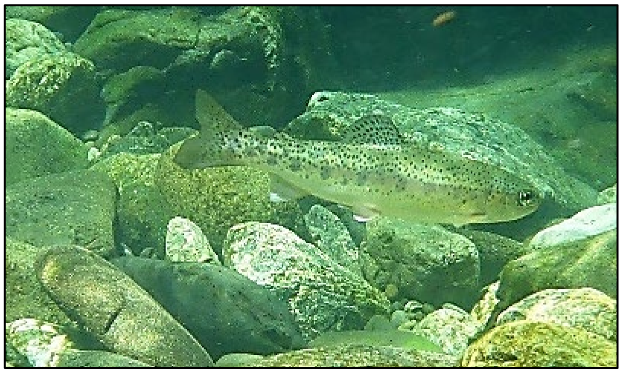
WORKING DRAFT TECHNICAL MEMORANDUM ◦ APRIL 2021

Potter Valley Project Feasibility Study: Potential Ecosystem and Fisheries Responses to Project Alternatives



PREPARED FOR

Potter Valley Project Planning Agreement Parties
California Trout
Humboldt County
Mendocino County Inland Water and Power Commission
Round Valley Indian Tribes
Sonoma County Water Agency

PREPARED BY

Stillwater Sciences
McMillen Jacobs Associates
McBain Associates Applied River Sciences
M3
pH
Geosyntec

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Cover photos (clockwise from top left): overview of critical riffle site on the Eel River, juvenile Chinook Salmon observed on the Eel River, overview of Cape Horn Dam, juvenile *O. mykiss* observed on the Eel River.

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PREFACE

This Technical Memorandum was prepared for the Potter Valley Project Planning Agreement Parties (Parties) by the Consultant Team as part of the Feasibility Study Report. A feasibility study examines the practicability of a potential action or actions in meeting agreed upon common goals. The principal function of a feasibility study is to help potential project proponents focus on potential project components and narrow down the possibilities to determine if the project should be pursued or not. Therefore, a feasibility study reflects a snapshot in time at the beginning of a path that starts with the broad cursory analyses of a feasibility study, transitions toward more refined analyses of an increasingly focused project plan, and hopefully ends with implementation of the best possible project that meets programmatic goals in a cost-effective manner.

The purpose of this Technical Memorandum is to provide feasibility-level technical information on how potential Water Supply Scenarios and infrastructure options could affect both the Eel River and the Russian River from a physical, thermal, and ecological standpoint, and to outline potential fisheries restoration opportunities within and outside the Potter Valley Project. As described in the Alternatives Development and Project Plan Technical Memorandum, various operational and infrastructure options were developed, then bundled into a range of Feasibility Study Alternatives consistent with "dams remain" and "dam(s) removed" categories as desired by the Parties. This Technical Memorandum is not intended to be an exhaustive or comprehensive analysis of all Project options and Feasibility Study Alternatives but describes the initial analyses that the Parties undertook to evaluate the feasibility of assuming ownership of the Project and a potential Project Plan that best met the Shared Objectives of the Planning Agreement.

This Technical Memorandum reflects the consultant work product that was intended to be purely informational and is thus not binding on any of the Parties. Initial ecological evaluations contained in this Technical Memorandum are based on conceptual designs of capital improvements and initial analyses of potential hydrologic and physical changes to the rivers. The initial evaluations incorporate limited input from the Parties and no direct input from resource agencies or other stakeholders, and potential actions contained within the Project Plan are not optimized for the best benefit/cost tradeoffs. In addition, this Technical Memorandum will not be filed with the Federal Energy Regulatory Commission (FERC) as the basis for compliance under the Integrated License Process or other FERC rules. While this Technical Memorandum has contributed to the information available to the Parties, the Parties have not solely relied on this document for justification for any decision the Parties have made or will make regarding the Project, FERC filings, or cooperative agreements. More detailed studies will be conducted (and are currently being conducted) thorough development and implementation of FERC-approved study plans, as well as additional engineering and ecological studies outside of the FERC process. Accordingly, this Technical Memorandum reflects an initial step that will be expanded and built upon in the coming years with additional studies, analysis, synthesis, and ultimately decisions by the Parties on proceeding with a Project Plan for the Potter Valley Project.

1 INTRODUCTION

1.1 Background and Purpose

The Potter Valley Project (Project) is an inter-basin hydroelectric project in the upper Eel River and the upper Russian River watersheds (Figure 1-1). The Project began diverting water in 1908 when Cape Horn Dam and the Van Arsdale Diversion were built. Water from the upper Eel River is diverted into the upper Russian River, where much of it flows through Potter Valley and into Lake Mendocino via the East Branch Russian River (EBRR). Scott Dam, creating Lake Pillsbury, was built in 1922 approximately 12 miles upstream of Cape Horn Dam. The storage in Lake Pillsbury and managed releases downstream allow year-round diversions to occur. Scott Dam is a complete barrier to fish passage, fragmenting fish and aquatic habitat connectivity between reaches upstream and downstream of the dam. In the Russian River watershed, Coyote Valley Dam was completed in 1958 and created Lake Mendocino. Coyote Valley Dam releases water into the Russian River immediately upstream of the confluence with the West Branch Russian River (WBRR). The Eel River and Russian River watersheds are thus inherently connected. Potter Valley and the Russian River are dependent on water diversions from the Eel River, while those same diversions (and dams) likely impair Eel River fish production and ecological function.

Historic anadromous salmon and steelhead populations in the Eel River likely exceeded a million returning salmon and steelhead in good years and have been reduced to about 3,500 fish in recent years (Yoshiyama and Moyle 2010, California Department of Fish and Wildlife [CDFW] 2019, CalTrout 2019). A decline of salmonid populations in the Eel River has been linked to various causes, such as historic logging practices, catastrophic flooding and sediment loading from historic logging practice, and salmonid over-harvesting (Yoshiyama and Moyle 2010). Since its construction, the Project has likely combined with those effects and further contributed to the decline of salmonid populations and ecological conditions in the Eel River. Project effects include:

- Impediments to migration for salmonids and other native fishes (low streamflow and structural);
- Inaccessible habitat (blocked by dam);
- Lentic habitats that support non-native predators (Sacramento Pikeminnow [*Ptychocheilus grandis*] and black bass [*Micropterus spp.*]);
- Altered streamflow and temperature regimes;
- Poor water quality; and
- Reduced ecological function.

However, the Project also may improve some areas for fish habitat by:

- Discharging cool water during summer months between Scott and Cape Horn dams and
- Allowing for the use of “blockwater” to trigger upstream and downstream salmonid migration.

As stated in the Preface, the purpose of this Technical Memorandum is to provide information to the Parties on how proposed Water Supply Scenarios and corresponding infrastructure changes may affect both the Eel River and the Russian River. Section 2 provides an overview of focal species and their life history timing and listing status. Section 3 evaluates different Water Supply Scenarios and compares hydrological, thermal, and geomorphic conditions expected under each.

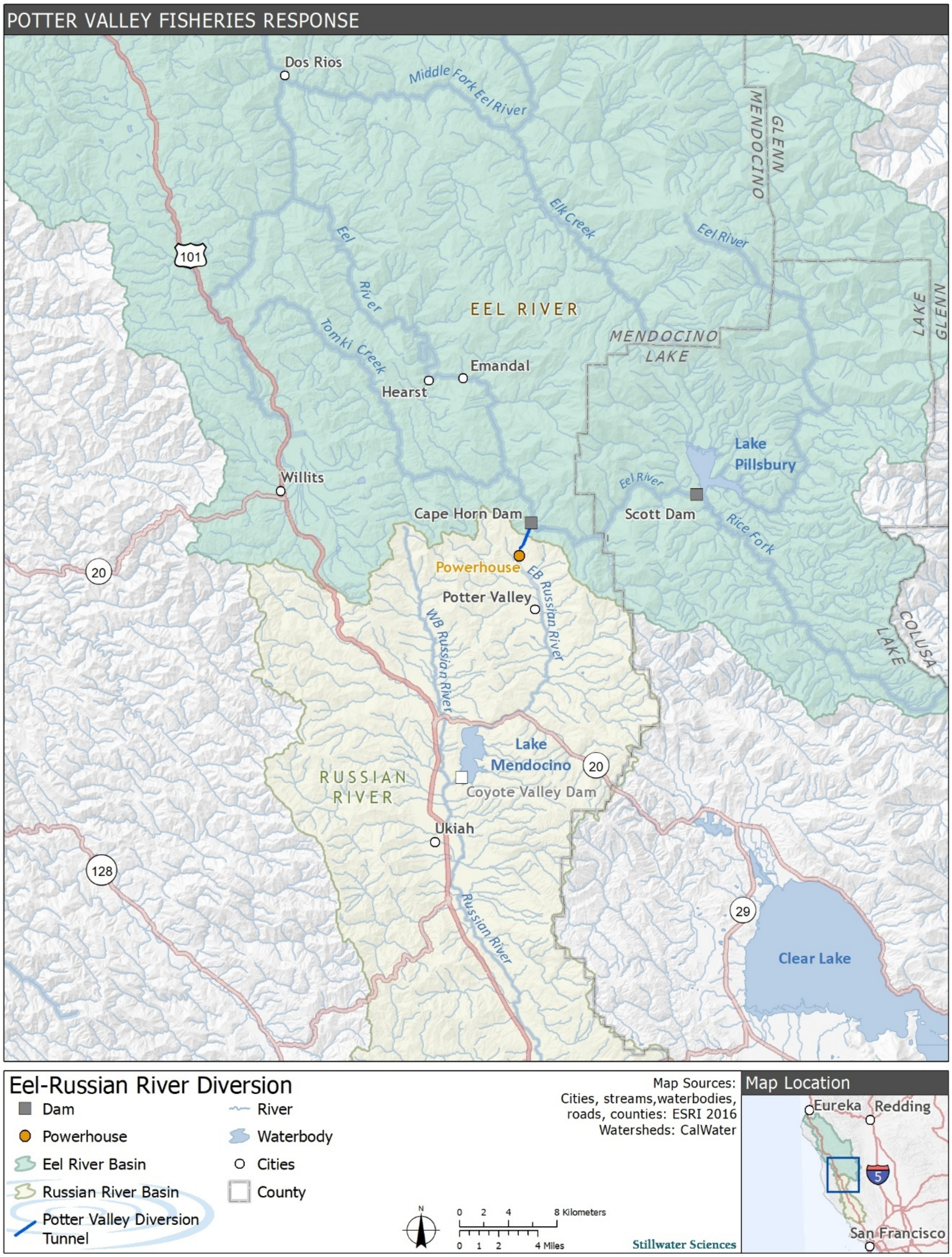

Figure 1-1. Potter Valley Project vicinity.

Section 4 describes various ecological tradeoffs and summarizes expected biological and ecological responses to the different Feasibility Study Alternatives. Section 5 focuses on salmonid productivity trade-offs of the different Feasibility Study Alternatives. Finally, Section 6 outlines a potential framework for developing and implementing key fisheries restoration strategies within the Eel River basin outside of the Project boundary (and excluded from obligations of the new FERC license).

1.2 Overall Approach

We first describe expected and predicted hydrologic, water temperature, and geomorphic differences, and then analyze ecological and salmonid productivity tradeoffs in response to the Water Supply Scenarios and associated infrastructure changes developed by the Ad Hoc Committee Water Supply Working Group (Table 1-1). Analyses focused on differences between:

- Baseline Scenario (current operations; referred to as Baseline hereafter)
- Scenario 4B (modified Reasonable and Prudent Alternatives (RPA)/FIRO and fish flows)
- Scenario 2 (run-of-the-river/Forecast Informed Reservoir Operations [FIRO] and fish flows)
- Scenario 3 (decommissioning)

This approach allowed comparison of future conditions to both existing conditions and the condition in which the Project is not relicensed by the Parties. In addition to Water Supply Scenarios, we also evaluated tradeoffs related to the Feasibility Study Alternatives (see the Alternatives Analysis and Project Plan Technical Memorandum) to account for potential infrastructure changes (Table 1-2). When appropriate, we compared:

- Alternative 1 (Scott Dam and Cape Horn Dam remain)
- Alternative 2 (Scott Dam removed and Cape Horn Dam remains)
- Alternative 3 (Scott Dam and Cape Horn Dam removed with alternative diversion infrastructure at or upstream of Van Arsdale Diversion intake)

For additional information on both the Water Supply Scenarios and Feasibility Study Alternatives, refer to the Alternatives Description and Project Plan Technical Memorandum developed for the Feasibility Study (Stillwater Sciences et al. 2021).

Table 1-1. Matrix of Water Supply Scenarios modified from the Ad Hoc Committee Water Supply Working Group (Addley et al. 2019).

<table>
<tr><th colspan="3" rowspan="2">Modeling Scenarios</th><th colspan="3">Russian River & Lake Mendocino Alternatives</th></tr>
<tr><th>Current Operations</th><th>Lake Mendocino FIRO (Hybrid) with Fish Flow Project Operations[1]</th><th>Raise Coyote Valley Dam</th></tr>
<tr><td rowspan="6">Potter Valley Project Alternatives</td><td rowspan="3">Dams Remain</td><td rowspan="2">Current Operations[2]</td><td>Baseline: Existing Climate (n=1)</td><td></td><td rowspan="2"></td></tr>
<tr><td>Baseline FC: Future Climate (n=4)</td><td></td></tr>
<tr><td>Revised Operations[3]</td><td>Scenario 4: Existing Climate (n=1)</td><td>Scenario 4B: Existing Climate (n=1)</td><td></td></tr>
<tr><td rowspan="3">Dam(s) Removed</td><td rowspan="2">Run-of-the-river[4]</td><td></td><td>Scenario 2: Existing Climate (n=1)</td><td></td></tr>
<tr><td></td><td>Scenario 2FC: Future Climate (n=4)</td><td></td></tr>
<tr><td>Decommission[5]</td><td>Scenario 1: Existing Climate (n=1)</td><td>Scenario 3: Existing Climate (n=1)</td><td>Scenario 5: Preliminary analysis, Existing Climate</td></tr>
</table>

Note: Red outlined boxes are potential future Water Supply Scenarios analyzed by this technical memorandum, and green outlined boxes are potential Baseline and Potter Valley Project (Project) Decommission Water Supply Scenarios analyzed in the hydrology section of this technical memorandum.

[1] Lake Mendocino Forecast Informed Reservoir Operations (FIRO) and Sonoma County Water Agency Fish Habitat Flows and Water Rights Project Draft Environmental Impact Report (State Clearinghouse No. 2010092087) (Fish Flow EIR) Assumptions: Maximum allowed reservoir elevation during November-March flood reserve space raised from 68,400 acre-feet (ac-ft) to 80,050 ac-ft. Reduces Lake Mendocino releases in all years except driest year by up to 80 cubic feet per second (cfs). Achieve unmet Potter Valley Irrigation District (PVID) demands (up to 15,320 ac-ft) via PVID pumpback from Lake Mendocino.

[2] Current operations: Scott Dam and Cape Horn Dam stay in place, streamflows and diversions based on 2002 Biological Opinion Reasonable and Prudent Alternative (RPA) flows, maximum diversion = 170 cfs based on model calibration mass balance. Russian River flows based on 2008 Biological Opinion RPA and 1986 Decision 1610, existing flood control rule curve (no FIRO).

[3] Project Revised Operations Assumptions: 1) allow discretionary Project diversions when Scott Dam is spilling up to 170 cfs, 2) reduce Eel River minimum instream flow “floor” by up to 50 cfs in winter and spring, and 3) reduce minimum instream flow on the East Branch Russian River year-round by various amounts for different water year types.

[4] Run-of-the-river Assumptions: Remove Scott Dam; continue Van Arsdale diversions with a maximum Project diversion of 300 cfs resulting from capital projects that improve diversion reliability; achieve unmet PVID demands (up to 15,320 ac-ft) via in-valley storage, aquifer storage and recovery, pumpback from Lake Mendocino, or other means.

[5] Decommission Assumptions: Scott Dam, Cape Horn Dam, and Project Diversion would be completely removed, no water diversions from Eel River to Russian River, Eel River streamflows would be unimpaired.

Table 1-2. Potter Valley Project Feasibility Study Alternatives and corresponding Water Supply Scenarios.

Feasibility Study Alternative:	**Existing Conditions**	**Alternative 1**	**Alternative 2**	**Alternative 3**	**Decommission**
Water Supply Scenario:	**Baseline**	**Scenario 4B**	**Scenario 2**	**Scenario 2**	**Scenario 3**
Disposition of Dams and Diversion	• Scott Dam Remains • Cape Horn Remains • Existing Diversion	• Scott Dam Remains • Cape Horn Remains • Existing Diversion	• Scott Dam Removed • Cape Horn Remains • Existing Diversion	• Scott Dam Removed • Cape Horn Removed • Alternative Diversion	• Scott Dam Removed • Cape Horn Removed • No Diversion

2 FOCAL SPECIES

This assessment focused on fall-run Chinook Salmon (*Oncorhynchus tshawytscha*) and summer-run and winter-run steelhead (*O. mykiss*). We also assessed Pacific Lamprey (*Entosphenus tridentatus*), Foothill Yellow-legged Frogs (*Rana boylii*), and Northwestern Pond Turtles (*Actinemys marmorata)*. These species are either listed as Threatened or as a Species of Concern under the Federal or California State Endangered Species Acts (Table 2-1). Coinciding with listing status, these species were focused on because are they are most impacted by the operations and infrastructure of the Project. In addition to direct analysis on these species, we examined other important aspects of the ecosystem that are affected by the Project and influence ecological and salmonid health. For example, we assessed how different scenarios would either facilitate or inhibit populations of non-native predatory fish (Sacramento Pikeminnow, Largemouth Bass [*Micropterus salmoides*], and Smallmouth Bass [*Micropterus dolomieu*]) found in the upper Eel and Russian rivers. We further analyzed processes (e.g., geomorphic) or habitats (e.g., riparian zones) that support focal species and will be affected by changes to streamflow and infrastructure corresponding to the Feasibility Study Alternatives.

Table 2-1. Federal and California State listing status of species discussed in this document.

Species (population)	Listing status[1]	Known presence by watershed[2]	
		Eel River	Russian River
Fall-run Chinook Salmon (California Coastal ESU)	FT	N	N
Summer- and Winter-run Steelhead (Northern California DPS)	FT, SSC, SC	N	NA
Winter-run Steelhead (Central California Coast DPS)	FT	NA	N
Coho Salmon (Southern Oregon/Northern California Coast ESU)	FT, ST	N	NA
Pacific Lamprey	SSC, FSS	N	N
Sacramento Pikeminnow	None	I	N
Largemouth Bass	None	I	I
Smallmouth Bass	None	X	I
Foothill Yellow-legged Frog	SSC, FSS	N	N
Northwestern Pond Turtle	SSC, FSS	N	N
American Bullfrog	None	I	I
Red-eared Sliders	None	X	I

Notes: ESU = evolutionarily significant unit; DPS = distinct population segment

[1] FT = Federally listed as Threatened, FE = Federally listed as Endangered, FSC = Federal Species of Concern, FC = Federal candidate species, ST = State Threatened, SSC = California Species of Special Concern, SC = State candidate species, FSS = USDA Forest Service Sensitive.

[2] N=Native, I = Introduced, NA = not within the range of the ESU, X = Not present.

Understanding the life history timing of focal species is important for assessing the effects of various Feasibility Study Alternatives. Generalized life history timing summaries are provided below for Chinook Salmon, steelhead, Pacific Lamprey, and Coho Salmon (Table 2-2 through Table 2-7).

Table 2-2. Generalized life history timing of fall-run Chinook Salmon in the Eel River watershed.

Life stage	Month											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Adult migration[1,2,3,4]	░									░	▓	▓
Spawning[1,2,3]	░	░									▓	▓
Incubation[5]	░	░	░	░							░	░
Fry rearing[1,2,5,6]		░	░	░	░	░						
Juvenile outmigration[1,2,6]			░	▓	▓	░	░					

[1] VTN (1982)

[2] SEC (1998)

[3] PG&E (2017)

[4] CDFW unpublished data (1996–2017)

[5] Moyle et al. 2017 and assumed based on spawning time and presence of fry reported by VTN (1982) and Beak (1986)

[6] Beak (1986)

░ = Span of activity

▓ = Peak of activity

Table 2-3. Generalized life history timing of winter-run steelhead in the Eel River watershed.

Life stage	Month											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Adult migration[1,2,3]	▓	▓	▓	░	░						░	░
Spawning[1,3,4]	▓	▓	▓	░	░							░
Adult outmigration (kelt)[2,5,6]		░	░	░	░							
Incubation[7,8]	░	░	░	░	░	░						
Fry[9]			░	░	░	░	░					
Juvenile rearing[1,2,6]	░	░	░	░	░	░	░	░	░	░	░	░
Smolt outmigration[1,2,10]			░	▓	▓	░						

[1] VTN (1982)
[2] SEC (1998)
[3] CDFW unpublished data (1996–2017)
[4] Busby et al. (1996)
[5] Teo et al. 2013
[6] Moyle et al. 2017
[7] Shapovalov and Taft (1954)
[8] Barnhart (1991)
[9] assumed based on expected time of emergence from redd gravels
[10] Beak (1986)

░ = Span of activity
▓ = Peak of activity

Table 2-4. Generalized life history timing of summer-run steelhead in the Eel River watershed.

Life stage	Month											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Summer-run adult migration[1,2,3]			░	▓	▓	░						
Summer-run holding[3,4,5]						░	░	░	░	░		
Fall-run and half-pounder entry and holding in lower Eel River [4,6,7]							░	░	░	░		
Spawning[3,4,5]	▓	▓	▓	░								░
Adult outmigration (kelt)[3,8,9]		░	░	░	░							
Incubation[5,10]	░	░	░	░	░							
Fry[11]			░	░	░	░	░					
Juvenile rearing[3,8]	░	░	░	░	░	░	░	░	░	░	░	░
Smolt outmigration[8,12,13]			░	▓	▓	░						

[1] Everest (1973)
[2] Busby et al. (1996)
[3] Moyle et al (2017)
[4] Roelofs (1983)
[5] Barnhart (1991)
[6] Kajtaniak and Gruver (2020)
[7] Hodge et al. (2014)
[8] SEC (1998)
[9] Teo et al. (2013)
[10] Shapovalov and Taft (1954)
[11] assumed based on expected time of fry emergence
[12] VTN (1982)
[13] Beak (1986)

░ = Span of activity
▓ = Peak of activity

Table 2-5. Generalized life history timing of Pacific Lamprey (river-maturing ecotype) in the Eel River watershed.

Life stage	**Month**											
	Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Adult migration[1,2,3,4]	▓	▓	▓	▓	▓	░	░				░	░
Pre-spawning holding[4,5,6]	░	░	░			░	░	░	░	░	░	░
Spawning[1,2]			░	▓	▓	░	░					
Incubation[8,9]			░	░	░	░	░					
Larval drift and settlement[10,11,12]				░	░	░	░	░				
Larval rearing[2] (ammocoete)	░	░	░	░	░	░	░	░	░	░	░	░
Macrophthalmia outmigration[14,15]	░	░	░	░	░						░	░

[1] Stillwater Sciences (2010)
[2] Stillwater Sciences and WNRD (2016)
[3] Parker (2018)
[4] Robinson and Bayer (2005)
[5] Clemens et al. (2012)
[6] McCovey (2011)
[7] Starcevich et al. (2014)
[8] Meeuwig et al. (2005)
[9] Brumo (2006)
[10] Harvey et al. (2002)
[11] White and Harvey (2003)
[12] Brumo et al. (2009)
[14] van de Wetering (1998)
[15] Goodman et al. (2015)

░ = Span of activity
▓ = Peak of activity

Table 2-6. Generalized life history timing of Coho Salmon in the Eel River watershed, based primarily on the South Fork Eel River and other northern California streams.

Life stage	**Month**											
	Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Adult migration[1,2]	▓	░									░	▓
Spawning[1,2]	▓	░									░	▓
Incubation[3,4]	░	░	░	░	░						░	░
Juvenile rearing[1,4]	░	░	░	░	░	░	░	░	░	░	░	░
Smolt outmigration[1,5,6]			░	▓	▓	░						

[1] Ricker et al. (2014)
[2] Guczek et al. (2019)
[3] Murray and McPhail (1988)
[4] Moyle et al. (2017)
[5] Mendocino Redwood Company (2002)
[6] Vaughn (2005)

░ = Span of activity
▓ = Peak of activity

Table 2-7. Generalized life history timing of Chinook Salmon, steelhead, and Pacific Lamprey in the upper Russian River watershed.

Species	**Life stage**	**Month**											
		Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Chinook Salmon (fall-run)	Adult migration[1,2]	░								░	░▓	▓░	░
	Spawning[1,2]	░										▓	▓
	Incubation[1,2]	▓										▓	▓
	Juvenile rearing[1,2]			░	░	░							
	Smolt outmigration[1,2]		░	░	░▓	▓░	░	░					
Steelhead (winter-run)	Adult migration[1,2]	▓	▓	▓	░	░							░
	Spawning[1,2]	░	░	░	░								░
	Incubation[1,2]	░	░	░	░								░
	YOY (Age - 0) rearing[1]	░	░	░▓	▓	▓	░	░	░	░	░	░	░
	Juvenile (Age - 1+) rearing[1]	░	░	░	░	░	░	░	░	░	░	░	░
	Smolt outmigration[1,2]	░	░	▓	▓	▓	▓░						░
Pacific Lamprey	Spawning[2]	░	░	░									
	Incubation[2]	░	░	░	░								
	Emergence[2]		░	░	░	░							
	Larval rearing[2] (ammocoete)	░	░	░	░	░	░	░	░	░	░	░	░
	Macrophthalmia outmigration[2]	░	░									░	░

[1] ESA 2015

[2] SCWA 2016

░ = Span of activity

▓ = Peak of activity

3 PHYSICAL AND THERMAL ANALYSES

The direct physical effects of dams on rivers are generally to the hydrology, thermal regime, and geomorphic structure of the river (Poff et al. 2007; Magilligan and Nislow 2005; Palmer and Ruhi 2019). The changes to these physical and thermal river characteristics are what drives change to fish and aquatic ecological function (Bunn and Arthington 2002; Power et al. 1996). In this section we assess the hydrological, thermal, and geomorphic changes that will be the foundation for the Ecological Analyses (Section 4) and the Salmonid Productivity Tradeoffs Analysis (Section 5).

3.1 Hydrological Analysis

The Jared Huffman Ad Hoc Committee Water Supply Working Group prepared a HEC-ResSim model which predicted streamflow at various locations of interest in the Russian River and Eel River (Addley et al. 2019). The objective of Section 3.1 is to highlight the key aspects of hydrology that will affect ecological processes and salmonid productivity in the Eel and Russian rivers in response to different Feasibility Study Alternatives and Water Supply Scenarios

3.1.1 Approach

We evaluated each Water Supply Scenario from HEC-ResSim model hydrology data for the Eel River and the Russian River by developing an automated script in the R statistical programing language. We used an approach that summarized hydrology to estimate changes in streamflow regimes based on:

- Daily average records for the entire period of record for two water year types to analyze general changes;
- Using an example wetter (2011) and drier (2015) water year type to analyze specific changes on a year-to-year basis; and
- The flood frequency changes based on daily average streamflows.

The daily average record analysis for both the Eel River and the Russian River evaluated daily average streamflow from water years 1911 to 2017 and compared wetter water years (n=43) and drier water year (n=28) on the Eel River, and normal water years (n=95) and drier water years (n =12) on the Russian River. The Russian River evaluation used normal water years as opposed to wetter water years because no water years in the period of record were classified as wet.

The flood frequency analysis followed methods outlined in U.S. Geological Survey (USGS) Bulletin 17C (England et al. 2019) to rank and assign recurrence intervals to yearly peak flows from water years 1911 to 2017 for both the Eel River and the Russian River and develop a log-Pearson III fit flood frequency curve. Daily average flow data was used for the analysis since the model does not output instantaneous peak values.

Modeled streamflow used for the hydrological analysis was from the following HEC-ResSim computational nodes: downstream of Scott Dam (gage E-2), downstream of Cape Horn Dam (gage E-11), the Russian River immediately downstream of the WBRR confluence, and the Russian River at Hopland (Figure 3-1). These locations were selected because changes to streamflow there were determined to be greatest and potentially have the most effect on ecological function on both rivers.

The analysis was run for Scenario 2 (Run-of-the-river/FIRO and fish flows), Scenario 4B (Modified RPA/FIRO and fish flows), Scenario 3 (Decommission), and Baseline (Current Operations) as shown in Table 1-1. To reduce clutter, the Water Supply Scenarios were plotted as follows:

1. Scenario 3 with Scenario 4B, and
2. Baseline with Scenario 4B and Scenario 2.

We analyzed each Water Supply Scenario using four key hydrograph components that can affect the ecological function and salmonid productivity in rivers:

1. Winter and spring peak flows;
2. Spring recession limb;
3. Summer baseflow; and
4. Fall streamflows.

For each location, we summarized the key changes to the streamflow regime that are anticipated for each Water Supply Scenario.

3.1.2 Eel River from Scott Dam to Cape Horn Dam (gage E-2)

Each Water Supply Scenario is meaningfully different from one another between Scott Dam and Cape Horn Dam (except for Scenario 2 and 3, which are the same). These hydrologic changes likely result in important ecological effects on the Eel River between the two dams. If Scott Dam is removed under Scenario 2 or Scenario 3, then hydrological changes associated with unimpaired streamflow will be most pronounced for the ecological function of the Eel River from Scott Dam to Cape Horn Dam. Downstream of Cape Horn Dam, Scenario 2 and Scenario 3 would differ because Scenario 2 would include diversion into the Russian River, while Scenario 3 would cease diversion and reflect unimpaired streamflows. Downstream of Scott Dam, Scenario 2 and Scenario 3 are the same hydrology because Scott Dam would be removed under both Scenario 2 and Scenario 3. For this reason, Scenario 2 was not plotted to reduce clutter on figures below.

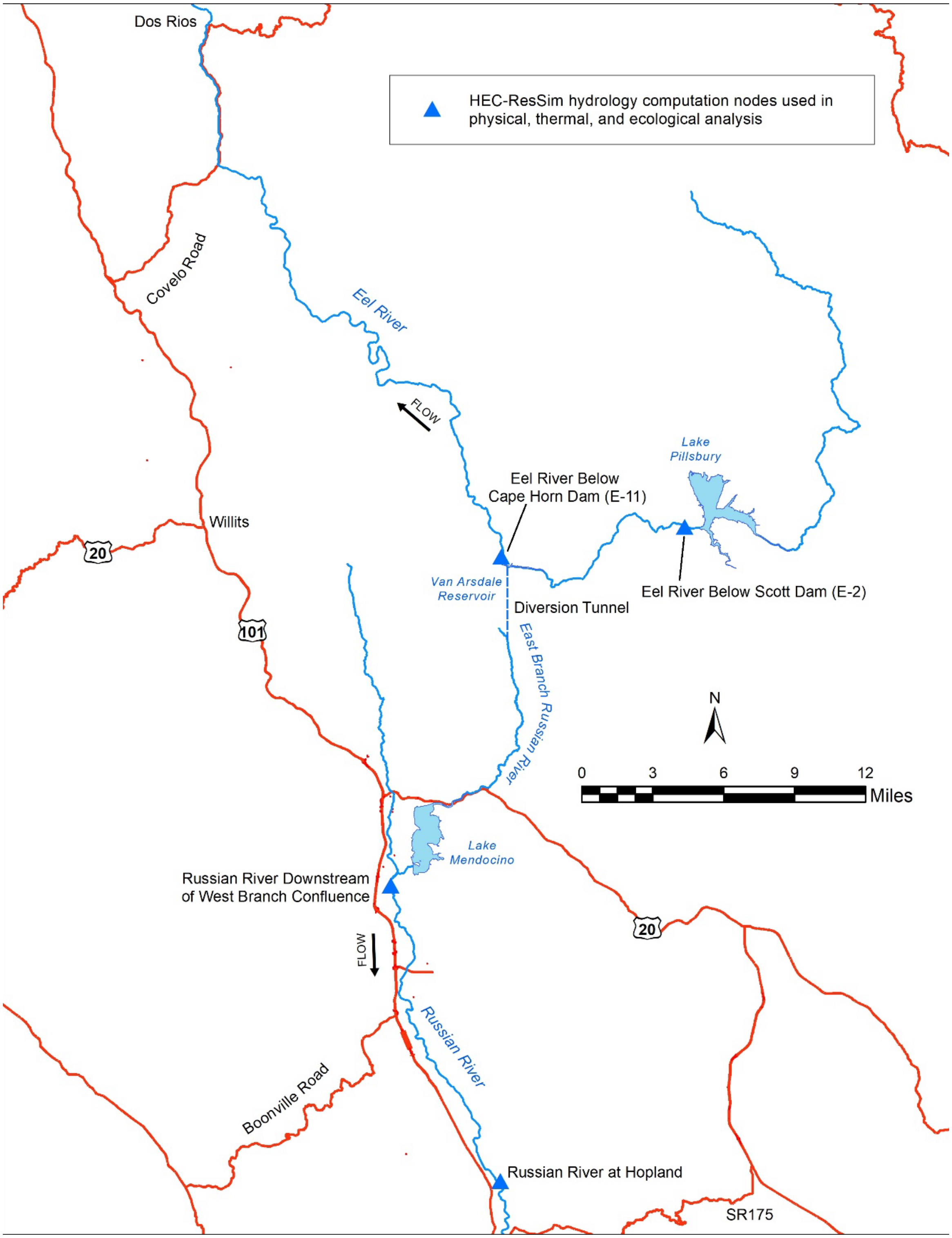

Figure 3-1. HEC-ResSim Hydrology computation nodes used in the hydrological analysis.

3.1.2.1 Daily Average Record

Winter and spring peak flows (Figure 3-2)

- Winter and spring peak flows are typically higher under Scenario 2 and Scenario 3 (Dam(s) Removed) than Baseline and Scenario 4B (Dams Remain).
- Spring storms result in higher magnitude streamflows under Scenario 2 and Scenario 3 than Scenario 4B, which retains Scott Dam.
- All trends are similar in both wetter and drier years, but there are higher magnitude differences in peak flows under wetter years when comparing Scenario 2 and Scenario 3 to Baseline or Scenario 4B.

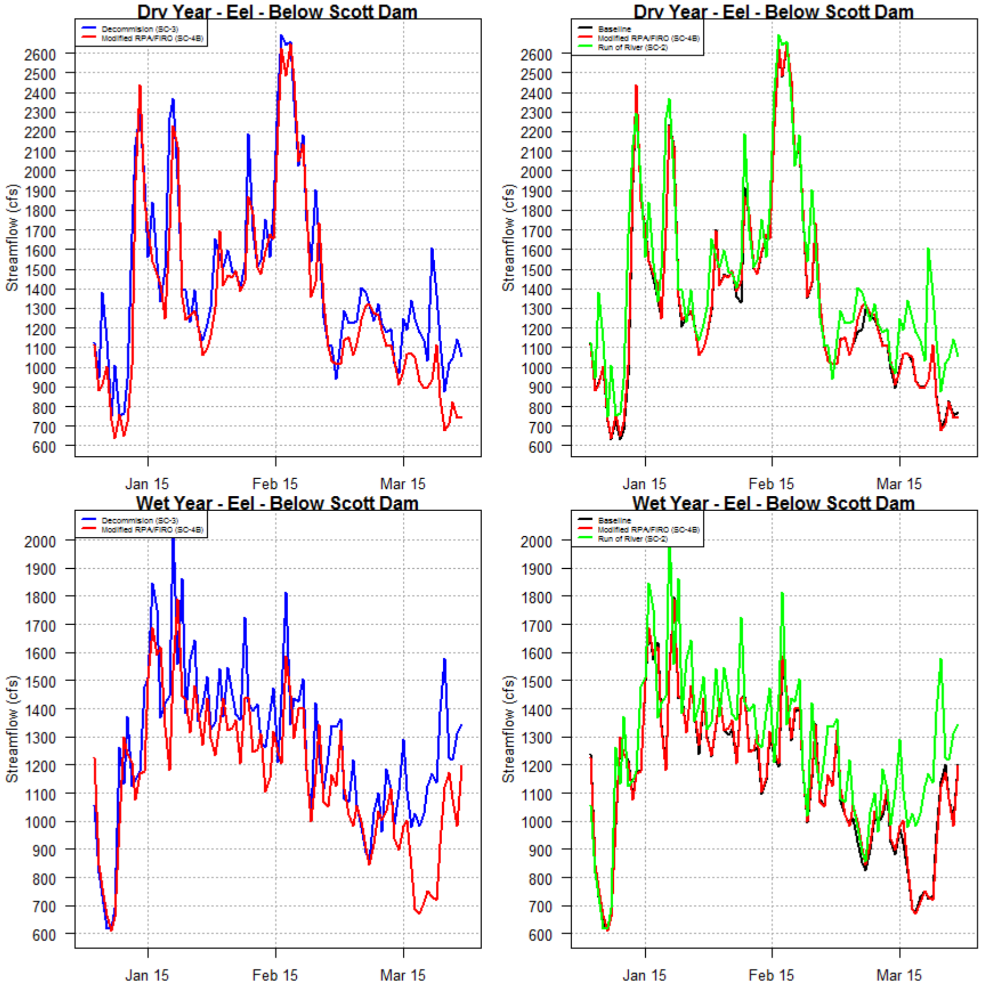

Figure 3-2. Daily average streamflow for drier (dry and very dry) and wetter (wet and very wet) years on the Eel River downstream of Scott Dam from water years 1911–2017 during winter and spring storm period. Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right. Scenario 2 is not presented on the left column of plots because it is identical to Scenario 3 at this location.

Spring recession (Figure 3-3)

- The spring recession limb is shorter for Scenario 4B and Baseline (i.e., baseflow is reached earlier) than Scenario 2 and Scenario 3.
- Under Scenario 4B and Baseline, which retain Scott Dam, there are higher late spring recession streamflows during late spring (May and June) than Scenario 2 and Scenario 3, where Scott Dam is removed.
- For wetter water year types, the spring recession lasts longer under Scenario 2 and Scenario 3 than under Scenario 4B and Baseline (i.e., base streamflow is reached later).

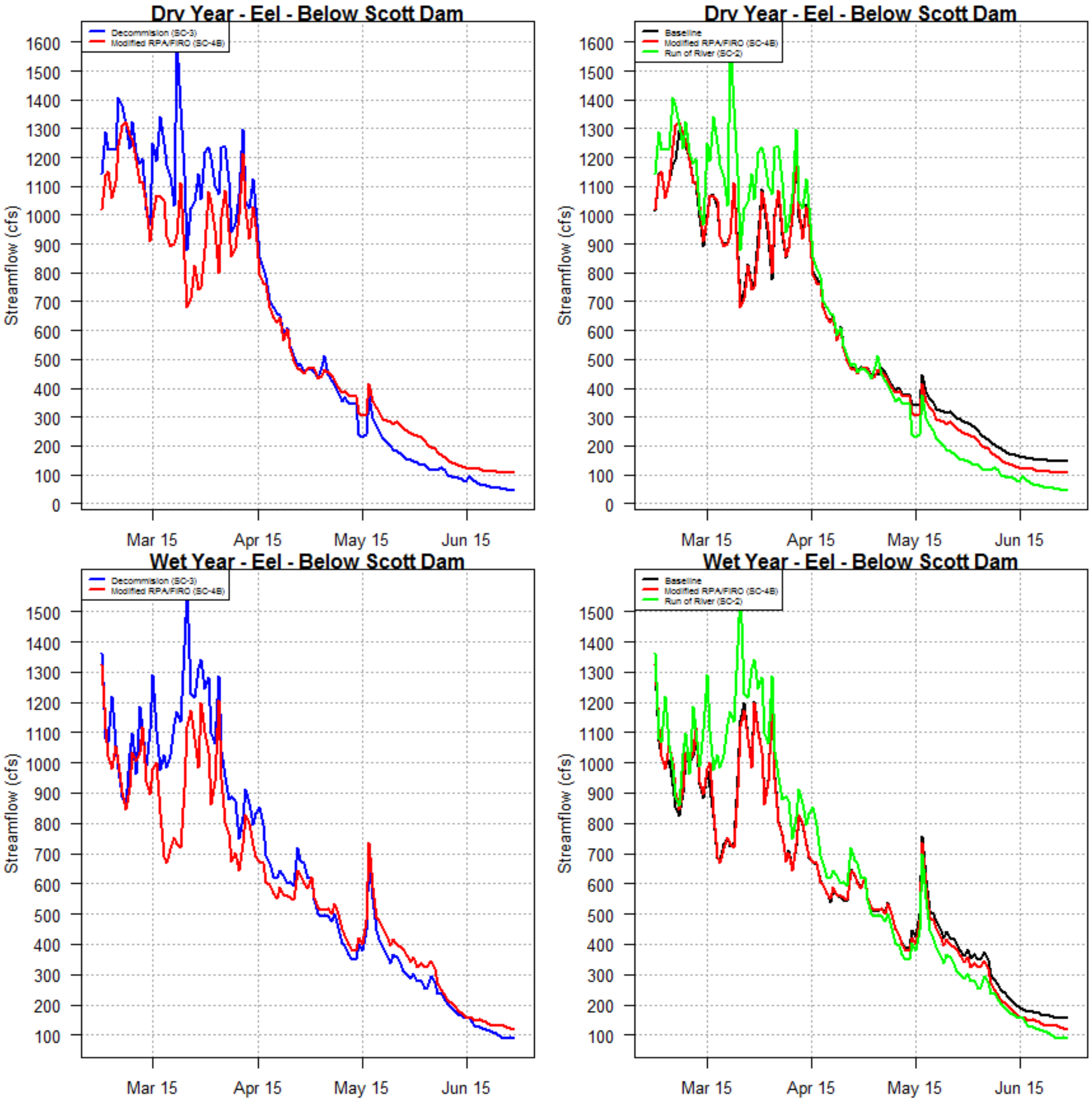

Figure 3-3. Daily average streamflow for drier and wetter years on the Eel River downstream of Scott Dam from water years 1911–2017 during spring recession period. Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right. Scenario 2 is not presented on the left column of plots because it is identical to Scenario 3 at this location.

Summer baseflows (Figure 3-4)

- Summer baseflows are approximately 100 cubic feet per second (cfs) higher under Scenarios 4B and Baseline than under Scenario 2 and Scenario 3.
- Scenario 4B has increasing streamflows earlier in the summer for wetter years than Baseline or Scenario 2 and Scenario 3.

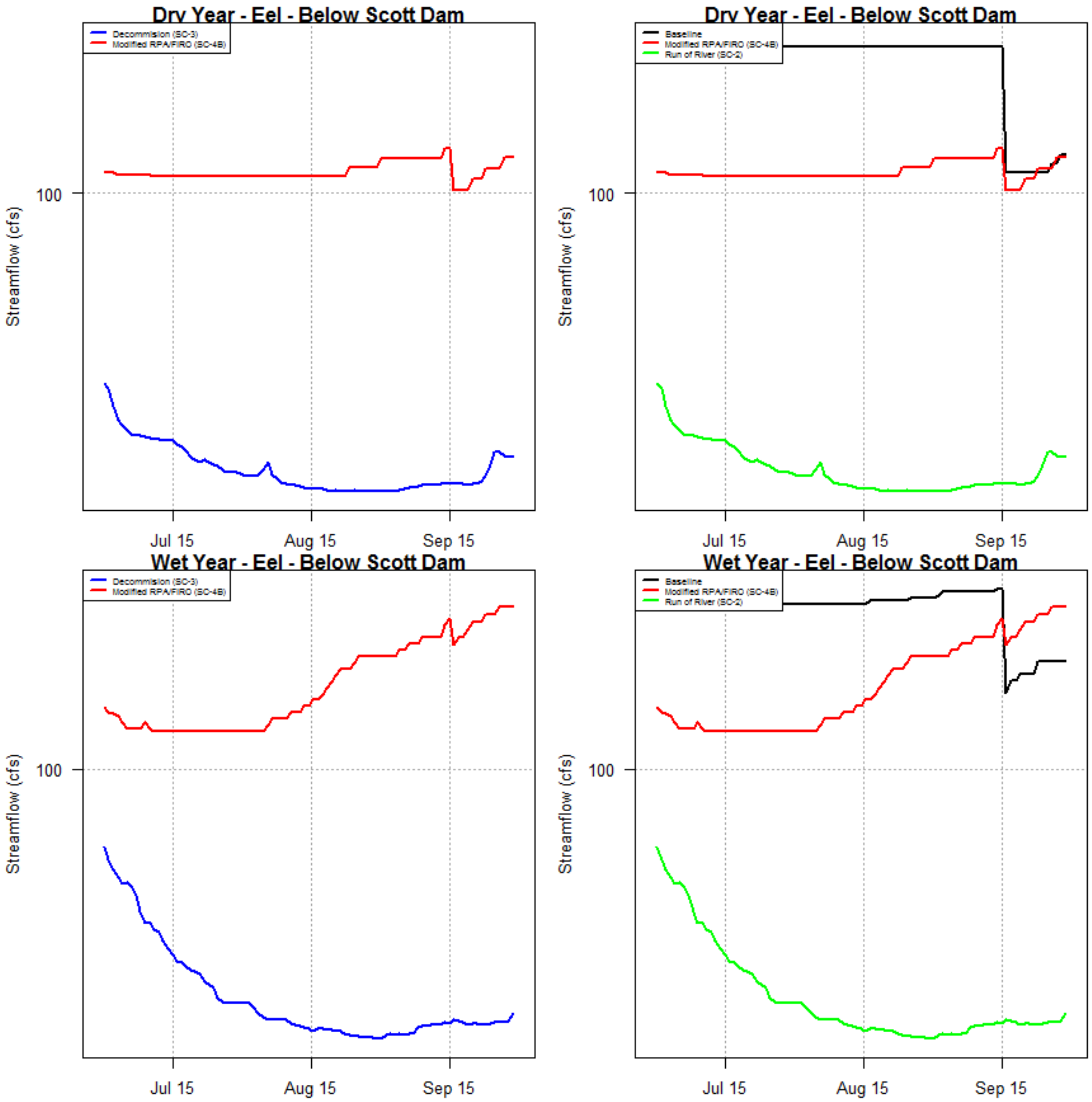

Figure 3-4. Daily average streamflow for drier and wetter years on the Eel River downstream of Scott Dam from water years 1911–2017 during summer baseflow period. Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right. Scenario 2 is not presented on the left column of plots because it is identical to Scenario 3 at this location.

Fall streamflows (Figure 3-5)

- Baseflow in early fall (October to mid-November) is higher under Scenario 4B and Baseline than Scenario 2 and Scenario 3 for both wetter and drier water years.
- Peak flows in late fall (mid-November through December) are higher under Scenario 2 and Scenario 3 than under Scenario 4B and Baseline for wetter water years.
- Late Fall streamflows are similar in magnitude under all Water Supply Scenarios for drier water years, while early fall streamflows are truncated by Scott Dam.

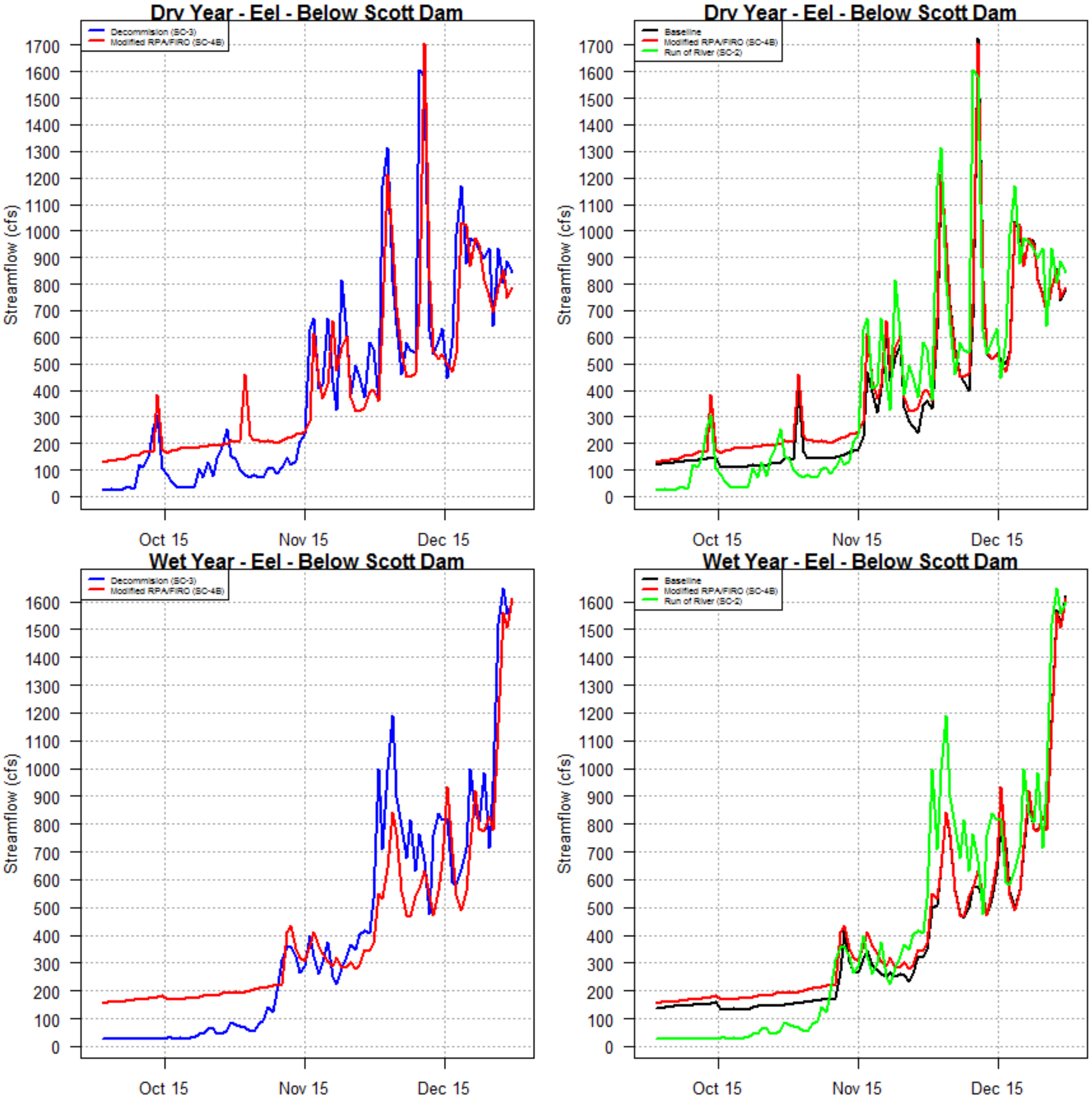

Figure 3-5. Daily average streamflow for drier and wetter years on the Eel River downstream of Scott Dam from water years 1911–2017 during fall streamflow period. Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right. Scenario 2 is not presented on the left column of plots because it is identical to Scenario 3 at this location.

3.1.2.2 Example Water Years (2011 and 2015)

Winter and spring peak flows (Figure 3-6)

- Winter and spring peak flows are 1,000 cfs to 1,500 cfs higher under Scenario 2 and Scenario 3 for the drier water year, and over 3,000 cfs higher for wetter water year.
- Baseflow is slightly higher under Scenario 4B and Baseline for the drier water year.

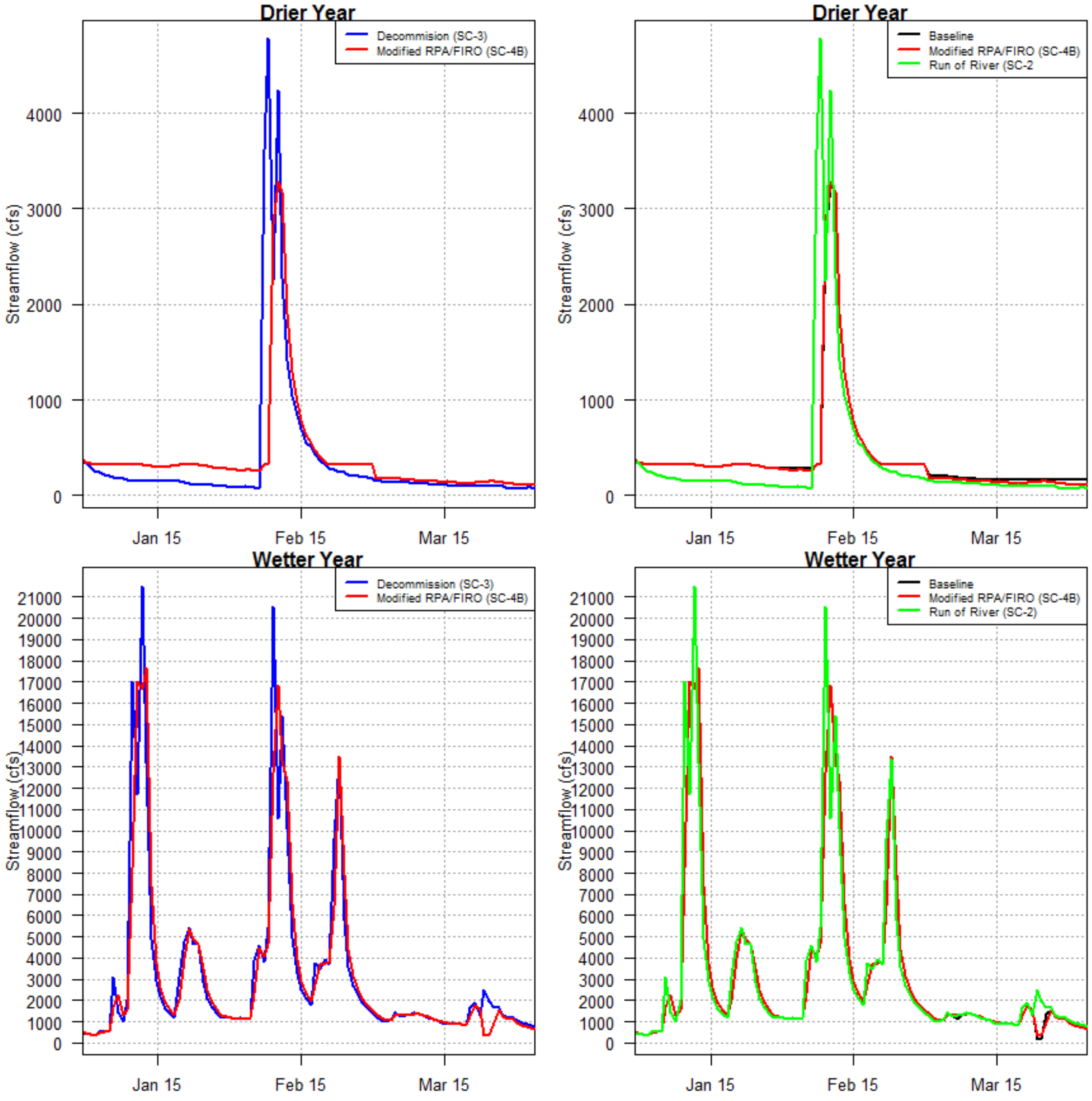

Figure 3-6. Example water years for drier (2015) and wetter (2011) years on the Eel River downstream of Scott Dam during spring and winter storm period. Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right. Scenario 2 is not presented on the left column of plots because it is identical to Scenario 3 at this location.

Spring recession (Figure 3-7))

- The spring recession limb is roughly the same duration for all Water Supply Scenarios in the drier water year.
- Under Scenario 4B and Baseline, spring recession streamflow is higher during later spring (May and June) than under Scenario 2 and Scenario 3, but the magnitudes of peak flows events are higher.
- For the wetter water year, the spring recession lasts longer under Scenario 2 and Scenario 3 than under Scenario 4B and Baseline (i.e., base streamflow is reached later).

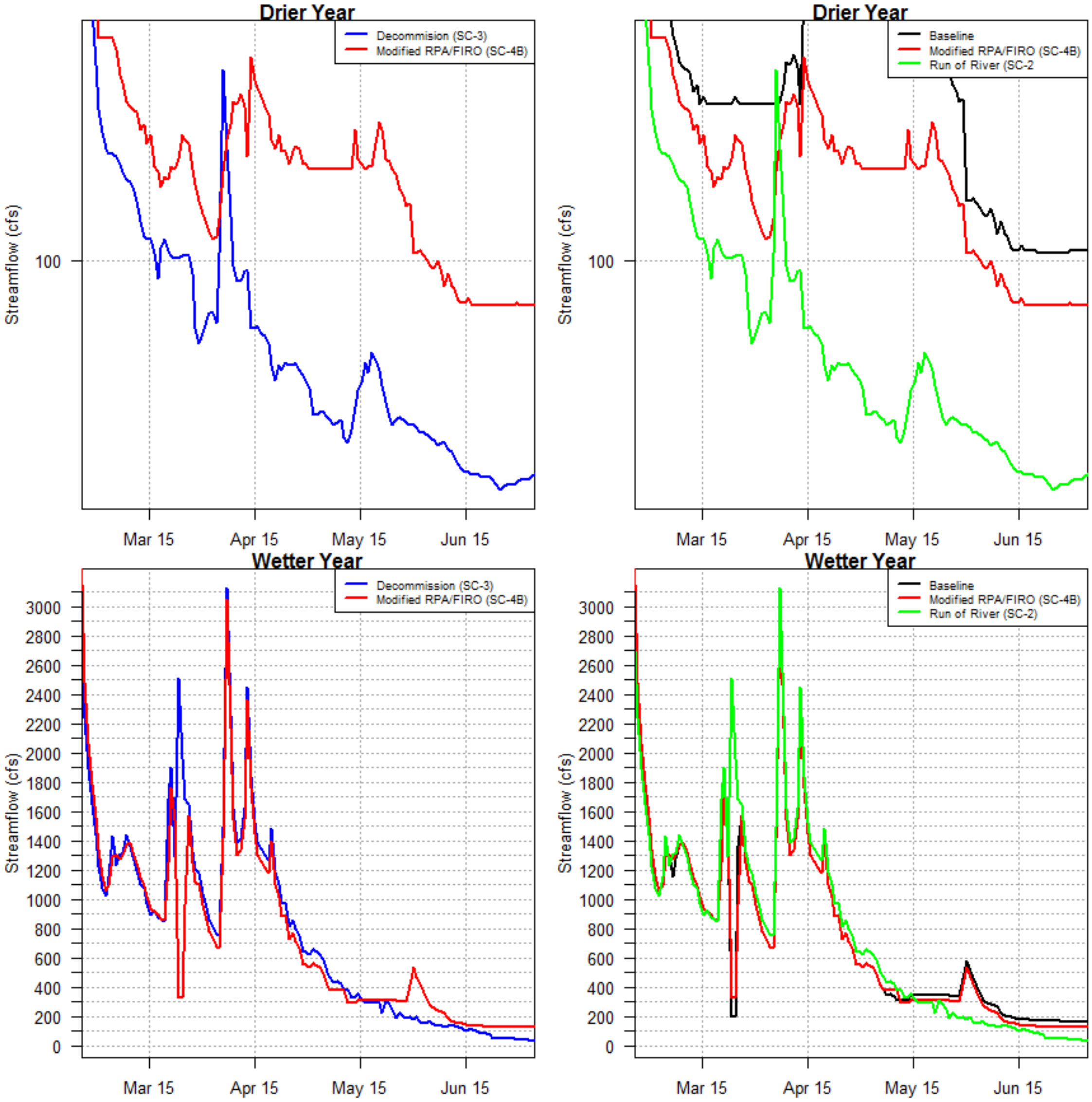

Figure 3-7. Example water years for drier (2015) and wetter (2011) years on the Eel River downstream of Scott Dam during the spring recession period. Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right. Scenario 2 is not presented on the left column of plots because it is identical to Scenario 3 at this location.

Summer baseflows (Figure 3-8)

- Summer baseflows are approximately 100 cfs higher under Scenarios 4B and Baseline than under Scenario 2 and Scenario 3.
- For Scenario 3 in the wetter year, baseflow condition is not reached until nearly halfway through summer months.

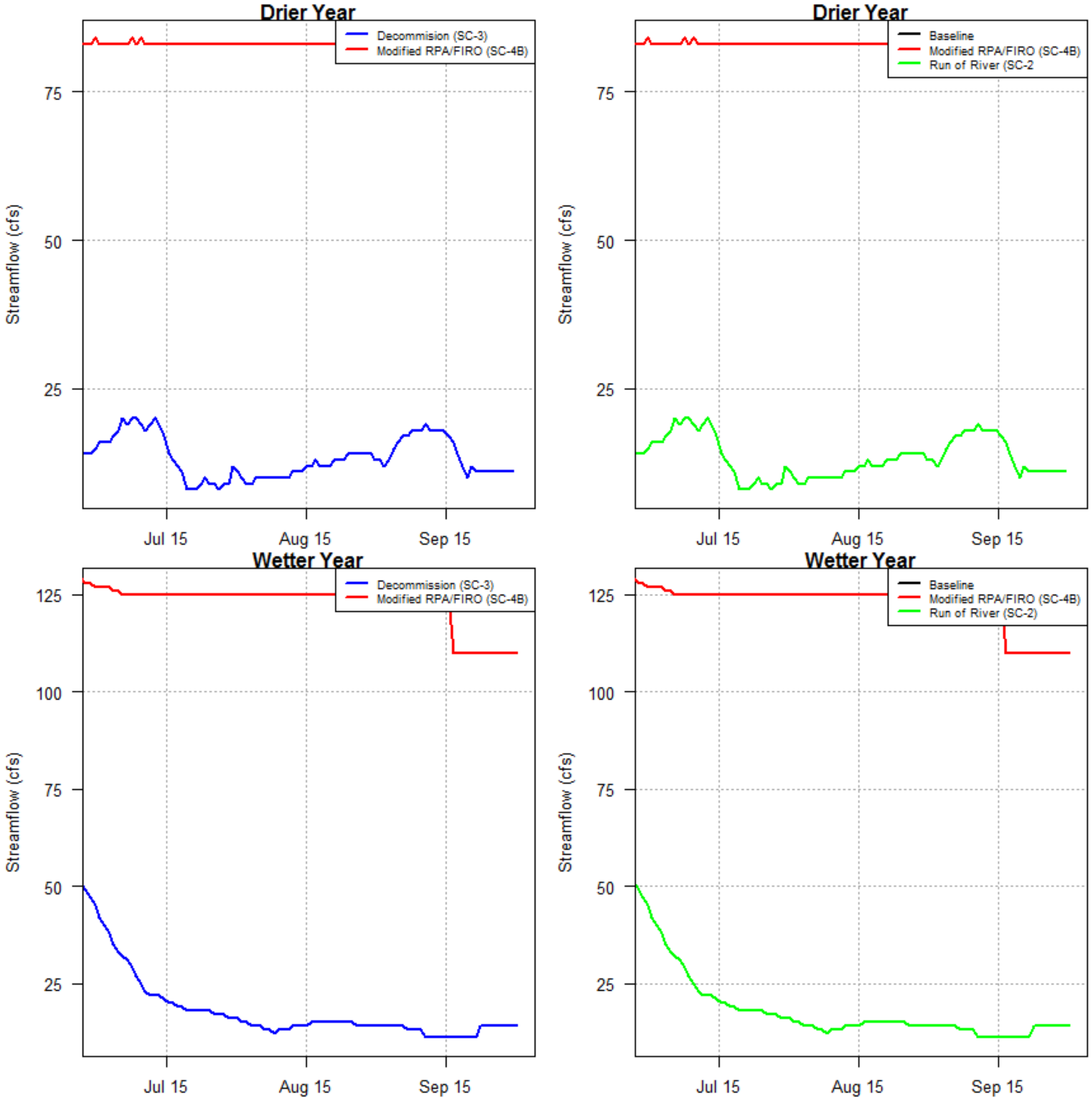

Figure 3-8. Example water years for drier (2015) and wetter (2011) years on the Eel River downstream of Scott Dam during summer baseflow period. Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right. Scenario 2 is not presented on the left column of plots because it is identical to Scenario 3 at this location.

Fall streamflow (Figure 3-9)

- Small streamflow peaks occur earlier in the fall season for Scenario 2 and Scenario 3.
- The magnitude of late fall peak flows is approximately 3,000 cfs higher for Scenario 2 and Scenario 3 in the drier year and as much as 5,000 cfs higher in the wetter year.

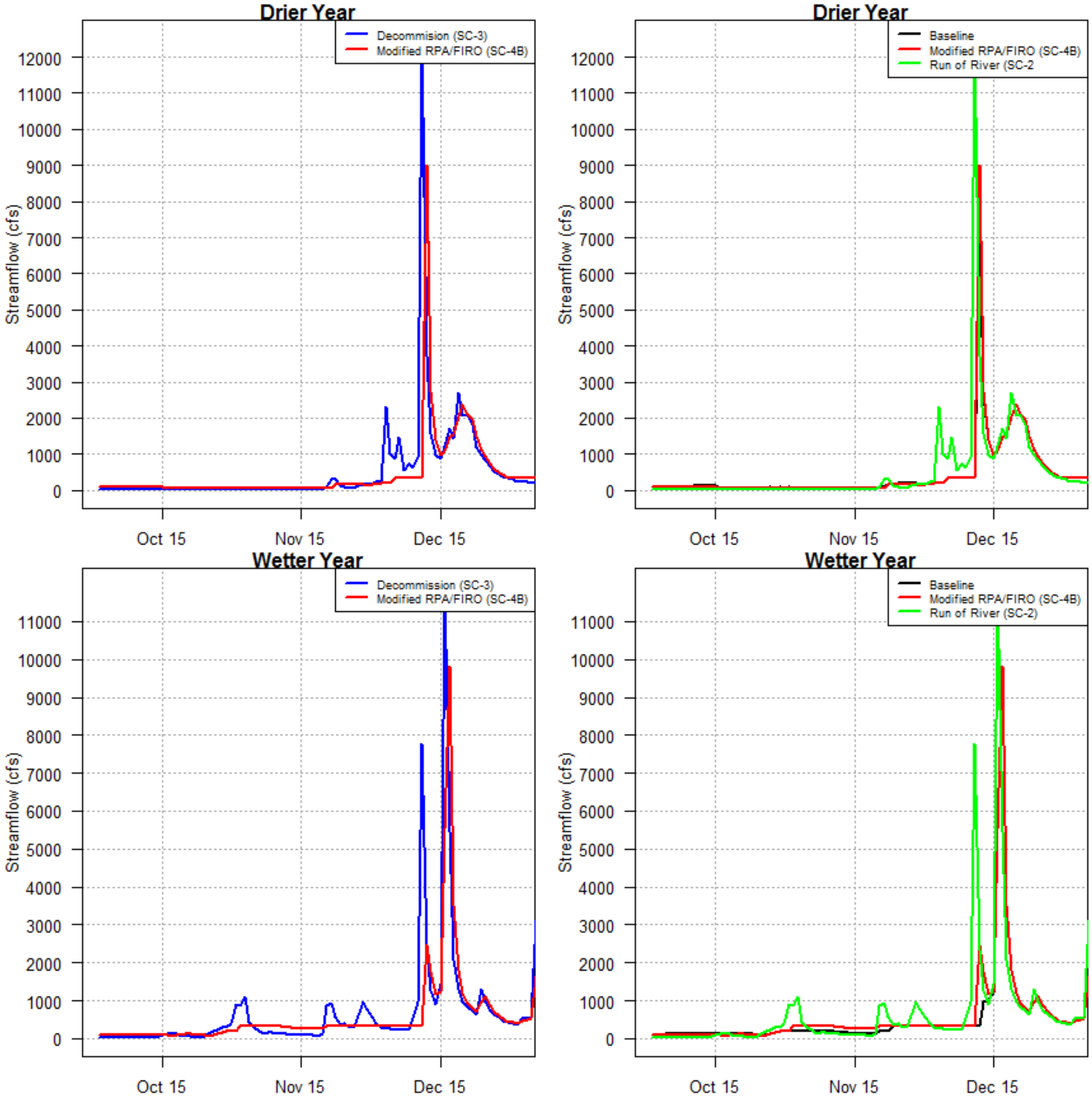

Figure 3-9. Example water years for drier (2015) and wetter (2011) years on the Eel River downstream of Scott Dam during fall streamflow period. Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right. Scenario 2 is not presented on the left column of plots because it is identical to Scenario 3 at this location.

3.1.2.3 Flood Frequency

The flood frequency plot for downstream of Scott Dam (gage E-2, Figure 3-10) illustrates that the Scenario 2 and Scenario 3 peak daily average streamflow magnitudes are higher than the Baseline or Scenario 4B streamflows for nearly all recurrence intervals, with the greatest discrepancy in the 5- to 15-year flood recurrence range. Baseline and Scenario 4B streamflow magnitudes are nearly identical across the entire flood recurrence range.

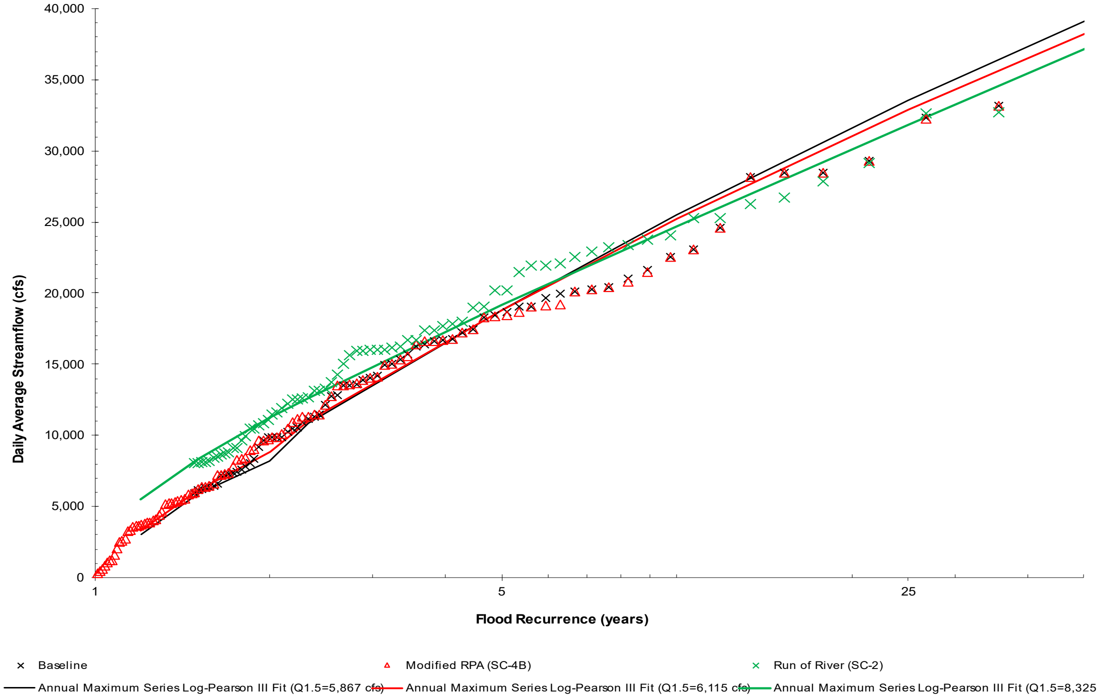

Figure 3-10. Flood frequency analysis for all Water Supply Scenarios at gage E-2 (Scenario 3 is the same as Scenario 2 at gage E-2). X-axis is log-scale.

3.1.3 Eel River Downstream of Cape Horn Dam (gage E-11)

Each Water Supply Scenario is different for downstream of Cape Horn Dam and is regulated by the diversion schedule. The streamflows are not altered as much here as they are downstream of Scott Dam, but the timing and magnitude of diversion could affect fish and ecology.

3.1.3.1 Daily Average Record

Winter and spring peak flows (Figure 3-11)

- In general, peak flows are highest under Scenario 3 for both wetter and drier water years. Scenario 3 peak flows may be as much as 400 cfs higher than Scenario 2 or Scenario 4B.
- Baseline, Scenario 2, and Scenario 4B hydrographs are similar for the majority of the winter and spring seasons, with the greatest discrepancy occurring in mid-March.
- All trends are similar in both wetter and drier years, but there are larger differences under wetter year Water Supply Scenarios.

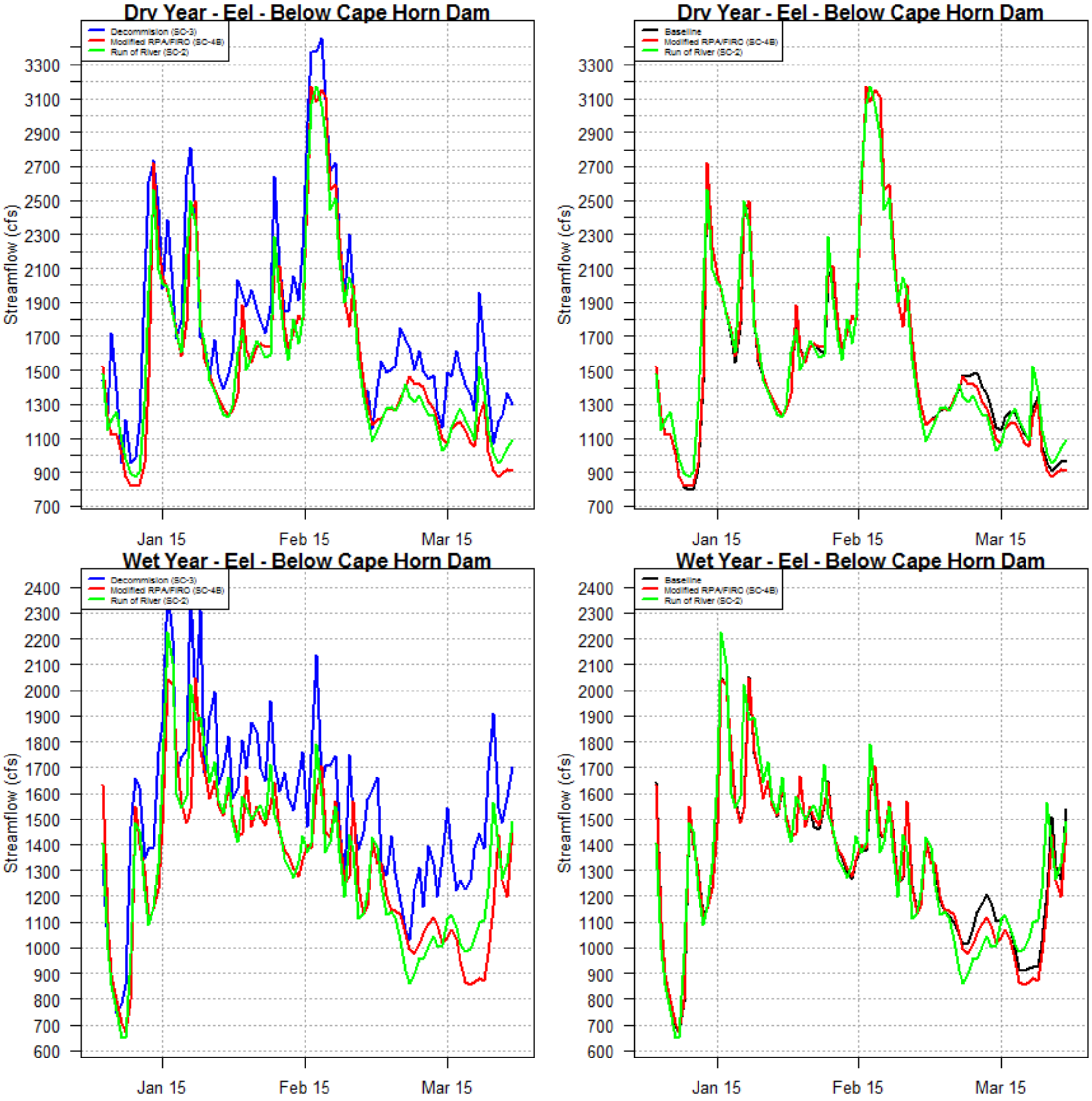

Figure 3-11. Daily average streamflow for drier and wetter years on the Eel River downstream of Cape Horn Dam from water years 1911-2017 during spring and winter storm period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Spring recession (Figure 3-12)

- Streamflow magnitude of the spring recession limb is highest under Scenario 3 for both wetter and drier water years.
- Streamflow magnitude of the spring recession limb is lowest under Scenario 2 for both wetter and drier water years.
- For wetter water year types, the spring recession lasts longer under Scenario 3 than under other Water Supply Scenarios (i.e., base streamflow is reached later).

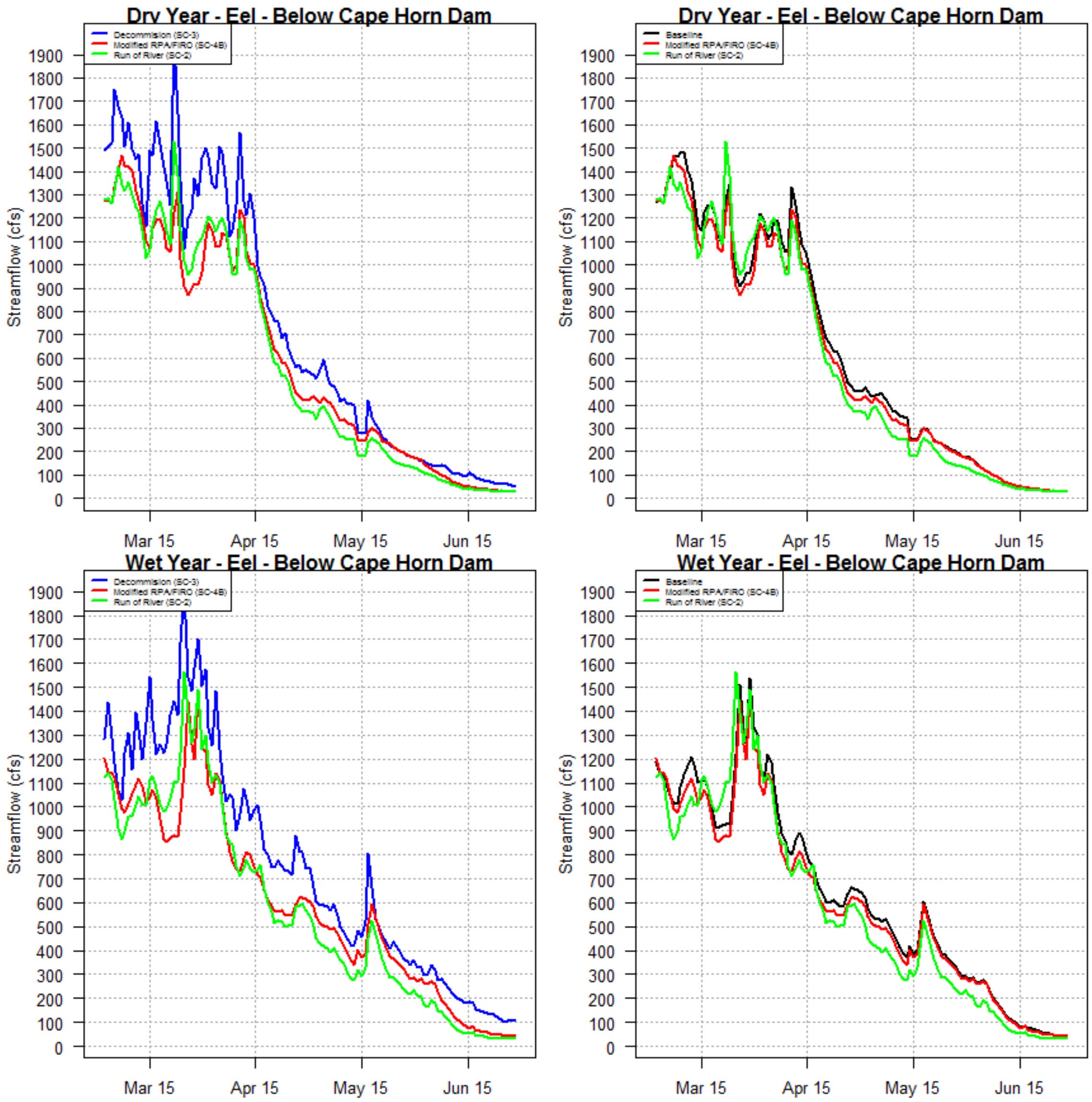

Figure 3-12. Daily average streamflow for drier and wetter years on the Eel River downstream of Cape Horn Dam from water years 1911-2017 during spring recession period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Summer baseflows (Figure 3-13)

- Magnitudes of summer baseflows for all Water Supply Scenarios are within about 5 cfs of each other for both wetter and drier water years.
- Scenario 3 and Scenario 4B do not reach baseflow condition until roughly mid-July to mid-August.

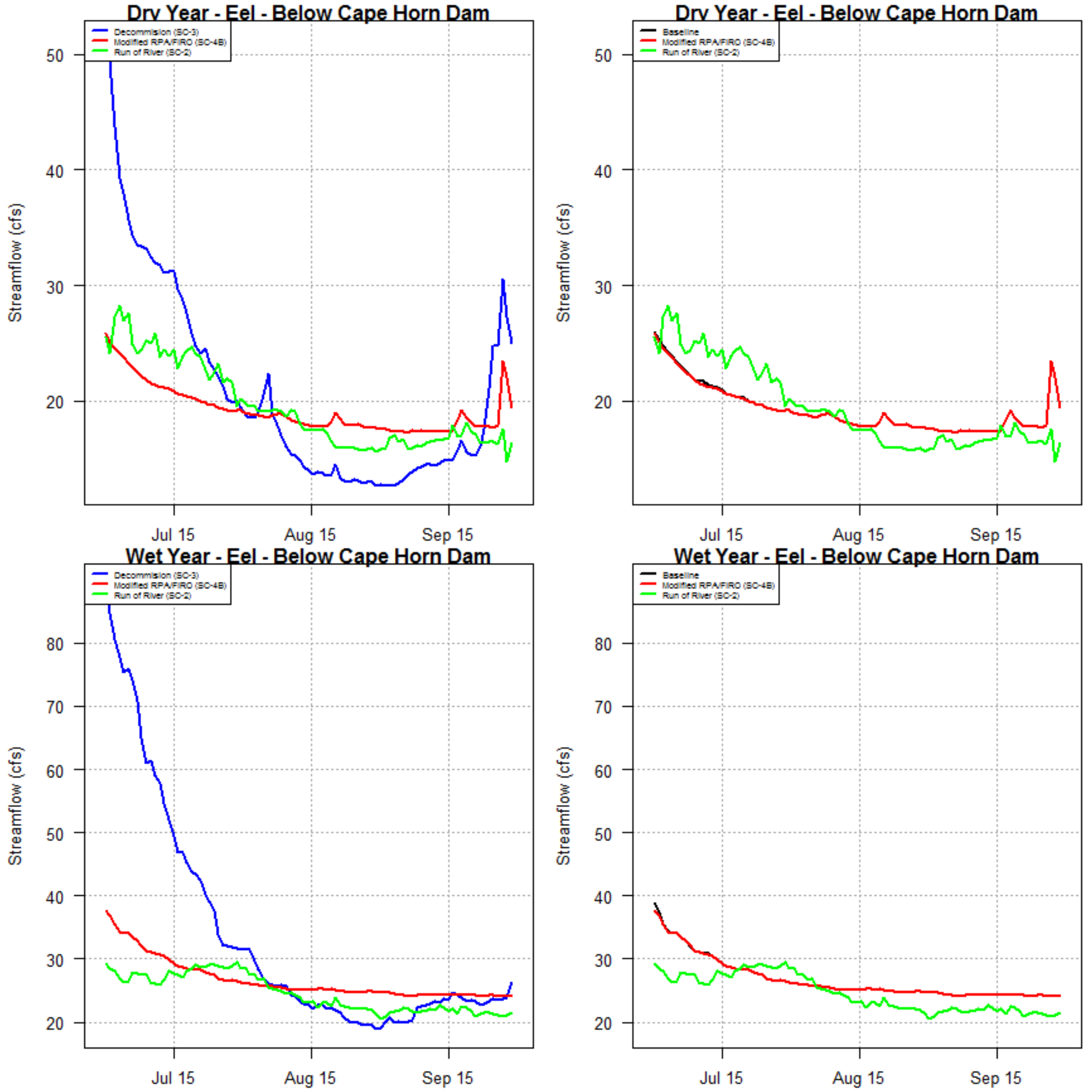

Figure 3-13. Daily average streamflow for drier and wetter years on the Eel River downstream of Cape Horn Dam from water years 1911-2017 during summer baseflow period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Fall streamflow (Figure 3-14)

- Magnitudes of fall peak flows are lower under Water Supply Scenarios which retain Scott Dam for both wetter and drier water years.
- For all water year types, magnitudes of fall peak flows are typically lower under Scenario 2 than Scenario 3, but higher than Scenario 4B and Baseline.

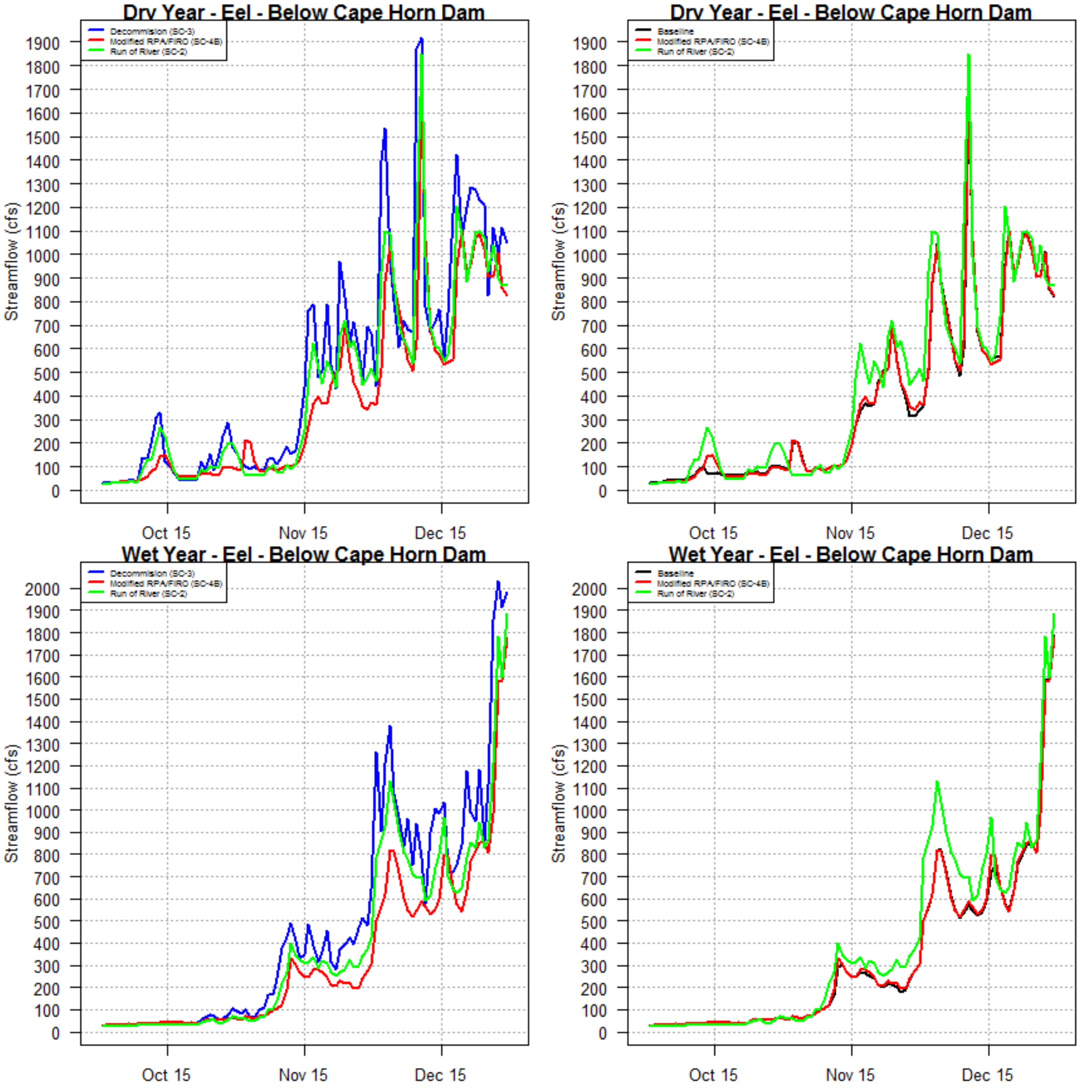

Figure 3-14. Daily average streamflow for drier and wetter years on the Eel River downstream of Cape Horn Dam from water years 1911-2017 during fall streamflow period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

3.1.3.2 Example Water Years (2011 and 2015)

Winter and spring peak flows (Figure 3-15)

- Peak flows are roughly 2,000 cfs higher under Scenario 3 for both the wetter and drier water years.
- Baseline, Scenario 2, and Scenario 4B hydrographs are very similar for the majority of the winter and spring seasons, with the greatest difference occurring in mid-March and at hydrograph peaks.
- All trends are similar in both wetter and drier years, but there are larger differences under wetter year Water Supply Scenarios.

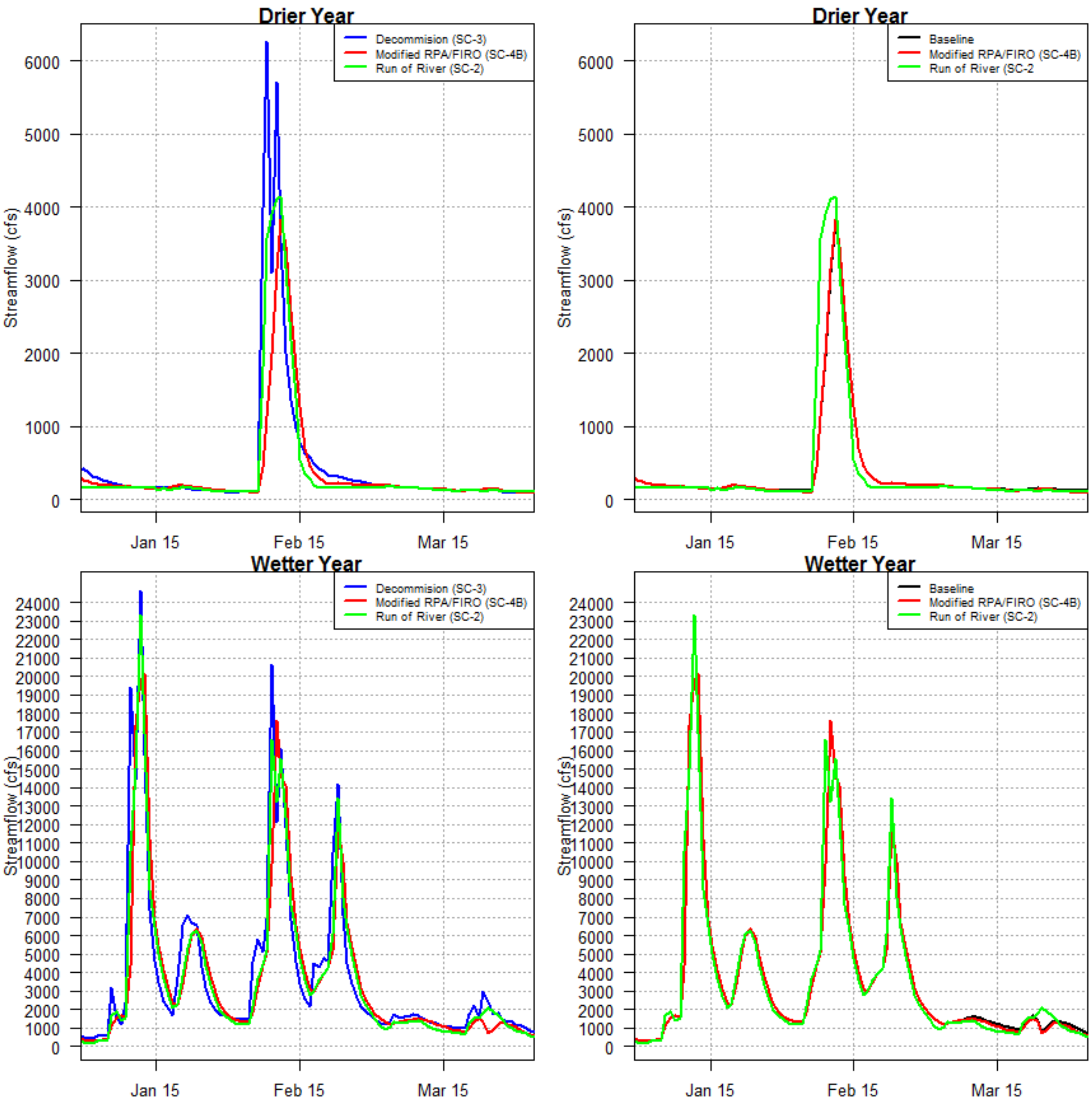

Figure 3-15. Example water years for drier (2015) and wetter (2011) years on the Eel River downstream of Cape Horn Dam during the winter and spring storm period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Spring recession (Figure 3-16)

- Peak flows in the spring recession limb are highest under Scenario 3 for both the wetter and drier water years.
- All trends are similar in both the wetter and drier years, but there are larger differences under the wetter water year Water Supply Scenarios.

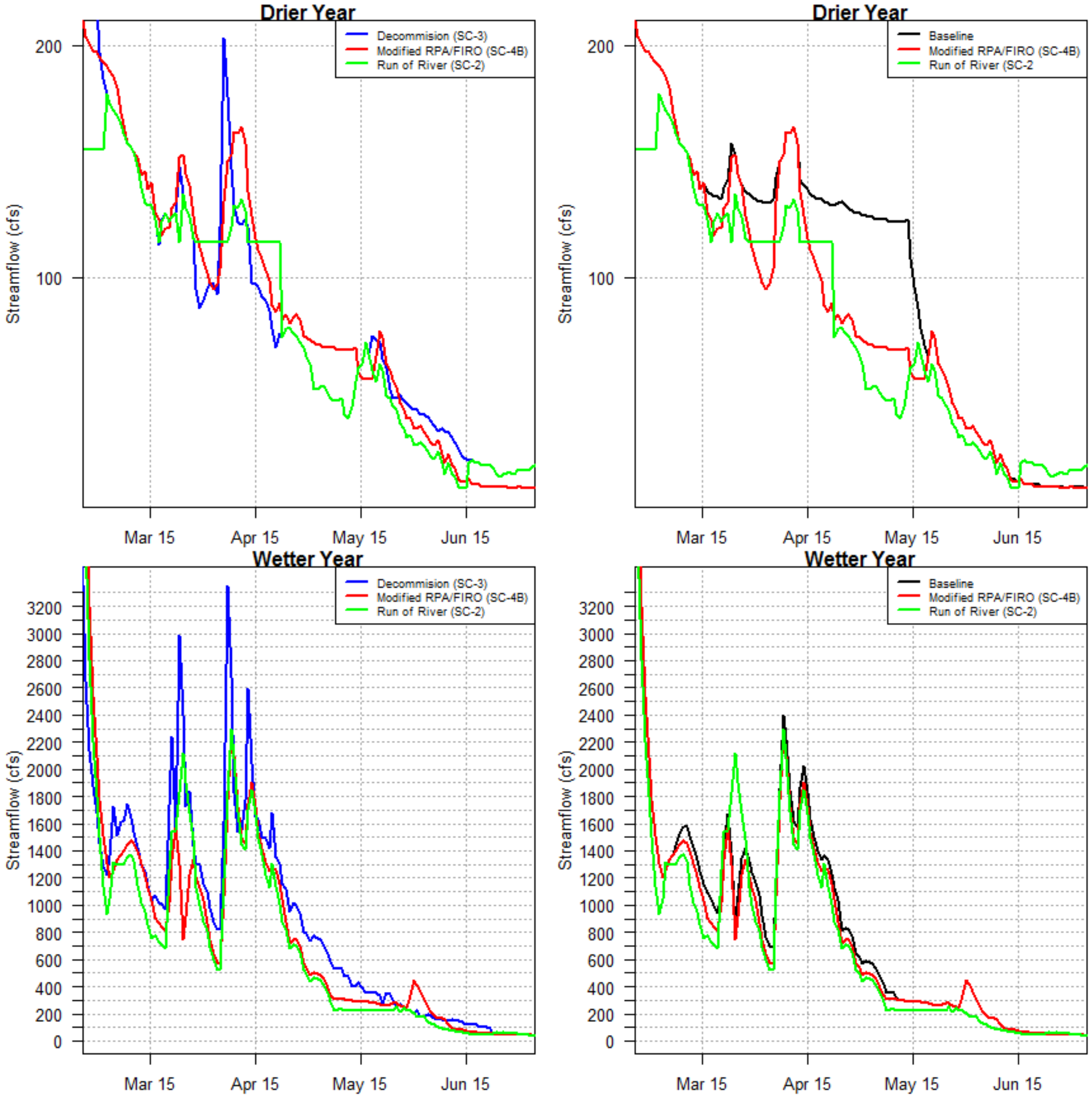

Figure 3-16. Example water years for drier (2015) and wetter (2011) years on the Eel River downstream of Cape Horn Dam during the spring recession period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Summer baseflows (Figure 3-17)

- Summer baseflows are approximately 20 cfs higher under Scenarios 4B and Baseline than under Scenario 2 and Scenario 3 for the wetter water year.
- Summer baseflows are approximately 10 cfs lower under Scenarios 4B and Baseline than under Scenario 2 and Scenario 3 for the drier water year.

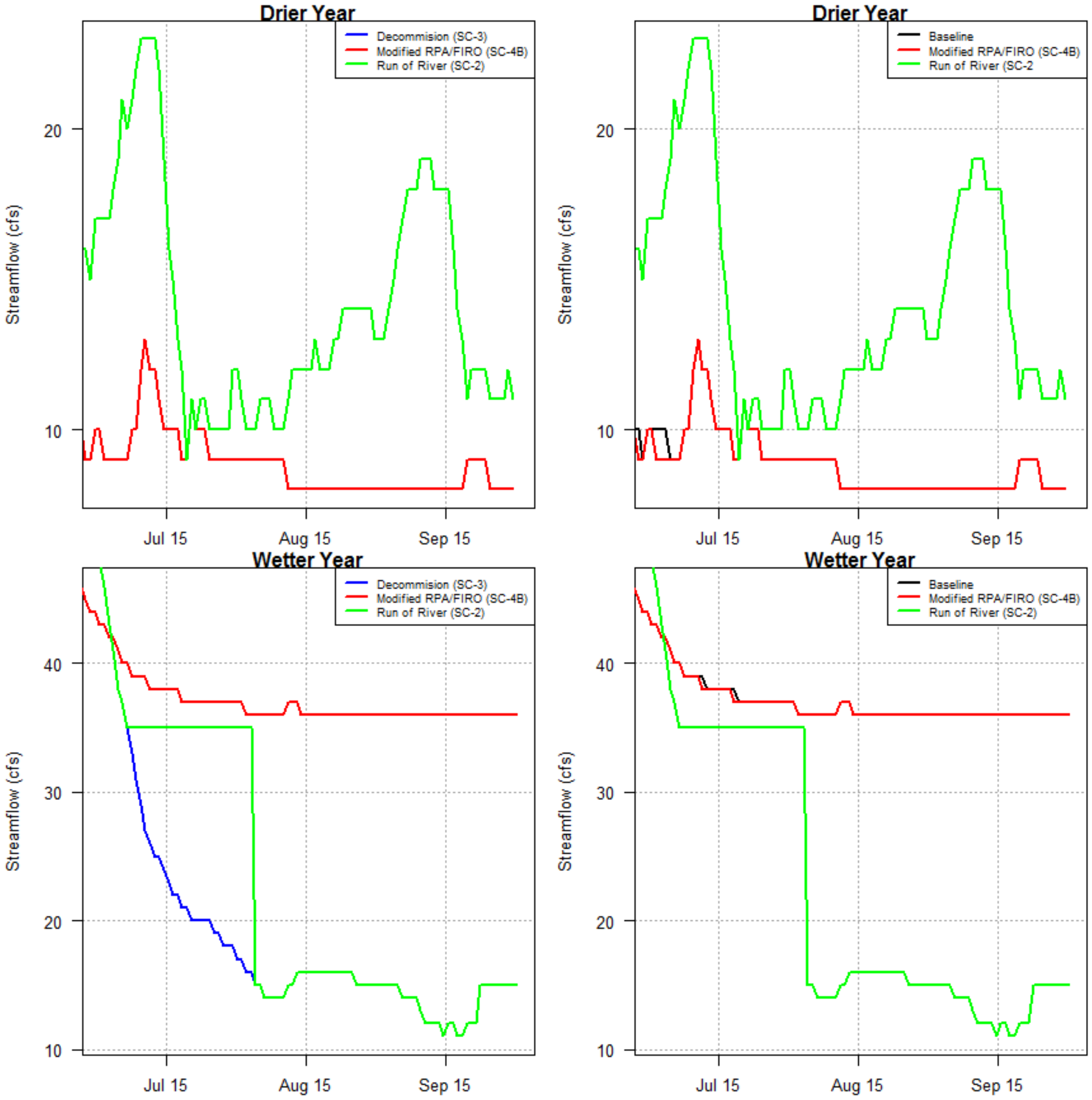

Figure 3-17. Example water years for drier (2015) and wetter (2011) years on the Eel River downstream of Cape Horn Dam summer baseflow period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Fall streamflow (Figure 3-18)

- The magnitude of late fall peak flows is approximately 6,000 cfs higher for Scenario 3 in the drier year and as much as 5,000 cfs higher in the wetter year.
- All trends are similar in both the wetter and drier years.

