Science Bulletin

Oregon Department of Fish and Wildlife

Number 2022-1
Fish Division

2022 Assessment of Naturally Produced Summer Steelhead in the Umpqua River Basin

Disclaimer

Data, analyses, and conclusions inherently have assumptions, limitations, and context associated with them. These add to the complexity and nuance of results and should be recognized. The Oregon Department of Fish and Wildlife shall not be held liable for improper or incorrect use or interpretation of the data, analyses, and conclusions described and contained herein.

Data were compiled from a variety of sources and care was taken in the creation of these datasets and their treatment. The data and analyses contained herein represent the best information available at the time the assessment was completed. However, data, analyses, and conclusions may be revised as new data, analyses, or information become available.

This report should be cited as:

ODFW (Oregon Department of Fish and Wildlife). 2022. 2022 assessment of naturally produced summer steelhead in the Umpqua River basin. Science Bulletin 2022-1. ODFW, Salem.

ODFW prohibits discrimination on the basis of race, color, national origin, age, sex or disability. If you believe you have been discriminated against as described above in any program, activity or facility, or if you desire further information, please contact the Deputy Director, Fish & Wildlife Programs, ODFW, 4034 Fairview Industrial Dr. SE, Salem, OR 97302, or call 503-947-6000, or write to the Chief, Public Civil Rights Division Department of the Interior, 1849 C Street NW, Washington, DC 20240.

The information in this report will be furnished in alternate format for people with disabilities, if needed. Please call 503-947-6002 or e-mail odfw.info@odfw.oregon.gov to request an alternate format.

2022 Assessment of Naturally Produced Summer Steelhead in the Umpqua River Basin

Oregon Department of Fish and Wildlife
Conservation and Recovery Program
28655 Highway 34
Corvallis, OR 97333

April 1, 2022

TABLE OF CONTENTS

TABLES

FIGURES

SUMMARY

After a record low return of adult, naturally produced North Umpqua River summer steelhead to Winchester Dam in 2021, we conducted an expedited assessment to determine: whether there has been a change in the current status of the population, the cause of the downturn, and the future outlook for the population. Primary outcomes and conclusions include:

CURRENT STATUS – *The downturn in 2021 was unprecedented, but consistent with downturns in other summer steelhead populations. The population exhibits high viability and is expected to rebound as environmental conditions improve. However, this expectation should be tempered based on environmental conditions changing and the potential for more frequent downturns.*

1. The escapement estimate of 449 summer steelhead for 2021 was below the critical abundance threshold (1,200 fish) and was the lowest escapement estimate in the record from 1946–2021.

2. Trend analysis indicated significant correlations between North Umpqua summer steelhead and other summer steelhead populations in Oregon, meaning that these other populations were also experiencing declines, and suggesting that large-scale ocean and/or climate drivers play a substantial role in regulating returns.

3. Population viability modeling indicated that this population is not at any immediate risk of extinction, although there are limitations to this approach noted below. No simulations fell below the critical abundance threshold of 1,200 fish. However, population simulations only periodically rose above the desired abundance threshold of 4,200 fish.

    - Note that the most recent, low return years were unable to be included in this specific analysis because all age classes from recent spawn years have not had time to return. Population viability modeling is dependent on the relationship between spawners and the adult fish they produce, known as recruits, which for steelhead return as adults over a period of several years (up to seven years after spawned). Additionally, population viability modeling determines extinction risk based on existing data and its inherent variability resulting from conditions when the data were collected; the model cannot account for conditions in the future that are different in magnitude or frequency, so precaution should be taken interpreting viability model results in the face of regime shifts such as ocean and climate change.

CAUSE OF DOWNTURN – *The primary cause of the downturn was low ocean survival due to poor marine conditions. A secondary cause was poor freshwater conditions that impacted survival during multiple years of juvenile outmigration and, likely, during 2021 when adults were returning to spawn. Additional potential causes are non-native fish predation on outmigrants (as exacerbated by stream temperature changes), disease, and thiamine deficiency. Incidental fishing mortality and predation by pinnipeds and avian predators do not appear to be factors in*

the downturn. There is no indication that the hatchery summer steelhead program has negatively affected naturally produced summer steelhead, although there are more hatchery fish on natural spawning grounds than the management threshold allows.

4. Primary Cause: Marine conditions played the most significant role in the 2021 downturn. The North Pacific Gyre Oscillation (NPGO) index, an indicator of a climate pattern that substantially influences ocean conditions, is strongly correlated with steelhead recruitment. The NPGO index recently declined to the lowest level observed in the last 70 years. Other marine conditions experienced during ocean rearing by recently returning steelhead cohorts were also poor in 2015-2019 according to multiple marine “stoplight” indicators developed by NOAA.

5. Secondary Cause: Multiple indicators of freshwater stream flow and temperature conditions were poor during 2015-2017, overlapping with the poor ocean conditions of 2015-2019. These poor freshwater conditions occurred during the years when 2021 returning adults were rearing and out-migrating to the ocean as juveniles. Other coastal basins with summer steelhead (the Siletz and the Rogue rivers) also experienced poor freshwater conditions during this period, but in general, freshwater indicators in those two basins were more favorable than those in the North Umpqua. Additionally, water temperatures on the mainstem Umpqua and North Umpqua rivers were particularly poor in 2021 when adults were returning to spawn; although there are no direct data for confirmation, these temperatures may have led to in-river pre-spawning mortality and/or re-entry into the ocean.

6. Other potential contributing factors: In addition to the primary and secondary causes for the 2021 downturn, other factors may be contributing to the recent decline. These include:

    - Potential predation on juvenile steelhead by smallmouth bass and striped bass may be exacerbated by warming conditions, particularly where steelhead outmigrate through bass habitat and when water temperatures exceed 15.5^{o} C. Anecdotal reports suggested the striped bass population may be increasing.

    - No examinations for disease have been conducted for naturally produced Umpqua summer steelhead. However, given the known prevalence of various disease outbreaks at Rock Creek Hatchery and the tendency of disease to worsen in large rivers subject to summer warming, disease should be considered as a potential limiting factor.

    - Thiamine deficiency is considered a potential limiting factor, but very little work on this topic relative to salmonids in the western U.S. has been published. Steelhead appear to require much more thiamine than other salmonids. A small number of winter steelhead eggs from two of three ODFW hatcheries sampled recently were thiamine deficient, and some pilot work is ongoing at the Oregon Hatchery Research Center to explore this issue.

7. Incidental mortality attributable to catch-and-release angling is likely low and was incorporated into the datasets used to develop spawner-recruit curves and population viability analyses. These analyses indicated that the population is viable. Current permanent (no retention of naturally produced fish and terminal gear restrictions) and temporary (angling closures during low returns or poor conditions such as high water temperatures) regulations likely minimize angling-related mortality on naturally produced North Umpqua summer steelhead.

8. There was no evidence to suggest that predation by pinnipeds or birds has increased or decreased in recent years. Harbor seal counts were available for the Umpqua estuary only through 2014 and did not indicate a substantial change in population size for the periods surveyed. Sea lions are not known to frequent the area, and the most recent (2012) surveys of double-crested cormorants in the estuary and lower river yielded estimates of <1,000 birds.

9. The summer steelhead hatchery program does not appear to be negatively impacting natural production.

   - Hatchery smolt releases have: a) no clear relationship with recruitment of naturally produced summer steelhead based on potential interaction at the smolt stage and b) a marginally significant positive relationship with recruitment of naturally produced summer steelhead based on potential interaction at the adult stage.

   - Model simulations indicate that the extinction risk of naturally produced summer steelhead is not affected by hatchery program size or the relative reproductive success of hatchery fish.

   - Hatchery fish are more segregated on natural spawning grounds than previously reported "un-weighted" basin-wide estimates of the proportion of hatchery-origin spawners (pHOS; 34%, 9-year average) or proportions of hatchery summer steelhead counted at Winchester Dam (41%, 9-year average 2013-2021) indicate. Hatchery proportions are near zero in the Steamboat Creek sub-basin, a preferred spawning sub-basin for over half of the naturally produced spawners; estimates are higher (>75%) in the Rock Creek sub-basin, where hatchery smolts are acclimated and released. After accounting for spatial segregation of natural and hatchery origin spawners, the most recent "weighted" basin-wide pHOS estimate is 17% (9-year average).

   - The weighted pHOS estimate is higher than the ≤10% target in the *Coastal Multi-Species Conservation and Management Plan*. Analyses indicate that reducing the hatchery program or removing more hatchery fish at Rock Creek Hatchery should reduce pHOS below the target level.

FUTURE OUTLOOK – *Data to inform models are not currently spatially and temporally robust enough to predict specific impacts of climate and ocean changes on North Umpqua summer steelhead. Upper tributary freshwater habitat should still be able to support juvenile steelhead even though instream flows and temperature are expected to decrease and increase, respectively, and fires are expected to affect flows and temperature. Changing ocean conditions are likely to impact steelhead marine growth and survival, but the effects are highly uncertain. Overall, there are significant challenges and some positive indications of resilience. It is unclear at this time how freshwater and marine changes will interact to affect North Umpqua summer steelhead, which have a diverse life history that has potential to buffer changes.*

10. Species distribution modeling of both summer and winter juvenile steelhead in freshwater suggests the current rearing strongholds are in the lower Smith River and upper watersheds of the North Umpqua basin. Future predictions indicate little change in the distribution of this species even though future conditions will be warmer throughout the Umpqua basin (specifically in the lower mainstem Umpqua and North Umpqua rivers, but also in North Umpqua tributary streams).

11. Modeling of stream temperatures and stream flows under future climate and post-fire conditions suggested that stream temperatures will continue to increase under both scenarios due to increased air temperatures and immediate loss of riparian shading vegetation, respectively. These effects could diminish in time when the vegetation regrows, but our modeling focused on the mean monthly response over the 30 years post-fire.

12. We modeled fire simulations intended to capture the scale of the 2020 and 2021 fires in Canton Creek and Rock Creek in which ~15% and ~73% of the watersheds were burned, respectively. Post-fire simulations suggested an increase in instream flows; however, this effect was due to the loss of vegetation on the landscape. The increase in fire activity and subsequent temporary increases in instream flows is unlikely to offset a modeled climatic shift in instream flow availability, which predicted decreased flows throughout the year.

13. Marine conditions are expected to change in the future, but the effect of these changes on steelhead growth and survival are difficult to predict. Projections for sea surface temperature and marine heat waves suggest that conditions will be more challenging for steelhead in the future. Projections for upwelling indicate more stable near-term conditions and long-term changes that could increase marine productivity in the near-shore zone.

INTRODUCTION

The North Umpqua population of summer steelhead *Oncorhynchus mykiss* is one of only four naturally produced[1] summer steelhead populations on the Oregon Coast. This culturally and ecological important population is the largest of two such populations that constitute Oregon's Coastal summer steelhead species management unit (SMU; Figure 1). Coastal summer steelhead have been designated a sensitive species by the Oregon Department of Fish and Wildlife (ODFW) (ODFW 2021), and summer steelhead are an Oregon Conservation Strategy species (ODFW 2016a). Winter steelhead are much more widely distributed in the Coastal area, and are part of their own, separate SMU.

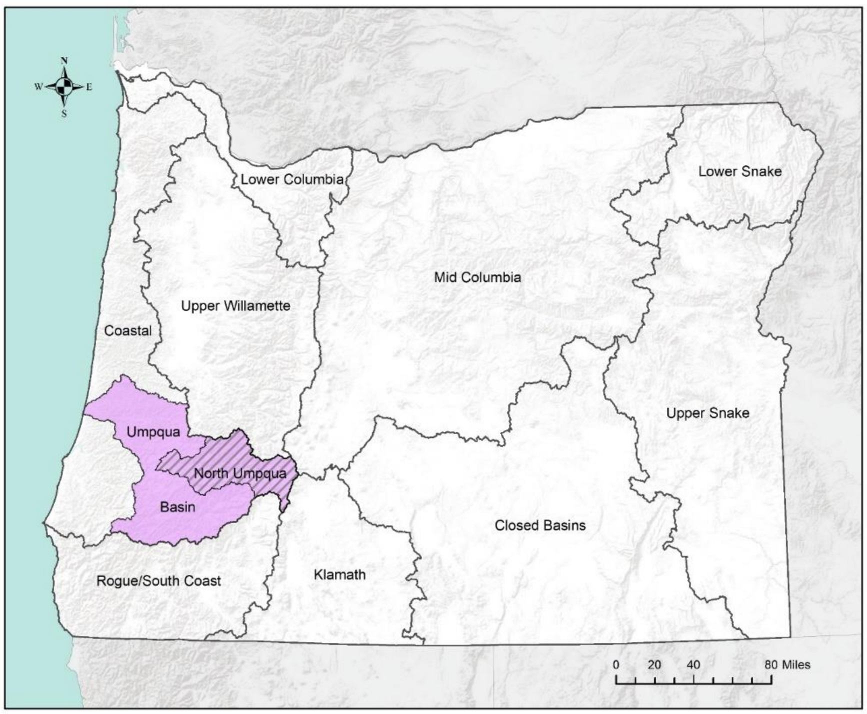


Figure 1. Generalized map of Oregon Species Management Units (SMUs). The Umpqua and North Umpqua basins are within the Coastal SMU.

In addition to their ecological significance, North Umpqua summer steelhead support valuable angling opportunities. The Umpqua River is arguably one of the most important freshwater fisheries in the state, offering opportunities for spring and fall Chinook salmon *O. tshawytscha*, coho salmon *O. kisutch*, summer and winter steelhead, cutthroat trout *O. clarkii*, American shad *Alosa sapidissima*, striped bass *Morone saxatilis*, smallmouth bass *Micropterus dolomieu*, and

[1]We use this term as a convention throughout to indicate fish that were born in the wild. Naturally produced fish may have hatchery-origin or naturally produced parents. The term "wild" is used where referring to known wild fish (i.e., before the implementation of hatchery programs).

others. However, angling for summer steelhead is particularly popular, in part because of their return to the river as adults during periods of fair weather. Public demand for increased opportunities to harvest summer steelhead, in conjunction with management goals for escapement, led to the initiation of a summer steelhead hatchery program in 1958 (fully described in the **Hatchery** section of this report). Mark-selective fisheries have been seasonally implemented in portions of the North and mainstem Umpqua rivers since 1999 to increase protections for naturally produced summer steelhead (ODFW 1999).

Management of the Coastal summer steelhead SMU is guided by the *Coastal Multi-Species Conservation and Management Plan* (CMP), adopted by the Oregon Fish and Wildlife Commission in 2014. During development of the CMP, the naturally produced North Umpqua population of summer steelhead was thoroughly assessed and considered to be viable, with no indicators of concern for the population itself relative to all viable salmonid population parameters considered in the assessment (ODFW 2014). Nevertheless, the CMP directs management actions to achieve increases in abundance that are intended to support improved angling opportunity and as a buffer against future threats. To this end, the CMP established two abundance criteria, a desired abundance target that represents a mean future spawner abundance goal and a critical abundance threshold that represents the point at which the conservation of the population could be in jeopardy if an observed downward trend continues. Based on the record of adult passage counts at Winchester dam, after adjustments to account for losses prior to spawning, the CMP set the desired and critical abundance targets at 4,200 and 1,200, respectively.

As with all salmonid populations, abundance of naturally produced North Umpqua summer steelhead has cycled through periods of relatively high and low abundance since counts of returning adult summer steelhead began at Winchester Dam (river kilometer 190) in 1946. Returns declined moderately from 2012 to 2020 but did not fall below the CMP critical abundance threshold (ODFW 2020). However, counts in 2021 declined dramatically, with a final escapement estimate of 449 naturally produced fish. This estimate is well below the critical abundance threshold and is the smallest return ever recorded.

Given this dramatic decline, ODFW initiated an expedited assessment of the population and investigation into factors that may have contributed to the decline. In this report we evaluate the following for North Umpqua summer steelhead: (1) current population status; (2) potential limiting factors (LFs), including marine conditions, freshwater habitat, predators, hatchery effects, and harvest; and (3) future outlook for freshwater and marine habitat. Where data were readily available, we developed models and conducted quantitative statistical assessments. In other cases, we reviewed the available literature and provided qualitative assessments.

CURRENT POPULATION STATUS

North Umpqua Summer Steelhead Life History

Summer steelhead are differentiated from winter steelhead by the timing of entry to freshwater. Adult winter steelhead on the Oregon Coast typically enter freshwater in November to April, before spawning in December through May (ODFW 2014). In contrast to this relatively tight temporal coupling of adult migration and spawning, adult North Umpqua summer steelhead enter freshwater beginning as early as late spring and progressing through late fall (McGie 1994). These fish are counted as they pass Winchester Dam in May through November. After passing the dam, North Umpqua summer steelhead hold in freshwater through the summer and fall prior to spawning in the following year (January through May).

A summer-returning life history strategy for steelhead can be advantageous when the distance from the ocean to spawning habitats is long or where a natural barrier favors passage during summer flow conditions. Winter velocity barriers likely favor a summer returning life history in key production areas of the North Umpqua basin, which together with availability of cold water, constrains production of summer steelhead to areas upstream from Winchester Dam (Figure 2). For example, the Steamboat Creek sub-basin supports a substantial proportion of the total production of the North Umpqua summer steelhead population. This is also the case in the Siletz River (the other Coastal SMU population), where winter flows create a velocity barrier at Siletz Falls, favoring a summer-returning steelhead population in the upper portion of that basin. In cases such as these, spawning distributions for summer steelhead can be more constrained than their more broadly distributed winter-run counterparts.

North Umpqua summer steelhead rear in freshwater until migrating out of the North Umpqua basin primarily as age-2 and age-3 smolts. Smolt outmigration occurs from January through July (ODFW 2005), typically peaking in late spring (March – June; Daily 1990). Spawners predominantly return as 2-salt and 3-salt adults, but smaller proportions return after only one year or after spending 4 years at sea (ODFW Unpublished Data). The protracted and variable freshwater and ocean rearing periods mean that adult spawners returning in a given year are derived from multiple spawning years and experienced different combinations of conditions in freshwater and in the ocean.

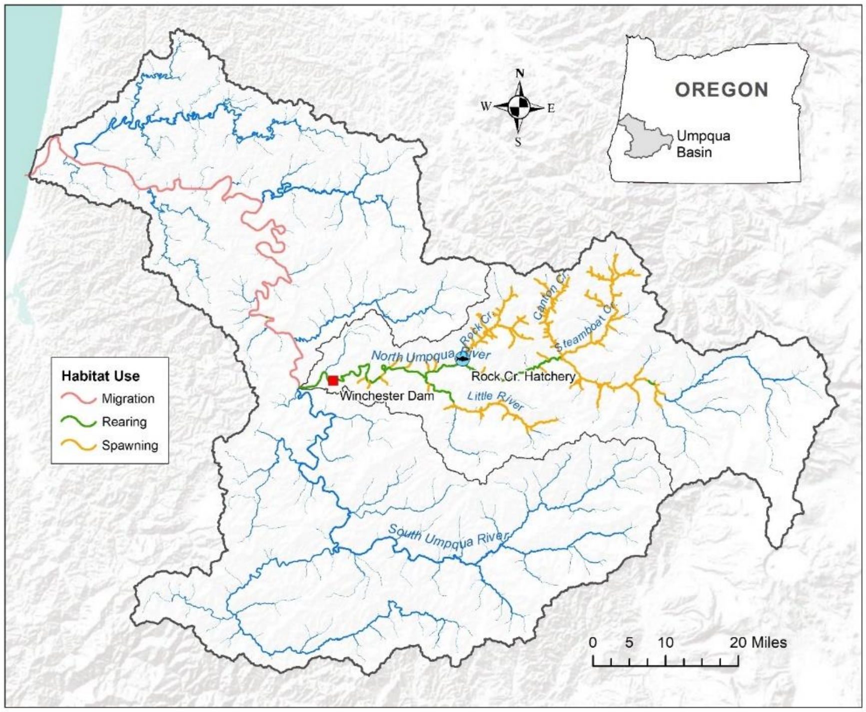


Figure 2. The Umpqua River basin and associated features.
Habitat use designations are from the ODFW fish distribution layer for summer steelhead.

The summer steelhead life history discussed above poses several challenges for managers, particularly when monitoring is based on counts of returning adults passing a dam, as is the case for North Umpqua summer steelhead. These include:

- Passage counts in a given year at Winchester Dam indicate spawners in the following year (i.e., Spawn Year = Count Year +1), and counts must be adjusted to account for losses that occur prior to spawning.
- Dam counts do not provide information on productivity of the different sub-basins available for spawning upstream from Winchester Dam. Differential use of the North Umpqua sub-basins by both naturally produced and hatchery summer steelhead is further discussed in the **Hatchery** section of this assessment.
- The diverse freshwater and ocean age structure for North Umpqua summer steelhead can effectively spread risk for the population, but it can also pose challenges for identifying statistically significant environmental covariates for use in assessing population dynamics or forecasting future returns.

In this assessment, we have addressed these challenges to the extent feasible with the available data and each is discussed further in the sections below.

Abundance

The CMP status assessment concluded that the North Umpqua summer steelhead population was viable with no indicators of concern with the viability results. In this report, we provide an updated assessment of population performance relative to CMP abundance targets, evaluation of recent abundance trends, and evaluation of an updated population viability analysis. Current and projected future distributions for juveniles are also addressed (**Species Distribution Modeling** section).

The CMP established a Desired Abundance target and a Critical Abundance threshold for the North Umpqua summer steelhead population. Desired Abundance is the mean future wild spawner abundance goal, which actions across management categories identified in the CMP are trying to attain. Desired Abundance for populations has no direct harvest management implications. The Desired Abundance target (4,200) was set to the 75th percentile of the lognormal distribution of naturally produced spawner abundance estimates for the period from 1946-2009. Attaining Desired Abundance would require an increase in the median abundance of ~30% over the median abundance for the same period (3,200). This increase in Desired Abundance is intended to strengthen populations, provide greater resiliency to future changes in limiting factors such as exacerbating climate and ocean change and development associated with human population growth or expansion, and provide more consistent and improved fisheries.

The Critical Abundance threshold for the North Umpqua summer steelhead population was set at 1,200. This threshold represents an historically low wild spawner abundance, below which long-term persistence becomes uncertain. The Critical Abundance threshold was calculated as (SMSY+20%)*0.5, where SMSY is the spawner abundance at maximum sustainable yield from the stock-recruit relationship for the population, the 20% increase in SMSY is a risk buffer for uncertainty in the estimation of SMSY and future threats to populations and taking half of the resulting product is consistent with Critical Abundances identified in other plans. Note that Critical Abundance levels are intended to be high enough to allow time for management actions to be implemented to improve a population's status before risk becomes too great, but not too high that they unnecessarily constrain fisheries when viability is not at risk.

The CMP's abundance thresholds are assessed based on estimates of naturally produced escapement. Escapement is estimated by adjusting summer period passage counts of naturally produced steelhead at Winchester Dam (May 1 to November 30) to account for a change in counting methods in 1992 and fishery mortality. Fishery mortality includes direct harvest (when permitted) and, after prohibition of harvest, an estimate of mortality incidental to catch-and-release angling. Abundance is assessed annually and reported in the CMP's Wild Fish Monitoring Summaries (available at https://www.dfw.state.or.us/fish/crp/coastal_multispecies.asp).

From 2014 through 2020, annual escapement estimates were lower than Desired Abundance but higher than Critical Abundance (Table 1). However, the escapement estimate for 2021 was below Critical Abundance, and it was the lowest escapement estimate in the record from 1946 – 2021 (Figure 3).

Table 1. Naturally produced spawner escapement estimates for North Umpqua summer steelhead. Population estimates are since approval of the *Coastal Multi-Species Conservation and Management Plan* (CMP). Naturally produced spawner escapement estimates are summer period counts at Winchester Dam (May 1 – November 30), adjusted to account for fishery-related mortality above Winchester Dam. The CMP Desired Abundance Target and Critical Abundance Threshold are 4,200 and 1,200, respectively (see CMP, Table AIII:2).

Run Year	Naturally Produced Escapement
2014	2,182
2015	1,598
2016	3,652
2017	2,472
2018	1,820
2019	1,924
2020	1,452
2021[P]	449

[P]2021 Estimate is provisional and subject to change.

Trend analysis of a population with cyclic abundance can produce results that are highly dependent on the period considered. For example, there is no significant trend evident in the North Umpqua naturally produced summer steelhead abundance time series from 1946-2021, but there are cycles of higher and lower abundances in the record (Figure 3). Assessment of trend using different start and end points would yield very different conclusions about the trajectory of the population. For this reason, we draw comparisons to other summer steelhead abundance time series to set a context for the recent decline in abundance of North Umpqua summer steelhead.

The North Umpqua summer steelhead population was not alone in experiencing a substantial downturn in 2021. A similar substantial decline in abundance was observed in 2021 in the Siletz summer steelhead population, the other population of summer steelhead within the Coastal SMU. There, the 2021 abundance estimate for natural origin fish fell to the 7th percentile for the period of record (1994-2021) (Figure 4). Hatchery origin returns to both Winchester Dam (North Umpqua) and Siletz Falls in 2021 were the lowest on record (Siletz, 1994-2021; Winchester, 1959-2021).

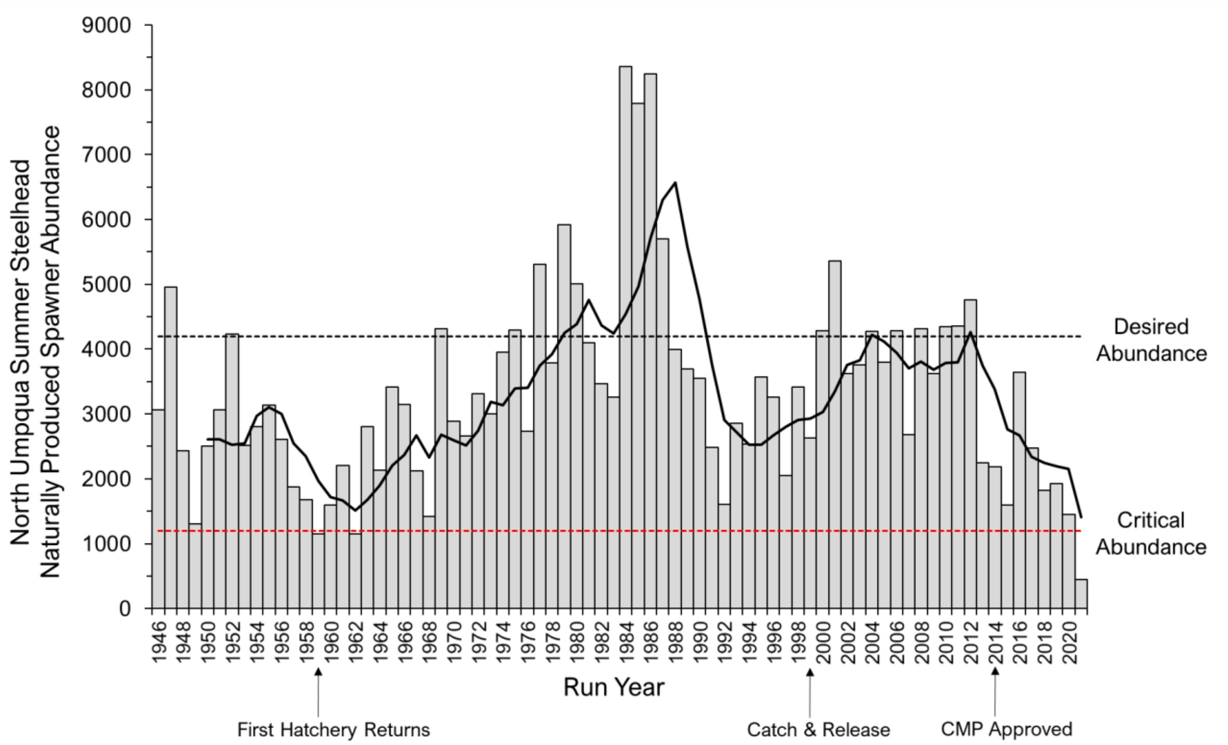


Figure 3. Naturally produced spawner abundance, North Umpqua summer steelhead (run years 1946 -2021). Abundance estimates are based on counts of naturally produced summer steelhead at Winchester Dam, adjusted to account for a change in counting methods in 1992 and fishery mortality. The solid black line is the running five-year geometric mean of abundance estimates. The dashed black and dashed red lines are Coastal Multi-Species Conservation and Management Plan (CMP) targets for Desired Abundance and Critical Abundance, respectively. Arrows indicate the first hatchery origin adult returns (1959), implementation of catch and release regulations for unmarked summer steelhead throughout the basin (1999), and approval of the CMP (2014).

Population trends in the Siletz and the North Umpqua mirrored each other closely from 2003 to 2014 (Figure 5). Both populations declined beginning in 2013, but while the trend in the North Umpqua has steadily declined over the last eight years, the population trend in the Siletz leveled off around 2016. Both populations saw a very sharp decline in 2021, but since the Siletz did not experience the declines observed in the previous four years in the North Umpqua, the status of the Siletz is not much lower than it was three years ago. Trends in the Rogue River differs markedly from the other two basins. When the North Umpqua and Siletz were peaking

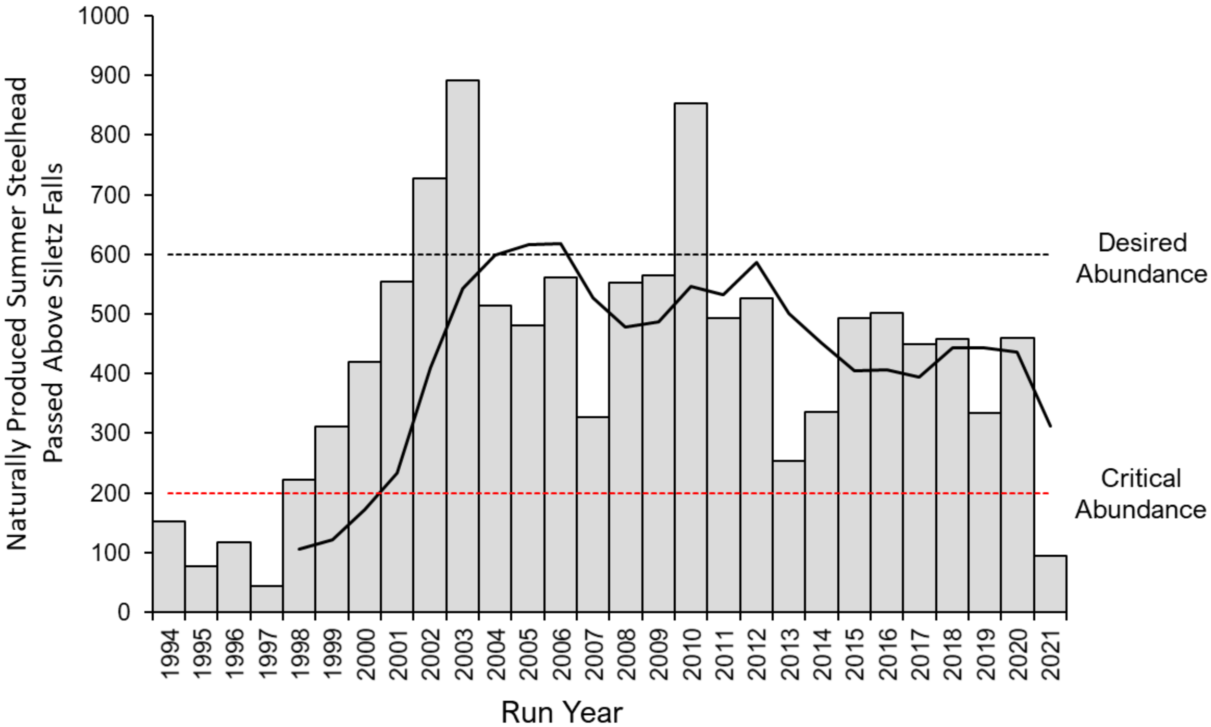
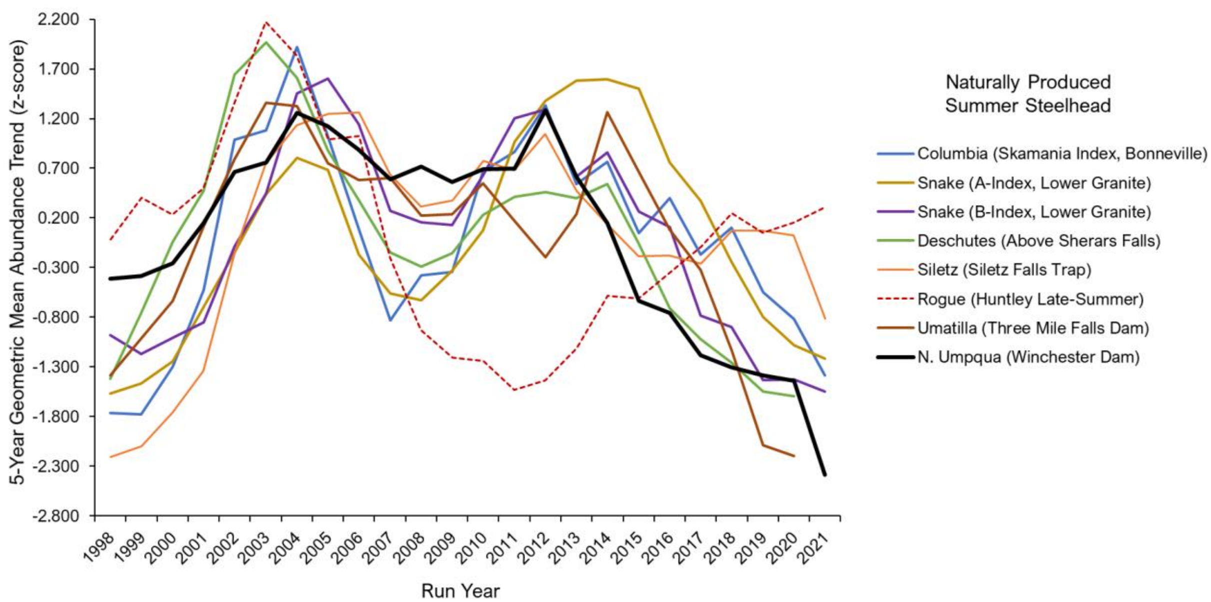


Figure 4. Naturally produced summer steelhead counts at Siletz Falls (run years 1994-2021).
The solid black line is the running five-year geometric mean of abundance estimates. The dashed black and dashed red lines are Coastal Multi-Species Conservation and Management Plan (CMP) targets for Desired Abundance and Critical Abundance, respectively.



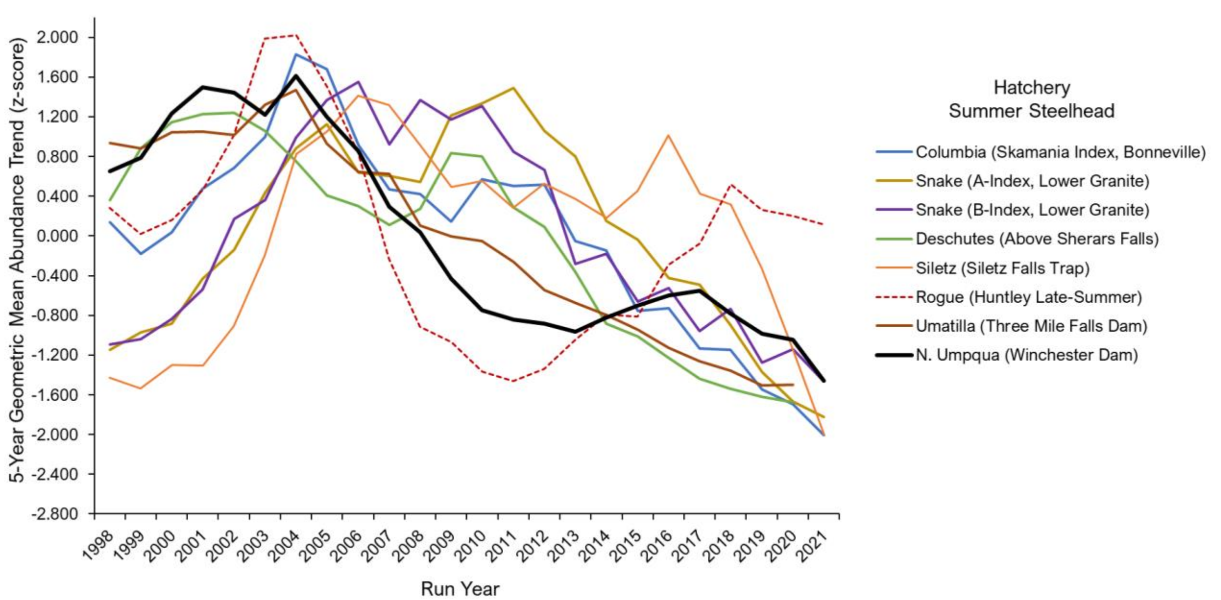


Figure 5. Naturally produced and hatchery summer steelhead abundance trends (run years 1998-2021). Lines are standardized (z-scores) trends of 5-year geometric means for Skamania Index summer steelhead passage at Bonneville Dam, A-Index and B-Index summer steelhead passage at Lower Granite Dam, Umatilla River summer steelhead at Three Mile Falls Dam, Deschutes River summer steelhead migrating above Sherar's Falls, trap counts of Siletz summer steelhead at Siletz Falls, Rogue late-summer steelhead at Huntley Park, and North Umpqua summer steelhead passage at Winchester Dam. Data for 2021 in all populations is provisional and subject to change as estimates are finalized; 2021 estimates for the Deschutes and Umatilla trends are not yet available.

around 2012, Rogue trends were at their lowest. Trends in the Rogue have been increasing since then, while they have been declining in the other two basins.

Comparison of trends among several naturally produced summer steelhead runs indicated significant correlations between North Umpqua summer steelhead and most of the abundance trends analyzed (Table 2). This broad regional coherence among naturally produced summer steelhead abundance trends suggests a role of large-scale ocean and/or climate drivers in recent trends. Despite significant distances among population areas, recent declines are evident across most of the trends assessed. However, the coastal populations (Siletz and North Umpqua) appear to have declined more abruptly from 2020 to 2021 relative to the mean abundances of their respective time series (Figure 5). Interpreting trends in hatchery origin summer steelhead abundance can be complicated by changes in smolt release numbers or other hatchery management changes that differ among populations. Abundance trends for hatchery summer steelhead are somewhat less coherent than those of their naturally produced counterparts. However, most hatchery trends indicate recent declines, with many starting to decline prior to recent declines in complementary trends for naturally produced fish (Figure 5).

We emphasize that 2021 abundance estimates for most trends are provisional or not yet available. It is also notable that not all abundance trends are significantly correlated (i.e., Rogue River late-summer steelhead). Summer steelhead in the Rogue River are thought to occupy a different ocean distribution than more northerly steelhead stocks and they have a half-pounder life history strategy unique to steelhead populations in the Rogue, Klamath, and Eel rivers in the Pacific Northwest. These factors may combine to yield a different response to ocean conditions or regional climate drivers than observed in other northwest summer steelhead populations.

The CMP assessed the trend of the North Umpqua summer steelhead population as having a low probability of decline. However, the decline over the past decade resulted in the 2021 return abundance that is less than 40% of previous lows (Figure 3). This and the steep nature of the decline relative to other correlated abundance trends warrant a re-assessment of population viability. Although population viability analyses are not generally expected to substantially change over short periods of time, we reassessed population viability using an updated stock-recruit function derived from the most current and complete data available. This analysis is described below.

Table 2. Pearson's correlation coefficients for summer steelhead abundance trends.
Trends (standardized 5-year geometric means, z-scores) are for 1998-2021, when data are available for all trends (i.e., starting with run-year 1994). Trends are Rogue River late-summer steelhead (Huntley Park Index), Snake River A-Index and B-Index summer steelhead at Lower Granite Dam (JCRM 2022), Columbia River Skamania Index, Umatilla summer steelhead at Three Mile Falls Dam, Deschutes summer steelhead (above Sherar's Falls), Siletz summer steelhead (Siletz Falls passage count), and North Umpqua summer steelhead at Winchester Dam. All trends are naturally produced adult summer steelhead. All data for 2021 are preliminary and subject to change. Estimates for 2021 are not yet available for the Deschutes and Umatilla rivers. Green shading indicates significant correlation (p-value < 0.05); yellow shading indicates one marginally significant result (p-value = 0.051). Unshaded cells indicate no significant correlation.

Abundance Trend	Rogue	Snake (A-Index)	Snake (B-Index)	Columbia (Skamania)	Umatilla	Deschutes	Siletz
Snake (A-Index)[a]	-0.240						
Snake (B-Index)[a]	-0.086	0.757					
Columbia (Skamania)[b]	0.095	0.818	0.805				
Umatilla[c]	0.175	0.615	0.789	0.663			
Deschutes[d]	0.346	0.497	0.714	0.695	0.879		
Siletz[e]	-0.037	0.625	0.760	0.780	0.498	0.413	
North Umpqua[f]	0.007	0.433	0.839	0.610	0.742	0.818	0.518

[a]Wild A-Index and B-Index summer steelhead passage at Lower Granite Dam. Data are from Table 13 in the 2022 Joint Columbia River Management Staff Report (JCRM 2022); 2021 data are preliminary; passage is through Dec 31, 2021, and subcomponent determination is based on visual sampling.

[b]Wild Skamania Index summer steelhead passage at Bonneville Dam. Data are from Table 12, JCRM 2022. 2021 estimate is preliminary.

[C]Summer steelhead count at Three Mile Falls Dam; 2021 estimate is not yet available.

[d]Mark-recapture estimate of natural origin summer steelhead migration above Sherars Falls. 2021 estimate is not yet available.

[e]Passage count of unmarked summer steelhead at Siletz Falls Trap.

[f]Passage count of unmarked steelhead during the summer counting period (May 1 - November 30) at Winchester Dam.

Population Viability

Data. As indicated in the previous section, North Umpqua summer steelhead spawner and recruit abundance are estimated based on counts at Winchester Dam from 1992-2020. Counting methods changed in 1992 as did ocean conditions related to the Pacific Decadal Oscillation (PDO) and the North Pacific Gyre Oscillation (NPGO; Litzow et al. 2020) making the early 1990s a logical starting point for this analysis. At the time the analysis was conducted, 2020 was the most recent year with final Winchester Dam counts. Spawner abundance is based on escapement of both hatchery and naturally produced adults since the product of a hatchery fish reproducing with a naturally produced fish or two hatchery fish reproducing will be indistinguishable from a pure naturally produced fish (i.e., no fin clip). Recruit abundance is based on the number of naturally produced returns in any given year prior to harvest. Each of these numbers is based on counts at Winchester Dam, adjusted to account for harvest, hatchery summer steelhead passing Winchester Dam during the winter counting period, and retention of hatchery fish at Rock Creek Hatchery. Summer steelhead counted at Winchester Dam in a given year typically spawn early in the following calendar year (e.g., 1993 spawners are based on 1992 counts).

Annual age compositions used to assign naturally produced returners to spawning year and to generate age compositions for population viability analysis (PVA) were derived from scale data acquired in the Volunteer Steelhead Scale Project from 1983-1993 (ODFW unpublished data). The average annual age composition was applied to pre-harvest abundance of naturally produced returns (pre-harvest) to get recruits by spawning year.

Stock-Recruitment Analysis. Stock-recruitment analysis is an accepted technique in fisheries science used to relate the number of spawners in a population to the number of “recruits” they are expected to produce that will survive to a given size/age (Ricker 1954; Hilborn and Walters 2003; Haddon 2011). In this analysis, “recruits” are defined as the number of offspring produced in a given year from a group of spawners (including both hatchery and naturally produced fish) that survived to spawn or would have had they not been harvested. For a detailed explanation of how age compositions are used to calculate recruits by spawning year, see Table A-II: 6 and accompanying text in the CMP (p. 143-144). Due to the life history diversity of steelhead, recruits from a given spawning year return as adults over a period of several years. For this analysis, full recruit information is available for spawning years 1993-2014. Spawning years after 2014 could not be included in the analysis because not enough years have elapsed to allow all recruit age classes to return (e.g., age-7 fish from the 2015 spawning year did not return until 2021, and the 2021 dam count was not finalized prior to this analysis).

After the number of recruits have been estimated, it is possible to relate the abundance of spawners in a given year to the abundance of recruits produced in that spawning year. It is also possible to plot this relationship and fit one of a variety of non-linear recruitment functions (i.e., curves) to the resulting data. For the purposes of this analysis, we examined the fit of density-dependent model with (Ricker) and without (Beverton-Holt; Logistic Hockey Stick) "over-compensation" (i.e., a descending right side of the recruitment curve; Figure S1) as well as a simple density-independent model. These models were parameterized as follows:

$$Ricker: R = \boldsymbol{\alpha} S e^{-\boldsymbol{\beta} S} e^{\boldsymbol{\epsilon}}$$

$$Beverton-Holt: R = \frac{\boldsymbol{\alpha} S}{1 + \boldsymbol{\beta} S} e^{\boldsymbol{\epsilon}}$$

$$LogisticHockeyStick: R = \boldsymbol{\alpha} \theta \gamma \left(1 + e^{-1\boldsymbol{\theta}}\right) \left(\frac{S}{\boldsymbol{\theta} \gamma} - log\left(\frac{1 + e^{(S - \boldsymbol{\gamma})/(\boldsymbol{\theta}\gamma)}}{1 + e^{-1\boldsymbol{\theta}}} \right) \right) e^{\boldsymbol{\epsilon}}$$

$$Density - Independent: R = \boldsymbol{\alpha} S e^{\boldsymbol{\epsilon}}$$

where *R* is recruits, *S* is spawners and *e* is the base of the natural logarithm and Ɛ is the error term. All models were fit using maximum-likelihood estimation (MLE).

Attempts to fit these models using Bayesian methods were unsuccessful or not credible. More advanced methods such as Bayesian analysis can be very "data hungry," requiring large quantities of information to reliably converge and produce reasonable parameter estimates. Exploratory attempts to fit Beverton-Holt and Ricker models to these data resulted in models that failed to converge, produced unacceptable numbers of divergent transitions, and/or resulted in non-nonsensical parameter estimates. This indicates that the data do not contain sufficient information to estimate parameters using Bayesian methods.

Stock-recruitment curves generate two primary parameters, $\boldsymbol{\alpha}$ and $\boldsymbol{\beta}$. The parameter $\boldsymbol{\alpha}$ is referred to as the "intrinsic productivity" and represents the number of recruits produced per spawner at a level where there are no density dependent effects. This makes it a measure of population growth rate. The parameter $\boldsymbol{\beta}$ is the change in curvature of the recruitment curve as the number of spawners increases. This number is usually very small and does not have an obvious real-world meaning.

The number of recruits at equilibrium (*Neq*) is a more understandable metric that represents the number of recruits the population should produce at equilibrium. This is visually represented as the point at which the recruitment curve (shown in black in Figures S1-S4) intersects the 1:1 line (shown in red in Figures S1-S4) and is calculated as:

$$Neq = \frac{log(\boldsymbol{\alpha})}{\boldsymbol{\beta}}$$

This is the number of recruits the population should hover around when all other factors (ocean conditions, climate, habitat, harvest, passage, etc.) are equal.

All four models provided reasonable parameter estimates and visual fit to the data was good (Figures S1-S4) when models were fit using maximum likelihood estimation.

We used AICc to rank models. AICc assesses how well the model fits the data while applying a penalty for increasing complexity. This prevents "overfitting," a potentially serious problem in stock-assessment were the model fits existing data too well and has low or no ability to capture future variability resulting in poor predictions of future conditions. AICc is calculated as:

$$AICc = AIC + \frac{2k^2 + 2k}{n - k - 1} = -2ln(L) + 2k + \frac{2k^2 + 2k}{n - k - 1}$$

Maximum likelihood estimation calculates the negative log likelihood $-ln(L)$, which represents how well the model fits the data, as part of the model fitting procedure. We then calculate the penalty factor for the number of parameters $2k + \frac{2k^2+2k}{n-k-1}$ and add it to the negative log likelihood.

The Ricker and Beverton-Holt models are essentially indistinguishable according to AICc (Table S1). Others have little to no support in the data (AICc >= 4).

We performed further analyses using the Beverton-Holt curve (Figure S2). Ricker curves require over-compensation. There is no indication in the data that over-compensation is occurring, nor can we determine a defensible biological rationale to support it. Also, preliminary analyses showed the Beverton-Holt curve outperformed the Ricker curve once covariates were incorporated to each.

Covariates can improve fit and accuracy of a model. However, too many covariates or covariates with spurious correlations can lead to overfitting and ultimately reduce fit and accuracy. Moreover, examining multiple potential relationships from many possible covariates (i.e., "data dredging") also creates the possibility of developing spurious correlations. We only considered a selected suite of covariates with established relationships and/or with an *a priori* biological rationale for why they might impact the stock recruit relationship. Statistical significance was set at the $\alpha \le 0.1$ level when screening covariates to err on the side of inclusion where relationships may be weak on average but critically important in particular years.

The steelhead life cycle is divided into both a freshwater residency period and an ocean residency period. For the freshwater period, we tested covariates for minimum summer flow and maximum weekly mean temperature (MWMT). These relationships were examined for the first two years of a year classes' life (corresponding to their freshwater residency as juveniles) and for the fifth and sixth year of life (corresponding to the conditions they would encounter as returning spawners during the peak ages for return according to age compositions). The indicators for stream flow and temperature that we used were the annual minimum flow (cfs) at the stream gage on the Umpqua River near Elkton and the mean NorWest/PRISM index of the MWMT at the three important holding pools. For the saltwater period, we examined NPGO and Coastal Upwelling Transport Index (CUTI; a measure of upwelling). These relationships were examined for the fifth through seventh years of a year classes' life. This corresponds to the conditions they would encounter during the duration of their ocean residency. The NPGO index has recently reached some of the lowest values (poorest conditions) since 1950, surpassing previous lows in the early 1990s that also coincided with low returns of North Umpqua summer steelhead and other coastal salmon runs (Figure 6).

PDO, sea surface temperature (SST), two other measures of upwelling (Upwelling Index and Biologically Effective Upwelling Transport Index; BEUTI) and copepod biomass were all considered but ultimately discarded as potential covariates. PDO and SST both correlated very closely with NPGO so including them would simply complicate the model without adding any additional predictive information. Also, recent research suggests that, since ~1990, NPGO has a stronger relationship to salmonid survival than PDO (Kilduff et al. 2015, Litzow et al. 2018). All measures of upwelling were highly correlated (Figure S5). CUTI was selected since it shows the highest correlation, on average, with the other two measures.

