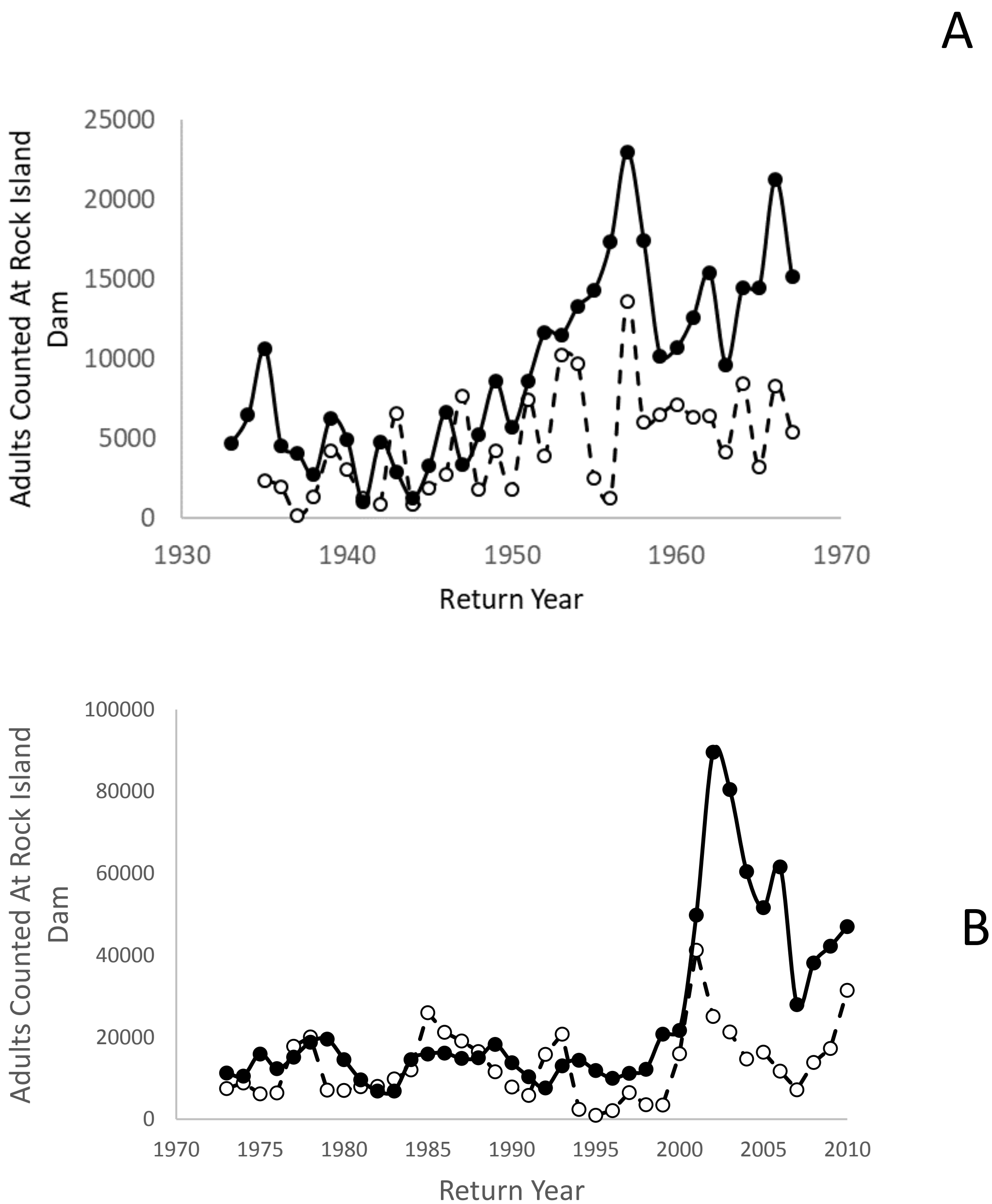


Figure 3.25 A&B. Counts of hatchery and natural origin adult summer (black circles) and spring Chinook (open circles) at Rock Island Dam from 1933 through 1968 (part A) and from 1973 through 2010 (part B). Jacks (3-yr-old males) are included in the counts made from 1933 to 1950. (Data are from Chelan PUD www.chelanpud.org/departments/yourPUD /RIAnnualFishCount.pdf)

Return numbers can provide a general demographic overview of how population abundance varies overtime, but grouping returns by brood year provides a number of important advantages. Foremost among those is the development of recruitment curves that can determine intrinsic productivity and the population carrying capacity in existing habitat (ISAB 2015-1). Additionally, smolt-to-recruit (SARs) estimates for each brood year can be made. SAR values could be used to directly test the effects of the summer spill program on survival and also allow a more refined assessment of how summer and spring Chinook abundance may be related. The advantages of using brood year returns as opposed to annual returns have led us to make several suggestions the researchers may wish to undertake if they are not already underway:[4]

- Add harvest values (by age) to the return numbers for years 1989 – 2016
- Evaluate trends of abundance by brood year rather than by return year.
    - Anadromous spring Chinook reach maturation at ages 3, 4, and 5. Summer Chinook also mature at these ages and a few may return at age 6. If age data are available, it will be possible to examine the abundance of fish returning from the 1986 brood year up to the 2010 brood year. This would create a dataset with 25 brood years as opposed to the existing one with 28 return years.
- Use return data to populate brood year returns by subbasin rather than aggregating them into a single number for each race of Chinook salmon. Summer and spring Chinook returning to the Wenatchee River pass over two fewer dams than those returning to the Methow and Okanogan River and therefore may be affected by different stressors.

3.3.1.2. Adult Entrance Timing, Mainstem Migration Speed and In-River Survival

Populations of Upper Columbia River spring Chinook enter the Columbia estuary and pass over Bonneville Dam at consistently earlier dates than summer Chinook (Murdoch 2017, Sorel et al. 2017). A comparison of the cumulative passage of dates of Upper Columbia spring and summer Chinook over Bonneville Dam, for instance indicates that this difference is greater than 40 days and often exceeds 50 days throughout the migration period (Table 3.10). Sorel et al. (2017) found that survival of adult Chinook moving upstream from Astoria to Bonneville (rkm 44 to rkm 233) was low and decreased substantially from 2010 to 2015. Stressors and threats in the

[4] Our request to compare the survival and life history strategies of spring and summer Chinook returning to upper Columbia subbasins occurred after our site visit to the upper Columbia. We appreciate the efforts undertaken to carry out this unexpected and new assignment. The scientists performing this work indicated that additional comparisons will be made in the future. It is likely that yearly return data will be deconstructed and assembled into brood year survival and abundance data. The suggestions for future analyses were made to affirm the importance of producing and analyzing brood year information.

estuary and lower river had a substantial impact on adult survival and there was a “gradient of risk.” Stocks that had early arrival and transit timing were more vulnerable than those that exhibited later migration timing patterns.

Table 3.10. Percentage of Upper Columbia River spring and summer Chinook passing over Bonneville Dam by date. Data provided by Andrew Murdoch (pers. comm.)

	Cumulative Percentage and Arrival Dates of Upper Columbia Spring & Summer Chinook Passing Bonneville Dam				
Race of Chinook	5%	25%	50%	75%	90%
Spring	April 20	May 1	May 13	June 4	June 25
Summer	June 13	June 24	July 4	July 21	Aug 5
Difference (days)	54	54	52	47	41

Migration timing varied slightly from one year to the next, but the order in which populations entered the Columbia was consistent (Sorel et al. 2017). Additionally, the amount of time needed to travel from the estuary to Bonneville decreased from late-winter and early spring to mid-June. Fish entering the river in late March took an average of 30 to 40 days to pass through the Lower River, while those making this passage in mid-June needed only 5 to 10 days to transit this same portion of the river (Sorel et al. 2017). Thus, it is likely that adult Upper Columbia River spring Chinook may spend over a month in this part of the river compared to a week or less for summer Chinook.

Timing of river entry and residency in the lower river strongly affect population-specific survival rates (Table 3.11). Lower survival may be related to increases in pinniped abundance in the estuary and lower river. Surveys conducted by WDFW and ODFW indicated that up to 2,000 to 3,000 California sea lions and 1,000 Stellar sea lions were in the lower Columbia River near Astoria during the spring in 2015. Since the 1980’s “California sea lions have been moving in increasing numbers farther and farther up the Columbia River—first to the Astoria area then to the Cowlitz and on to Bonneville Dam, 145 miles from the river mouth”(wdfw.wa.gov/help/questions/261/Are+sea+lions+native+to+the+Columbia+River%3F). The continuing temporal increase in pinniped numbers may be an important factor limiting Upper Columbia spring Chinook abundance.

Table 3.11. Return year and arrival timing relationships to the survival of adult Chinook salmon in the lower Columbia River. Upper Columbia River spring Chinook stocks are shown in bold. Data are from Sorel et al. 2017. Upper Columbia River populations are in bold font.

Chinook Salmon Population	Lower River Survival Rates 2010 - 2012	Lower River Survival Rates 2013 - 2015
Early Arriving Populations Lemhi River Marsh Creek Upper Grande Ronde River Catherine Creek Tucannon **Methow**	69% - 81%	50% - 70%
Intermediate Arriving Populations Upper Salmon River Big Creek Minam River **Entiat River** **Wenatchee River**	79% - 88%	67% - 85%
Late Arriving Populations Pahsimeroi River Upper South Fork Salmon River East Fork South Fork Salmon River Secesh River Imnaha River Lostine River	84% - 92%	83% - 92%

3.3.1.3. Pre-Spawning Mortality and Hatchery Wild Composition on Spawning Grounds

[This section was updated April 10, 2018; see update statement and italicized text. Table 3.12 was removed.]

Survival and migration speed of adult spring and summer Chinook in the mainstem of the Columbia River were compared from McNary Dam to Rocky Reach Dam. Survival rates for adult Chinook passing through different reaches of the mainstem were estimated by using "conversion rates", obtained from the Columbia River DART website. For Chinook originating above Rocky Reach Dam, these rates were calculated by dividing the number of PIT-tagged adults detected passing Rocky Reach Dam by the number detected passing McNary Dam, a *distance of 292 rkm. Conversion rates were 0.94 for spring Chinook and 0.88 for summer Chinook, a difference that was not statistically significant ($t = 2.365$, $p = 0.150$).*

The conversion rates for spring and summer Chinook were not corrected for harvest between McNary and Rocky Reach dams. Sport harvest in the Upper Columbia River primarily focuses on summer Chinook and largely does not occur when spring Chinook are migrating. From 2010-2016, an average of 8 spring Chinook were harvested annually from McNary Dam to Rocky Reach Dam, in contrast to an average of 1,600 summer Chinook harvested and retained annually (data from Paul Hoffarth and Travis Maitland, WDFW). Sport harvest amounts to 3.3% of the adults that potentially would have passed through Rocky Reach Dam during 2010-2016 (annual adult passage plus sport harvest). If conversion rates for summer Chinook were increased by 3.3% to account for the effect of harvest, the conversion rates for spring and summer Chinook during this period would be similar (0.94 and 0.91, respectively).

CRITFC reports migration rates for spring and summer Chinook in the Upper Columbia River to BPA (Fryer et al. 2011, 2012, 2013a, 2013b, 2015a, 2015b, 2017). Based on their estimates, mean travel time was 9.53 days for spring Chinook from 2009-2016 and 8.87 days for summer Chinook. Statistically, these travel times do not differ significantly ($t = 1.349$, $p = 0.226$). We expected travel times of spring Chinook to be longer than travel times for summer Chinook because river flows in the Columbia are typically higher and water temperatures lower during the upstream migration period for spring Chinook, but the observed migration rates do not support this conclusion.

Upper Columbia River spring Chinook also enter spawning tributaries earlier than summer Chinook and hold in different areas prior to spawning. Approximately 50% of the spring Chinook entering the Methow and Entiat rivers arrive by the first week in May. Spring Chinook returning to the Wenatchee arrive slightly later, usually in the last two weeks of May. On average, 50% of the summer Chinook arrive during the first week in July, approximately 40+ days later. Spring Chinook typically move into the upper reaches of streams and hold in areas with deep cover that are adjacent to their eventual spawning locations. In contrast, summer Chinook hold in mainstem pools (Murdoch 2017).

Pre-spawning mortality of Upper Columbia River spring and summer Chinook females has been estimated for the past 16 years. These estimates were calculated by dividing redd numbers by the number of females entering a subbasin. The assumption was made that each surviving female would construct a single redd. The overall average ratio of redds per female for summer Chinook was 0.85 (95% C.I. = 0.75 – 0.95), indicating that pre-spawning mortality for summer Chinook in Upper Columbia subbasins is around 15%. Spring Chinook females were observed to have substantially lower pre-spawning survival rates. The ratio of redds per female for this race was 0.54 (95% C.I. = 0.46 – 0.62) which implied that on average close to half (46%) of the spring

Chinook females perished prior to spawning. Additionally, there is some evidence of density dependent mortality in spring Chinook as there was a positive relationship between pre-spawning mortality and female abundance. Density dependent mortality was not observed in summer Chinook females (Murdoch 2017).

UCR scientists plan to determine what habitat attributes are needed by spring Chinook while they hold and mature prior to spawning (Murdoch 2017). For example, do cold water refuges, baseflows, and water temperatures limit survival, or is survival dependent upon complex in-river cover, dense riparian vegetation and cut banks, or a combination of all of these or other factors? Such information could guide future habitat restoration actions and improve pre-spawning survival rates.

Hatchery programs for both summer and spring Chinook occur in the Upper Columbia to supplement natural populations or produce fish exclusively for harvest. In supplementation programs, hatchery adults are allowed to escape into natural spawning areas. Hatchery fish produced for harvest may also arrive on spawning grounds. Proportions of hatchery origin spring and summer Chinook adults spawning in nature (pHOS) and proportions of natural origin adults used as hatchery broodstock (pNOB) in the Wenatchee, Methow, and Okanogan have been estimated annually from 1989 to 2016.

These two statistics, pHOS and pNOB, are commonly used to calculate a ratio referred to as the Proportionate Natural Influence or PNI.[5] PNI values approximate the degree to which a population of salmon consisting of both hatchery and natural origin spawners maintains the original natural population's genetic properties. The higher the PNI value the greater the genetic affinity to the natural population. The Hatchery Scientific Review Group, which co-developed this concept, recommends that PNI values in integrated hatchery programs should be greater than 0.67. PNI values for spring and summer Chinook were estimated from mean values provided by Andrew Murdoch (Table 3.13). The advent of selective sport fisheries above Priest Rapids Dam since 2014 have substantially reduced pHOS values (Murdoch 2017) so current PNI values are likely greater than those shown in Table 3.12. Nonetheless, past PNI

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[5] The general formula for PNI is: pNOB/(pNOB + pHOS). Recently the pHOS term has been modified in recognition that hatchery fish may not produce as many adult offspring as natural origin adults. This new term is referred to as $pHOS_{eff}$ which equals RRS * $pHOS_{census}$. RRS is an estimate of the relative reproductive success of hatchery adults when compared to natural origin counterparts. Suppose that RRS is estimated to be 0.8, in this instance a census pHOS of 50% would be reduced down to 40% resulting in a higher PNI than if the original $pHOS_{census}$ value had been used. For a detailed explanation of the PNI concept see HSRG (Hatchery Scientific Review Group). 2014. On the Science of Hatcheries: An updated perspective on the role of hatcheries in salmon and steelhead management in the Pacific Northwest. A. Appleby, H.L. Blankenship, D. Campton, K. Currens, T. Evelyn, D. Fast, T. Flagg, J. Gislason, P. Kline, C. Mahnken, B.Missildine, L. Mobrand, G. Nandor, P. Paquet, S. Patterson, L. Seeb, S. Smith, and K. Warheit. June 2014.
Available online: hatcheryreform.us

values suggest that Upper Columbia River spring Chinook populations in the Wenatchee and Methow were likely exposed to higher levels of potential domestication selection than summer Chinook.

Table 3.13. Estimated pNOB, pHOS, and PNI values for Upper Columbia River Chinook salmon populations; mean pNOB and pHOS values (1989 – 2016) were obtained from Andrew Murdoch (pers. comm.).

Upper Columbia River Subbasin	Race of Chinook Salmon	% Natural Origin Broodstock (pNOB)	% Hatchery Origin Spawners (pHOS)	Proportionate Natural Influence
Wenatchee River	Summer	92.4	13.8	0.869
	Spring	54.6	45.0	0.548
Methow	Summer	74.3	33.1	0.691
	Spring	37.0	57.4	0.391
Okanogan	Summer	74.3	40.7	0.646

3.3.1.4. Hybridization and Redd Superimposition

In 2006, the USFW began a RME study to measure redd superimposition and hybridization rates between summer and spring Chinook salmon in the Entiat subbasin. Adult summer and spring Chinook were found contemporaneously in portions of the Entiat River. Stream surveys were conducted from August 9 through November 15, and an average of 441 spring and summer Chinook redds were counted during 2006-2014. Generally, 60 or more were built when both races were in the river together, suggesting hybridization was possible on ~14% of the Chinook redds. Male summer Chinook could have been present at 9% (17/197) of the spring Chinook redds, while spring Chinook males were potentially present at 18% (45/244) of the summer Chinook redds (Figure 3.26; Desgroseillier et al. 2017). Hybridization was assessed on approximately 1,000 Chinook juveniles leaving the Entiat River in 2011 and 3.6% (32) were found to be hybrids. Additional hybrids were discovered rearing in the Entiat during the winter. To date it has not been possible to determine whether hybrids have returned as adults or contributed to the recruitment of juveniles. However, an examination of the effects of hybridization between fall-run Chinook salmon (tule and up-river bright Chinook) in the Columbia Basin (Smith and Engle 2011) concluded that hybrid juveniles were not viable and thus likely reduce overall population productivity (Desgroseillier et al. 2017). Whether the same is true for spring x summer hybrids is not known.

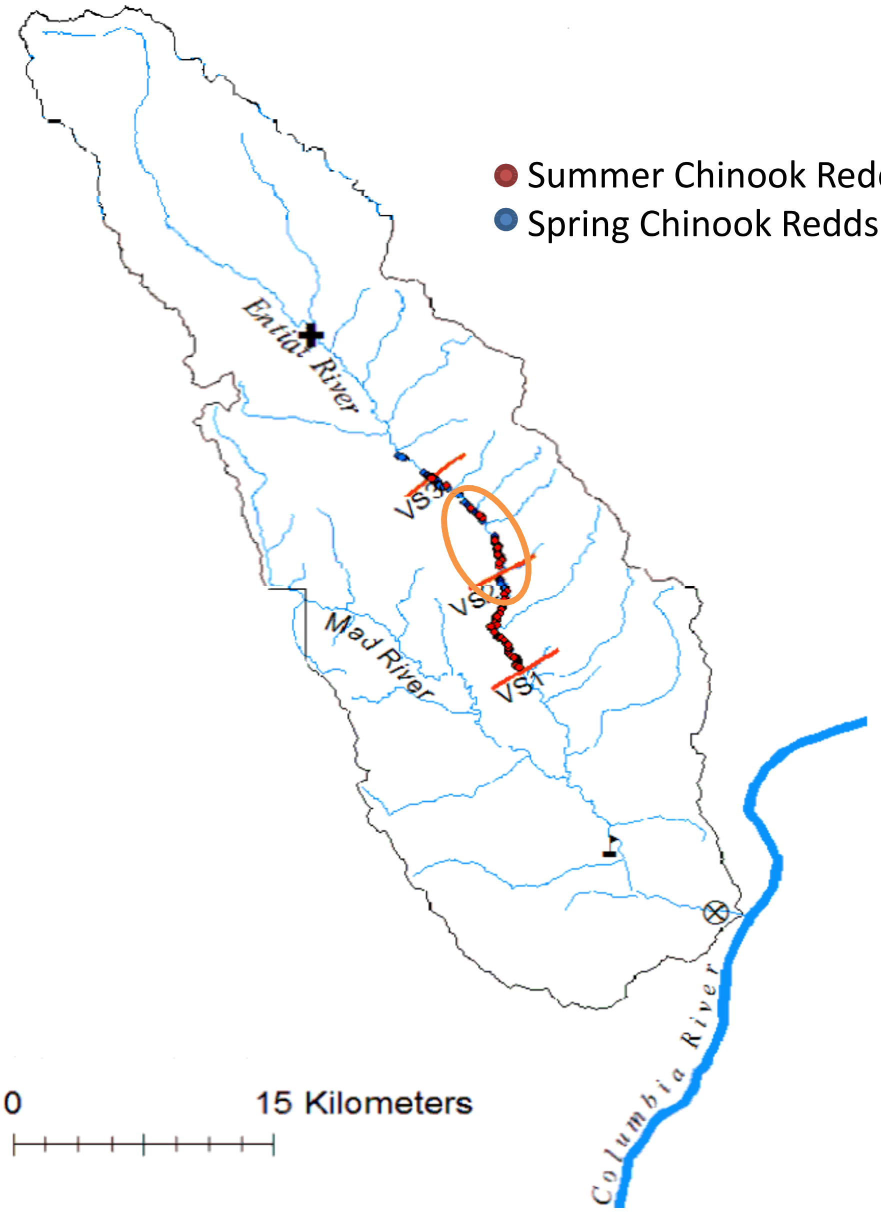

Figure 3.26. The spatial distribution of spring and summer Chinook salmon redds in the Entiat subbasin. Overlap of redds occurred in Valley Segments (VS) 2-3. Orange oval indicates area where temporal and spatial overlap occurred. (Image taken directly from Desgroseillier et al. presentation (2017)).

Redd superimposition or the multiple use of a spawning site within a spawning season was also examined in the Entiat River. Because of differences in maturation timing, summer Chinook females may excavate nest pockets in locations that were recently used by spring Chinook. Murdoch (2017) reported that USFW researchers had observed superimposition on ~19% of the spring Chinook redds they surveyed in the Entiat. The impact that superimposition may have on egg-to-fry survival can be variable and depends upon a number of factors. For instance, Chinook eggs are fairly immune to mechanical shock if embryos have accrued 190 temperature units C^O prior to dislodgement or movement (Jensen 2003). Additionally, nest depth is related to female length and larger females are expected to create deeper nests (Steen and Quinn 1999). Because spring Chinook are generally smaller than summer Chinook, their deposited eggs may be particularly vulnerable to dislodgement due to the digging activities of larger summer Chinook. Finally, the majority of eggs deposited by a female are often placed into her first two nests (de Gaudemar et al. 2000). These will be located at the tail end of a redd and are often covered by a substantial layer of gravel placed there by the final digging actions of a female. Typically, newly arriving females are attracted to the anterior portions of pre-existing redds. This is where the previous occupant has created a depression or "pot" which accelerates stream flows and accentuates water interchange into the substrate. Knowing when and which portions of a redd were disturbed would help clarify the potential damage caused by multiple usage of the same spawning area.

Temporal and spatial overlap information on adult spring and summer Chinook is likely available for the Wenatchee and Methow subbasins. This information should be used to determine the degree of overlap that exists between these two races in the above subbasins. Since spawner density is known to affect distribution patterns (Murdoch 2017), multiple year assessments should take place. Additionally, analyses on hybridization rates and the occurrence of redd superimposition similar to those performed in the Entiat are encouraged in the Wenatchee and Methow subbasins. Such analyses would increase our understanding of the relative importance of these factors on spring Chinook productivity. Are redd superimposition or hybridization major factors constraining spring Chinook abundance or do they occur rarely and are of little demographic importance?

3.3.2. Comparisons between Juvenile Summer and Spring Chinook Salmon

Juvenile trapping operations occur throughout the Upper Columbia Basin and are implemented to keep track of the abundance and timing of juvenile salmonid migrations. Such data are not only useful for forecasting future adult abundances but also allow managers and researchers to potentially detect changes in freshwater productivity, survival, growth, and alterations in juvenile life histories. Recently, genetic methods have been used to identify the racial origin of out-migrating juvenile Chinook salmon. These studies revealed that both juvenile spring and summer Chinook had more diverse out-migration patterns than previously thought.

3.3.2.1. Juvenile Life Histories

In Upper Columbia River subbasins that support both summer and spring Chinook juveniles, fisheries managers have assumed that subyearling emigrants leaving in the early summer (e.g., June and July), are summer Chinook while those emigrating as yearlings in late fall through the early spring are spring Chinook. Similarly, 0+ spring Chinook smolts were believed to leave beginning in August, peaking as new yearlings in mid- to late October and finishing as 1+ migrants in late December (Desgroseillier et al. 2017). Recent genetic analyses of tissue samples from juvenile migrants revealed a much more complicated suite of emigration patterns than expected (Desgroseillier et al. 2017).

Samples collected from mid-September through November revealed the expected upsurge in spring Chinook juveniles. However, a surprisingly high percentage (~57%) of the subyearlings leaving the Entiat in July were spring Chinook. This meant that thousands of juveniles that previously had been assumed to be summer Chinook were actually spring Chinook. Samples obtained from juveniles rearing in the Entiat during the summer and winter of 2012, 2013, and 2014 indicated that the percentage of each race rearing in different valley segments of the Entiat remained relatively constant from one year to the next. During the summer sampling period, a positive correlation between rkm and the proportion of spring Chinook juveniles rearing in the subbasin was found. This tendency was not seen during the winter sampling period suggesting that spring Chinook juveniles were moving downstream, most likely seeking overwintering rearing areas. Surprisingly, a relatively high percentage (~12%) of summer Chinook overwintered in the Entiat subbasin (Desgroseillier et al. 2017).

The results of the genetic analyses on sampled Chinook juveniles have a number of management implications that were emphasized by Desgroseillier et al. (2017). First, undervaluing the number of spring Chinook juveniles produced from the Entiat has inflated SAR values and simultaneously underestimated egg-to-fry survival and emigrant per redd values. Incorrect values in these metrics may disguise density dependence, obscure locations in the life cycle where bottlenecks are occurring, and possibly lead to incorrect productivity or recruit/spawner ratios and erroneous conclusions about the capacity of the environment to produce juvenile spring and summer Chinook.

Second, the distribution patterns shown in Table 3.14 led Desgroseillier et al. (2017) to make two general recommendations about where habitat restoration for Chinook juveniles ought to be focused. If the goal is to enhance spring Chinook capacity during their first spring and summer in freshwater, then upper portions of the Entiat should be targeted for restoration where spring Chinook are concentrated. Conversely, restoration actions designed to provide overwintering habitat for juvenile Chinook should be distributed throughout the subbasin. This strategy would also benefit overwintering summer Chinook.

Table 3.14. Distribution patterns of summer and spring Chinook salmon juveniles rearing in the Entiat subbasin during the summer (August) and winter (March) as revealed by genetic analyses of 542 samples (405 from the summer period and 137 from the winter period). See Figure 3.26 for locations of valley segments. Percentage data are from Tom Desgroseillier (pers. comm.)

		Summer Sampling Period (August)		Winter Sampling Period (March)	
Sampling Area (Valley Segment)	Location (River Kilometer)	% Summer Chinook	% Spring Chinook	% Summer Chinook	% Spring Chinook
VS-1	4.2	47	53	0	100
VS-1	7.4	52	48	0	100
VS-1	10.3	52	48	0	100
VS-1	23.4	8	92	43	57
VS-2	28.1	52	48	29	71
VS-2	31.6	24	76	20	80
VS-3	36.7	12	88	6	94
VS-3	40.2	13	87	8	92
VS-3	42.7	5	95	0	100
VS-3	44.6	5	95	8	92

The abundance of yearling summer Chinook in the Entiat appears to be greater than abundances in other Upper Columbia subbasins. Genetic assignments of yearling Chinook rearing in the Wenatchee River over six sampling years (2004 – 2007 and 2012) indicated a lower level of yearling summer Chinook (mean of 3.2%, range 0.9% - 7.2%). Similarly, a single year of genetic analyses of overwintering Chinook in the Methow also showed just 1.7% of the yearlings were summer Chinook (Murdoch 2017). The Entiat is a smaller and less complex subbasin than the Wenatchee and Methow subbasins, which may explain why a substantial percentage of spring Chinook emigrate as subyearlings from the Entiat. However, the higher relative incidence of yearling summer Chinook in the Entiat remains to be explained. It appears that the upper portions of valley segment 1 (VS-1) and valley segment 2 (VS-2) are favored by this race as overwintering locations (Table 3.14). A careful inventory of habitat attributes in this portion of the Entiat might provide information about overwintering habitat and discern if the high percentage of summer Chinook in this portion of the river simply is due to a paucity of juvenile spring Chinook at these locations.

Rotary screw trap (RST) data from the Upper Columbia subbasins indicated that a substantial portion of Chinook juveniles exit their natal streams as subyearlings in late June and July. This exodus of subyearling Chinook prompted Desgroseillier et al. (2017) to investigate where both spring and summer subyearlings went after they left the Entiat subbasin. PIT tag detections at tributary arrays and at mainstem dams on the Columbia were used to help decipher the migratory and rearing strategies used by parr from the two races of Chinook. Additionally, the residency and migratory patterns exhibited by Entiat spring chinook parr were compared to those exhibited by spring Chinook parr originating from the Chiwawa (Wenatchee subbasin), Chewuch, and Twisp rivers (both in the Methow subbasin). This evaluation was conducted to see if subbasin origin affected parr migratory behavior. The migratory patterns of yearling smolts produced by both races were also tracked and compared.

Juvenile trapping data showed that large numbers of subyearling Chinook out-migrate as fry (< 50 mm FL) for weeks to months from the Entiat and the Wenatchee subbasins (Murdoch 2017; Desgroseillier et al. 2017). No assessments have been made on the percentage of the fry that are of summer or spring origin. Additionally, little is known about fry behavior and survival once they enter mainstem reservoirs. More is known about parr (> 50 mm) who continuously out-migrate throughout summer and fall months. Desgroseillier et al. (2017) found that spring Chinook subyearlings from the Entiat migrated downstream and overwintered in mainstem reservoirs. None were observed directly migrating to the estuary as subyearlings. Conversely subyearling summer Chinook parr that emigrated in mid-July through early September often migrated directly downstream to the estuary. Summer Chinook that emigrated from the Entiat later in the fall also migrated downstream but often overwintered in mainstem reservoirs prior to entering the estuary during the following spring.

The general tendency of Entiat subyearling spring Chinook to enter the estuary as yearlings was similar to the life history patterns of spring Chinook in the Twisp, Chiwawa, and Chewuch subbasins. PIT tag detections on these fish showed that a large portion moved downstream ≤150 rkm before overwintering. Unlike the Entiat fish, however, almost none of the Methow or Wenatchee subyearlings left their natal streams prior to overwintering in downstream reaches in the subbasins (Desgroseillier et al. 2017)

Fish that stayed in their natal subbasins and emigrated as yearlings entered the mainstem from early February through the end of May. Spill to facilitate juvenile emigration in the mainstem begins in mid-April and by that time almost 60% of the yearling Chinook smolts (largely spring Chinook smolts) have already entered the mainstem. Conversely, less than 10% of summer Chinook subyearlings have entered the mainstem before the onset of spill. Thus, the later migration timing of summer subyearlings matches well with the increased spill regime (Murdoch 2017) and they are expected to have lower mortality rates due to dam passage.

The life history patterns revealed by RST trapping and PIT Tag detections showed that the life history patterns of both summer and spring Chinook juveniles in the Upper Columbia are more complex than originally thought. Assumed and observed life history patterns of Entiat spring and summer Chinook are compared in Table 3.15. This table and data from the Murdoch (2017) and Desgroseillier et al. (2017) indicate that:

- Summer Chinook have greater juvenile life history diversity than spring Chinook and therefore may have greater intrinsic resilience than spring Chinook
- Summer Chinook enter the estuary as subyearlings and yearlings over a much broader span of time than spring Chinook
- Later out-migration timing of summer Chinook subyearlings coincides with the increased spill regime that was established to reduce dam related mortalities. Their outmigration timing is also better synchronized with increased flows and higher turbidity due to the spring run-off.
- Smaller size at emigration may make summer Chinook less vulnerable to bird predation if birds prefer larger prey (e.g., Caspian Terns)
- Summer Chinook typically rear for shorter periods in tributaries than spring Chinook and are less likely to experience capacity limitations or survival bottlenecks in subbasins
- Contrary to previous thought, some Upper Columbia spring Chinook juveniles rear in mainstem reservoirs for prolonged periods
- The lower 235 rkm between Bonneville Dam and the ocean was identified as a feeding area for juvenile salmonids where competition for resources with other species and age classes may occur

Table 3.15. Comparisons between assumed and observed life history patterns in Upper Columbia River summer and spring Chinook juveniles. Newly acquired data indicate unexpected mainstem use by spring Chinook and tributary rearing and overwintering by summer Chinook. Table modified from Desgroseillier et al. (2017).

CLASSIC LIFE HISTORY (ASSUMED) OF ENTIAT CHINOOK SALMON								
		Season						
RACE	LOCATION	FALL	WIN	SPR	SUM	FALL	WIN	SPR
Summer	Gravel	X	X					
	Emergence			X				
	Tributary			X	X	X		
	Columbia R.				X	X		
	Estuary				X	X		
	Ocean				X	X		
Spring	Gravel	O	O					
	Emergence			O				
	Tributary			O	O	O	O	O
	Columbia R.							O
	Estuary							O
	Ocean							O

OBSERVED LIFE HISTORY OF ENTIAT CHINOOK SALMON								
		Season						
RACE	LOCATION	FALL	WIN	SPR	SUM	FALL	WIN	SPR
Summer	Gravel	X	X					
	Emergence			X				
	Tributary			X	X	X	X	X
	Columbia R.			X	X	X	X	X
	Estuary			X	X	X		X
	Ocean			X	X	X		X
Spring	Gravel	O	O					
	Emergence			O				
	Tributary			O	O	O	O	O
	Columbia R.			O	O	O	O	O
	Estuary							O
	Ocean							O

Studies demonstrate that mainstem reservoirs and the lower reach of the Columbia provide important rearing and overwintering areas for Upper Columbia River Chinook (e.g., Desgroseillier et al. 2017) and illustrate the need for additional mainstem research. Some studies have assessed the effects of channel catfish, smallmouth bass, and walleye predation on juvenile salmonids, but holistic, regional multi-species studies are needed to understand cumulative effects (Sanderson et al. 2009). Additionally, possible effects of competition between juvenile salmonids and American Shad for planktonic food in the lower river and estuary is yet to be determined.

3.3.2.2. Juvenile Life History Strategies and the Production of Adults

Both Murdoch (2017) and Desgroseillier et al. (2017) examined the juvenile life history strategies employed by adult salmon returning to Columbia River subbasins. Desgroseillier et al. (2017) compared adult return rates (SARs) among three types of juveniles returning to the Entiat; (1) yearlings, (2) subyearlings that emigrated from the Entiat in June – September, and (3) subyearlings that left during the October-November out-migration period. They also compiled data on Entiat adults that were grouped by rearing area; (1) yearlings that stayed in the Entiat, (2) reservoir yearlings, and (3) subyearlings that immigrated directly to the ocean. Out of the two yearling strategies employed, those that resided in mainstem reservoirs had higher SAR values than yearlings that had remained in the Entiat prior to out-migration. Additionally, juveniles that emigrated from the Entiat and entered the estuary as subyearlings (summer Chinook) also had higher SAR values than yearling fish staying in the Entiat. Juveniles that reared in reservoirs and out-migrated as yearlings had the highest SAR values (Table 3.16). At present the factors responsible for these patterns in SAR values are not entirely understood.

Table 3.16. Smolt-to-adult return rates (SARs) in Chinook salmon returning to the Entiat that adopted different life histories (yearling and subyearling) and juvenile rearing areas (natal stream, reservoir, and ocean) by brood year. Data from Tom Desgroseillier (pers. comm.)

	SARs in Yearling and Subyearlings			Rearing Area Effects on SARs		
Brood Year	Yearling	Sub Yearling (Jun – Sep)	Sub Yearling (Oct –Nov)	Natal Stream Yearling	Reservoir Yearling	Ocean Type Subyearling
2006	0.80%	0.12%	0.56%	0.80%	2.99%	N/A
2007	1.34%	N/A	0.17%	1.34%	3.64%	N/A
2008	0.90%	N/A	0.20%	0.90%	N/A	N/A
2009	0.28%	0.11%	0.20%	0.28%	0.75%	0.51%
2010	0.82%	0.17%	0.26%	0.82%	2.81%	1.18%
2011	1.08%	0.36%	0.39%	1.08%	1.20%	2.25%

Murdoch (2017) examined the effect of juvenile life history on adult production with a different approach. He documented the percentage of adults that had utilized a yearling, reservoir yearling, and subyearling juvenile strategy on a return year basis. This was estimated for adult Chinook returning to the Wenatchee, Methow, and Okanogan Rivers for return years 1993 – 2015. Percentage values from Andrew Murdoch (pers. comm.) were used in Kendall's Tau correlations to see if the proportion of adults using the subyearling, reservoir, and yearling strategies were similar in each subbasin for each return year. The proportion of adults using each strategy in each return year was remarkably similar among the three subbasins (Table 3.17). Additionally, a substantial increase in the subyearling strategy was observed along with a corresponding decrease in the yearling and reservoir yearling strategies in adult Chinook from the early 1990s to the present. Representative scatter plots illustrate these trends in Figure 3.27.

Table 3.17. Results of Kendall's Tau correlations conducted on the percentages of adults possessing different juvenile life histories that returned in the same year but to different Upper Columbia subbasins (performed to see if adults returning in the same year but to different subbasins used similar juvenile life histories). Percentage data obtained from Murdoch presentation (2017).

Subbasins	Correlation	Juvenile Life History Type: Subyearling Tau	Subyearling p Value	Reservoir Rearing Tau	Reservoir Rearing p Value	Yearling Tau	Yearling p Value
Wenatchee & Methow	% of adults in the Wenatchee vs. % of adults in the Methow with the same juvenile life history	0.836	<.01	0.737	<.01	0.605	<.01
Wenatchee & Okanogan	% of adults in the Wenatchee vs. % of adults in the Okanogan with the same juvenile life history	0.787	<.01	0.649	<.01	0.512	<.01
Methow & Okanogan	% of adults in the Methow vs. % of adults in the Okanogan with the same juvenile life history	0.810	<.01	0.735	<.01	0.728	<.01

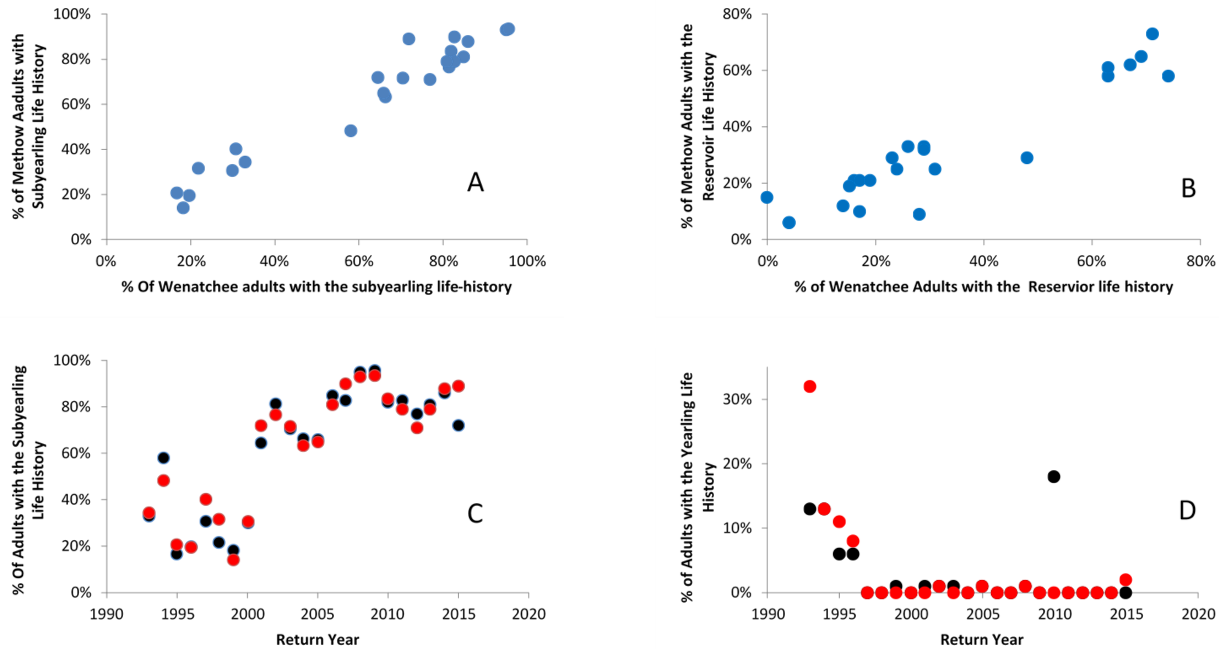

Figure 3.27. Percentage of natural origin adults returning in the same year to the Wenatchee and Methow subbasins that utilized the juvenile subyearling (A) and reservoir (B) strategies and percentage of Wenatchee (black circles) and Methow (red circles) of naturally produced adults that used the subyearling (C) and yearling (D) life histories by return year. Data from Murdoch (2017).

The marked increase in the subyearling strategy and corresponding decrease of the yearling strategy in all three subbasins might be due to conditions in the subbasins or conditions they encountered in the mainstem after leaving their natal subbasins. Exploring how environmental and biological variables (e.g., flow, water temperature, water travel time, predator abundance, habitat attributes) influence the survival of fish using these different life history strategies may help identify population bottlenecks and guide future restoration actions.

3.4. Pinniped Predation

3.4.1. Background Information

The Upper Columbia Salmon Recovery Board (UCSRB) Recovery Plan (Final 9-13-2007) identifies pinnipeds**,** including harbor seals (*Phoca vitulina*), California sea lions (*Zalophus californianus*), and Stellar sea lions (*Eumetopia jubatus*), as the primary marine mammals preying on Chinook and steelhead originating from the Upper Columbia Basin. As part of the Recovery plan, "the UCSRB supports immediate adoption of more effective predator control programs, including lethal removal when necessary, of the marine and avian predators that have the most significant negative impacts on returns of Upper Columbia Basin ESA-listed salmonid fish stocks."

The 2016 five-year Upper Columbia status report provides an updated review of information on pinniped predation and predatory pinniped management (through 2015), including the following conclusions and recommendations:

> "The effect of marine mammal predation on the productivity and abundance of Columbia River basin salmon and steelhead stocks has not been quantitatively assessed at this time. The absolute number of animals preying upon salmon and steelhead throughout the lower Columbia River and Willamette River is not known. The information available since the last status review clearly indicates that predation by pinnipeds on listed stocks of Columbia River basin salmon [including Upper Columbia spring Chinook] and steelhead, as well as eulachon, has increased at an unprecedented rate. So while there are management efforts to reduce pinniped predation in the vicinity of Bonneville Dam, this management effort is insufficient to reduce the severity of the threat, especially pinniped predation in the Columbia River estuary (river miles 1 to 145), and at Willamette Falls. Recommendations: (1) expand pinniped monitoring efforts to assess predator-prey interactions between pinnipeds and listed species, (2) maintain predatory pinniped management actions at Bonneville Dam to reduce the loss of up- river listed salmon and steelhead stocks, (3) complete life-cycle/extinction risk modeling to quantify predation rates by predatory pinnipeds on listed salmon and steelhead stocks in the Columbia River and Willamette River, and (4) expand research efforts in the Columbia River estuary on survival and run timing for adult salmonids migrating through the lower Columbia River to Bonneville Dam."

3.4.2. ISAB's Past Findings, Conclusions, and Recommendations

Most of the theoretical and existing evidence available to the ISAB to answer pinniped predation questions in the current review has been thoroughly reviewed in past ISAB reports on food webs (ISAB 2011-1), density dependence (ISAB 2015-1), critical uncertainties (ISAB/ISRP 2016-1), predation metrics (ISAB 2016-1), and Interior Columbia Basin life-cycle models (ISAB 2013-5, ISAB 2017-1). Therefore, we briefly summarize ISAB's relevant past findings, conclusions, and recommendations.

The ISAB's Food Web Report (ISAB 2011-1) included a review of information through 2010 on the increasing abundance of pinniped predators and predation by sea lions at Bonneville Dam. The ISAB concluded that an important knowledge gap is the lack of data on rates of predation by apex predators (birds, fish, marine mammals) on the region's fish (salmon, steelhead, lamprey, sturgeon, eulachon, etc.) resources. These data are needed to better understand food web dynamics and total ecosystem productivity in the Columbia River Basin. The ISAB recommended: (1) using models to estimate system-scale consequences of increases or decreases in predation rates, (2) quantifying predator abundance in space and time and using bioenergetic and community process models to estimate system-scale consequences, and (3) determining the effects of predation on salmonids by piscivorous fish, seabirds, and mammals and whether these effects are additive or compensatory; doing this by comparing predation rates as a percent of the juveniles passing Bonneville Dam or PIT tag detectors in the lower estuary with appropriate smolt-to-adult survival estimates (SARs) for different ESUs and run timings.

The ISAB's Density Dependence Report (ISAB 2015-1) reviewed evidence of predation effects on density dependence of salmon populations in the Upper Columbia and Snake river basins, and identified the lack of information about predation on adult salmon by pinnipeds (seals and sea lions) as a primary data gap. The ISAB considered predation on adult salmon during upstream migration to be of particular concern because it may reduce the potential spawning population more than an equivalent rate of predation at earlier life stages. By the time adult salmon enter the Columbia River estuary, they have already survived numerous threats in both freshwater and marine environments, and all are potentially valuable for harvest or spawning. Evidence reviewed by the ISAB showed that the escapement goal for spring Chinook counted at Bonneville Dam (115,000 fish) had been met or exceeded since 2008 despite evidence that predation of salmon by pinnipeds is increasing. Moreover, there is evidence for density dependence over the entire life cycle for most interior populations of spring and summer Chinook salmon (24 out of 26 populations remained strongly compensatory; Zabel and Cooney 2013), even though depensatory mortality[6] likely occurs at some life stages. However, among Upper Columbia spring Chinook populations evaluated (Entiat, Methow, and Wenatchee), only the Wenatchee population did not exhibit a significant density-dependent relationship (Zabel and Cooney 2013).

