Appendices to the 2020 Columbia River System Biological Opinion

Appendix A – Tributary Habitat Technical Foundation and Analytical Methods

Appendix B – Avian Predation Management

Appendix C – Life-cycle Model Outputs

Appendix A – Tributary Habitat Technical Foundation and Analytical Methods

A.1 Introduction

This appendix summarizes recent information on the scientific basis for understanding how tributary habitat actions can improve salmonid abundance, productivity, spatial structure, and diversity within the Columbia and Snake River basins. It also describes the methods and factors we considered in our analysis of the effects of tributary habitat improvement actions in this biological opinion (opinion). Finally, it discusses important considerations in implementation of tributary habitat improvement actions.

A.2 Scientific Basis for Tributary Habitat Improvement Actions

The National Marine Fisheries Service (NMFS) has determined, based on best available science, that by identifying the factors limiting habitat function, and by strategically implementing actions to alleviate those limiting factors, habitat function will improve and, ultimately, the abundance, productivity, spatial structure, and diversity of salmon and steelhead will improve as well. In most cases, near-term benefits would be expected in abundance and productivity. Depending on the population and the scale and type of action, benefits to spatial structure and diversity might also accrue, either in the near term or over time. In addition, while we expect abundance, productivity, spatial structure, and diversity to respond positively to strategically implemented tributary habitat improvement actions, other factors also influence these viability parameters (e.g., ocean conditions) and any resulting trends. Throughout this discussion, when we discuss improvements in abundance and productivity, it is implied that spatial structure and diversity might also respond positively, and that other factors, as noted here, might influence any resulting trends.

The fundamental relationship between fish, freshwater habitat, and population response provides the basis for implementation of tributary habitat improvement actions, including those implemented under multiple Columbia River System (CRS) biological opinions. This relationship was articulated in Appendix C of the 2007 Comprehensive Analysis (USACE et al. 2007, Appendix C, Attachment C-1 and Annexes 1-3) and reiterated in NMFS' previous CRS biological opinions, including the 2008 biological opinion (NMFS 2008), its 2010 and 2014 supplements (NMFS 2010, 2014), and the 2019 biological opinion (NMFS 2019).[1] Below, we summarize and update the findings and discussion in those documents.

[1] In biological opinions before 2019, the CRS was referred to as the Federal Columbia River Power System, or FCRPS.

A.2.1 2014 Supplemental Biological Opinion

In the 2014 supplemental biological opinion, we described our knowledge of the basic relationships between fish and their tributary habitat, and the findings in the scientific literature about how changes in fish habitat affect fish populations. We evaluated multiple lines of evidence, including literature on the physical and biological effectiveness of improvement actions, correlation analyses, and preliminary results from monitoring in the Columbia River basin designed to evaluate the effects of various actions on tributary habitat limiting factors and on salmon and steelhead population response. We noted that the outcomes of habitat improvement are well documented and support our determination that the strategic implementation of actions to alleviate habitat-related limiting factors will improve habitat function and, ultimately, the freshwater survival of salmon and steelhead.[2] We also noted that long-term studies were underway in the Columbia River basin to further validate and contribute to adaptive management and implementation of tributary habitat improvement actions (NMFS 2014).

We determined in the 2014 supplemental biological opinion that tributary habitat improvement actions have been well documented to provide benefits to fish at the stream-reach scale. We also noted that studies examining changes in salmon and steelhead survival at the population scale were less numerous, in part because directly measuring survival in response to habitat improvement at the watershed scale is complex, costly, and generally requires lengthy periods of action implementation and habitat and fish response monitoring. We found that available studies at the population or watershed scale supported our determination that tributary habitat improvements would lead to improved freshwater survival, as did correlation analyses that examined relationships between habitat improvement actions and fish abundance. We also determined that preliminary results from research, monitoring, and evaluation (RME)

[2] In the 2008 biological opinion and its 2010 and 2014 supplements, we characterized the benefits of tributary habitat improvement actions at the population level primarily in terms of their effect on freshwater survival, either life-stage-specific or total egg-to-smolt survival. We also assumed, based on best available information, that these improvements would carry on to direct improvements in recruits per spawner (R/S) and therefore contribute to achieving metrics, such as R/S > 1, that were used as one part of the analysis in those biological opinions. In the 2019 biological opinion and this current opinion, we characterize the effects of tributary habitat improvement actions at the population level primarily in terms of changes in population abundance, productivity, spatial structure, and diversity. We then qualitatively relate these population-level changes to effects to the species or designated critical habitat. This approach is consistent with our section 7 regulations, which direct NMFS to formulate the agency's biological opinion as to whether a proposed action is likely to: 1) reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing its numbers, reproduction, or distribution; or 2) appreciably diminish the value of designated or proposed critical habitat for the conservation of the species (50 CFR 402.02). The approach is also consistent with our longstanding use of "viable salmonid population" (VSP) parameters (McElhany et al. 2000) to evaluate Pacific salmon and steelhead population viability. The four VSP parameters (abundance, productivity, spatial structure, and diversity) encompass the species' "reproduction, numbers, or distribution," and are commonly used to evaluate long-term risk of extinction and population status relative to Endangered Species Act (ESA) recovery goals. All of these population parameters could affect survival and also mitigate extinction risk by making populations more resilient, and this is why we use these factors to assess the status of populations, which in turn informs the evaluation of species status.

implemented under the 2008 biological opinion appeared to support our determination (NMFS 2014).

In the 2014 supplemental biological opinion, we concluded that the best available scientific literature indicated that many habitat improvement actions (such as increasing instream flow, improving access to blocked habitat, reducing mortality from entrainment at water diversion screens, placing logs and other structures to improve stream structure, and restoring off-channel and floodplain habitat) can improve tributary habitat quantity and quality over relatively short time periods. We also concluded that for other habitat improvements, such as actions that address the source of fine sediment in spawning areas (e.g., road decommissioning) and the restoration of riparian vegetation, it may take decades to realize their full benefit (NMFS 2014).

In addition, we concluded that the best available scientific literature supports the approach of improving tributary habitat to increase survival of salmon and steelhead at the population scale, and we noted that preliminary results from tributary habitat RME conducted under the 2008 biological opinion provided evidence that the Action Agencies' habitat improvements were correctly targeting and improving degraded conditions and providing benefits to fish (NMFS 2014).

A.2.2 Updated Findings on Scientific Basis for Tributary Habitat Improvement Actions

Literature on fish habitat restoration is extensive and has been summarized recently in the context of salmon and steelhead recovery in the Columbia River basin (see Roni et al. 2002, 2008, 2014). More recently, Hillman et al. (2016) conducted a review of published and unpublished literature on the effectiveness of habitat improvements that built upon those earlier reviews. Pess and Jordan (2019) also summarized findings on habitat restoration actions at the site and reach scale. Roni (2018) reviewed studies that specifically examined fish movement in relation to river or floodplain habitat restoration or improvement. Haskell et al. (2019) and Bennett et al. (2016) summarized key findings from 16 intensively monitored watersheds (IMWs) and evaluated IMWs as a method for evaluating the effects of tributary habitat improvement actions, and the Bureau of Reclamation recently completed a report summarizing RME results in the Methow River basin (BOR 2019). In addition, the Independent Scientific Advisory Board (ISAB 2015) evaluated density dependence (the relationship between population density and population growth rate) in the Columbia River basin, including density dependence in tributary habitats, and evaluated research and recovery efforts for Upper Columbia River spring-run Chinook salmon, including questions related to the prioritization and effectiveness of habitat improvement actions for that evolutionarily significant unit (ESU) (ISAB 2018). Below, we briefly describe results from recently published literature and from monitoring and evaluation conducted in conjunction with CRS tributary habitat actions, and present our conclusions regarding how that information supports the basis of the CRS tributary habitat improvement actions.[3]

A.2.2.1 Response at the Stream-Reach Scale

Hillman et al. (2016) searched for literature not reviewed in the earlier efforts by Roni et al. (2002, 2008, 2014) and identified papers that provided quantitative information on physical and biological response to habitat improvement. They provided an annotated bibliography for each paper they reviewed, summarizing key findings and reporting quantitative changes in physical and biological parameters, including changes in fish habitat and fish abundance where available. They summarized the results for major categories and subcategories of improvement actions. Below, we summarize these findings with extreme brevity and provide a few anecdotal examples from recent monitoring for illustrative purposes; readers are referred to Hillman et al. (2016), Pess and Jordan (2019), Roni et al. (2002, 2008, 2014, 2018), and specific monitoring reports and publications for more detail.

Barrier removal: Habitat conditions have been shown to respond quickly to barrier removal, and positive effects are usually long term or even permanent. Reviews of the effectiveness of habitat improvements have consistently reported removal of barriers or installation of fish passage as one of the most effective methods for increasing fish numbers (Roni et al. 2014, Hillman et al. 2016, Pess and Jordan 2019). Examples abound, and we include just a few here for illustration:

- In many years, low flows and obstructions blocked Loup Loup Creek, the southernmost tributary of the Okanogan River, making it impassable to fish trying to reach habitat in the creek's upper reaches. Agreements on water use, removal of culverts, and alteration of another barrier to improve fish passage reopened the creek in 2010. Increasing numbers of juvenile steelhead have been documented by annual snorkel surveys in the creek, and adult steelhead are also returning, with an average of 22 adult steelhead returning to the creek each year from 2012 to 2016 (OBMEP 2018).

[3] Under the 2008 biological opinion and the Adaptive Management Implementation Plan (adopted as part of the 2008 biological opinion and its 2010 supplement), the Action Agencies implemented an extensive tributary habitat monitoring program (under Reasonable and Prudent Alternative (RPA) Actions 56 and 57), paired with fish population status monitoring (under RPA Action 50), to define the benefits of habitat improvements (NMFS 2008, 2010, 2014). This RME program was part of an adaptive management approach designed to inform and shape future habitat actions so they deliver increasingly meaningful and cost-effective results (BPA and BOR 2013). The program was described briefly in the 2014 supplemental biological opinion (and in more detail in BPA et al. [2013], BPA [2013], and BPA and BOR [2013]). Hillman et al. (2016), Bennett et al. (2016), and Haskell et al. (2019) incorporated recent results from that RME program. The Action Agencies continued an RME program, although at a reduced level, under the 2019 biological opinion (NMFS 2019), and they propose to continue, under their 2020 proposed action (BPA et al. 2020, USACE 2020), to implement a tributary RME program and to engage in a collaborative process to develop and implement a Columbia River basin tributary habitat RME strategy.

- Evaluations of culvert removal projects in Washington State, including two sites in the Columbia River basin, have indicated increases in juvenile coho salmon numbers within 2 years of culvert removal or replacement (O'Neal et al. 2016).
- In an evaluation of fish numbers above and below former impassable culverts at 32 sites in the interior Columbia River basin, no differences were detected in numbers of steelhead or other salmonids above and below the formerly impassable barriers (Hillman et al. 2016). This suggests that culvert replacement has been effective in allowing juvenile salmonids access to formerly blocked habitat.

Instream structures: Actions to improve stream complexity include the placement of structures such as logs, logjams, cover structures, and boulders, and the addition of gravel. Most published literature on placement of instream structures is related to placement of large wood, and the vast majority of these studies show a positive response for habitat and salmonid fishes (Hillman et al. 2016, Pess and Jordan 2019). The increase in fish abundance in improved habitats was typically related to an increase in habitat capacity, and not due to a redistribution of fish from other habitats of the same stream reach (Polivka et al. 2015, Roni 2018). The lack of response or small decrease in abundance reported in some studies appears to be largely because watershed processes (e.g., sediment, water quality, etc.) were not addressed, because monitoring had not occurred long enough to show results, or because treatments had resulted in little change in physical habitat. Although more research specific to these action types for Chinook salmon and steelhead in the Columbia River basin is needed, available studies show that the effects of instream structures are generally rapid (1 to 3 years), often occurring during the first high-flow event (Hillman et al. 2016).

Several studies (Clark et al. 2017, 2019) evaluated projects throughout the Columbia River basin that involved adding large wood to streams. The evaluations included snorkel surveys to examine fish numbers in sites where debris had been added as part of restoration, compared to numbers from surveys at control sites where no improvements had been made. Clark et al. (2017) found nearly double the density of juvenile steelhead in streams with wood structures compared to those without. The improvement in fish numbers was consistent among various sites and watersheds. In addition, the restored reaches included more pools and larger pools, signaling that the woody debris helps add needed complexity to rivers and streams by altering river flows that shape and scour the streambeds. Clark et al. (2019) found that the proportion of pool area and the amount of pools, large wood, and pool-forming large wood were significantly higher in treated reaches than in control reaches. Juvenile Chinook salmon, coho salmon, and steelhead abundances were also significantly higher in treated reaches than in control reaches.

In three tributaries of Asotin Creek, a tributary of the Snake River in southeast Washington, from 2012 to 2016, scientists installed more than 650 log structures made of wood debris held in place by log piles driven into the stream bottom in an effort to add complexity to the stream and provide habitat for fish. Early monitoring documented a 28.8 percent increase in juvenile steelhead abundance in areas with the wood debris compared to those without, and modeling suggests that the carrying capacity of the streams has increased by 50 percent following addition of the debris. Initial results also suggest that the productivity of fish populations may be increasing, as reflected by the number of surviving smolts per female spawning in the research stretches of streams. Researchers will continue tracking the number of smolts per female to determine whether the increases continue in the long term (Griswold and Phillips 2018).

One relatively new type of instream habitat improvement with promising results is the addition of structures to mimic the hydro-geomorphic effects of beaver dams. The importance of beaver dams for creating habitat is well documented. “Beaver support structures” or “beaver dam analogs” can have similar effects on stream velocity, surface water level and routes, ground water level, sediment sorting, water table, and riparian vegetation. Results from one such project in Bridge Creek, in the John Day River basin, have shown that beaver dam analogs can lead to aggradation of incised channels, increased side channels, increased floodplain area, and increased groundwater levels, as well as to construction of actual beaver dams (Pollock et al. 2012, DeVries et al. 2012, Bouwes et al. 2016b). Results from this study in Bridge Creek also show significant increases in the density, survival, and production of steelhead following construction of beaver dam analogs (Bouwes et al. 2016b). Studies outside the Columbia River basin suggest that instream structures can help to restore human-impacted river ecosystems, primarily through altering the abundance and biomass of consumers and resources in the food web (Thompson et al. 2017).

Floodplain habitat reconnection: Studies on the effectiveness of floodplain habitat reconnection have consistently shown rapid recolonization of newly accessible habitat by salmonids and other fishes, and fish rearing in such habitats can have higher growth rates than those rearing in the mainstem. Success of these projects depends on their connection with the main channel and their morphology and depth, as well as on addressing water quality and other upstream problems, although more monitoring of such projects in the Columbia River basin is needed (Hillman et al. 2016). Examples with positive results include remediation and reconnection of former mining dredge ponds to the Yankee Fork of the Salmon River in central Idaho, where the reconnected habitats quickly became home to juvenile salmon and spawning steelhead (Bellmore et al. 2012). Also, improvements coordinated by the Confederated Tribes of the Umatilla Indian Reservation on Catherine Creek in northeast Oregon increased the habitat capacity for juvenile salmon by roughly two to eight times in terms of usable area for fish, depending on the time of year. Biologists recorded immediate improvements in favorable habitat measures, such as the frequency of slow-water pools and the amount of large woody debris (CTUIR 2017). The floodplain of the upper Chilliwack River watershed, a tributary to the Fraser River in British Columbia, was extensively restored from 1996 to 2000 through off-channel habitat restoration. Researchers estimated that 27 to 34 percent of the total production of out-migrating coho salmon smolts in the watershed could be attributed to the newly created habitat (Ogston et al. 2014). Floodplain reconnections can also improve hyporheic flow and the ecosystem processes that allow restoration of habitat and water quality to occur. Singh et al. (2018) showed that levee setbacks in a reach of the Yakima River improved the hyporheic connection between surface and groundwater in the Yakima floodplain, which demonstrates that levee setback can provide a valuable hydrologic tool to restore ecosystem processes in previously leveed rivers.

Modeling efforts for the Columbia River basin suggest that there has been an estimated 26 percent decrease basinwide in floodplain channel area from historical conditions, and that reconnecting historical floodplains currently used for agriculture could increase side-channel habitat by 25 percent and spring Chinook salmon parr rearing capacity by 9 percent over current estimates (Bond et al. 2019). While individual watersheds throughout the Columbia River basin vary greatly in habitat factors that limit salmon recovery, large-scale estimates of restoration potential like this one are useful in making decisions about long-term restoration goals (Bond et al. 2019).

Riparian planting: Riparian habitat improvement through riparian planting and silvicultural treatment, invasive species control, and riparian fencing and grazing management can lead to increased shade, improved bank stability, reduction of fine sediment, reduced temperatures, and improvement of water quality (Roni et al. 2014, Hillman et al. 2016, NMFS 2017b). While some benefits of riparian planting begin to accrue after 10 to 15 years (Justice et al. 2017, Pess and Jordan 2019), the full effects of riparian plantings on habitat conditions can take more than 50 years to accrue, in part because of the long lag time between tree growth and any change in channel conditions, delivery of large wood, and shading effects on temperature. As a result, few studies have examined the response of instream habitat or fish to riparian planting or thinning. One retrospective study (Lennox et al. 2011) found that project age was positively correlated with both riparian vegetation and fish habitat at enhanced sites. Riparian enhancement actions are also often critical for success of other enhancement actions (e.g., floodplain, instream) (Hillman et al. 2016). Justice et al. (2017) found through modeling that a combination of riparian restoration and channel narrowing could both reduce stream temperatures and increase the abundance of Chinook salmon parr in the Upper Grande Ronde River and Catherine Creek in northeast Oregon. They utilized a stream temperature model to explore potential benefits of channel and riparian restoration on stream temperatures in the Upper Grande Ronde River and Catherine Creek basins, and concluded that restoration of such streams could more than make up for an expected increase in summer stream temperature through 2080.

Livestock exclusions: Monitoring of livestock exclusions to improve riparian areas shows that habitat response can occur relatively rapidly (<5 years), but studies of fish response have been variable. Lack of fish response has been attributed to short duration of monitoring, small size of exclosures, and the influence of upstream processes (Hillman et al. 2016). In one example, researchers are tracking the condition of seven livestock exclusion projects across the Columbia River basin, although only two of the sites include observations from before the livestock exclusion projects for comparison. Where data are available, comparisons show a reduction in erosion and a slight increase in canopy cover, which matches broader findings on the relationship between grazing and riparian health. In a study of 261 grazed and ungrazed watersheds, those without grazing impacts demonstrated more stable banks, deeper pools, and reduced amounts of fine sediment. Researchers expect to see continued improvements in riparian conditions in areas where grazing impacts have been controlled, although the full response may take several years (O'Neal et al. 2017).

Reduction of excess fine sediment: Reduction of excess fine sediment is usually accomplished through road enhancement, agricultural treatments, and riparian enhancement. Actions such as road decommissioning, removal, and upgrading are successful in decreasing fine sediment delivery to streams. Little monitoring and evaluation has been done to examine the response of fish or other biota to road treatments (Hillman et al. 2016).

Flow augmentation: Reduced flows can affect adult and juvenile salmonids by blocking fish migration, stranding fish, reducing rearing habitat availability, and increasing summer water temperatures (NMFS 2017b). Flow protection or enhancement reduces these impacts, and the literature shows an obvious and clear relationship between increased flows and increases in fish and macroinvertebrate production (Hillman et al. 2016). The response is most dramatic in stream reaches that were previously dewatered or too warm to support fish because of water withdrawals, and the most successful projects are those in which acquired flows are held in trust long term or in perpetuity (Hillman et al. 2016). Studies of fish movements following increases in instream flows show rapid fish colonization of newly accessible habitats and illustrate the success of these projects (Roni et al. 2008). For example, ongoing studies in the Lemhi River basin show increased spawner and juvenile fish numbers following enhancement of instream flows in tributaries (Uthe et al. 2017, Appendix A of Griswold and Phillips 2018). Similarly, re-watering a previously dewatered reach of the Bridge River in British Columbia led to increases in juvenile Pacific salmon and riparian plant growth following enhancement of instream flows (Hall et al. 2011; Bradford et al. 2011, cited in Hillman et al. 2016). The effects of flow augmentation on habitat conditions depend on the amount of flow within the channel, how much water is added, and how long it remains in the stream (e.g., is the water removed downstream? is the augmentation perpetual or for a limited time of year or number of years?). Augmented flow in dewatered channels or in streams too warm to support fish will have the greatest effects on habitat condition. In addition, augmenting flood flows can improve floodplain connectivity and off-channel conditions (Hillman et al. 2016).

Nutrient enhancement: Hillman et al. (2016) did a comprehensive review of literature on the topic of nutrient enhancement. They concluded that while additional study is needed, available studies suggest that nutrient enrichment through the addition of inorganic nutrients, salmon carcasses, or an increase in spawning fish can increase primary and secondary production and fish growth and, possibly, survival of salmonids in oligotrophic streams.

Acquisition and protection: Protection of high-quality riparian/floodplain habitat (e.g., through conservation easements and acquisitions) helps to maintain riparian vegetation, reduce delivery of sediments and pollutants to streams, and maintain bank stability and water quality. The most favorable responses come from protecting large areas of streamside habitat in perpetuity, addressing upstream processes that negatively affect downstream habitat conditions, and regulating/managing activities within the protected areas (Hillman et al. 2016).

A.2.2.1.1 Summary: Response at the Stream-Reach Scale

Extensive literature continues to document benefits to habitat and fish at the stream-reach scale as a result of habitat improvements. Barrier removal and the placement of large woody debris and other types of instream structures are known to improve instream habitat and increase numbers of trout and juvenile coho salmon and steelhead. Various floodplain habitat improvement and enhancement techniques also show positive responses.

Relatively little work has been done to examine the physical and biological response in streams from riparian planting, flow augmentation, sediment reduction (road removal), or acquisition and protection. Additional monitoring or focused studies examining the effects of these methods, and their cost-effectiveness, would be beneficial. Additional studies of instream structures and floodplain reconnection specific to Chinook salmon in the Columbia River basin may also be warranted; many studies of those techniques have been done in coastal coho streams, but the existing body of literature provides confidence in the effects of those actions.

Addition of inorganic or organic nutrients or salmon carcasses has been shown to increase primary and secondary production and fish growth, although few studies have documented increased fish numbers. Moreover, studies to date in the Columbia River basin have not shown an increase in fish numbers. Thus, these techniques should still be considered experimental and in need of additional evaluation.

In studies where no response to habitat improvement actions has been shown, it appears to be largely because watershed processes (e.g., sediment, water quality, etc.) were not also addressed, because monitoring had not occurred for long enough to show results, or because treatments had resulted in little change in physical habitat (Hillman et al. 2016). These findings highlight factors important to the success of habitat improvement actions: ensuring that upstream and watershed processes (such as sediment and water quality) have been addressed, understanding what factors are limiting fish production, and ensuring that the total amount or extent of treatment is adequate.

Roni (2018) reviewed literature that informed a key uncertainty about the effects of tributary habitat improvement: whether fish concentrate around restoration projects (i.e., whether fish move into restored habitat but without an increase in their total abundance) or whether restoration actually increases total abundance. Based on his review, he concluded that existing literature provides little evidence to support the view that river restoration leads to concentration of fish at restoration sites. Instead, he found that the literature does suggest that restoration may lead to increased survival, increased abundance, or both. Roni (2018) notes that the scientific literature suggests that fish response to restoration varies greatly depending on the watershed template, location and characteristics of the habitat restoration, and the life history and limiting factors for the species being addressed.

A.2.2.2 Response at the Population/Watershed Scale

Population- or watershed-scale monitoring projects are rare compared to reach-scale monitoring, because population-scale monitoring is challenging and costly, requiring robust large-scale monitoring and implementation designs; long-term monitoring, coordination, and funding commitments; and large and extensive treatments. However, a number of IMWs have been established throughout the Columbia River basin and the Pacific Northwest.

IMWs are large-scale experiments with well-developed, long-term monitoring designed to determine population/watershed-scale fish and habitat responses to enhancement actions. Although most IMWs would not be expected to show full results yet, a few have demonstrated fish responses at the watershed scale. Ten have detected improvements in juvenile fish metrics, and four have documented significant increases in adult salmon abundance. Adult increases were demonstrated primarily in IMWs that included removals of dams or barriers that opened new areas for colonization. Some IMWs have shown little or no response even after intense treatment and monitoring (e.g., Tenmile and Fish Creeks). This is believed to be because broader watershed-scale factors, such as floods and road failures, limited the success of the restoration actions implemented, or, in some cases, because design and procedural issues during monitoring limited the ability of the IMW to detect responses to enhancement.

Below, we highlight notable results from some of the longer-running IMWs in the Columbia River basin. Because enhancement actions have generally not been implemented for a long enough period for habitat and fish populations to respond and to allow full evaluation, most IMWs are several years away from definitive conclusions regarding enhancement effectiveness. Several summaries of results to date have been completed (Roni et al. 2014; Hillman et al. 2016, 2019; Bennett et al. 2016; Griswold and Phillips 2018; Haskell et al. 2019), and readers are referred to those summaries, as well as to publications cited therein and below regarding specific IMWs for additional detail.

Asotin Creek IMW: The goal of the Asotin Creek IMW is to test the effectiveness of adding large wood to increase habitat complexity and steelhead production. Researchers added large wood to treatment sections and compared the treated sections to control sections. Although monitoring is still ongoing, initial results show significant improvements in habitat complexity. The frequency of large wood has increased by 185 percent in treated sections compared to control sections, and the structures are creating hydraulic and geomorphic responses. It is too early to evaluate changes in steelhead production, but researchers have documented a significant (250 percent) increase in juvenile abundance in treatment reaches compared to control reaches. Practitioners also report increases in juvenile growth, survival, and productivity, as well as in numbers of adults and redds (Haskell et al. 2019). The remainder of the study will focus on estimating changes in productivity and other life-history characteristics of steelhead, and identifying the causal mechanisms of changes (Bouwes et al. 2016a).