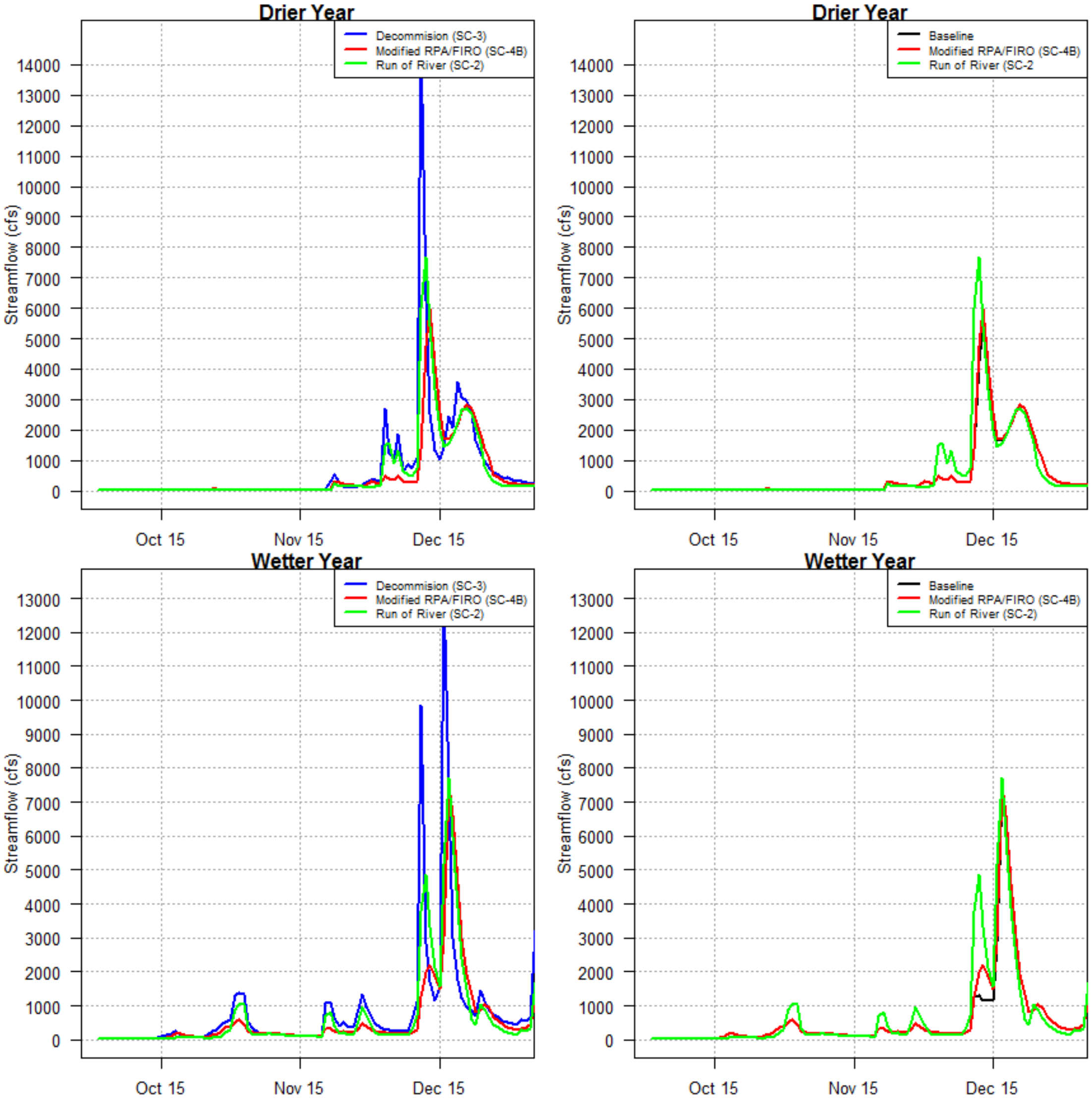

Figure 3-18. Example water years for drier (2015) and wetter (2011) years on the Eel River downstream of Cape Horn Dam during fall streamflow period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

3.1.3.3 Flood Frequency

The flood frequency plot for downstream of Cape Horn Dam (gage E-11**,** Figure 3-19) illustrates that Scenario 3 peak daily average streamflow magnitudes are higher than the Baseline, Scenario 2, and Scenario 4B streamflows for all recurrence intervals up to about the 10-year flood. Baseline, Scenario 2, and Scenario 4B streamflow magnitudes are nearly identical across the entire flood recurrence range.

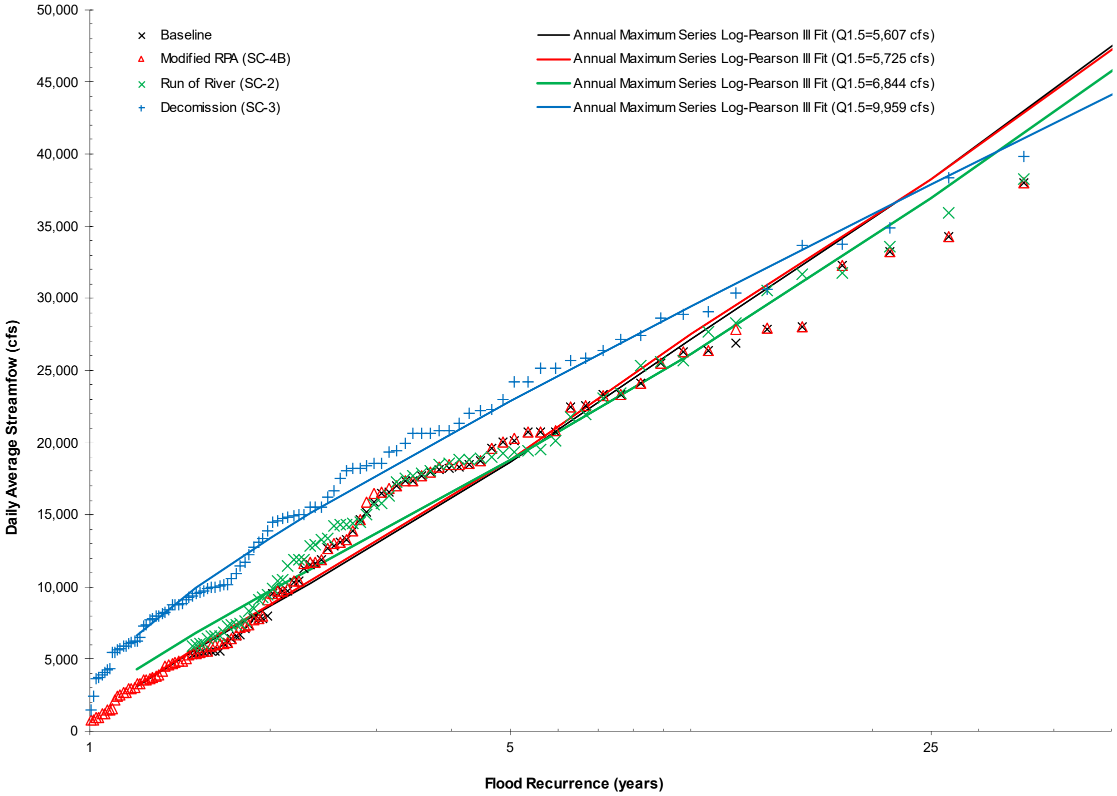

Figure 3-19. Flood frequency analysis for all Water Supply Scenarios (Baseline [black], Scenario 2 [green], Scenario 3 [blue], and Scenario 4B [red]) at gage E-11. X-axis is log-scale.

3.1.4 Russian River Downstream of the West Branch Confluence

There are no proposed infrastructure changes downstream of Coyote Valley Dam associated with the Project. In general, streamflow changes relative to Baseline can be described as decreased under Scenario 3, increased under Scenario 2, and nearly identical under Scenario 4B. There are very few changes to time and duration of streamflows, and changes are typically represented by differences in streamflow magnitudes.

3.1.4.1 Daily Average Record

Winter and spring peak flows (Figure 3-20)

- Scenario 3 consistently has lower streamflow magnitude compared to all other Water Supply Scenarios for both normal and drier water years.
- Scenario 2 has the highest peak flows for normal water years.
- Scenario 4B has the highest peak flows for drier water years.
- All trends are similar in both wetter and drier years, but there are larger differences under wetter year Water Supply Scenarios.

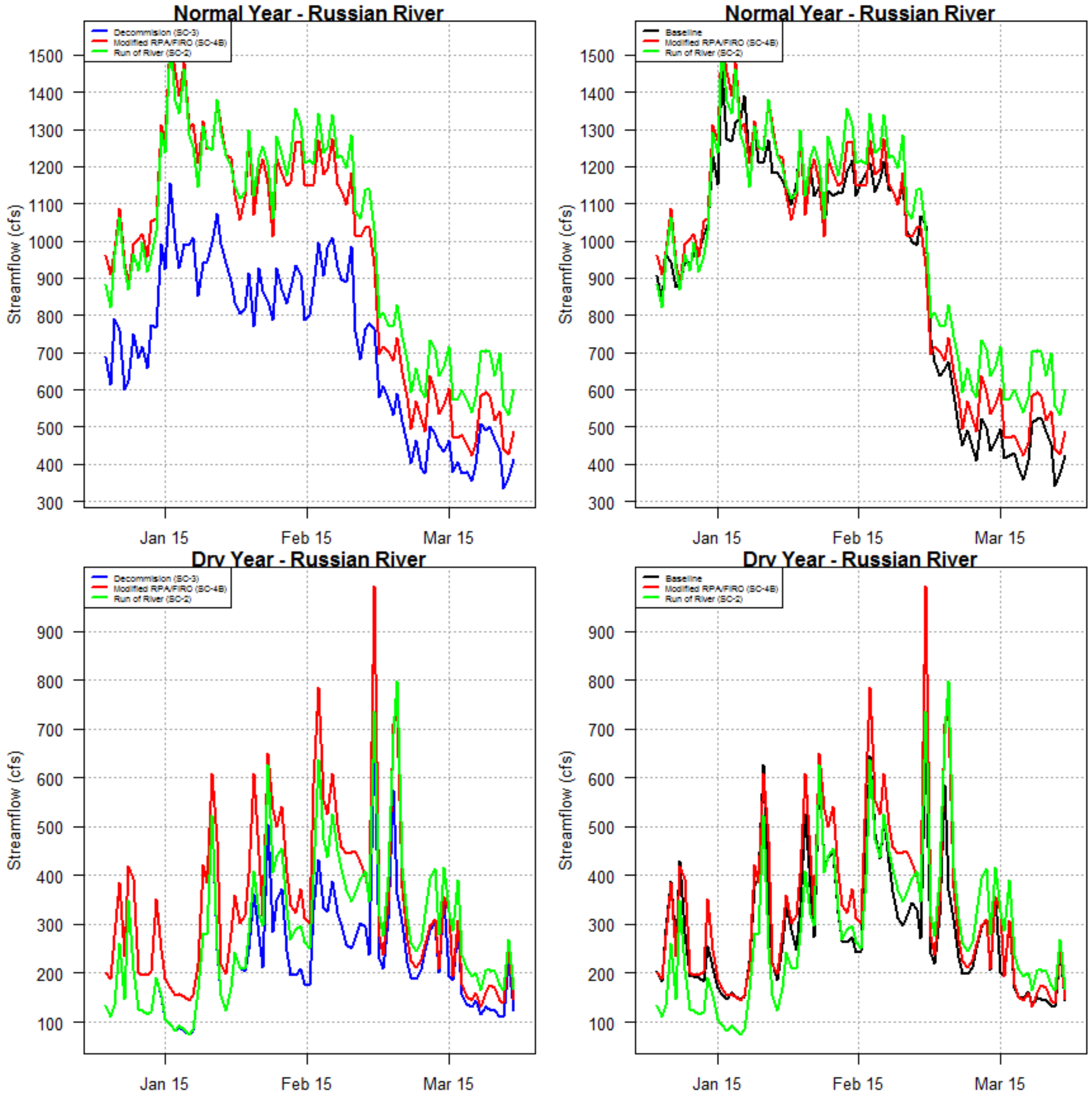

Figure 3-20. Daily average streamflow for normal and drier years on the Russian River downstream of the West Branch Russian River confluence from water years 1911-2017 during winter and spring storm period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Spring recession (Figure 3-21)

- Scenario 2 has the highest streamflow magnitude of any Water Supply Scenario throughout the entire spring recession limb for both normal and drier water years.
- Scenario 3 has the lowest streamflow magnitude of any Water Supply Scenario throughout the entire spring recession limb for both normal and drier water years.
- The hydrographs of the Baseline and Scenario 4B are very similar for both water year types, with the exception of higher Scenario 4B peak flows in March and April during normal water years.

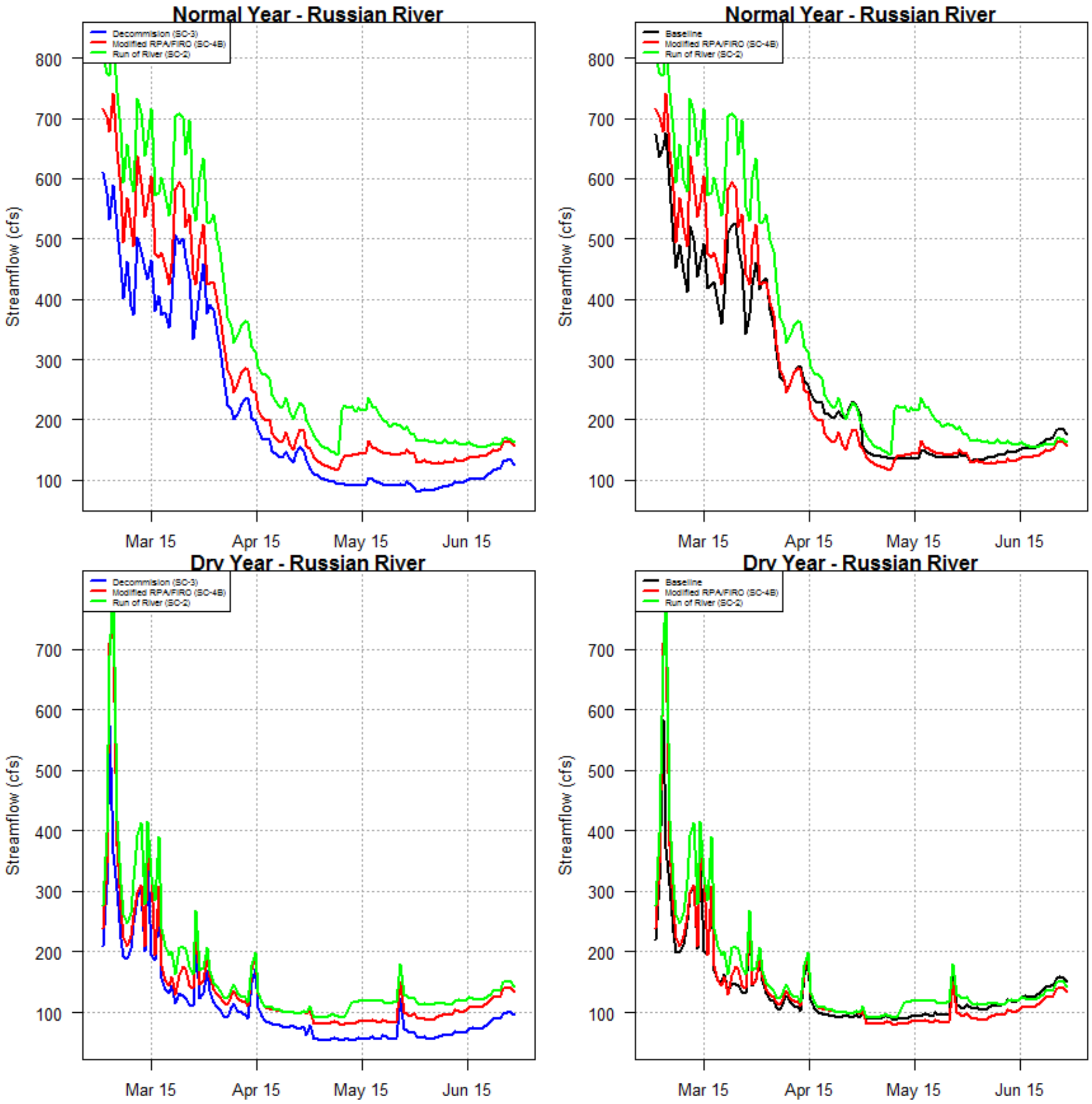

Figure 3-21. Daily average streamflow for normal and drier years on the Russian River downstream of the West Branch Russian River confluence from water years 1911-2017 during the spring recession period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Summer baseflows (Figure 3-22)

- Summer baseflows are approximately 25 cfs higher under Baseline than under Scenario 2 and Scenario 4B for both normal and drier water years.
- Summer baseflows are between 25 cfs and 50 cfs lower under Scenario 3 than under Scenario 2 and Scenario 4B for both normal and drier water years.
- General trends in the hydrographs of each Water Supply Scenario are similar, though streamflow magnitudes differ.

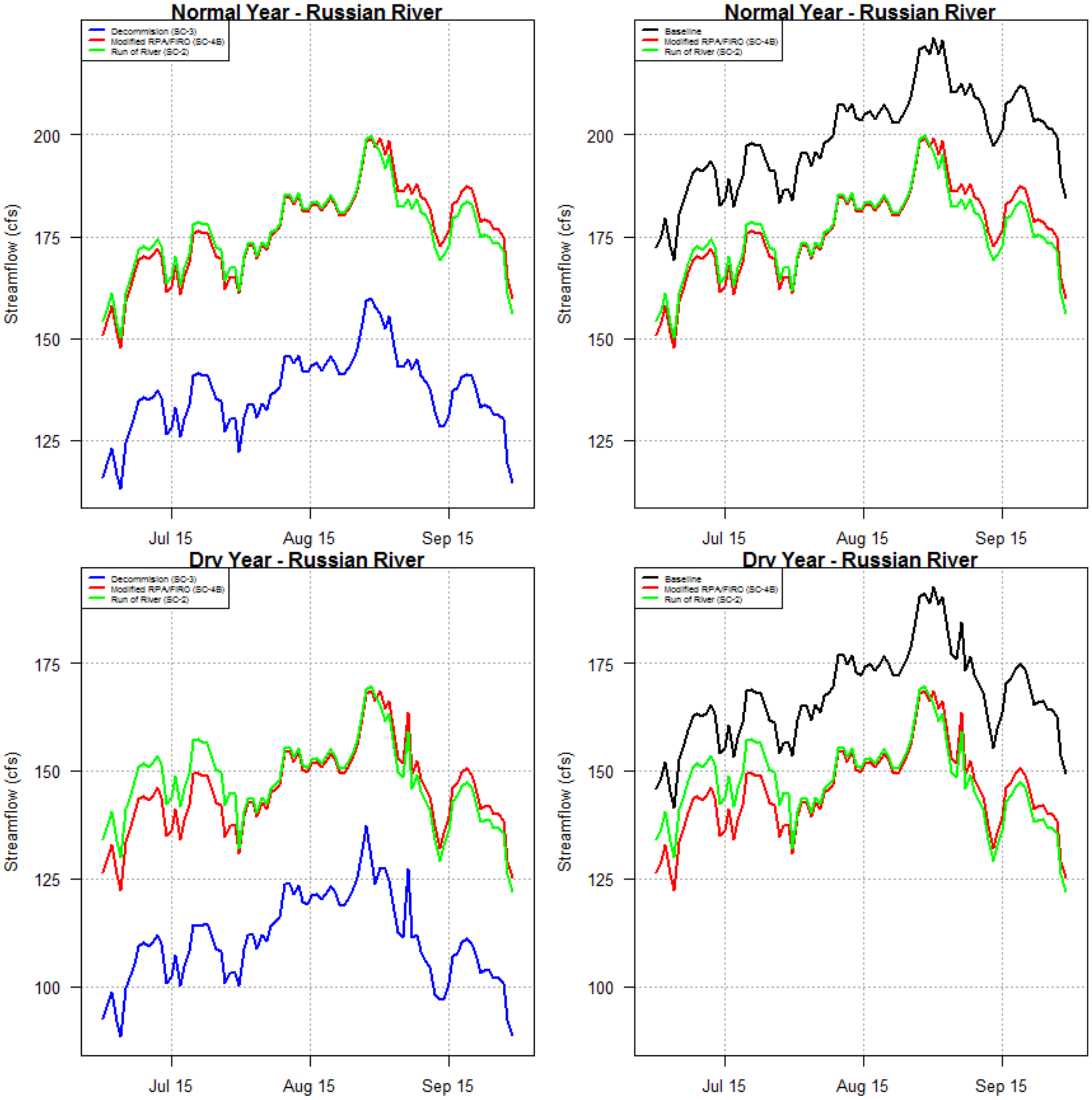

Figure 3-22. Daily average streamflow for normal and drier years on the Russian River downstream of the West Branch Russian River confluence from water years 1911-2017 during summer baseflow period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Fall streamflow (Figure 3-23)

- Streamflow magnitudes are generally higher under continued diversion Water Supply Scenarios (Baseline, Scenario 4B, Scenario 2) than Scenario 3 for both normal and drier water years.
- Peak flows in late Fall (mid-November through December) are higher under Scenario 2 and Scenario 4B relative to Scenario 3 for wetter water years, and similar in magnitude under all Water Supply Scenarios for drier water years.
- Scenario 4B streamflow is consistently about 50 cfs greater than Scenario 3 streamflow conditions for both normal and drier water years.
- The ramp-up to 300 cfs from mid-October to the start of November under Baseline and Scenario 4B during normal water years does not occur under Scenario 2 or Scenario 3.The ramp up in release is due to the flood control releases caused by the Lake Mendocino guide curves in the hydrology model. The guide curve decreases as winter approaches, and forces additional releases during mid-October to lower the reservoir to meet the winter flood control guide curve.

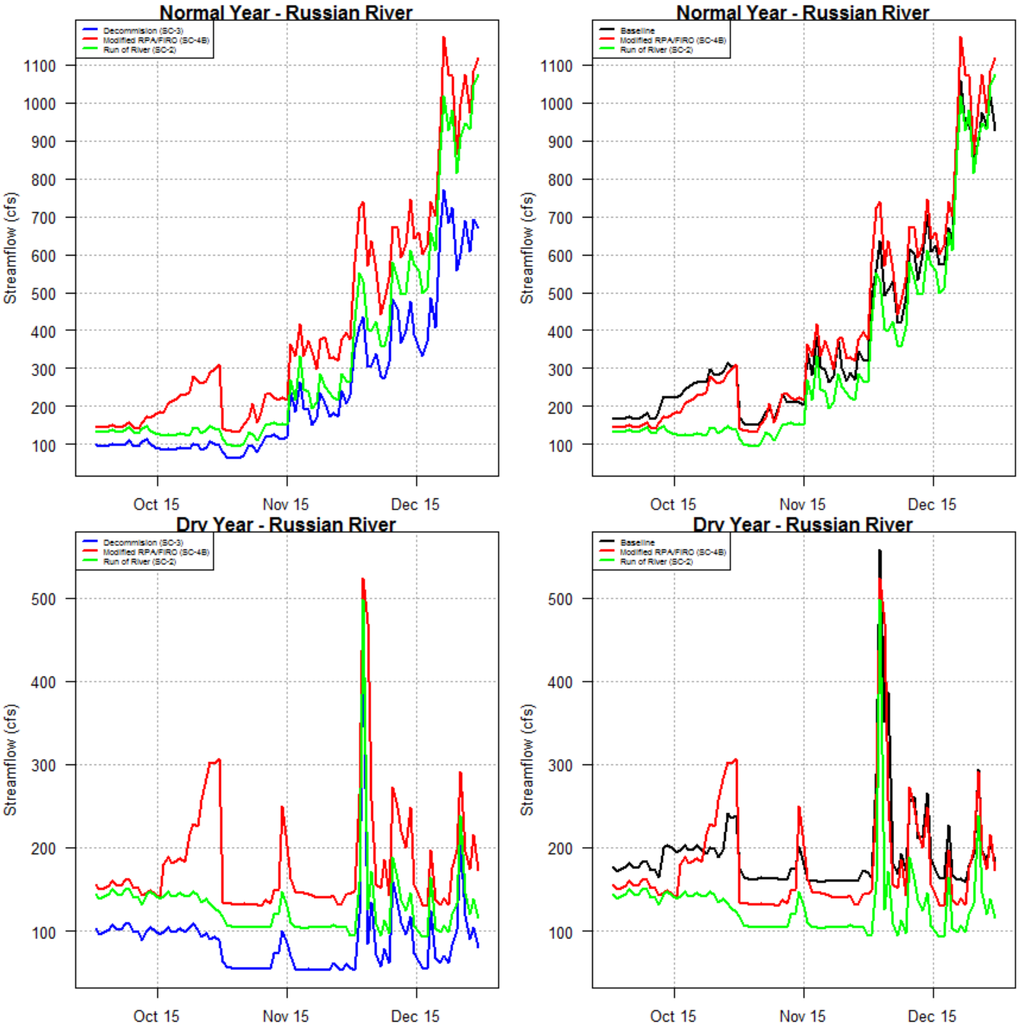

Figure 3-23. Daily average streamflow for normal and drier years on the Russian River downstream of the West Branch Russian River confluence from water years 1911-2017 during fall streamflow period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

3.1.4.2 Example Water Years (2011 and 2015)

Winter and spring peak flows (Figure 3-24)

- Scenario 3 has consistently lower streamflow magnitude compared to all other Water Supply Scenarios for both the wetter and drier years.
- Scenario 2 streamflows are lower than Scenario 4B and Baseline, but higher than Scenario 3 streamflows.
- All trends are similar in both the wetter and drier years, but there are larger differences for the wetter year.

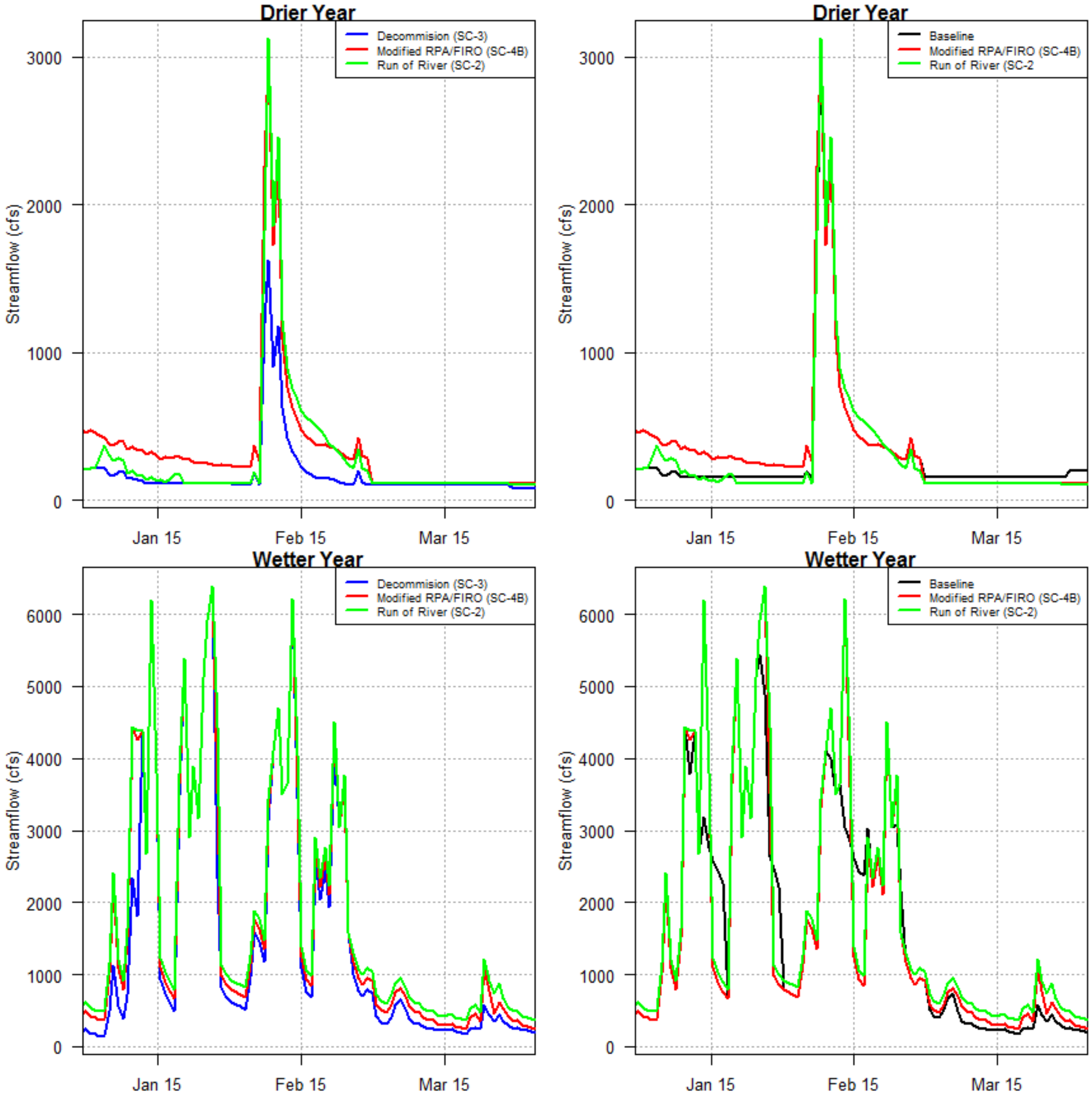

Figure 3-24. Example water years for drier (2015) and wetter (2011) years on the Russian River downstream of the West Branch Russian River confluence during the spring and winter storm period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Spring recession (Figure 3-27)

- Scenario 2 has the highest streamflow magnitude of any Water Supply Scenario throughout the entire spring recession limb for the wetter year, but the lowest for drier years.
- Scenario 3 has the lowest streamflow magnitude of any Water Supply Scenario throughout the entire spring recession limb for both the wetter and drier water years.
- During the drier year under Scenario 3, an abrupt drop in streamflow occurs at the beginning of April that does not occur until later (beginning of May) under other Water Supply Scenarios.
- During the wetter year under Scenario 2, a streamflow peak of about 375 cfs occurs at the beginning of May that does not occur under other Water Supply Scenarios. These occur because the Project is diverting 300 cfs into the EBRR, and the Lake Mendocino flood control guide curves in the hydrology model are also increasing linearly between March 1 to May 10, allowing some of that diverted water to be stored but some released as flood control releases. Starting on May 10th, the guide curve stops increasing, so all inflows need to be released for flood control.

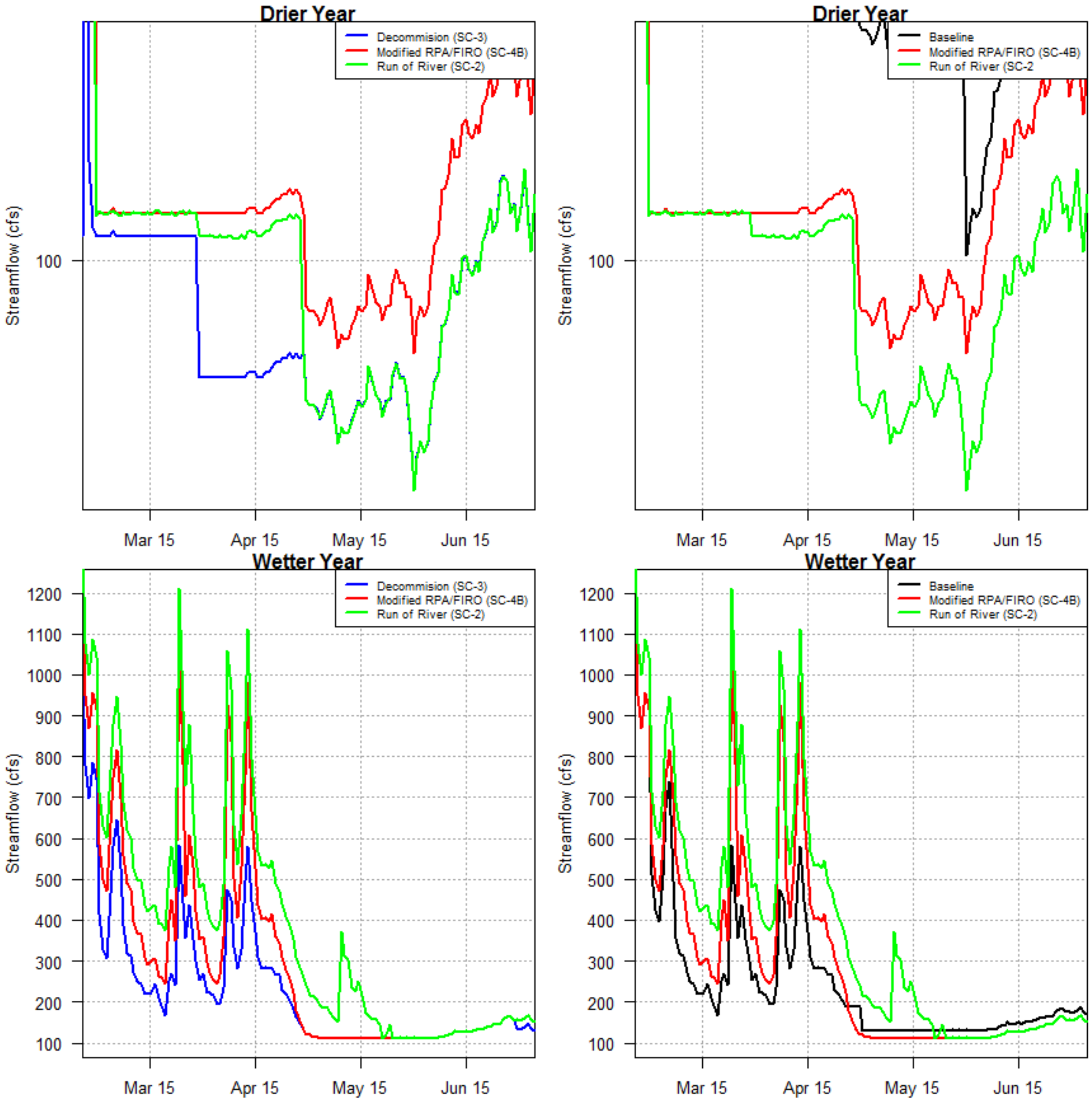

Figure 3-25. Example water years for drier (2015) and wetter (2011) years on the Russian River downstream of the West Branch Russian River confluence during the spring recession period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Summer baseflows (Figure 3-26)

- Summer baseflows are highest under Baseline for both the wetter and drier years.
- Summer baseflows are lowest under Scenario 3 for both the wetter and drier years.
- The hydrographs for Scenario 2 and Scenario 3 are identical for the drier year.
- The hydrographs for Scenario 2 and Scenario 4B are identical for the wetter year.
- General trends in the hydrographs of each Water Supply Scenario are similar, though streamflow magnitudes differ.

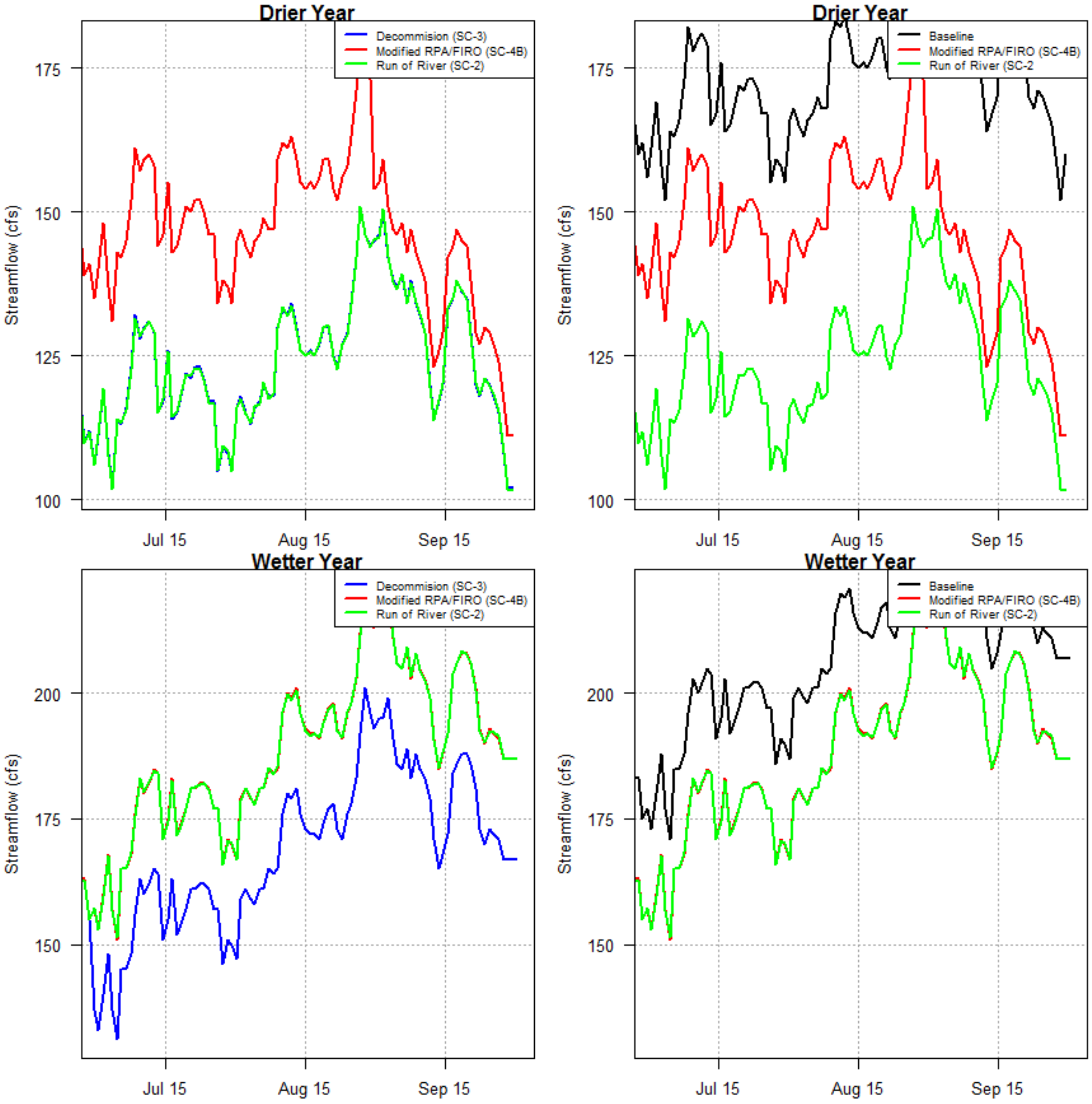

Figure 3-26. Example water years for drier (2015) and wetter (2011) years on the Russian River downstream of the West Branch Russian River confluence during summer baseflow period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Fall streamflow (Figure 3-27)

- General trends in the hydrographs of each Water Supply Scenario are similar.
- Peak flows under Scenario 4B are generally higher than other Water Supply Scenarios for both the wetter and drier years by as much as 1,500 cfs.

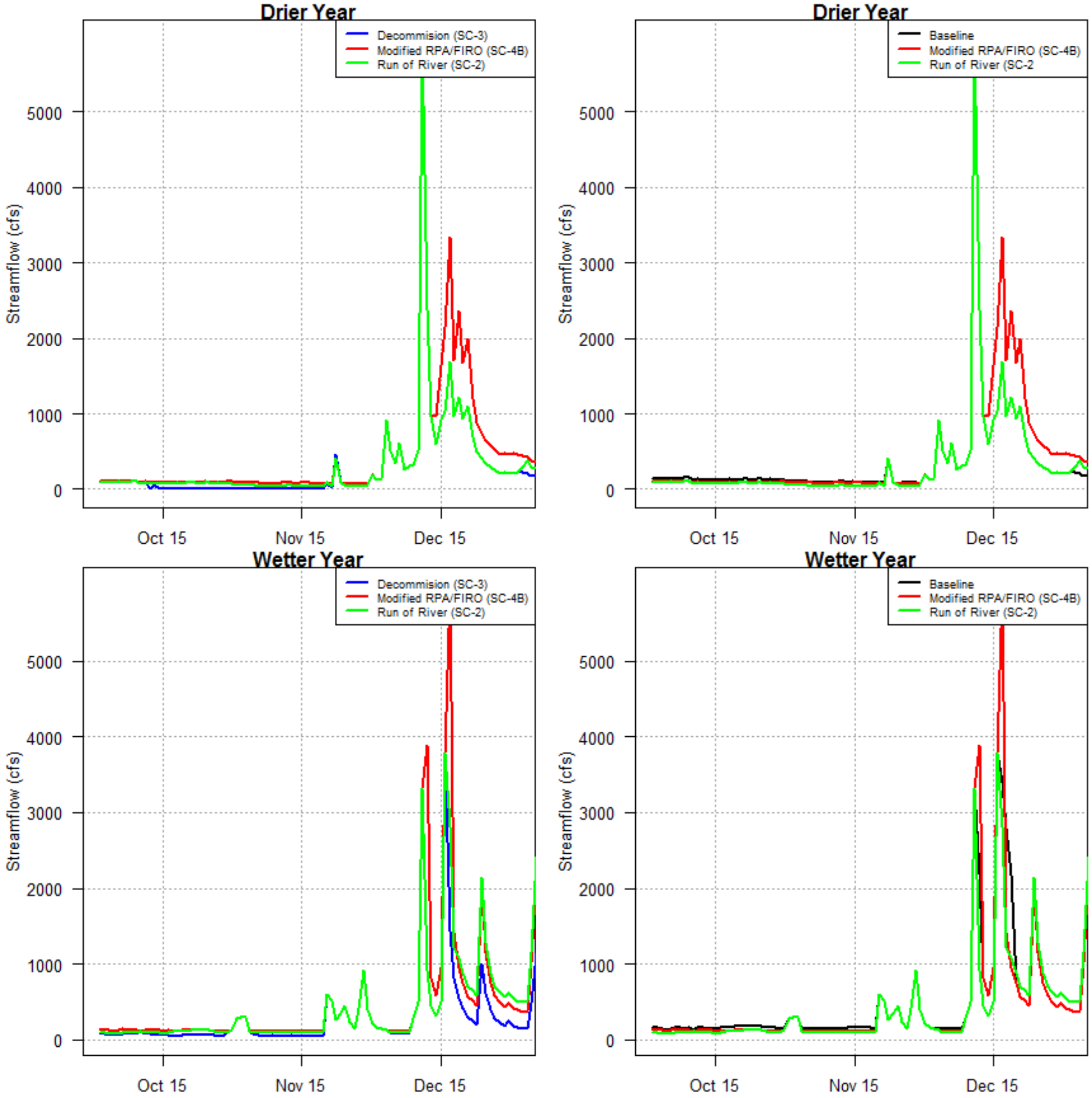

Figure 3-27. Example water years for drier (2015) and wetter (2011) years on the Russian River downstream of the West Branch Russian River confluence during fall streamflow period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

3.1.4.3 Flood Frequency

The flood frequency plot for Russian River (Figure 3-28) illustrates that Scenario 2 and Scenario 4B peak daily average streamflow magnitudes are higher than the Baseline and Scenario 3 streamflows for all recurrence intervals up to about the 5-year flood. Baseline streamflows are the lowest of all Water Supply Scenarios in the 2- to 5- year recurrence interval range. Streamflow magnitudes of all Water Supply Scenarios are nearly identical for streamflows above the 5-year recurrence interval.

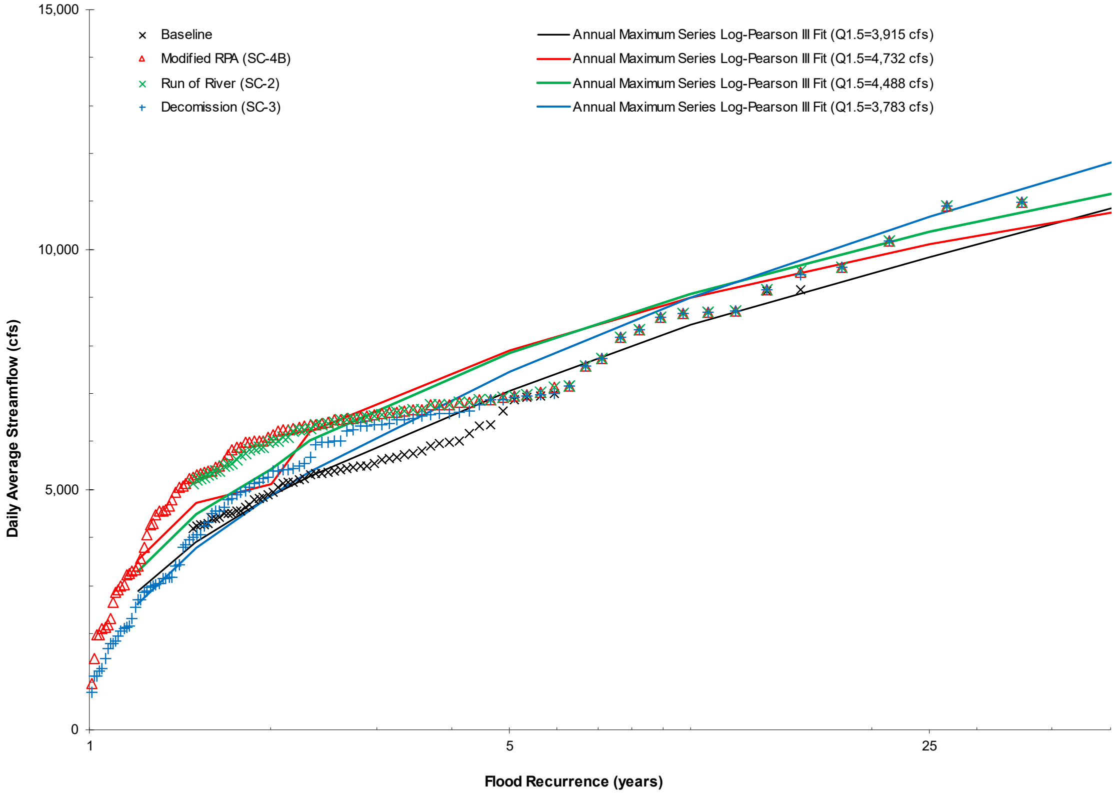

Figure 3-28. Flood frequency analysis for all Water Supply Scenarios at Russian River downstream of the West Branch Russian River confluence. X-axis is log-scale.

3.1.5 Russian River at Hopland

There are no proposed infrastructure changes downstream of Coyote Valley Dam associated with the Project. Hopland is approximately 18.5 river miles downstream of Coyote Valley Dam and can be influenced by sources other than dam releases (e.g., tributaries and accretion). In general, both Scenario 2 and Scenario 4B had higher streamflows across all seasons and water year types relative to Scenario 3 and were lower relative to the Baseline. The relative changes were smaller in magnitude than those observed downstream of the WBRR confluence (Section 3.1.4).

3.1.5.1 Daily Average Record

Compared to the Russian River downstream of the WBRR confluence, tributary influence caused Project-induced hydrology changes to be dampened out at this downstream node.

Winter and spring peak flows (Figure 3-29)

- Scenario 2 and Scenario 4B had higher streamflows relative to Scenario 3 and lower streamflows relative to Baseline.
- The largest difference was between Scenario 2 and Scenario 4B relative to Scenario 3 under normal water year types. Scenario 3 was 300 to 400 cfs lower during January and February.
- Differences between Scenario 2 and Scenario 4B relative to Baseline were minimal.

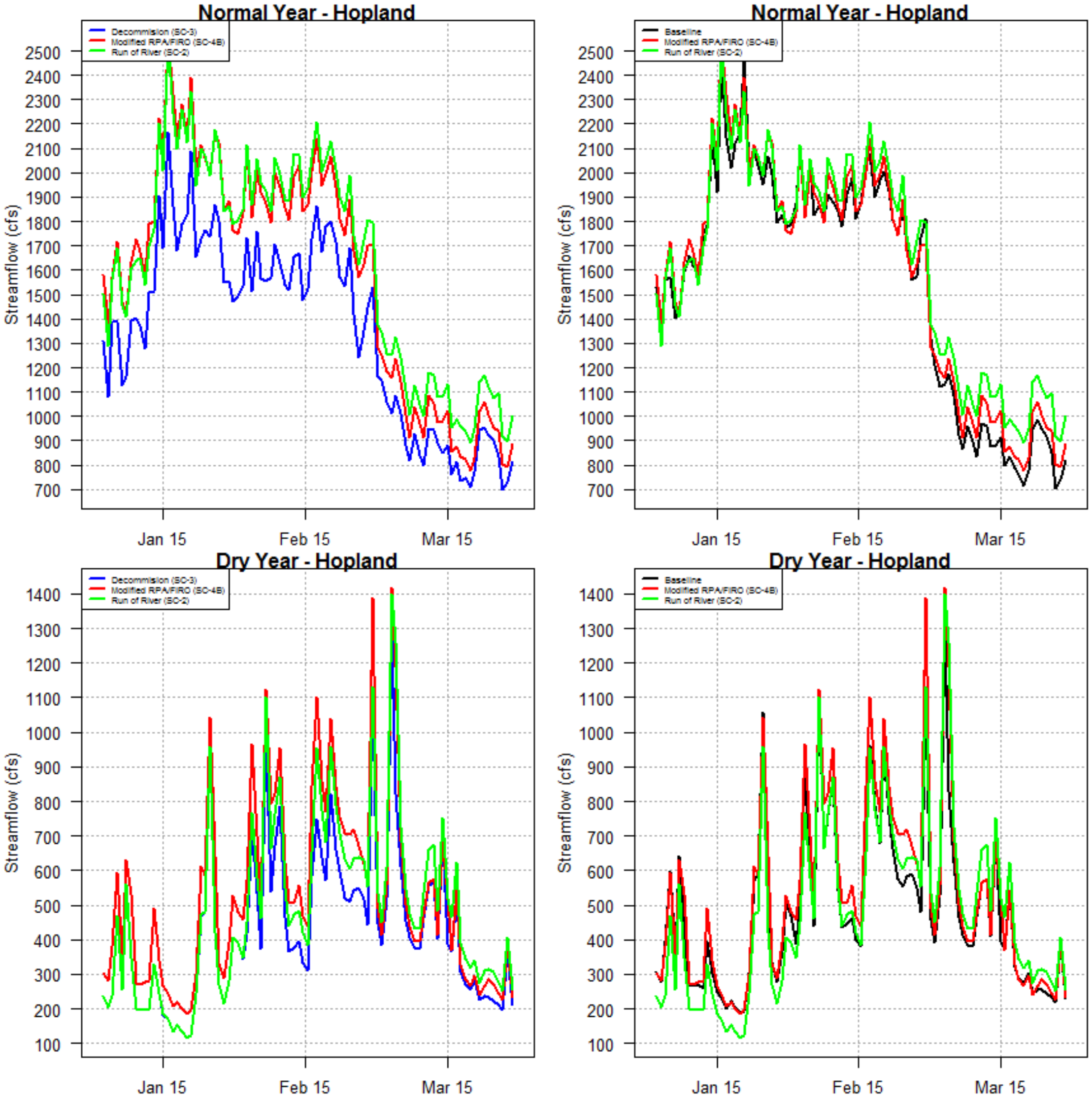

Figure 3-29. Daily average streamflow for normal and drier years on the Russian River at Hopland from water years 1911–2017 during winter and spring storm period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Spring recession (Figure 3-30):

- Modeled streamflow in Scenario 2 and Scenario 4B were generally higher than both Scenario 3 and Baseline.
- In drier water years, the magnitude of difference between Scenario 2 and Scenario 4B relative to Scenario 3 and Baseline were smaller than in normal water years.

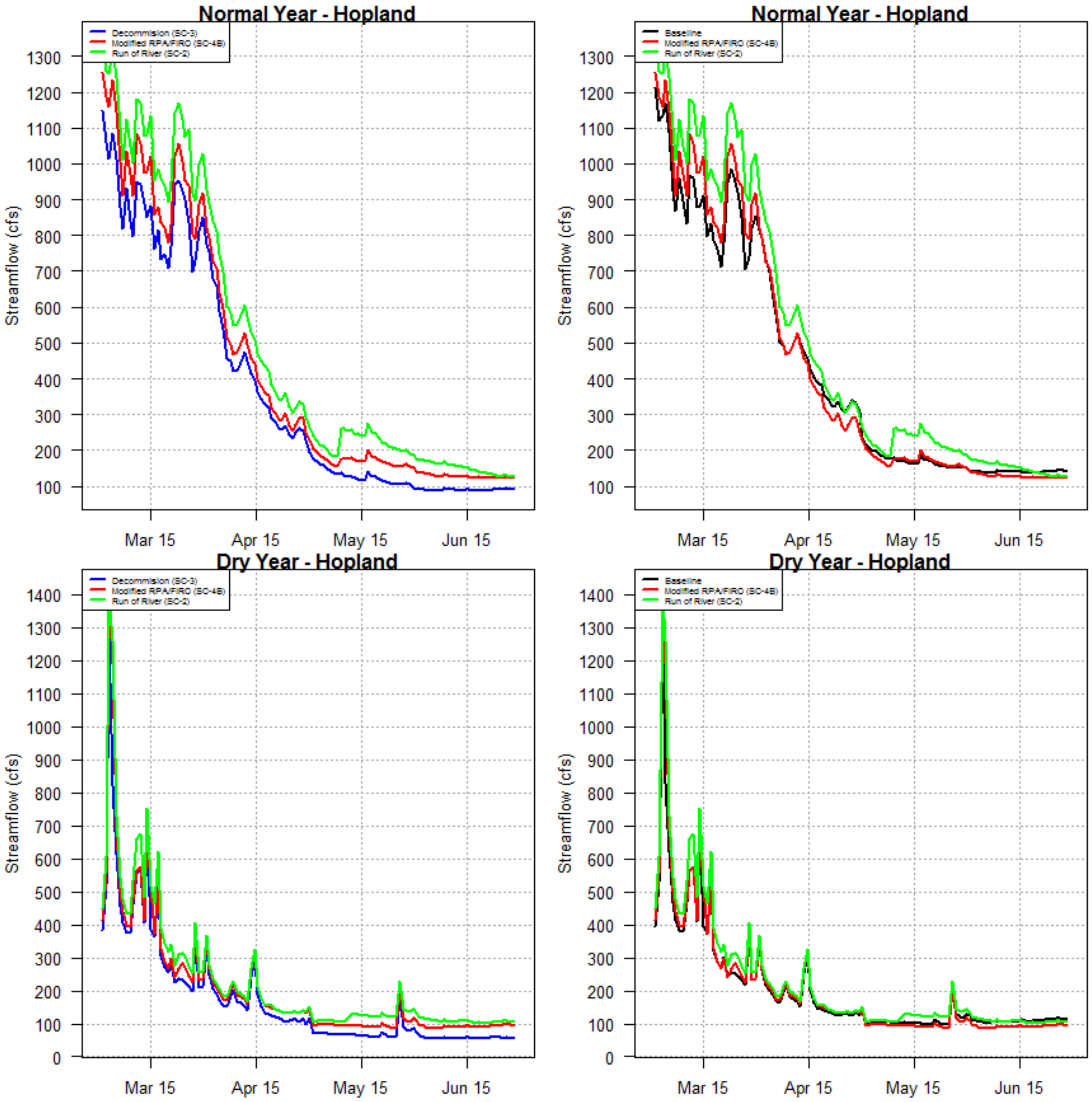

Figure 3-30. Daily average streamflow for normal and drier years on the Russian River at Hopland from water years 1911–2017 during the spring recession period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Summer baseflows (Figure 3-31):

- Relative to Scenario 3, Scenario 2 and Scenario 4B were approximately 25 cfs higher in both normal and drier years.
- Relative to Baseline, Scenario 3 and Scenario 4B were approximately 25 cfs lower in both normal and drier years.

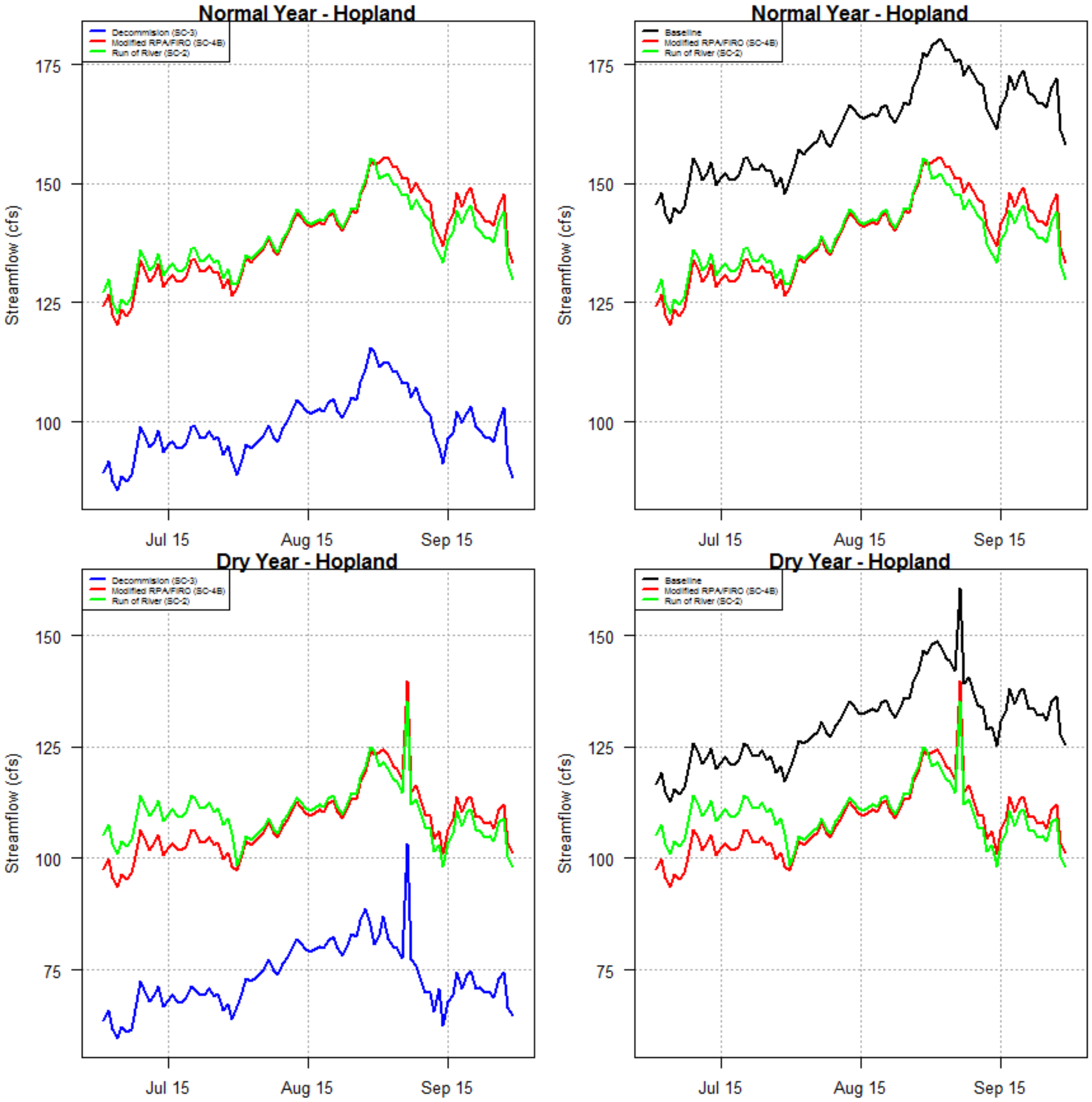

Figure 3-31. Daily average streamflow for normal and drier years on the Russian River at Hopland from water years 1911–2017 during summer baseflow period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Fall streamflow (Figure 3-32)

- During fall streamflows, Scenario 4B and Scenario 2 had higher magnitude streamflow events (~300 cfs) relative to Scenario 3 for both normal and drier water year types.
- Some early (mid to late October) fall freshets were only observed in Scenario 4B.
- In normal and drier water years, Scenario 2 was typically lower than Baseline, while Scenario 4B was higher, but differences were low in magnitude (~100 cfs).

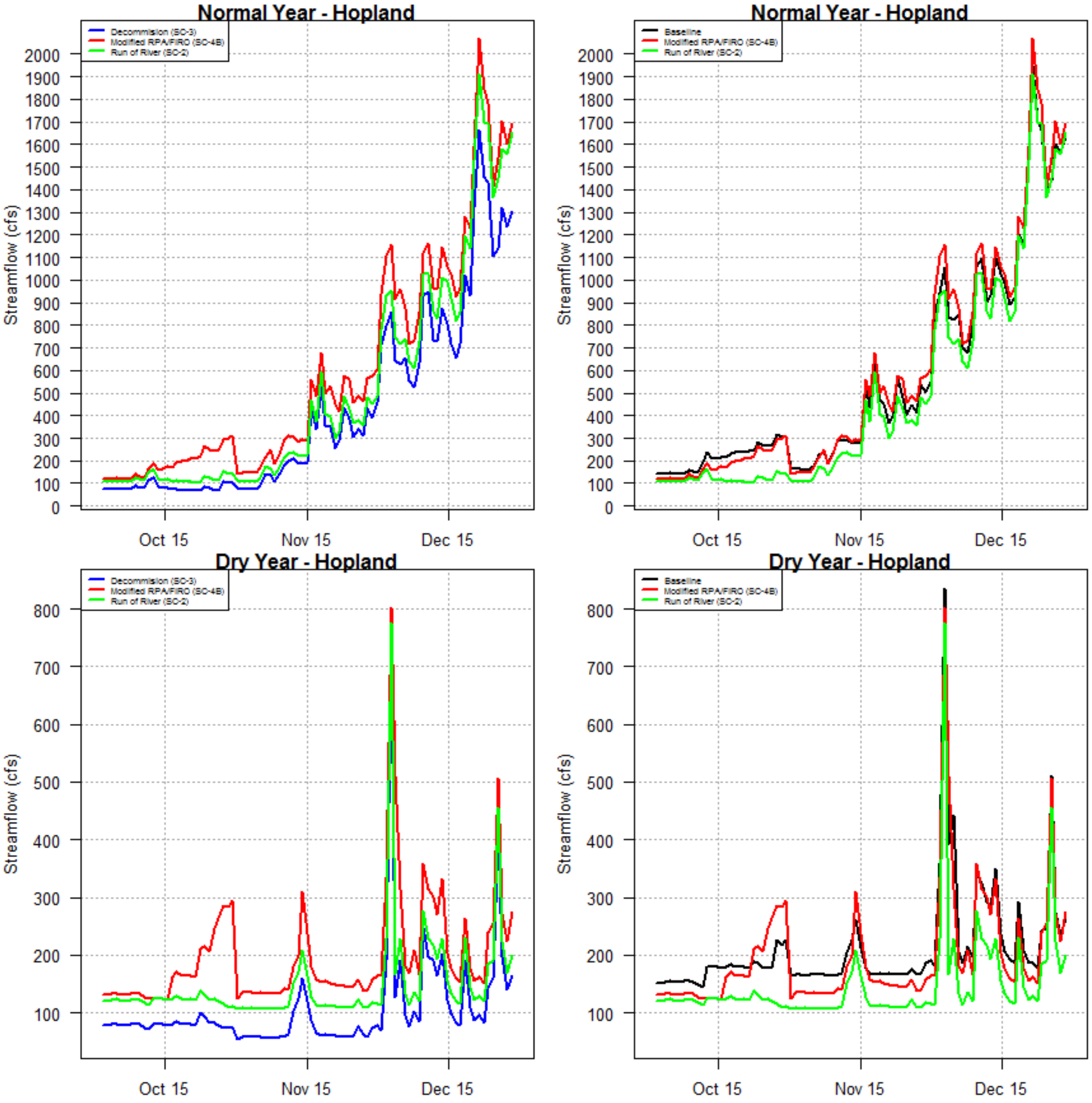

Figure 3-32. Daily average streamflow for normal and drier years on the Russian River at Hopland from water years 1911-2017 during fall streamflow period.

3.1.5.2 Example Water Year (2011 and 2015)

Winter and spring peak flows (Figure 3-33)

- Differences among Water Supply Scenarios were very small for the example water years.
- Magnitudes of spring and winter peak flows were not meaningfully different among Water Supply Scenarios for both example years.
- Drier years had streamflow events approximately half that of the wetter years, approximately 5,500 cfs relative to approximately 11,000 cfs.

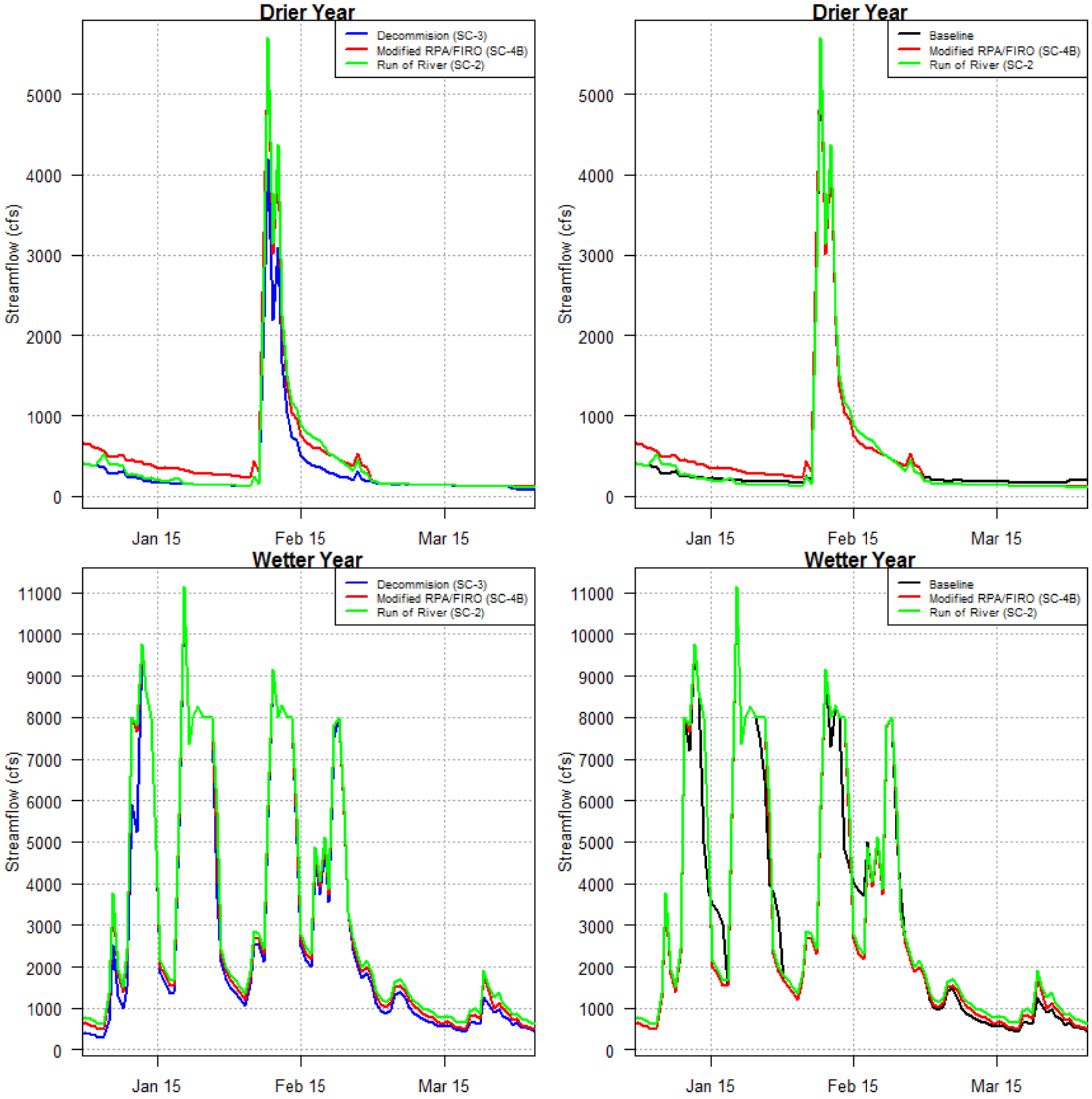

Figure 3-33. Example water years for drier (2015) and wetter (2011) years on the Russian River at Hopland during the winter and spring storm period.

Spring recession (Figure 3-34)

- Spring recession streamflows for Scenario 2 and Scenario 4B were constantly higher than Scenario 3 in both example water year types.
- During drier years, Baseline had higher streamflows than both Scenario 2 and Scenario 4B by approximately 25 cfs.
- During wetter years, Scenario 2 and Scenario 4B were higher than Baseline.

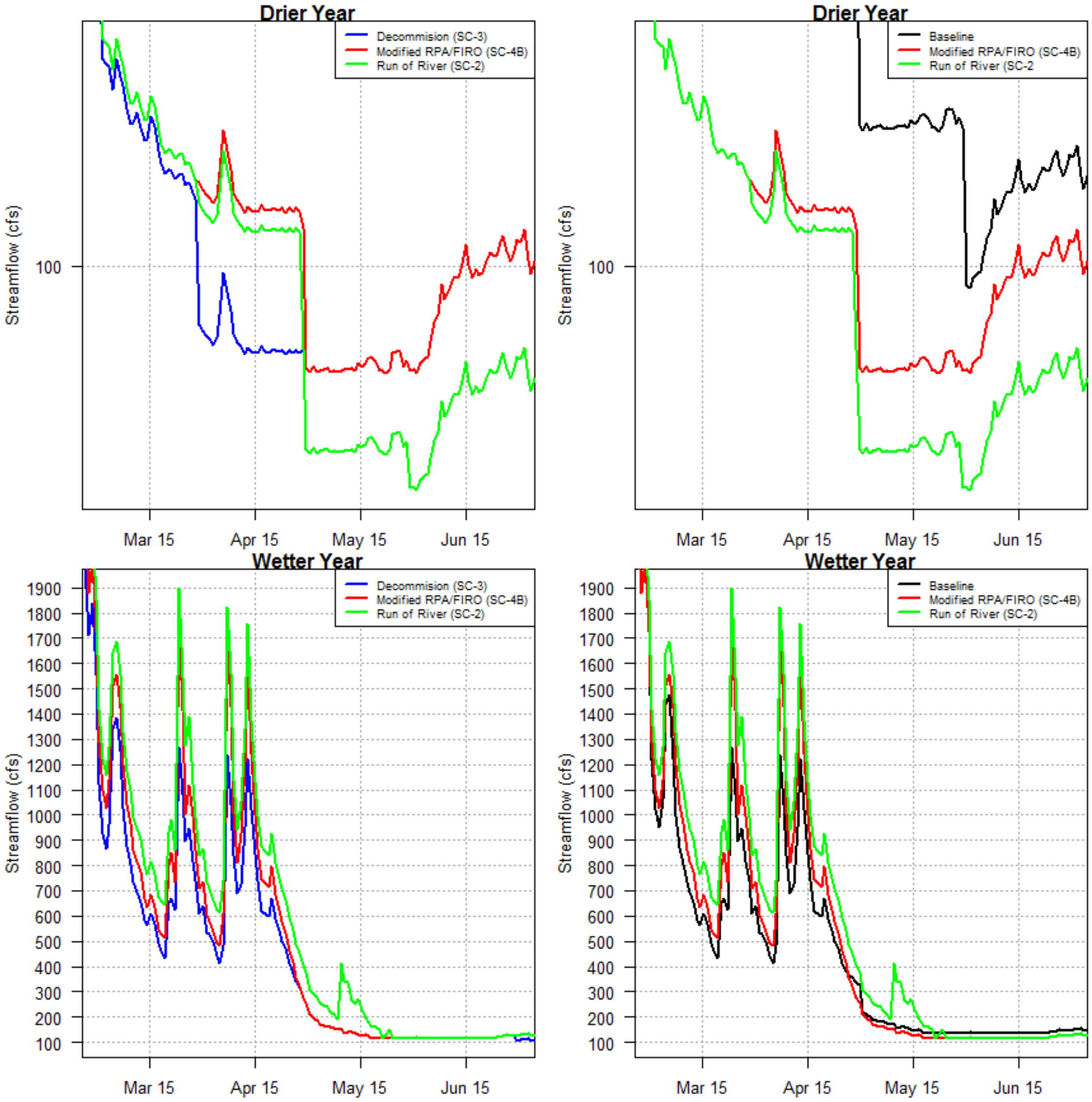

Figure 3-34. Example water years for drier (2015) and wetter (2011) years on the Russian River at Hopland during the spring recession period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

Summer baseflows (Figure 3-35)

- Scenario 3 and Scenario 2 were identical for drier years while Scenario 4B was identical to Scenario 2 in wetter years.
- Scenario 2 was lower than Scenario 4B for drier years and higher than both Scenario 3 and Scenario 4B for wetter years.
- Baseline was 30 to 50 cfs higher than the other Water Supply Scenarios.

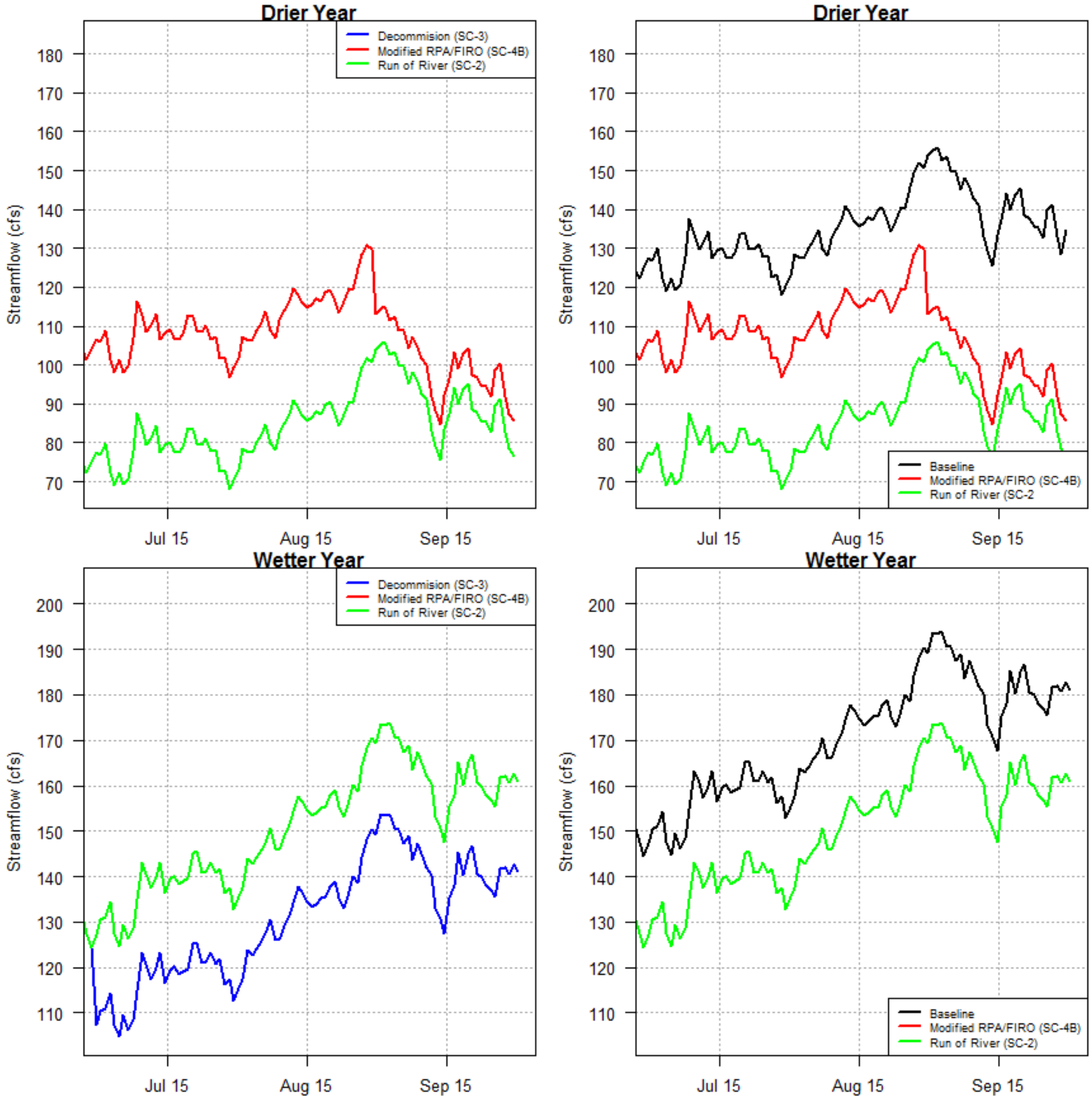

Figure 3-35. Example water years for drier (2015) and wetter (2011) years on the Russian River at Hopland during summer baseflow period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right. Note: when streamflow for two or more scenarios are identical, one or more data series may be hidden. Scenario 3 is hidden in the top right plot because it is identical to Scenario 2. In wetter years Scenario 4B is hidden because it is identical to Scenario 2 (bottom row of plots).

Fall streamflow (Figure 3-36)

- The drier year had a significant winter storm event modeled on December 10 through 12 which caused the larger y-axis scale in drier year plots relative to the wetter year plots for the fall season.
- There were minimal differences between all Water Supply Scenarios during the fall streamflow season.
- The largest differences were later in fall and early winter (end of December), when Scenario 4B and Baseline had higher streamflow magnitudes than Scenario 2, but in general this was only during drier years.

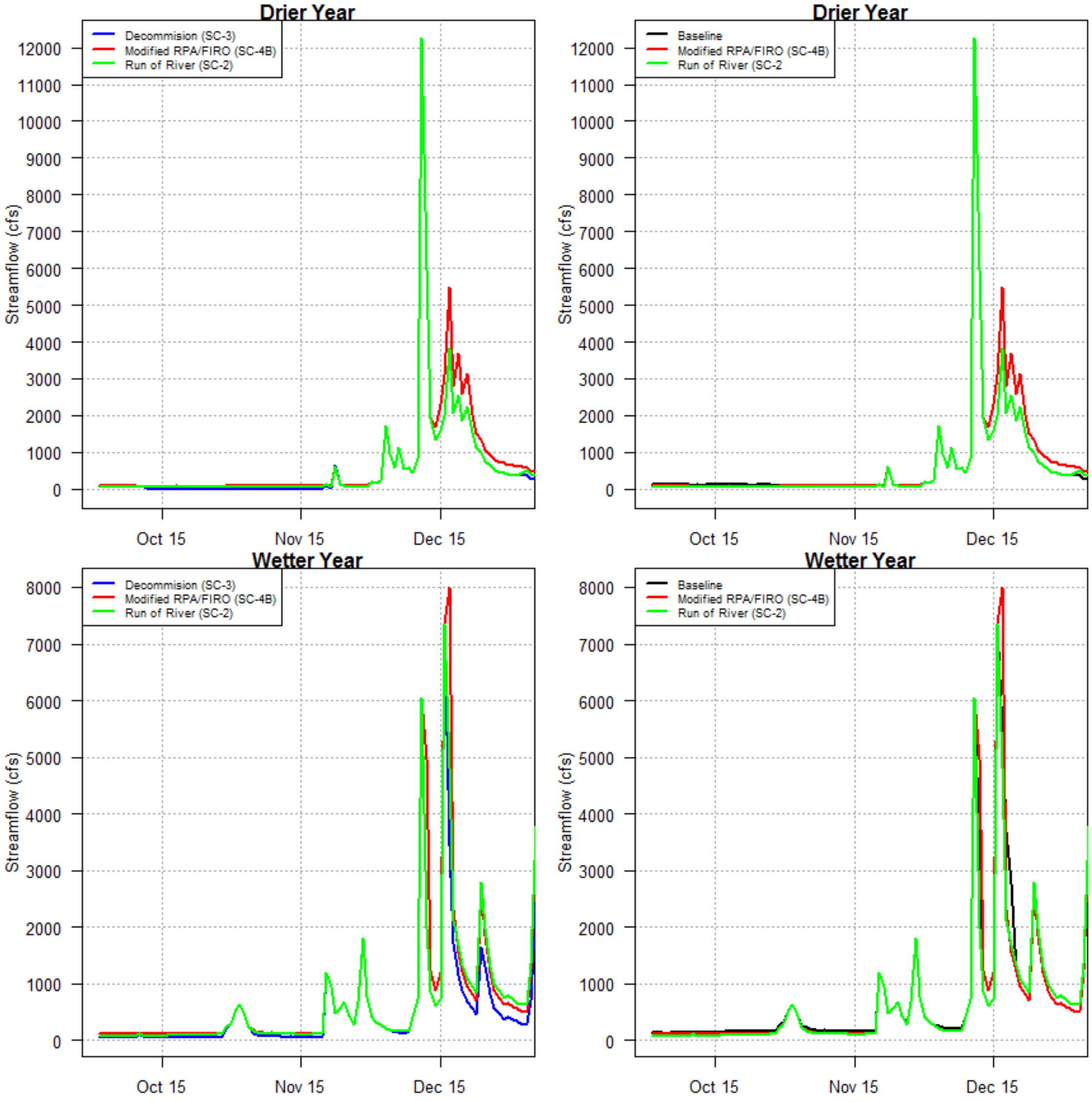

Figure 3-36. Example water years for drier (2015) and wetter (2011) years on the Russian River at Hopland during fall streamflow period. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right.

3.1.5.3 Flood Frequency Analysis

The flood frequency analysis shows similarity between all Water Supply Scenarios. Thus, the same flood intervals are expected to continue at Hopland on the Russian River under each Water Supply Scenario. Examination of the data showed that the flood events each year were similar, and likely are driven by the additional streamflows from tributaries and runoff between Coyote Valley Dam and Hopland.

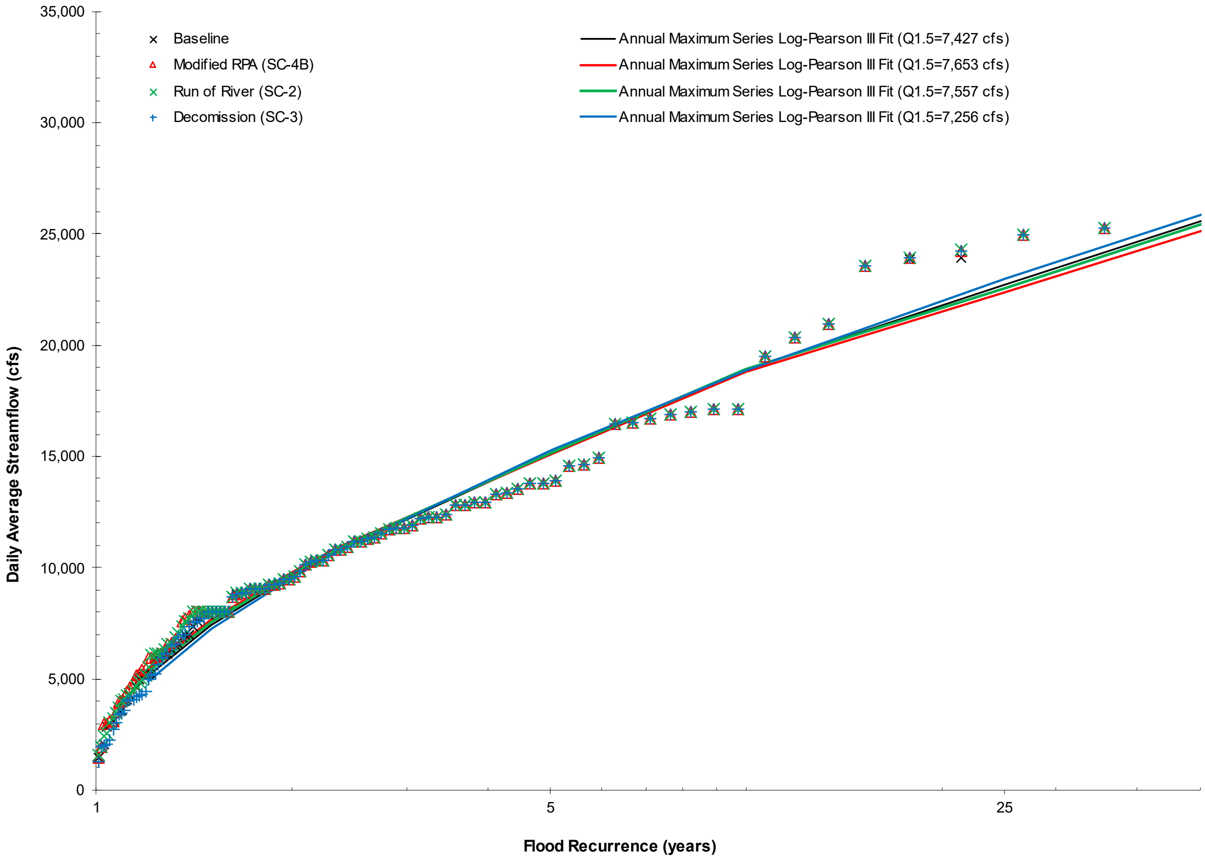

Figure 3-37. Flood frequency analysis for Russian River at Hopland for water years 1911 to 2017.

3.2 Water Temperature

This section utilizes available water temperature data and other information to describe expected general differences in water temperatures between Water Supply Scenarios, with a focus on understanding effects of dam removal on salmonid productivity in Eel River reaches downstream of Scott Dam. Section 3.2.2 describes expected differences in water temperatures amongst Water Supply Scenarios in the reach from Scott Dam to Cape Horn Dam, and Section 3.2.3 describes water temperatures from Cape Horn Dam to Tomki Creek. The potential effects of water temperature differences on salmonid productivity in the upper Eel River are outlined in Section 5.2. A description of water temperature considerations on the Russian River downstream of Coyote Valley Dam is presented in Section 3.2.4.

Changes in water temperatures downstream of Scott Dam are expected to affect productivity of anadromous salmonids by influencing their distribution, behavior, growth, and survival (SEC 1998, McCullough 1999, Myrick and Cech 2004, Marine and Cech 2004, Sloat and Reeves 2014). Water temperature also mediates ecological interactions (predation and competition) between salmonids, Sacramento Pikeminnow, and other non-native fish (Reese and Harvey 2002); influences juvenile salmonid outmigration timing (SEC 1998); and is a key factor in controlling numerous other ecological processes (Olden and Naiman 2010).

To help put potential effects of water temperatures expected under Water Supply Scenarios in context, Table 3-2 lists water temperature thresholds for physiological or ecological variables most relevant to salmonid productivity in the upper Eel River.

Table 3-1. Water temperature thresholds for key physiological or ecological variables important for growth and survival of juvenile steelhead and Chinook Salmon.

Physiological or ecological variable	Water temperature range or threshold (ºC)	References
Juvenile steelhead: high growth rate	15–19 [a]	Myrick and Cech (2001)
Juvenile steelhead: chronic to acute stress	>21	Cech and Myrick (1999)
Juvenile steelhead: reduced growth in presence of Sacramento Pikeminnow	>19	Reese and Harvey (2002)
Juvenile Chinook Salmon: high growth rate	15–19 [a]	Brett et al. (1982), Myrick and Cech (2004)
Juvenile Chinook Salmon: chronic to acute stress	>20	Brett (1952), Baker et al. (1995), Marine and Cech (2004)
Juvenile Chinook Salmon: predation vulnerability increases	>17	Marine and Cech (2004)

Notes: ºC = degrees Celsius

[a] Depending on food ration.

3.2.1 Approach

Recent thermograph data from the upper Eel River (PG&E 2017a, PG&E unpublished data) were used to approximate water temperatures expected under Baseline and Scenario 4B (Dams Remain). Under these scenarios, water temperatures in reaches of the upper Eel River downstream of Scott Dam are expected to be similar to recently measured temperatures, which are affected to varying degrees by coldwater releases from Lake Pillsbury, depending on water year type, season, and location (NMFS 2002, PG&E 2017a).

Since modelled predictions of water temperature were not available for Scenario 2 and Scenario 3 (Dam(s) Removed), expected differences in water temperature conditions were inferred based on general assumptions about changes in temperature following dam removal (e.g., absence of coldwater pool), modelled hydrology, existing temperature data from unaffected locations upstream and downstream of Lake Pillsbury, and the assumption of a gradual downstream increase in water temperature. General water temperature conditions expected under Scenario 2 and Scenario 3 were predicted based on interpretation of thermograph data from relevant locations upstream of Lake Pillsbury (unaffected by Project), between Scott and Cape Horn Dam, and downstream of Cape Horn Dam. Without coldwater releases, water temperatures in the reach from Scott Dam to Cape Horn Dam are expected to be intermediate between those currently recorded at the unaffected Bloody Rock site (upstream of Lake Pillsbury) and the site below Tomki Creek (about 4 miles downstream of Cape Horn Dam). While there is some year-to-year variation, the influence of coldwater releases from Lake Pillsbury generally extends downstream to Cape Horn Dam, with negligible influence expected downstream of Tomki Creek (PG&E 2017a).

To help demonstrate interannual variability and the potential range of water temperatures under different Water Supply Scenarios, this assessment used data for example wet (2011) and very dry (2015) water years. The assessment evaluated the following time periods, when changes in water temperature are expected to have the greatest effects on the key life stages that drive population productivity of Chinook Salmon and steelhead:

- March through June (juvenile salmonid spring growth and outmigration period);
- July through September (steelhead summer rearing period encompassing annual thermal maximum); and
- October through January (Chinook Salmon migration and spawning).

Year-round water temperature data are available for gage E-2 downstream of Scott Dam between 2010 and 2017 (PG&E unpublished data). However, available water temperature data from other sites in the upper Eel River are limited to the period from June to October in most years, with data collection starting as early as May in some years (PG&E 2017a). Therefore, evaluation of water temperatures focused primarily on spring through early fall, when data are available and when temperature differences between scenarios and effects on salmonid productivity are expected to be most pronounced.

3.2.2 Eel River from Scott Dam to Cape Horn Dam

3.2.2.1 March through June (Juvenile Growth and Outmigration)

Under Baseline and Scenario 4B, spring water temperatures in the reach between the dams are assumed to be similar to those recently recorded at E-2 gage, where daily mean water temperatures are typically about 7 to 11°C in March and April and about 11 to 16°C in May and June (PG&E, unpublished data, Figures 3-38 and 3-39). In May of some recent years (e.g., 2014), blockwater releases consisting of a combination of relatively warm surface water released through the spill gates and cooler needle valve releases were conducted to increase downstream temperatures and stimulate earlier emigration of juvenile Chinook Salmon. During these managed releases, daily mean water temperatures at the E-2 site can reach 15 to 17°C. These warmer releases are likely representative of water temperatures expected under Scenario 2 and Scenario 3, but could also occur under Baseline and Scenario 4B if similar surface water releases were to be implemented.

Spring water temperatures in the reach between the dams are artificially lower than would be expected in the absence of Scott Dam. Water temperature data from May and June (limited to a subset of years) indicate that measured temperatures between the dams are generally lower and more stable during this period than at sites both upstream of Lake Pillsbury and downstream of Cape Horn Dam (PG&E 2017a). For example, in May and June of 2015 (a drier year), daily mean water temperatures at E-2 ranged from about 12.5 to 18°C, while temperatures were 12 to 23°C at Bloody Rock and 14 to 25°C near Tomki Creek (Figure 3-39). Early spring water temperature data are not available at sites other than the E-2 gage, and it is uncertain if water temperatures between the dams in March and April are also artificially lower than they would be in the absence of Scott Dam.

In addition, compared with the consistently cold (but gradually warming) temperatures between the dams, relatively pronounced diel and day-to-day fluctuations are observed in May and June at sites not affected by the coldwater releases (PG&E unpublished data). These fluctuations likely reflect the considerable air temperature variations that occur in the spring. In general, under Scenario 2 and Scenario 3, it is assumed that springtime water temperatures between the dams will be intermediate between those typically recorded in the Eel River at Bloody Rock and Tomki Creek, and that temperatures will vary considerably within and between years relative to Baseline and Scenario 4B.

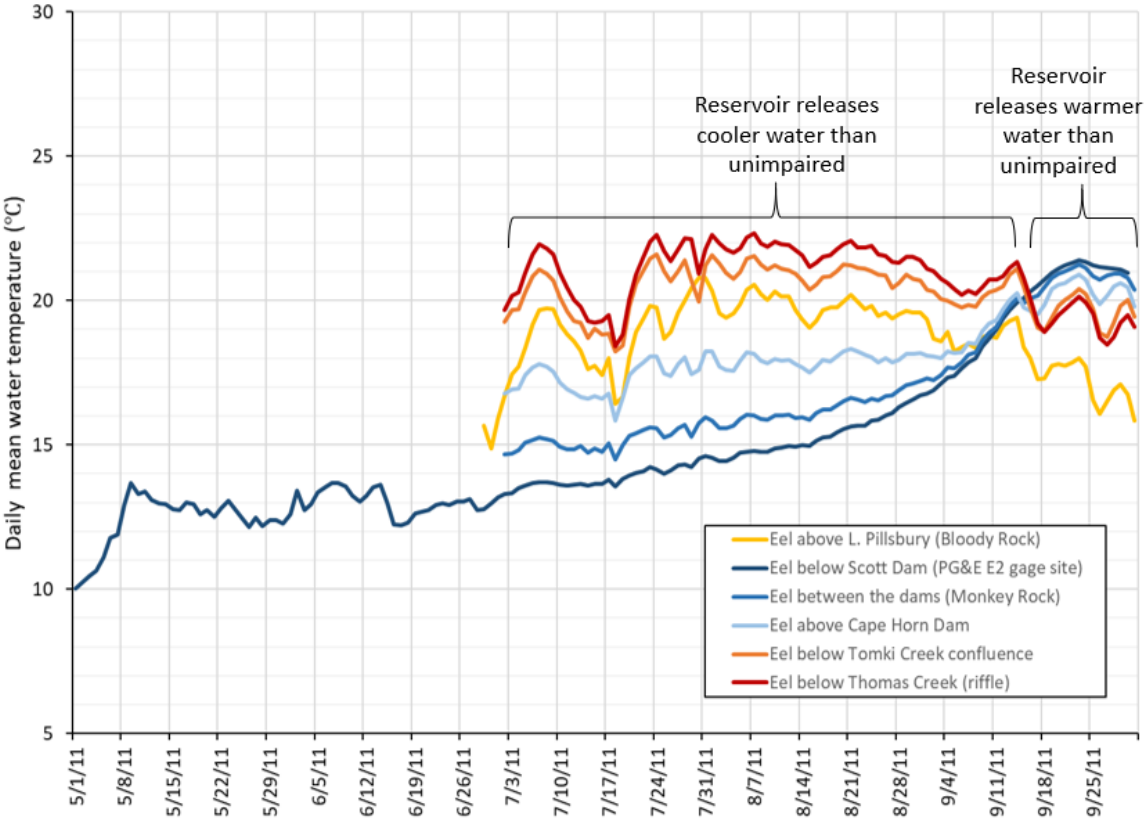

Figure 3-38. Daily mean water temperature at select sites in the upper Eel River from May through September 2011, an example wet water year (Data source: PG&E unpublished data).

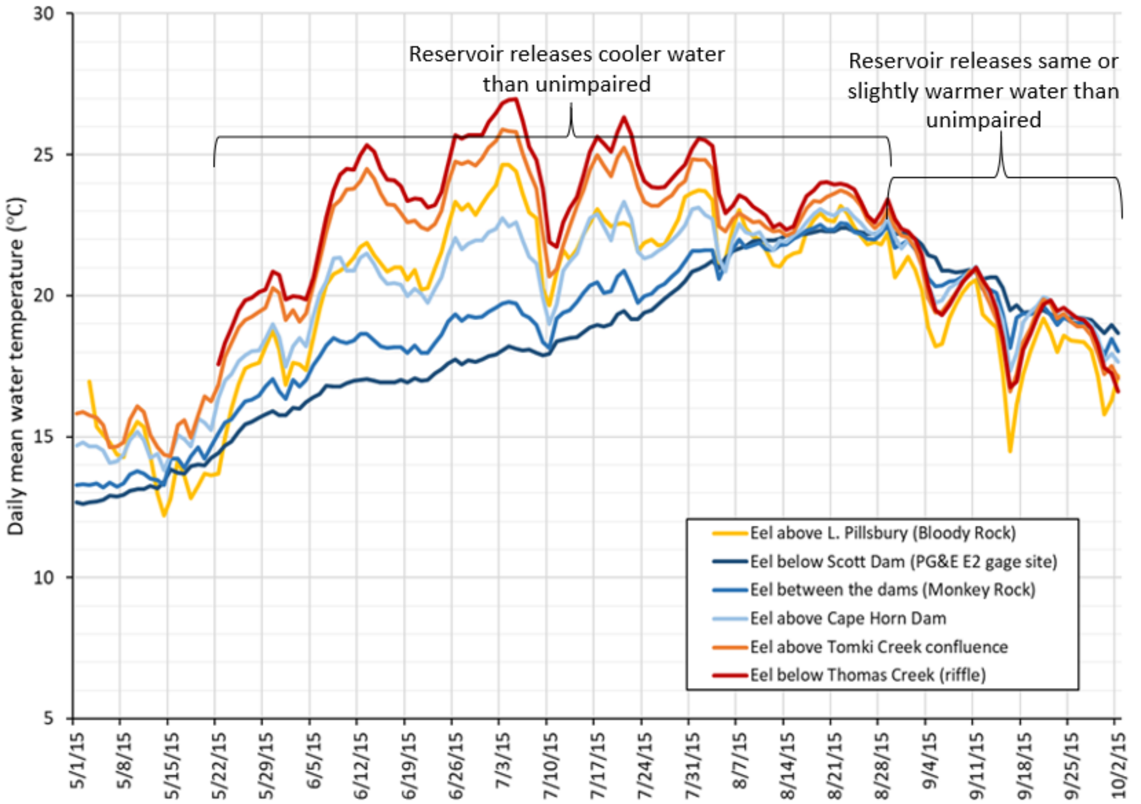

Figure 3-39. Daily mean water temperature at select sites in the upper Eel River from May through September 2015, an example very dry water year (Data source: PG&E unpublished data).

3.2.2.2 July through September (Summer Rearing)

Under Scenario 2 and Scenario 3, summer water temperatures are expected to be substantially higher than recently measured and under Baseline and Scenario 4B. With Scott Dam in place, summer water temperatures in the reach between the dams are typically considerably lower than both upstream and downstream sites not affected by the Project (PG&E 2017a, Figures 3-38 and 3-39). For example, during the wet year of 2011, daily mean water temperatures at sites between the dams ranged from approximately 14 to 18°C during the hottest period, while temperatures were approximately 19 to 21°C at a site upstream of Lake Pillsbury near Bloody Rock and 20 to 22°C near Tomki Creek (Figure 3-38). During the hottest period in the very dry water year of 2015, daily mean water temperatures at sites between the dams ranged from approximately 17 to 20°C, while temperatures were approximately 23 to 25°C at Bloody Rock and 25 to 26°C near Tomki Creek (Figure 3-39).

Under Scenario 2 and Scenario 3, without coldwater releases from Lake Pillsbury, summer water temperatures in the reach from Scott Dam to Cape Horn Dam would generally be expected to be intermediate between those recently measured at the Bloody Rock and below Tomki Creek sites. Thus, daily mean temperatures in the reach between the dams during the hottest periods of most years would be likely range from about 20 to 24°C. This estimate assumes a gradual increase in water temperature moving from upstream to downstream. This assumption is generally supported by limited water temperature data collected upstream of Lake Pillsbury and the minimally affected reach downstream of Cape Horn Dam, although coldwater tributaries and springs, hyporheic inputs, and thermal stratification in deep pools can cause locally cooler water temperatures (PG&E 2017a, PG&E unpublished data, Figures 3-38 and 3-39).

