US Army Corps of Engineers®
Portland District

WILLAMETTE VALLEY SYSTEM OPERATIONS AND MAINTENANCE

DRAFT PROGRAMMATIC ENVIRONMENTAL IMPACT STATEMENT

APPENDIX E: FISH AND AQUATIC HABITAT

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Supporting Information for Biological Input Parameters Used for Modeling of the Willamette Valley System EIS Downstream Fish Passage Measures in the Fish Benefit Workbook (FBW)

1.1 SPRING CHINOOK SALMON -

1.1.1 DETROIT & BIG CLIFF

Assumptions:

- Yearling stage begins in January
- Baseline includes spilling for temperature management, which is equivalent to the spring spill measure 714. It is assumed that these measures are identical.

a. No Action Alternative (NAA or Baseline) / Measure 714 (Use spillway to pass fish in the spring).

Run timing –

Schedules were developed separately for a) when reservoir fills sufficiently for surface spill (see Run timing **IF SPILL OCCURS**) and b) if no surface spill available (see Run timing **IF NO SPILL**) in a given year. This is based on the assumption that few fish would pass in the spring or summer in years when no surface spill is available under measure 714, and instead fish would pass in the fall via the turbines or RO as the reservoir is drafted. During the target spill period (June to October), most water years in the period of record fall into one of two categories: 75% of the days providing spill, or <30% of the days providing spill. The FBW will apply the spill run timing in years with 75% of the days providing spill, otherwise apply the non-spill year run timing for a given year in the period of record.

*Run timing **IF SPILL OCCURS** (reservoir fills above spillway crest for a portion of the run season):*

- Fry – applied Alden (2014) for baseline conditions. Assume fry distribute along reservoir shorelines upon entry in spring, and most become available to pass in June based on Monzyk et al (2010-2014) fry distribution data.
- Subyearlings - adjusted original Alden (2014) timing to reflect more spring passage. Assume most fry mature into subs stage and become more pelagic and widely distribute in reservoir in June. References in Hansen et al. 2017 (Khan et al. 2012, Romer et al. 2013, Beeman and Adams 2015) –indicate fish will use the spillway when it's operated.
- Yearlings – Adjusted original Alden (2014) timing. Yearlings have been shown to migrate quickly through reservoirs. The Alden (2014) timing (which used CGR as a surrogate) was adjusted with upstream trap data for DET (Romer et al. 2016). Assumed yearlings are

seeking to leave in winter and spring. Some yearlings will be available and pass with spill (Romer et al. 2013).

Run timing for ***IF NO SPILL.***

- Fry – Applied the Alden (2014) timing for fry.
- Subyearling - Applied the Alden (2014) run timing, which was also used in Detroit Configuration/Operation Plan 2.0 Reevaluation (USACE 2019).
- Yearling - Alden (2014) timing was adjusted with upstream trap data for DET (Romer et al. 2016). Alden (2014) used CGR screwtrap data as surrogate. Yearlings have been shown to migrate quickly through reservoir.

DPE (Dam Passage Efficiency) –

Applied USGS (Beeman et al. 2014b) data from Table 11, using averages of dam passage efficiencies from the spring and fall studies weighted by sample sizes. However, there are no studies of fish passage efficiency with Detroit reservoir drafted below 1450. The target elevation for measures 40 and 720 is 1375. Original proposed DPE values are currently 0.4 when the pool is between 1363 and 1424 ft and 0.27 when the pool is at 1341 to 1362. DPE values for Detroit Dam when the pool elevation is near the spillway crest and turbine penstocks is up to 0.77.

Table 1-1. Revised Dam Passage Efficiency inputs applied:

Pool Elevation	DPE	Note
1574	0.77	Max pool
1541	0.77	Spillway crest
1540	0.03	
1500	0.04	
1450	0.27	50' over top of penstock
1425	0.77	6' over top of penstock
1415	0.3	40' over top of RO
1375	0.77	25' over top of RO
1340	0.77	Upper RO

Note the DPE at elevation 1425 (6’ over the top of the penstocks) may be too high for Measures 40 and 720 considering that some adjustment may be needed to compensate for the fact that FBW is a daily model, yet the intent of the proposed operations when drafting below 50’ of depth over the penstocks is that turbines will only be operated during the daytime for 8 hrs.

Route effectiveness (RE)– Applied Alden (2014).

Alden rationale for their recommended RE values states "Data are based on Khan et al. 2012 and Beeman preliminary 2013. The values were set up such that at spill levels of greater than 30%, approximately 90 percent of the fish pass via the spillway. When the RO and Turbines (no spillway) is operating that analysis was based on Beaman wherein at a 70% turbine, 30% RO flow split; 88% of the fish passed the turbines 12% through the RO". The Alden RE estimates may be somewhat conservative for the spillway and RO. Beeman and Adams (2015) estimated spillway RE at 3.05 during the spring study period in 2013, when most fish passed at night over the spillway. The average spillway flow (552 cfs) to turbine flow (606 cfs) ratio was approximately 0.90 on during the night in this period. Turbine RE was estimated at 0.99 and regulating outlet RE was estimated at 1.62 during the fall study period, when most fish passed via the turbines. We did not revise inputs from the Alden 2014 recommendations however due to the lack of readily available information to estimate RE for different flow ratios using the Beeman and Adams results.

Route survival –

For turbines, Beeman and Adams (2015) estimated survival from the forebay Detroit Dam to Big Cliff forebay at 62.2% in the fall of 2013 when 120 of 122 fish that passed used the turbines. Turbine flows were generally greater than 1000 cfs. Therefore, a survival rate of 62.2% was applied for turbine passage at flows of 1000cfs for all life stages. Applied Alden (2014) for flows <1000cfs, which was based on Normandeau (2010) and utilized rainbow trout as a surrogate for subs/yearlings.

For regulating outlets (ROs), Applied Alden (2014) survival rates, which were based on Normandeau (2010) and utilized rainbow trout as a surrogate for subs/yearlings.

For spill, the high range of the Alden (2014) estimates was used. Normandeau (2010) data indicated higher survival. Survival estimates by Beeman and Adams (2015) was also considered. They modeled survival from the forebay Detroit Dam to Big Cliff forebay as 71.6% based on detections of acoustic tagged juvenile Chinook. However did not account for route of passage. Most of the fish passage events detected occurred during the period when surface spill was occurring and those fish with known routes of passage nearly all used the spillway.

Re-regulation mortality, applied the same value as used by Corps (2015) of 15%. Beeman and Adams (2015) estimated juvenile Chinook survival from Detroit Dam tailrace downstream to Minto Dam as 0.67 to 0.74, or inversely a mortality of 0.26 to 0.33. We assume this estimate includes mortality occurring below Big Cliff Reservoir. Fischer et al. 2019 estimated mortality through Dexter Reservoir (which reregulates flows below Lookout Point Dam), at about 2%. Big Cliff Resevoir is smaller than Dexter. Oligher and Donaldson (1966) conducted Big Cliff Kaplan turbine unit tests to determine what effect various operating conditions would have on survival of fish passing through this type of turbine. Average survival from all tests in Oct. 1964 was 91.1 percent at 91 ft. head, 94.5 percent at 81 ft. head, and 89.7 percent at 71 ft. head. Average survival from all tests in May 1966 was 92.2 percent at 91 ft. head, 89.8 percent at 81 ft. head, and 90.6 percent at 71 ft. head. Therefore, we expect the 26%-33% mortality rate range is likely high since it also includes mortality occurring below Big Cliff. Therefore, we applied 15% reregulation mortality, as used previously in USACE (2015).

b. Measure 392+105: FSS with SWS –

Flow range determined in the Detroit Design Documentation Report (DDR) for the Floating Screen Structure (FSS) is 1,000 – 5,600 CFS, with all flow to the Selective Withdrawal Structure (SWS) going through FSS to avoid competing flow. Above 5,600 through the FSS we are not in NMFS fry criteria anymore and would want lower survival for fry → here we assume that above 5,600, water would be drawn in from a low-level inlet and assume no fish in that part of the water column.

Run timing -

- Fry - Applied the Alden (2014) timing for a floating structure.

- Subyearlings – Adjusted the Alden (2014) baseline timing with downstream passage from the Willamette Project Configuration/Operations Plan (USACE 2015, p 48, Appendix K). Assumed some fry would mature to subyearling stage in spring and be available to pass. Data indicates growth rates can be high in DET Reservoir; Breitenbush tributary data indicate by May-June fish would have grown >60 mm (Monzyk et al. 2015). Adjusted subyearling timing accordingly.

- Yearlings – same as baseline

Dam Passage Efficiency - above minimum conservation pool–

DPE within the pool elevation operating range of the FSS was estimated separately for each alternative. The method and results are described in Attachment A of this Chapter.

Table 1-2. Dam Passage Efficiency Values by Alternative:

Alternative	DPE within the FSS pool elevation operating range
1	0.569
2	*TBD – pending finalization of alternative and RES-SIM results*
3a and 3b	Not applicable
4	*TBD – pending finalization of alternative and RES-SIM results*

Dam Passage Efficiency, below minimum conservation pool - applied DPE values from Detroit (DET) baseline

Route Effectiveness – Applied Alden (2014). Assumes no surface spill and all flow through the FSS.

Route survival – 98% for all life stages for the fish passage route (FSS). Other routes same as baseline. The FSS is assumed to have a passage survival of 98% for all target species collected, based on structures operating in the Northwest similar to the FSS concepts being considered for the WVS EIS (see USACE 2015 section 2.5.5).

c. **Measure 40 – Deep fall drawdown to 10ft over the top of the upper RO's – Target start date 15 Nov and maintained for three weeks.**

Run timing *- same as baseline.*

Dam Passage Efficiency – same as baseline.

Route effectiveness – same as baseline.

Route survival – same as baseline.

d. **Measure 720: Spring delay refill with target elevation at 10' over the top of the upper RO's. May 1 to May 21 at target elevation.**

Run timing –

- Fry – Same as Detroit (DET) FSS (measure 392)
- Subyearlings – Same as DET FSS (measure 392)
- Yearlings – Same as baseline

Dam Passage Efficiency – *Same as baseline*

Route Effectiveness *– Same as baseline*

Route Survival *– Same as baseline*

1.1.2 FOSTER

Assumptions

- Yearling stage begins in January
- Baseline includes spilling for temperature management, which is equivalent to the spring spill measure 714. It is assumed that these measures are identical.

a. **Baseline**

Run timing –

Same as used in the Foster Downstream Fish Passage EDR (2016). Alden (2014) recommendation was based on fry data from Monzyk (2012) and for subyearling and yearling data from Wagner and Ingram (1973). Adjustments to Alden timing made considered data presented by Monzyk and Romer (2013 and 2014) above and below resevoir screwtrapping. We assume subs (>60 mm) are from those that entered the reservoir as fry, grew, and then move further from shore in May- June then emigrate.

Dam Passage Efficiency –

Applied data from Liss et al. (2020). Also see Alden (2014). Fry and sub-yearlings. Liss et al. did not include data for fry; assumed same for fry. Values at different elevations given the presence of a weir were taken from Liss et al. (2020) for the weir (SPE), low pool (min con), and the turbines. Liss et al. assumed low pool conditions when sub-yearlings pass. Therefore, we used the average DPE observed over 3 years.

Turbine passage was averaged from observations of passage from Liss et al. (2020) over low pool conditions (ie, calculated using FPE, Fish Passage Proportion). DPE was available for yearlings under high and low pool conditions. Therefore, DPE was taken to be the midpoint between low and high DPE values over 3 years and two pool elevations for yearlings using PNNL 2020.

Route Effectiveness – Applied Alden (2014)

Route survival –

Applied averages of estimated survival for subs (CK0) and yearlings (CK1) for each route from Liss et al. (2020). Low and high pool survival estimates were available for yearling Chinook, and so the average across both pool elevations was applied.

b. Measure 392

Run timing - Same as baseline.

Dam Passage Efficiency –

Measure 392 for Foster Dam is a concept of either further improving the fish weir operated in Spillbay 4 or constructing a dedicated fish collection and bypass pipe in the same vicinity as the fish weir, with either concept operating at about 600 cfs. Until further refinement of this concept, we assumed a DPE consistent with the highest DPE measured at the dam for steelhead to date of 0.76 as reported in Table 5.6 of Liss et al. (2020).

Route Effectiveness – Applied Alden (2014)

Route survival –

For spillway and turbines, used same values as for baseline. For fish passage route, assumed 98%, where fish passage concept is either a modified overflow weir or a dedicated fish pipe (see USACE 2015 section 2.5.5).

1.1.3 GREEN PETER

a. Baseline

- Not applicable – no fish outplanted above dam.

b. Measure 392: GPR FSS –

Run timing – same as DET timing for Measure 392.

Dam Passage Efficiency –

DPE within the pool elevation operating range of the FSS was estimated separately for each alternative. The method and results are described in Appendix A of this document. DPE values by Alternative when above minimum conservation pool:

Table 1-3. Dam Passage Efficiency by Alternative within the FSS.

Alternative	DPE within the FSS pool elevation operating range
1	0.544
2	TBD – pending finalization of alternative and RES-SIM results
3a and 3b	Not applicable
4	TBD – pending finalization of alternative and RES-SIM results

Below minimum conservation pool elevation, applied DPE values from baseline adjusted on depths to outlets for GPR.

Route effectiveness –

Applied DET RE values due to similarity in dam configuration. Local data on RE for existing routes at GPR not available.

Route survival –

98% for fish passage route (see USACE 2015, section 2.5.5). Spillway, turbines and RO assumed the same as DET due to similar dam configuration.

c. Measure 714 and 721: Spring/summer spill

Run timing – Applied DET baseline timing for years with and without spill.

Dam Passage Efficiency –

Data is not available for DPE of juvenile Chinook at Green Peter Dam. Applied DPE values from DET to GPR based on DPEs for similar depths to outlets at GPR. Assumed highest DPE when pool surface elevation $\leq$ depth over top of outlet.

Route effectiveness –

Applied DET RE values due to similarity in dam configuration. Local data on RE for existing routes at GPR not available.

Route survival –

Applied route survival from DET due to similarity in dam configuration. No site-specific data on juvenile downstream passage survival for spillway, turbines and ROs.

d. Measure 40 (deep fall drawdown)
   - Same as 714 and 721)

e. Measure 720 (spring delay refill)
   - Same as 714 and 721

1.1.4 COUGAR

Assumptions:

   - Yearling stage begins in January

a. Baseline

Run timing

- Fry – Applied Alden (2014)
- Subyearlings – Applied Alden (2014)
- Yearlings – Applied Alden (2014). Also see CGR 2.0 DDR, Romer et al. 2013 and Hansen et al. 2017.

Dam Passage Efficiency –

Applied DPE as used in CGR 2.0 DDR (USACE, 2020). DPE estimates developed based on passage rates reported in Beeman et al. 2013 and 2014. For diversion tunnel DPE, RO passage rates reported by Beeman et al. were applied for the diversion tunnel based on similar depths to the outlet except when very near or below the top of the diversion tunnel, in which case estimated DPE was based on passage rates observed by Nesbit et al. (2014) for Fall Creek Dam outlet works at low pool elevations. After modeling with initial assumptions, DPE input values were further reviewed to adjust assumptions to better reflect field data and the new operational scenarios included in the EIS (M40 and 720). Due to lack of data on Chinook passage when the pool elevation is very near the top of the RO, information on juvenile Chinook passage from Fall Creek Reservoir was applied considering that both outlets are located in close proximity to the bottom of the pool.

Table 1-4. Dam Passage Efficiency Values Applied by Elevation.

Pool elevation	Previous DPE	DPE	Revised 9/23 DPE
1690	0.1	0.135	0.135
1635			0.2
1571	0.2	0.2	0.3
1570	0.42	0.16	0.5
1532	0.42	0.33	0.6
1516	0.6	0.6	0.75
1500	0.7	0.7	0.8
1450	0.1	0.1	
1425	0.299	0.299	0.299
1400	0.5	0.5	0.5
1360	0.6	0.6	0.6
1337	0.7	0.7	0.7
1321	0.8	0.8	0.8
1310	0.95	0.95	0.95
1290	0.95	0.95	0.95

Route Effectiveness –

Applied Alden (2014). These values were derived from Beeman et al. (2013 and 2014a) data. The overall value from 2011 and 2012 were averaged to obtain RO effectiveness value of 91.45%. The estimate applies for flows ranging from 48% to 73%, as this was the range of flows the data was collected over. Values for flows above and below the range were shaped based on professional opinion. The use of professional opinion should have little effect as the project should operate within the published ranges very often. [*NOTE: Below 1571, the RO bypass gate is opened. Effectiveness in this case should be equivalent to the best Surface Flow Outlets, ~6.0 (ENSR 2007, Johnson et al. 2009.)*

Route Survival –

Fry: Applied Alden (2014).

Subs and yearlings: Adjusted USACE 2015 (see Appendix K) values down to 36% based upon the Beeman (2012) radio-telemetry work. 60% seems very high based on all available data, while Alden's 29% seems very low. CGR EDR explains why COP HI-Z tag data is likely estimated high due to premature inflation of tags, and that barotrauma sheer stress was high, and why that value should be adjusted downward. CGR EDR: "This, coupled with modeling of the chance of turbine strike at different fork lengths, indicate that the chances of yearling Chinook surviving turbine passage at Cougar Dam are certainly less than 50% and likely in the 30-40% range (Duncan 2010a, Carlson 2010).” Used 30% as low and 40% as high estimate bracket.

b. Measure 392: CGR FSS –

Run timing -

- Fry – Applied Alden (2014)
- Subyearlings – Same as DET FSS timing for subyearlings.
- Yearlings – Revised from Alden (2014) in consideration of Romer et al. (2013-2016) above-reservoir screw trap data for CGR.

Dam Passage Efficiency –

DPE within the pool elevation operating range of the FSS was estimated separately for each alternative (see Appendix A).

Table 1-5. Dam Passage Efficiency values by Alternative for measure 392.

Alternative	DPE within the FSS pool elevation operating range
1	Not applicable
2	Not applicable
3a and 3b	Not applicable
4	*0.864*

Below the operating elevation range of the FSS (minimum conservation pool) - applied DPE values as used in the baseline.

Route Effectiveness –

Applied Alden (2014). Assumes no surface spill and all flow through the FSS when pool between min and max conservation elevations.

Route survival –

Fish passage route 98% for all life stages (see USACE 2015 section 2.5.5). Same as baseline for other routes.

c. Measure 40: Deeper fall drawdowns to 10 ft over top of upper RO’s AND to diversion tunnel (1290’) – target start 15 Nov for three weeks. Assumes RO structural improvements for fish passage survival.

Run timing –

Fry – Same as baseline

Subyearlings – Same as baseline

Yearlings – Same as baseline

Dam Passage Efficiency – Same as baseline

Route Effectiveness – Same as baseline

Route Survival –

Used Nesbit (2014) survival data for diversion tunnel, and Alden (2014) parameter estimates for other routes.

d. Measure 720: Delay refill with pool held at 10 ft above top of upper RO's – target May 1 to May 21 at target elevation.

Run timing –

- Fry – used Cougar head of reservoir data from Monzyk et al. (2011) and Romer et al. 2012-2016.
- Subyearlings – Same as DET FSS timing for subyearlings.
- Yearlings – Run timing revised from Alden (2014) in consideration of Romer et al. (2013-2016) above-reservoir screw trap data for CGR.

Dam Passage Efficiency – Same as baseline

Route Effectiveness – Same as baseline

Route Survival – Same as baseline

e. Measure 720: Spring drawdown to diversion tunnel (1290') target May 1 to May 21 at target elevation.

Run timing –

- Fry – used Cougar head of reservoir data from Monzyk et al. 2011, and Romer et al. 2012-2016. Notes: Most fry emigrate into CGR Reservoir during April and May. RES-SIM models of a 1290 delay refill indicates the reservoir elevation will be much higher than 1290 during these months in several years. Fry will therefore distribute along the reservoir shoreline (Monzyk et al. 2011-2015), and then many will pass once the reservoir is less than about 20 feet over the diversion tunnel.
- Subyearlings – Same as DET FSS timing for subyearlings. Notes: Fry mature into the parr stage and become pelagic in June (Monzyk et al. 2011-2015). We expect some will pass

when the reservoir is within 50ft of depth over the DT, and most will pass once the reservoir is within 25 of the top DT, based on radio-telemetry study at Fall Creek Dam (Nesbit et al. 2014).

- Yearlings – Run timing revised from Alden (2014) in consideration of Monzyk et al. 2011 and Romer et al. (2012-2016) above-reservoir screw trap data for CGR.

Dam Passage Efficiency – Same as baseline

Route Effectiveness – Same as baseline

Route Survival – Same as baseline

1.1.5 HILLS CREEK

Assumptions:

- The spillway will not be used under the NAA and Measure 392.
- Measures 714 and 479 assume spillway modified to improve fish survival and feasibility for long-term use.
- Yearling stage begins in January.

f. Baseline

Run timing -

- Fry – Applied Alden (2014) for CGR baseline run timing
- Subyearlings – Applied Alden (2014) for CGR baseline run timing
- Yearlings – Revised run timing applied in the COP for HCR (USACE 2015, Appendix K) based on the assumption that the yearling stage begins in January.

Dam Passage Efficiency –Applied DPE from CGR for similar depths to outlets using data from Beeman et al. (2013; see Table 9). Assumes no surface spill is occurring since the spillway at HCR is not used (i.e. designed only for emergency use).

Route Effectiveness – Same as CGR for each route, due to similarity in dam configuration.

Route Survival – Used Alden 2014 (based on CGR RO survival estimates). Assumes no surface spill. Alden estimates could be high, considering RO configuration at HCR would be expected to result in higher injury and mortality. Life cycle model sensitivity analysis will further assess the parameters estimates and influence on the model results.

g. Measure 714 –

Use a modified spillway to pass fish in the spring –From May 1 until July 1 (or as long as hydrology supports during the conservation season), operate the spillway 24 hrs/day as the primary outlet, with turbines and ROs as secondary. This measure assumes structural modifications to the spillway to make it feasible to operate, and safer for fish to pass over.

Run timing -

- Fry – Same as baseline
- Subyearlings – Used similar approach as for DET, measure 714: If 'no spill': same as HCR baseline. If spill: used DET spill timing for baseline/measure 714.
- Yearlings – Same as HCR baseline

Dam Passage Efficiency –

Updated baseline DPE estimates to include operation of a modified spillway. Adjusted DET DPE down for above spillway crest at high pool due to the fact that at HCR the max pool is higher above crest than DET max pool over the DET spillway crest (i.e fish must sound to greater depths when at HCR max pool).

Route Effectiveness –

Spillway same as DET since this measure assumes modifications to the spillway. Other routes same as CGR for each route, due to similarity in dam configuration.

Route Survival -

Spillway – Assumed spillway will be newly designed with fish survival in mind; anticipate slightly higher survival than DET. Used the high end of the DET range, as reported for sensor fish/balloon tag data (Normandeau, 2010); 48 hr survival was 64 – 84% at different gate openings. [Data also reported in Hansen et al. (2017) data synthesis.]

RO and turbines – Utilized Alden (2014)

h. Measure 479: Modify Existing Outlets –

Re-design spillway gates and channel to allow for low-flow releases when lake is above spillway crest. This would provide more normative temperatures during the summer through the release of warmer water during the summer and saving cooler deeper water for the fall. Won't change total flow, but less hydropower. Hit 1495 by Feb 26 on current rule curve.

Run Timing, DPE, RE, and Route Survival - same as for measure 714 (spring spill).

i. Measure 392: Floating screen structure

Run timing – same as for DET Measure 392

Dam Passage Efficiency –

Fish passage within the FSS – DPE within the pool elevation operating range of the FSS was estimated separately for each alternative. The method and results are described in Appendix A of this document.

Table 1-6. Hills Creek DPE values by Alternative.

Alternative	DPE within the FSS pool elevation operating range
1	Not applicable
2	Not applicable
3a and 3b	Not applicable
4	0.791

Below minimum conservation pool - applied DPE values from baseline

Route Effectiveness - RE for FSS from CGR Measure 392, other routes same as baseline

Route Survival - FSS 98% for all life stages, other routes same as baseline.

j. Measure 304: Augment flows by tapping the power pool

Run Timing, DPE, RE, and Route Survival - same as HCR Baseline.

k. Measure 40: Deep fall drawdown to 10 ft above the top of the RO by NOV15 –

Target start date 15 Nov and maintained for three weeks. Assumed not to affect run timing of yearlings.

Run timing -

- Fry – same as Baseline.
- Subyearlings – same as DET baseline ‘no spill” timing, which has peak passage in Nov. when reservoir elevation low.
- Yearlings – same as HCR Baseline. This measure would end before Jan.

Dam Passage Efficiency, RE, and Route Survival - same as HCR Baseline.

l. Measure 720: Delay refill to 10 ft above the top of the RO May 1 to May 21

Run timing -

- Fry – same as baseline.
- Subyearlings – same as DET Measure 392.
- Yearlings – same as DET Measure 392.
- Dam Passage Efficiency, RE, and Route Survival - same as HCR Baseline.

1.1.6 LOOKOUT POINT & DEXTER

Assumptions:

- Yearling stage begins in January.

m. Baseline

Run timing - Same as DET baseline, all lifestages:

Dam Passage Efficiency –

Based on DPE values used for DET, adjusted for outlet elevations at Lookout Point (LOP). Also considered Fischer et al. (2019) estimated DPE was 31% for October released fish and 58% for December-released fish, when forebay surface elevations in October were about 850ft, and ranged from 822 to 837 ft in December.

Table 1-7. Revised DPEs inputs applied

Pool elevation	DPE	Note
934	0.77	Max pool
926	0.77	
887.5	0.77	Spillway crest
887	0.10	
825	0.58	Min cons.
819	0.58	Min power
780	0.30	Below power pool; 44' over top of RO
761	0.77	25' over top of RO
724	0.77	RO invert

Route Effectiveness – Applied Alden (2014)

Route Survival –

RO survival rates assumed are the same as for DET baseline, all lifestages, since no data is available for LOP RO survival. For turbines at lower flows, also used DET data since recent PNNL acoustic telemetry studies estimated survival only for moderate to high flows levels (Fischer et al. 2019). For higher flows, used Fischer et al. (2019), who estimated survival of turbine-passed fish to the Lookout Point tailwaters at 77.9% (SE = 3.9) for October released fish (n = 134) and 82.3% (SE = 3.4) for December-released fish (n = 331). Survival of turbine-passed fish (n = 83) to the Lookout Point tailrace was 78.4% (SE = 4.7) for February-released fish. For spillway survival, also used Fischer et al. (2019), who estimated survival of pooled February and April-released fish passing via Spill Bay 3 on April 29, 2018 (n = 66) was 98.7% (SE = 5.5).

Reregulation Reservoir and Dam Passage Mortality for Dexter- for all life stages, applied 26%. Fischer et al. (2019) estimated survival of Chinook subs and yearlings, from the Lookout Point tailwaters to Dexter Dam forebay ranged from 88.5% (SE=4.3) to 93.0% (SE = 6.8) to 88.5% (SE=4.3) among the study release groups. Survival for fish passing Dexter Dam was not estimated. For fish released in October and December, the joint probability of migration and survival from Lookout Point tailrace to the Corvallis array was 0.435 and 0.443, respectively. However, since this estimate includes survival within a significant river reach downstream of Dexter Dam, we considered passage survival data from Big Cliff Dam (the reregulation dam below Detroit Dam which also has Kaplan turbines). Beeman and Adams (2015) estimated juvenile Chinook survival from Detroit Dam tailrace downstream to Minto Dam as 0.67 to 0.74. Considering the Beeman and Adams mortality estimate would be somewhat lower if it was for just Big Cliff Dam, and the very low mortality estimated in Dexter Reservoir by Fischer (2019), we applied a re-regulation mortality estimate of 26%.

PNNL survival estimate summary (Tables from Fischer et al. 2019)

Table 4.2. Sample Sizes (N) and Estimated ViRDCt Survival Probabilities ($\hat{S}$) from Lookout Point to the Lookout Point Immediate Tailrace Array (LPT array) and to Dexter for Acoustic-Tagged CH0 Released into the Lookout Point Reservoir in October and December 2017. Detection probabilities (p) of each detection array (LPT and Dexter) are also shown. Virtual release groups (V_1) were formed by release month and route of passage at Lookout Point. Standard errors (SEs) of survival estimates are shown in parentheses. All detection probability SEs were ≤0.01. Superscripts indicate the model that was used to estimate survival.

		Lookout Point to Immediate Tailrace		Lookout Point to Dexter	
V_1 group	N	$\hat{S}$ (SE)	p	$\hat{S}$ (SE)	p
Oct turbines	134	0.779 (0.039)[a]	0.99	0.724 (0.039)[b]	1.00
Dec turbines	331	0.823 (0.024)[c]	1.00	0.727 (0.025)[d]	1.00

(a) Reduced ViRDCt model
(b) CJS model
(c) Tag life-adjusted ViRDCt model
(d) Tag life-adjusted CJS model

Table 5.2. Sample Sizes (N) and Estimated ViRDCt Survival Probabilities ($\hat{S}$) from Lookout Point Passage to the Lookout Point Immediate Tailrace (LPT Array) and to Dexter for Acoustic-Tagged CH1 Released into the Lookout Point Reservoir in February and April 2018. Detection probabilities (p) of each detection array (LPT and Dexter) are also shown. Virtual release groups (V_1) were formed by month of release and route of passage at Lookout Point. Standard errors (SEs) of survival estimates are shown in parentheses. All detection probability SEs were ≤0.01. Superscripts indicate the model used to estimate survival.

		Lookout Point to Immediate Tailrace		Lookout Point to Dexter	
V_1 Group	N	$\hat{S}$ (SE)	p	$\hat{S}$ (SE)	p
February turbines	83	0.784 (0.047)[a]	1.00	0.699 (0.050)[b]	1.00
April turbines	11	0.654 (0.189)[e]	1.00	0.441 (0.143)[a]	1.00
Feb & April spillway	66	0.987 (0.055)[c]	1.00	0.884 (0.070)[c]	1.00
Spill and April Pooled	77	0.942 (0.057) [c]	1.00	0.822 (0.047)[c]	1.00

(a) Reduced ViRDCt model
(b) CJS model
(c) Tag life-adjusted ViRDCt model
(e) Full ViRDCt model

a. **Measure 392 + 105: Structure (FSS) with SWS** – Assumes design concept from DET scaled to LOP turbine capacity.

Run timing –

Fry – Same as baseline.
Subyearlings – Same as DET measure 392.
Yearlings – Same as DET measure 392.

Dam Passage Efficiency –

Dam Passage Efficiency within the pool elevation operating range of the FSS was estimated separately for each alternative. The method and results are described in Appendix A of this document.

Table 1-8. Dam Passage Efficiency values by Alternative

Alternative	DPE within the FSS pool elevation operating range
1	0.824
2	0.824
3a and 3b	Not applicable
4	*0.964*

Note: Below minimum conservation pool - applied DPE values from baseline

Route Effectiveness – Same as DET measure 392.

Route Survival –

Fish passage: 98% for all life stages. Other routes same as baseline.

n. Measure 166: Use lowest ROs in fall and winter drawdowns to reduce water temperatures below dams

Run timing, DPE, Route Effectiveness and Route Survival – same as LOP baseline

o. Measure 714 and 721: Use spillway to pass fish in the spring

Run timing, DPE, Route Effectiveness and Route Survival – same as LOP baseline

p. Measure 40: Deep fall drawdown to 10' over the top of the RO - on 15 Nov. (Anytime from 15 Oct – 15 Dec.)

Run timing, DPE, Route Effectiveness and Route Survival – same as LOP baseline

q. Measure 720 – Spring drawdown to lowest outlet for downstream passage – June 1-22.

Run timing -

- Fry – Same as LOP baseline. Reservoir is smaller in spring, but assume fry remain along shorelines until June (see Monzyk and Romer 2011-2015).
- Subyearlings – New. Assume majority of subs passing in June, when recruitment to the subyearling stage (>50mm size obtained, and more pelagically distributed) primarily occurs per Monzyk et al. 2010-2015).
- Yearlings – Same as LOP baseline.

Dam Passage Efficiency, Route Effectiveness and Route Survival – same as LOP baseline

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1.1.7 Chinook Attachment A

Fish Benefits Workbook (FBW) Dam Passage Efficiency (DPE) Calculations for Floating Screen Structures, Willamette Valley System EIS and ESA consultation fish effects analysis.

Floating screen structures (FSS) are dynamic in that they can accommodate varying elevations while taking advantage of available outflows. The FSS design includes two screened flumes or barrels that can accommodate a wider range of inflows better than a single flume design. Data on the fish collection efficiency of these and similar structures is limited but growing. For spring Chinook salmon, a target species for passage at Willamette dams, a wide range of collection rates have been observed among floating surface collectors operating in the Pacific Northwest (Kock et al. 2019). Some of these differences would be attributable to differences in designs and local conditions, making comparisons difficult among existing surface collectors. Kock et al. (2019) used a hierarchical log-linear regression to identify which design aspects most successfully predicted dam passage efficiency. They are: effective forebay size at a distance 500 meters from the dam face (ha), entrance size (m^2), collector inflow (m^3/s), and the presence of nets that improve fish guidance or efficiency (See Table 1-9 adapted from Kock et al. 2019). While this model is heavily focused on physical attributes of dam configuration and proposed engineering design dimensions for a collector, it is important to recognize that the collectors discussed in the EIS and the BA have yet to be successfully implemented and there is considerable risk and uncertainty about the realized effectiveness of these structures. Under modeled and simulated conditions, these collectors are expected to perform reasonably, but real time management or unobserved conditions could impact the effectiveness of proposed collectors, particularly in cases where the predictor variables represent the highest extremes of the functional relationships described in Kock et al. (2019). For this reason, dam passage efficiency should be interpreted in the lens of perfect information and actual results may vary.

Table 1-9. Coefficients for each significant predictor of fish collection efficiency. *

Variable	Coefficient estimate	SE	t-value	P-value
Intercept (Chinook Salmon)	-0.923	0.356	NA	NA
Coho Salmon	0.876	0.371	2.361	0.023
Sockeye Salmon	0.631	0.383	1.647	0.107
Steelhead	1.474	0.539	2.737	0.009
Lead nets	0.848	0.313	2.705	0.009
Inflow	0.492	0.068	7.188	<0.001
Effective forebay area	-1.086	0.183	-5.945	<0.001
Entrance area	0.991	0.233	4.254	<0.001
Effective forebay area x entrance area	2.112	0.362	5.835	<0.001

Notes: * Adapted Table 7 from Kock et al. 2019.

** Table 7 Coefficient estimates, SEs, and tests of significance for the effect of each predictor variable on fish collection efficiency (FCE) from Kock et al. 20

Forebay size for application of the Kock et al. regression model was estimated following the methods described by Kock et al. (2019). An FSS has been designed for Detroit and for Cougar; however, FSS's are also measures proposed for several other projects for the Willamette Systems EIS. The most relevant information about what inflows and entrance sizes may be reasonably expected comes from the design plans for Detroit and Cougar.

Forebay Size

Similar to Kock et al. (2019), effective forebay size was calculated as the water surface area from the face of the dam to the area 500m from the dam face. This was calculated for each project of interest:

Table 1-10. Effective forebay size for several Willamette Systems projects

Project	Size	Unit
Hills Creek	55.4	Ha
Green Peter	20.9	Ha
Cougar	27.6	Ha
Foster	47.9	Ha
Detroit	24.2	Ha
Lookout Point	35.4	Ha

Inflow and Entrance Specifications

We used Detroit and Cougar and scaled the designs and operations to the projects for which they were most similar.

Minimum and maximum flows through the FSS for DET and CGR were based on design flow ranges as documented in the DDRs. The FSS inflow operating range for a Hills Creek Dam FSS were assumed from the Cougar Dam FSS design, given the similarity in dam configuration and turbine capacity. Total FSS inflow capacity for GRP and LOP were determined by scaling based on the DET design flow. This was accomplished by dividing the DET total design flow by the DET turbine capacity, and then multiplying the result with the total turbine capacity flow at GRP and LOP. Due to the frequency at which flows can be less than 1000 cfs from GRP Dam, it was assumed that pumped flow would be used to supplement the FSS inflows up to 1000 cfs for the minimum FSS operating range at GRP.

Table 1-11. Detroit specifications. *

Project	Max total turbine capacity at min con	FSS V-screen design flow	Scaler (design flow / turbine capacity)
DET	4960	4600 (double barrel)	0.927

Note: * Green Peter and Lookout Point do not currently have an FSS design. Therefore, proposed FSS's at these locations were scaled to the Detroit FSS based on turbine capacity.

Table 1-12. Proposed Green Peter and Lookout FSS specifications *

Project	Max total turbine capacity at min con	DET FSS Scaler	Estimated Double V-screen design flow	Total V-screen design flow assumed for EIS
LOP	8100	.927	7509	6000
GPR	4420	.927	4097	4000

Note: * Proposed FSS specifications for Green Peter and Lookout scaled to the Detroit FSS design.

Adjusted down design flow, based on Kock et al. 2019 model of FSC fish guidance efficiency indicating efficiency would be high assuming a double V-screen designed of 6000 cfs.

Min con = Minimum Conservation Pool.

Table 1-13. Minimum and maximum flows through each FSS structure by project *

Project	Minimum FSS flow *	Maximum FSS flow *	Notes
Detroit FSS[1]	1000	5600	Per Detroit DDR
Cougar FSS[2]	300	1000	Per Cougar DDR
Green Peter FSS	1000	4000	Based on DET FSS scaler * GPR turbine capacity (See table above)
Lookout Pt FSS	1350 (equivalent to cavitation limit for DEX)	6000	Based on DET FSS scaler * LOP turbine capacity, adjusted based on Kock et al. FSC model (see table above)
Hills Creek FSS	300	1000	Assumed from CGR DDR

Notes: 1 Detroit FSS: There are two entrances in the FSS, capable of handling flow ranges from 1,000 cfs to 5,600 cfs. The design flow rate for fish collection operations is 4,500 cfs, with each channel operating at a flow of 2,250 cfs. Future provisions for pumped attraction flow will accommodate 1,000 cfs to drive flow through the FSS and continue attracting and collecting fish from the forebay. – per Final DDR.

2 Cougar FSS: There are two entrances on the Dual Entrance Angled FSS, with the starboard collection channel sized to pass 400 cubic feet per second (cfs) and the port collection channel sized to pass 600 cfs. Including two entrances instead of only one allows for better control of hydraulic conditions over the full range of design flows (300 to 1,000 cfs). – per 90% DDR.

* Flows are in cubic feet per second (cfs).

We applied these scalers at other projects of interest. Entrance size for a conceptual FSS at Hills Creek Dam was assumed from the Cougar Dam FSS design given the similarity in dam configuration and turbine capacity. These scaled relationships provided the most likely dimensions for an FSS at each project of interest based on available information (Table 4). Due to the frequency at which flows can be less than 1000 cfs from Green Peter Dam, it was assumed that pumped flow would be used to supplement the FSS inflows up to 1000 cfs for the minimum FSS operating range at GRP.

Table 1-14. Estimated dimensions of FSS entrances, minimum, and maximum outflow capacities. *

Project	Entrance area	Maximum FSS flow	Minimum FSS Flow
DET FSS	1776	5600	1000
GPR FSS	1268	4000	1000
LOP FSS	1902	6000	1350
CGR FSS	1938	1000	300
HCR FSS	1938	1000	300

Note: * Dimension estimates are based on turbine capacities and the relationship between entrance size and inflows.

Dimensions are indicated in Imperial units (square feet) but were converted to Metric for use in the log regression.

* Flows are in cubic feet per second (cfs).

It is important to note that entrance area is given for two flumes operating. When the FSS is operated at minimum inflow, only one barrel may operate. At these times, it was assumed that the entrance area is reduced by half. To investigate what flows were most likely at each project, we examined Res-Sim output for the period of record during peak fish passage times: April 1 – July 1 and September 1 to December 1. We developed a frequency distribution by binning dam discharge by 100 cfs increments. If the most frequently occurring flow was less than two times the minimum flow at a given project, we assumed single barrel operation and reduced the entrance size by half.

FCE Calculator

Once we had calculated the dimensions of each potential collector, we used these in the log-linear regression model from Kock et al. We adapted a spreadsheet “FCE Calculator” which captures the regression coefficients and log transformations to predict DPE.

Logistic regression equation for factors affecting FCE (from Kock et al. 2019)

$$lp = c_1 + c_2 \cdot I_{\text{coho}} + c_3 \cdot I_{\text{sockeye}} + c_4 \cdot I_{\text{steelhead}} + c_5 \cdot L + c_6 \cdot F + c_7 \cdot A + c_8 \cdot E + c_9 \cdot A \cdot E$$

$$FCE = \frac{\exp(lp)}{1+\exp(lp)}$$

Figure 1-1. Logistic regression equation used to predict DPE *(indicated as FCE, here)*.

The spreadsheet calculator allows the user to input their own values into the regression. These values are standardized per Kock et al. using the mean and standard error from their hierarchical analysis. Since data do not currently exist for collectors in the Willamette, we used the mean and standard deviation of multiple collectors evaluated in Kock et al. (see Supplement 3 in Kock et al. 2019) to approximate a standardized estimate (ie, $\frac{x-\bar{x}}{sd}$). These standardized inputs are then log transformed and imputed to the log regression equation for each proposed collector. The regression result (*lp*) must be untransformed from log space to provide DPE, here indicated as *FCE* in the reference text. All inputs were converted to Metric prior to analysis.

Table 1-15. Example of FCE calculator run. *

Variables	Coefficient	To Equation	Input Values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	0	0
c_6 Inflow =	0.492	1.392	28.316847
c_7 Effective forebay area =	-1.086	0.567	24.2
c_8 Entrance area =	0.991	-0.408	82.497864
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Notes: Users may input data into the white cells. Blue cells carry user inputs, log transform, standardize, and pass to the logistic regression (red cells). Lp is the log transformed DPE whereas FCE is the untransformed result.

lp = 0.279; FCE = 0.569

Calculation and justification for inflows through each collector

The FCE calculator was used to predict DPE for each structure where an FSS is proposed in Alternatives 1 and 4. Although the model is informative in that it can integrate information from very different collector types based on specific design features common to all collectors, the model assumes constant inflow through the collector. There are two main reasons that we expect variable inflows through proposed collectors: 1) The USACE conducts power peaking at several projects (Green Peter, Lookout Point, and Detroit dams) where hourly outflows change dramatically over the course of 24 hours, and 2) available water in a given year does not necessarily support the hypothesis that the collector would run at optimal capacity at all times.

To evaluate what flows might be expected, we examined the frequency of the daily average outflows predicted by Res-Sim and binned by 100 cfs intervals, under alternatives 1 and 4. As expected, the most frequently occurring outflows were substantially less than the optimal capacity assumed for each collector. In some cases, the flows were below the capacity needed to run even one barrel of an FSS. In these cases, we assumed supplemental pumps would be required to increase the inflow to minimum operating capacity (one barrel); however, at power peaking projects, the daily average may not accurately reflect hours of the day when inflows could also be quite high.

We used hourly outflow information from DBQuery to determine hourly outflow pattens in a deficit, sufficient, and adequate year type. Each year was then divided into different fish passage seasons: spring (April 1-July 1) and fall (September 1-December 1). We calculated the quantiles for hourly outflows (Table 1-16) and plotted the median hourly outflow by season (Figure 1-2).

Table 1-16. Detroit Abundant Year (2011) Spring and Fall Hourly Outflow Quantiles. *

Season	0%	25%	50%	75%	100%
Spring	0	0	1.97	2.075	4.38
Fall	0	0	1.95	2.14	5.21

Note: * Quantiles for hourly outflows at Detroit in an abundant year type (2011) in the spring and fall.

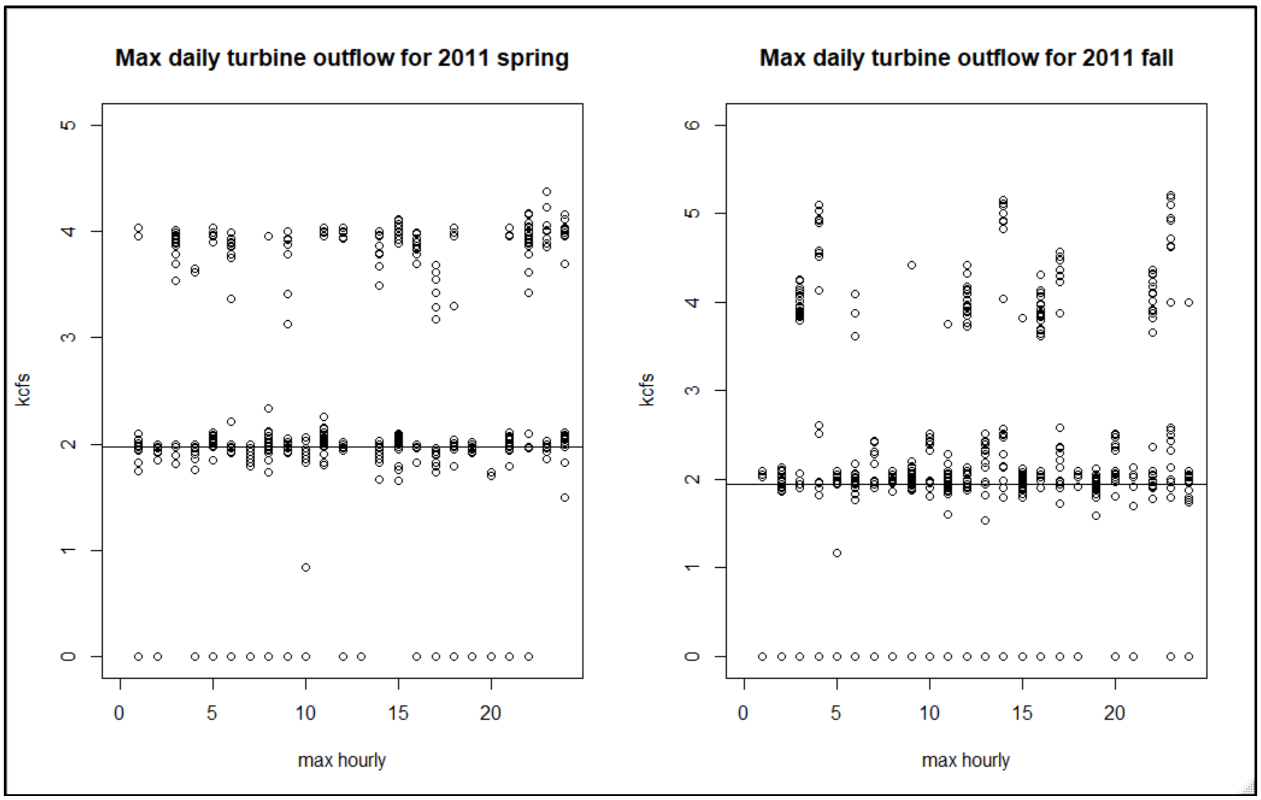

Figure 1-2. Detroit Spring and Fall Median Abundant Water Year Hourly Outflows.

Detroit Spring (Left) and Fall (Right) Median Abundant Water Year Hourly Outflows. The open dots represent the median hourly outflow. The solid line represents the median outflow for all data points.

In general, less than 25% of the hourly outflow data was above the optimal inflow capacity for Detroit. We show the abundant year type here to demonstrate that even under ideal conditions, the FSS would still operate below optimal capacity for most of the time. Therefore, we deemed it inappropriate to assume optimal capacity. We consulted with the Kock et al.

team to help determine reasonable inflows. The team agreed, it would be inappropriate to assume optimal capacity most of the time. They indicated that it was more reasonable to use the most frequently occurring daily outflow from Res-sim--with the caveat that the PDT should consider limiting power peaking at night when fish are most likely to pass and when variable flows would have the greatest impact of DPE. Furthermore, the team believed that the orientation of the collector (parallel to the dam face rather than perpendicular) would likely act as an efficient guidance structure and recommended utilizing the model coefficient for guide nets (see Kock et al. 2019).

We incorporated these suggestions into the current FCE calculator used to estimate DPE (see FBW, Appendix A sent to Cooperators on 03 June 2021). The results for DPE are presented with and without guide nets (see example in Table 1-17). In general, DPE improved 25%-30% when fish guidance considerations were included.

Table 1-17. Dam Passage Efficiency calculation for an FSS at Detroit for Alternative 4. *

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.467	29.73269
c_7 Effective forebay area =	-1.086	0.567	24.2
c_8 Entrance area =	0.991	-0.408	82.49786
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Notes: Estimates are for Chinook. The cells in red represent that log probability and DPE assuming a guidance structure.

lp = 1.353; FCE = 0.795; W/O LN = 0.587; percent change = 0.261289

Dam Passage Efficiencies for Alternative 1

Chinook

Table 1-18. Dam Passage Efficiency calculation for an FSS at Detroit under Alternative 1

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.392	28.316847
c_7 Effective forebay area =	-1.086	0.567	24.2
c_8 Entrance area =	0.991	-0.408	82.497864
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Notes: lp = 1.279; FCE = 0.782.

Table 1-19. Dam Passage Efficiency calculation for an FSS at Green Peter under Alternative 1

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.392	28.316847
c_7 Effective forebay area =	-1.086	0.638	20.9
c_8 Entrance area =	0.991	-0.582	58.900502
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Notes: lp = 1.175; FCE = 0.764

Table 1-20. Dam Passage Efficiency calculation for an FSS at Cougar under Alternative 1.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	0.615	16.9901082
c_7 Effective forebay area =	-1.086	0.495	27.6
c_8 Entrance area =	0.991	0.310	180.046014
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Note: lp = 1.147; FCE = 0.759

Table 1-21. Dam Passage Efficiency calculation for Lookout Point FSS at under Alternative 1

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	0	0
c_6 Inflow =	0.492	1.849	38.22774345
c_7 Effective forebay area =	-1.086	0.329	35.4
c_8 Entrance area =	0.991	-0.365	88.350753
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Note: lp = 0.541; FCE = 0.632

Table 1-22. Dam Passage Efficiency calculation for an FSS at Hills Creek under Alternative 1.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	0.177	12.74258115
c_7 Effective forebay area =	-1.086	-0.096	55.4
c_8 Entrance area =	0.991	0.310	180.046014
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Note: lp = 0.119; FCE = 0.530

Steelhead

Table 1-23. Dam Passage Efficiency calculation for an FSS at Detroit under Alternative 1.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	1	1
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.392	28.316847
c_7 Effective forebay area =	-1.086	0.567	24.2
c_8 Entrance area =	0.991	-0.408	82.497864
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Notes: lp = 2.279; FCE = 0.907

Table 1-24. Dam Passage Efficiency Calculation for a Green Peter FSS Under Alternative 1.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	1	1
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.392	28.316847
c_7 Effective forebay area =	-1.086	0.638	20.9
c_8 Entrance area =	0.991	-0.582	58.900502
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Notes: lp = 2.175; FCE = 0.898

Dam Passage Efficiencies for Alternative 2 – to be inserted after alternative description completed and RES-SIM hydrology results available

Dam Passage Efficiencies for Alternative 3a and 3b– to be inserted after alternative description completed and RES-SIM hydrology results available

Dam Passage Efficiencies for Alternative 4– to be inserted after alternative description completed and RES-SIM hydrology results available

Chinook

Table 1-25. Dam Passage Efficiency calculation for a Lookout Point FSS under Alternative 4.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	2.932	77.87132925
c_7 Effective forebay area =	-1.086	0.329	35.4
c_8 Entrance area =	0.991	0.286	176.701506
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Note: lp = 3.274; FCE = 0.964

Table 1-26. Dam Passage Efficiency calculation for a Detroit FSS under Alternative 4

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.467	29.73269
c_7 Effective forebay area =	-1.086	0.567	24.2
c_8 Entrance area =	0.991	-0.408	82.49786
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Note; lp = 1.353; FCE = 0.795

Table 1-27. Dam Passage Efficiency calculation for an FSS at Hills Creek under Alternative 4.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	0.177	12.74258115
c_7 Effective forebay area =	-1.086	-0.096	55.4
c_8 Entrance area =	0.991	0.310	180.046014
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Note: lp = 0.119; FCE = 0.530

Table 1-28. Dam Passage Efficiency calculation for an FSS at Cougar under Alternative 4.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.314	26.90100465
c_7 Effective forebay area =	-1.086	0.495	27.6
c_8 Entrance area =	0.991	0.310	180.046014
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Note: lp = 1.847; FCE = 0.864

Table 1-29. Dam Passage Efficiency calculation for an FSS at Detroit under Alternative 4

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	1	1
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.467	29.73269
c_7 Effective forebay area =	-1.086	0.567	24.2
c_8 Entrance area =	0.991	-0.408	82.49786
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Note: lp = 2.353; FCE = 0.913

Supporting Information for Biological Input Parameters Used for Modeling of the Willamette Valley System EIS Downstream Fish Passage Measures in the Fish Benefit Workbook (FBW)

1.2 ---WINTER STEELHEAD---

1.2.1 DETROIT & BIG CLIFF

Assumptions:

- Steelhead lifestages
  - Fry/early parr (June, year-0 to December, year - 0)
  - Parr (December, year-0 to December, year - 1)
  - Smolt (December, year-1 to December, year - 2).
- Mortality for Big Cliff reservoir and dam is 15% as utilized in the Engineering Design Report (EDR) for Detroit fish passage (USACE 2017a).
- Baseline includes spilling for temperature management, which is equivalent to the spring spill measure 714. It is assumed that these measures are identical.

a. No Action Alternative (i.e. Baseline) / Measure 714 (Use spillway to pass fish in the spring).

Run timing –

Downstream juvenile winter steelhead passage timing data for Detroit reservoir and dam is limited to studies which released artificially reared surrogates artificially reared from wild winter steelhead brood. Therefore timing inputs were developed by review of information from Green Peter and Foster dams where study of wild juvenile steelhead downstream passage has occurred. Romer et al. (2016) described that the “Typical life-history patterns observed for naturally-produced winter steelhead are dominated by age-2 smolts in the Columbia and Snake rivers as well as coastal Oregon streams (Busby et al. 1996). In the South Santiam River, juvenile O. mykiss migrate into Foster Reservoir at age-0, age-1, or age-2 and rear for a variable amount of time before exiting the reservoir. In the spring, only age-1 and age-2 fish are present in the basin. The first age-0 juveniles typically begin entering the reservoir in late June soon after emergence, and this age-class continues to enter the reservoir through the rest of the year (Romer et al. 2015). Juveniles can exit Foster Reservoir at any of the three age-classes, although age-2 smolts are the primary age class that continues to the Columbia River estuary (discussed later in this report)”. Passage patterns observed at Green Peter Dam however we assume are more representative of how steelhead would be expected to use Detroit Reservoir, given both are larger than Foster Reservoir and operated for flood risk management. Wagner and Ingram (1973) observed that 69-88% of the juvenile winter steelhead passing downstream at Green Peter Dam in April and May. We calculated percentages observed monthly from Table 9 in Wagner and Ingram (Table 1-30, below) and used this as the primary basis for passage assumptions at Detroit and Green Peter dams. The average annual size of emigrating steelhead during the years 1969 to 1971 ranged from 176 mm to 197 mm. We assumed some age-0's would pass in their first summer but most in their first fall/winter; and that age-1's and age-2’s would pass in spring. Information from studies of passage of winter steelhead at Foster Dam (Monzyk et al. 2017, Romer et al. 2017), and passage of tagged juvenile winter steelhead artificially reared and released into Detroit Reservoir (Beeman et al. 2013; Johnson et al. 2016) support the assumption that most juvenile winter steelhead would pass Detroit Dam in spring.

Table 1-30. Green Peter Dam Wild Reared Steelhead 1968-1971. *

Month	1968	1969	1970	1971	Avg
Jan	0%	3%	1%	0%	1%
Feb	nd	0%	3%	2%	2%
Mar	nd	3%	12%	1%	6%
Apr	24%	32%	30%	27%	28%
May	60%	43%	39%	61%	51%
Jun	10%	18%	13%	9%	12%
Jul	1%	0%	0%	0%	0%
Aug	nd	nd	nd	nd	nd
Sep	nd	nd	nd	nd	nd
Oct	0%	0%	0%	0%	0%
Nov	0%	0%	1%	0%	0%
Dec	4%	1%	0%	0%	1%

Notes: * Percentages of wild reared juvenile winter steelhead enumerated at the juvenile evaluation station at Green Peter Dam prepared from catch data in Table 9 from Wagner and Ingram (1973).

ND = no data.

The percentages of wild juvenile winter steelhead passing Green Peter Dam in 1969-1971 is very consistent with patterns of juvenile steelhead collected in the lower Santiam (Whitman et al. 2017; see Figure 5). Monitoring of wild juvenile winter steelhead migrating downstream into Foster Reservoir and passage Foster Dam although showed the majority of wild juvenile winter steelhead emigrate into Foster Reservoir as age-0 in early summer, most passed downstream at Foster Dam at Age 2 primarily in the spring (Monzyk et al. 2017). Romer et al. (2017) reports migration timing from screwtrapping into Foster Reservoir consistent with Monzyk et al. (2017), however screwtrapping below Foster Reservoir was found unreliable for assessing timing of wild juvenile winter steelhead since the trap did not collect fish passing over the spillway. Therefore, we adopted the monthly averages for Age 1 and Age 2 steelhead calculated from Wagner and Ingram.

For Age-0, we applied above reservoir catch patterns reported by Romer et al. (2017; see Figure 15), showing most Age-0 entering between July and December with most in August to October. However, Hughes et al. (2017) provided reservoir residency time for active tagged juveniles of up to 3 weeks in Foster Reservoir. Due to the larger size of Detroit Reservoir and smaller size of age-0 fry, we shifted the timing of reservoir entry one month forward, to account for reservoir residency and rearing of Age-0 steelhead prior to arrival in the dam forebay and their availability to pass downstream.

Comparison or run-timing information:

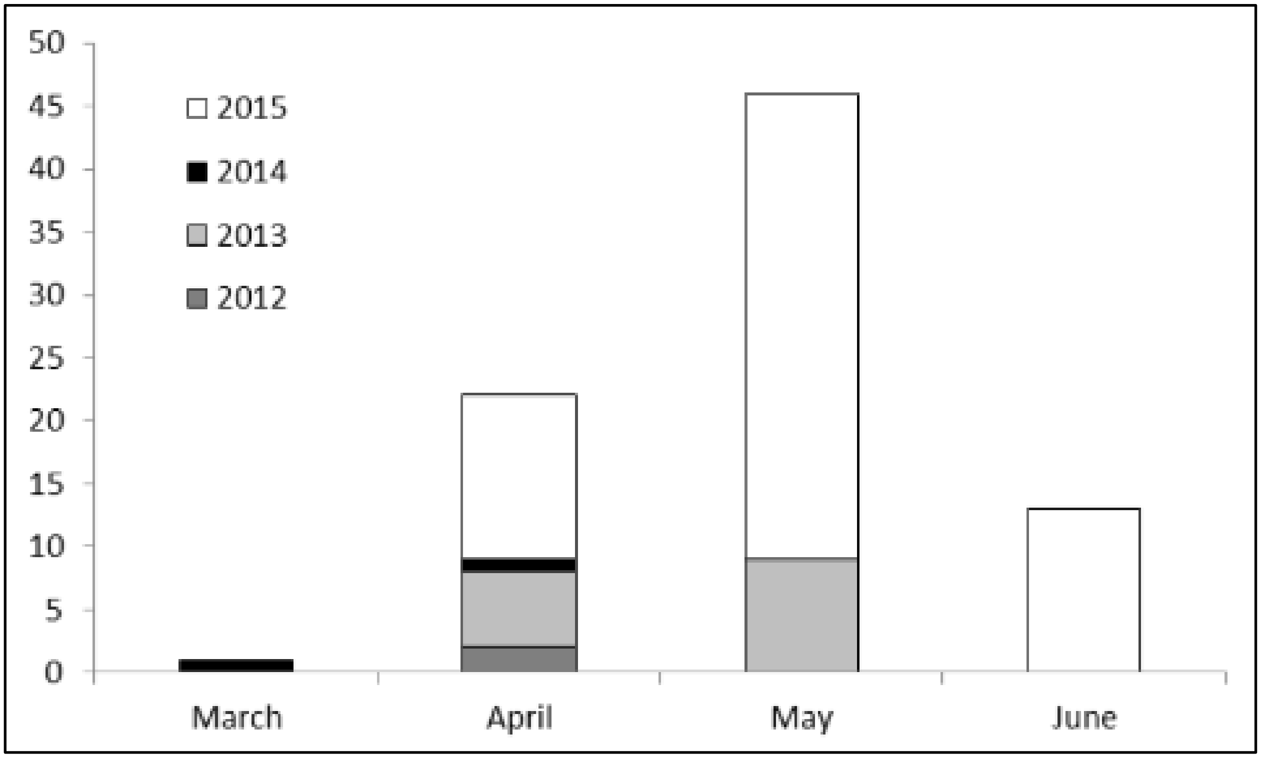

Figure 1-3. Monthly Steelhead smolt detections at Willamette Falls or the Columbia Estuary. *Steelhead smolt detections by month (N-82) at Willamette Falls or the Columbia Estuary during seaward migration. Year corresponds to the year of migration (or detection), not to year tagged (Romer et al. 2016; Figure 15).*

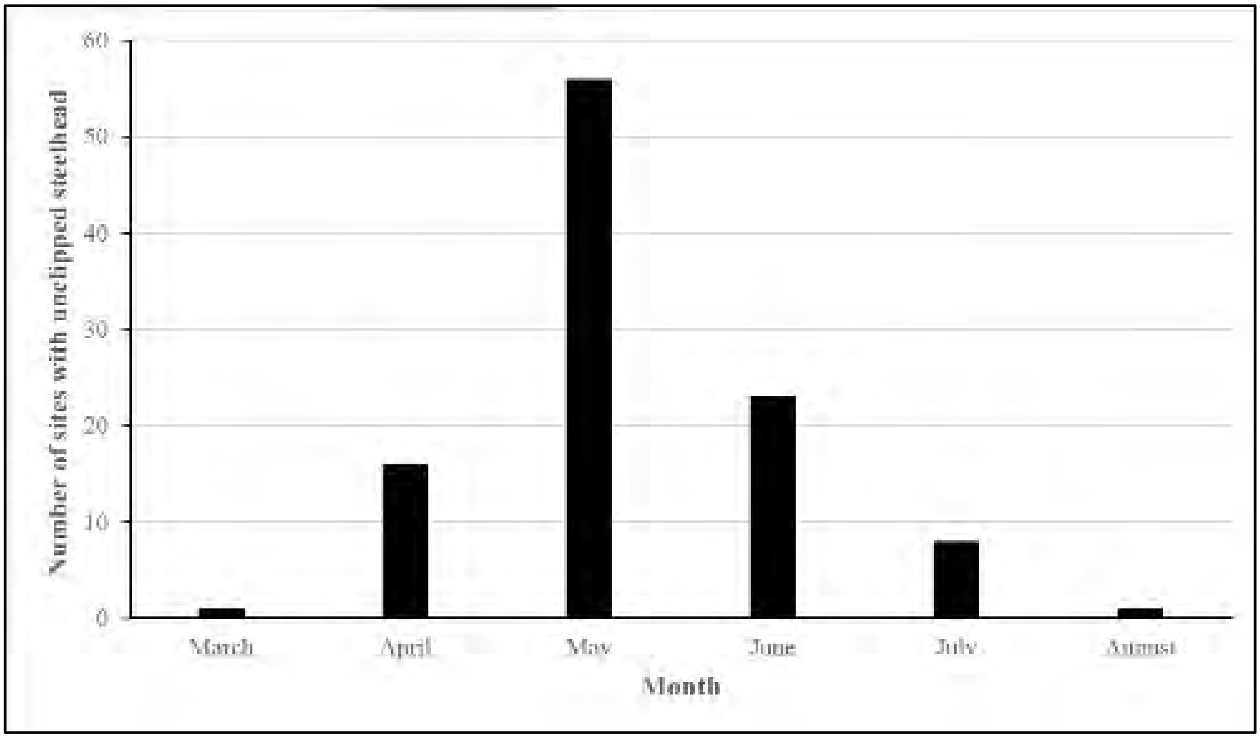

Figure 1-4. Scinc sites where unclipped juvenile steelhead were present, by Month. *Figure 5 from Monzyk et al. (2017)*

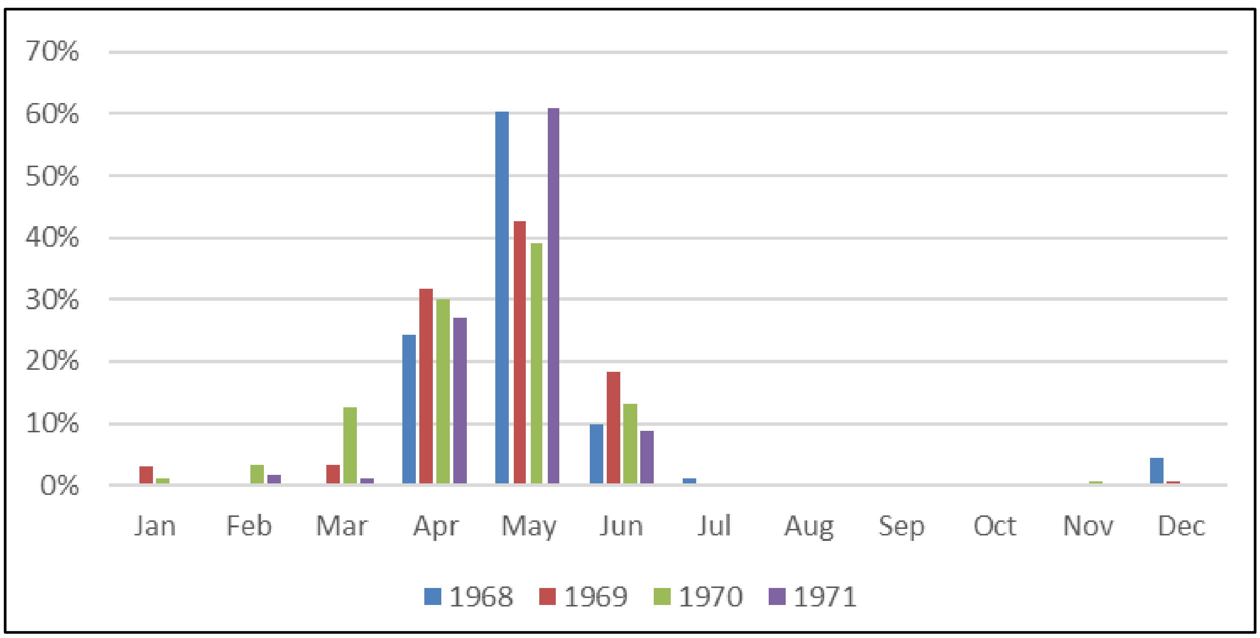

Figure 1-5. Juvenile Winter Steelhead Downstream Passage at Green Peter Dam. *Figure reproduced from data in Table 9, Wagner and Ingram (1973).*

Dam Passage Efficiency – DPE

Beeman and Adams (2015) estimated DPE for steelhead in spring 2013 at Detroit Dam at 0.678, during which time all active tagged steelhead passed over the spillway which was operating through much of the study period. Their study also released active tagged steelhead in the fall, however no steelhead passed Detroit Dam during the fall study period when the reservoir was being drafted down to the minimum conservation pool elevation. As summarized by Beeman and Adams (2015), "The near lack of passage of tagged steelhead during the fall study period may be related to the use of a summer-run stock, but results from tagged winter-run steelhead at Foster Dam were similar to those we report, suggesting it is a seasonal phenomenon".

Evaluations of juvenile steelhead passage at Foster Dam shows a strong preference for surface routes. Liss et al. (2020) estimated DPE from active tag hatchery steelhead (both summer and winter run) released into Foster Reservoir).

The fish weir provides a passage route downstream at the water surface and was modified in 2018. Other outlets at Foster Dam (spillbays and turbine penstocks) require fish to pass at different depths depending on the reservoir surface elevation. During low pool conditions of the Liss et al. study, with the new weir operating in 2018, DPE ranged from 0.43–0.53 for steelhead. The pool surface elevation was about 613', with depths to the spillway crest of about 16' and to the top of the turbine penstock of about 22'. For high pool operation in summer, also with the new weir operating, DPE for steelhead was 0.38.

Nearly all steelhead that passed downstream used the weir during the high pool study period. The pool elevation was about 635', with depths to the spillway crest of about 38' and to the top of the turbine penstock about 44'. Based on the combination of Beeman and Adams (2015) estimate for DPE at Detroit when above the spillway crest, the DPE estimates for Foster Dam from Liss et al, and Chinook DPE estimates for water depths to outlets beyond those covered by the previous references, we applied the Table 1-31 DPE estimates for Detroit Dam:

Table 1-31. Steelhead DPE estimates for Detroit Dam.

Pool Elevation	DPE	Note
1574	0.48	Max pool. 33' over spillway crest. Depth to top of outlet shallower than 33’ but depends on gate opening. Used the mid-value of .48 from the Foster DPE range of .43-.53 from Liss et al 2020, and no competing flows present
1557	0.68	15’ over spillway crest. Used Beeman and Adams DPE estimate since moderate depth to outlet and no competing flows present.
1541	0.68	Spillway crest. Used Beeman and Adams DPE estimate since shallow depth to outlet and no competing flows present.
1540	0.03	140’ over top of penstock. Value from Chinook DPE inputs.
1500	0.48	50’ over top of penstock. Used the mid-value of .48 from the Foster DPE range of .43-.53 from Liss et al 2020, and no competing flows present
1450	0.68	25' over top of penstock. Used Beeman and Adams 2015 DPE estimate since shallow depth to outlet.
1424	0.24	1 ft below min power pool. 74' over top of RO
1400	0.48	50’ over top of RO. Used the mid-value of .48 from the Foster DPE range of .43-.53 from Liss et al 2020, and no competing flows present
1375	0.68	25' over top of RO
1340	0.68	Upper RO. Used Beeman and Adams DPE estimate since shallow depth to outlet.

Route effectiveness –

The Beeman and Adams 2015 report of the 2013 study included a spillway effectiveness value of 2.92 for steelhead released into tributaries above Detroit Reservoir, and 8.84 for fish released into the head of Detroit Reservoir (but there were few fish from which to make the estimate). Therefore’ an average of the two estimates, weighted by the sample size, was used of 3.74 for the spillway RE value. In the 2013 study, no steelhead passed downstream when the pool was below the spillway crest during the fall study and therefore RE values were applied from Alden 2014 for the RO and turbines. The turbine RE value recommended by Alden of 1.16 for Detroit Dam is similar to their recommended RE value for Foster turbines of 1.0. Having the RO as a lower RE value of 0.542 at flow ratios of less than one makes sense, since this would occur when turbines are also operating at a much shallower depth.

Route survival –

For turbines and ROs, applied the same values used in Alden (2014) for this dam. For spillway survival, Beeman et al. (2015) estimated survival at Detroit Dam of 0.78 (range 0.70 to 0.95) for active-tagged juveniles with a size representative of parr and smolt. Since tagged fish passed over the spillway in this study we are applying the estimate of 0.78 for Detroit spillway for all lifestages of juvenile winter steelhead, also assuming age-0 survival would be this rate or higher due to their smaller size.

b. Measure 392+105: FSS with SWS

Flow range determined in the Detroit Design Documentation Report (DDR) for the Floating Screen Structure (FSS) is 1,000 – 5,600 CFS, with all flow to the Selective Withdrawal Structure (SWS) going through FSS to avoid competing flow. Above 5,600 through the FSS we are not in NMFS fry criteria anymore and would want lower survival for fry -- here we assume that above 5,600, water would be drawn in from a low-level inlet and assume no fish in that part of the water column.

Run timing –

We adjusted timing to align with average monthly surface spill operations in spring to account for the increased attraction from surface spill. For measure 392, we adjusted baseline run timing back one month, assuming more normative run timing for all life stages with an FSS operating throughout the year when above the minimum conservation pool elevation.

Dam Passage Efficiency –

Above minimum conservation pool– DPE within the pool elevation operating range of the FSS was estimated separately for each alternative. The method and results are described in Appendix A of this document.

Table 1-32. Dam Passage Efficiency values by Alternative.

Alternative	DPE within the FSS pool elevation operating range
1	.907
2a and 2b	.94
3a and 3b	Not applicable
4	.91

Note: Dam Passage Efficiency, below minimum conservation pool - applied DPE values from DET baseline.

Route Effectiveness –

Applied same values as used for baseline RE for existing routes. For the FSS per measure 392, applied the Applied Alden (2014) value of 13.11. Alden provided the rationale for the 13.11 value stating “steelhead collection effectiveness for surface type collectors and bypasses in the Columbia and Snake Rivers ranged from 5.3-24.6, with an average of 13.11 (See table in spreadsheet). This value was based on a flow ratio of 0.04. The 13.11 value was used for all flow ratios. At a flow ratio of 0.2 through the FSS the 13.11 value results in 78% of the steelhead entering the collector”.

Route survival –

98% for all life stages for the fish passage route (FSS). Other routes same as baseline. The FSS is assumed to have a passage survival of 98% for all target species collected, based on structures operating in the Northwest similar to the FSS concepts being considered for the WVS EIS (see USACE 2015 section 2.5.5).

Measure 40 – Deep fall drawdown to 10ft over the top of the upper RO's – Target start date 15 Nov and maintained for three weeks.

Run timing - same as baseline.

Dam Passage Efficiency – same as baseline.

Route effectiveness – same as baseline.

Route survival – same as baseline.

Measure 720: Spring delay refill with target elevation at 10' over the top of the upper RO's. May 1 to May 21 at target elevation.

Run timing – Same as Measure 392

Dam Passage Efficiency – Same as baseline

Route Effectiveness – Same as baseline

Route Survival – Same as baseline

1.2.2 FOSTER

- Baseline includes spilling for temperature management, which is equivalent to the spring spill measure 714. It is assumed that these measures are identical.
- Lifestage definitions same as DET

a. Baseline

Run timing –

Information from Romer et al. (2017) and previous reports from their screw trap monitoring efforts consistently show the majority of juvenile wild winter steelhead that enter Foster reservoir are age-0 fish while age-2 fish appear to comprise the majority of fish exiting the reservoir. Romer et al. points out that this suggests that the reservoir serves as rearing habitat for a large portion of the juvenile population. Therefore, the above reservoir screwtrap data is not necessarily representative of timing of passage from Foster Reservoir to downstream of Foster Dam. The below Foster Dam screwtrap operated for a few years below the turbines also may be of limited value since most steelhead prefer to pass over the fishweir or the spillways. However, Monzyk et al. (2017) reported that travel time from Foster Dam to Willamette Falls was about 6 days (based on PIT detections), and therefore Willamette Falls Passage timing would be reasonable for estimating monthly Foster Dam passage timing. They reported detections of PIT tagged juvenile steelhead, that were released above Foster Dam, occurred March to June at Willamette Falls with a monthly pattern very similar to that observed by Wagner and Ingram (1973) for Green Peter Dam passage (see comparison of run timing in figures presented above for Detroit Run Timing). Therefore, we used the same run timing applied for Green Peter Dam for Foster Dam.

Dam Passage Efficiency –

Applied data from Liss et al. (2020). The fish weir provides a passage route downstream at the water surface. Other outlets require fish to pass at variable depths. During low pool, with the new weir operating in 2018, DPE ranged from 0.43–0.53 for steelhead. The pool elevation was about 613’, with depths to the spillway crest of about 16’ and to the top of the turbine penstock about 22’. For high pool operation in summer, with the new weir operating in 2018, DPE for steelhead was 0.38. Nearly all steelhead that passed downstream used the weir during the high pool study period. The pool elevation was about 635’, with depths to the spillway crest of about 38’ and to the top of the turbine penstock about 44’. We assumed the lower end of the DPE range of estimates for a high pool DPE, the higher end of the DPE estimates for the low pool DPE and applied a value from the middle of the DPE estimate range for an elevation between low and high pool. We did not distinguish DPE among parr and smolt lifestages assuming the active tag data are applicable to both parr and smolts. We assumed fry would show a similar preferences for passing at lower pool elevations when depths to outlets are lower.

Table 1-33. Foster Baseline Measure Dam Passage Efficiency

Pool Elevation	Fry	parr	smolt
635	0.38	0.38	0.38
623	0.43	0.43	0.43
613	0.53	0.53	0.53

Route Effectiveness –

Applied Alden (2014), which included the rationale that “Draft hydroacoustic data collected in 2013 indicate that 54% of the fish passed the dam through the weir, with 23% through the spillway. Effectiveness values were set to achieve 54% passage through the weir (fish passage structure at a flow of ratio of 20%. It was assumed that the weir passed 20% of the flow during the testing period, but this will need to be confirmed when data are available. Data is based primarily on Chinook and not steelhead. Liss et al. (2020) assessed passage efficiency of hatchery-reared winter steelhead outfitted with active tags. Average values across the three study years for fish weir effectiveness was 4.44 and was 1.97 for the spillway (see Table S.3; Liss et al. 2020, copied below). These newer data are consistent with the previous values applied by Alden for the weir and spillway of 4.8 and 2.0, respectively. However, the estimates provided by Liss et al. also show that passage effectiveness varies between low and high pool and among years.

Table 1-34. Table S.3 from Liss et al. 2020.

(Table S.3 continued)								
	STH2 – Spring						S-STH – Spring	
	2015		2016		2018		2018	
Metric	Low Pool	High Pool	Low Pool	High Pool	Low Pool	High Pool	Low Pool	High Pool
DPE	**0.432 (0.026)**	**0.762 (0.021)**	**0.529 (0.035)**	**0.667 (0.024)**	**0.464 (0.023)**	**0.378 (0.028)**	**0.439 (0.043)**	**0.519 (0.026)**
FPE	**0.355 (0.026)**	**0.749 (0.022)**	**0.375 (0.035)**	**0.649 (0.025)**	**0.319 (0.022)**	**0.371 (0.028)**	**0.341 (0.041)**	**0.517 (0.026)**
SPE || Dam	0.852 (0.034)	0.994 (0.006)	0.739 (0.053)	1.000 (0.000)	0.683 (0.032)	0.982 (0.013)	0.776 (0.055)	0.995 (0.005)
FWE || Dam	0.426 (0.048)	0.971 (0.013)	0.434 (0.060)	0.973 (0.014)	0.318 (0.032)	0.973 (0.016)	0.328 (0.062)	0.979 (0.011)
SBE || Dam	0.426 (0.048)	0.023 (0.012)	0.304 (0.055)	0.027 (0.014)	0.365 (0.033)	0.009 (0.009)	0.448 (0.065)	0.016 (0.009)
Fish Weir Effect.	2.908 (0.325)	5.992 (0.079)	4.782 (0.656)	7.353 (0.102)	2.160 (0.218)	3.430 (0.055)	2.228 (0.419)	3.451 (0.037)
Spill Bay Effect.	0.947 (0.106)	0.102 (0.050)	0.753 (0.137)	0.146 (0.072)	0.903 (0.082)	0.046 (0.046)	1.109 (0.162)	0.081 (0.046)
Spillway Effect.	1.429 (0.057)	2.534 (0.015)	1.493 (0.107)	3.120 (0.000)	1.238 (0.058)	2.037 (0.026)	1.407 (0.099)	2.064 (0.011)

Route survival –

Applied averages of estimated survival for subs and parr for each route from Liss et al. (2020). Low and high pool survival estimates were available for yearlings, and so the average across both pool elevations was applied.

b. Measure 392

Run timing - Same as baseline.

Dam Passage Efficiency –

Measure 392 for Foster Dam is a concept of either further improving the fish weir operated in Spillbay 4 or constructing a dedicated fish collection and bypass pipe in the same vicinity as the fish weir, with either concept operating up to about 600 cfs. Until further refinement of this concept, we assumed a DPE consistent with the highest DPE measured at the dam for steelhead to date of 0.76 as reported in Table 5.6 of Liss et al. (2020).

Route Effectiveness –

Applied Alden (2014)

Route survival –

For spillway and turbines, used same values as for baseline. For fish passage route, assumed 98%, where fish passage concept is either a modified overflow weir or a dedicated fish pipe (see USACE 2015 section 2.5.5).

1.2.3 GREEN PETER

Lifestage definitions same as DET

a. Baseline

Not applicable – no fish outplanted above dam.

b. Measure 392: GPR FSS –

Run timing – same as DET timing for Measure 392.

Dam Passage Efficiency –

DPE within the pool elevation operating range of the FSS was estimated separately for each alternative. The method and results are described in Chinook Attachment A of this Chapter. Dam Passage Efficiency values by Alternative when above minimum conservation pool.

Table 1-35. Green Peter Dam Passage Efficiency

Alternative	DPE within the FSS pool elevation operating range
1	0.898
2a and 2b	Not applicable
3a and 3b	Not applicable
4	Not applicable

Below minimum conservation pool elevations, we applied DPE values from baseline for similar depths to outlets at GPR.

Route effectiveness –

Applied DET RE values due to similarity in dam configuration. Local data on RE for existing routes at GPR not available.

Route survival –

Route survival was 98% for fish passage route (see USACE 2015, section 2.5.5). Spillway, turbines and RO assumed the same as DET due to similar dam configuration.

c. Measure 714 and 721: Spring/summer spill

Run timing –

Applied DET baseline timing.

Dam Passage Efficiency –

Applied DPE input values developed for DET baseline adjusted for depths to outlets at GPR. Assumed highest DPE when pool surface elevation < depth over top of outlet.

Route effectiveness –

Applied DET RE values due to similarity in dam configuration. Local data on RE for existing ***routes at GPR not available.***

Route survival –

Applied route survival from DET due to similarity in dam configuration. No site specific data on juvenile downstream passage survival for spillway, turbines and ROs.

d. Measure 40 (deep fall drawdown)

Same as 714 and 721)

e. Measure 720 (spring delay refill)

Same as 714 and 721

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1.2.4 Steelhead Attachment A

Fish Benefits Workbook (FBW) Dam Passage Efficiency (DPE) Calculations for Floating Screen Structures, Willamette Valley System EIS and ESA consultation fish effects analysis

Floating screen structures (FSS) are dynamic in that they can accommodate varying elevations while taking advantage of available outflows. The FSS design includes two screened flumes or barrels that can accommodate a wider range of inflows better than a single flume design. Data on the fish collection efficiency of these and similar structures is limited but growing. For spring Chinook salmon, a target species for passage at Willamette dams, a wide range of collection rates have been observed among floating surface collectors operating in the Pacific Northwest (Kock et al. 2019). Some of these differences would be attributable to differences in designs and local conditions, making comparisons difficult among existing surface collectors. Kock et al. (2019) used a hierarchical log-linear regression to identify which design aspects most successfully predicted dam passage efficiency. They are: effective forebay size at a distance 500 meters from the dam face (ha), entrance size (m^2), collector inflow (m^3/s), and the presence of nets that improve fish guidance or efficiency (See Table 1 adapted from Kock et al. 2019). While this model is heavily focused on physical attributes of dam configuration and proposed engineering design dimensions for a collector, it is important to recognize that the collectors discussed in the EIS and the BA have yet to be successfully implemented and there is considerable risk and uncertainty about the realized effectiveness of these structures. Under modeled and simulated conditions, these collectors are expected to perform reasonably, but real time management or unobserved conditions could impact the effectiveness of proposed collectors, particularly in cases where the predictor variables represent the highest extremes of the functional relationships described in Kock et al. (2019). For this reason, dam passage efficiency should be interpreted in the lens of perfect information and actual results may vary.

Table 1-36. Coefficients for each significant predictor of fish collection efficiency.

TABLE 7. Coefficient estimates, SEs, and tests of significance for the effect of each predictor variable on fish collection efficiency (FCE).

Variable	Coefficient estimate	SE	*t*-value	*P*-value
Intercept (Chinook Salmon)	−0.923	0.356	NA	NA
Coho Salmon	0.876	0.371	2.361	0.023
Sockeye Salmon	0.631	0.383	1.647	0.107
Steelhead	1.474	0.539	2.737	0.009
Lead nets	0.848	0.313	2.705	0.009
Inflow	0.492	0.068	7.188	<0.001
Effective forebay area	−1.086	0.183	−5.945	<0.001
Entrance area	0.991	0.233	4.254	<0.001
Effective forebay area × entrance area	2.112	0.362	5.835	<0.001

Note: Table 7 adapted from Kock et al. 2019 showing the coefficients for each significant predictor of fish collection efficiency.

Forebay size for application of the Kock et al. regression model was estimated following the methods described by Kock et al. (2019). An FSS has been designed for Detroit and for Cougar; however, FSS's are also measures proposed for several other projects for the Willamette

Systems EIS. The most relevant information about what inflows and entrance sizes may be reasonably expected comes from the design plans for Detroit and Cougar.

Forebay size

Similar to Kock et al. (2019), effective forebay size was calculated as the water surface area from the face of the dam to the area 500m from the dam face. This was calculated for each project of interest:

Table 1-37. Effective forebay size for several Willamette Systems projects

Project	Size	Unit
Hills Creek	55.4	Ha
Green Peter	20.9	Ha
Cougar	27.6	Ha
Foster	47.9	Ha
Detroit	24.2	Ha
Lookout Point	35.4	Ha

Inflow and Entrance Specifications

We used Detroit and Cougar and scaled the designs and operations to the projects for which they were most similar.

Minimum and maximum flows through the FSS for DET and CGR were based on design flow ranges as documented in the DDRs. The FSS inflow operating range for a Hills Creek Dam FSS were assumed from the Cougar Dam FSS design, given the similarity in dam configuration and turbine capacity. Total FSS inflow capacity for GRP and LOP were determined by scaling based on the DET design flow. This was accomplished by dividing the DET total design flow by the DET turbine capacity, and then multiplying the result with the total turbine capacity flow at GRP and LOP. Due to the frequency at which flows can be less than 1000 cfs from GRP Dam, it was assumed that pumped flow would be used to supplement the FSS inflows up to 1000 cfs for the minimum FSS operating range at GRP.

Table 1-38. Detroit specifications used for Green Peter and Lookout Point Scaling. *

Project	Max total turbine capacity at min con	FSS V-screen design flow	Scaler (design flow / turbine capacity)
DET	4960	4600 (double barrel)	0.927

Note: Green Peter and Lookout Point do not currently have an FSS design. Therefore, proposed FSS's at these locations were scaled to the Detroit FSS based on turbine capacity.

Table 1-39. Proposed FSS specifications for Green Peter and Lookout. *

Project	Max total turbine capacity at min con	DET FSS Scaler	Estimated Double V-screen design flow	Total V-screen design flow assumed for EIS
LOP	8100	.927	7509	6000
GPR	4420	.927	4097	4000

Note: * Proposed FSS specifications for Green Peter and Lookout, scaled to the Detroit FSS design.
LOP Adjusted down design flow, based on Kock et al. 2019 model of FSC fish guidance efficiency indicating efficiency would be high assuming a double V-screen designed of 6000 cfs.

For Detroit and Green Peter, when dam outflows are below the minimum operational flow, it is assumed that minimum flows are supplemented and recirculated with pumped flow from forebay.

Table 1-40. Minimum and maximum flows through each FSS structure by project *

Project	Minimum FSS flow **	Maximum FSS flow **	Notes
Detroit FSS[1]	1000	5600	Per Detroit DDR
Cougar FSS[2]	300	1000	Per Cougar DDR
Green Peter FSS	1000	4000	Based on DET FSS scaler * GPR turbine capacity (See table above)
Lookout Pt FSS	1350 (equivalent to cavitation limit for DEX)	6000	Based on DET FSS scaler * LOP turbine capacity, adjusted based on Kock et al. FSC model (see table above)
Hills Creek FSS	300	1000	Assumed from CGR DDR

Notes: * Minimum and maximum flows (cfs) through each FSS structure by project. For Detroit and Green Peter, whendam outflows are below the minimum operational flow, it is assumed that minimum flows are supplemented and recirculated with pumped flow from forebay

** All flows shown in cubic feet per second (cfs).

1. Detroit FSS: There are two entrances in the FSS, capable of handling flow ranges from 1,000 cfs to 5,600 cfs. The design flow rate for fish collection operations is 4,500 cfs, with each channel operating at a flow of 2,250 cfs. Future provisions for pumped attraction flow will accommodate 1,000 cfs to drive flow through the FSS and continue attracting and collecting fish from the forebay. – per Final DDR.

2. Cougar FSS: There are two entrances on the Dual Entrance Angled FSS, with the starboard collection channel sized to pass 400 cubic feet per second (cfs) and the port collection channel sized to pass 600 cfs. Including two entrances instead of only one allows for better control of hydraulic conditions over the full range of design flows (300 to 1,000 cfs). – per 90% DDR.

We applied these scalers at other projects of interest. Entrance size for a conceptual FSS at Hills Creek Dam was assumed from the Cougar Dam FSS design given the similarity in dam configuration and turbine capacity. These scaled relationships provided the most likely dimensions for an FSS at each project of interest based on available information (Table 4). Due to the frequency at which flows can be less than 1000 cfs from Green Peter Dam, it was assumed that pumped flow would be used to supplement the FSS inflows up to 1000 cfs for the minimum FSS operating range at GRP.

Table 1-41. Estimated FSS entrance dimensions, minimum and maximum outflow capacities *

Project	Maximum FSS flow (cfs)	Entrance area (sq ft)	Minimum FSS Flow (cfs)
DET FSS	5600	1776	1000
GPR FSS	4000	1268	1000
LOP FSS	6000	1902	1350
CGR FSS	1000	1938	300
HCR FSS	1000	1938	300

Notes: 1. Estimated dimensions for FSS entrances, minimum, and maximum outflow capacities based on turbine capacities and the relationship between entrance size and inflows.

2. Dimensions are indicated in Imperial units but were converted to Metric for use in the log regression.

Entrance area is given for two flumes operating. When the FSS is operated at minimum inflow, only one barrel may operate. At these times, the entrance area is reduced by half. We examined Res-Sim output for the period of record during peak fish passage times: April 1 – July 1 and September 1 to December 1 to estimate each project's most likely flows. We developed a frequency distribution by binning dam discharge by 100 cfs increments. If the most frequently occurring flow was less than two times the minimum flow at a given project, we assumed single barrel operation and reduced the entrance size by half.

FCE Calculator

Once we had calculated the dimensions of each potential collector, we used these in the log-linear regression model from Kock et al. We adapted a spreadsheet "FCE Calculator" which captures the regression coefficients and log transformations to predict DPE.

Logistic regression equation for factors affecting FCE (from Kock et al. 2019)

$$lp = c_1 + c_2 \cdot I_{\text{coho}} + c_3 \cdot I_{\text{sockeye}} + c_4 \cdot I_{\text{steelhead}} + c_5 \cdot L + c_6 \cdot F + c_7 \cdot A + c_8 \cdot E + c_9 \cdot A \cdot E$$

$$FCE = \frac{\exp(lp)}{1 + \exp(lp)}$$

Figure 1-6. Logistic regression equation used to predict DPE (indicated as FCE, here).

The spreadsheet calculator allows the user to input their own values into the regression. These values are standardized per Kock et al. using the mean and standard error from their hierarchical analysis. Since data do not currently exist for collectors in the Willamette, we used the mean and standard deviation of multiple collectors evaluated in Kock et al. (see Supplement 3 in Kock et al. 2019) to approximate a standardized estimate (i.e., $\frac{x-\bar{x}}{sd}$). These standardized inputs are then log transformed and imputed to the log regression equation for each proposed collector. The regression result (*lp*) must be untransformed from log space to provide DPE (Dam

Passage Efficiency will be indicated as FCE within Chapter 1). All inputs were converted to Metric prior to analysis.

Table 1-42. Example of FCE calculator run.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	0	0
c_6 Inflow =	0.492	1.392	28.316847
c_7 Effective forebay area =	-1.086	0.567	24.2
c_8 Entrance area =	0.991	-0.408	82.497864
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Notes: lp = 0.279; FCE = 0.569

Calculation and justification for inflows through each collector

The FCE calculator was used to predict DPE for each structure where an FSS is proposed in Alternatives 1 and 4. Although the model is informative in that it can integrate information from very different collector types based on specific design features common to all collectors, the model assumes constant inflow through the collector. There are two main reasons that we expect variable inflows through proposed collectors: 1) The USACE conducts power peaking at several projects (Green Peter, Lookout Point, and Detroit dams) where hourly outflows change dramatically over the course of 24 hours, and 2) available water in a given year does not necessarily support the hypothesis that the collector would run at optimal capacity at all times.

To evaluate what flows might be expected, we examined the frequency of the daily average outflows predicted by Res-Sim and binned by 100 cfs intervals, under alternatives 1 and 4. As expected, the most frequently occurring outflows were substantially less than the optimal capacity assumed for each collector. In some cases, the flows were below the capacity needed to run even one barrel of an FSS. In these cases, we assumed supplemental pumps would be required to increase the inflow to minimum operating capacity (one barrel); however, at power peaking projects, the daily average may not accurately reflect hours of the day when inflows could also be quite high.

We used hourly outflow information from DBQuery to determine hourly outflow pattens in a deficit, sufficient, and adequate year type. Each year was then divided into different fish passage seasons: spring (April 1-July 1) and fall (September 1-December 1). We calculated the quantiles for hourly outflows (Table 1-43) and plotted the median hourly outflow by season (Figure 1-7).

Table 1-43. Spring and Fall Quantiles for Detroit hourly outflows in an abundant year. *

Season	0%	25%	50%	75%	100%
Spring 2011	0	0	1.97	2.075	4.38
Fall 2011	0	0	1.95	2.14	5.21

Note: * Quantiles for hourly outflows at Detroit in an abundant year type (2011) in the spring and fall.

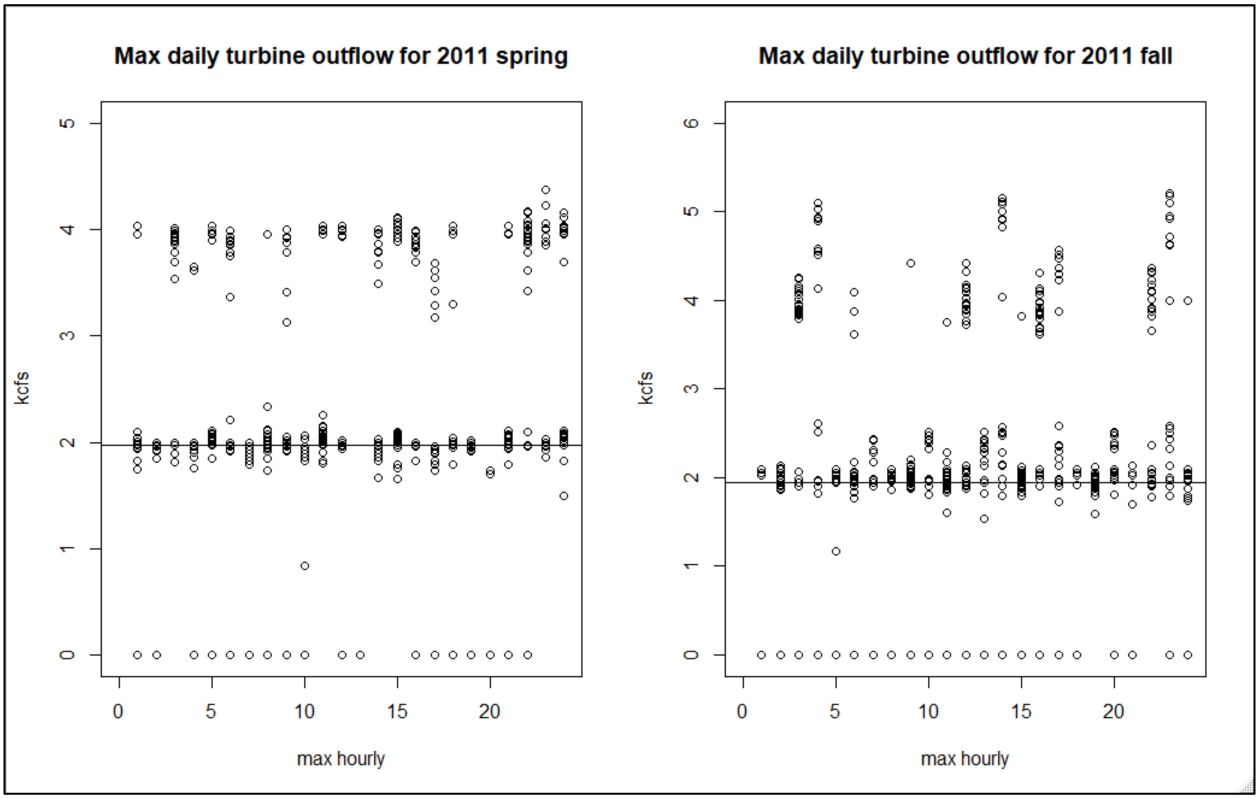

Figure 1-7. Detroit Median Hourly Spring and Fall Outflows in Abundant Water Years. *Median hourly outflows from Detroit for an abundant water year type (2011) in spring (left) and fall (right). The open dots represent the median hourly outflow. The solid line represents the median outflow for all data points.*

In general, less than 25% of the hourly outflow data was above the optimal inflow capacity for Detroit. We show the abundant year type here to demonstrate that even under ideal conditions, the FSS would still operate below optimal capacity for a majority of the time. Therefore, we deemed it inappropriate to assume optimal capacity. We consulted with the Kock et al. team to help determine reasonable inflows. The team agreed, it would be inappropriate to assume optimal capacity most of the time. They indicated that it was more reasonable to use the most frequently occurring daily outflow from Res-sim--with the caveat that the PDT should consider limiting power peaking at night when fish are most likely to pass and when variable flows would have the greatest impact of DPE. Furthermore, the team believed that the orientation of the collector (parallel to the dam face rather than perpendicular) would likely act as an efficient guidance structure and recommended utilizing the model coefficient for guide nets (see Kock et al. 2019).

We incorporated these suggestions into the current FCE calculator used to estimate DPE (see FBW, Appendix A sent to Cooperators on 03 June 2021). The results for DPE are presented with and without guide nets (see example in Table 2). In general, DPE improved 25%-30% when fish guidance considerations were included.

Table 1-44. DPE Calculation for an FSS at Detroit for Alternative 4.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	0	0
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.467	29.73269
c_7 Effective forebay area =	-1.086	0.567	24.2
c_8 Entrance area =	0.991	-0.408	82.49786
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Notes: Estimates are for Chinook. The cells in red represent that log probability and DPE assuming a guidance structure.

lp = 1.353; FCE = 0.795; W/o LN = 0.587; percent change = 0.261289

Dam Passage Efficiencies for Alternative 1

Table 1-45. DPE calculation for an FSS at Detroit under Alternative 1.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	1	1
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.392	28.316847
c_7 Effective forebay area =	-1.086	0.567	24.2
c_8 Entrance area =	0.991	-0.408	82.497864
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Note: lp = 2.279: FCE = 0.907

Table 1-46. Dam Passage Efficiency calculation for an FSS at Green Peter under Alternative 1.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	1	1
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.392	28.316847
c_7 Effective forebay area =	-1.086	0.638	20.9
c_8 Entrance area =	0.991	-0.582	58.900502
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Notes lp = 2.175; FCE = 0.898

Dam Passage Efficiencies for Alternative 2a and 2b

Table 1-47. Dam Passage Efficiency calculation for a Detroit FSS Alternatives 2a and 2b.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	1	1
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.849	38.22774345
c_7 Effective forebay area =	-1.086	0.567	24.2
c_8 Entrance area =	0.991	-0.408	82.497864
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Notes: lp = 2.736; FCE = 0.939

Dam Passage Efficiencies for Alternative 3a and 3b– *Not applicable*

Dam Passage Efficiencies for Alternative 4

Table 1-48. Dam Passage Efficiency calculation for a Detroit FSS under Alternative 4.

Variables	Coefficient	To equation	Input values
c_1 (Chinook salmon) =	-0.923	1	1
c_2 (coho salmon) =	0.876	0	0
c_3 (sockeye salmon) =	0.631	0	0
c_4 (steelhead) =	1.474	1	1
c_5 Lead nets =	0.848	1	1
c_6 Inflow =	0.492	1.467	29.73268935
c_7 Effective forebay area =	-1.086	0.567	24.2
c_8 Entrance area =	0.991	-0.408	82.497864
c_9 Effective forebay area x entrance area =	2.112	-2.273	n/a

Notes: lp = 2.353; FCE = 0.913

CHAPTER 2 - FISH BENEFIT WORKBOOK RESULTS

CHINOOK

2.1 CHINOOK NO ACTION ALTERNATIVE (NAA OR BASELINE)

2.1.1 North Santiam - Detroit

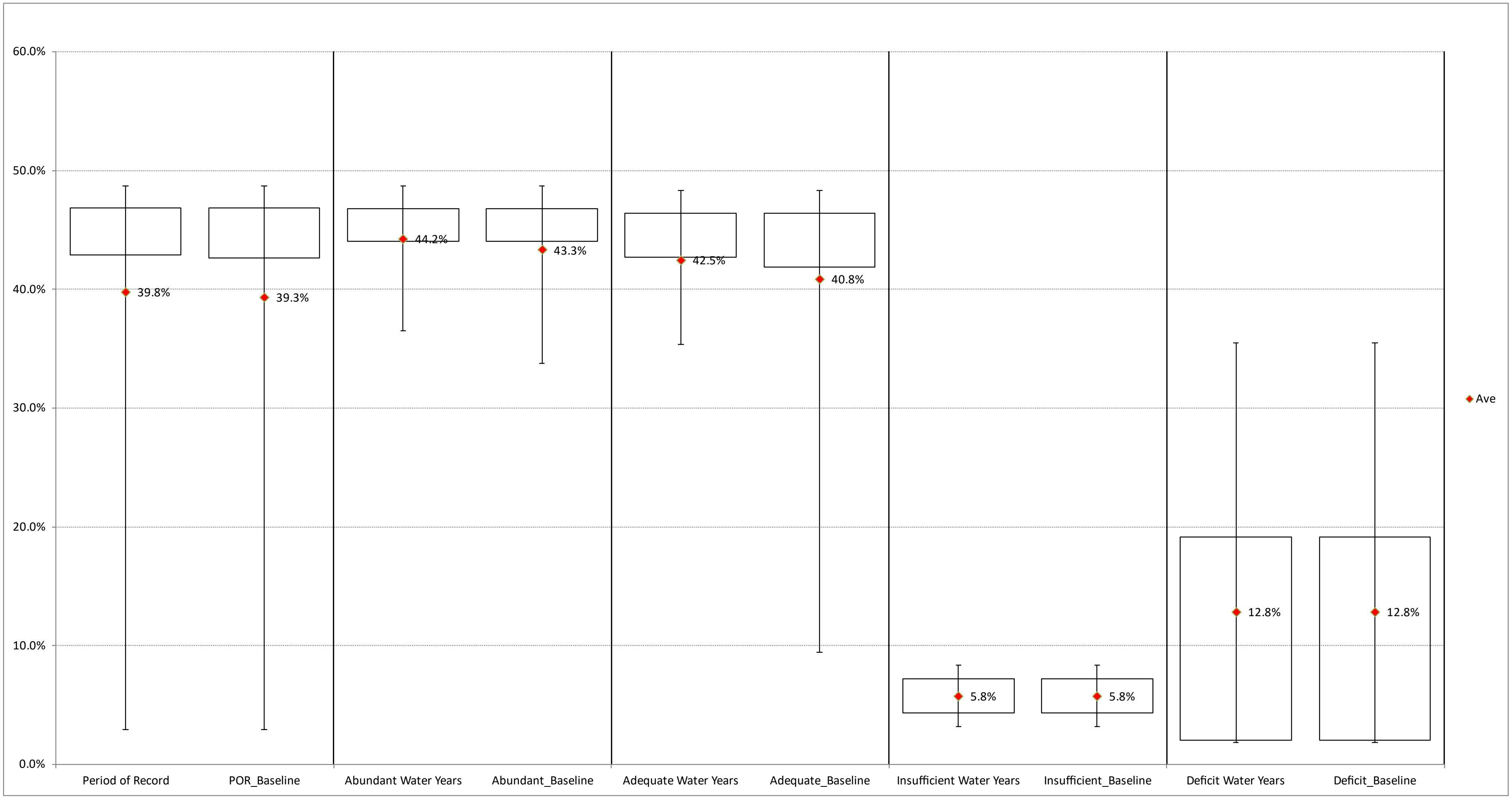

Figure 2-1. Detroit Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Detroit for juvenile spring Chinook fry under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

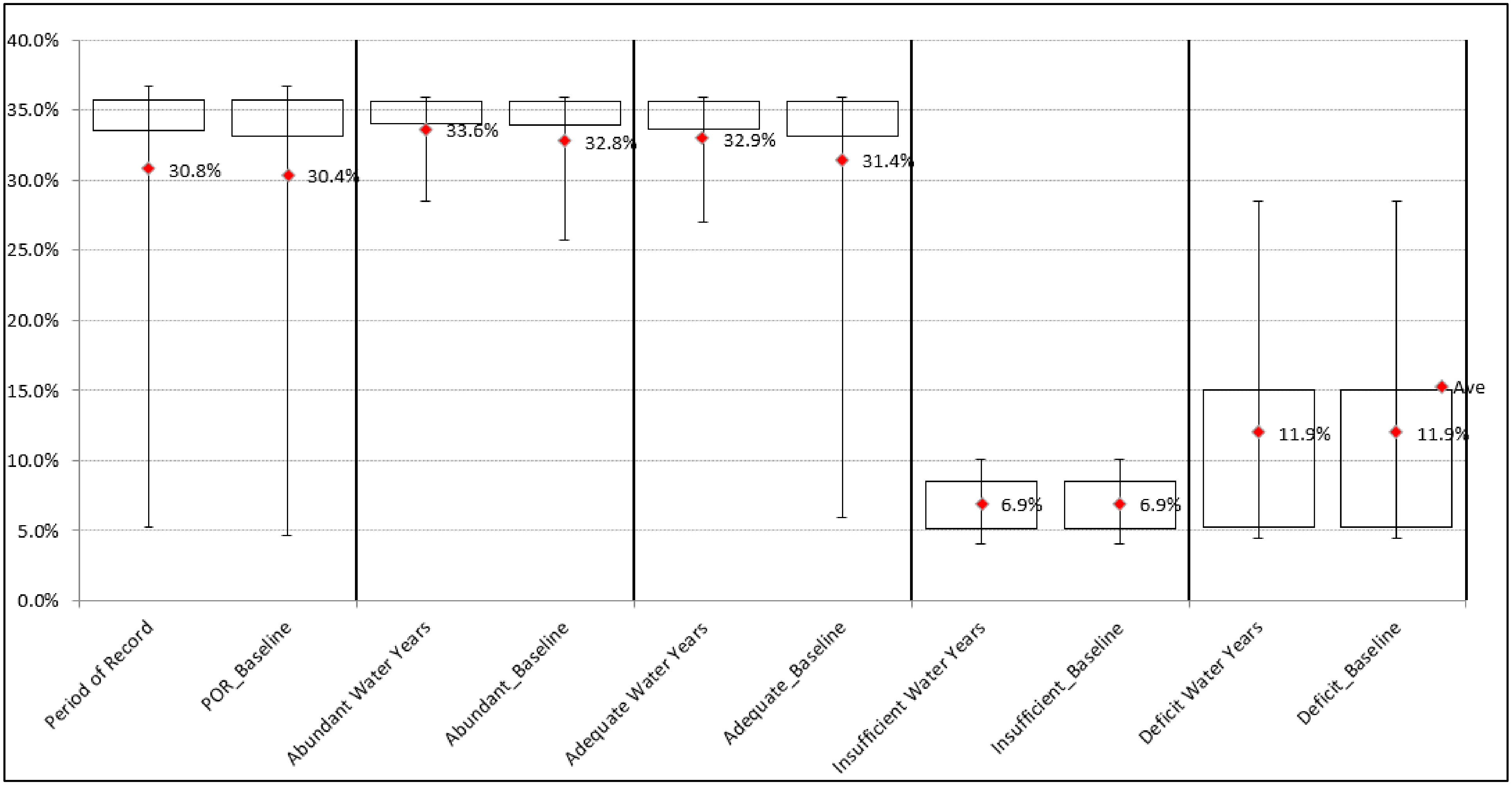

Figure 2-2. Detroit Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Detroit for juvenile spring Chinook sub-yearlings under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

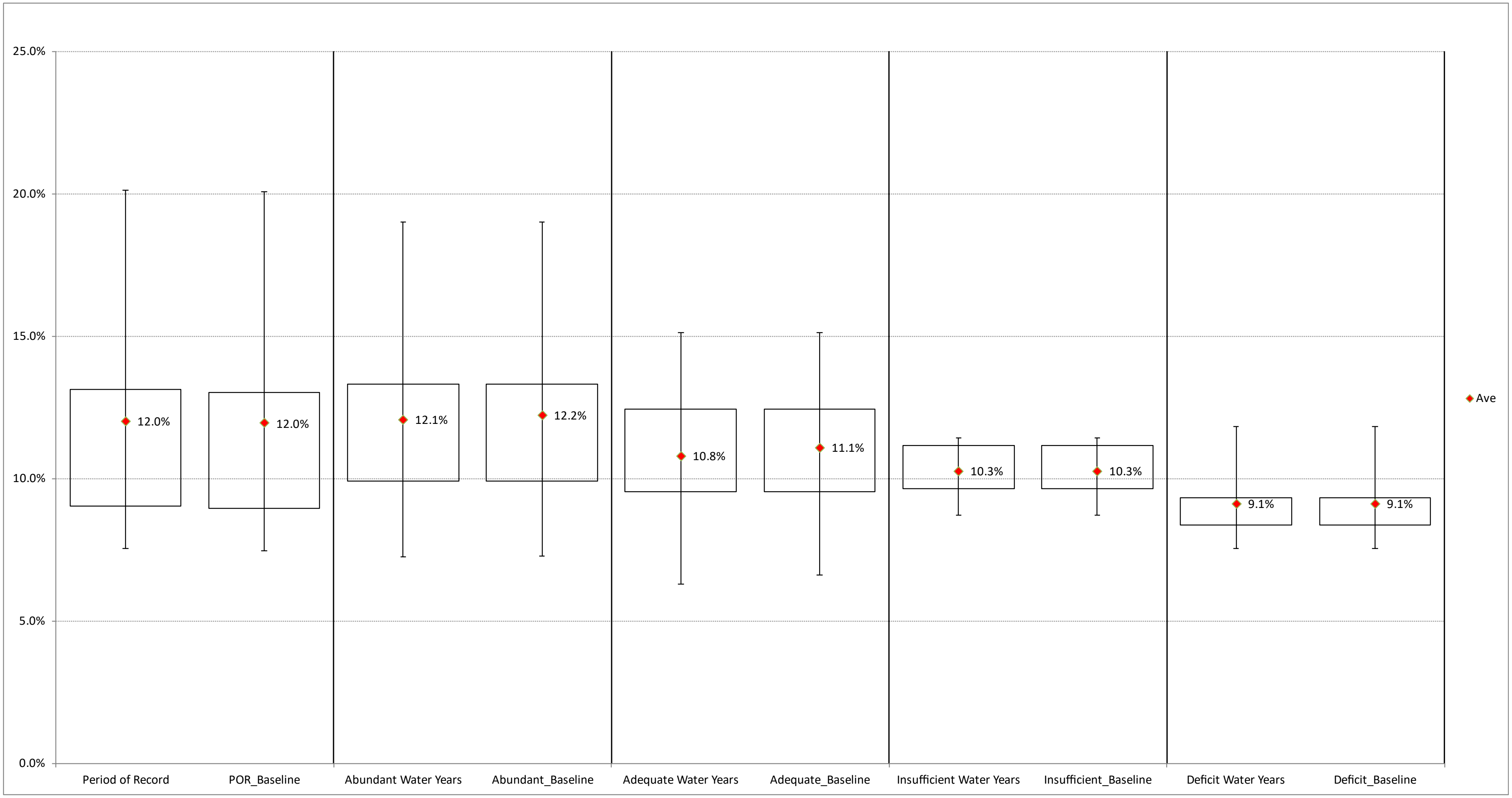

Figure 2-3. Detroit Juvenile Spring Chinook Yearlings Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Detroit for juvenile spring Chinook yearlings under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

South Santiam - Foster

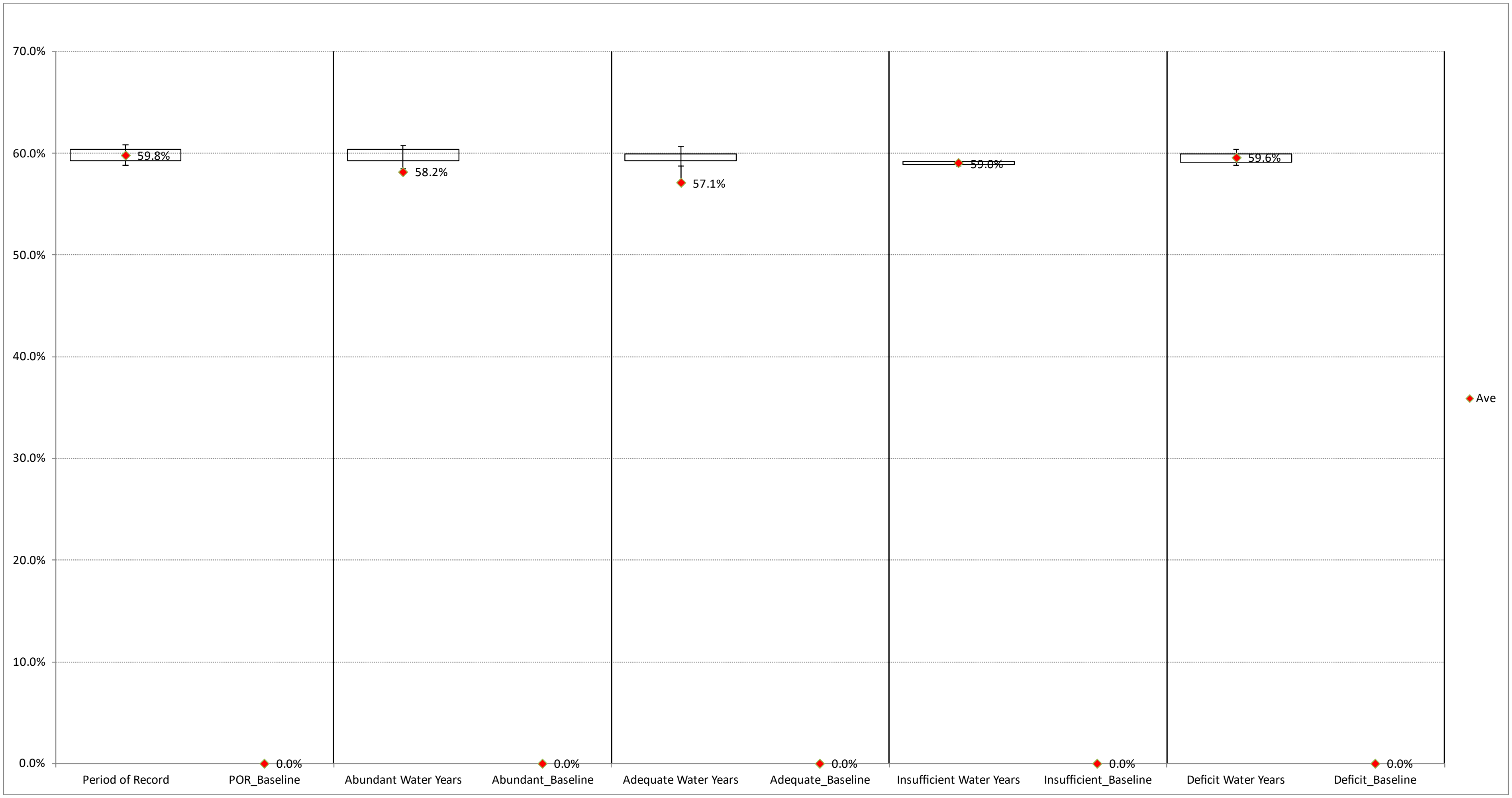

Figure 2-4. Foster Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Foster for juvenile spring Chinook fry under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

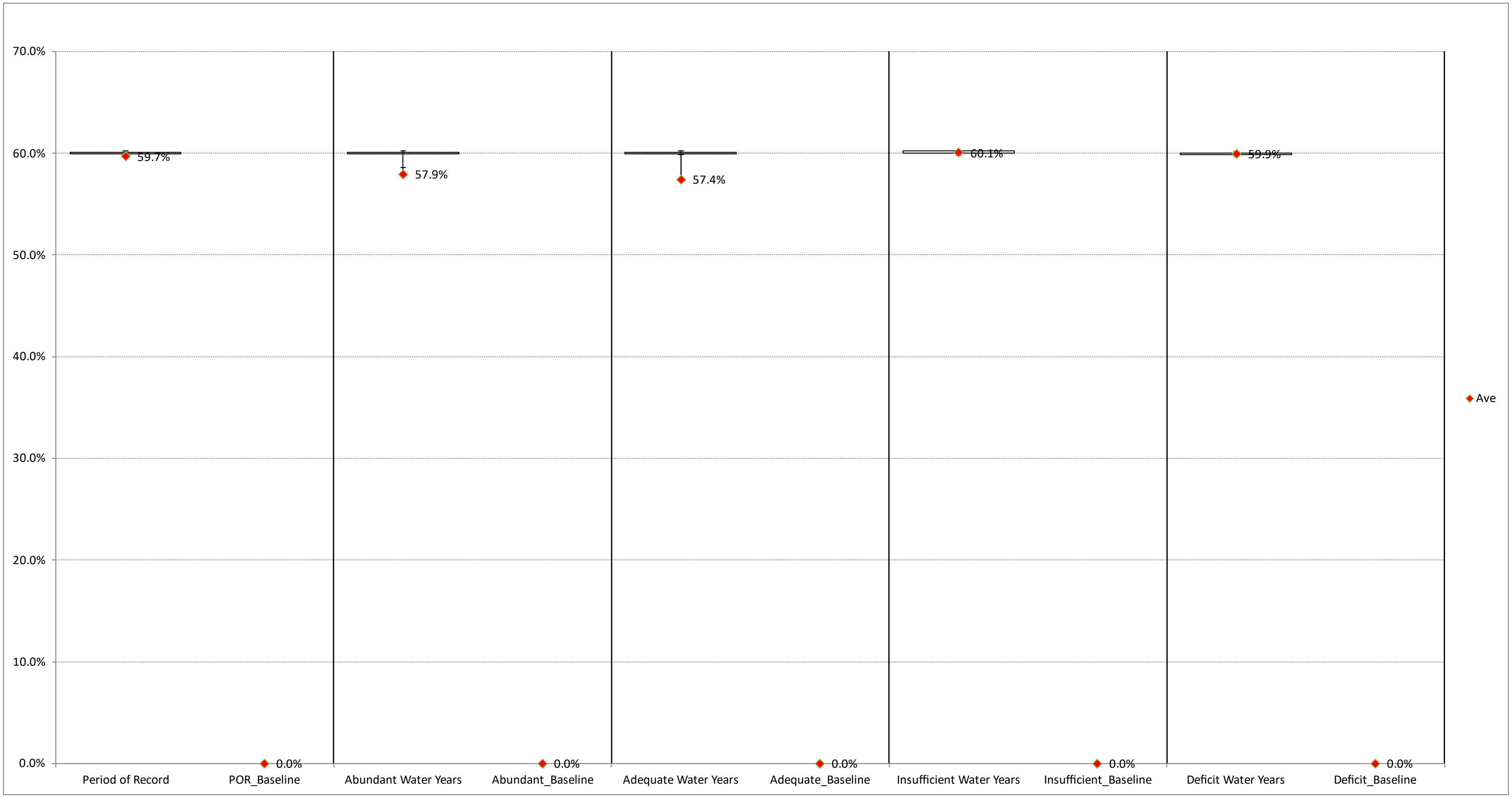

Figure 2-5. Foster Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Foster for juvenile spring Chinook sub-yearlings under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

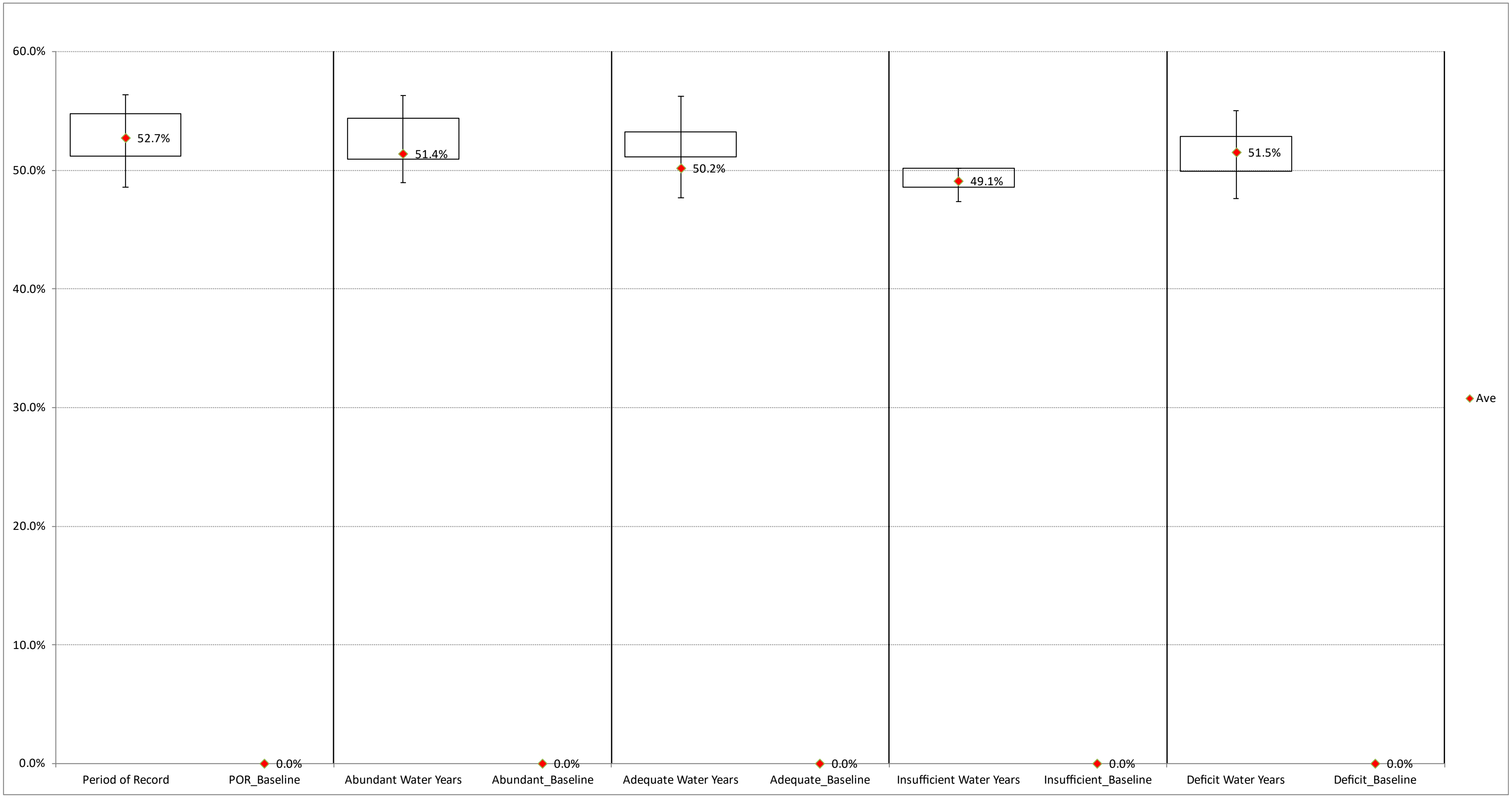

Figure 2-6. Foster Juvenile Spring Chinook Yearlings Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Foster for juvenile spring Chinook yearlings under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.* **South Santiam – Green Peter**

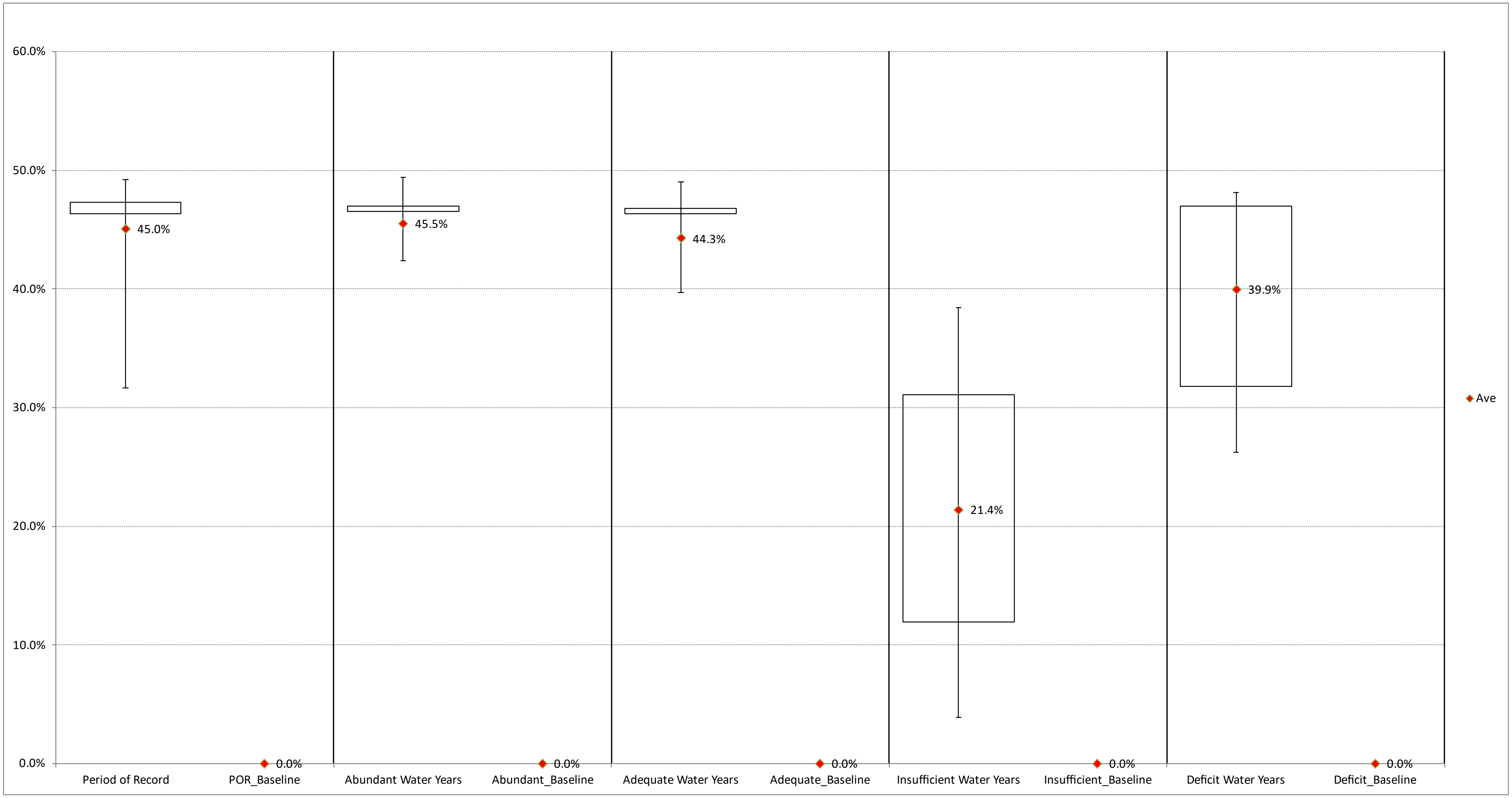

Figure 2-7. Green Peter Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Green Peter for juvenile spring Chinook fry under the No Action Alternative. The mean is given by the point estimate (filled dot).*

Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.

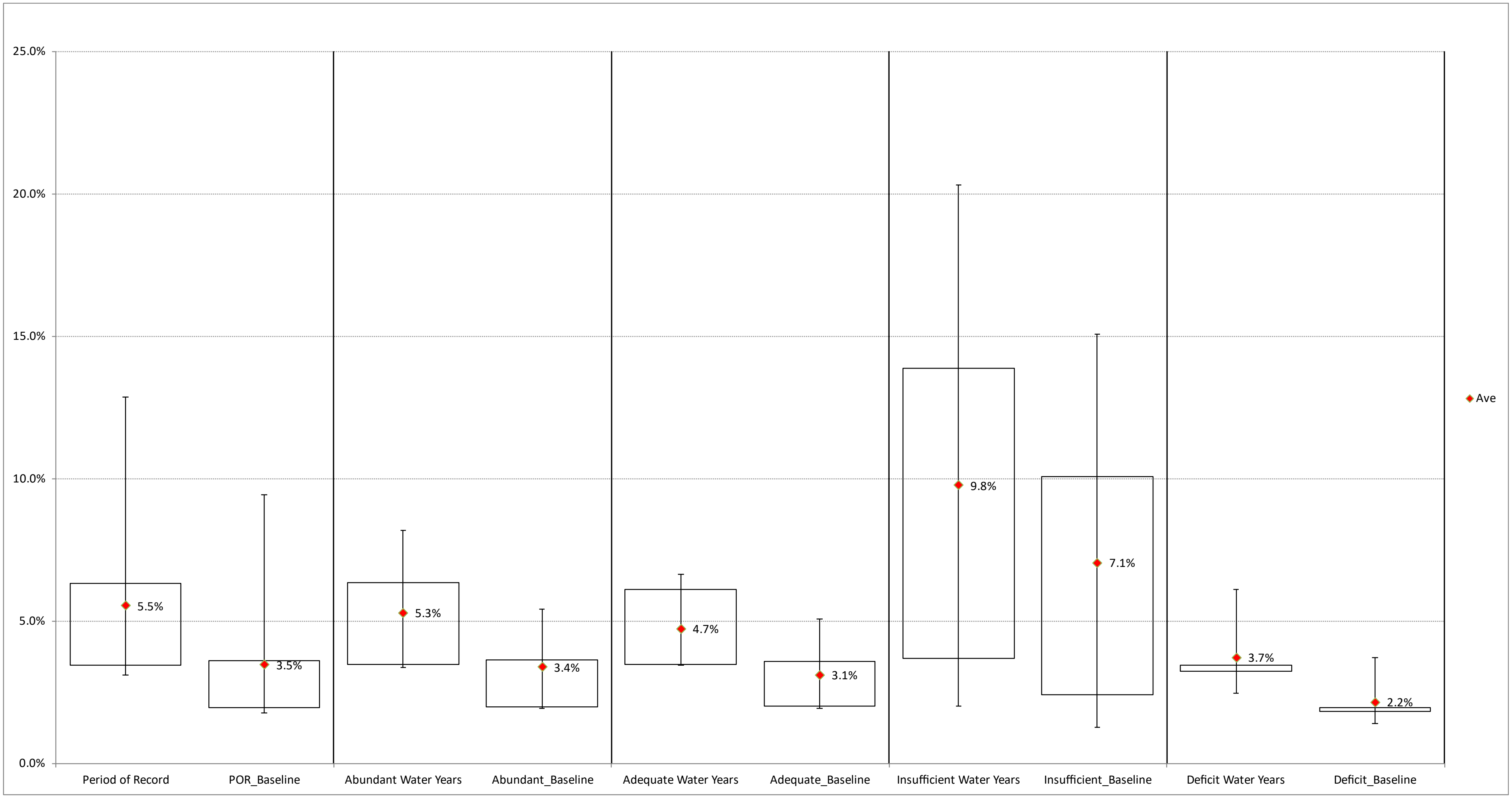

Figure 2-8. Green Peter Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Green Peter for juvenile spring Chinook sub-yearlings under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

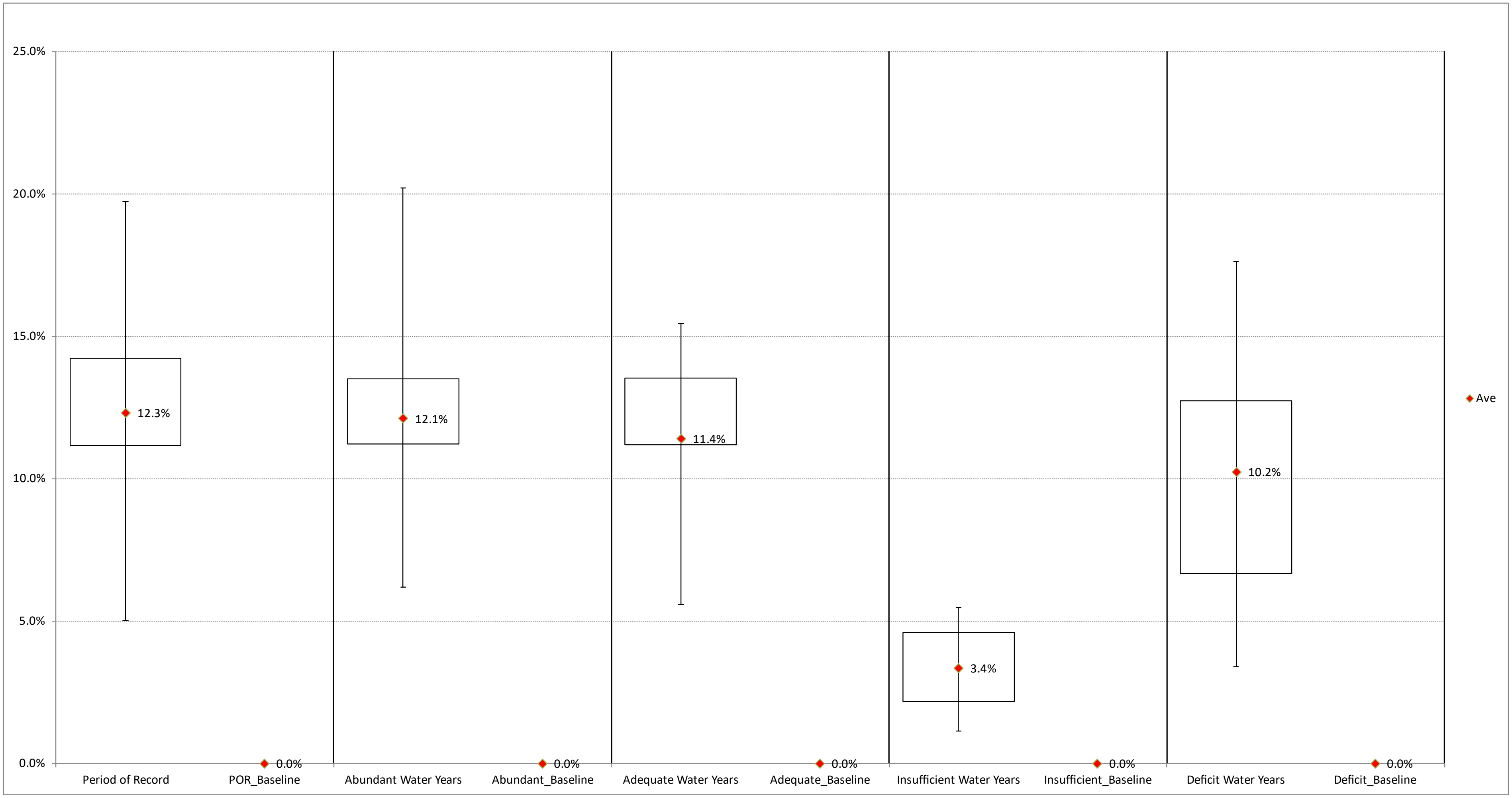

Figure 2-9. Green Peter Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Green Peter for juvenile spring Chinook yearlings under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.* **McKenzie – Cougar**

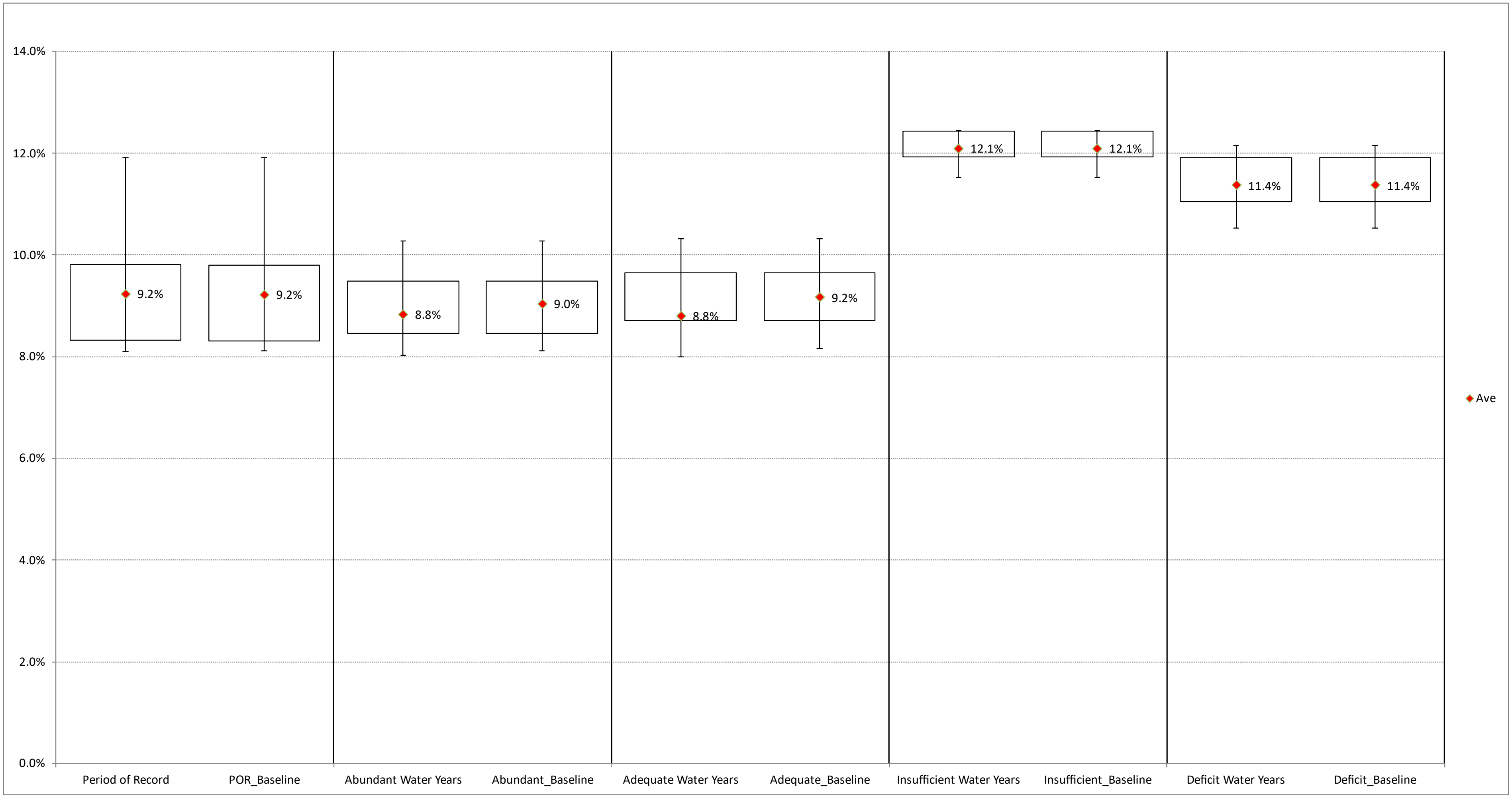

Figure 2-10. Cougar Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Cougar for juvenile spring Chinook fry under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

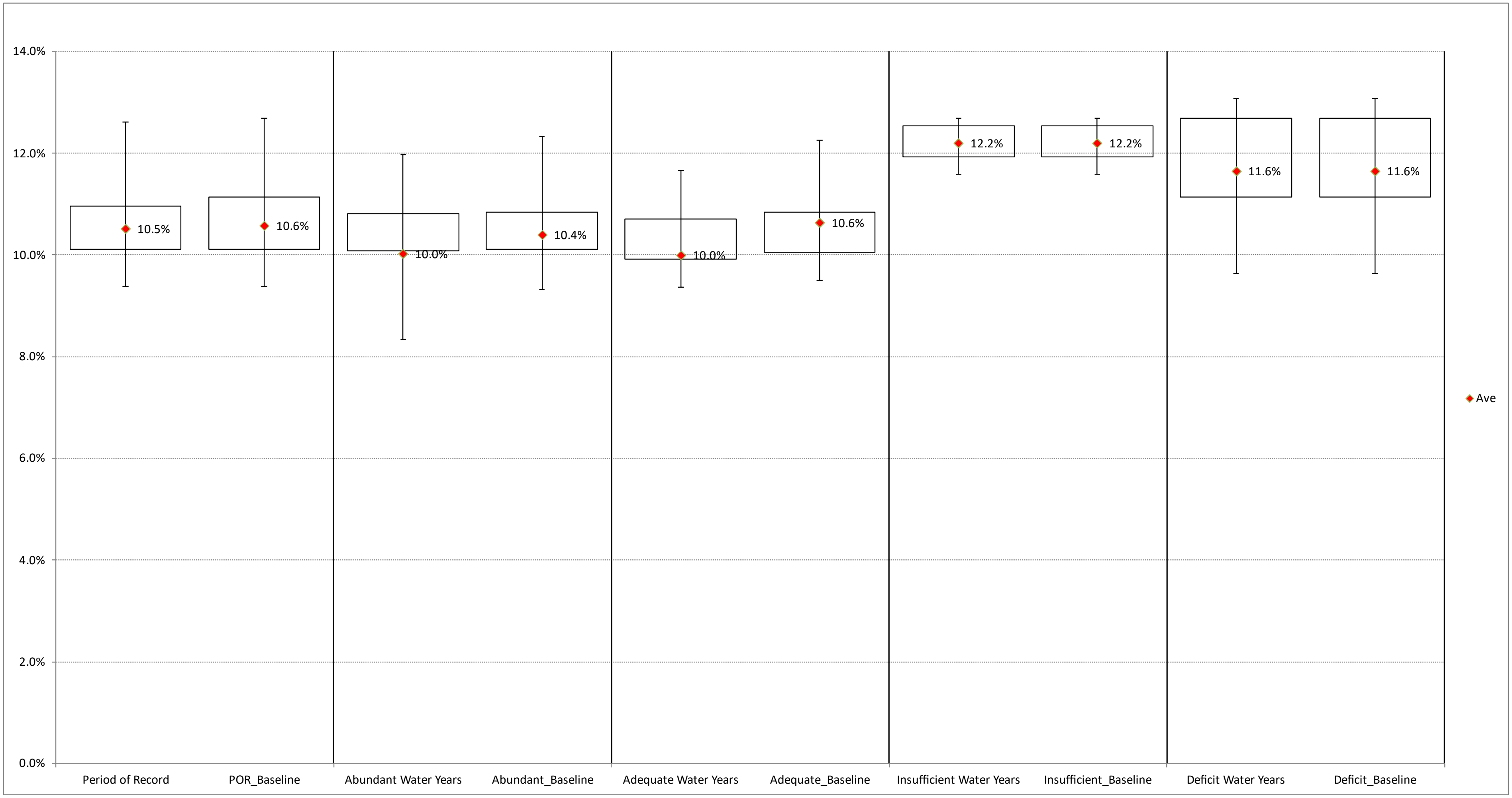

Figure 2-11. Cougar Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Cougar for juvenile spring Chinook sub-yearlings under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

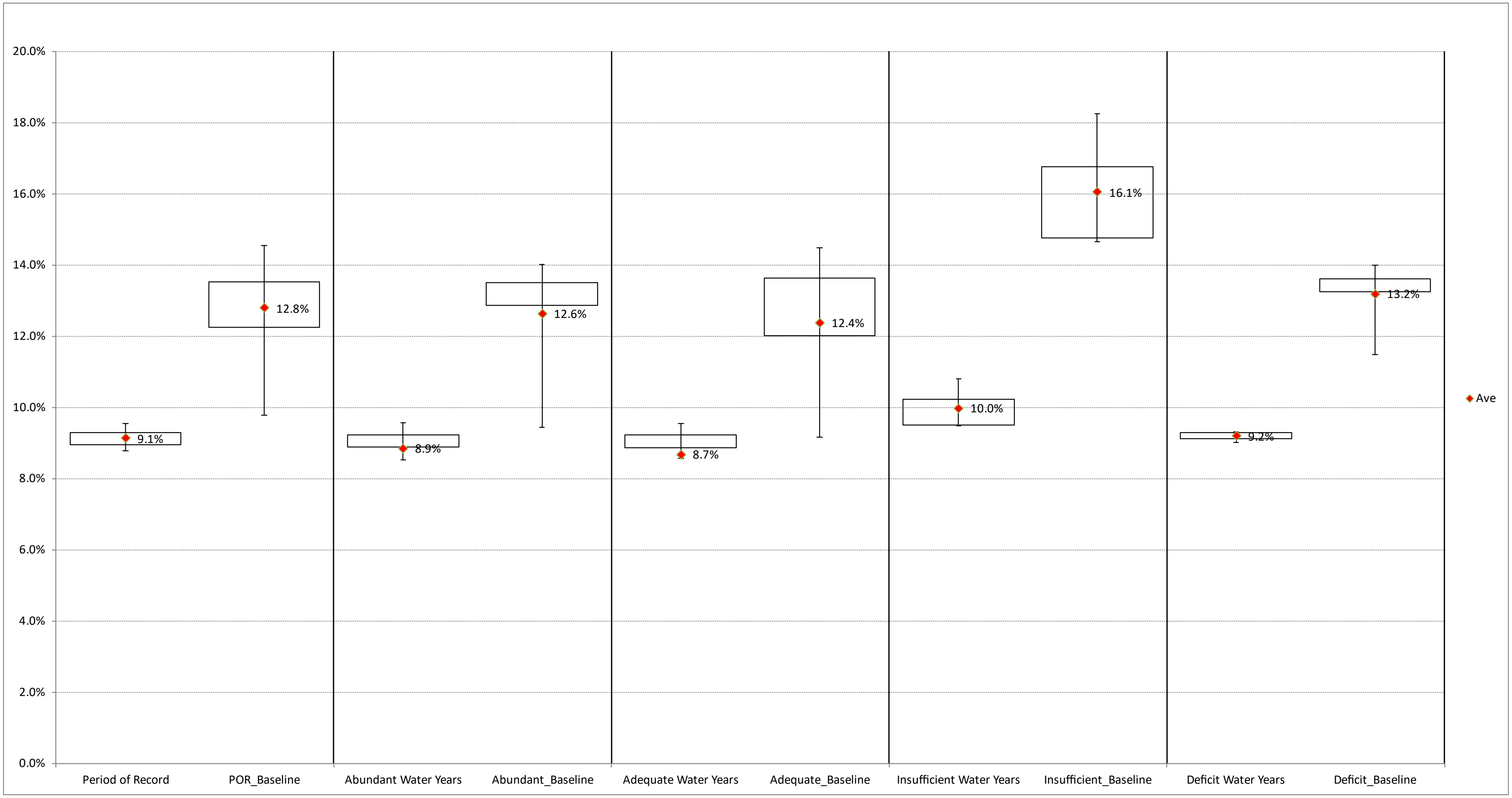

Figure 2-12. Cougar Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Cougar for juvenile spring Chinook yearlings under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.1.4 Middle Fork - Lookout Point

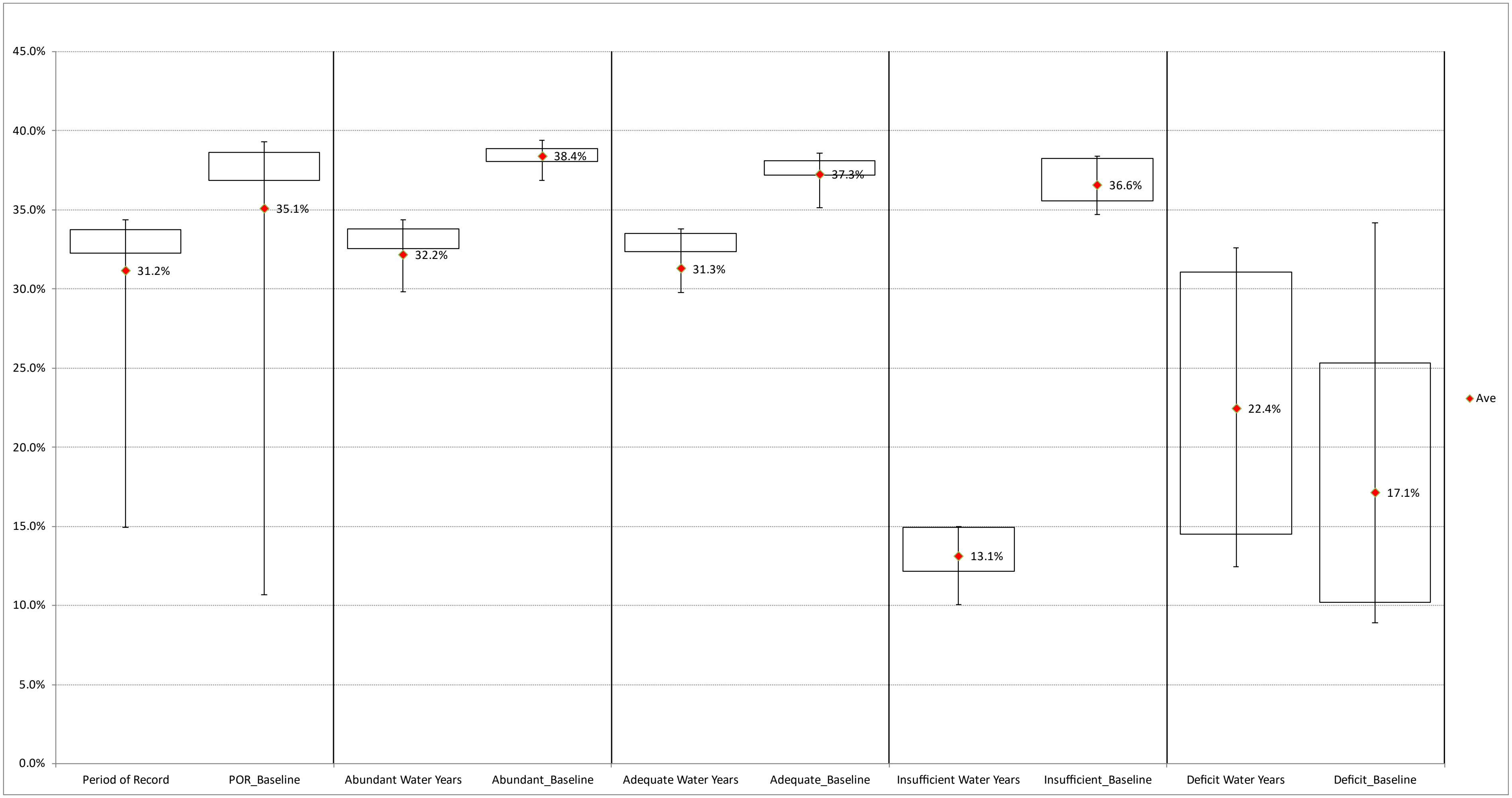

Figure 2-13. Lookout Point Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook fry under the No Action Alternative. The mean is given by the point estimate (filled*

dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.