MWMT in the sixth year of a year classes' life produced the only marginally significant relationship (Figure S6; $P = 0.09$; $R^2 = 0.14$) with recruits. No other combination of MWMT or minimum summer flow and year had any significant relationship (*P*-value) or demonstrated any explanatory power (R^2). This makes biological sense as most fish return at age 5 or 6 and some spawn multiple times, returning at both ages. Umpqua summer steelhead return through spring and late-summer meaning many would experience high water temperatures potentially impacting their spawning success the following spring.

NPGO in the fourth year of a year classes' life produced the strongest and the most significant relationship (Figure S7; $P = 0.00024$; $R^2 = 0.50$) with recruits. Since age compositions suggest that Umpqua summer steelhead leave for the ocean as smolts between ages two and three, year four would be the first full year of ocean conditions all sub-adult steelhead would experience prior to their starting to return to freshwater at age four and a half and five and a half.

Other research has suggested that the relationship between survival and NPGO is highest in the first year in the ocean corresponding to ages two and three (Burke et al. 2013; Beamish and Mahnken 2001; Holtby et al. 1990; Wells et al. 2008). While both age 2 and age 3 produced significant relationships ($P < 0.05$), they did not explain nearly as much variation ($R^2 = 0.18$ and 0.28, respectively) as did age 4. Therefore, we chose to continue this analysis with NPGO corresponding to age 4 since a plausible biological rationale exists to explain its effectiveness.

Beverton-Holt model fits improved when MWMT and NPGO were included both singly and together in the model (Figures S8, 7, and S9; Table S2). The Beverton-Holt model with only NPGO substantially outperformed all other models except the Beverton-Holt model with MWMT and NPGO (ΔAICc >> 4). However, including MWMT still does not produce a more informative model (ΔAICc = 1.36) despite the additional parameter. The original Beverton-Holt model with no covariates is included for reference (Figure S2). The Beverton-Holt model with only NPGO was used for the remainder of this analysis since it was the best performing and less complicated model.

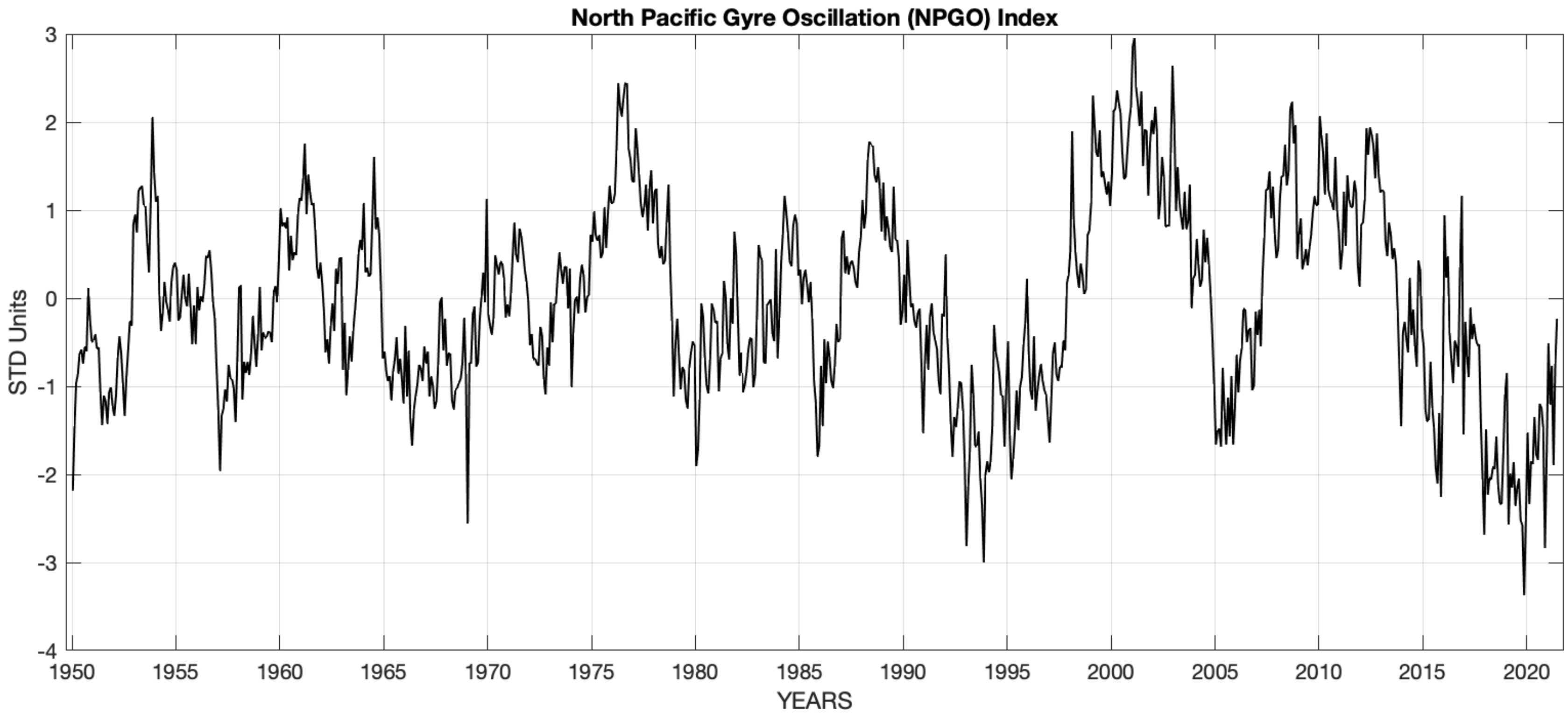


Figure 6. The North Pacific Gyre Oscillation Index(Di Lorenzo et al. 2008) from January 1950 – January 2022. Values after June 2020 are preliminary. Figure downloaded from http://www.o3d.org/npgo/ on 3/17/2020.

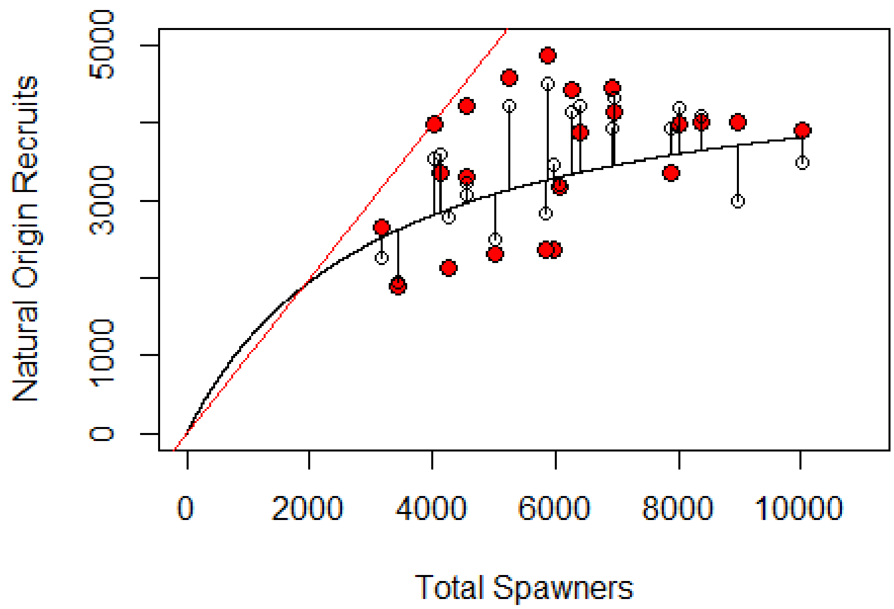


Figure 7. Beverton-Holt curve with NPGO included as covariate.
A Beverton-Holt curve with NPGO included as a covariate fit to spawner-recruit data for the North Umpqua summer steelhead population from 1992-2014. The red line represents the 1:1 relationship where spawner and recruit numbers are equal along their respective axes. The black line represents the fit of the Beverton-Holt curve to the data. The "lollipop" lines coming off the fit curve shows how the addition of NPGO adjusts the predicted values for each year. Open circles represent the new predicted values.

Population Viability Analysis. A Population Viability Analysis (PVA) is a model that uses the stock-recruitment relationship developed above to simulate population abundances into the future. The stock-recruitment relationship is used in the PVA to capture population dynamics and simulate spawner abundance through time. Therefore, the process used to estimate recruits from spawners in the stock-recruitment analysis is simply reversed to estimate spawners from recruits in the PVA (see ***Data*** and ***Stock-Recruitment Analysis*** sections above). The analytical steps are as follows:

1) take spawner abundance in a given year (naturally produced recruits combined with hatchery escapement)
2) use the stock-recruit function to estimate recruits from total spawners
3) use the age composition to apportion recruits to their proper return year (steelhead in the Umpqua mature are 4,5,6 and 7+ years old based on observed age compositions)
4) take the sum of age classes across a given return year to estimate spawner abundance (i.e. - run size) in the next year

It is important to incorporate randomness (i.e. - stochasticity) into PVA models. Without some degree of randomness in parameters and input variables, eventually all models would converge, and extinction risk would be functionally zero. Age composition and NPGO are both randomly varied, within their observed ranges to introduce elements of randomness into the PVA. A random error term is also introduced to capture random error within the parameters of the model.

Extinction occurs in the PVA model when spawner abundance drops below a pre-determined quasi-extinction threshold (QET) for three consecutive years. In a separate process, at critically low spawner abundances recruitment is set to zero simulating the reproductive failure threshold (RFT). These thresholds allow us to account for processes like inbreeding depression, genetic drift, juvenile mortality and other processes that drive populations to extinction at very low abundances (Gilpin and Soulé 1984; Courchamp et al. 2008; Jamieson and Allendorf 2012).

QET/RFT were defined in the CMP as 250 for “large” populations such as North Umpqua summer steelhead. Regardless of the output of the PVA, if *Neq* is less than 500, a population is considered at moderate risk for extinction. This is termed the minimum equilibrium threshold (MET) and reflects the level at which insufficient genetic diversity exists to allow a population to respond to new or changing conditions.

The purpose of the PVA, as implemented here, is to determine current status, not forecast the future. Since the PVA uses past conditions to model extinction risk, it implicitly assumes that conditions when the data was collected will continue for at least the duration of the 100-year simulation. The model is not intended to capture the effects of climate change, ecosystem disruption, habitat alterations or changes in fisheries, for instance.

An approximate but effective way to gauge whether the PVA is simulating the observed population dynamics is to simply superimpose the observed data of the results of a single 100-year simulation. If the simulated data visually matches the observed data, then we can be reasonably certain that the PVA is capturing the dynamics of the population reasonably well.

For our model, the mean and variance of the observed data (red line), for the most part, “look like” the mean and variance of the simulated data (black line; Figure S10). The simulated data do appear to trend slightly lower. This does, in fact, match the data, with the first few years being slightly higher until a drop off in more recent years.

Our PVA ran 100,000 repetitions of a 100-year simulation using the Beverton-Holt model with NPGO as a covariate. For each repetition, hatchery spawners are specified based on an established, density independent relationship between smolt releases and hatchery returns and a random smolt release abundance selected based on recent smolt releases. Age composition was randomly selected from the range of age compositions observed during sampling years. NPGO was also randomly varied within its observed range. Random error was also incorporated into the simulations. We then calculated the number of simulations that resulted in local "extinction" (i.e. - population abundance below QET/RFT = 250) and the *Neq* for comparison with benchmarks from the CMP.

PVA Results. The North Umpqua summer steelhead population never dropped below QET/RFT nor the critical abundance threshold (CAT = 1,200) during any of the 100,000 simulations in the PVA model (Figure 8). *Neq* was 1,473, which is well above the MET threshold of 500. However, the population only sporadically rose above the desired abundance threshold (DAT = 4,200). Most simulated values fell between the critical and desired thresholds.

The PVA model suggests that this population is not at any immediate risk of extinction based on a QET/RFT of 250 and a *Neq* > 500. This would correspond to an Abundance and Productivity (A&P) score of 1 per the CMP (page 149). Simulation results also suggests the population is expected to remain above the critical abundance threshold (i.e., no simulations fell below CAT). However, the population was at or above the desired abundance threshold in only 16% of simulations. These results are consistent with the status determinations made in the CMP, though it should be noted that an older and slightly longer data set (1974-2004) was used in those analyses. The present analysis incorporated the most recent complete data (spawners through 2014 and recruits through 2020) to provide the most up-to-date assessment of current conditions.

We emphasize once more that PVA model results reflect population status under recent observed conditions. Should future conditions change from the conditions under which this data was collected, the status of this population could also change. The potential impacts of changing conditions on North Umpqua summer steelhead are discussed in the **Future Outlook** section below.

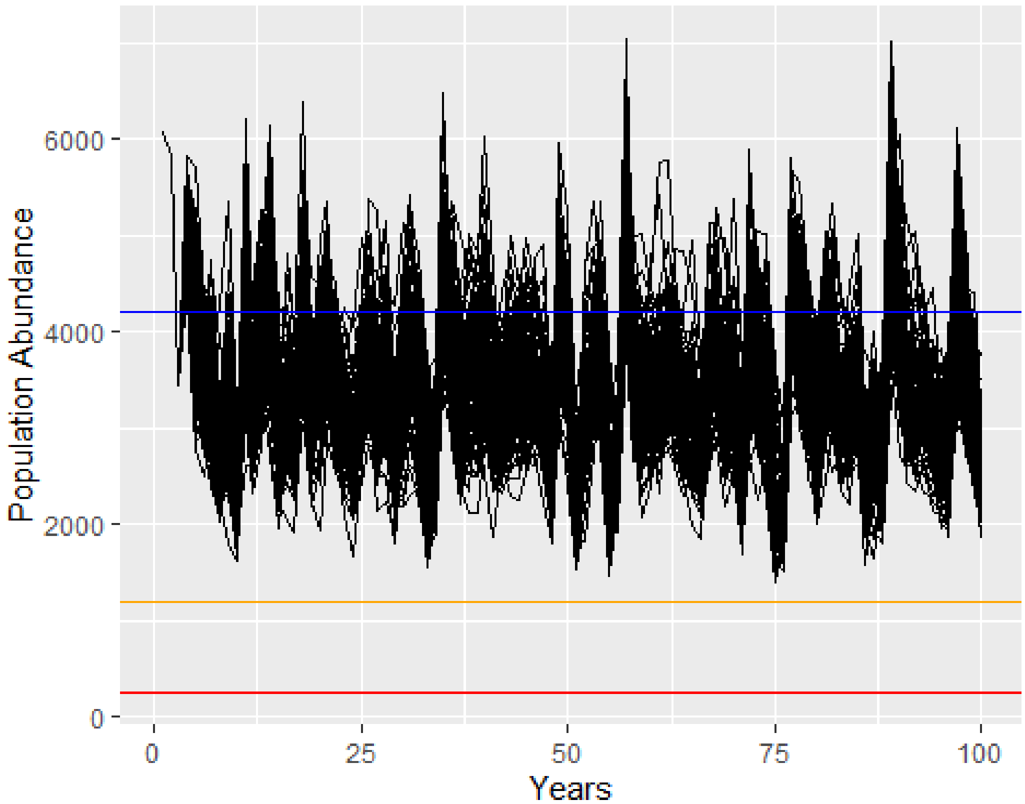


Figure 8. 100-year simulation using a Beverton-Holt curve with NPGO included as a covariate. A sub-sample of 100 of the 100,000 repetitions of the 100-year simulation is displayed. The red line is the QET/RFT, the orange line is the critical abundance threshold, and the blue line is the desired abundance threshold.

Summary: Current Population Status

North Umpqua summer steelhead have experienced cycles of low and high abundance throughout the historical record (ca. 1946). In 2021, the naturally produced return to Winchester Dam was the lowest on record, and low abundances of naturally produced summer steelhead were observed at counting stations or populations areas beyond the Umpqua basin. Since the mid-1990s (standardizing all populations to the same base period), the abundance trend for North Umpqua summer steelhead is significantly correlated with summer steelhead abundance trends across a broad geography.

Correlations among broadly distributed abundance trends implicates large-scale processes (e.g., ocean conditions, regional climate drivers) as overarching moderators of cyclical abundances of summer steelhead. However, more localized conditions can be expected to either moderate or enhance the response of individual populations to these larger-scale drivers of abundance. The recent declining trend in North Umpqua summer steelhead is consistent with recent declines in other populations areas, but the magnitude of the decline in the two Coastal SMU populations (North Umpqua, Siletz) in 2021 may have been larger than that observed in other areas. This can be confirmed only after run year 2021-22 abundance estimates become available for other population areas.

Population viability modeling indicated that the North Umpqua summer steelhead population is not currently at any immediate risk of extinction, nor is the population expected to frequently fall below the CMP critical abundance threshold. However, the population is not expected to consistently meet or exceed the desired abundance target at its current productivity and capacity. These results are consistent with the status determinations made in the CMP and the expectation in the CMP that improvements would be necessary to achieve a median abundance equal to the desired abundance target.

For the model to be accurate in the future, one must assume that ocean, freshwater, climate, and management conditions will remain consistent. However, we know this is likely not the case and expect environmental conditions to be more frequently unfavorable (see **Future Outlook**). Even though this is expected, environmental conditions will generally be within the range of those this population has experienced historically the majority of the time, albeit more frequently associated with conditions that have resulted in lower returns historically but at levels which the population can still self-sustain.

LIMITING FACTORS (CAUSE OF THE DOWNTURN)

Limiting factors are defined as biological, physical, or chemical conditions altered to such an extent by anthropogenic activities that they impede achievement of population biological performance goals. Generally, each limiting factor has the potential to affect the abundance, productivity, spatial structure, and diversity of fish populations. To determine the cause of the 2021 downturn, we evaluated the set of potential limiting factors considered in the CMP, with the exception that marine conditions were also included given the large effect they can have on a population[2]. In this evaluation we selected limiting factors based on professional judgment of ODFW biologists and co-managers, and the assessments were informed by scientific literature, data analyses and interpretation, and professional experience.

Hatchery

Program Description. The North Umpqua summer steelhead hatchery program ("Stock 55") was initiated in the 1950s to augment harvest opportunity in the mainstem Umpqua and North Umpqua rivers. The current program goal is to provide 5,000 hatchery origin steelhead primarily for angler harvest, while maintaining a hatchery stock that is genetically and ecologically similar to the wild population (ODFW 2020a). The program was established using local wild broodstock with the first collections at Winchester Dam in 1955 (OSGC 1956). Naturally produced broodstock have been incorporated since program inception (McGie 1994).

[2] Marine conditions were not assessed as a specific, manageable Limiting Factor in the CMP because there is no ability to control these through local management actions. Marine conditions were considered in the CMP's assessment of potential climate change impacts.

Brood fish are collected primarily at Winchester Dam and supplemented as needed with fish collected at Rock Creek Hatchery (ODFW 2016b).

The first release of approximately 66,000 Stock 55 summer steelhead smolts in 1958 was followed in 1959 by the first one-salt hatchery origin adult summer steelhead passing Winchester Dam. Smolt releases increased through 1970, peaking with a release of 197,768 smolts in 1969 (Figure 9). Increases in smolt releases were mirrored by an increasing trend in hatchery-origin adult summer steelhead passing Winchester Dam through 1972 (Figure 10).

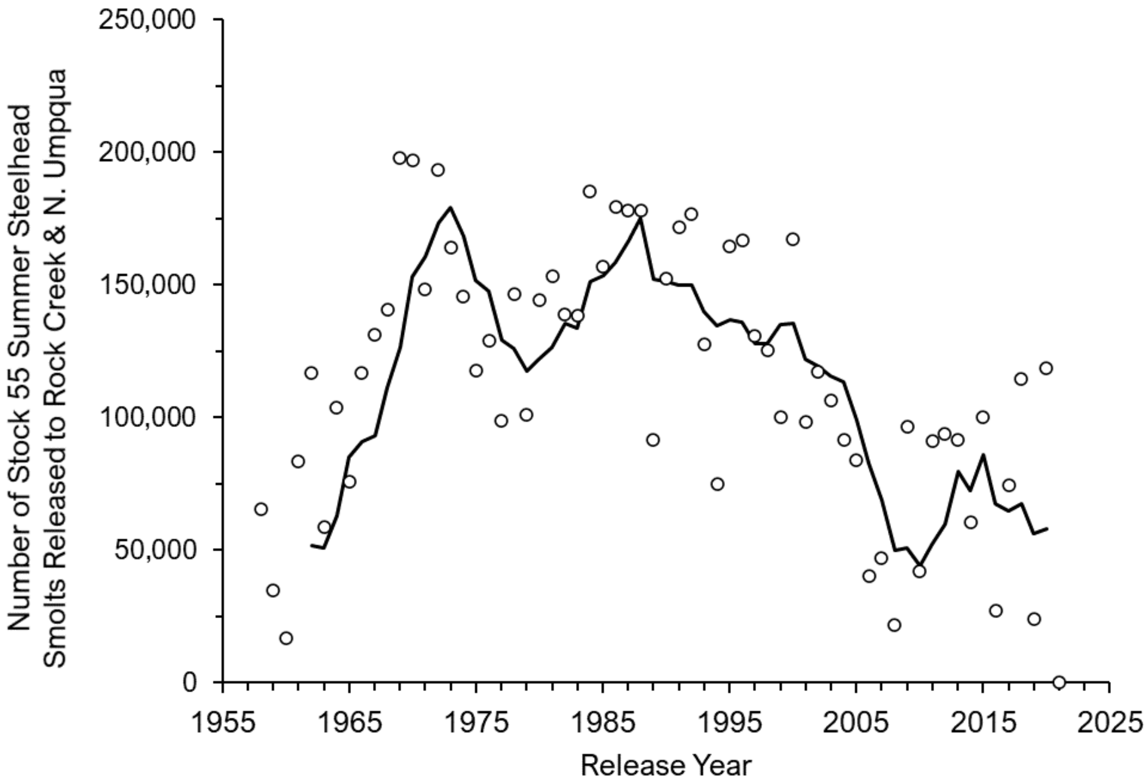


Figure 9 Calendar-year releases of Stock 55 summer steelhead smolts into the North Umpqua basin (1958-2021). The solid line shows the running 5-year moving geometric mean. Data from 1958-1977 and 1983 are from unpublished hatchery records documented in McGie (1994); data from 1978-1982 and 1984-2021 are from the ODFW Hatchery Management Information System. No smolts were released in 2021 due to hatchery losses incurred during the 2020 Archie Creek Fire.

After 1970, the annual release goal for Stock 55 summer steelhead was set at 150,000 smolts to reduce risk to the naturally produced summer steelhead following several years of hatchery returns that were well above returns of naturally produced fish (Anderson et al. 1986). Anderson et al. (1986) attributed lower hatchery returns in 1981-1983 (See Figure 10) to a poor initial response to the movement of the rearing program from Bandon Hatchery to Rock Creek Hatchery and poor ocean rearing conditions. After these lower returns, the annual release goal was increased to 168,000, which is reflected in generally higher releases during the period from 1984 to 2000 (Figure 9; but note exceptions in 1989 & 1994). Releases during this period ranged from approximately 75,000 to 185,000 smolts, averaging 147,806. The 168,000 smolt annual release goal was formalized in the North Umpqua Fish Management Plan (Anderson et al. 1986), approved by the Oregon Fish and Wildlife Commission in 1986. Hatchery returns briefly rebounded before dropping again in the early 1990s, a period of generally poor conditions for ocean rearing (Figure 10).

The CMP (ODFW 2014), approved by the Oregon Fish and Wildlife Commission in 2014, authorized annual releases of 165,000 summer steelhead smolts to the North Umpqua basin. However, post-CMP releases averaged approximately 76,500 smolts from 2015 to 2020 (Range = 23,872 – 118,716; Figure 9). No Stock 55 steelhead smolts were released into the North Umpqua basin in 2021 due to rearing losses caused by the Archie Creek wildfire in 2020. Corresponding hatchery returns have been below the program goal (5,000 hatchery-produced steelhead to the fishery; ODFW 2016b), generally approximating returns near program inception and at other low points in the early 1980s and early 1990s. The provisional estimate of hatchery origin summer steelhead passage at Winchester Dam in 2021 (180 fish) is the lowest on record (Figure 10).

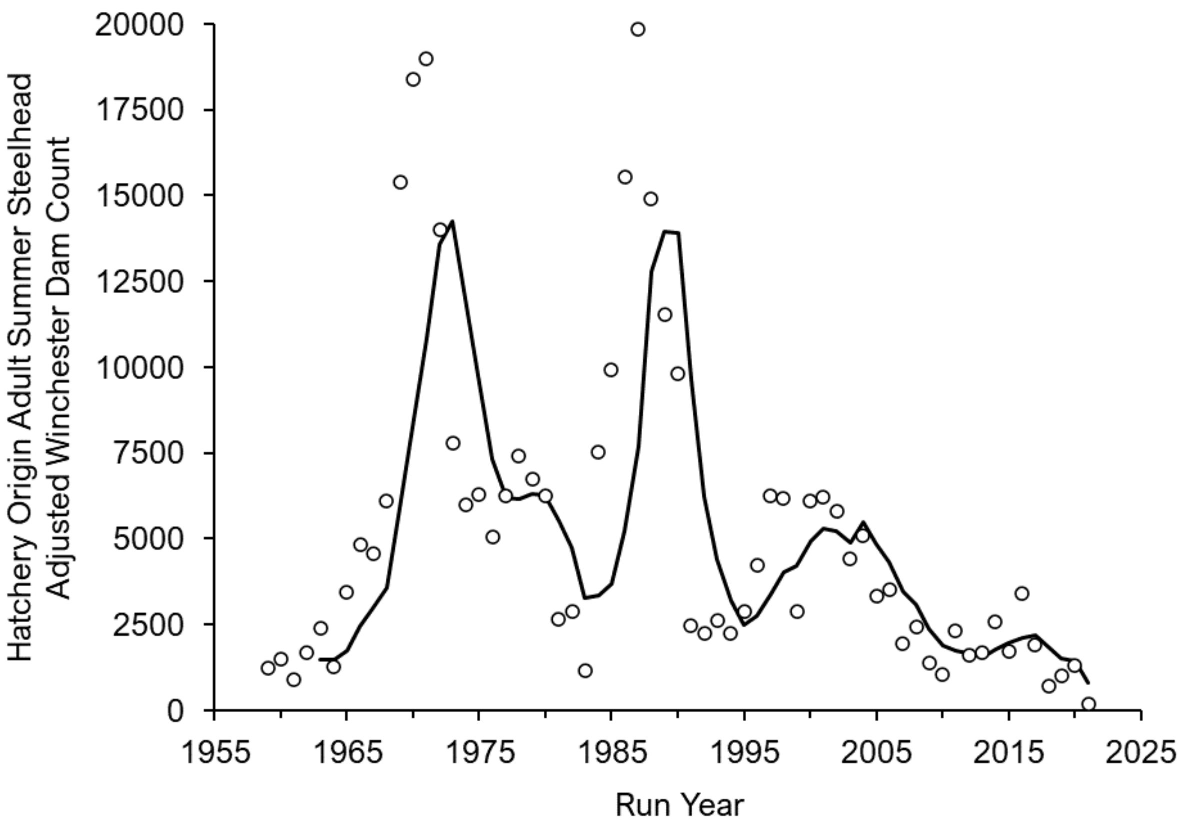


Figure 10. Hatchery summer steelhead passage at Winchester Dam, 1959-2021.
Dam counts were adjusted to account for hatchery summer steelhead passing Winchester Dam during the winter counting period and for a change in counting methods in 1992.

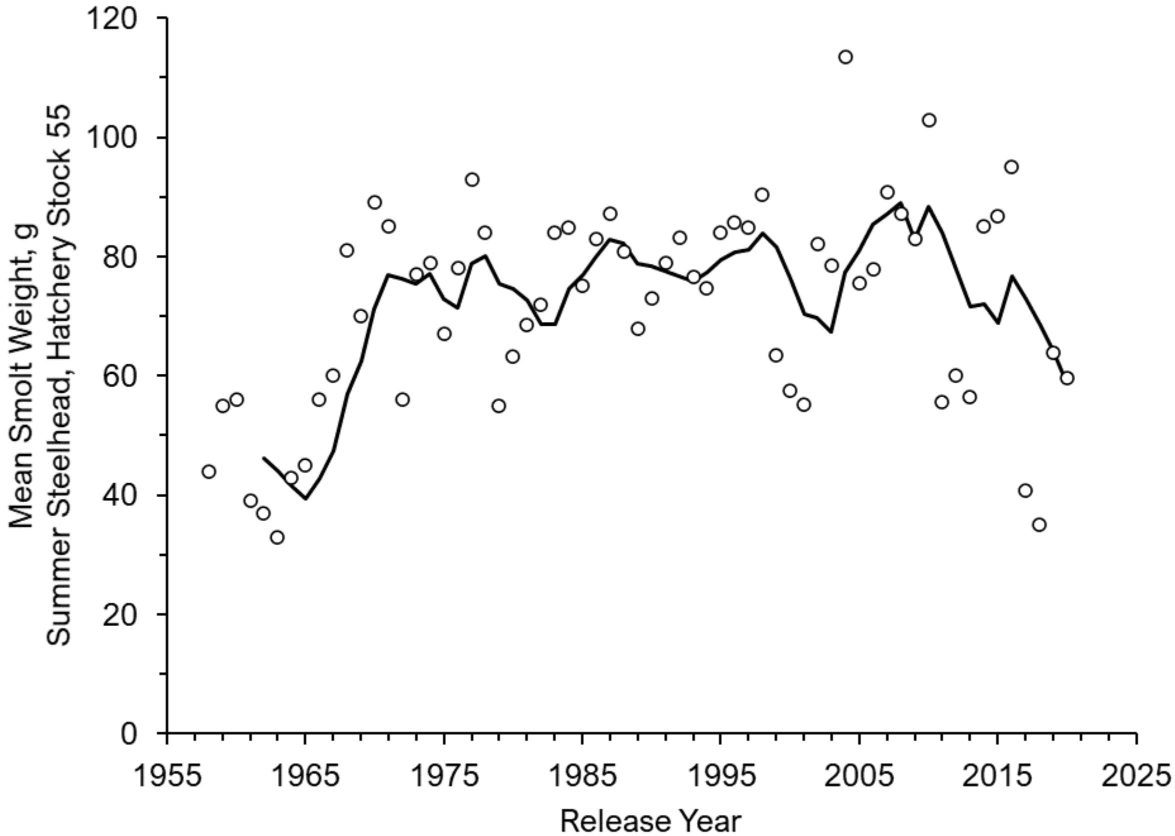


Figure 11. Mean weight of Stock 55 summer steelhead smolts released to the North Umpqua River (1958-2020). The solid line shows the running 5-year moving geometric mean. Data from 1958-1977 and 1983 are from unpublished hatchery records documented in McGie (1994); data from 1978-1982 and 1984-2021 are from the ODFW Hatchery Management Information System. No smolts were released in 2021 due to hatchery losses incurred during the 2020 Archie Creek Fire.

None of the smolt release numbers provided above include hatchery summer steelhead released as fry or fingerlings. However, hatchery records indicate that Stock 55 summer steelhead fry and fingerlings were released into the North Umpqua basin in some years prior to 1988 (McGie 1994). Surplus fry and fingerlings were also released into the Steamboat Creek sub-basin in 1961 (OSGC 1962). Due to their small sizes at release, these fry and fingerlings are not expected to have contributed strongly to subsequent adult returns (McGie 1994). Since 1988 all releases have been smolts, except for a small number of unfed fry (<2,000) released annually as a part of ODFW's Salmon and Trout Enhancement Program school education program (ODFW 2016b).

The release goal for Stock 55 summer steelhead is currently 165,000 smolts at 6 fish per pound (ODFW 2020a), or an average weight of 75.5 grams. Since the initial smolt release in 1958, smolts have averaged 71.2 ± 4.3 (95% CI) grams. Average smolt sizes were substantially smaller during early program implementation and in some recent years, most notably 2017 and 2018 (Figure 11).

Hatchery-Origin Spawners. The CMP established a target of ≤10% for the percent of hatchery origin summer steelhead on the spawning grounds (pHOS) for the North Umpqua summer steelhead population. The CMP provides an exception to this target within a four-mile radius around Rock Creek Hatchery, where the target is <30% or <60% for significant or less significant wild spawning areas, respectively. "Significant wild spawning areas" are determined based on the quality of the habitat within the area and the proportion of all spawning habitat that it represents. The appropriate target for the four-mile radius around Rock Creek Hatchery is 60% because water temperatures are likely to limit spawning and rearing, and because this area represents a low proportion of the spawning habitat used by naturally produced spawners (See discussion of spatial segregation below).

Counts of natural and hatchery origin summer steelhead passing Winchester Dam can provide an estimate of the proportion of the total summer steelhead passage comprised of hatchery origin fish. However, this proportion is not an estimate of pHOS because it does not account for removal of adults that occurs prior to spawning (i.e., fishery-related mortality; retention at hatchery facilities). Even after accounting for these losses, the proportional escapement of hatchery origin steelhead relative to the combined escapement of natural and hatchery origin steelhead reflects pHOS only to the degree that natural and hatchery origin steelhead share a common spawning distribution in both time and space.

Calculating pHOS. The CMP specifies that spatial segregation of spawning will be incorporated into pHOS estimates:

> "*(I)n determining the population-level pHOS estimate, the spawning habitats will be weighted by their wild fish use (i.e., areas with a higher proportion of the wild spawning population will be weighted heavier than areas with low percentages of the wild spawning population; quality and proportion of spawning habitat within an area may also be considered in the weighting). This will allow for higher pHOS levels in areas that are not significantly used by wild fish but must be offset by pHOS estimates that are lower in areas that are more heavily used by the wild population to meet the population-wide/majority spawning area target.*"

Since adoption of the CMP and absent spawning ground surveys in the North Umpqua, ODFW has published basin-wide pHOS estimates that are based on the proportional escapement of hatchery origin summer steelhead relative to the combined escapement of natural and hatchery origin summer steelhead (i.e., counts at Winchester Dam adjusted for losses prior to spawning), while also reporting that these estimates do not account for the likelihood of significant segregation of hatchery origin spawners (ODFW 2020b, 2021a). In this assessment, we have accounted for the segregation of naturally produced and hatchery origin spawners to the extent feasible with the available data.

We are not aware of data indicating a substantial difference in spawning timing by natural and hatchery origin summer steelhead in the North Umpqua basin. Any differences in timing may be minimized by use of broodstock collected over the duration of the summer counting period (ODFW 2016b). Therefore, in the following analysis we assume no temporal segregation of spawners by origin.

Radio telemetry tracking of adult North Umpqua summer steelhead indicated a high degree of segregation of spawning areas favored by natural and hatchery origin summer steelhead. Loomis et al. (2003) captured, radio-tagged, and released 140 naturally produced steelhead in the summer counting period (May 1 through Nov 30) in 1998 - 2000 and 98 hatchery origin steelhead during the summer counting period in 1999 and 2000.[3] Fish were captured at Winchester Dam, released (after a recovery period), tracked several times weekly, and, where possible[4], assigned as spawners to one of five North Umpqua sub-basins: Lower North Umpqua River (Winchester Dam to Deadline Falls, excluding the Rock Creek and Little River sub-basins), Little River, Rock Creek, Upper North Umpqua River (Upstream of Deadline Falls, excluding the Steamboat Creek sub-basin), or Steamboat Creek.

Over the course of the telemetry study, 92 naturally produced and 20 hatchery origin summer steelhead were able to be assigned as spawners to one of the five sub-basins (Tables S3 and S4). Approximately 74% of the naturally produced summer steelhead spawned upstream of Deadline Falls, most (52%) in the Steamboat Creek sub-basin (Table S3). Conversely, 90% of hatchery origin summer steelhead spawned downstream from Deadline Falls, most (80%) in the Rock Creek sub-basin (Table S4). No hatchery origin summer steelhead were assigned as spawners to either the Little River or Steamboat Creek sub-basins (Table S4).

Consistent with the guidance in the CMP, we used the telemetry-inferred distribution proportions to account for segregation of natural and hatchery origin summer steelhead as follows:

(1) Counts of summer steelhead at Winchester Dam (May 1 through November 30) were adjusted to account for harvest[5] and hatchery origin summer steelhead passing the dam during the winter counting period (December 1 through April 30). These adjusted counts represent escapement estimates for natural and hatchery origin summer steelhead

---

[3] Loomis et al. (2003) considered steelhead to be summer steelhead if they passed Winchester Dam from June 1 through December 31 or winter steelhead if they passed the dam from January 1 through May 30. In the present analysis, we used counting periods consistent with those used to discriminate summer and winter steelhead at Winchester Dam (Summer = May 1 - Nov 30; Winter = Dec 1 - Apr 30).

[4] Assignment of radio-tagged adjust summer steelhead as a spawners in a specific sub-basin was not possible where fish were lost to tracking, left they system prior to spawning, or were harvested.

[5] After the harvest of wild summer steelhead was prohibited, a fishery mortality rate for naturally produced summer steelhead was estimated by applying a 10% incidental mortality rate to an encounter rate set equal to the harvest rate of hatchery summer steelhead. Harvest rates of hatchery summer steelhead upstream from Winchester Dam are based on the count of hatchery summer steelhead upstream from Winchester Dam (adjusted for hatchery summer steelhead passing during the winter counting period) and angler reported harvest of hatchery summer steelhead upstream from Winchester Dam.

upstream from Winchester Dam prior to removal of hatchery fish at Rock Creek Hatchery.

(2) Natural and hatchery origin escapement estimates from Step 1 were apportioned to the five sub-basins using telemetry-based distribution proportions (i.e., Tables S3 and S4).
(3) Hatchery fish removed at Rock Creek Hatchery were subtracted from the hatchery origin escapement estimate for the Rock Creek sub-basin. Annual numbers of hatchery summer steelhead retained at Rock Creek Hatchery are based on data from ODFW's Hatchery Management Information System, HMIS.
(4) Sub-basin pHOS estimates were calculated as sub-basin hatchery escapement divided by the combined sub-basin escapement of natural and hatchery origin summer steelhead.
(5) A basin-wide estimate of pHOS was calculated from sub-basin pHOS estimates, weighted by each sub-basin's proportion of naturally produced spawners (i.e., Table S3).

This analysis is similar to those reported in ODFW (2005) and ODFW (2020b), with some exceptions. ODFW (2005) used the Loomis et al. (2003) telemetry data to calculate an annual weighted pHOS for 2001-2005 (5-year average = 17%). However, ODFW (2005) assigned tagged steelhead to run-timing (summer vs. winter-run) based on a summer counting period from June 1 to December 31. In our present analysis, we used proportions based on the Winchester Dam summer counting period (May 1 to November 30). ODFW (2005) also assumed that 34% of hatchery summer steelhead that were not harvested were diverted to Rock Creek hatchery. Our analysis accounted for retention of hatchery fish at Rock Creek hatchery using records in ODFW's Hatchery Management Information System, which indicated lower rates of retention than assumed in ODFW (2005). Re-calculation of the ODFW (2005) assessment period using our methods yields an average weighted pHOS of 21% for 2001-2005.

Our present analysis accounts for removal of hatchery fish at Rock Creek Hatchery after hatchery fish are apportioned among basins. Both previous analyses (ODFW 2005, 2020b) accounted for removal of hatchery fish at Rock Creek Hatchery before apportioning the hatchery-origin escapement to sub-basins. This results in an overestimate of hatchery escapement to Rock Creek and an underestimate of hatchery escapement to the Lower and Upper North Umpqua sub-basins. ODFW (2020b) also used a slightly different apportionment of hatchery fish due to inclusion of one hatchery fish in the Lower North Umpqua sub-basin that was presumed lost to harvest (Loomis et al. 2003).

Assessing pHOS. The CMP pHOS targets are assessed as 9-year moving averages to account for interannual variation (ODFW 2014). The most recent 9-year average of the proportion of hatchery origin summer steelhead at Winchester Dam (unadjusted summer-period counts, 2013-2021) is 41%. After accounting for harvest, retention of hatchery fish at Rock Creek hatchery, and hatchery summer steelhead passing Winchester Dam during the winter counting period, the corresponding basin-wide 9-year average pHOS above Winchester Dam is reduced to 34% (Table 4). This estimate is substantially higher than the basin-wide pHOS target set by the CMP (≤10%), but as previously discussed, it overestimates risk by failing to account for spatial segregation of hatchery and naturally produced fish during spawning.

When apportioned to sub-basins, escapement estimates for natural and hatchery origin summer steelhead illustrate the differences in the spatial distributions of natural and hatchery origin summer steelhead among the North Umpqua sub-basins (Figure 12). These distributions result in a substantial heterogeneity of pHOS within the North Umpqua basin (Table 4; Figure 13). When this heterogeneity is accounted for (i.e., weighting pHOS by the sub-basin proportions of naturally produced escapement), the most recent 9-year basin-wide average of pHOS is 17%. Contributing annual estimates of weighted pHOS over the 9-year period from 2013 through 2021 range from 12 to 21% (Table 3).

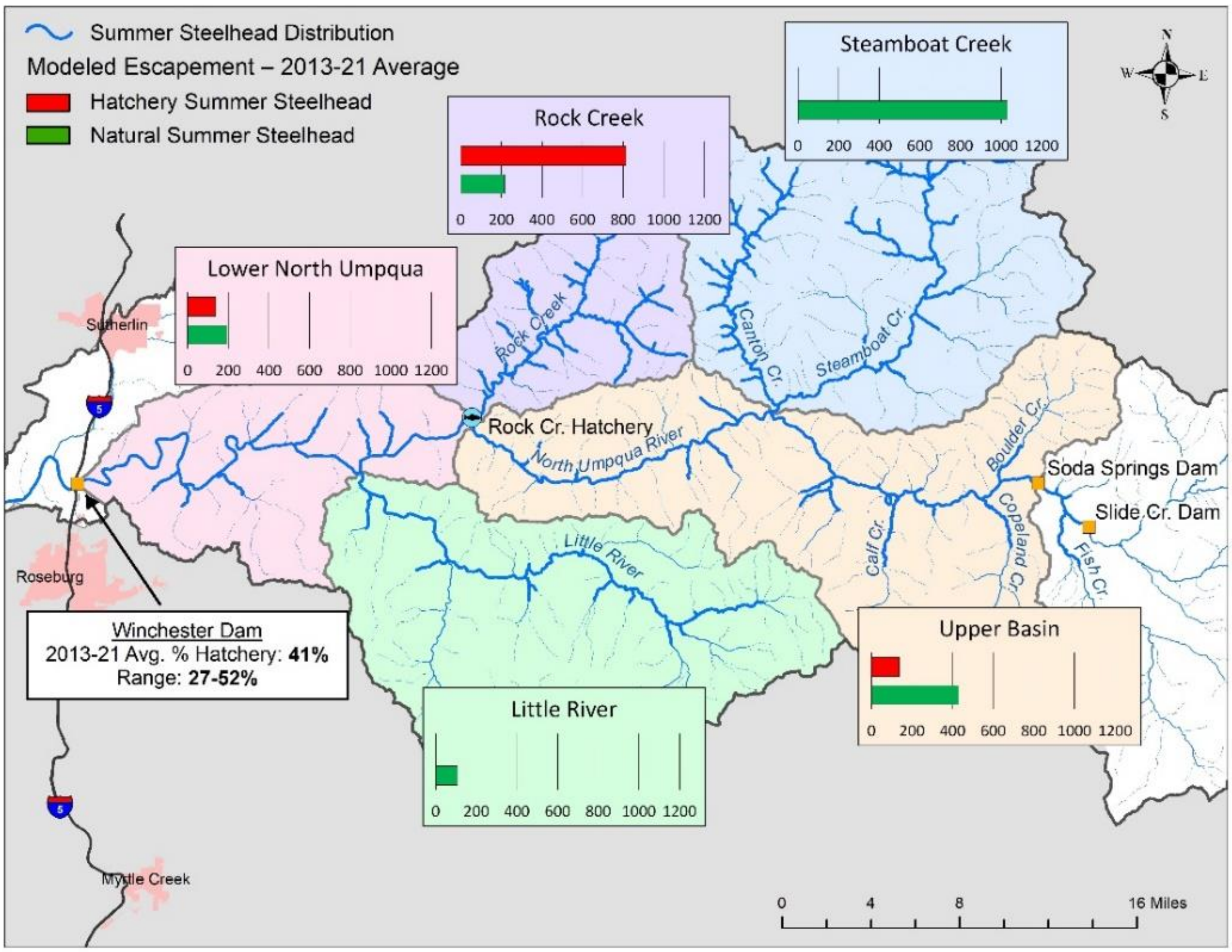


Figure 12. Proportion of natural and hatchery origin steelhead in North Umpqua sub-basins. Proportions during the summer counting period and estimates of escapement by origin for five North Umpqua sub-basins. Proportion of hatchery origin steelhead at Winchester Dam during the summer counting period (May 1 – November 30) and estimates of escapement by origin (natural = green; hatchery = red) for five North Umpqua sub-basins. Sub-basin escapement estimates are basin-wide escapement estimates for natural and hatchery origin summer steelhead, apportioned to sub-basins based on telemetry data reported in Loomis et al. (2003). Estimates are 9-year averages corresponding to the most recent 9-year period used for assessment of pHOS targets (2013-2021).

Table 3. Annual and 9-year average estimates of hatchery proportions in the North Umpqua. Proportions are shown for unadjusted passage counts at Winchester Dam and estimates of unweighted and weighted basin-wide percent hatchery origin spawners (pHOS) for North Umpqua summer steelhead, 2013-2021. Unweighted basin-wide pHOS estimates are based on counts of hatchery and naturally produced summer steelhead counted at Winchester Dam, adjusted to account for harvest, hatchery summer steelhead passing the dam during the winter counting period, and retention of hatchery fish at Winchester Dam. These estimates, as reported in the CMP Wild Fish Monitoring reports (ODFW 2020b) assume no spatial segregation of hatchery and natural-origin spawners. Weighted basin-wide estimates of pHOS are based on the five sub-basin pHOS estimates, weighted based on the proportion of the total naturally produced escapement spawning in each sub-basin. These estimates account for spatial segregation of hatchery and naturally produced spawners based on telemetry data reported in Loomis et al. (2003).

Run Year	Hatchery Proportion (Winchester Dam)	Unweighted Basin-wide pHOS (%)	Weighted Basin-wide pHOS (%)
2013	41	35	17
2014	52	42	21
2015	50	40	20
2016	47	41	20
2017	42	34	18
2018	27	20	12
2019	33	25	15
2020	45	38	19
2021[P]	29	30	15
Average	41	34	17

[P]Data for return-year 2021 are provisional and subject to change.

Table 4. Annual and 9-year average estimates of pHOS for North Umpqua sub-basins (2013-2021).

Estimates are based on counts of hatchery and naturally produced summer steelhead counted at Winchester Dam, adjusted to account for harvest, hatchery summer steelhead passing the dam during the winter counting period, and retention of hatchery fish at Winchester Dam. Basin-wide escapement estimates are apportioned across the five sub-basins based on telemetry data reported in Loomis et al. (2003).

Run Year	Percent Hatchery Origin Spawners				
	Lower N. Umpqua	Rock Creek	Little River	Upper N. Umpqua	Steamboat Creek
2013	39	79	0	23	0
2014	50	83	0	31	0
2015	47	82	0	28	0
2016	45	83	0	27	0
2017	41	78	0	24	0
2018	25	62	0	13	0
2019	32	70	0	18	0
2020	45	81	0	27	0
2021[P]	30	75	0	16	0
Average	39	77	0	23	0

[P]Data for run-year 2021 are provisional and subject to change.

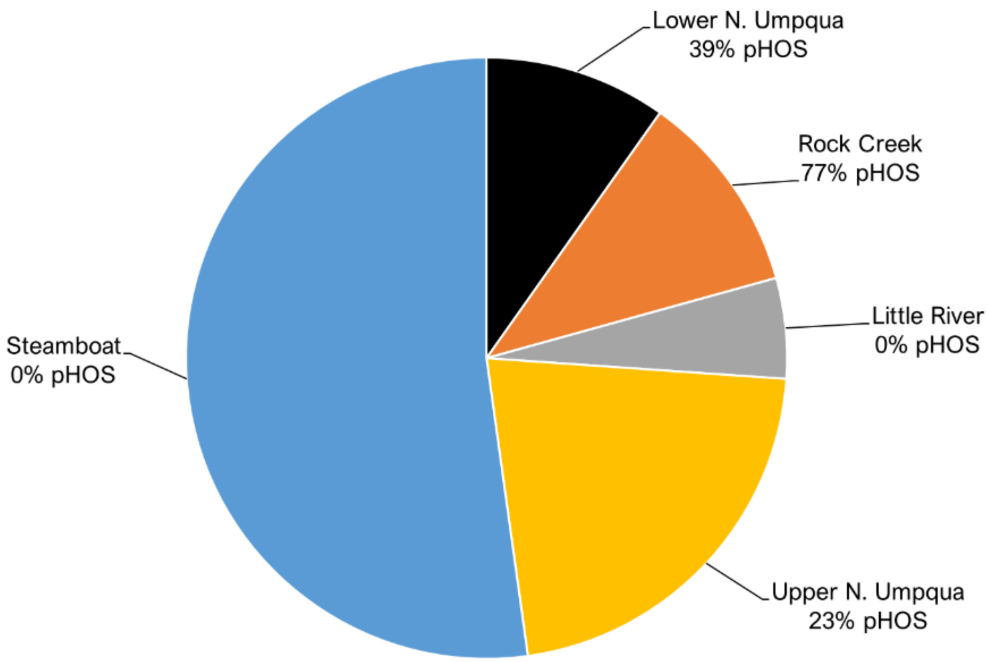


Figure 13. Proportional spawning use of North Umpqua sub-basins by hatchery summer steelhead. Estimates of the percent hatchery origin spawners (pHOS) are the most recent 9-year averages, 2013-2021, based on telemetry-inferred apportionment (Loomis et al. 2003) of basin-wide natural- and hatchery-origin escapement to the five sub-basins.