Bridge Creek IMW: The Bridge Creek IMW is designed to test whether constructing beaver dam analogs to encourage natural beaver dam development can improve habitat in Bridge Creek, a deeply incised stream in the John Day River basin that has limited riparian vegetation and poor habitat complexity and quality. The hypothesis is that the analogs and beavers will aggrade the channel and thereby alter hydrology, temperature, geomorphology, and vegetation to improve habitat conditions for steelhead. Researchers saturated four reaches on Bridge Creek with beaver dam analogs and identified control reaches. Monitoring occurred three years before treatment (2007 to 2009) and four years after treatment (2010 to 2013). In 2013, researchers counted 236 beaver dams in Bridge Creek. About half of these were made by beavers; the others were functioning beaver dam analogs (overall an eight-fold increase over the pre-treatment conditions). Treated reaches had higher water levels and deeper pools, lower water temperatures, large upstream dam pools and downstream plunge pools, and large increases in inundation area, thermal refugia, and side channels. The beaver complexes also created greater variability in water depths, channel widths, and temperatures, indicating an increase in habitat complexity. These changes translated into changes in fish density, density-dependent decreases in growth, and increases in juvenile survival. Four years following treatment, juvenile production had increased in Bridge Creek by 175 percent relative to the control. The treatments had no negative effects on upstream or downstream migration of juvenile or adult steelhead (ISEMP/CHaMP 2015, 2016; Bouwes et al. 2016b).

Entiat River IMW: Researchers and stakeholders determined that reduced instream complexity was the primary concern limiting Chinook salmon and steelhead production in the lower 26 miles of the Entiat River. Current land uses (primarily agriculture, roads, and residential development) restrict habitat improvement options in this portion of the river, so an engineered approach is being used to increase complexity, including adding rocks and large wood to the river, and reconnecting the floodplain by breaching levees where possible. After completion of two of four planned rounds of habitat actions, affecting about 14 percent of the targeted stretch of river, habitat monitoring showed a significant increase in the volume of wood in the Entiat River (ISEMP/CHaMP 2015, 2016), and fish were using treated areas on a seasonal basis at a fine scale (R.D. Nelle, U.S. Fish and Wildlife Service, personal communication, cited in Hillman et al. 2016). Polivka et al. (2015) also found that both Chinook salmon and steelhead were more abundant in improved pools than in untreated pools in early summer, but this difference was mostly absent by September. Polivka and Claeson (2020) surveyed reaches of the Entiat River treated with engineered logjams and reaches without treatments to determine if restoration had increased habitat capacity. They found that the density of juvenile Chinook salmon and steelhead was 3.1 and 2.7 times greater, respectively, in treated habitat compared to untreated habitat. To distinguish whether these density differences were actual increases in capacity rather than fish moving from poor habitat to good habitat, they compared density in unrestored habitat in both treated and untreated reaches. They found no differences for either species, confirming that the increased density in restored habitat units did not come from depletion of unrestored habitat in the same reach. Thus, they concluded that the restoration had increased the habitat capacity of the reach at the scale of pools created by engineered logjams.

The Entiat River has not yet experienced the high post-treatment flows needed to affect channel morphology as hypothesized. Furthermore, the enhancement plan is not yet complete. Whether the enhancement plan can be implemented as originally designed is questionable because landowner constraints currently limit the completion of the implementation plan. The Entiat IMW illustrates many of the challenges of implementing enhancement actions under a structured monitoring design (Hillman et al. 2016).

Lemhi River IMW: In the Lemhi River, stakeholders and researchers determined that insufficient instream flow, loss of access to historically important habitat, and simplification of mainstem habitat were the primary ecological concerns for Chinook salmon and steelhead productivity (ISEMP/CHaMP 2015, 2016). Researchers developed a plan to remove or reduce migration barriers, maintain or enhance riparian conditions, decrease fine sediment and temperatures, increase tributary connections, and improve habitat quality. Twenty-two types of habitat improvement actions were planned in high-priority watersheds. Tributary water diversions have been replaced with mainstem diversions, allowing tributaries to be reconnected to the mainstem, reducing total water withdrawals, and allowing cooler tributary water to enter the mainstem Lemhi River. In addition, tributary passage conditions have been improved, providing access to relatively intact public lands. Fish and habitat monitoring are underway to detect life-stage-specific responses to individual habitat actions and the accumulated effects of multiple actions at the population scale.

The reconnection of tributaries to the Lemhi River nearly doubled the length of stream available to Chinook salmon and steelhead (ISEMP/CHaMP 2016). Minimum instream flow agreements have addressed passage impediments and reduced temperatures in the upper mainstem Lemhi River. Overall, restoration has resulted in a 22 percent increase in wetted stream area and a 19 percent increase in pool habitat compared to pre-treatment conditions. Adult steelhead have moved into each of the five reconnected tributaries, and these tributaries are producing anadromous juveniles. Researchers have also documented the presence of adult Chinook salmon in two of the five reconnected tributaries, and juvenile Chinook salmon in all reconnected tributaries (Hillman et al. 2016, Haskell et al. 2019). This is the first occurrence of juvenile salmon in four of the five tributaries since the mid-2000s. The IMW team has reported an increase in juvenile Chinook salmon productivity (Uthe et al. 2017, Haskell et al. 2019). Overall, work in the Lemhi River basin has increased the summer rearing capacity for parr by 62 percent. Monitoring information and modeling results now indicate that juvenile Chinook salmon rearing habitat, particularly winter habitat, is currently limiting in the lower Lemhi River. As a result, habitat improvement efforts have shifted to improve habitat in the lower Lemhi River (Hillman et al. 2016).

Fish Creek IMW: Fish Creek, a tributary to the Clackamas River in Oregon, was one of the earliest IMWs. The goals of enhancement were to increase the amount of pool habitat for summer and winter rearing and the amount of habitat for anadromous salmonids (Chinook and coho salmon and steelhead). Intensive monitoring of enhancement activities began in 1982 and continued through 1995. Some preliminary enhancement activities occurred in 1983, but most work (large wood placement, off-channel pond construction) occurred from 1986 to 1988. This included placement of 500 large wood structures covering much of the anadromous zone of Fish Creek. Despite intensive monitoring of habitat and numbers of parr, smolts, and adults, significant changes in fish numbers were not detected after enhancement. There were rapid increases in pool habitat following placement of instream structures, but no significant increases in coho salmon or steelhead parr or smolts were detected. Chinook salmon were present only during the initial years of the study, and their response to enhancement could not be examined.

Floods in the winter of 1995 to 1996 damaged or destroyed many of the instream structures, and road failures and other broader watershed-scale factors and processes following enhancement appear to have limited the success of the habitat actions. These results highlighted the need for 1) addressing watershed-scale processes, 2) having a control watershed, and 3) not relying solely on statistical significance to determine fish response to enhancement (Reeves et al. 1997, cited in Hillman et al. 2016).

Methow River IMW: In the Methow River basin, analysis indicated that insufficient instream flows, floodplain connectivity, and off-channel habitat; fish passage barriers; high levels of fine sediments; and degraded riparian conditions limited salmonid productivity. As a result, more than 120 improvement and protection projects have been implemented within the basin since 1999. Actions include augmenting flow, screening of water withdrawals, improving fish passage, reconnecting floodplains and side channels, improving riparian habitat, and placing of instream structures. A collection of studies designed to examine different reach-scale enhancement measures and limiting factors was carried out (see, e.g., Bellmore et al. [2013], Benjamin and Bellmore [2016], Bellmore et al. [2017], and Martens and Connolly [2014]). Results were summarized in a report by the Bureau of Reclamation (BOR 2019) and include the following:

- Chinook salmon and steelhead out-migrant abundances showed high variability across years. Spring Chinook salmon out-migrant abundances appeared to be trending upward over time, but steelhead did not show a strong trend. Egg-to-smolt survival increased for spring Chinook in the Twisp River (BOR 2019).
- Increasing the hydrologic connectivity between off-channel habitat and the mainstem increased use by target species (BOR 2019).
- A greater percentage of invertebrate food sources were consumed by salmonids in side channels compared to the main channel. Increasing connectivity may increase fish production because of the abundant food resources frequently found in side channels (BOR 2019).
- Hyporheic upwelling moderates surface-water temperatures and can increase fish production. Placing side channels in areas likely to receive upwelling, such as the inside of a meander bend, has been shown to increase groundwater connectivity (BOR 2019).
- Side-channel enhancement projects with sufficiently deep pools and large wood improved habitat suitability and carrying capacity. Chinook salmon and steelhead carrying capacity of reconnected side channels was 251 percent higher, on average, than in the main channel (based on food availability) (Bellmore et al. 2013), and densities (but not survival) increased for both Chinook salmon and steelhead following habitat action implementation. The side channels also provide escape cover for juvenile Chinook salmon and steelhead from predatory fish species (Haskell et al. 2019). Target species densities in the mainstem and side-channels of the Methow River were positively associated with deep pools with large wood and overhead cover (BOR 2019).

- On the Entiat River, early-summer rearing density of juvenile Chinook and steelhead was approximately doubled at placed large wood structures compared to untreated areas (BOR 2019).
- Channel reconstruction and large wood enhancement in a small stream can increase spawning densities, total fish production, and the consumption of invertebrate food resources (BOR 2019).

Potlatch Creek IMW: In the Potlatch Creek basin, research indicated that low flows and dewatering were the primary factors affecting steelhead production in the lower basin, while a lack of habitat complexity was limiting steelhead production in the upper basin (Bowersox and Biggs 2012, Heekin 2013). In the lower basin (Big Bear Creek), low flows and dewatering are being addressed by removing fish passage barriers to open inaccessible habitat and developing water-release strategies from headwater reservoirs. In the upper basin (East Fork Potlatch River), habitat enhancement actions, including woody debris treatments and meadow rehabilitation, are being implemented to improve habitat complexity. After about 25 percent of planned treatments had been completed, preliminary results suggest that juvenile steelhead densities were greater within treatment reaches than in control reaches in the upper basin, and steelhead redds had been found above the site of the Dutch Flat Dam removal in the lower watershed. These and other fish and habitat responses indicate the potential for future population-level responses (Uthe et al. 2017, Haskell et al. 2019).

Wind River IMW: In the Wind River basin, researchers and stakeholders identified impaired fish passage, reduced abundance of instream woody debris, increased sedimentation and scour, and reduced channel stability and habitat complexity as the primary concerns limiting Chinook salmon and steelhead production (Coffin 2014, Buehrens and Cochran 2015). A collaborative enhancement and monitoring program initiated in the 1990s included the removal of Hemlock Dam on Trout Creek in 2009 and Martha Creek Dam in 2012, as well as the decommissioning of roads, addition of woody debris, removal of invasive plant species, enhancement of riparian areas, and improvement of fish passage at road crossings. Increases in steelhead adults and smolt density have been documented in Trout Creek (treated watershed) relative to the Wind River (untreated, reference watershed). For example, adult returns increased from 77 spawners in Trout Creek (pre-treatment) to 208 spawners in 2017 (after treatment), and smolt density increased 29 percent in Trout Creek (treatment site), while in the Wind River (reference site) it decreased 7.4 percent (Haskell et al. 2019).

A.2.2.2.1 Summary: Response at Population/Watershed Scale

Although the population/watershed-scale effectiveness monitoring projects are in varying stages of completeness, some are demonstrating habitat and fish responses. The most immediate responses have occurred where barriers were removed, resulting in increased spawning distributions of salmon and steelhead and increased juvenile life-history diversity. Projects that improved floodplain and side-channel connectivity have also shown significant benefits. For example, the use of beaver dam analogs and beavers to reconnect floodplain habitat and reduce channel incision have shown large improvements in juvenile steelhead abundance, survival, and production. Reconnecting side channels in the Methow River basin increased habitat area and fish capacity within treated reaches. Instream placement of large wood has, in general, increased habitat diversity by increasing pools and side channels, which has resulted in an increase in juvenile fish density and survival and, in some cases, reduced fish growth. At this time, nutrient enhancement has not been fully evaluated.

Researchers have noted both the utility and limitations of IMWs for evaluating population and watershed-scale responses (Bennett et al. 2016, Griswold and Phillips 2018, Haskell et al. 2019), and have concluded that successful IMWs appear to have the following characteristics: 1) implementers conduct watershed assessments and/or use modeling to identify problems and limiting factors (ecological concerns) within the watersheds before developing an enhancement plan; 2) implementers work with stakeholders and landowners to prioritize and sequence appropriate enhancement actions; 3) implementers use robust experimental designs and implement enhancement and monitoring plans within an adaptive management framework; 4) a large percentage of the watershed is improved; 5) projects are set up to identify causal mechanisms; 6) there is a commitment to long-term monitoring and funding (>10 years); and 7) enhancement, monitoring, funding, and implementation entities are well coordinated (Hillman et al. 2016). Factors that continue to make implementing IMWs a challenge include: lack of ability to control other management activities, coordination of enhancement activities and monitoring across multiple organizations, and funding (Roni et al. 2015). Excellent coordination among the various entities and stakeholders is needed to help maintain suitable control streams. Several authors of recent retrospective reports have highlighted the importance of coordination and communication between restoration action implementation programs and monitoring programs to ensure the proper placement and design of actions, as well as the potential of detecting results (Bennett et al. 2016, Hillman et al. 2016, Haskell et al. 2019). The majority of the region's IMWs have documented positive results from habitat implementation actions to either habitat parameters, fish parameters, or both. Some IMWs have not documented conclusive results, but this is due in large part to the long time periods necessary to affect habitat change and subsequent fish response (Haskell et al. 2019)

Finally, as noted by Chapman (1996), Reeves et al. (1997), and others (cited in Hillman et al. 2016), maintaining control streams is an important element of IMWs. Finding control streams is difficult, and there is no guarantee that control streams will remain suitable throughout the life of the project.

A.2.2.3 Density Dependence

The productivity of fish populations is density dependent, meaning that the productivity of a population declines as the density of fish in a habitat increases.[4] The productivity of a population

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[4] Productivity is used as an indicator of a population's ability to sustain itself or its ability to rebound from low numbers. The terms "population growth rate" and "population productivity" are interchangeable when referring to measures of population productivity over an entire life cycle. The indicator for productivity is the average number of surviving offspring per parent, which can be expressed as the number of recruits (adults) per spawner or the number of smolts per spawner.

will be lowest when a particular habitat is at capacity. At this point, further increases in abundance will not result in higher productivity (i.e., in more fish surviving to the next life stage), and in some cases increased abundance at this point could result in declines in productivity. For example, in freshwater habitats, as the density of smolts increases, increased competition for limited resources (e.g., food and shelter) drives survival down (or drives movement of fish to different habitats if available). In addition to a population being limited by the quantity or quality of a particular type of habitat (e.g., juvenile rearing habitat), the spatial patterns of habitat may also be limiting. Spawning and rearing habitat need to be in close enough proximity to each other for the fish to utilize them (Falke et al 2013).

The ISAB examined the question of density dependence and determined that "density effects on smolt production are now strongly evident at spawning abundances that are low relative to historical levels, implying that existing freshwater habitat is constraining the maximum sustainable size of the population" (ISAB 2015). The ISAB noted that dams and other development had limited fish to two-thirds of their historical habitat, and much of the habitat they could reach was degraded and could not support as many fish as it once did. The evidence of density dependence "suggests that habitat capacity has been greatly diminished," the ISAB concluded (ISAB 2015). The loss of habitat, "continuing changes to environmental conditions stemming from climate change, chemicals, and intensified land use appear to have further diminished the capacity of habitat that remains accessible" (ISAB 2015). The ISAB found that "the overall implication is that total adult returns of naturally spawning and hatchery fish may now be exceeding the carrying capacity of some areas of the Columbia Basin and its estuary" (ISAB 2015). In this case, improvements in tributary habitat capacity or productivity, if targeted at limiting life stage and limiting factors, would be likely to improve overall population abundance and productivity by removing a bottleneck on population growth.

A.2.2.4 Climate Resilience and Tributary Habitat Actions

Climate change is expected to adversely affect tributary habitat conditions for Columbia River basin salmon and steelhead (Climate Impacts Group 2004, Scheuerell and Williams 2005, Zabel et al. 2006, ISAB 2007). Likely changes have been characterized generally by the ISAB as follows:

- Warmer air temperatures will result in diminished snowpack and a shift to more winter/spring rain and runoff, rather than snow being stored until the spring/summer melt season.
- Lower snowpack will mean that watershed runoff decreases earlier in the season, resulting in lower stream flows in June through September. Peak river flows, and river flows in general, are likely to increase during the winter due to more precipitation falling as rain rather than snow.
- Water temperatures will rise, especially during the summer months, when lower stream flows co-occur with warmer air temperatures.

These changes will not be spatially homogeneous across the Pacific Northwest. Effects are difficult to predict, and some species are expected to be more vulnerable than others (Crozier et al. 2008, 2019, Waples et al. 2009, Lynch et al. 2016). Stream temperatures are expected to increase in most rivers, but the effect is expected to be greater where temperatures are already near the lethal or sub-lethal thresholds for salmon and steelhead, and lower in rivers where current temperatures are well below these thresholds (Beechie et al. 2013). Some rivers are expected to see large increases in peak flows, whereas others are expected to experience decreased low flows (Arnell 1999, Mantua et al. 2010). River flow is already becoming more variable in many rivers and is believed to negatively affect anadromous fish survival (Ward et al. 2015). Changes in stream temperature and flow regimes are also likely to lead to shifts in the distributions of native species and facilitate establishment of exotic species, affecting species interactions and predator-prey relationships (Lynch et al. 2016, Rehage and Blanchard 2016). How all these changes will affect freshwater ecosystems will depend on their specific characteristics and location (Crozier et al. 2008, Martins et al. 2012).

There has been some debate about the extent to which habitat restoration can compensate for anticipated shifts in temperature and hydrology. However, Beechie et al. (2013) concluded that past land and water uses have often degraded habitats to a greater degree than that predicted from climate change alone, presenting substantial opportunities to improve salmon habitats more than enough to compensate for expected climate change effects over the next several decades. Justice et al. (2017) demonstrated through modeling that a combination of riparian restoration and channel narrowing could reduce stream temperatures and increase the abundance of Chinook salmon parr in the Upper Grande Ronde River and Catherine Creek in northeast Oregon. They concluded that restoration of such streams could more than compensate for an expected increase in summer stream temperature through 2080. Crozier et al. (2019) looked at methods of increasing climate resilience for Pacific salmon and steelhead and concluded that reducing any anthropogenic stressor could improve response to climate change by improving the overall status of an ESU or distinct population segment (DPS) (in terms of abundance, productivity, spatial structure, and diversity) and thereby making the ESU or DPS more resilient and less vulnerable to stochastic extinction.

Beechie et al. (2013) reviewed pertinent literature to evaluate whether specific restoration action types would likely ameliorate climate change effects on flood flows, low flows, or stream temperature. They grouped restoration actions on the basis of the watershed processes or functions they attempt to restore and classified them as either likely or not likely to ameliorate a climate change effect on high stream flows, low stream flows, and stream temperature. They also reviewed restoration actions in the context of their ability to maintain or increase resilience of river ecosystems and salmon populations. Results of their review are briefly summarized in Table A.1-1.

Table A-1.1 Effects of tributary habitat improvement actions on ameliorating climate change effects and improving resilience (summarized from Beechie et al. [2013]).

Action Type	Effects of Action
Restoring riparian function	Restoring riparian areas through replanting of native trees can mitigate stream temperature increases via increased shading. While such actions generally do not directly ameliorate stream flow changes, removal of certain non-native species that use more water than native species and provide less shade can ameliorate both increased stream temperatures and decreased flows. Riparian restoration can lead to modest increases in habitat diversity over the long term via formation of pools or hiding cover. Riparian restoration can be expected to increase ecosystem resilience in the sense that rivers with intact riparian buffers can buffer ecological functions against changes in stream flow, but riparian restoration is unlikely to have significant effects on life history diversity and resilience.
Removing barriers	Removal of dams or other barriers can allow fish to access important upstream cool water habitats when downstream areas become too warm, thereby increasing habitat and life history diversity at the population and meta-population scales. Where dams or other structures contribute to reduced low flows or increased stream temperature, dam removal can also ameliorate low base flow and high temperatures by restoring downstream movement of sediment and water.
Reconnecting floodplains	Floodplain reconnection actions, which typically include reconnection or creation of side channels and sloughs, removal or setback of levees and dikes, and re-meandering of dredged or straightened channels, can ameliorate peak flow increases by storing flood water and reducing flood peaks or by increasing the availability of velocity and thermal refugia. Similarly, removing levees or re-meandering channels can ameliorate temperature increases by increasing the length of hyporheic flow paths beneath the floodplain, which can cool water during the summer. Restoring floodplain connectivity can also increase habitat diversity and facilitate increased life history diversity within a population, which has been linked to increased population resilience. Floodplain reconnection actions generally do not ameliorate base flow decreases.
Restoring incised stream channels	Restoration of incised stream channels can restore floodplain aquifer storage, increase summer base flow and decrease summer stream temperature, and increase availability of flood refugia. Some restoration techniques, such as use of beaver dams to increase sediment storage, have the added effects of increasing summer base flows, locally decreasing or buffering stream temperature, and increasing habitat diversity and productivity. Hence, restoration of incised channels has the potential to ameliorate climate-induced increases in stream temperature and effects on peak flows and low flows, and also to increase life history diversity through creation of off-channel and pond habitats.

Action Type	Effects of Action
Restoring stream flow regimes	**Flood flows** are caused by logging and forest roads, grazing activities, and impervious surfaces in urban areas, because water that would normally flow to streams slowly via subsurface flow is instead routed rapidly to streams through ditches. Actions to reduce routing of water directly from road ditches to the stream or, in an urban environment, to create additional stormwater retention structures or modify surface areas so that runoff is routed into groundwater storage rather than storm drains can help reduce these flood flows. Increased runoff and flood flows can also cause summer baseflows to decrease due to loss of infiltration and water storage in soils. Hence, reductions of grazing or logging effects on flood flows may also increase low flows in summer. **Low stream flows** are exacerbated by withdrawal of water from streams for irrigation or consumptive uses. Restoring water to streams through purchase of water rights or increased irrigation efficiency can dramatically increase low flows to streams and directly ameliorate climate-induced decreases in low stream flow or increased stream temperature. In some cases, restoring flow can also increase habitat diversity by restoring channel-forming flows that maintain habitat diversity and other ecological functions.
Improving stream structure and complexity	Actions such as adding large wood or spawning gravel to streams or re-meandering stream channels have been well documented to provide quick improvements in both physical habitat and fish production, although they do not restore the underlying disrupted process. They also do not generally ameliorate changes in temperature, base flow, or peak flows. Instream habitat actions can increase local habitat complexity (particularly if a large portion of the stream is treated), but such actions are unlikely to increase life history diversity or resilience of salmon populations.
Reducing erosion and sediment delivery to streams	In forested environments of the Pacific Northwest, sediment supply to stream channels is typically increased through surface erosion on unpaved roads or by increased landslides from roads or clearcuts. Climate-change-related increases in storm intensity and a shift from snow to rainfall may cause more frequent landslides in forest environments, especially where road management has not yet achieved reductions in landslide hazard. In croplands and grazing lands, efforts to reduce surface erosion can improve stream habitat by decreasing fine sediment in the streambed, increasing pool depth, or narrowing widened channels, but these actions will not ameliorate decreased low flows, increased flood magnitude, or increased stream temperature (although increased pool depth may create thermal refugia in rare cases). Moreover, these actions do little to increase habitat or life history diversity, except in cases where extremely high sediment supply has filled pools and reduced the diversity of habitat types.

Action Type	Effects of Action
Adding nutrients	Nutrient enrichment to compensate for lack of marine-derived nutrients from reduced salmon returns may be important to the productivity of naturally oligotrophic rivers where salmonid populations are food-limited. Nutrient additions, however, do not address the ultimate cause of low nutrient levels as a result of reduced salmon runs, and do not ameliorate climate change effects on stream flow, stream temperature, or habitat diversity. However, increased stream temperature increases the metabolism of juvenile fishes, which increases their food requirements, so where reduced nutrients and food resources have already compromised growth of juvenile salmonids, actions to increase nutrient supply may indirectly ameliorate temperature effects on salmonid growth rates. Maintaining this effect would require a consistent, long-term nutrient supplementation program.

A.2.2.5 Detecting Fish Response to Tributary Habitat Actions

Measuring the effects of habitat improvement for fish and other aquatic and riparian biota is “one of the great challenges of river and stream conservation” (ISAB 2018). Detecting a fish response to habitat actions is challenging for many reasons: actions encompass a wide range of locations, intensities, and sizes; they are implemented over a long period of time; fish may move in and out of a watershed throughout any evaluation period; the number of juveniles and the time they enter a watershed is dependent on variable environmental conditions in any given year; some actions, such as riparian planting and conservation easements, may take decades before they provide their full ecological benefits; and other actions, such as large wood placements and floodplain reconnection, provide more immediate benefits, but also evolve and accrue benefits over time. In addition, large areas of a watershed need to be improved to detect fish responses (e.g., Roni et al. [2010] reported that more than 20 percent of a watershed would need to be improved to measure a population/watershed-scale response to enhancement), and pre-treatment data and reference, or control, watersheds are needed to detect treatment effects at a watershed scale. Further, many habitat actions have complex effects that play out over large scales and are confounded by other effects (including climate and ocean conditions) that also influence salmon survival.

Importantly, this does not mean that the actions are not providing a benefit, especially when viewed in the context of long-term implementation of habitat improvement actions. Actions may be having a benefit even though that benefit cannot be detected in modeling or monitoring results for various reasons, including countervailing effects such as ocean conditions or increased predation, variability in life-stage survivals, the fact that not a large enough portion of a watershed or the right factors have yet been treated, and, in the case of models, uncertainty in assumptions or parameters.

The ISAB included a cogent discussion of this topic in its report on spring Chinook salmon in the upper Columbia River (ISAB 2018). Noting the complexities and constraints involved in large-scale experiments designed to measure a population response to habitat improvement actions, they discuss how other ways of evaluating possible fish responses can be used to argue for positive benefits that cannot be measured directly owing to other complexities or confounding factors. The monitoring results described above, and many others, are promising in that they address exactly the kinds of factors the ISAB recommended considering: they demonstrate that habitat is changing in response to the actions, that fish are using the restored habitat, that densities and growth rates are responding, and that improvements in survival have been measured in some cases.