Surprising new evidence indicated a steady decline in estuarine survival of the combined runs of adult middle and Upper Columbia River spring Chinook and Snake River spring/summer Chinook (from 90% in 2010 to 69% in 2013) (Wargo Rub et al. 2014). Survival was consistently higher for Chinook arriving late in the run compared to those returning early or at the peak, when predation by pinnipeds would have been more intense (Wargo Rub et al. 2014). The declining survival rates also coincided with the growing presence of sea lions and seals in the estuary. The number of sea lions identified at haul out sites near Astoria in 2013 was five times that observed during each of the previous three years, and a still larger number was observed in 2014 (Wargo Rub et al. 2014). The ISAB concluded that there were no reliable estimates of total pinniped abundances in the Columbia River estuary, integrated over all seasons, and the impacts of pinniped predation on salmonids in the Columbia River were still unknown or largely speculative. For example, in theory, even if pinniped predation on interior populations of

[6] Mortality caused by individual predators is typically depensatory, which means that the impact on a salmon prey population from individual pinniped predators is highest when fewer prey are present and decreases when more salmon prey are available because predators become satiated and reduce their feeding rate. This typical functional response can be offset by an increase in the number of predators due to aggregation in the short term or increased predator reproduction and abundance in the long term.

spring/summer Chinook was reduced, salmon productivity (recruits per spawner) would not increase because escapement goals (at Bonneville) were met and most populations exhibited density dependence. Because depensatory mortality may pose a threat to ESA-listed populations, the ISAB recommended further quantification of mortality and evaluation of life-cycle recruitment in salmon populations targeted by pinnipeds. Further studies were needed to track pinniped abundance in the estuary, and to confirm that salmon mortality attributed to pinnipeds is depensatory, as expected, and as suggested by studies to date.

The ISAB's Critical Uncertainties Report (ISAB/ISRP 2016-1) identified predation as a new critical uncertainty for the Council's Fish and Wildlife Program and two critical questions for this uncertainty: (1) To what extent is the viability or abundance of native fish and wildlife populations in the Columbia River Basin jeopardized by predation? and (2) How effectively can undesirable impacts of predation be ameliorated by management actions including hydrosystem operations, habitat modifications, and predator population control? These critical questions were included in the Council's 2017 Research Plan.

The ISAB's Predation Metrics Report (ISAB 2016-1) reviewed and recommended potential alternative metrics for evaluating and comparing the effects of predation at different stages in the life cycle of anadromous salmon and steelhead in the Columbia River Basin. The ISAB considered compensatory mortality to be the most important uncertainty to address when developing a predation metric. The ISAB reviewed evidence for mechanisms of compensation, including (1) density dependent survival due to factors other than predation, (2) selective predation based on fish size and condition, and (3) switching behavior of predators, which may be caused by a change in abundances of alternative prey species or when secondary predators increase predation on salmon following control of the primary predator. Considerable compensation in predation-related mortality may occur between juvenile and adult life stages, but additional compensation may also occur during the subsequent spawner-to-smolt stage, indicating the need to consider predation within the context of the entire life cycle.

A review and comparison of three alternative metrics using a standard set of evaluation criteria revealed that a single metric would not be adequate for evaluating all goals. The ISAB recommended: (1) using and further refining two types of metrics currently in use in the Basin, that is, *Equivalence-factor metrics* (for example, adult equivalents), which can be used to compare the effects of predation on salmon and steelhead at different points in their life cycle, and a *population growth rate metric* (also called *delta-lambda,* $\Delta\lambda$), which can be used to compare how different predation scenarios affect rates of population recovery or decline; (2) adjusting the *equivalence-factor metrics* and the *population growth rate metric* ($\Delta\lambda$) to account for assumed or estimated compensation in mortality, and (3) placing predation mortality in the context of a life-cycle model that can be used to evaluate multiple factors affecting salmon survival and interactions among those factors in modeled scenarios and verified with data. This approach could help guide research, monitoring, and evaluation of predation throughout the salmonid life cycle, both to provide the data necessary to parameterize and verify models, and to refine metrics.

The ISAB's Life Cycle Modeling Report (ISAB 2017-1) reviewed NOAA's 2017 report on Interior Columbia Basin Life Cycle Modeling, and builds on a previous ISAB review in 2013 (ISAB 2013-5). In Chapter 6.a of the NOAA report (Population-specific pinniped predation; Sorel et al. 2017), modelers used tag-recapture data to estimate population-specific survivals of adult spring-summer Chinook salmon migrating from the mouth of the Columbia River to Bonneville Dam. The modeling results indicated a decline in Chinook salmon survival as pinniped density increased. Early-migrating spring Chinook salmon populations that entered the lower river in late winter or early spring had a higher risk of mortality from pinniped predation than later migrating populations. Compared to a baseline period (1998-2012), apparent survival of adult Upper Columbia spring Chinook in 2013-2015 decreased substantially, i.e., average survival was 22% lower than survival during the baseline period for the earliest-arriving population (Methow) and 11% lower for intermediate-arriving populations (Entiat and Wenatchee). Investigators speculated that a 22% decrease in survival would significantly affect population viability if sustained.

The ISAB's review concluded that treatment and communication of assumptions and uncertainties in the analysis were not fully addressed by the authors. For example, salmon mortality due to handling and tagging is presumed to be equal in all years, without discussion of how variable mortality might affect estimates. Small sample sizes necessitated simplifying assumptions like constant year effects across populations and constant variances within populations across years, but discussion was limited on robustness to violation of these assumptions. Uncertainty remains as to whether other causal factors or estimation error—for example, disease, permanent straying below Bonneville of upriver fish, underestimation of harvest, artifact of learned behavior by predators (Wargo-Rub et al. 2014)— may be involved in the apparent decline in survival of adult Upper Columbia spring Chinook from the river mouth to Bonneville Dam. The ISAB's recommendations for some specific areas of future research and implementation of results in life-cycle models included: (1) apply the results of Sorel et al. (2017) to existing life-cycle models, as recommended by Sorel et al. (2017) and ISAB 2016-1 (Predation Metrics Report) and evaluate the risk of extinction caused by pinniped predation and tradeoffs between fishing mortality and this predation mortality and the risk of extinction; (2) extend the model to include parameters for marine mammal predation on adult salmon in the Columbia River Plume. Explore spatial variation in predation pressure in the plume and lower river. (3) investigate relationships of fishing and marine mammal predation on adult salmon; (4) investigate relationships of forage fish density and marine mammal predation on adult salmon survival; and (5) develop methods to investigate marine mammal predation on the juvenile freshwater/estuary/early ocean life stages.

A preliminary life-cycle model for Upper Columbia spring Chinook in the Wenatchee subbasin, reviewed by ISAB in 2013 (ISAB 2013-5), was updated (Jorgensen et al. 2017). Two pinniped predation scenarios of adult salmon survival were modeled, and results indicated the biggest increase in spawner abundance occurred when estuarine adult survival increased to historical levels, as might result from reduced pinniped predation, and when the hatchery programs resumed operations to current levels. The ISAB's review (ISAB 2017-1) concluded:

> "Given the many assumptions and fixed parameter values, the [model] outputs should be used in a **qualitative** fashion only, e.g., to rank actions in terms of effectiveness. It would be difficult to assign a **quantitative gain** to any of the scenarios that were performed."
>
> "Similarly, it will be difficult to use the model in its current stage of development for adaptive management, because it will be unknown whether a failure to see a predicted response to a management action in the real world is due to stochastic variability or to inadequacy of the life-cycle model."

3.4.3. Conclusions about Pinniped Predation

Are pinnipeds potentially a significant source of mortality for Upper Columbia spring Chinook?

The best available scientific evidence, as reviewed by ISAB in past reports (see above, ISAB's Past Findings, Conclusions, and Recommendations) and in new (2017) reports and publications, indicates that pinnipeds are *potentially* a significant source of mortality for Upper Columbia spring Chinook adults. However, population-specific estimates of predation impacts on Upper Columbia spring Chinook were not available to the ISAB at the time of this review. The estimated consumption of combined populations of Chinook salmon by pinnipeds in the Columbia River increased sharply over the past decade, likely exceeding removals by fisheries (Chasco et al. 2017) (Figure 3.28). Potential impacts varied by pinniped species and salmon life stage. For example, in 2015 estimated biomass (metric tonnes, t) and numbers of Chinook salmon consumed in the Columbia River estuary varied among harbor seals (14 t, including 1000 adults and 312,000 smolts), California sea lions (219 t, including 46,000 adults), and Steller sea lions (227 t, including 47,000 adults) (Chasco et al. 2017). In 2017, the reduced salmonid runs and persistently high numbers of pinnipeds in the latter part of the season suggests that the total impact by pinnipeds on the year's salmonid run may be large (Tidwell et al. 2017) (Figure 3.29).

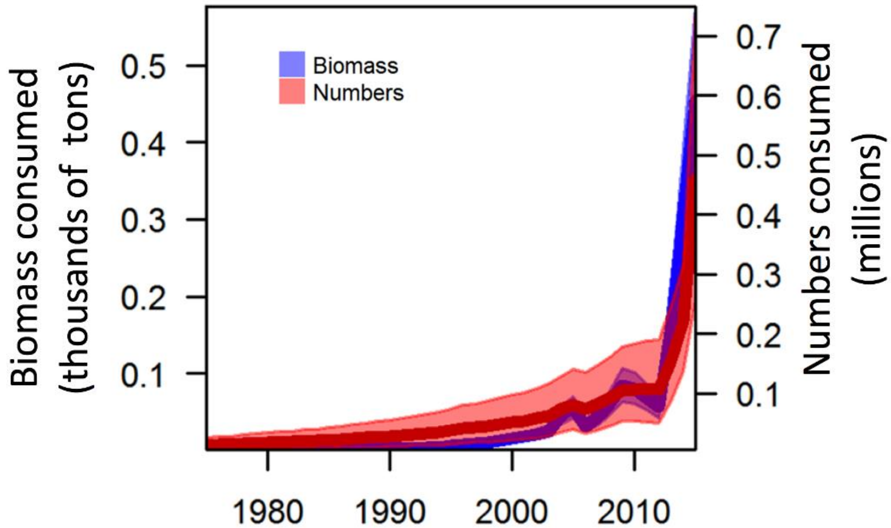

Figure 3.28. Estimates of consumption of Chinook salmon by pinniped predators in the Columbia River, with uncertainty, in terms of the biomass (primary axis) and number (secondary axis) of Chinook salmon consumed, 1975-2015. Source: Figure 7, Chasco et al. (2017).

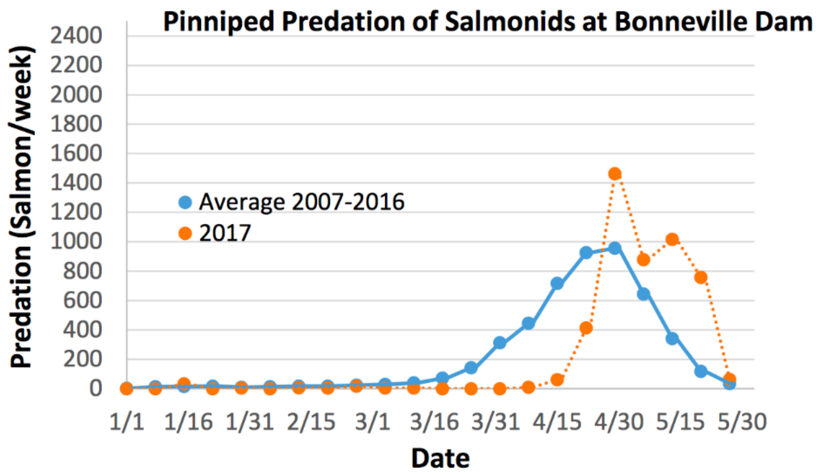

Figure 3.29. Preliminary estimated pinniped predation at Bonneville Dam in 2017, compared to the 10-yr average (Tidwell et al. 2017). Between 13 May and 2 June 2017 an estimated 1837 ± 130 adult salmonids were consumed (10-yr average estimate of 494 adult salmonids for the same time period). From 1 January to 2 June 2017, an estimated, but not adjusted, 4993 ± 234 adult salmonids (82% spring Chinook) were consumed by pinnipeds. The combined runs were delayed relative to previous years and smaller than the long-term average. As such, the duration of pinniped presence and levels of fish predation were protracted relative to the 10-year average. The reduced salmonid runs and persistently high numbers of pinnipeds in the latter part of the season suggests that the total impact by pinnipeds on 2017's salmonid run may be large.

The Chasco et al. (2017) estimates of in-river consumption of adult (ocean age two and greater) Chinook salmon by sea lions in the Columbia River from January to August in 2015 was 65,000 (49,000–81,000), were lower than the most recent direct, tagging-based estimate of 95,000 (61,000–127,000) spring/summer Chinook (Michelle Rub, pers. comm., August 7, 2017, as cited by Chasco et al. 2017). However, the Chasco et al. model estimated that additional sea lion predation on Columbia River Chinook salmon in the ocean may be larger than previously documented (e.g., approximately 70,000 ocean age-1 salmon) during the same period in 2015.

Updated tagging-based estimates of in-river (estuary and lower river below Bonneville) mortality of upriver spring/summer Chinook salmon adults peaked in 2014 and 2015 (~100,000 fish) and decreased in 2016 and 2017 to relatively low levels (~22,000 fish) similar to those in the base period, 2010-2013 (~31,000 fish) (Figure 3.30) (Wargo Rub et al. presentation to ISAB, December 8, 2017, Northwest Power and Conservation Council, Portland, OR). Radio telemetry tracking results indicate the highest mortality occurs in and near the estuary and in the Bonneville tailrace. The mortalities are thought to be due to pinniped predation, but additional covariates potentially influencing salmon survival include spill, abundance of smelt, and fin clip status.

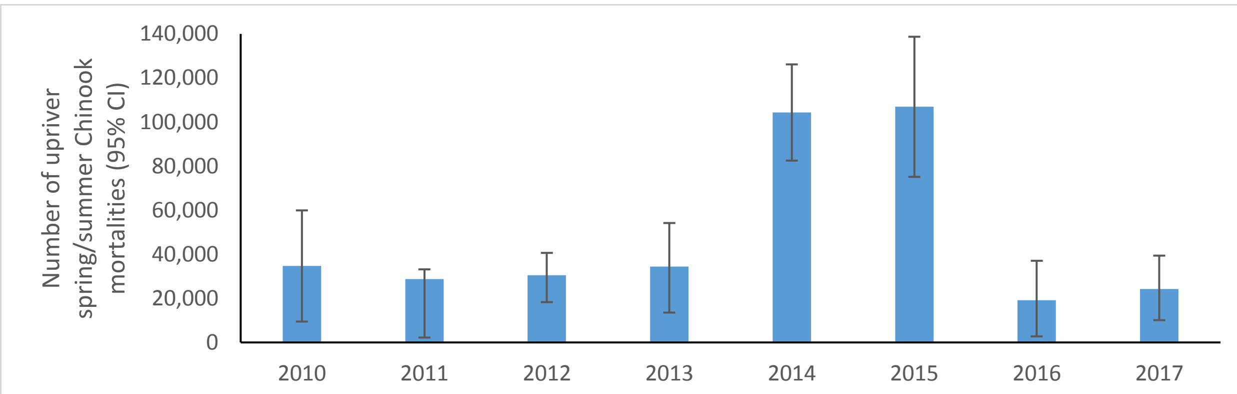

Figure 3.30. The estimated number of upriver spring/summer Chinook mortalities (95% confidence interval) in the Columbia River below Bonneville Dam

Can the effect of pinniped predation of Upper Columbia spring Chinook be quantified?

Efforts to quantify the effects of pinniped predation on Upper Columbia spring Chinook are ongoing and continue to make progress (e.g., Sorel et al. 2017). However, additional data and evaluation of uncertainties in the estimates and model structure are needed to further improve estimates of pinniped predation on Upper Columbia River spring Chinook salmon. Chasco et al. (2017) developed a bioenergetics/life-cycle modeling approach to quantify trends (1975-2015) in predation of Chinook salmon by three species of pinnipeds (harbor seals, California sea lions, and Steller sea lions) and killer whales (*Orcinus orca*) in eight regions of the Northeastern Pacific, including the Columbia River. Chasco et al. (2017) addressed uncertainty in key parameters of the model related to predator abundance, diets, and bioenergetics. For example, the many assumptions about the fraction of energy in pinniped diets derived from Chinook salmon resulted in large coefficients of variation in sensitivity analyses (Chasco et al. 2017, see Supplementary Information). The structural uncertainty in the model formulation was not addressed. Chasco et al. (2017) suggested using a multi-model approach to address structural uncertainty, e.g., by comparing their bioenergetics approach to other methods such as individual-based models or time series modeling.

3.5. Recommendations

Identifying limiting factors

The ISAB recommends integrating the results of the different approaches—limiting factors analysis, density dependence analysis, and life-cycle modeling—for identifying limiting factors to guide future revision of the Biological Strategy of the RTT and the Recovery Plan for spring Chinook salmon. This integration will require a collaborative process that includes significant participation by experts, practitioners, and management teams from all Hs. To date, analysis has focused on limiting factors related to habitat conditions. The limitations of four Hs on spring Chinook populations are considered in isolation.

We recommend the UCSRB, RTT, and co-managers to consider an effort to synthesize the results of the different approaches for identifying limiting factors to guide future revision of the Biological Strategy of the RTT and the coordination of actions for spring Chinook salmon by the UCSRB.

If return of adult spawners or recruitment substantially limit recovery in the Upper Columbia, then discussions of the effects of harvest on escapement between co-managers and participants in the UCSRB could strengthen future approaches to improve recovery efforts.

Integration of Biophysical and Economic Evaluation of Recovery Actions

Biophysical limitations on spring Chinook are considered in limiting factors analysis, density dependence analysis, and life-cycle modeling. Economic analysis—relaxing which constraint can most cost-effectively improve abundance in the shortest period of time—has rarely has been used to measure the operational importance of different limiting factors. We recommend the development of integrated analyses in which biophysical analyses of limiting factors and economic analysis of the costs associated with different management actions to better inform recovery actions for UCR spring Chinook salmon. Such analyses would provide quantitative estimates of the relative cost-effectiveness of alternative recovery actions

Snake River and Upper Columbia River spring Chinook salmon

The ISAB recommends continued comparison of Chinook recovery in both ESUs to determine which restoration actions are most effective. Rigorous RME programs are essential to understand the trends and factors that influence them in these two basins. The lower number of populations and total abundances of the Upper Columbia River spring Chinook ESU potentially expose them to greater threats than the Snake River spring/summer Chinook ESU.

Spring and summer Chinook in the Upper Columbia River

The ISAB recommends continued investigations of the effects of summer Chinook on spring Chinook in Upper Columbia River subbasins, including effects of hatchery practices, redd superimposition, competition, outmigration behavior, relative rates of survival and behavior in the mainstem Columbia, and the relative effects of pinnipeds and harvest in the lower river and estuary. Summer Chinook may affect the recovery of spring Chinook through competition, redd superimposition, and introgression. In addition, differences in life histories, habitat relationships, survival, population demographics, and responses to potential limiting factors by summer Chinook may reveal critical factors for the recovery of spring Chinook salmon in the Upper Columbia River.

Analysis of natural origin return numbers for spring and summer Chinook provide a general demographic overview of how population abundance varies over time. However, grouping returns by brood year provides a number of important advantages. Juveniles are counted by cohort or year class, and adults should be counted on a similar basis in demographic analyses. Foremost among those is the development of recruitment curves that can determine intrinsic productivity and the population carrying capacity in existing habitat (ISAB 2015-1). Additionally, smolt-to-recruit (SARs) estimates for each brood year can be made. SAR values could be used to directly test the effects of the summer spill program on survival and also allow a more refined assessment of how summer and spring Chinook abundance may be related. Advantages of using brood year returns as opposed to annual returns have led us to make several suggestions the researchers may wish to undertake if they are not already underway:[7]

- Add harvest values (by age) to the return numbers for years 1989 – 2016
- Evaluate trends of abundance by brood year rather than by return year.
    - Anadromous spring Chinook reach maturation at ages 3, 4, and 5. Summer Chinook also mature at these ages and a few may return at age 6. If age data are available, it will be possible to examine the abundance of fish returning from the 1986 brood year up to the 2010 brood year. This would create a dataset with 25 brood years as opposed to the existing one with 28 return years.

[7] Our request to compare the survival and life history strategies of spring and summer Chinook returning to upper Columbia subbasins occurred after our site visit to the upper Columbia. We appreciate the effort undertaken to carry out this unexpected and new assignment. The scientists performing this work indicated that additional comparisons will be made in the future. It is likely that yearly return data will be deconstructed and assembled into brood year survival and abundance data. The suggestions for future analyses were made to affirm the importance of producing and analyzing brood year information.

- Use return data to populate brood year returns by subbasin rather than aggregating them into a single number for each race of Chinook salmon. Summer and spring Chinook returning to the Wenatchee River pass over two fewer dams than those returning to the Methow and Okanogan River and therefore may be affected by different stressors.

- Temporal and spatial overlap information on adult spring and summer Chinook is likely available for the Wenatchee and Methow subbasins. This information should be used to determine the degree of overlap that exists between these two races in the above subbasins. Since spawner density is known to affect distribution patterns (Murdoch 2017) multiple year assessments should take place. Additionally, analyses on hybridization rates and the occurrence of redd superimposition similar to those performed in the Entiat are encouraged. They would help determine the relative importance of these factors on spring Chinook productivity. Are they major factors constraining spring Chinook abundance or occur rarely and are of little demographic importance?

Pinniped predation

The ISAB reiterates its recommendations from past reviews (see 3.4.2) and recommends proceeding with pinniped recommendations in the 2016 five-year Upper Columbia status report:

> "(1) expand pinniped monitoring efforts to assess predator-prey interactions between pinnipeds and listed species, (2) maintain predatory pinniped management actions at Bonneville Dam to reduce the loss of up- river listed salmon and steelhead stocks, (3) complete life-cycle/extinction risk modeling to quantify predation rates by predatory pinnipeds on listed salmon and steelhead stocks in the Columbia River and Willamette River, and (4) expand research efforts in the Columbia River estuary on survival and run timing for adult salmonids migrating through the lower Columbia River to Bonneville Dam."

The ISAB considers the second recommendation a necessary precautionary measure while better data are collected.

The ISAB recommends identifying and investigating other potentially significant sources of mortality of Upper Columbia spring Chinook smolts and adults in the Columbia River plume/ocean shelf habitats, estuary, and lower mainstem and tributaries. New information from NOAA's tagging and modeling efforts revealed important data gaps, including lack of population-specific survival estimates for Upper Columbia spring Chinook.

The ISAB recommends use of a variety of approaches to quantify pinniped predation impacts, such as the ongoing tagging studies and coast-wide bioenergetics/life-cycle modeling. New information from NOAA's tagging and modeling efforts revealed some important data gaps, including the lack of population-specific estimates for UCR spring Chinook salmon. To evaluate assumptions and reduce uncertainties in estimates, modeling efforts would benefit from better information on the abundance and spatial-temporal distribution of pinnipeds, particularly in the lower river and estuary, pinniped body mass and digestion efficiencies, and the fraction of Chinook salmon prey in pinniped diets, as well as better estimates of natural-origin smolt production, smolt and adult spatial distribution and residence time, size-at age and growth, and energy density of UC spring Chinook. Comparison of multiple models could reduce structural uncertainty (e.g., comparing a bioenergetics approach to individual-based models or time series models).

4. Prioritization and Effectiveness of Habitat Restoration and Enhancement

Questions submitted to ISAB:

Are habitat recovery actions being prioritized and sequenced strategically, given existing knowledge and data gaps?

Is there evidence that past projects have improved habitat for this ESU?

How should habitat projects be prioritized and what types of habitat projects should be prioritized in the future? Why?

How well are actions in other management sectors (all H's, i.e., habitat and hydrosystem, hatcheries, and harvest) aligned with recovery efforts? Specific input to inform development and refinement of the Upper Columbia's proposed prioritization framework for projects would be much appreciated.

Many habitat protection, restoration, and enhancement projects have been planned and completed in the Upper Columbia River region since spring Chinook salmon were listed as an endangered species in 1999. In large part, this is because restoration of tributary habitat is considered a key requirement for recovery of this salmon stock and one of the most feasible actions possible to mitigate for degradation of the hydrosystem by dams, native and nonnative predators, and poor and changing ocean conditions, among other stressors. Toward that end, a total of $74 million dollars from all funding sources was been spent between 1996 to 2012 to plan and carry out these habitat projects (UCSRB 2014b).

The effects of habitat restoration interact with those of actions to address the other three Hs—Hatchery, Hydropower, and Harvest—making it difficult to separate their effects and determine which actions provide optimum benefits. As a result, biologists and managers have focused on smolts produced from UCR subbasins as one measure of the effectiveness of habitat projects, because it is relatively free from influence by other mitigation actions, except for introduction of hatchery fish (UCSRB 2007). A key measure of smolt productivity is smolts per redd, determined by dividing smolt output measured using screw traps by redd abundance measured using field surveys or estimated from counts of adults. Another measure is parr-to-smolt survival, measured by detections of PIT-tagged fish at PIT-tag antennas in lower river locations or at the first downstream dam, although the accuracy of this measure can be affected by conditions in the mainstem Columbia River for outmigrants.

Habitat restoration consists of a wide range of actions, many of which require decades or longer to see their full effects. Therefore, detecting and summarizing the effects of these efforts is expected to be difficult. In this section, we discuss how habitat restoration projects can be prioritized, the evidence that the most commonly-used types produce better physical habitat and more fish, three primary methods used to set priorities for habitat projects, and how habitat restoration is aligned with other Hs to achieve recovery.

4.1. Are Recovery Actions Being Prioritized and Sequenced Strategically?

The Upper Columbia Salmon Recovery Board (UCSRB) is the primary entity that prioritizes habitat restoration actions for spring Chinook in the Upper Columbia River. Formed in 1999, the UCSRB is a partnership of counties, tribes, state and federal agencies, and local conservation groups. They have established several key organizational components and processes:

- rigorous structure for governance
- regional technical team
- explicit recovery plan and biological strategy
- economic analysis of regional cost/benefit considerations
- staff for implementation
- staff for fiduciary management
- a framework for adaptive management
- regular schedule of meetings
- outreach and education

The UCSRB is one of eight regional salmon recovery organizations as part of the State of Washington's Governor's Salmon Recovery Office. The UCSRB consists of five members: one County Commissioner from each of Chelan, Douglas, and Okanogan counties and one representative from each of the Colville Confederated Tribes and the Yakama Nation. The UCSRB has a staff that includes an executive director, science program manager, natural resource program manager, fiscal manager, and natural resources program coordinator.

The UCSRB established the Regional Technical Team (RTT) to recommend regional strategies, evaluate monitoring data, review salmon recovery projects, and guide monitoring plans. The RTT developed the Upper Columbia Biological Strategy as a framework for identifying, designing, and implementing habitat projects to recover native salmonid stocks (RTT 2014). The UCSRB also works with local partners to share new syntheses of Upper Columbia science and assist the community in linking healthy communities with restoration of the aesthetic, economic and cultural values in the Upper Columbia. Each county either individually or collectively holds public meetings, open houses, workshops, and informational sessions. Both general and technical information are provided in brochures, websites, and published documents.

Since 2010, the ISRP has participated in six iterative reviews of the UCSRB's Upper Columbia Programmatic Habitat Project (BPA project #2010-001-00). Most recently, in 2017, the ISRP reviewed the project as part of the Northwest Power and Conservation Council's Review of Umbrella Habitat Restoration Projects (ISRP 2017-2; also see ISRP 2014-10, ISRP 2014-5 [covers review history], ISRP 2013-11, ISRP 2010-28, and ISRP 2010-12). The ISRP's reviews have focused on the UCSRB's project selection process, Biological Strategy, monitoring and evaluation approach, and results. In the course of those reviews, the ISRP and UCSRB have discussed and addressed project prioritization issues such as conflicts of interest concerns with RTT members; multi-year funding and the ability to pursue large, complex projects; and other steps to improve administrative efficiency.

In the 2017 Umbrella Project Review, the ISRP stated "the UCSRB has developed an open and transparent system for solicitation, review, and project design that serves as a useful model for other Umbrella Projects. The formal coordination among the Executive Teams, Project Teams, Design Teams and Regional Technical Team (UCRTT) is commendable." However, the ISRP noted some opportunities to improve the UCSRB project selection process including the need for 1) a description of how projects will be prioritized and sequenced, given limited funding, to maximize the effectiveness of the program and 2) a "plan for the systematic collection of data relevant to limiting factors and project design before projects are selected." This ISAB review is not intended to duplicate or rehash those ISRP reviews, but the ISAB offers some additional recommendations for improving the project prioritization process, particularly to maximize the impact of available funds over short and long time frames.

4.1.1. Components for Effective Prioritization of Recovery Actions

If financial resources and time are unlimited, prioritization is unnecessary. Prioritization is only necessary when there are constraints. Since financial resources and time are those constraints, the focus of prioritization should be to achieve the greatest restoration benefit per dollar and per year. Prioritization and sequencing of recovery and research efforts should reflect both the intended objectives (the recovery goals) and the realities of existing resource constraints. The UCSRB Recovery Plan must assess how well these projects and other actions are achieving the desired changes in fish abundance, productivity, and risk of extinction. Project success can be framed "in terms of time, cost, and quality/performance (scope)" (de Wit 1988), three key criteria that have been used effectively to measure success of project management for more than 60 years (Figure 4.1).

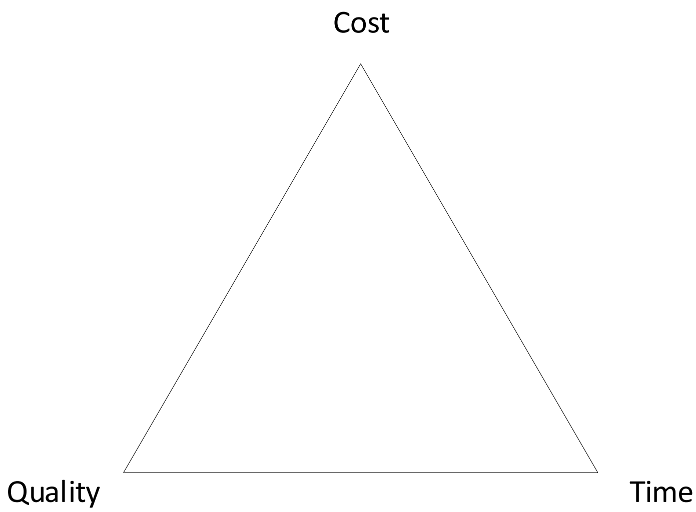

Figure 4.1. Framework for assessing project management success (from Atkinson 1999).

One succinct version of this widely-used set of criteria seeks to "achieve the project objectives on time and to the specified cost, quality and performance" (Atkinson 1999). This framework recognizes that additional criteria will be important at the local level, such as benefits or costs for specific stakeholder groups, sources of funding restricted to particular uses, and individual landowner considerations.

(1) The cost constraint recognizes that resources are limited so choosing actions that contribute most per unit of available resources will go farthest toward achieving overall success. Tradeoffs are inherent in project selection. Choosing to start or expand one project generally will result in fewer resources for other projects. Resources for habitat restoration are limited as are resources for research, monitoring, and evaluation. One critical decision involves the allocation of scarce resources directly for restoration versus data collection and research that may improve later decisions. Does one decide to act, or does one decide to gather more information/data so that a better-informed decision can be made in the future (see ISAB/ISRP 2016-1, Appendix A)? In some cases, the contribution of a given action toward ultimate success will depend on complementary actions or on actions taken elsewhere that are outside the control of the project managers (e.g., harvest, hydrosystem, hatcheries).

(2) The time constraint recognizes that the timeliness of a given action or its contribution to recovery goals can be as important as cost. To paraphrase a legal maxim, "recovery delayed is recovery denied." The value of the project goal may be highly time sensitive. The UCSRB

Recovery Plan (2007) suggests that reclassification of (improved) abundance “could occur” within 5-15 years (as of 2007) and that recovery of Upper Columbia spring Chinook could occur within 10-30 years. These timelines come with major caveats regarding out-of-ESU conditions, but nevertheless convey the sense of urgency, similar to the NPCC Fish and Wildlife Program goal of 5 million salmon and steelhead per year by 2025.

The time dimension also recognizes that sustained increases in abundance contribute more toward goals than temporary increases. More immediate improvements in abundance also should carry more weight in decision making than improvements that are delayed or have slow responses to project actions (Arrow 1965; Farrow and Zerbe 2013; Arrow et al. 2013).

(3) The most uncertain and complex criterion for project success is the quality/performance dimension. Given the state of empirical research in the region, there is great uncertainty about likely success in terms of increased abundance for a wide range of restoration actions due to other limiting factors, density dependence, out-of-ESU conditions, and other factors. In addition, the quality, scale, and siting of actions can have a major effect on success. Life-cycle models have the potential to incorporate information at all life stages and estimate the impact of actions that affect specific life stages on aggregate abundance through time, while accounting for limiting factors at other life stages. Though currently in their early developmental stages, life-cycle models eventually may be one of the most comprehensive ways to evaluate the benefits from specific restoration actions.

While most attention is focused on determining which kinds of actions will be biologically beneficial toward recovery of Upper Columbia spring Chinook, it would be inaccurate to conclude that this quality/performance element outweighs the elements of cost and time. A project of type A that is estimated to improve abundance by 10% will be twice as valuable as a project of type B that improves abundance by 5%. But if project B costs one-third as much as project A, greater progress toward increased fish abundance will be achieved by doing three projects of type B rather than one project of type A, with the same total resources. Similarly, a project type Z that improves abundance by 10% but only after a 20-year lag should have a lower priority than a similar project Q that for the same cost is expected to achieve the same outcome in 10 years. The real world contains more complicated alternatives where the impact, timing, and costs do not lend themselves to simple ranking. Instead, they require quantitative comparisons in terms of their estimated benefits, costs, and timeliness, especially when uncertainty is high.

There are two general points here. First, prioritizing actions require judgments about their likely biological performance, but equal or greater overall success may be achieved if the other two elements (i.e., cost and time) are given appropriate weight. Second, the tradeoffs between quality/performance versus cost versus timeliness should be explicitly evaluated at all stages of planning and decision making. Evaluating these tradeoffs correctly involves weighting all three factors in a manner that considers multiplicative interactions and time discounting. Indeed, more detailed optimization models such as goal programming can be applied (Tamiz et al. 1998). Frameworks for dealing with projects involving uncertainty about methods and goals have also been developed (Turner and Cochrane 1993).

The focus of our recommendations here is cost-effectiveness analysis rather than "benefit-cost analysis," which is not being proposed. Benefit-cost analysis is a method in which both benefits and costs are estimated in comparable monetary units, which reflect society's valuation or "willingness to pay" for a given action as well as for the costs of that action. This would allow a project to be evaluated in terms of its present value of net benefits in dollars. However, estimating the monetary value of many environmental resources is difficult and likely to be controversial. The kinds of values at issue would include cultural and aesthetic values, indirect use value, and non-use values such as existence value (see NRC 2005).

To the extent that the decision to undertake restoration activities has been made (e.g., ESA listings and recovery plans), we can avoid the step of monetizing the benefits of alternative actions and instead ask: which restoration activities would be most effective per dollar of funding? This prioritization is the focus of cost-effectiveness analysis, where CE, the cost-effectiveness ratio, is the ratio of estimated benefits (in units of adult fish, smolts, redds, or other appropriate biological changes) to estimated project costs. In this report, we focus on how to implement cost-effectiveness analysis to prioritize alternative actions and maximize the effectiveness of program resources.

4.1.2. Cost-effectiveness Estimates that Account for Time

Two challenges arise when attempting to prioritize projects based on costs and benefits that change through time. First, benefits of different actions are difficult to estimate, such as the expected increased numbers of smolts across different time periods. Second, many individuals may not be familiar with time discounting used in both benefit-cost analysis and cost-effectiveness analysis. However, there are simple methods to address these challenges that are transparent, consistent with standard practices, and can be uniformly applied to all projects.

The first challenge, the difficulty in quantifying the expected improvements in smolt numbers or other VSP parameters, is a genuine problem for judging the likely merits of different projects. Rather than using a somewhat arbitrary cost index (see section 4.3.3 on RTT Strategy below), there are advantages to simply estimating the expected benefits, even if this is simply asking experts each to independently identify the highest and lowest realistic estimate, and then calculating the midpoint (Conroy and Peterson 2013). Advantages of this approach are that: 1) the estimated benefits are all in the same units (e.g., expected increase in smolts per year) and 2) the estimates produced form a basis for discussion about whether they are too high or too low.

The second challenge, prioritizing projects because time and resources are scarce, involves application of discounting future costs or benefit in some form. Given the high degree of uncertainty in benefits and their timing, a comprehensive project cost analysis model does not seem appropriate. Nevertheless, a simplified framework that upholds standard practices for project evaluation is possible, as illustrated in Box 4.1:

Box 4.1
Example of a Simplified Framework
For Evaluating Cost Effectiveness of Projects

Step 1: Estimate the effect that the project is expected to produce during each of four different time periods. For example, estimate the average annual increase in number of smolts during years 1-10, 11-25, 26-50, and 51-100.

Step 2: Estimate the initial cost of the project in dollars. For cases where there will be ongoing costs such as maintenance costs in future periods, convert these future dollars into their "present value" by discounting them using an appropriate discount rate such as 3% (Moore et al. 2004; EPA 2010).

Step 3: Sum the average values in Step 1 across all time periods, and divide this total by the cost estimate from Step 2. The result is a measure of cost-effectiveness, where near-term abundance benefits have been given somewhat more weight than mid-term or long-term abundance benefits. This differentiation in weighting is a result of the different lengths of the four time periods, which effectively discounts future benefits at approximately a 2.5% social discount rate. Other groupings of time periods are possible, but the number of years in each grouping should increase in the future. The resulting ratio can be interpreted as a measure of the biological benefits per dollar of cost, and these can be ranked from highest to lowest across projects.

Step 4: If the above analysis was done for, say, mid-range estimates of the abundance benefits in Step 1, one could repeat the analysis for low and high estimates, and perhaps for an alternative discount rate. Compare the ranking of projects with these different assumptions to assess the sensitivity of the ratios to the assumptions. More effort could then be placed on gaining information about assumptions that affect the outcome more. If these cost-effectiveness ratios are very similar, then other criteria for ranking projects may be judged to be more important. But if there are large differences between the highest and lowest cost-effectiveness ratios, choosing the most cost-effective options will be important.

By quantifying benefits and comparing each project's cost-effectiveness in this way, project selection can be done to achieve the greatest benefit for limited resources and time. Indeed, having these explicit metrics to guide decisions can also help guard against problems with incompatible incentives that lead to wasteful uses of public resources. Strategies of "study, tinker, and hope," or giving in to pressure to spread funds evenly over interest groups can frequently lead to inefficient use of public resources. For example, decisions based on political acceptance or jurisdictional equity can lead to the lowest possible benefits for society (Wu et al. 2003). Habitat investments with cumulative effects need to be carefully targeted to achieve the maximum benefits (Wu et al. 2000). These issues have been important for a range of conservation programs including riparian restoration (Watanabe et al. 2005).

Choosing projects based only on their relative biological benefits, while ignoring the differences in cost, can lead to large losses in effectiveness. For example, a retrospective analysis of conservation in California found that accounting for land costs in prioritizing conservation actions would have quadrupled the number of distinct species and tripled the number of threatened and endangered species protected under the plan (Underwood et al, 2009). Similar results have been found for vertebrate conservation, marine reserves, and salmon conservation (Boyd et al. 2015).

4.2. Evidence of Physical and Biological Responses to Past Habitat Restoration Projects

4.2.1. Approaches for Measuring Effects of Habitat Restoration

Measuring the effects of habitat restoration for fish and other aquatic and riparian biota is one of the great challenges of river and stream conservation. Large-scale experiments have inherent constraints that make the basic requirements of control, replication, and randomization at best difficult and at worst impossible to meet. Likewise, in the best case, researchers can measure true survival of salmonids (as a key measure of fitness) in response to habitat actions (see Bouwes et al. 2016 for an example) using a Before-After-Control-Impact (BACI) design (Manly 2001), but in many cases this is impossible. Therefore, researchers may instead consider measuring other correlates of fitness in the hierarchy of possible responses by fish. These range from short-term measures over relatively small spatial scales like habitat use or growth, to longer-term measures like smolt production for fish that traverse entire subbasins during their juvenile life history.

This hierarchy can be framed as a series of questions:

1. Do fish use the area treated or the habitat created?
2. Is fish abundance increased by the treatment, while not drawing fish away from adjacent untreated areas?
3. Has the treatment increased fish growth or physiological condition?
4. Has the treatment increased survival of fish through one or more life stages?
5. Has a set of treatments increased smolt production for an entire population?

Well-designed and executed studies of any of these responses are difficult and take years to complete, analyze, and publish, so researchers have been able to complete relatively few for spring Chinook in the Upper Columbia. Lacking this direct evidence, two other sources of information may be available. First, evidence may be available from research in other watersheds and in some cases for other salmonid races or species to assess which habitat restoration actions are likely to be most effective.

However, a difficult problem is that many habitat actions have complex effects on salmon life cycles that play out over large scales of space and time, and are confounded by many other effects within subbasins and across the entire range of habitats that salmon occupy. Therefore, when direct evidence is unavailable, a second approach is to assess whether improvements to habitat are likely to have positive effects on key life stages based on research at smaller spatial scales and shorter time scales, and to use inductive reasoning to argue for positive benefits that cannot be measured directly owing to other confounding factors. As an example, fine sediment is a known mortality factor for salmonid eggs and embryos (e.g., Reiser and White 1988). Therefore, habitat restoration that reduces fine sediment should increase survival of eggs and embryos, even if other mortality factors later in the life cycle mask the effects of the reduced sediment (Hillman et al. 2016).

4.2.1.1. A hierarchy of approaches for testing the effects of habitat restoration

The challenge of field experiments

A key problem for field experiments on habitat restoration is how to rigorously measure responses, especially those by fish, when replicated experiments are difficult or impossible. Experiments are the gold standard of the scientific method and require the hallmarks of control, replication, and randomization. However, true controls are often difficult or impossible because physical effects of treatments may flow into downstream reaches, so controls must often be upstream where physical conditions like temperature are different at the outset.

Moreover, fish may swim among study reaches, so that "treatment" fish may become "control" fish and violate the assumption that reaches are independent.

Likewise, replication is difficult or impossible, because reaches are often inherently different from the start (e.g., differences among side channels chosen as replicates). In addition, reaches of sufficient length to encompass all relevant fish population processes are usually long and very difficult to sample, so at best only a few replicates can be measured.

Finally, randomization is often overlooked and is also difficult to achieve. In theory, the entire population of sampling units (e.g., disconnected side channels in the basin) would be defined beforehand, and a random subset then chosen for treatment (e.g., reconnection) and compared to another random subset chosen as unmanipulated controls. It is rare that such randomization is done in large field experiments on habitat restoration. At best, replicate reaches are chosen haphazardly from among those available, with no intended bias, or replicates include all suitable study reaches or units in the subbasin.

What is possible?

Given these inherent challenges for measuring the effects of habitat restoration on fish and fish populations, what are the options? Fish respond to their environment through a hierarchy of responses, from behavior (including movement) to physiology, growth, abundance, and reproduction. Each is a measure of fitness, ranging from those operating at short time scales and small spatial scales to the entire lifetime fitness of fish carried out over the entire hydrosystem and ocean ecosystem. Thus, what is possible is to work from first principles and measure components of this hierarchy of responses, framed as a series of questions (see Table 4.1).

1. Do fish use the area treated or the habitat created? – If fish do not occupy the area treated by the habitat project or the habitat created, then they cannot reap any potential benefits to fitness, so this is a basic measure of project success. An ideal design is to estimate true abundance of fish in randomly chosen sites that are treated by habitat restoration projects versus another randomly chosen subset not treated (controls). This treatment-control comparison could be strengthened by also measuring fish density before treatments to confirm that fish abundance is similar among sites before any treatment (i.e., in a Before-After-Control-Impact [BACI] design; Stewart-Oaten et al. 1986; Manly 2001). In many cases, investigators estimate relative abundance by methods such as snorkeling supplemented with netting, but these are underestimates with unknown accuracy. Methods that include estimating capture probability are superior because they allow also estimating true abundance (White 2005). The most precise estimates of true abundance are achieved when the sites are enclosed to allow closed-population estimators to be used, such as mark-recapture and removal estimates (White 2008).

As an example of this design, Desgroseillier and Albrecht (2016) estimated density of juvenile Chinook salmon in five connected side channels of the Entiat River by electrofishing and seining during three seasons (spring, summer, fall) over three years and compared them to the density in adjacent mainstem habitats. Most of the side channels had been enhanced by reconnecting them or adding wood to increase their complexity. Density was about five times higher in off-channel than mainstem habitats during summer, although true abundance was not estimated and methods for estimating density in mainstem habitats were not reported.

2. Is fish abundance increased by habitat treatments, without affecting abundance in adjacent habitats? – One concern often raised by biologists and managers is that fish may move to use treated areas or habitat structures (i.e., be “attracted”), thereby depleting abundance from adjacent untreated reaches. This issue is difficult to address because fish can move from many different distances, including long distances from treatments (Gowan and Fausch 1996), so measuring any effects of depletion in adjacent habitats is challenging. However, one argument countering this concern is based on the first principles of density-dependent population regulation in salmonids (e.g., Elliott 1994), which supports the contention that habitat left vacant by fish that move will be filled by subordinate fish that would have died had they not found suitable habitat. For example, if this mechanism did not operate then no angler would be allowed to go fishing, for fear that habitat left vacant by fish caught would never be filled.

Nevertheless, new technology and redoubled efforts allow addressing this question, ideally using a Before-After-Control-Impact (BACI) design based on estimates of true abundance and density. For example, Polivka et al. (2015) measured density of juvenile Chinook and steelhead in “treatment” pools created by log or rock habitat structures (N=10), and compared them to adjacent “control” pools not associated with structures (N=15), as well as “natural” pools in other similar segments with no habitat treatments (N=41). Estimates of relative abundance based on snorkeling and netting over several seasons of the five years after treatment showed, for example, that density of juvenile Chinook was higher in treatment pools associated with habitat structures than adjacent control pools or natural pools in early and mid-summer. Moreover, densities were similar in control versus natural pools, indicating no detectable depletion of fish from control pools by moving to treatment pools. However, outmigration of summer Chinook juveniles eliminated these differences by late summer. This emphasizes the complexity of detecting responses for fish like salmon with complex life cycles that include use of many different habitats dispersed across watersheds and oceans. Here again, estimates of true abundance, if they are possible, would improve the strength of these inferences.