The lower weighted pHOS estimates relative to unweighted estimates are largely attributable to (1) the high relative weighting of the Steamboat Creek sub-basin, where pHOS is lowest (near zero) but where the proportion of naturally produced escapement is highest (52%) and (2) lower relative weighting for the Rock Creek sub-basin, where pHOS is highest (>75%) but where the proportion of North Umpqua naturally produced escapement is only 11% (Figure 13). Our basin-wide weighted estimates of pHOS include the four-mile radius around Rock Creek Hatchery, where higher pHOS is allowable (<60% in portions of the Lower North Umpqua and Rock Creek sub-basins). Data were insufficient to exclude this area from the calculation or to provide for spatial weighting at a fine enough spatial resolution to account for expected higher pHOS in this area.

The analysis above depends on the assumption that the sub-basin distributions of natural and hatchery steelhead are stationary through time. When segregated by run year, the telemetry data indicates some interannual variability, but the general pattern of naturally produced summer steelhead favoring the upper basin and hatchery origin fish favoring the lower basin was consistent through the study (Figure S11). The most interannual variability is associated with the proportional use of the Upper North Umpqua sub-basin (Above Deadline Falls, excluding the Steamboat Creek sub-basin) and Lower North Umpqua sub-basin (excluding the Rock Creek and Little River sub-basins). Understanding hatchery fish use of the Upper and Lower North Umpqua sub-basins (outside of the Rock Creek sub-basin) is complicated by the low sample sizes of tagged hatchery fish spawning in these sub-basins (n = 2 in each sub-basin during one year of the study; Figure S11). However, several additional lines of reasoning, described below, suggest that the telemetry data generally reflects recent conditions.

The importance of the Steamboat Creek sub-basin to the productivity of the North Umpqua summer steelhead population was recognized over 80 years ago, when Steamboat Creek and its tributaries were closed to all angling to protect summer steelhead (OSGC 1936).[6] Prior to the start of the Stock 55 summer steelhead hatchery program, the 1951 Annual Report of the Oregon State Game Commission, Fishery Division noted, "(s)ummer steelhead of the Umpqua system are peculiar in that practically the entire run enters the Steamboat Creek drainage to spawn" (OSGC 1952). The significance of this sub-basin to naturally produced summer steelhead has since been cited by numerous authors (e.g., OSGC 1959; Collins 1971; Anderson et al. 1986). The telemetry-based estimate that 52% of naturally produced summer steelhead spawn in the Steamboat Creek sub-basin may be an underestimate or it may reflect a broadening of the summer steelhead spawning distribution following naturalization of hatchery origin spawners in areas outside of the historical primary spawning distribution.

The general lack of hatchery-origin spawners in the Steamboat Creek sub-basin is also supported by observations during surveys of adult summer steelhead holding in pools during summer. In 1962, adult steelhead observed holding below Steamboat Falls were exclusively naturally produced fish (OSGC 1963) despite an unweighted basin-wide hatchery proportion of 46%.[7] More recent peak counts of holding adults in the Steamboat Creek sub-basin are significantly correlated with escapement of naturally produced summer steelhead above Winchester Dam ($r^2 = 0.67$, $P < 0.001$; Figure S12), but not with escapement of hatchery origin summer steelhead ($r^2 = 0.06$, $P = 0.183$; Figure S13). Observations of marked hatchery origin steelhead have been rare on these surveys (8 of 24 years), with peak counts of marked fish ranging from zero to <0.3% of the observed peak of holding adults between 1998 and 2021 (when observation notes on fin-clip status were available).

The use of the Rock Creek sub-basin as the primary spawning destination by hatchery-origin summer steelhead is plausible given that most hatchery smolts are volitionally released into Rock Creek from the hatchery (ODFW 2016b), which is situated low in the Rock Creek sub-basin. Telemetry-based estimates of sub-basin pHOS (Table 3) are similar to proportions of hatchery origin fish among adult steelhead observed during snorkel surveys of summer holding pools (Table S5). These surveys indicate that the proportion of hatchery origin fish is higher for

[6] 1932 is often cited as the year in which angling was prohibited in the Steamboat Creek sub-basin, but a review of Oregon angling regulations indicates the closure began in 1936.
[7] The basin-wide unweighted hatchery proportion is calculated as the escapement of hatchery origin summer steelhead divided by the total escapement of natural and hatchery origin steelhead. Escapement estimates are based on Winchester Dam counts, adjusted to account for harvest, retention at hatchery facilities, hatchery summer steelhead passing the dam during the winter counting period, and a change in counting methods prior to 1992. This calculation assumes no temporal or spatial segregation of natural and hatchery origin fish.

fish holding below the Rock Creek diversion dam than for fish holding above the dam (Table S5), an observation that generally aligns with data presented in Loomis et al. (2003).

There is no independent means by which to corroborate telemetry-based pHOS estimates in the other North Umpqua sub-basins. Summer steelhead passing Soda Springs Dam (Upper North Umpqua sub-basin) have been counted by video since 2013. However, the hatchery proportion of summer steelhead observed at the dam is not an estimate of the proportion of hatchery fish on spawning grounds; the counts are not escapement estimates, and they do not account for exploratory movements of steelhead prior to selection of spawning locations (e.g., fallbacks are observed at the counting station). Since 2013, the hatchery proportion at Soda Springs Dam averaged 35%, while the 9-year average of telemetry-based estimates of pHOS above Deadline Falls (excluding the Steamboat Creek sub-basin) was 23%.

Our analysis, which considers spatial segregation to the extent currently possible, indicates that the risk of interactions between hatchery and naturally produced summer steelhead on the spawning grounds is lower than indicated by the proportion of hatchery origin summer steelhead in passage counts at Winchester Dam and the pHOS surrogate reported previously derived from these data. However, the most recent basin-wide pHOS estimates, weighted to account for distributions of naturally produced spawners (17%, 9-year average; Table 3), are higher than the CMP target (≤10%). Estimates of pHOS tend to be lower in the upper basin and higher in the lower basin, reflecting the differential preferences of natural and hatchery origin spawners.

We expect pHOS to be near zero in the Steamboat Creek sub-basin, the preferred spawning sub-basin for more than half of naturally produced spawners; this conclusion is supported by observations of holding adults during summer snorkel surveys. The pHOS in the Little River sub-basin is also expected to be near zero, though this sub-basin is expected to be favored by fewer than 6% of naturally produced summer steelhead spawners. There is greater uncertainty about the proportions of natural and hatchery origin summer steelhead spawning in the lower and upper North Umpqua sub-basins, but higher pHOS in the Rock Creek sub-basin is supported by surveys of adult summer steelhead holding in Rock Creek pools. Data are insufficient to assess the hatchery hotspot target (<60%) in the four-mile radius around Rock Creek Hatchery (Rock Creek and Lower North Umpqua sub-basins).

> *Select pHOS Management Options*. Given that the most recent 9-year weighted, basin-wide average of pHOS is above the threshold established in the CMP, we analyzed several management options for reducing pHOS to a level below the threshold. The options considered are ones that could be readily analyzed, but are not the only options available to reduce pHOS.
>
> To estimate a smolt release consistent with achieving a basin-wide weighted pHOS of ≤ 10%, we related the proportion of hatchery steelhead in the summer period count at Winchester Dam to contributing smolt releases, assuming returns as 1-salt (10%), 2-salt

(66.5%), 3-salt (20%) and 4-salt (3.5%) adults. We used a power function to fit the relationship between contributing smolt releases and subsequent hatchery proportions in summer-period steelhead counts at Winchester Dam (Figure 14). The power function provided a more realistic intercept relative to linear or logarithmic alternatives. A 10% weighted pHOS corresponds to a contributing release of approximately 30,000 smolts (Figure 14). This is approximately 40% of the average annual release since approval of the CMP.

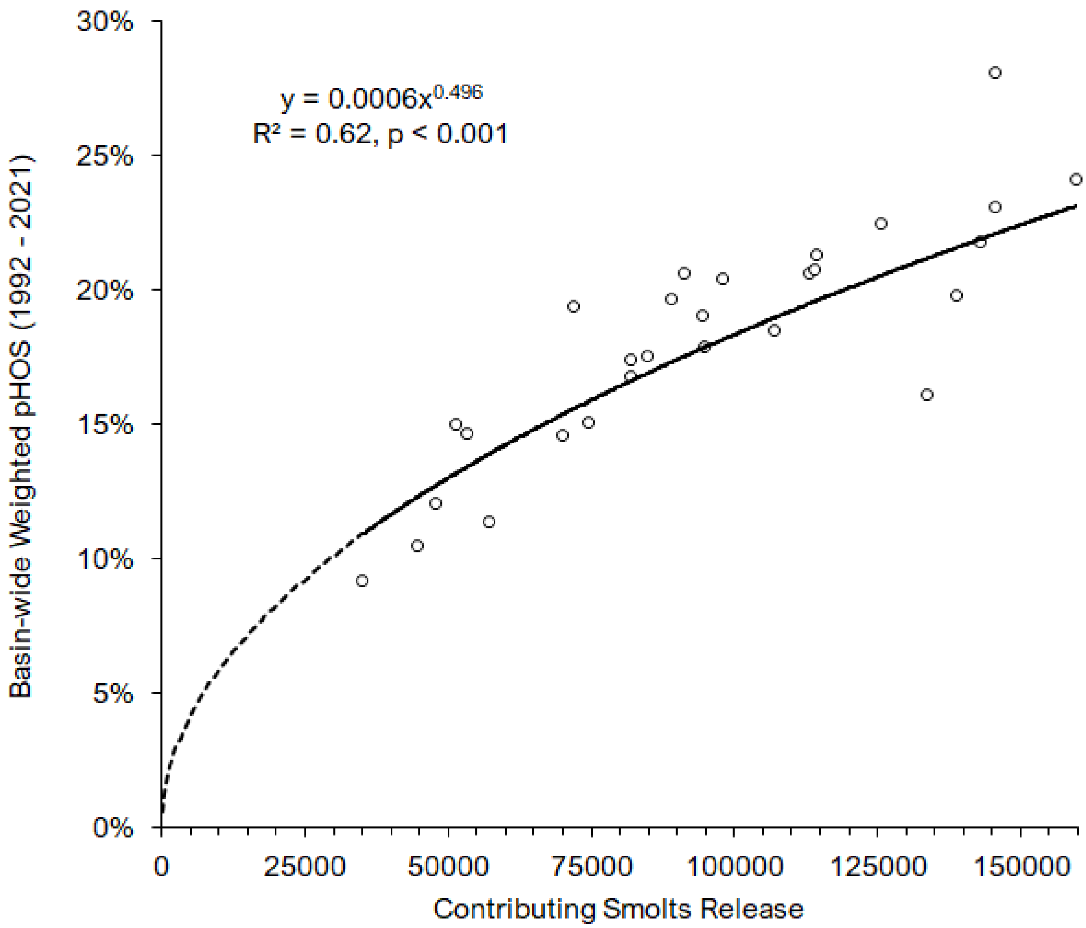


Figure 14. Relationship between smolt release numbers and percent hatchery origin summer steelhead. Relationship between contributing smolt release numbers and basin-wide weighted estimates of the percent hatchery origin summer steelhead, return years 1992-2021. Contributing smolt releases were smolt releases one to four years prior to each run year, weighted by a static ocean age structure for returning adults (1-salt (10%), 2-salt (66.5%), 3-salt (20%) and 4-salt (3.5%). The solid line indicates the trend through the range of the available data; the dashed line indicates the predicted trend below the range of the available data (subject to greater uncertainty).

In a separate analysis using the past nine years of annual sub-basin pHOS estimates, we also simulated the removal of hatchery fish in Rock Creek to achieve a sub-basin pHOS of 10%. Reducing pHOS to 10% in the Rock Creek sub-basin would have reduced the basin-wide weighted pHOS estimate to approximately 10%. Efficacy of achieving significant pHOS reductions through removal of hatchery origin steelhead in Rock Creek is dependent on the feasibility of removing additional hatchery fish at the hatchery and diversion dam. The scenario simulated here was set to an arbitrary reduction intended to reflect a significant but not complete removal of hatchery fish. This is a hypothetical scenario, and we also did not assess the feasibility of achieving this magnitude of removal. Reduced smolt releases and increased removal of hatchery fish in the Rock Creek basin are independent potential management interventions that were not evaluated in concert.

Additional Hatchery Impact Analyses. As discussed in the CMP, the level of naturally spawning hatchery fish is currently the most feasible way of assessing hatchery risk, but it may not capture the magnitude of all potential risks that hatchery fish might have on naturally produced populations (e.g., competition, predation, disease). Additionally, estimates of pHOS capture potential for interbreeding between hatchery and naturally produced fish, but they are not a direct measure of interbreeding or impacts to reproductive success. Although our analyses accounts for spatial segregation to the extent possible, the available data do not allow for evaluation of pHOS at any finer spatial resolution. The analyses above also depend on continued stationarity of spawning distributions of hatchery and wild fish despite recent changes to habitat conditions (e.g., wildfires). Given the limitations of the pHOS metric and the data available for the current pHOS analysis, we have further assessed hatchery impacts by evaluating (1) the influence of hatchery smolt releases on recruitment of naturally produced summer steelhead, and (2) population viability under various levels of both smolt releases and relative reproductive success of hatchery origin spawners.

Influence on Natural Recruitment. North Umpqua summer steelhead spawner and recruit abundance were both estimated based on counts at Winchester Dam from 1992-2020. Annual age compositions used to assign naturally produced returners to spawning year and to generate age compositions for population viability analysis (PVA) were derived from scale data acquired in the Volunteer Steelhead Scale Project from 1983-1993 (ODFW unpublished data). Data collection and processing details are given in the **Population Viability** section of this report. The same data sets are used here.

Hatchery release data were taken from ODFW’s Hatchery Management Information System (HMIS). Where possible this information was cross-referenced and reconciled with other data sources including McGie (1994) and the Umpqua River summer steelhead Hatchery Genetic Management Program (HGMP) (ODFW 2016b).

We tested three hypotheses to determine whether hatchery smolt releases could affect abundance and productivity of naturally produced steelhead: 1) hatchery smolts could impact naturally produced smolts during the year of their concurrent outmigration, 2) hatchery smolts could affect naturally produced spawners during the year that the bulk of the hatchery released smolts would return as adults (~age 4), or 3) both mechanisms. If hatchery releases are impacting outgoing naturally produced smolts, we would expect those effects to align with the year in which both hatchery smolts and naturally produced smolts are out-migrating. For example, hatchery smolts released in 1995 would align with naturally produced recruits from spawning year 1993 since the bulk of those naturally produced smolts would out-migrate as age-2 fish in 1995. Figure S14 shows there is no relationship between hatchery releases and naturally produced recruits (aligned for the years in which both would have out-migrated together as smolts), suggesting that hatchery smolt releases are not systematically impacting co-occurring naturally produced smolts.

If hatchery releases are impacting naturally produced steelhead when they return as spawners, we would expect those effects to align with the spawning year in which most hatchery smolts (2 saltwater years) would be returning. For example, hatchery origin smolts released in 1990 and returning as 2-salt spawners during the 1992 run year would impact naturally produced recruits from the1993 spawning year. One potential mechanism for this would be competition with, or displacement of, naturally produced spawners by hatchery origin spawners from a given smolt release class during their concurrent return year. Figure 15 shows the relationship for this alignment. The relationship was marginally significant (*P* = 0.087) and had low explanatory power (R^2 = 0.14).

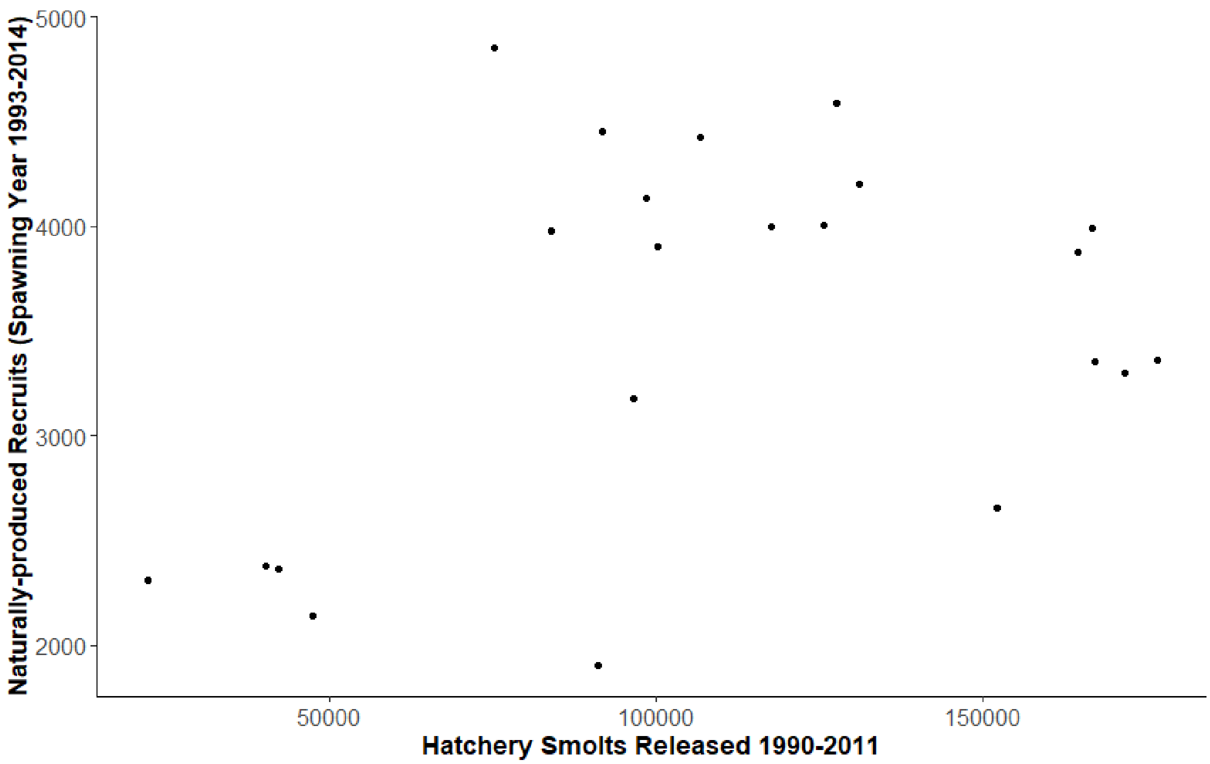


Figure 15. Relationship between hatchery smolts released and naturally produced recruits.
The comparison was made for the spawning year when most hatchery smolts returned as spawners.

The trend in Figure 15 toward higher naturally produced recruits at higher hatchery smolt releases may seem counter-intuitive, but it is important to consider that naturally produced recruits include offspring of hatchery fish that stray to the spawning grounds. These fish could increase the overall number of naturally produced recruits even if they have lower relative reproductive success, particularly if they increase the spatial distribution of summer steelhead spawning and rearing. These topics are explored in greater detail in the next section.

Population Viability Under Different Hatchery Scenarios. Multiple studies have demonstrated that hatchery steelhead have lower relative reproductive success (*RRS*) than their naturally produced counterparts (Berntson et al. 2011, Araki et al. 2007). *RRS* refers to hatchery spawners producing fish that survive and recruit to spawn themselves at lower rates than naturally produced fish. This can have implications for population persistence especially if *RRS* is relatively low and proportion of hatchery origin spawners (pHOS) is relatively high.

It is sometimes possible to estimate the *RRS* of hatchery fish (*RRSh*) empirically by separating naturally produced and hatchery fish and multiplying hatchery fish by a term ($\boldsymbol{\psi}$) representing *RRSh* (Falcy and Suring 2018). *RRSh* must be bounded between zero and one. Adding *RRSh* into the Beverton-Holt model derived in the **Population Viability** section yields:

$$Beverton - Holt: R = \frac{\boldsymbol{\alpha}(NO + \psi H)}{1 + \boldsymbol{\beta}(NO + \psi H)} e^{\boldsymbol{\epsilon}}$$

Attempts to derive *RRSh* empirically for the North Umpqua summer steelhead population were unsuccessful. We attempted multiple methods to constrain the *RRSh* parameter to the range [0,1]. However, these models often failed to converge on a finite solution. Moreover, when unconstrained models did converge to biologically plausible parameter estimates, they showed extreme sensitivity to initial parameter values making estimated parameter values very unstable and highly dependent on initial values. Therefore, we were unable to empirically derive *RRSh* for this population from available stock-recruitment data. Studies conducted on Oregon steelhead populations cited above measured a range of *RRSh* between 0.3-0.6. Given uncertainty about *RRSh*, we used a PVA model to evaluate how variation in *RRSh* could affect viability results for North Umpqua summer steelhead. In simulations, we also varied hatchery smolt release size to see how the two factors interacted to influence viability outcomes.

The PVA used the stock-recruitment relationship developed in the **Population Viability** section to simulate population abundances over a 100-year timeframe. We incorporated randomness into this iteration of the model by randomly varying age composition, NPGO, and harvest rates above and below Winchester Dam for naturally produced and hatchery fish, independently. We also introduced a random error term to capture variability within the parameters of the model. As discussed above, without some degree of randomness in parameters and input variables, eventually all models would converge and extinction risk would be functionally zero.

Extinction occurs in the PVA model when spawner abundance drops below a pre-determined quasi-extinction threshold (QET) for three consecutive years. In a separate process, at critically low spawner abundances recruitment is set to zero simulating the reproductive failure threshold (RFT). QET/RFT were defined in the CMP as 250 for "large" populations such as North Umpqua summer steelhead (ODFW 2014).

The PVA model assumes that conditions and the relationship between parameters and data, other than the randomly varying parameters and data (e.g., age composition or NPGO), will remain constant for at least the duration of the 100-year simulation. The model is not intended to capture the effects of climate change, ecosystem disruption or habitat alterations.

This PVA ran 1,000 repetitions of a 100-year simulation using the Beverton-Holt model with NPGO as a covariate at each combination of a range of possible *RRSh* values from zero to one by 0.1 increments and smolt release abundance values from zero to 165,000 (current target) by 1,000 fish increments. For each repetition, age composition was randomly selected from the range of age compositions observed during sampling years. NPGO was also randomly varied within its observed range. Harvest rates above and below Winchester Dam for naturally produced and hatchery fish were each modeled independently (i.e., 4 separate values) assuming a beta distribution ($\boldsymbol{\beta}[0,1]$). Estimating a theoretical distribution based on observed values allowed values to be more extreme than just those sampled while still falling within the range of plausible outcomes. Hatchery broodstock removal was modeled similarly. Random error was also incorporated into the simulations. We then calculated the number of simulations that resulted in local "extinction" (i.e., population abundance below QET/RFT = 250).

Results indicate that, regardless of variations in *RRSh* and smolt release abundance, no modeled simulation (i.e. - 0 out of ~181.5 million; 0%) showed the North Umpqua summer steelhead population dropping below QET/RFT. A high percentage of simulations (>90%) remained below the desired abundance threshold (DAT) regardless of *RRSh* or smolt release abundance. This is a result of the stock-recruitment curve having an *Neq* below the DAT (see **Population Viability** section). The population will tend towards equilibrium at a level below the DAT, only rising above on occasions when environmental conditions are favorable. Simulations with lower *RRSh* and smaller smolt releases tended to have lower abundance of naturally produced steelhead, consistent with the trend in Figure 15. This could be considered a worst-case scenario because the PVA model does not include compensatory mechanisms that could improve the productivity of naturally produced fish at lower hatchery fish abundance.

Our analysis could be improved if additional data on the proportion of naturally produced recruits produced by naturally produced spawners or hatchery spawners (i.e., hatchery fish that spawn in the wild with naturally produced fish or other hatchery fish) were available for this population. Even at annual smolt releases of zero, the stock-recruitment relationship used here was still derived assuming an annual addition of hatchery spawners. We expect that the stock-recruit relationship might change in the absence of annual infusions of hatchery spawners but at present have no ability to parse out recruits into progeny of naturally produced or hatchery spawning and develop separate stock-recruit curves.

Within the range of *RRSh* provided in the literature and historic annual smolt release abundance, the Umpqua summer Steelhead population appears unlikely to decline below "quasi-extinction" thresholds. However, it is also unlikely to meet or exceed desired abundance thresholds on a regular basis. This assumes ocean conditions, freshwater habitat conditions, climate and human influence remain largely stable in the immediate future. Any change in these factors could change the underlying population dynamics resulting in a change in the range or frequency of possible outcomes.

Harvest and Fishery Mortality

North Umpqua summer steelhead are not harvested in significant numbers in ocean fisheries (PFMC 2021). Historically, they were intercepted to varying degrees in commercial net fisheries for shad in the lower Umpqua River (FCO & OSGC 1946). Recreational fishery regulations have been adjusted over time to increase protection for naturally produced summer steelhead in the mainstem Umpqua and North Umpqua rivers, including:

- 1936 – Steamboat Creek and its tributaries closed to all angling (OSGC 1936)
- 1952 – North Umpqua above Winchester Dam closed to all angling from a floating device; Fly Water section established from Rock Creek deadline to Soda Springs (No fishing from a floating device and fly only) (OSGC 1952)
- 1990 – Harvest of unmarked summer steelhead prohibited in Fly Water section of North Umpqua River (July 1 – October 31) (ODFW 1990)
- 1999 – Harvest of unmarked steelhead prohibited Fly Water section of North Umpqua (May 1 – December 31) and in the Mainstem Umpqua River, including tidewater, from mouth upstream to confluence with the North and South Forks (ODFW 1999).

Harvest of wild summer steelhead is currently prohibited throughout the Umpqua basin. However, mortality incidental to catch-and-release angling has been incorporated into the population abundance dataset used for the population viability analysis, presented in the **Population Viability** section. That PVA model indicated the North Umpqua summer steelhead population is not at any immediate risk of extinction, supporting the status determination in the CMP.

Mortality of North Umpqua summer steelhead incidental to catch-and-release angling is calculated as follows. Harvest rates of hatchery summer steelhead upstream from Winchester Dam are based on the count of hatchery summer steelhead upstream from Winchester Dam, adjusted to account for hatchery summer steelhead passing Winchester Dam during the winter counting period, and angler reported harvest of hatchery summer steelhead. Reported harvest is adjusted by a factor of 0.625 to account for upward bias in expanded harvest card estimates. With the assumptions that (1) all hatchery summer steelhead caught are subsequently retained (i.e., harvest rate = encounter rate) and (2) wild summer steelhead are encountered in the fishery at the same rate as hatchery summer steelhead, the harvest rate of hatchery summer steelhead is applied as an encounter rate for wild summer steelhead. A 10% incidental pre-spawning mortality rate is then applied to the estimate of wild summer steelhead encountered by anglers upstream from Winchester Dam so that a catch-and-release fishery mortality rate for naturally produced summer steelhead is calculated as:

$$M_N = HR_H \times StS_N \times 0.1,$$

where

M_N = Fishery Mortality Rate for naturally produced Summer Steelhead
HR_H = Harvest Rate of Hatchery Summer Steelhead
StS_N = Count of naturally produced Summer Steelhead at Winchester Dam, and
0.1 = 10% mortality rate incidental to catch-and-release angling.

Since implementation of catch-and-release regulations for naturally produced summer steelhead, we estimated an average fishery-related mortality of 2% for naturally produced summer steelhead above Winchester Dam and 1% in the mainstem Umpqua River and the North Umpqua River below Winchester Dam. This is similar to an estimate of the catch-and-release fishery mortality for wild summer steelhead entering Idaho (5%, Marshal 2001). These calculations are sensitive to both the assumed incidental mortality rate and the assumed encounter rate.

Twardek et al. (2018) evaluated both mortality and sub-lethal physiological effects of catch-and-release angling on summer steelhead in the Bulkey River, British Columbia. Harvest of naturally produced fish in this population was prohibited in 1991, so catch-and-release regulations have been in effect for approximately the same timeframe that harvest of unmarked North Umpqua summer steelhead was significantly curtailed. After being caught, steelhead were fitted with radio tags, and apparent survival was estimated based on telemetry-inferred patterns of movement. Incidental mortality prior to spawning was estimated at 15% (4.5% within 3 days + 10.5% over winter). We note that these mortality rates are the result of all sources contributing of mortality, including the effects of angling; they are not mortality rates solely attributable to angling. Others have estimated catch-and-release mortality rates for steelhead closer to 5% (e.g., Hooton 1987; Nelson et al. 2005), but these were winter-run populations, which may be subject to different angling conditions. We assumed a 10%

incidental mortality rate to be conservative. Adjusting this further to 15% would raise our fishery mortality estimates by approximately 1%, on average.

The assumed encounter rate can have a significant influence on estimates of fishery mortality (Figure S15), and encounter rates can vary substantially depending on site-specific aspects of the fishery (e.g., access, conditions, etc.). For example, McCormick et al. (2020) applied a 50% encounter rate for modeling catch-and-release angling in Fish Creek, Idaho, though noting it was likely higher than actual encounter rates. A 50% encounter rate and a 10% pre-spawn mortality rate would increase our estimate of fishing mortality to 5%; a 100% encounter rate would increase the estimate to 10%. We are not aware of data to determine a catch-and-release encounter rate for naturally produced summer steelhead in the North Umpqua River. Our current basis on hatchery fishery data provides a site-specific anchor to the North Umpqua River, but additional efforts to quantify encounter rates could help to improve modeled assessments of fishery impacts and population viability.

It is notable that Twardek et al. (2018) estimated mortality rates under conservative restrictions on terminal gear (single, barbless hook, artificial lures only). These restrictions are intended to reduce handling stress and the incidence of deep hooking, which can substantially increase near-term mortality following release. These are comparable to terminal gear restrictions in the North Umpqua fly water section (Deadline Falls to Soda Springs) (single barbless artificial fly, January 1 – June 30 and October 1 – December 31; single, barbless, unweighted, artificial fly, July 1 – September 30). These regulations, and other tools, including angling closures during low returns or poor conditions (e.g., high water temperatures) will continue to be important for minimizing angling-related mortality for North Umpqua summer steelhead.

Other Species

Limiting factors can result from direct or indirect interactions between the species under consideration and other species. Examples of these interactions are predation, competition, prey/food source changes or limitations, disease, and others. A number of these interactions were considered.

Non-Native Fish Predation. Smallmouth bass were likely introduced to the Umpqua River in the early 1970s. They were first documented in the South Umpqua River near Roseburg in 1977; by 1985 a popular fishery had developed throughout most of the Umpqua and South Umpqua rivers (Daily 1990). Creel surveys and other observations suggested that their numbers have increased over time (particularly in the 1990s) and they have been documented in parts of the North Umpqua River (ODFW 2009, ODFW 2010). They have not likely become established in summer steelhead spawning strongholds high in the basin.

There is a robust amount of literature on smallmouth bass predation on juvenile salmonids; substantial losses have been demonstrated in the Columbia, Snake, and Yakima rivers where resources were available to conduct large-scale, multi-year studies. In the Columbia River, Tabor et al. (1993) estimated juvenile salmonids made up 59% of the diet of smallmouth bass by weight, and 65% of bass sampled during May and June had consumed salmonids. Erhardt et al. (2018) documented similar results in the Snake River and concluded smallmouth bass were significant predators there, and predation had increased 15-fold from the mid-1990s, likely explaining mortality of subyearling Chinook salmon that was previously unaccounted for.

Other studies have indicated less-substantial impacts. In the Yakima River, Washington, Fritts and Pearsons (2004) estimated 200,405 juvenile salmonids were consumed annually from 1998 to 2001 by smallmouth bass; the majority were wild subyearling juvenile Chinook salmon *O. tshawytscha*. However, total predation never exceeded 0.6% of the production of wild and hatchery salmonids combined. Rieman et al. (1991) concluded that, of the 2.7 million juvenile salmonids consumed by predatory fishes in John Day Reservoir (Columbia River basin), only 9% could be attributed to smallmouth bass. Two studies (Summers and Daily 2001; ODFW 2007) concluded predation by smallmouth bass was negligible in the Willamette River.

The distribution of juvenile summer steelhead and extent of habitat overlap with smallmouth bass is not precisely known for the Umpqua River. Juvenile steelhead likely rear for 1-3 years in or near their natal habitat, which is presumed to be less suitable for smallmouth bass. However, at minimum, steelhead smolts must traverse bass habitat as they outmigrate to the ocean, so encounters then are highly likely.

To adequately determine whether smallmouth bass predation on juvenile North Umpqua summer steelhead has increased we would need adequate data on bass population dynamics, prey abundance, and diet samples. Unfortunately, these data do not exist apart from anecdotal reports that suggest larger populations or increased feeding activity. We can, however, assess whether conditions that would favor predation by bass have increased over time. Bass activity is closely tied to stream temperatures. In the Umpqua River, smallmouth bass are thought to feed most actively as stream temperatures rise above 15.5°C and are most active over 18.0°C (Daily 1990). Juvenile summer steelhead migrate out of the North Umpqua from January through July (ODFW 2005), likely peaking March 1 – June 30 (Daily 1990). We plotted historic stream temperature from the Elkton gage for 1986 to 1992 to compare to smallmouth bass feeding thresholds. During the early part of the outmigration period, stream temperatures were too cold for bass to be active; however, there was a period of roughly three months when temperatures were high enough for bass to be actively feeding (Figure 16). Daily (2009) also concluded Umpqua River smallmouth bass were “not actively feeding during the early part of the juvenile salmonid migration.”

Temperature monitoring at the Elkton stream gage ended in 1992. In recent years (since 2007), temperature has been monitored at the Idleyld gage and the Winchester gage (since 2017) which are situated above bass distribution. To assess more recent stream temperature conditions in the lower Umpqua River, we combined the NorWest stream temperature model for June with annual PRISM air temperature data. The NorWest model provides an estimate of average June stream temperature from 1993 to 2012. We extracted the NorWest estimate for a

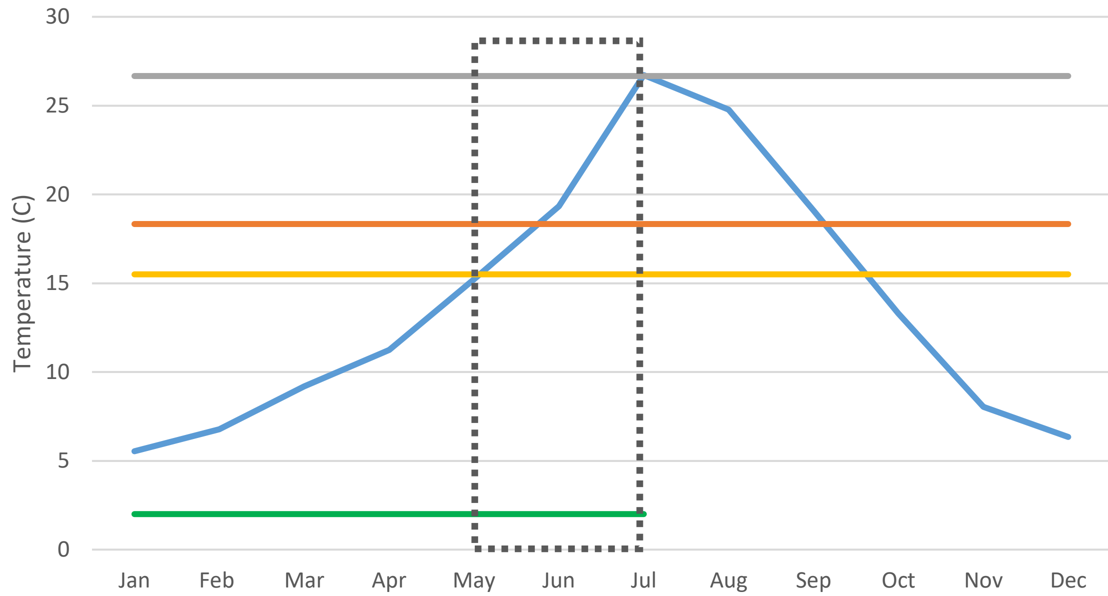


Figure 16. Average stream temperature by month (1986-1992) on the Umpqua River near Elkton. Average stream temperature by month (1986-1992) at the stream gage on the Umpqua River near Elkton (blue line). Juvenile summer steelhead migrate between January and July (green line). Bass begin actively feeding at 15.5°C (yellow line) and are most active between 18 and 27°C (orange and grey lines). The box (dashed lines) highlights the period when summer steelhead are out-migrating and temperatures in the migration corridor are warm enough for bass to be feeding.

stream reach (Common Identifier 24526610), which lies just below the confluence of the mainstem Umpqua River and the Smith River. The results (an index of mean temperature) suggested June stream temperatures in that reach are nearly always above 15.5° C in June and have increased through time (Figure 17), implying late-migrating naturally produced juvenile summer steelhead are at increased risk of predation by smallmouth bass. For comparison, in the Siletz River the June temperature index never reached the lower threshold for bass activity and the June temperature index is similar in the Rogue River to that in the Umpqua (Figure 17; note that, to our knowledge, smallmouth bass have not been introduced to the Rogue or Siletz basins).

Anecdotal angling reports received by Umpqua District staff in 2021 suggested a rebound of the striped bass population. Striped bass were introduced to San Francisco Bay in 1879 and migrated to Oregon by about 1914 (Parks 1978; Wydoski and Whitney 2003). Historic records of substantial commercial harvest of striped bass in the Coos, Coquille, Siuslaw, Smith and Umpqua river basins were maintained until 1976 when commercial harvest was prohibited. Annual catches topped out at about 75,000 fish, with the majority of the landings occurring in the Umpqua basin in the last several decades of the fishery (Parks 1978). Striped bass subsequently supported sport fisheries that declined steadily through the mid-2000s in most areas. However, an active and apparently growing fishery is present in the Coquille River basin. Striped bass were investigated informally as a potential factor in the decline of Coquille fall Chinook salmon during 2018-19, and some removal efforts were implemented there in 2021 (ODFW, unpublished data).

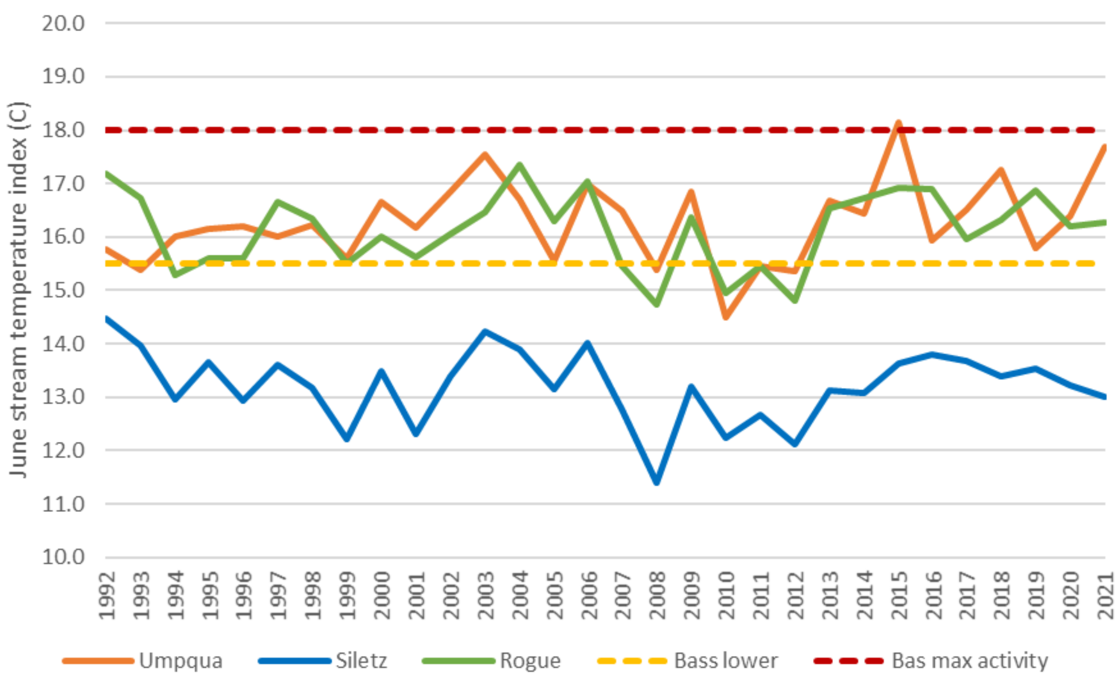


Figure 17. Stream temperature index at the head of tide in June in three coastal rivers.
Bass do not feed as actively when water temperatures are below 15° C and they are most active around 18° C.

Predation on juvenile salmonids by striped bass has been extensively documented. For example, striped bass were estimated to have consumed 383,000 juvenile salmonids over three months (April-June) in Coos Bay during 1963; this evaluation was conducted at a time when managers were considering enhancements to the striped bass fishery (Johnson et al. 1992). Considering the huge diversity of fish, invertebrates, and non-food items (cigarette butts, feathers, and wood) striped bass are known to ingest (Walter and Austin 2003; Nobriga and Feyrer 2008), they are serious predators and a threat to salmonids wherever they co-occur. No substantive studies or surveys for striped bass have been conducted in the Umpqua basin for decades.

Increasing temperatures in the Umpqua River during the summer steelhead outmigration period has likely exacerbated interactions with smallmouth bass, and whatever predation occurs may be compounded by the suspected increase in striped bass abundance.

Pinniped Predation. Steelhead tend to school less than other salmonids and occupy the upper layer of the water column (Ruggerone et al. 1990; Burgner et al. 1992; Beeman and Maule 2006) making them more susceptible to predation (Collis et al. 2001). Harbor seals *Phoca vitulina richardii* are one of the primary predators of juvenile steelhead in the ocean (Moore and Berejikian 2017; Thomas et al. 2017). ODFW monitored the distribution and abundance of harbor seals near the Umpqua from 1977 to 2014 (Figure S16). Harbor seal populations appear to have been stable from the early 1990s through the end of the dataset in 2014. Absent any data indicating increased predation by pinnipeds, it does not appear that this is an important driver of recent trends. The Umpqua is not regularly frequented by sea lions.

Avian Predation. Steelhead are particularly susceptible to predation by piscivorous birds (Collis et al. 2001; Evans et al. 2012, 2016; Freschette et al. 2012). The probability of predation by Caspian terns and California gulls/ring-billed gulls on steelhead smolts was two to five times higher than on juvenile salmon during outmigration in the Columbia River and on the California coast (Evans et al. 2012, 2016; Freschette et al. 2012). This might be due the larger size of steelhead smolts (Quinn 2005; Hostetter et al. 2012) and the fact that they are more surface oriented than salmon smolts (Winkler 1996; Cuthbert and Wires 1999; Beeman and Maule 2006; Pollet et al. 2012).

Extensive studies of predation on salmonid smolts by Caspian Terns, California Gulls, Ring-billed Gulls and Double Crested Cormorants have been conducted in the Columbia River. Colonial waterbird predation in these reaches is often the greatest source of steelhead mortality during outmigration, accounting for 28% to 91% of all steelhead smolt mortality during outmigration (Evans et al. 2019). Extensive Caspian Tern colonies are located in the Columbia River and in Puget Sound. During the 1980s, a rapid increase also occurred at other Oregon coastal colonies. Immigration of birds from other regions, reduced persecution in coastal areas, and protection of nesting habitat may have been contributing factors associated with this increase (Carter et al. 1995).

Though Caspian terns are common in bays and estuaries along the coast during spring and fall migration, no large colonies are present on the coast or on the Umpqua River, and sampling on Winchester Bay suggests relatively few Caspian terns and gulls feed there compared to other Oregon estuaries. However, the largest colony of double-crested cormorants on the Oregon coast is located on Bolon Island near confluence of the Smith River and the Umpqua River in the upper reaches of the Umpqua estuary Figure S17). It should be noted, however, that this colony is still more than an order of magnitude smaller than those on the Columbia River estuary, and their abundance in Winchester Bay during 2016-2021 was about half of what it was during 2003-2015 (Figure S18).

ODFW conducted a diet study in Winchester Bay in April and May 2012–2014. Coho were the most frequent prey of cormorants with estimates of around 70,000 smolts taken in 2013 and 2014. However, no salmonids were detected in the cormorant diet in 2015, and steelhead were never found in the stomachs of the 110 double-crested cormorants sampled in any year (Lyons, 2010). Double-crested cormorants move in response to colony disturbance, but there has been no known management of colonies on the central and south portions of the Oregon Coast. Bald eagles have apparently contributed to colony failures and abandonments at several sites along the Oregon Coast, but double-crested cormorants have not moved to other locations in the Umpqua River basin in response to recent declines at the Bolon Island colony. Given this evidence it does not appear that avian predation is driving recent declines.

Disease. ODFW has not conducted any recent disease monitoring for naturally produced Umpqua summer steelhead, either juveniles or adults. However, disease is known to be prevalent in the associated hatchery programs, indicating that disease hosts are present in at least some portion of the basin and that conditions are often within the suitable range for pathogens. It is unknown whether this is translates into increased prevalence and/or survival issues for naturally produced fish.

The disease outbreaks at the hatchery were not exclusive to steelhead and not uncommon to other species being reared in hatchery systems. Diagnoses involved external parasites: *ichthiopthirius, ichthyobodo, trichodina, epistylus*; internal parasites: sanguinicola, trematodes, *nanophyetus*; external fungus as saprolegnia and internal fungus as *Phoma herbarum*; gram-negative bacterial infections: Flavobacteriosis (cold water disease), Columnaris disease, and (rarely) opportunistic environmental gram-negative infections such as *Aeromonas/ Pseudomonas* complex, and one incidence of Furunculosis (*Aeromonas salmonicida* infection). Non-infectious disease such as drop-out disease and coagulated yolk in the fry is sometimes observed (ODFW, unpublished data).

Thiamine Deficiency. Multiple species and stocks of Pacific salmon have experienced declines in the number of returning adults over a wide region of the Pacific Northwest due to poor marine survival (low smolt-to-adult survival rates) (Welch et al 2018). This reduced survival could be partly caused by thiamine deficiency complex (TDC). Thiamine (vitamin B_1) is an essential vitamin with an integral role in many metabolic processes. Most animals acquire thiamine through their diet, so this deficiency can arise if food is thiamine deficient. Additionally, some prey species produce large amounts of thiaminase, which breaks down thiamine in the predators that eat them. TDC has been studied in the Baltic Sea and in the Laurentian Great Lakes where it has caused high rates of juvenile mortality in hatchery raised salmonids (Brown et al. 2005b). Thiamine treatments (immersion or injection) for adults and eggs have been used to increase survival rates (e.g., Fitzsimmons et al. 2005).

There have been very few studies conducted to date on thiamine deficiencies in salmonids in the western U.S. Serious deficiencies were identified in Alaskan Chinook salmon eggs (Honeyfield et al. 2016) and thiamine was associated with increased juvenile mortality for Chinook salmon from California's Central Valley (https://www.fisheries.noaa.gov/west-coast/science-data/monitoring-thiamine-deficiency-california-salmon). Research from Lake Ontario indicated that steelhead may require more thiamine than other salmonids (Futia and Rinchard 2019).

In a small pilot study, ODFW biologists recently found that winter steelhead eggs were thiamine deficient in two of three ODFW hatcheries sampled. Supplementation trials are ongoing at the

Oregon Hatchery Research Center to determine if adding thiamine to the eggs could help with early fry mortality and disease.

Marine Conditions

Marine conditions when juvenile salmonids enter the ocean can have profound consequences for their survival. Conditions that favor strong upwelling of cold, nutrient-rich water provide abundant food for growing fish, resulting in rapid growth and high survival rates. When warm, stagnant water dominates, food becomes scarce, and predators become more abundant (Brodeur et al. 2005; Schwing et al. 2005).

There are dozens of possible indices of marine conditions that can be used to evaluate the favorability of ocean habitats for juvenile steelhead. We took two approaches to examine conditions for steelhead. The first is an overview of multiple marine indicators to provide a qualitative assessment. The second involves mathematical modeling using one or more select marine indicators to quantitatively determine how much of the variability in steelhead returns can be explained by marine conditions.

The National Oceanographic and Atmospheric Administration (NOAA) provides summaries of multiple marine indicators commonly referred to as "stoplight tables" of ocean indicators (Table 5; Figure 18). Indicators are grouped into three categories: Climate and Atmospheric Indicators such as PDO, Local Physical Indicators like SST, and Local Biological Indicators like salmon catches or plankton. Each indicator is rated "good" (green), "fair" (yellow), or "poor" (red) relative to its impact on the marine survival of juvenile salmonids entering the Northern California Current. These indicators are then combined by calculating the mean of the ranks of the indicators. This mean is then characterized as good, fair, or poor. Five additional indicators are depicted on the chart but are not included as principal components including upwelling indices and spring transition date. Additional information about the methods used to measure and calculate the indicators can be found at https://www.fisheries.noaa.gov/west-coast/science-data/ocean-indicators-methods-and-background-materials.

The stoplight table (Figure 18) reveals that conditions were poor for many indicators during 2003-2005 and for 2014-2019. Since most summer steelhead from the North Umpqua River return to spawn at age 5 or 6, the five-year period from 2014 to 2019 was long enough to affect all cohorts and eliminate the possibility that a strong remaining cohort could reinforce the affected cohorts.

For the modeling effort, PDO, sea surface temperature (SST), two other measures of upwelling (Upwelling Index and Biologically Effective Upwelling Transport Index; BEUTI) and copepod biomass were all discarded as potential covariates. PDO and SST both correlated very closely with NPGO so including them would simply complicate the model without adding any additional predictive information. All the measures of upwelling were highly correlated (Figure S5). CUTI was selected for modeling since it had the highest correlation, on average, with the other two measures.

We selected the NPGO and the Coastal Upwelling Transport Index (CUTI; a measure of upwelling) for modelling. CUTI did not have a significant relationship with steelhead returns, but NPGO had the strongest and most significant relationship with recruits of all covariates tested ($p = 0.0002$; $R^2 = 0.50$). As mentioned previously, NPGO has recently been at some of the lowest levels in the history of the index (Figure 6). More detail on mathematical modeling is included in the discussion of covariates in the **Population Viability** section.

Given that marine conditions tend to be the strongest driver of salmonid survival, it is not surprising that returns have been poor in recent years, and all natural origin steelhead populations north of the Rogue River have declined, though none as precipitously as those in the North Umpqua River (Figure 5).

Table 5. Indicators included in the NOAA Ocean Stoplight Table.