Notably, under current conditions, water temperatures between the dams gradually rise throughout the summer, typically peaking at 20 to 22°C in late August or September (versus July for reaches less affected by the Project), depending on water year. Particularly in wetter water years, such as 2011 and 2017, late summer and early fall water temperatures between Scott Dam and Cape Horn Dam often significantly exceed those of both upstream and downstream sites, suggesting releases from Scott Dam are artificially warm at this time and that the coldwater pool in Lake Pillsbury is insufficient to sustain optimal salmonid rearing conditions through the summer.

In some years, thermal stratification occurs in the mainstem Eel River in certain deep pools downstream of Cape Horn Dam during summer low flows (Kubicek 1977, PG&E 2016). For example, in June through August 2015, daily average water temperatures in a pool located upstream of Fish Creek (approximately 23 river miles downstream of Cape Horn Dam) ranged from approximately 12 to 15°C on the bottom and 22 to 28°C in a nearby riffle (Figure 3-40) (PG&E 2016). Such thermal stratification would be more likely to occur in deep pools between Scott Dam and Cape Horn Dam under the reduced summer streamflows predicted for Scenario 2 and Scenario 3 (Section 3.1.2).

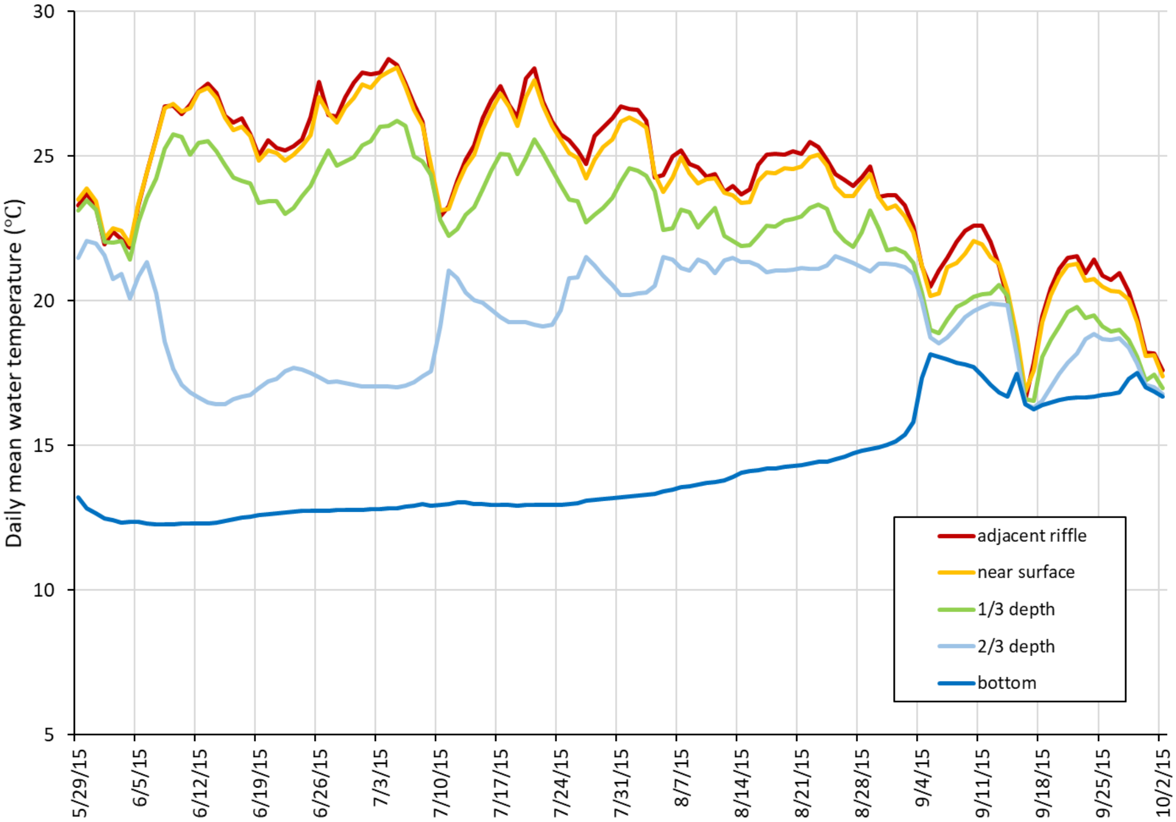

Figure 3-40. Thermal stratification observed in a pool located in the Eel River upstream of Fish Creek (approximately 23 river miles downstream of Cape Horn Dam) during summer 2015 (Data source: PG&E 2016).

3.2.2.3 October through January (Fall Migration and Spawning)

With the exception of the year-round data at the E-2 gage, water temperature data from PG&E monitoring sites in the upper Eel River (both upstream and downstream of Scott Dam) are limited after September. Data from the E-2 gage indicate that daily mean water temperature typically peaks there in September at approximately 20 to 22°C before dropping gradually throughout the fall and winter to a low of about 5 to 7°C, usually in January. This pattern would likely continue under Baseline and Scenario 4B. Data from unaffected locations in the upper Eel River (e.g., Bloody Rock stie and site below Tomki Creek) are not available to allow inferences to be made about fall and winter temperatures under Scenario 2 and Scenario 3. Overall, minimal differences in fall and winter temperatures are expected between Water Supply Scenarios.

3.2.3 Eel River Downstream of Cape Horn Dam

3.2.3.1 March through June (Juvenile Growth and Outmigration)

Recent thermograph data collected in May (available during 2014–2016 and 2007) at a sub-set of monitoring sites downstream of Cape Horn Dam indicate that water temperatures generally ranged from about 14 to 20°C in that month, depending on site and time (PG&E 2017a, PG&E unpublished data). Data from June (generally available at all sites from 2005 through 2017) indicate considerable interannual variation in water temperatures during that month (PG&E 2017a). In general, daily mean water temperatures in early June typically ranged from about 16 to 24°C, depending on location and year. In late June, daily mean temperatures generally ranged from about 18 to 26°C (PG&E 2017a). Available data from both May and June show a general upstream-to-downstream warming trend downstream of Cape Horn Dam.

Due to the overall similarity in streamflows downstream of Cape Horn Dam between Baseline and Scenario 4B (Section 3.1.3), May and June water temperatures under those Water Supply Scenarios would be expected to be similar to those measured downstream of Cape Horn Dam in recent years. Since spring streamflows under Scenario 2 are predicted to be somewhat lower than those in Baseline and Scenario 4B, water temperatures under Scenario 2 may be higher than the other scenarios, to some extent.

Spring diversions at Van Arsdale Reservoir under Baseline, Scenario 4B, and Scenario 2 would typically result in substantially lower streamflow downstream of Cape Horn Dam compared with Scenario 3 (Section 3.1.3). For example, during June of wetter water years, streamflow downstream of Cape Horn Dam would typically be 50 to 150 cfs for Baseline, Scenario 4B, and Scenario 2, versus 100 to 300 cfs under Scenario 3 (Section 3.1.3). Limited spring water temperature data and lack of water temperature modeling limit understanding of the magnitude of effects of these flow differences, but it can generally be inferred that temperatures would be cooler under the higher streamflows of Scenario 3 relative to the other scenarios.

3.2.3.2 July through September (Summer Rearing)

The relatively minor differences in summer streamflows downstream of Cape Horn Dam between Water Supply Scenarios (Section 3.1.3) are expected to have a minor influence on water temperatures in the reach. Additionally, as described above, water temperatures downstream of Cape Horn Dam, particularly below Tomki Creek, are expected to be minimally affected by coldwater releases in the summer compared with the reach between the dams.

For all Water Supply Scenarios, summer water temperatures are expected to be similar to recently measured values, with daily mean temperatures ranging from about 18 to 28°C depending on site, month, and year (PG&E 2017a, PG&E unpublished data). Thermograph data indicate an upstream-to-downstream warming trend downstream of Cape Horn Damn and generally cooler water temperatures in wet water years relative to dry water years.

3.2.3.3 October through January (Fall Migration and Spawning)

Lack of water temperature data and modeling limit inferences about the influence of the differences in fall and early winter streamflows downstream of Cape Horn Dam between Water Supply Scenarios on water temperatures (Section 3.1.3). With the potential exception of early October, limited early fall water temperature data indicate that for all Water Supply Scenarios, daily mean temperatures will likely remain below the thresholds for key physiological or ecological variables important for growth and survival of juvenile steelhead and Chinook Salmon (Table 3-1).

3.2.4 Russian River

Water temperature on the Russian River downstream of Coyote Valley Dam is regulated by releases from Coyote Valley Dam. Coyote Valley Dam has one hypolimnetic release point at the bottom of the dam, which maintains a relatively cool water temperature downstream until air temperature and tributary accretion become more influential (SCWA 2016). Because neither the Water Supply Scenarios or the Feasibility Study Alternatives impact any change to the release point to the river, any changes to release water temperature in the Russian River are not anticipated to be ecologically relevant and would only occur over a short stretch of river.

The main influence of the Water Supply Scenarios would be on the reservoir elevation of Lake Mendocino. The higher the reservoir elevation, the larger the hypolimnion and associated coldwater pool in Lake Mendocino would be available for releases to the Russian River. Under Baseline conditions, the released water remains colder than other sections of the river until mixing of the stratified water layers occurs in late summer/ early fall for normal and dry water years (SCWA 2016). Under Scenario 2, Scenario 3, and Scenario 4B, FIRO operations are expected to hold the reservoir higher and would increase the depth of the hypolimnion and provide more volume to be released over the summer (Addley et al. 2019, Delaney et al. 2020). It is not expected that this would result in any change to the water temperature of the hypolimnion but would result in a larger volume of cold water within the hypolimnion. While the volume of water is greater in the reservoir, changes to average summer releases from Coyote Valley Dam (going from 150 cfs to 130 cfs proposed in the Fish Habitat Flows and Water Rights Project Draft Environmental Impact Report; See Table SC4B-4 in SCWA 2016) are expected to be the main driver in preserving the coldwater pool in the summer during normal years.

Because changes to water temperature in the Russian River are anticipated to be minimal in response to Water Supply Scenarios and Feasibility Study Alternatives, and because the reservoir water temperature model was not available, we decided that additional analysis was not needed. Instead, we focused our attention on the Eel River, where changes to water temperature in response to Water Supply Scenarios and Feasibility Study Alternatives are expected to be more significant.

3.3 Fluvial Geomorphology

The effects of dams on downstream streamflow and sediment transport regimes are well documented (Williams and Wolman 1984, Power et al. 1996, Meitzen et al. 2013). In this section, we first provide a general overview of river response to dams. Next, we provide a discussion on potential changes to the Eel and Russian rivers in response to the Feasibility Study Alternatives and Water Supply Scenarios.

In salmon-bearing streams, which are typically alluvial or semi-alluvial with substantial coarse sediment supply, impoundment typically results in reduced coarse sediment supply (from reservoir trapping) to the reaches downstream of the dam. In addition, streamflow regulation and/or diversion can cause significant changes to the downstream flow regime that vary by the type/size of dam and project operations. For example, a large storage reservoir will likely have larger effects to streamflow and sediment regime than a small run-of-the-river diversion dam. Effects of reduced coarse sediment supply and impaired streamflow regime immediately downstream of the dam typically include:

- A simplified channel (reduced topographic and structural complexity, reduced habitat variability);
- Riffle coarsening;
- Fossilization of alluvial features;
- Reduced rates of channel migration;
- Reduction of fine sediment supply for overbank deposition; and
- Reduction in the amount and quality of spawning gravels available for anadromous salmonids (Graf 2006).

High streamflow releases from the dam can still transport sediment downstream, just less frequently, at a lower transport rate, and with lower duration of transport. Larger dams usually trap 100 percent of upstream coarse sediment supply, and due to the less-frequent high streamflow releases from the dam and lack of coarse sediment supply from reaches upstream of the dam, mobilized bed sediments downstream of the dam are not replaced. In some reaches, reduced coarse sediment supply may result in incision to bedrock and a complete loss of channel dynamism and floodplain connectivity. Bed surface coarsening typically occurs due to finer material being selectively transported downstream, and the frequency of bed mobilization decreases as grain size increases over time.

Changes in streamflow and sediment supply often cause changes in aquatic habitat for salmonids. Reduced peak flows result in a less dynamic channel and reduced floodplain function. These effects often contribute to poorer habitat conditions for anadromous salmonids and other aquatic species (such as the Foothill Yellow-legged Frog). In addition to blocking upstream fish migration access, dams and streamflow regulation often result in degraded spawning habitat (reduced coarse sediment supply results in impaired spawning riffles), poor-quality rearing habitat downstream of the dam (channel simplification decreases hydraulic refugia and disconnected floodplains reduce access to food and cover resources) and less-productive benthic macroinvertebrate habitat (armored streambed results in reduced interstitial habitat).

Changes in streamflow and sediment supply can affect riparian vegetation downstream of the dam, which can then translate into changes in channel form and processes, and ultimately change habitat for salmonids and other aquatic and terrestrial species. For example, if tributaries downstream of the dam are still delivering fine sediment, encroaching riparian vegetation, less regularly scoured by high streamflows and coarse sediment movement, may trap fine sediment and grow into riparian berms. This riparian encroachment and berm-building process serves to further simplify streamflow patterns and constrict streamflow into a smaller channel than previously existed. Most of the effects listed above decay longitudinally with distance downstream of the dam as tributaries offset the sediment deficit and contribute to a more natural hydrograph.

3.3.1 Approach

The following sections describe our hypotheses of how fluvial process, channel form, and sediment budget would change from the current conditions under Water Supply Scenarios and the Feasibility Study Alternatives. Four reaches are described:

1. Eel River currently inundated by Lake Pillsbury
2. Eel River from Scott Dam to Cape Horn Dam
3. Eel River downstream of Cape Horn Dam
4. Russian River downstream of the WBRR confluence

The time scale of the discussion assumes evolution many years after dam removal, and thus does not distinguish between differing potential methods and timelines for dam removal (i.e., rapid versus phased). There is limited available information on fluvial geomorphology in the upper Eel River, so much of the discussion is based on literature review and field experience that we apply to the upper Eel River based on its specific characteristics. To provide a framework for considering linkages between these fluvial geomorphic changes to river ecology, each subsection focuses on three components of fluvial geomorphic change: 1) Fluvial Processes, 2) Sediment Budget, and 3) Channel Form. These changes to fluvial geomorphic processes and form cause corresponding ecological changes as discussed above.

3.3.2 Eel River Currently Inundated by Lake Pillsbury

The historic Eel River channel within Lake Pillsbury has been buried by approximately 21 million cubic yards (yd^3) of sediment over the 100 years since Scott Dam was completed. We estimate that approximately 12 million yd^3 of that sediment would be mobilized if Scott Dam is removed and no mechanical removal of this sediment occurs. The fate of this sediment can have important implications for ecological function and river channel morphology in downstream reaches.

Either by mechanical sediment management or natural river erosion into the reservoir sediment deposits, fluvial processes immediately following Scott Dam removal would be dominated by the river incising into remaining reservoir sediments, eventually returning close to or at its original pre-dam location and grade. The most dynamic and unstable period will occur with the first storm following removal and will be scaled to the magnitude of that storm. The process of channel incision will be different between the Eel River arm and the Rice Fork arm. The Rice Fork arm is very confined by adjacent hillslopes, and thus is anticipated to largely incise without the ability to laterally migrate, such that most of the sediment stored in the Rice Fork arm will be transported downstream. The Eel River arm is much wider, and thus it is anticipated to migrate substantially across the reservoir sediments as it incises. Based on observations of similar dam removal projects, we expect that the channel will return to its original location in 2 to 5 years, depending on the sequence of storms during that period. The channels will have very high sediment transport rates, and the longitudinal profile will be simple (lacking substantial pools) as the channel processes its sediment. Once the channels return to their original profile location and sediment supply recedes to background levels, more complex topography can be expected as roughness elements cause pools and riffles to re-form.

During this transition, the local sediment budget would be very dynamic, with output greatly exceeding input (and thereby reducing sediment storage) within the Lake Pillsbury inundation area. Once the river has down cut to its original bed elevation and equilibrium grade, the sediment budget would still be slightly out of balance as the river continues to adjust laterally through migration into remaining terraces. Once the channel stabilizes laterally, the sediment budget would approach a balanced state where output is approximately equal to input from the upper watershed.

Channel form and habitat quality during this transition period will be poor and transient as sediment is evacuated from the former reservoir and the channel incises. Once the channel returns to its original profile, channel form and habitat quality and quantity should begin to improve as the channel stabilizes and begins interacting with bedrock roughness elements and riparian vegetation begins to re-establish. The Rice Fork and smaller tributaries should stabilize earlier than the Eel River arm due to the large width of the Eel River arm. The time scale for the transition to a more stable channel form with improved habitat could be 5 to 10 years, driven by the time of channel incision to the original profile and ability of riparian vegetation to re-establish and contribute to stability.

Under Baseline and Scenario 4B (Dams Remain), the reservoir will remain, so no changes would be expected.

3.3.3 Eel River from Scott Dam to Cape Horn Dam

Under current conditions, the reach between Scott Dam and Soda Creek reflects typical channel response to upstream streamflow and sediment regulation (Williams and Wolman 1984). Instream alluvial bars and floodplains are coarsened and less mobile, and dense riparian vegetation confines and isolates the low streamflow channel, rendering it virtually incapable of meandering within its pre-dam floodplain in areas with wider valley walls (e.g., between Scott Dam and Soda Creek). In more confined reaches downstream of Soda Creek, riparian encroachment continues, with smaller scale bars within the riparian vegetation. Substantial sediment supply from Soda Creek, coupled with infrequent high streamflows capable of moving the sediment, contributes to more active alluvial features (bars, riffles, floodplains) downstream of Soda Creek as the river enters a confined canyon. Based on field observations and aerial photography, this more confined canyon, combined with the channelizing effect of the riparian vegetation, fosters higher sediment transport rates and low amounts of coarse sediment storage (bars). Near the downstream end of the canyon, sediments from Soda Creek and other downstream tributaries accumulate in valley expansion areas via alternate bars and other bar features, including storage within Van Arsdale Reservoir.

With removal of Scott Dam, hydrology will revert to an unimpaired streamflow regime, and the magnitude of commonly occurring floods (in the 1.5 to 5-year range) will increase compared to existing conditions, as these events will no longer be captured or attenuated by Lake Pillsbury. For example, the predicted 1.5-year streamflow would increase from approximately 6,000 to 9,000 cfs (Section 3.1). The combination of increased flood magnitude and increased sediment supply should greatly enhance bed mobilization magnitude, duration, and frequency. In addition, the unimpaired streamflow regime and increased sediment load should increase scour and removal of encroaching riparian vegetation in the reach, facilitating a transition into a more natural morphology over time, with open cobble/boulder bars and less vegetation.

With removal of Scott Dam, the change to the sediment budget will be dramatic. Even if a substantial amount of Lake Pillsbury erodible sediments is removed and stockpiled prior to dam removal, the sediment supply will reflect unimpaired conditions based on the sediment supply from the upper Eel River watershed. Accordingly, the sediment budget will be substantially greater after dam removal. In the first 2 to 5 years after dam removal, as the river incises through the reservoir sediments, the sediment budget will become much greater and will likely cause substantial changes to channel form in the reach. Reach-scale aggradation (sedimentation) is likely to occur in areas with greater valley width (Soda Creek area) and could cause short-term filling of the channel with sediment. Reaches with narrower valley width should route this sediment more efficiently and have less aggradation risk, but local deposition will certainly occur. The general amount of channel aggradation should decay in the downstream direction as streamflows increase and the proportion of “new” sediment decreases compared to background, pre-dam removal levels.

Lateral migration will likely increase in those reaches with greater valley width, and it is probable that new cobble/gravel bars will form over the entire reach. The channel morphology is expected to evolve into something similar to what currently exists on the Eel River upstream of Lake Pillsbury, with exposed cobble/gravel bars, forced meanders within a confined valley, deep pools (after the sediment budget decreases after reservoir sediments are evacuated), and sparse riparian vegetation along the channel margins. The transition from a riparian encroached channel and more seasonably warming water temperatures should improve physical habitat for Foothill Yellow-legged Frog reproduction. In addition, a more variable streamflow regime, combined with frequently mobilized bed substrates, should reduce the frequency and magnitude of excessive algal blooms in the reach and promote invertebrate productivity.

During and after Scott Dam removal, suspended sediment concentrations will be extremely large as the river down cuts through the reservoir sediments. Depending on how the dam is removed (quickly versus phased over years), the magnitude, duration, and frequency of extremely high suspended sediment concentration events will vary and could exceed 200,000 milligrams per liter (mg/L). The general strategy for dam removal in similar projects along the west coast is to rapidly remove the dam and allow a single pulse of high suspended sediment concentration, rather than a prolonged event and/or multiple events of high suspended sediment that would result from a phased dam removal. The predicted suspended sediment concentrations of these various dam removal options will be evaluated in the next phase of the project and compared to biological thresholds for suspended sediment. After the river stabilizes upstream of Scott Dam, we expect to see seasonal changes in suspended sediment and turbidity. Currently, Lake Pillsbury releases in the summer are often much more turbid from biological sources than the river upstream of Lake Pillsbury, and removal of Lake Pillsbury should cause this reach to reflect less turbid summer conditions like the reach upstream of Lake Pillsbury. In contrast, during winter high streamflows, Lake Pillsbury captures suspended sediments, and corresponding downstream releases are less turbid than the river upstream of the reservoir. However, during very large floods, the entire reservoir can become turbid and remain so for long periods of time, causing prolonged high turbidity releases downstream.

Under Scenario 4B, high streamflow hydrology will not appreciably change, and thus the fluvial geomorphology of the Eel River from Scott Dam to Cape Horn Dam will likely remain similar to existing conditions.

3.3.4 Eel River Downstream of Cape Horn Dam

Under current conditions, the reach downstream of Cape Horn Dam is slightly wider valley, and benefits from cumulative streamflow and sediment contributions from upstream tributaries and larger tributaries in the reach. The fluvial geomorphic effect of Scott Dam continues to decay downstream of the Tomki Creek confluence and is difficult to impossible to detect downstream of the Middle Fork Eel River confluence.

The existing sediment budget in this reach is less affected by Scott Dam due to these cumulative tributary sediment contributions. We also hypothesize that most sediment is currently routing through Van Arsdale Reservoir based on active bars on the inside bend of the reservoir and abrasion of the concrete on Cape Horn Dam on the right (east) side of the dam. Therefore, the sediment budget in this reach is increasing longitudinally, and is much larger than the upstream reach. Accordingly, the channel morphology is also more reflective of a river with more sediment and greater disturbance regime. Some riparian encroachment is still evident in some areas, but the river is transitioning to open boulder/cobble/gravel bars, with limited vegetation on the bars, and a more mobile bed surface.

Under Scenario 2 and Scenario 3, changes to fluvial process, sediment budget, and channel morphology would be similar to that described in Section 3.3.3, with a few differences. With Cape Horn Dam removed, the potential sediment budget would increase again for the short term if the 1 million yd^3 of sediment currently stored in the reservoir were allowed to flush downstream. This sediment would quickly route downstream but would likely cause some local aggradation and channel simplification (shallower pools, more lateral movement) for a few years. When considering the channel response to sediment from upstream of Scott Dam, the valley width in this reach is wider than the upstream reach, and the degree of change from existing conditions would be less because the river is already less encroached than upstream of Cape Horn Dam. There would also be a slight difference in high streamflow hydrology in this reach as the increased diversion capacity (from 240 cfs to 300 cfs) would slightly reduce the magnitude of winter high streamflows, but the approximately 60 cfs reduction in winter streamflows is considered minor and would almost certainly have no effect on downstream fluvial processes or channel morphology.

For Scenario 2 where Scott Dam is removed and Cape Horn Dam remains, the primary difference would be that the 1 million yd^3 of sediment in Van Arsdale Reservoir would not be flushed downstream. In this case, the changes to this reach would be less pronounced than the changes described above, with less risk of pool filling and retaining a more stable channel.

3.3.5 Russian River Downstream of the West Branch Confluence

The Russian River downstream of the WBRR confluence has been regulated by Lake Mendocino since 1958. Lake Mendocino traps all sediment from the East Branch Russian River, yet the WBRR continues to deliver considerable amounts of sediment to the river. Despite streamflow and sediment contribution from the WBRR, this reach is extremely encroached with riparian vegetation and has incised considerably over the years.

The only potential changes from the Project on this reach would be hydrologic changes since the reach is physically isolated from any geomorphic change from the Project. Hydrologic modeling of various Water Supply Scenarios by Addley et al. (2019) shows that the high streamflow regime is not significantly affected by the Project. The biggest changes to hydrology are due to implementation of FIRO and the proposed Fish Flow EIS/EIR streamflow regime. Potential geomorphic changes from those two non-Project hydrologic changes are not described here but are expected to be minor.

4 ECOLOGICAL TRADEOFFS

The Project has changed the hydrologic, physical, chemical, and ecological processes of the upper Eel River and Russian River to varying degrees. Project facilities and operations have blocked fish passage, altered the natural streamflow and temperature regime, affected river geomorphology, facilitated establishment of non-native species, and changed the quality of habitat available to native anadromous salmonids, resident species, and amphibian populations. The Project also supplies the Eel River from Scott Dam to Cape Horn Dam with coldwater hypolimnetic streamflow releases that can support rearing salmonids over the summer in some years. The dams also give natural resource managers the opportunity to use blockwater streamflow releases to trigger upstream and downstream migration events if needed.

This evaluation of ecological tradeoffs aims to provide information to the Parties on how different Feasibility Study Alternatives and Water Supply Scenarios may affect ecosystem components and river function in both the Eel River and the Russian River. The goal of this section is to address how different Water Supply Scenarios and infrastructure changes would affect or change each of the analysis topics listed below based on the outcomes of Water Supply Scenario 2 (Alternative 2 and Alternative 3) and Scenario 4B relative to Baseline and Scenario 3.

We provide an overview of the potential changes to:

1. Salmonid Flow-Habitat Relationships
2. Chinook Salmon Migration
3. Pacific Lamprey Responses
4. Non-Native Predator Responses
5. Herpetological Species Responses
6. Benthic Macroinvertebrates Responses
7. Riparian Habitat
8. Cyanobacteria and Algal Growth

One of the most important questions is the amount of habitat available to anadromous Chinook Salmon and steelhead upstream of Scott Dam if it were removed relative to the assumed benefits of the cool tailwater habitat created downstream of Scott Dam. That specific topic is not addressed in this section but is explicitly analyzed in Section 5.

4.1 Approach

We reviewed relevant information from peer-reviewed literature, Eel River and Russian River specific reports, data, and existing models (when available) to develop anticipated outcomes for each topic listed above at three hydrological focal areas listed below. The analyses were supported and informed by a summary of modeled (HEC-ResSim) Water Supply Scenarios (Table 1-1) and based on what infrastructure would be removed or updated under the Feasibility Study Alternatives (Table 1-2). We analyzed Scenario 2, Scenario 4B, Scenario 3, and Baseline coupled with Feasibility Study Alternatives 1, 2, and 3 (Table 4-1). This allowed comparison of potential future conditions to both existing conditions and a situation in which the project is not relicensed by the Parties (assumed to be Scenario 3 with decommissioning). The potential outcomes are projected for conditions once all the effects of Feasibility Study Alternatives have occurred and the river has achieved equilibrium (e.g., after Lake Pillsbury sediment is flushed). We analyzed responses for three focal areas (Figure 3-1) that were assumed to be most susceptible to alterations by infrastructure and Water Supply Scenarios:

1. Eel River from Scott Dam to Cape Horn Dam – This location would be significantly altered in both streamflow and infrastructure under Scenario 2 (Alternative 2 and Alternative 3) and Scenario 3. Removal of Scott Dam would return the Eel River to unimpaired streamflow and sediment routing conditions and open access to previously blocked off habitat. The additional removal of Cape Horn Dam would return the Eel River downstream of Scott Dam to a more natural channel by removing Van Arsdale Reservoir.
2. Eel River Downstream of Cape Horn Dam – This location would also be significantly altered by either the removal of Cape Horn Dam and/or Scott Dam allowing unimpeded fish passage under Scenario 3 or Scenario 2 (Alternative 3) and changes to streamflow based on different diversion schedules for Scenario 2 with dam retention (Alternative 2) and Scenario 4B (Alternative 1).
3. Russian River Downstream of the WBRR Confluence – This location would be altered by changing streamflow releases from Coyote Valley Dam under Scenario 2, Scenario 3, and Scenario 4B. No infrastructure changes due to the Feasibility Study Alternatives are anticipated.

Table 4-1. Water Supply Scenarios used in evaluating ecological trade-offs for the Eel and Russian rivers and how Water Supply Scenarios and Feasibility Study Alternatives (Table 1-1 and Table 1-2) are expected to affect habitat and streamflow.

Focal Areas	**Water Supply Scenario (and Feasibility Study Alternative)**	**Description**	**Alterations to Streamflows and Habitat**
Eel River from Scott Dam to Cape Horn Dam	Baseline (Existing Conditions)	Current Operations; Scott Dam Remains, Cape Horn Dam Remains, Existing Diversion	No alterations; included to compare the current streamflow regime to other Water Supply Scenarios to gage outcomes.
	Scenario 2 (Alternative 2)	Scott Dam Removed; Cape Horn Remains; Existing Diversion	Opens access to habitat upstream of Scott Dam; hydrology just downstream of Scott Dam would be similar to unimpaired but backwatering from Van Arsdale Reservoir would remain.
	Scenario 2 (Alternative 3)	Scott Dam Removed; Cape Horn Remains; Alternative Diversion	Returns the river to an unimpaired streamflow regime in the focal area. Hydrology is the same as Scenario 3.
	Scenario 3 (Decommissioning)	Scott Dam removed, Cape Horn Dam Removed, No Diversion	Returns the river to an unimpaired streamflow regime.
	Scenario 4B (Alternative 1)	Scott Dam Remains, Cape Horn Dam Remains, Diversion Remains	Hydrology similar to Baseline in focal area.
Eel River Downstream of Cape Horn Dam	Baseline (Existing Conditions)	Current Operations; Scott Dam Remains, Cape Horn Dam Remains, Existing Diversion	No alterations; included to compare the current streamflow regime to other Water Supply Scenarios to gage outcomes.
	Scenario 2 (Alternative 2)	Scott Dam Removed; Cape Horn Remains; Existing Diversion	Similar to Baseline in focal area.
	Scenario 2 (Alternative 3)	Scott Dam Removed; Cape Horn Remains; Alternative Diversion	Diversion changes streamflow downstream of Cape Horn Dam but with a more natural shape to the hydrograph.
	Scenario 3 (Decommissioning)	Scott Dam removed, Cape Horn Dam Removed, No Diversion	Returns the river to an unimpaired flows regime.
	Scenario 4B (Alternative 1)	Scott Dam Remains, Cape Horn Dam Remains, Diversion Remains	Similar to Baseline but with an alternative diversion strategy (amount of streamflow and timing).
Russian River Downstream of the West Branch Russian River Confluence	Baseline (Existing Conditions)	Current Operations; Scott Dam Remains, Cape Horn Dam Remains, Existing Diversion	No alterations; included to compare the current streamflow regime to other Water Supply Scenarios to gage outcomes.
	Scenario 2 (Alternative 2)	Scott Dam Removed; Cape Horn Remains; Existing Diversion	Similar to Baseline in focal area.
	Scenario 2 (Alternative 3)	Scott Dam Removed; Cape Horn Remains; Alternative Diversion	Similar to Baseline in focal area.
	Scenario 3 (Decommissioning)	Scott Dam removed, Cape Horn Dam Removed, No Diversion	No diversion from the Eel River. Scenario 3 would have the greatest effect to streamflow and habitat in focal area.

Focal Areas	Water Supply Scenario (and Feasibility Study Alternative)	Description	Alterations to Streamflows and Habitat
	Scenario 4B (Alternative 1)	Scott Dam Remains, Cape Horn Dam Remains, Diversion Remains	Similar to Baseline but the altered diversion strategy would alter hydrograph.

While all Water Supply Scenarios at each location were considered, our analysis focused on the most important changes that would drive a significant response in the target of the analysis (e.g., adult Chinook Salmon migration). Each analysis includes a general background and introduction, which provides a general overview of the science and data available to inform the analysis. Next, each location was analyzed using literature review, expert opinion, and/or existing data/models, and categorized by specific mechanisms at each analysis location. For some of the analyses (e.g., hydrological analysis, riparian plant species, streamflow habitat relationships, and fish passage), specific approaches and methods are detailed within the section below because they utilized existing models or data, and thus provide a quantitative analysis instead of a qualitative analysis.

We evaluated the potential outcome of Water Supply Scenarios and Feasibility Study Alternatives on each metric of the analysis if the remainder of the system were held constant. For example, we assessed benthic macroinvertebrates and food webs in the context of fisheries completely independent of predation or water temperature. A more thorough discussion of what metrics may be most limiting to salmonid productivity is available in Section 5.

4.2 Salmonid Flow-Habitat Relationships

The effects of hydrologic changes of Water Supply Scenarios on aquatic habitat on the Eel River and Russian River were explored at each node (Figure 3-1). On the Eel River, we selected one weighted usable area (WUA) study site (Table 4-2 and Figure 4-1) downstream of Scott Dam (Trout Creek site) which is the most directly affected by changes to Scott Dam operations. We selected two WUA study sites downstream of Cape Horn Dam: Cape Horn site and Big Bend site, which are immediately downstream of Cape Horn Dam and downstream of the Project, respectively. For the Russian River, we chose two WUA study sites that represent the geomorphic characteristics of the upper Russian River, the reach between the confluence with the West Branch Russian River, and the confluence with Dry Creek near Healdsburg. The Ukiah site, which is mostly alluvial, and the Hopland site, which is influenced by bedrock/constriction (SCWA 2016).

Table 4-2. Sites selected for the flow-habitat relationship analysis; the Water Supply Scenario comparison used to evaluate flow-habitat relationships; and the life stage-specific time periods evaluated for both the Eel and Russian river watersheds.

Flow-habitat Evaluation Site	**Hydrological Source (node)**	**Water Supply Scenario Comparisons**	**Time Period Evaluated**[3]
Eel River			
Trout Creek	Scott Dam (gage E-2)	Scenario 2/3 to Scenario 4B Scenario 2/3 to Baseline Baseline to Scenario 4B	Oct 1–Dec 31 (Chinook Spawning) Jan 1–May 31 (Chinook Juvenile Rearing) Dec 1–Apr 30 (Steelhead Spawning) Jan 1–Dec 31 (Steelhead Juvenile Rearing) Mar 1–Jul 31 (Steelhead Fry Rearing)
Cape Horn	Cape Horn Dam (gage E-11)	Scenario 3 to Scenario 2 Scenario 3 to Scenario 4B Baseline to Scenario 2 Baseline to Scenario 4B	
Big Bend	Above Outlet Creek[1]		
Russian River			
Ukiah	Russian River downstream of WBRR confluence[2]	Scenario 3 to Scenario 2 Scenario 3 to Scenario 4B Baseline to Scenario 2 Baseline to Scenario 4B	Mar 1–May 31 (Chinook Fry and Juvenile Rearing) Jan 1–Dec 31 (Steelhead fry and Juvenile Rearing)
Hopland	Russian River at Hopland		

[1] Hydrology at Above Outlet Creek was calculated by using adding accretion from gage E-11 to Above Outlet Creek to each gage E-11 Water Supply Scenario. Accretion from gage E-11 to Outlet Creek was developed by subtracting Unimpaired streamflow from Above Outlet Creek from gage E-11 Scenario 3 (PG&E Unimpaired Data).

[2] Hydrology at Ukiah was calculated by adding HEC-ResSim outputs for Baseline from West Branch Russian River (WBRR) and East Branch Russian River Downstream of Lake Mendocino.

[3] The time periods evaluated for the Eel River sites are based on an earlier version of the Eel River salmonid life history tables presented in Section 2 which have since been refined based on additional information and therefore do not exactly match the timings applied to analyses in Sections 4.2 and 4.3. Overall, we do not expect the differences in timing to impact results of these analyses, since the stream flows evaluated are still representative of seasonal habitat conditions expected for each life stage.

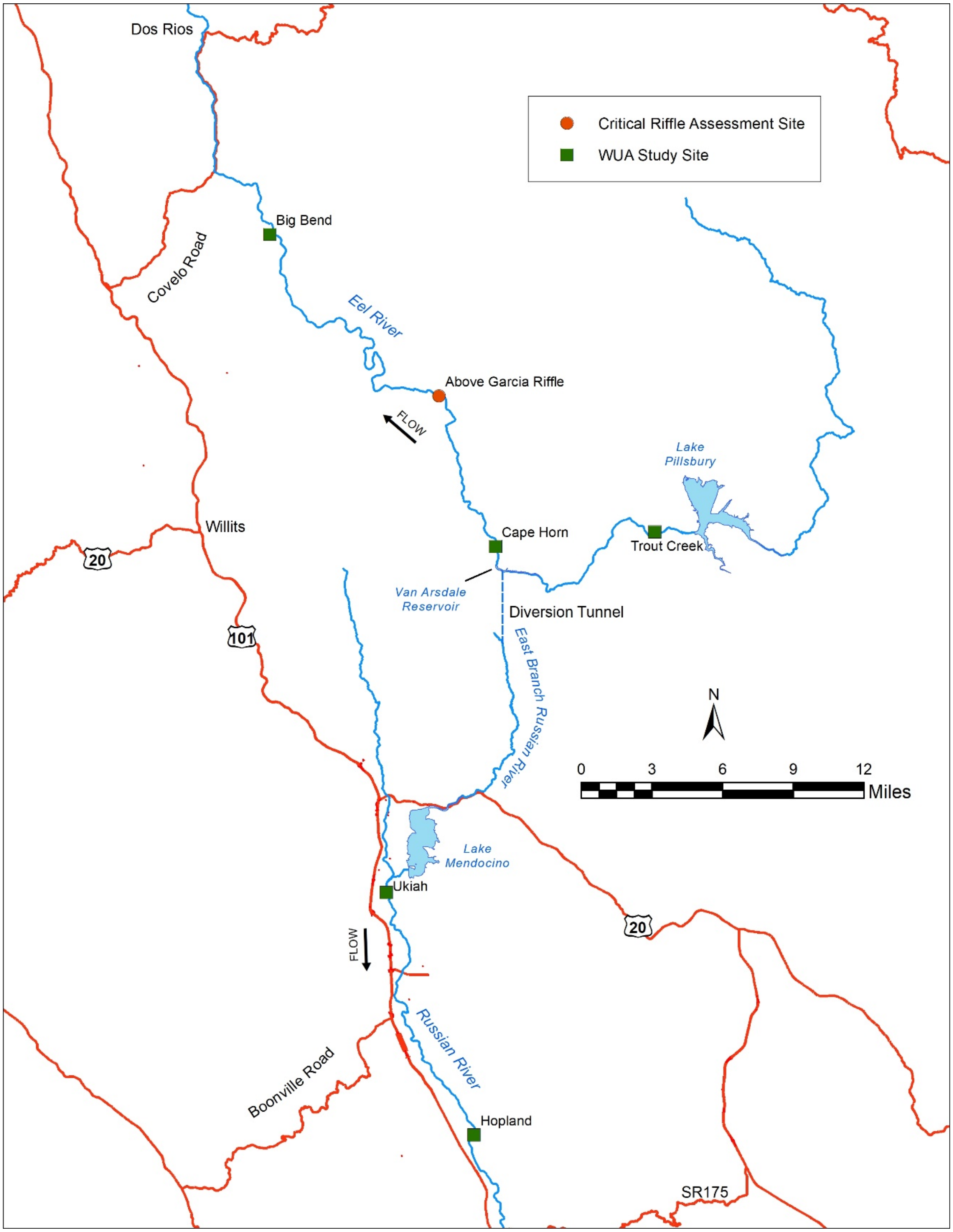

Figure 4-1. Weighted useable area and critical riffle analysis (CRA) sites on the upper Eel River and the upper Russian River where ecological effects of Water Supply Scenarios were evaluated. Fish passage was evaluated throughout the Russian River, therefore no CRA site for the Russian River is depicted on this map.

Our approach used modeled daily HEC-ResSim streamflow data from the four Water Supply Scenarios spanning water years 1911 to 2017 and WUA curves developed for the Eel River (Figure 4-2 through Figure 4-4, VTN 1982) and the Russian River (Figure 4-5 and Figure 4-6, SCWA 2016). We computed daily WUA habitat values from daily streamflows for all Water Supply Scenarios and years, then compared the changes in WUA habitat between the Water Supply Scenarios.

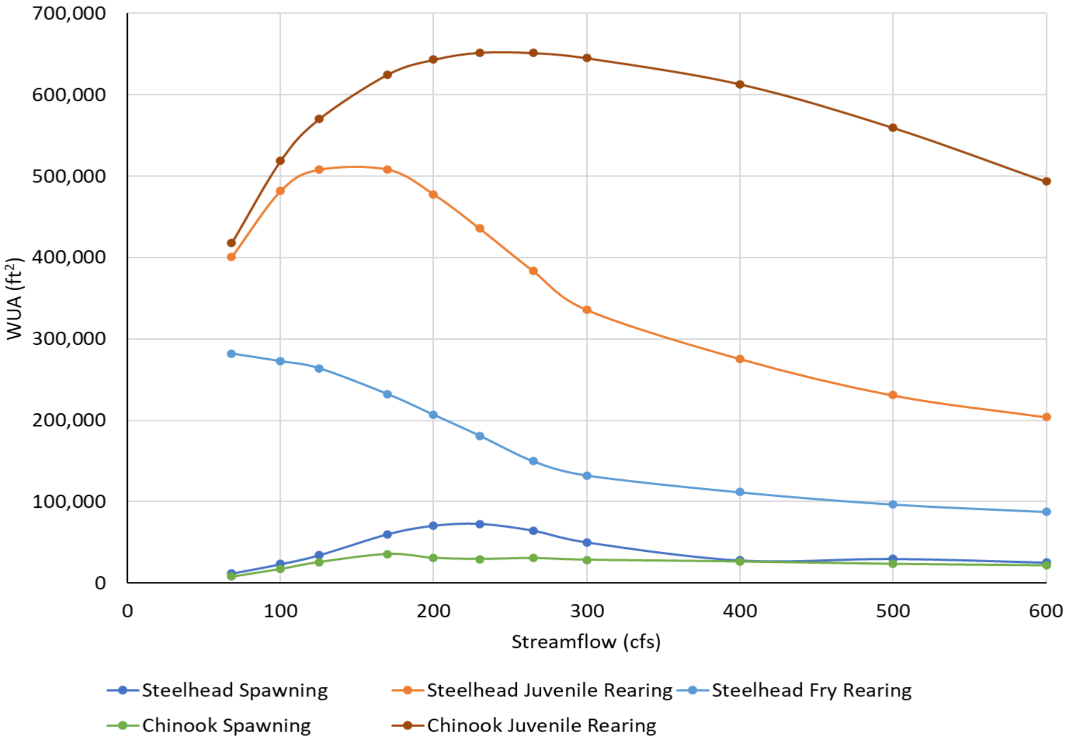

Figure 4-2. Weighted usable area (WUA) curves (feet squared [ft^2]) for the Trout Creek site on the Eel River. Curves were derived from WUA tables reported in Appendix G of VTN (1982).

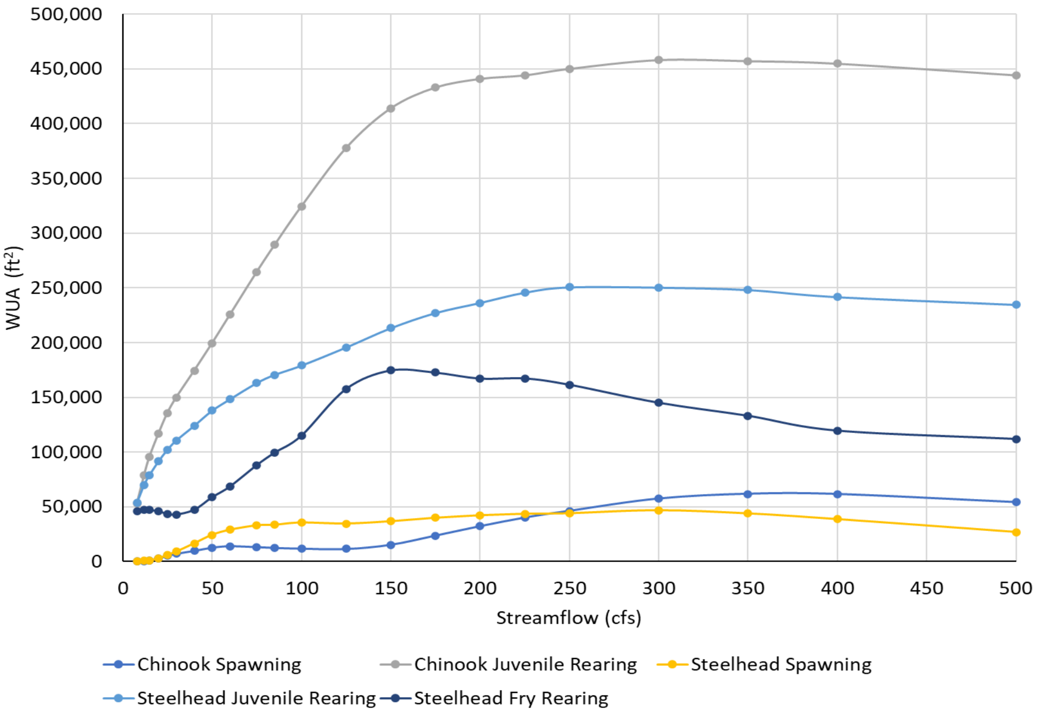

Figure 4-3. Weighted usable area (WUA) curves (feet squared [ft^2]) for the Cape Horn site on the Eel River. Curves were derived from WUA tables reported in Appendix G of VTN (1982).

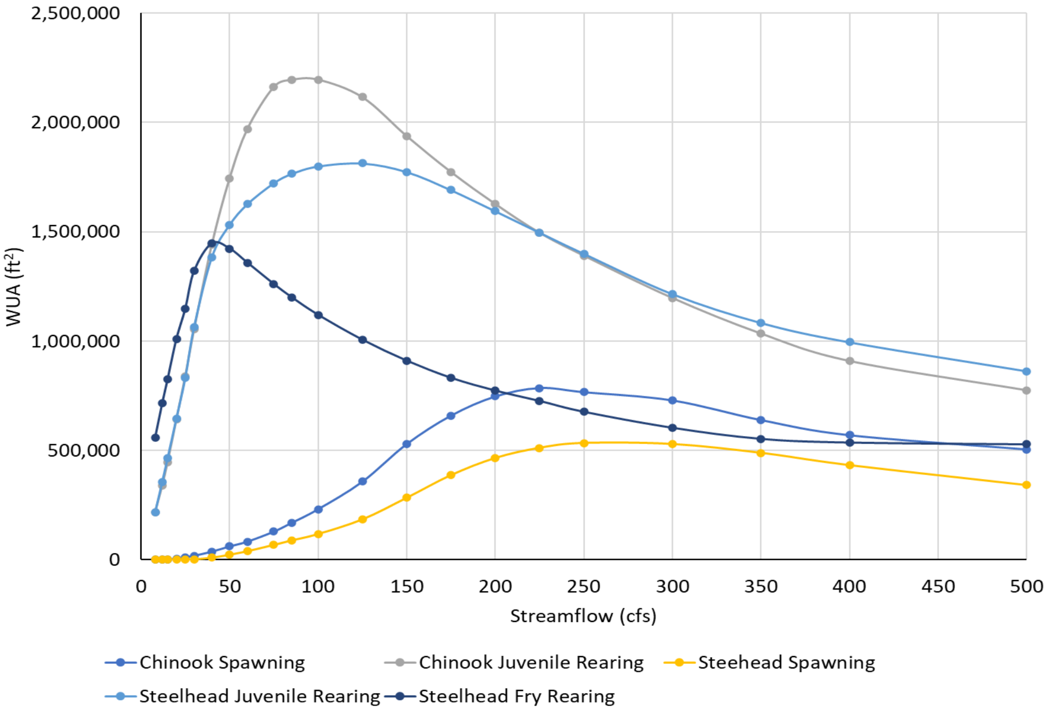

Figure 4-4. Weighted usable area (WUA) curves feet squared [ft^2]) for the Big Bend site on the Eel River. Curves were derived from WUA tables reported in Appendix G of VTN (1982).

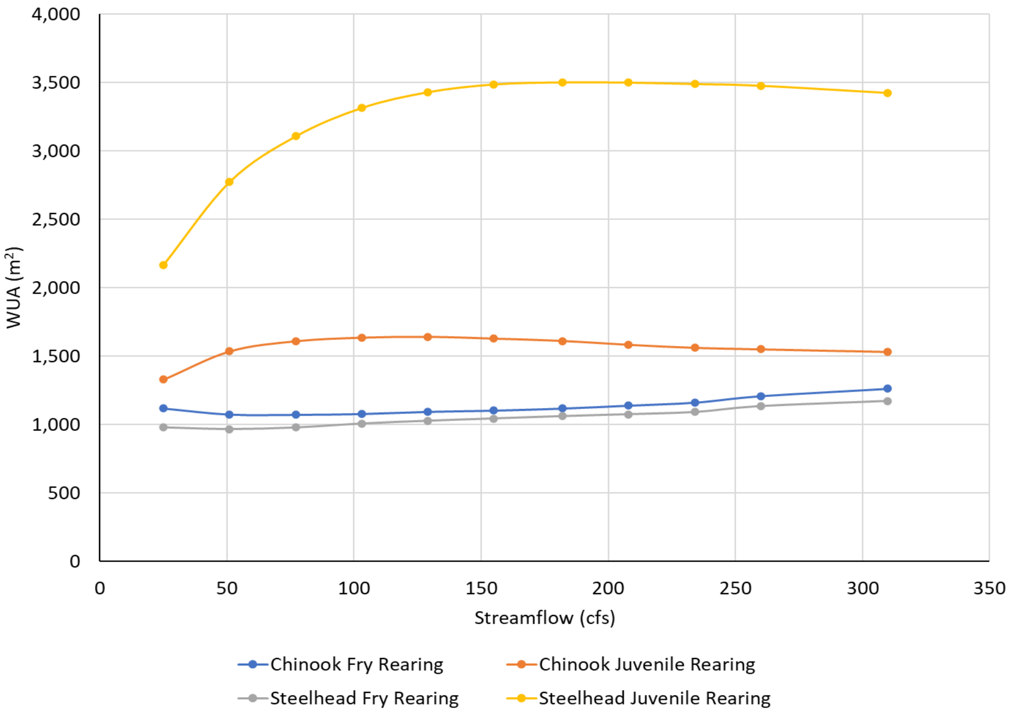

Figure 4-5. Weighted usable area (WUA) curves (m^2) for the Ukiah site on the Russian River. Curves were derived from WUA tables reported in the Fish Flow Draft Environmental Impact Report (SCWA 2016).

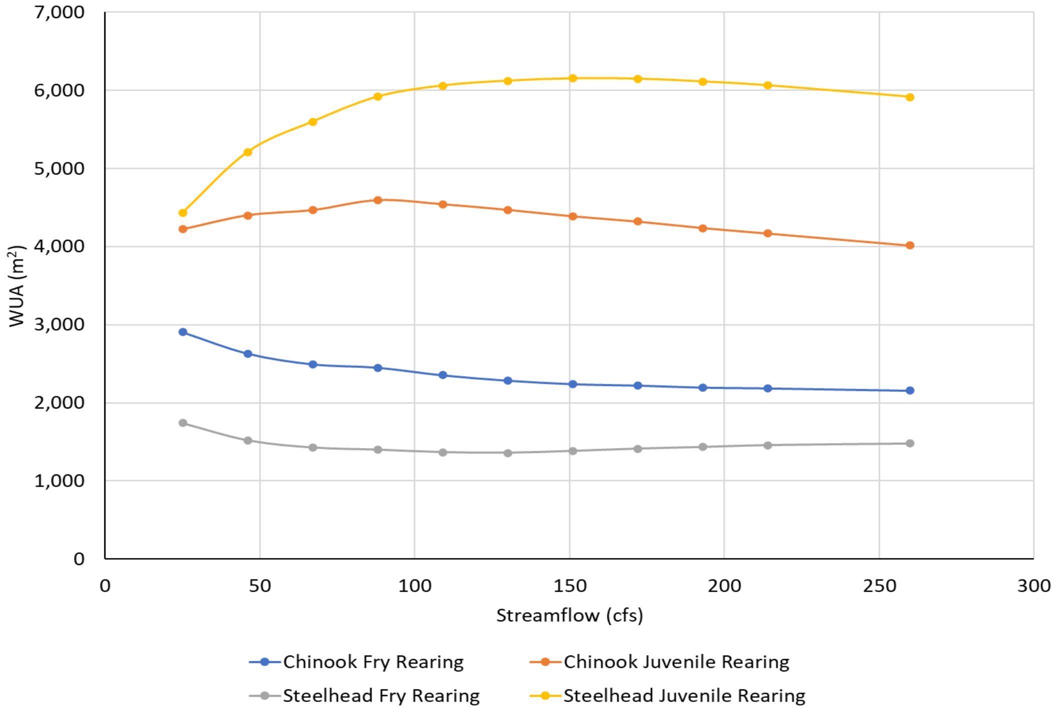

Figure 4-6. Weighted usable area (WUA) curves (m^2) for the Hopland site on the Russian River. Curves were derived from WUA tables reported in the Fish Flow Draft Environmental Impact Report (SCWA 2016).

We calculated the percent difference in available WUA habitat between four comparisons of Water Supply Scenarios: from Scenario 3 to Scenario 2, Scenario 3 to Scenario 4B, Baseline to Scenario 2, and Baseline to Scenario 4B (Table 4-2). We evaluated the Water Supply Scenarios in these combinations to (1) evaluate how habitat would change under the proposed Scenarios compared to full decommissioning (Scenario 3 to Scenario 2, Scenario 3 to Scenario 4B) and (2) evaluate how habitat would change under the proposed Scenarios compared to no change in hydrology (Baseline to Scenario 2, and Baseline to Scenario 4B). The streamflow data used for the analysis was extracted from the HEC-ResSim model at hydrological focal points closest to the sites (Figure 4-1). In the Eel River, we evaluated percent changes in habitat for Chinook Salmon spawning and juvenile rearing, and for steelhead spawning, juvenile rearing, and fry rearing. In the Russian River, we evaluated changes in habitat for Chinook Salmon juvenile and fry rearing and steelhead juvenile and fry rearing (Table 4-2). Steelhead and Chinook Salmon spawning were not analyzed in the Russian River because WUA curves were only available for juvenile and fry life stages (SCWA 2016).

We developed an automated R script (R Core Team 2015) to summarize and analyze the streamflow data. For the four Water Supply Scenarios, we calculated daily habitat time series by assigning a WUA value from the usable area-to-streamflow relationship curves for each day in the evaluated period (water years 1911–2017). From these data sets, we computed the percent change in WUA for the period in which the target species and life stages were present (Table 2-2 and Table 2-5), and then we binned and plotted the frequency of days which had a percent change. A negative percent change indicated a decrease in habitat, and a positive percent change indicated an increase. There were limitations on the days which could be analyzed because WUA curves did not extend to low enough or high enough streamflow, and without further information, we could not extend the curves. Thus, days where streamflows were not within the streamflow range on WUA curves (<68 cfs and >600 cfs at the Trout Creek site, <8 cfs and >500 cfs for the Cape Horn and Big Bend sites, <25 cfs and >310 cfs for the Ukiah site, and <25 cfs and >360 cfs for the Hopland site) were excluded from the analysis. Because of this WUA curve limitation, we chose to include months where more than 60 percent of days were within the streamflow range for the WUA curves. We selected this cutoff because we deemed that 60 percent or more of the data would show changes in habitat availability. Therefore, February and March for the Eel River and January, February, and March for all years (water years 1911–2017) for the Russian River were excluded from the analyses because more than 40 percent of the days in each month were outside the range of streamflows on WUA curves.

4.2.1 Eel River from Scott Dam to Cape Horn Dam (gage E-2)

While we present some examples and representative figures of the results of analysis here, figures of all the comparisons for this section can be found in Appendix A. There is no difference in water supply for Scenario 2 and Scenario 3 downstream of Scott Dam, and the Water Supply Scenarios are used interchangeably in this section (labeled as Scenario 2/3 from this point on in this section of the report). The differences between Scenario 2 and Scenario 3 are due to the Potter Valley diversion at Cape Horn Dam under Scenario 2 and the differences in their hydrographs become apparent downstream of Cape Horn Dam. See Section 3.1 for further details.

4.2.1.1 Spawning Habitat: Trout Creek Site

The greatest percent change in Spawning WUA between Water Supply Scenarios occurred when comparing Scenario 2/3 to Baseline and Scenario 2/3 to Scenario 4B. Chinook Salmon spawning (Oct 1–Dec 31) and steelhead spawning (Dec 1–Apr 30) saw frequent but small magnitude (+/- 10%) increases to habitat when Water Supply Scenarios changed from Baseline to Scenario 2/3

(Figure 4-7) and Scenario 2/3 to Scenario 4B (Figure 4-8), but higher magnitudes of decreased spawning habitat also occurred under Scenario 2/3 relative to Baseline. Overall, this appears as a decrease in spawning habitat under Scenario 2/3 relative to Scenario 4B. The increase in habitat for Chinook Salmon and steelhead spawning is explained by the lower daily streamflow under Baseline and Scenario 4B compared to Scenario 2/3 (Figure 3-2 and Figure 3-3) during their spawning periods because WUA curves for both species peak at lower streamflows (170 cfs for Chinook Salmon and 230 cfs for steelhead) and decrease as streamflows increase. Thus, higher streamflows typically associated with Scenario 2/3 likely drove decreases to spawning habitat. Baseline to Scenario 4B saw little percent change in spawning habitat because the two Water Supply Scenarios are generally similar downstream of Scott Dam throughout the year. Streamflows greater than 600 cfs were unable to be analyzed because they were not included in the WUA curves; it is unclear how spawning habitat availability would change at higher streamflows where WUA curves did provide information.

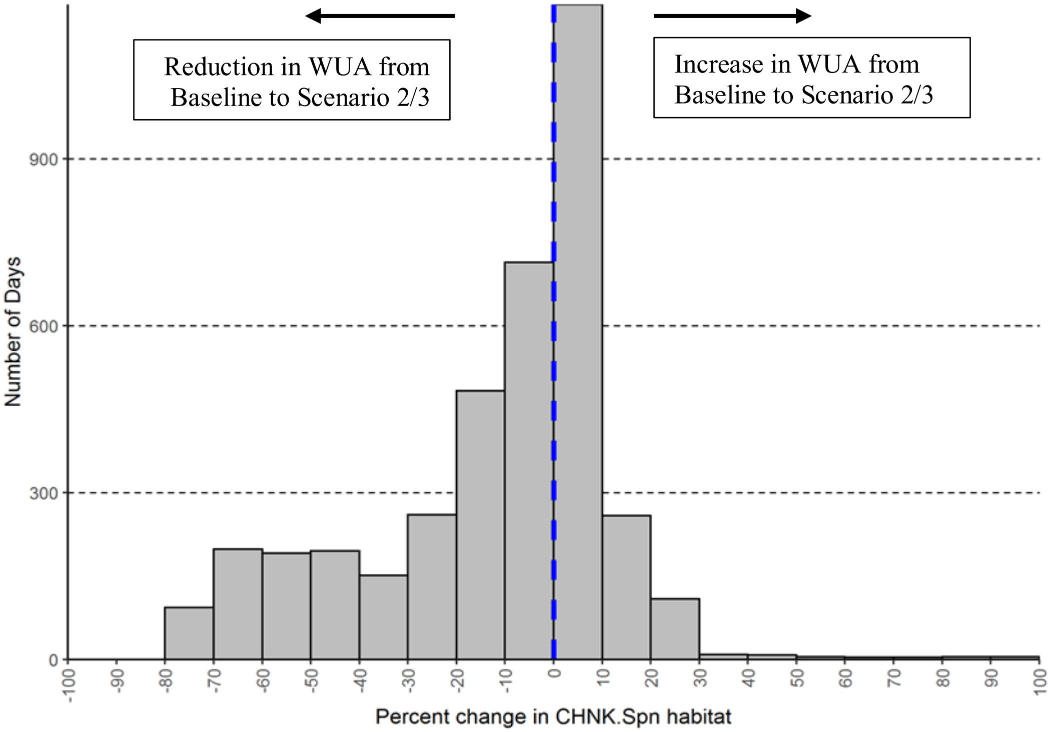

Figure 4-7. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2/3 for Chinook Salmon spawning habitat at the Trout Creek site.

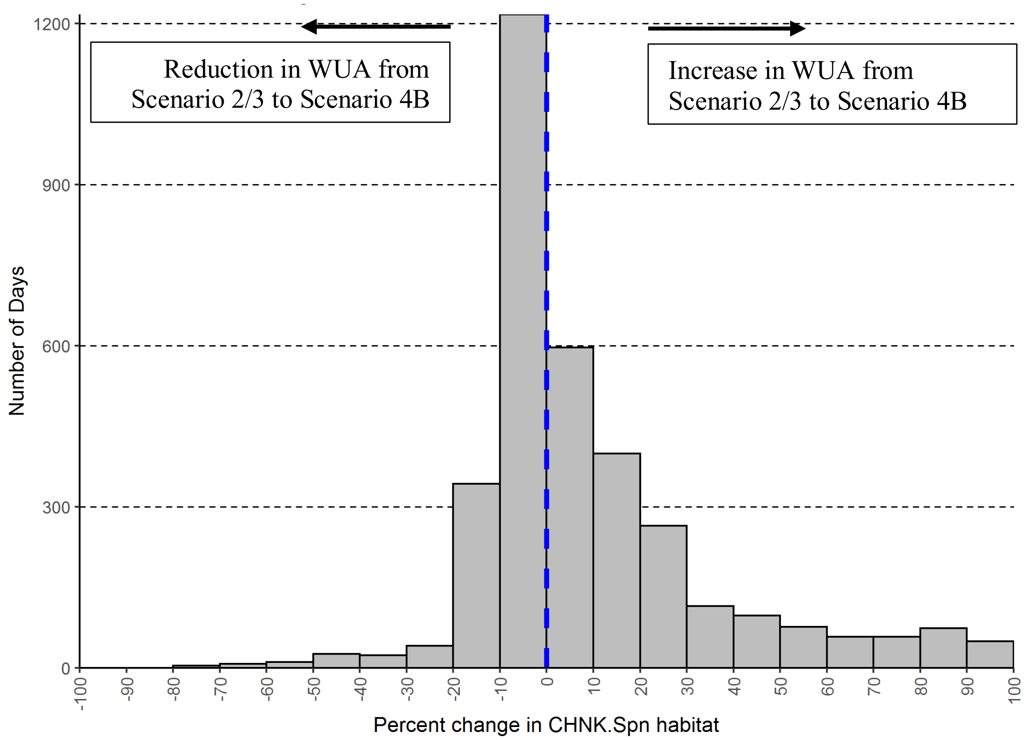

Figure 4-8. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 2/3 to Scenario 4B for Chinook Salmon spawning habitat at the Trout Creek site.

4.2.1.2 Juvenile and Fry Habitat: Trout Creek Site

Chinook Salmon juvenile rearing habitat (Jan 1–May 31) saw very little change under all Water Supply Scenario comparisons because the relationship between streamflow and WUA was generally flat across the range of streamflows (Figure 4-2).

Steelhead juvenile (Jan 1–Dec 31) and fry rearing (Mar 1–Jul 31) habitat typically increased in Scenario 2/3 relative to Baseline, but there were also days which decreased (Figure 4-9). The increases in habitat for steelhead juvenile and fry rearing is explained by comparing WUA curves to the daily average streamflow for each Water Supply Scenario. During the fry rearing period and much of the juvenile rearing period, daily streamflow under Scenario 2/3 is lower than Baseline and Scenario 4B, thus habitat is increased for most of the year under Scenario 2/3. The transition to when Scenario 2/3 habitat becomes less than Baseline and Scenario 4B occurs around mid-April and continues through the end of October (Figure 3-3 though Figure 3-5). WUA curves for both life stages peak at lower streamflows (170 cfs for juveniles and 68 cfs for fry) and decrease as streamflows increase. Importantly, low streamflows (<68 cfs) were not evaluated in the analysis because streamflows below 68 cfs were not included on WUA curves for the Trout Creek site. Given the inability to evaluate data below 68 cfs, there is uncertainty in the results for the low streamflow range.

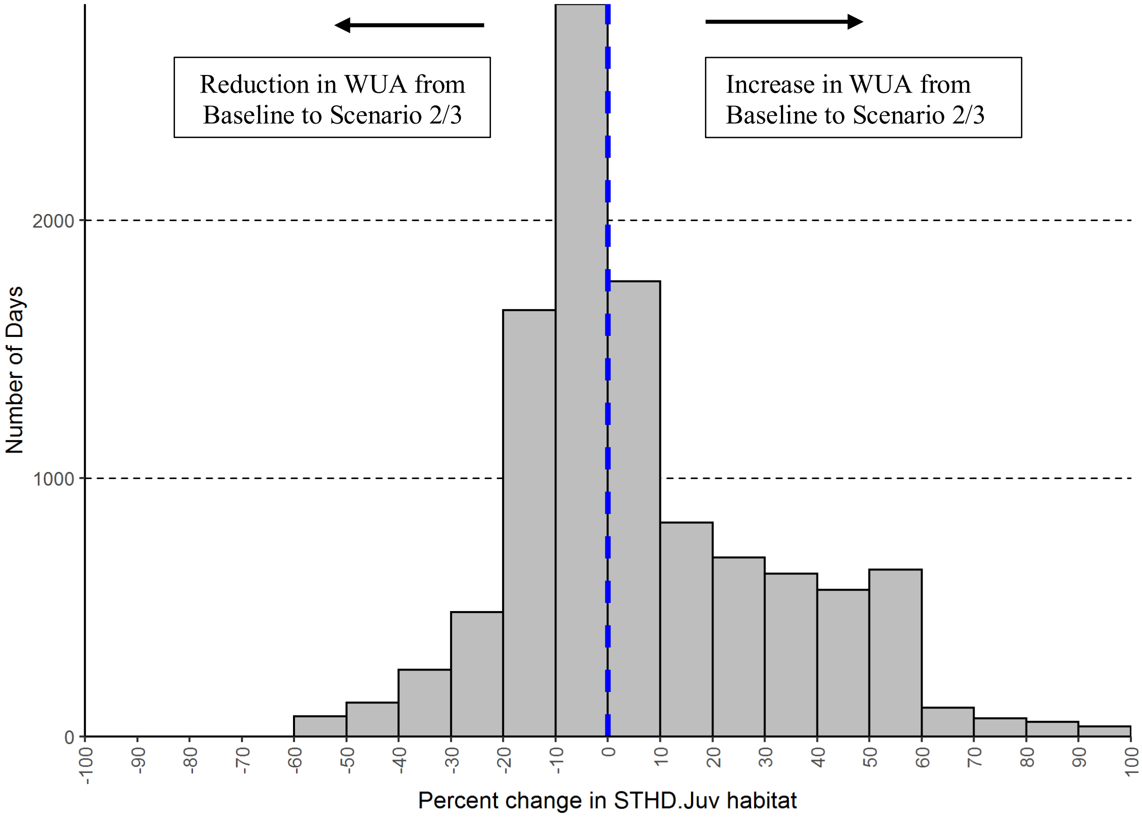

Figure 4-9. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2/3 for steelhead juvenile habitat at the Trout Creek site.

4.2.2 Eel River Downstream of Cape Horn Dam (gage E-11)

At the Cape Horn Site, very little change in habitat occurred for all species and life stages when comparing Baseline to Scenario 4B because their hydrology is similar. The only noticeable change was minimally decreased streamflow under Scenario 4B between December 1 through mid-May when the minimum streamflow is reduced to 45 cfs from 100 cfs under Baseline (Figure 3-11 and Figure 3-12). Because of the similarities between Scenario 4B and Baseline, we focused this section on comparisons between Baseline, Scenario 3, and Scenario 2, and presented notable examples of changes in habitat. All figures for this section can be found in Appendix A.

4.2.2.1 Spawning Habitat: Cape Horn Dam Site

Baseline to Scenario 2 had a higher frequency of days that decreased in Chinook Salmon spawning habitat relative to Baseline (Figure 4-10) due to hydrology under Baseline being lower than Scenario 2 during the Chinook Salmon spawning period (Oct 1–Dec 31) for both wetter and drier water year types (Figure 3-14). WUA for Chinook Salmon spawning was the greatest at higher streamflows and therefore WUA under Scenario 2 was greater than Baseline (Figure 4-3). WUA for steelhead spawning was similar for all streamflows and therefore there was little change in habitat for all Water Supply Scenarios compared (Figure 4-3).

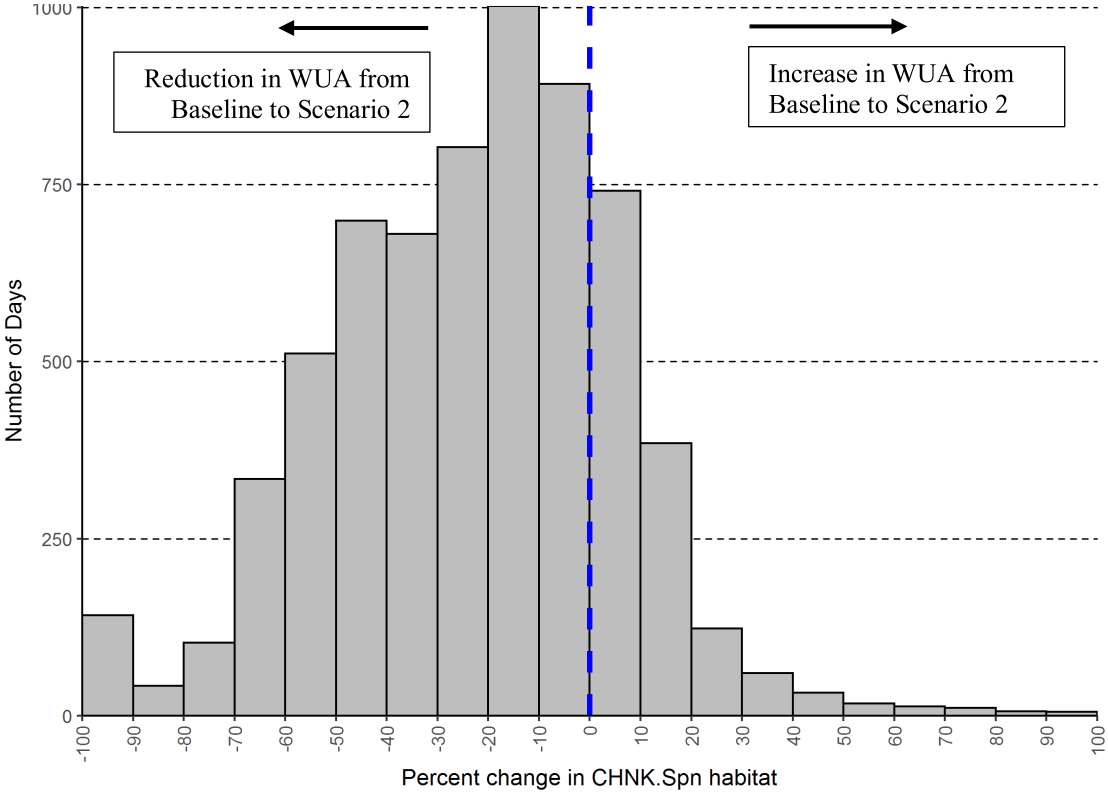

Figure 4-10. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for Chinook Salmon spawning habitat at the Cape Horn site.

4.2.2.2 Juvenile and Fry Habitat: Cape Horn Dam Site

There were both increases and decreases in steelhead fry rearing habitat relative to Scenario 2, Scenario 4B, and Baseline for fry and juvenile steelhead (Figure 4-11). This was driven by average streamflow that was greater than 100 cfs (representing small decreases in habitat) during the first half of the rearing period that was included in the analysis (April–May, Figure 3-12), and less than 100 cfs (representing large changes to habitat) during the second half (June through July, Figure 3-12 and Figure 3-13). Thus, the magnitude of changes to fry WUA occurring in June and July were greater than those in April–May. The steelhead fry rearing WUA curve increases until 150 cfs, then begins to decrease at higher streamflows. Under Scenario 3, the recession limb is more gradual and retains higher (>150 cfs) streamflow in the river for a longer period, specifically during wetter years, representing decreases in WUA. Alternatively, increases appear to be driven by higher streamflows closer to the 150 cfs WUA peak for steelhead fry during the spring recession in drier years under Scenario 3 relative to other Scenarios.

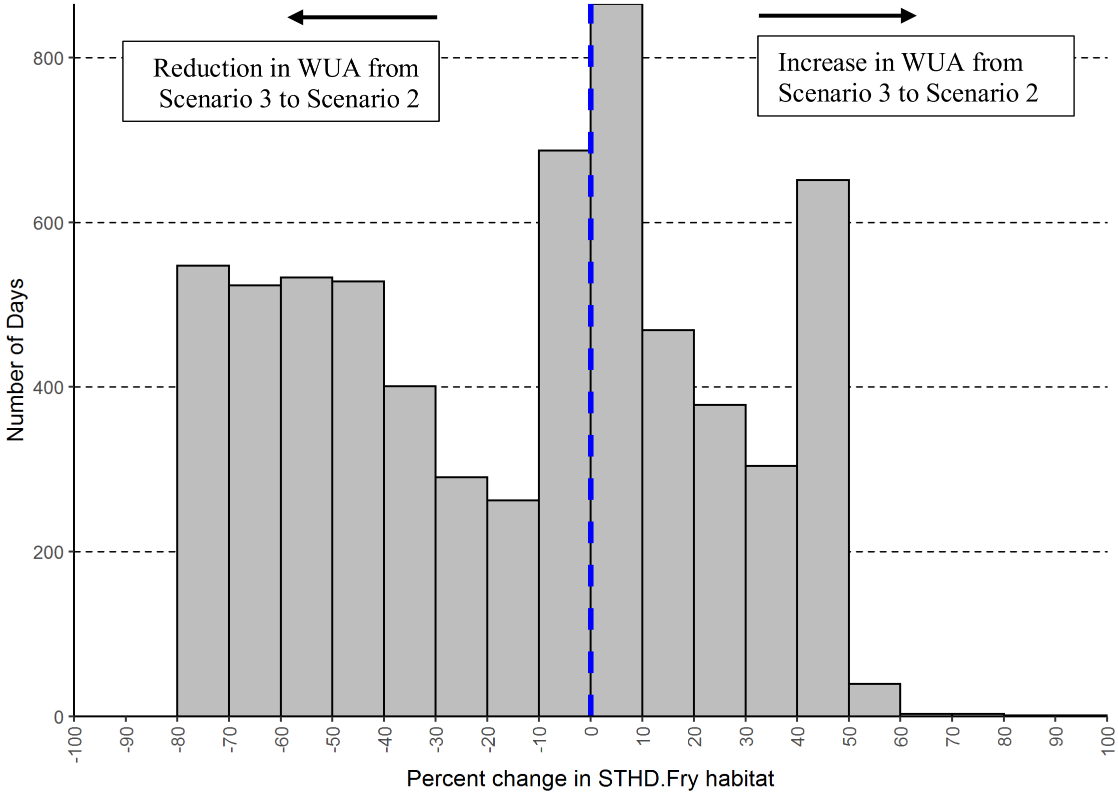

Figure 4-11. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for steelhead fry habitat at the Cape Horn Dam site.

Neither juvenile Chinook Salmon or juvenile steelhead had large or clear changes between the Water Supply Scenarios. For example, Chinook Salmon juvenile rearing habitat saw very little change under all Water Supply Scenario comparisons because their WUA curve was generally flat at streamflow greater than 125 cfs (Figure 4-3), and daily average streamflow under all Water Supply Scenarios during rearing (January 1–May 31; no days between February 1–March 31 were included in analysis) was greater than 100 cfs (Figure 3-11 and Figure 3-13).

4.2.2.3 Spawning Habitat: Big Bend (Outlet Creek) Site

Results show that decreases in spawning habitat for both steelhead and Chinook Salmon can be expected under Scenario 2, Scenario 3, and Scenario 4B relative to Baseline (ex. Figure 4-12), which are driven by lower streamflows in each Water Supply Scenario relative to Baseline during the fall spawning season (Figure 3-12). These differences indicate that some of the streamflow habitat relationships are affected by increases in streamflow from tributary inputs between Cape Horn Dam and Outlet Creek (primarily Tomki Creek). Streamflow under Scenario 2 is typically lower during Chinook Salmon spawning season, which is likely driving the reductions in WUA habitat (Figure 4-12, Table 2-2) Additionally, the range of streamflow within the WUA curve was limited at this site (<8 cfs and >500 cfs) and only a few days were able to be analyzed.

There were no clear direction changes in steelhead spawning habitat across the Water Supply Scenario comparisons.

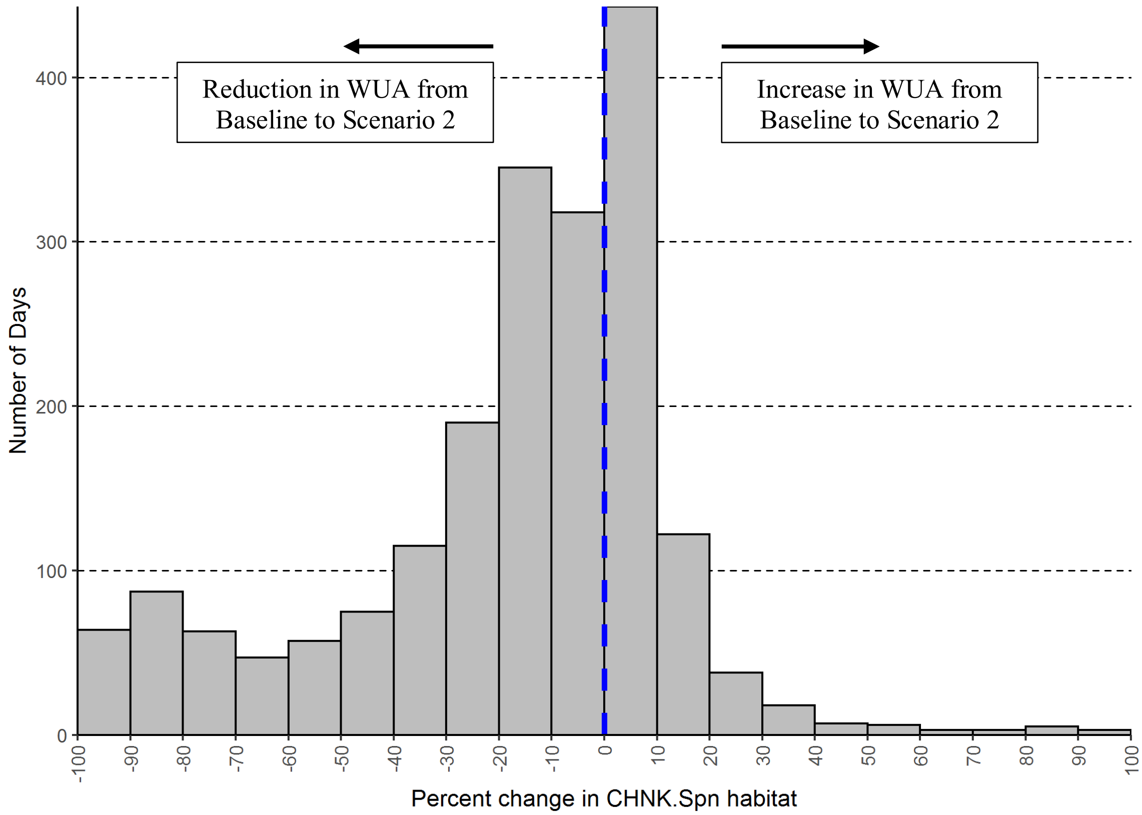

Figure 4-12. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for Chinook Salmon spawning habitat at the Big Bend site.

4.2.2.4 Fry and Juvenile Habitat: Big Bend (Outlet Creek) Site

Very little change in habitat occurred under both comparisons for juvenile and fry life stages. The exception was for Chinook Salmon juvenile rearing and steelhead fry rearing, where Scenario 2 typically resulted in higher habitat values when compared to Scenario 3 (e.g., Figure 4-13). Both Chinook Salmon juvenile rearing and steelhead fry rearing occur during the spring recession limb (Figure 3-12) and WUA curves for both at the Big Bend site peak at low streamflow and gradually decrease as streamflows increase (Figure 4-4). Under Scenario 2, lower streamflows in the spring likely drive the habitat changes.

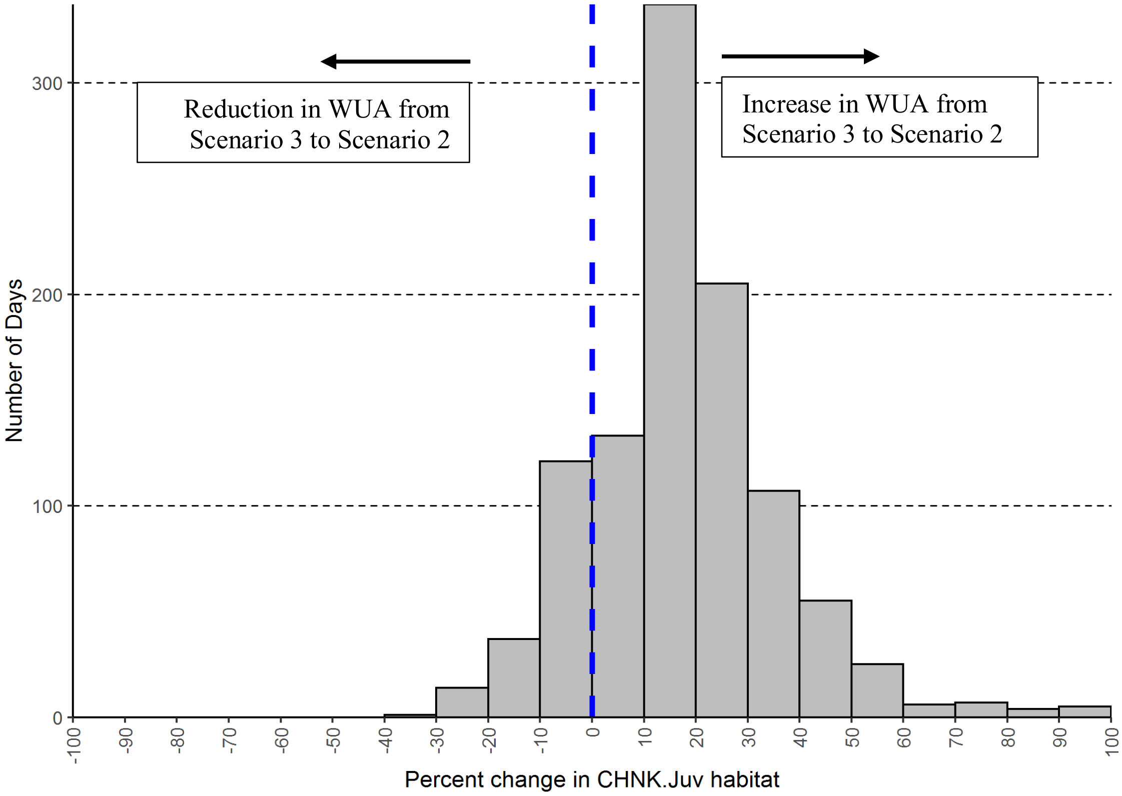

Figure 4-13. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for Chinook Salmon juvenile habitat at the Big Bend site.

4.2.3 Russian River Downstream of the West Branch Confluence

Minimal differences in WUA were computed on the Russian River. Thus, all species and life histories are discussed under only one heading for each site.

4.2.3.1 Ukiah Site

There was very little change in habitat for all evaluated species and life stages under all Water Supply Scenario comparisons. This was mainly due to the shape of the WUA curves for all species and life stages at the Ukiah site and not because there were not differences in the hydrographs downstream of the WBRR confluence. WUA for Chinook Salmon juveniles and fry rearing (Mar 1–May 31), and steelhead fry rearing (Jan 1–Dec 31) were relatively flat, with very little difference between the highest and lowest WUA values (difference between lowest and highest streamflows were 190 m^2 for Chinook Salmon fry, 206 m^2 for steelhead fry, and 311 m^2 for Chinook Salmon juveniles, Figure 4-5). Steelhead juvenile rearing (Jan 1–Dec 31) had the greatest difference between the highest and lowest WUA values (1,335 m^2), and correspondingly, the percent change in WUA charts for steelhead rearing had more change in habitat relative to the other species and life stages under Baseline to Scenario 2, Scenario 3 to Scenario 2, and Scenario 3 to Scenario 4B (Figure 4-14 to Figure 4-16).

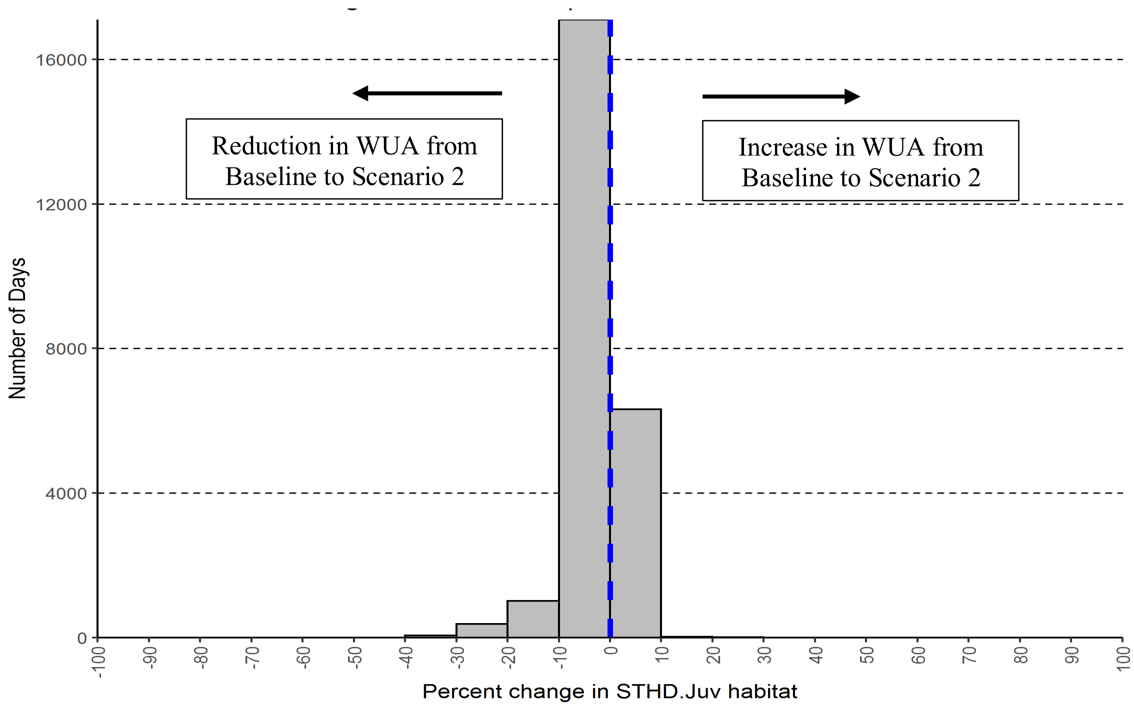

Figure 4-14. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead juvenile habitat at the Ukiah site.

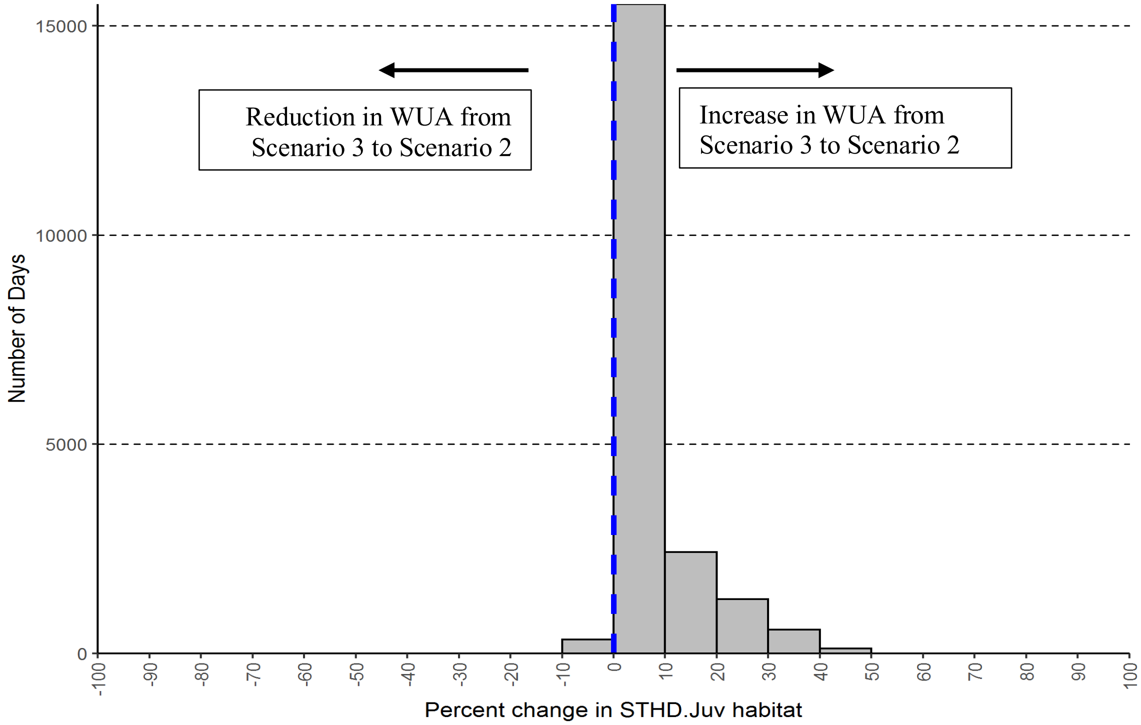

Figure 4-15. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for steelhead juvenile habitat at the Ukiah site.

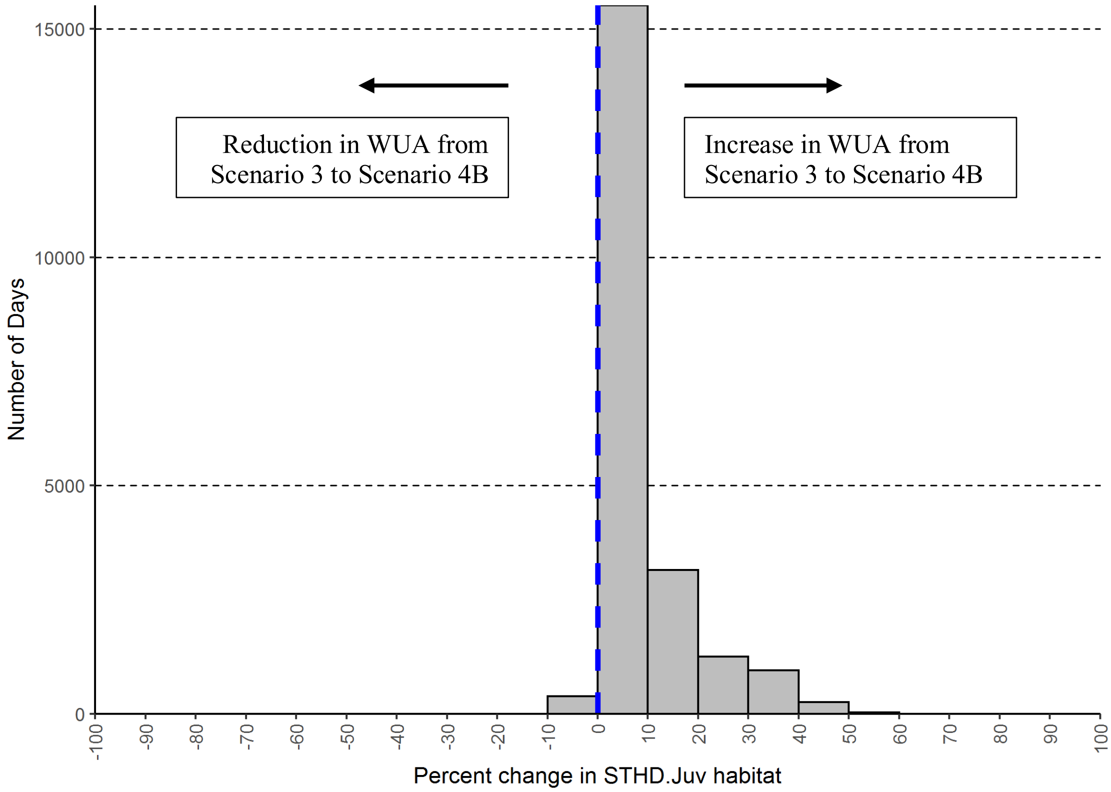

Figure 4-16. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead juvenile habitat at the Ukiah site.

Of all the Water Supply Scenarios, Scenario 3 saw the greatest reduction in streamflow and WUA relative to Scenario 4B or Scenario 2. However, the changes in WUA are low at all streamflows evaluated and fish habitat should not be significantly affected by changes in streamflow. The exception would be for steelhead juvenile rearing when streamflows fall below 100 cfs. The greatest reduction in steelhead juvenile rearing habitat would most likely occur under Scenario 3 for the drier water year type in the fall and during the late spring to early summer when streamflow generally decreases noticeably below 100 cfs (Figure 3-22).

4.2.3.2 Hopland Site

Similar to the Ukiah site, there was very little change in habitat for all evaluated species and life stages under all Water Supply Scenario comparisons. Steelhead juvenile rearing had less change in habitat at the Hopland site than the Ukiah site under Baseline to Scenario 2, Scenario 3 to Scenario 2, and Scenario 3 to Scenario 4B (Figure 4-17). This would suggest that tributary inputs between the Ukiah and Hopland site reduced the effects of the differences on streamflows between the three comparisons on steelhead rearing habitat as you move downstream of Coyote Valley Dam.

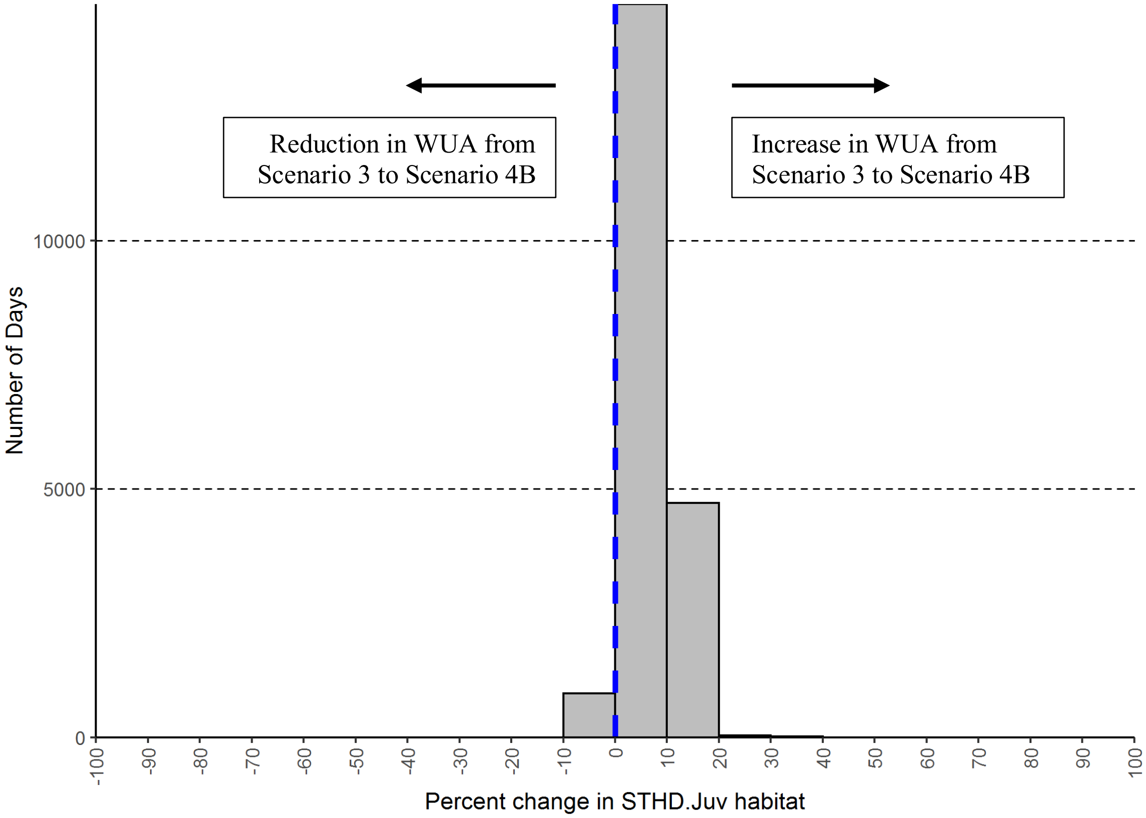

Figure 4-17. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead juvenile habitat at the Hopland site.

4.2.4 Summary

For the Eel River downstream of Scott Dam, WUA analysis suggests that under Scenario 3 and Scenario 2 there is likely to be a decrease in spawning habitat for Chinook Salmon and no clear change to steelhead spawning habitat. The decrease in Chinook Salmon spawning habitat is likely driven by lower (unimpaired) fall streamflows associated with Scenario 3 and Scenario 2 when Chinook are spawning. For juvenile Chinook Salmon and juvenile/fry steelhead rearing results were mixed. With some comparisons resulting in more habitat under Scenario 3 and Scenario 2 relative to Baseline and Scenario 4B, with changes varying among water year types.

Downstream of Cape Horn Dam, spawning habitat typically remained unchanged across Water Supply Scenarios, while there were both increases and decreases for juvenile and fry life history stages for Chinook Salmon and steelhead. There were differences between WUA analysis results between Cape Horn and Big Bend sites, suggesting that tributary accretion that occurs between Cape Horn and Big Bend from Tomki Creek can be influential for habitat estimations.

The Russian River saw small changes for nearly all comparisons, with the distribution of days which represent a percent increase or decrease for all life histories centered around 0. Juvenile rearing habitat for steelhead is predicted to be reduced under Scenario 3 and Scenario 2 due to reduced water deliveries from the Project relative to Baseline and Scenario 4B. However, even those changes were typically less than 10 percent.

4.3 Chinook Salmon Migration

We evaluated hydrological effects on fish passage in the upper Eel River and the upper Russian River to assess how the four Water Supply Scenarios would affect upstream passage of returning adult Chinook Salmon. We focused this analysis on streamflow-related effects and not infrastructure-related changes. We also evaluated the effect of Water Supply Scenarios on streamflow magnitude downstream of Cape Horn Dam related to upstream migration of adult Chinook Salmon holding in the lower Eel River.

4.3.1 Critical Riffle Analysis

The critical riffle analysis (CRA) focused on Chinook Salmon because they are typically moving upstream during fall when streamflows are low, and in the Eel River could be affected by attenuation of fall freshet streamflows by Scott Dam and Lake Pillsbury and diversions at Cape Horn Dam. Steelhead migration occurs in the winter and spring when streamflow is higher, and passage is not anticipated to be limiting. We used modeled streamflow data from the four Water Supply Scenarios spanning water years 1911 to 2017 and Chinook Salmon passage thresholds developed for the Eel River and the Russian River. For the Eel River, we used the passage threshold of 124 cfs from the CRA, which was identified as the recommended minimum streamflow for providing passage at the most limiting critical riffle site (Above Garcia Riffle, Figure 4-1 and Figure 4-18) based on passage criteria reported (Thompson 1972, VTN 1982, SEC 1998). For the upper Russian River, we used the passage streamflow recommendation of 105 cfs, developed by measuring maximum riffle crest depths in the upper Russian River and observing when adult Chinook Salmon reached spawning grounds (SCWA 2016, J. Smith, Sonoma Water Fish Biologist, pers. comm.). During the migration period for all water years evaluated, we assigned a binary outcome of passage to the daily average streamflow record: 1 if streamflow was greater than or equal to the passage threshold, or 0 if daily average streamflow was less. We calculated the average number of passable days per year for each Water Supply Scenario and plotted the results in bar charts for all years and by water year type. We used modeled HEC-ResSim streamflow data for each Water Supply Scenario from nodes near the area where passage was evaluated. For the Eel River, streamflow downstream of Cape Horn Dam (gage E-11, Figure 3-1) was used, while for the Russian River streamflow near Healdsburg (USGS 11464000) was used.

Figure 4-18. Photo taken in 2018 of a riffle near the “Above Garcia Riffle” listed in VTN (1982). At the time the photo was taken, the riffle used in the critical riffle analysis in 1982 was not in the same location. This riffle is one of two riffles closest to the original location.