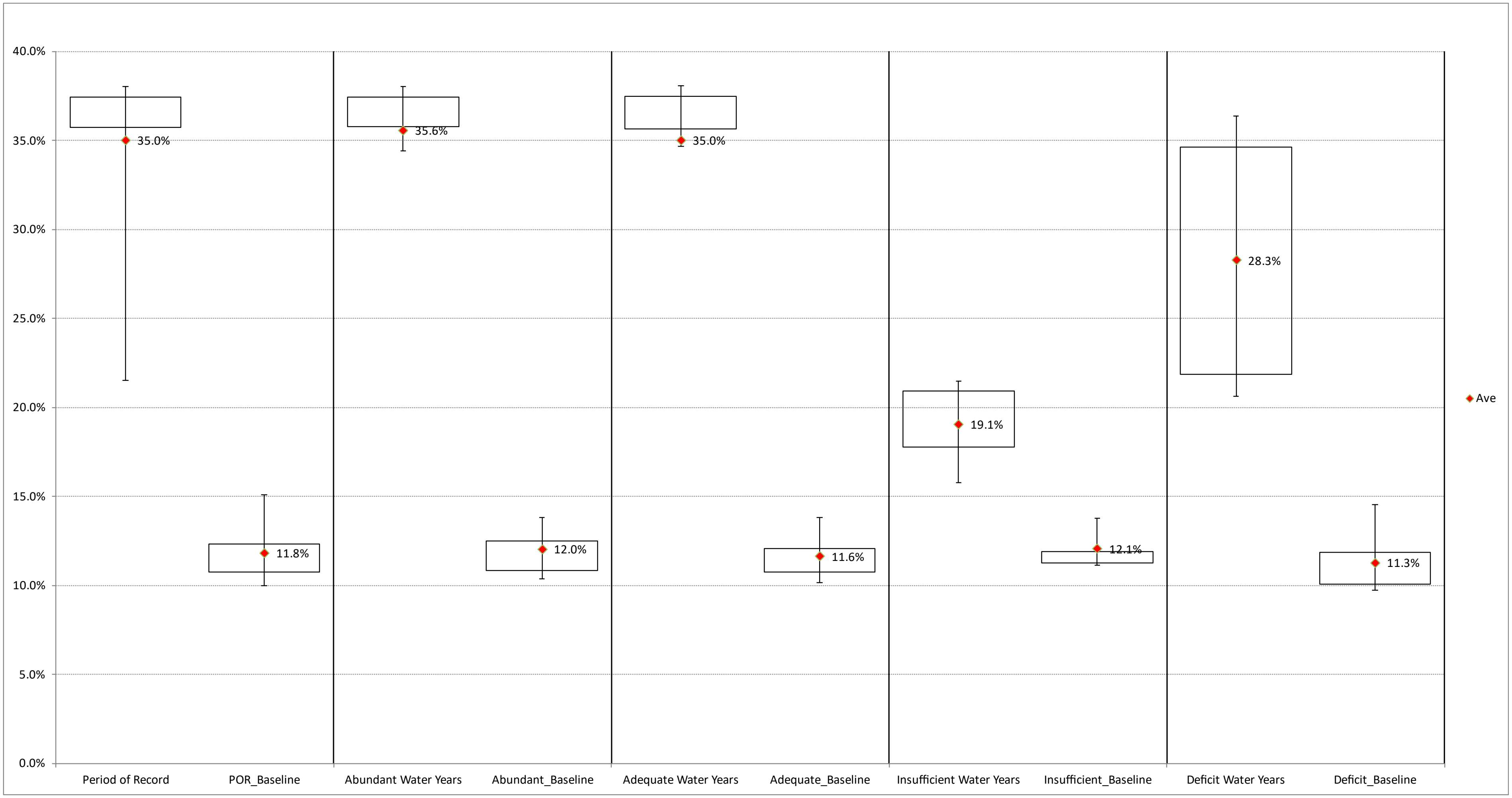

Figure 2-14. Lookout Point Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook sub-yearlings under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

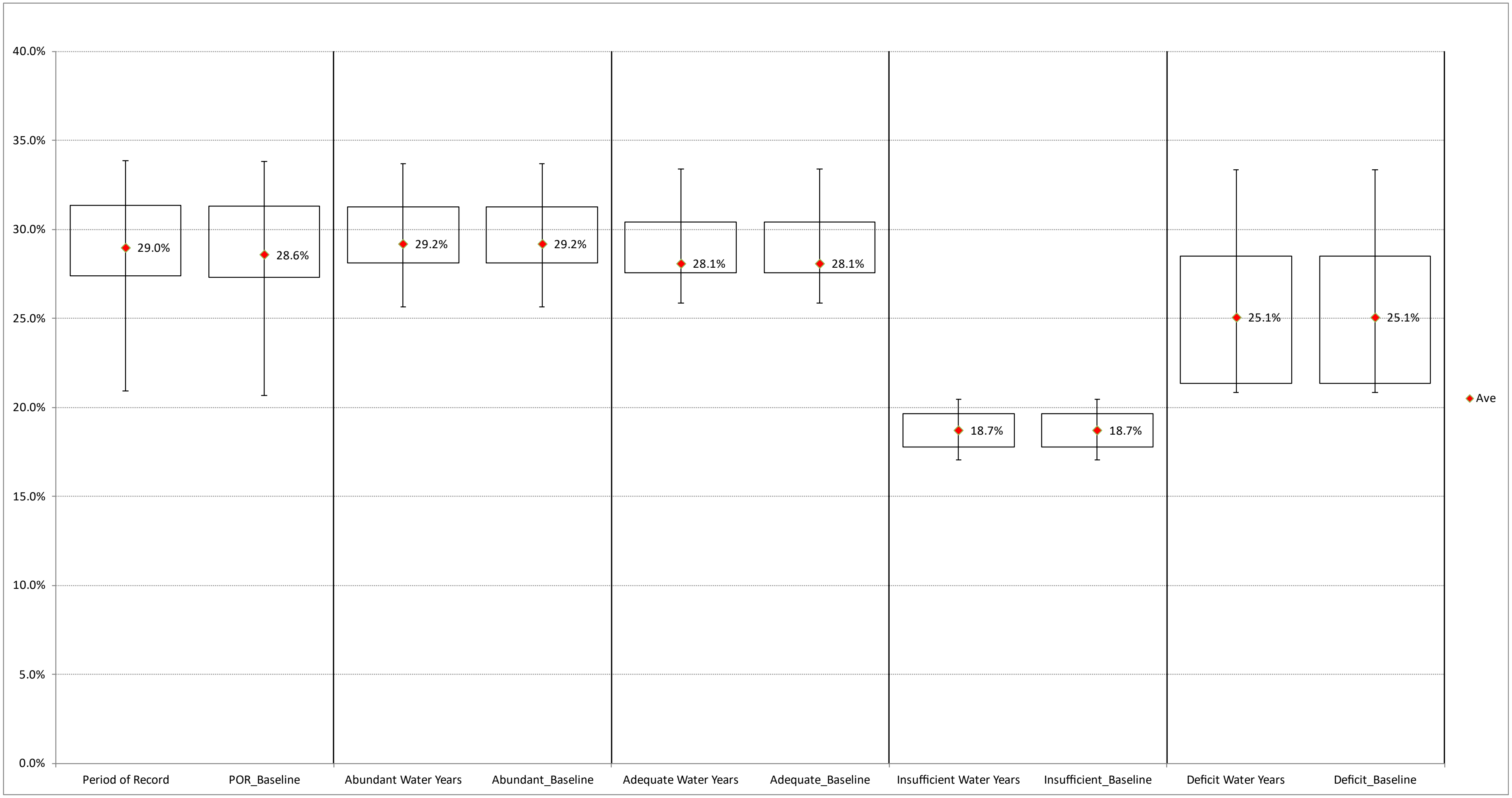

Figure 2-15. Lookout Point Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook yearlings under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.1.5 Middle Fork- Hills Creek

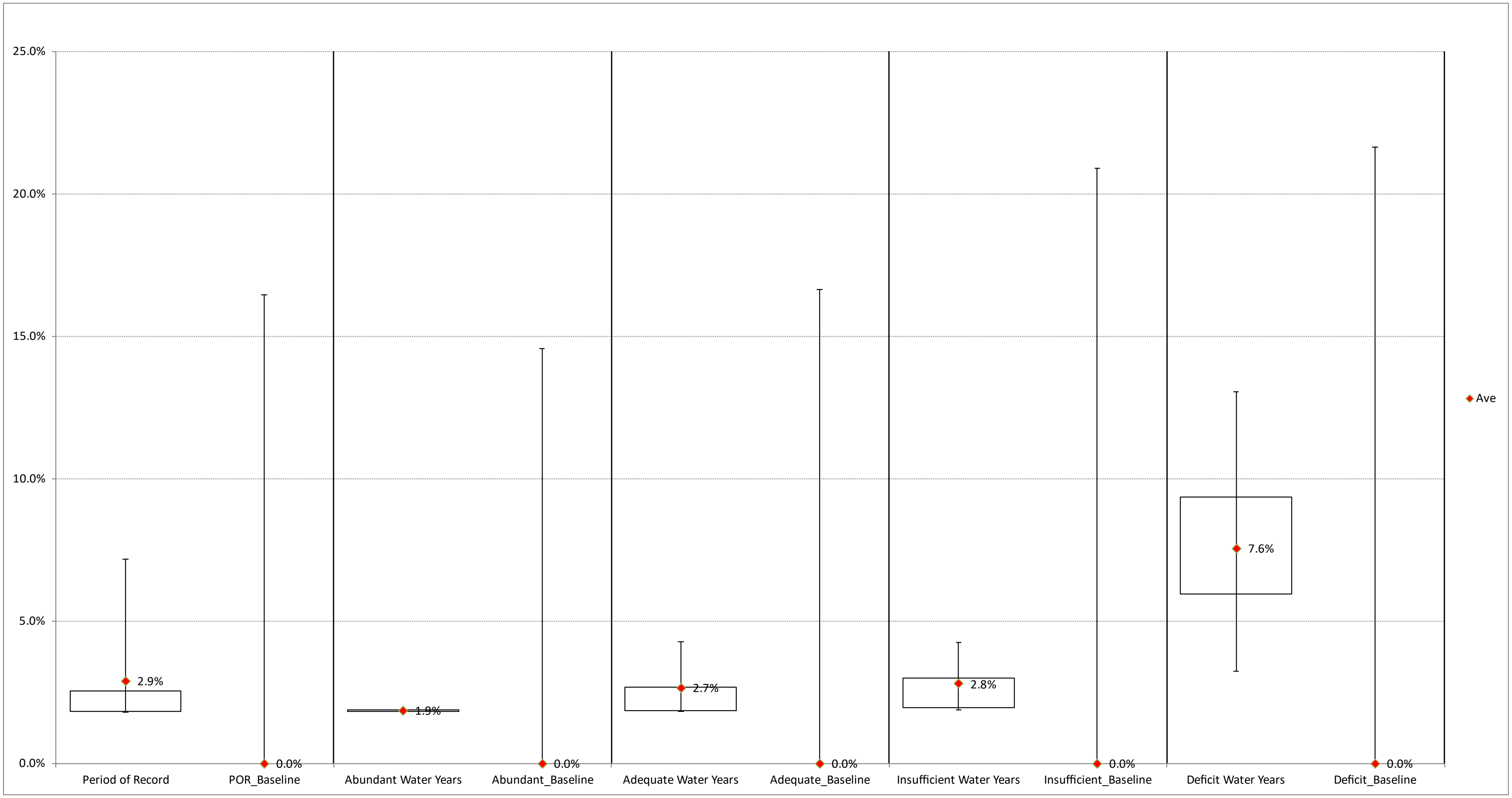

Figure 2-16. Downstream dam passage survival at Hills Creek for juvenile spring Chinook fry under the No Action Alternative. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook fry under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

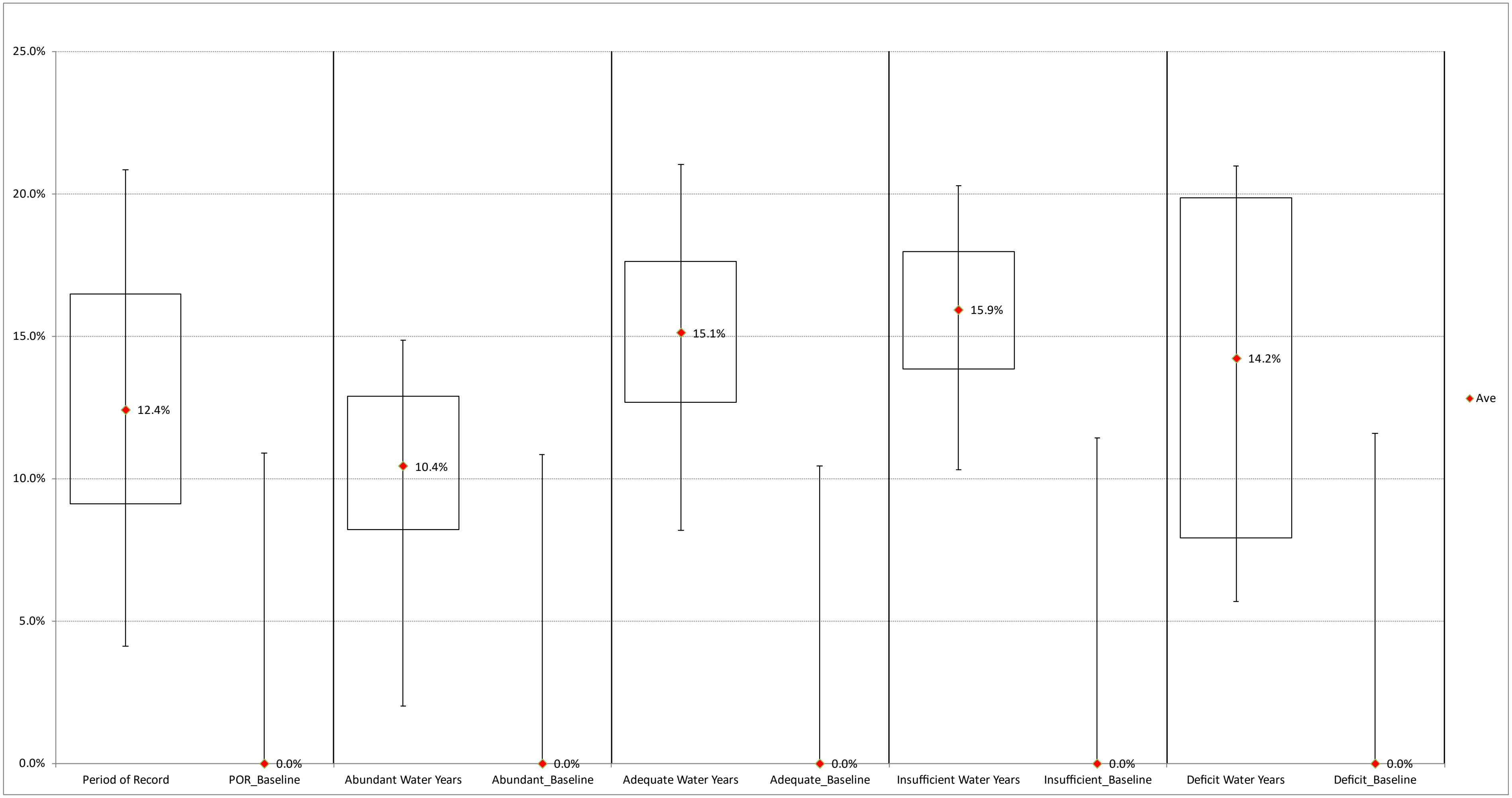

Figure 2-17. Hills Creek Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook sub-yearlings under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

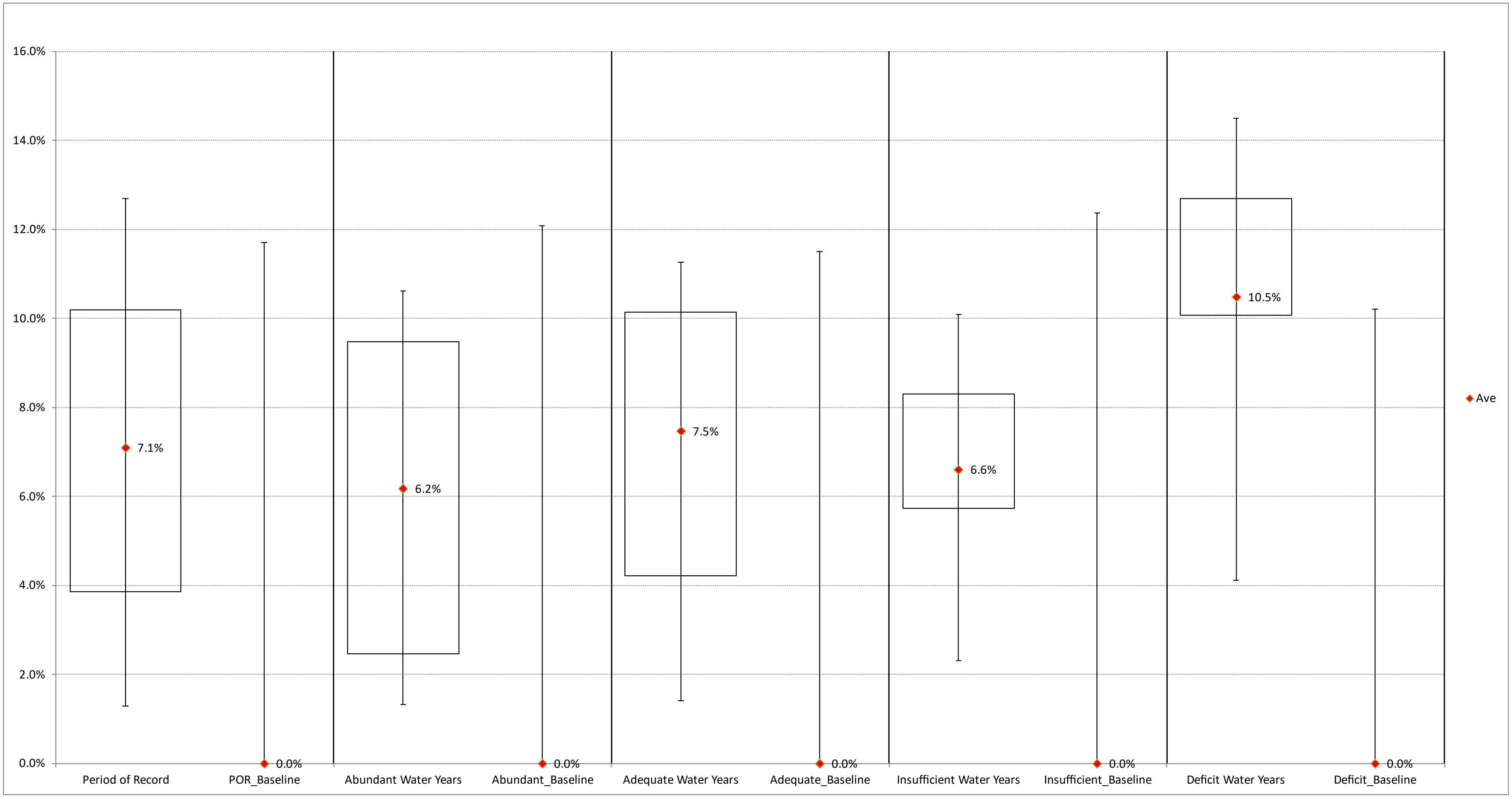

Figure 2-18. Hills Creek Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under the No Action Alternative. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook yearlings under the No Action Alternative. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

CHINOOK

2.2 CHINOOK ALTERNATIVE 1

2.2.1 North Santiam - Detroit

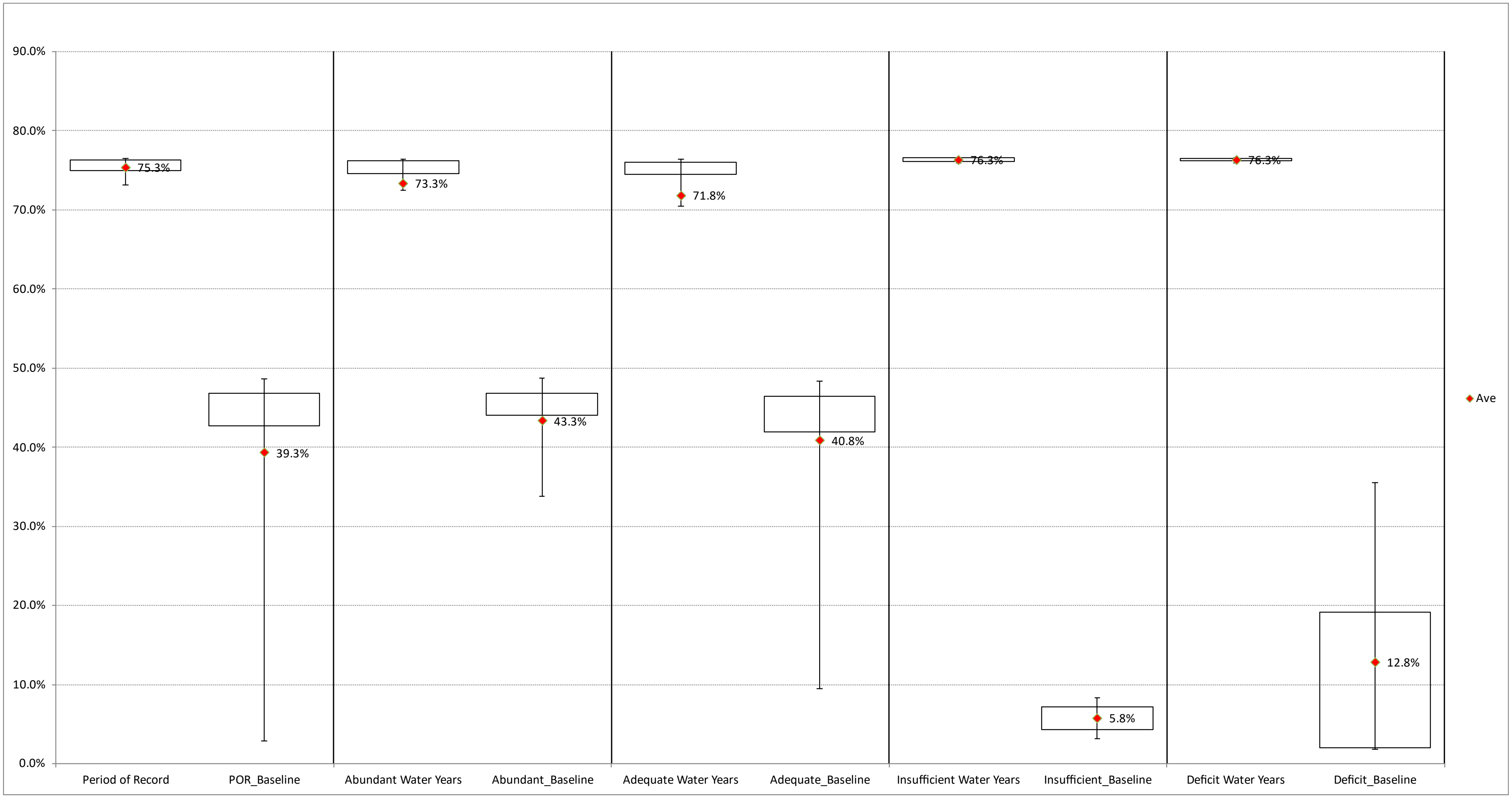

Figure 2-19. Detroit Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 1. *The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

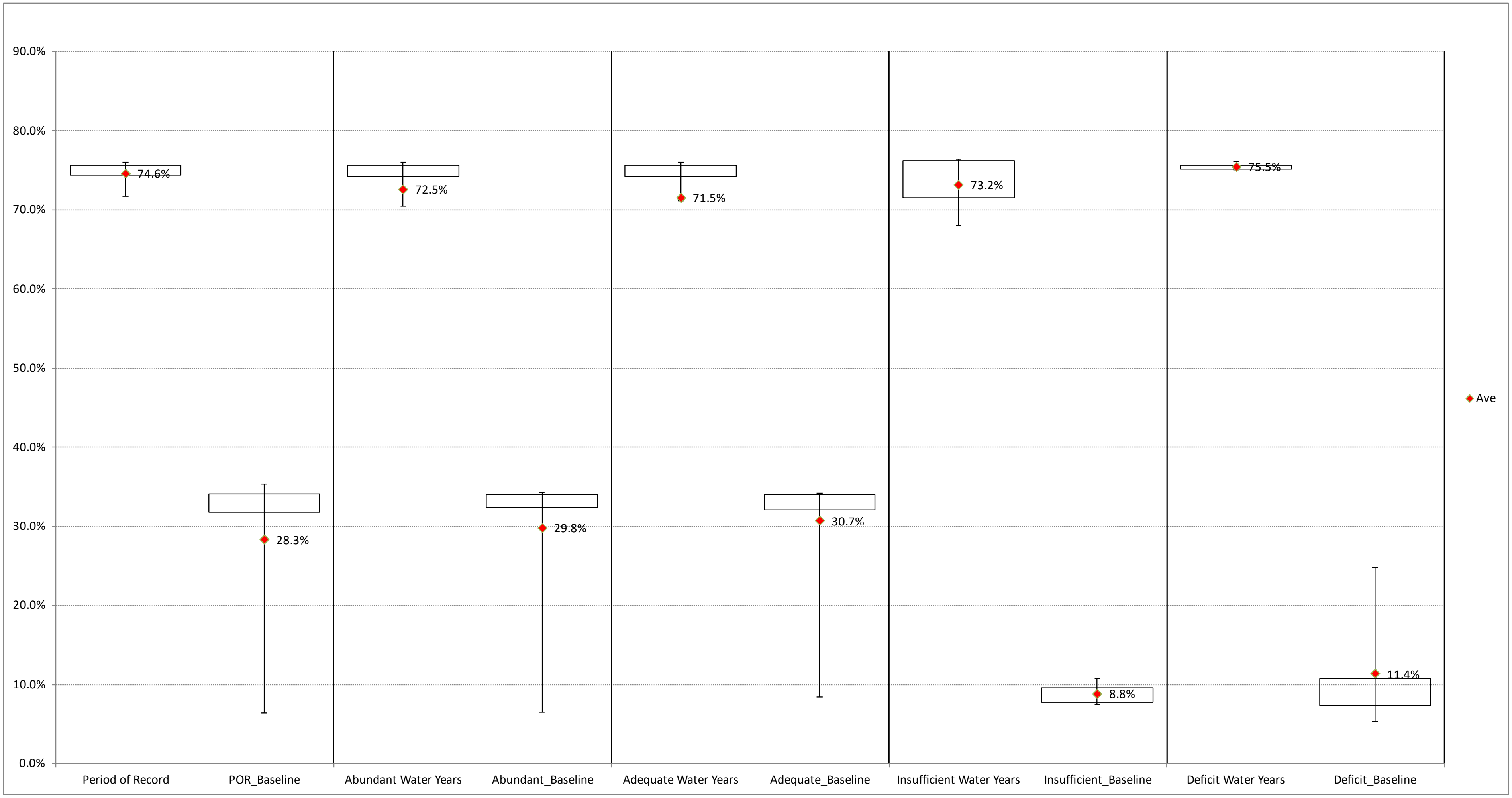

Figure 2-20. Detroit Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Detroit for juvenile spring Chinook sub-yearling under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

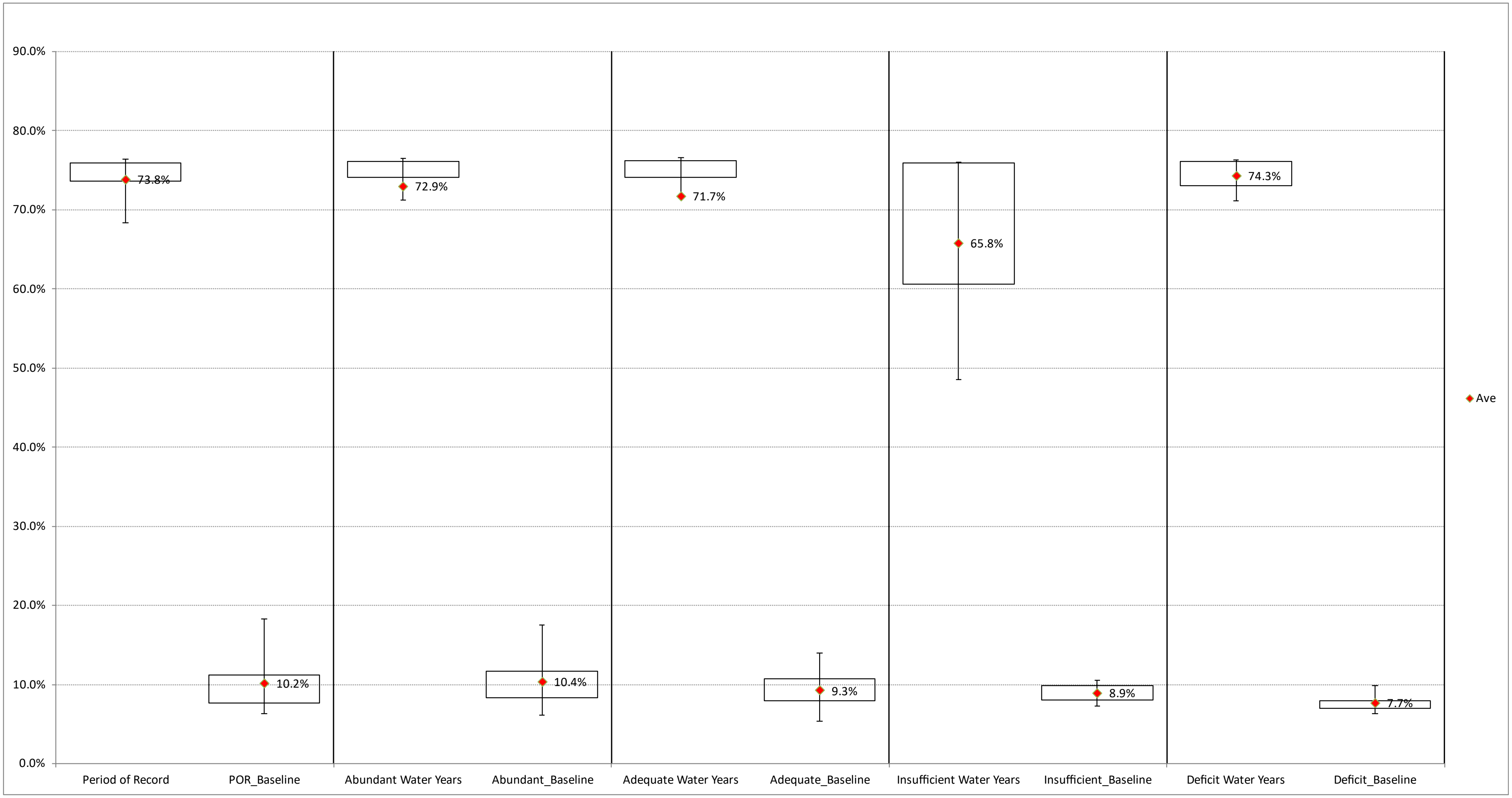

Figure 2-21. Detroit Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Detroit for juvenile spring Chinook yearling under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.2.2 South Santiam - Foster

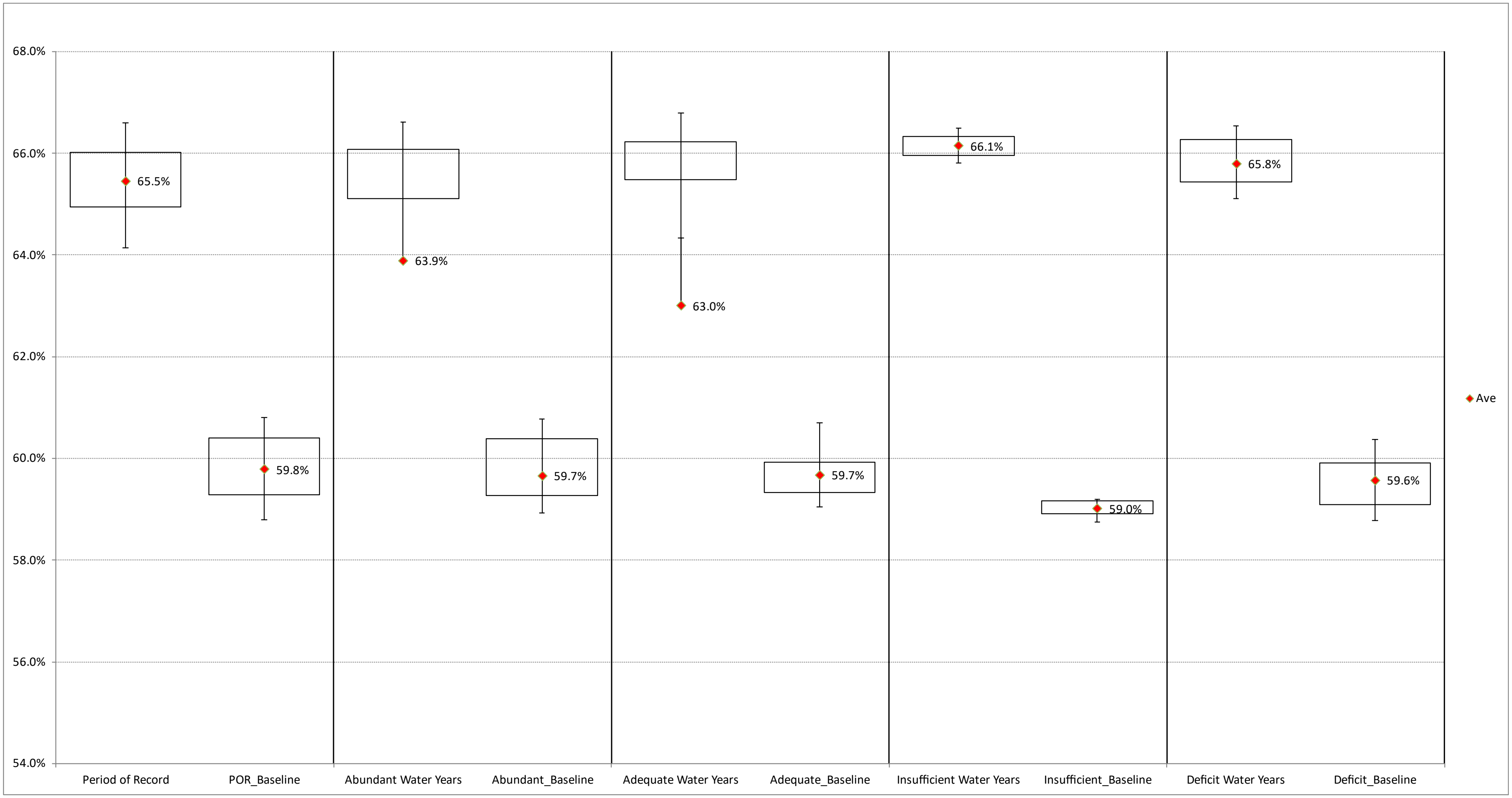

Figure 2-22. Foster juvenile spring Chinook fry Downstream dam passage survival under Alternative 1. *Downstream dam passage survival at Foster for juvenile spring Chinook fry under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

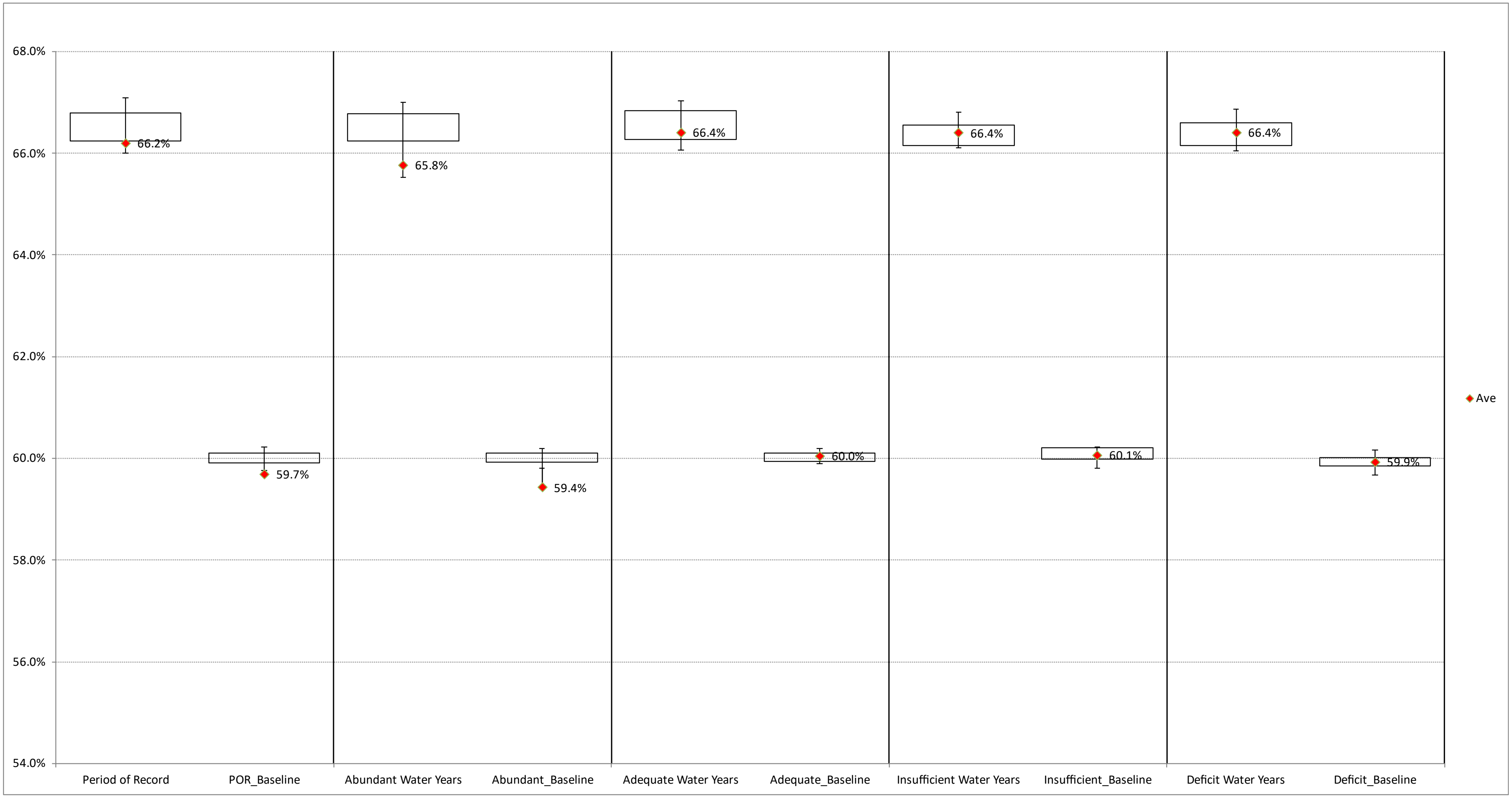

Figure 2-23. Foster Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Foster for juvenile spring Chinook sub-yearlings under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

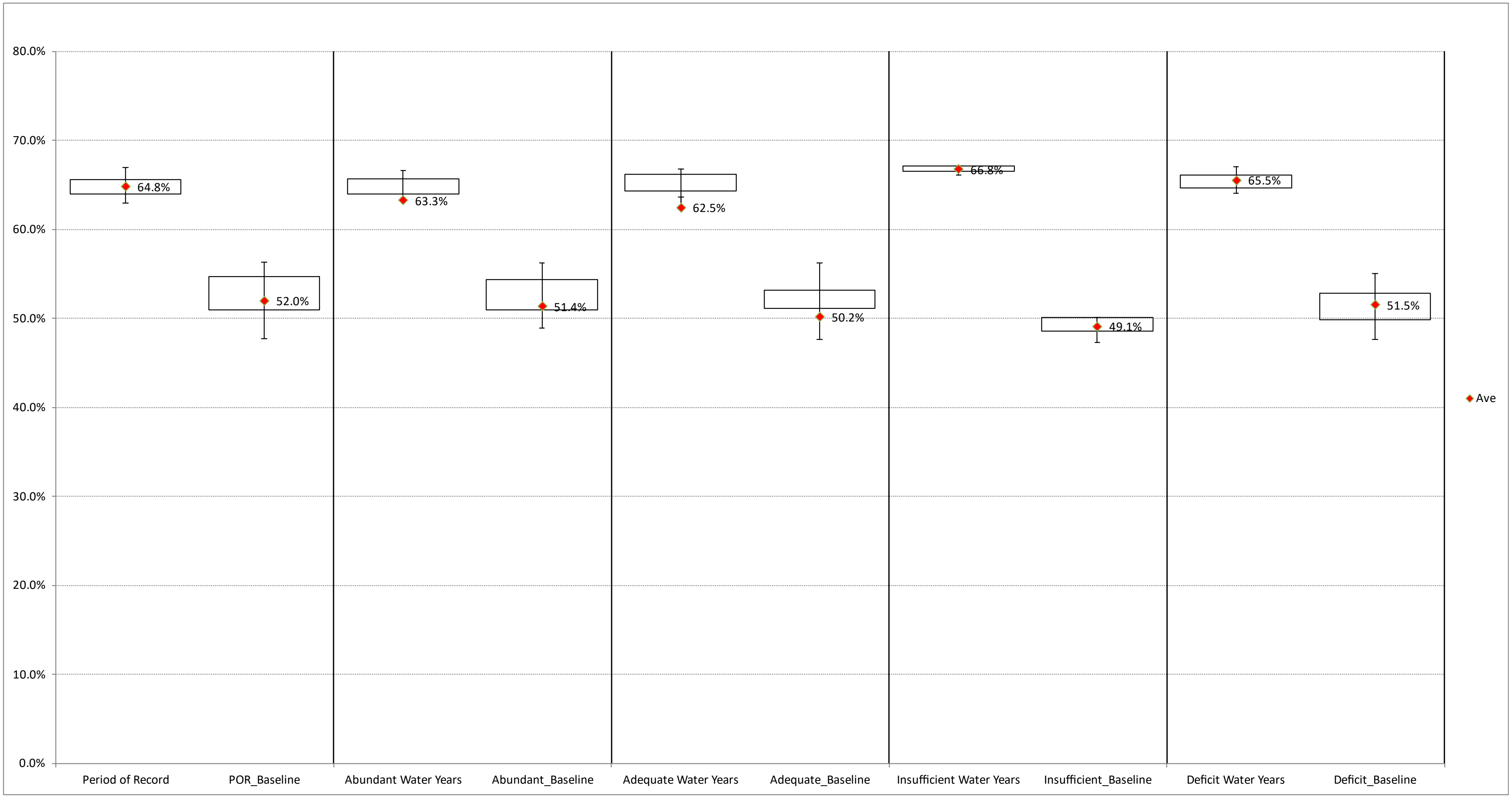

Figure 2-24. Foster Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Foster for juvenile spring Chinook yearlings under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.2.3 South Santiam – Green Peter

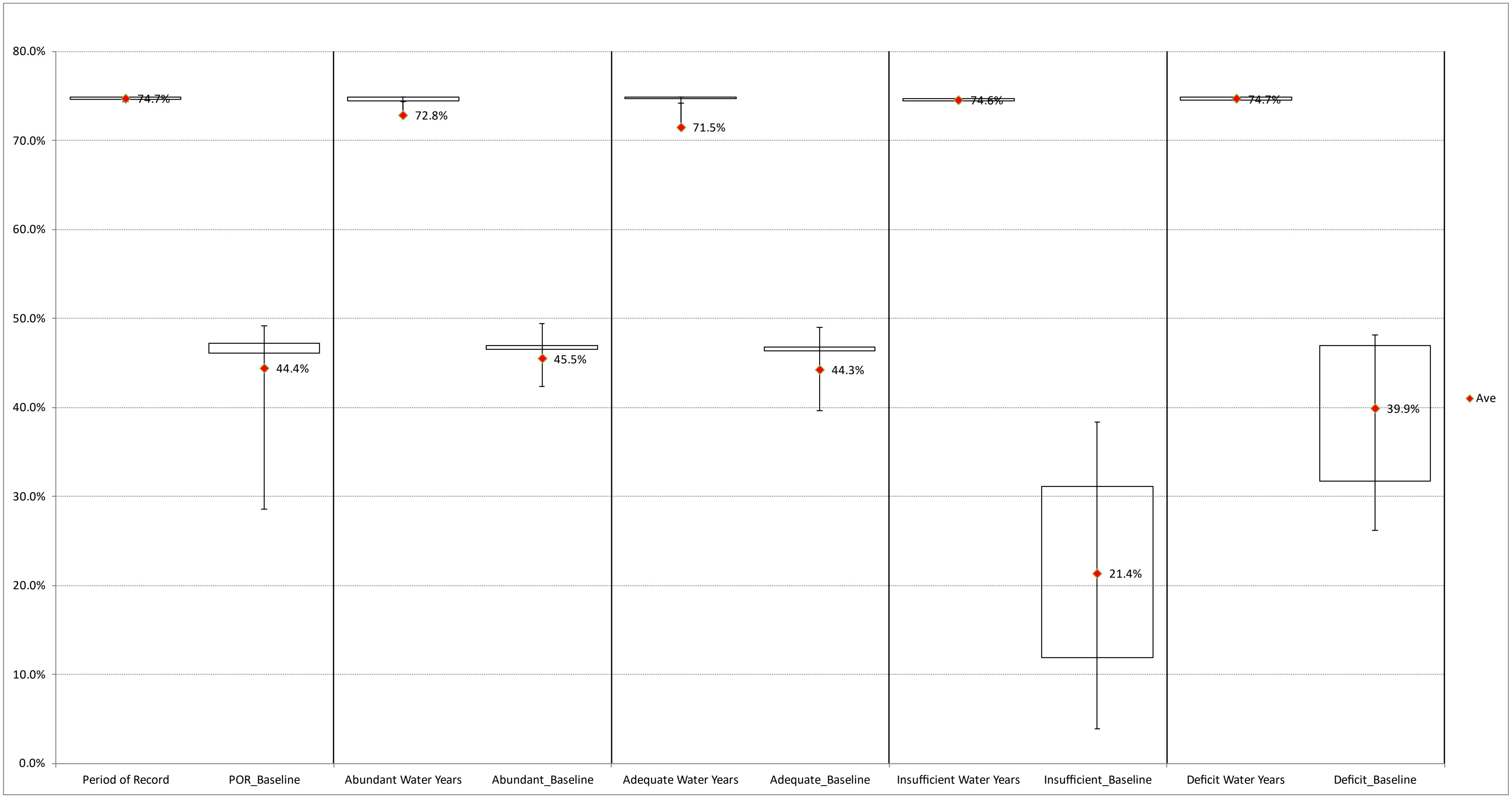

Figure 2-25. Green Peter Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Green Peter for juvenile spring Chinook fry under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

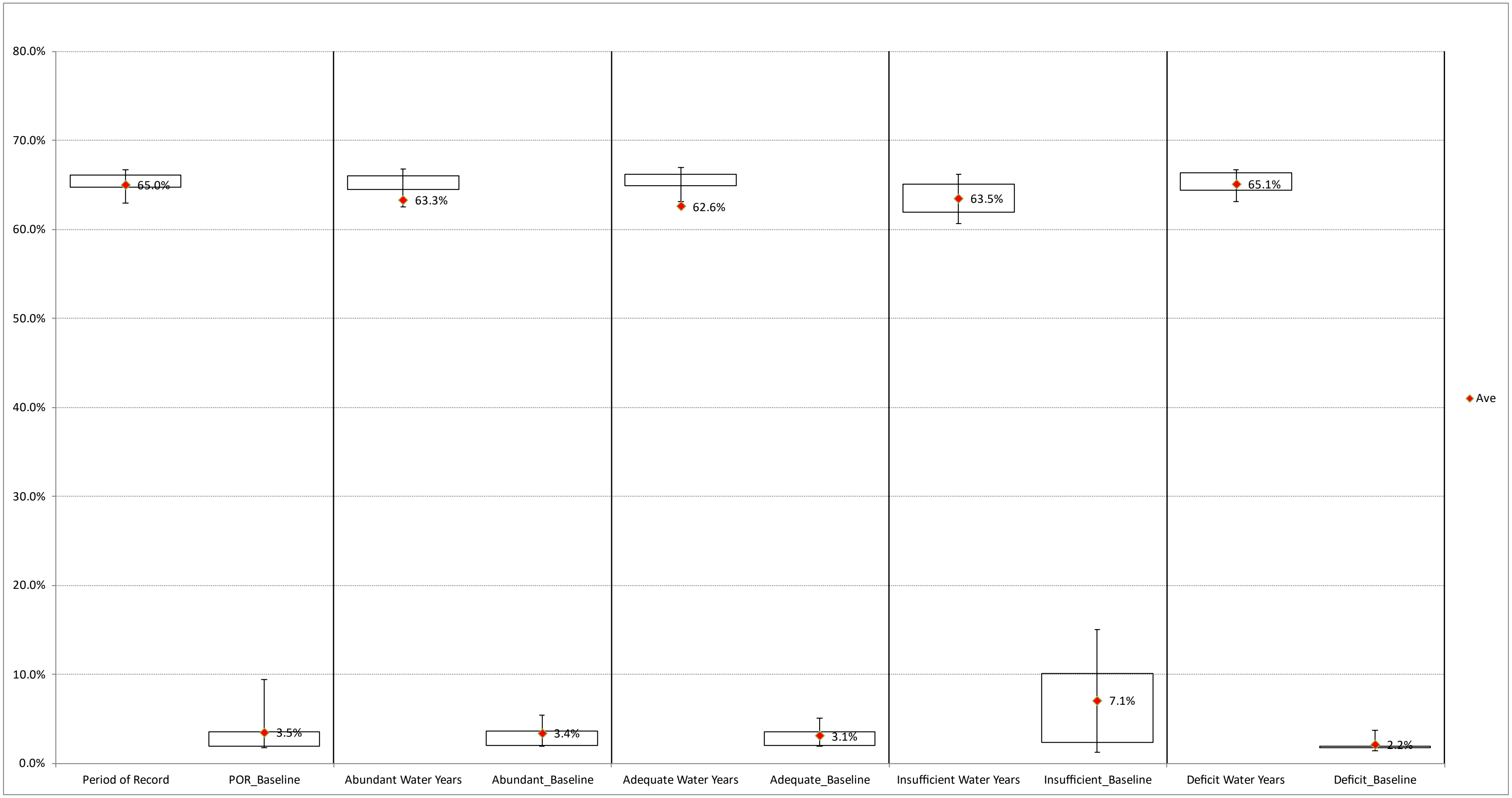

Figure 2-26. Green Peter Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Green Peter for juvenile spring Chinook sub-yearling under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

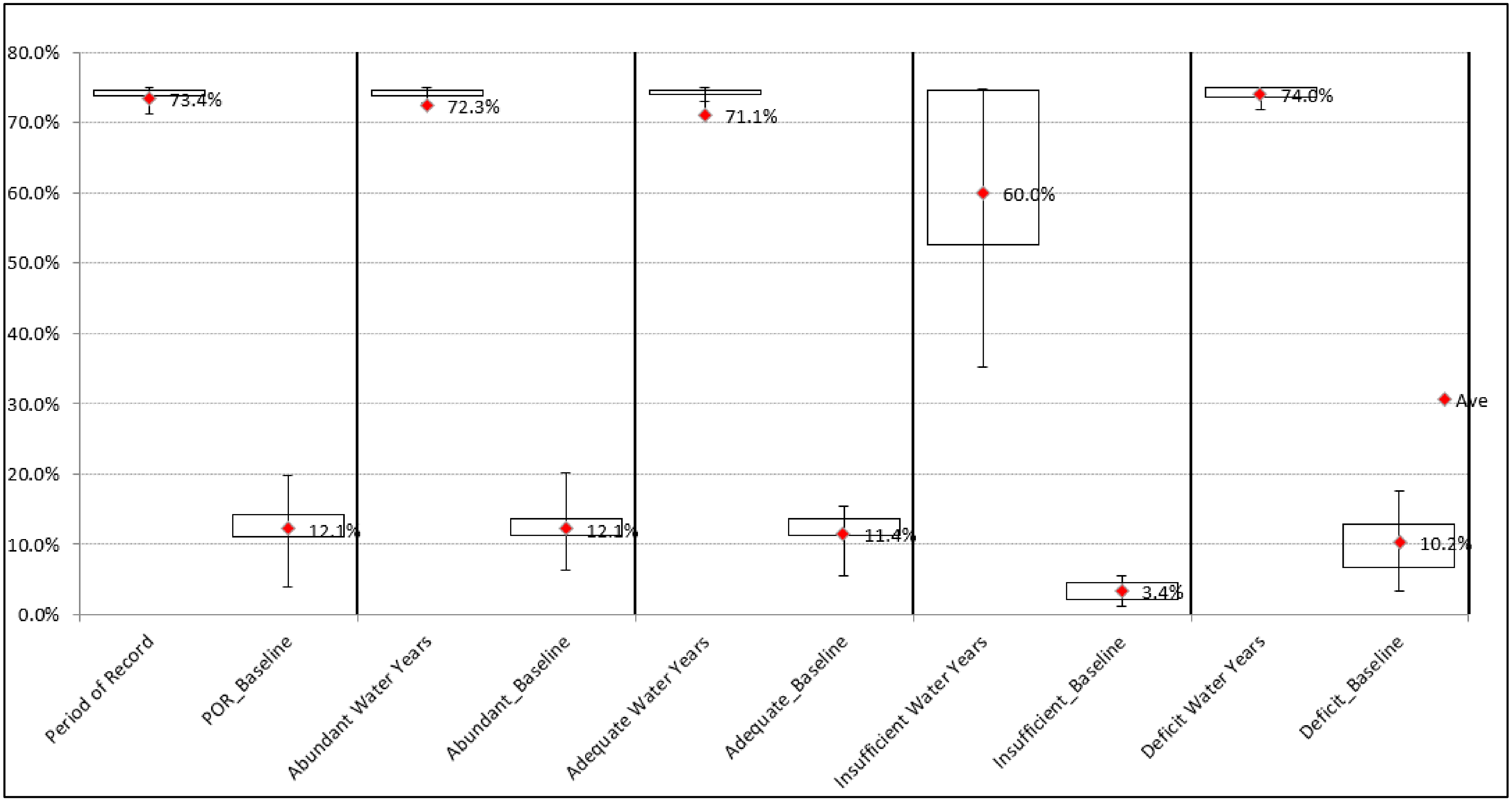

Figure 2-27. Green Peter Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Green Peter for juvenile spring Chinook yearling under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.2.4 McKenzie – Cougar

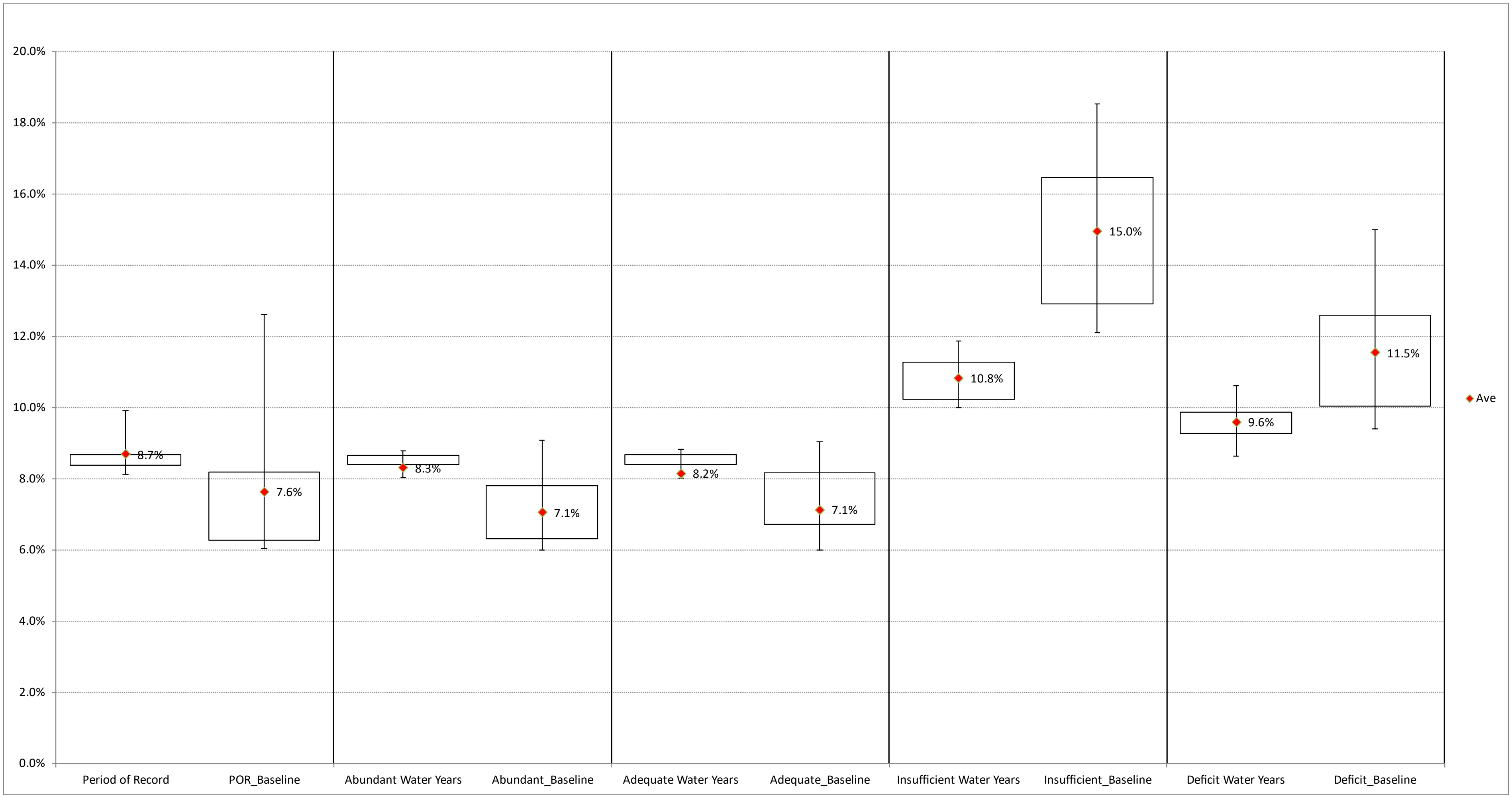

Figure 2-28. Cougar Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Cougar for juvenile spring Chinook fry under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

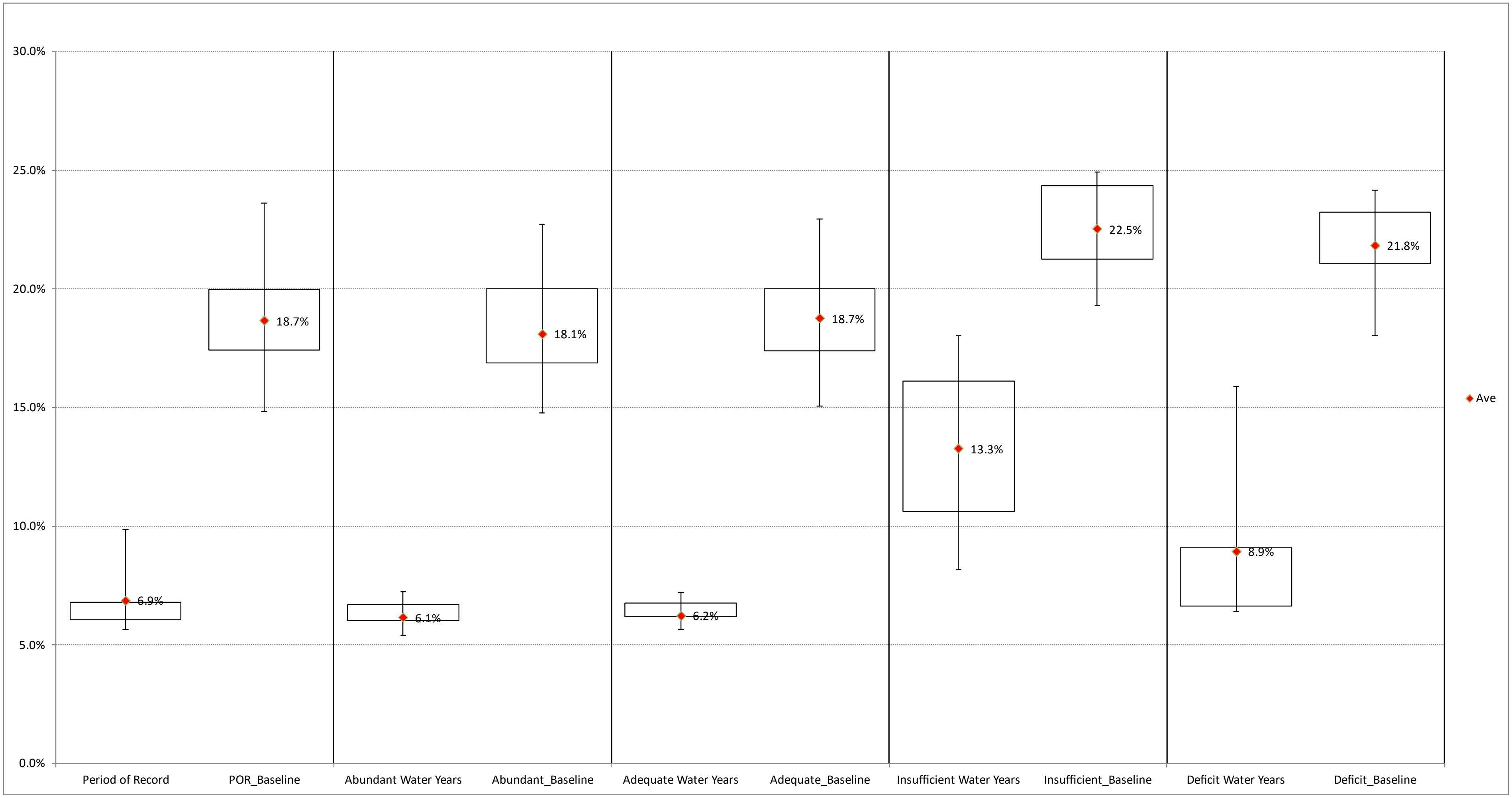

Figure 2-29. Cougar Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Cougar for juvenile spring Chinook sub-yearlings under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

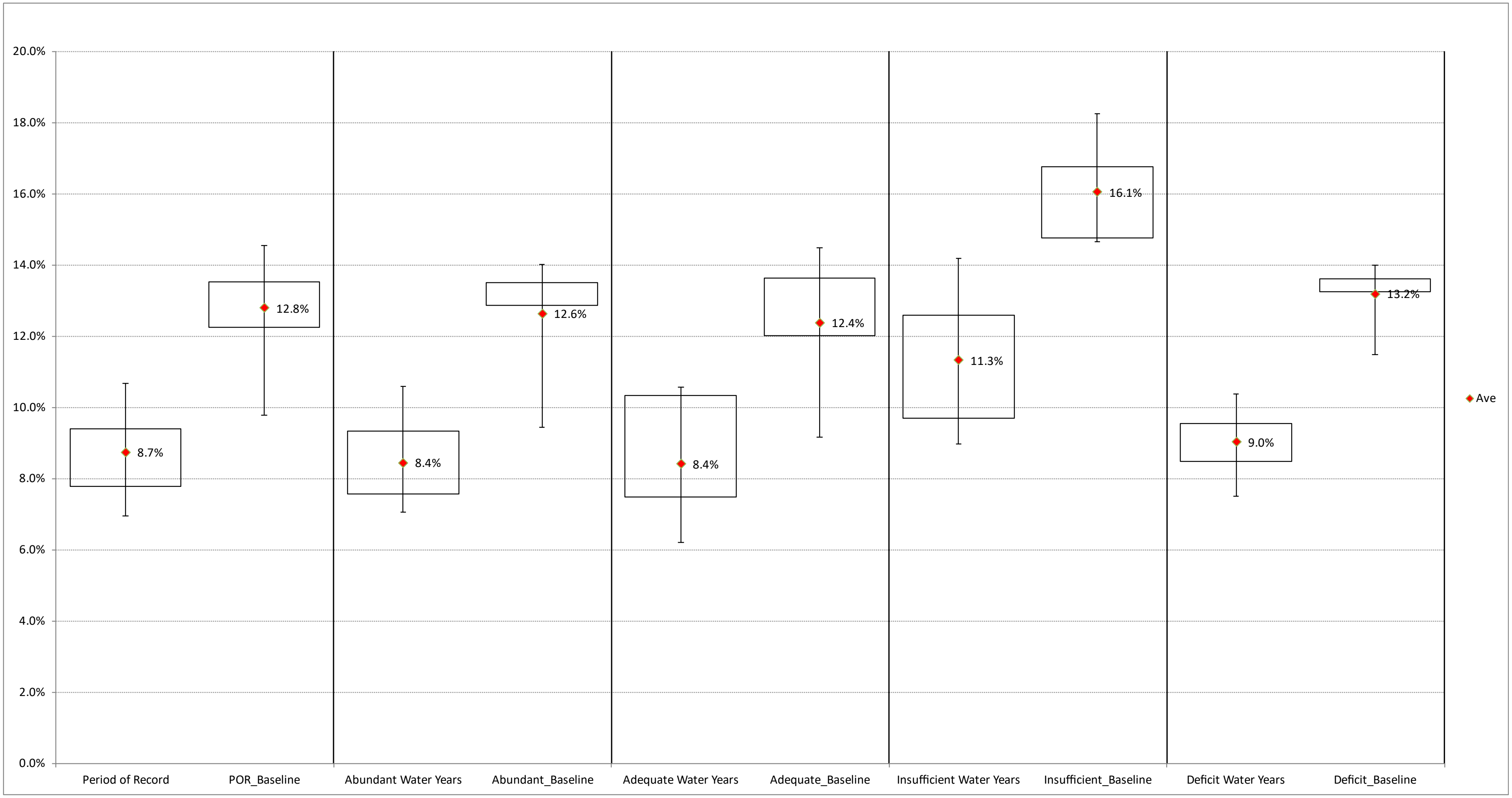

Figure 2-30. Cougar Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Cougar for juvenile spring Chinook yearlings under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.2.5 Middle Fork – Lookout Point

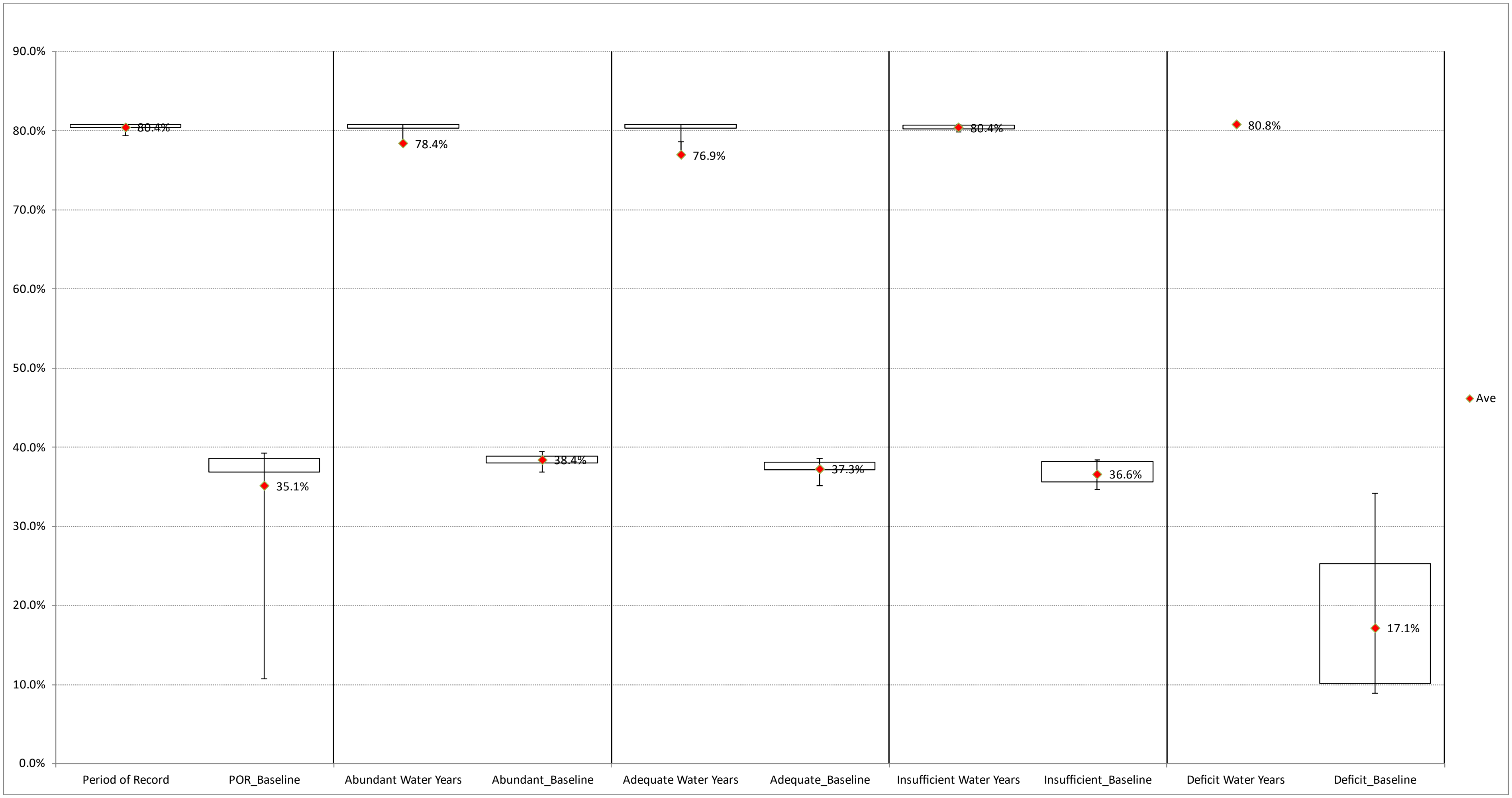

Figure 2-31. Lookout Point Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook fry under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

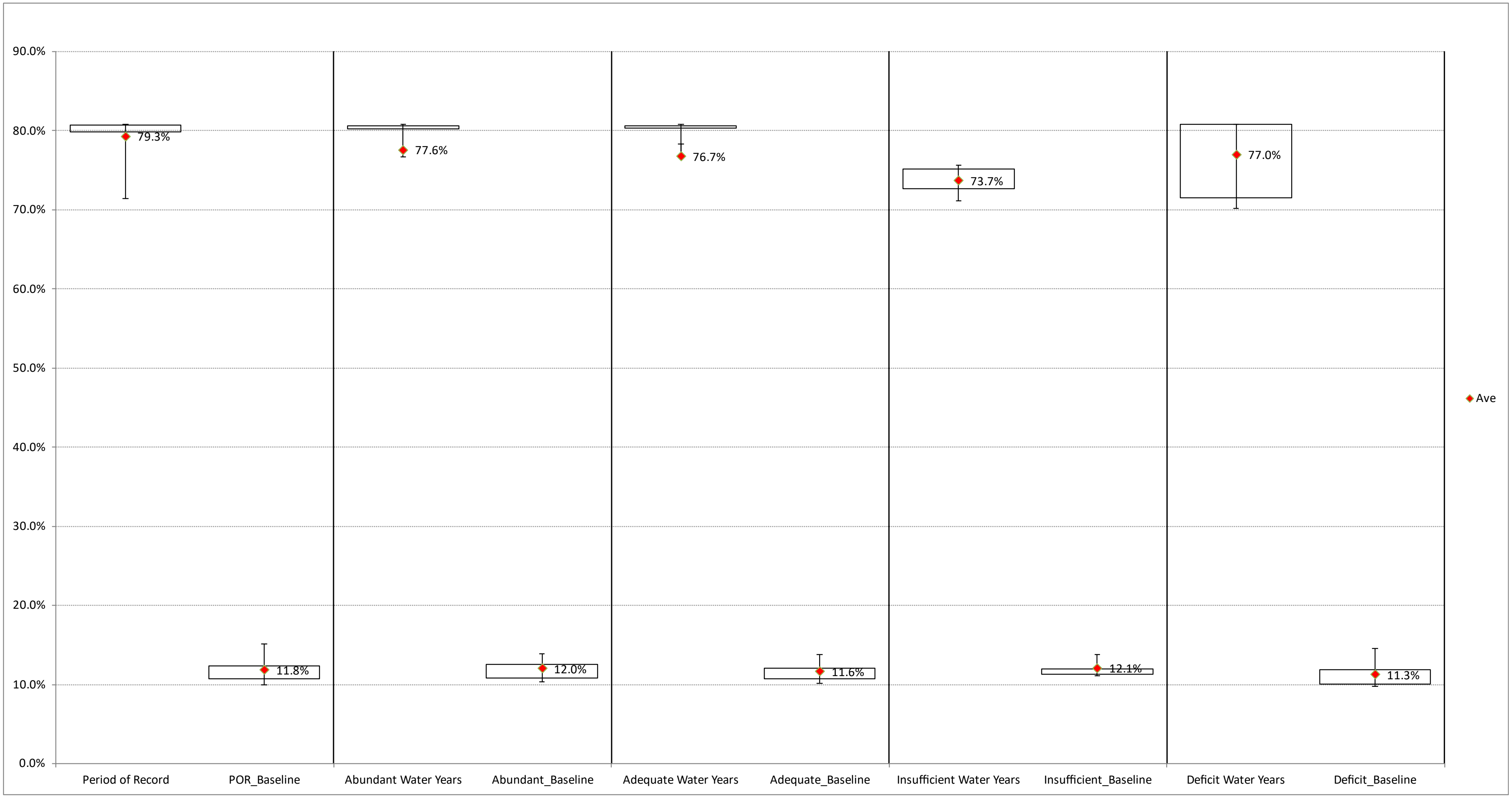

Figure 2-32. Lookout Point Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook sub-yearlings under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

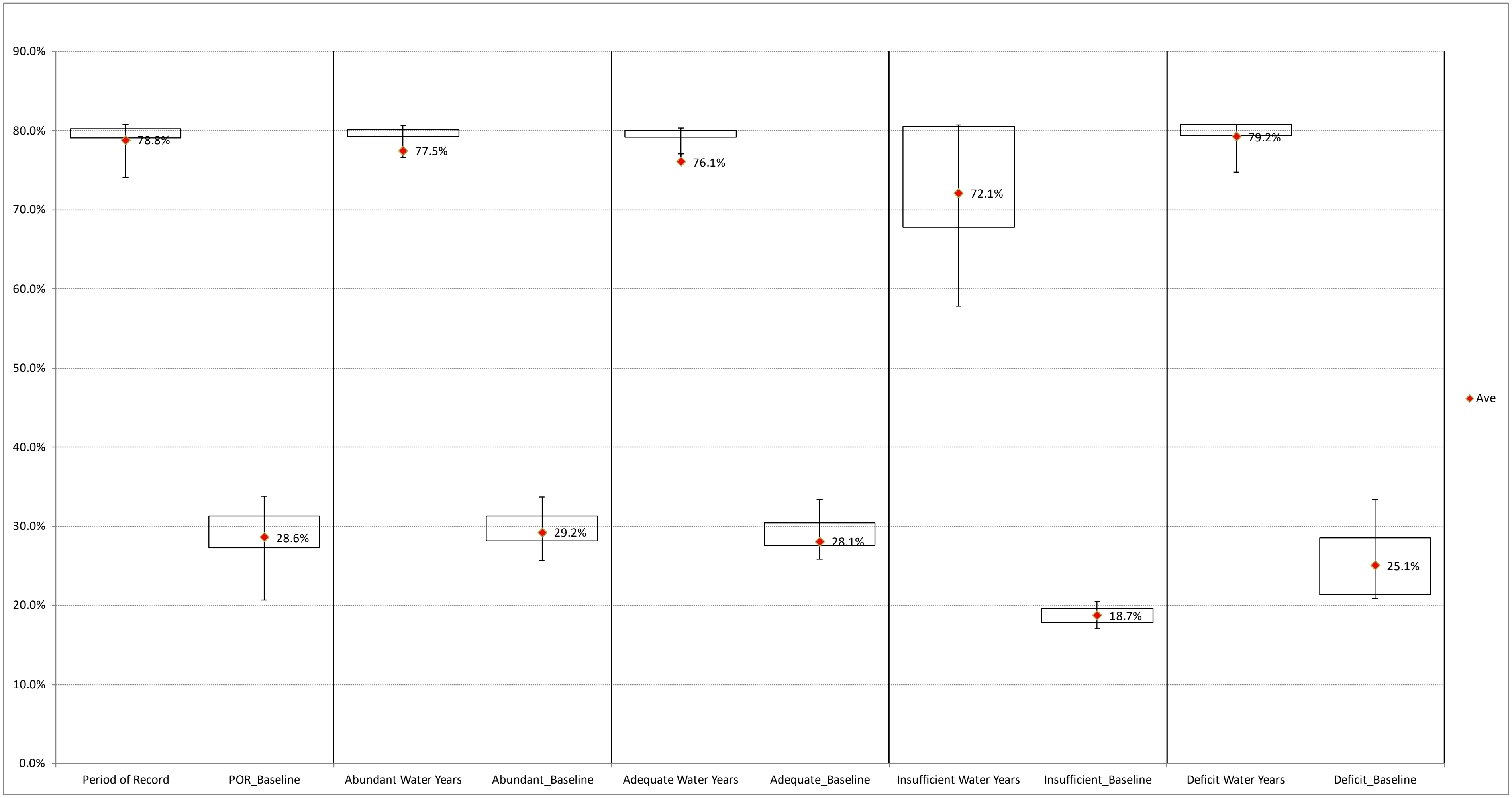

Figure 2-33. Lookout Point Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook yearlings under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

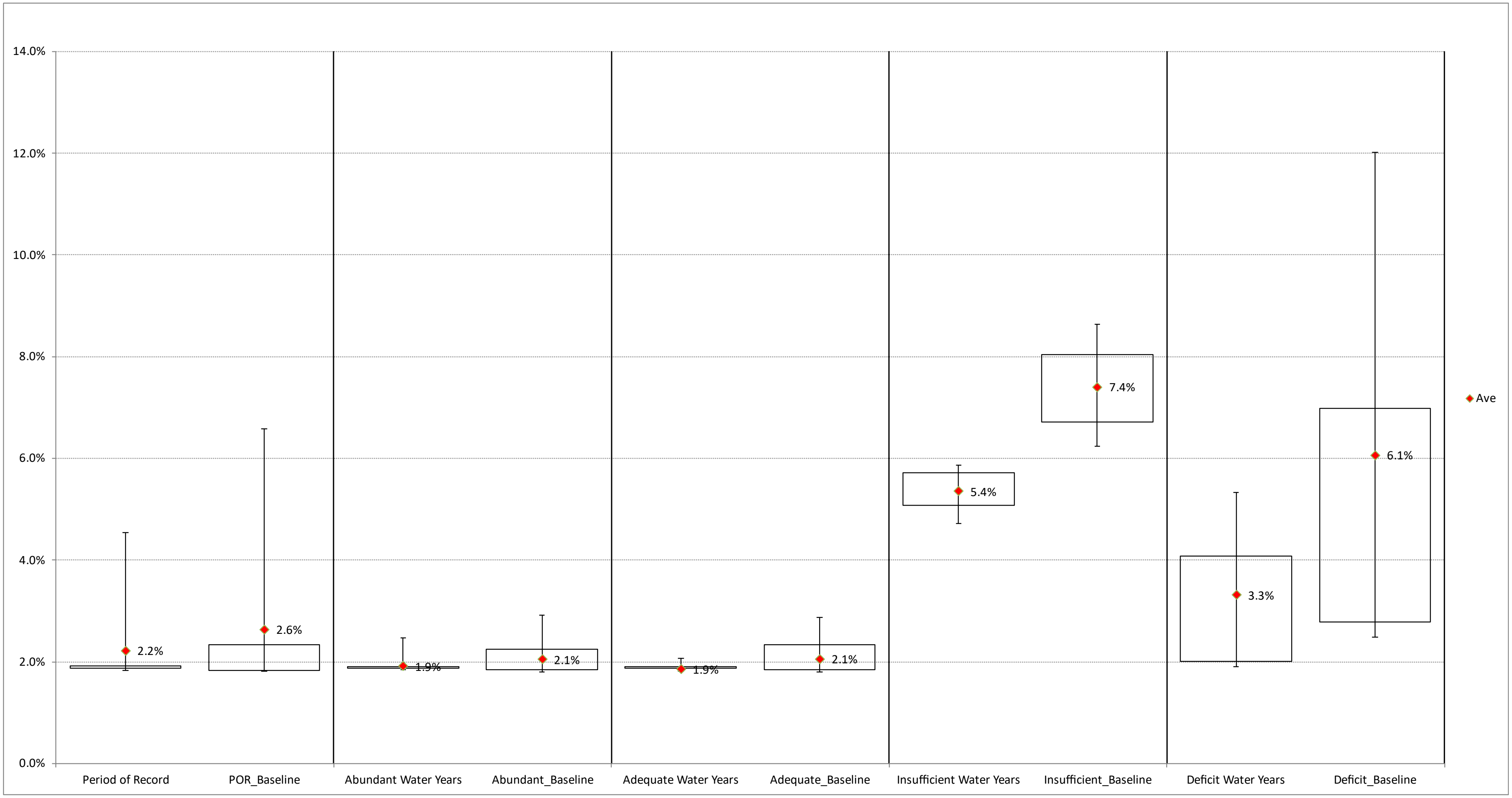

Figure 2-34. Hills Creek Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook fry under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

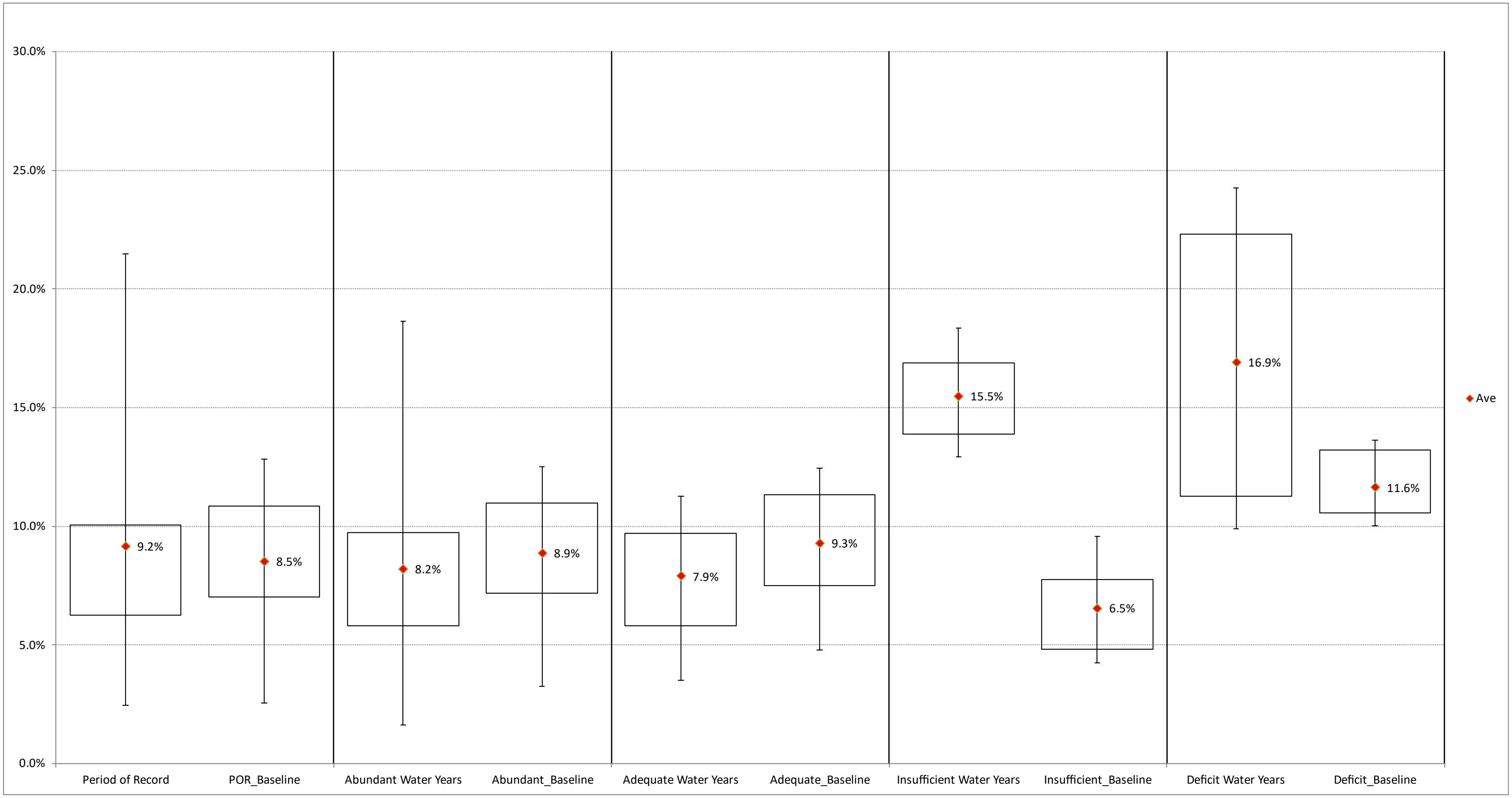

Figure 2-35. Hills Creek Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook sub-yearlings under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

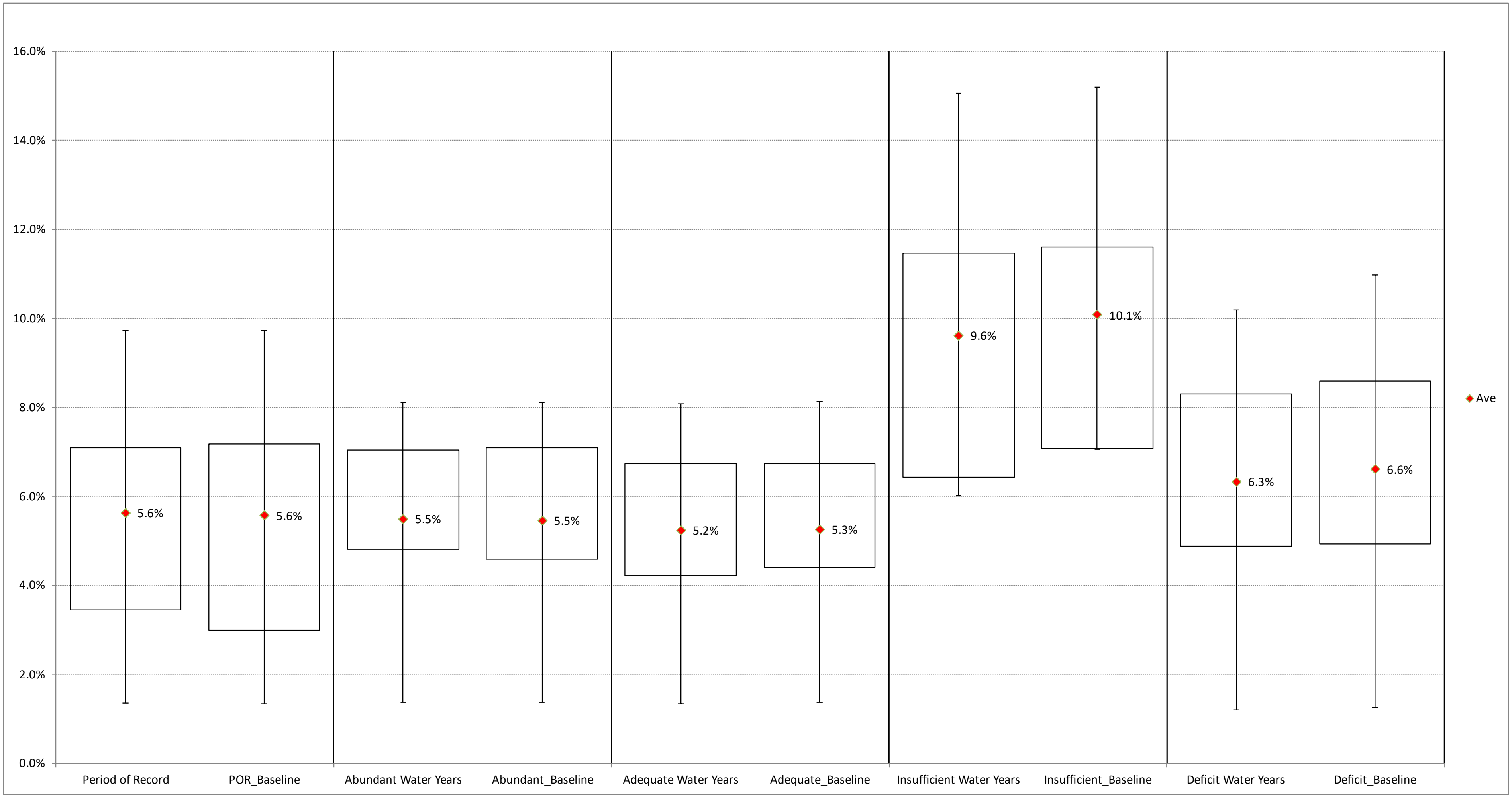

Figure 2-36. Hills Creek Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook yearlings under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

CHINOOK

2.3 CHINOOK ALTERNATIVE 2A

2.3.1 North Santiam - Detroit

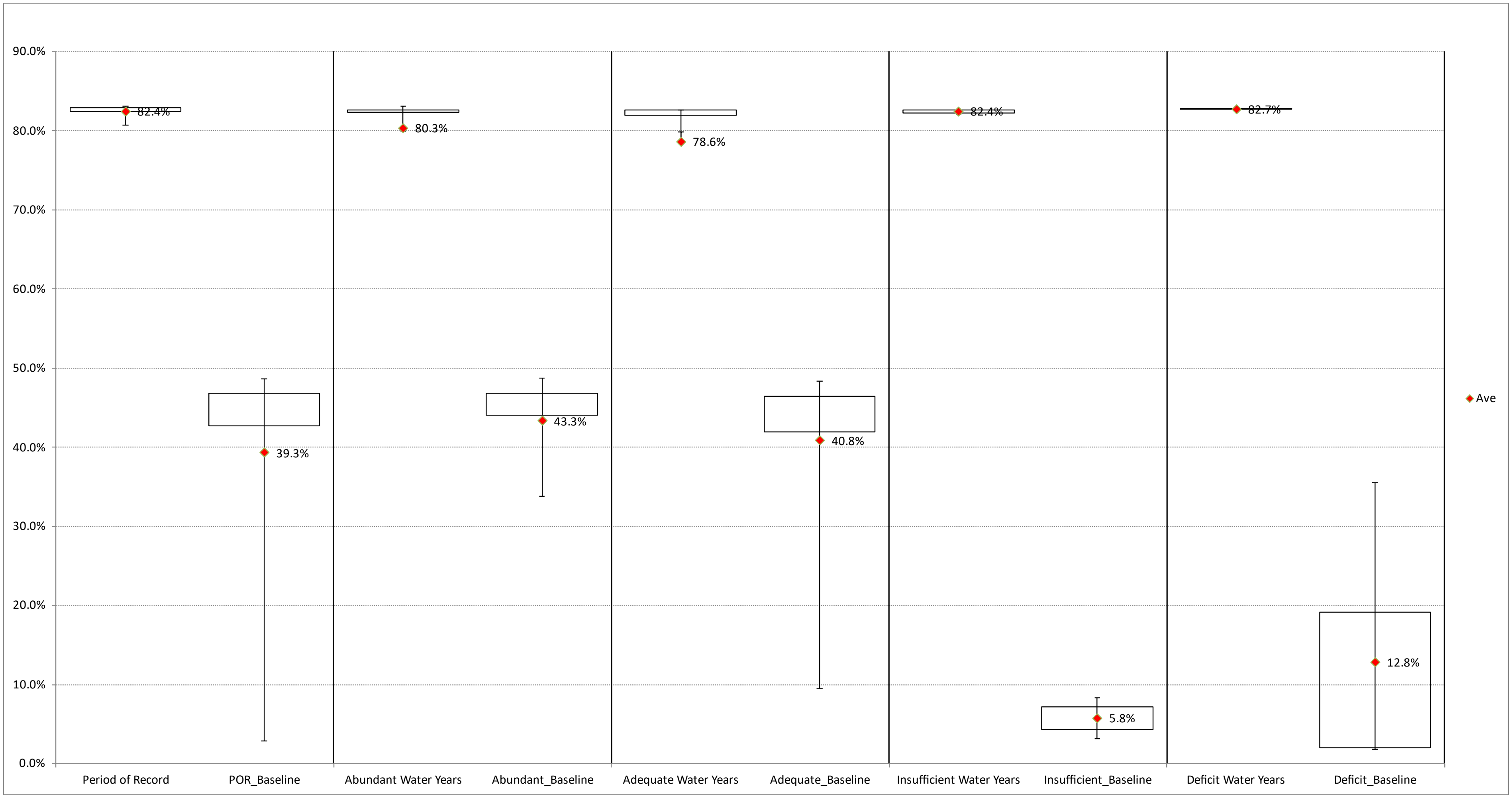

Figure 2-37. Detroit Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Detroit for juvenile spring Chinook fry under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

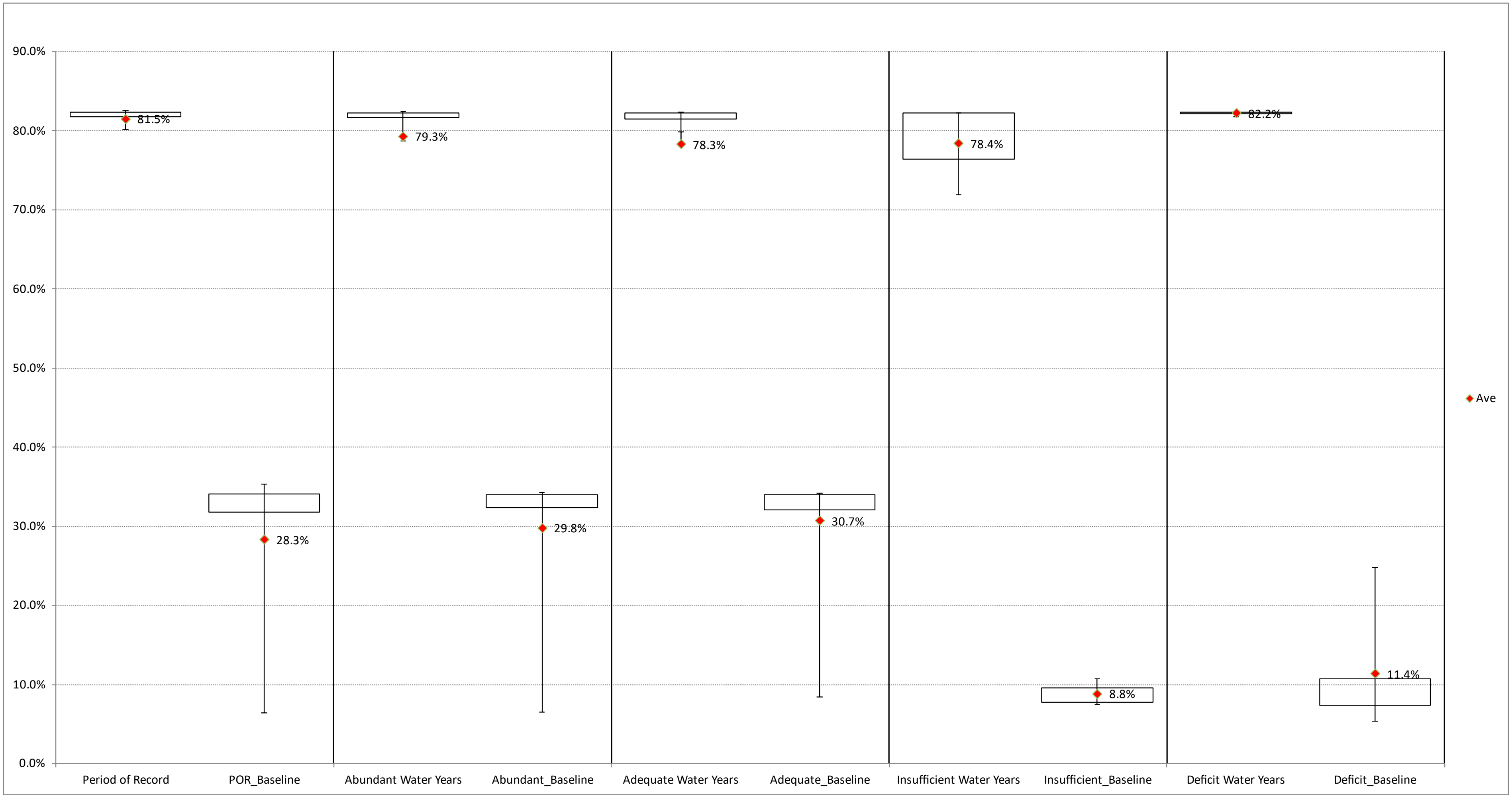

Figure 2-38. Detroit Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Detroit for juvenile spring Chinook sub-yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

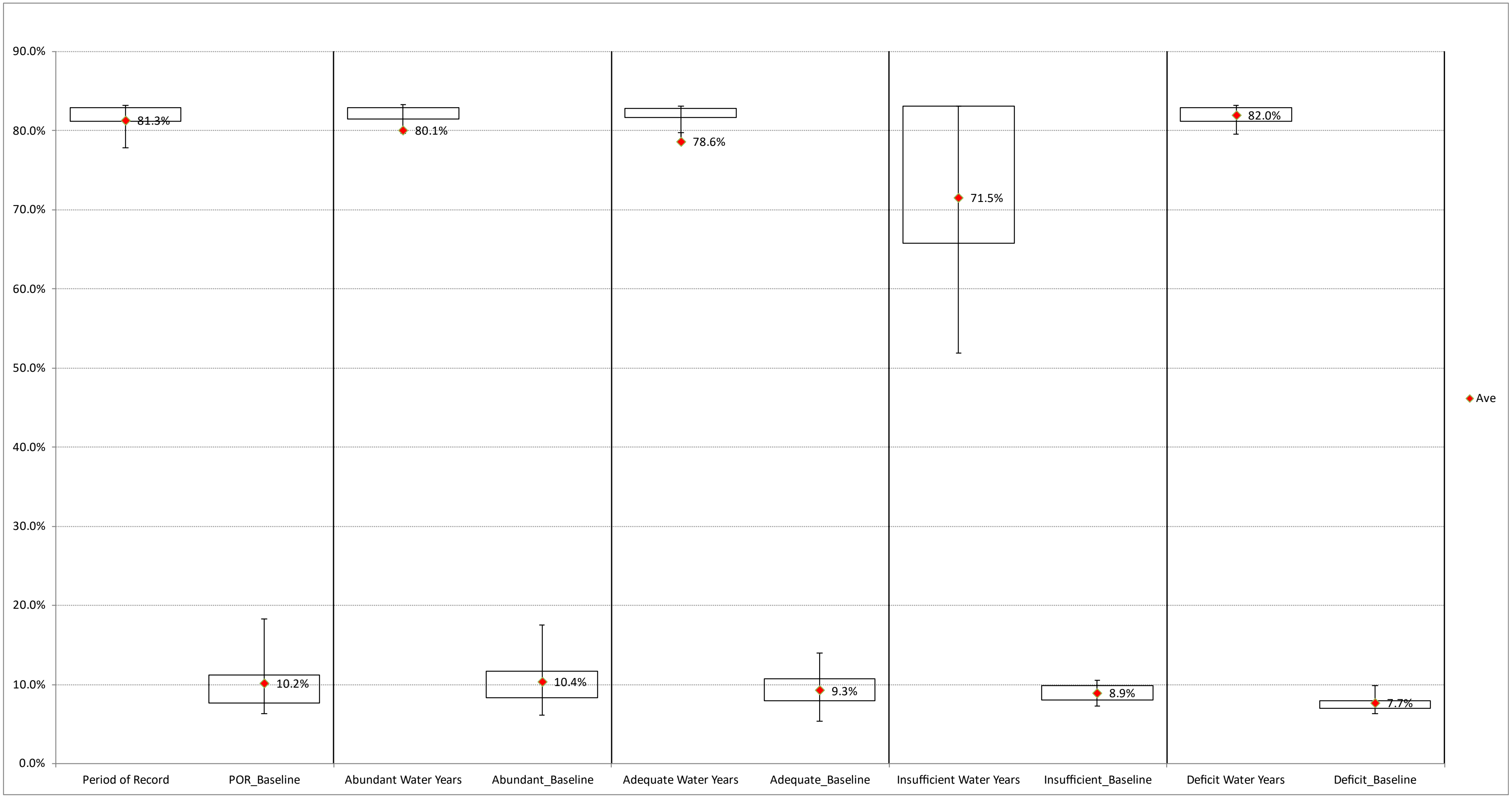

Figure 2-39. Detroit Juvenile Spring Chinook Yearlings Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Detroit for juvenile spring Chinook yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.3.2 South Santiam – Foster

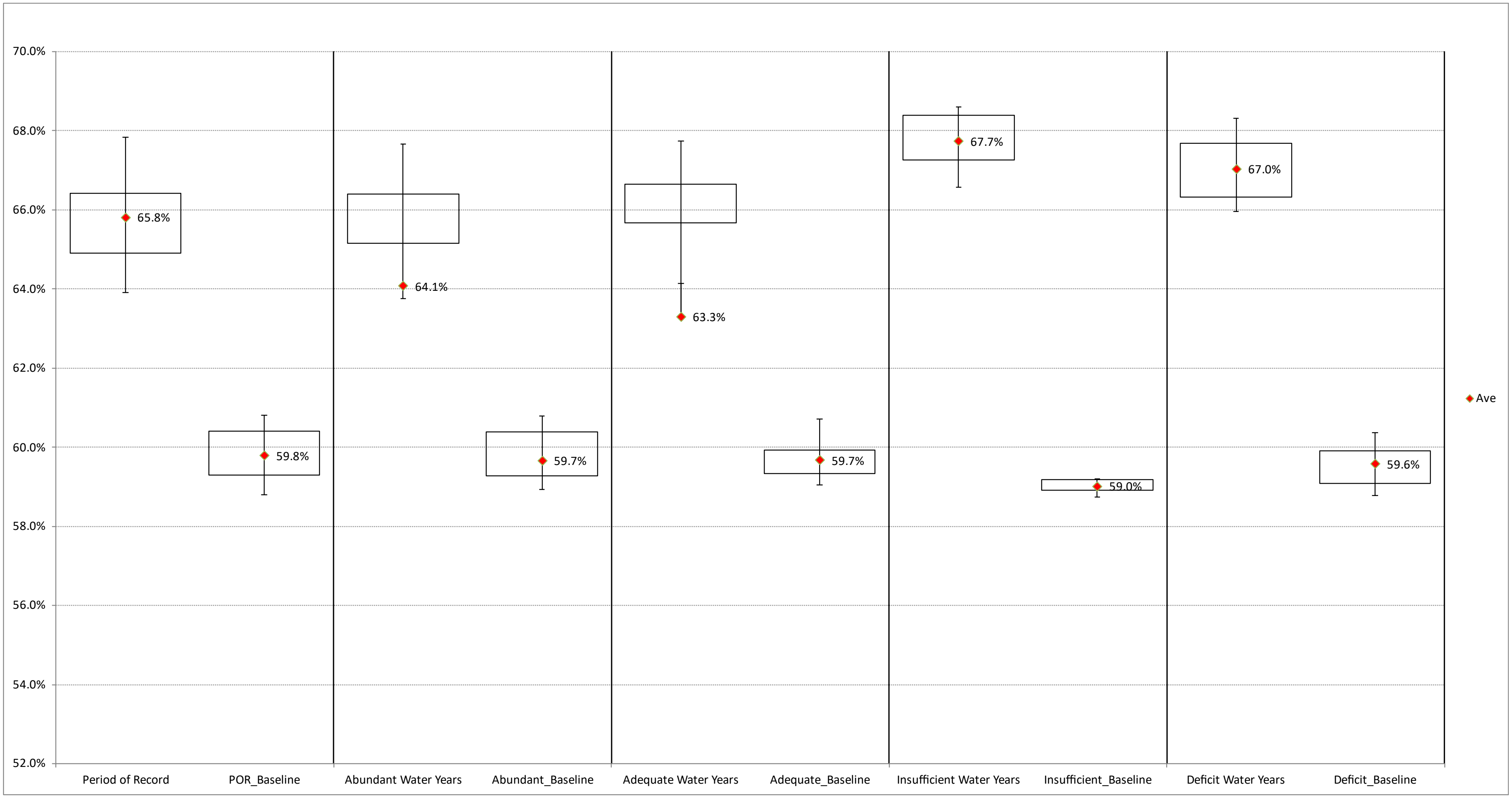

Figure 2-40. Foster Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Foster for juvenile spring Chinook fry under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

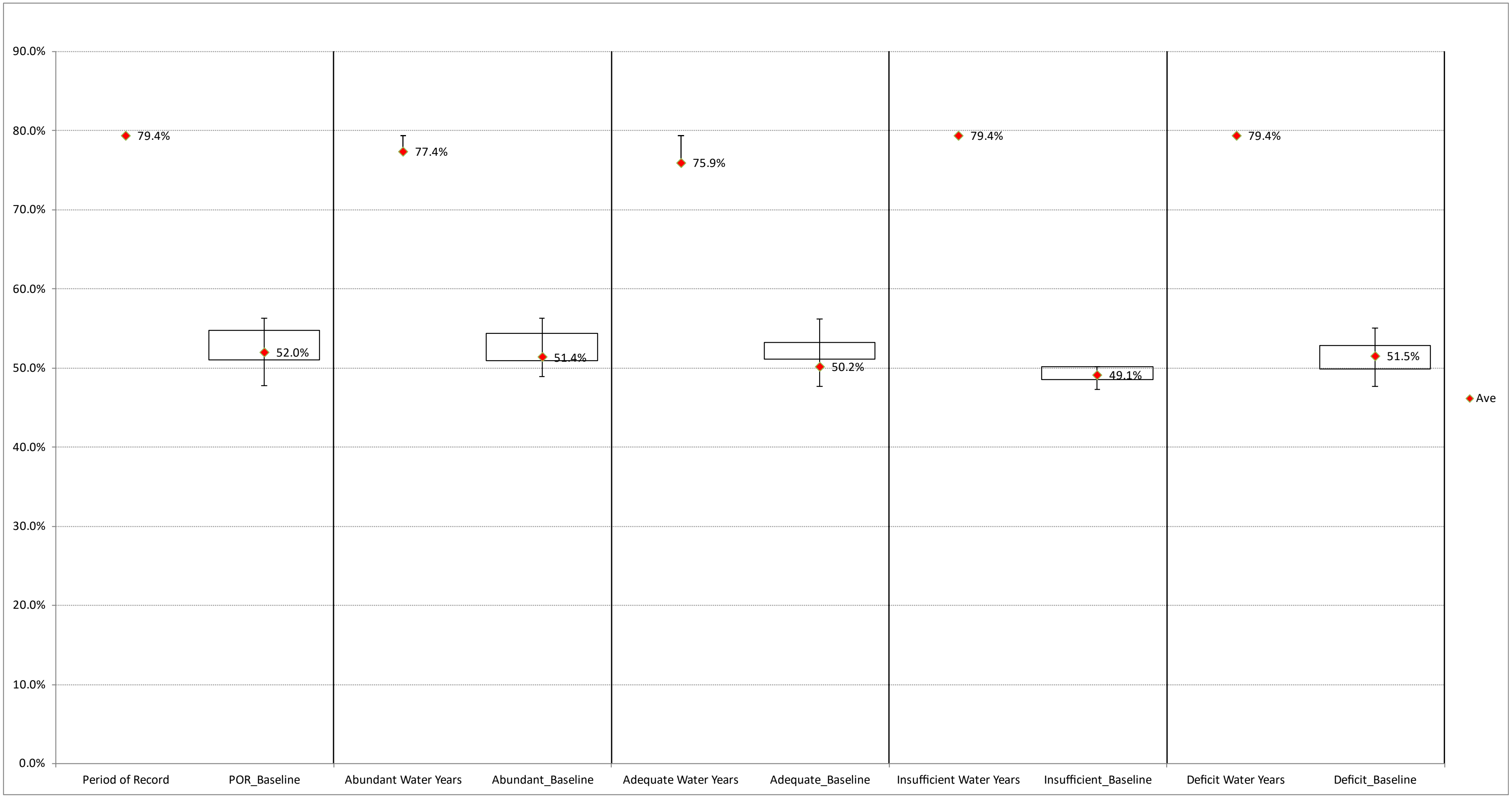

Figure 2-41. Foster Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Foster for juvenile spring Chinook sub-yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

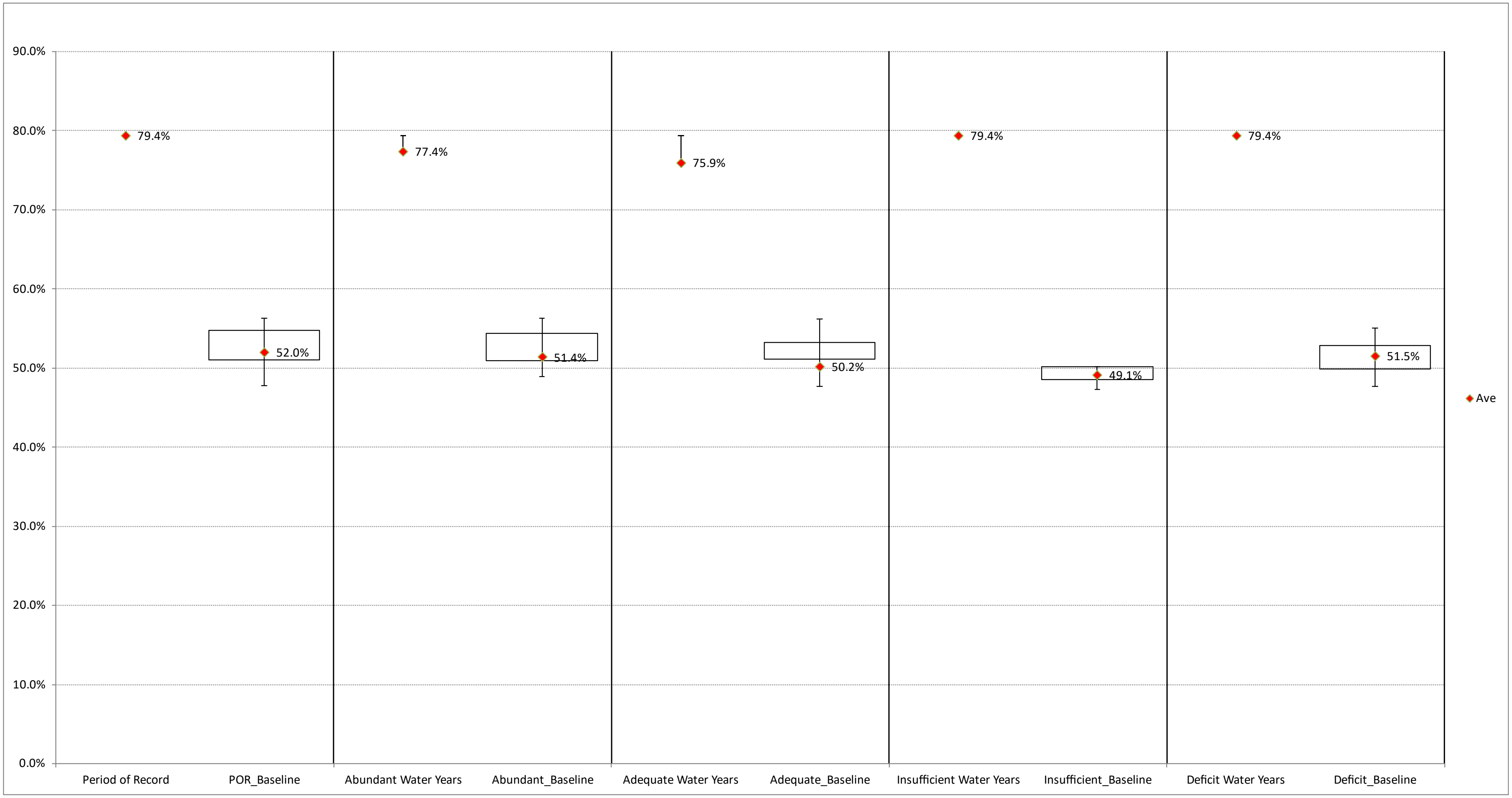

Figure 2-42. Foster Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Foster for juvenile spring Chinook yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.3.3 South Santiam – Green Peter

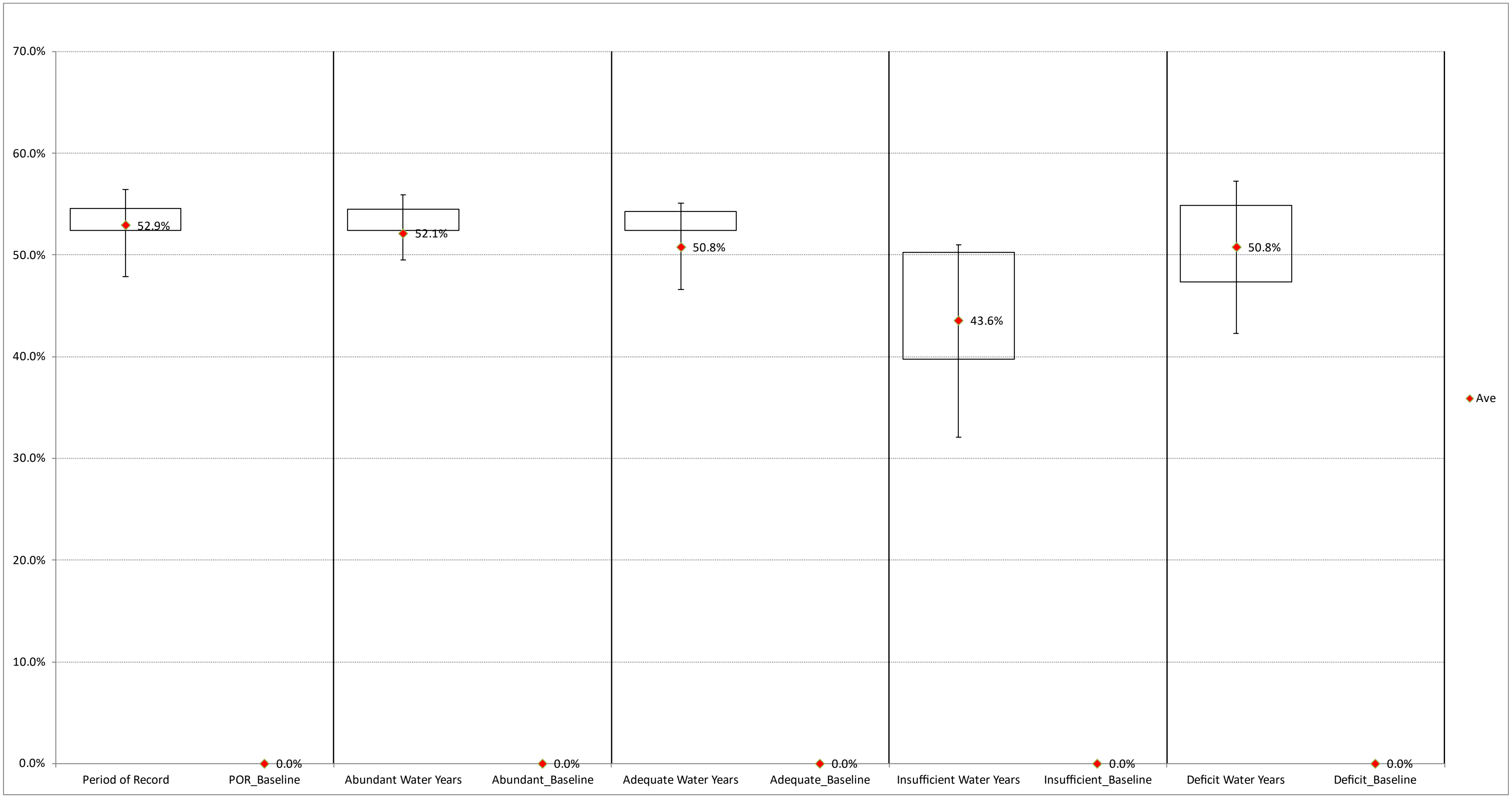

Figure 2-43. Green Peter Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Green Peter for juvenile spring Chinook fry under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

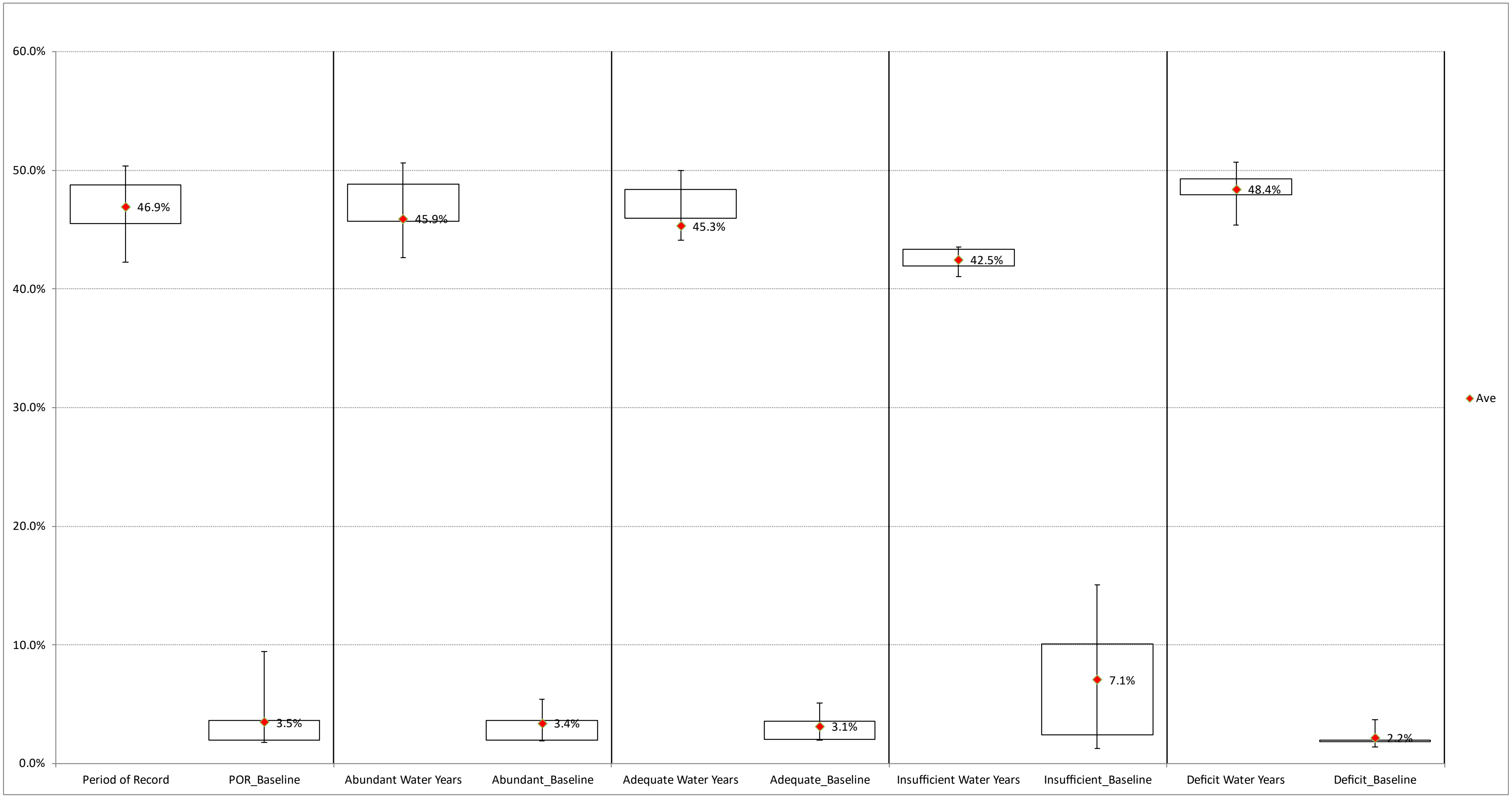

Figure 2-44. Green Peter Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Green Peter for juvenile spring Chinook sub-yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

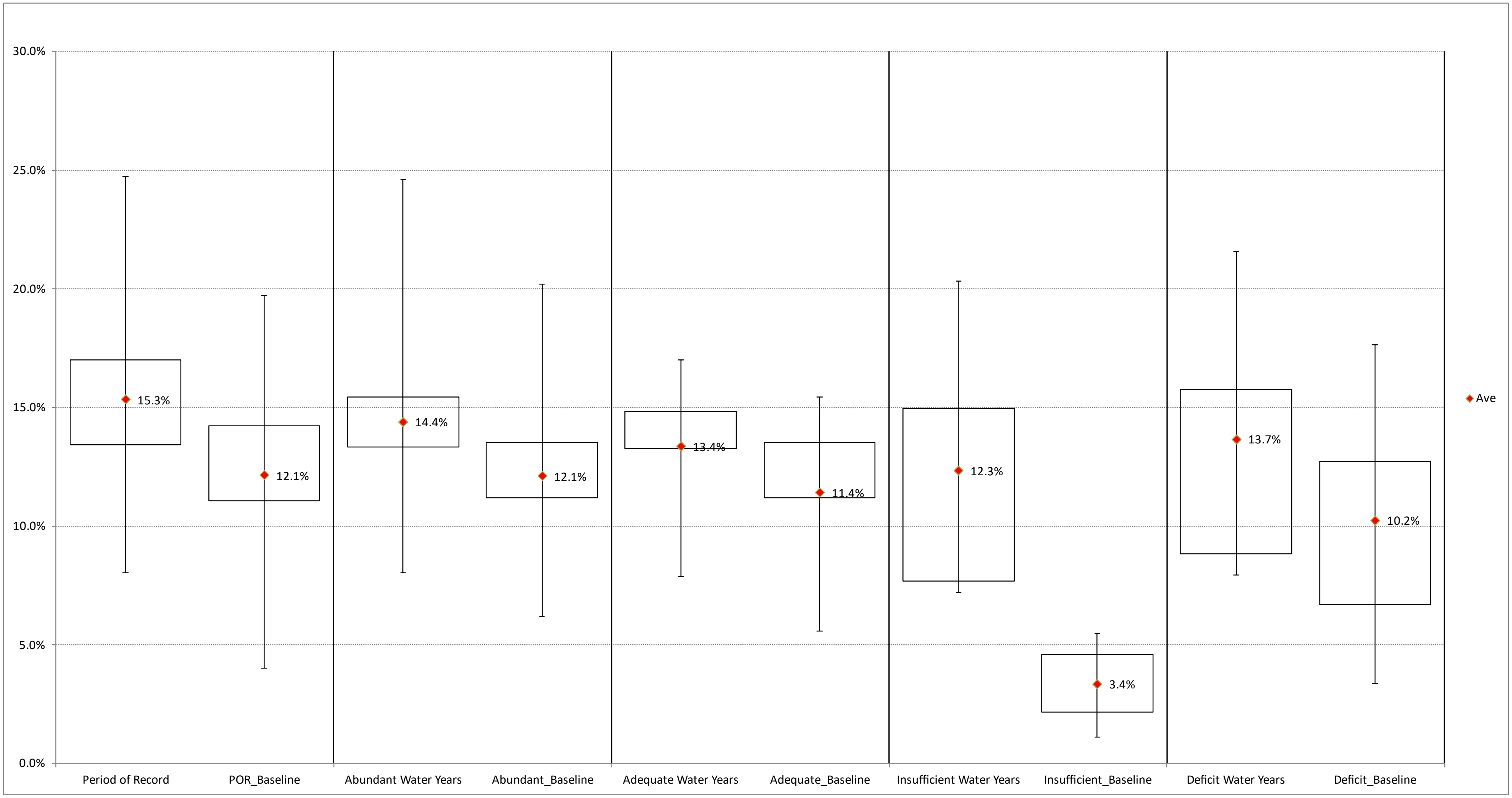

Figure 2-45. Green Peter Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Green Peter for juvenile spring Chinook yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.3.4 McKenzie - Cougar

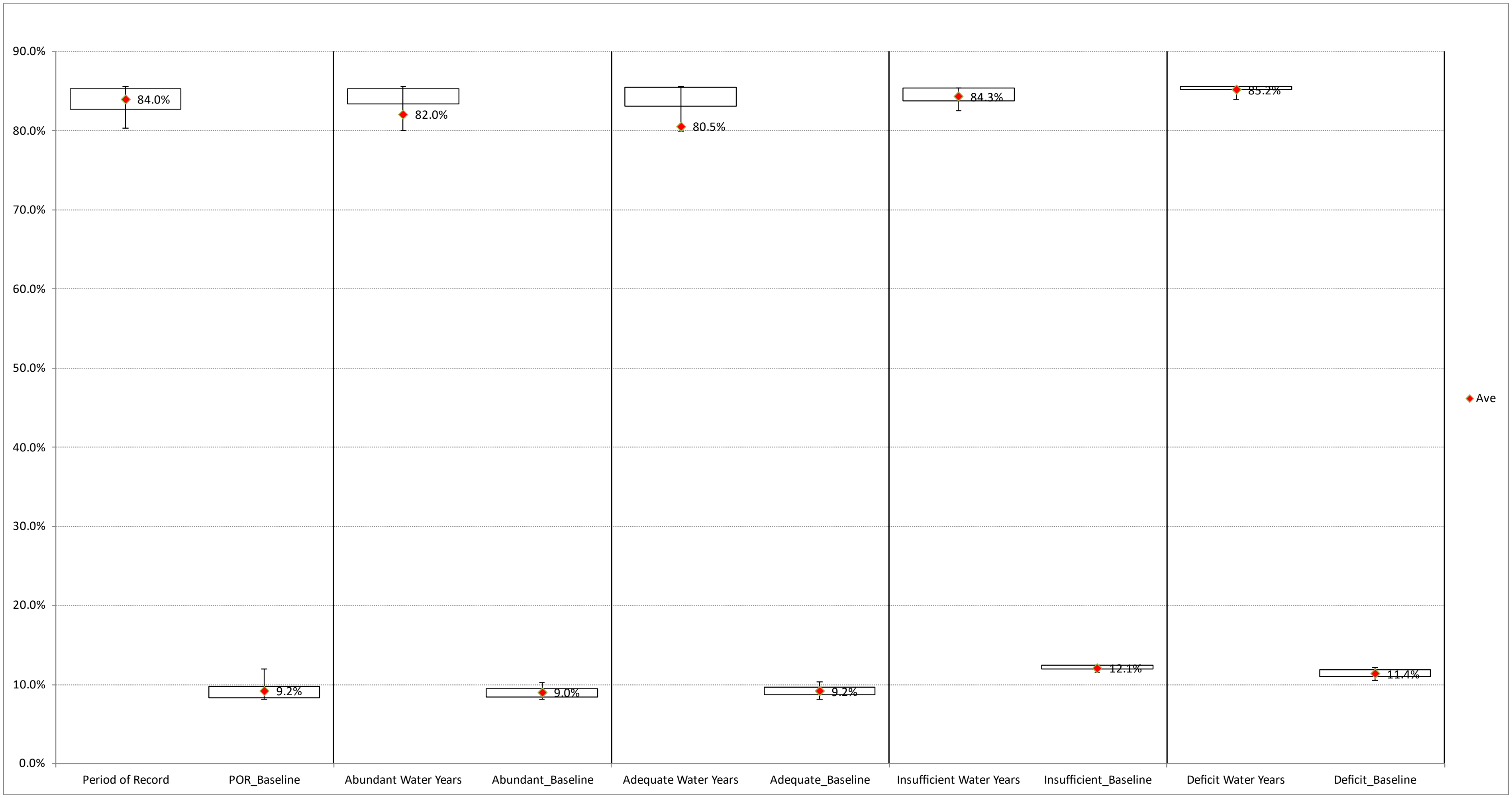

Figure 2-46. Cougar Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Cougar for juvenile spring Chinook fry under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

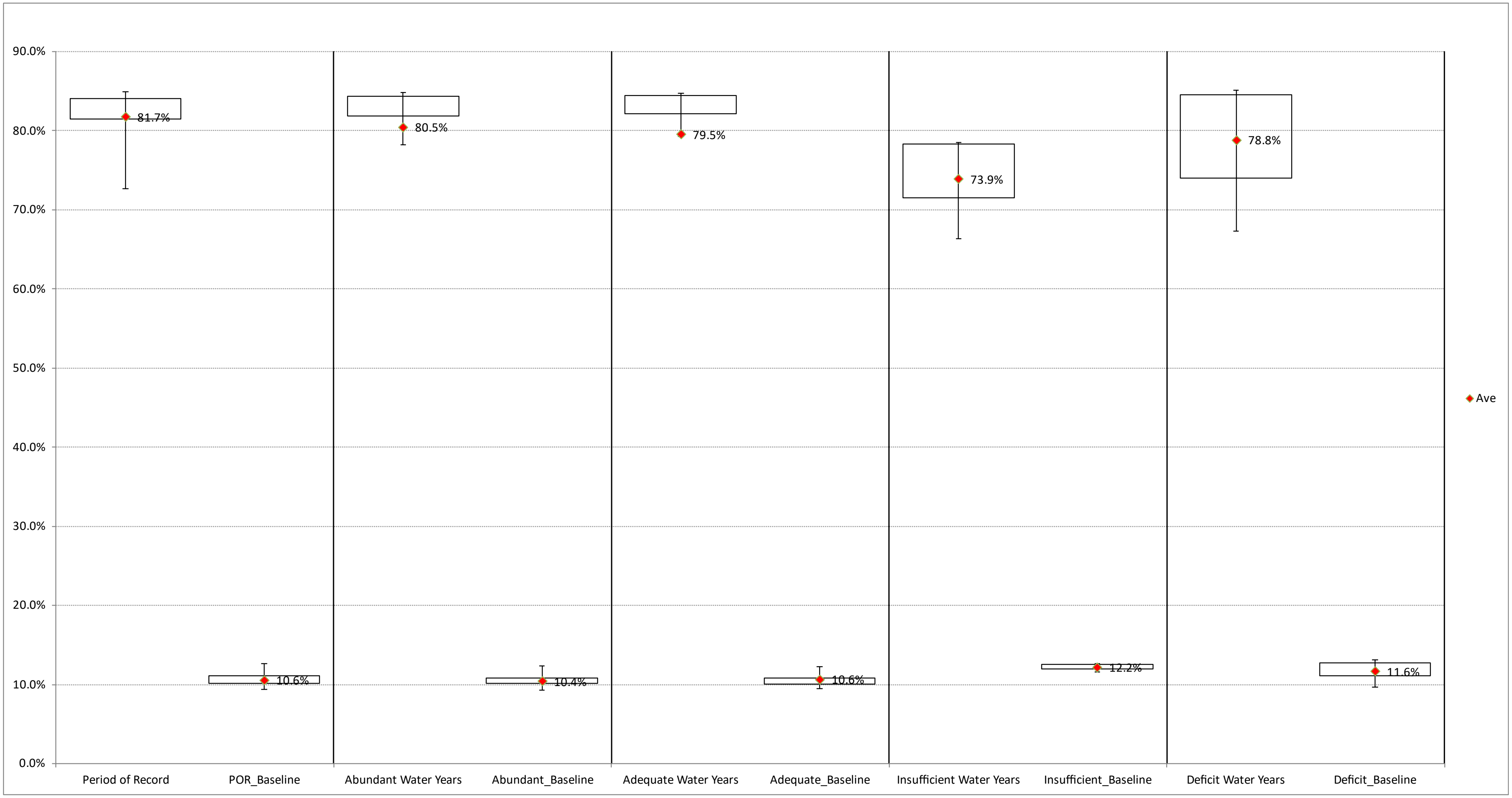

Figure 2-47. Cougar Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Cougar for juvenile spring Chinook sub-yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

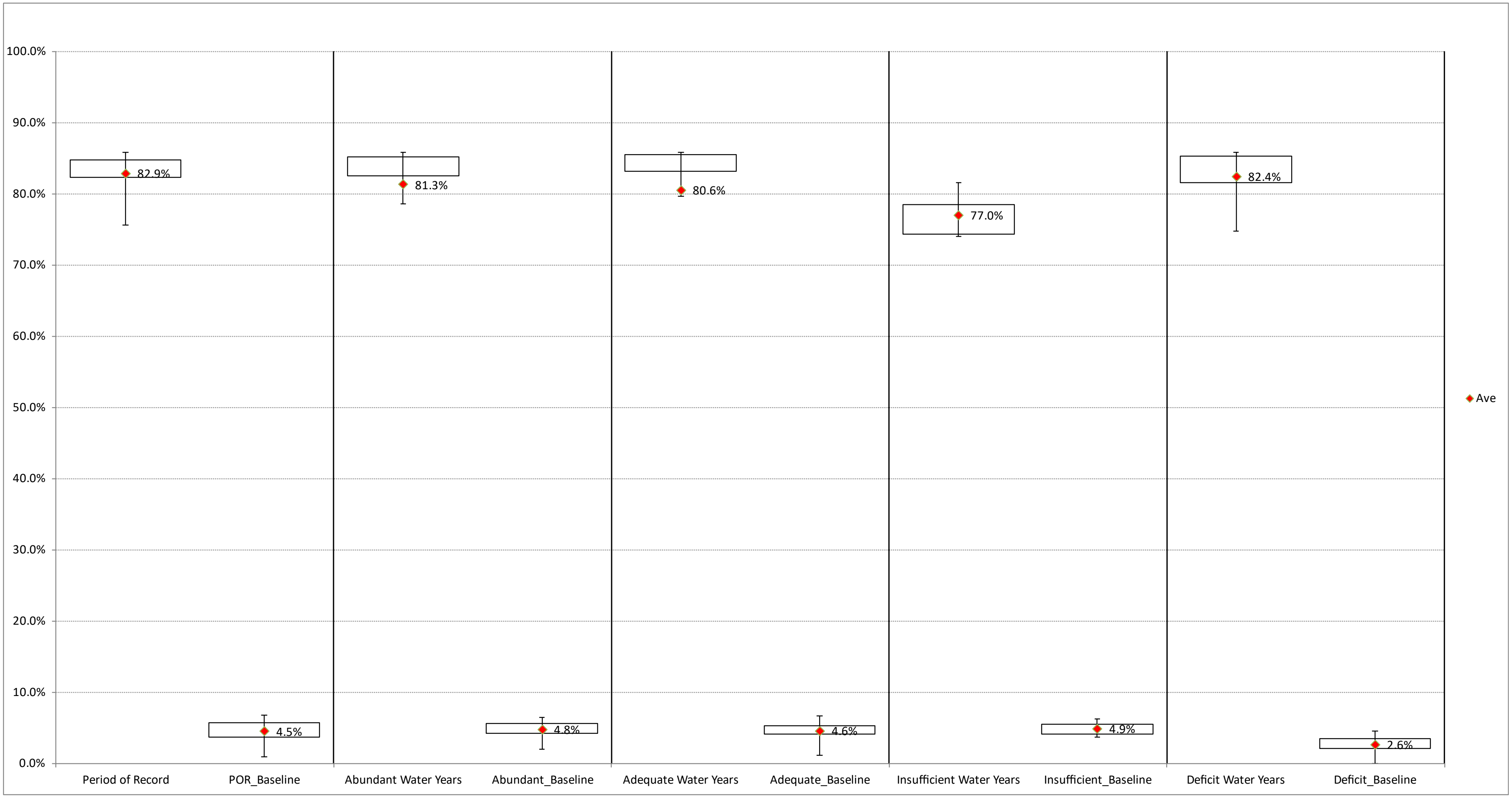

Figure 2-48. Cougar Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Cougar for juvenile spring Chinook yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.3.5 Middle Fork – Lookout Point

Figure 2-49. Lookout Point Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook fry under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

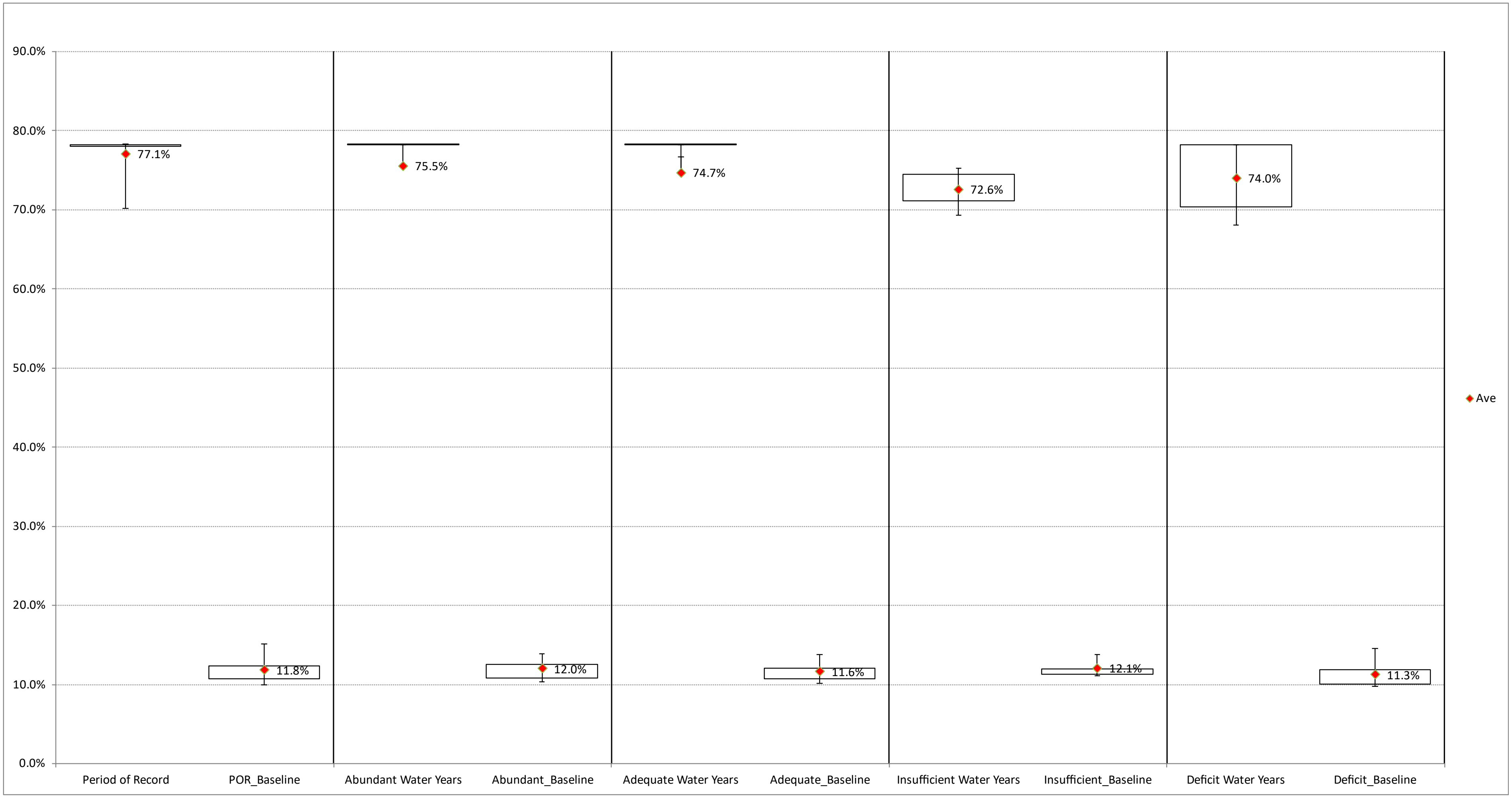

Figure 2-50. Lookout Point Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 2a. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook sub-yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

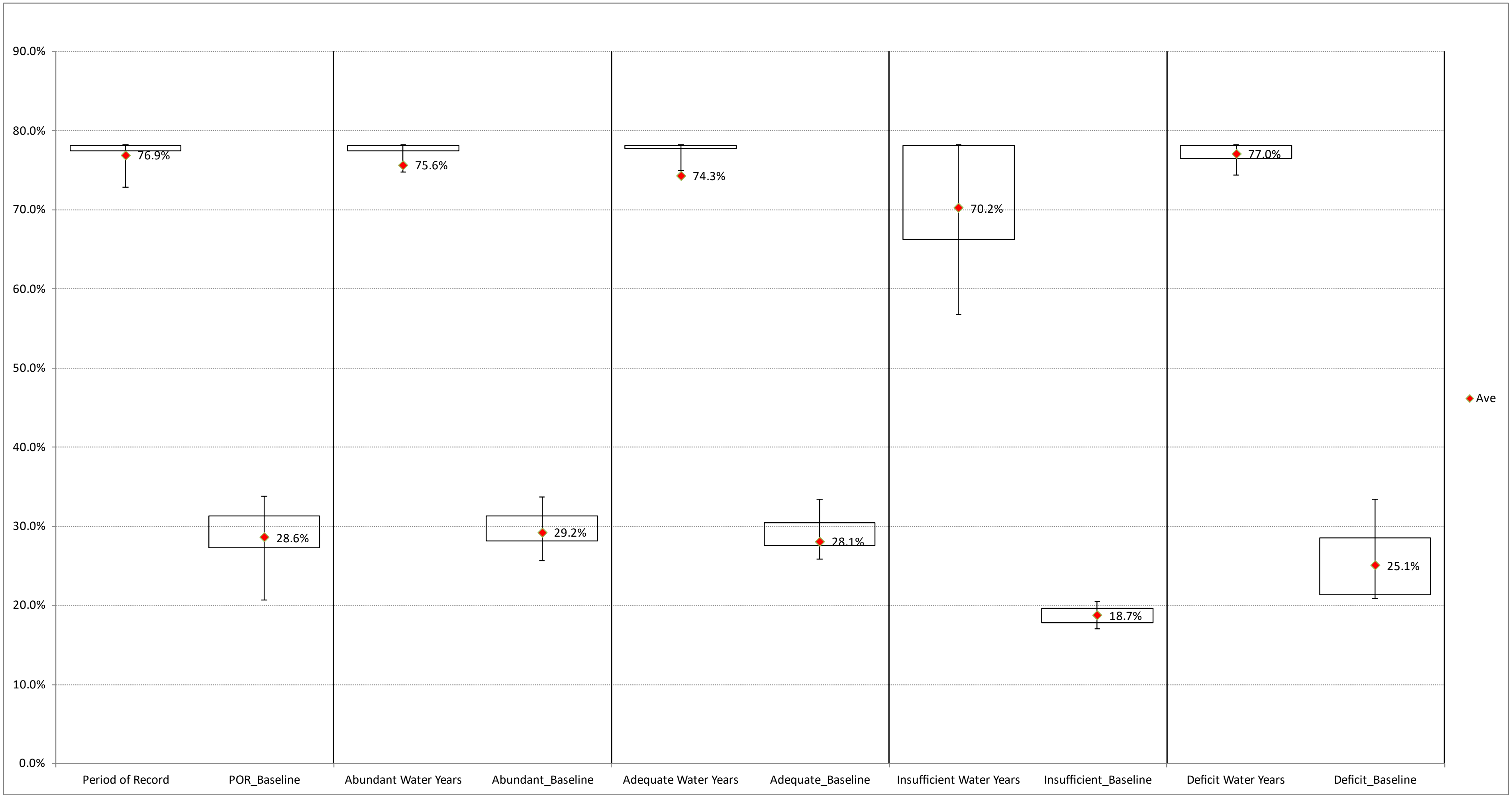

Figure 2-51. Lookout Point for juvenile spring Chinook yearling Downstream dam passage survival at s under Alternative 2a. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

CHINOOK

2.3.6 ALTERNATIVE 2B

2.3.7 North Santiam – Detroit

See Alternative 2a

2.3.8 South Santiam – Foster

See Alternative 2a

2.3.9 South Santiam – Green Peter

See Alternative 2a

2.3.10 McKenzie – Cougar

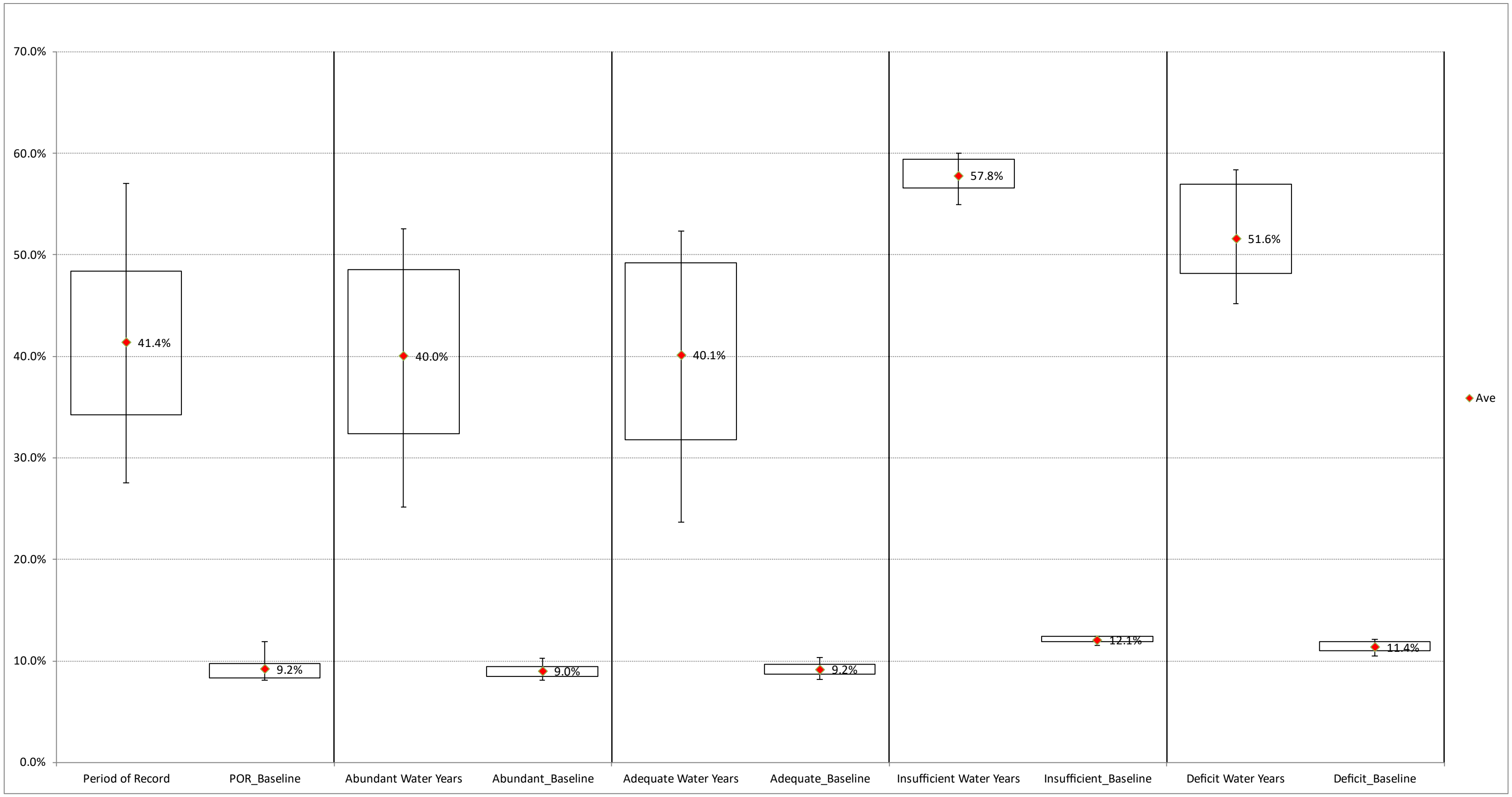

Figure 2-52. Cougar Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 2b. *Downstream dam passage survival at Cougar for juvenile spring Chinook fry under Alternative 2b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

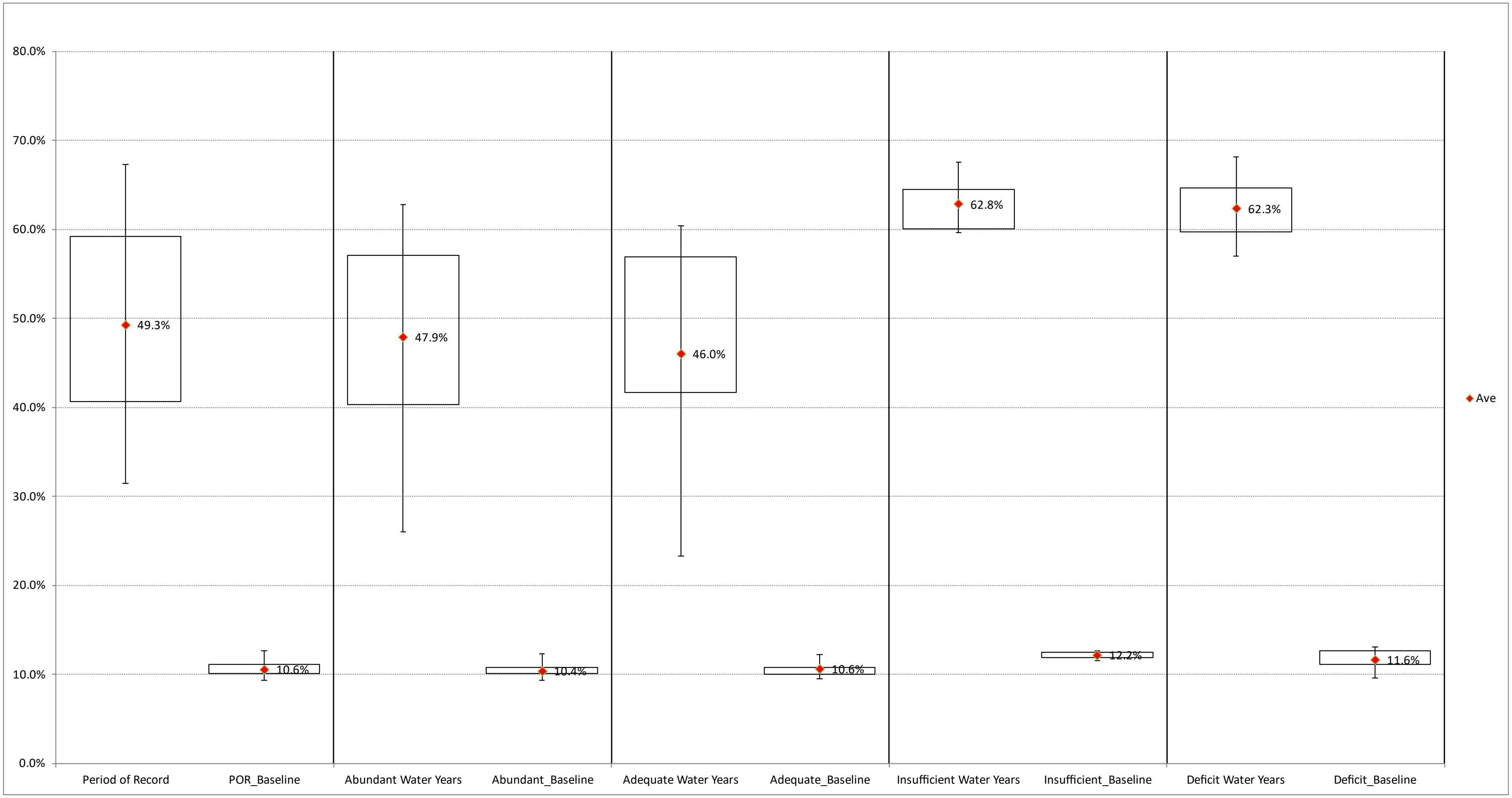

Figure 2-53. Cougar Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 2b. *The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

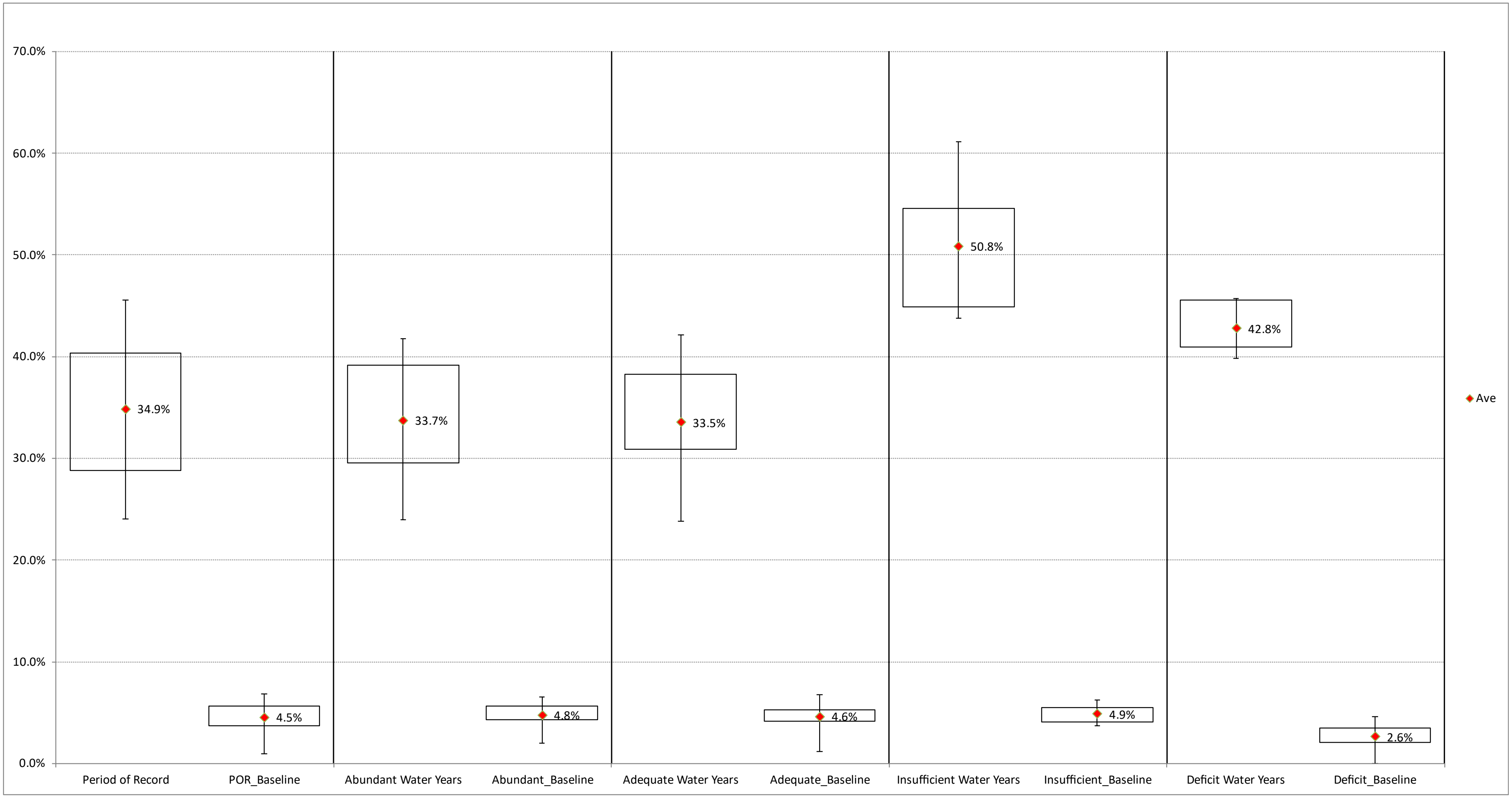

Figure 2-54. Downstream dam passage survival at Cougar for juvenile spring Chinook yearlings under Alternative 2b. *The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.3.11 Middle Fork – Lookout Point

See Alternative 2a

CHINOOK

2.3.12 ALTERNATIVE 3a

2.3.13

2.3.14 North Santiam – Detroit

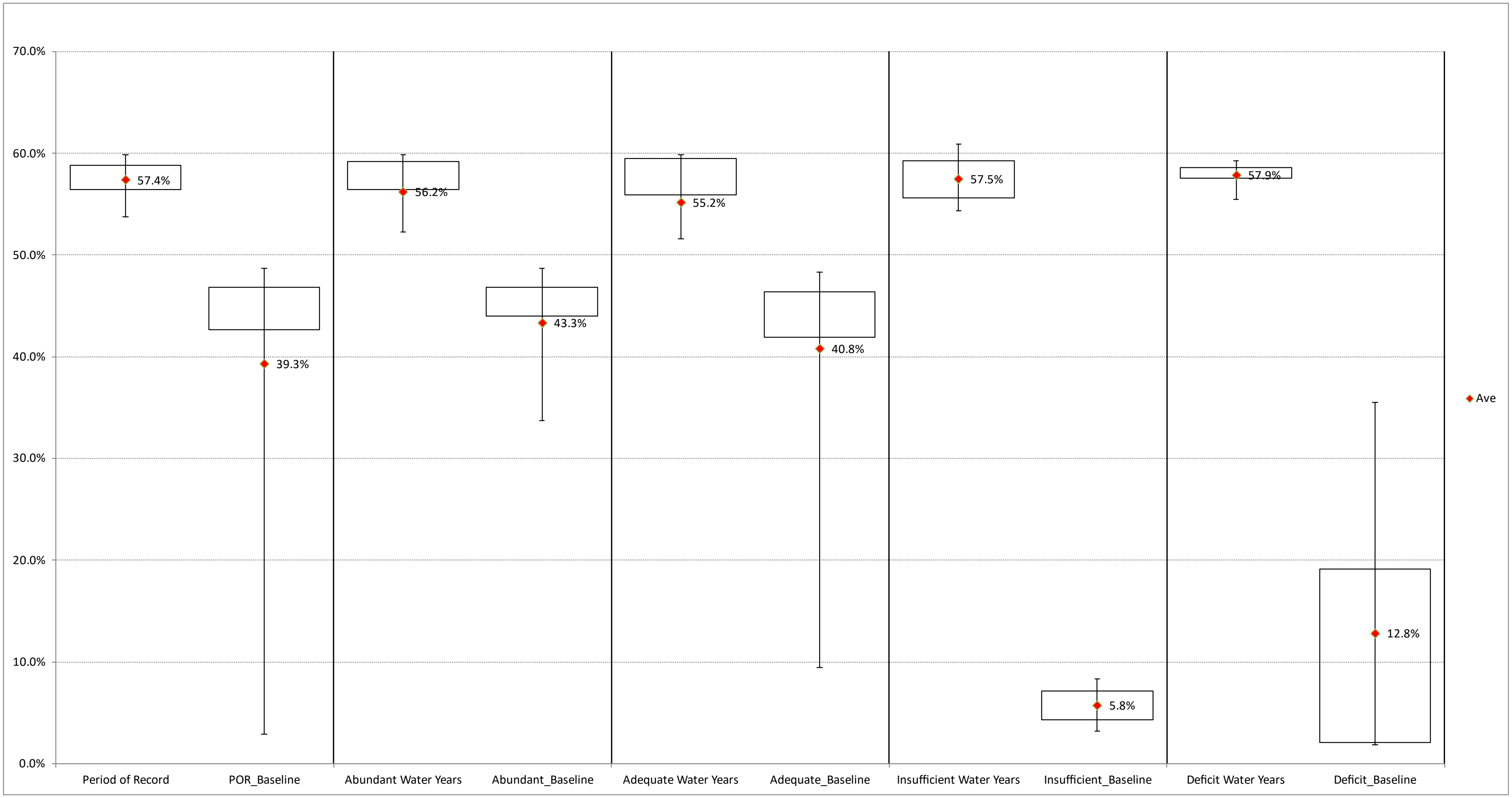

Figure 2-55. Detroit Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Detroit for juvenile spring Chinook fry under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

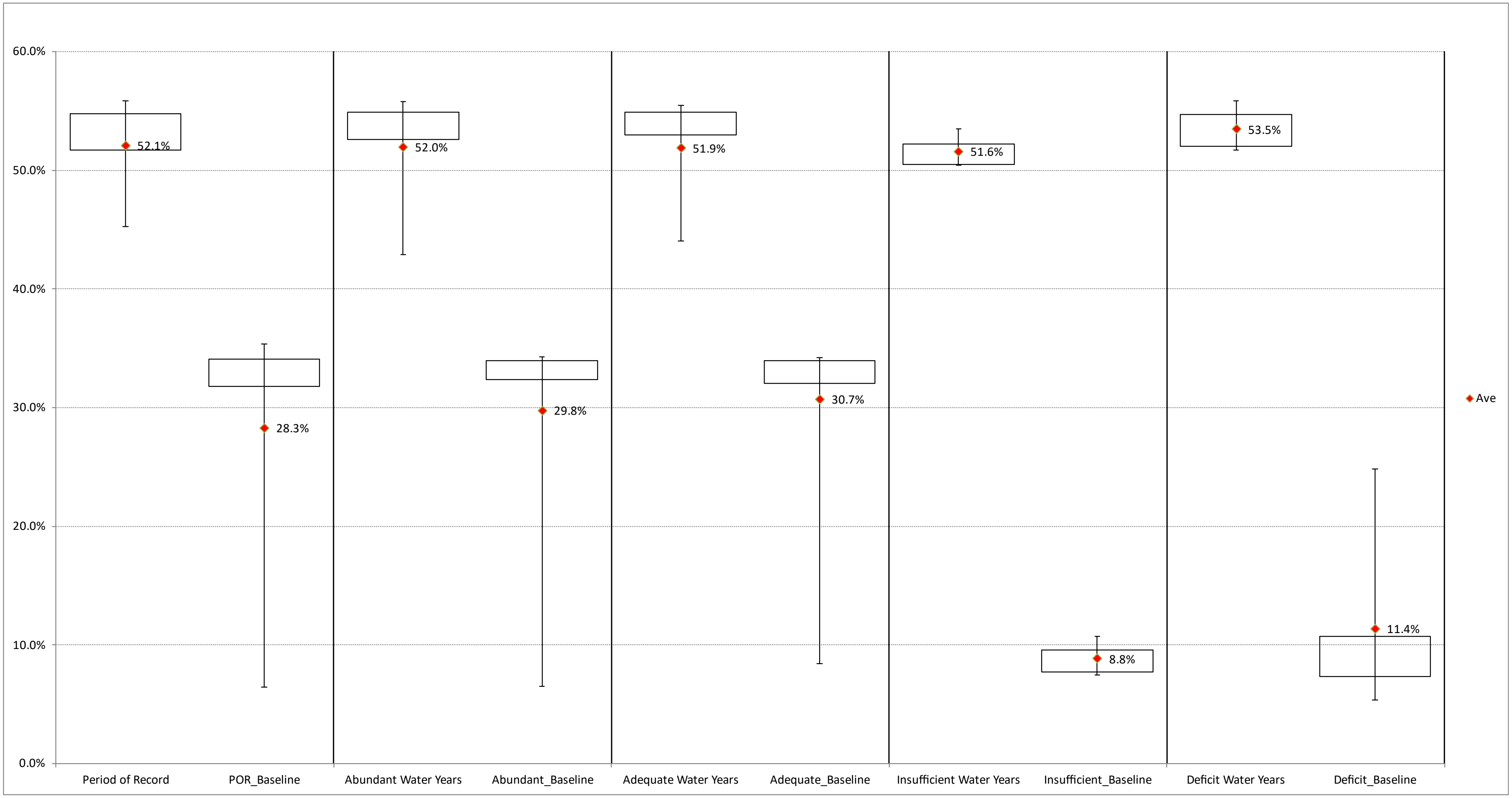

Figure 2-56. Detroit Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Detroit for juvenile spring Chinook sub-yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

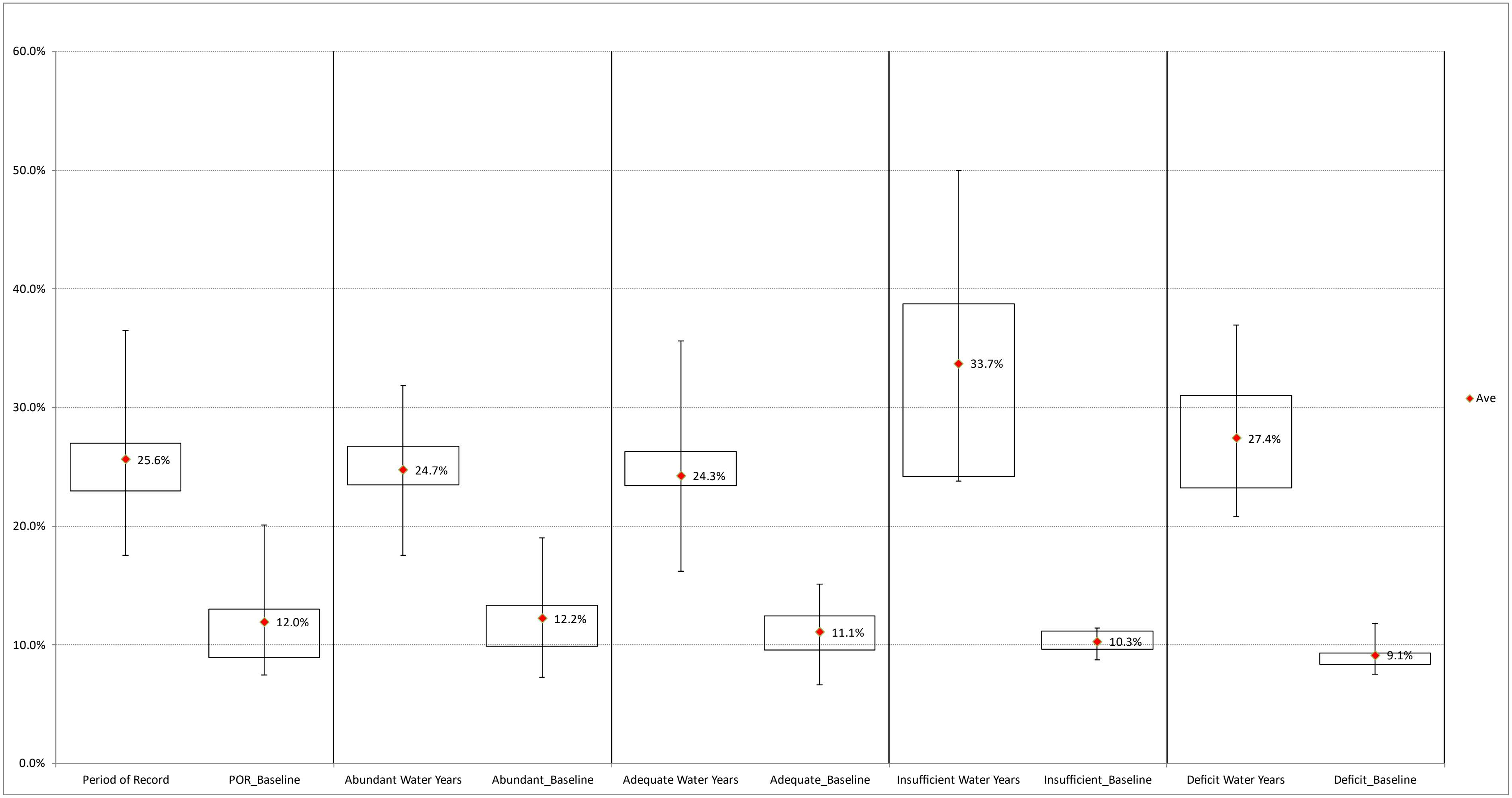

Figure 2-57. Detroit Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Detroit for juvenile spring Chinook yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.3.15 South Santiam - Foster

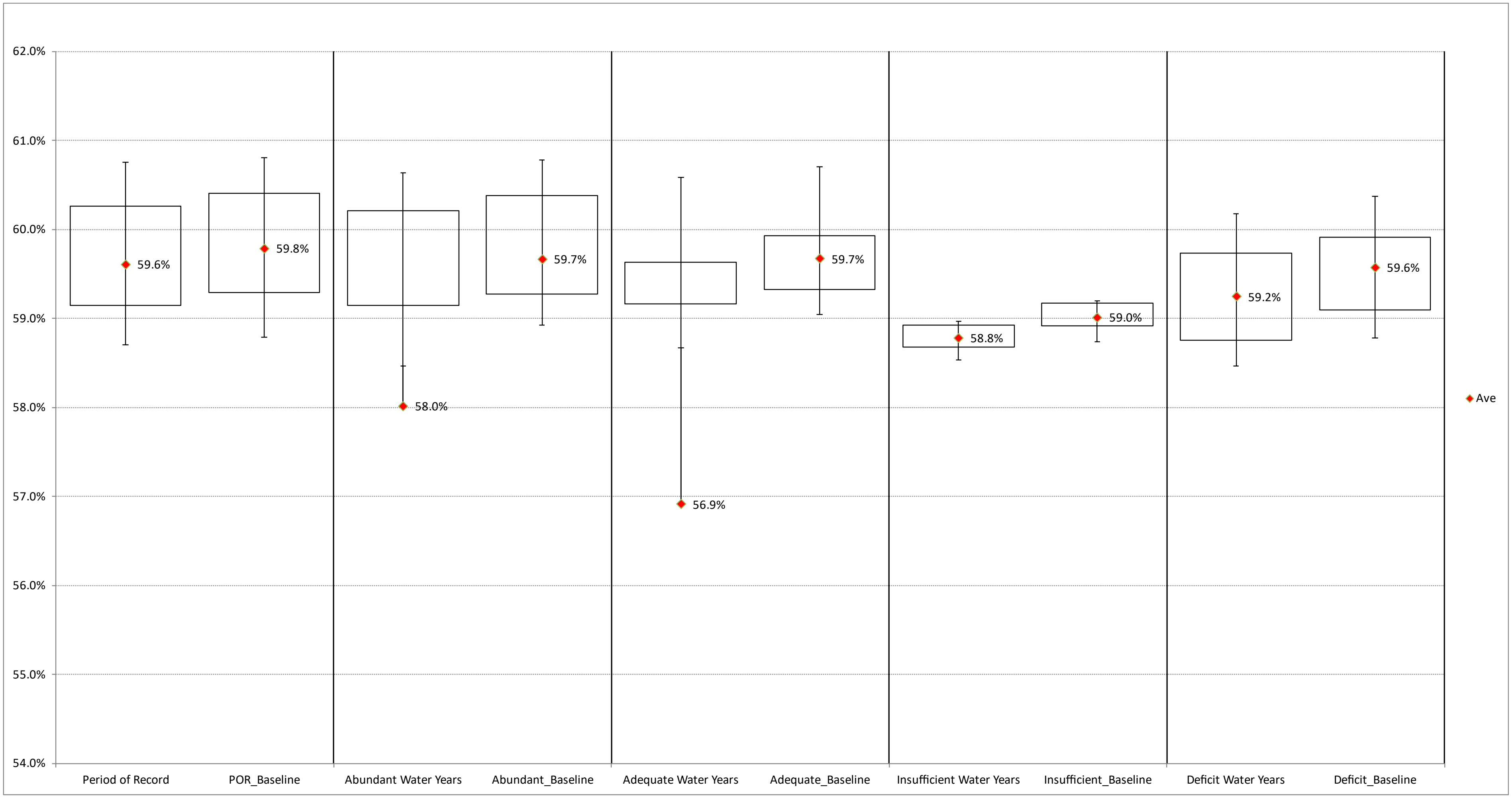

Figure 2-58. Foster Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Foster for juvenile spring Chinook fry under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

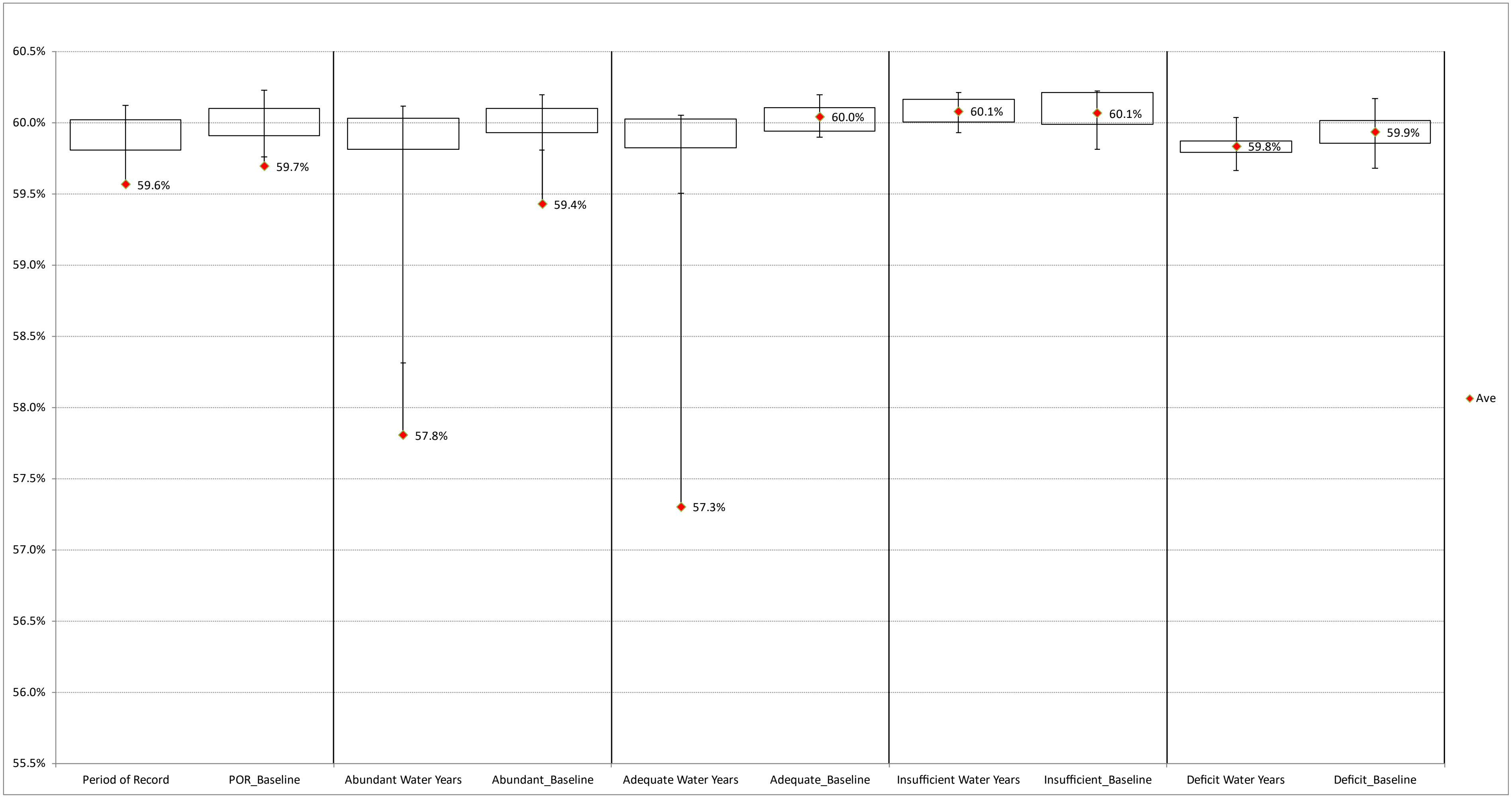

Figure 2-59. Foster Juvenile Spring Chinook Sub-Yearlings Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Foster for juvenile spring Chinook sub-yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

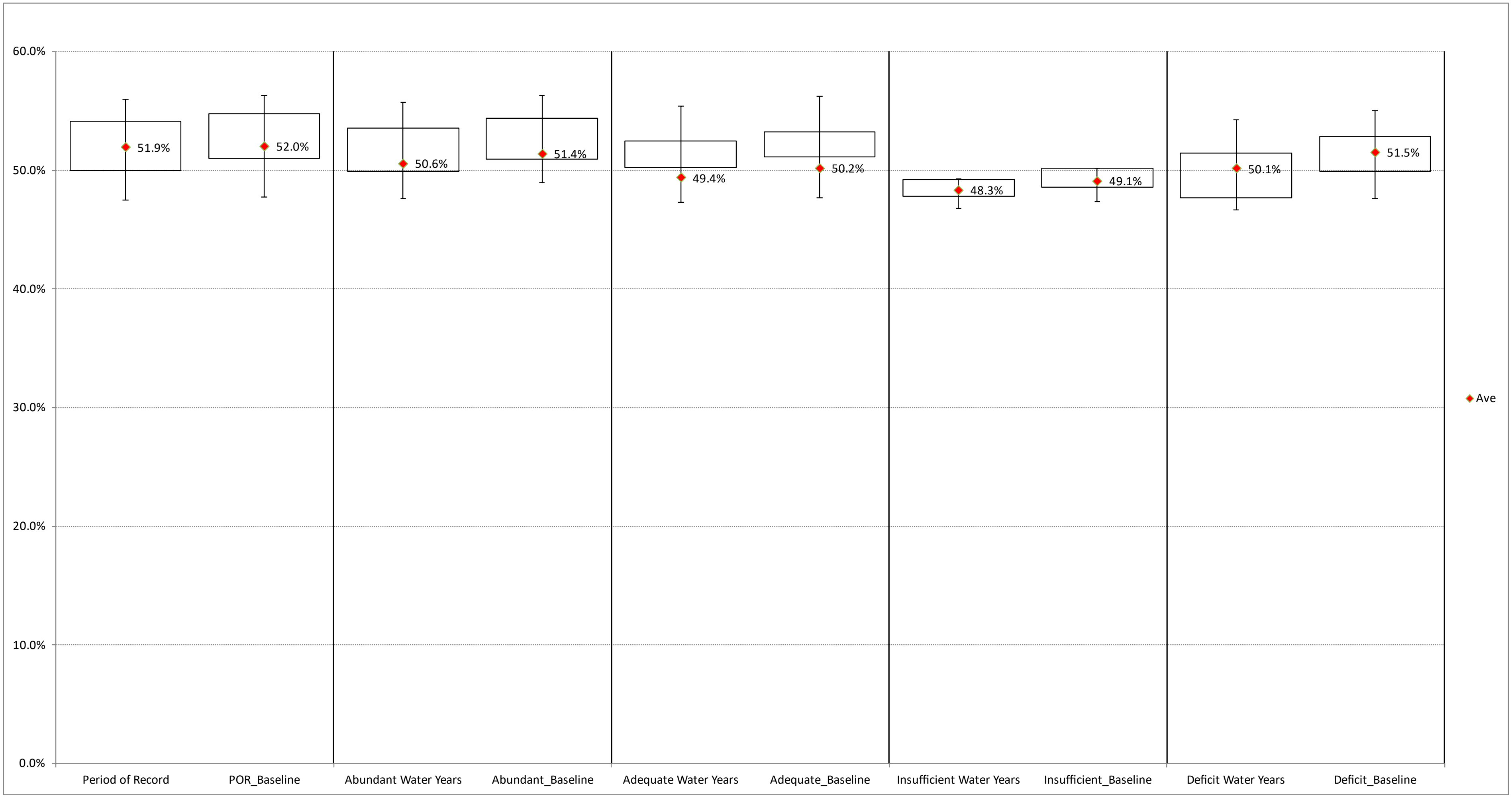

Figure 2-60. Foster For Juvenile Spring Chinook Yearlings Downstream Dam Passage Survival At Under Alternative 3a. *Downstream dam passage survival at Foster for juvenile spring Chinook yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.3.16 South Santiam – Green Peter

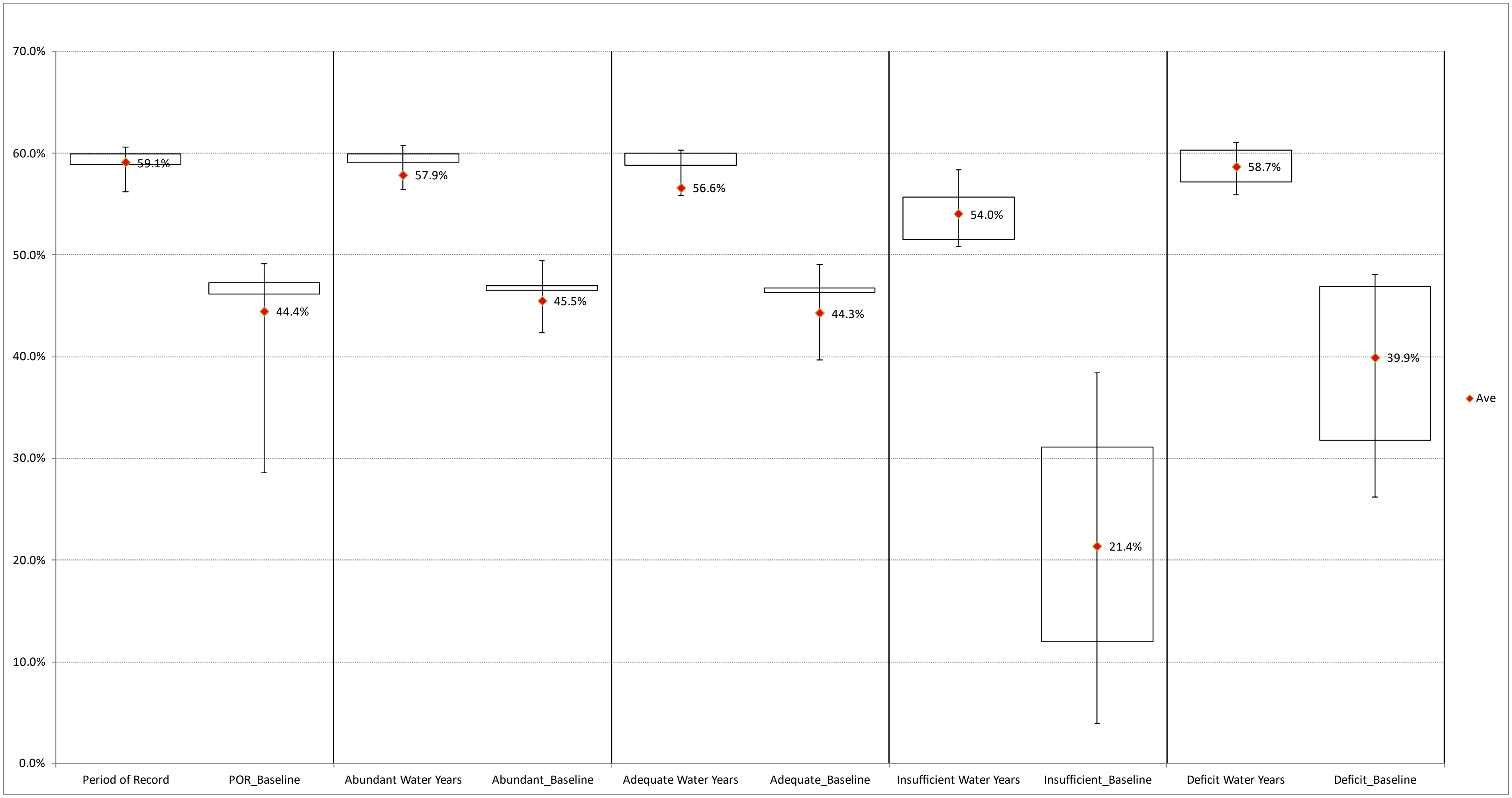

Figure 2-61. Green Peter Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Green Peter for juvenile spring Chinook fry under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