Climate and Atmospheric Indicators
- PDO (Sum Dec-March)
- PDO (Sum May-Sept)
- ONI (Average Jan-June)

Local Physical Indicators
- SST NDBC buoys (Â°C; May-Sept)
- Upper 20 m T (Â°C; Nov-Mar)
- Upper 20 m T (Â°C; May-Sept)
- Deep temperature (Â°C; May-Sept)
- Deep salinity (May-Sept)

Local Biological Indicators
- Copepod richness anomaly (no. species; May-Sept)
- N. copepod biomass anomaly (mg C m-3; May-Sept)
- S. copepod biomass anomaly (mg C m-3; May-Sept)
- Biological transition (day of year)
- Nearshore Ichthyoplankton (Log mg C 1,000 m-3; Jan-Mar)
- Nearshore/offshore Ichthyoplankton community index (PCO axis 1 scores; Jan Mar)
- Chinook salmon juvenile catches (no. km-1; June)
- Coho salmon juvenile catches (no. km-1; June)

Principal Component scores (PC1)

Principal Component scores (PC2)

Ecosystem Indicators not included in the mean of ranks or statistical analyses
- Physical Spring Trans. UI based (day of year)
- Physical Spring Trans. Hydrographic (day of year)
- Upwelling Anomaly (Sum April-May)
- Length of Upwelling Season UI based (days)
- Copepod Community Index (MDS axis 1 scores; May-Sept)

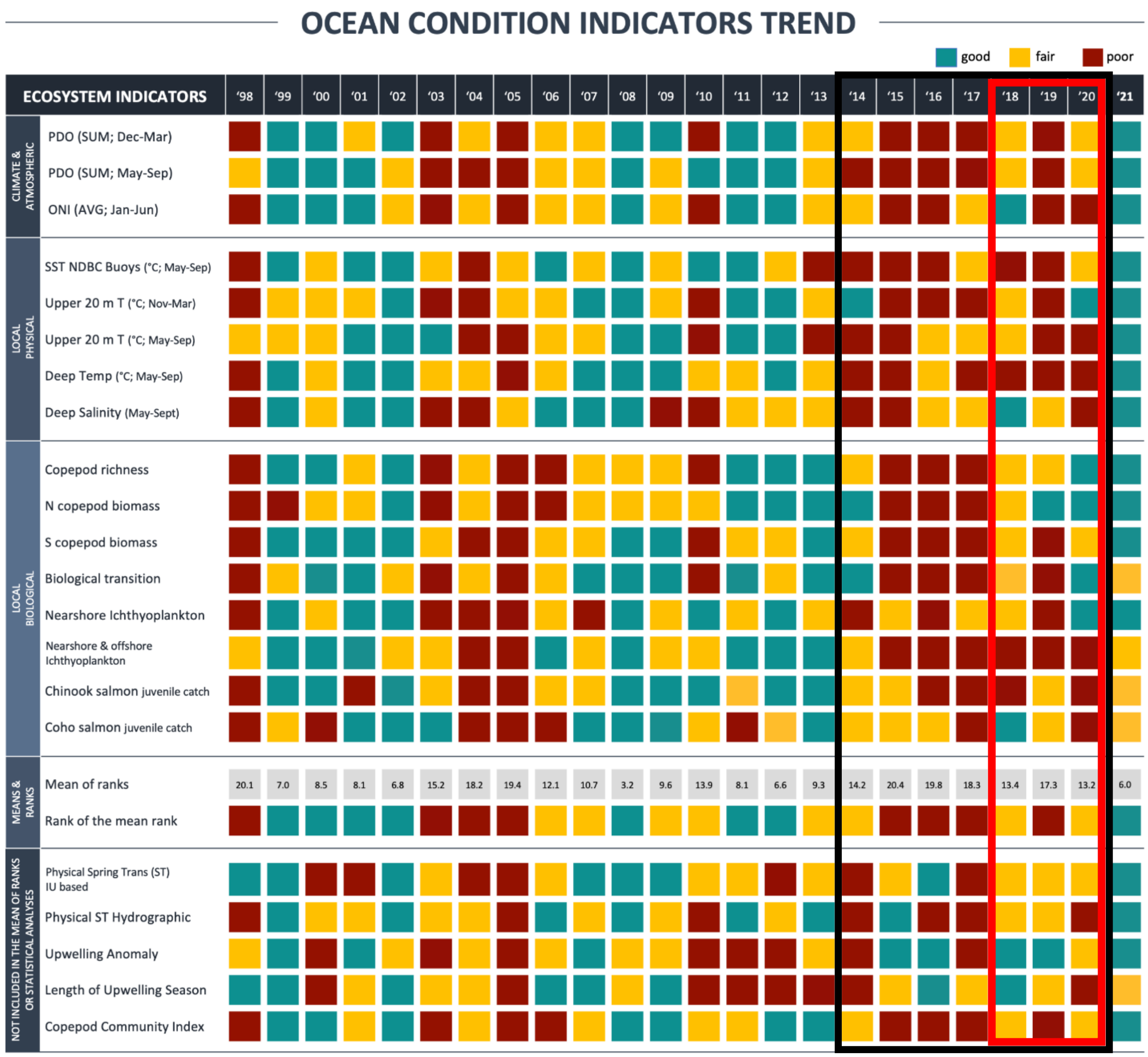


Figure 18. NOAA Ocean Stoplight Table for 1998-2021. Ocean conditions experienced by fish returning during the period of decline are outlined in black. Ocean conditions experienced by fish returning in 2021 are outlined in red. Available at: https://www.fisheries.noaa.gov/content/ocean-conditions-indicators-trends

Freshwater Conditions

We extended the NOAA marine stoplight approach to freshwater conditions by creating our own freshwater stoplight table. Tables like this are useful in providing a quick, overall impression of the condition of a suite of interrelated variables over time. This table includes indicators of streamflow and stream temperature during different life-history stages during the species' freshwater residence (Tables 6 and S6; Figure 19). To assess stream temperature conditions during the period of reference (1992 to 2020), we combined the NorWest stream temperature model with PRISM air temperature data. The NorWest model provides a single estimate of stream temperature using observations collected during the period from 1993 to 2012. We extracted PRISM air temperature data and NorWest modeled stream temperature estimates for two locations and seasonal periods. To assess conditions during smolt outmigration we extracted mean June air temperature data at COMID 24526610 in the lower Umpqua near the head of tidal influence. To assess conditions during summer holding (adults and juveniles) we extracted mean July air temperature data at three holding pools in the North Umpqua (100-fish, 5-mile and Upper Bend pools). In each case we calculated the mean temperature observed in the PRISM air temperature data for the period between 2002 and 2011 (to mirror the period used for NorWest models). We then divided the PRISM value for each year between 1992 and 2020 by the mean value to generate a deviation from the mean and multiplied this annual deviation by the appropriate NorWest modeled estimate to provide an index of stream temperature at the points of interest for each year. As in the marine stoplight table, freshwater conditions were poor for many indicators during 2013- 2016 and 2018 (Figure 19). Most fish spawned during this period would return as adults between 2018 and 2021.

The MWMT Index included in the table was also considered as a covariate in the PVA analysis, where it was found to have a marginally significant relationship with recruits ($P = 0.09$; $R^2 = 0.14$; see **Current Population Status** section). Though this relationship is much weaker than that with the NPGO, which explained roughly 50% of the variation, a covariate that explains 14% of the variation in steelhead returns is not necessarily inconsequential, especially when run sizes are small or when it contributes to additive effects with other stressors. The only other freshwater covariate tested in the PVA analysis was summer minimum flow, which was not found to have a significant correlation with recruits.

We compared the freshwater stoplight tables for two other summer steelhead population areas (Siletz and Rogue rivers; Figures S19 and S20) to the table for the North Umpqua (Figure 19, which includes mean ranks for each river basin and for marine conditions). We could not directly compare streamflow conditions among different basins because of differences in the size of the drainage area and location of the stream gage within the basin. Consequently, we assumed for each basin that the historical flow regime would be most advantageous, flow higher than the historical regime would be intermediate, and flow lower than the historical regime would be least advantageous (Table 6). We calculated the 25th and 75th percentiles of flow for the historical regime and set thresholds for stream flow such that years with flow below the 25th percentile were categorized as poor (red), years with flow greater than the 75th percentile were categorized as fair (yellow), and years with flow between the 25th and 75th percentiles were categorized as good (green). We were able to more directly compare the temperature indices and to make category thresholds relevant to biological limits. June category thresholds are related to the temperatures at which smallmouth bass most actively feed and summer holding thresholds are related to optimal and lethal temperatures for salmon and trout (Table 6). The freshwater stoplight tables suggest that freshwater conditions were more favorable for summer steelhead in the Siletz and even in the Rogue compared to the North Umpqua, particularly after 2012.

Table 6. Metrics for the Freshwater Stoplight Table.

Life stage	Metric	Justification
eggs, fry, parr	Max_cfs	Highest flows limit movement, disrupt redds, and can injure, kill or displace juvenile fish. Even if there isn't direct mortality or injury, high flows might disrupt feeding behavior and require greater energy expenditures. Assume negative linear relationship with flow.
adult migration	Access Days	Flows high enough to allow adults to migrate into the system are advantageous. Assume linear relationship with number of days of accessible conditions during the adult migration.
migration (in/out)	CFM	Shifts in the center of flow movement can disrupt phenological relationships between fish and prey, predators, and conditions for outmigration. We assumed that CFM in the midrange for the period from 1992-2010 is most advantageous, later CFM is fair, and earlier CFM is poor.
migration (in/out)	Day50	Similar to CFM
migration (in/out)	Day75	Similar to CFM
migration (in/out)	Dur25to75	Wider intervals between date of 25th percentile flow and 75th percentile flow indicates a more even temporal distribution of precipitation. Assume linear relationship with interval duration.
outmigration	SprFlow	Spring flow is advantageous for juvenile outmigration and might provide protection from predation by bass in the lower river. Assume linear relationship with flow.
outmigration	JuneT_Indx	Smallmouth bass in the migration corridor are more active at higher temperatures. Assume negative linear relationship with stream temperature index with category thresholds at 16 degrees C and 17 degrees C
outmigration	Out Migration	Spring flow and temperature both contribute to speed of outmigration and susceptibility to predation by bass. Average of both categories. Assume negative linear relationship.
summer holding	Min_cfs	Lowest flows restrict movement, exacerbate high temperatures, limit accessible habitat, and increase the success of predators. Assume linear relationship with flow.
summer holding	MWMT_Indx	In addition to causing direct mortality, high maximum temperatures can cause higher incidence of disease, disrupt feeding and growth, and make fish more susceptible to predation. Assume negative linear relationship with temperature.

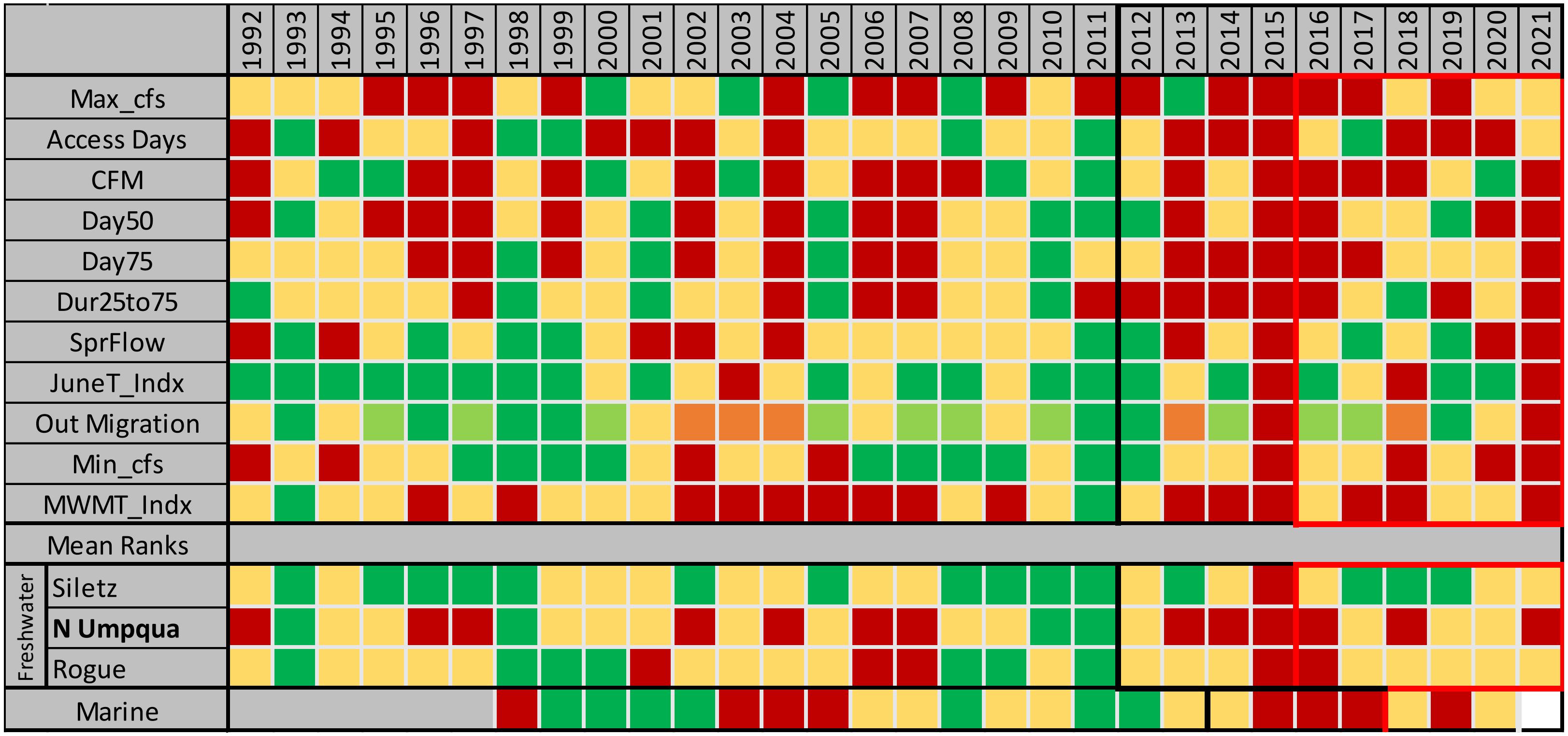


Figure 19. Freshwater stoplight table for North Umpqua summer steelhead. All flow metrics were calculated using measurements at the stream gage on the Umpqua River near Elkton. Temperature indices are from the head of tide in June, and in holding pools in July and August. The mean ranks of freshwater conditions for two other summer steelhead populations (Siletz and Rogue) and marine conditions are included for reference.

We highlighted comparisons of stream temperature since it is easier to make direct comparisons of temperature and to make biologically relevant distinctions (Figure 20). There was a clear separation in the maximum stream temperature index in areas where adult steelhead hold over the summer. Maximum holding temperatures in the Siletz hovered around suitable holding temperatures for steelhead, and in one year even the maximum temperature fell in the optimal temperature range for steelhead. Stream temperatures in the Rogue were higher than in the Siletz, but only exceeded the 22-degree threshold three times in the 30-year time series. In the North Umpqua, by contrast, the stream temperature index in holding areas was well above the maximum temperature for steelhead almost every year and exceeded the threshold by a greater margin, with the highest temperature observed occurring in 2021. This index is an approximation, and adults can find cooler water in deep pools (especially in the upper parts of the North Umpqua basin), but clearly steelhead are living much closer to their temperature limits in the North Umpqua than in the Siletz or even the Rogue.

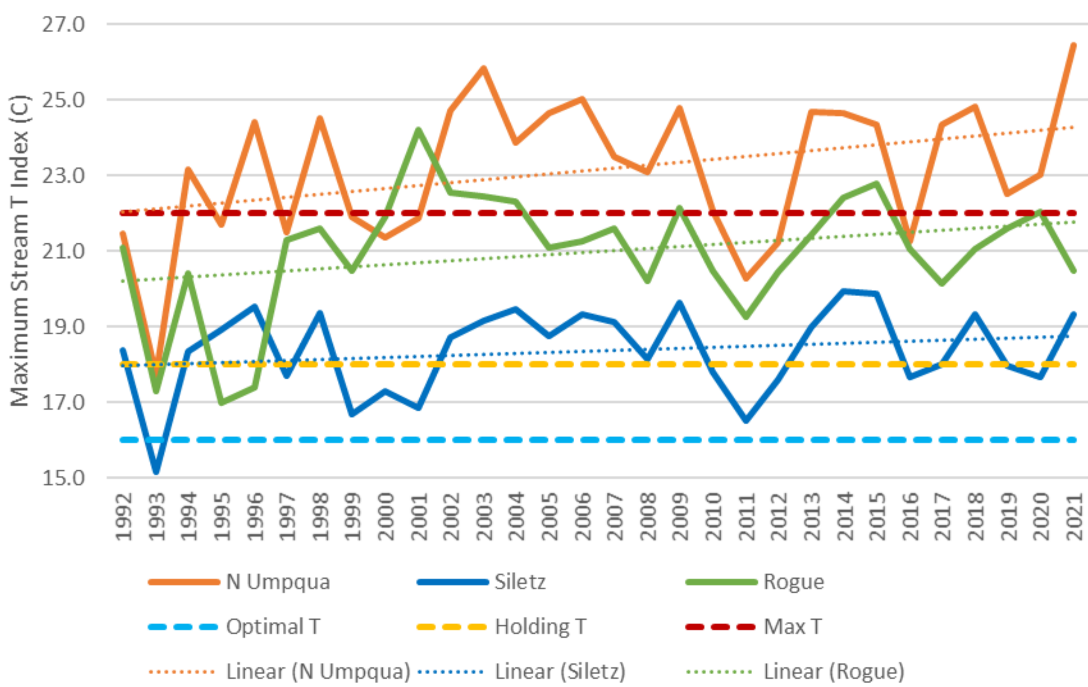


Figure 20. Comparison of MWMT in holding areas for adult steelhead in three coastal rivers.
The maximum temperature index (MWMT) for the North Umpqua, Siletz, and Rogue rivers is shown. Optimal water temperature for salmon and trout is indicated by the blue dashed line, suitable temperature for holding is indicated by the yellow dashed line, and the maximum temperature threshold for salmon and trout is indicated by the red dashed line.

In the next section we also compared the populations using temperature data from stream gages in each population (Figure 21). The advantage of the stream gage data is that these are direct observations of stream temperature. The drawback is that there are very few places with long time-series of stream temperature observations, and the places where they exist are not always the most instructive locations for the questions posed. The modeled stream temperature utilized in this section has the drawback of uncertainty, but the advantage of being available everywhere on the stream network and for the entire period of interest. We have employed both approaches in an effort to balance their advantages and disadvantages.

Additional Stream Temperature Analyses

We used two additional sources of stream temperature to evaluate within and outside basin status and trends: long-term continuous records of temperature recorded at gage or thermistor sites, and thermal infrared imagery.

Comparison of Gage Data. To posit the Umpqua basin within a larger regional context, stream temperature data was collated from USGS gages in the Umpqua and Rogue basins with the longest continuous record (Table S7; Figure 21). The USGS gage in the Siletz does not collect stream temperature therefore we formatted thermistor data obtained from an ODFW life cycle monitoring trap location on Mill Creek, a tributary of the Siletz River (Table S7; Figure 21). Some inconsistencies occur through time specifically with the Siletz data only available through 2018 and the data at Winchester unavailable for 2015 and 2016. We calculated daily average stream temperature across the summer (June through September) season. When all locations were compared, we found that temperatures in the Lower North Umpqua River located at Winchester had consistently warmer temperatures throughout the summer and through time, with comparable average daily temperatures in the Rogue River at Agness (Figure S21; Figure 22). When looking at all years combined (and variability reflected in confidence intervals around each location curve) we see that these two locations were over a degree higher on average in June, with the warmest temperatures peaking in July (Figure S21). The North Umpqua at Soda Springs gage had the lowest daily averages and lowest maximum average season temperatures followed by Rogue River at Prospect (Figure 22, Figure S21).

In addition to evaluating daily average stream temperatures, we compared maximum stream temperatures at Winchester through time (2017 through 2021) specifically contrasting previous years to 2021 temperatures during the spring and early summer months when adult summer steelhead would be migrating (March – June). High stream temperatures can slow, delay, or stop migration and increase pre-spawn mortality. Generally, the monthly mean maximum temperature has increased through time with 2021 having the warmest temperatures observed. Temperatures over the last five years during the months of March - May were all within a thermally suitable range for adult salmonids (min 7.68 °C in 2019, max 16.95 °C in 2021). However, June and July temperatures in 2021 were warmer than all previous years (June min 17.26 °C in 2017, max 22.62 °C in 2021; July min 22.6 °C, max 25.18 °C). This can be observed in Figure S27 which shows the distribution of daily average summer temperatures (June – September) at Winchester from 2017 – 2021 (Figure 20 also indicates this with other data). The mainstem Umpqua regularly sees temperatures >20 °C during the summer season; however, the duration of water temperatures over 22 °C has also increased through time. For example, in 2021, the temperature at the Winchester gage was greater than 22 °C 24 hours per day from June 21 – June 30, with temperatures reaching 29.1 °C during the heat dome event.

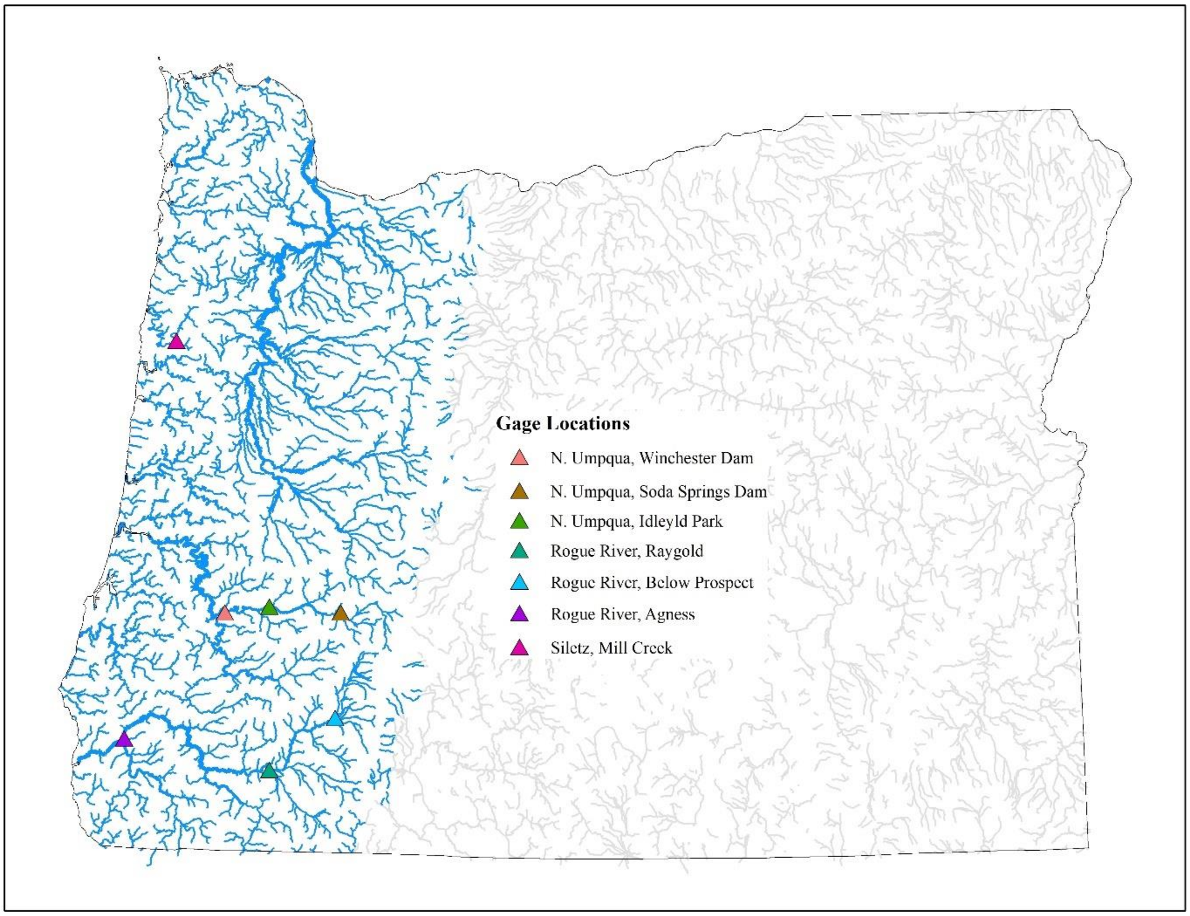


Figure 21. Hydrological gage or thermistor locations used in the stream temperature comparison.

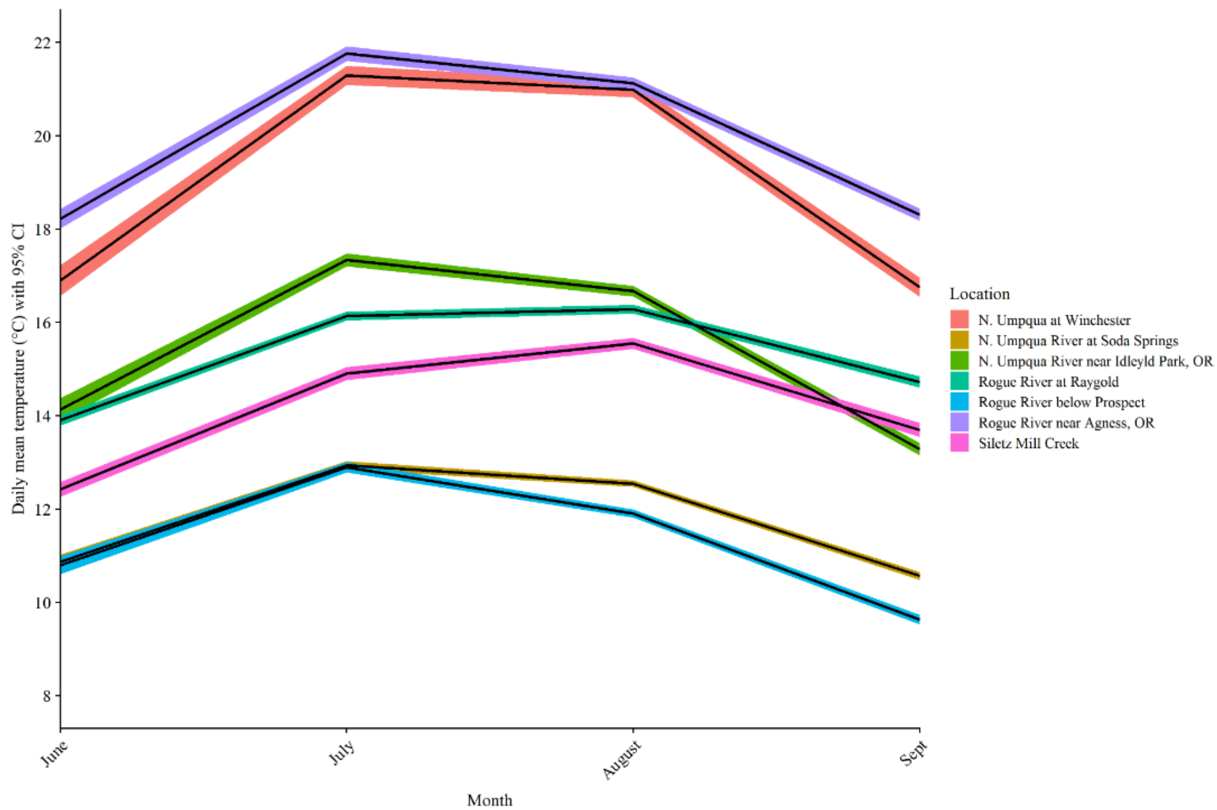


Figure 22. Daily mean summer temperature for locations within the Siletz, Umpqua, and Rogue basins. Temperature during summer (June – September) is averaged across all years (2008-2021). Shaded area represents 95% confidence intervals.

Longitudinal Profiles. Airborne thermal infrared (TIR) imagery measures the radiant temperature of surface water and is acquired using a fixed wing aircraft fitted with a TIR camera. Images are collected during the summer season (July and August), when air temperatures are generally warmest and when the weather is most optimal for flights. TIR imagery was collected in the North Umpqua in 2002 on July 23, 25, and 26. The majority of the TIR surveys were conducted by Watershed Sciences Inc., Corvallis, OR (USA) (now NV5, https://www.nv5.com/geospatial/; Table S8). Because of technological advances during this time, different TIR sensors were used (pre-2001 used scanned arrays; post 2001 used focal plane arrays); however, wavelength range, radio-metric calibration and sensitivity were all similar (Fullerton et al. 2015). For each flight, thermistors were distributed instream to ground-truth the remotely sensed temperatures and all thermal image data were georeferenced. These data have been used extensively to map thermal heterogeneity in Oregon (Fullerton et al. 2018; Fullerton et al. 2015; Torgersen et al. 2012) and can highlight locations that have cooler surficial water temperatures, implying a potential cold-water patch for salmonids to use a thermal refuge. See Torgersen et al. (2001) and Monk et al. (2013) for more details.

Fullerton et al. (2015) georeferenced the thermal image data and subsampled water temperatures from the images at approximately 150–200-m intervals along the thalweg of each river. We took those thermal image points and plotted longitudinal profiles for several different rivers/creeks in the North Umpqua basin (Figure S22). Each section had a different profile dependent on the position of the stream/river segment in the basin, influenced largely by elevation and gradient. For the mainstem Umpqua segments, the lower Umpqua had the warmest stretch with a minimum temperature of 24.6 ^{o}C and a maximum of 27.4 ^{o}C. From Winchester dam to Steamboat Creek, the minimum temperature was 17.1 ^{o}C and a maximum of 24.8 ^{o}C. The upper segment from Steamboat Creek to the approximate location of Toketee Dam had the coolest temperatures with a minimum temperature of 10.4 ^{o}C and a maximum temperature of 16.2 ^{o}C. In the tributaries, Little River and Steamboat had the coolest minimum temperatures of 16.2 ^{o}C and 16.4 ^{o}C, respectively, with maximum temperatures reaching 25.5 ^{o}C in the Little River and 22.8 ^{o}C in Steamboat (Table S8). Canton Creek and Rock Creek had minimum temperatures of 17.6 ^{o}C and 18.1 ^{o}C and maximum temperatures of 22.4 ^{o}C and 21.7 ^{o}C, respectively (Table S8). These data can be useful in identifying potential cold-water patches (e.g., reaches of stream with a difference $\geq$2 ^{o}C when compared to adjacent reaches) where fish can hold for thermal respite. When looking at the mainstem sections, there appear to be some cool sections in the lower mainstem before the confluence with the North Umpqua (Figure S23). Temperatures decrease distinctly moving upstream into the North Umpqua with the contribution of cooler tributary flow helping to maintain conditions within reasonable thermal thresholds for salmonids.

Instream Physical Habitat Condition

There are current physical stream habitat data in the North Umpqua watersheds in general and within the distribution of summer steelhead, specifically. Some census surveys were conducted in the early 1990s and starting in 1998, programmatic physical habitat surveys have been conducted within the Oregon Coast, including the Umpqua basin, as part of the Oregon Plan for Salmon and Watersheds. The sites were selected using a Generalized Random Tessellation Stratification (GRTS) sampling design with an imposed rotating panel design where sites are surveyed annually, every three years, every nine years, or once only. We compiled all surveys conducted in the North Umpqua basin, plotting a selection of habitat metrics (Table S9). In summary, the Upper North Umpqua River and Steamboat Creek had the highest percentage of secondary channel area, which can provide strong rearing conditions particularly during high flow conditions, though the distribution was broad (Figure S24). The Upper North Umpqua River also had the lowest pool frequency (more fast water than deep pools; Figure S25) indicating less area for holding. The Lower North Umpqua had the lowest gradient, channel shading and wood volume (Figures S24 and S26) which can indicate low complexity and increased temperatures. The tributaries of the North Umpqua had the deepest riffles (Little River) and the deepest (>1m) pools per kilometer (Steamboat Creek; Figures S24 and S25), which can translate to higher quality spawning habitats and thermal refuge opportunities.

Summary: Limiting Factors

Our assessment indicated that the risk posed by hatchery fish, as measured by pHOS, is lower than indicated by the hatchery proportion at Winchester Dam. This is attributable to spatial segregation of hatchery and wild fish. Hatchery fish prefer to spawn closer to the site of their release in the lower basin (below Deadline Falls, excluding the Little River sub-basin), notably Rock Creek; wild fish prefer to spawn in the upper basin, notably in the Steamboat Creek sub-basin. However, after accounting for differing spatial distributions of natural and hatchery origin summer steelhead, basin-wide pHOS exceeds the CMP target of ≤10%. The presence of hatchery origin spawners is expected to be near zero in the Steamboat Creek sub-basin preferred by naturally produced spawners, but pHOS is likely higher in some other sub-basins, particularly in the Rock Creek sub-basin where hatchery smolts are released. While there are no indications of negative demographic impacts from recent smolt releases, analyses indicate that attainment of the CMP's basin-wide pHOS targets would require smolt releases <30,000 and/or removal of additional hatchery origin fish by anglers or through management actions.

Beyond simply assessing pHOS targets, our assessment also considered demographic impacts of smolt releases on 1) naturally produced smolts during the year of their concurrent outmigration and 2) returning spawners during the year that most of the hatchery smolts would return to spawn. We found no measurable impact on naturally produced smolts. We did identify a marginally significant positive relationship between naturally produced adult recruits and hatchery smolt releases, suggesting some impact of hatchery smolt releases on naturally produced recruits. This could occur through the contribution of hatchery x hatchery and hatchery x naturally produced mating pairs to the overall abundance of naturally produced recruits.

We also modeled population viability incorporating a range of relative reproductive success for hatchery spawners and annual releases of between 0 and 165,000 hatchery smolts. Within the ranges modeled for both variables, no simulations showed the North Umpqua summer steelhead population dropping below the "quasi-extinction" threshold. However, simulation results indicated the population was unlikely to consistently meet or exceed desired abundance thresholds (as expected until more of the CMP actions across all limiting factors are implemented). All population viability modeling in this assessment, including this modeled scenario, assumes stationary conditions and no compensatory mechanisms. Results should be considered as indicators of the current relative risk, not predictions of future status.

Fishery mortality is an unlikely driver of recent trends in North Umpqua summer steelhead. Harvest of naturally produced summer steelhead has been significantly reduced above Deadline Falls for over 30 years and prohibited throughout the Umpqua basin for over 20 years. No angling has been allowed in the Steamboat Creek sub-basin (including Canton Creek) for 85 years, and angling is also prohibited in Rock, Calf, and Copeland creeks. Pre-spawn mortality incidental to catch-and-release angling was accounted for in datasets used for stock-recruit and population viability analyses, which indicated a low risk of extinction for the population. Current regulations (no retention of naturally produced fish and terminal gear restrictions) and other tools, including angling closures during low returns or poor conditions (e.g., high water temperatures) will continue to be important for minimizing angling-related mortality for North Umpqua summer steelhead as the future is shaped by climate change.

Other species and factors (bass predation and disease) were assessed, but conclusions were limited by data availability. Late-migrating juvenile summer steelhead may be at increased risk of encountering actively feeding smallmouth bass due to warming water temperatures in June. Predation by smallmouth bass may be compounded by striped bass predation given anecdotal reports of a recent rebound of the striped bass population. Disease outbreaks have occurred at Rock Creek Hatchery, but there has been no recent monitoring of disease for naturally produced Umpqua summer steelhead.

Ocean conditions are important limiting factors for North Umpqua summer steelhead. Conditions across a suite of ocean ecosystem indicators were generally poor for the period experienced by recently returning adults, and the NPGO index had the strongest and most significant relationship with adult recruits of all ocean and freshwater variables tested. Other marine-related limiting factors (pinniped predation and avian predation) did not appear to be important drivers of recent trends though data were limited for these factors. Thiamine deficiency may be related to low marine survival in Pacific salmon, and thiamine-deficient winter steelhead eggs have been observed in a pilot study on the Oregon Coast. However, data are not available to further assess thiamine deficiency as a limiting factor operating on North Umpqua summer steelhead.

Freshwater ecosystem indicators related to stream flows and temperatures also indicated that poor conditions for many indicators were experienced by adult summer steelhead returning to the North Umpqua River from 2018 to 2021, with 2021 having the highest temperatures observed. We found a marginally significant relationship between adult summer steelhead returns and MWMT for the sixth year of a year classes' life, but no clear relationships with MWMT for the first two years of a year classes' life. This suggests that the effects of summer maximum stream temperatures on adult returns is more apparent when they are holding as adults before spawning than when they reared or out-migrated as juveniles (which is likely a more complex relationship, with multiple years contributing to returns and outmigration likely affected by temperatures earlier in the season).

When we compared river temperature data from gages across the region in other basins with summer steelhead populations, we found that temperatures were warmest in Lower North Umpqua at Winchester, peaking in late July most years, and exceeded only occasionally by the Rogue River near Agness. The magnitude of stream temperatures and the effect of acute warming events is a concern. 2021 was the warmest year to date and maximum temperatures at Winchester Dam exceeded 24^{o} C, with maximum recorded temperatures near 26^{o} C (Figure

S27). Impacts of high water temperatures on migrating adult summer steelhead, such as those that likely occurred in 2021, may become an increasingly important limiting factor.

Analyses could be improved with additional data on smallmouth bass and striped bass predation and population dynamics in the Umpqua basin, encounter rates for naturally produced fish in catch-and-release fisheries above and below Winchester Dam, separating naturally produced recruits produced by naturally produced spawners versus naturally produced recruits produced by hatchery origin spawners and distributions of adult hatchery origin steelhead in the lower and upper North Umpqua sub-basins. Genetic studies would greatly improve our understanding of the demographic interactions between natural origin and hatchery steelhead.

Our assessment considered limiting factors related to ocean conditions (ocean productivity indicators, pinniped predation, and avian predation), freshwater conditions (stream flows and temperature, predation by smallmouth bass and striped bass, disease), hatcheries, harvest and fishery mortality, and reduced survival caused by thiamine deficiency complex. Ocean productivity and summer temperatures were significant covariates for recent demographic trends. We conclude recent declines in abundance of North Umpqua summer steelhead are likely a response to recent poor conditions for both ocean and freshwater rearing and outmigration of juveniles. Also, an already low abundance of returning adults in 2021 may have been further impacted by high water temperatures in the mainstem Umpqua and North Umpqua rivers during the summer migration period and holding prior to spawning. Freshwater stream conditions likely exacerbate known potential threats to steelhead such as predation and disease. Both hatchery and naturally produced summer steelhead appear to have been impacted by these factors.

FUTURE OUTLOOK

Oregon's native anadromous fish species encounter myriad stresses impacting their viability, persistence, and survival. Anthropogenic influence on natural environments is widespread, degrading freshwater rearing and spawning habitats, altering migratory corridors and blocking access to historic habitats via barriers such a dams, culverts, and tide gates. Compounding these existing challenges is a rapidly changing climate, which is altering the magnitude and variability of stream flows and temperatures, as well as the marine conditions, experienced by these species. Stream flows and temperatures are increasing during late winter and early spring but decreasing in the summer. As winters warm, snowpack in the Cascade Range continues to decline, reducing by 20% since 1950 (Siirila-Woodburn et al. 2021). Decreasing snowpack means less water flowing in streams during the summer and even more pressure given the competing water needs (e.g., water withdrawals for agriculture and well drilling for drinking water). Additionally, with rising temperatures increasing the rate of evaporation of water from soils and plants which translates to less water that will drain from the ground into river and streams (EPA 2022). While there has been a surge of restoration actions in the last few decades striving to improve habitat conditions and increase habitat area for fish rearing and refuge, improvements in water quantity and actions to reduce stream temperatures has not kept pace with the current rate of change.

Local populations of summer steelhead are particularly vulnerable to changes in climate given their life history. Adults migrate through warm river corridors often during peak temperatures in the summer and juvenile offspring are spending summers in increasingly warm tributary habitats. In this section, we examine the vulnerability of summer steelhead in the Umpqua basin considering current habitat conditions and projections of future conditions. Here we present 1) a species distribution model developed to assess current and future probability of presence given habitat conditions; 2) a comparison of stream temperatures in the Umpqua relative to other locations where steelhead are distributed as well as a summary of physical habitat conditions across the North Umpqua watersheds; 3) modeled stream flow using a mechanistic model (VELMA) to explore different scenarios (including wildfires) that could influence the resilience of this species in this basin; and 4) the outlook for future marine conditions based on projections for several metrics potentially linked to steelhead marine survival.

Species Distribution Modeling

Species distribution models (SDMs) are models that relate species distributional data (e.g., occurrence, abundance) at a particular location with information on the environmental conditions at the same location. Unlike current distribution maps that only convey where we expect a species to occur under current conditions, SDMs can be used a) to explain relationships between species and habitat, b) for prediction to locations without species data, or c) for predictions of distribution under different conditions (e.g., future climate). The probability of occurrence is obtained from the models and can be interpreted as habitat suitability or habitat availability depending on whether the habitat is accessible to the species. Like all modeling approaches, there are a number of assumptions and uncertainties. Some of the more prominent uncertainties specific to SDMs include: species occurrence data are always incomplete, there is imperfect knowledge of the factors driving occurrence, species-environment relationships are plastic, and if considering scenarios of future climate, there is uncertainty among model predictions based on emission assumptions (Wenger et al. 2013). Given these limitations, SDMs should be treated as hypotheses to test and validate. Additional sampling and modeling should be conducted to improve the model, making the process of developing SDMs iterative (Jarnevich et al. 2015). With that in mind, SDMs can be a useful management tool, informing habitat restoration and protection needs and other decisions.

Species distribution modeling can leverage spatial and temporal datasets, providing an approach to monitoring species by focusing efforts on locations with highest probability of presence given suitable habitat conditions. Because these models are used to infer relationships between species occurrence and environmental conditions as well as predict occurrence at unsampled locations, the environmental data that is used needs to have full coverage across the study area. The best data for this purpose is often considered landscape data, collected at broad resolutions with less specificity at localized or proximate stream conditions, which also strongly influence species distributions (e.g., physical habitat data, localized flows and temperatures, competing species, etc.). In the absence of detailed local data, SDMs can provide a place to start understanding how and why species distribute themselves throughout a stream network.

Study Area and Steelhead Data. The Umpqua basin is dominated by three distinct regions based on landscape and climatic features: the Coast Range, Klamath Mountains, and the Western Cascades (OSU 2021) (Figure 23A). Coast Range region elevations range from sea level to approximately 2,000 ft with a climate dominated by moderate wet winters and cooler dry summers and precipitation ranging 75 – 80 inches per year at the mouth of the Umpqua to approximately 53 – 60 in further inland. The Klamath Mountain region elevations range between 500-1000 ft, the climate is more variable with warmer temperatures in summer and cooler temperatures in the winter compared to the Coast Range. Precipitation in this region is strongly influenced by elevation with drier conditions in lowland valleys often resulting in stream drying. The North Umpqua basins are within the Western Cascades region. This region is characterized by rugged mountains and deeply cut valleys with elevations ranging from 1,000 – 5,000 ft. At high elevations, most of the precipitation falls as snow with annual ranges between 50 – 90 inches. The North Umpqua originates in the high Cascades with its source in porous basaltic lavas of the High Cascade geologic province flowing through tertiary volcanic formations of the Western Cascade Range (OSU 2021).

Juvenile steelhead survey data have been collected in the Umpqua basin since 1998 through the present (2022) as part of programmatic monitoring by ODFW. Individual sites were originally selected as part of a random, spatially balanced survey design within the distribution coho and steelhead (ODFW 2015). Sites ranged from 0.33 – 1.4 km (average 1.0 km) in length and were surveyed once during the summer season (July – September). At each site, most pools were snorkeled and all salmonids were identified and enumerated. Initially, (pre-2010) only pools that were ≥40 cm in maximum depth were snorkeled. However, in 2010 after some evaluation conducted in the Smith River (Umpqua), this criterion was lowered to include pools ≥20 cm in maximum depth (Constable 2009). These data cannot differentiate between summer and winter runs of steelhead. We modeled surveys within the Lower and North Umpqua Basins, summarizing the final results for the North Umpqua watersheds only to specifically target summer steelhead. Steelhead were not counted officially until 2002; therefore, in total, there were 20 years of data (2002 – 2021) (Figure 23B).

Spatial Landscape and Modeling. All spatial data used in this analysis were formatted to a modified version of the National Hydrograph Dataset (NHD) high resolution (1:24k) stream network. Survey reaches as well as all the landscape predictor variables were georeferenced on this stream network. We evaluated the influence of several different spatially explicit landscape variables as predictors of steelhead distribution (Figure S28). We were challenged in combining all prospective predictor variables as they resided on three different stream networks (Table

S10). Stream temperature data (NorWeST; Isaak et al. 2017) and flow permanence data (PROSPER; Jeager et al. 2019) were modeled on the 1:100k NHD. Slope and stream size were obtained from the NHD high resolution (1:24k) network. Finally, solar radiation was obtained from TerrainWorks (SolMean; Benda et al. 2007) and was on a custom 1:24k stream network. Due to this, there were streams that did not get included in the analysis. We georeferenced as much of the spatial data as possible to the NHD high resolution dataset (Table S10; Figure 23).

We initially explored several different statistical models and response variables (e.g., count, occurrence) from zero-inflated, negative binomial (ZINB) models to generalized additive models (GAMs) and generalized linear mixed models (GLMMs). Given the low sample size across the study area (Lower and Northern Umpqua watersheds) and our inability to get good model fit with the more sophisticated models, we chose to model the probability of the presence (using a binary 0/1 response) of juvenile steelhead, fitting a logistic regression model (a type of generalized linear model) with static spatial predictor variables (Table S11). We took the maximum observation across the period when delineating the presence/absence response, e.g., a surveyed reach needed to have at least one presence across the period to be considered present.

A series of 14 candidate models were fit to the data and Akaike information criterion (AIC) was used to compare models. The model with the lowest AIC value was chosen. All models were fit using the lme4 package in R Studio (Bates et al. 2011). Prior to developing the candidate models, we evaluated multi-collinearity among the predictor variables by calculating the variance inflation factor (VIF). Variables with high VIF values were not included together in the same model. Given the dearth of steelhead data, and more specifically summer steelhead, we kept the models simple (e.g., low number of predictors per model) to avoid overfitting.

To evaluate model fit, we assessed simulated residuals and evaluated the accuracy of our model by assessing how well the model held up the assumptions of logistic regression. Interpreting standard residual plots from these models often provides inaccurate results. We therefore used the DHARMa package in R to simulate residuals from our best model (Hartig 2017). Using this package, new data are simulated from the fitted model for each observation. An empirical cumulative density function is calculated for the simulated residuals describing the possible values given the predictor combination. The quantiles of scaled simulated residuals were tested against residuals from a uniform distribution. We then evaluated the frequency of simulated values, QQ plots of the observed versus the expected and the residual versus predicted values. Finally, we quantified model performance by evaluating how well the model classified presence versus absence, using a confusion matrix to evaluate binomial classification results. A confusion matrix consists of four different counts: true positive, true negative, false positive, false negative. These four counts can then be used to calculate class statistic measures to quantify model performance: accuracy, precision, prevalence, misclassification rate, true positive rate and true negative rate (see Table S12 for descriptions).

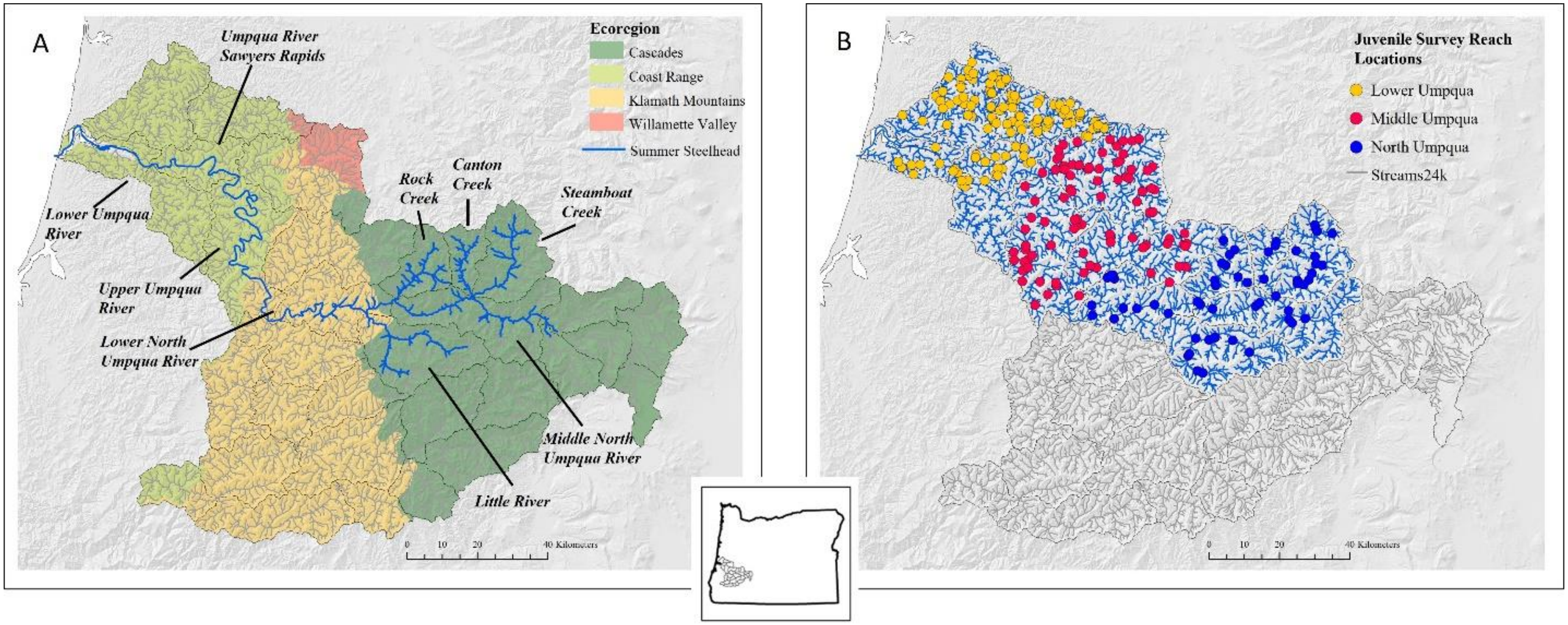


Figure 23. Background maps for the species distribution model.
Panel A: distribution of summer steelhead within the Umpqua basin across ecoregions and watersheds and Panel B: juvenile steelhead survey sites used in the spatial distribution model.