Table 4.1. A hierarchy of approaches for measuring effects of habitat enhancement on salmonids in streams and rivers. Each approach addresses a key question about how fish respond, and at what spatial scale the response is measured. Abbreviations: CH=Chinook; TRT=treatment; CTL= control; IMW=intensively monitored watershed; BDA=beaver dam analogs

Approach	Theoretical design	Actual design	Outcome	Constraints
1. Do fish use the area treated or the habitat created?	Replicated BACI design, based on sampling of enclosed areas to estimate true fish density	Replicated TRT-CTL comparison, N=5 "replicate" side channels, not chosen randomly, spring, summer, fall 2015 (methods for estimating density in mainstem not reported)	**Density of juvenile CH higher in enhanced off-channel habitats than mainstem habitats in the Entiat R.** (Desgroseillier and Albrecht 2016)	Use of habitats by fish varies seasonally. Unknown whether use of habitat causes increased fitness
2. Is fish abundance increased by the treatment, while not affecting controls?	Replicated BACI design, based on sampling of enclosed areas to estimate true fish density	TRT-CTL comparisons 5 yrs post-TRT (N=10 TRT pools, N=11 CTL, N=41 natural pools), based on relative abundance from snorkeling and seining	**Density of juvenile CH higher in treatment pools associated with habitat structures** in early and mid-summer, and similar in adjacent untreated control pools compared to natural pools in untreated reaches (Polivka et al. 2015)	True controls difficult because of fish movement (experimental units are open), and effects may vary seasonally
3. Has the treatment increased fish growth or physiological condition (as a measure of fitness)?	Comparison of growth rate of individuals known to have used restored habitat vs. "control" fish that used untreated areas	TRT-CTL comparisons for 5 yr post-TRT, N=238 TRT fish, N>3000 CTL fish	**Juvenile CH recaptured in restored pools grew faster than those not recaptured or that used untreated pools**, although total annual growth was apparently similar	Methods for analysis are still being developed and tested

4. Has the treatment increased survival of fish through one or more life stages?	Barker capture-recapture model to analyze a replicated BACI design	Replicated TRT sites using Barker model; N=8 "replicate" side channels, not randomly selected, no controls, 3 yrs post-TRT sampling	**Juvenile CH survival in side channels was higher during summer than winter, and higher in upstream than downstream segments of the Entiat River** (Desgroseillier and Albrecht 2016; Grote and Desgroseillier 2016)	Precise estimates of true survival require sufficient recaptures of permanent emigrants (e.g., smolts; Conner et al. 2015)
5. Has a set of treatments increased smolt production for an entire population?	Estimates of smolt production (smolts/redd) for many years pre- vs. post-treatment	No example found		High sampling and process variance on smolt production estimates limits power to detect all but large differences pre- vs. post-treatment. Effects of habitat restoration are confounded with those of hatchery releases.

3. Has the treatment increased physiological condition or growth? – Physiological condition or growth are more integrative measures of the importance of fish habitat than is abundance at a point in time. Growth of juvenile salmon, for example, is known to confer advantages for subsequent survival during smolting and ocean rearing. Therefore, a useful measure is to compare whether the growth rate of individuals that were marked and recaptured during a season in "restored" pools (those created by habitat restoration structures made from logs or boulders) is greater than those that were marked in restored or unrestored pools and never seen again (i.e., had no site fidelity near habitat structures).

Polivka and Mihaljevic (unpublished manuscript) reported that juvenile Chinook salmon in the lower Entiat River that showed fidelity to restored pools (15-60 d between recaptures) reached a larger size earlier in the season than those never recaptured, based on data from five separate years during 2009-2016. Nevertheless, by the end of the growing season individuals did not differ in length, regardless of habitat selection behavior. One explanation for this last result might be that smaller fish that used unrestored habitat had higher mortality, although this was not addressed by the authors.

4. Has the treatment increased survival of fish through one or more life stages? – A still more integrative measure of fitness, and a key component for any population model, is survival. Those fish that gain benefits from habitat restoration are expected to survive better during periods when they are using treated areas as well as perhaps after they leave the area and use other habitats. However, survival is estimated from marking and recapturing fish, so how can it be measured for marked fish that leave, or for those that immigrate but are not marked? The lack of information on these two groups could introduce unknown bias into estimates of survival.

Recent advances in modeling data from capture-recapture sampling allow better estimation of survival based not only on marking and recapturing animals in defined reaches, but also detections beyond those reaches using fixed PIT-tag antennas, mobile PIT-tag detectors, or other sources of recaptures. The Barker model (Barker 1997; Barker and White 1999; Conner et al. 2015) has been successfully used to incorporate diverse sources of recaptures that occur continuously in time (instead of only during discrete sampling occasions) to estimate true survival that is not confounded with emigration, rather than only apparent survival where the two are confounded, as estimated using Cormack-Jolly-Seber (CJS) models. As for all models, the precision of these estimates increases when more fish are marked and more effort is expended to detect both fish within reaches and those that emigrate permanently (e.g., smolts that emigrate from the system; Conner et al. 2015).

Desgroseillier and Albrecht (2016) and Grote and Desgroseillier (2016) marked juvenile Chinook salmon with PIT tags in six side channels along the Entiat River over 3 years, and detected them in subsequent sampling as well as retrieving the detections at downstream dams during smolt outmigration. Analysis revealed that their survival was higher during summer than winter, and higher in side-channels farther upstream in the basin than those downstream. This method holds promise to greatly improve the understanding of juvenile salmon survival through smolt outmigration and has been successfully used in other Intensively-Monitored Watersheds for juvenile steelhead (Bouwes et al. 2016). It is especially useful for steelhead, because they use tributaries longer than juvenile Chinook and so are more frequently detected, and are more likely to be detected as they emigrate permanently by smolting downstream (C. Saunders, Utah State University, personal communication).

5. Have the entire set of habitat restoration treatments increased smolt production for a salmon population? – The ultimate goal of habitat restoration is to increase the smolts per redd, or smolt production, for the population of salmon in an entire subbasin, such as the Wenatchee, Entiat, or Methow rivers. This requires measuring two quantities: the number of redds produced each year for that population in the subbasin and the smolts that outmigrate from the juveniles that emerge from those redds.

While conceptually simple, measuring these quantities is technically challenging and must be conducted for long periods to have hope of detecting relevant changes owing to habitat restoration (see below). Although estimating total number of redds is relatively straightforward, based on stratified random sampling or complete census, it requires substantial field effort. In contrast, measuring smolt output is done using floating screw traps, which sample only a small volume of the outmigrant smolts, or with fixed PIT-tag antennas located near the downstream ends of subbasins. As a result, the measurement variance on estimates of smolts tends to be large (e.g., Grote and Desgroseillier 2017), and when added to the process variance (e.g., natural fluctuations in smolt output year to year), results in large total variances. The upshot is that many years of sampling before and after habitat treatments are needed to detect even large changes in smolt production, based on power analyses (see below).

At present monitoring is underway in the Upper Columbia subbasins to generate these data, but they have not been analyzed to address whether there is evidence for a positive effect of habitat restoration. Analysis in the Methow River subbasin showed that egg-to-emigrant survival increased in the Twisp River over time, a tributary in which among the highest proportion of habitat available and treatable had received habitat restoration (U.S. Bureau of Reclamation Unpublished Data). However, there was also evidence for density-dependent survival of spring Chinook emigrants and smolts in this tributary, suggesting that rearing habitat or food may still be limiting. Overall, the problems described elsewhere with lack of suitable control reaches, movement of fish among treatment and control reaches, the large extent of treatments required to detect a response, and lag times for responses to occur remain difficult challenges for detecting effects of habitat restoration.

The influences of density dependence and hatchery fish

Each of these measures of the success of habitat restoration may also be complicated by density-dependent processes. For example, lower smolt output could result from too few adults, or, alternatively, too many adults. In the former case, if segments or whole subbasins are “under-seeded,” then the carrying capacity will not be achieved (see Bellmore et al. 2013 for an example in the Methow River, based on estimates of food production in floodplain habitats), and the predicted responses to habitat restoration may not occur. In the second case, high densities of adults can produce too many juveniles for the available carrying capacity, which then survive poorly and reduce smolt output, owing to food, space, disease, or other limitations at high densities, again confounding responses to habitat restoration.

Hatchery-reared juvenile salmon also increase density and place demands on food and space, thereby contributing to density-dependent processes. A worst-case scenario is where these hatchery juveniles usurp resources that would be used by wild juvenile salmon and yet have lower fitness and die at greater rates at some life stage, thereby wasting resources that could have supported wild salmon. Here again, this may eliminate some or all of the expected response to habitat restoration.

4.2.2. Evidence of Fish Responses to Habitat Restoration of Key Limiting Factors

Key limiting factors for juvenile spring Chinook salmon and steelhead production have been addressed by using seven main types of restoration actions grouped under the umbrella of habitat: 1) removing barriers to connectivity, 2) increasing streamflow, 3) reconnecting floodplains, side channels, and off-channel habitats, 4) restoring habitat complexity using log or boulder structures, 5) managing fine sediment, 6) restoring nutrients, and 7) controlling nonnative species.

Here we focus on evidence for fish responses to three types of projects that constitute the majority of habitat restoration actions: restoring connectivity (13% of all projects; UCSRB 2014a), reconnecting floodplains and off-channel habitats (15%), and adding habitat structures (21%). We address only briefly several other types of restoration projects (which together make up 17% of projects) and do not discuss projects that address assessment and design (17%).

Habitat protection projects (the remaining 17%) are an additional category considered to be of overwhelming importance and value by the UCSRB (2014b) and the Regional Technical Team in their Biological Strategy (RTT 2014). The objective of these projects is to protect existing areas with high ecological integrity where natural processes still occur. As of 2012, 46 projects were completed that protected 3,379 acres (2,728 floodplain acres) and 47 miles of stream (UCSRB 2014b). It is clear that preventing degradation of habitat that is already functioning is among the highest priorities, although it must be pursued judiciously, and not to the exclusion of restoring other key habitats (RTT 2014).

4.2.2.1. Restoring connectivity

The UCSRB set a high priority on restoring connectivity by removing barriers to upstream movement, or modifying them to become passible by fish, owing to the potential to allow access to large amounts of habitat quickly. A recent report indicates that by 2012, 93 barriers had been removed, opening 282 miles of previously blocked habitat (UCSRB 2014b). To date, this one type of restoration restored far more length of habitat than all other types summed together.

Success of projects that restore connectivity depends on the presence of suitable habitat upstream, a downstream source population, and the distance upstream available for colonization (Hillman et al. 2016). Studies within and outside the Columbia River Basin indicate that colonization of upstream habitats opened to fish can be rapid, and that abundance is similar above and below barriers after connectivity is established, indicating that fish have recolonized the opened habitat. O'Neal et al. (2016) reported that fish passage projects in nine rivers from across a wide range of river types in Washington and Oregon produced significant increases in densities of juvenile coho salmon and all salmonids combined, based on comparisons for 1 year pre-treatment versus 1, 2, and 5 years post-treatment, as well as increasing post-treatment trends for these groups through time ($P < 0.10$ for each). However, no such differences were detected for juvenile Chinook salmon. It is unclear at how many locations Chinook salmon were present, which may have reduced the sample size for this species and the power to detect the differences.

A good example of restored connectivity in the Upper Columbia is the modification of water diversions to allow fish passage on Beaver Creek, a tributary of the lower Methow River, although this project affected steelhead more than Chinook salmon. The diversions were modified by creating rock-vortex weirs that impounded enough water to allow diversions for irrigation, while simultaneously allowing fish to pass upstream. Many juvenile steelhead and some juvenile Chinook salmon ascended the newly opened stream segment over the 4-year monitoring period (Martens and Connolly 2010). During the final years of the evaluation adult steelhead returned which had emigrated from the segment as parr, thereby confirming that an anadromous population of this species had been re-established (Weigel et al. 2013). No information was available on adult Chinook salmon returns.

Overall, studies from both within the Upper Columbia River region (UCSRB 2014b), and other locations within and outside the Columbia River Basin (reviewed in Roni et al. 2008, 2014; Hillman et al. 2016), have reported that colonization of newly opened habitat by anadromous salmonids like spring Chinook salmon can be rapid and that densities can reach equilibrium within a few years. Although few detailed studies of colonization were found for spring Chinook in the UCR, this habitat restoration action remains among the highest priority and the most certain to have positive biological benefits.

4.2.2.2. Reconnecting floodplains and off-channel habitats

Chinook salmon typically inhabit larger rivers with floodplains, and habitats in these off-channel areas provide important rearing areas for many salmonids (Hillman et al. 2016). Restoration projects may include reconnecting side-channels and off-channel ponds and wetlands, removing or moving levees to allow channels to meander and connect with the floodplain, or constructing off-channel habitats and restoring meanders. By 2012, more than 2700 acres of off-channel habitat had been protected, 117 more acres reconnected, and 11 miles of off-channel streams restored, representing the second largest number of restoration projects (UCSRB 2014b).

In general, studies of the effects of these projects across the Pacific Northwest show rapid recolonization of the newly accessible habitat by various salmonids (Roni et al. 2008; Hillman et al. 2016). Chinook salmon are reported to use side channels fed by surface water more than ponds or wetlands with little flow, at least in some regions (Pess et al. 2008).

Several studies in the Upper Columbia provide direct data on the biological benefits of off-channel habitats for Chinook salmon and other salmonids. Desgroseillier and Albrecht (2016) and Grote and Desgroseillier (2015) reported higher densities of juvenile Chinook salmon in six off-channel habitats of the Entiat River (most were created or enhanced based on natural floodplain features) compared to main channel habitats, indicating that fish colonized and used the newly accessible habitats. Survival of juvenile spring Chinook salmon in the off-channel habitats was similar during winter compared to summer (20% vs. 24%, respectively), despite the harsher winter conditions. Moreover, survival in two off-channel habitats nearer the headwaters was 10-15 percentage points higher during summer and winter than three of these habitats near the river mouth, though the authors did not discuss reasons for the difference.

Upwelling groundwater sources common in floodplain habitats also enhance the food web that supports Chinook salmon growth. Meija et al. (2015) reported that sites “gaining” groundwater in the Methow River were warmer and had more nitrogen, periphyton, and benthic invertebrates than sites in downwelling (“losing”) reaches. This resulted in nearly twice the growth rate of hatchery juvenile Chinook salmon in enclosures during a field experiment, as well as faster growth by free-ranging wild Chinook in gaining versus losing reaches.

Floodplains also store organic matter and nutrients deposited during floods, and so are rich sources of production of invertebrates that feed fish. Bellmore et al. (2013) measured the trophic basis of production for juvenile Chinook salmon, steelhead, and other fishes in five floodplain channels of different types (e.g., a side channel connected downstream and a disconnected channel not scoured by floods during the study). Based on extensive sampling and analysis of fish (consumers) and their invertebrate prey, they found that in the main channel of the Methow River about half to two-thirds of the production of invertebrate foods available to support anadromous Chinook and steelhead was consumed by other native fish, including mountain whitefish (*Prosopium williamsoni*) and several sculpin species (*Cottus* spp.). Moreover, these non-target fishes consumed 95% of the total prey consumed by fish in this habitat. In addition, production available to the anadromous salmonids in the side channels on the floodplain was more than 2.5 times that in the main channel, even though much of it was not used, apparently because of low densities of these anadromous salmonids (i.e., underseeding). This indicates that floodplain reconnection potentially provides additional food resources to spring Chinook juveniles.

Following on this work, Bellmore et al. (2017) developed a model of the Methow River ecosystem, the Aquatic Trophic Productivity Model, and used it to evaluate three restoration scenarios: restoring riparian vegetation, nutrient augmentation by adding salmon carcasses, and reconnecting side channels. Their results showed that reconnecting side channels had much larger potential effects on native fish biomass (an estimated 31% increase) than carcass addition (18%), and that riparian revegetation would have little effect (2%). However, this result was altered in model runs that included, in addition, invasions by nonnative New Zealand mudsnails (*Potamopyrgus antipodarum*) or potential nonnative fish predators such as smallmouth bass *(Micropterus dolomieu*). Responses in fish biomass were still greatest for reconnecting side channels, except in the worst-case scenario where both snails and fish invaded, which reduced predicted fish biomass below that predicted for carcass additions under this same invasion scenario.

Overall, restoring and reconnecting floodplains provides a wide range of benefits, especially for rearing juvenile Chinook salmon, and is a high priority for habitat restoration. The models of Bellmore et al. (2013, 2017) will prove useful to help managers think broadly about which restoration actions are most likely to benefit target species like Chinook salmon and also to compare their relative cost-effectiveness.

4.2.2.3. Adding habitat complexity using structures

Adding instream structures such as boulders and engineered log jams is the most common type of habitat restoration project, making up more than a fifth of all projects including those for habitat protection and for assessment and design. By 2012, these projects added 518 structures, created 180 pools, and enhanced 22 miles of stream in the UCR (UCSRB 2014b). Despite this activity, adding habitat structures to rivers and streams has also received the greatest scrutiny from restoration scientists, who often question whether they actually increase fish numbers or biomass (e.g., Stewart et al. 2009; Whiteway et al. 2010; Roni et al. 2015). The general hypothesis is that adding structural complexity to channels will increase survival during summer and overwinter by providing refuges from flow, visual isolation from conspecifics and aquatic predators, and overhead cover from terrestrial predators (Fausch 1993).

Although some analyses have reported equivocal effects (Stewart et al. 2009; but see comments on flaws in this study by Whiteway et al. 2010), most report positive results overall (Whiteway et al. 2010; Roni et al. 2015). For example, Hillman et al. (2016) reported that about 90% of the 83 studies they reviewed showed positive effects of placing habitat structures made of large wood on physical habitat, and about 70-80% reported positive effects on juvenile or adult salmonids (N=67 and 33 studies, respectively; Fig. 4.2), with less than 3% of studies showing negative effects in any of these cases (the rest were equivocal). Even with an unknown publication bias against negative results (i.e., investigators finding no effect or a negative effect are unlikely to attempt publication or successfully publish results), these results are biologically significant.

Likewise, it is unrealistic to expect that instream structures will improve physical habitat and increase fish numbers or biomass equally in streams of all types across all biomes, just as we would not expect all cancer drugs tested to be equally effective for all cancers or across all patient groups (Mukherjee 2010). For example, O'Neal et al. (2016) showed positive effects of adding structures on three characteristics of physical habitat (pool area, pool depth, volume of wood), but no effects for juveniles of four salmonid species (Chinook, steelhead, coho, bull trout), perhaps because the 12 projects they analyzed were scattered across various ecoregions in Washington and Oregon. None were in the Upper Columbia River.

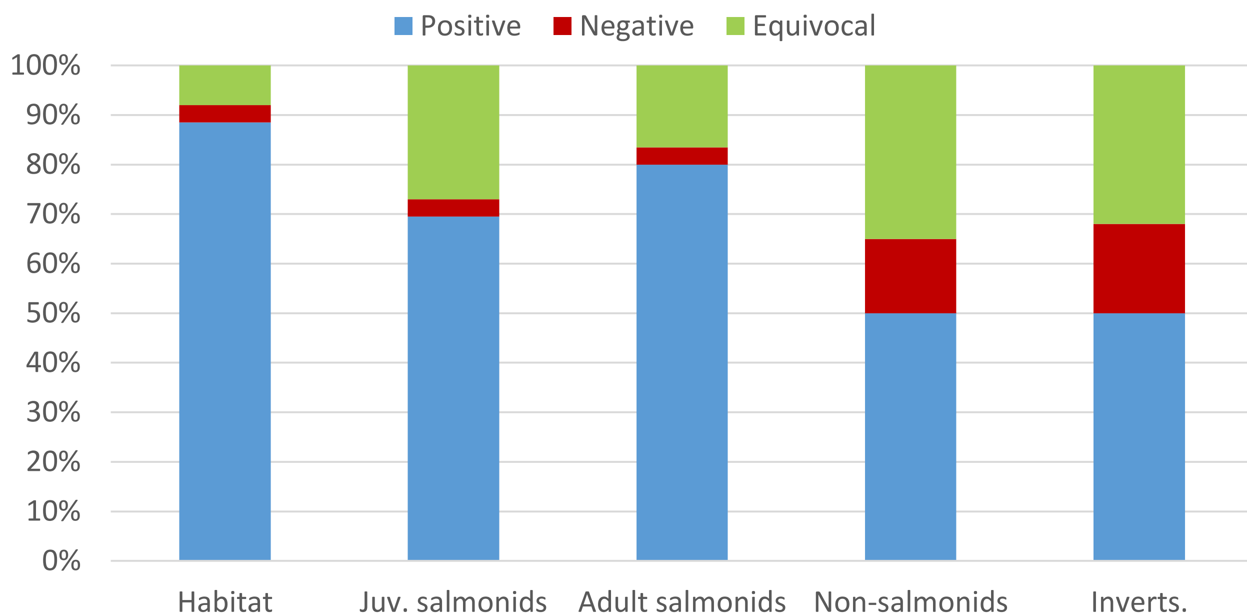

Figure 4.2. Proportion of published studies of placing structures made of large wood that reported positive effects, negative effects, or no change (equivocal) in physical habitat, fish (juvenile and adult salmonids, or non-salmonids), or macroinvertebrate density or diversity (Inverts). Number of studies (n) was 83, 67, 33, 17, and 21 for each case, from left to right. Some studies reported responses for several categories (from Hillman et al. 2016, after Roni et al. 2015).

Several studies provide direct data on the effectiveness of instream structures in the UCR. Polivka et al. (2015) reported that pools associated with instream structures made of boulders or logs had higher abundance of juvenile Chinook salmon during early and mid-summer, compared to adjacent natural pools without structures, or natural pools in other nearby segments with similar habitat. However, many of these juveniles were apparently summer Chinook, which outmigrate by late summer, resulting in no detectable differences just afterwards. Nevertheless, this work serves to address the concern that structures may draw fish away from adjacent areas and argues for the view that the structures increase carrying capacity rather than simply redistributing fish already present.

Roni et al. (2008, 2014, 2015) and Hillman et al. (2016) reviewed the large number of studies on instream structures in other parts of the Columbia River Basin and beyond. The best example to date of a comprehensive design and analysis for detecting the effects of instream habitat structures on anadromous salmonids in the Columbia River Basin is the research conducted under the CHaMP/ISEMP program in the Bridge Creek IMW in the John Day River basin of central Oregon. Beavers create important habitat in this high-desert basin, and beaver-dam analogs (BDA) have been used there to mimic the effects of beaver dams as well as to encourage beavers to build their own dams (Pollock et al. 2014).

A 7-year field experiment based on a BACI design, including four replicates of treatment reaches with BDA and three reference reaches in a similar watershed nearby (which, therefore, were fully independent) was used to measure the density, growth, survival, and production (g/m/day) of juvenile steelhead in response to the beaver dam analogs (Bouwes et al. 2016). Fish were PIT-tagged in study reaches but also detected with fixed and mobile PIT-tag antennas in other reaches and downstream locations, allowing use of the Barker model (a modification to the Cormack-Jolly-Seber model for open populations) to estimate true survival rates. Analyses detected much greater density of juvenile steelhead (an increase of 81 fish/100 m), lower growth (owing to density-dependence, given the higher density), a 52% increase in survival, and a 175% increase in juvenile steelhead production in the treatment reaches compared to those in reference stream reaches. In addition, the structures moderated diel temperatures because of increased water storage and interaction with groundwater, benefiting salmonids by buffering temperature extremes and providing thermal refuges (Weber et al. 2017). This example from the CHaMP/ISEMP program is one of the most comprehensive and successful evaluations of instream structures completed for an anadromous salmonid population to date, anywhere in the world.

4.2.3. Application of Life-Cycle Models to Integrate Habitat Restoration with Other Effects

An important goal of the UCSRB is to use Life Cycle Models (LCM) to analyze the effects of habitat restoration that is coincident with other effects likely to be caused by hatcheries, hydropower, harvest, and variable ocean conditions (see Roni et al. in press for review of methods for assessing habitat restoration). Toward this end, the CHaMP/ISEMP group is developing a LCM for the Entiat River, which is also an Intensively Monitored Watershed (IMW).

The Entiat River IMW was established in 2011, and habitat projects were conducted in 2012 and 2014. Saunders et al. (2017) reported on development of a LCM and its use in a preliminary analysis, which is similar to the LCM developed for the John Day River IMW. They used this model to analyze the effects of a subset of the habitat projects conducted in 2012 in the Entiat River. The analysis showed that this subset of projects increased rearing capacity for juvenile Chinook (spring and summer runs combined) an average of 7% across the sites treated, amounting to a <1% increase in carrying capacity for the whole basin. When combined with an assumed 2% increase in overwinter survival expected owing to the restoration, the predicted abundance of spawners increased about 25%, from about 230 to 290, but not enough to reach the goal of 500 spawners set in the Recovery Plan.

Despite these model assumptions and predictions, the analysis of empirical data to date detected no change in juvenile Chinook salmon survival, and also had low statistical power to detect changes in abundance (S. Walker, TerrAqua, personal communication). A post-hoc analysis of statistical power indicated that, given the variability measured to date, only very large year-over-year changes in Chinook abundance (i.e., ≥60%) owing to restoration could be detected with statistical confidence. A further analysis using the LCM to estimate the increases in current habitat capacity and survival needed to reach the recovery goal of 500 spawners indicated that a 10% increase in each was predicted to meet the goal, and that a 5% increase in each might achieve the goal.

Firmer conclusions from this work must await further analysis of the habitat actions completed to date. However, the preliminary results from the IMW in both the John Day and Entiat rivers (Saunders et al. 2017) indicate that the habitat restoration scenarios evaluated to date, especially those using log and boulder structures, are unlikely to have sufficiently large effects alone to increase either habitat capacity or survival enough to reach recovery goals. Although adding habitat complexity using these structures is an important objective as part of a comprehensive habitat restoration strategy, the relatively larger biological effects of restoring fish passage by removing barriers and restoring and reconnecting floodplains are likely to be more important goals in the near term. Likewise, managing riparian zones and uplands to deliver large wood, and managing channel geomorphology to allow storing it in floodplains and channels, are important long-term goals.

4.2.4. Sample Designs and Sample Sizes Needed to Detect Biologically Significant Effects

Data on fish abundance, growth, survival, and smolt production are inherently variable, owing to both sampling error (variability owing to the sampling process) and process error (e.g., annual variability due to years with different temperature or flow regimes). Process error is often much greater than sampling error, so many years of data are needed to detect effects with acceptable statistical power and confidence, even in a simple design comparing effects before vs. after habitat restoration.

If a BACI design can be used, perhaps by employing an adjacent control basin like the Chiwawa River basin used for the Entiat River IMW (S. Walker, personal communication), the analysis is more complex. In summary, however, with as much as 5 years of data for the periods before and after the treatment, and typical levels of sampling and process error, it is likely that only a doubling or halving of abundance could be detected with acceptable statistical power and confidence (see Appendix E for discussion of BACI designs).

In a similar analysis, Paulsen and Fisher (2005) reported that given five control sites and three treatment sites at which 10 years of data were collected before habitat manipulations and 7-9 years of data collection occurred afterwards, an investigator can detect a 30% increase in parr-to-smolt survival (based on tagging juveniles and detecting them at a downstream dam) at acceptable levels of statistical power (80%) and significance (alpha=0.05).

This example shows that a key requirement is a long run of pre-treatment data, which is counterintuitive to most practitioners and unlikely to be available or planned. That is, when funds are available for habitat restoration, most practitioners want to complete projects, rather than wait to measure pre-treatment data and risk losing funds. However, O'Neal et al. (2016) showed that for analyses of habitat restoration, adding 1 year of pre-treatment measurements increased the power to detect differences more than including up to 100 years of post-treatment data. Increasing pre-treatment data from even 1 year to 2 years improved power by about 30%, and 5 years of data before and after treatments appeared optimal.

4.3. Approaches for Prioritizing Habitat Projects

4.3.1. Ecosystem Diagnosis and Treatment (EDT) Model

The Ecosystem Diagnosis and Treatment (EDT) model is a life-cycle habitat model that can characterize salmonid ecosystems temporally and spatially (ICF International 2017; Roni et al. in press). The model is built to assess salmon performance in terms of capacity (i.e., maximum population size), productivity (i.e., survival at low density), and diversity (i.e., life history trajectories linked to coordinates of time and space; Mobrand et al. 1997). EDT uses a multiple-stage Beverton-Holt production model with species and life-stage specific performance benchmarks for optimal environmental conditions (Blair et al. 2009). The environmental conditions considered include physical habitat and water quality conditions as well as food availability.

To characterize the environment, life stage survival factors are derived from environmental attributes based on raw data that comes from field data (preferred), information derived from other attributes, expert opinion, or hypotheses (Blair et al. 2009). Environmental attributes include hydrologic characteristics (e.g., flow, hydrologic regime), stream habitat structure (e.g., channel morphometry, habitat type, obstructions, sediment), water quality (e.g., chemistry, temperature), and biological community effects (e.g., biotic interactions with competitors, predators, parasites and pathogens, and macroinvertebrates). The model also considers the percent of key habitat (i.e., primary habitat types such as pools, glides, riffles) for different life stages within each geographic unit, and a food factor that incorporates alkalinity, benthic community richness, riparian function, and salmon carcasses. The life stage survival, key habitat, and food factors are used to adjust the performance benchmarks to evaluate current or potential habitat scenarios (Blair et al. 2009). The results can be related to the Viable Salmonid Population (VSP) criteria of McElhany et al. (2000; see Arterburn 2017; ICF International 2017).

The EDT model was used for the Fish and Wildlife Program's 2004 Subbasin Plans for the Entiat, Methow and Okanogan watersheds (see plans for each basin in www.nwcouncil.org/fw/subbasinplanning). More recently in the Upper Columbia Basin, the Okanogan Basin Monitoring and Evaluation Program (OBMEP) has been integrated with EDT to track trends in data quality and data collection needs, identify priority areas for habitat protection and restoration, provide a method for evaluating biological effectiveness of restoration actions, and to evaluate strategies for reintroducing extirpated species (Colville Confederated Tribes 2013). In collaboration with the Colville Confederated Tribes, ICF International has developed Salmonid Population Report Cards for populations and reaches to visually summarize OBMEP data and EDT output (Colville Federated Tribes 2013; Arterburn and Klett 2015; Arterburn 2017; Eric Doyle, ICF International, personal communication).

EDT modules can be downloaded without charge. However, the development of an EDT application for a particular basin requires an initial investment with ICF International who currently maintains the software. For example, an EDT application for the Methow River basin was started in 2016, with the initial release expected in early 2018. Developing the Methow EDT model involved upgrading the EDT model for the Methow Subbasin Plan to the latest version of EDT, reconfiguring the reaches based on lessons learned from the Okanogan EDT model, and incorporating new habitat data collected since 2004. The approximate cost for developing the Methow EDT model is $250,000. Once the application has been built, maintenance costs of $100,000 per monitoring cycle are likely (Eric Doyle, ICF International, personal communication), although the costs decline over time as model use is standardized in a basin (e.g., monitoring cycle costs in the Okanogan are estimated at <$60,000; J. Arterburn, Colville Tribes, personal communication).

4.3.1.1. Applicability of EDT for ranking of habitat projects

The EDT approach seems well suited to ranking and evaluating the effectiveness of habitat projects. It has the advantage of incorporating food availability, and the authors suggest that it can be linked to other models of land use that predict input variables to EDT like sediment, thereby addressing factors advocated by ISAB as important for assessing salmon recovery (ISAB 2011-1; ISAB 2011-4; Naiman et al. 2012). The approach incorporates rule-based relationships between species survival and environmental characteristics that are based on prevailing scientific knowledge (ICF International 2017). It is able to evaluate the habitat needs for discrete life stages and has been designed to provide data that relate to VSP criteria. The model can be used to indicate reaches and actions that could improve VSP criteria, thus providing a means to prioritize actions.

ICF International (2017) is clear that the model is not a predictive model (i.e., it does not forecast future conditions), and it is difficult to validate the model against measured data (Steel et al. 2009). A review of subbasins plans by ISAB and ISRP concluded that EDT results can be considered hypotheses that should be tested (ISAB/ISRP 2004-13). Thus, it seems plausible that if long-term monitoring of fish responses and environmental conditions is done after projects are implemented, then empirical estimates of the VSP parameters could eventually be compared against predictions from the model to validate them, and indicate model parameters that require refinement. ICF International (2017) describes this type of validation exercise conducted for five steelhead and four salmon populations (Rawding 2004). Model estimates of productivity (alpha) and capacity (beta) for the Beverton-Holt curve were within the 95% confidence limits for the nine populations.

In the Okanogan Basin, removal of culvert barriers and an irrigation knife gate on Loup Loup Creek enabled fish passage in 2014. The EDT model had estimated that the additional habitat should support an adult abundance of 27 summer steelhead, and in 2014, exactly 27 wild summer steelhead returned to the assessment unit. Over time, the EDT model predicted a 5-fold increase in juvenile abundance with the restoration actions on Loup Loup Creek, but OBMEP has found an increase closer to 10 times the abundance prior to restoration (J. Arterburn presentation to ISAB, July 20, 2017). Although in this case the EDT predictions of adult abundance happened to be the same number of steelhead seen in 2014, EDT estimates should be considered as relative, rather than absolute estimates (ISRP/ISAB 2004).

Sensitivity analyses were performed on EDT by the U.S. Bureau of Reclamation (Reclamation), Washington Department of Fish and Wildlife (WDFW), and NOAA Fisheries (Steel et al. 2009; McElhany et al. 2010). Steel et al. (2009) reported that Reclamation found that EDT was relatively insensitive to flow attributes in the Yakima basin, which were important to their decision-making, so they developed additional models to predict values for several habitat and temperature attributes incorporated in EDT that are most influenced by changes in stream flow. A sophisticated sensitivity analyses conducted by NOAA Fisheries indicated that prediction intervals for estimates of abundance, productivity and capacity had large ranges, leading McElhany et al. (2010) to caution against using EDT for management decisions that require a high degree of confidence in these outputs, such as setting harvest limits or endangered species goals. However, the model was fairly robust in ranking the highest-priority reaches for preservation or restoration (Steel et al. 2009; McElhany et al. 2010), so it is likely to be useful for this goal in the Upper Columbia. McElhany et al. (2010) also reported that the model required a large amount of time to calculate outputs for the sensitivity analysis, which could be a constraint for model users.

One concern reported by McElhany et al. (2010) was the limited ability for model users to adjust some input parameters. ICF International has revised the model since the assessments by McElhany et al. (2010) and Steel et al. (2009) to create a more modular structure to the model that allows the rule structure of the model to be modified (Eric Doyle, ICF International, personal communication). The presentation by Arterburn (2017) to the ISAB indicated that the Okanogan River application of EDT has incorporated adjustments to input parameters. Additionally, because OBMEP raw data are integrated with EDT, the model can be part of an adaptive management strategy to refine and update the relationships between environmental attributes and survival factors. EDT includes a rating for the quality of Level 1 data (i.e., raw data and observations), noting that empirical field data are the most desirable (Blair et al. 2009; Colville Federate Tribes 2013). The OBMEP report cards provide an overview of data quality so that progress can be tracked on obtaining data for the models. The report cards also show progress towards goals as well as priorities of projects both spatially and by project types.

Another concern reported by ISRP/ISAB (2004) was the limited application of EDT for hatchery interactions. EDT does include hatchery releases in its accounting for density dependence and competition. For more detailed assessment of hatchery-wild interactions and impacts to populations, the Colville Tribes uses the productivity and capacity values from EDT with the All-H Analyzer (AHA) spreadsheet model (J. Arterburn, Colville Tribes, personal communication). The AHA model is available from Lars Mobrand (see Appendix G of ICF Jones & Stokes 2010 for a description).

Should EDT continue to be developed and applied in the Upper Columbia Basin and the rest of the Columbia River Basin, a scientific evaluation of the species habitat rules might be prudent. At the time that EDT was developed in the 1990s, much less data were available to establish the rules. Satellite imagery, monitoring data, and data from experimental studies could be useful for revising or updating EDT rules.

4.3.2. Habitat Suitability Indices (HSI)

Habitat suitability index (HSI) approaches were initially developed in the 1980s (Roloff and Kernohan 1999). For aquatic species, HSIs typically represent preferences of each species for instream variables (e.g., velocity, depth, substrate, and cover) at different life stages (Ahmadi-Nedushan et al. 2006). Indices generally range from 0 (no preference) to 1 (maximum preference) for a particular habitat condition and can be combined to develop a composite suitability index. For example, depth, velocity and cover HSIs might be combined by multiplying the individual indices, which would result in zero suitability if any of the variables were considered unsuitable (Ahmadi-Nedushan et al. 2006). Such composite indices assume that all variables are equally important and that all environmental variables are independent with no interactions. To address the first assumption, weights can be used to incorporate relative importance of variables. However, interactions among variables cannot be addressed with HSI (see Ahmadi-Nedushan et al. 2006 for an analysis of several alternative approaches to HSI to address interactions).

There have been criticisms of HSI models used for evaluating habitat and population responses to habitat quality (Roloff and Kernohan 1999; Ahmadi-Nedushan et al. 2006). Although the models do quantify habitat suitability, this metric may or may not be directly correlated to fish abundance and biomass (Rose et al. 2015) because other factors like food may have overriding effects. As Booth et al. (2016) point out, more habitat does not guarantee desired biological gains, especially if a clear understanding of the relationship between habitat and the life history of the target species is lacking. Roloff and Kernohan (1999) assessed 58 HSI model validation studies and found that the applications failed to account for variability in input data, which in turn affects HSI output interpretation. Other shortcomings noted were model applications at inappropriate spatial scales, inability to account for density-dependent effects, and lack of sufficient duration of observations to test model performance (Roloff and Kernohan 1999).

In the mid-Columbia River Basin, McHugh et al. (2017) applied a spawning HSI to estimate spawner capacity in egg equivalents for steelhead in the Middle Fork John Day Basin. This HSI model was used with inputs of flow depth and velocity from a hydraulic model (Delft3D) to assess the quality of spawning habitat across space within reaches. The spawning capacity was then upscaled from reaches to the entire basin using statistical modeling based on bankfull width. The demographic parameters predicted from these analyses were used as input to the life-cycle model (LCM) for the basin. The combined models were used to assess riparian restoration scenarios that considered 1) baseline (or status quo) conditions, 2) best-case riparian revegetation, channel adjustment, and instream flow acquisition, 3) complete maturation of current (i.e., 2008) vegetation, and 4) targeted placement of instream wood structures. This modeling effort also included use of a net rate of energy intake (NREI) drift-foraging model of juvenile steelhead habitat capacity (McHugh et al. 2017). A similar approach has been used by Saunders et al. (2017) in the Entiat River basin for spring Chinook. Use of an HSI model for spawner capacity may be among the best applications of this method, because fish are selecting habitat for a defined period based on a few key variables that can be measured and modeled (i.e., depth, velocity, substrate, and cover).

In the Upper Columbia Basin, HSI models have only been applied at the spatial scale of individual projects. Welch (2017) described the use of HSI models with hydraulic models for project selection and design, indicating that Bonneville Power Administration (BPA) would be requesting this type of analysis as part of statements of work. They noted that HSI models were useful for bridging the gap between biologists who could identify habitat suitability criteria and engineers who needed to design projects.

Busch et al. (2013) developed an intrinsic potential (IP) model for spawning fall Chinook salmon in the Lower Columbia River evolutionarily significant unit (ESU) that is a variation of habitat suitability models (see Roni et al. in press for descriptions of these model types). Their IP model described potential habitat by using 10-m digital elevation models to rate habitat characteristics of channel confinement, channel width, and channel gradient. Busch et al. (2013) performed a sensitivity analysis of the shapes of suitability curves and found that results for some watersheds were very sensitive to curve shapes. Correlations between model results and available fish abundance data at the population level were not significant, and they were unable to examine the ability of the model to predict fish data spatially due to lack of data. Nevertheless, output from the IP model was significantly correlated with corresponding EDT output at the reach and population level.

4.3.2.1. Applicability of HSI for ranking of habitat projects

The HSI-LCM applications in the Middle Fork John Day Basin were designed to assess what type of restoration project (e.g., riparian restoration, addition of wood) was better suited to improve population abundance of steelhead, but the applications did not appear to be used to prioritize placement of such projects (McHugh et al. 2017). Welch (2017) indicated that HSI models coupled with hydraulic models would be required for project selection and design in the Upper Columbia Basin but did not describe how these approaches would be used to prioritize habitat projects (e.g., what criteria would be used, how projects would be evaluated). Roni et al. (in press) provide a comprehensive review comparing methods used to prioritize habitat restoration.

4.3.3. Regional Technical Team (RTT) Biological Strategy

The Upper Columbia Regional Technical Team (RTT) started as an ad-hoc group of scientists who wanted to coordinate and prioritize restoration actions (Kahler 2017). Representatives of the agencies involved have agreed to “check their agency affiliations at the door.” The RTT Biological Strategy involves using what they term planning science, review science, and adaptive management in salmon recovery. Planning science involves using tools like ecological models (e.g., EDT, Shiraz, Physical Habitat Simulation [PHABSIM], and life-cycle models), reach assessments, published literature, and local research and expertise to identify limiting factors, improve the biological strategy, inform recovery plans and habitat actions, and develop the RTT priorities spreadsheet. Reviewing science involves RTT evaluation of projects once they have been proposed. As part of adaptive management, the RTT synthesizes monitoring and makes recommendations to the UCSRB, regional co-managers, and the Upper Columbia River Umbrella Project (Kahler 2017).

The RTT’s Biological Strategy (RTT 2014) states that the highest priority for *protecting* biological productivity is to sustain natural geo-fluvial processes, such as unrestricted channel migration, floodplain function, and adequate stream flows, especially when these processes are already functioning at a high level. In contrast, the highest priority for *increasing* biological productivity is to restore stream channel complexity and floodplain function. The RTT recognizes that there must be a pragmatic balance between protection and restoration actions, with recovery as the goal.

The Biological Strategy includes a description of how projects are prioritized and rated. The major subbasins in the UCR (Wenatchee, Entiat, Methow, Okanogan) are divided into assessment units, each of which is a portion of the subwatershed (e.g., lower Wenatchee River) or a tributary (Nason Creek). Assessment units are assigned one of four priorities, with the top priority assigned to areas with high quality functioning habitat. Within these units, protection and restoration actions are defined in two tiers, with the top tier designating the highest priority actions that could be accomplished under ideal conditions, such as when most land ownership is public. For example, Tier 1 protection actions protect high-quality geo-fluvial processes, and Tier 1 restoration actions restore fluvial geomorphic processes. The report has tables that indicate the priorities for addressing ecological concerns (i.e., limiting factors) for each assessment unit, although coordination of the priorities with the Viable Salmonid Population (VSP) criteria was not clear.

4.3.3.1. Assessment of RTT Biological Strategy for prioritization of habitat projects

The ISAB raises two major concerns about the ranking criteria used to prioritize projects. Analysis of cost-effectiveness is not quantitative and is given a low weighting in the final rankings. Potential projects are ranked sequentially by two groups within the UCSRB. First, cost effectiveness is considered, but only after the technical criteria have been scored. The RTT developed two scoring rubrics to assess priorities for protection projects and restoration projects. For protection projects, seven criteria are scored and weighted, with the highest weights given to criteria protecting habitat at locations important to fish and protecting habitats that would result in the greatest loss of freshwater habitat capacity or fish survival if they were lost. For restoration projects, 10 criteria are scored and weighted, with most weight on criteria considering how well projects address the ecological concerns (i.e., limiting factors), whether they are appropriately located and scaled, and the expected benefits to habitat capacity and fish survival. Several criteria also address the lag time before benefits of restoration projects are expected and the duration they will persist. However, for each rubric, cost effectiveness is considered in imprecise terms, but not explicitly calculated, and makes up only 5% of the total score. The project rankings by the RTT are then submitted to the Citizens' Advisory Committee, which considers benefits to fish (the RTT ranking; 40%), project longevity (20%), project scope (10%), community support (17%), and economics (13%). As with the RTT, the cost effectiveness analysis is not quantitative and is given a low weight. Final funding decisions are made by the UCSRB and the Salmon Recovery Board of the state of Washington. Reportedly, cost effectiveness is considered by these regional entities, but there is no explicit analysis and the evaluations are not documented. Independent estimates of cost-effectiveness metrics, like those described in section 4.1, are not calculated in either local or regional rankings and decisions.

The second major concern is the lack of a formal approach for weighting both cost and time elements of projects. As discussed in Section 4.1, projects that produce benefits sooner rather than later, over a much longer period, or at a fraction of the cost of other projects, need to be weighted proportionally to allow maximizing objectives within constraints of limited time and resources. Cost differences between projects are as important as benefit differences for achieving program objectives. Cost-effectiveness should be determined using the ratio of biological benefits (in ecological units) to costs (in monetary units), instead of implicitly addressing them in other scoring components. Providing resources equally among geographic units instead of accounting for the relative cost effectiveness of different actions can lead to the lowest possible benefits to society (Wu et al. 2003).

Coordination of the priorities with the Viable Salmonid Population (VSP) criteria also was not clear. Despite the understandable hesitation to estimate the numbers of additional fish that a project will produce in years 1-10, 11-25, and 26-50, those estimates are needed to be able to give more weight to benefits that come sooner rather than later, by discounting future benefits (see Section 4.1.2). These time-weighted estimated benefits are then divided by project costs to produce a time-weighted measure of benefits per dollar of cost. This metric allows prioritizing projects to achieve the greatest overall benefit for the lowest cost in the shortest time period. Similar methods are required for projects in most federal and state agencies (EPA 2010).

4.4. Coordination and Interactions with Other Hs

The Upper Columbia Salmon Recovery Board (USCRB) works with partners to facilitate recovery actions in the Wenatchee, Entiat, and Methow basins, including creating an overall operational structure, developing explicit strategies, and coordinating actions. The UCR Recovery Plan for spring Chinook salmon is designed to increase the capacity of the habitat to produce outmigrants (smolts/redd), resulting in greater numbers of returning adults (UCSRB 2014b). However, other limiting factors such as mortality during downstream migration, poor ocean conditions, or predation may reduce these benefits. Hence, habitat actions must be coordinated to complement actions in other sectors and address the most important limiting factors and interactions among factors.

The UCSRB has developed one of the better examples of a well-documented process for prioritizing restoration projects and coordinating recovery actions within groups responsible for habitat, hydrosystem, hatcheries, and harvest. The regional recovery plan, improved limiting factor assessment, life-cycle models, and monitoring provide critical information for recovery actions for spring Chinook salmon. One of the greatest challenges for the UCSRB is coordination of the different groups in the three basins that are responsible for the different Hs. Currently, there is no process for regularly discussing the findings and actions of the different groups and integrating the separate efforts into a coordinated action plan.

4.4.1. Operational Structure

As described for the prioritization of recovery actions, the UCSRB has the responsibility to develop the Recovery Plan for the UCR and coordinate habitat actions with hatchery operations, harvest, and hydropower (www.ucsrb.org/). The RTT developed the Upper Columbia Biological Strategy as a regional framework for developing actions to recover native salmonids stocks (RTT 2014). The RTT provides the technical basis for the regional strategy and reviews habitat projects at several stages of development and implementation.

Coordination by the UCSRB also considers all watershed plans in the Upper Columbia Region to address habitat objectives of the recovery plan (www.digitalarchives.wa.gov/GovernorLocke/gsro/regions/upper.htm). The UCSRB submits project lists to the Salmon Recovery Funding Board of the state of Washington. In addition, the Board receives funding for habitat actions through the Umbrella Projects of the NPCC. Counties and co-managers lead the implementation of the subbasin plans. The UCSRB attempts to align local watershed plans and the regional subbasin plans into the regional salmon recovery plan.

4.4.2. Strategies

Creating the Recovery Plan relies on integrating actions across all sectors affecting salmon and steelhead (harvest, hatcheries, hydropower, and habitat) as well as integrating actions beyond the boundaries of the Upper Columbia region (e.g., lower Columbia, estuary, and ocean). While the Recovery Plan includes specific actions for habitat, it acknowledges that actions in freshwater tributary habitat are not likely to achieve recovery on their own and should not be the sole focus of recovery efforts. Achieving viable salmonid populations requires recovery actions across all H-sectors.