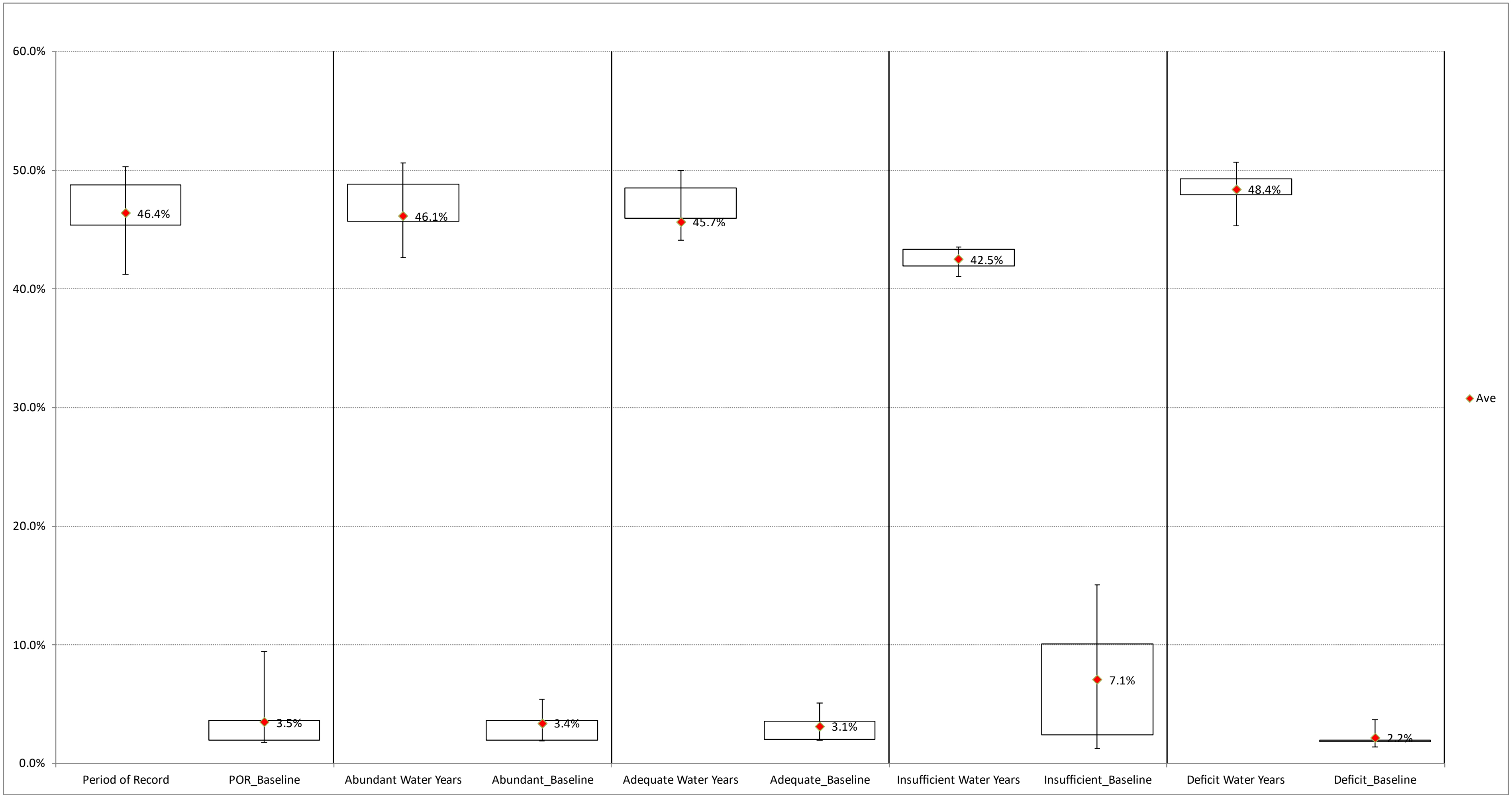

Figure 2-62. Green Peter Juvenile Spring Chinook Sub-Yearlings Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Green Peter for juvenile spring Chinook sub-yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

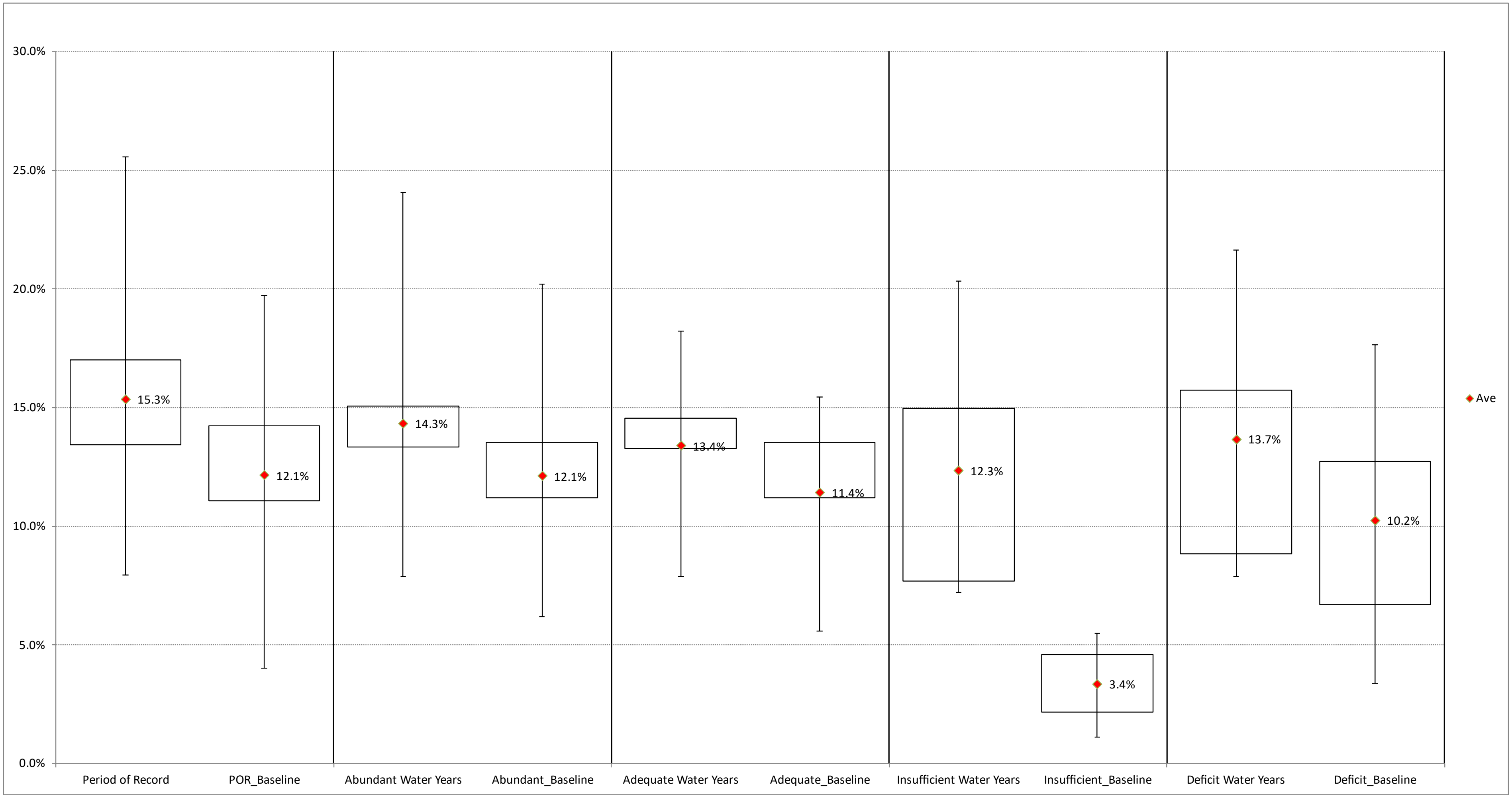

Figure 2-63. Green Peter Juvenile Spring Chinook Yearlings Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Green Peter for juvenile spring Chinook yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.3.17 McKenzie - Cougar

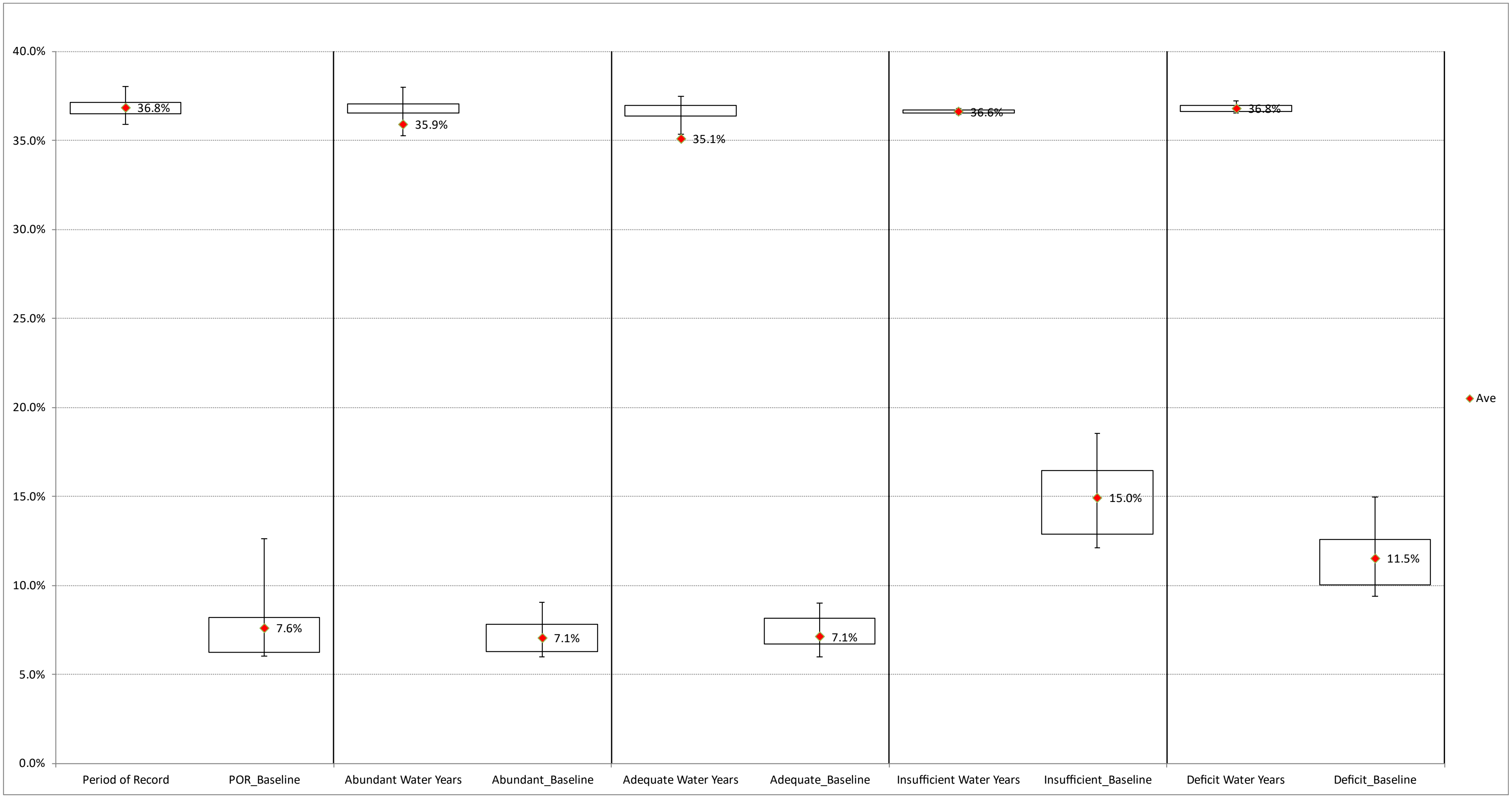

Figure 2-64. Cougar Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Cougar for juvenile spring Chinook fry under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

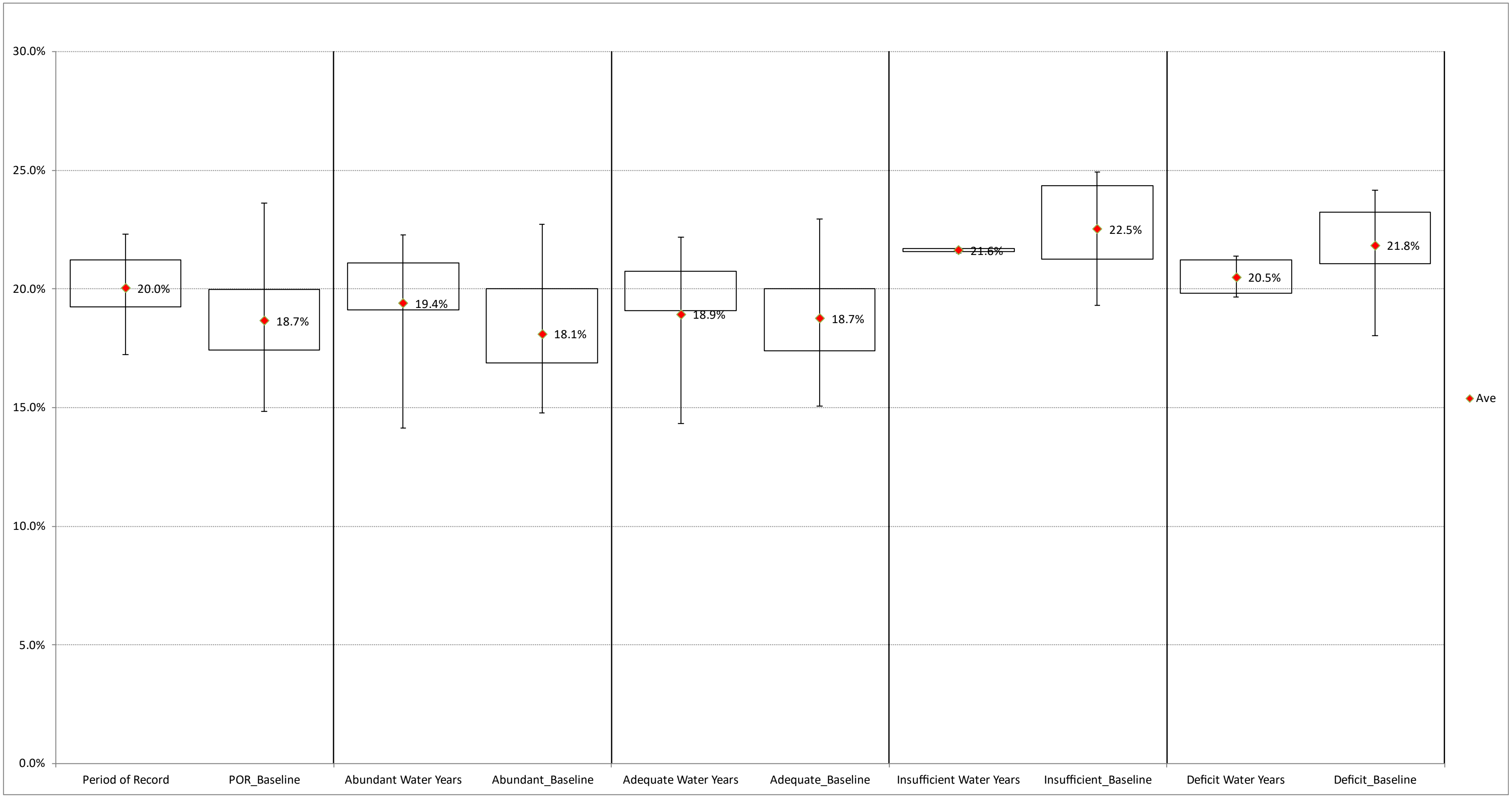

Figure 2-65. Cougar Juvenile Spring Chinook Sub-Yearlings Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Cougar for juvenile spring Chinook sub-yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

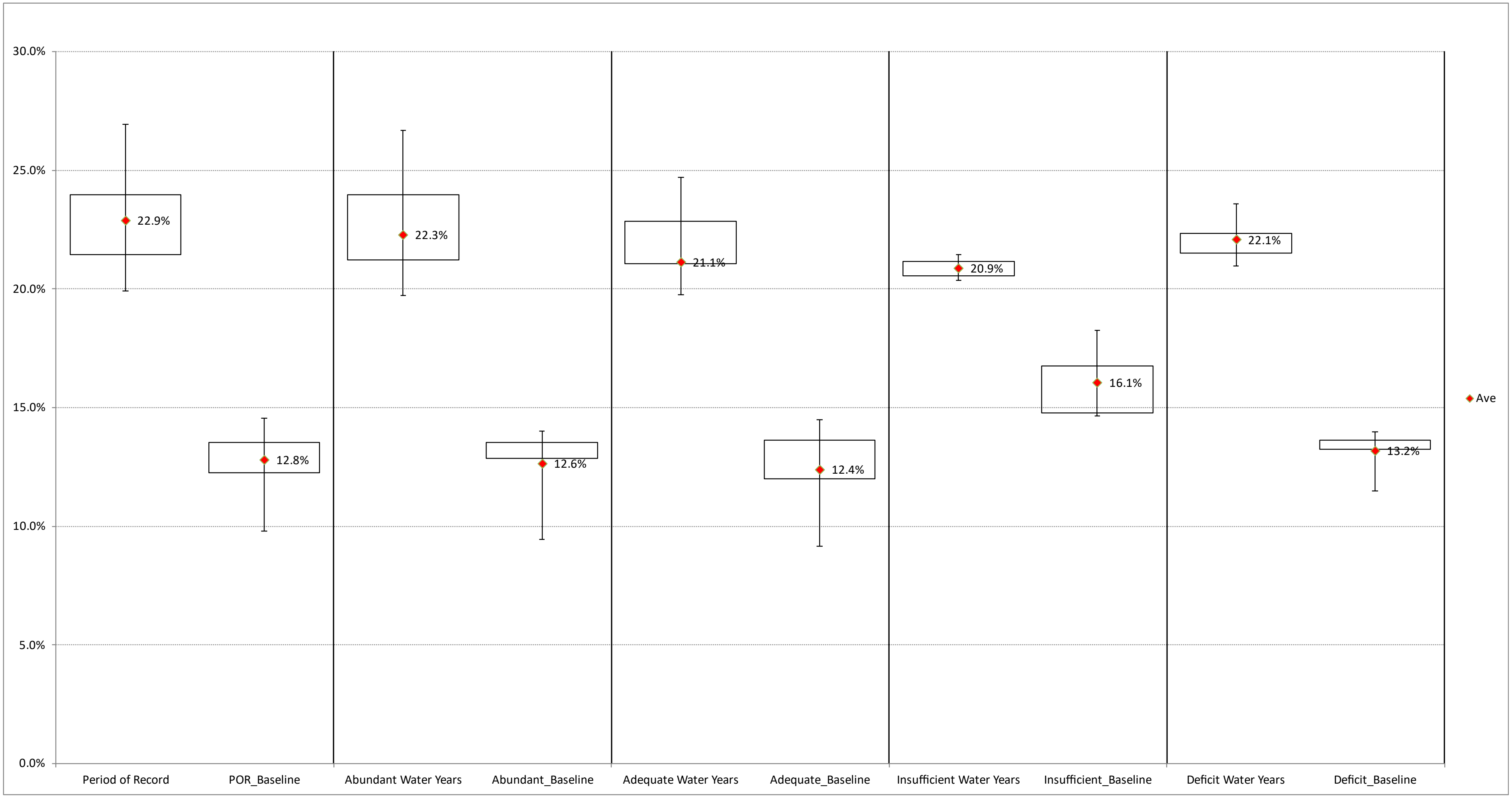

Figure 2-66. Cougar Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Cougar for juvenile spring Chinook yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.3.18 Middle Fork – Lookout Point

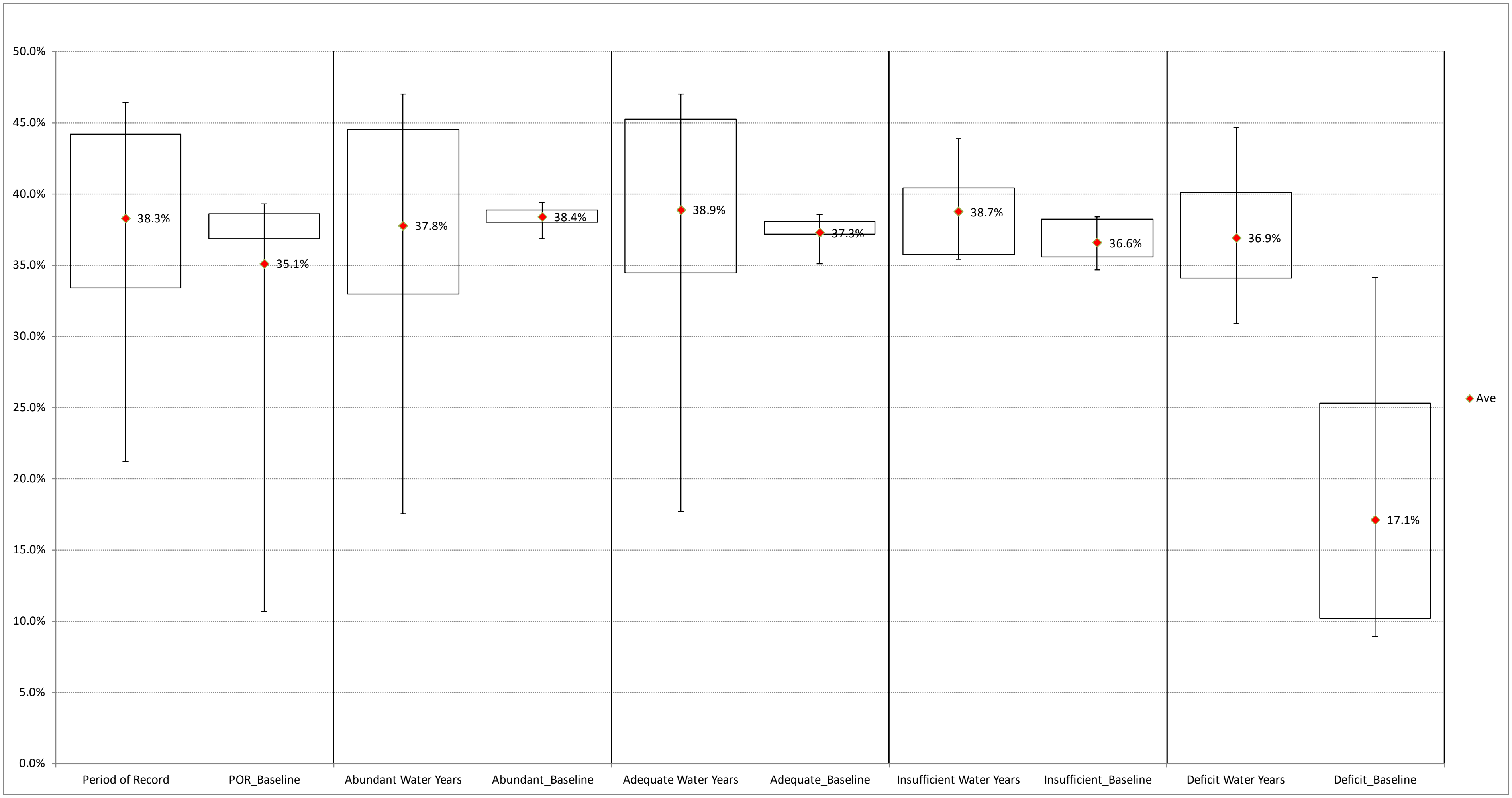

Figure 2-67. Lookout Point Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook fry under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

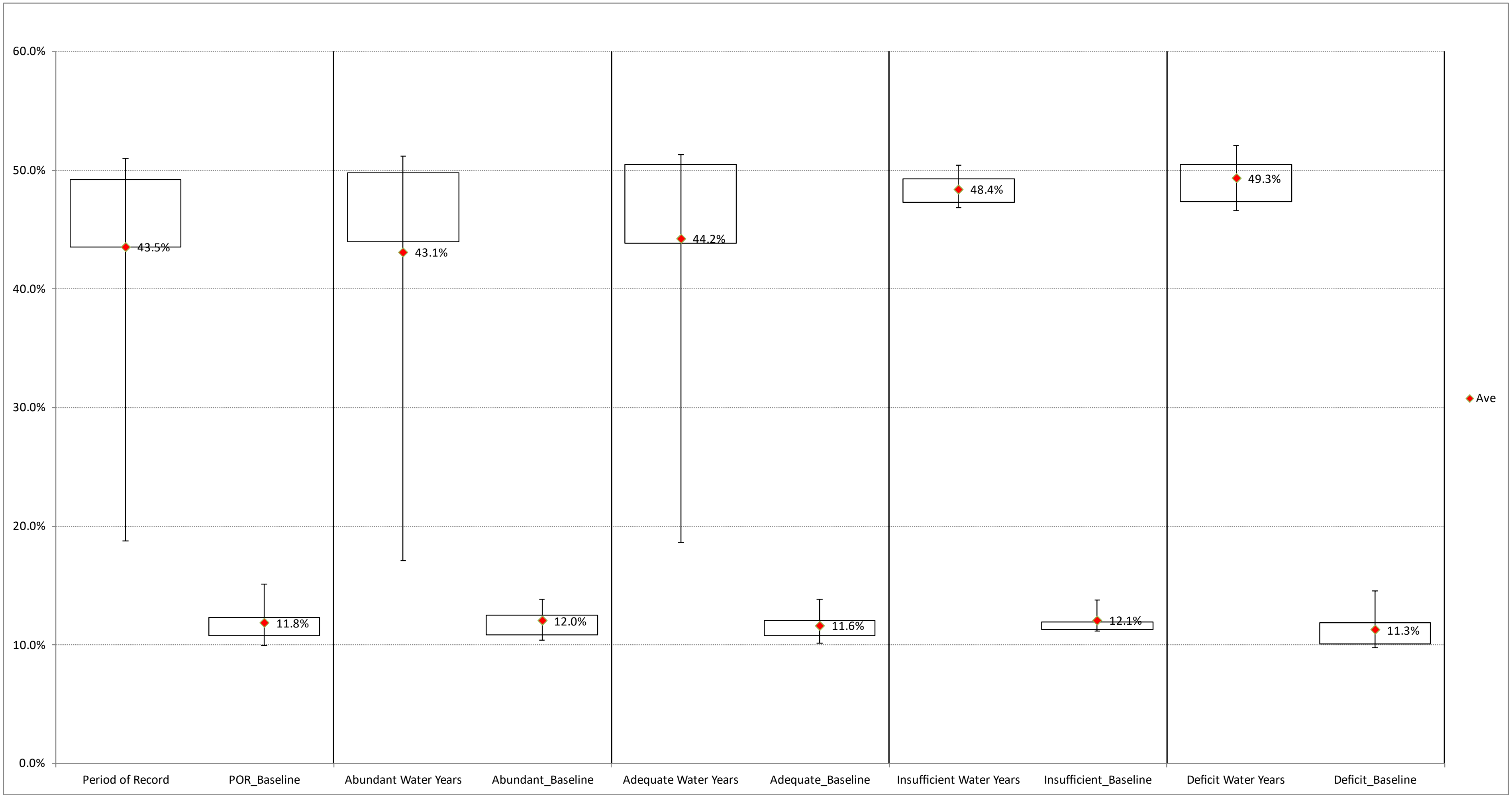

Figure 2-68. Lookout Point Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook sub-yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

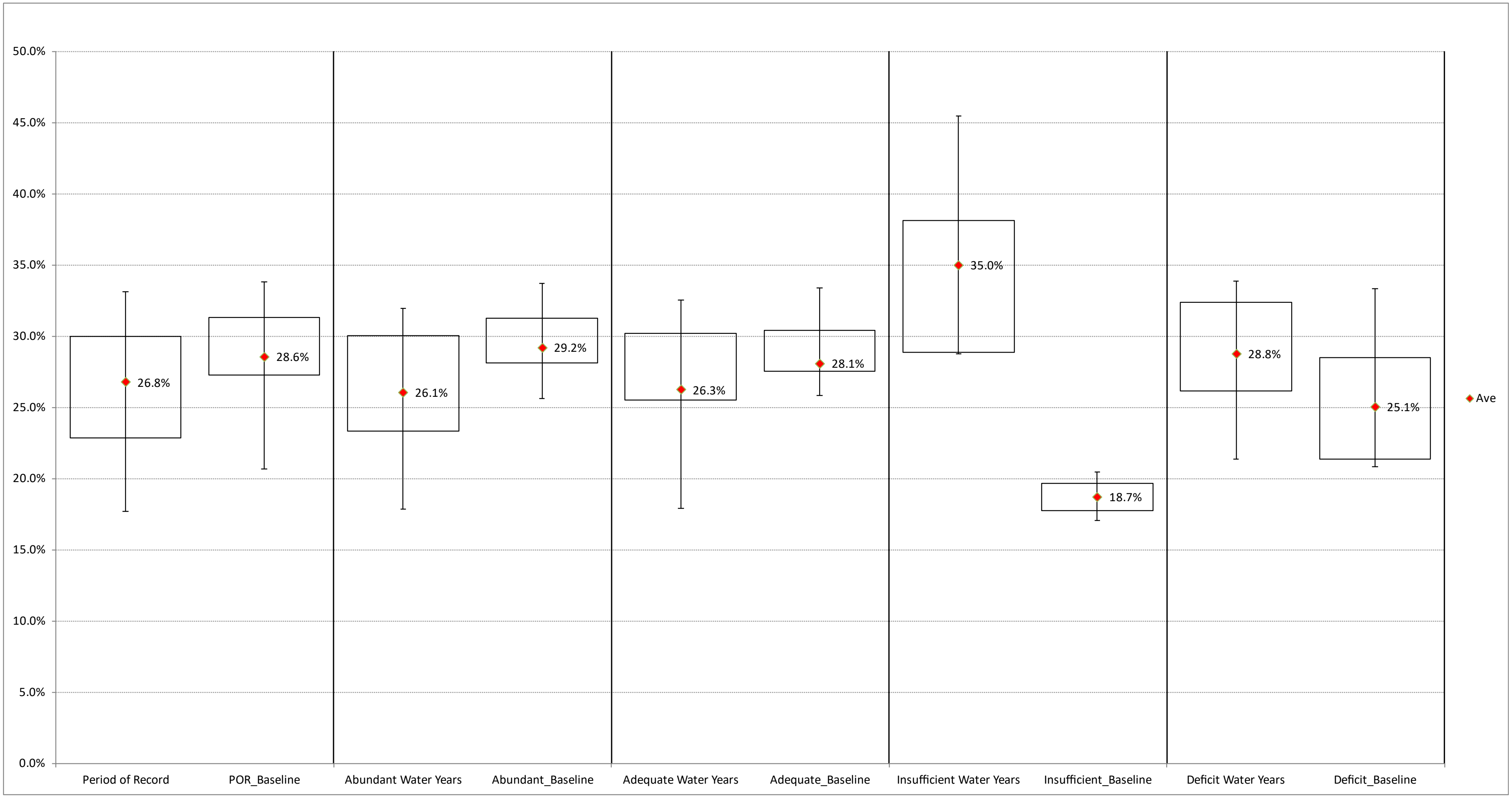

Figure 2-69. Lookout Point Juvenile Spring Chinook Yearlings Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

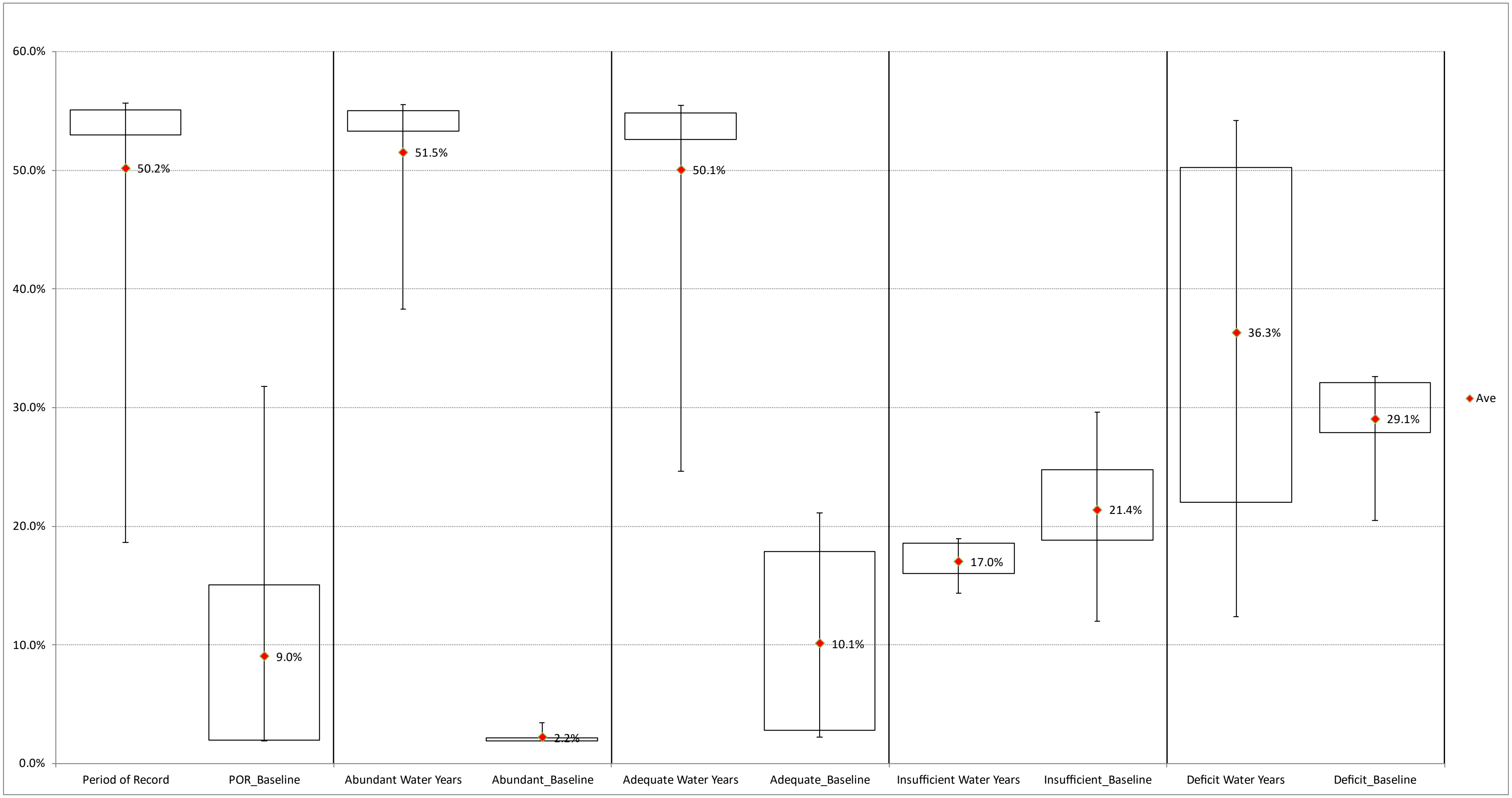

Figure 2-70. Hills Creek Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook fry under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

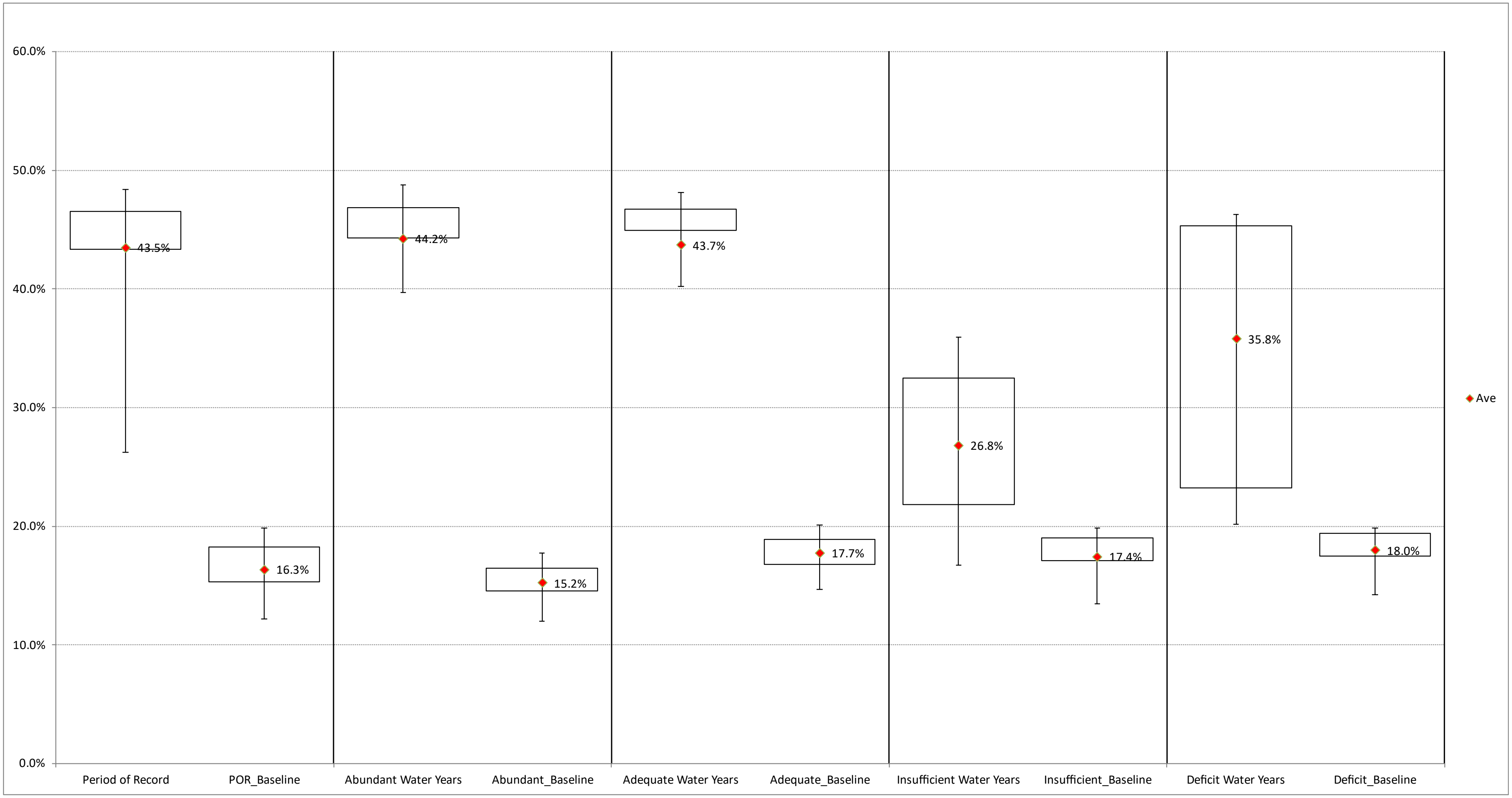

Figure 2-71. Hills Creek Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 3a. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook sub-yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

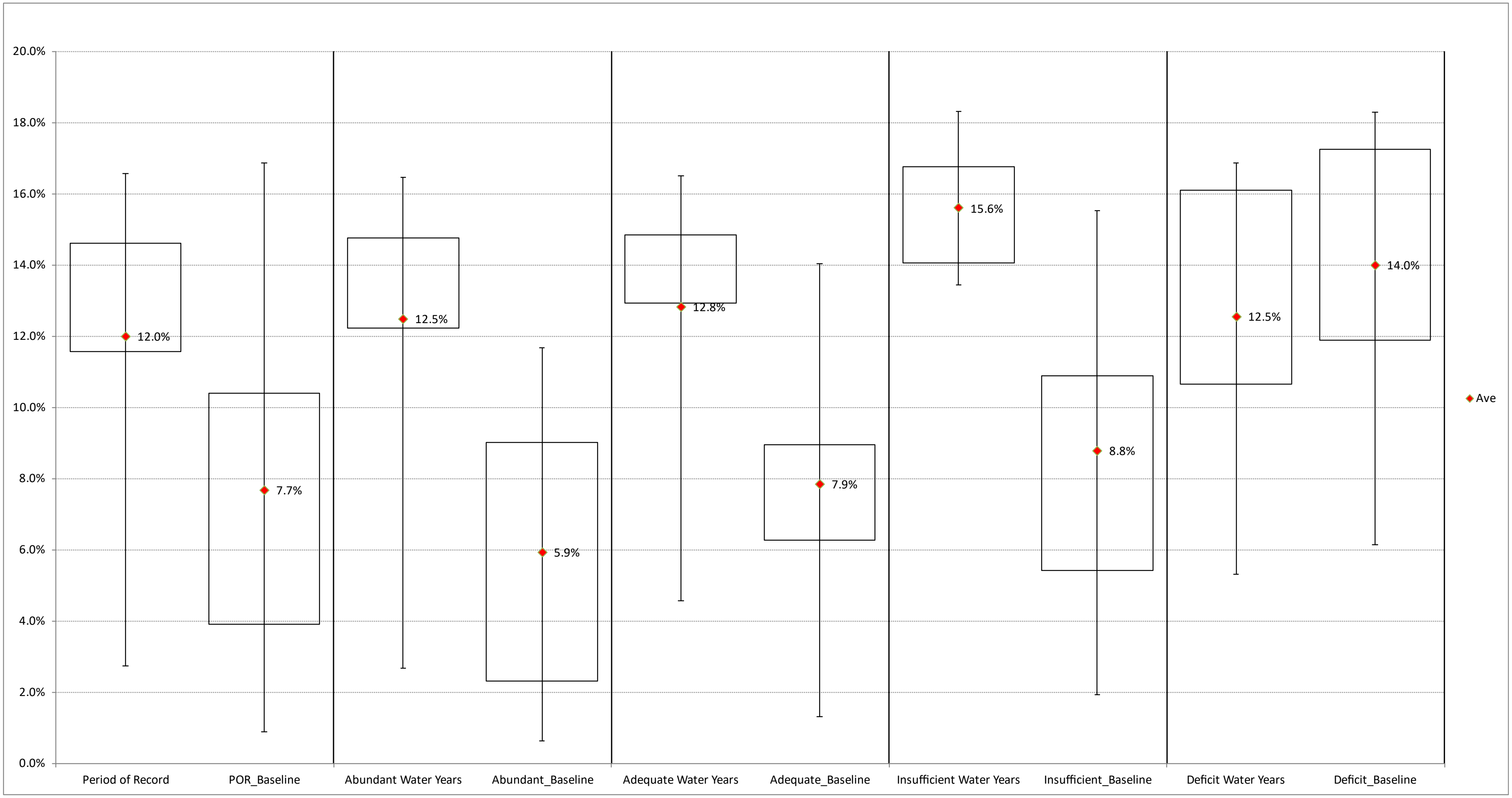

Figure 2-72. Downstream dam passage survival at Hills Creek for juvenile spring Chinook yearlings under Alternative 3a. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

CHINOOK

2.4 CHINOOK ALTERNATIVE 3B

2.4.1 North Santiam – Detroit

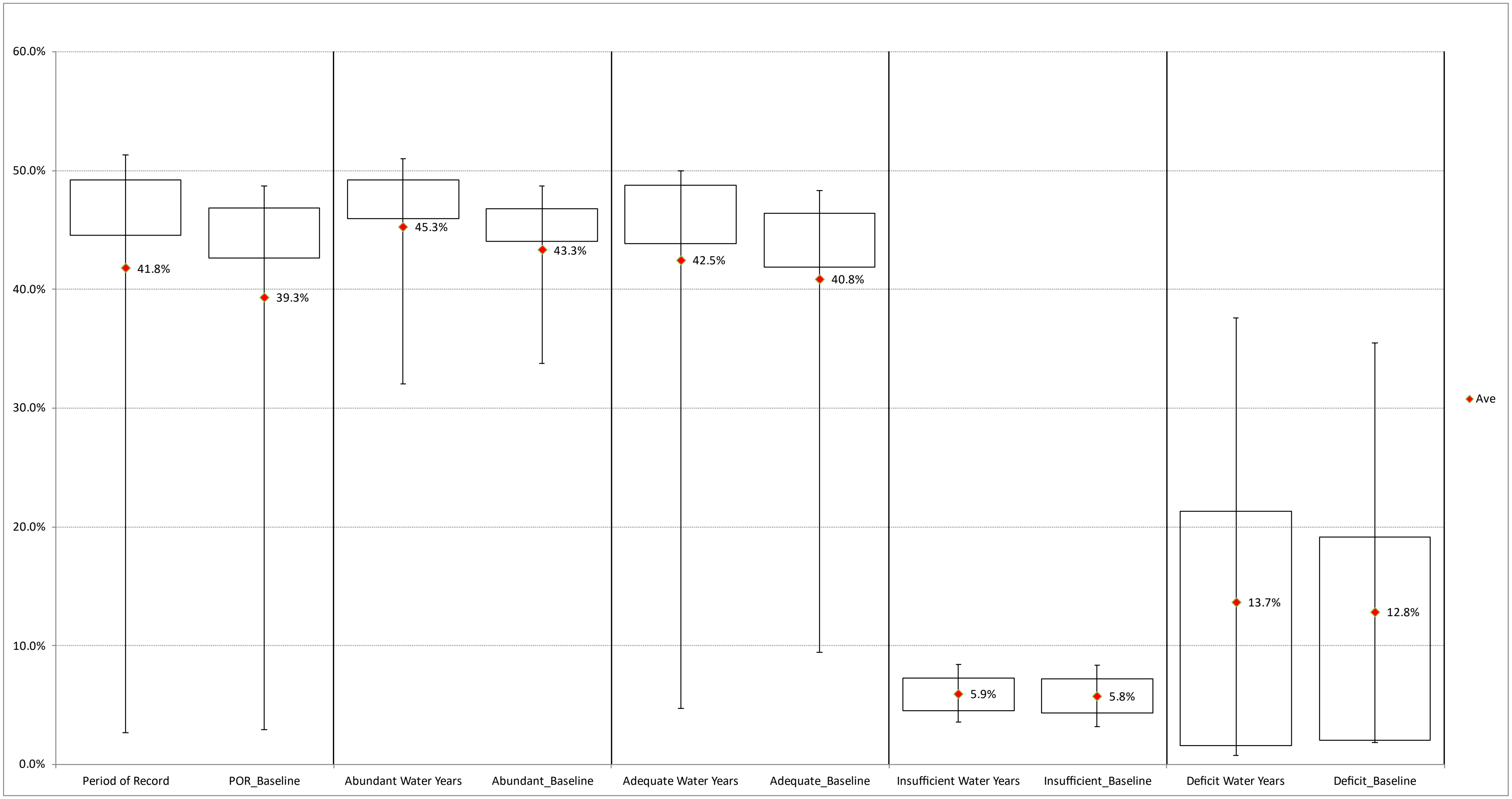

Figure 2-73. Detroit Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Detroit for juvenile spring Chinook fry under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

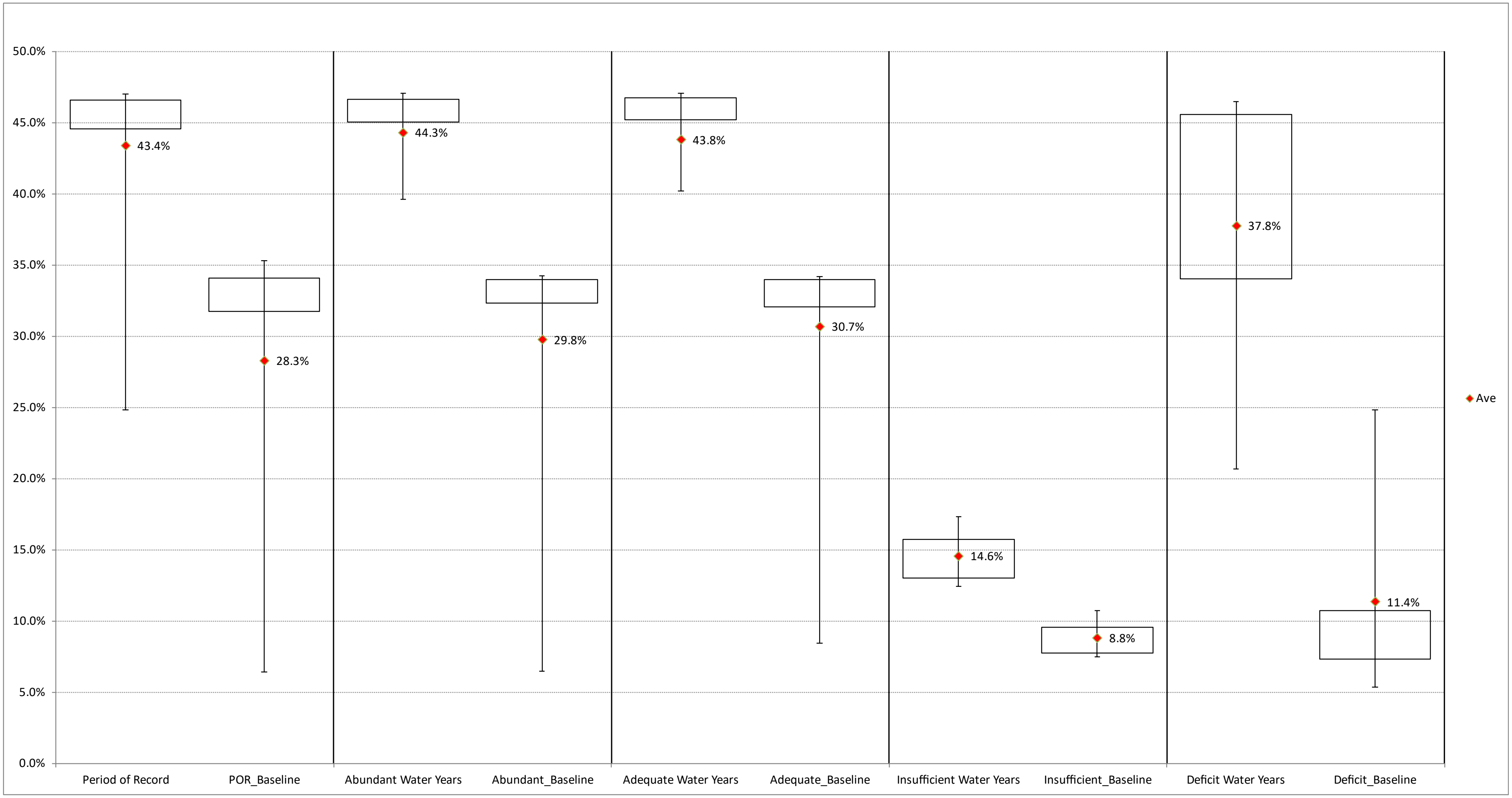

Figure 2-74. Detroit Juvenile Spring Chinook Sub-Yearlings Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Detroit for juvenile spring Chinook sub-yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

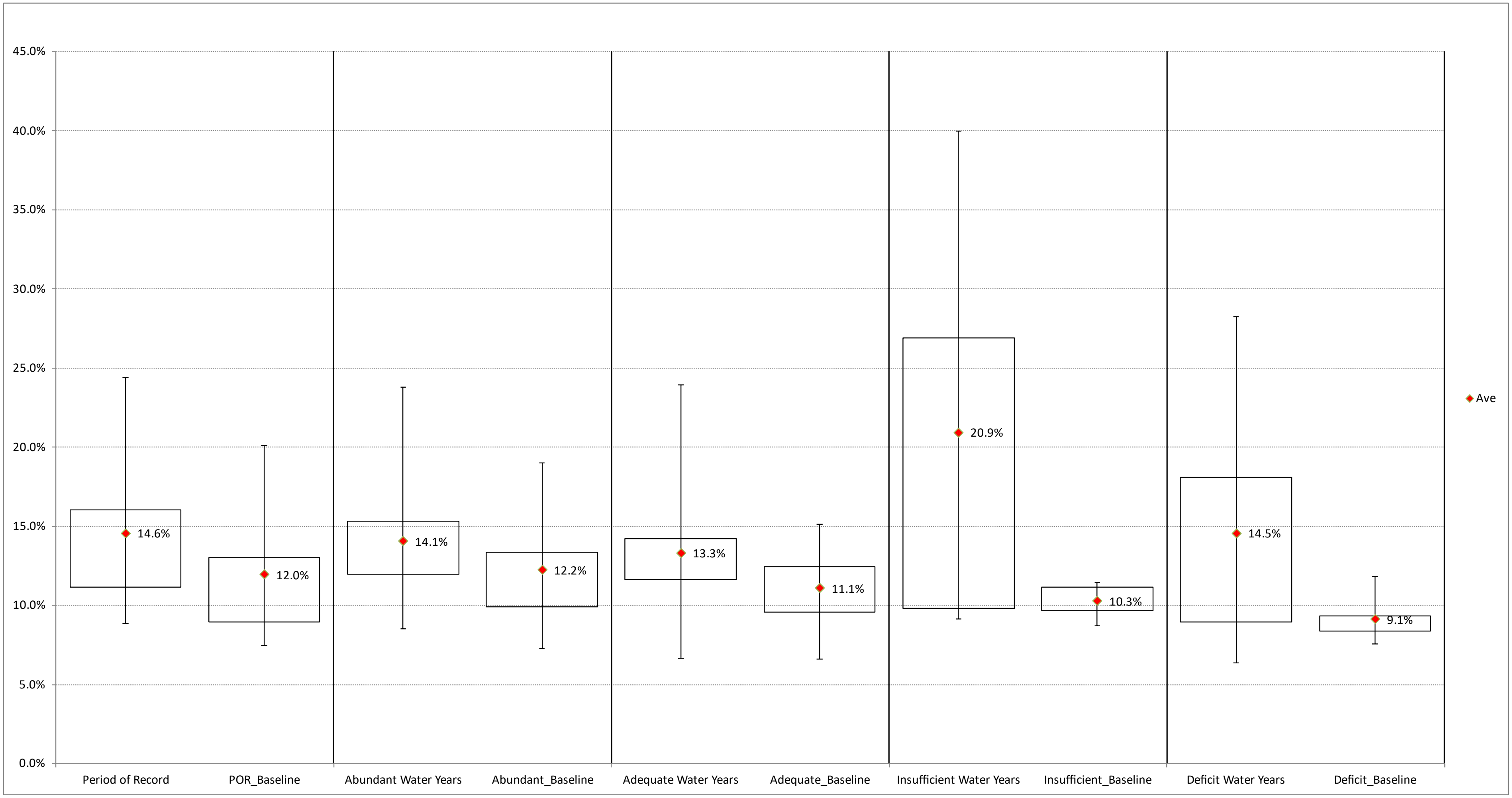

Figure 2-75. Detroit Juvenile Spring Chinook Yearlings Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Detroit for juvenile spring Chinook yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.4.2 South Santiam – Foster

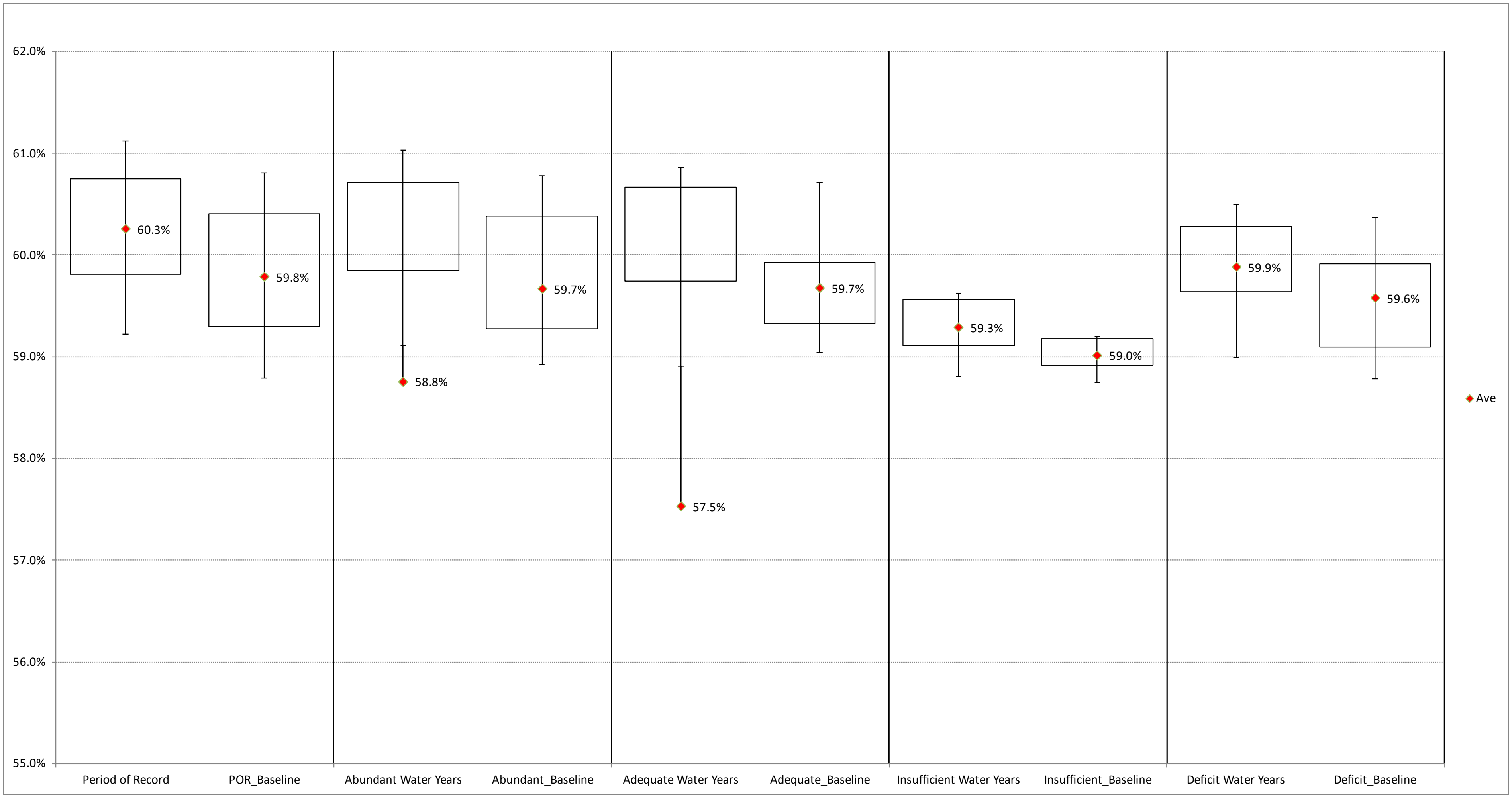

Figure 2-76. Foster Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Foster for juvenile spring Chinook fry under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

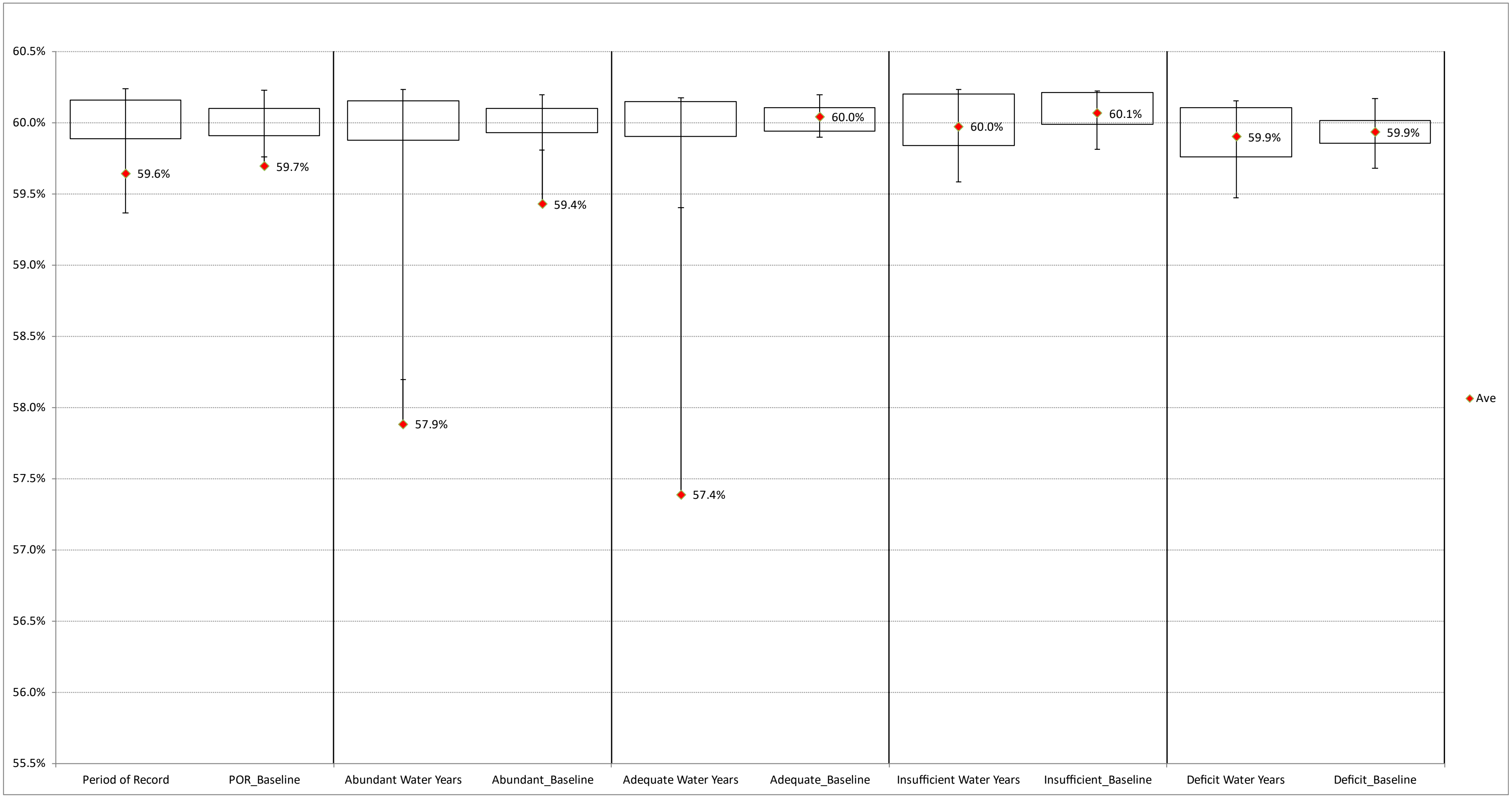

Figure 2-77. Foster Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Foster for juvenile spring Chinook sub-yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

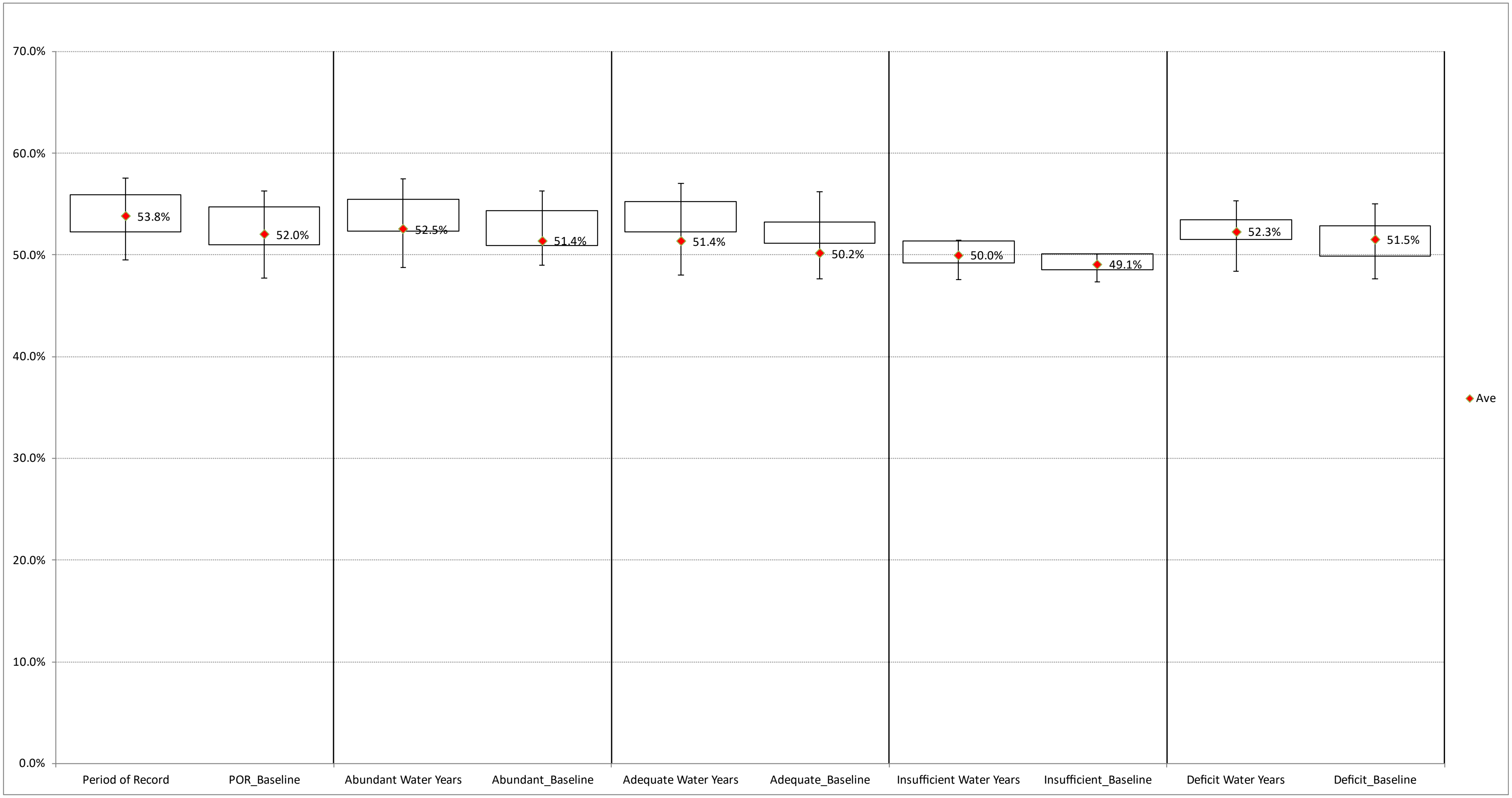

Figure 2-78. Downstream dam passage survival at Foster for juvenile spring Chinook yearlings under Alternative 3b. *Downstream dam passage survival at Foster for juvenile spring Chinook yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.4.3 McKenzie – Cougar

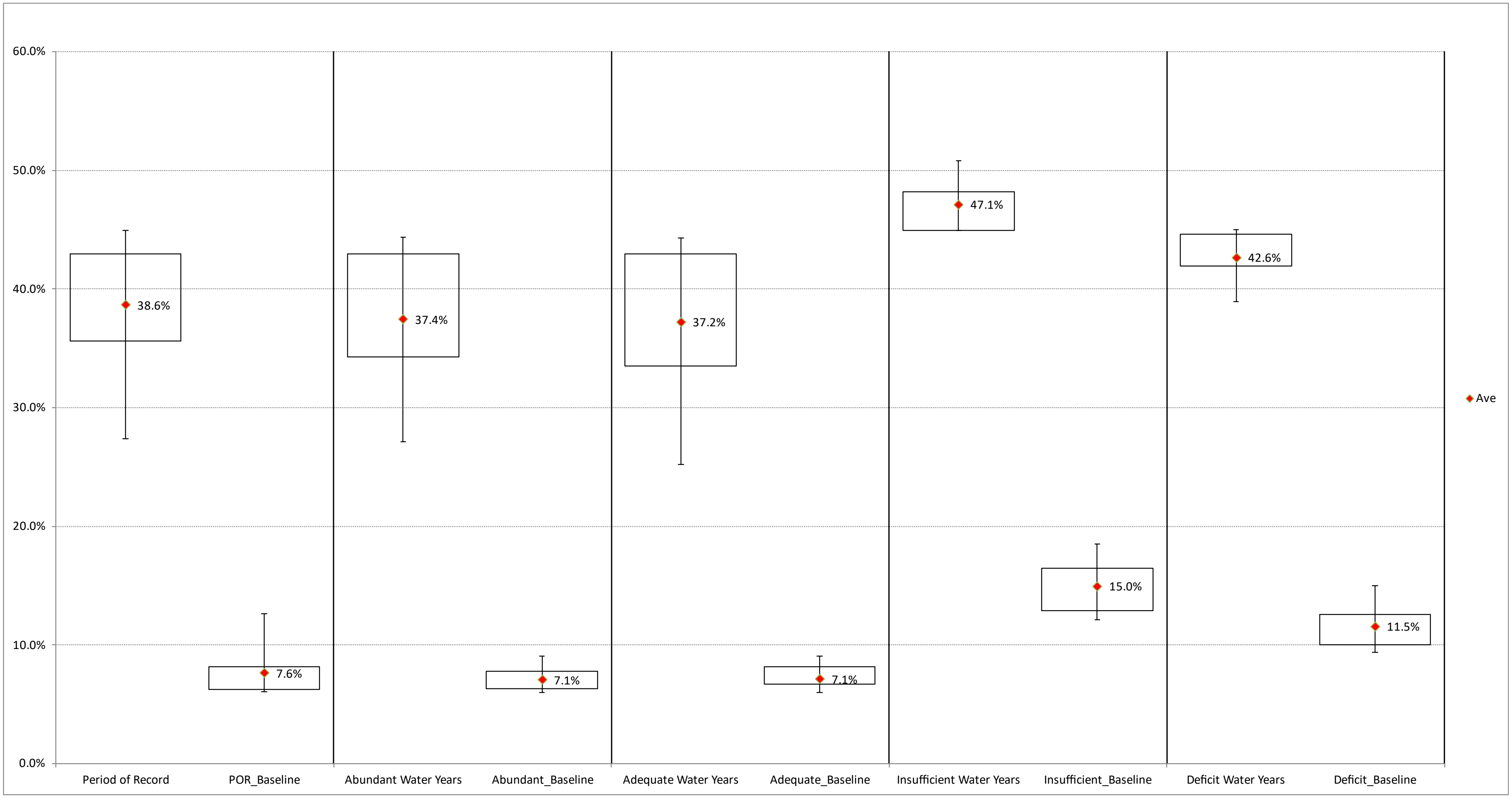

Figure 2-79. Cougar Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Cougar for juvenile spring Chinook fry under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

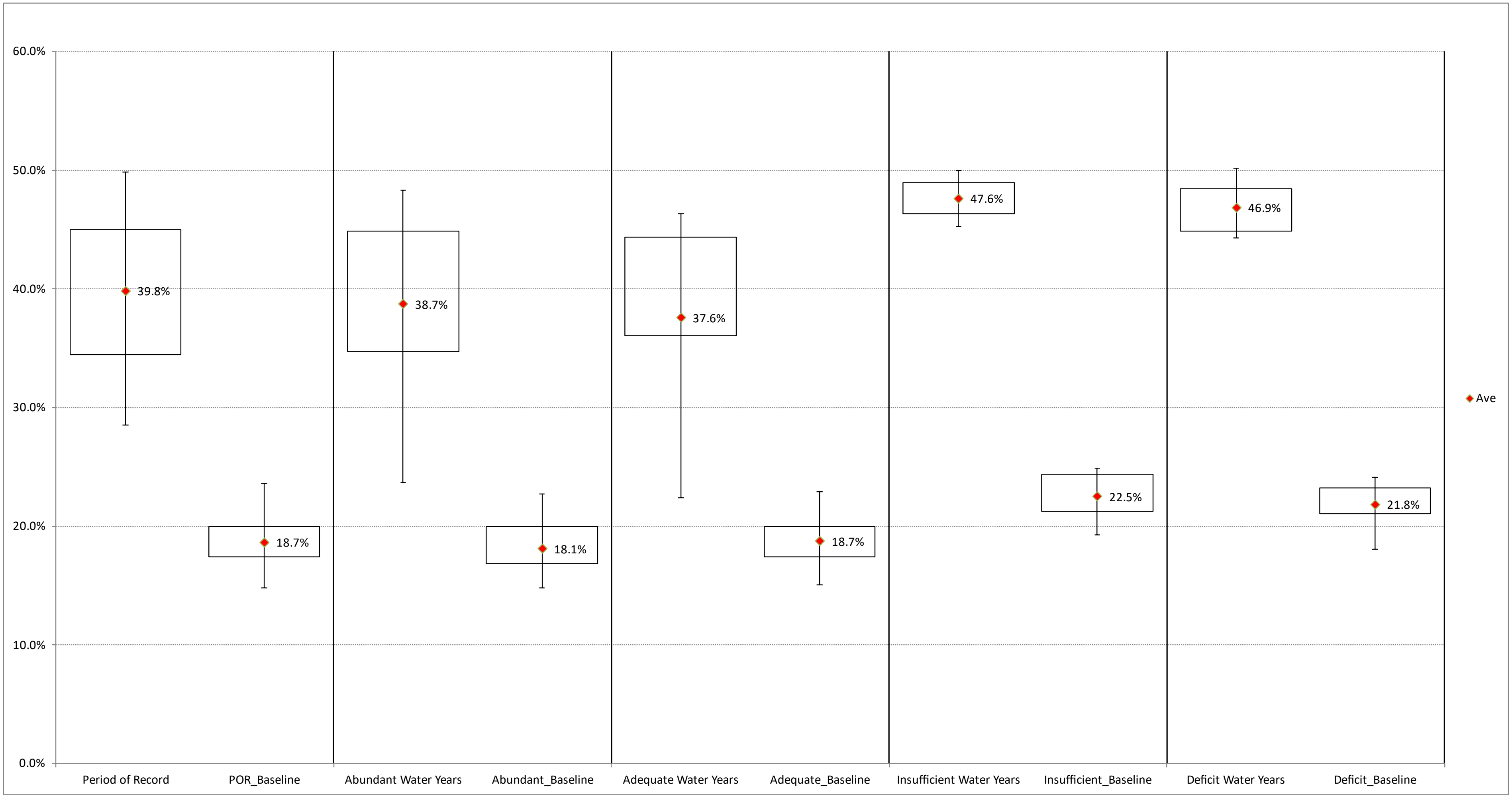

Figure 2-80. Cougar Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Cougar for juvenile spring Chinook sub-yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

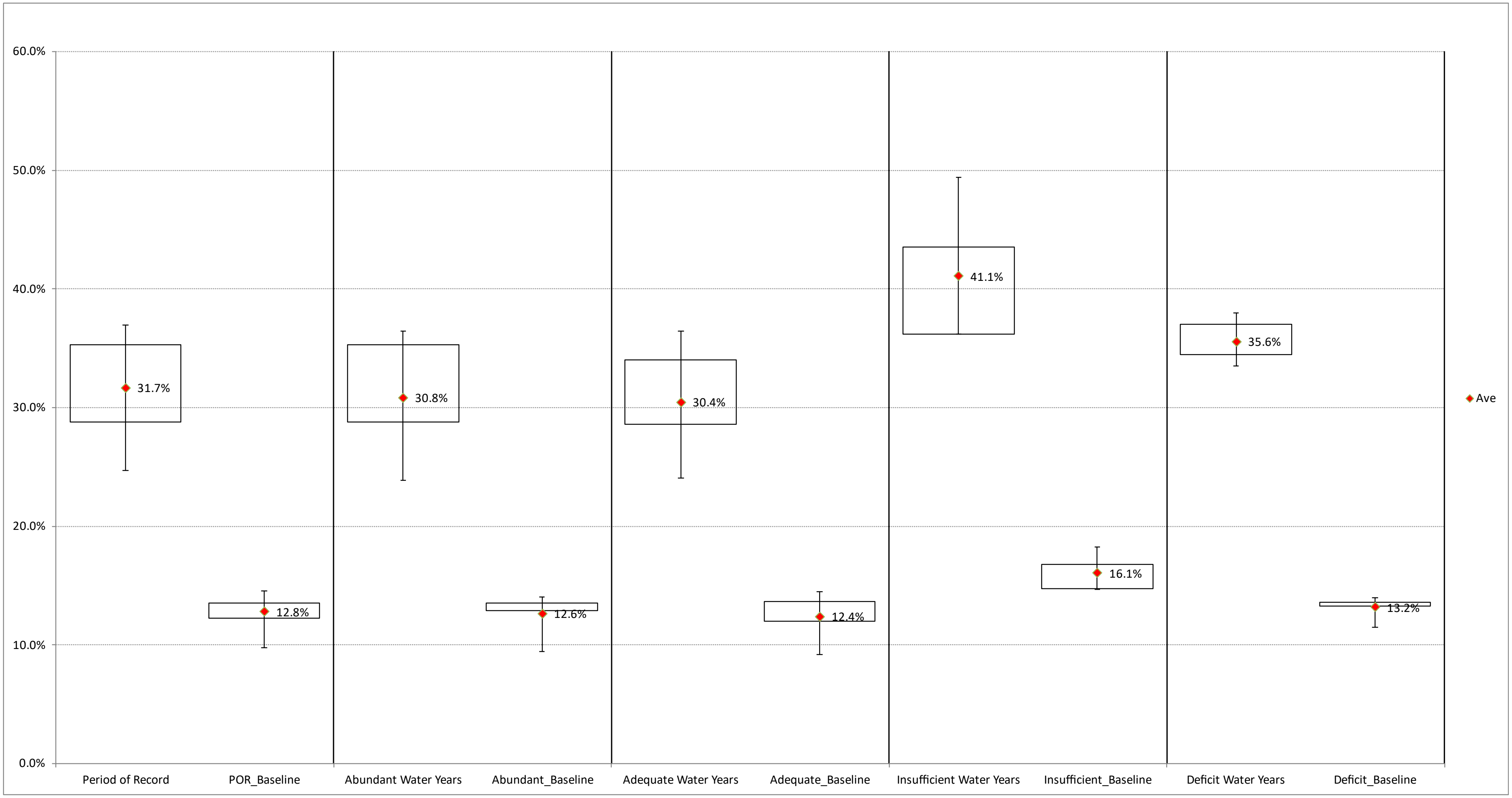

Figure 2-81. Cougar Juvenile Spring Chinook Yearlings Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Cougar for juvenile spring Chinook yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.4.4 Middle Fork – Lookout Point

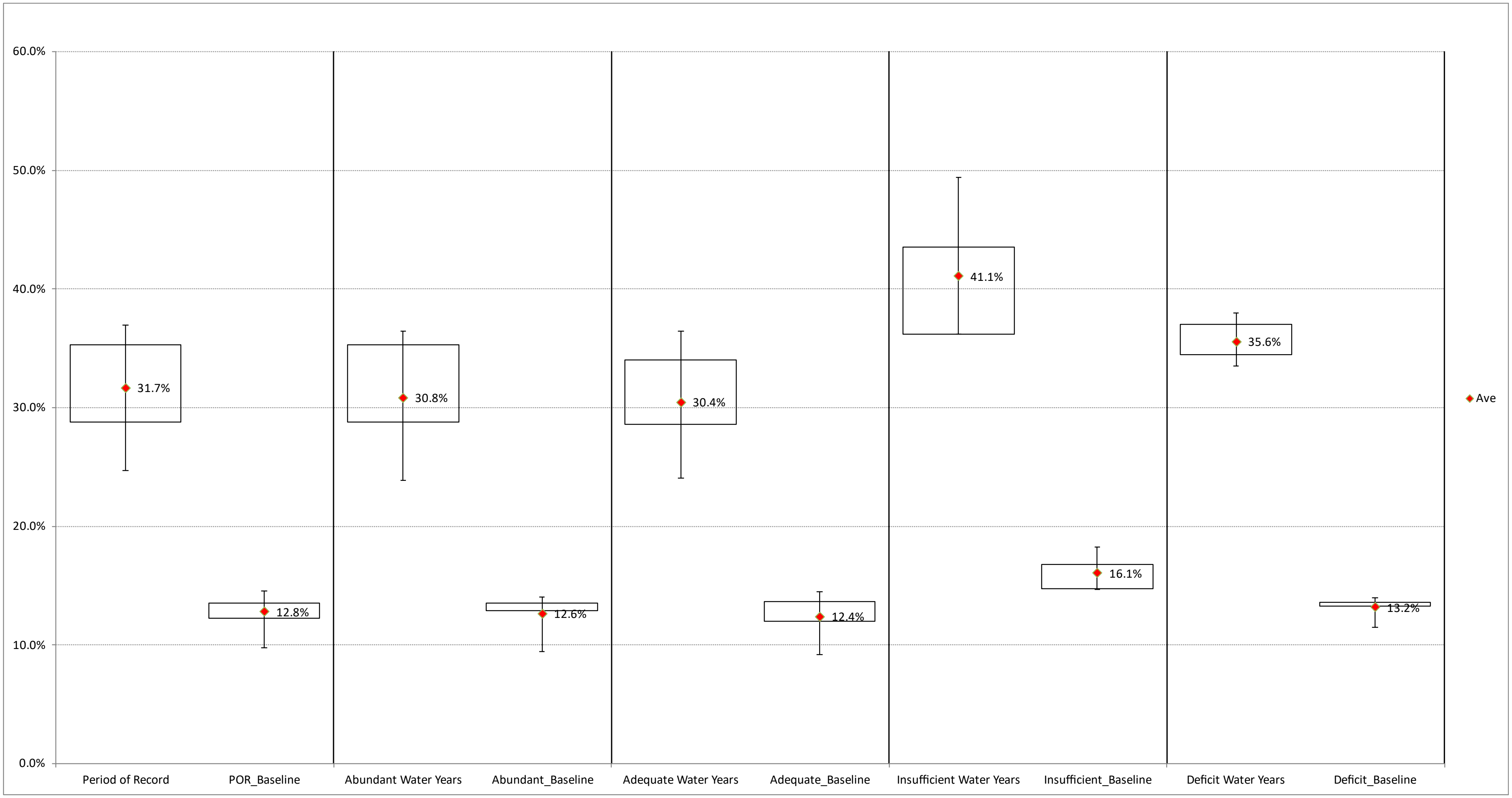

Figure 2-82. Lookout Point Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook fry under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

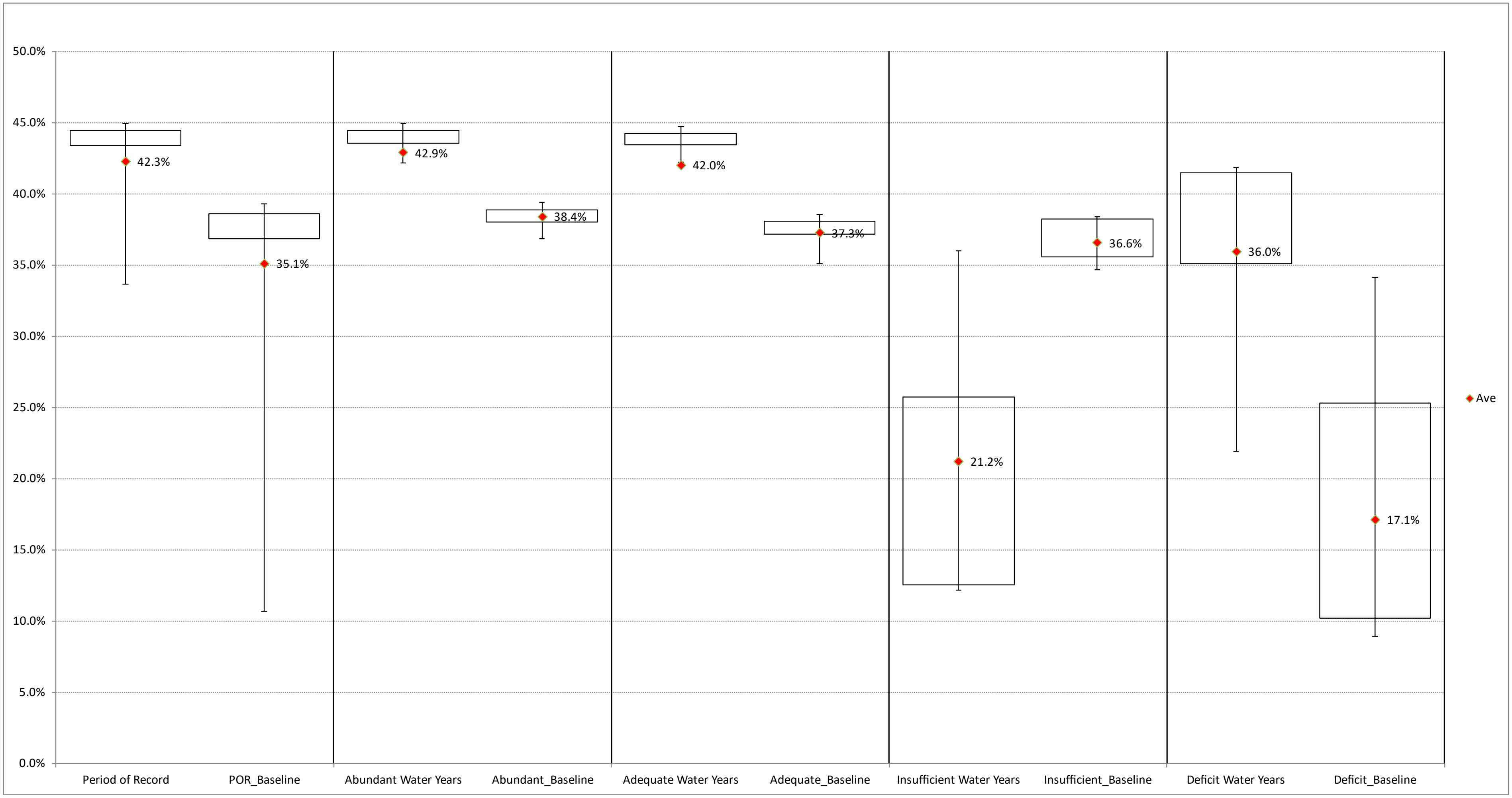

Figure 2-83. Lookout Point Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook sub-yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

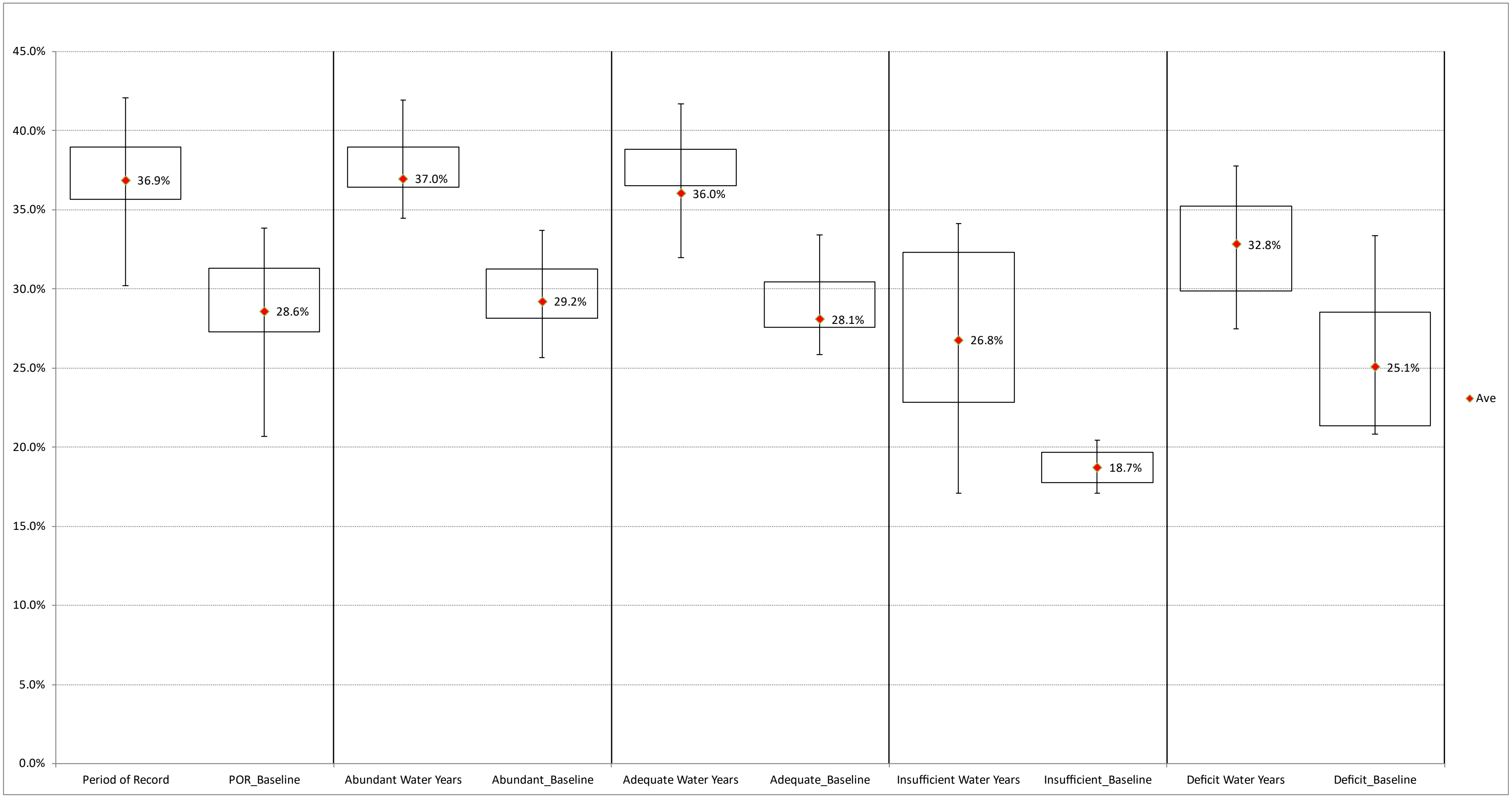

Figure 2-84. Lookout Point for juvenile spring Chinook yearling Downstream dam passage survival under Alternative 3b. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

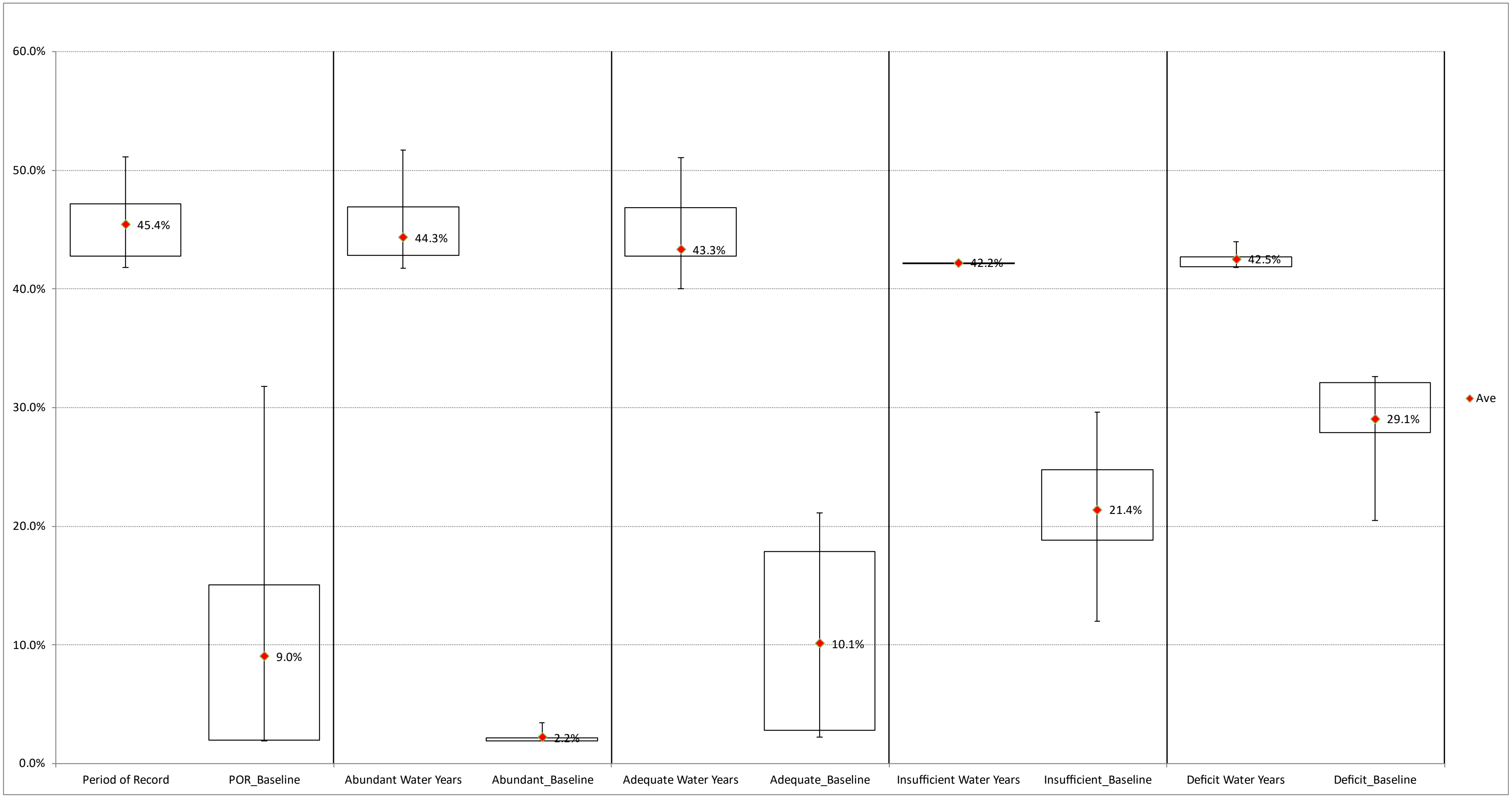

Figure 2-85. Hills Creek Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook fry under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

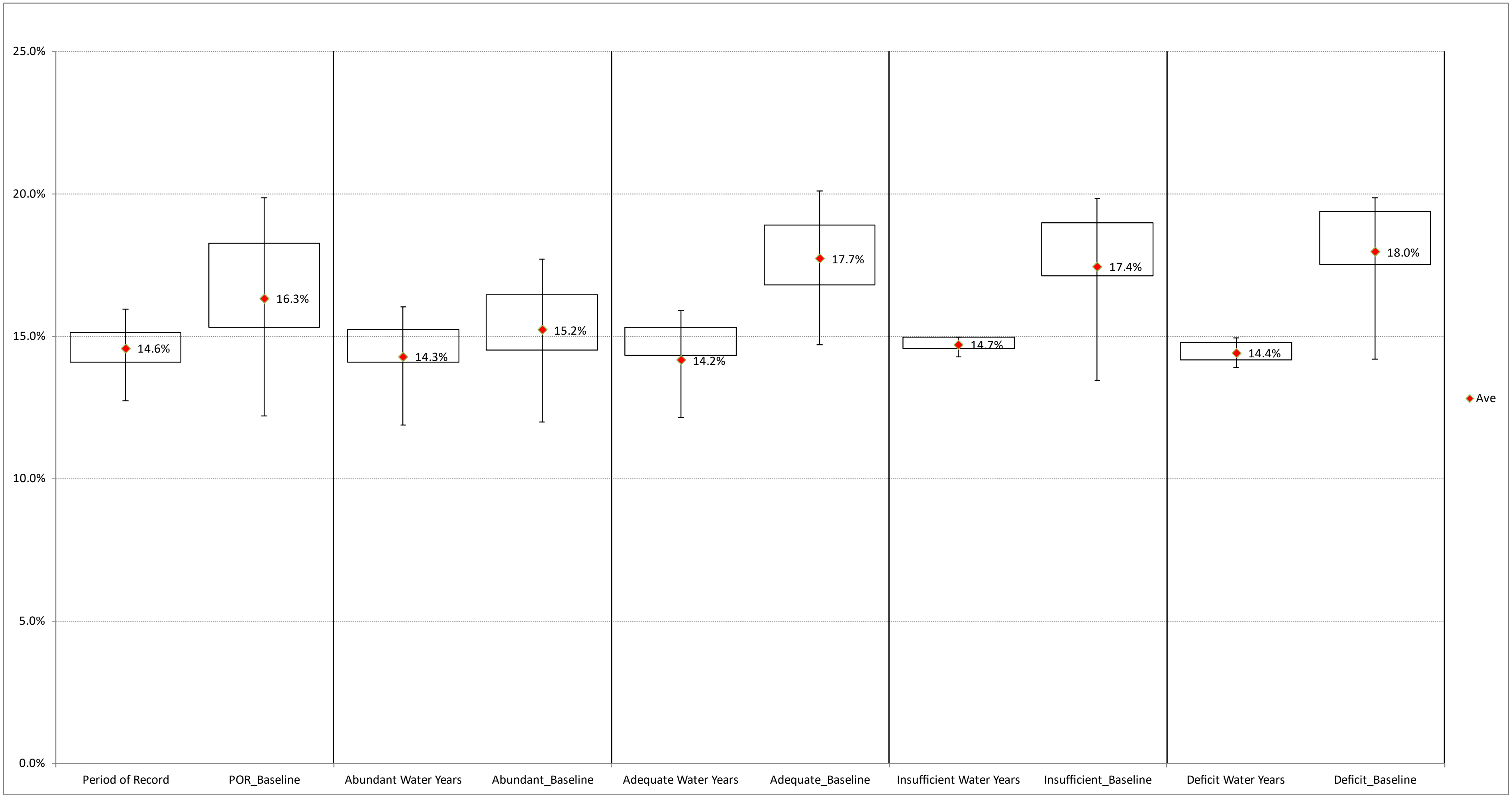

Figure 2-86. Hills Creek For Juvenile Spring Chinook Sub-Yearlings Downstream Dam Passage Survival At Under Alternative 3b. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook sub-yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

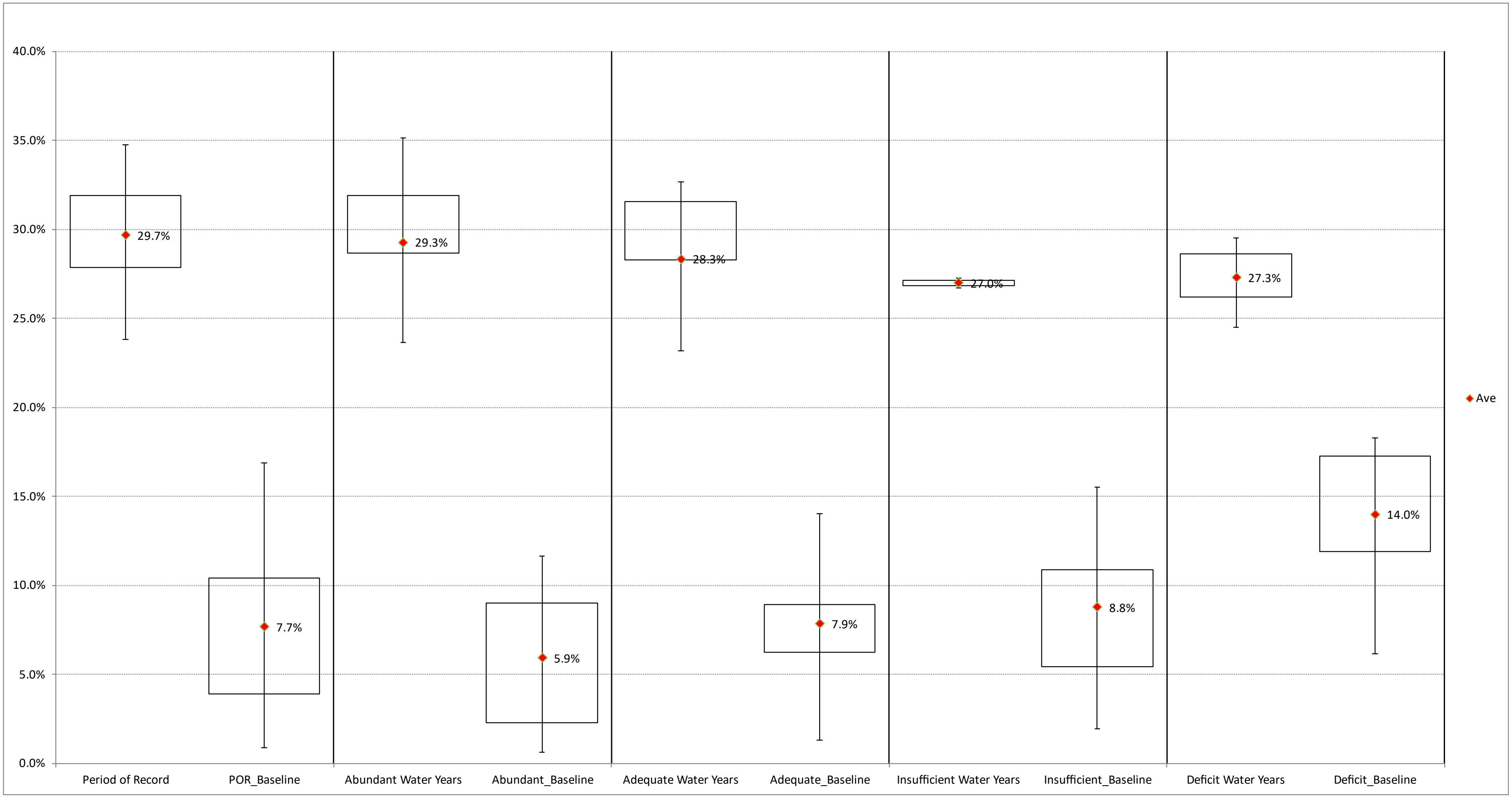

Figure 2-87. Hills Creek Juvenile Spring Chinook Yearlings Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

CHINOOK

2.5 CHINOOK ALTERNATIVE 4

2.5.1 North Santiam – Detroit

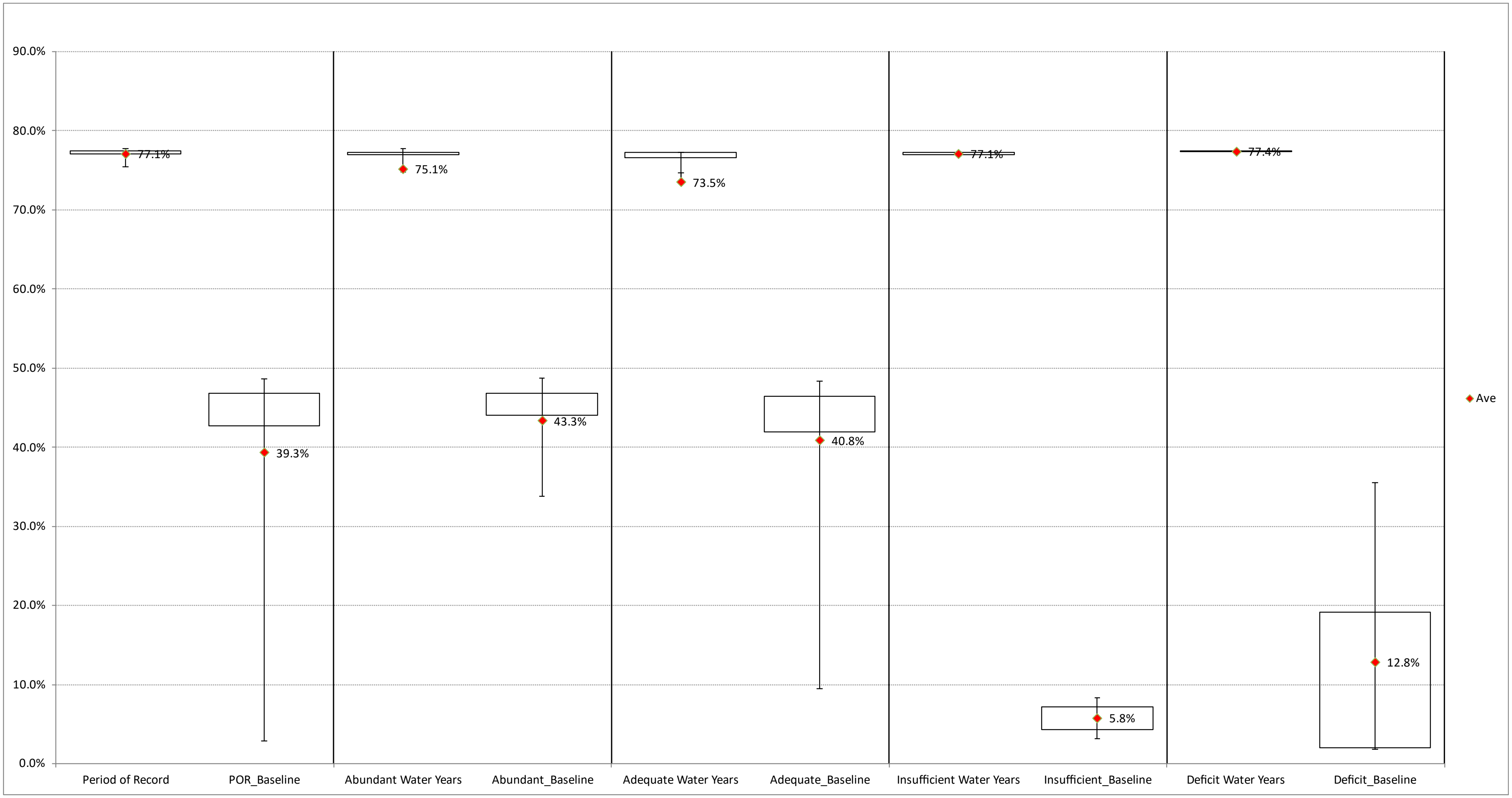

Figure 2-88. Detroit Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Detroit for juvenile spring Chinook fry under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

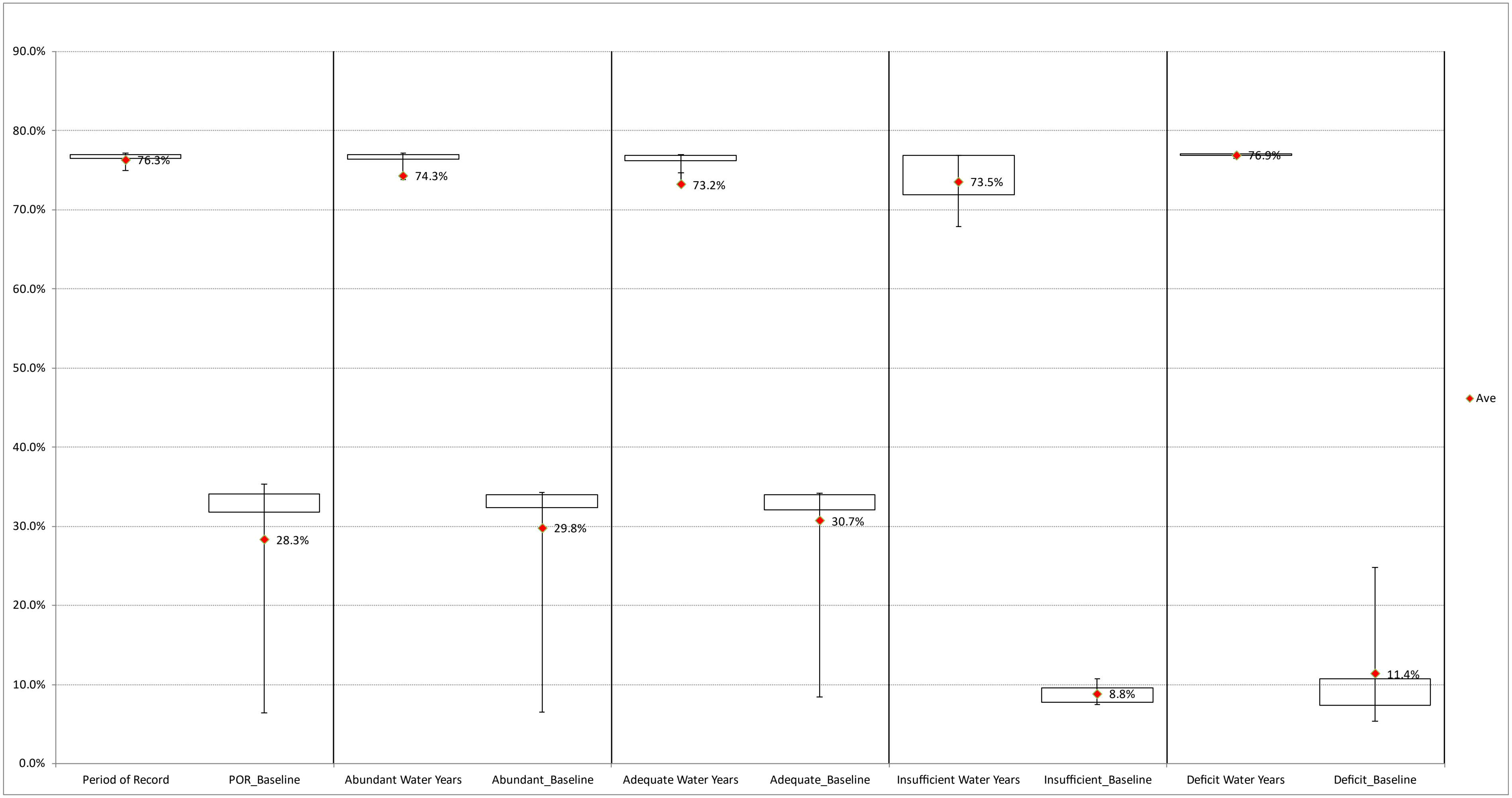

Figure 2-89. Detroit Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Detroit for juvenile spring Chinook sub-yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

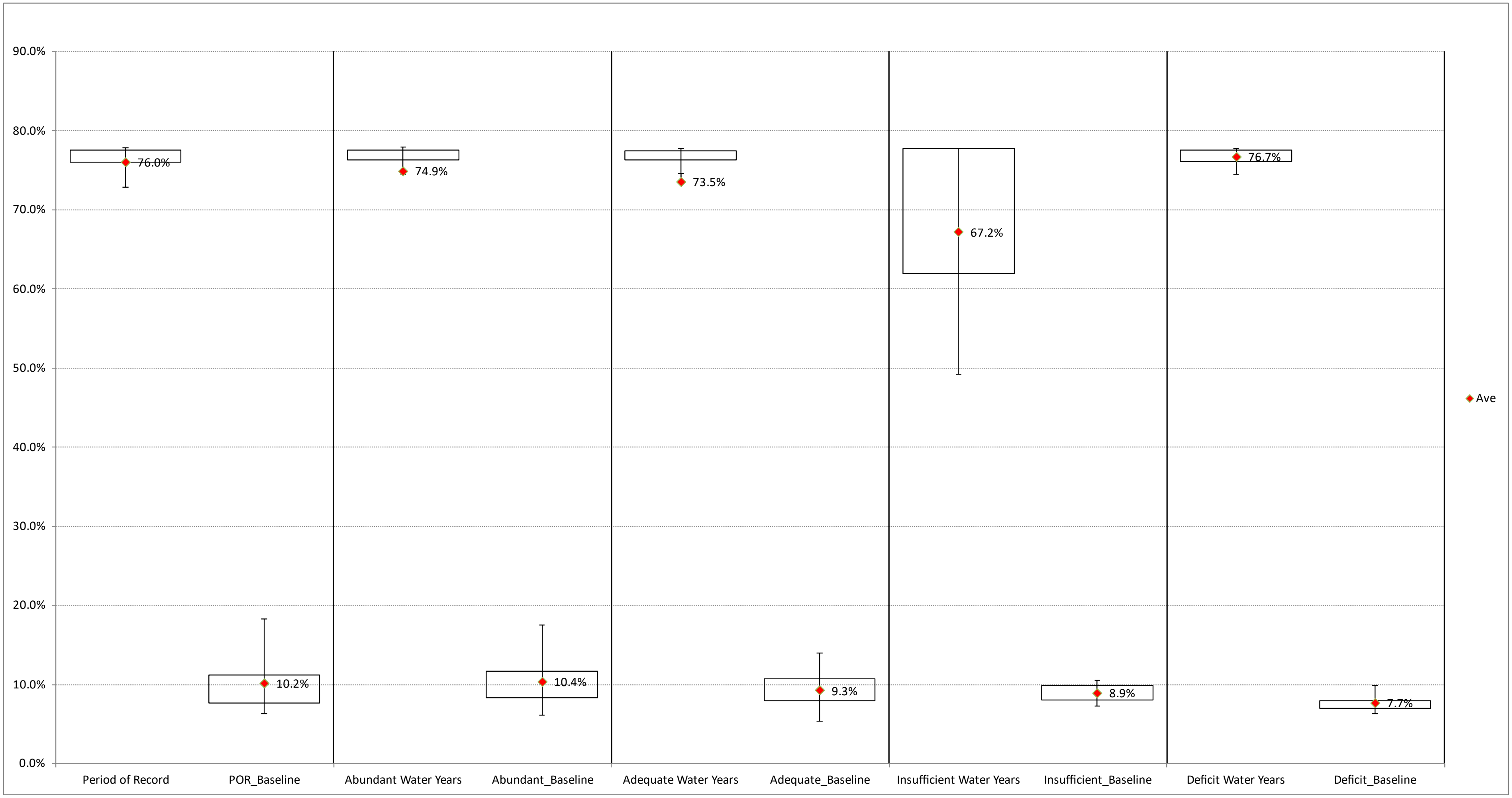

Figure 2-90. Downstream dam passage survival at Detroit for juvenile spring Chinook yearlings under Alternative 4. *Downstream dam passage survival at Detroit for juvenile spring Chinook yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.5.2 South Santiam – Foster

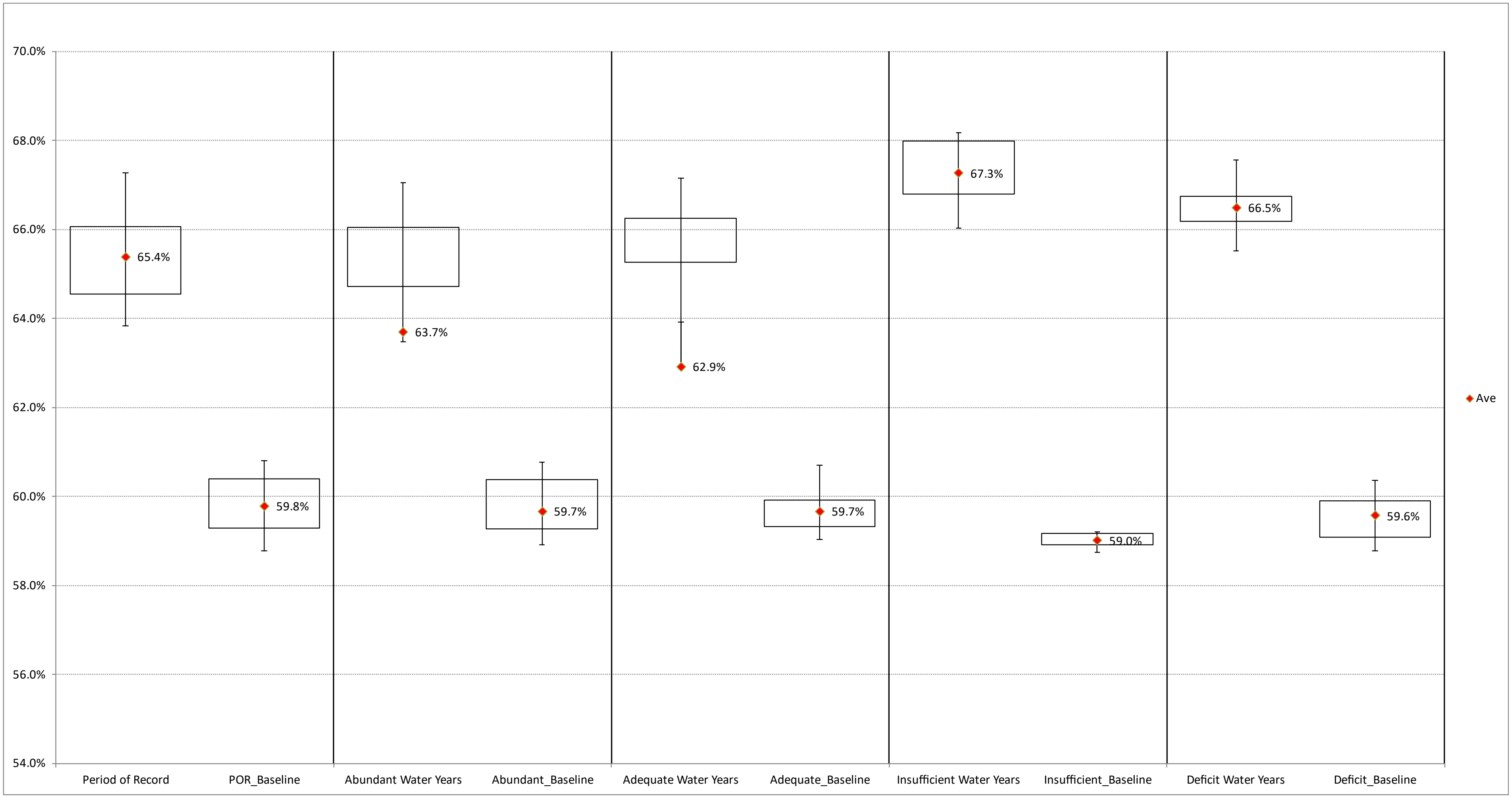

Figure 2-91. Downstream dam passage survival at Foster for juvenile spring Chinook fry under Alternative 4. *Downstream dam passage survival at Foster for juvenile spring Chinook fry under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

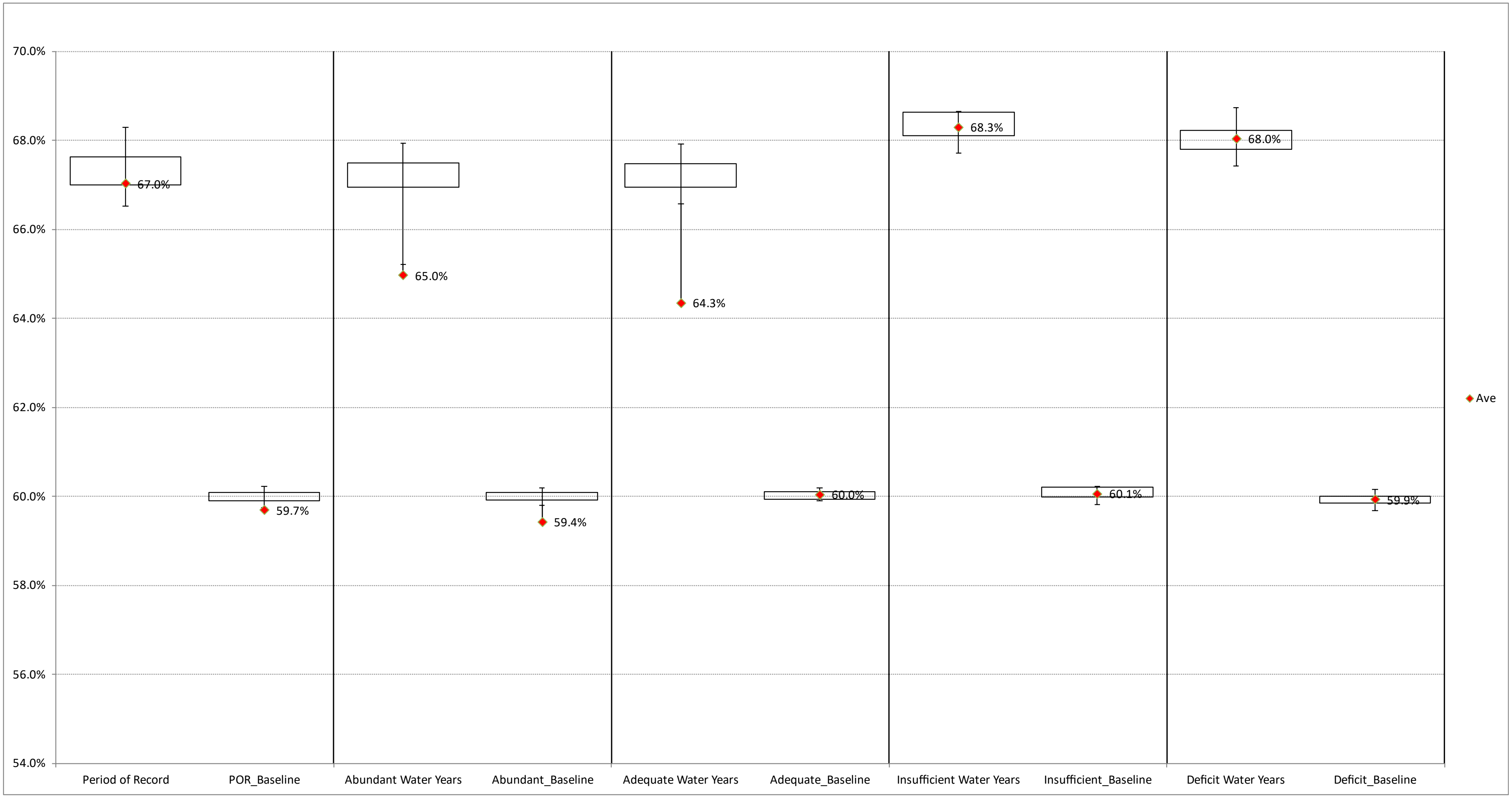

Figure 2-92. Foster Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Foster for juvenile spring Chinook sub-yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

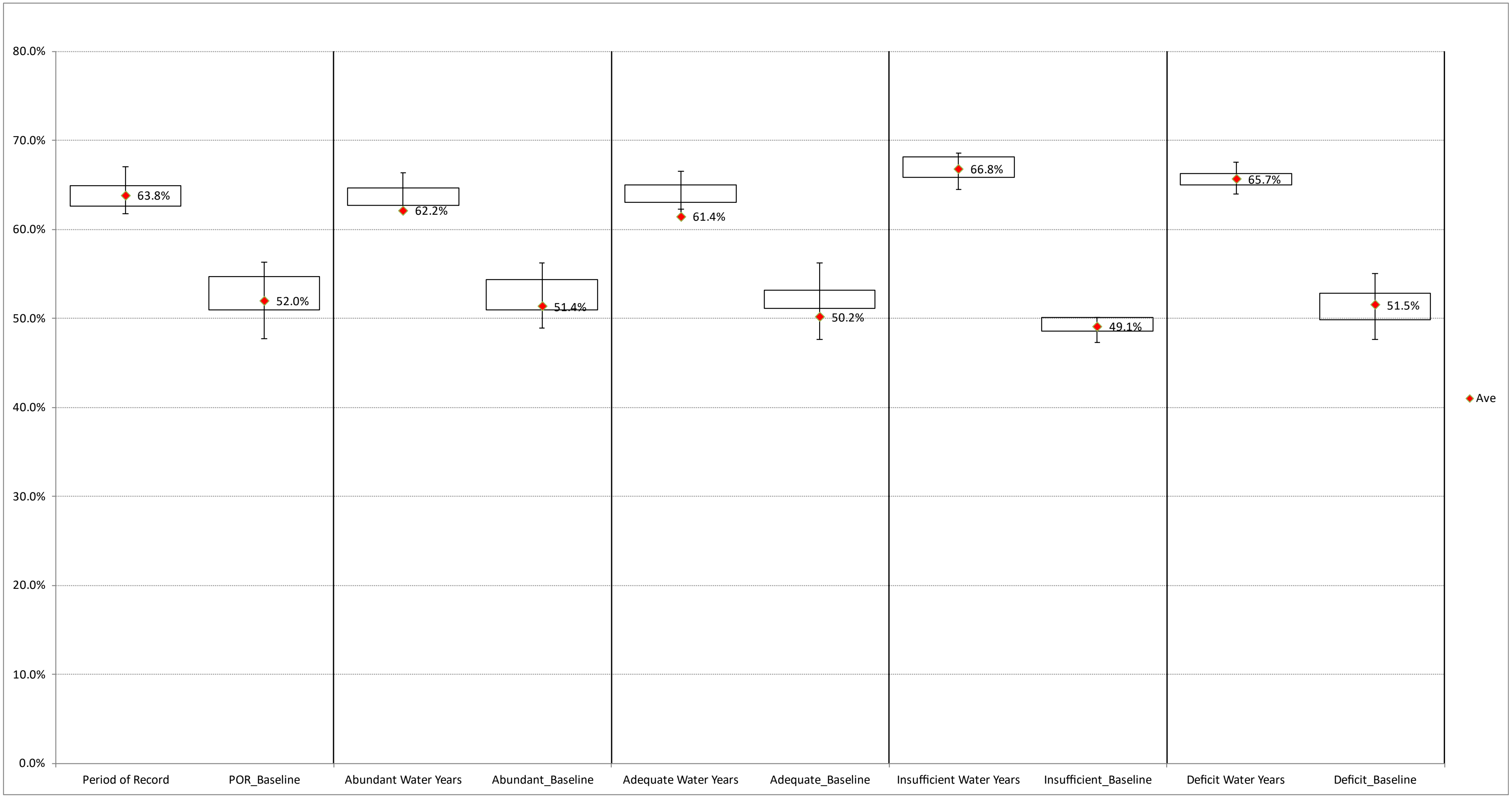

Figure 2-93. Foster Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Foster for juvenile spring Chinook yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.5.3 South Santiam – Green Peter

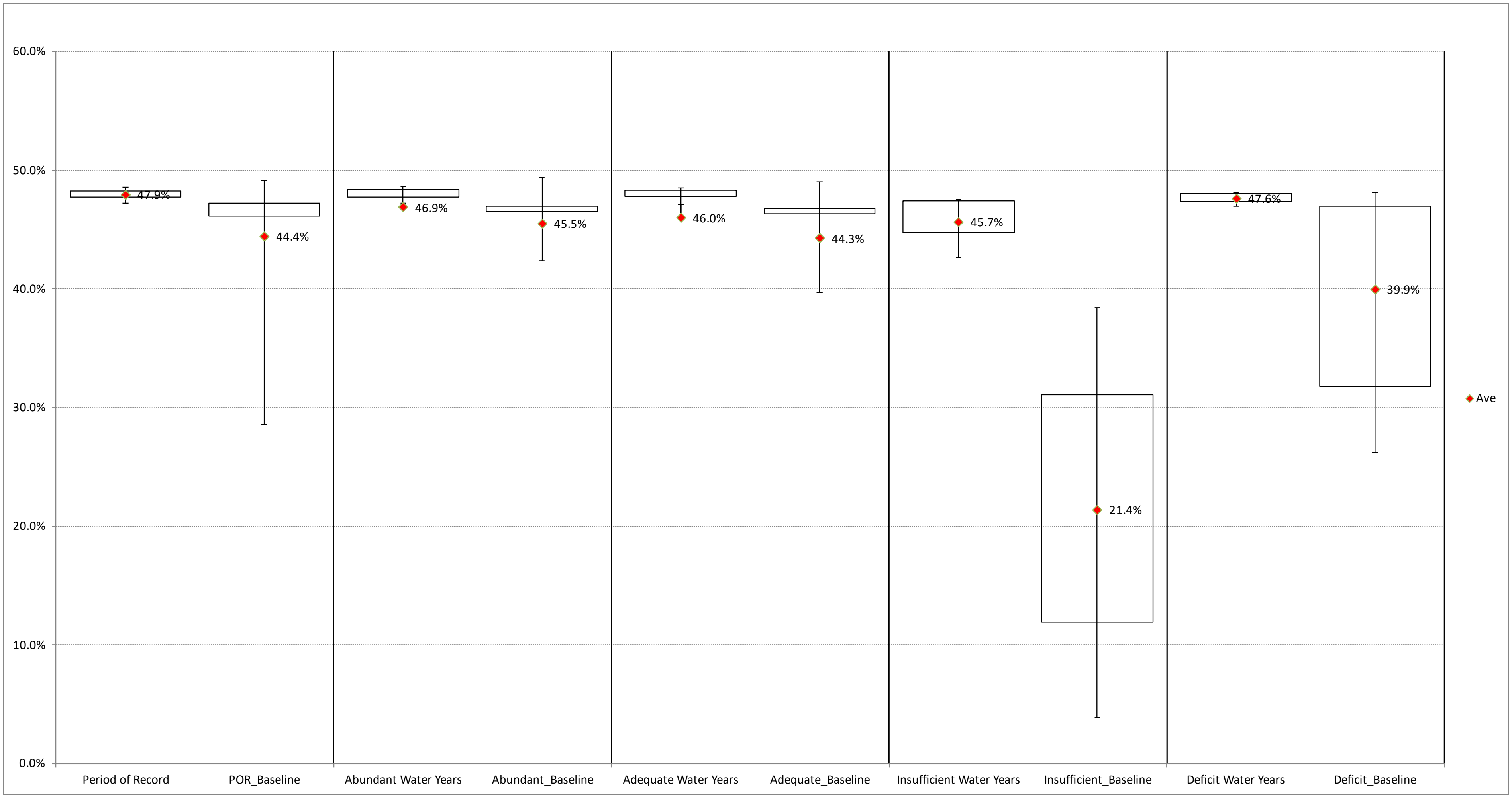

Figure 2-94. Green Peter Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Green Peter for juvenile spring Chinook fry under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

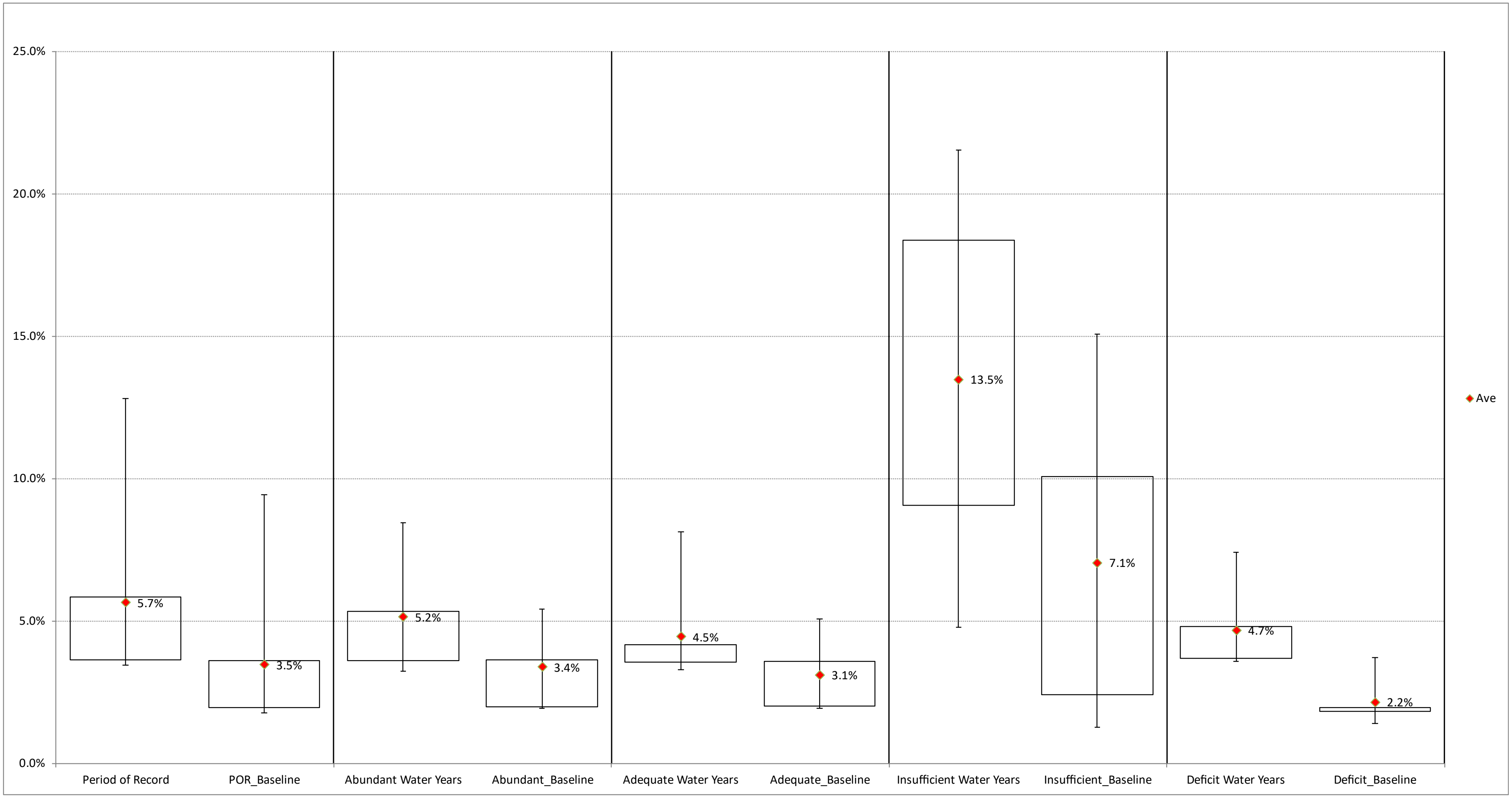

Figure 2-95. Downstream dam passage survival at Green Peter for juvenile spring Chinook sub-yearlings under Alternative 4. *Downstream dam passage survival at Green Peter for juvenile spring Chinook sub-yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

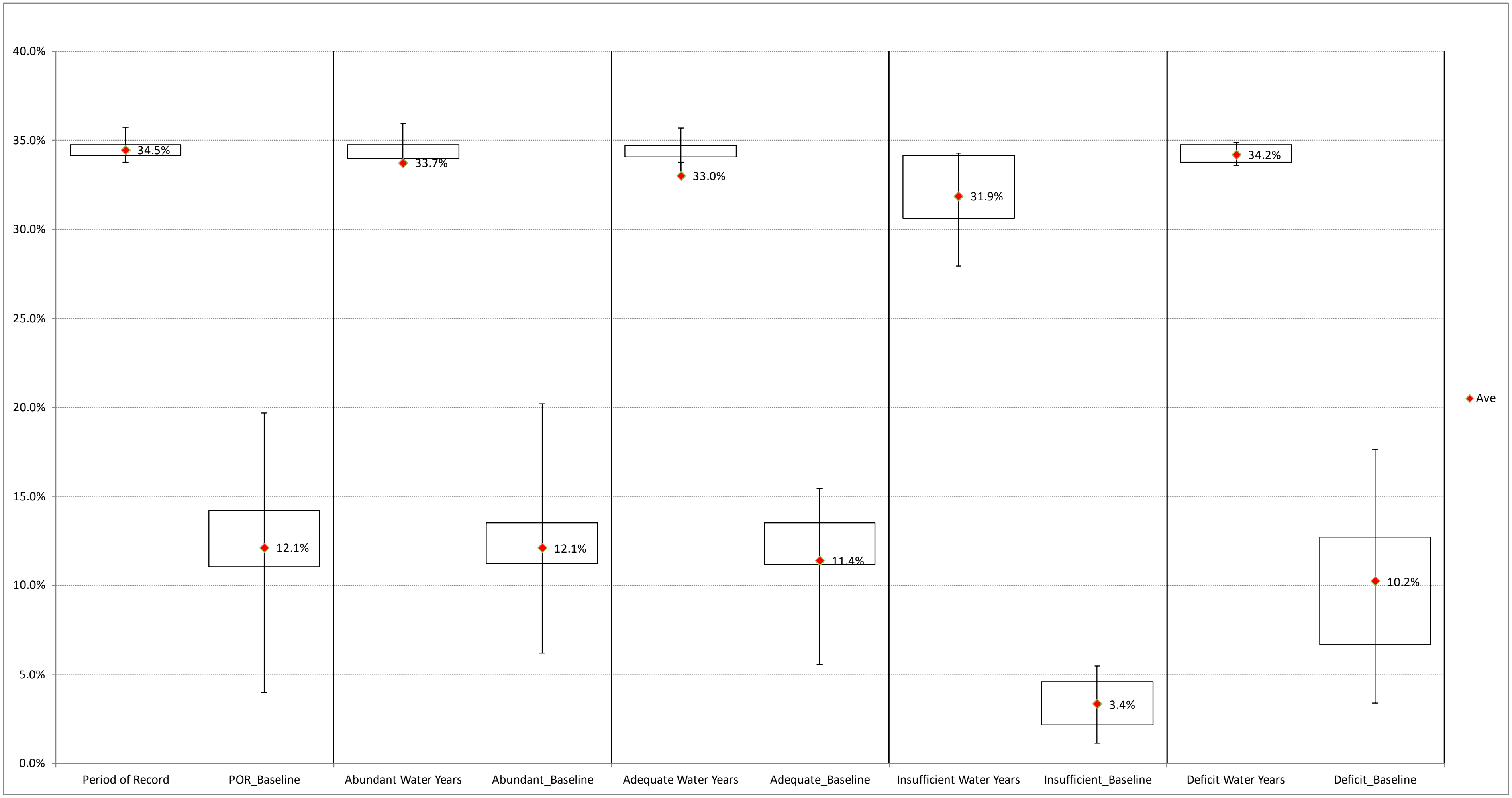

Figure 2-96. Green Peter Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Green Peter for juvenile spring Chinook sub-yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.5.4 McKenzie - Cougar

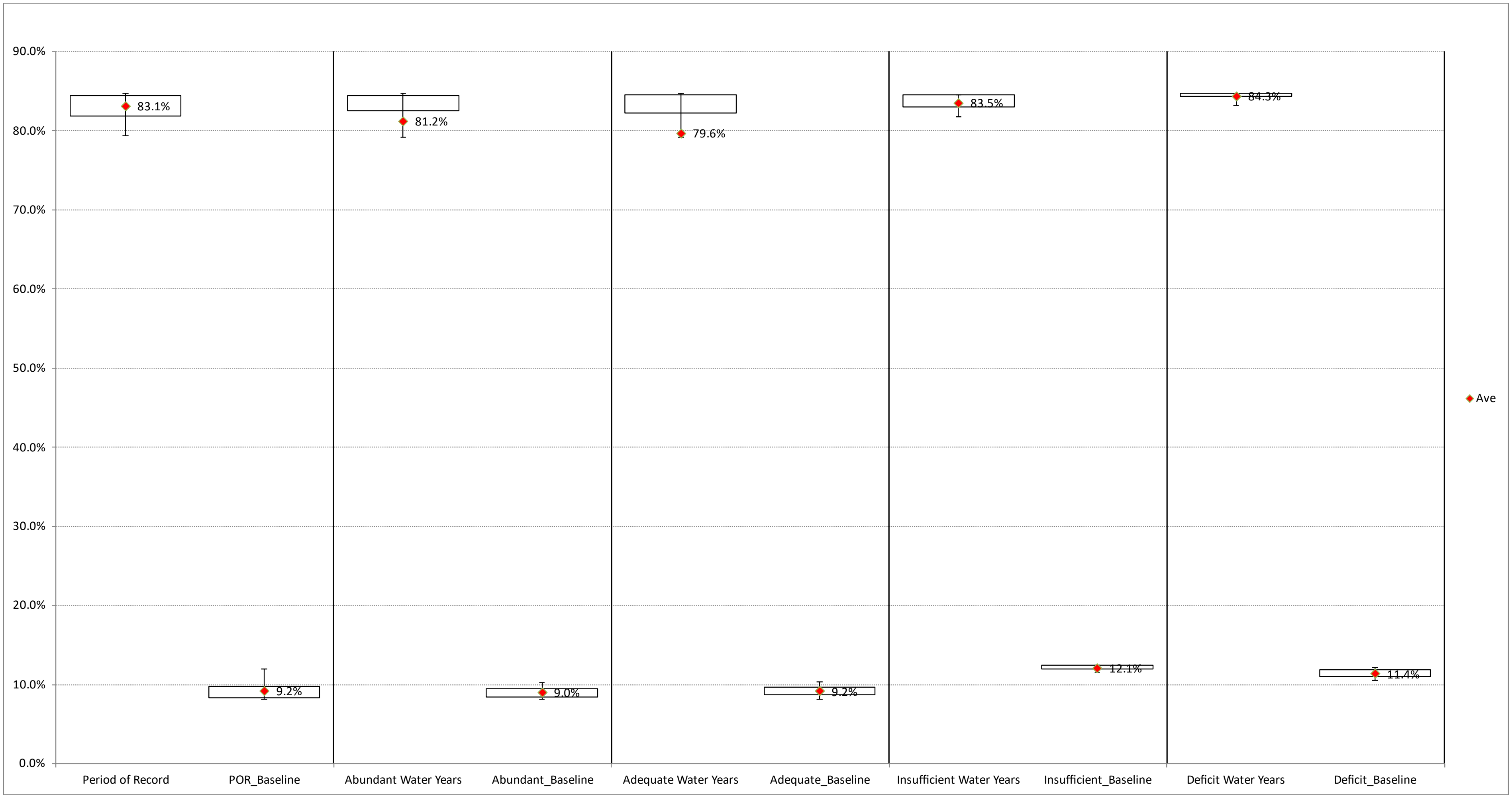

Figure 2-97. Cougar Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Cougar for juvenile spring Chinook fry under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

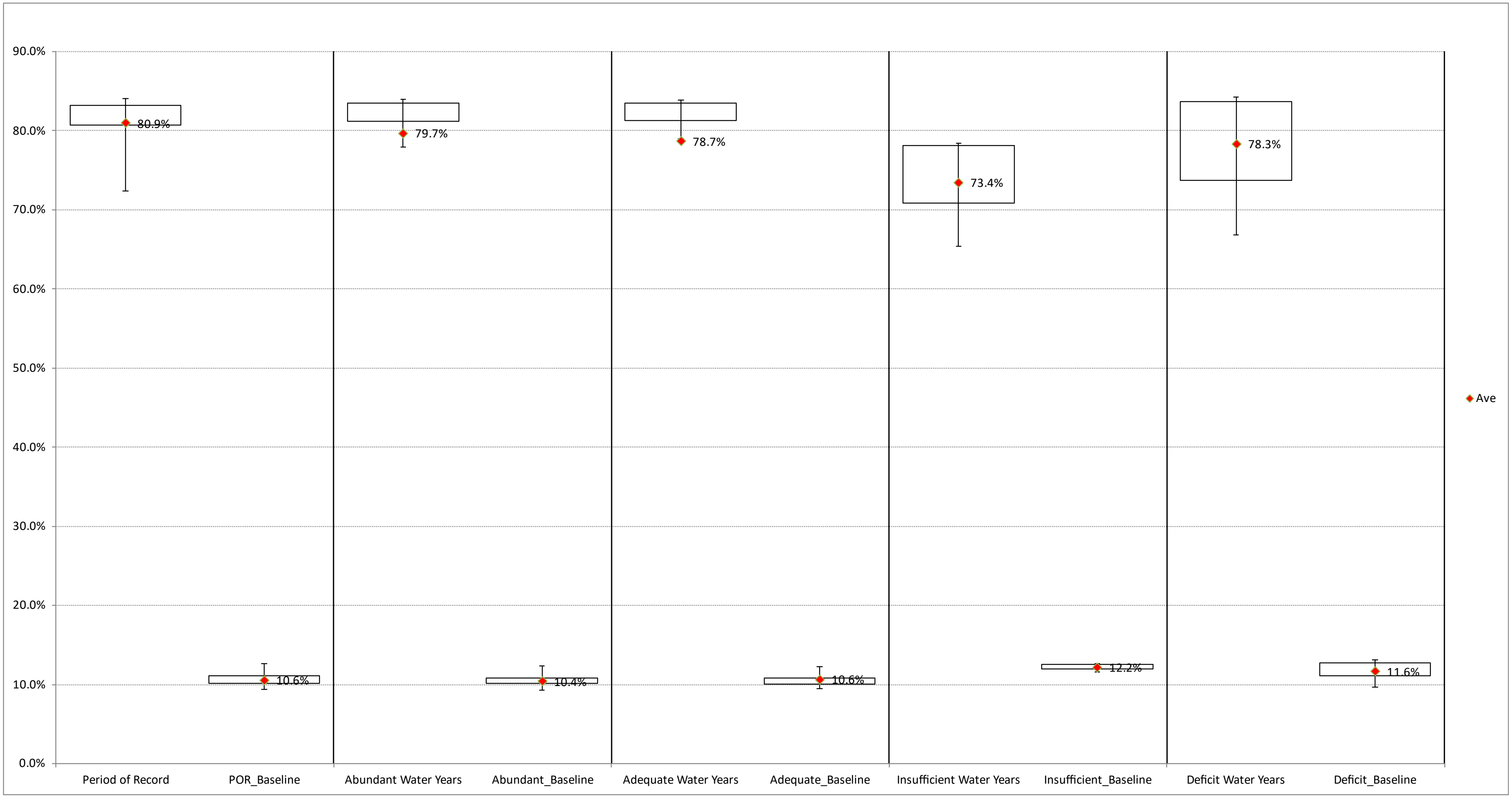

Figure 2-98. Cougar Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Cougar for juvenile spring Chinook sub-yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

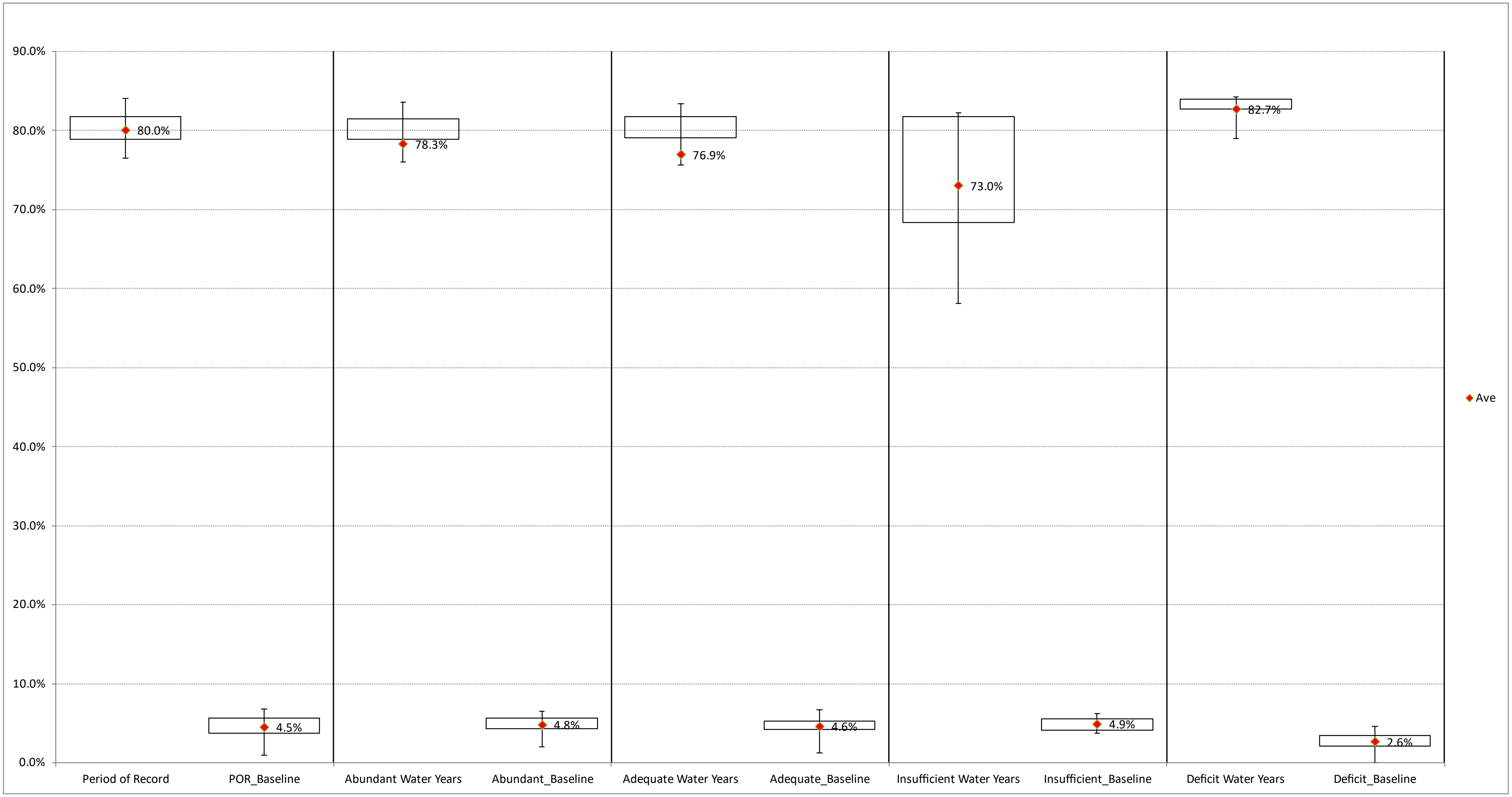

Figure 2-99. Cougar Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Cougar for juvenile spring Chinook yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.5.5 Middle Fork – Lookout Point

Figure 2-100. Lookout Point Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook fry under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

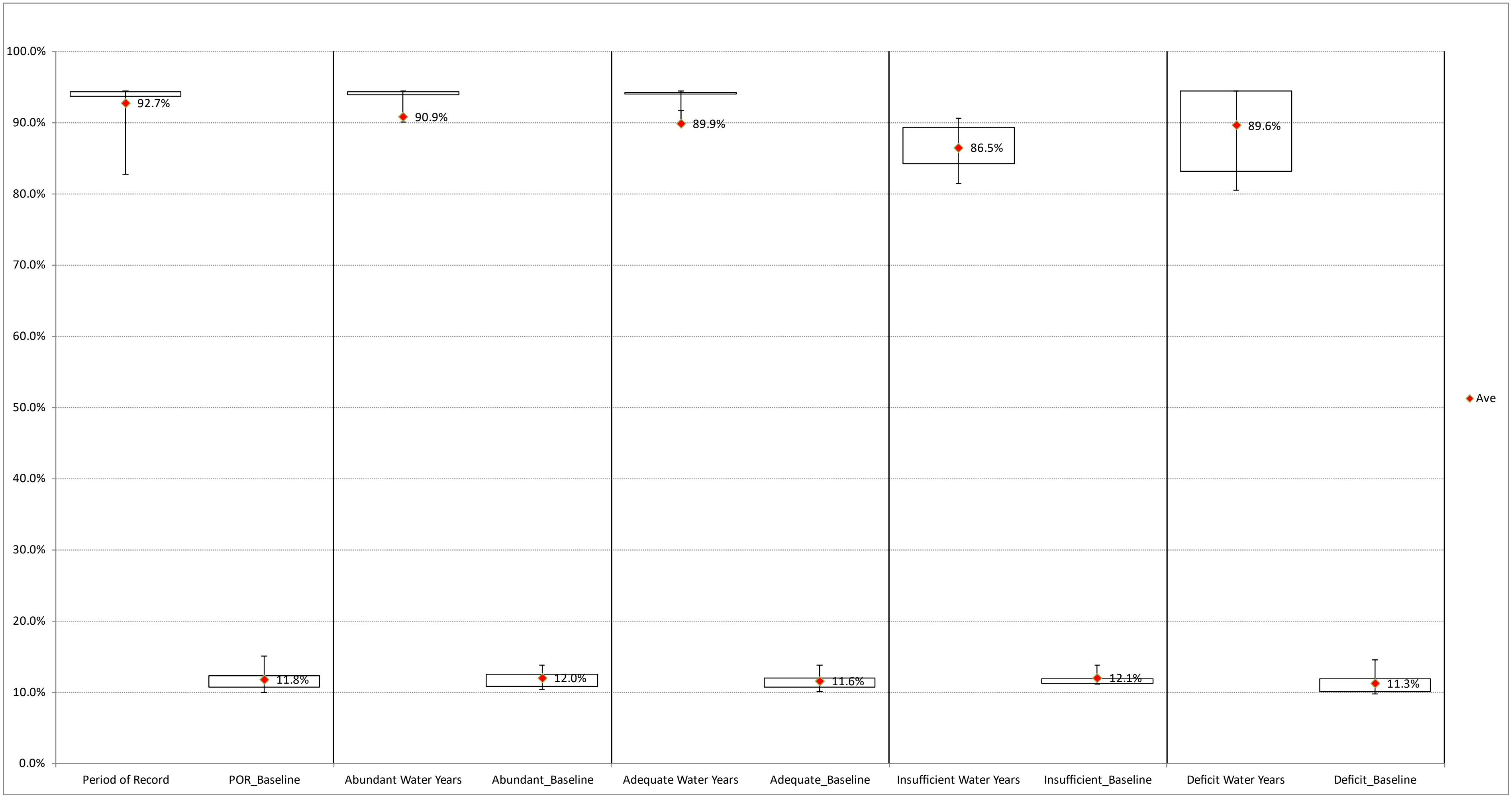

Figure 2-101. Lookout Point Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook sub-yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

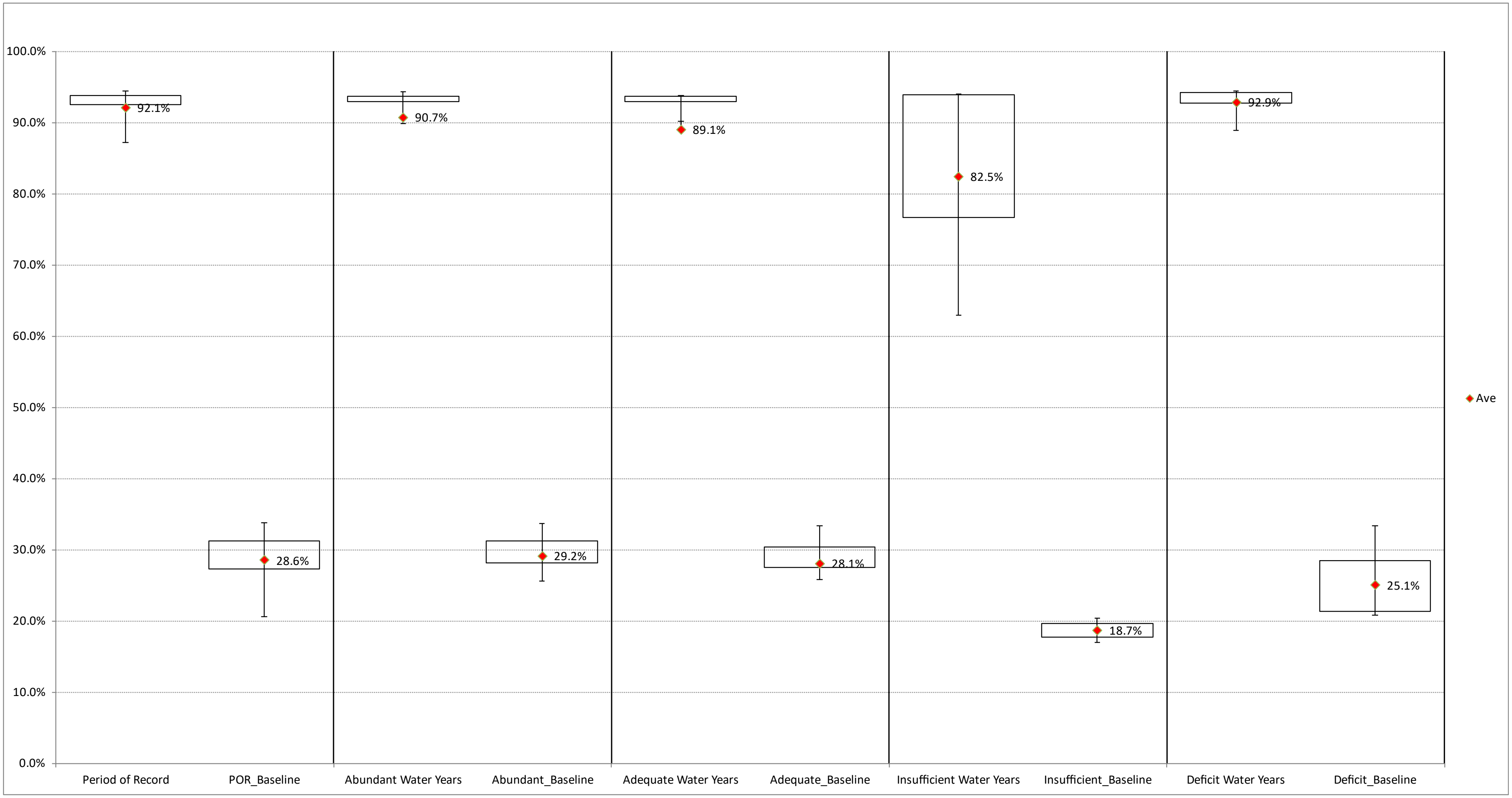

Figure 2-102. Lookout Point Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Lookout Point for juvenile spring Chinook yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.5.6 Middle Fork – Hills Creek

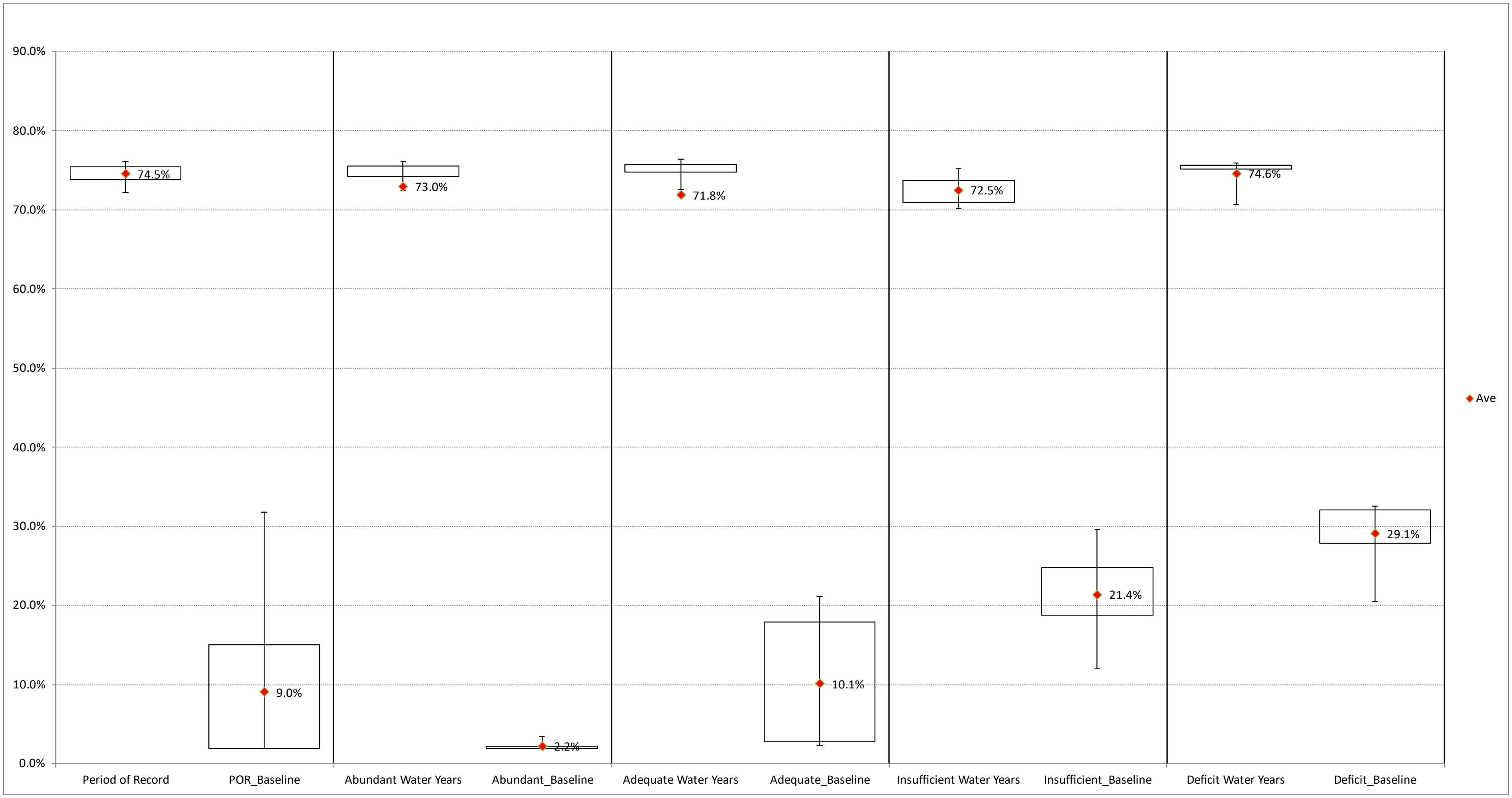

Figure 2-103. Hills Creek Juvenile Spring Chinook Fry Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook fry under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

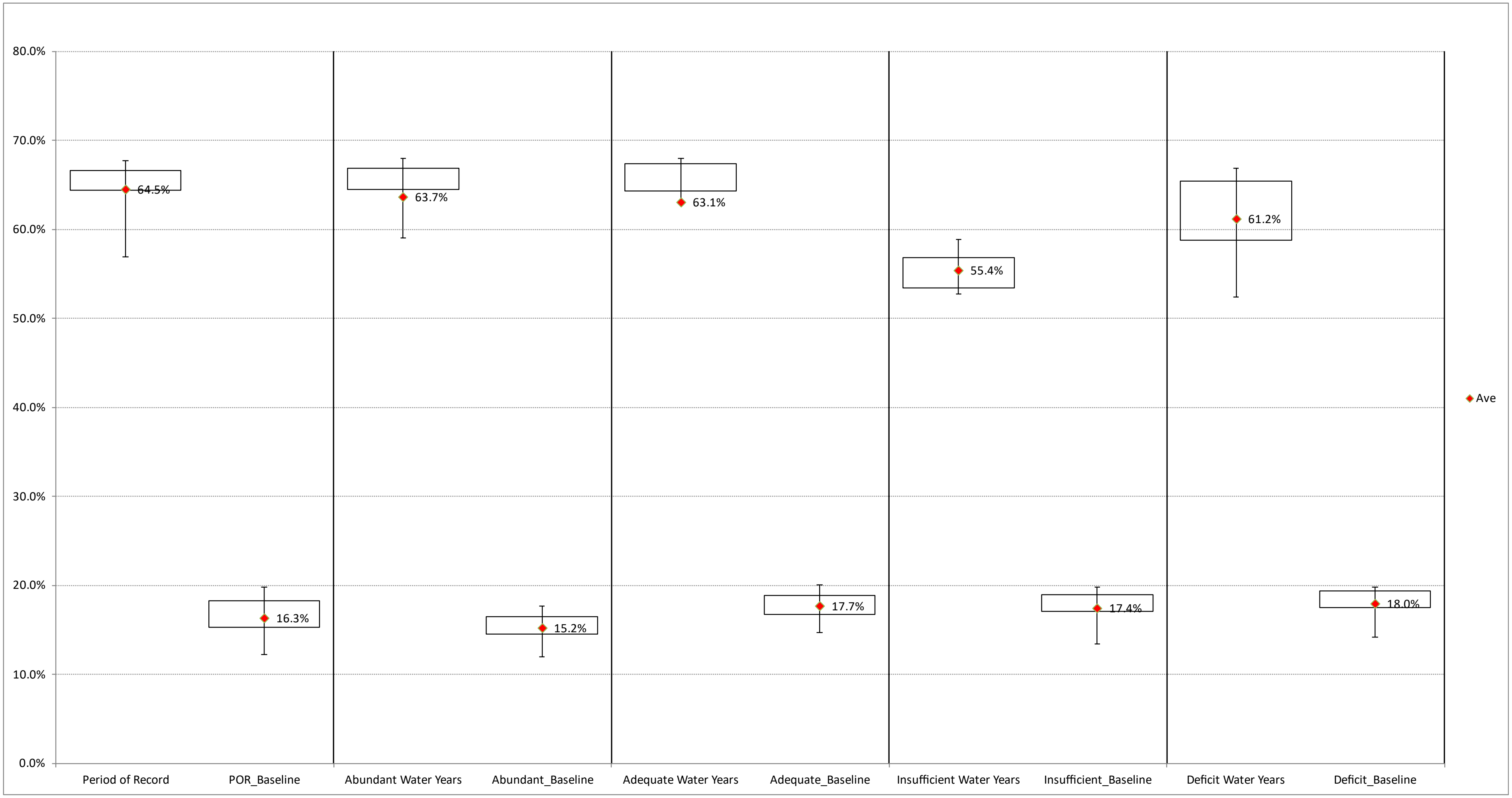

Figure 2-104. Hills Creek Juvenile Spring Chinook Sub-Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook sub-yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

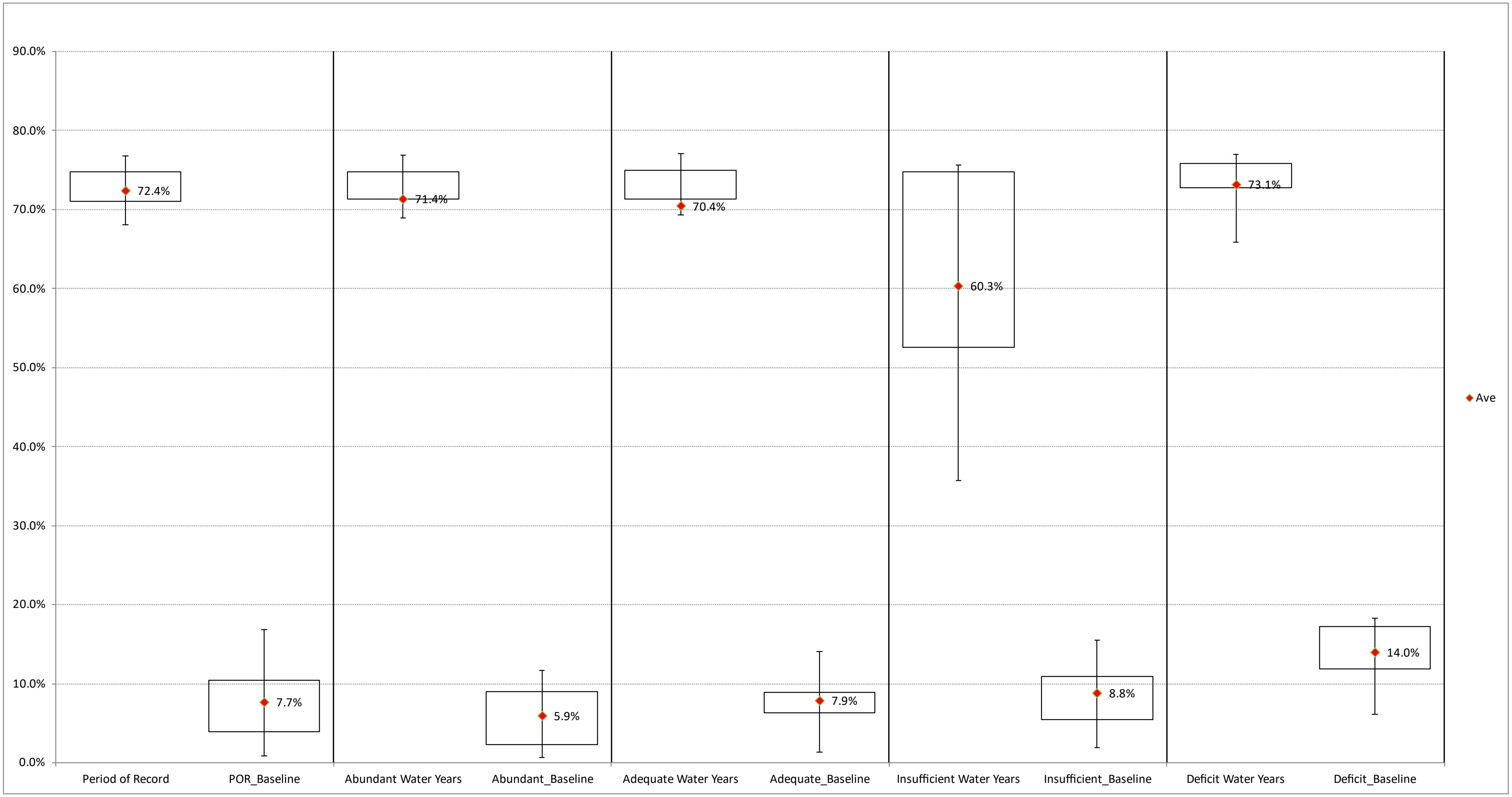

Figure 2-105. Hills Creek Juvenile Spring Chinook Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Hills Creek for juvenile spring Chinook yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