A.2.3 Conclusion Regarding Scientific Basis for Tributary Habitat Improvement Actions

To draw conclusions about the benefits of tributary habitat improvements, we evaluated multiple lines of evidence, including knowledge of the basic relationships between fish and their tributary habitat, findings in the scientific literature about how changes in fish habitat affect fish populations, literature on the physical and biological effectiveness of tributary habitat improvement actions, correlation analyses, results from monitoring in the Columbia River basin designed to evaluate the effects of various actions on tributary habitat limiting factors and on salmon and steelhead population response, and the results of life-cycle models. All of this information continues to confirm our findings in the 2008 biological opinion, its 2010 and 2014 supplements, and the 2019 biological opinion regarding habitat and fish response. Overall, the weight of evidence continues to support the basis for implementing tributary habitat improvement actions, and our previous conclusion that many habitat restoration actions can improve salmon abundance and productivity over relatively short periods. Examples of such actions include increasing instream flow, improving access to blocked habitat, reducing mortality from entrainment at water diversion screens, placing of logs and other structures to improve stream structure, and restoring off-channel and floodplain habitat. For other habitat improvements, such as reduction of excess fine sediment in spawning areas and restoration of riparian vegetation, it may take decades to realize their full benefit.

A.3 Analysis of Effects of Tributary Habitat Actions

A.3.1 Methods Used in 2008 Biological Opinion and its Supplements

The approach used in the 2008 biological opinion for analyzing the effects of tributary habitat actions relied on using expert judgment to estimate the change in habitat function as a result of implementing habitat improvement actions, and then using an empirically based model to estimate the overall change in habitat function and a corresponding change in egg-to-smolt survival that would result from that change in habitat function. A monitoring and evaluation program was in place to track the effects of tributary habitat actions and to provide input for the adaptive management framework within which the Action Agencies implemented habitat improvement actions.

This method was developed by the Remand Collaboration Habitat Workgroup, which was convened in 2006 at the request of the Policy Work Group formed as part of the court-ordered remand of NMFS' 2004 biological opinion for the CRS. Members of this workgroup represented the states, tribes, and Federal agencies (including NMFS) involved in the remand collaboration process and were selected for their technical expertise. The workgroup developed methods based on both expert opinion and review of scientific information (such as known egg-to-smolt survival relationships for Chinook salmon and steelhead) that could be applied consistently to all populations. For additional detail on the methods used in the 2008 biological opinion, see Section 3.1.1.6 of NMFS (2014); Appendix C of the 2007 Comprehensive Analysis (Appendix C, Attachment C-1 and Annexes 1–3, USACE et al. [2007]); and Appendix C of Milstein et al. (2013).

A.3.2 Methods Used in 2019 and 2020 Biological Opinions

We noted in the 2014 supplemental biological opinion that life-cycle models (actually, a suite of models within a common framework) were under development and should be available for future CRS analyses (NMFS 2014). These models were developed through the Adaptive Management Implementation Plan process and have been peer-reviewed by the ISAB (ISAB 2013, 2017).

Life-cycle models are increasingly being used in an effort to better predict the outcome of various management scenarios in relation to Pacific Northwest salmonids. By modeling multiple stages and transitions, life-cycle models can determine where bottlenecks in survival or capacity limit recovery, or make projections about population abundance and extinction risk under various scenarios of potential future conditions. Life-cycle models are well-suited to management of salmonid populations because the salmonid life cycle encompasses large geographic ranges and multiple opportunities to address human impacts. Developing effective management strategies involves balancing a range of potential actions across life stages, habitat types, and anthropogenic impacts. The full life-cycle modeling framework used in this opinion is documented in Zabel and Jordan (2020), with additional detail on modeling of tributary habitat improvement actions provided in Pess and Jordan (2019).

The life-cycle modeling effort includes the development of several tributary habitat models in collaboration with key state and tribal scientists. These models represent an evolving method to estimate salmonid population response to habitat improvement actions. They allow detailed estimation of juvenile habitat capacity and survival, making it possible to evaluate changes in capacity and survival under various management or restoration scenarios. All the models are framed in the matrix life-cycle modeling format originally described by the Interior Columbia Technical Recovery Team (ICTRT) and Zabel (2007), although each is adapted to use the different levels of information available to populate its freshwater life stages (Zabel et al. 2017, Pess and Jordan 2019, Zabel and Jordan 2020).

In the 2019 biological opinion and in this opinion, we consider results of these tributary habitat models for some spring Chinook salmon populations in evaluating the effects of actions implemented to date and the effects of the proposed action. We expect to continue model development for additional populations in the future. As noted below, we also anticipate using these tributary habitat models to inform implementation of proposed tributary habitat actions over the 15-year implementation period of this opinion.

In addition to using life-cycle models to evaluate tributary habitat actions for some populations, we also evaluated proposed tributary habitat actions using qualitative considerations. Both the quantitative methods and qualitative considerations are described below.

A.3.2.1 Quantitative Methods: Modeling the Effects of Tributary Habitat Actions

Using life-cycle models to estimate fish population response to a suite of tributary habitat actions involves the following steps:

1. Estimate life-stage-specific habitat capacities: To estimate how a population will respond to various types and intensities of tributary habitat improvement actions, modelers first need to estimate life-stage-specific habitat capacity, or how many fish a system might support at a specific life stage under historical, current, or proposed habitat conditions. This requires a compilation of available data on parameters such as life-stage-specific capacity, survival, and abundance. Models can then be developed at the appropriate level of detail given the available data and understanding of limiting factors. If data to parameterize a model are lacking, modelers must choose whether to collect the necessary data or to utilize the parameters and functional relationships from nearby basins or the general literature to inform the model. Zabel et al. (2017) and Zabel and Jordan (2020) describe and compare different methods to estimate juvenile rearing capacity at several spatial scales and extents. Pess and Jordan (2019) provide additional discussion on the approaches used to estimate juvenile rearing capacity.

2. Calibrate life-cycle models to fish data and current conditions: To make models more accurately reflect fish data and current conditions, modelers calibrate them, meaning that they adjust model parameters based on available data. Calibration techniques range from straightforward to complex. For example, a simple approach would be to develop life-cycle-model parameters independently based on the literature and reach-scale data, and then adjust the reach-scale parameters to produce population-scale predictions that are in closer agreement with basin-scale fish data. More complex approaches involve the use of statistical model fitting, where statistical techniques, such as state-space models, are used to derive parameters directly from local fish abundance data, where available. This approach allows for uncertainty in the data to be carried through all stages of the model and be reflected in the outputs. Approaches to calibration are discussed in more detail in Pess and Jordan (2019).

3. Evaluate how habitat restoration scenarios would change habitat capacity and survival: If managers develop several restoration scenarios, modelers can evaluate how each restoration scenario would change habitat capacity and productivity from existing or historical conditions. They do this by comparing the current or proposed stream condition to experimental or observational data that can inform how habitat capacity, fish growth, or fish survival changes under different habitat scenarios.

Habitat is typically evaluated by looking at habitat quantity (e.g., stream channel area, pool frequency), habitat quality (e.g., floodplain condition, fine sediment levels, riparian condition), environmental conditions (e.g., stream temperature, streamflow), indicators of habitat quality (e.g., adjacent land use), and causes of habitat degradation (e.g., water diversions and barriers). Each of these variables can have an impact on salmon habitat capacity and survival at one or more life stages.

Habitat changes between restoration scenarios and current or historical conditions are then translated into changes in life-stage capacity or survival. For example, addition of wood structures to a channel may increase both summer and winter rearing capacity and change both summer and winter rearing life-stage survival rates. By contrast, a change in spawning gravel quality by decreasing percentage of fine sediment would not alter spawning capacity, but would increase egg-to-fry survival. Pess and Jordan (2019) document the methods used for translating habitat actions into life-cycle model inputs. They also contrast methods and results using "data rich" and "data poor" environments in the Upper Grande Ronde, Wenatchee, and Upper Salmon River basins.

In general, changes in habitat quantity translate into changes in habitat capacity, and changes in habitat quality translate into changes in life-stage survival. The functional relationships between a habitat change and the corresponding change in capacity or survival are typically developed from literature or from local empirical relationships. For example, numerous studies of fine sediment effects on egg-to-fry survival show that egg-to-fry survival decreases with increasing fine sediment, and both general and species-specific equations can be developed to translate changes in fine sediment into a change in survival. On the other hand, local data may indicate that smolt production of a particular species is related to a stream parameter such as summer streamflow, and the statistical relationship between streamflow and survival can be used to quantify rearing survival in a life-cycle model. For additional detail on translating habitat quantity into habitat capacity estimates and translating habitat quality into survival estimates, see Pess and Jordan (2019).

4. Use life-cycle models to evaluate differences in fish production among scenarios: Finally, the changes in capacity and survival from the restoration scenarios are used as inputs to a life-cycle model to assess the overall change in salmon abundance and productivity (and, potentially, change in spatial structure and diversity) that would result from the restoration scenarios. For example, modelers might estimate that reconnecting a certain amount of floodplains will increase parr capacity in a particular stream by 10 percent. That information then becomes an input to a life-cycle model to evaluate whether that 10 percent increase in parr capacity will result in an increase in adult abundance or, alternatively, in falling below a quasi-extinction threshold. If there is strong density dependence after the parr stage (e.g., in overwinter survival), then the increased parr capacity might not produce many additional adults. In other cases, there might be a proportional increase in adult abundance.

Life-cycle models can vary considerably in complexity, particularly in the number and specificity of life stages included in the model. In general, more complex models allow for a greater range of restoration scenario development; however, they also require more data. Conversely, less complicated models accommodate a more limited range of restoration scenarios, but require less input data.

For this opinion, we considered modeling of the effects of tributary habitat actions implemented from 2009 through 2015 and proposed tributary habitat actions for certain populations in the Grande Ronde/Imnaha and Upper Salmon River major population groups (MPGs) of the Snake River spring/summer Chinook salmon ESU; for the Wenatchee River population in the Upper Columbia River spring-run Chinook salmon ESU, we considered the effects of tributary habitat actions implemented from 2009 through 2018 and proposed tributary habitat actions (see Cooney et al. [2020b], Jordan et al. [2020], and Jorgensen and Bond [2020]).[5] In some cases, modeling of additional habitat action scenarios, such as scenarios involving longer-term strategic implementation of actions, or scenarios involving random implementation of actions (see Pess and Jordan 2019), was also available and informed our understanding of the context for the proposed tributary habitat action.

For the Wenatchee spring-run Chinook salmon population, modelers evaluated how certain tributary habitat actions implemented in 2009 through 2018, as well as how certain actions proposed for implementation in 2021 through 2036, might change juvenile rearing capacity in major tributaries to the Wenatchee River (Jorgensen et al. 2013, 2017; Pess and Jordan 2019; Jorgensen and Bond 2020). The model can only assess the benefits to juvenile rearing capacity of actions to improve access and stream complexity. Actions implemented and proposed for implementation are broader in scope than those evaluated, but because the model does not evaluate the effects of actions such as returning flow to the stream, screening diversions, and restoring riparian areas, potential benefits of those types of actions are not included in the model results. Further, the modeling may not have captured all benefits attributable to the specific actions that were evaluated (Jorgensen and Bond 2020). There were four habitat actions completed during the period of 2009 through 2018 that were located in the spawning and rearing areas evaluated and that were quantifiable into changes in habitat capacity. These actions translated to a 7.6 percent increase in capacity in the Nason Creek watershed and a 1.7 percent increase in the White River watershed (Jorgensen and Bond 2020).

For the proposed actions, the modelers also had to make assumptions about what portion of the anticipated actions (which the Action Agencies identified at the MPG level) would be implemented in the Wenatchee River population, and where, when, and how. These assumptions are described in Jorgensen and Bond (2020). For example, modelers assumed that actions would be implemented in the same watersheds where they had been in the past (i.e., the White River,

[5] Although habitat improvement actions were underway in this ESU before 2009, modelers used 2009 as a starting point because they viewed actions completed before then as less likely to yield benefits as a result of having been more opportunistic, smaller actions implemented without the benefit of comprehensive tributary and reach assessments and other planning tools. In addition, systematic monitoring data to describe habitat conditions for use in the life-cycle models were not available prior to 2009. Metrics for actions completed in 2019 were not yet available at the time the life-cycle modeling was completed.

Nason Creek, and the Chiwawa River within the Wenatchee River subbasin); that habitat access actions would open habitat of type and quality similar to that currently available; and that actions to improve complexity would be implemented in locations adjacent to habitat currently in moderate or good condition. Based on model results, the proposed actions would increase juvenile rearing capacity by 3 percent in the Chiwawa River, 5.2 percent in Nason Creek, and 3.5 percent in the White River (Jorgensen and Bond 2020). (Projected changes in abundance of natural-origin spawners and extinction risk for this population as a result of these actions are captured in the life-cycle modeling results discussed in Section 2.6.3.1.12 of this opinion.)

For the Upper Salmon River spring/summer Chinook salmon MPG, modelers evaluated how certain tributary habitat actions implemented from 2009 through 2015 would affect juvenile rearing and spawning capacity in the Pahsimeroi, North Fork, East Fork, Upper Mainstem, and Yankee Fork populations. The model evaluated instream actions (i.e., actions to improve stream complexity and/or floodplain/side-channel connectivity) and actions to improve access. Because this model does not evaluate the effects of actions such as returning flow to the stream, screening diversions, or restoring riparian areas, benefits of such actions are not included in the model results. Further, the modeling may not have captured all benefits attributable to the specific actions that were evaluated. Modeling methods, assumptions, and results are documented in Pess and Jordan (2019) and in Jordan et al. (2020). Based on the model results, actions implemented in 2009 through 2015 increased juvenile rearing capacity by 7 percent in the Lemhi, 9.4 percent in the Pahsimeroi, 2 percent in the North Fork, less than 1 percent in the East Fork and Upper Mainstem, and 1 percent in the Yankee Fork. The actions increased spawning capacity by less than 0.5 percent in most of these populations, and by 2.1 percent in the Lemhi population (Jordan et al. 2020).

Modelers also evaluated the effects of some types of the proposed tributary actions on populations in this MPG. The proposed tributary habitat actions for 2021 through 2036 for this MPG include flow protection and enhancement, screening of diversions, access, stream complexity, and riparian habitat improvement. As noted above, however, the model can only assess the benefits to juvenile rearing and adult spawning capacity of instream actions to improve stream complexity or floodplain/side-channel connectivity and actions to improve access, so the effects of actions such as returning flow to the stream, screening diversions, and restoring riparian areas are not included in the model results. For the analysis, modelers assumed that the Action Agencies' efforts would be focused on the Lemhi, Pahsimeroi, and Upper Mainstem populations. Modelers also made other assumptions, documented in Pess and Jordan (2019) and Jordan et al. (2020) (e.g., habitat access projects were assumed to open habitat of similar type and quality to that currently available, and complexity actions were applied to improve the quality of habitat currently in moderate or good condition). Based on model results, the proposed actions would increase juvenile rearing capacity by 12.3 percent in the Lemhi, 19.8 percent in the Pahsimeroi, and 10.5 percent in the Upper Mainstem; spawning capacity would increase by 7.8 percent in the Lemhi, 19.8 percent in the Pahsimeroi, and 6.9 percent in the Upper Mainstem. (Projected changes in abundance of natural-origin spawners and extinction risk for this population as a result of these actions are captured in the life-cycle modeling results discussed in Section 2.2.3.1.12. of this opinion.)

For the Grande Ronde/Imnaha River spring/summer Chinook salmon MPG, modelers evaluated: 1) the impacts of tributary habitat actions implemented in 2009 through 2015 (Pess and Jordan 2019, Cooney et al. 2020); 2) the impacts of the tributary habitat actions in the proposed action for this opinion (Cooney et al. 2020); 3) several scenarios of long-term implementation of tributary habitat actions, including specific actions called for in Appendix A (Northeast Oregon management unit) of the Snake River Spring/Summer Chinook Salmon and Steelhead ESA Recovery Plan (NMFS 2017b), and a scenario focused on restoring stream structure and reducing temperatures through the combined effects of riparian shade and achieving natural channel structure and width/depth ratios (Pess and Jordan 2019). Modeling methods, assumptions, and results are documented in Pess and Jordan (2019) and Cooney et al. (2020b). Modelers concluded, for example, that in Catherine Creek, the actions implemented in 2009 to 2015 would increase summer parr rearing capacity by 21 percent within a few years of implementation. For the Catherine Creek population, the proposed actions would increase summer-rearing capacity by an additional 75 percent after full implementation (i.e., by 2036), and for the Upper Grande Ronde population, by 26 percent. As benefits of actions implemented under the proposed action continue to accrue, functional parr capacity would increase by a total of 100 percent in Catherine Creek and 33 percent in the Upper Grande Ronde at 24 years after full implementation (Cooney et al. 2020b; Pess and Jordan 2019). (Projected changes in abundance of natural-origin spawners and extinction risk as a result of tributary habitat actions are captured in the life-cycle modeling results discussed in Section 2.2.3.1.12 and Appendix C of this opinion, and in Zabel and Jordan 2020.)

Generally the modeling shows that 1) actions implemented in 2009 through 2018 will have small-to-moderate positive effects on habitat capacity; 2) implementation of the proposed tributary habitat actions evaluated in this biological opinion will have small-to-large positive effects on abundance and extinction risk; 3) implementation of actions at similar or enhanced levels of effort for a longer time period, consistent with recovery plan priorities and best principles of watershed restoration would have even greater benefits (e.g., see discussion of life-cycle modeling results for the Grande Ronde spring/summer Chinook salmon MPG in Pess and Jordan 2019).

A.3.2.2 Qualitative Considerations Used in 2019 and 2020 Biological Opinions

In addition to considering the results of life-cycle modeling of salmonid population response to habitat improvement actions, we also used qualitative considerations to evaluate tributary habitat actions. The qualitative considerations included the following factors:

A.3.2.2.1 Extent to Which Actions Address Identified Limiting Factors or Life Stages

A limiting factor is a factor that controls the growth, abundance, or distribution of a population in an ecosystem. Tributary habitat improvement actions will be most beneficial if targeted at the factor or life stage that is most limiting. We considered the extent to which tributary habitat actions implemented and proposed for implementation addressed identified limiting factors and life stages. Our qualitative evaluation of this factor for the proposed tributary habitat action is necessarily coarse in scale since the proposed actions are identified at the MPG scale. However, based on the Action Agencies' past record of implementation, their stated commitment to continue to improve strategic implementation, and the types of actions they have identified for implementation, we are confident that, in general, actions to be implemented will target limiting factors that have been identified using best available information.

A.3.2.2.2 Potential to Improve Tributary Habitat Conditions

Our qualitative evaluation also considered the potential for improvements in tributary habitat capacity and/or productivity in the targeted populations. This consideration is important because it speaks to the potential to achieve improvements in abundance, productivity, spatial structure, and diversity as a result of implementing tributary habitat improvements. Our evaluation of the potential to improve tributary habitat conditions was informed by ESA recovery plans for interior Columbia Basin salmon and steelhead (UCSRB 2007; NMFS 2009, 2015, 2017a, 2017b), the most recent ESA 5-year status reviews (NMFS 2016a, 2016b, 2016c), the focal population analysis for Snake River spring/summer Chinook salmon (Cooney et al. 2020a; also see additional discussion below), the ISAB's 2015 examination of density dependence in salmon and steelhead in the Columbia River basin (ISAB 2015), and other information. Again, our qualitative evaluation of the proposed tributary habitat action is necessarily coarse since the proposed actions are identified at the MPG scale. However, based on our evaluation and on the Action Agencies' statements regarding the populations where they intend to focus implementation of tributary habitat improvement actions (BPA et al. 2020), actions will target populations where there is potential to improve tributary habitat productivity.

A.3.2.2.3 Role of Populations in ESA Recovery Scenario

NMFS has completed ESA recovery plans for all listed salmon and steelhead in the Columbia River basin (UCSRB 2007; NMFS 2009, 2013a, 2015, 2017a, 2017b). These recovery plans provide: 1) recovery goals, 2) management actions to achieve the goals, and 3) estimates of the time and cost required to carry out the actions. The plans also provide additional information to help frame and prioritize recovery actions, including descriptions of the status of the species; identification of limiting factors and threats; and "scenarios" for recovery. Recovery scenarios are based on the biological viability criteria developed by technical recovery teams (TRTs) to define conditions that, when met, will describe viable populations and species.[6]

The biological viability criteria are consistent with the hierarchical population structure that is critical to the resilience and long-term survival of salmon and steelhead. Each ESU or DPS consists of multiple independent populations that spawn in different watersheds throughout the

[6] NMFS appointed two TRTS for Columbia Basin: the Willamette/Lower Columbia TRT and the Interior Columbia TRT. This discussion focuses on the ICTRT, but the work of both Columbia Basin TRTs, and of all West Coast TRTs, was based on the same scientific principles (e.g., McElhany et al. 2000) and was generally consistent with each other.

ESU's or DPS's range. Additionally, within an ESU or DPS, independent populations are organized into larger groups known as major population groups (MPGs). MPGs are groups of populations that share similarities within the ESU or DPS (ICTRT 2005). The viability criteria are designed to assess risk for abundance/productivity and spatial structure/diversity at the population level. These population-level assessments are then considered in the context of criteria for how many and which populations within an MPG need to be at what status for the MPG as a whole to have a low risk of extinction, consistent with de-listing. The viability criteria developed by the Interior Columbia Technical Recovery Team (ICTRT) are summarized briefly below and outlined in detail in Interior Columbia recovery plans (NMFS 2017a, 2017b, 2015, 2009; UCSRB 2007) and the ICTRT's technical report (ICTRT 2007).

ESU/DPS viability criterion: All extant MPGs and any extirpated MPGs critical for proper functioning of the ESU or DPS should be at low risk.

MPG-level viability criteria: The following six criteria should be met for an MPG to be regarded as at low risk:

1. At least one-half of the historical populations within the MPG (with a minimum of two populations) should meet viability standards.[7]
2. At least one population should be classified as highly viable.[8]
3. Viable populations within an MPG should include some populations that are classified (based on historical intrinsic potential) as "very large," "large," or "intermediate." In particular, very large and large populations should be at or above their composite historical fraction within each MPG.
4. All major life-history strategies (e.g., spring and summer run timing) that were present historically within the MPG should be represented in populations meeting viability requirements.
5. Remaining MPG populations should be maintained with sufficient abundance, productivity, spatial structure, and diversity to provide for ecological functions and to preserve options for ESU/DPS recovery.
6. For MPGs with only one population, the population must be highly viable.

Population-level criteria: The ICTRT also defined population-level criteria for evaluating the status of the individual populations. These criteria describe a viable population based on

[7] This means that, based on evaluation of population-level abundance/productivity, spatial structure, and diversity, using methods recommended by the ICTRT, a population should have a low (<5 percent) risk of extinction over a 100-year time frame.

[8] This means that, based on evaluation of population-level abundance/productivity, spatial structure, and diversity, using methods recommended by the ICTRT, a population should have a very low (<1 percent) risk of extinction over a 100-year time frame.

the four viable salmonid population (VSP) parameters (abundance, productivity, spatial structure, and diversity).

Thus, the criteria for determining whether an MPG is at low risk allow for some flexibility in terms of which populations will be targeted for a particular recovery level to achieve a low risk MPG. The ESA recovery plans provide some additional guidance on which populations are targeted for viable, highly viable, or maintained status to achieve ESA recovery. This is relevant to the effects analysis because, in general, efforts focused on populations that need to achieve viable or highly viable status will be more valuable to near-term and long-term recovery efforts.

A.3.2.2.4 NMFS Focal Population Analysis

To provide strategic guidance for implementation of recovery plans, NMFS has developed the concept of focal populations. The intent of this concept is to develop and apply criteria to identify populations where tributary habitat recovery efforts should be focused in the short term (i.e., a 5- to 10-year time frame) to contribute to both near-term improvements and long-term recovery goals. This concept and the method used to identify focal populations are described in detail in Cooney et al. (2020a).

The importance of sequencing or prioritizing restoration and recovery efforts over time to optimize conservation outcomes has gained increased attention in the conservation literature (e.g., see Drechsler and Wissel 1998, Willi et al. 2006, McBride et al. 2010, Wilson et al. 2011, Aitken et al. 2013). Specifically, it is important to explicitly consider the role of starting conditions and inherent limitations on available resources when determining how to maximize gains toward long-term goals. It is also important to consider the time required for restoration actions to achieve desired improvements in habitat conditions and the associated lags in benefits to fish. In many ways, the basic principles for multi-population-level sequential planning strategies parallel the advice regarding protection and restoration within populations (e.g., Beechie et al. 2010).

Using ESA recovery plans and ICTRT work as starting points, supplemented by new information and additional considerations, NMFS developed an approach to identify short-term opportunities to benefit key populations, consistent with longer-term ESA recovery goals. Criteria for identifying focal populations include: 1) VSP characteristics (abundance, productivity, spatial structure, and diversity; intrinsic potential [population size and complexity]; and meta-population characteristics); 2) current population status (quasi-extinction risk, current abundance relative to minimum thresholds for recovery, hatchery supplementation, and gaps between current status and target status to achieve recovery goals); 3) relative habitat improvement potential; and 4) climate change vulnerabilities. For details on these criteria and how they were applied, see Cooney et al. (2020a).

The focal population concept is intended to complement other approaches to help prioritize activities in the basin in support of recovery plan implementation, CRS-related actions, and other processes. For example, results from the focal population analysis could contribute to sequencing future efforts to develop more strategic action plans at the population or MPG level. Accordingly, in our qualitative evaluation of the effects of tributary habitat actions under this opinion, we considered alignment between the Action Agencies' efforts and the focal populations. We expect to work with the Action Agencies and co-managers over time to more closely align tributary habitat efforts with focal populations.

A.3.2.2.5 Action Agencies' Track Record of Implementation

Our qualitative evaluation also considered the Action Agencies' track record of implementation, their relationships with local implementing partners, and their commitment to continuing to improve implementation through adaptive management. Under the 2008 biological opinion, the Action Agencies implemented substantial tributary habitat improvement actions; increased investments in the tributary habitat; and improved the scope, biological rigor, and collaborative regional effort directed at implementing tributary habitat actions (BPA et al. 2013, 2016; NMFS 2014). They continued implementation of these actions under the 2019 biological opinion (NMFS 2019, BPA et al. 2020). The Action Agencies have stated their commitment to continuing to improve strategic implementation of tributary habitat actions, consistent with best available science related to habitat restoration; to convene a tributary habitat steering committee to oversee program implementation and a tributary technical team to provide scientific input to program implementation; to report on implementation using metrics that will allow NMFS to evaluate the success of the actions; and to conduct RME to assess tributary habitat conditions, limiting factors, action effectiveness, and to address associated critical uncertainties (BPA et al. 2020).