Management of salmon and steelhead requires coordinated decision-making to achieve recovery while honoring treaty and reserved rights and meeting legal and regulatory requirements. The UCSRB developed an Integrated Recovery Program to align management and restoration actions across all entities involved in UCR salmon recovery. The Integrated Recovery Program tracks and reports information about actions implemented for recovery of salmon and steelhead across management and geographic boundaries. Habitat Reports document project results, recent outcomes to support “All-H” collaboration, and progress toward integrated recovery.

The UCSRB established the Upper Columbia Programmatic Habitat Project (UCPHP) to implement reach-based habitat restoration projects in high-priority areas in the Upper Columbia Region (Umbrella Project). The UCPHP coordinates regional actions to meet mitigation obligations under the 2008 FCRPS Biological Opinion (BiOp) with recovery needs and priorities identified in the Upper Columbia Spring Chinook Salmon and Steelhead Recovery Plan (Recovery Plan; UCSRB 2007), and the Biological Strategy (UCRTT 2013).

The Recovery Plan of the UCSRB includes a general analysis of the benefits of salmon recovery and the economic costs of recovery actions to other sectors in the region (Chapter 6 Social/Economic Considerations, p 250). The economic benefits of recovery of salmon and steelhead were quantified based on economic information and analyses from the Columbia River Basin, including the Snake River basin. Economic costs were quantified for the agricultural sector of the three basins. Although the RTT review process provides relatively limited information on cost effectiveness of specific projects, the UCSRB as a whole includes some consideration of the economic factors associated with salmon recovery.

4.4.3. Coordinating Actions

The UCSRB tries to be informed about the individual strategies related to habitat, hydrosystem, hatchery, and harvest working groups, but coordination of the different groups that have been working for more than 20 years is complex. The UCR includes a substantial number of separate coordinating committees that are independent and report to different entities. Each of the Habitat Conservation Plans (HCPs) for Chelan and Douglas PUDs has a Coordinating Committee, Hatchery Committee, and Tributary Committee. Grant County has the Priest Rapids Coordinating Committee, a Hatchery Committee, and a Habitat Committee. The RTT also coordinates habitat assessments and actions in the UCR. There is no coordinating committee for harvest actions in the UCR. Each PUD also has a separate committee to address issues related to non-native invasive species. While membership overlaps on the coordinating committees and some of the committees have joint meetings, the ISAB understands that decision-making processes are separate and reporting and coordination between the committees is not systematic.

The UCSRB meets regularly and includes participants from the habitat, hatchery, hydrosystem, and harvest sectors who review future projects and coordinate actions. The RTT also meets regularly to review information and coordinate activities. Reviews by the RTT provide some communication between habitat and hatcheries, and to some degree harvest, but there is no systematic coordination of the actions between the RTT and the PUDs. The PUDs provide important funds for hatchery and habitat because of funding obligations of the PUDs based on responsibilities for impacts related to the reservoirs, dams, and tailraces. The PUD operations are assumed to lower survival of salmon and steelhead passing through the hydrosystems of the upper Columbia River by up to 9%, 7% for juvenile mortality, and 2% for adult mortality. Licensing agreements require the PUDs to mitigate for 7% of the 9% decrease in survival and 2% is mitigated by habitat restoration actions (FERC 2002). The PUDs provide funds for the hatcheries and monitor hatchery releases to document the hatchery mitigation. The 7% mitigation obligation can be reduced by improving survival from upstream rearing areas through the dams, which is monitored annually and adjusted based on 4-yr averages. They also provide funds for habitat restoration projects in the basin, but the outcomes are not assessed and monitoring for habitat conditions and fish-habitat relationships is not specifically funded.

Hatchery managers and co-managers are involved in coordination with hydrosystem operations and are funded in large part by hydrosystem obligations. Though managers of the hydrosystem and hatcheries actively participate in the RTT, hatchery operations potentially influence the effectiveness of habitat restoration actions. Studies in the UCR have found that hatchery releases potentially reduce fitness of juvenile spring Chinook (see Chapter 5), which would reduce the effectiveness of habitat restoration. Continued management refinements to reduce effects of hatcheries on fitness of spring Chinook in the three basins must be coordinated with habitat restoration projects to improve recovery efforts.

The Recovery Plan is designed to coordinate habitat restoration actions with harvest. Both ocean and in-river harvest in the Columbia River basin are coordinated and managed through the United States v. Oregon Management Agreement under the jurisdiction of the federal court. Harvest intercepted 5 to 18% of returning spring Chinook salmon during 1977 to 2013 (NOAA Salmon Population Summary database). Washington Department of Fish and Wildlife (WDFW) recommends harvest levels for UCR stocks within this Management Agreement. WDFW reports their harvest management plans to the RTT and UCSRB, but there is no major or regular process for feedback from the RTT and UCSRB to harvest management decisions. Though the Management Agreement ultimately determines the degree to which harvest influences numbers of returning adults, collaborative discussions of the influence of adult return rates on spring Chinook recovery in the UCR and potential harvest management options would strengthen recovery efforts of the UCSRB.

4.5. Recommendations

Here we address the questions posed by the Council, after rearranging them to group several similar ones together:

- **Are habitat recovery actions being prioritized and sequenced strategically, given existing knowledge and data gaps? How should habitat projects be prioritized?**

The ISAB found the UCSRB’s system for soliciting, reviewing, and designing restoration projects to be scientifically sound with regard to habitat conditions and effects of hatcheries and the hydrosystem. Assessments of limiting factors are used to prioritize recovery actions, and the recent history of restoration is relatively consistent with the rankings of limiting factors. Current methods of prioritization (e.g., EDT, HSI, RTT Biological Strategy) are useful.

The life-cycle models that have been developed for the three UCR basins show great potential for integrating multiple factors that determine the abundance of spring Chinook throughout their life histories and across their full geographic range. The recent ISAB review of the life-cycle models (ISAB 2017-1) commended the recent progress of the life-cycle modeling effort and expansion to new areas in response to previous ISAB recommendations. The ISAB review noted that models are always a tradeoff between realism and simplicity and do not include all sources of variation. They concluded "the models can be used for ranking scenarios, but their predicted results may not be accurate." The capacity to rank scenarios and integrate multiple factors across the full life history of spring Chinook strengthens the prioritization of projects and coordination of actions across participants in the UCSRB Recovery Strategy.

The UCSRB focuses most attention on determining which kinds of actions will be biologically beneficial toward recovery of Upper Columbia spring Chinook and assesses fish abundance, productivity, and risk of extinction. The procedure for characterizing cost effectiveness is not quantitative and does not provide a rigorous basis for prioritizing actions. If funds are unlimited, there would be no need to prioritize actions, but that clearly is not the case. The criteria used by the RTT and the Citizens Advisory Committee are vague and the results are weighted, so they have little influence on project priorities. Cost effectiveness is considered by the UCSRB and the Salmon Recovery Board, but there is no explicit analysis and the evaluations are not documented.

The ISAB recommends applying a transparent cost-effectiveness analysis of each project proposed, perhaps by using the approach in the simple example we described (see section 4.1.2) as a starting point. The lack of rigorous cost effectiveness analysis and its minor influence on prioritization of restoration actions for all Hs limit the USCRB and participants in their recovery efforts. This is a common deficiency throughout the Columbia Basin, but such analyses would allow the program to use its limited resources more effectively. A transparent method for estimating the biological effectiveness of projects and their costs, including a discounting for time should be included in the Recovery Program to improve the way priorities are set.

The existing prioritization process could be strengthened by incorporating explicit analysis of performance, time, and cost in a cost-effectiveness assessment. Studies have shown that doing so can improve outcomes by an order of magnitude. However, this will require empirical or other ways of estimating biological benefits as well as estimating project costs. Effectiveness in terms of smolts per adult (i.e., freshwater productivity) is difficult to estimate, as are the effects over time, but developing a clear method would better highlight knowledge gaps. In the interim, eliciting estimates from a set of experts could provide an objective starting point for assessing relative cost effectiveness of projects.

- **Is there evidence that past projects have improved habitat for this ESU?**

The ISAB found that there is sufficient evidence that protecting habitat, removing barriers to restore connectivity, and reconnecting side channels and floodplains have positive effects on anadromous salmonids, including spring Chinook salmon. Projects at different scales within the Upper Columbia provide strong evidence that structures that increase pools and habitat complexity can increase fish production, survival, and abundance. Effects of log and boulder structures should be measured to understand effects of specific types of structures in particular watersheds.

The ISAB recommends designing rigorous experiments and continued careful monitoring to measure the effectiveness of habitat restoration practices in the Upper Columbia subbasins across a hierarchy of biological responses, including use of habitats by fish, and their abundance, growth, survival, and productivity. VSP measurements should be compared against model predictions to verify and improve modeling approaches.

- **What types of habitat projects should be prioritized in the future? Why?**

Projects that restore and sustain key fluvial and ecological processes should be prioritized, given predictions for future climate and building on the success of the projects completed so far. A key goal will be to provide habitats that are resilient to changing conditions and extreme events, and ones that provide connected habitats sufficient to support the full range of life history diversity among spring Chinook in the Upper Columbia.

Empirical data and modeling from the Upper Columbia and other locations within and beyond the Columbia River Basin support ranking habitat protection as a high priority, followed by removing barriers, and reconnecting floodplains and side channels. Increasing habitat complexity using log and boulder structures is a useful short-term approach, but a long-term strategy is needed to restore processes that maintain channel complexity and supply and retain large wood in rivers. Less information was available on projects that increase instream flows or address water quality, although these can also be effective.

- **How well are actions in other management sectors (all H's, i.e., habitat and hydrosystem, hatcheries, and harvest) aligned with recovery efforts? Specific input to inform development and refinement of the Upper Columbia's proposed prioritization framework for projects would be much appreciated.**

The ISAB encourages the UCSRB and its participants to develop a systematic, collective process for coordination of the actions, monitoring efforts, and decisions across the numerous working groups and coordinating committees in the three subbasins. The UCSRB has developed a useful process for prioritizing restoration projects and coordinating recovery actions. The regional recovery plan, limiting factors assessment, life-cycle models, and monitoring provide critical information for recovery actions. However, a continued challenge is coordinating groups in the three subbasins responsible for the four Hs. More than 16 independent coordinating committees and several other major working groups make critical decisions on recovery actions. Currently, there is no formal process for integrating their separate efforts into a coordinated action plan across the three subbasins.

Hatchery supplementation has not increased spring Chinook abundance or productivity, and genetic diversity has decreased compared to historical diversity. Coordination between habitat restoration actions and emerging information about the effects of hatchery supplementation should remain a critical component of spring Chinook salmon recovery in the Upper Columbia.

If return of adult spawners or recruitment substantially limit recovery in the Upper Columbia, then discussions of the effects of harvest on escapement between co-managers and participants in the UCSRB could improve recovery efforts. More dialogue between the RTT and harvest co-managers under U.S. v. Oregon could align habitat restoration actions with returns of adult spawners needed for recovery.

5. Research, Monitoring, Evaluation, and Validation

Questions submitted to ISAB:

Is an RME framework in place that can adequately address the questions about prioritization and coordination?

Can this RME framework provide suitable data to test and validate hypotheses, inform management decisions, and confirm that limiting factors were correctly identified and are being addressed effectively? If not, what changes need to be made to the RME Framework?

To what extent has the fitness of the Upper Columbia spring Chinook ESU been negatively or positively affected by historical and current hatchery programs in this ESU?

> To what extent have contemporary supplementation programs provided a demographic benefit to the natural populations?
>
> Is the current methodology in the PUD hatchery monitoring and evaluation program sufficient to answer the questions above?

5.1. Research, Monitoring, and Evaluation (RME) Programs

The Upper Columbia River Recovery Plan is designed to assess all Hs, and partners collectively contribute information based on a variety of research and monitoring programs. Currently, there is no RME plan that encompasses all Hs and their related working groups, and there is no process to coordinate monitoring efforts across the subbasins and address information needs. Much of the RME program is funded through the responsibilities of the PUDs under licensing agreements. As a result, it is largely focused on assessing hatchery practices and the effects of hatcheries on spring Chinook populations. While this is a critical aspect of recovery in the UCR, it does not address all actions of the recovery program. In this chapter, we review the RME program associated with hatcheries in the UCR, but we note the need for integration and coordination of RME efforts and development of future studies that address all Hs.

5.1.1. RME for Hatchery Programs in the Upper Columbia River

In the Upper Columbia, integrated and safety-net hatchery programs are being used to supplement indigenous populations in an effort to increase natural origin recruit (NOR) abundance, preserve distribution patterns, and maintain genetic differences among the region's Chinook populations. In integrated programs, NOR adults are annually introduced into hatchery broodstocks and in some instances the percentage of hatchery fish (pHOS) allowed to spawn in nature is also controlled. Similarly, NORs are used to start safety-net programs. However, once a safety-net program has become established NORs are not typically incorporated into their broodstock. Instead, F1 adults produced from supplementation hatcheries are used (Maier 2017). Safety-net fish act as a genetic and demographic reserve for their corresponding integrated hatchery program. Thus, when returns of adults originating from integrated programs are low, fish from a safety-net hatchery are used to safeguard and continue supplementation. Conversely, in years of high abundance, safety-net programs can function like segregated hatcheries with surplus fish available for harvest.

Segregated Chinook hatchery programs specifically designed to augment harvest are also being implemented in the Upper Columbia. Unlike the conservation and safety-net programs that rely on native broodstock, both within and out-of-subbasin fish are used in harvest augmentation hatcheries (Hillman et al. 2017). Hatchery programs are also being used to reintroduce extirpated salmonids into Upper Columbia subbasins. Mid-Columbia River coho, for example, are being reintroduced to the Wenatchee and Methow subbasins, and efforts to establish spring Chinook and sockeye salmon in the Okanogan subbasin are also occurring. As of 2015, there were 24 active hatchery programs in the Upper Columbia. Seven are conservation programs, two for spring Chinook, one for summer/fall Chinook, and four for steelhead with a total annual production goal of ~2.1 million fish. Four safety-net efforts accompany the conservation programs, two each for spring Chinook and steelhead. In combination, the safety-net programs release approximately 900 thousand smolts per year. The three reintroduction programs have a combined release goal of 6.7 million smolts; 5 million of those are sockeye that are being reestablished into portions of the Okanogan subbasin. Smolts produced for the harvest augmentation programs in the Upper Basin account for over half (>60%) of the ~25 million hatchery fish released each year. A little less than 2 million of these fish are spring Chinook, and the remaining 14 million are summer/fall Chinook (Table 5.1).

The use of hatcheries for conservation purposes is not without controversy. For example, a question that has received considerable scrutiny is whether the influx of hatchery fish on natural spawning grounds alters the intrinsic productivity of a natural population. Previous studies have shown that artificial culture can alter the behavior (Fleming and Gross 1992; Lura et al. 1993; Fleming et al. 2000), morphology (Petersson et al. 1996; Busack et al. 2007) and physiology (Fleming and Petersson 2001; Knudsen et al. 2006) of hatchery salmonids. These changes have been linked to reduced fitness when the fish spawn in nature (Christie et al. 2014). Additionally, in some instances, significant genetic changes, due to small founder sizes and variation in family size, have reduced genetic diversity in hatchery fish (Naish et al. 2008;

Christie et al. 2012). Genetic changes can be passed onto future generations and thus decreases in fitness of natural populations can occur. The magnitude and predicted impacts of genetic alterations as well as possible epigenetic effects due to hatchery conditions are important questions that warrant further investigation.

Table 5.1. The hatchery programs in the Upper Columbia River as of 2015 (from Maier 2017; and Hillman et al. 2017)

Species	Program	Subbasin	Program Components	Goals	Production Goal	Operator[a]
Spring Chinook	Wenatchee Spring Chinook	Wenatchee	Nason, White[b], Chiwawa	Conservation	269,026	WDFW
	Wenatchee Spring Chinook	Wenatchee	Nason	Safety-Net	98,670	WDFW
	Methow Spring Chinook	Methow	Methow, Twisp, Chewuch	Conservation	223,765	DPUD
	Winthrop Spring Chinook	Methow		Safety-Net	400,000	FWS
	Leavenworth Spring Chinook	Wenatchee		Harvest	1,200,000	FWS
	Chief Joseph Spring Chinook	Okanogan		Harvest	700,000	CCT
	Chief Joseph Spring Chinook	Okanogan		Reintroduction	200,000	CCT
Steelhead	Wenatchee Steelhead	Wenatchee		Conservation	123,650	WDFW
	Wenatchee Steelhead	Wenatchee		Safety-Net	123,650	WDFW
	Wells Steelhead	Methow, Columbia	Methow, Wells	Safety-Net	260,000	DPUD
	Twisp Steelhead	Methow	Twisp	Conservation	48,000	DPUD
	Winthrop Steelhead	Methow		Conservation	200,000	FWS
	Okanogan Steelhead	Okanogan		Conservation	100,000	WDFW
Coho	Mid-Columbia Coho	Wenatchee, Methow	Wenatchee, Methow	Reintroduction	1,500,000	YN
Summer/Fall Chinook	Priest Rapids Fall Chinook	Columbia		Harvest	7,300,000	WDFW
	Ringold Springs Fall Chinook	Columbia		Harvest	3,500,000	WDFW
	Chelan Falls Summer Chinook	Columbia		Harvest	576,000	WDFW
	Wells Summer Chinook	Columbia		Harvest	804,000	WDFW
	Wenatchee Summer Chinook	Wenatchee		Harvest	500,000	WDFW
	Entiat Summer Chinook	Entiat		Harvest	400,000	FWS
	Methow Summer Chinook	Methow		Harvest	200,000	WDFW
	Chief Joseph Summer/Fall Chinook	Columbia		Harvest	900,000	CCT
	Chief Joseph Summer/Fall Chinook	Okanogan		Conservation/Harvest	1,200,000	CCT
Sockeye	Okanogan Sockeye	Okanogan		Reintroduction/Harvest	5,000,000	ONA

[a] Operators: CCT = Confederated Tribes of the Colville Reservation; DPUD = Douglas County Public Utility District; FWS = U.S. Fish and Wildlife Service; ONA = Okanogan Nation Alliance; WDFW = Washington Department of Fish and Wildlife; YN = Yakama Nation

[b] Last release of juveniles from a White River captive brood program occurred in 2015

Alternatively, results from the Idaho Salmon Supplementation Study (ISRP 2016-9) showed that the ability of spring Chinook supplementation programs to increase NOR abundance is not only influenced by adult traits and stock origins but it is also linked to incubation sites and where juvenile fish rear and grow. When the Idaho study first began over 23 years ago, it was believed that the Salmon and Clearwater River subbasins would be capable of producing greater numbers of spring Chinook if more adult spawners were added. However, it soon became apparent that salmonid populations in these subbasins were already at or near capacity because density dependent effects were observed (Walters et al. 2013; Venditti et al. 2015). Thus, when rearing environments are fully occupied or possess deleterious environmental conditions, no increase in abundance through supplementation should be expected. Consequently, assessments of hatchery supplementation need to include provisions that will allow researchers to: (a) examine and characterize the habitats supplemented fish are occupying, (b) determine if density dependent effects are occurring, and (c) appraise the traits and performance of hatchery origin adults and their subsequent offspring

These concerns and the desire to determine the efficacy of the supplementation efforts taking place in the Upper Columbia, led fisheries managers, PUD scientists, tribal, state, federal and private researchers to develop a comprehensive monitoring and evaluation plan. When the plan was initially introduced, it possessed eight major objectives (Murdoch et al. 2011). Each objective covered an uncertainty associated with supplementation and raised one or more management questions. Answers to those questions could be used to evaluate the effects of ongoing supplementation efforts (Table 5.2). The data or metrics needed to resolve each management question were generally described along with proposed analytical methods. These general descriptions were buttressed by appendices that provided detailed information on the metrics and analytical approaches being used to answer the questions shown in Table 5.1.

5.2. Collecting Suitable Data, Testing and Validating Hypotheses

The Monitoring and Evaluation Plan developed by Murdoch et al. (2011) emphasized that the most important goals of the Upper Columbia hatchery supplementation program were to increase total spawning abundance and NOR recruitment in supplemented populations. Appendices were attached to the M&E Plan and were used to delineate how NOR abundance and recruitment rates will be assessed (Murdoch et al. 2011). Two of the appendices described the methods used to estimate abundance and recruitment rates of spring Chinook NORs in the Chiwawa and summer Chinook in the Wenatchee, Methow, and Okanogan subbasins. Numerous factors unrelated to supplementation (e.g., ocean environment, droughts, floods, and conditions in juvenile rearing areas and the freshwater migration corridor, etc.) can influence NOR abundance and recruitment rates. Use of reference populations can account for these factors. However, reference populations must not be undergoing supplementation, and both the supplemented and reference populations need to be experiencing similar environmental conditions. The proponents recognized these requirements and one of the appendices (their Appendix C) explains and provides results of the analytical and graphical methods used to choose appropriate reference populations.

Power analyses, for example, were conducted to help ascertain which reference populations should be selected for specific supplementation efforts. In these analyses statistical power (i.e., probability of rejecting a false null hypothesis was set at 0.80), Type-1 error probability (i.e., risk of rejecting a true null hypothesis of no difference was fixed at 0.05), and the number of years to include in the power analyses were either 5, 10, 15, 20, or 50. A modified Before-After-Control design that included replication before and after supplementation in reference and treated populations was used. Differences between reference and supplemented (treatment) populations were quantified by using ratio scores, which were calculated by dividing treatment values by reference population values.

Table 5.2. The objectives and management questions that were presented in the monitoring and evaluation program of Murdoch et al. (2011)

Objective	Supplementation Uncertainties	Management Questions
1	Abundance, Recruitment, and Productivity	1. Has supplementation increased the total number of spawners within the supplemented population? 2. Has supplementation increased NORs within the supplemented population? 3. Has the program increased adult productivity of NORs in the supplemented population?
2	Migration and Spawning Characteristics	1. Is the migration timing of NOR and hatchery origin recruits (HOR) fish similar? 2. Is the spawn timing of NOR and HOR females similar? 3. Is the distribution of redds similar for NORs and HORs?
3	Genetic and Phenotypic Characteristics	1. Are allele frequencies of HOR fish similar to those of NORs? 2. Do genetic distances among subpopulations within a supplemented population remain the same over time? 3. Does the ratio of effective population size (N_e) to population size (N) remain constant over time? 4. Is age-at-maturity of HORs and NORs similar within each sex? 5. Is length-at-age of HORs and NORs similar within each sex?
4	Adult Recruits/Spawner (R/S)	1. Does the hatchery recruitment rate (HRR) exceed the recruitment rate (NRR) achieved by NORs? 2. Does the HRR exceed target values?
5	Stray Rates	1. Is the stray rate of HORs less than 5% for the total brood year return? 2. Is the stray rate of HORs less than 5% of the spawning escapement within other independent populations? 3. Is the stray rate of hatchery fish less than 10% of the spawning aggregate within non-target spawning areas within the target population?
6	Characteristics of Released Hatchery Juveniles	1. Are the fork lengths and weights of released fish equal to program goals? 2. Is the number of fish released equal to the project goal?
7	Freshwater Productivity	1. Does the number of juveniles produced per redd decrease as the proportion of hatchery spawners (pHOS) increases?
8	Harvest	1. Is the harvest on hatchery fish produced from the Wells summer Chinook program great enough to manage natural spawning of hatchery fish but low enough to sustain the hatchery program? 2. Is the escapement of hatchery fish from supplementation programs (conservation and safety-net) in excess of broodstock needs to provide opportunities for terminal harvest?

Aspin-Welch t-tests or randomization tests were used to evaluate the null hypothesis that the mean ratio score before supplementation did not differ from the mean ratio score observed during supplementation. Results from these tests were employed to determine the minimal change in mean ratio scores that indicated an effect. These minimal detectable differences or MDD scores were used to identify and rank possible reference streams. Preference was given to reference streams possessing low MDD scores. This same approach was used to find appropriate reference populations that could be used to examine the effects of varying pHOS levels on freshwater productivity (question under Objective 7). The use of power analyses to indicate the amount of difference in mean productivity that would need to occur before the effects of supplementation can be detected is a laudable approach. As expected, MDD values decreased as more years were added to the comparisons. This result supports the value of long-term monitoring and evaluation programs.

We found that the methods and approaches in the RM&E Plan's Appendix C were well described and generally appropriate. A detailed appraisal containing questions and suggestions for improving the methods and approaches found in this appendix along with similar reviews of some of the other appendices in the RM&E Plan can be found in our Appendices C to H. We commend the authors for describing the statistical procedures that are being proposed for each question under all eight of the RM&E plan's objectives. It was somewhat surprising, however, that ANOVA or Kruskal-Wallis tests were being considered when examining possible differences in NOR and HOR migration timing, spawn timing, and redd distributions (Objective 2). Plainly, these tests could be used to compare mean values in these traits. However, Kolmogorov-Smirnov (K-S-test) two sample tests could be used to examine the agreement between two cumulative distributions, for example, the redd distributions across the length of a stream, migration or spawn timing over Julian dates by HOR and NOR fish. Such tests would provide more information to the researchers than comparisons of means. Similarly, we had a few concerns about how CWT recoveries and PIT tag detections were being used to estimate straying rates (see our Appendix C for further details). The use of allele frequencies and subsequent F_{ST} and factorial correspondence tests to examine genetic diversity, population structure and changes in effective population size (Ne) relative to census size (N) seem rigorous and well founded. Details of the methods used to track possible genetic changes due to supplementation were presented in Appendix F in the RM&E plan (Murdoch et al. 2011).

Two additional regional objectives (9 and 10) were added to the 2011 Monitoring and Evaluation plan (Murdoch et al. 2011) and were more fully explained in Hillman et al. (2017). Objective 9 is designed to examine the possible transfer of pathogens from hatchery programs to natural fish. Three possible pathways for disease transmission were identified: (a) horizontal transmission from hatchery effluent, (b) vertical transmission on spawning grounds from hatchery females to their offspring, and (c) horizontal transmission from hatchery fish to wild counterparts in the migration corridor. The pathogen of foremost concern is Bacterial Kidney Disease or BKD (*Renibacterium salmoninarum*). A series of tasks, including assembling fish health data on hatchery broodstock and examining possible relationships between hatchery conditions and disease profiles are being proposed to address this objective. As of yet, hypotheses, metrics, and methods that address the possible risk that hatchery fish may pose as disease vectors are still under development.

Objective 10 was instituted to examine possible effects of ecological interactions between post-release hatchery fish and valuable native fish taxa (referred to as non-target taxa of concern or NTTOC) that are not part of a hatchery program (Ham and Pearsons 2001). First, native fish taxa that may be impacted by hatchery fish need to be identified. Second, containment objectives, or levels of acceptable impact for each NTTOC need to be established. Third, data on abundance, biomass (kg/km), fish weight, length, and condition by age, etc. are needed from reference and treated areas to detect temporal changes in NTTOC populations. Typically, such information is gathered as part of a monitoring and evaluation program dedicated toward assessing hatchery impacts. If containment objectives have been exceeded, efforts are made to determine the causation and if necessary adaptive management is used to alter hatchery practices to reduce impacts.

Such a program has been in place in the Yakima River subbasin since the early 1990's. It is being used to ascertain whether hatchery spring Chinook salmon have affected rainbow trout, mountain whitefish, mountain suckers, and other resident fish species in the Yakima River and its tributaries (Pearsons and Temple 2010). Two independent procedures, an expert panel or Delphi approach and the Predation Competition Disease-Risk 1 (PCD Risk-1) computer model, were proposed for examining impacts on NTTOC by hatchery fish in the Wenatchee, Methow, and Okanogan subbasins (Pearsons et al. 2012).

The hatchery programs identified in Table 5.1 are expected to release 20 million or more salmon and steelhead annually. Five NTTOC species that are located in 25 geographically distinct regions were identified in the Upper Columbia. Containment objectives were established for each species that varied from <5% for spring Chinook, steelhead trout, and Pacific lamprey, <10% for summer Chinook and sockeye salmon, and <41% for cutthroat trout (Pearsons et al. 2012). An interagency group consisting of PUD, state, federal, and tribal scientists populated the databases on hatchery programs, NTTOC populations, and environmental attributes that were needed to run the PCD Risk-1 model (Mackey et al. 2014).

Out of a potential 526 interactions between hatchery and NTTOC fish, 202 were successfully completed. Interactions between Pacific lamprey and cutthroat trout with hatchery salmonids were not performed due to a lack of abundance information on these two NTTOC species. Additionally, interactions between hatchery fish from the Chief Joseph Hatchery and NTTOCs were not assessed (Mackey et al. 2014). Programing language interactions with a Windows Environment ostensibly prevented an additional 134 assessments from being completed. The PDC Risk_1 model has since had its computer language updated and is now being used by NOAA Fisheries to assess hatchery risk in its Biological Opinions.

None of the NTTOC-hatchery program interactions exceeded previously established containment objectives (Mackey et al. 2014). Estimated impacts to NTTOC rarely exceeded 1% (Mackey et al. 2014). This finding, along with issues associated with who should be on the expert panel terminated the planned Delphi assessment. It is important to recognize that the outputs from the PCD Risk-1 model are estimates or hypotheses of expected outcomes. It is our understanding that existing data could be used to validate some of the outcomes produced by the model. We recommend that this work be undertaken and would think results produced from such an effort would be of interest to NOAA Fisheries and others who are concerned about possible hatchery-NTTOC interactions. Ideally, field sampling similar to that being conducted in the Yakima subbasin should also be occurring in selected Upper Columbia River locations. Data from such efforts would help to further test the accuracy of the initial PCD Risk-1 modelling outputs.

5.3. Changes in the RME Process

Hatchery programs in the Upper Columbia have three general goals to: (a) increase the abundance of ESA-listed NORs (e.g., spring Chinook and steelhead) while simultaneously maintaining or increasing the spatial distribution, adult productivity, and genetic properties of those stocks; (b) enhance the number of non-listed NOR salmonids (e.g., summer Chinook) in the Upper Columbia without compromising their spatial distribution, adult productivity and genetic integrity; and (c) increase opportunities for harvest through segregated hatchery programs, and when abundance allows, via surplus adults produced from safety-net programs. Harvest augmentation programs are expected to include actions that will segregate hatchery fish destined for harvest from natural spawning populations.

The RM&E plan developed by Murdoch et al. (2011) was designed to determine if these management goals were being accomplished. The plan was modified as new information, approaches, and tools became available. A requirement was built into the RM&E plan to develop and combine these changes once every five years to refine the original monitoring and evaluation plan. The latest five-year update of the plan was released in November by Hillman et al. (2017). The objectives of the revised plan are the same as those presented by Murdoch et al. (2011). However, the original objectives have been re-ordered and are now placed into three categories: Risk, In-Hatchery, and In-Nature indicators. Risk Assessment indicators are being used to determine if hatchery operations pose risks to natural populations. In-Hatchery indicators establish whether artificial production goals are being accomplished, and In-Nature indicators evaluate hatchery fish performance after release (Maier 2017).

What is monitored and evaluated has also been tailored to the type of hatchery program being assessed. Evaluations made on conservation efforts are centered on how a program influences NOR abundance and productivity. Safety-net programs are appraised by examining the long-term fitness of supplemented populations while harvest augmentation is reviewed by determining whether the program has provided harvest opportunities. Specific metrics are associated with each type of M&E. For conservation and safety-net programs, NOR recruitment rates, abundance, and the production of juveniles per redd are the primary metrics being used. These metrics are referred to as productivity indicators by Hillman et al. (2017) who acknowledge that they can be difficult to measure and typically vary in space and time.

When productivity indicators are not available, stray rates, HOR recruitment (R/S), size-at-age data, run timing, spatial distribution patterns and smolt size are used to help evaluate conservation and safety-net programs (Hillman et al. (2017). These metrics are referred to as monitoring indicators. They are not considered as important as productivity indicators but provide additional insights into program performance. Objectives 1 and 7 of the 2011 plan provided the impetus for collecting the revised plan's productivity indicators. The remaining six original objectives spurred the collection of information used to create the current M&E plan's monitoring indicators (Table 5.3).

Table 5.3. Productivity and monitoring indicators along with objectives, and targets for Upper Columbia River hatchery programs (from Hillman et al. (2013) and Maier (2017))

Type of Hatchery Program	Objective	Indicator	Target	Program Goals: Rebuild Natural Populations	Program Goals: Maintain Genetic Diversity	Program Goals: Opportunity For Harvest
Conservation & Safety-net Productivity Indicators	Determine if the program has increased the number of naturally spawning adults	Abundance of natural-origin spawners	Increase	✓		✓
		Adult productivity (NRR)	No decrease	✓		
	Determine if the proportion of hatchery-origin fish affects freshwater productivity	Residuals vs. pHOS[a]	No relationship	✓		
		Juveniles per redd vs. pHOS[a]	No relationship	✓		
Conservation & Safety-Net Program Monitoring Indicators	Determine if run timing and distribution meets objectives	Migration timing	No difference	✓	✓	
		Spawn timing	No difference	✓	✓	
		Redd distribution	No difference	✓	✓	
	Determine if program has affected genetic diversity and population structure	Allele frequency (hatchery vs. wild)	No difference		✓	
		Genetic distance between populations	No difference		✓	
		Effective population size	Increase		✓	
		Age and size at maturity	No difference		✓	
	Determine if hatchery survival meets expectations	HRR	HRR > NRR	✓		
		HRR	HRR ≥ Goal	✓		
	Determine if stray rate of hatchery-origin fish is acceptable	Out of basin	≤ 5%	✓	✓	
		Within basin	≤ 10%	✓	✓	
	Determine if hatchery-origin fish were released at program targets	Size and number	= Target	✓		
	Provide harvest opportunities when appropriate	Harvest	Escapement goals			✓
Harvest Program Monitoring Indicators	Determine if hatchery survival meets expectations	HRR	HRR > NRR			✓
		HRR	HRR ≥ Goal			✓
	Determine if stray rate of hatchery-origin fish is acceptable	Out of basin	≤ 5%		✓	
		Within basin	≤ 10%		✓	
	Determine if hatchery-origin fish were released at program targets	Size and number	= Target			✓
	Provide harvest opportunities when appropriate	Harvest	Escapement goals			✓

[a] We recommend that these two indicators be changed to juveniles per redd. See our Appendix C for further details

Metrics or indicators (see Table 5.3) used to evaluate harvest augmentation efforts differ from those used to appraise conservation programs. The central goals of harvest programs are to bolster fishing opportunities and segregate hatchery adults from natural spawning populations. Consequently, performance indicators related to this type of program are quantification of harvest opportunities, HOR recruitment rates, and stray rates. Regardless of whether a hatchery program is based on conservation or harvest objectives both productivity and monitoring indicators will be used for evaluation purposes (Hillman et al. 2017). Moreover, the current plan is structured to allow direct assessments of management actions on monitoring and productivity indicators. For instance, management decisions about the brood sources used in hatcheries will influence monitoring indicators such as stray rates, run timing, and the genetic properties of the supplemented populations. These in turn may affect, recruitment rates of NORs, NOR abundance, and juveniles per redd which are considered productivity measures.

The overarching goal of the hatchery-related M&E program is to identify empirical associations between management decisions such as brood stock sources with productivity indicators like NOR recruitment and abundance. The plan refers to these as “chain of causation” connections. These connections between management actions and fish performance are used in the program’s formal adaptive management process. If management actions are shown to be ineffective or are preventing the program from achieving one or more of its primary goals, e.g., rebuilding natural populations, maintaining genetic diversity, or providing for harvest opportunities, alternative actions are proposed, implemented, and subsequently evaluated.

A number of management strategies and M&E approaches have ensured that the effects of the hatchery programs in the Upper Columbia are carefully appraised and adjusted when needed. One notable management strategy was to create two committees (HCP Hatchery Committee and Priest Rapids Coordinating Committee Hatchery Subcommittee) to provide oversight of hatchery operations and production goals. Annual reporting of M&E results, establishment of a five-year review process for the M&E program, the capacity to refine M&E methods in a collaborative and transparent fashion, and the production of a well thought out adaptive management process are other key management components. Additionally, the M&E program is well crafted. It incorporates reference or control populations, predetermined statistical analyses, along with an active and productive research program. In combination, these elements permit effective changes to the M&E program as well as objective assessments of the numerous Upper Columbia River hatchery programs.

5.4. Uncertainties Associated with Hatchery Fish Interactions in the Upper Columbia

5.4.1. Sufficiency of the Current M&E Program

The approaches and methods described in Murdoch et al. (2011) and Hillman et al (2017) are generally appropriate to detect changes in fitness and abundance in the spring Chinook populations undergoing supplementation. Upon review, we found areas where the methods currently being used could be improved. These suggestions are provided in our Appendices C to E. In section 5.4.3 below, several of these suggestions are briefly described.

5.4.2. Effect of Historical and Current Hatchery Programs on Fitness of the Upper Columbia Spring Chinook ESU

This question has three parts. First, what is meant by fitness? Second what factors could be used to determine whether fitness has been negatively or positively affected by historical or current hatchery programs? Finally, what do the data and analyses produced by the RM&E program and ancillary research tell us about how previous, and ongoing hatchery programs, have influenced the fitness of individuals in the three extant spring Chinook populations in the ESU? In a formal genetic sense, fitness measures the capacity of an individual to pass its genes through time. It is typically measured by the number of adult offspring an individual genotype produces relative to the contributions of other genotypes; the more offspring the greater an individual's fitness (Wilson 1975). In the above question, a broader definition of fitness is implied. One based on the overall capacity of fish spawning in the same population to be self-sustaining and thus capable of perpetuating their genes through time.

The M&E plan tracks temporal changes in abundance, productivity, spatial structure, and genetic diversity (VSP parameters) relative to control or reference populations. Examination of trends in these parameters is one way to ascertain if increases or decreases in fitness can be associated with past and existing hatchery operations. A suite of factors, however, influences VSP parameters. Environmental conditions experienced under artificial culture and genetic changes due to domestication can affect age-at-maturation, sex ratios, size at maturity, choice of spawning location, genetic diversity, and other traits. At the same time, environmental conditions experienced by HOR and NOR adults and their offspring living in nature can also alter VSP parameters. Disentangling the influences of these factors on VSP parameters has to occur before long-term effects on fitness due to exposure to hatchery conditions can be estimated. Life Cycle Models appear to be a promising way to tease out the effects caused by contemporary hatchery programs. Estimates of the potential impacts of earlier fish cultural practices are more problematic since they rely on historical accounts and speculations about possible consequences.

5.4.2.1. Possible effects of historical fish culture activities on the fitness of Upper Columbia River spring Chinook

Formerly robust populations of anadromous salmonids in the Upper Columbia had declined substantially by the late 19th Century. Deteriorating conditions caused by agricultural practices, mining, logging, the construction of impassable dams, unscreened irrigation diversions, and overharvest prompted the Washington Department of Fish and Game to construct two hatcheries, one in the Wenatchee subbasin and another in the Methow. Both were in operation by 1899. The Wenatchee hatchery was located on Chiwaukum Creek about 40 miles upstream from the Columbia River. This facility was closed in 1904 due to extreme weather conditions at the site and difficulty in obtaining spring Chinook broodstock (Wahle and Pearson 1984). To circumvent these difficulties a new hatchery was built near the town of Leavenworth, approximately 25 miles upstream from the Columbia. It began operation in 1913 and was abandoned in 1931. Collecting broodstock was difficult at this site and was probably one of the reasons the hatchery was closed (Wahle and Pearson 1984).

The first Methow hatchery was located at the mouth of the Twisp River. Coho were an important species cultured at the hatchery as approximately 12 million coho eggs were taken from 1904 -1914. In 1915, an impassible dam was constructed near the mouth of the Methow River. The hatchery was subsequently moved downstream from the dam, but the coho run was strongly impacted. Egg takes from this species declined, and no coho eggs were obtained after 1920. The hatchery continued to operate until 1931, incubating eggs and releasing steelhead, Chinook, and chum salmon fry obtained from other river sources (Wahle and Pearson 1984).

Survival rates of fish produced from these early hatcheries undoubtedly were low. Hatchery fish were typically released as fry, eggs from outside stocks were commonly imported which further reduced potential survival, and little research on improving salmon culture methods was performed during the early 20th century. Consequently, few (if any) hatchery origin adults were presumably produced and available to interbreed with NORs. However, acquisition of native broodstock to support these programs could have reduced spatial structure, genetic diversity, and overall abundance of anadromous fishes returning to the Wenatchee and Methow subbasins. Information on when and where broodstock were collected, proportion of the population that was removed, and actual number taken are needed to evaluate the full impact of broodstock mining. Blankenship et al. (2007) report that the Leavenworth state hatchery used eggs from non-native stocks (Willamette River spring Chinook and lower Columbia River hatchery fall Chinook), suggesting that problems with obtaining adequate numbers of broodstock were prevalent. Perhaps the physical challenges (e.g., freshets, weir maintenance, etc.) faced by early fish culturists reduced the number of adults they could obtain. This, and their reliance on out-of-basin stocks, would have limited the impacts of their collection efforts on native salmonids. Alternatively, local abundances could have been so low it made obtaining broodstock difficult. In the latter situation, removal of NORs could substantially alter VSP parameters.

In 1939, Grand Coulee Dam blocked anadromous fish from using ancestral spawning and rearing areas in more than 1,000 linear river miles (Fish and Hanavan 1948). To mitigate for this loss, a second phase of hatchery construction and use began under the Grand Coulee Fish Maintenance Project (GCFMP). A detailed account of the accomplishments and setbacks of the first nine years of the GCFMP can be found in Fish and Hanavan (1948). They report that natural salmon runs in the Wenatchee, Entiat, Methow, and Okanogan subbasins had been "decimated" over the previous 30 to 40 years. Efforts were underway in the 1930s to remove or place fishways on impassable dams and to install screening on major irrigation ditches. Because of these improvements and scarcity of natural fish, it was determined that Upper Columbia subbasins would be appropriate locations for translocating adult salmonids destined for natal areas upstream of the Grand Coulee Dam.

Adult salmon were captured at Rock Island Dam from 1939 – 1943. Originally, these fish were supposed to be transported to holding areas in Icicle Creek and used for broodstock at a central hatchery located at Leavenworth. Neither the Leavenworth hatchery nor any of its proposed sub-hatcheries intended for the Entiat, Methow, and Okanogan were completed in time to receive fish in 1939. Adult salmon were instead hauled and released into enclosed areas in the Wenatchee, Entiat, and Okanogan Rivers. For example, in the Wenatchee River, spring Chinook and steelhead were placed into the lower 16 miles of Nason Creek. Additionally, summer Chinook were placed into an 18-mile portion of the river located below Lake Wenatchee, and sockeye were also introduced into Lake Wenatchee. Summer Chinook and steelhead were placed into a 15-mile stretch of the Entiat River, and additional sockeye were transported to Lake Osoyoos in the Okanogan subbasin. Approximately 100 linear miles of stream were impounded and used for natural production. Routine inspections of these areas showed that fish were successfully spawning. This led the GCFMP to continue to release adults into these locations until 1944 when adults were no longer intercepted at Rock Island Dam (Table 5.4).

Table 5.4. The number and spawning success of spring and summer Chinook transplanted into the Wenatchee River from 1939-1943. Data from Fish and Hanavan (1948).

Year	Race of Chinook	Location	Number Released	Number of Carcasses Sampled	% Spawned
1939			3,957	423	77
1940			3,165	574	67
1941	Spring	Nason Creek	1,251	417	37
1942			1,014	255	51
1943			1,191	243	86
1939			3,498	1,052	83
1940			752	169	93
1941	Summer	Upper Wenatchee	446	94	60
1942			3,050	776	39
1943			386	No Data[a]	No Data[a]

[a] flooding destroyed the rack in the upper Wenatchee in 1943, no sampling occurred

By 1940, the Leavenworth Hatchery and its satellite hatcheries in the Entiat and Methow were ready to receive fish. From 1940 – 1943, adult spring and summer Chinook collected at Rock Island Dam were used as broodstock at the Leavenworth Hatchery. Additionally, from 1940 – 1947 eyed Chinook salmon eggs from the McKenzie River, Big White Salmon River, Spring Creek, and from the Wind River (Carson Hatchery) were imported into the hatchery (Fish and Hanavan 1948). Initially Chinook eggs from these various sources were also shipped from the Leavenworth Hatchery into the Entiat and Methow hatcheries for further incubation, rearing, and release. That practice stopped once the value of using locally adapted fish for broodstock was recognized. Sporadic importation of Chinook eggs from Lower River Chinook populations, however, continued at Leavenworth until 1985 (Potter 2016). Although spring Chinook have been produced at Leavenworth since 1940, sockeye were the main species raised at the hatchery. By the early 1970s the sockeye program was terminated due to disease concerns (Potter 2016). From this point on, Chinook became the principal species cultured at Leavenworth. Currently, they are being used in a segregated harvest augmentation program.

The overall effect of the Grand Coulee Fish Maintenance Project on the fitness of Upper Columbia River Chinook populations can be hypothesized if the project is regarded as a large salmonid transplant effort. The results of past attempts to transplant salmonids were summarized by Withler (1982) who observed,

> "Transplants within the Pacific salmons' normal range have been singularly unsuccessful in producing new anadromous stocks, except where natural colonization has been prevented by an obvious physical barrier. In only one of

> many transplants into watersheds where there was no physical obstruction to upstream migration has a new run developed: sockeye fry and fingerlings of Skagit River origin planted in 1937 in Issaquah Creek (Lake Sammamish) and Cedar River (Lake Washington) developed into self-perpetuating runs."

Quinn (2005) offers a possible explanation for why such actions persistently end in failure by hypothesizing that accessible habitat in the areas being colonized are already fully occupied. The source of the fish being transplanted is also critical. Return rates of adult coho from 13 paired releases of local and non-local hatchery fish showed that there was an exponential decline in recovery rates with transfer distance. The farther away a non-local stock was from its natal location, the poorer its recovery rate was (Reisenbichler 1988; Quinn 2005). Mismatches with other factors associated with local adaptation (e.g., resistance to parasites and diseases, migration and reproductive timing, energy reserves at the adult stage, egg size, size and age at maturity, adult morphology, seawater entry date, etc.) may have contributed to transplant failures.

The extensive history of past transplant failures cast doubts on whether the GCFMP successfully preserved any of the genetic heritage of the salmonid stocks located above Grand Coulee Dam. Small extant populations of salmonids existed in Upper Columbia prior to the GCFMP (Fish and Hanavan 1948). It is possible that these fish had fully seeded the poor juvenile habitat then available. Additionally, the spring and summer Chinook captured at Rock Island Dam and used as hatchery broodstock or natural spawners, originated from multiple upstream spawning areas. Fish obtained from distant upstream populations would probably fare worse than those originating from populations closer to their transplant locations (Reisenbichler 1988), potentially reducing genetic diversity substantially.