Using the best model, we predicted the probability of juvenile steelhead presence within the North Umpqua watersheds. We excluded the uppermost North Umpqua watersheds (e.g., Fish Creek, Clearwater Creek, Upper North Umpqua, and the headwaters) as they were not sampled in the original dataset. We substituted metrics representing modeled future conditions. For future predictions, we had two potential covariates to leverage: temperature and precipitation. For stream temperature, we used the 2080 (2070 – 2099) projections from the NorWeST stream temperature dataset which are based on an ensemble of Global Climate Models (GCM) and the A1B climate scenario (Isaak et al 2017). For spring precipitation, we used a multi-model mean derived from 20 Statistically Downscaled CMIP5 models (Abatzoglou 2013). We modeled the 2070-2099 mean using the high emissions (RCP 8.5) scenario for both predictor variables.

Best Model Description. Based on the best model selected using AIC, the probability of juvenile steelhead presence was influenced by the total spring precipitation, distance from the ocean, and maximum weekly maximum temperatures (Table S11; Figure S29). Overall, occurrence was higher at sites in the upper watersheds of the North Umpqua and sites located in the Lower Smith River watershed in the Lower Umpqua. The probability of juvenile steelhead presence was higher in reaches where spring precipitation was higher and in reaches with lower maximum weekly maximum temperatures (Table S11; Figure S30).

Model diagnostics were good, with well-distributed residuals and normal QQ plots (Figure S31). Overall, the model's accuracy was 75%. Over 90% of observations classified as presence were modeled as presence. The misclassification rate of the model was 24% (Table S12). However, it appeared the model did not predict absences well, having a 20% false positive rate.

Prediction and Future Projection. Using the best model, we predicted the probability of presence across all streams in the North Umpqua watersheds. The probability of juvenile steelhead presence is relatively high throughout streams in the North Umpqua, specifically Rock Creek, Canton Creek and Steamboat Creek (Figure 24A). Probability of presence is low from the outlet with the Pacific Ocean and moving upstream through the lower Umpqua River and into the lower North Umpqua River. Predictions of presence increased upstream of Winchester Dam and into the Middle North Umpqua River. We estimated >70% probability of presence throughout the Rock Cr, Canton Cr, and Steamboat Cr. watersheds.

To understand how future temperature and precipitation might influence the distribution of this species, we projected the model into the future using 2080 projections of maximum weekly maximum temperature and spring precipitation. Future predictions of the probability of juvenile steelhead were not vastly different from current predictions. There was little change in the probability of presence in most streams; locations where the current probability of presence is relatively high remained so in the future (Figures 24A and 24B). In some tributaries in the North Umpqua watersheds, very small decreases (<1%) in the probability of presence were predicted (Figure 24B).

Results. The probability of juvenile steelhead presence is generally high, with probabilities greater than 75% in the North Umpqua watersheds. Presence of juvenile steelhead is lower in the mainstem North Umpqua River with the probability of presence increasing moving upstream, particularly after the confluence with Rock Creek. Based on this model, spring precipitation, stream temperature, and distance from the ocean were the most influential predictors, with reaches in the upper portions of the watersheds having more suitable habitats, lower maximum weekly maximum temperatures, and higher average spring precipitation, providing better conditions for juvenile steelhead. Future projections of distribution are not considerably different from current conditions; the strongholds in the North Umpqua remain the same.

While this model is a good first step to understanding how juvenile summer steelhead are distributed in the Umpqua, there are several issues. First, summer steelhead were not distinguished in the field surveys from winter steelhead; therefore, this model represents a more generalized distribution of *O*. *mykiss*. We presented the predicted occurrence of juvenile steelhead in North Umpqua watersheds where summer steelhead are primarily thought to occur. While there is no direct evidence suggesting that habitat preferences differ between summer and winter juvenile steelhead, the predominance of this species in the North Umpqua is evident in the population structure derived from adult spawner information. Summer steelhead adults enter rivers as sexually immature fish, waiting as many as 10 months in freshwater before spawning. These summer-run fish therefore are smaller in size and have the time to migrate longer distances to access habitats and spawn in the upper North Umpqua watershed. Winter-run fish are mature when they enter freshwater between November and May, spawning shortly after they arrive.

Next, we had relatively small sample sizes (sites snorkeled for juvenile steelhead) over time and therefore we were not able to adequately incorporate the effects of temporally variable climate conditions on instream habitats and how that influences the distribution of juvenile steelhead. Two of the most influential covariates for juvenile salmonid survival are stream temperature and flow. For both, we used the best available data which had broad spatial coverage but did not have a temporal component (data were averaged across 2002-2011). Spring precipitation emerged as a stronger covariate than mean summer flow, but at best it correlates to runoff and natural flow, not accounting for any water withdrawals that might occur in the watersheds.

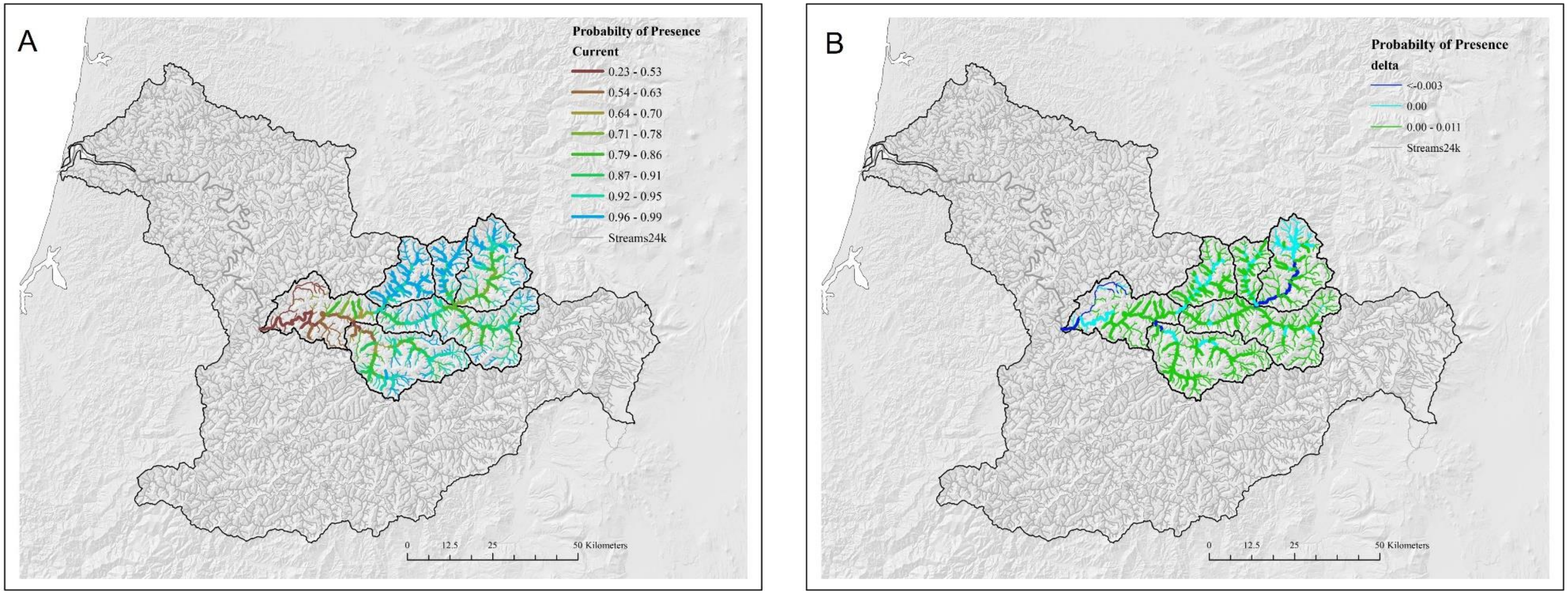


Figure 24. Spatial distribution of the current and future probability of juvenile steelhead presence.
The current probability of juvenile steelhead presence (A) and the change in the probability of presence in the future (“delta”: current minus future; B) in the North Umpqua are shown.

While we were not able to model the probability of juvenile steelhead presence through time due to sample size, we can characterize the temporal variability of temperature and precipitation relative to juvenile steelhead occurrence. Using the NorWest/PRISM temperature index (see **Limiting Factors** section of this report) and total spring precipitation we observed how varying climatic conditions influenced the occurrence of juvenile steelhead (Figure S32A and S32B). Juvenile steelhead predominantly occupied habitats with MWMT below 21 ^{o}C, though they clearly accessed cooler water when available as seen in 2011 and 2012 when wetter than normal spring conditions likely resulted in median MWMT temperatures around 17 ^{o}C . In 2013, stream temperatures warmed across the region, which coincided with lower abundances overall. In 2015, the basin received lower than average spring precipitation, which likely influenced both the ability of adult summer steelhead to access some of the higher quality habitats and the survival of juvenile summer steelhead holding over the summer in the tributaries. Oregon continues to experience increasingly warm thermal conditions, with maximum temperatures in streams increasing each year (see **Limiting Factors** section).

Improvements to the summer steelhead distribution model would include thorough and localized stream temperature data and flow estimates that account for water withdrawals and flow permanence and consistent biological sampling or surveying of summer steelhead abundances and occurrence throughout the North Umpqua River.

Current, Future, and Post-Fire Streamflow Simulations

The Visualizing Ecosystem Land Management Assessments (VELMA) model is a spatially distributed ecohydrological model developed by the U.S. Environmental Protection Agency (EPA) (Abdelnour et. al 2011; 2013; McKane et al. 2014). The VELMA model links hydrologic and terrestrial processes to create an ecosystem-based model capable of modeling landscape conditions as well as allowing the evaluation of the effects of disturbances (e.g., fire, timber harvest, or climate change). Recently, VELMA has been integrated with Penumbra, a mechanistic model that simulates ground-level solar irradiance (Halama et al. 2018). When coupled, these models allow for daily stream temperature modeling. While both models have individually been published in peer-reviewed journals, the coupled model has yet to be submitted for publication. For this assessment, we separately examined how 1) future climate and 2) large-scale fires might affect the timing and quantity of streamflow as well as stream temperature in the North Umpqua. Given the limited time for this assessment and the significant staff and computational time necessary to develop and run the model for a location, only three study watersheds were selected for this analysis: Rock Creek, Steamboat Creek, and Canton Creek. (Figure 25). These watersheds were selected as areas of focus to represent environmental conditions affecting some of the strongest sub-basins for summer steelhead (Steamboat and Canton) or areas recently impacted by wildfire (Rock Creek and Canton). The results of the species distribution analysis suggests that all three of these watersheds had a strong probability of steelhead juvenile presence; also note again that the available data did not distinguish between winter and summer steelhead, so the analysis provided a generalized distribution of *Oncorhynchus mykiss*.

Landcover Covariates. VELMA-Penumbra requires several spatial data inputs to initialize the landscape. These inputs include a digital elevation model (DEM), land cover species, pervious surface, land cover age, biomass, and underlying soil texture (Figure S33).

Hydrological flows of the watersheds were derived from a 30m DEM acquired via the U.S. Geological Survey (USGS). At this resolution, each pixel in the dataset represents a 30m-by-30m square on the earth's surface. Each watershed was conditioned using the EPA's pre-processing program called JPDEM. JPDEM "flat processes", or conditions, the "raw" DEM to be compatible with the VELMA model by removing sinks or large, flat areas – ensuring water will move seamlessly to the outlet. Outlets of the stream network were selected to correspond to the 1:100k NHD. These designated outlets correspond to confluences across the network, so output data can be linked spatially back to the stream network.

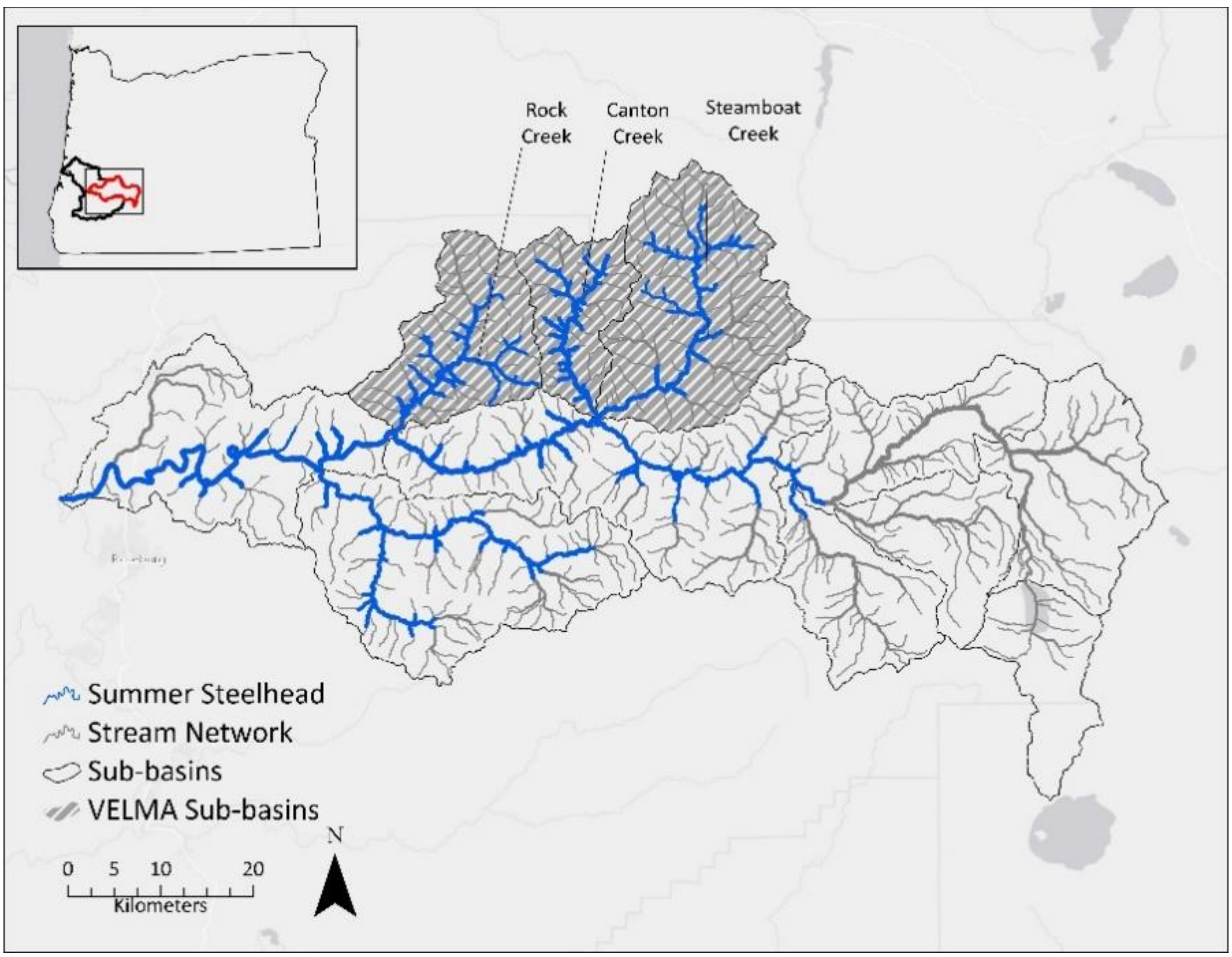


Figure 25. Location of subbasins included in the VELMA analyses.

Land cover spatial data were collected to characterize the terrestrial processes within each watershed. Upland portions of the North Umpqua watershed are dominated by coniferous forests, so a species cover map was created by reclassifying the Landscape Ecology, Modeling, Mapping & Analysis (LEMMA) gradient nearest neighbor (GNN) species dataset to represent two cover types: conifer and alder (Ohmann et al. 2011). These simplified land cover parameters have been calibrated and proven successful at modeling flow within numerous watersheds along the Pacific Northwest. Land cover age and land cover biomass inputs were derived from LandTrendr products that correspond to conditions in 1990 and thus all simulations were set to begin on January 1st of that year (Kennedy et al. 2010). A pervious surface landcover dataset was derived by calculating the inverse of the Multi-Resolution Land Characteristics Consortium (MRLC) impervious surface layer. To account for effects of anthropogenic development on the landscape a pervious surface data layer was generated with values that ranged from 0 (completely impermeable) to 1 (completely permeable). Lastly, a soil texture layer was derived by combining the U.S. Geological Survey's (USGS) Soil Survey Geographic Database (SSURGO) and State Soil Geographic (STATSGO) soil datasets, and then simplifying the soil texture output to 5 classifications: sandy loam, loam, clay loam, silt loam, and silty clay loam. Both SSURGO and STATSGO are soil databases, but SSURGO is a higher resolution, county-based dataset, while STATSGO is a lower resolution, statewide dataset. Where available, SSURGO data were always prioritized, but when SSURGO data were absent, STATSGO data were used to fill gaps to create a continuous soil texture layer. The five soil textures were selected based on their prevalence throughout the Oregon coastal region. Any smaller areas of differing soil textures not included in the list above were combined with the majority neighboring soil texture for simplicity.

Climate Data. Daily precipitation (mm) and temperature (°C) data were collected from eleven separate climate scenarios. Sites for these data collections were randomly selected within each watershed so that no two sites were < 4km from each other (the resolution of the PRISM dataset; Figure S34). Historical, or current, climate conditions were represented by daily PRISM data collected from 1990-2020. Future climate scenarios were collected from the University of California Merced Climatology Lab's website for the years 2065-2095. Future climate data were generated by the Multivariate Adaptive Constructed Analogs (MACA) climate downscale process and collected from 10 different 8.5 Representative Concentration Pathways (RCP) climate scenarios: CanESM2 (Canada), CCSM4 (USA), CNRM-CM5 (France), CSIRO-Md3-6-0 (Australia), GFDL-ESM2M (USA), HadGEM2-CC365(United Kingdom), HadGEM2_ES365 (United Kingdom), inncm4 (Russia), IPSL-CM5A-MR (France), and MICRO5 (Japan). These ten future climate scenarios were selected as they appeared to perform most robustly in the Pacific Northwest for temperature and precipitation (River Management Joint Operating Committee (RMJOC): Bonneville Power Administration, United States Army Corps of Engineers, United States Bureau of Reclamation 2018). While these data are available so that any one of the individual scenarios could be modeled with VELMA, the mean of the temperature and precipitation data for the ten scenarios were used for this analysis as future climate projections can vary substantially. All other VELMA input data was kept constant between future and historical models to isolate the effect of future climate conditions on streamflow.

Disturbance. To best characterize the reality of the environmental disturbances that occur annually within the watersheds that would affect the hydrological results of VELMA, timber harvest disturbances were implemented by identifying areas of harvest via the change detection data available through download via the LandTrendr algorithm on Google Earth Engine (Kennedy et al. 2018). Landtrendr detects change annually, so large blocks of change (excluding areas known to be fire disturbances) were assumed to be the result of timber clearcut harvest for model simplification. Parameters to simulate clearcuts and fire are available in Table S13. Fire boundaries were identified the National Interagency Fire Center's wildland fire perimeters (Figure S35). Large fires occurring in each watershed analyzed include the Archie Creek fire and the Chaos Fire complex. Both disturbances were initiated in 2020 for ease of modeling, though in reality, the Chaos Fire complex occurred in 2021. Post-fire, models were run for an additional 30 years with climate conditions repeating from those of 1990-2020. Fire analyses were therefore run for 61 years (1990-2051).

VELMA Model Evaluation. Three metrics were used to evaluate the performance: the Nash-Sutcliffe efficiency (NSE) (Nash and Sutcliffe, 1970), the ratio of the root mean square error to the standard deviation of measured data (RSR), and Kling-Gupta Efficiency (KGE, Gupta et al. 2009). Among the three watersheds, only one gage station had available data during the period of analysis – USGS gage station #1431670 located at the outlet of Steamboat Creek (Figure S36). Therefore, evaluation was only conducted for this watershed's simulation. Because of the proximity of the watersheds as well as their similarity in soil and landcover, we assumed that Steamboat's model performance was comparable to the other two watersheds. When the 30 years of daily simulation were compared to 30 years of collected gage data (the first year of VELMA was excluded as it is considered a "run-up" year), the NSE was 0.54, the RSR value was 0.68, and the KGE value was 0.49 (Figure S37). All of these metrics met criteria for a good model as asserted in Moriasi et al. 2007, which states an NSE > 0.50 and RSR ≤ 0.70 indicate a satisfactory fit. Though the NSE was lower than expected, based on previous modeling efforts on the Oregon Coast, this could be due to the inclusion of harvest scenarios not previous modeled. The NSE is traditionally used to evaluate hydrological models, though it is sensitive to high and low flow comparisons, alternatively, the KGE metric is increasingly being used to evaluate model performance, though the two metrics cannot be directly compared. Generally, a KGE value > 0.3 is considered a good predictive model, and because our model met all three metric standards, we are satisfied with the simulation's ability to model flow (Knoben et al. 2019).

Temperature Evaluation. The usage of the linked VELMA-Penumbra to model stream temperature is new and has not yet been published in a peer-reviewed journal; any conclusions based on our analysis should take this into consideration. Two locations were used to evaluate stream temperature modeling: one on a small tributary of East Fork Rock Creek and the other on Canton Creek (Figure S36). Thermistor data for East Fork Rock Creek were limited to May-September of 2006, May-September 2007, and July-August 2009. Conversely, Canton Creek thermistor data were available for several years in the summers of 2002, and 2004-2011; however, year-round data becomes available from the summer of 2012 to 2019 (Figure S38). When compared to thermistor data, the East Rock Creek temperature model yielded a NSE of 0.24, KGE value of 0.72, and RSR value of 0.87. These values fall short of the acceptable metrics for a behavioral model, though the small size of the stream as well as the paucity of observed data could contribute to its low performance. The performance of the Canton Creek simulation was slightly better with an NSE of 0.64, KGE of 0.69, and RSR of 0.60. These results met our criteria listed above, and multiple years of temperature data validated our simulation. Interestingly, the Canton Creek simulation resulted in a consistent over-estimation of temperature – an opposite trend observed anecdotally from NorWest, and the smaller, Rock Creek East Fork simulation under-estimated temperature. Because the linked models have not been extensively validated, more information is needed to be able to make direct conclusions from these results.

Effect of Future Climate. The effects of climate change were evaluated for Rock Creek and Canton Creek by keeping all environmental conditions the same between 1990-2020 and 2065-2085 including initial age and biomass values as well as annual harvest events. Only temperature and precipitation values were altered at the climate stations (Figure S34). Daily results were averaged across months for the 30 years of simulation for comparison of the projected changes in timing and amount of flow. In both watersheds, there was a projected decrease in flows in spring, summer, and fall seasons (Figure 26). Modeling suggested that winter flows would be less affected than the other seasons. Due to the uncertainty of the temperature modeling, future climate stream temperatures were not included in this analysis, though numerous sources agree they are expected to rise with increasing air temperatures and decreasing flows.

Effect of Fire. Climate change is expected to increase the frequency and intensity of wildfires on the landscape (Meehel et al. 2007; IPCC 2007). While these intense fires can increase temperatures during the fire's duration, they can also alter long-term stream temperatures via the destruction of riparian vegetation which decreases stream shading and subsequently increases stream temperatures until/if riparian vegetation regrows. The effects of large fires on the landscape on instream temperature and flow were evaluated for Rock Creek and Canton Creek by comparing flows between 1990-2020 and 2021-2051 after implementing a single, large forest fire in 2020. Climate variables were re-started so that the climate in 1990 was the same as 2021 to attempt to isolate the effect of loss of vegetation vs. change in climate. In all simulations, both stream temperatures and flows were shown to increase following wildfire events (Figure 27). These results are consistent with other, in-situ and modeled studies on the impact on streams following high-severity wildfires (Leonard et al. 2017). In addition to their potentially devasting effects on human infrastructure and threat to human life, these large fires will have a potentially dramatic effect on the available habitat for salmonids by quickly increasing stream temperatures. This analysis focused on the direct effect of wildfire on instream temperature and flows. There are potential short-term benefits of fire on fish production, such as increase in instream wood for habitat and potential increase in beaver activity (Falke et al. 2015; Flitcroft et al. 2016) that would potentially lower temperatures, but these were not analyzed here.

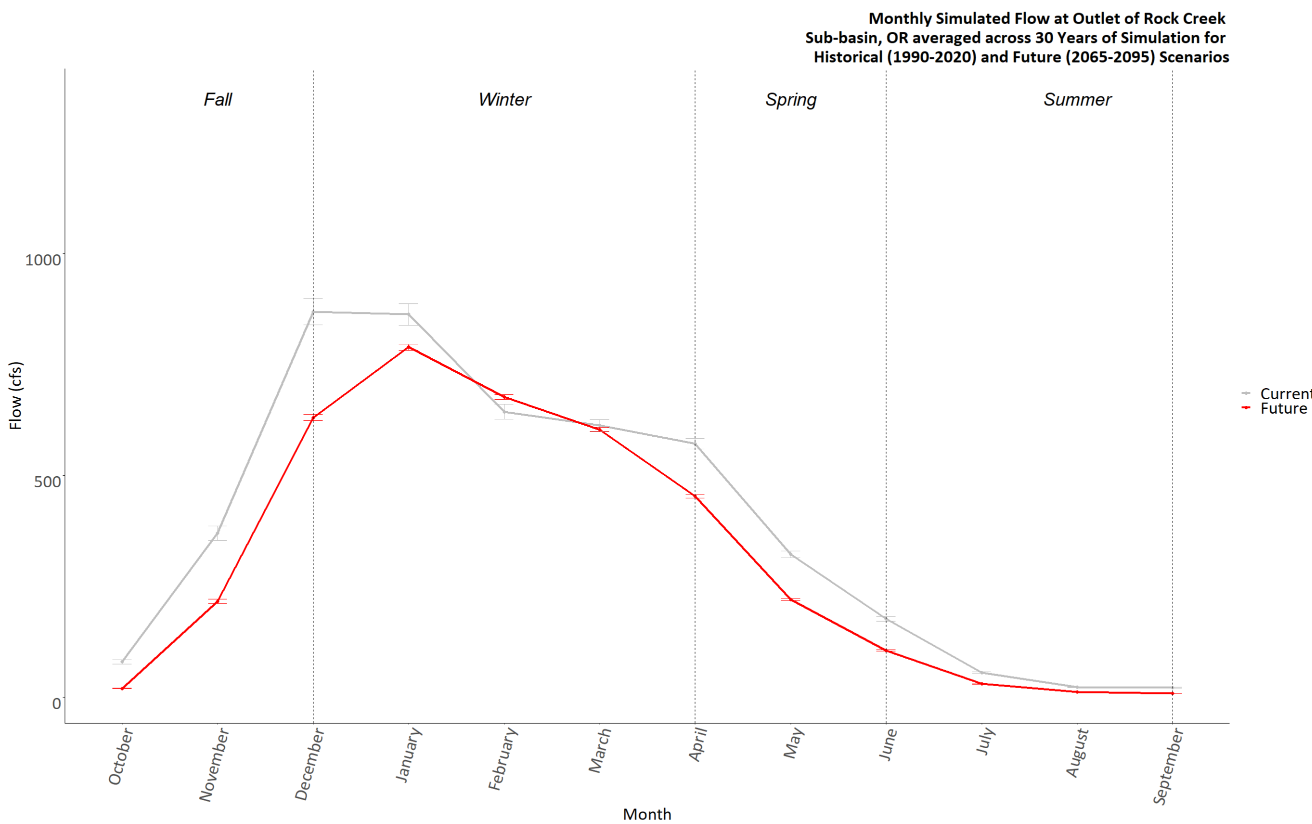


Figure 26. Climate change simulation at the outlet of Rock Creek.

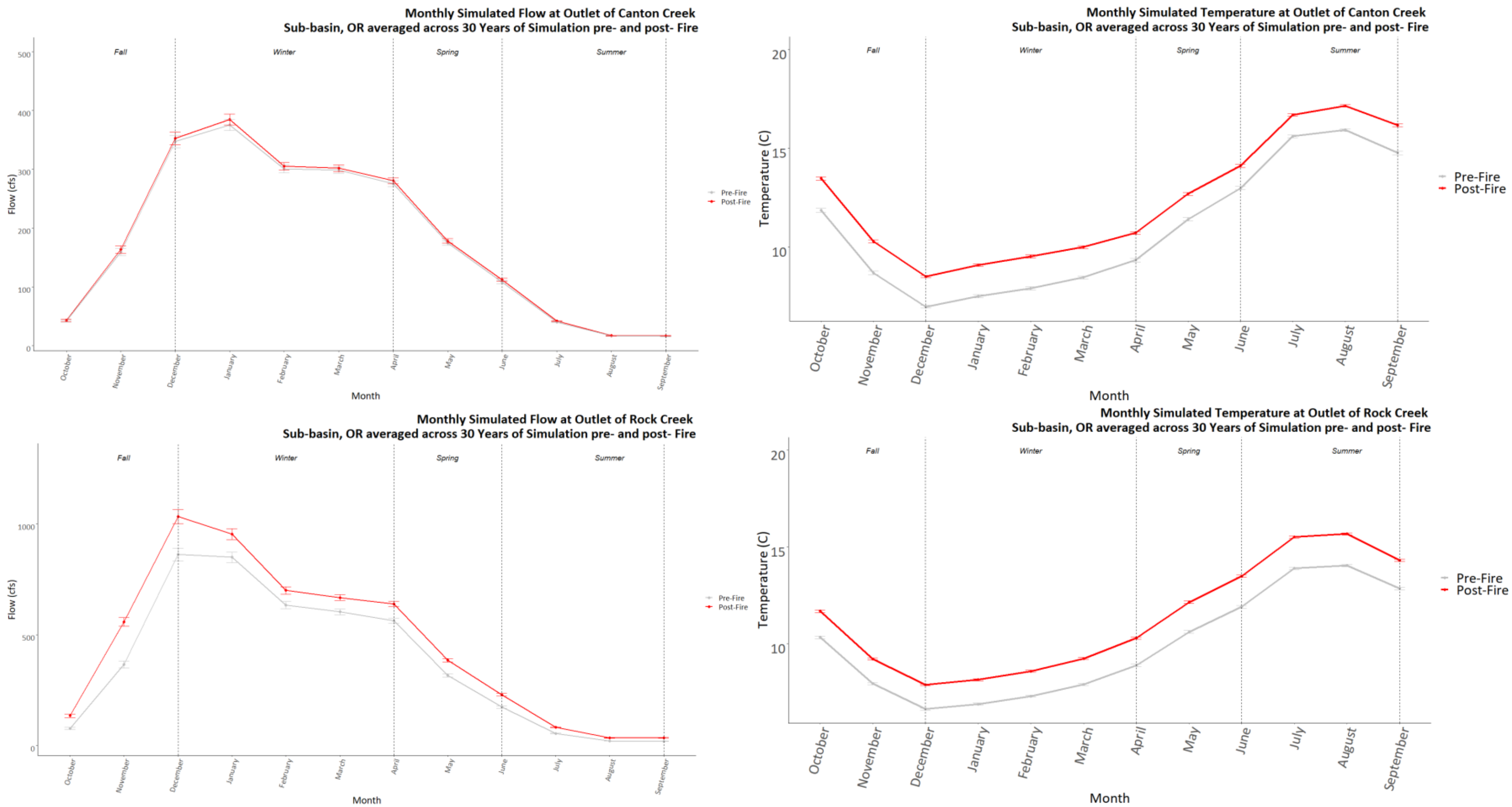


Figure 27. Pre- and post-fire flow and temperature simulations in Canton Creek and East Rock Creek.
Monthly pre- and post-fire simulation results with standard error for stream temperature (right) and flow (flow) at the outlets of Canton Creek (top) and East Rock Creek (bottom).

Future Marine Conditions

As discussed in the **Limiting Factors** section, marine survival is a major driver of abundance and productivity for anadromous salmonids, including North Umpqua summer steelhead. Steelhead marine survival is linked to complex physical and ecological processes occurring at spatial scales ranging from local estuarine and nearshore environments to large portions of the north Pacific Ocean (Friedland et al. 2014; Wilson et al. 2021). These processes are being affected by anthropogenic climate change and impacts are expected to continue or accelerate in the future (IPCC 2019). Changes in the ocean environment could affect marine growth and survival for anadromous salmonids through many pathways (Crozier et al. 2019; Wainwright and Weitkamp 2013). The following sub-sections outline changes expected to occur for several key metrics (*sea surface temperature*, *upwelling*, *ocean acidification*, and *sea level rise*) linked to steelhead marine habitat, based on currently available science.

Sea Surface Temperature. Steelhead marine survival and recruitment is generally correlated with patterns of sea surface temperature (SST) and large-scale climate indices such as NPGO (see **Current Population Status** section; Friedland et al. 2014; Scheuerell et al. 2020; Wilson et al. 2021). Under the Representative Concentration Pathway (RCP) 8.5 scenario (IPCC 2014; Schwalm et al. 2020), annual average sea surface temperature in the northeastern Pacific is projected to increase by 2.4–3.6°F by the end of century (Figure 28). The largest increases are expected to occur in the northern portion of the area. These changes could reduce overall high-seas habitat for steelhead (Abdul-Aziz et al. 2011), but population-scale effects will depend on marine distribution patterns that are poorly understood. Additionally, marine heatwaves have doubled in frequency since 1982 and are increasing in intensity. They are projected to further increase in frequency, duration, extent and intensity. Their frequency will be 20 times higher at 3.6°F warming, compared to pre-industrial levels. They would occur 50 times more often if emissions continue to increase strongly (IPCC 2019).

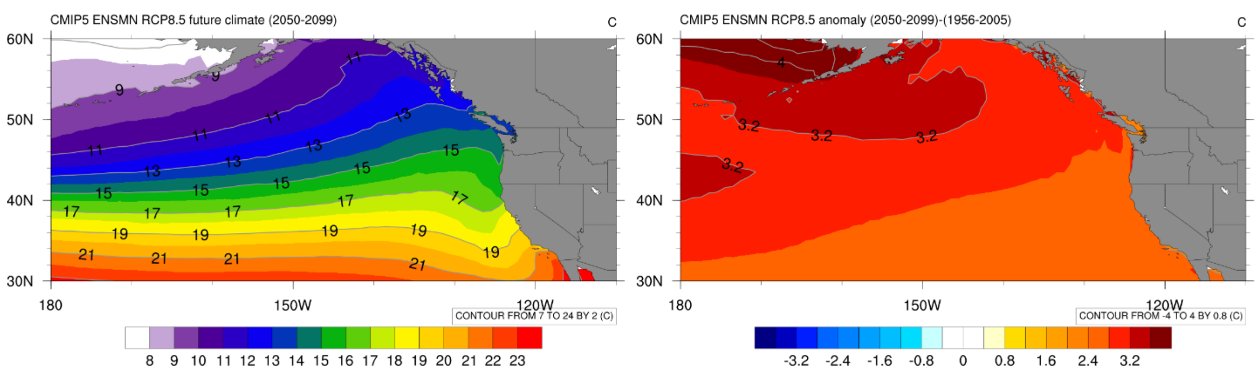


Figure 28. Maps showing a sea surface temperature (SST) projection and change from the recent reference period. Left Panel: Projected future SST (°C) for the period 2050-2099 based on RCP 8.5 scenario; Right Panel: Difference in mean SST between future (2050-99) and reference period (1956-2005). SST interpolated on a 1°x1° grid for the entire year. Figure downloaded from the Climate Change Web Portal of the NOAA Physical Sciences Laboratory (https://psl.noaa.gov/ipcc/ocn/).

Upwelling. At ocean entry, juvenile salmonid survival is linked to the occurrence and intensity of upwelling which drives the input and retention of cold, nutrient-rich waters to the euphotic zone. This, in turn increases primary productivity resulting in a lipid-rich food-web (Wells et al. 2016). The most recent models suggest that in the northern California Current System (CCS), upwelling will become more intense in the spring and less intense in the summer as a result of anthropogenic climate change (Rykaczewski et al. 2015). Changes in upwelling due to climate change will emerge primarily late in the second half of the century (Brady et al. 2017). There remains uncertainty in these predictions because the ensembles include relatively coarse-resolution global models from which it is difficult to resolve local dynamics in the CCS.

Ocean Acidification. Ocean pH levels have been declining as a result of uptake of carbon dioxide from the atmosphere (IPCC 2019). The California Current Large Marine Ecosystem (CCLME) is experiencing greater ocean acidification because of the combination of upwelling currents that transport dissolved inorganic carbon rich water from the deep ocean and high productivity of the shelf that increases potential for remineralization (Chan et al. 2017). Within the CCLME, the nearshore region (<10 km from shore) is most strongly affected by current, and likely future, acidification resulting in reduced abundance and increased corrosion in the shells of calcifying organisms (Feely et al. 2016). In offshore areas (>10 miles from shore) within the ocean feeding grounds of steelhead, surface pH is expected to decrease by 0.24-0.32 units by the end of the century (Figure 29). Data collected off the Oregon coast indicate that ocean pH is significantly lower within the nearshore area (<10 miles from shore) (pH 7.43) than the global mean (pH 8.1), and it is expected that this area will be more susceptible to acidification (Chan et al. 2017). Although direct impacts of pH have been shown for many taxa in the CCLME (Busch and McElhany 2016), there is considerable uncertainty in projecting future impacts.

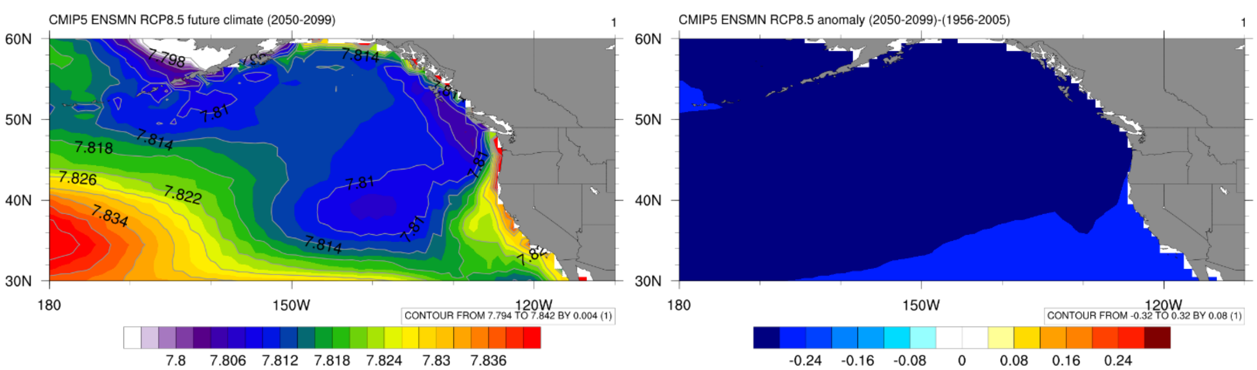


Figure 29. Maps showing an ocean acidity (pH) projection and change from the recent reference period. Left Panel: Projected future surface pH for the period 2050-2099 based on the RCP 8.5 scenario; Right Panel: Difference in mean surface pH between future (2050-99) and reference period (1956-2005). Surface pH interpolated on a 1°x1° grid for the entire year. Figure downloaded from the Climate Change Web Portal of the NOAA Physical Sciences Laboratory (https://psl.noaa.gov/ipcc/ocn/).

Sea Level Rise. Estuarine habitats will be affected by ongoing sea level rise, which is projected to continue past 2100 under all likely climate scenarios (IPCC 2019). Under RCP 8.5, the projected global mean sea level (GMSL) rise is 0.71 m (likely range: 0.51–0.92 m) for 2081–2100 (IPCC 2019). Effects of sea level rise on estuary habitat for juvenile salmonids are complex and depend on estuary topography and anthropogenic impacts that constrain tidal influence and habitat development (Flitcroft et al. 2013). Many estuaries have limited scope for migration and are thus expected to experience major shifts in estuarine habitat type with sea level rise (Thorne et al. 2018). On the Oregon coast, out-migrating steelhead appear to move quickly through estuaries but can experience significant mortality in these habitats (Romer et al. 2013). It is uncertain how changes in estuarine habitat due to sea level rise might affect predation risk for steelhead smolts in estuaries.

Summary: Future Outlook

We were somewhat limited in our ability to assess the vulnerability of summer steelhead to changing climate conditions. Most of the data leveraged in this analysis were collected for other purposes and were therefore not at the spatial or temporal resolutions needed to make precise estimates. There was limited continuous data representing two of the most influential environmental variables affecting distribution, survival, and resilience of this species: water temperature and stream flow.

Despite this, we were able to complete assessments utilizing sophisticated models. One of these climate change assessments indicated generally high probabilities of juvenile steelhead presence within the distribution of summer steelhead in the North Umpqua mainstem and tributaries of the Middle North Umpqua (e.g., Canton and Steamboat creeks). These tributaries have cool stream temperatures, deep pools, and high stream flow permanence. While future projections did show increases in stream temperature and decreases in spring precipitation, these watersheds will likely remain a stronghold for this species. These stronghold areas for juvenile rearing are likely to be resilient to climate-driven changes in temperatures and flow, though intense wildfires in these areas would further increase temperatures.

However, stream temperature data from multiple sources indicate very warm conditions throughout the mainstem Umpqua River and lower North Umpqua River during times when these areas are used as the primary migratory corridors for adult summer steelhead. These climate- or fire-driven increases to temperatures may become an increasingly important limiting factor through impacts to migrating adult summer steelhead. These conditions could also potentially affect steelhead smolts if they out-migrate late or encounter unusually warm conditions in the spring. Some cold-water patches are evident, but thermal conditions do not generally improve until cooler tributaries in the upper watersheds begin to contribute flow.

Additionally, we projected the effect of large and powerful fires on the landscape, which increased stream temperatures and flows in all simulations. These fires are expected to become more frequent in the future, with potential for dramatic effect on the available habitat for salmonids by rapidly increasing stream temperatures. Even though other future climate scenario modeling suggests a reduction in the availability of instream flows throughout much of the year, large fires are projected to increase instream flows due to the loss of terrestrial vegetation that would otherwise take up much of the water. However, this effect would be temporary, as vegetation would regrow. Additionally, the increase in flow that would result from a large fire would also be accompanied by negative effects to instream water quality (such as increased sedimentation and “flashiness” to the system). So, increased flows due to wildfires are not expected to mitigate the effects of a shifting climatic regime reducing instream flows, but rather exacerbate an already stressed system as the loss of shading will compound stream warming and the regrowth of vegetation will eventually reduce flows again, as middle-aged forests stands uptake more water (and reduce instream flows) than other ages.

While there is considerable uncertainty in future climate projections, it is clear that water temperatures in the Umpqua basin will continue to increase and will be a critical factor for all native species.

In the ocean, climate change is altering key physical processes and these effects are expected to continue or accelerate in the future (IPCC 2019). Changes in the ocean environment pose a major risk to Pacific salmon and steelhead, but assessing vulnerability to these effects is complicated by high uncertainty about marine ecosystem responses (Crozier et al. 2019). For North Umpqua summer steelhead, future projections suggest that ocean conditions are likely to become more challenging. Projections for increasing SST and more frequent marine heat waves, in particular, have relatively high confidence and are likely to impact marine survival and result in greater population variability based on our current understanding. Climate-driven changes in upwelling are less certain and intensification of upwelling could have positive effects on nearshore marine productivity (Wainwright and Weitkamp 2013). The effects of ocean acidification and sea level rise are much harder to predict.

CONCLUSION

The historic downturn in 2021 was caused by very poor marine conditions influencing survival, and likely exacerbated by extreme freshwater temperature conditions in 2021 that impacted returning adults that hold through the summer before spawning. Freshwater temperature and flow conditions were also very poor during the years when the 2021 returns were outmigrating as juveniles. Other limiting factors may also be at play, including, but not limited to, non-native fish predation in the lower Umpqua River. Although the local hatchery program was not found to have a negative effect on the naturally produced population, the management threshold established to limit hatchery influence on natural production has been exceeded (pHOS).

Based on the most current population data and modeling, North Umpqua summer steelhead have the intrinsic productivity needed to rebound if conditions allow. Going forward, ocean conditions have improved significantly since 2019. We expect a modest improvement in marine survival for the cohort returning in 2022, but those fish returning in 2023 and 2024 will have experienced very favorable ocean conditions and likely high marine survival, which is expected to offset some of the poor freshwater conditions those fish experienced. Conversely, freshwater conditions remain poor and fish rearing and outmigrating in 2022 may have lower survival, so we will continue to monitor freshwater and marine environmental conditions for this population.

The expectation of this population to remain viable over the long term is dependent on the assumption that marine, freshwater, and other conditions will continue to fluctuate within a range consistent with the magnitude and frequency of past conditions. Analyses into future conditions indicate that marine and freshwater conditions are changing. We are not able at this time to quantitatively analyze how changes in these multiple, complex, and interacting factors that vary over space and time in broadly different locations will affect the North Umpqua summer steelhead population, which has diverse life history characteristics that have contributed to its resiliency. However, qualitatively our analyses suggest that risk to viability will be highest when the population experiences the following conditions for a sustained period that encompasses at least 2-3 generations: very poor freshwater conditions during rearing and outmigration followed by very poor ocean conditions (i.e., extreme events) and then very poor freshwater conditions during migration and holding prior to spawning.

Continued research, monitoring, and analyses are warranted to understand the effects of these environmental changes on population status.

REFERENCES

Abatzoglou, J.T. and T.J. Brown. 2012. A comparison of statistical downscaling methods suited for wildfire applications. International Journal of Climatology 32:772-780.

Abatzoglou, J. T. 2013. Development of gridded surface meteorological data for ecological applications and modelling. International Journal of Climatology 33: 121–131.

Abdelnour, M. Stieglitz, F. Pan, and R. McKane. 2011. Catchment hydrological responses to forest harvest amount and spatial pattern. Water Resources Research 47:1-18.

Abdul-Aziz, O. I., N. J. Mantua, and K. W. Myers. 2011. Potential climate change impacts on thermal habitats of Pacific salmon (*Oncorhynchus* spp.) in the North Pacific Ocean and adjacent seas. Canadian Journal of Fisheries and Aquatic Sciences 68:1660-1680.

Anderson, D.M., D.P. Lischia, and D.W. Loomis. 1986. Fish Management Plan, North Umpqua River (below Soda Springs Dam). ODFW, Portland.

Araki, H., W.R. Ardren, E. Olsen, B. Cooper, and M.S. Blouin. 2007. Reproductive success of captive-bred steelhead trout in the wild: evaluation of three hatchery programs in the Hood River. Conservation Biology. 21:181-190.

Atcheson, M.E., K.W. Myers, and D.A. Beauchamp. 2011. Bioenergetic response by steelhead to variation in diet, thermal habitat, and climate in the North Pacific Ocean. Transactions of the American Fisheries Society 141: 1081-1096.

Auth, T. D., E. A. Daly, R. D. Brodeur, and J. L. Fisher. 2018. Phenological and distributional shifts in ichthyoplankton associated with recent warming in the northeast Pacific Ocean. Global Change Biology 24: 259– 272.

Bates, D., Maechler, M., Bolker, B., Walker, S., Christensen, R.H.B., Singmann, H., Dai, B., Scheipl, F. and Grothendieck, G., 2011. Package 'lme4'. *Linear mixed-effects models using S4 classes. R package version*, *1*(6).

Beamish, R.J. and C. Mahnken. 2001. A critical size and period hypothesis to explain natural regulation of salmon abundance and the linkage to climate and climate change. Progress In Oceanography 49: 423–437.

Beeman, J. W., and A. G. Maule. 2006. Migration depths of juvenile Chinook Salmon and steelhead relative to total dissolved gas supersaturation in a Columbia River reservoir. Transactions of the American Fisheries Society 135: 584– 594.

Benda, L., Miller, D., Andras, K., Bigelow, P., Reeves, G. and Michael, D., 2007. NetMap: a new tool in support of watershed science and resource management. Forest Science 53:206-219.

Berntson, E.A., R.W. Carmichael, M.W. Flesher, E.J. Ward, and P. Moran. 2011. Diminished reproductive success of steelhead from a hatchery supplementation program (Little

Sheep Creek, Imnaha Basin, Oregon). Transactions of the American Fisheries Society 140:685-698.

Brady, R. X., M. A. Alexander, N. S. Lovenduski, and R. R. Rykaczewski. 2017. Emergent anthropogenic trends in California Current upwelling. Geophysical Research Letters 44: 5044-5052.

Brodeur, R.D., Fisher, J.P., Emmett, R.L., Morgan, C.A. and Casillas, E., 2005. Species composition and community structure of pelagic nekton off Oregon and Washington under variable oceanographic conditions. Marine Ecology Progress Series, 298, pp.41-57.

Brown, R. F., B. E. Wright, S. D. Riemer, and J. Laake. 2005a. Trends in abundance and current status of harbor seals in Oregon: 1977-2003. Submitted to Marine Mammal Science, July 2004. Draft manuscript available from ODFW Marine Mammal Research Program, 7118 NE Vandenberg Ave., Corvallis, OR, 97330.

Brown, S.B., J.D. Fitzsimmons, D.C. Honeyfield, and D.E. Tillitt. 2005b. Implications of thiamine deficiency in Great Lakes salmonines. Journal of Aquatic Animal Health 17:113-124.

Burgner, R. L., J. T. Light, L. Margolis, T. Okazaki, and S. Ito. 1992. Distribution and origins of steelhead trout (*Oncorhynchus mykiss*) in offshore waters of the North Pacific Ocean. International North Pacific Fisheries Commission Bulletin 51.

Burke B.J., W.T. Peterson, B.R. Beckman, C. Morgan, E.A. Daly, and M. Litz. 2013. Multivariate models of adult Pacific salmon returns. PLoS ONE 8(1): e54134. https://doi.org/10.1371/journal.pone.0054134

Busch, D. S., and P. McElhany. 2016. Estimates of the direct effect of seawater PH on the survival rate of species groups in the California Current ecosystem. PLoS ONE 11

Carter, H. R., A. L. Sowls, M. S. Rodway, U. W. Wilson, R. W. Lowe, G. J. McChesney, F. Gress, and D. W. Anderson. 1995. Population size, trends, and conservation problems of the double-crested cormorant on the Pacific coast of North America. Colonial Waterbirds 18:189–215.

Chan, F., J. A. Barth, C. A. Blanchette, R. H. Byrne, F. Chavez, O. Cheriton, R. A. Feely, et al. 2017. Persistent spatial structuring of coastal ocean acidification in the California Current system. Scientific Reports 7: 1-7.

Collins, C.S. 1971. North Umpqua Case Study: An Introduction. Pp. 225-226 In Krygier, J.T. and J.D. Hall (eds.). Proceedings of a Symposium - Forest Land Uses and Stream Environment. October 19-21, 1970. Oregon State University, Corvallis.