4.3.1.1 Eel River Downstream of Cape Horn Dam

For all water years combined, the average number of days when upstream Chinook Salmon passage was possible in the upper Eel River ranged from a low of 31 days for Scenario 2 to a high of 38 days for Baseline for the 92 days possible during the October 1 to December 31 migration window. Across all Water Supply Scenarios, very dry water year types had the least average number of passable days (except for Baseline, which had 26 passable days), and very wet water year types had the most average number of passable days (Figure 4-19). Of the Water Supply Scenarios, the average number of days when upstream adult Chinook Salmon passage was possible was greatest for Baseline apart from very wet water year types, where Scenario 3 had one day more (49 days) than Baseline for the average number of days passable. During very dry water years, Baseline had the greatest number of passable days, with 8 to 11 more passable days compared with the other Water Supply Scenarios. This was driven by the increased daily average streamflow during the fall upstream migration period compared with other Water Supply Scenarios, which have lower late summer streamflows. Scenario 2 had the least average number of passable days for all water year types due to the increased Project diversion (140 cfs–300 cfs maximum diversion).

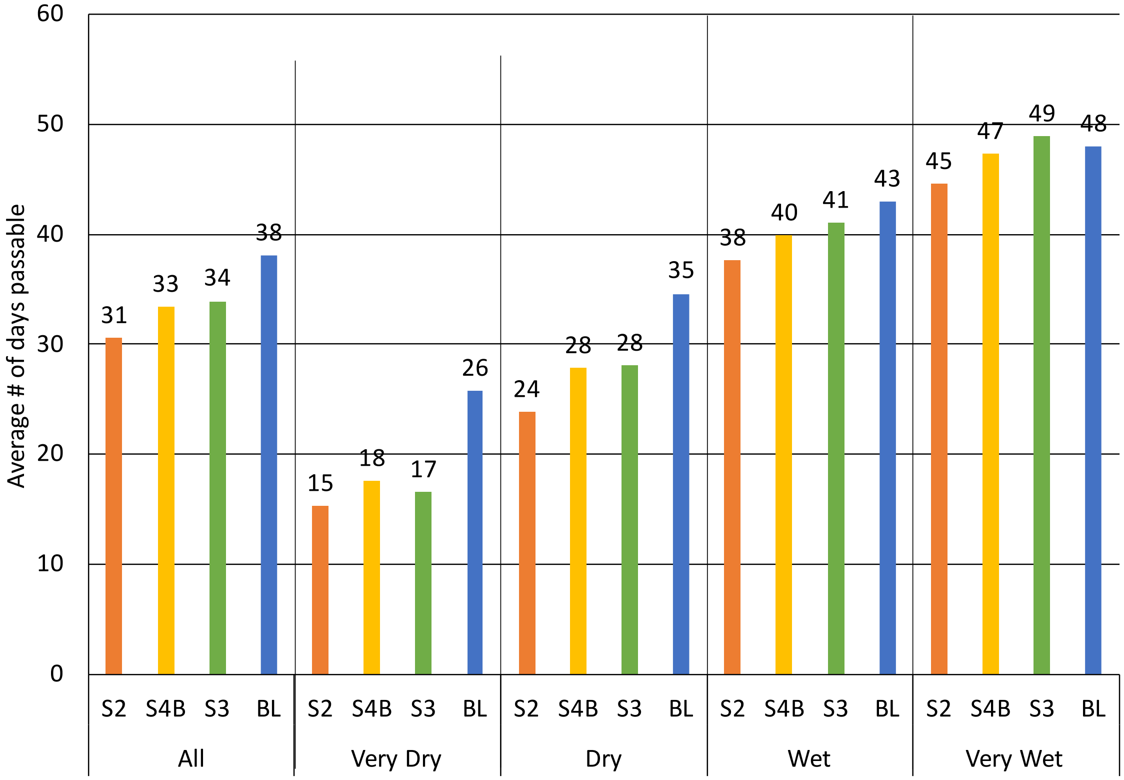

Figure 4-19. Average number of days where streamflow was greater than the passage threshold (124 cfs) in the Eel River downstream of Cape Horn Dam during fall-run Chinook Salmon upstream migration for the four Scenarios (S2 = Scenario 2, S4B = Scenario 4B, S3 = Scenario 3, BL = Baseline). Fall-run Chinook Salmon migration window for the Eel River is Oct 1-Dec 31, or 92 days total.

4.3.1.2 Russian River Downstream of the West Branch Confluence

For all water years combined, the average number of days where upstream adult Chinook Salmon passage was possible in the Upper Russian River ranged from 76 days for Scenario 3 to 137 days for Baseline for the 153 days possible during the September 1 to January 31 migration window. The average number of days passable for all Water Supply Scenarios increased from critical to dry to normal water year types (Figure 4-20). Scenario 3 had the least number of passable days compared to all Water Supply Scenarios for all the water year types, while Baseline had the most. During critical water year types, Baseline increased the average number of passable days from 27 days to 72 days. Scenario 3 would have the greatest effect on fall-run Chinook Salmon spawning in the upper Russian River regardless of water year type.

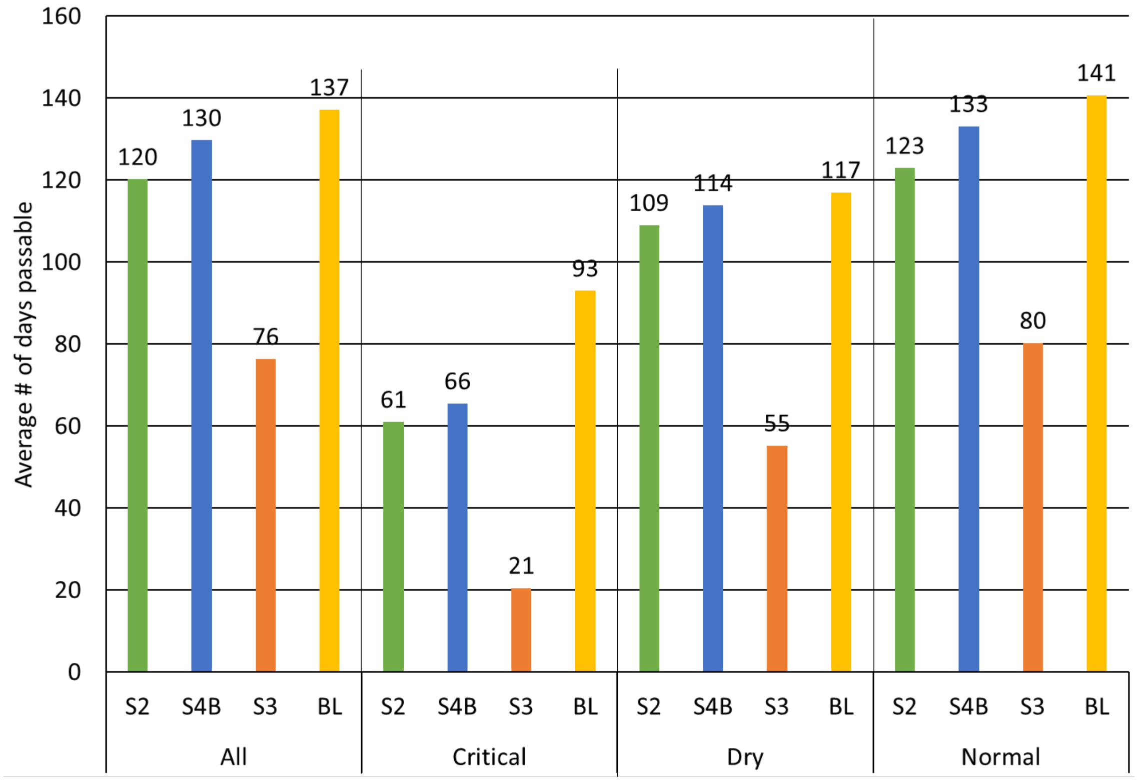

Figure 4-20. Average number of days where streamflow was greater than the passage threshold in the upper Russian River (105 cfs) during fall-run Chinook Salmon upstream migration for the four Scenarios (S2 = Scenario 2, S4B = Scenario 4B, S3 = Scenario 3, BL = Baseline). Fall-run Chinook Salmon migration window for the Russian River is Sept 1-Jan 31 and 153 days total per year.

4.3.2 Effect of Fall Streamflow Magnitude to Fish Migration in the Lower Eel River

Chinook Salmon typically migrate into the Eel River from the ocean in the fall. During low streamflows, they hold in the lower Eel River near Fortuna. Once fall storms provide higher streamflows (freshets), their upstream migration is triggered and they move upstream to spawning habitat. Because the Project can reduce the magnitude of fall freshets (Figure 4-21), it is possible that the magnitude of streamflows required to trigger Chinook Salmon upstream migrations could be affected by the Water Supply Scenarios. Typically, adult Chinook Salmon arrive at the Van Arsdale Fish Station when streamflow at the Scotia gage (USGS Gage 11477000) exceeds approximately 200 cfs five days prior to arrival (S. Harris, retired CDFW Fish Biologist, pers. comm.). Therefore, we expect a streamflow magnitude of 200 cfs at the Scotia gage to trigger Chinook Salmon holding in the the lower river to migrate upstream.

For each Water Supply Scenario, we estimated the potential change in timing and frequency of flows exceeding 200 cfs at Scotia. We calculated the streamflow at Scotia for each Water Supply Scenario from the HEC-ResSim model by adding by the corresponding accretion from Cape Horn Dam (gage E-11) to Scotia. Accretion was estimated by subtracting the measured streamflow at Scotia from measured streamflow at Cape Horn Dam (gage E-11). The hydographs between Cape

Horn Dam and Scotia were compared to determine if the reduced streamflows caused by the Water Supply Scenarios would lessen streamflows to a point where Chinook Salmon upstream migration would decrease. Specifically, we examined peak flows at Scotia relative to the streamflow magnitude change caused by the Water Supply Scenarios to search for instances where streamflow would be reduced to less than 200 cfs. This analysis was run for two example water years, a wetter year (2011) and a drier year (2017).

The reduction of streamflows from the Water Supply Scenarios caused virtually no reduction to streamflows at Scotia (Figure 4-22), thus none of the Water Supply Scenarios would likely delay the upstream migration cue for Chinook Salmon. Streamflows were reduced by the Project by approximately 300 cfs and 200 cfs for Scenario 2 and Scenario 4B, respectively, during the drier years (Figure 4-22). During wetter years, streamflows were reduced by 300 cfs and 750 cfs for Scenario 2 and Scenario 4B, respectively (Figure 4-22). However, these streamflows represented only 3 percent and 6 percent of the total streamflow at Scotia, respectively. Based on this analysis, we conclude that the reduction of streamflow caused by the Project is minimal relative to large tributary and Eel River watershed accretion that takes place between the Project and the lower Eel. It is possible Baseline and Scenario 4B, which increase streamflows to 100 cfs on December 1, could improve Chinook Salmon migration in years when freshet storms are extremely late in the year.

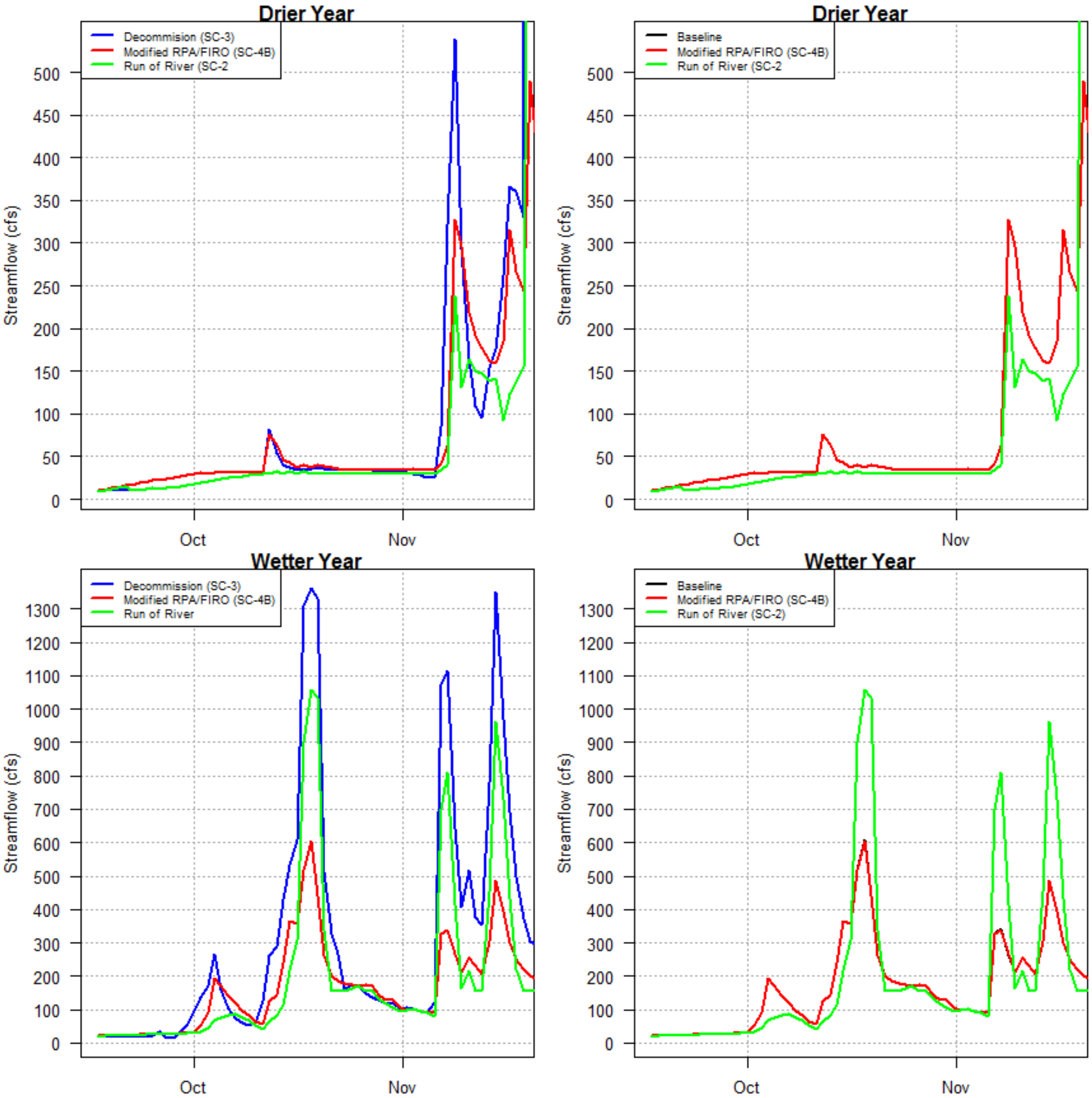

Figure 4-21. Fall freshet streamflow for each Water Supply Scenario during wetter (2017) and drier (2015) years downstream of Cape Horn Dam. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right. Note that Baseline (black line) is similar to Scenario 4B (red line) and black line is obscured by red line.

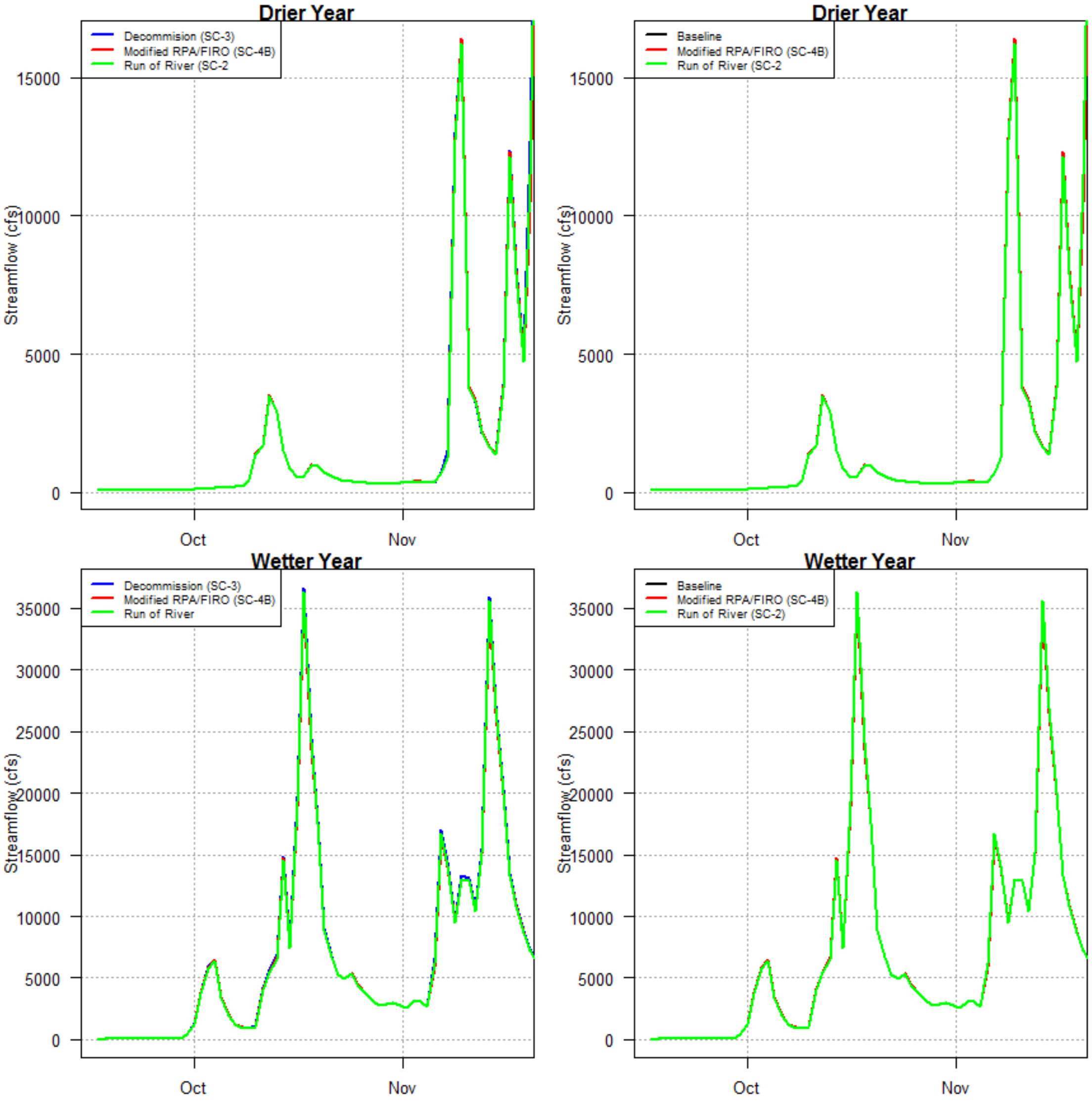

Figure 4-22. Fall freshet streamflow from each Water Supply Scenario during a wetter (2017) and drier (2015) years at Scotia. Scenario 2 (green), Scenario 3 (blue) and Scenario 4B (red) are presented on the left, and Baseline (black), Scenario 2 (green), and Scenario 4B (red) are presented on the right. Note that all scenarios are similar such that only the green line (Scenario 2) appears visible.

4.3.3 Summary

Changes in streamflow magnitude can affect the number of days that adult salmonids have suitable conditions for successful passage. Lower streamflows in the fall that are typically associated with Scenario 3 may result in fewer passable days in the fall at critical riffles in both the Eel River and Russian River. The analysis of streamflow at Scotia suggests that Project operations have little to no effect on the magnitude of streamflows in the lower Eel River watershed during fall and are not expected to influence migration. However, retention of the dams truncates some of the fall freshets and could delay fish migration, specifically during early fall flows.

4.4 Pacific Lamprey Responses

The Pacific Lamprey is one of the most widely distributed species along the Pacific Rim (Reid and Goodman 2015). In the Eel River watershed, the species is both a key component of the aquatic ecosystem and a culturally important food source for Native American tribes (WNRD and Stillwater Sciences 2016). Historically, the species was so abundant in the Eel River watershed that it was named "Eel River" by European settlers, who mistook lampreys for eels (W.W. Elliott & Co. 1881). However, the Pacific Lamprey population has declined substantially in response to widespread habitat degradation, barriers to fish passage, invasive predators, and water diversions (Stillwater Sciences 2010, Goodman and Reid 2012). Due to its cultural and ecological importance and in response to its rapid decline, the Pacific Lamprey has recently gained more recognition as a priority species for conservation and restoration, both in the Eel River and across its native range (Goodman and Reid 2012, WNRD and Stillwater Sciences 2016).

Pacific Lamprey are anadromous and migrate from the ocean into the upper mainstem Eel River and its tributaries to spawn. During this migration, which occurs primarily in the winter and spring, Pacific Lamprey can be affected by altered streamflow regimes from dams and diversions, as well as barriers to fish passage (Goodman and Reid 2012, Stillwater Sciences 2014, Goodman et al. 2015). Larval (ammocoete) Pacific Lamprey spend approximately five years in fresh water before out-migrating to the ocean in the spring. This protracted freshwater rearing period exposes the species to a wide range of environmental conditions, making them susceptible to habitat alteration, altered streamflow regimes, poor water quality, and predation by non-native fish (Goodman and Reid 2012, Stillwater Sciences 2014). After undergoing a metamorphosis to the juvenile stage (macrophthalmia), Pacific Lamprey emigrate to the ocean, typically between fall and spring and in conjunction with streamflow peaks (Goodman et al. 2015). Timing of outmigration and survival during this life stage are expected to be influenced by changes in streamflow.

As with salmonids, there is considerable uncertainty in how Pacific Lamprey would respond to the Water Supply Scenarios. Like other native fishes, Pacific Lamprey evolved with a natural hydrograph and rely on hydrologic cues to initiate key aspects of the life cycle such as adult migration to spawning areas and juvenile migration to ocean. Pacific Lamprey also have many of the same general ecological and habitat requirements as salmonids, such as cool water temperatures, adequate streamflow, volitional passage, and cover from predators. For these reasons, they are generally expected to respond in a similar manner as salmonids to physical, hydrological, and thermal changes associated with differences in Water Supply Scenarios. Key responses expected by Pacific Lamprey are described below.

4.4.1 Eel River Upstream of Scott Dam

Due to lack of upstream passage at Scott Dam, Pacific Lamprey cannot access the upper Eel River watershed, which has considerable amounts of suitable spawning and rearing habitats. No estimate of suitable and accessible Pacific Lamprey habitat upstream of Scott Dam has been made, but due to similar habitat requirements, it can be assumed that they would utilize a similar extent of channel as salmon and steelhead and potentially more due to their ability to climb over certain features that prevent salmonid passage. Access to these new habitats under Scenario 2 and Scenario 3 would likely result in a considerable increase in Pacific Lamprey production from the upper Eel River. If volitional passage were reestablished through addition of a fish ladder without removing Scott Dam, the design would need to consider passage requirements specific to adult Pacific Lamprey. Likewise, steps would need to be taken to allow downstream passage of out-migrating juvenile lampreys through the reservoir and past the dam. Additionally, with Scott Dam in place, larval and juvenile lampreys would be at risk of increased predation by Sacramento Pikeminnow and other non-native predators present in Lake Pillsbury.

4.4.2 Eel River from Scott to Cape Horn Dam (gage E-2)

In the reach between Scott Dam and Cape Horn Dam, the primary factors related to differences in Water Supply Scenarios expected to affect Pacific Lamprey are:

- Improved access to spawning and rearing habitats in the reach due to modifications of adult upstream fish passage conditions at Cape Horn Dam;
- Changes in larval rearing habitat area, growth, and survival due to alterations to streamflow and water temperatures; and
- Changes in timing of and survival during juvenile outmigration due to altered hydrology.

Scenarios that improve lamprey passage at Cape Horn Dam will maximize access to upstream habitats, potentially increasing lamprey production from the reach. Recent modifications to fish passage infrastructure at Cape Horn Dam, consisting of a tube that bypasses the fishway, have greatly improved upstream passage (Figure 4-23**,** D. Goodman, USFWS, pers. comm., Goodman and Reid 2017). However, this system is considered a temporary solution because its operation can be interrupted by high streamflows and debris. If Cape Horn Dam is retained (Scenarios 2 and 4b), a more permanent fishway that accounts for lamprey passage requirements will be necessary to provide uninterrupted passage at the site across a range of streamflows.

Larval lamprey are expected to be affected primarily by the lower summer streamflow and warmer water temperatures projected to occur between Scott Dam and Cape Horn Dam due to loss of storage and coldwater releases under Scenario 2 and Scenario 3 (Section 3.2). The exact effects of lower streamflows are uncertain, but it is possible that a decrease in the overall area of wetted channel could decrease the suitable area of depositional, fine-sediment larval rearing habitat, which is often located along stream margins, in side-channels, or in alcoves. This reduced habitat area, along with warmer water temperatures, could negatively impact growth and survival. On the other hand, larval lamprey in the reach could benefit from changes in channel condition under Scenario 2 and Scenario 3. The unimpaired hydrograph and absence of Scott Dam may create a more dynamic channel with greater habitat complexity, which could lead to sorting and deposition of fine sediment larval habitats that stay wetted at lower streamflows.

Since their outmigration is highly associated with streamflow cues (Goodman et al. 2015), out-migrating juvenile lamprey are also expected to be affected by changes in the seasonal timing or magnitude of peak flows associated with the Water Supply Scenarios, particularly during the winter and spring. In general, the higher, longer duration, and more variable wetter season streamflows under Scenario 2 and Scenario 3 are expected to improve juvenile lamprey survival during outmigration in the reach compared with the lower, more stable, and colder flows that would occur under Baseline and Scenario 4B.

Figure 4-23. Tubes at the Van Arsdale Fish Station used to facilitate upstream passage of Pacific Lamprey over Cape Horn Dam.

4.4.3 Eel River Downstream of Cape Horn Dam (gage E-11)

Higher wet season diversion rate at the Van Arsdale Diversion under Baseline, Scenario 2, and Scenario 4B and resulting lower winter streamflows downstream of Cape Horn Dam could have negative impacts on Pacific Lamprey relative to the natural hydrograph of Scenario 3, particularly during the spring when the differences are most pronounced (Section 3.2). Lower and muted spring peak flows may affect outmigration cues for juvenile lamprey and also make them more vulnerable to predation by Sacramento Pikeminnow. Sudden decreases in streamflow associated with the diversion could also affect spawning adult Pacific Lamprey in the reach downstream of Cape Horn Dam by reducing the area of suitable habitat or dewatering redds. The relatively small differences in summer streamflows between the Water Supply Scenarios in this reach are not expected to result in significant differences in the quantity or quality of larval lamprey summer rearing habitats.

4.4.4 Russian River Downstream of the West Branch Confluence

There are no proposed infrastructural changes that will improve or impede passage of Pacific Lamprey on the Russian River. Thus, any changes will be caused by changes to the streamflow regime. Under Scenario 3, streamflow is typically lower during all phases of the hydrograph compared to the other Water Supply Scenarios. Lower streamflows could affect the larval life stage most significantly. While level of effects are uncertain, lower streamflows could decrease the overall area of wetted channel, in turn reducing the area of depositional, fine-sediment larval rearing habitat, which is often located along stream margins, in side-channels, or in alcoves. This reduced habitat area could negatively impact growth and survival.

4.4.5 Summary

Removal of Scott Dam (Scenario 2 and Scenario 3) would provide Pacific Lamprey access to large areas of high-quality holding, spawning, and rearing habitats upstream and likely contribute to recovery of the species. Retention of Scott Dam and construction of a fishway (Scenario 4B) would also allow access to these habitats but would (1) require special design considerations for upstream and downstream passage, (2) decrease passage efficiency, and (3) lower survival due to predation in the reservoir relative to Scenario 2 and Scenario 3. Retention of Cape Horn Dam (Scenarios 2 and 4B) would require construction of a permanent fishway that accounts for lamprey passage requirements across a range of streamflows and that can withstand high flows and sediment loads.

There is considerable uncertainty in Pacific Lamprey responses to expected differences in hydrology, water temperatures, and geomorphology between Water Supply Scenarios, but in general, Scenario 2 and Scenario 3 are expected to benefit the species by reestablishing access to the upper watershed and creating a more natural hydrological regime and greater habitat complexity in downstream reaches. Downstream of Cape Horn Dam, the higher wetter season diversion rate under Baseline, Scenario 2, and Scenario 4B and resulting lower streamflows could have minor negative impacts on Pacific Lamprey relative to Scenario 3 by (1) altering outmigration cues for juveniles, (2) making outmigrants more vulnerable to predation by Sacramento Pikeminnow, and (3) affecting spawning behavior and success.

4.5 Non-native Predator Responses

Invasive species are one of the most ecologically destructive threats to freshwater ecosystems and cost millions of dollars a year (Pimentel et al. 2000). Non-native predators can reduce native salmonid populations through direct predation (Zimmerman 1999, Nakamoto and Harvey 2003, Sanderson et al. 2009, Cavallo et al. 2013) and indirect competition (Reese and Harvey 2002). For example, Striped Bass reduce the survival of Chinook Salmon in California's Central Valley (Cavallo et al. 2013) and Smallmouth Bass prey on out-migrating juvenile Chinook Salmon in the Columbia River (Zimmerman 1999). Sacramento Pikeminnow were introduced into the upper Eel River via Lake Pillsbury in the late 1970s and have since expanded throughout much of the Eel River watershed (SEC 1998, Brown and Moyle 1997, Kinziger et al. 2014). Pikeminnow consume and compete with juvenile salmonids and are thought to be a barrier to salmonid recovery in the upper Eel River (Reese and Harvey 2002, Brown and Moyle 1997, NMFS 2002, Nakamoto and Harvey 2003, PG&E 2018a). Pikeminnow are likely to have particularly adverse effects on native species such as Pacific Lamprey, Sacramento Sucker (*Catostomus occidentalis*), and sculpin (*Cottus sp.*) that tolerate warmer water temperatures than salmonids and have a higher degree of overlap with pikeminnow during the summer (Brown and Moyle 1991, White and Harvey 2001). Largemouth Bass, which also prey on salmonids, are also present in the upper Eel River, particularly in Lake Pillsbury and the vicinity of Van Arsdale Reservoir and Cape Horn Dam infrastructure (PG&E 2020). Pikeminnow and bass are also known predators of the native Foothill Yellow-legged Frog (Ashton and Nakamoto 2007, Paoletti 2009).

Hydroelectric projects, including the Project, can create ideal habitats for these non-native predators, which prefer slow and warm waters, rather than swift and cold waters (Brown and Moyle 1981, Moyle and Baltz 1985, Moyle 2002). Slow-moving habitats created by Lake Pillsbury and Van Arsdale Reservoir, coupled with lower streamflows downstream of Cape Horn Dam, have provided ideal habitat for pikeminnow and bass (*Micropterus*) to thrive (Brown and Moyle 1997, SEC 1998, PG&E 2020).

The different Water Supply Scenarios analyzed consist of various streamflow regimes and infrastructural changes that are expected to alter the habitat and survival of non-native predators. On the Eel River, we focus analysis on pikeminnow and Largemouth Bass. On the Russian River, Sacramento Pikeminnow are native and part of the natural aquatic community. However, both Largemouth Bass and Smallmouth Bass reside in the Russian River and prey on juvenile Chinook Salmon, Coho Salmon, and steelhead. In this section, we discuss the anticipated responses of non-native predators to infrastructure, habitat, and streamflow changes expected under each Water Supply Scenario in the upper Eel and Russian rivers.

4.5.1 Eel River Upstream of Scott Dam

Lake Pillsbury provides warm water and lentic habitat that is ideal for pikeminnow and bass. Preliminary electrofishing and gill netting data from Lake Pillsbury collected for relicensing Study AQ 9 – Fish Populations indicate that there are numerous large pikeminnow and bass in the reservoir (Figure 4-24, PG&E unpublished data, 2018). During high streamflow events, spills from Lake Pillsbury can move larval, juvenile, and adult life stages of non-native fish from Lake Pillsbury downstream into the Eel River (J. Fuller, NMFS Fisheries Biologist, pers. comm., January 2020). While unclear how critical the flux of individuals from Lake Pillsbury is to the success of non-native fish populations downstream of Scott Dam, the removal of Scott Dam under Scenario 2 and Scenario 3 would remove a large area of high-quality habitat for these species and likely result in decreased recruitment and smaller populations throughout the upper Eel River. The decrease in population of non-native predators and competitors is expected to improve juvenile salmonid growth and survival rates upstream of Lake Pillsbury if anadromy were restored in this reach.

4.5.2 Eel River from Scott to Cape Horn Dam (gage E-2)

Large numbers of pikeminnow are found between Scott Dam and Cape Horn Dam (PG&E 2020). Van Arsdale Reservoir and infrastructure associated with the diversion and Cape Horn Dam are considered “hot-spots” for both pikeminnow and bass (PG&E 2020, Figure 4-21). We identified three key mechanisms that could affect non-native predators in this reach in response to the different Water Supply Scenarios:

1. Removal or modification of infrastructure may reduce habitat and predation opportunities for pikeminnow and bass;
2. Removal of Scott Dam would increase habitat connectivity and access to preferred cooler water habitats for salmonids, thereby reducing overlap with pikeminnow and bass; and,
3. Seasonal changes in streamflow and temperatures could affect distribution, habitat suitability, survival, and bioenergetics of pikeminnow and bass.

Each of these mechanisms is examined in the ensuing sections and potential implications for salmonid productivity are discussed in Section 5.

Figure 4-24. Photo of large Sacramento Pikeminnow taken in Van Arsdale Reservoir at the diversion structure during relicensing studies in 2018.

4.5.2.1 Infrastructure Modification

The removal of Scott Dam, coupled with the removal, modification, or replacement of Cape Horn Dam, would likely represent the most important change to non-native predator habitat in the reach between dams. The slow-moving and warming surface waters of Van Arsdale Reservoir, along with infrastructure associated with the diversion, dam, and fishway, create ideal habitats for pikeminnow and bass. Both the diversion intake and fishway infrastructure effectively concentrate juvenile salmonids and other native fish in a small area and make them more susceptible to predation (NMFS 2002, D. Goodman, USFWS, pers. comm. February 2020). These conditions result in high densities of large pikeminnow and bass in and around the Project infrastructure (PG&E 2020). Length at age data from Brown and Moyle (1997) suggest that pikeminnow in the reach between Scott and Cape Horn dams may also grow more rapidly than pikeminnow in other portions of the Eel River watershed. High concentrations of large pikeminnow have also been observed at other dams, such as Red Bluff Diversion Dam in the Sacramento River, where streamflow patterns and confused or injured fish passing the dam attract them (Brown and Moyle 1981). Likewise, predation rates by Striped Bass (*Morone saxatilis*) on juvenile Chinook Salmon have been shown to be heightened at diversion dam infrastructure (Sabal et al. 2016) and likely occurs for pikeminnow as well.

Under Scenario 2 (with Alternative 3) or Scenario 3, Van Arsdale Reservoir would be eliminated, and the river would return to a natural channel consisting of riffles, runs, and pools. Removing the large area of low velocity pool habitat that pikeminnow and bass thrive in would eliminate a large portion of the habitat available in the reach between the dams, reducing their numbers in the reach. Removal of Cape Horn Dam would also allow juvenile salmonids to migrate past the former dam site across the entire width of the river using natural cover, likely reducing predation caused by predators concentrating at Cape Horn Dam, the diversion intake, and the Van Arsdale Fishway relative to Baseline.

With the retention of Cape Horn Dam under Scenario 2 (Alternative 2) and Scenario 4B, modifications to the dam, the fishway, and diversion infrastructure would be made to improve upstream and downstream passage efficiency of juvenile salmonids and should be designed to minimize predator habitat and predation hotspots. These infrastructure changes would likely result in reduction of predator populations and increased salmonid survival relative to Baseline.

4.5.2.2 Overlap with Salmonids

Under Scenario 2 and Scenario 3, Scott Dam would be removed, allowing salmonids unimpeded access to many more miles of habitat in cooler and higher gradient reaches of the upper mainstem Eel River, the Rice Fork, and tributaries (Section 5). Brown and Moyle (1991) suggest that in systems where both pikeminnow and salmonids co-evolved, there is minimal overlap between salmonid rearing habitat and pikeminnow habitat. The former generally occurs in cooler and smaller tributaries, while the latter occurs in the warmer, lower gradient reaches of mainstem of rivers. In general, pikeminnow are restricted to streams with summer water temperatures of 18 to 28°C (64–82°F, Brown and Moyle 1997, Harvey et al. 2002, Moyle 2002). These studies also suggest that where there is overlap between species, there is reduced growth and higher mortality of salmonids. Reduced growth can occur by juvenile salmonids occupying less energetically efficient habitat as they try to avoid predation by pikeminnow. Pikeminnow and bass are not likely to utilize the smaller and colder headwater streams of the upper Eel River, providing rearing habitats where juvenile salmonids have refuge from non-native predation. However, out-migrating smolts would still have to contend with predation risks as they move into the warmer, low-gradient mainstem reaches downstream where these non-native predators resided.

4.5.2.3 Sacramento Pikeminnow Response to Streamflow and Temperature Changes

Under Baseline and Scenario 4B, hypolimnetic releases from Lake Pillsbury create cooler and elevated streamflow between Scott and Cape Horn Dams relative to that expected under Scenario 2 and Scenario 3 (Sections 3.1 and 3.2). Competition and predation by pikeminnow (and likely bass) on juvenile salmonids is reduced in water temperatures less than 18°C (Reese and Harvey 2002). Increased metabolic rate and decreased gastric evacuation time with increasing temperature indicates that pikeminnow feed more actively with increasing water temperatures (Vondracek 1987). Under Scenario 2 and Scenario 3, streamflows would be unimpaired upstream of Cape Horn Dam, and they would be reduced in the summer by approximately 70 to 120 cfs for drier and wetter years, respectively (Section 3.1). Along with decreased flows, daily average summer water temperatures are expected to be approximately 14 to 20°C (depending on water year) under Baseline and Scenario 4B and increase to 20 to 25°C under Scenario 2 and Scenario 3 (Section 3.2). These increased water temperatures are expected to increase the competitive advantage of pikeminnow over juvenile salmonids and increase predation in the reach between the dams relative to Baseline. However, as described above, these water temperature related changes may be partially or fully offset by (1) a reduction in preferred physical habitat for non-native predators through elimination of Van Arsdale Reservoir and modification of infrastructure and (2) a reduction in spatial overlap between non-native predators and rearing salmonids due to access to colder tributaries under Scenario 2 and Scenario 3.

4.5.3 Eel River Downstream of Cape Horn Dam (gage E-11)

Pikeminnow predation on juvenile salmonids downstream of Cape Horn Dam is believed to be a significant source of mortality (SEC 1998, NMFS 2002). Brown and Moyle (1997) found that salmonids were present in 100 percent of diet samples of pikeminnow larger than 100 millimeters (mm) collected near the Outlet Creek confluence during the outmigration season. NMFS (2002) suggests that lower streamflows coupled with very warm (>24°C, PG&E 2016, Section 3.4) water temperatures in the reach create ideal habitat for pikeminnow to thrive and prey on juvenile salmonids. The pool downstream of Cape Horn Dam and the mouth of the fishway also create conditions that attract large pikeminnow and Largemouth Bass that can prey upon migrating salmonids (Stillwater Sciences, pers. obs., PG&E 2020). We evaluate two key mechanisms that may affect non-native predators downstream of Cape Horn Dam under different Water Supply Scenarios:

1. Infrastructure modification at Cape Horn Dam
2. Changes to streamflow regime downstream of Cape Horn Dam

4.5.3.1 Infrastructure Modification at Cape Horn Dam

As with the reach upstream of Cape Horn Dam, Under Scenario 2 (Alternative 3) or Scenario 3, removal of Cape Horn Dam, the fishway, and associated infrastructure would likely reduce non-native predator population sizes, preferred habitat area, and predation opportunities immediately downstream of Cape Horn Dam. Removal of Cape Horn Dam would also allow unimpeded juvenile fish passage across the entire river channel, rather than concentrated at the fishway where predation rates on juvenile salmonids are presumed to be high. Additionally, it is possible that the area of preferred non-native predator habitat in the large pool downstream of Cape Horn Dam would be reduced along with predation opportunities on juvenile salmonids that become stunned/injured when spilling over the face of the dam with the return to a natural river corridor at the site. Together, these changes are expected to reduce the abundance of non-native predators and predation on salmonids relative to Water Supply Scenarios where Cape Horn Dam is retained.

4.5.3.2 Changes in Streamflow Regime Downstream of Cape Horn Dam

Differences in streamflows between Water Supply Scenarios, particularly in the spring and summer, may affect pikeminnow habitat area, abundance, ecological interactions with salmonids and other native fishes in the reach downstream of Cape Horn Dam. The magnitude of these effects is difficult to predict, but Water Supply Scenarios that decrease streamflow—thereby increasing water temperature and area of low-velocity habitats preferred by pikeminnow—are generally expected to bolster pikeminnow populations and reduce salmonid survival and habitat in the reach. This generality is supported by annual fish monitoring conducted by PG&E at sites downstream of Cape Horn Dam between 2005 and 2017, which indicates that at each site, pikeminnow densities (based primarily on smaller, juvenile fish) were higher in drier water years compared with wetter water years (Figure 4-25). It is unclear if larger and older fish, which are more likely to prey on juvenile salmonids, also follow this trend. In contrast, juvenile steelhead densities were higher in wetter water years, especially closer to Cape Horn Dam, compared with drier water years (Figure 4-26).

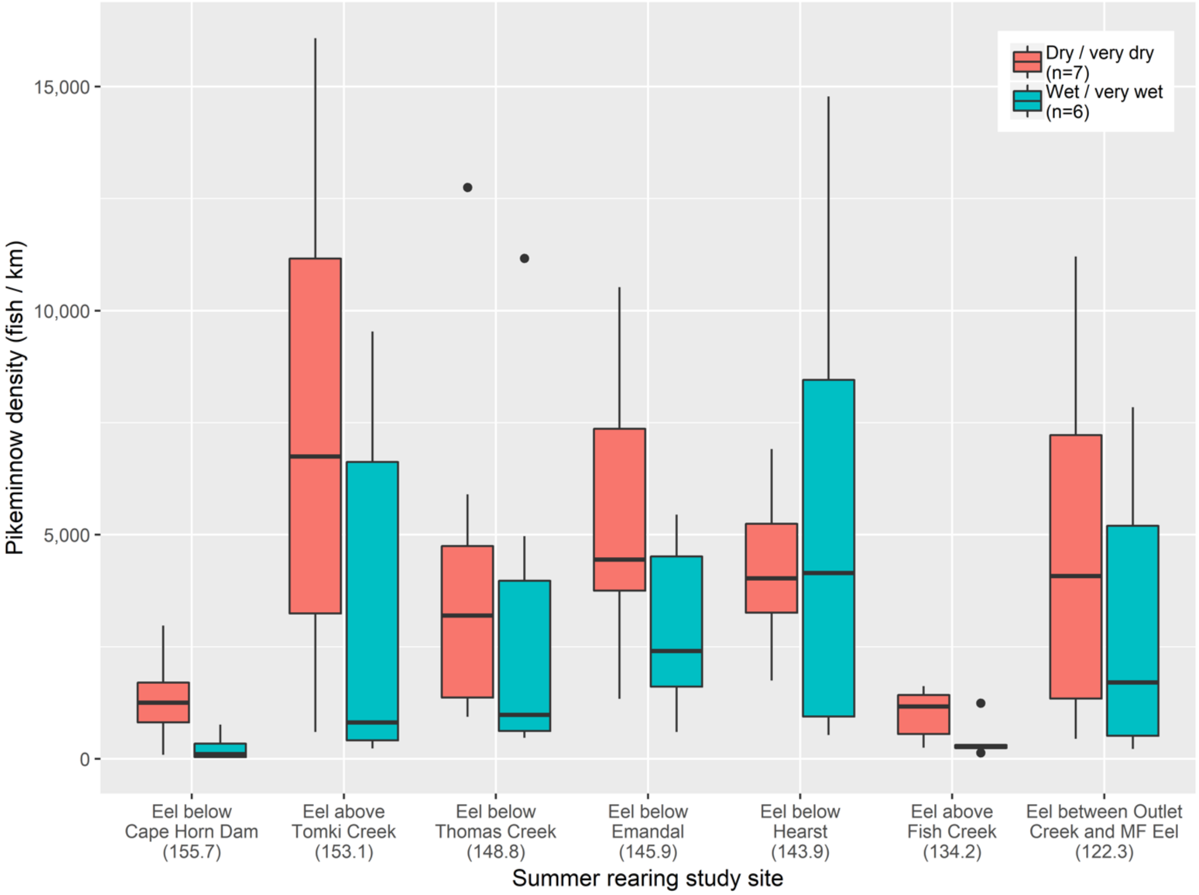

Figure 4-25. Sacramento Pikeminnow densities at select sites in the upper Eel River from 2005–2017 by drier and wetter water year types (Data source: PG&E 2018). All size classes were lumped and catches were dominated by smaller, juvenile fish. The median is indicated by horizontal bars, 25th and 75th percentiles by box edges, minimum and maximum by vertical lines (whiskers), and outliers by points.

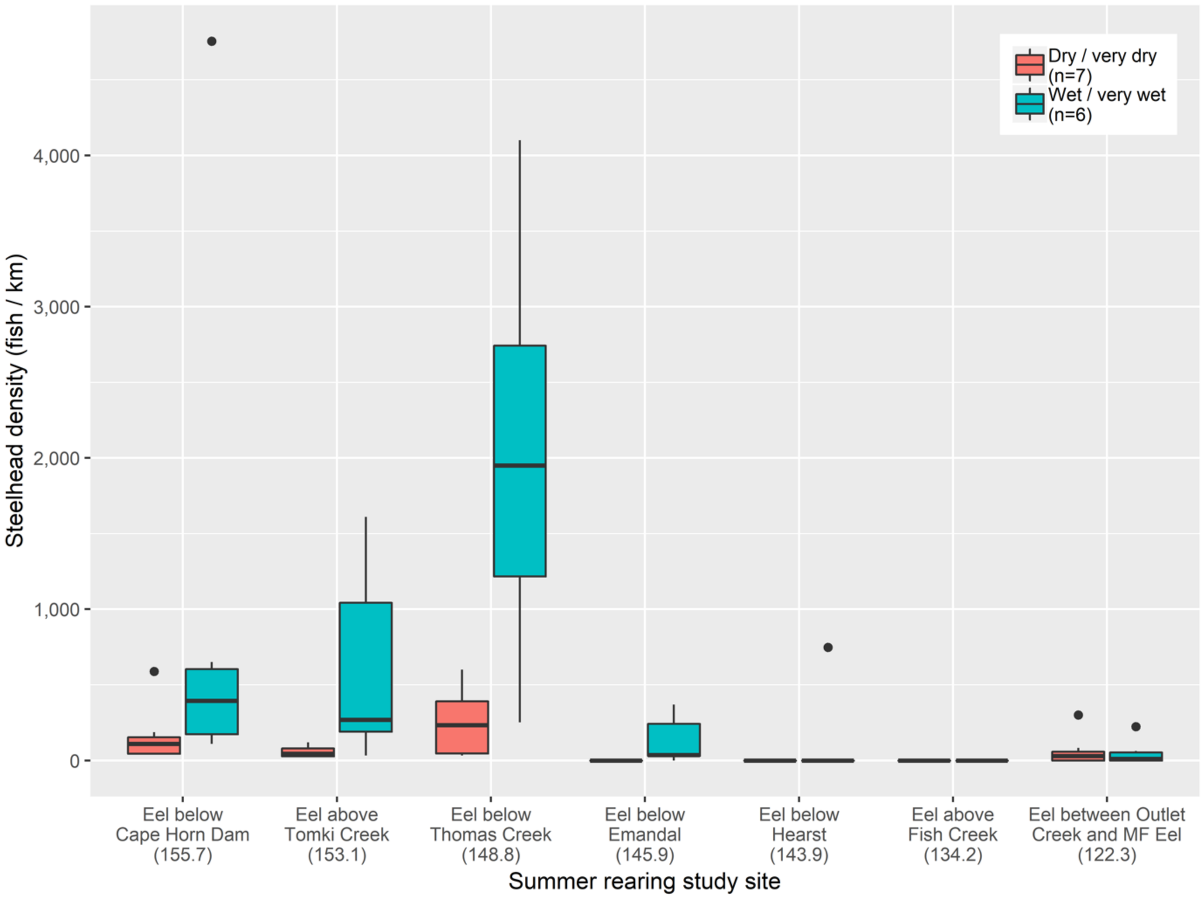

Figure 4-26. Juvenile steelhead densities at select sites in the upper Eel River from 2005–2017 by drier and wetter water year types (Data source: PG&E 2018). All size classes lumped.

During the spring, the unimpaired spring recession streamflows downstream of Cape Horn Dam under Scenario 3 would typically be higher than under both Scenario 2 and Scenario 4B, which have springtime diversion at Van Arsdale (Section 3.1). Depending on water year type, and when spring diversions cease, Scenario 2 streamflows would typically be about 100 to 300 cfs lower than Scenario 3 in March and April and 0 to 200 cfs lower in May and June. These lower streamflows, along with likely warmer temperatures, may:

1. Increase the area of preferred pikeminnow habitat while also reducing salmonid habitat and promoting overlap between the two;
2. Increase pikeminnow spawning success and larval recruitment; and
3. Increase their competitive advantage over juvenile steelhead due to warmer water temperatures.

During the summer, under Scenario 2 and Scenario 3, streamflow downstream of Cape Horn Dam is projected to be similar to or up to about 20 cfs lower than under Baseline and Scenario 4B depending on water year (Section 3.1). The effects of lower summer streamflows on pikeminnow populations are uncertain. Since streamflow and water temperature conditions expected under Baseline and Scenario 4B already create very favorable conditions for pikeminnow, even lower streamflows may not create additional benefits to the pikeminnow population.

Changes in the winter streamflow regime may also affect pikeminnow populations in the reach. Loss of reservoir attenuation under Scenario 2 and Scenario 3 is projected to result in higher peak flows than under the Baseline and Scenario 4B and Feasibility Study Alternatives. Peak flows under Scenario 3 would be even larger than under Scenario 2 due to lack of diversion at Van Arsdale Reservoir. The effects of these higher winter peak flows on pikeminnow are uncertain, but generally expected to result in fewer pikeminnow caused by reduced recruitment in higher streamflows. Pikeminnow abundance in the reach is generally higher in drier water years (Figure 4-25), which could be due in part to reduced overwinter survival or larval recruitment in wetter years with high peak flows.

Effects of expected changes in pikeminnow populations under different Water Supply Scenarios on salmonid population productivity downstream of Cape Horn Dam are discussed in Section 5.2.

4.5.4 Russian River Downstream of the West Branch Confluence

The Russian River has native Sacramento Pikeminnow, which co-evolved with juvenile salmonids and are not presumed to be a critical source of mortality for salmonids. However, Smallmouth and Largemouth Bass are found in the Russian River and are considered to be a significant source of predation on salmonids. Largemouth Bass are mostly concentrated in the lower portion of the Russian River and are outside the effect of the Project (SCWA 2016). Smallmouth Bass are distributed throughout the Russian River and likely prey on juvenile salmonids; studies from the Columbia River have suggested that they can be an important predator on juvenile Chinook Salmon (Zimmerman 1999). There are no proposed infrastructure changes to the Russian River side of the Project, thus we do not anticipate any changes to their habitat or distribution from that source. The primary mechanism that could affect non-native predators on the Russian River due to differences in Water Supply Scenarios is changes to the streamflow regime.

4.5.4.1 Changes to Streamflow Regime

During drier years, streamflows will be reduced by 50 cfs under Scenario 2 and 3 relative to Baseline and Scenario 4B. Studies describing the rates of predation and competition on juvenile salmonids by Smallmouth Bass are limited, and it is unclear how this may change with decreased streamflow. Similar to pikeminnow on the Eel River, one hypothesis is that under lower streamflows there would less habitat, which would increase overlap between juvenile salmonids and Smallmouth and Largemouth Bass and may increase predations rates. However, there are several reasons why there would likely be minimal change to predation rates. First, turbidity is typically very high downstream of Coyote Valley Dam (SCWA 2016) and likely outweighs any effect of streamflow change because highly turbid water reduces predation rates (Gregory and Levings 1998). Second, the streamflow change may not be significant enough to reduce river stage to a point where significant overlap could occur between the predator and prey.

4.5.5 Summary

Predation and competition by non-native predatory fish are critical factors limiting recovery of native fish populations in the upper Eel River. Removal of both dams and their reservoirs would eliminate large areas of warm, lentic habitat and infrastructure that favor non-native predator populations and facilitate predation on native fish. Providing unimpeded passage for juvenile salmonids to coldwater habitat upstream of Cape Horn Dam and Scott Dam would also reduce overlap between non-native predatory fish and these species. Effects of differences in streamflow between Water Supply Scenarios on predatory fish are uncertain, but Water Supply Scenarios that decrease streamflow and increase water temperature and low velocity habitat are generally expected to favor pikeminnow populations over native species. In the reach between Scott Dam and Cape Horn Dam, loss of coldwater hypolimnetic releases from Lake Pillsbury in Scenario 2 and Scenario 3 is expected to increase the competitive advantage of pikeminnow over juvenile salmonids and increase predation in the reach relative to Baseline and Scenario 4B. However, these effects may be offset by a reduction in lentic habitat, removal of predation hotspots, salmonid access to the upper watershed, and reduced overlap between native and non-native species under Scenario 2 and Scenario 3. Downstream of Cape Horn Dam, lower spring streamflows due to the Van Arsdale Diversion under Scenario 2 and Scenario 4B are generally expected to increase the area of preferred pikeminnow habitat relative to natural spring hydrology of Scenario 3.

Overall, differences between evaluated Water Supply Scenarios are expected to have minimal changes to existing predation rates in the Russian River.

4.6 Herpetological Responses

Riverine amphibians and herpetofauna in the Eel River watershed evolved to succeed in a Mediterranean climate with wet winters and dry summers, with annual streamflow variability influencing cohort success. Timing, magnitude, and duration of hydrologic events as well as aquatic thermal regime influence population-level dynamics of native herpetofauna such as Foothill Yellow-Legged Frogs and Northwestern Pond Turtles (Wheeler et al. 2015, Ashton et al. 2015). These two native species are particularly sensitive to streamflow management, and they receive protection in the states they occur and are under review for federal ESA listing. Non-native bullfrogs, turtles, and sport fishes can gain competitive advantage over native focal herpetofauna in managed aquatic habitats, which can be a significant source of direct (predation) and indirect (competition) mortality on native species (e.g., Fuller et al. 2011, Lambert et al. 2019). The following paragraphs discuss native and non-native herpetofauna that may be affected by Water Supply Scenarios and Feasibility Study Alternatives. Descriptions in this section references the Feasibility Study Alternatives and corresponding Water Supply Scenarios presented in Table 1-2.

- Foothill Yellow-legged Frog (FYLF; *Rana boylii*); native: This obligate river-breeding frog relies on edgewaters adjacent to flowing water for reproduction, preferring gravel and cobble bars with open canopy. Egg masses and tadpoles are at risk of scour or desiccation mortality from stage changes, and developmental rate is influenced by water temperature (Catenazzi and Kupferberg 2013). Timing of oviposition and subsequent metamorphosis are critical in dictating cohort success. When streamflow and temperature regimes are altered by dam operations, natural cues can be decoupled from river conditions leading to poor choices in timing and placement of egg masses (Lind et al. 1996). Hypolimnetic releases that keep water unnaturally cold through the summer growth period reduce growth rate and lower cohort success (Wheeler et al. 2015). Streamflow management strategies that recognize timing of biological patterns and resource productivity improve cohort success for FYLF (Railsback et al. 2016). These are frogs of lotic systems (they do not disperse across reservoirs), so Lake Pillsbury effectively isolates each of the inlet arms and downstream populations. Altered hydrology and channel morphology can facilitate invasive aquatic predators known to prey on FYLF, including bullfrogs (Fuller et al. 2011), Sacramento Pikeminnow (Ashton and Nakamoto 2007), and bass (Paoletti 2009).
- Northwestern Pond Turtle (NWPT; *Actinemys marmorata*); native: Occurs in a variety of habitats across the west coast states. It is always associated with access to temporary or

permanent freshwater, including rivers, creeks, marshes, ponds, lakes, and reservoirs, but also relies on terrestrial habitats to complete its life history (Bury et al. 2012). When a river is impounded, lotic waters become lentic, which can increase opportunities for NWPT, but can also favor invasive species. A streamflow regime that includes high winter streamflows and annual streamflow variation maintains terrestrial habitats and provides variable habitat for benthic macroinvertebrates, providing a prey base for native herpetofauna (Poff et al. 1997, Poff and Zimmerman 2010). Young turtles are especially vulnerable to invasive aquatic predators, including bullfrog and bass, and nest predation may increase in areas where human infrastructure or habitation promotes meso-carnivore release (e.g., Soulé et al. 1988).

- American Bullfrog (bullfrog; *Lithobates catesbeianus*); invasive: From east of the Rockies, bullfrogs are invasive in the west, and as voracious predators, disrupt novel ecosystems (Adams and Pearl 2007). In the upper Eel River, bullfrogs require year-around lentic waters for successful rearing, which is provided by Lake Pillsbury and Van Arsdale Reservoir. Scouring winter streamflows and summer drought natural to west coast rivers can limit bullfrog proliferation. Reservoirs and other impoundments provide stable lentic conditions favored by this invasive predator, and downstream of dams, minimum summer baseflow requirements and reduction of scouring winter streamflows allows bullfrogs to persist to the detriment of native amphibians (Fuller et al. 2011).
- Red-eared Slider (RES; *Trachemys scripta elegans*); invasive: Red-eared Sliders are native to the eastern United States. This invasive turtle out-competes and out-produces native turtles (Lambert et al. 2019). Potential for hybridizing with or transfer of parasites and diseases to native NWPT is uncertain. Reported observations in Potter Valley, Lake Mendocino, and Clear Lake warrant vigilance if Eel River reservoirs remain in place, as this invasive turtle often thrives in anthropogenically modified habitats once established. Removal of one or both dams would reduce invasion risk by this turtle.

4.6.1 Eel River Upstream of Scott Dam

Lake Pillsbury fragments a historically continuous FYLF population and promotes proliferation of non-native aquatic predators. Feasibility Study Alternatives which preserve Scott Dam (Baseline and Scenario 4B) would continue to provide suitable habitat for NWPT as well as non-native predatory fishes and bullfrogs (Section 4.5). Removal of Scott Dam (Scenarios 2 and 3) will reconnect fragmented FYLF populations now isolated in the tributaries upstream of the reservoir and enable them to interact with populations and access habitat downstream of Scott Dam. Miles of channel inundated beneath the reservoir will become available for colonization by FYLF.

A survey conducted in June 2018 confirmed NWPT use the reservoir, including young turtles, indicating a reproducing population (S. Kufperberg and D. Ashton, pers. obs.). No non-native turtles were detected in or around the reservoir. Turtle activity was focused in the arms of the river forks and little use was documented near the dam, west shore, or wetlands at the north shore at time of survey. NWPT preference for inlet tributary arms, and away from the dam and deeper waters, suggests minimal effects to turtles from Scott Dam removal. There may be some reduction in suitable lentic habitat for NWPT, but ample pools should remain in the river, which may be enhanced by beaver activity, as observed in the Rice Fork and upper Eel River (June 2018, S. Kupferberg and D. Ashton). Scott Dam removal would remove habitat that favors non-native aquatic predators (e.g., bullfrogs, bass) which prey upon and compete with native herpetofauna.

4.6.2 Eel River from Scott Dam to Cape Horn Dam

Impaired streamflows of the Baseline degrade habitat quality for FYLF and NWPT, as explained in the following sections. Anticipated outcomes of streamflow and thermal regime changes from Scott Dam to Van Arsdale under Scenario 4B and Scenario 2 (Alternatives 2 and 3) are discussed first, followed by discussion of Van Arsdale Reservoir.

4.6.2.1 Eel River Downstream of Scott Dam

Under Baseline, downstream of Scott Dam, peak flow events are attenuated and less variable, leading to riparian encroachment and reduction in quality of FYLF breeding and rearing habitat. Unnaturally high and stable summer baseflows with coldwater hypolimnetic releases from the reservoir slows development and growth of FYLF eggs and tadpoles. Conditions under Scenario 4B are very similar to Baseline and minimal change for focal species is expected. Scenario 2 (Alternatives 2 and 3) remove Scott Dam, resulting in unimpaired streamflow conditions between the dams. Higher and more frequent winter peak flows will reduce riparian encroachment and maintain open bars used by FYLF for breeding and rearing. Scouring winter streamflows can limit bullfrog persistence by flushing out overwintering tadpoles. High winter streamflows can increase spring and summer diatom crop that feeds FYLF tadpoles, as well as benthic macroinvertebrates which are later consumed by adult FYLF (Power et al. 2008, 2013). Warmer, lower, variable summer streamflows will improve tadpole growth leading to earlier metamorphosis and larger froglet sizes. Higher water temperatures and a more gradual streamflow recession in the spring should increase FYLF reproduction success. Return to natural streamflow and thermal regimes coupled with more natural riparian recruitment and geomorphic function downstream of Scott Dam site to the footprint of Van Arsdale Reservoir is expected to benefit native herpetofauna and discourage non-native aquatic predators (bullfrogs, bass, and pikeminnow).

4.6.2.2 Van Arsdale Reservoir

Scenario 2 (Alternative 2) keeps the impoundment behind Cape Horn Dam, which will continue to fragment FYLF populations upstream and downstream of the dam while, at the same time, promoting non-native aquatic predators. Van Arsdale Reservoir expands the amount of suitable habitat for NWPT, but also promotes non-native bullfrogs. FYLF and NWPT are more adapted for life in the riverine environment than are non-native bullfrogs, and it is likely that sufficient NWPT habitat would remain even without Van Arsdale Reservoir. For example, a deep backwater along bedrock on the north shore may still hold ponded water for NWPT after Cape Horn Dam’s removal.

4.6.3 Eel River Downstream of Cape Horn Dam

Removal of Cape Horn Dam (Scenario 2 [Alternative 3]) would result in higher winter peak flows and lower, variable summer streamflows. This is expected to reduce riparian encroachment and maintain coarse sediments, enhancing quantity and quality of FYLF breeding and rearing habitats. Increased peak flows will promote formation and maintenance of pools used by NWPT. Large periodic floods form and maintain nesting habitat for NWPT. Outcomes for options which preserve Cape Horn Dam and diversions to the Russian River are dependent on streamflow management. Higher water temperatures in the spring and summer along with a more gradual spring streamflow recession into less impaired summer baseflows are most likely to benefit FYLF.

4.6.4 Russian River Downstream of the West Branch Confluence

No significant change is expected for NWPT under the different Water Supply Scenarios, particularly downstream of Coyote Dam where agriculture and residential development adjacent to the river channel have already reduced suitability of terrestrial habitats required by NWPT. Reduced summer flows under Scenario 2 (Alternatives 2 and 3) are expected to reduce proliferation of non-native aquatic predators, including bullfrogs. In the long-term, native herpetofauna are expected to benefit from reduction in competition, predation, and disease proliferation. Although reduced summer flows could lead to localized negative impacts to native herpetofauna during periods of extreme drought when animals are concentrated in remaining pools (Adams et al. 2017), these extreme conditions can give locally adapted native species a competitive advantage over non-native aquatic predators. And while no changes are expected for FYLF under Scenario 4B (Alternative 1), reduced and variable summer flows downstream of Coyote Dam in Scenario 2 (Alternatives 2 and 3) are expected to decrease egg mass scour mortality and increase developmental rate for FYLF eggs and tadpoles (Lind et al. 1996, Wheeler et al. 2015). No geomorphic changes are expected, but higher peak flows and lower, variable summer flows under Scenario 2 (Alternatives 2 and 3) are expected to reduce riparian encroachment, improving habitat suitability for FYLF and NWPT.

4.6.5 Summary

Removal of Scott Dam should enhance habitat for FYLF and NWPT through habitat reconnection, reversing riparian encroachment, and reduction of non-native predators and competitors. The return to a natural hydrograph with warmer spring and lower summer baseflows would likely improve growth and reproductive success for FYLF. Removal of Cape Horn Dam and Van Arsdale Reservoir (Scenario 2 [Alternative 3]) would reduce habitat for non-native aquatic predators, while also improving, expanding and reconnecting habitat and populations of FYFL downstream of Scott Dam. The conversion of lentic habitat (Lake Pillsbury and Van Arsdale Reservoir) back to a lotic condition may reduce the quantity of NWPT habitat; however NWPT are well adapted to utilize pools in-river that are likely to remain once the channel returns to equilibrium.

4.7 Benthic Macroinvertebrate Responses

Fish growth is a function of food availability, consumption efficiency, metabolic rates, and water temperature (Railsback and Rose 1999, Fausch 1984, Plumb and Moffit 2015). Benthic macroinvertebrates make up a significant portion of food resources for juvenile salmonids in California (Rundio and Lindley 2008, McCarthy et al. 2009), including the Eel River (Uno and Power 2015). When environmental conditions such as temperature, velocity, and depth (a function of streamflow) and food availability are sufficient, juvenile salmonids occupy energetically profitable foraging positions (Hughes and Dill 1990, Piccolo et al. 2014, Naman et al. 2016) and grow to size classes that increase the probability of survival during outmigration and ocean rearing (Bond et al. 2008, Zimmerman et al. 2015). In addition, food availability can be an important factor in determining migration patterns for some salmonids, such as native *Oncorhynchus mykiss* (Benjamin et al. 2013).

Dams alter the natural streamflow regime (Bunn et al. 2002, Poff et al. 1997, Poff and Zimmerman 2010, Palmer and Ruhi 2019) and affect the biodiversity, abundance, size, and phenology of benthic macroinvertebrates (Dewson et al. 2007, Power et al. 2015, Naman et al. 2016, Caldwell et al. 2018; Jansen et al. 2020). To understand the effects that altered streamflow regimes downstream of dams may be having on benthic macroinvertebrates' productivity in regulated rivers, it is useful to explore how key components of the hydrograph influence benthic macroinvertebrates in unregulated areas.

In the Eel River watershed, the benthic macroinvertebrate community assemblage differed downstream of Cape Horn Dam from a comparable, unregulated site on the Middle Fork, and these differences were in part due to less green filamentous algae, *Cladophora*, and its associated benthic macroinvertebrates, such as Chironomidae, downstream of the dam (Jansen et al. 2020). In the South Fork Eel River, changes to the abundance, diversity, and timing of food for salmonids are linked to the magnitude of winter floods (Power et al. 2015). In years with streamflows that are high enough to mobilize the bed, armored grazer caddisflies (*Dicosmoecus gilvipes*) are lost to scour, allowing diatoms and filamentous algae to grow during spring without intense grazing (Power et al. 2008). Diatoms and algae then serve as the energy base to high-quality benthic macroinvertebrates (Finlay et al. 2002), which are readily consumed and provide quality energy for juvenile salmonids.

During years with low winter streamflows, armored grazers remain intact and reduce algal growth, which supports fewer high-quality prey items for juvenile salmonids (Power et al. 2008, 2013). In addition, higher streamflows (which inundate floodplains) can result in both increased habitat for aquatic macroinvertebrates (Jeffres et al. 2008, Rosenfeld et al. 2008) and flux of terrestrial macroinvertebrates (Baxter et al. 2005). In general, the natural streamflow regime that captures high winter streamflows and annual streamflow variation typically promotes improved and variable habitat for macroinvertebrates (Poff et al. 1997, Poff and Zimmerman 2010).

Habitat quality can also contribute to the abundance and diversity of benthic macroinvertebrates (Naman et al. 2017). Typically, riffle habitats with gravel and cobbles that have well-oxygenated interstitial space promote abundance of benthic macroinvertebrates, which are an important food source for juvenile salmonids (Gore et al. 2001, Naman et al. 2016).

In this assessment, we use modeled daily streamflow data from each Water Supply Scenario coupled with technical and peer reviewed literature to assess how each Water Supply Scenario may affect benthic macroinvertebrates.

4.7.1 Eel River Upstream of Scott Dam

The river channel within the Lake Pillsbury footprint is currently saturated with fine sediment, and reservoir hypolimnetic conditions are typically anoxic in summer, and contribute no invertebrates to salmonid productivity. If Scott Dam remains, there would be no change. If Scott Dam is removed (Scenario 2 and 3), the soft sediment would either be mechanically removed and/or flushed out in storm events and eventually the substrate and channel would return to pre-dam conditions. As the channel reformed under the unimpaired flow regime, the invertebrate community and productivity would be one of the first biological communities to shows signs of recovery (Carlson et al. 2018).

4.7.2 Eel River from Scott Dam to Cape Horn Dam

Baseline streamflow conditions downstream of Scott Dam are represented by lower peak flow events and less variation, with higher and stable summer baseflows relative to unimpaired streamflow scenarios (Scenarios 2 and 3). The higher summer streamflows have relatively cooler water temperature because water is sourced from hypolimnetic releases from Scott Dam. A 1993 PIT tag study (SEC 1998) found high growth rates of 0.7 to 0.9 mm/day for juvenile steelhead; however, there were only two recaptures. SEC (1998) also used fish scale analysis of returning adults to infer that the majority of those returning individuals reared in the habitat between Scott Dam and Cape Horn Dam. While observed fish growth rates (n=2) and the scale analysis suggest there is adequate benthic macroinvertebrate productivity to produce successful salmonids, the data is limited by having only two recaptures and requires further validation. Unfortunately, quantitative information on benthic macroinvertebrate production is unavailable.

Conditions under the modified RPA (Scenario 4B) are very similar to Baseline and minimal change is expected. Under Scenario 3 and Scenario 2, the initial flush of sediment would likely result in stark declines to invertebrate productivity like other dam removal efforts (Carlson et al. 2017, Morley et al. 2020). However, after sediment loading behind Lake Pillsbury was cleared and the river returned to a more natural state, there would be higher magnitude winter peak flows (~1,200 cfs increase in drier years and ~5,000 cfs increase in wetter years), more frequent winter peak flows, and lower but more variable summer streamflows, and the invertebrate community would be expected to recover rapidly (Carlson et al. 2017). Based on the summarized literature above, there are two proposed mechanisms that could alter benthic macroinvertebrate productivity and subsequently fish growth potential downstream of Scott Dam: (1) food web changes, and (2) benthic macroinvertebrate habitat increases.

4.7.2.1 Food Web Changes

The lower baseflows may warm water temperatures under an unimpaired streamflow scenario with Scott Dam removed (Scenarios 2 and 3). The warmer water temperatures will likely increase the rate at which benthic macroinvertebrates transition through their life history (Uno and Stillman 2020), with effects on the timing of their emergence and potential relevance to fish (Uno and Power 2015) and riparian consumers (Uno 2016). There would also be an increase in the frequency of winter streamflows that scour grazers and allow for higher rates of diatom and filamentous algae production in the spring and summer. The increased stands of diatoms create a rich food source for highly digestible and preferred benthic macroinvertebrates for rearing juvenile salmonids (Power et al. 2015). Jansen et al. (2020) show that algal growth and community structure of invertebrates differ between regulated and unregulated sections of the Eel River. We caution that it is unclear what the streamflow threshold is for scour of invertebrates to occur downstream of Scott Dam, and diatom and algal growth can be limited by multiple factors beyond the abundance of grazers (e.g., light limitation, water temperature, streamflow). Additionally, the Eel River from Scott Dam to Cape Horn Dam may not be directly comparable to the South Fork Eel River, where many of the food web studies mentioned above have been conducted.

Spill and discharge from dams can provide lentic food components to downstream consumers, such as zooplankton, which are easily captured and nutrient rich (Lieberman and Horn 1998; Simpkins and Hubert 2000). If the flux of zooplankton from Lake Pillsbury to the tailwater downstream of Scott Dam is greater than the benthic macroinvertebrates production downstream of Scott Dam, it may currently represent an important food source for juvenile fish, especially during winter when benthic macroinvertebrates' growth is low. Removal of Scott Dam would remove any current zooplankton source from Lake Pillsbury. Additional data on the concentration of zooplankton in the tailwater downstream of Scott Dam and diet/isotopic data on juvenile salmonids is needed to determine if (1) the current flux of zooplankton to the tailwater is significant and (2) if its loss would negatively affect juvenile salmonids.

4.7.2.2 Benthic Macroinvertebrate Habitat Increases

SEC (1998) documented that only 2.6 kilometers (km, 14%) of the total habitat between Scott Dam and Cape Horn Dam is considered riffle habitat. The remaining 16.2 km is pool or run habitat, which is not likely to produce high amounts of invertebrates for foraging fish relative to riffles (Naman et al. 2017). Under the Scenario 3 and Scenario 2 (with Cape Horn Dam alternative diversion), both dams would be removed, and much of this habitat would be returned to a natural series of riffle-pool sequences, which supports benthic macroinvertebrate community diversity and productivity. Similarly, removing dams would allow the full spectrum of substrate sizes to pass and move through the river (Section 3.3), providing more habitat for benthic macroinvertebrates' production.

4.7.3 Eel River Downstream of Cape Horn Dam

Under current conditions, the habitat downstream of Cape Horn Dam supports very little rearing habitat for juvenile salmonids and fewer fish were documented by electrofishing surveys in this reach than those between the two dams (SEC 1998, PG&E 2018a, J. Fuller, National Marine Fisheries Service [NMFS] Fisheries Biologist, pers. comm.). Fewer fish being observed in this section of river is likely a function of water temperature, habitat homogeneity, and non-native predators; thus, it is unclear how important benthic macroinvertebrate changes would be to fish productivity in this reach. In this section we describe the limited site-specific research at Cape Horn Dam and contextualize how the different Water Supply Scenarios may elicit a response in benthic macroinvertebrates.

4.7.3.1 Streamflow-Driven Changes

With Scott Dam in place (Baseline and Scenario 4B), downstream of Cape Horn Dam, winter and spring storms are truncated by up to 2,000 cfs during drier years and 4,000 cfs in wetter years. There is potential for similar scour-driven changes to algal growth and benthic macroinvertebrate community structure described in Section 4.7.2.1.

Jansen et al. (2020) showed that even under high streamflow conditions in 2017, when the percent difference between unimpaired streamflow sites on the Middle Fork Eel River and impaired streamflow at the study site downstream of Cape Horn Dam was low, benthic macroinvertebrate communities differed. The authors posit that the Project influences algal growth and benthic macroinvertebrate community structure, but they were not able quantify a specific streamflow relationship. Additional data or streamflow to invertebrate production experiments would help identify any possible causation of the differences.

The spring recession limb is an important time for benthic macroinvertebrate productivity and fish growth. Site-specific research has suggested that streamflows >80 cfs are needed to maintain riffle habitat for benthic macroinvertebrate production during spring (O'Dowd and Trush 2016). Downstream of Cape Horn Dam, during wetter years, this criterion is met by nearly all Water Supply Scenarios, while during drier years, Baseline and Scenario 4B hold the 80 cfs criterion longer than other Scenarios (Figure 3-15), suggesting less benthic macroinvertebrate productivity under Scenario 2 and 3 during drier years.

4.7.4 Russian River Downstream of the West Branch Confluence

In general, Scenarios 2 and 3 have lower streamflow than Baseline and Scenario 4B (Section 3.1). One possible effect is that during the driest years, extremely low summer streamflows under Scenario 3 may reduce habitat connectivity and wetted channel area, decreasing delivery and production of benthic macroinvertebrates. However, this situation is likely incredibly rare and unlikely to be significant. Overall, no significant change to benthic macroinvertebrate productivity is anticipated among the different scenarios.

4.7.5 Summary

Benthic macroinvertebrates are the main source of food and growth potential for juvenile salmonids. Feasibility Study Alternatives and Water Supply Scenarios may positively or negatively influence benthic macroinvertebrate community structure and abundance depending on water year type and location. Downstream of Scott Dam, Scenario 2 and Scenario 3 have potential to (1) increase the benthic macroinvertebrate diversity between the two dams through increased winter scour streamflows and (2) alter benthic macroinvertebrate habitat by creating additional riffle-pool sequences that provide benthic macroinvertebrate productivity and foraging habitat for salmonids. Downstream of Cape Horn Dam, lower streamflows associated with Scenarios 2 and 3 may reduce riffle connectivity and limit benthic macroinvertebrate production in the spring, but higher winter flows would support natural food web progressions. On the Russian River, significant changes to benthic macroinvertebrate diversity or production are not anticipated under any Water Supply Scenarios.

4.8 Riparian Habitat

Vegetation along the Eel River is sparse within the canyon walls upstream of Lake Pillsbury. Conifers and woody deciduous trees are rare (Figure 4-27). Deciduous shrubs occasionally occur on broad, expansive gravel bars. Plants that may establish along the low streamflow channel are frequently scoured and cannot survive annual patterns of floods and low summer and fall streamflow. Woody plant recruitment occurs within a narrow window of opportunity that is created through annual desiccation, large channel maintaining floods, and smaller floods that annually scour seedlings away from the low water channel margin.

Downstream of Scott Dam and Cape Horn Dam, there is a noticeable increase in conifers and deciduous woody plants within the canyon walls (Figure 4-28). Regulated streamflow releases are higher during the summer months and moderate size floods are smaller in magnitude relative to unimpaired streamflows (Section 3.1). The long-term effect of streamflow regulation has been woody plant encroachment downstream of Scott Dam.

Vegetation encroachment occurs when plant establishment progresses toward the low water channel after streamflow impairment or during an extended drought period. A reduction in high streamflows reduces flood scour and inundation-induced riparian mortality, which allows riparian vegetation to initiate and survive in channel locations that would normally be scoured by floods. When plants establish along a stream or river margin and are not removed, the channel loses its dynamic ability to rework its bed and banks, thus becoming static (Thorne et al. 1996). Regardless of the magnitude of hydrologic impairment on river streamflow fluctuations, an increase in vegetation is often documented in regulated rivers (Gordon and Mentemeyer 2006, Bejarano and Sordo-Ward 2011, Casado et al. 2016).

Vegetation encroachment is often considered to be a negative aspect of streamflow regulation. Effects that have been associated with vegetation encroachment include: channel narrowing; bank stabilization; reductions in wetted edge length; channel bed coarsening; changes in bed form variation; changes in location and abundance of grasses, forbs, shrubs, and trees; and changes to the food web structure (Power et al. 1996, DeWine and Cooper 2007, Vesipa et al. 2017; Bejarano et al. 2020). Overall, the environmental changes associated with vegetation encroachment have led to reduced variability in biodiversity and habitat structure (i.e., functional complexity) where it has been documented. The effects of vegetation encroachment may increase depth and velocity across a range of discharges, reduce flood capacity, and decrease aquatic habitat and riparian area (Shafroth et al. 2002, Gordon and Mentenmeyer 2006; Casado et al. 2016).

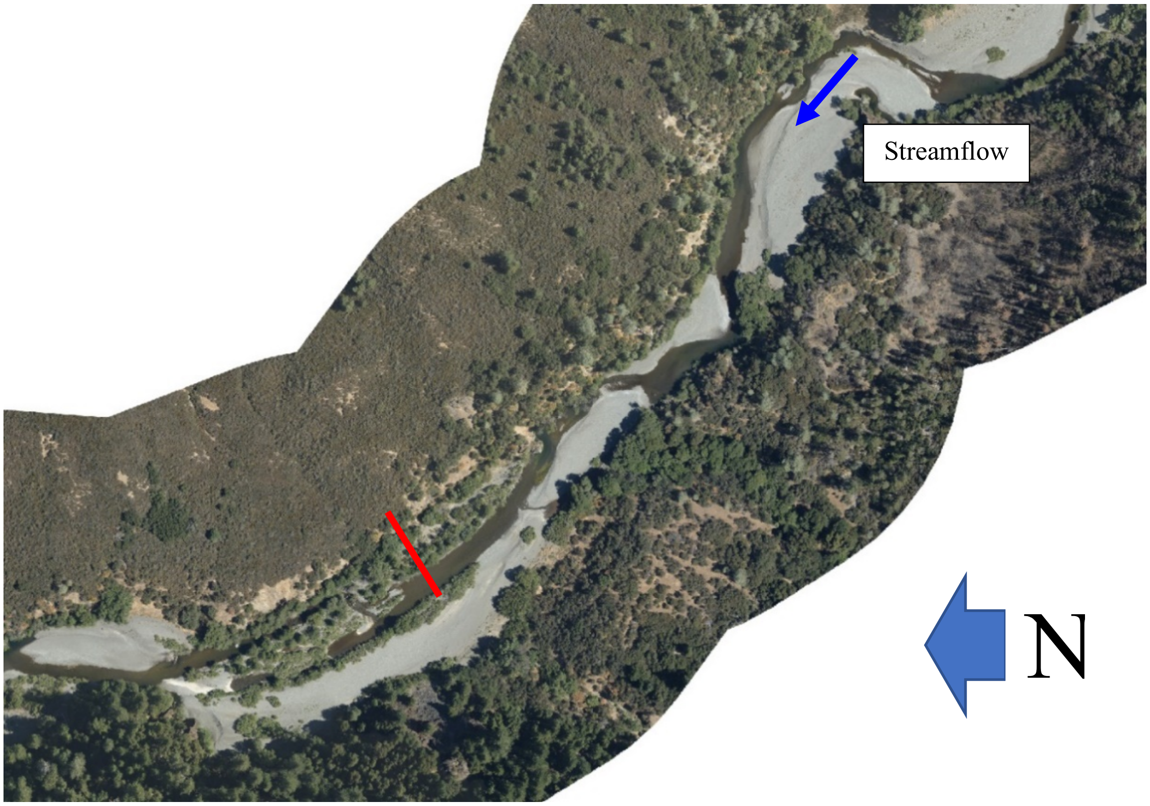

Figure 4-27. Eel River riparian vegetation and channel pattern upstream of Lake Pillsbury. For reference, the red line is approximately 230 feet long.

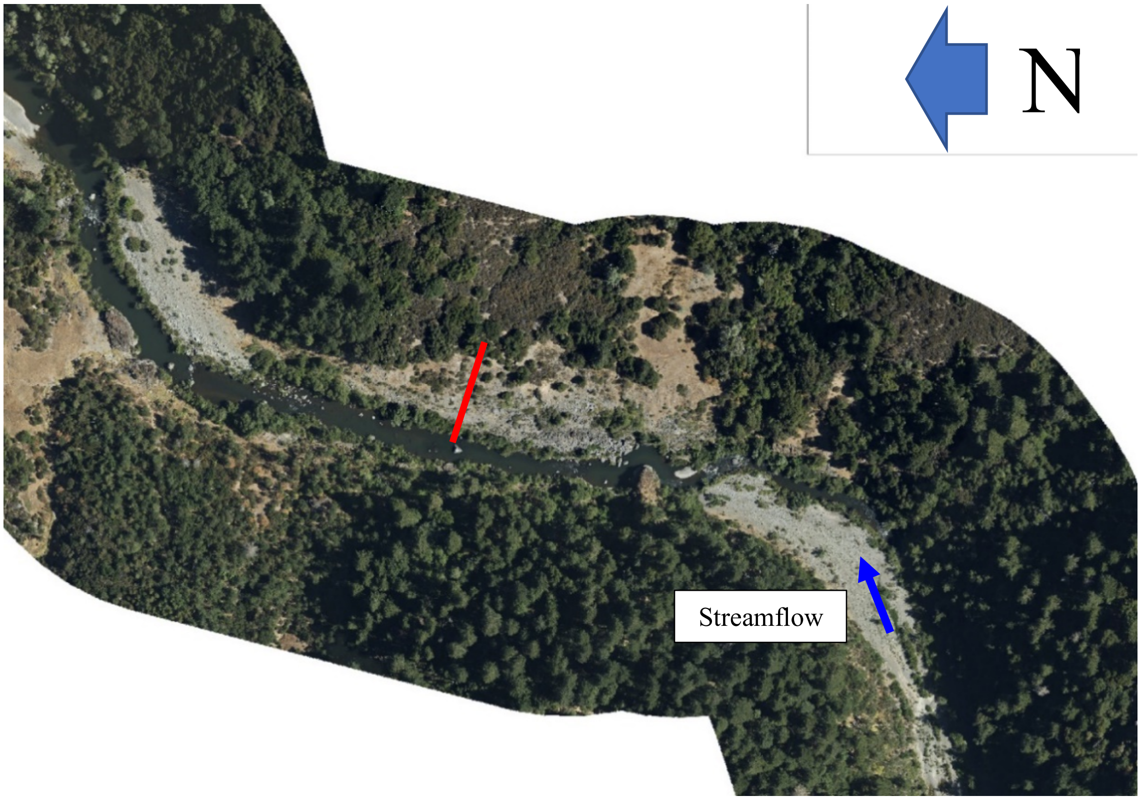

Figure 4-28. Eel River riparian vegetation and channel pattern downstream of Scott Dam. For reference, the red line is approximately 230 feet long.

Successful woody plant recruitment is the result of first year seedling survival and the pattern of floods and drought in the years following. Plant recruitment occurs when a plant germinates, survives through the first growing season (initiation), establishes, and survives to reproduction. Streamflow timing, magnitude, duration, frequency and rate of change influence species selection, bank location, recruitment frequency, and vegetation structure. The successful initiation of cottonwoods and willows relies on the annual cycle of late spring snowmelt floods and the timing and rate of receding water (Bradley and Smith 1986, Scott et al. 1993). The successful initiation of ashes, alders, and maples is tied to winter storm patterns and stable but elevated streamflow between rainfall-driven flood events. After the first growing season, successful recruitment is the result of surviving bed scour and deposition and continued access to groundwater during the growing season.

Relationships between successful woody plant recruitment, annual and interannual streamflow patterns, and growing proximity to the low water channel are well described in the literature (Bradley and Smith 1986, Kondolf and Wilcock 1996, McBain and Trush 1997, Mahoney and Rood 1998). The pattern of streamflow during and after seed dispersal is the primary factor influencing successful plant initiation regardless of plant species. Streamflow that recedes too quickly can kill a young plant whose roots cannot grow fast enough to keep up with the recession (McBride et al. 1988, Mahoney and Rood 1992, Segelquist et al. 1993). Streamflows can occur in a pattern that will allow one species to successfully recruit, but not another species simply because streamflows are not timed to satisfy the needs of that species' seed dispersal period. The relationship between seed dispersal and streamflow timing, magnitude, duration, rate of change, and frequency are primary processes that drive vegetation patterns along the Eel River.

The window of seedling establishment upper limit is a function of desiccation, the lower limit a function of bed scour. Large floods are important in creating and maintaining seedbeds on floodplains and scour channels and in facilitating seedling germination of some woody plant species higher away from the low water edge, where they would be less susceptible to bed scour. Seedlings that germinate higher on the bank during years of channel maintaining floods run the risk of desiccation. Some plants annually try to grow along the low water edge. Before regulation, seedlings growing along the low water margin were scoured annually or biannually, limiting seedling establishment along the summer low water edge. Before regulation on the Eel River, the window of opportunity between bed scour and desiccation was likely much wider and higher on the channel bank than post-dam. With the completion of Scott Dam and nearly constant summer streamflow, the window of opportunity shifted from the valley margin and upper bars to the low water channel margin.

4.8.1 Approach

We evaluated five focal species: white alder (*Alnus rhombifolia*), arroyo willow (*Salix lasiolepis*), Fremont cottonwood (*Populus fremontii*), shiny willow (*S. lasiandra*), narrowleaf willow (*S. exigua*), and dusky willow (*Salix melanopsis*). The species selected were representative of other plants that are characteristic of Eel River and Russian River riparian vegetation. Therefore, if the life history requirements for the selected plants were met, then the life history requirements of other commonly associated species were also assumed to be met (Lambeck 1997). Current streamflow may support the life history needs of one or more focal species, yet future conditions may not favor them and may favor other focal species. A shift in focal species dominance could create a measurable shift in dominant canopy species and reduce the presence of a homogenous band of encroaching vegetation. We performed four analyses to determine how hydrological changes would affect riparian vegetation in the three different focal reaches (Eel River from Scott Dam to Cape Horn Dam, Eel River downstream of Cape Horn Dam, and Russian River downstream of the WBRR confluence):

1. Changes to flood frequency and magnitude
2. Changes to streamflow recession rate
3. Changes to minimum daily average streamflow
4. Changes to streamflow duration

For each analysis, we evaluated how future riparian vegetation would respond to the Water Supply Scenarios at each of the three hydrologic focal areas (Table 4-3). A primary assumption of the riparian evaluation is that changes in focal mechanisms would create a shift in species recruitment frequency, channel bank location, and/or abundance, and that these changes would be observable at a landscape scale. For example, if future streamflow became more like unimpaired streamflow upstream of Lake Pillsbury, the vegetation patterns observed downstream of Scott Dam would become more reflective of vegetation patterns upstream of Lake Pillsbury over time.

4.8.2 Changes in Flood Frequency and Magnitude

Changes in flood magnitude frequency rates were compared between the four Water Supply Scenarios. The analysis was conducted for the Eel River from Scott Dam to Cape Horn Dam , Eel River downstream of Cape Horn Dam, and the Russian River downstream of the WBRR confluence. This evaluation assumes that changes in large floods (greater than 10-year recurrence), moderate floods (3 to 10-year), and smaller floods (less than 3-year) between Water Supply Scenarios could remove encroaching vegetation and inhibit the re-colonization of gravel bars, reducing the future risk of re-encroachment. Flood frequency analyses were conducted for each Water Supply Scenario at each location using the annual daily average stream maximum for the 106-year hydrologic period of record. The annual daily average maximum was sorted largest to smallest and summary flood frequency statistics calculated. For this analysis, we relied on the flood frequency figures in Section 3.2; results from these figures were similar between the Eel River from Scott Dam to Cape Horn Dam and the Eel River downstream of Cape Horn Dam, thus they were grouped.

Table 4-3. Water Supply Scenarios used to evaluate future riparian vegetation response.

River	Focal Areas	Water Supply Scenarios Considered
Eel River	From Scott Dam to Cape Horn Dam	Baseline
		Scenario 2/3
		Scenario 4B
	Downstream of Cape Horn Dam	Baseline
		Scenario 2
		Scenario 3
		Scenario 4B
Russian River	Downstream of the West Branch Russian River Confluence	Baseline
		Scenario 2
		Scenario 3
		Scenario 4B

4.8.2.1 Eel River from Scott Dam to Cape Horn Dam and Eel River Downstream of Cape Horn Dam

The flood frequency results for the Eel River from Scott Dam to Cape Horn Dam and the Eel River downstream of Cape Horn dam were similar (see Section 3.1). The biggest changes between Baseline, Scenario 3, and Scenario 2 for the Eel River from Scott Dam to Cape Horn Dam and the Eel River downstream of Cape Horn Dam were an approximately 10 percent increase in the 5 to 10-year flood magnitudes above Baseline and an approximately 40 percent increase in the 1.5-year flood magnitude. Increases in the 1.5- to 10-year flood magnitudes would mean that the effective discharge and channel-forming floods will be reached more often than they are under the Baseline and Scenario 4B. A magnitude increase in more frequently recurring floods leads to an increase in bank erosion, channel migration in wider reaches, removal of encroaching mature vegetation, and overall decrease of vegetation establishing within the canyon walls downstream of Scott and Cape Horn Dams.

4.8.2.2 Russian River Downstream of the West Branch Confluence

On the Russian River, flood frequency changes between Baseline, Scenario 2, and Scenario 3 are most pronounced in the 2- to 6-year flood magnitudes (Section 3.1). There is an approximately 20 percent increase in the 2- to 3-year flood magnitudes above Baseline and an approximately 40 percent increase in the 1.5-year flood magnitude. Scenario 4B and Scenario 2 increase the 1.5-year flood magnitudes 15 percent to 20 percent, which would mean that the effective discharge and channel-forming floods will be reached more often than they are under Baseline. However, the 1.5-year flood magnitude is decreased in Scenario 3 (~3% decrease in flood magnitude). A magnitude increase in more frequently recurring floods in the Scenarios 4B and 2 could lead to an increase in bank erosion, channel migration in wider reaches, removal of encroaching mature vegetation, and an overall decrease in establishing vegetation along annual low water channel margins downstream in reaches downstream of the WBRR confluence.

4.8.3 Changes in Streamflow Recession Rates

Changes in streamflow recession rates that influence new riparian establishment were compared between Water Supply Scenarios. This evaluation assumes that woody plants growing in the riparian corridor are vulnerable to changes in streamflow recession rates during seed dispersal and the remainder of the growing season. Scenario 2 and Scenario 3 were compared to Baseline for the Eel River from Scott Dam to Cape Horn Dam and the Eel River downstream of Cape Horn Dam. The hydrology of the modified Scenario 4B is similar to Baseline and was not compared here because we assumed that the change would be similar. The changes to streamflow recession rates were similar between the Eel River from Scott Dam to Cape Horn Dam and the Eel River downstream of Cape Horn Dam, and thus are presented together.

Streamflow was converted into river water surface elevations and daily changes in elevation were calculated for each year. River elevation data were summarized using the minimum, maximum, median, average, 25th percentile, and 75th percentile stage change for each month within each year of the 107-year period of record. For a young seedling to survive, streamflow recession should occur at a rate less than or equal to the maximum seedling root growth rate. Survival has been shown to decrease when streamflow elevations recede faster than 0.10 ft/day for cottonwoods and at 0.03 ft/day for willows (Mahoney and Rood 1991, Segelquist et al. 1993, Mahoney and Rood 1998, Amlin and Rood 2002, Stella et al. 2010). River elevation recessions greater than -0.10ft/day were considered the maximum survivable recession. The number of times that streamflow recession exceeded the maximum survivable recession rate (-0.10 ft/day) were counted and box and whisker charts constructed to illustrate whether differences were significant between Water Supply Scenarios. Data was not sorted by water year class, which may illustrate greater contrast in streamflow management changes.

4.8.3.1 Eel River from Scott Dam to Cape Horn Dam and the Eel River Downstream of Cape Horn Dam

The analysis for the Eel River from Scott Dam to Cape Horn Dam suggests that there will be an increase in the frequency of daily average stage recessions exceeding 0.10 ft/day in all months except September under Scenario 2 and Scenario 3. The box and whisker charts for the Eel River from Scott Dam to Cape Horn Dam suggest that even with changes in frequency of streamflow recession rates greater than -0.10 ft/day, the change is not significantly different from Baseline (Figure 4-29).

The analysis for the Eel River downstream of Cape Horn Dam suggests that there will be an increase in the frequency of daily average stage recessions exceeding 0.10 ft/day in all months except May and June with Scenario 2 and June with Scenario 3. The box and whisker charts for the Eel River downstream of Cape Horn Dam suggest that even with changes in frequency of fatal streamflow recession rates, the change is not significantly different from Baseline (Figure 4-30).

4.8.3.2 Russian River Downstream of the West Branch Confluence

Upon preliminary examination of the hydrographs, there were no changes to streamflow that would represent a significant number of 0.10 ft/day changes in elevation. Based on visual hydrograph analysis, we anticipate minimal changes to riparian zone recruitment across the different Water Supply Scenarios in this reach.

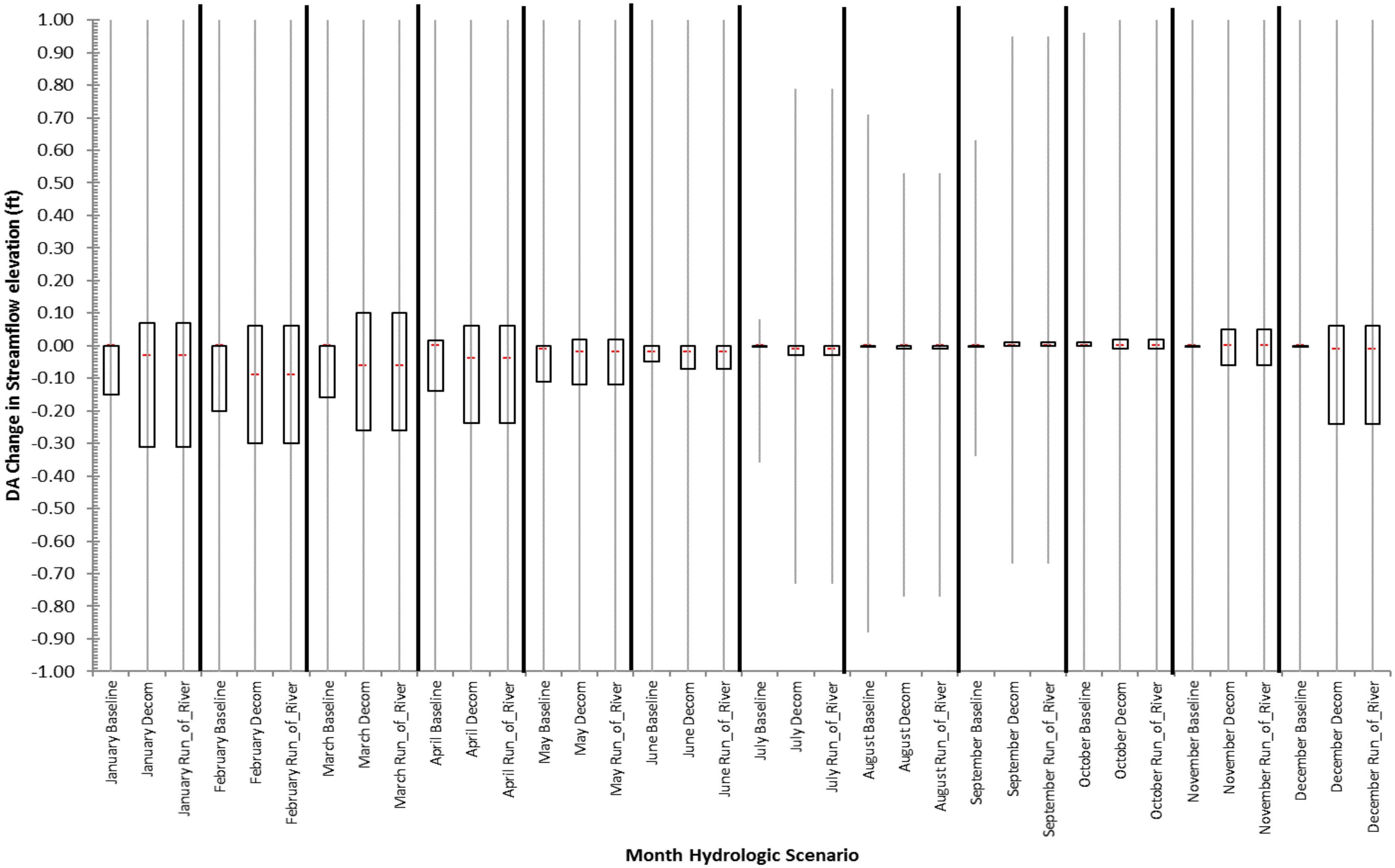

Figure 4-29. Changes in daily average (DA) streamflow recession rates for the Eel River from Scott Dam to Cape Horn Dam for each Water Supply Scenario. Scenario 3 and Scenario 2 results are the same for the Eel River from Scott Dam to Cape Horn Dam (both unimpaired).