On the other hand, under some circumstances, transplants of Chinook salmon within their native range have created self-sustaining populations. In their review of previous transplant attempts, Fedorenko and Shepherd (1986) identified a number of factors that led to successful transplants. The most important factors were close ties to a hatchery program that provided annual releases of one million or more juveniles, program persistence of a decade or more, use of three or more donor populations and hybrids among the donor stocks, extensive juvenile rearing and acclimation at release locations, and the production of relatively large and healthy juveniles. Overall, the GCFMP program and its subsequent hatchery program met these requirements. But other requirements identified by Fedorenko and Shepherd may not have been met (i.e., use of geographically close donor stocks (<100 km) with similar freshwater migratory routes, presence of suitable spawning and rearing areas, adequate forage base). Currently, it is unknown if any of the genetic legacy contained in the salmonid populations native to areas above Grand Coulee Dam was retained. This question could be resolved if scale samples from adults used as broodstock at Leavenworth and its satellite hatcheries during 1940-1943 were taken and are still available. Procedures now make it possible to extract DNA from scales, and genetic signatures from these historic samples could be compared to those obtained from contemporary spring and summer Chinook salmon in the Upper Columbia.

Beginning in the 1960s, additional efforts were initiated to mitigate for habitat losses due to the construction and operation of the five mid-Columbia River PUD dams. Three hatcheries were established: Chelan, Columbia Basin, and Wells. Possible effects of the Columbia Basin and Chelan hatcheries are not considered here since Chinook are not cultured at either of these facilities. Along with hatchery construction, spawning channels for summer/fall Chinook were also built in the vicinity of Priest Rapids (1963), Rocky Reach (1961), and Wells (1967) dams. The channels were operated for just a few years before it was decided that they were not functioning as designed. A hatchery facility was subsequently built at Priest Rapids and portions of its spawning channel are now being used for fish rearing purposes. Similarly, the Turtle Rock spawning channel (Rocky Reach) was converted into a rearing facility for juvenile salmonids. The Wells spawning channel was removed during a recent renovation of the Wells Hatchery. In the late 1980s and early 1990s, the Eastbank Hatchery with five satellite facilities and the Methow Hatchery with two satellite ponds were constructed (SRT 1998-33). Potential effects of these newer hatcheries on the fitness of NOR spring Chinook in the Upper Columbia are discussed below.

5.4.2.2. Possible effects of current fish culture activities on VSP parameters (fitness) of Upper Columbia River spring Chinook

Ongoing Monitoring and Evaluation in the Upper Columbia is designed to assess how hatchery operations may affect VSP parameters. Murdoch et al. (2011) and Hillman et al. (2017) presented results on responses of VSP parameters to supplementation for Methow and Wenatchee spring Chinook populations, respectively. Besides these assessments, research in the UCR included: (a) the validity of using redd counts and carcass data to estimate population abundance (Murdoch et al. 2010), (b) the relative reproductive success of HOR and NORs spawning in nature (Williamson et al. 2010), (c) factors responsible for differences in reproductive success (Hughes and Murdoch 2017), (d) the reproductive success of hatchery-origin precocious parr (Ford et al. 2015), and (e) stray rates of HOR and NOR adults (Ford et al. 2015). These studies provided additional insights into how supplementation may be affecting VSP parameters in Upper Columbia spring Chinook.

The M&E plan (see Table 5.3) prioritizes two VSP parameters, abundance (NOR and total) and productivity. These parameters are evaluated in supplemented populations by comparing them to values and trends in reference populations. Hillman et al. (2017) emphasized that abundance and productivity are rarely measured in the field. Instead, other indicators (e.g., redd counts, fish counts at weirs, scales, tags, etc.) are used to estimate them. Those estimates often rely on assumptions (e.g., fish/redd, pre-spawning loss, marking rates, etc.), which increases their variability (Hillman et al. 2017). Consequently, if disparities are detected, they likely are substantial because the elevated variation in the parameters being compared reduces the power to detect differences. Also, as described above (5.4.2.1), spring Chinook populations in the Upper Columbia have been exposed to a long history of hatchery transplants and supplementation. Potentially, many NORs and HORs could share the same hatchery ancestors, potentially narrowing genetic differences between them. It also enhances the possibility that the origin of observed divergences is due to environmental differences between hatchery and natural environments rather than by genetic changes.

In the Methow, three spring Chinook supplementation efforts located on the Twisp, Chewuch, and Methow Rivers were evaluated by Murdoch et al. (2011). Supplementation began in 1992 and has been modified since its inception. In its current configuration, a composite broodstock of Chewuch and Methow adults, is used for the Methow and Chewuch programs. Broodstock for the Twisp program is obtained from the Twisp River. All the eggs for these programs are incubated at the Methow Hatchery. Methow juveniles are released into the Methow River while those designated for the Chewuch and Twisp are transported to their rivers for final acclimation and rearing. In the early years (1992 - 1999) of the Methow supplementation project, the National Fish Hatchery at Winthrop reared and released "Carson" stock spring Chinook as part of a segregated harvest augmentation program. In 2000, the Winthrop Hatchery switched to using local broodstock. At present, spring Chinook adults (F1) produced by the Methow supplementation program are used as broodstock at the Winthrop Hatchery as part of a safety net program.

Results of some of the comparisons performed by Murdoch et al. (2011) are summarized in Table 5.5. They show that the overall abundance of spring Chinook in the three populations did not increase relative to reference populations. At the same time, supplementation did not affect productivity. Additionally, NOR abundance in the Chewuch and Methow did not change although it decreased in the Twisp. These results differ somewhat from those obtained from an evaluation of a spring Chinook supplementation program in the Imnaha River. Like the Methow programs, NORs in the Imnaha did not increase due to supplementation. But the overall abundance of spring Chinook in the Imnaha did increase and productivity decreased. Reduced productivity in the Imnaha was linked to differences in spawning locations, spawn timing, and decreases in age- and size-at-maturation in hatchery fish (orafs.org/wp-content/uploads/2015/03/ Session-2-5-Hoffnagle.pdf). In contrast, NORs and HORs spawned at similar times and places in the Methow. Additionally, no differences in size were detected when age and sex were held constant. These similarities were likely responsible for the maintenance of productivity in the supplemented Methow populations.

In the Wenatchee subbasin, there are three spring Chinook hatchery programs. Two of these are conservation efforts, located at the Chiwawa River and nearby at Nason Creek. A segregated harvest augmentation endeavor located at the Leavenworth Hatchery is the third hatchery program. The M&E plan (Murdoch et al. 2011, Hillman et al. 2017) contains a comprehensive account of the analytical steps and procedures being used to evaluate the ecological and genetic effects of the conservation programs on NOR spring Chinook in the Wenatchee River. An evaluation of the Chiwawa supplementation program is included in the M&E plan to illustrate how the proposed analytical methods should be applied. Supplementation influences on overall adult abundance (NORs + HORs), NOR abundance, and productivity in the Chiwawa were investigated. Multiple steps were taken to select appropriate reference populations that could act as controls in the analyses. After the reference populations were selected, analyses using trends, mean differences, ratios, rates, and spawner recruit models were used in the evaluation process. Additional steps were also undertaken to account for possible density-dependent and carrying capacity effects (Hillman et al. 2017). This approach meant that multiple independent analyses were used to examine the same effects of the Chiwawa project.

The principal results of these different analyses were quite similar. There was no evidence that the supplementation program in the Chiwawa increased overall abundance or the abundance of NORs.

Table 5.5. Summary of some of the assessments made on NOR and HOR spring Chinook originating from the Methow, Chewuch, and Twisp supplementation programs. Abundance and productivity assessments are made relative to what occurred in reference streams. Other comparisons are based on observations made on fish returning to the Methow. Data from Murdoch et al. (2011)

Supplementation Program	Parameter	Finding
Twisp	Overall Abundance	Declined relative to reference populations
Chewuch	Overall Abundance	No evidence of an increase
Methow	Overall Abundance	No evidence of an increase
Twisp	NOR Abundance	No evidence of a change
Chewuch	NOR Abundance	Declined relative to reference populations
Methow	NOR Abundance	No evidence of a change
Twisp	Productivity	No evidence of a change
Chewuch	Productivity	No evidence of a change
Methow	Productivity	No evidence of change
Twisp	Redd Distribution	NORs & HORs have similar distributions
Chewuch	Redd Distribution	NORs & HORs have similar distributions
Methow	Redd Distribution	Lower downstream sites by HORs in some yrs.
Twisp	Mean Brood year Stray Rate	36% -mainly within the Methow subbasin
Chewuch	Mean Brood year Stray Rate	39% -mainly within the Methow subbasin
Methow	Mean Brood year Stray Rate	3%
Twisp	Age At Maturity	No differences found
Chewuch	Age At Maturity	NOR & HOR females no difference; HOR males younger
Methow	Age At Maturity	NOR & HOR females no difference; HOR males younger
Twisp	Size At Maturity	Within a sex: 4-yr-old HORs & NORs were similar in size
Chewuch	Size At Maturity	Within a sex: 4-yr-old HORs & NORs were similar in size
Methow	Size At Maturity	Within a sex: 4-yr-old HORs & NORs were similar in size

Twisp	Genetic Diversity	Remain distinct from other spring Chinook in Methow
Chewuch	Genetic Diversity	Decreasing over time, closely related to Methow stock
Methow	Genetic Diversity	Closely related to Winthrop and Chewuch stocks
Twisp	Effective Population Size	Ratio of N_e/N remains constant as expected
Chewuch	Effective Population Size	Ratio of N_e/N remains constant as expected
Methow	Effective Population Size	Ratio of N_e/N declined and is not related to abundance

As in the Methow, there was no evidence that productivity was affected by supplementation. A non-significant negative trend was found between increasing levels of pHOS and productivity; suggesting that HORs may not be as productive as NORs. After carrying out their exhaustive evaluations, however, the authors state, “based on these analyses, there is no strong evidence that the supplementation program has significantly benefited or harmed the natural spring Chinook population” (Hillman et al. 2017).

Before concluding that the current supplementation efforts are not appreciably influencing VSP parameters in NOR populations three other issues need to be briefly discussed. First, straying rates in some hatchery programs are quite high. For example, HORs produced by the Chewuch and Twisp projects were found to have stray rates greater than 35% (Table 5.6). It was noted, that many of these fish were recovered at the Methow Hatchery, the location where they were incubated and reared for a period of time. Few strayed into other Upper Columbia River subbasins (Murdoch et al. 2011). Nevertheless, concerns were raised about how straying and hatchery operations may be reducing genetic diversity among Methow subbasin spring Chinook stocks. For instance, the genetic diversity of the Chewuch population is decreasing and becoming similar to the Methow. This may be due to straying. Significant straying (37%) has also been observed in HORs produced by the Chiwawa program. Clearly, some degree of straying is beneficial as it may expand the spatial diversity of a population. Excessive straying however, has the potential to erode stock specific adaptations and lead to lower productivity (Ford et al. 2015).

Table 5.6 Average stray rates by brood year in spring Chinook salmon produced by Upper Columbia River hatchery programs. Percentage values equal the proportion of the fish that were not detected in target watersheds. In-basin and out-of-basin straying is included in the percentage value. Data are from the Upper Columbia Salmon Recovery Board; Hillman et al. 2016; Snow et al. 2016; Humling et al. 2016; and Cooper et al. 2017.

Spring Chinook Hatchery Program	Brood Years Evaluated	Average Brood Year Stray Rate
Chiwawa	Brood Years 2005 -2009	37%
Methow	Brood Years 2005 -2009	3%
Twisp	Brood Years 2005 -2009	36%
Chewuch	Brood Years 2005 -2009	39%
Winthrop	Brood Years 2005 -2009	36.5%
Leavenworth	Brood Years 2005 -2009	4%

Research conducted on the Wenatchee River using parentage analysis compared straying tendencies in NORs and HORs (Ford et al. 2015). Results produced by this study raised a number of important questions about the roles of habitat quality, parental origins in naturally spawning fish, and possible genetic changes caused by hatchery conditions on the proclivity to stray. More of this work should be performed to provide additional insights into the factors that induce straying and on how it may be reduced. Dittman et al. (2015) recount that laboratory and field studies indicate that embryonic imprinting on natal waters occurs in salmonids from hatching to emergence. They hypothesize that straying could be reduced or that HORs could be directed to spawning sites by exposing salmonid alevins to artificial odors or to waters from desired spawning locations while the fish are incubating in a hatchery environment. This idea capitalizes on the fact that adult salmon sequentially follow a set of stream odors to natal spawning areas, moving upstream until they reach sites with odors that were imprinted upon at the embryonic stage. In theory, HORs returning to a centralized incubation hatchery (e.g., Methow) could be guided to targeted spawning areas by using additional upstream olfactory cues. We hope that a pilot study examining this possibility can be undertaken in the future, perhaps in one of the Upper Columbia spring Chinook hatchery programs.

Second, the tendency for hatchery conditions to induce precocious maturation in spring Chinook also may affect VSP parameters in supplemented populations. In some circumstances, precocious maturation rates of hatchery males can be greater than 90% (Shearer et al. 2006; Larsen et al. 2004, 2013; Harstad et al. 2014; Don Larsen pers. comm.), an order of magnitude greater than such rates in nature (Larsen et al. 2013). Consequently, large numbers of precocious males can be released from hatchery programs. Not much is known about their capacity to reproduce in nature. Recently, Ford et al. (2015) used parentage analysis to evaluate the breeding success of hatchery origin precocious parr in the White River. Anadromous males produced about five times as many offspring per spawner as precocious parr. Nevertheless, approximately 33% of the progeny produced were apparently fathered by precocious males because of the large number of precocious males in the spawning population (Ford et al. 2015).

Possible causes (e.g., genetic, epigenetic, environmental, and genetic x environmental interactions) of early maturation in hatchery fish are not completely known (Ford et al. 2015). Ford et al. (2015) discussed some possible repercussions of early maturation. If early maturation has a genetic origin, successful reproduction could be of concern because the genotypes present in precocious parr may not be representative of those that would have existed if the fish had experienced a typical anadromous life cycle. If on the other hand, maturity is largely a response to rearing conditions, early maturing males would likely not contribute deleterious genetic changes to supplemented populations. Additionally, Ford et al. (2015) suggest:

> "If early maturity is the result of genotype x environmental interactions such that some genotypes that would not have matured early in the wild do mature early when reared in the hatchery, then successful reproduction by early maturing fish may in *fact even be necessary to avoid genetic changes due to hatchery rearing.*"

Studies to evaluate the factors responsible for early maturity in hatchery fish are needed before the conservation implications of this phenomenon can be fully understood.

Third, three assessments were performed that examined possible genetic changes caused by the supplementation programs that started in the early 1990s. Two of those efforts were focused on spring Chinook. One compared genetic samples obtained from Methow spring Chinook prior to supplementation (1992-93) to those obtained up to 2006 (Small et al. 2007). Samples came from the Winthrop National Fish Hatchery (WNFH) and from natural and hatchery fish collected in the Methow, Twisp, and Chewuch Rivers. Spring Chinook from the Twisp Hatchery and Twisp River were genetically differentiated from the Methow-Chewuch group. The Methow population was closely related to the Carson stock, the out-of-basin stock that was used at the WNFH until 2001. Additionally, the Methow and Chewuch populations became more similar over time because of a change in how broodstock was acquired for those programs. Beginning in 2001, adults obtained from the Methow and Chewuch River were aggregated to form a single broodstock. Other than this change, no significant differences between pre- and post-supplementation periods were found. Genetic diversity measures (heterozygosity and allelic richness) were unchanged and the Twisp population still remained differentiated from the Methow-Chewuch group. Temporal changes in effective population size (N_e) were not detected and appear to be stable in the Methow and Chewuch populations. However, N_e is declining in the Twisp population and is precipitously low. This decline has been slowed somewhat by supplementation. Also, Small et al. (2007) note that strays from the

Methow-Chewuch population are entering the Twisp and are likely reducing differentiation among the three populations.

A similar evaluation was undertaken on spring Chinook returning to the Chiwawa River (Blankenship et al. 2007). As with the Small et al. (2007) study, genetic samples obtained pre- and post-supplementation were analyzed and compared. Post-supplementation census size was similar to what it had been prior to supplementation. Additionally, the genetic diversity of NORs appeared to be unaffected by supplementation, hetrozygosities are high, and contemporary N_e is similar or perhaps slightly higher than it was before supplementation (Blankenship et al. 2007). There was slight but statistically significant differentiation among spring Chinook in the Chiwawa River, White River, and Nason Creek. Yet, 99.3% of the genetic variation observed was within samples, and very little could be linked to population differences (Blankenship et al. 2007). This was probably caused by large amounts of gene flow among these populations or by very recent divergence (Blankenship et al. 2007).

A final genetic assessment examined the population structure of Upper Columbia summer Chinook from the Wenatchee, Methow, and Okanogan Rivers (Kassler et al. 2011) to determine if supplementation had impacted the genetic structure of these populations. Population differentiation among these populations was not observed in any of the pre- or post-supplementation samples. This suggests that genetic homogenization or extensive gene flow among these populations occurred prior to contemporary supplementation. Pairwise F_{ST} comparisons between the summer Chinook populations and eight fall Chinook and two summer Chinook populations (Entiat and Chelan River) were also performed. F_{ST} values were less than 0.01 for summers vs. falls from Hanford Reach, Lower Yakima, Priest Rapids, and Umatilla. These low F_{ST} values probably occurred because in the 1970s and 80s fall Chinook were commonly spawned with Upper River summer Chinook. This created a single homogenized summer/fall population. There was also very little evidence of differentiation between the Chelan River and Entiat populations and summer Chinook returning to the Wenatchee, Methow, and Okanogan. No change in N_e from the 1993 to 2008 was found. Kassler et al. (2011) conclude their assessment by stating that the genetic diversity of summer Chinook in the Upper Columbia has not been altered by the ongoing supplementation program.

In summary, the spring Chinook supplementation program that began in the 1990s in the Methow has decreased genetic differentiation among the Methow, Chewuch, and Twisp populations. On the other hand, genetic diversity and effective population sizes have been largely maintained in these populations. Similarly, genetic diversity and N_e values have not been affected by supplementation in the Chiwawa River. Unlike the Methow, however, the spring Chinook populations in the Wenatchee (Chiwawa and White Rivers and Nason Creek) appear to have been homogenized prior to the current supplementation effort. Genetic analyses of Upper

River summer Chinook also indicate that summer Chinook passing over Rock Island Dam originate from one homogenized population, created by the artificial mixing of endemic summer and fall run fish. As with the other two genetic assessments, contemporary supplementation has maintained N_e and genetic diversity in Upper Columbia summer Chinook. Consequently, it appears that historical events prior to the current supplementation efforts were largely responsible for decreasing genetic diversity and homogenizing Upper Columbia spring, summer and fall Chinook.

The supposition that Upper Columbia spring Chinook have lost a substantial amount of their past genetic diversity was recently supported by genetic analyses (Johnson et al. 2018). They compared mitochondrial DNA (mtDNA) haplotype richness, haplotype and nucleotide diversity, and population differentiation among ancient Upper Columbia and Snake River Chinook to contemporary Chinook populations in the Upper Columbia and Snake subbasins. Compared to their ancestral predecessors, Chinook in both the Upper Columbia and Snake subbasins have lost genetic diversity. Chinook from the Upper Columbia experienced greater genetic losses than those returning to the Snake subbasin. Haplotype and nucleotide diversity in contemporary Upper Columbia Chinook were about 1/3rd of their ancestral values. Alternatively, present day Chinook from the Snake subbasin retained about 2/3^{rds} of their ancestral haplotype and nucleotide diversity. Haplotype richness (presence and proportion of haplotypes) in the Upper Columbia subbasin has also decreased. Loss of genetic diversity experienced by Upper Columbia Chinook theoretically reduces their capacity to respond to new selection pressures; e.g., to climate change and its associated effects on water temperatures, flow regimes, and dynamic weather events.

The results obtained by Johnson et al. (2018) came from a relatively small collection of ancient samples. They nevertheless suggest that a substantial amount of genetic diversity has been lost. Some still remains though. Five matriarchal haplotype lineages still exist in Upper Columbia Chinook salmon, and managers are aware of the benefits of using local stocks in conservation, safety-net, and harvest augmentation programs. So far, current hatchery operations have not increased overall abundance or the abundance of NORs. At the same time, productivity of these populations appears to be stable. The current M&E plan provides information for researchers and managers to use in an adaptive management program. It is hoped that ongoing management practices will maintain current levels of genetic diversity and that natural selection will be allowed to once again act as a creative agent, yielding additional genetic differentiation and population structure in Upper Columbia Chinook.

5.4.3. Recommendations

The ISAB recommends that the UCSRB consider the design of their overall RME program for all Hs in their future efforts to develop a systematic, collective process for coordination of the actions, monitoring efforts, and decisions across the numerous working groups and coordinating committees in the three subbasins (see Recommendations in Chapter 4). The RME program is funded largely through the responsibilities of the PUDs under licensing agreements. As a result, it is largely focused on assessing hatchery practices and the effects of hatcheries on spring Chinook salmon populations. While this is a critical aspect of recovery in the Upper Columbia, it does not address all actions of the recovery program. Currently, there is no RME Plan that encompasses all Hs and their related working groups, and there is no process to coordinate monitoring efforts across the subbasins and address information needs related to all Hs.

Approaches and methods of the RTT, PUDs, and regional fisheries agencies are generally appropriate and can be used to answer questions about effects of hatcheries and the hydrosystem, but analyses could be improved. Upper Columbia RME planning efforts among the PUDs, WDFW, UCSRB resulted in a thoughtful process to identify reference populations that could be utilized to help assess how hatchery supplementation efforts were influencing total spawner abundance, natural origin spawner abundance, and productivity (recruits per spawner) in supplemented streams.

In many instances in the Upper Columbia, the RME program compares a supplemented population to three or more reference populations, but the results of the individual assessments are not aggregated. A more sophisticated analytical approach that simultaneously examines data from the supplemented population and all reference populations would improve the precision of the estimates and increase the power to detect effects.

The RME efforts track temporal changes in abundance, productivity, spatial structure, and genetic diversity (VSP parameters) relative to control or reference populations. However, many factors (e.g., environmental conditions experienced under artificial culture, genetic changes due to domestication, environmental conditions experienced by adults and their offspring in natural streams) can also alter VSP parameters. Influences of these factors on VSP parameters must be disentangled before long-term effects of hatchery practices on fitness can be estimated. Life-cycle models are a promising way to evaluate the relative effects of ecological concerns and human actions to better design and prioritize information needs and potential effectiveness of restoration actions.

Assessing potential impacts of early fish cultural practices is problematic because they rely on historical accounts and speculations about possible consequences. Recent genetic studies of Columbia River Chinook indicate hatchery practices and effects of mainstem and tributary dams have reduced genetic diversity of Upper Columbia spring Chinook populations. Genetic analyses offer increasing potential to quantify the influences of past practices on the fitness of anadromous salmon and steelhead populations.

Throughout this chapter we identified analytical methods that Upper Columbia managers and researchers may wish to consider. Greater detail and more comprehensive discussions on recommended statistical approaches can be found in our Appendices C to E.

Statistical and Analytical Suggestions

1. When comparing the migration timing, spawn timing, and redd distributions of NOR and HORs spawning in nature, we recommend the use of Kolmogorov-Smirnov Two-Sample Tests (K-S test). K-S tests are used to compare two frequency distributions, e.g., the distribution of NOR and HOR redds throughout a stream basin, or timing differences by Julian dates in migration and spawning. These tests are sensitive to a variety of differences that may exist in the two distributions being compared, not just central location (means) but also to skewness, dispersion, etc. (Siegel 1956).

2. Measures of uncertainty (e.g., standard errors) need to be calculated for stray rates. The methodology for estimating stray rates for Chinook based on CWT expansions for tagging proportions and catch searched is appropriate. Yet, standard errors are needed to help put calculated rates into perspective. Currently, it is unclear if a 17% stray rate has an uncertainty of ± 1% or ± 15%.

3. The process used to select reference populations employed both statistical (e.g., presence of a long-time series in abundance and productivity estimates, harvest estimates, etc.) and biological (similar life history traits, few or no hatchery fish, similar trends in freshwater habitat, etc.) criteria. Statistical criteria have little bearing on the suitability of a reference stream. All else being equal, biological criteria should be given more weight when selecting reference populations.

4. At present, separate analyses are being used to compare treated (supplemented) populations with a single reference population. Often, treated populations are compared to three or more suitable reference populations. The results of these comparisons, however, are never aggregated. We recommend that data from a treatment population and all of its reference populations be compared simultaneously. This will lead to improved precision and also increase the power to detect differences. Such models can be fit using standard software (e.g., R). However, we recommend that a Bayesian Analysis be used. It will allow the incorporation of prior beliefs on the value of supplementation, integration of multiple performance measures, and can more easily deal with unequal variances in the different populations.

5. Differences between treatment and reference populations are quantified by using ratio scores. These were calculated by dividing values from the treatment population by those from a reference population. The authors recommend that these ratios be

analyzed using original values. However, many biological effects are multiplicative rather than additive. To account for multiplicative effects the log(T/R) should be used rather than arithmetic values.

6. The M&E Plan makes extensive use of null hypothesis statistical testing to evaluate the impact of hatchery supplementation efforts in the Upper Columbia. These tests are being used to detect both positive and adverse effects. Null hypothesis testing is not really designed to assess "no impact" hypotheses. Given a large enough sample size, it is almost certain that that some effect will be detected, but it may not be biologically relevant. To directly assess a "no impact" hypothesis, a modification to the Null Hypothesis Statistical Test framework may be useful, namely **Equivalence Testing** Refer to (en.wikipedia.org/wiki/Equivalence_test) and our Appendix C for further details.

7. The greatest disadvantage of the Null Hypothesis approach is that it does not provide a direct answer to the question—how certain is it that supplementation has an effect or not had an effect? The p-value does not serve this purpose. The p-value measures the consistency of the data with no effect, which can be easily misinterpreted. We think a Bayesian approach may be more useful. The posterior belief about a parameter provides a direct interpretation on the question of interest. For example, consider again the *productivity indicator* of abundance of natural spawners (first row of Table 1 in Hillman et al. 2017). Under the NHST framework, a p-value of 0.02 indicates that the observed data is not consistent with no effect—hardly an interpretation that can be interpreted with ease. Under a Bayesian framework, a posterior belief of 0.98 that the mean abundance has increased under supplementation is easily interpreted. For most of the analyses proposed in the M&E plan a Bayesian analysis is easily implemented.

Suggestions for Additional Research

1. Outputs from the PCD-Risk_1 Model that was used to evaluate potential interactions between hatchery fish and non-target-taxa of concern (NTTOC) in the Upper Columbia are hypotheses or expected outcomes. We recommend that existing data be used to validate some of the outcomes produced by the model. Results from such analyses would be of real interest to NOAA-Fisheries and others that are using the model to examine possible hatchery fish x NTTOC interactions in the Basin. Ideally, field sampling similar to that occurring in the Upper Yakima River could also be started in selected areas in the Upper Columbia. Data from such efforts could be used to further test the accuracy of the initial PCD-Risk_1 Model outputs.

2. Continued development and validation of the life-cycle model being developed for Wenatchee River spring Chinook is encouraged. The model is designed to evaluate the

effects of hatchery domestication, climate change, pinniped predation, ocean conditions, and freshwater habitat on the population dynamics of Wenatchee spring Chinook. At present, many of the model's parameter values are fixed with only a few being estimated by an ad hoc calibration method. Given its many assumptions and fixed parameters its current outputs can only be used qualitatively (ISAB 2017-1). The model, however, appears to have the flexibility to incorporate new data, and efforts to refine it are underway. When completed, it has the potential to be a valuable tool for management.

3. It is unknown if any of the genetic legacy from Chinook stocks native to areas of the Columbia River above the Grand Coulee Dam were retained by the Grand Coulee Fish Maintenance Project. There may be one possible way to resolve this question. If scales were collected on the spring Chinook spawned at Leavenworth from 1940 – 1943, and if they are still available, it should be possible to extract DNA from these samples. Results from these analyses could be compared to the genetic profiles that exist on spring Chinook currently in the Upper Basin.

4. Dittman et al. (2015) have proposed that hatchery fish can be directed to targeted spawning locations if they are exposed to artificial odors or to waters from desired spawning locations from hatching to yolk absorption. We encourage Upper Columbia researchers to continue working with Dittman and colleagues and implement a pilot project to test the feasibility of this approach. It has the potential to greatly reduce straying and possibly eliminate the need for expensive acclimation sites.

5. Further Investigations that explore the factors responsible for early maturation in male spring Chinook salmon are also encouraged. Ford et al. (2015) indicates that the conservation benefits and risks associated with releasing precocious parr are affected by the factors that drive early maturation in hatchery stocks. Consequently, a thorough understanding of what enhances precocious maturation in hatchery stocks would help determine if anything should be done to control the release of early maturing male parr.

6. Modeling

> **Questions submitted to ISAB:**
>
> Are the life-cycle and habitat models in development for the Upper Columbia ESU useful for informing the identification, prioritization, and evaluation of restoration actions? At what resolution scale can this guidance be applied, for example, watershed, population, or reach scale? Are there other approaches that would be useful?

6.1. Specific Comments about Life-Cycle Models

There are four life-cycle models developed for the Upper Columbia ESU (Table 6.1). All of these life-cycle models are based on a standard life-cycle formulation with different emphases at different life stages. For example, the Methow River model uses a food web at the juvenile stage; the Wenatchee model simulates each population and hatchery releases; and the ISEMP models 17 stream reaches in the Entiat, where habitat capacity in juvenile rearing habitat is modeled using NREI and spawning capacity modeled using HSI.

The resolution of the life-cycle models varies spatially and temporally (Table 6.1). The Methow River life-cycle model estimates changes in growth of the juvenile fish at a daily level, but the models for the Wenatchee and Entiat River use monthly time steps. Consequently, the impacts of very short intervention (e.g., adding nutrients at specific points in the growth of juveniles) can be assessed for the Methow River, but not the other systems. Similarly, the spatial scale is 1-km reaches for the Methow River, and so restoration actions at this fine scale can be modeled; but only hatchery operations at the individual stream or modification to the physical attributes at the site level are modelled for the Wenatchee and Entiat Rivers, respectively.

All life-cycle model parameters are typically specified by the user based on actual data collected for a particular life stage from the system being modeled or collected from other systems. None of the life-cycle models are “fit” to the entire data set (as done in the CSS models (Fish Passage Center, 2017)) and tuning or calibration is needed to match the model output to typical data seen over multiple years. Because none of these life-cycle models are fit to actual data from these systems, the results in baseline runs may match the pattern of changes over time, but suffered from noticeable bias (i.e., they typically under-predicted abundance).

Table 6.1. Summary of features of the life-cycle models developed for the Upper Columbia ESU

Feature	**Food web model**	**Wenatchee model**	**ISEMP model**	**EDT**
Chapter in life-cycle review	Chapter 6c Benjamin et al. (2017)	Chapter 9b	Chapter 9d	N/A
Species	Spring Chinook	Spring Chinook	Entiat spring Chinook	Okanogan summer steelhead and summer and fall-run Chinook
Life cycle stages considered	Eggs, Juvenile (food web), smolts, adults	Parrs, smolts, adults x major fish production area and 2 hatchery programs	Eggs, fry, parr, smolts (multiple ages), adults	Eggs, fry, parr, smolts (multiple ages), spawners, adults (several year classes)
Geographic locations	Methow River	Wenatchee River	Entiat River	Okanogan River
Type of model	Deterministic	Deterministic	Deterministic	Analytical rule-based model
Spatial and temporal resolution	1 km reach. Daily time steps.	Stream. Monthly at early stages; Yearly at adult stages	Site. Monthly at earlier stages. Yearly at adults stages.	Reach and watershed Monthly
Inputs to the model	Physical and hydraulic conditions of the stream. Structure and composition of riparian zone. Marine nutrients from adult salmon.	Hatchery Operations Spawners in each production area.	All parameters are input to the model as given values.	Environmental attributes (e.g., flow, stream corridor structure, water quality, biological community), rules for survival factors
State variables	Spawners, eggs, biomass of periphyton, organic matter, aquatic invertebrates, juvenile spring Chinook, juvenile steelhead, and other fish; number of smolts produced; number of adults in the ocean	Spawners, eggs, parr, smolts, adults (3 ocean ages) by production area and hatchery program	Spawners, eggs, parr, smolts, adults (3 oceans ages)	Spawners, eggs, parr, smolts, adults (up to 6 ocean ages)
Limiting factors that can be modified in the model.	Food availability for juveniles from food web model. Space for juveniles.	Hatchery operations. Parr capacity.	None directly, but all indirectly by modifying parameter values.	Environmental variables (see inputs above)

Feature	Food web model	Wenatchee model	ISEMP model	EDT
Restoration actions that can be modeled (examples)	Addition of nutrients via salmon analogues. Addition/Removal of predators and/or non-native species. Habitat restoration actions to modify physical aspects of the stream and/or riparian conditions such as a side channel.	Hatchery operations. Any life-cycle parameter can be changed based on other models that link restoration or hydrosystem operations to parameters of this model.	Increased habitat capacity and increased juvenile survival (no specific actions)	Addition of nutrients via salmon carcasses; alteration of water quality; habitat restoration; alterations of flow

There are many reasons why model output may match the pattern of observed changes over time but still have noticeable bias. Although the models are useful for ranking scenarios, they should not be used to predict abundances. For example, density dependence in a later life stage may not have been modelled, which would limit increases through the actual life cycle. Increases in productivity in earlier life stages (e.g., from habitat improvements) potentially would be less than projected because of limitations outside the natal habitat. To the extent that downstream effects operate approximately equally on fish affected by upstream changes, the life-cycle models may be useful only to rank restoration actions in terms of their relative effect on fish abundance. The assumption of approximately proportional effects would be tenable if upstream actions result in incremental changes to productivity, rather than dramatic changes, so that density dependence (and other effects) have approximately proportional impacts on improved productivity. As the life-cycle models improve with better calibration and inclusion of other feedbacks and nonlinearities, these relative rankings could change. A sensitivity analysis (not done for all models) could identify which restoration action has the potentially largest impact on the response variable (e.g., SAR or adult abundance).

Similarly, the models may be useful to rank the relative impact of different actions on (a limited set of) different life stages that are close together in the life cycle because different limiting factors are assumed. This can be done in two ways.

First, some limiting factors are part of each model. For example, a food web model is linked with the life-cycle model for the Methow River to investigate the impact of changes in nutrients (such as by adding salmon analogues), improving riparian conditions, or adding side channel habitat. The model predicted that adding nutrients resulted in substantial impacts and adding side channel habitat results in the most improvement in mean abundance. The Wenatchee model predicted that continued hatchery operations had the largest impact.

Second, for all models, the values of parameters that describe transitions among life stages, which are based on published values, can be modified. For example, COMPASS could be used to predict impacts of changes in hydrosystem operations on in-river survival and this revised value can then be fed into the life cycle models. This was the approach used in the Wenatchee model where changes in estuary survival or juvenile survival are made. The Wenatchee model predicted that increasing estuarine survival lead to large improvements. Improvements in these models over time will require calibration/validation against time series data.

However, comparing the benefits of increase estuary survival to benefit to improvement in habitat restoration actions on the spawning grounds may be less useful because of the many intermediate stages between the spawning grounds and estuary.

The modeled number of fish alive at the various stages in the life-cycle models may tend to overstate the benefits of actions because compensatory effects (both within a population due to density dependence and across populations sharing the same habitat) are rarely modeled past the smolt stage. Consequently, limiting conditions downstream may reduce the actual benefit from restoration actions.

In general, judging the validity and accuracy of a life cycle model by comparing its prediction with observed outcomes from actual management actions is always challenging. For example, the model may predict that a management action should result in a 20% increase in the number of spawners, but only 10% increase was seen. Did this discrepancy occur because the model is inadequate (e.g., model is too simple, key limiting factors were not included, interactions with other species were not modeled, etc.) or because of natural variation in the life cycle of the fish (e.g., marine survival was lower than predicted) even if the model is basically a reasonable representation. Evaluation of the actual benefit of actual restoration actions using the life-cycle models against observed outcomes will be difficult because not all sources of variation are accounted for (indeed, most of the models are deterministic).

The life-cycle models will be in a continual state of refinement and improvement. Focused studies (e.g., fish-in and fish-out studies on habitat improvements) provide information on the benefits of management actions over short time and small spatial scales but cannot be readily scaled up with incorporation into a model that represents larger scales. Initial life-cycle models can be used to scale up management actions to larger spatial (e.g., entire river) and temporal scales (e.g., entire life cycles) but rarely model non-linearities and feedback mechanism (e.g., density dependence) in more than one stage. At this point, the models are useful for ranking the relative benefit of management actions at the population level but may not perform well when predicting exact benefits. This is the current state of life-cycle models in the Upper Columbia. These models are also useful to identify the stages on which the life cycle is most sensitive and identify potential scenarios for improvements. A side-benefit of developing a life-cycle model also lies in determining which data are missing about the population of interest and need to be collected. The next steps in modelling are to better calibrate the model to actual life-cycle data both at fine and large temporal and spatial scales and to include more complex relationships in those stages where the model is most sensitive.

6.1.1. Food Web Model for the Methow River

A food-web model was integrated with life-cycle model for the Methow River to study the impact of physical and hydraulic conditions of the steams, the structure and composition of the riparian zone, and marine nutrient subsides from returning adult salmon on spring Chinook (Benjamin et al. 2017). This model has two linked submodels. The food web model is a mass-balance model that operates on a theoretical 1-km reach of the Methow River. The output from the food web model (biomass of juvenile Chinook produced) is linked to a simple life-cycle model that incorporates eggs, juvenile, smolts, and adult fish. Both submodels are deterministic.

The parameters that drive the life-cycle model are determined from many studies (see Table 2 of Bellmore et al. (2017)), but were not determined by fitting empirical data to the model. The current model simulated 9 adults returning and spawning in a 1-km reach and resulted in 794 smolts migrating. The spawners/km input to the model is less than ½ that observed in the Methow River (23 spawners/km). The modeled number of smolts produced/adults (88 smolts/adult) is only about 1/10 of the empirical estimates of 900 smolts/adult. Modelled mean weight of smolts is 9.8 g, which is comparable to empirical values. The model suggested multiple time periods when Chinook salmon would migrate downstream including mid-April, mid-summer, and winter when growth of juvenile fish is stalled or negative.

There are two key limiting factors that are directly modeled and of interest for restoration activities. First is the food availability predicted by the food web model. This depends on inputs of adult spawners (marine nutrients), and conditions of the stream. All of these could be affected by habitat restoration actions plus experimental actions (e.g., nutrient additions via salmon carcass analogues or improving riparian conditions, predator removal, etc.). The second limiting factor is space available for juveniles, which is determined by available wetted area of the reach divided by average length-based territory size. Restoration actions could increase the available wetted areas by direct modifications of the stream.

Bellmore et al. (2017) evaluated several scenarios. Improving riparian conditions contributed little to improvements in production; adding nutrients results in substantial (18%)

improvement; and additional side channel habitat had the biggest impact (31%) improvement. The latter results were highly dependent on the presence/absence of nonnative snails and fish.

6.1.2. Wenatchee Model

This Wenatchee River life-cycle model is a standard Leslie-matrix type model that simulates the movement of fish among five life stages (parr, smolt, and years in the ocean) but is extended to include three fish production areas (Chiwawa River, Nason Creek, and White River) and two hatchery operations (Chiwawa River and Nason Creek) (Jorgensen et al. (2017)). Once smolts are produced by hatchery programs or in each of the fish production areas, they share common hydrosystem, estuary, ocean survival, harvest, and maturation schedules (Figure 1 in Jorgensen et al. (2017)).

The parameters of the model are mostly determined from other studies (Table 5 of Jorgensen et al. (2017)). The authors attempted to calibrate their model by allowing four parameters (parr-smolt survival, survival on first year of entry to ocean, up river survival, and pre-spawn survival) to vary (Table 2 of Jorgensen et al. (2017)), but as noted in the ISAB review of the life-cycle models (ISAB 2017-1) the proposed calibration procedure may not result in optimal choices.

The life-cycle model is conceptually detailed enough to investigate the consequences of habitat improvements by simply changing relevant parameters of the model (starting on their page 27). For example,

- increasing capacity in habitats (in Table 5, the Beverton-Holt "a" parameter)
- improvements in parr-> smolt survival through the hydrosystem (multiple parameters)
- pre-spawning mortality

However, these restoration actions never appear directly in the model and the estimated impacts of these above actions must be provided, typically, as the result of other models. For example, the link from habitat restoration to parr capacity is from Bond and Nodine (unpublished report, NWFSC); improvements to in-river survival from changes to hydrosystem operations are modeled via the COMPASS models, etc. The only actions directly modeled here are hatchery operations.

Resolution is at the stream level, i.e., each population has a separate "branch" in the LCM. This can be aggregated up to the watershed level to investigate the aggregate effect. It is not possible to model impacts at the "reach" level.

Seven different scenarios were modeled: reduced hatchery production; reduced harvest; habitat improvements that increased juvenile survival; increased spill in the hydrosystem;

reductions in pinniped predation; or changes in ocean survival. The largest improvements in abundance occurred when estuarine adult survival was increased to historical levels (e.g., simulating conditions from reduced pinniped predation) and when hatchery operations continued operations at historical levels. Appendix A of Jorgensen et al. (2017) describes work in progress that may relate spawning success, egg-to-fry survival, and life history proportions as functions of habitat and weather covariates measured in the three stream areas, but this has not yet been implemented.

The biggest problem with the current model is the unorthodox fitting procedure (a non-standard calibration procedure that does NOT work as advertised) and relying on fixed parameter values (Table 5). No information is presented regarding whether the results from the LCM actually fit any available data.

Sensitivity analysis of the model may be difficult to interpret because the authors divided by the SE of beta rather than the SD of the X values and then they do a further standardization. The authors used a sensitivity analysis to identify which parts of life cycle are most sensitive. The sensitivity analysis could also identify the impact of changes in parameters as part of restoration actions, but the authors did not pursue this.

6.1.3. ISEMP Model for the Entiat River

The ISEMP model consists of a general life-cycle model with specific implementations for the Entiat spring Chinook (Saunders et al. 2017). Two other populations were also modeled (the middle fork of the John Day river and Lemhi spring Chinook), but these populations are not part of the Upper Columbia basins. The general model is flexible enough to model multiple “sites” within a stream, where a “site” is user defined (e.g., part of a stream). For example, the model for the Entiat River used 17 reaches as “sites.” The model is a standard life-stage specific Leslie-type matrix model with a Beverton-Holt spawner recruit relationship.

The input parameters are specified to the model (e.g., Table 4 of Saunders et al. (2017)), and there is no model fitting to actual data using the life cycle model. For example, in-river survival as a function of spill/flow would be obtained from the COMPASS or CSS models and then used as is. The model has the ability to allow for some stochasticity in the various parameter values, but again, these are provided as input, e.g., the standard error from a model fitting exercise done elsewhere.

The life-cycle model can be “linked” to other models. For example, the model for the Middle Fork John Day steelhead and Entiat spring Chinook links habitat capacity to changes in habitat using hydraulic and habitat models (Net-rate of energy intake (NREI) and habitat suitability indices (HSI); page 24 of Saunders et al. (2017)). The NREI model is used for estimating juvenile rearing habitat, and the HSI model is used to estimate spawning capacity for individual reaches after restoration actions are performed. These modeled results then are extrapolated to the watershed level using GRTS weights (page 24 of Saunders et al. (2017)) and relationships to covariates such as temperature, aquatic gross primary production, bank full width, etc.

The base model for the Entiat Chinook also tended to underestimate abundance (model values around 200 vs actual values near 300; Figures 10 and 11 of Saunders et al. (2017)). Results can be used to rank scenarios but not predict absolute numbers of fish.

Three different scenarios were investigated using the Entiat Chinook model broadly classified as improvements in habitat complexity (large woody debris) and improvements in juvenile survival. The NREI model was again used to model changes in capacity to habitat changes. A hypothesized 2-percentage-point increase in juvenile survival was used given that there is no model that links juvenile survival to habitat changes. The model predicted virtually no change due to habitat improvements; the model predicted a modest improvement when juvenile survival increased. Under no scenario did abundance meet the recovery objectives (Saunders et al. 2017).

See bitbucket.org/mtnahorniak/champ-isemp-life-cycle-model/wiki/Home for more info on the actual model.

6.1.4. Ecosystem Diagnosis and Treatment (EDT) Model

See Section 4.3.1 for a description of the EDT model. It is being used in the Okanogan River basin to prioritize restoration projects for summer Steelhead and summer and fall-run Chinook (Colville Federated Tribes 2013; Arterburn and Klett 2015; Arterburn 2017). In the Okanogan basin, the model is being applied to look at reach and watershed-level actions. EDT is also being used to develop a life cycle model for the Methow basin. With EDT, rules are applied to relate environmental variables (i.e., flow, stream corridor structure, water quality, and biological community) to survival factors for different life stages. Thus, unlike other life cycle models that use statistical assessments to reduce the number of environmental variables in the models, EDT includes all that have available input data. The environmental variables for EDT are also more comprehensive than with other life-cycle models by including physical and ecological variables as well as considering how relationships may vary for different life stages.

6.1.5. Recommendations

The ISAB recommends continued development of the life-cycle models, incorporation of more recent information on fish habitat relationships, and development of scenarios that more completely represent the restoration actions and factors that are likely to influence recovery. The life-cycle models should be continually refined and improved. We recommend using the life-cycle models to rank proposed restoration actions and incorporate their results in analysis of cost effectiveness.

Some restoration actions are river specific, but other actions are common across the models. It would be helpful to develop a set of scenarios for these actions (e.g., incorporate recent restoration project types, use Comparative Survival Study predicted results for in-river survival from changes in spill/flow to evaluate overall impacts of changes in river survival).

Sensitivity analyses should be performed on all models to identify which limiting factors are most important. These sensitivity analyses should use a standardized set of options. The models should be calibrated to earlier life-cycle stages. For example, the food web models should be calibrated to actual data on smolts produced rather than trying to calibrate for the entire life-cycle. This will improve confidence in the direct benefits of some restoration actions.

The next steps in modelling are to better calibrate the models to actual life-cycle data both at fine and large temporal and spatial scales and to include more complex relationships in life-cycle stages where the models are most sensitive. For example, the NREI models should be calibrated to actual data on smolts produced rather than trying to calibrate it using summary measures such as spawners produced. This will improve confidence of the direct benefits of some restoration actions (e.g., changes in physical habitat does lead to measurable increases in productivity), which are fed into the larger life-cycle model. The Lemhi model is such an example of a calibration at a finer resolution.

We recommend leveraging the experience gained in applying the EDT models in the Okanogan and Methow subbasins if the EDT models are developed for the other subbasins. The species habitat rules in the EDT model should be evaluated closely if the model is used.

Where possible, multiple models can be compared to better understand and quantify uncertainties and relationships between limiting factors and responses in the basin.