A.3.2.2.6 Short-term Negative Effects of Implementing Tributary Habitat Improvement Actions

We considered short-term negative effects that could result from implementation of tributary habitat improvement actions. Tributary habitat improvement actions will have long-term beneficial effects at the action and subbasin scale. Adverse effects during construction are expected to be minor, occur only at the project scale, and persist for a short time (no more and typically less than a few weeks). Examples of such short-term effects include sediment plumes, localized and brief chemical contamination from machinery, and the destruction or disturbance of some existing riparian vegetation. These impacts will be limited by the use of the practices described in the Habitat Improvement Programmatic Consultation (NMFS 2013b, 2020). The positive effects of these actions on habitat function and salmon and steelhead populations (e.g., restored access, improved water quality and hydraulic processes, restored riparian vegetation, and enhanced channel structure) will be long term.

A.3.2.3 Conclusion Regarding Methods Used to Analyze Effects

NMFS has determined that the approach used to evaluate the effects of proposed tributary habitat improvement actions is based on best available science. The qualitative considerations used in our analysis are comprehensive and based on best available information. In addition, we considered quantitative life-cycle model results that represent an improved method to estimate salmonid population response to a series of habitat improvement actions.

The life-cycle models were developed in collaboration with key state and tribal scientists, and have been independently peer-reviewed by the ISAB (ISAB 2013, 2017). They allow detailed estimation of juvenile habitat capacity and survival, making it possible to evaluate changes in capacity and survival under various management or restoration scenarios. They are based to a greater extent on population-specific empirical relationships than were the methods used in the 2008 biological opinion, and they are based on a more complex and realistic representation of fish-habitat relationships and timing of benefits than the methods used in the 2008 biological opinion. Under the methods used in the 2008 biological opinion, expert judgment provided a large part of the determination of habitat function in all locations, given the limited extent of readily available empirical data and information. We expect that expert opinion will continue to play a role in the process of estimating habitat benefits (e.g., in estimating how specific actions or suites of actions will change habitat), but we also expect greater reliance on empirically and mathematically derived relationships as they evolve.

A.4 RME and Adaptive Management

The 2008 biological opinion and its 2010 and 2014 supplements (NMFS 2008, 2010, 2014) contained a robust and sizable RME program designed to evaluate tributary habitat conditions, fish use of tributaries, and the effects of tributary habitat improvement actions on habitat and fish. The collection, assembly, and analysis of data from that program has enhanced our understanding of the effects of tributary habitat improvement actions and provided new information to incorporate into decision-making on habitat action prioritization and design. Such RME continued to play an important role in implementation of tributary habitat actions under the 2019 CRS biological opinion and will continue to do so under the current proposed action.

The Action Agencies will continue to monitor habitat status and trends, conduct compliance and implementation monitoring, support habitat action effectiveness monitoring, and fund fish and habitat monitoring and research projects with regional partners to address critical uncertainties. The Action Agencies have also committed to engaging in a collaborative process with other regional partners to develop and implement a Columbia River basin tributary habitat RME strategy that will align with and directly support implementation of tributary habitat improvement actions.

In addition, the Action Agencies have committed to an adaptive management/decision support framework in which habitat action implementation will be guided by the Tributary Habitat Steering Committee, with input from a newly formed Tributary Technical Team. Comprehensive reviews of program implementation will occur at 5-year intervals, with the explicit recognition that such periodic reviews provide an opportunity to consider program adjustments based on NMFS' 5-year status reviews, new climate and fisheries science, and the ongoing development of life-cycle models and other tools for identifying, prioritizing, and evaluating the projected benefits of suites of actions (Appendix D of BPA et al. 2020).

A.5 Implementation Considerations

In the 2008 biological opinion and its 2010 and 2014 supplements, we described recent findings in the literature regarding approaches to watershed restoration. That literature emphasized the need to incorporate proper planning (including assessing the natural potential of a system and using that information to direct action location, design, and selection), sequencing, and prioritization into decision frameworks to best achieve habitat objectives. We noted the four principles outlined in Beechie et al. (2010) to help ensure that restoration was guided toward sustainable actions: 1) address the root causes of degradation, 2) be consistent with the physical and biological potential of the site, 3) scale actions to be commensurate with the environmental problems, and 4) clearly articulate the expected outcomes.

Recent literature, monitoring information, and life-cycle modeling continue to reinforce the principles outlined in Beechie et al. (2010) and the principle that if the wrong action is implemented in the wrong place or at the wrong time, desired habitat conditions will not be observed or sustained over time. Based on their literature review, Hillman et al. (2016) concluded that the actions that are most effective at producing desired habitat conditions are those that:

1. Address the life stage and habitat condition limiting fish performance. Salmonid response to habitat enhancement is based on whether or not the enhancement actions address the specific life stage and habitat factors limiting that population's performance.

2. Consider, and are implemented in context with, fluvial and geomorphic conditions and are sequenced such that the effects of enhancement actions on habitat conditions are not limited by upstream watershed processes. Habitat improvement actions, including protection projects, are ineffective or the effects are short-lived if unaddressed upstream watershed processes degrade treatment sites.

3. Treat a large percentage of the stream or watershed. The literature indicates that the largest biological benefits are associated with treating more than 20 percent of a watershed. Treating small portions of degraded habitat has little biological effect at the watershed scale, and the treatments are often overwhelmed by upstream degraded habitat conditions. (Roni [2018] also noted that the total amount of restoration and the connectivity of the restored habitats are important drivers of population- or watershed-level response to restoration.)

4. Derive from detailed watershed assessments to determine disrupted processes and lost habitat. Limiting factors analysis, watershed assessments, reach assessments, and habitat and life-cycle modeling are tools that can be used to identify threats, problems, and limiting factors within a watershed.

5. Are implemented in the context of a watershed implementation plan that prioritizes locations and types of actions. The sequencing of actions needs to consider degraded

watershed processes and threats, and limiting life stages and habitats. The literature identified degraded upstream watershed processes as the most common factor affecting the success of enhancement projects.

6. Include adequate coordination among stakeholders, landowners, funding and monitoring entities, and implementers. A lack of landowner support can derail a well-designed implementation plan.

7. Incorporate effectiveness monitoring at a subset of projects. Monitoring data collected under an adaptive management framework provide information needed to determine if enhancement work should continue as planned or be refocused or redirected.

Hillman et al. (2016) also summarize findings regarding specific types of habitat improvement actions, and note the importance of protecting high-quality habitat and prioritizing the reconnection of spawning and rearing areas (particularly areas with high intrinsic potential). They recommend that use of instream structures be implemented in concert with actions that improve watershed processes. Instream structures often provide benefits that are realized more quickly than actions that improve watershed processes, but it is important that they be sized appropriately for the channel and designed to mimic natural accumulations.

The core components of these findings were reinforced by the ISAB in its review of spring Chinook salmon in the upper Columbia River (ISAB 2018). They noted that while further analysis of limiting factors was needed, "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." They further noted that "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."

Pess and Jordan (2019) elaborate on some of these themes and demonstrate the utility of life-cycle modeling in both data-rich and data-poor situations for evaluating and choosing among alternative restoration scenarios. They note that it is the combined effect of all restoration actions that will determine the potential magnitude of change in salmon populations, and demonstrate how alternative restoration scenarios at the watershed-scale can be developed and evaluated to determine which suite of actions will likely provide the largest benefit to salmon populations. The purpose of these analyses is to help focus restoration efforts on the types, location, and level of actions that lead to a measurable and significant improvement to salmon populations. Specific methods for these analyses depend on local habitat and fish data availability, and may range from simple analyses based on coarse spatial and/or temporal resolution data to more detailed evaluations with higher resolution data. Therefore, while the richness of the data will determine the analysis type used to evaluate the salmon population response to a suite of potential restoration actions, we do have tools available for both data-rich and data-poor situations. Further, learning from data-rich scenarios will inform and support decision making in data-poor scenarios.

Another important consideration in identifying and prioritizing tributary habitat improvement actions is how those actions can contribute to climate adaptation and resilience for salmonids. Beechie et al. 2013 developed a decision support process for adapting salmon recovery plans that incorporates: 1) local habitat factors limiting salmon recovery, 2) scenarios of climate change effects on stream flow and temperature, 3) the ability of restoration actions to ameliorate climate change effects, and 4) the ability of restoration actions to increase habitat diversity and salmon population resilience.

We support the conclusions and evaluation approaches noted above regarding effective implementation of tributary habitat improvement actions, and we expect that the Action Agencies will continue working to implement the tributary habitat improvement actions consistent with the recommendations noted above so that their effectiveness will be enhanced.

A.6 Conclusion

For this opinion, we reviewed the literature on habitat restoration and re-affirmed the strong technical foundation for the tributary habitat program. We evaluated RME information and found that it also supported the foundation of the program. We determined that the methods we use to evaluate the effects of tributary habitat actions are based on best available science and information. We evaluated the effects of proposed tributary habitat actions quantitatively for some populations and qualitatively for all populations within the context of our understanding of limiting factors, habitat improvement potential, and recovery plan and focus population frameworks. We then qualitatively related these population-level changes to effects on the species or designated critical habitat. We considered short-term negative effects that could result from implementation of habitat improvement actions. We also considered the Action Agencies' track record of implementation, as well as the strategic framework within which the Action Agencies were committing to implement the tributary habitat improvement actions. In addition, we considered the adequacy of the RME and adaptive management framework proposed to evaluate and support implementation of tributary habitat actions.

Over time, understanding of habitat limiting factors has improved, along with the tools and processes for identifying, prioritizing, and coordinating the locations and types of actions that will provide the greatest improvements. The completion of ESA recovery plans for all ESA-listed Columbia Basin salmon and steelhead, the continued development of life-cycle models and additional tools that the Action Agencies and others have developed (e.g., tributary and reach assessments), should further enhance the ability to implement actions within a strategic framework. The Action Agencies' continued development and support of the local partnerships and the implementation infrastructure they have developed over the past 10-plus years should also contribute to this effort. Thus, we expect that future habitat restoration actions will target actions strategically to address limiting factors in a manner that contributes to both short-term and long-term benefits to VSP parameters, with a focus on populations that are important to achieving long-term recovery goals.

Implementation of the tributary habitat actions analyzed in this opinion, if implemented as described in the proposed action, will provide near-term and long-term benefits to the targeted populations. Actions implemented to ameliorate limiting factors for any population would provide localized habitat benefits and potential improvements in abundance and productivity for the targeted population. Where such actions are implemented consistent with the strategic approach outlined in the proposed action (i.e., consistent with ESA recovery plan population priorities and the best available science [e.g., watershed assessments] and modeling information that informs questions related to what kind of actions will be most beneficial where, in what sequence, and at what scale), these benefits would be enhanced. In addition, certain types of actions are also likely to increase climate change resilience. For example, actions to restore riparian vegetation, streamflow, and floodplain connectivity and to re-aggrade incised stream channels can ameliorate temperature increases, base flow decreases, and peak flow increases, and thereby improve stream conditions, habitat diversity, and population resilience to certain effects of climate change. Improvements in tributary habitat are likely to contribute to improvements in all four VSP parameters for the targeted populations. While it is possible that effects of some actions, such as actions to improve stream flow or remove barriers to passage, could be immediate, for other actions, benefits will take several years to fully accrue (and could take 50 years or more for actions such as restoring riparian areas)—and fish populations also need sufficient time to respond. Therefore, it is unlikely that the full benefits of these actions will be realized in the timeframe of this proposed action. Further, to yield significant improvements, it is necessary to implement a large scale and scope of habitat improvement actions (e.g., implementation over a 25-year time period or longer) and to implement actions throughout a large portion of each watershed. Thus, it is important to consider the results of the habitat actions to be implemented under this proposed action in the context of the effects of long-term implementation of habitat actions.

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Appendix B – Avian Predation Management

Note: In some cases, predation rate estimates in this appendix differ from those reported in the 2019 CRS Biological Opinion. The 2019 CRS Biological Opinion used results from some reports with relatively small datasets or generated using methods that are now out-of-date. For example, before 2014, most predation rates were reported as minimum estimates because they had not been corrected for the proportion of tags that the birds deposited at off-colony locations (e.g., loafing, roosting, and foraging sites) or the proportion of deposited fish tags that were lost (i.e., not detected) before scanning the colony at the end of nesting season (e.g., blown off the colony during wind storms, washed away during flooding events, or otherwise damaged during the course of nesting season) (Evans and Payton (2020a). In draft tables produced for the regionally supported Avian Predation Synthesis Report, to be completed in September 2020, Evans and Payton (2020a) report updated predation rate estimates that have been corrected for PIT tag deposition and detection probabilities for all colonies and years where adequate data were available for analysis. This retrospective analysis of historical PIT tag datasets provides a more accurate and standardized estimate of avian predation rates across years, waterbird species, and colony locations than those previously reported in the researchers' annual reports or NMFS' biological opinions.

In addition, the CRS Action Agencies and U.S. Fish and Wildlife Service requested a change in the definitions of the pre-management and post-management time periods for the managed colonial waterbird colonies. In Evans and Payton (2020a), the management periods begin when actions to reduce colony size were initiated, regardless of whether those actions resulted in a significantly reduced colony size that year.

B.1 Introduction

Research shows that Caspian terns (*Hydroprogne caspia*) and double-crested cormorants (*Phalacrocorax auritus*) nesting on East Sand Island in the Columbia River estuary have consumed more than 10 to 20 percent of the juvenile Endangered Species Act (ESA)-listed Chinook and steelhead migrating from the interior Columbia Basin in some years (USACE 2015a; Evans et al. 2016). In response to these findings, NMFS provided several management measures in the 2008 FCRPS[1] biological opinion and 2010 and 2014 supplemental biological opinions to reduce the predation rates (NMFS 2008, 2010, 2014):

- RPA Action 45—The FCRPS Action Agencies will implement the Caspian Tern Management Plan. East Sand Island tern habitat will be reduced from 6.5 to 1.5–2 acres.

[1] In earlier biological opinions, the federal Columbia River System (CRS) was referred to as the Federal Columbia River Power System or FCRPS.

- RPA Action 46—The FCRPS Action Agencies will develop a cormorant management plan encompassing additional research, development of a conceptual management plan, and implementation of warranted actions in the estuary.
  - This RPA action was modified in 2014 to read: "The FCRPS Action Agencies will develop a cormorant management plan (including necessary monitoring and research) and implement warranted actions to reduce cormorant predation in the estuary to Base Period levels (no more than 5,380 to 5,939 nesting pairs on East Sand Island."
- RPA Action 47—Inland Avian Predation: The FCRPS Action Agencies will develop an avian management plan (for Double-Crested Cormorants, Caspian Terns, and other avian species as determined by RME) for Corps-owned lands and associated shallow-water habitat.

During the 2018 Court-ordered remand of the 2014 supplemental opinion, the Action Agencies proposed to implement the management plans that resulted from the RPA actions. In this appendix, we document the Action Agencies' progress with respect to the management plans and current estimates of smolt predation rates.

B.2 Effects of Avian Predator Colonies in the Columbia River Estuary

B.2.1 Caspian Terns Nesting on East Sand Island

Terns first nested on East Sand Island in the lower Columbia River estuary in 1984, following the deposition of fresh dredged material at the eastern tip of the island in 1983. By 1985, vegetation covered the nesting site making it unsuitable for terns and by 1986 the colony had shifted to Rice Island, another dredged-material disposal site 16 miles upriver. In 1999 and 2000, the Corps' used social attraction mechanisms (decoys and pre-recorded callbacks) to move terns back to East Sand Island from Rice Island, in order to decrease the numbers of juvenile salmon and steelhead consumed by the terns (USACE 2015b).

This work was challenged under the National Environmental Policy Act (NEPA) by the Seattle Audubon Society, National Audubon Society, American Bird Conservancy, and Defenders of Wildlife. In 2002, the parties involved in the lawsuit reached a settlement agreement, which allowed the Corps' to continue to use social attraction devices to induce the terns to nest on East Sand Island, but also required the Corps' and U.S. Fish and Wildlife Service (USFWS) to produce an Environmental Impact Statement (EIS) for a plan to manage the terns in the long term. Subsequently, the federal agencies completed the Caspian Tern Management to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary Final Environmental Impact Statement (USFWS et al. 2005). The USFWS and Corps' each issued their own records of decision (RODs) in 2006 (USFWS 2006, USACE 2006). Collectively, these documents are called the Caspian Tern Management Plan.

The Caspian Tern Management Plan called for the redistribution of 60 percent of the East Sand Island colony to new habitat (islands) to be constructed in Oregon, California, and Washington at a nesting area ratio of 2 new acres per acre reduction on East Sand Island. Because Caspian terns nested on an average of five acres from 2001 to 2004 on East Sand Island, approximately seven to eight acres of new suitable habitat would need to be created to reduce the East Sand Island habitat by 1 to1.5 acres (USFWS et al. 2005). Plans to create new habitat in Washington State were unattainable so a modified alternative was selected that involved constructing seven acres of new habitat in Oregon and California and reducing East Sand Island habitat to 1.5 to 2 acres so that a larger number of terns would remain on the island (3,125 to 4,375 pairs, assuming an average nesting density of 0.55 pairs/m^2).

Despite the subsequent reduction in nesting habitat on East Sand Island, the tern population continued to increase, with the number of nesting pairs on the island in 2013 (7,000) and 2014 (6,200) at twice that predicted in the 2005 EIS (USFWS et al. 2005) and in the Corps' 2006 Record of Decision (USACE 2015b). The Corps' responded by preparing a smaller habitat area (1 acre) to reduce the size of the tern colony as prescribed in the Caspian Tern Management Plan with the expected outcome of 3,125 to 4,375 breeding pairs. Even with the Corps' efforts, terns further increased their nesting density at East Sand Island to 1.36 nests/m^2 in 2016 (Roby et al. 2018). Thus, numbers of breeding pairs remained above the Caspian Tern Management Plan's objective of 3,125 to 4,375 pairs.

The situation on East Sand Island changed in 2017. After increasing for a decade, the Caspian tern nesting density declined, dropping to 0.97 nests/m^2, with a peak size of about 3,500 breeding pairs in early June 2017. However, this decline in nesting at East Sand Island was met by an increase in tern interest at Rice Island, 16 miles upriver, where about 1,000 pairs of terns roosted, or tried to nest.[2] The increase in activity at Rice Island indicated that some of the birds displaced from East Sand Island had remained within the estuary, but were too far from the mouth of the river to include a larger proportion of marine forage fishes in their diet (i.e., diet was almost entirely salmonids). The Corps has increased dissuasion efforts and was able to keep terns from successfully nesting on Rice Island during the 2015 to 2019 nesting seasons (Harper and Collis 2018, USACE 2019).

By mid-June 2017, the size of the tern colony on East Sand Island was declining rapidly and all nesting attempts had failed (Roby et al. 2018). Terns completely abandoned the colony at the end of the month and stayed away for 10 days, an unprecedented event at this location. A smaller wave of nesting activity began in July and continued through early September with researchers reporting up to several hundred nests with eggs present on the colony at one time.

[2] Caspian tern activity at Rice Island had increased in 2015 following the placement of dredged material. The Corps built a berm on the downstream portion of the island in September 2015 to reduce line-of-sight visibility from areas where terns were prospecting for nests to the river. This reduced Caspian tern loafing or roosting in the upland placement area from about 6,000 individuals in 2015 to about 1,000 in 2017 (Evans et al. 2018a). A small proportion of these birds laid eggs, all of which were collected under annual depredation permits from USFWS.

However, these nests also failed and no Caspian tern young were raised on East Sand Island in 2017. This nesting failure could have been related to an increase in the rate of disturbance by bald eagles (*Haliaeetus leucocephalus*) and gulls (*Larus* spp.). Eagles harass terns to get them to drop fish they are bringing back to the nest (kleptoparasitism), and when other terns leave their nests during an eagle disturbance, gulls prey on the exposed eggs and chicks.

In 2018, the estimate of peak of peak colony size was 4,959 breeding pairs (Roby et al. 2019). This was significantly more than the estimate of 3,500 breeding pairs in 2017, but much lower than any other year since 2000 when the colony relocated to East Sand Island. Nevertheless, the estimated size of the East Sand Island tern colony in 2018 was still substantially larger than the target colony size of up to 4,375 breeding pairs specified in the management plan (USACE 2015). In 2018, nesting density was 1.23 nests/m^2, significantly higher than the 0.97 nests/m^2 in 2017, but lower than the 1.36 nests/m^2 in 2016 (Roby et al. 2019). Nesting density was slightly lower (was 1.11 nests/m^2) in 2019 (Roby et al. 2020). Peak colony size was reached in mid-June, 2019, at 3,860 breeding pairs, within the range targeted by the management plan (3,125 to 4,375 pairs).

C.2.1.1 Smolt Predation Rates by East Sand Island Caspian Terns

Evans and Payton (2020a) compare average annual predation rates before and after initiation of the Caspian Tern Management Plan (Table B-1). The findings indicate that predation rates have been, on average, significantly lower during the management period (2008 to 2018) than before management (2000 to 2007). Predation rates on steelhead, although variable, were linearly related to colony size, indicating that management actions to reduce numbers of terns on East Sand Island have resulted in lower annual predation rates at this colony (Evans et al. 2018a).

The presence of approximately 1,000 terns attempting to nest at Rice Island in 2017 (Evans et al. 2018a), and smaller numbers roosting or trying to nest on Rice, Miller, and Pillar Islands in 2018 and 2019 (Harper and Collis 2018, USACE 2019), offset the lower predation rates at the East Sand Island colony to an unknown degree (Evans et al. 2018a). Because pre-management per capita predation rates on salmonids by nesting terns were two to three times higher at Rice Island than at East Sand Island, the USACE performs hazing and dissuasion activities to ensure that nesting attempts on Rice Island are not successful. The forage base at East Sand Island includes several species of marine fishes so that the same number of terns is likely to eat fewer salmonids at this location (Roby et al. 2002).

Table B-1. Average annual predation rates (with 95 percent credible intervals) on Snake River (SR), Upper Columbia River (UCR), Upper Willamette River (UWR), Mid-Columbia River (MCR), Lower Columbia River (LCR), and Columbia River (CR) salmonids by Caspian terns at East Sand Island. The pre-management time period is defined as the period before actions to reduce colony size were first implemented. Comparable estimates are not available (NA) for some time periods. An asterisks (*) indicates statistically credible differences between management periods for a salmonid species. Source: Evans and Payton (2020a).

Salmonid ESU/DPS	Pre-management Period 2000-2007	Management Period 2008-2018
SR sockeye salmon[a]	NA	1.8% (1.4-2.2)
SR spring/summer Chinook salmon	5.2% (4.6-6.0)	2.1% (1.9-2.4)*
UCR spring Chinook salmon	4.3% (3.7-5.1)	1.9% (1.6-2.2)*
SR fall Chinook salmon	2.9% (2.4-3.4)	1.0% (0.8-1.2)*
SR steelhead	25.3% (22.7-28.3)	10.7% (9.8-12.0)*
UCR steelhead[b]	17.2% (15.2-19.5)	11.0% (10.0-12.5)*
MCR steelhead[c]	17.1% (14.0-22.0)	10.1% (9.1-11.4)*
LCR Chinook salmon[d]	4.1% (3.2-5.6)	2.5% (2.2-2.8)*
LCR steelhead[e]	15.2% (11.7-20.7)	10.4% (9.4-11.4)*
LCR coho salmon[f]	2.6% (1.4-4.3)	3.1% (2.6-3.8)
CR chum salmon	NA	NA
UWR Chinook salmon[g]	1.4% (0.7-2.4)	1.7% (1.3-2.1)
UWR steelhead	NA	NA

[a] Predation rate estimates for SR sockeye salmon were not available in 2000-2008 and in 2016-2017.

[b] Predation rate estimates for UCR steelhead were not available in 2001.

[c] Predation rate estimates for MCR steelhead were not available in 2000-2006.

[d] Predation rate estimates for LCR Chinook salmon were not available in 2000-2006.

[e] Predation rate estimates for LCR steelhead were not available in 2000-2006.

[f] Predation rate estimates LCR coho salmon were not available in 2000-2006 and in 2012-2016.

[g] Predation rate estimates for UWR Chinook salmon were not available in 2000-2006 and in 2017.

C.2.1.1.1 Tern Predation on Lower Columbia River Chinook and Coho Salmon and Steelhead

The predation rates in Evans and Quinn (2020) are based on the number of passive integrated transponder (PIT) tags detected on East Sand Island as a proportion of the number of tagged smolts that passed Bonneville Dam (or Sullivan Dam at Willamette Falls for Upper Willamette River Chinook salmon). Relatively few PIT-tagged smolts from Lower Columbia River ESUs and DPSs pass Bonneville Dam—which lies upriver from most of the spawning and rearing areas used by these species—so this method has not previously been applied. However, Sebring et al. (2013) estimated predation rates on specially PIT-tagged subyearling hatchery smolts from the Lower Columbia River (LCR) Chinook salmon ESU during 2002 to 2010. After recovering PIT tags from the East Sand Island tern colony, they estimated a minimum average annual predation rate[3] on these hatchery smolts of 4 percent. Sebring et al. (2010) also estimated a 3 percent minimum predation rate on PIT-tagged hatchery-origin LCR coho salmon by terns nesting at East Sand Island.

[3] Sebring et al. (2013) provide minimum predation rates because their estimates were not corrected for PIT-tag detection probabilities on East Sand Island. In addition, the samples were not representative of the Lower Columbia River Chinook salmon ESU at-large because they consisted only of subyearling hatchery-origin fish.

More recently, Evans and Payton (2020a) estimated Caspian tern predation rates for LCR Chinook salmon, LCR coho salmon, and LCR steelhead using PIT tag recoveries adjusted for deposition rates as estimated in the past for interior Columbia salmonids (e.g., Evans et al. 2019a). Evans and Payton (2020a) estimate a tern predation rate on LCR Chinook salmon for the pre-management period (4.1 percent) that is very similar to the minimum predation rate estimate (4.0 percent) in Sebring et al. (2013). However, Evans and Payton (2020a) report much higher average tern predation rates on LCR steelhead: 15.2 percent on LCR steelhead before the start of colony management on East Sand Island and 10.4 percent during the management period. Their estimate of predation rates on LCR coho salmon appear unchanged since the start of tern colony management (3.1 percent compared to 2.6 percent during the pre-colony management period; the difference is not statistically credible, Table B-1)).