STEELHEAD

2.6 STEELHEAD NO ACTION ALTERNATIVE (NAA OR BASELINE)

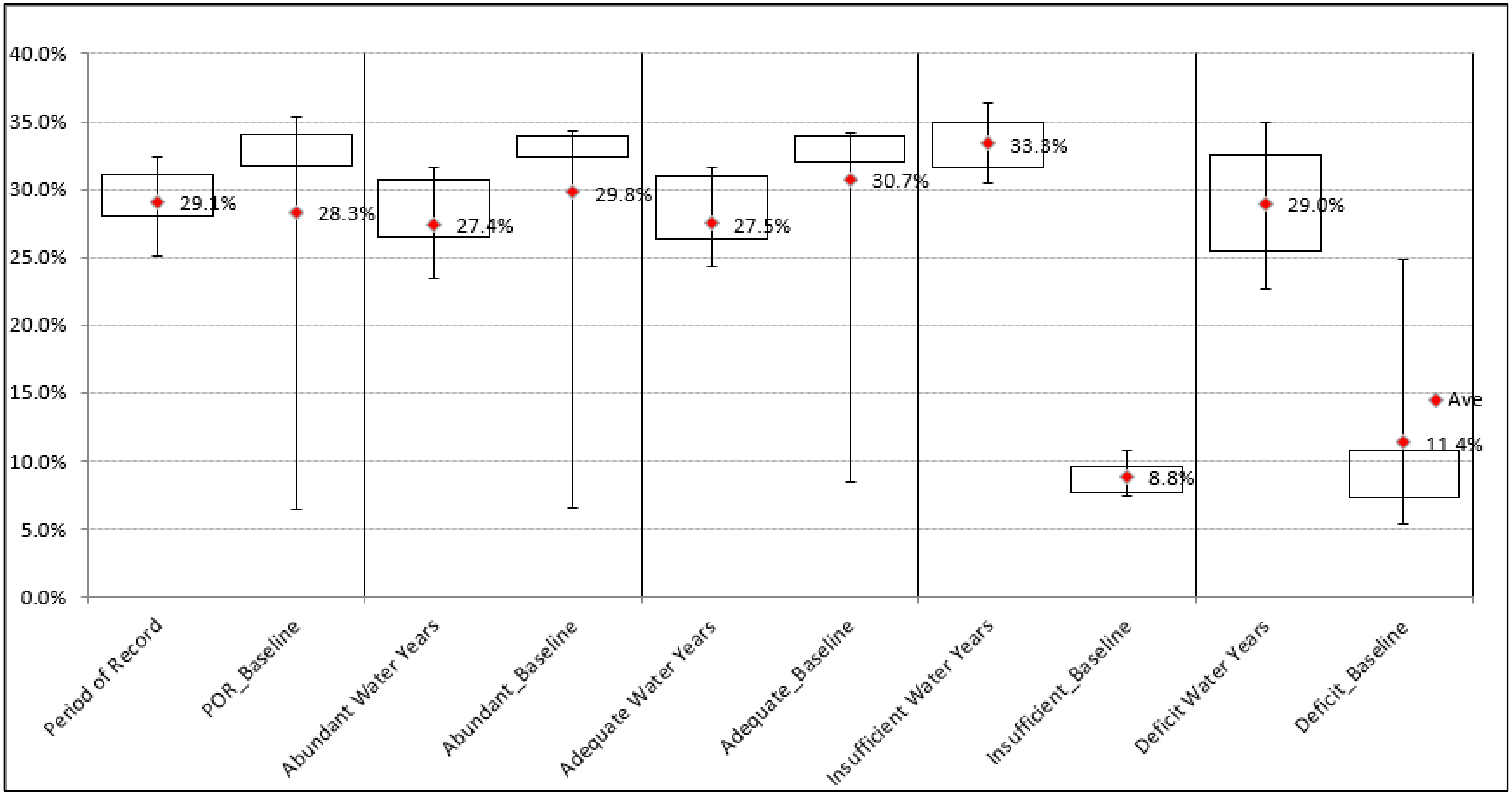

Figure 2-106. Detroit Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under the NAA. *Downstream dam passage survival at Detroit for juvenile winter steelhead sub-yearlings under the NAA. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

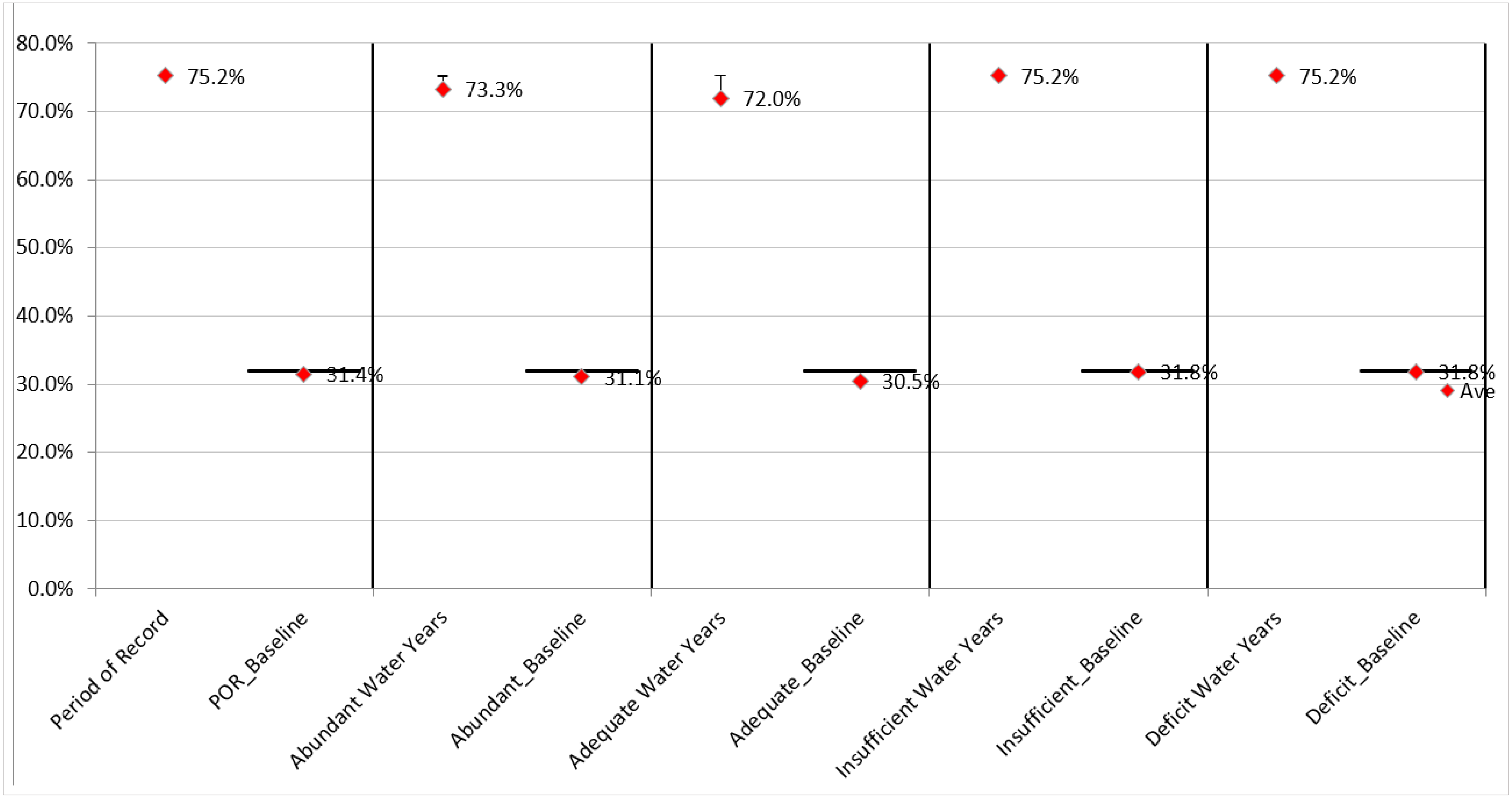

Figure 2-107. Detroit Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under the NAA. *The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

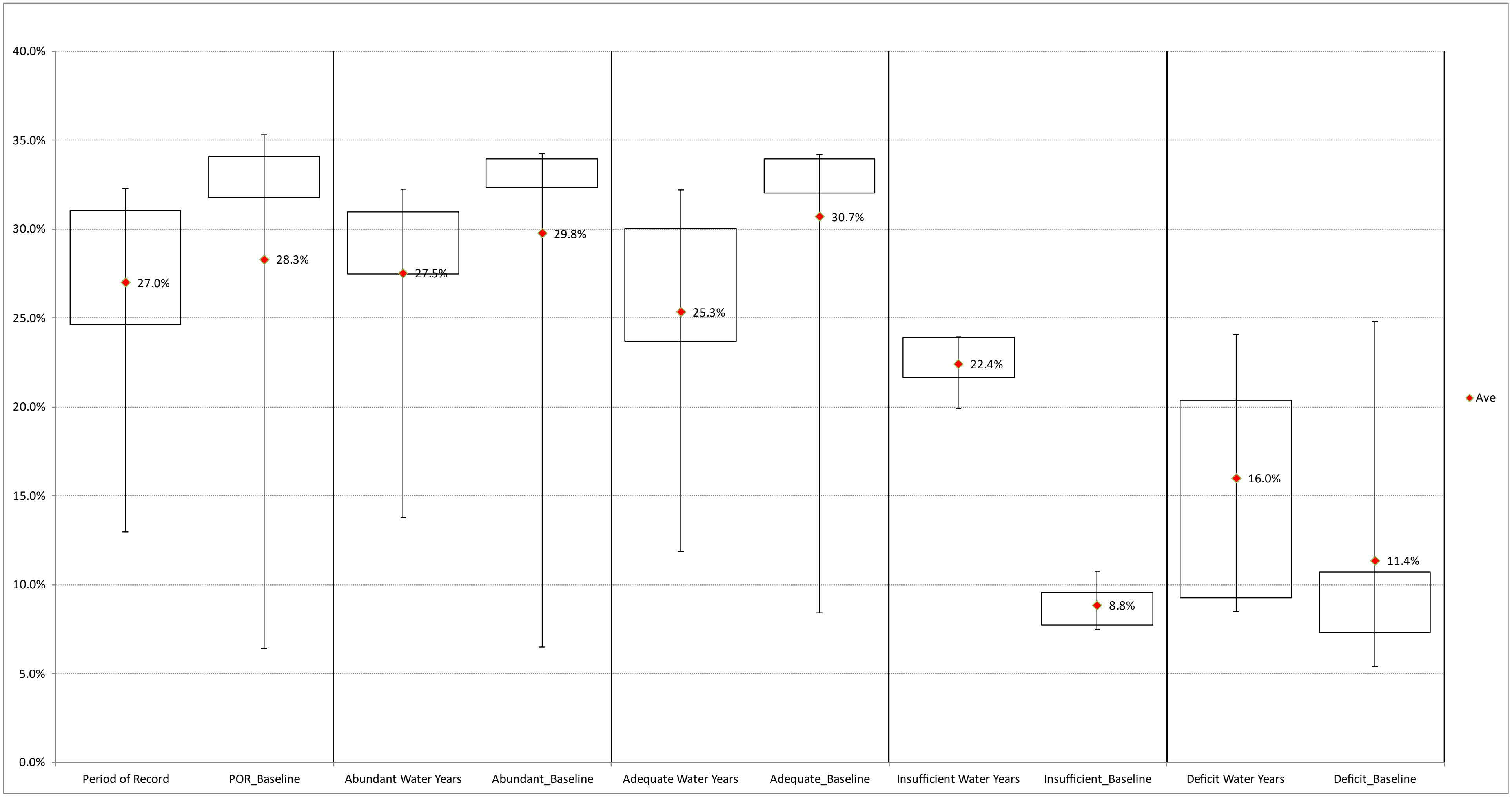

Figure 2-108. Detroit For 2-Year-Old Juvenile Winter Steelhead Downstream Dam Passage Survival At Under the NAA. *Downstream dam passage survival at Detroit for juvenile winter steelhead 2 year olds under the NAA. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

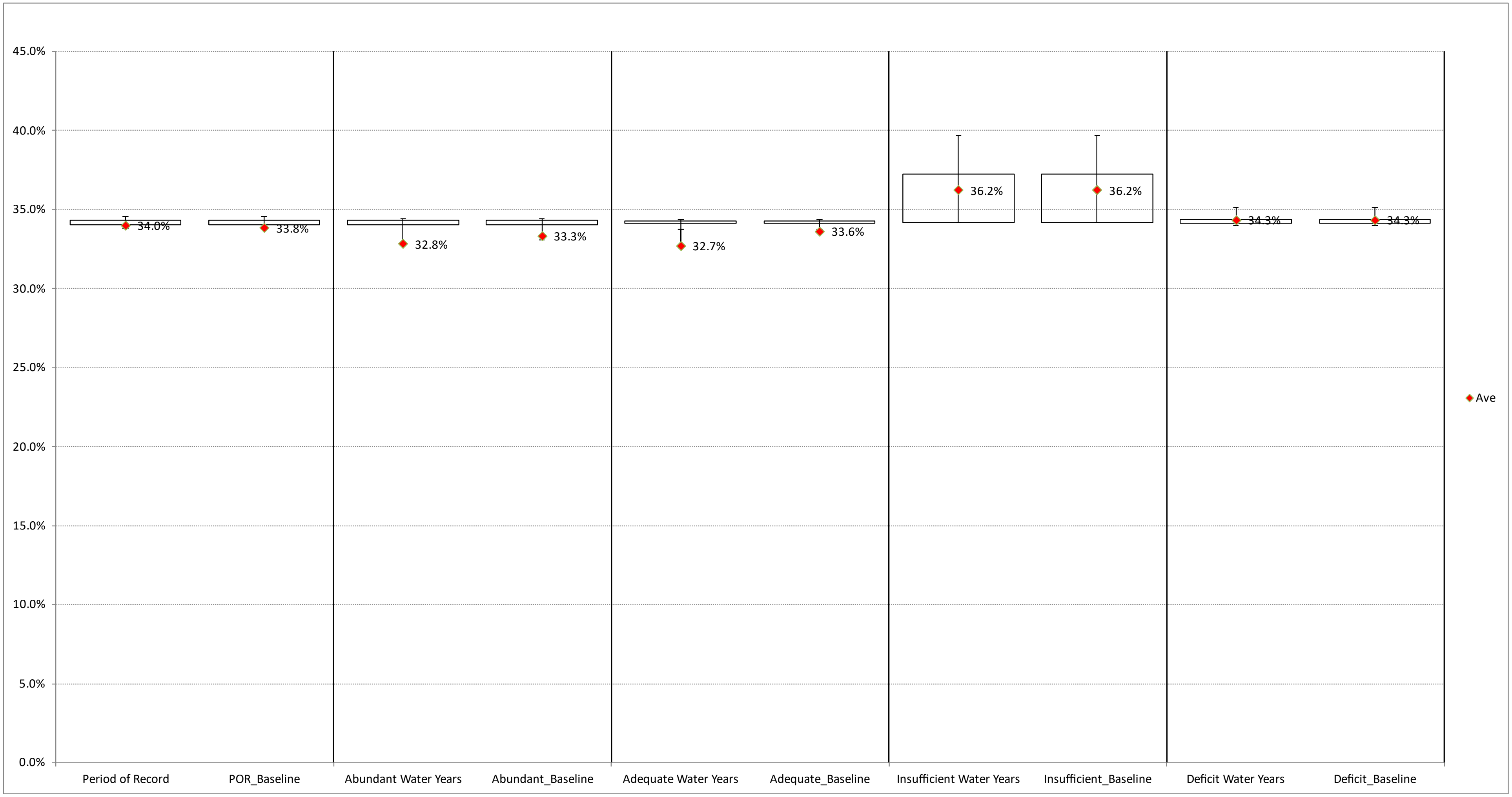

Figure 2-109. Foster Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under the NAA. *Downstream dam passage survival at Foster for juvenile winter steelhead sub-yearlings under the NAA. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

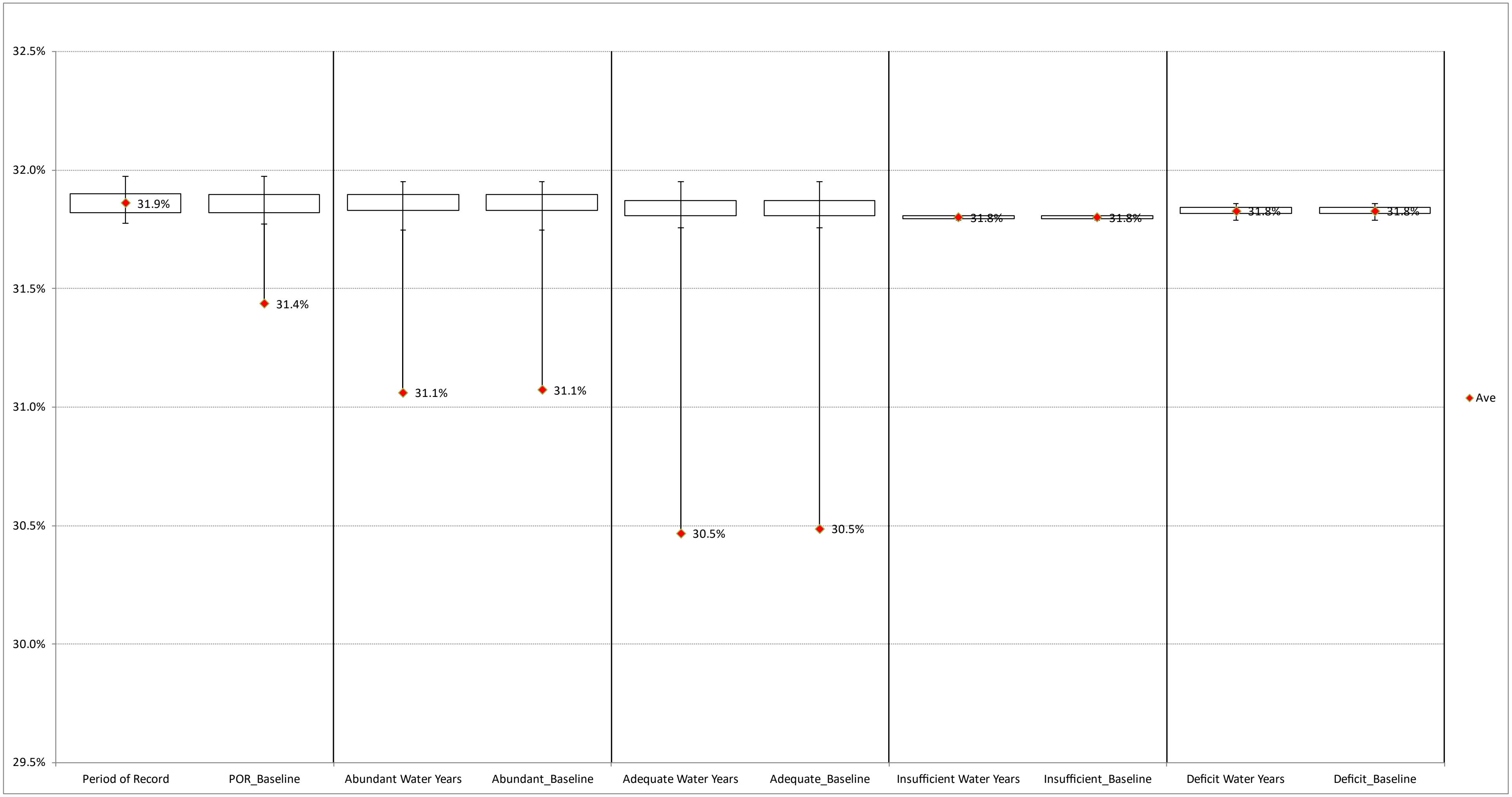

Figure 2-110. Foster Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under the NAA. *Downstream dam passage survival at Foster for juvenile winter steelhead yearlings under the NAA. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

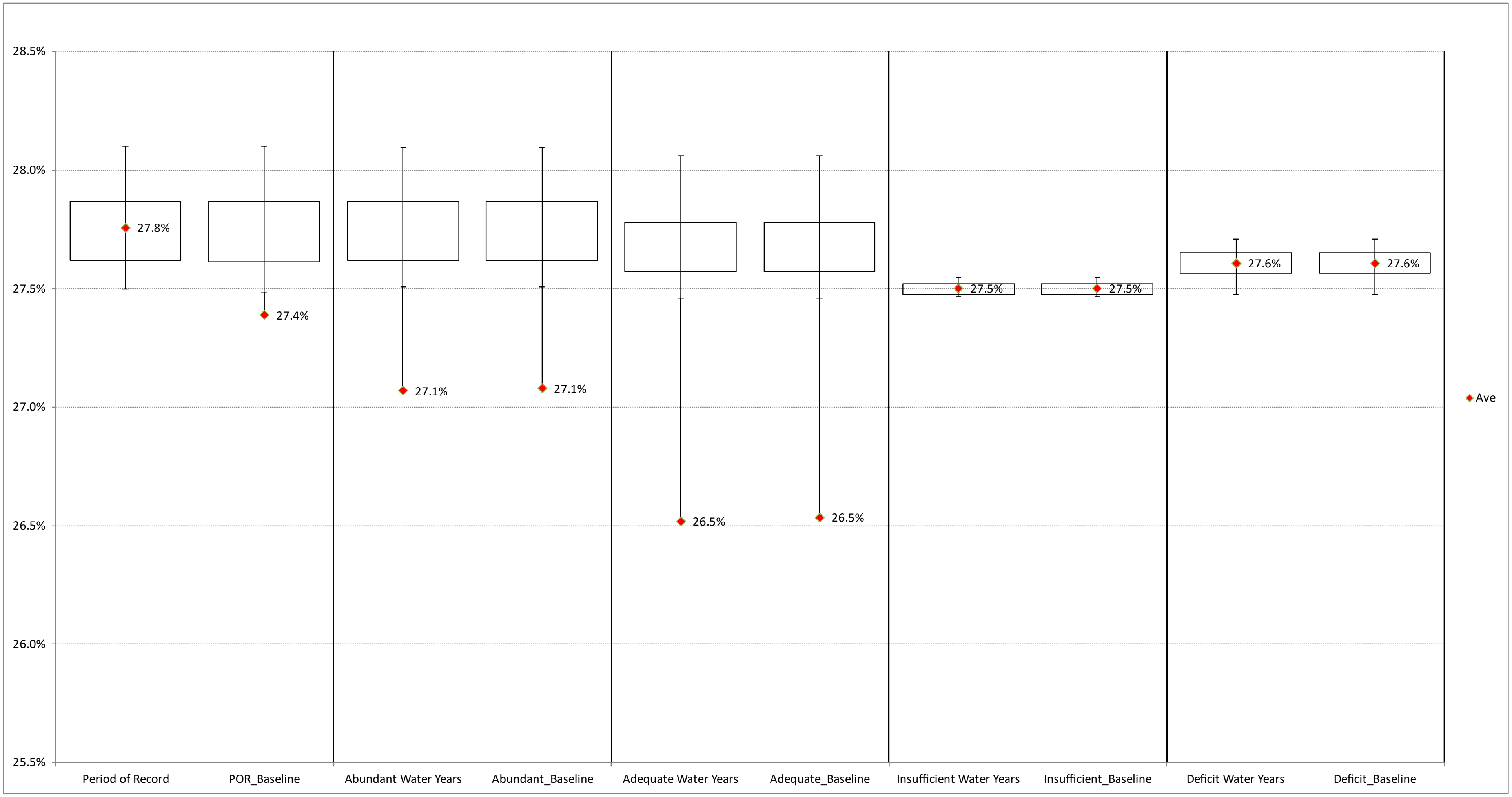

Figure 2-111. Foster 2-Year-Old Juvenile Winter Steelhead Downstream Dam Passage Survival Under the NAA. *Downstream dam passage survival at Foster for juvenile winter steelhead 2 year olds under the NAA. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

STEELHEAD

2.7 STEELHEAD ALTERNATIVE 1

2.7.1 South Santiam – Foster

Figure 2-112. Foster Juvenile Winter Steelhead Sub-Yearlings Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Foster for juvenile winter steelhead sub-yearlings under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

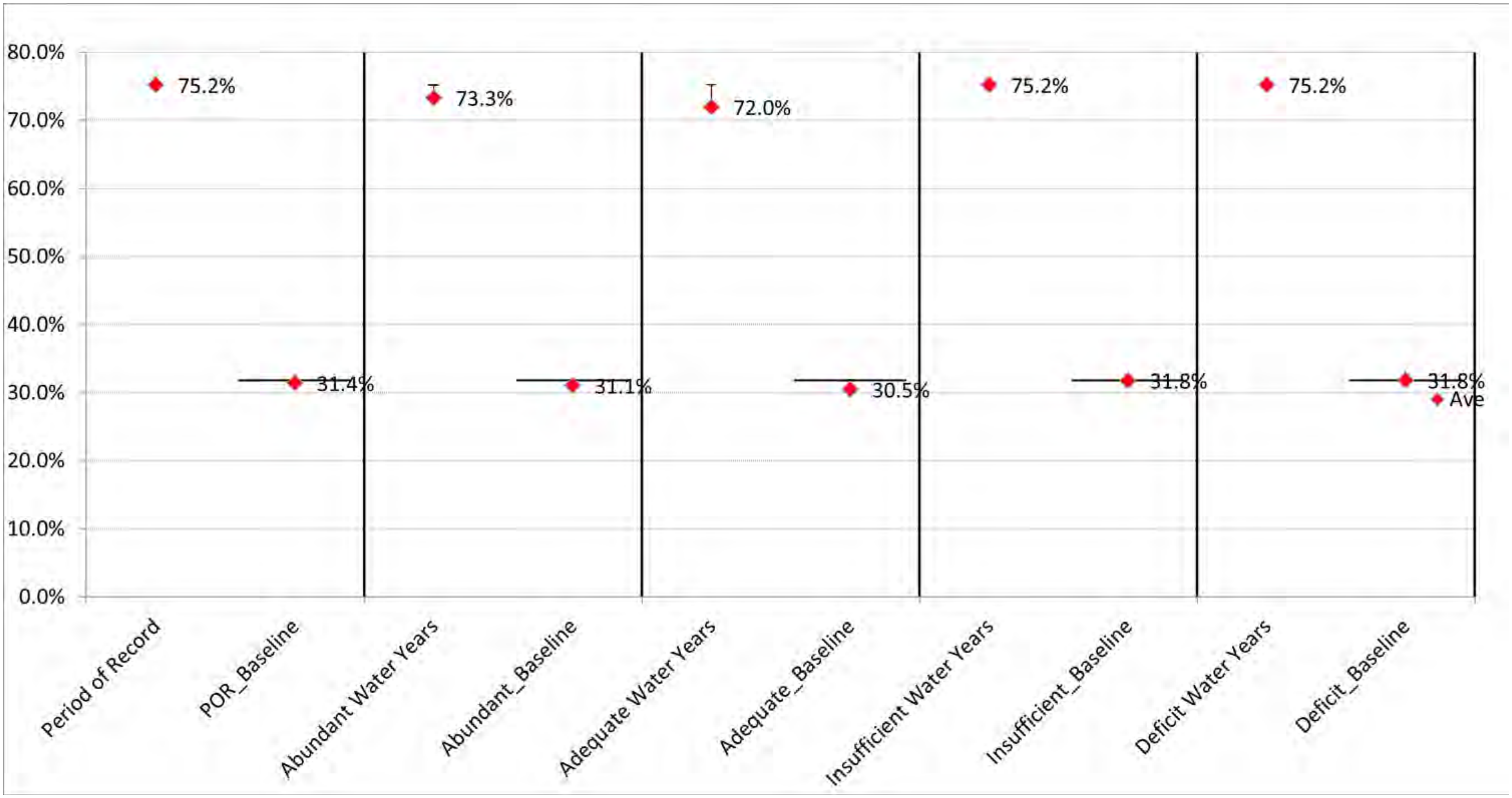

Figure 2-113. Foster Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Foster for juvenile winter steelhead yearlings under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

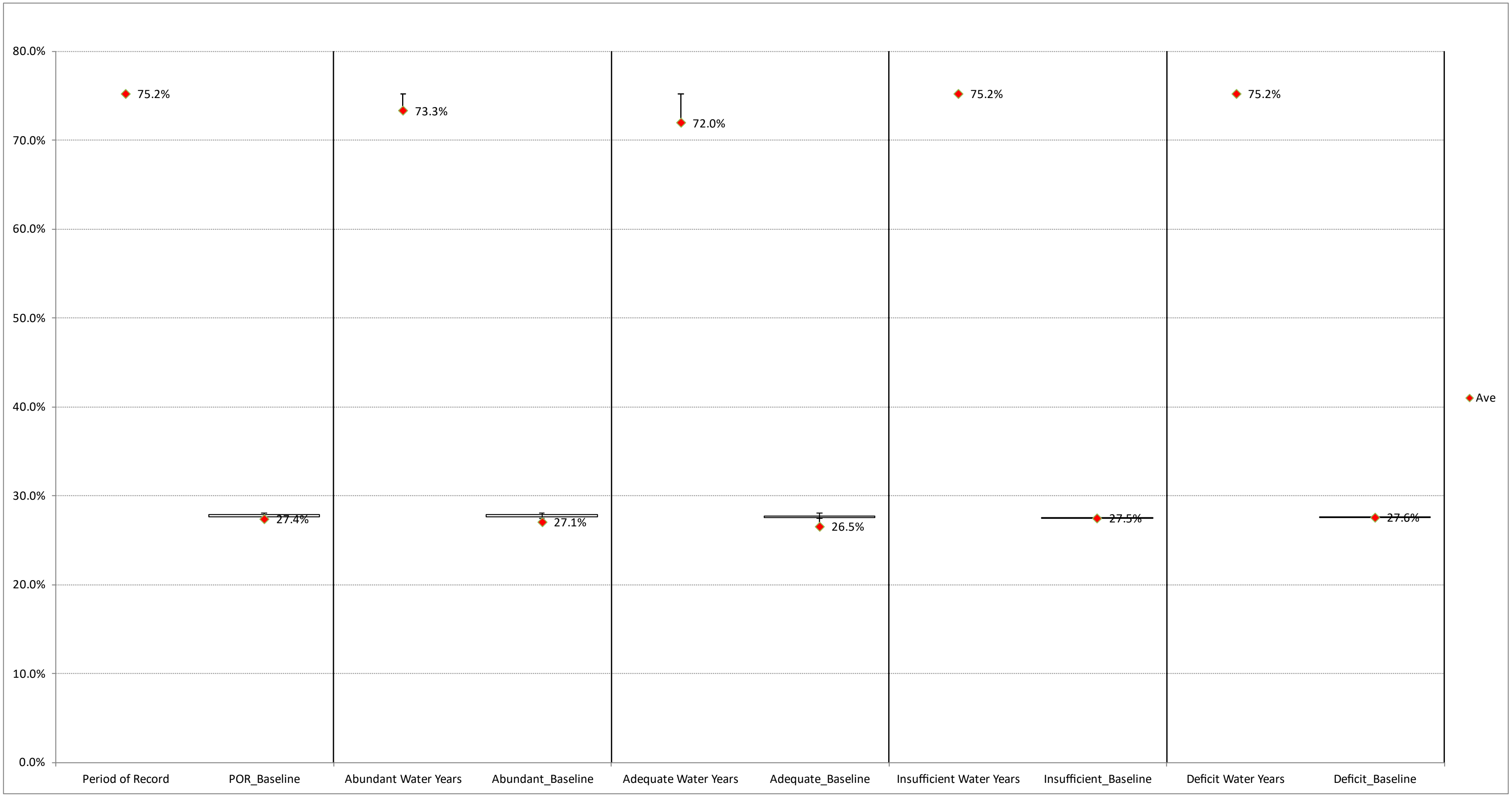

Figure 2-114. Foster 2-Year-Old Juvenile Winter Steelhead Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Foster for juvenile winter steelhead 2 year olds under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.7.2 South Santiam – Green Peter

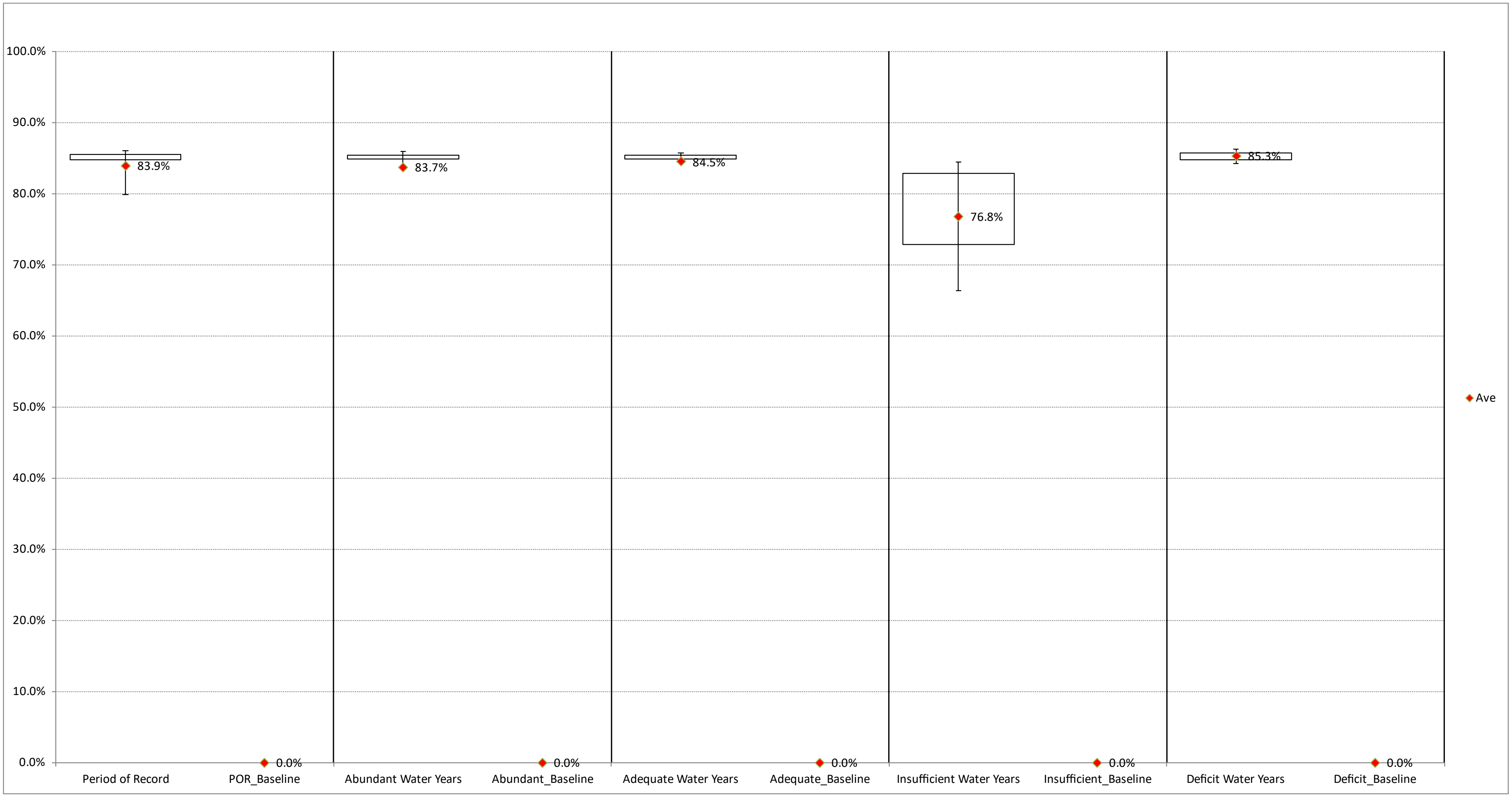

Figure 2-115. Green Peter Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Green Peter for juvenile winter steelhead sub-yearlings under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

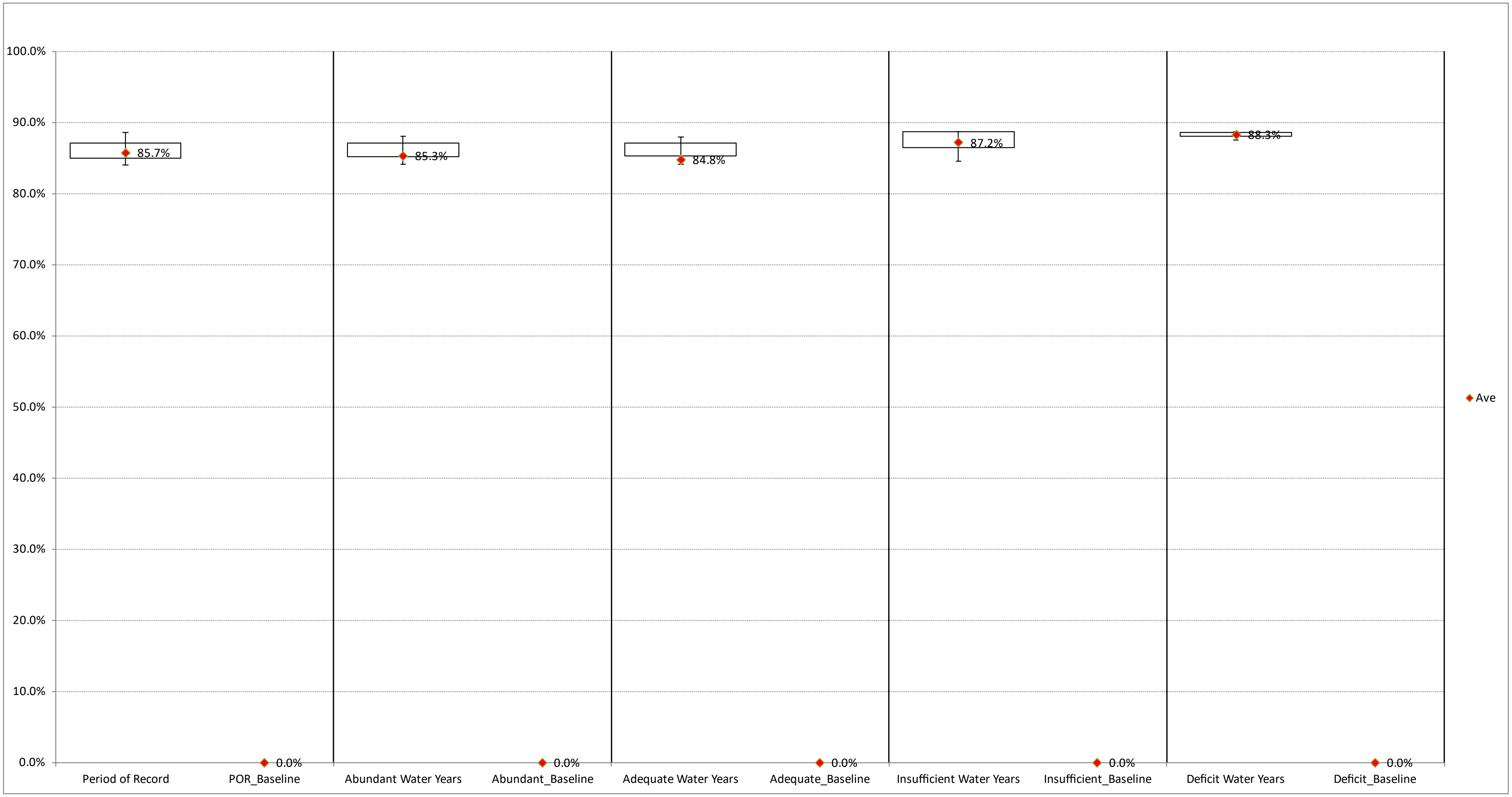

Figure 2-116. Green Peter Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Green Peter for juvenile winter steelhead yearlings under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

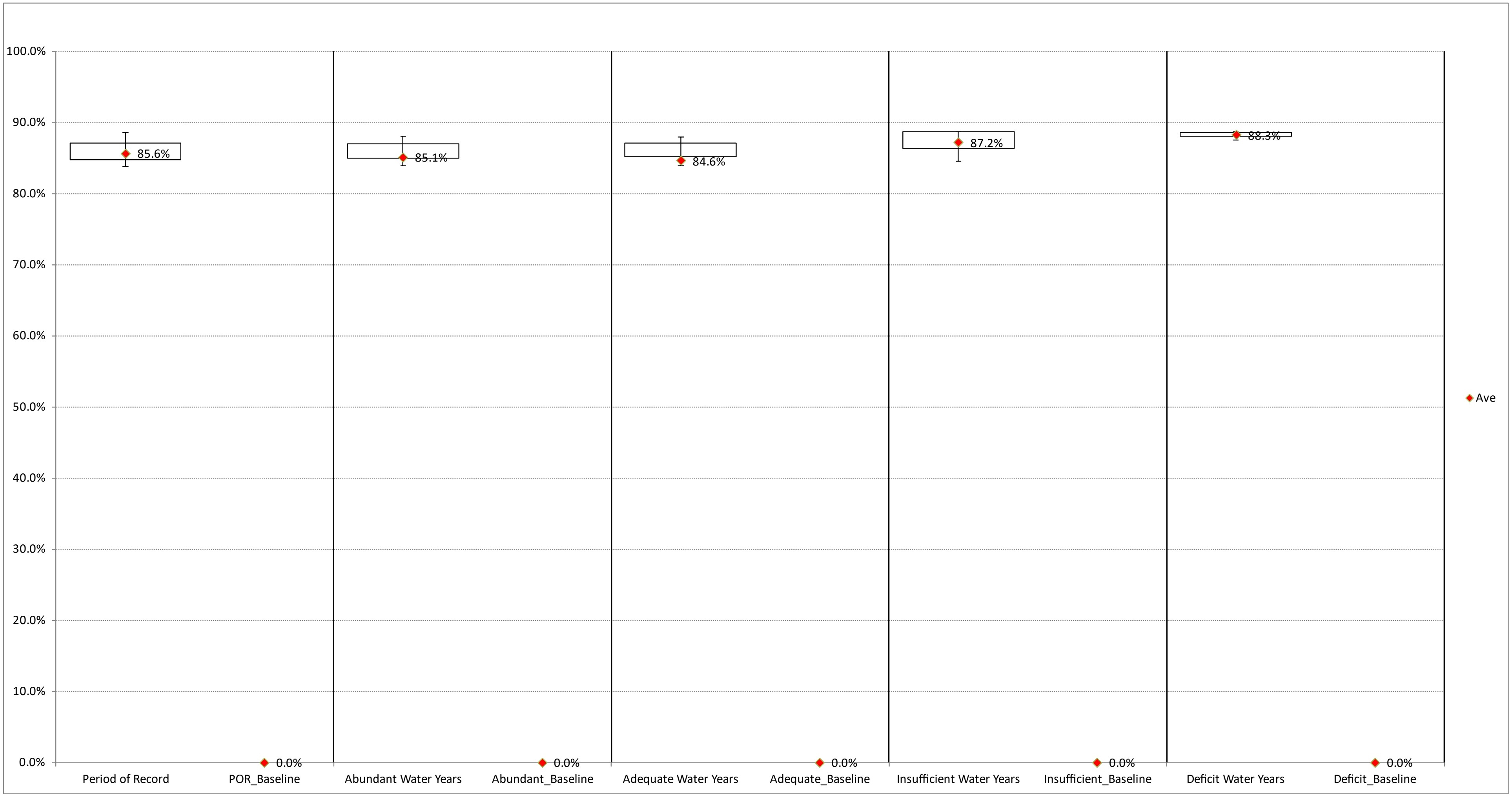

Figure 2-117. Green Peter 2-Year-Old Juvenile Winter Steelhead Downstream Dam Passage Survival Under Alternative 1. *Downstream dam passage survival at Green Peter for juvenile winter steelhead 2 year olds under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

STEELHEAD

2.8 STEELHEAD ALTERNATIVE 2A AND 2B

2.8.1 North Santiam - Detroit

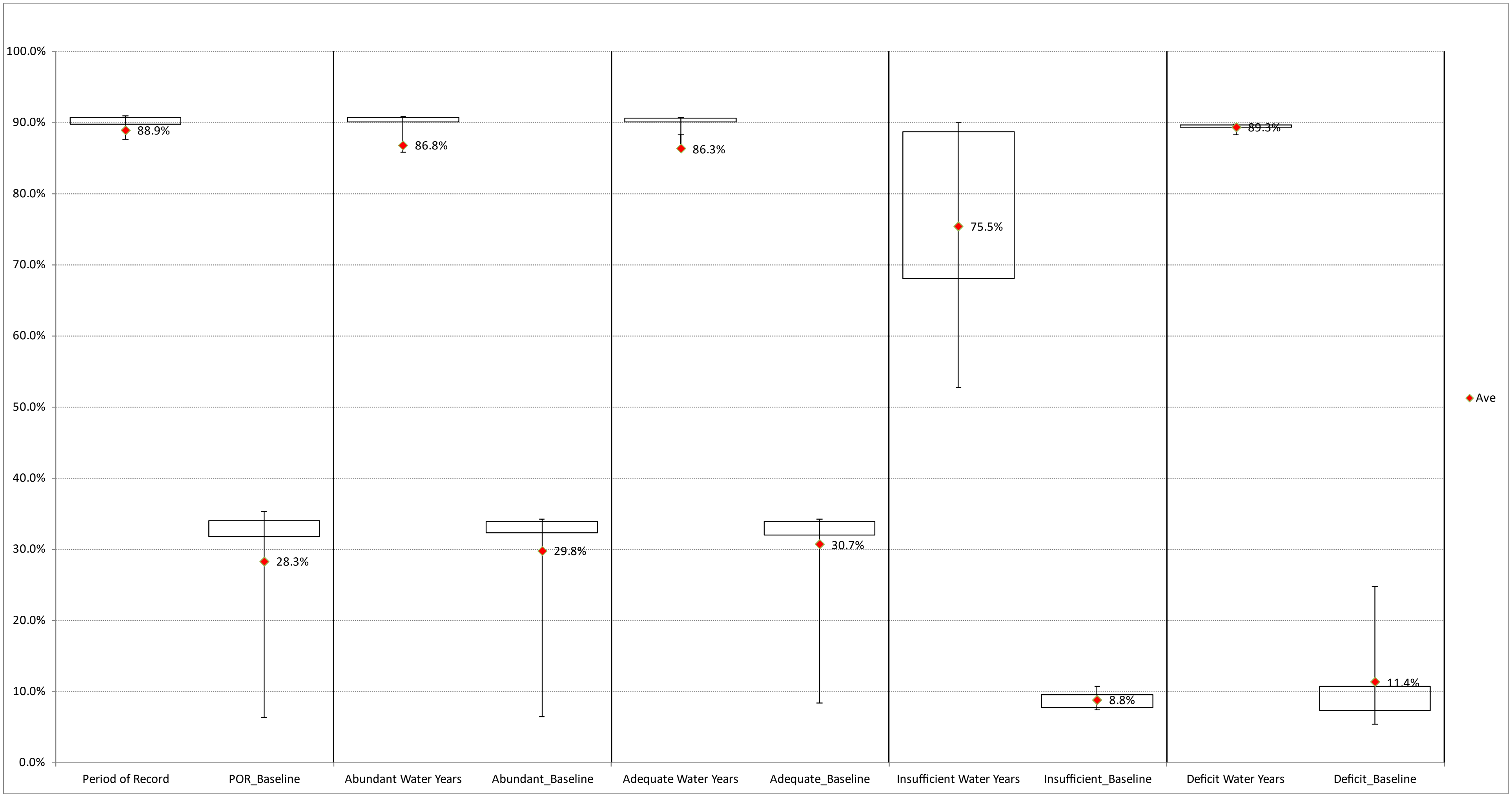

Figure 2-118. Detroit Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under Alternative 2a and 2b. *Downstream dam passage survival at Detroit for juvenile winter steelhead sub-yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

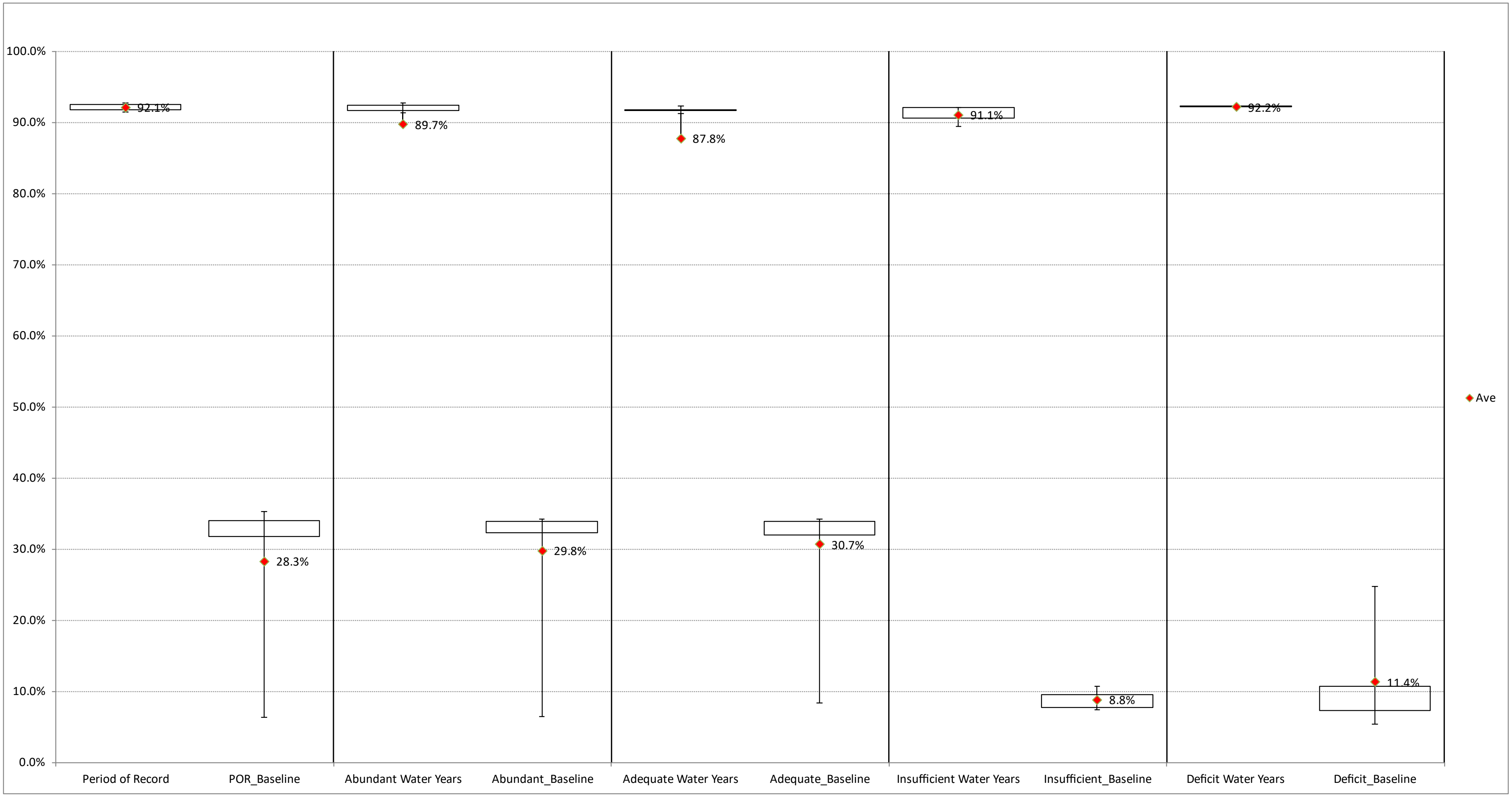

Figure 2-119. Detroit Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 2a and 2b. *Downstream dam passage survival at Detroit for juvenile winter steelhead yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

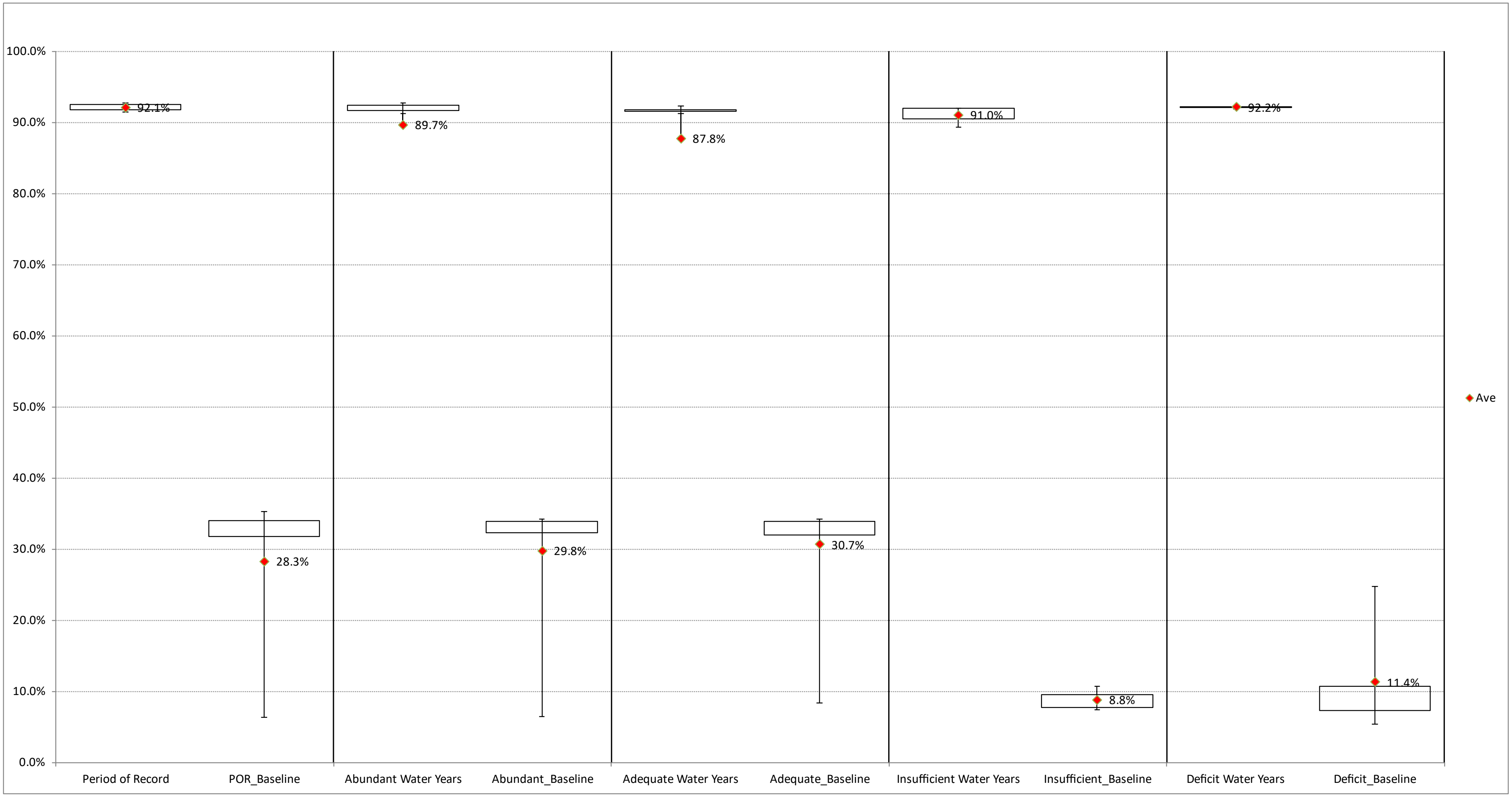

Figure 2-120. Detroit 2-Year-Old Juvenile Winter Steelhead Downstream Dam Passage Survival Under Alternative 2a and 2b. *Downstream dam passage survival at Detroit for juvenile winter steelhead 2 year olds under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.8.2 South Santiam – Foster

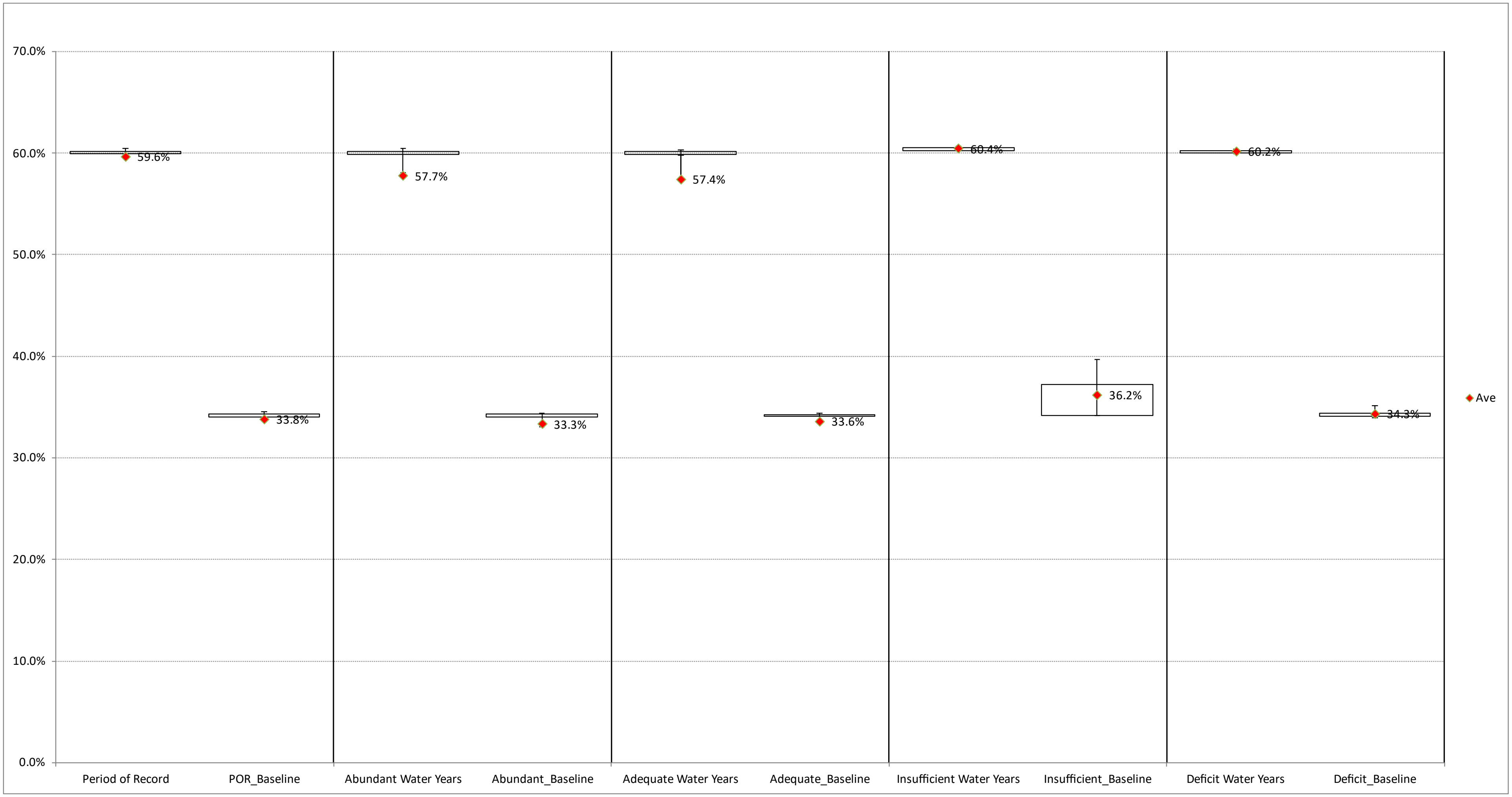

Figure 2-121. Foster Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under Alternative 2a and 2b. *Downstream dam passage survival at Foster for juvenile winter steelhead sub-yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

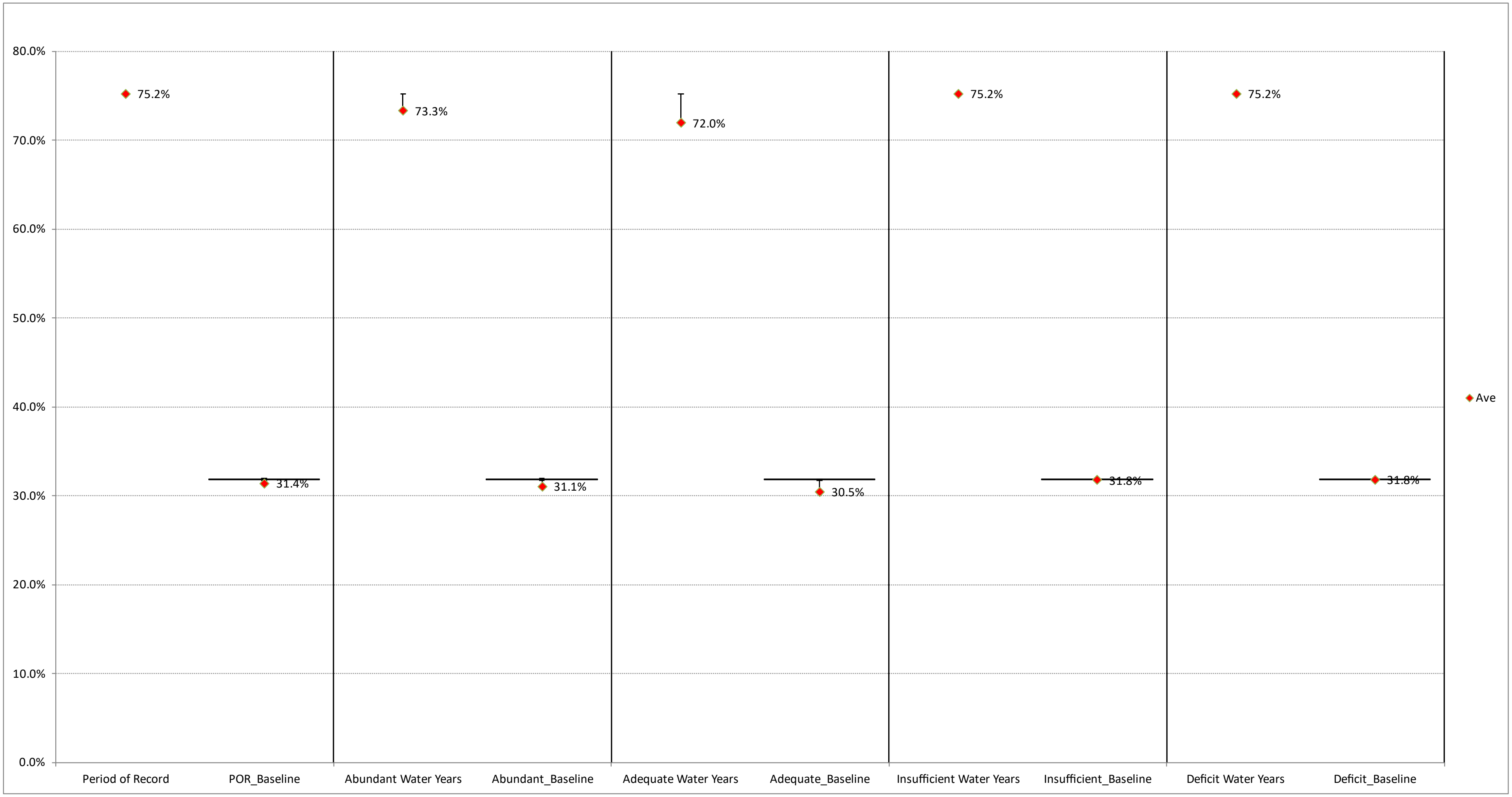

Figure 2-122. Foster Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 2a and 2b. *Downstream dam passage survival at Foster for juvenile winter steelhead yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

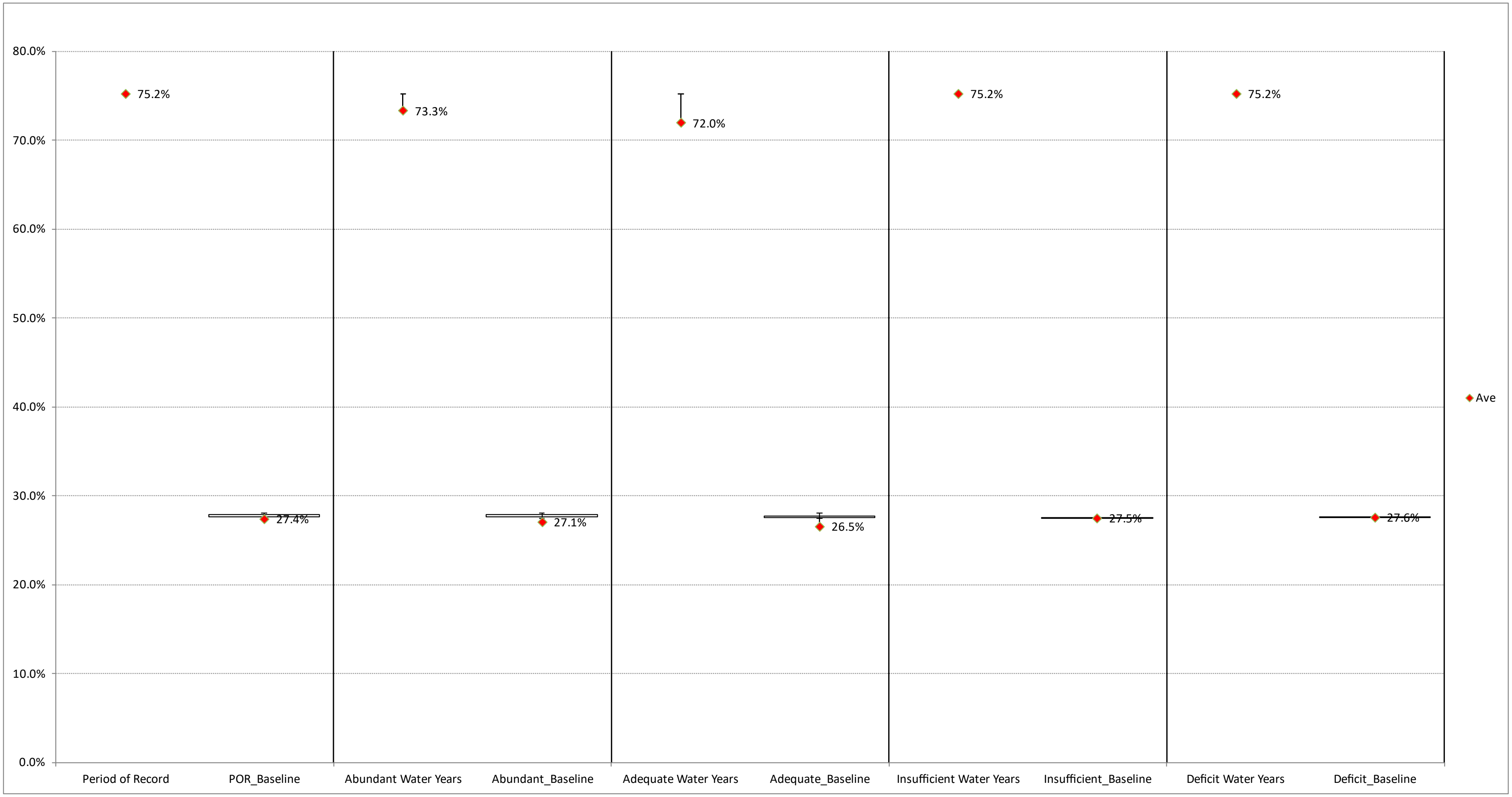

Figure 2-123. Foster 2-Year-Old Juvenile Winter Steelhead Downstream Dam Passage Survival Under Alternative 2a and 2b. *Downstream dam passage survival at Foster for juvenile winter steelhead 2 year olds under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.8.3 South Santiam – Green Peter

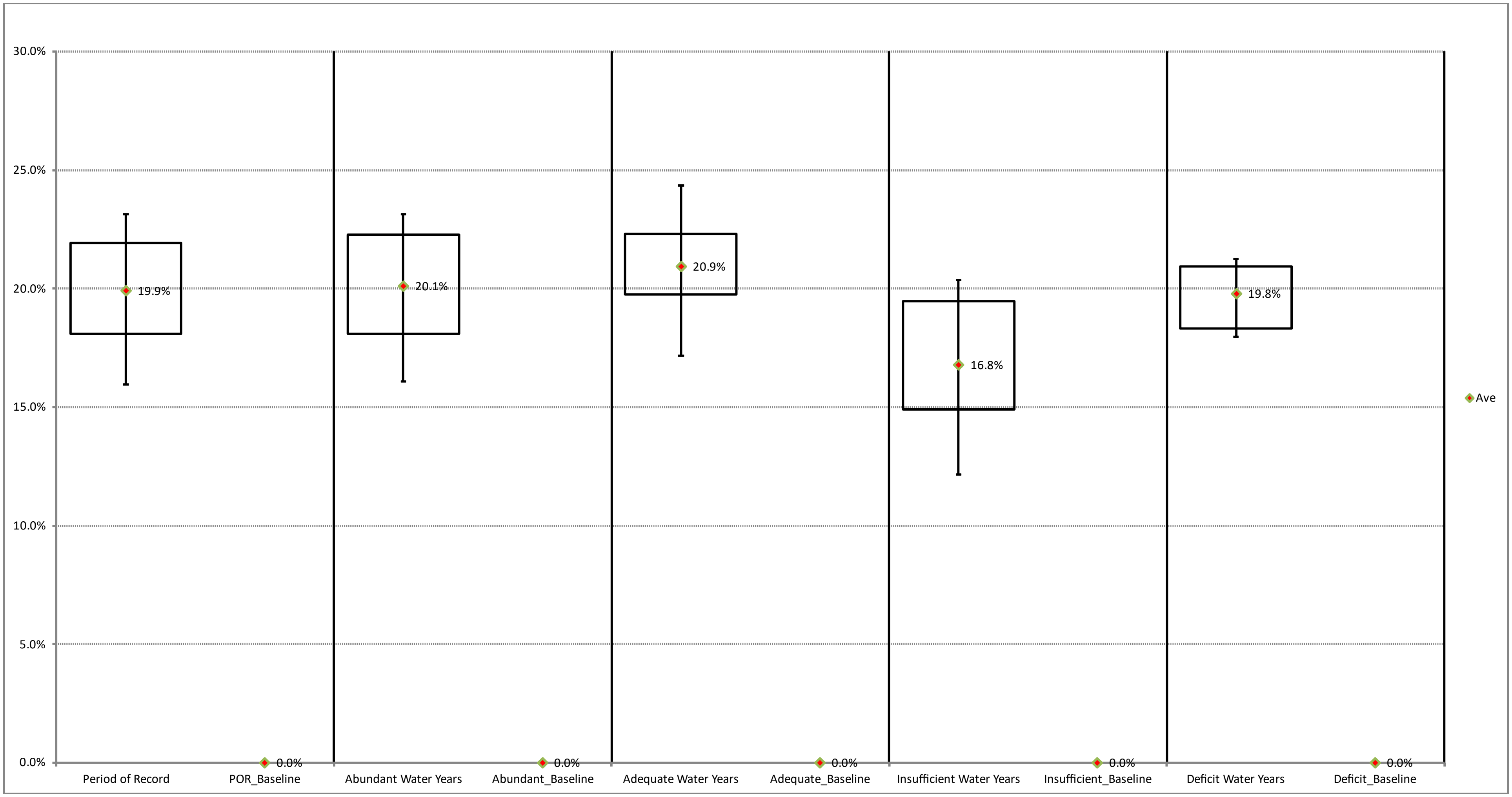

Figure 2-124. Green Peter Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under Alternative 2a and 2b. *Downstream dam passage survival at Green Peter for juvenile winter steelhead sub-yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

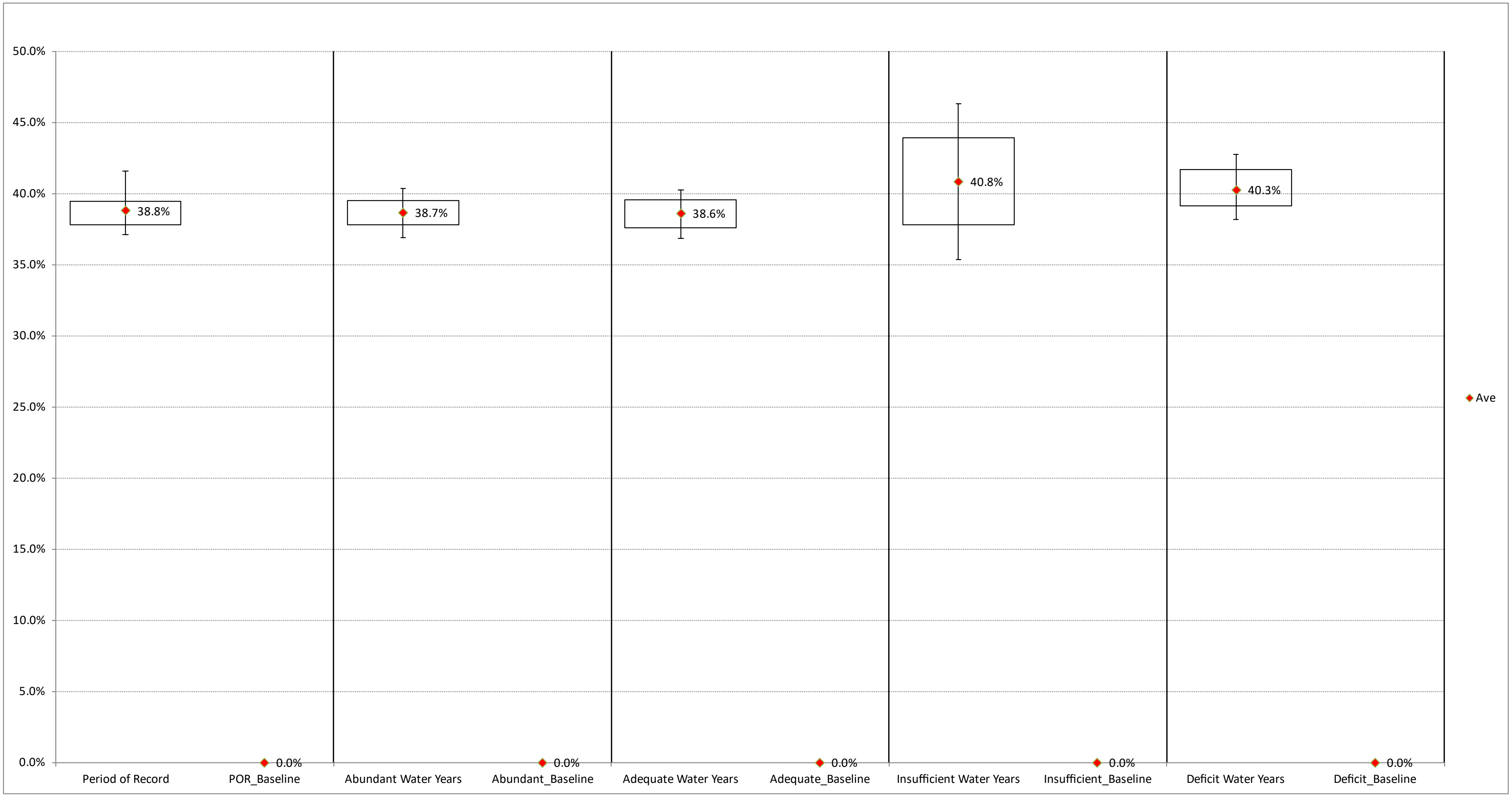

Figure 2-125. Green Peter Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 2a and 2b. *Downstream dam passage survival at Green Peter for juvenile winter steelhead yearlings under Alternative 2a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

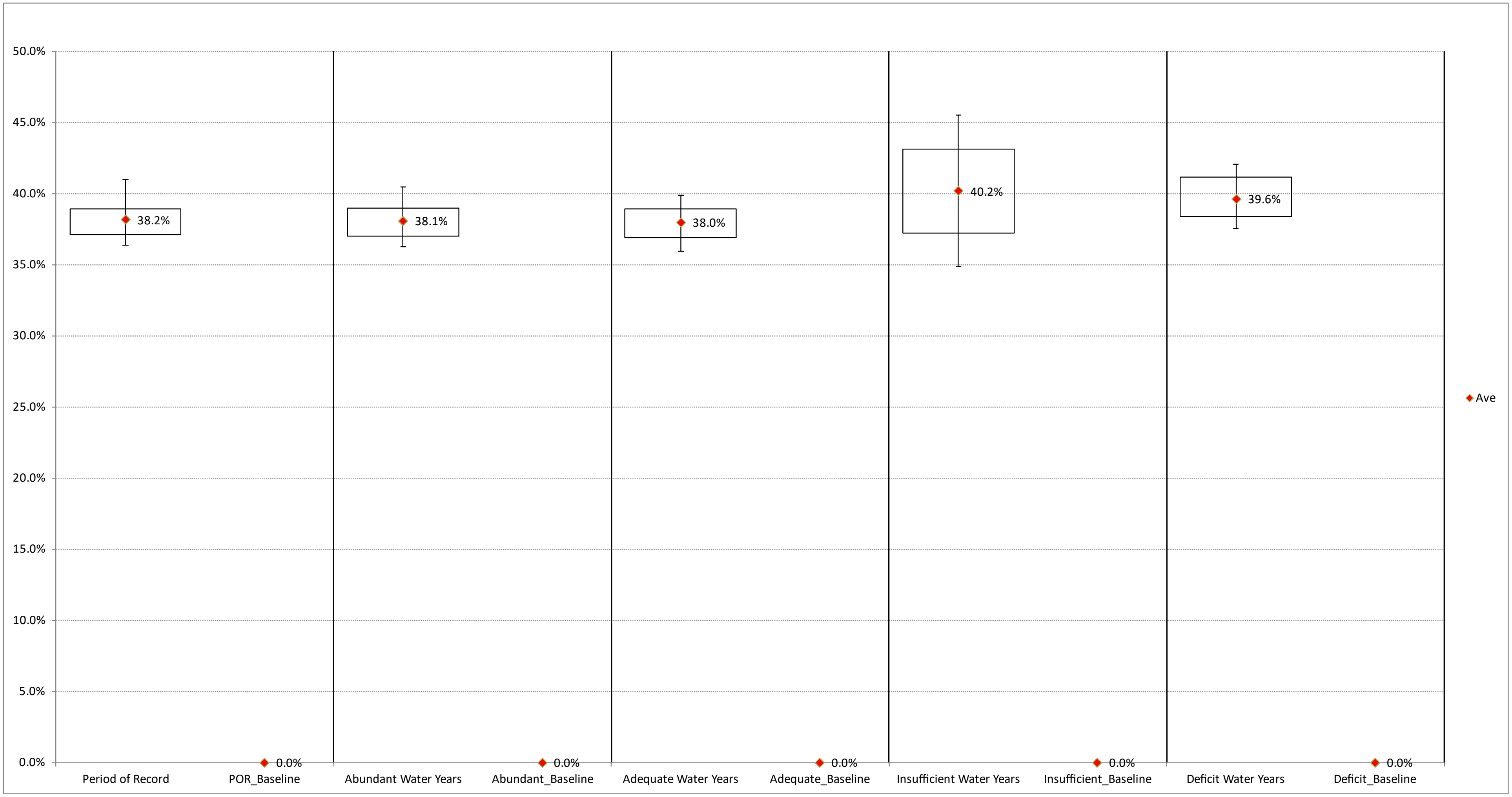

Figure 2-126. Green Peter 2-Year-Old Juvenile Winter Steelhead Downstream Dam Passage Survival Under Alternative 2a and 2b. *Downstream dam passage survival at Detroit for juvenile winter steelhead 2 year olds under Alternative 1. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

STEELHEAD

2.9 STEELHEAD ALTERNATIVE 3A

2.9.1 North Santiam – Detroit

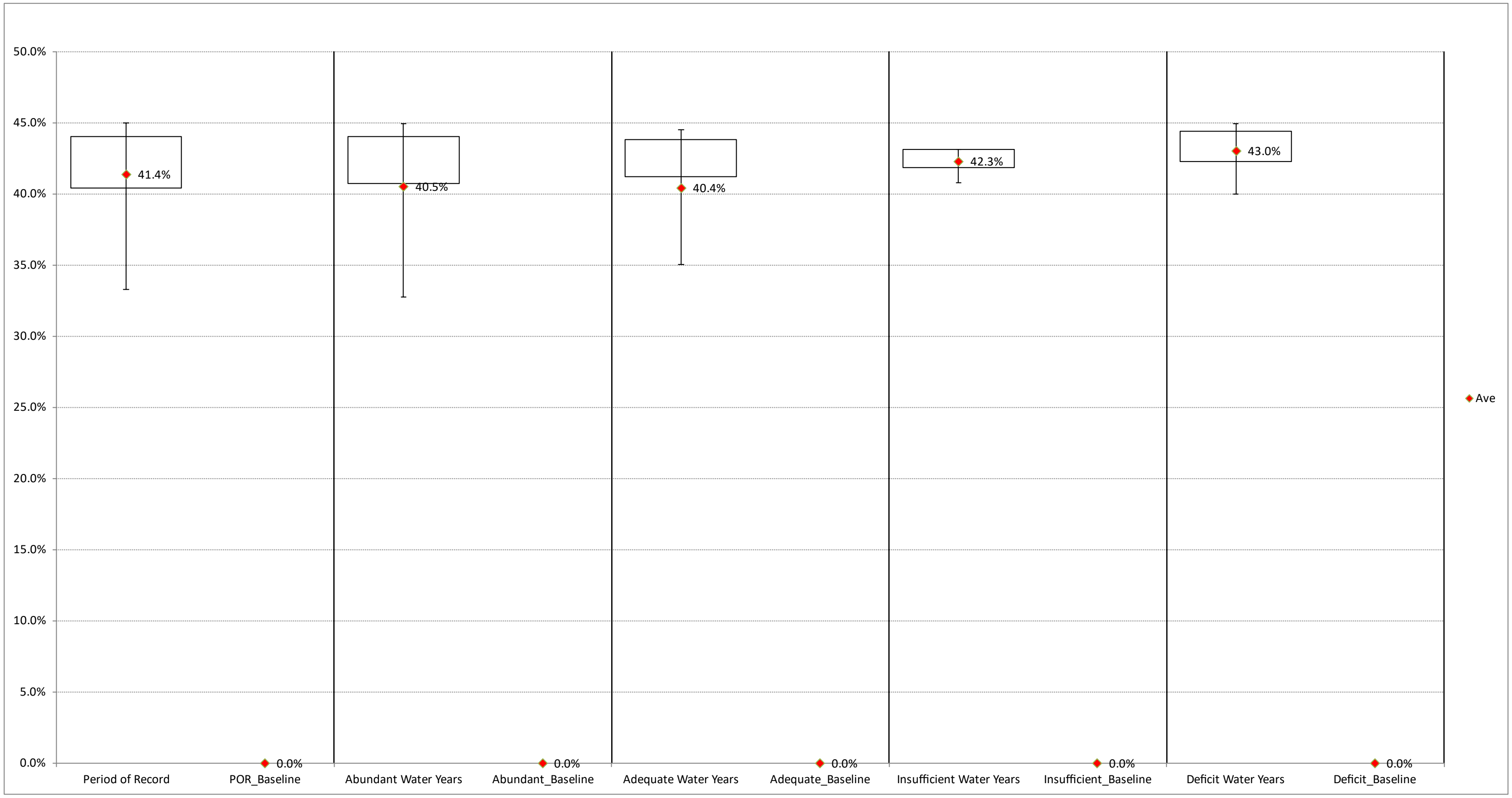

Figure 2-127. Detroit Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under Alternative 3a. Downstream dam passage survival at Detroit for juvenile winter steelhead sub-yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel

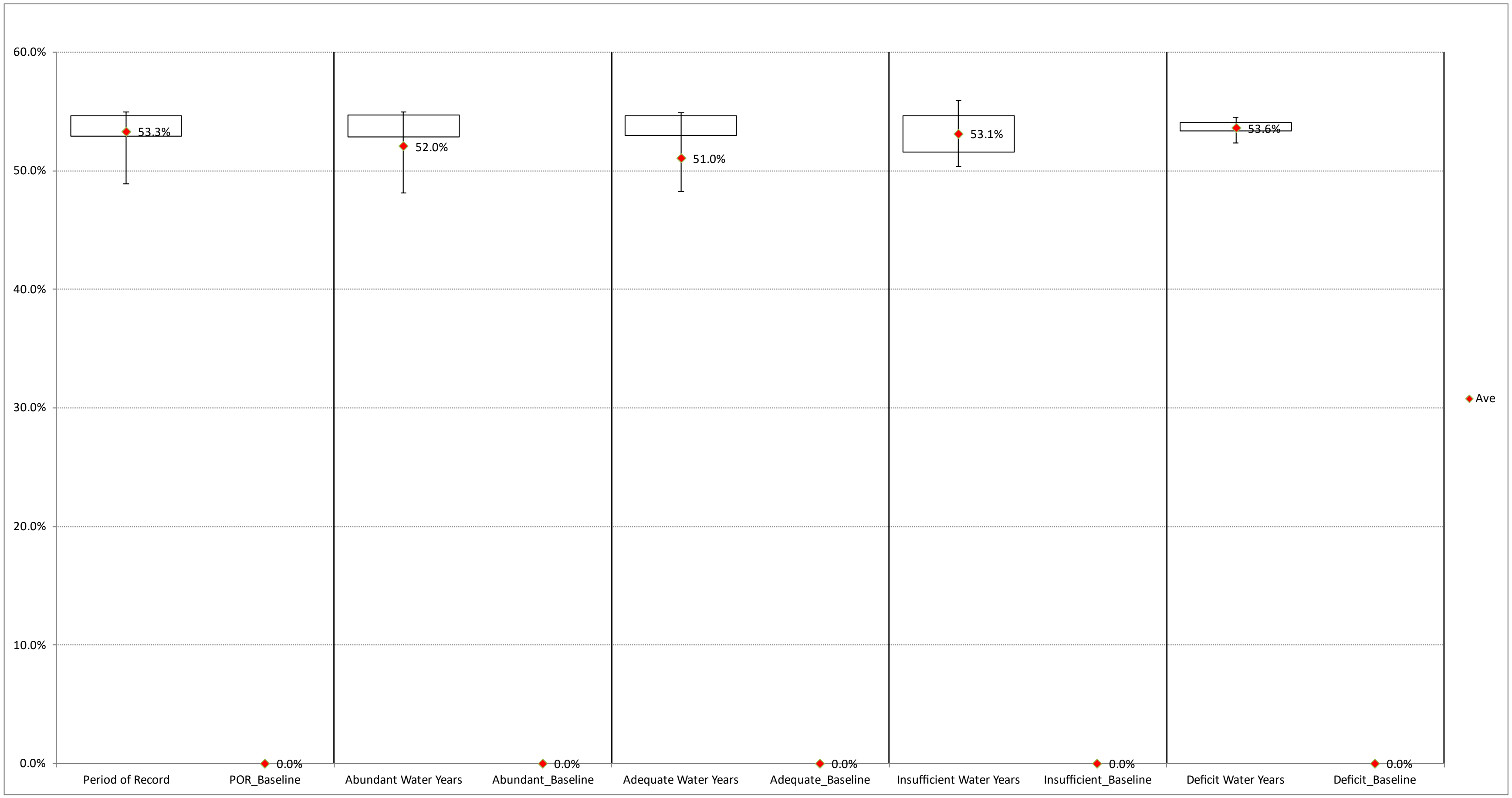

Figure 2-128. Detroit Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 3a. Downstream dam passage survival at Detroit for juvenile winter steelhead yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel

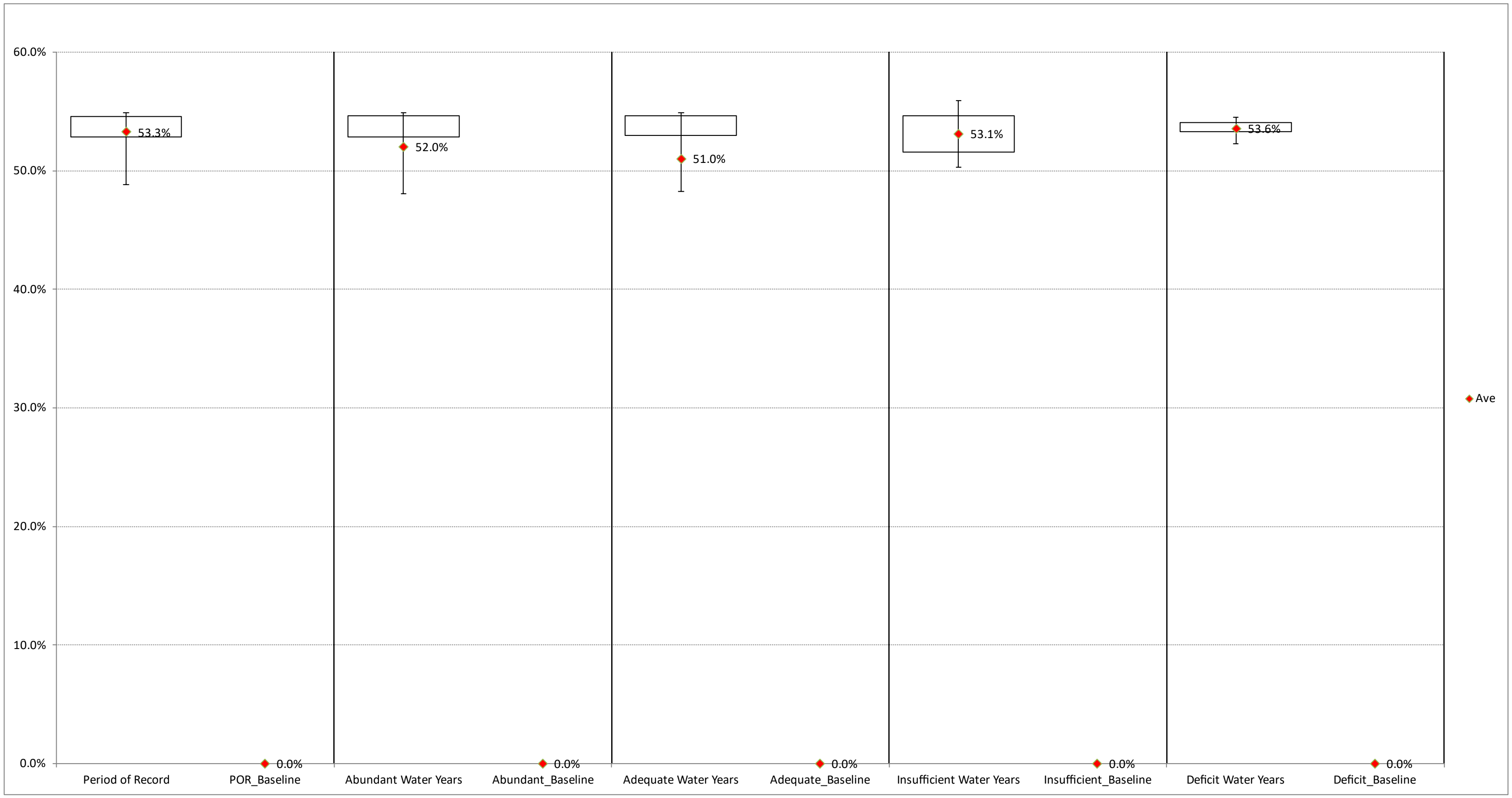

Figure 2-129. Detroit 2-Year-Old Juvenile Winter Steelhead Downstream Dam Passage Survival Under Alternative 3a. Downstream dam passage survival at Detroit for juvenile winter steelhead 2 year olds under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel

South Santiam – Foster

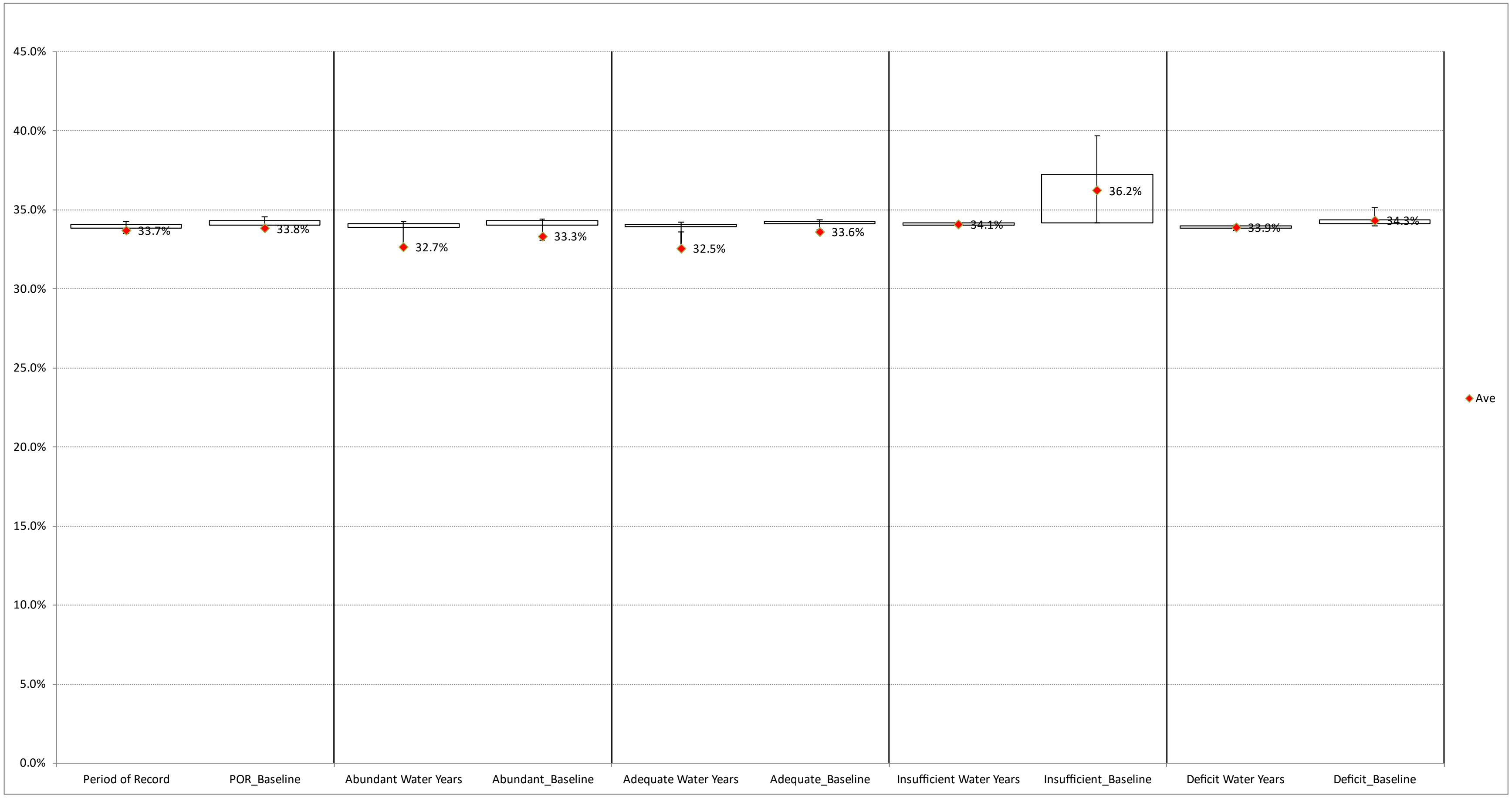

Figure 2-130. Foster Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under Alternative 3a. Downstream dam passage survival at Foster for juvenile winter steelhead sub-yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel

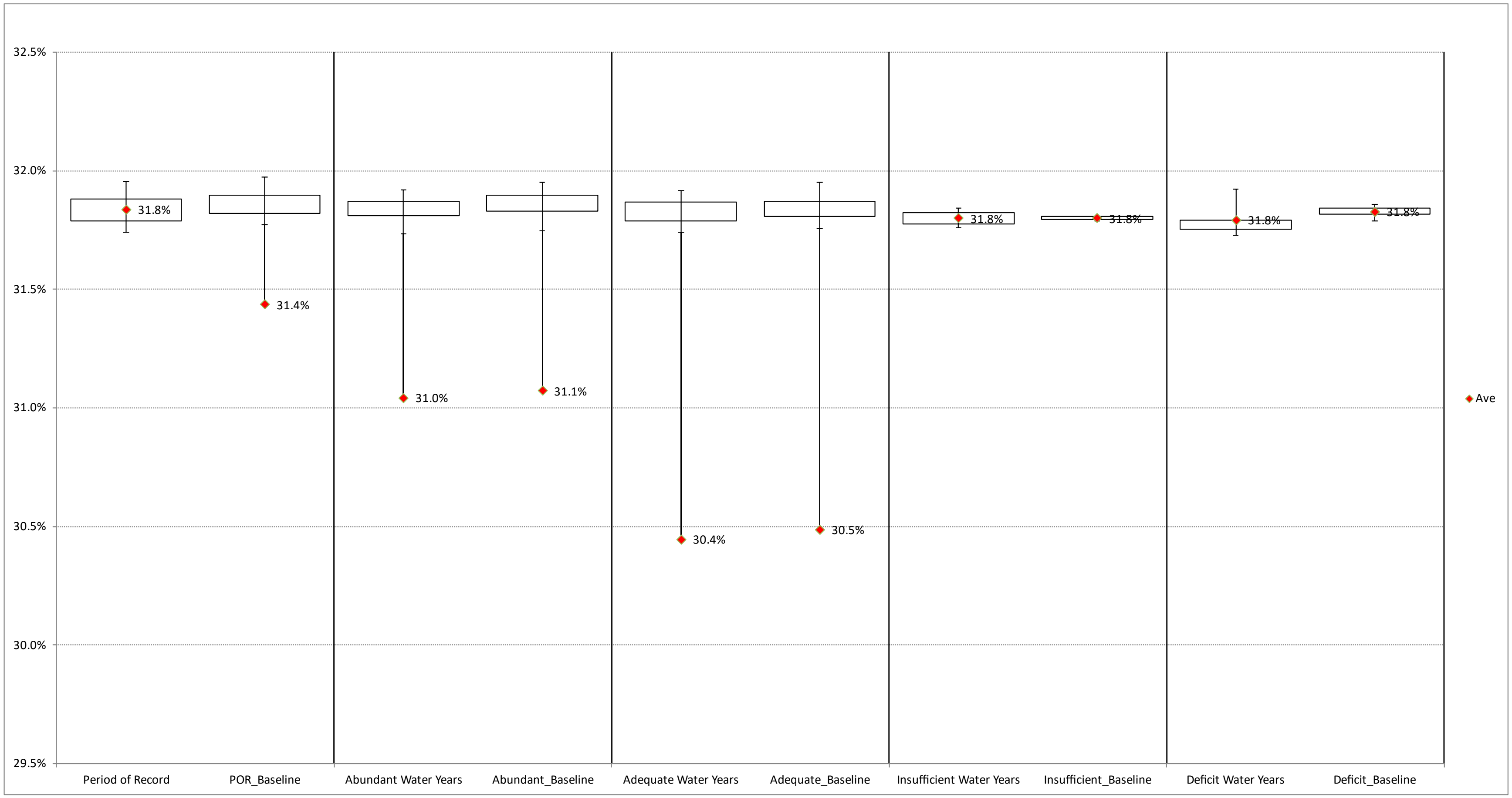

Figure 2-131. Foster Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 3a. Downstream dam passage survival at Foster for juvenile winter steelhead yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel

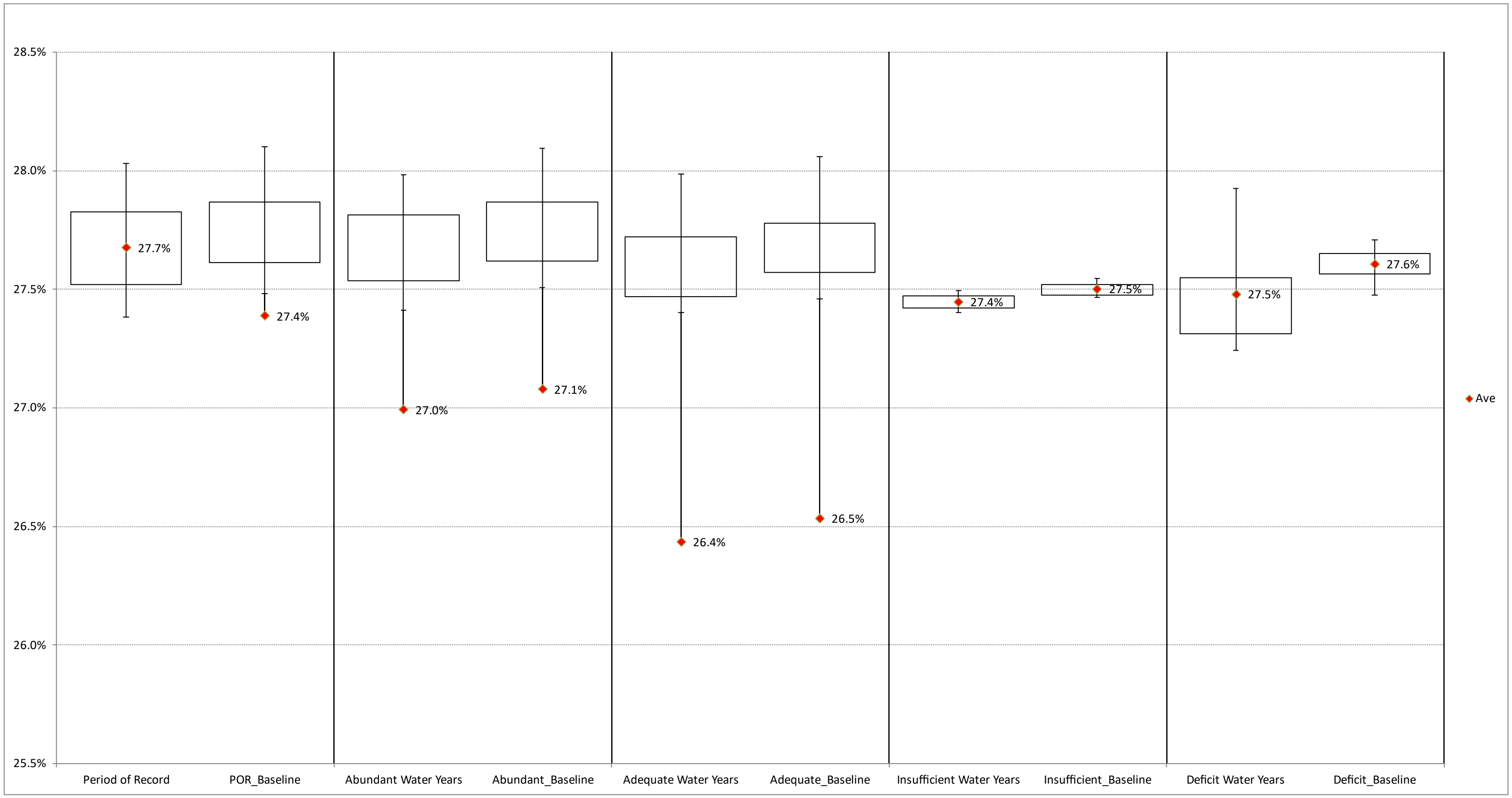

Figure 2-132. Foster Juvenile Winter Steelhead 2 Year Old Downstream Dam Passage Survival Under Alternative 3a. Downstream dam passage survival at Foster for juvenile winter steelhead 2 year olds under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel

2.9.2 South Santiam – Green Peter

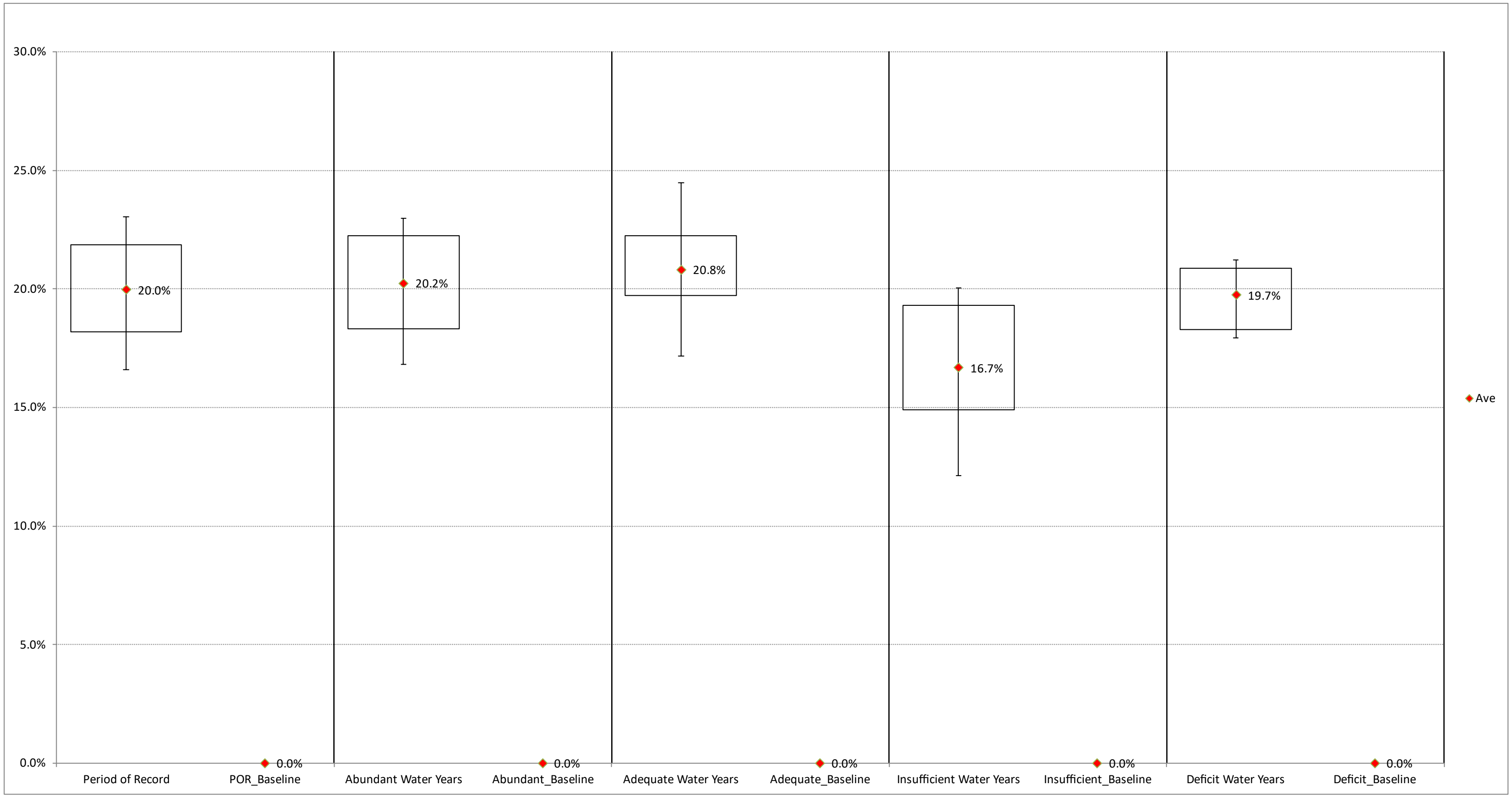

Figure 2-133. Green Peter Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under Alternative 3a. Downstream dam passage survival at Green Peter for juvenile winter steelhead sub-yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel

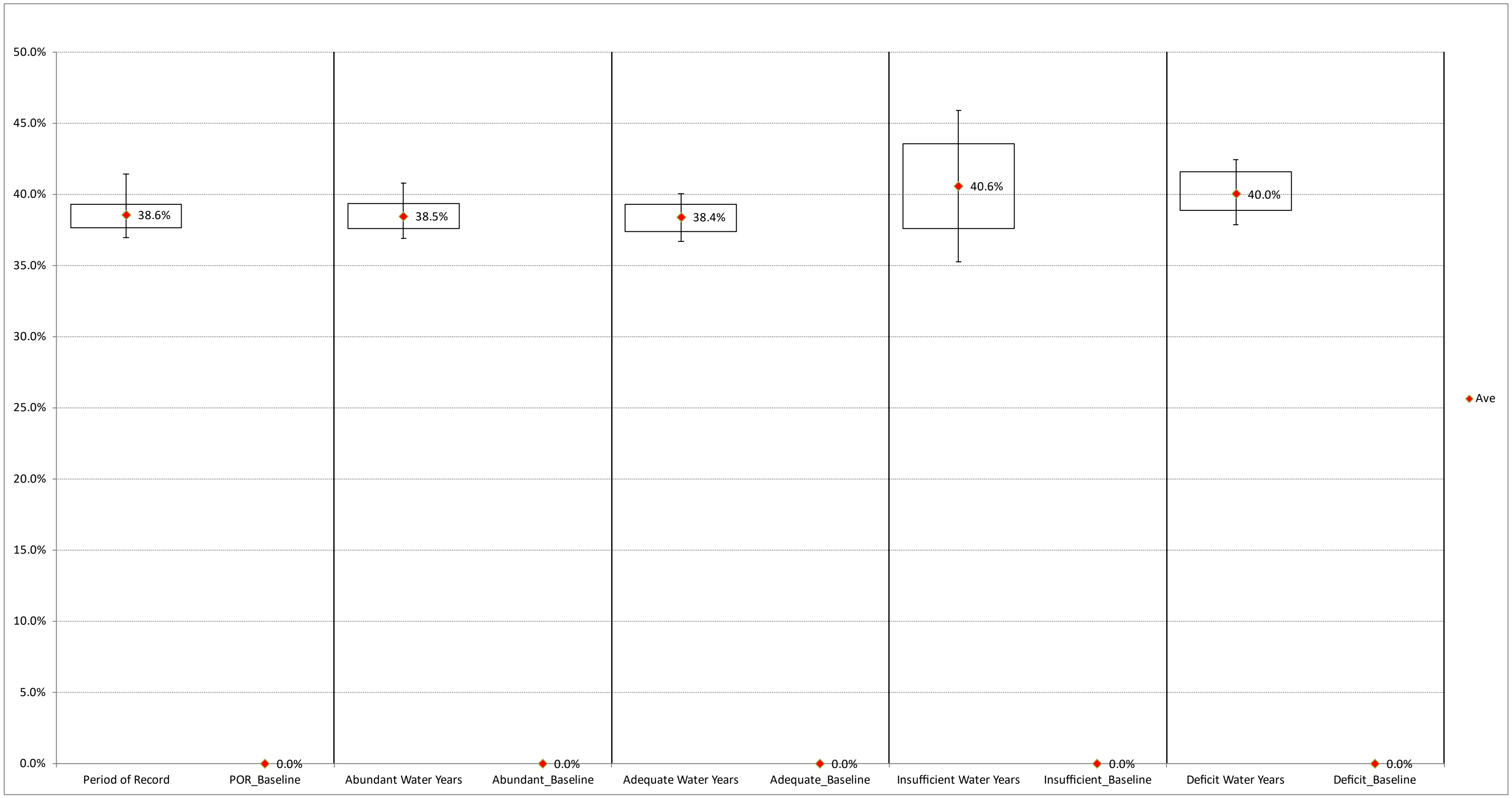

Figure 2-134. Green Peter Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 3a. Downstream dam passage survival at Green Peter for juvenile winter steelhead yearlings under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel

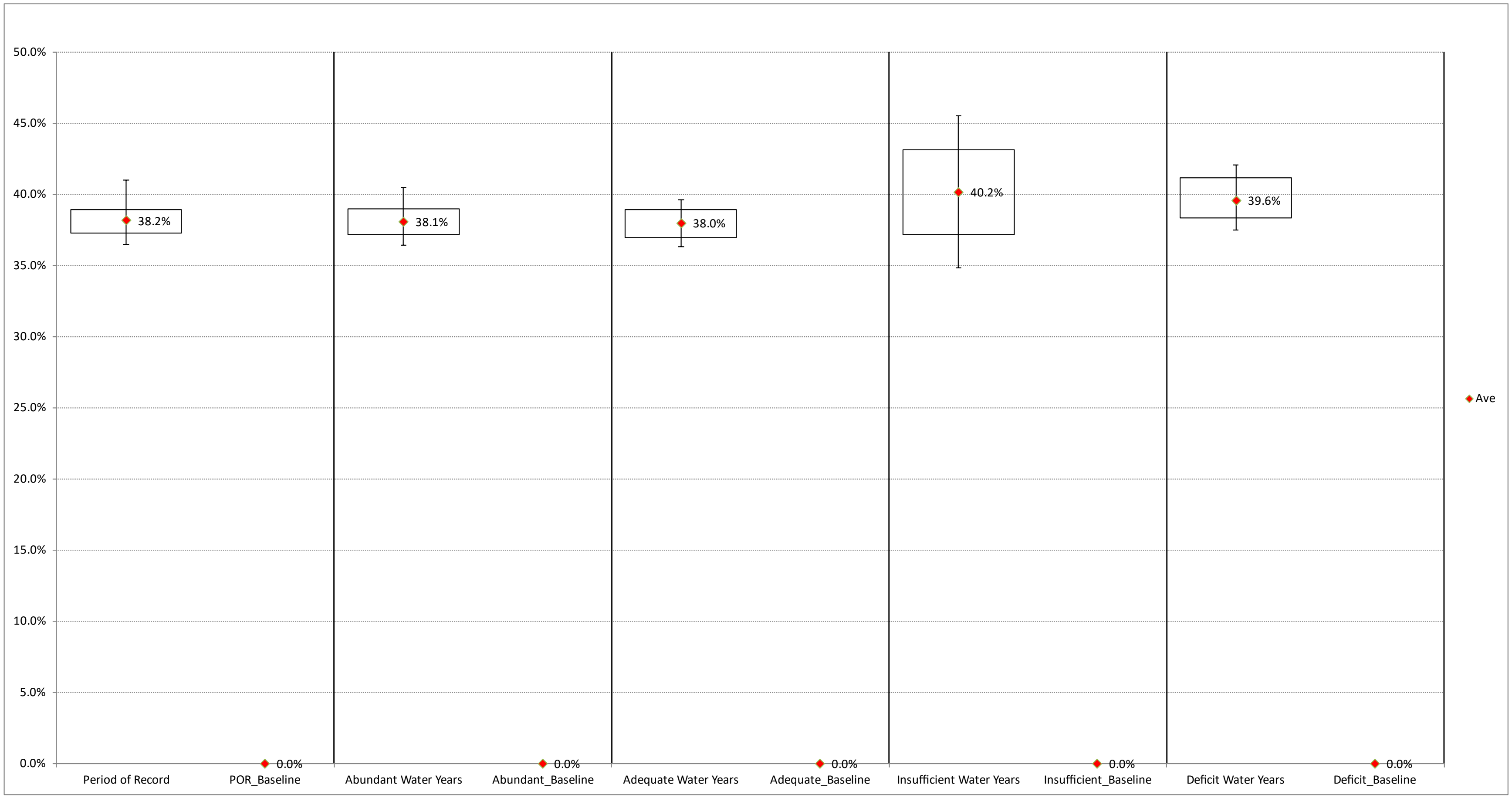

Figure 2-135. Green Peter 2-Year-Old Juvenile Winter Steelhead Downstream Dam Passage Survival Under Alternative 3a. Downstream dam passage survival at Green Peter for juvenile winter steelhead 2 year olds under Alternative 3a. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel

STEELHEAD

2.10 STEELHEAD ALTERNATIVE 3B

2.10.1 North Santiam – Detroit

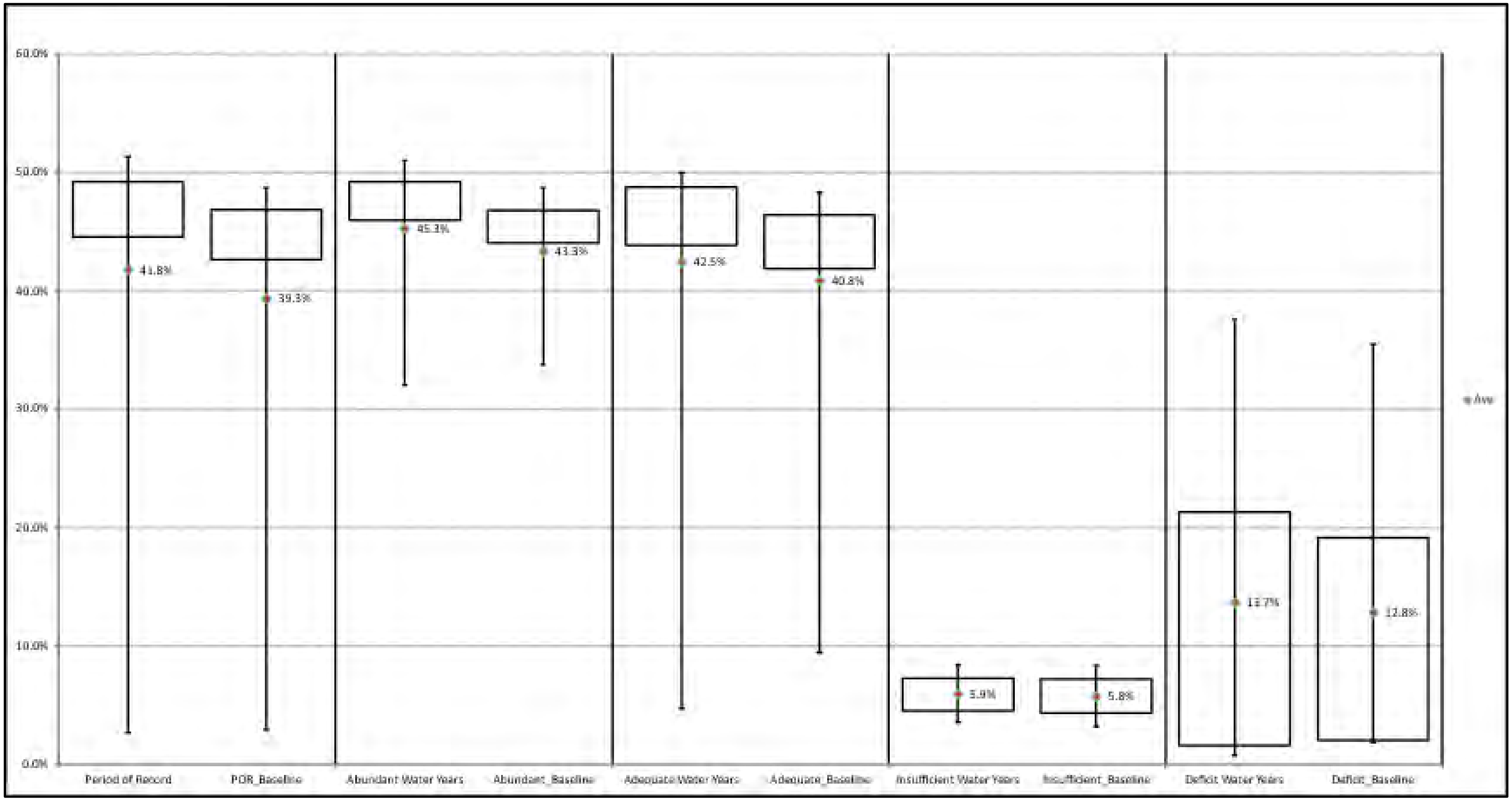

Figure 2-136. Detroit Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Detroit for juvenile winter steelhead sub-yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

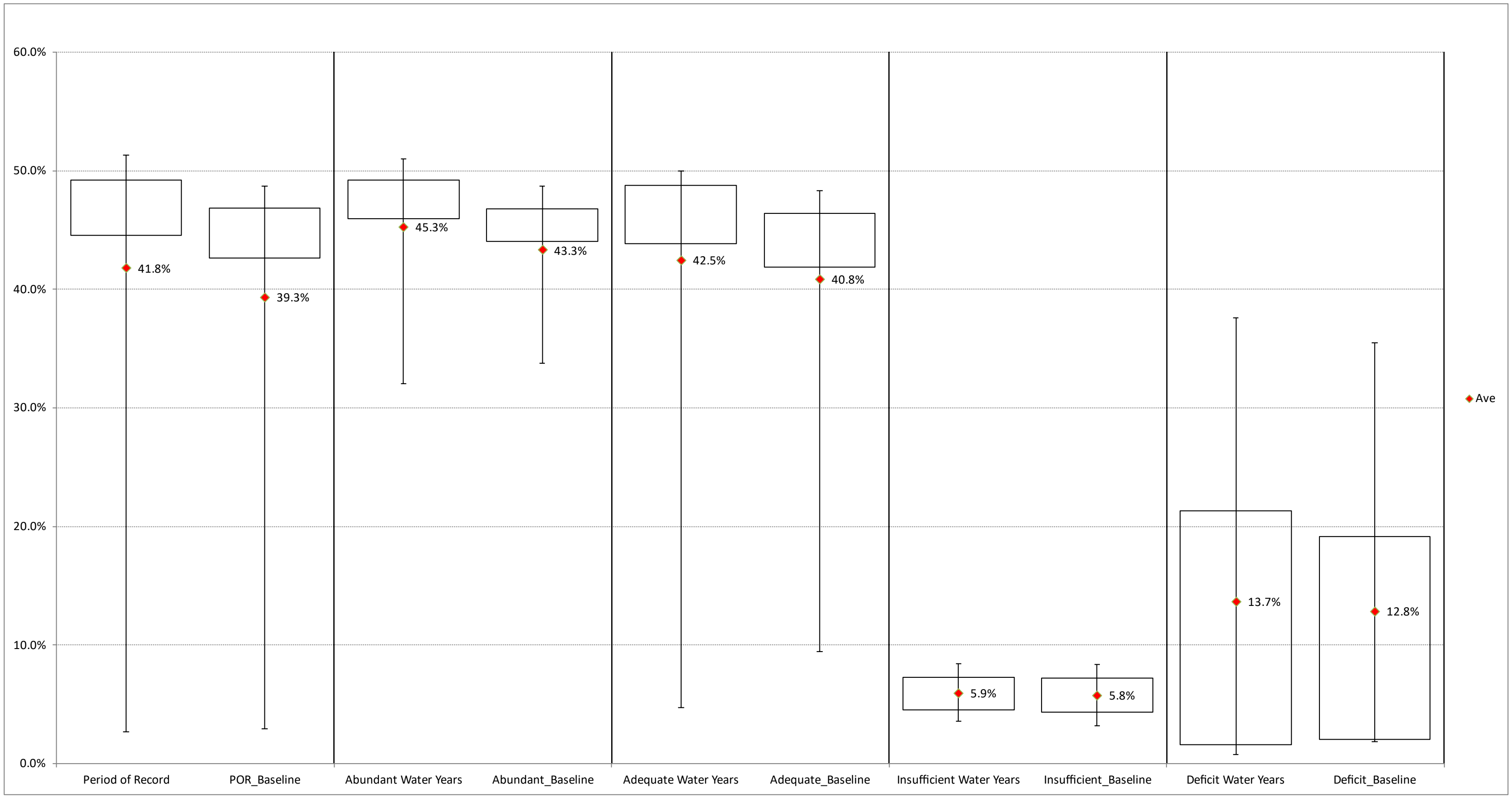

Figure 2-137. Detroit Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Detroit for juvenile winter steelhead yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

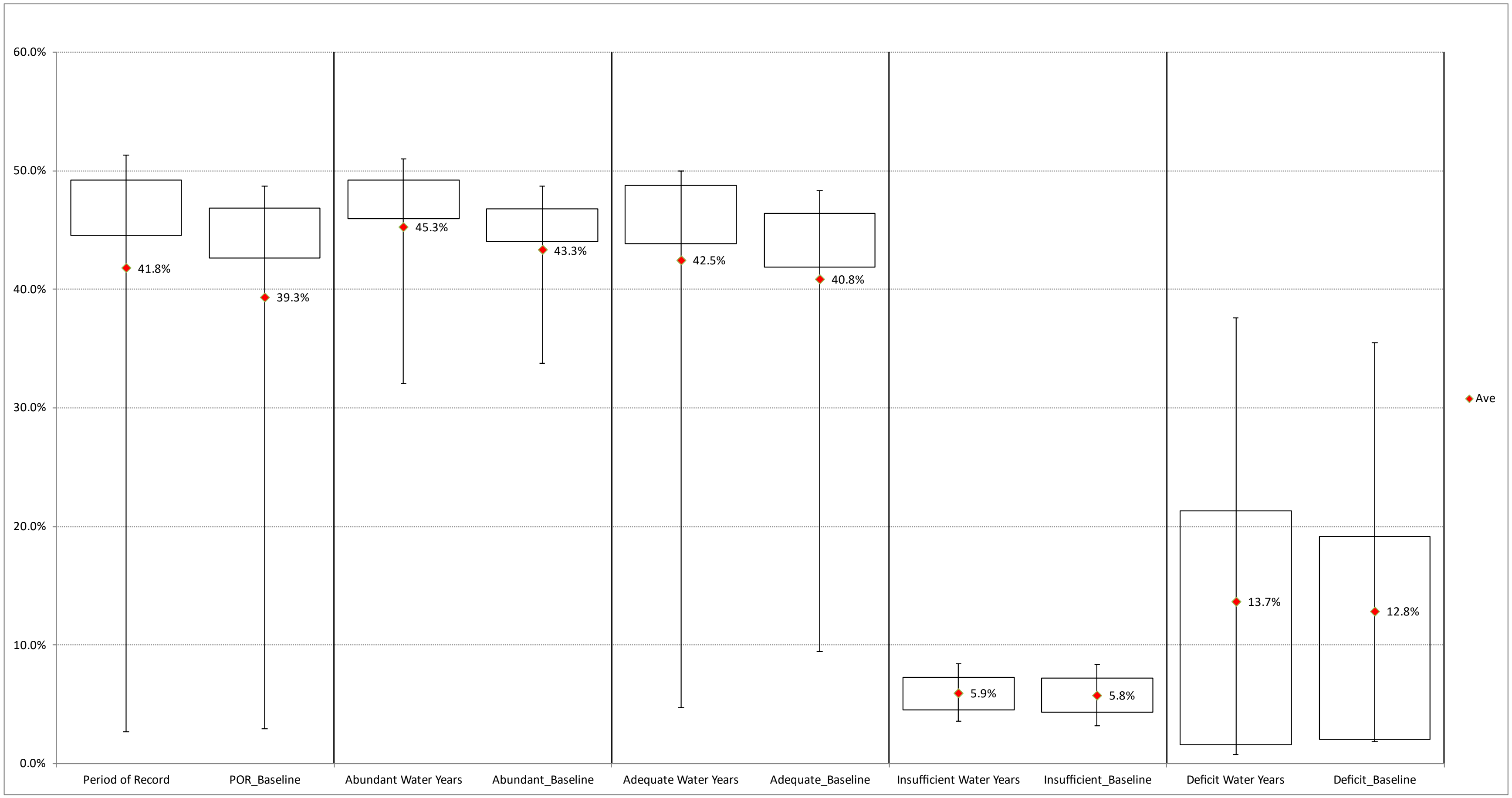

Figure 2-138. Detroit 2-Year-Old Juvenile Winter Steelhead Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Detroit for juvenile winter steelhead 2 year olds under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.10.2 South Santiam – Foster

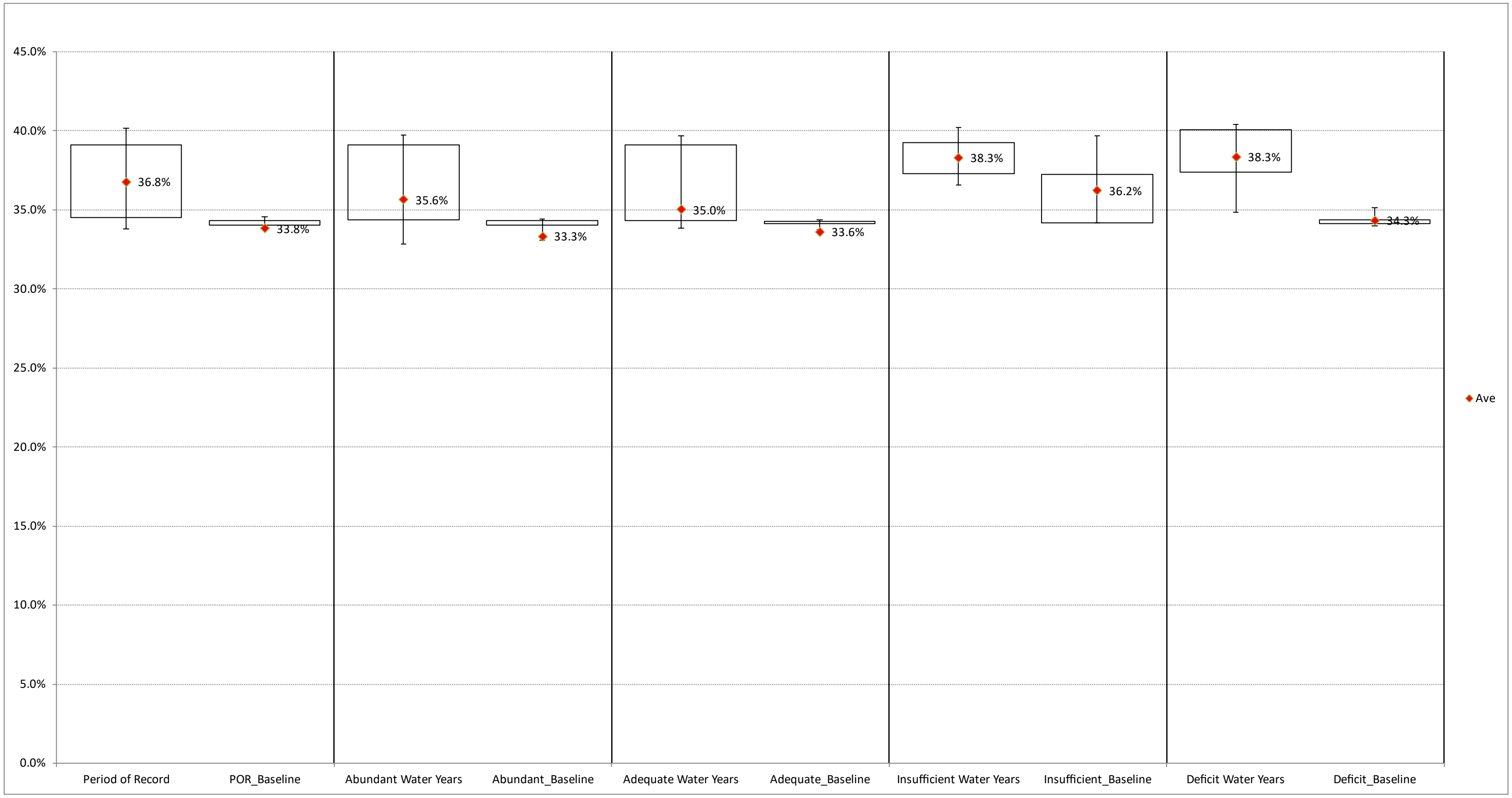

Figure 2-139. Foster Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Foster for juvenile winter steelhead sub-yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

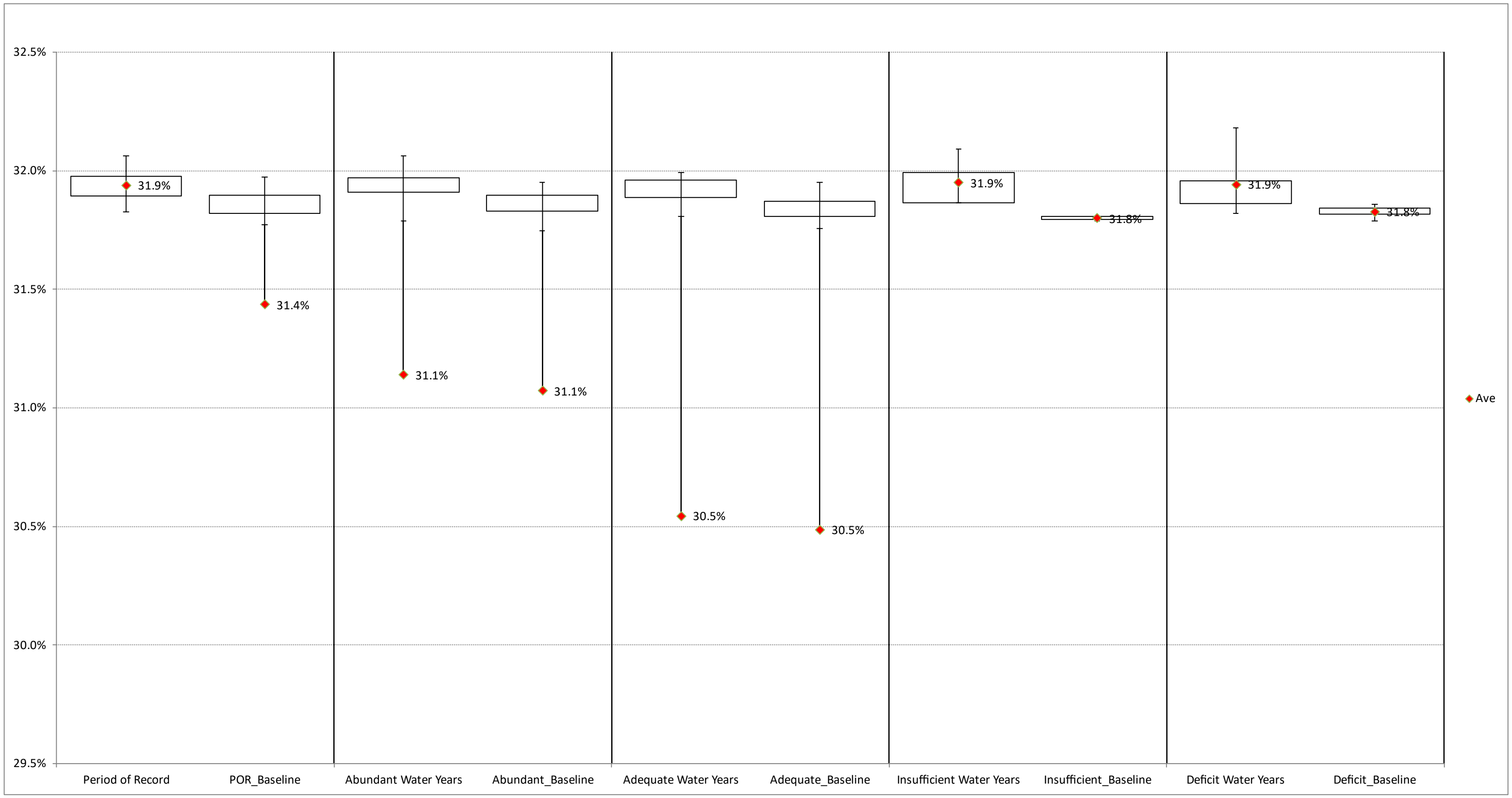

Figure 2-140. Foster Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Foster for juvenile winter steelhead yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

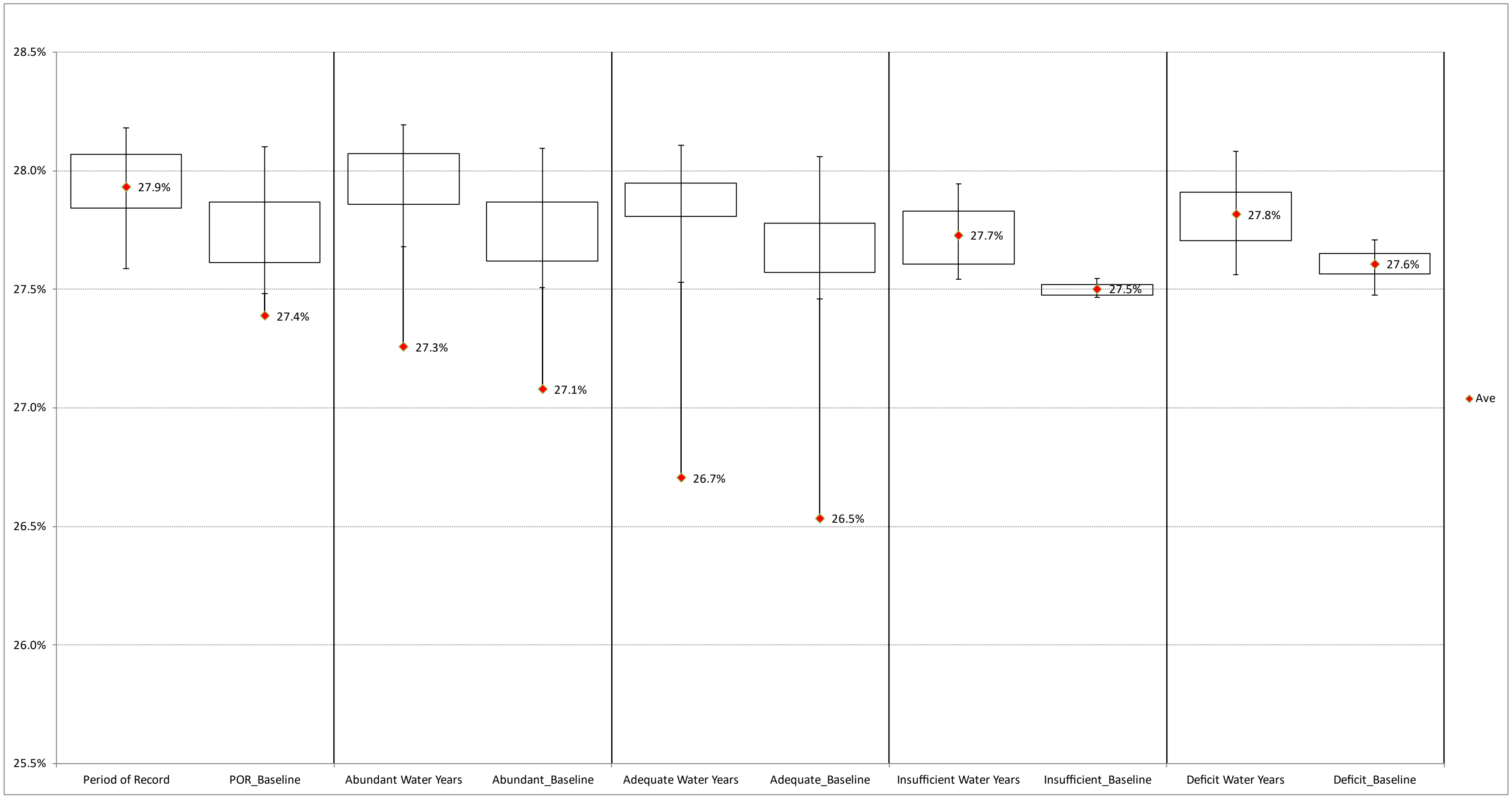

Figure 2-141. Foster 2-Year-Old Juvenile Winter Steelhead Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Foster for juvenile winter steelhead 2 year olds under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.10.3 South Santiam – Green Peter

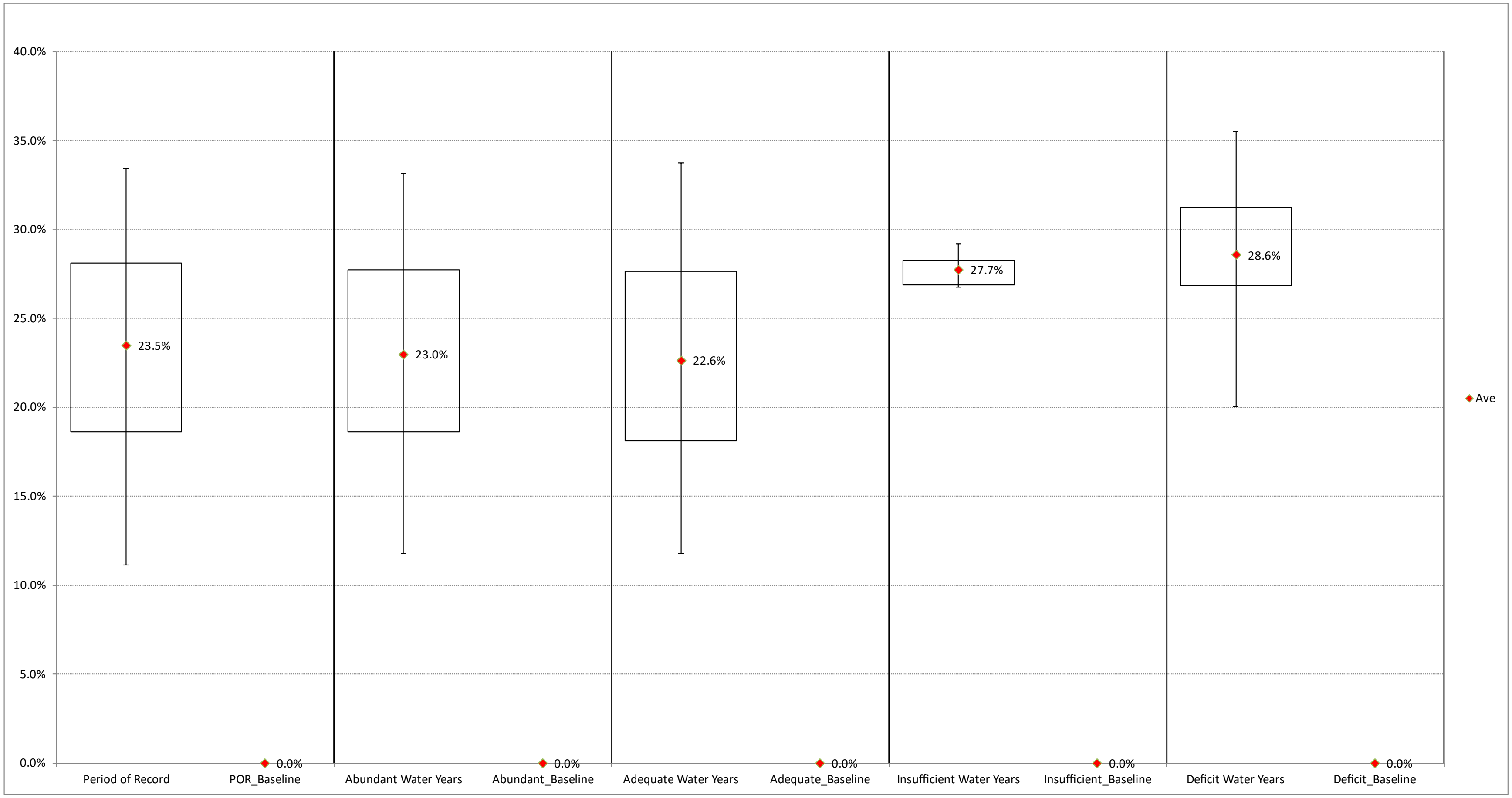

Figure 2-142. Green Peter Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Green Peter for juvenile winter steelhead sub-yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

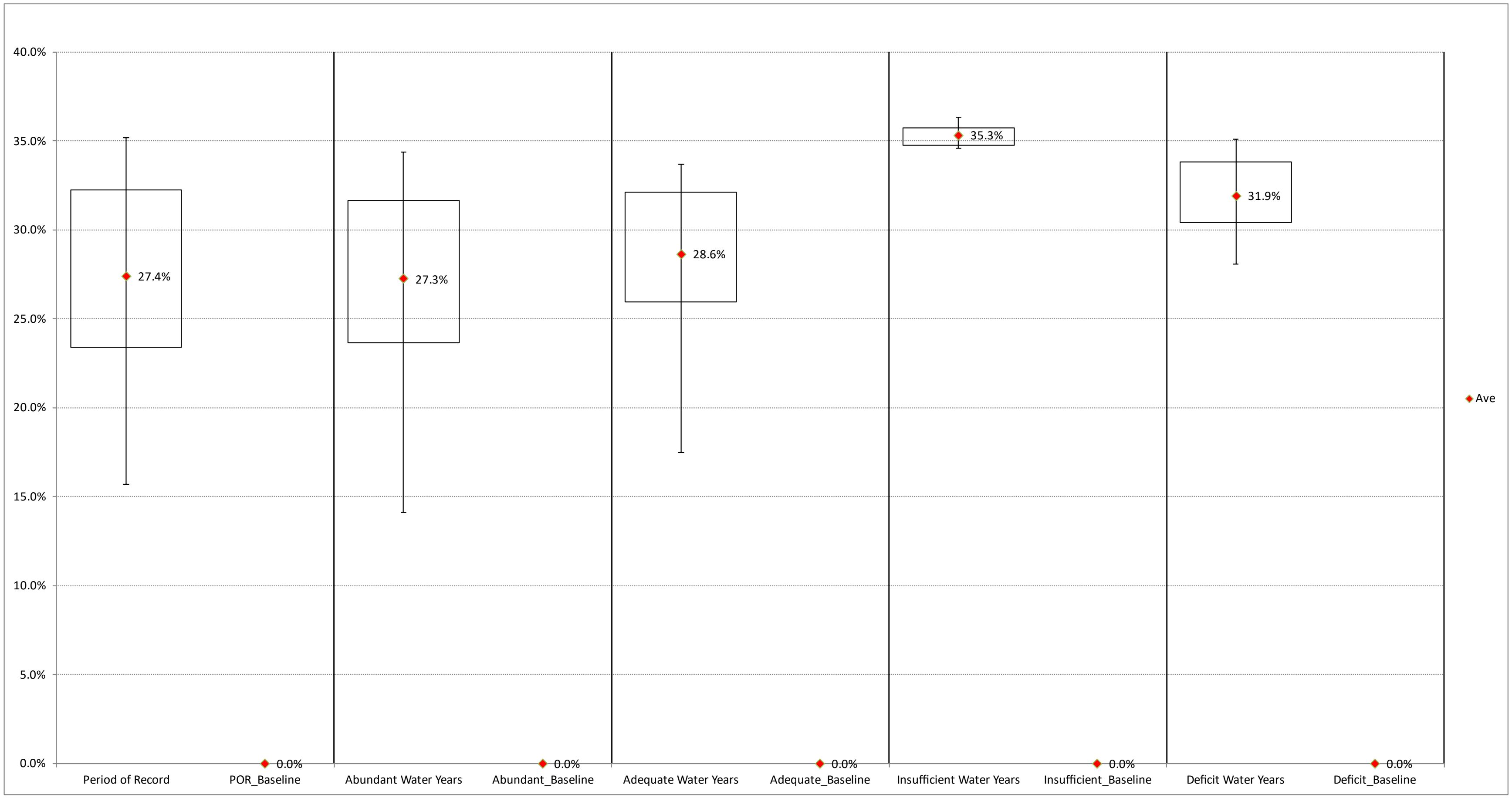

Figure 2-143. Green Peter Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Green Peter for juvenile winter steelhead yearlings under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

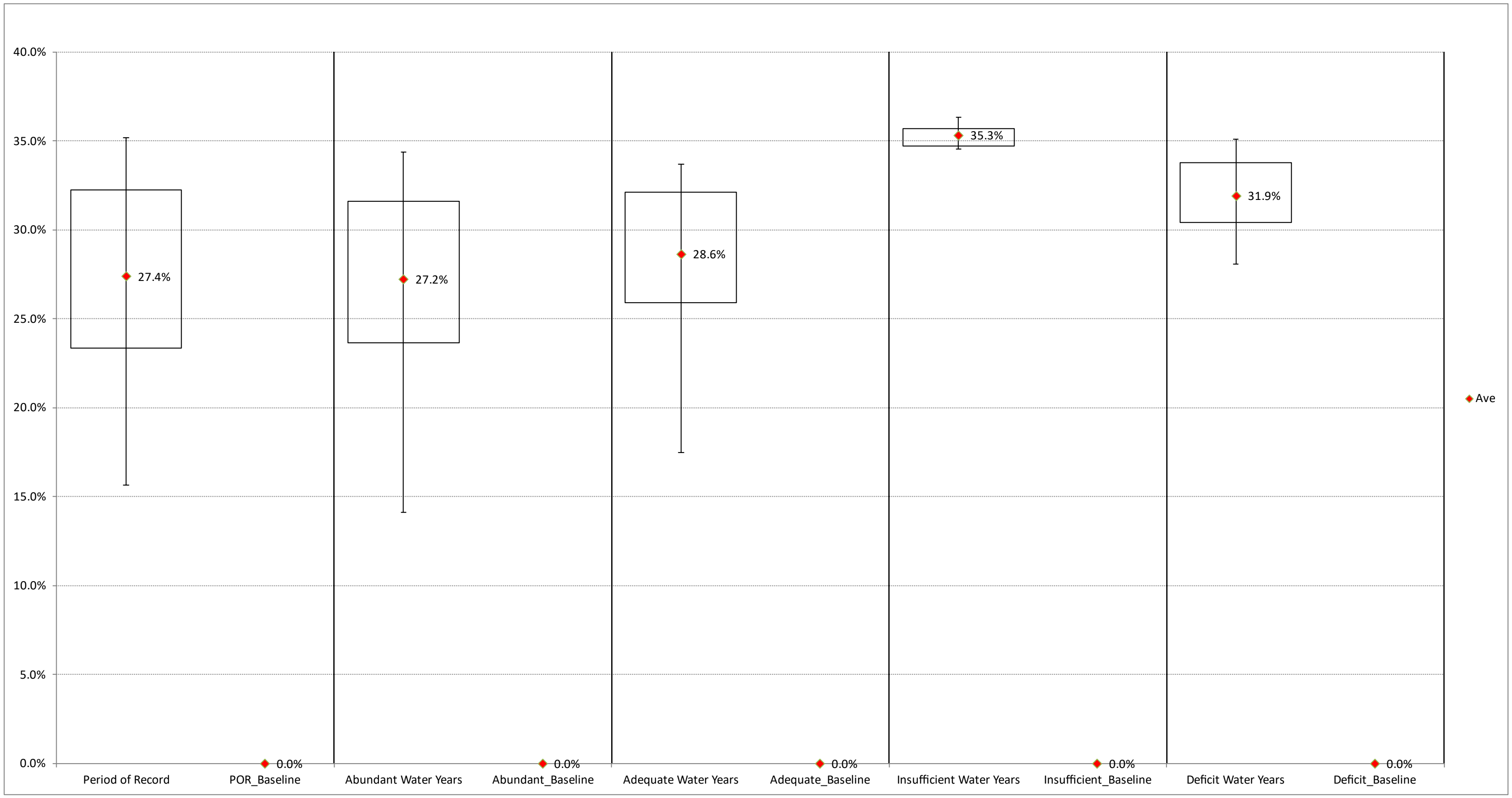

Figure 2-144. Green Peter 2-Year-Old Juvenile Winter Steelhead Downstream Dam Passage Survival Under Alternative 3b. *Downstream dam passage survival at Green Peter for juvenile winter steelhead 2 year olds under Alternative 3b. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

STEELHEAD

2.11 STEELHEAD ALTERNATIVE 4

2.11.1 North Santiam – Detroit

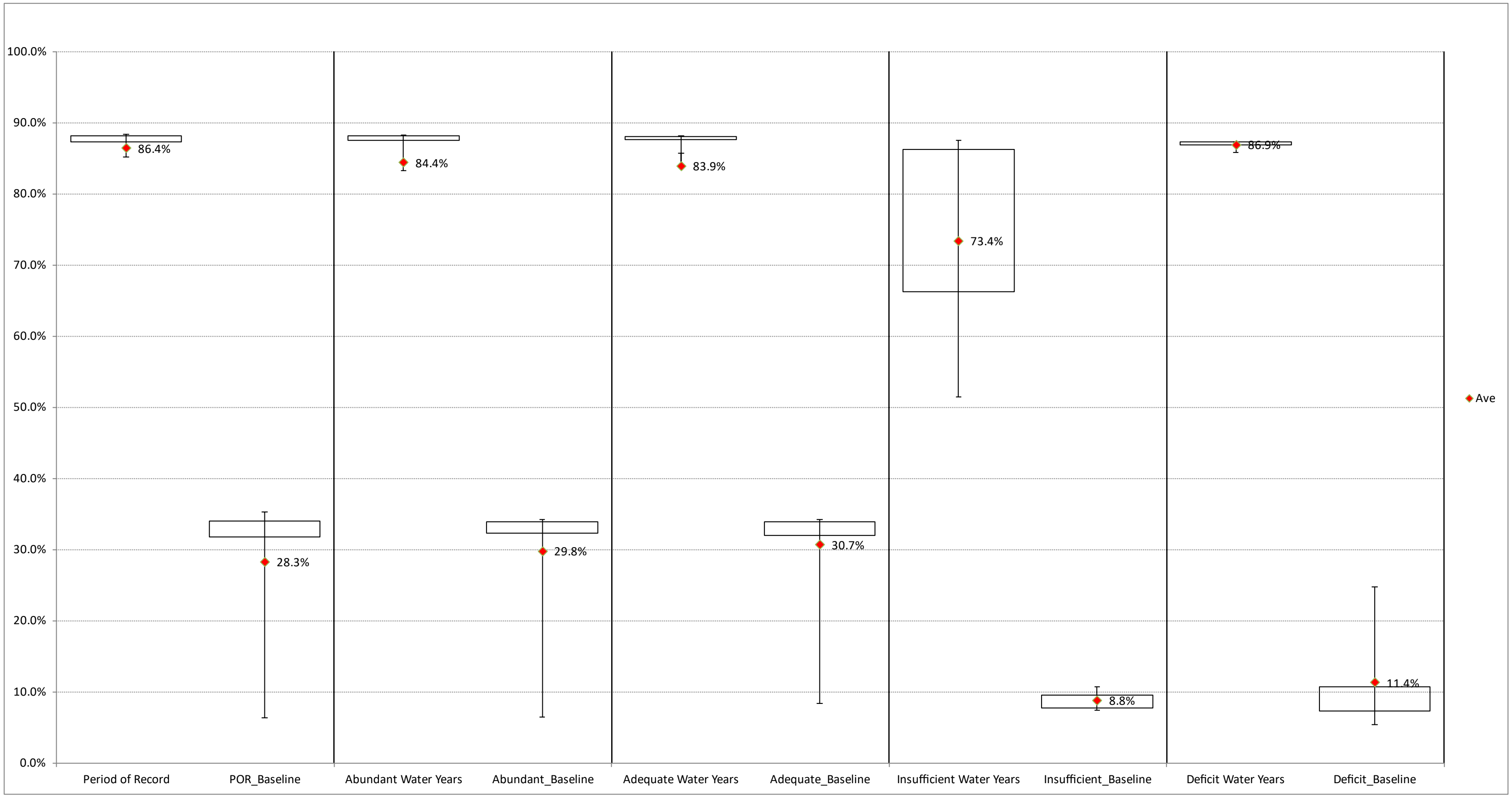

Figure 2-145. Detroit Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Detroit for juvenile winter steelhead sub-yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

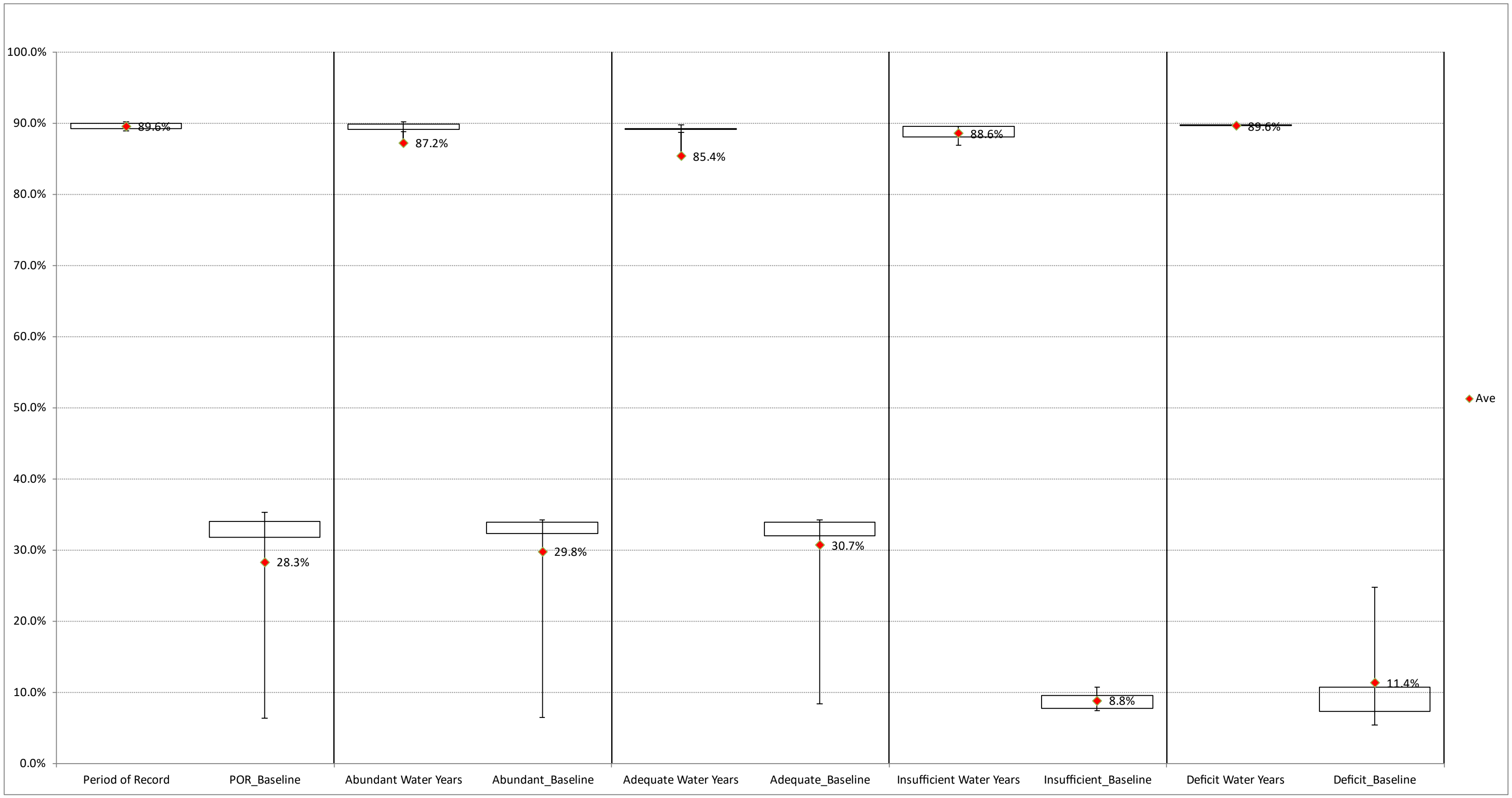

Figure 2-146. Detroit Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Detroit for juvenile winter steelhead yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

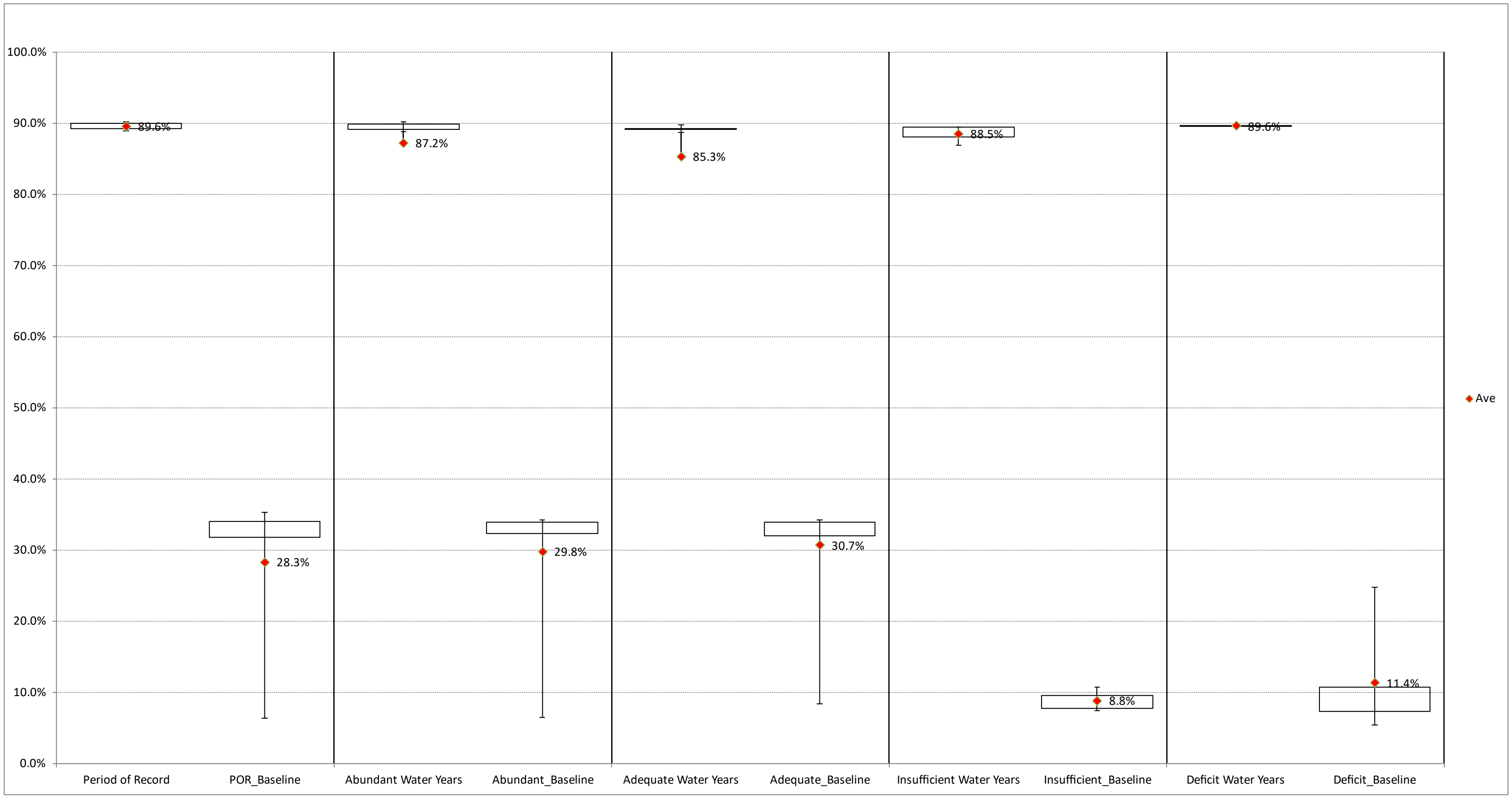

Figure 2-147. Detroit Juvenile Winter Steelhead 2-Year-Old Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Detroit for juvenile winter steelhead 2 year olds under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

South Santiam – Foster

Figure 2-148. Foster Juvenile Winter Steelhead Sub-Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Foster for juvenile winter steelhead sub-yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

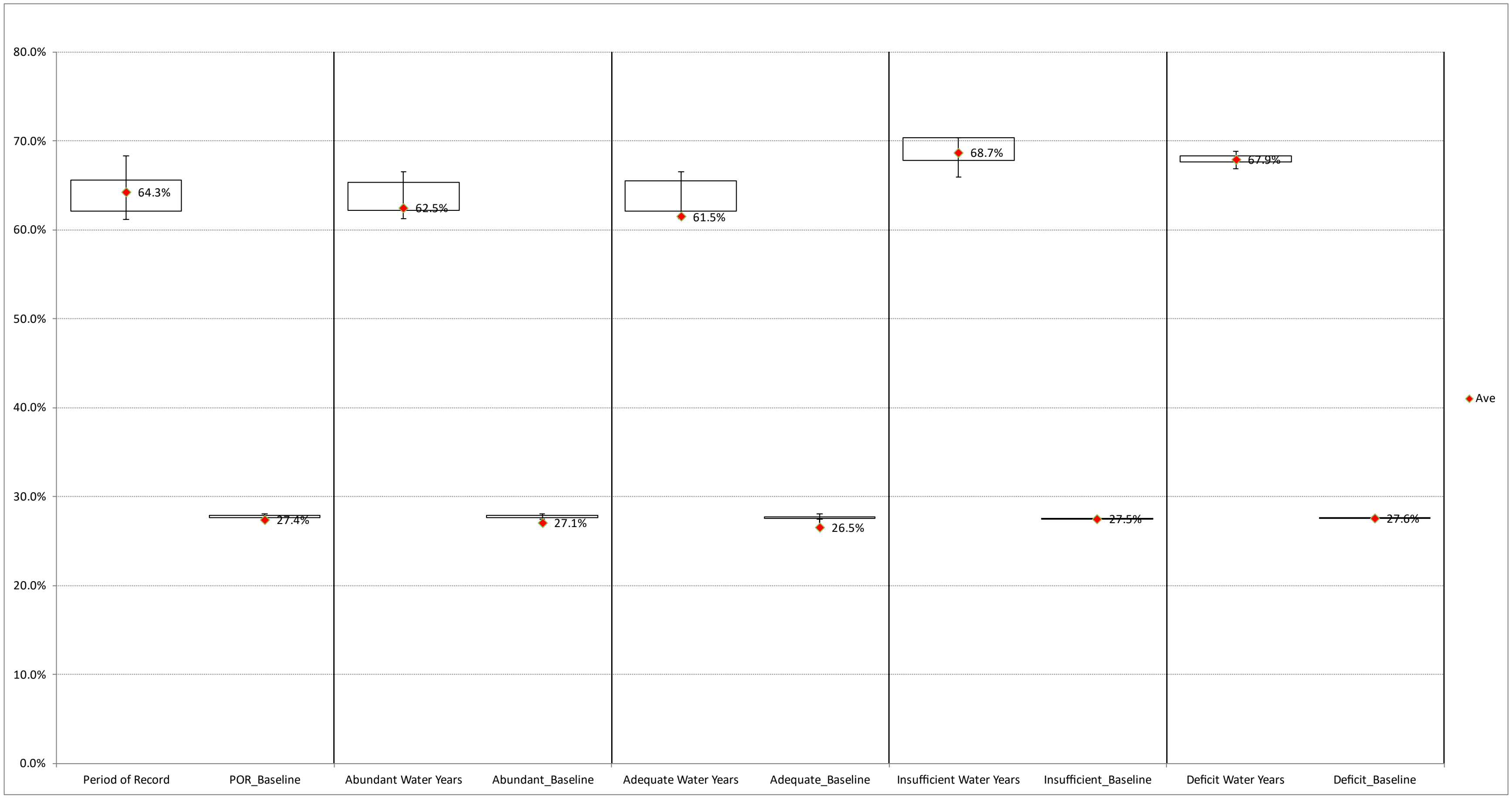

Figure 2-149. Foster Juvenile Winter Steelhead Yearling Downstream Dam Passage Survival Under Alternative 4. *Downstream dam passage survival at Foster for juvenile winter steelhead yearlings under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

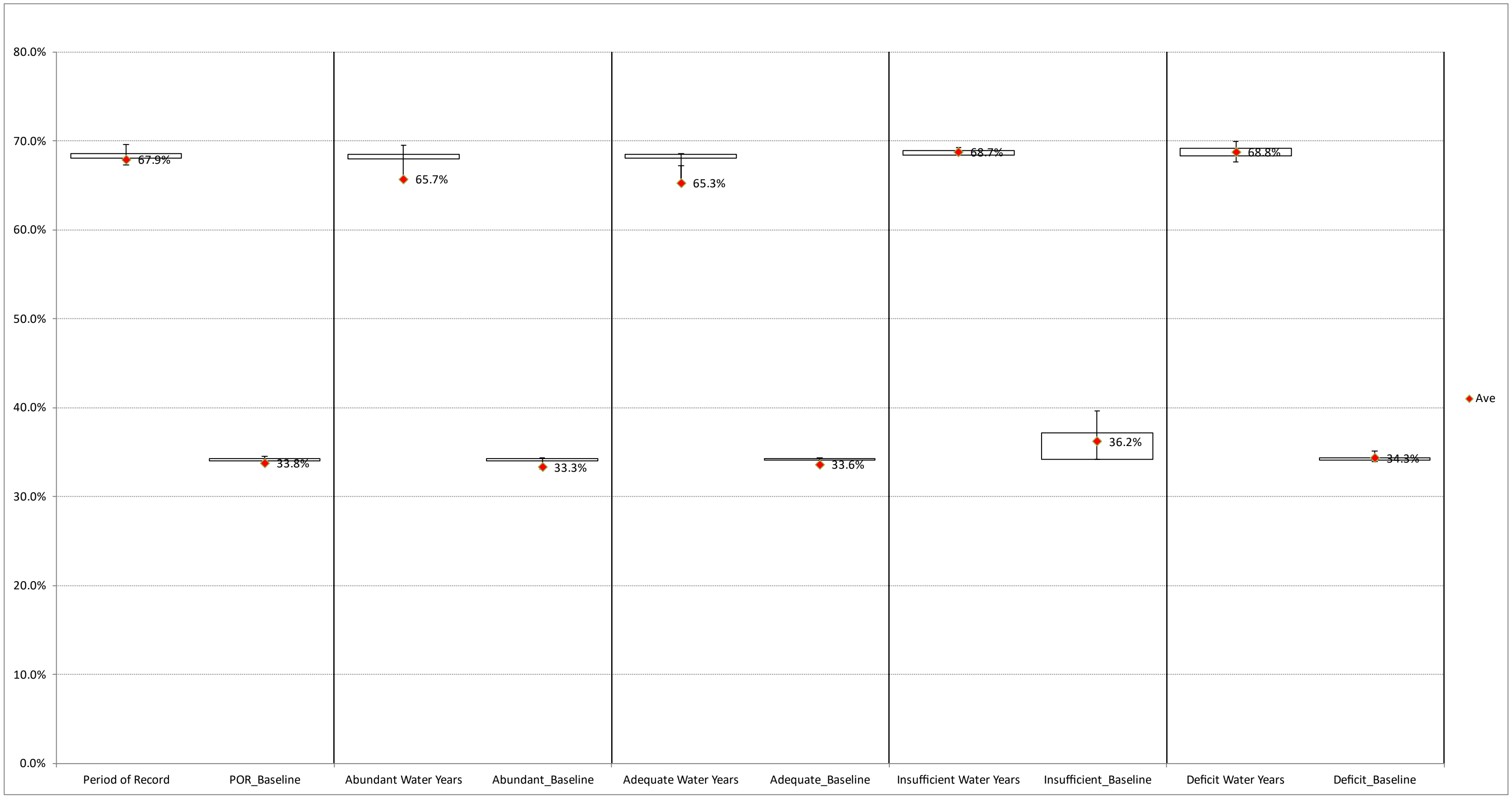

Figure 2-150. Foster for 2-year-old juvenile winter steelhead Downstream dam passage survival under Alternative 4. *Downstream dam passage survival at Foster for juvenile winter steelhead 2 year olds under Alternative 4. The mean is given by the point estimate (filled dot). Survival probabilities are given for the period of record (far left), compared to hydrologic year types denoted in each panel.*

2.11.2 South Santiam – Green Peter

(Same as Alternative 2a)

CHAPTER 3 - WVS EIS BULL TROUT ASSESSMENT

3.1 INTRODUCTION

Among dams included in the WVS, bull trout populations currently exist above Cougar and Hills Creek dams. These populations are stable or increasing (Zymonas et al. 2021). The U.S. Fish and Wildlife Service (USFWS) also plans to reintroduce bull trout above Detroit Dam (C. Allen, pers. comm. insert date 2021). For purposes of the WVS EIS, we assumed bull trout have been reintroduced above Detroit Dam, given the 30-year time horizon of the EIS effects analysis.