Collis, K., D. D. Roby, D. P. Craig, B. A. Ryan, and R. D. Ledgerwood. 2001. Colonial waterbird predation on juvenile salmonids tagged with passive integrated transponders in the

Columbia River estuary: vulnerability of different salmonid species, stocks, and rearing types. Transactions of the American Fisheries Society 130:385–396.

Courchamp, F., L. Berec, and J. Gascoigne. 2008. Alee effects in ecology and conservation. Oxford University Press, Oxford, UK.

Crozier, L. G., M. M. McClure, T. Beechie, S. J. Bograd, D. A. Boughton, M. Carr, T. D. Cooney, et al. 2019. Climate vulnerability assessment for Pacific salmon and Steelhead in the California Current large marine ecosystem. PLoS ONE 14.

Cuthbert, F. J., and L. R. Wires. 1999. Caspian tern (Hydroprogne caspia), version 2.0. In A. F. Poole and F. B. Gill, editors. The birds of North America. Cornell Lab of Ornithology, Ithaca, New York. Available: https://doi.org/10.2173/bna.403. (August 2019).

Daily, K. 1990. Umpqua River Smallmouth Bass Investigation. ODFW, Portland.

Di Lorenzo E., Schneider N., Cobb K. M., Chhak, K, Franks P. J. S., Miller A. J., McWilliams J. C., Bograd S. J., Arango H., Curchister E., Powell T. M. and P. Rivere, 2008. North Pacific Gyre Oscillation links ocean climate and ecosystem change. Geophysical Research Letters. 35: L08607, doi:10.1029/2007GL032838.

EPA. 2022. https://www.epa.gov/climate-research.Date accessed February 24, 2022.

Erhardt, J.M, K.F. Tiffan, and W.P. Connor. 2018. Juvenile Chinook Salmon mortality in a Snake River reservoir: Smallmouth Bass predation revisited. Transactions of the American Fisheries Society 147:316-328.

Evans, A. F., N. J. Hostetter, K. Collis, D. D. Roby, D. E. Lyons, B. P. Sandford, R. D. Ledgerwood, and S. Sebring. 2012. Systemwide evaluation of avian predation on juvenile salmonids from the Columbia River based on recoveries of passive integrated transponder tags. Transactions of the American Fisheries Society 141: 975– 989.

Evans, A. F., Q. Payton, A. Turecek, B. M. Cramer, K. Collis, D. D. Roby, P. J. Loschl, L. Sullivan, J. Skalski, M. Weiland, and C. Dotson. 2016. Avian predation on juvenile salmonids: spatial and temporal analysis based on acoustic and passive integrated transponder tags. Transactions of the American Fisheries Society 145: 860– 877.

Evans, A.F., Q. Payton, B.M. Cramer, K. Collins, N.J. Hostetter, D.D. Roby, and C. Dotson. 2019. Cumulative effects of avian predation on Upper Columbia River Steelhead. Transactions of the American Fisheries Society 148: 896-913.

Falcy, M.R., and E. Suring. 2018. Detecting the effects of management regime shifts in dynamic environments using multi-population state-space models. Biological Conservation 221: 34-43.

Falke, J.A., R.L. Flitcroft, J.B. Dunham, K.M. McNyset, P.F. Hessburg, G.H. Reeves, and C.T. Marshall. 2015. Climate change and vulnerability of bull trout (*Salvelinus confluentus*) in a fire-prone landscape. Canadian Journal of Fisheries and Aquatic Sciences 72:304-318.

Feely, R. A., S. R. Alin, B. Carter, N. Bednaršek, B. Hales, F. Chan, T. M. Hill, et al. 2016. Chemical and biological impacts of ocean acidification along the West Coast of North America. Estuarine, Coastal and Shelf Science 183: 260-270.

Fish Commission of Oregon (FCO) and Oregon State Game Commission (OSGC). 1946. The Umpqua River Study. Joint Report of the Fish Commission of Oregon and Oregon State Game Commission, Portland, Oregon.

Fitzsimmons, J.D., B. Williston, P. Amcoff, L. Balk, C. Pecor, H.G. Ketola, J.P. Hinterkopf, and D.C. Honeyfield. 2005. The effect of thiamine injection on upstream migration, survival, and thiamine status of putative thiamine-deficient coho salmon. Journal of Aquatic Animal Health 17:48-58.

Flitcroft, R, K. Burnett and K. Christensen. 2013. A simple model that identifies potential effects of sea-level rise on estuarine and estuary-ecotone habitat locations for salmonids in Oregon, USA. Environmental Management 52:196-208.

Flitcroft, R.L., J.A. Falke, G.H. Reeves, P.F. Hessburg, K.M. McNyset, and L.E. Benda. 2016. Wildfire may increase habitat quality for spring Chinook salmon in the Wenatchee River subbasin, WA, USA. Forest Ecology and Management 359:126-140.

FR (U.S. Office of the Federal Register). 1998. Endangered and Threatened Species: Threatened Status for Two ESUs of Steelhead in Washington, Oregon, and California. Federal Register 63:53 (March 19, 1998):13347-13371.

Freschette, D., A. K. Osterback, S. A. Hayes, M. H. Bond, J. W. Moore, S. A. Shaffer, and J. T. Harvey. 2012. Assessing avian predation on juvenile salmonids using passive integrated transponder tag recoveries and mark–recapture methods. North American Journal of Fisheries Management 32: 1237– 1250.

Friedland, K. D., B. R. Ward, D. W. Welch, and S. A. Hayes. 2014. Postsmolt growth and thermal regime define the marine survival of steelhead from the Keogh River, British Columbia. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 6:1-11.

Fritts, A.L., and T.N. Pearsons. 2004. Smallmouth bass predation on hatchery and wild salmonids in the Yakima River, Washington. Transactions of the American Fisheries Society 133:880-895.

Futia, M.H., and J. Rinchard. 2019. Evaluation of adult and offspring thiamine deficiency in salmonine species from Lake Ontario. Journal of Great Lakes Research 45: 811-820.

Gilpin, M.E., and M.E. Soulé. 1984. Minimum viable populations: processes of species extinction. Pp. 19-34 In Soulé (Ed.) Conservation Biology: the science of scarcity and diversity. Sinauer, Sunderland, MA.

Haddon, M. 2011. Modelling and Quantitative Methods in Fisheries, 2nd Ed. Chapman & Hall/CRC Press, Boca Raton, FL.

Halama, J.J., R.E. Kennedy, J.J. Graham, R.B. McKane, B.L. Barnhart, K.S. Djang, P.B., Pettus, A.F. Brookes, P.C. Wingo. Penumbra: A spatially distributed, mechanistic model for simulating ground-level incident solar energy across heterogeneous landscapes. PLoS ONE 13(12): e0206439.

Hartig, F. and Hartig, M.F., 2017. Package 'DHARMa'. *R package*.

Hicks, B.J., R.L. Beschta, and R. Harr. 1991. Long-term changes in streamflow following logging in western Oregon and associated fisheries implications. Water Resources Bulletin 27: 217–226.

Hilborn, R. and C.J. Walters (Eds). 2003. Quantitative Fisheries Stock Assessment: choice, dynamics & uncertainty. Kluwer Academic Publishers, Norwell, MA.

Holtby, L.B., B.C. Andersen and R.K. Kadowaki. 1990. Importance of Smolt Size and Early Ocean Growth to Interannual Variability in Marine Survival of Coho Salmon (Oncorhynchus kisutch). Canadian Journal of Fisheries and Aquatic Sciences 47: 2181–2194.

Honeyfield, D.C., J.M. Murphy, K.G. Howard, W.W. Strasburger, and A.C. Matz. 2016. An exploratory assessment of thiamine status in western Alaska Chinook Salmon (*Oncorhynchus tshawytscha*). North Pacific Anadromous Fish Commission Bulletin No. 6:21-31.

Hooton, R. S. 1987. Catch and release as a management strategy for steelhead in British Columbia. Pages 143– 156 In R. A. Barnhart and T. D. Roelofs, Eds. Proceedings of catch and release fishing—a decade of experience. Humboldt State University, Arcata, California.

Hostetter, N. J., A. F. Evans, D. D. Roby, and K. Collis. 2012. Susceptibility of juvenile steelhead to avian predation: the influence of individual fish characteristics and river conditions. Transactions of the American Fisheries Society 141: 1586– 1599.

IPCC. 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.

IPCC. 2019. IPCC Special Report on the Ocean and Cryosphere in a Changing Climate [H.-O. Pörtner, D.C. Roberts, V. Masson-Delmotte, P. Zhai, M. Tignor, E. Poloczanska, K. Mintenbeck, A. Alegría, M. Nicolai, A. Okem, J. Petzold, B. Rama, N.M. Weyer (eds.)].

Isaak, D.J., Wenger, S.J., Peterson, E.E., Ver Hoef, J.M., Nagel, D.E., Luce, C.H., Hostetler, S.W., Dunham, J.B., Roper, B.B., Wollrab, S.P. and Chandler, G.L. 2017. The NorWeST summer stream temperature model and scenarios for the western US: A crowd-sourced database and new geospatial tools foster a user community and predict broad climate warming of rivers and streams. Water Resources Research 53: 9181-9205.

Jaeger, K.L., R. Sando, R.R. McShane, J.B. Dunham, D.P. Hockman-Wert, K.E. Kaiser, K. Hafen, J.C. Risley, and K.W. Blasch. 2019. Probability of Streamflow Permanence Model (PROSPER): A spatially continuous model of annual streamflow permanence throughout the Pacific Northwest. *Journal of Hydrology X*, 2:100005.

Jamieson, I.G., and F.W. Allendorf. 2012. How does the 50/500 rule apply to MVPs? Trends in Ecology and Evolution 27:578-584.

Jarnevich, C.S., Stohlgren, T.J., Kumar, S., Morisette, J.T. and Holcombe, T.R., 2015. Caveats for correlative species distribution modeling. *Ecological Informatics*, *29*, pp.6-15.

Johnson, J.H., A.A. Nigro, and R. Temple. 1992. Evaluating enhancement of striped bass in the context of potential predation on anadromous salmonids in Coos Bay, Oregon. North American Journal of Fisheries Management 12:103-108.

Kennedy, R.E., Z. Yang, and W.B. Cohen. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 1. LandTrendr - temporal segmentation algorithms. Remote Sensing of Environment 114(12): 2897–2910.

Kennedy, R.E., Z. Yang, N. Gorelick, J. Braaten, L. Cavalcante, W.B. Cohen, and S. Healey. 2018. Implementation of the LandTrendr algorithm on Google Earth Engine. Remote Sensing of Environment *10*, 691.

Kilduff, D.P., E. Di Lorenzo, L.W. Botsford, and S.L.H. Teo. 2015. Changing central Pacific El Niños reduce stability of North American salmon survival rates. Proceedings of the National Academy of Sciences of the United States of America 112: 10962-10966.

Knoben, W.J., J.E. Freer, and R.A. Woods. 2019. Technical note: Inherent benchmark or not? Comparing Nash-Sutcliffe and Kling-Gupta efficiency scores. Hydrology and Earth System Sciences 23(10):4323-4331.

Leonard, J.M., Magaña, H.A., Bangert, R.K. et al. 2017. Fire and Floods: The recovery of headwater stream systems following high-severity wildfire. Fire Ecology 13:62–84.

Litzow, M.A., Ciannelli, L, Puerta, P., Wettstein, J.J., Rykaczewski, R.R. and M. Opiekun. 2018. Non-stationary climate–salmon relationships in the Gulf of Alaska. Proceedings of the Royal Society B Biological Sciences 285: 20181855.

Litzow, M.A., Hunsicker, M.E., Bond, N.A., Burke, B.J., Cunningham, C.J., Gosselin, J.L., Norton, E.L., Ward, E.J. and S.G. Zador. 2020. The changing physical and ecological meanings of North Pacific Ocean climate indices. Proceedings of the National Academy of Sciences of the United States of America 117:7665-7671.

Loomis, D., J. Raasch, R. Thompson, and B. Ryan. 2003. A Radio Telemetry Study of Steelhead in the North Umpqua River Basin, 1998-2001. ODFW, Roseburg.

Lyons, D.E. 2010. Bioenergetics-based predator-prey relationships between piscivorous birds and juvenile salmonids in the Columbia River estuary. Unpubl. Ph.D. dissertation, Oregon State University, Corvallis, Oregon. Available on-line at www.birdresearchnw.org

Marshall, S.L. 2001. Estimates of the Incidental Mortality of Wild Steelhead Caught and Released by Idaho Anglers and Recommendations for Establishing Annual Take Limits Under Section 10(a)(1)(B) of the Endangered Species Act. Idaho Department of Fish and Game, Boise, Idaho.

McCormick, J.L., M.E. Dobos, B.J. Bowersox and T. Copeland. 2020. Evaluation of management strategies for an incidental catch-and-release steelhead fishery. North American Journal of Fisheries Management, 41(2): 498-512.

McKane, R., A. Brookes, K., Djang, M. Stieglitz, A. Abdelnour, F. Pan, J. Halama, P. Pettus, and D. Phillips. 2014. Velma Version 2.0: User Manual and Technical Documentation. Environmental Protection Agency Office of Research and Development National Health and Environmental Effects Research Laboratory, Corvallis, OR, USA. Document control number L-PESD-30840-QP-1-2.

McGie, A.M. 1994. Stock-Recruitment in Summer-Run Steelhead of the North Umpqua River, Oregon. Information Report 94-5. ODFW, Portland.

Moore ME, and BA Berejikian, 2017. Population, habitat, and marine location effects on early marine survival and behavior of Puget Sound steelhead smolts. Ecosphere 8: e01834.

Moore, R.B., McKay, L.D., Rea, A.H., Bondelid, T.R., Price, C.V., Dewald, T.G., and Johnston, C.M., 2019, User's guide for the national hydrography dataset plus (NHDPlus) high resolution: U.S. Geological Survey Open-File Report 2019–1096, 66 p.

Moriasi, D.N., J.G. Arnold, M.W. Van Liew, R.L. Bingner, R.D. Harmel, and T.L. Veith. 2007. Model evaluation guidelines for systemic quantification of accuracy in watershed simulations. American Society of Agricultural and Biological Engineers. 50:885-900.

Nelson, T. C., M. L. Rosenau, and N. T. Johnston. 2005. Behaviour and survival of wild and hatchery-origin winter steelhead spawners caught and released in a recreational fishery. North American Journal of Fisheries Management, 25: 931– 943.

Nobriga, M.L., and F. Feyrer. 2008. Diet composition in San Francisco Estuary striped bass: does trophic adaptability have its limits? Environmental Biology of Fishes 83:495-503.

Ohmann, J.L., M.J. Gregory, H.M. Roberts, B.C. Warren, R.E. Kennedy, and Z. Yang. 2012. Mapping change of older forest with nearest-neighbor imputation and Landsat time-series. Forest Ecology and Management 272:13-25.

Oregon State University Libraries & Press and Institute for Natural Resources. 2021. Oregon Explorer. Accessed on [February 26, 2020]. URL: http://oregonexplorer.info/

ODFW (Oregon Department of Fish and Wildlife). 1990. Oregon Sport Fishing Regulations, 1990 Edition. ODFW, Salem.

ODFW. 1999. 1999 Oregon Sport Fishing Regulations. ODFW, Salem.

ODFW. 2005. Oregon Native Fish Status Report – Volume II. ODFW Fish Division, Salem.

ODFW. 2007. Biology, behavior, and resources of resident and anadromous fish in the lower Willamette River. Information Report 2007-03. ODFW, Salem.

ODFW. 2008. Winchester Dam fish Counts. ODFW, Roseburg.

ODFW. 2009. Rock Creek Hatchery Operations Plan. ODFW, Salem.

ODFW. 2010. Natural Resources Information Management Program (NRIMP) URL: http://nrimp.dfw.state.or.us/nrimp/default.aspx?p=259

ODFW. 2014. Coastal Multi-Species Conservation and Management Plan. ODFW, Salem.

ODFW. 2016a. Oregon Conservation Strategy. ODFW, Salem.

ODFW. 2016b. Hatchery and Genetic Management Plan (HGMP), Umpqua River Basin Summer Steelhead Program. ODFW, Salem.

ODFW. 2020a. Rock Creek Hatchery, Program Management Plan 2020. ODFW, Salem.

ODFW. 2020b. Coastal Multi-Species Conservation and Management Plan Wild Fish Monitoring Summaries, 2014-2019. ODFW, Salem.

ODFW. 2021a. Coastal Multi-species Conservation and Management Plan: Wild fish monitoring summaries 2014-2020. ODFW, Salem.

ODFW. 2021b. ODFW Sensitive Species List 2021. ODFW, Salem.

OSGC (Oregon State Game Commission). 1936. Official Synopsis of Oregon Fishing Laws, Effective April 5, 1936. Oregon State Game Commission, Portland, Oregon.

OSGC. 1952. Annual Report of the Oregon State Game Commission, Fishery Division – 1951. OSGC, Portland.

OSGC. 1952. Oregon Angling Regulations 1952. Oregon State Game Commission, Portland, Oregon.

OSGC. 1956. Annual Report of the Oregon State Game Commission, Fishery Division – 1955. OSGC, Portland.

OSGC. 1962. Annual Report of the Oregon State Game Commission, Fishery Division – 1961. OSGC, Portland.

OSGC. 1963. Annual Report of the Oregon State Game Commission, Fishery Division – 1962. OSGC, Portland.

Parks, N.B. 1978. The Pacific Northwest commercial fishery for striped bass, 1922-74. Marine Fisheries Review 40:18-20.

PFMC (Pacific Fishery Management Council). 2000. Identification and description of essential fish habitat, adverse impacts, and recommended conservation measures for salmon. Appendix A to Amendment 14, Pacific Coast Salmon Fishery Management Plan. Available at http://www.pcouncil.org/wp-content/uploads/99efh2.pdf

PFMC. 2013. Pacific Coast Fishery Ecosystem Plan for the U.S. Portion of the California Current Large Marine Ecosystem. Available at http://www.pcouncil.org/ecosystem-based-management/fep/

PFMC. 2014. Appendix A to the Pacific Coast Salmon Fishery Management Plan as modified by Amendment 18 to the Pacific Coast Salmon Plan: Identification and Description of Essential Fish Habitat, Adverse Impacts, and Recommended Conservation Measures for Salmon. Available at: https://www.pcouncil.org/documents/2019/08/salmon-efh-appendix-a.pdf/

PFMC. 2021. Pacific Coast Salmon Fishery Management Plan for Commercial and Recreational Salmon Fisheries off the Coasts of Washington, Oregon, and California as Revised through Amendment 21 (Effective September 2021). Available at: https://www.pcouncil.org/documents/2016/03/salmon-fmp-through-amendment-20.pdf/

Pollet, I. L., D. Shutler, J. W. Chardine, and J. P. Ryder. 2012. Ring-billed gull (*Larus delawarensis*), version 2.0. In A. F. Poole, editor. The birds of North America. Cornell Lab of Ornithology, Ithaca, New York. Available: https://doi.org/10.2173/bna.33. (August 2019).

PRISM Climate Group, Oregon State University, https://prism.oregonstate.edu, data created 4 Feb 2014, accessed 16 Jan 2022.

Quinn, T. P. 2005. The behavior and ecology of Pacific salmon and trout. University of Washington Press, Seattle.

Ricker, W.E. 1954. Stock and recruitment. Journal of the Fisheries Research Board of Canada 11: 559-623.

Rieman, B.E., R.C. Beamsderfer, S. Vigg, and T.P. Poe. 1991. Estimated loss of juvenile salmonids to predation by northern squawfish, walleyes, and smallmouth bass in John Day Reservoir, Columbia River. Transactions of the American Fisheries Society 120:448-458.

River Management Joint Operating Committee (RMJOC): Bonneville Power Administration, United States Army Corps of Engineers, United States Bureau of Reclamation. 2018. Climate and Hydrology Datasets for RMJOC Long-Term Planning Studies: Second Edition (RMJOC-II) Part I: Hydroclimate Projections and Analyses. page 36.

Romer, J. D., C. A. Leblanc, S. Clements, J. A. Ferguson, M. L. Kent, D. Noakes, and C. B. Schreck. 2013. Survival and behavior of juvenile steelhead trout (*Oncorhynchus mykiss*) in two estuaries in Oregon, USA. Environmental Biology of Fishes 96:849-863.

Ruggerone, G. T., T. P. Quinn, I. A. McGregor, and T. S. Wilkinson. 1990. Horizontal and vertical movement of adult steelhead trout, Oncorhynchus mykiss, in Dean and Fisher channels, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 47:1963–1969.

Rykaczewski, R. R., J. P. Dunne, W. J. Sydeman, M. Garcia-Reyes, B. A. Black and S. J. Bograd. 2015. Poleward displacement of coastal upwelling-favorable winds in the ocean's eastern boundary currents through the 21st century. Geophysical Research Letters 42: 6424-6431.

Scheuerell, M. D., C. P. Ruff, J. H. Anderson, and E. M. Beamer. 2021. An integrated population model for estimating the relative effects of natural and anthropogenic factors on a threatened population of steelhead trout. Journal of Applied Ecology 58: 114-124.

Schwalm, C.R., S. Glendon and P.B. Duffy. 2020. RCP8.5 tracks cumulative CO_2 emissions. Proceedings of the National Academy of Science of the USA 117: 19656–19657.

Schwing, F.B., D.M. Palacios, and S.J. Bograd. 2005. El Niño impacts on the California Current ecosystems. U.S. CLIVAR Newsletter, Vol. 3, No. 2, 5-8.

Siirila-Woodburn, E.R., Rhoades, A.M., Hatchett, B.J. *et al.* 2021. A low-to-no snow future and its impacts on water resources in the western United States. *Nat Rev Earth Environ* **2,** 800–819.

Sobocinski, K.L., N.W. Kendall, C.M. Greene, and M.W. Schmidt. 2020. Ecosystem indicators of marine survival in Puget Sound steelhead trout. Progress in Oceanography 188:102419.

Summers, J., and K. Daily. 2001. Willamette River bass diet study, spring 2000. ODFW, Portland.

Tabor, R. A., R. S. Shively, T. P. Poe. 1993. Predation on juvenile salmonids by smallmouth bass and northern squawfish in the Columbia River near Richland, WA. North American Journal of Fisheries Management 13:831-838.

Thomas, A.C., B.W. Nelson, M.M. Lance, B.E. Deagle, and A.W. Trites, 2017. Harbour seals target juvenile salmon of conservation concern. Canadian Journal of Fisheries and Aquatic Sciences 74: 907−921.

Thorne, K., G. MacDonald, G. Guntenspergen, R. Ambrose, K. Buffington, B. Dugger, C. Freeman, C. Janousek, L. Brown, J. Rosencranz, J. Holmquist, J. Smol, K. Hargan and J. Takekawa. 2018. US Pacific coastal wetland resilience and vulnerability to sea-level rise. Science Advances 4:eaao3270.

Twardek, W.M., T.O. Gagne, L.K. Elmer, S.J. Cooke, M.C. Beere and A.J. Danylchuk. 2018. Consequences of catch-and-release angling on the physiology, behaviour and survival of wild steelhead *Oncorhynchus mykiss* in the Bulkey River, British Columbia. Fisheries Research, 206(2018): 235-246.

Wainwright, T. C., and L. A. Weitkamp. 2013. Effects of climate change on Oregon Coast Coho Salmon: Habitat and life-cycle interactions. Northwest Science 87: 219-242.

Walter, J.F., and H.M. Austin. 2003. Diet composition of large striped bass (*Morone saxatilis*) in Chesapeake Bay. Fishery Bulletin 101:411-423.

Welch, D.W., M.H. Futia, J. Rinchard, A.K. Teffer, K.M. Miller, S.G. Hinch, D.C. Honeyfield. 2018. Thiamine Levels in Muscle and Eggs of Adult Pacific Salmon from the Fraser River, British Columbia. Journal of Aquatic Animal Health 30:191-200.

Wells, B.K., C.B. Grimes, J.G. Sneva, S. McPherson, J.B. Waldvogel. 2008. Relationships between oceanic conditions and growth of Chinook salmon (Oncorhynchus tshawytscha) from California, Washington, and Alaska, USA. Fisheries Oceanography 17:101–125.

Wells, B. K., J. A. Santora, I. D. Schroeder, N. Mantua, W. J. Sydeman, D. D. Huff, and J. C. Field. 2016. Marine ecosystem perspectives on Chinook salmon recruitment: a synthesis of empirical and modeling studies from a California upwelling system. Marine Ecology Progress Series 552: 271-284.

Wenger, S.J., C.H. Luce, A.F. Hamlet, D.J. Isaak, and H.M Neville. 2010. Macroscale hydrologic modeling of ecologically relevant flow metrics. Water Resources Research. 46: W09513. doi:10.1029/2009WR008839.

Wenger, S.J., Som, N.A., Dauwalter, D.C., Isaak, D.J., Neville, H.M., Luce, C.H., Dunham, J.B., Young, M.K., Fausch, K.D. and Rieman, B.E., 2013. Probabilistic accounting of uncertainty in forecasts of species distributions under climate change. Global Change Biology, 19:3343-3354.

Wilson, S. M., T. W. Buehrens, J. L. Fisher, K. L. Wilson, and J. W. Moore. 2021. Phenological mismatch, carryover effects, and marine survival in a wild steelhead trout *Oncorhynchus mykiss* population. Progress in Oceanography 193 (2021), Article 102533.

Winkler, D. W. 1996. California gull (*Larus californicus*), version 2.0. In A. F. Poole and F. B. Gill, editors. The birds of North America. Cornell Lab of Ornithology, Ithaca, New York. Available: https://doi.org/10.2173/bna.259. (August 2019).

Wydoski, R.S., and R.R. Whitney. 2003. Inland Fishes of Washington, 2nd edition. American Fisheries Society, Bethesda, MD, and University of Washington Press, Seattle and London.

APPENDIX 1 – SUPPLEMENTAL TABLES

Table S1. AICc values for Ricker (RK), Beverton-Holt (BH0), Logistic Hockey Stick (LHS) and density independent (aS) stock recruitment models for the North Umpqua summer steelhead population, 1992-2014.

	AICc	ΔAICc	wtAICc
RK	6.763548	0.0000000	0.4854707
BH0	7.000490	0.2369423	0.4312327
LHS	10.769842	4.0062936	0.0654949
aS	13.375198	6.6116498	0.0178017

Table S2. AICc values for Beverton-Holt stock recruitment models with and without covariates including MWMT and NPGO both singly and combined for the North Umpqua summer steelhead population, 1992-2014.

	AICc	ΔAICc	wtAICc
BH.NPGO	-5.877144	0.000000	0.6514098
BH5.NPGO.MWMT	-4.521083	1.356061	0.3306659
BH.MWMT	1.428502	7.305646	0.0168831
BH	7.000490	12.877634	0.0010411

Table S3. Number and proportion of naturally produced summer steelhead spawning in North Umpqua sub-basins as determined by radio telemetry. Fish were tagged at Winchester Dam in 1998, 1999, and 2000. Ninety-two naturally produced summer steelhead tagged during the Winchester Dam summer counting period (May 1 – November 30) were assigned as spawners to one of the five spawning basins. Data are from Loomis et al. (2003).

	Number of Tagged Naturally Produced Summer Steelhead Tracked to Spawning Subbasin					
Month Tagged	Lower North Umpqua	Rock Creek	Little River	Upper North Umpqua	Steamboat Creek	Total
May	2	0	0	0	0	2
June	3	1	0	5	16	25
July	0	4	0	5	17	26
August	0	2	2	3	9	16
September	0	2	2	3	3	10
October	3	1	0	2	2	8
November	1	0	1	2	1	5
All Months	9	10	5	20	48	92
Proportion of Total	0.098	0.109	0.054	0.217	0.522	-

Table S4. Number and proportion of hatchery summer steelhead spawning in North Umpqua sub-basins as determined by radio telemetry. Fish were tagged at Winchester Dam in 1998, 1999, and 2000. Twenty hatchery summer steelhead tagged during the Winchester Dam summer counting period (May 1 – November 30) were assigned as spawners to one of the five spawning basins. Data are from Loomis et al. (2003).

	Number of Tagged Hatchery Summer Steelhead Tracked to Spawning Subbasin					
Month Tagged	Lower North Umpqua	Rock Creek	Little River	Upper North Umpqua	Steamboat Creek	Total
May	1	0	0	0	0	1
June	0	3	0	2	0	5
July	0[a]	7	0	0	0	8
August	0	3	0	0	0	3
September	0	1	0	0	0	1
October	1	2	0	0	0	3
November	0	0	0	0	0	0
All Months	2	16	0	2	0	20
Proportion of Total	0.100	0.800	0.000	0.100	0.000	-

[a]Loomis et al. (2003) note that one hatchery fish observed in the lower North Umpqua in July was likely lost to harvest rather than settled on a spawning location.

Table S5. Percent of adult summer steelhead holding in Rock Creek comprised of hatchery-origin fish. Estimates are based on observations of fin-clip status during summer snorkel surveys with data segregated by location relative to the Rock Creek diversion dam (above or below) and for all data pooled.

Run Year	Below Rock Creek Dam		Above Rock Creek Dam		Above & Below Rock Creek Dam	
	Annual	9-year Average	Annual	9-year Average	Annual	9-year Average
2008	50	-	20	-	24	-
2009	86	-	71	-	73	-
2010	83	-	41	-	44	-
2011	60	-	82	-	82	-
2012	100	-	72	-	74	-
2013	100	-	68	-	73	-
2014	100	-	77	-	78	-
2015	50	-	51	-	51	-
2016	98	81	77	62	82	65
2017	91	85	76	68	78	70
2018	84	85	48	66	64	69
2019	64	83	55	67	59	71
2020	81	85	86[a]	68[a]	85[a]	71[a]
2021	NS[b]	NS[b]	100	71	NA[c]	NA[c]

[a]Due to the Archie Creek Fire, surveys in 2020 were on a single date; surveys below the diversion dam covered a subset of survey area.
[b]NS = Not Surveyed; There were no snorkel surveys below the diversion dam in 2021.
[c]NA = Not Applicable; No snorkel surveys below the diversion dam in 2021; no full basin estimate.

Table S6. Flow and temperature metrics.

Metric	Description
Max_cfs	Maximum daily mean flow
Access Days	Number of days during the migration period when flow was above the threshold for access. The threshold for accessible flows is based on the relationship between flow at the Elkton gage and the arrival of coho.
CFM	Center of Flow Movement.
Day50	Day on which cumulative flow reaches 50% of the total for the water year.
Day75	Day on which cumulative flow reaches 75% of the total for the water year.
Dur25to75	Number of days between the day of 25% of annual flow and the day that it of 75% of annual flow.
SprFlow	Sum of flow during the spring season.
JuneT_Indx	Stream temperature index in June below the confluence of the Smith River and the Umpqua near Reedsport (COMID = 24526610).
Out Migration	Average of category designations for spring flow and June temperature index (1, 2 or 3 for low, medium, and high).
Min_cfs	Minimum daily mean flow
MWMT_Indx	Maximum weekly mean temperature index at the three most important holding pools in the North Umpqua (100 Fish, 5 Mile and Upper Bend).

Table S7. USGS gage and Mill Creek Siletz thermistor location details. Elevation recorded in feet (*) or feet above stage height (**).

Agency	Site Name	Site ID	Elevation	Drainage Area (mi^2)	Location
ODFW	Mill Creek, Siletz Trap	N/A	731.627*	13	Lat 45°48'43", Long 123°47'35"
USGS	N. Umpqua at Winchester	14319500	372.97 **	1,344	Lat 43°16'20", Long 123°24'40"
USGS	N. Umpqua, Soda Springs	14316460	1,720**	437	Lat 43°18'22", Long 122°30'42"
USGS	N. Umpqua River, Idleyld Park	14317450	780**	886	Lat 43°19'29", Long 122°59'55"
USGS	Rogue River at Raygold	14359000	1,121.78**	2,053	Lat 42°26'15", Long 122°59'10"
USGS	Rogue River below Prospect	14330000	1,964.56**	379	Lat 42°43'47", Long 122°30'54"
USGS	Rogue River at Agness	14372300	113.81**	3,939	Lat 42°34'43", Long 124°03'25"

Table S8. Thermal infrared imagery (TIR) statistics include the starting and ending kilometer of each flight, total river kilometers flown and the min, max and mean (standard deviation; SD) of water temperature measured from the TIR in each watershed.

				Temperature (^oC)			
Watershed (HUC10)	Start km TIR	End km TIR	River Km	Mean (SD)	Min	Max	Note
Canton Creek	0	17.01	17	20.52 (1.06)	17.6	22.4	
Little River	0	40.65	40	21.04 (2.19)	16.2	25.5	
Rock Creek	0.02	20.47	20	20.38 (0.61)	18.1	21.7	
Steamboat Creek	0.12	31.12	31	20.17 (1.34)	16.4	22.8	
Lower Umpqua River	32	175	143	25.9 (0.53)	24.6	27.4	Below Winchester
Lower North Umpqua River	0.07	84.3	84.2	20.7 (2.33)	17.1	24.8	Winchester to Steamboat
Middle North Umpqua River	83.48	119.66	36	14.78 (1.04)	10.4	16.2	Steamboat to Toketee

Table S9. Physical habitat data available in North Umpqua watersheds.

Stream Name	Total Surveys	Max Surveys	Min Year	Max Year	Survey Type	Project
Canton Creek	9	1	1998	2018	Oregon Plan	Aquatic Inventories
Little River	23	3	1999	2019	Oregon Plan	Aquatic Inventories
Lower North Umpqua River	23	4	1999	2020	Oregon Plan	Aquatic Inventories
Middle North Umpqua River	40	4	1998	2018	Oregon Plan	Aquatic Inventories
Rock Creek	14	2	1998	2017	Oregon Plan	Aquatic Inventories
Steamboat Creek	31	3	1999	2018	Oregon Plan	Aquatic Inventories
Upper North Umpqua River	20	2	1999	2017	Oregon Plan	Aquatic Inventories

Table S10. Predictors considered in spatial distribution modeling of juvenile steelhead in the Lower and North Umpqua watersheds.

Predictor	Description	Units	Data Source
Steelhead occurrence	Presence or absence of juvenile steelhead within survey reach.	0 or 1	ODFW
Flow Permanence	Average probability of flow permanence	Probability	PROSPER; Jaeger et al. 2019
Maximum Weekly Maximum Temperature	Average maximum weekly stream temperature	Degrees C	NorWeST; Isaak et al. 2017
Mean Summer Flow	Average stream flow conditions	cubic feet/sec	VIC; Wnger et al. 2010
Distance to Ocean	Linear distance from surveyed reach to ocean	Kilometers	ODFW
Total Spring Precipitation	Sum of spring precipitation	Millimeters	PRISM Climate Group 2014
Total Winter Precipitation	Sum of winter precipitation	Millimeters	PRISM Climate Group 2014
Solar Radiation	Current shade thermal energy, averaged downstream	Watt-hours/m2	NetMap; Benda et al. 2007
Stream Size	Stream size based on Strahler stream order Small=1, 2,3; Med/large =>4	Factor; small, med/large	NHD; Moore et al. 2019
Slope	Slope of stream segment	rise/run	NHD; Moore et al. 2019

Table S11. Generalized linear model results from best model selected using AIC.

Coefficients	Prob	Logit Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	0.398	-0.410	1.946	-0.211	0.832
Temperature	0.464	-0.142	0.074	-1.912	0.055
Distance	0.501	0.007	0.002	2.912	0.003
Spring Precip.	0.502	0.008	0.002	3.927	0.000

Table S12. Confusion matrix diagnostics of best model.

Measure	Value	Description
Accuracy	0.753	How often classified correctly
Misclassification rate	0.247	How often classified incorrectly
True Positive Rate	0.750	When true presence, rate predicted present
False Positive Rate	0.200	When true absence, rate predicted present
Precision	0.982	When present, how often correct
Prevalence	0.701	How often presence occurs in sample

Table S13. Parameters used in VELMA disturbance assessment.

Disturbance Type			
Harvest		Fire	
ageThreshold	0-500	activationProbability	1
biomassAgStemNharvestFraction	0.95	ageThreshold	0
biomassAgStemNoffsiteFraction	0.85	ashFraction	0.7
biomassBgStemNharvestFraction	0.95	burnLossK	0.7
biomassBgStemNoffsiteFraction	0	currentDayPrecipThreshold	100
biomassLeafNharvestFraction	0.95	enableAutomaticSpatialReporting	FALSE
biomassLeafNoffsiteFraction	0	filterMapFullName	Fire extent spatial file
biomassRootNharvestFraction	0.95	fuelThreshold	0
biomassRootNoffsiteFraction	0	initializeActiveJdays	Day of disturbance
enableAutomaticSpatialReporting	FALSE	initializeActiveYears	2020
filterMapFullName	Harvest extent spatial file	initializeAgeModification	1
initializeActiveJdays	*Day at which disturbance occurred* (152)	initializeCoverIds	1
initializeActiveLoops	1	initializeFilterIds	1
initializeActiveYears	year	modelClass	BurnLsrDisturbanceModel
initializeAgeModification	0	moistureThreshold	0.7
initializeCoverIds	1	setFixedSeed	
initializeFilterIds	1		
modelClass	HarvestLsrDisturbanceModel		

APPENDIX 2 – SUPPLEMENTAL FIGURES

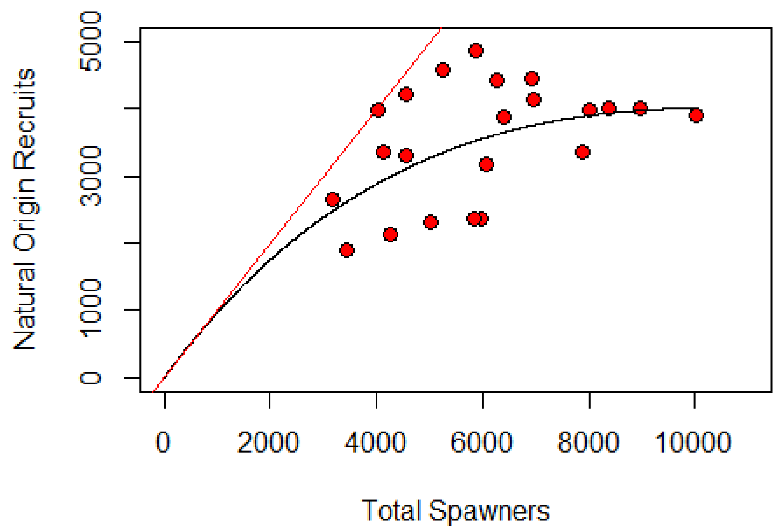


Figure S1. A Ricker curve fit to spawner-recruit data for the North Umpqua summer steelhead population from 1992-2014. The red line represents the 1:1 relationship where spawner and recruit numbers are equal along their respective axes. The black line represents the fit of the Ricker curve to the data.

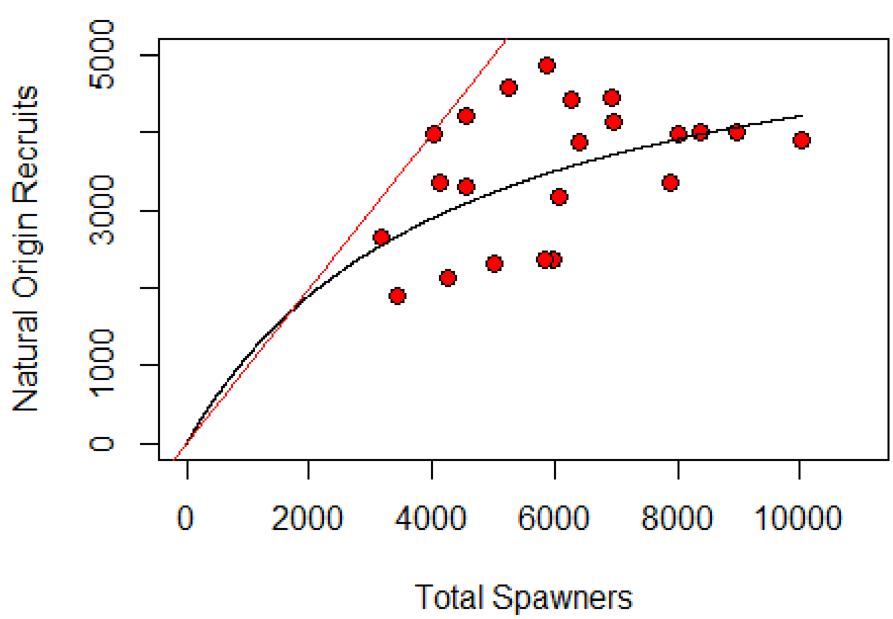


Figure S2. A Beverton-Holt curve fit to spawner-recruit data for the North Umpqua summer steelhead population from 1992-2014. The red line represents the 1:1 relationship where spawner and recruit numbers are equal along their respective axes. The black line represents the fit of the Beverton-Holt curve to the data.

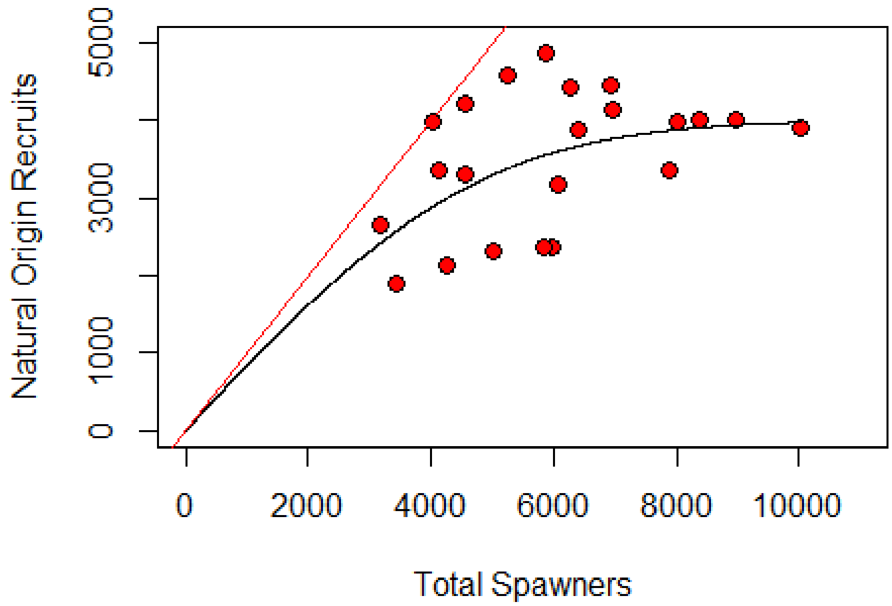


Figure S3. A Logistic Hockey Stick curve fit to spawner-recruit data for the North Umpqua summer steelhead population from 1992-2014. The red line represents the 1:1 relationship where spawner and recruit numbers are equal along their respective axes. The black line represents the fit of the Logistic Hockey Stick curve to the data.

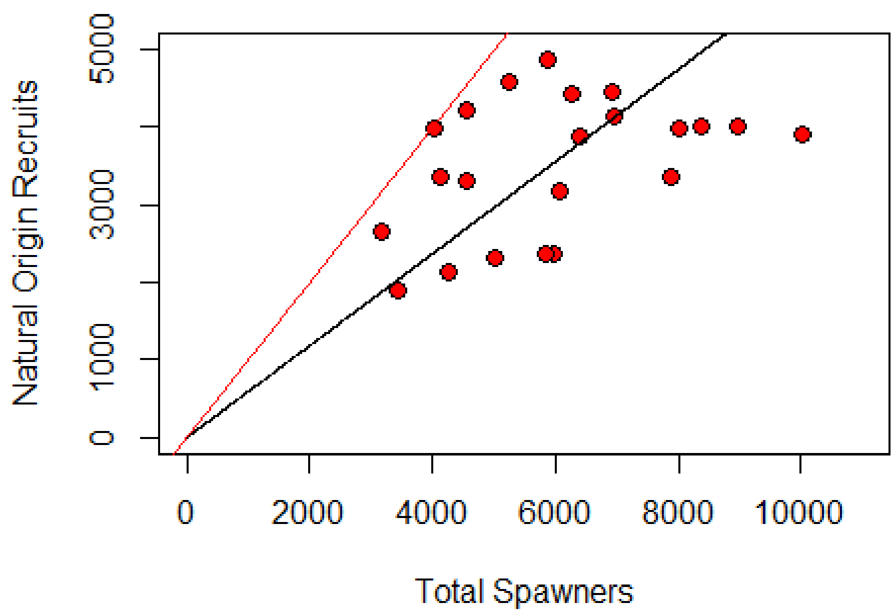


Figure S4. A density-independent curve fit to spawner-recruit data for the North Umpqua summer steelhead population from 1992-2014. The red line represents the 1:1 relationship where spawner and recruit numbers are equal along their respective axes. The black line represents the fit of the density independent curve to the data.

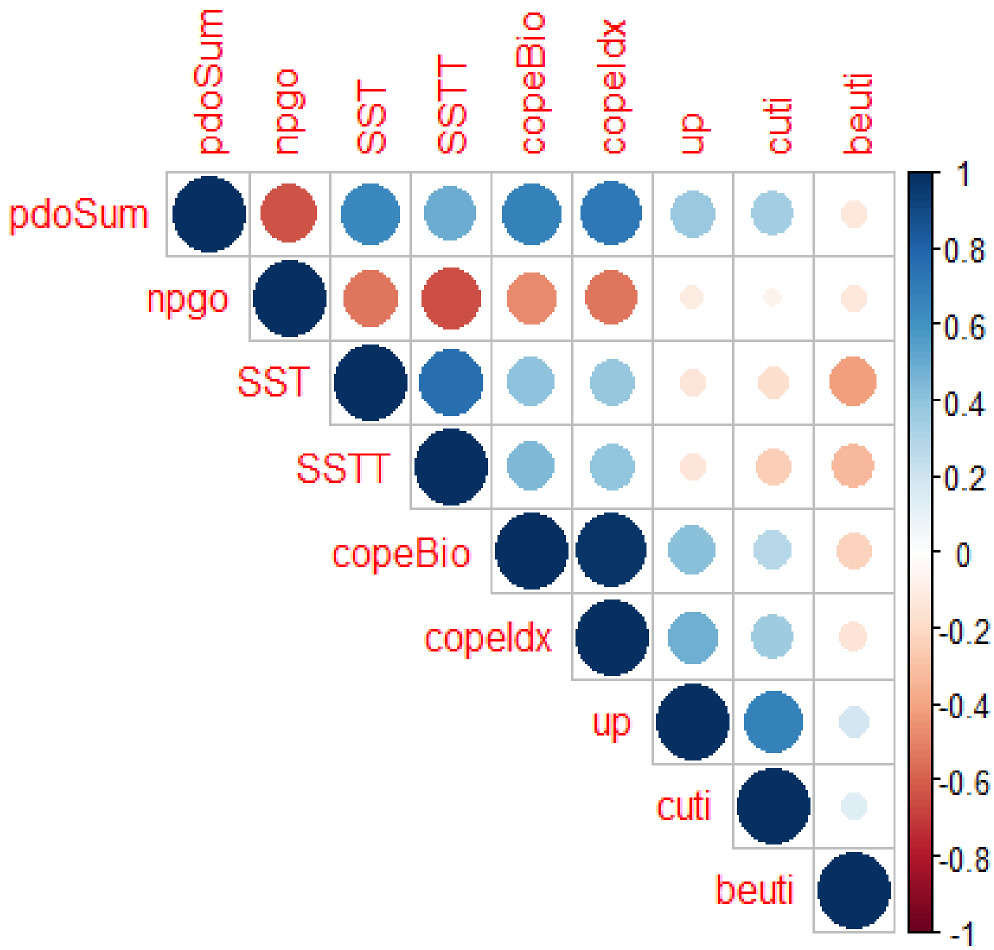


Figure S5. A correlation matrix of potential marine covariates to the stock recruitment relationship for Umpqua summer steelhead, 1992-2014. Larger circles indicate higher correlation and color indicates whether the correlation is positive (i.e., covariates increase or decrease together) or negative (i.e., increase in one covariate produces a decrease in the other or vice versa).

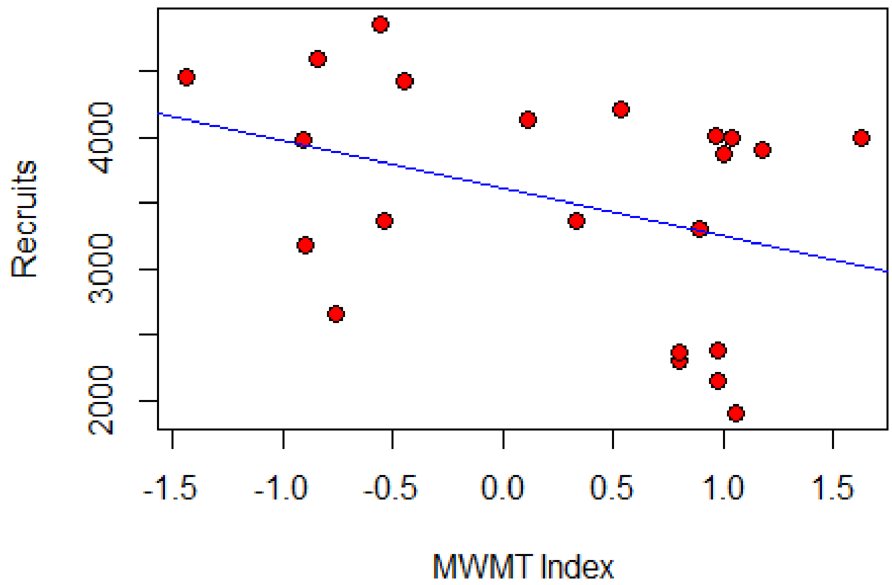


Figure S6. Linear regression of Umpqua summer steelhead recruits from 1992-2014 on maximum weekly mean water temperature corresponding to 5-year-old fish. The blue line represents the linear regression line.