C.2.1.2 Summary—Impacts of Caspian Terns in the Columbia River Estuary

The nesting attempts by terns on Rice Island in recent years indicate that this species' response to habitat reduction on East Sand Island has been in flux. However, the Corps has adjusted dissuasion efforts at dredge material islands in the lower Columbia River to ensure that terns do not nest successfully except on East Sand Island. The long term success of the management plan in reducing smolt predation is likely to depend on whether nesting densities remain low on East Sand Island and, if they do, whether birds move to areas outside the Columbia River basin rather than upstream to sites like Rice Island or the interior Columbia River plateau. Resightings of previously banded Caspian terns during the 2017 and 2018 nesting seasons showed that some moved from the estuary to the plateau or to Puget Sound, but that others were still coming to East Sand Island from other colonies in the Columbia basin and elsewhere in the Pacific Flyway (Roby et al. 2018, 2019a). One tern that was banded as a fledgling on East Sand Island and resighted there in 2016, was later seen at Corps'-constructed islands at Don Edwards National Wildlife Refuge in San Francisco Bay in 2017. The opposite occurred in 2018 when a tern banded in northern San Francisco Bay was spotted on East Sand Island (Roby et al. 2019). Given these movements, it may take more time for the colony on East Sand Island to stabilize. Based on the information in the upcoming Avian Predation Synthesis Report, NMFS will discuss with the state and tribal fish and wildlife managers whether the current colony management effort is adequate or whether the region should evaluate additional management actions.

The average annual predation rates reported in Evans and Payton (2020) indicate that these have been, on average, significantly lower during the management period (2008 to 2018) than before management (2000 to 2007). Predation rates on steelhead, although variable, were linearly related to colony size, indicating that management actions to reduce numbers of terns on East Sand Island have resulted in lower annual predation rates at this colony (Evans et al. 2018a). This is significant because steelhead are especially vulnerable to tern predation.

B.2.2 Double-crested Cormorants Nesting on East Sand Island

The double-crested cormorant colony on East Sand Island increased nearly threefold during 1997 to 2013 to about 14,900 breeding pairs (Turecek et al. 2018). The estimated per-capita smolt consumption by cormorants on East Sand Island was about four times higher than that of Caspian terns before management, both due to the larger number of breeding pairs and the higher food requirement of larger individual cormorants (Roby et al. 2013). Under 2008 RPA action 46 (as modified in the 2014 Supplemental FCRPS biological opinion), the Corps' developed the *Double-crested Cormorant Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary* (Cormorant Management Plan). The Cormorant Management Plan (USACE 2015a) called for a two-phased approach:

- Phase I – Reduce East Sand Island colony to 5,380-5,939 breeding pairs by implementing primarily lethal methods to reduce the population (i.e., four years of culling adults and three years of nest loss through egg oiling).
- Phase II – Transition to lower maintenance, primarily non-lethal techniques to ensure colony size does not exceed 5,380-5,939 breeding pairs. This would be accomplished via habitat modifications to reduce the availability of nesting habitat, supported with human hazing and limited egg take (500 eggs on East Sand Island and 250 at dredged placement sites in the upper estuary) to support the objectives of habitat modification and ensure the colony size does not exceed management objectives.

Phase I of the effort was expected to last four years, ending with a colony size of about 5,600 breeding pairs. However, the dispersal and subsequent nesting failure of this colony in 2016 and 2017 (see below) triggered the decision to end culling after April 2017, the third year of the implementation (see Table 5-3 in USACE 2015b). Early implementation of Phase II therefore began in 2018.

B.2.2.1 Double-crested Cormorant Colony Abandonment and Dispersal in 2016 and 2017; Late Nesting in 2018

In 2016, double-crested cormorants began nesting on East Sand Island in late April, but were absent between the week of 17 May and late June (Anchor QEA 2017). Researchers suspended ground surveys to avoid disrupting the remaining individuals, but reinitiated surveys when the larger numbers of birds returned. Peak numbers (19,544 double-crested cormorants; 9,772 nests) were present during the first week of July (Anchor QEA 2017). During the period when cormorants were absent from East Sand Island, many were observed on the Astoria-Megler Bridge, about 9 miles upriver. Counts on the bridge peaked at over 4,000 cormorants and about 550 nests in mid-June 2016 (Anchor QEA 2017).

Another abandonment and dispersal event occurred during the 2017 breeding season. Cormorants began staging and loafing on the beaches at the western end of East Sand Island in mid-April, but were unable to establish a colony (Turecek et al. 2018). They did have two brief periods of attendance during mid-May and early-June, but frequent disturbances by bald eagles resulted in partial or complete flushes to nearby beaches or to rafts on the water, usually in nearby Baker Bay. After a dispersal event in early June, cormorants did not sustain colony attendance except for 544 nests late in the season, but these did not appear to fledge any chicks. However, they did nest successfully on the Astoria–Megler Bridge with a peak count of about 6,000 individuals and more than 800 nests on 11 July (MacDonald 2017). Smaller numbers nested on the Lewis and Clark Bridge (Longview, WA); aids to navigation between Tongue Point, Oregon, and Skamokawa, Washington; and the electrical transmission towers near the Sandy River delta, Oregon.

The nesting chronology of the East Sand Island cormorant colony was roughly one month delayed in 2018 compared to that in 2004 to 2014, before implementation of the Management Plan (Turecek et al. 2019). Cormorants began loafing on the north beaches of East Sand Island in large numbers on 15 April, were first seen in the designated colony area on 9 May, and began roosting overnight on 15 May. Frequent predation pressure from bald eagles limited formation of the colony until early July. Cormorants responded to these disturbances by dispersing to nearby beaches and elsewhere off colony and/or retreating from bald eagles that landed within the colony. During flushes, bald eagles and gulls would walk through the colony and depredate eggs in the nests left unattended. Ultimately, nesting success in 2018 was estimated to be 1.8 young raised/active nest, identical to the average nesting success for the double-crested cormorant colony on East Sand Island before management (Turecek et al. 2019). In addition, a peak number of 1,736 breeding pairs nested on the Astoria-Megler Bridge in 2018 (Turecek et al. 2019). Altogether, a total of 8,485 pairs of double-crested cormorants attempted to breed in the Columbia River estuary in 2018, far higher than the average 5,660 breeding pairs envisioned by the management plan.

The Corps reduced monitoring effort at the East Sand Island double-crested cormorant colony in 2019, obtaining estimates of colony attendance and nesting success from aerial photographs. The USDA APHIS staff conducting hazing outside the established “sanctuary” at the west end of the island also provided occasional on-the-ground observations.

B.2.2.1.1 Phase II of the Double-crested Cormorant Management Plan: Terrain Modification at East Sand Island

The Corps implemented terrain modification at East Sand Island, as anticipated in the Double-crested Cormorant Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary, between December, 2018, and March, 2019. About 125,000 cubic yards of upland material were excavated from an embayment on the western end of East Sand Island, reducing the elevation to 7.5 feet below the mean higher high water elevation. The length of the existing rock revetment along the southern shoreline was increased by 2,800 linear feet and a 600 linear foot revetment was built along the western shoreline to support integrity of the island and Columbia River Federal Navigation Channel. The purpose of the terrain modification project was to allow for more frequent inundation of the island, reducing the extent of habitat suitable for nesting by double-crested cormorants.

The Corps reported a peak population of about 4,500 individual double-crested cormorants occupying and attempting to nest on East Sand Island during the month of June, with a peak of 350 nests observed on June 25, but all were abandoned four days later (Scalfani 2019). The first chicks were observed in early September and no more than 51 nests successfully fledged young. In comparison, a peak number of 3,542 active double-crested cormorant nests were observed on the Astoria-Megler Bridge on June 4, 2019, more than double the number seen in previous year (Scalfani 2020).

B.2.2.2 Smolt Predation Rates by East Sand Island Cormorants

Before 2016, most double-crested cormorants in the estuary nested on East Sand Island and researchers were able to estimate predation rates for the estuary as a whole based on PIT-tag recoveries at that location. Since 2016, however, large numbers of cormorants have abandoned East Sand Island and moved to the Astoria–Megler Bridge and other locations upstream for much of the breeding season (MacDonald 2017, Turecek et al. 2019). Estimates of smolt consumption calculated by the researchers for 2016 through 2018 therefore must be considered underestimates because cormorants spent little time on the island during the peak smolt outmigration (Evans et al. 2018a). Although Evans and Payton (2020a) show predation rates by cormorants on East Sand Island during 2016 to 2018 that are lower (and statistically significant) compared to those before management actions were initiated (Table B-2), the post-management estimate of impact may be less than half that of the large number of birds foraging from the Astoria-Megler Bridge.[4]

[4] Collis et al. (2002) observed that double-crested cormorants nesting on the Astoria-Megler Bridge were more likely to feed in the upstream areas used by those nesting on Rice Island in 1997 to 1998. Those cormorants consumed three times more salmonids than those on East Sand Island because their foraging range did not overlap or overlapped to a lesser degree with marine fishes such as herring, surfperch, and flounder in the lower estuary. Cormorants nesting on the Astoria-Megler Bridge, about midway between Rice and East Sand Islands, may still be able to take advantage of marine forage fish, but to an unquantified amount.

Table B-2. Average annual predation rates (with 95 percent credible intervals) by double-crested cormorants nesting on East Sand Island prior to (Pre) and following (Post) implementation of management actions. Management actions included lethal take of eggs and adults and passive dissuasion during Phase I and egg take and passive dissuasion (only) during Phase II. An asterisks (*) indicates statistically credible differences between management periods for a salmonid species. NA denotes that estimates were not available during that time period Source: Evans and Payton (2020a).

Salmonid ESU/DPS	Pre	Post, Phase I[a]	Post, Phase II[a]
	2003-2014	2015-2017	2018
SR sockeye salmon[b]	4.2% (3.3-5.3)	2.4% (1.4-4.0)	0.9% (0.4-1.9)*
SR spr/sum Chinook salmon	4.6% (4.1-5.3)	6.8% (5.3-9.4)	0.5% (0.3-0.8)*
UCR spr Chinook salmon	3.8% (3.2-4.6)	4.1% (3.2-5.8)	0.6% (0.3-1.2)*
SR fall Chinook salmon	2.7% (2.3-3.2)	3.7% (2.6-5.4)	0.9% (0.5-1.6)*
SR steelhead	7.2% (6.3-8.5)	6.8% (5.3-9.4)	0.5% (0.3-0.9)*
UCR steelhead	6.3% (5.5-7.2)	5.8% (4.5-8.1)	0.7% (0.4-1.4)*
MCR steelhead[c]	7.5% (6.3-9.3)	5.4% (4.0-7.7)	0.4% (0.1-1.0)*
LCR Chinook salmon[d]	27.5% (24.3-30.7)	8.7% (6.2-12.1)*	7.3 (4.8-11.6)*
LCR steelhead[e]	5.4% (4.5-6.3)	5.0% (3.7-6.9)	0.6% (0.3-1.0)*
LCR coho salmon[f]	15.0% (12.2-18.1)	0.2% (0-0.7)*	0.3% (0.1-0.8)*
CR chum salmon	NA	NA	NA
UWR Chinook salmon[g]	1.8% (1.3-2.6)	1.4% (0.6-2.9)	NA
UWR steelhead	NA	NA	NA

[a] Predation rate estimates during the post-management periods are minimum estimates due to en masse dispersal or redistribution events or because colony formation was delayed until after the peak smolt outmigration period (Evans and Payton 2020a).

[b] Predation rate estimates were not available in 2003 to 2008 and in 2016 to 2017.

[c] Predation rate estimates were not available in 2003 to 2006 and in 2017.

[d] Predation rate estimates were not available in 2003 to 2006.

[e] Predation rate estimates were not available in 2003 to 2006.

[f] Predation rate estimates were not available in 2003 to 2006.

[g] Predation rate estimates were not available in 2003 to 2006 and in 2012 to 2015.

[h] Predation rate estimates were not available in 2003 to 2006 and in 2017.

B.2.2.1.1 Cormorant Predation on Lower Columbia River Chinook and Coho Salmon and Steelhead

Sebring et al. (2013) estimated predation rates by double-crested cormorants on PIT-tagged subyearling hatchery Chinook salmon from the LCR Chinook salmon ESU during 2002 to 2010. Based on PIT-tag recoveries from East Sand Island, minimum predation rates on these hatchery smolts averaged 10 percent. Lyons et al. (2014) estimated an average predation rate of 26 percent on LCR Chinook salmon during 2007 to 2010. Predation rates differed by rearing type, averaging 29 percent on hatchery-origin Chinook salmon and 11 percent on natural-origin Chinook salmon.

Sebring et al. (2012) estimated a 10 percent minimum predation rate on PIT-tagged hatchery-origin LCR coho salmon by cormorants nesting at East Sand Island in 2010. Lyons et al. (2014) estimated an average predation rate of 28 percent on the LCR coho salmon ESU by this colony during 2007 to 2010, weighting estimates by the relative abundances of hatchery- and natural-origin fish originating upstream and downstream of Bonneville Dam. Predation rates differed by rearing type, averaging 30 percent on hatchery-origin and 10 percent on natural-origin coho salmon.

More recently, Evans and Payton (2020a) estimated double-crested cormorant predation rates for LCR Chinook salmon, LCR coho salmon, and LCR steelhead using the same PIT tag analyses (i.e., including adjustments for deposition rates) they have performed in the past for interior Columbia salmonids (see Table B-2). The estimate for cormorant predation on LCR Chinook salmon for the pre-management period (27.5 percent) is much higher than the minimum predation rate (10 percent) described in Sebring et al. (2013), but similar to that in Lyons et al. (2014), 28 percent. Evans and Payton (2020a) reported somewhat lower cormorant predation rates on LCR coho salmon: 15.0 percent during the pre-management period and <1 percent during each of the post-management periods. The pattern for LCR steelhead differed—an average predation rate of 5.4 percent by cormorants on East Sand Island before the start of colony management and 5.0 and 0.6 percent during the Phase I and Phase II post-management periods, respectively. As described above for the interior species, due to the movement of large numbers of cormorants from East Sand Island to the Astoria–Megler Bridge beginning in 2016, data for the pre- and post-management periods are not directly comparable.

B.2.2.3 Summary—Impacts of Double-crested Cormorants in the Columbia River Estuary

The movement of double-crested cormorants from the East Sand Island colony to the Astoria-Megler Bridge in 2016 to 2018 indicates that this species' response to colony management activities and frequent disturbance by bald eagles, often followed by gulls taking eggs and chicks, remains in flux. The terrain modification action in Phase II of the management plan was completed before the 2019 nesting season to ensure that this colony does not exceed the management plan objective of no more than 5,380-5,939 nesting pairs. At the same time, the number of birds nesting on the Astoria–Megler Bridge has continued to grow so that an estimated 6,319 nesting pairs (including 4,103 on East Sand Island and 1,736 on the Astoria-Megler Bridge) were in the estuary during 2018 and up to 4,666 (including 399 on East Sand Island and 3,542 on the Astoria Megler Bridge) in 2019 (Scalfani 2020). As a result of these movements to upstream sites, the predation rates shown in Table B-2, which are based on PIT tag collections from East Sand Island, underestimate double-crested cormorant predation pressure in the estuary as a whole.

B.2.3 Summary—Effects of Avian Predator Colonies in the Columbia River Estuary

The nesting attempts by up to 1,000 terns on East Sand Island in the last few years indicate that these birds are moving around the estuary in response to habitat reduction on East Sand Island. The Corps has increased dissuasion efforts to keep prospecting terns from nesting on Rice Island and, at the same time, the numbers of nesting pairs on East Sand Island have declined. These are indications that the tern management plan, in combination with predation pressure from bald eagles and gulls, has been successful at reducing the number of terns in the estuary and thus smolt predation. Continued monitoring indicates that this pattern is continuing. If it does not, additional management actions may be needed to protect listed salmonids. The movement of cormorants from East Sand Island to nesting sites farther upstream in the estuary over the same period, whether as a result of bald eagle pressure or management actions, indicates that implementation of the management plan may not have reduced predation rates on listed salmonids.

B.3 Effects of Avian Predator Colonies on the Interior Columbia Plateau

B.3.1 Management Activities at Inland Caspian Tern Colonies

Predation on salmonids by piscivorous waterbirds nesting in the Columbia basin upstream of Bonneville Dam (i.e., interior Columbia plateau) became a concern when colonies became established at Crescent Island in McNary Reservoir and on Goose Island in Potholes Reservoir. Roby et al. (2011) estimated total predation on salmonids at Crescent Island during 2004 to 2009 of 330,000 to 500,000 smolts per year. Annual predation rates on UCR steelhead by terns nesting on Goose Island (Potholes Reservoir) averaged 15.7 percent during 2007 to 2013 (Collis et al. 2018). As a result of these impacts, NMFS required the Corps and Reclamation to develop an Inland Avian Predation Management Plan (IAPMP) in RPA action 47.

The objective of the IAPMP is to reduce predation on ESA-listed salmonids by Caspian terns nesting at Goose and Crescent Islands while taking actions to prevent terns from forming new colonies and/or expanding existing colonies where feasible (USACE 2014). In general, the IAPMP aims to reduce predation on interior Columbia basin salmonids to less than 2 percent for each listed ESU/DPS per tern colony per year. The primary management goal during the first phase was to fully dissuade terns from nesting on Goose Island with a long term goal of reducing the colony to less than about 40 breeding pairs (to meet the <2 percent predation rate objective). During the first year of implementation (2014), The Corps set up passive dissuasion (ropes and flagging) and conducted active hazing. However, 156 pairs nested on a nearby islet called Northwest Rocks (Roby et al. 2015). Terns also tried to nest at other colony sites they had used previously (Crescent Island in McNary Reservoir, the Blalock Islands in John Day Reservoir, Twinning Island in Banks Lake, and Harper Island in Sprague Lake) and a small island in Lenore Lake. Only the colonies at Crescent Island and the Blalock Islands succeeded in raising young. The colony on Crescent Island was the largest in the interior Columbia plateau region that year (474 breeding pairs).

The second phase of the IAPMP called for development of suitable alternative Caspian tern nesting habitat in areas where predation on ESA-listed species would be lower before dissuading terns from Crescent Island. Similar to efforts on Goose Island, the short-term goal was to dissuade terns from nesting with a long-term goal of less than about 40 nesting pairs (to achieve the less than 2 percent predation rate objective) (USACE 2014). The Corps identified a potential site at Don Edwards San Francisco Bay National Wildlife Refuge and modified this area for tern nesting during winter 2014 to 2015, so the Action Agencies were able to begin dissuading terns from Crescent Island during the 2015 breeding season. Passive dissuasion. Hazing successfully prevented nesting on Crescent Island and most of the nesting at Goose Island. However, the number of terns at the Blalock Islands was ten times higher in 2015 than the year before and resightings of colored leg-bands indicated that large numbers had moved there from Crescent Island (many of these individuals nested at the Blalock Islands again in 2016). Terns also came to the interior plateau from East Sand Island in the estuary, and from additional Corps'-constructed colony sites in southeastern Oregon and northeastern California in 2015 when those areas experienced severe drought.

Terns displaced from Crescent Island continued to relocate to the unmanaged colony sites at the Blalock Islands and to a limited degree (i.e., below the 40 pairs per colony threshold) at Badger Island in 2017 (Collis et al. 2018). Overall, the number of pairs of Caspian terns at each colony in the interior Columbia plateau region during 2017 represented a 19 percent decline compared to the pre-management period (Figure B-1). Numbers of pairs were the same or lower at these colonies in 2018 when an estimated 491 breeding pairs of Caspian terns nested at four breeding colonies (Blalock Island, Badger Island, Harper Island in Sprague Lake, and an unnamed island in Lenore Lake; Collis et al. 2019). This represented a 44 percent decline in the size of the breeding population compared pre-management average (2005 to 2013) and a 28 percent decline when compared to the management period (2014 to 2017).

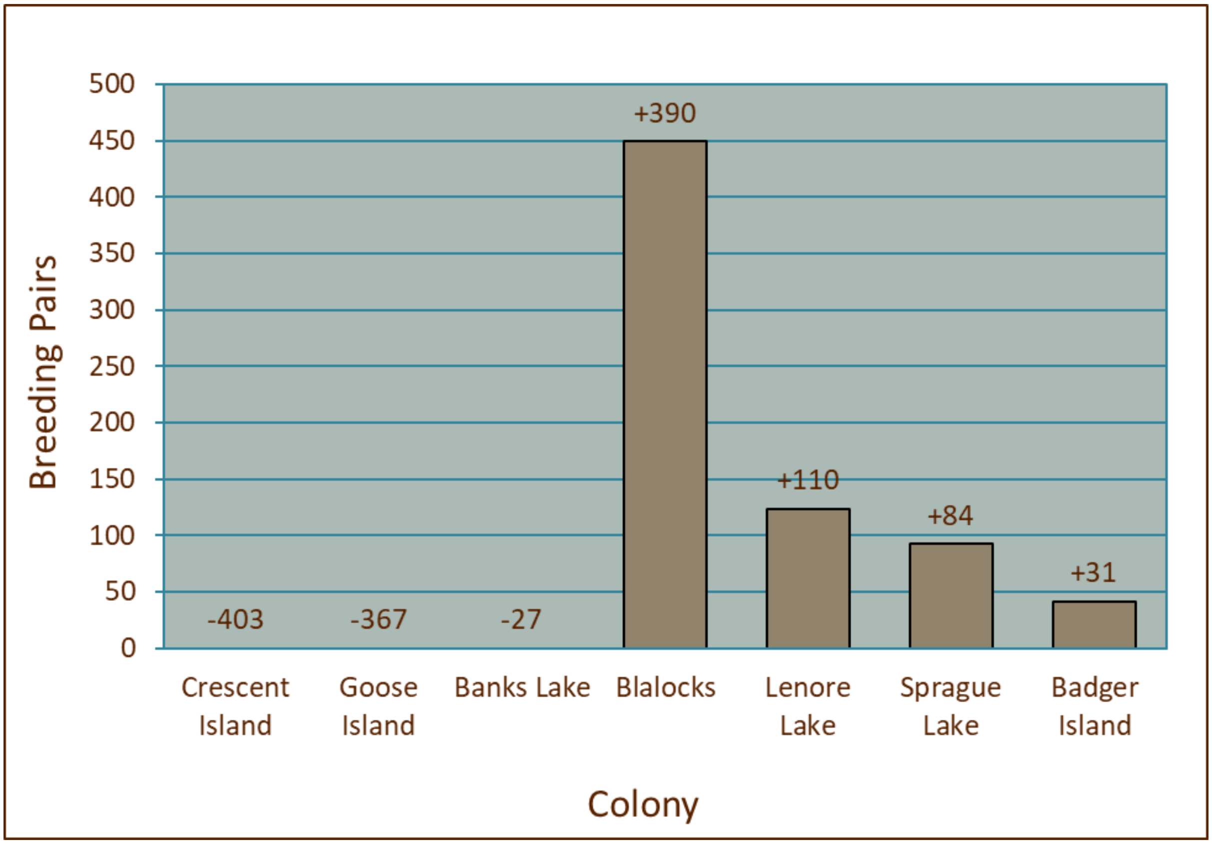

Figure B-1. Sizes of Caspian tern nesting colonies (numbers of breeding pairs) in the interior Columbia plateau region during the 2017 breeding season. The number over each bar indicates the change in colony size in 2017 compared to the average colony size before tern management (2005 to 2013). Source: Collis et al. (2018).

B.3.2 Salmonid Predation Rates at Interior Columbia Plateau Tern Colonies

In 2017, the goal of the IAPMP to reduce ESU/DPS-specific predation rates to less than 2 percent was achieved at Goose Island for the third consecutive year and Crescent Island for the fourth (Collis et al. 2019). As a result, Caspian tern predation on UCR steelhead has declined from about 2 to 15 percent at these three managed colonies (Table B-3). However, predation rates on other ESA-listed salmonids (especially SR sockeye salmon and SR steelhead) by terns on the Blalock Islands have been higher since colony management began at Goose and Crescent Islands (Table B-4).

Table B-3. Average annual predation rates by Caspian terns at managed colonies in the interior Columbia plateau region prior to (Pre) and following (Post) implementation of management actions. Only ESA-listed salmonids migrating from the Snake River (SR) and Upper Columbia River (UCR), including steelhead, sockeye salmon, and spring (spr), summer (sum), and fall run Chinook salmon, are within foraging distance of these colonies. NC denotes that no colony existed during a time period. Management actions were implemented on Goose Island in Potholes Reservoir during 2014 to 2018, on an unnamed island in northeastern Pothole Reservoir in 2017 to 2018, and on Crescent Island during 2015 to 2018. Source: Evans and Payton (2020a)

ESU/DPS	Goose Island			North Potholes Island			Crescent Island		
	Pre 2007-2013	Post 2014-2019	Last 3 Yrs 2017-2019	Pre 2007-2013	Post 2016	Last 3 Yrs 2017-2019	Pre 2007-2014	Post 2015-2019	Last 3 Yrs 2017-2019
SR Sockeye	<0.1%	<0.1%	<0.1%	NC	<0.1%	<0.1%	1.5% (1.2-2.0)	<0.1%	<0.1%
SR spr/sum Chinook	<0.1%	<0.1%	<0.1%	NC	<0.1%	<0.1%	0.8% (0.7-1.0)	<0.1%	<0.1%
UCR spr Chinook	2.5% (1.7-3.6)	<0.1%	<0.1%	NC	0.1% (0.1-0.3)	<0.1%	0.5% (0.3-0.9)	<0.1%	<0.1%
SR fall Chinook	<0.1%	<0.1%	<0.1%	NC	<0.1%	<0.1%	1.0% (0.9-1.2)	<0.1%	<0.1%
SR steelhead	<0.1%	<0.1%	<0.1%	NC	<0.1%	<0.1%	4.5% (4.1-5.1)	<0.1%	<0.1%
UCR steelhead	15.7% (14.1-18.9)	<0.1%	<0.1%	NC	4.1% (2.9-6.3)	<0.1%	2.5% (2.2-2.9)	<0.1%	<0.1%

Table B-4. Average annual predation rates (95% credible intervals) by Caspian terns at the Blalock Islands, and unmanaged colony, prior to (Pre) and following (Post) implementation of management activities at Goose and Crescent Islands. Impacts are shown for ESA-listed salmonids migrating from the Snake River (SR) and Upper Columbia River (UCR), including steelhead, sockeye salmon, and spring (spr), summer (sum), and fall Chinook salmon runs. Source: Evans and Payton (2020a)

ESU/DPS	Blalock Islands		
	Pre 2007-2013	Post 2014-2019	Last 3 Years 2017-2019
SR sockeye salmon	0.2% (0.1-0.4)	1.6% (1.0-2.5)	1.8% (0.7-4.0)
SR spr/sum Chinook salmon	0.1% (0.1-0.2)	0.7% (0.5-0.9)	0.6% (0.4-0.9)
UCR spr Chinook salmon	<0.1%	0.6% (0.5-0.9)	0.8% (0.5-1.3)
SR fall Chinook salmon	<0.1%	0.7% (0.6-1.1)	0.9% (0.6-1.4)
SR steelhead	0.5% (0.4-0.9)	3.7% (3.1-4.6)	3.1% (2.4-4.1)
UCR steelhead	0.5% (0.3-0.7)	4.3% (3.5-5.6)	4.5% (3.4-6.1)

B.3.3 Gull Colonies on the Interior Columbia Plateau

When the IAPMP (USACE 2014) was developed, estimates of salmonid predation by gulls nesting in the interior Columbia plateau region did not exceed 2 percent per listed ESU/DPS per colony per year (Table B-5). The largest impact by California and ring-billed gulls (*Larus californicus* and *L. delawarensis*, respectively) was from the colony nesting at Miller Rocks, a group of rock outcroppings and small islands in The Dalles Reservoir. The Corps concluded that, in comparison to Caspian terns nesting at Goose and Crescent Islands, the benefits to ESA-listed salmonids through reductions in predation by avian predators, such as gulls nesting on Miller Rocks, would be substantially lower (Lyons et al. 2011).