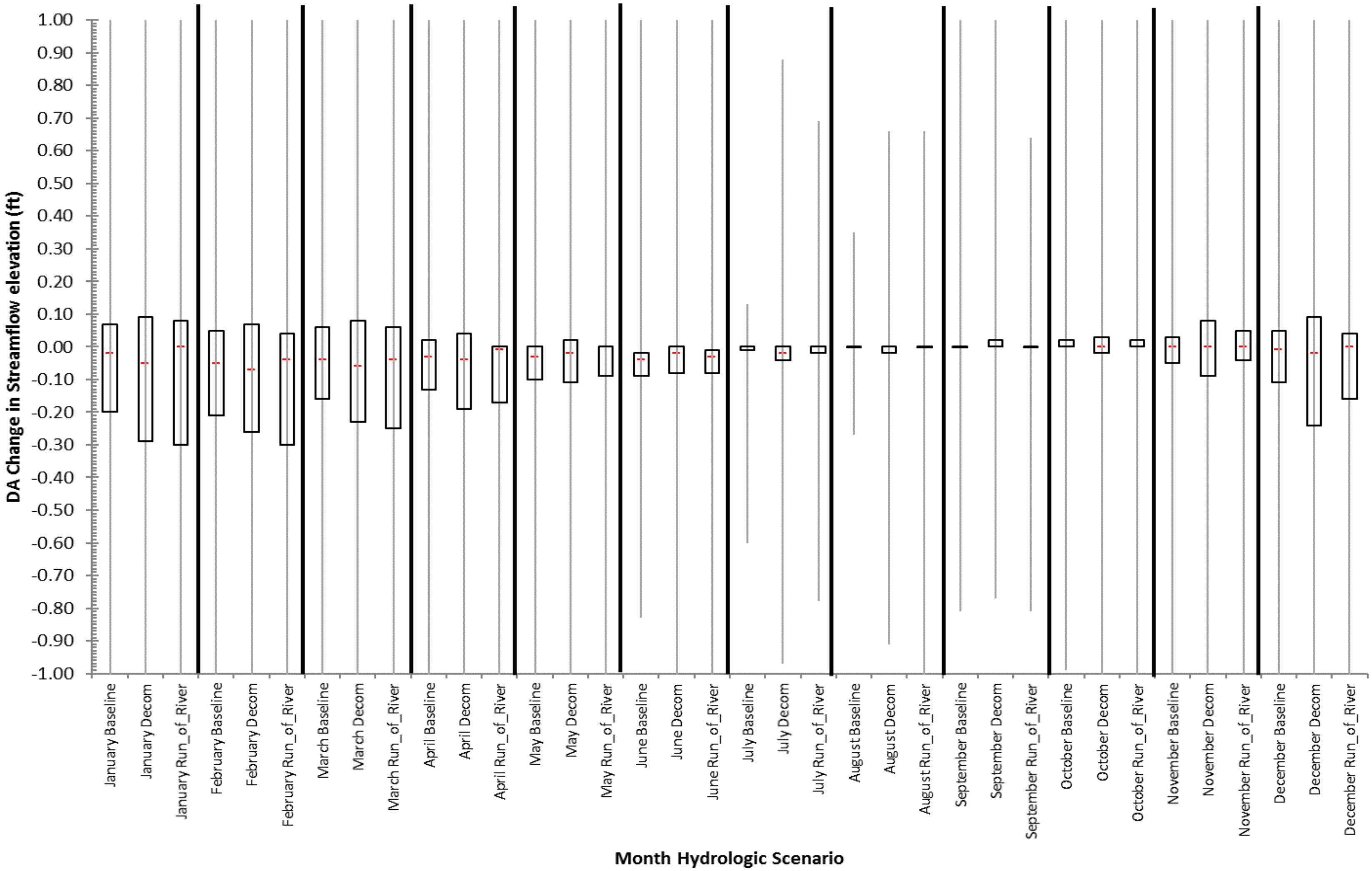

Figure 4-30. Changes in daily average (DA) streamflow recession rates for the Eel River downstream of Cape Horn Dam for each Water Supply Scenario.

4.8.4 Changes in Minimum Daily Average Streamflow

An evaluation of minimum streamflow was conducted using the minimum annual daily average streamflow for the daily average discharge data period of record. Annual minimum streamflow typically occurs during summer months. The minimum streamflow evaluation assumes that woody plants that disperse seeds in the summer will be susceptible to changes in streamflow minimums and potentially establish at different locations on the riverbank. Lower minimum streamflow magnitudes can result in reduced risk of future encroachment because seeds are dispersed within areas that are likely to be scoured by higher streamflows in the water. The annual minimum daily average streamflow was extracted from each year and sorted largest to smallest for the period record. Annual daily average minimum data were summarized using the minimum, maximum, median, average, 25th percentile, and 75th percentile stage change for each month within each year of the 107-year period of record for each Water Supply Scenario. Scenario 4B, Scenario 2, and Scenario 3 Water Supply Scenarios were compared to Baseline for the Eel River from Scott Dam to Cape Horn Dam, the Eel River downstream of Cape Horn Dam, and the Russian River downstream of the WBRR confluence.

Several Water Supply Scenarios did not have a significant change in streamflow. There is not a significant difference between the Scenario 4B and Baseline for the Eel River from Scott Dam to Cape Horn Dam or the Eel River downstream of Cape Horn Dam. On the Russian River, there is no significant difference between Scenario 2 and Baseline on the Russian River. No change over existing conditions is expected between these Water Supply Scenarios.

4.8.4.1 Eel River from Scott Dam to Cape Horn Dam and the Eel River Downstream of Cape Horn Dam

Average streamflows under Scenario 2 and Scenario 3 are the same and are significantly lower than the Scenario 4B and Baseline (Figure 4-31). Under lower unimpaired streamflow (Scenario 2 and Scenario 3), narrowleaf willow seedlings would initiate lower in the channel downstream of Scott Dam where they would be more vulnerable to flood scour in the water, thereby resulting in future reduction of encroachment risk. The difference in the minimum daily average streamflow downstream of Cape Horn Dam is similar to of the Eel River from Scott Dam to Cape Horn Dam, suggesting similar changes to riparian encroachment (Figure 4-32).

4.8.4.2 Russian River Downstream of the West Branch Confluence

On the Russian River, minimum daily average streamflows under Scenario 2 and Scenario 3 are lower than Baseline and Scenario 4B. Under Scenario 2, narrowleaf willow and dusky willow seedlings could initiate lower in the channel where they would be more vulnerable to winter flood scour, thereby resulting in a reduction in future riparian encroachment risk.

Minimum daily average streamflows on the Russian River under Scenario 3 are lower than the Baseline and Scenario 2 (Figure 4-33). Under Scenario 3, narrowleaf willow and dusky willow seedlings would initiate lower in the channel relative to Scenario 2 and Baseline. It is possible that Scenario 3 would not result in a significant reduction in future encroachment risk.

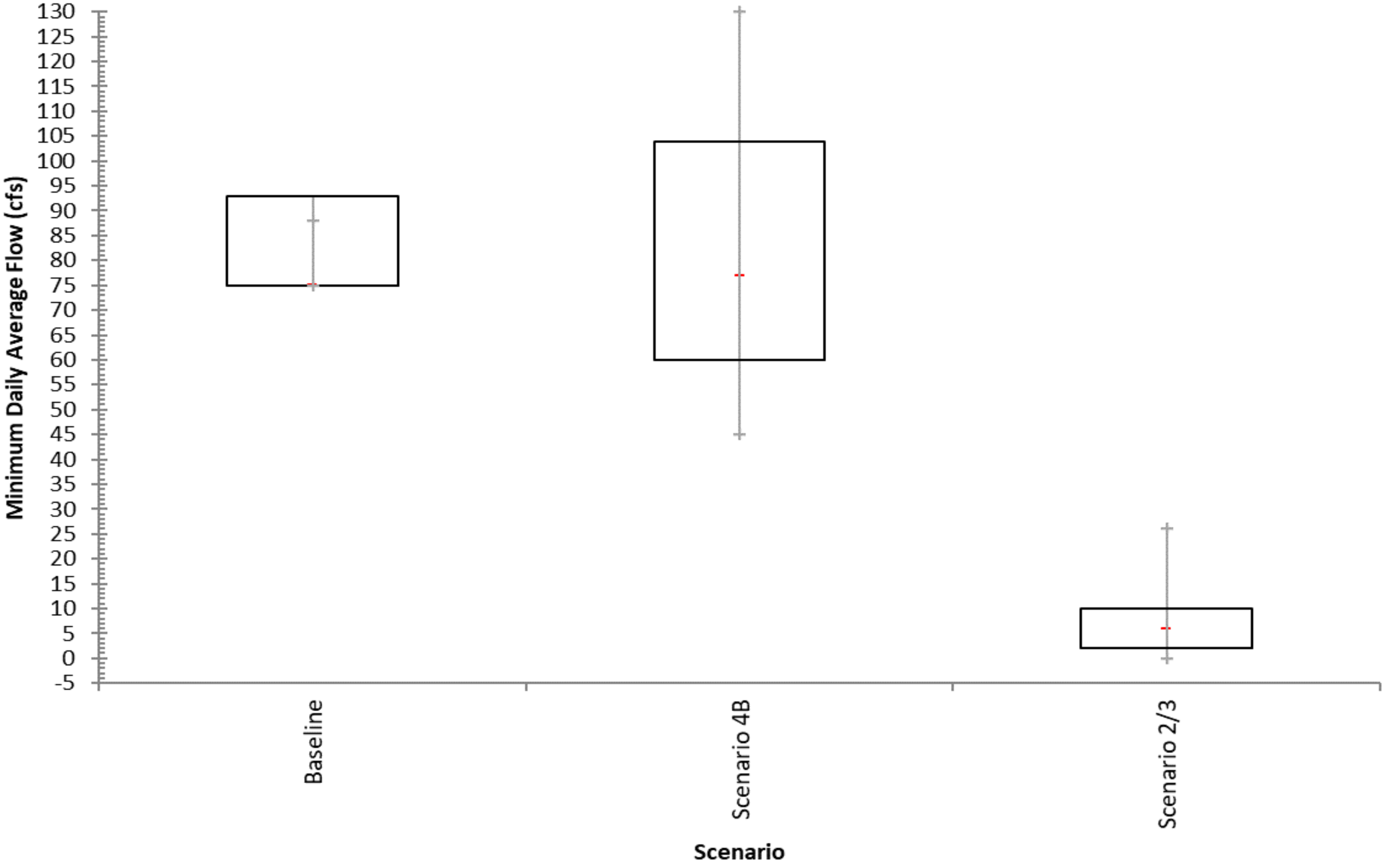

Figure 4-31. Annual minimum daily average streamflow on the Eel River from Scott Dam to Cape Horn Dam in the Baseline, Scenario 4B, and Scenario 2/3 Water Supply Scenarios (n=107 years).

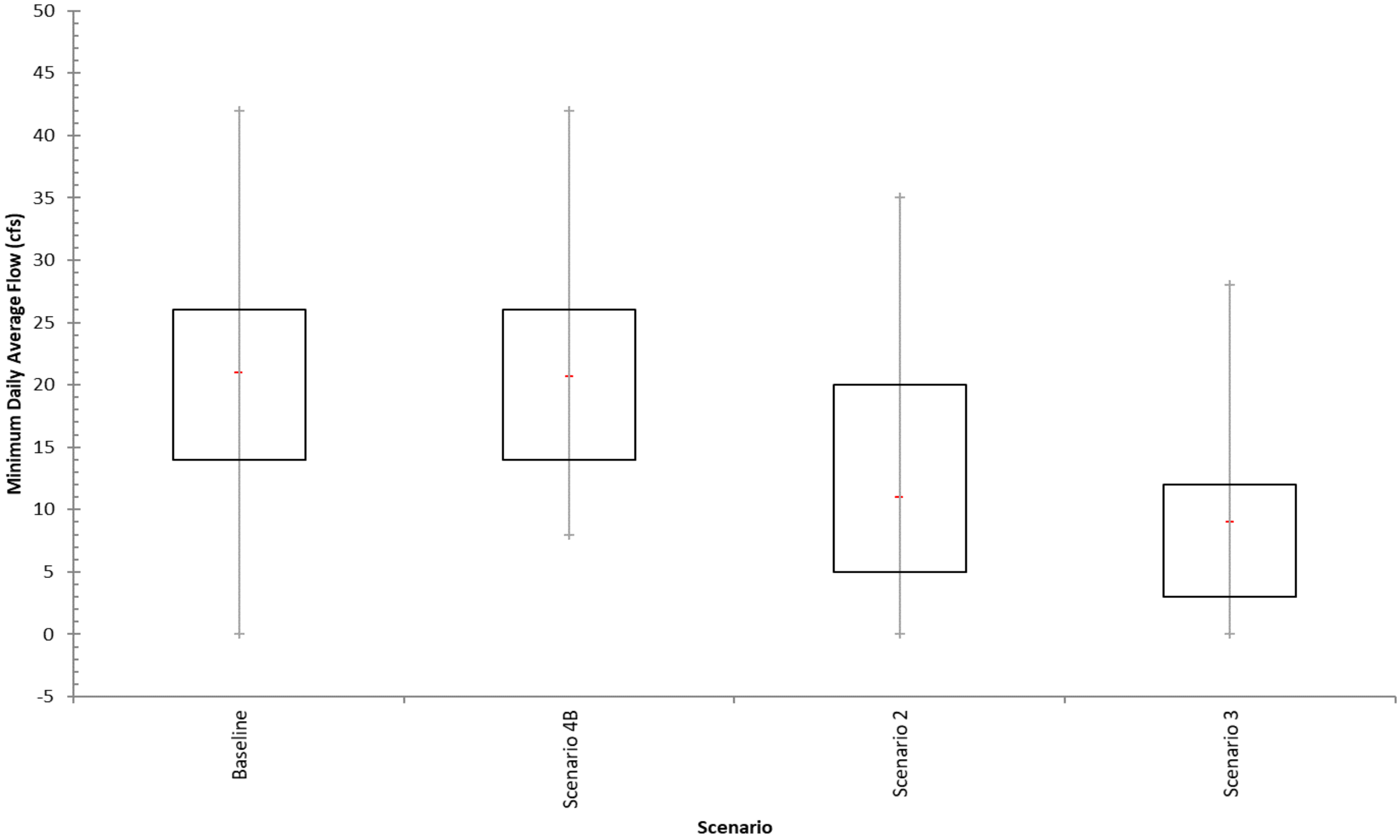

Figure 4-32. Annual minimum daily average streamflow on the Eel River downstream of Cape Horn Dam in the Baseline, Scenario 4B, Scenario 2 and Scenario 3 hydrologic modeling outputs (n=107 years).

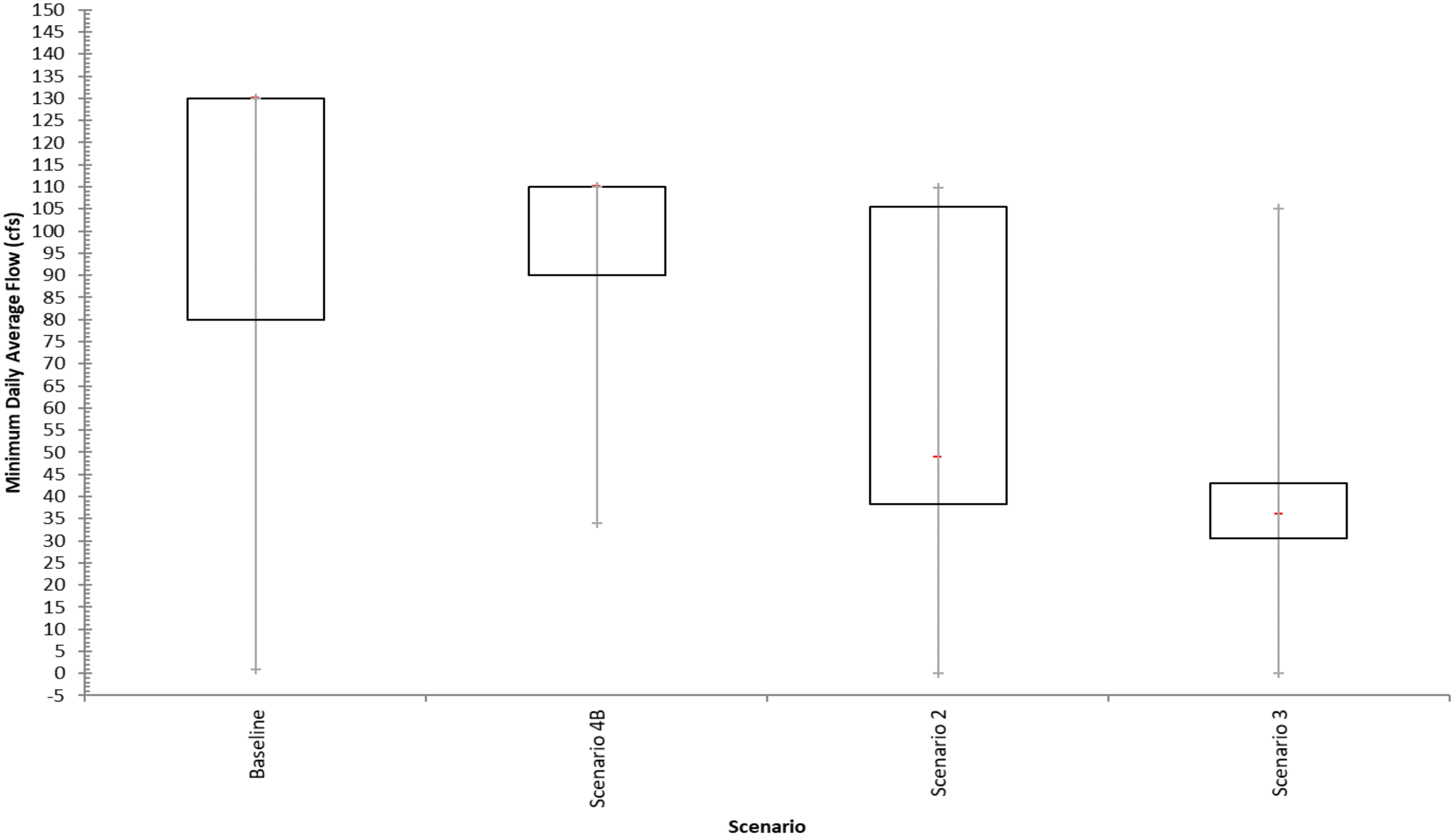

Figure 4-33. Annual minimum daily average streamflow on the Russian River downstream of the West Branch Russian River confluence in Baseline, Scenario 4B, Scenario 2, and Scenario 3 hydrologic modeling outputs (n=107 years).

4.8.5 Changes in Streamflow Duration

Differences in the duration of streamflow during the summer was evaluated between Water Supply Scenarios. The summer streamflow duration evaluation assumes that woody plants that disperse seeds in the summer will be susceptible to changes in streamflow minimums and potentially establish at different locations on the riverbank. Narrowleaf willow and dusky willow disperse seeds during the summer months and can take advantage of moist substrate along the water edge. Constant summer streamflow magnitudes facilitate the establishment of woody species at one bank location/elevation. Variable summer streamflow that continues to recede facilitates woody plant establishment at a range of bank locations and promotes young seedling initiation lower in the channel, where they would be more prone to scour mortality in the winter.

Days with the same consecutive streamflow during the summer months were counted and compared between Water Supply Scenarios. Scenario 4B, Scenario 2, and Scenario 3 were compared to Baseline for the Eel River from Scott Dam to Cape Horn Dam, the Eel River downstream of Cape Horn Dam, and the Russian River downstream of the WBRR confluence.

4.8.5.1 Eel River from Scott Dam to Cape Horn Dam

Summer streamflow under the Baseline for the Eel River from Scott Dam to Cape Horn Dam is nearly constant for 65 consecutive days in most years. Baseline provides stable and constant soil moisture along the wetted channel edge, creating favorable conditions for narrowleaf willow and dusky willow encroachment.

Summer baseflows for the Eel River from Scott Dam to Cape Horn Dam under Scenario 4B are constant for 30 to 65 days during the July 15 to September 15 period. In some years, there is an increase in streamflow magnitude after the period of constant streamflow. Increased streamflow after a period of constant streamflow could inundate and drown young seedlings if the water is deep enough to completely inundate seedlings, which could reduce encroachment. For the Eel River from Scott Dam to Cape Horn Dam, summer streamflow under the Scenario 2 and Scenario 3 continues to recede throughout the summer into fall. Scenario 2 and Scenario 3 are most likely to promote a variable riparian fringe along the water edge and inhibit re-encroachment in the future.

4.8.5.2 Eel River Downstream of Cape Horn Dam

Summer streamflow under Baseline for the Eel River downstream of Cape Horn Dam is nearly constant for 75 days (July 15 to September 15) with 3 cfs or less variation during the entire 75-day period. Baseline provides stable and constant soil moisture along the wetted channel edge, creating favorable conditions for narrowleaf willow and dusky willow encroachment.

Summer streamflow for the Eel River downstream of Cape Horn Dam under the Scenario 4B is near constant for 45 to 70 days during the July 15 to September 15 period with 4 cfs or less of variation. In some years, there is an increase in streamflow after the period of constant streamflow. Increased streamflow after a period of constant streamflow could inundate and drown young seedlings if the water is deep enough. It is possible that the period of inundation after the period of constant streamflow could cause enough mortality to inhibit narrowleaf willow and dusky willow from encroaching the summer low water channel margins.

Summer streamflow for the Eel River downstream of Cape Horn Dam under Scenario 2 is near constant for 30 to 50 days in the late summer and early fall. The period of constant streamflow occurs after narrowleaf willow and dusky willow disperse seeds, but constant streamflow at the same discharge would sustain soil moisture and provide suitable growing conditions for seedlings that established earlier in the summer. There would be no apparent difference in riparian response between Baseline, Scenario 4B, and Scenario 2 for the Eel River downstream of Cape Horn Dam.

Summer streamflow for the Eel River downstream of Cape Horn Dam under Scenario 3 continues to recede throughout the summer into fall (i.e., there is no period of constant streamflow during the summer months). Scenario 3 is the most likely to inhibit re- encroachment in the future, but would likely have the least amount of riparian vegetation within the active channel.

4.8.5.3 Russian River Downstream of the West Branch Confluence

On the Russian River downstream of the WBRR confluence, summer streamflow in all Water Supply Scenarios has high daily variation moving from early spring into late summer and fall. It is unlikely that there is any significant difference in summer streamflow duration/constancy between Water Supply Scenarios. All Water Supply Scenarios on the Russian River downstream of the WBRR confluence are likely to promote a variable riparian fringe along the low flow water edge, and the reduced flood regime caused by Lake Mendocino will likely continue to foster future riparian encroachment.

4.8.6 Summary

The different Water Supply Scenarios and Feasibility Study Alternatives will cause varying riparian vegetation responses. The most significant changes will be above Scott Dam within the current inundation limits of Lake Pillsbury and from Scott Dam to Cape Horn Dam under a Scott Dam removal scenario. The removal of Scott Dam would restore the current channel that is under Lake Pillsbury and patchy riparian vegetation is likely to return after the channel stabilizes. For the Eel River from Scott Dam to Cape Horn Dam, a return to unimpaired flow would result in scour and desiccation of the existing vegetation that has encroached on the channel. The resulting vegetation community after Scott Dam removal is likely to transform to patchy vegetation patterns currently observed above Lake Pillsbury. Minor changes in riparian vegetation are expected for the Eel River downstream of Cape Horn Dam, as tributary accretion added to mainstem river flows currently inhibit riparian encroachment. Closer to Cape Horn Dam, riparian vegetation conditions are expected to change, and the extent of change will get less with increasing distance downstream from Cape Horn Dam. There are no major changes predicted to riparian vegetation on the Russian River, as flows from all Water Supply Scenarios are anticipated to be minimal.

4.9 Cyanobacteria and Algal Growth

Cyanobacteria (blue-green algae) naturally occur in the upper Eel River (Bouma-Gregson et al. 2018) and are considered a significant threat to water quality and freshwater ecosystem function because they can produce toxins that harm wildlife and human health (Heisler et al. 2008, O'Neil et al. 2012, Breinlinger et al 2020). It is unclear what effect the Project has on cyanobacterial blooms in the mainstem Eel River, but unpredictable, and potentially negative interactions between cyanobacteria, water quality and higher order organisms occur in other freshwater systems (Breinlinger et al. 2020). Dams in general facilitate cyanobacterial blooms through stable and reduced flow regimes downstream, creation of artificial lentic habitats that have low light attenuation and long residence times, trapping of nutrients, and promotion of hypolimnetic anoxia

(Burford et al. 2007). Observations of cyanobacterial blooms and detections of their toxins have been observed in the upper Eel River assessment area (Figure 4-34, Asarian and Higgins 2018, Jansen et al. 2020). The objective of this section is to assess how different Water Supply Scenarios and Feasibility Study Alternatives may affect cyanobacterial and other algal blooms because their growth (1) could vary with Project operations and infrastructure, (2) is a public and animal health issue, (3) is an aesthetic and recreational issue, (4) serves as an ecological indicator, and (5) is an important part of the ecological community in the Eel River (Power et al. 2015).

Figure 4-34. Algal bloom from Lake Pillsbury, photo taken in September 2018.

4.9.1 Eel River from Scott Dam to Cape Horn Dam (gage E-2)

The limnology of Lake Pillsbury affects biological and ecological processes on the Eel River downstream of Scott Dam. Hypolimnetic water released from Lake Pillsbury is cooler than unimpaired streamflow and may have increased nutrient concentrations. Released water from Lake Pillsbury has a higher magnitude and is more stable than unimpaired conditions during summer, but results in lower magnitude winter floods.

The release of higher summer baseflows that are cooler than unimpaired streamflows should reduce the probability of algal blooms. Algal sample collections from the Trout Creek Campground (located between Scott Dam and Cape Horn Dam) showed that cyanobacteria are present and had the highest concentration of microcystin toxins (microcystin) collected throughout the entire Eel River watershed in 2016 (Asarian and Higgins 2018). The detection of microcystin is surprising because taxa that produce it do not commonly grow in lotic systems that have cooler water temperatures (Keith Bouma-Gregson, California State Water Board, pers. comm.). Asarian and Higgins (2018) suggest two possible explanations: (1) direct release of microcystin from Lake Pillsbury, or (2) the presence of *Phoridium,* which is a benthic algal species that thrives in stable streamflow conditions (Wood et al. 2017) and has been observed in high concentrations at the Trout Creek site in the Eel River (Asarian and Higgins 2018). *Phoridium* typically releases anatoxins (not the toxin detected at Trout Creek) but has also been observed to release microcystin in the South Fork Eel River (Bouma-Gregson et al. 2018). Thus, it is unclear what mechanisms drove the 2016 bloom, nor have any algal samples been taken since. Additional sampling for toxins across water years would help identify if there is a meaningful effect of the Project on cyanobacterial blooms in this section of river.

By removing Scott Dam and Lake Pillsbury, Scenario 2 and Scenario 3 would eliminate the flux of any direct toxins to the Eel River downstream of Scott Dam. Similarly, higher magnitude winter events may reduce the probability of widespread and dense *Phoridium* blooms, which thrive in stable streamflow conditions. Under Baseline and Scenario 4B, there would be no meaningful changes to streamflow and no infrastructure changes thus algal growth conditions would be similar to what is observed today.

4.9.2 Eel River Downstream of Cape Horn Dam (gage E-11)

It is currently unclear how the retention of Cape Horn Dam and Van Arsdale Reservoir would affect algal blooms. Reduced and stable streamflow has been hypothesized to facilitate cyanobacterial algae blooms in the South Fork Eel River, but the blooms also occur naturally (Power et al. 2015). Specifically, it is posited that lower winter peak flows and stable summer streamflow could promote larger and more frequent blooms of cyanobacteria (Power et al. 2015). The hypothesis is based on work from the South Fork Eel River, and it remains unclear if the lower winter flows or more stable streamflows downstream of Cape Horn Dam would replicate those mechanisms because they have not been exclusively tested in the upper Eel River. However, Jansen et al. (2020) detected more toxin-producing cyanobacterial genera downstream of Cape Horn Dam than in the Middle Fork Eel River, which is not impounded. Further downstream, microcystin toxins were detected downstream of Outlet Creek in 2014 and upstream of Outlet Creek when resampled in 2016 (Asarian and Higgins 2018, Bouma-Gregson et al. 2018). Interestingly, toxins were only detected downstream of Outlet Creek in 2014 (drier year) and not detected in 2016 (wetter year), which suggests that the higher streamflows downstream of Cape Horn Dam in 2016 may reduce the probability of blooms in some years.

Downstream of Cape Horn Dam, winter floods will likely be highest for Scenario 3 and lower under Scenario 2, Scenario 4B, and Baseline by 30 percent in drier water years. Scenario 4B and Baseline have higher summer flows in wetter years. If mechanisms present on the South Fork Eel River also function similarly downstream of Cape Horn Dam, then the higher winter flows under Scenario 3, specifically in drier years, would reduce the probability of blooms. Alternatively, the higher summer streamflows under Baseline and Scenario 4B may currently reduce algal bloom probability, but it is unclear how higher and more stable flows interact with algal growth. We also caution that many variables that could impact algal blooms independent of Project operations (e.g., nutrient concentration) that were not available to be analyzed.

4.9.3 Russian River Downstream of the West Branch Confluence

Cyanobacterial blooms do not typically occur in the Russian River downstream of the WBRR confluence, which is likely due to the relatively high streamflows (>100 cfs) released year-round from Coyote Valley Dam. While it is possible that releases may have high nutrient concentrations, they do not commonly manifest into cyanobacterial blooms. Under Scenarios 2 and 4B, streamflow would decrease by 20 to 50 cfs downstream of Coyote Valley Dam. Given the limited information, there is no indication that streamflow would be reduced to a point where the river would become fractionated and create pools that could promote harmful algal blooms.

4.9.4 Summary

Cyanobacterial blooms occur naturally throughout the Eel River watershed, but Project operations, along with other variables, have the potential to promote or reduce blooms. Scenarios 2 or 3 are likely to reduce the probability of algal blooms downstream of Scott Dam because removal of the reservoir would eliminate any flux of any cyanobacteria toxins produced in the reservoir to the downstream river. In addition, the increased winter flow magnitudes under Scenario 2 and Scenario 3 would result in higher and more variable flows compared to Baseline and Scenario 4B. Downstream of Cape Horn Dam, higher winter flows under Scenario 3 may decrease algal growth, specifically in drier years, but Scenario 4B and Baseline have higher summer flows in wetter years that could decrease algal growth. A more comprehensive data set using multiple year and site data collection would help determine if and how the Project influences cyanobacterial blooms. The dynamics which trigger blooms are difficult to understand in the upper Eel River without specific studies designed to examine them, thus the potential changes described here come with uncertainty.

5 SALMONID PRODUCTIVITY TRADEOFFS

This evaluation is intended to provide a general understanding of the expected tradeoffs between Dam(s) Removed (Scenario 2 and Scenario 3) and Dams Remain (Baseline and Scenario 4B) (Table 1-1) in terms of potential anadromous salmonid production for the upper Eel River. The evaluation was separated into two assessments focusing on affected reaches of the upper Eel River upstream and downstream of Scott Dam, since the factors affecting anadromous salmonid populations and availability of data to inform these assessments differ between areas. The two areas include: (1) stream reaches that would become accessible to anadromous salmonids upstream of Scott Dam if fish passage were provided, and (2) reaches of the Eel River from Scott Dam downstream to the Tomki Creek confluence, where population productivity is expected to be significantly influenced by altered hydrology, water temperatures, and channel changes associated with the Project. Assessments specific to these two areas are reported in Section 5.1 (“Upstream of Scott Dam”) and Section 5.2 (“Downstream of Scott Dam”) such that relative salmonid productivity tradeoffs for each area can be compared for each Water Supply Scenario. This evaluation is mostly qualitative, relying on published literature and professional judgement, although quantitative information is used to the extent possible to support individual assessments and conclusions. The analyses assume a future steady-state or equilibrium condition expected after any short-term impacts resulting from dam removal have passed.

The focal species for this evaluation include fall-run Chinook Salmon and steelhead (summer- and winter-run), with emphasis on evaluating life stages for which habitat limitations or density-independent survival within the affected reaches are most likely to limit overall population productivity (i.e., the number of adults returning to spawn). Generalized life history timing for these species and life stages in the Eel River are presented in Section 2.

For fall-run Chinook Salmon, differences in Water Supply Scenarios that affect the spawning, incubation, and outmigration life stages are most likely to affect population productivity, since fry and early juveniles typically outmigrate in the spring soon after emergence and do not rear through the summer.

For steelhead, summer and winter rearing habitats for older juvenile life stages (age-1+ and older) are potentially most limiting within the affected reaches. As steelhead grow, they tend to prefer deeper habitats and defend a larger territory (requiring more space), resulting in lower habitat carrying capacity for older and larger juveniles. Steelhead spawning habitat is not likely to limit population productivity compared with older juvenile rearing habitat since a high number of eggs can fit into a relatively small redd area. For this reason, rearing habitat carrying capacity for older juveniles would likely be far less than the number of fry or age-0 juveniles that could be produced from available spawning habitat.

5.1 Eel River Upstream of Scott Dam

The evaluation of productivity tradeoffs for reaches upstream of Scott Dam relies primarily on existing estimates of habitat carrying capacity and production potential for reaches that would become accessible with construction of fish passage facilities at Scott Dam, or with Scott Dam removal. Estimates of potential anadromous salmonid habitat and productivity potential are summarized from a series of studies over time which have used differing methodologies to estimate capacity in reaches above Scott Dam. In addition, and where available, a brief overview of methods used to develop the estimates is provided. Due to the wide range of values in reported estimates of production potential for reaches upstream of Scott Dam, a comparative estimate of production potential is developed to provide context using recent credible values of accessible stream length and potential fish density (see Section 5.1.2).

5.1.1 Summary of Habitat Quantity and Salmonid Abundance Estimates

Existing information on available habitat and potential production capacity for Chinook Salmon and steelhead for reaches upstream of Scott Dam are summarized in Table 5-1. For Chinook Salmon, estimates of available habitat length upstream of Scott Dam range from approximately 35 to 93 mi (56 to 150 km) and estimates of potential production range from approximately 800 to 10,000 adults (spawners). For steelhead, estimates of available habitat length range from approximately 36 to 365 mi (60 to 584 km) and estimates of potential production range from approximately 500 to 25,000 adults (spawners).

Cooper (2017) provides estimates of both available habitat and potential production for Chinook Salmon and steelhead that can be summarized to correspond with the Feasibility Study Alternatives. Based on Cooper (2017), removal of Scott Dam could provide an approximately 26 percent increase in Chinook Salmon adult production and 13 to 15 percent increase in steelhead adult production compared with passage at Scott Dam via fish ladder (with Lake Pillsbury remaining).

In addition to the information reported in Table 5-1, Becker and Reining (2009) report that *O. mykiss* have been observed in almost all streams upstream of Scott Dam historically. No quantities of habitat or densities were reported. Their report was based on an extensive literature review, data sheets, newspaper articles, and personal communications. A brief description of methods used to develop potential production estimates is presented in Table 5-2.

Table 5-1. Estimates of available habitat and potential production of Chinook Salmon and steelhead for reaches upstream of Scott Dam. Conversions from source data are presented in parentheses.

Source		**Estimated Habitat**				**Estimated Fish Production Potential**			
		Chinook Spawning	**Chinook Rearing**	**Steelhead Spawning**	**Steelhead Rearing**	**Chinook Spawning**	**Chinook Rearing**	**Steelhead Spawning**	**Steelhead Rearing**
CDFW (1979) [a, b]		N/A	N/A	N/A	N/A	2,300	N/A	2,500	N/A
VTN (1982) [c]	Historic	35.7 mi (60.4 km)				2,499	N/A	3,356	N/A
	Current	25.2 mi (40.6 km)				1,250	N/A	1,499	N/A
USFS and BLM (1995) [a]		100 mi (161 km)				N/A	N/A	N/A	N/A
Higgins (2010) [d, e]		N/A				3,092	N/A	N/A	N/A
Becker and Reining (2009) [d]		411 km (255 mi)				N/A	N/A	N/A	N/A
Spence et al. (2008, 2012) [f]		60.6 km		347.8 km		1,212	N/A	6,956	N/A
NMFS (2016) [g]		59.8 km		317.5 km (197.3 mi)		1,196	N/A	6,400	N/A
Cooper (2017) [h]	Scenario 1	127 km (79 mi)		463 km (288 mi)	291 km (181 mi)	4,593	201,426	1,044 – 2,044	57,374
	Scenario 2	111 km (69 mi)		437 km (272 mi)	233 km (145 mi)	3,655	160,322	907 – 1,815	49,858
	Scenario 3	89 km (55 mi)		318 km (198 mi)	179 km (111 mi)	1,487	65,200	507 – 1,014	27,848
	Reservoir	16 km (10 mi)		26 km (16 mi)	26 km (16 mi)	938	14,104	137 – 229	7,516
Cooper et al. (2020)	1938-1955 (high)[i]	151 km		460 km	291 km	10,123	N/A	26,016	N/A
	1955-1963 [j]					1,075	N/A	10,899	N/A
	1964-1975 [k]					2,172	N/A	3,321	N/A
	UCM [l]	151 km		460 km	291 km	4,593	201,426	1,281	57,374
	UCM [m]	100 km		315 km	178 km	1,487	65,200	622	27,848
FitzGerald et al. (2020)		144 km (90 mi)		584 km (365 mi)		1,242–3,314	N/A	256–5,370	N/A

[a] Information as cited in Cooper (2017)

[b] Information as cited in VTN (1982)

[c] Historic estimate based on no reservoir present and density for historic high escapement in Tomki Creek. Current estimate based on area upstream of Lake Pillsbury and density for recent high escapement in Soda Creek. Note that VTN (1982) reports that another 23 miles of "minor channel" exists (p.291 and Table 3.10-2, Cooper [2017]

reports 22.7 miles), but VTN does not include these for calculating the production estimates reported for steelhead. 10.5 miles of channels were reported to be inundated by Lake Pillsbury.

[d] Information as cited in Cooper et al. (2020)

[e] Includes production from reaches between Scott Dam and Cape Horn Dam

[f] Production estimates reported in NMFS (2016) are based on the IP model described by Spence et al. (2008, 2012), with updates. Therefore, NMFS (2016) replaces estimates reported by Spence et al. (2008, 2012). Chinook Salmon habitat length and production potential reported by Spence et al. (2008, Table 8 p.79). Chinook Salmon historical production calculations: 555.9 km - 495.3 km = 60.6 km; 60.6 km * 20 fish/km = 1,212 fish. Steelhead habitat length and production potential reported in Spence et al. (2012, Table 6, p.12). Steelhead historical production calculations: 349.6 km – 1.8 km = 347.8 km; 347.8 km * 20 fish/km = 6,956 fish.

[g] Chinook Salmon habitat length of 59.8 km is reported in Appendix G (Table 1, p.8), and reported as updated/revised estimate from Spence et al. (2008, 2012). Chinook Salmon production potential of 796 spawning adults calculated based on 20 fish/km. Steelhead habitat length and production potential reported in Table 22 (Page 138).

[h] Three study scenarios: 1) removal of Scott Dam, 2) fish ladder at Scott Dam, 3) removal of Scott Dam and no fish passage at Bloody Rock Roughs (i.e., in years with lower streamflow, Bloody Rock roughs is a barrier to migration and limits access to upstream habitat). Chinook Salmon and steelhead habitat length and production potential for "Reservoir" are calculated values (Scenario 1 - Scenario 2) for reaches inundated by Lake Pillsbury. These calculated values for reservoir are not reported in Cooper (2017).

[i] Based on historic high adult escapement reported from Benbow Dam Fisheries Station (BDFS) on SF Eel River during 1938–1955, and converted to adult density of 67 fish/km for Chinook Salmon and 57 fish/km for steelhead based on estimate of available habitat upstream of Benbow Dam

[j] Based on average adult escapement reported from BDFS on SF Eel River during 1955–1963, and converted to adult density of 7 fish/km for Chinook Salmon and 24 fish/km for steelhead based on estimate of available habitat upstream of Benbow Dam

[k] Based on average adult escapement reported from BDFS on SF Eel River during 1964–1975, and converted to adult density of 14 fish/km for Chinook Salmon and 7 fish/km for steelhead based on estimate of available habitat upstream of Benbow Dam

[l] Based on Unit Characteristic Method (UCM) assuming Scott Dam removal and passage at Bloody Rock Roughs

[m] Based on UCM assuming Scott Dam removal and no passage at Bloody Rock Roughs

Table 5-2. Overview of methods for reported estimates of potential Chinook Salmon and steelhead production capacity upstream of Scott Dam.

Source	**Methods Overview**
CDFW (1979)[a]	Estimates historical spawner abundances above Scott Dam. Methods of how estimates were determined are not described in Cooper (2017).
VTN (1982)	The "Mitigation Study" evaluated the quality and quantity of spawning and rearing habitat lost due to construction of Scott Dam. The length of formerly accessible habitat was estimated based on review of CDFW stream survey reports and reconnaissance-level field surveys (both aerial and ground). Estimates of production potential were calculated based on observed spawning densities multiplied by estimates of stream length. Resulting estimates of useable "major" channel length were 25.2 miles upstream of Lake Pillsbury and 10.5 miles inundated by the reservoir. An additional 23 miles of "minor" channel was reported, although not included for estimating production potential. Historic run size estimates used densities of 70 fish/mi for Chinook Salmon (based on highest recorded escapement for Tomki Creek) and 94 fish/mi for steelhead (based on 1981 estimate of 42 fish/mi from Soda Creek and "adjusted to the 1930 decade levels." Current run size estimates used densities of 35 fish/mi for Chinook Salmon and 42 fish/mi for steelhead.
USFS and BLM (1995)[a]	Cooper (2017) notes that methods of how estimates were determined "were not elaborated upon."
Spence et al. (2008, 2012)	Estimates "projected population abundances" based on intrinsic habitat potential (IP, see NMFS 2016). Density of 20 spawners/km used for both Chinook Salmon and steelhead. Spence et al. (2012) reports revised estimates to those reported in Spence et al. (2008) for steelhead based on refinement of curve relating suitability to mean annual discharge.
NMFS (2016)	NMFS (2016) builds on the work reported in Spence et al. (2008, 2012) and should be considered the most recent revision of this work. Estimated production potential was based on IP, which utilizes large-scale habitat suitability indices including slope, valley constraint, and annual discharge to determine the potential range of accessible habitat for salmonids. Potentially suitable stream habitat was mapped in GIS and production potential was based on a density of 20 spawners/km for both Chinook Salmon and steelhead.
Cooper (2017)	This study focused on estimating production potential of available habitat upstream of Scott Dam. The study area was stratified into reach types based on gradient and drainage area. Habitat was mapped, and habitat variables were collected in 10% of each habitat type in 25 randomly selected study reaches. Production capacity was estimated using the Unit Characteristic Method (UCM, Cramer and Ackerman 2009), which uses habitat unit-scale and reach-scale suitability indices (HSI, or scalars) to modify density based on habitat characteristics. The model multiplies a baseline standardized fish density by habitat unit area and adjusts density according to HSIs of

Source	Methods Overview
	habitat parameter relationships. Estimated densities of parr were used to estimate densities of spawners using survival rates between life stages. Potential water temperature limitations were incorporated into the model using a relationship developed for Coho by Cramer et al. (2012). The temperature scaler has value of 0.95 at 16°C and reduces to 0.05 at 23°C.
Cooper et al. (2020)	This study builds on the work reported in Cooper (2017), including minor adjustments to estimates of accessible habitat area and application of the UCM in estimating production capacity. In addition, estimates of potential adult/spawner capacity were reported based on historic adult counts at Benbow Dam Fisheries Station (BDFS) on the SF Eel River during 1938–1975. Adult counts at BDFS and estimated habitat availability upstream were used to estimate adult/spawner density, which were then applied to estimated habitat availability upstream of Scott Dam to estimate habitat capacity. Three density estimates were used to represent a range of observed values: (1) historic high adult count during 1938-1955, (2) average adult count during 1955-1963, and (3) average adult count during 1964–1975. Resulting densities for these periods were 67, 7, and 14 adults/km for Chinook Salmon and 57, 24, and 7 adults/km for steelhead, respectively.
FitzGerald et al. (2020)	This study focused on thermal habitat suitability of available habitat upstream of Scott Dam and evaluating the relative importance within the Eel River watershed. The study area included stream reaches downstream of barriers based on NMFS barrier coverage and streams considered suitable based on gradient according to Spence et al. (2008). The study stratified the channel network within the study area into reach types based on gradient and drainage area and assigned a "productivity level" (ideal, productive, fairly productive, poor) for spawning and rearing life stages to each reach type. Water temperature conditions were based on the spatial stream network (SSN) NorWeST model (based on Isaak et al. 2017). This regional water temperature model predicted mean monthly water temperatures for 1-km segments. Assessment of potential water temperature conditions were based on three water year types: warm (2015), average (2002-2011), cool/drought (2011). Temperature thresholds used for evaluating rearing habitat suitability included Optimal (10–15°C) and Tolerable (16–23°C). In addition, water temperatures ≥18°C were suboptimal due to increased predation pressures from Sacramento Pikeminnow. Comment: The NorWeST model is a coarse-scale model used to predict mean monthly water temperatures over large geographic area. Note: this methods overview and the numbers reported in Table 5-1 above are based on an unpublished draft report that is subject to change.

[a] Information as cited in Cooper (2017)

5.1.2 Discussion and Synthesis

Acknowledging the wide range of reported estimated values of potential production capacity for accessible reaches upstream of Scott Dam (Tables 5-1 and 5-2), this section attempts to provide context for reported estimates based on a review of currently available information and methods and applies a consistent approach to estimating potential production capacity to facilitate comparison.

A common approach used to estimate potential production capacity of habitat is applying estimates of linear density (fish/km) to estimates of suitable and accessible stream length. Recent studies use techniques for estimating available habitat that are well-documented and far more refined than earlier estimates. Based on these recent studies, about 93 mi (150 km) of habitat for Chinook Salmon and about 310 mi (500 km) for steelhead (estimated based on Cooper et al. 2020 and FitzGerald et al. 2020) would be available in accessible stream reaches upstream of Scott Dam.

Reported estimates of adult (spawner) density for the Eel River vary considerably based on the methods used and supporting data available (Tables 5-1 and 5-2). Reported densities used to estimate potential historic production capacity range up to 67 fish/km for Chinook Salmon and 57 fish/km for steelhead (Cooper et al. 2020). Reported densities used for estimating potential current production capacity range up to 22 fish/km for Chinook Salmon and 26 fish/km for steelhead (VTN 1982). Differences between historic and current estimates are a function of anthropogenic factors such as land and resource management (exacerbated by large floods) and development (roads, levees, dams), which have contributed directly to habitat degradation over the past century, as well as indirect factors such as pollution and climate change.

Applying a consistent density estimate is useful for assessing relative differences in potential production capacity. NMFS (2016) and others have used 20 fish/km for estimating potential production capacity under current conditions both for Chinook Salmon and steelhead in the Eel River, which is a reasonable approximate value based on the reported range of densities and considering the supporting data and associated methods. Applying a density of 20 fish/km to estimates of suitable habitat length upstream of Scott Dam, and for the entire Eel River watershed, are summarized in Table 5-3. Doubling the density estimate to 40 fish/km may provide a rough approximation of densities that could be achieved after implementing comprehensive and coordinated restoration throughout the watershed. While near the lower end of the range of reported values for potential historic conditions (41–67 fish/km; VTN 1982, Cooper 2017, Cooper et al, 2020), 40 fish/km may more appropriately represent an achievable value accounting for habitat limitations under current conditions. Note that densities for “historic conditions” referred to here are likely substantially less than peak densities for Chinook Salmon during pre-disturbance conditions, when the Eel River watershed may have supported 100,000 to as many as 800,000 adults (Yoshiyama and Moyle 2010), equating to an average density of 40 to 320 fish/km.

Table 5-3. Potential production capacity estimates for current conditions based on generalized estimates of available habitat and adult density (20 fish/km).

Location	Species	Available Habitat (km)	Potential Production Capacity
Upstream of Scott Dam	Chinook Salmon	150	3,000
	Steelhead	500	10,000
Eel River watershed	Chinook Salmon	2,500[a]	50,000
	Steelhead	5,000[a]	100,000

Notes: km = kilometers

[a] from Yoshiyama and Moyle 2010

For comparison, northern California's Smith River has relatively healthy populations of Chinook Salmon and steelhead, estimated at about 21,250 and 15,500 adults, respectively, based on the average for two years of adult escapement monitoring (Larson 2013). Drainage area of the Smith River watershed is approximately 1,860 km^2, compared with 9,542 km^2 for the Eel River watershed. Applying the number of fish per km^2 from the Smith River to the Eel River results in Chinook Salmon and steelhead populations of about 110,000 and 80,000, respectively. Back-calculated linear densities from these estimates results in about 44 and 16 fish/km for Chinook Salmon and steelhead, respectively. These estimates provide another rough approximation of potential production capacity for the Eel River watershed with comprehensive restoration.

The assessments described in Table 5-1 and Table 5-2 focus on estimates of production potential based on channel characteristics and habitat potential within accessible reaches. Potential limitations to production potential caused by high summer water temperature, predation by non-native fish species, and the influence of water temperature on predation potential were also considered in some studies (e.g., Cooper 2017, FitzGerald et al. 2020). However, there are numerous other considerations that could affect potential production that are not addressed in this analysis including:

Dams Remain – Water Supply Scenario 4B

- Predation in the reservoir by non-native species and maintaining habitat that supports non-native predator populations could largely negate the production benefits of providing fish passage. Predator populations in Lake Pillsbury likely seed predator populations in habitat upstream and downstream from the reservoir.
- Upstream migration delay at dam/fishway – some proportion of fish may be delayed while attempting to pass dam/fishway, others may avoid passage or be unable to negotiate fishway.
- Downstream migration delay at reservoir – potentially difficult for juveniles (and kelts) to navigate reservoir, locate dam, and navigate pathway downstream of dam.
- Alternative downstream passage options, such as tributary collection, would likely have low capture probability during some periods (e.g., periods with high streamflow, woody debris, and turbidity).

Dam(s) Removed – Water Supply Scenario 2 and Scenario 3

- Increased life history diversity, or the portfolio effect (Schindler et al. 2010), which provides greater overall population resiliency. For example, under current conditions, most of the fall-run Chinook Salmon population is presumably ocean-type (i.e., adults mature in the ocean, migrate upstream in fall to spawn, and juveniles outmigrate soon after emergence in early spring), but upstream of Lake Pillsbury, it is possible that a fall-migrant component (juveniles rear over summer and outmigrate in fall/winter) could establish (like some Klamath River fall-run Chinook Salmon). Similar properties could be applied for increased life history diversity in steelhead.

5.1.3 Additional Data Needs, Uncertainties, and Recommended Additional Studies

Additional data on site-specific conditions should be collected, including hydrology, water temperature, available/accessible habitat (migration barriers), habitat conditions (habitat type frequency and distribution, spawning and rearing habitat availability, cover and complexity), and distribution and abundance of non-native predators. Most of these data would be provided from implementation of the FERC-approved study plan with proposed modifications and new studies.

It would also be useful to synthesize existing information and newly developed information from site-specific studies and use it within a limiting factors framework to evaluate site-specific production potential using a population dynamics life-cycle model. This type of approach would also lend itself to identifying where habitat conditions are currently in good condition and where restoration would potentially be appropriate for improving conditions.

5.1.4 Summary

Reported estimates of available habitat and potential production capacity upstream of Scott Dam vary significantly. However, recent studies use techniques for estimating available habitat that are well-documented and more refined than earlier estimates. For the Dam(s) Removed condition, about 93 mi (150 km) of habitat for Chinook Salmon and about 310 mi (500 km) for steelhead (estimated based on Cooper et al. 2020 and FitzGerald et al. 2020) would become accessible based on these recent studies. Assuming an adult density of 20 fish/km, which was used by NMFS in recovery planning to estimate potential production capacity and is within the range of observed densities, this indicates a potential production capacity upstream of Scott Dam of approximately 3,000 Chinook Salmon and 10,000 steelhead. These estimates include habitat within the Lake Pillsbury footprint, which could require years to recover and become fully productive.

The Dams Remain (Water Supply Scenario 4B) condition would provide access to habitat upstream of Scott Dam; however, production capacity would be reduced due to habitat inundated by Lake Pillsbury, which includes about 10 mi (16 km) of habitat for Chinook Salmon and 16 mi (26 km) for steelhead (Cooper 2017). This loss in potential production capacity would include approximately 320 Chinook Salmon and 520 steelhead, or 10 percent and 5 percent, respectively, of total production estimated for reaches upstream of Scott Dam.

Additional losses to production capacity of habitat upstream of Scott Dam would be expected under Baseline and Scenario 4B, particularly resulting from predation by non-native predators. Predation by non-native species would likely be particularly problematic in Lake Pillsbury and areas around project infrastructure where juvenile fish would be concentrated during outmigration. The degree to which predation by non-native predators in Lake Pillsbury (and reaches upstream) will reduce production of juvenile and smolt produced from reaches upstream is uncertain and could be substantial. We did not attempt to estimate potential production losses to predation. If extreme, predation of juvenile salmonids by non-native predators in Lake Pillsbury could reduce, or even eliminate the potential production benefits of providing fish passage and has the potential for creating a population sink.

5.2 Eel River Downstream of Scott Dam

For reaches downstream of Scott Dam, potential salmonid productivity tradeoffs between Water Supply Scenarios were qualitatively assessed by describing how differences in hydrology (Section 3.1), water temperature (Section 3.2), and various ecological changes (Section 4) are expected to influence salmon and steelhead habitat carrying capacity, growth, and survival. The assessment primarily focused on differences in hydrology and water temperature because these variables are expected to differ substantially between Water Supply Scenarios and are critical determinants of fish distribution, physical habitat suitability, growth, and ecological interactions such as competition and predation with non-native Sacramento Pikeminnow. Likely effects of expected changes in fluvial geomorphology (Section 3.3) under Dam(s) Removed scenarios were also considered in some cases. Importantly, the assessment did not account for potential short-term impacts to habitat or water quality related to dam removal (e.g., turbidity and sediment flushing from Lake Pillsbury) or other infrastructure changes. Instead, it evaluated habitat conditions expected once the channel and sediment supply reach an equilibrium condition.

The assessment primarily compared expected differences in salmonid productivity between Dams Remain (Baseline and Scenario 4B) and Dam(s) Removed (Scenario 2 and Scenario 3), but also included some comparison of differences between specific scenarios where relevant.

The assessment focused on the following reaches of the mainstem Eel River, where population productivity is expected to be most significantly influenced by altered hydrology and water temperatures associated with the Project:

- Scott Dam to Cape Horn Dam (Section 5.2.1)
- Downstream of Cape Horn Dam (Section 5.2.2)

Hydrological and resulting ecological differences between Water Supply Scenarios may influence salmonid productivity downstream of Tomki Creek, but these effects are expected to be relatively minor compared with upstream reaches, and decay further downstream.

This assessment focused on life stages for which changes in hydrology and water temperature in these reaches are expected to have the greatest effects on population productivity, mainly Chinook Salmon spawning, incubation, and outmigration and steelhead juvenile rearing and outmigration. To facilitate evaluation of how differences between Water Supply Scenarios may affect these key life stages, the assessment was broken into the following periods:

- March through June (juvenile salmonid spring growth and outmigration period);
- July through September (steelhead summer rearing period encompassing thermal maximum); and
- October through January (Chinook Salmon fall migration and spawning).

Section 5.2.3 provides a synopsis of key positive and negative effects of dam removal on population productivity of Chinook Salmon and steelhead downstream of Scott Dam and a discussion of magnitude of these effects relative to providing access to historical habitats upstream of Scott Dam.

5.2.1 Eel River from Scott Dam to Cape Horn Dam

5.2.1.1 March through June (Juvenile Growth and Outmigration)

Higher March and April streamflows and higher magnitude late winter and early spring flow peaks (Section 3.1), coupled with warmer spring water temperatures (Section 3.2), are expected under Scenario 2 and Scenario 3. These differences in early spring conditions are expected to facilitate outmigration of juvenile Chinook Salmon and steelhead from the reach between Scott Dam and Cape Horn Dam and likely improve their survival in the reach compared with the lower, more stable, and cooler spring streamflows expected to occur under Baseline and Scenario 4B. Various studies have found increased migration rate and survival of outmigrating smolts with increasing streamflow (e.g., Michel et al. 2015, Cordoleani et al. 2017). However, as described in Section 5.2.3, the benefits of higher stream flows in the reach between the dams may be offset by the lower streamflows downstream of Cape Horn Dam (due to significant spring diversion at Van Arsdale) under Scenario 2, relative to the unimpaired flows under Scenario 3.

The moderately lower streamflows downstream of Scott Dam projected for late spring (May and June) under the natural hydrographs of Scenario 2 and Scenario 3 are not expected to have large effects on juvenile salmonid habitat carrying capacity, growth, or survival relative to Baseline and Scenario 4B. Depending on water year type and changes in water temperature, effects on salmonid productivity could be positive or negative. As described below, the lower streamflows and warmer water temperatures associated with the natural hydrographs of Scenarios 2 and 3 could increase juvenile salmonid growth and encourage earlier migration, but in drier water years, temperatures would exceed stressful levels earlier, adversely impacting growth and survival.

The loss of the ability to implement spring blockwater releases under Scenario 2 and Scenario 3 is expected to have little to no overall effect on salmonid production, since the spring blockwater releases are designed to mimic a more natural hydrograph and raise water temperatures, both of which would occur under Scenario 2 and Scenario 3.

The warmer spring water temperatures projected to occur in the reach between Scott Dam and Cape Horn Dam under Scenario 2 and Scenario 3 in the absence of coldwater releases are expected to have positive effects on juvenile salmonid populations by encouraging earlier and more natural emigration timing and improving growth rates during the critical period leading up to emigration. Beak (1986) found that timing of both juvenile Chinook Salmon and steelhead emigration from the Eel River above Cape Horn Dam was delayed and prolonged relative to unregulated Tomki Creek due to coldwater releases. Subsequent analyses of 6 years of data collected in the 1980s indicated that the date when 50 percent of the Chinook Salmon population emigrated from the upper Eel River (May 7) was on average 19 days later than from Tomki Creek (April 17), where daily water temperatures were on average 3°C warmer and had greater diel fluctuations than downstream of Scott Dam during the period leading up to emigration (SEC 1998).

Such emigration delays, which would likely continue to occur under Scenario 4B, are expected to substantially reduce survival by increasing the likelihood that juvenile salmonids would encounter thermally stressful or lethal water temperatures in downstream reaches of the Eel River. For example, daily mean water temperatures at many sites downstream of Cape Horn Dam reach levels stressful to juvenile salmonids by late May or early June and may reach lethal levels during drier water years (PG&E 2017a, Section 3.2). Delayed migration during periods with lower, less turbid streamflows and warmer water temperatures also means that juvenile salmonids are more susceptible to predation by Sacramento Pikeminnow compared with earlier migrating individuals, particularly in drier years (Brown and Moyle 1981, Brown and Moyle 1997, SEC 1998). Predation by pikeminnow has the potential to be a significant source of mortality on juvenile salmonids during emigration from the upper Eel River. In May 1989, Brown and Moyle (1997) found that salmonids were present in 100 percent of diet samples collected from Sacramento Pikeminnow greater than 100 mm captured in the mainstem Eel River near the Outlet Creek confluence.

The warmer temperatures expected from Scott Dam to Cape Horn Dam under Scenario 2 and Scenario 3 are also expected to improve spring growth rates of juvenile salmonids rearing there relative to Baseline and Scenario 4B, particularly during March and April when coldwater releases from the reservoir often range from 7 to 11°C, considerably colder than values that maximize growth of both Chinook Salmon and steelhead (Section 3.2). Improved growth rates in the spring during the critical period leading to outmigration may further encourage earlier emigration and increase smolt-to-adult survival, which is highly correlated with size at outmigration (Peterman 1982, Ward et al. 1989, Cordoleani et al. 2017).

As described in Section 4.7, the higher magnitude, longer duration winter peak flows expected under Dam(s) Removed Scenario may also promote higher food quality and greater diversity of the benthic macroinvertebrate community during the critical spring growth period for salmonids (Power et al. 2015, Jansen et al. 2020). Peak flows would scour armored grazers, making more niche space for diatom growth. Diatom growth sustains a more digestible and energetically profitable food base for juvenile salmonids (Power et al. 2015, Jansen et al. 2020). Thus, production of high-quality food items may increase and improve fish growth.

Warmer spring water temperatures between Scott Dam and Cape Horn Dam under Scenario 2 and Scenario 3 could also result in negative impacts on juvenile salmonids in the reach, such as earlier onset of predation and competition by Sacramento Pikeminnow. Increased exposure to pikeminnow may lower survival or growth of juvenile salmonids, particularly in May and June and in drier water years. Likely effects of water temperature and streamflow on Sacramento Pikeminnow populations and ecological interactions with salmonids are further discussed in Section 4.5 and under the summer rearing section below.

5.2.1.2 July through September (Juvenile Summer Rearing)

The substantially lower summer streamflow and warmer water temperatures projected to occur between Scott Dam and Cape Horn Dam due to loss of storage and coldwater releases under Scenario 2 and Scenario 3 are expected to primarily affect juvenile steelhead because juvenile Chinook Salmon generally do not over-summer in the upper Eel River. The exact effects of lower streamflow are uncertain, but by reducing the overall area of the wetted channel, lower flows are expected to alter the distribution and area of suitable summer rearing habitats, likely decreasing overall habitat carrying capacity in the reach, especially for age-1 and older steelhead. Lower streamflows and less habitat area may also increase competition and predation from non-native Sacramento Pikeminnow in the reach. Analysis of flow-habitat relationships (Section 4.2) suggests that from mid-April through October, suitable habitat area for steelhead juvenile would typically decrease under Scenario 2 and Scenario 3 relative to Baseline. However, the lower range of streamflows could not be analyzed due to limitations in the WUA curve applied, and thus the magnitude of habitat loss is unclear. Additionally, as described below, utilization of suitable physical habitat may be further limited by warm water temperatures in some years or presence of pikeminnow.

In combination with the likely decrease in suitable physical habitat area, warmer summer water temperatures in the reach under Scenario 2 and Scenario 3 may have several negative impacts on the juvenile steelhead, potentially decreasing densities and eventual smolt production from the reach relative to Baseline and Scenario 4B. Without Scott Dam, daily mean summer water temperatures are generally expected to increase from typical baseline values of 14 to 20°C (depending on water year) to about 20 to 24°C—approaching or exceeding levels where juvenile steelhead may experience sublethal effects such as reduced growth, reduced competitive ability, behavioral alterations, and/or increased susceptibility to disease and predation (Section 3.2). Of particular concern is increased competition and predation by Sacramento Pikeminnow (Section 4.5), which have largely displaced steelhead in reaches downstream of Cape Horn Dam (SEC 1998, PG&E 2018). In addition to increased direct predation on steelhead at warmer temperatures, presence of large predatory Sacramento Pikeminnow has been shown to cause juvenile steelhead to increase use of higher velocity riffles relative to slower habits that may be more productive (Brown and Moyle 1991). A laboratory study by Reese and Harvey (2002) found that at water temperatures of 20 to 23°C, growth of territorially dominant juvenile steelhead was reduced by more than 50 percent in the presence of equal numbers of similarly-sized pikeminnow, however, no growth reduction was observed at 15 to 18°C. On the other hand, in the absence of Scott Dam under Scenario 2 and Scenario 3, juvenile steelhead may move into cooler upstream tributaries, where competition and predation would be reduced due to less suitable water temperatures and habitat for pikeminnow (NMFS 2016, Fitzgerald et al. 2020).

Under Scenario 2 and Scenario 3, in the reach between Scott Dam and Cape Horn Dam, summer streamflow would likely be similar to that recently observed in the Eel River just downstream of Cape Horn Dam, and water temperatures would likely be slightly cooler. For this reason, under Scenario 2 and Scenario 3, juvenile steelhead summer densities in the reach between the dams are generally expected to be similar to or marginally higher than those recently observed at the uppermost summer fish monitoring sites downstream of Cape Horn Dam (PG&E 2018, Figure 4-26). Juvenile steelhead density data from recent summer sampling in the reach between Scott Dam and Cape Horn Dam (PG&E 2020) are more qualitative in nature and not directly comparable to the quantitative estimates of steelhead densities at sites downstream of Cape Horn Dam (PG&E 2018). Thus, it is not possible to quantify the expected differences in steelhead densities between Water Supply Scenarios by comparing densities observed upstream and downstream of Cape Horn Dam. Nonetheless, with Scott Dam in place (Baseline and Scenario 4B), steelhead summer rearing densities are expected to be considerably higher above Cape Horn Dam than in downstream reaches due to more suitable water temperatures. This generality is supported by more quantitative densities estimates and reach comparisons from the early 1980s, which indicated high densities and growth rates of steelhead rearing between the dams (VTN 1982, SEC 1998); although these data were likely collected before the complete establishment of Sacramento Pikeminnow.

Reduction in steelhead densities downstream of Scott Dam under Scenario 2 and Scenario 3 due to lower flows and impacts from pikeminnow competition are expected to be most pronounced in dry water years. Annual juvenile steelhead density estimates conducted by PG&E at sites downstream of Cape Horn Dam between 2005 and 2017 indicate that at each site, densities were higher in wetter water years compared with drier water years (Figure 4-26). In contrast, pikeminnow densities (based primarily on smaller, juvenile fish) were higher in drier water years (Figure 4-25). The mechanisms behind these patterns are unclear but may be related to higher winter and spring flows reducing pikeminnow spawning success and juvenile recruitment. These patterns warrant more thorough examination due to their potentially important implications for Sacramento Pikeminnow and steelhead management in Project-affected reaches.

There are several trade-offs and uncertainties that could help offset the potential loss of juvenile steelhead production between Scott Dam and Cape Horn Dam under Scenario 2 and Scenario 3. As described above, removal of Scott Dam would result in connectivity between this reach and cooler mainstem and tributary habitats upstream of Scott Dam, allowing juvenile steelhead to move to locations that limit exposure to higher temperatures and pikeminnow predation and competition. Likewise, infrastructure associated with Cape Horn Dam and Scott Dam, as well as Van Arsdale Reservoir, have been observed to concentrate predatory Sacramento Pikeminnow and Largemouth Bass (Section 4.5). Removal or modification of these predatory hotspots and the habitats created by the reservoirs may reduce pikeminnow populations and/or predation on juvenile steelhead in the reach. It is also possible that unimpaired winter and spring flows in the absence of reservoir attenuation will reduce overwinter survival, spawning success, or juvenile recruitment of pikeminnow and other predatory fishes and depress their populations, lessening their impacts on steelhead in the summer. Additionally, with the reduced summer streamflows of dam removal, thermal stratification in summer is more likely to occur in deep pools between Scott Dam and Cape Horn Dam. Thermal stratification has been documented in pools in the mainstem Eel River downstream of Cape Horn Dam (Kubicek 1977, PG&E 2016, Figure 3-40) and also in the Middle Fork Eel River, where these pools provided refuge habitat for juvenile steelhead in reaches that would otherwise be uninhabitable (Nielson et al. 1994).

Notably, substantial changes in channel form that could affect habitat area and suitability are likely to occur in the reach downstream of Scott Dam under Scenario 2 and Scenario 3 (Section 3.3). Currently (and under Scenario 4B), alluvial bars and floodplains are coarsened and less mobile, and riparian encroachment inhibits channel migration. With the restoration of natural sediment supply and unimpaired peak flows following removal of Scott Dam, lateral channel migration will increase and channel morphology in the reach will be similar to the Eel River upstream of Lake Pillsbury, with more lateral channel migration and exposed cobble/gravel bars. The more natural, dynamic channel, along with increased supply of large wood, is also expected to result in greater overall habitat complexity and potentially greater hyporheic exchange that could moderate summer water temperatures. The length of time after dam removal required for the channel to adjust and equilibrate will depend on sediment management and dam removal methods, as well as hydrologic conditions.

Finally, the warmer water temperatures may increase the incidence of anadromy versus residency in *O. mykiss* in the reach between the dams, since temperature directly influences metabolism, growth, and lipid content (Kendall et al. 2015). Anadromy has been shown to predominate in streams with warmer, stressful summer water temperatures, with residency being more common in cooler streams without stressful temperatures (Sogard et al. 2012).

5.2.1.3 October through January (Adult Fall Migration and Spawning)

The effects of differences in fall and early winter streamflows between Water Supply Scenarios on salmonid productivity between Scott Dam and Cape Horn Dam are unclear, but would likely have the greatest effect on migrating, spawning, and incubating Chinook Salmon life stages. While differences in early winter streamflow regime could have minor influences on steelhead migration behavior, these differences would likely have minimal effects on overall steelhead population productivity.

The lower early fall streamflows expected in the reach under Scenario 2 and Scenario 3 (Section 3.3) could obstruct Chinook Salmon passage at shallow riffles and limit spatial distribution of earlier spawners, particularly in drier years. Additionally, loss of the ability to implement fall blockwater releases under Scenario 2 and Scenario 3 would preclude discretionary flow releases intended to promote adult Chinook Salmon migration or reduce incidence of disease. This reduced ability to manage fall streamflows could have negative impacts on Chinook Salmon production in some years.

The higher and naturally variable late fall and early winter streamflow under Scenario 2 and Scenario 3 could have variable effects on Chinook Salmon spawning habitat area and spawning success. The flow-habitat analysis (Section 4.2) suggests a general decline in suitable spawning habitat area at a site near Trout Creek between Baseline and Scenario 2 and Scenario 3. The predicted decline in suitable spawning habitat is likely driven by increased water depths and velocities associated with higher and less muted streamflows during the November and December spawning period under Scenario 2 and Scenario 3 relative to Baseline. A study evaluating Chinook Salmon spawning habitat in the Trinity River, however, suggests that ascending baseflows of a natural hydrograph could increase the abundance and quality of spawning habitat compared with lower and more stable, regulated streamflows, particularly at unconfined locations where higher streamflows improve lateral connectivity (Goodman et al. 2018). The authors also hypothesized that because naturally ascending baseflows facilitate spatial separation of preferred habitats over the spawning season, they may reduce the risk of redd superimposition.

The higher magnitude, longer duration peak flows expected to occur in late fall and early winter under Scenario 2 and Scenario 3 could have a range of effects on Chinook Salmon spawning and egg incubation. One potential negative impact would be increased likelihood of redd scour or sand infiltration during years with early flood events. However, in the long term, removal of Scott Dam is expected to promote a more naturally meandering, dynamic, and hydraulically complex channel that promotes gravel sorting and potentially increases area of suitably sized spawning substrates (Section 3.3).

Overall, any potential reductions in Chinook Salmon pre-spawning survival, spawning habitat area, and spawning success in the reach are likely to be offset by positive effects of a more natural fall and winter hydrograph in the absence of the dam and access to a large amount of spawning habitat upstream of Scott Dam.

5.2.2 Eel River Downstream of Cape Horn Dam

5.2.2.1 March through June (Juvenile Growth and Outmigration)

Due to the enlarged diversion tunnel and lower streamflows downstream of Cape Horn Dam during the spring recession period, the effects of Scenario 2 on salmonid production relative to the Baseline and Scenario 4B are expected to be considerably different compared with the effects of Scenario 3 (no diversion). For this reason, Scenario 2 and Scenario 3 are evaluated separately in this section.

The generally lower spring flows downstream of Cape Horn Dam due to the larger diversion under Scenario 2 would have a range of effects on salmonid productivity in the reach, depending on water year and month. In general, the lower spring flows of Scenario 2 are expected to have low to moderate negative impacts on outmigrating and juvenile rearing life stages of Chinook Salmon and steelhead relative to Baseline and Scenario 4B. As described above, based on various studies in other river systems, survival of juvenile salmonids during emigration is expected to be reduced at lower streamflows. However, despite the greater diversion capacity under Scenario 2, during spring high flow events, streamflow downstream of Cape Horn Dam would generally be higher compared with Baseline and Scenario 4B, potentially facilitating juvenile emigration through the reach during these events.

The unimpaired spring flows downstream of Cape Horn Dam under Scenario 3 would typically be substantially higher than under both Scenario 2 and Scenario 4B, which have significant spring diversion at Van Arsdale (Section 3.1). Depending on water year type and when spring diversions cease, Scenario 2 streamflows would typically be about 100 to 300 cfs lower than Scenario 3 in March and April and 0 to 200 cfs lower in May and June. These substantially lower flows would likely lower survival of outmigrating salmonids, especially in wetter years when the diversion is retained well into June and Scenario 2 flows would often be less than 50 percent of the unimpaired flows of Scenario 3. Under Scenario 2, the large difference in spring flows between reaches upstream and downstream of Cape Horn Dam during some years would result in sudden changes in environmental conditions for emigrating, likely reducing the benefits of unimpaired spring flows in the reach between Scott Dam and Cape Horn Dam.

In addition to affecting outmigration timing and survival, diversion-related spring flow reductions may decrease area of suitable rearing habitat for juvenile steelhead and reduce inundation of riffles and benthic macroinvertebrate production, thereby reducing juvenile salmonid spring growth (O’Dowd and Trush 2016). As described in Section 4.5, spring flow reductions and associated increases in water temperature could also increase the area of Sacramento Pikeminnow preferred habitat, decrease the competitive advantage of steelhead versus pikeminnow, and potentially increase pikeminnow spawning success and larval recruitment.

The likely warmer spring water temperatures downstream of Cape Horn Dam under Scenario 2 compared with Scenario 3 could have a range of negative or positive effects on salmonid growth, depending on water year type. Under Scenario 2, late spring water temperatures could more rapidly exceed levels stressful to salmonids, causing sublethal effects. However, in early spring, warmer water temperatures could improve juvenile salmonid growth rates, resulting in larger size at outmigration and increased survival. Some of these potentially negative impacts of diversion are expected to be offset in reaches downstream of Tomki Creek due to tributary accretion and a more natural hydrograph. In addition, during drier years such as 2015, spring streamflow differences between Scenario 2 and Scenario 3 are typically less pronounced due to earlier cessation of the diversion.

As described above, the option to implement spring blockwater releases would no longer exist under Dam(s) Removed Water Supply Scenarios. The loss of these discretionary, protective flow releases could negatively impact juvenile salmonid survival and smolt production under both Scenario 2 and Scenario 3, particularly during drier years. However, as described above, the more natural spring hydrograph of Scenario 3 would likely limit these potential impacts relative to Scenario 2.

5.2.2.2 July through September (Juvenile Summer Rearing)

The relatively minor differences in summer and early fall streamflows between Scenario 2 and Scenario 3 and Baseline and Scenario 4B downstream of Cape Horn Dam (Section Eel River Downstream of Cape Horn Dam (gage E-11)) are not expected to have significant effects on production of salmonids in the reach. Under all Water Supply Scenarios, conditions for juvenile steelhead summer rearing are expected to be marginal in this reach due to warm water temperatures and presence of large numbers of Sacramento Pikeminnow. Overall densities of steelhead in this reach are expected to continue to be relatively low under Dam(s) Removed scenarios, particularly in drier water years. During some years, the minor differences in streamflow and water temperature between Water Supply Scenarios could affect availability of suitable steelhead rearing habitat and affect growth rates by altering delivery or composition of food resources (Section 4.7). The overall effect of these differences on steelhead productivity is unknown but expected to be small compared with the effect of differences in summer streamflows and temperatures in the reach upstream of Cape Horn Dam.

5.2.2.3 October through January (Adult Fall Migration and Spawning)

Due to significant diversion at Van Arsdale under Scenario 2, once minimum flow thresholds are exceeded in the fall, effects of Scenario 2 on salmonid production in the reach between Cape Horn Dam and Tomki Creek are expected to be considerably different compared with Scenario 3. For this reason, these Dam(s) Removed scenarios are assessed separately in this section.

Prior to significant fall rains and re-initiation of the diversion, the minor differences in streamflow between the various Water Supply Scenarios are not projected to have considerable effects on salmonid production. The muted and lower early season flow peaks that would occur in the fall of some water years due to the larger diversion of Scenario 2 could delay migration and reduce passage of adult Chinook Salmon at critical riffles relative to the other Water Supply Scenarios, particularly Scenario 3. An analysis of the number of days considered passable by adult Chinook Salmon at a critical riffle downstream of Cape Horn Dam suggests passage may be more limited by Scenario 2 compared with the other Water Supply Scenarios (Section 4.3).

Flow-habitat analyses conducted for flows less than 500 cfs at a site downstream of Cape Horn Dam suggest differences in flow during the spawning season result in a general decline in Chinook Salmon spawning habitat between Baseline and Scenario 2 (Section 4.3.1.1). However, it is unclear whether this site is representative of spawning habitat conditions in the larger reach, and as discussed above, the generally more natural ascending baseflows expected under Scenario 2 could increase the abundance of spawning habitat compared with lower and more stable regulated streamflows (Goodman et al. 2018).

Under Scenario 2, larger later peak flows that exceed the capacity of the diversion would occur due to loss of reservoir attenuation. These fall peak flows would typically be substantially higher than the small early fall freshet peaks expected under the Baseline and Scenario 4B. Peak flows under Scenario 3 would be even larger than Scenario 2 due to lack of diversion. As described for the reach above Cape Horn Dam, these higher peak flows could increase the likelihood of redd scour during years with significant flood events that occur during the Chinook Salmon egg incubation period.

5.2.3 Summary

In reaches downstream of Scott Dam, changes in streamflow, water temperature, and fluvial geomorphology under Dam(s) Removed scenarios (Scenario 2 and Scenario 3) are expected to have a range of potential effects on productivity of Chinook Salmon and steelhead populations relative to Dams Remain scenarios (Baseline and Scenario 4B, Table 5-4). Potential population-level effects of streamflow and water temperature differences between scenarios vary by reach, season, and water year type. In some cases, effects are expected to be minimal or neutral because a life stage is not present or impacts to the life stage do not influence smolt production or adult returns. For example, higher winter streamflows under Scenario 2 and Scenario 3 could facilitate adult steelhead migration through the study reaches, but this effect would likely have minimal population-level consequences, since juvenile rearing habitat quantity and quality is expected be more limiting to steelhead production. In other cases, it is unclear whether the net effect of a difference in streamflow or temperature would be positive or negative because insufficient information is available to fully assess effects. For example, uncertainties in the effects of Scenario 2 and Scenario 3 on spring water temperature adds uncertainty in assessing direction and magnitude of temperate effects on growth and survival relative to Baseline and Scenario 4B.

The effects of dam removal expected to have the greatest influence on Chinook Salmon population productivity in reaches downstream of Scott Dam are primarily related to (1) changes in streamflow and water temperatures during the spring juvenile outmigration period and (2) changes in streamflow during the fall adult migration period. Substantially higher flows in March and April downstream of Scott Dam, along with likely increased water temperatures during that time, are expected to lead to an earlier and more natural juvenile emigration timing and increase survival during outmigration. However, the substantially lower spring flows predicted downstream of Cape Horn Dam under Scenario 2 could slow emigration and reduce survival, offsetting potential benefits to outmigrating juvenile Chinook Salmon relative to the somewhat higher spring flows of Scenario 4B, and especially the unimpaired flows of Scenario 3.

During the Chinook Salmon adult migration period, the lower early fall flows expected both upstream and downstream of Cape Horn Dam may restrict passage at shallow riffles and limit ability of earlier spawning adults to reach upstream spawning grounds, particularly in drier years. This potential negative impact would likely be greatest downstream of Cape Horn Dam and for Scenario 2, where in some years fall diversion would considerably mute small, early flow peaks (relative to Scenario 4B and especially Scenario 3). In both reaches, the higher late fall and winter flow peaks expected under Scenario 2 and Scenario 3 in some years due to loss of reservoir attenuation could also negatively affect Chinook Salmon production due to reduction of suitable spawning habitat area or increased redd scour; however, a more natural ascending hydrograph in the fall and a more complex, dynamic channel may offset this potential effect. Overall, the direction and magnitude of changes in Chinook Salmon population productivity in reaches downstream of Scott Dam resulting from dam removal are expected to vary by water year but be relatively small in most years.

The effects of dam removal expected to have greatest potential influence on steelhead population productivity are primarily related to (1) differences in streamflow and water temperatures during the spring juvenile outmigration period and (2) changes in streamflow and water temperature during the summer rearing period. Substantially higher streamflows in March and April downstream of Scott Dam are expected to lead to more natural emigration timing and improve survival. However, as with emigrating juvenile Chinook Salmon, the substantially lower spring streamflows downstream of Cape Horn Dam under Scenario 2 could offset these benefits by slowing emigration and exposing steelhead to greater Sacramento Pikeminnow predation. The largest effect of dam removal on the steelhead population would likely be related to substantially lower summer streamflow and warmer water temperatures in the reach between Scott Dam and Cape Horn Dam, which would likely reduce overall summer habitat carrying capacity and survival of juvenile steelhead in the reach. These effects are expected to reduce overall steelhead smolt production from the reach between Scott Dam and Cape Horn Dam, but the magnitude of this reduction is unclear due to considerable uncertainty in water temperatures and habitat and ecological changes resulting from removal of Scott Dam. In reaches downstream of Cape Horn Dam, only minor changes are expected in summer water temperatures due to dam removal and thus the overall effect on juvenile steelhead productivity in this reach is expected to be relatively small. Overall, steelhead population productivity in reaches downstream of Scott Dam, particularly in the 12-mile reach upstream of Cape Horn Dam, is expected to moderately decline under Dam(s) Removed scenarios (particularly Scenario 2).

Any potential declines in both steelhead and Chinook Salmon population productivity resulting from dam removal would be compensated for by the increased productivity resulting from access to the extensive, high-quality, coldwater habitats in the upper Eel River and tributaries upstream of Scott Dam (Section 5.1). Even if all current Chinook Salmon and steelhead production from the reach between the dams was lost, overall population productivity from the upper Eel River watershed would likely increase considerably with access to these historically available habitats. Moreover, removal of Scott Dam would provide access to suitable coldwater reaches capable of supporting summer-run steelhead, augmenting a rare ecotype and adding to the overall genetic diversity of steelhead in the upper Eel River. Finally, for both steelhead and Chinook Salmon populations, access to reaches above Scott Dam with variable environmental and habitat conditions would likely promote greater juvenile life history diversity and increase overall population resilience relative to Baseline and Scenario 4B.

Table 5-4. Summary of potential effects of Scenario 2 and Scenario 3 (Dam(s) Removed) on streamflow, water temperature, and salmonid productivity in reaches downstream of Scott Dam relative to Baseline and Scenario 4B (Dams Remain).

Reach	**Water Supply Scenario comparison**	**Time period**	**Streamflow and temperature differences**	**Potential effects on salmonid habitat carrying capacity, growth, or survival.**	**Projected effect on population productivity**	
					Chinook Salmon	**Steelhead**
Scott Dam to Cape Horn Dam	Scenario 2 and Scenario 3 compared with Baseline and Scenario 4B	March–June	Substantially higher flows in March and April in all years, including more higher magnitude, longer duration peak flows unaffected by reservoir attenuation	Higher migration rates and survival of juvenile Chinook Salmon and steelhead within reach	Positive	Positive
			Moderately lower flows in May and June	Overall, minimal direct effects on juvenile salmonid habitat carrying capacity growth or survival; although warmer late spring water temperatures due to lower flows could have positive or negative effects on growth and survival	Mixed, unclear	Mixed, unclear
			Loss of ability to manage spring flows (blockwater releases)	Minimal effects, since spring blockwater releases are designed to mimic more natural hydrograph and raise water temperatures, both of which would occur in the reach under Scenario 2 and Scenario 3	Neutral	Neutral
			Warmer spring water temperatures	Earlier, more natural emigration timing with lower likelihood of encountering stressful or lethal temperatures or high predation rates in downstream reaches; increased survival during emigration	Positive	Positive
				Increased spring growth rates for juvenile Chinook Salmon and steelhead and higher smolt-to-adult survival	Positive	Positive
				Earlier onset of and increased predation and competition from non-native predators, especially in late spring and drier years	Negative	Negative

Reach	Water Supply Scenario comparison	Time period	Streamflow and temperature differences	Potential effects on salmonid habitat carrying capacity, growth, or survival.	Projected effect on population productivity	
					Chinook Salmon	Steelhead
Scott Dam to Cape Horn Dam	Scenario 2 and Scenario 3 compared with Baseline and Scenario 4B	July–September	Lower streamflow due to lack of storage, especially in drier years	Reduced physical habitat area for juvenile steelhead and likely reduced carrying capacity	Neutral	Negative
			Warmer summer water temperatures that exceed level stressful to juvenile steelhead	Increased sublethal effects such as reduced growth, reduced competitive ability, behavioral alterations, and/or increased susceptibility to disease	Neutral	Negative
				Reduced densities that are more comparable to sites downstream of Cape Horn Dam	Neutral	Negative
			Cooler late summer and early fall water temperatures, particularly in wetter water years	Assume relatively minimal, but likely positive, effect on growth and production due to relatively short period that unnaturally warm late-summer temperatures occur under current conditions	Neutral	Positive
		October–January	Lower early fall flows	Restricted passage of adult Chinook Salmon at shallow riffles and limited spatial distribution of earlier spawners, particularly in drier years	Negative	Neutral
			Generally higher and more variable late fall and early winter flows associated with a natural hydrograph	Potential range of effects on Chinook Salmon spawning habitat area and spawning success	Mixed, unclear	Neutral
			Higher magnitude, longer duration peak flows unaffected by reservoir attenuation	Potential increase in scour of Chinook Salmon redds, but more complex, dynamic channel	Mixed, unclear	Neutral
			Lost ability to manage fall flows (blockwater releases)	Potentially delayed adult migration and/or higher incidence of pre-spawning mortality in some years	Negative	Neutral

Reach	Water Supply Scenario comparison	Time period	Streamflow and temperature differences	Potential effects on salmonid habitat carrying capacity, growth, or survival.	Projected effect on population productivity	
					Chinook Salmon	Steelhead
Downstream of Cape Horn Dam	Scenario 2 compared with Baseline, Scenario 4B, and Scenario 3	March–June	Lower spring streamflow downstream of Cape Horn Dam due to higher diversion rate, particularly compared with Scenario 3	Lowered migration rates and survival of juvenile Chinook Salmon and steelhead within reach, particularly compared with Scenario 3	Negative	Negative
			Warmer water temperatures due to lower flows, particularly compared with Scenario 3	Earlier onset of and increased predation and competition from non-native predators	Negative	Negative
				Warmer early spring water temperatures could improve juvenile salmonid growth rates, resulting in larger size at outmigration and increased survival. Warmer late spring water temperatures could exceed levels stressful to salmonids, causing sublethal effects.	Mixed, unclear	Mixed, unclear
			Loss of ability to manage spring flows (blockwater releases) with removal of Scott Dam	Effect uncertain, but would limit the ability of discretionary protective measures to increase survival of outmigrating salmonids in the reach during drier years	Negative	Negative
	Scenario 3 compared with Baseline and Scenario 4B	March–June	Higher spring streamflow downstream of Cape Horn Dam relative to other Water Supply Scenarios due removal of diversion	Higher migration rates and survival of juvenile Chinook Salmon and steelhead within reach	Positive	Positive
			Cooler water temperatures due to higher flows	Later onset of and reduced predation and competition from non-native predators	Positive	Positive
				Cooler early spring water temperatures could reduce juvenile salmonid growth rates, resulting in smaller size at outmigration. Cooler late spring water temperatures improve growth rates and minimize sublethal effects of warm temperatures.	Mixed, unclear	Mixed, unclear
			Loss of ability to manage spring flows (blockwater releases) with removal of Scott Dam	Effect uncertain, but would limit the ability of discretionary protective measures to increase survival of outmigrating salmonids in the reach during drier years	Negative	Negative
	Scenario 2 and Scenario 3 compared with Baseline and Scenario 4B	July–September	Minor and variable differences in streamflow between Water Supply Scenarios, depending on water year and month	Overall effect on juvenile steelhead productivity is unknown but expected to be small compared with effects of changes reduced streamflows and increased temperatures in the reach upstream of Cape Horn Dam	Neutral	Mixed, unclear

Reach	Water Supply Scenario comparison	Time period	Streamflow and temperature differences	Potential effects on salmonid habitat carrying capacity, growth, or survival.	Projected effect on population productivity	
					Chinook Salmon	**Steelhead**
Downstream of Cape Horn Dam	Scenario 2 compared with Baseline, Scenario 4B, and Scenario 3	October–January	Muted early season small flow peaks and lower baseflows due to larger diversion relative to other Water Supply Scenarios	Potential downstream effects to migrating adult Chinook Salmon, including delayed migration and reduced passage at critical riffles, particularly relative to Scenario 3	Negative	Neutral
			Generally higher and more variable late fall and early winter flows compared Baseline and Scenario 4B (but not Scenario 3)	Potential range of effects on Chinook Salmon spawning habitat area and spawning success	Mixed, unclear	Neutral
			Higher magnitude, longer duration peak flows unaffected by reservoir attenuation compared with Baseline and Scenario 4B (but not Scenario 3)	Potential increase in scour of Chinook Salmon redds	Negative	Neutral
	Scenario 3 compared with Baseline and Scenario 4B	October–January	Higher and more natural early season flow peak compared with all other Water Supply Scenarios	Potentially improved early season fish passage and lower pre-spawn mortality	Positive	Neutral
			Higher and more variable late fall and early winter flows compared with all other Water Supply Scenarios	Potential range of effects on Chinook Salmon spawning habitat area and spawning success	Mixed, unclear	Neutral
			Higher magnitude, longer duration peak flows unaffected by reservoir attenuation or diversion (relative to all Water Supply Scenarios)	Potential increase in scour of Chinook Salmon redds, but more complex, dynamic channel	Mixed, unclear	Neutral

6 SUMMARY OF ECOLOGICAL EVALUATION OF WATER SUPPLY SCENARIOS AND DAM REMOVAL ALTERNATIVES

The Eel River Watershed contains approximately 2,500 km of historically accessible channels with suitable habitat for Chinook Salmon and approximately 5,000 km for steelhead (Cooper et al. 2020; Fitzgerald et al. 2020). Eel River fisheries hold substantial cultural and economic values for both tribal and non-tribal people and communities. Historically, tribal communities relied heavily on the Eel River fishery resources for subsistence and were integral to their beliefs of being interconnected with nature and seasonal patterns. The Project licensing provides an opportunity to balance water supply reliability, cultural values, economics, fisheries, and ecological health of the Eel and Russian rivers. The recovery of anadromous fish populations in the Eel River, while maintaining Russian River water supply reliability, would provide tremendous cultural, social, and ceremonial value (both tangible and intangible) for tribal and non-tribal communities.

The objectives of the Feasibility Study ecological and fisheries analyses described in this document were to understand how changes to the Project's infrastructure, streamflow, and restoration would affect the fisheries and ecology of the Eel and Russian rivers. The analyses were intended to inform the Parties regarding how Feasibility Study Alternatives and Water Supply Scenarios would affect fish productivity and river ecology. Based on the analyses presented here, there are a range of ecological and fisheries responses which may occur depending on the Water Supply Scenario or Alternative being considered (Table 1-1) that varies spatially and temporally. To help synthesize key conclusions, a synopsis of primary ecological and fisheries responses to Water Supply Scenarios and Feasibility Study Alternatives (Table 1-1) are provided below for both the Eel River (Table 6-1) and Russian River (Table 6-2). Immediately below we provide a written summary of the key points for the Feasibility Study Alternatives assessed here.

6.1 Alternatives Summary

Retention of Scott Dam and Lake Pillsbury with inclusion of a fish passage strategy would provide access to habitat above Lake Pillsbury. However, fish production would be limited by:

1. Lost habitat inundated by Lake Pillsbury, which includes about 16 km of habitat for Chinook Salmon and 26 km for steelhead (Cooper 2017, Table 6-1);
2. Continued physical and ecological effects of the reservoir, including alteration of natural hydrology and water temperatures, interruption of downstream sediment and wood supplies, and predation on juvenile salmonids; and
3. Reduced fitness or potential mortality of migrating fish associated with reduced upstream and downstream passage efficiency through or around the reservoir (depending on the fish passage strategy selected).

The retention of Scott Dam would provide a continued coldwater pool, which could benefit summer rearing salmonids between Scott and Cape Horn dams. The benefits must be considered in parallel with other effects of water temperature and flow releases from Scott Dam. For example, the coldwater pool likely reduces spring growth rates due to lower water temperatures and can postpone the outmigration timing of juvenile fish, increasing their risk of exposure to high water temperatures in the mainstem Eel River.

Removal of Scott Dam (Alternative 2) would provide unimpeded access to about 150 km of spawning habitat for Chinook Salmon and about 500 km for steelhead, or about 6 percent and 10 percent of historically accessible habitat, respectively (Cooper et al. 2020, Fitzgerald et al. 2020). Estimates of potential production for reaches upstream of Scott Dam range from approximately 800 to 10,000 adult Chinook Salmon and 500 to 25,000 adult steelhead based on a variety of approaches (Table 6-1). Removal of Scott Dam would also provide unimpeded passage for Pacific Lamprey and other native fishes. Removal of Scott Dam would also return a natural flow regime to the Eel River between Lake Pillsbury and Van Arsdale Reservoir, which would restore natural ecological processes.

Retention of Cape Horn Dam in Alternative 2 would still impact flow regimes downstream, require fish to navigate a fish passage system, and provide lentic habitat that supports non-native predators. Removal of Cape Horn Dam and Van Arsdale Reservoir and replacement with an alternative diversion (Scenario 2, Alternative 3) would improve fish passage, restore physical and ecological processes, reconnect fragmented habitat, and improve conditions for native species, but at a greater economic cost and with increased uncertainty in water supply reliability.

We caution that improvements to Project infrastructure and operations will not guarantee restoration of the Eel River watershed's fisheries and ecology. For comprehensive salmonid restoration in the Eel River watershed to be successful, restoration efforts need to be coordinated and use a common framework for identifying and prioritizing restoration actions that will be most effective. In Section 7, we provide a high-level overview of an Eel River watershed restoration strategy.

Table 6-1. Comparison of ecological and fisheries responses to Water Supply Scenarios and Feasibility Study Alternatives for the Eel River. Red boxes represent a negative change or maintaining poor conditions for the target of the analysis. Blue boxes represent a neutral or unclear response (e.g., multi-directional) for the target of the analysis. Green boxes represent positive change or maintaining good conditions for the target of the analysis. All results here are relative to existing infrastructure and Baseline Water Supply Scenario.

Water Supply Scenario	**Scenario 4B (Revised Operations; Modified RPA/FIRO and Fish Flow EIR)**	**Scenario 2 (Run-of-the-river; FIRO and Fish Flow EIR)**	
Feasibility Study Alternatives	**Alternative 1**	**Alternative 2**	**Alternative 3**
Disposition of Dams	**Scott Dam Remains Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Removed with alternative diversion infrastructure**
Flow-Habitat Relationships (Weighted Usable Area [WUA], Water Temperature Was Not Evaluated Here)			
Chinook and Steelhead Spawning Habitat	Chinook and steelhead spawning WUA habitat was greater under Alternative 1 because WUA was greatest at streamflows that occur during the spawning season for both species compared to the other Feasibility Study Alternatives.	Steelhead and Chinook spawning WUA habitat in Scenario 2 would decrease downstream of Scott Dam and Cape Horn Dam relative to Baseline or Scenario 4B operations.	
Juvenile Chinook Rearing Habitat	Retention of both dams will have little effect on Chinook juvenile rearing WUA habitat. Many days with higher and lower streamflows are unable to analyzed due to WUA curve limitations.	Retention of both dams will have little effect on Chinook juvenile rearing WUA habitat. Many days with higher and lower streamflows were unable to analyzed due to WUA curve limitations.	
Juvenile Steelhead Rearing Habitat	Retention of both dams would not significantly reduce WUA habitat downstream of Scott Dam or Cape Horn Dam. However, many days with lower streamflow are unable to be analyzed downstream of Scott Dam.	Steelhead juvenile rearing habitat is expected to increase downstream of Scott Dam.	
		Steelhead juvenile rearing habitat is expected to be similar downstream of Cape Horn Dam.	
Steelhead Fry Rearing Habitat	Moderate increases to steelhead fry WUA habitat are expected under Scenario 4B relative to Baseline but would decrease relative to Scenario 2.	Fry habitat is expected to increase under Scenario 2 downstream of Scott Dam and is due to lower summer and early spring baseflows.	
		Downstream of Cape Horn Dam, steelhead fry habitat is expected to remain similar to Baseline.	

Water Supply Scenario	**Scenario 4B (Revised Operations; Modified RPA/FIRO and Fish Flow EIR)**	**Scenario 2 (Run-of-the-river; FIRO and Fish Flow EIR)**	
Feasibility Study Alternatives	**Alternative 1**	**Alternative 2**	**Alternative 3**
Disposition of Dams	**Scott Dam Remains Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Removed with alternative diversion infrastructure**
Fish Migration			
Critical Riffle Passage	The number of days that riffles are passable by adult Chinook Salmon is on average 5 days fewer than Baseline, out of a total possible of 38 days, but likely not a significant change.	The number of days that riffles are passable by adult Chinook Salmon is on average 7 days fewer than Baseline, out of a total possible of 38 days, but likely not a significant change. In dry years up to 11 days fewer than Baseline, out of a total possible of 26.	
Streamflow Migration Cues	No evidence of significant streamflow changes from Project operations under any Water Supply Scenario at the Scotia gage.	No evidence of significant streamflow changes from Project operations under any Water Supply Scenario at the Scotia gage.	
Pacific Lamprey	Retention of both dams with construction of a fishway at Scott Dam would allow access to large areas of upstream habitat, but would: (1) require design considerations for lamprey passage (2) decrease passage efficiency, and (3) lower survival due to reservoir predation relative to Scenario 2.	Removal of Scott Dam would provide lamprey unimpeded access to large areas of holding, spawning, and rearing habitats and reduce predation on outmigrating juveniles in Lake Pillsbury. Retention of Cape Horn Dam would require a permanent fishway that accounts for lamprey and can withstand high streamflows and sediment loads. Facilitating downstream juvenile passage and survival at Cape Horn Dam would also require consideration.	Removal of both dams would provide lamprey unimpeded access to large areas of holding, spawning, and rearing habitats and reduce predation on outmigrating juveniles in both reservoirs.

Water Supply Scenario	**Scenario 4B (Revised Operations; Modified RPA/FIRO and Fish Flow EIR)**	**Scenario 2 (Run-of-the-river; FIRO and Fish Flow EIR)**	
Feasibility Study Alternatives	**Alternative 1**	**Alternative 2**	**Alternative 3**
Disposition of Dams	**Scott Dam Remains Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Removed with alternative diversion infrastructure**
Non-native predators	Retention of both dams would continue to create warm and lentic habitats that favor Sacramento Pikeminnow and black bass populations and facilitate predation on native fish and increase overlap with juvenile salmonids by restricting their access to coldwater habitats above Scott Dam.	Removal of Scott Dam and Lake Pillsbury would eliminate large areas of high-quality habitat for Sacramento Pikeminnow and black bass, decreasing their populations throughout the upper Eel River.	Removal of both dams would eliminate warm and lentic habitats and infrastructure that favor non-native predator populations and facilitate predation on native fish.
	Maintaining cold water reduces competitive advantage of Sacramento Pikeminnow in the reach between dams relative to Feasibility Study Alternatives 2 and 3.	Retaining Cape Horn Dam and Van Arsdale Reservoir would maintain ideal habitats for predators and predation hotspots associated with infrastructure at these locations.	
		Loss of coldwater releases from Scott Dam is expected to increase competitive advantage of Sacramento Pikeminnow over salmonids in the reach between dams relative to Scenario 4B. These effects may be offset by removal of lentic habitat and predation hotspots, smaller predator populations, and reduced overlap with native fish from juvenile salmonids rearing in tributaries where fewer Sacramento Pikeminnow are present.	
Herpetofauna			
Foothill Yellow-legged Frogs	Continued population fragmentation, reduced habitat quality (by riparian encroachment and coarse sediment deficit) and quantity (by inundation under reservoir). Elevated and cold summer baseflows reduce egg and tadpole development rates. Lentic habitats promote non-native predators that consume tadpoles.	Re-connect fragmented populations, increased habitat availability and quality, naturalized streamflow and temperature regime, and reduced riparian encroachment downstream of Scott Dam would benefit reproduction and development of all life stages.	Re-connect fragmented populations, increased habitat availability and quality, naturalized streamflow and temperature regime, and reduced riparian encroachment downstream of Scott and Cape Horn Dam would benefit reproduction and development of all life stages.

Water Supply Scenario	**Scenario 4B (Revised Operations; Modified RPA/FIRO and Fish Flow EIR)**	**Scenario 2 (Run-of-the-river; FIRO and Fish Flow EIR)**	
Feasibility Study Alternatives	**Alternative 1**	**Alternative 2**	**Alternative 3**
Disposition of Dams	**Scott Dam Remains Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Removed with alternative diversion infrastructure**
Northwestern Pond Turtle (NWPT)	No change to NWPT habitat through retention of Lake Pillsbury. Also retain non-native competitors and predators.	Loss of lentic habitat, but not considered significant enough to affect NWPT, will discourage non-native competitors and predators	
American Bullfrog	Retaining the reservoirs promotes non-native bullfrogs and other non-native aquatic predators to the detriment of native herpetofauna.	Reduce rearing habitat of predator/non-native American Bullfrogs in Lake Pillsbury and downstream to Van Arsdale. Van Arsdale Reservoir would still support American Bullfrogs. Higher winter streamflows could negatively affect American Bullfrogs.	Reduce rearing habitat of predator/non-native American Bullfrogs in Lake Pillsbury and Van Arsdale. Higher winter streamflows could negatively affect American Bullfrogs.
Benthic macroinvertebrates	Cold water released maintain benthic macroinvertebrate productivity currently observed.	Increased winter scour flows and longer spring recessions would likely improve diverse and palatable macroinvertebrate productivity downstream of Scott Dam.	Increased winter scour flows and longer spring recessions would improve diverse and palatable benthic macroinvertebrate productivity downstream of both dams.
Riparian Vegetation	Fewer floods, elevated and stable summer baseflow will promote willow and conifer encroachment.	Increased floods, lower and variable summer baseflows would reduce willow and conifer encroachment.	
Cyanobacteria and algal growth	Retention of both dams may facilitate blooms through stable streamflows and retention of lentic habitats.	Reduced winter scour flows and less variation would retain existing algal growth conditions. Removal of Lake Pillsbury would reduce algal growth conditions in the reservoir.	Removal of Cape Horn Dam would reduce slow-moving lentic habitat and reduce probability of any blooms.