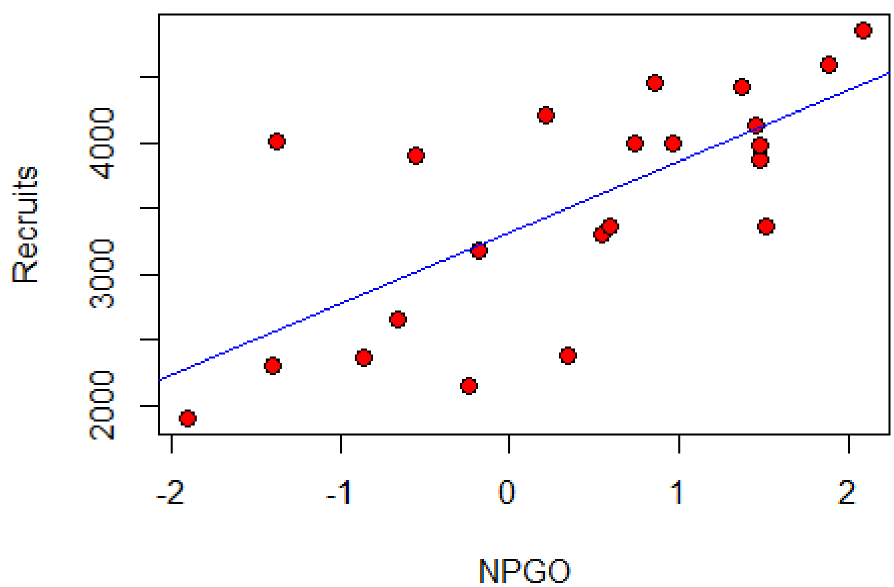


Figure S7. Linear regression of Umpqua summer steelhead recruits from 1992-2014 on North Pacific Gyre Oscillation (NPGO) corresponding to 5-year-old fish. The blue line represents the linear regression line.

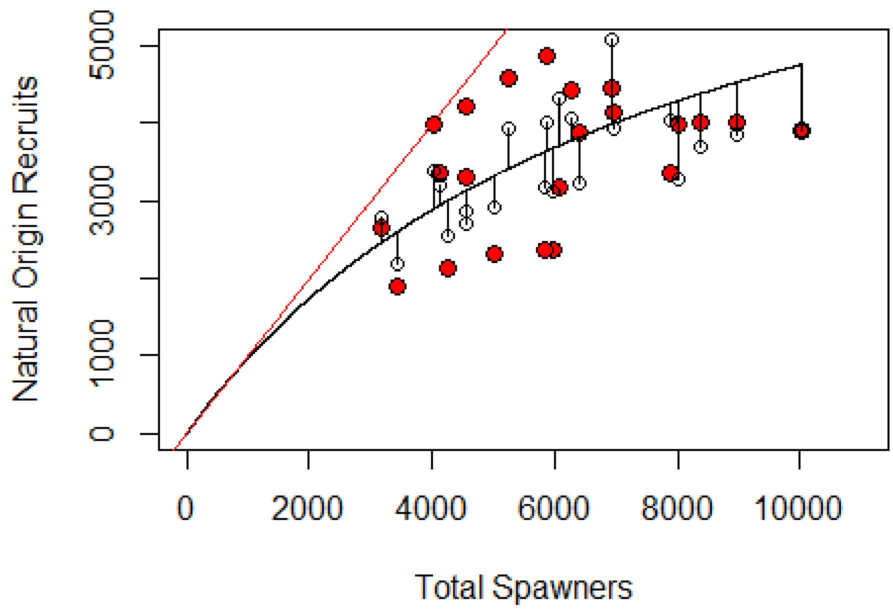


Figure S8. A Beverton-Holt curve with MWMT included as a covariate fit to spawner-recruit data for the North Umpqua summer steelhead population from 1992-2014. The red line represents the 1:1 relationship where spawner and recruit numbers are equal along their respective axes. The black line represents the fit of the Beverton-Holt curve to the data. The “lollipop” lines coming off the fit curve shows how the addition of MWMT adjusts the predicted values for each year. Open circles represent the new predicted values.

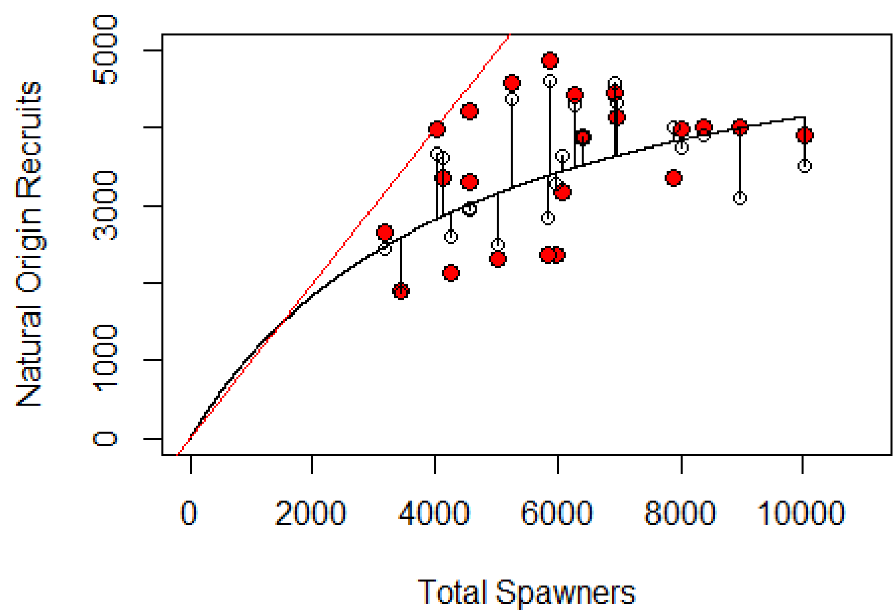


Figure S9. A Beverton-Holt curve with MWMT and NPGO included as covariates fit to spawner-recruit data for the North Umpqua summer steelhead population from 1992-2014. The red line represents the 1:1 relationship where spawner and recruit numbers are equal along their respective axes. The black line represents the fit of the Beverton-Holt curve to the data. The "lollipop" lines coming off the fit curve shows how the addition of NPGO and MWMT adjusts the predicted values for each year. Open circles represent the new predicted values.

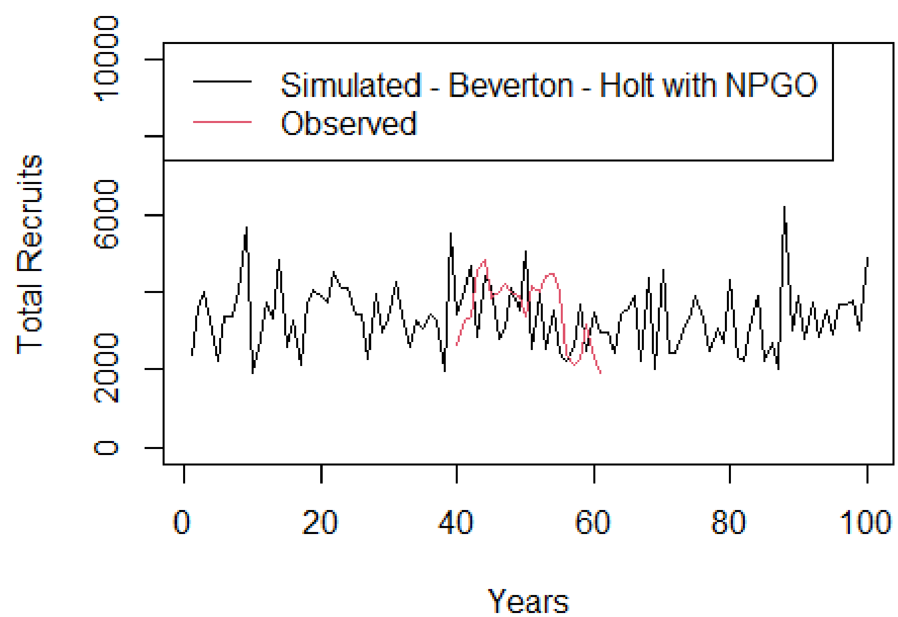


Figure S10. A single 100-year simulation using a Beverton-Holt curve with NPGO included as a covariate. The red line is the actual, observed abundance from 1992-2014 superimposed on the simulated data for reference.

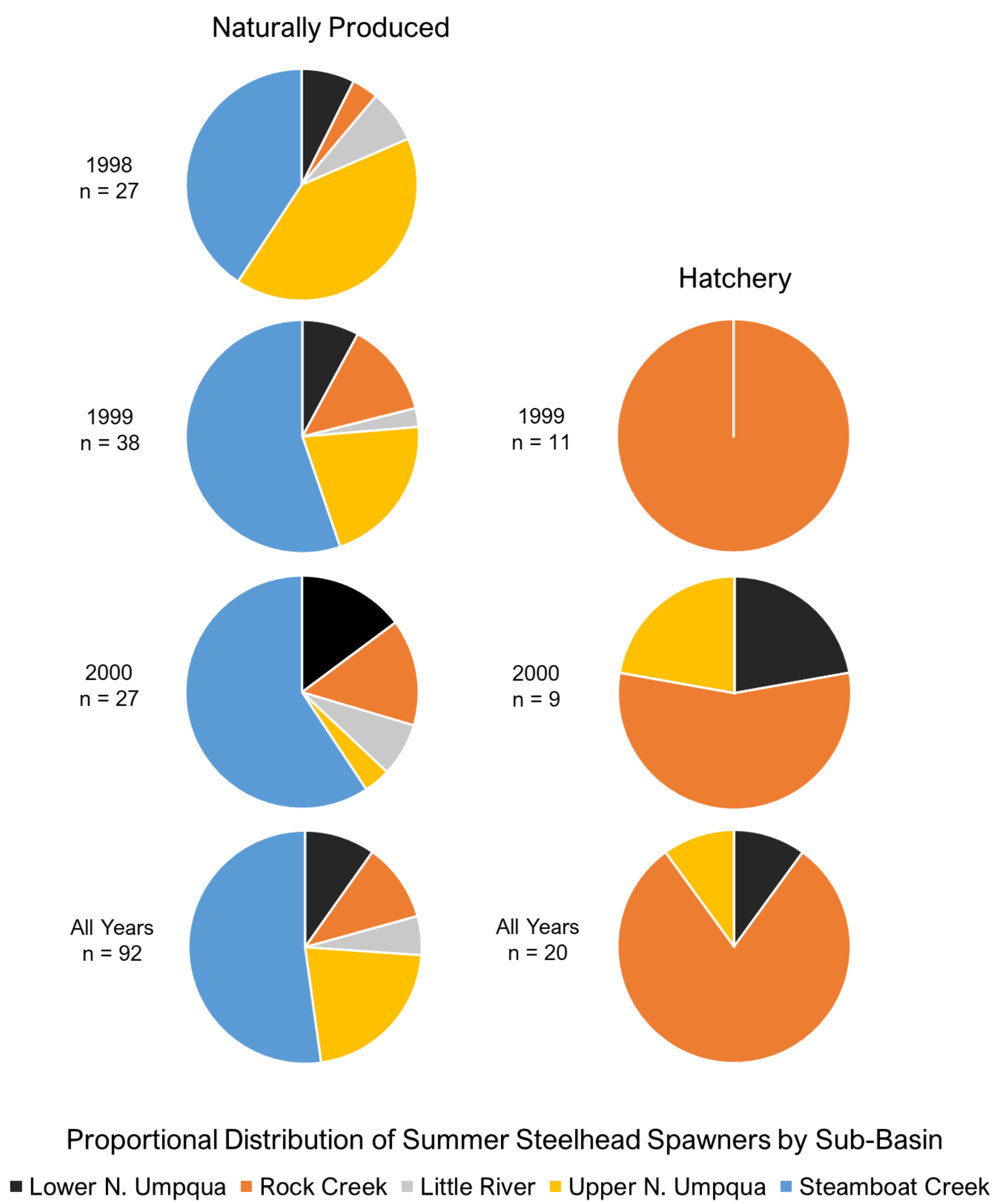


Figure S11. Proportional use of five North Umpqua sub-basins for spawning by natural and hatchery origin summer steelhead, inferred from telemetry data collected by Loomis et al. (2003) in run-years 1998, 1999, and 2000. Fish were considered summer steelhead if they were tagged at Winchester Dam during the summer counting period (May 1 – November 30). No hatchery origin steelhead were tagged in 1998.

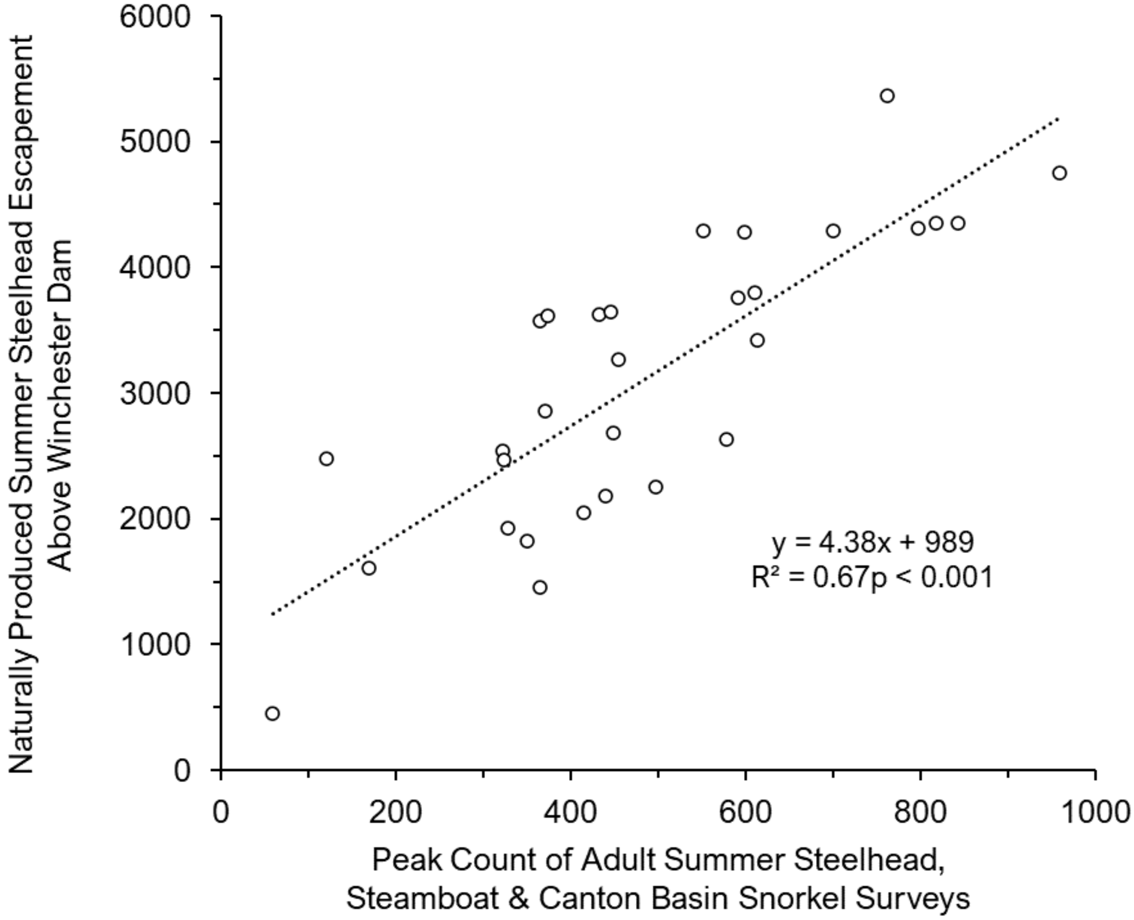


Figure S12. Relationship between the peak count of adult summer steelhead observed holding in the Steamboat Creek sub-basin (including Canton Creek) during summer snorkel surveys and escapement of naturally produced summer steelhead above Winchester Dam, 1991-2021. Note that the peak count of holding adults in 2021 (59) is likely an undercount due to limited surveys (June only).

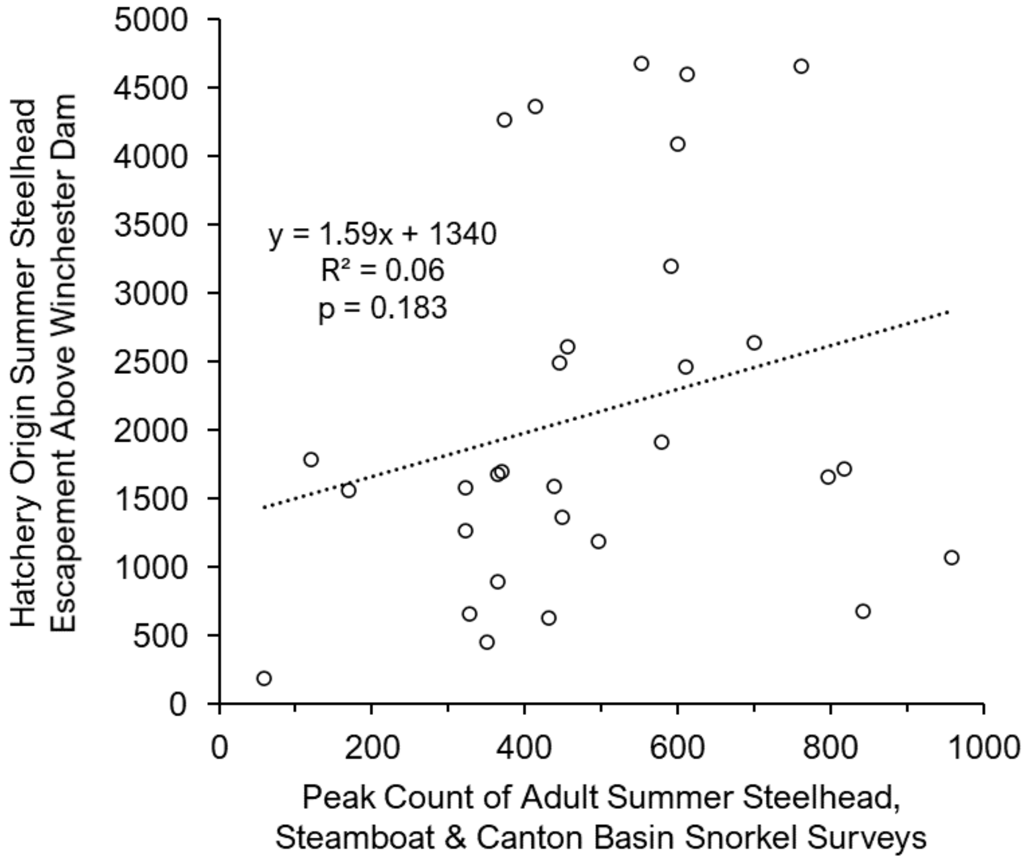


Figure S13. Relationship between the peak count of adult summer steelhead observed holding in the Steamboat Creek sub-basin (including Canton Creek) during summer snorkel surveys and escapement of hatchery origin summer steelhead above Winchester Dam, 1991-2021. Note that the peak count of holding adults in 2021 (59) is likely an undercount due to limited surveys (June only).

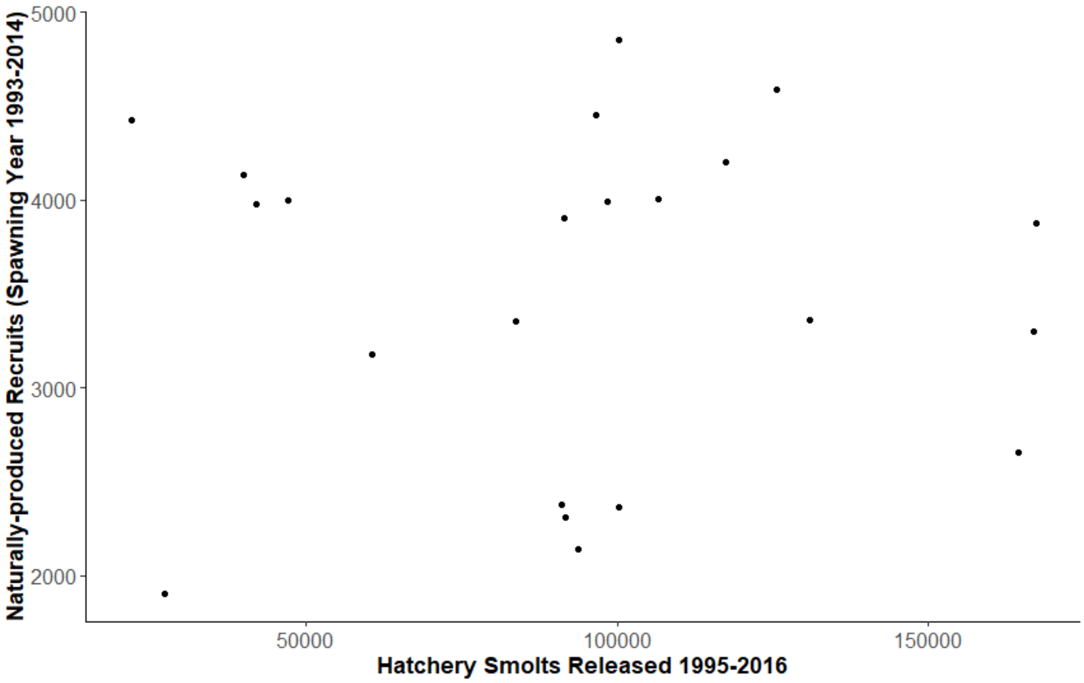


Figure S14. Relationship between natural-origin recruits and hatchery smolt releases aligned for the year in which both would have out-migrated together as smolts.

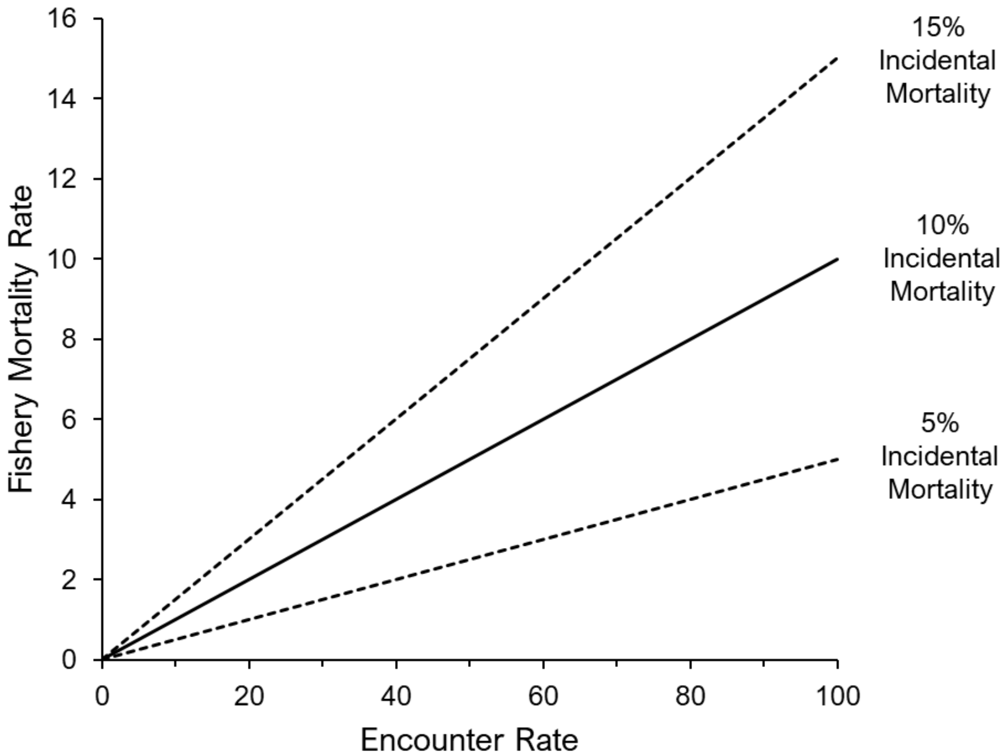


Figure S15. Hypothetical fishery mortality rates based on encounter rates and three levels of pre-spawning mortality incidental to catch and release angling.

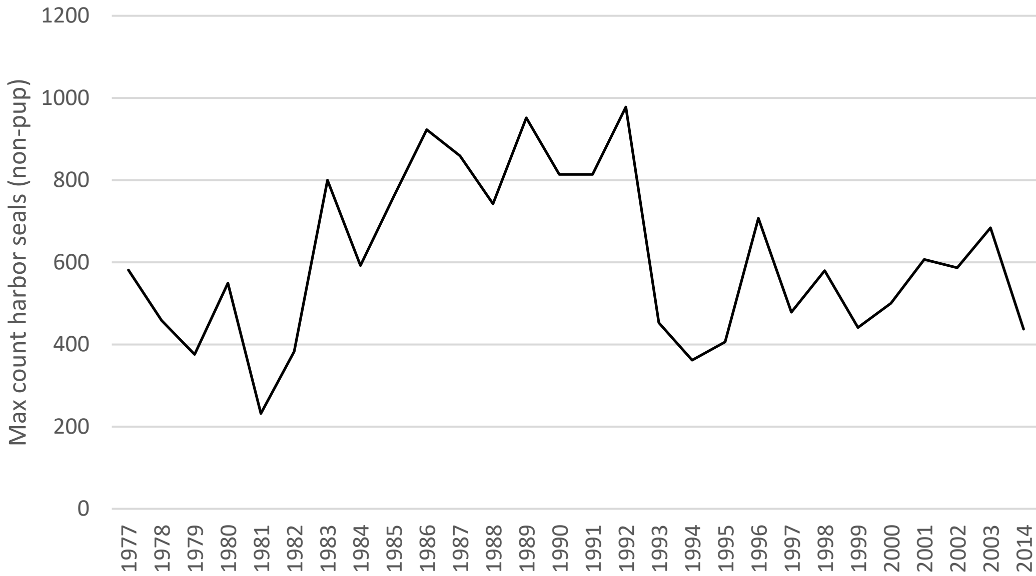


Figure S16. Maximum counts of harbor seals (excluding pups) at the Umpqua River.

Figure S17. Bolon Island Double-crested Cormorant colony, June 2003 (Photograph by Roy W. Lowe).

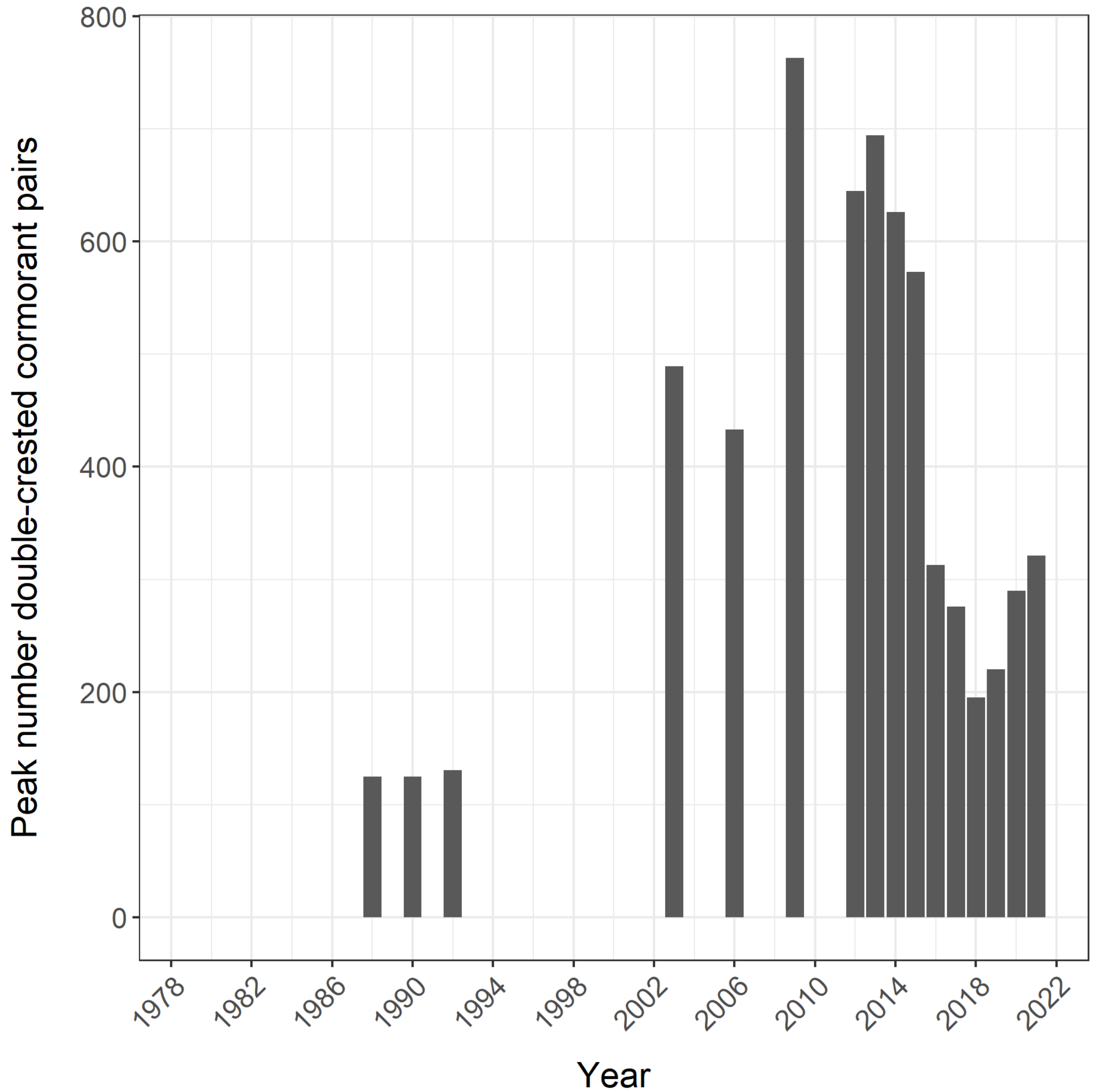


Figure S18. Peak number of double-crested cormorant pairs observed at all Umpqua basin colonies.

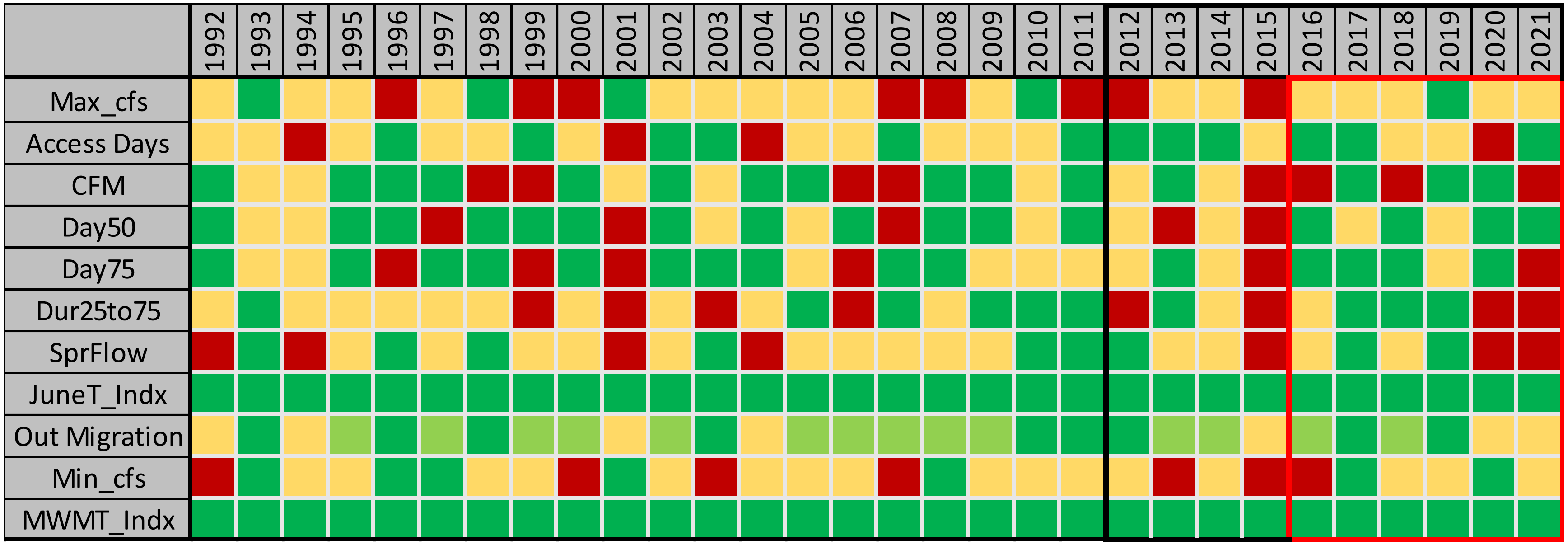


Figure S19. Freshwater stoplight table for the Siletz summer steelhead population. Flow metrics are based on the stream gage on the mainstem Siletz River. Temperature indices are from the head of tide in June, and in steelhead holding areas in the summer.

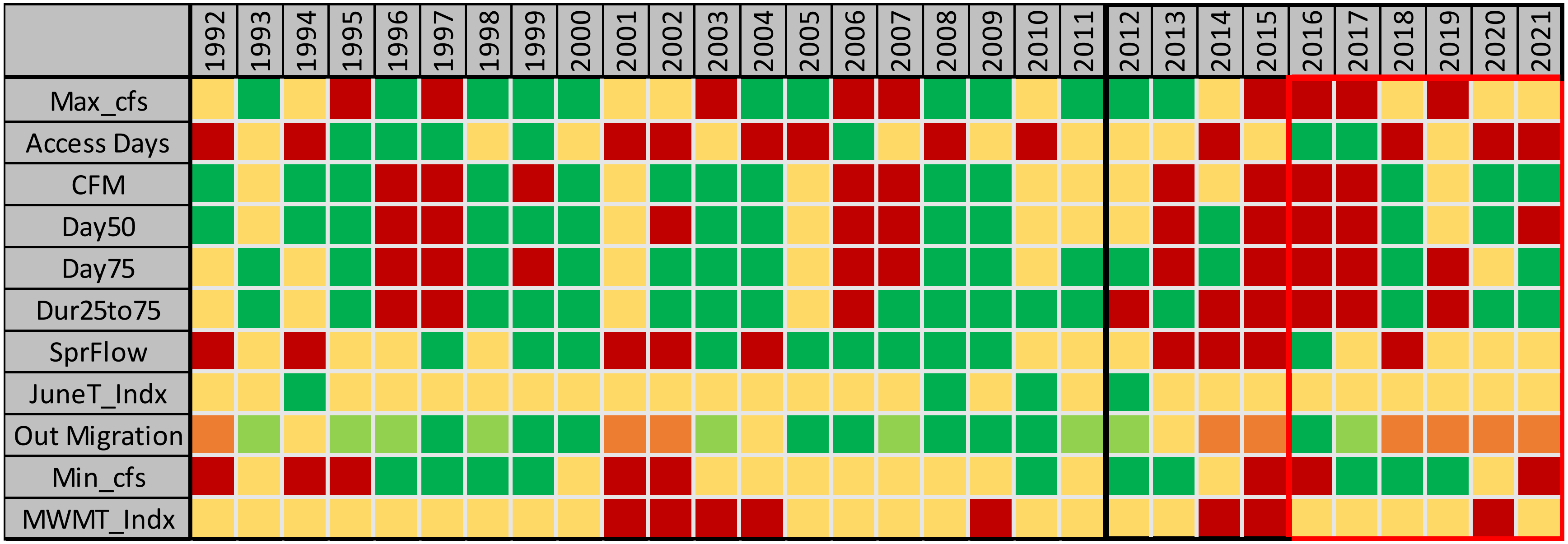


Figure S20. Freshwater stoplight table for Rogue summer steelhead populations. Flow metrics are based on the stream gage on the Rogue River near Agness. Temperature indices are from the gage at Agness in June; summer maximum temperature was measured at the stream gages on the Applegate River near Wilderville and on the Rogue River near Dodge Bridge.

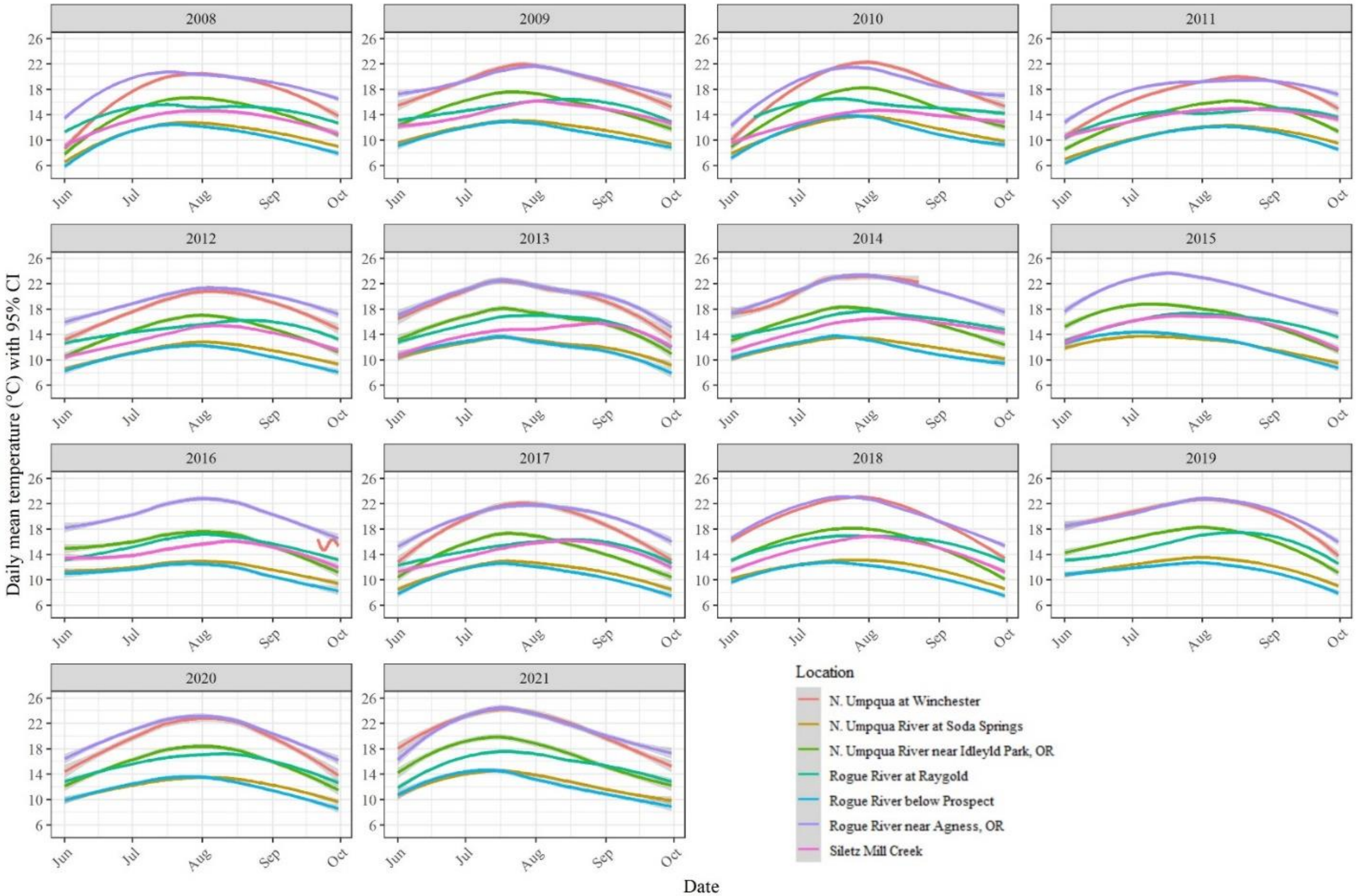


Figure S21. Daily mean temperature during summer (June – September) for locations within the Siletz, Umpqua, and Rogue basins, 2008-2021. Shaded areas represent 95% confidence intervals.

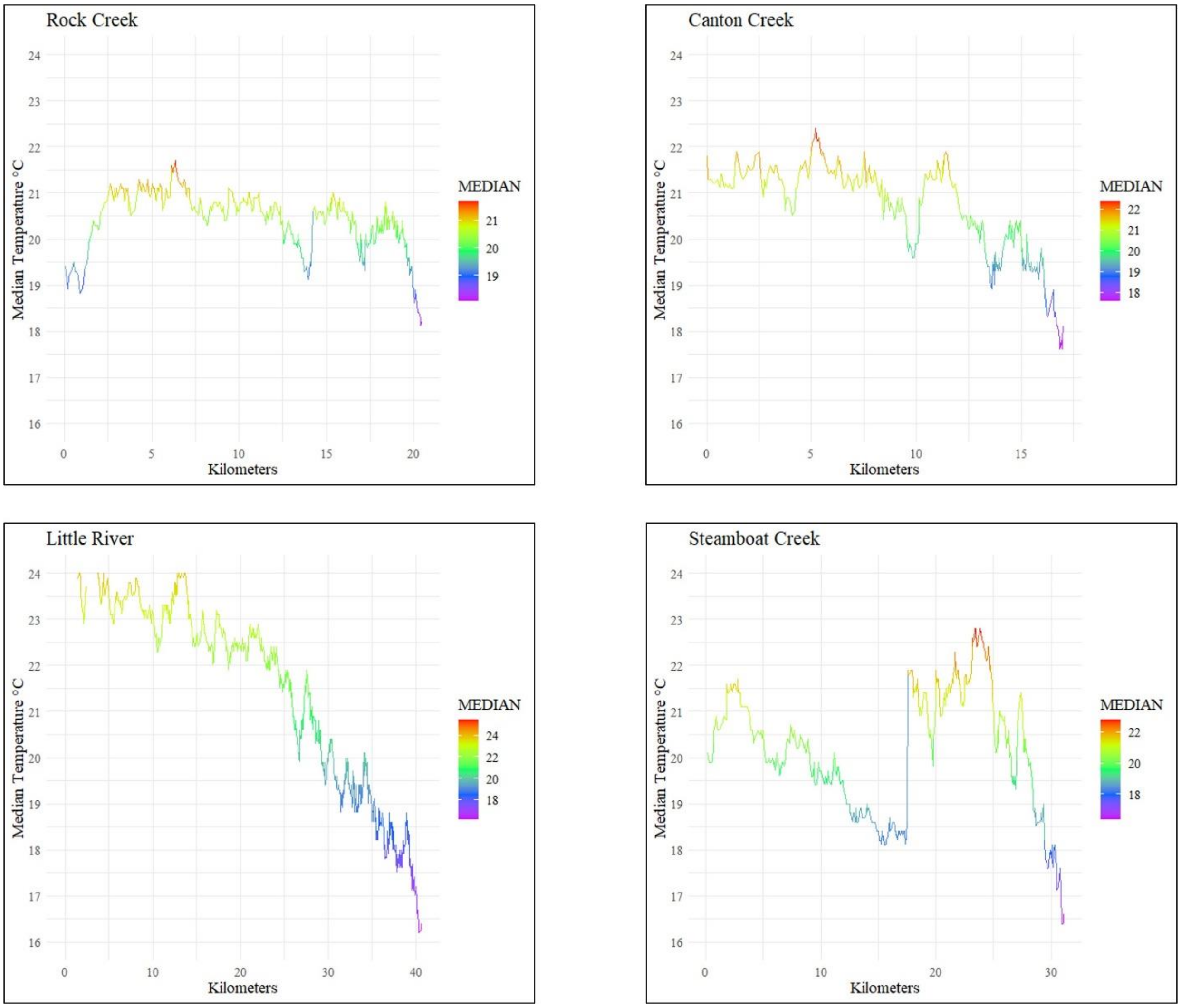


Figure S22. Longitudinal profiles of the median stream temperature obtained from thermal infrared imagery for tributary streams in the North Umpqua River.

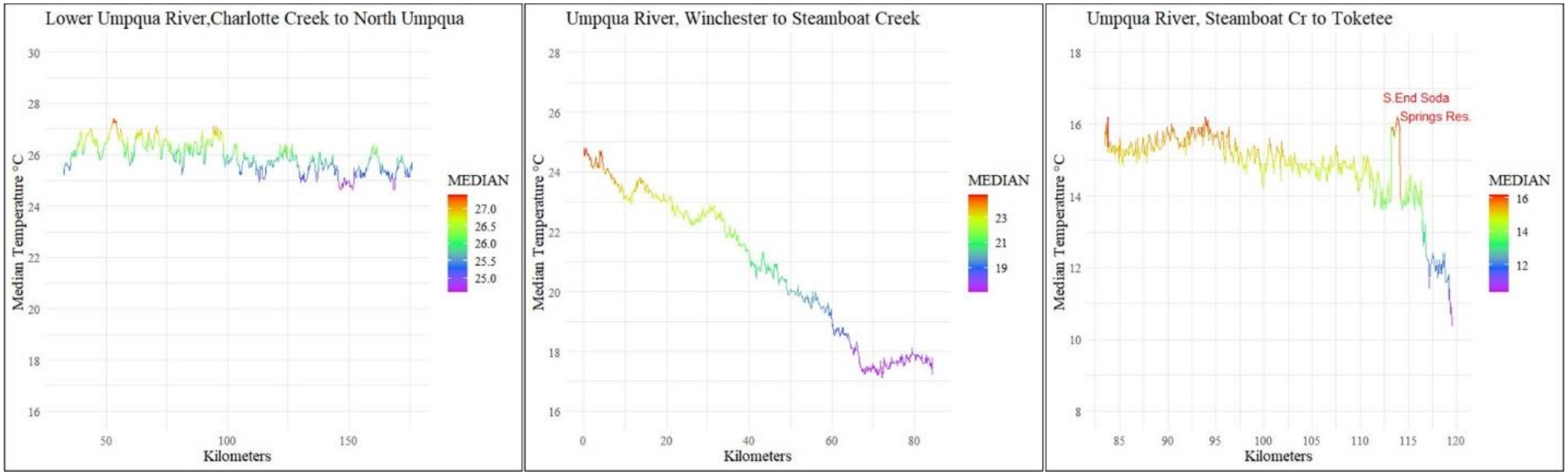


Figure S23. Longitudinal profiles of the median stream temperature obtained from thermal infrared imagery for mainstem Umpqua River sections.

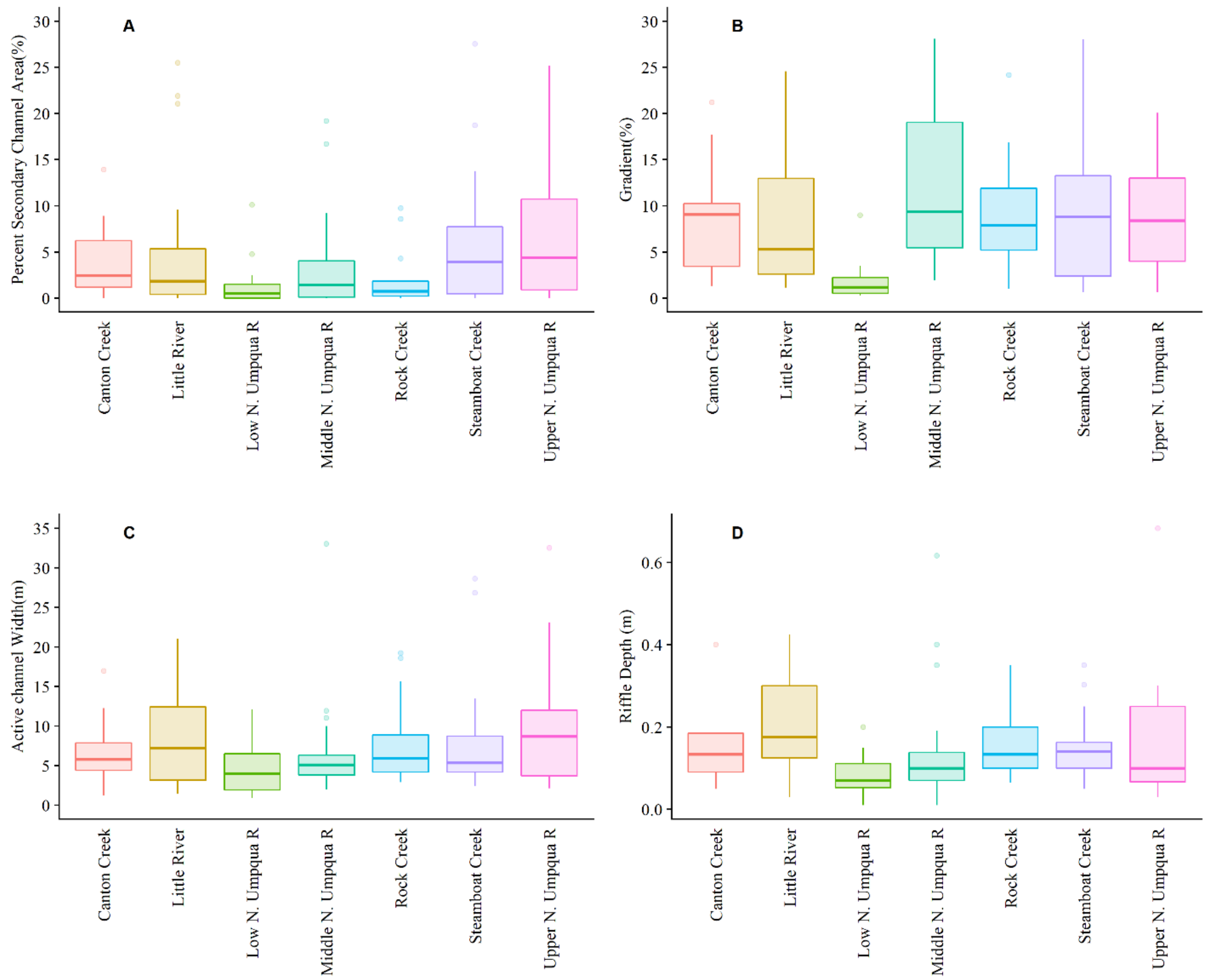


Figure S24. Distribution of different physical habitat metrics describing stream morphology. Box plots show distribution within the 25th and 75th percentiles, the median (center line), and the 95% confidence intervals. The data describe site conditions for 1998–2020.