Table B-5. Average annual predation rates on ESA-listed Snake River (SR) and Upper Columbia River (UCR) salmonids by California and ring-billed gulls at Miller Rocks, The Dalles Reservoir, 2007 to 2010, adjusted to account for the fraction of each salmonid species transported around the interior Columbia plateau waterbird colonies as part of the Corps' juvenile salmonid transportation program. Sources: Lyons et al. (2011), USACE (2014).

Chinook			Sockeye	Steelhead	
SR spr/sum	**SR fall**	**UCR spr**	**SR**	**SR**	**UCR**
0.3%	0.3%	0.4%	0.6%	1.2%	1.6%

While implementing the IAPMP, the Corps' made an effort to prevent nesting by California and ring-billed gulls on Goose and Crescent Islands. This decision was based on the theory that gulls attract prospecting Caspian terns and, thus, could limit the efficacy of dissuasion efforts (USACE 2014; Roby et al. 2016). The effort was partially successful, with gulls dispersing from Crescent Island to add to numbers at the colonies on Island 20 (McNary Reservoir), Miller Rocks (The Dalles Reservoir), and the Central Blalock Islands (John Day Reservoir) in 2015 (the gull colony on Goose Island has remained relatively stable in recent years) (Collis et al. 2018). Gull predation rates, like those of terns, were generally higher for juvenile steelhead than for salmon (Table B-6).[9]

[9] Due to improvements in estimation methods, the predation rates in Table B-6 are more accurate than those in Table B-5.

Table B-6. Average annual predation rates (95 percent credible intervals) by unmanaged California and ring-billed gulls (LAXX) colonies on Island 20 and Badger Island (McNary Reservoir) on PIT-tagged Snake River (SR) and Upper Columbia River (UCR) salmonids. Source: Evans and Payton (2020a).

ESU/DPS	Island 20 2013-2019	Badger Is. 2015-2019	Crescent Is. 2007-2014	Blalock Is. 2013-2019	Miller Rocks 2007-2019
SR sockeye	0.9% (0.3-1.7)	2.7% (1.2-4.8)	2.2% (1.3-3.2)	2.0% (1.0-3.0)	6.2% (4.8-7.7)
SR spr/sum Chinook	0.2% (0.1-0.3)	0.5% (0.3-0.6)	1.0% (0.8-1.2)	0.2% (0.1-0.2)	1.2% (1.1-1.4)
UCR spr Chinook	0.5% (0.1-1.2)	1.6% (0.7-2.9)	1.2% (0.4-2.3)	0.4% (0.2-0.6)	2.1% (1.7-2.4)
SR fall Chinook	0.2% (0.1-0.4)	0.9% (0.5-1.4)	0.6% (0.4-0.9)	0.4% (0.3-0.6)	2.0% (1.8-2.4)
SR steelhead	1.6% (1.2-1.9)	3.3% (2.5-4.3)	4.8% (4.1-5.6)	2.4% (2.0-3.0)	7.2% (6.5-8.1)
UCR steelhead	4.1% (3.3-4.9)	5.0% (3.5-6.9)	5.8% (5.0-6.9)	3.9% (3.0-4.8)	8.2% (6.9-9.3)

The following average annual predation rates have exceeded 2 percent per listed ESU/DPS per gull colony per year:

- Island 20 – UCR steelhead;
- Badger Island – SR sockeye salmon, SR steelhead, and UCR steelhead;
- Crescent Island – SR sockeye salmon, SR steelhead and UCR steelhead;
- Blalock Islands – SR steelhead and UCR steelhead; and
- Miller Rocks – SR sockeye salmon, UCR spring Chinook salmon, SR steelhead, and UCR steelhead.

Consumption rates by gulls from colonies in the interior Columbia plateau region during 2015 were significantly higher than those observed at the same colonies in previous years, with a roughly two- to five-fold increase in some cases (Roby et al. 2016). Consumption rates for gulls nesting on Miller Rocks were the highest of any gull colony evaluated.

Further research is needed to understand whether gulls disproportionately consume weak or compromised smolts, especially near dams, or prey on fish from the general outmigrant population. In either case, smolt predation rates at certain gull colonies have continued to be some of the highest associated with any piscivorous waterbird colony in the interior Columbia Plateau region since multi-predator species studies were initiated in 2007. Management of gull predation is not addressed in the current avian predation management plans for the Columbia plateau or the estuary.

B.3.4 Summary—Impacts of Avian Predator Colonies on the Interior Columbia Plateau

As discussed in Collis et al. (2019), management actions to eliminate breeding colonies of Caspian terns on Goose Island in Potholes Reservoir and on Crescent Island in McNary Reservoir—formerly the largest breeding colonies for the species in the interior Columbia plateau region—were successful in 2017 and 2018. As a result, predation on juvenile salmonids by Caspian terns nesting at these two sites was effectively eliminated. Overall, numbers of breeding Caspian terns on the interior Columbia plateau decreased by 44 percent from pre-management levels due to the management of colonies on Goose and Crescent Islands through 2018. However, resightings of banded Caspian terns in previous years show that most terns that were displaced from Goose and Crescent Islands have remained in the region and many have tried to nest at unmanaged colony sites. Most notable has been the post-management increase in the size of the formerly small breeding colony in the Blalock Islands. Caspian terns nesting in the Blalock Islands during 2015 to 2018 consumed sufficient numbers of juvenile salmonids to at least partially off-set reductions in smolt consumption due to tern management at Goose and Crescent Islands. Based on results during the first five years of implementation of the IAPMP, tern predation rates in the interior Columbia are decreasing, but it appears that the over-all goal of the management plan to reduce predation rates to less than 2 percent per tern colony on ESA-listed ESU/DPS per year will not be fully realized until nesting habitat is reduced at the currently unmanaged colony sites, especially in the Blalock Islands. Under the 2020 proposed action (BPA et al. 2020), the Action Agencies will increase the normal forebay operating range at John Day Dam by 2 feet during April 10 through June 1 or June 15 to deter Caspian terns from nesting at the Blalock Islands Complex. The purpose of this operation is to reduce predation pressure on spring migrating, ESA-listed juvenile salmon and steelhead by deterring Caspian terns from nesting in the Blalock Islands Complex during this period.

In addition, average annual predation rates by gulls have exceeded those at tern colonies on the interior Columbia plateau, at least for SR and UCR steelhead. Reductions in gull predation rates at the colony level were considered not warranted when the IAPMP was developed and there are no regional plans to manage these colonies.

B.4 Is Caspian Tern or Double-crested Cormorant Predation Additive or Compensatory?

An unstated assumption in many predator control programs is that reducing predation during one life stage increases the survival of prey over a longer portion of its life cycle. In the current context, this would suggest that ensuring that fewer smolts are eaten by Caspian terns or double-crested cormorants during their outmigration would increase the number of adult returns (e.g., as measured by SARs). If so, avian predation would be considered an "additive" source of mortality. Alternatively, if the smolts "saved" from predation succumbed to other sources of mortality such as predators in the ocean, disease, or starvation, then avian predation would be considered a "compensatory" source of mortality.

The completely additive and completely compensatory hypotheses are illustrated in the left and right panels, respectively, of Figure B-2 (adapted by Evans et al. 2019b from Anderson and Burnham 1976 and Sandercock et al. 2011). The first graph shows an additive relationship between survival and predation rate (slope = -1). If tern and/or cormorant predation is a completely additive source of mortality for Columbia Basin salmonids, smolt survival will decrease linearly as tern or cormorant predation increases. However, if tern and/or cormorant predation is completely compensatory, then smolt survival would remain constant as predation by terns or cormorants increases up to a critical threshold, above which smolt survival would decline.

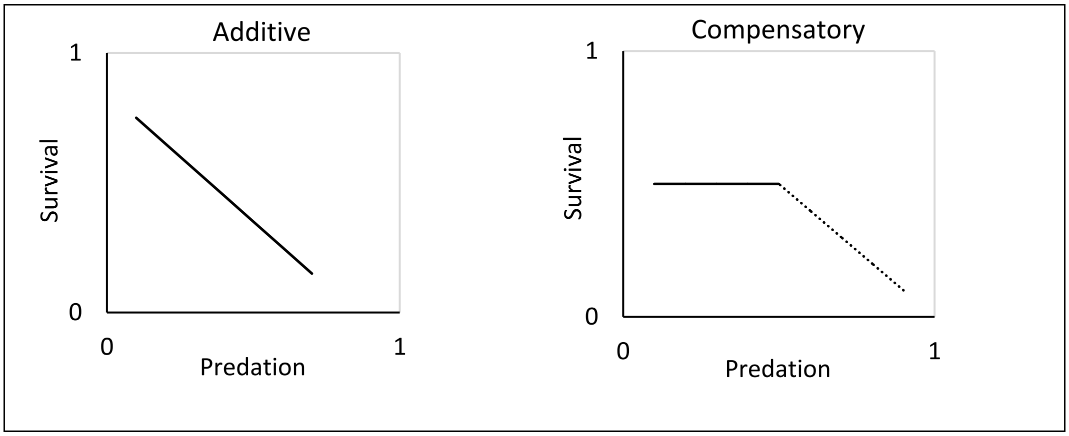

Figure B-2. Graphical representation of the completely additive and compensatory hypotheses for the effects of avian predation on the survival of salmonid populations to adulthood, adapted by Evans et al. (2019b) from Anderson and Burnham (1976) and Sandercock et al. (2011).

In recent years, researchers in the Columbia basin have asked if tern or cormorant predation is additive, or whether it is compensatory to some degree. If the latter, reducing predation rates by culling birds or reducing colony sizes may not translate into an increase in survival to adulthood. There is evidence that, at least for steelhead, fish condition, size, and rearing history may affect the vulnerability of fish to avian predation (Hostetter et al. 2012) and it is likely that predation losses to avian predators is somewhat compensatory due to these vulnerabilities. NMFS can use information on degree of compensation to assess whether the Action Agencies' avian predator control programs are affecting the number of returning adults. In the following sections, we review the modeling studies by Haeseker et al. (2020) and Evans et al. (2019b) that describe these relationships.

B.4.1 Estimating Correlations between Avian Predation Rates and Adult Returns for SRB Steelhead

Haeseker et al. (2020) applied a random effects model to a 16-year mark-recapture-recovery data set to assess whether predation by Caspian terns and double-crested cormorants on SRB steelhead smolts constituted an additive or compensatory source of mortality. Haeseker et al. (2020) state that a negative correlation between the avian predation and survival processes would be consistent with the hypothesis that avian predation is having an additive effect on survival. They found that, for both colonies, the estimated correlation between the predation rate and survival rate of steelhead was near zero, claiming that this indicates that mortality due to avian predation is compensatory–that smolts not eaten by terns and cormorants nesting on East Sand Island would have died anyway due to other causes.

Fish Management staff at the Columbia River Inter-Tribal Fish Commission have reviewed this paper, identifying several methodological concerns that weaken the results and cautioning against acceptance of the conclusions in this paper. The concerns expressed in Skiles (2020) are summarized here:

- **<u>Ocean survival</u>**
  - The authors assumed an annual survival rate of 0.8 for steelhead that remain in the ocean from the first to second year, but this value, which was published for salmon in Ricker (1976), may not be appropriate for steelhead.
  - Ocean conditions have changed dramatically since 1976 and it is unlikely that the conditions that resulted in a survival rate of 0.8 in 1976 are the same as in the period of this study (2000 to 2015 outmigrants and their first year returns).
  - The authors should have conducted a sensitivity analysis to their assumption that first to second year ocean survival is constant because, although most of the variation in ocean survival does occur in the first year, it is not true that first to second year survival never changes.
- **<u>Covariates for the survival model</u>**
  - The authors "evaluated 4 candidate environmental indices previously identified in other studies as being associated with salmonid survival," but only exploring models with 4 potential covariates to explain survival is too narrow for this statistical investigation. Other potential covariates would include fish length, avian colony size, and forage fish abundance.
  - Since all fish in this study were detected at Bonneville Dam as juveniles, a term indicating arrival day at Bonneville Dam was used as a covariate. This variable acts as a proxy for other covariates (e.g., spill percentage, temperature, and degree of smoltification) that change throughout the migration season. However, other studies that have used arrival day as a covariate also include non-linear effects of this variable by inclusion of a quadratic term. It is likely that inclusion of a quadratic term would have explained more variation in survival, but also may have diminished the significance of other covariates that were found to be statistically significant.
- **<u>Modeling survival in the covariate model</u>**
  - The model likely underestimates the true variation in survival by assuming that the survival term is constant within cohorts and across years. The authors built a multivariate normal model for survival and predation with a standard covariance matrix, but the right hand side of this equation is missing several terms. A true "full model" would take a more complicated form and would include additive effects of yearly and weekly cohorts and potentially multiplicative effects of year and cohorts.
- **<u>Hypothesis test of the correlation coefficient</u>**
  - The authors built a multivariate model with a standard covariance matrix. They interpreted estimates of the correlation coefficient that were near zero with credible intervals that overlapped zero as indicating compensatory mortality, and

negative estimates of the correlation coefficient with credible intervals that did not overlap zero as indicating additive mortality. However, the logic of this interpretation is not correct. The authors failed to reject the null hypothesis that mortality is compensatory as the estimated confidence intervals for the correlation coefficient included zero, but from a statistical hypothesis testing framework this does not imply that mortality is compensatory.

- **Assessing additive and compensatory mortality is difficult at this life stage**
    - The authors should have given more attention to the idea that it is very difficult to determine whether avian mortality is additive or compensatory given that this source of mortality is small compared to all other sources of mortality that occur throughout the life-cycle of steelhead.
    - The authors conducted their study over multiple years of the steelhead life-cycle. Conducting this study over a shorter period (i.e., for individual years) might have resulted in different conclusions.
    - A power analysis could have been conducted to determine what sample sizes were necessary to detect additive mortality if it indeed existed. Or, a simulation analysis could have been conducted where additive mortality was assumed in the simulated data and the models used by the authors evaluated to determine if they could detect this effect.

Although NMFS has not yet reviewed the Haeseker et al. (2020) paper to this extent, the concerns described in Skiles (2020) appear valid. We expect that this paper will receive additional regional review in the context of adaptive management for avian predation management at the East Sand Island and inland colony sites in the coming year.

B.4.2 Joint Mortality and Survival Model

Payton et al. (2020) also studied the relationship between avian predation and SARs, in this case using the Joint Mortality and Survival Model described in Payton et al. (2019). In contrast to Haeseker et al. (2020), these authors found "strong evidence that Caspian tern predation was an additive source of mortality for all spatial scales, years [2008 to 2015], and life-stages (smolt, SAR) evaluated." This modeling framework looks at the effect of tern predation at multiple colonies rather than just the single large colony on East Sand Island as in Haeseker et al. (2020). Enlarging the scope of the study to colonies above Bonneville Dam, some of which have also been subject to management measures, probably increased the ratio of the signal of avian predation to that of other factors that affect the likelihood of adult returns. There are several other important differences between this approach and the one taken in Haeseker et al. (2020). In addition to looking at effects on UCR instead of SRB steelhead, Payton et al. (2020) analyzed the degree of additivity (or compensation) for each annual cohort of outmigrants rather than across a multiyear study period. Payton et al. (2020) reported that Caspian tern predation may have been a partially additive source of mortality to (Figure 4 in Payton et al. 2020). Although the statistical model used in this study was reviewed before publication in the journal Environmental and Ecological Statistics, our understanding of its implications for adaptive management will also benefit from regional discussion and review.

B.4.3 Compensatory versus Additive Mortality in the 2008/2010/2014 FCRPS Biological Opinions and 2019 CRS Biological Opinion

The RPA developed for the 2008 FCRPS biological opinion and its 2010 and 2014 supplements employed multiple measures to improve the survival of ESA-listed salmonids. This included efforts to improve hydrosystem structures and operations, tributary and estuary habitat quality, and hatchery practices, and reduce avian, fish, and pinniped predation. NMFS did not quantitatively assume any compensatory mortality in assessing the benefits of predation management as applied to Caspian terns in the 2008 FCRPS biological opinion and in the 2014 FCRPS biological opinion, stated there was no clear indication that the case would be different, or substantial, for predation by double-crested cormorants.

As described above, the approaches taken more recently by modelers investigating the degree to which Caspian tern predation in the Columbia basin may be additive versus compensatory vary widely. They require more regional review before we can apply their findings to fisheries management or incorporate them into life cycle models such as those used in the 2020 opinion for SR spring/summer Chinook salmon and UCR spring Chinook salmon. The Action Agencies propose to continue implementing the avian predation management plans described in sections B.2 and B.3, maintaining the reduced amounts of nesting habitat achieved for terns and cormorants on East Sand Island and continuing to dissuade terns from nesting on Goose and Crescent Islands on the interior Columbia plateau (BPA et al. 2020, USACE et al. 2020). Thus, we expect that any reduced avian predation rates achieved under the 2008 FCRPS biological opinion and associated RPA will continue. Although work remains, we expect that at least some of the predation that is occurring is additive and contributes to increased SARs.

B.5 Summary—Avian Predation Management in the Columbia Basin

The region’s success in improving the survival of juvenile salmonids by managing the size of avian predator colonies is uncertain, but data from the 2018 and 2019 field seasons indicate that numbers of terns in the Columbia basin and their smolt predation rates have decreased (Harper and Collis 2018, Roby et al. 2019, Turecek et al. 2019). There is uncertainty because many Caspian terns moved to nearby locations in recent years rather than leaving the Columbia Basin. However, the Corps has been successful at keeping terns from nesting on Rice, Miller Sands, and Pillar Islands in 2018 and 2019 and the number of terns on East Sand Island has been much lower than any other year since 2000, indicating an overall reduction in the number in the estuary. Nevertheless, the estimated size of the East Sand Island colony in 2018 was substantially larger than the target colony size of up to 4,375 breeding pairs. On the interior Columbia plateau, the long term success of tern management efforts will depend on whether the Action Agencies successfully maintain the passive dissuasion established under the Inland Avian Predation Management Plan. With respect to double-crested cormorants, the number nesting on East Sand Island has declined, but large numbers have moved to the Astoria-Megler Bridge where per capita predation rates on salmonids is likely to be even higher than before colony management.

The 2020 Avian Predation Synthesis Report will help the adaptive management teams consider whether the Action Agencies or other regional parties should change their implementation strategies, including whether new measures should be assessed that could further reduce predation pressure. As discussed in Section B.4.3, an important question in evaluating the success of these programs is whether avian predation is an additive or compensatory source of mortality. That is, do reductions in smolt predation rates by Caspian terns or double-crested cormorants result in higher adult returns because avian predation adds to the other sources of smolt mortality, or are many of the smolts eaten by birds destined to die before returning as adults regardless of the level of avian predation? Haeseker et al. (2020) and Payton et al. (2020) have modeled these relationships, but these papers need more review. NMFS' position in this biological opinion is that, given the magnitude of bird predation on smolts, especially steelhead, in the Columbia Basin, it is likely that some of the individuals consumed by birds could otherwise have survived to adulthood. Therefore, even if avian predation is partially compensatory, we expect that the current and potential future efforts to limit the size of these tern and cormorant colonies are contributing to increased SARs for some populations of the listed ESUs/DPSs.

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Appendix C - Life-cycle Model Outputs

Note: This appendix contains modelling outputs in tabular form for abundance, QET50, and QET30 for all populations modelled. More detailed description of the model outputs, including references, are presented in the main document. See Section 2.2.3.1.12 Life-Cycle Models (SR spring/summer Chinook salmon), Section 2.5.3.1.12 Life-Cycle Modeling (SR fall Chinook salmon), and Section 2.6.3.1.12 Life-Cycle Modeling (UCR spring-run Chinook salmon). A complete description of the models is presented in Zabel 2020 and Perry 2020.

C.1 Abundance

A time period of 24 years forward from 2020 was selected as a reasonable timeframe to assess parameters generated by the models, including the geomean spawner abundance and the quasi-extinction risk threshold (QET). The period of 24 years includes approximately 6 generations of fish which would have experienced the proposed action as juveniles and returned to their natal streams as adults.

The abundances presented are the geomean spawner abundance for years 15 through 24 of the 24-year period of analysis. As noted, they represent either only natural-origin spawners, or natural- and hatchery-origin spawners. In the case of fall Chinook salmon, the abundance represents only female spawners. For all other populations modeled, the number represents both male and female spawners.

The QET is an estimate of the probability of a population reaching abundance levels for four consecutive years that may be too small to effectively reproduce—especially in larger basins where spawning adults might have more difficulty finding one another. Small populations are also more at risk from demographic stochasticity, genetic processes, and environmental variability. Because the exact number at which this condition occurs for Chinook salmon populations is unknown (and is likely variable due to a number of factors), past biological opinions (e.g., NMFS 2008a) provided QET projections for 50, 30, 10, and 1 individual. In this opinion, NMFS presents QET projections for 30 and 50 adults (for four consecutive years in the projected abundance estimates over the next 24 years) as a useful means of illustrating differences resulting from factors affecting the abundance and productivity of the modeled populations.

C.2 Climate Modelling

To account for anthropogenic carbon emissions, we extracted trends from global climate model (GCM) projections of RCP4.5 and RCP8.5 emission scenarios. The climate scenarios were modelled using the ensemble approach, as advocated by the Intergovernmental Panel on Climate Change (IPCC 2014). This approach addresses uncertainty in model assumptions by using as many different models as possible. There are 26 GCMs available for each emissions scenario from Coupled Model Intercomparison Project CMIP5, available from NOAA's Earth Systems Research Laboratory (Alexander et al. 2018). Scientists at the University of Washington downscaled output from 10 of those GCMs using multiple downscaling methods, and processed the output through four different hydrological models to produce 80 different time series for naturalized flow across the Columbia River Basin (RMJOC 2018, Chegwidden et al. 2019). Different GCMs and hydrological models projected more or less change in a given environmental variable, reflecting differences in model characteristics. To capture this range of environmental projections, we modeled population responses to the lower quartile, mean, and upper quartile time series available for each emissions scenario. Thus we represented model uncertainty by including examples of relatively slow warming, relatively fast warming, and the ensemble mean projection.

To calculate the impact of climate change, tri-monthly divergences were calculated from a reference period of 2005 to 2025 mean for each time series. Then a 20-year running mean of the resulting annual anomalies was calculated for each time series. The 25th, 50th, and 75th quantiles of the differences were selected across all time series. These quantiles represent the spread across climate models of low, medium, and high rates of change in climate conditions under the assumptions of RCP4.5 and RCP8.5.