Water Supply Scenario	**Scenario 4B (Revised Operations; Modified RPA/FIRO and Fish Flow EIR)**	**Scenario 2 (Run-of-the-river; FIRO and Fish Flow EIR)**	
Feasibility Study Alternatives	**Alternative 1**	**Alternative 2**	**Alternative 3**
Disposition of Dams	**Scott Dam Remains Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Removed with alternative diversion infrastructure**
Salmonid productivity upstream of Scott Dam	Retention of both dams, with a new fishway at Scott Dam and redesigned passage infrastructure at Cape Horn Dam, would (1) allow access to large areas of spawning and coldwater rearing habitats upstream; and (2) promote life-history diversity, including the potential to establish summer-run steelhead.	Removal of Scott Dam would (1) allow access to large areas of spawning and coldwater rearing habitats upstream; (2) promote life-history diversity, including the potential to establish summer-run steelhead; (3) allow unimpeded passage for adult upstream migration and juvenile outmigration; and (4) decrease predation and competition from Sacramento Pikeminnow and black bass by reducing overlap between non-native predators and salmonids, and eliminate Lake Pillsbury as a predatory population source.	Removal of Scott Dam would (1) allow access to large areas of spawning and coldwater rearing habitats upstream; (2) promote life-history diversity, including the potential to establish summer-run steelhead; (3) allow unimpeded passage for adult upstream migration and juvenile and kelt outmigration; and (4) decrease predation and competition from pikeminnow and bass by reducing overlap between non-native predators and salmonids, and eliminate Lake Pillsbury as a predatory population source. Removal of Cape Horn Dam and Van Arsdale Reservoir would (1) allow unimpeded passage by all life stages at the site and (2) reduce predation on juvenile salmonids by removing predatory fish hotspots.
	Both upstream adult and downstream juvenile salmonid passage efficiency would be reduced through a natural fishway, fish ladder, or "trap and haul" at Scott Dam. Survival of outmigrating juveniles would likely be significantly reduced by imperfect downstream passage and high levels of predation in Lake Pillsbury and Van Arsdale Reservoir.	Retention of Cape Horn Dam would decrease efficiency of adult passage into the newly accessible habitats above Scott Dam and reduce smolt production due to high rates of predation by Sacramento Pikeminnow and black bass at dam infrastructure and in Van Arsdale Reservoir. Downstream passage by both juvenile salmonids and steelhead kelts may also be negatively impacted by the dam and diversion infrastructure.	

Water Supply Scenario	**Scenario 4B (Revised Operations; Modified RPA/FIRO and Fish Flow EIR)**	**Scenario 2 (Run-of-the-river; FIRO and Fish Flow EIR)**	
Feasibility Study Alternatives	**Alternative 1**	**Alternative 2**	**Alternative 3**
Disposition of Dams	**Scott Dam Remains Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Removed with alternative diversion infrastructure**
Salmonid productivity downstream of Scott Dam			
Juvenile Rearing - Scott Dam to Cape Horn Dam	Retained bottom releases with artificially cool water temperatures between Scott and Cape Horn dams would maintain growth and survival benefits to juvenile steelhead in the summer.	Natural hydrograph (upstream of Cape Horn Dam) with warmer spring and summer water temperatures between dams. Reduced growth and survival of juvenile steelhead in the summer, but better growth conditions for both steelhead and Chinook in spring. Improved connectivity with colder upstream habitats and decreased predation due to removal of Scott Dam.	Natural hydrograph (upstream of Cape Horn Dam) with warmer spring and summer water temperatures between dams. Reduced growth and survival of juvenile steelhead in the summer, but better growth conditions for both steelhead and Chinook in spring. Complete connectivity with colder upstream habitats and predation reduced due to removal for both dams and associated infrastructure.
	Coldwater releases may suppress growth of salmonids in the spring. Retaining Scott Dam limits connectivity with colder upstream habitats that provide refuge from non-native predators. High risk of predation due to Van Arsdale Reservoir and Cape Horn Dam infrastructure and larger predator populations.	High risk of predation due to Van Arsdale Reservoir and Cape Horn Dam infrastructure.	
Juvenile Rearing - Downstream of Cape Horn Dam	Spring rearing habitat conditions, growth, and survival could be limited by lower streamflows associated with continued diversion at Van Arsdale relative to Scenario 3.	Spring rearing habitat conditions, growth, and survival could be limited by lower streamflows and higher water temperatures associated with increased diversion at Van Arsdale relative to Scenario 4B and especially Scenario 3.	

Water Supply Scenario	**Scenario 4B (Revised Operations; Modified RPA/FIRO and Fish Flow EIR)**	**Scenario 2 (Run-of-the-river; FIRO and Fish Flow EIR)**	
Feasibility Study Alternatives	**Alternative 1**	**Alternative 2**	**Alternative 3**
Disposition of Dams	**Scott Dam Remains Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Removed with alternative diversion infrastructure**
Juvenile Outmigration - Scott Dam to Cape Horn Dam	Coldwater releases in spring could delay juvenile outmigration, lowering survival due to stressful or lethal temperatures and high predation rates in downstream reaches. Juveniles would need to outmigrate past Cape Horn Dam, increasing injury risk, migration delay, and predation due to infrastructure.	Unimpaired hydrograph and warmer water temperatures would lead to earlier, more natural juvenile outmigration timing and increased survival in reach.	Upstream of Cape Horn Dam, unimpaired hydrology and warmer water temperatures is expected to lead to earlier, more natural juvenile outmigration timing and could increase survival downstream. In the absence of Cape Horn, juveniles could outmigrate unimpeded in a free-flowing channel with reduced predatory fish habitat.
	Retained ability to implement discretionary streamflow releases designed to benefit facilitate outmigration and improve survival.	Juveniles would need to outmigrate past Cape Horn Dam, increasing injury risk, migration delay, and predation due to infrastructure	
Juvenile Outmigration - Downstream of Cape Horn Dam	Juvenile Chinook and steelhead survival during outmigration could be limited by continued spring diversion at Van Arsdale and increased water temperatures relative to Scenario 3.	Juvenile Chinook and steelhead survival during outmigration could be limited by increased spring diversion at Van Arsdale and increased water temperatures relative to Scenario 4B and especially Scenario 3. This impact may be partially off-set by earlier outmigration timing.	
Adult Migration and Spawning - Scott Dam to Cape Horn Dam	Higher early fall streamflows expected to maintain sufficient passage conditions for adult Chinook Salmon in reach. Lower and more stable late fall and early winter streamflows may improve spawning success and embryo survival relative to Scenario 2 and Scenario 3, but uncertain.	Passage of adult Chinook Salmon at shallow riffles may be limited relative to Scenario 4B and Baseline due to lower early fall streamflows, particularly in drier years. Minimal or neutral effects on steelhead migration through riffles expected. Effects of generally higher and more variable late fall and early winter streamflows associated with a natural hydrograph on spawning success are uncertain.	
	Delayed passage of adult salmon and steelhead at Cape Horn Dam.	Delayed passage of adult salmon and steelhead at Cape Horn Dam.	Unimpeded adult salmon and steelhead passage at Cape Horn Dam site.

Water Supply Scenario	**Scenario 4B (Revised Operations; Modified RPA/FIRO and Fish Flow EIR)**	**Scenario 2 (Run-of-the-river; FIRO and Fish Flow EIR)**	
Feasibility Study Alternatives	**Alternative 1**	**Alternative 2**	**Alternative 3**
Disposition of Dams	**Scott Dam Remains Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Remains with existing diversion infrastructure**	**Scott Dam Removed Cape Horn Removed with alternative diversion infrastructure**
Adult Migration and Spawning - Downstream of Cape Horn Dam	Muted early fall streamflow peaks due to the diversion may delay early season migration and reduce passage at critical riffles for Chinook Salmon (relative to Scenario 3, but not Scenario 2).	Muted early fall streamflow peaks and lower baseflows due to larger diversion relative to all other Water Supply Scenarios may delay early season migration and reduce passage at critical riffles for Chinook Salmon. Generally higher and more variable late fall and early winter streamflows compared with Scenario 4B due to loss of reservoir attenuation would have uncertain effects on Chinook spawning success.	
	Minimal or neutral effects on steelhead migration and spawning success expected.		
Discretionary Blockwater	Retained ability to implement discretionary streamflow releases aimed at improving juvenile survival in spring and adult passage in fall.	Lost ability to implement discretionary streamflow releases.	

Table 6-2. Comparison of ecological and fisheries responses to the Feasibility Study Alternatives and Water Supply Scenarios for the Russian River. Red boxes represent a negative change or maintaining poor conditions for the target of the analysis. Blue boxes represent a neutral or unclear response (e.g., multi-directional) for the target of the analysis. Green boxes represent positive change or maintaining good conditions. Alternatives 2 and 3 are grouped in the Russian River because there are not infrastructure changes proposed and the Water Supply Scenario is the same at both locations for the target of the analysis.

Water Supply Scenario	**Scenario 4B (Revised Operations; Modified RPA/FIRO and Fish Flow EIR)**	**Scenario 2 (Run-of-the-river; FIRO and Fish Flow EIR)**	
Feasibility Study Alternatives	**Alternative 1**	**Alternative 2**	**Alternative 3**
Disposition of Dams	**Scott Dam Remains Cape Horn Remains**	**Scott Dam Removed Cape Horn Remains**	**Scott Dam Removed Cape Horn Removed**
Flow-Habitat Relationships (Weighted Usable Area [WUA])			
Juvenile Chinook and Steelhead Rearing Habitat	Minimal change to juvenile salmonid WUA habitat relative to Baseline.	Minimal changes to juvenile salmonid WUA habitat relative to Baseline. Some increases in habitat relative to Scenario 3.	
Steelhead Fry Rearing Habitat	Minimal change to steelhead fry WUA habitat relative to Baseline.	Minimal change to steelhead fry WUA habitat relative to Baseline.	
Fish migration			
Critical Riffle Passage	Slight decreases in number of passable days for Scenario 4B relative to Baseline.	Average decrease of 17 days that are passable under Scenario 2 relative to Baseline.	
Pacific Lamprey	No infrastructure changes and minimal changes to streamflow; no significant change predicted relative to Baseline.	Lower streamflows associated with Scenarios 2 and 3 relative to Baseline and Scenario 4B; could decrease wetted habitat for ammoceates, though this would likely only occur under critical year types. No quantitative analysis was available.	
Non-native predators	No changes are expected to occur for any non-native predator populations or interactions with salmonids.	No changes are expected to occur for any non-native predator populations or interactions with salmonids.	

Water Supply Scenario	**Scenario 4B (Revised Operations; Modified RPA/FIRO and Fish Flow EIR)**	**Scenario 2 (Run-of-the-river; FIRO and Fish Flow EIR)**	
Feasibility Study Alternatives	**Alternative 1**	**Alternative 2**	**Alternative 3**
Disposition of Dams	**Scott Dam Remains Cape Horn Remains**	**Scott Dam Removed Cape Horn Remains**	**Scott Dam Removed Cape Horn Removed**
Herpetofauna	Scenario 4B has stable streamflows which promote non-native American Bullfrogs that prey on native herpetofauna.	Lower streamflows under Scenario 2 could give native FYFL and NWPT a competitive advantage over non-natives.	
Benthic macroinvertebrates	No major changes are anticipated for benthic macroinvertebrates.	In general, no major changes are anticipated to occur for benthic macroinvertebrates.	
Riparian habitat	Higher summer baseflows likely promote riparian encroachment but would not change relative to Baseline.	Higher peak flows and lower summer baseflows under Scenario 2 would increase scour and reduce encroachment, but changes are anticipated to be minimal.	
Cyanobacteria and algal growth	No change expected relative to Baseline, which does not promote cyanobacterial algal growth.	No significant change expected to occur relative to Baseline.	

7 FISHERIES RESTORATION STRATEGIES

Fish habitat and production potential throughout much of the Eel River watershed have been significantly impaired from past and recent intensive land use practices (NMFS 2014, 2016; Eel River Forum 2016). Notable recent and ongoing land uses in the watershed include grazing, industrial timber management, rural and residential development, gravel extraction, and widespread cannabis cultivation. These activities, along with historical widespread disturbance of the landscape from intensive logging, forest community alteration, and construction of roads and railways, followed by large floods in the 1955 and 1964, have caused extensive changes to much of the watershed. Impacts from these changes include widespread landslides, channel aggregation and simplification, loss of riparian vegetation, increased water temperatures, and altered hydrology. These alterations have contributed to widespread fish habitat loss and population declines (CDFW 2014; NMFS 2014, 2016; Eel River Forum 2016). Additionally, the United States Environmental Protection Agency (USEPA) has listed all Eel River subwatersheds as impaired on the federal Clean Water Act 303(d) list, primarily for excessive sediment and increased water temperatures (Eel River Forum 2016). The ecology of the Eel River watershed has also been fundamentally altered by presence of non-native species, most notably Sacramento Pikeminnow (Section 4.5).

In response to the pervasive fish habitat degradation and population declines, the Parties propose to investigate additional restoration and conservation opportunities within the Eel River watershed beyond the Project boundary that will further contribute to the recovery and preservation of tribally, commercially, and recreationally important fisheries. These potential actions would be in addition to the restoration actions slated to occur within the Project boundary under a new FERC license, but are entirely separate from any agreements or measures required by a new FERC license. Implementing major coordinated restoration programs both within and outside of the Project boundary is critical for recovering native fish populations within the entire Eel River watershed and reinvigorating tribal, commercial, and recreational fisheries.

This section provides a concise summary of key fisheries restoration strategies and concepts for the Eel River watershed outside the Project boundary but does not address actions within the Project boundary. For the purposes of this assessment, the Project boundary is defined as the inundation footprint of Lake Pillsbury and the mainstem Eel River from Scott Dam downstream to the Middle Fork Eel River confluence. Accessible stream reaches upstream of Lake Pillsbury are also excluded from this summary, as are Project operations (e.g., flow releases) and infrastructure (dams and diversions), which are addressed in the Feasibility Study Report (Stillwater Sciences et al. 2020). The Fisheries Restoration Plan for the Feasibility Study includes the following restoration actions to be undertaken within the Project boundary:

1. Restoration of anadromous fish access to habitat upstream of Scott Dam via removal of Scott Dam;
2. Management of sediment and vegetation in the Lake Pillsbury footprint to restore historic riverine and riparian habitat along the Eel River and minimize effects to aquatic resources downstream of Scott Dam;
3. Restoration of natural physical and biological processes within the reservoir footprint and reaches downstream of Scott Dam via removal of Scott Dam and additional restoration actions;
4. Modifications to Cape Horn Dam to improve upstream and downstream fish passage; and
5. Modifications to Van Arsdale Diversion infrastructure to reduce risk of fish entrainment.

Fisheries restoration opportunities within the Project boundary are also outlined in the NMFS (2016) Coastal Multispecies Recovery Plan and the Eel River Action Plan (Eel River Forum 2016).

The fisheries restoration strategies described herein focuses on native fish species in the Eel River watershed with current or historical commercial, recreational, or tribal harvest values, including fall-run Chinook Salmon, Coho Salmon, summer-run/winter-run steelhead, Pacific Lamprey, and Green Sturgeon (*Acipenser medirostris*). Restoration and preservation of habitats and ecological processes that benefit other native fish and aquatic species should also be considered as part of a holistic restoration planning process.

Salmonid restoration in the Eel River watershed is more likely to succeed if restoration efforts are coordinated and use a common framework for identifying and prioritizing restoration actions that will most effectively increase populations of focal species and restore ecological processes. The restoration strategies, concepts, and project types discussed herein are based in part on existing frameworks (e.g., Bradbury et al. 1995) and fisheries recovery plans developed by state and federal (NMFS 2014, 2016) resource agencies, Indian tribes, the Eel River Forum (2016), and other stakeholders. Importantly, this initial fisheries restoration strategy does not provide a full list of restoration actions or attempt to identify specific locations for implementation. Development of a comprehensive fishery restoration strategy for the Eel River watershed that identifies, describes, and prioritizes specific locations and actions needed to most efficiently restore focal fish populations will require a long-term, coordinated effort between the Parties, state, federal, and tribal entities, and other key stakeholders in the basin (e.g., Eel River Forum, NGOs). Accordingly, the strategy proposed herein will be refined and expanded with input from these stakeholders.

This section outlines six key steps for developing a comprehensive and coordinated watershed-wide restoration strategy for the Eel River and provides initial ideas and concepts that can serve as a starting point for each step.

1. *Select restoration planning areas:* Divide the watershed into geographical focal areas (or subwatersheds) to serve as the starting point for planning, prioritization, implementation, and monitoring processes.
2. *Identify key limiting factors:* Develop and apply a framework for identifying key factors limiting population productivity (i.e., population bottlenecks) for each species within each restoration planning area. Targeting restoration actions that most directly address habitat limitations or improve survival of life stages that will increase the number of returning adults is the most cost-effective means of working toward recovery of a population. However, within this concept, it is important to recognize the importance of restoring a range of habitats and ecological processes that maintain and improve overall life history diversity and population resilience.
3. *Identify and describe key restoration actions*: Identify types of restoration actions that will most directly and efficiently address limiting factors (actions expected to result in increased smolt production and adult returns) for each species in the focal area.
4. *Prioritize project types and locations for implementing restoration*: Develop and apply a prioritization process for identifying locations where implementing key restoration actions will have the greatest benefit to focal fish populations.
5. *Implement restoration projects*: Design, permit, and construct priority habitat enhancement and restoration projects.

6. *Monitor Restoration:* Monitor project evolution and fish population and habitat response to restoration.

If implemented at an appropriate scale and timeframe, such a framework and the broad restoration strategies discussed herein would collectively advance recovery of native fish populations throughout the Eel River watershed.

7.1 Select Restoration Planning Areas

A watershed-wide fisheries restoration strategy for a large river basin, such as the Eel River, should be divided into focal planning areas within which key restoration actions can be identified, prioritized, and implemented. Ultimately, the boundaries of these restoration planning areas should be selected with consensus from fisheries resources agencies and other key stakeholders based on factors such as distribution of focal fish species, climate, geology, hydrology, geomorphic characteristics, existing restoration planning processes, and the jurisdiction and shared interest of stakeholders. As a starting point for discussion of this topic, the Eel River watershed could be divided into the following planning areas:

- Lower Eel River and Estuary, including the mainstem Eel River and tributaries downstream of the Van Duzen River;
- Van Duzen River watershed;
- Lower Mainstem Eel River, including the mainstem Eel River and tributaries from the South Fork Eel River downstream to the Van Duzen River confluence;
- South Fork Eel River watershed;
- Middle Mainstem Eel River, including the mainstem Eel River and tributaries from the Middle Fork Eel River downstream to the South Fork Eel River confluence;
- North Fork Eel River watershed;
- Middle Fork Eel River watershed.
- Upper Mainstem Eel River, including the mainstem Eel River and tributaries from Scott Dam downstream to the Middle Fork Eel River confluence; and
- Upper Eel River watershed, including the mainstem Eel River and tributaries upstream of Scott Dam.

As described in Section 7.4, as part of an overall Eel River watershed fisheries restoration strategy, planning areas thought to be most important for protecting and recovering focal fish populations may be initially prioritized for focusing restoration resources. A restoration prioritization process would then be applied within priority planning areas to systematically identify both general (e.g., subwatersheds or reaches) and specific locations where implementing restoration would most efficiently help recover focal fish populations.

7.2 Identify Key Limiting Factors

For fisheries restoration strategies to be most successful and cost-effective, they should focus on the life stages of focal species for which habitat carrying capacity, growth, or survival limit population productivity. This limiting factors framework allows identification of restoration actions most likely to have direct benefits to focal species populations by increasing adult returns or building resilience by promoting life history diversity. Species-specific life-cycle conceptual models that integrate life history, habitat capacity, growth, and survival (e.g., Williams 2010) could be used to systematically identify population bottlenecks and determine which life stages should be targeted in different parts of the watershed. Such models could also be used as a tool to evaluate the relative benefits of proposed actions.

In the absence of these specific conceptual life-cycle models, state and federal recovery plans (e.g., NMFS 2014, 2016) and other research provide a general understanding of life history, population dynamics, habitat conditions, and key threats to focal, native fish species in the Eel River watershed. This body of existing work indicates that habitat carrying capacity and survival during the following life stages are likely to limit salmon and steelhead population productivity and therefore should be the primary focus of restoration efforts (in the general locations specified):

- *Fall-run Chinook Salmon*: spawning habitat quality, embryo incubation, juvenile growth, and smolt outmigration survival (mainstem Eel River, major tributaries, and the estuary);
- *Coho Salmon*: juvenile summer and winter rearing habitat, juvenile growth, and smolt outmigration survival (cold, low-gradient tributaries to the South Fork Eel and Van Duzen rivers, mainstem river corridors downstream of spawning streams, and the estuary);
- *Winter-run steelhead*: age-1 and older juvenile summer and winter rearing habitat, juvenile growth, smolt outmigration survival, and kelt outmigration survival (throughout accessible and thermally suitable reaches in the watershed and estuary); and
- *Summer-run steelhead*: age-1 and older juvenile summer and winter rearing habitat, juvenile growth, smolt outmigration survival, kelt outmigration survival, and adult holding (accessible and thermally suitable reaches in the Middle Fork Eel and Van Duzen rivers and estuary).

As with salmonids, more complete life-cycle models that integrate new data are needed to identify key limiting factors for Pacific Lamprey and Green Sturgeon in the Eel River. An initial limiting factors conceptual model for Pacific Lamprey indicates that the following are likely among the most important freshwater factors limiting Pacific Lamprey adult returns in the Eel River watershed, and thus should be the focus of restoration (Stillwater Sciences 2014):

- Adult access to spawning habitats interrupted by migration barriers in tributaries;
- Availability of stable fine-sediment larval rearing habitats and survival and growth during the larval stage; and
- Survival of juveniles during outmigration to the ocean.

Additional studies are needed to better understand the factors that limit the size of the Green Sturgeon population in the Eel River, but available information (Israel and Klimley 2008, Stillwater Sciences and WNRD 2017) suggest they may include:

- Quantity and quality of unimbedded cobble and large gravel spawning substrates in the mainstem Eel River (egg and embryo survival are likely impaired by fine sediment);
- Access to higher quality mainstem Eel River spawning habitats (generally upstream of the South Fork Eel River confluence) can be restricted by low streamflows in some years; and
- Reduced growth and survival during the larval and fry stages due to high fine sediment loads, lower dissolved oxygen, suboptimal water temperatures, and predation.

In addition to identifying targeted restoration actions that most directly and efficiently increase population productivity of native focal fish species in focal planning areas, a holistic restoration strategy should (1) recognize the importance of restoring a range of habitats across the watershed to help maintain and improve life history diversity and overall population resilience and (2) strive to concurrently address and remediate the root causes of aquatic ecosystem degradation on a landscape-scale.

7.3 Identify and Describe Key Restoration Actions

Ultimately, restoration actions that directly address key limiting factors and increase populations of each focal fish species should be identified and described based on existing information and utilizing the species conceptual models described above. Based on known threats to focal species and existing recovery plans and restoration planning processes, fisheries restoration actions should strive to achieve the following:

- Reduce fine sediment inputs, promote substrate sorting, and reduce fine sediment storage in substrates for all species.
- Increase low-velocity winter and spring rearing habitats for juvenile Coho Salmon.
- Improve or maintain thermal suitability of summer rearing habitats for Coho Salmon and steelhead.
- Increase or maintain natural instream flows that support focal species ecological needs, particularly during the spring and summer.
- Increase growth and survival of rearing juvenile salmonids and outmigrating smolts.
- Develop and implement effective methods to control the Sacramento Pikeminnow and other non-native predators.
- Restore the estuary.

Restoration actions that help achieve each of these objectives and the portions of the Eel River watershed and general channel features where they should be applied are summarized below.

Notably, in addition to the restoration-focused actions described below, identification and preservation of stream reaches and/or subwatersheds with existing high-quality fish habitats should be an integral part of a larger, holistic strategy to recover fish populations in the Eel River watershed.

7.3.1 Reduce Fine Sediment Inputs and Promote Substrate Sorting

The Eel River has the highest recorded average suspended sediment load per unit area of any river of its size or larger in the lower U.S. (Lisle 1990). This very large fine sediment load impacts both spawning and rearing habitat quality and juvenile growth. Elevated fine sediments can decrease embryo survival during incubation by reducing intragravel permeability, dissolved oxygen concentrations, and the rate of removal of metabolic wastes (Ringler and Hall 1975, Taggert 1976). Elevated fine sediment can also reduce the availability of interstitial spaces used by steelhead and other salmonids for cover and high-flow refuge (Bustard and Narver 1975, Ligon et al. 2016). Other effects of elevated coarse and fine sediments may include reduced benthic macroinvertebrate production, pool filling, and habitat simplification (Crouse et al. 1981). High fine sediment loads can also serve to reduce the growth and survival of juvenile salmonids (Suttle et al. 2004).

This restoration action is primarily focused on ameliorating sediment impacts on egg-to-fry survival of Chinook Salmon in mainstem reaches and larger spawning tributaries and increasing unembedded cobble-boulder winter rearing habitat for juvenile steelhead throughout the watershed. Sediment impacts to these life stages are considered important factors limiting population productivity of these species. Reducing fine sediments throughout the mainstem Eel River is also expected to be particularly important for Green Sturgeon egg-to-fry survival. The

Coastal Multispecies Recovery Plan (NMFS 2016) outlines these actions related to fine sediment reduction:

- Investigate the degree to which fine sediment limits spawning habitat quality and smolt production; conduct a literature review and, if needed, bulk sample stream substrates to better define the impacts of fine sediment.
- Coordinate with state and federal agencies and other partners to develop stream-specific plans that identify and prioritize opportunities to reduce fine sediment contributions, such as road sediment source assessments, road upgrades or removals, and improvements in timberland management.
- Implement prioritized sediment reduction projects in coordination with watershed partners.

Additional actions to reduce fine sediment include upgrading or removing damaged or improperly sized culverts and improving management of roads and other developments associated with cannabis cultivation.

In addition to addressing sediment sources, spawning and winter rearing habitat quality can also be improved by implementing restoration projects that increase overall channel complexity to promote sediment sorting. For example, adding large wood to simplified channels can create hydraulic complexity; promote local scour, deposition, sediment sorting, and in-channel sediment storage; and meter sediment transport (Wohl and Scott 2016).

7.3.2 Increase Low-Velocity Winter Rearing Habitats

This restoration strategy is focused primarily on increasing winter carrying capacity, survival, and growth of juvenile Coho Salmon, but restoration is also expected to benefit juvenile steelhead by providing winter refuge habitat and creating deep pools for summer rearing. Associated benefits related to restoring low-velocity rearing habitats include fine sediment deposition on floodplains and sorting and deposition of spawning gravels, which would benefit all focal species. Since this action is primarily aimed at restoring winter rearing habitat for Coho Salmon, it should be focused in cold, low-gradient, and unconfined tributaries to the Lower Eel, South Fork Eel, and Van Duzen rivers and mainstem river corridors downstream of spawning areas. Opportunities to improve Coho Salmon winter rearing habitat in tributaries upstream of the South Fork Eel River (e.g., Outlet and Tomki creeks) should also be considered if winter habitat capacity is deemed to limit productivity in those watersheds.

Increasing low-velocity winter refuge habitat includes restoration actions designed to improve both instream habitats (i.e., increasing pool frequency by creating pools within other habitat units or adding complexity to existing pools) and off-channel habitats (e.g., creating alcoves, side-channels, or off-channel ponds and improving floodplain connectivity). Restoration actions needed to achieve these different types of winter refuge habitat will depend on existing habitat limitations and physical attributes of the channel, but may include:

- *Accelerated wood recruitment*. This involves placing unanchored whole trees or rootwads into the channel with minimal engineering by directional falling of riparian trees or placement of wood with heavy equipment (Carah et al. 2014). This approach can be an effective alternative that reduces costs related to engineering, design, permitting, administration, and transport (Carah et al. 2014).
- *Engineered instream wood structures*. This action entails methodical design and construction of wood structures to achieve site-specific restoration objectives. Engineered wood structures designed to create low-velocity winter rearing habitat for Coho Salmon are typically intended do one or more of the following: (1) create complex, low-velocity pool

habitat in areas currently lacking complex pools, (2) help create or reconnect side-channels or alcoves, (3) improve access by juvenile fish to low velocity floodplain habitat across a range of streamflows, and (4) provide escape cover for juveniles to reduce predation.

- *Creation of off-channel habitats and floodplain connectivity*. Coho Salmon occur at higher densities and have higher survival rates in off-channel habitats compared with instream habitats (Nickelson et al. 1992, Lestelle 2007), which appears to be due to a combination of superior refuge from high flows and turbidity as well as improved feeding and growth conditions (Lestelle 2007). Ponds formed by oxbows or beavers or constructed through restoration have the highest Coho Salmon winter survival and growth values reported in the literature and typically have a large carrying capacity relative to main channel habitats (Lestelle 2007). Alcoves and backwaters have similar characteristics that create protected over-wintering refuge for juvenile Coho Salmon. For these reasons, creating, enhancing, and providing access to off-channel habitats should be a high priority action. Restoration actions aimed at improving the quality and quantity of low-velocity habitat for winter rearing of juvenile salmonids in off-channel and floodplain areas may include (1) creating and maintaining connected, complex side channels or alcoves; (2) creating off-channel ponds; and (3) increasing overall frequency and duration of floodplain connectivity with the main channel during winter flows. In reaches that have been channelized or incised, channel spanning engineered log jams or Beaver Dam Analogues (BDAs) may be used to facilitate upstream sediment accumulation and increase the frequency at which flow overtops the banks. These features can provide greater access to high-quality winter rearing habitats, increase deposition of fine sediment on the floodplains, promote groundwater recharge, and generally dissipate stream energy.

In many instances, these categories of winter refuge habitat restoration will overlap and work in concert. For example, a strategically placed large wood jam may create a pool while improving connectivity with adjacent floodplains and constructed off-channel habitats.

7.3.3 Improve or Maintain Thermal Suitability of Summer Rearing Habitats

This restoration strategy is focused primarily on increasing summer carrying capacity, survival, and growth of juvenile Coho Salmon and steelhead at locations where water temperatures are suboptimal. The primary strategies for improving or maintaining thermal suitability of summer rearing habitat for Coho Salmon include restoration of the riparian canopy, augmenting summer streamflows and hyporheic exchange (Section 7.3.4), and locating and restoring habitat near coldwater refugia such as cold tributaries or seeps. Increasing connectivity between spawning and winter rearing habitats and thermally suitable summer rearing habitats by removing barriers to juvenile fish migration is another important strategy for maximizing suitable summer rearing habitat.

7.3.4 Increase or Maintain Instream Flows

Identifying instream flow needs for focal species and implementing actions to protect and augment instream flows during the spring recession and summer and fall dry season are critical restoration strategies in the Eel River watershed, which experiences large inter-annual variation in precipitation and streamflow and is vulnerable to severe and prolonged drought related to climate change, water diversions, and other factors. Actions that protect and enhance instream flows help maintain thermal suitability, delivery of high-quality food resources, habitat quality, reduce predation, and passage conditions for salmonids and other native species. These actions may include the following:

- Reducing legal and illegal surface water diversions and implementing instream flow policy based on the best available flow-ecology science, water use efficiency engineering, permitting, and enforcement.
- Implementing water conservation projects aimed at reducing consumptive water use in the spring and summer and improving the efficiency of water distribution systems.
- Creating off-channel water storage (e.g., tanks and ponds) that is supplied during the wet season and allows forbearance of surface water diversion during the dry season.
- Implementing upslope forest management practices that reduce evapotranspiration and increase or extend groundwater storage.
- Improving roads and road drainage infrastructure to reduce accelerated rainfall runoff.
- Implementing in-channel or off-channel projects that help elevate or recharge the water table, such as BDAs and large wood jams.
- Increasing overall channel complexity and floodplain connectivity through wood loading and other instream restoration actions.
- Managing high-density, even-aged riparian forest stands to help offset the impacts of elevated evapotranspiration rates on shallow groundwater.
- Monitoring streamflow and water temperature to inform subwatershed-scale water management, policy, and compliance.

7.3.5 Increase Growth and Survival of Juveniles and Smolts

In addition to physical habitat carrying capacity for key life stages, fish growth rate is an important element of overall population productivity for anadromous fish (Section 4.7). Size at outmigration to the ocean has been correlated with ocean survival, and thus juvenile growth conditions are an important factor affecting the number of adult salmon and steelhead that return (Peterman 1982, Ward et al. 1989, Bond et al. 2008, Jones et al. 2014, Zimmerman et al. 2015). For this reason, improving access to abundant and high-quality food resources and water temperatures that optimize growth rates is an important restoration strategy for each of the target species. To achieve this strategy, more research is needed to understand seasonal food resources and growth rates of focal species in different parts of the basin and their relationship to instream flows. This research and overall understanding of fish food and growth could be informed by the life-history conceptual models described in Section 7.2. As the factors limiting growth (e.g., food resources, streamflow, temperature) are better understood, the knowledge can be incorporated into the various restoration strategies and specific actions described herein to increase growth and survival for focal species.

One of the most direct and important strategies for improving juvenile growth and increasing smolt and ocean survival is restoring the lower mainstem Eel River and estuary. Due to their abundant food resources and position before ocean entry in the anadromous life cycle, estuaries are extremely important habitats for increasing fish growth and population productivity. Increasing survival of juvenile salmonids and other priority fish species rearing in and migrating through the lower Eel River and estuary is critical for improving basin-wide productivity. Estuaries provide critical rearing habitats and a transition zone for anadromous species undergoing physiological changes before entering the ocean (Bond et al. 2008, CDFG 2010, Hughes et al. 2014). Historically, the lower mainstem Eel River and estuary provided high-quality refugia and feeding opportunities and supported high densities of juvenile salmonids (CDFG 2010).

Key objectives for improving fish productivity in the lower mainstem and estuary include implementing actions that improve physical habitat quantity and quality for fry and pre-smolt rearing and increase growth of pre-smolts and survival of smolts during outmigration. Priority habitat restoration actions in the lower mainstem Eel River and estuary would include and build upon those being implement in several ongoing large-scale restoration projects, including the Salt River Ecosystem Restoration Project, the Eel River Estuary Preserve, the Ocean Ranch Unit of the Eel River Wildlife Area, and the Cannibal Island Restoration Area. These projects include removing or setting back levees, removing or modifying tide gates to restore tidal exchange and improve fish access to slough habitats, and restoring slough connectivity to tidal marshes and floodplains. These actions will increase the overall quality and extent of estuarine habitats and should be expanded into additional portions of the estuary. Restoration actions that improve juvenile fish habitat in the freshwater-estuarine transition zone are also needed, particularly improving access to off-channel and floodplain habitats and increasing overall habitat complexity in the mainstem channel (e.g., more frequent and deeper pools and more cover from predators).

To help prioritize restoration actions in the estuary, additional research and monitoring are needed to better understand the factors limiting growth and survival of juvenile fish there. These actions include monitoring seasonal distribution of focal species and better describing the conditions influencing survival and growth in the estuary (e.g., water quality, food production, habitat availability, predation etc.). Once limiting factors have been identified, the priority would be to implement actions that most efficiently address them. There is also a need to systematically monitor and assess fish populations in the estuary to track progress made toward addressing limiting factors.

7.3.6 Develop and Implement Effective Methods to Control Sacramento Pikeminnow and Other Non-Native Predators

As described in Section 4.5, predation and competition by non-native Sacramento Pikeminnow are important factors limiting recovery of salmonids, lampreys, and other native fish species throughout the Eel River watershed. Addressing this issue is critical for improving survival of depressed native fish populations while physical habitat restoration actions are implemented on a large scale. Priority actions to address this issue include:

- Evaluate efficacy and lessons learned from past and current Sacramento Pikeminnow abundance monitoring and suppression efforts in the Eel River (e.g., Higgins 2020, PG&E 2020, Stillwater Sciences and WNRD 2020) and elsewhere in coordination with state and federal agencies, the Wiyot Tribe, PG&E, Eel River Recovery Project, and other partner organizations.
- Build on ongoing efforts to develop and implement a coordinated, holistic, and adaptive plan to actively suppress the Sacramento Pikeminnow population.
- Develop a coordinated approach to monitor trends in abundance of Sacramento Pikeminnow to track population response to suppression activities, streamflow management, environmental conditions, and restoration actions.
- Evaluate the feasibility of implementing a "Trojan Y" chromosome strategy to suppress or eradicate Sacramento Pikeminnow from the Eel River watershed.
- Expand understanding of Sacramento Pikeminnow seasonal movements, diet, spawning behavior and locations, and predation hotspots to inform and focus suppression strategies.

7.4 Prioritize Project Types and Locations for Implementing Restoration

In coordination with watershed stakeholders, an initial prioritization process should be developed and applied to systematically and objectively identify the planning areas (Section 7.1) in the Eel River where resources for restoration and conservation should be focused. This large-scale prioritization is needed to target portions of the watershed where restoration is most feasible and likely to have the largest and most rapid effect in terms of increasing overall productivity and resilience of focal fish populations. This initial prioritization of planning areas would likely take into consideration existing species presence and abundance, fish habitat conditions and intrinsic potential to support focal species, water temperature and stream flow constraints, level of watershed and channel disturbance, relative level of opportunity and constraints for implementing restoration (land ownership, accessibility, existing planning and project momentum etc.), and other ranking factors determined in coordination with watershed stakeholders.

After restoration planning areas are identified, a prioritization process should be developed and applied to each to objectively identify both general areas (e.g., subwatersheds or reaches) and specific locations where key restoration actions should be implemented to efficiently address factors limiting fish population productivity. The intrinsic potential of channels in the river network to support target species and life stages would be informed by fish distribution data, hydrology, sediment supply, drainage area, channel gradient and confinement, grain size, and water temperature. These factors, along with information on habitat conditions and ecological conditions (e.g., Sacramento Pikeminnow presence) would be evaluated to inform overall restoration need and appropriateness of different restoration actions for reach (e.g., channel size and confinement dictates suitability and design for large wood structures). The project selection and prioritization process should also consider cost and feasibility of implementing needed restoration actions based on factors such as accessibility, landowner cooperation, and permitting constraints.

Both for the initial large-scale prioritization and within planning-area prioritization process, species-specific life-cycle models and/or decision support tools could be applied to evaluate overall restoration potential and the relative benefits of various restoration strategies or proposed actions.

This prioritization process should integrate information from, be coordinated with, and build upon existing recovery planning and restoration prioritization efforts occurring in the watershed. In the South Fork Eel River watershed, for example, NMFS, CDFW, and other stakeholders recently implemented a restoration prioritization process for Coho Salmon, Chinook Salmon, and steelhead called Salmon Habitat and Restoration Priorities (SHaRP). That effort builds on federal recovery plans, available fish habitat and population monitoring data, and input from locally knowledgeable stakeholders to identify and prioritize the fish habitat restoration actions and locations within selected high priority subwatersheds, called tributary groups. An initial scoring matrix was developed to prioritize 19 tributary groups in the South Fork Eel River watershed based on biological importance, habitat condition, optimism and potential, and integrity and risk. NMFS and CDFW brought together technical experts, restoration practitioners, and stakeholders with knowledge of the highest-ranking tributary groups to identify and prioritize site-specific restoration needs, actions, and key locations. The products from SHaRP, including a prioritized list of site-specific actions for each tributary group, are anticipated in 2021. The agencies intend to apply the process more broadly across portions of the Eel River watershed and other coastal watersheds in northern California. While the restoration prioritization framework described herein is intended to be applied at a larger scale and consider non-salmonid species, it could integrate relevant/applicable aspects of the SHaRP process.

7.5 Implement Restoration Projects

After identifying high-priority projects and locations, the following provides a broad and general stepwise process for successfully implementing habitat restoration projects:

- Confirm that the potential project type and location are included in the focus and eligibility requirements of state and federal programs that fund fish habitat restoration.
- Conduct outreach and coordinate with local stakeholders and agency representatives to develop project concepts, gain support for the project, and understand linkages to related projects in the area.
- Obtain funding.
- Assess current and historical project site conditions, conduct appropriate technical analyses (e.g., hydrologic, hydraulic, geomorphic, and biological), and engineer project design elements within a process that leads to design plans, specifications, and costs suitable for permitting and construction implementation.
- Obtain regulatory permits required by local, state, and federal agencies.
- Construct the project.
- Conduct effectiveness monitoring (Section7.3.6).
- Disseminate project information to help inform future restoration projects in the planning area and beyond.

7.6 Monitor Biological and Physical Response to Restoration

Regular and long-term fish population monitoring is essential for tracking status of focal fish species, gaging overall success of a restoration program at recovering their populations, and adapting geographical and species emphasis of restoration accordingly (i.e., devoting limited funding where it is needed most). The primary ongoing salmon and steelhead population monitoring programs in the Eel River that have and continue to help address this need include:

- Long-term counts of adult salmon and steelhead at Van Arsdale Fisheries Station in the Upper Eel River (1922–present).
- Ongoing salmonid monitoring:
    - Juvenile steelhead summer rearing monitoring conducted annually since 2005 in late summer by backpack electrofishing and snorkel surveys at established index sites in the mainstem Eel River between Cape Horn Dam and the Middle Fork Eel River (PG&E 2018).
    - Chinook Salmon carcass surveys conducted annually since 1978 in index reaches of the upper mainstem Eel River and Tomki Creek (PG&E 2017b).
- Sonar monitoring of adult salmon and steelhead abundance conducted in the lower mainstem Eel River above the South Fork confluence and in the lower South Fork Eel River during the migration periods in 2019–2021 water years (Kajtaniak and Gruver 2020, Metheny 2020).
- Annual fall snorkel surveys conducted in the lower mainstem Eel River, focused on adult Chinook Salmon (Eel River Recovery Project 2011–2018).
- Annual Coho Salmon spawning surveys conducted in the South Fork Eel River since 2010 as part of the California Coastal Salmonid Monitoring Program (Guczek et al. 2019)

- Annual summer-run steelhead adult snorkel counts in the Middle Fork Eel River (1960s–present) and Van Duzen River (2011–present).

To monitor progress toward fisheries recovery, it is important that these and other monitoring programs in the basin are continued and ideally expanded to include other key watersheds, such as the Van Duzen River, where little regular salmonid monitoring occurs.

There have been a recent and ongoing counts of adult Pacific Lamprey passing the new lamprey passage system at Cape Horn Dam in the upper Eel River (Goodman and Reid 2017), as well as recent efforts to study and monitor Pacific Lamprey in lower portions of the Eel River watershed (Stillwater Sciences and WNRD 2016). Recent research has also been conducted to improve understanding of the status of Green Sturgeon in the Eel River (Stillwater Sciences and WNRD 2017). However, additional and more regular and widespread monitoring is needed to adequately track the status of these species and their response to restoration strategies in this large river system. In addition to funding and expanding existing monitoring programs, one means of expanding population monitoring data for Pacific Lamprey and Green Sturgeon is to extend ongoing and planned sonar monitoring through the early summer period to capture the migration periods of these species.

In addition to the regular population-level monitoring described above, monitoring of physical and biological responses to specific restoration projects is needed, both to determine the level of success and to adapt similar restoration projects in the future. Coordinating with and expanding on ongoing efforts to monitor water temperatures and instream flows throughout the Eel River watershed is also critical, both to identify locations to target enhancement and for monitoring response of these variables.

Three primary ways to monitor the success of restoration projects, based on Duffy (2006), include:

- *Implementation monitoring*: monitoring to document the fulfillment of contract obligations or compliance with regulations or laws.
- *Effectiveness monitoring*: monitoring to document trends in resource condition following a management action. Effectiveness monitoring is most often associated with physical or chemical processes and habitats.
- *Validation monitoring:* monitoring to document the response of biota to restoration actions. Validation monitoring ideally establishes cause-and-effect relationships between restoration actions and biota.

Ideally, an integrated watershed-wide fisheries and restoration monitoring program would be developed and funded to help design, plan, and coordinate needed fish, habitat, and restoration monitoring efforts and promote adaptive management of restoration efforts across the Eel River watershed.

7.7 Summary

The strategy outlined above provides an initial strategic framework for fisheries restoration in the Eel River watershed. This strategy is intended to address fundamental constraints and opportunities from the watershed’s headwaters downstream to its estuary and to identify the most cost-effective and feasible fishery restoration actions. The Eel River watershed is large, as are its restoration needs. The foundational plans that address fisheries recovery (NMFS 2014, 2016; Eel River Forum 2016) recommended habitat restoration throughout the Eel River watershed, although these plans do not precisely identify and prioritize specific projects. Implementation of a comprehensive restoration strategy for the Eel River and restoring fisheries in the watershed will take many decades of concerted effort on behalf of the Parties and watershed partners.

Fundamental to the Shared Objectives of the Two-Basin Solution is improving habitat conditions in the Eel River to support significant recovery of anadromous fish populations. To achieve restoration goals that lead to significant fishery recovery, restoration needs to be coordinated and substantial to make a significant improvement to fish populations. The fisheries restoration strategies described above (Section 7) provide an initial framework for developing a comprehensive restoration plan and process for the Eel River watershed. This strategy is intended to address fundamental constraints and opportunities from the watershed's headwaters downstream to its estuary and to identify the most cost-effective and feasible fishery restoration actions. The Eel River watershed is large, as are its restoration needs. The foundational plans that address fisheries recovery (NMFS 2014, 2016; Eel River Forum 2016) recommended habitat restoration throughout the Eel River watershed, although these plans do not precisely identify and prioritize specific projects. Implementing the strategy described above will not only build from these important foundational plans, but take the next step towards developing a logical and strategic transition from broad restoration goals to watershed-specific and site-specific restoration and preservation projects.

Successful restoration leading to species recovery will also require an efficient management and decision-making process, substantial stakeholder engagement, and employ a scientific basis to inform decision-making (another Shared Objective). A comprehensive and coordinated approach to restoration will also focus restoration needs where it is most likely to result in meaningful improvements, therefore making the most efficient and effective use of available resources. In addition, a coordinated approach will leverage new and existing partnerships to encourage additional investment in restoration and recovery. Large-scale coordinated restoration is needed to support ecological processes and promote synergistic effects that can be realized with a properly functioning ecosystem. In the absence of a large-scale coordinated restoration effort, restoration actions may remain dispersed, and at a scale that is less likely to reach thresholds where clear improvements toward population recovery are realized.

Lastly, implementing a comprehensive restoration strategy for the Eel River and significantly restoring fisheries in the watershed will take many decades of concerted effort on behalf of the Parties and watershed partners. Implementation of the Two-Basin Solution reflects an important milestone and opportunity for fishery recovery opportunities in Project area, and will likely facilitate synergy, attention, and resources to the overall Eel River recovery effort. Accordingly, the accelerating restoration efforts on the Eel River (including those provided by the Two-Basin Solution) provides a unique and critical opportunity to achieve significant population recovery of the Eel River fishery.

8 REFERENCES

Adams, M. J., and C. A. Pearl. 2007. Problems and opportunities managing invasive bullfrogs: is there any hope? Pages 679–693 in F. Gherardi, editor. Biological invaders in inland waters: Profiles, distribution, and threats. Springer Publishing.

Adams, A.J., A.P. Pessier, and C.J. Briggs. 2017. Rapid extirpation of a North American frog coincides with an increase in fungal pathogen prevalence: historical analysis and implications for reintroduction. Ecology and evolution, 7(23), pp.10216-10232.

Addley, C., C. Delaney, J. Emery, M. Lent, S. McBain, J. Mendoza, P. Pyle, D. Seymour, and A. Ticlavilca. 2019. Results of initial water supply modeling for Potter Valley Project and Russian River alternatives. Prepared by the Water Supply Modeling Subgroup for the Potter Valley Project Huffman Ad-Hoc Committee Water Supply Working Group, Prepared on May 22, 2019, Updated on February 20, 2020.

Amlin, N. M., and S. B. Rood. 2002. Comparative tolerances of riparian willows and cottonwoods to water-table decline. Wetlands 22: 338–346.

Andrews, E. D. 1986. Downstream effects of Flaming Gorge Reservoir on the Green River, Colorado and Utah. Geological Society of America Bulletin 97:1,012–1,023.

Asarian, E. and P. Higgins. 2018. Eel River Cooperative Cyanotoxin Analysis Summary 2013–2017.

Ashton, D. T., J. B. Bettaso, and H. H. Welsh Jr. 2015. Changes across a decade in size, growth, and body condition of Western Pond Turtle (*Actinemys marmorata*) populations on free-flowing and regulated forks of the Trinity River in northwest California. Copeia 103: 621–633.

Ashton, D., and R. J. Nakamoto. 2007. Rana boylii (Foothill Yellow-legged Frog) Predation. Herpetological Review 38: 442.

Baker, P. F., T. P. Speed, and F. K. Ligon. 1995. Estimating the influence of temperature on the survival of chinook salmon smolts (*Oncorhynchus tshawytscha*) migrating through the Sacramento-San Joaquin River Delta of California. Canadian Journal of Fisheries and Aquatic Sciences 52: 855–863.

Barnhart, R. A. 1991. Steelhead Oncorhynchus mykiss. Pages 324–336 in J. Stolz and J. Schnell, editor. Trout. Stackpole Books, Harrisburg, Pennsylvania.

Baxter, C. V., K. D. Fausch, and W. C. Saunders. 2005. Tangled webs: reciprocal flows of invertebrate prey link streams and riparian zones. Freshwater Biology 50: 201–220.

Beak Consultants, Inc. 1986. Article 41 studies to determine the effects of water temperature on downstream migration of anadromous salmonids in the upper Eel River below Lake Pillsbury. Prepared for Pacific Gas and Electric Company, San Ramon, California.

Becker, G. S., and I. J. Reining. 2009. Steelhead trout/rainbow trout (*Oncorhynchus mykiss*) resources of the Eel River watershed, California. Cartography by D. A. Asbury. Center for Ecosystem Management and Restoration, Oakland, California.

Bejarano, M. D., and A. Sordo-Ward. 2011. Riparian woodland encroachment following flow regulation: A comparative study of Mediterranean and Boreal streams. Knowledge and Management of Aquatic Ecosystems 402: 1–15.

Benjamin, J.R., P.J. Connolly, J.G. Romine, and R.W. Perry. 2013. Potential effects of changes in temperature and food resources on life history trajectories of juvenile Oncorhynchus mykiss. Transactions of the American Fisheries Society. 142:208-220.

Benn, P. C., and W. D. Erskine. 1994. Complex channel response to flow regulation: Cudgegong River below Windamere Dam, Australia. Applied Geography 14: 153–168.

Birken, A. S., and D. J. Cooper. 2006. Processes of Tamarix invasion and floodplain development along the lower Green River, Utah. Ecological Applications 16: 1,103–1,120.

Bohorquez, P., and J. D. Del Moral-Erencia. 2017a. 100 Years of competition between reduction in channel capacity and streamflow during floods in the Guadalquivir River (southern Spain). Remote Sensing 9: 727.

Bohorquez, P., and J. D. Del Moral-Erencia. 2017b. Parametric study of trends in flood stages over time in the regulated Guadalquivir River (years 1910–2016). European Water 59: 145–151.

Bond, M. H., S. A. Hayes, C. V Hanson, and R. B. MacFarlane. 2008. Marine survival of Steelhead (*Oncorhynchus mykiss*) enhanced by a seasonally closed estuary. Canadian Journal of Fisheries and Aquatic Sciences 65: 2,242–2,252.

Bouma-Gregson, K., R. M. Kudela, and M. E. Power. 2018. Widespread anatoxin-a detection in benthic cyanobacterial mats throughout a river network. PLOS ONE 13:e0197669.

Bradbury, W., W. Nehlsen, T. Nickleson, K. Moore, R. Hughes, D. Heller, J. Nicholas, D. Bottom, W. Weaver, R. Beschta. 1995. Handbook for Prioritizing Watershed Protection and Restoration to Aid Recovery of Native Salmon. Published by Pacific Rivers Council, Eugene, OR. 47 p.

Bradley, C. E., and D. G. Smith. 1986. Plains cottonwood recruitment and survival on a prairie meandering river floodplain, Milk River, southern Alberta and northern Montana. Canadian Journal of Botany 64: 1,433–1,442.

Breinlinger, S., T. J. Phillips, B. N. Haram, J. Mareš, J. A. Martínez Yerena, P. Hrouzek, R. Sobotka, W. M. Henderson, P. Schmieder, S. M. Williams, J. D. Lauderdale, H. D. Wilde, W. Gerrin, A. Kust, J. W. Washington, C. Wagner, B. Geier, M. Liebeke, H. Enke, T. H. J. Niedermeyer, and S. B. Wilde. 2021. Hunting the eagle killer: A cyanobacterial neurotoxin causes vacuolar myelinopathy. Science 371:eaax9050

Brett, J. R. 1952. Temperature tolerance in young Pacific salmon genus Oncorhynchus. Journal of the Fisheries Research Board of Canada 9: 265–323.

Brett, J. R., W. C. Clarke, and J. E. Shelbourn. 1982. Experiments on thermal requirements for growth and food conversion efficiency of juvenile Chinook salmon (*Oncorhynchus tshawytscha*). Canadian Technical Report of Fisheries and Aquatic Sciences No. 1127. Prepared by Department of Fisheries and Oceans, Nanaimo, B.C.

Brown, L. R., and P. B. Moyle. 1981. The impact of squawfish on salmonid populations: A review. North American Journal of Fisheries Management 1: 104–111.

Brown, L. R., and P. B. Moyle. 1991. Changes in Habitat and Microhabitat Partitioning within an Assemblage of Stream Fishes in Response to Predation by Sacramento Squawfish (*Ptychocheilus grandis*). Canadian Journal of Fisheries and Aquatic Sciences 48: 849–856.

Brown, L. R., and P. B. Moyle. 1997. Invading species in the Eel River, California: successes, failures, and relationships with resident species. Environmental Biology of Fishes 49: 271–291.

Brumo A. F. 2006. Spawning, larval recruitment, and early life survival of Pacific Lampreys in the South Fork Coquille River, Oregon. Master's thesis. Oregon State University, Corvallis.

Brumo, A.F., L. Grandmontagne, S.N. Namitz, and D.F. Markle. 2009. Evaluation of approaches used to monitor Pacific Lamprey spawning populations in a coastal Oregon stream. Pages 204–222 in L. R. Brown, S. D. Chase, M. G. Mesa, R. J. Beamish, and P. B. Moyle, editors. Biology, management, and conservation of lampreys in North America. American Fisheries Society, Symposium 72, Bethesda, Maryland.

Bunn, S.E., and A.H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30: 492–507.

Bustard, D.R., and D.W. Narver. 1975. Aspects of the winter ecology of juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). Journal of the Fisheries Research Board of Canada 32: 667–680.

Burford, M.A., S.A. Johnson, A.J. Cook, T.V Packer, B.M. Taylor, and E.R. Townsley. 2007. Correlations between watershed and reservoir characteristics, and algal blooms in subtropical reservoirs. Water Research 41: 4,105–4,114.

Bury, R.B., H.H. Welsh, D.J. Germano, and D.T. Ashton, editors., 2012. Western pond turtle: biology, sampling techniques, inventory and monitoring, conservation, and management. Society for Northwestern Vertebrate Biology. Northwest Fauna 7: 1–126.

Busby, P.J., T.C. Wainwright, G.J. Bryant, L.J. Lierheimer, R.S. Waples, F.W. Waknitz, and I.V. Lagomarsino.1996. Status Review of west coast steelhead from Washington, Idaho, Oregon, and California. NOAA Technical Memorandum NMFS-NWFSC-19. National Marine Fisheries Service, Seattle, Washington.

Caldwell, T.J., G.J. Rossi, R.E. Henery, and S. Chandra. 2018. Decreased streamflow impacts fish movement and energetics through reductions to invertebrate drift, body size, and abundance. River Research and Applications 34: 1–12.

CalTrout. 2019. Adult Salmonid SONAR monitoring program; South Fork Eel River, Tributary to Eel River. Report to California Department of Fish and Wildlife Fisheries Restoration Grant Program.

Carah, J.K., C.C. Blencowe, D.W. Wright, and L.A. Bolton. 2014. Low-cost restoration techniques for rapidly increasing wood cover in coastal coho salmon streams. North American Journal of Fisheries Management 34: 1,003–1,013.

Carlson, P.E., S. Donadi, and L. Sandin. 2018. Responses of macroinvertebrate communities to small dam removals: Implications for bioassessment and restoration. Journal of Applied Ecology 55:1896–1907.

Catenazzi, A., and S.J. Kupferberg. 2013. The importance of thermal conditions to recruitment success in stream-breeding frog populations distributed across a productivity gradient. Biological Conservation168: 40-48.

Casado, A., J.L. Peiry, and A. Campo. 2016. Geomorphic and vegetation changes in a meandering dryland river regulated by a large dam, Sauce Grande River, Argentina. Geomorphology 268: 21–34.

Cavallo, B., J. Merz, and J. Setka. 2013. Effects of predator and flow manipulation on Chinook salmon (*Oncorhynchus tshawytscha*) survival in an imperiled estuary. Environmental Biology of Fishes 96: 393–403.

CDFG (California Department of Fish and Game). 2010. Lower Eel River Watershed Assessment. Coastal Watershed Planning and Assessment Program. Department of Fish and Game.

CDFW (California Department of Fish and Wildlife). 2014. South Fork Eel River Watershed Assessment. Coastal Watershed Planning and Assessment Program. Fortuna, CA.

CDFW. 2019. Preliminary DRAFT of the beginning of Chinook Salmon Migration from the Lower Eel River to Portions of the Eel River watershed upstream of the Mainstem Eel River DIDSON monitoring site. November 25th through November 30th, 2019.

Cech, J.J. and C.A. Myrick. 1999. Steelhead and Chinook salmon bioenergetics: temperature, ration, and genetic effects. Project Number UCAL-WRC-W-885. University of California Water Resources Center.

Clemens, B.J., M.G. Mesa, R.J. Magie, D.A. Young, and C.B. Schreck. 2012. Pre-spawning migration of adult Pacific Lamprey, Entosphenus tridentatus, in the Willamette River, Oregon, USA. Environmental Biology of Fishes 93: 245–254.

Comiti, F., M. Da Canal, N. Surian, L. Mao, L. Picco, and M. A. Lenzi. 2011. Channel adjustments and vegetation cover dynamics in a large gravel bed river over the last 200 years. Geomorphology 125: 147–159.

Cooper, D.J., D.C. Andersen, and R.A. Chimner. 2003. Multiple pathways for woody plant establishment on floodplains at local to regional scales. Journal of Ecology 91: 182–196.

Cooper, E.J. 2017. An estimation of potential salmonid habitat capacity in the upper mainstem Eel River, California. Master's Thesis. Humboldt State University.

Cooper, E.J., A.P. O'Dowd, J.J. Graham, D.W. Mierau, W. Trush, R. Taylor. In preparation. Salmonid habitat and population capacity estimates for Steelhead trout and Chinook salmon upstream of Scott Dam in the Eel River, California.

Cordoleani, F., J. Notch, A. McHuron, A. Ammann, and C. Michel. 2017. Movement and survival of wild Chinook salmon smolts from Butte Creek during their out-migration to the ocean: comparison of a dry year versus a wet year. Transactions of the American Fisheries Society. 147: 171–184.

Cramer, S.P., and N.K. Ackerman. 2009. Prediction of stream carrying capacity for steelhead trout: the Unit Characteristic Method. American Fisheries Society Symposium, 71: 255–288.

Cramer, S.P., J. Vaughan, M. Teply, and S. Duery. 2012. Potential gains in anadromous salmonid production from restoration of Beaver Creek (Sandy River Basin, Oregon). Prepared by Cramer Fish Sciences for U.S. Army Corps of Engineers, Portland District, Oregon.

Crouse, M. ., C.A. Callahan, K. W. Malueg, S. E. Dominguez. 1981. Effects of fine sediment on growth of juvenile coho salmon in laboratory streams. Transactions of the American Fisheries Society. 110: 281–286.

Cui, Y., D. B. Booth, J. Monschke, S. Gentzler, J. Rodifer, B. Greimann, and B. Cluer. 2016. Analyses of the erosion of fine sediment deposit for a large dam-removal project: an empirical approach. International Journal of River Basin Management DOI: 10.1080/15715124.2016.1247362.

Delaney, C. J., R. K. Hartman, J. Mendoza, M. Dettinger, L. Delle Monache, J. Jasperse, F. M. Ralph, C. Talbot, J. Brown, D. Reynolds, and S. Evett. 2020. Forecast Informed Reservoir Operations Using Ensemble Streamflow Predictions for a Multipurpose Reservoir in Northern California. Water Resources Research 56:e2019WR026604.

DeWine, J. M., and D. J. Cooper. 2007. Effects of river regulation on riparian box elder (Acer negundo) forests in canyons of the upper Colorado River Basin, USA. Wetlands 27: 278–289.

Dewson, Z. S., A. B. W. James, and R. G. Death. 2007. Invertebrate community responses to experimentally reduced discharge in small streams of different water quality. Journal of the North American Benthological Society 26: 754–766.

Duffy, W. G. 2006. Protocols for monitoring the response of anadromous salmon and steelhead to watershed restoration in California. California Cooperative Fish Research Unit, Humboldt State University.

Eel River Forum. 2016. The Eel River Action Plan: A Compilation of Information and Recommended Actions.

England, J.F., Jr., Cohn, T.A., Faber, B.A., Stedinger, J.R., Thomas, W.O., Jr., Veilleux, A.G., Kiang, J.E., and Mason, R.R., Jr. 2019. Guidelines for determining flood flow frequency—Bulletin 17C (ver. 1.1, May 2019): U.S. Geological Survey Techniques and Methods, book 4, chap. B5, 148 p.

ESA. 2015. Dry Creek Habitat Enhancement Phase III, Mile Three 30% Design Report. Prepared for the Sonoma County Water Agency.

Everest, F. H. 1973. Ecology and management of summer steelhead in the Rogue River. Oregon State Game Commission. Fishery Research Report Number 7. Final Report. Oregon State Game Commission, Corvalis, Oregon, 48 pp.

Everest, F. H., and W. R. Meehan. 1981. Forest management and anadromous fish habitat productivity. Transactions of the North American Wildlife and Natural Resources Conference 46: 521–530

Fausch, K. D. 1984. Profitable stream positions for salmonids: relating specific growth rate to net energy gain. Canadian Journal of Zoology 62: 441–451.

Finlay, J. C., S. Khandwala, and M. E. Power. 2002. Spatial scales of carbon flow in a river food web. Ecology 83: 1,845–1,859.

FitzGerald, A., D. Boughton, J. Fuller, S. John, B. Martin, L. Harrison, N. Mantua. 2020. Thermal and habitat suitability for anadromous salmonids in the dammed and inaccessible Upper Mainstem Eel River subbasin in the Eel River watershed, California, Fisheries Ecology Division, Southwest Fisheries Science Center, National Marine Fisheries Service, Santa Cruz, California, 51 pp.

Friedman, J. M., W. R. Osterkamp, and W. M. J. Lewis. 1996. The role of vegetation and bed-level fluctuations in the process of channel narrowing. Geomorphology 14: 341–351.

Friedman, J. M., W. R. Osterkamp, M. L. Scott, and G. T. Auble. 1998. Downstream effects of dams on channel geometry and bottomland vegetation: Regional patterns in the Great Plains. Wetlands 18: 619–633.

Fuller, T. E., K. L. Pope, D. T. Ashton, and H. H. Welsh. 2011. Linking the distribution of an invasive amphibian (*Rana catesbeiana*) to habitat conditions in a managed river system in northern California. Restoration Ecology 19: 204-213.

Geosyntec Consultants. 2019. Draft Sampling and Analysis Plan, Lake Pillsbury and Van Arsdale Reservoir. Prepared for California State Coastal Conservancy.

Goodman, D. H., and S. B. Reid. 2012. Pacific Lamprey (*Entosphenus tridentatus*) assessment and template for conservation measures in California. US Fish and Wildlife Service, Arcata, California.

Goodman, D. H. and S. B. Reid. 2017. Climbing above the competition: innovative approaches and recommendations for improving Pacific Lamprey passage at fishways. Ecological Engineering, 107 224–232.

Goodman, D. H., S. B. Reid, N. A. Som, and W. R. Poytress. 2015. The punctuated seaward migration of Pacific Lamprey (*Entosphenus tridentatus*): environmental cues and implications for streamflow management. Canadian Journal of Fisheries and Aquatic Science 72: 1–12.

Goodman D. H., N. A. Som, and N. J Hetrick. 2018. Increasing the availability and spatial variation of spawning habitats through ascending baseflows. River Research and Applications 34: 844–853.

Gordon, E., and R. Meentemeyer. 2006. Effects of dam operation and land use on stream channel morphology and riparian vegetation. Geomorphology 82: 412–429.

Gore, J. A., J. B. Layzer, and J. I. M. Mead. 2001. Macroinvertebrate instream flow studies after 20 years: A role in stream management and restoration. Regulated Rivers: Research & Management 542: 527–542.

Graf, W. L. 2006. Downstream hydrologic and geomorphic effects of large dams on American rivers. Geomorphology 79: 336–360.

Grams, P. E., and J. Schmidt. 2002. Streamflow regulation and multi-level flood plain formation: Channel narrowing on the aggrading Green River in the eastern Uinta Mountains, Colorado and Utah. Geomorphology 44: 337–360.

Gregory, R. S., and C. D. Levings. 1998. Turbidity reduces predation on migrating juvenile Pacific Salmon. Transactions of the American Fisheries Society 127: 275–285.

Guczek, J., S. Powers, and M. Larson. 2019. Results of regional spawning ground surveys and estimates of salmonid redd abundance in the South Fork Eel River, Humboldt and Mendocino Counties, California, 2018–2019. California Coastal Salmonid Monitoring Program Annual Report prepared in partial fulfillment of California Department of Fish and Wildlife Fisheries Restoration Grant Program Grantee Agreement Number: P1510507.

Nakamoto, R. J., and B. C. Harvey. 2003. Spatial, seasonal, and size-dependent variation in the diet of Sacramento pikeminnow in the Eel River, Northwestern California. California Fish and Game 89: 30–45.

Harvey, B. C., J. L. White, and R. J. Nakamoto. 2002. Habitat relationships and larval drift of native and nonindigenous fishes in neighboring tributaries of a coastal California river. Transactions of the American Fisheries Society 131: 159–170.

Heisler, J., P. M. Glibert, J. M. Burkholder, D. M. Anderson, W. Cochlan, W. C. Dennison, Q. Dortch, C. J. Gobler, C. A. Heil, E. Humphries, A. Lewitus, R. Magnien, H. G. Marshall, K. Sellner, D. A. Stockwell, D. K. Stoecker, and M. Suddleson. 2008. Eutrophication and harmful algal blooms: A scientific consensus. Harmful Algae 8: 3–13.

Hodge, B. W., M. A. Wilzbach, and W. G. Duffy. 2014. Potential fitness benefits of the half-pounder life history in Klamath River steelhead. Transactions of the American Fisheries Society 143: 864–875.

Higgins, P. 2010. Eel River fall Chinook salmon monitoring project final 2010 report. Prepared for Friends of Eel River, Arcata, California.

Higgins, P. 2020. Sacramento Pikeminnow South Fork Eel River: ERRP index reach 2020 trend monitoring survey. Eel River Recovery Project.

Hughes, B. B., M. D. Levey, J. A. Brown, M. C. Fountain, A. B. Carlisle, S. Y. Litvin, C. M. Greene, W. N. Heady and M. G. Gleason. 2014. Nursery Functions of U.S. West Coast Estuaries: The State of Knowledge for Juveniles of Focal Invertebrate and Fish Species. The Nature Conservancy, Arlington, Virginia.

Hughes, N. F., and L. M. Dill. 1990. Position choice by drift-feeding salmonids: model and test for arctic grayling (*Thymallus arcticus*) in subarctic mountain streams, interior Alaska. Canadian Journal of Fisheries and Aquatic Sciences 47: 2,039–2,048.

Isaak, D. J., S. J. Wenger, E. E. Peterson, J. M. Ver Hoef, D. E. Nagel, C. H. Luce, S. W. Hostetler, J. B. Dunham, B.B. Roper, S. P. Wollrab, G. L. Chandler, D. L. Horan, and S. Parkes-Payne. 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: 9,181–9,205.

Israel, J. A., and A. P. Klimley. 2008. Life history conceptual model – North American green sturgeon (*Acipenser medirostris*). Sacramento-San Joaquin Delta Regional Ecosystem Restoration Implementation Plan.

Jansen, L. S., A. O'Dowd, and K. Bouma-Gregson. 2020. A comparison of benthic algal and macroinvertebrate communities in a dammed and undammed Mediterranean river (Eel River watershed, California, USA). River Res. Appl. n/a. doi:10.1002/rra.3695.

Jeffres, C. A., J. J. Opperman, and P. B. Moyle. 2008. Ephemeral floodplain habitats provide best growth conditions for juvenile Chinook salmon in a California river. Environmental Biology of Fishes 83: 449–458.

Johnson, B. L., W. B. Richardson, and N. T. J. 1995. Past, present, and future concepts in large river ecology. BioScience 45: 7.

Johnson, W. C. 1998. Adjustment of riparian vegetation to river regulation in the Great Plains, USA. Wetlands 18: 608–618.

Johnson, W. C. 2000. Tree recruitment and survival in rivers: Influence of hydrological processes. Hydrological Processes 14: 3,051–3,074.

Jones, K. K., T. J. Cornwell, D. L. Bottom, L. A. Campbell, and S. Stein. 2014. The contribution of estuary-resident life histories to the return of adult Oncorhynchus kisutch. Journal of Fish Biology 85: 52–80.

Kajtaniak, D., and J. Gruver. 2020. Lower mainstem Eel River Chinook Salmon Monitoring Project, Final Report: sonar estimation of California Coastal (CC) Chinook Salmon (*Oncorhynchus tshawytscha*) and Northern California (NC) steelhead (Oncorhynchus mykiss) abundance in the lower mainstem Eel River, Humboldt County, California, 2019–2020.

Kendall, N. W., J. R. McMillan, M. R. Sloat, T. W. Buehrens, T. P. Quinn, G. R. Pess, K. V. Kuzishchin, M. M. McClure, and R. W. Zabel. 2015. Anadromy and residency in steelhead and rainbow trout (*Oncorhynchus mykiss*): a review of the processes and patterns. Canadian Journal of Fisheries and Aquatic Sciences 72: 319–342.

Kinziger, A. P., R. J. Nakamoto, and B. C. Harvey. 2014. Local-scale invasion pathways and small founder numbers in introduced Sacramento pikeminnow (*Ptychocheilus grandis*). Conservation Genetics 15: 1–9.

Kondolf, G. M., and P. Wilcock. 1996. The flushing flow problem: Defining and evaluating objectives. Water Resources Research 32: 2,589–2,599.

Kubicek, P. F. 1977. Summer water temperature conditions in the Eel River system, with reference to trout and salmon. Master’s Thesis. Humboldt State University, Arcata, California.

Kui, L., J. C. Stella, P. B. Shafroth, P. K. House, and A. C. Wilcox. 2016. The long-term legacy of geomorphic and riparian vegetation feedbacks on the dammed Bill Williams River, Arizona, USA. Ecohydrology 2017: e1839.

Lambeck, R. J. 1997. Focal species: a multi species umbrella for nature conservation. Conservation Biology 11: 849–856

Lambert, M. R., J. M. McKenzie, R. M. Screen, A. G. Clause, B. B. Johnson, G. G. Mount, H. B. Shaffer, and G. B. Pauly. 2019. Experimental removal of introduced slider turtles offers new insight into competition with a native, threatened turtle.

Larson, Z. 2013. Operation of dual frequency identification sonar (DIDSON) to monitor adult anadromous fish migrations in the Smith River, California: 2-year pilot study. Prepared by Zack Larson and Associates, Crescent City California.

Lestelle, L. C. 2007. Coho salmon (*Oncorhynchus kisutch*) life history patterns in the Pacific Northwest and California. Prepared by Biostream Environmental, Poulsbo, Washington for U.S. Bureau of Reclamation, Klamath Area Office.

Lieberman, D., and M. Horn. 1998. Pre- and Post-Operational Effects of a Temperature Control Device on Physical, Chemical, and Biological Attributes of Shasta Lake, California: Phase 1, spring 1995 through fall 1997. Prepared by: United States Department of the Interior, Geological Survey, in cooperation with the U.S. Bureau of Reclamation, Reston, Virginia.

Ligon, F. K., R. J. Nakamoto, B. C. Harvey, and P. F. Baker. 2016. Use of streambed substrate as refuge by steelhead or rainbow trout (*Oncorhynchus mykiss*) during simulated freshets. Journal of Fish Biology 88: 1,475–1,485.

Lind, A. J., H. H. Welsh, Jr., and R. A. Wilson. 1996. The effects of a dam on breeding habitat and egg survival of the foothill yellow-legged frog (*Rana boylii*) in Northwestern California. Herpetological Review 27: 62–67.

Lisle, T. E. 1990. The Eel River, Northwestern California: high sediment yields from a dynamic landscape. Pages 311–314 in M. G. Wolman and H. G. Riggs, editors. Surface water hydrology, v O-1. The geology of North America. Geological Society of America.

Magdaleno, F., and A. Fernandes. 2010. Hydromorphological alteration of a large Mediterranean river: Relative role of high and low flows on the evolution of riparian forests and channel morphology. River Research and Applications 27: 374–387.

Magilligan, F. J., and K. H. Nislow. 2005. Changes in hydrologic regime by dams. Geomorphology 71: 61–78.

Mahoney, J. M., and S. B. Rood. 1991. A device for studying the influence of declining water table on poplar growth and survival. Tree Physiology 8: 305–314.

Mahoney, J. M., and S.B. Rood. 1992. Response of a hybrid poplar to water table decline in different substrates. Forest Ecology and Management 54: 141–156.

Mahoney, J. M., and S. B. Rood. 1998. Streamflow Requirements for Cottonwood Seedling Recruitment- An Integrative Model. Wetlands 18: 634–645.

Mahoney, J. M., and K. Rood. 1992. Response of a hybrid poplar to water table decline in different substrates. Forest Ecology and Management 54: 141–156.

Major, J. J., J. E. O'Connor, C. J. Podolak, M. K. Keith, K. R. Spicer, S. Pittman, H. M. Bragg, J. R. Wallick, D. Q. Tanner, A. Rhode, and P. R. Wilcock. 2012, Geomorphic response of the Sandy River, Oregon, to removal of Marmot Dam: U.S. Geological Survey Professional Paper 1792.

Marine, K., and J. Cech, Jr. 2004. Effects of high water temperature on growth, smoltification, and predator avoidance in juvenile Sacramento River Chinook Salmon. North American Journal of Fisheries Management 24: 198–210.

McBain & Trush Inc. 1997. Trinity River Maintenance Flow Study Final Report. Hoopa Valley Tribe, Arcata, California.

McBride, J. R., and J. Strahan. 1984a. Establishment and survival of woody riparian species on gravel bars of an intermittent stream. American Midland Naturalist 112: 235–245.

McBride, J. R., and J. Strahan. 1984b. Fluvial Geomorphic Processes, Bank Stabilization and Controlled Streamflow Effects upon Riparian Vegetation along Dry Creek. Appendix I, U.S. Army Corps of Engineers, Dry Creek Vegetation Plan, , San Francisco, California.

McBride, J. R., N. Sugihara, and E. Norberg. 1988. Growth and Survival of Three Riparian Woodland Species in Relation to Simulated Water Table Dynamics. Department of Forestry and Resource Management, University of California Berkeley, California.

McCarthy, S. F., J. J. Duda, J. M. Emlen, G. R. Hodgson, D. A. Beauchamp. 2009. Linking Habitat Quality with Trophic Performance of Steelhead along Forest Gradients in the South Fork Trinity River Watershed, California. Transactions of the American Fisheries Society, 183:3, 506-521

McCovey Jr., B. W. 2011. A small-scale radio bio-telemetry study to monitor migrating Pacific Lamprey (*Lampetra tridentata*) within the Klamath River basin. Prepared by Yurok Tribal Fisheries Program, Hoopa, California.

McCullough, D. A. 1999. A review and synthesis of effects of alterations to the water temperature regime on freshwater life stages of salmonids, with special reference to Chinook salmon. Columbia River Inter-Tribal Fish Commission, Portland, Oregon.

Meeuwig, M. H., J. M. Bayer, and J. G. Seelye. 2005. Effects of temperature on survival and development of early life stage Pacific and western brook lampreys. Transactions of the American Fisheries Society 134: 19–27.

Meitzen, K. M., M. W. Doyle, M. C. Thoms, and C. E. Burns. 2013. Geomorphology within the interdisciplinary science of environmental flows. Geomorphology 200: 143–154.

Mendocino Redwood Company, LLC. 2002. Outmigration of Juvenile Salmonids from Hollow Tree Creek, Mendocino County, California (2000-2002). Fort Bragg.

Merritt, D. M., and D. J. Cooper. 2000. Riparian vegetation and channel change in response to river regulation: a comparative study of regulated and unregulated streams in the Green River Basin, USA. Regulated Rivers: Research & Management 16: 543–564.

Merritt, D. M., M. L. Scott, L. Poff, G. T. Auble, and D. A. Lytle. 2010. Theory, methods and tools for determining environmental flows for riparian vegetation: Riparian vegetation-flow response guilds. Freshwater Biology 55: 206–225.

Metheny, M. 2020. Adult Salmonid Sonar Monitoring Program. South Fork Eel River, Tributary to Eel River. Field Note, August 3, 2020.

Michel C. J., A. J. Ammann, S. T. Lindley, P. T. Sandstrom, E. D. Chapman, M. J. Thomas, G. P. Singer, A. P. Klimley, and R. B. MacFarlane. 2015. Chinook salmon outmigration survival in wet and dry years in California's Sacramento River. Canadian Journal of Fisheries and Aquatic Sciences. 72: 1,749–1,759.

Morley, S. A., M. M. Foley, J. J. Duda, M. M. Beirne, R. L. Paradis, R. C. Johnson, M. L. McHenry, M. Elofson, E. M. Sampson, R. E. McCoy, J. Stapleton, and G. R. Pess. 2020. Shifting food web structure during dam removal—Disturbance and recovery during a major restoration action. PLOS ONE 15.

Moyle, P. B. 2002. Inland Fishes of California. 2nd edition. University of California Press, Berkley.

Moyle, P. B., and D. M. Baltz. 1985. Microhabitat use by an assemblage of California stream fishes: Developing criteria for instream flow determinations. Transactions of the American Fisheries Society 114: 695–704.

Moyle, P., R. Lusardi, P. Samuel, and J. Katz. 2017. State of the Salmonids: Status of California's Emblematic Fishes 2017. Center for Watershed Sciences, University of California, Davis and California Trout, San Francisco, California.

Murray, C.B., and McPhail, J. D. 1988. Effects of incubation temperature on development of five species of Pacific salmon (*Oncorhynchus*) embryos and alevins. Canadian Journal of Zoology 66: 266–273.

Myrick, C. A., and J. J. Cech. 2001. Temperature effects on chinook salmon and steelhead: a review focusing on California's Central Valley populations. Prepared by Department of Fishery and Wildlife Biology, Colorado State University, Fort Collins and Department of Wildlife, Fish, and Conservation Biology, University of California, Davis for the Bay-Delta Modeling Forum. http://www.sfei.org/modelingforum/.

Myrick, C. A., and J. J. Cech. 2004. Temperature effects on juvenile anadromous salmonids in California's central valley: what don't we know? Reviews in Fish Biology and Fisheries 14: 113–123.

Nakamoto, R. J., and B. C. Harvey. 2003. Spatial, seasonal, and size dependent variation in the diet of Sacramento Pikeminnow in the Eel River, Northwester, California. California Fish and Game 89:30–45.

Naman, S. M., J. S. Rosenfeld, and J. S. Richardson. 2016. Causes and consequences of invertebrate drift in running waters: from individuals to populations and trophic fluxes. Canadian Journal of Fisheries and Aquatic Sciences 73: 1–14.

Naman, S. M., J. S. Rosenfeld, L. C. Third, and J. S. Richardson. 2017. Habitat-specific production of aquatic and terrestrial invertebrate drift in small forest streams: implications for drift-feeding fish. Canadian Journal of Fisheries and Aquatic Sciences 74: 1,208–1,217.

Nickelson, T. E., J. D. Rodgers, S. L. Johnson, and M. F. Solazzi. 1992. Seasonal changes in habitat use by juvenile Coho Salmon (*Oncorhynchus kisutch*) in Oregon coastal streams. Canadian Journal of Fisheries and Aquatic Sciences 49: 783–789.

Nielson J. L., T. E. Lisle, and V. Ozaki. 1994. Thermally stratified pools and their use by steelhead in Northern California streams. Transactions of the American Fisheries Society 123: 613–626

NMFS (National Marine Fisheries Service). 2002. Biological Opinion for the Potter Valley Project. National Marine Fisheries Service, FERC Project No. 77-110

NMFS. 2014. Final Recovery Plan for the Southern Oregon/Northern California Coast Evolutionarily Significant Unit of Coho Salmon (*Oncorhynchus kisutch*). National Marine Fisheries Service. Arcata, California.

NMFS. 2016. Coastal Multispecies Recovery Plan. National Marine Fisheries Service, West Coast Region, Santa Rosa, California.

O'Brien, J. S., and P. J. Currier. 1987. Channel Morphology and Riparian Vegetation Changes in the Big Bend Reach of the Platte River in Nebraska. Prepared by the Engineering Research Center, Colorado State University and The Platte River Whooping Crane Habitat Maintenance Trust, Fort Collins, Colorado.

Olden, J. D., and R. J. Naiman. 2010. Incorporating thermal regimes into environmental flows assessments: modifying dam operations to restore freshwater ecosystem integrity. Freshwater Biology 55:86–107.

O'Dowd, A., and W. J. Trush. 2016. Friends of the Eel River Blockwater Investigation Final Memo. Humboldt State University River Institute, Arcata, California.

O'Neil, J. M., T. W. Davis, M. A. Burford, and C. J. Gobler. 2012. The rise of harmful cyanobacteria blooms: The potential roles of eutrophication and climate change. Harmful Algae 14: 313–334.

Paoletti, D.L. 2009. Responses of Foothill Yellow-legged frog (Rana boylii) Larvae to an Introduced Predator. Master's Thesis. Oregon State University, Corvallis.

Palmer, M., and A. Ruhi. 2019. Linkages between flow regime, biota, and ecosystem processes: Implications for river restoration. 365: DOI: 10.1126/science.aaw2087.

Parker K. A. 2018. Evidence for the genetic basis of inheritance of ocean and river-maturing ecotypes of Pacific Lamprey (*Entosphenus tridentatus*) in the Klamath River, California. Master's thesis. Humboldt State University, Arcata, California.

Pasquale, N., P. Perona, R. Francis, and P. Burlando. 2014. Above-ground and below-ground Salix dynamics in response to river processes. Hydrological Processes 28: 5,189–5,203.

Pelzman, R. J. 1973. Causes and Possible Prevention of Riparian Plant Encroachment on Anadromous Fish Habitat. Environmental Services Branch Administrative Report No. 73-1, California Department of Fish and Game, Redding, California.

Peterman, R. M. 1982. Model of salmon age structure and its use in preseason forecasting and studies of marine survival. Canadian Journal of Fisheries and Aquatic Sciences 39: 1,444–1,452.

PG&E (Pacific Gas and Electric Company). 2016. Article 52(a). Summer Water Temperature Monitoring Results, 2015. Addressing NMFS Measure 8 (in part) and License Article 57. Potter Valley Hydroelectric Project, FERC Project No. 77.

PG&E. 2017a. Pottery Valley Hydroelectric Project FERC Project No. 77. Relicensing Pre-Application Document (PAD). Volume 1: Public Information Sections 107.

PG&E. 2017b. Article 53. Chinook Salmon Carcass Survey Results, 2015/16. Potter Valley Hydroelectric Project, FERC Project No. 77. June

PG&E. 2018. Summer Rearing Monitoring Results, 2017. Addressing NMFS Measure 8 (in part). Potter Valley Hydroelectric Project, FERC Project No. 77. June

PG&E. 2020. Article 52(a). Pikeminnow Monitoring and Suppression Results, 2019. Addressing NMFS RPA Section G.2 and Measures 1 and 2 (in part). Potter Valley Hydroelectric Project, FERC Project No. 77.

Piccolo, J. J., B. M. Frank, and J. W. Hayes. 2014. Food and space revisited: The role of drift-feeding theory in predicting the distribution, growth, and abundance of stream salmonids. Environmental Biology of Fishes 97: 475–488.

Pimentel, D., L. Lach, R. Zuniga, and D. Morrison. 2000. Environmental and economic costs of nonindigenous species in the United States. BioScience 50: 53.

Plumb, J. M., and C. M. Moffitt. 2015. Re-estimating temperature-dependent consumption parameters in bioenergetics models for juvenile Chinook Salmon. Transactions of the American Fisheries Society 144: 323–330.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. BioScience 47: 769–784.

Poff, N. L., J. D. Olden, D. M. Merritt, and D. M. Pepin. 2007. Homogenization of regional river dynamics by dams and global biodiversity implications. Proc. Natl. Acad. Sci. 104: 5,732–5,737.

Poff, N. L., and J. K. H. Zimmerman. 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwater Biology 55: 194–205.

Power, M. E. 1990. Effects of fish in river food webs. Science 250: 811–814.

Power, M. E., W. E. Dietrich, and J. C. Finlay. 1996. Dams and downstream aquatic biodiversity: Potential food web consequences of hydrologic and geomorphic change. Environmental Management 20: 887–895.

Power, M. E., M. S. Parker, and W. E. Dietrich. 2008. Seasonal reassembly of a river food web: floods, droughts and impacts of fish. Ecological Monographs 78: 263–282.

Power, M. E., J. R. Holomuzki, and R. L. Lowe. 2013. Food webs in Mediterranean rivers. Hydrobiologia 719: 119–136.

Power, M. E., K. Bouma-Gregson, P. Higgins, and S. M. Carlson. 2015. The thirsty eel: Summer and winter flow thresholds that tilt the Eel River of Northwestern California from salmon-supporting to cyanobacterially degraded states. Copeia 103: 200–211.

R Core Team, X. 2015. R: A language for environment and statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Railsback, S. F., and K. A. Rose. 1999. Bioenergetics modeling of stream trout growth: temperature and food consumption effects. Transactions of the American Fisheries Society 128: 241–256.

Railsback, S. F., B. C. Harvey, S. J. Kupferberg, M. M. Lang, S. McBain, and H. H. Welsh, Jr. 2016. Modeling potential river management conflicts between frogs and salmonids. Canadian Journal of Fisheries and Aquatic Sciences 73: 773–784.

Reese, C. D., and B. C. Harvey. 2002. Temperature-dependent interactions between juvenile Steelhead and Sacramento Pikeminnow in laboratory streams. Transactions of the American Fisheries Society 131: 599–606.

Reid, S. B., and D. H. Goodman. 2015. Detectability of Pacific Lamprey occupancy in western drainages: Implications for distribution surveys. Transactions of the American Fisheries Society 144: 315–322.

Ricker, S. J., D. Ward, and C. W. Anderson. 2014. Results of Freshwater Creek Salmonid Life Cycle Monitoring Station 2010–2013. California Department of Fish and Game, Anadromous Fisheries Resource Assessment and Monitoring Program, Arcata, California.

Ritchie, A. C., J. A. Warrick, A. E. East, C. S. Magirl, A. W. Stevens, J. A. Bountry, T. J. Randle, C. A. Curran, R. C. Hilldale, J. J. Duda, G. R. Gelfenbaum, I. M. Miller, G. R. Pess, M. M. Foley, R. McCoy, and A. S. Ogston. 2018. Morphodynamic evolution following sediment release from the world's largest dam removal. Scientific Reports 8: 13,279.

Ringler, N. H., and J. D. Hall. 1975. Effects of logging on water temperature and dissolved oxygen in spawning beds. Transactions of the American Fisheries Society 104: 111–121.

Ritter, J. R. 1968. Changes in the Channel Morphology of Trinity River and Eight Tributaries, California, 1961-65. Open-File Report, U.S. Department of the Interior, Menlo Park, California.

Robinson, T. C. and J. M. Bayer. 2005. Upstream migration of Pacific lampreys in the John Day River, Oregon: behavior, timing, and habitat use. Northwest Science 79: 106–119.

Roelofs, T. D. 1983. Current status of California summer steelhead (*Salmo gairdneri*) stocks and habitat, and recommendations for their management. Report to USDA Forest Service Region 5.

Rosenfeld, J., E. Raeburn, P. C. Carrier, and R. Johnson. 2008. Effects of side channel structure on productivity of floodplain habitats for juvenile Coho Salmon. Canadian Journal of Fisheries and Aquatic Science 28: 1,108–1,119.

Rundio, DE, Lindley, ST. 2008. Seasonal patterns of terrestrial and aquatic prey abundance and use by Oncorhynchus mykiss in a California coastal basin with a Mediterranean climate. Transactions of the American Fisheries Society. 137, 467-480

Sabal, M., S. Hayes, J. Merz, and J. Setka. 2016. Habitat alterations and a non-native predator, the Striped Bass, increase native Chinook Salmon mortality in the Central Valley, California. North American Journal of Fisheries Management 36: 309–320.

Sanderson, B. L., K. A. Barnas, and A. M. W. Rub. 2009. Nonindigenous species of the Pacific Northwest: An overlooked risk to endangered salmon? BioScience 59: 245–256.

Schindler, D. E., R. Hilborn, B. Chasco, C. P. Boatright, T. P. Quinn, L. A. Rogers, and M. S. Webster. 2010. Population diversity and the portfolio effect in an exploited species. Nature 465: 609–612.

Scott, M. L., M. A. Wondzell, and G. T. Auble. 1993. Hydrograph characteristics relevant to the establishment and growth of western riparian vegetation. Pages 237-246 in Proceedings of the Thirteenth Annual American Geophysical Union Hydrology Days. Hydrology Days Publications, Atherton, California.

Scott, M. L., J. M. Friedman, and G. T. Auble. 1996. Fluvial process and the establishment of bottomland trees. Geomorphology 14: 327–339.

Scott, M. L., G. T. Auble, and J. M. Friedman. 1997. Flood dependency of cottonwood establishment along the Missouri River, Montana, USA. Ecological Applications 7: 677–690.

SCWA (Sonoma County Water Agency). 2016. Fish Habitat Flows and Water Rights Project Draft Environmental Impact Report. July 2016.

SEC (Steiner Environmental Consulting). 1998. Potter Valley Project Monitoring Program, Effects of Operations on Upper Eel River Anadromous Salmonids. Prepared for Pacific Gas and Electric Company. Steiner Environmental Consulting FERC No 77:604.

Segelquist, C. A., M. L. Scott, and G. T. Auble. 1993. Establishment of Populus deltoides under simulated alluvial groundwater declines. American Midland Naturalist 130: 274–285.

Shafroth, P. B., J. C. Stromberg, and D. T. Patten. 2002. Riparian vegetation response to altered disturbance and stress regimes. Ecological Applications 12: 107–123.

Shapovalov, L., and A. C. Taft. 1954. The life histories of the steelhead rainbow trout (*Salmo gairdneri gairdneri*) and silver salmon (*Oncorhynchus kisutch*) with special reference to Waddell Creek, California, and recommendations regarding their management. Fish Bulletin 98. California Department of Fish and Game.

Sherrard, J. J., and W. D. Erskine. 1991. Complex response of a sand-bed stream to upstream impoundment. Regulated Rivers 6: 53–70.

Simpkins, D. G., and W. A. Hubert. 2000. Drifting invertebrates, stomach contents, and body conditions of juvenile Rainbow Trout from fall through winter in a Wyoming tailwater. Transactions of the American Fisheries Society 129: 1,187–1,195.

Sloat, M.R., and G. H. Reeves. 2014. Individual condition, standard metabolic rate, and rearing temperature influence Steelhead and rainbow trout (*Oncorhynchus mykiss*) life histories. Canadian Journal of Fisheries and Aquatic Sciences. 71: 491–501.

Sogard, S. M., J. E. Merz, W. H. Satterthwaite, M. P. Beakes, D. R. Swank, E. M. Collins, R. G. Titus, and M. Mangel. 2012. Contrasts in habitat characteristics and life history patterns of *Oncorhynchus mykiss* in California's central coast and central valley. Transactions of the American Fisheries Society 141: 747–760.

Soulé, M. E., D. T. Bolger, A. C. Alberts, J. Wright, M. Sorice, S. Hill. 1988. Reconstructed dynamics of rapid extinctions of chaparral-requiring birds in urban habitat islands. Conservation Biology 2: 75–92.

Spence, B. C., E. P. Bjorkstedt, J. C. Garza, J. J. Smith, D. G. Hankin, D. Fuller, W. E. Jones, R. Macedo, T. H. Williams, and E. Mora. 2008. A framework for assessing the viability of threatened and endangered salmon and steelhead trout in the north-central California coast recovery domain. NOAA Technical Memorandum, NMFS. Southwest Fisheries Science Center, Santa Cruz, California.

Spence, B. C., E. P. Bjorkstedt, S. Paddock, and L. Nanus. 2012. Updates to biological viability criteria for threatened steelhead populations in the north-central California coast recovery domain. NMFS. Southwest Fisheries Science Center, Santa Cruz, California.

Starcevich, S. J., S. L. Gunckel, and S. E. Jacobs. 2014. Movements, habitat use, and population characteristics of adult Pacific Lamprey in a coastal river. Environmental Biology of Fishes 97: 939–953.

Stella, J. C., J. Battles, J. McBride, and B. K. Orr. 2010. Riparian seedling mortality from simulated water table recession, and the design of sustainable flow regimes on regulated rivers. Restoration Ecology 3: 1–11.

Stillwater Sciences. 2010. Pacific Lamprey in the Eel River basin: a summary of current information and identification of research needs. Prepared by Stillwater Sciences, Arcata, California for Wiyot Tribe, Loleta, California.

Stillwater Sciences. 2014. A conceptual framework for understanding factors limiting Pacific lamprey production in the Eel River basin. Prepared by Stillwater Sciences, Arcata, California for Wiyot Tribe, Loleta, California.

Stillwater Sciences and Wiyot Tribe Natural Resources Department. 2016. Monitoring Pacific Lamprey in lower Eel River basin: pilot surveys and recommendations for long-term monitoring. Prepared by Stillwater Sciences, Arcata, California and Wiyot Tribe Natural Resources Department, Table Bluff, California for U.S. Fish and Wildlife Service, Sacramento, California.

Stillwater Sciences and Wiyot Tribe Natural Resources Department. 2017. Status, distribution, and population of origin of green sturgeon in the Eel River: results of 2014–2016 studies. Prepared by Stillwater Sciences, Arcata, California and Wiyot Tribe, Natural Resources Department, Loleta, California, for National Oceanic and Atmospheric Administration, Fisheries Species Recovery Grants to Tribes, Silver Springs, Maryland.

Stillwater Sciences. 2019. South Fork Ten Mile River Coho Salmon Restoration Project: Phase 1 Pre-treatment Monitoring. Prepared by Stillwater Sciences, Arcata, California for The Nature Conservancy, San Francisco, California.

Stillwater Sciences et al. 2021. Potter Valley Project Feasibility Study: Alternatives Description and Project Plan. Prepared by Stillwater Sciences and McBain Associates, Arcata, California; McMillen Jacobs Associates, Boise, Idaho; M.Cubed, Davis, California; Princeton Hydro, South Glastonbury, Connecticut; and Geosyntec Consultants, Oakland, California for the Potter Valley Project Planning Agreement Parties.

Stillwater Sciences and Wiyot Tribe Natural Resources Department. 2020. Evaluation of Population Monitoring and Suppression Strategies for Invasive Sacramento Pikeminnow in the South Fork Eel River. Prepared by Stillwater Sciences, Arcata, California and Wiyot Tribe Natural Resources Department, Table Bluff, California for U.S. Fish and Wildlife Service, Sacramento, California.

Suttle, K. B., M. E. Power, J. M. Levine, and C. McNeely. 2004. How sediment in riverbeds impairs growth and survival of juvenile salmonids. Ecological Applications 14: 969–974.

Tagart, J. V. 1976. The survival from egg deposition to emergence of coho salmon in the Clearwater River, Jefferson County, Washington. Master's thesis. University of Washington, Seattle.

Teo, S.L.H., P.T. Sandstrom, E.D. Chapman, R.E. Null, K. Brown, P. Klimley, and B.A. Block. 2013. Archival and acoustic tags reveal the post-spawning migrations, diving behavior, and thermal habitat of hatchery-origin Sacramento River steelhead kelts (*Oncorhynchus mykiss*). Environmental Biology of Fishes. 96: 175–187

Thompson, K. 1972. Determining stream flows for fish life. Pages 31–50 in Proceedings of the instream flow requirement workshop. Pacific Northwest River Basin Commission, Vancouver, Washington.

Thorne, C. R., R. G. Allen, and A. Simon. 1996. Geomorphological river channel reconnaissance for river analysis, engineering and management. Transactions of the Institute of British Geographers 21: 469–483.

Uno, H., and M. E. Power. 2015. Mainstem-tributary linkages by mayfly migration help sustain salmonids in a warming river network. Ecology Letters 18: 1,012–1,020.

Uno, H. 2016. Stream thermal heterogeneity prolongs aquatic-terrestrial subsidy and enhances riparian spider growth. Ecology 97: 2.547–2,553.

Uno, H. and J. Stillman. 2020. Lifetime eurythermy by seasonally matched thermal performance of developmental stages in an annual aquatic insect. Oecologia 192: 647–656.

Van de Wetering, S. J. 1998. Aspects of life history characteristics and physiological processes in smolting Pacific Lamprey (*Lampetra tridentata*) in a central Oregon coast stream. Master's thesis. Oregon State University. Corvallis.

Vaughn, H. 2005. Sproul Creek Downstream Migrant Trapping Program Report. Prepared for the Eel River Salmon Restoration Project, Miranda, California. October 18.

Vesipa, R., C. Camporeale, and L. Ridolfi. 2017. Effect of river flow fluctuations on riparian vegetation dynamics: Processes and models. Advances in Water Resources 110:10.1016/j.advwatres.2017.1009.1028.

Vondracek, B. 1987. Digestion rates and gastric evacuation times in relation to temperature of the Sacramento Squawfish, Ptychocheilus grandis. U.S. National Marine Fisheries Service Fishery Bulletin 85: 159–163.

VTN (VTN Oregon, Inc.). 1982. Potter Valley Project (FERC No. 77) Fisheries Study. Final Report Vols. I & II. Prepared for Pacific Gas and Electric Company, San Ramon, CA.

USDI (U.S. Department of the Interior), U.S. Department of Commerce, and National Marine Fisheries Service. 2013. Klamath dam removal overview report for the Secretary of the Interior. An assessment of science and technical information.

USFS (United States Forest Service) and BLM (Bureau of Land Management). 1995. Watershed analysis report for the middle fork Eel River watershed.

Ward, B. R., P. A. Slaney, A. R. Facchin, and R. W. Land. 1989. Size-biased survival in steelhead trout (*Oncorhynchus mykiss*): back-calculated lengths from adults' scales compared to migrating smolts at the Keogh River, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 46: 1,853–1,858.

Wheeler, C.A., J. B. Bettaso, D. T. Ashton, and H. H. Welsh. 2015. Effects of water temperature on breeding phenology, growth, and metamorphosis of foothill yellow-legged frogs (*Rana boylii*): A case study of the regulated mainstem and unregulated tributaries of California's Trinity River. River Research and Applications 31: 1,276–1,286.

White, J. L., and B. C. Harvey. 2001. Effects of an introduced piscivorous fish on native benthic fishes in a coastal river. Freshwater Biology. 46: 987–995.

White, J. L., and B. C. Harvey. 2003. Basin-scale patterns in drift of embryonic and larval fishes and lamprey ammocoetes in two coastal rivers. Environmental Biology of Fishes 67: 369–378.