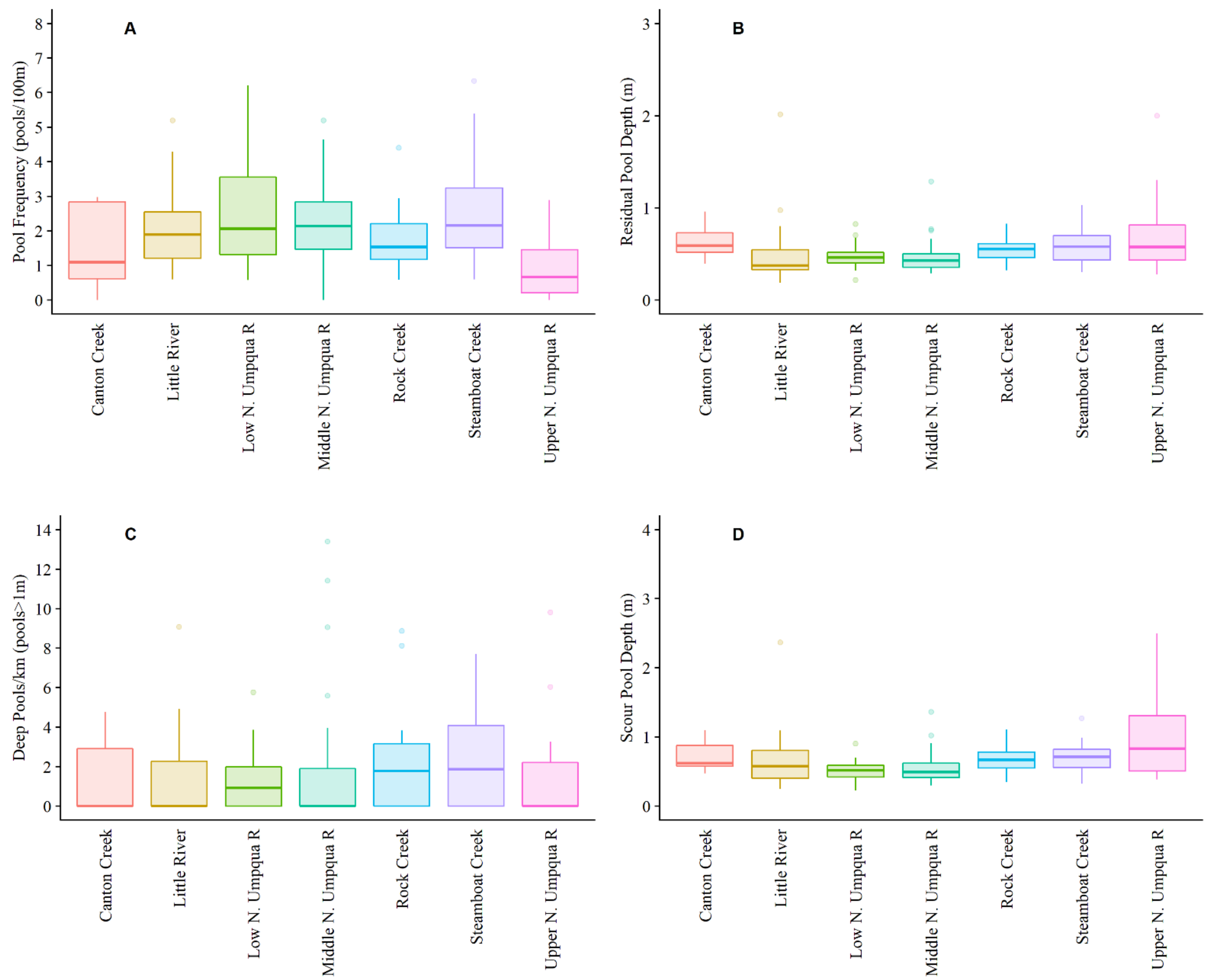


Figure S25. Distribution of different physical habitat metrics describing pools. Box plots show distribution within the 25th and 75th percentiles, the median (center line), and the 95% confidence intervals. The data describe site conditions for 1998–2020.

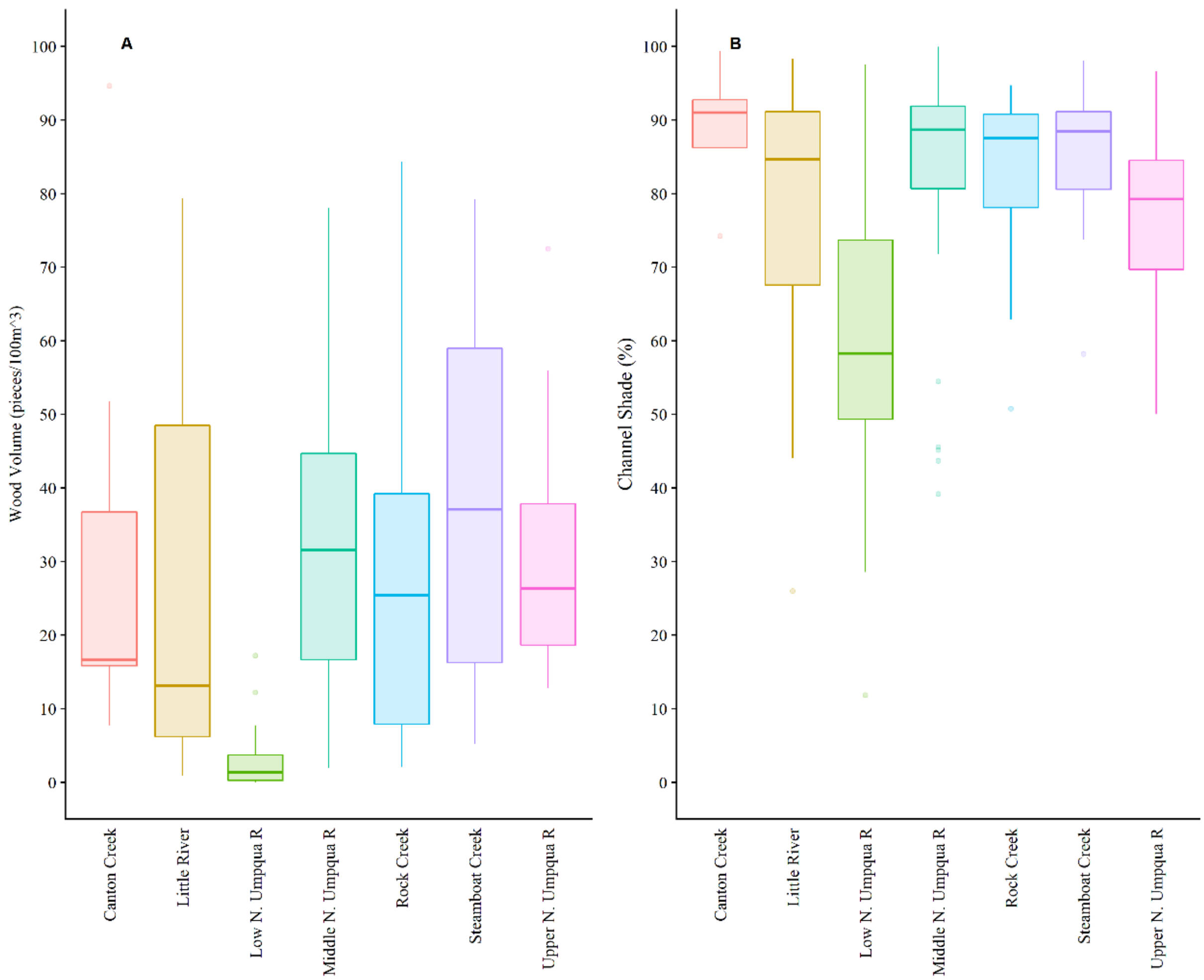


Figure S26. Distribution of different physical habitat metrics describing large wood and shading. Box plots show distribution within the 25th and 75th percentiles, the median (center line), and the 95% confidence intervals. The data describe site conditions for 1998–2020.

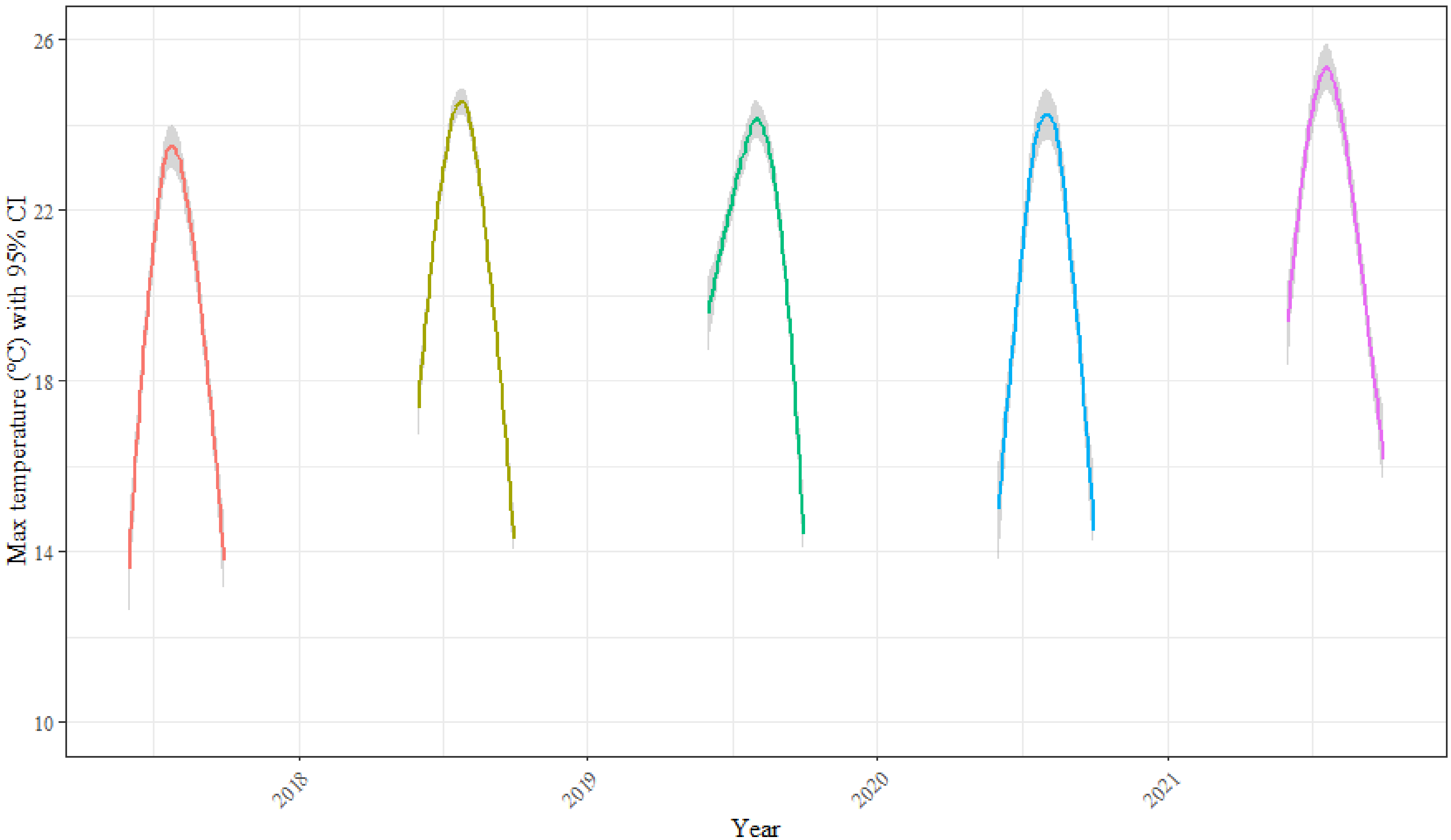


Figure S27. Distribution of maximum temperatures for June – September at Winchester Dam from 2017-2021. Each year is represented by a different color (*2017=red, 2018=yellow, 2019=green, 2020=blue, 2021=purple*).

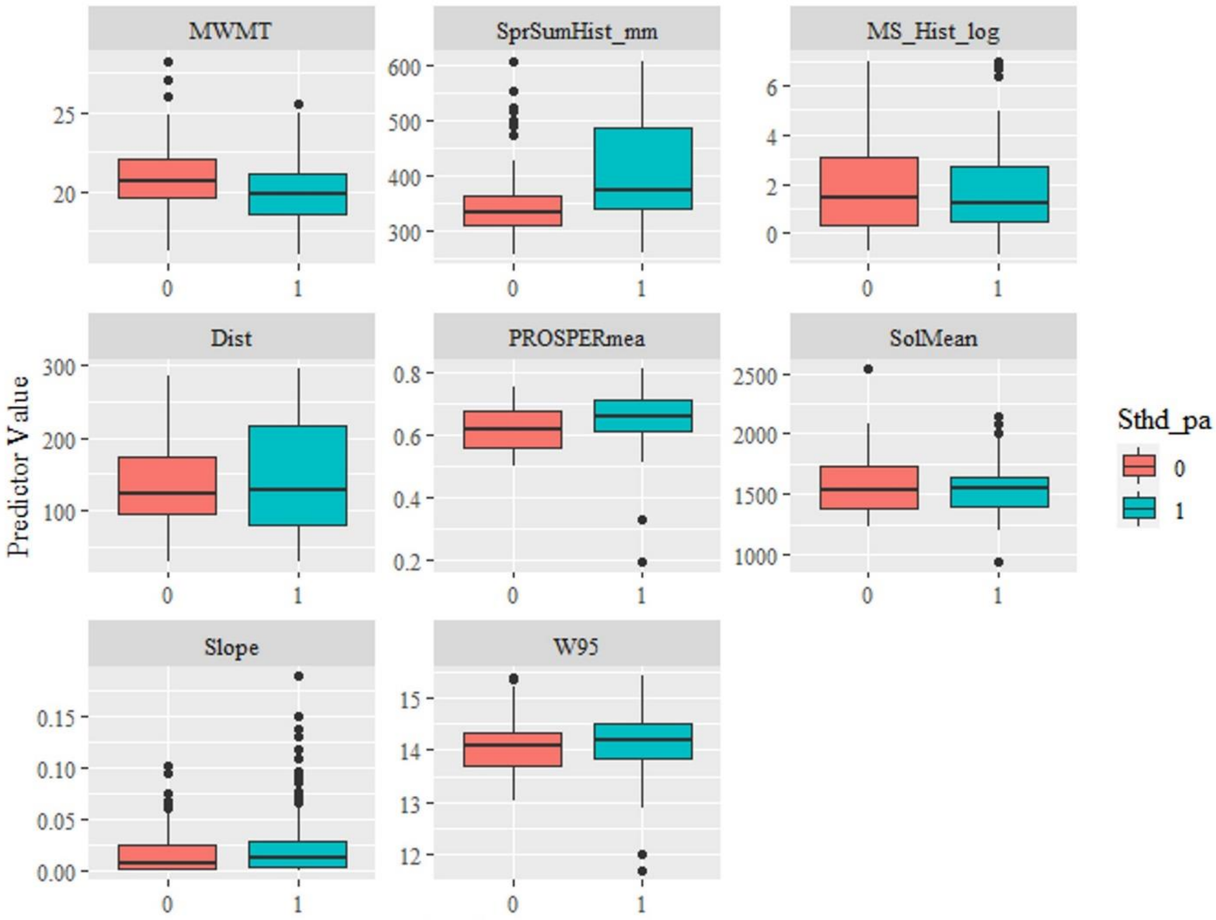


Figure S28. Box plots displaying distribution of predictor values for sites where steelhead were present (green; 1) and where they were absent (orange; 0).

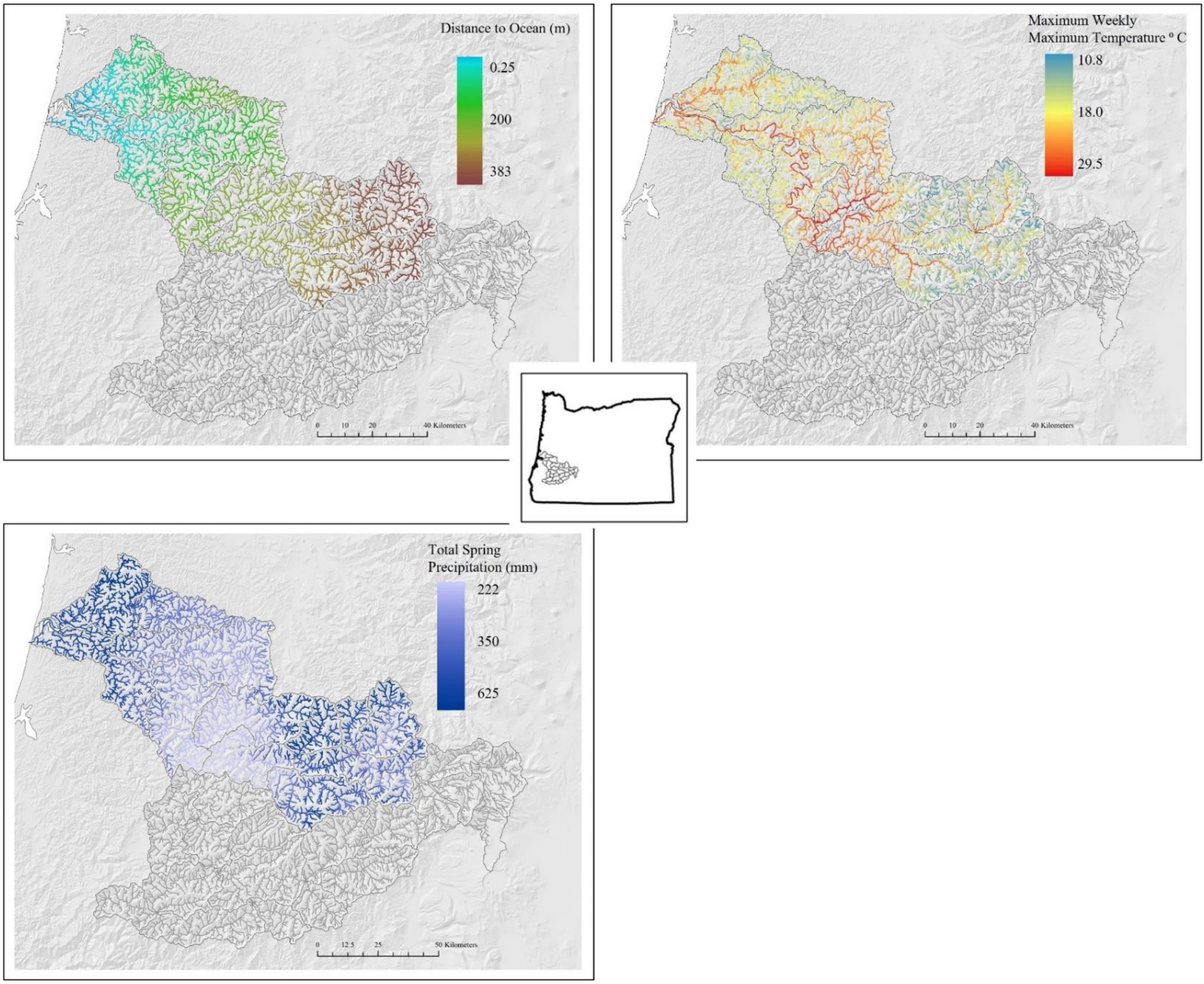


Figure S29. Spatial distribution of the predictor variables included in the final juvenile steelhead distribution model.

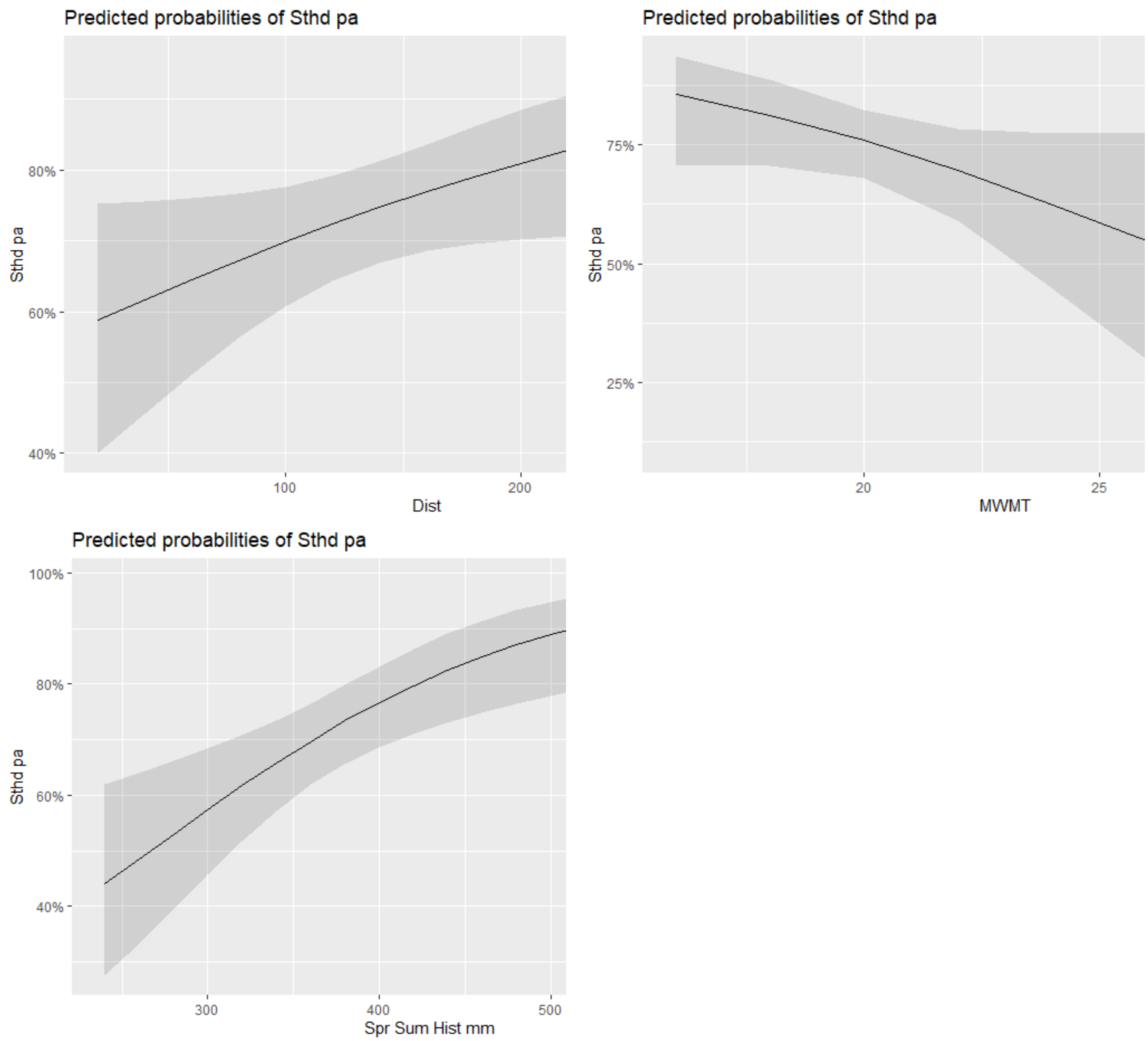
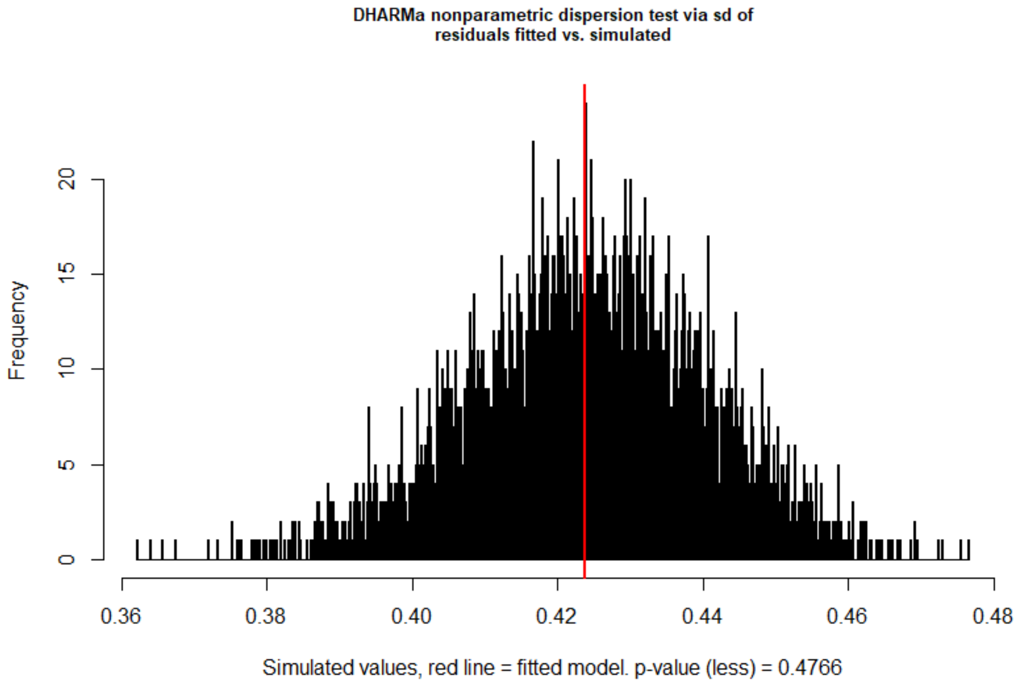


Figure S30. Marginal effect plots of predictors from best species distribution model. Marginal effects describe how change in the response (presence/absence of steelhead) is related to change in each predictor variable. Y-axis = probability of juvenile steelhead presence, x-axis = predictor variable values. The gray area represents 95% confidence intervals around marginal effect.



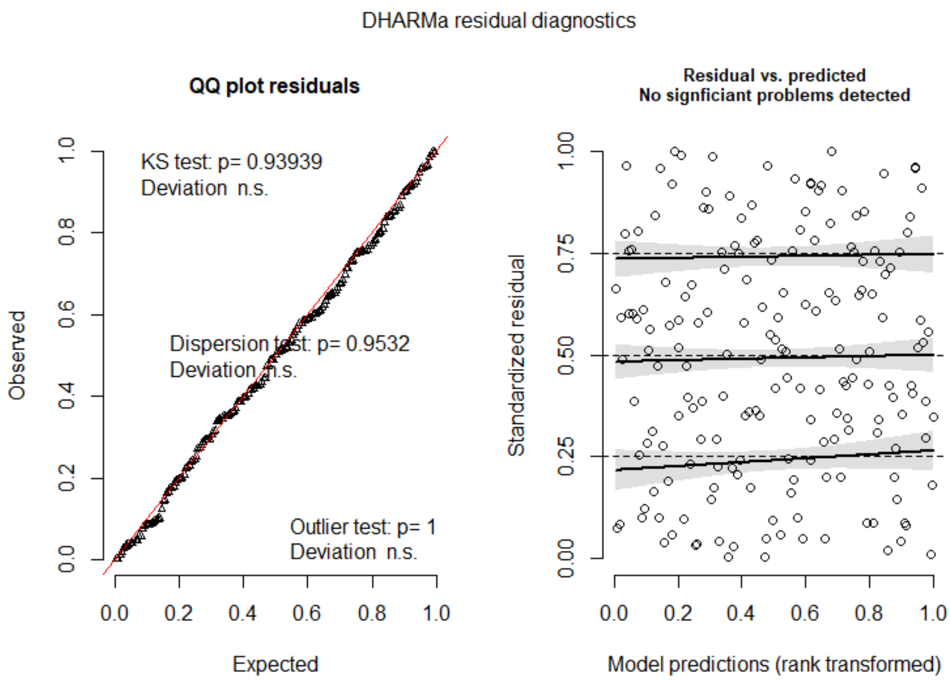


Figure S31. Plots summarizing residuals from the best model predicting juvenile steelhead throughout the Lower and North Umpqua watersheds. Simulated residuals (top figure) displaying the distribution of simulated values, QQ plot (bottom left) which can detect deviations from expected distribution and tests distribution, dispersion, and outliers (none were significant) and the plot of the residuals against predicted (right).

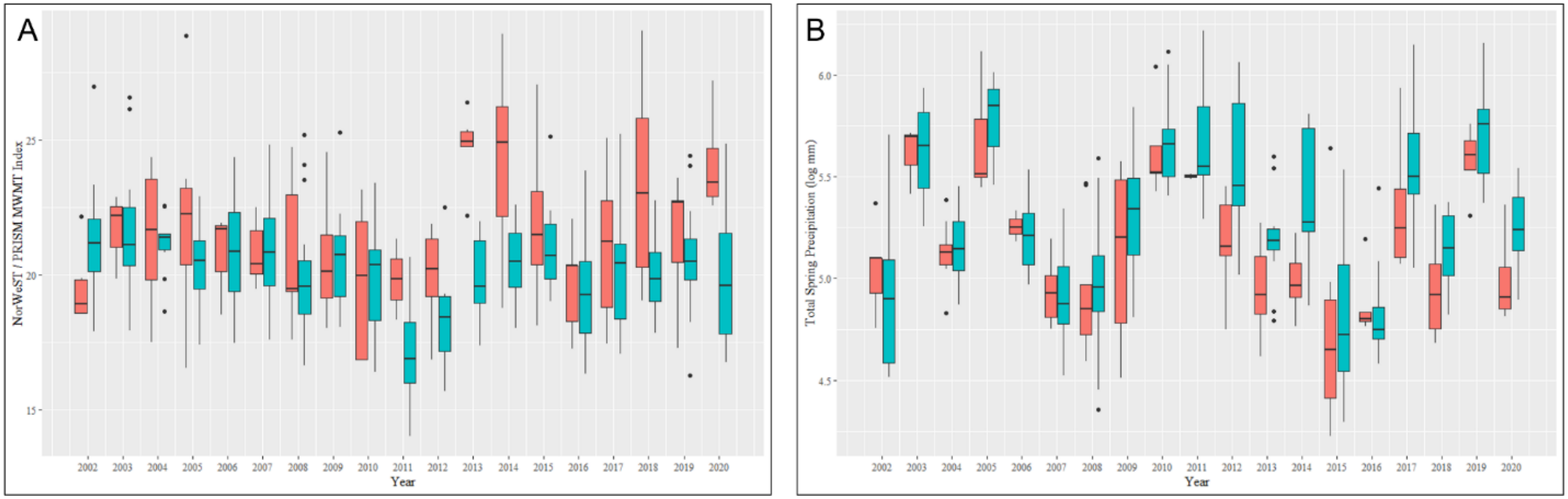


Figure S32. Box plots displaying the presence (*green*) and absence (*orange*) of juvenile steelhead varying through time (x-axis) relative to the NorWeST/MWMT Index (A) and the total amount of spring precipitation (B).

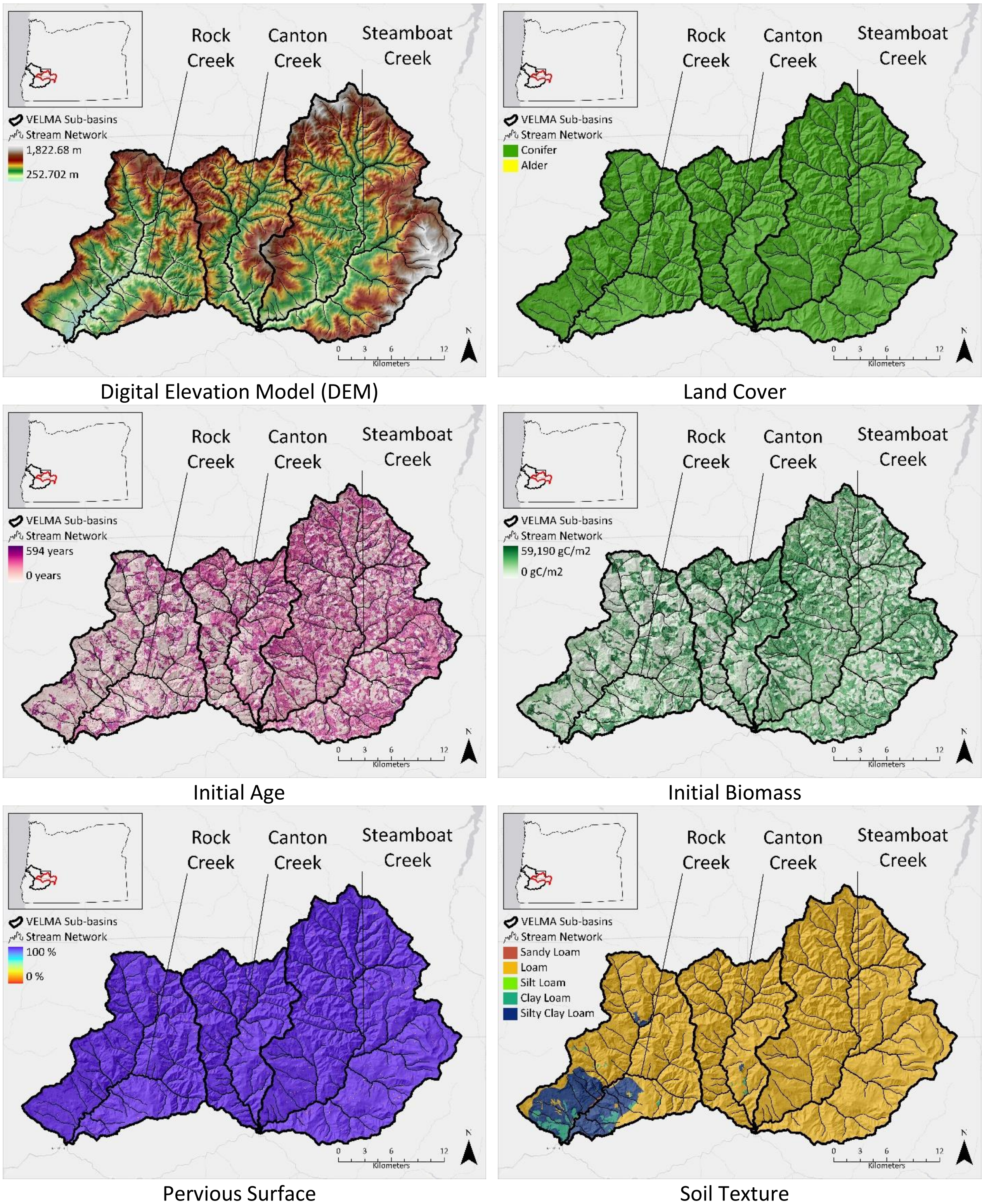


Figure S33. Spatial data inputs to initiate VELMA in each of the watersheds.

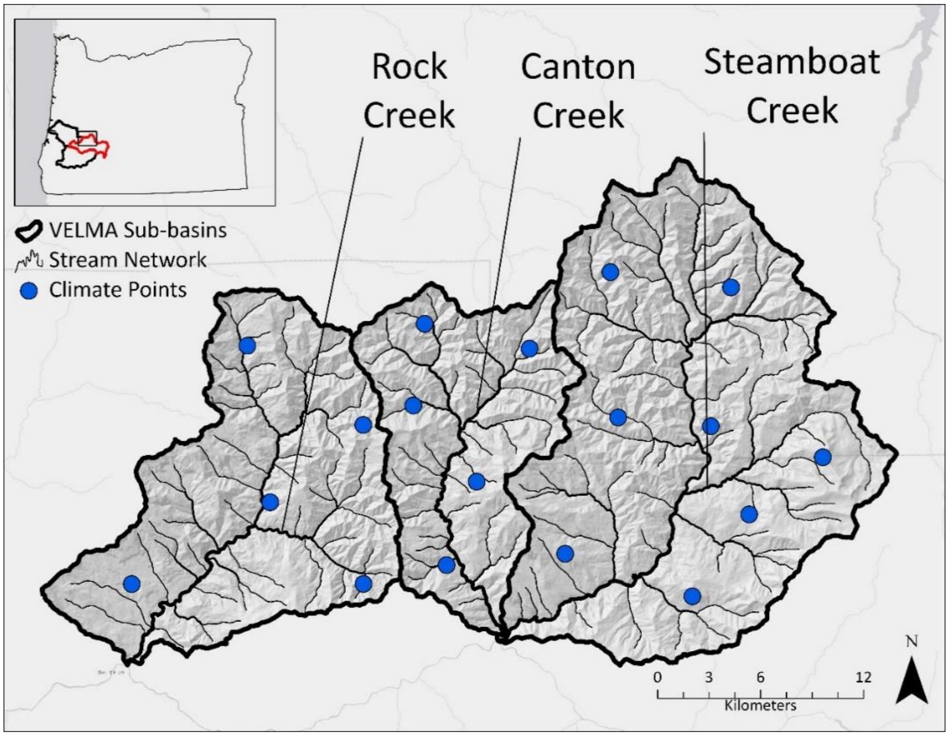


Figure S34. Locations at which daily temperature and precipitation data for current (PRISM-derived) and future (MACA – derived) were collected and inputted to the model.

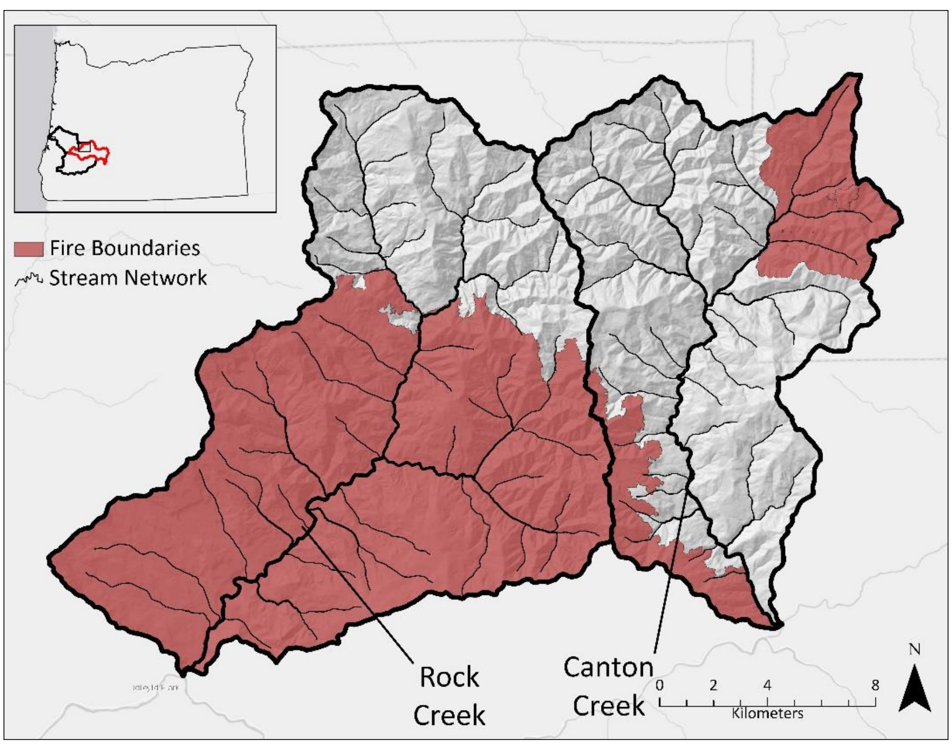


Figure S35. Areas burned to implement pre-and post- fire analysis in Canton and Rock Creek.

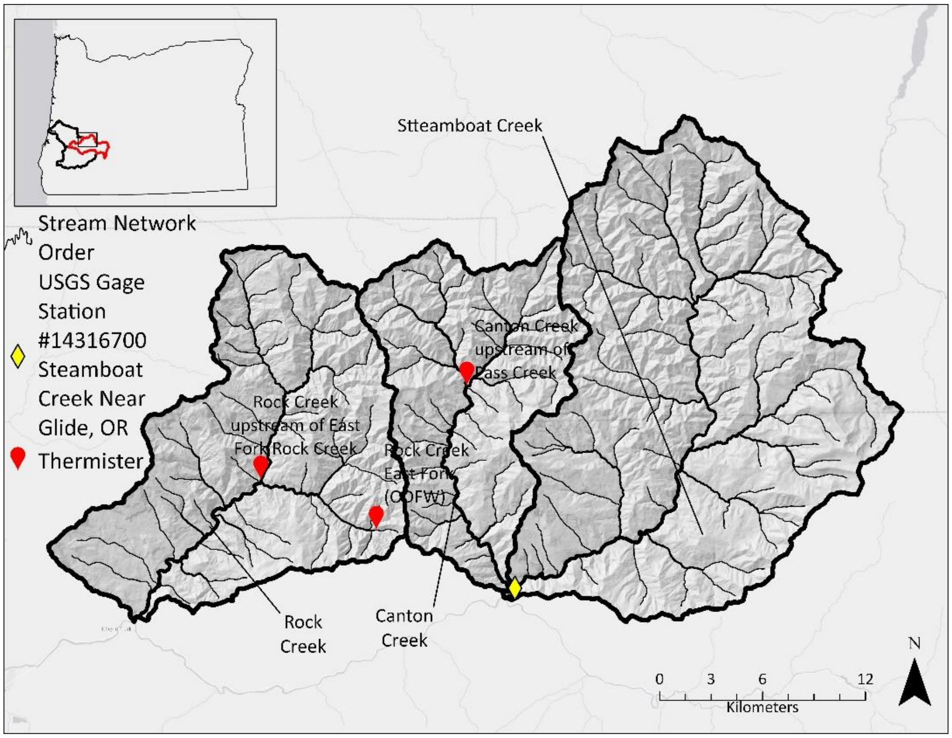


Figure S36. Locations of validation sites for temperature model (red) and flow model (yellow). The Penumbra-VELMA model results for temperature were compared to thermistor data collected at Canton Creek upstream of Pass creek and East Fork Rock Creek. The results for flow model were compared to the only continuous gage in the study area located at the outlet of Steamboat Creek.

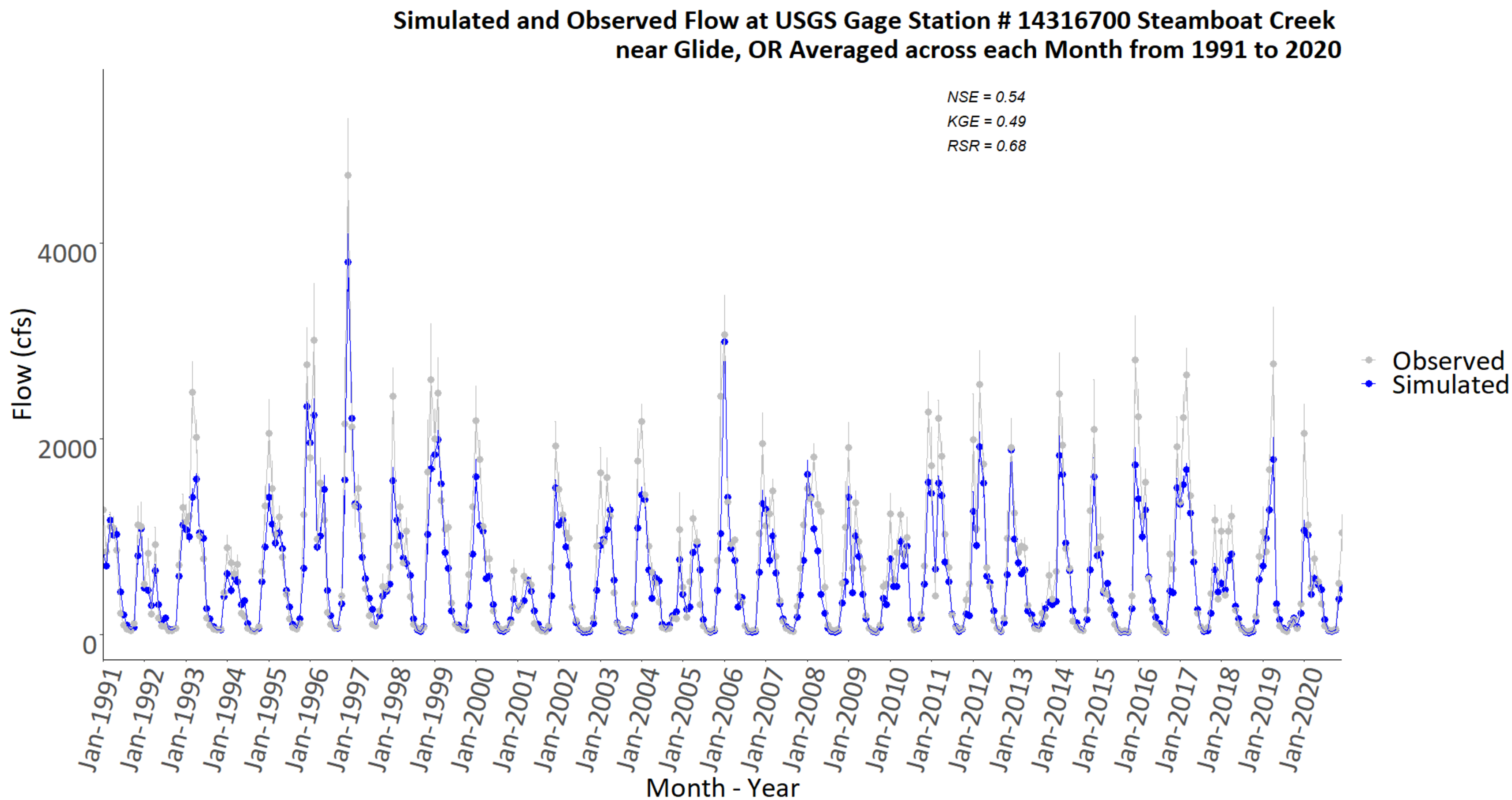
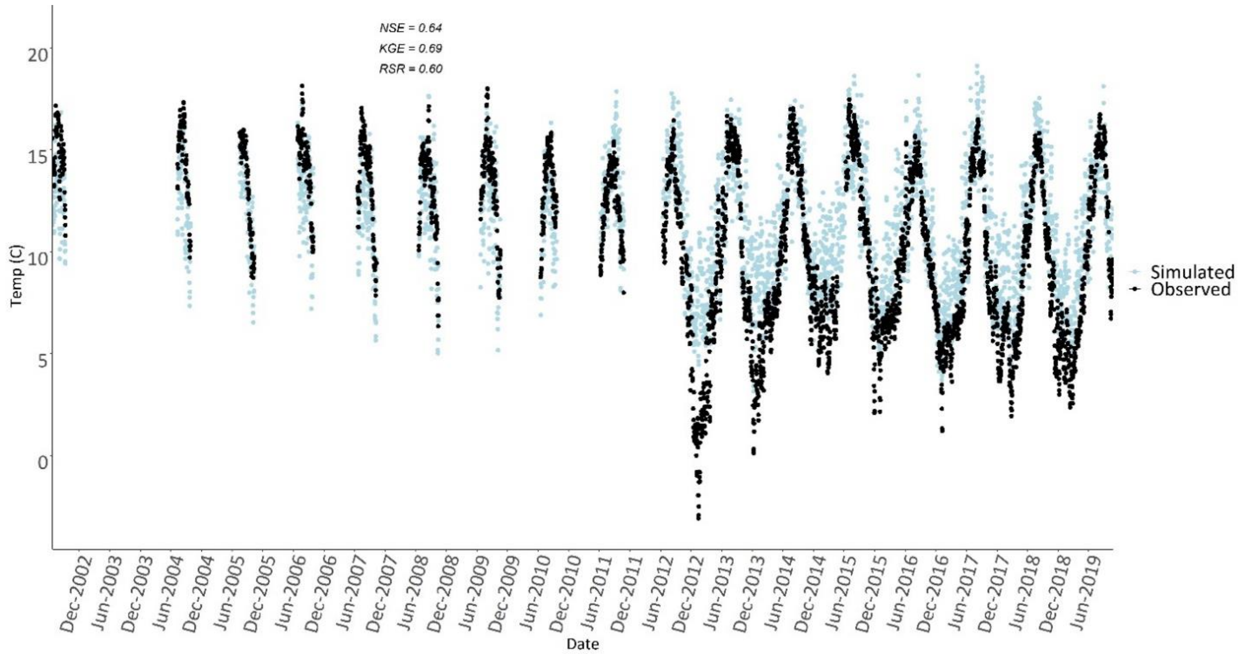


Figure S37. Simulated (blue) and observed (grey) flow at USGS Gage Station # 14316700 (Steamboat Creek near Glide), averaged each month for 1991-2020.



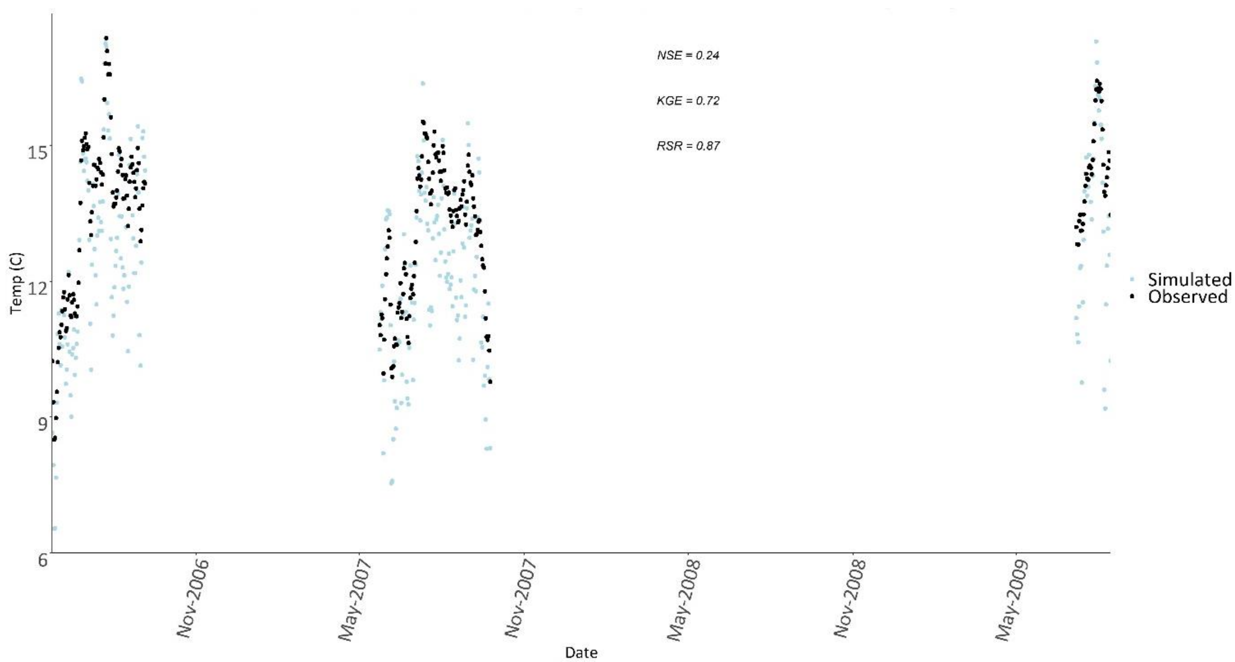


Figure S38. Temperature evaluation compared to available thermistor data for Canton Creek (top) and East Fork Rock Creek (bottom).

OREGON

Fish & Wildlife

4034 Fairview Industrial Drive SE
Salem, OR 97302