To assess the effects of the WVS EIS alternatives on bull trout, a habitat assessment framework was developed following the principles and approaches applied by Schaller et al. (2014), with additional considerations of reservoir and fish passage conditions at large dams, and limiting factors documented in the Oregon Bull Trout Recovery Strategy (USFWS and others insert year).

Schaller et al. (2014) surveyed biologists with knowledge of bull trout to identify and weight variables affecting aquatic habitat conditions for bull trout. Scores were defined for assessing each of the variables for different lifestage needs of bull trout, and then applied with the weighting factors to assess habitat conditions in river reaches of interest.

The highest weighted variables identified by Schaller et al. (2014) were surface flow, water temperature and passage impediments (see Table 3.17 in Schaller et al. 2014), indicating these were considered the most important variables by the biologists surveyed. Other viable weightings were much smaller, indicating they would have much less of an influence when comparing effects among alternatives in an assessment. We therefore focused the habitat assessment for the WVS EIS on surface flow, water temperature and passage conditions.

For purposes of the WVS EIS bull trout assessment, habitat reaches were defined consistent with those recently applied by ICF (2022) when modeling habitat conditions using the Ecosystem Diagnostic and Treatment (EDT) model. This allowed for the application of information on habitat conditions for variables of interest already summarized by ICF to be used.

We assumed all bull trout would utilize reservoirs that are located downstream of each un-impounded river reach being assessed. This is based on Zymonas et al. (2021) reporting that most bull trout populations in the Willamette Basin are adfluvial.

Additional variables not explicitly considered by Schaller et al. (2014) which are important considerations when assessing reservoir use by bull trout are predation and fisheries. Both predation and harvest are included as primary threats to recovery of bull trout in the Upper Willamette. Reservoirs of the WVS include piscivorous fishes known to prey on salmonids, including pike minnow, walleye and smallmouth bass. Predation risk was scored based on the piscivorous fish species present in each reservoir. Local sport fisheries increase the risk of stress, injury, and mortality. Evidence of injury from hook and line capture of bull trout has been reported for bull trout in Hills Creek and South Fork McKenzie (ODFW 2021; Zymonas et al. 2021). Since the USACE of Engineers does not have any authority to change sport fishing regulations, we assumed current fisheries regulations and level of fishing effort (pressure) would continue under each WVS EIS alternatives. Predation risk and fisheries variable scores were used to decrement the value of the habitat scores.

Compared to Schaller et al. 2014 we modified how passage impediments were considered to better account for conditions found at large dams currently and under each WVS EIS. We characterized passage at dams for bull trout as either not available, partially available, or fully available. Under the no passage available category, we assumed poor downstream passage conditions and no upstream passage are provided. For passage to be fully available, we assumed both effective up and downstream passage is present. Other conditions were assumed to fall in the partially available passage category. For the WVS EIS, operational downstream passage with upstream passage would be in this category. Passage condition categories were scored and used as an adjustment factor for scoring habitat conditions available below WVS dams where bull trout currently reside upstream (Cougar Dam and Hills Creek) or where they are being considered for reintroduction (Detroit Dam).

Bull trout habitat score = [above principal dam hydrology score + temperature score * reach length * predation risk factor * fisheries risk factor] + [below principal dam hydrology score + temperature score * reach length * predation risk factor * fisheries risk factor * passage condition factor]

3.1.1 Exposure to Limiting Factors and Risks Under Different Dam Passage Conditions

In order for access to additional habitat to be beneficial it must lead to increases in abundance, productivity (adult recruitment), and diversity. Expanded distribution could also reduce risks from catastrophic events (e.g. large wild fires or landslides), if spawning can still occur, and the expansion in distribution does not reduce productivity or spawner abundance over time for the primary population.

Bull trout collected at the Cougar adult fish facility documents that some individuals will move downstream of the dam and some return and are effectively collected and moved back upstream (e.g., Zymonas et al. 2021). Most of those returning are mature adults, based on their size. However, data is lacking on the growth and survival for bull trout that move below WVS dams, and it is not possible to determine if the rate of mortality for individuals moving below principal dams is greater than the rate of recruitment or spawners returning and spawning in the principal population. Benefits of providing passage and access to habitat below WVS dams could include access to additional rearing/maintenance habitat or spawning habitat, access to other spawning populations, increase in distribution reducing risks from catastrophic events (e.g., wildfires, large landslides). However, there are also many risks for bull trout that move downstream which act to diminish the potential benefits of accessing additional habitat below dams. These include injury or mortality from passage at large dams or diversion dams, the inability to move back upstream of dams lacking passage facilities, exposure to poor habitat conditions (e.g., higher water temperatures), injury or mortality from predators or angling. We also did not find any evidence of satellite populations having resulted from bull trout volitionally moving below principal dams, however this may be due to limited survey effort.

The Oregon Bull Trout Recovery Strategy prepared by USFWS and others lists the following statewide limiting factors, and those specifically identified for bull trout in the Upper Willamette. Exposure to all these known limiting factors would be expected to increase with access below dams for bull trout below Cougar, Hills Creek and Detroit/Big Cliff dams.

Table 3-1. Limiting factors identified in the Oregon Bull Trout Recovery Strategy

Statewide Limiting Factors	Upper Willamette Limiting Factors
Temperature Flow Barriers Human development	Altered flow and geomorphic processes Entrainment and fish passage Illegal harvest Prey base Hybridization and competition Predation

Note: Data provided by USFWS and Others, 2021.

To assess the value of improving access to habitat downstream of primary dams, effects on population abundance, productivity, diversity and distribution should be considered. It is not known if the current rate of downstream emigration is equal to or greater than the return rate for the Cougar and Hills Creek populations. In the Deschutes River, where cool water temperatures are maintained by significant ground water inputs, return rates of bull trout passing downstream of Round Butte Dam have been high (insert pers comm). However, higher water temperatures and multiple other limiting factors exist below WVS dams, as referenced above from the Oregon Bull Trout Recovery Strategy. If emigrate rates are greater than return rates above dams, then the existing populations will decline unless satellite spawning areas are established downstream. When reviewing Zymonas et al. 2021, there is not any evidence effecting spawning below WVS dams is occurring. Moreover, very few locations existing where spawning below WVS dams could potentially occur, and these would be expected to be negatively impacted by climate change (insert ref). Habitat quality below dams would be assumed to further degrade over the 30 year time period of the WVS EIS, due to predicted climate change effects on precipitation and air temperatures leading to changes in hydrology, water temperatures, fire, competition with warmwater and exotic fishes, landuse and development, among other factors.

Lacking emigration and upstream return rates of bull trout at WVS dams, we assume that risks of mortality are high for emigrants passing below dams due to the numerous limiting factors present, prediction in further habitat degradation, and that there would not be spawning below dams. Since existing bull trout populations above Cougar and Hills Creek dams, which are currently stable or increasing, rely on reservoirs for rearing and foraging, we also considered the extent that reservoir conditions would change in each alternative. A fish passage measure which results in a reservoir pool which is largely drained would be expected to significantly affect rearing and forage opportunity. Passage measures which maintain a reservoir year-round were assumed not to significantly affect rearing and forage opportunity.

Based on the above, we categorized risks for bull trout populations residing above Detroit and Hills Creek dams as high for those providing increased access to habitat below dams (improved passage at dams). For Cougar, due to the maintenance of cooler water below Cougar Dam and the Upper McKenzie watershed, we scored the risk level for WVS EIS alternatives with improved dam passage as moderate if the reservoir is maintained, and high if the reservoir is significantly reduced.

For alternatives where fish passage is not changed from existing conditions, we categorized the risks as low. This is primarily based on available information showing existing populations of bull trout above Cougar and Hills Creek as stable or increasing, and the assumption that habitat conditions will degrade and known limiting factors will be exacerbated below dams with climate change.

3.2 ASSESSMENT RESULTS

Table 3-2. Reach Scores

Reach	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4
HCR habitat score	10.06	10.03	10.11	10.11	19.41	20.39
Population Risk	Low	Low	Low	High	High	High
Risk factors	1,2,3	1,2,3	1,2,3	2,4	2,4	2,4
CGR habitat score	18.96	19.00	26.88	26.90	22.78	23.02
Population Risk	Moderate	Moderate	High	Moderate	High	Moderate
Risk factors*	4,5	4,5	4,5	4,5	4,5	4,5
DET habitat score	61.31	79.40	81.54	80.94	71.23	69.89
Population Risk	High	High	High	High	High	High
Risk factors	2,4	2,4	2,4	2,4	2,4	2,4

Notes: *Risk Factors:

1. Low survival of emigrants but low rate of emigration, climate change will further reduce survival
2. No spawning populations downstream, consider translocation for potential genetic exchange
3. Limited distribution of population infers risk from catastrophic events (e.g. wildfire; landslide)
4. Moderate to high level of emigration with low survival downstream, climate change will further reduce
5. Spawning populations downstream within sub-basin allowing for potential genetic exchange

Table 3-3. Percent change in scores from NAA

Reach	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4
HCR habitat score	0%	0%	0%	93%	103%	134%
Population Risk	Low	Low	Low	High	High	High
Risk factors*	1,2,3	1,2,3	1,2,3	2,4	2,4	2,4
CGR habitat score	0%	42%	42%	20%	21%	38%
Population Risk	Moderate	Moderate	High	Moderate	High	Moderate
Risk factors	4,5	4,5	4,5	4,5	4,5	4,5
DET habitat score	30%	33%	32%	16%	14%	30%
Population Risk	High	High	High	High	High	High
Risk factors	2,4	2,4	2,4	2,4	2,4	2,4

Notes: *Risk Factors:

1. Low survival of emigrants but low rate of emigration, climate change will further reduce survival
2. No spawning populations downstream, consider translocation for potential genetic exchange
3. Limited distribution of population infers risk from catastrophic events (e.g. wildfire; landslide)
4. Moderate to high level of emigration with low survival downstream, climate change will further reduce
5. Spawning populations downstream within sub-basin allowing for potential genetic exchange

Table 3-4. Reach Scores by Alternative and Sub-Basin

Reach	Extent	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Middle Fork	Above HCR	10.06	10.03	10.11	10.11	9.30	10.09	10.03
Middle Fork	Below HCR	0.00	0.00	0.00	0.00	10.12	10.30	13.48
All Middle Fork	Risk*	Low	Low	Low	Low	High	High	High
McKenzie	Above CGR	11.83	11.86	12.15	12.16	11.95	12.07	11.86
McKenzie	Below CGR	7.13	7.14	14.73	14.74	10.83	10.95	14.27
All McKenzie	Risk*	Moderate	Moderate	Moderate	High	Moderate	High	Moderate
North Santiam	Above Detroit	41.52	41.32	42.47	42.47	42.79	41.49	41.69
North Santiam	Below Detroit	19.79	38.08	39.07	38.48	28.44	28.40	37.79
All North Santiam	Risk*	High	High	High	High	High	High	High

Note: Risk captures the entire reach, both above and below the focus location.

1. Low survival of emigrants but low rate of emigration, climate change will further reduce survival
2. No spawning populations downstream, consider translocation for potential genetic exchange
3. Limited distribution of population infers risk from catastrophic events (e.g. wildfire; landslide)
4. Moderate to high level of emigration with low survival downstream, climate change will further reduce
5. Spawning populations downstream within sub-basin allowing for potential genetic exchange

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Table 3-5. Reach definitions

Reach	Reach length*
Hills Creek above Hills Creek Reservoir	16.573
Hills Creek below Hills Creek Dam	24.457
North Fork Middle Fork River	27.011
North Santiam and Breitenbush rivers above Detroit Reservoir	33.185
North Santiam River below Big Cliff Dam	50.31

Note: * Reach Length from EDT; ICF (2022)

Flow and temperature scores adapted from EDT (ICF 2022) for river reaches above and below reservoirs where bull trout reside or are proposed for reintroduction. EDT rankings, in general, occur on a scale of 0 to 4 with 0 being the best and 4 being the worst. To adapt the rankings to the appropriate scale for this analysis, we took the inverse of the rankings, lower scores being worse than high scores.

Table 3-6. Intra-annual Low Flow and Temperature by Alternative. *

2015 Alternative	Extent	Intra-annual low flow	Temperature
NAA 2015	Above HCR	0.836625	0.7825
NAA 2015	Below HCR	0.77725	0.685
NAA 2015	Above CGR	0.81	0.9425
NAA 2015	Below CGR	0.81875	0.9
NAA 2015	Above Detroit	0.816	0.83475
NAA 2015	Below Detroit	0.8175	0.835
Alt1 2015	Above HCR	0.83775	0.776
Alt1 2015	Below HCR	0.78125	0.69925
Alt1 2015	Above CGR	0.81125	0.94475
Alt1 2015	Below CGR	0.819	0.903
Alt1 2015	Above Detroit	0.81075	0.83175
Alt1 2015	Below Detroit	0.819	0.771
Alt2a 2015	Above HCR	0.85025	0.776
Alt2a 2015	Below HCR	0.8475	0.6915
Alt2a 2015	Above CGR	0.8525	0.9465
Alt2a 2015	Below CGR	0.8725	0.903
Alt2a 2015	Above Detroit	0.85675	0.83175
Alt2a 2015	Below Detroit	0.86225	0.769
Alt2b 2015	Above HCR	0.85025	0.776
Alt2b 2015	Below HCR	0.8475	0.6885

2015 Alternative	Extent	Intra-annual low flow	Temperature
Alt2b 2015	Above CGR	0.8525	0.94775
Alt2b 2015	Below CGR	0.8725	0.90475
Alt2b 2015	Above Detroit	0.8565	0.83175
Alt2b 2015	Below Detroit	0.85975	0.74675
Alt 3a 2015	Above HCR	0.7955	0.70025
Alt 3a 2015	Below HCR	0.78375	0.68725
Alt 3a 2015	Above CGR	0.82625	0.9435
Alt 3a 2015	Below CGR	0.838	0.903
Alt 3a 2015	Above Detroit	0.86925	0.83175
Alt 3a 2015	Below Detroit	0.859	0.7245
Alt3b 2015	Above HCR	0.847	0.776
Alt3b 2015	Below HCR	0.82175	0.676
Alt3b 2015	Above CGR	0.839	0.9485
Alt3b 2015	Below CGR	0.8545	0.9055
Alt3b 2015	Above Detroit	0.8175	0.83175
Alt3b 2015	Below Detroit	0.8255	0.7555
Alt4 2015	Above HCR	0.8375	0.77575
Alt4 2015	Below HCR	0.78075	0.6885
Alt4 2015	Above CGR	0.80975	0.946
Alt4 2015	Below CGR	0.81725	0.903
Alt4 2015	Above Detroit	0.82575	0.83175
Alt4 2015	Below Detroit	0.83125	0.74675

Notes: Hills Creek (HCR); Cougar (CGR); Detroit (DET).

Table 3-7 shows below dam habitat availability assumptions and habitat adjustment factors for bull trout populations above WVS dams based on passage conditions included in each WVS EIS alternative. Abbreviations: d/s = downstream; AFF = adult fish collection facility.

Table 3-7. Detroit Passage and Downstream Habitat Availability, and Passage Adjustment

Alternative	passage up/down	d/s habitat availability	Passage adjustment factor
NAA	AFF/limited operational	Partial availability	0.50
1	AFF/floating structure	Full availability	1
2a	AFF/floating structure	Full availability	1
2b	AFF/floating structure	Full availability	1
3a	AFF/operational	Partial availability	0.75
3b	AFF/operational	Partial availability	0.75
4	AFF/floating structure	Full availability	1

Notes: Abbreviations: d/s = downstream; AFF = adult fish collection facility.
Hills Creek (HCR); Cougar (CGR); Detroit (DET).

Table 3-8. Cougar Passage and Downstream Habitat Availability, and Passage Adjustment

Alternative	passage up/down	d/s habitat availability	Passage adjustment factor
NAA	AFF/limited operational	Partial availability	0.5
1	AFF/limited operational	Partial availability	0.5
2a	AFF/floating structure	Full availability	1.0
2b	AFF/operational	Partial availability	0.75
3a	AFF/operational	Partial availability	0.75
3b	AFF/operational	Partial availability	0.75
4	AFF/floating structure	Full availability	1.0

Table 3-9. Hill's Creek Passage and Downstream Habitat Availability, and Passage Adjustment

Alternative	passage up/down	d/s habitat availability	Passage adjustment factor
NAA	None/Existing	Not available	0
1	None/Existing	Not available	0
2a	None/Existing	Not available	0
2b	None/Existing	Not available	0
3a	AFF/operational	Partial availability	0.75
3b	AFF/operational	Partial availability	0.75
4	AFF/floating structure	Full availability	1

Table 3-10. General Predation Risk Adjustment Factor

Adjustment factor	Description
1	Little to no predation
0.75	Moderate level of predation
0.5	High level of predation

Table 3-11. Predation risk adjustment factor scores for WVS reservoirs.

Reservoir	Adjustment Factor	Basis for score
Lookout Point	0.5	Large established populations of northern pikeminnow, walleye, largemouth bass, crappie (Brandt et al. 2016; Monzyk et al. 2014; Monzyk et al. 2013)
Detroit/Big Cliff	1	Potentially piscivorous fish species present: rainbow trout, cutthroat trout, brown bullhead, and sculpin; rainbow trout dominate, but evidence of fish in diet low (Monzyk et al. 2012). A low risk of predation was concluded; however, this may be an underestimate.

Table 3-12. Fisheries Adjustment Factor (Presence of Fisheries, Target Sport Species)

Adjustment factor	Description
1	Little to no fishing present = little to no risk of injury or mortality
0.75	Moderate level of fishing= moderate risk of injury or mortality
0.5	High level of fishing= high risk of injury or mortality

Table 3-13. Fisheries Risk Adjustment Factor Scores for WVS Reservoirs.

Reservoir	Adjustment Factor	Basis for score
Lookout Point	0.75	Moderate level of fishing assumed based on online review of comments at the following websites:
Detroit/Big Cliff	0.5	High levels of fishing targeting stocked trout in Detroit Reservoir

	1		2a		2b		3a		3b		4	
	Up	**Down**	**Up**	**Down**	**Up**	**Down**	**Up**	**Down**	**Up**	**Down**	**Up**	**Down**
DEX								spring spill		spring spill		
LOP		floating structure		floating structure		floating structure		spring drawdown fall drawdown		spring spill fall drawdown		floating structure
HCR							AFF lamprey passage	spring spill fall drawdown	AFF lamprey passage	spring drawdown fall drawdown	AFF lamprey passage	floating structure
FCR								add spring spill				
CGR	add lamprey passage		add lamprey passage	floating structure	add lamprey passage	spring drawdown fall drawdown (DT)	add lamprey passage	spring drawdown fall drawdown (RO)	add lamprey passage	spring drawdown fall drawdown (DT)	add lamprey passage	floating structure
BLU							AFF lamprey passage	fall drawdown	AFF lamprey passage	fall drawdown		
FOS		modified fish weir		modified fish weir		modified fish weir						modified fish weir
GPR	AFF	floating structure	AFF lamprey passage	spring spill fall drawdown	AFF lamprey passage	spring spill fall drawdown	AFF lamprey passage	spring spill fall drawdown	AFF lamprey passage	spring drawdown fall drawdown		
BCL		fish collected at DET		fish collected at DET		fish collected at DET		spring spill		spring spill		fish collected at DET
DET		floating structure		floating structure		floating structure		spring drawdown fall drawdown		spring spill fall drawdown		floating structure

Figure 3-1. Fish passage measures included in the WVS EIS, by dam and alternative. *Blank cells equal NAA fish passage conditions.*

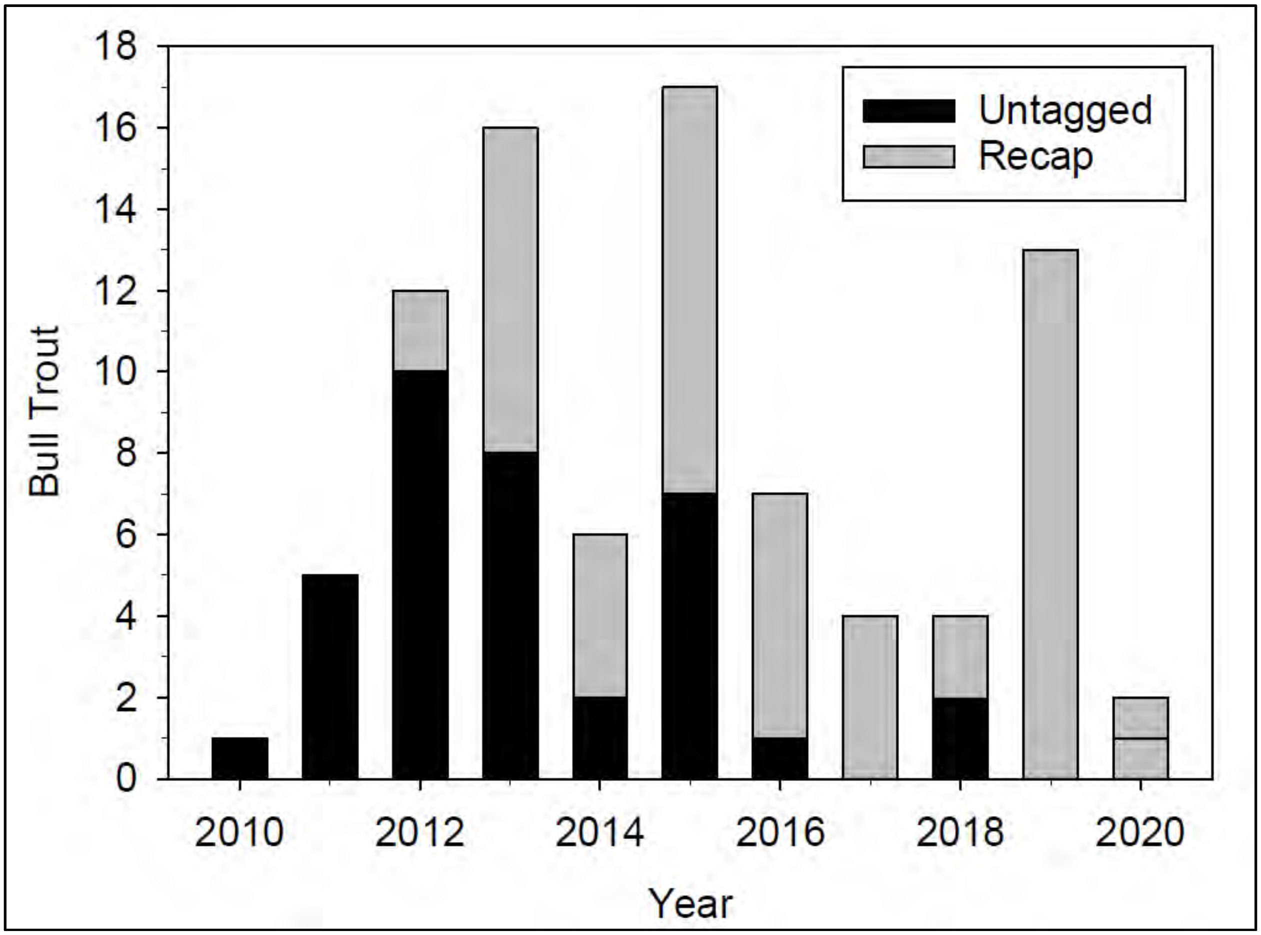


Figure 3-2. Bull Trout collected annually at the Cougar Dam upstream fish passage facility. *Number of Bull Trout collected each year at the Cougar Dam upstream fish passage facility, including fish previously PIT-tagged ("Recap") and fish without a PIT tag when captured ("Untagged). Figure and figure caption reproduced from Zymonas et al. (2021), Figure 1.3.*

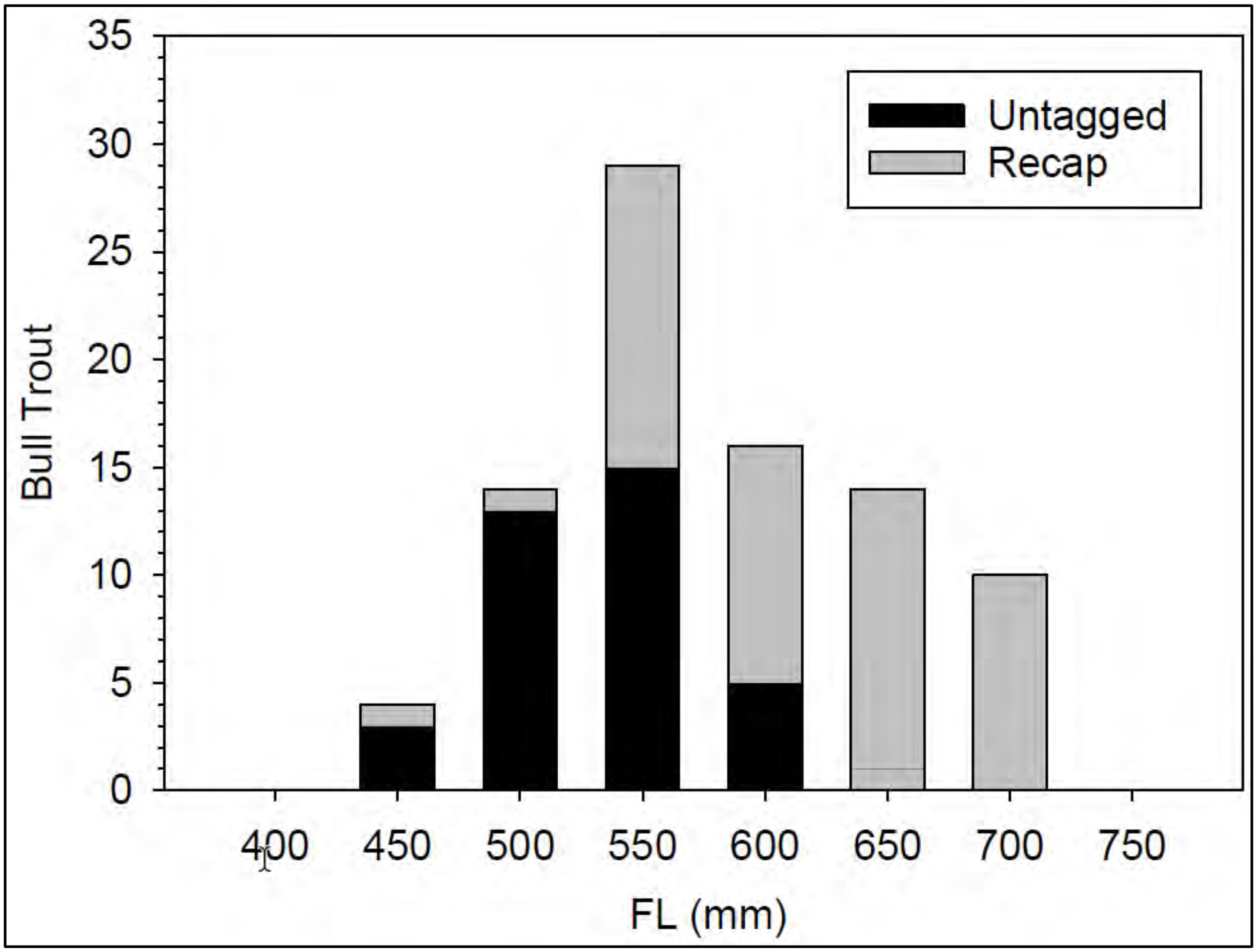


Figure 3-3. Cougar Dam Upstream Fish Passage Bull Trout Length Frequency. *Length frequency distribution for all Bull Trout (N = 87) collected at the Cougar Dam upstream fish passage Length frequency distribution for all Bull Trout (N = 87) collected at the Cougar Dam upstream fish passage facility, including fish previously PIT-tagged ("Recap") and fish without a PIT tag when captured ("Untagged"). Figure and caption information reproduced from Zymonas et al. (2021), Figure 1.4.*

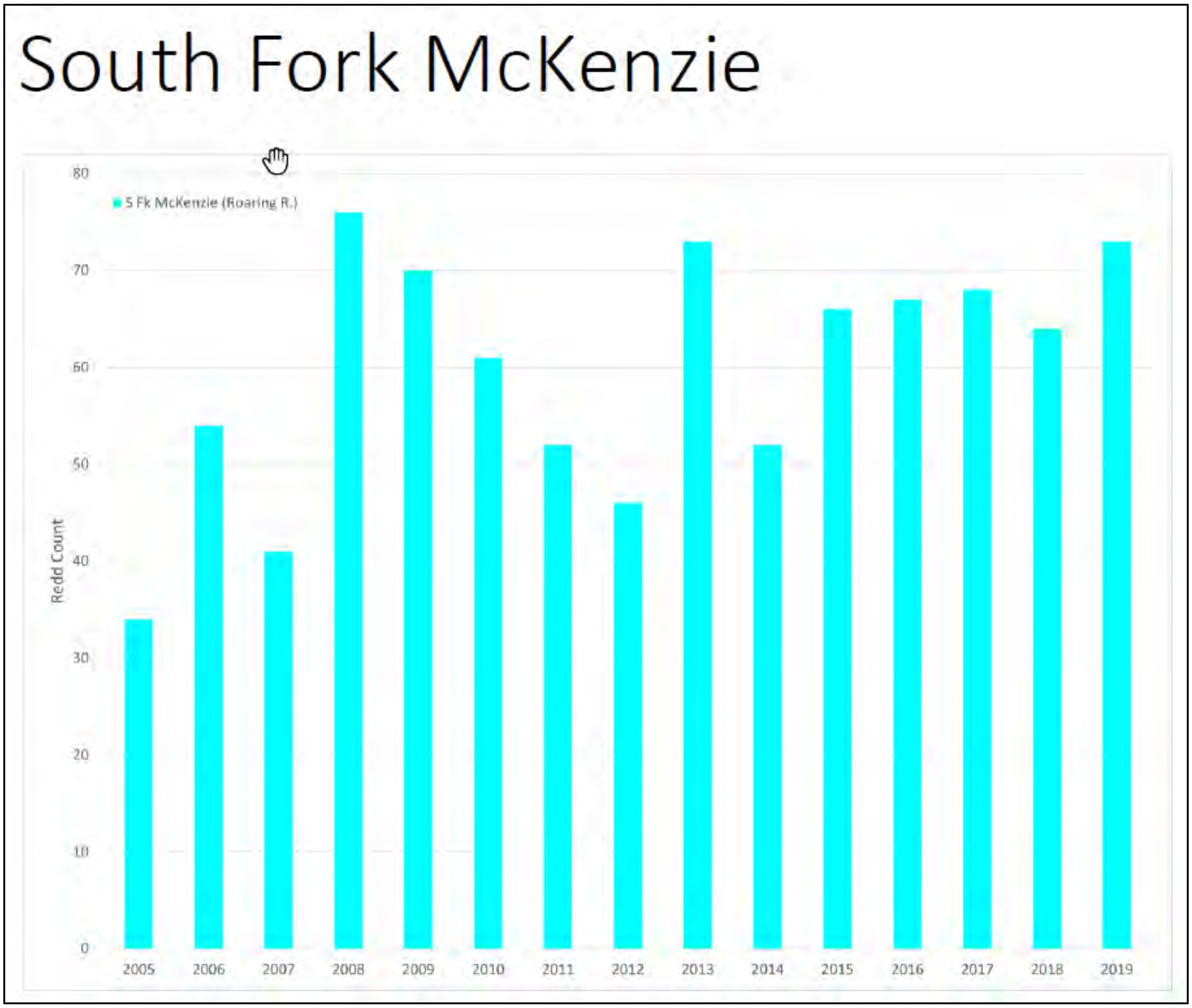


Figure 3-4. Annual redd counts for bull trout in the Roaring River. *Annual redd counts for bull trout in the Roaring River, a tributary of the South Fork McKenzie above Cougar Dam. Figure copied from Harrison and Zymonas 2021.*

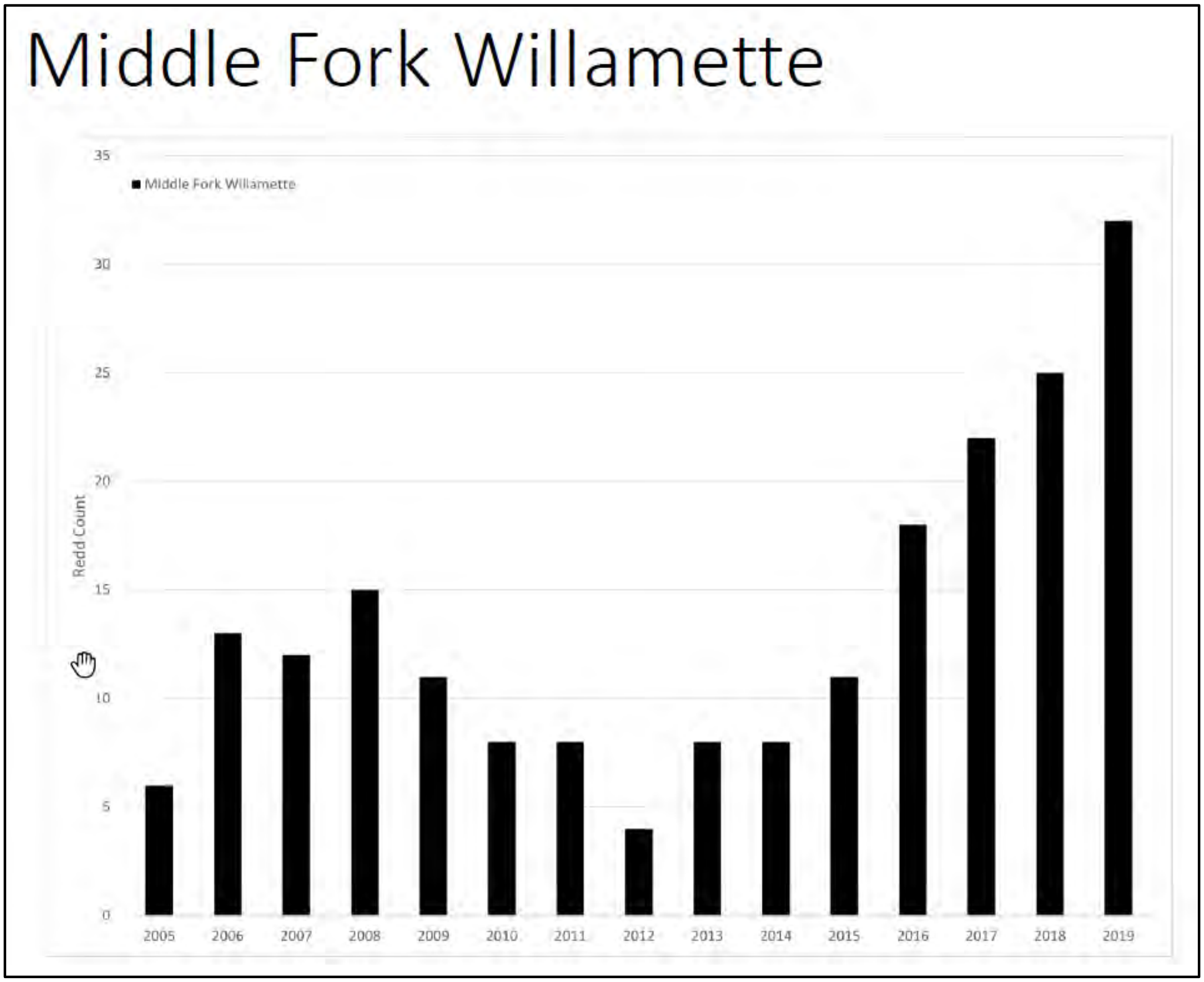


Figure 3-5. Annual Middle Fork Willamette Basin Bull Trout Redd Counts. *Annual redd counts for bull trout in the Middle Fork Willamette Basin, above Hills Creek Dam. Figure copied from Harrison and Zymonas 2021.*

Table 3.1.Description of surface flow categories and criteria for rating the quality of surface flow within reaches of the South Fork and Mainstem Walla Walla rivers as well as Mill and Yellowhawk creeks.

Surface Flow Categories	Numeric Rating	Percentage Rating Criteria	Explanation
High Quality	5	> 80.0%	Near normative, less-diverted streamflows inferring high quantity and quality of habitat for bull trout at all life stages.
Good Quality	4	> 60.0 – 80.0%	Partially-diverted streamflows inferring good quantity and quality of habitat for bull trout at all life stages and for all strategies and actions in terms of surface flow.
Fair Quality	3	> 40.0 – 60.0%	Moderately-diverted streamflows inferring a fair quantity and quality of habitat for bull trout at all life stages and for all strategies and actions in terms of surface flow.
Low Quality	2	> 20.0 – 40.0%	Very depleted streamflows inferring low quantity and quality of habitat for bull trout at all life stages and for all strategies and actions in terms of surface flow.
Poor Quality	1	≤ 20.0 %	Severely depleted streamflows inferring a poor quantity and quality of habitat for bull trout at all life stages and for all strategies and actions in terms of surface flow.

Figure 3-6. Habitat rating criteria from Schaller et al. (2014)

Table 3.3. Description of temperature categories and associated numeric rating.

Temperature Categories	Numeric Rank	Explanation
High Quality	5	Optimal temperature conditions for bull trout at a given life stage to exhibit a certain strategy or conduct a particular action and thrive.
Good Quality	4	Temperature conditions, albeit not optimal, that allow bull trout at a given life stage to exhibit a certain strategy or conduct a particular action and thrive.
Fair Quality	3	Tolerable temperature conditions that likely allow bull trout at a given life stage to exhibit a certain strategy or conduct a particular action and persist.
Low Quality	2	Temperature conditions, albeit tolerable, likely only marginally allow bull trout at a given life stage to exhibit a certain strategy or conduct a particular action and persist.
Poor Quality	1	Temperature conditions that severely limit the ability of a bull trout at a given life stage to exhibit a certain strategy or conduct a particular action and may be lethal or may inhibit the long-term persistence of the population.

Table 3.4. Criteria for rating the quality of temperature within reaches of the South Fork Walla Walla River and Mainstem Walla Walla River as well as Mill and Yellowhawk creeks.

Activity/Process/Action	Poor Quality (1)	Low Quality (2)	Fair Quality (3)	Good Quality (4)	High Quality (5)
Adult Spawning	>16°C*	>10 - 16°C*	>7 - 10°C*	≤5°C*	>5 - 7°C*
Juvenile Rearing, Foraging and Growth	>22°C	>18 -22°C and ≤1°C	1 – 6°C and >16 – 18°C	>6 - 10°C and >12 - 16°C	>10 - 12°C
Fluvial Adult Upstream Migration	>26°C and ≤5°C	>20 - 26°C and >5 - 11°C	>16 to 20°C	>11 to 14°C	>14 - 16°C
Adult Foraging and Maintenance	>26°C	>20 -26°C and ≤1°C	1 – 6°C and >18 – 20°C	>6 - 14°C and >16 - 18°C	>14 - 16°C
Fluvial Adult Downstream Migration	>26°C	>20 -26°C and ≤1°C	>16 -20°C	>1 - 6°C and >10 - 16°C	>6 – 10°C
Fluvial Sub-adult Downstream Migration	>26°C	>20 -26°C and ≤1°C	>16 -20°C	>1 - 6°C and >10 - 16°C	>6 – 10°C
Fluvial Sub-adult Lower River Evacuation	>26°C	>20 -26°	>18 - 20°C	>16 - 18°C	< 16°C
Fluvial Sub-adult Rearing, Foraging and Growth	>26°C	>20 -26°C and ≤6°C	6 - 10°C and >18 - 20°C	>10 -14°C and >16 - 18°C	>14 - 16°C

Figure 3-7. Habitat rating criteria from Schaller et al. (2014) (cont.)

CHAPTER 4 - ASSESSMENT OF CLIMATE CHANGE EFFECTS ON FISH FROM WVS

4.1 INTRODUCTION

Crozier et al. (2019) conducted a comprehensive climate vulnerability assessment for Pacific salmon and steelhead (Oncorhynchus spp.) for distinct population segments (DPSs) in the U.S. They followed the climate vulnerability assessment method developed by Hare et al. (2016), which is now being implemented for U.S. marine and anadromous species by NOAA

Fisheries (Link et al. 2015). The 2019 assessment was based on three components of vulnerability: 1) biological sensitivity, which is a function of individual species characteristics; 2) climate exposure, which is a function of geographical location and projected future climate conditions; and 3) adaptive capacity, which describes the ability of a DPS to adapt to rapidly changing environmental conditions.

Crozier et al. found that in general, DPSs with the highest sensitivity and exposure and lowest adaptive capacity were the most vulnerable to climate change. For spring Chinook DPSs assessed, their findings suggest a potential range contraction toward the coast for anadromous life histories unless access to higher-elevation habitats is restored and habitat quality in rearing areas and migration corridors is improved (Herbold et al. 2018). Steelhead DPSs considered tended to score lower in sensitivity than Chinook in the same region and were found to have an intermediate vulnerability between high and moderate.

Upper Willamette River spring Chinook (UWR Chinook) endure a temperature-stressed adult migration and summer holding period and were specifically found to be highly vulnerable to temperature increases due to long adult migrations in spring and summer through highly modified rivers, along with exposure to high summer stream temperatures during the holding period prior to spawning. Under existing fish passage conditions at dams in the Willamette, this DPS was found to have a very high overall vulnerability, very high biological sensitivity, high climate exposure and a moderate adaptive capacity. Access to high elevation habitat to reduce effects of climate change has also been found important by others (Myers et al. 2018; Fitzgerald et al. 2021). Overall, Myers et al. 2018 summarized that climate change is expected to reduce UWR Chinook adult abundance in the North Santiam River, South Santiam River, McKenzie River, and Middle Fork Willamette River, and stated additional factors not included in their life cycle model will likely influence the response of populations to climate change through 2040 and 2080, with a net effect of these factors likely be an increase the risk of extinction (further decrease abundance). Compared to UWR Chinook, Upper Willamette River winter steelhead (UWR steelhead) were found to have a high overall vulnerability, high biological sensitivity, high climate exposure and moderate adaptive capacity.

Table 4-1. Climate Change Vulnerability in Chinook and Steelhead

Vulnerability	UWR Chinook	UWR steelhead
Overall vulnerability	Very high	High
Biological sensitivity	Very high	High
Climate exposure	High	High
Adaptive capacity	Moderate	Moderate

Since vulnerability was assessed as higher for UWR Chinook than for UWR steelhead, we focused our assessment of climate change for the WVS EIS on this species and assumed results from this approach would be somewhat conservative for considering these effects for UWR steelhead. We further assumed the scoring for spring Chinook overall vulnerability would be found relatively similar when compared across alternatives for bull trout since both species are dependent on adequately cold water to complete their life cycle largely found above WVS dams, there is no known bull trout spawning habitat below dams where the reside or are proposed for reintroduction, and conditions below dams being assessed here would apply for rearing bull trout. Although relative results would be expected to be similar for bull trout, climate vulnerability bull trout would likely be somewhat underestimated when assuming scores for UWR Chinook due to the especially cold-water requirements of bull trout.

4.2 METHODS – OVERVIEW

To assess the vulnerability of spring Chinook salmon to climate effects under each WVS EIS alternative we followed the principles of the Crozier et al. framework, with the same objective to characterize the relative degree of threat posed by each component of vulnerability. We used results from lifecycle models applied in the WVS EIS to characterize population viability for existing climate conditions, and then assessed how attributes relating to species exposure sensitivity, and adaptive capacity would change among WVS EIS alternatives when factoring in the effects of climate change.

For sensitivity, Crozier et al. assessed different life-stages of each DPS, ocean acidification, population viability, hatchery influence and other stressors. Among these attributes, we focused on population viability since this measure of population performance accounts for the major attributes expected to change across WVS EIS alternatives. Extinction risk estimates were taken from life cycle modeling completed for each EIS alternative by UBC and NWFSC. The proportion of hatchery origin spawners strongly relates to extinction risk estimates in population models. Extinction risk is low when population replace rates are near or greater than 1, and when this occurs, managers plan to reduce or elimate outplanting of hatchery origin adults.

For exposure, we focused our assessment on differences in freshwater attributes included by Crozier et al.: stream temperature, summer water deficiency, flooding and hydrologic regime. To characterize freshwater conditions for our assessment for each WVS EIS alternative, we considered regulated and unregulated stream reaches separately in order to account for the influence of reservoirs and fish passage conditions when assessing effects of climate change.

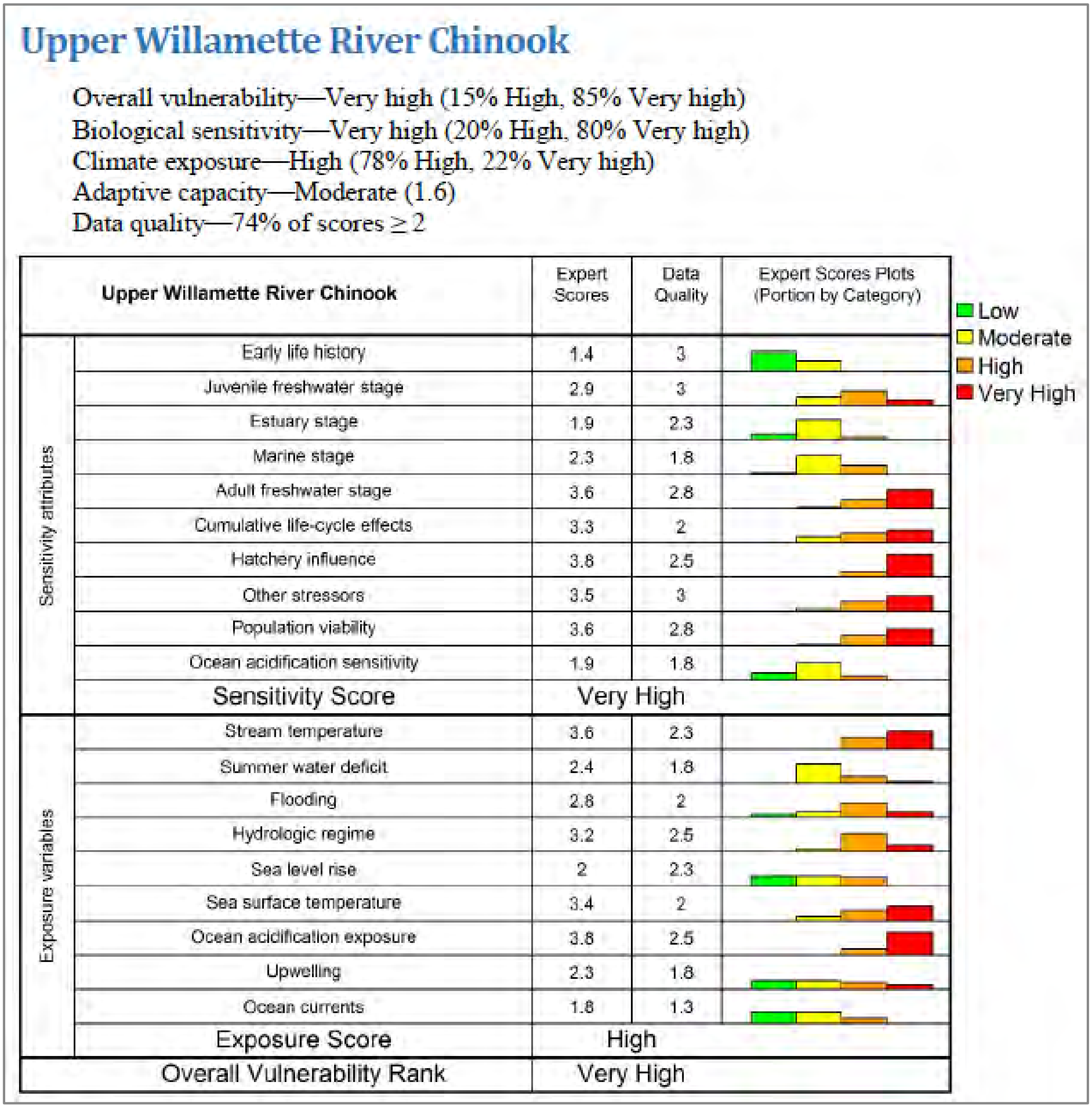
Crozier et al. used the evapotranspiration differential (potential minus actual), also known as the summer water deficit to assess effects of climate change on summer stream flows. We adjusted their scores, for above dam river reaches where passage for Chinook was included in an alternative, based on predicted changes in winter and summer precipitation and summer air temperatures. For below dam reaches, reservoirs have an important effect on summer flows and therefore we applied a qualitative assessment of reservoir storage availability with future climate change as a proxy for stream flow below dams.

We considered the availability of High Cascade base flows to inform their potential influence on the resiliency in the Santiam, McKenzie and Middle Fork Willamette sub-basins. Table 1 in Tague and Grant (2004) summarizes these contributions by streams in the Willamette River Basin. McKenzie, followed by the North Santiam sub-basin, had the highest percentage of high Cascade base flows. Middle Fork had High Cascade base flow contributions only tributaries below Hills Creek Dam, which would not contribute to areas used for adult holding and spawning of spring Chinook salmon. Redd capacities changed very little in the North Santiam and McKenzie above WVS dams under future climate change temperature scenarios (Bond et al. 2017), and so we assumed the resiliency due to the greater contribution of High Cascade base flow in these sub-basins is reasonably reflected in the assessment under the attributes where redd capacities are applied (see below).

Using the definitions of attributes from Crozier et al. 2019, we assumed the following specific attributes would not be different among EIS alternatives, and therefore applied results for these attributes from Crozier et al. 2019:

- Ocean acidification
- Sea surface temperature
- Hydrologic regime
- Cumulative life-cycle effects
- Adaptive capacity

For the other attributes assessed, criteria were developed to categorize each attribute for each alternative from a low to very high. Criteria for assigning these categories are provided below. The categorized bins were then assigned a numerical value (low = 1, moderate = 2, high = 3, very high = 4). Finally overall vulnerability was determined by multiplying the numeric values for sensitivity, exposure and adaptive capacity, and assigning a total score for each alternative based on the product. The product values were converted to cumulative vulnerability categories using the scoring logic from Crozier et al (2019) presented in their Table 3 (copy below).



Upper Willamette River Chinook

Overall vulnerability—Very high (15% High, 85% Very high)
Biological sensitivity—Very high (20% High, 80% Very high)
Climate exposure—High (78% High, 22% Very high)
Adaptive capacity—Moderate (1.6)
Data quality—74% of scores ≥ 2

	Upper Willamette River Chinook	Expert Scores	Data Quality	Expert Scores Plots (Portion by Category)
Sensitivity attributes	Early life history	1.4	3	
	Juvenile freshwater stage	2.9	3	
	Estuary stage	1.9	2.3	
	Marine stage	2.3	1.8	
	Adult freshwater stage	3.6	2.8	
	Cumulative life-cycle effects	3.3	2	
	Hatchery influence	3.8	2.5	
	Other stressors	3.5	3	
	Population viability	3.6	2.8	
	Ocean acidification sensitivity	1.9	1.8	
	Sensitivity Score	Very High		
Exposure variables	Stream temperature	3.6	2.3	
	Summer water deficit	2.4	1.8	
	Flooding	2.8	2	
	Hydrologic regime	3.2	2.5	
	Sea level rise	2	2.3	
	Sea surface temperature	3.4	2	
	Ocean acidification exposure	3.8	2.5	
	Upwelling	2.3	1.8	
	Ocean currents	1.8	1.3	
	Exposure Score	High		
	Overall Vulnerability Rank	Very High		

Figure 4-1. Climate vulnerability assessment results for Upper Willamette spring Chinook salmon. *Climate vulnerability assessment results for Upper Willamette spring Chinook salmon reproduced from Crozier et al. 2019. Note the assessment results assume current fish passage conditions at WVS dams, representative of the WVS EIS NAA.*

4.3 METHODS - CRITERIA APPLIED FOR ASSESSMENT OF ATTRIBUTES

Using the definitions of attributes from Crozier et al. 2019, we assumed the following specific attributes would not be different among EIS alternatives, and therefore applied results for these attributes from Crozier et al. 2019:

- ocean acidification

- sea surface temperature
- hydrologic regime
- cumulative life-cycle effects
- adaptive capacity

Stream Temperatures

Bond et al. 2016 estimated changes in redd capacity for UWR spring Chinook population affected by WVS dams in future water temperature scenarios for years 2040 and 2080. The percent of future available spawning habitat in each EIS alternative, from Bond et al for 2040 and 2080 scenarios, were used to score stream temperature effects above each dam where improvements to downstream fish passage are included as a measure in an alternative. Water temperature effects below dams are accounted for in extinction risk estimates from life cycle models applied for assessing population viability.

Table 4-2. Percent of accessible future Chinook spawning habitat above WVS dams

-	<50%	50-74%	>=75%
Vulnerability criteria	High	Moderate	Low

Summer Water Deficit

Crozier et al. used the evapotranspiration differential (potential minus actual), also known as the summer water deficit. We applied their scores for above dam river reaches where passage for Chinook was included in an alternative, adjusted for change in precipitation patterns, air temperatures, and availability of high Cascade base flows.

For below dam reaches, reservoirs have an important effect on summer flows and therefore we applied a qualitative assessment of reservoir storage availability with future climate change as a proxy for stream flow below dams.

Adult Freshwater Stage

Bond et al. 2016 estimated change in redd capacity was assessed along with the resiliency of fish passage and temperature management at dams. Downstream fish passage resiliency of each alternative was assessed based on the type of downstream fish passage operations included (specifically the number of spring deep drawdowns) and the number of downstream fish passage structures included in each alternative. Spring deep drawdowns were assumed resilient to climate change since drawing the reservoir down low in spring can occur in both wet and dry year types whereas surface spill operations require adequate inflows to refill reservoir between February and May. The resiliency of water temperature management at each dam was assessed based on the number of water temperature management structures included in each alternative.

Table 4-3. Criteria used to assess climate change resiliency of Dam downstream fish passage.

	Resiliency		
Criteria	Low	Moderate	High
Flexibility in DSP ops	spring deep drawdowns at 1 or fewer dams	spring deep drawdowns at 2-3 dams	spring deep drawdowns at 4-5 or more dams
No. of DSP structures	0-1 dams	2-2.5 dams	3 or more dams

4.3.2 Population Viability

For a viable population, assumed 3 populations need to be at low extinction risk. This is a conservative application of the UWR 2011 Recovery Plan delisting criteria: "a. At least two populations in the ESU and DPS meet Population viability criteria (see 2 below), b. The average of all population extinction risk category scores with the ESU or DPS is 2.25 or greater." The minimum number of populations with low extinction risk (<0.05) from results of modeling by UBC and NWFSC was used for assessing this attribute.

Table 4-4. Number of populations with low risk of extinction (p <0.5)

Number of Populations	3	2	1
Vulnerability criteria	Low	Moderate	High

Hatchery influence

The same scores applied for population viability were applied for hatchery influence. When population extinction risk is low when estimated in UBC and NWFSC lifecycle models, this reflects that cohort replacement for natural origin spawners is near 1 and that fish passage has improved allowing release of hatchery fish above dams to be reduced.

Other stressors

Considered change in attributes highlighted by Crozier for UWR Chinook: above dam habitat access, survival of transported fish, PSM, non-native fishes and contaminants. we applied above dam future habitat availability under future temperature scenarios from Bond et al. 2016 for above dam habitat access where fish passage is improved in an EIS alternative (see criteria under “stream temperatures” above). For PSM, we assessed the number of new adult traps at WVS dams meeting NMFS criteria as a proxy for managing transport survival and timing in each alternative (see table below). For resiliency in temperature management at dams, we assessed the number of structures included in each alternative, assuming structures allow for more flexibility in managing water temperature discharged at a range of pool elevations compared to operations using existing dam outlets. For contaminants and non-natives, we based scores on results from Crozier et al. 2019.

Table 4-5. Number Of Adult Traps Compared to Vulnerability

Number of traps	≤5	6	7
Vulnerability criteria	High	Moderate	Low

Table 4-6. Number of Temperature Structures Compared to Vulnerability Criteria

Number of temperatures Structures	1	2	3
Vulnerability criteria	High	Moderate	Low

Table 4-7. Copy of Table 3 from Crozier et al. (2019) used to convert scores to cumulative vulnerability categorical ratings.

Table 3. Logic rule for ranking sensitivity and exposure components and cumulative vulnerability. We used the logic rule across attributes to assign a numeric score and vulnerability category to sensitivity and exposure components (top section). We then used the product of the numeric component scores to assign cumulative vulnerability for each DPS (bottom section).

Overall sensitivity or exposure score	**Numeric score**	**Logic rule**
Very High	4	More than 3 attribute means ≥ 3.5
High	3	More than 2 attribute means ≥ 3
Moderate	2	More than 2 attribute means ≥ 2.5
Low	1	All other scores
Cumulative vulnerability	**Component product**	**Component combinations**
Very High	≥12	Very high/high or Very high/very high
High	8-11	Very high/moderate or High/high
Moderate	4-6	Very high/low, High/moderate, or Moderate/moderate
Low	≤ 3	High/low, Moderate/low, or Low/low

Table 4-8. METHODS - Data sources used for assessment of each attribute.

Attribute Type	Attribute	Data Source	Data type
Exposure Attributes	Ocean Acidification[1]	Crozier et al. 2019	(see Crozier et al. 2019)
Exposure Attribute	Stream Temperature	Bond et al. 2016	Spring Chinook redd capacity above WVS dams under 2040 and 2080 projected stream temperatures
Exposure Attribute	Sea Surface Temperature[1]	Crozier et al. 2019	(see Crozier et al. 2019)
Exposure Attribute	Hydrologic Regime[1]	Crozier et al. 2019	(see Crozier et al. 2019)
Sensitivity Attribute	Adult Freshwater Stage	Bond et al. 2016; Measures in each EIS alternative	Spring Chinook redd capacity above WVS dams under 2040 and 2080 projected stream temperatures; number of fish passage structures in each EIS alternative; number of spring deep reservoir drawdowns; number of temperature towers.
Sensitivity Attribute	Cumulative Life-Cycle Effects[1]	Crozier et al. 2019	(see Crozier et al. 2019)
Sensitivity Attribute	Population Viability	UBC 2022; NWFSC 2022	Extinction risk estimates for each spring Chinook population based on lifecycle models
Sensitivity Attribute	Hatchery Influence	UBC 2022; NWFSC 2022	Extinction risk estimates for each spring Chinook population based on lifecycle models. Assume extinction risk estimates inversely related to pHOS since outplanting of hatchery fish will reduce as cohort replacement for natural origin returns is achieved.
Sensitivity Attribute	Other Stressors	Bond et al. 2016; Measures in each EIS alternative	Spring Chinook redd capacity above WVS dams under 2040 and 2080 projected stream temperatures; number of NMFS-criteria adult collection facilities, non-native fishes and contaminants; number of temperature management structures at dams
Sensitivity Attribute	Adaptive Capacity[1]	Crozier et al. 2019	(see Crozier et al. 2019)

4.4 RESULTS

The cumulative vulnerability of UWR Chinook was rated as high to very high across the WVS EIS alternatives. These high and very high ratings reflect scores included for ocean acidification, seas surface temperature, hydrologic regime and cumulative life-cycle effects. Among the alternatives, 2b and 4 received the lowest cumulative vulnerability scores (10.0). These results were driven by better (lower) population viability and hatchery influence scores as compared to the other alternatives. Alternative 3a and 3b had the highest vulnerability scores (14.9). Vulnerability scores for 3a and 3b reflect the poor results for the summer water deficit below dam's attribute, population viability and hatchery influence attributes when compared to the other alternatives.

Table 4-9. Attribute categorization results for assessment of climate vulnerability of Upper Willamette spring Chinook salmon.

Attribute Type	Attribute	NAA[1]	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Exposure Attributes	ocean acidification[1]	Very high	Very high	Very high	Very high	Very high	Very high	Very high
Exposure Attributes	stream temperature	Very High	Moderate	Low	Low	Low	Low	Low
Exposure Attributes	sea surface temperature[1]	High	High	High	High	High	High	High
Exposure Attributes	hydrologic regime[1]	High	High	High	High	High	High	High
Exposure Attributes	summer water deficit_above dams[1]	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate
Exposure Attributes	summer water deficit_below dams	Moderate	Moderate	Moderate	Moderate	High	High	Moderate
Sensitivity Attributes	adult freshwater stage	Very High	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate
Sensitivity Attributes	cumulative life-cycle effects[1]	Very High	Very High	Very High	Very High	Very High	Very High	Very High
Sensitivity Attributes	population viability	Very High	Moderate	Low	Moderate	High	High	Low
Sensitivity Attributes	hatchery influence	Very High	Moderate	Low	Moderate	High	High	Low
Other types	other stressors	High	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate
Adaptive type	Adaptive Capacity[1]	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate

Note: [1] Results for the NAA and attributes marked with a (1) are adopted from Crozier et al. 2019.

Table 4-10. Vulnerability Results with Assessment Categories as Numeric Scores. *

Attribute	NAA[1]	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Exposure Attributes	High	2.7	2.5	2.5	2.7	2.7	2.5
Ocean Acidification[1]	Very high	4.0	4.0	4.0	4.0	4.0	4.0
Stream Temperature	Very High	2.0	1.0	1.0	1.0	1.0	1.0
Sea Surface Temperature[1]	High	3.0	3.0	3.0	3.0	3.0	3.0
Hydrologic Regime[1]	High	3.0	3.0	3.0	3.0	3.0	3.0
Summer Water Deficit_Above Dams[1]	Moderate	2.0	2.0	2.0	2.0	2.0	2.0
Summer Water Deficit_Below Dams	Moderate	2.0	2.0	2.0	3.0	3.0	2.0
Sensitivity Attributes	Very High	2.4	2.0	2.4	2.8	2.8	2.0
Adult Freshwater Stage	Very High	2.0	2.0	2.0	2.0	2.0	2.0
Cumulative Life-Cycle Effects[1]	Very High	4.0	4.0	4.0	4.0	4.0	4.0
Population Viability	Very High	2.0	1.0	2.0	3.0	3.0	1.0
Hatchery Influence	Very High	2.0	1.0	2.0	3.0	3.0	1.0
Other Stressors	High	2.0	2.0	2.0	2.0	2.0	2.0
Adaptive Capacity[1]	Moderate	2.0	2.0	2.0	2.0	2.0	2.0
Overall Vulnerability	Very High	12.8	10.0	12.0	14.9	14.9	10.0
Overall Vulnerability	Very High	Very High	High	Very High	Very High	Very High	High

Notes: Overall vulnerability results based on conversion of assessment categories to numeric scores.

Results from Crozier et al. (2019) are applied for the NAA.

Results for attributes noted with a superscript 1 are also from Crozier et al. (2019), assuming these attributes would not be changing under each WVS EIS alternative.

4.4.1 ADDITIONAL SUPPORTING INFORMATION

Table 4-11. Distribution of fish passage among WVS EIS alternatives as applied for the fish climate change assessment.

	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
NS_DET	n	y	y	y	y	y	y
SS_FOS	y	y	y	y	y	y	y
SS_GRP	n	y	y	y	y	y	n
MCK_CGR	n	n	y	y	y	y	y
MF_LOP	n	y	y	y	y	y	y
MF_HCR	n	n	n	n	y	y	y

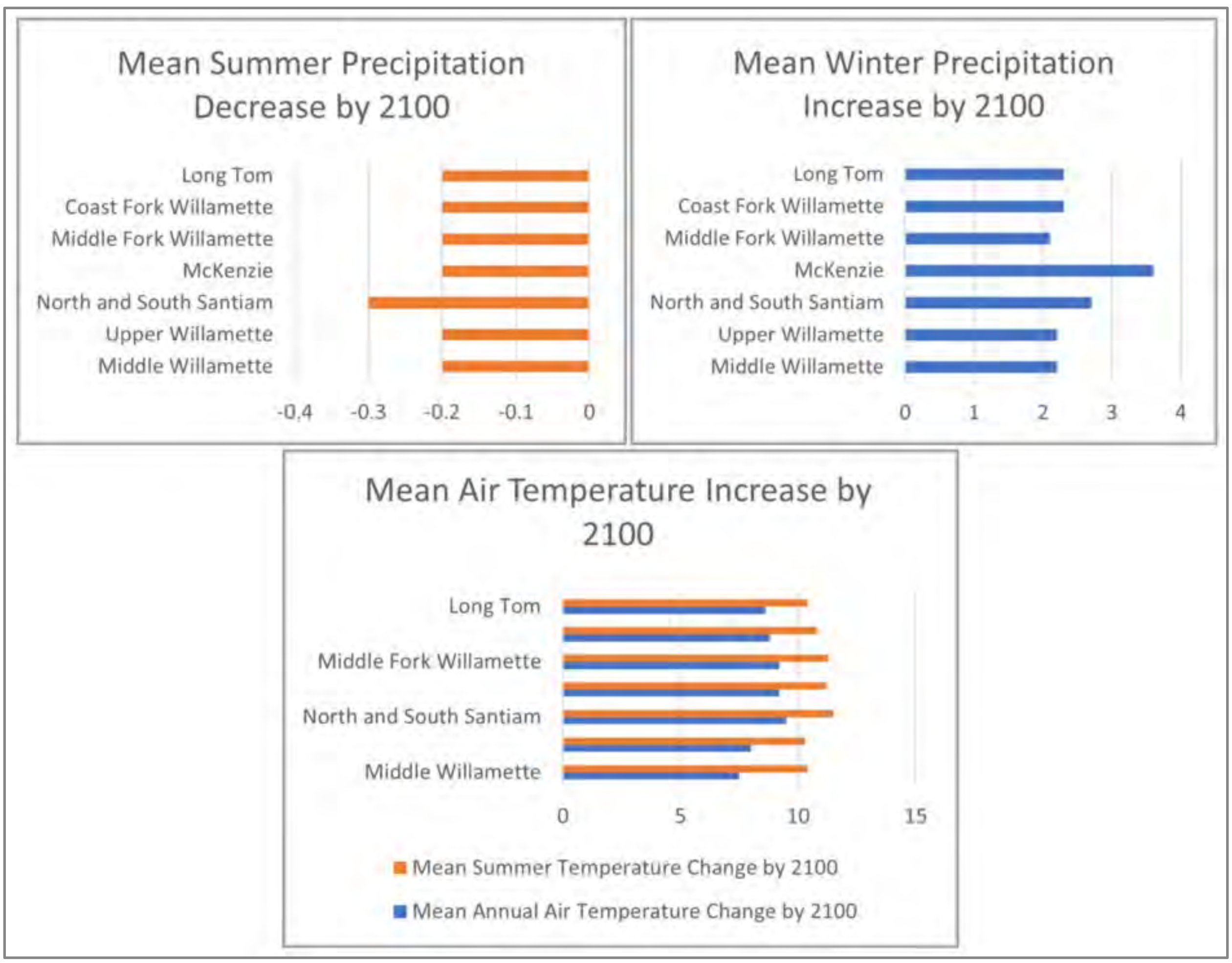


Figure 4-2. Temperature and Precipitation Changes in Upper Willamette River watersheds. *Summary of changes in air temperatures and precipitation for Upper Willamette River watersheds affected by the WVS. Northwest Climate Toolbox; RCP8.5; Data Source: MACAv2.*

Table 4-12. Contribution of High Cascade base flows. *

Table 1. Watersheds

Basin and Watershed	USGS Gauge Number	Drainage Area, mi^2	Elevation, feet	Percent High Cascade	Period of Record	Mean August, mm/month	Mean Annual, mm/year	Slope b	Intercept a	R2	RMSE	n
Middle Fork Willamette												
Fall	14150300	118	844	0%	1963.09.01–1999.09.30	11.23	1210.21	1.39	−3.40	0.78	0.93	3341
Hills	14144900	52.7	1631	9%	1958.10.01–1981.10.01	17.96	1433.72	1.53	−3.90	0.74	0.88	3211
Salmon	14146500	117	1462	52%	1986.10.01–1994.06.13	36.89	2150.63	2.04	−5.56	0.70	0.89	1892
Salt	14146000	113	1246	63%	1933.10.01–1951.09.30	30.68	1130.31	2.02	−5.33	0.64	0.95	3175
McKenzie												
Gate	14163000	47.6	764	0%	1966.10.01–1990.09.30	17.23	1103.92	1.46	−3.78	0.78	0.87	3313
Blue (at Tidbits)	14161100	45.8	1387	3%	1963.09.01–1999.09.30	13.60	2130.37	1.38	−3.50	0.80	0.90	3251
Blue	14161000	11.5	1960	3%	1947.10.01–1955.09.30	19.54	1700.99	1.20	−3.31	0.75	0.87	1950
Lookout	14161500	24.1	1377	16%	1963.09.01–1999.09.30	17.76	1689.55	1.42	−3.81	0.79	0.87	3226
Springfield[a]	14164000	1066	554	40%	1911.05.01–1915.03.31	56.15	2829.25	2.06	−5.89	0.61	0.85	2008
Coburg[a]	14165500	1337	392	46%	1944.10.01–1972.09.30	49.57	2678.66	2.18	−6.20	0.73	0.89	3268
Walterville[a]	14163900	1081	600	58%	1989.10.01–1999.09.30	32.94	1850.81	1.87	−4.89	0.52	1.36	2292
Leaburg[a]	14163150	1030	710	61%	1989.10.01–1999.09.30	30.28	1787.63	1.88	−4.82	0.54	1.32	2396
Vida[a]	14162500	930	856	68%	1924.10.01–1999.09.30	71.32	1750.04	2.44	−6.87	0.69	0.93	3080
S.F. McKenzie (above Cougar)	14159200	160	1710	68%	1957.10.01–1987.09.30	44.44	1191.64	2.30	−6.32	0.71	0.94	3320
Horse	14159100	149	1426	83%	1962.10.01–1969.09.30	59.63	1532.56	2.73	−7.59	0.60	0.89	1649
Mckenzie Bridge[a]	14159000	348	1419	88%	1910.10.01–1994.09.30	101.40	2084.72	3.02	−9.28	0.53	0.83	2458
Clear[a]	14158500	92.4	3015	95%	1937.10.01–1999.09.30	96.59	2853.57	2.10	−7.23	0.60	0.76	3421
Belknap[a]	14158700	146	2602	95%	1957.10.01–1962.09.30	93.17	1485.27	3.16	−9.85	0.63	0.74	1220
South Santiam												
Quartzville	14185900	99.2	1050	3%	1965.08.10–1999.09.30	43.39	1837.78	1.38	−3.47	0.80	0.93	3321
North Santiam												
L.N. Santiam	14182500	112	655	5%	1931.10.01–1999.09.30	16.59	2003.09	1.37	−3.42	0.81	0.90	3268
E. Humbug	14178700	7.32	2050	0%	1978.08.01–1994.07.10	15.01	1316.71	1.49	−3.81	0.82	0.86	3346
Breitenbush	14179000	108	1574	46%	1932.06.01–1987.10.01	46.27	2350.72	1.82	−5.06	0.72	0.91	3313
Santiam	14178000	216	1591	78%	1928.10.01–1999.09.30	64.57	1877.80	2.24	−6.51	0.67	0.90	3215
Clackamas												
Clackamas	14208000	136	2040	82%	1920.04.01–1970.09.30	56.13	1989.92	2.69	−7.46	0.67	0.90	3105
Fish	14209700	45.2	940	3%	1963.09.01–1999.09.30	12.34	1697.19	1.46	−3.54	0.84	0.82	2368
Oak	14208500	54	3140	85%	1915.10.01–1928.09.30	79.97	2091.87	2.30	−7.20	0.47	0.86	2264
Roaring	14209600	42.4	1040	30%	1966.01.28–1968.09.30	41.91	1370.95	1.74	−4.83	0.61	0.94	622

[a]Gauge on main stem.

Source: Tague and Grant 2004.

Table 4-13. Spring Chinook spawning habitat and predicted change 2040 and 2080. *

Tributary	Redd Capacity Estimates	Redd Capacity Estimates	Redd Capacity Estimates	Percent of total habitat	Percent of total habitat	Percent of total habitat	Percent reduction in redd capacity	Percent reduction in redd capacity
Timeframe	1993-2011 avg temp	2040 projected temp	2080 projected temp	1993-2011 avg temp	2040 projected temp	2080 projected temp	2040 projected temp	2080 projected temp
North Santiam Below Detroit	22,693	19,388	12,712	59%	55%	45%	15%	44%
North Santiam Above Detroit	15,602	15,602	15,602	41%	45%	55%	0%	0%
North Santiam Total	38,295	34,990	28,314	100%	100%	100%	9%	26%
South Santiam Below Foster	8787	4213	2060	59%	69%	69%	52%	77%
South Santiam Above Foster	4,504	1,640	923	30%	27%	31%	64%	80%
South Santiam Above Green Peter	1508	257	0	10%	4%	0%	83%	100%
South Santiam Total	14799	6110	2983	100%	100%	100%	59%	80%
McKenzie Below Cougar and Trail Bridge dams	44,480	39,439	32,698	89%	88%	86%	11%	26%
McKenzie Above Cougar Dam	5,423	5,423	5,416	11%	12%	14%	0%	0%
McKenzie Total	49,903	44,862	38,114	100%	100%	100%	10%	24%
Middle Fork Below Fall Cr/Dexter/Lookout Point dams	8,813	3,801	1,418	8%	4%	1%	57%	84%
Middle Fork Above Fall Creek Dam	3,419	1,220	579	3%	1%	1%	64%	83%
Middle Fork Above Dexter/Lookout Point dams	72,937	70,649	68,691	65%	68%	70%	3%	6%
Middle Fork Above Hills Creek Dam	27,532	27,525	26,803	24%	27%	27%	0%	3%
Middle Fork Total	112,701	103,195	97,491	100%	100%	100%	8%	13%

Notes: Estimated habitat for spring Chinook salmon spawning (redd capacity) in primary spawning tributaries affected by WVS dams, and predicted change in capacity from projected water temperatures in 2040 and 2080.

Sources: Redd capacity data reproduced from Bond et al. (2017).

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CHAPTER 5 - FLOW-SURVIVAL MODEL

Evaluation of the Effectiveness of Alternative Willamette River Flow Regimes for Supporting At-risk Salmonids

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5.1 INTRODUCTION

Stream flow regulation is one of the most important issues facing natural resource managers and planners in the Pacific Northwest. In recent years, a rapidly growing human population has led to increased water demands from agriculture, industry, and municipalities. These changes are evident in the Willamette Valley where human population growth has increased more than 15% during just the last decade. Increased water demands likely have unintended negative impacts on the aquatic resources in the Willamette Valley. Natural resource managers and planners can be effective at meeting the multiple water needs only if they are informed as to the nature and extent of potential impacts of water management actions on their objectives. This requires the ability to assess the suitability of the aquatic environment to support biota, coupled with a reliable assessment of the potential effects of future management actions.

We initiated a process to support the development of instream flow recommendations for the Willamette Basin Review (WBR) feasibility study and address the National Marine Fisheries Service 2008 Biological Opinion for the US Army Corps of Engineers (USACE) Willamette Project (DeWeber and Peterson 2020). Specifically, the original objectives of the project were to work with scientists, managers, and other stakeholders to identify ecological objectives and hypothesized flow-ecology relationships, review existing information on biological and anthropogenic instream flow needs in the river, and develop a framework for identifying analyses or assessments to review information and hypotheses on instream flow needs. This process began in April 2016, when a group of managers and scientists (Instream Flow Science Group) met to discuss the instream flow needs for the Willamette Basin. Since then, we have worked with an interdisciplinary team of scientists and other subject matter experts to develop decision support models (DSMs) for evaluating the effects of flows on juvenile and adult salmonids. We then use DSMs to evaluate the relative effectiveness of alternative flow and thermal management regimes provided by the USACE on spring Chinook salmon and winter Steelhead trout to assess their relative effectiveness.

Here, we describe the results of our cooperative effort with the USACE and their collaborators and contractors in support of Willamette Valley System Environmental Impact Statement (WVS EIS). Our ultimate goal was to support assessments of the response of salmonids to alternative flow regimes using existing DSMs. In what follows, we provide brief

descriptions of the management area, the DSMs, and DSM inputs. We then report the results of simulations under seven alternative flow management regimes. Detailed descriptions of the DSMs including sources of information can be found in Peterson et al. (2021).

5.2 METHODS

Management area. - The DSM simulated the dynamics of salmonids within the Willamette River upstream of Willamette Falls Dam and the main salmon-bearing tributaries: the North and South Santiam Rivers, the McKenzie River, the Middle Fork Willamette River, and Fall Creek. We included only the areas above Willamette Falls Dam and below the USACE projects in the DSM (i.e., the model extent). The mainstem and tributaries were subdivided into 18 sections (Figure

1) that varied in length from 7 to 69 km. These sections served as the grain for modeling the ecological dynamics within each.

5.2.1 Decision support models

The water management season in the Willamette Basin runs from April 1 to October 1, so flow management in a given water year affects juvenile salmonids from previous brood years and returning adults for the current brood year. Therefore, we developed four DSMs for adult Chinook salmon, juvenile Chinook salmon, returning Steelhead adults, and outmigrating Steelhead smolt. The models are documented in Peterson et al. (2021) and are briefly described below. All models operated on a weekly time step that began on the eighth week of the year and ran through April of the following year. The values used for initial conditions (e.g., number of returning adults salmon) were based on the upper range of observed values in the two past decades and expert judgement. All modeling was conducted using R statistical software (R Core Team 2021).

Adult Chinook salmon DSM. –

The adult Chinook salmon model was a stochastic model that tracked the number of adult salmon passing Willamette Falls and traveling upstream to spawning reaches where they remained until spawning. Following spawning, the model tracked the number of redds until the eggs hatched. The model began with a randomly generated number of adult salmon passing Willamette Falls (90,000 +/- 35,000). The distribution of adults among the four spawning tributaries, and the passage week were randomly assigned using empirical models that were fitted to observed telemetry and passage data, respectively. Once assigned to a tributary, the model tracked each returning tributary and passage week 'cohort' through the stream sections until they reached the spawning tributary section. Movement rate was modeled using an empirical model fit to existing telemetry data and an en route survival model was developed from published sources. Fish held in the spawning reaches until the second week in September when they were randomly assigned one of three consecutive spawning weeks.

Survival prior to spawning was modeled using an empirical model fitted to ODFW spawning ground survey data. Spawning females created a redd in available habitat as if insufficient habitat were available (i.e., redd capacity exceeded), females spawning in weeks two and three destroyed a previous redd through superimposition. The redds survived until hatching the following winter or spring depending on thermal exposure.

Juvenile Chinook salmon DSM.-

The juvenile Chinook salmon model is a stochastic stage-based model that tracks the number of fish by tributary of origin and six size classes: (fork length): <60 mm, 60-75 mm, 75-90 mm, 90-105 mm, 105-120 mm, and >120 mm. The model begins with a randomly generated number of redds that vary by tributary: North (420 +/- 206) and South (544 +/- 276) Santiam Rivers, the McKenzie River (1173 +/- 577), and the Middle Fork Willamette River (32 +/- 15). Rearing subyearling Chinook salmon from the previous brood year are also randomly generated for each tributary (10000 +/- 3000) and are randomly assigned to size classes based on observed sizes in historical seine data collected by ODFW. Transition between stages and river sections were estimated using survival, growth, and movement submodels that were fit using juvenile data collected by ODFW (1999-2018). Fry swim-up from redds in each tributary was modeled as a function of accumulated degree days and assigned to the smallest size class. Juvenile Chinook salmon used all available habitat and when habitat capacity was exceeded, individuals would move to the next downstream segment.

Subyearling Chinook salmon 105-120 mm fork length left the basin in March-July, whereas yearlings >120 mm emigrated from the basin October-April. Juveniles that survived to pass Willamette Falls were transformed into adult equivalents using a transition function that was fitted to juvenile tagging and adult return data collected by ODFW from 1999-2018.

Adult Steelhead trout DSM.-

The adult Steelhead trout model was a stochastic model that tracked returning adults, eggs, and six size classes of juvenile fish: <60 mm, 60-75 mm, 75-90 mm, 90-105 mm, 105-120 mm, and >120 mm. The model simulated dam passage, movement, and survival using submodels that were primarily drawn from published sources and expert judgment. The model also made several assumptions that were necessary given the lack of information on Steelhead trout in the Willamette Basin and their interactions with resident conspecifics, Rainbow trout. The model assumed that all spawning and rearing occurred in the mainstem North and South Santiam Rivers and downstream reaches and that there were no interactions or competition for habitat or other resources with Rainbow trout. Similar to the adult Chinook salmon DSM, the model began by randomly generating returning adults (20,000 +/- 9,849) and assigning them to a spawning tributary and a return week and then by following each cohort through the stream sections until they reached the spawning tributary section. Survival and movement of adults through the sections and prespawn survival of adults was estimated using information from published studies. Surviving adult females were randomly assigned to spawning weeks based on expert judgment of spawning timing. Females created redds in all available spawning habitat and when habitat was limited, redd superimposition occurred. Eggs in redds survived until hatching with the time until emergence depending on thermal exposure.

Emerging fry were assigned to the smallest size class. Similar to the juvenile Chinook salmon DSM, juvenile Steelhead trout used all available habitat and they moved to the next downstream river section if all the available habitat was used. Juvenile Steelhead trout survived and grew until the last week in April. The model output included the total number of juvenile Steelhead trout produced.

Steelhead smolt outmigrant DSM.-

The Steelhead trout smolt outmigrant model was a stochastic model that tracked outmigrants from the North and South Santiam Rivers to Willamette Falls. It was relatively simple and included three components; outmigrant timing, movement, and survival submodels. For this exercise, we used the earlier outmigrant timing submodel and the survival submodel that was fitted using tagging data collected by ODFW from 1999-2018.

5.2.2 Model inputs

Juvenile Steelhead trout habitat suitability criteria were based on values from a meta- analysis of published literature (White et al. 2022). Juvenile Chinook habitat suitability criteria were based on observations of fish habitat use in the Willamette River during base flows and adjusted for incomplete detection (Hansen et al. in review). We also created three sets of habitat suitability criteria: narrow, median, and broad, to capture the variability in habitat suitability in the literature (White et al. 2022). The narrow criterium included a relatively restricted range of depths and velocities that were considered to be suitable for habitat, whereas the broad category included a wide range of suitable habitat depths and velocities. The range of suitable conditions for the median category were midway between the narrow and broad categories. The simulations described below were run separately using each habitat suitability criteria.

Chinook salmon and Steelhead trout spawning and rearing habitat-discharge relationships were based on published relationships (from Gagner et al. 2014, River Design Group 2015, and White et al. 2022). Habitat models were not available for 6 of the 18 stream sections that were primarily in the McKenzie River basin. Habitat-discharge relations in these sections were estimated using expert judgment as detailed in Peterson et al. (2021).

The USACE provided estimated discharge and water temperature under seven flow management alternatives for three representative years: 2011, 2015, and 2016. Streamflows were estimated by USACE modelers using HEC-ResSim 3.3.1.124 (Klipsch et al. 2021). Water temperatures were modeled by US Geological Survey, Oregon Water Science Center scientists using CE-QUAL-W2 (Wells 2020).

5.2.3 Simulations

For each alternative flow regime, the DSMs were run for 10,000 iterations and the mean outcome was calculated by year across iterations. The 10,000 simulations were sufficient to obtain means that were within +/ 1% of the true mean value with 95% confidence, based on an analysis of means and standard deviations from preliminary simulations with 500 iterations. For each combination of outcome, tributary and juvenile habitat category, we ranked the outcomes from best (1) to worst (7) for each flow management alternative for each year and counted the number of instances where an alternative was top ranked. To facilitate comparisons among DSM outputs on very different scales (e.g., redds vs. smolt survival probabilities), we also calculated utility values which rescale the model outputs to values that range from zero (worst alternative) to one (best alternative). Finally, we combined the utilities across outputs two different ways: 1) equally weighting each response-specific utility (i.e., by 0.25 each) and summing the weighted utilities and 2) multiplying the utilities. The latter approach results in a combined utility of zero when any individual response utility was zero. We also calculated the expected loss of implementing a candidate flow regime relative to the optimal flow regime following Peterson et al. (2021). Values represent a decrease in the estimated outcome under a given flow management alternative relative to the best flow alternative (i.e., relative loss for the best flow management alternative is zero).

5.3 RESULTS

The tributary specific estimates for each flow management alternative can be found in Appendix A. The remainder of this section focuses on summaries across years and tributaries with an emphasis on interpreting patterns in the data.

The simulations indicated that, except for Steelhead trout smolt survival, no single flow alternative was best across all years and tributaries (Table 4-1). The NAA alternative was always best for Steelhead trout smolt survival, whereas each alternative was ranked best at least three times for Chinook salmon adult equivalents. Similarly, most of the alternatives were ranked best at least three times for Chinook salmon redds surviving until swim-up outcome.

Simulation results indicated minor differences for estimates under alternative juvenile habitat categories. A comparison of identical flow management alternatives under narrow, median, and broad habitat categories revealed differences in estimated Chinook salmon adult equivalents and age-1 Steelhead trout abundance averaging +/- 3.2% and 0.2%, respectively (Table 4-2).

Estimated differences between narrow and broad habitat categories were slightly greater, averaging +/- 4.2% and 0.3% for Chinook salmon adult equivalents and age-1 Steelhead trout, respectively. Nonetheless, the small differences in the estimated outcomes did change the rankings of flow management alternatives for the Chinook salmon adult equivalents outcome with NAA ranked as best under the narrow and median habitat definitions and Alt3b as best under the broad habitat definition (Table 4-3).

The utilities calculated using the basin-wide outcomes averaged across years indicated that the effects of the flow management alternatives varied by species and life history stage (Table 3). The NAA alternative was always best for Steelhead trout survival and Chinook salmon adult equivalents under the narrow and broad juvenile habitat categories. The alternative Alt4 was always best for Chinook salmon redds surviving until swim-up, whereas Alt3b was best for age-1 Steelhead trout. The combined utilities across outcomes indicated that alternatives NAA and Alt4 were consistently the top two flow management alternatives across juvenile habitat categories.

Utilities are intended to magnify the differences among decision alternatives to make patterns easier to identify, whereas relative loss estimates are intended to reveal the expected losses in outcomes that would occur if one of the alternatives were implemented in place of the best alternative for a given outcome. Simulation results suggested that implementing any one of the flow management alternatives other than NAA would result in Steelhead trout survival decreases by 3% or less (Table 4). In contrast, implementing alternatives Alt2b, Alt3a, and Alt3b would result in more than 10% fewer Chinook salmon redds surviving until swim-up compared to Alt4. Alternatives Alt1, Alt2a and Alt2b would lead to similar decreases in Chinook salmon adult equivalents relative to the best alternatives. Average relative loss across outcomes were lowest for NAA and Alt4 alternatives.

Tributary specific estimates also indicated that no one alternative was best among all tributaries with the exception of Steelhead trout smolt survival (Tables 5-7). The 2011 simulations indicated that the NAA was best in The McKenzie River, but Alt1, Alt4, and Alt3a were best for MF Willamette, SF Santiam, and NF Santiam rivers, respectively (Table 5). In contrast, the simulation results for 2015 were much more variable among flow management alternatives with coefficients of variation (standard deviation/mean) averaging 18% (Table 6) compared to results combined across years that averaged 4%. The corresponding utilities were similarly variable among tributaries and outcomes, except for Steelhead trout smolt survival that was always highest under NAA for all years (Table 8-10). The equally weighted and combined utilities indicated that the NAA was best for 2011 and 2016, while Alt4 alternative was best under 2015 conditions across tributaries for all outcomes and the two Chinook salmon outcomes, whereas NAA was best for the two Steelhead trout outcomes. The relative loss estimates reflected the variability among alternatives and simulation years. In 2011, relative loss estimates for Steelhead trout smolt survival was < 0.01 for all alternatives (Table 11). Average relative loss across all responses were also lower in 2011 and 2016 compared to 2015 with average loss across tributaries for all outcomes greater than 10% and in several instances greater than 20% (Tables 11-13).

The magnitude of the differences between the no action alternative (NAA) and the other flows was highly variable among years tributaries and responses (Tables 14-16). However, the differences between habitat definitions was relatively minor for Chinook salmon adult equivalents (Table 15), whereas differences among habitat definitions were much greater for age-1 Steelhead trout (Table 16).

5.4 DISCUSSION

Not surprisingly, the simulation results were consistent with those reported in Peterson et al. (2021) even though the juvenile Chinook salmon habitat definitions were different. The existing flow regulations (here, NAA) performed surprisingly well given the reported differences between existing flow management targets and optimal flows in deficit water years (Peterson et al. 2021). The effectiveness of the flow management alternatives also differed among outcomes and years. One potential reason for this variability may be due to partial controllability, that is the inability to perfectly implement management because parts of the system (e.g., inflows) are not completely under the control of managers. One approach to dealing with partial controllability may be to recast the decision as a hierarchical or multilevel decision problem, which would provide the framework to derive optimal decisions from hierarchically structured sequential decision-making processes (Chang et al. 2003, Wernz and Deshmukh 2012). Here, we envision an approach where a flow management alternative is implemented each year based on current conditions (e.g., snowpack, reservoir levels) at the start of the flow management season (the upper level). Flow management is then changed within the management season (lower-level decisions) at some predetermined intervals based on various system states (e.g., inflows, reservoir levels, dominant weather patterns). Such an approach is feasible to implement as it is similar to an ad hoc approach currently being used by the USACE and collaborating agencies to manage flows in the Umpqua Basin (G. Taylor, USACE, personal communication). All that is needed is to develop state-dependent management decision making rules using the existing models and, most importantly in-depth collaboration with water managers.

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5.5 TABLES

Table 5-1. Number of best outcome results of Simulations through repeated runs. *

Outcome	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Chinook salmon redds surviving until swim-up	2	2	5	0	1	1	1
Chinook salmon adult equivalents	6	5	3	5	5	8	4
Age-1 Steelhead trout	0	0	0	3	6	6	3
Steelhead trout smolt survival	6	0	0	0	0	0	0

Note: *The number of tributary, year, and juvenile habitat category simulations where each alternative was ranked best

Three habitat categories were used in simulations of Chinook salmon adult equivalents and Age-1 Steelhead trout so total rankings are three times greater than the other two responses.

Table 5-2. Outcomes of each decision support model habitat suitability category. *

Habitat Category	Flow Alternative	Surviving Chinook Redds	Chinook Salmon Adult Equivalents	Age-1 Steelhead	Steelhead Smolt Survival
Narrow	NAA	13294.1	8808.4	367856.6	93.2
Narrow	Alt1	13497.5	7254.5	353118.0	90.7
Narrow	Alt2a	13313.4	6984.2	359051.4	91.8
Narrow	Alt2b	12679.0	7775.9	368257.1	91.7
Narrow	Alt3a	12695.2	8308.4	371423.1	90.7
Narrow	Alt3b	12061.5	8562.6	392297.0	91.5
Narrow	Alt4	14488.2	7964.6	371273.3	91.8
Median	NAA	13294.1	8862.6	368108.4	93.2
Median	Alt1	13497.5	7155.5	354499.7	90.7
Median	Alt2a	13313.4	7241.5	358208.6	91.8
Median	Alt2b	12679.0	7296.1	368410.8	91.7
Median	Alt3a	12695.2	8166.4	371487.9	90.7
Median	Alt3b	12061.5	8083.1	391421.3	91.5
Median	Alt4	14488.2	7771.3	371748.7	91.8
Broad	NAA	13294.1	8058.6	369402.4	93.2
Broad	Alt1	13497.5	7156.8	353645.8	90.7
Broad	Alt2a	13313.4	7082.2	358831.7	91.8
Broad	Alt2b	12679.0	7655.4	368727.3	91.7
Broad	Alt3a	12695.2	7897.2	373287.3	90.7
Broad	Alt3b	12061.5	8065.8	394528.4	91.5
Broad	Alt4	14488.2	8029.7	371134.9	91.8

Note: Basin-wide outcomes for each of the four decision support models averaged across the three simulation years by juvenile salmonid habitat suitability category.

Table 5-3. DSM Utility Values Averaged for all Modeled Years and Habitat Categories *

Habitat Category	Flow Alternative	Surviving Chinook Redds	Chinook Salmon Adult Equivalents	Age-1 Steelhead Trout	Steelhead Smolt Survival	Combined Equal Weights	Multiplied Combined
Narrow	NAA	0.51	1.00	0.38	1.00	0.72	0.14
Narrow	Alt1	0.59	0.15	0.00	0.01	0.19	0.00
Narrow	Alt2a	0.52	0.00	0.15	0.44	0.28	0.00
Narrow	Alt2b	0.25	0.43	0.39	0.41	0.37	0.01
Narrow	Alt3a	0.26	0.73	0.47	0.00	0.36	0.00
Narrow	Alt3b	0.00	0.87	1.00	0.32	0.55	0.00
Narrow	Alt4	1.00	0.54	0.46	0.44	0.61	0.07
Median	NAA	0.51	1.00	0.37	1.00	0.72	0.13
Median	Alt1	0.59	0.00	0.00	0.01	0.15	0.00
Median	Alt2a	0.52	0.05	0.10	0.44	0.28	0.00
Median	Alt2b	0.25	0.08	0.38	0.41	0.28	0.00
Median	Alt3a	0.26	0.59	0.46	0.00	0.33	0.00
Median	Alt3b	0.00	0.54	1.00	0.32	0.46	0.00
Median	Alt4	1.00	0.36	0.47	0.44	0.57	0.04
Broad	NAA	0.51	0.99	0.39	1.00	0.72	0.14
Broad	Alt1	0.59	0.08	0.00	0.01	0.17	0.00
Broad	Alt2a	0.52	0.00	0.13	0.44	0.27	0.00
Broad	Alt2b	0.25	0.58	0.37	0.41	0.40	0.01
Broad	Alt3a	0.26	0.83	0.48	0.00	0.39	0.00
Broad	Alt3b	0.00	1.00	1.00	0.32	0.58	0.00
Broad	Alt4	1.00	0.96	0.43	0.44	0.71	0.13

Note: * 1. Utilities for basin-wide outcomes for each of the four decision support models averaged across the three simulation years and associated combined utilities by juvenile salmonid habitat suitability category with combined equal weights and multiplied combined.

* 2. Utilities range from 0 (worst) to 1 (best).

Table 5-4. DSM Relative loss for basin-wide outcomes averaged for 2011, 2015, 2016 and across outcomes. *

Habitat Category	Flow Alternative	Surviving Chinook Redds	Chinook Salmon Adult Equivalents	Age-1 Steelhead Trout	Steelhead Smolt Survival	Average Loss Across Outcomes
Narrow	NAA	0.08	0.00	0.06	0.00	0.04
Narrow	Alt1	0.07	0.18	0.10	0.03	0.09
Narrow	Alt2a	0.08	0.21	0.08	0.02	0.10
Narrow	Alt2b	0.12	0.12	0.06	0.02	0.08
Narrow	Alt3a	0.12	0.06	0.05	0.03	0.07
Narrow	Alt3b	0.17	0.03	0.00	0.02	0.05
Narrow	Alt4	0.00	0.10	0.05	0.01	0.04
Median	NAA	0.08	0.00	0.06	0.00	0.04
Median	Alt1	0.07	0.19	0.09	0.03	0.10
Median	Alt2a	0.08	0.18	0.08	0.02	0.09
Median	Alt2b	0.12	0.18	0.06	0.02	0.09
Median	Alt3a	0.12	0.08	0.05	0.03	0.07
Median	Alt3b	0.17	0.09	0.00	0.02	0.07
Median	Alt4	0.00	0.12	0.05	0.01	0.05
Broad	NAA	0.08	0.00	0.06	0.00	0.04
Broad	Alt1	0.07	0.11	0.10	0.03	0.08
Broad	Alt2a	0.08	0.12	0.09	0.02	0.08
Broad	Alt2b	0.12	0.05	0.07	0.02	0.06
Broad	Alt3a	0.12	0.02	0.05	0.03	0.06
Broad	Alt3b	0.17	0.00	0.00	0.02	0.05
Broad	Alt4	0.00	0.00	0.06	0.01	0.02

Note: * Relative loss for basin-wide outcomes for each of the four decision support models averaged across the three simulation years (2011, 2015, 2016) and the average loss across outcomes. Values are grouped by juvenile salmonid habitat suitability category (narrow, median, and broad).

Table 5-5. 2011 Averaged Tributary outcomes for each DSM by Alternative. *

DSM	Tributary	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Surviving Chinook Redds [1]	McKenzie	9934.1	9846.5	9782.4	9240.2	9850.0	9259.0	9880.6
Surviving Chinook Redds [1]	MF Willamette	121.1	172.9	127.6	136.8	92.7	141.3	109.8
Surviving Chinook Redds [1]	SF Santiam	3539.0	3914.1	2169.3	2222.0	2230.1	4959.9	5346.8
Surviving Chinook Redds [1]	NF Santiam	2229.3	2786.1	3001.6	2945.6	3598.1	2333.3	2960.2
Salmon Adult Equivalents [2]	McKenzie	3551.9	3579.1	3595.2	3593.5	3701.8	3751.5	3209.7
Salmon Adult Equivalents [2]	MF Willamette	321.8	240.7	340.1	285.2	249.4	269.7	262.0
Salmon Adult Equivalents [2]	SF Santiam	1198.9	1197.5	1053.1	896.3	1194.5	1773.5	1209.9
Salmon Adult Equivalents [2]	NF Santiam	1043.1	1002.5	990.4	1014.6	1003.5	1087.7	1021.5
Age-1 Steelhead [3]	SF Santiam	157563.7	130502.4	128146.5	126100.6	118726.8	157186.5	165694.1
Age-1 Steelhead [3]	NF Santiam	245045.5	227161.0	231294.1	250113.0	234094.3	219327.8	241204.6
Steelhead Smolt Survival [4]	SF Santiam	98.0	97.6	97.8	97.8	97.7	97.7	97.9
Steelhead Smolt Survival [4]	NF Santiam	98.3	98.0	98.2	98.2	98.0	97.9	98.1

Notes: Tributary-specific outcomes for each of the four decision support models (DSM) for simulation year 2011 averaged across juvenile salmonid habitat suitability categories.

[1] Chinook redds surviving until swim-up; [2] Chinook salmon adult equivalents; [3] Age-1 Steelhead trout; [4] Steelhead trout smolt survival.

Table 5-6. 2015 Averaged Tributary outcomes for each DSM by Alternative. *

DSM	Tributary	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Surviving Chinook Redds [1]	McKenzie	6945.1	7430.8	7575.6	7236.7	6935	6942.4	7456.7
Surviving Chinook Redds [1]	MF Willamette	8.3	19.9	19.4	12.7	7.8	6.2	13.3
Surviving Chinook Redds [1]	SF Santiam	1740.9	2386.1	2679.7	2630.3	2677.3	5.8	2380.7
Surviving Chinook Redds [1]	NF Santiam	1942.7	1210.6	2009.6	1865.9	1359.3	1788.3	1903.1
Salmon Adult Equivalents [2]	McKenzie	3334.0	2121.9	2184.5	2256.9	3604.9	4002.8	4057.9
Salmon Adult Equivalents [2]	MF Willamette	303.0	313.2	323.1	330.7	294.5	308.1	320.0
Salmon Adult Equivalents [2]	SF Santiam	1619.8	1883.5	2091.9	2448.8	1728.1	1635.2	1735.3
Salmon Adult Equivalents [2]	NF Santiam	1974.5	2276.2	1421.4	1762.0	1913.1	1888.4	2307.9
Age-1 Steelhead [3]	SF Santiam	138124	122819.7	142711.7	155092.5	254735.5	125382.4	158944.1
Age-1 Steelhead [3]	NF Santiam	207665.7	248224.6	241466.4	246517.5	119366.2	301489.7	210303.3
Steelhead Smolt Survival [4]	SF Santiam	89.3	85.3	86.4	86.3	85.9	85.6	86.8
Steelhead Smolt Survival [4]	NF Santiam	89.6	87	88.2	88.1	85.1	89.3	88.1

Notes: * Tributary-specific outcomes for each of the four decision support models for simulation year 2015 averaged across juvenile salmonid habitat suitability categories.

[1] Chinook redds surviving until swim-up; [2] Chinook salmon adult equivalents; [3] Age-1 Steelhead trout; [4] Steelhead trout smolt survival.

Table 5-7. 2016 Averaged Tributary outcomes for each DSM by Alternative. *

DSM	Tributary	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Surviving Redds[1]	McKenzie	8383.5	8116.7	8534.5	7657.7	7596.2	7751.3	8398.3
Surviving Redds[1]	MF Willamette	34.9	39.5	31.6	40.6	19.0	43.0	33.8
Surviving Redds[1]	SF Santiam	2790.1	2476.1	1696.3	1780.0	1692.6	1109.6	2785.1
Surviving Redds[1]	NF Santiam	2213.4	2093.2	2312.5	2268.6	2027.5	1844.4	2196.2
Salmon Adult Equivalents[2]	McKenzie	4887.5	4466.7	4674.6	5039.5	5409.8	5016.5	4939.7
Salmon Adult Equivalents[2]	MF Willamette	322.9	348.9	333.9	347.1	318.4	336.7	322.5
Salmon Adult Equivalents[2]	SF Santiam	2171.7	2064.1	1758.6	2463.4	2107.9	2539.3	2272.6
Salmon Adult Equivalents[2]	NF Santiam	5000.5	2072.5	2541.1	2289.5	2846.1	2102.0	2106.7
Age-1 Steelhead[3]	SF Santiam	148206.9	126394.5	117470.1	116896.8	194178.5	89132.3	147952.8
Age-1 Steelhead[3]	NF Santiam	208761.7	206161.4	215002.8	210674.7	195097.1	285728.1	190058.1
Steelhead Smolt Survival[4]	SF Santiam	91.9	87.4	89.1	89.0	88.3	87.9	89.3
Steelhead Smolt Survival[4]	NF Santiam	92.0	89.1	91.0	90.9	89.2	90.5	90.6

Notes: * Tributary-specific outcomes for each of the four decision support models for simulation year 2015 averaged across juvenile salmonid habitat suitability categories.

[1] Chinook redds surviving until swim-up; [2] Chinook salmon adult equivalents; [3] Age-1 Steelhead trout; [4] Steelhead trout smolt survival.

Table 5-8. 2011 Tributary-specific utilities for each DSM for based on values in Table 4-5. *

DSM	Tributary	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Surviving Chinook Redds [1]	McKenzie	1.00	0.87	0.78	0.00	0.88	0.03	0.92
Surviving Chinook Redds [1]	MF Willamette	0.35	1.00	0.44	0.55	0.00	0.61	0.21
Surviving Chinook Redds [1]	SF Santiam	0.43	0.55	0.00	0.02	0.02	0.88	1.00
Surviving Chinook Redds [1]	NF Santiam	0.00	0.41	0.56	0.52	1.00	0.08	0.53
Salmon Adult Equivalents [2]	McKenzie	0.63	0.68	0.71	0.71	0.91	1.00	0.00
Salmon Adult Equivalents [2]	MF Willamette	0.82	0.00	1.00	0.45	0.09	0.29	0.21
Salmon Adult Equivalents [2]	SF Santiam	0.34	0.34	0.18	0.00	0.34	1.00	0.36
Salmon Adult Equivalents [2]	NF Santiam	0.54	0.12	0.00	0.25	0.13	1.00	0.32
Age-1 Steelhead [3]	SF Santiam	0.83	0.25	0.20	0.16	0.00	0.82	1.00
Age-1 Steelhead [3]	NF Santiam	0.84	0.25	0.39	1.00	0.48	0.00	0.71
Steelhead Smolt Survival [4]	SF Santiam	1.00	0.00	0.50	0.50	0.25	0.25	0.75
Steelhead Smolt Survival [4]	NF Santiam	1.00	0.25	0.75	0.75	0.25	0.00	0.50
Combined utility [5]	All outcomes	0.65	0.39	0.46	0.41	0.36	0.50	0.54
Combined utility [5]	Chinook only	0.51	0.50	0.46	0.31	0.42	0.61	0.45
Combined utility [5]	Steelhead only	0.92	0.19	0.46	0.60	0.24	0.27	0.74

Notes: * Tributary-specific utilities for each of the four decision support models for simulation year 2011 based on values in Table 4-5.

[1] Chinook redds surviving until swim-up; [2] Chinook salmon adult equivalents; [3] Age-1 Steelhead trout; [4] Steelhead trout smolt survival; [5] Combined utility (equal weight)

Table 5-9. 2015 Tributary-specific utilities for each DSM for based on values in Table 4-6. *

DSM	Tributary	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Surviving Chinook Redds [1]	McKenzie	0.02	0.77	1.00	0.47	0.00	0.01	0.81
Surviving Chinook Redds [1]	MF Willamette	0.15	1.00	0.96	0.47	0.12	0.00	0.52
Surviving Chinook Redds [1]	SF Santiam	0.65	0.89	1.00	0.98	1.00	0.00	0.89
Surviving Chinook Redds [1]	NF Santiam	0.92	0.00	1.00	0.82	0.19	0.72	0.87
Salmon Adult Equivalents [2]	McKenzie	0.63	0.00	0.03	0.07	0.77	0.97	1.00
Salmon Adult Equivalents [2]	MF Willamette	0.24	0.52	0.79	1.00	0.00	0.37	0.70
Salmon Adult Equivalents [2]	SF Santiam	0.00	0.32	0.57	1.00	0.13	0.02	0.14
Salmon Adult Equivalents [2]	NF Santiam	0.62	0.96	0.00	0.38	0.55	0.53	1.00
Age-1 Steelhead [3]	SF Santiam	0.12	0.00	0.15	0.24	1.00	0.02	0.27
Age-1 Steelhead [3]	NF Santiam	0.48	0.71	0.67	0.70	0.00	1.00	0.50
Steelhead Smolt Survival [4]	SF Santiam	1.00	0.00	0.28	0.25	0.15	0.07	0.38
Steelhead Smolt Survival [4]	NF Santiam	1.00	0.42	0.69	0.67	0.00	0.93	0.67
Combined utility [5]	All outcomes	0.49	0.47	0.60	0.59	0.33	0.39	0.65
Combined utility [5]	Chinook only	0.40	0.56	0.67	0.65	0.34	0.33	0.74
Combined utility [5]	Steelhead only	0.65	0.28	0.45	0.46	0.29	0.51	0.45

*Note: * Tributary-specific utilities for each of the four decision support models for simulation year 2015 based on values in Table 4-6.

[1] Chinook redds surviving until swim-up; [2] Chinook salmon adult equivalents; [3] Age-1 Steelhead trout; [4] Steelhead trout smolt survival; [5] Combined utility (equal weight)

Table 5-10. 2016 Tributary-specific utilities for each DSM for based on values in Table 4-7. *

DSM	Tributary	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Surviving Chinook Redds [1]	McKenzie	0.84	0.55	1.00	0.07	0.00	0.17	0.85
Surviving Chinook Redds [1]	MF Willamette	0.66	0.85	0.53	0.90	0.00	1.00	0.62
Surviving Chinook Redds [1]	SF Santiam	1.00	0.81	0.35	0.40	0.35	0.00	1.00
Surviving Chinook Redds [1]	NF Santiam	0.79	0.53	1.00	0.91	0.39	0.00	0.75
Salmon Adult Equivalents [2]	McKenzie	0.45	0.00	0.22	0.61	1.00	0.58	0.50
Salmon Adult Equivalents [2]	MF Willamette	0.15	1.00	0.51	0.94	0.00	0.60	0.14
Salmon Adult Equivalents [2]	SF Santiam	0.53	0.39	0.00	0.90	0.45	1.00	0.66
Salmon Adult Equivalents [2]	NF Santiam	1.00	0.00	0.16	0.07	0.26	0.01	0.01
Age-1 Steelhead [3]	SF Santiam	0.56	0.35	0.27	0.26	1.00	0.00	0.56
Age-1 Steelhead [3]	NF Santiam	0.20	0.17	0.26	0.22	0.05	1.00	0.00
Steelhead Smolt Survival [4]	SF Santiam	1.00	0.00	0.38	0.36	0.20	0.11	0.42
Steelhead Smolt Survival [4]	NF Santiam	1.00	0.00	0.66	0.62	0.03	0.48	0.52
Combined utility [5]	All outcomes	0.68	0.39	0.44	0.52	0.31	0.41	0.50
Combined utility [5]	Chinook only	0.68	0.52	0.47	0.60	0.31	0.42	0.57
Combined utility [5]	Steelhead only	0.69	0.13	0.39	0.36	0.32	0.40	0.37

*Note: * Tributary-specific utilities for each of the four decision support models for simulation year 2016 based on values in Table 4-7.

[1] Chinook redds surviving until swim-up; [2] Chinook salmon adult equivalents; [3] Age-1 Steelhead trout; [4] Steelhead trout smolt survival; [5] Combined utility (equal weight)

Table 5-11. 2011 Tributary-specific utilities for each DSM for based on values in Table 4-5. *

DSM	Tributary	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Surviving Chinook Redds [1]	McKenzie	0.00	0.01	0.02	0.07	0.01	0.07	0.01
Surviving Chinook Redds [1]	MF Willamette	0.30	0.00	0.26	0.21	0.46	0.18	0.36
Surviving Chinook Redds [1]	SF Santiam	0.34	0.27	0.59	0.58	0.58	0.07	0.00
Surviving Chinook Redds [1]	NF Santiam	0.38	0.23	0.17	0.18	0.00	0.35	0.18
Salmon Adult Equivalents [2]	McKenzie	0.05	0.05	0.04	0.04	0.01	0.00	0.14
Salmon Adult Equivalents [2]	MF Willamette	0.05	0.29	0.00	0.16	0.27	0.21	0.23
Salmon Adult Equivalents [2]	SF Santiam	0.32	0.32	0.41	0.49	0.33	0.00	0.32
Salmon Adult Equivalents [2]	NF Santiam	0.04	0.08	0.09	0.07	0.08	0.00	0.06
Age-1 Steelhead [3]	SF Santiam	0.05	0.21	0.23	0.24	0.28	0.05	0.00
Age-1 Steelhead [3]	NF Santiam	0.02	0.09	0.08	0.00	0.06	0.12	0.04
Steelhead Smolt Survival [4]	SF Santiam	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Steelhead Smolt Survival [4]	NF Santiam	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Average Relative Loss	All outcomes	0.13	0.13	0.16	0.17	0.17	0.09	0.11
Average Relative Loss	Chinook only	0.19	0.16	0.20	0.23	0.22	0.11	0.16
Average Relative Loss	Steelhead only	0.02	0.08	0.08	0.06	0.09	0.05	0.01

*Note: * Tributary-specific utilities for each of the four decision support models for simulation year 2011 based on values in Table 4-5.

[1] Chinook redds surviving until swim-up; [2] Chinook salmon adult equivalents; [3] Age-1 Steelhead trout; [4] Steelhead trout smolt survival

Table 5-12. 2015 Tributary-specific utilities for each DSM for based on values in Table 4-6. *

DSM	Tributary	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Surviving Chinook Redds [1]	McKenzie	0.08	0.02	0.00	0.04	0.08	0.08	0.02
Surviving Chinook Redds [1]	MF Willamette	0.58	0.00	0.03	0.36	0.61	0.69	0.33
Surviving Chinook Redds [1]	SF Santiam	0.35	0.11	0.00	0.02	0.00	1.00	0.11
Surviving Chinook Redds [1]	NF Santiam	0.03	0.40	0.00	0.07	0.32	0.11	0.05
Salmon Adult Equivalents [2]	McKenzie	0.18	0.48	0.46	0.44	0.11	0.01	0.00
Salmon Adult Equivalents [2]	MF Willamette	0.08	0.05	0.02	0.00	0.11	0.07	0.03
Salmon Adult Equivalents [2]	SF Santiam	0.34	0.23	0.15	0.00	0.29	0.33	0.29
Salmon Adult Equivalents [2]	NF Santiam	0.14	0.01	0.38	0.24	0.17	0.18	0.00
Age-1 Steelhead [3]	SF Santiam	0.46	0.52	0.44	0.39	0.00	0.51	0.38
Age-1 Steelhead [3]	NF Santiam	0.31	0.18	0.20	0.18	0.60	0.00	0.30
Steelhead Smolt Survival [4]	SF Santiam	0.00	0.04	0.03	0.03	0.04	0.04	0.03
Steelhead Smolt Survival [4]	NF Santiam	0.00	0.03	0.02	0.02	0.05	0.00	0.02
Average relative loss	All outcomes	0.21	0.17	0.14	0.15	0.20	0.25	0.13
Average relative loss	Chinook only	0.22	0.16	0.13	0.15	0.21	0.31	0.10
Average relative loss	Steelhead Only	0.19	0.19	0.17	0.16	0.17	0.14	0.18

Notes: * Tributary-specific utilities for each of the four decision support models for simulation year 2015 based on values in Table 4-6.

[1] Chinook redds surviving until swim-up; [2]Chinook salmon adult equivalents; [3] Age-1 Steelhead trout; [4] Steelhead trout smolt survival

Table 5-13. 2016 Tributary-specific utilities for each DSM for based on values in Table 4-7. *

DSM	Tributary	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
Surviving Chinook Redds [1]	McKenzie	0.02	0.05	0.00	0.10	0.11	0.09	0.02
Surviving Chinook Redds [1]	MF Willamette	0.19	0.08	0.27	0.06	0.56	0.00	0.21
Surviving Chinook Redds [1]	SF Santiam	0.00	0.11	0.39	0.36	0.39	0.60	0.00
Surviving Chinook Redds [1]	NF Santiam	0.04	0.09	0.00	0.02	0.12	0.20	0.05
Salmon Adult Equivalents [2]	McKenzie	0.10	0.17	0.14	0.07	0.00	0.07	0.09
Salmon Adult Equivalents [2]	MF Willamette	0.07	0.00	0.04	0.01	0.09	0.03	0.08
Salmon Adult Equivalents [2]	SF Santiam	0.14	0.19	0.31	0.03	0.17	0.00	0.11
Salmon Adult Equivalents [2]	NF Santiam	0.00	0.59	0.49	0.54	0.43	0.58	0.58
Age-1 Steelhead [3]	SF Santiam	0.24	0.35	0.40	0.40	0.00	0.54	0.24
Age-1 Steelhead [3]	NF Santiam	0.27	0.28	0.25	0.26	0.32	0.00	0.33
Steelhead Smolt Survival [4]	SF Santiam	0.00	0.05	0.03	0.03	0.04	0.04	0.03
Steelhead Smolt Survival [4]	NF Santiam	0.00	0.03	0.01	0.01	0.03	0.02	0.02
Average relative loss	All outcomes	0.09	0.17	0.19	0.16	0.19	0.18	0.15
Average relative loss	Chinook only	0.07	0.16	0.20	0.15	0.23	0.20	0.14
Average relative loss	Steelhead only	0.13	0.18	0.17	0.18	0.10	0.15	0.15

Notes: * Tributary-specific utilities for each of the four decision support models for simulation year 2016 based on values in Table 4-7.

[1] Chinook redds surviving until swim-up; [2] Chinook salmon adult equivalents; [3] Age-1 Steelhead trout; [4] Steelhead trout smolt survival

Table 5-14. NAA compared to action alternatives for redds[1] and Steelhead smolt survival[2] *

Alternative	Year	Redds surviving[1] McKenzie	Redds surviving[1] MF Willamette	Redds surviving[1] SF Santiam	Redds surviving[1] NF Santiam	Steelhead smolts surviving[2] SF Santiam	Steelhead smolts surviving[2] NF Santiam
Alt1	2011	-87.6	51.8	375.1	556.8	-0.4	-0.3
Alt1	2015	485.7	11.6	645.2	-732.1	-4.0	-2.6
Alt1	2016	-266.8	4.6	-314.0	-120.2	-4.5	-2.9
Alt2a	2011	-151.7	6.5	-1369.7	772.3	-0.2	-0.1
Alt2a	2015	630.5	11.1	938.8	66.9	-2.9	-1.4
Alt2a	2016	151.0	-3.3	-1093.8	99.1	-2.8	-1.0
Alt2b	2011	-693.9	15.7	-1317.0	716.3	-0.2	-0.1
Alt2b	2015	291.6	4.4	889.4	-76.8	-3.0	-1.5
Alt2b	2016	-725.8	5.7	-1010.1	55.2	-2.9	-1.1
Alt3a	2011	-84.1	-28.4	-1308.9	1368.8	-0.3	-0.3
Alt3a	2015	-10.1	-0.5	936.4	-583.4	-3.4	-4.5
Alt3a	2016	-787.3	-15.9	-1097.5	-185.9	-3.6	-2.8
Alt3b	2011	-675.1	20.2	1420.9	104.0	-0.3	-0.4
Alt3b	2015	-2.7	-2.1	-1735.1	-154.4	-3.7	-0.3
Alt3b	2016	-632.2	8.1	-1680.5	-369.0	-4.0	-1.5
Alt4	2011	-53.5	-11.3	1807.8	730.9	-0.1	-0.2
Alt4	2015	511.6	5.0	639.8	-39.6	-2.5	-1.5
Alt4	2016	14.8	-1.1	-5.0	-17.2	-2.6	-1.4

Notes: * The magnitude of the difference between predictions under the no action alternative (NAA) and each alternative (i.e., alternative minus NAA estimate) for Chinook salmon redds surviving until swim-up and Steelhead smolt survival to Willamette Falls Dam

[1] Chinook salmon redds surviving until swim-up; [2] Steelhead trout smolts surviving to Willamette Falls Dam.

Table 5-15. Comparing NAA and each Action alternative for Chinook adult equivalents by habitat and tributary. *

Alternative	Year	McKenzie Narrow**	McKenzie Median**	McKenzie Broad**	MF Willamette Narrow**	MF Willamette Median**	MF Willamette Broad**	SF Santiam Narrow**	SF Santiam Median**	SF Santiam Broad**	NF Santiam Narrow**	NF Santiam Median**	NSF Santiam Broad**
Alt1	2011	-68.7	-31.4	181.7	-63	-92.4	-87.8	8.4	-15.5	2.8	414.9	-463.6	-73.2
Alt1	2015	-1241.6	-1207.6	-1187.1	-48.8	5.5	73.9	542	-430.7	679.8	675.4	211.9	17.9
Alt1	2016	-370	-151.6	-740.9	36.5	7.6	34	-64.8	-97	-161.1	-4482.2	-2856.6	-1445.2
Alt2a	2011	-89.9	71.1	148.7	-6.1	62	-1	-10.8	-422.6	-3.9	-38.3	-34.3	-85.4
Alt2a	2015	-1143.4	-1094.1	-1211	-15.4	36.1	39.5	1124	151.7	140.4	-293.3	-891.3	-474.7
Alt2a	2016	-162.6	17.2	-493.2	30.4	4.9	-2.2	-1084.4	-47.4	-107.7	-3783	-2716.5	-878.7
Alt2b	2011	-44.8	22.5	146.9	-49.6	-55.5	-4.8	6	-464.2	-449.6	2.9	2.7	-91.1
Alt2b	2015	-1114.5	-1031.9	-1084.8	-19	32.2	69.7	1180.2	106.5	1200.2	162.3	-846.2	46.4
Alt2b	2016	162.7	452	-158.8	48.7	23.4	0.6	497.5	447.1	-69.5	-3930	-3388.2	-814.7
Alt3a	2011	157.4	94.9	197.3	-69.7	-73.6	-73.8	461.7	-23.2	-451.8	213.9	48.9	-381.5
Alt3a	2015	321.4	222.3	269.2	-25.5	-32.8	32.8	109.5	-457.9	673.1	656.7	-819.1	-21.7
Alt3a	2016	380.2	1152.5	34.2	4.5	-20.8	2.8	-40.1	-40	-111.5	-3670.2	-2139.7	-653.1
Alt3b	2011	128	207.7	263	-62.8	-37.1	-56.5	871.7	415.2	437	88.8	24.4	20.7
Alt3b	2015	705.3	751.9	549.3	-1.2	5.3	11	446.2	-473	73	43.3	-405.4	104
Alt3b	2016	228.1	268.3	-109.4	33.1	8.8	-0.3	623.9	246.9	232	-3842	-3351.4	-1502
Alt4	2011	-349.9	-404	-272.8	-59.7	-64.3	-55.5	460	-455.1	28.1	9.3	-48.5	-25.7
Alt4	2015	774.5	747.9	649.3	-20.5	2.4	68.9	638.1	-418.5	126.9	117.6	161.7	721
Alt4	2016	230.2	137.9	-211.6	38.5	-17.5	-22	13.2	-63.2	352.5	-4382.8	-2852.8	-1445.6

Notes: *The magnitude of the difference between predictions under the no action alternative (NAA) and each alternative (i.e., alternative minus NAA estimate) for Chinook salmon adult equivalents by habitat definition and tributary.

** Narrow, Median and Broad refer to juvenile salmonid habitat suitability categories, or habitat definition

Table 5-16. Comparing NAA And Action Alts. For Age-1 Steelhead by Habitat and Tributary. *

Alternative	Year	SF Santiam Narrow**	SF Santiam Median**	SF Santiam Broad**	NF Santiam Narrow**	NF Santiam Median**	NF Santiam Broad**
Alt1	2011	-28218.9	-27057.8	-25907.4	-18776.6	-15478.8	-19398.1
Alt1	2015	-18097.1	-13654.2	-14161.5	43925.6	37812.9	39938.2
Alt1	2016	-26636.9	-20042	-18758.3	3588.1	-2406.2	-8982.7
Alt2a	2011	-30360.4	-29678.6	-28212.6	-14546.8	-10106.7	-16600.5
Alt2a	2015	3276.5	4391.9	6094.6	39703	30411	31288.3
Alt2a	2016	-31173.7	-30730.6	-30306.1	6685.8	6013.4	6024.3
Alt2b	2011	-32719.2	-31909	-29761.1	5865.3	8002.7	1334.6
Alt2b	2015	14189.6	18798.1	17917.9	42000.3	37064.1	37491.2
Alt2b	2016	-31564.9	-31262.5	-31102.9	3430.4	213.6	2095
Alt3a	2011	-39897.3	-38099.1	-38514.5	-12324.6	-8022.8	-12506.2
Alt3a	2015	115099.5	115761.5	118973.5	-84657.5	-90623.4	-89617.4
Alt3a	2016	45226.8	45215.6	47472.4	-12747.4	-14093.3	-14152.9
Alt3b	2011	-1152.7	1171.2	-1150.3	-26026.8	-25017.2	-26109.1
Alt3b	2015	-16023.2	-11609.2	-10592.5	97479.5	92473.9	91518.8
Alt3b	2016	-59316.6	-60135.9	-57771.3	78361.1	73055.7	79482.4
Alt4	2011	8552.8	7511.7	8326.6	-4631	-463.7	-6427.8
Alt4	2015	15862.4	23418.3	23179.6	10357.3	-852.2	-1592.3
Alt4	2016	-1336.6	-215.6	789.9	-18554.7	-18477.6	-19078.3

Notes: * The magnitude of the difference between predictions under the no action alternative (NAA) and each alternative (i.e., alternative minus NAA estimate) for age-1 Steelhead trout by habitat definition and tributary.

** Narrow, Median and Broad refer to juvenile salmonid habitat suitability categories, or habitat definition

5.6 FIGURES

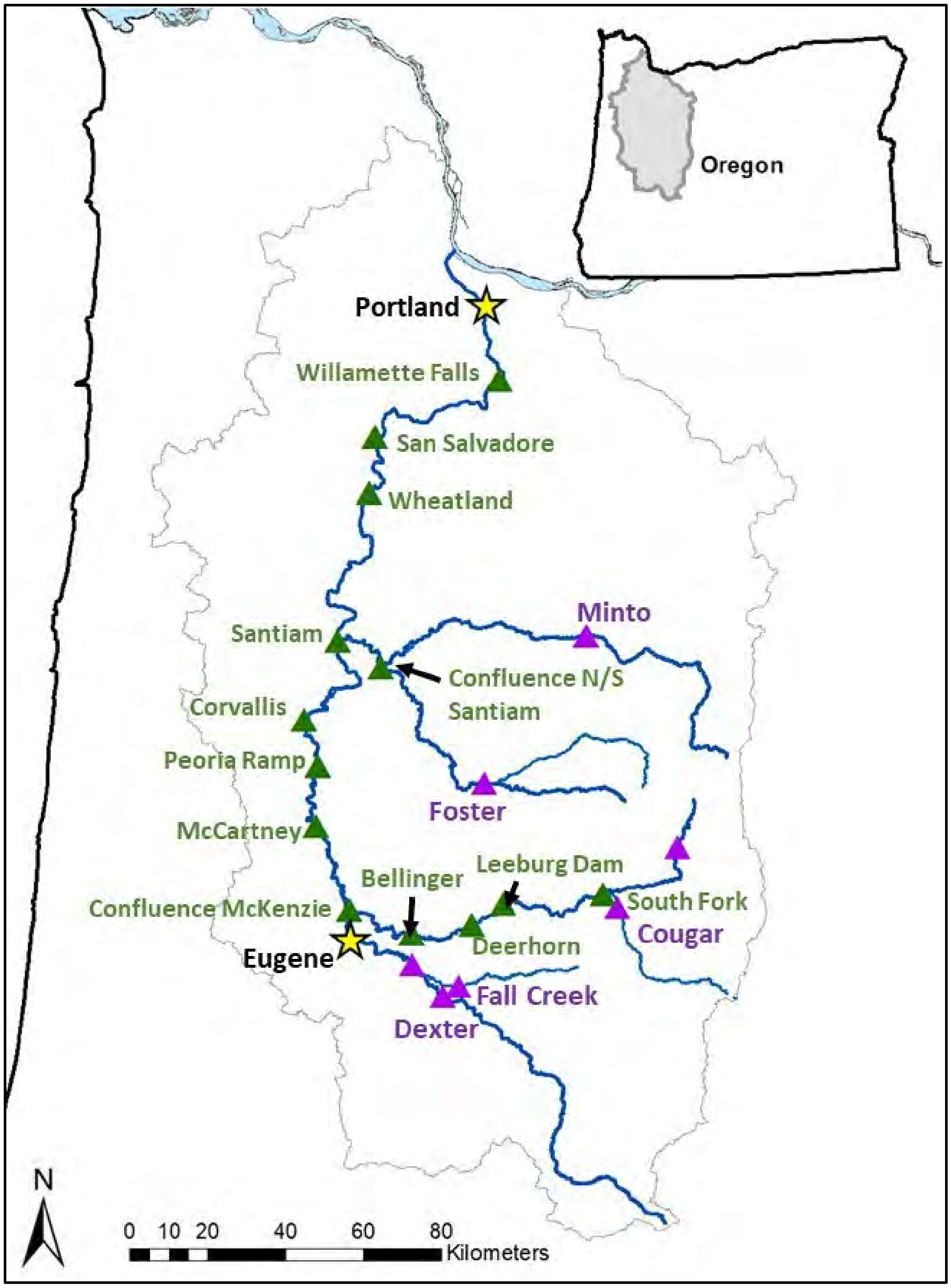

Figure 5-1. The upper Willamette Basin with section starts and USACE Projects. *The upper Willamette Basin with section starts (green triangles) and US Army Corps of Engineers Projects (purple triangles). Location of major cities in the basin are shown as yellow stars (reproduced from Peterson et al. 2021).*

5.7 CHAPTER 5 ATTACHMENT A: TRIBUTARY SPECIFIC ESTIMATES FOR EACH FLOW MANAGEMENT ALTERNATIVE BY JUVENILE HABITAT CATEGORY.

Table 5-17. Mean Chinook Outcomes by Alternative, Year, and Narrow Habitat Definition *

Alternative	Year	Redds surviving[1] McKenzie	Redds surviving[1] MF Willamette	Redds surviving[1] SF Santiam	Redds surviving[1] NF Santiam	Chinook adult equivalents[2] McKenzie	Chinook adult equivalents[2] MF Willamette	Chinook adult equivalents[2] SF Santiam	Chinook adult equivalents[2] NF Santiam
NAA	2011	9934.1	121.1	3539.0	2229.3	3615.3	322.4	901.8	1007.5
NAA	2015	6945.1	8.3	1740.9	1942.7	3337.1	331.5	1466.3	1776.9
NAA	2016	8383.5	34.9	2790.1	2213.4	4874.9	302.1	2129.5	6360.0
Alt1	2011	9846.5	172.9	3914.1	2786.1	3546.6	259.4	910.2	1422.4
Alt1	2015	7430.8	19.9	2386.1	1210.6	2095.5	282.7	2008.3	2452.3
Alt1	2016	8116.7	39.5	2476.1	2093.2	4504.9	338.6	2064.7	1877.8
Alt2a	2011	9782.4	127.6	2169.3	3001.6	3525.4	316.3	891.0	969.2
Alt2a	2015	7575.6	19.4	2679.7	2009.6	2193.7	316.1	2590.3	1483.6
Alt2a	2016	8534.5	31.6	1696.3	2312.5	4712.3	332.5	1045.1	2577.0
Alt2b	2011	9240.2	136.8	2222.0	2945.6	3570.5	272.8	907.8	1010.4
Alt2b	2015	7236.7	12.7	2630.3	1865.9	2222.6	312.5	2646.5	1939.2
Alt2b	2016	7657.7	40.6	1780.0	2268.6	5037.6	350.8	2627.0	2430.0
Alt3a	2011	9850.0	92.7	2230.1	3598.1	3772.7	252.7	1363.5	1221.4
Alt3a	2015	6935.0	7.8	2677.3	1359.3	3658.5	306.0	1575.8	2433.6
Alt3a	2016	7596.2	19.0	1692.6	2027.5	5255.1	306.6	2089.4	2689.8
Alt3b	2011	9259.0	141.3	4959.9	2333.3	3743.3	259.6	1773.5	1096.3
Alt3b	2015	6942.4	6.2	5.8	1788.3	4042.4	330.3	1912.5	1820.2
Alt3b	2016	7751.3	43.0	1109.6	1844.4	5103.0	335.2	2753.4	2518.0
Alt4	2011	9880.6	109.8	5346.8	2960.2	3265.4	262.7	1361.8	1016.8
Alt4	2015	7456.7	13.3	2380.7	1903.1	4111.6	311.0	2104.4	1894.5
Alt4	2016	8398.3	33.8	2785.1	2196.2	5105.1	340.6	2142.7	1977.2

Notes: * Estimated mean outcomes under the seven flow management alternatives by year using the narrow habitat definition.

[1]Chinook salmon redds surviving until swim-up; [2]chinook Salmon Adult Equivalents.

Table 5-18. Mean Steelhead Outcomes by Alternative, Year, and Narrow Habitat Definition *

Alternative	Year	Age-1 Steelhead[1] SF Santiam	Age-1 Steelhead[1] NF Santiam	Steelhead smolts surviving[2] SF Santiam	Steelhead smolts surviving[2] NF Santiam
NAA	2011	158560.2	244259.5	98.0	98.3
NAA	2015	140566.1	204113.7	89.3	89.6
NAA	2016	148749.7	207320.6	91.9	92.0
Alt1	2011	130341.3	225482.9	97.6	98.0
Alt1	2015	122469.0	248039.3	85.3	87.0
Alt1	2016	122112.8	210908.7	87.4	89.1
Alt2a	2011	128199.8	229712.7	97.8	98.2
Alt2a	2015	143842.6	243816.7	86.4	88.2
Alt2a	2016	117576.0	214006.4	89.1	91.0
Alt2b	2011	125841.0	250124.8	97.8	98.2
Alt2b	2015	154755.7	246114.0	86.3	88.1
Alt2b	2016	117184.8	210751.0	89.0	90.9
Alt3a	2011	118662.9	231934.9	97.7	98.0
Alt3a	2015	255665.6	119456.2	85.9	85.1
Alt3a	2016	193976.5	194573.2	88.3	89.2
Alt3b	2011	157407.5	218232.7	97.7	97.9
Alt3b	2015	124542.9	301593.2	85.6	89.3
Alt3b	2016	89433.1	285681.7	87.9	90.5
Alt4	2011	167113.0	239628.5	97.9	98.1
Alt4	2015	156428.5	214471.0	86.8	88.1
Alt4	2016	147413.1	188765.9	89.3	90.6

Notes: * Estimated mean outcomes under the seven flow management alternatives by year with the narrow habitat definition

[1] Age-1 Steelhead trout; [2]Steelhead trout smolt survival

Table 5-19. Mean Chinook Outcomes by Alternative, Year, and Narrow Habitat Definition *

Alternative	Year	Redds Surviving[1] Mckenzie	Redds Surviving[1] MF Willamette	Redds Surviving[1] SF Santiam	Redds Surviving[1] NF Santiam	Chinook Adult Equivalents[2] Mckenzie	Chinook Adult Equivalents[2] MF Willamette	Chinook Adult Equivalents[2] SF Santiam	Chinook Adult Equivalents[2] NF Santiam
NAA	2011	9934.1	121.1	3539.0	2229.3	3571.3	322.8	1347.6	1032.2
NAA	2015	6945.1	8.3	1740.9	1942.7	3298.1	307.5	1981.5	2273.0
NAA	2016	8383.5	34.9	2790.1	2213.4	4674.0	332.6	2171.4	5275.8
Alt1	2011	9846.5	172.9	3914.1	2786.1	3539.9	230.4	1332.1	568.6
Alt1	2015	7430.8	19.9	2386.1	1210.6	2090.5	313.0	1550.8	2484.9
Alt1	2016	8116.7	39.5	2476.1	2093.2	4522.4	340.2	2074.4	2419.2
Alt2a	2011	9782.4	127.6	2169.3	3001.6	3642.4	384.8	925.0	997.9
Alt2a	2015	7575.6	19.4	2679.7	2009.6	2204.0	343.6	2133.2	1381.7
Alt2a	2016	8534.5	31.6	1696.3	2312.5	4691.2	337.5	2124.0	2559.3
Alt2b	2011	9240.2	136.8	2222.0	2945.6	3593.8	267.3	883.4	1034.9
Alt2b	2015	7236.7	12.7	2630.3	1865.9	2266.2	339.7	2088.0	1426.8
Alt2b	2016	7657.7	40.6	1780.0	2268.6	5126.0	356.0	2618.5	1887.6
Alt3a	2011	9850.0	92.7	2230.1	3598.1	3666.2	249.2	1324.4	1081.1
Alt3a	2015	6935.0	7.8	2677.3	1359.3	3520.4	274.7	1523.6	1453.9
Alt3a	2016	7596.2	19.0	1692.6	2027.5	5826.5	311.8	2131.4	3136.1
Alt3b	2011	9259.0	141.3	4959.9	2333.3	3779.0	285.7	1762.8	1056.6
Alt3b	2015	6942.4	6.2	5.8	1788.3	4050.0	312.8	1508.5	1867.6
Alt3b	2016	7751.3	43.0	1109.6	1844.4	4942.3	341.4	2418.3	1924.4
Alt4	2011	9880.6	109.8	5346.8	2960.2	3167.3	258.5	892.5	983.7
Alt4	2015	7456.7	13.3	2380.7	1903.1	4046.0	309.9	1563.0	2434.7
Alt4	2016	8398.3	33.8	2785.1	2196.2	4811.9	315.1	2108.2	2423.0

Notes: * Estimated mean outcomes under the seven flow management alternatives by year using the median habitat definition.

[1]Chinook salmon redds surviving until swim-up; [2]chinook Salmon Adult Equivalents.

Table 5-20. Mean Steelhead Outcomes by Alternative, Year, and Median Habitat Definition *

Alternative	Year	Age-1 Steelhead[1] SF Santiam	Age-1 Steelhead[1] NF Santiam	Steelhead Smolts Surviving[2] SF Santiam	Steelhead Smolts Surviving[2] NF Santiam
NAA	2011	157485.3	242693.4	98.0	98.3
NAA	2015	137000.5	209659.7	89.3	89.6
NAA	2016	148689.9	208796.5	91.9	92.0
Alt1	2011	130427.5	227214.6	97.6	98.0
Alt1	2015	123346.3	247472.6	85.3	87.0
Alt1	2016	128647.9	206390.3	87.4	89.1
Alt2a	2011	127806.7	232586.7	97.8	98.2
Alt2a	2015	141392.4	240070.7	86.4	88.2
Alt2a	2016	117959.3	214809.9	89.1	91.0
Alt2b	2011	125576.3	250696.1	97.8	98.2
Alt2b	2015	155798.6	246723.8	86.3	88.1
Alt2b	2016	117427.4	209010.1	89.0	90.9
Alt3a	2011	119386.2	234670.6	97.7	98.0
Alt3a	2015	252762.0	119036.3	85.9	85.1
Alt3a	2016	193905.5	194703.2	88.3	89.2
Alt3b	2011	158656.5	217676.2	97.7	97.9
Alt3b	2015	125391.3	302133.6	85.6	89.3
Alt3b	2016	88554.0	281852.2	87.9	90.5
Alt4	2011	164997.0	242229.7	97.9	98.1
Alt4	2015	160418.8	208807.5	86.8	88.1
Alt4	2016	148474.3	190318.9	89.3	90.6

Notes: * Estimated mean outcomes under the seven flow management alternatives by year with the median habitat definition

[1] Age-1 Steelhead trout; [2]Steelhead trout smolt survival

Table 5-21. Mean Chinook Outcomes by Alternative, Year, and Broad Habitat Definition *

Alternative	Year	Redds Surviving[1] Mckenzie	Redds Surviving[1] MF Willamette	Redds Surviving[1] SF Santiam	Redds Surviving[1] NF Santiam	Chinook Adult Equivalents[2] Mckenzie	Chinook Adult Equivalents[2] MF Willamette	Chinook Adult Equivalents[2] SF Santiam	Chinook Adult Equivalents[2] NF Santiam
NAA	2011	9934.1	121.1	3539.0	2229.3	3469.2	320.2	1347.3	1089.6
NAA	2015	6945.1	8.3	1740.9	1942.7	3366.7	270.1	1411.7	1873.5
NAA	2016	8383.5	34.9	2790.1	2213.4	5113.6	333.9	2214.3	3365.6
Alt1	2011	9846.5	172.9	3914.1	2786.1	3650.9	232.4	1350.1	1016.4
Alt1	2015	7430.8	19.9	2386.1	1210.6	2179.6	344.0	2091.5	1891.4
Alt1	2016	8116.7	39.5	2476.1	2093.2	4372.7	367.9	2053.2	1920.4
Alt2a	2011	9782.4	127.6	2169.3	3001.6	3617.9	319.2	1343.4	1004.2
Alt2a	2015	7575.6	19.4	2679.7	2009.6	2155.7	309.6	1552.1	1398.8
Alt2a	2016	8534.5	31.6	1696.3	2312.5	4620.4	331.7	2106.6	2486.9
Alt2b	2011	9240.2	136.8	2222.0	2945.6	3616.1	315.4	897.7	998.5
Alt2b	2015	7236.7	12.7	2630.3	1865.9	2281.9	339.8	2611.9	1919.9
Alt2b	2016	7657.7	40.6	1780.0	2268.6	4954.8	334.5	2144.8	2550.9
Alt3a	2011	9850.0	92.7	2230.1	3598.1	3666.5	246.4	895.5	708.1
Alt3a	2015	6935.0	7.8	2677.3	1359.3	3635.9	302.9	2084.8	1851.8
Alt3a	2016	7596.2	19.0	1692.6	2027.5	5147.8	336.7	2102.8	2712.5
Alt3b	2011	9259.0	141.3	4959.9	2333.3	3732.2	263.7	1784.3	1110.3
Alt3b	2015	6942.4	6.2	5.8	1788.3	3916.0	281.1	1484.7	1977.5
Alt3b	2016	7751.3	43.0	1109.6	1844.4	5004.2	333.6	2446.3	1863.6
Alt4	2011	9880.6	109.8	5346.8	2960.2	3196.4	264.7	1375.4	1063.9
Alt4	2015	7456.7	13.3	2380.7	1903.1	4016.0	339.0	1538.6	2594.5
Alt4	2016	8398.3	33.8	2785.1	2196.2	4902.0	311.9	2566.8	1920.0

Notes: * Estimated mean outcomes under the seven flow management alternatives by year using the broad habitat definition.

[1]Chinook salmon redds surviving until swim-up; [2]chinook Salmon Adult Equivalents.

Table 5-22. Mean Steelhead Outcomes by Alternative, Year, and Broad Habitat Definition *

Alternative	Year	Age-1 Steelhead[1] SF Santiam	Age-1 Steelhead[1] NF Santiam	Steelhead Smolts Surviving[2] SF Santiam	Steelhead Smolts Surviving[2] NF Santiam
NAA	2011	156645.7	248183.5	98.0	98.3
NAA	2015	136805.4	209223.6	89.3	89.6
NAA	2016	147181.0	210167.9	91.9	92.0
Alt1	2011	130738.3	228785.4	97.6	98.0
Alt1	2015	122643.9	249161.8	85.3	87.0
Alt1	2016	128422.7	201185.2	87.4	89.1
Alt2a	2011	128433.1	231583.0	97.8	98.2
Alt2a	2015	142900.0	240511.9	86.4	88.2
Alt2a	2016	116874.9	216192.2	89.1	91.0
Alt2b	2011	126884.6	249518.1	97.8	98.2
Alt2b	2015	154723.3	246714.8	86.3	88.1
Alt2b	2016	116078.1	212262.9	89.0	90.9
Alt3a	2011	118131.2	235677.3	97.7	98.0
Alt3a	2015	255778.9	119606.2	85.9	85.1
Alt3a	2016	194653.4	196015.0	88.3	89.2
Alt3b	2011	155495.4	222074.4	97.7	97.9
Alt3b	2015	126212.9	300742.4	85.6	89.3
Alt3b	2016	89409.7	289650.3	87.9	90.5
Alt4	2011	164972.3	241755.7	97.9	98.1
Alt4	2015	159985.0	207631.3	86.8	88.1
Alt4	2016	147970.9	191089.6	89.3	90.6

Notes: * Estimated mean outcomes under the seven flow management alternatives by year with the broad habitat definition

[1] Age-1 Steelhead trout; [2]Steelhead trout smolt survival

CHAPTER 6 - ECOSYSTEM DIAGNOSIS AND TREATMENT

Final

ECOSYSTEM DIAGNOSIS & TREATMENT TO SUPPORT THE WILLAMETTE VALLEY SYSTEMS ENVIRONMENTAL IMPACT STATEMENT

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Contents

Acronyms and Abbreviations

Abbreviation ..... Definition

AltX.................... Alternative (1, 2a, 2b, 3a, 3b, 4)
AWQMS............. Ambient Water Quality Monitoring System
BLM .... Bureau of Land Management
DO .... Dissolved Oxygen
EDT .... Ecosystem Diagnosis & Treatment
EDT2 .... Ecosystem Diagnosis & Treatment, Version 2
EDT3 .... Ecosystem Diagnosis & Treatment, Version 3
EIS....................... Environmental Impact Statement
ESA .... Endangered Species Act
FBW .... Fish Benefit Workbook
HARD .... Habitat Attribute Reach Dataset
B-IBI.................... Benthic Index of Biotic Integrity
ICF....................... ICF International, Inc.
MFW................... Middle Fork Willamette
NAA .... No Action Alternative
ODFW .... Oregon Department of Fish and Wildlife
ORDEQ................ Oregon Department of Environmental Quality
PREDATOR.......... Predictive Assessment Tool for Oregon
R2 .... R2 Resource Consultants, Inc.
TDG..................... Total Dissolved Gas
TSS...................... Total Suspended Solid
USACE................. United States Army Corps of Engineers
USFS .... United States Forest Service
USGS................... United States Geological Survey
WVS.................... Willamette Valley System
WY .... Water Year

6.1 INTRODUCTION

The Willamette Basin is Oregon's largest river basin and is composed of a network of streams and rivers that drain the western slopes of Cascade Mountains and the eastern slopes of the Oregon Coast Range. Flowing approximately 300 miles from headwaters to the mouth, the mainstream Willamette River flows northward and joins the Columbia River in Portland, Oregon. Some of the major tributaries joining the Willamette include the North and South Santiam River, the McKenzie River, and the Coast and Middle-Fork of the Willamette. The Willamette and its tributaries support historic runs of native salmonids including chinook Salmon (spring-run; Oncorhynchus tshawytscha) and winter steelhead (O. mykiss).

Nearly 70% of Oregon's population lives within the Willamette Basin and land-use ranges from highly urban to farmlands and forestry. The river system is a multi-purpose waterway that provides ports for commercial barges, irrigation for crops, recreational opportunities, and fishery. Since settlement by Europeans in the 1800's, the Willamette River Basin has undergone extensive changes through urbanization and installation of dams. Within the Willamette basin thirteen dams administered by the U.S. Army Corps of Engineers (USACE) were established and comprise the Willamette Valley System (WVS). These dams are part of a resource system that provides flood risk management, hydrological power generation, irrigation, and recreational opportunities. The WVS system currently operates under the most recent evaluation. Due primarily to extensive habitat loss and negative impacts from the federal dams, there have been basin-wide declines in once abundant spring Chinook and winter steelhead populations. In 1999, spring Chinook and steelhead in the Willamette Basin were listed under the Endangered Species Act (ESA) and a review and update of federal-dam management within the WVS became a priority.

The most recent Environmental Impact Statement (EIS) for the WVS was completed in 1980. Since then, dam operations have been modified and structural improvement for fish passage and temperature control have been implemented to address impacts of the WVS on ESA-listed fish. In 2008, the National Marine Fisheries Service issued a Biological Opinion for the Willamette River Basin Flood Control Project and the USACE has now reinitiated formal consultation under Section 7 of the ESA. The Portland District USACE is preparing a new EIS to address the continued operations and maintenance of the WVS and to meet obligations to avoid jeopardizing the continued existence of listed species. In development of the EIS, the USACE is evaluating several options to update dam management to help protect salmon, steelhead, and bull trout listed under the ESA.

Purpose

This report was compiled to support development of the EIS to update the management of federal dams in the WVS system using the Ecosystem Diagnosis & Treatment (EDT) model to evaluate habitat potential for spring Chinook and winter steelhead in the Santiam, McKenzie, and Middle- Fork Willamette tributaries.

Habitat potential is a measure of the ability of the habitat, through its physical and biological characteristics, to potentially support the species being modeled by EDT. Habitat potential can be looked at through the lens of habitat quantity (capacity), habitat quality (productivity), or equilibrium abundance, which is calculated based on productivity and capacity and so summarizes both. Diversity is also an indicator of habitat potential, in terms of the breadth of spawning areas and life histories for a species that could be supported by the habitat.

These analyses evaluated the habitat potential for both spring Chinook (all basins) and winter steelhead (Santiam Basin only) under proposed alternatives. Seven alternatives were considered, including a No Action Alternative (NAA). Each of these management alternatives was modeled under wet (represented by 2011), dry (represented by 2015), and normal (represented by 2016) water-year (WY) conditions. The combinations of management alternatives and WY conditions comprise a total of 21 scenarios modeled by EDT for both spring Chinook and winter steelhead.

Project Area

The WVS includes thirteen multi-purpose dams managed by the USACE in the Willamette Basin. Ten of the 13 WVS dams lie within the Santiam, McKenzie, and Middle Fork Willamette basins and are encompassed in the project area evaluated through EDT (Figure 1-1). The primary purpose of WVS dams is flood risk management. Secondary purposes include hydropower, recreation, irrigation, municipal and industrial water supply, fish and wildlife habitat and water quality. The update of WVS management is geared towards balancing the purposes of the dams, with the continued survival of spring Chinook and winter steelhead (USACE, 2022).

In general, current dam management during winter months maintains reservoirs at their lowest elevations to allow for the temporary storage of rain and snow melt. When high flow events occur, the outflow from the dams is coordinated to reduce peak flow and river stages at downstream locations. In spring, WVS dams are managed to fill reservoirs to increase the amount stored for conservation purposes and flood risk management. In summer, stored water is used for recreation on the reservoirs and some water is released into the downstream water to produce hydroelectric power and provide water for irrigation and municipal. During drier summer and fall months, release of stored water augments lower river flows to improve water quality and conditions for fish and wildlife. In fall, water is again drawn down to minimum levels in preparation for the flood season (USACE 2022).

The Santiam River, including both North and South Santiam basins, supports both winter steelhead and spring Chinook populations. Federal dams on North Santiam include Big Cliff and Detroit.

Detroit Dam, completed in 1953, is located 48 miles upstream of the confluence with Willamette River and 13 miles upstream of Mill City, Oregon. Big Cliff Dam, completed in 1954, is located below Detroit Dam. This dam is used to regulate the large flows of water released from Detroit and is also used to generate hydropower. WVS dams in the South Santiam basin are Foster and Green Peter.

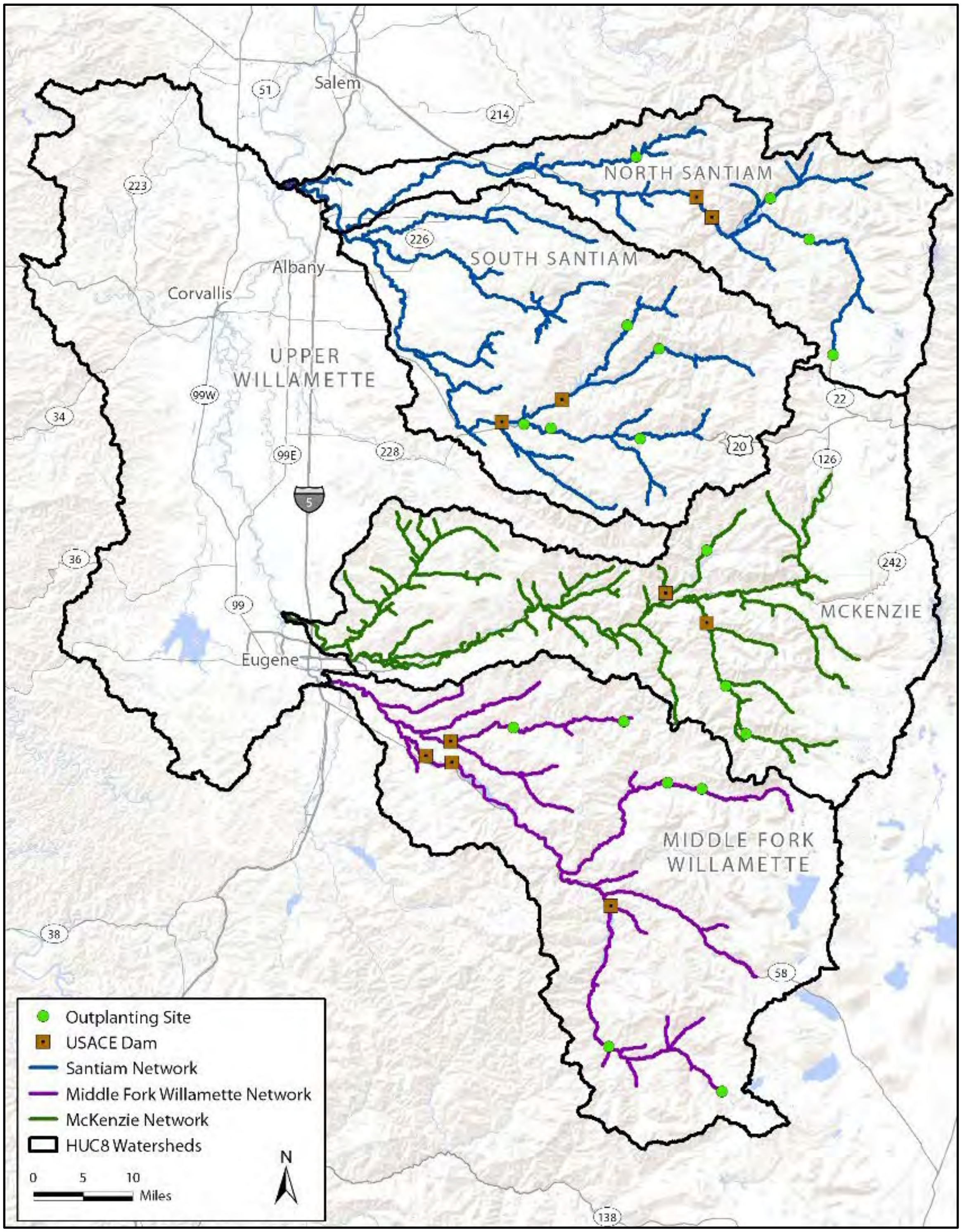


Figure 6-1. Stream network modeled in the Santiam, McKenzie, and Middle Fork Willamette basins.

Foster Dam, completed in 1968, is located about 30 miles upstream from Albany, Oregon. Green Peter Dam, completed in 1966, is seven miles upstream of Foster and is on the Middle Santiam River.

McKenzie and Middle-Fork Willamette support Spring Chinook populations. Cougar and Blue River dams are on the McKenzie. Cougar Dam, completed in 1963, is located on the South Fork of the McKenzie River. Blue River Dam, completed in 1969, is located on a tributary of the McKenzie River and lies about 38 miles east of Eugene, OR.

Four federal dams are within the Middle Fork Willamette basin: Fall Creek, Dexter, Lookout Point and Hills Creek. Fall Creek Dam was built in 1965 and is on a major tributary to the Middle Fork Willamette. Dexter Dam is on the Middle Fork Willamette, one mile downstream of Lookout Point dam. Dexter, built in 1955, is a re-regulating dam that operates to provide consistent flows into the river downstream by managing the fluctuations of outflow from Lookout Point dam which was completed in 1954. Hill's Creek was completed in 1961 and is also located on the Middle Fork of the Willamette River.

6.2 METHODS

6.2.1 Ecosystem Diagnosis & Treatment (EDT)

EDT is a spatially explicit deterministic model used to evaluate habitat conditions relevant to the life stages of the modeled fish species in river reaches through time (Blair et al. 2009).

Overall, three basic components are used in EDT to characterize a watershed: the system geometry (river network), the habitat attributes, and the life histories of the fishes evaluated (Figure 6-2).

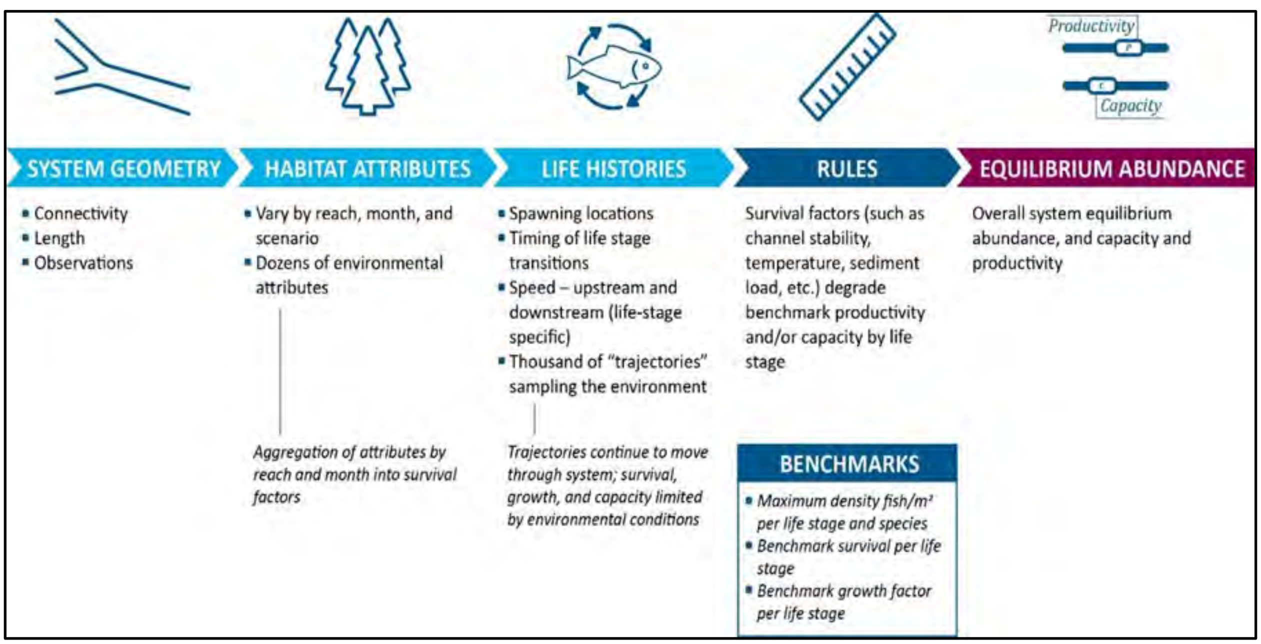


Figure 6-2. Ecosystem Diagnosis and Treatment Framework

The life history component of the model describes and defines, per species evaluated, where the species can spawn, the timing of life stage transitions, and the rate of movement through the system per life stage (Table 2-1). For each species, hundreds to thousands of trajectories are run using the model. For this analysis, approximately 3,000 trajectories were run for spring Chinook and 2,000 for steelhead. Each trajectory demonstrates a specific and realistic life history pattern that could be expressed by that species in the system. Each trajectory starts in one spawning location, has a certain number of days in the egg life stage, a certain number of days until emergence to fry, and specific locations and timings for movements and transitions to additional life stages until returning as a spawner. Collectively, all the trajectories for each species evaluated (termed a 'trajectory set') encompasses a full range of viable spawning locations and specific life history patterns throughout the considered basins.

Table 6-1. Parameters Defining Life Cycle Models for EDT Populations

Parameter	Description	Life Cycle Application	Units
Spawning Reach	Reach locations allowed for spawning trajectories start distributed among these reaches	Trajectories begin as eggs and end as spawners in these locations	EDT reach
Duration	Defines minimum and maximum amount of time trajectory may spend in a life stage	Defined specifically for each life stage	Days
Transition Time Window	Time periods during which one life stage may transition to another	Defined for spawning and for transitions between life stages (egg to fry; marine to migrant prespawner, and so on)	Dates
Speed	Speed at which life stage may move up or downstream	Defined for each life stage	kilometers per day
Location Window	Locations at which one life stage may transition to another	Defined for transitions between life stages	River kilometers (relative to mouth)

Overall, system geometries and trajectory sets remain static among scenarios. Therefore, changes in model results among scenarios are not due to differences in life history configurations or changes to stream networks, but to the habitat modeled. Habitat attributes vary among scenarios, and the interaction of the components of the model for different scenarios is what drives differences in population performance. Overall, the life history trajectories for species are affected in their productivity and capacity by life stage due to habitat conditions (e.g., temperatures that are too high, too much fine sediment, not enough benthic invertebrates) as compared to benchmark values of productivity and capacity (Figure 6-3).

Survival values in the Pacific Ocean are entered as fixed survival rates to complete the species life history.

Ultimately, the EDT model results in population level estimates of capacity, productivity, diversity, and equilibrium abundance by scenario.

Capacity in EDT describes how large a population can grow given the quantity and quality of habitat (Figure 5-3). In EDT, capacity is the asymptotic limit to abundance reflecting habitat area, habitat type (e.g., pools, riffles), food, and productivity.

Productivity in EDT is density-independent survival (intrinsic productivity discussed in McElhany et al. 2000). Productivity under a given set of conditions is the slope of the abundance line of a Beverton-Holt production function graph at its origin (Figure 5-2). Productivity reflects the quality of habitat in reaches and across months throughout the model, according to the life stages of the fish species being evaluated. Productivity is a function of habitat attributes such as temperature, large wood, and water quality that affect survival of life stages. Within the Beverton-Holt formulation, calculation of equilibrium abundance requires a productivity of at least 1 (spawners = progeny). Life history trajectories with productivities less than 1 are considered non-sustainable and do not enter into calculations of abundance.

Equilibrium abundance (Neq) is calculated based on productivities and capacities, and the Neq is the point where the abundance curve crosses the spawner-progeny replacement line (Figure 5-3; Lestelle et al. 2004). The estimate of potential fish performance in EDT reflects habitat conditions from spawning grounds all the way downstream, and back up to spawning grounds as returning adults, spanning the entire life history of the species.

One additional parameter calculated in EDT is diversity. Diversity in EDT is the proportion of sustainable life history trajectories that are used to calculate equilibrium abundance. EDT diversity relates to the breadth of suitable habitat within the spatial unit and the variation in modeled life histories within the population. A lower diversity indicates that the calculated abundance relies on an increasingly narrow range of suitable habitat and life histories within the population. Populations in EDT with higher diversity are assumed to have greater resiliency to environmental perturbations compared to those with lower diversity.

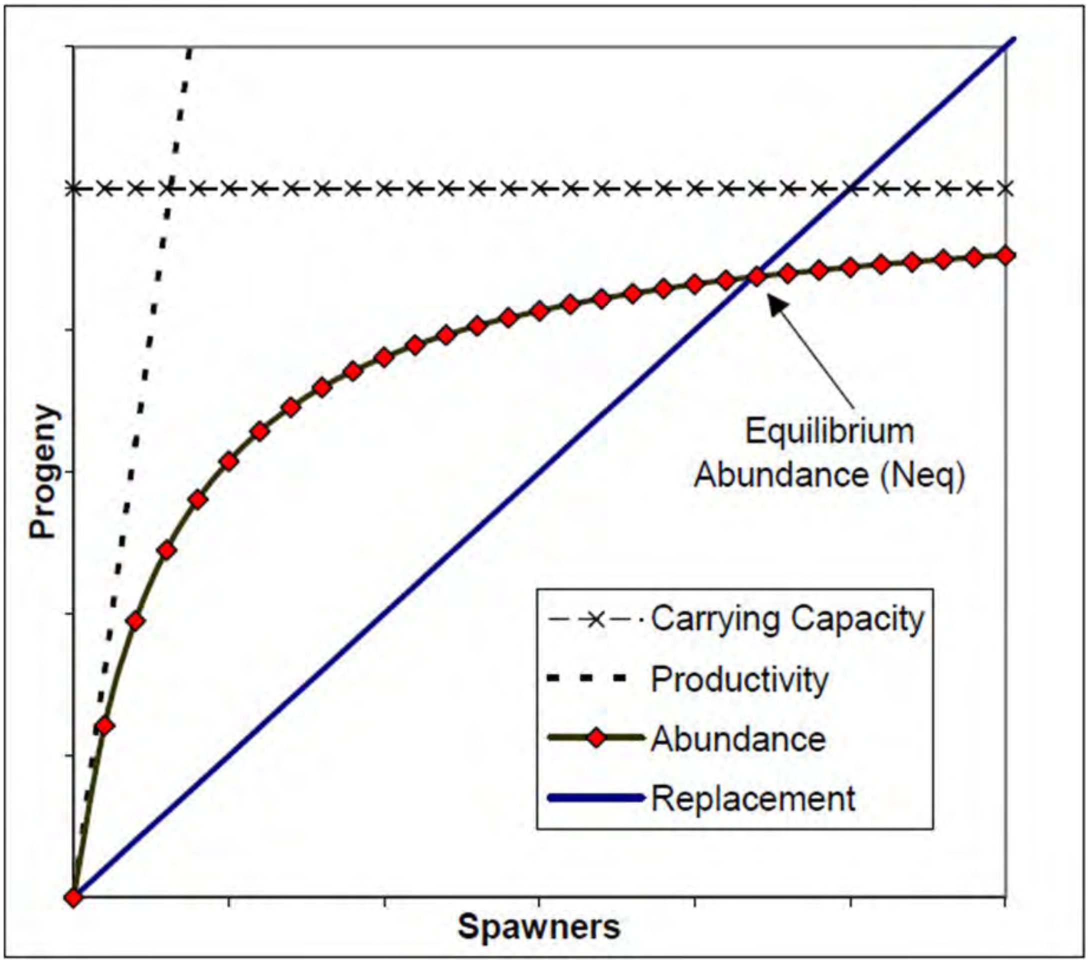


Figure 6-3. Example Beverton-Holt Production Function (from Lestelle et al. 2004).

EDT is a habitat-based model that is used to evaluate habitat potential for a modeled species by looking at habitat quantity (capacity), habitat quality (productivity), and/or equilibrium abundance. Diversity is also an indicator of habitat potential measured as the breadth of spawning areas and life histories for a species that could be supported by the habitat.