7. Appendices

Appendix A. List of Presentations to ISAB

ISAB July 20, 2017 Meeting, Wenatchee, Washington (agenda)

BACKGROUND

- Overview: The 10 Essential Understandings about Upper Columbia River Spring Chinook Salmon - Todd Pearsons and Peter Graf (Grant County PUD)
- Status and Life History of Upper Columbia River Spring Chinook - Andrew Murdoch (WDFW) and Tom Desgroseillier (USFWS)

QUESTION 1: IDENTIFICATION OF LIMITING FACTORS

- The Role of Science in Salmon Recovery - Tom Kahler (Regional Technical Team [RTT] and Douglas County PUD)

QUESTION 3: RESEARCH, MONITORING AND EVALUATION (RM&E)

- Tributary Habitat RM&E: Upper Columbia spring Chinook Salmon - Jeremy Cram (RTT)
- Chelan, Douglas and Grant County Public Utility Districts' Hatchery Monitoring, Evaluation and Research Programs - Catherine Willard (Chelan PUD), Tom Kahler (Douglas PUD), and Peter Graf (Grant PUD)
- Wenatchee Spring Chinook Relative Reproductive Success Study - Andrew Murdoch (WDFW)

QUESTION 4: MODEL DEVELOPMENT

- Methow Basin Spring Chinook Matrix Model - Greg Mackey (Douglas PUD)
- Ecosystem Diagnosis and Treatment: Okanogan and Methow habitat status and trends - John Arterburn (Confederated Tribes of the Colville Reservation)

QUESTION 2: HABITAT RECOVERY ACTION PRIORITIZATION, EFFECTIVENESS, AND ALIGNMENT WITH OTHER H'S (HYDRO, HATCHERIES, AND HARVEST)

- Habitat Restoration and Protection in the Upper Columbia: overview of processes, partners, and results - Greer Maier (UCSRB)
- SARs and Juvenile Metrics of Upper Columbia Stocks from the Comparative Survival Study (CSS) - Dan Rawding (WDFW)
- Mid-Columbia Hydroelectric Projects

- Wells Dam: Douglas PUD's Achievement and Long-term Maintenance of No Net Impact (NNI) for the Wells Hydroelectric Project Via the Wells Habitat Conservation Plan (HCP) - Tom Kahler (Douglas County PUD)
- Rocky Reach and Rock Island Dams Survival Study Results - Lance Keller (Chelan County PUD)
- Wanapum and Priest Rapids Dams - Curt Dotson (Grant County PUD)

- Leavenworth Complex Hatchery Programs - Bill Gale (USFWS)

ISAB September 15, 2017 Meeting, Portland, Oregon (agenda)

- Comparison of Upper Columbia and Snake River spring Chinook status - Mike Ford, NOAA Fisheries, ISAB Ex Officio
- Comparison of Upper Columbia spring and summer Chinook life histories and management - Andrew Murdoch, WDFW
- Use of Habitat Suitability Indices for project selection and design in the Upper Columbia - Sean Welch, BPA

ISAB October 27, 2017 Meeting, Portland, Oregon (agenda)

- Life History Diversity of Upper Columbia Spring Chinook - Tom Desgroseillier (USFWS)
- Methow River Subbasin Overview
  - Habitat limiting factors and restoration: Methow watershed ramblings – John Crandall (Methow Salmon Recovery Foundation)
  - Hatchery and wild fish numbers and analyses – Charlie Snow (WDFW)
- Losses of Methow-origin spring Chinook in Zone 6 by brood year, origin, and ocean age – Tom Kahler (Douglas County PUD)

ISAB December 8, 2017 Meeting, Portland, Oregon (agenda)

- Juvenile Spring Chinook and Operations of Rocky Reach and Rock Island Dams – Lance Keller, Chelan County PUD
- Competing tradeoffs between increasing marine mammal predation and fisheries harvest of Chinook salmon – Brandon Chasco, OSU (article)

- Survival of adult spring/summer Chinook salmon (Oncorhynchus tshawytscha) through the estuary and lower Columbia River amid a rapidly changing predator population – Michelle Wargo Rub, NOAA Fisheries
- Spring Period: *U.S. v Oregon* Chinook management – Jeromy Jording, NOAA Fisheries

Appendix B. Comparison of Ocean and Estuary Life Histories

Table B.1. Comparison of smolt-to-adult (ocean and estuary) life history traits and status of Upper Columbia (UC) spring Chinook and Snake River spring/summer Chinook

Life history trait/status	Description	UC spring Chinook	Snake R spring/summer Chinook	Source
Smolt freshwater age	Smolt age (years) based on CWT and PIT data collected from smolts in the lower estuary, river mouth, and ocean off OR and WA, 2007-2015	Yearling (age 1) migrants; identical to Snake spring/summer	Yearling (age 1) migrants; identical to UC spring	Weitkamp et al. 2015 and many others
Hatchery smolt releases	Mean number (millions of fish), % adipose fin clipped, release date, release weight (g), based on coded wire tag recovery data, 2007-2011	3.1 million, 67.6% clipped, April 24, 28.7 g; different than Snake spring	Spring/summer: 9.8/2.3 million, 92.3/93.9% clipped, April 20/April 9, 62.7g/23.0 g; SN different than UC spring	Weitkamp et al. 2015; Regional Mark Information System (RMIS; www.rmpc.org/)
Smolt Downstream migration rate	km/day, based on 2006-2011 recoveries of CWT fish in the estuary (75th percentile of rate)	24.6 km/day; similar to Snake spring/summer	25.4 km/day; similar to UC spring	Fisher et al. 2014
Smolt downstream migration rate to lower estuary/mouth	Based on CWT data (n=65 Upper Columbia R spring, n = 58 Snake R spring, 2006-2011	42.1 days, 830 km, 21.6 km/day; similar to Snake spring	51.1/63.3 days, 853/1100 km, 18.1/19.0 km/day; Snake spring similar to UC spring	Weitkamp et al. 2015 (see also Fisher et al. 2014)

Life history trait/status	Description	UC spring Chinook	Snake R spring/summer Chinook	Source
Smolt use of shallow tidal freshwater & estuary habitat	GSI (microsatellite DNA) of juvenile Chinook in shallow water tidal lower Columbia River and estuary, January 2002-September 2007	Not detected (Roegner et al. 2014); Not included in Roegner et al. (2012) analysis because of low (<0.9%) probability of correct GSI assignment; similar to Snake spring/summer	<1% of yearling sample (n=36), total sample n=2,174 yearling, fry, fingerling; Roegner et al. 2012) in shallow water habitats in tidal lower river and estuary; similar to UC spring	Roegner et al. 2012, Roegner and Teel 2014
Smolt use of deep water (channel) estuary habitat	Based on genetic stock id (microsatellite DNA (>0.8 probability of correct GSI assignment), Teel et al. 2015) and CWT	Present in deep water (channel) habitat; similar to Snake spring/summer	Present in deep water (channel) habitat; similar to UC spring	Weitkamp et al. 2015
Smolt size and timing in estuary	Mean size and date at capture in lower estuary, UCR timing data include both Mid and Upper Columbia stocks, 2006-2011	Smaller and earlier than Snake R, but high within stock variation	Larger and later than UC spring, but high within stock variation	Weitkamp et al. 2015
Smolt survival in estuary	Estuary survival (S.oa); see additional discussion in 3.1.4	not estimated by CSS	Estimated by CSS	CSS 2017

Life history trait/status	Description	UC spring Chinook	Snake R spring/summer Chinook	Source
Ocean dispersal pattern of juveniles during March-Nov	Based on coded-wire tag recovery data, 1995-2006	Rapid northward migration during 1st 4 mos, by late summer not found south of Vancouver I., B.C., rarely caught on continental shelf in fall; assumed to be identical to Snake/summer	Rapid northward migration during 1st 4 mos, by late summer not found south of Vancouver I., B.C., rarely caught on continental shelf in fall; assumed to be identical to UC spring	Fisher et al. 2014; Weitkamp et al. 2015
Mean ocean distance traveled by juveniles (km) during March-Nov	Mean km, based on coded-wire tag recovery data for juveniles in their 1st ocean spring-fall, 2006-2011	201.9 km; less than Snake spring/summer	395.2 km; more than UC spring	Fisher et al. 2014
Catch of sub-adults and adults in coastal ocean fisheries	Numbers of fish based on coastal recoveries of coded wire tagged fish, 1979-2004	Rare, suggesting a mainly offshore ocean existence; identical to assumption for Snake spring/summer after the first few months at sea	Rare, suggesting a mainly offshore ocean existence; identical to assumption for UC spring after the first few months at sea	Weitkamp et al. 2010; Sharma and Quinn 2012; Fisher et al. 2014; Pacific States Marine Fisheries Commission, Regional Mark Information System (RMIS; www.rmpc.org) CTC 2017

Life history trait/status	Description	UC spring Chinook	Snake R spring/summer Chinook	Source
Ocean survival	1st ocean year (S.o1); a constant survival rate for subadults in subsequent ocean years is assumed; estimated smolt-to-adult (SAR) available for both ESUs, 2008-2014; , see additional discussion in 3.1.4 and 3.2.1.4.	(S.o1) not calculated; estimated SARs of both ESUs similar, 2008-2014	Estimated SARs of both ESUs similar, 2008-2014	CSS 2017
Adult run timing	Arrival at Bonneville Dam (average median ordinal day) based on genetic stock identification (GSI) and PIT tag data	GSI = May 7; PIT = April 22-May 6, categorized as early run timing along with two Snake River spring Chinook groups: Rapid R.-Clearwater R. and Lower Snake	Rapid R.-Clearwater R. (GSI: May 7, PIT: April 21-30); Lower Snake (GSI: May 10; PIT: April 23); MF Salmon (GSI: May 30; PIT: May 3), SF Salmon (GSI: June 3, PIT: May 28), Upper Salmon (GSI: June 3; PIT: May 22)	Hess et al. 2014 (GSI based on 2004-2007 data; PIT based on 1996-2001 data from Keefer et al. (2004)
Age at return	Chinook that passed Bonneville Dam	3, 4, and 5 yr olds (4 yr olds are dominant age group); similar to Snake spring/summer	3, 4, and 5 yr olds (4 yr olds are dominant age group); similar to UC spring	Hess et al. 2014 (GSI based on 2004-2007 data; PIT based on 1996-2001 data from Keefer et al. (2004)

Life history trait/status	Description	UC spring Chinook	Snake R spring/summer Chinook	Source
Adult size at return	Average fork length (cm) at Bonneville Dam	73.2 cm; similar to Snake spring Chinook	Rapid R Clearwater R (71.3 cm); MF Salmon (74.2 cm), SF Salmon (74.1 cm), Upper Salmon (75.7 cm); Similar to UC spring Chinook	Hess et al. 2014 (GSI based on 2004-2007 data; PIT based on 1996-2001 data from Keefer et al. (2004)
Adult returns (% wild returns)	Average number returning to Columbia River mouth	1980s-20,343 adults (38% wild); 1990s-9,501 (20% wild), 2000s-21,712 (10% wild); decreasing trend similar to Snake spring/summer	1980s-39,849 (48% wild), 1990s-29,904 (38% wild), 2000s-110,827 (27% wild), decreasing trend similar to UC spring Chinook	ODFW/WDFW 2017
% of aggregate adult upriver spring Chinook run		15% since 1980, 11% of recent 10-yr avg; decreasing percentage different than Snake spring/summer	48% since 1980; 53% recent 10-yr average; increasing percentage different the UC spring	ODFW/WDFW 2017

Appendix C. Review of Appendix C in Murdoch et al. (2011)

Appendix C of Murdoch et al. (2011) presents methods for comparing abundance and productivity (the performance measure) between reference and supplemental populations.[8] There are many potential reference populations and there is concern that the set of potential reference populations differ in their "closeness" to the supplemental population on the performance measure during the period before supplementation is started.

A measure of the effect of supplementation is computed for each pair of reference vs. supplemented stream. Individual estimates of effect size are obtained for each pair of supplemental vs. reference stream.

This review examines each section in the Appendix C of Murdoch et al. (2011) and makes some recommendations to improve the document.

C.1. Selecting Reference Populations

The authors are commended in their careful search for reference streams using a number of criteria listed on page 199 of Appendix C of Murdoch et al. (2011). Some of the criteria are biologically based (e.g., similar life history) while other criteria are statistical (e.g., accurate abundance estimates or a long time series). The statistical criteria have little bearing on the suitability of a reference stream; the impacts of these criteria are on the precision of the estimates. For example, if there is substantial measurement error in estimates of abundance or if the time series is short, estimates of the supplementation effect are still unbiased, but the standard error of the supplementation effect may be so large that effects of supplementation have low power to be detected (refer to Appendix E). With suitable methodology, there is some benefit to including reference streams in the analysis that have substantial measurement error or short time series. All else being equal, the biological criteria should be given more weight in selecting the reference streams than the statistical criteria.

The appendix uses several measures of "closeness":

1. Correlation in performance measure (e.g., abundance) during the period before supplementation occurs
2. Graphical comparison of trends over time
3. Comparing the slopes of any trends over time
4. Comparing the minimal detectable difference

[8] The material in Appendix C of Murdoch et al. (2011) was essentially duplicated in Appendix 6 of Hillman et al. (2017).

The graphical methods compare population (potential reference vs. supplementation stream) on three measures (abundance, natural origin recruits, and adult productivity). The comparison was done on both the actual scale and natural logarithm scale. A priori, a logarithmic scale would be preferred. Abundance values vary from year-to-year based on year-specific effects (see Appendix E). In many biological systems, these effects are multiplicative, i.e., in a good year the number of spawners may increase from 100 to 150 in one stream and from 50 to 75 in another stream. While the arithmetic difference is not equal (differences of 50 vs. 25), both streams experience a 50% increase in abundance or a change of log(1.5)=0.4 on the logarithmic scale resulting in more "parallelism" in their responses over time as noted by the authors on page 202 of Murdoch et al. (2011).

Next, a correlational analysis was used to evaluate reference streams vs. the supplementation stream. The authors are too pessimistic about the need for a high correlation between the performance measure in reference and supplemental streams prior to the period of supplementation. In the case of BACI designs, the purpose of the reference stream is to simply show the change in the mean response over time between the before and after periods. This difference is compared to the change in the MEAN response in the supplemental stream between the before and after periods (see Appendix E). If this differential change in the MEAN (the BACI effect) is not zero, then there is evidence that supplementation has an effect.

Consequently, the correlation in actual responses between the two streams could be zero and a valid estimate of the BACI effect can still be obtained. There is no statistical reason why a correlation of at least 0.6 is needed (page 209 of Murdoch et al. 2011). However, the PRECISION (the standard error) of the BACI estimate is affected by the correlation in responses between the streams during the before period, which is largely driven by any year-specific effects that are common to both populations.

Consequently, it is NOT necessary to select reference populations that have a high correlation with supplemental streams as long as the MEAN performance measure is estimated well in both periods (before vs. after).

The standard BACI design assumes that the performance measure is in steady state prior to and after the supplementation starts (see Appendix E). The authors (page 209 of Murdoch et al. 2011) computed the trend in all streams and examined if the trend could be distinguished from the trend observed in the supplementation stream. The standard statistical technique to do this is Analysis of Covariance (ANCOVA), but it is unclear from the text if the authors actually did this. The authors did a trend analysis on both the original and logarithmic scale. Again, *a priori*, the logarithmic scale should be the preferred scale. It turns out that a BACI analysis is still valid in the presence of parallel trends if the effect of supplementation is a shift in the intercepts without any changes in the slopes. If the effect of supplementation is a change in the slopes, then a BACI analysis (or the equivalent paired difference approach used by the authors) will need to be modified.

Too often studies are conducted without a proper investigation of the power of a particular design. The power analysis conducted by the authors (page 212 of Murdoch et al. 2011) is appropriate based on the empirical estimates of variation computed by pairing of the supplemental stream with each of the reference streams. The power analysis of the ratio would be more appropriate when computed on the log(scale), i.e., log(T/R) as this then corresponds to the paired-difference approach on the log(performance measures). The resulting minimal detectable difference can be re-expressed on the original scale by taking the anti-logarithms which would be more easily understood. For example, if the minimal detectable difference of log(T/R) is 0.60, this implies that a multiplicative change of exp(0.60)=1.8x can be detected.

Finally, the authors develop a procedure to score which reference stream is “best” based on 5 criteria (top of page 220 of Murdoch et al. 2011). The criteria can again be divided into biological criteria (pre pNOS and post pNOS) and statistical criteria (correlation, relative difference in slopes, and a coefficient of variation). Each criterion is considered to be independent in developing a metric (Table 11 of Murdoch et al. 2011). Unfortunately, the three statistical criteria are all highly correlated and some double counting is taking place. For example, if the response measure is highly correlated between the two streams, then the trends will be similar and the CV value also higher. The statistical criteria also influence the precision of the BACI estimate and have no impact on the bias of the BACI estimate. The statistical criteria should be given little weight in deciding on an appropriate reference stream.

C.2. Analysis Methods

C.2.1. Analysis of Trends

The authors compared the slopes within each supplemental-reference pair for each period (before and after) using what appears to be Analysis of Covariance (ANCOVA), but this technique was never explicitly mentioned. Unfortunately, the analysis did not directly examine if supplementation had a positive effect – rather the authors hoped that a parallel trend in the before period would be replaced by a non-parallel trend in the after supplementation period.

A direct examination of the relevant hypothesis is possible using a variant of ANCOVA combining data from both periods and both streams using a similar approach as a BACI analysis. A model of the form:

$$Y = Site * Period + Year * Site * Period$$

could be fit. Here the term Site:Period corresponds to the 4 intercepts for the 4 different trends (2 sites x 2 periods) and the term Year*Site*Period would directly test (and estimate) if the CHANGE in slope for the supplemental stream between the before and after periods is the same as the CHANGE in slope for the reference stream between the before and after periods.

The author's and the above analysis allowed the intercept for the trend lines to vary between periods leading to disconnected lines see in Figure 8 (and others in Murdoch et al. 2011) when supplementation starts. It is also possible to modify the above analysis to allow for a broken stick trend line where the trend lines for a site in the before and after periods must join at the start of supplementation. This model should also be explored.

All of the analyses can be done on the original scale or the logarithmic scale, but as noted earlier, the logarithmic scale is preferred.

C.2.2. Analysis of BACI Data

The authors used a paired-difference approach where the difference, ratio, or difference annual change was computed for each year, followed by a t-test of the difference/ratio comparing the mean difference/ratio/change before and after supplementation started. This is equivalent to a BACI analysis using a mixed linear model (see Appendix E.1).

It is not necessary to test if variances are equal (see p.231 of Murdoch et al. 2011) in the before vs. after periods as the Aspin-Welch version of the t-test performs well even if variances are unequal. The tests for normality may also be superfluous because the sample size is reasonable in both periods, so the central limit theorem implies that the results are robust to non-normality.

It is puzzling that the authors were unable to perform a one-sided randomization test, as randomization tests, per se, are not restricted to one-sided hypotheses?

We agree with the authors (p. 232 of Murdoch et al. 2011) that no adjustment is needed for autocorrelation because of the small number of years measured.

Separate analyses were done on the three performance measures on both the original and log-transformed values using differences/ ratios/ changes. As noted earlier, the analyses on the log-scale will be preferred. Results from such analyses (e.g., Table 15 of Murdoch et al. 2011) should be converted to the multiplicative change (e.g., an effect size of .701 on the log-scale corresponds to an exp(.701)=2x change on the original scale.

C.2.3. Corrections for Density Dependence

Two different corrections for density dependence were employed.

In the first method, the recruits/spawner were adjusted by the carrying capacity. This method will only have an impact when the number of spawners exceeds capacity.

In the second method, a fraction of carrying capacity was computed. Because all values of NOR for a stream are divided by the same constant, the results from this analysis will be identical when a log-transform is used to the analysis of log(NOR) seen earlier because log(NOR/K)=log(NOR) – log(K) and log(K) is constant for all years. This was noted by the authors in a footnote on page 251 pf Murdoch et al. (2011). Analyses using this correction therefore provide no new information.

Both of these methods require an estimate of capacity. This was estimated using a Ricker, Beverton-Holt, and a hockey-stick model. The authors use a non-linear regression method to fit these models and compare models using information theoretic methods (AICc). All models had virtually the same AICc values indicating that there was little to distinguish among them, but the authors decided to use only the results from the hockey-stick model. A more natural approach would be to use model averaging to average the estimated capacity from all three models.

The analysis of the corrected-for-density dependence measures uses the same methodology as other measures and similar comments apply.

C.2.4. Comparing Stock Recruitment Curves

The authors used a randomization approach to examine if there was evidence that the stock-recruitment curves differed between supplemented and reference streams when each pair of streams was examined.

The authors tested two hypotheses – first that the model parameters are indistinguishable between streams and second that the fitted curve differed somewhere along the curve. The randomization procedure used an ad hoc discrepancy measure.

It is no more difficult to actually fit models with both parameters in common, only one parameter in common, or no parameters in common and use AICc to rank these models to determine the degree of support for a common stock-recruitment curve. This approach would be preferred to that taken by the authors.

The second hypothesis is redundant – if the underlying parameters cannot be distinguished between the two populations, then the results curves cannot be distinguished. The second hypothesis can be dropped.

C.2.5 Analyses without Reference Populations

The authors discuss several approaches to look at changes between the before and after period without a reference stream.

First, a before vs. after comparison of the slopes in each period using a t-test. This is a variant of ANCOVA (despite this method not being mentioned by the authors). As well, models where the trend lines match at the start of supplementation should also be considered.

Second is a before vs. after comparison of the means using a two-sample t-test. This is standard.

Third is a comparison of stock-recruitment curves. The authors used an ad hoc randomization approach rather than an AICc directed approach as noted earlier.

Fourth is a correlational approach looking at the relationship between pHOS and productivity using a standard correlational analysis. This is standard but likely has low power to detect effects.

Lastly is a comparison to standards. No analysis is presented for this approach.

The key problem with before vs. after studies is the effects of periods are completely confounded with temporal changes in any other variables between the two periods. Consequently, these methods should seldom be used.

C.2.6. Review of Conclusions in Appendix C

The authors did painstaking work to identify appropriate reference streams based on biological and statistical criteria. More weight should be given to the biological criteria and less to the statistical criteria. The statistical criteria affect the precision of the estimates, but not the bias of the estimate of the BACI effect.

The authors recommended analyses based on paired differences on the original scale (T-R) or ratios (T/R). We disagree with the last recommendation. Many effects in biological populations are multiplicative rather than additive, and log(T/R) or log(T)-log(R) would be more appropriate, i.e., analyze the log(response measure).

The first corrections for density dependence seem sensible, but the second has no impact if the analysis is done on the logarithmic scale because the new response measure is a simple scaling of the original data. The second method is akin to analyzing the data in '000s of fish vs. an analysis using the actual number of fish. Both methods rely on estimates of carrying capacity. The AICc criterion indicated that several models seem to fit the data equally well – rather than selecting the carrying capacity from a single model, a model-averaged values should be used.

The comparison of stock-recruitment curves among pairs of populations was performed using a randomization test using an ad-hoc measure of discrepancy. AICc methods should be used to fit and rank 4 models that differ in the number of common parameters between the two stocks.

C.3. Summary and Recommendations

For the most part, the analyses are suitable and the conclusions drawn appropriate. However, the report can be improved in the following ways:

1. Give more weight to biological criteria and less weight to statistical criteria to selecting reference streams. Reference streams are still valuable if they match on the biological criteria but not the statistical criteria.

2. Many biological systems have effects that operate multiplicatively. Consequently, analysis of the log(performance) measure are often most suitable. In the paired difference analysis, this is equivalent to an analysis on the log(T/R).

3. Some of the statistical methods used to estimate carrying capacity and compare stock-recruitment curves need revision

4. Some of the statistical methods to compare trend lines between before and after periods need revision.

5. Rather than presenting estimates of the supplementation effects in tables, a graphical presentation should be created showing the estimate and measure of precision to make the results more readable.

6. There are separate analyzes in this paper for each pair of supplemented stream and reference stream, but the results are never aggregated together. A more sophisticated analysis that used the combined set of the supplemental stream and all reference streams, would lead to estimates with improved precision and have increased power to detect effects (see Appendix E). In most cases, these models can be fit using standard software (e.g., R), but there are some advantages to using Bayesian methods. For example, a Bayesian analysis could incorporate prior belief on the benefit of supplementation; integrate several performance measures; and more easily deal with unequal variances over time in the different streams. Furthermore, the output from a Bayesian analysis would provide a degree of belief rather than a simple yes/no from standard hypothesis testing.

Appendix D. Additional Editorial Comments on Murdoch et al. (2011)

These additional comments do not relate directly to the charge to the ISAB in Section V (RME) but are provided for completeness.

D.1. Editorial Comments on Appendix A of Murdoch et al. (2011)

The authors first develop a method to estimate the number of NOR in a multistep process.

(1) Total number of spawners returning is estimated in a multistep process from redd counts. Redd counts from a single sample are adjusted to account for detection efficiency from a single sample. Each redd accounts for a single female. An expansion factor is used to adjust for the male:female adults ratio and then for the jacks:total population ratio (see comment below on page 166).

(2) The total escapement is then multiplied by the sex ratio seen in broodstock and carcass surveys to estimate the number of wild fish. Here some problems could occur because the total escapement includes jacks, but because jacks are smaller, they may be underrepresented in carcass or excluded in broodstock surveys. The first problem is noted on the bottom of page 169. There are also some editorial problems as noted below where the numbers in the text for examples don't match the Tables.

(3) The estimated wild fish from (2) are then multiplied by the age distribution to get the number of wild fish by brood year x age. See comments below on p. 169 of Murdoch et al. (2011)

(4) The brood year x age values are summed over all ages for brood year to get the number of wild fish returning for a particular brood year.

(5) The values in (4) are adjusted for harvest using either a hatchery indicator stock or a harvest report from the Joint Staff reports. It is also adjusted for handling mortality (recreational, commercial, or non-selective harvests) as well.

The NRR is computed by natural returns for a brood year / total escapement that produced these returns.

The methods used seem reasonable. The key problems are making sure that the various expansion factors are sensible and correct as used. There are many places where biases can be introduced such as non-random sampling (harvest, carcass searches). There is no easy way to estimate the potential size of biases that may occur, but this may be moot given the extreme drop in the stocks seen in Figure 1 of Appendix A of Murdoch et al. (2011).

There are quite a few places where the numbers in the text don't agree with the numbers in the tables as shown below so some editorial changes are needed.

p.166. The expansion factor for jacks isn't quite correct. The correct expansion factor is 1/(1-p) where *p* is the proportion of jacks. For example, if jacks are 50% of the run, then the number of adults must be doubled to account for jacks + adults. If jacks are 14% of the run, then correct expansion factor is 1/(1-.14)=1.16 not 1.14.

p.166. It is not necessary to make the assumption that male mate with only one female.

p.167. Something doesn't add up. They claim the proportion of wild fish is 0.24, yet Table 2 of Appendix A of Murdoch et al. (2011) has a value of 0.21. They also claim a wild return of 589, but 0.24 x 1725 = 414. The difference isn't the wild fish retained in the brood stock – Table 2 of Appendix A of Murdoch et al. (2011) has this as 115 fish.

p.169. Values used in brood year example don't match Table 3 of Appendix A of Murdoch et al. (2011). For example the text has a proportion of 0.020 for age-3 fish in 2004, but Table 3 of Appendix A of Murdoch et al. (2011) has 0.060.

D.2. Editorial Comments on Appendix B of Murdoch et al. (2011)

Stray rates for steelhead and sockeye are based on location of last detection of fish that have PIT-tags. It wasn't clear if all streams in the system have PIT-tag arrays to detect fish – if not, then stray rates may be underestimated. For example, if 10% of fish go to a stream without a PIT-tag array, then these fish are "invisible" and are not counted as strays. See for example, p.187 of Murdoch et al. (2011) where the authors comment on new PIT-tag array sites being established over time. The methodology seems ok.

Stray rates for Chinook are based on CWT expansions for tagging proportions and catch searched. Methodology seems OK.

However, statements such as found on page 192 of Murdoch et al. (2011) that the stray rate is more than 5% based on these raw estimates are unfounded because no measures of uncertainty were presented. For example, there is no standard error presented for any estimate of stray rates so it is unclear if the 17% stray rate estimate has an uncertainty of ±1 percentage point or ±15 percentage points.

The authors then looked at what fraction of given escapement are strays. This was not done for PIT-tagged studies as the fraction tagged is unknown. This was done using CWT for Chinook**.** **Again, no estimates of uncertainty are produced so it is difficult to evaluate if the average proportion of strays meets targeted values.**

There is a comment on the bottom of page 194 of Murdoch et al. (2011) about changes in survey effort being confounded with estimates of stray rates. The Tables would be strengthened if some estimates of survey effort were added to see when the survey effort expanded dramatically.

D.3. Editorial Comments on Appendix C of Murdoch et al. (2011)

p.3. As noted in the review of Appendix C of Murdoch et al. (2011), it is not necessary that reference streams “track” supplemental streams with a high correlation. The choice of reference streams should be based more on biological criteria than statistical criteria.

p.3. As noted in the review of Appendix C of Murdoch et al. (2011), we suggest they use the log(T/R) rather than just T/R. There are two reasons. First, many effects operate multiplicatively, and so analyses on the log() scale are preferred. Secondly, the analysis of T/R could give quite different results than the analysis of R/T (e.g., 2/1 is quite different than 1/2 relative to no effect of a ratio of 1). However log(2)= -log(1/2) so the analysis of log(T/R) and log(R/T) is symmetric and only the sign changes.

So all of the hypotheses listed on pages 3 of Appendix C of Murdoch et al. (2011) onwards about mean ratio scores would be better done using mean log(ratio) scores etc.

p.4. Assumption of normality is not important given the relatively large number of years of data available.

p. 5. It is not clear how they intend to compare the distribution of redd locations. If they want to measure the mean river location that is not really a test of distribution of the redd locations. It is not clear here what statistical method will be used for hypothesis 2.3. Figure 7 of Appendix C of Murdoch et al. (2011) indicates that a simple test of the MEAN river location will be used so the hypothesis needs to be rewritten in terms of the mean river mile of redds.

p.6. Hypothesis 3.2 appears to be a way to test for no trend. Similarly, Hypothesis 3.3 is a way to state no trend.

p.6. Hypothesis 3.4 should likely test if the MEAN ages of maturity are equal. Looking ahead at page 28, this is what is done, i.e., testing if the mean age changed. There is no reason why a Kruskall-Wallis test (see page 28) needs to be used for this comparison as a regular two-way ANOVA as used for mean Julian date of return or spawning or redd locations will work just fine.

p.10. A hypothesis-testing framework seem an odd way to determine if management goals are being met. Is the management goal stated in terms of long-term averages or is there a specific target that must be met? It seems to us that this is a yes/no answer without the need for a formal hypothesis testing framework.

p.19. All four reference streams should be used together in a single analysis as outlined in our review of Appendix C rather than this ad hoc weighting scheme. This would give an overall value for Table 6 rather than each individual supplementation vs individual reference stream result. You also have improved power to detect effects – for example, none of the individual comparisons for productivity showed evidence of an effect, but a combined analysis may show evidence of an effect.

p.28. No reason why a Kruskall-Wallis test should be used. Use a two-way ANOVA in a similar fashion to test if the mean Julian date, or mean redd location differed between groups.

p.29. Graphs are labeled as "mean total age." Likely should be just "mean age."

p.31. See earlier comments about comparing management objectives through hypothesis testing. The proportion of time management objectives are not met is important. Each of the values in Table 8 (the HRR and NRR) needs a measure of uncertainty to compare against management objectives of 4.

Similar comments as above about other population analyses. Some specific comments are:

p. 52. Figure 27. What happened to the redd distribution in 2010?

p. 57. Figure 33. Need SE on the bars.

p. 58. The authors concluded that a small sample size can cause an effect to be detected. This is unlikely to be true.

Appendix E. Review of BACI Analysis

This appendix reviews the details of a BACI analysis to understand what assumptions are being made and the impact of violations of these assumptions.

E.1. One Supplemented and One Reference Stream

Consider first a BACI design with one stream receiving supplementation and the other stream serving as reference with both streams measured for 10 years before and 10 years after supplementation is started (in year 11). For simplicity, assume that the mean number of spawners on the spawning grounds does not change between the before and after periods for the control stream, but increases for the supplemental stream.[9] All analysis takes place on the logarithmic scale so that the effect of augmentation is a multiplicative effect rather than an additive effect.

Suppose that the TRUE mean log(number of spawners) in both streams in both periods (before vs. after) is shown in Table E.1.

Table E.1. Mean abundance in two streams, before and after supplementation is started

Stream	Mean during BEFORE period Log(mean)	Mean during AFTER period log(mean)
Reference	200	200
	5.30	5.30
Supplemented	300	400
	5.70	5.99

The BACI analysis looks for a differential change, i.e., the difference in change in the log(mean) for the reference stream between the before and after periods vs. the change in the log(mean) for the supplemented stream, i.e.

$$\left(\log(mean)_{sup,after} - \log(mean)_{sup,before}\right) - \left(\log(mean)_{ref,after} - \log(mean)_{ref,before}\right)$$

or in this case

$$(5.99 - 5.70) - (5.30 - 5.30) = 0.28$$

[9] The analysis and conclusions are identical if the control stream has a change in the mean between the before and after periods.

representing an exp(0.28) = 1.33x times larger (multiplicative) change in the mean abundance in the supplemented stream (a 1.33x increase) compared to the reference stream (a 1.0x increase). This is known as the BACI effect.

Actual data will not fall exactly on the means and there are several sources of variation (Figure E.1).

- year-specific effects (e.g., weather) which affects returns to both streams equally. In the simple BACI experiment, these effects have a variance of σ^2_{year} which affects streams simultaneously. For example, if ocean conditions are such that abundances in a year increase, the same increase should be seen in all streams.
- year-site interaction effects that represent the non-parallelism of response between the supplemented and reference sites with a variance of $\sigma^2_{site-year}$ which is assumed the same for all years and all sites.
- measurement error where the actual abundance is measured with error. This is completely confounded with the year-site interaction effects and so will be "ignored."

Figure E.2 presents three scenarios of simulated data where the year-specific variance is the same in all scenarios, but the year-site interaction variance differs. The figure also presents the correlation between the measurements along with estimates of the BACI effect (see below)

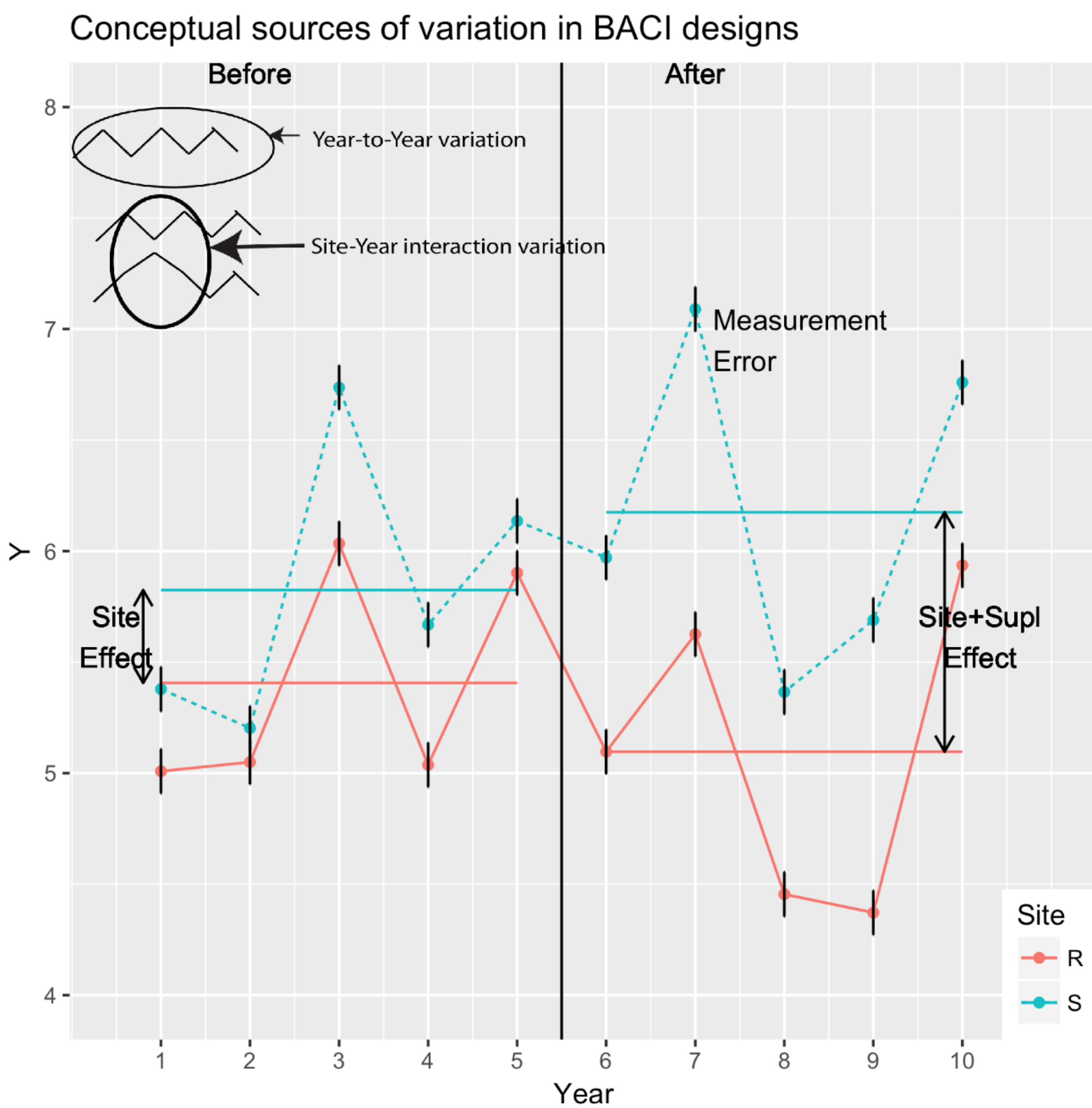

Figure E.1. Conceptual sources of variation in a BACI experiment with 1 supplemental stream (S) and one reference stream (R). Year-to-year variation represents year-specific effects that are equal in both streams. Site-year interaction represents non-parallelism in the response over time. This is confounded with measurement error and cannot be separated. The difference in the means during the before period is due to intrinsic site-to-site differences. The difference in the means during the after periods is due to the intrinsic site differences plus the effect of supplementation.

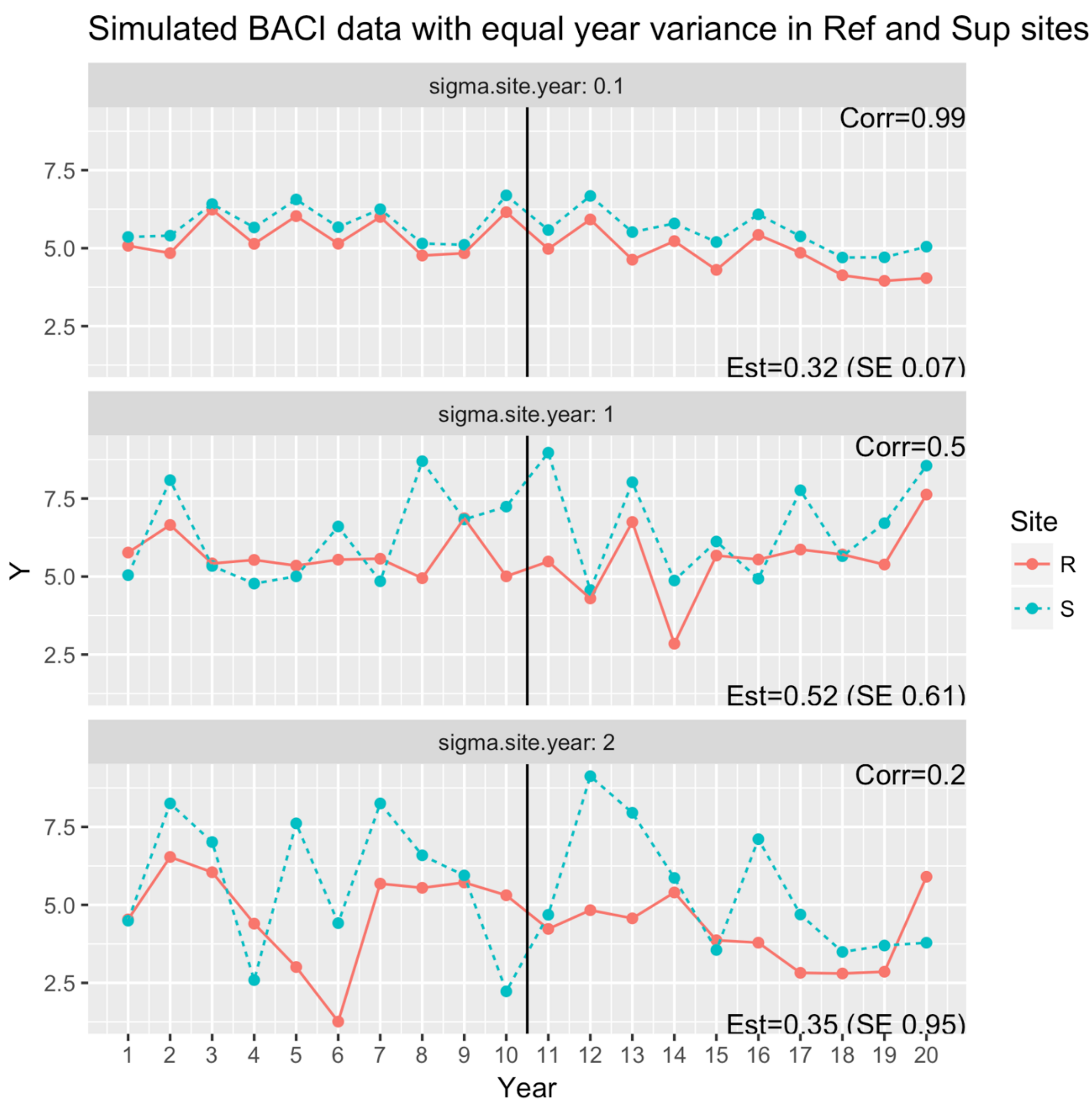

Figure E.2. Simulated results under three scenarios. In all scenarios, the variance of the year-specific effect is held fixed ($\sigma^2_{year} = 1$) while the year-site interaction effects vary as shown on the plot. The study was conducted using the means shown in Table E.1 with measurements taken 10 years before the supplementation program began and 10 years after the supplementation started.

There are two equivalent ways to estimate the BACI effect. First (as was done in Appendix C of Murdoch et al. 2011), the yearly values for the reference and supplemented streams are paired and the difference between the two (e.g., supplemented – reference measurement) is taken. This reduces the data to a single "difference" for each year, and two-sample t-test that the mean difference is the same in the before vs. the after period can be computed.

Second, a formal BACI model can be file (refer to Section 12.9 in people.stat.sfu.ca/~cschwarz/Stat-650/Notes/PDFbigbook-R/R-part013.pdf). The model is (using a standard modelling syntax)

$$Y = Site + Period + Site{:}Period + Year(R)$$

where *Site* refers to the site effects (reference vs. control), *Period* refers to the effects of before vs. after periods, and *Site:Period* represents the site-period interaction where the change between the before and after period is the difference for the reference and supplemental streams and represents the BACI effect; *Year(R)* represents the random year-specific effects which are assumed to affect both sites equally. This model is a Mixed Linear Model and can be fit using standard software (e.g., *lmer()* in *R*). A contrast of the *Site:Period* terms yields the BACI effect.

The estimated BACI effects are shown in bottom right of Figure E.2. The true value of the BACI effect (see above) is 0.28. In all scenarios, the estimated BACI effects are unbiased estimates of the true BACI effect (0.28), but the standard error of the estimates varies considerably.

In addition, the (theoretical) correlation between the readings in the two streams during the before period is shown in the upper right part of each plot for each scenario. It can be shown that the correlation is found a $\frac{\sigma_{year}^2}{\sigma_{year}^2+\sigma_{site-year}^2}$. Intuitively, the BACI effect is more precise when the correlation is larger. However, the degree of correlation has NO IMPACT on the bias of the estimates. Indeed, a valid BACI estimate can still be found even if the correlation between the readings in the two streams is 0. **The correlation between the readings in the two streams during the before period will affect the precision of the estimates, but the BACI estimates are still unbiased. Secondly, the precision of the BACI estimates "incorporates" the correlation between the measurements in the two streams prior to supplementation. The standard error provides a natural and defensible way to "rank" the reference streams.**

E.2. One Supplemented and Multiple Reference Streams

The design can be extended by including several reference streams and simulated data in the case with 1 supplemental stream and 3 reference streams is shown in Figure E.3. The BACI effect represents the differential change between the mean for the supplemental site vs. the differential change in the mean over ALL reference sites.

This BACI estimate can again be formed in two (equivalent) ways. First, three separate paired analyses can be done as before using the supplemented stream vs. each of the reference streams. This gives three estimates of the BACI effect, and they must now be combined. Because we are assuming that the year-specific effect and year-site interaction effect variances are the same for all stream, the overall BACI effect is found as a simple average of the three individual BACI estimates (i.e., a weighted average with equal weights). However, the estimates are NOT independent (the same supplemental steam is used in all three pairs) so the standard error of this weighted average is not easily found.[10]

Secondly, a more complex mixed-linear model can be fit:

$$Y = Class + Period + Class{:}Period + Site(R) + Year(R)$$

where *Site(R)* refers to the random site effects (the single supplemental stream and the 3 reference streams); *Class* refers to the two classifications of the sites (supplemental vs. reference); *Period* refers to the effects of before vs. after periods, and *Class:Period* represents the class-period interaction where the change between the before and after period is the different for the reference and supplemental streams categories and represents the BACI effect; *Year(R)* represents the random effect of year. As before this model can be fit using standard software (e.g., *lmer()* in *R)* and the BACI estimates found directly without having to average the estimates from the three pairs.

Referring to Figure E.3, we once again can show that all estimates are unbiased; the precision of the BACI estimates is improved relative to Figure E.2 (more than 1 reference stream incorporated); the standard errors of the overall BACI estimate is found automatically without having to average the results from the paired analysis. Once again, **the correlation between the readings among the streams during the before period will affect the precision of the estimates, but the BACI estimates are still unbiased. Secondly, the precision of the BACI estimates "incorporates" the correlation between the measurements among the streams prior to supplementation. This more complex model will automatically "average" the individual supplementation estimates from several reference streams.**

[10] The theoretical covariance between the three estimates depends only on the year-specific effects variance. In theory, this covariance could be used to find the standard error of the weighted average.