C.3 Model Results

SNAKE RIVER FALL CHINOOK ESU

Snake River Fall Chinook					
	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Abundance Estimates					
Proposed Action	78	2592	8222	26714	266393
QET30 Estimates					
Proposed Action	0.00	0.00	0.00	0.00	0.18
QET50 Estimates					
Proposed Action	0.00	0.00	0.00	0.00	0.38

UPPER COLUMBIA SPRING CHINOOK ESU

Upper Columbia Spring Chinook					
	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Abundance Estimates					
Wenatchee River					
Proposed Action	182	339	532	885	1588
Proposed Action+17%	232	458	739	1144	2173
QET30 Estimates					
Wenatchee River					
Proposed Action	0.00	0.00	0.00	0.00	0.00
Proposed Action+17%	0.00	0.00	0.00	0.00	0.00
QET50 Estimates					
Wenatchee River					
Proposed Action	0.00	0.00	0.00	0.00	0.01
Proposed Action+17%	0.00	0.00	0.00	0.00	0.01

SNAKE RIVER SPRING SUMMER CHINOOK

Middle Fork Salmon River MPG Abundance Estimates					
	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Bear Valley					
Proposed Action	121	242	412	663	1284
Proposed Action+17%	150	306	518	818	1519
Proposed Action+ 35%	190	375	622	993	1770
RCP8.5 low	61	149	243	391	829
RCP8.5 low +17%	79	188	303	480	1005
RCP8.5 low + 35%	99	226	371	575	1151
RCP8.5 mean	53	116	193	340	698
RCP8.5 mean +17%	69	145	238	423	852
RCP8.5 mean + 35%	85	176	292	510	986
RCP8.5 high	38	95	156	265	525
RCP8.5 high +17%	49	120	192	329	661
RCP8.5 high + 35%	62	142	229	397	800
RCP4.5 low	70	151	258	427	794
RCP4.5 low + 17.5%	90	189	320	532	965
RCP4.5 low + 35%	109	228	389	652	1156
RCP4.5 mean	63	134	225	384	714
RCP4.5 mean + 17.5%	78	171	276	478	876
RCP4.5 mean + 35%	91	206	336	577	1080
RCP4.5 high	52	111	191	330	670
RCP4.5 high + 17.5%	64	144	237	409	794
RCP4.5 high + 35%	77	171	290	502	945
Big Creek					
Proposed Action	82	135	194	282	469
Proposed Action+17%	96	158	228	330	551
Proposed Action+ 35%	112	181	262	380	647
RCP8.5 low	52	88	123	177	303
RCP8.5 low +17%	61	102	144	205	356
RCP8.5 low + 35%	69	118	164	234	407
RCP8.5 mean	41	76	109	160	259
RCP8.5 mean +17%	47	88	127	186	297
RCP8.5 mean + 35%	52	100	146	211	342
RCP8.5 high	36	60	89	129	221

Middle Fork Salmon River MPG Abundance Estimates					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
RCP8.5 high +17%	42	69	104	148	257
RCP8.5 high + 35%	48	78	118	171	296
RCP4.5 low	55	93	134	191	307
RCP4.5 low + 17.5%	64	109	157	221	355
RCP4.5 low + 35%	72	124	178	252	414
RCP4.5 mean	44	84	120	175	283
RCP4.5 mean + 17.5%	52	98	139	206	338
RCP4.5 mean + 35%	58	112	160	236	391
RCP4.5 high	40	74	109	153	249
RCP4.5 high + 17.5%	46	85	128	181	291
RCP4.5 high + 35%	53	98	146	207	333
Camas Creek					
Proposed Action	19	37	54	82	139
Proposed Action+17%	22	44	64	98	165
Proposed Action+ 35%	26	51	75	114	193
RCP8.5 low	14	24	35	51	90
RCP8.5 low +17%	16	28	41	60	107
RCP8.5 low + 35%	18	33	47	70	123
RCP8.5 mean	11	20	30	44	81
RCP8.5 mean +17%	12	23	35	51	95
RCP8.5 mean + 35%	14	27	41	60	111
RCP8.5 high	8	16	25	37	62
RCP8.5 high +17%	10	19	29	44	74
RCP8.5 high + 35%	11	22	34	50	85
RCP4.5 low	14	25	37	55	96
RCP4.5 low + 17.5%	17	29	43	64	112
RCP4.5 low + 35%	19	33	49	74	130
RCP4.5 mean	12	22	33	49	88
RCP4.5 mean + 17.5%	14	26	39	58	105
RCP4.5 mean + 35%	17	30	45	68	122
RCP4.5 high	10	19	29	45	77
RCP4.5 high + 17.5%	12	23	34	54	92
RCP4.5 high + 35%	14	26	40	62	108
Loon Creek					
Proposed Action	29	52	74	106	191
Proposed Action+17%	34	60	86	123	224

Middle Fork Salmon River MPG Abundance Estimates					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
Proposed Action+ 35%	39	69	100	142	260
RCP8.5 low	20	35	49	68	114
RCP8.5 low +17%	23	41	57	80	135
RCP8.5 low + 35%	26	47	66	92	154
RCP8.5 mean	17	29	41	60	100
RCP8.5 mean +17%	19	33	48	70	121
RCP8.5 mean + 35%	22	38	55	80	138
RCP8.5 high	13	23	34	49	81
RCP8.5 high +17%	15	26	39	57	94
RCP8.5 high + 35%	17	30	45	66	108
RCP4.5 low	20	36	51	74	127
RCP4.5 low + 17.5%	23	42	60	86	148
RCP4.5 low + 35%	26	48	69	100	172
RCP4.5 mean	17	31	46	66	110
RCP4.5 mean + 17.5%	21	37	54	77	127
RCP4.5 mean + 35%	23	42	62	89	151
RCP4.5 high	16	27	40	60	101
RCP4.5 high + 17.5%	18	31	47	69	119
RCP4.5 high + 35%	21	36	53	79	139
Marsh Creek					
Proposed Action	83	154	269	445	850
Proposed Action+17%	101	191	330	542	1046
Proposed Action+ 35%	121	225	395	642	1247
RCP8.5 low	52	101	157	244	544
RCP8.5 low +17%	62	123	191	298	664
RCP8.5 low + 35%	75	145	228	362	783
RCP8.5 mean	41	90	138	226	435
RCP8.5 mean +17%	50	109	170	280	535
RCP8.5 mean + 35%	57	131	204	330	654
RCP8.5 high	32	71	115	181	357
RCP8.5 high +17%	41	85	140	223	436
RCP8.5 high + 35%	48	102	166	265	527
RCP4.5 low	50	103	165	262	519
RCP4.5 low + 17.5%	62	126	203	325	640
RCP4.5 low + 35%	72	154	242	391	756
RCP4.5 mean	45	95	151	262	483

Middle Fork Salmon River MPG Abundance Estimates					
	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
RCP4.5 mean + 17.5%	55	115	186	317	594
RCP4.5 mean + 35%	66	138	223	375	720
RCP4.5 high	41	84	145	236	461
RCP4.5 high + 17.5%	51	103	176	291	562
RCP4.5 high + 35%	60	124	209	345	682
Sulphur Creek					
Proposed Action	23	47	72	118	235
Proposed Action+17%	28	59	90	145	299
Proposed Action+ 35%	33	70	108	175	363
RCP8.5 low	15	28	44	67	138
RCP8.5 low +17%	19	35	55	81	167
RCP8.5 low + 35%	22	41	65	98	205
RCP8.5 mean	11	23	37	63	126
RCP8.5 mean +17%	14	28	46	76	152
RCP8.5 mean + 35%	16	33	54	92	189
RCP8.5 high	9	19	31	49	101
RCP8.5 high +17%	11	23	38	61	124
RCP8.5 high + 35%	14	27	46	72	150
RCP4.5 low	15	29	45	72	143
RCP4.5 low + 17.5%	18	36	56	89	172
RCP4.5 low + 35%	22	43	69	106	213
RCP4.5 mean	13	26	42	70	140
RCP4.5 mean + 17.5%	16	32	52	86	168
RCP4.5 mean + 35%	19	38	62	102	211
RCP4.5 high	11	23	38	61	132
RCP4.5 high + 17.5%	13	28	47	76	165
RCP4.5 high + 35%	15	34	56	91	201

Middle Fork Salmon River MPG QET 30 Estimates					
	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Bear Valley					
Proposed Action	0.00	0.00	0.01	0.01	0.02
Proposed Action+17%	0.00	0.00	0.00	0.01	0.01
Proposed Action+ 35%	0.00	0.00	0.00	0.00	0.01
RCP8.5 low	0.01	0.02	0.03	0.04	0.07
RCP8.5 low +17%	0.00	0.01	0.02	0.03	0.05
RCP8.5 low + 35%	0.00	0.00	0.01	0.02	0.03
RCP8.5 mean	0.02	0.04	0.05	0.07	0.09
RCP8.5 mean +17%	0.01	0.02	0.03	0.05	0.06
RCP8.5 mean + 35%	0.00	0.01	0.02	0.03	0.05
RCP8.5 high	0.05	0.07	0.09	0.11	0.14
RCP8.5 high +17%	0.02	0.04	0.06	0.07	0.10
RCP8.5 high + 35%	0.01	0.02	0.03	0.04	0.07
RCP4.5 low	0.00	0.02	0.03	0.04	0.06
RCP4.5 low + 17.5%	0.00	0.01	0.02	0.02	0.04
RCP4.5 low + 35%	0.00	0.00	0.01	0.01	0.03
RCP4.5 mean	0.01	0.03	0.05	0.06	0.08
RCP4.5 mean + 17.5%	0.00	0.01	0.02	0.04	0.06
RCP4.5 mean + 35%	0.00	0.01	0.02	0.02	0.04
RCP4.5 high	0.03	0.05	0.06	0.08	0.10
RCP4.5 high + 17.5%	0.01	0.02	0.03	0.04	0.06
RCP4.5 high + 35%	0.00	0.01	0.02	0.02	0.04
Big Creek					
Proposed Action	0.00	0.01	0.02	0.03	0.05
Proposed Action+17%	0.00	0.00	0.01	0.01	0.03
Proposed Action+ 35%	0.00	0.00	0.00	0.01	0.02
RCP8.5 low	0.02	0.04	0.05	0.07	0.10
RCP8.5 low +17%	0.00	0.02	0.03	0.04	0.06
RCP8.5 low + 35%	0.00	0.01	0.02	0.03	0.04
RCP8.5 mean	0.05	0.07	0.09	0.11	0.14
RCP8.5 mean +17%	0.03	0.05	0.07	0.08	0.11
RCP8.5 mean + 35%	0.02	0.03	0.05	0.06	0.09
RCP8.5 high	0.09	0.13	0.15	0.17	0.21
RCP8.5 high +17%	0.06	0.08	0.11	0.13	0.16
RCP8.5 high + 35%	0.03	0.06	0.07	0.09	0.12
RCP4.5 low + 17.5%	0.01	0.02	0.03	0.05	0.06
RCP4.5 low + 35%	0.00	0.01	0.02	0.03	0.05

Middle Fork Salmon River MPG QET 30 Estimates					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
RCP4.5 mean	0.04	0.06	0.08	0.09	0.12
RCP4.5 mean + 17.5%	0.02	0.04	0.06	0.07	0.10
RCP4.5 mean + 35%	0.01	0.02	0.03	0.04	0.06
RCP4.5 high	0.06	0.09	0.11	0.13	0.17
RCP4.5 high + 17.5%	0.03	0.05	0.07	0.09	0.12
RCP4.5 high + 35%	0.02	0.04	0.05	0.06	0.09
Camas Creek					
Proposed Action	0.49	0.54	0.57	0.60	0.65
Proposed Action+17%	0.38	0.42	0.46	0.49	0.54
Proposed Action+ 35%	0.28	0.33	0.37	0.40	0.45
RCP8.5 low	0.65	0.69	0.72	0.75	0.79
RCP8.5 low +17%	0.55	0.60	0.64	0.67	0.71
RCP8.5 low + 35%	0.47	0.51	0.54	0.58	0.62
RCP8.5 mean	0.71	0.76	0.79	0.81	0.85
RCP8.5 mean +17%	0.64	0.68	0.72	0.75	0.79
RCP8.5 mean + 35%	0.55	0.60	0.63	0.66	0.71
RCP8.5 high	0.82	0.85	0.88	0.90	0.93
RCP8.5 high +17%	0.75	0.79	0.82	0.85	0.88
RCP8.5 high + 35%	0.66	0.71	0.74	0.77	0.81
RCP4.5 low	0.68	0.72	0.75	0.78	0.82
RCP4.5 low + 17.5%	0.56	0.60	0.63	0.67	0.72
RCP4.5 low + 35%	0.47	0.51	0.54	0.58	0.63
RCP4.5 mean	0.71	0.75	0.77	0.80	0.83
RCP4.5 mean + 17.5%	0.63	0.67	0.71	0.73	0.78
RCP4.5 mean + 35%	0.53	0.58	0.61	0.64	0.69
RCP4.5 high	0.73	0.77	0.80	0.83	0.87
RCP4.5 high + 17.5%	0.66	0.71	0.73	0.76	0.81
RCP4.5 high + 35%	0.57	0.61	0.65	0.68	0.72
Loon Creek					
Proposed Action	0.25	0.30	0.33	0.36	0.40
Proposed Action+17%	0.17	0.21	0.24	0.27	0.32
Proposed Action+ 35%	0.12	0.15	0.18	0.20	0.24
RCP8.5 low	0.41	0.46	0.49	0.53	0.57
RCP8.5 low +17%	0.31	0.36	0.40	0.43	0.48
RCP8.5 low + 35%	0.22	0.27	0.30	0.33	0.37
RCP8.5 mean	0.51	0.56	0.59	0.62	0.67
RCP8.5 mean +17%	0.41	0.46	0.50	0.53	0.57

Middle Fork Salmon River MPG QET 30 Estimates					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
RCP8.5 mean + 35%	0.33	0.37	0.41	0.44	0.49
RCP8.5 high	0.67	0.71	0.74	0.77	0.81
RCP8.5 high +17%	0.56	0.61	0.64	0.67	0.72
RCP8.5 high + 35%	0.46	0.50	0.53	0.57	0.62
RCP4.5 low	0.43	0.47	0.50	0.54	0.58
RCP4.5 low + 17.5%	0.31	0.36	0.40	0.43	0.47
RCP4.5 low + 35%	0.23	0.27	0.30	0.33	0.38
RCP4.5 mean	0.50	0.54	0.57	0.61	0.66
RCP4.5 mean + 17.5%	0.39	0.44	0.47	0.51	0.55
RCP4.5 mean + 35%	0.31	0.35	0.38	0.41	0.46
RCP4.5 high	0.54	0.59	0.62	0.66	0.71
RCP4.5 high + 17.5%	0.43	0.48	0.52	0.55	0.60
RCP4.5 high + 35%	0.34	0.39	0.42	0.45	0.50
Marsh Creek					
Proposed Action	0.00	0.02	0.03	0.04	0.06
Proposed Action+17%	0.00	0.01	0.01	0.02	0.04
Proposed Action+ 35%	0.00	0.00	0.01	0.02	0.03
RCP8.5 low	0.02	0.04	0.06	0.07	0.10
RCP8.5 low +17%	0.01	0.03	0.04	0.05	0.07
RCP8.5 low + 35%	0.00	0.01	0.02	0.04	0.05
RCP8.5 mean	0.05	0.08	0.10	0.12	0.15
RCP8.5 mean +17%	0.03	0.05	0.06	0.08	0.11
RCP8.5 mean + 35%	0.02	0.03	0.04	0.06	0.08
RCP8.5 high	0.08	0.12	0.14	0.16	0.20
RCP8.5 high +17%	0.05	0.08	0.10	0.12	0.14
RCP8.5 high + 35%	0.03	0.05	0.07	0.09	0.11
RCP4.5 low	0.03	0.05	0.07	0.09	0.12
RCP4.5 low + 17.5%	0.01	0.03	0.04	0.06	0.08
RCP4.5 low + 35%	0.00	0.01	0.02	0.03	0.05
RCP4.5 mean	0.04	0.06	0.07	0.09	0.12
RCP4.5 mean + 17.5%	0.02	0.04	0.05	0.07	0.09
RCP4.5 mean + 35%	0.01	0.02	0.03	0.05	0.07
RCP4.5 high	0.06	0.09	0.11	0.13	0.16
RCP4.5 high + 17.5%	0.03	0.05	0.07	0.09	0.11
RCP4.5 high + 35%	0.01	0.03	0.04	0.06	0.08
Sulphur Creek					
Proposed Action	0.34	0.39	0.42	0.45	0.50

Middle Fork Salmon River MPG QET 30 Estimates					
	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Proposed Action+17%	0.24	0.29	0.32	0.35	0.39
Proposed Action+ 35%	0.18	0.22	0.25	0.28	0.32
RCP8.5 low	0.55	0.60	0.63	0.66	0.70
RCP8.5 low +17%	0.42	0.47	0.50	0.54	0.58
RCP8.5 low + 35%	0.33	0.38	0.41	0.44	0.48
RCP8.5 mean	0.63	0.68	0.71	0.74	0.78
RCP8.5 mean +17%	0.53	0.58	0.62	0.65	0.70
RCP8.5 mean + 35%	0.43	0.47	0.51	0.54	0.59
RCP8.5 high	0.71	0.75	0.78	0.81	0.85
RCP8.5 high +17%	0.61	0.66	0.69	0.72	0.76
RCP8.5 high + 35%	0.51	0.56	0.59	0.62	0.67
RCP4.5 low	0.54	0.59	0.62	0.66	0.71
RCP4.5 low + 17.5%	0.44	0.48	0.52	0.55	0.60
RCP4.5 low + 35%	0.33	0.37	0.41	0.44	0.49
RCP4.5 mean	0.58	0.63	0.66	0.69	0.74
RCP4.5 mean + 17.5%	0.46	0.51	0.54	0.57	0.62
RCP4.5 mean + 35%	0.38	0.42	0.45	0.49	0.54
RCP4.5 high	0.62	0.66	0.69	0.72	0.76
RCP4.5 high + 17.5%	0.50	0.55	0.58	0.61	0.66
RCP4.5 high + 35%	0.41	0.45	0.49	0.52	0.57

Middle Fork Salmon River MPG					
QET 50 Estimates					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
Bear Valley					
Proposed Action	0.01	0.03	0.04	0.05	0.08
Proposed Action+17%	0	0.01	0.02	0.02	0.04
Proposed Action+ 35%	0	0	0.01	0.02	0.03
RCP8.5 low	0.05	0.08	0.09	0.11	0.14
RCP8.5 low +17%	0.02	0.04	0.06	0.07	0.1
RCP8.5 low + 35%	0.01	0.02	0.04	0.05	0.07
RCP8.5 mean	0.1	0.13	0.15	0.18	0.21
RCP8.5 mean +17%	0.05	0.07	0.09	0.11	0.14
RCP8.5 mean + 35%	0.03	0.05	0.06	0.08	0.11
RCP8.5 high	0.17	0.21	0.23	0.26	0.3
RCP8.5 high +17%	0.1	0.14	0.16	0.19	0.22
RCP8.5 high + 35%	0.06	0.09	0.11	0.13	0.16
RCP4.5 low	0.05	0.08	0.1	0.12	0.15
RCP4.5 low + 17.5%	0.03	0.05	0.07	0.09	0.11
RCP4.5 low + 35%	0.01	0.02	0.04	0.05	0.07
RCP4.5 mean	0.07	0.1	0.12	0.14	0.18
RCP4.5 mean + 17.5%	0.04	0.06	0.08	0.1	0.13
RCP4.5 mean + 35%	0.02	0.04	0.05	0.06	0.09
RCP4.5 high	0.11	0.14	0.17	0.19	0.23
RCP4.5 high + 17.5%	0.07	0.09	0.11	0.13	0.16
RCP4.5 high + 35%	0.03	0.06	0.07	0.09	0.12
Big Creek					
Proposed Action	0.06	0.09	0.11	0.13	0.16
Proposed Action+17%	0.03	0.05	0.06	0.08	0.11
Proposed Action+ 35%	0.01	0.03	0.04	0.06	0.08
RCP8.5 low	0.16	0.2	0.23	0.26	0.31
RCP8.5 low +17%	0.1	0.13	0.16	0.18	0.22
RCP8.5 low + 35%	0.06	0.09	0.11	0.13	0.16
RCP8.5 mean	0.23	0.278	0.3	0.33	0.38
RCP8.5 mean +17%	0.16	0.2	0.23	0.26	0.3
RCP8.5 mean + 35%	0.11	0.14	0.17	0.19	0.23
RCP8.5 high	0.36	0.41	0.45	0.48	0.53
RCP8.5 high +17%	0.27	0.32	0.35	0.38	0.43
RCP8.5 high + 35%	0.19	0.23	0.26	0.29	0.33
RCP4.5 low	0.17	0.21	0.24	0.27	0.3
RCP4.5 low + 17.5%	0.1	0.13	0.15	0.18	0.22

Middle Fork Salmon River MPG					
QET 50 Estimates					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
RCP4.5 low + 35%	0.06	0.09	0.11	0.13	0.17
RCP4.5 mean	0.2	0.24	0.27	0.3	0.34
RCP4.5 mean + 17.5%	0.14	0.17	0.2	0.22	0.27
RCP4.5 mean + 35%	0.09	0.12	0.14	0.17	0.2
RCP4.5 high	0.24	0.29	0.32	0.35	0.4
RCP4.5 high + 17.5%	0.17	0.22	0.24	0.27	0.32
RCP4.5 high + 35%	0.13	0.16	0.18	0.21	0.25
Camas Creek					
Proposed Action	0.8	0.83	0.86	0.88	0.91
Proposed Action+17%	0.72	0.76	0.78	0.81	0.85
Proposed Action+ 35%	0.62	0.67	0.71	0.73	0.78
RCP8.5 low	0.9	0.93	0.95	0.96	0.98
RCP8.5 low +17%	0.85	0.88	0.9	0.92	0.95
RCP8.5 low + 35%	0.78	0.82	0.84	0.87	0.9
RCP8.5 mean	0.92	0.94	0.96	0.97	0.99
RCP8.5 mean +17%	0.88	0.91	0.93	0.95	0.97
RCP8.5 mean + 35%	0.84	0.87	0.89	0.92	0.94
RCP8.5 high	0.96	0.97	0.98	0.99	1
RCP8.5 high +17%	0.94	0.96	0.97	0.98	0.99
RCP8.5 high + 35%	0.9	0.93	0.95	0.96	0.98
RCP4.5 low	0.9	0.92	0.94	0.96	0.98
RCP4.5 low + 17.5%	0.84	0.87	0.89	0.91	0.94
RCP4.5 low + 35%	0.78	0.82	0.85	0.87	0.91
RCP4.5 mean	0.91	0.93	0.95	0.96	0.98
RCP4.5 mean + 17.5%	0.87	0.89	0.91	0.93	0.96
RCP4.5 mean + 35%	0.819	0.85	0.87	0.9	0.93
RCP4.5 high	0.93	0.95	0.96	0.98	0.99
RCP4.5 high + 17.5%	0.88	0.91	0.92	0.94	0.97
RCP4.5 high + 35%	0.83	0.86	0.88	0.91	0.93
Loon Creek					
Proposed Action	0.63	0.67	0.7	0.73	0.77
Proposed Action+17%	0.53	0.58	0.61	0.64	0.69
Proposed Action+ 35%	0.42	0.47	0.5	0.53	0.59
RCP8.5 low	0.79	0.83	0.85	0.87	0.91
RCP8.5 low +17%	0.7	0.75	0.78	0.8	0.84
RCP8.5 low + 35%	0.61	0.65	0.68	0.71	0.75
RCP8.5 mean	0.83	0.86	0.89	0.91	0.93

Middle Fork Salmon River MPG					
QET 50 Estimates					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
RCP8.5 mean +17%	0.77	0.8	0.83	0.85	0.88
RCP8.5 mean + 35%	0.69	0.73	0.76	0.79	0.83
RCP8.5 high	0.89	0.92	0.94	0.95	0.97
RCP8.5 high +17%	0.85	0.88	0.9	0.92	0.95
RCP8.5 high + 35%	0.8	0.84	0.86	0.88	0.91
RCP4.5 low	0.78	0.81	0.84	0.86	0.9
RCP4.5 low + 17.5%	0.69	0.73	0.76	0.79	0.83
RCP4.5 low + 35%	0.59	0.64	0.67	0.7	0.75
RCP4.5 mean	0.82	0.86	0.88	0.9	0.93
RCP4.5 mean + 17.5%	0.75	0.79	0.82	0.84	0.88
RCP4.5 mean + 35%	0.67	0.72	0.75	0.77	0.82
RCP4.5 high	0.84	0.87	0.89	0.91	0.94
RCP4.5 high + 17.5%	0.79	0.82	0.85	0.87	0.9
RCP4.5 high + 35%	0.71	0.75	0.78	0.81	0.84
Marsh Creek					
Proposed Action	0.05	0.09	0.11	0.13	0.16
Proposed Action+17%	0.03	0.05	0.06	0.08	0.11
Proposed Action+ 35%	0.01	0.03	0.04	0.05	0.08
RCP8.5 low	0.15	0.19	0.21	0.24	0.28
RCP8.5 low +17%	0.09	0.12	0.14	0.17	0.2
RCP8.5 low + 35%	0.05	0.08	0.09	0.11	0.15
RCP8.5 mean	0.2	0.24	0.26	0.29	0.34
RCP8.5 mean +17%	0.14	0.17	0.2	0.23	0.27
RCP8.5 mean + 35%	0.09	0.12	0.15	0.17	0.2
RCP8.5 high	0.28	0.32	0.36	0.39	0.44
RCP8.5 high +17%	0.18	0.22	0.25	0.28	0.32
RCP8.5 high + 35%	0.12	0.16	0.18	0.21	0.24
RCP4.5 low	0.16	0.19	0.22	0.25	0.29
RCP4.5 low + 17.5%	0.1	0.13	0.15	0.17	0.21
RCP4.5 low + 35%	0.06	0.08	0.1	0.12	0.16
RCP4.5 mean	0.19	0.23	0.26	0.29	0.33
RCP4.5 mean + 17.5%	0.12	0.15	0.18	0.21	0.24
RCP4.5 mean + 35%	0.07	0.1	0.12	0.14	0.18
RCP4.5 high	0.2	0.24	0.27	0.3	0.35
RCP4.5 high + 17.5%	0.14	0.17	0.2	0.23	0.27
RCP4.5 high + 35%	0.08	0.11	0.13	0.16	0.19
Sulphur Creek					

Middle Fork Salmon River MPG					
QET 50 Estimates					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
Proposed Action	0.66	0.71	0.74	0.77	0.81
Proposed Action+17%	0.55	0.6	0.63	0.66	0.7
Proposed Action+ 35%	0.43	0.48	0.51	0.54	0.59
RCP8.5 low	0.83	0.86	0.88	0.9	0.93
RCP8.5 low +17%	0.74	0.78	0.8	0.83	0.87
RCP8.5 low + 35%	0.63	0.68	0.71	0.74	0.79
RCP8.5 mean	0.83	0.86	0.89	0.91	0.94
RCP8.5 mean +17%	0.76	0.8	0.83	0.85	0.88
RCP8.5 mean + 35%	0.69	0.73	0.76	0.79	0.82
RCP8.5 high	0.9	0.92	0.94	0.96	0.98
RCP8.5 high +17%	0.85	0.88	0.9	0.92	0.94
RCP8.5 high + 35%	0.78	0.82	0.84	0.86	0.89
RCP4.5 low	0.82	0.85	0.88	0.9	0.93
RCP4.5 low + 17.5%	0.73	0.77	0.8	0.83	0.87
RCP4.5 low + 35%	0.62	0.67	0.7	0.73	0.78
RCP4.5 mean	0.82	0.86	0.88	0.91	0.93
RCP4.5 mean + 17.5%	0.74	0.78	0.81	0.83	0.87
RCP4.5 mean + 35%	0.65	0.7	0.73	0.76	0.81
RCP4.5 high	0.84	0.87	0.89	0.91	0.94
RCP4.5 high + 17.5%	0.77	0.81	0.84	0.86	0.9
RCP4.5 high + 35%	0.689	0.73	0.76	0.78	0.82