Williams, G. J. 2010. Life History Conceptual Model for Chinook salmon and Steelhead. DRERIP Delta Conceptual Model. Sacramento (CA): Delta Regional Ecosystem Restoration Implementation Plan. http://www.dfg.ca.gov/ERP/drerip_conceptual_models.asp

Williams, G. P., and M. G. Wolman, 1984. Downstream effects of dams on alluvial rivers, US Geological Survey Professional Paper 1286, Washington, D.C.

Wohl, E., and D. N. Scott. 2016. Wood and sediment storage and dynamics in river corridors. Earth Surf. Process. Landforms 42: 5–23.

WNRD (Wiyot Tribe Natural Resources Department) and Stillwater Sciences. 2016. Wiyot Tribe Pacific Lamprey adaptive management plan framework. Prepared by Wiyot Tribe Natural Resources Department, Table Bluff Reservation, Loleta, California and Stillwater Sciences, Arcata, California for U.S. Fish and Wildlife Service, Sacramento, California.

Wood, S. A., J. Atalah, A. Wagenhoff, L. Brown, K. Doehring, R. G. Young, and I. Hawes. 2016. Effect of river flow, temperature, and water chemistry on proliferations of the benthic anatoxin-producing cyanobacterium Phormidium. Freshwater Science 36:63–76.

Elliott, W.W. & Co. 1881. History of Humboldt County, California: with Illustrations Descriptive of its Scenery, Farms, Residences, Public Buildings, Factories, Hotels, Business Houses, Schools, Churches, etc., from Original Drawings, including Biographical Sketches. San Francisco: W.W. Elliot & Co.

Yoshiyama, R. M., and P. B. Moyle. 2010. Historical review of Eel River anadromous salmonids, with emphasis on Chinook salmon, Coho Salmon and Steelhead. University of California, Center for Watershed Sciences.

Zahar, Y., A. Ghorbel, and J. Albergel. 2008. Impacts of large dams on downstream flow conditions of rivers: Aggradation and reduction of the Medjerda channel capacity downstream of the Sidi Salem dam (Tunisia). Journal of Hydrology 351: 318–330.

Zimmerman, M. P. 1999. Food habits of Smallmouth Bass, Walleyes, and Northern Pikeminnow in the Lower Columbia River Basin during outmigration of juvenile anadromous salmonids. Transactions of the American Fisheries Society 128: 1,036–1,054.

Zimmerman, M. S., C. Kinsel, E. Beamer, E. J. Connor, and D. E. Pflug. 2015. Abundance, survival, and life history strategies of juvenile Chinook Salmon in the Skagit River, Washington. Transactions of the American Fisheries Society 144: 627–641.

APPENDIX A
Weighted Usable Area Analysis

Figures

A-1 EEL RIVER

A-1.1 Downstream of Scott Dam (gage E-2)

A-1.1.1 Trout Creek Site

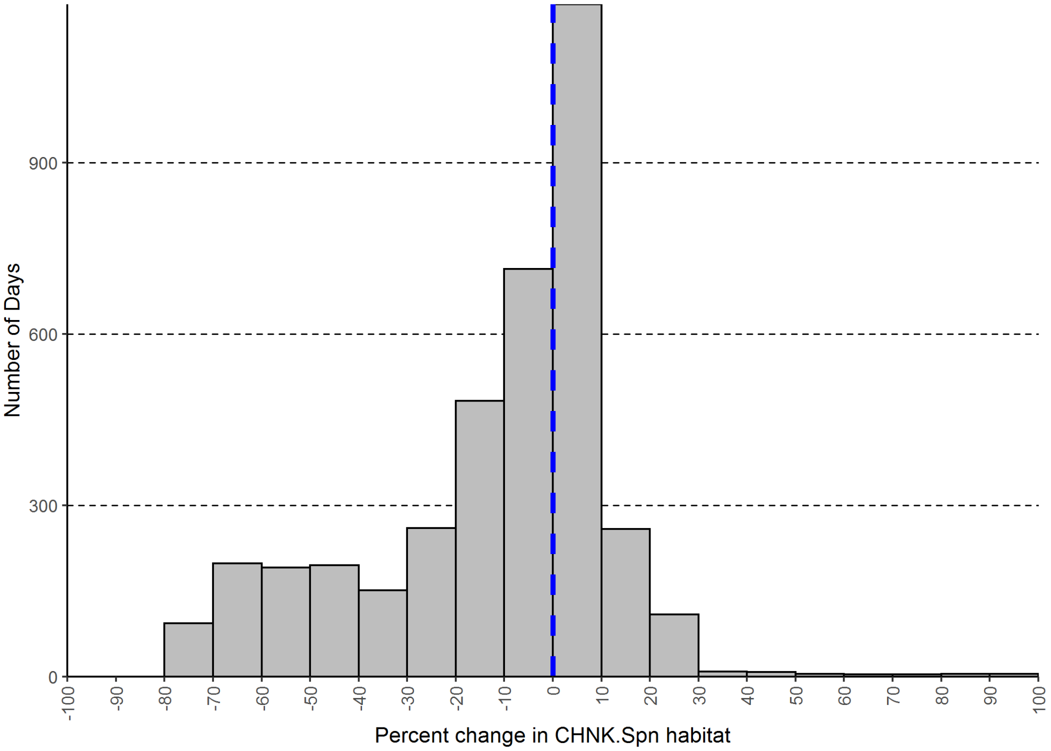

Figure A1-1. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for Chinook Salmon spawning habitat at the Trout Creek site.

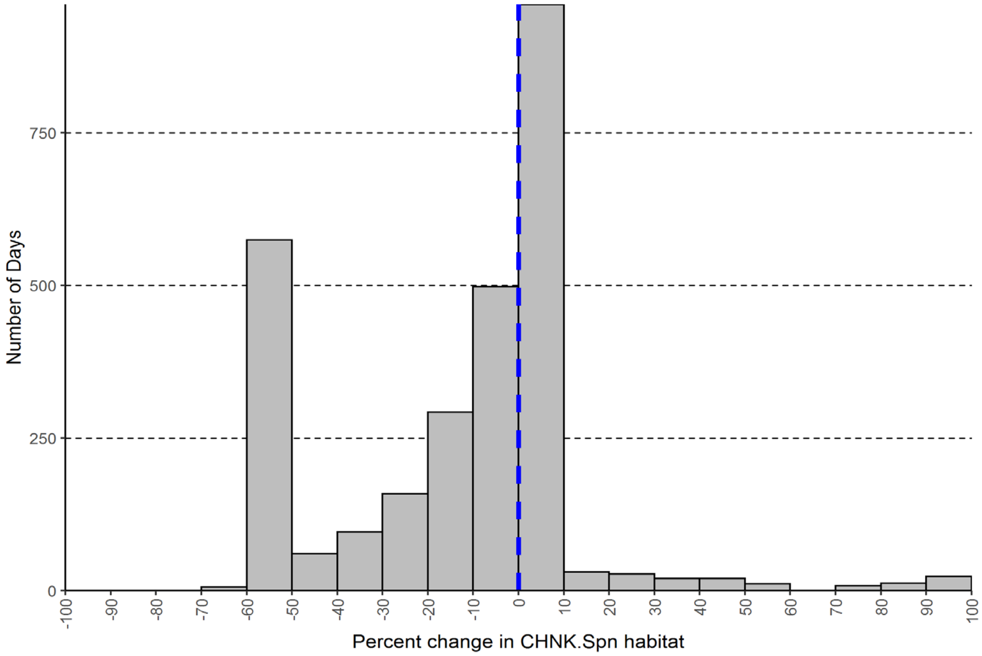

Figure A1-2. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for Chinook Salmon spawning habitat at the Trout Creek site.

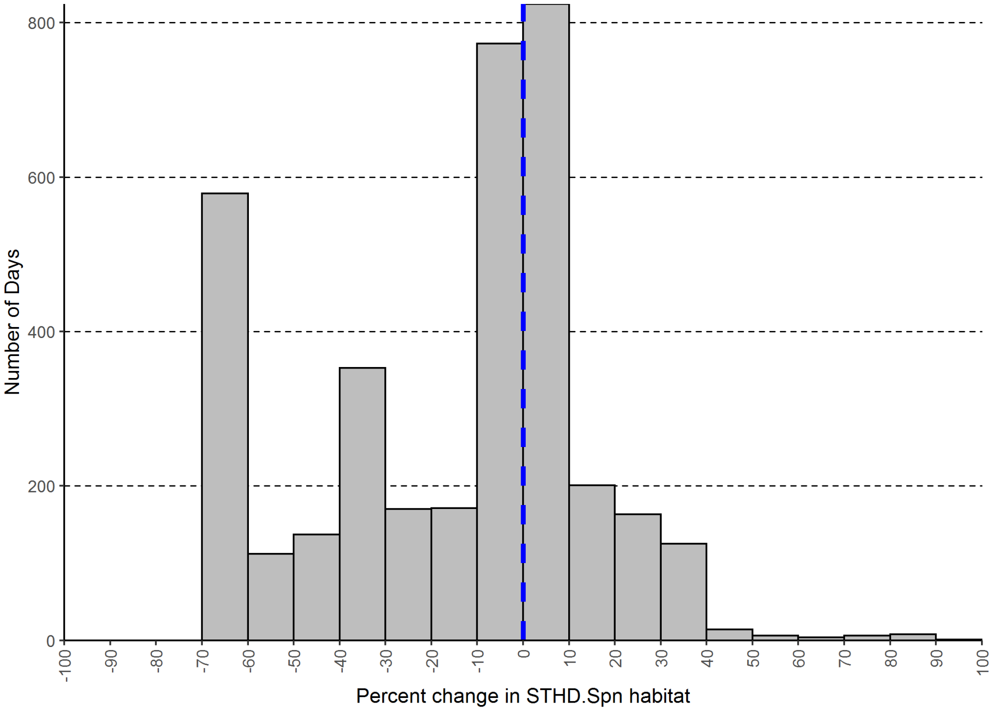

Figure A1-3. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead spawning habitat at the Trout Creek site.

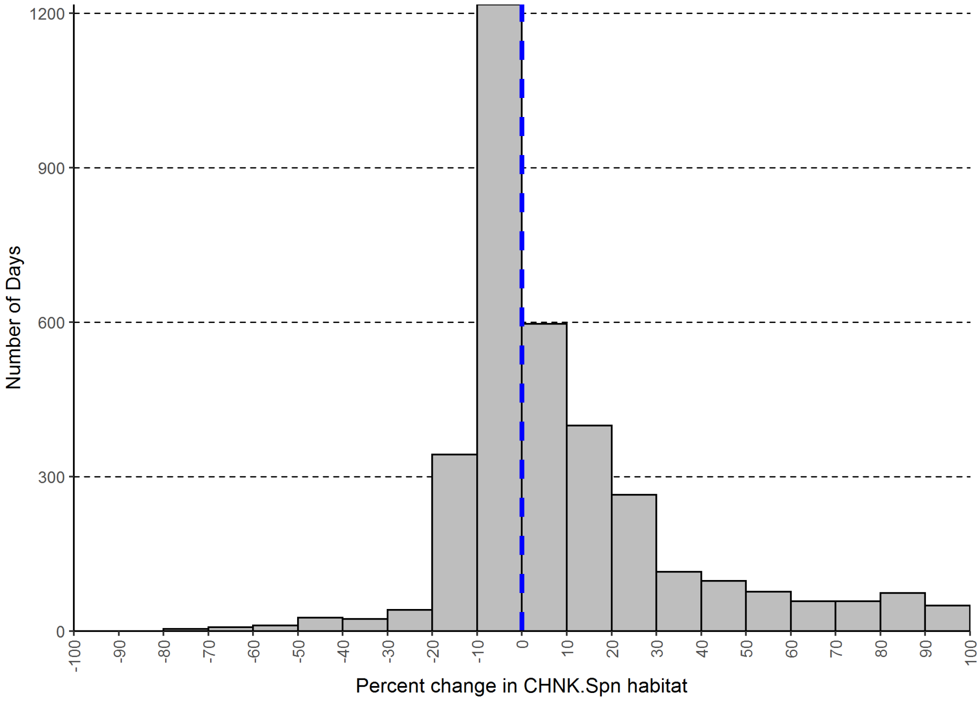

Figure A1-4. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for Chinook Salmon spawning habitat at the Trout Creek site.

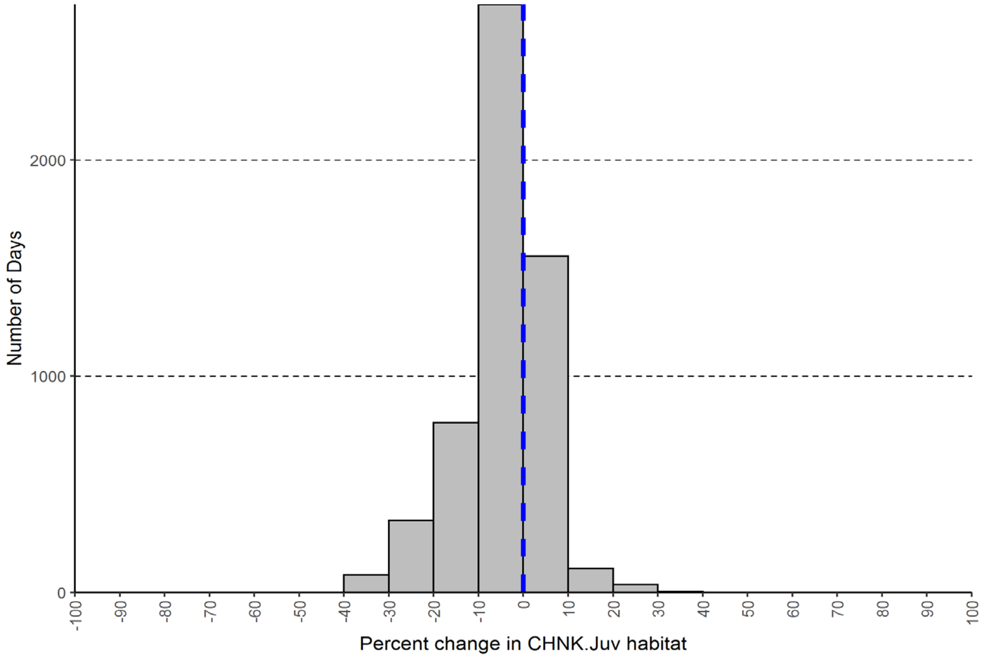

Figure A1-5. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for Chinook Salmon juvenile habitat at the Trout Creek site.

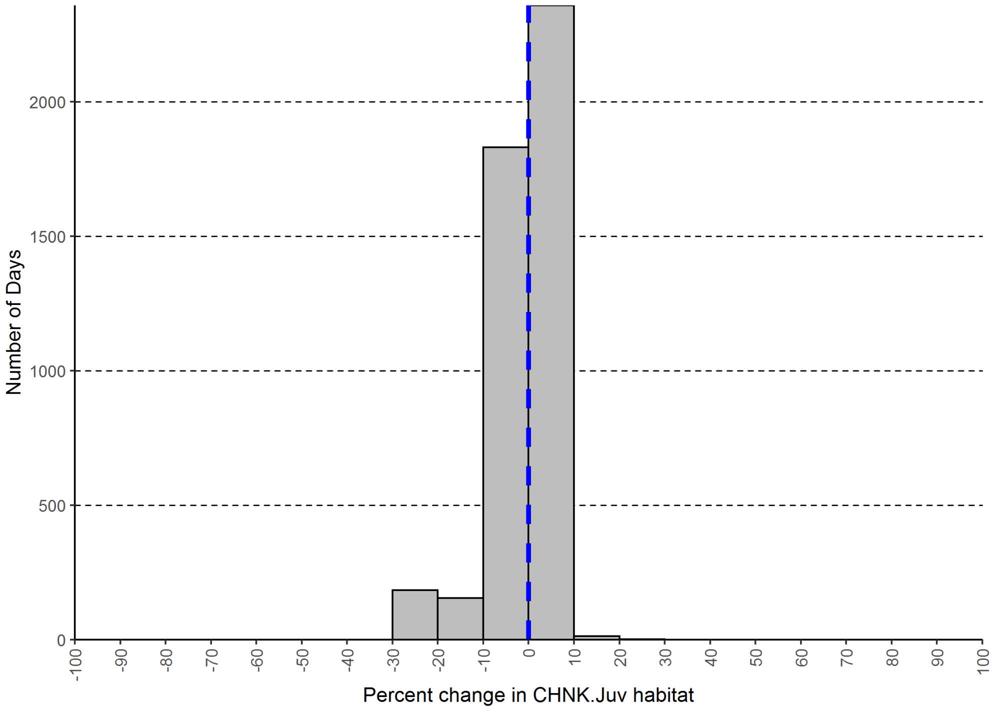

Figure A1-6. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for Chinook Salmon juvenile habitat at the Trout Creek site.

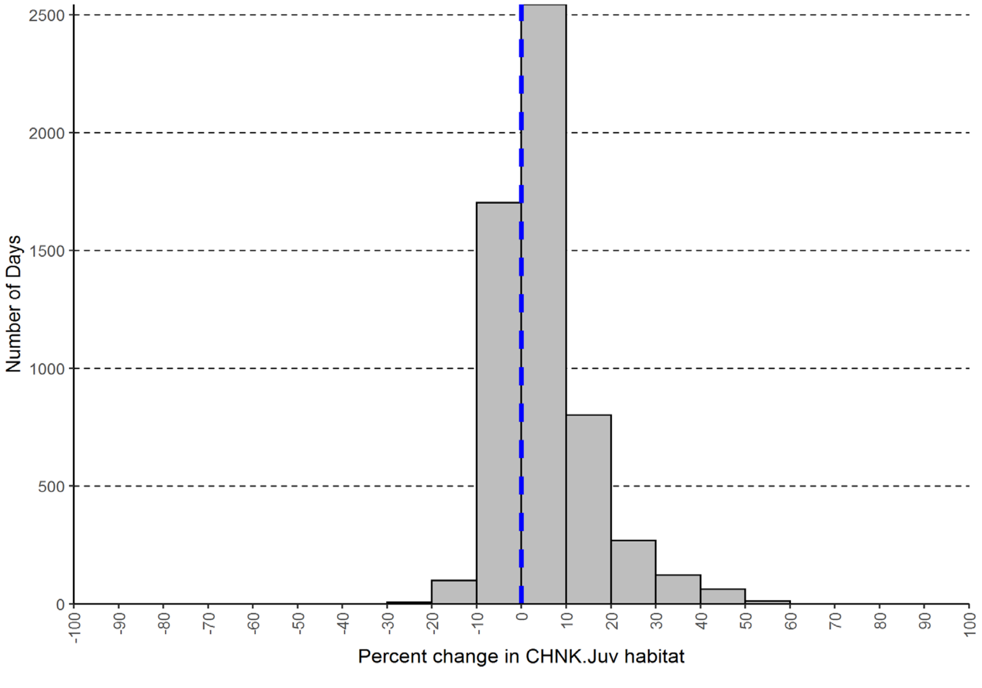

Figure A1-7. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for Chinook Salmon juvenile habitat at the Trout Creek site.

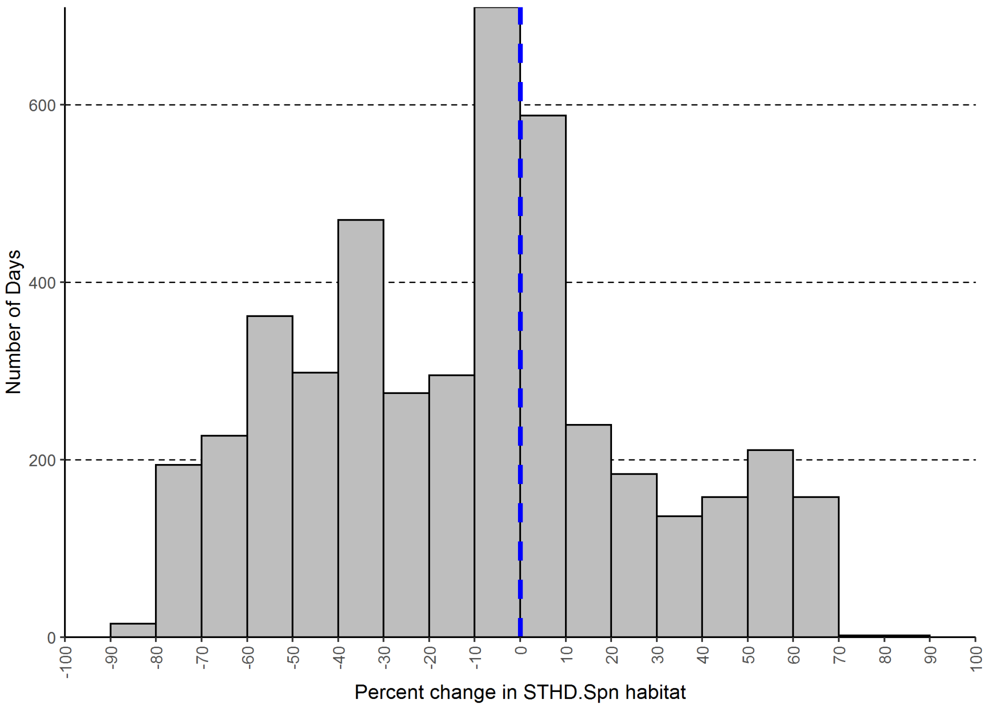

Figure A1-8. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead spawning habitat at the Trout Creek site.

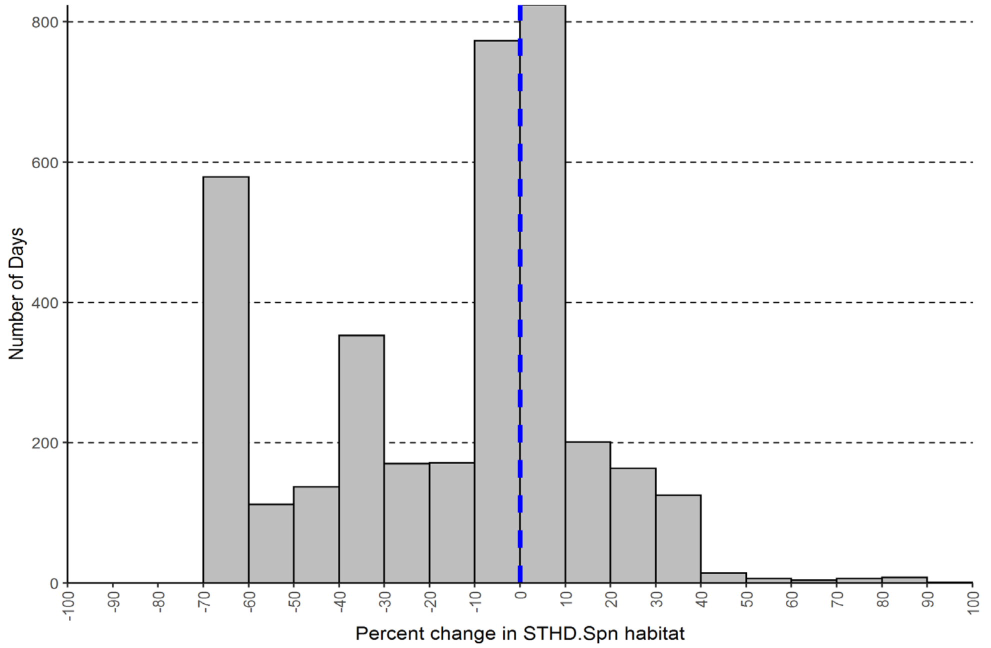

Figure A1-9. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead spawning habitat at the Trout Creek site.

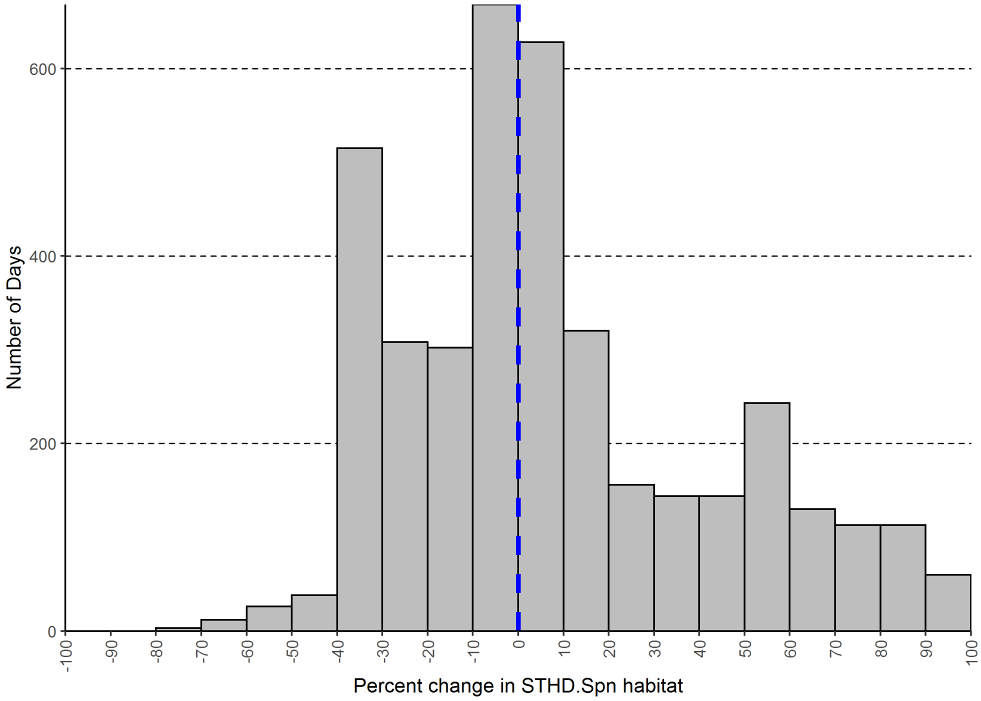

Figure A1-10. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead spawning habitat at the Trout Creek site.

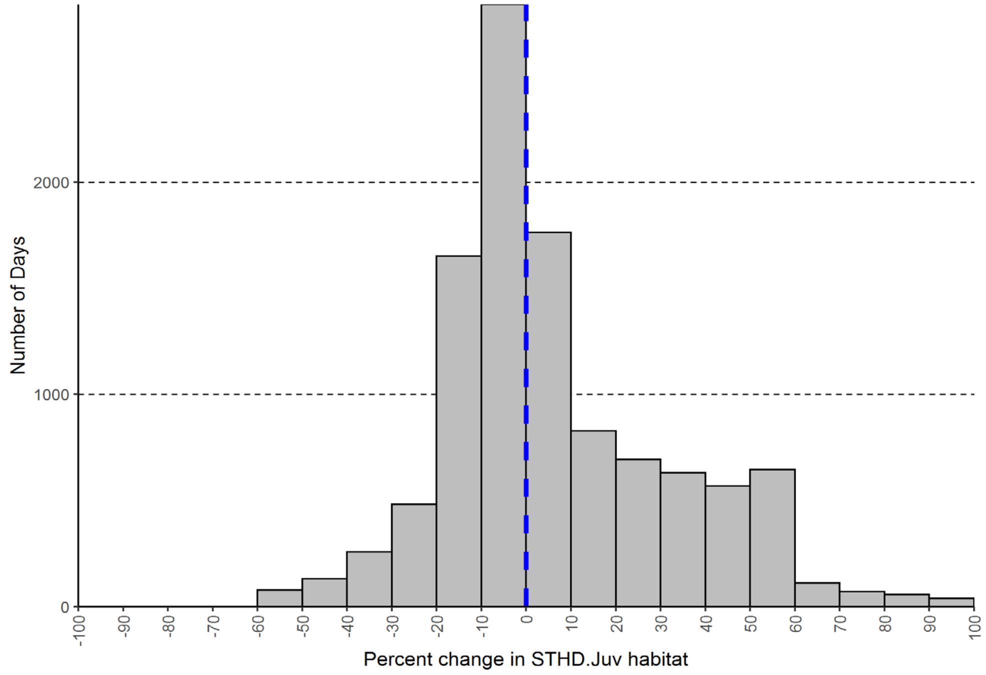

Figure A1-11. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead juvenile habitat at the Trout Creek site.

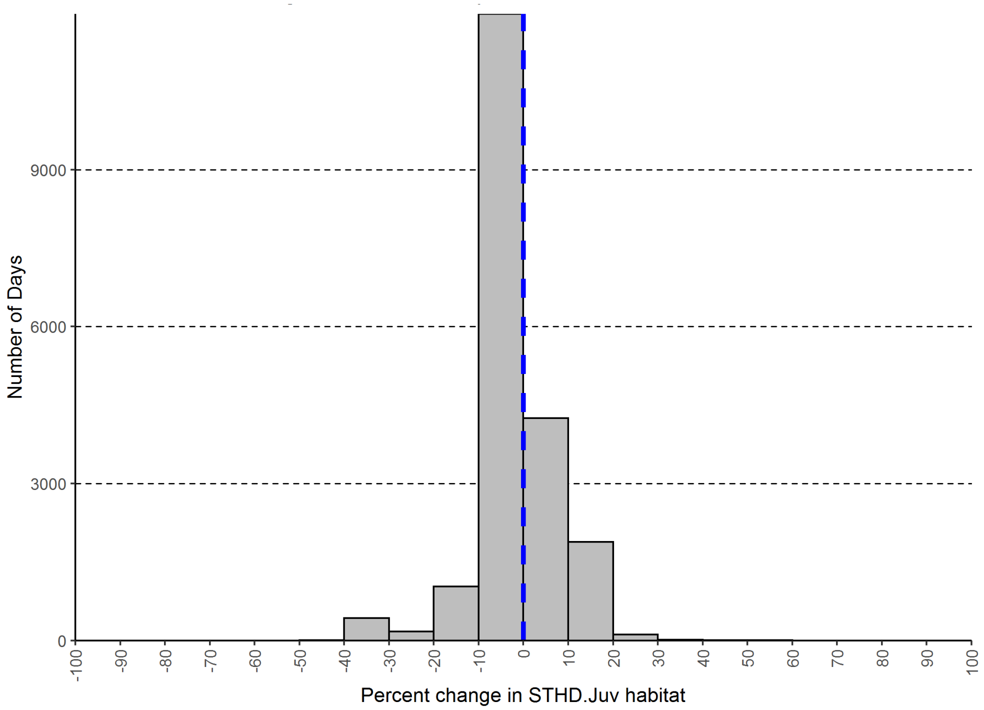

Figure A1-12. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead juvenile habitat at the Trout Creek site.

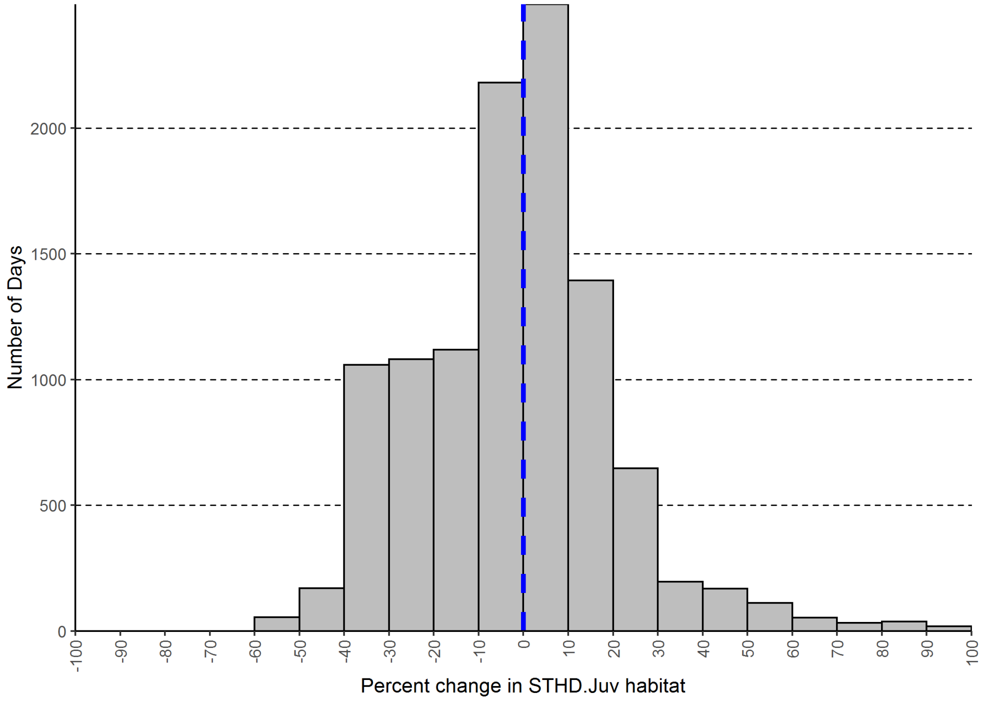

Figure A1-13. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead juvenile habitat at the Trout Creek site.

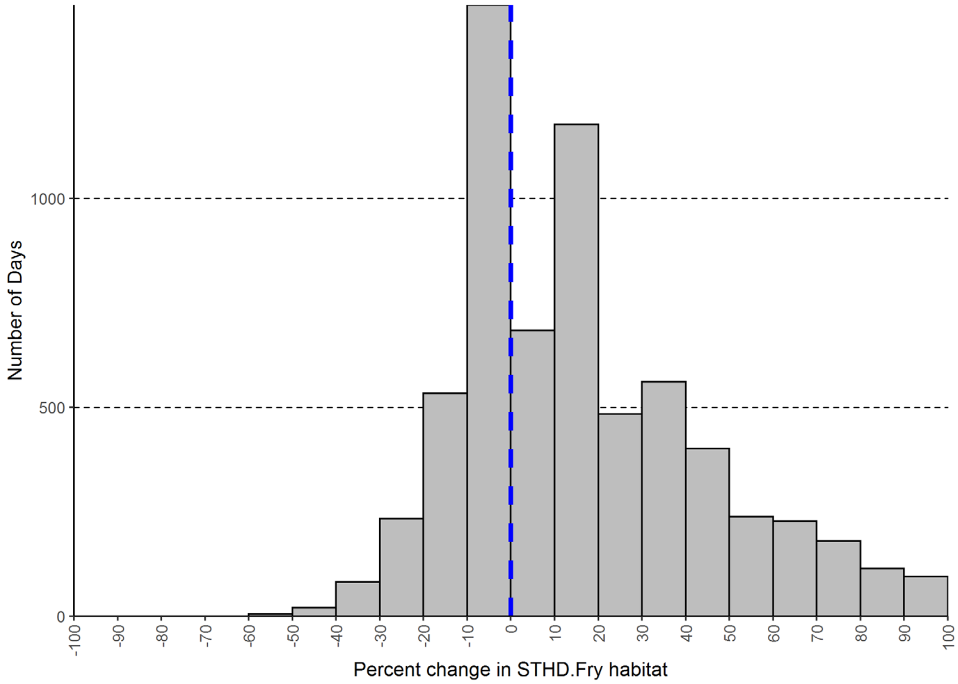

Figure A1-14. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead fry habitat at the Trout Creek site.

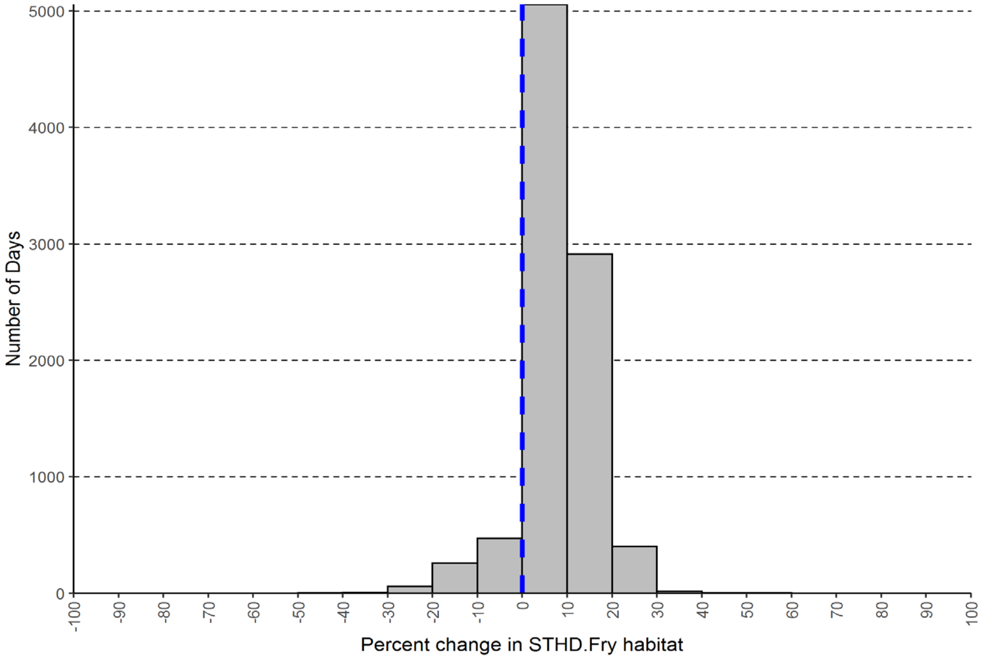

Figure A1-15. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead fry habitat at the Trout Creek site.

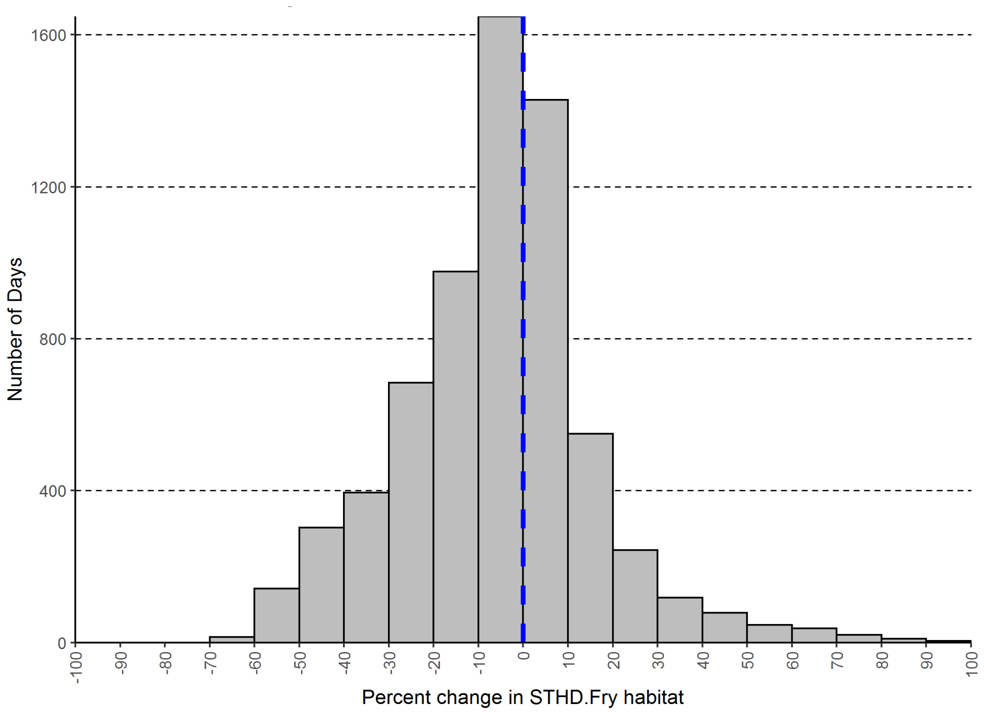

Figure A1-16. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead fry habitat at the Trout Creek site.

A-1.2 Downstream of Cape Horn Dam (gage E-11)

A-1.2.1 Cape Horn Dam Site

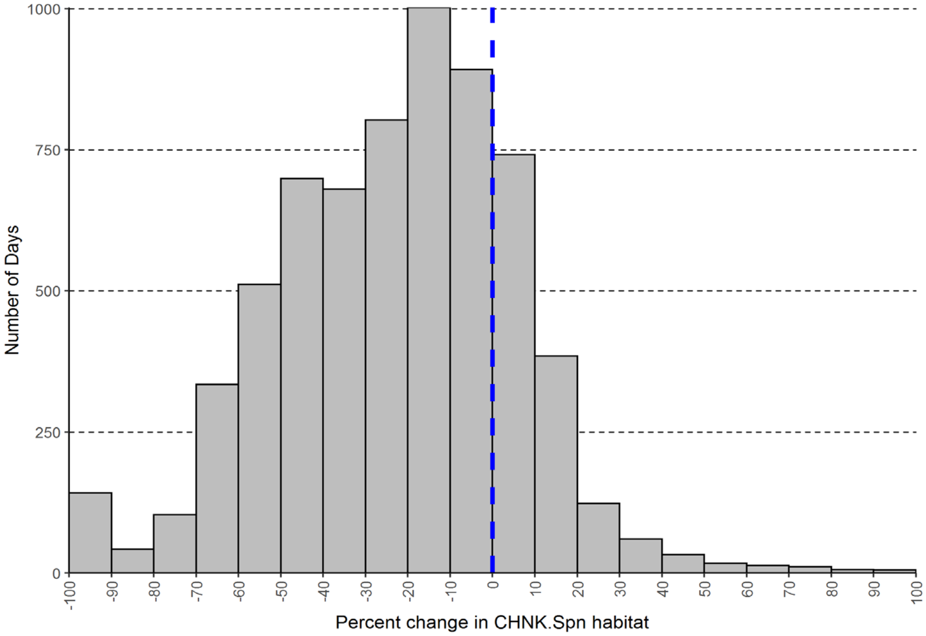

Figure A1-17. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for Chinook Salmon spawning habitat at the Cape Horn Dam site.

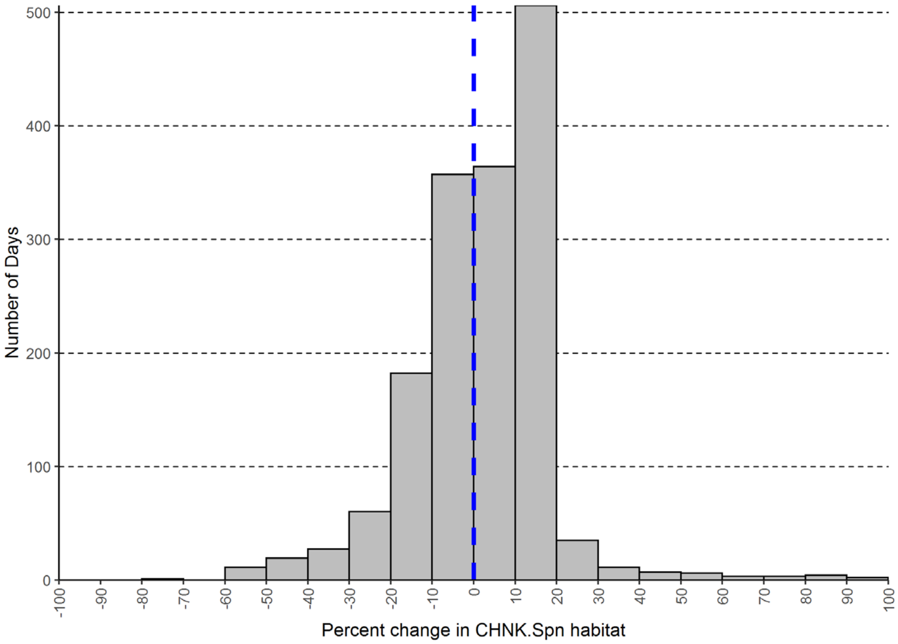

Figure A1-18. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for Chinook Salmon spawning habitat at the Cape Horn Dam site.

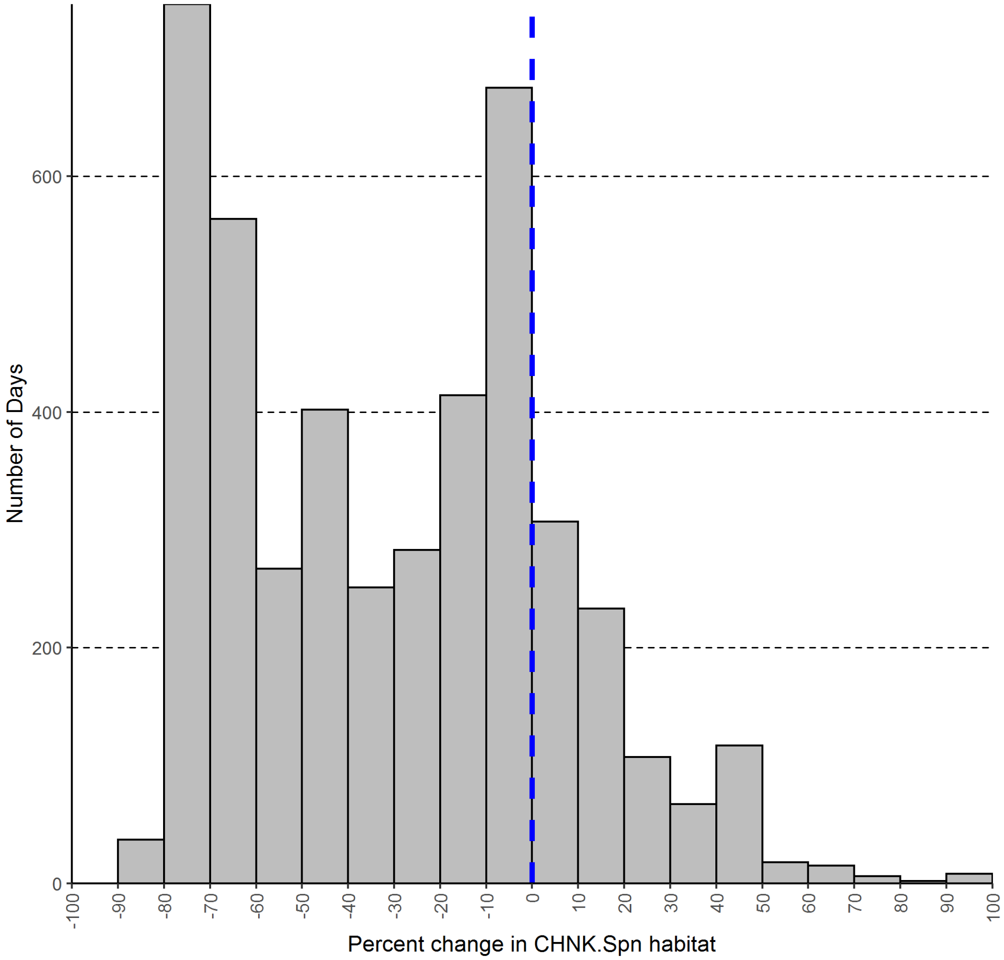

Figure A1-19. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for Chinook Salmon spawning habitat at the Cape Horn Dam site.

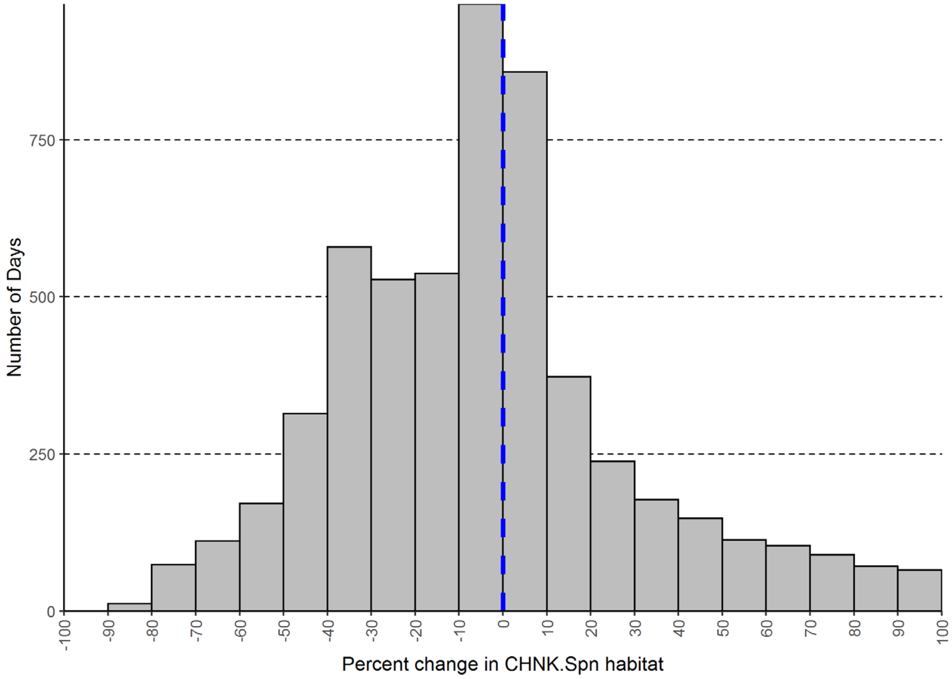

Figure A1-20. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for Chinook Salmon spawning habitat at the Cape Horn Dam site.

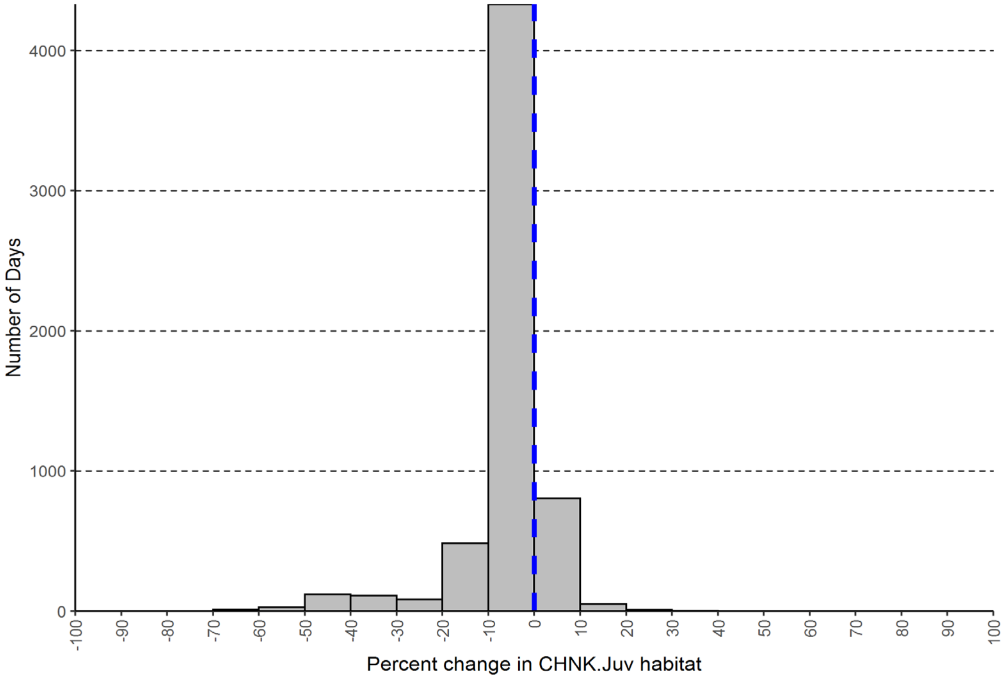

Figure A1-21. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for Chinook Salmon juvenile habitat at the Cape Horn Dam site.

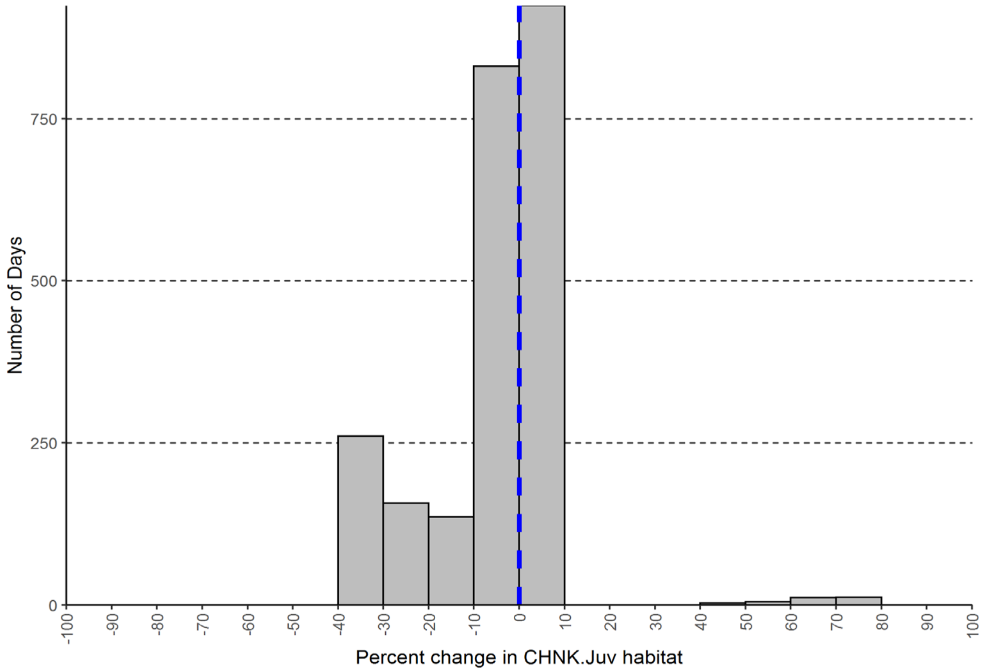

Figure A1-22. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for Chinook Salmon juvenile habitat at the Cape Horn Dam site.

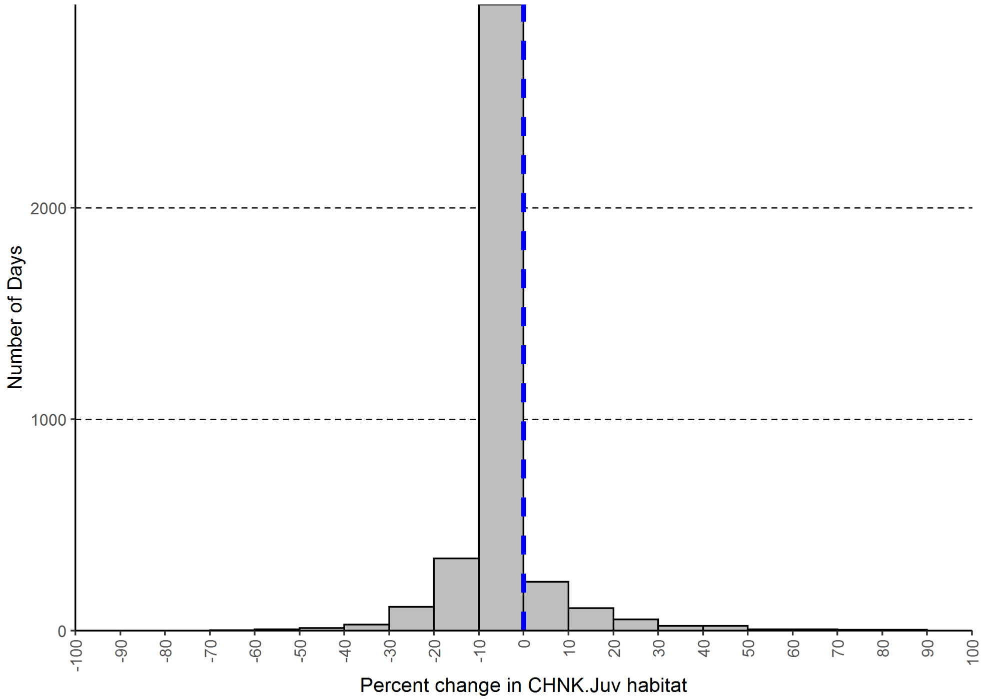

Figure A1-23. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for Chinook Salmon juvenile habitat at the Cape Horn Dam site.

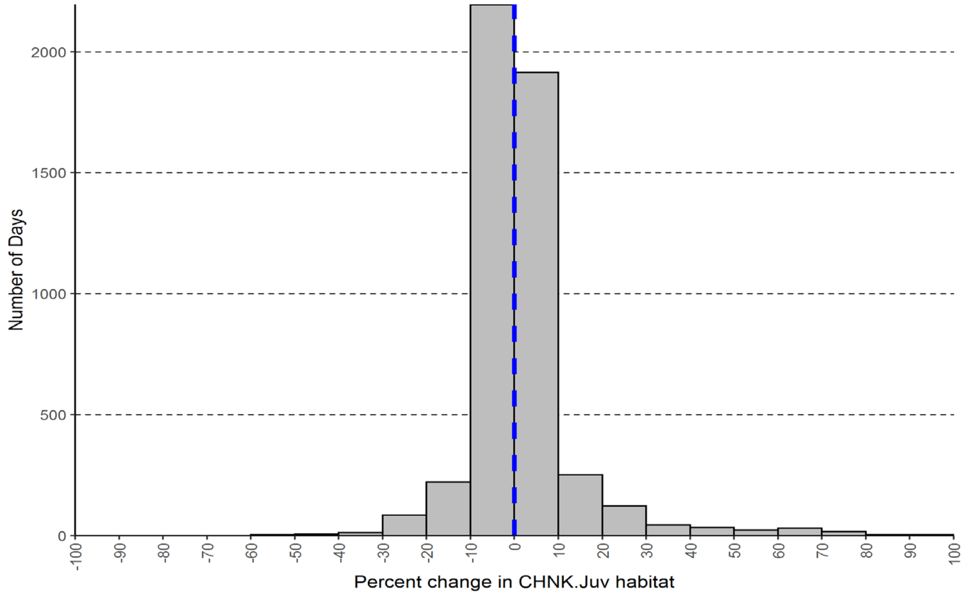

Figure A1-24. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for Chinook Salmon juvenile habitat at the Cape Horn Dam site.

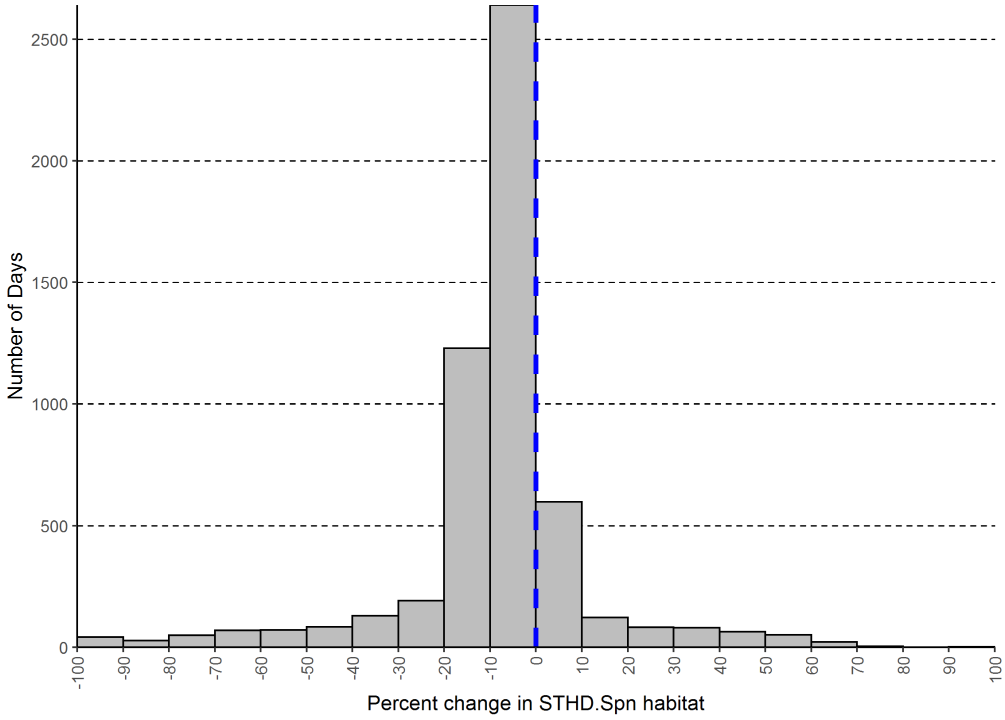

Figure A1-25. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead spawning habitat at the Cape Horn Dam site.

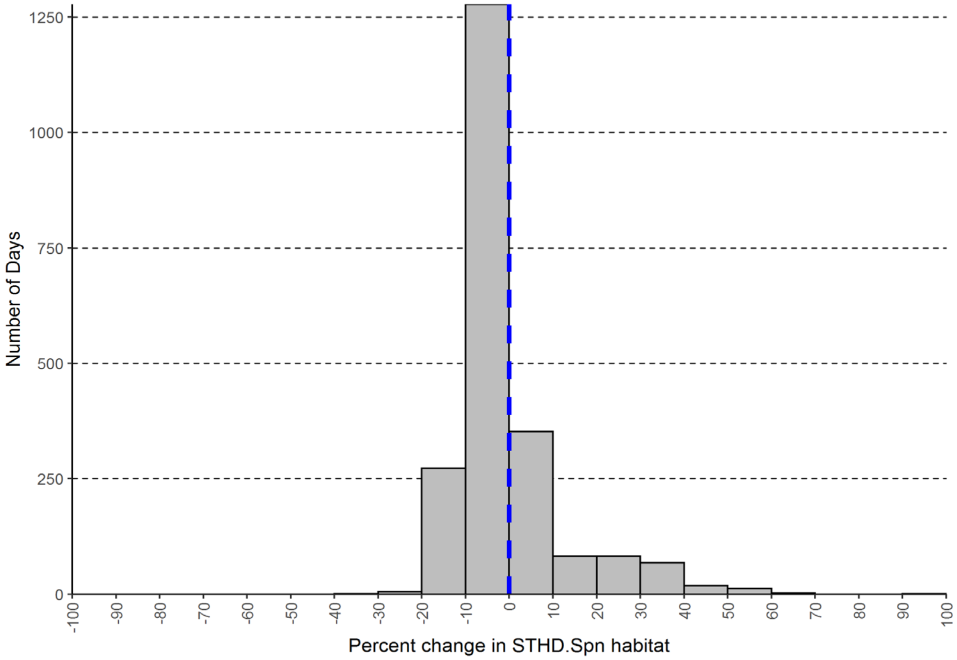

Figure A1-26. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead spawning habitat at the Cape Horn Dam site.

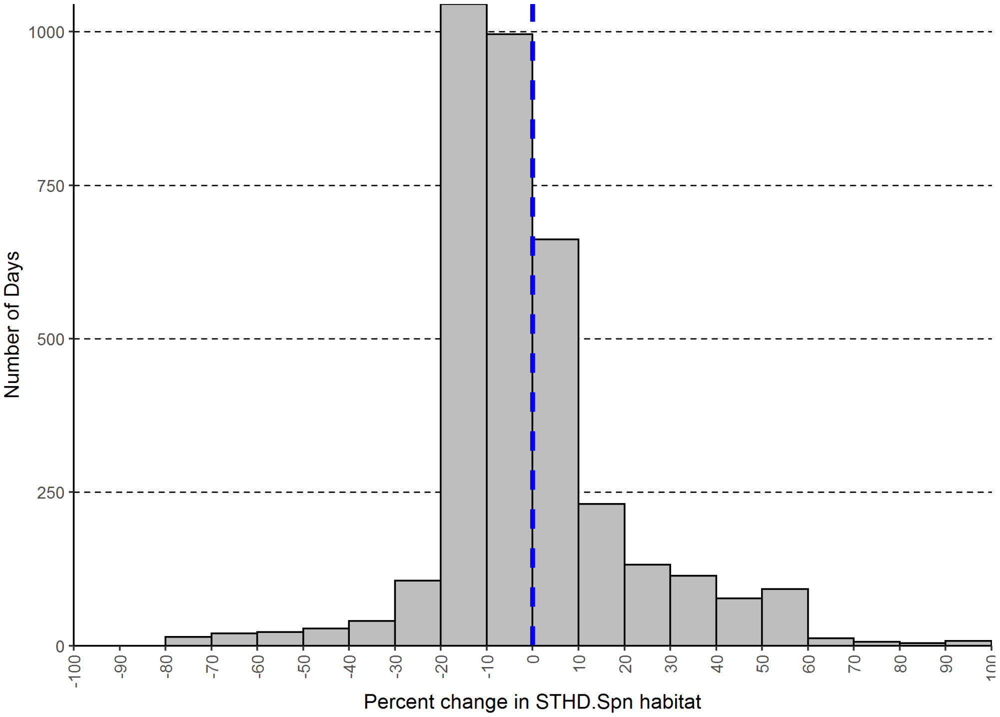

Figure A1-27. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for steelhead spawning habitat at the Cape Horn Dam site.

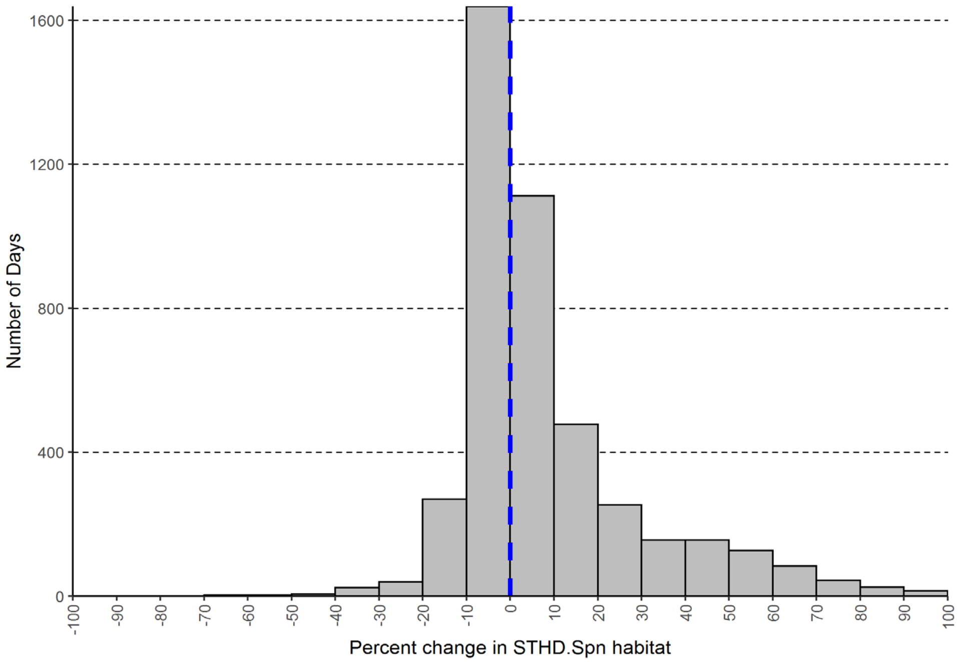

Figure A1-28. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead spawning habitat at the Cape Horn Dam site.

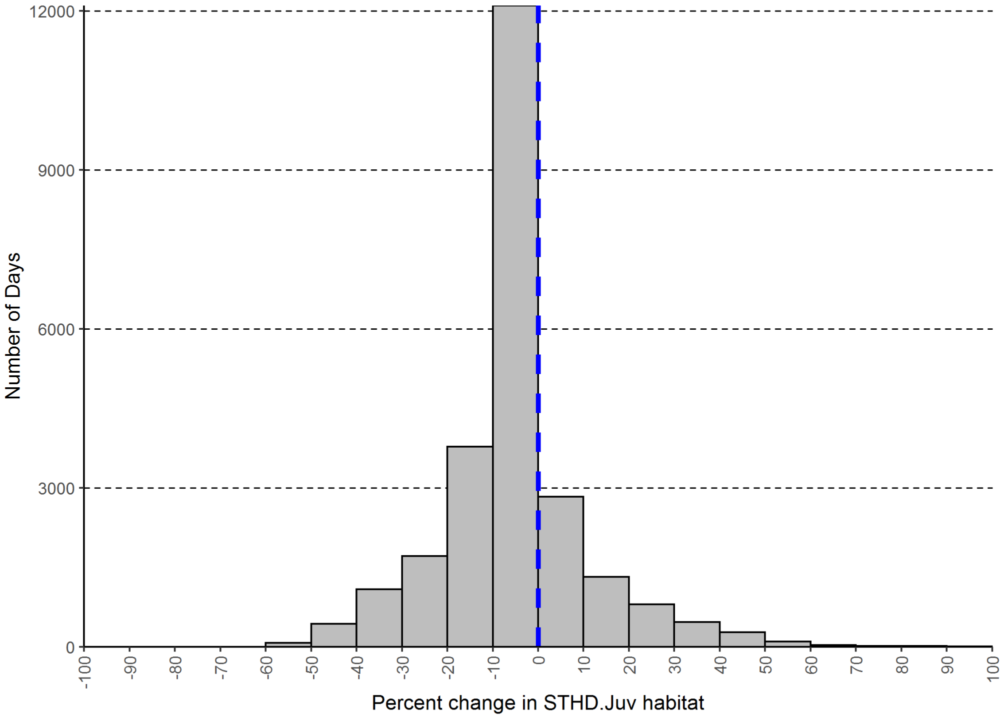

Figure A1-29. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead juvenile habitat at the Cape Horn Dam site.

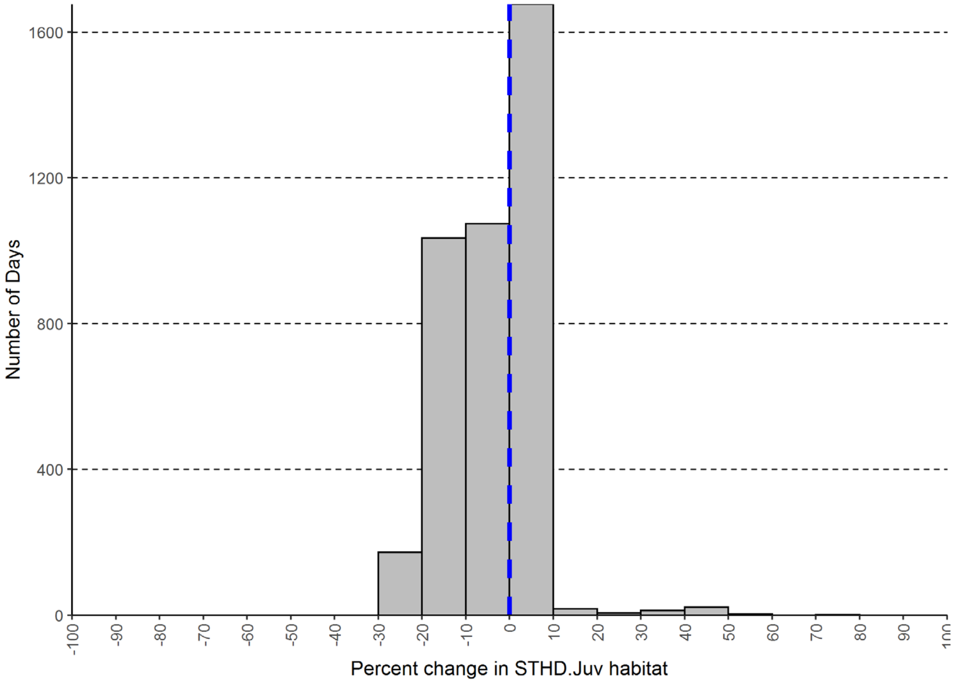

Figure A1-30. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead juvenile habitat at the Cape Horn Dam site.

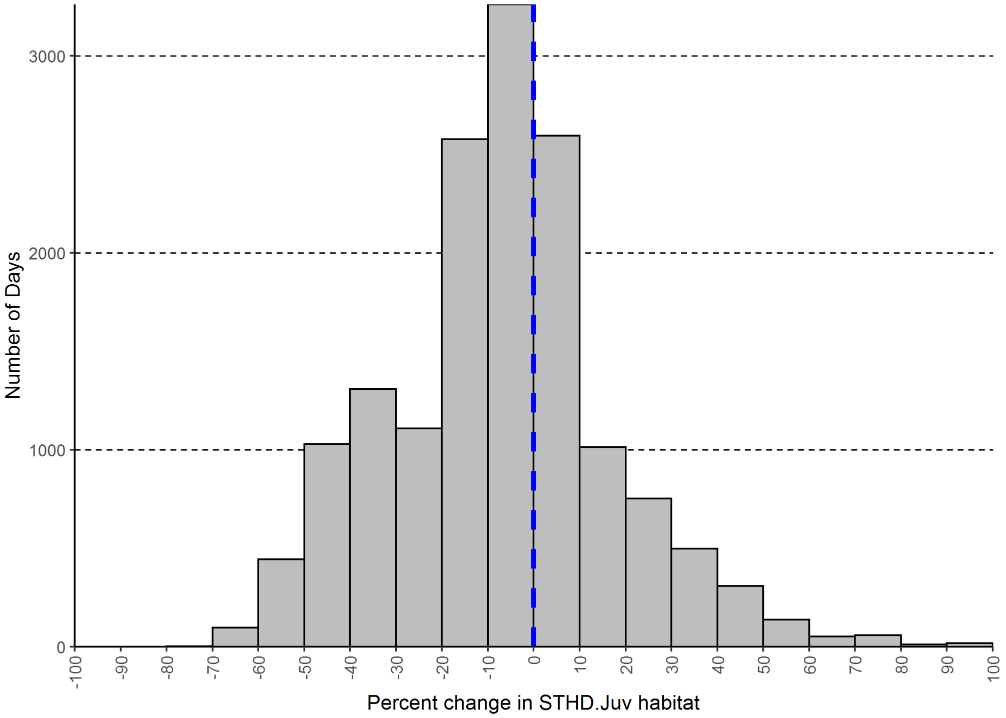

Figure A1-31. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for steelhead juvenile habitat at the Cape Horn Dam site.

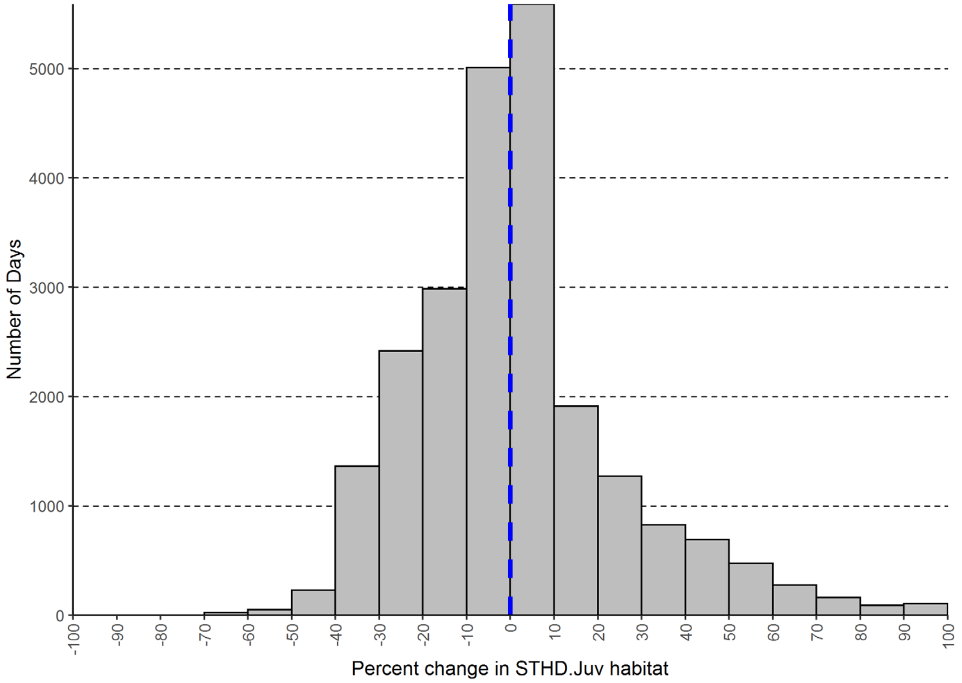

Figure A1-32. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead juvenile habitat at the Cape Horn Dam site.

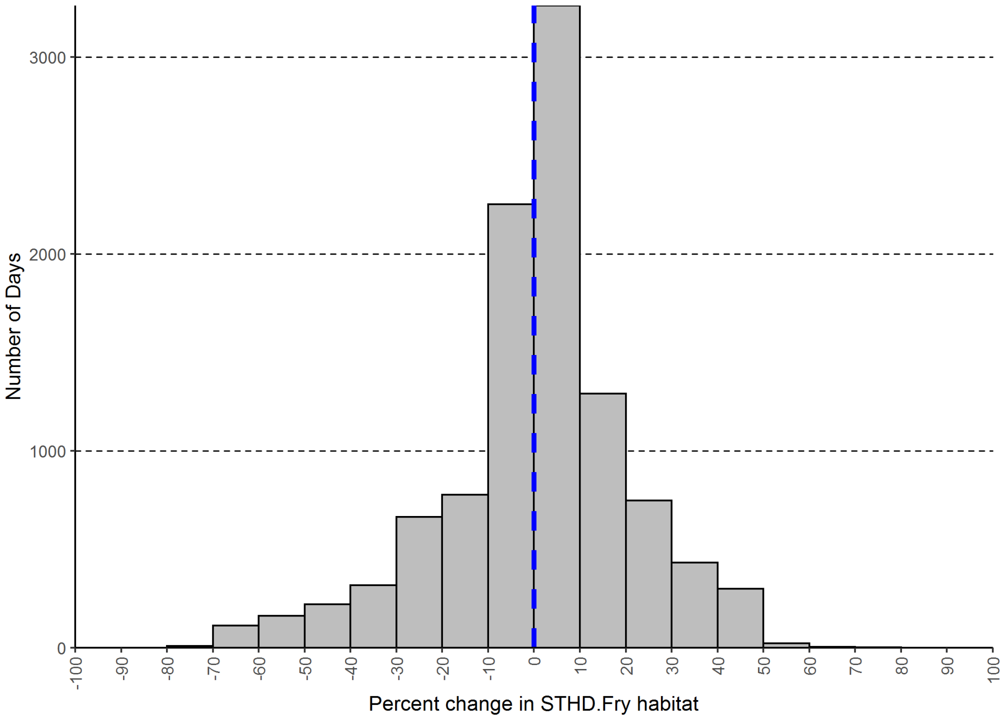

Figure A1-33. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead fry habitat at the Cape Horn Dam site.

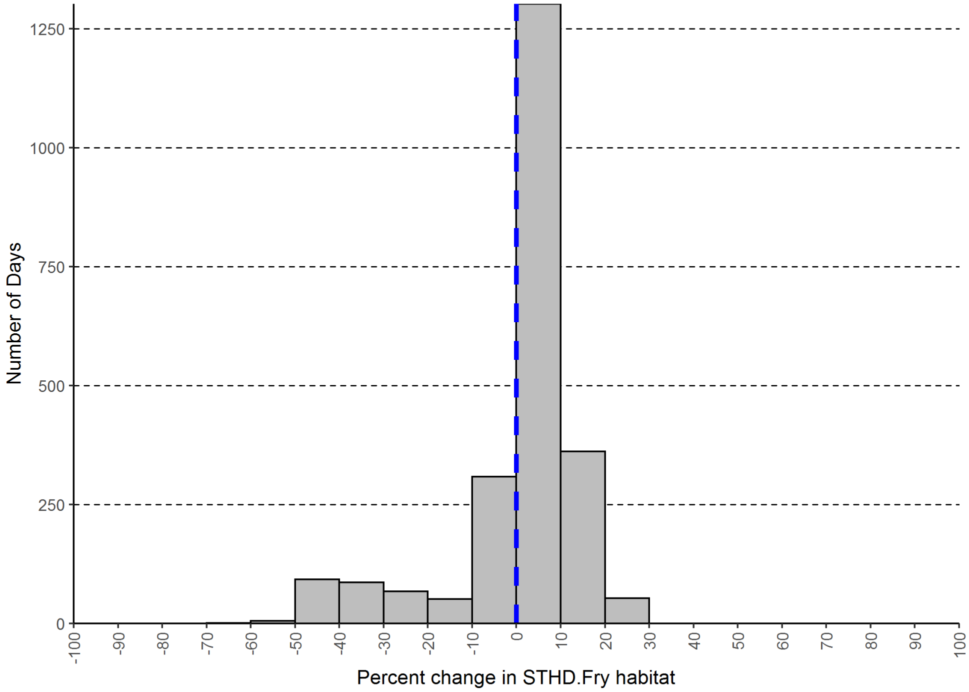

Figure A1-34. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead fry habitat at the Cape Horn Dam site.

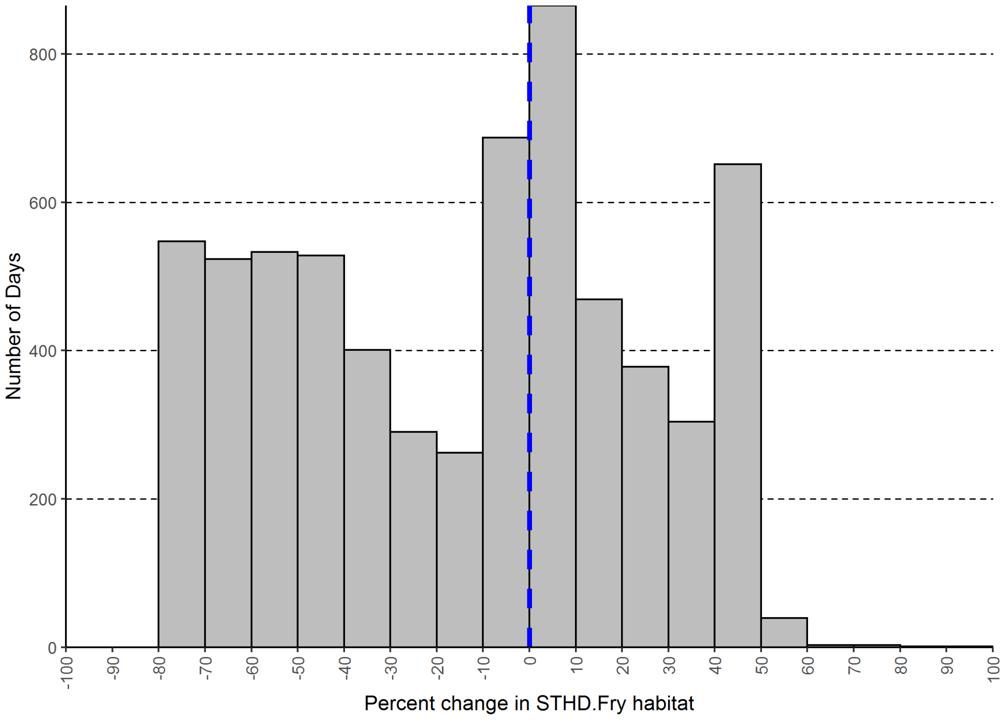

Figure A1-35. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for steelhead fry habitat at the Cape Horn Dam site.

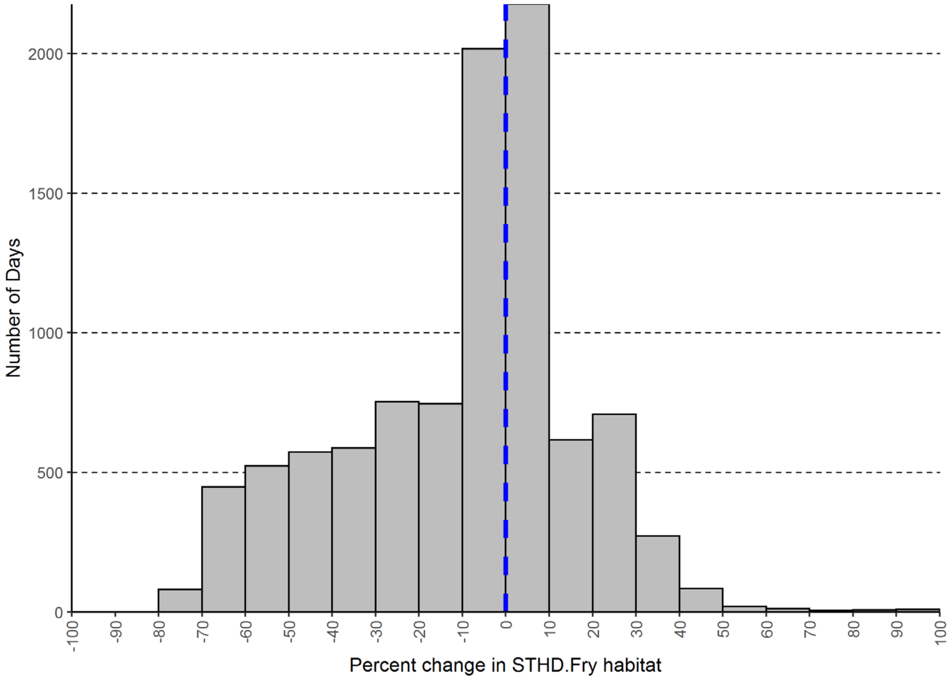

Figure A1-36. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead fry habitat at the Cape Horn Dam site.

A-1.2.2 Big Bend Site

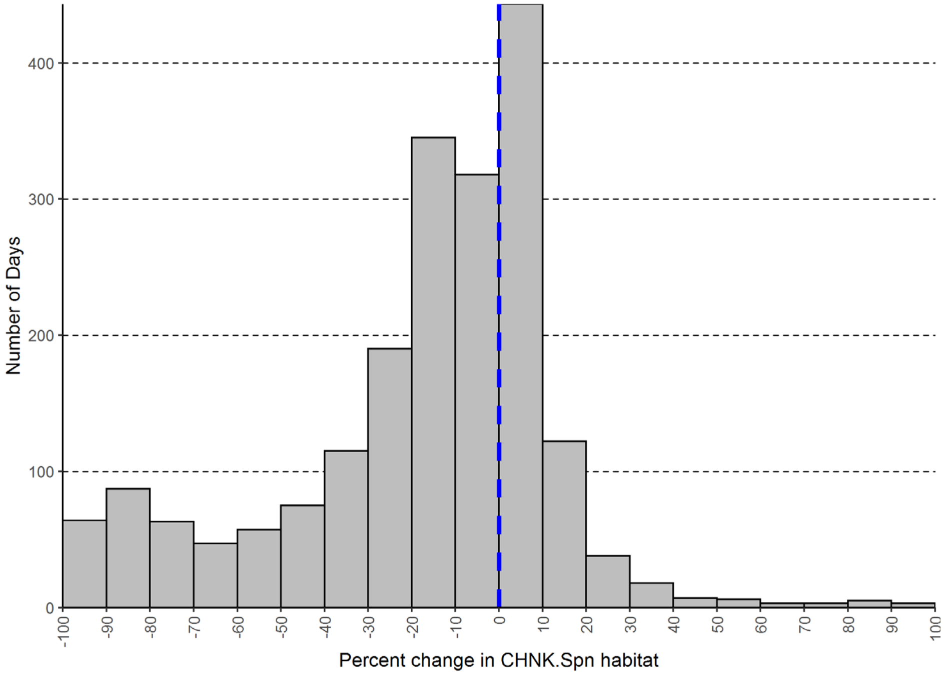

Figure A1-37. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for Chinook Salmon spawning habitat at the Big Bend site.

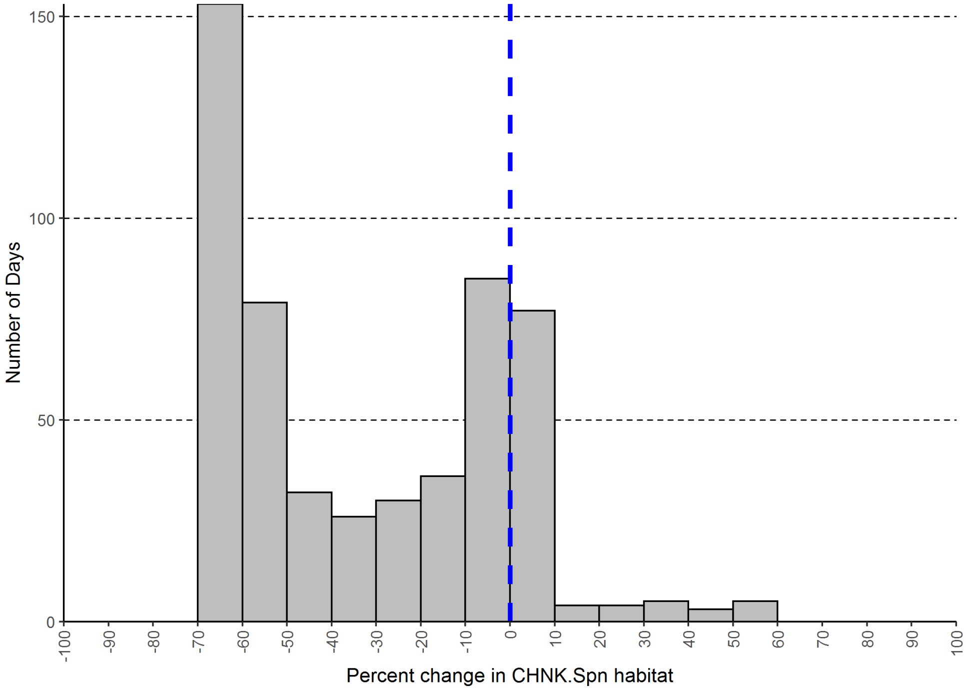

Figure A1-38. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for Chinook Salmon spawning habitat at the Big Bend site.

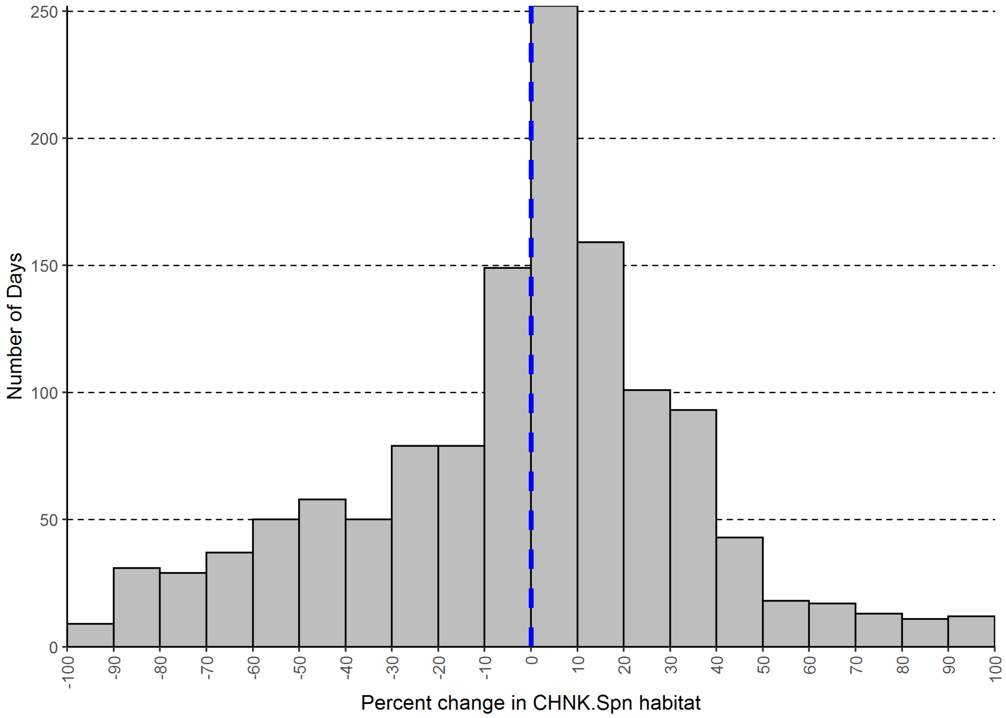

Figure A1-39. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for Chinook Salmon spawning habitat at the Big Bend site.

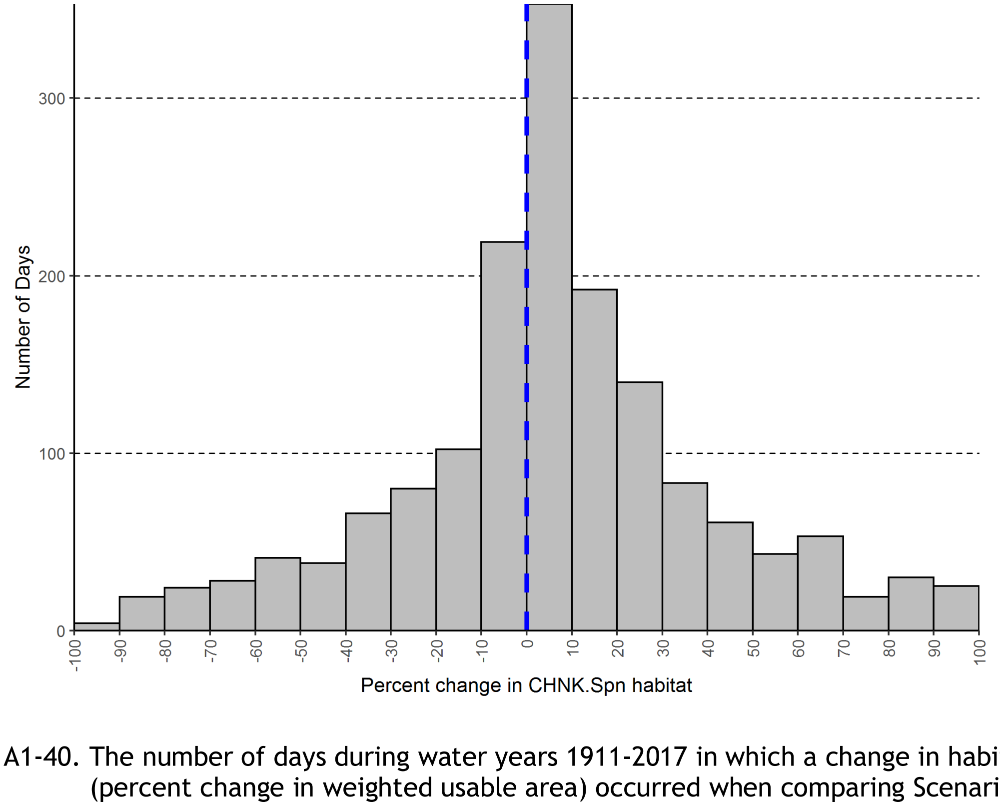

Figure A1-40. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for Chinook Salmon spawning habitat at the Big Bend site.

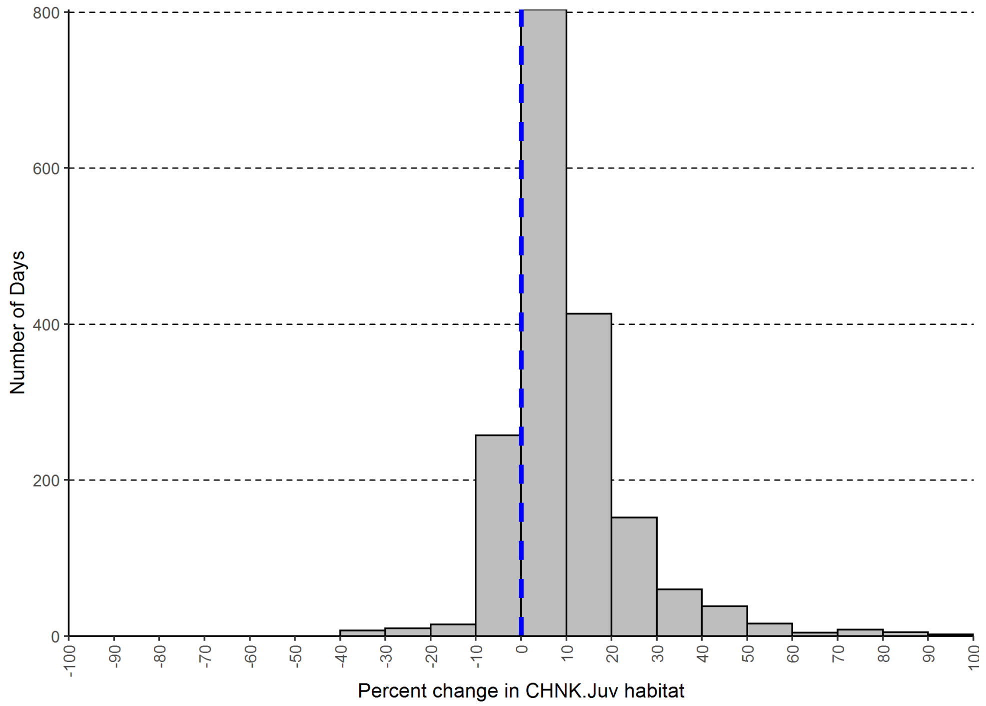

Figure A1-41. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for Chinook Salmon juvenile habitat at the Big Bend site.

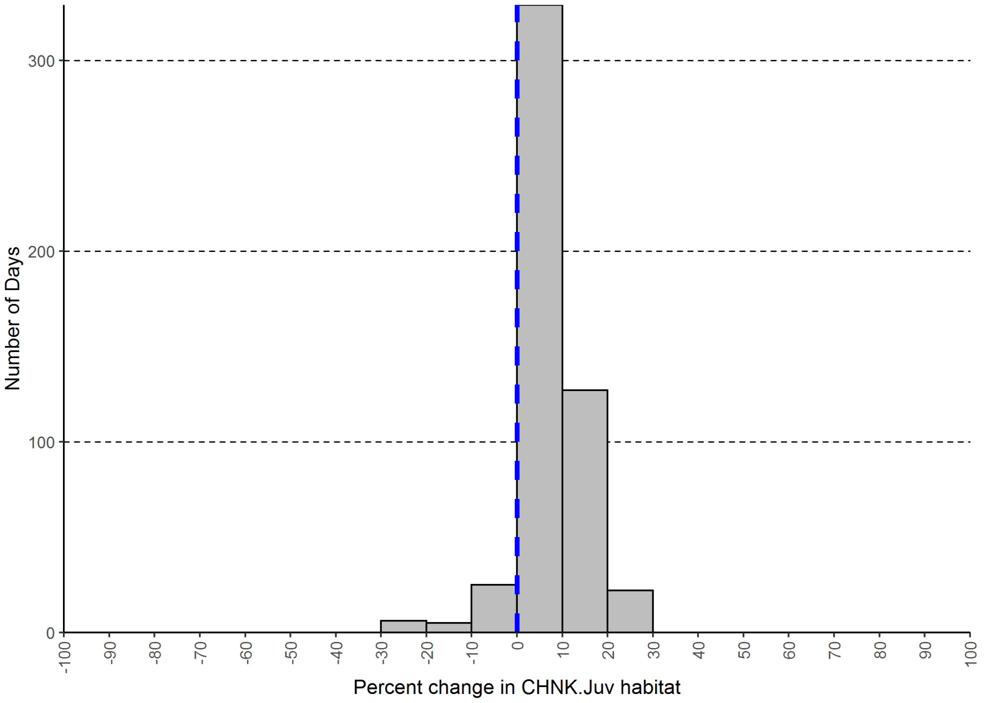

Figure A1-42. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for Chinook Salmon juvenile habitat at the Big Bend site.

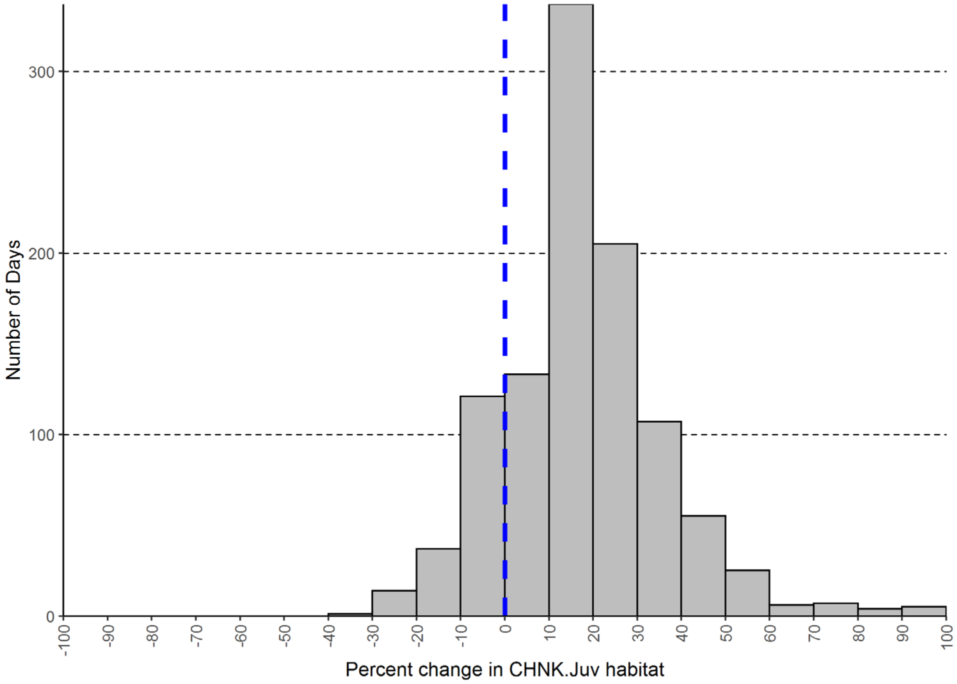

Figure A1-43. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for Chinook Salmon juvenile habitat at the Big Bend site.