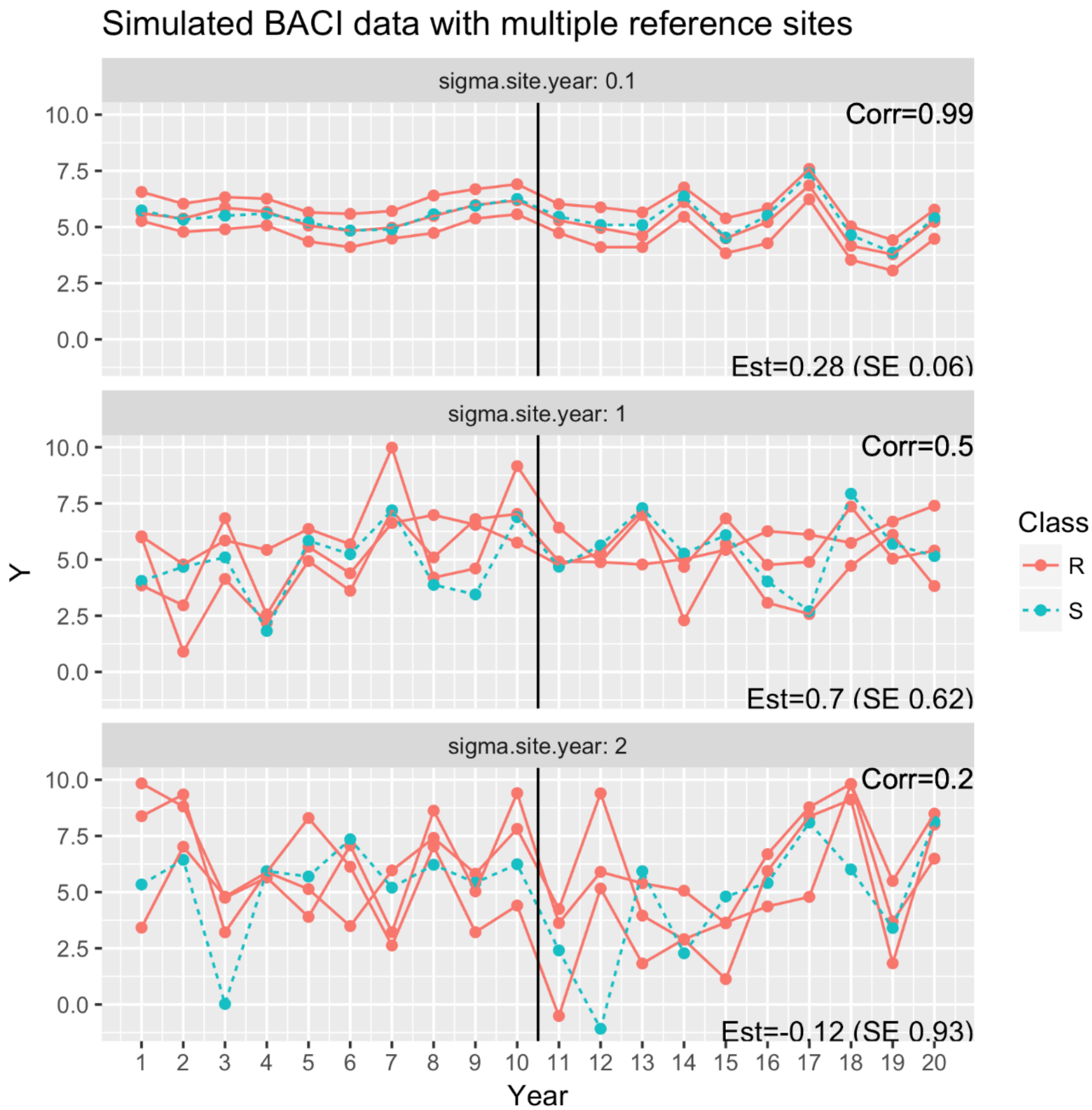

Figure E.3. Simulated results under three scenarios with 1 supplemented stream and 3 reference streams. In all scenarios, the variance of the year-specific effect is held fixed ($\sigma^2_{year} = 1$) while the year-site interaction effects vary as shown on the plot. The study was conducted using the means shown in Table E.1 with measurements taken 10 years before the supplementation program began, and 10 years after the supplementation started.

Both of the above analyses assumed that the year-specific effects had the same variance for all streams, so that if the year-specific effect caused an increase in the log(abundance) in one stream, it would cause the same increase in log(abundance) in the other streams. For example, if the year-specific effect caused a doubling of abundance, then the increase in log(abundance) would be log(2)=0.70 in both streams. A consequence of this assumption is that the variability across time in all streams is equal. This assumption may be violated where the variability in one stream is much larger/small than the variability in another stream. For example, year-specific effect could result in an increase/decrease in log(abundance) in one stream by a factor of 0.7 (doubling or halving), but in the same year, could cause change in log(abundance) by a factor of 0.35 (1.4x change). There would still be a high correlation on the year specific effects.

Figure E.4 shows simulated data illustrating these ideas. In these scenarios, the reference stream has a smaller variation over time than the supplemental stream. The year effects between the two streams are correlated as shown on the plot. For example, in the lower panel, there is a high correspondence in the two streams so that when the abundance in one stream increases, the abundance in the other stream also increases, but not by the same quantity. Similarly, decreases in abundance are matched. However, the BACI design “forces” the year effect to be equal and so any non-parallelism seen here is assumed to be site-year variation and so the “empirical” correlation smaller than the actual correlation.

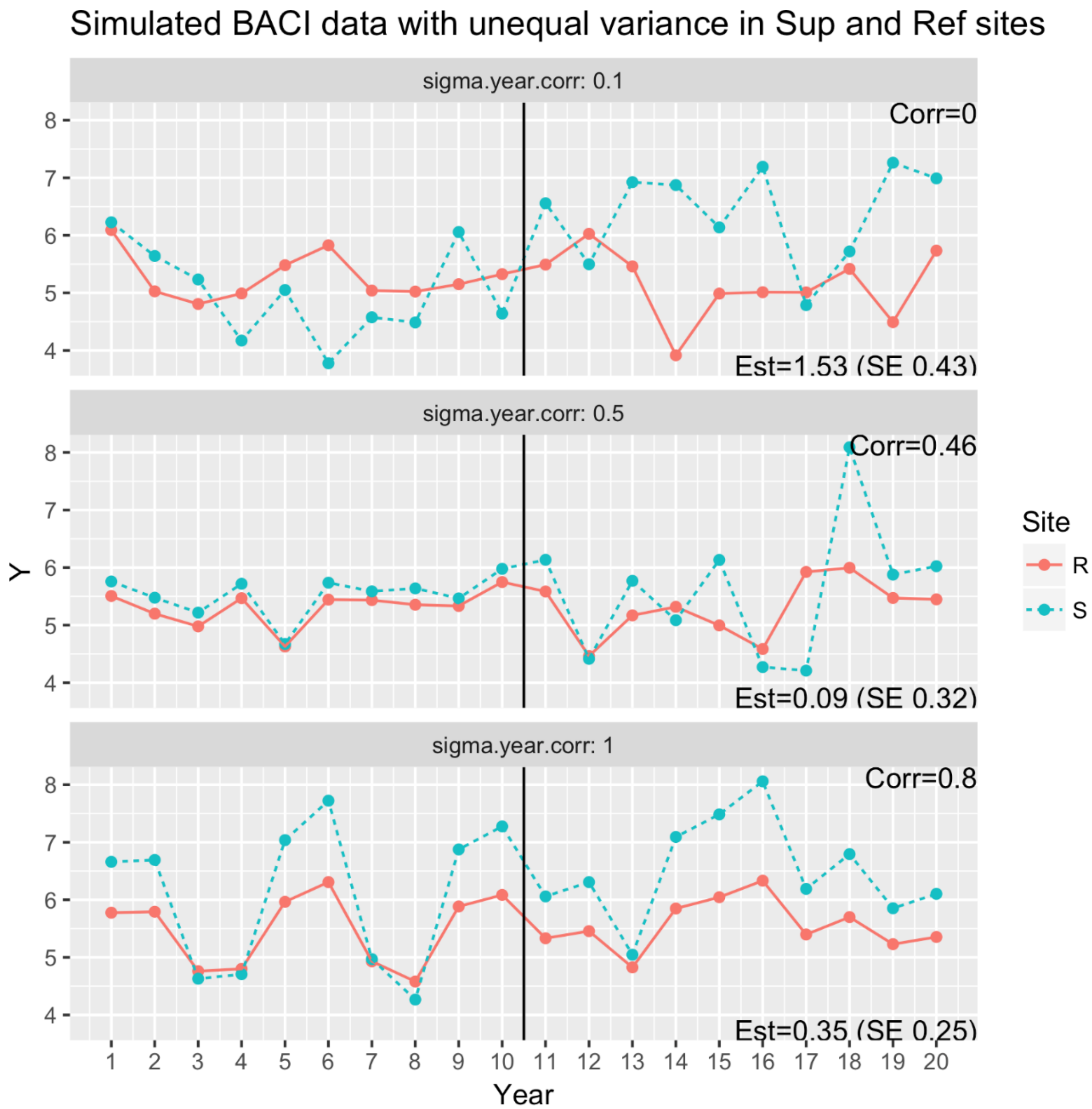

Figure E.4. Simulated results under three scenarios with 1 supplemented stream and 1 reference stream. The variability in the two streams differs ($\sigma^2_{sup,year} = 1;\ \sigma^2_{ref,year} = .25$) with different correlations between the year specific effects. The study was conducted using the means shown in Table E.1 with measurements taken 10 years before the supplementation program began, and 10 years after the supplementation started. The panel label shows the correlation when the data was generated; the upper right value shows the correlation when the variability is divided into a common year effect and site-year interactions.

There are again two equivalent ways to analyze this data. In the paired analysis, the effect of the unequal variances is "hidden" because the variance of the difference in a particular year is $\sigma^2_{sup,year} + \sigma^2_{ref,year} + 2\rho\sigma_{sup,year}\sigma_{sup,year}$ and so it doesn't matter if the variances are equal or different – the difference automatically has the appropriate variance.

If a BACI model is fit, the non-parallelism is assumed to be the effect of a common year-effect and a site-year interaction. While the BACI analysis “forces” a common year effect, it also adds a “year-site” effect to account for the non-parallelism and so again the final results are identical to the paired analysis. This non-parallelism is again represented by the correlation in the log(abundance) during the before period. **Even with unequal variances, it can be shown that the BACI estimates are unbiased and that the standard error of the estimate is related to the correlation of the log(abundance). Once again, the standard error of the estimate is the best measure for weighting the results from several reference streams because it incorporates all sources of uncertainty.**

Appendix F. Review of Hillman et al. 2017

F.1. Introduction

Hillman et al. (2017) present a five-year update to the Monitoring and Evaluation Plan being used to evaluate the PUD hatchery programs in the Upper Columbia. Many of the methods in their revised M&E Plan are similar to those presented in Murdoch et al. (2011). Here we concentrate on issues that were not raised in our review of Appendix C that was part of Murdoch et al. (2011). This review provides general comments about the Hillman et al. document, followed by a chapter-by-chapter detailed review.

F.1.1. General Comment about Null Hypothesis Significance Testing

Hillman et al. (2017) make extensive use of null hypothesis statistical test (NHST, Appendix G) to evaluate the impact of the supplementation program in the Upper Columbia. This is used in two ways.

First is assessing if the impact of supplementation results in a change to a parameter that measures the population. For example, consider the productivity indicator of abundance of natural spawners (first row of Table 1 in Hillman et al. 2017). The supplementation program should result in an increase in the mean number of natural spawners (the target change). The null hypothesis is no change in the mean number of spawners from pre- to post-supplementation. The test statistic is related to the observed difference in the sample mean number spawners from the pre- and post-supplementation program. A large observed difference (post-pre), accompanied by a small p-value, is evidence that the mean number of spawners has increased with supplementation. A power analysis is used to ensure that enough data (e.g., years of data pre- and post-supplementation) are collected to detect a certain desirable change (e.g., an increase by at least 100 spawners).

Second, the program may wish to detect an adverse consequence from supplementation. For example, consider the monitoring indicator of run timing (third row of Table 1 in Hillman et al. 2017). The supplementation program should result in no difference in migration timing between hatchery and natural fish. The null hypothesis is again no difference in some attribution of run timing (such as the mean arrival at the spawning grounds) between hatchery and natural fish. The test statistic is related to the observed difference in the mean run timing. A large difference in the mean run timing, accompanied by a small p-value, is evidence of an adverse consequence of the supplementation program. Now a power analysis is used to ensure that enough data are collected to detect an undesirable change (e.g., a change in run timing by 10 days).

In both cases, rather than just relying on a "reject" or "not reject" the null hypothesis dichotomy, it is also useful to present estimates of the effect size along with measures of uncertainty (e.g., standard errors or confidence intervals). When an effect is detected, the estimated effect size will indicate if this is a large, biologically important effect or a small (biologically non-important) effect that just happened to be detected because of a large sample size. The actual p-value is NOT informative, i.e., the p-value could be 0.01 in both cases. Similarly, if an effect is not detected, the estimated effect size will indicate if this is because it is a small effect (with small uncertainty) or a potential large effect but with such a large uncertainty, it was not detected. Again, the actual p-value is not informative, i.e., the p-value could be 0.80 in both cases.

Consequently, when reporting the results from hypothesis testing, both the p-value and estimates of effect size and measures of uncertainty must be reported. This last point needs to be emphasized more in Hillman et al. (2017).

Null hypothesis testing is not really designed to assess "no impact" hypotheses. Given a large enough sample size, it is almost certain that some effect will be detected, but it may not be biologically relevant. To directly assess a "no impact" hypothesis, a modification to the NHST framework may be useful, namely **equivalence testing**. The null hypothesis is defined as an effect large enough to be deemed interesting, specified by an equivalence bound. The alternative hypothesis is any effect that is less extreme than said equivalence bound. The observed data is statistically compared against the equivalence bounds. In the two one-sided t-tests (TOST) approach, two one-sided tests are performed with the null hypothesis being that the effect size is less than/greater than the lower/upper bounds respectively. If both tests are "rejected," then there is evidence that the actual effect size is within the zone of indifference. Equivalence testing reduces the misinterpretation of p-values larger than the alpha level as support for the absence of a true effect. Furthermore, equivalence tests can identify effects that are statistically significant but biologically unimportant. This method should be considered as an alternative to regular NHST when the target direction in an indicator is no change.

The greatest disadvantage of the NHST framework is that it does NOT provide a direct answer to the actual question – how certain is it that supplementation did or didn't have an effect? The p-value does NOT serve this purpose – the p-value measures the consistency of the data with no effect, which is easily misinterpreted. Here a Bayesian approach may be useful. The posterior belief about the parameter provides a direct interpretation of the question of interest. For example, consider again the productivity indicator of abundance of natural spawners (first row of Table 1 in Hillman et al., 2017). Under the NHST framework, a p-value of 0.02 indicates that the observed data is not consistent with no effect – hardly an interpretation that can be understood with ease. Under a Bayesian framework, a posterior belief of 0.98 that the mean abundance has increased under supplementation is easily interpreted. For most of the analyses used in this document, a Bayesian analysis is easily implemented.

F.2. Review of Section 1 of Hillman et al. (2017)

Section 1 of Hillman et al. (2017) presents an overview of the monitoring and evaluation plan. As noted by the authors, any management action needs to be evaluated for its effectiveness and a suite of outcomes from management is presented in their Figure 1 and their Table 1 (called indicators in their document). These outcome variables have been divided into two classes – monitoring indicators and productivity indicators (using their terminology) with the main distinction being that productivity indicators are the primary metrics to assess if conservation and safety-net goals have been met, with the monitoring indicators being used if the productivity indicators are not available or results are equivocal.

Their Table 1 presents a summary of the direction in which each of the monitoring and productivity indicators should be moving if the management actions are successful. These directions drive the statistical hypotheses created to evaluate the program. For example, the target direction for the abundance of natural spawners is an increase. The statistical hypothesis associated with this indicator will have a null hypothesis of NO change and an alternate hypothesis of an increase after supplementation. Thus the “rejection” of the hypothesis will provide evidence of a program success. On the other hand, the target direction for adult productivity (NRR) is no decrease. The statistical hypothesis associated with this indicator will have a null hypothesis of no change and an alternate hypothesis of a decrease in NRR after supplementation. Now failure to reject the hypothesis will be an indication of program success! As noted previously, the null hypothesis significance testing framework is less useful when monitoring for no change.

All indicator variables should relate directly to the populations such as abundance, juveniles per redd, pHOS etc. which are easy to interpret. But, their Table 1 also includes some indicators that cannot be directly measured in the populations such as Residuals vs. pHOS or Effective population size or Genetic Distance, which are statistical measurements and are much harder to interpret. Presumably, the latter should be given less weight when evaluating management actions, but this is never indicated in the document. These indicators need to be replaced by parameters that are directly measurable (see later sections below).

Their Table 1 also includes some indicators for which it is not obvious what is being compared. For example, indicators such as migration timing could be compared in terms of the entire distribution, of the mean arrival time, or of selected percentiles. These are explained in more detail later in the document, but their Table 1 could be updated with footnotes to indicate upon which dimension these indicators will be compared.

There is no guidance provided in Section 1 on how to weight the monitoring indicators

> "In the event that the statistical power of tests that involve productivity indicators is insufficient to inform sound management decisions, some of the monitoring indicators may be used to guide management."

Are changes in migration timing and distribution, for instance, more important than changes in redd distribution? This section could be improved by ranking the indicator variables in terms of importance to assessing the program.

To evaluate the impact of management actions, there is a need to account for temporal change to separate out the effect of management actions and temporal changes that would have occurred in the absence of management actions. The key design for such comparison is a Before–After-Impact-Control (BACI) design (and variants). A reference population is required that does not undergo management actions in order to measure temporal effect. There is a brief discussion in their Section 1 on how to assess impact in the absence of suitable reference populations. The authors should review Wiens and Parker (1995) for advice on dealing with such situations and update this section of the document.

F.3. Review of Section 2 (Adult Productivity) of Hillman et al. (2017)

This section provides a detailed protocol on how to assess the Adult Productivity indicators. For each indicator, a detailed monitoring question is provided along with formal statistical hypotheses to be examined from the monitoring study. For each indicator, what variables are measured or derived is given along with the frequency with which the indicator is measured.

F.3.1. Review of Section 2.1 (Natural Replacement Rate (NRR))

The NRR is defined as

$$NRR = \frac{\text{Natural Origin Returns}}{\text{Natural+Hatchery Spawners}}$$

The number of natural origin recruits (NOR) is computed on a brood year basis. From their Table 1, the NRR should not decrease if management actions are successful.

A set of six statistical hypotheses are presented – the authors indicate that the choice among the hypotheses will depend on the quality and quantity of the data available.

Because the target direction for this indicator is no increase, the statistical null hypothesis of no change was translated to a null hypothesis of program success, and a "rejection" of the hypothesis would provide evidence of a program failure.

Presumably, the intent is actually the reverse, i.e., until evidence provides elsewise, the status quo is that supplementation has no impact. All of the hypotheses should then be reversed in this section. If the intent is actually to assume equivalency until proven otherwise, then equivalence testing should be used.

This reversal of the null hypothesis implies that standard power computations are not informative because they provide information on the ability to detect failure and not success.

As noted in the previous review of Appendix C, hypotheses about the ratio of the NRR between supplemented and reference streams should be expressed in terms of the log(ratio). (Hypotheses 4 and 5) and difference in the slopes of NRR should be analyzed on the log() or logit() scale for the same reason (Hypothesis 2).

Hypothesis 6 is not a proper statistical hypothesis because it depends on data (the residual from a fit) and not the underlying population metrics (the mean NRR or slope of the NRR line). This needs to be recast by hypothesizing that the addition of pHOS has no impact on the stock recruitment curve, i.e., similar to a multiple regression hypothesis that a certain variable has no marginal impact on the fit. A correlation analysis is not appropriate here.

The list of possible statistical methods should include a BACI analysis and a paired t-test method when a reference stream is available (see companion review of Appendix C). Both methods are equivalent when there is a single reference stream, but a BACI analysis can be extended when there are multiple reference streams.

The authors suggest that

> “Correlation analysis will examine associations between hatchery adult composition and NRRs.”

There does not appear to be a definition of hatchery adult composition in this document. Does this mean pNOB (the proportion of the broodstock consisting of NORs)? If the hatchery adult composition is a proportion of some sort, then ordinary regression analysis should be used rather than correlational analysis as it will be more informative.

The suggestion that productivity should be correlated with other factors such as ocean productivity in a particular five-year period will have essentially no power to detect effects because of the very small sample size (5 years).

F.3.2. Review of Section 2.2 (Natural Origin Recruits (NOR))

The NOR is defined as number of natural origin returns (NOR) computed on a brood year basis. From their Table 1, the NOR should increase if management actions are successful.

A set of six statistical hypotheses are presented – the authors indicate that the choice among the hypotheses will depend on the quality and quantity of the data available.

Because the target direction for this indicator is an increase, the statistical null hypothesis of no change implies a null hypothesis of program failure, and a "rejection" of the hypothesis would provide evidence of a program success. This is a situation where the null hypothesis significance testing framework is most often used. The resulting p-value does not have an easy interpretation, and a Bayesian approach may be more useful.

As noted in the companion review of Appendix C, hypotheses about the ratio of the NOR between supplemented and reference streams should be expressed in terms of the log(ratio). (Hypotheses 4 and 5) and difference in the slopes of NRR should be analyzed on the log()scale for the same reason (Hypothesis 2).

Hypothesis 6 again suffers from the problem that the status quo is that supplementation is working and a "rejection" of the hypothesis is an indicator of failure (see previous section). If this is the intent, then equivalence testing is preferred over a simple hypothesis test.

The list of possible statistical methods should include a BACI analysis and a paired t-test method as noted previously.

Hatchery adult composition is not defined and a regression analysis may be more informative (see previous comments).

The suggestion that productivity should be correlated with other factors such as ocean productivity in a particular five-year period will have essentially no power to detect effects because of the very small sample size (5 years).

F.4. Review of Section 3 (Juvenile Productivity) of Hillman et al. (2017)

This section is structured similarly to Section 2 of Hillman et al. (2017) (reviewed earlier).

The key productivity indicator is the number of juveniles/redd. There are two monitoring questions. First, is there a general change in juveniles/redd between the years with and without supplementation. Second is there a specific relationship between juveniles/redd and the proportion of hatchery spawners (pHOS).

Their Table 1 is not consistent with this section. The first productivity indicator for juvenile productivity in Table 1 is Residuals vs. pHOS; the second indicator is juveniles per redd vs. pHOS. Neither of these are indicators, but rather statistical tests, and both should be replaced simply by juveniles/redd.

A set of six statistical hypotheses are presented for the general change and two for the specific relationship – the authors indicate that the choice among the hypotheses in each group will depend on the quality and quantity of the data available.

F.5.1. General Investigation of Impact of Supplementation on Juvenile Productivity

All of the hypotheses are framed in terms of detecting negative effects of supplementation. Equivalence testing may be a more suitable methodology rather than standard null hypothesis testing. The hypotheses in this section need to be updated in conjunction with changes to their Table 1.

As noted in the companion review of Appendix C, hypotheses about the ratio of the juveniles/redd between supplemented and reference streams should be expressed in terms of the log(ratio). (Hypotheses 4 and 5) and the difference in the slopes of juveniles/redd should be analyzed on the log()scale for the same reason (Hypothesis 2).

Hypothesis 6 is not a proper statistical hypothesis because it depends on data (the residual from a fit) and not the underlying population metrics (the mean juveniles/redd or slope of the juveniles/redd line). This needs to be recast by hypothesizing that the addition of pHOS has no impact on the stock recruitment curve, i.e., similar to a multiple regression hypothesis that a certain variable has no marginal impact on the fit. A correlation analysis is not appropriate here.

Hypothesis 6 also suffers from the problem that the status quo is that supplementation is working and a "rejection" of the hypothesis is an indicator of failure (see previous section). If this is the intent, then equivalence testing is preferred over a simple hypothesis test.

F.5.2. Specific Investigation of the Relationship between pHOS and Juvenile Productivity

The target direction for this indictor is no relationship. As noted earlier, equivalence testing may be a more suitable framework.

There are two hypotheses presented. The first hypothesis in this section is identical to Hypothesis 6 of the previous section and suffers from the same deficiencies.

The list of possible statistical methods should include a BACI analysis and a paired t-test method (both equivalent when there is a single reference stream) when a reference stream is available (see companion review of Appendix C).

Hatchery adult composition is not defined in this document, and a regression analysis will be more informative (see previous sections).

F.6. Review of Section 4 (Natural environment monitoring indicators) of Hillman et al. (2017)

This section is structured similarly to their Section 2 (reviewed earlier) and covers indicator variables that the authors state are to be used if the productivity indicators are equivocal

F.6.1. Review of Section 4.1 (Hatchery Replacement Rates (HRR)

The key variables to measure are the Hatchery Replacement Rates (HRR) and the Natural Replacement Rates where both are computed on a brood year basis.

A set of two hypotheses are created comparing the HRR to NRR and to a target value.

The notation used for the hypothesis test make it unclear exactly what is being done. For example, the two hypotheses are stated as:

> "Statistical Hypothesis 3.2.1:
>
> Ho3.2.1.1: HRR Year x > NRR Year x
>
> Statistical Hypothesis 3.2.2:
>
> Ho3.2.2.1: HRR ≥ Target Value identified in Appendix 2"

What does "Year x" refer to? If the intent is to compare the HRR to the NRR or to the target for EACH brood year, then this is NOT a statistical hypothesis testing problem. Assuming that error in counting returns from brood years is negligible, the HRR in year x either is or is not greater than the NRR in year x and either does or does not exceed the target vale for year x. There is no uncertainty in this comparison.

It is even more confusing when the suggested statistical methods are considered:

- "For Q3.2.1 use graphic analysis and paired-sample quantile tests to compare HRR to NRR
- For Q3.2.2 use graphic analysis and one-sample quantile tests to compare HRR to the target value."

Now it appears that a series of yearly data are being compared using pairing by brood years?

Assuming that a multi-year comparison is being done, then what quantiles will be compared? For example, suppose we wish to compare the 90th percentile of the HRR to the target and we find that there is no evidence that the 90th percentile exceeds the target values. It is unclear how useful this will be for management purposes. About the only quantile that would seem to be useful would be the .50 quantile (or the median).

As in previous sections, an analysis on the log() scale would be preferred.

Presumably the time series consists of years with and without supplementation taking place. Are the comparisons restricted to the years with supplementation only?

The proposed correlational analysis of HRR with extraneous factors will not be useful.

F.6.2. Review of Section 4.2 (Proportion of Hatchery Origin - pHOS)

The key variable to measure is the proportion of Hatchery Origin Spawners (pHOS) and the hypotheses compares pHOS to a target value.

No statistical methods are proposed. This type of comparison is very similar to the comparison of HRR to standards and similar methods can be used.

This indicator variable does not appear in their Table 1 and needs to be added.

F.6.3. Review of Section 4.3 (Run Timing, Spawn Timing, and Spawning Distribution)

These three performance indicators have similar hypotheses and will be analyzed in similar ways. Detailed comments on the analysis of run timing are provided, and similar comments will apply to the analysis of spawn timing or spawning distribution.

Unlike the previous sections where one value was obtained per year, the data for these metrics consists of a set of values for every year of the study. For example, the analysis of run timing would use a dataset consisting of the arrival time of every fish at the measuring location for every year.

One simple way to analyze this data is to reduce each year's multiple values (the actual arrival times of individual fish) to a single number (such as the mean, median, or percentiles). Then the usual paired t-test, BACI analysis, or regression analyses can be used to examine if the mean of the yearly means, median, or percentiles are equal between the hatchery and naturally produced population. These types of comparisons were conducted with the indicator variables of earlier sections.

Another approach would be to use two-sample Kolmogorov-Smirnov tests to annually examine differences in cumulative distributions. The K-S test is sensitive to any type of difference in the distributions from which the samples are drawn – e.g. location (central tendency) in dispersion, skewness and so forth. K-S tests would allow differences in a single year to be detected. It becomes more complicated when multiple years are involved. Consultation with a statistician is advised to determine how best to detect persistent differences across multiple years. Consequently, Hypotheses 1 and 2 of this section are difficult to assess with simple statistical methods – it will be likely that some sort of randomization procedure (e.g., bootstrapping) will be needed. These randomization methods will have to done very carefully, as there are several levels in the analysis that need to be accounted for – within year variation and across year variation while keeping the paired nature of the data present.

F.6.3.1. Specific Comments about Comparing Run Timing

There are three hypotheses listed for this monitoring indicator. Hypothesis 3 uses summary statistics for each year's distribution in the comparisons. Hypotheses 1 and 2 are actually the same hypothesis – equality of the migration timing is identical to equality of the cumulative distribution.

F.6.3.2. Specific Comments about Comparing Spawn Timing

The first two hypotheses are similar to the comparable hypotheses for the run timing indicator. The third hypothesis needs further refinement as it is not clear what is being examined. It appears that this will be a type of ANCOVA where a regression line between spawn time and elevation is compared between hatchery and natural fish. However, how are multiple years of data included in this comparison? As in comparing multiple years of distributions, there are no simple statistical methods and the ones proposed in this document are inappropriate.

F.6.3.3. Specific Comments about Comparing Spatial Distribution of Redds

Similar methods could be used for comparing spatial distribution of redds in a single stream. In river systems that possess multiple tributaries distribution patterns become more difficult to assess. However, we don't believe this is an insurmountable problem if K-S are used. In the Wenatchee for example, there are two spring Chinook supplementation projects of interest, the Chiwawa and Nason Creek. These tributaries could be segmented into equal sized sections and redds counts of HORs and NORs falling into each stream segment could be made in each tributary and used to create cumulative frequency distributions. Similarly, in the Methow, distributions of hatchery and natural fish in the Twisp, Chewuch, and portions of the Methow are of interest. The K-S tests would be done separately for each tributary or main river. It would also be possible to compare distributions for multiple segments, but this will not be straightforward. In this case, bootstrapping appears to be a useful option.

This portion of the document indicates that a percent overlap in distribution will be used without any details on how this will be computed. The document then indicates that chi-square test will be performed. This may be suitable for a single year of data, but there is no simple way to include multiple years of data into the analysis. More details are needed here.

F.6.4. Review of Section 4.4 (Stray rates)

Some care is needed about the definition of stray rates. Section 10 (Glossary) defines stray rate as:

> "The rate at which fish spawn outside of natal rivers or the stream in which they were released."

By this definition, the numerator is the number of fish that spawned in a natal stream and the denominator is the number of fish from that natal area that spawned elsewhere than the natal stream. In order to measure the stray rate, the origin of spawners at ALL possible spawning sites for that natal spawner set must be measured.

However, in this section, the term "stray rate" is used for the proportion of spawners in a natal stream that came from other natal streams, i.e., the same denominator is used but now the numerator is the number of fish from other natal streams spawning here. For example:

> "…hatchery strays from other populations cannot make up more than 5% of the spawning escapement within a non-target, recipient population."

This is quite different from the definition in the glossary; determination of the natal origin of all fish in a single spawning aggregate is now needed.

F.6.4.1. Specific Comments about Brood Year Stray Rates

It is not clear from the document which definition of stray rate is being used. For example, the first monitoring question is:

> "Q6.1.1: What is the brood-year stray rate of hatchery fish?"

This appears to be based on the first definition of stray rate. Consequently, much more justification is needed that all possible spawning areas have been searched to find strays from a natal brood. The specific hypothesis is also problematic:

> "Ho6.1.1.1: None. "

This implies that a single stray fish from a hatchery brood would be enough to indicate that the hatchery program has failed. Indeed, in the list of potential analyses no actual statistical methods are listed.

F.6.4.2. Specific Comments about Among-population Return-year Stray Rates

Now the second definition of a stray-rate is being used, i.e., what fraction of a spawning aggregate is composed of fish from other natal areas.

Their Table 1 indicates that that target for the indicator is that < 5% must be out-of-basin. However, the hypothesis listed in this section has a null hypothesis where more than 5% of spawning aggregate is made up of strays (which is again backwards).

The suggested analyses are 1-sample quantile tests based on the yearly proportions which is appropriate, but such tests do not test the null hypothesis as noted above.

F.6.4.3. Specific Comments about Within-population Return-year Stray Rates

The same comments apply as in the previous section.

F.6.5. Review of Section 4.5 (Population Genetics)

The preamble to this section indicates:

> "…it is important to monitor the genetic status of the natural populations to determine if there are signs of changes in genetic distance among populations, changes in allele frequencies, and to estimate effective population size."

Hence monitoring takes place of the natural population only and the genetic composition of hatchery fish is not relevant.

F.6.5.1. Specific Comments about Allele Frequency

Despite the emphasis on genetic changes in the natural populations, the hypotheses in this section compare the allele frequency of natural, hatchery, and broodstock fish. It is not clear how such a comparison helps in assessing the effects of the supplementation program.

This section appears to list three alternate hypotheses in an attempt to list the alternatives if the null hypothesis is not tenable. There are actually 4 alternate hypotheses, but it is not necessary to list the individual comparisons because the tests for the null hypothesis are non-specific.

These comparisons appear to be done only for a specific year and will not use the entire time series. The list of possible statistical analyses cannot deal with multiple years of data.

F.6.5.2. Specific Comments about Genetic Distance between Populations

This now compares the genetic distances at two time points, but the proposed analyses does not support an analysis of more than two years of data.

F.6.5.3. Specific Comments about Effective Spawning Populations

The specific monitoring question is:

> Q7.3.1:“Is the ratio of effective population size (Ne) to spawning population size (N) constant over time?”

This differs from the target listed in Table 1 where an increasing ratio is of interest.

However, the specific statistical hypothesis being investigated is much more specific indicating that each ratio of effective population size to spawning population size must be equal over time. This “equality” does not allow for year-specific effects on this ratio and is inappropriate.

This hypothesis can be simply investigated using a regression analysis on the observed values.

F.6.6. Review of Section 4.6 (Phenotypic traits)

This section assesses the potential effects of domestication by examining size at maturity, age at maturity, sex ratio, and fecundity (among other variables).

F.6.6.1. Specific Comments about Age at Maturity

Assuming that fish are collected at random from the spawners so that the sample is representative of the entire population, this dataset again is two dimensional with multiple fish collected on multiple years.

As with the monitoring indicator for run timing, the analysis of such data is complicated, but this section ignores these complications. The simplest approach would be to compute the mean age of maturity, or the proportion of fish at age 3 for natural and hatchery populations for each year, and then use a paired t-test to compare these summary measures over time. However, this section is silent, again, on how to deal with pre- and post- supplementation data.

The document appears to suggest that some sort of chi-square test will be used to compare the frequency distribution (e.g., proportion of 3, 4, 5 years fish) without having to compute a mean, but chi-square tests are not appropriate for multiple years of data.

F.6.6.2. Specific Comments about Size at Maturity

Again, a key assumption is that fish are collected at random from the spawning grounds. These data are multi-dimensional – multiple years of data; each year of data has data from two genders, multiple ages, and hatchery vs. natural populations; multiple fish collected in each combination of year, gender, age, origin.

The statistical hypotheses need to be restated in terms of the mean size or mean length at maturity rather than simply in terms of size or length.

The proposed analysis method using a 3-factor ANOVA is not appropriate because of the multiple years of data and complex nesting and pairing occurring. For example, year specific effects do not operate on individual fish, but rather on groups of fish and individual fish are pseudo-replicates. The authors need to consult with a statistician on the proper analysis of this data.

F.6.6.3. Specific Comments about Fecundity at Size

The monitoring question in this section looks at the relationship between fecundity or gonadal mass and size of the fish and compares if these are the same for hatchery and natural fish.

Again, the data is multidimensional with fish collected over multiple years and multiple fish collected in each year. A type of regression analysis will be needed, but ordinary ANCOVA is not appropriate because of the multiple years of data collected. A random effect ANCOVA will be needed with year random effects on both the intercept and slope for the hatchery and natural populations, but a correlation will need to be assumed among the random effects for the two populations in a single year. The authors need to consult with a statistician on the proper analysis of this data.

F.6.6.4. Specific Comments about Sex Ratio

The monitoring question compares the sex ratio for hatchery and natural populations. The main concern is again how to incorporate multiple years of data into the analysis – the proposed use of a simple chi-square test is not appropriate.

F.7. Review of Section 5 (Hatchery environment monitoring indicators) of Hillman et al. (2017)

This section is structured similarly to their Section 2 (reviewed earlier) and covers indicator variables for hatchery operations such as size and number of fish.

F.7.1. Specific comments on Size at Release of Hatchery Fish

This section appears to be comparison of the number of fish released and mean size of fish released to target values for each year. Random samples of fish collected at release are used.

The analysis appears to be a simple t-test comparing the mean weight /size of fish to hatchery targets for each year. However, the analysis is not appropriate because fish are not individually randomly selected; rather groups of fish are selected and this grouping has not been accounted for in the proposed analyses. A random effect, sub-sampling t-test will be required to assess these hypotheses.

How will multiple years of data be analyzed to see if the hatchery program is meeting its goals?

F.7.2. Specific Comments on Coefficient of Variation (CV) of Hatchery Fish Released

This section is similar to the previous except now the CV of weight or size is the response variable. Similar comments apply.

F.7.3. Specific Comments about Condition Factor (K) of Hatchery Fish Released

This section is similar to the previous except now condition factor is the response variable. A regression analysis will be needed to estimate K, and then a one-sample t-test will be used to compare to the hatchery target. Similar comments as in the previous section also apply here.

F.8. Review of Section 6 (Harvest monitoring indicators) of Hillman et al. (2017)

This section (very) briefly discusses monitoring of harvest and how it will be compared to targets to meet program goals.

There are two levels of assessment. First each year's harvest and escapement can be compared to program targets. No statistical tests are needed as the harvest/escapement either does or does not meet program targets.

A much more interesting question is how to use multiple years of data. Here it is not clear at all how to combine multiple years of data into a sensible monitoring indicator. Is the proportion of time program targets are met useful? Is the mean overage or underage compared to program targets useful?

This section appears to be interested in comparing overage/underage to specific quantiles (which ones -- only the median seems useful) using a 1-sample quantile test).

F.9. Review of Section 7 (Regional Objectives) of Hillman et al. (2017)

In this section, the incidence of disease and non-target taxa are of concern.

This section on the incidence of disease is still under development without any hypotheses because there is currently no disease management plan.

This section simply points to another report on risk assessment of the impact of hatchery programs on non-target taxa. This second report was not reviewed.

F.10. Review of Section 8 (Adaptive management) of Hillman et al. (2017)

No comments.

F.11. Review of Appendix 1 (Estimation of carrying capacity)

Refer to the companion document evaluating Appendix C of the UCR for comments on fitting the stock-recruitment curves to estimate population capacity.

The authors then distinguish between population capacity and habitat capacity. In particular, the authors note:

> "The fact that there are actual recruitment data above the estimated population capacity indicates that habitat capacity must be greater than the population capacity, or that measurement error is high. The former explanation is more likely than the latter."

Another explanation is that capacity is not a fixed ceiling value each year but that year-specific effects (random noise in the stock recruitment models) can temporarily raise or lower capacity in a particular year. For example, in a very dry year, the capacity of a stream to support smolt production may be lowered compared to a very wet year. So, it is unclear, why the authors are worried about the observed fluctuations about the median stock recruitment curve. While the quantile regression method may identify the theoretical maximum capacity under ideal year-specific effects, how is this useful in deciding if management actions are useful because it is not possible to modify the year-specific random effects? On the contrary, the lower 10% quantile would be of more interest because the "minimum" capacity that could be expected due to random year effects and so represents the worst-case scenario.

The authors used regression methods to estimate the 90th percentile based on least squares but indicated they were unable to estimate the reference intervals (prediction intervals) for the hockey-stock model because the hockey-stick model is not "linear." This is actually incorrect – the hockey stock model is linear (in the mathematical sense) even though the actual curve is not a straight line. Consequently, standard methods for finding reference intervals should also work for this model.

In the recommendation section, the authors state:

> "When AICc values are not appreciably different, then select the model that is most useful (e.g., Ricker and smooth hockey stick models are easier to work with than the Beverton-Holt model)."

A better approach would be to use model averaging. All three models have the same ease of use if maximum likelihood methods are used rather than non-linear least squares for model fitting.

> "The percentage of the reference interval should be set using the error in the estimation of the recruits and the level of desire to exclude anomalous data. For example, if the

> 95% confidence interval is approximately 10% of the recruitment estimate, then the reference interval should be set at 90% (e.g., RI = 100% - C.I.%).

The authors are confusing confidence intervals (uncertainty in estimates of population parameters) and prediction (reference) intervals (range of future values). This statement is not correct.

F.12. Review of Appendix 12 to 15

No comments.

F.13. Review of Appendix 6 (Identifying and analyzing reference populations)

Refer to the companion document on evaluating Appendix C of the UCRR.

F.14. Editorial Comments

Tables 1 and 2. The bottom left cell under Objectives simply says “Appropriate.” It is not clear what this means.

Appendix G. Null Hypothesis Significance Testing (NHST)

This document makes extensive use of null hypothesis statistical test (NHST) to evaluate the impact of the supplementation program in the Upper Columbia. The general paradigm starts with a variable (a *monitoring* or *productivity indicator* in this document) that is related to an underlying parameter of the model. A null hypothesis posits no change in the underlying parameter attributable to the supplementation program. The *indicator* is measured and the observed value of the *indicator* is compared to the sampling distribution of the *indicator* assuming no impact of the supplementation program. If the observed value of the *indicator* is unusual compared to the sampling distribution under no effect (e.g., a small p-value), then this is taken as evidence for an effect of the supplementation program.

This process works reasonably well when the effect of supplementation indeed leads to a change in the underlying parameter. For example, consider the *productivity indicator* of abundance of natural spawners (first row of Table 1 in Hillman et al, 2017). The supplementation program should result in step change in the mean number of natural spawners (the target change is an *increase* – first row of table 1 in Hillman et al, 2017). The null hypothesis is no change in the MEAN number of spawners from pre- to post- supplementation (ignoring for now problems of temporal change) as indicated by $Ho_{1.2.1.3}$ (page 11 in Hillman et al. 2017). The test statistic is related to the observed difference in the sample mean number spawners from the pre- and post-supplementation program. A large observed difference (post-pre), accompanied by a small p-value, is evidence that the mean number of spawners has increased with supplementation.

There are two possible errors that could be made. First is a false positive, i.e., you incorrectly conclude that there is evidence of an effect when in reality there is none. The false positive rate is controlled by the alpha level (typically set to 0.05) and is independent of sample size. Second is a false negative where an actual effect exists, but it was not detected. This is controlled by the sample size (number of years measured before and after supplementation was implemented), with a larger sample size having a reduced false negative probability.

Failure to detect an effect (i.e., a large p-value) could be the result of non-existent effect of supplementation or a small effect of supplementation that was masked by variability in the data). A power analysis would consider what size of effect is important to detect with a high probability (e.g., what sample size is needed to ensure that an increase in the mean by 100 spawners is detected with high probability).

Rather than just relying on a “detect” and “not detected” dichotomy, it is also useful to present estimates of the effect size along with measures of uncertainty (e.g., standard errors or confidence intervals). This is important in both cases. When an effect is detected, the estimated effect size will indicate if this is a large, biologically important effect, or a small (biologically non-important) effect that just happened to be detected because of a large sample size. The actual p-value is NOT informative, i.e., the p-value could be .01 in both cases. So any hypothesis testing results must provide both the p-value and estimates of effect size.

This framework performs less well when the supposed effect of supplementation is non-existent (e.g., target impacts of *no difference* in Table 1 of Hillman et al. 2017). Now the hypothesis testing framework is set up to detect deleterious effects of supplementation. A small p-value indicates evidence of deleterious effect and a large p-value indicates no evidence of deleterious effect. A power analysis indicates the sample size necessary to detect a certain deleterious effect. Again, effect size estimates are important to distinguish between detecting a large negative effect with a small sample size or a small (biologically unimportant) effect with a large sample size.

8. Literature Cited

Ahmadi-Hedushan B, A St-Hilaire, M Berube, E Robichaud, N Thiemonge, and B Bobee. 2006. A review of statistical methods for the evaluation of aquatic habitat suitability for instream flow assessment. River Research and Applications 22:503-523.

Andonaegui C. 1999. Salmon and steelhead habitat limiting factors report for the Entiat Watershed Water Resource Inventory Area (WRIA) 46, Version 3. Washington State Conservation Commission, Headquarters Office, Olympia, Washington.

Andonaegui C. 2001. Salmon, steelhead, and bull trout habitat limiting factors for the Wenatchee subbasin (Water Resource Inventory Area 45) and Portions of WRIA 40 within Chelan County (Squilchuck, Stemilt and Colockum drainages). Final draft report. Washington State Conservation Commission.

Arrow KJ. 1965. Criteria for social investment. Water Resources Research 1.1:1-8.

Arrow K et al. 2013. Determining benefits and costs for future generations. Science 341.6144:349-350.

Arterburn J. 2017. Okanogan and Methow habitat status and trends. Presentation to the Independent Scientific Advisory Board on July 20, 2017.

Arterburn J and R Klett. 2015. 2015 Chinook habitat status and trend report cards for 2013 monitoring. www.cctobmep.com/obmep_publications.php#edt_reports

Atkinson R. 1999. Project management: cost, time and quality, two best guesses and a phenomenon, it's time to accept other success criteria. International Journal of Project Management 17.6: 337-342.

Barker RJ. 1997. Joint modeling of live-recapture, tag-resight, and tag recovery data. Biometrics 53:666-667.

Barker RJ and G White. 1999. Joint analysis of live and dead encounters of marked animals. Program MARK, https://sites.warnercnr.colostate.edu/gwhite/wp-content/uploads/sites/73/2017/04/LiveDead.pdf

Barnas KA, SL Katz, DE Hamm, MC Diaz, and CE Jordan. 2015. Is habitat restoration targeting relevant ecological needs for endangered species? Using Pacific Salmon as a case study. Ecosphere 6(7):110. dx.doi.org/10.1890/ES14-00466.1

Beechie TJ, DA Sear, JD Olden, GR Pess, JM Buffington, H Moir, P Roni, and MM Pollock. 2010. Process-based principles for restoring river ecosystems. BioScience 60(3):209-222.

Bellmore JR, CV Baxter, KD Martens, and PJ Connolly. 2013. The floodplain food web mosaic: a study of its importance to salmon and steelhead with implications for their recovery. Ecological Applications 23:189–207.

Bellmore JR, JR Benjamin, M Newsom, JA Bountry, and D Dombroski. 2017. Incorporating food web dynamics into ecological restoration: a modeling approach for river ecosystems. Ecological Applications 27:814-832.

Benjamin JR, JR Bellmore, EW Whitney, and J Dunham. Incorporating food web dynamics into life cycle modeling. *In* R Zabel et al. Interior Columbia Basin Life Cycle Modeling, May 27, 2017 Draft Report, National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle.

Blair GR, LC Lestelle, and LE Mobrand. 2009. The Ecosystem Diagnosis and Treatment Model: a tool for assessing salmonid performance potential based on habitat conditions. American Fisheries Society Symposium 71:289-309.