South Fork Salmon River MPG					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
Abundance Estimates					
Secesh River					
Proposed Action	154	364	556	916	1843
Proposed Action+17%	198	446	688	1139	2180
Proposed Action+ 35%	256	539	840	1357	2528
RCP8.5 low	89	194	323	529	1049
RCP8.5 low +17%	112	243	406	663	1281
RCP8.5 low + 35%	136	292	484	796	1495
RCP8.5 mean	74	173	286	471	1001
RCP8.5 mean +17%	92	213	353	583	1203
RCP8.5 mean + 35%	114	258	426	707	1412
RCP8.5 high	57	136	235	391	763
RCP8.5 high +17%	70	168	295	488	939
RCP8.5 high + 35%	85	205	352	594	1112
RCP4.5 low	87	201	349	566	1161
RCP4.5 low + 17.5%	110	251	429	706	1414
RCP4.5 low + 35%	132	301	521	851	1639
RCP4.5 mean	87	186	316	515	1028
RCP4.5 mean + 17.5%	104	232	389	644	1270
RCP4.5 mean + 35%	126	277	469	778	1470
RCP4.5 high	68	167	287	493	986
RCP4.5 high + 17.5%	86	207	361	609	1216
RCP4.5 high + 35%	108	252	436	733	1456
QET30 Estimates					
Secesh River					
Proposed Action	0.00	0.00	0.01	0.01	0.02
Proposed Action+17.5%	0.00	0.00	0.00	0.01	0.02
Proposed Action+ 35%	0.00	0.00	0.00	0.01	0.02
RCP8.5 low	0.00	0.01	0.02	0.03	0.05
RCP8.5 low +17%	0.00	0.00	0.01	0.02	0.03
RCP8.5 low + 35%	0.00	0.00	0.00	0.01	0.02
RCP8.5 mean	0.01	0.02	0.03	0.05	0.07
RCP8.5 mean +17%	0.00	0.01	0.02	0.03	0.05
RCP8.5 mean + 35%	0.00	0.01	0.01	0.02	0.04
RCP8.5 high	0.02	0.03	0.05	0.06	0.09
RCP8.5 high +17%	0.01	0.02	0.03	0.04	0.06
RCP8.5 high + 35%	0.00	0.01	0.02	0.04	0.05
RCP4.5 low	0.00	0.01	0.02	0.04	0.06

South Fork Salmon River MPG					
	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
RCP4.5 low + 17.5%	0.00	0.00	0.01	0.02	0.03
RCP4.5 low + 35%	0.00	0.00	0.00	0.01	0.02
RCP4.5 mean	0.00	0.01	0.02	0.03	0.05
RCP4.5 mean + 17.5%	0.00	0.00	0.01	0.02	0.03
RCP4.5 mean + 35%	0.00	0.00	0.01	0.01	0.02
RCP4.5 high	0.00	0.02	0.03	0.04	0.06
RCP4.5 high + 17.5%	0.00	0.00	0.01	0.02	0.03
RCP4.5 high + 35%	0.00	0.00	0.00	0.01	0.02
QET50					
Secesh River					
Proposed Action	0.00	0.02	0.03	0.04	0.06
Proposed Action+17.5%	0.00	0.00	0.01	0.02	0.03
Proposed Action+ 35%	0.00	0.00	0.00	0.01	0.02
RCP8.5 low	0.03	0.05	0.06	0.08	0.10
RCP8.5 low +17%	0.01	0.02	0.03	0.05	0.07
RCP8.5 low + 35%	0.00	0.01	0.02	0.03	0.05
RCP8.5 mean	0.04	0.07	0.08	0.10	0.13
RCP8.5 mean +17%	0.02	0.04	0.06	0.07	0.10
RCP8.5 mean + 35%	0.01	0.03	0.04	0.05	0.07
RCP8.5 high	0.08	0.11	0.12	0.15	0.18
RCP8.5 high +17%	0.05	0.07	0.09	0.11	0.14
RCP8.5 high + 35%	0.03	0.04	0.06	0.07	0.10
RCP4.5 low	0.02	0.04	0.06	0.08	0.10
RCP4.5 low + 17.5%	0.01	0.03	0.04	0.05	0.08
RCP4.5 low + 35%	0.00	0.02	0.03	0.04	0.06
RCP4.5 mean	0.04	0.06	0.08	0.10	0.12
RCP4.5 mean + 17.5%	0.01	0.02	0.04	0.05	0.08
RCP4.5 mean + 35%	0.00	0.01	0.02	0.03	0.05
RCP4.5 high	0.05	0.08	0.10	0.12	0.15
RCP4.5 high + 17.5%	0.03	0.04	0.06	0.08	0.10
RCP4.5 high + 35%	0.01	0.02	0.03	0.05	0.06

Grand Ronde MPG Abundance Estimates					
	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Natural and Hatchery Origin Spawners					
Catherine Creek					
Proposed Action	280	491	679	950	1429
Proposed Action+17%	366	584	810	1115	1746
Proposed Action+ 35%	429	692	965	1307	1991
RCP8.5 low	205	354	489	655	999
RCP8.5 low +17%	269	419	582	773	1273
RCP8.5 low + 35%	315	496	680	889	1448
RCP8.5 mean	170	274	388	542	890
RCP8.5 mean +17%	202	335	459	653	1038
RCP8.5 mean + 35%	239	396	543	753	1192
RCP8.5 high	130	216	306	428	649
RCP8.5 high +17%	158	270	366	509	775
RCP8.5 high + 35%	189	312	428	593	935
RCP 4.5 low	220	357	517	703	1073
RCP 4.5 low + 17%	272	445	616	848	1292
RCP 4.5 low + 35%	345	531	712	986	1457
RCP 4.5 med	186	308	435	610	1016
RCP 4.5 med + 17%	232	369	511	730	1175
RCP 4.5 med + 35%	292	443	614	835	1350
RCP 4.5 high	155	242	360	503	823
RCP 4.5 high + 17%	181	299	433	597	956
RCP 4.5 high + 35%	221	354	516	697	1118
Lostine					
Proposed Action	355	519	658	846	1181
Proposed Action+17%	415	605	751	965	1328
Proposed Action+ 35%	475	681	858	1083	1483
RCP8.5 low	254	375	480	610	883
RCP8.5 low +17%	309	439	552	704	1017
RCP8.5 low + 35%	355	505	629	795	1148
RCP8.5 mean	187	289	377	529	740
RCP8.5 mean +17%	224	335	440	602	861
RCP8.5 mean + 35%	261	384	505	684	962
RCP8.5 high	139	222	302	404	595
RCP8.5 high +17%	161	261	353	473	681

Grand Ronde MPG Abundance Estimates					
	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
RCP8.5 high + 35%	193	304	408	534	783
RCP 4.5 low	240	384	502	649	912
RCP 4.5 low + 17%	288	456	587	743	1037
RCP 4.5 low + 35%	335	518	664	830	1150
RCP 4.5 med	216	324	438	579	840
RCP 4.5 med + 17%	253	384	501	662	949
RCP 4.5 med + 35%	297	440	578	748	1068
RCP 4.5 high	171	265	365	474	706
RCP 4.5 high + 17%	196	310	424	547	802
RCP 4.5 high + 35%	230	359	488	619	910
Upper Grande Ronde					
Proposed Action	265	426	567	768	1164
Proposed Action+17%	317	510	668	895	1333
Proposed Action+ 35%	379	595	767	1035	1545
RCP8.5 low	185	292	387	513	817
RCP8.5 low +17%	229	351	462	612	927
RCP8.5 low + 35%	270	409	530	699	1059
RCP8.5 mean	126	212	293	434	654
RCP8.5 mean +17%	162	260	353	510	770
RCP8.5 mean + 35%	194	305	415	592	885
RCP8.5 high	94	158	226	310	496
RCP8.5 high +17%	111	193	271	372	584
RCP8.5 high + 35%	139	230	324	434	684
RCP 4.5 low	179	302	415	561	830
RCP 4.5 low + 17%	217	366	491	654	966
RCP 4.5 low + 35%	258	426	563	753	1094
RCP 4.5 med	157	250	346	477	746
RCP 4.5 med + 17%	192	303	411	567	857
RCP 4.5 med + 35%	224	356	481	653	993
RCP 4.5 high	116	188	275	379	615
RCP 4.5 high + 17%	143	232	334	457	716
RCP 4.5 high + 35%	168	275	391	528	822
Wild origin spawners					
Catherine Creek					
Proposed Action	126	226	313	446	679
Proposed Action+17%	166	268	376	526	835
Proposed Action+ 35%	196	320	451	620	958

Grand Ronde MPG Abundance Estimates					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
RCP8.5 low	88	160	222	302	469
RCP8.5 low +17%	120	190	266	358	602
RCP8.5 low + 35%	141	228	315	415	689
RCP8.5 mean	72	122	172	247	416
RCP8.5 mean +17%	89	150	207	300	487
RCP8.5 mean + 35%	105	178	247	348	565
RCP8.5 high	53	93	134	194	298
RCP8.5 high +17%	67	117	162	231	359
RCP8.5 high + 35%	81	137	191	272	437
RCP 4.5 low	95	161	237	326	502
RCP 4.5 low + 17%	121	202	284	393	613
RCP 4.5 low + 35%	155	243	330	462	694
RCP 4.5 med	80	136	197	281	475
RCP 4.5 med + 17%	102	166	233	338	556
RCP 4.5 med + 35%	130	201	282	389	642
RCP 4.5 high	65	106	162	228	383
RCP 4.5 high + 17%	78	132	197	273	449
RCP 4.5 high + 35%	98	159	234	321	526
Lostine River					
Proposed Action	102	173	233	325	513
Proposed Action+17%	129	206	281	394	612
Proposed Action+ 35%	146	246	336	448	701
RCP8.5 low	66	114	159	219	351
RCP8.5 low +17%	89	143	193	269	422
RCP8.5 low + 35%	108	171	232	313	505
RCP8.5 mean	45	84	123	188	292
RCP8.5 mean +17%	60	102	148	221	357
RCP8.5 mean + 35%	74	124	178	259	406
RCP8.5 high	34	62	92	135	224
RCP8.5 high +17%	41	77	114	166	267
RCP8.5 high + 35%	51	95	137	195	323
RCP 4.5 low	68	122	170	232	373
RCP 4.5 low + 17%	89	153	205	279	452
RCP 4.5 low + 35%	107	179	248	325	517
RCP 4.5 med	61	99	143	204	344
RCP 4.5 med + 17%	72	125	177	250	401
RCP 4.5 med + 35%	90	146	208	292	468

Grand Ronde MPG Abundance Estimates					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
RCP 4.5 high	42	75	114	165	280
RCP 4.5 high + 17%	52	95	139	199	330
RCP 4.5 high + 35%	65	114	165	233	387
Minam River					
Proposed Action	1	29	217	853	2999
Proposed Action+17%	2	86	465	1453	3779
Proposed Action+ 35%	7	170	843	2155	4762
RCP8.5 low	1	18	123	535	1752
RCP8.5 low +17%	3	52	268	881	2563
RCP8.5 low + 35%	8	119	450	1275	3293
RCP8.5 mean	1	11	91	374	1410
RCP8.5 mean +17%	2	30	190	610	1904
RCP8.5 mean + 35%	6	68	331	934	2473
RCP8.5 high	1	8	61	237	990
RCP8.5 high +17%	1	25	127	424	1399
RCP8.5 high + 35%	4	58	230	664	1806
RCP 4.5 low	1	20	139	562	1874
RCP 4.5 low + 17%	2	60	311	950	2491
RCP 4.5 low + 35%	5	122	555	1385	3350
RCP 4.5 med	1	14	110	387	1667
RCP 4.5 med + 17%	2	40	238	681	2278
RCP 4.5 med + 35%	6	102	401	1058	3103
RCP 4.5 high	1	11	71	319	1198
RCP 4.5 high + 17%	2	30	158	564	1636
RCP 4.5 high + 35%	4	67	305	870	2221
Upper Grande Ronde					
Proposed Action	34	50	66	87	123
Proposed Action+17%	39	57	74	99	142
Proposed Action+ 35%	46	66	85	114	161
RCP8.5 low	26	37	48	61	93
RCP8.5 low +17%	30	43	55	72	104
RCP8.5 low + 35%	34	49	63	80	120
RCP8.5 mean	20	29	39	52	81
RCP8.5 mean +17%	23	34	46	60	92
RCP8.5 mean + 35%	26	39	51	68	101
RCP8.5 high	16	24	31	42	61
RCP8.5 high +17%	19	28	36	48	70

Grand Ronde MPG Abundance Estimates					
	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
RCP8.5 high + 35%	21	32	42	56	78
RCP 4.5 low	25	39	51	66	96
RCP 4.5 low + 17%	30	45	59	76	111
RCP 4.5 low + 35%	34	52	65	84	123
RCP 4.5 med	23	34	44	57	84
RCP 4.5 med + 17%	27	39	51	68	99
RCP 4.5 med + 35%	31	44	58	75	113
RCP 4.5 high	18	27	37	50	75
RCP 4.5 high + 17%	21	32	43	56	85
RCP 4.5 high + 35%	24	37	48	64	94
Wenaha River					
Proposed Action	5	58	195	616	2248
Proposed Action+17%	22	140	446	1254	3533
Proposed Action+ 35%	56	301	844	2105	5345
RCP8.5 low	6	38	106	286	1308
RCP8.5 low +17%	22	94	247	627	2077
RCP8.5 low + 35%	49	199	482	1109	3056
RCP8.5 mean	4	24	68	219	973
RCP8.5 mean +17%	12	63	170	459	1701
RCP8.5 mean + 35%	31	131	328	812	2455
RCP8.5 high	2	15	50	152	657
RCP8.5 high +17%	7	41	116	316	1107
RCP8.5 high + 35%	18	86	240	544	1793
RCP 4.5 low	6	39	112	335	1346
RCP 4.5 low + 17%	20	100	271	685	2286
RCP 4.5 low + 35%	50	213	505	1226	3389
RCP 4.5 med	5	30	88	265	1233
RCP 4.5 med + 17%	17	74	201	516	2079
RCP 4.5 med + 35%	46	165	399	945	3066
RCP 4.5 high	3	19	61	189	732
RCP 4.5 high + 17%	10	50	140	419	1407
RCP 4.5 high + 35%	26	107	289	703	1986

Grand Ronde MPG QET 50					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
Natural and Hatchery Origin Spawners					
Catherine Creek					
Proposed Action	0.00	0.00	0.00	0.00	0.01
Proposed Action+17%	0.00	0.00	0.00	0.00	0.00
Proposed Action+ 35%	0.00	0.00	0.00	0.00	0.00
RCP8.5 low	0.00	0.00	0.00	0.00	0.01
RCP8.5 low +17%	0.00	0.00	0.00	0.00	0.00
RCP8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP8.5 mean	0.00	0.00	0.00	0.01	0.01
RCP8.5 mean +17%	0.00	0.00	0.00	0.00	0.01
RCP8.5 mean + 35%	0.00	0.00	0.00	0.00	0.00
RCP8.5 high	0.00	0.00	0.00	0.00	0.01
RCP8.5 high +17%	0.00	0.00	0.00	0.00	0.00
RCP8.5 high + 35%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 low	0.00	0.00	0.00	0.00	0.01
RCP 4.5 low + 17%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 med	0.00	0.00	0.00	0.00	0.00
RCP 4.5 med + 17%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 med + 35%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 high	0.00	0.00	0.00	0.00	0.00
RCP 4.5 high + 17%	0.00	0.00	0.00	0.00	0.01
RCP 4.5 high + 35%	0.00	0.00	0.00	0.00	0.00
Lostine					
Proposed Action	0.00	0.00	0.00	0.00	0.00
Proposed Action+17%	0.00	0.00	0.00	0.00	0.00
Proposed Action+ 35%	0.00	0.00	0.00	0.00	0.00
RCP8.5 low	0.00	0.00	0.00	0.00	0.00
RCP8.5 low +17%	0.00	0.00	0.00	0.00	0.00
RCP8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP8.5 mean	0.00	0.00	0.00	0.00	0.00
RCP8.5 mean +17%	0.00	0.00	0.00	0.00	0.00
RCP8.5 mean + 35%	0.00	0.00	0.00	0.00	0.00
RCP8.5 high	0.00	0.00	0.00	0.00	0.00
RCP8.5 high +17%	0.00	0.00	0.00	0.00	0.00
RCP8.5 high + 35%	0.00	0.00	0.00	0.00	0.00

Grand Ronde MPG QET 50					
	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
RCP 4.5 low	0.00	0.00	0.00	0.00	0.00
RCP 4.5 low + 17%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 med	0.00	0.00	0.00	0.00	0.00
RCP 4.5 med + 17%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 med + 35%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 high	0.00	0.00	0.00	0.00	0.00
RCP 4.5 high + 17%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 high + 35%	0.00	0.00	0.00	0.00	0.00
Upper Grande Ronde					
Proposed Action	0.00	0.00	0.00	0.00	0.00
Proposed Action+17%	0.00	0.00	0.00	0.00	0.00
Proposed Action+ 35%	0.00	0.00	0.00	0.00	0.00
RCP8.5 low	0.00	0.00	0.00	0.00	0.00
RCP8.5 low +17%	0.00	0.00	0.00	0.00	0.00
RCP8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP8.5 mean	0.00	0.00	0.00	0.00	0.00
RCP8.5 mean +17%	0.00	0.00	0.00	0.00	0.00
RCP8.5 mean + 35%	0.00	0.00	0.00	0.00	0.00
RCP8.5 high	0.00	0.00	0.00	0.00	0.00
RCP8.5 high +17%	0.00	0.00	0.00	0.00	0.00
RCP8.5 high + 35%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 low	0.00	0.00	0.00	0.00	0.00
RCP 4.5 low + 17%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 med	0.00	0.00	0.00	0.00	0.01
RCP 4.5 med + 17%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 med + 35%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 high	0.00	0.00	0.00	0.00	0.00
RCP 4.5 high + 17%	0.00	0.00	0.00	0.00	0.00
RCP 4.5 high + 35%	0.00	0.00	0.00	0.00	0.00
Wild origin spawners					
Catherine Creek					
Proposed Action	0.03	0.05	0.07	0.08	0.10
Proposed Action+17%	0.01	0.02	0.02	0.03	0.05
Proposed Action+ 35%	0.00	0.01	0.01	0.02	0.04

Grand Ronde MPG QET 50					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
RCP8.5 low	0.03	0.05	0.06	0.08	0.09
RCP8.5 low +17%	0.00	0.01	0.02	0.04	0.05
RCP8.5 low + 35%	0.00	0.01	0.01	0.02	0.04
RCP8.5 mean	0.05	0.07	0.09	0.10	0.14
RCP8.5 mean +17%	0.02	0.03	0.05	0.06	0.09
RCP8.5 mean + 35%	0.00	0.02	0.02	0.04	0.05
RCP8.5 high	0.08	0.11	0.13	0.15	0.19
RCP8.5 high +17%	0.03	0.06	0.07	0.09	0.12
RCP8.5 high + 35%	0.01	0.03	0.05	0.06	0.08
RCP 4.5 low	0.02	0.04	0.06	0.07	0.10
RCP 4.5 low + 17%	0.00	0.02	0.03	0.04	0.05
RCP 4.5 low + 35%	0.00	0.01	0.01	0.02	0.03
RCP 4.5 med	0.05	0.07	0.09	0.11	0.13
RCP 4.5 med + 17%	0.02	0.04	0.05	0.06	0.08
RCP 4.5 med + 35%	0.00	0.01	0.01	0.02	0.04
RCP 4.5 high	0.07	0.09	0.11	0.13	0.17
RCP 4.5 high + 17%	0.02	0.04	0.05	0.07	0.10
RCP 4.5 high + 35%	0.00	0.01	0.02	0.03	0.05
Lostine River					
Proposed Action	0.00	0.00	0.00	0.01	0.01
Proposed Action+17%	0.00	0.00	0.00	0.00	0.00
Proposed Action+ 35%	0.00	0.00	0.00	0.00	0.01
RCP8.5 low	0.00	0.01	0.01	0.02	0.03
RCP8.5 low +17%	0.00	0.00	0.00	0.00	0.01
RCP8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP8.5 mean	0.03	0.04	0.06	0.08	0.10
RCP8.5 mean +17%	0.00	0.01	0.02	0.03	0.05
RCP8.5 mean + 35%	0.00	0.00	0.01	0.01	0.02
RCP8.5 high	0.10	0.12	0.14	0.17	0.19
RCP8.5 high +17%	0.04	0.07	0.08	0.09	0.12
RCP8.5 high + 35%	0.01	0.03	0.04	0.05	0.07
RCP 4.5 low	0.00	0.00	0.01	0.02	0.03
RCP 4.5 low + 17%	0.00	0.00	0.00	0.01	0.02
RCP 4.5 low + 35%	0.00	0.00	0.00	0.00	0.01
RCP 4.5 med	0.00	0.01	0.02	0.03	0.05

Grand Ronde MPG QET 50					
	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
RCP 4.5 med + 17%	0.00	0.00	0.01	0.02	0.03
RCP 4.5 med + 35%	0.00	0.00	0.00	0.01	0.02
RCP 4.5 high	0.05	0.07	0.08	0.10	0.12
RCP 4.5 high + 17%	0.01	0.02	0.04	0.05	0.07
RCP 4.5 high + 35%	0.00	0.01	0.02	0.03	0.05
Minam River					
Proposed Action	0.22	0.25	0.28	0.31	0.35
Proposed Action+17%	0.12	0.14	0.17	0.20	0.23
Proposed Action+ 35%	0.07	0.09	0.12	0.14	0.16
RCP8.5 low	0.27	0.31	0.34	0.38	0.41
RCP8.5 low +17%	0.15	0.18	0.20	0.23	0.26
RCP8.5 low + 35%	0.09	0.11	0.14	0.15	0.18
RCP8.5 mean	0.32	0.36	0.39	0.42	0.45
RCP8.5 mean +17%	0.23	0.26	0.29	0.32	0.37
RCP8.5 mean + 35%	0.13	0.15	0.18	0.20	0.25
RCP8.5 high	0.35	0.40	0.43	0.47	0.52
RCP8.5 high +17%	0.26	0.30	0.33	0.37	0.41
RCP8.5 high + 35%	0.14	0.18	0.21	0.23	0.29
RCP 4.5 low	0.25	0.30	0.33	0.35	0.40
RCP 4.5 low + 17%	0.14	0.18	0.20	0.23	0.28
RCP 4.5 low + 35%	0.08	0.11	0.14	0.16	0.18
RCP 4.5 med	0.29	0.33	0.37	0.41	0.44
RCP 4.5 med + 17%	0.18	0.22	0.25	0.28	0.31
RCP 4.5 med + 35%	0.10	0.14	0.16	0.18	0.23
RCP 4.5 high	0.34	0.38	0.41	0.44	0.48
RCP 4.5 high + 17%	0.23	0.26	0.29	0.32	0.37
RCP 4.5 high + 35%	0.13	0.16	0.18	0.21	0.25
Upper Grande Ronde					
Proposed Action	0.40	0.45	0.48	0.50	0.54
Proposed Action+17%	0.29	0.33	0.37	0.40	0.43
Proposed Action+ 35%	0.21	0.24	0.27	0.29	0.34
RCP8.5 low	0.53	0.58	0.61	0.65	0.69
RCP8.5 low +17%	0.38	0.43	0.45	0.48	0.53
RCP8.5 low + 35%	0.26	0.29	0.32	0.35	0.40
RCP8.5 mean	0.66	0.69	0.72	0.74	0.78

Grand Ronde MPG QET 50					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
RCP8.5 mean +17%	0.53	0.56	0.59	0.62	0.68
RCP8.5 mean + 35%	0.42	0.47	0.50	0.52	0.57
RCP8.5 high	0.77	0.80	0.83	0.85	0.88
RCP8.5 high +17%	0.69	0.72	0.75	0.78	0.82
RCP8.5 high + 35%	0.55	0.59	0.63	0.65	0.69
RCP 4.5 low	0.51	0.55	0.58	0.61	0.64
RCP 4.5 low + 17%	0.36	0.40	0.44	0.46	0.51
RCP 4.5 low + 35%	0.26	0.30	0.33	0.35	0.39
RCP 4.5 med	0.59	0.63	0.66	0.69	0.74
RCP 4.5 med + 17%	0.47	0.51	0.54	0.57	0.63
RCP 4.5 med + 35%	0.34	0.37	0.41	0.44	0.49
RCP 4.5 high	0.69	0.72	0.74	0.76	0.79
RCP 4.5 high + 17%	0.58	0.61	0.65	0.69	0.74
RCP 4.5 high + 35%	0.46	0.49	0.53	0.55	0.59
Wenaha River					
Proposed Action	0.14	0.17	0.20	0.22	0.25
Proposed Action+17%	0.05	0.07	0.09	0.10	0.14
Proposed Action+ 35%	0.01	0.02	0.03	0.04	0.07
RCP8.5 low	0.19	0.24	0.26	0.29	0.33
RCP8.5 low +17%	0.06	0.08	0.11	0.13	0.16
RCP8.5 low + 35%	0.01	0.02	0.03	0.05	0.07
RCP8.5 mean	0.31	0.34	0.37	0.41	0.46
RCP8.5 mean +17%	0.12	0.15	0.18	0.21	0.23
RCP8.5 mean + 35%	0.04	0.06	0.07	0.08	0.11
RCP8.5 high	0.39	0.43	0.46	0.49	0.53
RCP8.5 high +17%	0.21	0.24	0.27	0.30	0.34
RCP8.5 high + 35%	0.08	0.11	0.13	0.15	0.18
RCP 4.5 low	0.20	0.23	0.25	0.28	0.32
RCP 4.5 low + 17%	0.06	0.09	0.11	0.12	0.16
RCP 4.5 low + 35%	0.00	0.02	0.04	0.05	0.06
RCP 4.5 med	0.25	0.30	0.33	0.35	0.40
RCP 4.5 med + 17%	0.09	0.13	0.15	0.17	0.19
RCP 4.5 med + 35%	0.02	0.04	0.05	0.06	0.08
RCP 4.5 high	0.31	0.36	0.39	0.43	0.47
RCP 4.5 high + 17%	0.16	0.19	0.21	0.24	0.27

Grand Ronde MPG QET 50					
	Percentiles				
Population, scenario	5%	25%	50%	75%	95%
RCP 4.5 high + 35%	0.06	0.08	0.09	0.12	0.15

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