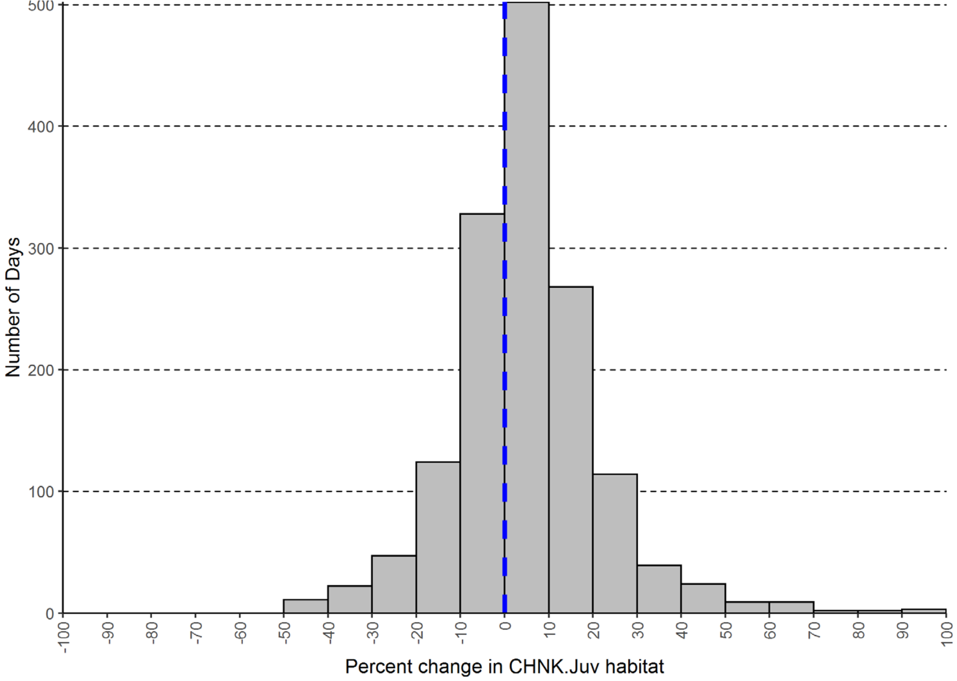

Figure A1-44. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for Chinook Salmon juvenile habitat at the Big Bend site.

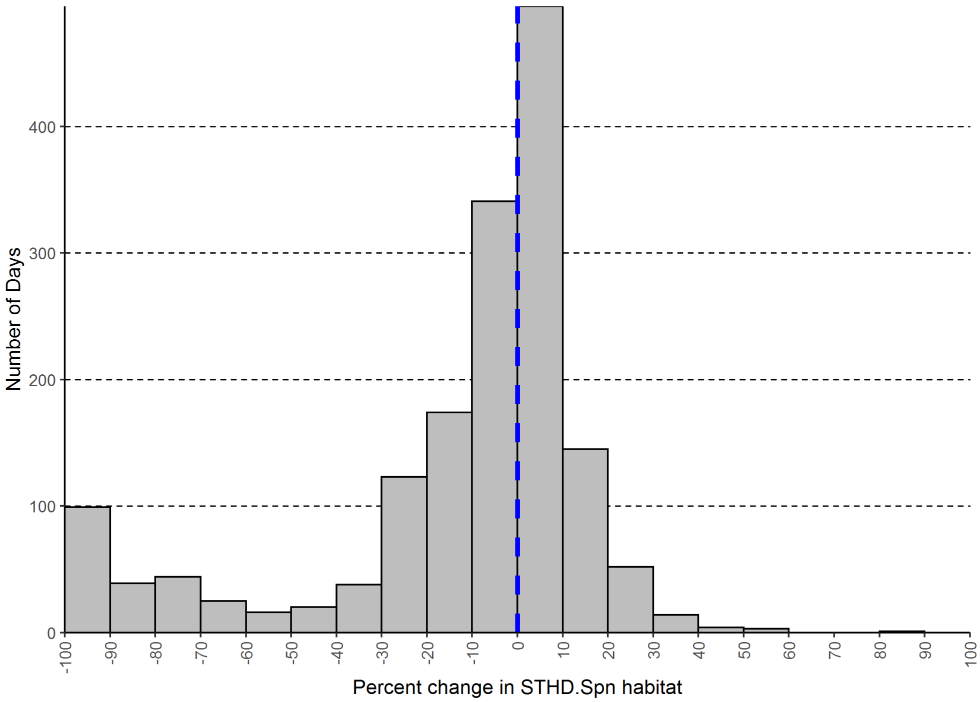

Figure A1-45. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead spawning habitat at the Big Bend site.

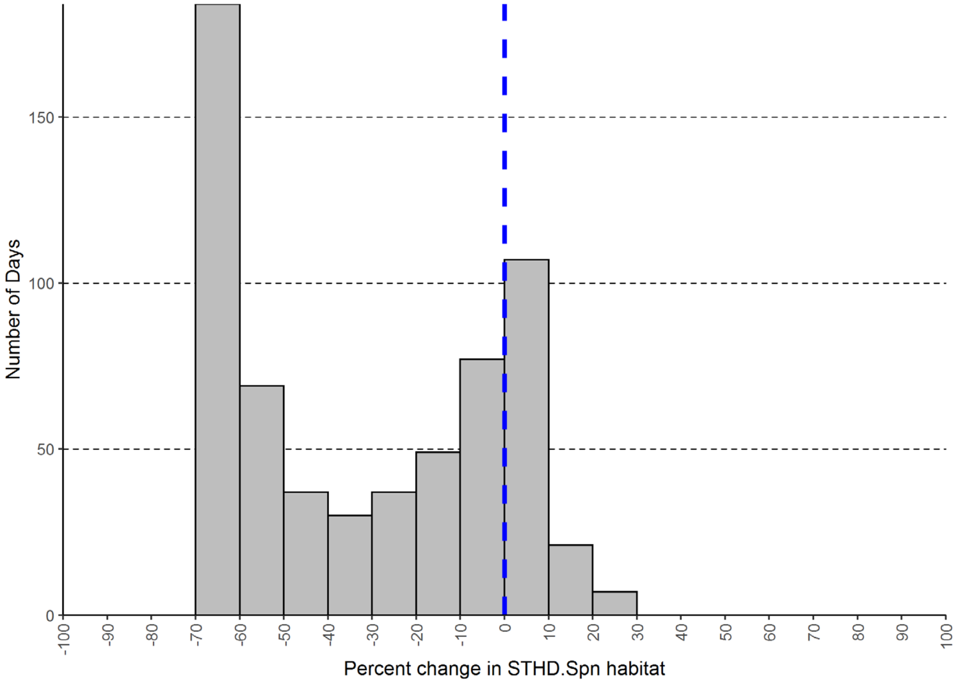

Figure A1-46. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead spawning habitat at the Big Bend site.

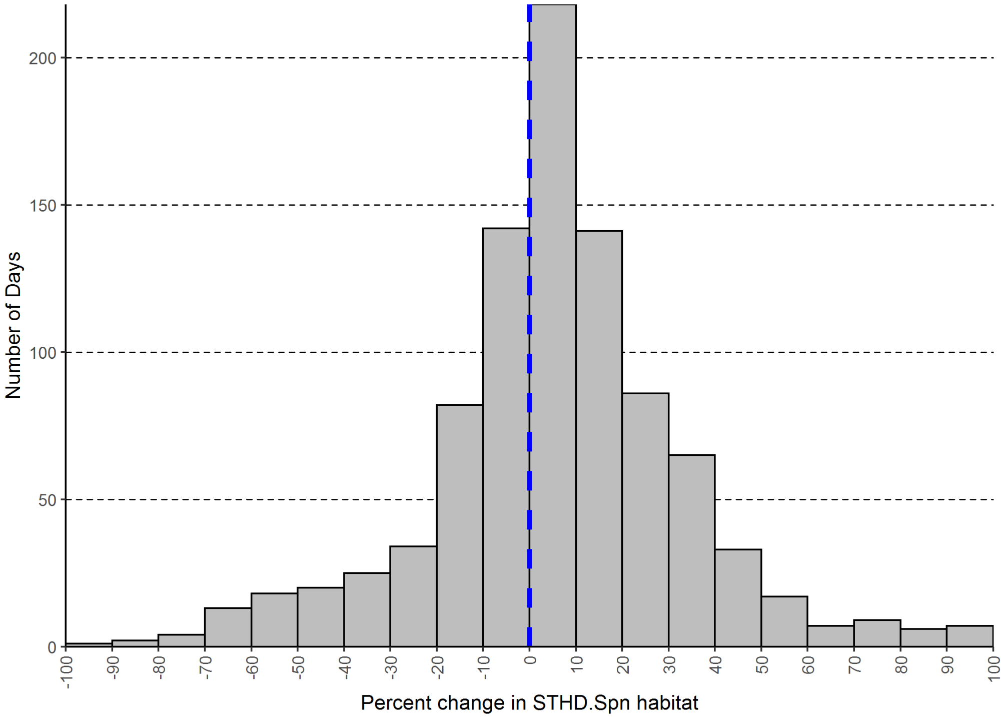

Figure A1-47. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for steelhead spawning habitat at the Big Bend site.

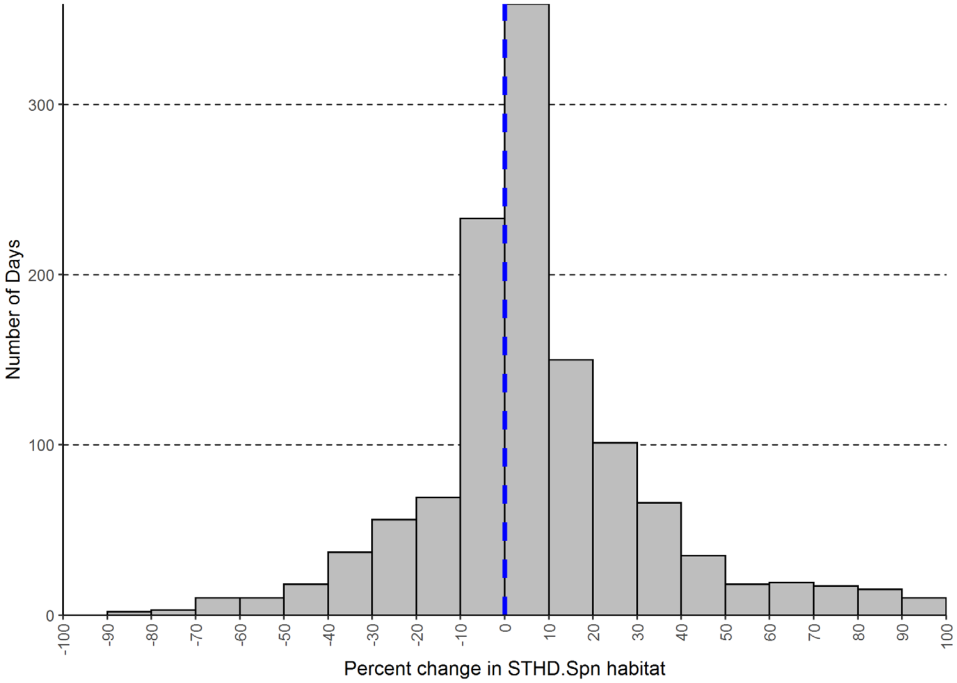

Figure A1-48. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead spawning habitat at the Big Bend site.

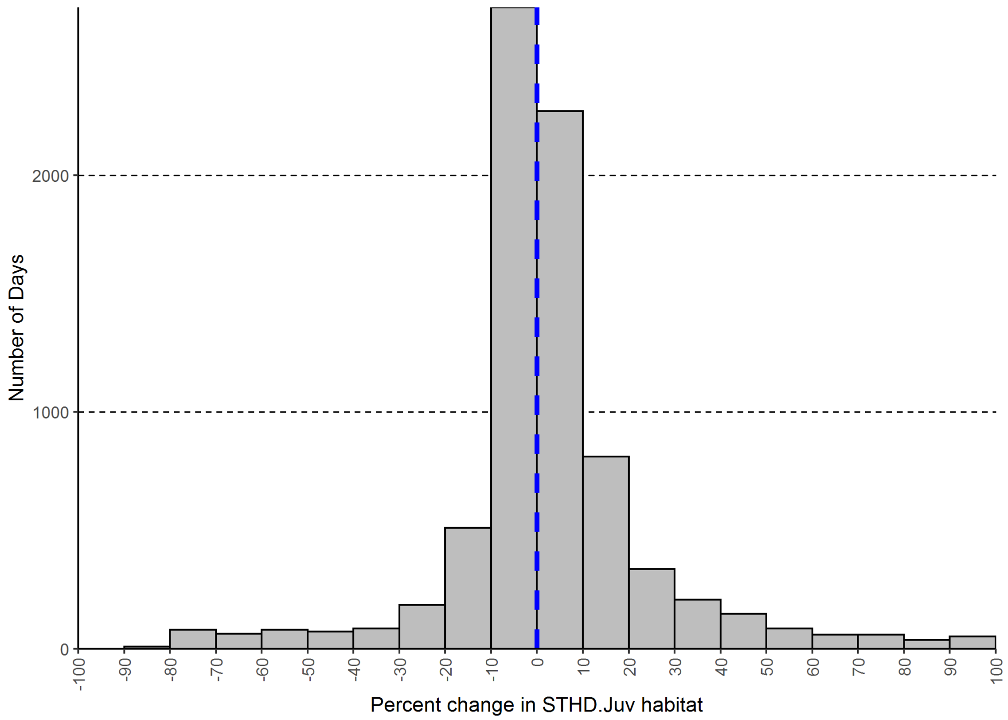

Figure A1-49. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead juvenile habitat at the Big Bend site.

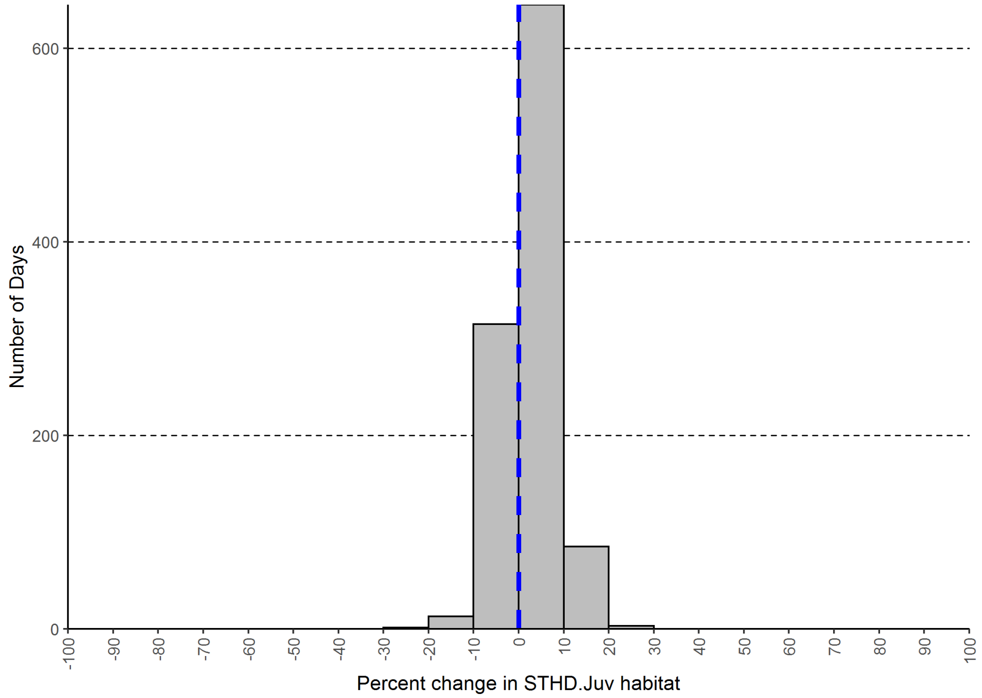

Figure A1-50. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead juvenile habitat at the Big Bend site.

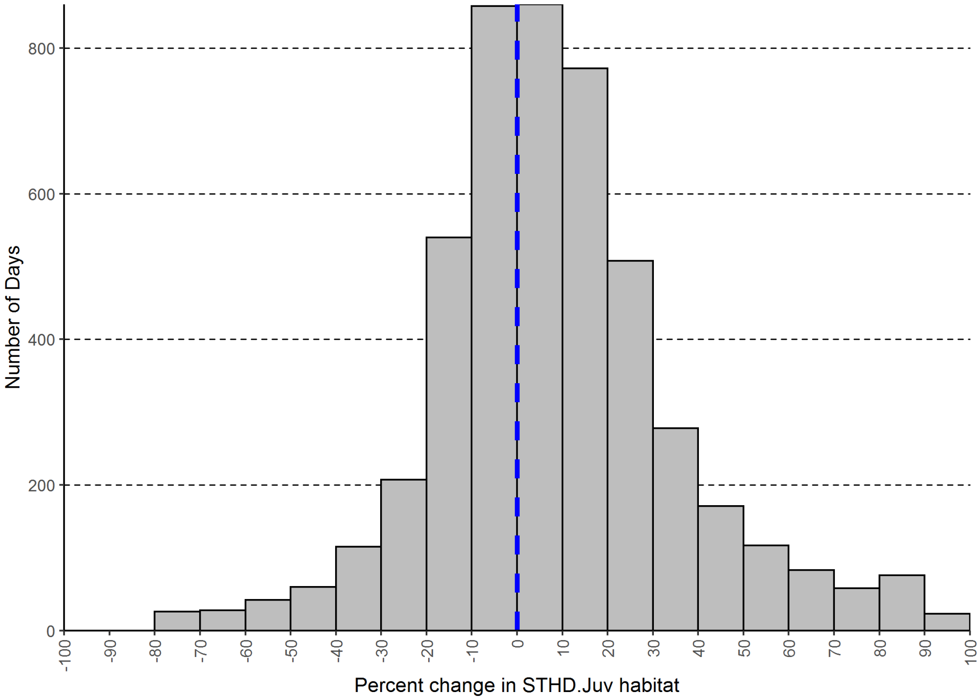

Figure A1-51. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for steelhead juvenile habitat at the Big Bend site.

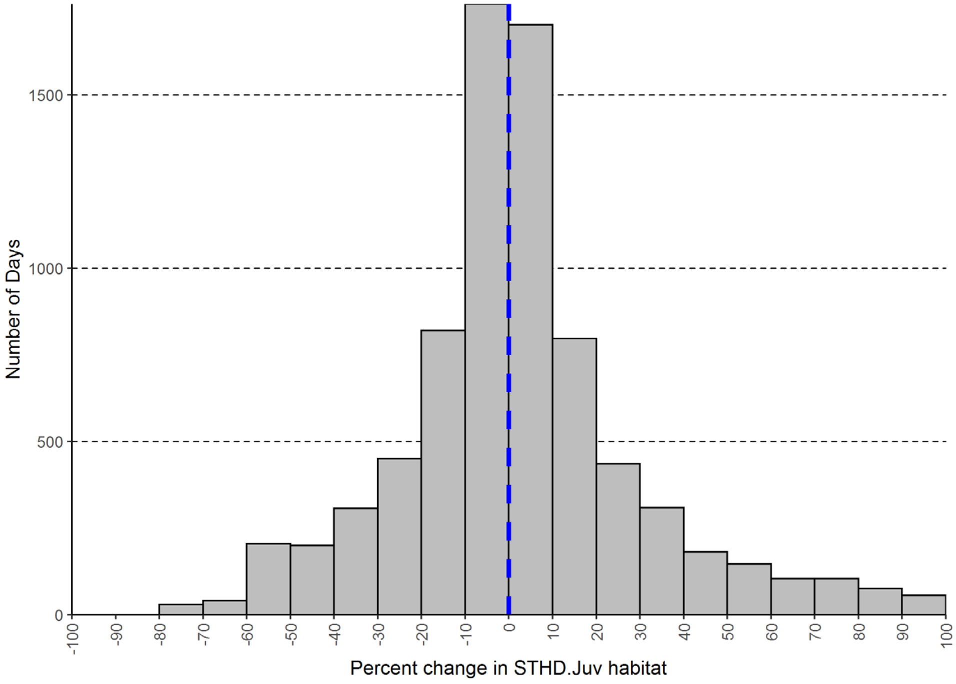

Figure A1-52. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead juvenile habitat at the Big Bend site.

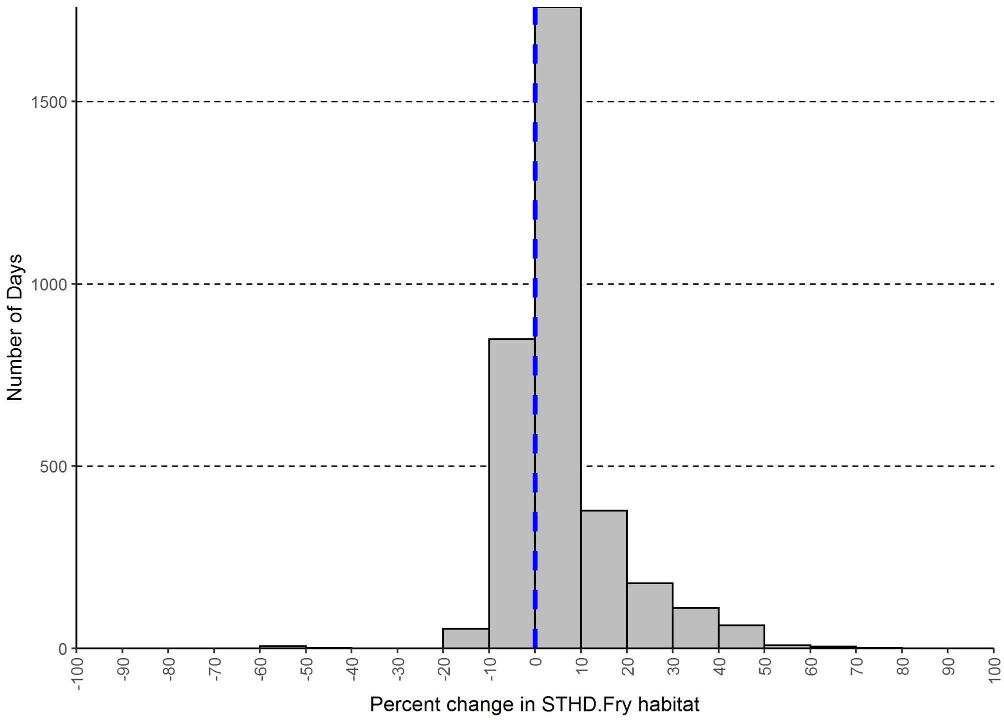

Figure A1-53. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead fry habitat at the Big Bend site.

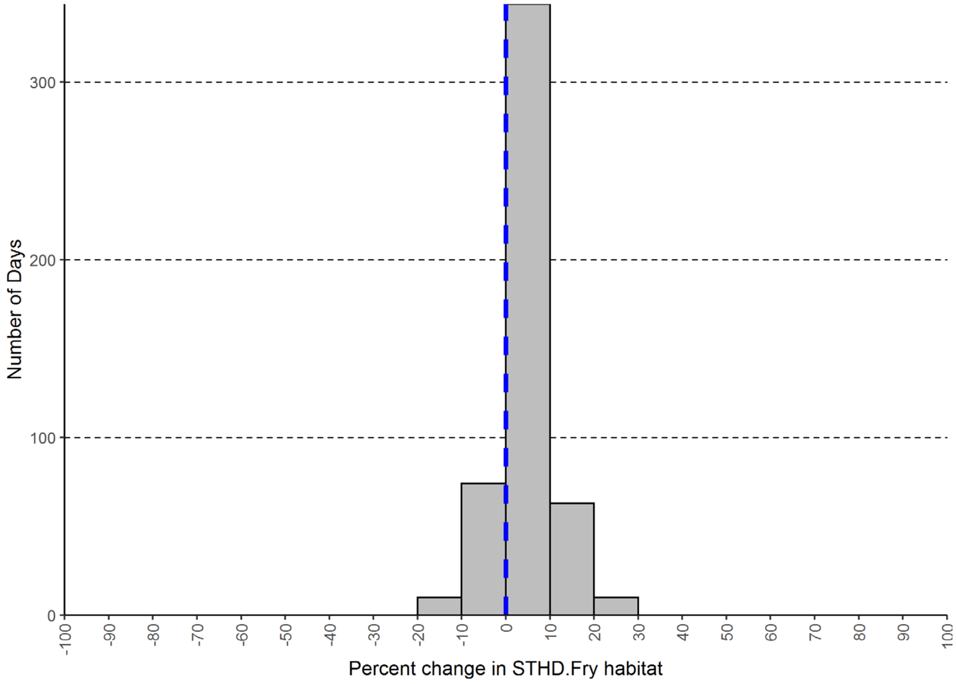

Figure A1-54. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead fry habitat at the Big Bend site.

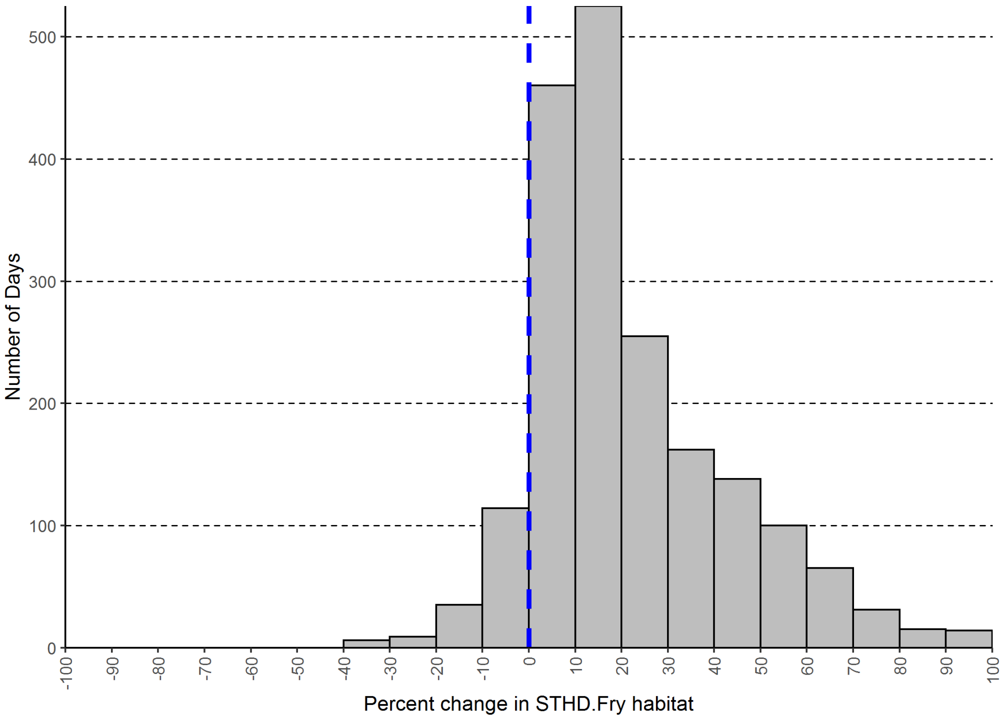

Figure A1-55. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for steelhead fry habitat at the Big Bend site.

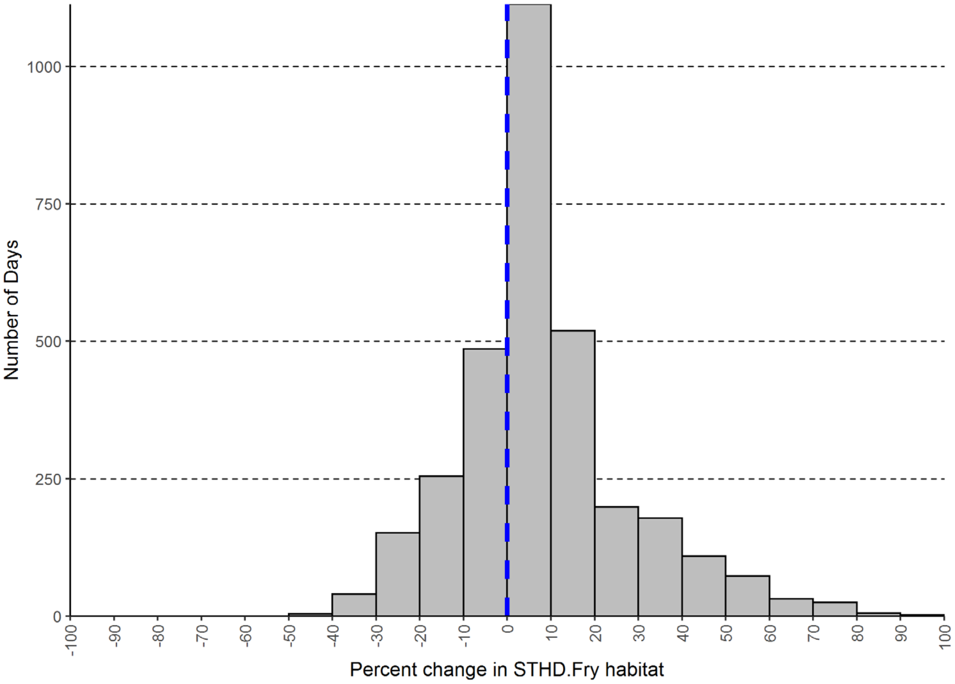

Figure A1-56. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead fry habitat at the Big Bend site.

A-2 RUSSIAN RVER

A-2.1 Russian River Downstream of West Branch Confluence

A-2.1.1 Ukiah Site

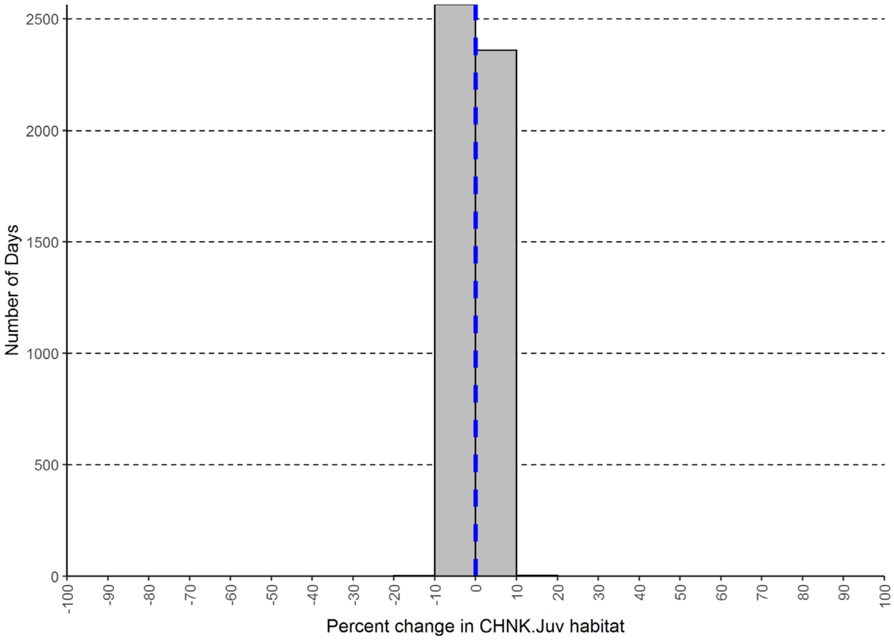

Figure A2-1. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for Chinook Salmon juvenile habitat at the Ukiah site.

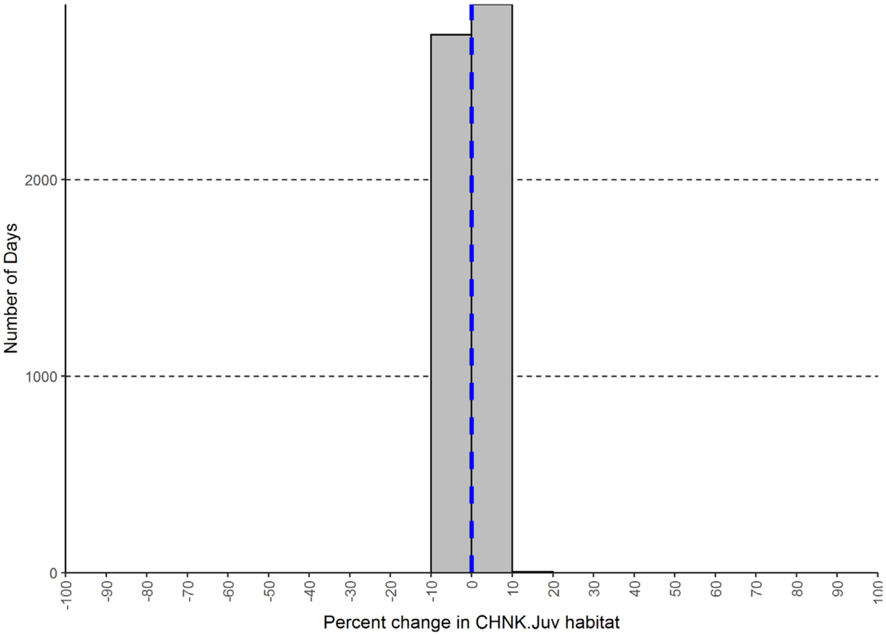

Figure A2-2. The number of days, sorted by a range of magnitudes, where a change in habitat occurred when water supply scenario was changed from Baseline to Scenario 4B for Chinook Salmon juvenile habitat at the Ukiah site.

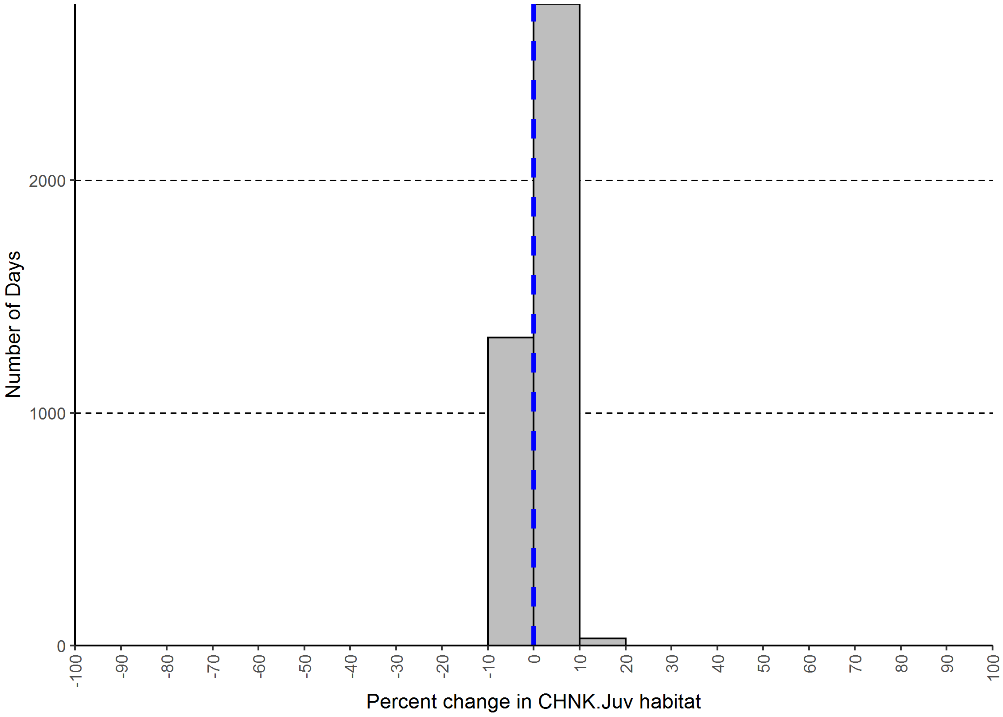

Figure A2-3. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for Chinook Salmon juvenile habitat at the Ukiah site.

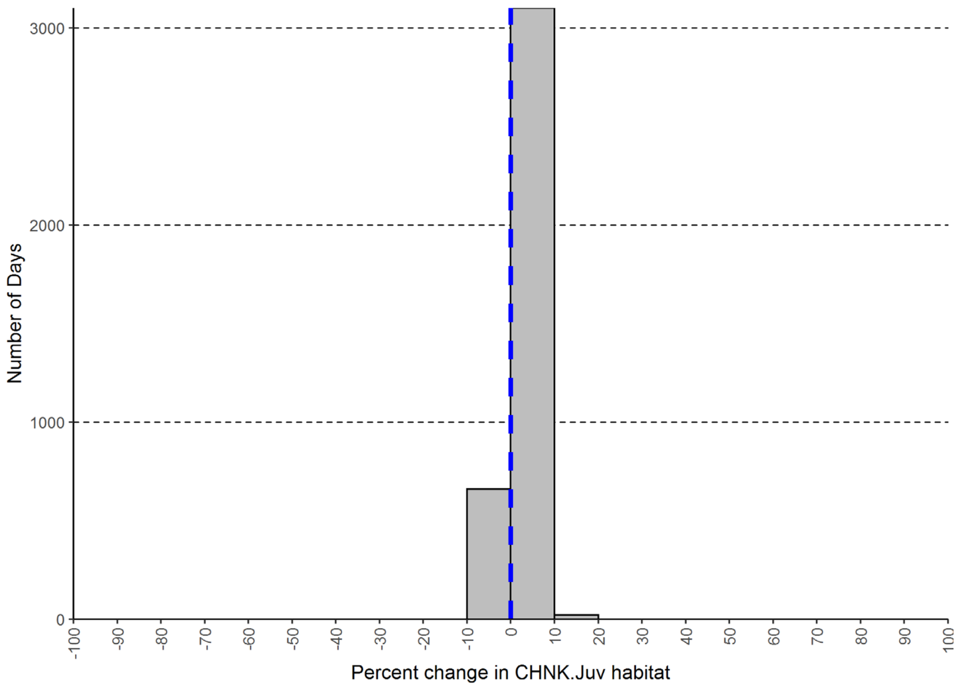

Figure A2-4. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for Chinook Salmon juvenile habitat at the Ukiah site.

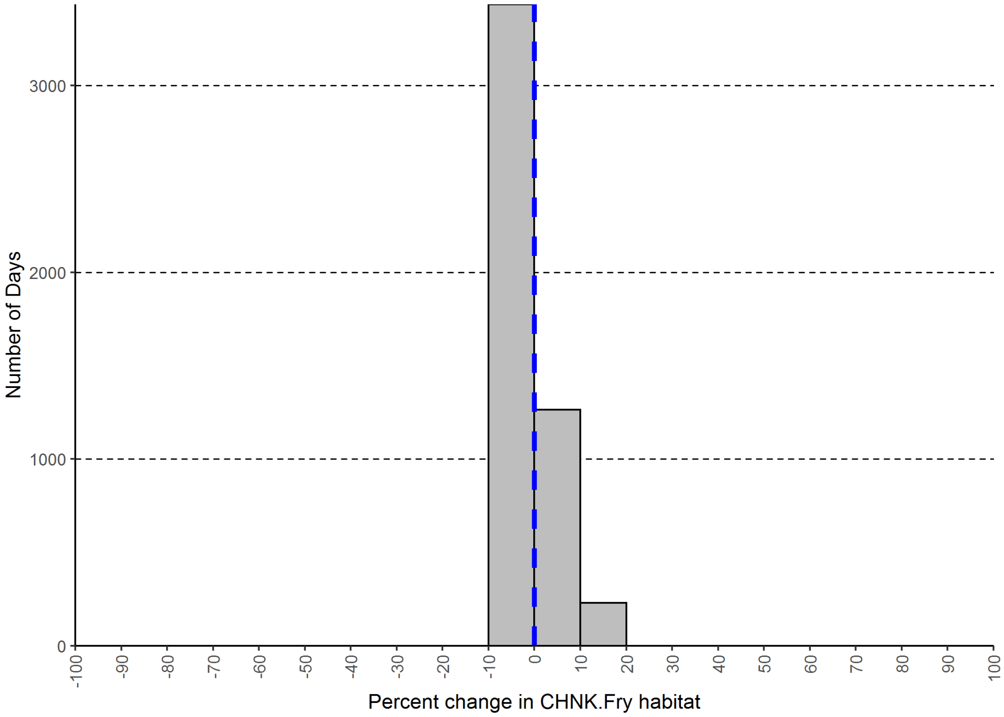

Figure A2-5. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for Chinook Salmon fry habitat at the Ukiah site.

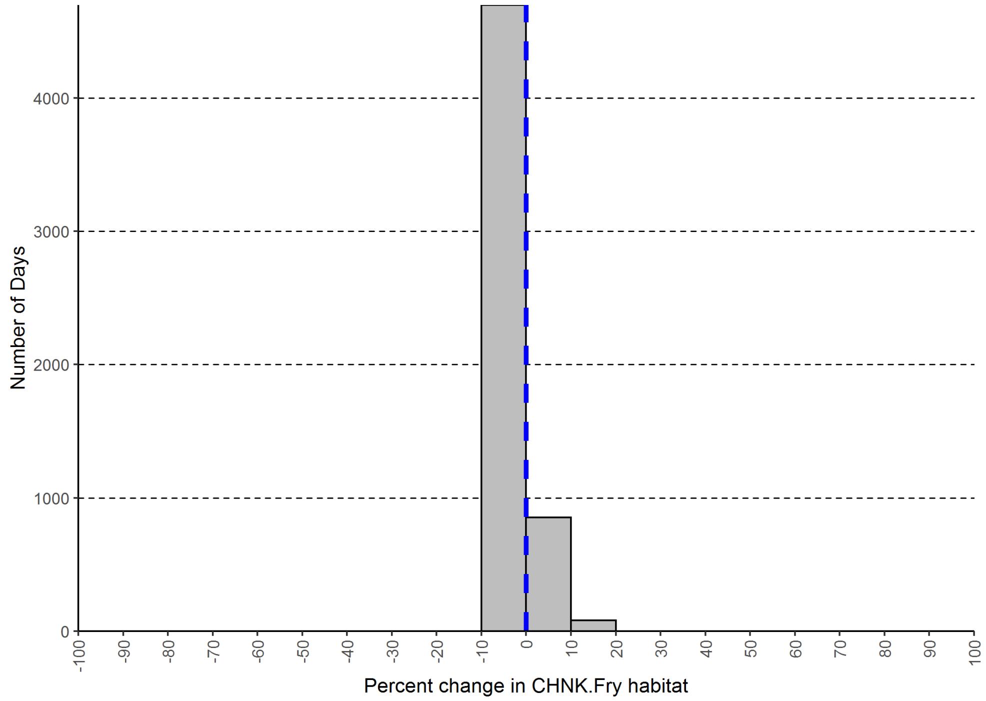

Figure A2-6. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for Chinook Salmon fry habitat at the Ukiah site.

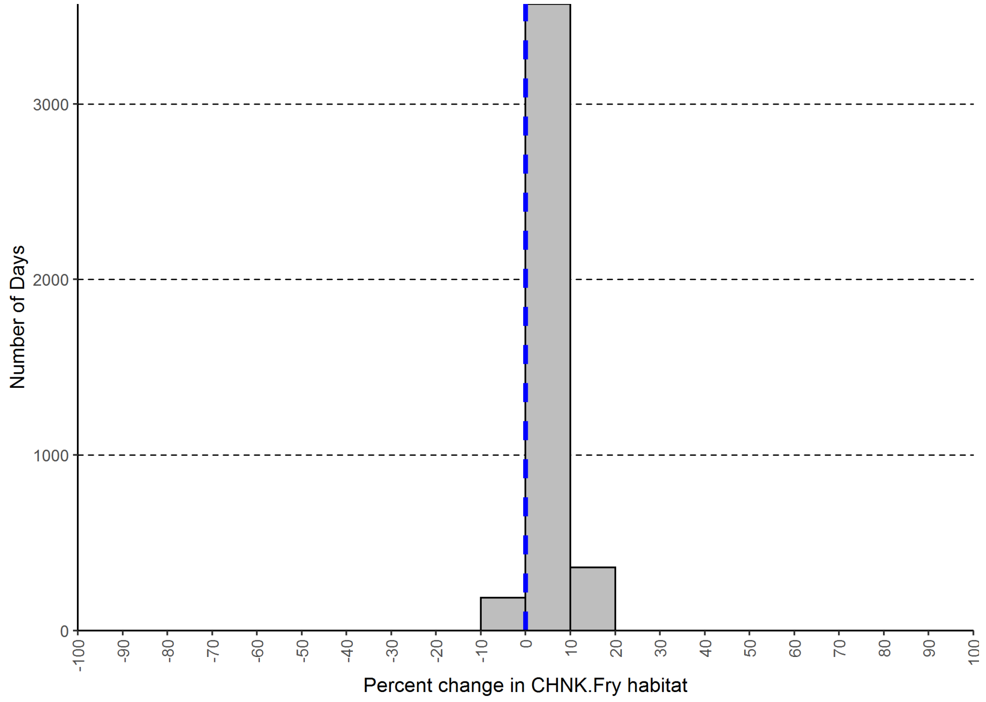

Figure A2-7. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for Chinook Salmon fry habitat at the Ukiah site.

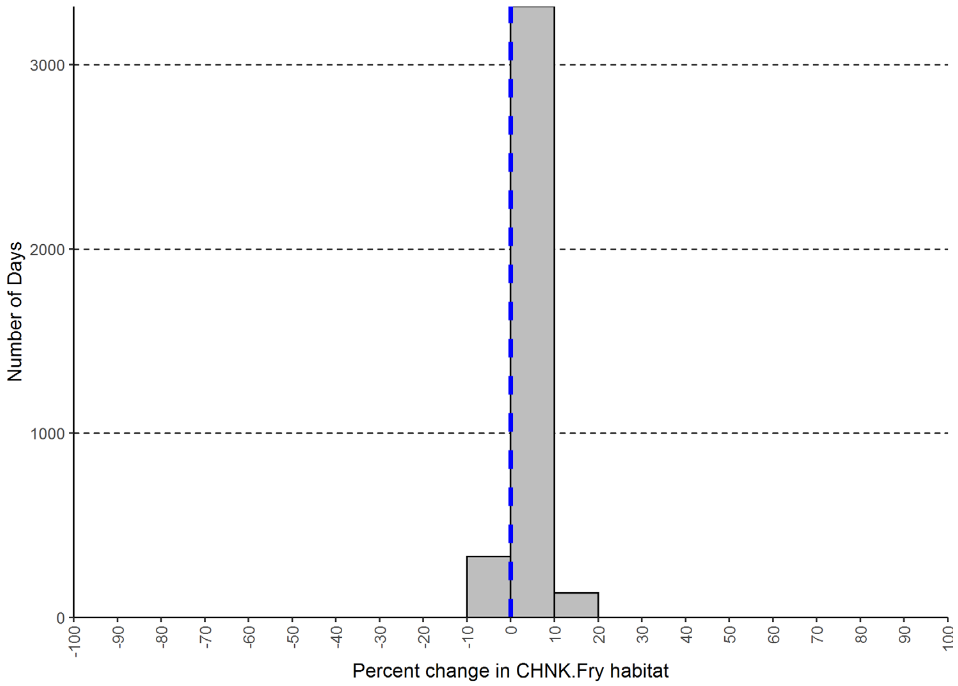

Figure A2-8. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for Chinook Salmon fry habitat at the Ukiah site.

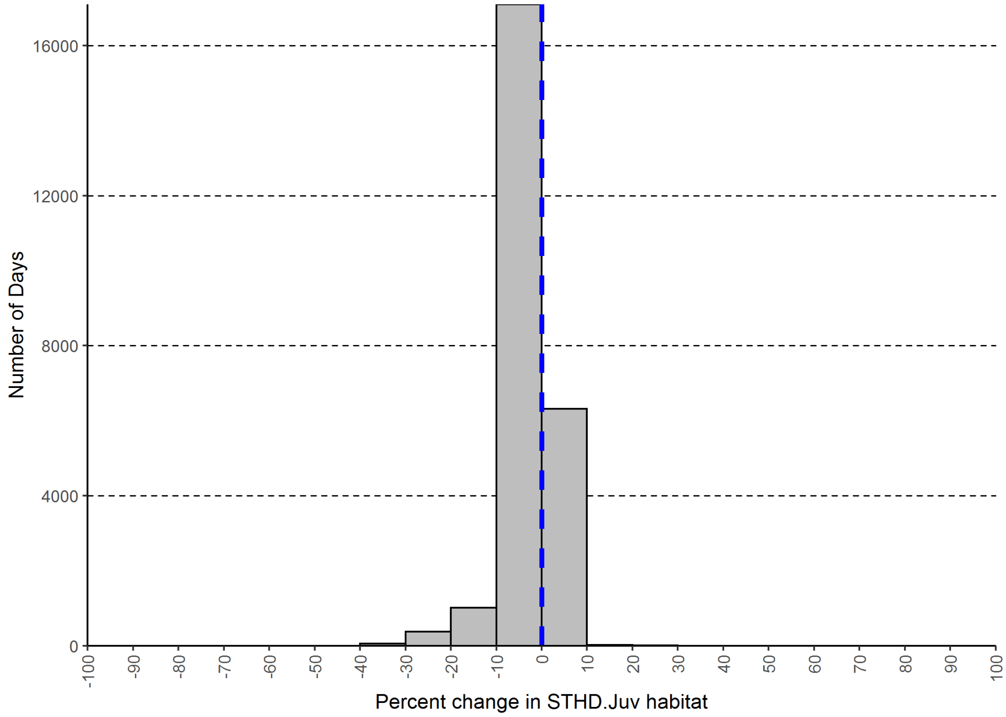

Figure A2-9. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead juvenile habitat at the Ukiah site.

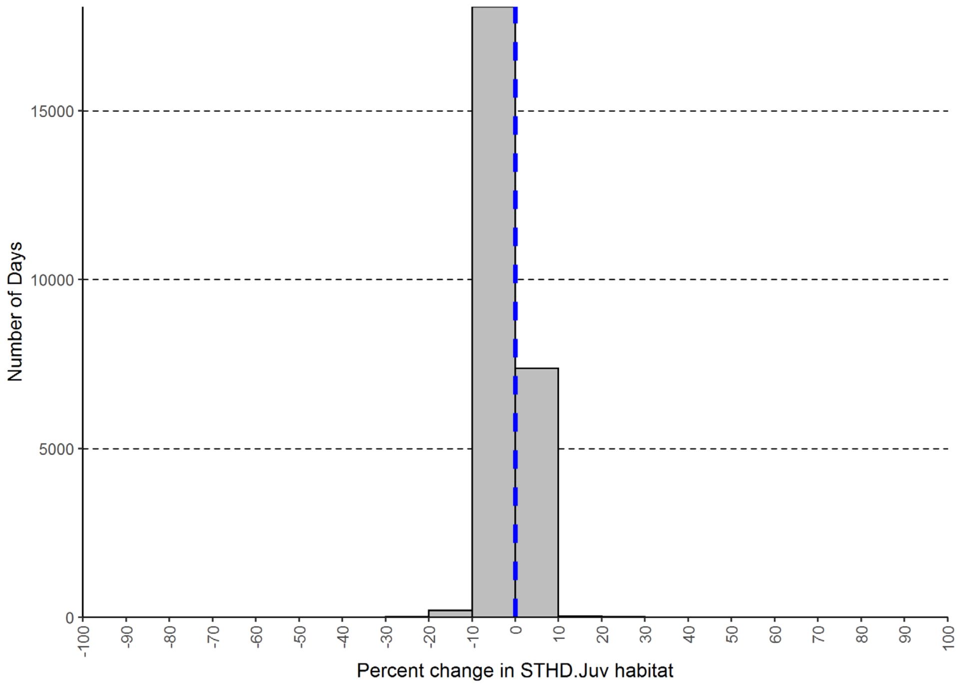

Figure A2-10. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead juvenile habitat at the Ukiah site.

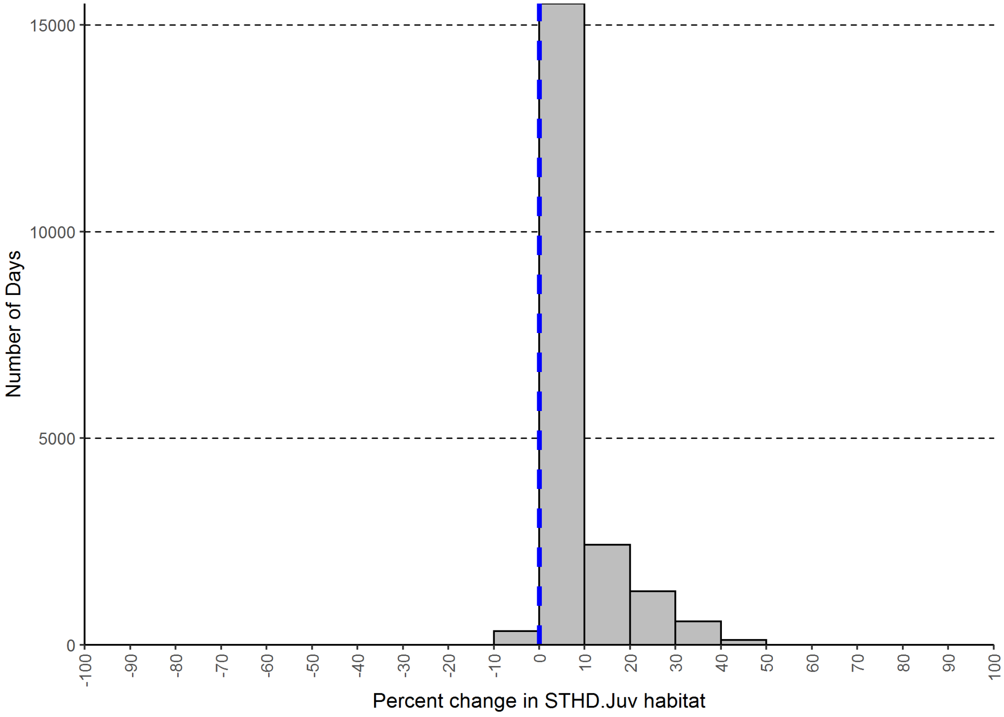

Figure A2-11. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for steelhead juvenile habitat at the Ukiah site.

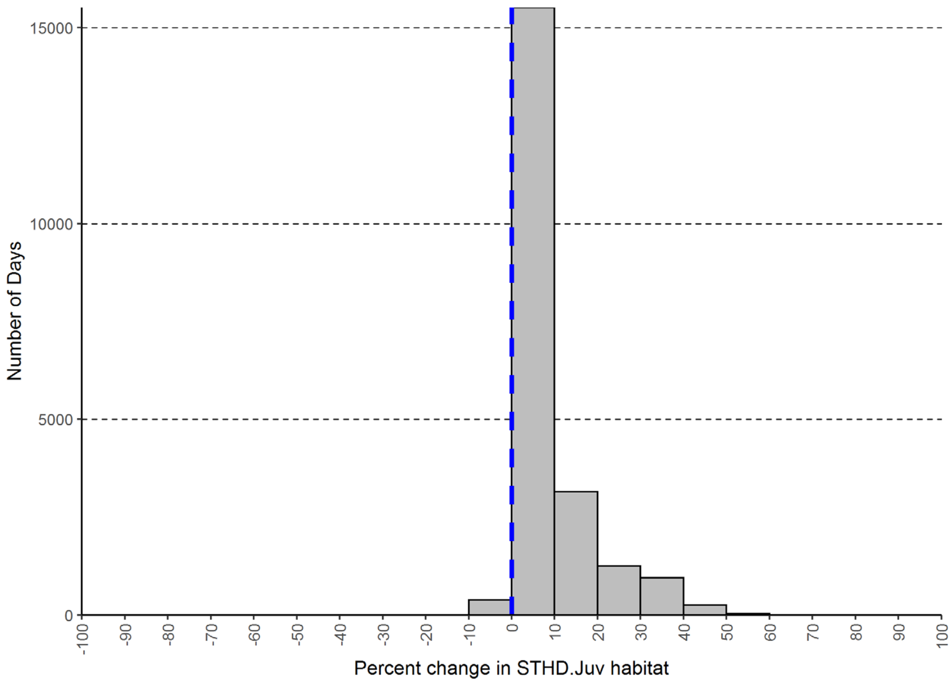

Figure A2-12. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead juvenile habitat at the Ukiah site.

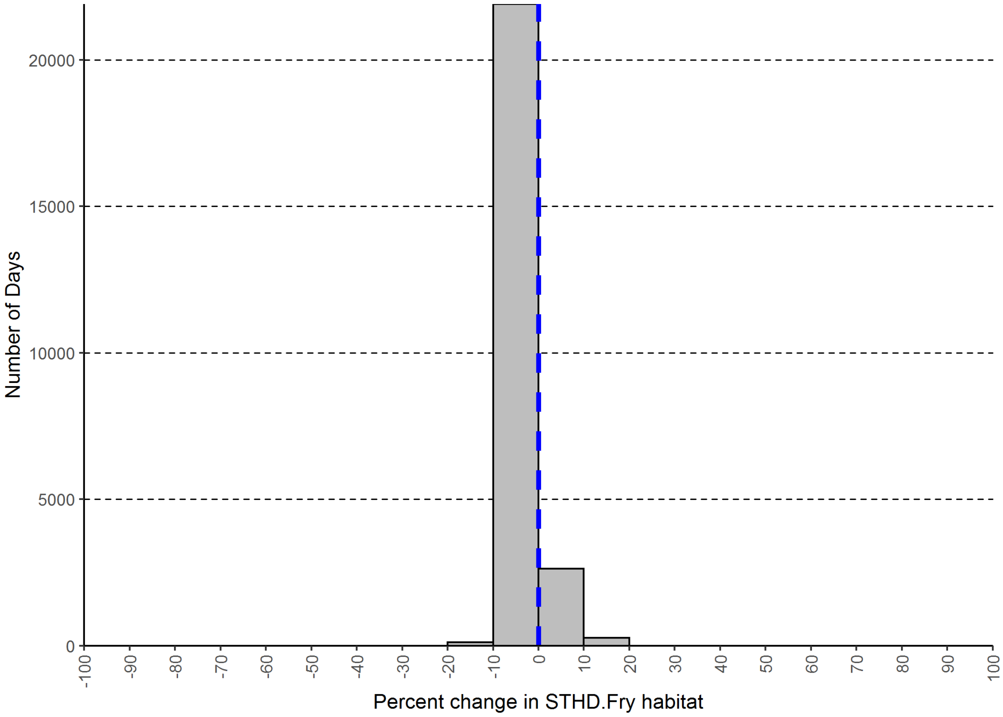

Figure A2-13. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead fry habitat at the Ukiah site.

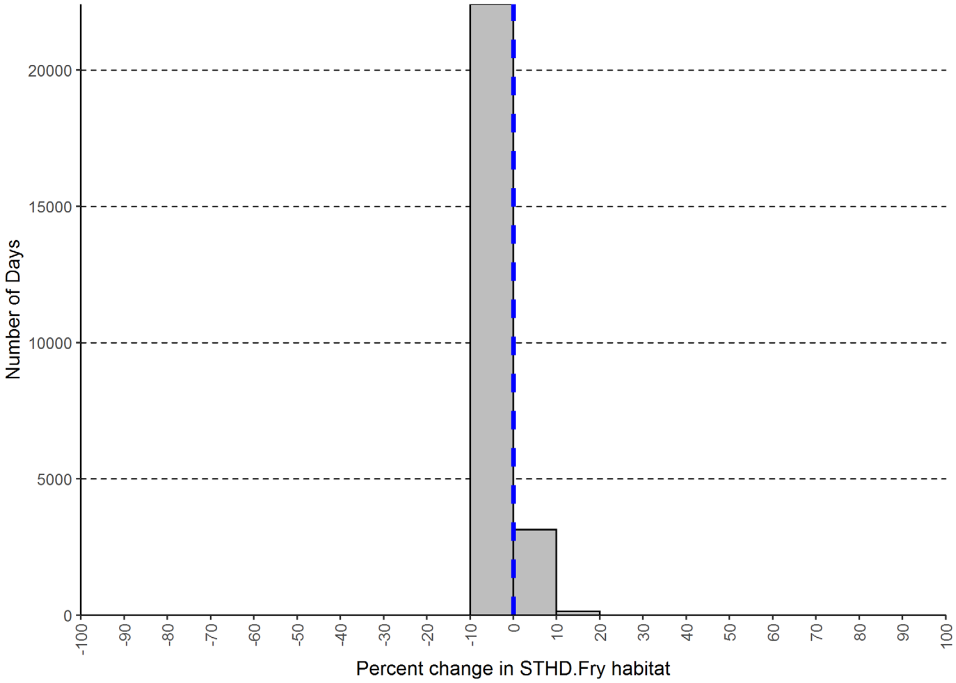

Figure A2-14. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead fry habitat at the Ukiah site.

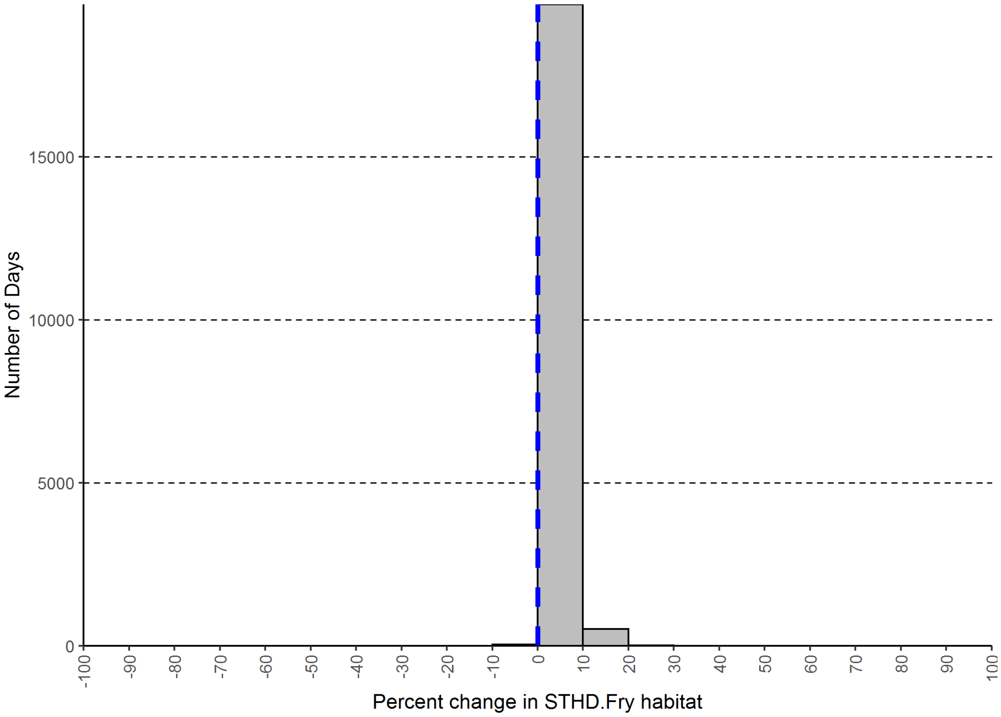

Figure A2-15. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for steelhead fry habitat at the Ukiah site.

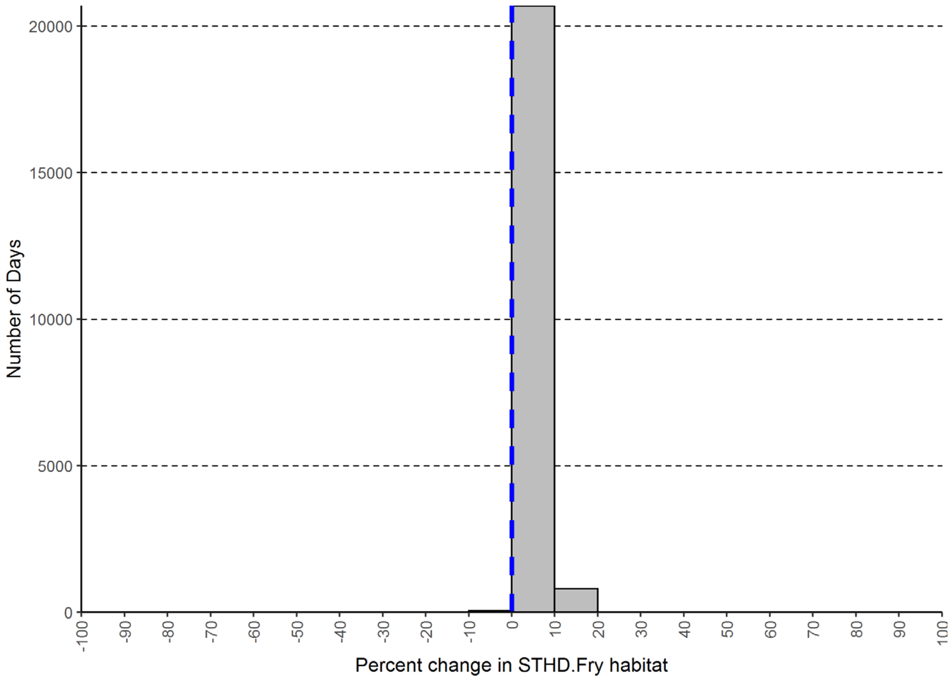

Figure A2-16. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead fry habitat at the Ukiah site.

A-2.1.2 Hopland Site

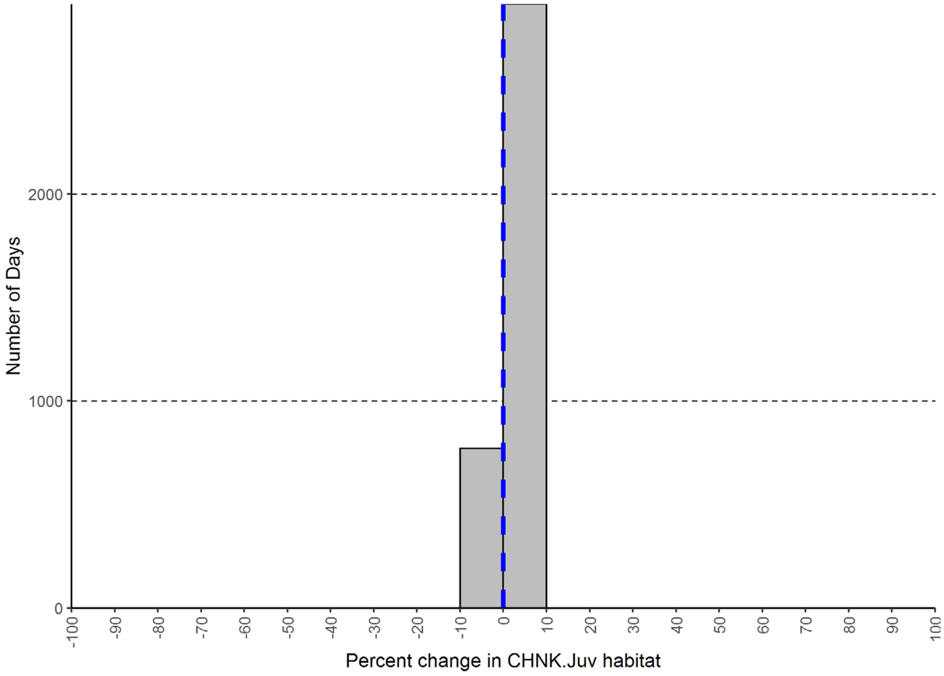

Figure A2-17. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for Chinook Salmon juvenile habitat at the Hopland site.

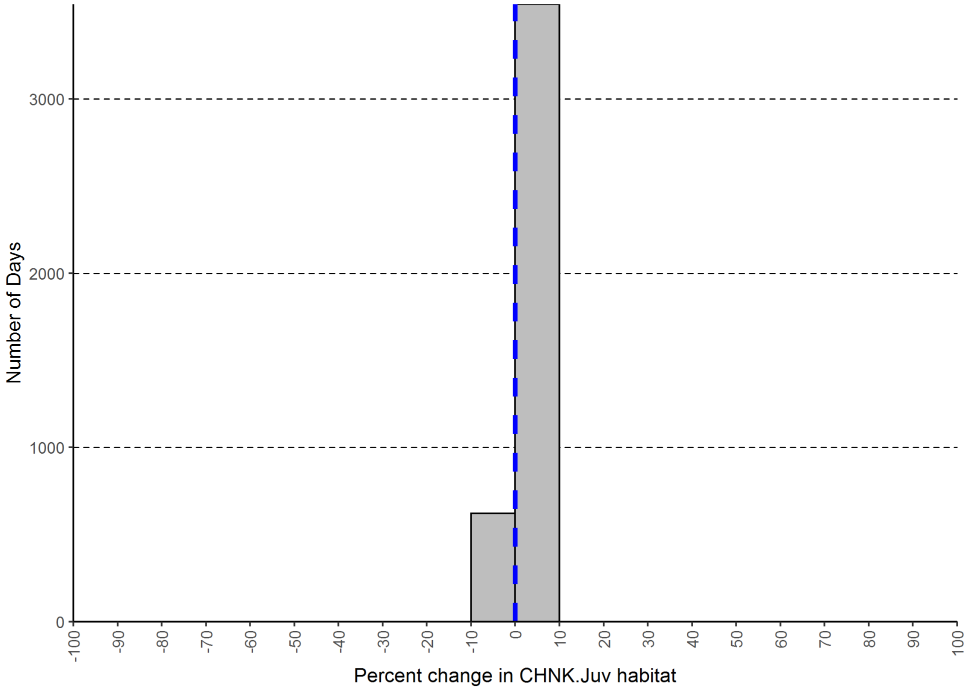

Figure A2-18. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for Chinook Salmon juvenile habitat at the Hopland site.

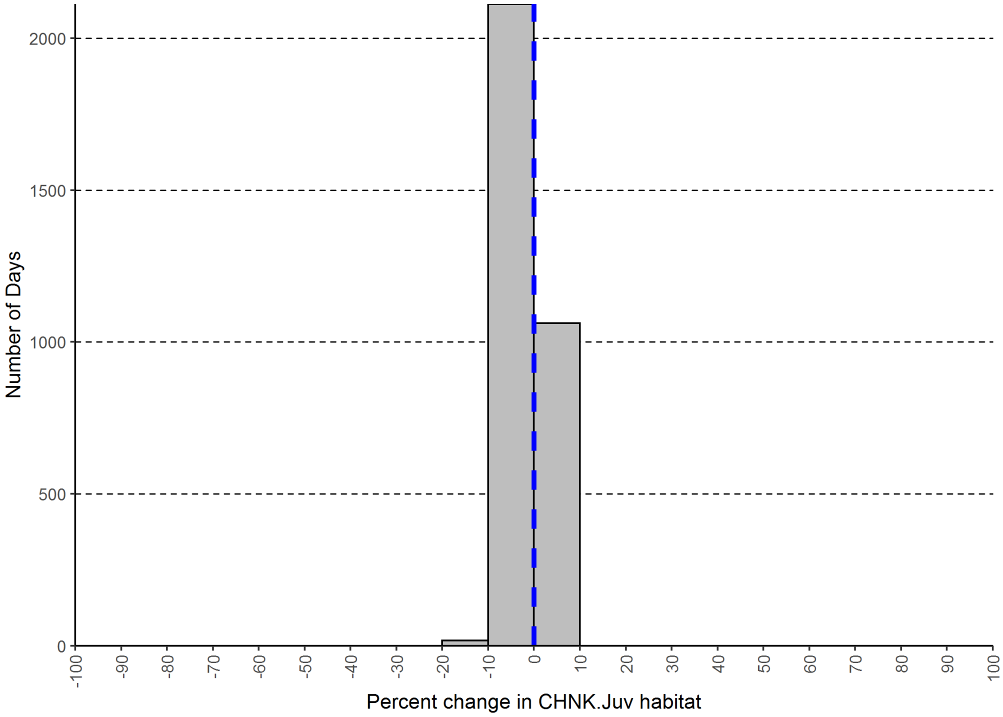

Figure A2-19. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for Chinook Salmon juvenile habitat at the Hopland site.

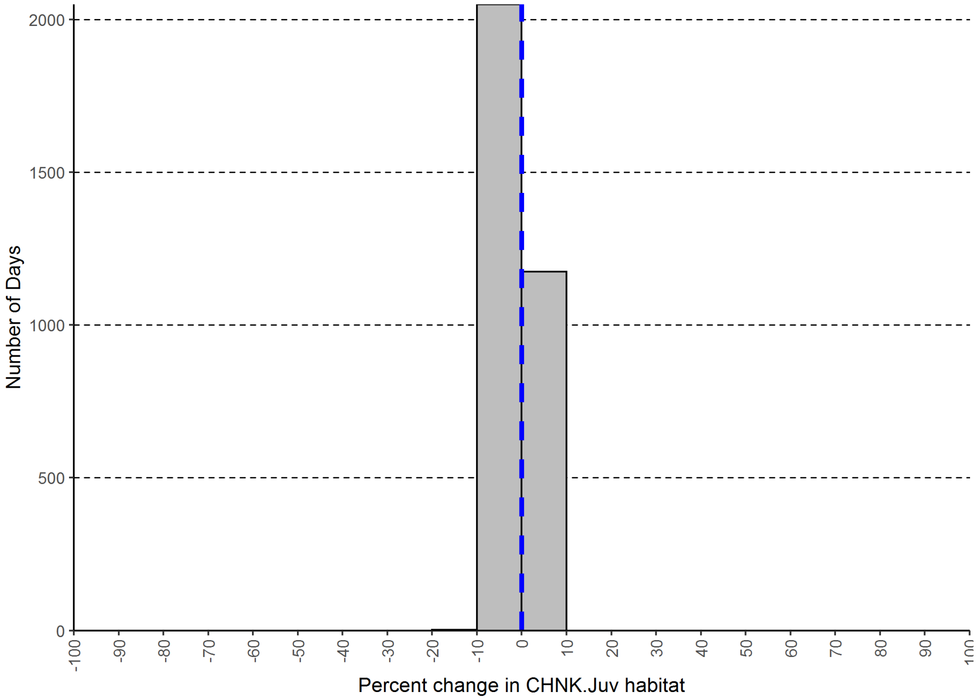

Figure A2-20. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for Chinook Salmon juvenile habitat at the Hopland site.

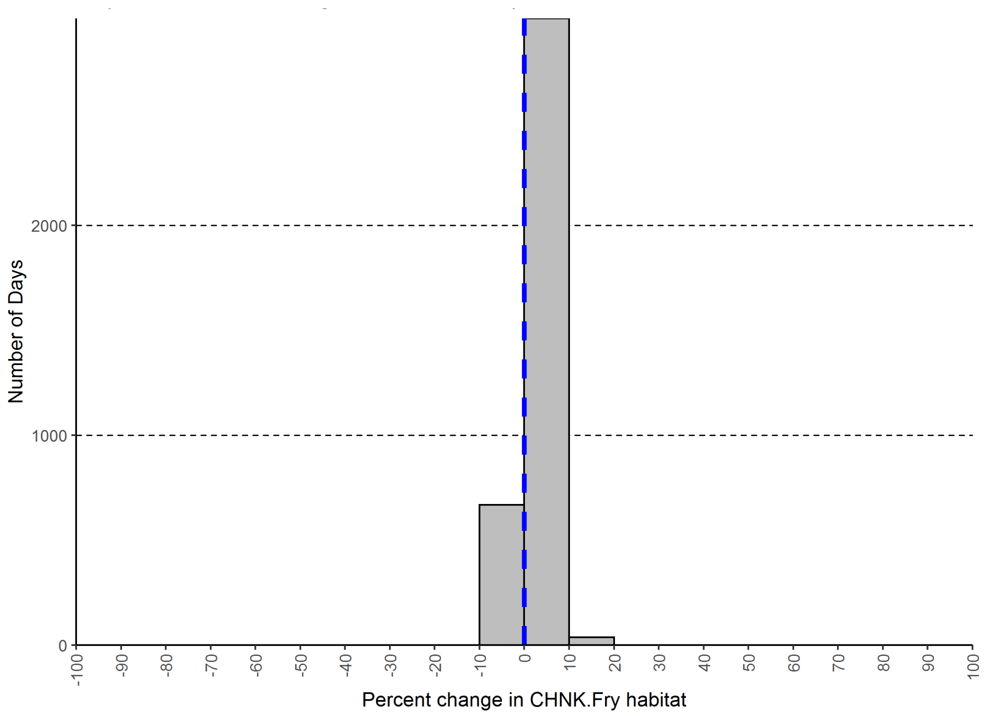

Figure A2-21. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for Chinook Salmon fry habitat at the Hopland site.

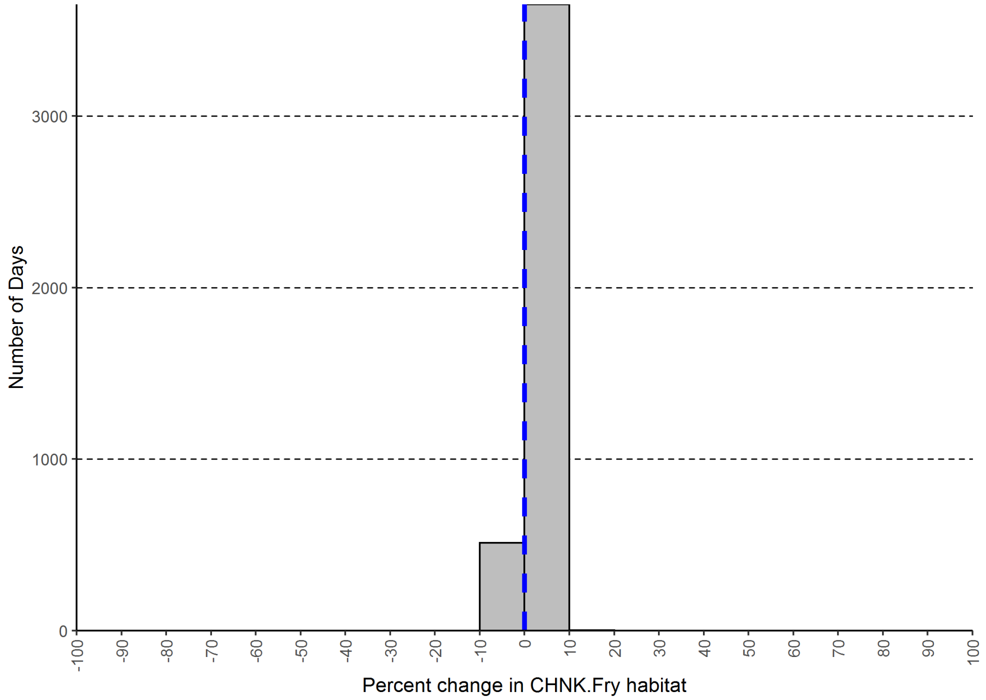

Figure A2-22. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for Chinook Salmon fry habitat at the Hopland site.

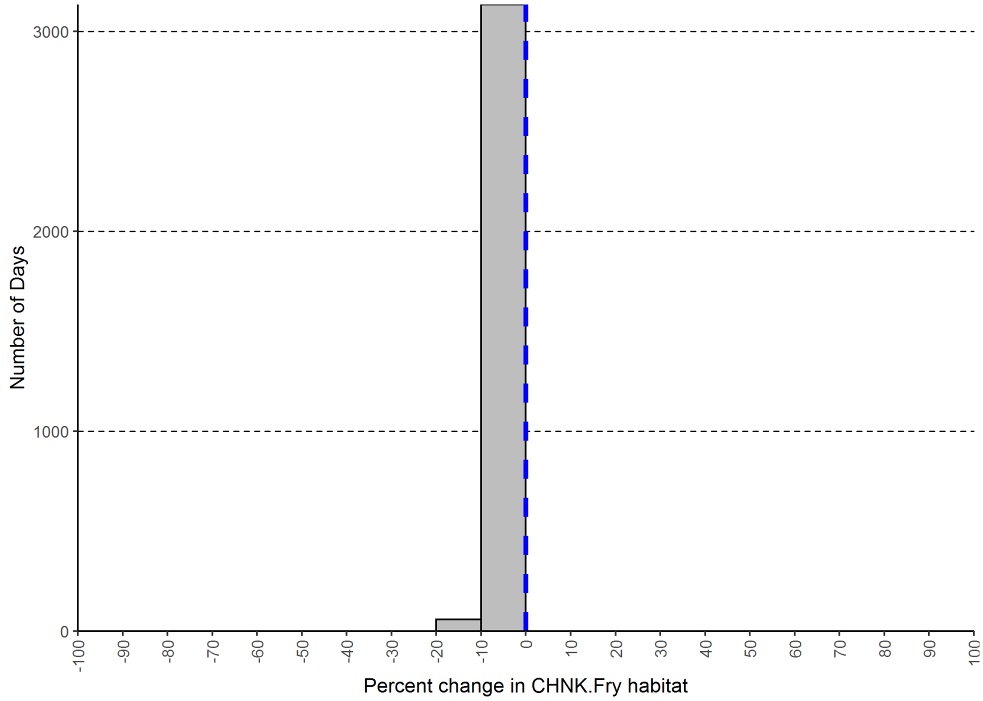

Figure A2-23. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for Chinook Salmon fry habitat at the Hopland site.

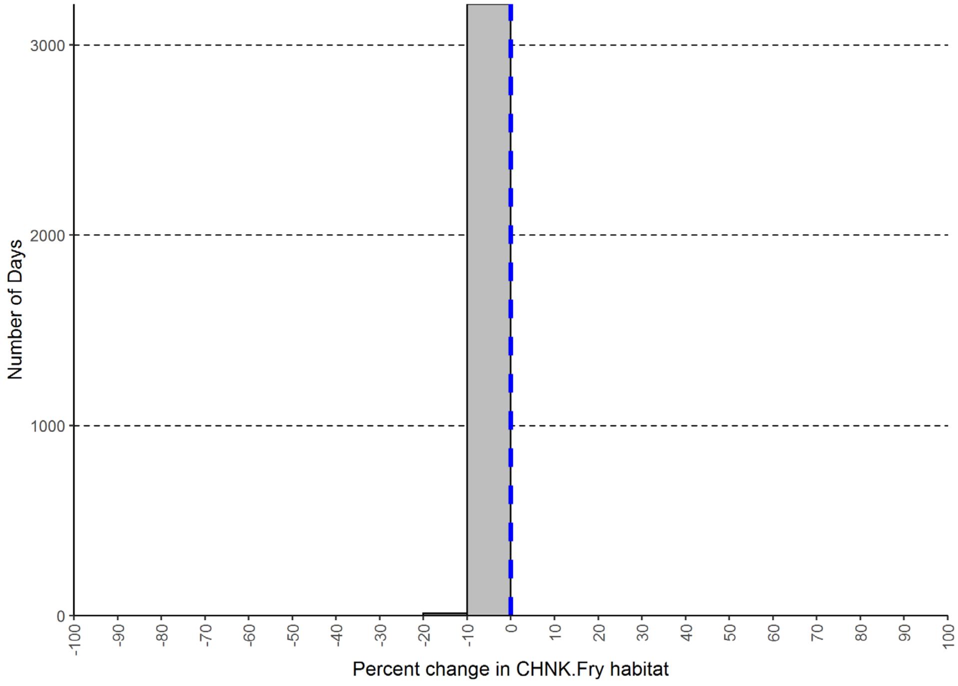

Figure A2-24. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for Chinook Salmon fry habitat at the Hopland site.

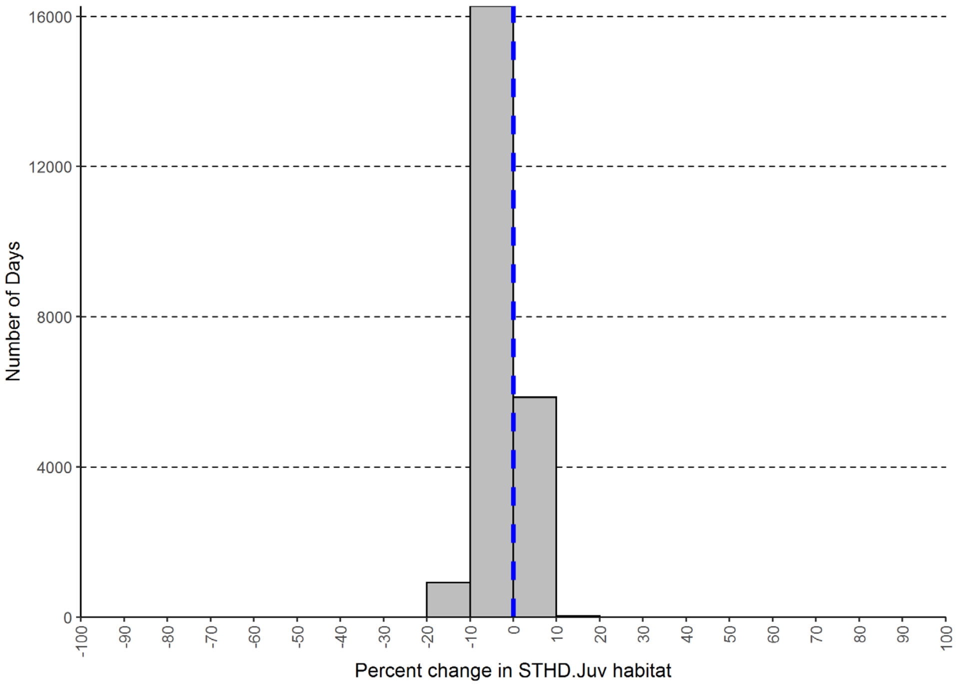

Figure A2-25. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead juvenile habitat at the Hopland site.

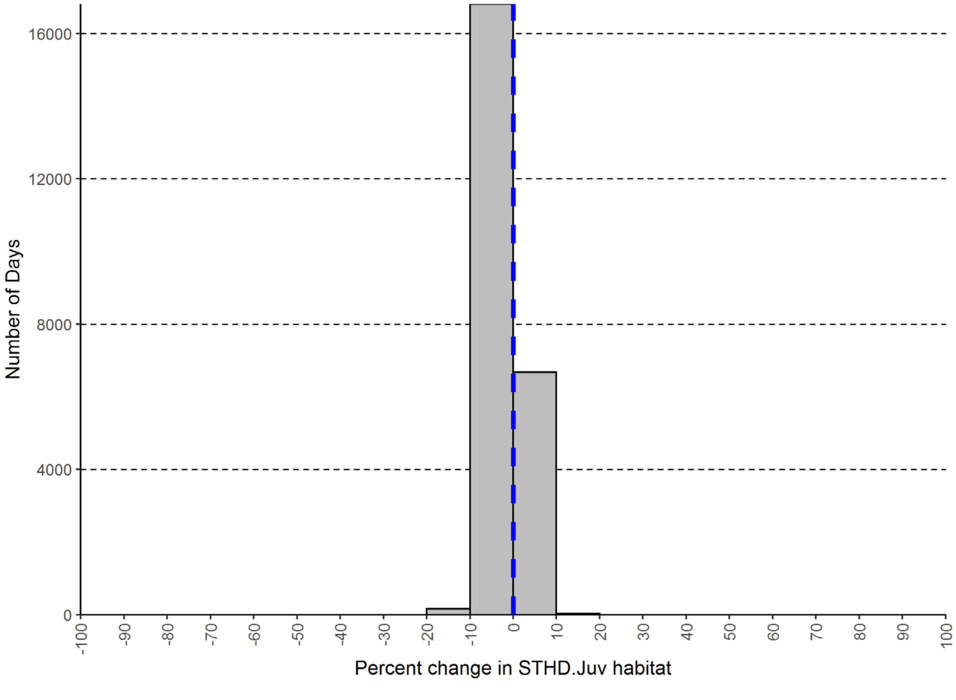

Figure A2-26. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead juvenile habitat at the Hopland site.

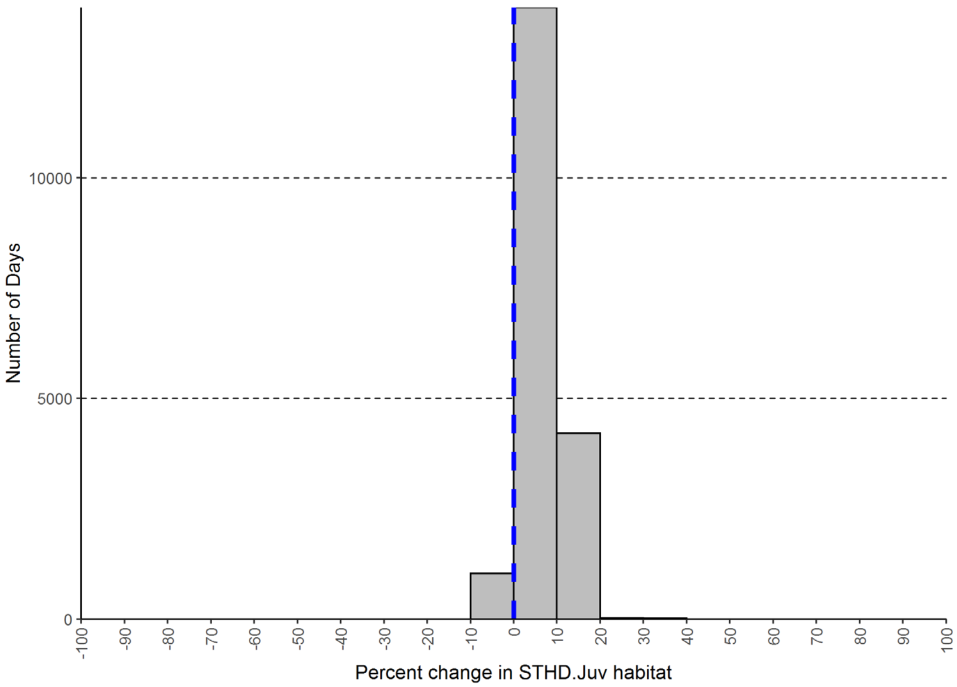

Figure A2-27. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for steelhead juvenile habitat at the Hopland site.

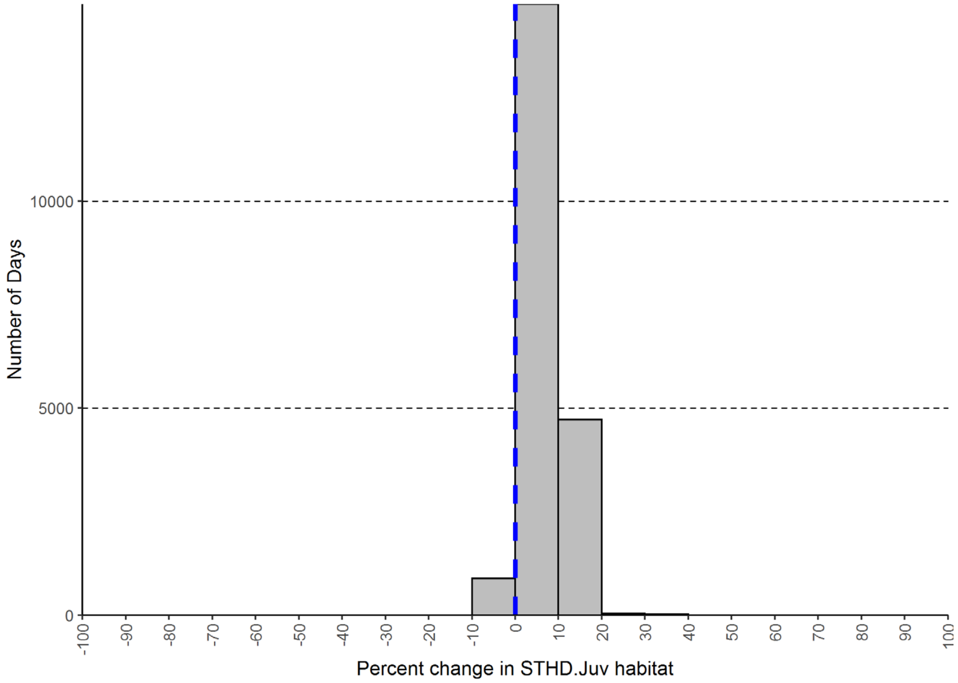

Figure A2-28. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead juvenile habitat at the Hopland site.

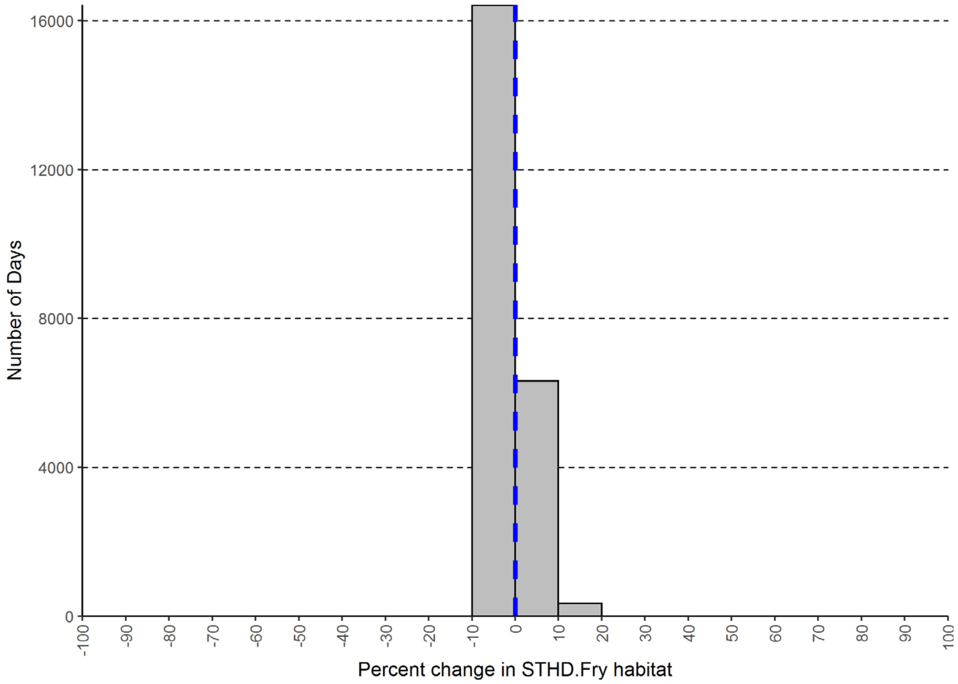

Figure A2-29. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 2 for steelhead fry habitat at the Hopland site.

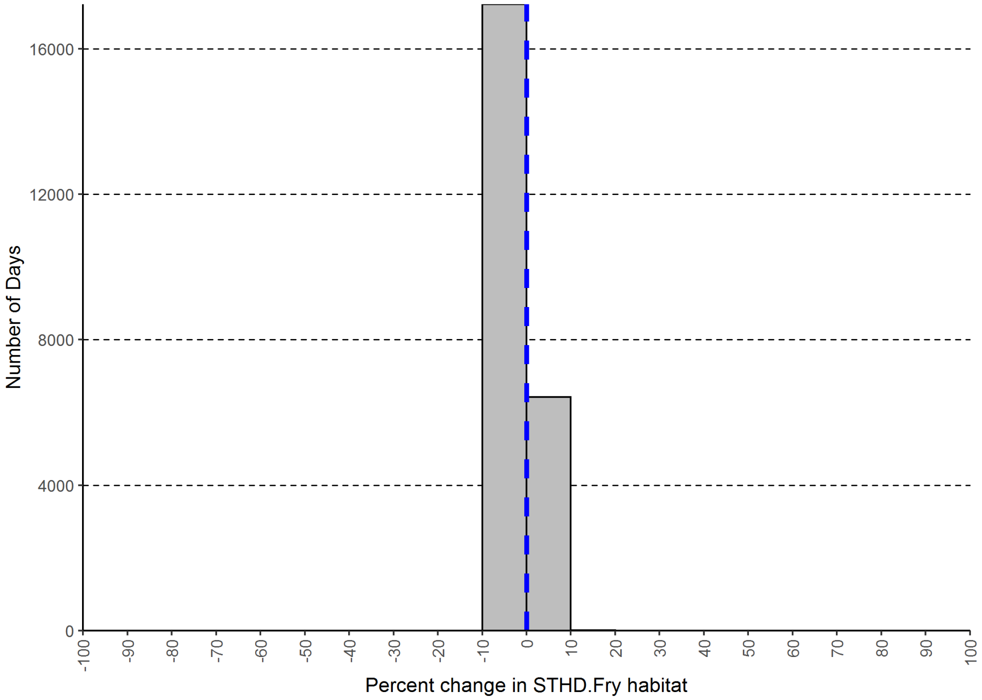

Figure A2-30. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Baseline to Scenario 4B for steelhead fry habitat at the Hopland site.

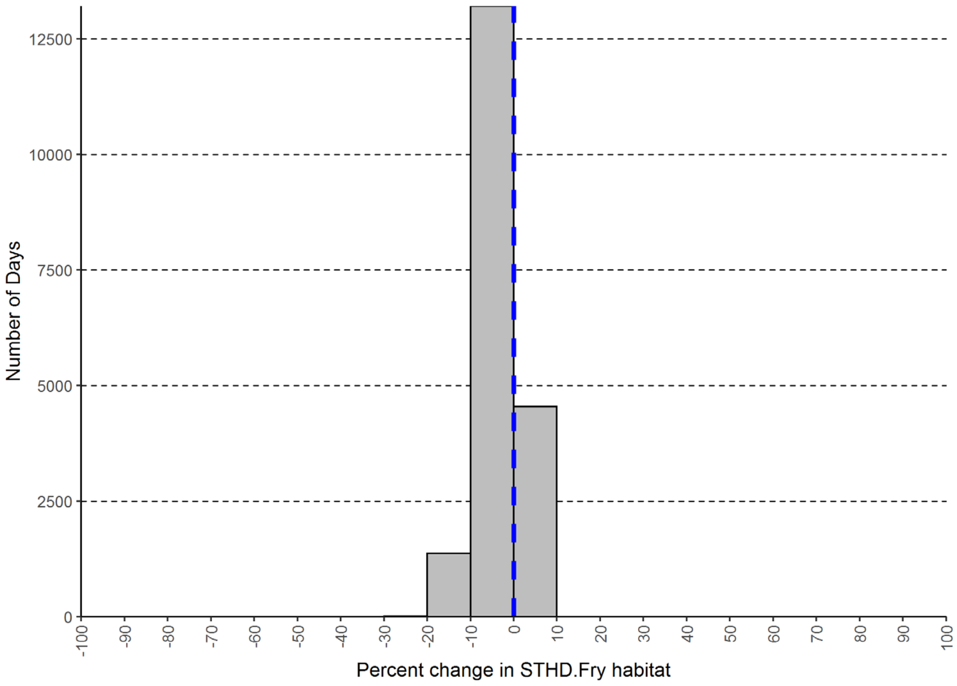

Figure A2-31. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 2 for steelhead fry habitat at the Hopland site.

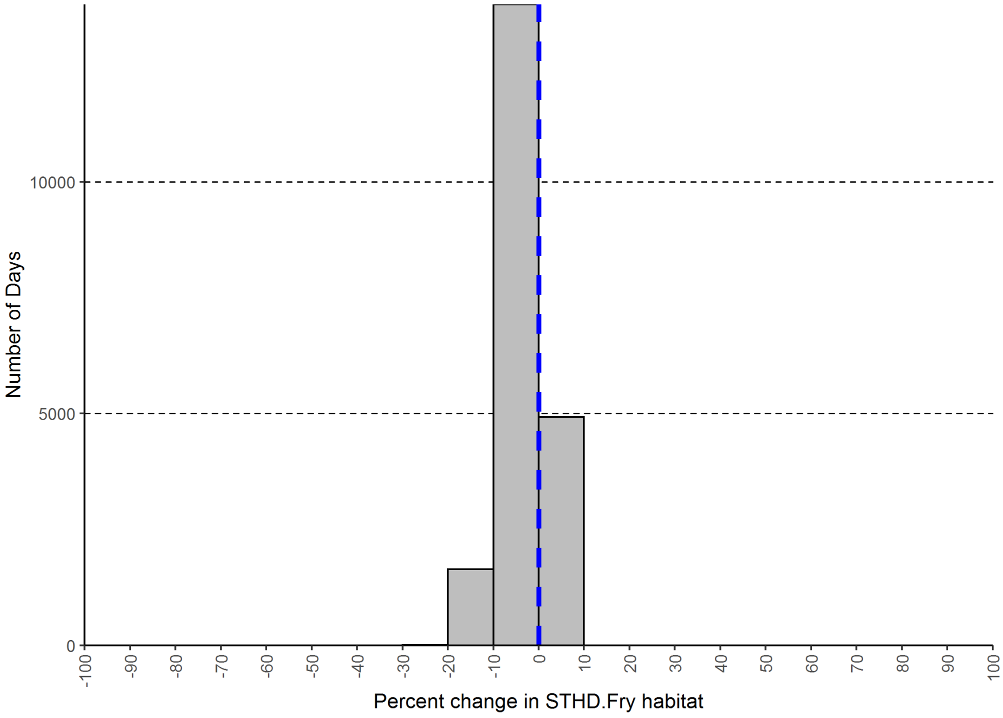

Figure A2-32. The number of days during water years 1911-2017 in which a change in habitat (percent change in weighted usable area) occurred when comparing Scenario 3 to Scenario 4B for steelhead fry habitat at the Hopland site.