Blankenship SM, J Von Bargen, KI Warheit, and AR Murdoch. 2007. Assessing the genetic diversity of natural Chiwawa salmon and evaluating the effectiveness of its supportive hatchery supplementation program. Appendix J *in* Hillman T, M Miller, M Johnson, C Moran, J Williams, M Tonseth, C Willard, S Hopkins, B Ishida, C Kamphaus, T Pearsons, and P Graf. 2016. Monitoring and evaluation of the Chelan and Grant County PUDs hatchery programs: 2015 annual report. Report to the HCP and PRCC Hatchery Committees, Wenatchee and Ephrata, WA.

Booth DB, JG Scholz, TJ Beechie, and SC Ralph. 2016. Integrating limiting-factors analysis with process-based restoration to improve recovery of endangered salmonids in the Pacific Northwest USA. Water 8:174, doi:10.3390/w8050174.

Bouwes N, N Weber, CE Jordan, WC Saunders, IA Tattam, C Volk, JM Wheaton, and MM Pollock. 2016. Ecosystem experiment reveals benefits of natural and simulated beaver dams to a threatened population of steelhead (*Oncorhynchus mykiss*). Nature Scientific Reports 6:28581 | DOI: 10.1038/srep28581.

Boyd J, R Epanchin-Niell, and J Siikamäki. 2015. Conservation planning: a review of return on investment analysis. Review of Environmental Economics and Policy 9.1: 23-42.

Busack C, CM Knudsen, G Hart, and P Huffman. 2007. Morphological differences between adult wild and first-generation hatchery upper Yakima River spring Chinook salmon. Transactions of the American Fisheries Society 136:1076-1087.

Busch DS, M Sheer, K Burnett, P McElhany, and T Cooney. 2013. Landscape-level model to predict spawning habitat for lower Columbia River fall Chinook salmon (*Oncorhynchus tshawytscha*). River Research and Applications 29:297-312. DOI: 10.1002/rra.1597

Chasco BE, IC Kaplan, AC Thomas, A Acevedo-Gutiérrez, DP Noren, MJ Ford, MB Hanson, JJ Scordino, SJ Jeffries, KN Marshall, AO Shelton, C Matkin, BJ Burke, and EJ Ward. 2017. Scientific Reports 7: 15439, DOI:10.1038/s41598-017-14984-8.

Christie MR, MJ Ford, and MS Blouin. 2014. On the reproductive success of early-generation hatchery fish in the wild. Evolutionary Applications 7:888-896.

Christie MR, ML Marine, RA French, RS Waples, and MS Blouin. 2012. Effective size of a wild salmonid population is greatly reduced by hatchery supplementation. Heredity 109:254-260.

Cleary PJ, M Edwards, IP Wilson, E Shoudel, and D Bright. 2014. Evaluation of spring Chinook salmon *Oncorhynchus tshawytscha* supplementation in the Lostine River, Oregon 2012 Annual Report (January 2012 to December 2012). U.S. D.O.E. Bonneville Power Administration Division of Fish and Wildlife, Portland, Oregon 97208. BPA Project Number: 1998-007-02. BPA Contract Number: 0005502.

Colville Confederated Tribes. 2013. OBMEP/EDT Habitat Status and Trends Report for the 2009 Monitoring Cycle: Okanogan Summer/Fall Chinook. Omak, WA: Colville Confederated Tribes. ICF 00579.12. Prepared by ICF International, Seattle, WA.

Conner MM, SN Bennett, WC Saunders and N Bouwes. 2015. Comparison of tributary survival estimates of steelhead using Cormack-Jolly-Seber and Barker Models: Implications for sampling efforts and designs. Transactions of the American Fisheries Society 144:34-47.

Conroy, MJ and JT Peterson. 2013. Decision making in natural resource management: a structured, adaptive approach. John Wiley & Sons, Ltd.

Cooper, M, M Humling, G Fraser, and H Potter. 2017. Hatchery Evaluation Plan for the Leavenworth Hatchery Complex. Version 01. U.S. Fish and Wildlife Service, Leavenworth WA.

Crump C, L Naylor, A VanSickle, B Startzel-Holt. 2017. Annual Progress Report. Lower Snake River Compensation Plan Confederated Tribes of the Umatilla Indian Reservation. Evaluation Studies for 1 January 2016 to 31 December 2016. Evaluation of Reestablishing Natural Production of Spring Chinook Salmon in Lookingglass Creek, Oregon. Using a Local Stock (Catherine Creek). Lower Snake River Compensation Plan, CTUIR Project No. 475, FWS Agreement F14AC00013.

CSS 2017. Comparative Survival Study of PIT-tagged Spring/Summer/Fall Chinook, Summer Steelhead, and Sockeye. DRAFT 2017 Annual Report. BPA Contract #19960200.

CTC (Chinook Technical Committee). 2017. Annual report of catch and escapement for 2016. Pacific Salmon Commission Joint Chinook Technical Committee Report TCCHINOOK 17-2.

de Gaudemar B, SL Schroder, and EP Beall. 2000. Nest placement and egg distribution in Atlantic salmon redds. Environmental Biology of Fishes 57:37-47.

de Wit A. 1988. Measurement of project success. International Journal of Project Management 6.3:164-170.

Desgroseillier TJ and NC Albrecht. 2016 DRAFT. Juvenile salmonid use of off-channel habitats within the Entiat River Basin, 2015. U.S. Fish and Wildlife Service, Leavenworth, Washington.

Desgroseillier T, M Cooper, G Fraser, and P DeHaan. 2017. Life history diversity of Entiat River subyearling Chinook salmon. Presentation to the ISAB, October 27, 2017.

Dittman AH, TN Pearsons, D May, RB Couture, and DLG Noakes. 2015. Imprinting of hatchery-reared salmon to targeted spawning locations: A new embryonic imprinting paradigm for hatchery programs. Fisheries 40:114-123

Elliott JM. 1994. Quantitative ecology and the brown trout. Oxford University Press, Oxford, U.K.

EPA US. 2010. Guidelines for preparing economic analyses. EPA. Washington, DC, US Environmental Protection Agency (updated 2014).

Evans, A, Q Payton, A Turecek, B Cramer, K Collis, D Roby, P Loschl, L Sullivan, M Weiland, J Skalski and R Townsend. 2015. Avian predation on juvenile salmonids in the Columbia River: A spatial and temporal analysis of impacts in relation to fish survival. Final Technical Report submitted to Grant County Public Utility District No. 2. www.birdresearchnw.org

Evans AF, Q Payton, A Turecek, B Cramer, K Collis, DD Roby, PJ Loschl, L Sullivan, J Skalski, M Weiland, and C Dotson. 2016. Avian predation on juvenile salmonids: spatial and temporal analysis based on acoustic and passive integrated transponder tags. Transactions of the American Fisheries Society 145: 860-877.

Farrow S and RO Zerbe. 2013. Principles and standards for benefit-cost analysis. Edward Elgar Publishing.

Faulkner JR, DL Widener, SG Smith, TM Marsh, and RW Zabel. 2017. Survival estimates for the passage of spring-migrating juvenile salmonids through Snake and Columbia River dams and reservoirs, 2016. BPA project 199302900. Northwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration.

Fausch KD. 1993. Experimental analysis of microhabitat selection by juvenile steelhead (*Oncorhynchus mykiss*) and coho salmon (*O. kisutch*) in a British Columbia stream. Canadian Journal of Fisheries and Aquatic Sciences 50:1198-1207.

Fedorenko AY and BG Shepherd. 1986. Review of salmon transplant procedures and suggested transplant guidelines. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1479, 151 pp.

FERC (Federal Energy Regulatory Commission). 2002. Anadromous Fish Agreement and Habitat Conservation Plan, Wells Hydroelectric Project, FERC License No. 2149, March 26, 2002.

Fish FF and MG Hanavan. 1948. A report upon the Grand Coulee fish maintenance project 1939-1947. U.S. Department of the Interior, Fish and Wildlife Service Special Scientific Report, Fish. 55, 63 pp.

Fisher JP, LA Weitkamp, DJ Teel, SA Hinton, M Trudel, JFT Morris, ME Thiess, RM Sweeting, JA Orsi, and EV Farley Jr. 2014. Early ocean dispersal patterns of Columbia River Chinook and Coho Salmon. Transactions of the American Fisheries Society 143:252–272.

Fleming IA and E Petersson. 2001. The ability of released hatchery salmonids to breed and contribute to the natural productivity of wild populations. Nordic Journal of Freshwater Research 75:71-98.

Fleming IA and MR Gross. 1992. Reproductive behavior of hatchery and wild coho salmon (*Oncorhynchus kisutch*) does it differ? Aquaculture 103:101-121.

Fleming IA, K Hindar, IB Mjolnerod, B Jonsson, T Balstad, and A Lamberg. 2000. Lifetime success and interactions of farm salmon invading a native population. Proceedings of the Royal Society of London B 267:1517-1523.

Ford, and et al. 2001. Upper Columbia River Steelhead and Spring Chinook Salmon Population Structure and Biological Requirements Final report, March 2001. Upper Columbia River Steelhead and Spring Chinook Salmon Biological Requirements Committee for the National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, WA.

Ford M. 2017. Status comparison of Upper Columbia spring Chinook and Snake River spring/summer Chinook. Presentation to the ISAB, Sept. 15, 2017. Portland, OR.

Ford MJ, A Murdoch, and M Hughes. 2015. Using parentage analysis to estimate rates of straying and homing in Chinook salmon (*Oncorhynchus tshawytscha*). Molecular Ecology 24:1109-1121.

Ford MJ, TN Pearsons, and A Murdoch. 2015. The spawning success of early maturing resident hatchery Chinook salmon in a natural river system. Transactions of the American Fisheries Society 144:539-548.

Fryer JK, J Mainord, J Whiteaker, and D Kelsey. 2011. Upstream migration timing of Columbia Basin Chinook and sockeye salmon and steelhead in 2009. Columbia River Inter-Tribal Fish Commission. Technical Report for BPA Project 2008-518-00, October 7, 2011. 78 p.

Fryer JK, J Whiteaker, and D Kelsey. 2012. Upstream migration timing of Columbia Basin Chinook and sockeye salmon and steelhead in 2010. Columbia River Inter-Tribal Fish Commission. Technical Report for BPA Project 2008-503-00, March 31, 2012. 79 p.

Fryer JK, J Whiteaker, and D Kelsey. 2013. Upstream migration timing of Columbia Basin Chinook and sockeye salmon and steelhead in 2011. Columbia River Inter-Tribal Fish Commission. Technical Report for BPA Project 2008-503-00, March 31, 2013. 86 p.

Fryer JK, J Whiteaker, and D Kelsey. 2013. Upstream migration timing of Columbia Basin Chinook and sockeye salmon and steelhead in 2012. Columbia River Inter-Tribal Fish Commission. Technical Report for BPA Project 2008-518-00, March 17, 2013. 97 p.

Fryer JK, J Whiteaker, and D Kelsey. 2015. Upstream migration timing of Columbia Basin Chinook and sockeye salmon and steelhead in 2013. Columbia River Inter-Tribal Fish Commission. Technical Report for BPA Project 2008-518-00, March 31, 2015. 100 p.

Fryer JK, J Whiteaker, and D Kelsey. 2015. Upstream migration timing of Columbia Basin Chinook and sockeye salmon and steelhead in 2014. Columbia River Inter-Tribal Fish Commission. Technical Report for BPA Project 2008-518-00, March 21, 2015. 106 p.

Fryer JK, H Wright, S Folks, R Bissanich, K Hyatt, M Stockwell, and J Miller. 2016. Limiting factors of the abundance of Okanagan and Wenatchee sockeye salmon in 2014. Technical Report 16-02. Columbia River Inter-Tribal Fish Commission. 131 p.

Fryer JK, J Whiteaker, and D Kelsey. 2017. Upstream migration timing of Columbia Basin Chinook and sockeye salmon and steelhead in 2015. Columbia River Inter-Tribal Fish Commission. Technical Report for BPA Project 2008-518-00, May 31, 2017. 119 p.

Gowan C and KD Fausch. 1996. Mobile brook trout in two high-elevation Colorado streams: re-evaluating the concept of restricted movement. Canadian Journal of Fisheries and Aquatic Sciences 53:1370-1381.

Grote JD and TJ Desgroseillier. 2016. Juvenile Salmonid Mark-Recapture Sampling within the Entiat River Intensively Watershed Study, 2015. U.S. Fish and Wildlife Service, Leavenworth, Washington.

Grote JD and TJ Desgroseillier. 2017. Juvenile salmonid out-migration monitoring on the Entiat River, 2016. U.S. Fish and Wildlife Service, Leavenworth, Washington.

Ham KD and TN Pearsons. 2001. A practical approach for containing ecological risks associated with fish stocking programs. Fisheries 25:15-23.

Hamm DE. 2012. Development and evaluation of a data dictionary to standardize salmonid habitat assessments in the Pacific Northwest. Fisheries 37:6–18. (cited in Barnas et al. 2015)

Harstad DL, DA Larsen, and BR Beckman. 2014. Variation in minijack rate among Columbia River Chinook salmon hatchery populations. Transactions of the American Fisheries Society 143:768-778.

Hess JE, JM Whiteaker, and JK Fryer. 2014. Monitoring stock-specific abundance, run timing, and straying of Chinook Salmon in the Columbia River using Genetic Stock Identification (GSI). North American Journal of Fisheries Management 34:184–201.

Hillman T, M Miller, M Johnson, C Moran, J Williams, M Tonseth, C Willard, S Hopkins, B Ishida, C Kamphaus, T Pearsons, and P Graf. 2016. Monitoring and evaluation of the Chelan and Grant County PUDs hatchery programs: 2015 annual report. Report to the HCP and PRCC Hatchery Committees, Wenatchee and Ephrata, WA.

Hillman T, P Roni, and J O'Neal. 2016. Effectiveness of tributary habitat enhancement projects. Report to Bonneville Power Administration, Portland, OR.

Hillman T, T Kahler, G Mackey, A Murdoch, K Murdoch, T Pearsons, M Tonseth, and C Willard. 2017. Monitoring and evaluation plan for PUD hatchery programs: 2017 update. Report to the HCP and PRCC Hatchery Committees, Wenatchee and Ephrata, WA.

Honea JM, MM McClure, JC Jorgensen, MD Scheuerell. 2016. Assessing freshwater life-stage vulnerability of an endangered Chinook salmon population to climate change influences on stream habitat. Climate Research 71: 127–137. doi: 10.3354/cr01434

Honea JM, JC Jorgensen, MM McClure, TD Cooney, K Engie, DM Holzer, and R Hilborn. 2009. Evaluating habitat effects on population status: influence of habitat restoration on spring-run Chinook salmon. Freshwater Biology 54:1576−1592.

Hughes MS, and AR Murdoch. 2017. Spawning habitat of hatchery spring Chinook salmon and possible mechanisms contributing to lower reproductive success. Transactions of the American Fisheries Society 146:1016-1027.

Humling M. 2016. Winthrop NFH spring Chinook and steelhead performance metrics [Memorandum]. Winthrop Field Office, Winthrop, WA. Dated May 2, 2016.

ICF International. 2017. Ecosystem Diagnosis & Treatment (EDT). edt.codeplex.com/documentation

ICF Jones & Stokes. 2010. Final Hatchery and Stocking Program Environmental Impact Report/Environmental Impact Statement. State Clearinghouse #2008082025. www.wildlife.ca.gov/Fishing/Hatcheries/EIR

ICTRT. 2007a. Viability Criteria for Application to Interior Columbia Basin Salmonid ESUs. Interior Columbia Basin Technical Recovery Team.

ICTRT. 2007b. Required Survival Rate Changes to Meet Technical Recovery Team Abundance and Productivity Viability Criteria for Interior Columbia River Basin Salmon and Steelhead Populations. Columbia Basin Technical Recovery Team.

ISAB (Independent Scientific Advisory Board). 2007-2. Climate change impacts on Columbia River Basin fish and wildlife. ISAB Climate Change Report. Northwest Power and Conservation Council, Portland, Oregon. Report ISAB 2007-2. 146 p. www.nwcouncil.org/fw/isab/isab2007-2.

ISAB (Independent Scientific Advisory Board). 2007-3. Human population impacts on Columbia River Basin Fish and Wildlife. ISAB Human Population Report. Northwest Power and Conservation Council, Portland, Oregon. Report ISAB 2007-3, 73 p. www.nwcouncil.org/fw/isab/isab2007-3.

ISAB (Independent Scientific Advisory Board). 2011-1. Columbia River Food Webs: Developing a broader scientific foundation for fish and wildlife restoration. Northwest Power and Conservation Council, Portland, Oregon. ISAB Report 2011-1. www.nwcouncil.org/fw/isab/isab2011-1.

ISAB (Independent Scientific Advisory Board). 2011-4. Using a comprehensive landscape approach for more effective conservation and restoration. Northwest Power and Conservation Council, Portland, Oregon. Report ISAB 2011-4. www.nwcouncil.org/fw/isab/isab2011-4.

ISAB (Independent Scientific Advisory Board). 2013-1. Review of the 2009 Columbia River Basin Fish and Wildlife Program. Northwest Power and Conservation Council, Portland, Oregon. Report ISAB 2013-1. www.nwcouncil.org/fw/isrp/isrp2013-1.

ISAB (Independent Scientific Advisory Board). 2013-5. Review of NOAA Fisheries' life-cycle models of salmonid populations in the Interior Columbia River Basin. Northwest Power and Conservation Council, Portland, Oregon. Report ISAB 2013-5. www.nwcouncil.org/fw/isab/isab2013-5.

ISAB (Independent Scientific Advisory Board). 2015-1. Density dependence and its implications for fish management and restoration programs in the Columbia River. Northwest Power and Conservation Council, Portland, Oregon. ISAB Report 2015-1. 246pp. www.nwcouncil.org/fw/isab/isab2015-1.

ISAB (Independent Scientific Advisory Board). 2016-1. Predation Metrics Report. Northwest Power and Conservation Council, Portland, Oregon. ISAB Report 2016-1. www.nwcouncil.org/fw/isab/isab2016-1/

ISAB (Independent Scientific Advisory Board). 2017-1. Review of NOAA Fisheries' Interior Columbia Basin Life-Cycle Modeling Draft Report. www.nwcouncil.org/fw/isab/isab2017-1.

ISAB (Independent Scientific Advisory Board). 2017-2. Review of Comparative Survival Study Draft 2017 Annual Report. Northwest Power and Conservation Council, Portland, Oregon. ISAB Report 2017-2. www.nwcouncil.org/fw/isab/isab2017-2.

ISAB/ISRP (Independent Scientific Advisory Board/Independent Scientific Review Panel). 2016-1. Critical uncertainties for the Columbia River Basin Fish and Wildlife Program. Northwest Power and Conservation Council, Portland, Oregon. ISAB Report 2016-1. 162pp. www.nwcouncil.org/fw/isab/isabisrp2016-1.

ISRP (Independent Scientific Review Panel). 2010-12. Response Request for the Upper Columbia Programmatic Habitat Project (2010-001-00). www.nwcouncil.org/fw/isrp/isrp2010-12.

ISRP (Independent Scientific Review Panel). 2010-28. Final Review of the Upper Columbia Programmatic Habitat Project (2010-001-00). www.nwcouncil.org/fw/isrp/isrp2010-28.

ISRP (Independent Scientific Review Panel). 2013-11. Geographic Review Final Report. Northwest Power and Conservation Council, Portland, OR. Report ISRP 2013-11. www.nwcouncil.org/fw/isrp/isrp2013-11.

ISRP (Independent Scientific Review Panel). 2014-5. Review of Progress Reports for Upper Columbia Programmatic Habitat Restoration Projects (#2010-001-00 and #2009-003-00). www.nwcouncil.org/fw/isrp/isrp2014-5.

ISRP (Independent Scientific Review Panel). 2014-10. Follow-Up Review of Progress Report for Upper Columbia Programmatic Habitat Restoration Projects (#2010-001-00 and #2009-003-00). Northwest Power and Conservation Council, Portland, Oregon. ISRP Report 2014-10. www.nwcouncil.org/fw/isrp/isrp2014-10.

ISRP (Independent Scientific Review Panel). 2016-9. Review of Idaho Supplementation Studies Project Completion Report 1991-2014. www.nwcouncil.org/fw/isrp/isrp2016-9.

ISRP (Independent Scientific Review Panel). 2017-2. Review of Umbrella Habitat Restoration Projects. www.nwcouncil.org/fw/isrp/isrp2017-2.

ISRP/ISAB (Independent Scientific Review Panel and Independent Scientific Advisory Board). 2004. Scientific Review of Subbasin Plans for the Columbia River Basin Fish and Wildlife Program. ISRP/ISAB 2004-13. www.nwcouncil.org/fw/isrp/isrpisab2004-13.

Jensen JOT. 2003. New mechanical shock sensitivity units in support of criteria for protection of salmonid eggs from blasting or seismic disturbance. Canadian Technical Report of Fisheries and Aquatic Sciences 2452. 19pp.

Johnson BM, BM Kemp, and GH Thorgaard. 2018. Increased mitochondrial DNA diversity in ancient Columbia River basin Chinook salmon *Oncorhynchus tshawytscha*. PLoS ONE 13(1): e0190059. doi.org/10.1371/journal.pone.0190059

Jorgensen J, A Murdoch, J Cram, M Sorel, T Hillman, G Maier, C Paulsen, T Cooney, R Zabel, and C Jordan. 2017. Chapter 9.b Wenatchee River spring-run Chinook salmon life-cycle model: hatchery effects, calibration, and sensitivity analyses. *In* R. Zabel et al. Interior Columbia Basin Life Cycle Modeling, May 27, 2017 Draft Report, National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle.

Kahler T. 2017. The role of science in salmon recovery. Presentation to the Independent Scientific Advisory Board on July 20, 2017.

Kassler T, S Blankenship, and AR Murdoch. 2011. Genetic structure of upper Columbia River summer Chinook and evaluation of the effects of supplementation programs. Washington Department of Fish and Wildlife Appendix M *in* Hillman T, M Miller, M Johnson, C Moran, J Williams, M Tonseth, C Willard, S Hopkins, B Ishida, C Kamphaus, T Pearsons, and P Graf. 2016. Monitoring and evaluation of the Chelan and Grant County PUDs hatchery programs: 2015 annual report. Report to the HCP and PRCC Hatchery Committees, Wenatchee and Ephrata, WA.

Keefer ML, CA Peery, MA Jepson, KR Tolotti, TC Bjornn, and LC Stuehrenberg. 2004. Stock-specific migration timing of adult spring–summer Chinook Salmon in the Columbia River basin. North American Journal of Fisheries Management 24:1145–1162.

Knudsen CM, SL Schroder, CA Busack, MV Johnston, TN Pearsons, WJ Bosch, and DE Fast. 2006. Comparison of life history traits between first-generation hatchery and wild upper Yakima River spring Chinook salmon. Transactions of the American Fisheries Society 135:1130-1144.

Lackey RT. 2017. Science and salmon recovery. pp. 69-94. *In* New Strategies for Wicked Problems: Science and Solutions in the 21st Century, EP Weber, DH Lach, and BS Steel, editors, Oregon State University Press, Corvallis, Oregon, 223 pp.

Larsen DA, BR Beckman, KA Cooper, D Barrett, M Johnston, P Swanson, and WW Dickhoff. 2004. Assessment of high rates of precocious male maturation in a spring Chinook supplementation hatchery program. Transactions of the American Fisheries Society 133:98-120.

Larsen DA, DL Harstad, AE Fuhrman, CM Knudsen, SL Schroder, WJ Bosch, DE Fast, and BR Beckman. *In Review.* Maintaining a wild phenotype in a conservation hatchery program for Chinook salmon: the effect of managed breeding on early male maturation.

Larsen DA, DL Harstad, CR Strom, MV Johnston, CM Knudsen, DE Fast, TN Pearsons, and BR Beckman. 2013. Early life history variation in hatchery-and natural-origin spring Chinook salmon in the Yakima River, Washington. Transactions of the American Fisheries Society 142:540-555.

Lichatowich J, R Williams, B Bakke, J Myron, D Bella, B McMillan, J Stanford and D Montgomery. 2017. *Wild Pacific Salmon: A Threatened Legacy.* jalich@comcast.net, Booklet funded by Fly Fishers International and Wild Fish Conservancy, Bemis Printing, St. Helens, OR.

Liss W, J Stanford, J Lichatowich, R Williams, C Coutant, P Mundy, and R Whitney. 2006. Developing a new conceptual foundation for salmon conservation. *In* Return to the River: Restoring Salmon to the Columbia River. Edited by R Williams, 51-98. Elsevier Academic Press, Burlington, MA.

Lura H, BT Barlaup, and H Saegrov. 1993. Spawning behavior of a farmed escaped Atlantic salmon (*Salmo salar*). Journal of Fish Biology 42:311-313.

Mackey G, TN Pearsons, MR Cooper, KG Murdoch, AR Murdoch, and TW Hillman. 2014. Ecological risk assessment of Upper-Columbia hatchery programs on non-target taxa of concern. Report produced by the Hatchery Evaluation Technical Team (HETT) for the HCP Wells Hatchery Committee, HCP Rocky Reach Hatchery Committee, HCP Rock Island Hatchery Committee, and the Priest Rapids Hatchery Sub-Committee. 89 pp.

Maier G. 2017. Upper Columbia Integrated recovery hatchery background summary. Upper Columbia Salmon Recovery Board, Wenatchee, WA. www.ucsrb.org/mdocuments-library/reports.

Manly, BFJ. 2001. Statistics for environmental science and management. New York: Chapman and Hall.

Martens KD and PJ Connolly. 2010. Effectiveness of a redesigned water diversion using rock vortex weirs to enhance longitudinal connectivity for small salmonids. North American Journal of Fisheries Management 30(6):1544–1552.

McElhany P, MH Ruckelshaus, MJ Ford, TC Wainwright, and EP Bjorkstedt. 2000. Viable salmonid populations and the recovery of evolutionary significant units. Seattle, WA: US Department of Commerce. NOAA Technical Memo NMFS-NWFSC-42. www.nwfsc.noaa.gov/assets/25/6190_06162004_143739_tm42.pdf

McElhany P, EA Steel, D Jensen, K Avery, N Yoder, C Busack, and B Thompson. 2010. Dealing with uncertainty in ecosystem models: lessons from a complex salmon model. Ecological Applications 20:465-482.

McHugh PA, WC Saunders, N Bouwes, CE Wall, S Bangen, JM Wheaton, M Nahorniak, JR Ruzycki, IA Tattam, and CE Jordan. 2017. Linking models across scales to assess the viability and restoration potential of a threatened population of steelhead (*Oncorhynchus mykiss*) in the Middle Fork John Day River, Oregon, USA. Ecological Modelling 355:24-38.

McMichael GA and BB James. 2017. Upriver bright predation bottleneck. Final report to the Letter of Agreement- Chinook Technical Committee, Pacific Salmon Commission, under contract CTC-2015-2. Mainstem Fish Research, LLC, Richland, Washington. 33 p. www.researchgate.net/publication/312320843_Upriver_Bright_Predation_ Bottleneck

Mejia FV, C Baxter, E Berntsen, and A Fremier. 2016. Linking groundwater – surface water exchange to food production and salmonid growth. Canadian Journal of Fisheries and Aquatic Sciences 73:1650-1660.

Mobrand LE, JA Lichatowich, LC Lestelle, and TS Vogel. 1997. An approach to describing ecosystem performance "through the eyes of salmon." Canadian Journal of Fisheries and Aquatic Sciences 54:2964-2973.

Moore MA et al. 2004. "Just give me a number!" Practical values for the social discount rate. Journal of Policy Analysis and Management 23.4: 789-812.

Mukherjee S. 2010. The Emperor of All Maladies: A Biography of Cancer. Simon and Schuster, New York. 571 p.

Mullan JW, KR Williams, G Rhodus, TW Hillman, and JD McIntyre. 1992. Production and habitat of salmonids in Mid-Columbia River tributary streams. U.S. Fish and Wildlife Service Monograph I, Leavenworth, WA.

Mullan JW. 1987. Status and propagation of Chinook salmon in the mid-Columbia River through 1985. USFWS Biological Report 89 (3), 111 pp. Leavenworth, WA.

Murdoch A. 2017. Upper Columbia spring and summer Chinook salmon: differences that may explain recent trends in abundance. Presentation to the ISAB, September 15, 2017.

Murdoch A, C Snow, C Frady, A Repp, M Small, S Blankenship, T Hillman, M Miller, and G Mackey. 2011. Evaluation of the hatchery programs funded by Douglas County PUD. Report to the Wells HCP Hatchery Committee, East Wenatchee, WA. 352 pp.

Murdoch AR, TN Pearsons, and TW Maitland. 2010. Estimating the spawning escapement of hatchery- and natural-origin spring Chinook salmon using redds and carcass data. North American Journal of Fisheries Management 30:361-375.

Myers JM et al. 1998. Status review of Chinook salmon from Washington, Idaho, Oregon and California. NOAA Technical Memorandum NMFS-NWFSC-35.

Naiman RJ, JR Alldredge, DA Beauchamp, PA Bisson, J Congleton, CJ Henny, N Huntly, R Lamberson, C Levings, EN Merrill, WG Pearcy, BE Rieman, GT Ruggerone, D Scarnecchia, PE Smouse, and CC Wood. 2012. Developing a broader scientific foundation for river restoration: Columbia River food webs. Proceedings of the National Academy of Sciences 109(52) 21201-21207.

Naish KA, JE Taylor, PS Levin, TP Quinn, JR Winton, D Huppert, and R Hilborn. 2008. An evaluation of the effects of conservation and fishery enhancement hatcheries on wild populations of salmon. Advances in Marine Biology 53:61-194.

Narum SR, JE Hess, and AP Matala. 2010. Examining genetic lineages of Chinook salmon in the Columbia River Basin. Transactions of the American Fisheries Society 139:1465-1477

National Research Council. 2005. Valuing ecosystem services: toward better environmental decision-making. National Academies Press.

NMFS (National Marine Fisheries Service). 2016. 5-Year Review: Summary & Evaluation of Snake River Sockeye Snake River Spring-Summer Chinook Snake River Fall-Run Chinook Snake River Basin Steelhead. National Marine Fisheries Service, West Coast Region, Portland, OR. www.westcoast.fisheries.noaa.gov/publications/status_reviews/salmon_steelhead/2016/2016_upper-columbia.pdf

NMFS (National Marine Fisheries Service). 2015. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest.

NOAA Module for the Ocean Environment. 2014. Prepared for the NOAA Fisheries, Northwest Regional Office. Prepared by the Northwest Fisheries Science Center, July 2014. www.westcoast.fisheries.noaa.gov/publications/recovery_planning/salmon_steelhead/domains/interior_columbia/snake/ocean_module.pdf

NPCC (Northwest Power and Conservation Council). 2004. Final Methow Subbasin Plan. November 2004, Northwest Power and Conservation Council. www.nwcouncil.org/fw/subbasinplanning/methow/plan

NPCC (Northwest Power and Conservation Council). 2014. Columbia River Basin Fish and Wildlife Program: 2009 amendments. Northwest Power and Conservation Council, Document 2014-12, Portland, Oregon. www.nwcouncil.org/fw/program/2014-12/program. (January 2015).

ODFW/WDFW. 2017. 2017 Joint Staff Report: Stock status and fisheries for spring Chinook, summer Chinook, sockeye, steelhead, and other species. Oregon Department of Fish and Wildlife/Washington Department of Fish and Wildlife, Joint Columbia River Management Staff, November 9, 2017. (wdfw.wa.gov/publications/01941)

O'Neal JS, P Roni, B Crawford, A Ritchie, and A Shelly. 2016. Comparing stream restoration project effectiveness using a programmatic evaluation of salmonid habitat and fish response. North American Journal of Fisheries Management 36:681-703.

Paulsen CM and TR Fisher. 2005. Do habitat actions affect juvenile survival? An information-theoretic approach applied to endangered Snake River Chinook salmon. Transactions of the American Fisheries Society 134:68-85.

Pearsons TN, AR Murdoch, G Mackey, KG Murdoch, TW Hillman, MR Cooper, and JL Miller. 2012. Ecological risk assessment of multiple hatchery programs in the Upper Columbia watershed using Delphi and modeling approaches. Environmental Biology of Fishes 94:87-100.

Pess GR, ML McHenry, TJ Beechie, and J Davies. 2008. Biological impacts of the Elwha River dams and potential salmonid responses to dam removal. Northwest Sci. 82 (Special Issue):72–90.

Petersson E, T Järvi, TNG Steffner, and B Ragnarsson. 1996. The effect of domestication on some life history traits of sea trout and Atlantic salmon. Journal of Fish Biology 48:776-791.

Polivka KM and JR Mihaljevic. 2017. Unpublished manuscript. A mechanistic growth model demonstrates the variable efficacy of in-stream habitat restoration structures for young-of-the-year salmonids.

Polivka KM, EA Steel, and JL Novak. 2015. Juvenile salmon and steelhead occupancy of stream pools treated and not treated with restoration structures, Entiat River, Washington. Canadian Journal of Fisheries and Aquatic Sciences 72:166-174.

Pollock MM, TJ Beechie, JM Wheaton, CE Jordan, N Bouwes, N Weber, C Volk. 2014. Using beaver dams to restore incised stream ecosystems. BioScience 64:279-290. https://doi.org/10.1093/biosci/biu036

Potter H. 2016. Leavenworth National Fish Hatchery Annual Report, 2016. U.S. Fish and Wildlife Service, Leavenworth WA.

Quinn TP. 2005. The behavior and ecology of Pacific salmon and trout. University of Washington Press, Seattle and London.

Rawding D. 2004. Comparison of spawner-recruit data with estimates of Ecosystem Diagnosis and Treatment (EDT) spawner-recruit performance. Vancouver, WA: Washington Department of Fish and Wildlife.

Reeves GH, FH Everest, and TE Nickelson. 1989. Identification of physical habitats limiting the production of coho salmon in western Oregon and Washington. Gen. Tech. Rep. PNW-GTR-245. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 18 p.

Regional Technical Team. 2014. A biological strategy to protect and restore salmonid habitat in the Upper Columbia Region. A Draft Report to the Upper Columbia Salmon Recovery Board.

Reisenbichler RR. 1988. Relation between distance transferred from natal stream and recovery rate for hatchery coho salmon. North American Journal of Fisheries Management 8:172-174.

Reiser DW and RG White. 1988. Effects of two sediment size-classes on survival of steelhead and Chinook salmon eggs. North American Journal of Fisheries Management 8:432-437.

Regional Mark Information System (RMIS). www.rmpc.org

Roegner GC, R McNatt, DJ Teel, and DL Bottom. 2012. Distribution, size, and origin of juvenile Chinook salmon in shallow-water habitats of the lower Columbia River and estuary, 2002–2007, Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 4:450-472.

Roegner GC and DJ Teel. 2014. Density and Condition of Subyearling Chinook Salmon in the lower Columbia River and estuary in relation to water temperature and genetic stock of origin. Transactions of the American Fisheries Society 143:1161-1176.

Roloff GJ and BJ Kernohan. 1999. Evaluating reliability of habitat suitability index models. Wildlife Society Bulletin 27: 973-985.

Roni P, GR Pess, TJ Beechie, and KM Hanson. 2014. Fish-habitat relationships and the effectiveness of habitat restoration. U.S. Dept. Commerce, NOAA Tech. Memo. NMFS-NWFSC-127.

Roni P, K Hanson, and TJ Beechie. 2008. Global review of the physical and biological effectiveness of stream habitat rehabilitation techniques. North American Journal of Fisheries Management. 28(3):856–890.

Roni P, T Beechie, G Pess, and K Hanson. 2015b. Wood placement in river restoration: fact, fiction, and future direction. Canadian Journal of Fisheries and Aquatic Sciences. 72:466-478.

Roni P, P Anders, TJ Beechie, and DJ Kaplowe. 2017. Review of tools for identifying, planning and implementing habitat restoration for Pacific salmon and steelhead. North American Journal of Fisheries Management *In Press*.

Rose KA, S Sable, DL DeAngelis, S Yurek, JC Trexler, W Graf, and DJ Reed. 2015. Proposed best modeling practices for assessing the effects of ecosystem restoration on fish. Ecological Modelling 300:12-29.

Rub AMW, D Huff, BP Sandford, MH Sorel, and RW Zabel. 2017. *In prep.* Interannual variation in survival of endangered salmon through the Columbia River Estuary. (Cited by Sorel et al. 2017).

Sanderson BL, KA Barnas, and M Wargo Rub. 2009. Nonindigenous species of the Pacific Northwest: an overlooked risk to endangered salmon? BioScience 59:245-256.

Saunders WC, NW Weber, N Bouwes, P McHugh, E Buhle, K See, B Lott, C Beasley, J White, S Pandit, P Nelle, T Desgroseillier, K van den Broek, M Armour, M Nahorniak, and C Jordan. 2017. ISEMP/CHaMP Life-cycle models – Entiat, John Day, Lemhi, Habitat Actions. *In* R Zabel, et al. Interior Columbia Basin Life Cycle Modeling, May 27, 2017 Draft Report, National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle.

Sharma R, and TP Quinn. 2012. Linkages between life history type and migration pathways in freshwater and marine environments for Chinook salmon, *Oncorhynchus tshawytscha*. Acta Oecologica 41:1–13.

Shearer KD, P Parkins, B Gadberry. BR Beckman, and P Swanson. 2006. The effects of growth rate/body size and a low lipid diet on the incidence of early sexual maturation in male spring Chinook salmon (*Oncorhynchus tshawytscha*). Aquaculture 252:545-556.

Siegel S. 1956. Nonparametric statistics for the behavioral sciences. McGraw-Hill Book Company, New York, Toronto, and London. 312 pp.

Small MR, K Warheit, C Dean, and A Murdoch. 2007. Genetic monitoring of Methow spring Chinook. Appendix D *in* Murdoch A, C Snow, C Frady, A Repp, M Small, S Blankenship, T Hillman, M Miller, and G Mackey. 2011. Evaluation of the hatchery programs funded by

Douglas County PUD. Report to the Wells HCP Hatchery Committee, East Wenatchee, WA. 352 pp.

Smith CT and R Engle. 2011. Persistent reproductive isolation between sympatric lineages of fall Chinook salmon in the White Salmon River, Washington. Transactions of the American Fisheries Society 140:699-715.

Snow C, C Frady, D Grundy, B Goodman, and A Murdoch. 2016. Monitoring and evaluation of the Wells Hatchery and Methow Hatchery programs: 2015 annual report. Report to Douglas PUD, Grant PUD, Chelan PUD and the Wells and Rocky Reach HCP Hatchery Committees and the Priest Rapids Hatchery Subcommittees, East Wenatchee, WA.

Sorel MH, AM Wargo Rub, and RW Zabel. 2017. Population-specific migration timing affects en route survival of Chinook salmon through a variable lower-river corridor. Chapter 6.a, Interior Columbia Basin Life Cycle Modeling, May 27, 2017 Draft Report, National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle.

Sorel MH, AM Wargo Rub, and RW Zabel. Temporal trends in marine mammal predation of Chinook salmon. Abstract. 2018 Upper Columbia Science Conference. Wenatchee, WA Feb. 23-24, 2018.

SRT (Scientific Review Team). 1998-33. Review of artificial production of anadromous and resident fish in the Columbia River Basin. www.nwcouncil.org/reports/1998/98-33.

Steel EA, P McElhany, NJ Yoder, MD Purser, K Malone, BE Thompson, KA Avery, D Jensen, G Blair, C Busack, MD Bowen, J Hubble, and T Kantz. 2009. Making the best use of modeled data: multiple approaches to sensitivity analysis of a fish-habitat model. Fisheries (Bethesda) 34:330-339.

Steen RP, and TP Quinn. 1999. Egg burial depth by sockeye salmon (*Oncorhynchus nerka*): implications for survival of embryos and natural selection of female body size. Canadian Journal of Zoology 75:836-841.

Stewart GB, HR Bayliss, and et al. 2009. Effectiveness of engineered in-stream structure mitigation measures to increase salmonid abundance: a systematic review. Ecological Applications. 19(4):931–941.

Stewart-Oaten A, WW Murdoch, and KR Parker. 1986. Environmental impact assessment: “pseudoreplication” in time? Ecology 67:929-940.

Tamiz M, D Jones, and C Romero. 1998. Goal programming for decision making: An overview of the current state-of-the-art. European Journal of Operational Research 111.3: 569-581.

Teel DJ, BJ Burke, DR Kuligowski, CA Morgan, and DM Van Doornik. 2015. Genetic identification of Chinook salmon: stock-specific distributions of juveniles along the Washington and Oregon Coasts, Marine and Coastal Fisheries 7:274-300.

Tidwell K, T Van Hevelingen, L Magill, and B van der Leeuw. 2017. Bi-weekly update: pinniped abundance and salmon predation at Bonneville lock and dam. Fisheries Field Unit U.S.

Army Corps of Engineers, Bonneville Lock and Dam Cascade Locks, OR 97014. pweb.crohms.org/tmt/documents/FPOM/2010/Task%20Groups/Task%20Group%20Pinnipeds/20170608-Bonneville%20Pinniped%20update.pdf

Turner RJ and RA Cochrane. 1993. Goals-and-methods matrix: coping with projects with ill-defined goals and/or methods of achieving them. International Journal of Project Management 11.2: 93-102.

UCSRB (Upper Columbia Salmon Recovery Board). 2007. Upper Columbia Spring Chinook Salmon and Steelhead Recovery Plan. Upper Columbia Salmon Recovery Board, Wenatchee, WA.

UCSRB (Upper Columbia Salmon Recovery Board). 2014a. Decision-Maker's Guide to the 2014 Integrated Recovery Program Habitat Report. Upper Columbia Salmon Recovery Board, Wenatchee, WA.

UCSRB (Upper Columbia Salmon Recovery Board). 2014b. Integrated Recovery Program Habitat Report. Upper Columbia Salmon Recovery Board, Wenatchee, WA.

Underwood EC, KR Klausmeyer, SA Morrison, M Bode, and MR Shaw. 2009. Evaluating conservation spending for species return: A retrospective analysis in California. Conservation Letters 2 (3): 130–37.

Venditti DA, KA Apperson, B Barnett, M Belnap, T Copeland, MP Corsi, WT Gross, L Janssen, R Santo, K Tardy, and A Teton. 2015. Idaho Supplementation Studies Project Completion Report 1991 – 2014. Idaho Department of Fish and Game Report 15-18. 150 pp.

Wahle RJ and RE Pearson. 1984. History of artificial propagation of coho salmon, *Oncorhynchus kisutch*, in the mid-Columbia River system. Marine Fisheries Review 46:34-43.

Walters AW, T Copland, and DA Venditti. 2013. The density dilemma: limitations on juvenile production in threatened salmon populations. Ecology of Freshwater Fish 22:508-519.

Waples RS, DJ Teel, JM Myers, and AR Marshall. 2004. Life-history divergence in Chinook salmon: historic contingency and parallel evolution. Evolution 58, 386.

Wargo Rub AM, L Gilbreath, D Teel, B Sandford, and L Charlton. 2014. Estimation of survival and run timing of adult spring/summer Chinook salmon from the Columbia River Estuary to Bonneville Dam: a cooperative effort between NOAA Fisheries and Columbia River commercial fishermen. Presentation of unpublished results to the Northwest Power and Conservation Council, Portland, OR (November 4, 2014). www.nwcouncil.org/media/7148426/f1.pdf.

Washington Department of Ecology. 1995. Wenatchee River watershed initial assessment. Draft, May 1995. Yakima, Washington.

Watanabe M, RM Adams, J Wu, JP Bolte, MM Cox, SL Johnson, and JL Ebersole. 2005. Toward efficient riparian restoration: integrating economic, physical, and biological models. Journal of Environmental Management, 75(2), 93-104.

Weber N, N Bouwes, MM Pollock, C Volk, JM Wheaton, G Wathen, J Wirtz, CE Jordan. 2017. Alteration of stream temperature by natural and artificial beaver dams. PLoS One 12(5): e0176313. doi.org/10.1371/journal.pone.0176313

Weigel DE, PJ Connolly, and et al. 2013. Colonization of Steelhead in a natal stream after barrier removal. Trans. Am. Fish. Soc. 142(4):920–930.

Weitkamp L, DJ Teel, M Liermann, SA Hinton, DM Van Doornik, and PJ Bentley. 2015. Stock-specific size and timing at ocean entry of Columbia River juvenile Chinook Salmon and steelhead: implications for early ocean growth. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 7:370–392.

Weitkamp LA. 2010. Marine distributions of Chinook salmon from the west coast of North America determined by coded wire tag recoveries. Transactions of the American Fisheries Society 139:147-170.

Welch S. 2017. Use of habitat suitability indices for project selection and design in the Upper Columbia. Presentation to the Independent Scientific Advisory Board of the Columbia River Basin on September 15, 2017.

White GC. 2005. Correcting wildlife counts using detection probabilities. Wildlife Research 32:211–216.

White GC. 2008. Closed population estimation models and their extensions in Program MARK. Environmental and Ecological Statistics 15:89–99.

Whiteway SL, PM Biron, A Zimmermann, O Venter, and JWA Grant. 2010. Do instream restoration structures enhance salmonid abundance? A meta-analysis. Canadian Journal of Fisheries and Aquatic Sciences 67(5):831–841.

Wiens JA and KR Parker. 1995. Analyzing the effects of accidental environmental impacts: approaches and assumptions. Ecological Applications 5:1069–1083.

Williams R, J Lichatowich, and M Powell. 2006. Diversity, structure, and status of salmon populations. *In* Return to the River: Restoring Salmon to the Columbia River. Edited by R Williams, 99-171. Elsevier Academic Press, Burlington, MA.

Williamson KS, AR Murdoch, TN Pearsons, EJ Ward, and MJ Ford. 2010. Factors influencing the relative fitness of hatchery and wild spring Chinook salmon (*Oncorhynchus tshawytscha*) in the Wenatchee River, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences 67:1840-1851.

Wilson EO. 1975. Sociobiology: the new synthesis. Belknap Press, Cambridge, Massachusetts.

Withler FC. 1982. Transplanting pacific salmon. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1079. 32 pp.

Wu J, RM Adams, and WG Boggess. 2000. Cumulative effects and optimal targeting of conservation efforts: steelhead trout habitat enhancement in Oregon. American Journal of Agricultural Economics 82.2: 400-413.

Wu J and WG Boggess. 2003. Measuring interactions among urbanization, land use regulations, and public finance. American Journal of Agricultural Economics 85:988-999.

Zabel R et al. 2017. Interior Columbia Basin Life-Cycle Modeling draft (May 23, 2017). NWFSC, Montlake, Seattle WA.

Zabel R and T Cooney. 2013. Appendix C: Recruits-per-Spawner in Base Versus Current Time Periods—Do they differ? In Appendix C in NOAA Fisheries. 2014. Endangered Species Act Section 7(a)(2) Supplemental Biological Opinion. Consultation on remand for operation of the federal Columbia River power system. www.westcoast.fisheries.noaa.gov/fish_passage/fcrps_opinion/federal_columbia_river_power_system.html.