6.2.2 General Model Information

Species Modeled

The two modeled species included spring-run Chinook salmon (spring Chinook) and winter-run steelhead (winter steelhead). Fish migrate up and down the mainstem Willamette River to enter or leave the Santiam, McKenzie, and Middle-Fork Willamette tributaries. Fish exiting the Willamette enter the Columbia River and ultimately the Pacific Ocean before returning to upstream basins to spawn as adults.

Spring Chinook Salmon

Spring-run Chinook salmon are present in Santiam (North and South), McKenzie, and Middle-Fork Willamette basins (Figure 5-4). Spring-run Chinook salmon are modeled in EDT to spawn in late August through October. The freshwater life cycle of spring Chinook in the Willamette basin varies from stream type yearlings to ocean types to fall outmigrants. Spring-run Chinook prespawning migrants leave the ocean and begin traveling back up the Columbia and Willamette mid-March through May, until they reach their spawning grounds again and hold until the spawn. Life history distribution of spring-run Chinook in the Willamette model is outlined in Table 5-2.

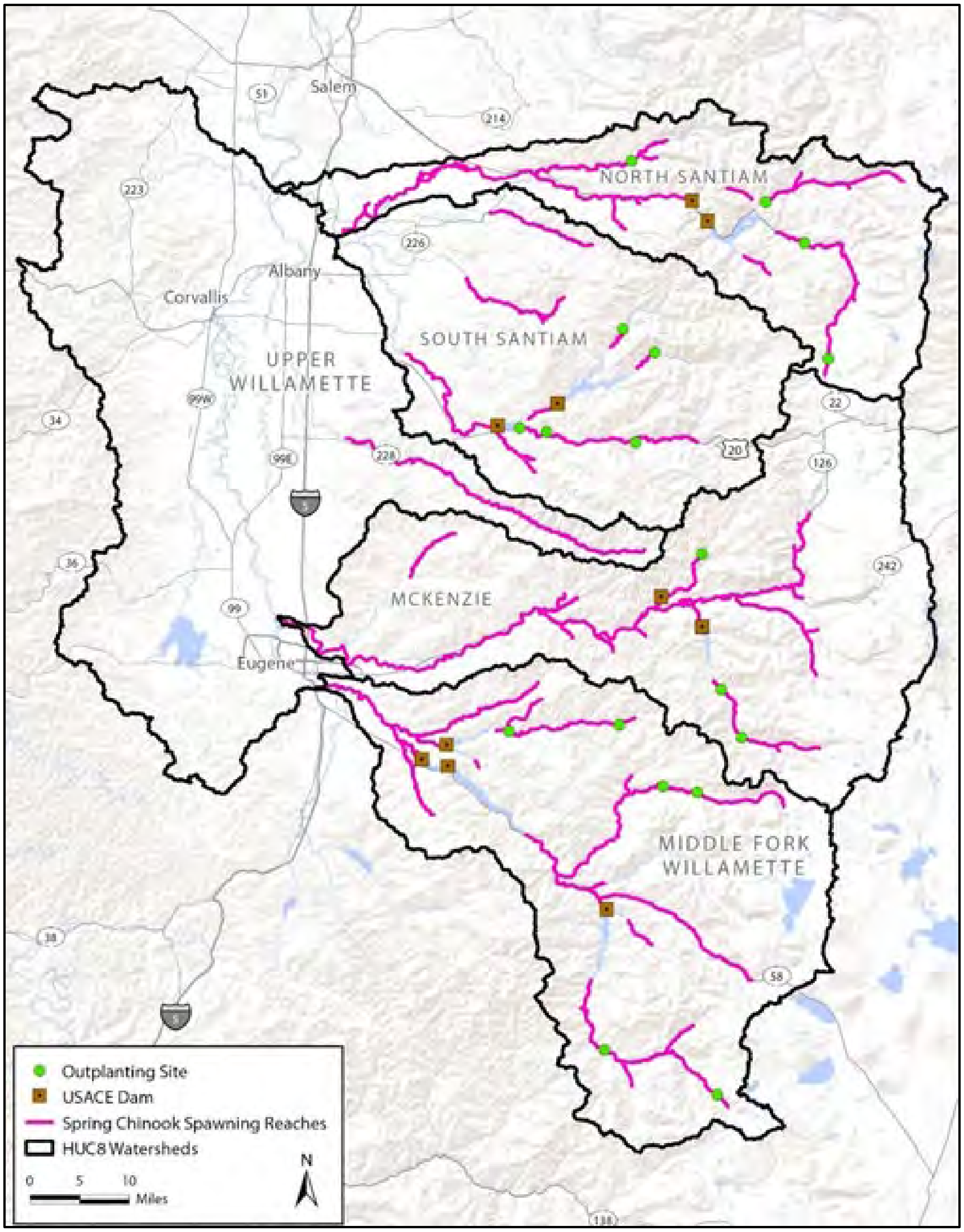


Figure 6-4. Spawning Reaches Used in EDT For Spring Chinook Salmon. *Map of spawning reaches used in EDT for spring Chinook in the Santiam, McKenzie and Middle Fork Willamette basins (includes reaches related to proposed outplanting).*

Table 6-2. Distribution of spring Chinook life cycle models in the Willamette trajectory set.

Life cycle model	Overall strategy	% Of trajectory set
1,2-yearling	Stream type resident	8.7
1,3-yearling	Stream type resident	28.9
1,4-yearling	Stream type resident	0.8
0,2-subyearling	Ocean type	4.6
0,3-subyearling	Ocean type	33.2
0,4-subyearling	Ocean type	3.2
0,2-fall migrant	Fall outmigrant	2.3
0,3-fall migrant	Fall outmigrant	16.6
0,3-fall migrant	Fall outmigrant	1.6

Winter Steelhead

Winter steelhead were modeled in both the north and south Santiam basins (Figure 2-4). Steelhead were modeled to spawn from March through May. Spending various amounts of time rearing in freshwater with both resident and transient life history patterns, they then were modeled to enter the ocean from the Columbia estuary from late April to late May. After spending time in the ocean, migrant prespawners were then modeled to re-enter freshwater in December and January. Life history distribution of winter steelhead in the Willamette model is outlined in Table 6-3.

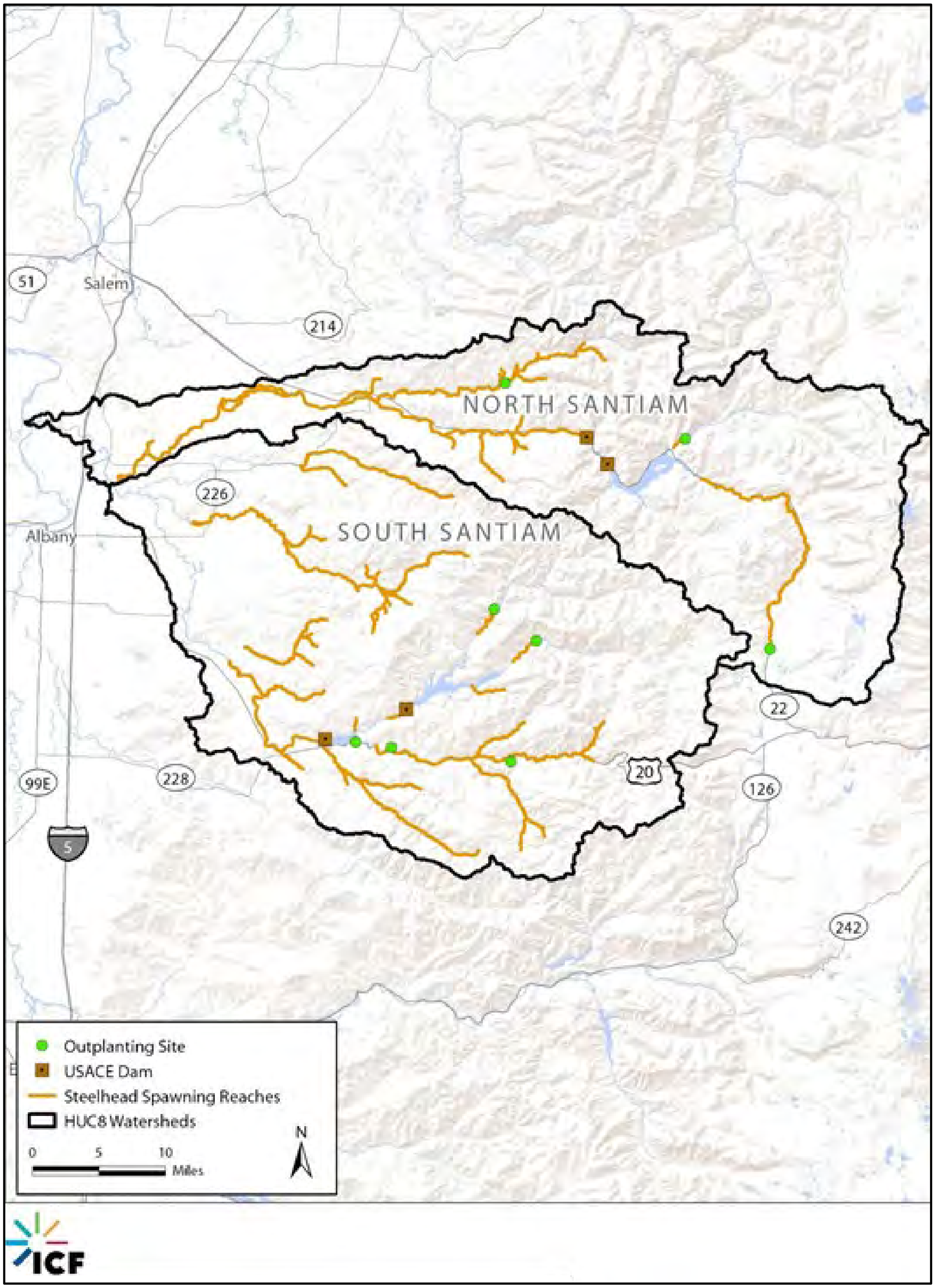


Figure 6-5. Map Of Spawning Reaches For Winter Steelhead In The Santiam Basin *(includes reaches related to proposed outplanting).*

Table 6-3. Distribution of steelhead life cycle models in the Willamette trajectory set.

Life cycle model	Overall strategy	% of trajectory set
1,1-resident	Stream type resident	15.75
1,2-resident	Stream type resident	7.73
1,3-resident	Stream type resident	0.02
2,1-resident	Stream type resident	15.75
2,2-resident	Stream type resident	7.73
2,3-resident	Stream type resident	0.02
3,1-resident	Stream type resident	2.01
3,2-resident	Stream type resident	0.99
3,3-resident	Stream type resident	0.003
1,1-transient	Transient rearing	15.75
1,2-transient	Transient rearing	7.73
1,3-transient	Transient rearing	0.02
2,1-resident	Transient rearing	15.75
2,2-resident	Transient rearing	7.73
2,3-resident	Transient rearing	0.02
3,1-resident	Transient rearing	2.01
3,2-resident	Transient rearing	0.99
3,3-resident	Transient rearing	0.003

6.2.3 Modeled Scenarios

To model the habitat potential of the species of interest the current habitat conditions within considered basins were established first. Then habitat conditions under alternative management scenarios were altered for each scenario to evaluate habitat potential under each scenario. There were existing EDT models for Willamette and McKenzie basins, however an EDT model had not been built for the Santiam basin prior to this assessment. Environment attribute data were imported from a prior version of EDT, for the McKenzie River, Middle Fork Willamette River, and mainstem Willamette. The Santiam basin network and habitat attributes describing the current condition for salmonids were built based on available information.

6.2.4 Current Condition

Mainstem Willamette

Data Transfer: EDT2 to EDT3

The Willamette River mainstem from the Middle Fork/Coast Fork confluence to the Columbia River confluence has been modelled in EDT2, the previous version of EDT. One ‘Habitat

Attribute Reach Dataset' (HARD) was imported from the Willamette River mainstem into EDT 3. Specifically, "Willamette_062906" was imported. It was last updated by Betsy Torell in 2006.

Additional Updates

Reach geometry was updated for the mainstem Willamette River to correct mismatched reach descriptions and lengths. Mainstem temperatures were updated using USGS continuous temperature gauge data for the last three years near Eugene and Portland.

McKenzie and Middle Fork Basins

Data Transfer: EDT2 to EDT3

The McKenzie River and the Middle Fork Willamette had been modelled in EDT2, the previous version of EDT. One HARD was imported from each of these subbasins into EDT3. From the McKenzie River subbasin the HARD "McKenzie 032404" was imported; it was last updated by Chip McConnaha and John Runyon in 2004. From the Middle Fork Willamette subbasin the HARD "MFW_CFW_121307" was imported; it was last updated by Betsy Torell in 2007.

Additional Updates

Maximum water temperature ratings were reviewed and updated to better reflect current conditions in the McKenzie and Middle Fork Basins.

Santiam Basin

Current habitat conditions for the Santiam Basin were described via multiple sources of data sources (Table 2-4). Several habitat attributes were compiled from the AWQMS (Aquatic Water Quality Monitoring System) managed by the ORDEQ (Oregon Department of Environmental Quality). This database manages statewide water quality data from a variety of sources and provides an assessment of the utility and reliability of these data (ORDEQ 2021). Only data rated highly reliable and of high quality were used to create habitat ratings for EDT attributes. Other sources of information included R2 Resource Consultants, Inc., USGS (United States Geological Survey) gauging stations, USACE (U.S. Army Corps of Engineers) reports and Access database, and ODFW (Oregon Department of Fisheries and Wildlife) Aquatic Inventories Project.

Table 6-4. Sources used for updating current conditions Willamette basin.

Data source	Basin(s)	EDT attributes
AWQMS (ORDEQ 2021)	North and South Santiam	Alkalinity, Metals in the Water, Temperature, Nutrients, Dissolved Oxygen, Total Suspended Solids
USGS (USGS 2020)	North and South Santiam	Flow (Historic and Current), Temperature
R2 Resource Consultants, Inc. (R2 2007; USACE 2007; USACE 2009)	North and South Santiam	Habitat Type, Large Wood, % Secondary Channel, Channel Width, Embeddedness, Fine Sediment (intra-gravel)
ODFW Aquatics Inventory Project (ODFW 2007)	North and South Santiam	Habitat Type, Large Wood, % Secondary Channel, Confinement, Channel Width, Channel Width, Embeddedness, Fine Sediment (intra-gravel)

Temperature

For the streams within the EDT network, daily water temperature parameters were downloaded from the AWQMS data management website from 2000-2021 (ORDEQ 2021). Data sources for the continuous temperature monitoring included: North Santiam Watershed Council, South Santiam Watershed Council, USGS, USFS, ORDEQ, SECOR International, and BLM (Bureau of Land Management). The maximum monthly temperature was used to assign a reach's EDT monthly index value (Lestelle 2005). When a reach's data did not encompass all months of the year, a relativized rating was developed using the pattern of a neighboring reach that had year-long data and similar ratings for overlapping months. For reaches with no temperature data, the nearest reach of similar size was selected as a proxy to represent conditions. When no nearest neighbor of similar size was available, a within-basin proxy was used (See Appendix A for proxy reaches used for the Maximum Temperature Attribute Rating for Santiam.).

Habitat Types and Wood

Habitat types within a reach, the amount of large wood per channel width (adjusted by reach length), and percent side channel habitat were derived from data from habitat surveys by either R2 Resource Consultants, Inc. or ODFW Aquatic Inventories Project. For R2 surveys, data were gathered for representative stream transects. We then referenced these transects to an EDT reach based on a combination of latitude and longitude and landmarks, such as tributary junctions. The data collected for the transects within an EDT reach were then summarized to derive EDT measurements and ratings for the model. ODFW habitat surveys gathered information for continuous stream lengths, that we could similarly georeference to EDT reaches. For each survey type, R2 or ODFW, we calculated, summarized, and rated the needed environmental attributes for habitat characterization in EDT.

Habitat types are expressed as a percentage of the stream bed, with each reach habitat type percentages totaling 100% when summed for an individual reach. We converted R2 and ODFW habitat types into EDT habitat types (Table 6-5) and calculated the percentage of each type within the reach. R2 and ODFW survey methods lacked an inventory of pool tail habitat which is an important habitat factor for salmonids. In accordance with previous EDT modeling and accepted methodology, we estimated pool tail habitat as 15% of total area of scour pool habitat. Accordingly, we also reduced the total area of scour pool habitat by 15% prior to calculating the percent sour pool habitat.

Table 6-5. Conversion of R2 and ODFW habitat types into EDT habitat types.

EDT Habitat Type	R2 Habitat Type	ODFW Habitat Types
Glide	Glide	Glide
Small Cobble Riffle	Riffle	Riffle
Large Cobble Riffle	Riffle with Pockets	-
Large Cobble Riffle	Cascades	Cascades
Large Cobble Riffle	Rapid	Rapids
Large Cobble Riffle	Step	Step/Falls
Backwater Pool	Backwater Pool	Dammed and Backwater Pool*
Scour Pool	Lateral Scour Pool	-
Scour Pool	Scour Pool (adj. by Pool Tail Area)	Scour Pool (adj. by Pool Tail Area)
Scour Pool	Plunge Pool	-
Pool Tails	15% of Total Scour Pool Area	15% of Total Scour Pool Area
Beaver Ponds	Dammed Pool (DP)	50% Dammed/Backwater Pool*

Note: *When beaver activity was noted in ODFW survey, 50% of ODFW Dammed/Backwater Pool habitat was assigned to Beaver Pond Habitat Type and the area of Backwater Pool was adjusted. If beaver activity was not noted, then no reach area was assigned to Beaver Ponds.

EDT attribute data includes percentage of side channel habitat within a reach. R2 survey protocol included categorization of primary, split channel, secondary channel, backwater, and additional channel types. Both single channel and primary channel types in the R2 surveys were considered mainstream channel under EDT classification. Secondary channels identified in the R2 surveys were considered side channel habitat. ODFW habitat survey protocols similarly identified and measured secondary channel habitat. We calculated percent side channel for EDT characterization as a proportion of total reach length.

In the EDT model, the amount of large wood per channel width is used to apply an overall reach rating for that attribute (Lestelle 2005). Under EDT attribute guidelines, large wood pieces included in the total count per reach are longer than 2m (~6.6 feet) and have diameter exceeding 0.1 m (0.33 ft). In the R2 survey protocol, the size of the large wood pieces counted in transect were nearly twice the size as the minimal piece considered in EDT; a piece had to be at least 3.7 m (12 feet) in length and >0.3 m (1 ft) diameter. We also looked in the habitat survey notes for any mention of log jams to help refine EDT rating. To make a rough correction to account for smaller wood pieces excluded by R2 surveys, we improved the EDT rating by 0.5 for each reach that had a rating better than 4 (In EDT the best rating for the large wood attribute is 0 and the worst 4.) Any reach with a rating of 4, was kept at the lowest rating. ODFW wood count protocol matched EDT protocol, so large wood ratings for reaches characterized by ODFW surveys were not adjusted.

Changes in Inter-annual Flow Variability (High and Low)

To assess the changes in inter-annual flow variability at high and low flows, the historic and current flow patterns are compared to assess whether current flows are different than historic patterns (Lestelle 2005). Where there are no changes, an index of 2 indicates current flow patterns did not vary from historic flows. When assessing changes in current conditions from historic peak flows, ratings higher than 2 indicate a shift toward higher peak discharge and ratings lower than 2 indicate reduced peaks. When assessing current condition changes from historic low flow, ratings higher than 2 indicate shifts toward more interannual variability/lower low flow discharge and ratings lower than 2 represent shifts towards less variability/increased low flows. Both historic and current discharge data were downloaded from USGS gauging stations (USGS 2020). For historic flow that was prior to dam installation, we used flows before 1950 in North Santiam and 1966 in South Santiam. To establish current flow patterns, we used available data for as close to a 30-yr record as we could obtain, from 1990 to the present day. Gauging stations were scattered throughout both North and South Santiam basins, providing a framework to assess EDT flow attributes. When reaches did not have a gauging station, we used ratings for a reach that was similar in size and within the basin. Many smaller streams had no flow data available; primarily those categorized as Low Flow <3 cfs (cubic feet per second), Headwaters, and Low Stream Order. For these streams, the month with the highest flow was rated 2 for the High Flow attribute rating and rated 2 during the month with the lowest flow for the Low Flow attribute rating (high and low flow months were identified based on existing flow data in the basin). The assumption was that these smaller waterways were not modified from historic flow patterns and a rating of 2 (neutral) was appropriate. A within basin scalar was then used to calculate the flow ratings for all other months of the year. For Mid-Stream Order and High Stream Order reaches, a similar proxy reach was used for flow ratings.

Bed Scour

Bed scour was calculated based on gradient and the high flow pattern of the reach. For all reaches with gradient more than or equal to 1%, the top rating was assumed to be 2. For all reaches less than 1%, the top rating was assumed to be 0, with little to no bed scour (Lestelle 2005). The high flow scalar pattern was then used to calculate the monthly bed scour value with the appropriate maximum rating for the reach.

Width

In EDT, the dimensional attribute of channel width is entered in monthly increments. We used flow data and channel measurements from habitat survey data to develop a scalar used to estimate monthly widths.

For each reach with width and flow data, the month with the maximum width was assigned to the month with the highest flow (USGS gauge data), which was usually January in the Santiam Basin. A scalar of 1 was assigned to (January) and a rough estimate of change in the scalar for each month was calculated based on flow pattern. Within a year, January's scalar was decreased each month until August (lowest flow month) and then increased by that same amount through December. For reaches with no flow data, width scalars were used based on proximity and stream-size.

For all reaches with habitat surveys, we used the measured bank full width as the maximum width for the month with the highest flow. For reaches that were not surveyed, the maximum width was the average bank full width of surveyed reaches of an Environmental Type within a Diagnostic Unit. When there were no survey-widths for an Environmental Type within a Diagnostic Unit, the maximum width for a reach was the average bankfull width for that Environmental Type for all surveyed reaches.

For each reach, monthly channel widths were then calculated by multiplying a month's scalar value by the maximum width.

Embeddedness and Fine Sediment (intra-gravel)

In EDT, embeddedness is rated on the extent (average %) that cobble and gravel particles are buried by fine sediment in both riffle and pool tailout habitat units where cobble and/or gravel substrates more often occur. The Fine Sediment (intra-gravel) attribute is rated based on the percentage of fine sediment within salmonid spawning substrates which are in pool tailouts, glides and small cobble-gravel riffles. For reaches surveyed by R2 methods, we looked at the estimated percent of silt found in small and large cobble riffles and scour pool habitat and used best judgement to assign the EDT Index value to embeddedness. For the fine sediment attribute rating, we looked at the percent silt and sand in scour pools, glides, and small cobble riffle habitat units (Lestelle 2005). For reaches surveyed using ODFW protocols, we used the percent sand, silt, and organics in the surface substrate of riffles, percent recorded for all habitats, and habitat-type percentages to inform the EDT ratings for both Embeddedness and Fine Sediment.

Benthos Diversity and Production

Benthic macroinvertebrate community data was downloaded from AWQMS (ORDEQ 2021). For all sampled sites, we obtained count data and calculated a B-IBI score to assess the macroinvertebrate community (Lestelle 2005). For some sites there were also O/E Ratio

(Observed taxa/Expected Taxa) and % Taxa Loss statistics provided using ORDEQ PREDATOR (Predictive Assessment Tool for Oregon) methods (ORDEQ 2020). When we had a O/E Ratio or % Taxa Loss that indicated worse or better condition than the B-IBI score indicated, we adjusted the EDT Rating down (improved) or up (worse) by 0.5 as indicated.

Alkalinity, Dissolved Oxygen, Nutrient Enrichment, and Turbidity

Most water chemistry data used to describe current conditions in the Santiam were retrieved from the AWQMS database (ORDEQ 2021). We downloaded available data from 2000-2021 and geolocated all sampling sites to EDT reaches. Many measurements were grab samples taken periodically through the years at varying months and frequency in the Santiam basins.

All total alkalinity measurements (as CaC03 mg/L) in the basin ranged between 10-40. These concentrations were consistent with those found on the westside of the Cascade Range, indicating a basin-wide EDT rating of 2 was appropriate (Lestelle 2005).

Dissolved Oxygen (DO, mg/L) was not measured extensively throughout the Santiam basin. Both grab samples by various organizations and continuous sampling by USGS showed no indication that DO was an impairment under EDT guidelines since concentrations remained above 8 mg/L throughout the year. Hence, DO was rated 0 for the entire basin (Lestelle 2005).

Concentration of Chlorophyll a (mg/L) served as a proxy for determining extent of nutrient enrichment within the basin. Although Chlorophyll a was measured at only a few sites and months through the years, concentrations rarely exceeded 3 mg/L indicating little to no nutrient enrichment in the basins based on the EDT rating scale (Lestelle 2005).

Heavy metal contamination was evaluated using available AQWMS water quality data. A suite of heavy metals that are potentially harmful to aquatic wildlife were measured including copper, zinc, selenium, silver, and several others. Most water samples showed either no heavy metals or levels below what would be harmful to salmonids under either short- or long-term exposure.

Where heavy metals were found in the water, we rated the reach and neighboring reaches 0.5, while all other reaches were rated 0.

Turbidity was assessed in the Santiam basin using Total Suspended Solid (TSS in mg/L) measurement. All available TSS data indicates that concentrations remain below <50 mg/L overall in both basins for all months (ORDEQ 2021). While primarily grab sample data was available for TSS, the few reaches that have measurements on a more regular basis indicates that the duration of any mildly increased concentration does not last for very long (i.e., short period of any elevated values, all under <50 mg/L), even during higher flow months. These two factors supported a 0 for the EDT Index Rating for all reaches and months (Lestelle 2005).

6.2.5 Alternative Scenarios

There are seven management alternatives that were modeled through EDT: No Action Alternative (NAA), Alternative 1 (Alt1), Alternative 2a (Alt2a), Alternative 2b (Alt 2b), Alternative 3a (Alt3a), Alternative 3b (Alt3b), and Alternative 4 (Alt4). Encompassed within these operational alternatives, there are measures to address fish-passage, water-quality, and flow. Fish passage is addressed under each alternative through either structural improvements and/or dam operations. The primary water quality issues addressed are maximum water temperature and potential harmful effects of increased Total Dissolved Gas (TDG) below dams. Flow management is aimed toward maintaining minimum flow to meet fish needs and aiding fish passage. A short description summarizing each alternative follows below (USACE, 2022) and details for individual dam-operations/modifications are provided in Appendix D. Measures common to the alternatives, excluding NAA, include hatchery improvements and gravel augmentation.

No Action Alternative (NAA)

The NAA alternative continues operations as they exist today at the ten dams included in the EDT modeling.

Alternative 1

Alt1 is focused on improving fish passage with storage-focused measures. Operations would increase the likelihood of WVS reservoirs refilling to their maximum pool levels in spring. Other measures include structural measures for fish passage, temperature control to allow for fish collection, and water temperature control over various water levels throughout the year.

Measures would reduce flows to Congressionally authorized minimum flows, add downstream fish passage structures, as well as water quality structures to allow for more water to be stored at certain reservoirs.

Alternative 2a and 2b

Alt2a and 2b aim to improve fish passage through modified operations and structural improvements. Other measures contribute to balancing water management flexibility to meet ESA-listed fish needs. Alt2a and 2b only differ in that 2a includes a structure at Cougar, while 2b does not.

Alternative 3a and 3b

Alt 3a and 3b are considered operations focused and are geared towards improving fish passage through water management rather than many structural modifications. Spring spill, spring drawdown, and deep fall reservoir drawdowns are parts of these operations plans. Deep drawdowns are when the water in a reservoir is lowered as much as possible during migration so fish can more easily find outlets to migrate through the dam. The primary difference between 3a and 3b are operations at Cougar Dam to aid fish passage. Under 3a there would be a deep drawdown drafting to 10 feet over the top of the Cougar Dam Regulating Outlet. Under 3b, there would be drafting to Cougar Diversion Tunnel for both deep fall and spring drawdown measures.

Alternative 4

Alt 4 is a structure-focused alternative that includes construction of upstream and downstream fish passage facilities, water temperature control towers, and structures to reduce Total Dissolved Gas (TDG) which can be harmful to fish.

For each alternative, the EDT network was characterized for three flow years: a wet year represented by WY 2011, a dry year represented by WY 2015, and a normal year represented by WY 2016. A combination of WY and management alternative are referred to as a scenario throughout this document. Scenarios differ in characterization of certain environmental attributes throughout the basin, especially in reaches downstream of managed dams.

Temperature, high and low flows, and fish passage at dams varied for effected reaches under each of these 21 scenarios. In addition, the EDT model accounted for: expansion of spawning areas through outplanting (fish-placement) and habitat improvement via gravel augmentation downstream from all project dams.

Maximum Temperature Ratings

Daily mean temperature for each scenario was provided to ICF by USACE via CE-QUAL-W2 model output. For the EDT model, we calculated temperature ratings for each month of a scenario at the reaches for which data were provided (Appendix B). After processing, modeled temperature-data for each scenario were used to rate maximum temperature for 20 EDT reaches in the Santiam basin, 13 in the Middle Fork basin, 52 in the McKenzie basin, and 16 in mainstem Willamette.

High and Low Flow Ratings

Willamette Basin stream flow was modeled using the Res-Sim program and was provided to ICF by USACE for both regulated and unregulated management under each scenario. EDT ratings for both Flow High and Flow Low ratings for each scenario were determined by comparing the differences between the unregulated and regulated conditions within each water year (Appendix C). In this way, the EDT rating for each scenario reflected the effect of each alternative management within a wet, dry, or normal water-year.

Fish Passage

Fish passage at several of the dams was modeled through a fish benefits workbook (FBW) by USACE. The average survival probability for different life-stages was used as the downstream-passage value at the dams. Chinook passage was modeled for Cougar, Detroit, Foster, Green Peter, Hills Creek, and Lookout Point. Passage for Steelhead was included for the Santiam Basin (Detroit, Foster, and Green Peter). At most dams, upstream passage was set at 100% for both species except for: Winter Steelhead at Green Peter under NAA and Alt 4 scenarios and Chinook Salmon at Hill's Creek under Alt 2a & Alt 2b scenarios.

Outplanting

Among the existing and planned outplanting sites, EDT accounted for any fish placements that would create areas of expansion (Table 2-6). One outplanting site in the McKenzie Basin would create a range expansion for Spring Chinook into the basin above Blue River dam (Figure 2-3). Within the Santiam Basin, placements of Steelhead would create range expansions above Green Peter dam in the South Santiam and above Detroit dam in the North Santiam (Figure 5-5).

Table 6-6. Range expansions from planned outplanting in the Santiam and McKenzie Basins.

USACE Project Name	River Basin	USACE Outplanting Site	Range Expansion	Alternative
Blue River	McKenzie	Lower release site 2-5 miles above head of reservoir	Spring Chinook	3
Detroit	North Santiam	Breitenbush USGS Gauge Site (#14179000)	Winter Steelhead	1,2,3,4
Detroit	North Santiam	Cooper's Ridge (Lower)	Winter Steelhead	1,2,3,4
Green Peter	South Santiam	Lower release site 2-5 miles above head of reservoir in Middle Santiam	Spring Chinook Winter Steelhead	1,2,3
Green Peter	South Santiam	Lower release site 2-5 miles above head of reservoir in Quartzville Creek	Spring Chinook Winter Steelhead	1,2,3

Gravel Augmentation

Additions of spawning-sized gravel to areas just below dams to improve spawning habitat was accounted for in all scenarios except No Action. The proportion of small riffle habitat type was increased 10%, with a corresponding decrease in other habitat types to keep all habitat types summing to 100%.

Total Dissolved Gas (TDG)

TDG is often elevated downstream of dams where spilling water and turbulence causes increased in dissolved gasses. When saturation levels are too high, fish can be negatively impacted. Results of TDG modeling are not included here, though experimental runs were conducted. The effects of the TDG were experimentally accounted for in the EDT model for reaches downstream from the Detroit, Big Cliff, Green Peter, Foster, Cougar, Hill's Creek, Lookout Point, and Dexter Dams. Daily TDG concentrations were provided by USACE for all scenarios and water years. Monthly average TDG was used to rate the conditions following the indexes in Table 5-6. In the case of Big Cliff and Detroit, fish within the reach downstream of both dams (reach North Santiam-11) would experience the accumulated effects of TDG from both dams; hence the rating was increased to that of Detroit's plus half that of Big Cliff.

Table 6-7. Index of TDG Ratings used in EDT models.

Index 0	Index 1	Index 2	Index 3	Index 4
Very low (average value typically would be <103 %)	Moderately low (average value typically would be 104 – 109 %)	Moderately high (average value typically would be 110 – 114 %)	High (average value typically would be 115 - 120 %)	High (average value typically would be >120 %)

6.3 RESULTS AND DISCUSSION

6.3.1 No Action

Spring Chinook

Under the No Action alternative, spring Chinook abundances ranged from 14,565 to 19,596 depending on the year type, with the most spring Chinook in wet years and the least in dry years (Figure 3-1 and Table 3-1). Diversity of spring Chinook ranged from 30.6% to 50.1%, again with the lowest diversity in dry years and the highest diversity in wet years (Figure 3-1 and Table 3-1). Capacity followed this same pattern, while productivity was modeled to be highest under dry years followed by wet and then normal years (Table 3-1). This lower productivity under normal and wet years is due to additional spawning reaches and life history patterns that contributed to the population in these model runs (higher diversity) and brought down the overall productivity while still contributing to the population.

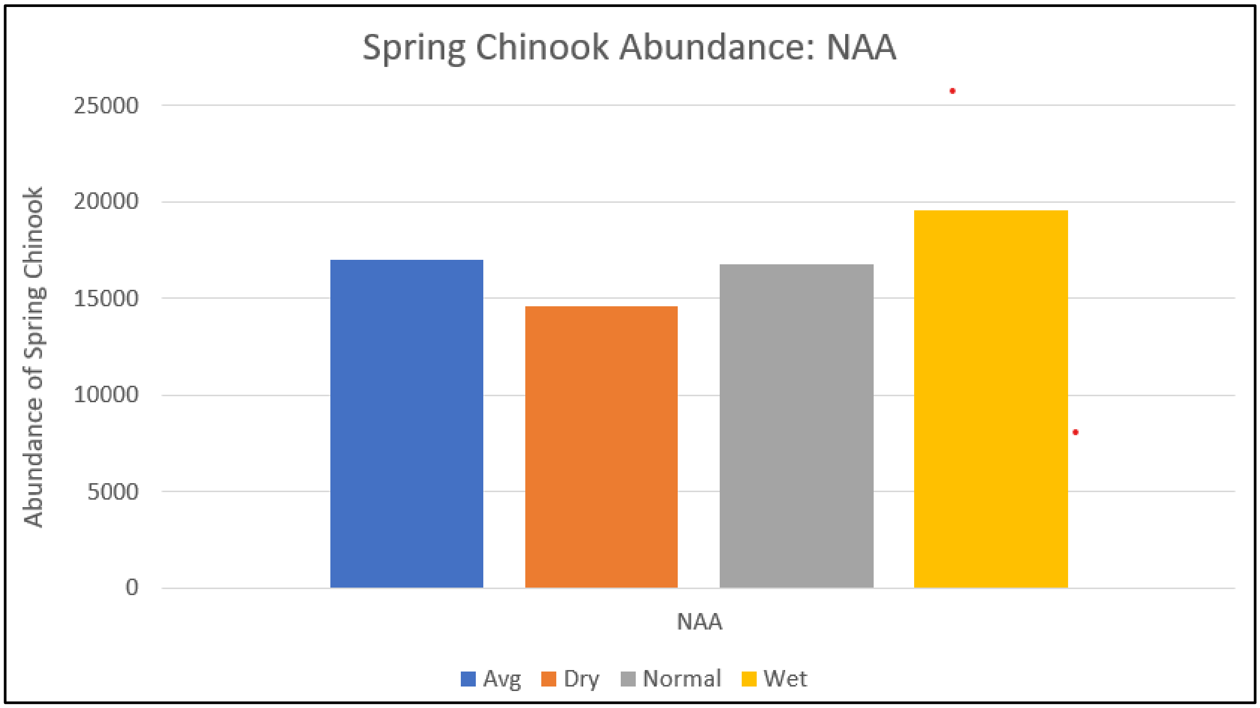


Figure 6-6. Abundance of spring Chinook for the NAA model.

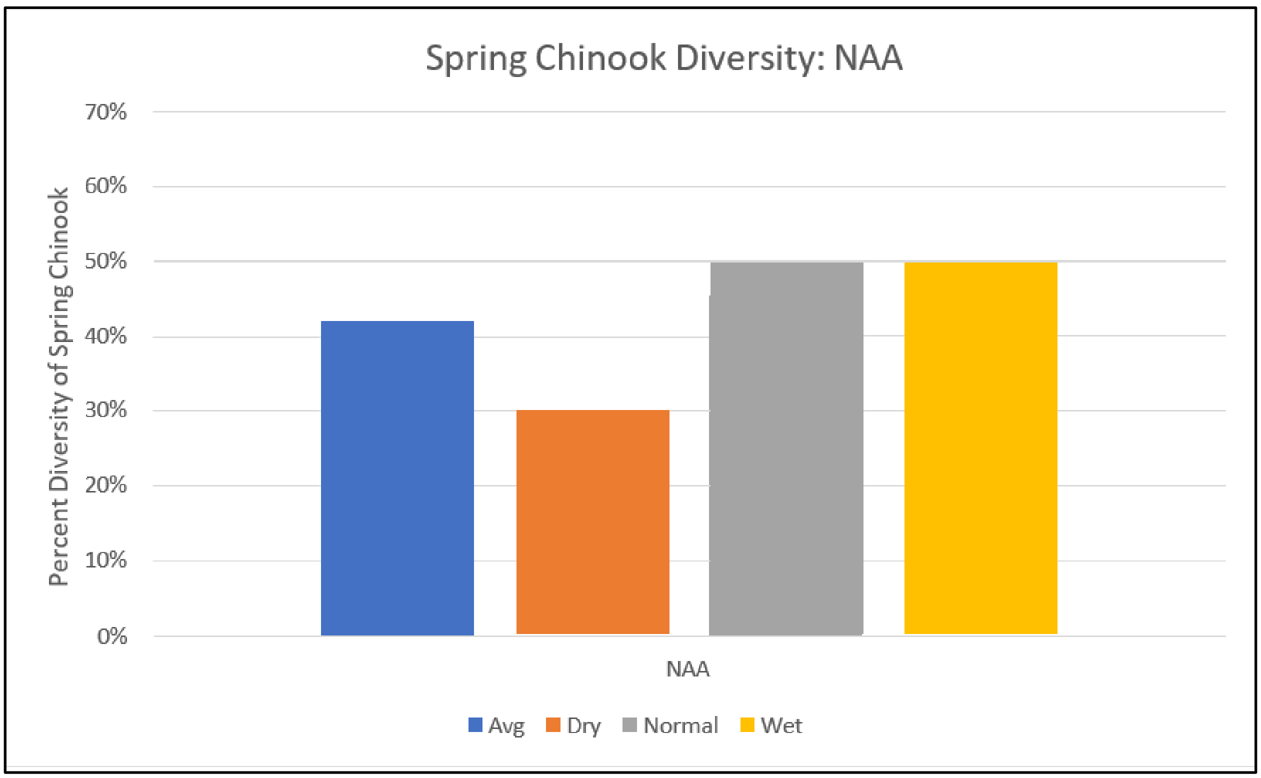


Figure 6-7. Diversity of Spring Chinook for the NAA model.

The majority of spring Chinook abundance was from the McKenzie watershed, followed by the North Santiam, Middle Fork, and South Santiam basins (Table 3-1).

Table 6-8. EDT modeling results for Spring Chinook under NAA alternative

Result	Water Year	All Subpopulations	North Santiam	South Santiam	McKenzie	Middle Fork Willamette
Abundance	Wet	19,596	1,488	638	16,544	935
Abundance	Dry	14,565	689	271	13,024	440
Abundance	Normal	16,727	1,299	496	14,193	699
Capacity	Wet	22,434	1,859	769	18,546	1,260
Capacity	Dry	16,597	817	416	14,741	623
Capacity	Normal	19,242	1,618	624	16,008	992
Productivity	Wet	7.9	5.0	5.9	9.3	3.9
Productivity	Dry	8.2	6.4	2.9	8.6	3.4
Productivity	Normal	7.7	5.1	4.9	8.8	3.4
Diversity	Wet	50.1%	69.9%	31.8%	67.7%	27.7%
Diversity	Dry	30.6%	22.4%	16.8%	64.6%	10.8%
Diversity	Normal	45.5%	66.8%	27.8%	64.5%	19.9%

Winter Steelhead

Under the No Action alternative, steelhead abundances ranged from 8,046 to 9,836 depending on the year type, with the most steelhead in normal years and the least in dry and wet years (Figure 5-8 and Table 6-8). Diversity of steelhead ranged from 70.3% to 73.4%, with the highest diversity in normal years and the lowest in dry years (Figure 3-4 and Table 3-2). Capacity was highest under normal years, while productivity was modeled to be highest under wet years (Table 3-2).

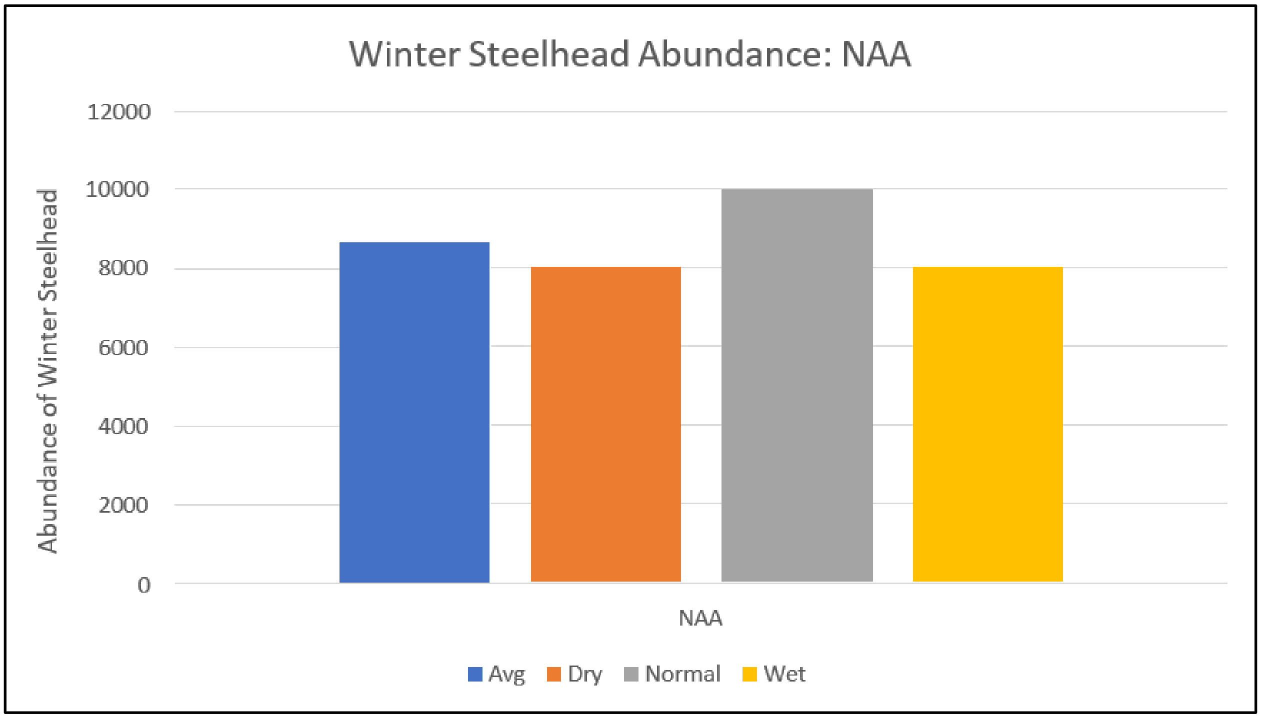


Figure 6-8. Abundance of winter steelhead for the NAA model.

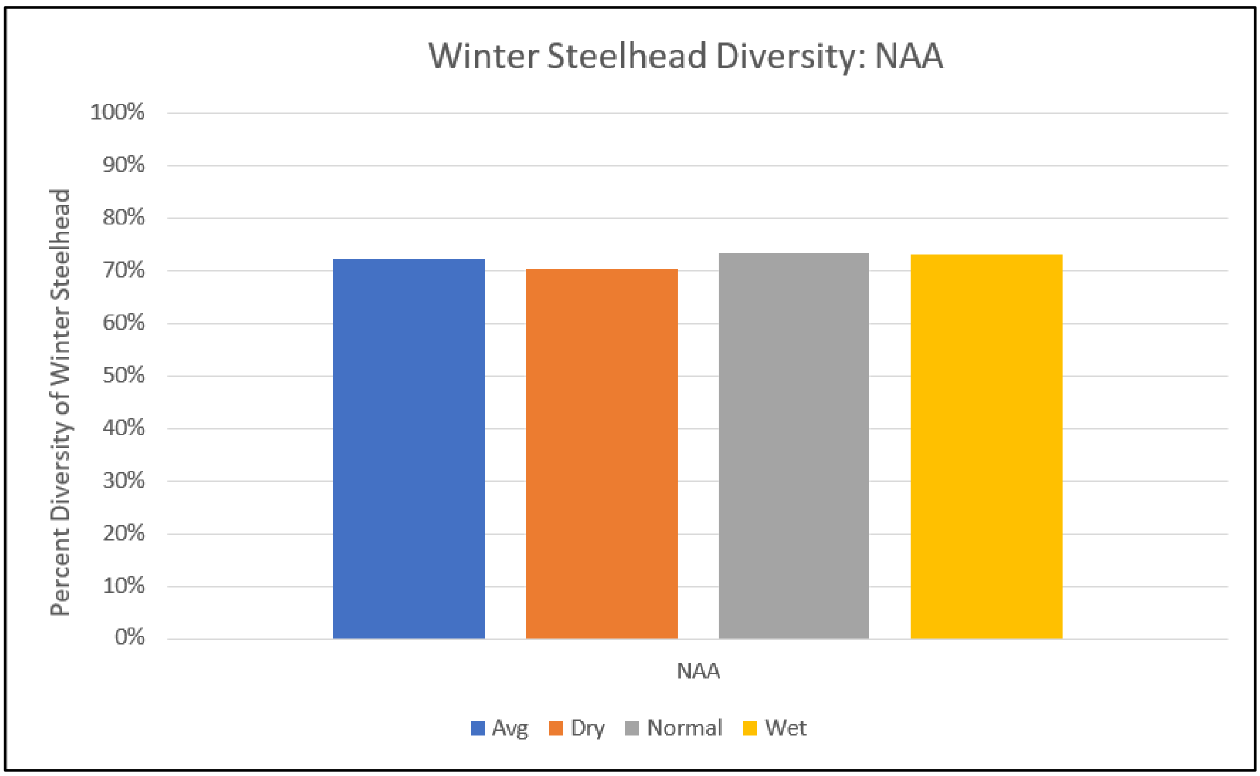


Figure 6-9. Diversity of winter steelhead for the NAA model

Abundance and diversity are both higher in South Santiam basin than North Santiam. While higher abundances are predicted for both North and South Santiam in normal water years under NAA management, North Santiam has slightly higher diversity under wet conditions and South Santiam under normal WY conditions (Table 6-10).

Table 6-9. EDT modeling results for winter steelhead under the NAA alternative.

Result	Water Year	All Subpopulations	North Santiam	South Santiam
Abundance	Wet	8,046	2,997	5,049
Abundance	Dry	8,072	2,971	5,105
Abundance	Normal	9,836	3,632	6,204
Capacity	Wet	8,539	3,173	5,366
Capacity	Dry	8,898	3,250	5,648
Capacity	Normal	10,487	3,874	6,613
Productivity	Wet	17.3	18.0	16.9
Productivity	Dry	10.8	11.6	10.4
Productivity	Normal	16.1	16.0	16.1
Diversity	Wet	73.1%	68.7%	75.7%
Diversity	Dry	70.3%	62.2%	75.1%
Diversity	Normal	73.4%	66.7%	77.4%

6.3.2 Alternatives

Spring Chinook

For spring Chinook, all alternatives showed higher abundance, diversity, and productivity under wet water-year conditions than either dry or normal (Table 5-10 through 5-15). McKenzie basin had higher abundances than any other basin under the three WY conditions modeled. Productivity tended to be higher in McKenzie within a WY condition, while there was some variation in diversity patterns. North Santiam also tended to have higher abundances, diversity, and productivity than South Santiam within a WY.

Under Alt1 management, abundance ranges between 15,756 during a dry year and 20,859 during a wet year (Table 5-10). Diversity was also highest in a wet year at 54.1% among all subpopulations as compared to 47.1% in a dry year. Productivity reflected the same pattern as both abundance and diversity; highest in a wet year at 7.9, lowest in a dry year at 7.4, with a normal year falling in between at 7.6. McKenzie basin had the highest abundance and diversity, followed by North Santiam, Middle Fork Willamette, and South Santiam with the lowest. Diversity in the North Santiam was slightly higher (74.1%) than that in McKenzie in a wet year (66.7%), but lower in a normal (63.6% in North Santiam; 64.2% in McKenzie) or dry year (61.6% in North Santiam; 65.1% in McKenzie).

Table 6-10. EDT modeling results for Spring Chinook under Alt1

Result	Water Year	All Subpopulations	North Santiam	South Santiam	McKenzie	Middle Fork Willamette
Abundance	Wet	20,859	1,953	853	16,480	1,584
Abundance	Dry	15,756	1,368	576	12,852	834
Abundance	Normal	16,738	1,451	624	13,502	1,074
Capacity	Wet	23,872	2,328	1,179	18,467	1,898
Capacity	Dry	18,210	1,652	829	14,566	1,164
Capacity	Normal	19,279	1,740	886	15,244	1,409
Productivity	Wet	7.9	6.2	3.6	9.3	6.0
Productivity	Dry	7.4	5.8	3.3	8.5	3.5
Productivity	Normal	7.6	6.0	3.4	8.8	4.2
Diversity	Wet	54.1%	74.1%	40.4%	66.7%	33.8%
Diversity	Dry	47.1%	61.6%	32.1%	65.1%	26.3%
Diversity	Normal	48.2%	63.6%	34.7%	64.2%	27.9%

Under Alt2a management abundance ranges between 18,453 in a dry year to 23,643 in a wet year (Table 6-11). Productivity was higher under wet conditions at 8.2, with both dry and normal year lower at 7.8. Diversity was higher under a wet year as well at 55.1%, with dry being the lowest at 47.0%, and normal between the two at 50.7%. Again, of the four basins modeled, McKenzie basin had the highest abundance and productivity and South Santiam the lowest. However, South Santiam had slightly higher diversity within each WY condition than Middle Fork Willamette.

Table 6-11. EDT modeling results for Spring Chinook under Alt2a.

Result	Water Year	All Subpopulations	North Santiam	South Santiam	McKenzie	Middle Fork Willamette
Abundance	Wet	23,643	2,057	824	19,338	1,467
Abundance	Dry	18,453	1,473	544	15,571	700
Abundance	Normal	19,929	1,630	566	16,612	1,018
Capacity	Wet	26,937	2,404	1,090	21,668	1,776
Capacity	Dry	21,162	1,750	848	17,596	968
Capacity	Normal	22,848	1,929	830	18,732	1,357
Productivity	Wet	8.2	6.9	4.1	9.3	5.8
Productivity	Dry	7.8	6.3	2.8	8.7	3.6
Productivity	Normal	7.8	6.5	3.1	8.8	4.0
Diversity	Wet	55.1%	68.3%	40.0%	74.7%	33.8%
Diversity	Dry	47.0%	58.0%	30.4%	70.3%	24.6%

Result	Water Year	All Subpopulations	North Santiam	South Santiam	McKenzie	Middle Fork Willamette
Diversity	Normal	50.7%	66.0%	35.1%	70.5%	27.9%

Under Alt2b management, abundances range from 17,440 in a dry year to 21,902 in a wet year (Table 5-12). Productivity was highest under wet conditions at 7.8, and lowest under normal conditions at 7.3. Within individual basins, productivity was highest under wet years and lowest in dry years. Diversity ranged between 46.6% in a dry year and 54.5% under a wet WY. Like Alt2a, McKenzie basin had higher abundance and productivity than any of the other basins and South Santiam the lowest. South Santiam also had slightly higher diversity within a WY condition than Middle Fork Willamette.

The major difference between Alt2 a and b, is that Alt2b did not include a fish-passage structure at Cougar in the McKenzie basin. Comparing abundances in McKenzie basin, they are higher under 2a management than 2b with values of 1,704 more in a wet year, 1,540 more in a normal year, and 993 more in a dry year. Productivity and diversity are slightly higher in the McKenzie basin under Alt2a management than Alt 2b under all WY conditions (Table 6-11 and 6-12).

Table 6-12. EDT modeling results for Spring Chinook under Alt2b.

Result	Water Year	All Subpopulations	North Santiam	South Santiam	McKenzie	Middle Fork Willamette
Abundance	Wet	21,902	2,056	787	17,634	1,474
Abundance	Dry	17,440	1,469	506	14,633	671
Abundance	Normal	18,325	1,630	532	15,072	992
Capacity	Wet	25,136	2,402	1,037	19,917	1,780
Capacity	Dry	20,128	1,746	809	16,645	928
Capacity	Normal	21,173	1,928	784	17,125	1,335
Productivity	Wet	7.8	6.9	4.2	8.7	5.8
Productivity	Dry	7.5	6.3	2.7	8.3	3.6
Productivity	Normal	7.4	6.5	3.1	8.3	3.9
Diversity	Wet	54.5%	68.1%	38.1%	74.2%	33.8%
Diversity	Dry	46.6%	58.1%	28.5%	70.4%	24.4%
Diversity	Normal	50.1%	66.0%	32.3%	70.8%	27.3%

Abundance under Alt3a management ranges between 15,515 and 20,923, with higher abundance in wet years than dry (Table 5-13). Productivity is 7.6 in a wet year and 7.3 in a normal year, with a dry year at 7.5. In individual basins, productivity is slightly higher under normal conditions than dry. Diversity ranges from 39.7% in a dry year to 50.3% in a wet year. McKenzie basin has higher abundance, diversity, and capacity than the other three basins.

North Santiam has higher abundance, diversity, and productivity in all water conditions than South Santiam basin. South Santiam has slightly higher diversity than Middle Fork Willamette under all WY conditions, as well as higher productivity in a wet year. While Middle Fork Willamette has slightly higher productivity than South Santiam in a dry water year.

Table 6-13. EDT modeling results for Spring Chinook under Alt3a.

Result	Water Year	All Subpopulations	North Santiam	South Santiam	McKenzie	Middle Fork Willamette
Abundance	Wet	20,923	1,463	733	17,691	920
Abundance	Dry	15,515	731	446	13,617	531
Abundance	Normal	17,102	861	494	14,800	754
Capacity	Wet	24,106	1,845	975	20,014	1,272
Capacity	Dry	17,914	932	687	15,525	771
Capacity	Normal	19,801	1,097	728	16,862	1,114
Productivity	Wet	7.6	4.8	4.0	8.6	3.6
Productivity	Dry	7.5	4.6	2.8	8.1	3.2
Productivity	Normal	7.3	4.7	3.1	8.2	3.1
Diversity	Wet	50.3%	54.7%	34.9%	76.6%	29.6%
Diversity	Dry	39.7%	41.0%	19.6%	73.2%	17.0%
Diversity	Normal	44.5%	44.6%	29.4%	74.7%	22.8%

Under Alt3b management, abundances ranged between 17,105 in a dry year and 21,422 in a wet year, with normal year abundance at 17,931 (Table 5-15). Diversity ranges between 53.7% in a wet year and 40.3% in a dry year, with normal year diversity at 49.3%. Productivity is highest in a dry year at 7.7 and lowest in a normal year at 7.3. While abundance and diversity are highest in a wet year at subpopulations level, productivity is highest in a dry year. This is because of the much lower diversity in all basins except McKenzie during dry years; McKenzie has the highest productivity among subpopulations, and this is weighted higher under dry years. Individual basin results show slightly higher productivity in wet years than a normal or dry year. Within each basin, abundance, productivity, and diversity are higher under wet years than either normal or dry years. In the North Santiam and Middle Fork Willamette diversity is highest in a wet year and lowest in a dry year. However, in South Santiam basin, diversity is higher during a dry year (21%) than a normal year (18%).

Table 6-14. EDT modeling results for Spring Chinook under Alt3b.

Result	Water Year	All Subpopulations	North Santiam	South Santiam	McKenzie	Middle Fork Willamette
Abundance	Wet	21,422	1,598	612	18,012	1,055
Abundance	Dry	17,105	936	368	15,100	503

Result	Water Year	All Subpopulations	North Santiam	South Santiam	McKenzie	Middle Fork Willamette
Abundance	Normal	17,931	1,338	362	15,306	757
Capacity	Wet	24,734	1,960	927	20,405	1,442
Capacity	Dry	19,646	1,159	572	17,189	725
Capacity	Normal	20,786	1,665	574	17,431	1,116
Productivity	Wet	7.5	5.4	2.9	8.5	3.7
Productivity	Dry	7.7	5.2	2.8	8.2	3.3
Productivity	Normal	7.3	5.1	2.7	8.2	3.1
Diversity	Wet	53.7%	71.8%	26.3%	76.7%	33.4%
Diversity	Dry	40.3%	41.2%	21.0%	75.1%	16.1%
Diversity	Normal	49.3%	69.9%	18.0%	75.8%	25.7%

One of the primary differences between Alt3a and Alt3b alternatives is the management of fish passage at Cougar in the McKenzie basin. Abundances in the McKenzie basin are higher under Alt3b management than Alt3a management, increasing 321 in a wet year, 531 in a normal year, and 1,483 in a dry year. Productivity is slightly higher (+0.1) under Alt 3a management than Alt3b in both wet and dry years and remains the same under normal WY conditions. In the McKenzie, diversity is slightly higher (0.1+) under Alt 3b management than 3a.

The other basins modeled also show different results under the Alt3a and 3b alternatives. Abundances, diversity, and productivity are higher for North Santiam and Middle Fork Willamette under Alt3b management, particularly under wet and normal year conditions (Table 6-13 and 6-14). Abundances, diversity, and productivity are higher under Alt3a management in the South Santiam basin, except for productivity in dry years which are equivalent (2.8) under both alternatives.

Under Alt4 management, abundances range from 18,493 in a dry year to 23,595 in a wet year (Table 6-15). Productivity is higher in a wet year at 8.0, than either dry or normal years at 7.6. Diversity is also higher in a wet year at 56.9% than either normal (54.2%) or dry (49.1%) years. Individual basin results reflect this same pattern with abundance, productivity, and diversity being higher in wet years and lower in dry years, with values lying between in normal years.

McKenzie basin has the highest abundance with North Santiam, Middle Fork Willamette, and South Santiam following. North Santiam has higher abundance, productivity, and diversity than South Santiam in all WY conditions.

Table 6-15. EDT modeling results for Spring Chinook under Alt4.

Result	Water Year	All Subpopulations	North Santiam	South Santiam	McKenzie	Middle Fork Willamette
Abundance	Wet	23,595	1,994	572	19,122	1,983
Abundance	Dry	18,493	1,427	355	15,358	1,282
Abundance	Normal	19,676	1,569	400	16,173	1,566
Capacity	Wet	26,958	2,351	749	21,435	2,423
Capacity	Dry	21,290	1,711	553	17,378	1,648
Capacity	Normal	22,651	1,874	547	18,259	1,972
Productivity	Wet	8.0	6.6	4.2	9.3	5.5
Productivity	Dry	7.6	6.0	2.8	8.6	4.5
Productivity	Normal	7.6	6.1	3.7	8.8	4.9
Diversity	Wet	56.9%	68.2%	33.0%	73.1%	46.8%
Diversity	Dry	49.1%	57.7%	21.0%	70.3%	38.8%
Diversity	Normal	54.2%	66.0%	28.4%	71.0%	44.5%

Comparing the alternatives with one another, Alt2a commonly had among the highest abundances in all WY conditions, except for Middle Fork Willamette basin where Alt4 had the highest abundances (Table 6-9 through 6-14). It varied among basins and WY conditions, on which alternative had the highest diversity or productivity.

In the McKenzie basin, abundances were highest in all WY conditions under Alt2a operations (Table 3-4). Diversity was highest under Alt3a/Alt3b operations in the McKenzie in wet years exceeding 76%, with Alt2a/Alt2b close behind at >74% for both alternatives. In a dry year, all alternatives except for Alt1, had diversity above 70% with Alt3b at the highest at 75%. In a normal year, Alt3a and Alt3b also had the highest diversity ranging between 74.7-75.8%.

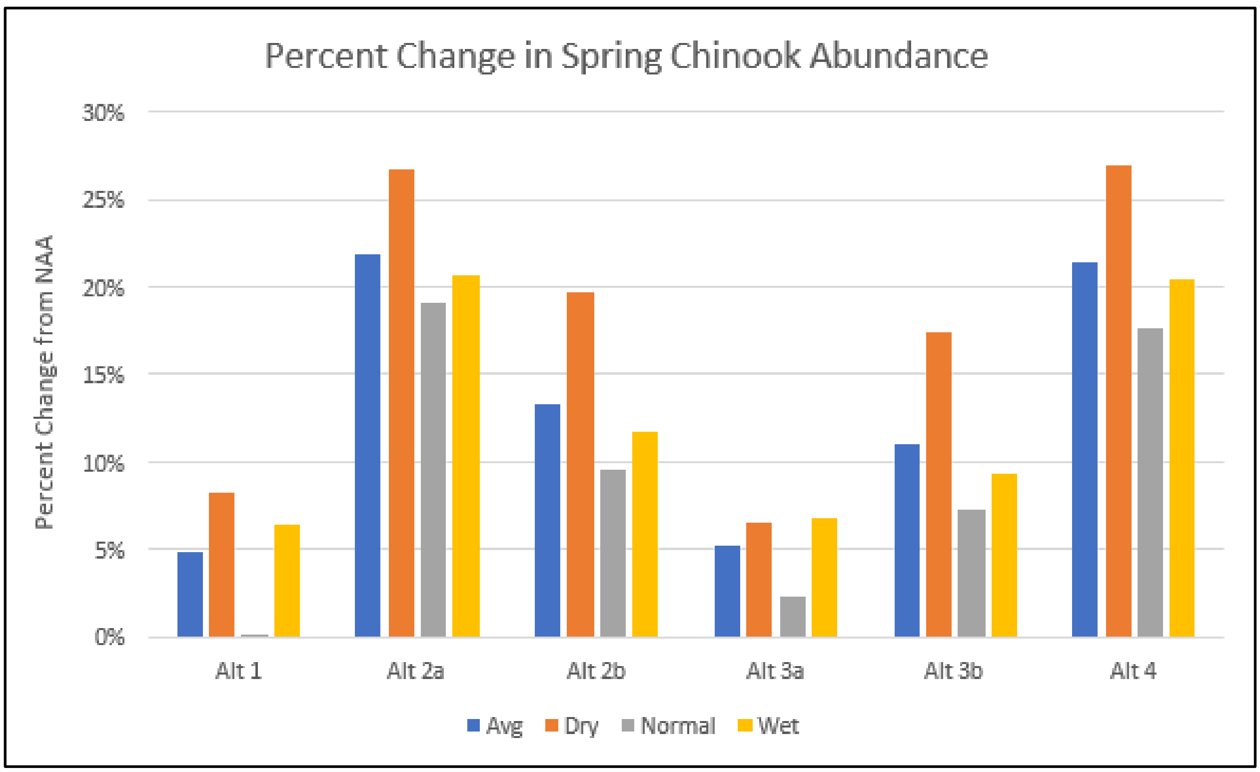
Productivity in the McKenzie was highest in wet and normal years under Alt2a, Alt1, and Alt4 management, which did not differ from the No Action at 9.3 in wet years and 8.8 in normal. In a dry year, Alt2a had the highest productivity at 8.7 with all other alternatives ranging between 8.1 and 8.6.

In the Middle Fork Willamette basin, Alt4 had the highest abundances and diversity under all WY conditions (Table 3-8). Productivity was highest under Alt4 in both dry (4.5) and normal (5.5) years, but highest under Alt1 in wet years at 6.0.

In the Santiam basin, North Santiam had the highest abundances under Alt2a, Alt2b, and Alt4 in all WY conditions, with Alt1 also being similarly high in wet years (Table 3-3 through 3-8). In the South Santiam, Alt2a, Alt2b, and Alt1 had the highest abundances in all WY conditions. In both basins, diversity was highest under Alt1 in most WY conditions with Alt3b highest in North Santiam in normal years. Productivity was highest under Alt2a and Alt2b, in all WY conditions in the North Santiam. In South Santiam, Alt 4, Alt2a, and Alt2b had the highest productivity among the alternatives in a wet year ranging from 4.1-4.7, but this did not exceed productivity under No Action which was 5.9. In a dry year, productivity was highest under Alt1. And in a normal year productivity was highest under Alt4 at 3.7, which again did not exceed that of No Action at 4.9.

In comparison with the No Action alternative, Alt2a, Alt2b, and Alt4 have the highest average percentage increase in spring Chinook abundance and diversity from NAA values (Figure 3-5 and 3-6). Increases were higher under dry year conditions than either wet or normal for both abundance and diversity. This reflects the higher relative change between the abundance and diversity under the No Action management alternative and those of the other proposed management alternatives in dry year conditions than either normal or wet years. Within a dry year, among all alternatives, abundance increased an average of 18% (range of 7-27%) while increasing an average of 9% in a normal year (range 0-18%) and 13% in a wet year (range 7-21%). In a dry year, among all alternatives, diversity increased an average of 47% over No Action management (range 30-60%), and less than 9% in a normal (range -2-19%) or wet (range 0-14%) year.

Table 6-16. Percent change in spring Chinook abundance of each alternative from NAA model.



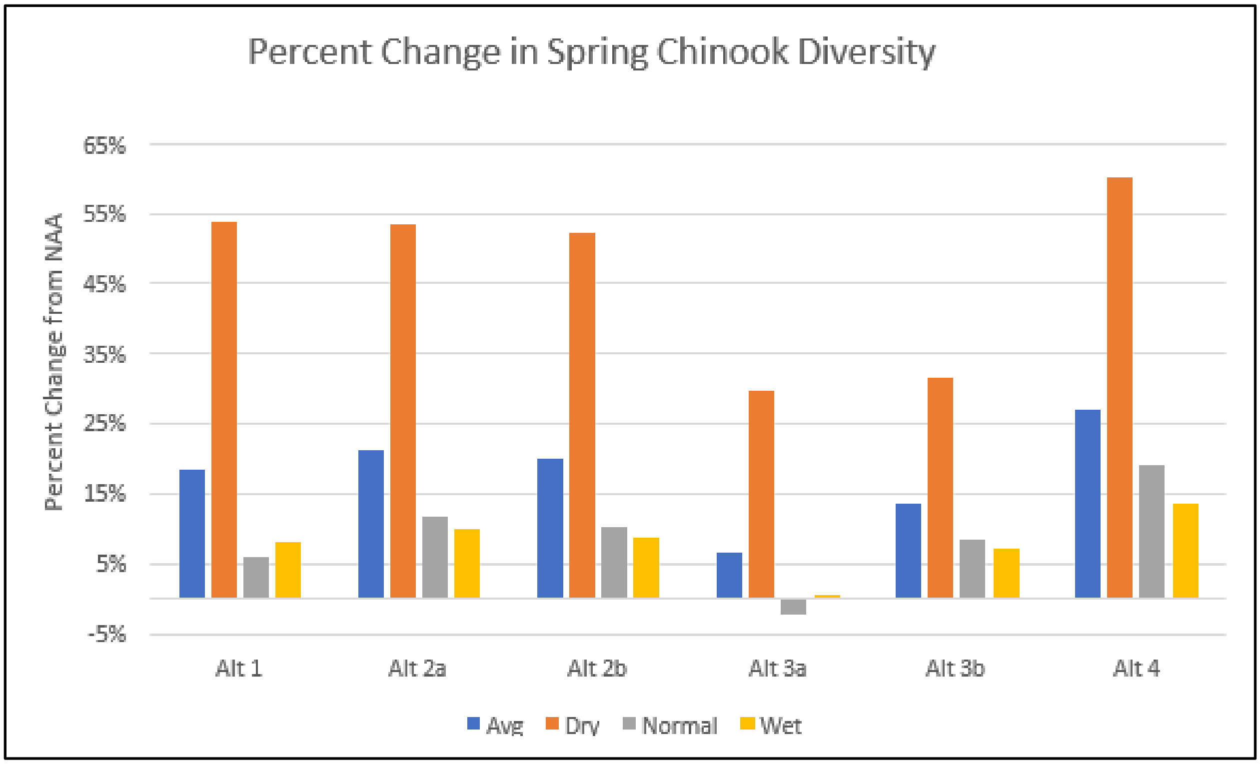


Figure 6-10. Percent change in spring Chinook diversity of each alternative from NAA model.

Winter Steelhead

Under the Alt1 alternative, abundance was highest under wet conditions at 15,515 and lowest under normal conditions at 14,335 (Table 3-9). Within each water-year condition and overall, abundance of steelhead is higher in the South Santiam basin than the North Santiam. South Santiam has highest abundance in a wet water year (8,196), and lowest abundance in a dry year (7,336. Abundance is higher in the North Santiam basin under wet conditions as well (7,303) but lowest in a normal year 6,858, respectively.

Diversity is high overall, exceeding 88% in all years and basins with the highest diversity occurring in wet years. North Santiam basin has higher diversity, ranging from 92.9% (normal) to 99.3% (wet), compared to South Santiam, which ranges from 88.3% (dry) to 90.8% (wet).

Productivity is also higher under wet year conditions at 22.4, followed by dry year productivity at 18.3 and the lowest during normal years at 17.8. This trend is also reflected in the individual basin with North Santiam having higher productivity than South Santiam in all water year conditions, particularly under wet conditions at 25.9. North Santiam also has higher productivity under wet conditions than either dry (24.0) or normal (23.1). South Santiam basin has higher diversity and productivity in a wet water year; however, both parameters are slightly higher (+0.1) in a normal water year than a dry water year (Table 6-17).

Table 6-17. EDT modeling results for winter steelhead under Alt1.

Result	Water Year	All Subpopulations	North Santiam	South Santiam
Abundance	Wet	15,515	7,303	8,196
Abundance	Dry	14,408	6,995	7,336
Abundance				
Capacity	Wet	16,240	7,596	8,644
Capacity	Dry	15,242	7,299	7,943
Capacity	Normal	15,212	7,168	8,043
Productivity	Wet	22.4	25.9	19.3
Productivity	Dry	18.3	24.0	13.1
Productivity	Normal	17.8	23.1	13.0
Diversity	Wet	94.0%	99.3%	90.8%
Diversity	Dry	90.2%	93.5%	88.3%
Diversity	Normal	90.1%	92.9%	88.4%

Under Alt2a management, overall abundance is highest in a wet year at 15,125 and lowest in a dry year at 13,860 (Table 6-18). Abundances in North and South Santiam basins are also higher in wet years than either normal or dry, with the lowest abundances in either basin occurring in dry years. Productivity is highest in wet years, both overall at 23.7 and within individual basins (26.7 in North Santiam and 20.9 in South Santiam). Diversity remained high in all water years, remaining >90% in all water year conditions. In the North Santiam, diversity was lowest under dry conditions at 93.6% and highest at 99.3% in wet conditions. In South Santiam, diversity ranged between 90.7% in a wet year and 88.4% in a dry year.

Table 6-18. EDT modeling results for winter steelhead under Alt2a.

Result	Water Year	All Subpopulations	North Santiam	South Santiam
Abundance	Wet	15,125	7,421	7,695
Abundance	Dry	13,860	6,997	6,745
Abundance	Normal	14,175	7,068	7,025
Capacity	Wet	15,792	7,709	8,082
Capacity	Dry	14,677	7,293	7,383
Capacity	Normal	15,011	7,380	7,632
Productivity	Wet	23.7	26.7	20.9
Productivity	Dry	18.0	24.6	11.6
Productivity	Normal	18.0	23.7	12.6
Diversity	Wet	93.9%	99.3%	90.7%
Diversity	Dry	90.4%	93.6%	88.4%

Result	Water Year	All Subpopulations	North Santiam	South Santiam
Diversity	Normal	91.8%	95.7%	89.5%

Under Alt2b management, abundance ranged bewteen 15,168 and 13,879 (Table 3-11). North Santiam had fairly similar results in abundances, diversity, and productivity under Alt2a and Alt2b managment. South Santiam had slighlty higher abundances ranging from 30-63 more individuals under Alt2b than Alt2a (Table 3-10 and 3-11). Productivity was also slighlty increased in the South Santiam under Alt2b management as compared to Alt2a management, and diversity remained fairly equivalent.

Table 6-19. EDT modeling results for winter steelhead under Alt2b.

Result	Water Year	All Subpopulations	North Santiam	South Santiam
Abundance	Wet	15,168	7,421	7,740
Abundance	Dry	13,879	6,993	6,775
Abundance	Normal	14,229	7,069	7,088
Capacity	Wet	15,833	7,709	8,124
Capacity	Dry	14,682	7,286	7,395
Capacity	Normal	15,059	7,381	7,678
Productivity	Wet	23.8	26.7	21.2
Productivity	Dry	18.3	24.9	11.9
Productivity	Normal	18.2	23.7	13.0
Diversity	Wet	93.9%	99.3%	90.7%
Diversity	Dry	90.2%	93.1%	88.5%
Diversity	Normal	91.8%	95.7%	89.5%

Under Alt3a management, abundance ranged from 12,930 in a wet year to 10,941 in a dry year (Table 3-12). Abundance was higher in the South Santiam than the North Santiam basin within each WY condition. Productivity was higher at 20.1 in wet years than either dry (12.6) or normal (13.0) years (Table 3-12). In the North Santiam basin, productivity was higher in a dry year at 16.0 than a normal year at 14.9. South Santiam basin had higher productivity in a normal year at 11.7 than a dry year at 10.4. Diversity was highest across the board during wet years, followed by normal and then dry.

Table 6-20. EDT modeling results for winter steelhead under Alt3a.

Result	Water Year	All Subpopulations	North Santiam	South Santiam
Abundance	Wet	12,930	6,094	6,836
Abundance	Dry	10,941	5,132	5,788
Abundance	Normal	11,547	5,368	6,172
Capacity	Wet	13,605	6,415	7,190

Result	Water Year	All Subpopulations	North Santiam	South Santiam
Capacity	Dry	11,882	5,474	6,407
Capacity	Normal	12,505	5,753	6,752
Productivity	Wet	20.1	20.0	20.3
Productivity	Dry	12.6	16.0	10.4
Productivity	Normal	13.0	14.9	11.7
Diversity	Wet	92.9%	99.3%	89.1%
Diversity	Dry	85.1%	82.4%	86.6%
Diversity	Normal	88.9%	91.1%	87.6%

Under Alt3b management, abundance was highest in a wet year at 12,568, and lowest in a dry year at 8,791. North and South Santiam also had highest abundance in wet years. North Santiam had lower abundance than South Santiam basin within each water year. Productivity and diversity followed the same pattern across subbasins, with the highest numbers in a wet year, followed by normal, and the lowest in a dry year. North Santiam had higher productivity and higher diversity than South Santiam basin within each WY condition.

Table 6-21. EDT modeling results for winter steelhead under Alt3b.

Result	Water Year	All Subpopulations	North Santiam	South Santiam
Abundance	Wet	12,568	6,021	6,546
Abundance	Dry	8,791	3,704	5,032
Abundance	Normal	11,135	5,588	5,469
Capacity	Wet	13,228	6,322	6,906
Capacity	Dry	9,736	4,006	5,730
Capacity	Normal	12,122	5,984	6,138
Productivity	Wet	20.0	21.0	19.2
Productivity	Dry	10.3	13.3	8.2
Productivity	Normal	12.3	15.1	9.2
Diversity	Wet	92.4%	99.3%	88.3%

Abundance under Alt4 management ranged from a high of 14,261 in a wet year to a low of 12,966 under dry conditions (Table 3-14). Productivity was also higher in a wet year at 24.4 than a dry year at 18.1. Diversity had a similar pattern as both abundance and productivity. North Santiam had higher abundance, diversity, and productivity than South Santiam within a WY condition. Diversity was higher in North Santiam, exceeding 93% in all WY conditions, while South Santiam remained below 79%.

Table 6-22. EDT modeling results for winter steelhead under Alt4.

Result	Water Year	All Subpopulations	North Santiam	South Santiam
Abundance	Wet	14,261	7,347	6,913
Abundance	Dry	12,966	6,933	5,953
Abundance	Normal	13,356	6,991	6,318
Capacity	Wet	14,870	7,637	7,232
Capacity	Dry	13,724	7,235	6,489
Capacity	Normal	14,130	7,307	6,823
Productivity	Wet	24.4	26.3	22.6
Productivity	Dry	18.1	24.0	12.1
Productivity	Normal	18.3	23.1	13.5
Diversity	Wet	86.2%	99.3%	78.4%
Diversity	Dry	83.3%	93.7%	77.1%
Diversity	Normal	84.4%	95.7%	77.7%

Winter steelhead perform better (higher abundances and diversity) during wet WYs under the proposed alternatives, which is a shift from better performance under normal conditions under No Action management (Table 3-2; Table 3-9 through 3-14). Abundances in North Santiam basin were highest, and comparable to each other, under Alt2a/2b management in wet and normal conditions and under Alt1/Alt2a/Alt2/Alt4 managements in dry year conditions. Under Alt2b management, abundance ranged between 15,168 and 13,879 (Table 3-11). North Santiam had similar results in abundances, diversity, and productivity under Alt2a and Alt2b management. South Santiam had slightly higher abundances ranging from 30-63 more individuals under Alt2b than Alt2a (Table 3-10 and 3-11). Productivity was also slightly increased in the South Santiam under Alt2b management as compared to Alt2a management, and diversity remained equivalent.

South Santiam had highest abundances under Alt1 management in all WY conditions (Table 3-9). The alternative management scenario that had the highest diversity values varied for each basin depending on WY condition. In a normal water year, South Santiam had the highest diversity under Alt2b (89.5%) and North Santiam under Alt3b (95.2%). In a dry year, South Santiam had the highest diversity under Alt2a (88.5%) and North Santiam under Alt3b (95.2%). In a wet year, South Santiam had the highest diversity under Alt1 (90.8%), and North Santiam's diversity did not vary, remaining at 99.3% diversity under all alternatives. Productivity in the North Santiam was highest under Alt2b in under normal conditions (23.7), as well as dry conditions (24.9). Under wet conditions, Alt2a and Alt2b had equivalent productivity in the North Santiam basin (26.7). In the South Santiam basin, the highest productivity under the proposed new alternatives was 13.5 under Alt4 management in a normal water year, 22.6 under Alt4 management in a wet water year, and 13.1 under Alt1 management in a dry water year.

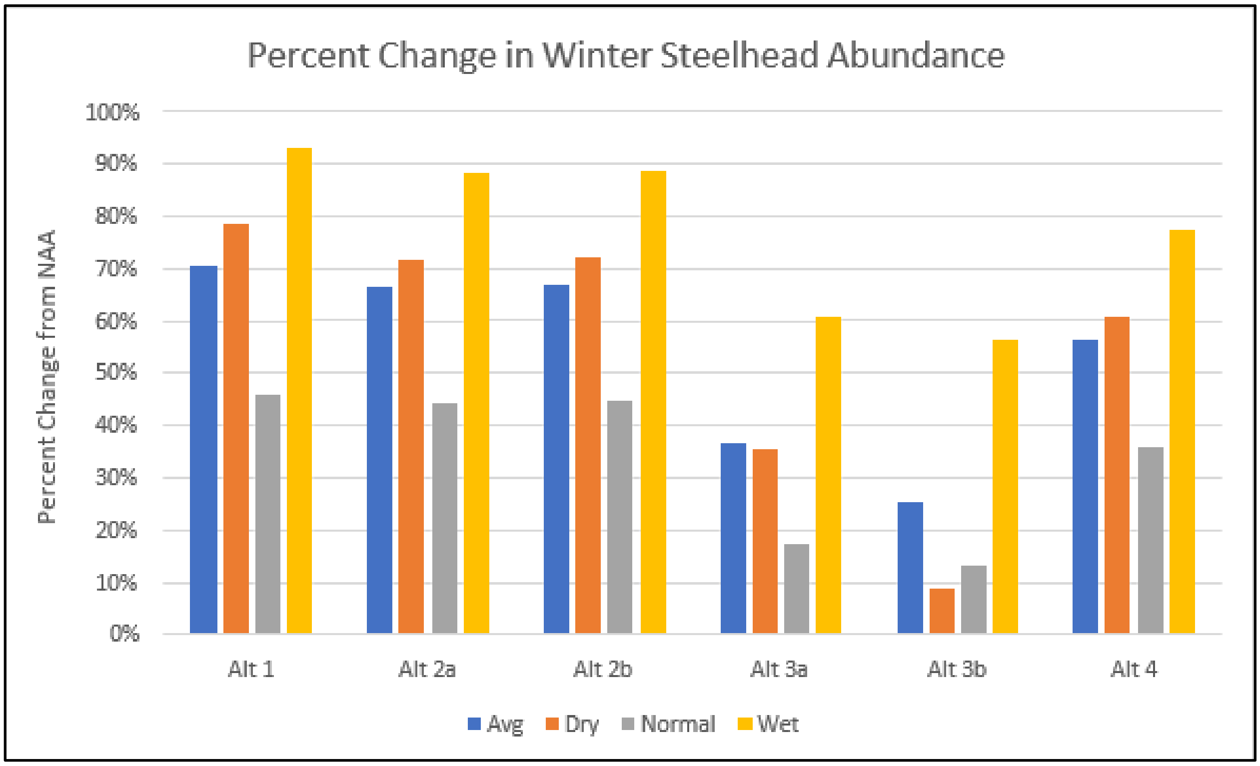


Figure 6-11. Percent change in winter steelhead abundance of each alternative from NAA model.

Alt1, Alt2a, Alt2b had the highest average percent increase in winter steelhead abundances over NAA, as well as within each WY condition (Figure 3-7). Increases in abundance from those under No Action management, were higher during wet year conditions for all management alternatives. Within a wet year, among alternatives, abundances increased an average of 77% (range 56-93%), while increasing an average of 34% in a normal year (range 13-46%), and 55% in a dry year (range 9-78%). Diversity of winter steelhead showed similar increases among all scenarios as compared to No Action with percent increases ranging between 15-29%. (Figure 3-8). Except for Alt3a, the percent increases in diversity from No Action management were highest in both dry and wet years than normal. Under Alt3a management, normal water years have the same percent increase in diversity of 21% over No Action management.

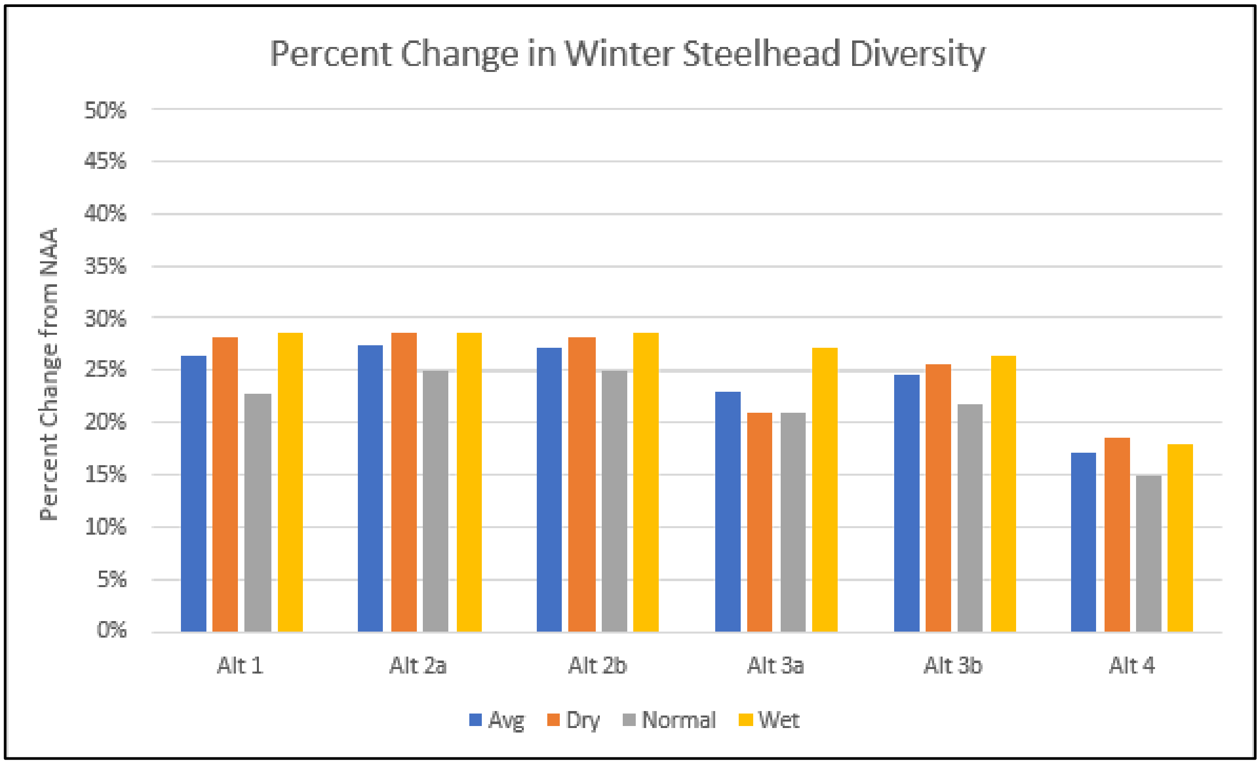


Figure 6-12. Percent change in winter steelhead diversity of each alternative from NAA model.

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6.4 CHAPTER 6-ATTACHMENT A: SANTIAM MAXIMUM TEMPERATURE RATINGS

Table A-1. Santiam Basin EDT reach temperature data and/or proxy rating descriptions

EDT Reach (Proxy Reach)	EDT Reach with Proxy Ratings
Blowout-3	Blowout-2
Brietenbush-3	NF Breitenbush-1, SF Breitenbush-1
Canal-1	Crabtree-6, Packers Gulch-1, Scott-1
Canyon-1	Wiley-3
Canyon-2	Bald Peter-1, Little Wiley-1, SF Scott-1, Wiley-4, Wiley-5
Crabtree-1	Bilyeu-1, Crabtree-2, Crabtree-3, Crabtree-4, Roaring River-1, Thomas-1, Thomas-2, Thomas-3
East Humbug-1	Evans-1, Fish-1, Little Sinker-1, Sinker-1
Hamilton-1	Ames-1, Hamilton-2
Hamilton-2	Wiley-1
Humbug-2	Humbug-1
Little North Santiam 2	Little North Santiam-3
McDowell-1	Crabtree-5, Hamilton-3, SF Crabtree-1, Thomas-4, Thomas-5, Wiley-2
Middle Santiam 6	Middle Santiam-7
Moose-1	Lewis-2, South Santiam-15
North Santiam-1	Chehulpum-1, North Santiam-2
North Santiam 4	North Santiam-5
North Santiam-8	North Santiam-9
North Santiam-9	North Santiam-10
North Santiam 14	Little North Santiam-4
North Santiam-17	Marion-1
Owl-1	Bald Barney-1, Bear-1, Camp-1, Cruiser-1, Green Mountain-1, Jack-1, Jackson-1, RB Trib 0004 (Roaring River), Rock-1 (Crabtree), Sixes-1, WF Rock-1 (Crabtree)
Quartzville 5	Quartzville-4

EDT Reach (Proxy Reach)	EDT Reach with Proxy Ratings
Rock-1	Cedar-1, Elkhorn-1, Little North Santiam-5, Little Rock-1, Mad-1, Minto-1
Santiam-2	Santiam-1
South Santiam-1	South Santiam-3
South Santiam-2	South Santiam-4, South Santiam-5, South Santiam-6
South Santiam-7	South Santiam-8
South Santiam-13	South Santiam-14
Stout-1	Snake-1
Two Girls-1	Harter-1, Suttle Camp Creek-1, White Rock-1

Note: * Summary of EDT reaches in the Santiam Basin with empirical temperature data used for maximum temperature ratings for current conditions and those reaches without measurement that were rated by proxy.

6.5 CHAPTER 6 ATTACHMENT B: MAXIMUM WATER TEMPERATURE RATINGS

Attachment B: Maximum Water Temperature Ratings

MEMORANDUM

To: USACE, Willamette Basin Project

From: Laura McMullen and Janel Sobota, ICF

Date: Updated February 22, 2022

Re: EDT Ratings of Maximum Temperature in the Willamette Basin Under Alternative Scenarios

Here we present the maximum water temperature EDT (Ecosystem Diagnosis and Treatment) ratings under the No Action Alternative (NAA), Alternative 1 (Alt1), Alternative 2a (Alt2a), Alternative 2b (Alt2b), Alternative 3a (Alt3a), Alternative 3b (Alt3b) and Alternative 4 (Alt4) scenarios for Wet (2011), Dry (2015), and Normal (2016) water-year conditions. Water temperature in the Willamette Basin was modeled using the CE-QUAL-W2 program and was provided to ICF by USACE (Contact: Laurel Stratton). For each management scenario and water-year condition (21 alternative scenarios total), mainstem water temperatures were calculated for the Willamette River from Willamette River Falls upstream to where the Willamette forks into the Middle Fork and Coastal Fork; the mouth of the Middle Fork of the Willamette River to Dexter Dam and then above Dexter to Hills Creek Reservoir; the mouth of Fall Creek (Middle Fork Willamette Basin) to Fall Creek Dam; McKenzie River from the mouth and up into the South Fork, stopping at Blue River Dam; South Santiam from the mouth to Foster Dam; Santiam/North Santiam from where the Santiam flows into the Willamette and up into the North Santiam to Detroit Dam.

Maximum Water Temperature in EDT Modeling

Output from the CE-QUAL-W2 model was then used to inform the EDT model. For EDT, maximum water temperature is one of many attributes used to define the conditions a fish (usually salmonid) will face on their way through the stream system. Water temperature is a crucial consideration in salmonid health and survival. Stream temperatures both directly and indirectly affect salmonid health as each species have a definite range of tolerances at different life stages and temperature affects different environmental factors, such as dissolved oxygen (lower concentrations at higher temperatures), which in turn impact salmonids. In EDT, maximum water temperature for each reach is incorporated into the model in monthly increments through a rating system based on salmonid health. The ratings are scaled from zero (best conditions) to four (worst conditions; Table 1). In general, summer months will have higher ratings when compared to than cooler months. For most river sections temperature data was derived from April through October and encompassed the warmest months of a year. For the river section of the Middle Fork of the Willamette above Lookout Point Lake up until Dexter Dam, maximum temperatures were calculated for all months of the year.

EDT ratings were assigned to each month's modeled maximum water temperature according to the protocol outlined in Table 1. The CE-QUAL-W2 output included 2,297 sites the system georeferenced to river miles. These monthly ratings were averaged for all the sites within an EDT reach, and each reach encompasses multiple sites. Ratings for 100 EDT reaches were developed for each of the 21 scenarios considered using the modeled maximum water temperatures. Graphs for all reaches are included in the appendices of this memo and initial results are described below.

Table 6-23. EDT ratings (Index Values) for maximum water temperatures.

Index 0	Index 1	Index 2	Index 3	Index 4
Warmest day < 10 C	Warmest day>10 C and <16 C	> 1 d with warmest day 22-25 C or 1-12 d with >16 C	> 1 d with warmest day 25-27.5 C or > 4 d (non-consecutive) with warmest day 22-25 C or >12 d with >16 C	> 1 d with warmest day 27.5 C or 3 d (consecutive) >25 C or >24 d with >21 C

Willamette River (Mainstem from Willamette Falls to Coastal/Middle Fork split)

There are 15 EDT reaches within this stretch of the mainstem of Willamette River, starting at Willamette Falls and ending at the split between the Coastal and Middle Fork Willamette (Appendix B-1).

Middle Fork Willamette River and Fall Creek

The Middle Fork of the Willamette, from the juncture with Willamette River until Dexter Dam, there are five EDT reaches (FR MFW-01 through -05; Appendix B-2).

McKenzie River

The river section from the mouth of the McKenzie River and into the South Fork up until the Blue River Dam, has 52 EDT reaches (Appendix B-3)

Santiam/North Santiam/South Santiam

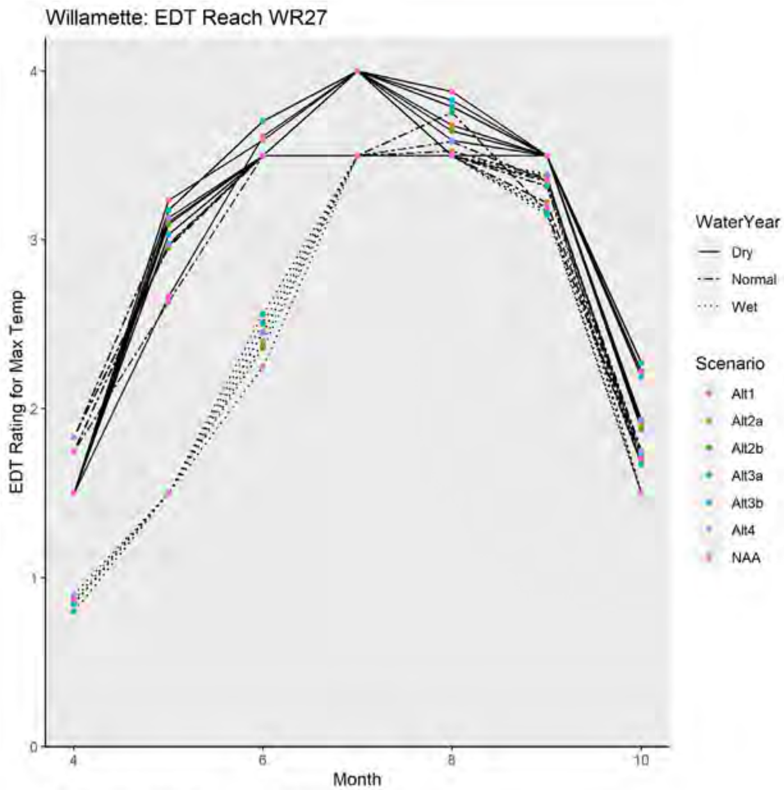
Where the mouth of the Santiam River upstream until the split between North and South Forks, there are two EDT Reaches. There are 11 reaches from the start of the North Santiam fork up to Detroit Dam. There are from seven reaches from the start of South Santiam until Foster Dam (Appendix B-4).

Maximum Water Temperature Ratings

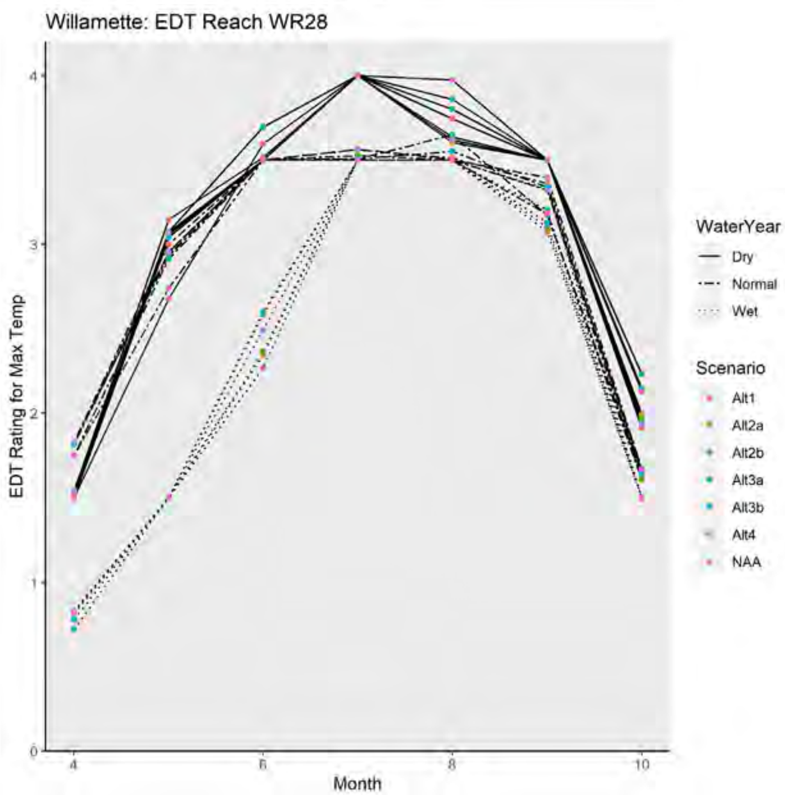
Graphs of the reaches maximum water temperature ratings for the modeled months of April through October are included in the following appendices. No matter which scenario, ratings generally increase from April to a peak in August/September and decline in October. In most reaches, wet-year conditions have lower (better temperature conditions for salmonids) in the spring and early summer than other water years no matter which scenario. Dry- and Normal water-year conditions generally mirror each other with similar ratings that vary by scenario.

Chapter 6 Attachment B-1: Willamette River (Willamette Falls to Coastal/Middle Fork split)

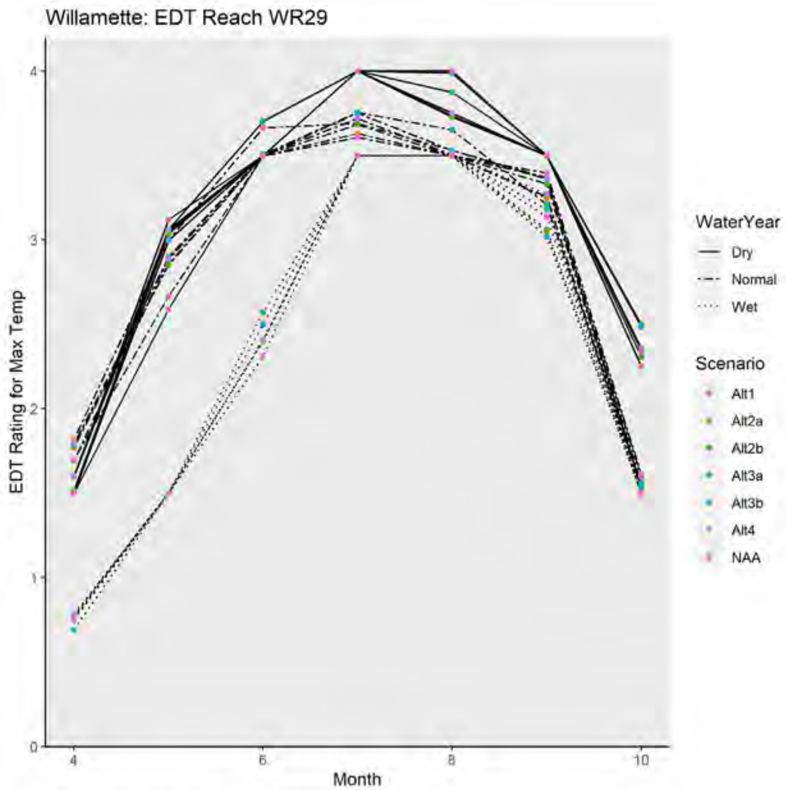
Graphs are organized downstream to upstream: EDT Reach WR27 is just above Willamette Falls; EDT Reach WR41 is where the Willamette splits into the Coast and Middle forks.

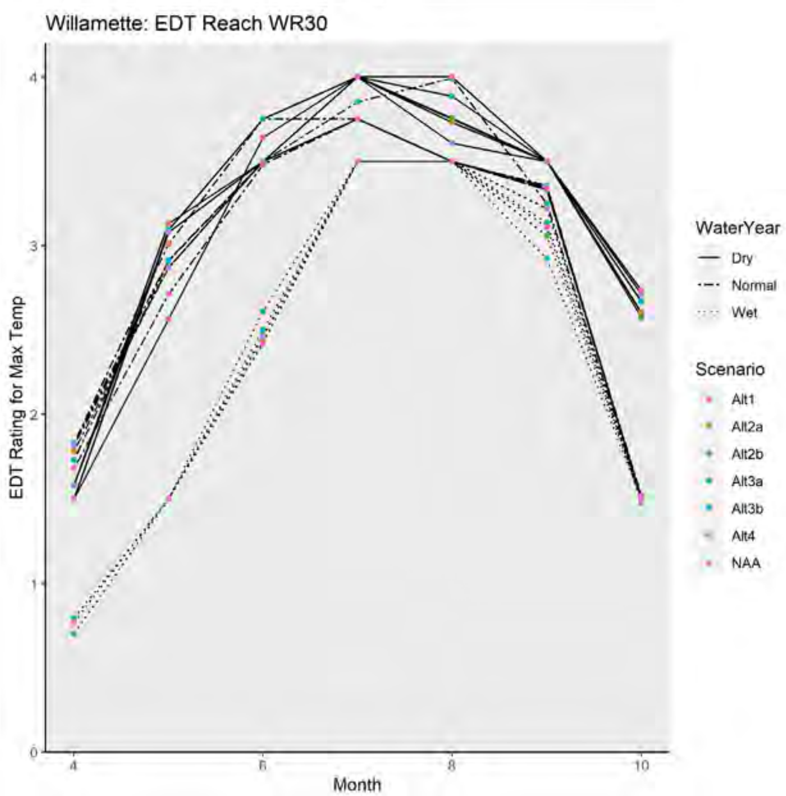


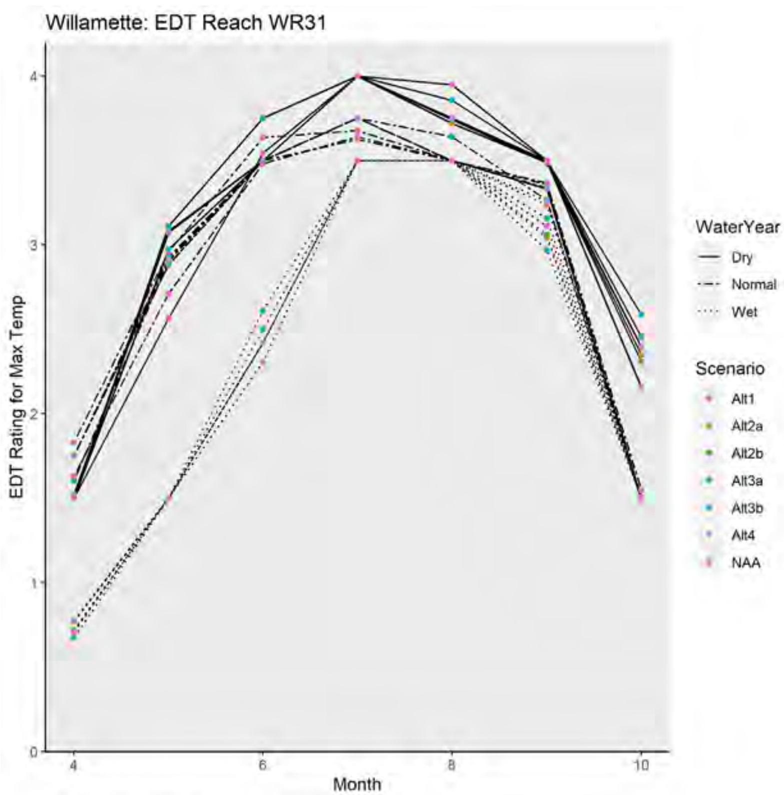
Willamette: EDT Reach WR27
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA



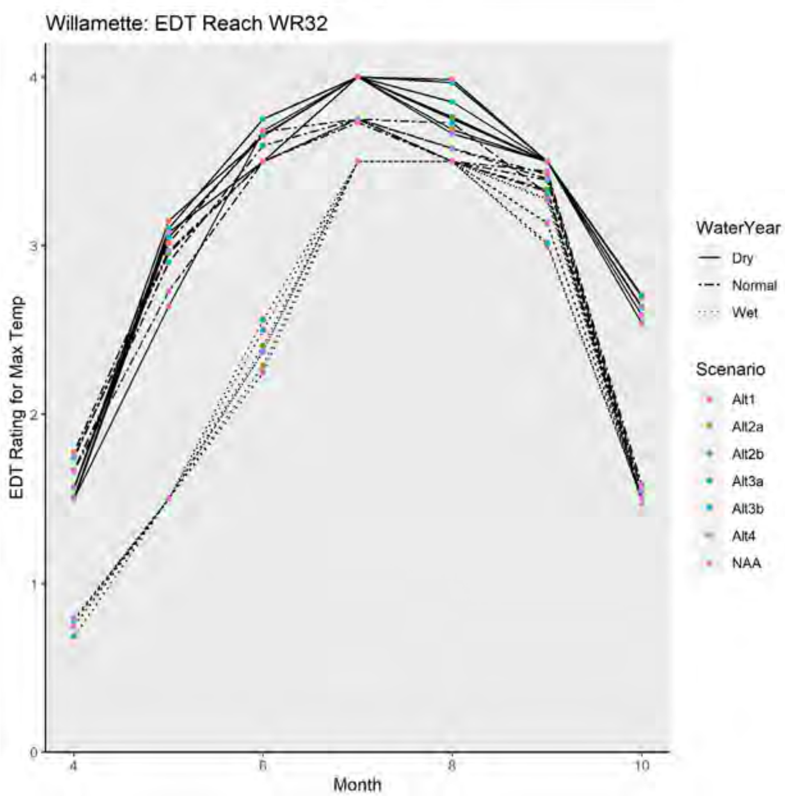
Willamette: EDT Reach WR28
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


Willamette: EDT Reach WR29
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

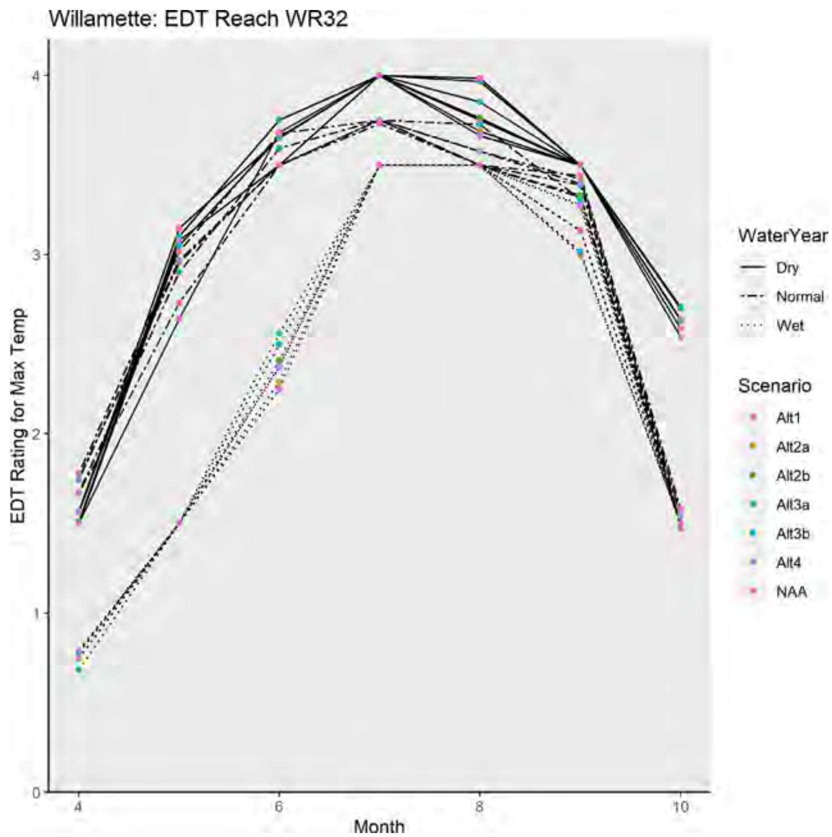

Willamette: EDT Reach WR30
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA



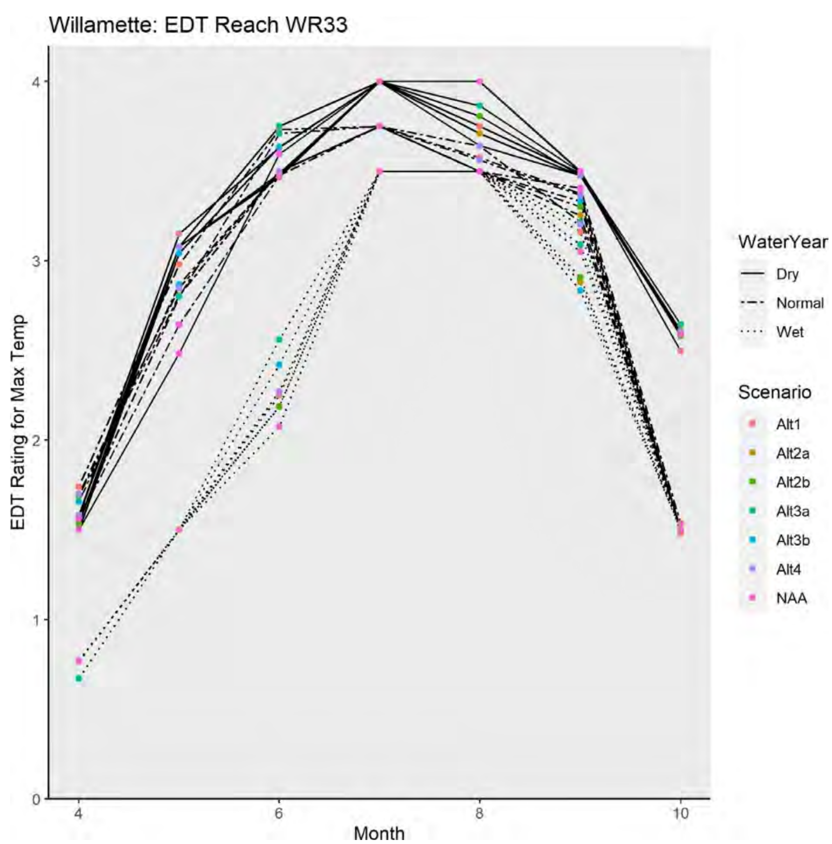
Willamette: EDT Reach WR31
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA



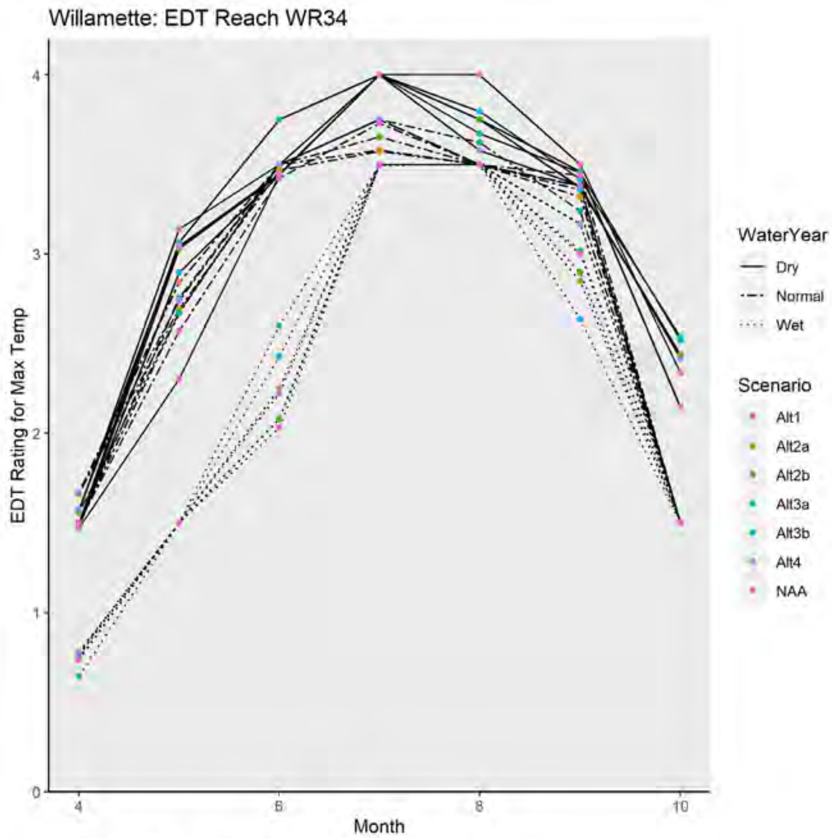
Willamette: EDT Reach WR32
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

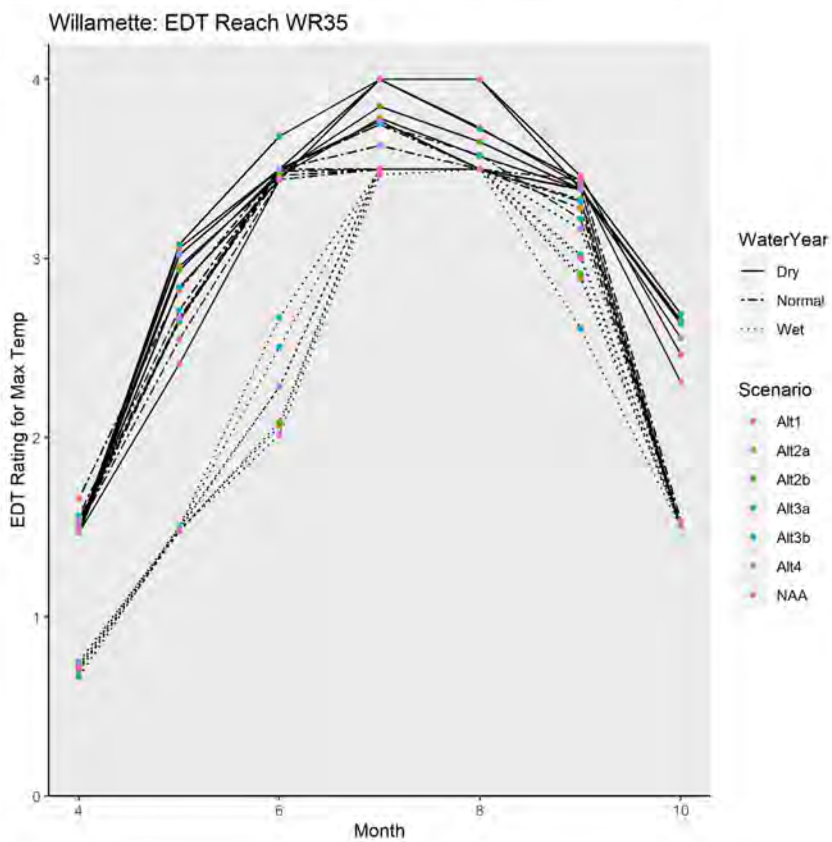


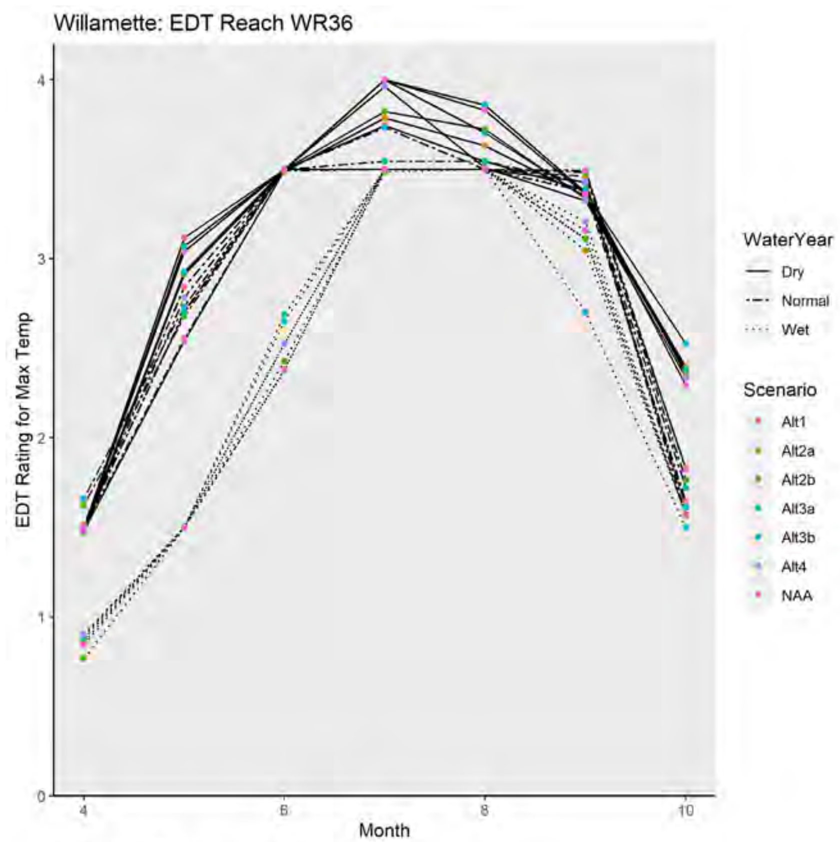
Willamette: EDT Reach WR32
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA



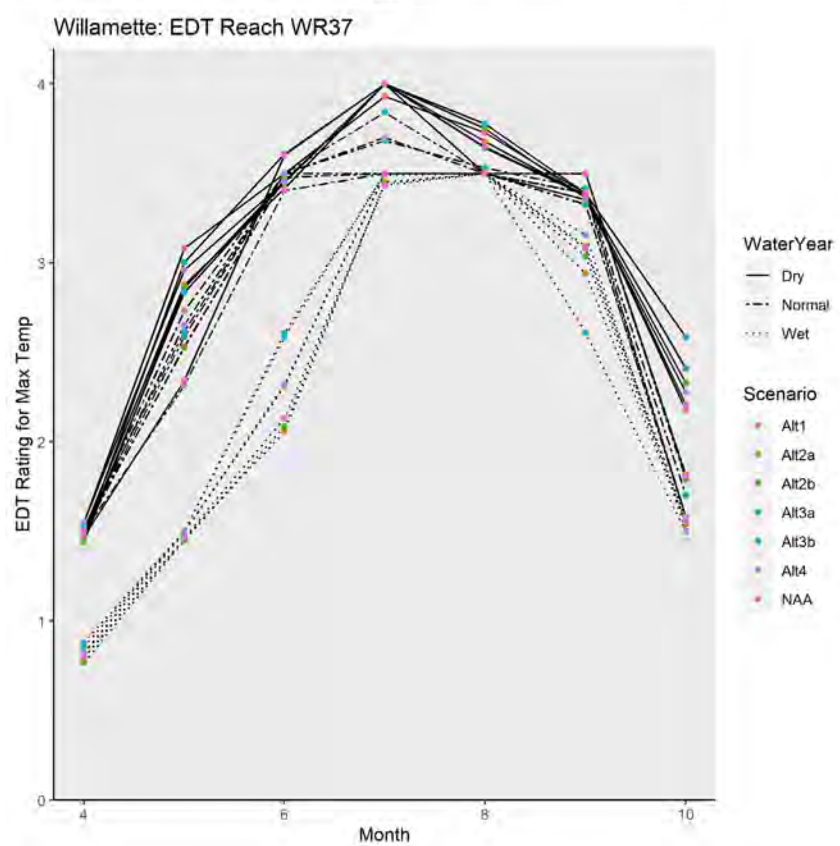
Willamette: EDT Reach WR33
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


Willamette: EDT Reach WR34
EDT Rating for Max Temp
0
1
2
3
4
4
6
8
10
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

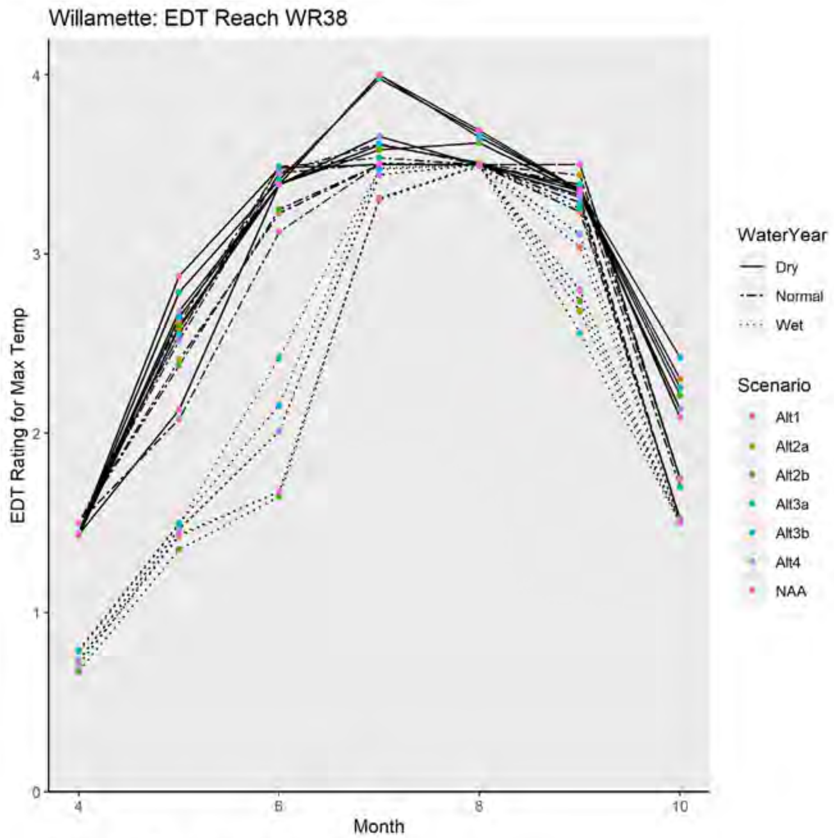

Willamette: EDT Reach WR35
EDT Rating for Max Temp
0
1
2
3
4
4
6
8
10
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

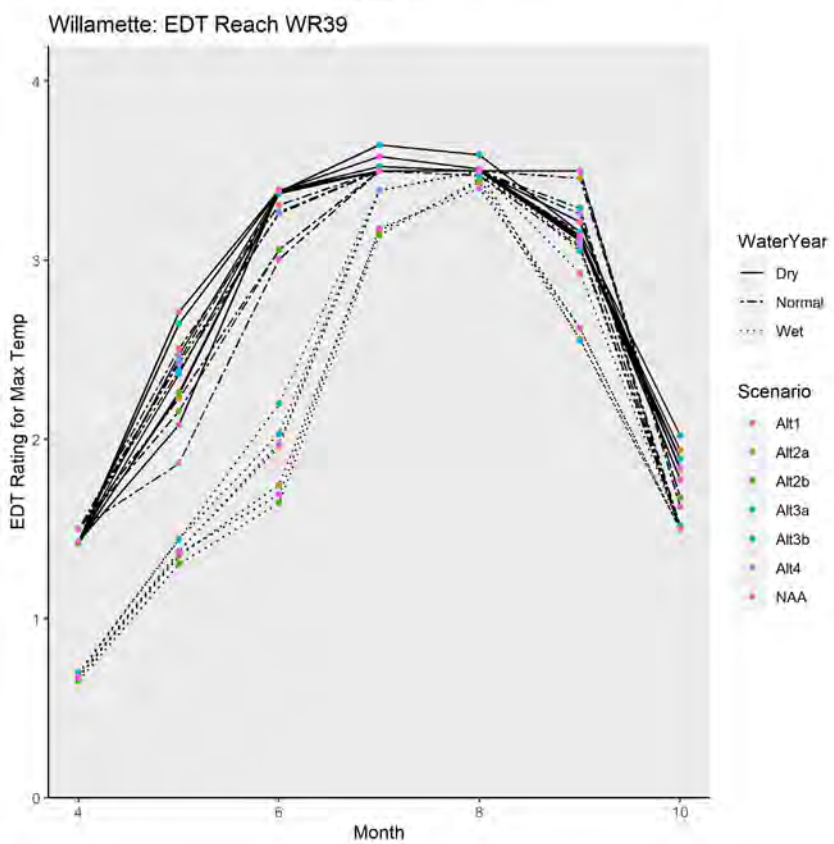


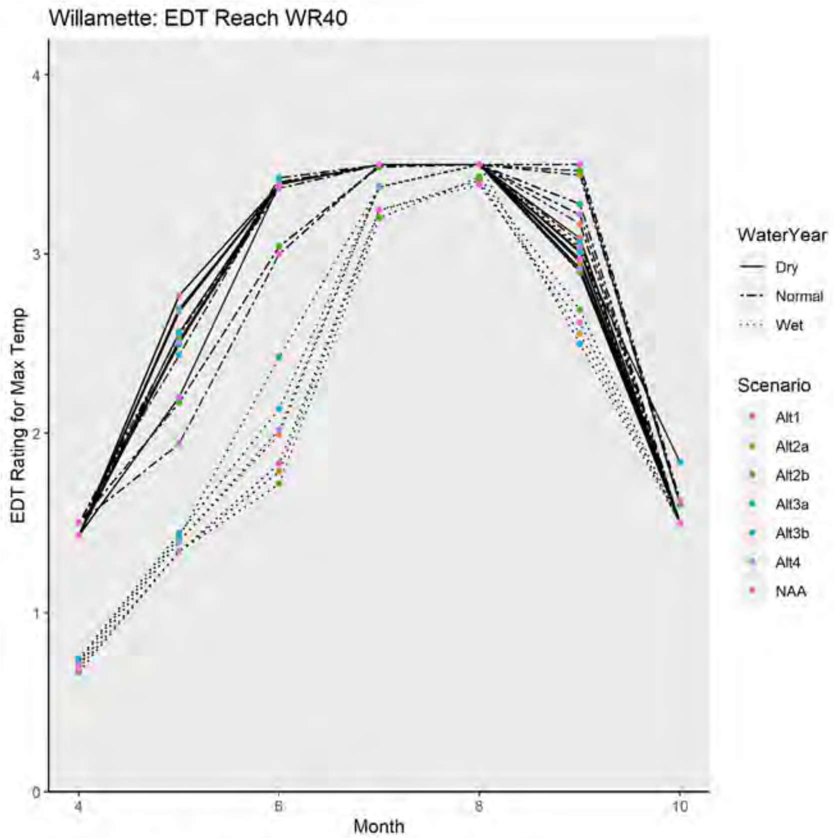
Willamette: EDT Reach WR36
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA



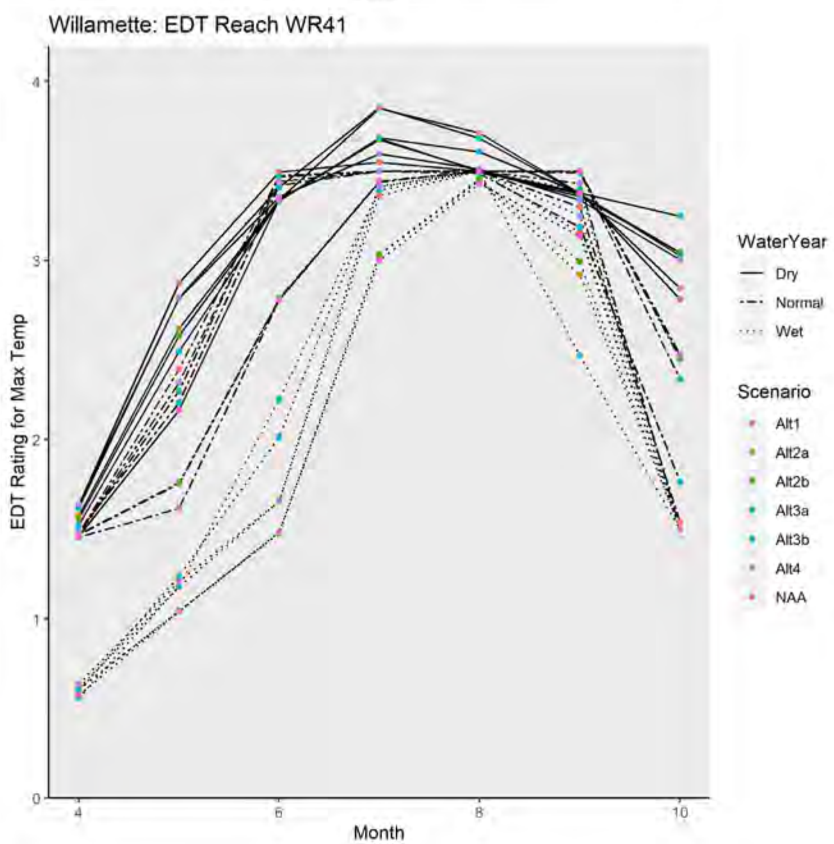
Willamette: EDT Reach WR37
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


Willamette: EDT Reach WR38
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


Willamette: EDT Reach WR39
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA



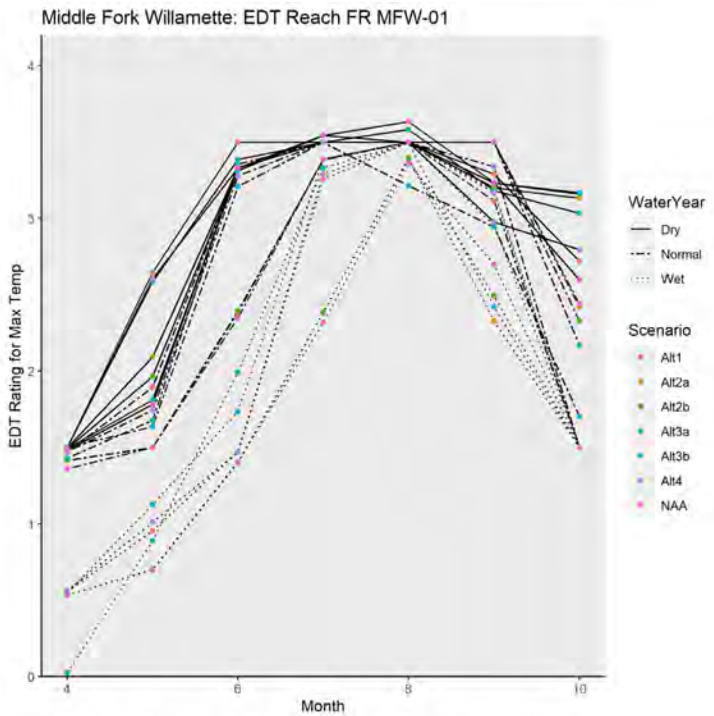
Willamette: EDT Reach WR40
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

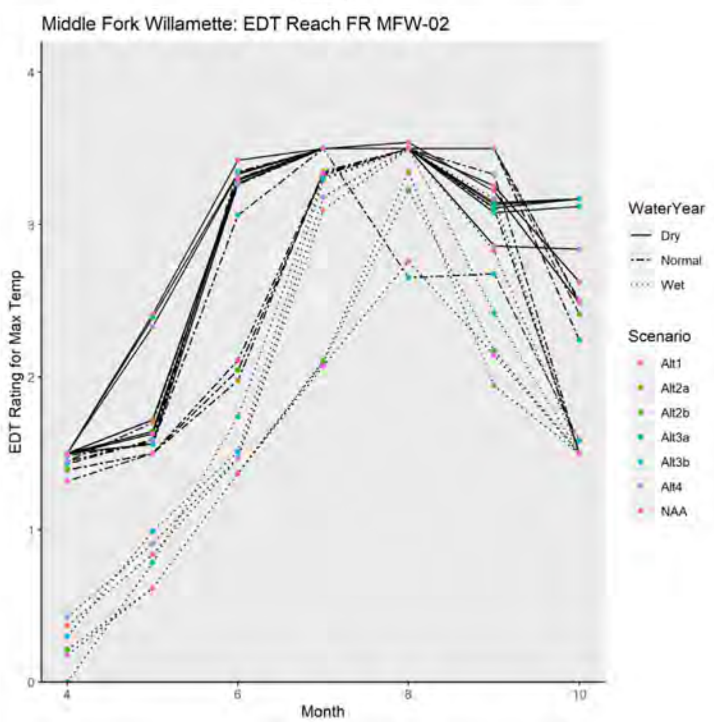


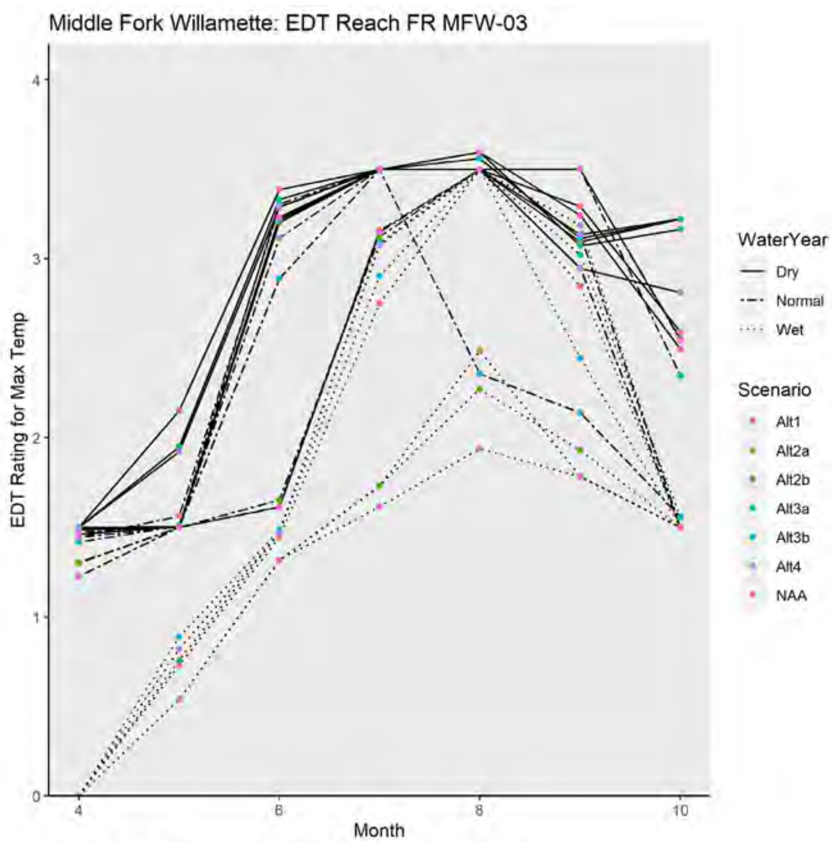
Willamette: EDT Reach WR41
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

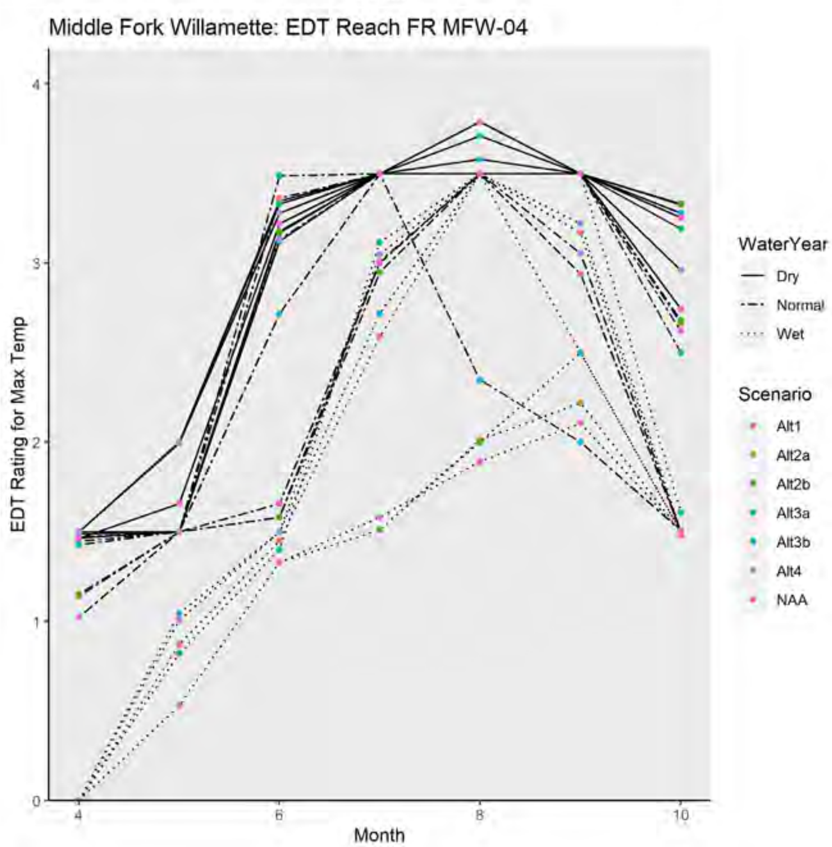
6.6 CHAPTER 6 ATTACHMENT B-2: MIDDLE FORK WILLAMETTE RIVER AND FALL CREEK

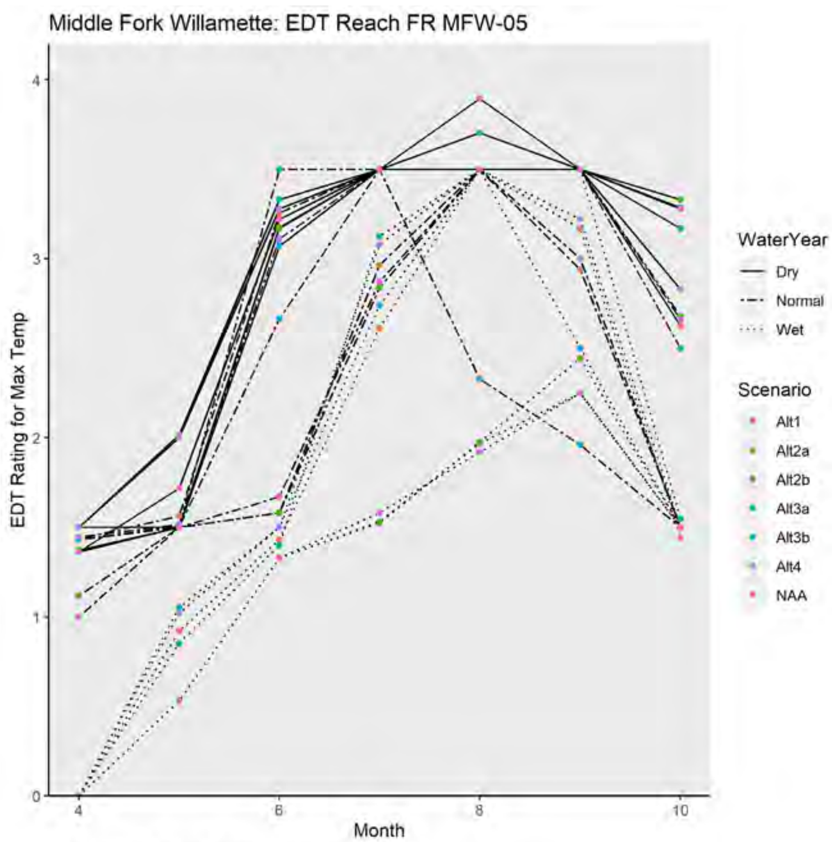
Graphs are organized downstream to upstream: FR MFW-01 through -05 are below Dexter; FR Fall-01A and -01B are Fall Creek below the dam; MFW-09 through -14 are between Dexter and Hills Creek.

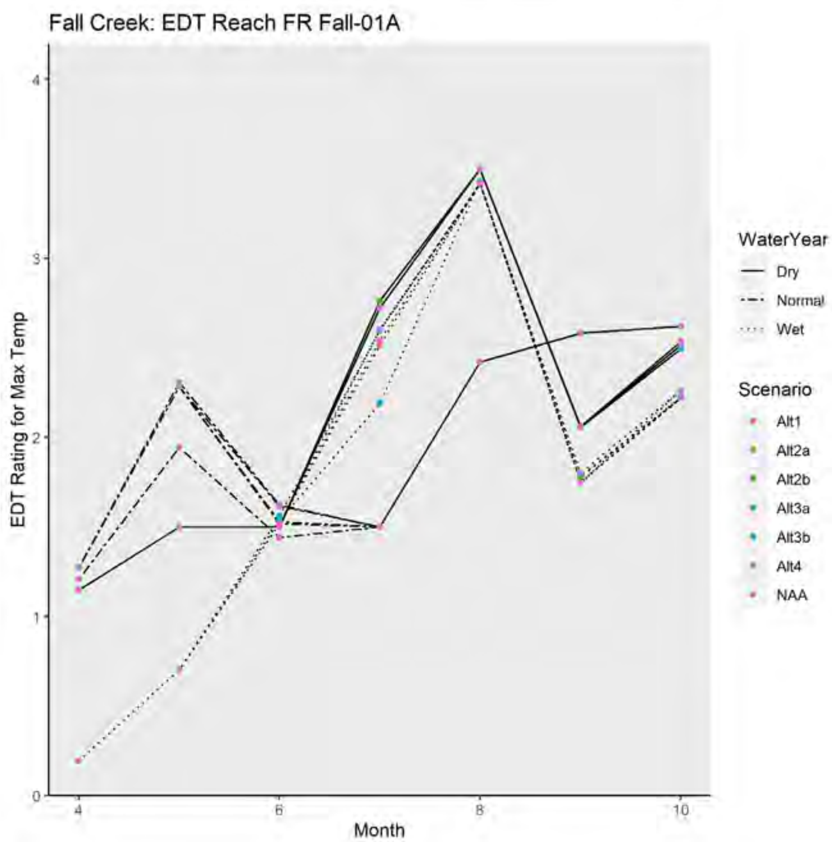


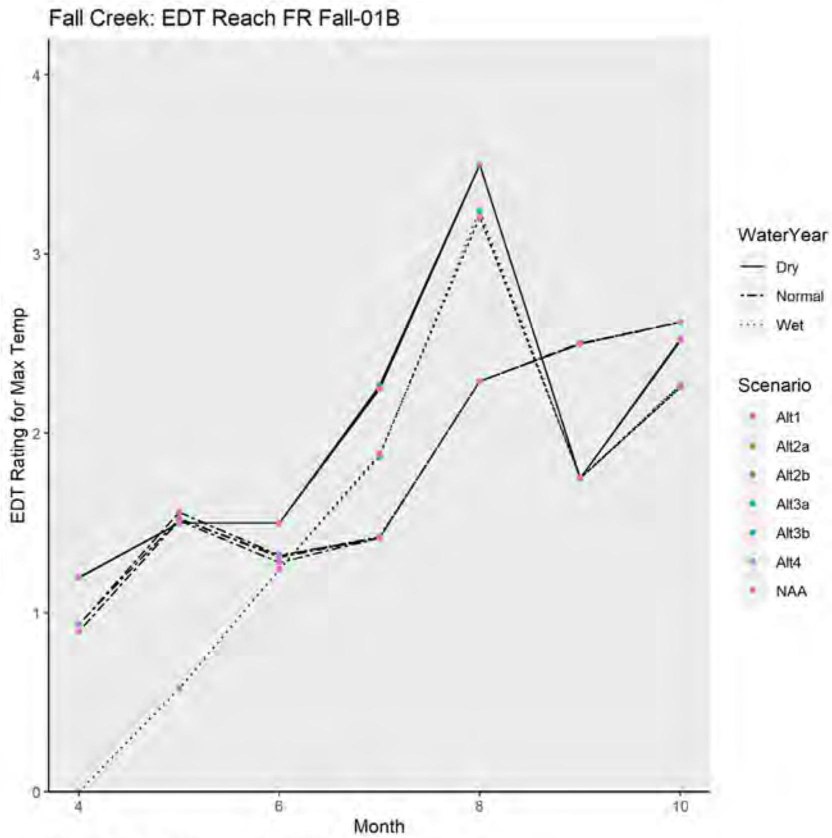


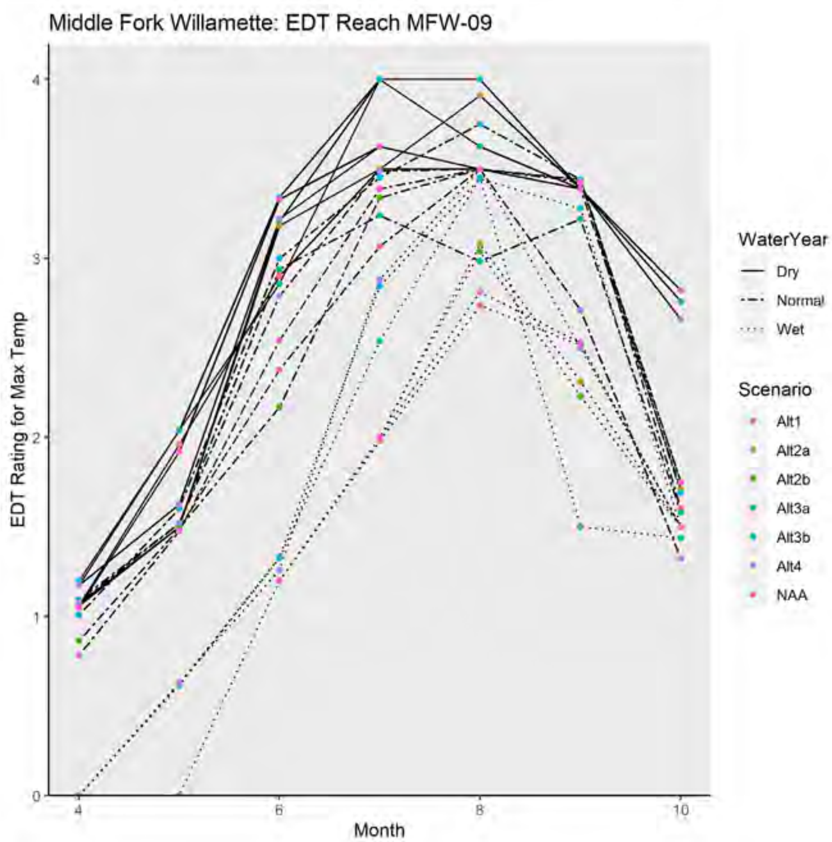

Middle Fork Willamette: EDT Reach FR MFW-03
EDT Rating for Max Temp
Month
0
1
2
3
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

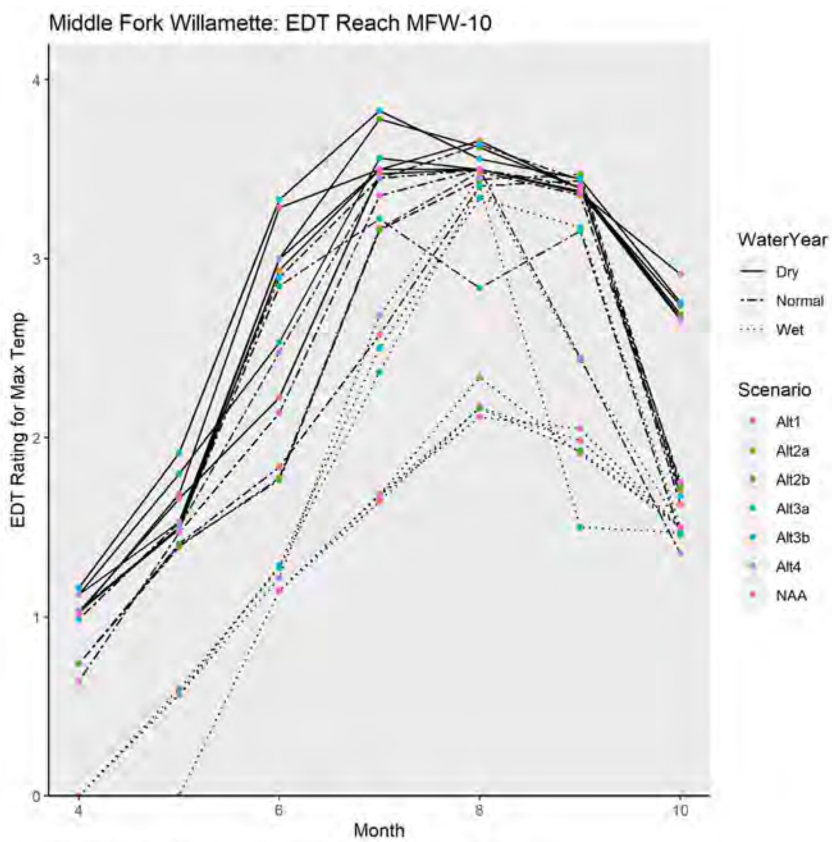

Middle Fork Willamette: EDT Reach FR MFW-04
EDT Rating for Max Temp
Month
0
1
2
3
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

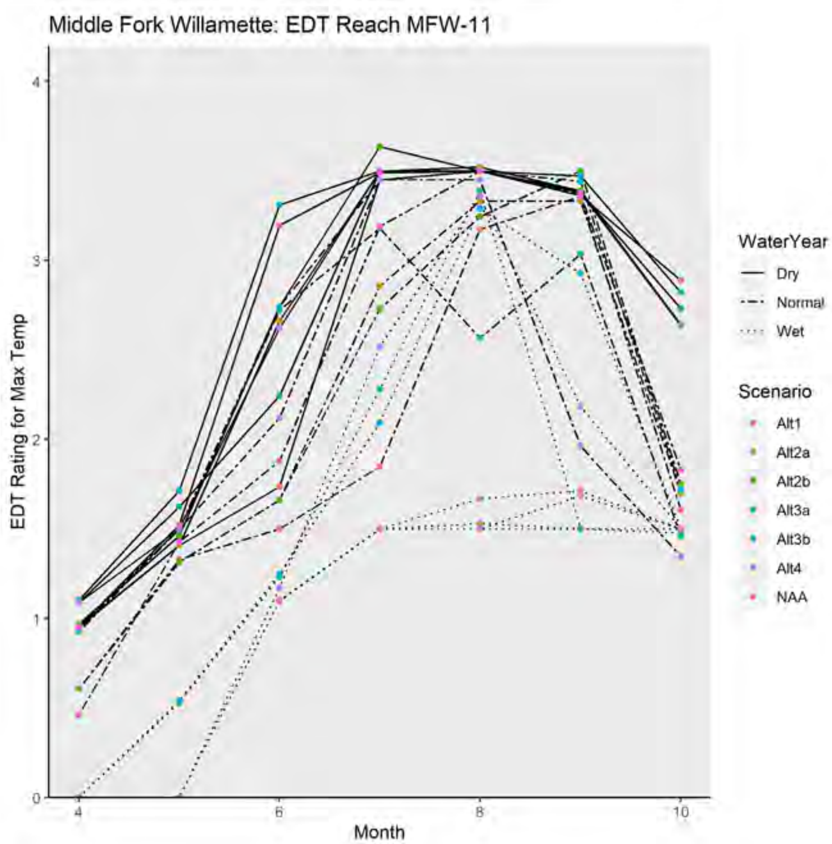

Middle Fork Willamette: EDT Reach FR MFW-05
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

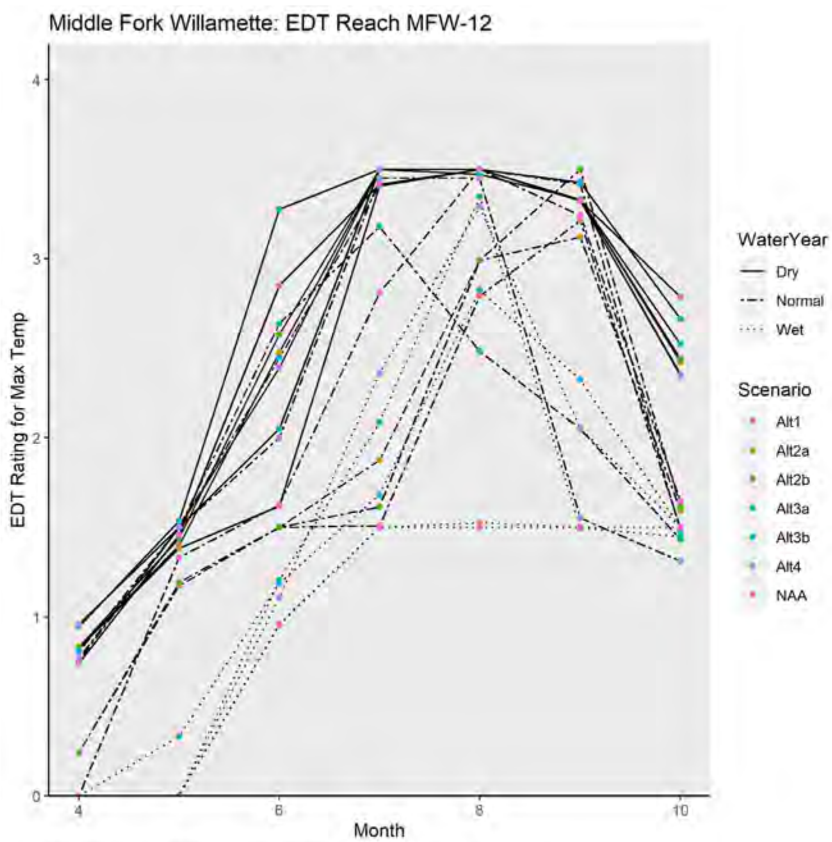

Fall Creek: EDT Reach FR Fall-01A
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

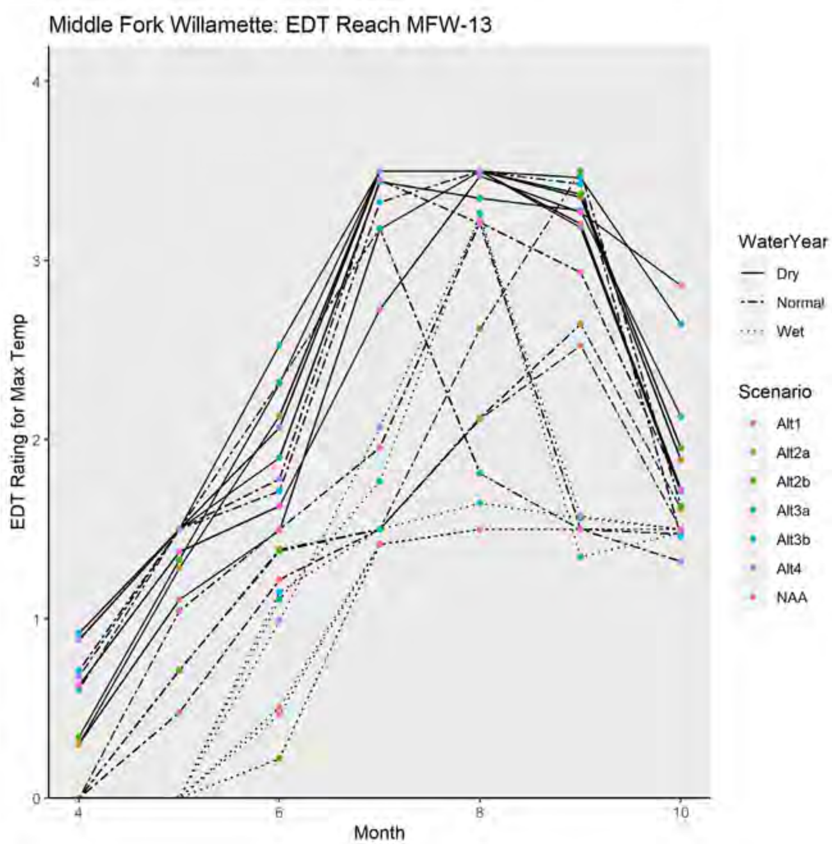

Fall Creek: EDT Reach FR Fall-01B
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

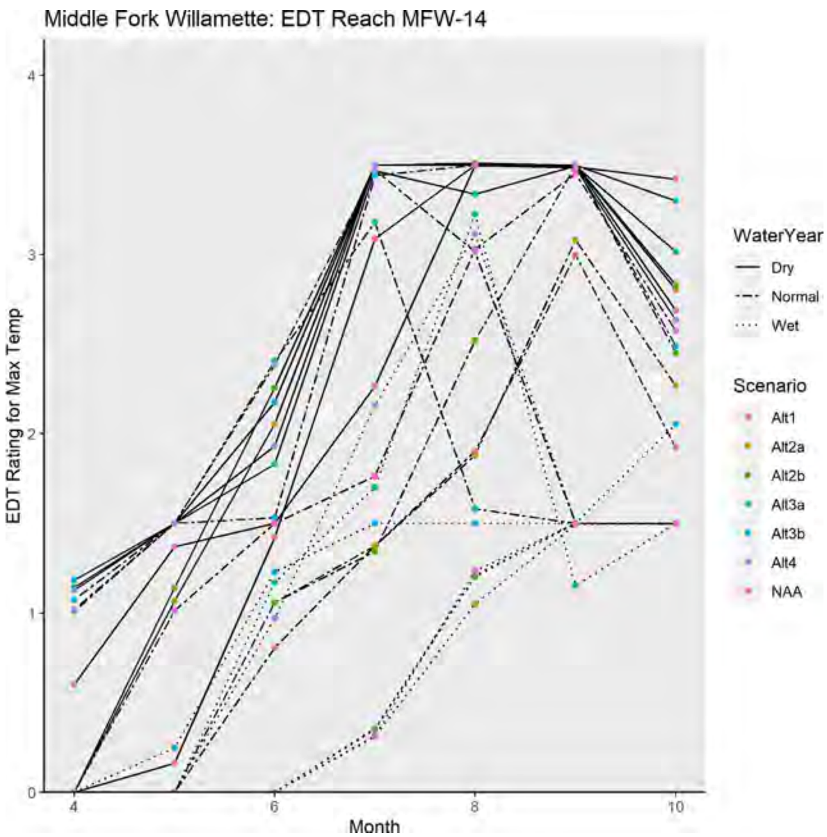

Middle Fork Willamette: EDT Reach MFW-09
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


Middle Fork Willamette: EDT Reach MFW-10
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


Middle Fork Willamette: EDT Reach MFW-11
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

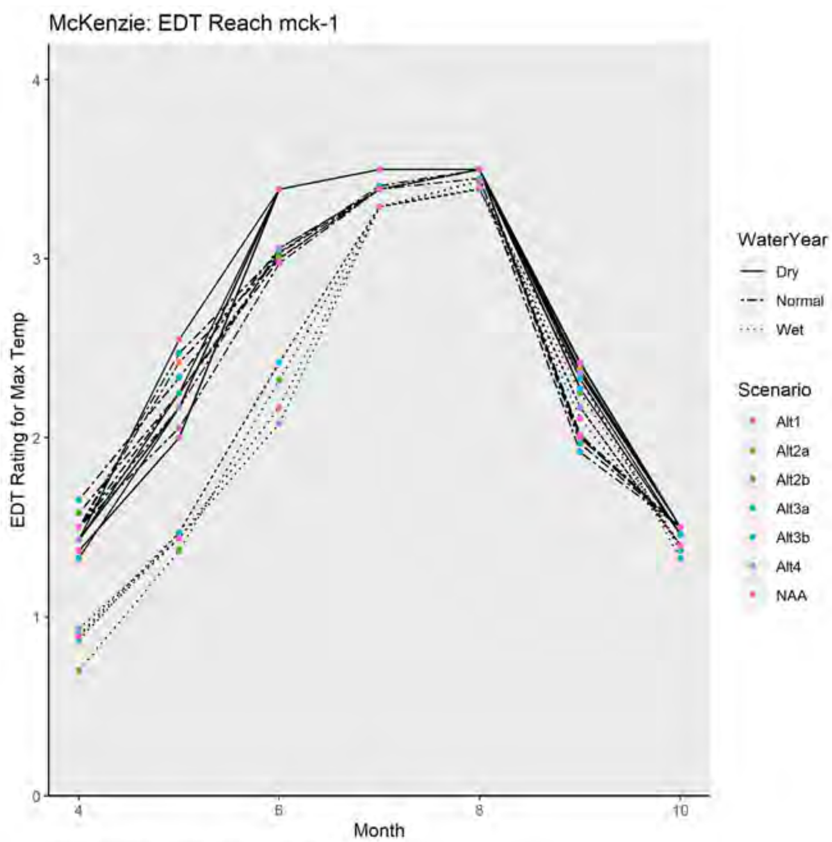

Middle Fork Willamette: EDT Reach MFW-12
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

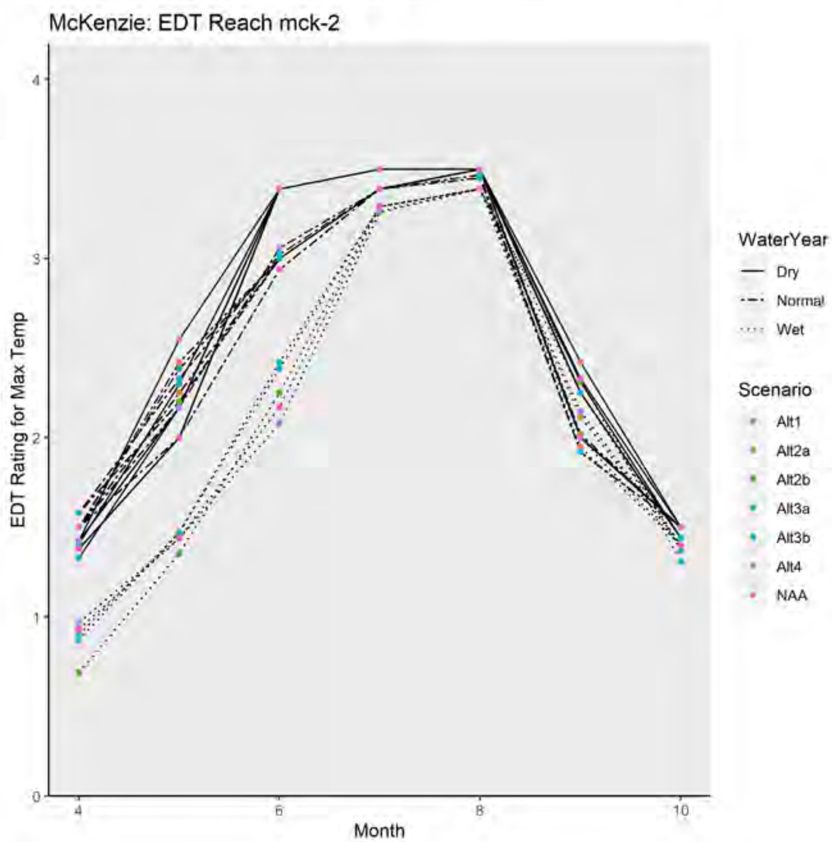

Middle Fork Willamette: EDT Reach MFW-13
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

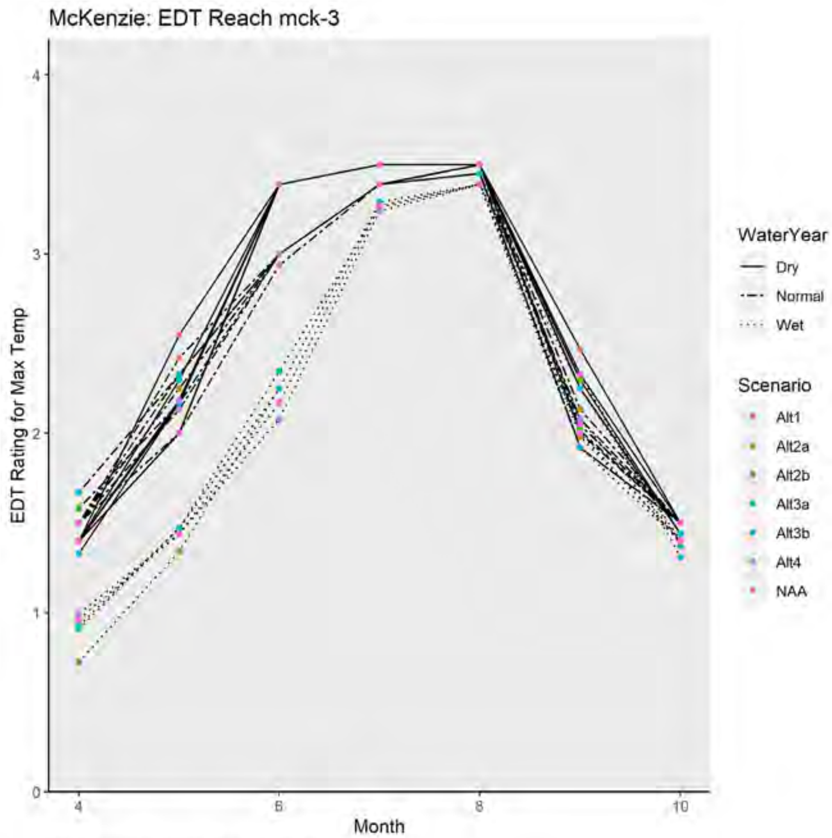


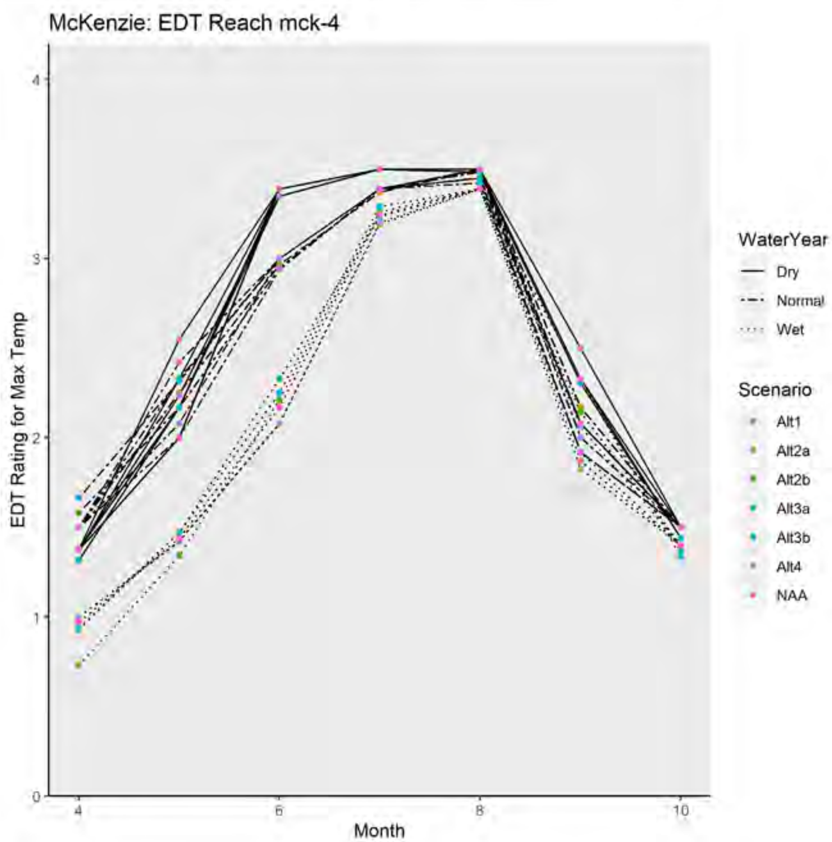
6.7 CHAPTER 6 ATTACHMENT B-3: MCKENZIE RIVER

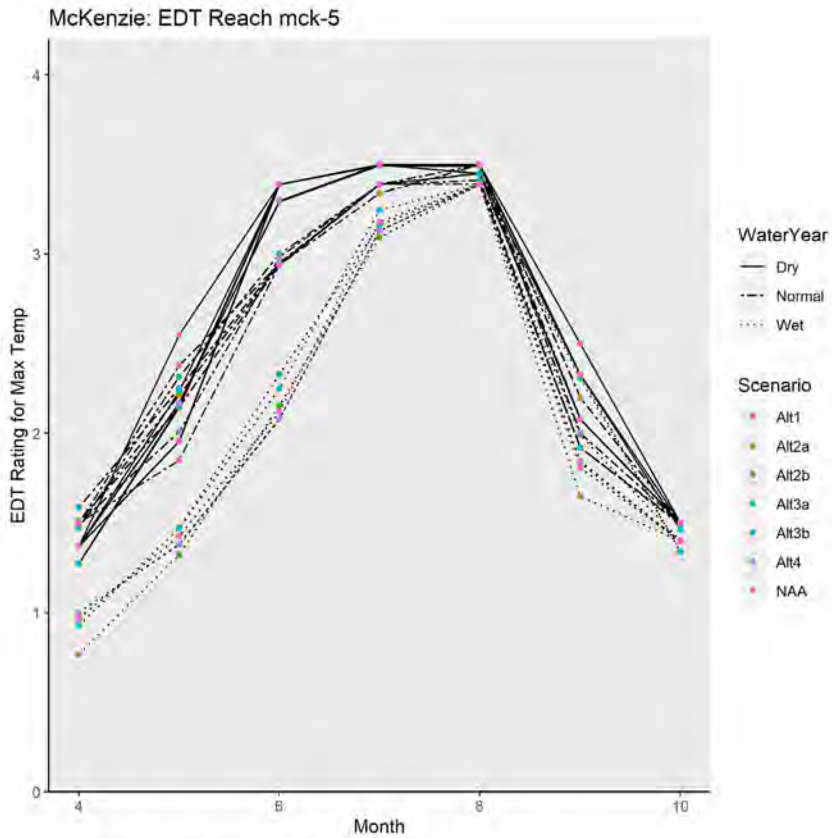
Graphs are organized downstream to upstream: mck-1 to mck-52 is the mouth of the McKenzie to South Fork of McKenzie River; McKenzie SF-1 and SF-2 are South Fork McKenzie before Cougar Dam.

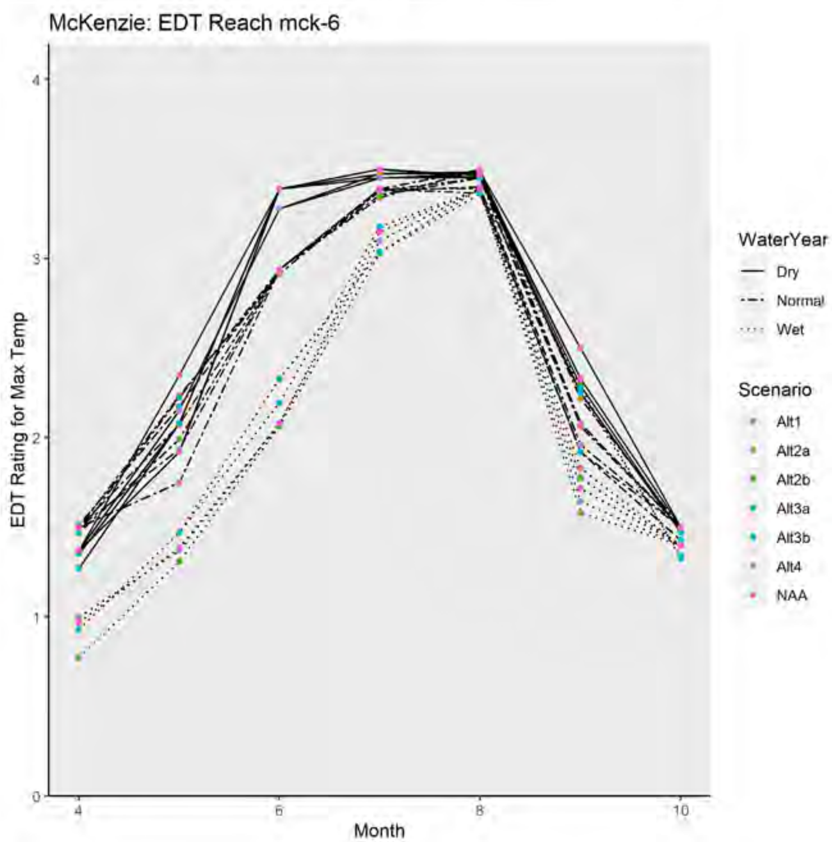

McKenzie: EDT Reach mck-1
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

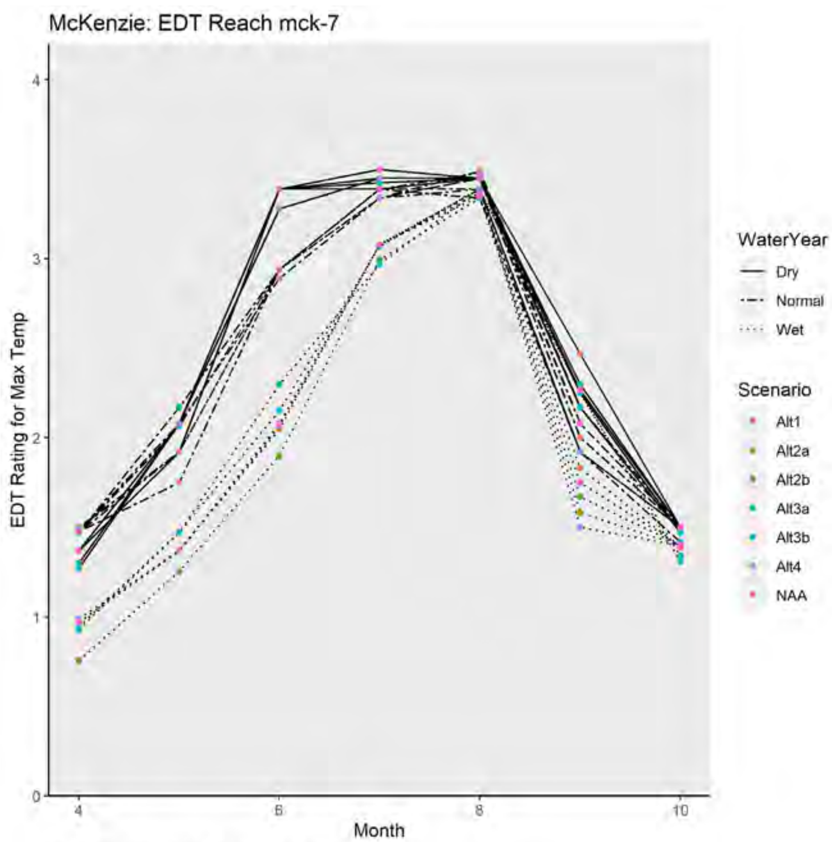

McKenzie: EDT Reach mck-2
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

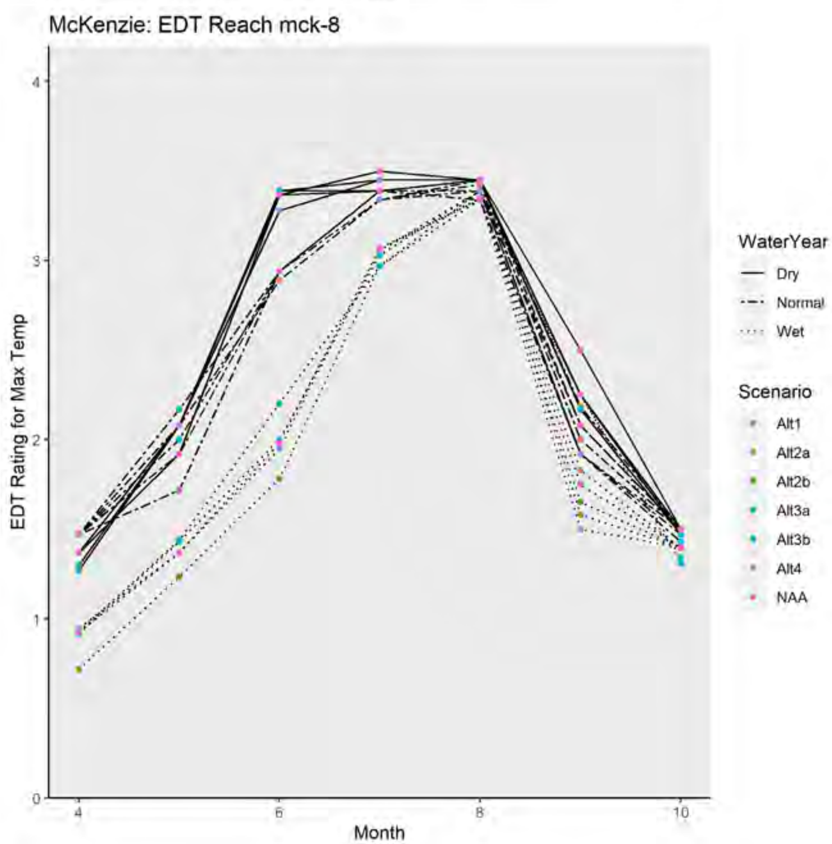

McKenzie: EDT Reach mck-3
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

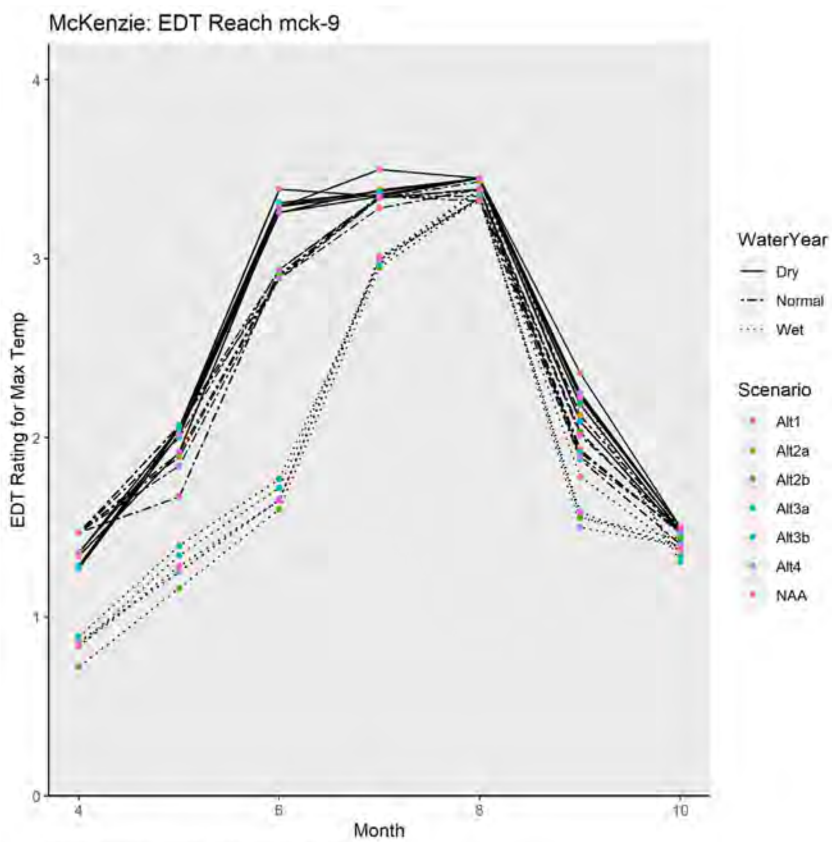

McKenzie: EDT Reach mck-4
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


McKenzie: EDT Reach mck-5
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

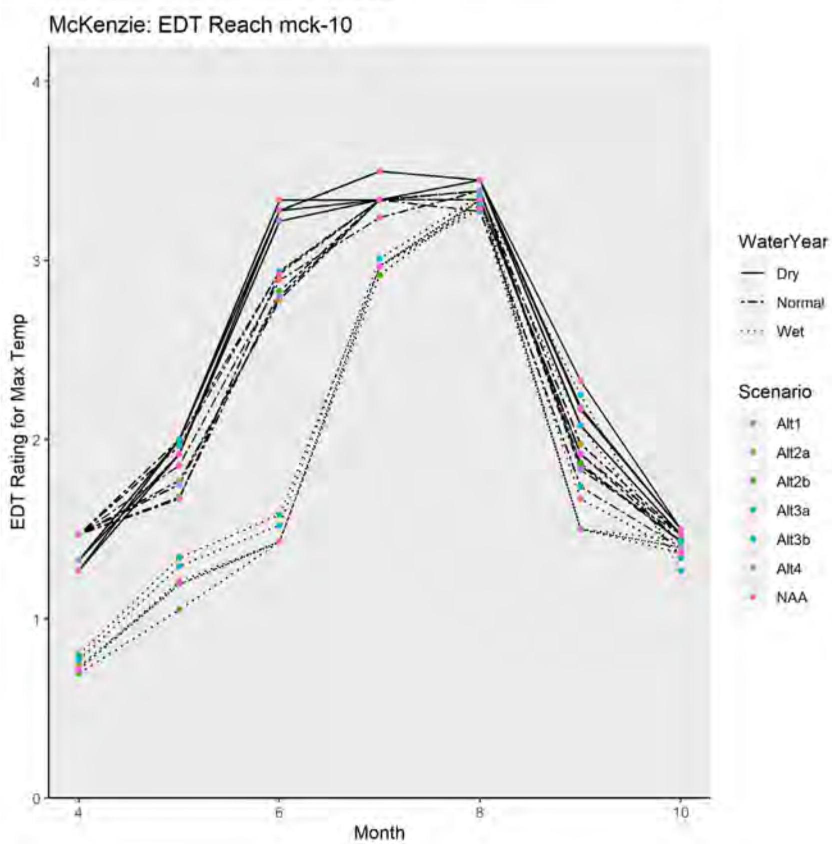

McKenzie: EDT Reach mck-6
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


McKenzie: EDT Reach mck-7
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

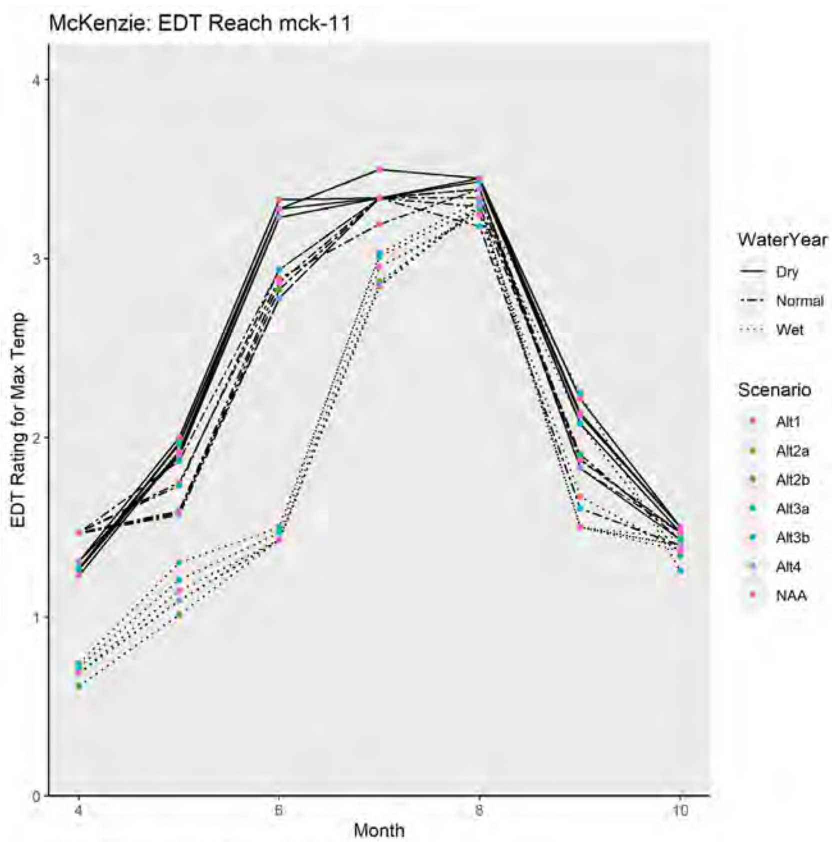

McKenzie: EDT Reach mck-8
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

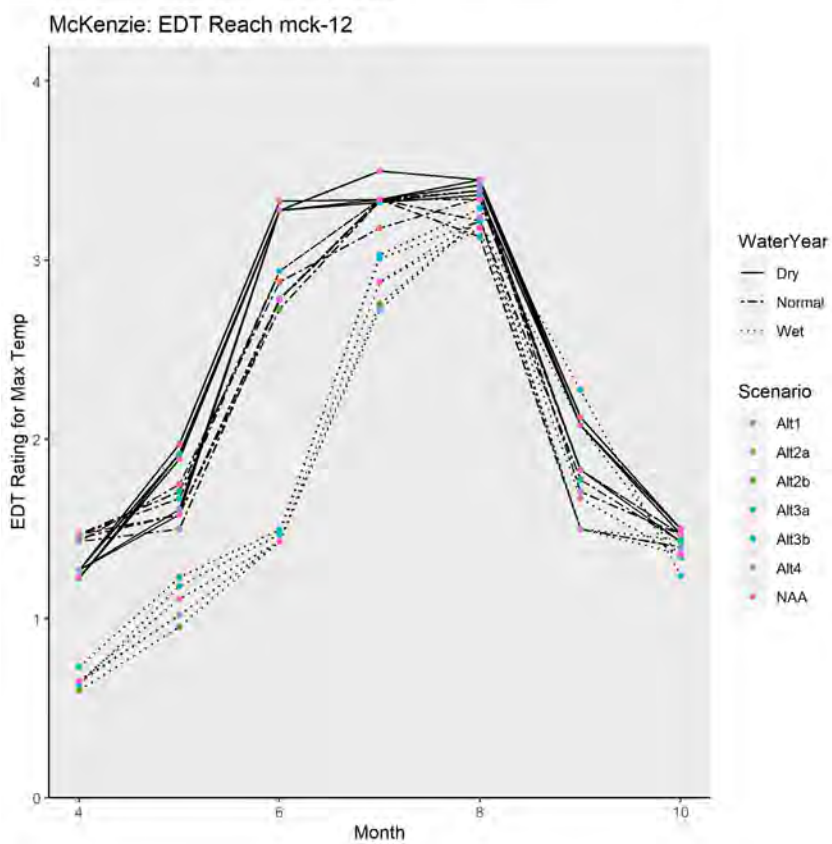


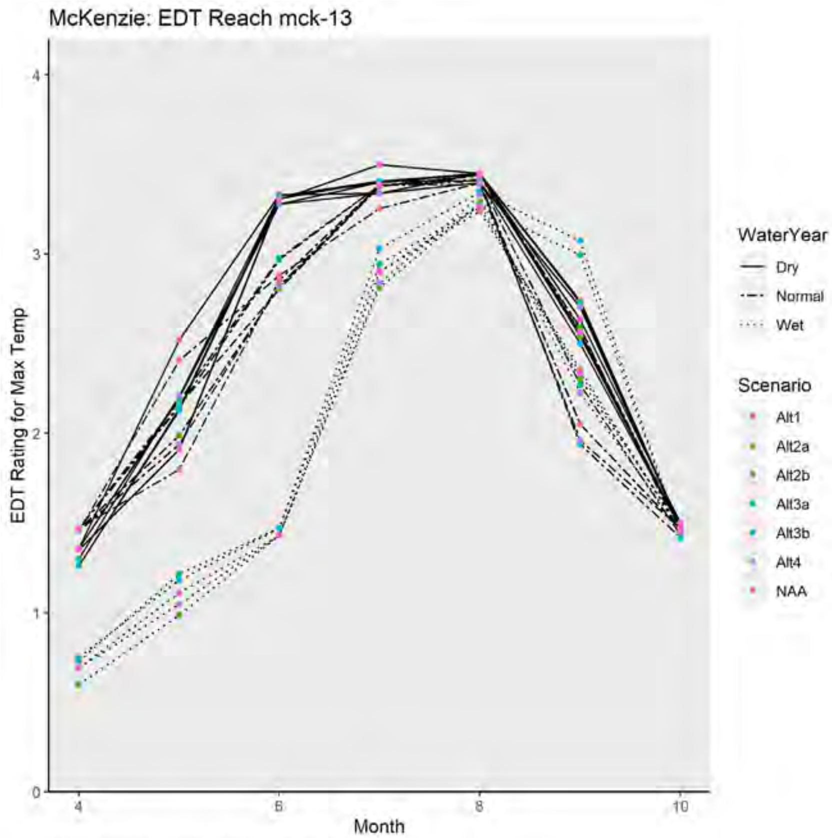
McKenzie: EDT Reach mck-9
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

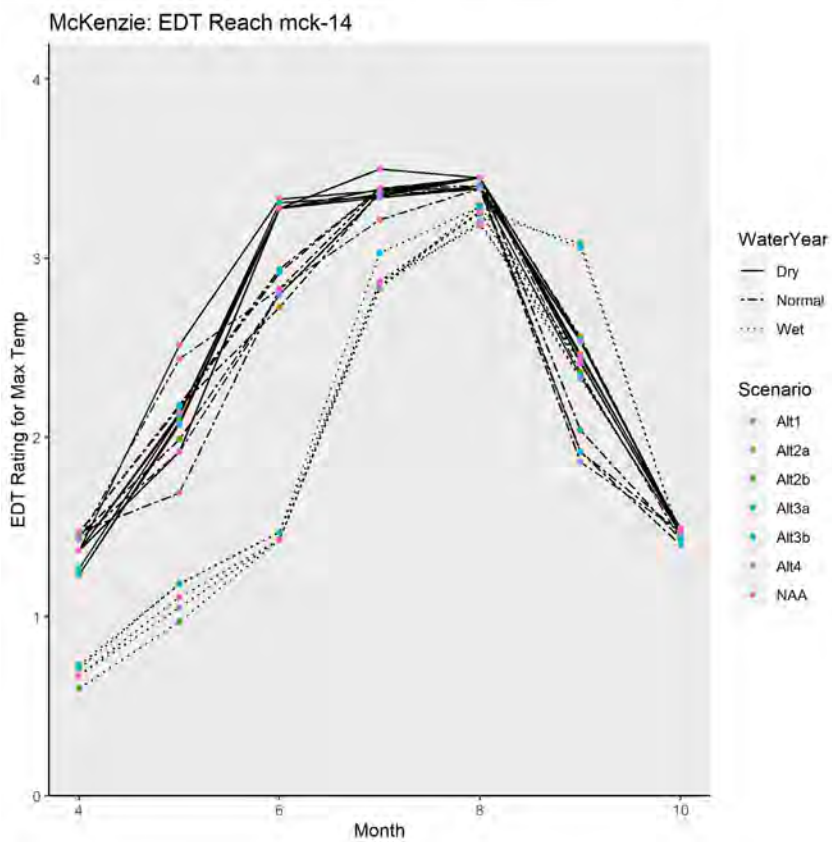


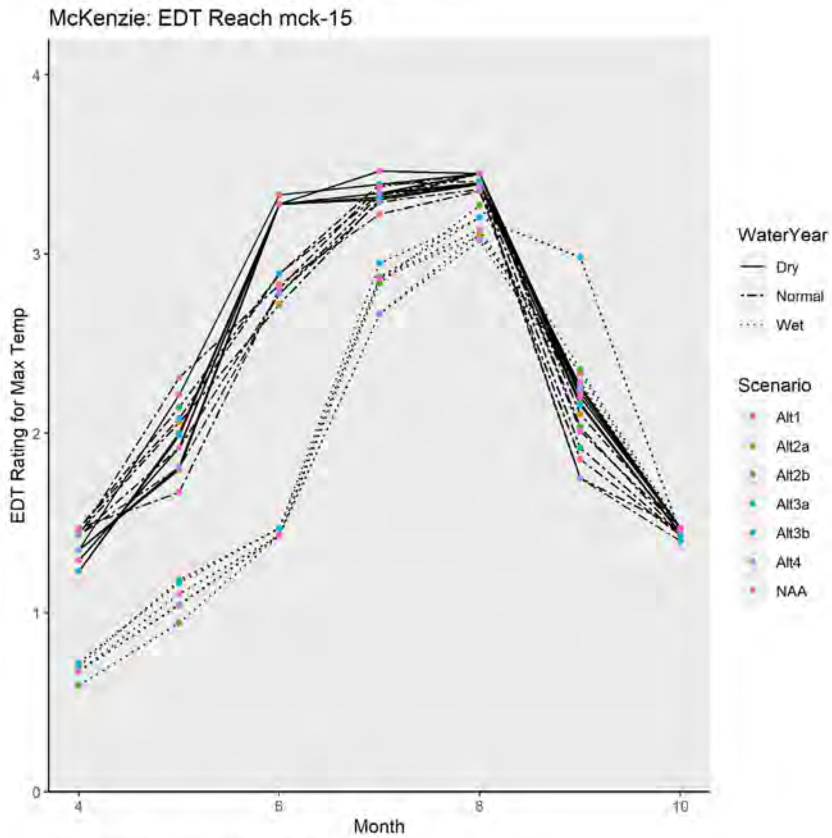
McKenzie: EDT Reach mck-10
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

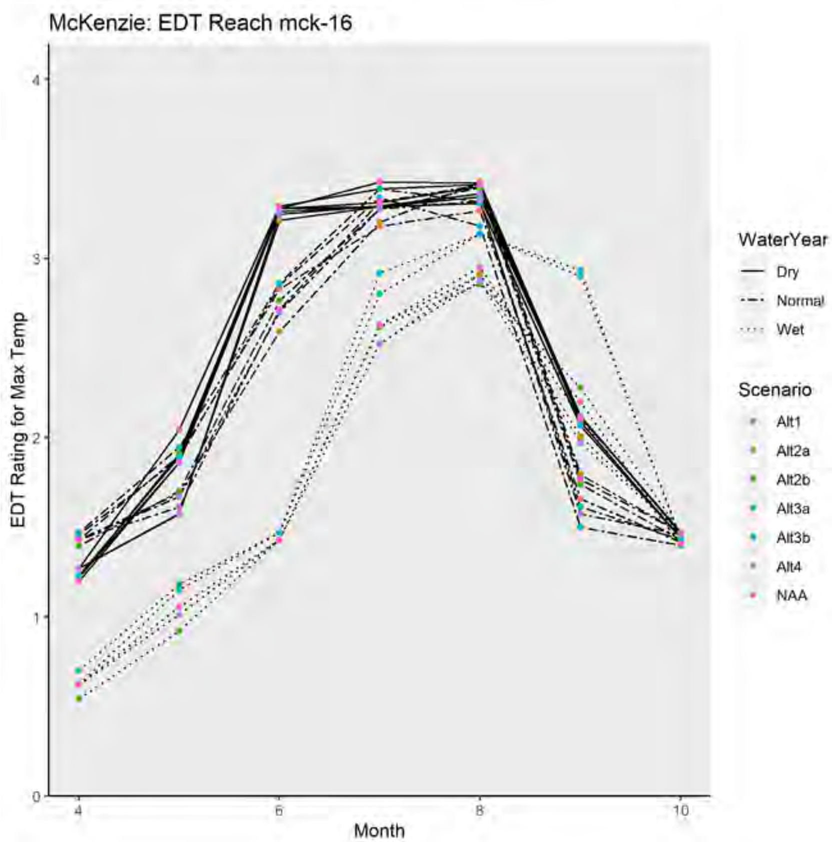

McKenzie: EDT Reach mck-11
EDT Rating for Max Temp
0
1
2
3
4
4
6
8
10
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

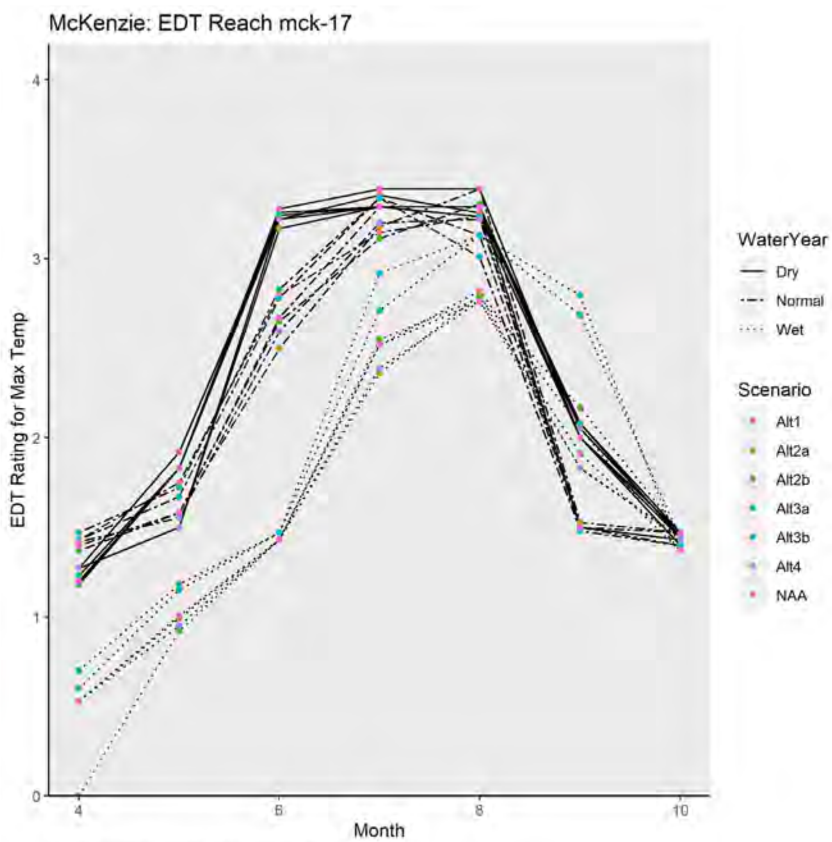

McKenzie: EDT Reach mck-12
EDT Rating for Max Temp
0
1
2
3
4
4
6
8
10
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

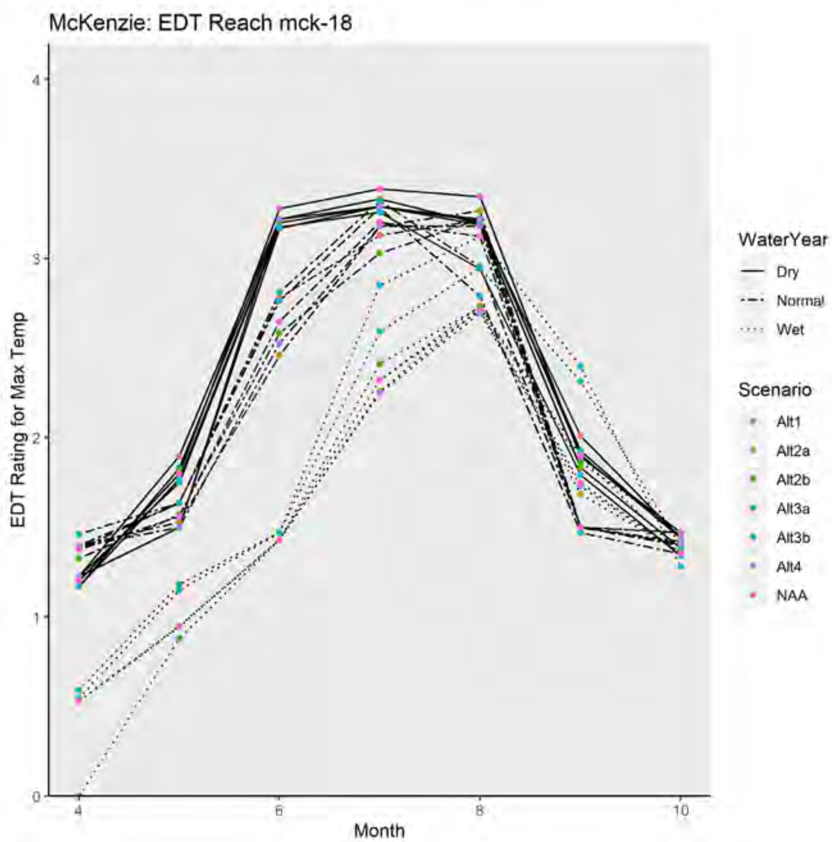

McKenzie: EDT Reach mck-13
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

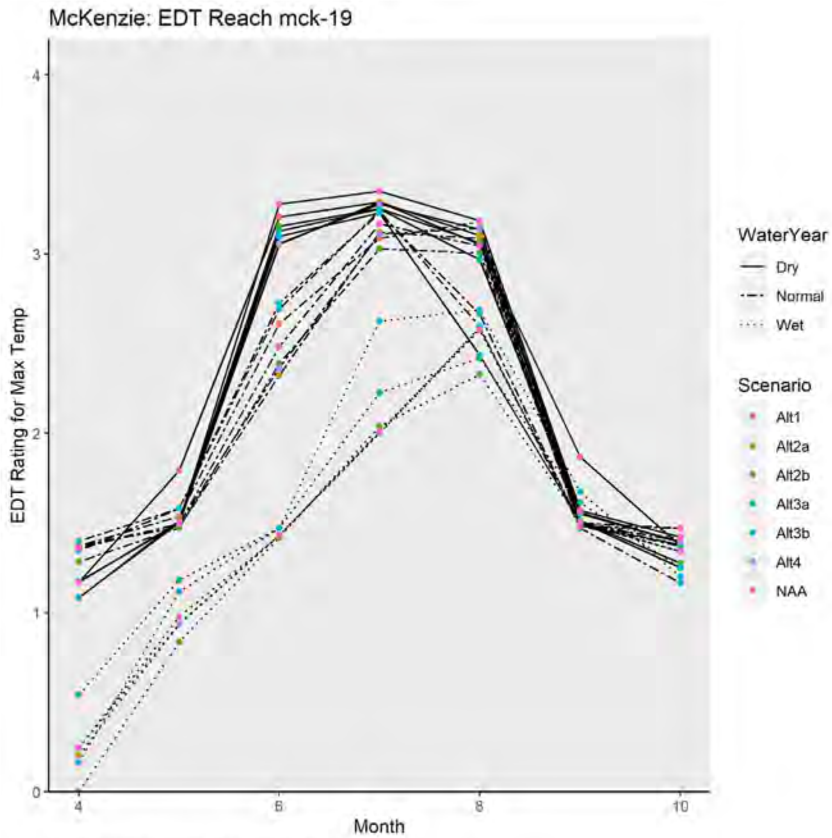

McKenzie: EDT Reach mck-14
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

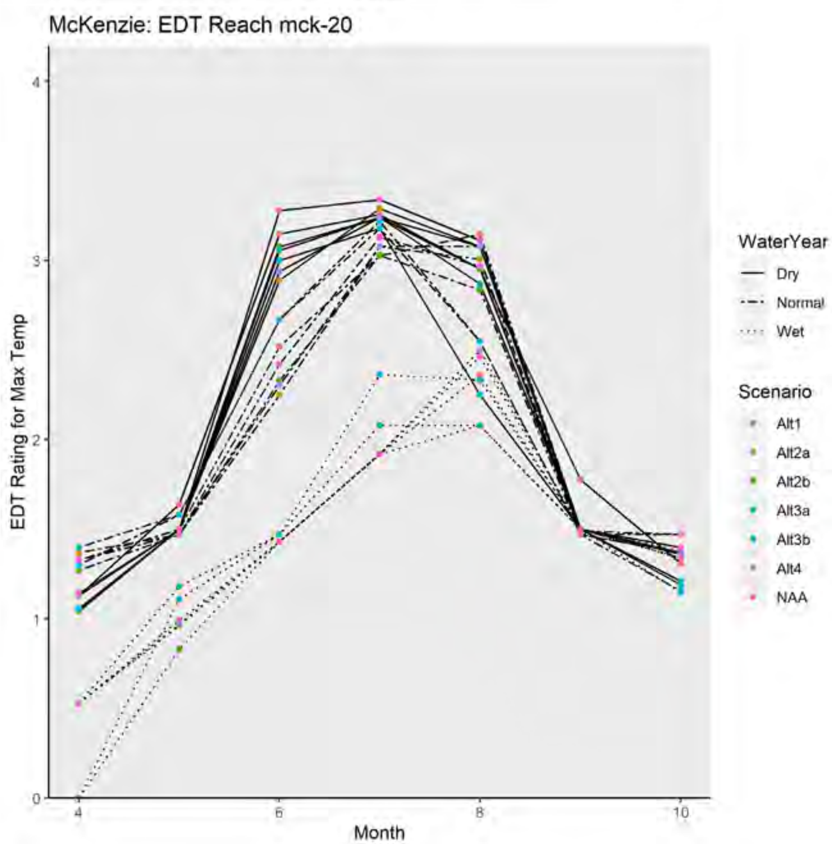

McKenzie: EDT Reach mck-15
EDT Rating for Max Temp
Month
0
1
2
3
4
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

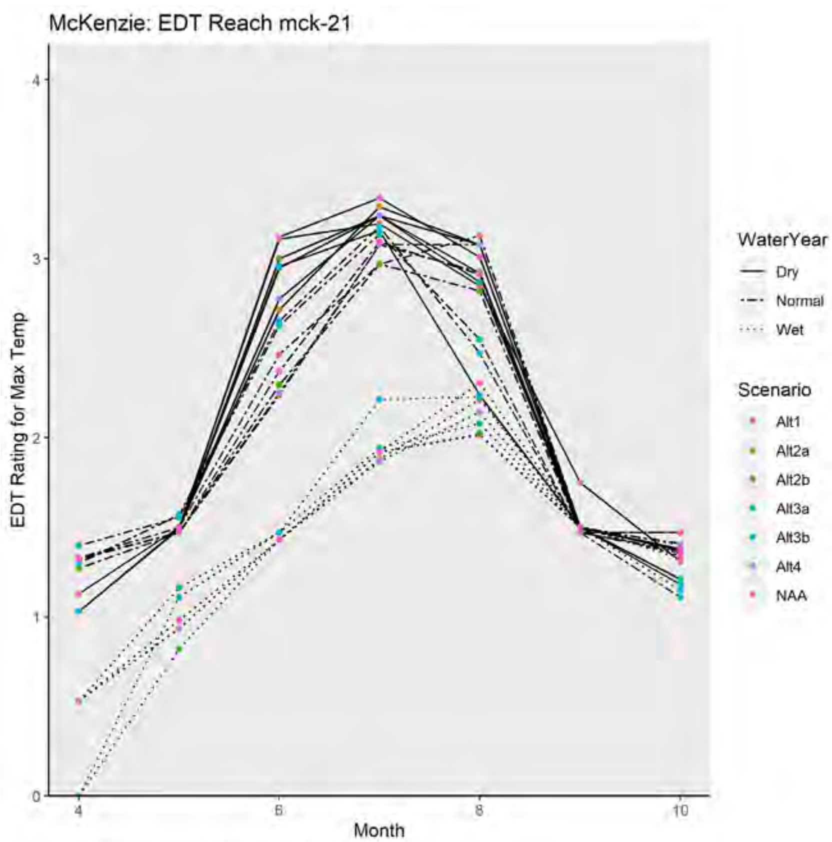

McKenzie: EDT Reach mck-16
EDT Rating for Max Temp
Month
0
1
2
3
4
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

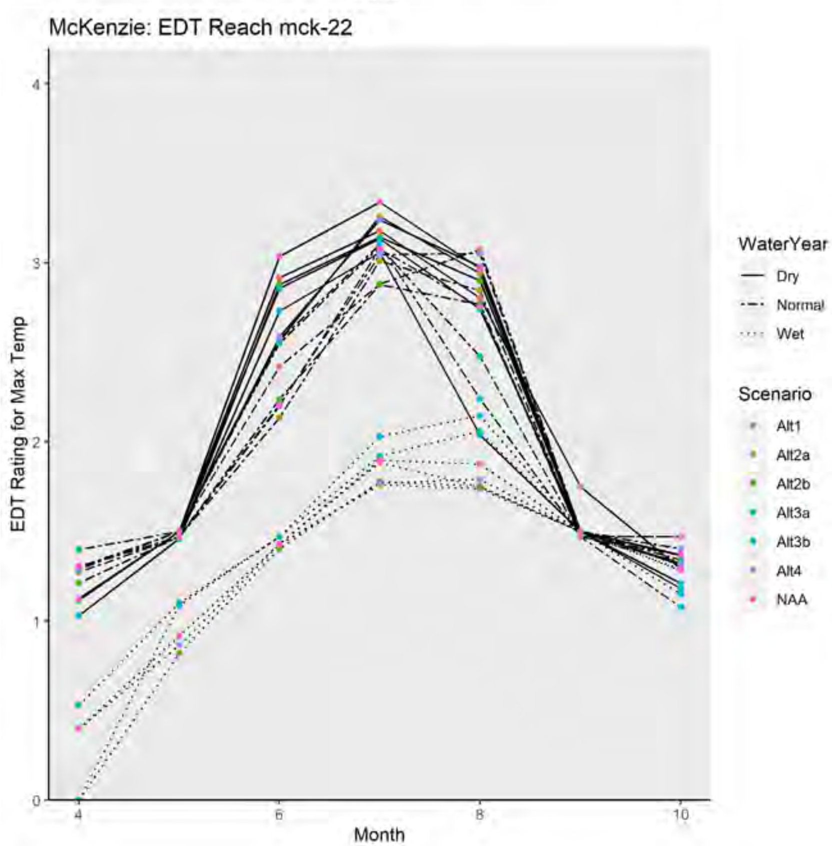

McKenzie: EDT Reach mck-17
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

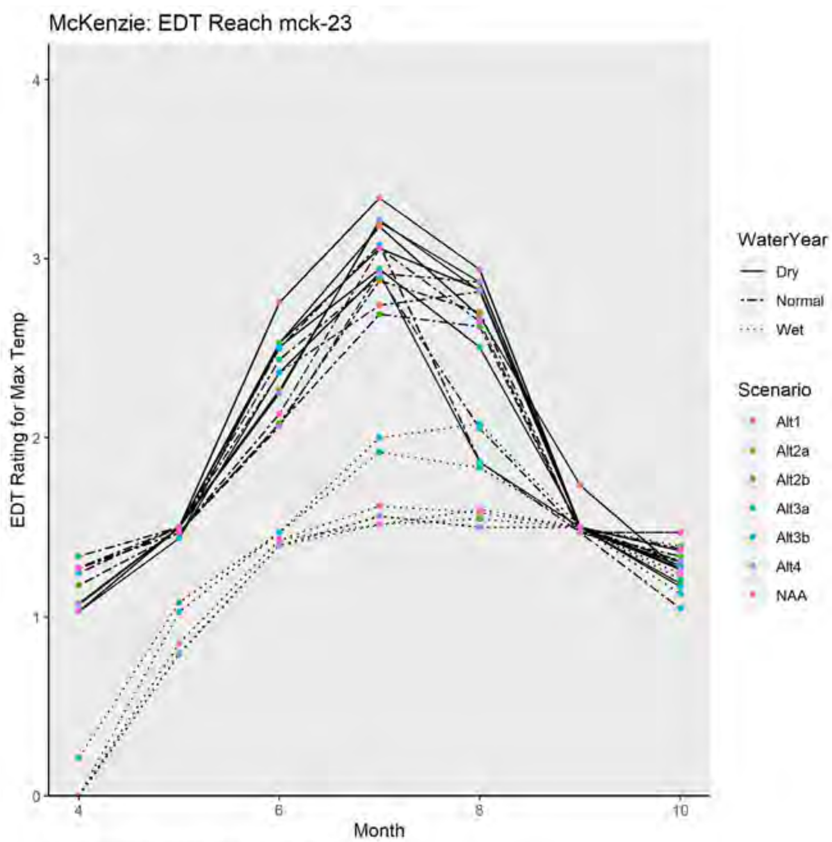

McKenzie: EDT Reach mck-18
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

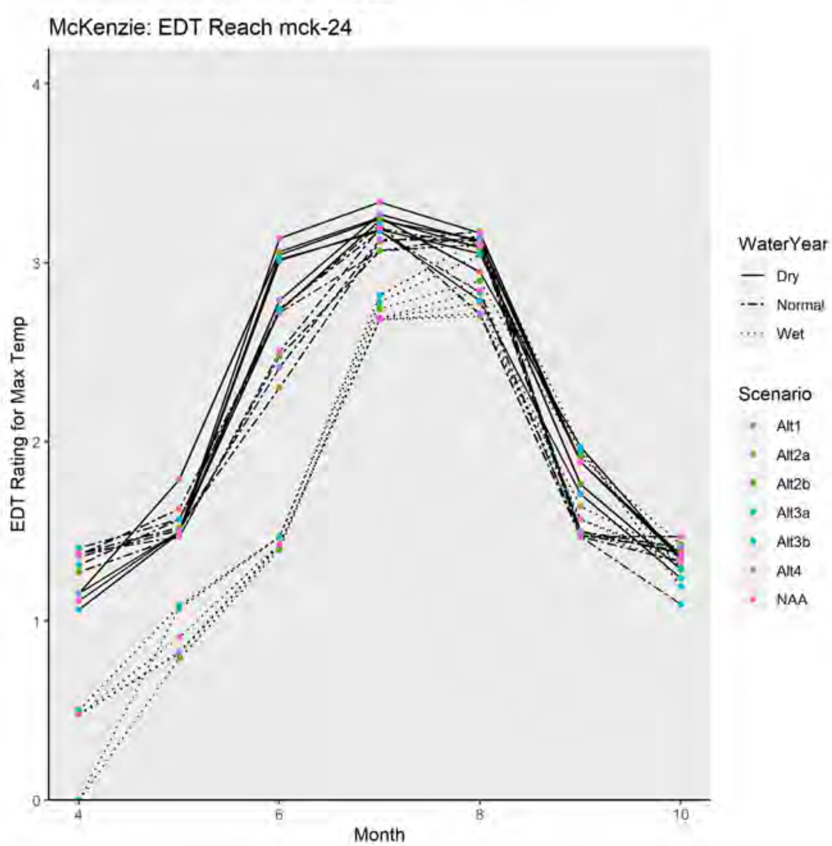

McKenzie: EDT Reach mck-19
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

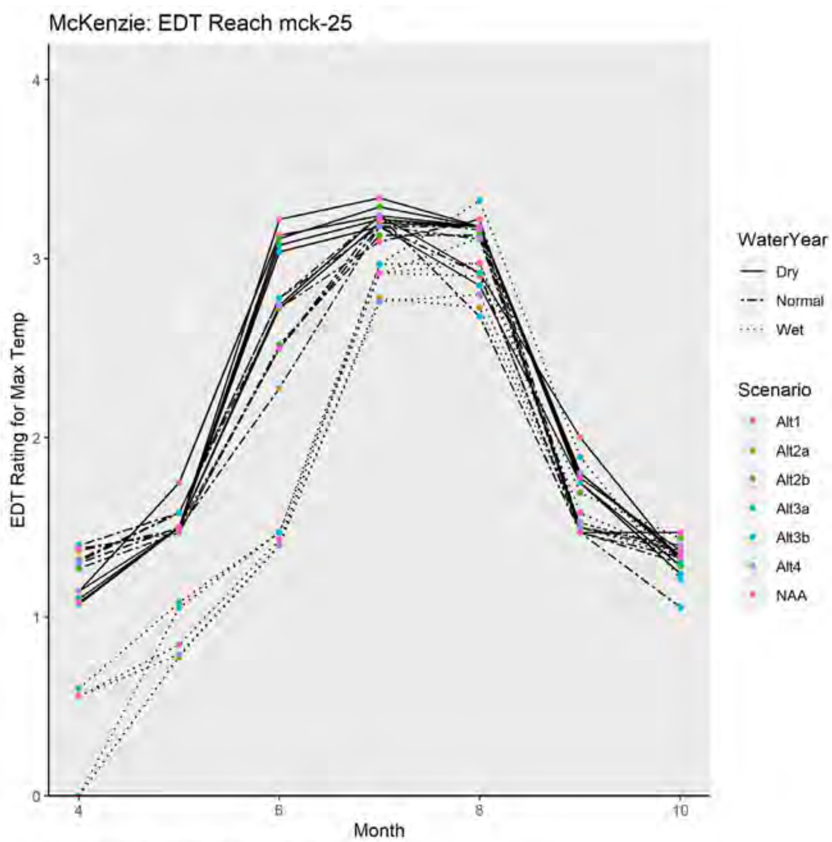

McKenzie: EDT Reach mck-20
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

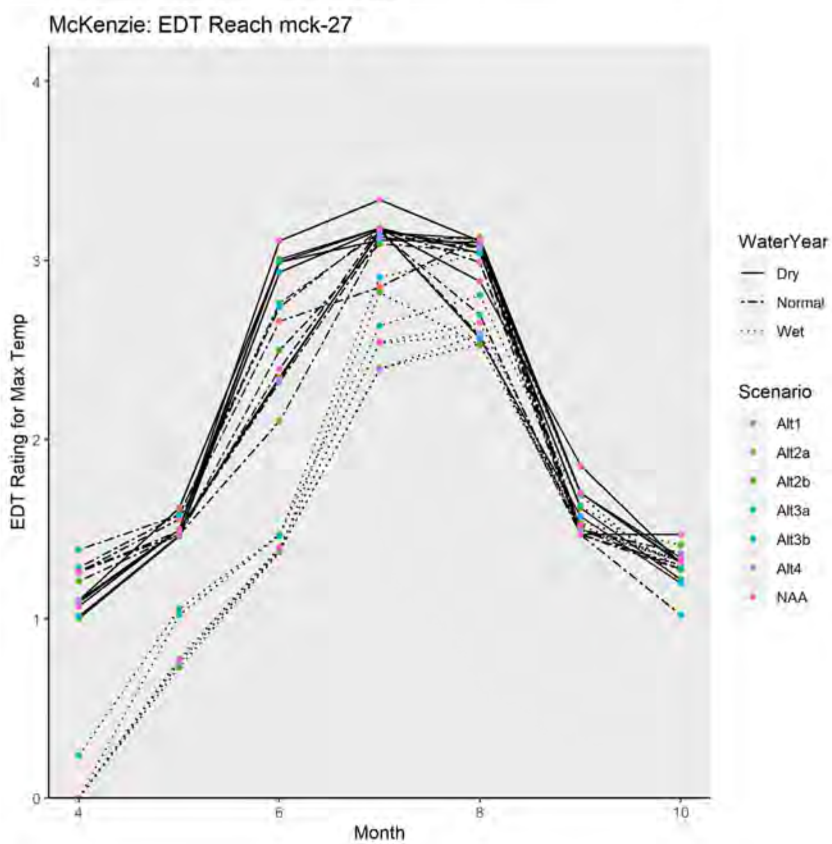

McKenzie: EDT Reach mck-21
EDT Rating for Max Temp
Month
4
6
8
10
0
1
2
3
4
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

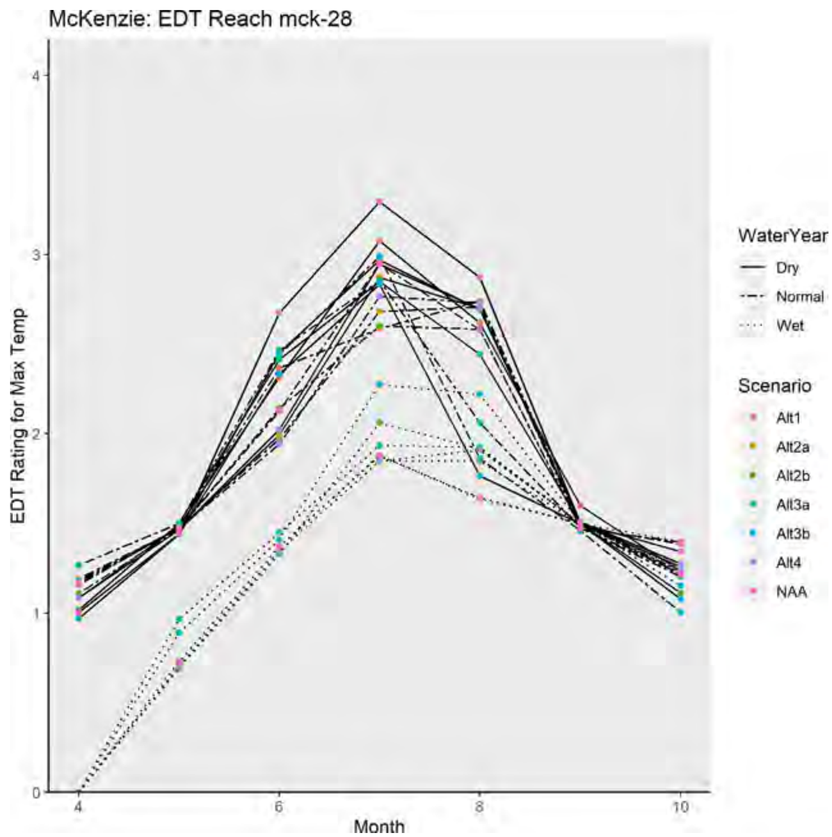

McKenzie: EDT Reach mck-22
EDT Rating for Max Temp
Month
4
6
8
10
0
1
2
3
4
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


McKenzie: EDT Reach mck-23
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

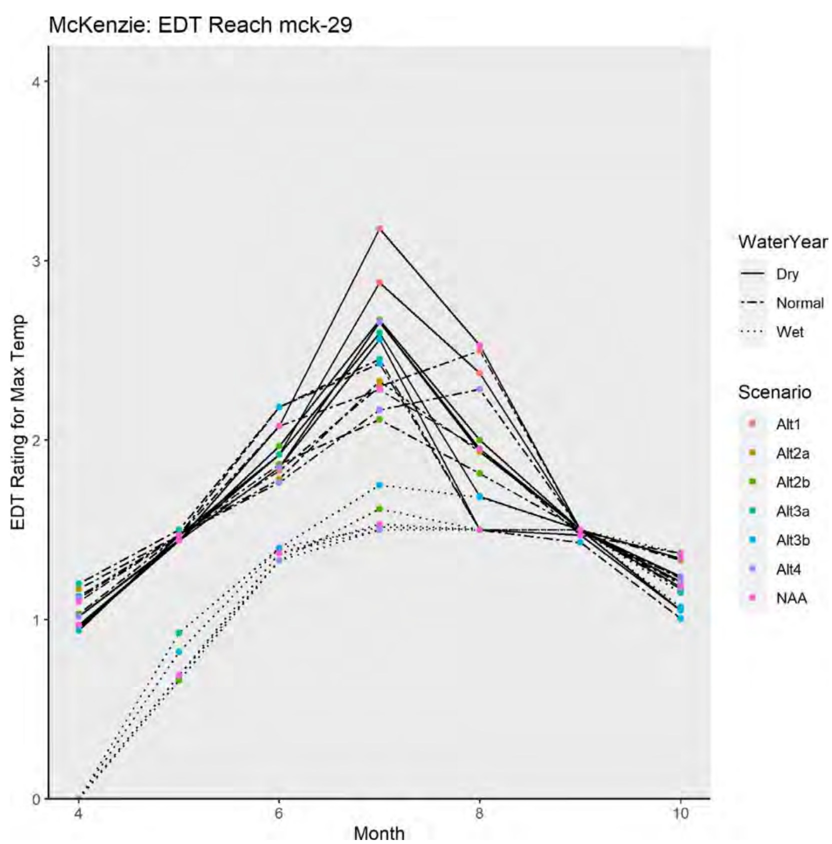

McKenzie: EDT Reach mck-24
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


McKenzie: EDT Reach mck-25
EDT Rating for Max Temp
0
1
2
3
4
4
6
8
10
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

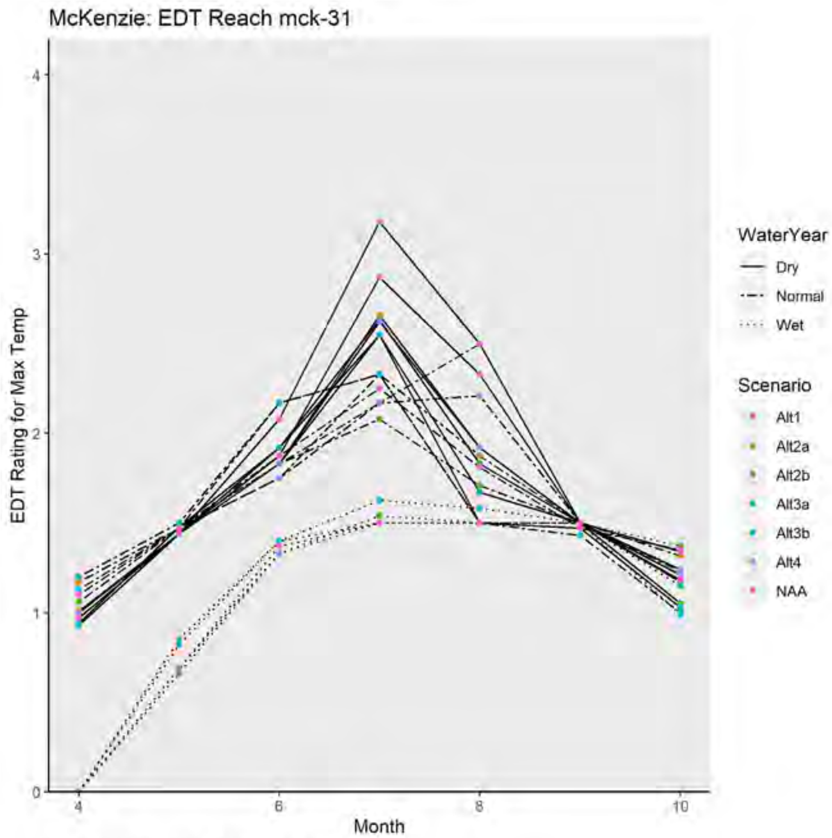

McKenzie: EDT Reach mck-27
EDT Rating for Max Temp
0
1
2
3
4
4
6
8
10
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

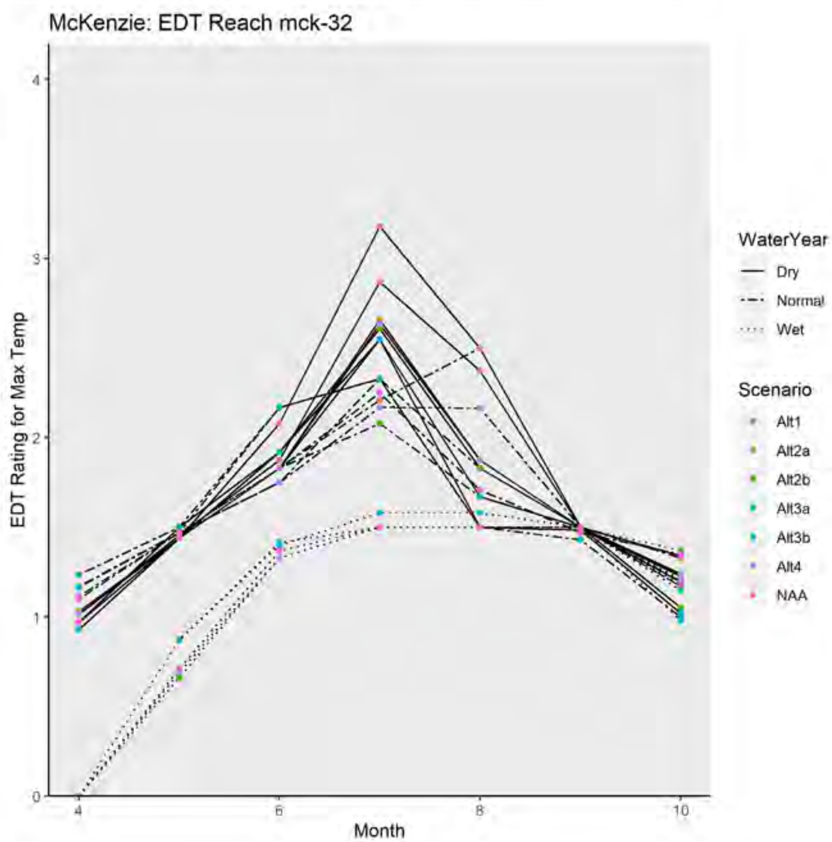


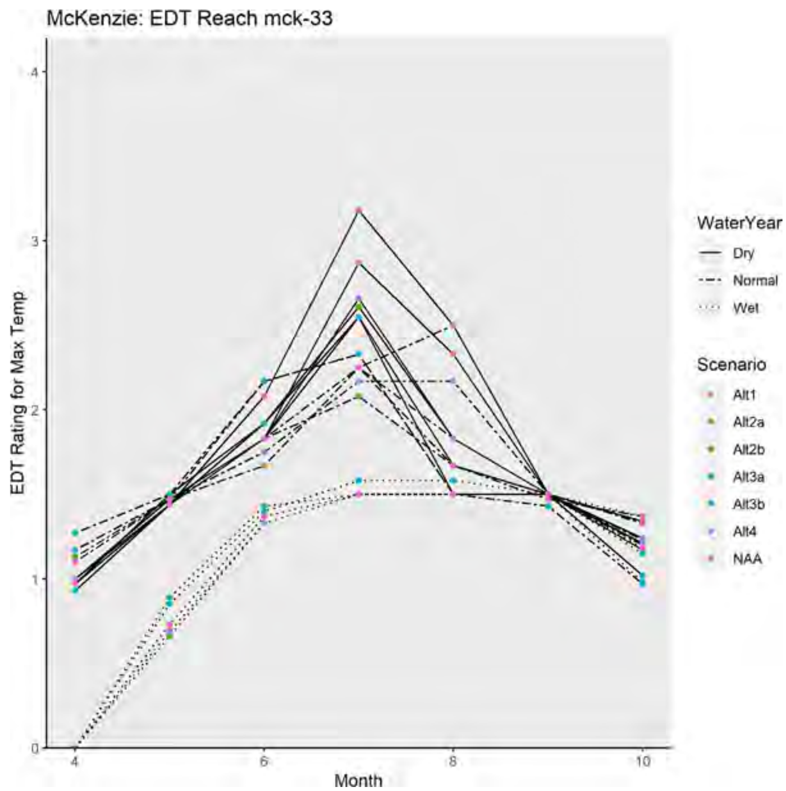
McKenzie: EDT Reach mck-28
EDT Rating for Max Temp
Month
0
1
2
3
4
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

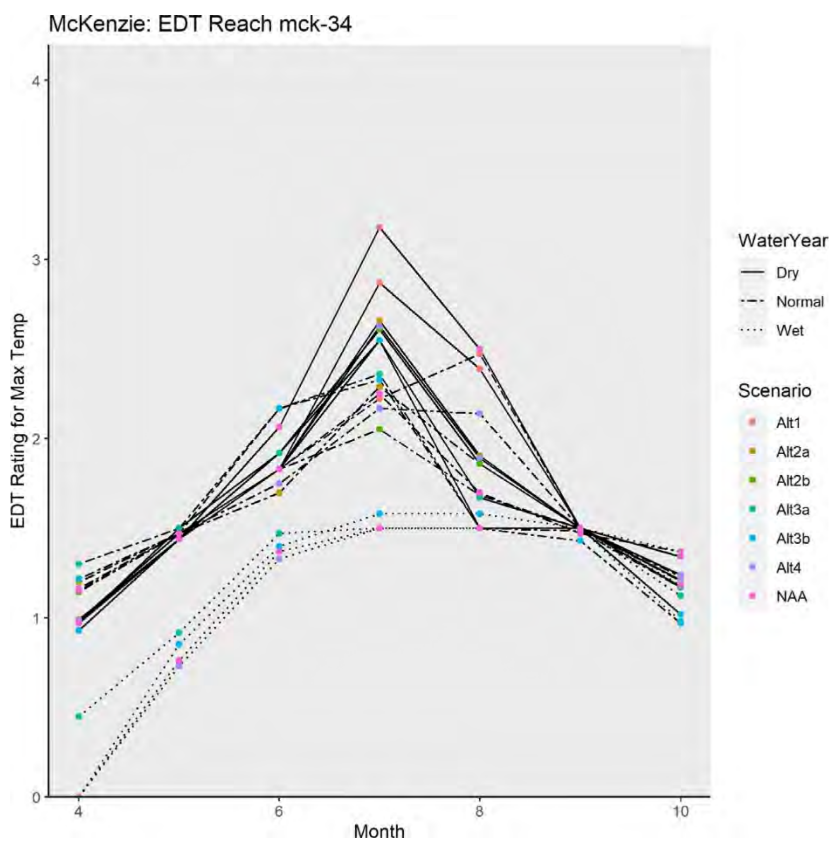


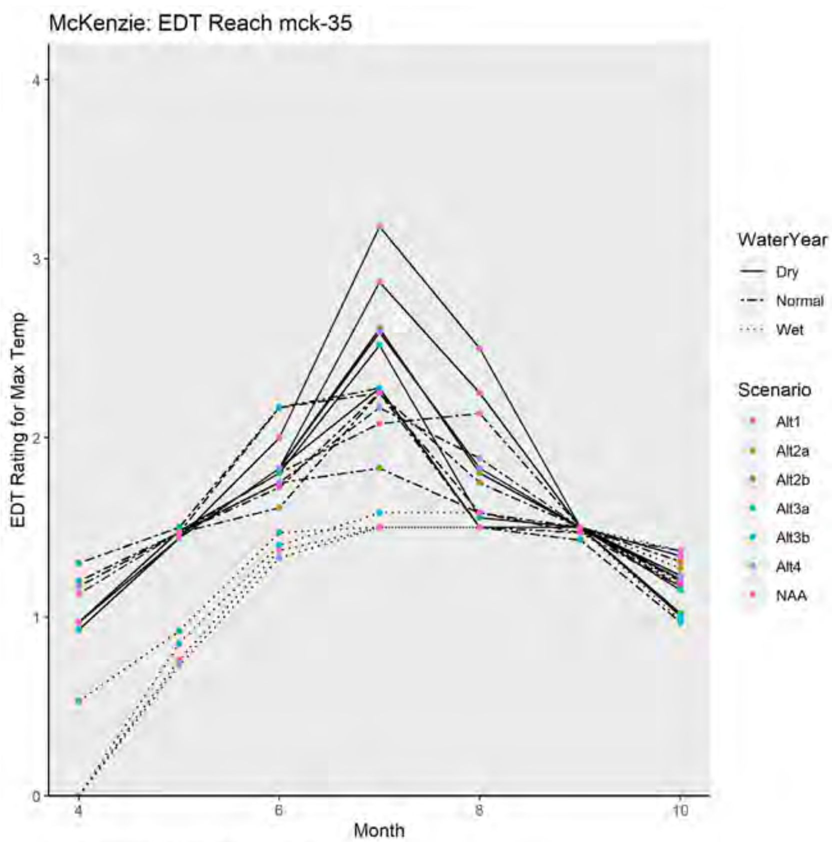
McKenzie: EDT Reach mck-29
EDT Rating for Max Temp
Month
0
1
2
3
4
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

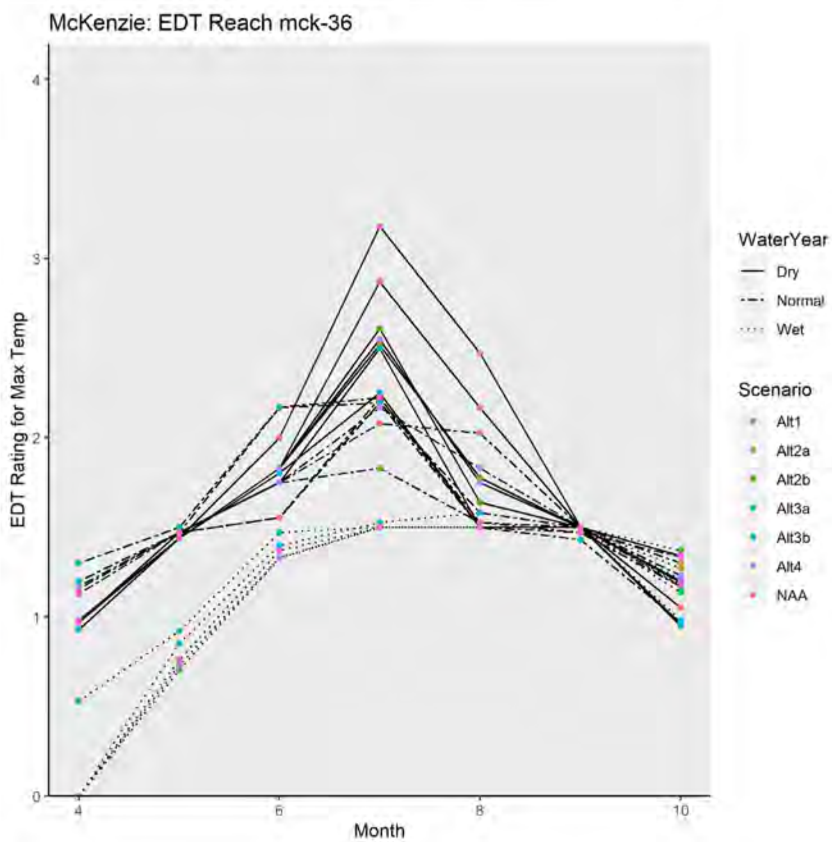

McKenzie: EDT Reach mck-31
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

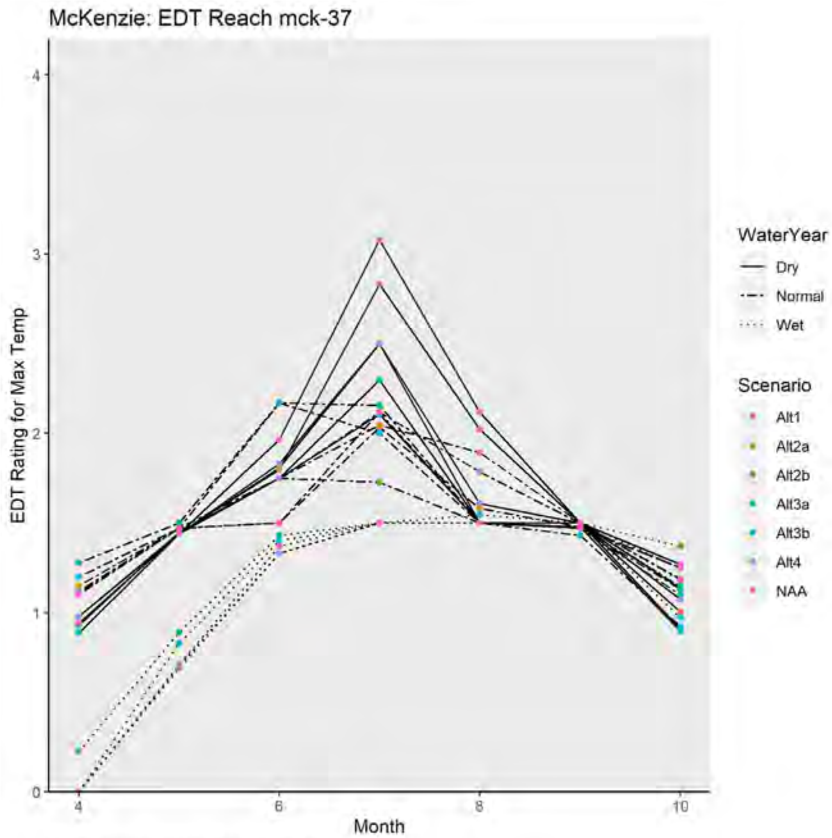

McKenzie: EDT Reach mck-32
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

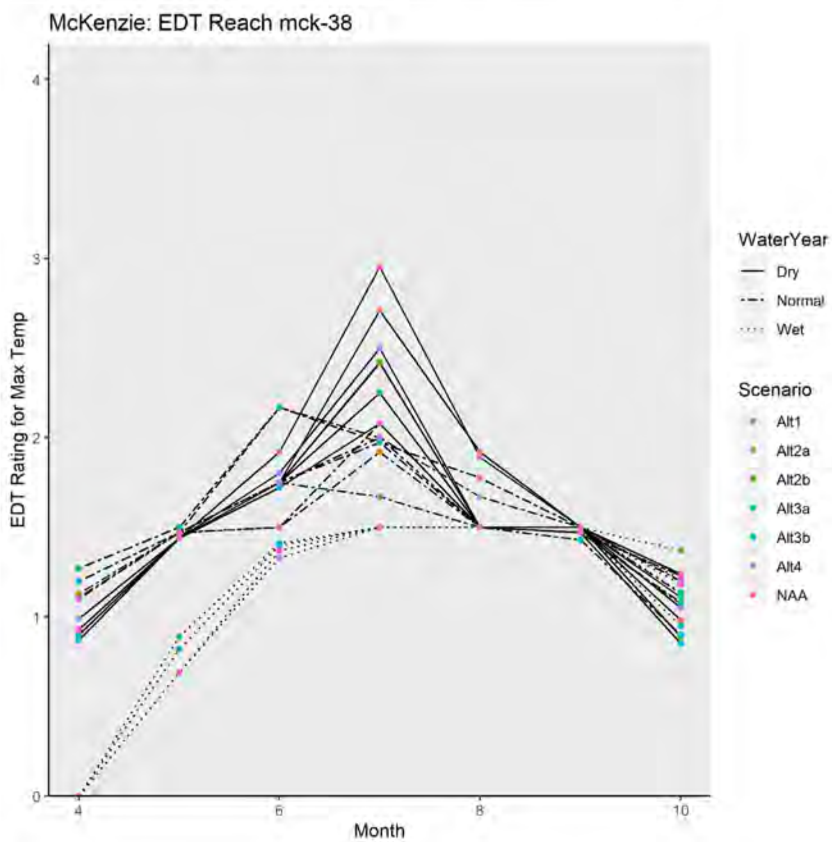

McKenzie: EDT Reach mck-33
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

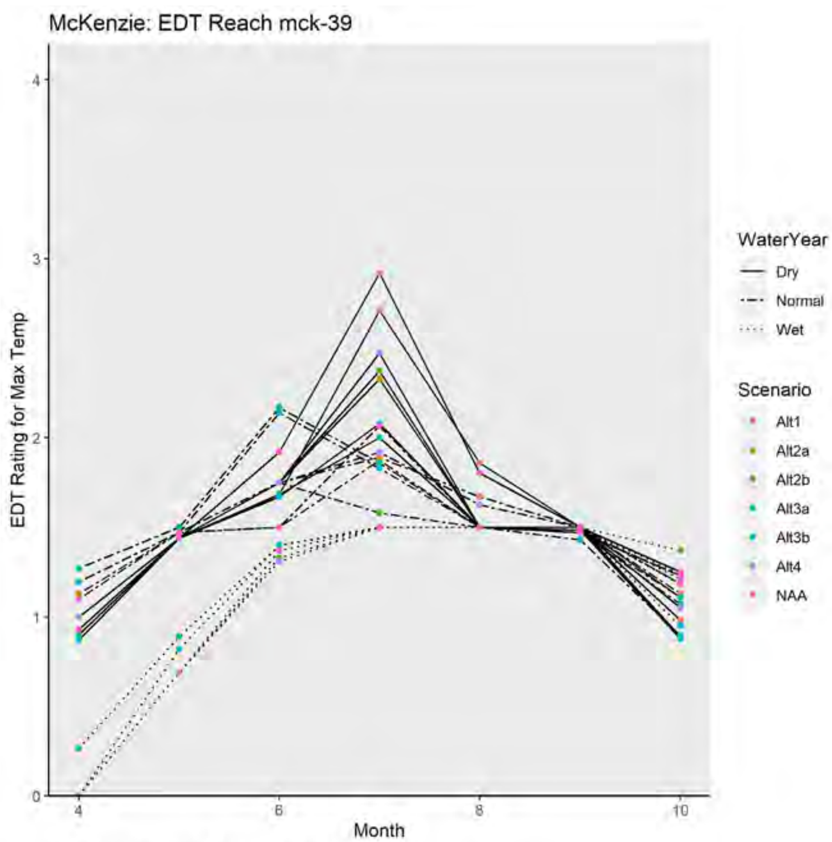

McKenzie: EDT Reach mck-34
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

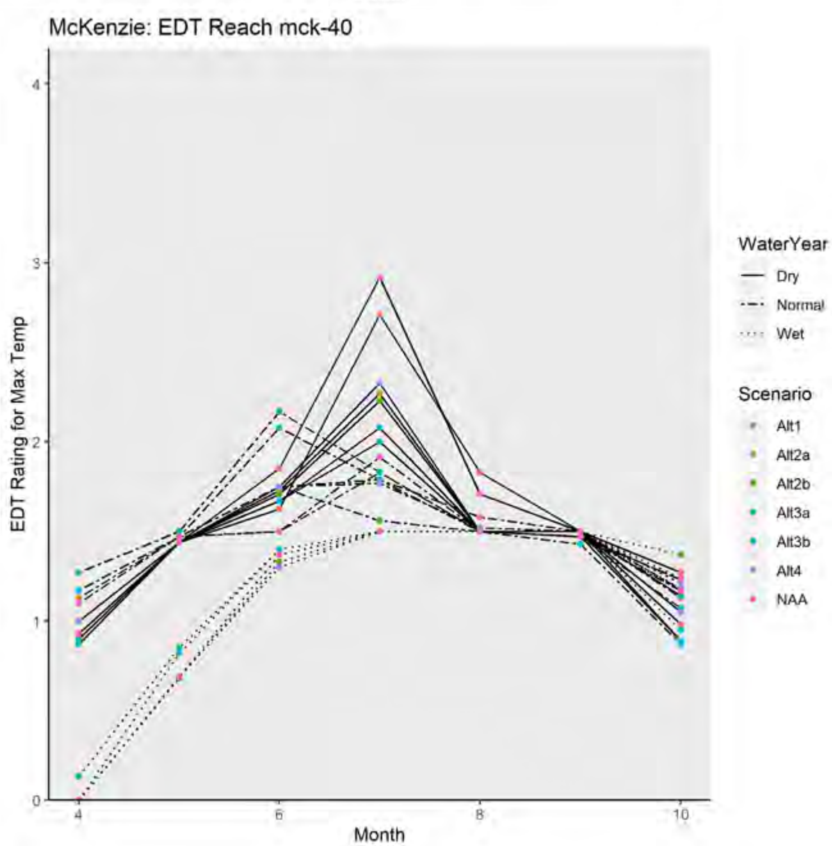

McKenzie: EDT Reach mck-35
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

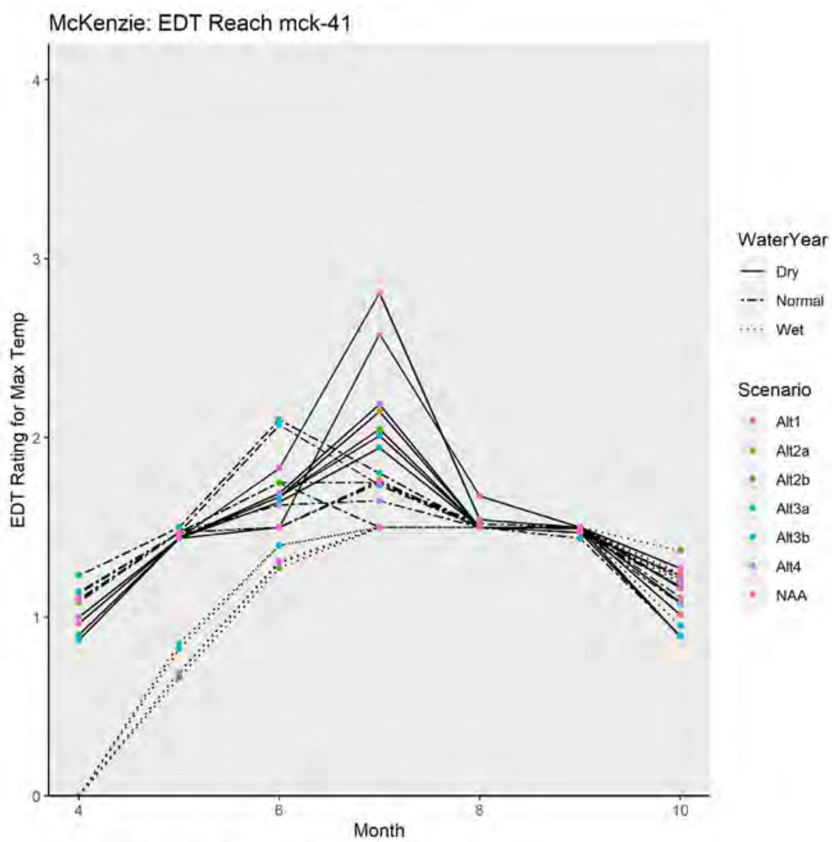

McKenzie: EDT Reach mck-36
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

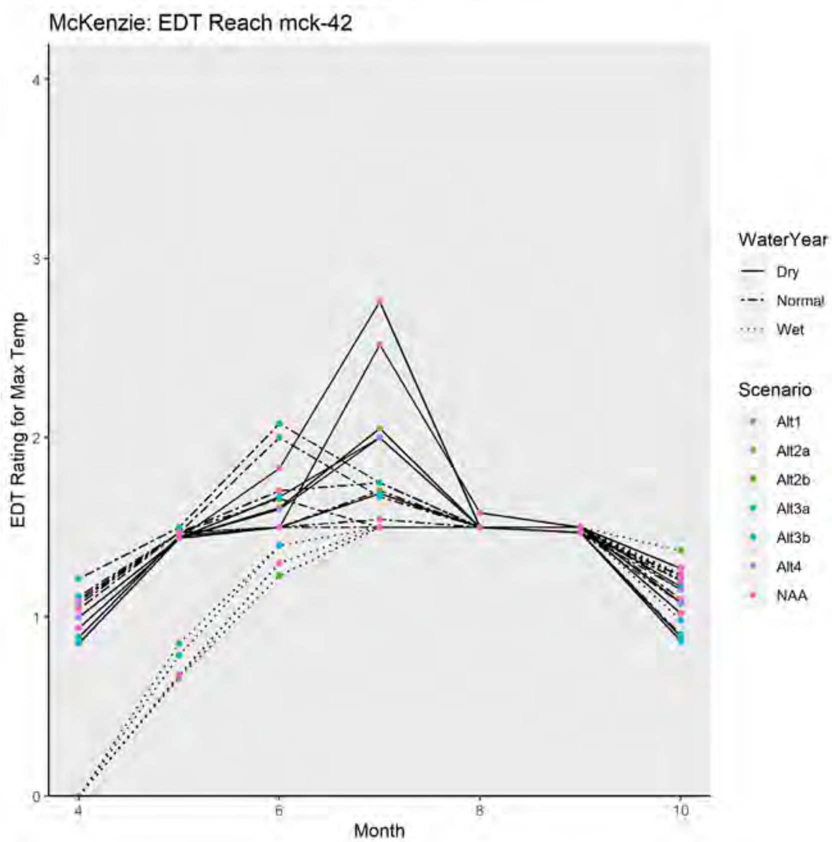

McKenzie: EDT Reach mck-37
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

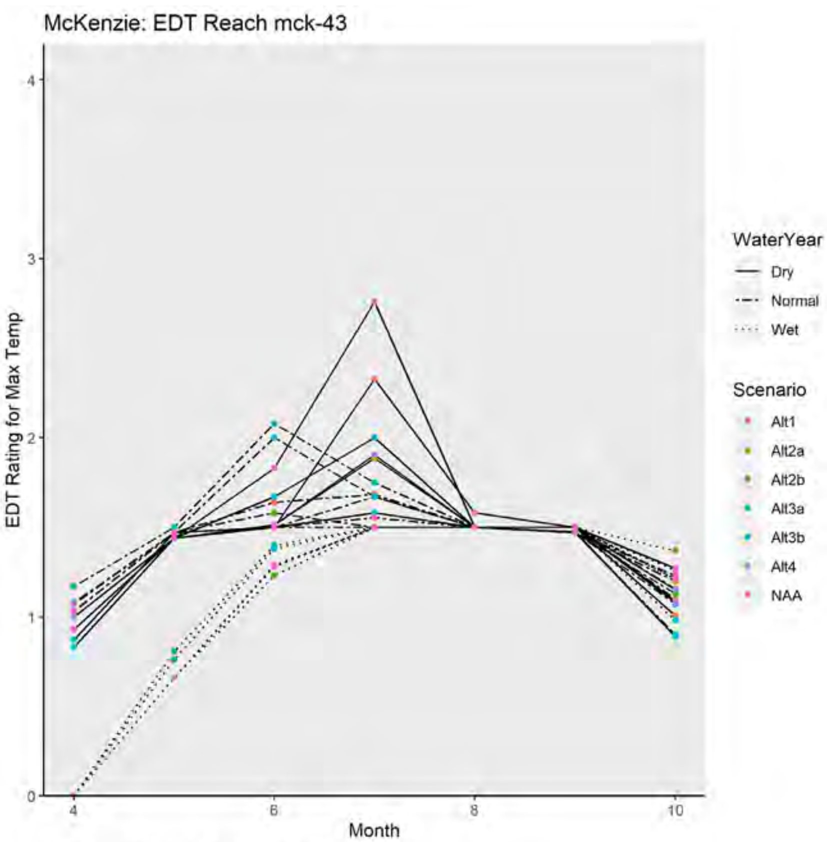

McKenzie: EDT Reach mck-38
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

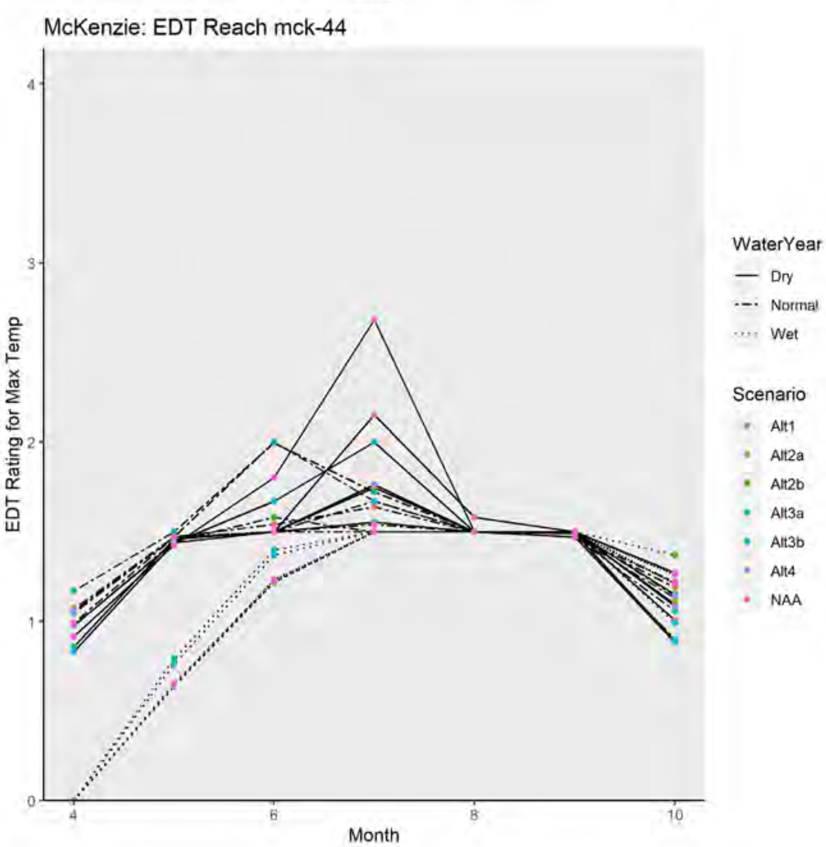

McKenzie: EDT Reach mck-39
EDT Rating for Max Temp
Month
0
1
2
3
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

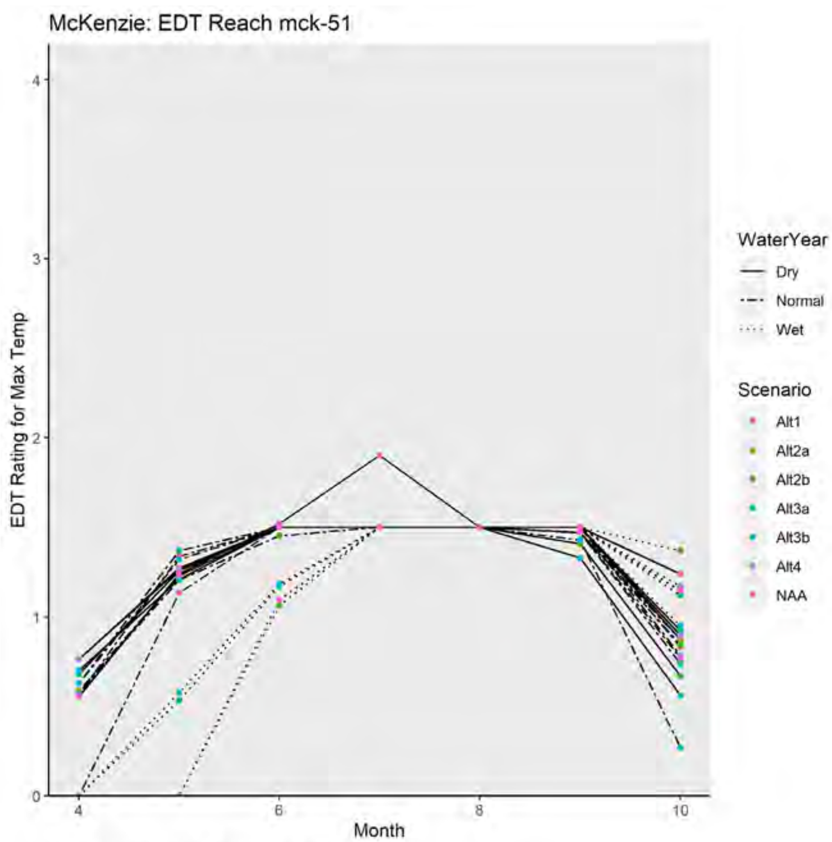

McKenzie: EDT Reach mck-40
EDT Rating for Max Temp
Month
0
1
2
3
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

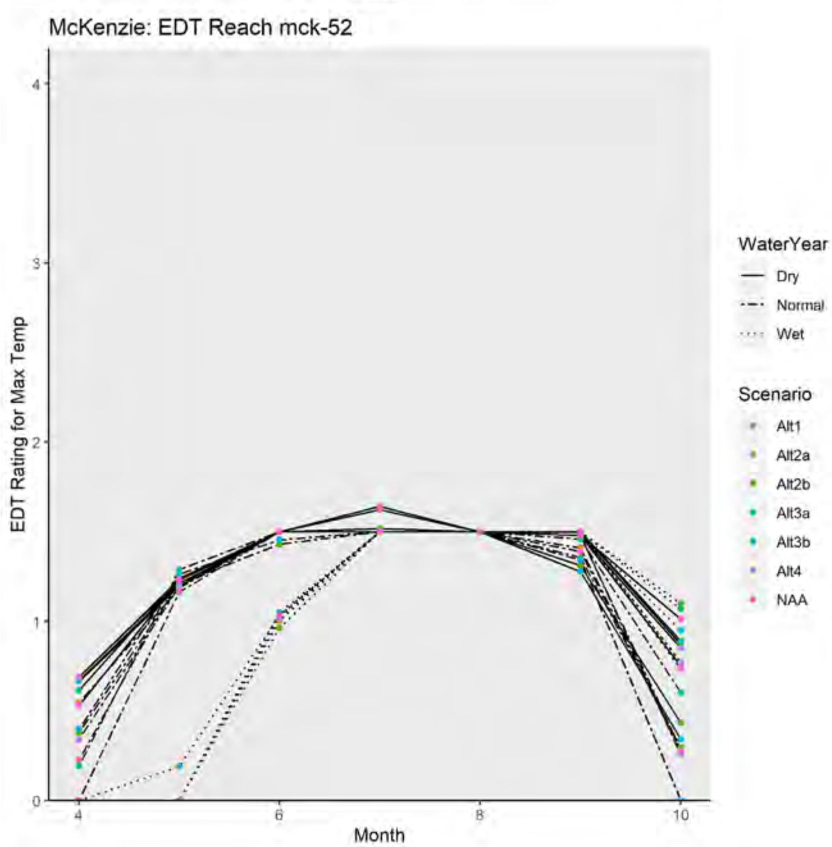

McKenzie: EDT Reach mck-41
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


McKenzie: EDT Reach mck-42
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

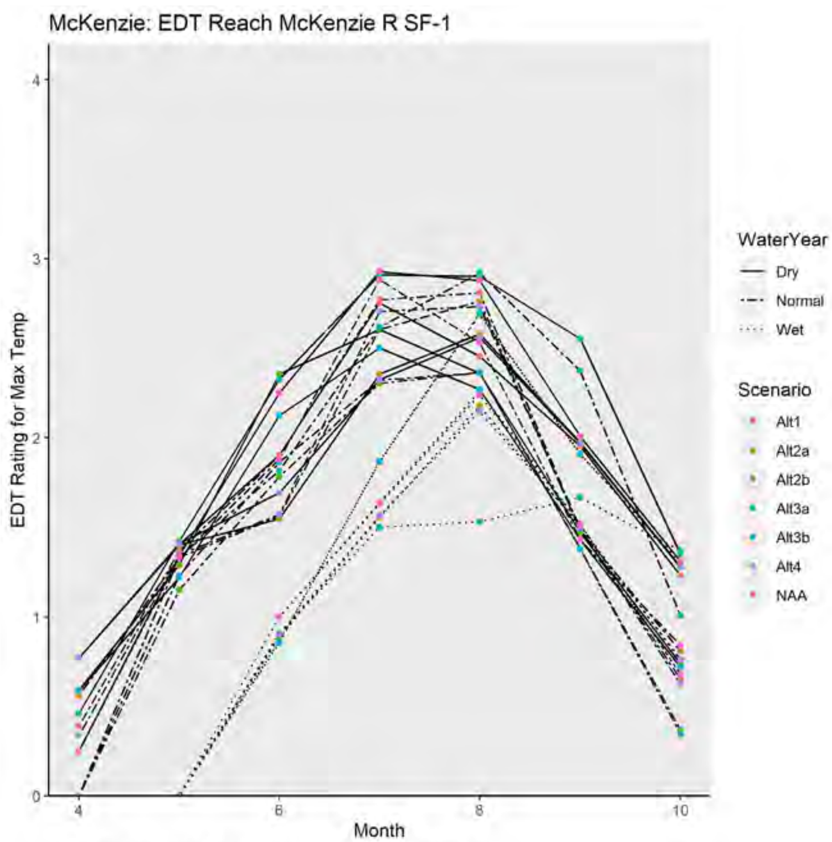

McKenzie: EDT Reach mck-43
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

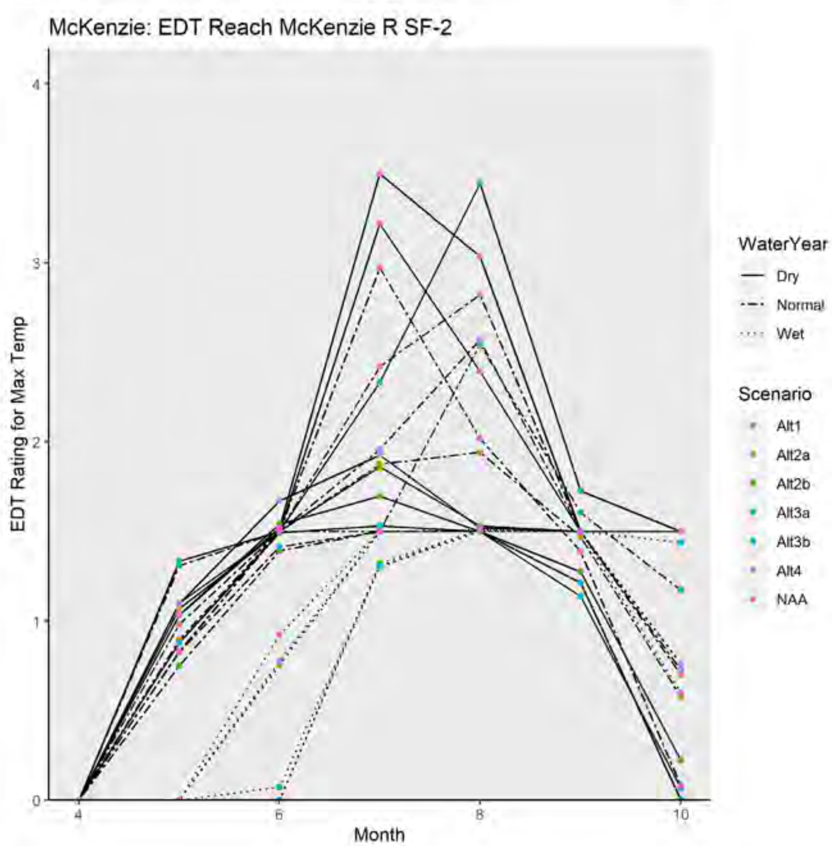

McKenzie: EDT Reach mck-44
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


McKenzie: EDT Reach mck-51
EDT Rating for Max Temp
Month
0
1
2
3
4
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


McKenzie: EDT Reach mck-52
EDT Rating for Max Temp
Month
0
1
2
3
4
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

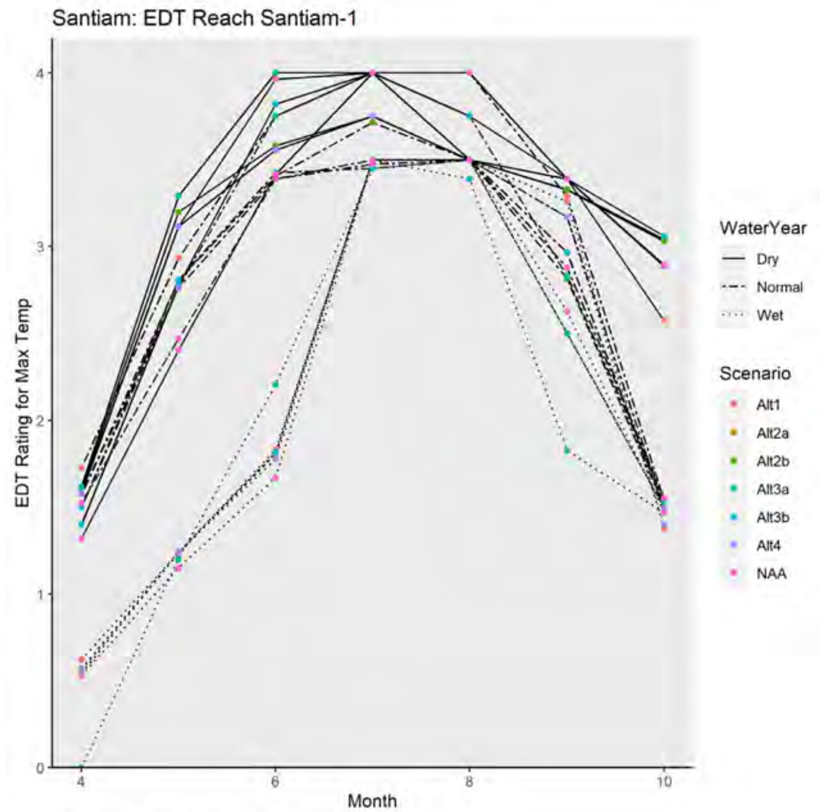
\\\\\\\

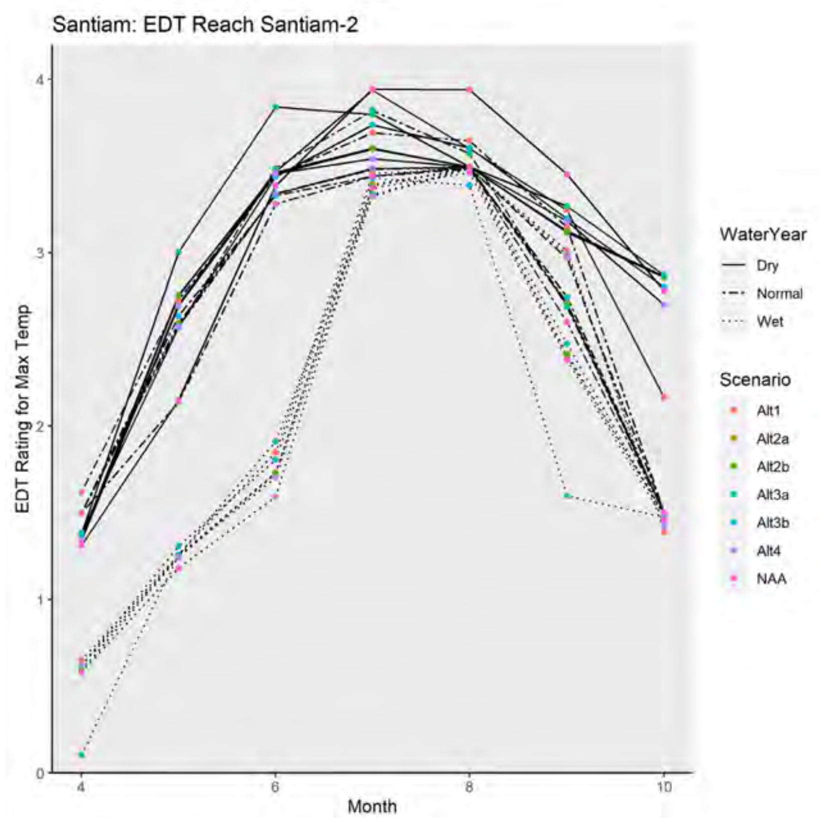

McKenzie: EDT Reach McKenzie R SF-1
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

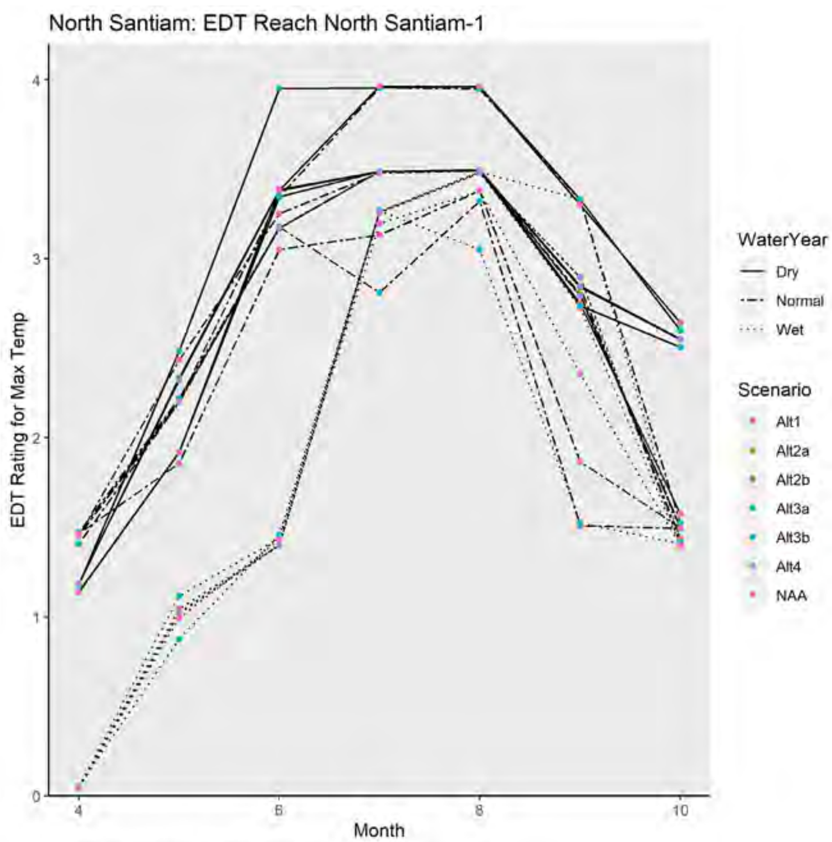

McKenzie: EDT Reach McKenzie R SF-2
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

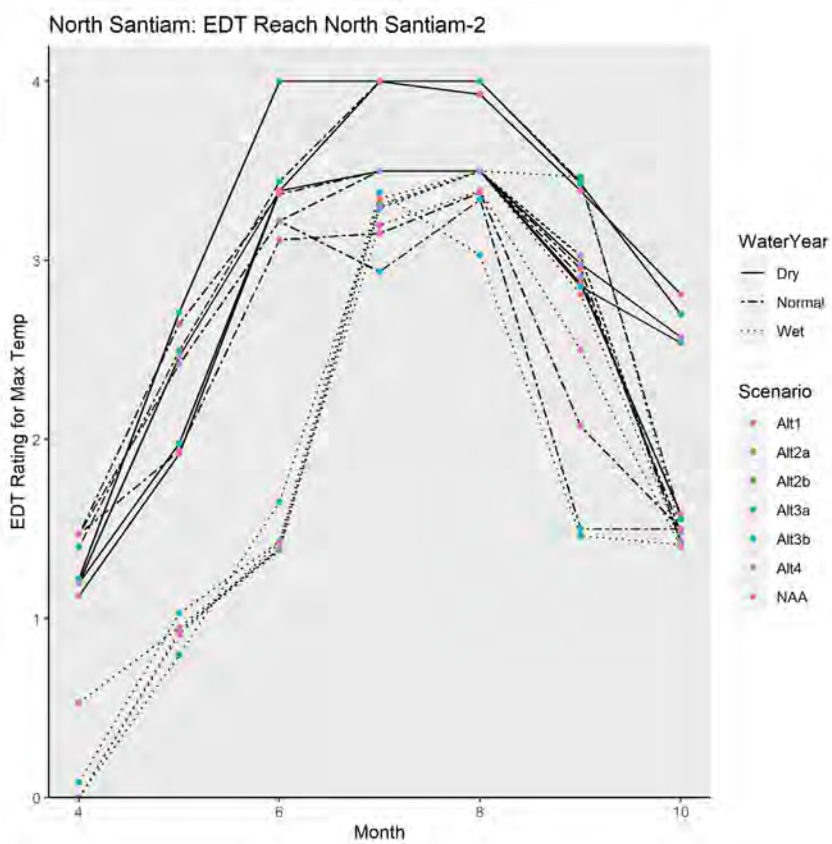
6.8 CHAPTER 6 ATTACHMENT B-4: SANTIAM RIVER

Graphs are organized downstream to upstream: Santiam-1 and -2 are the mainstem Santiam before the split into North and South Santiam; North Santiam-1 through -11 are below Detroit; South Santiam-1 through South Santiam-7 are below Foster.


Santiam: EDT Reach Santiam-1
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


Santiam: EDT Reach Santiam-2
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


North Santiam: EDT Reach North Santiam-1
EDT Rating for Max Temp
0
1
2
3
4
4
6
8
10
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA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EDT Rating for Max Temp
0
1
2
3
4
4
6
8
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Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
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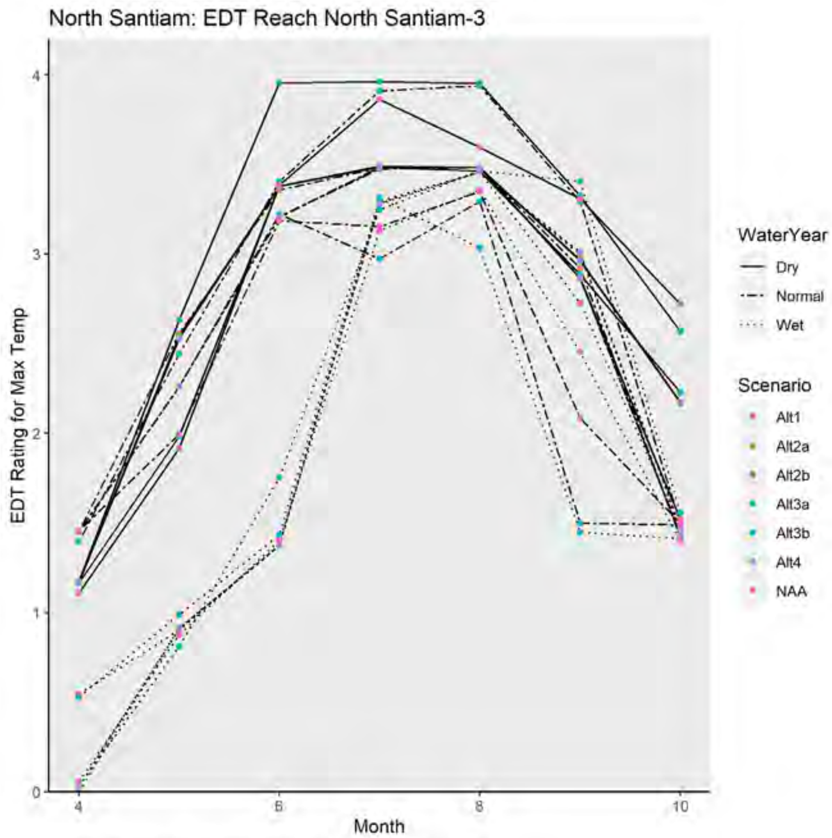
Alt3a
Alt3b
Alt4
NAA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EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
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Alt3a
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Alt4
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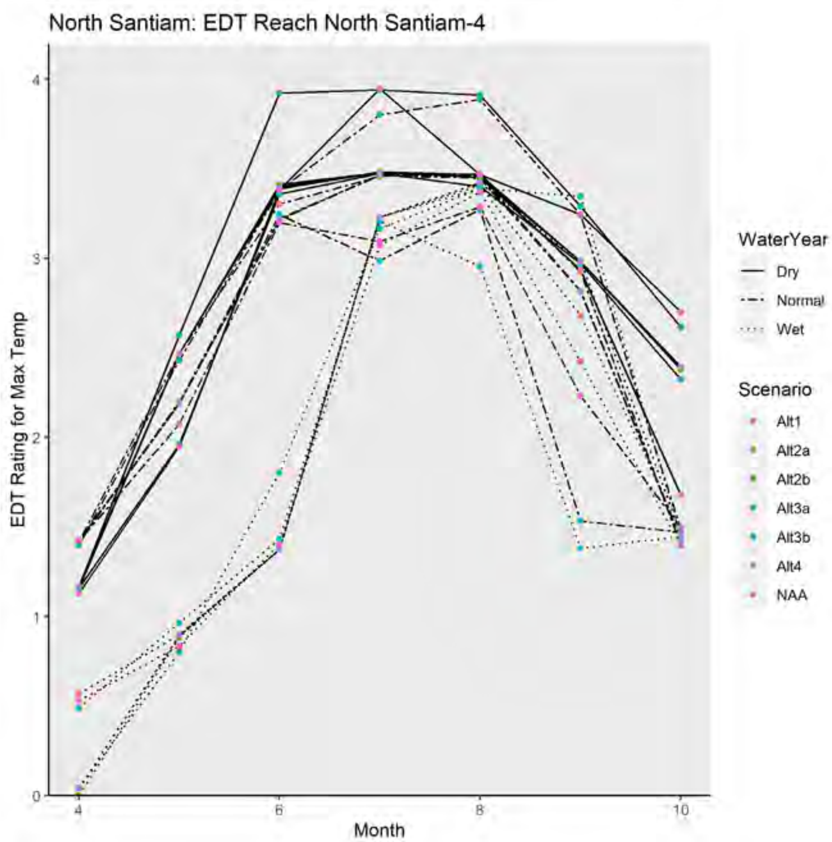
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EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
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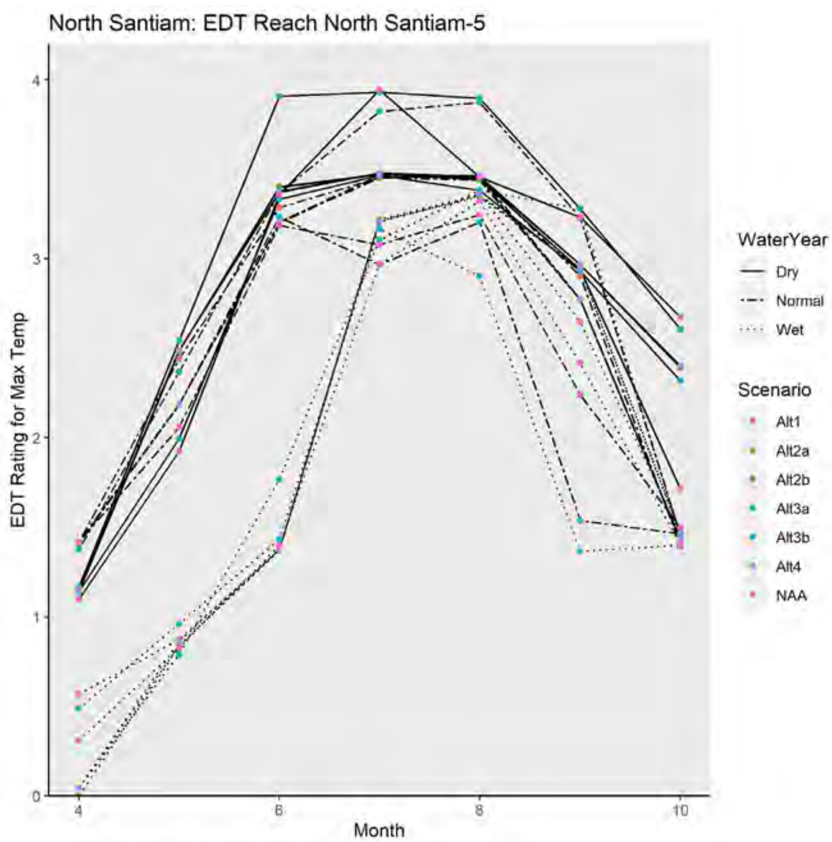
Alt3a
Alt3b
Alt4
NAA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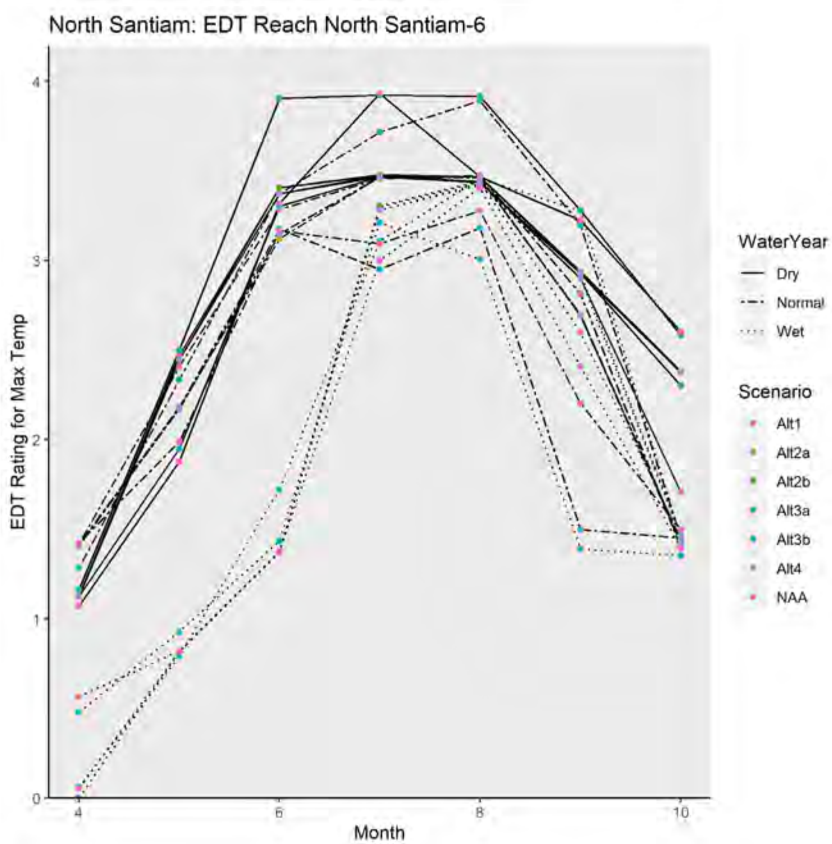
EDT Rating for Max Temp
Month
0
1
2
3
4
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA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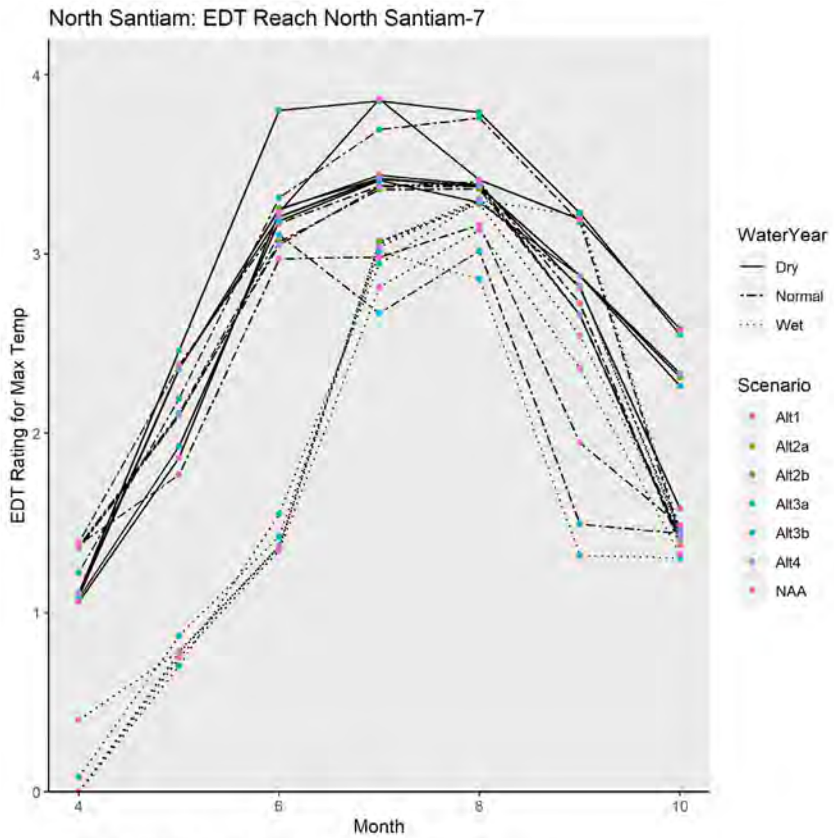
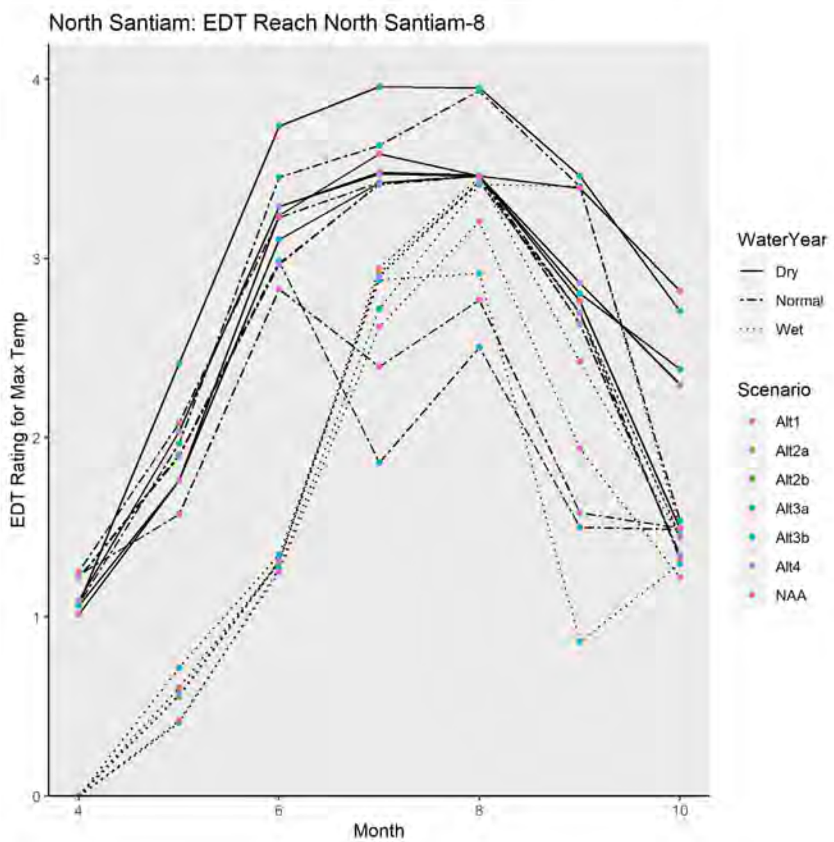
EDT Rating for Max Temp
Month
0
1
2
3
4
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


North Santiam: EDT Reach North Santiam-7
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA


North Santiam: EDT Reach North Santiam-8
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
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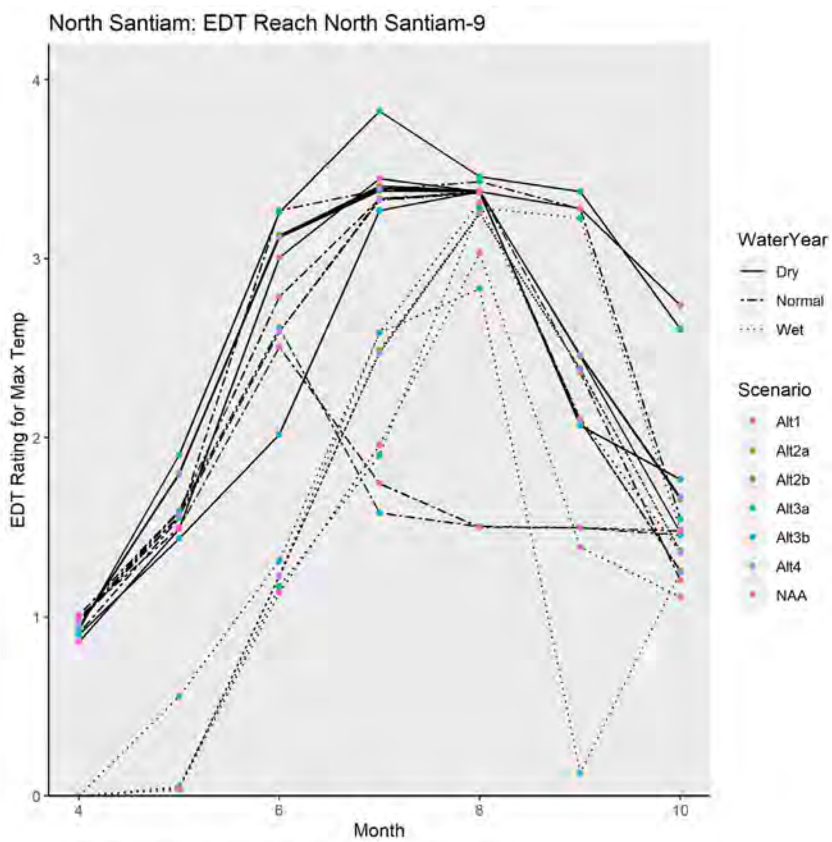
Alt3a
Alt3b
Alt4
NAA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EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
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Alt3b
Alt4
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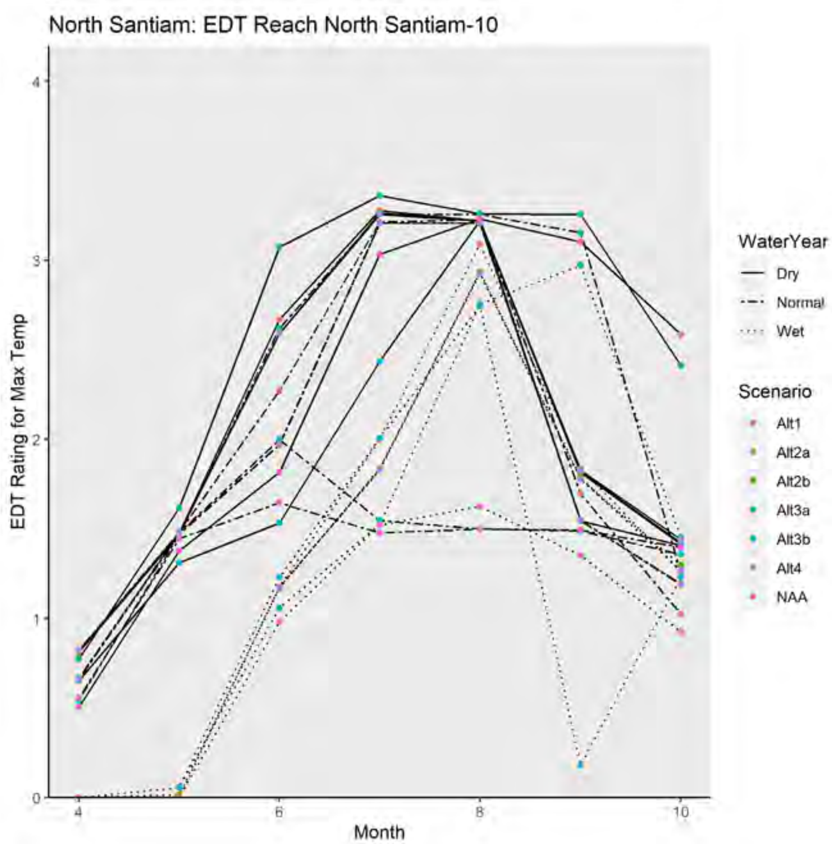
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EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
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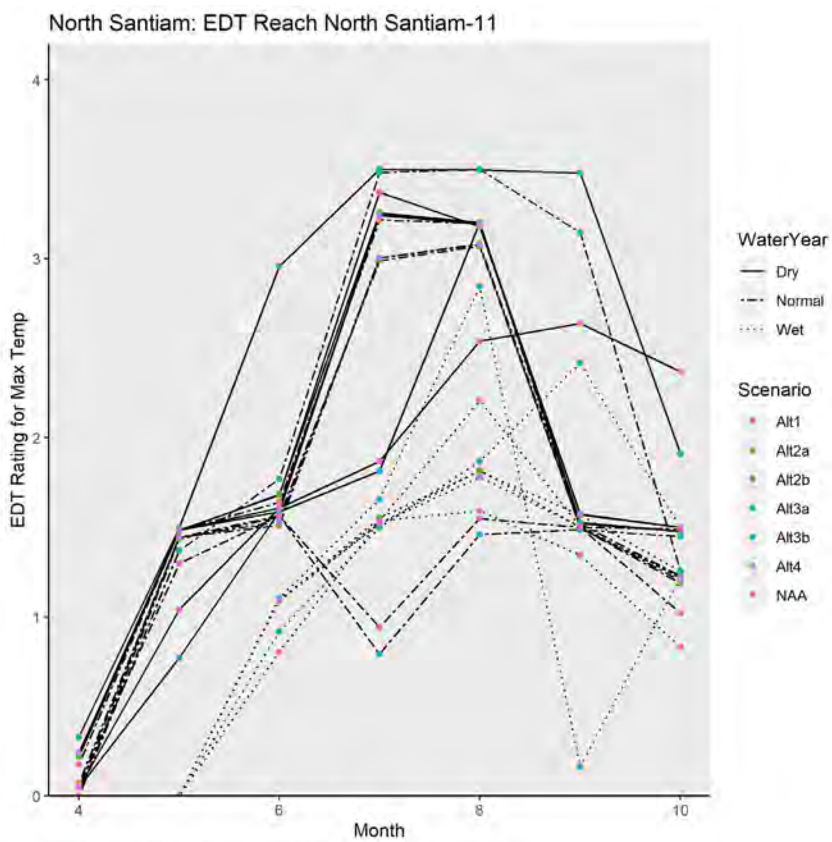
Alt3a
Alt3b
Alt4
NAA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EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
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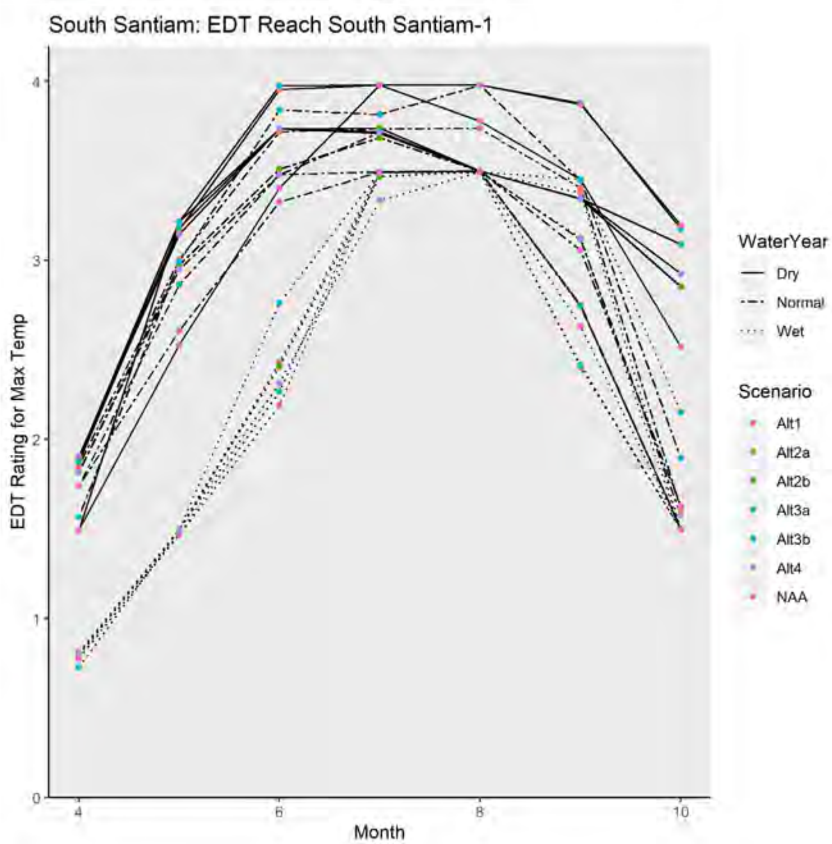
Alt2b
Alt3a
Alt3b
Alt4
NAA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EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
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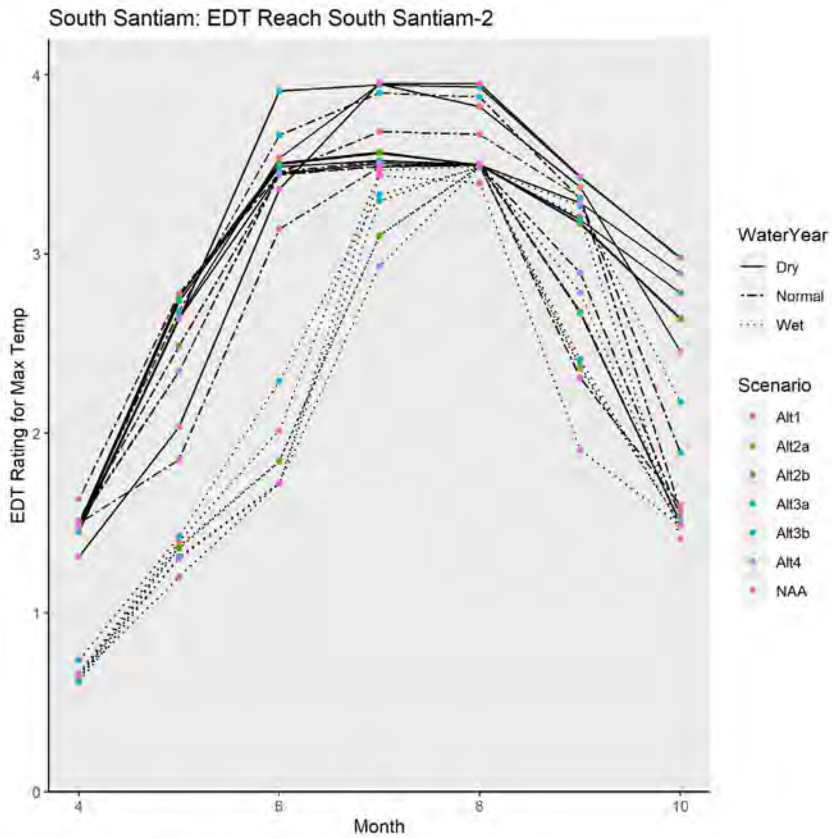
Alt3b
Alt4
NAA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EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
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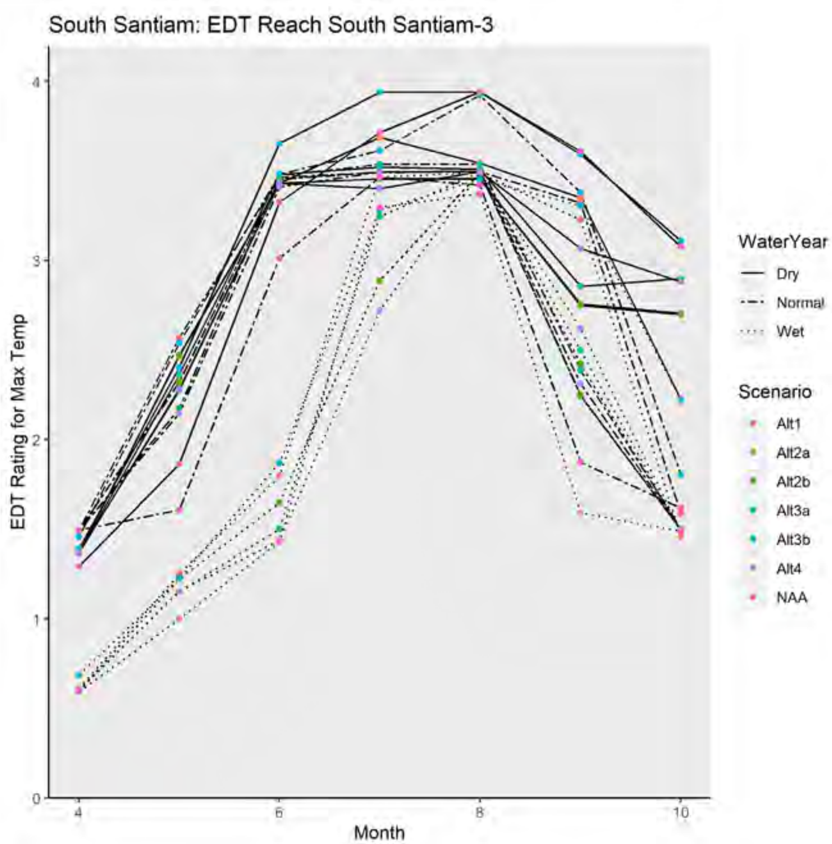
Alt3a
Alt3b
Alt4
NAA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EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
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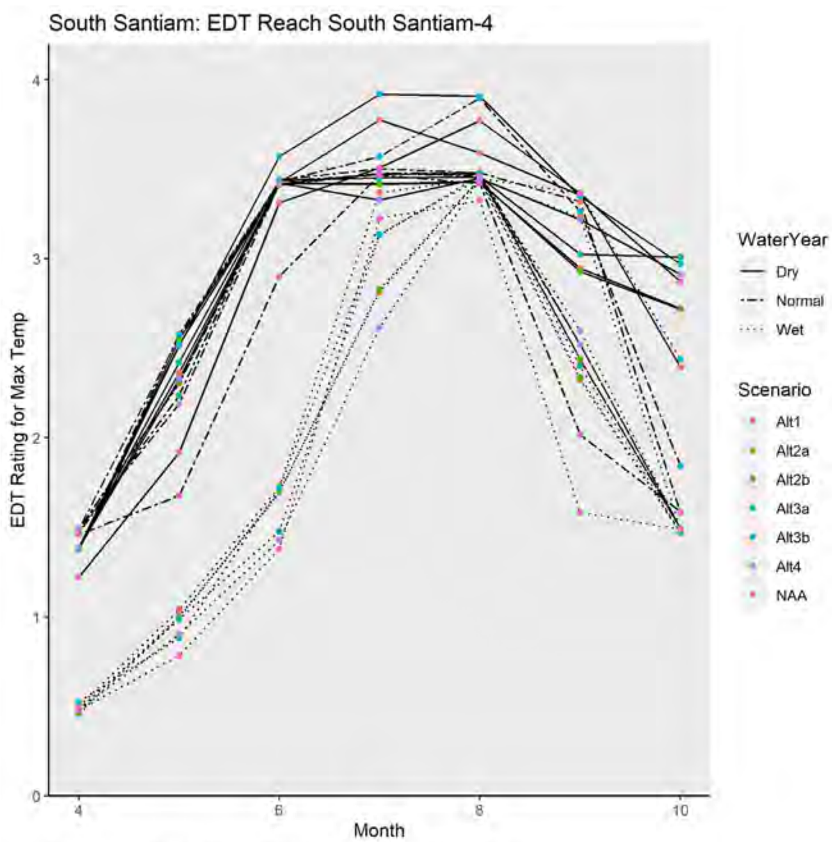
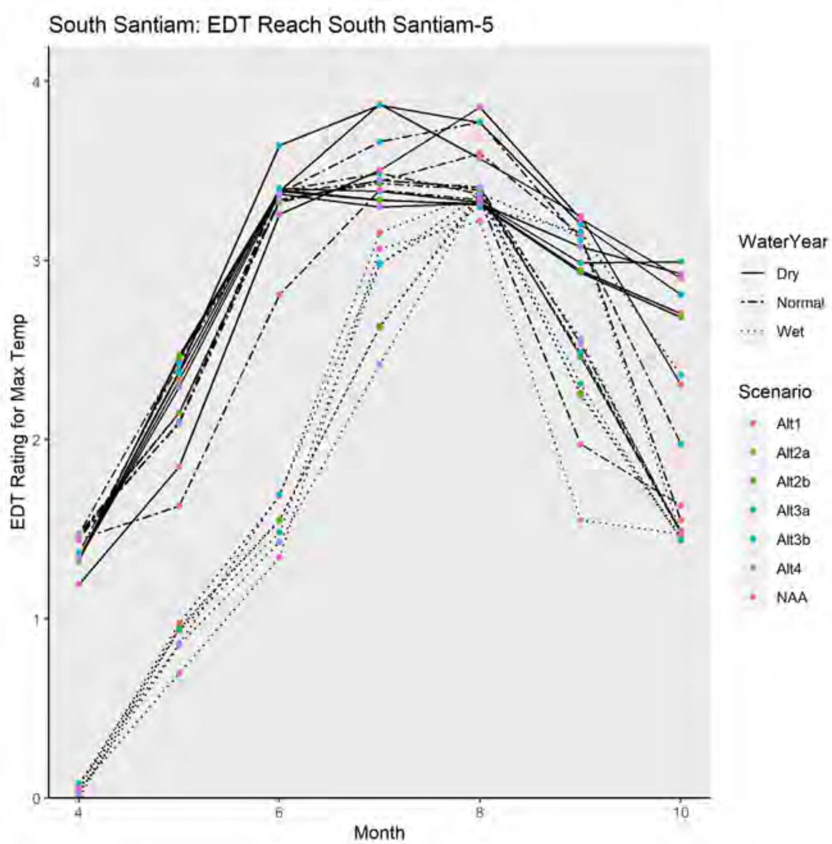
Alt3a
Alt3b
Alt4
NAA


South Santiam: EDT Reach South Santiam-4
EDT Rating for Max Temp
Month
0
1
2
3
4
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA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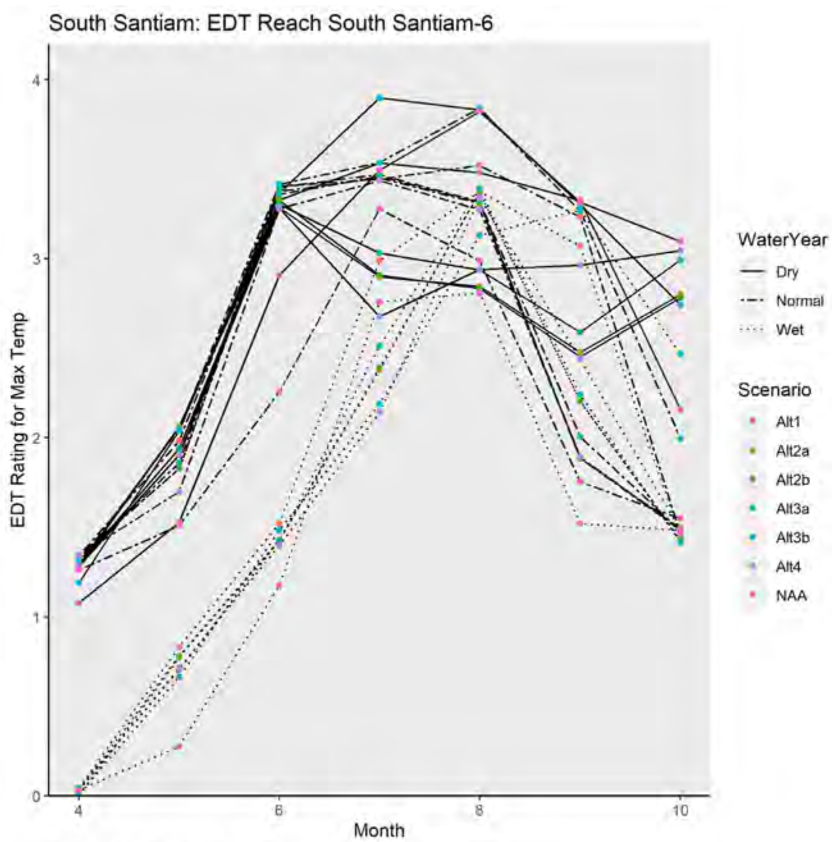
EDT Rating for Max Temp
Month
0
1
2
3
4
4
6
8
10
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA



South Santiam: EDT Reach South Santiam-6
EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
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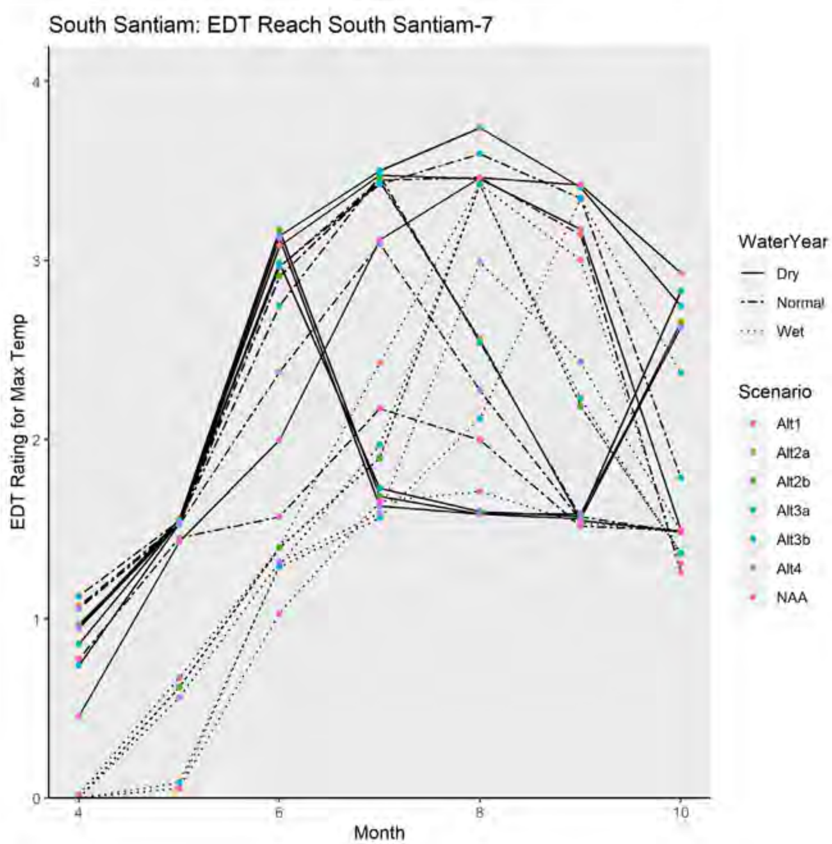
Alt3b
Alt4
NAA



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EDT Rating for Max Temp
Month
WaterYear
Dry
Normal
Wet
Scenario
Alt1
Alt2a
Alt2b
Alt3a
Alt3b
Alt4
NAA

6.9 CHAPTER 6-ATTACHMENT C: HIGH AND LOW FLOW RATINGS

MEMORANDUM

To: USACE, Willamette Basin Project

From: Laura McMullen and Janel Sobota, ICF

Date: Updated February 22, 2022

Re: EDT Ratings of Flow in the Willamette Basin Under Alternative Scenarios

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Here we present the EDT (Ecosystem Diagnosis & Treatment) ratings of changes in inter-annual variability of high and low flows under the No Action Alternative (NAA), Alternative 1 (Alt 1), Alternative 2a (Alt 2a), Alternative 2b (Alt 2b), Alternative 3a (Alt 3a), Alternative 3b (Alt 3b) and Alternative 4 (Alt 4) scenarios for Wet (2011), Dry (2015), and Normal (2016) water year conditions. Willamette Basin stream flow was modeled using the Res-Sim program and was provided to ICF by USACE for both regulated and unregulated management. Mean daily flow was modeled at each dam and several river sites downstream for a total of 19 sites.

Flow – Changes in Inter-annual Variability in High and Low Flows

Outputs from the Res-Sim model were used to calculate attribute ratings that will be input into the EDT model to describe habitat conditions that salmon are exposed to as they move through the system.

Changes in the timing and quantity of flow due to land uses and flow regulation, can affect responses of salmonids leading to changes in overall performance of their populations (LeStelle 2005). In EDT, two of the attributes used to describe relative change in flow during high and low flow periods related to land use effects on hydrological patterns are Flow High and Flow Low. For the purposes of this memo, the regulated flow patterns were compared to the unregulated flow within a water-year condition to rate the relative change. For example, flow in NAA, Normal (2016) was compared to unregulated flow for Normal (2016). There is a two-part process for calculating the ratings for these flow attributes by first determining the highest rating possible for the scenario and then establishing the monthly pattern- scalar that is used to calculate the other month's ratings.

To assess changes in inter-annual variability of high flows for an alternative, the change in discharge within the same water-year condition was compared between the alternative's regulated and unregulated flow and rated according to the protocol outlined in Table C-1. The index values for the high-flow EDT attribute are scaled to the unregulated state which are assigned a rating of 2. Shifts toward a higher peak discharge are represented by increases toward ratings of 3 and 4, reduced peaks by values of 0 and 1. The higher (closer to 4) the rating is, the more affect the rating would have on salmonid survival. For Flow High, we determine the percentage change of the regulated scenario from unregulated in the month with the highest flow (Flow High) which sets the highest rating possible and is assigned to the month with the highest mean flow. For example, Big Cliff Alt1, under Normal (2016) water-year conditions had a peak flow of 8,967 cfs (cubic feet per second) and the unregulated Normal (2016) peak was 7490 cfs. There is a +19% change in peak-flows between the regulated and unregulated flow so this site has rating of 2.48.

Changes in the inter-annual variability in low flows were evaluated within a water-year by the relative change in the 45-day low flow averages between regulated and unregulated flow and rated according to the protocol outlined in Table C-2. Like high-flow attribute ratings, a Rating Index of 2 would be assigned to low flows, during the lowest flow period, which were the same as the unregulated state. Ratings higher than 2 indicate a shift toward lower low-flow discharge and ratings less than 2 indicate increased low-flow discharge. Again, using the dataset for Normal (2016) water-year conditions for Big Cliff Alt1, the lowest 45-day average base flow was 1478 cfs and 1892 cfs for unregulated flow. There is a -22% relative change between the regulated and unregulated flow and a peak rating of 2.56 was assigned to this site for the Flow Low EDT attribute.

After setting the scale for the Flow High and Flow Low attributes, for the highest and lowest flow months respectively, values are scaled back towards 0 according to the flow pattern.

Table 6-24. EDT ratings (Index Values) for changes in inter-annual variability in high flows.

Index 0	Index 1	Index 2	Index 3	Index 4
Peak annual flows expected to be strongly reduced relative to an undisturbed watershed of similar size, geology, orientation, topography, and geography (or the pristine state for the watershed of interest); OR >40% and <100% decrease in Q_{2yr} based on a long time series (~40 yrs or longer with at least 20 yrs pertaining to a watershed development state) or as known by regulated flow levels. This condition is associated with flow regulation or water diversion projects.	Peak annual flows expected to be moderately reduced relative to an undisturbed watershed of similar size, geology, orientation, topography, and geography (or the pristine state for the watershed of interest); OR >20% and <40% decrease in Q_{2yr} based on a long time series (~40 yrs or longer with at least 20 yrs pertaining to a watershed development state) or as known by regulated flow levels. This condition is associated with flow regulation or water diversion projects.	Peak annual flows expected to be comparable to an undisturbed watershed of similar size, geology, orientation, topography, and geography (or the pristine state for the watershed of interest); OR <20% change in Q_{2yr} based on a long time series (~40 yrs or longer with at least 20 yrs pertaining to a watershed development state); OR <5% reduction in average T_{Qmean} compared to the undeveloped watershed state.	Peak annual flows expected to be moderately increased relative to an undisturbed watershed of similar size, geology, orientation, topography, and geography (or the pristine state for the watershed of interest); OR >20% and <40% increase in Q_{2yr} based on a long time series (~40 yrs or longer with at least 20 yrs pertaining to a watershed development state); OR >5% and <15% reduction in average T_{Qmean} compared to the undeveloped watershed state. This condition exemplified in some forested watersheds with high road density that experience significant rain on snow events, as the North Fork Stillaguamish River (Pess and others *in review*). Note: many managed forested watersheds in the Pacific Northwest exhibit slight, if any, increases in peak annual flows since logging commenced (see Ziemer and Lisle 1998).	Peak annual flows expected to be strongly increased relative to an undisturbed watershed of similar size, geology, orientation, topography, and geography (or the pristine state for the watershed of interest); OR >40% and <110%+ increase in Q_{2yr} based on a long time series (~40 yrs or longer with at least 20 yrs pertaining to a watershed development state); OR >15% and <45% reduction in average T_{Qmean} compared to the undeveloped watershed state. This condition exemplified in watersheds with significant urbanization (e.g., >20%).

Table 6-25. EDT ratings (Index Values) for changes in inter-annual variability in low flows.

Index 0	Index 1	Index 2	Index 3	Index 4
Average daily low flows expected to be strongly increased compared to an undisturbed watershed of similar size, geology, and flow regime (or the pristine state for the watershed of interest); OR >75% increase in the 45 or 60-day consecutive lowest average daily flow on a sufficiently long time series (~40 yrs or longer with at least 20 yrs pertaining to a watershed development state) or as known through flow regulation.	Average daily low flows expected to be moderately increased compared to an undisturbed watershed of similar size, geology, and flow regime (or the pristine state for the watershed of interest); OR >20% and <75% increase in the 45 or 60-day consecutive lowest average daily flow on a sufficiently long time series (~40 yrs or longer with at least 20 yrs pertaining to a watershed development state) or as known through flow regulation.	Average daily low flows expected to be comparable to an undisturbed watershed of similar size, geology, and flow regime (or the pristine state for the watershed of interest); OR <20% change in the 45 or 60-day consecutive lowest average daily flow on a sufficiently long time series (~40 yrs or longer with at least 20 yrs pertaining to a watershed development state).	Average daily low flows expected to be moderately reduced compared to an undisturbed watershed of similar size, geology, and flow regime (or the pristine state for the watershed of interest); OR >20% and <50% reduction in the 45 or 60-day consecutive lowest average daily flow on a sufficiently long time series (~40 yrs or longer with at least 20 yrs pertaining to a watershed development state) or as known through flow regulation.	Average daily low flows expected to be severely reduced compared to an undisturbed watershed of similar size, geology, and flow regime (or the pristine state for the watershed of interest); OR >50% and <=100% reduction in the 45 or 60-day consecutive lowest average daily flow on a sufficiently long time series (~40 yrs or longer with at least 20 yrs pertaining to a watershed development state) or as known through flow regulation.

Initial Findings

In the appendices, EDT ratings evaluating changes in inter-annual variability of high and low flow conditions between regulated and unregulated management within a water-condition are graphed for all scenarios. We can see that water-year conditions (Normal (2016), Dry (2015), and Wet (2011)) affect high and low flow ratings regardless of the management alternative for all sites and that ratings vary by management scenario.

Using Blue River in Appendix C-1, as an example to interpret the results, we explore some of the findings. Looking at ratings for changes in high flow under normal water-year conditions, the attribute varies under different management alternatives. In most winter months Alternatives 1-4 have slightly higher ratings than NAA. The ratings reflect that under the NAA alternative, the average high flows in Oct-Jan are lower than what would be expected under unregulated flows, while those of Alt 1 and Alt 4 are close to unregulated high-flows (rating of 2). Under both wet and dry water-years, high-flow ratings reflect a highly managed system (ratings <1) with reduced high-flows in both wet and dry years that do not differ greatly between alternatives. Of note is that flows in February, for all alternatives and water-year conditions, are lower than what might be expected in a naturally flowing system which is reflected in the “0” rating for the high-flow attribute.

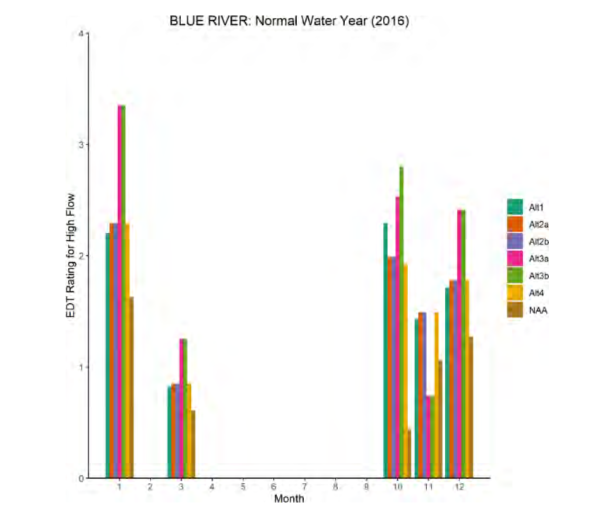
During the summer months under normal water-year conditions, ratings in the low flow attribute tend toward 2 and above indicating equivalent or slightly lower low-flows in the regulated vs. unregulated conditions. Under wet conditions, while individual scenarios vary slightly in their ratings, June and September ratings are often <1 indicating higher low-flows during that month than under unregulated conditions. In July and August all alternatives have ratings >2, indicating lower flows than unregulated conditions for those months. In drier conditions (Dry 2015), low-flow ratings in early spring (March-April) and later summer (Sept-Oct) have ratings >2 as well, lower flows than unregulated conditions. Within a dry year, the usually hotter summer months in the Pacific Northwest (July-August), ratings for Alt 3b remain higher than the other alternatives other than no-action.

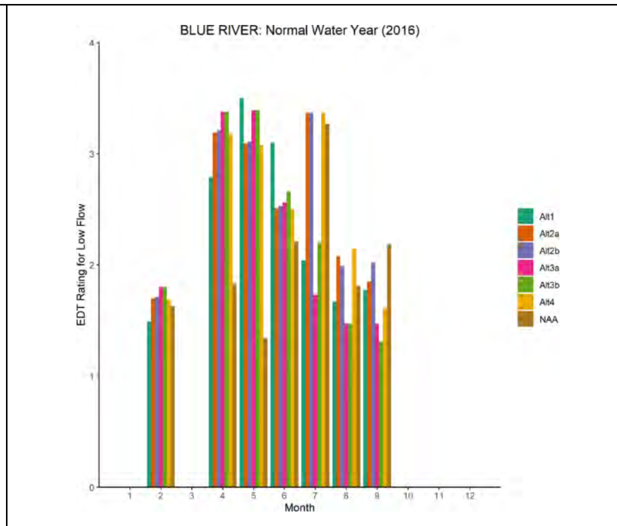
For most of the sites, ratings evaluating changes in high flow between regulated alternatives and unregulated flow reflect the water regulation in the systems. Water-year conditions strongly influence the changes in high-flow seen under the alternatives. Changes in low-flow conditions between unregulated and regulated alternatives are more variable under all water-year conditions than high-flow ratings and are also strongly influenced by management scenario.

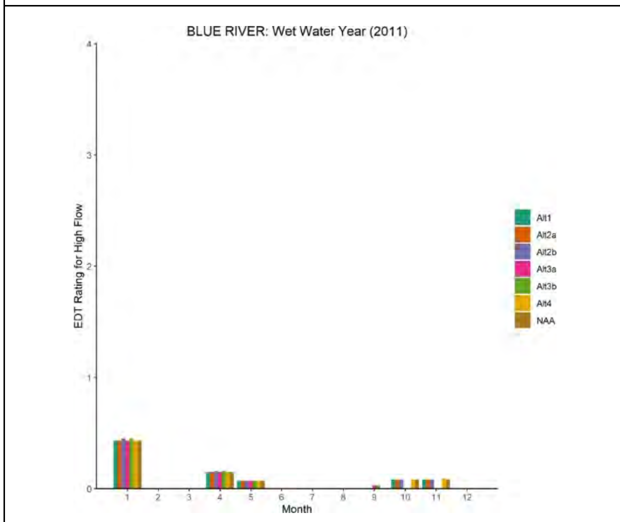
REFERENCES

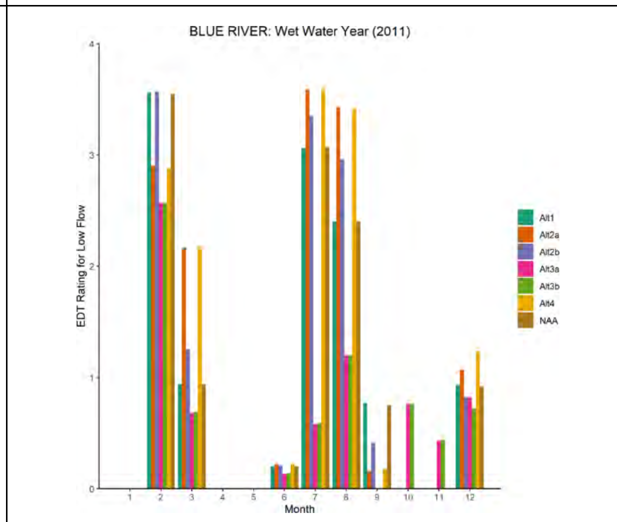
Lestelle, L.C. 2005. Guidelines for rating level 2 environmental attributes in Ecosystem Diagnosis and Treatment (EDT). Jones & Stokes Associates, Inc.

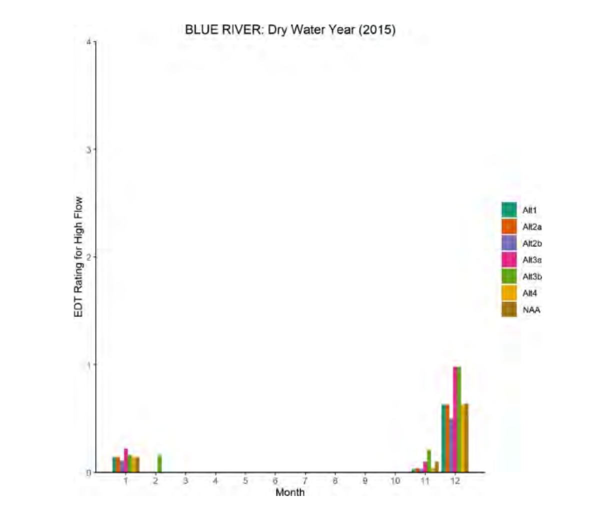
Chapter 6 Attachment C-1: Graphs of EDT Flow Ratings in McKenzie Watershed

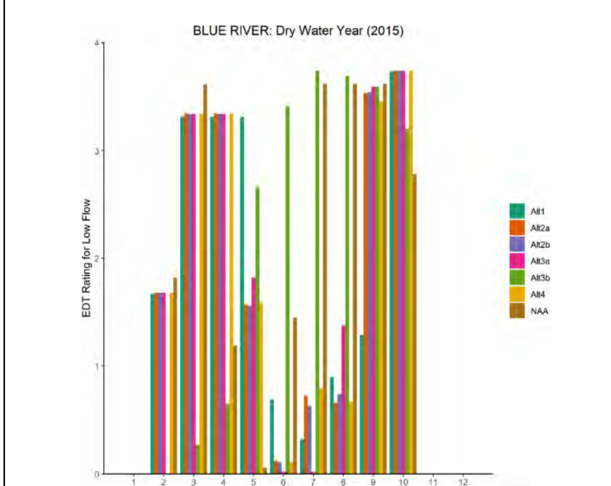




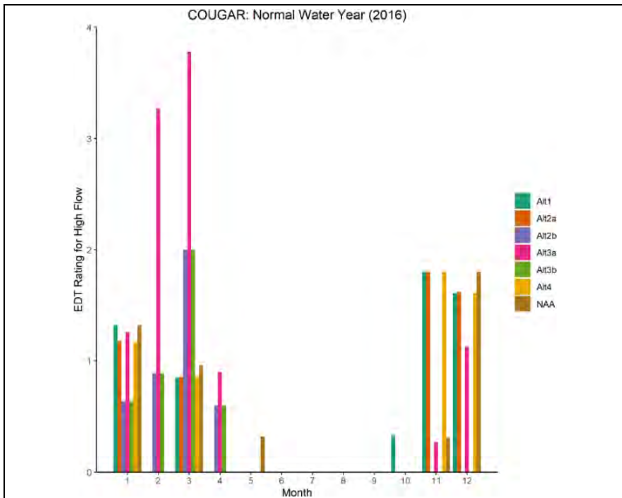


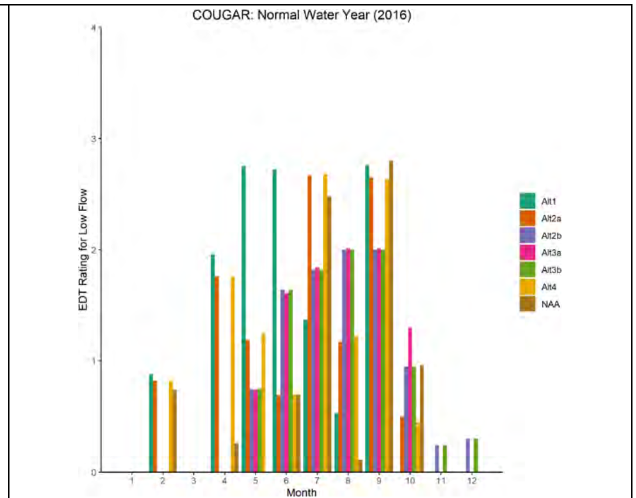


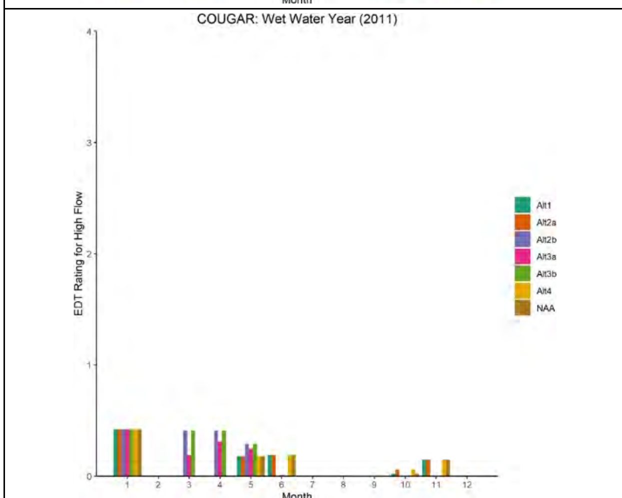


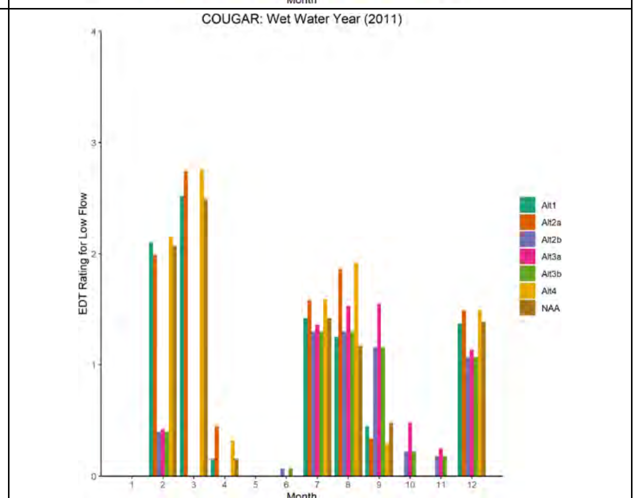


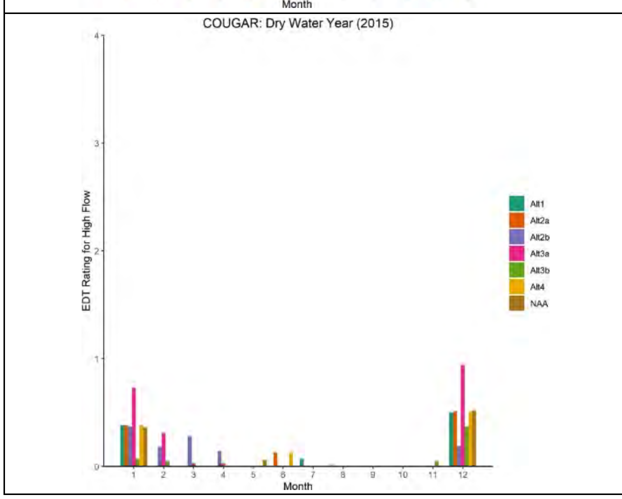
Chapter 6 Attachment C-1: Graphs of EDT Flow Ratings in McKenzie Watershed

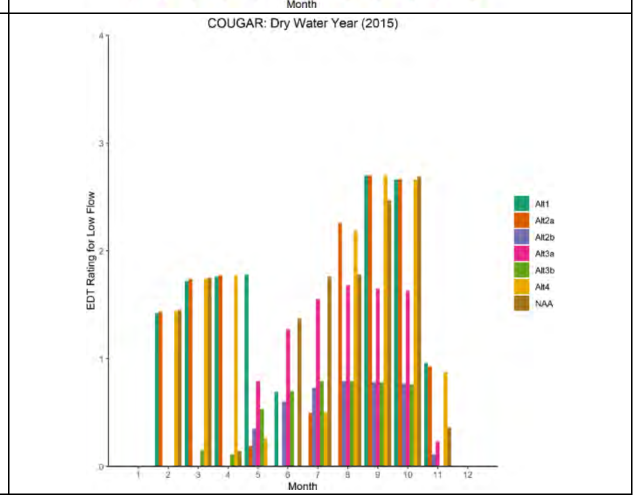




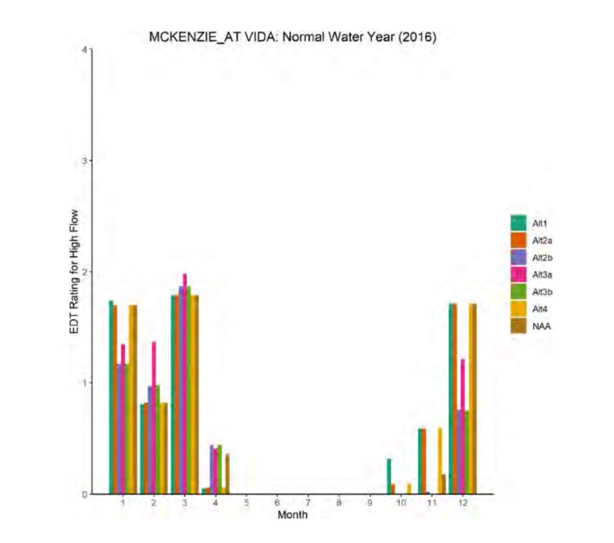


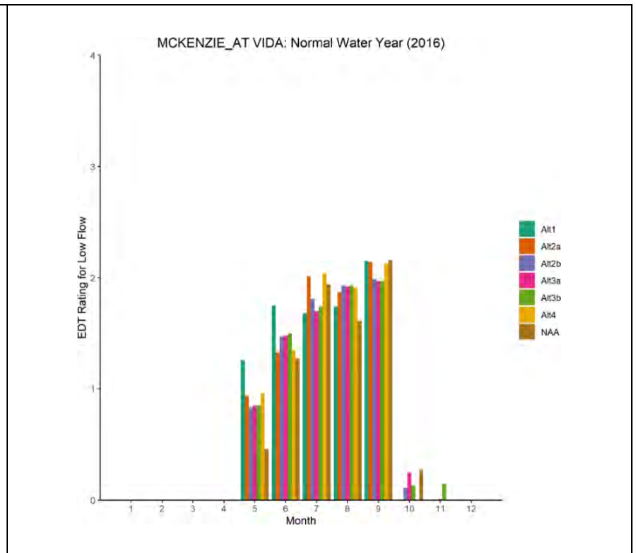


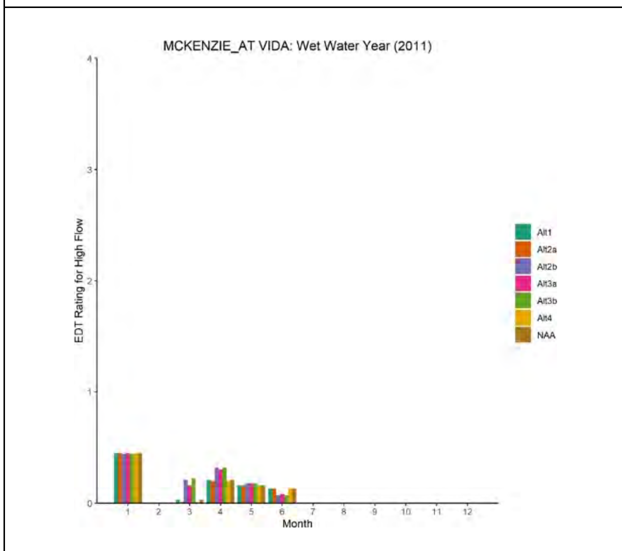


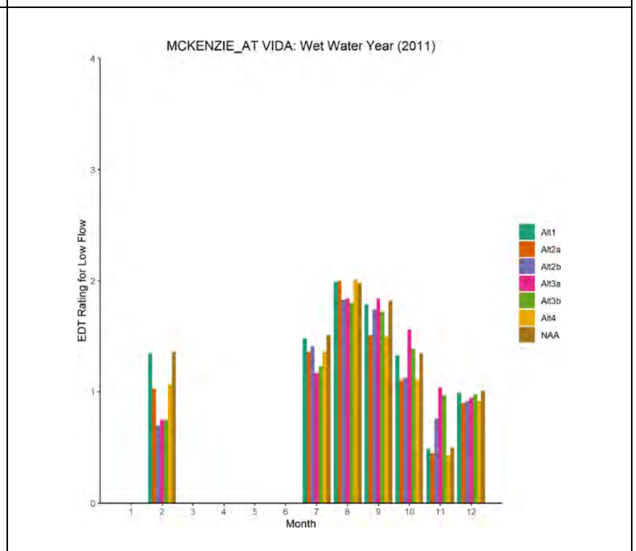


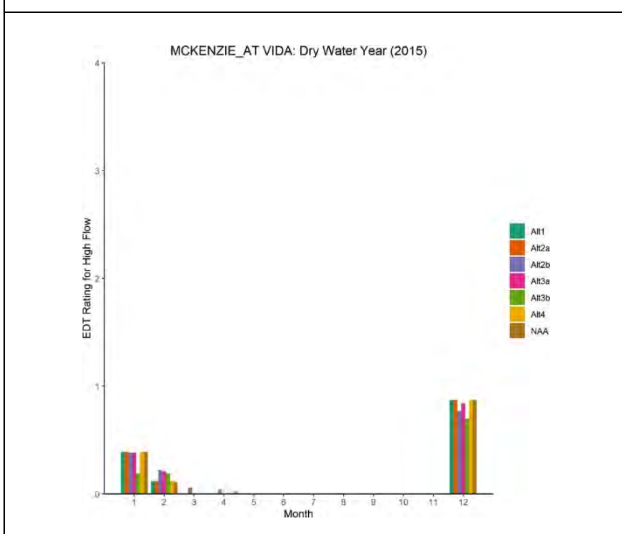
Chapter 6 Attachment C-1: Graphs of EDT Flow Ratings in McKenzie Watershed

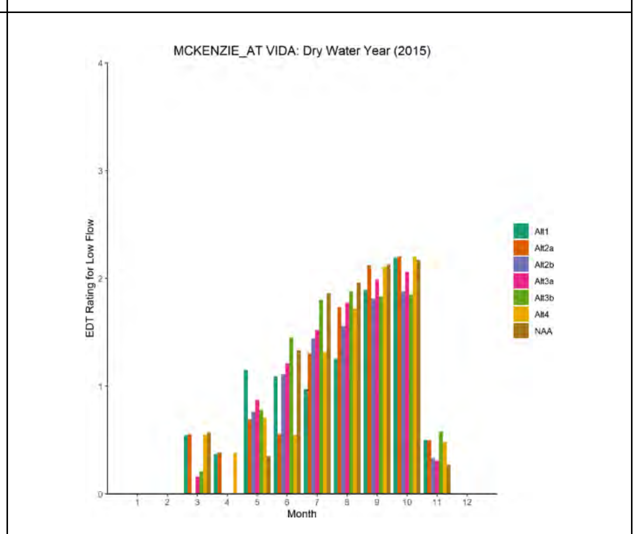




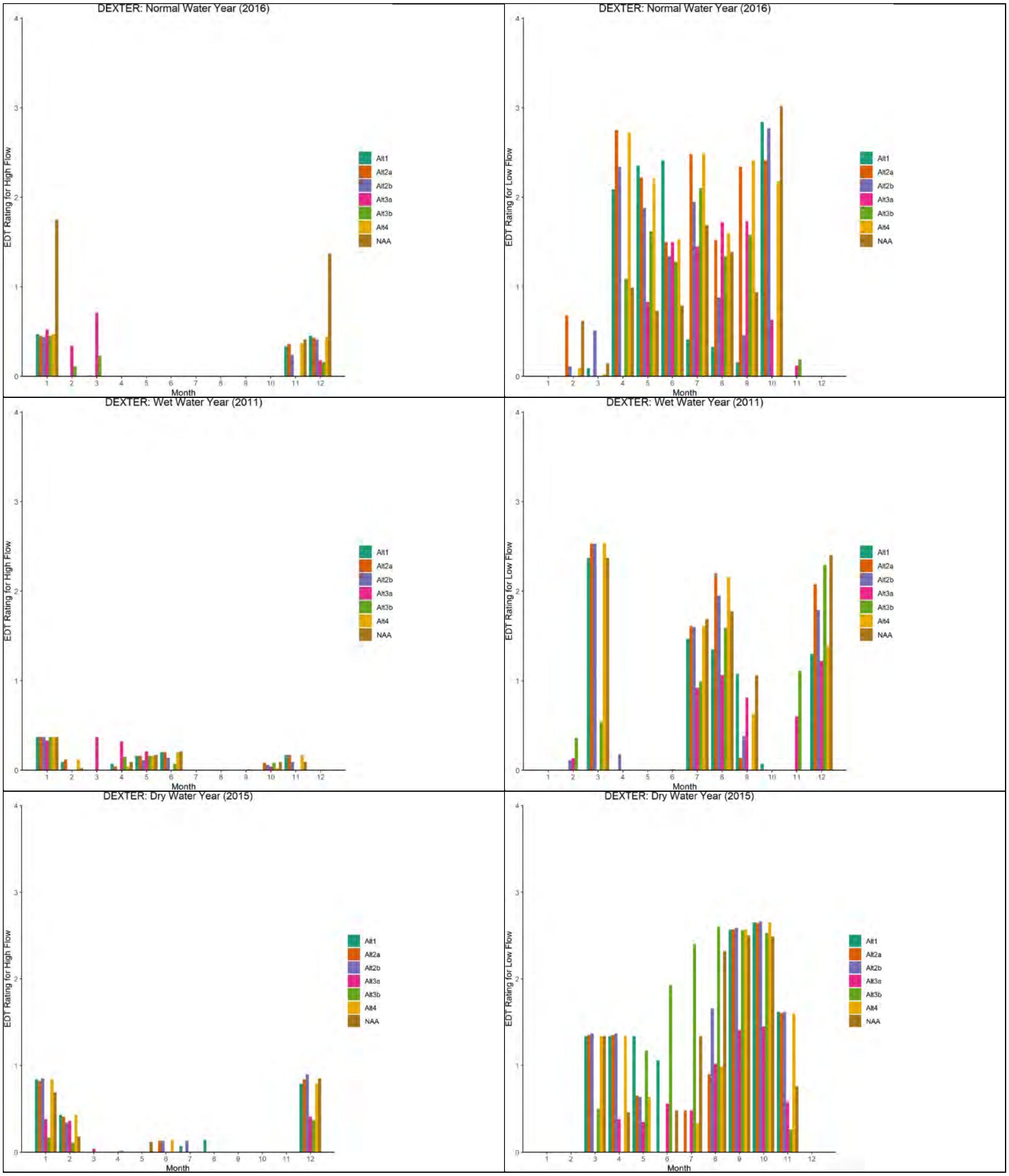




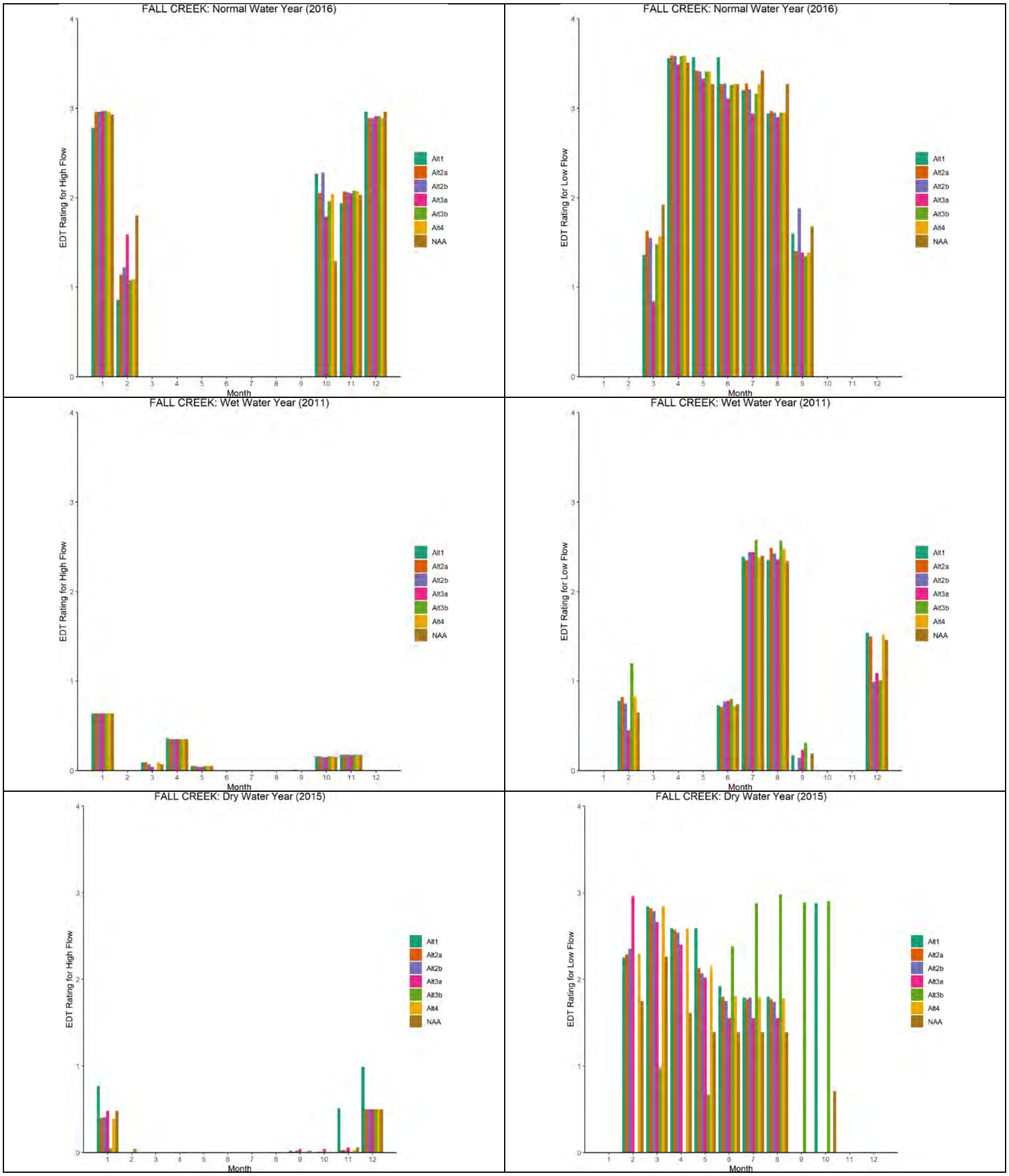




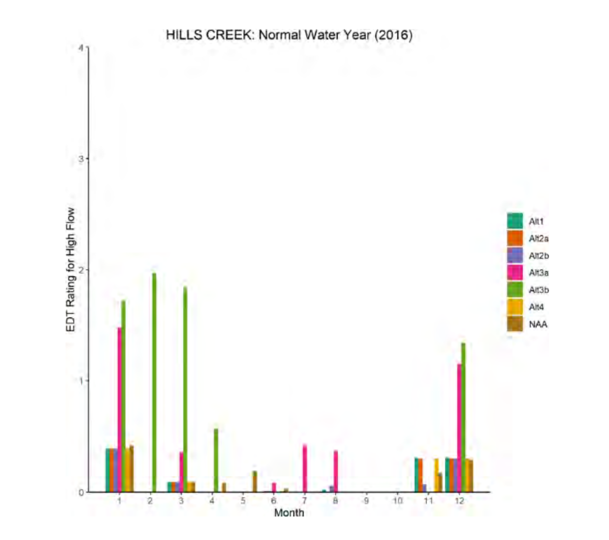
Chapter 6 Attachment C-2: Graphs of EDT Flow Ratings in Middle Fork Willamette Watershed

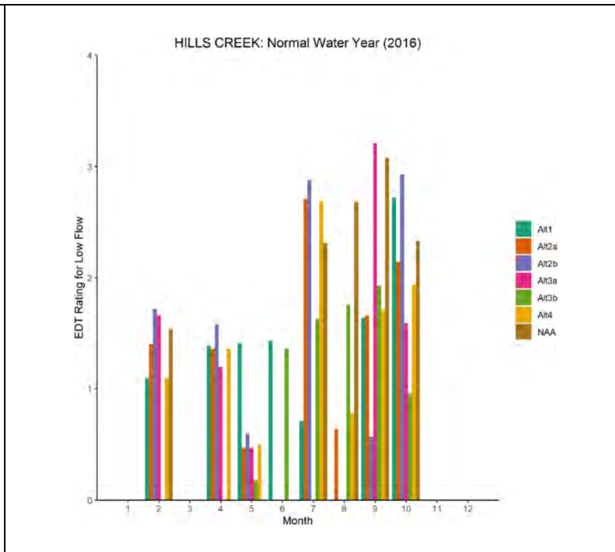


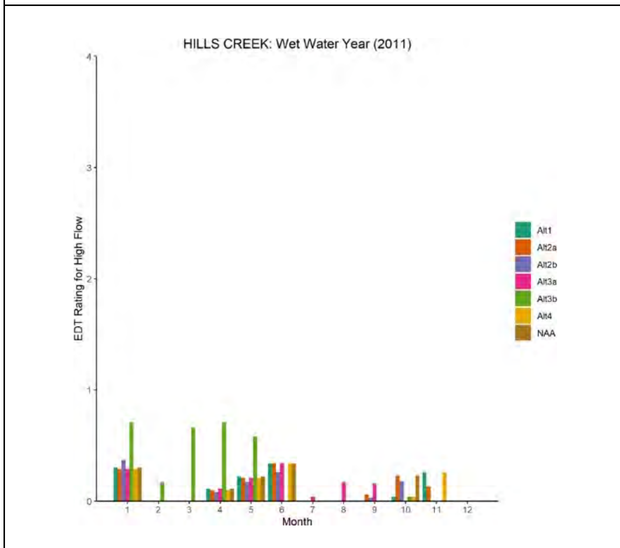
Chapter 6 Attachment C-2: Graphs of EDT Flow Ratings in Middle Fork Willamette Watershed

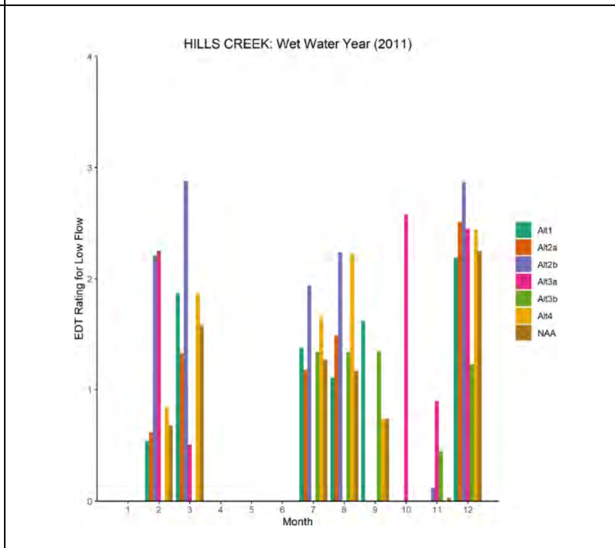


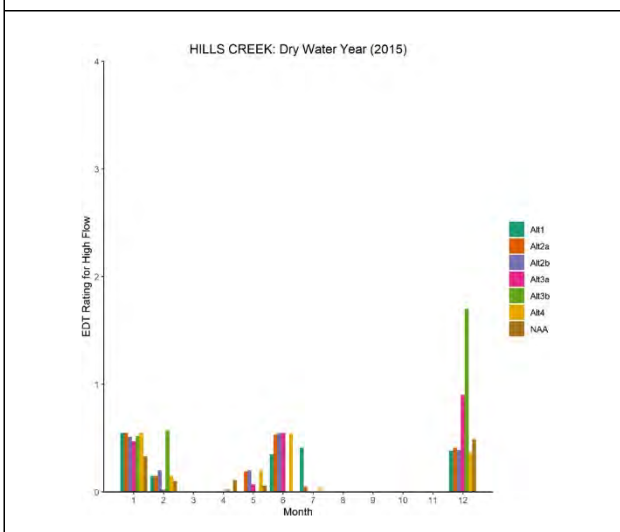
Chapter 6 Attachment C-2: Graphs of EDT Flow Ratings in Middle Fork Willamette Watershed

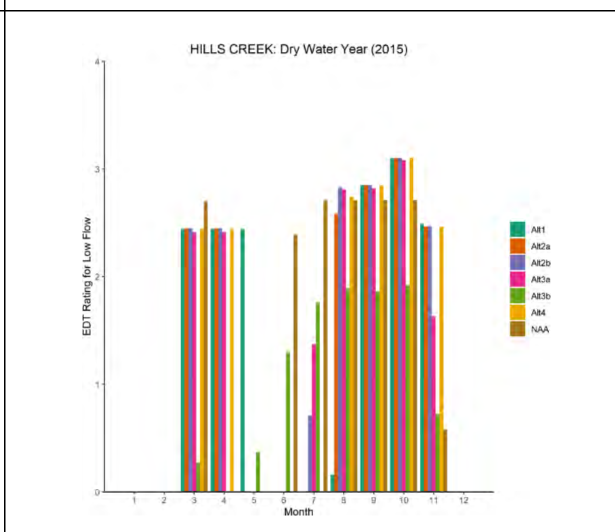




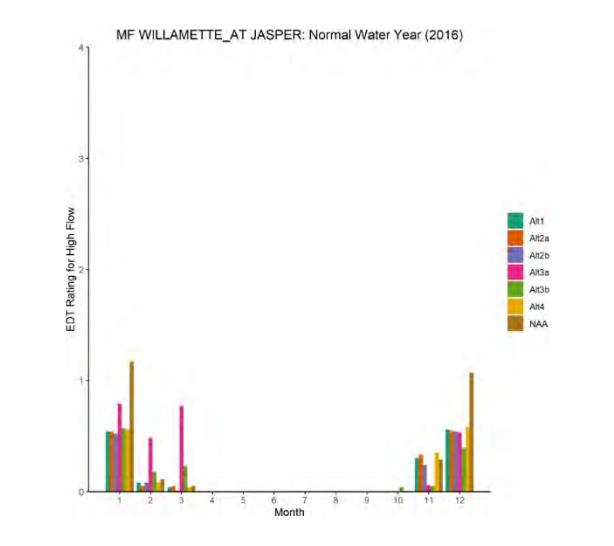


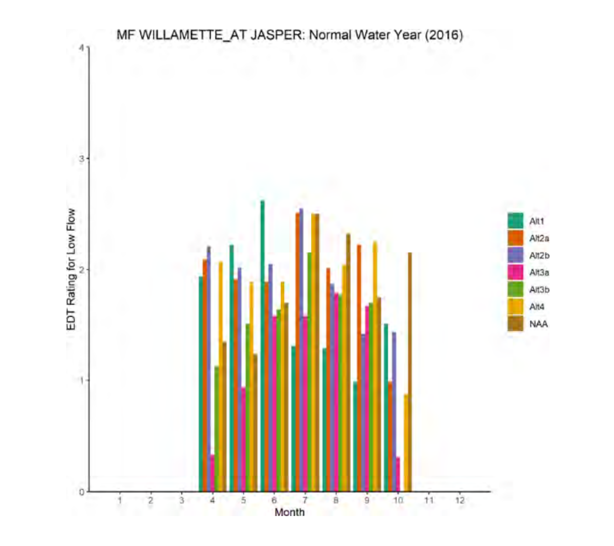


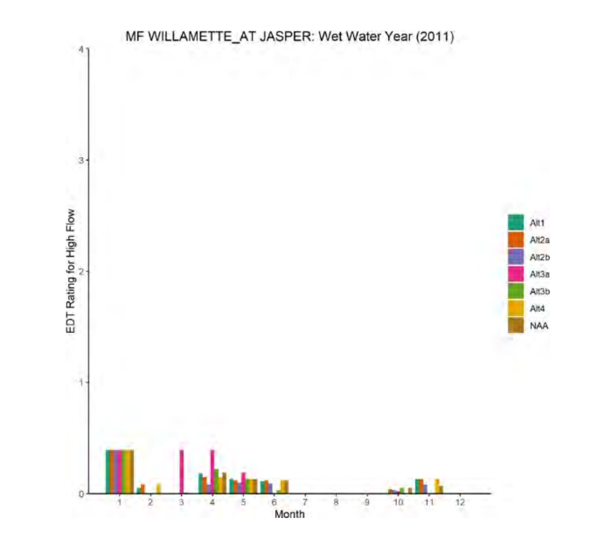


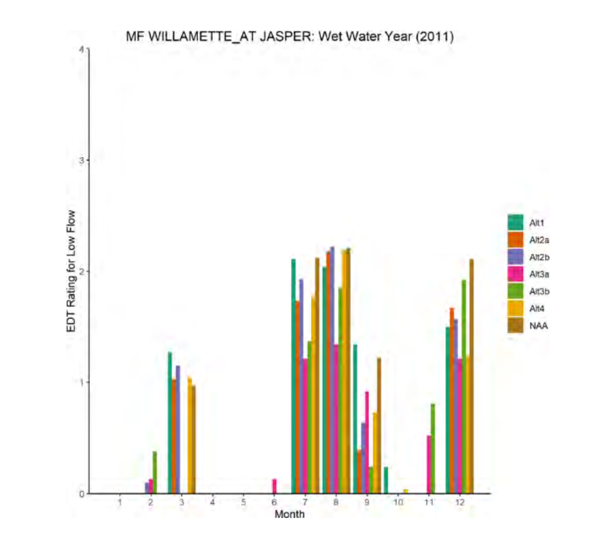


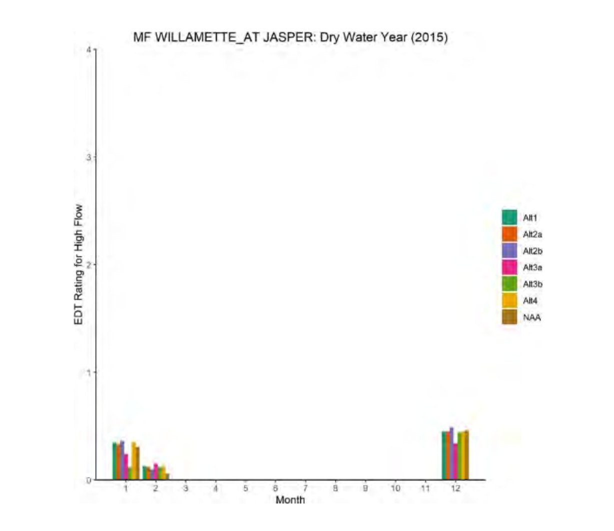
Chapter 6 Attachment C-2: Graphs of EDT Flow Ratings in Middle Fork Willamette Watershed

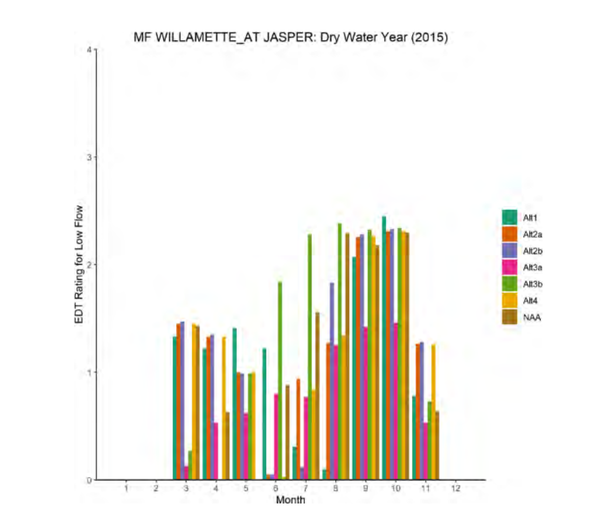




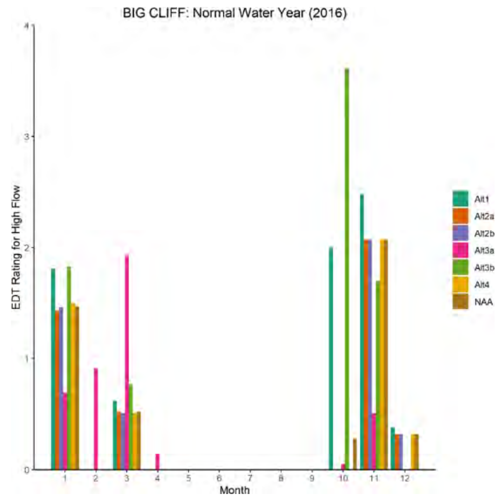


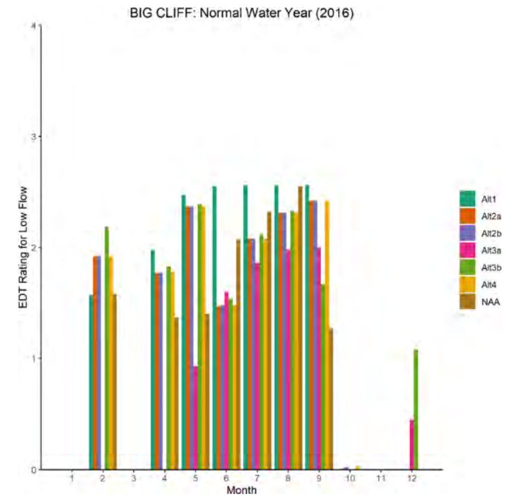


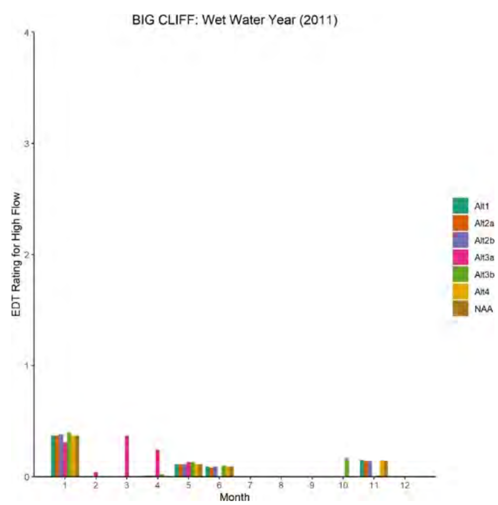


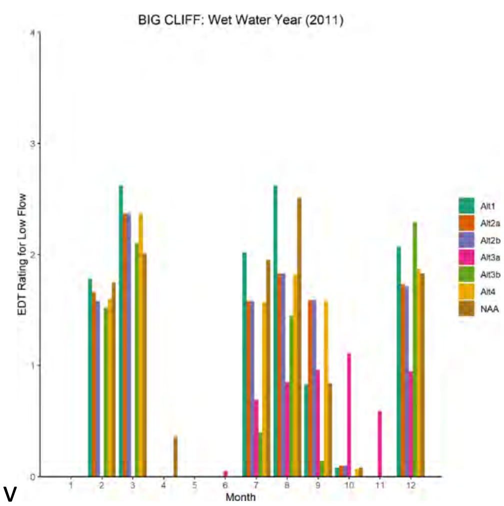


Chapter 6 Attachment C-2: Graphs of EDT Flow Ratings in Middle Fork Willamette Watershed









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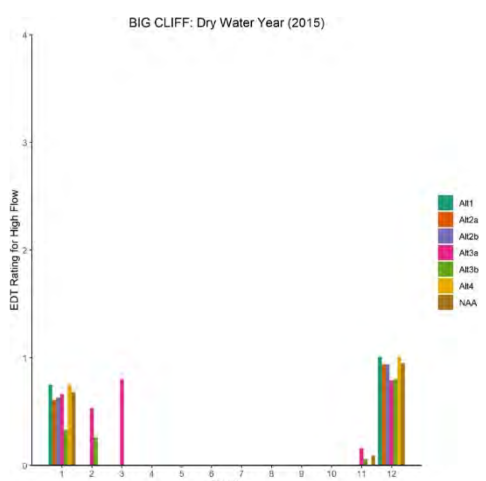
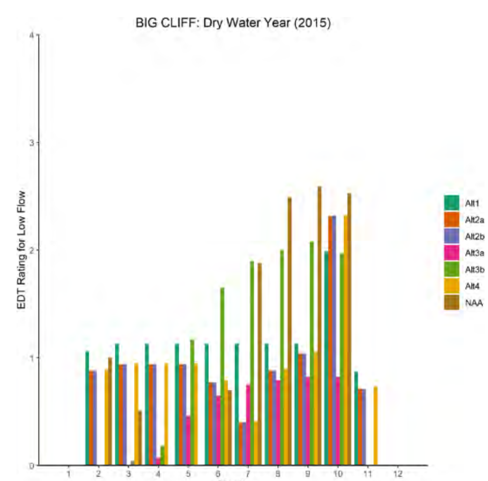
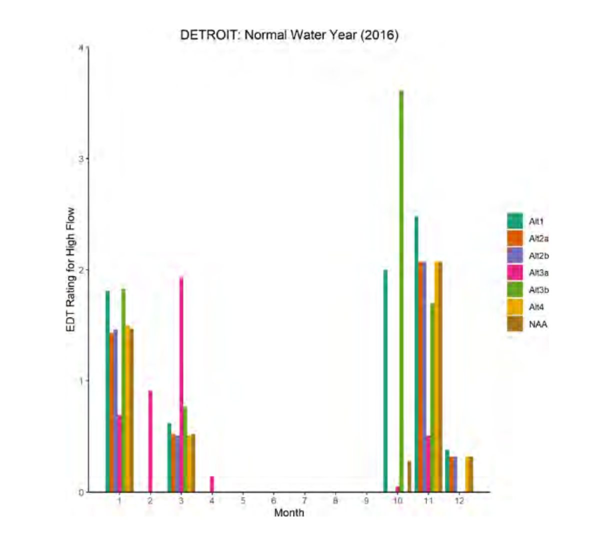
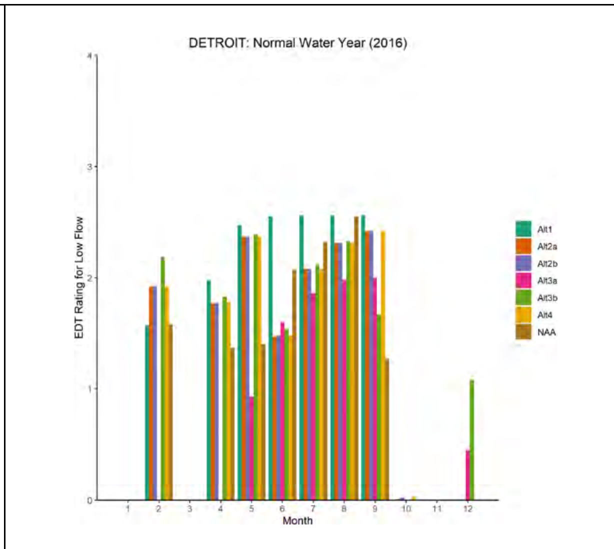
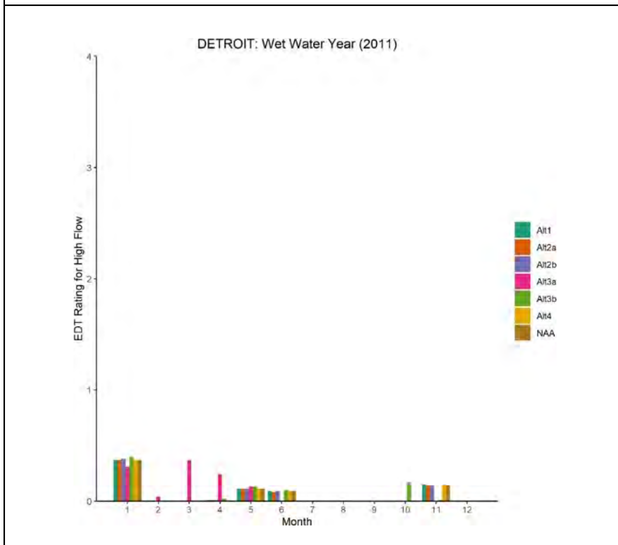


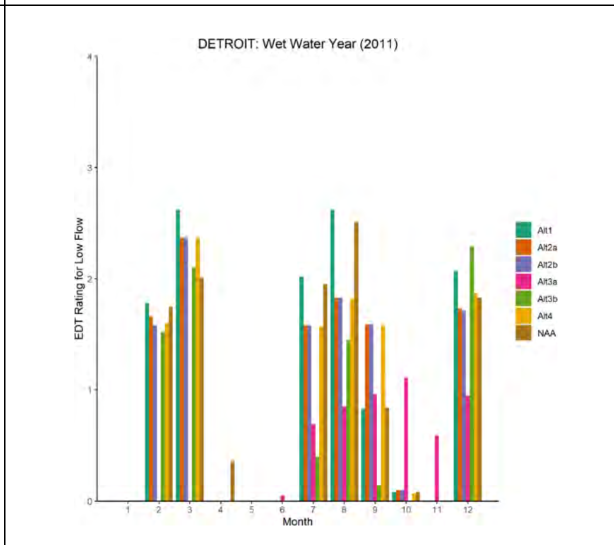


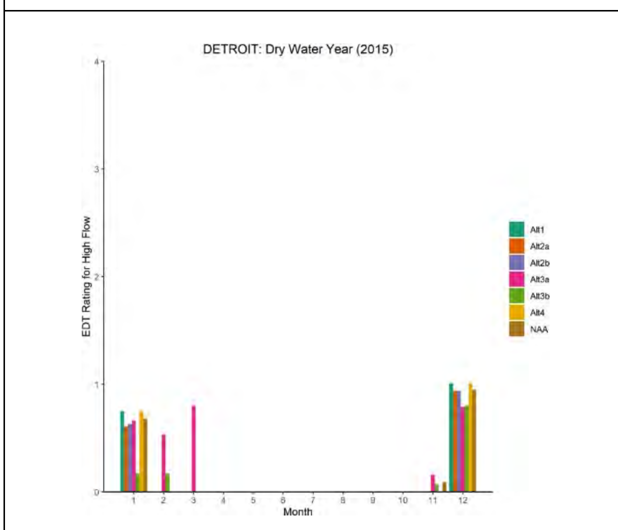
Chapter 6 Attachment C-3: Graphs of EDT Flow Ratings in North Santiam Watershed

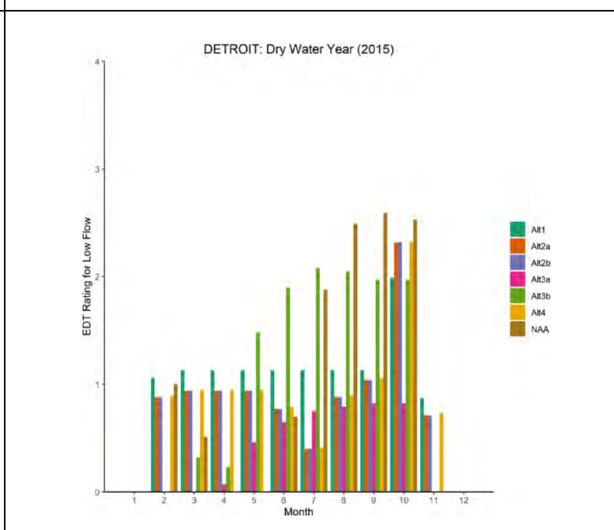




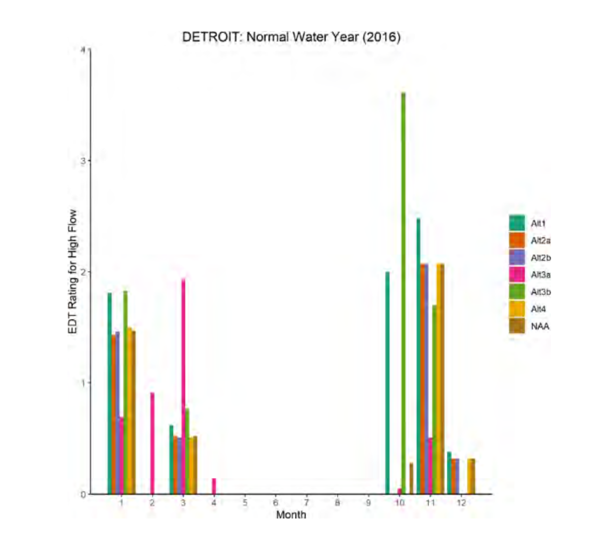


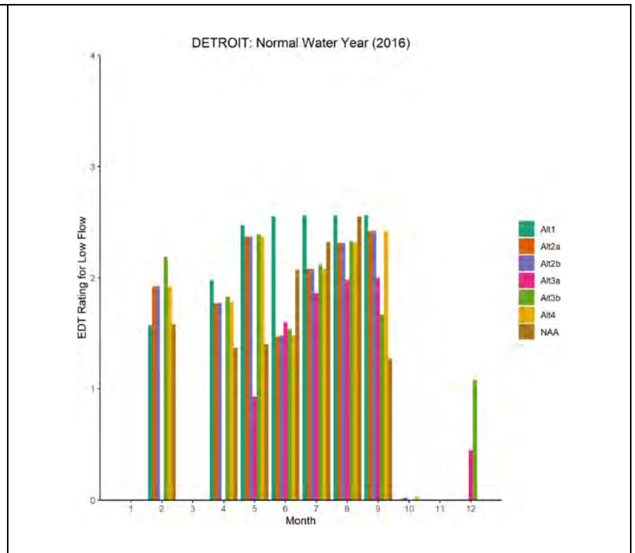


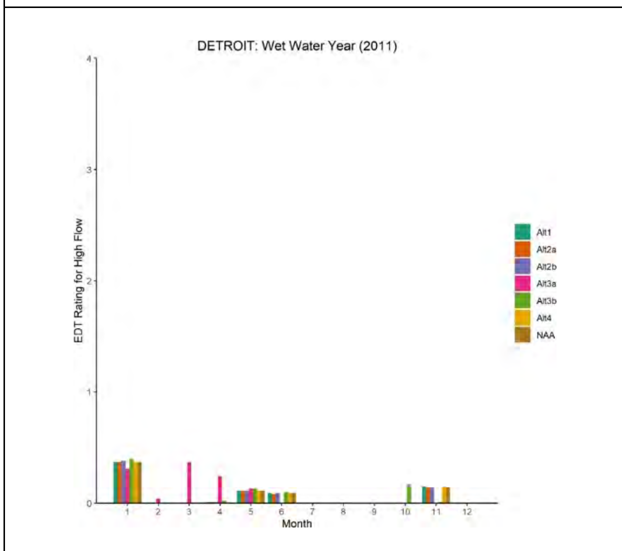


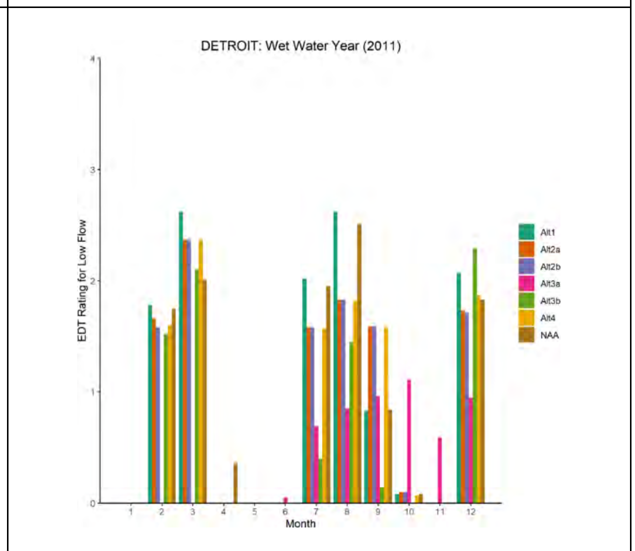


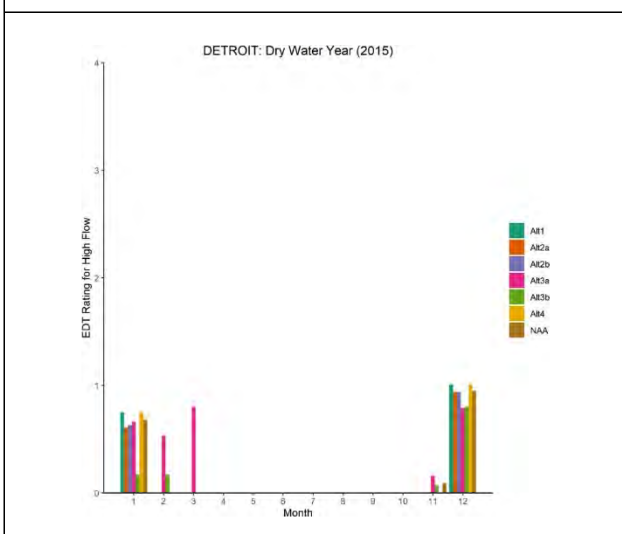
Chapter 6 Attachment C-3: Graphs of EDT Flow Ratings in North Santiam Watershed

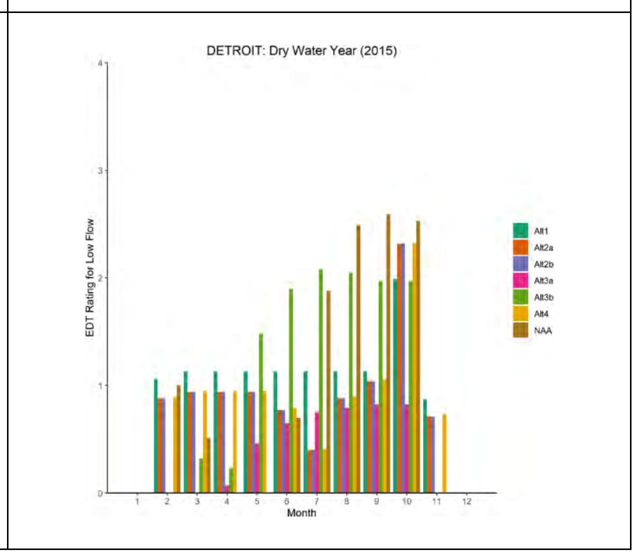




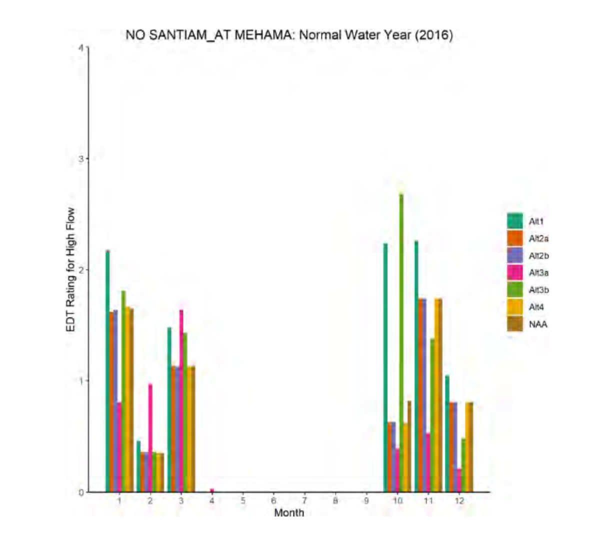


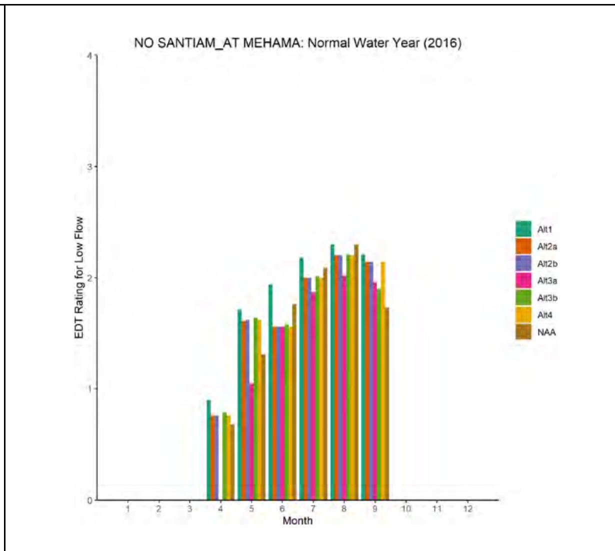


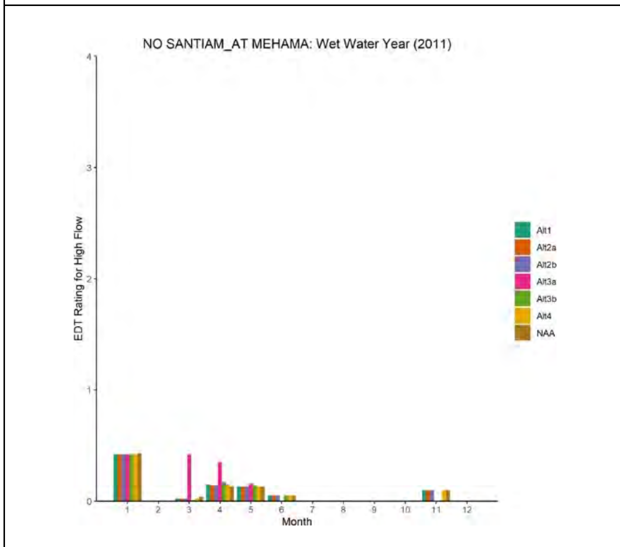


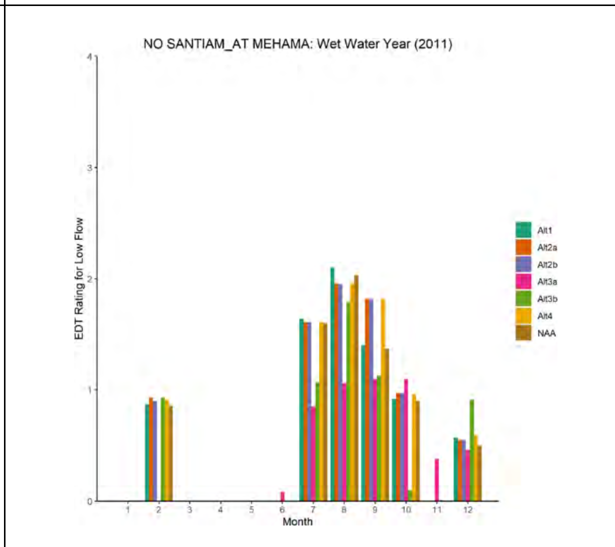


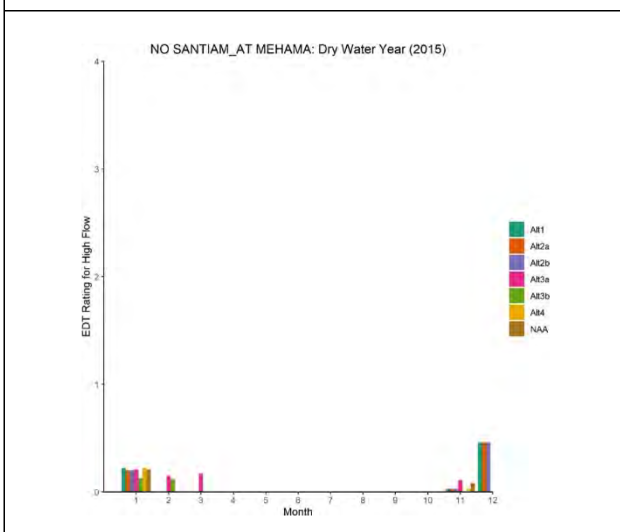
Chapter 6 Attachment C-3: Graphs of EDT Flow Ratings in North Santiam Watershed

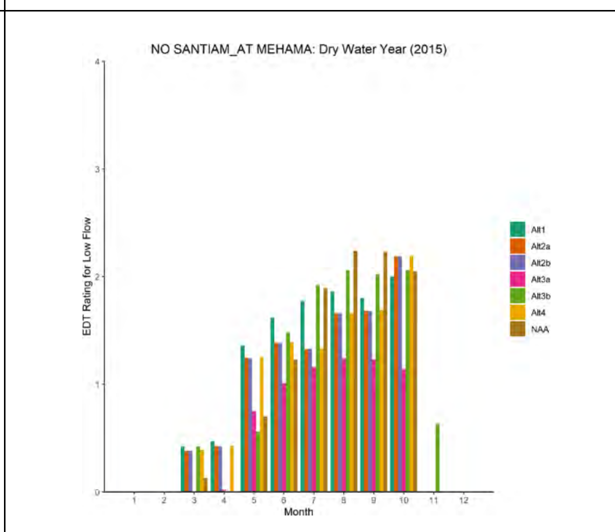




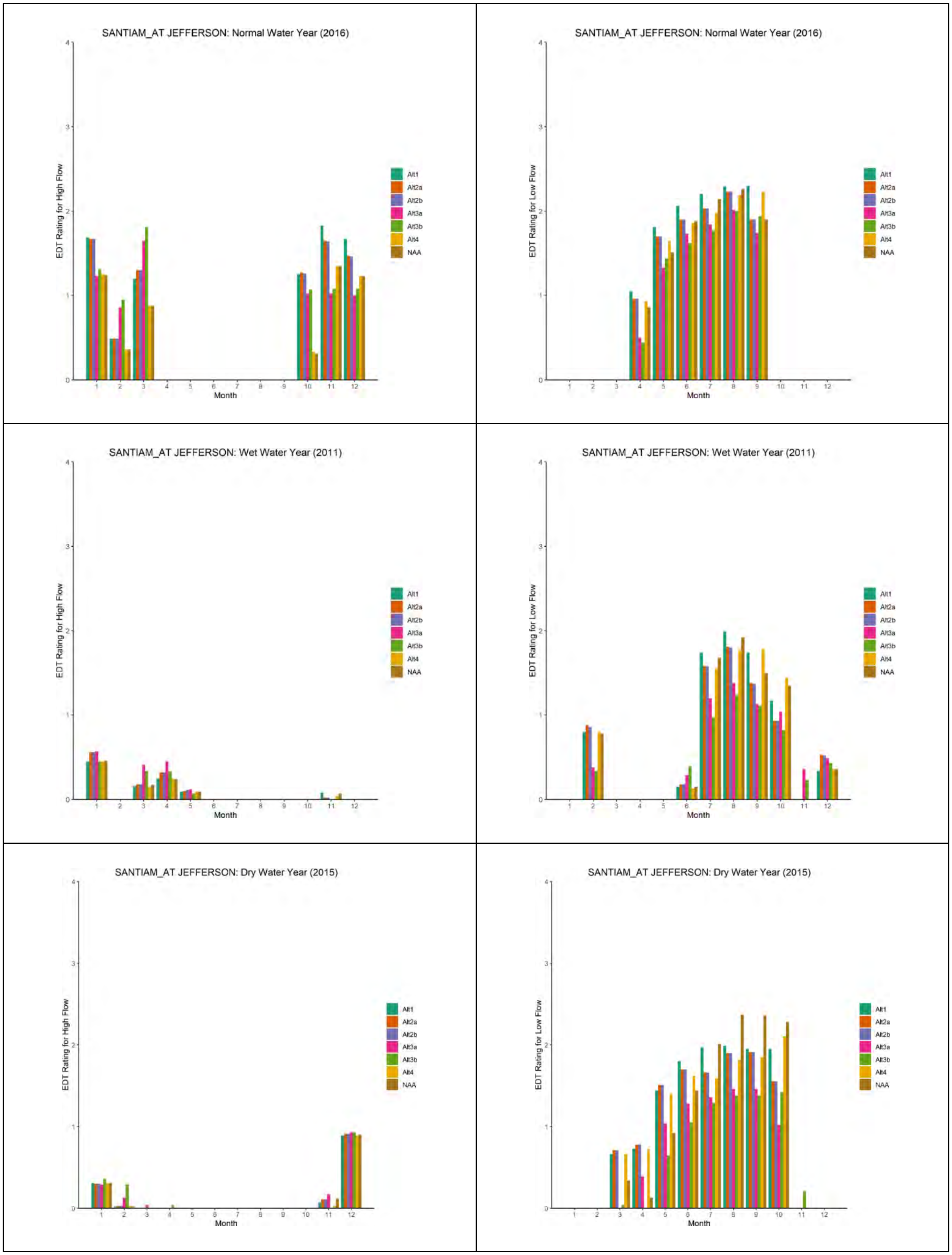




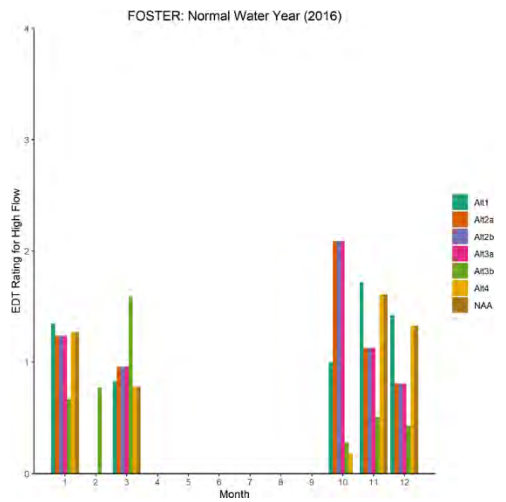


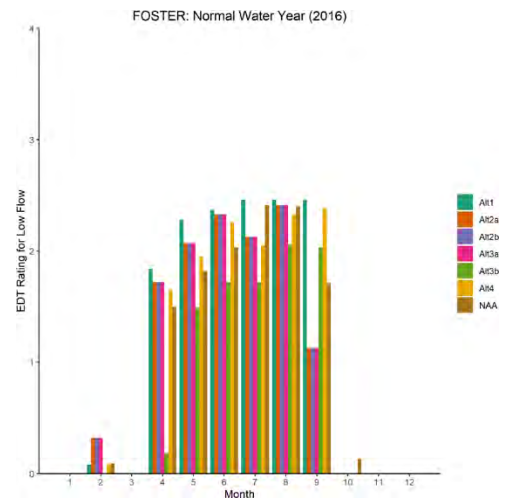


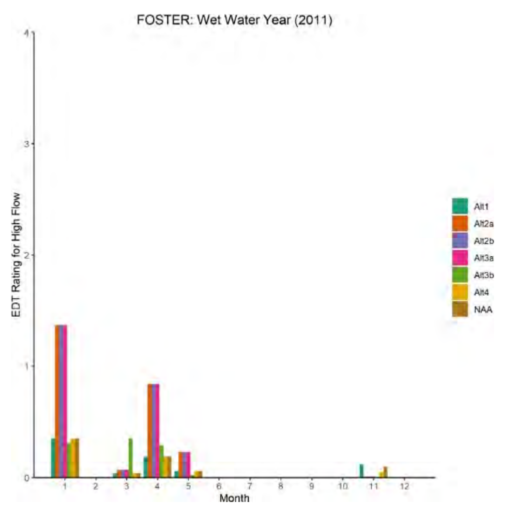
Chapter 6 Attachment C-3: Graphs of EDT Flow Ratings in North Santiam Watershed

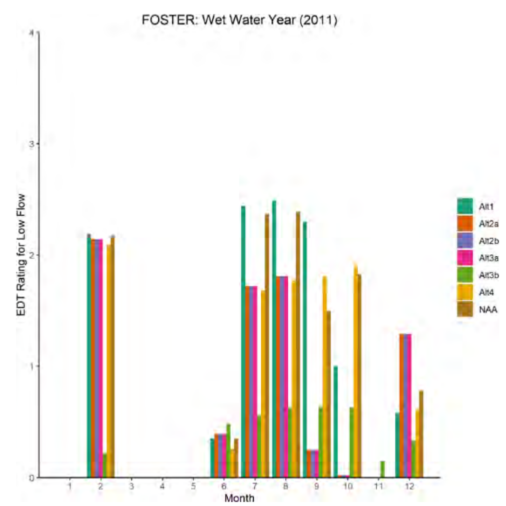


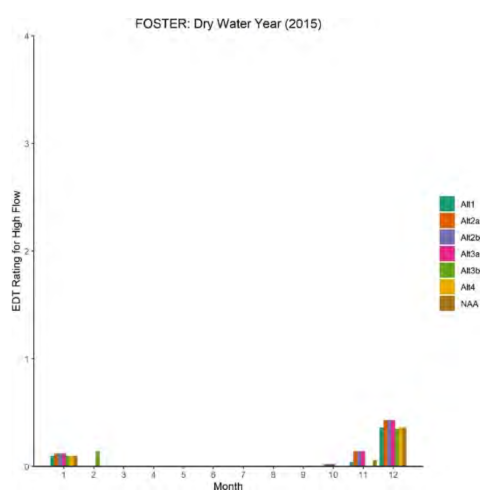
Chapter 6 Attachment C-4: Graphs of EDT Flow Ratings in South Santiam Watershed

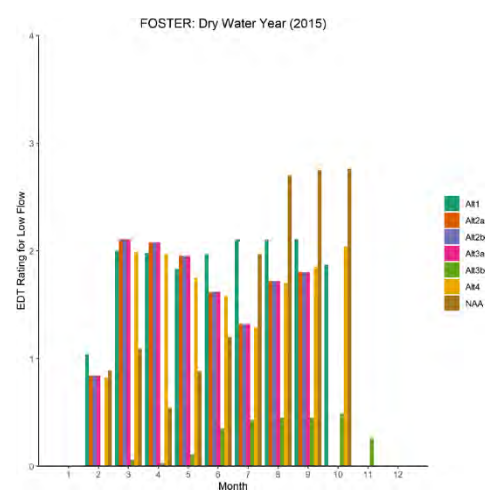




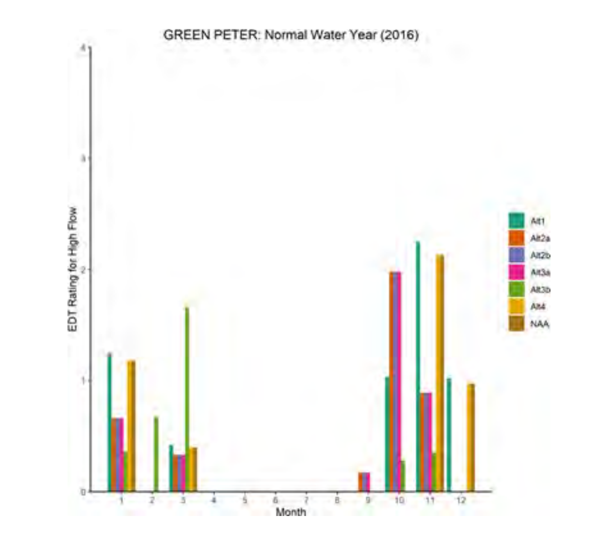


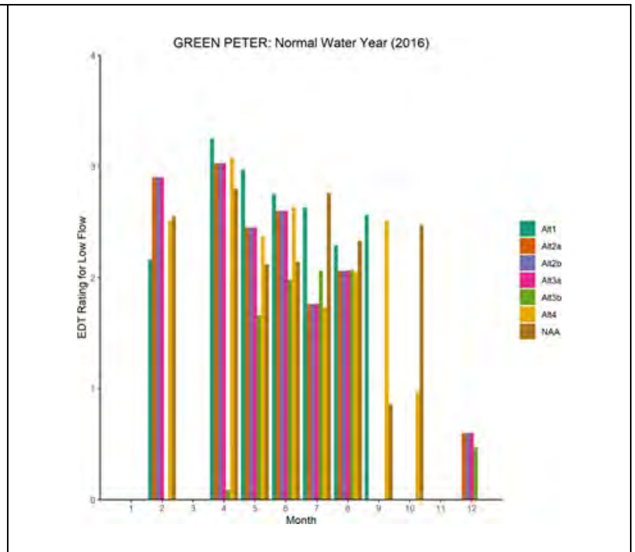


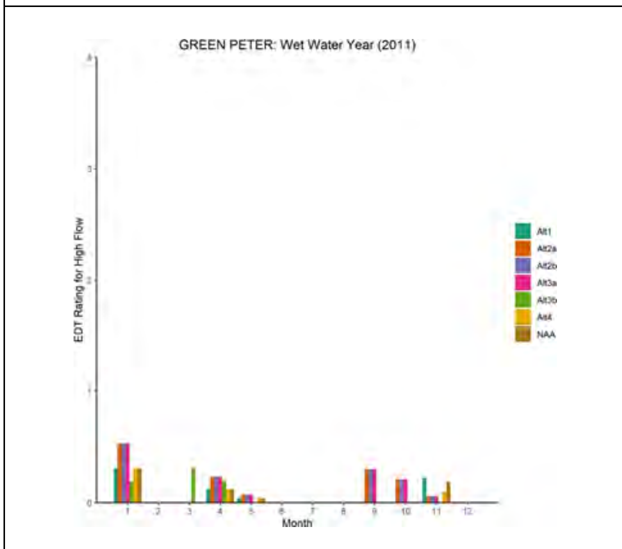


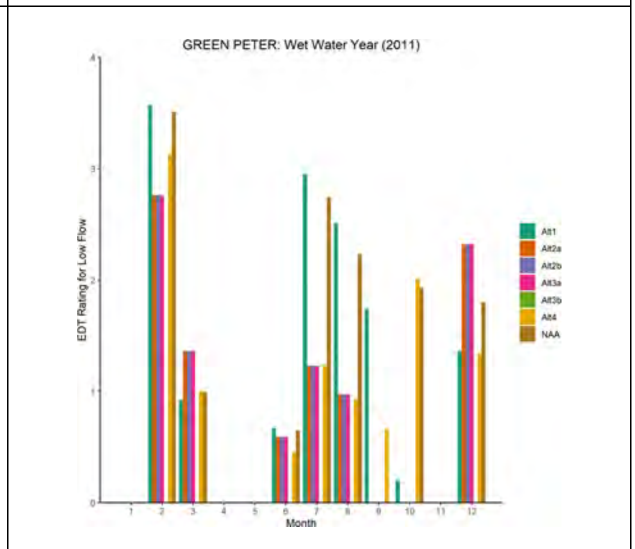


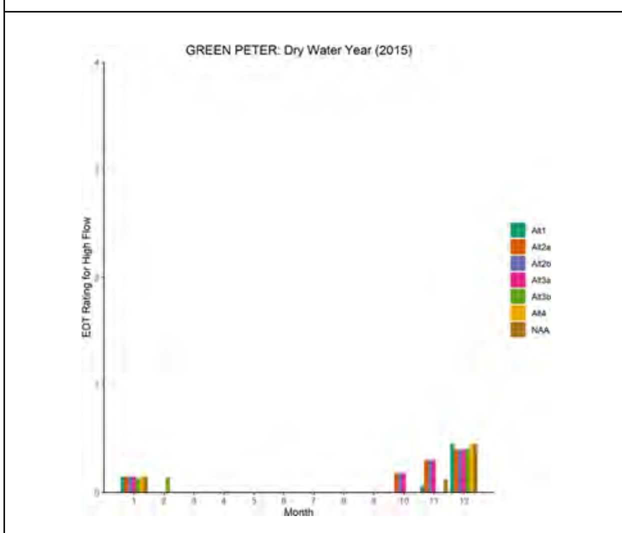
Chapter 6 Attachment C-4: Graphs of EDT Flow Ratings in South Santiam Watershed

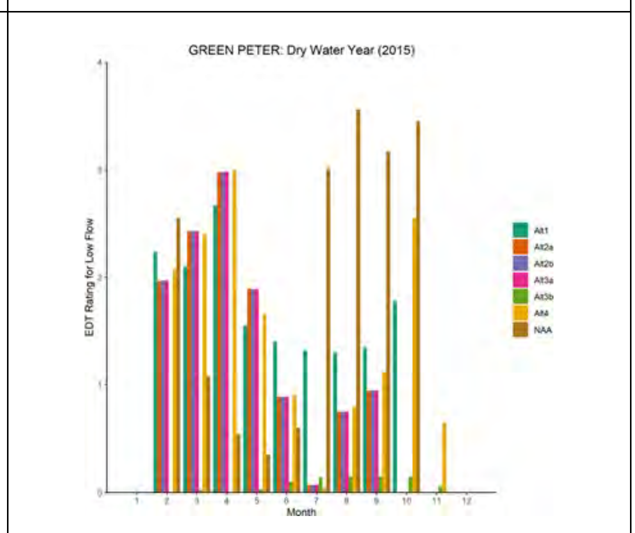




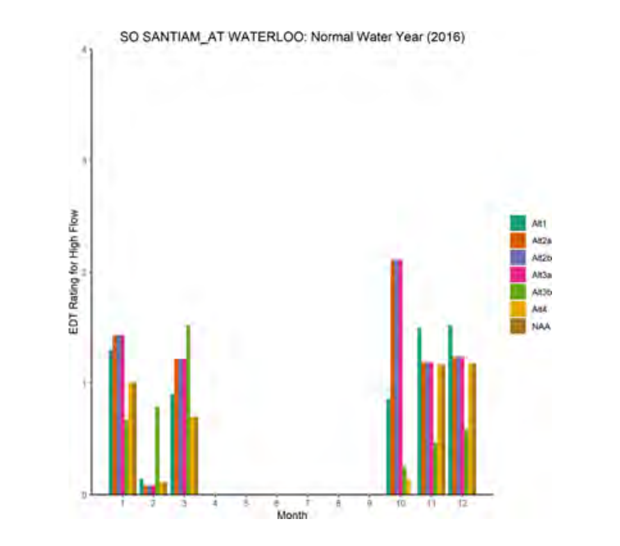


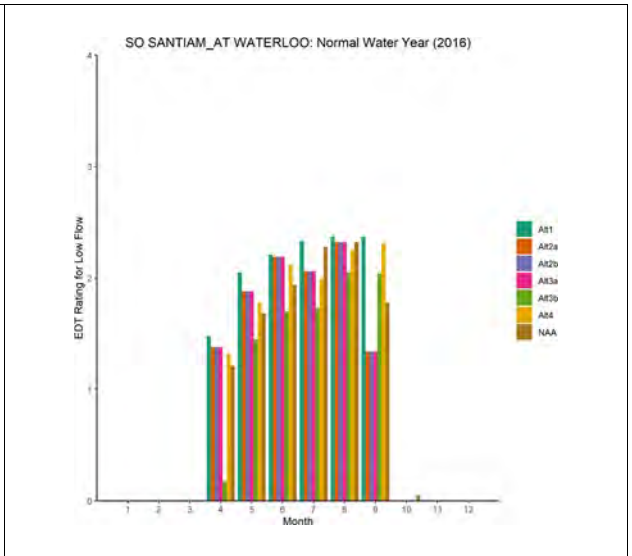


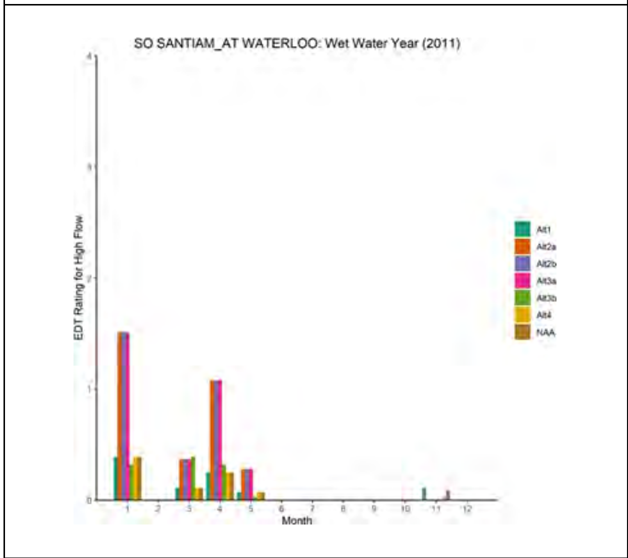


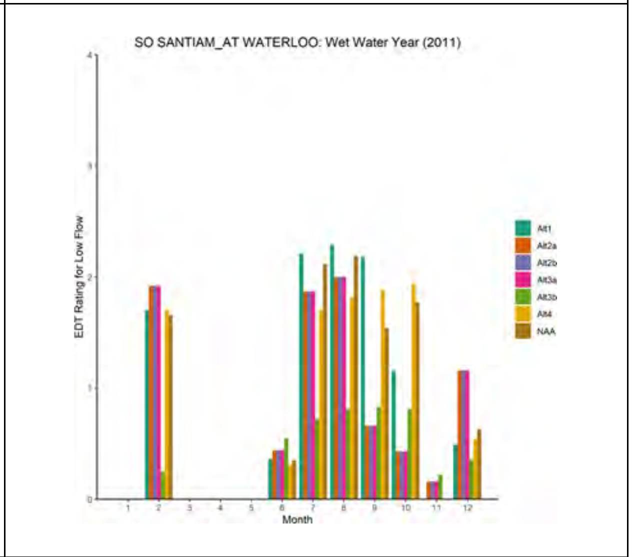


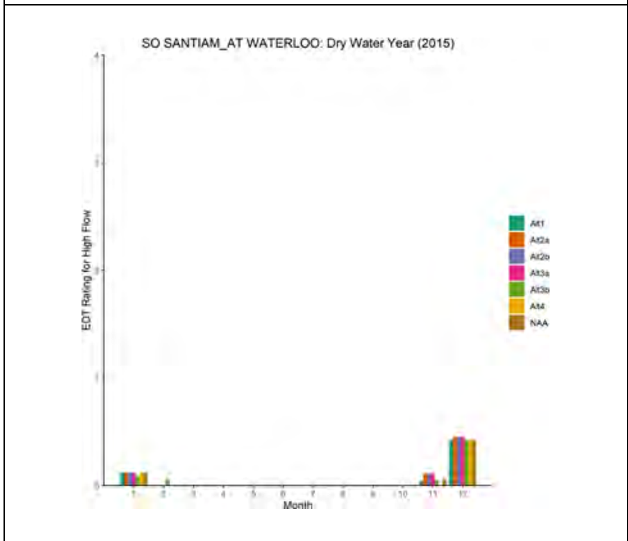
Chapter 6 Attachment C-4: Graphs of EDT Flow Ratings in South Santiam Watershed

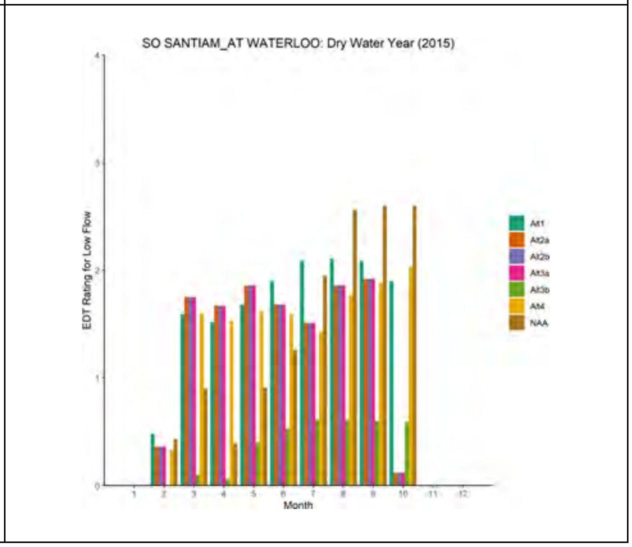




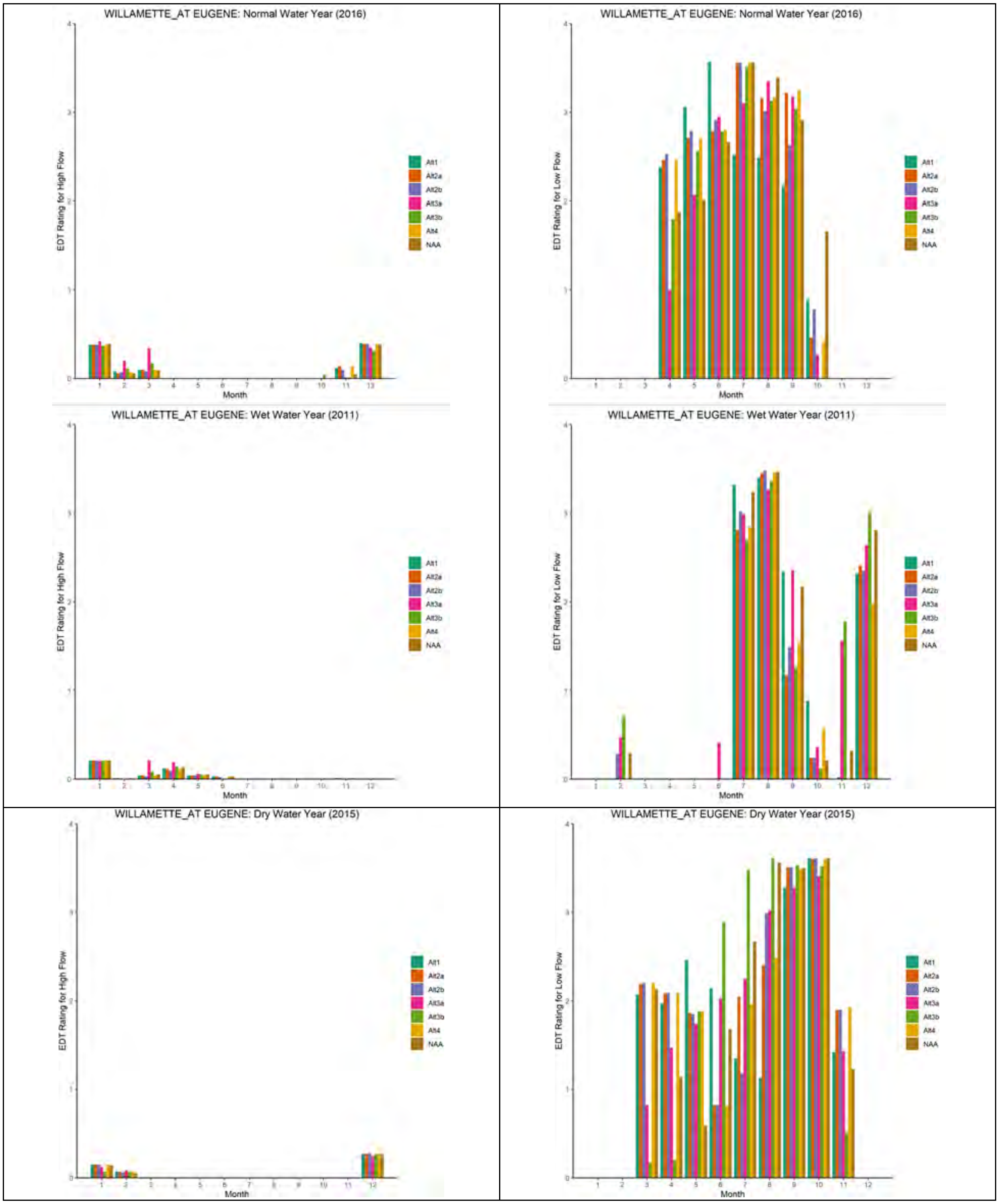






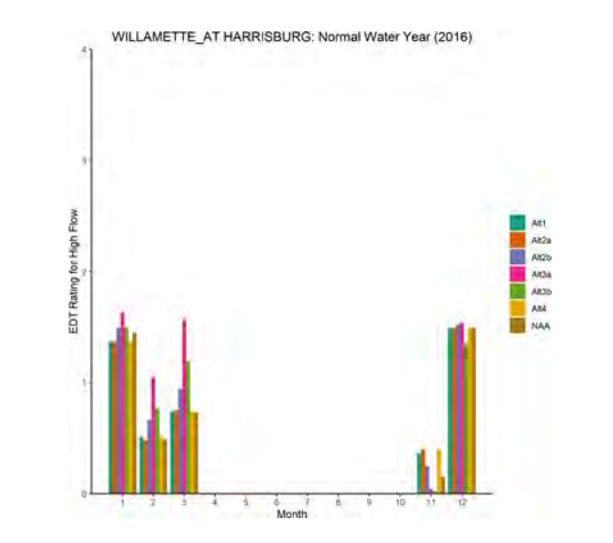


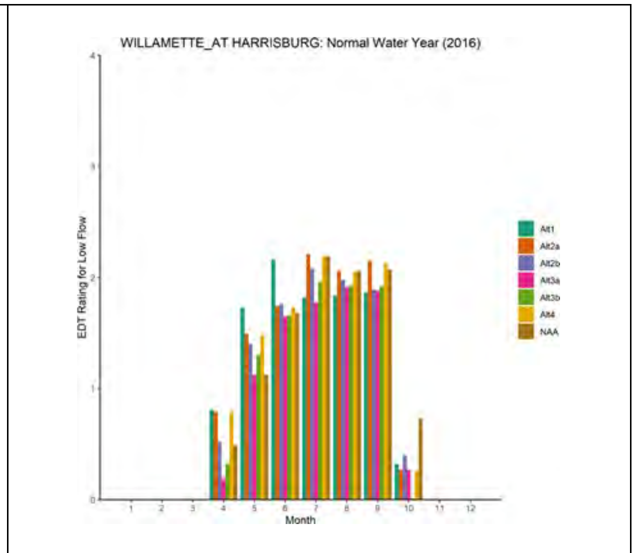
Chapter 6 Attachment C-5: Graphs of EDT Flow Ratings in Willamette River

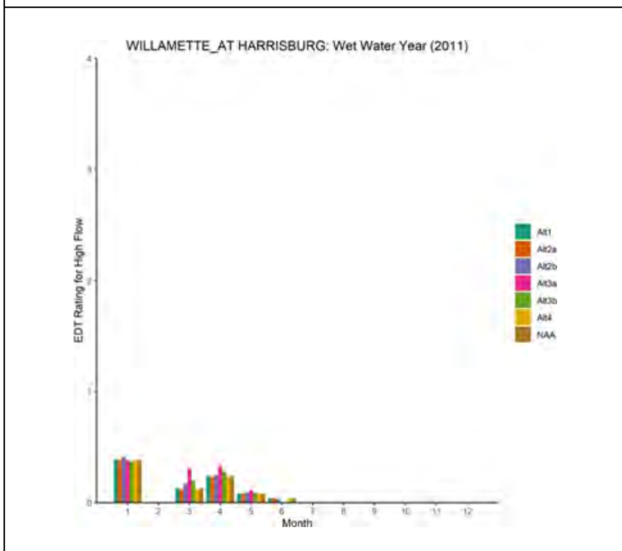


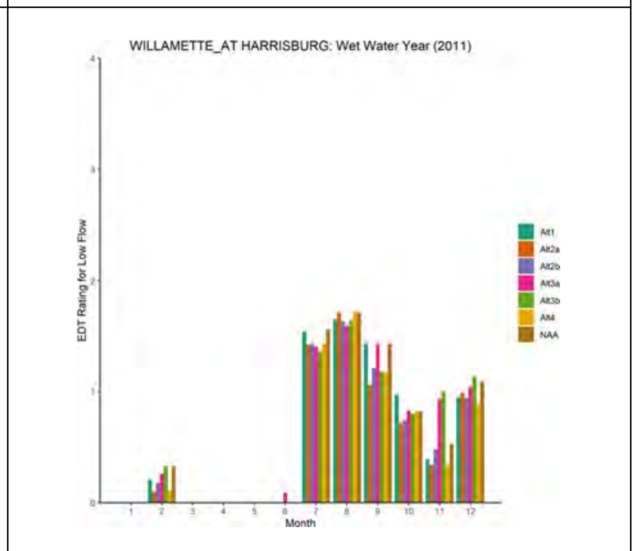
Note: Ratings are based on changes in regulated flow relative to unregulated flow (within the same water-year condition) at Harrisburg, the nearest flow data available

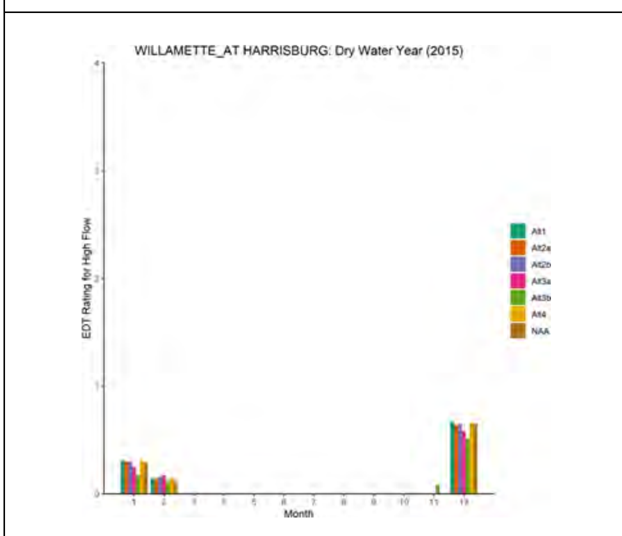
Chapter 6 Attachment C-5: Graphs of EDT Flow Ratings in Willamette River

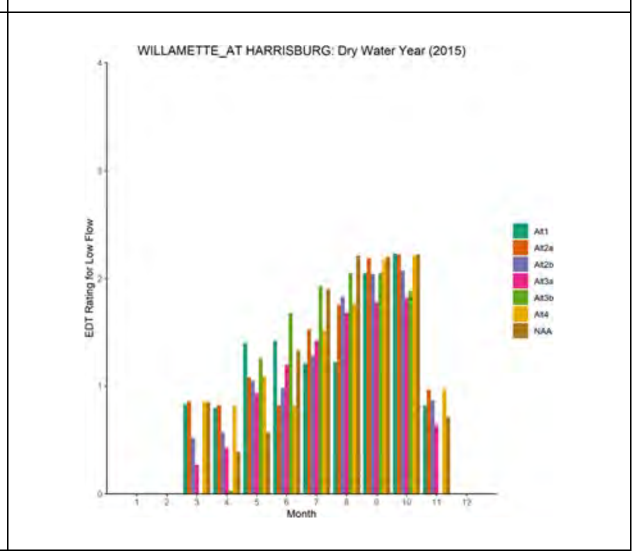




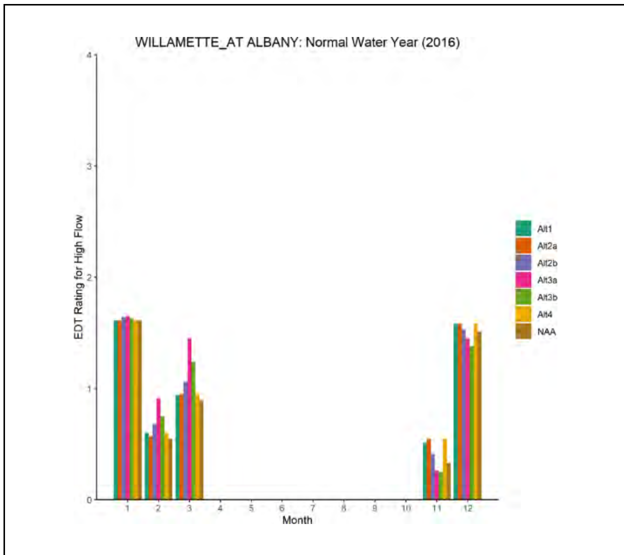


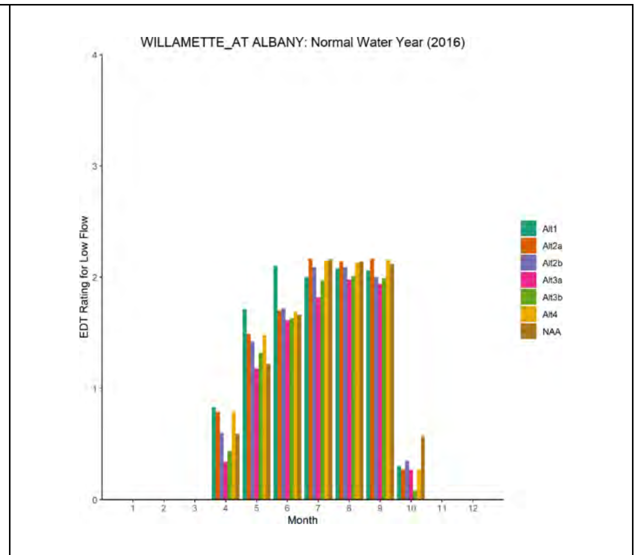


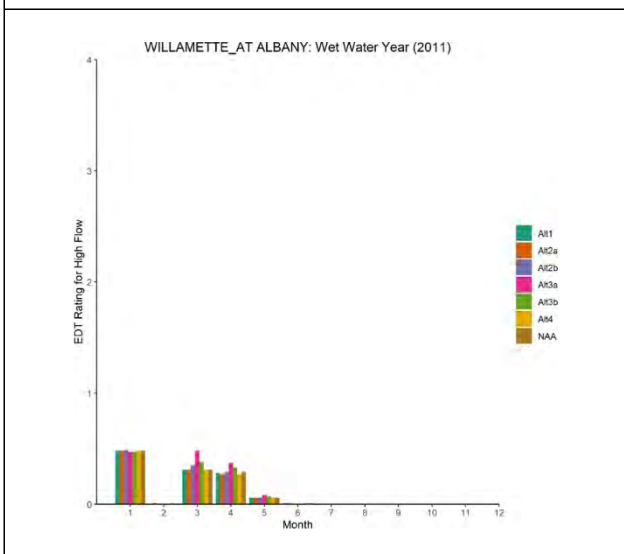


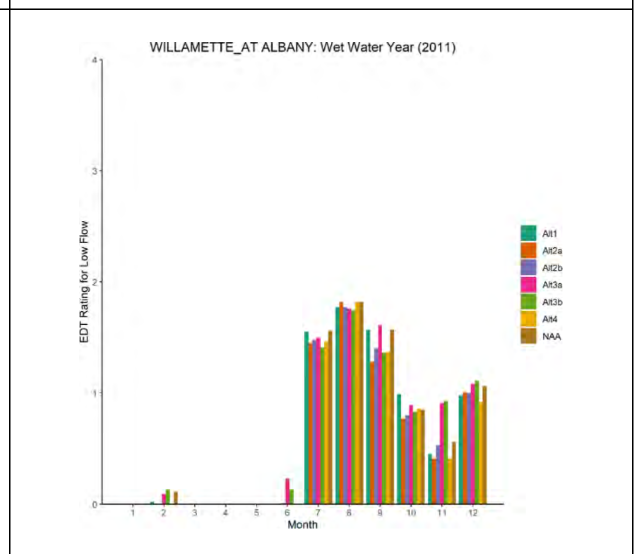


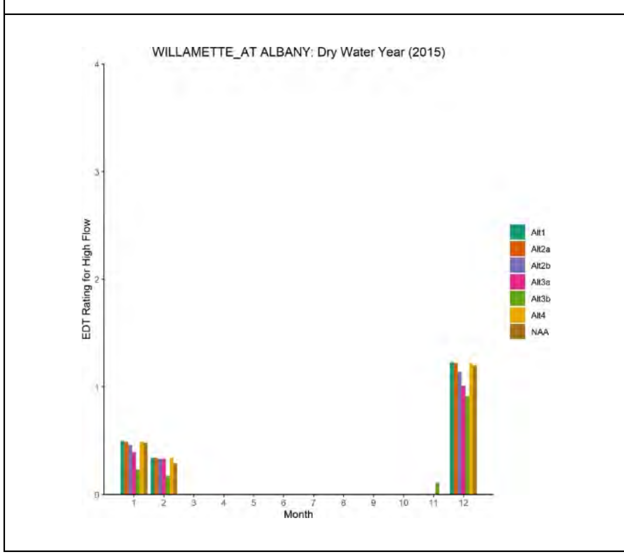
Chapter 6 Attachment C-5: Graphs of EDT Flow Ratings in Willamette River

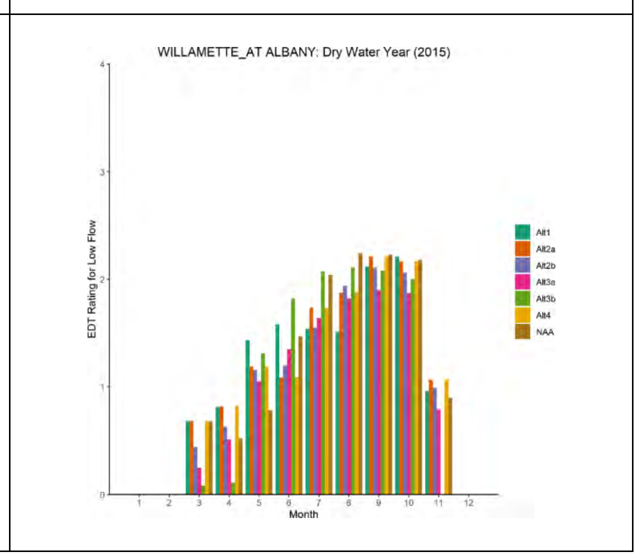




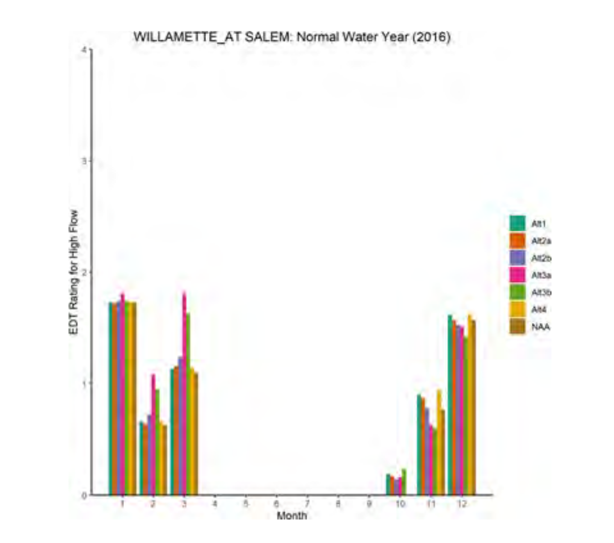


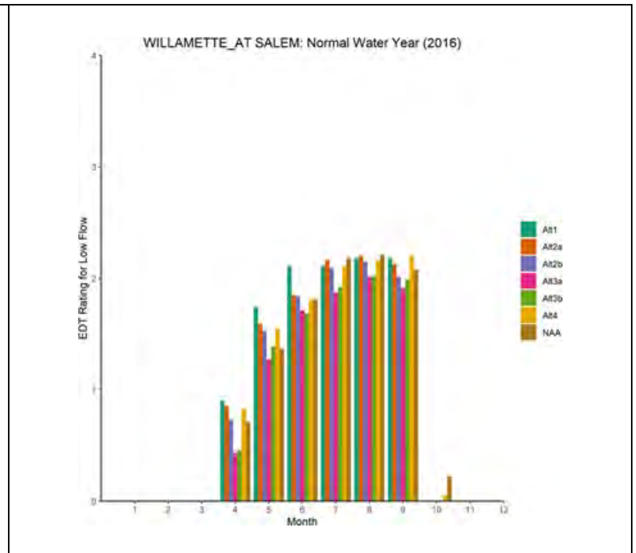


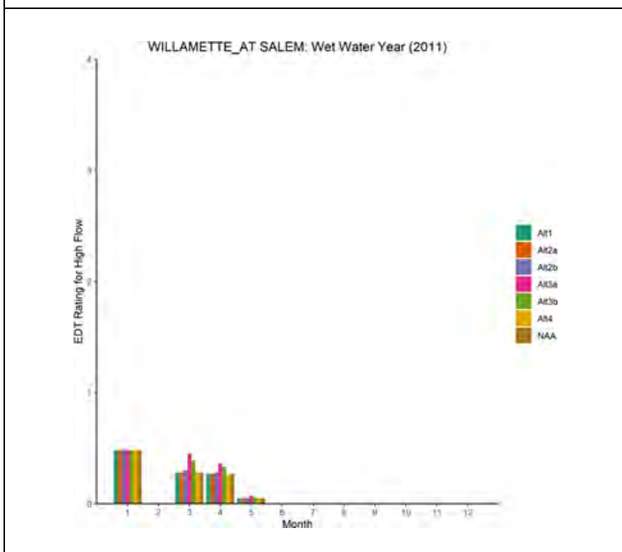


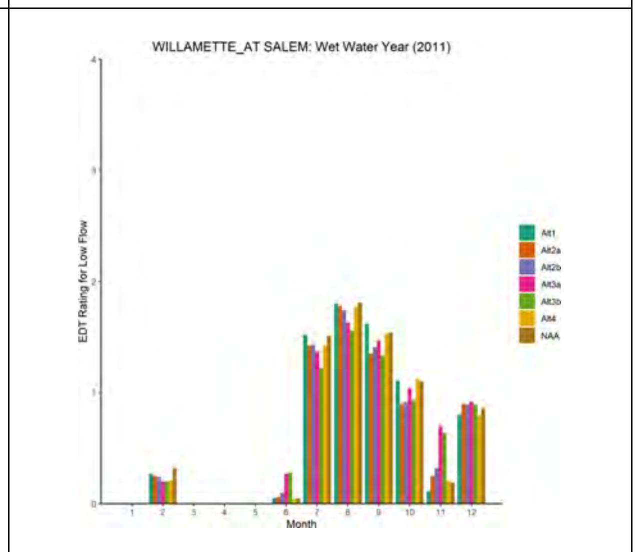


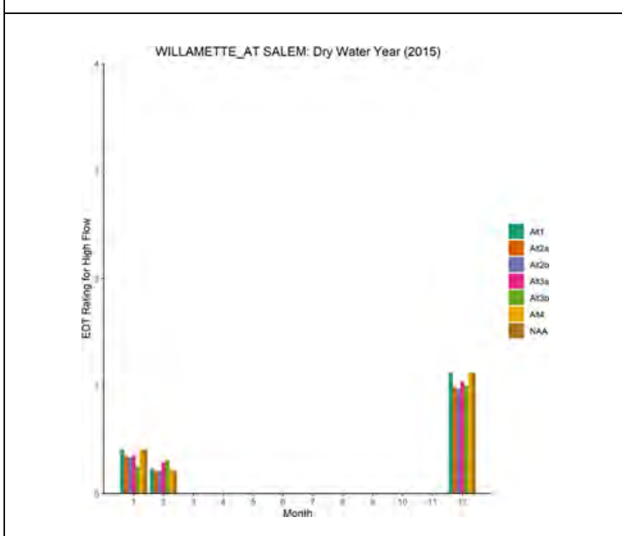
Chapter 6 Attachment C-5: Graphs of EDT Flow Ratings in Willamette River

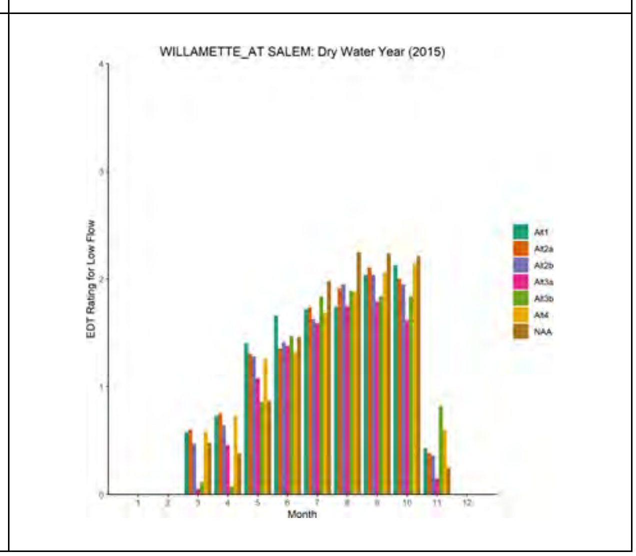












6.10 CHAPTER 6 ATTACHMENT D: USACE WVS EIS MEASUREMENTS

The following tables were provided by USACE and detail the measurements planned for each dam in the WVS system. The tables were modified to only include only those dams incorporated int the EDT-modeled scenarios. Included dams are DEX (Dexter), LOP (Lookout Point), HCR (Hills Creek), CGR (Cougar), BLU (Blue River), FOS (Foster), GPR (Green Peter), BCL (Big Cliff), DET (Detroit).

No Action Alternative

MEASURES	DEX	LOP	FCR	HCR	CGR	BLU	FOS	GPR	BCL	DET	Notes
WATER QUALITY MEASURES											
105. Construct temperature control tower		X						X		X	
166. Use lowest regulating outlets to discharge colder water during drawdown operations in fall and winter to reduce water temperatures below dams											
174. Structural improvements to reduce TDG	X	X			X		X	X	X	X	
479. Modify existing outlets to allow releases at varying depths for temperature control							X				
711. Mechanical degassing methods in fish collection/hatchery areas downstream of dams											
721. Use spillway for surface spill in summer											
FLOW MEASURES											
30. Change flows to provide effective biological benefit											
304. Augment flows by tapping power pool		X		X	X			X		X	
718. Augment instream flows by using inactive pool			X			X					
723. Reduce minimum flows to Congressionally authorized minimum flow requirements		X	X	X	X	X		X		X	
DOWNSTREAM PASSAGE MEASURES											
40. Deeper fall reservoir drawdowns to regulating outlets											
392. Construct structural downstream passage	*	X					X	X		X	* fish will be collected at LOP and transported below DEX
714. Pass water over spillway in spring											
720. Spring reservoir drawdown to regulating outlet or diversion tunnel at Cougar											
UPSTREAM PASSAGE MEASURES											
52. Provide Pacific lamprey passage and infrastructure	X				X						
639. Restore upstream and downstream passage at drop structures											
670. Update Dexter Adult Fish Facility using specs and handling practices that do not increase the risk of PSM and cease using CO_2											
722. Construct adult fish facility								X*			*722 @ GPR unless sorting @ FOS feasible

Alternative 1

MEASURES	DEX	LOP	FCR	HCR	CGR	BLU	FOS	GPR	BCL	DET	Notes
WATER QUALITY MEASURES											
105. Construct temperature control tower		X						X		X	
166. Use lowest regulating outlets to discharge colder water during drawdown operations in fall and winter to reduce water temperatures below dams											
174. Structural improvements to reduce TDG	X	X			X		X	X	X	X	
479. Modify existing outlets to allow releases at varying depths for temperature control							X				
711. Mechanical degassing methods in fish collection/hatchery areas downstream of dams											
721. Use spillway for surface spill in summer											
FLOW MEASURES											
30. Change flows to provide effective biological benefit											
304. Augment flows by tapping power pool		X		X	X			X		X	
718. Augment instream flows by using inactive pool			X			X					
723. Reduce minimum flows to Congressionally authorized minimum flow requirements		X	X	X	X	X		X		X	
DOWNSTREAM PASSAGE MEASURES											
40. Deeper fall reservoir drawdowns to regulating outlets											
392. Construct structural downstream passage	*	X					X	X		X	* fish will be collected at LOP and transported below DEX
714. Pass water over spillway in spring											
720. Spring reservoir drawdown to regulating outlet or diversion tunnel at Cougar											
UPSTREAM PASSAGE MEASURES											
52. Provide Pacific lamprey passage and infrastructure	X				X						
639. Restore upstream and downstream passage at drop structures											
670. Update Dexter Adult Fish Facility using specs and handling practices that do not increase the risk of PSM and cease using CO_2											
722. Construct adult fish facility								X*			*722 @ GPR unless sorting @ FOS feasible

Alternative 2a: Integrated Water Management Flexibility and ESA-Listed Fish Alternative

MEASURES	DEX	LOP	FCR	HCR	CGR	BLU	FOS	GPR	BCL	DET	Notes
WATER QUALITY MEASURES											
105. Construct temperature control tower										X	
166. Use lowest regulating outlets to discharge colder water during drawdown operations in fall and winter to reduce water temperatures below dams								X			
174. Structural improvements to reduce TDG											
479. Modify existing outlets to allow releases at varying depths for temperature control							X				
711. Mechanical degassing methods in fish collection/hatchery areas downstream of dams											
721. Use spillway for surface spill in summer								X			
FLOW MEASURES											
30. Change flows to provide effective biological benefit	X	X	X	X	X	X	X	X	X	X	
304. Augment flows by tapping power pool		X		X	X			X		X	
718. Augment instream flows by using inactive pool			X			X					
723. Reduce minimum flows to Congressionally authorized minimum flow requirements											
DOWNSTREAM PASSAGE MEASURES											
40. Deeper fall reservoir drawdowns to regulating outlets								X			
392. Construct structural downstream passage		X			X		X			X	
714. Pass water over spillway in spring								X			
720. Spring reservoir drawdown to regulating outlet											
UPSTREAM PASSAGE MEASURES											
52. Provide Pacific lamprey passage and infrastructure	X				X			X			
639. Restore upstream and downstream passage at drop structures											
670. Update Dexter Adult Fish Facility using specs and handling practices that do not increase the risk of PSM and cease using CO_2											
722. Construct adult fish facility								X			

Alternative 2b: Integrated Water Management Flexibility and ESA-Listed Fish Alternative

MEASURES	DEX	LOP	FCR	HCR	CGR	BLU	FOS	GPR	BCL	DET	Notes
WATER QUALITY MEASURES											
105. Construct temperature control tower										X	
166. Use lowest regulating outlets to discharge colder water during drawdown operations in fall and winter to reduce water temperatures below dams								X			
174. Structural improvements to reduce TDG											
479. Modify existing outlets to allow releases at varying depths for temperature control							X				
711. Mechanical degassing methods in fish collection/hatchery areas downstream of dams											
721. Use spillway for surface spill in summer								X			
FLOW MEASURES											
30. Change flows to provide effective biological benefit	X	X	X	X	X	X	X	X	X	X	
304. Augment flows by tapping power pool		X		X	X			X		X	
718. Augment instream flows by using inactive pool			X			X					
723. Reduce minimum flows to Congressionally authorized minimum flow requirements											
DOWNSTREAM PASSAGE MEASURES											
40. Deeper fall reservoir drawdowns to regulating outlets (Cougar diversion tunnel)					X			X			
392. Construct structural downstream passage		X					X			X	
714. Pass water over spillway in spring								X			
720. Spring reservoir drawdown to regulating outlet (diversion tunnel at Cougar)					X						
UPSTREAM PASSAGE MEASURES											
52. Provide Pacific lamprey passage and infrastructure	X				X			X			
639. Restore upstream and downstream passage at drop structures											
670. Update Dexter Adult Fish Facility using specs and handling practices that do not increase the risk of PSM and cease using CO_2											
722. Construct adult fish facility								X			

Alternative 3a: Operational Fish Passage Alternative

MEASURES	DEX	LOP	FCR	HCR	CGR	BLU	FOS	GPR	BCL	DET	Notes
WATER QUALITY MEASURES	*	*					*		*	*	***Spread Spill**
105. Construct temperature control tower											
166. Use lowest regulating outlets to discharge colder water during drawdown operations in fall and winter to reduce water temperatures below dams		X						X		X	
174. Structural improvements to reduce TDG											
479. Modify existing outlets to allow releases at varying depths for temperature control				X*		X*				X**	*Modify Spillway **lining the lower Ros
711. Mechanical degassing methods in fish collection/hatchery areas downstream of dams											
721. Use spillway for surface spill in summer		X		X		X	X	X		X	
FLOW MEASURES											
30. Change flows to provide effective biological benefit	X	X	X	X	X	X	X	X	X	X	
304. Augment flows by tapping power pool		X		X	X			X		X	
718. Augment instream flows by using inactive pool			X			X					
723. Reduce minimum flows to Congressionally authorized minimum flow requirements											
DOWNSTREAM PASSAGE MEASURES											
40. Deeper fall reservoir drawdowns to regulating outlets		X		X	X	X		X		X	
392. Construct structural downstream passage											
714. Pass water over spillway in spring	X		X	X				X	X		
720. Spring reservoir drawdown to regulating outlet		X			X					X	
UPSTREAM PASSAGE MEASURES											
52. Provide Pacific lamprey passage and infrastructure				X	X	X		X			
639. Restore upstream and downstream passage at drop structures											
670. Update Dexter Adult Fish Facility using specs and handling practices that do not increase the risk of PSM and cease using CO_2											
722. Construct adult fish facility				X		X		X			

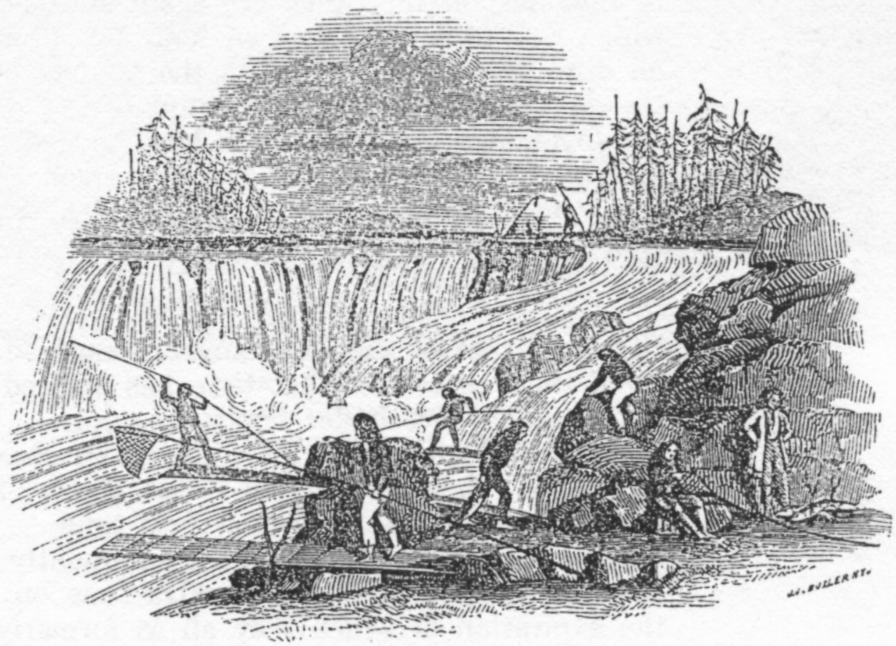
Alternative 3b: Operational Fish Passage Alternative

MEASURES	DEX	LOP	FCR	HCR	CGR	BLU	FOS	GPR	BCL	DET	Notes
WATER QUALITY MEASURES	*	*					*		*	*	***Spread Spill**
105. Construct temperature control tower											
166. Use lowest regulating outlets to discharge colder water during drawdown operations in fall and winter to reduce water temperatures below dams		X						X		X	
174. Structural improvements to reduce TDG											
479. Modify existing outlets to allow releases at varying depths for temperature control				X*	X***	X*				X**	*Modify Spillway **Lining the lower Ros ***modify diversion tunnel to make it safe/dam safety - CGR 2.0
711. Mechanical degassing methods in fish collection/hatchery areas downstream of dams											
721. Use spillway for surface spill in summer		X		X		X	X	X		X	
FLOW MEASURES											
30. Change flows to provide effective biological benefit	X	X	X	X	X	X	X	X	X	X	
304. Augment flows by tapping power pool		X		X				X		X	
718. Augment instream flows by using inactive pool			X			X					
723. Reduce minimum flows to Congressionally authorized minimum flow requirements											
DOWNSTREAM PASSAGE MEASURES											
40. Deeper fall reservoir drawdowns to regulating outlets		X		X	X	X		X		X	
392. Construct structural downstream passage											
714. Pass water over spillway in spring	X	X							X	X	
720. Spring reservoir drawdown to regulating outlet (diversion tunnel at Cougar)				X	X			X			
UPSTREAM PASSAGE MEASURES											
52. Provide Pacific lamprey passage and infrastructure				X	X	X		X			
639. Restore upstream and downstream passage at drop structures											
670. Update Dexter Adult Fish Facility using specs and handling practices that do not increase the risk of PSM and cease using CO_2											
722. Construct adult fish facility				X		X		X			

Alternative 4: Structural Fish Passage Alternative

MEASURES	DEX	LOP	FCR	HCR	CGR	BLU	FOS	GPR	BCL	DET	Notes
WATER QUALITY MEASURES							*				**TDG reduction included at FOS adult collection facility**
105. Construct temperature control tower		X		X						X	
166. Use lowest regulating outlets to discharge colder water during drawdown operations in fall and winter to reduce water temperatures below dams								X			
174. Structural improvements to reduce TDG	X	X*			X		X	X	X	X*	*incorporated into design of 105
479. Modify existing outlets to allow releases at varying depths for temperature control							X*				*FWWS and modified Fish Weir
711. Mechanical degassing methods in fish collection/hatchery areas downstream of dams	X*								X		*at adult fish facility
721. Use spillway for surface spill in summer								X			
FLOW MEASURES											
30. Change flows to provide effective biological benefit	X	X	X	X	X	X	X	X	X	X	
304. Augment flows by tapping power pool		X		X	X			X		X	
718. Augment instream flows by using inactive pool			X			X					
723. Reduce minimum flows to Congressionally authorized minimum flow requirements											
DOWNSTREAM PASSAGE MEASURES	*								*		*fish will be collected at upstream dam and transported downstream of re-reg dam
40. Deeper fall reservoir drawdowns to regulating outlets											
392. Construct structural downstream passage		X		X	X		X			X	
714. Pass water over spillway in spring											
720. Spring reservoir drawdown to regulating outlet											
UPSTREAM PASSAGE MEASURES											
52. Provide Pacific lamprey passage and infrastructure	X			X	X						
639. Restore upstream and downstream passage at drop structures											
670. Update Dexter Adult Fish Facility using specs and handling practices that do not increase the risk of PSM and cease using CO2											
722. Construct adult fish facility				X							

CHAPTER 7 - UPPER WILLAMETTE RIVER LIFE CYCLE MODELING



Willamette Falls

June 2022
Report to the United States Army Corps of Engineers

James Myers
Jeff Jorgensen
Morgan Bond
Richard Zabel

Northwest Fisheries Science Center (NFSC)
2725 Montlake Blvd. E
Seattle, Washington 98112

Executive Summary

Life cycle models have proven to be useful in predicting the consequences of specific actions or events on the viability of populations. The Northwest Fisheries Science Center (NFSC) has developed models to evaluate the benefits of providing juvenile and adult passage for ESA listed spring-run Chinook salmon and steelhead at several dams in the Upper Willamette River (UWR). In this report we utilize population-specific life cycle models to assess population viability under baseline (current or NAA) and proposed fish passage scenarios.

We analyzed spring-run Chinook salmon and winter-run steelhead populations in the Upper Willamette to highlight the ability of management efforts. Presently, conditions for salmon and steelhead in the Upper Willamette River ESU vary considerably across populations, with differing levels of natural production, abundance, and extinction risk. Similarly, the proportion of accessible historical spawning habitat for these populations varies among populations.

Overall, the majority of historical spawning habitat lies upstream of currently impassable dams. Accordingly, recovering listed Chinook salmon and steelhead will require the restoration of access to headwater regions for multiple populations.

The US Army Corps of Engineers (USACE) manages 13 dams and reservoirs in the Willamette River Basin. In April 2018, the National Marine Fisheries Service (NMFS) reinitiated formal Section 7 consultation on the operation and maintenance of the Willamette Valley Project. As a part of that process, the USACE is assessing a number of alternative scenarios for restoring access to headwater areas above USACE projects. This Life Cycle Model (LCM) analysis of passage alternatives in four UWR subbasins: North Santiam, South Santiam, McKenzie, and Middle Fork Willamette, is one of three approaches being employed by the USACE. Within each subbasin, the USACE specified structural or operational passage options for each dam. Further, the USACE estimated the effectiveness of each of these passage options for Chinook salmon and steelhead through the development of fish benefit workbooks (FBWs). Our life cycle model utilized these FBWs to estimate the abundance of natural-origin spawning adults over a 100-year period for each of the passage alternatives developed by the USACE. In addition to the effects of each passage option through the FBW, we also assessed the effects of passage options on Chinook salmon and steelhead via changes in downstream stream temperatures and total dissolved gas (TDG).

Irrespective of the passage option, the LCM captured variability in freshwater and ocean conditions, in addition to the influence of introgression and domestication by hatchery-origin Chinook salmon adults.

We produced model outputs to represent the four components of the Viable Salmonid Population (VSP) metrics: Productivity, Abundance, Life-History Diversity, and Spatial Structure. We rolled these component scores into an overall VSP score that represents how well the population performs under the various Alternatives.

Comparisons across the no action alternative (NAA) and six Alternatives (1,2a,2b,3a,3b, and 4) for all Chinook salmon and steelhead populations suggest that the highest improvement in overall VSP status would be obtained through Alternatives 1, 2a, 2b, and 4, with structural passage provided at least 2 projects (Figures 0.1 – 0.6). Results suggest that Alternative 4 would

produce the most populations (4) at full viability (VSP>3), while Alternatives 1 and 2b would produce the most populations (5) at moderate viability (VSP>2). Structural passage options, floating screen structures (FSS) or floating surface collectors (FSC), were predicted in the FBWs to have both high collection efficiencies and high passage survival, which is not well supported in empirical studies for Chinook and steelhead. Further, as part of some passage options collected juveniles were often transported around downstream dams to avoid additional passage mortalities and poor downstream water conditions (primarily TDG). Alternatives that relied solely on operational passage, 3a and 3b, did poorly compared to the other alternatives. It is beyond the scope of this report to detail differences between structural and operational passage at high head dams; however, it appears much of the inefficiency inherent in operational passage (as expressed in the FBW) comes from periods of time when the reservoir elevations are not ideal for passage through regulating outlets or via spill. Additionally, empirical studies indicate that in many cases these operational routes are often not necessarily "fish friendly". Finally, while structural passage options consistently provide good passage conditions, operational passage is most effective at certain reservoir elevations, and depending on hydrological conditions these passage windows vary in duration from year to year. Juvenile emigration timing has an important effect on over passage success, and there is considerable uncertainty in how juvenile fish will react to changes in reservoir conditions (elevation, flow, temperature, etc), and if and when fish will attempt to emigrate downstream.

Aside from the FBW, water quality conditions (temperature and TDG) below the dams varied depending on the passage option. Under certain conditions these factors had a significant effect on juvenile and adult survival. In some scenarios, TDG effects were mitigated with degassing structures. Changes in flow under different scenarios also affected downstream temperatures and survival. While we incorporated these passage option-specific temperature changes into the life- cycle model, we did not include any estimates of future temperature changes under a climate change scenario. Other factors influence survivals in the life cycle model, in most cases these were universal or nearly so across NAA and the Alternatives. They do provide additional sources of variability, which affects the probability of falling below the quasi-extinction threshold.

The underlying uncertainty in many of the parameters used in developing this life cycle model contributes to the overall uncertainty in the estimates of abundance and viability. Efforts to calibrate the NAA using recent population estimates were helpful in reducing some uncertainty, although NAA conditions rarely included adequate passage conditions at the high head dams.

Ultimately, because of the shared parameters across Alternatives, comparisons of the relative performance of each Alternative may be more appropriate. In this report we only present estimates for Chinook salmon and steelhead population abundance and viability under different passage options, which are combined in specific Alternatives for consideration by the USACE. We do not attempt to identify which Alternative is “best”.

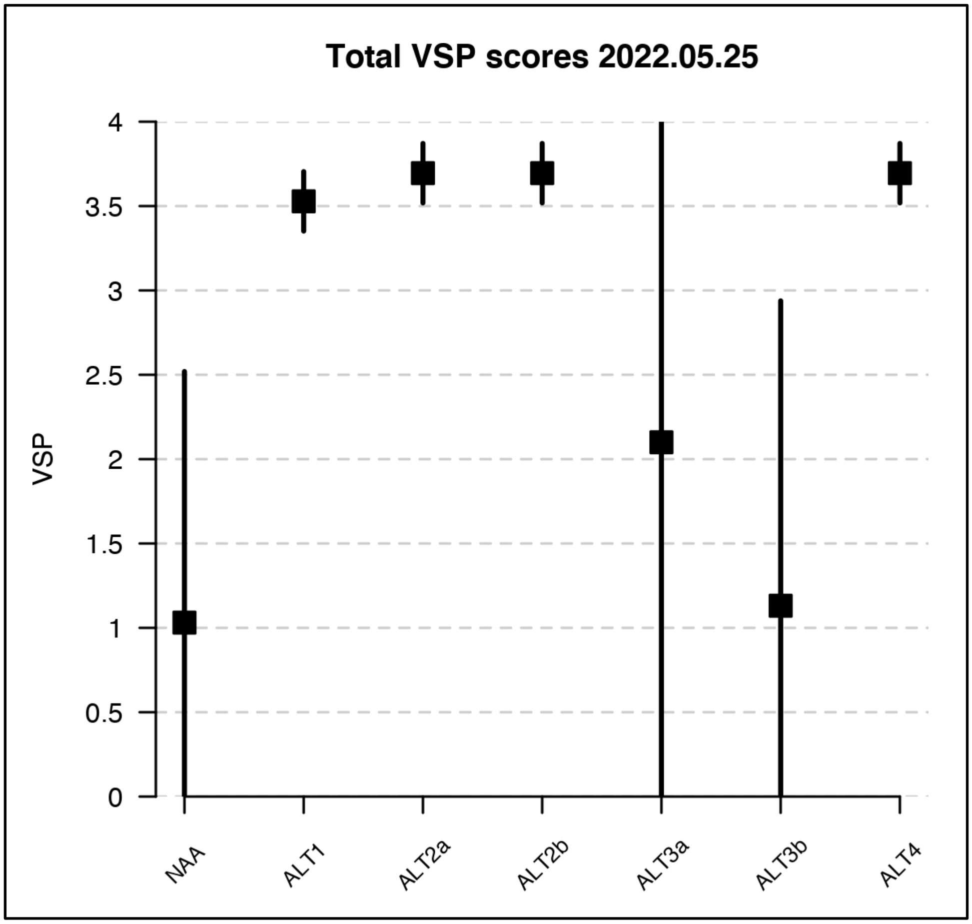


Figure 7-1. North Santiam River Chinook Total VSP scores for all passage alternatives. *Total VSP score for Chinook salmon in the North Santiam River under passage alternatives. (Median VSP with 95% confidence interval from 1000 LCM runs per alternative.)*

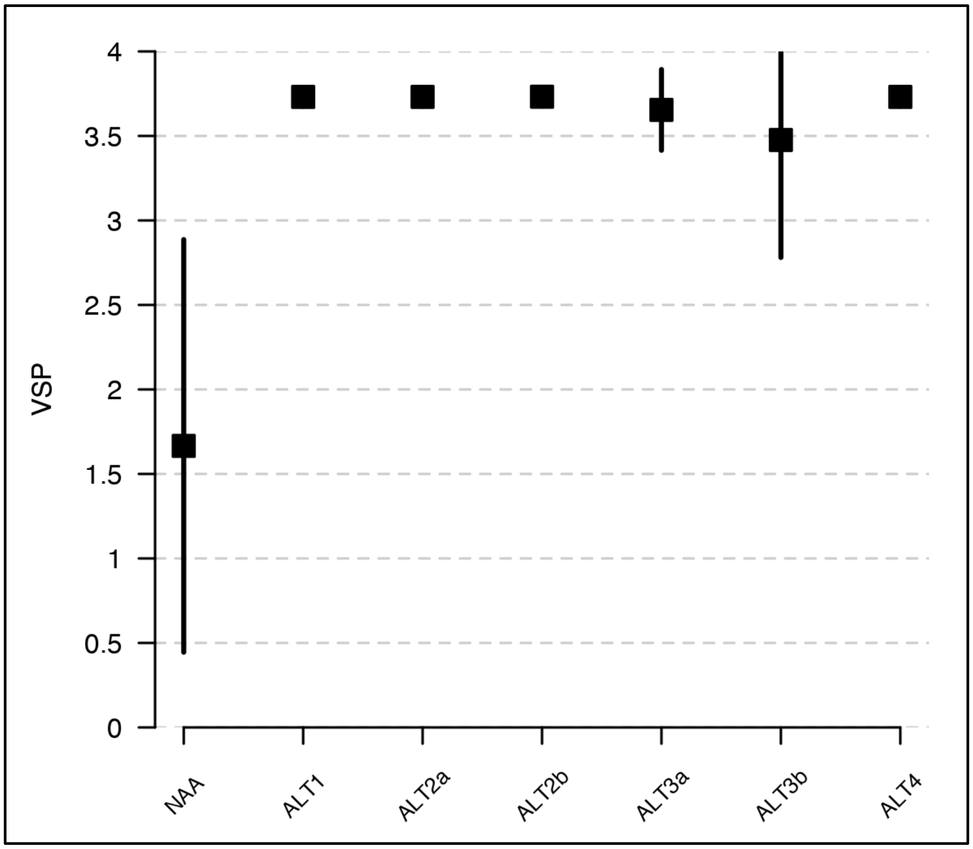


Figure 7-2. North Santiam River Winter Steelhead VSP Scores Under Passage Alternatives. *Total VSP score for winter steelhead in the North Santiam River under passage alternatives. (Median VSP with 95% confidence interval from 1000 LCM runs per alternative)*

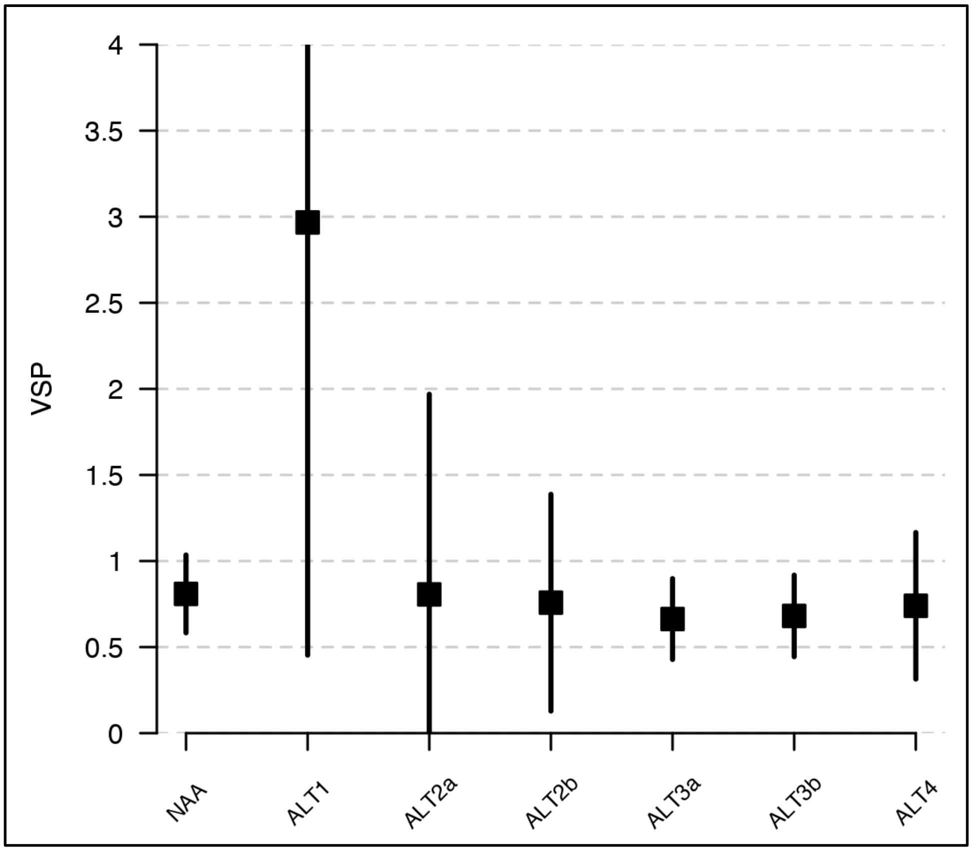


Figure 7-3. South Santiam River Chinook VSP scores under Passage Alternatives. *Total VSP score for Chinook salmon in the South Santiam River under passage alternatives. (Median VSP with 95% confidence interval from 1000 LCM runs per alternative)*

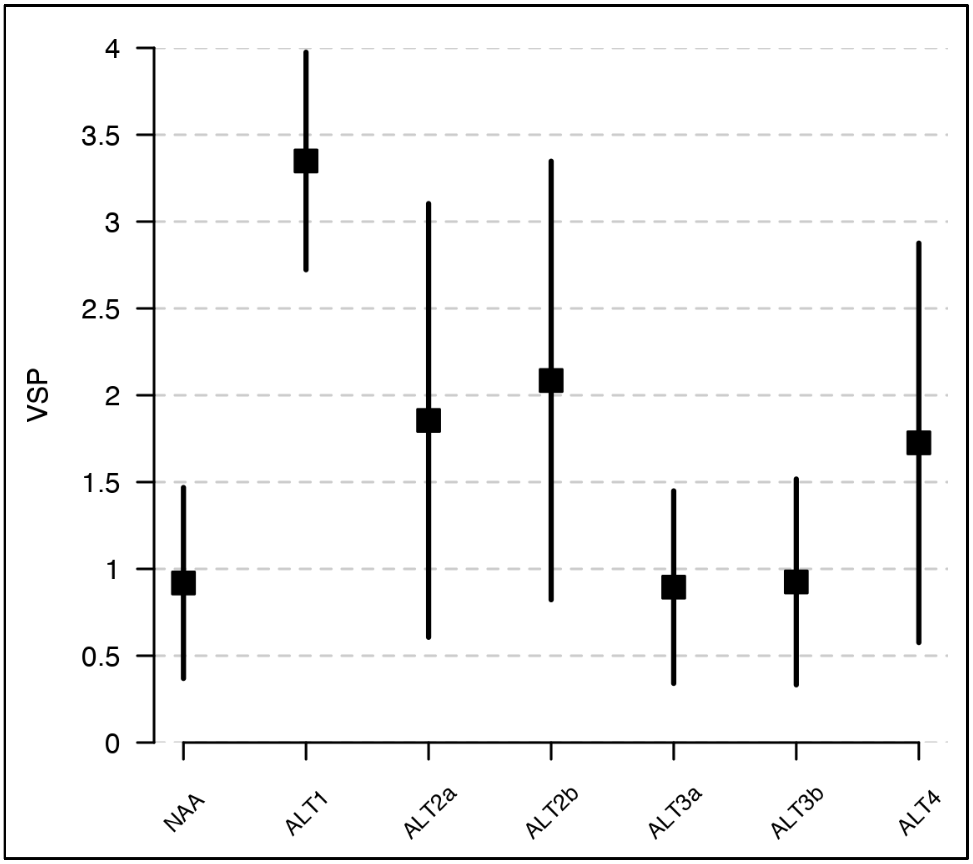


Figure 7-4. South Santiam River Winter Steelhead VSP Scores for Passage Alternatives. *Median total VSP score for winter steelhead in the South Santiam River under passage alternatives. (Median VSP with 95% confidence interval from 1000 LCM.)*

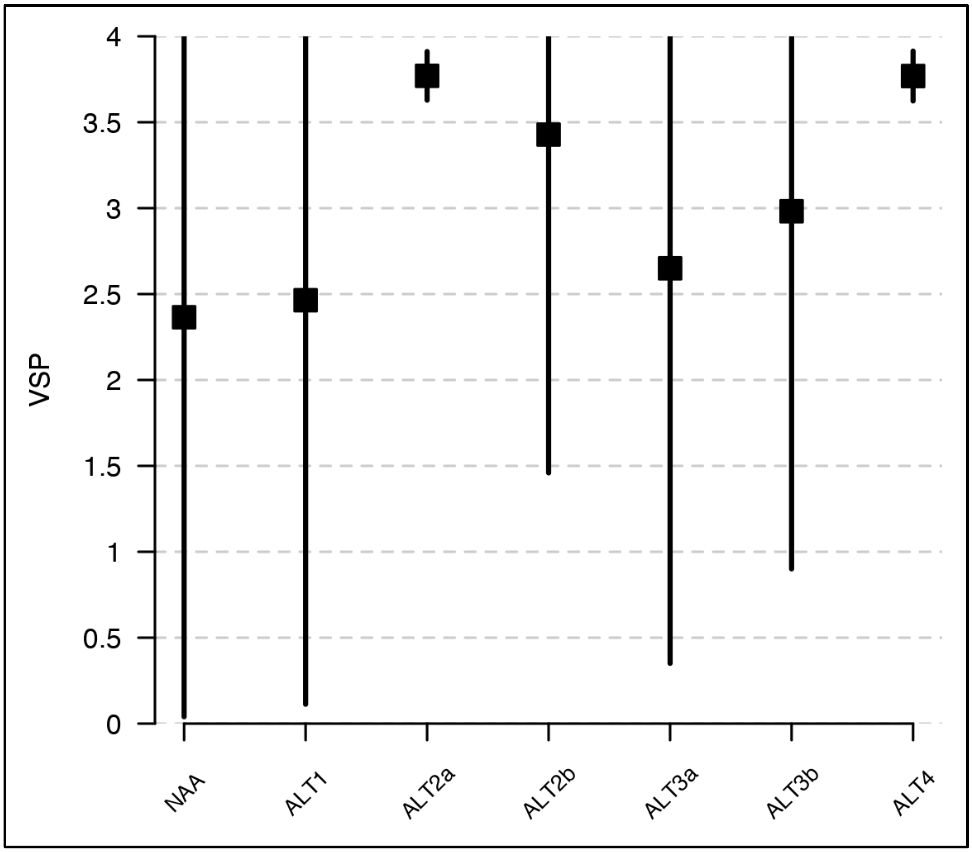


Figure 7-5. McKenzie River Chinook VSP scores under passage alternatives. *Total VSP score for Chinook salmon in the McKenzie River under passage alternatives. (Median VSP with 95% confidence interval from 1000 LCM runs per alternative)*

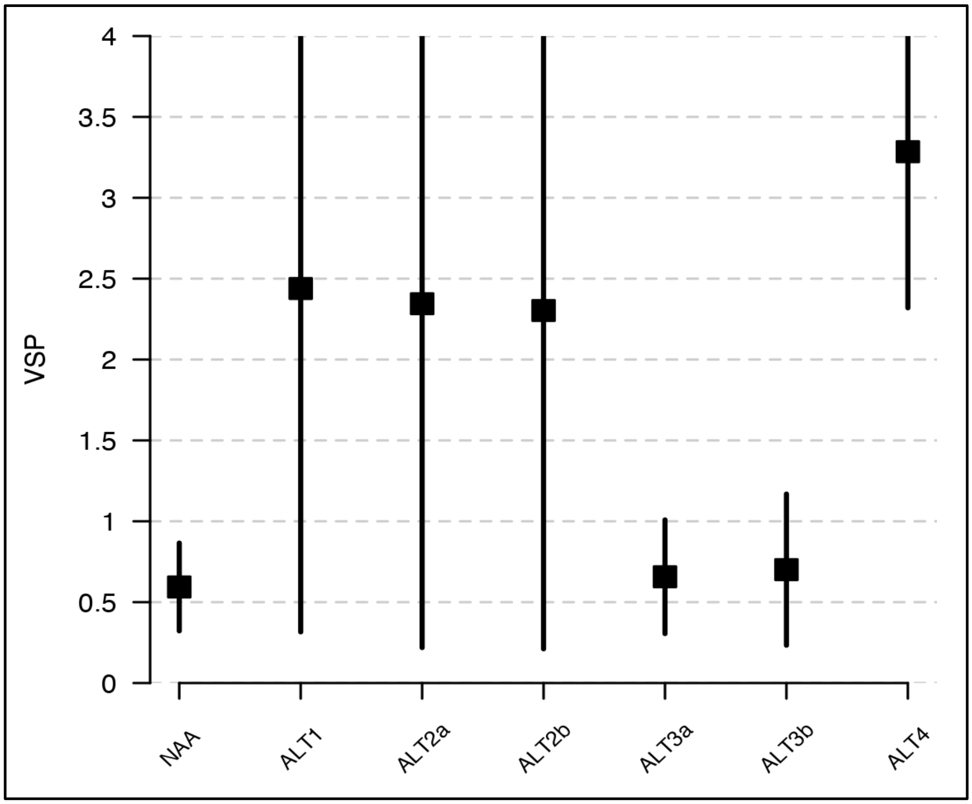


Figure 7-6. Middle Fork Willamette River Chinook VSP Scores Under Passage Alternatives
Total VSP score for Chinook salmon in the Middle Fork Willamette River under passage alternatives. (Median VSP with 95% confidence interval from 1000 LCM runs per alternative)

7.1 INTRODUCTION AND OVERVIEW

7.1.1 Introduction

Due to substantial declines in adult returns, the Upper Willamette River (UWR) spring-run Chinook salmon Evolutionarily Significant Unit (ESU) and UWR steelhead Distinct Population Segment (DPS) were listed as threatened under the U.S. Endangered Species Act (NMFS 1999a, b, 2005). Among the potential factors identified as responsible for the decline were the 13 dams and reservoirs operated by the Federal Government. These dams and reservoirs are managed for flood control, recreation, irrigation, fish and wildlife management, and power generation. A subsequent assessment of the effects of these dams resulted in a jeopardy determination by the National Marine Fisheries Service (NMFS) in the 2008 Biological Opinion (BiOp). The BiOp includes Reasonable and Prudent Alternatives (RPAs) for the US Army Corps of Engineers' (USACE) Willamette System (WS) describing actions that would avoid the likelihood of continued jeopardy to the species. These include providing both upstream and downstream fish passage for spring-run Chinook salmon and winter-run steelhead in sub-basins affected by Willamette System (WS) dams. In response to the BiOp, the USACE (Portland District) and regional partners have developed a suite of proposed actions designed to enhance Chinook salmon and steelhead population status. The actions are focused primarily on fish passage at dams, but also address other issues such as water temperature control, supplementation with hatchery fish, mainstem and tributary flows, and bank armoring.

In October 2015, the USACE completed the Willamette Configurations Operations Plan (COP) (USACE 2015) that identified a long-term strategy to address the NMFS BiOp requirements, and prepared a 5-year plan to implement the strategy. For downstream passage, the COP strategy includes major downstream passage actions at Cougar and Detroit dams, as well as continued reservoir operations at Fall Creek Dam and spill weir improvements at Foster Dam. It also included the continuation of research, monitoring and evaluation (RM&E) to investigate feasibility of providing effective fish passage in the Middle Fork Willamette River, although the benefits achieved in other sub-basins would be assessed prior to proceeding with fish passage actions in that sub-basin.

The development of the COP was based, in part, on the results from life cycle models simulating populations of spring-run Chinook salmon and winter-run steelhead under various fish passage scenarios. For this report, these population-specific life cycle models have been modified to incorporate the proposed structural and operational alternatives at the dams and fish passage systems for the Middle Fork, McKenzie, and the North and South Santiam basins, as specified in the 2022 WVS EIS analysis.

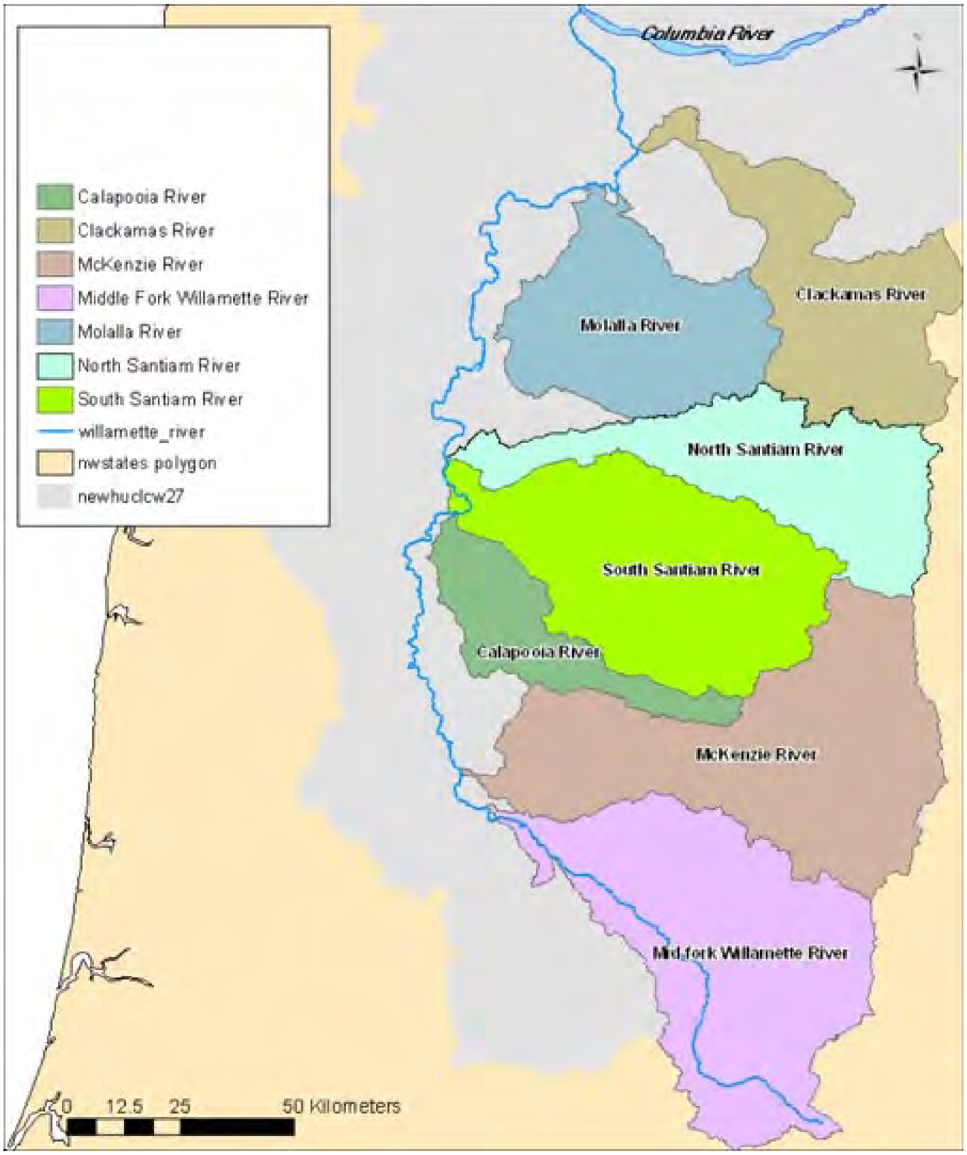


Figure 7-7. Upper Willamette River Basin Spring Chinook Populations

7.1.2 Life-Cycle Model

Upper Willamette River salmonid life-cycle models were originally developed for four populations of Chinook salmon (North Santiam River, South Santiam River, McKenzie River and Middle Fork Willamette River) and two populations of steelhead (North and South Santiam River) located in the Upper Willamette River (Figure 6.7; Zabel et al. 2015). Current versions of these population-specific models were developed in the programming language R (R_Core_Team 2013).

Both the Chinook and steelhead models are population-specific life cycle models (Figure 1.2) that follow a similar general structure as described below. The models are age-structured (annual), stage-based population viability models with stochastic elements. They consist of an array of individual abundances by age (from age 1 through 6 year olds) of the stream reaches within each population/river system that have consistent and significant fish production contributing to the population. Individuals in the array advanced to the next age class at each model annual time step by application of population-specific survival, productivity and fertility parameters. The demographic parameters were derived from studies in the basin, borrowed from similar nearby populations, derived from the model calibration process, informed from published studies of Chinook salmon, or were inferred through expert opinion. The parameters consisted of: demographic rates that determined productivity, survival, and capacity; life history splits into one of several juvenile rearing strategies; ocean maturation rates; harvest; and relative reproductive success of hatchery versus wild-origin fish. Some parameter values were unique to a particular tributary production area (e.g., egg capacities, and dam survival if fish in a tributary must navigate a project) while other parameters were shared and applied to fish from all tributary production areas (e.g., ocean maturation and survival, harvest).

In general, natural and hatchery origin adults return to their natal river and are allocated to the specific reach where they were born (straying to non-natal reaches or other rivers is not considered in the current model). Within each reach, some adults experience pre-spawn mortality (PSM, which is dependent on temperature and proportion of hatchery origin spawners (pHOS) and covered in Chapter 8), while others successfully spawn. A proportion of the eggs survive to the emergent fry stage (covered in chapter 2), and the fry are assigned to one of three freshwater life history strategies: those emigrating to the estuary as spring sub-yearlings, fall sub- yearlings, or as 2nd spring yearlings for Chinook salmon (Schroeder et al. 2016), and parr, yearling, and two-year old smolts, for steelhead. Juveniles produced upstream of dams rear in the reservoirs for varying durations depending on their life history trajectory and the method of juvenile downstream passage. Dam passage alternatives were provided by the USACE (described in chapter 3), and included monthly estimates of fish of each life stage approach dams, estimated dam passage efficiency and survival over a multiyear water year period of record. Reservoir rearing survivals were estimated by an expert panel (Zabel et al. 2015). Passage alternatives are fully described in the USACE' Configurations Operations Plan (COP) report (USACE 2015).

Juveniles emigrating through dams or incubating and rearing in the reach immediately below the dam are subject to mortality from gas bubble disease when total dissolved gas (TDG) exceeds specific supersaturation thresholds (covered in chapter 4). TDG levels are based on historical flow conditions that were also utilized in the development of the fish passage and survival fish benefit workbook (FBW) developed by the USACE (USACE 2015). Surviving juveniles from each reach are combined and move out past Willamette Falls to become smolts in the estuary.

Following the smolt stage, they enter the ocean as age 2 fish. Survival in the ocean based on relationships with ocean indicators and is described in chapter 6.5. For Chinook salmon, once in the ocean, a proportion of three-year-old fish are either harvested (covered in chapter 6.6), return to the river, die, or remain in the ocean to become four-year-old fish, and this routine is repeated for four-, five, and six-year-olds in the ocean. Steelhead initially returned to spawn after two, three, or four years in the ocean. Additionally, a small proportion of steelhead are repeat spawners, and return after one year in the ocean.

Returning adults are subject to mortality from harvest and other causes (disease, predation, TDG, etc.) prior to reaching their natal river (Chapters 6 and 7). The life cycle models are normally initialized for five to seven model years prior to initiating the 100-year period. Each alternative was then run 1000 times.

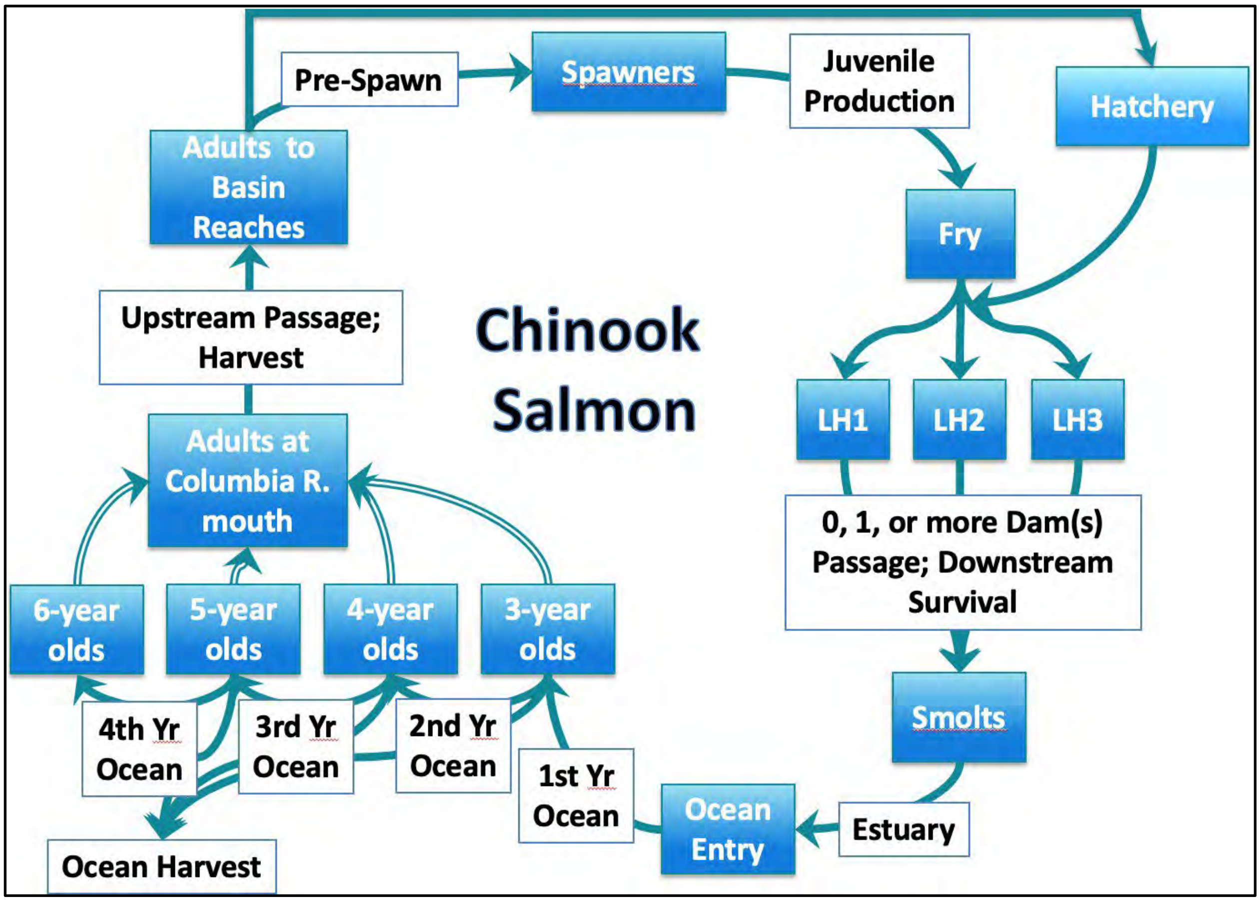


Figure 7-8. Spring Chinook Salmon Life Cycle Model for Natural Origin Fish. *Representation of the life cycle of spring Chinook salmon used in life cycle modeling corresponding to natural origin fish. Distinct life history trajectories are provided for the three primary freshwater juvenile life histories and four marine maturation times. In addition, different dam passage scenarios can be incorporated, with corresponding effects on upstream reservoir residency and survival and downstream dissolved gas effects on survival.*

The model also simulates concurrent hatchery production, described in Chapter 9. Specifically, juveniles produced from the hatchery are released as yearlings below the dam, and a proportion of those survive to the smolt stage. Hatchery origin fish also enter the ocean as two-year-olds, and like natural origin fish they are either harvested, return to the river, die, or remain in the ocean to become three-, four-, five-, and six-year-olds. Hatchery origin fish returning to the river will also experience freshwater harvest[1] and migration mortality, and those that survive will return either to the hatchery or to a reach to potentially spawn in the wild. The reproductive success of both hatchery and natural-origin adults was adjusted to reflect long-term domestication based on the contribution of hatchery-origin fish spawning naturally (Zabel et al. 2015). In addition, the number of hatchery juveniles released was decreased in response to an increase in the abundance of naturally produced adults, anticipating the reduced need for mitigation. In contrast, other than the limited hatchery production of "surrogate" winter steelhead (from naturally-produced adults), there has not been any production of winter steelhead in the Upper Willamette Basin since the 1990s.

Model parameters were established using values based on relationships developed for Willamette Valley fish or from values derived from the literature for other spring-run Chinook and steelhead salmon populations. Some parameters were adjusted such that baseline conditions (present-day reach-specific abundances) were reliably produced as output (Zabel et al. 2015). We describe the calibration process in chapter 10.

The data, particularly juvenile abundance and survival data, to support the models were sparse. For Chinook salmon, we relied on recent redd count data to fit the model under the NAA scenario. For steelhead, adult spawner counts were based on adult counts at Willamette Falls and tributary dams and expansions of redd index surveys (Falcy 2017, Mapes et al. 2017).

Ideally, we would have had more data (i.e., longer time series) and some data representing the juvenile phase, as was the case in the Crozier et al. (2021) analysis. Further, the NAA alternative does not necessarily capture the recent dam configuration and operations. Nonetheless, we feel that we reasonably represented the NAA alternative. Also, the alternatives from the Fish Benefit Workbook modeled dam passage alternatives and did not alter the parameters from throughout the life cycle. Thus, our results are reasonable representations of the relative effects of dam passage alternatives relative to NAA.

7.1.3 Technical Recovery Team (TRT) Population-Level Status

We evaluated model outputs based on the Viable Salmonid Population (VSP) conceptual approach (McElhany et al. 2000) to identifying and evaluating the status of salmon and steelhead population. This approach relies on the assessment of four population parameters: (1) abundance, (2) population growth rate (productivity), (3) population spatial structure, and (4) diversity. TRTs in each of the Recovery Domains developed metrics for these parameters, metrics that are often specific to the life history characteristics of each species and quantity and quality of information available. For listed salmonids in the Upper Willamette River Basin, McElhany et al. (McElhany et al. 2006, McElhany et al. 2007) present the relevant viability

[1] Hatchery origin Chinook salmon are externally marked (adipose fin clip) and subject to a selective fishery for marked fish. Unmarked (natural-origin) fish are supposed to be released if caught, but subject to handling and hooking mortality effects.

approach; however, it should be emphasized that, where new information has come to light, modifications in the viability criteria may be necessary.

7.1.4 Population Abundance and Productivity Criteria

In the 2003 viability report for the Willamette and Lower Columbia TRT (WLC TRT), McElhany et al. (2003) provided the following guidelines for abundance and productivity criteria:

Adult Population Productivity and Abundance Criteria Guidelines
• In general, viable populations should demonstrate a combination of population growth rate, productivity, and abundance that produces an acceptable probability of population persistence. Various approaches for evaluating population productivity and abundance combinations may be acceptable but must meet reasonable standards of statistical rigor. • A population with a non-negative growth rate and an average abundance approximately equivalent to estimated historical average abundance should be considered to be in the highest persistence category. The estimate of historical abundance should be credible, the estimate of current abundance should be averaged over several generations, and the growth rate should be estimated with an adequate level of statistical confidence. This criterion takes precedence over criterion 1.

These guidelines recognize that a variety of approaches may be taken to evaluating abundance and productivity and several methods were discussed in the 2003 report. McElhany et al. (2006) more fully explored the types of analyses and metrics that are useful for estimating a population's probability of persistence with reasonable statistical rigor and provide guidance on when each approach would be appropriate.

In the 2006 report, a generalized viability curve approach was presented. The "viability curve" approach utilized a combination of population abundance and productivity. This can be shown as an extinction risk iso-cline on a graph plotting population abundance against population productivity. Applying the viability curve approach requires two separate but closely related analyses. The first analysis is describing the functional relationship between abundance, productivity, and extinction risk. The second, related, analysis is determining the best metric for evaluating a given population relative to the viability curve.

7.1.5 Diversity and Spatial Structure and Overall VSP

The VSP approach also considers population diversity and spatial structure. In essence, high total VSP scores correspond to properly functioning populations that are not only abundant, but also represent historical life-history diversity and occupy the historical range.

Our overall approach was to develop a Life Cycle Model (LCM) for each population (Figure 1.3). We developed parameter sets that represented current (NAA) conditions and future management scenarios or alternatives. We used the LCM to estimate VSP component scores, and to then compute a total VSP scores. These VSP scores, then, were used to compare the performance of the various management alternatives (Figure 6-9).

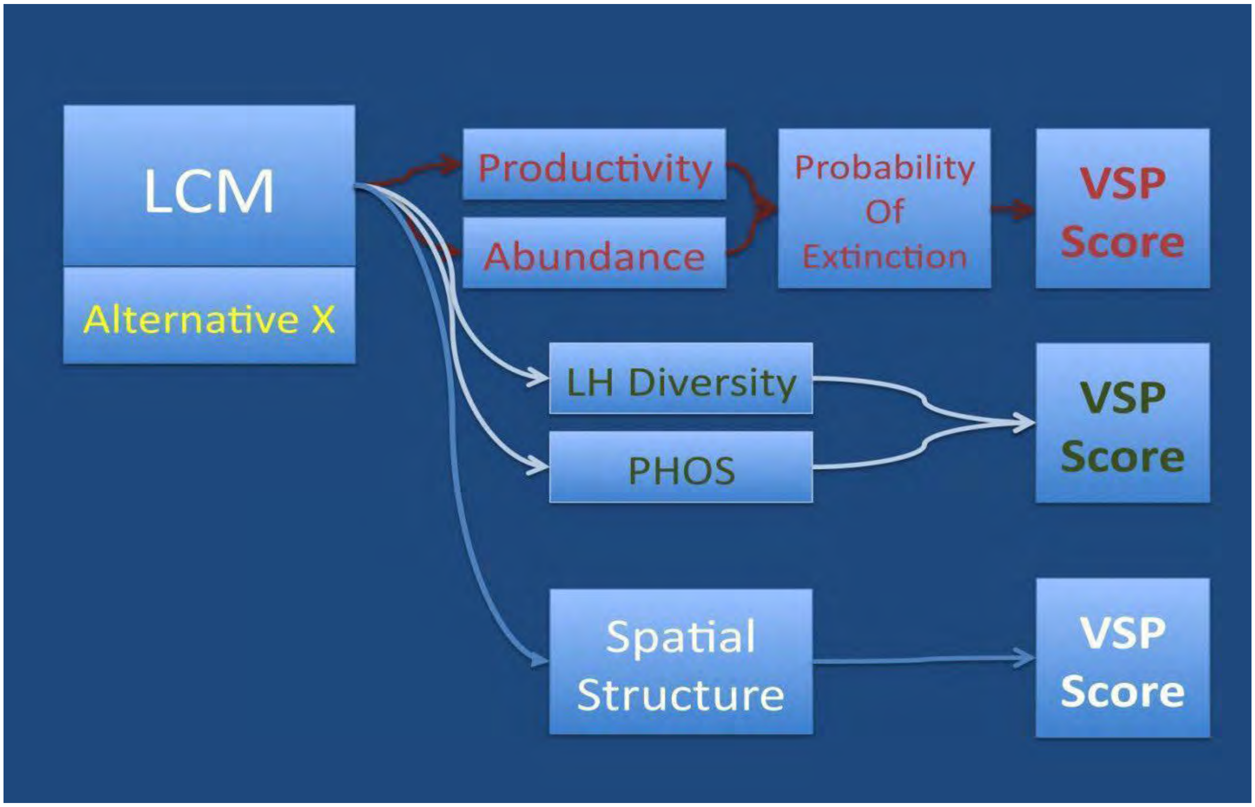


Figure 7-9. Schematic of how the Life-Cycle Model (LCM) feeds outputs for the VSP scoring. *LH Diversity is Life-History Diversity and PHOS is Proportion of Hatchery Origin Spawners.*

7.1.6 Estimating VSP Scores and Uncertainty from Productivity and Abundance Model Results

Before we describe our approach to estimating VSP based on productivity and abundance, we provide some definitions:

> *Run:* A single iteration of the model with a given set of parameters (drawn from distributions) for a set number of years (usually a hundred years). For each run we keep track of the population trajectory so we can calculate a suite of model metrics.
>
> *Alternative:* A specific set parameters that represent a particular management scenario. The "NAA" alternative represents current conditions. All other alternatives represent a proposed future management scenario.

For each alternative, we create 1000 times runs to capture the variability in model outputs. To represent productivity and abundance, we produce the following outputs:

Productivity: In keeping with previous analyses (e.g., Zabel and Jordan 2020), we calculate productivity as recruits (measured as returning spawners) per spawner measured at relatively low abundance. This represents the ability of a population to rebound at low abundance. At higher abundances, populations tend to hover about an equilibrium level, so recruits per spawner approaches unity, and does not distinguish among alternatives. Here we captured low abundance by taking the geometric mean of recruits per spawner for the first twenty-five years of each run. We chose this because we observed that this is a typical amount of time for a population to reach an equilibrium level from current conditions.

Abundance: With this measure, we are capturing population abundance at equilibrium. Accordingly, we measure abundance for years 26-100. In keeping with precedent (e.g., Zabel and Jordan, 2020), we calculated the geometric mean of abundance across each run. Geometric mean was used because population abundances tend to have a logarithmic distribution, characterized by peaks in abundance, and the geometric mean down-weights the peaks.

7.1.7 VSP Score for Abundance and Productivity

Consistent with the WLC TRT, we calculated the abundance and productivity VSP score as a measure of risk, as defined as the probability of falling below extinction thresholds. Before we describe the methods to do this, we provide some definitions:

Probability of extinction: We adopted the definition of quasi-extinction that was established by the WLC TRT as applied to the Willamette River populations. P(QET) is the probability of falling below the quasi-extinction threshold (QET) within T years, where $T = 100$. A population is considered to have fallen below the threshold if it drops below the QET threshold, on average per year, over a four-year period. We computed P(QET) for each alternative by compiling the proportion of 1000 runs that fell below the QET threshold. We chose 1000 runs because our estimates of P(QET) stabilized after that number of runs.

The quasi-extinction threshold: The quasi-extinction threshold is determined for a population based on its historical size and complexity of subpopulations. The WLC-TRT (McElhany et al 2007) set a QET of 250 spawners per year for the South Santiam, McKenzie, and Middle Fork Willamette River spring Chinook populations, and 150 for the North Santiam spring Chinook population. For the North Santiam and South Santiam winter steelhead populations, QETs were set at 200.

McElhany et al. (2000) provide guidelines on how to convert P(QET) into VSP scores (Figure 6-10) Table 1, from McElhany et al. (2000)), with 0 indicating a population is either extinct or at a very high risk of extinction, and 4 indicating a population is at very low risk of extinction.

Table 1 Population persistence probabilities associated with persistence categories (copied from 2003 viability report).

Population Persistence Category	Probability of Population Persistence in 100 Years	Description
0	0–40%	Either extinct or very high risk of extinction.
1	40–75%	Relatively high risk of extinction in 100 years.
2	75–95%	Moderate risk of extinction in 100 years.
3	95–99%	Low ("negligible") risk of extinction in 100 years (viable salmonid population).
4	>99%	Very low risk of extinction in 100 years.

Figure 7-10. Population presistence probabilities Associated with Persistence categories (PhotoCopied from 2003 viability report.

Based on the Table 1, we developed a piecewise linear translation between VSP score and probability of extinction (Figure 6.11). Thus, to estimate VSP scores for productivity and abundance VSPP&A, we first calculated P(QET) based on 1000 runs of a specified alternative. We then used the piecewise linear equation to convert P(QET) to VSPP&A.

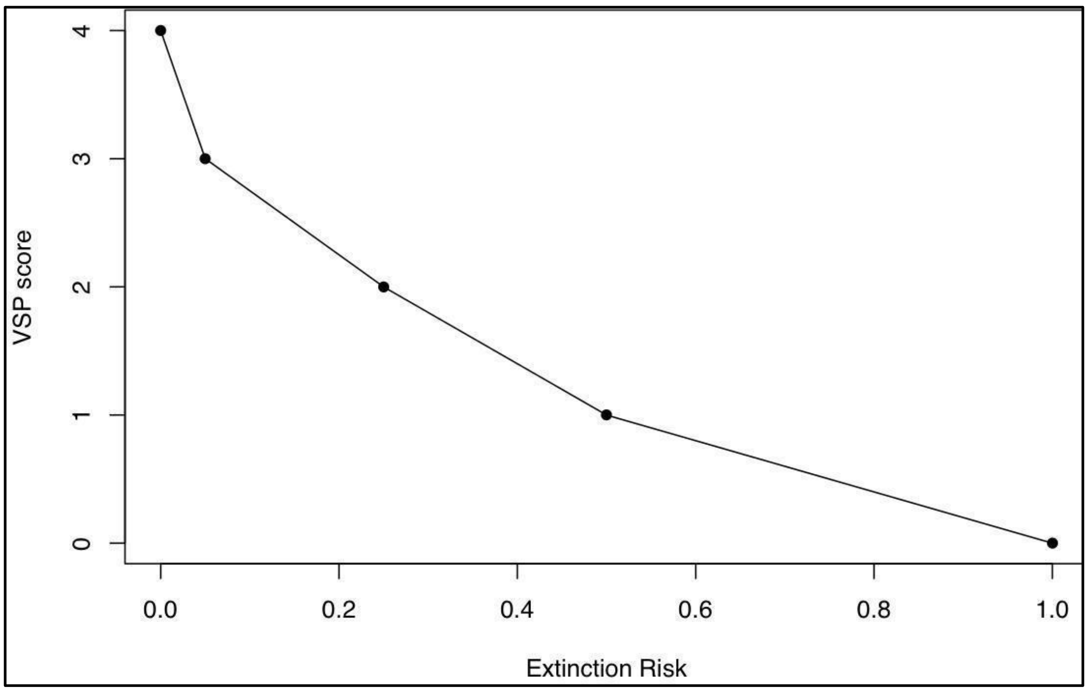


Figure 7-11. Relationship between VSP score for productivity and abundance and P(QET)

7.1.8 Calculating Uncertainty in VSP Scores for Productivity and Abundance

In addition to calculating $VSP_{P\&A}$ from life-cycle model output, we also estimated uncertainty about the scores. To do this, we adopted the following several-step approach. First, we defined a response surface that relates P(QET) to productivity and abundance for each population. Then we converted this surface to $VSP_{P\&A}$ scores. Then we plotted the 1000 runs per alternative on this surface to generate the distribution of VSP scores for an alternative. From this distribution of $VSP_{P\&A}$ scores, we could then produce estimates of variance and confidence intervals for $VSP_{P\&A}$ scores for a given alternative.

To generate the response surface, we used logistic regression to relate P(QET) to the Productivity and Abundance metrics, described above. For each of the 1000 runs within an alternative, we determined whether the individual run fell below QET. If it did, we designated it as 0; otherwise, it was designated as 1. We did this across all alternatives to create a data file with each line indicting whether the run fell below QET or not, and the mean Productivity and Abundance for the run. We then performed a logistic regression to develop a response surface for probability of extinction versus Productivity and Abundance using the following equation:

$$P(QET) = \beta_0 + \beta_P \cdot P^a + \beta_N \cdot N^b$$

where *P* is Productivity and *N* is abundance. Figure 7-12 demonstrates a response surface based on McKenzie River spring Chinook.

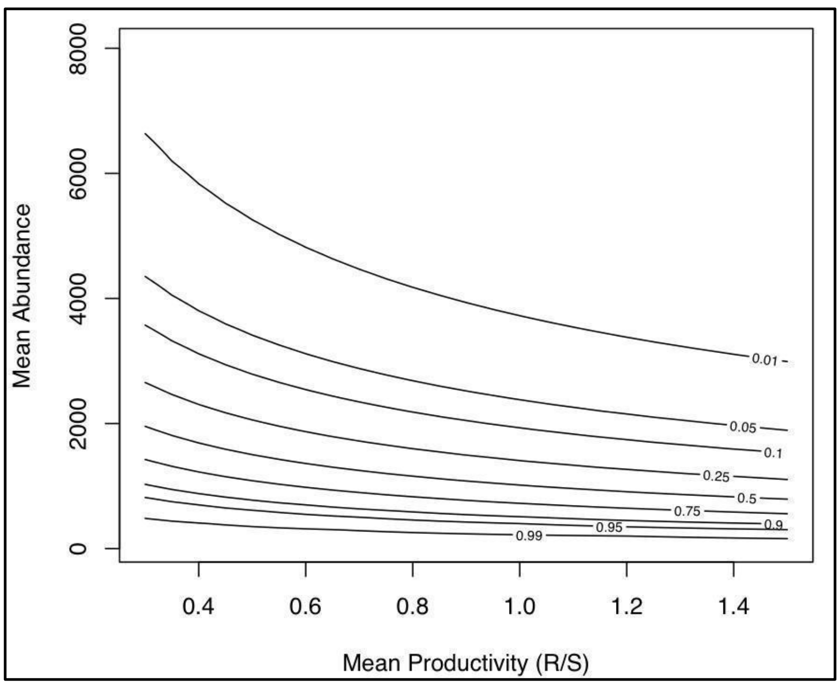


Figure 7-12. McKenzie spring Chinook population Isoclines of extinction probability by abundance versus productivity. *Isoclines of extinction probability on a plot of mean abundance versus mean productivity for the McKenzie spring Chinook population.*

Next, we translate the response surface for P(QET) to VSPP&A scores using the equation depicted in Figure 7-11. Figure 7-12 demonstrates a response surface of Productivity and Abundance versus VSPP&A for McKenzie spring Chinook.

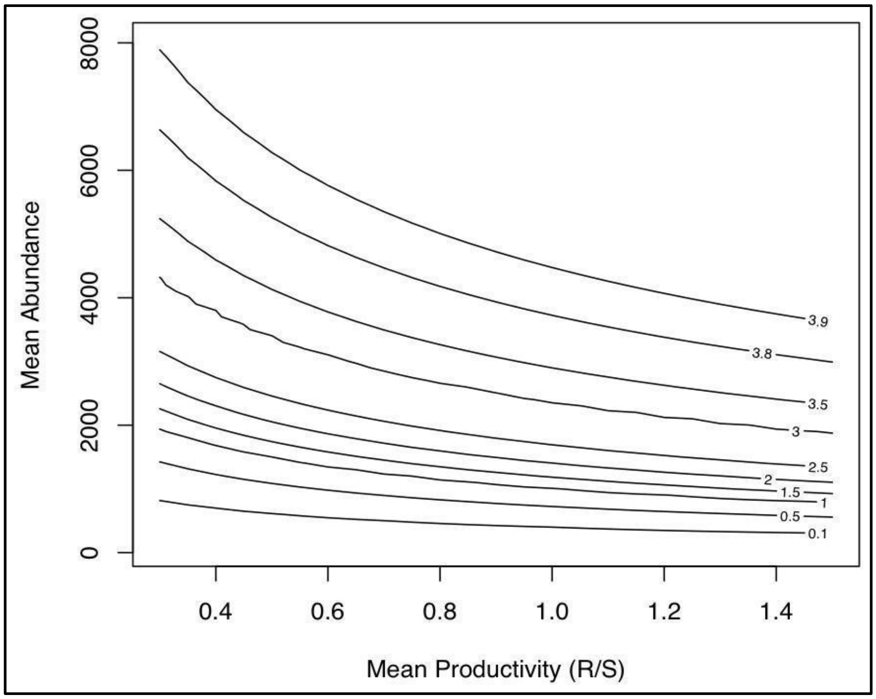


Figure 7-13. Response surface of VSP score for productivity and abundance versus mean productivity and abundance for the McKenzie spring Chinook population.

For a single alternative, we can then plot mean productivity and abundance for individual runs on the response surface, along with the grand mean for all runs (Figure 7-13).

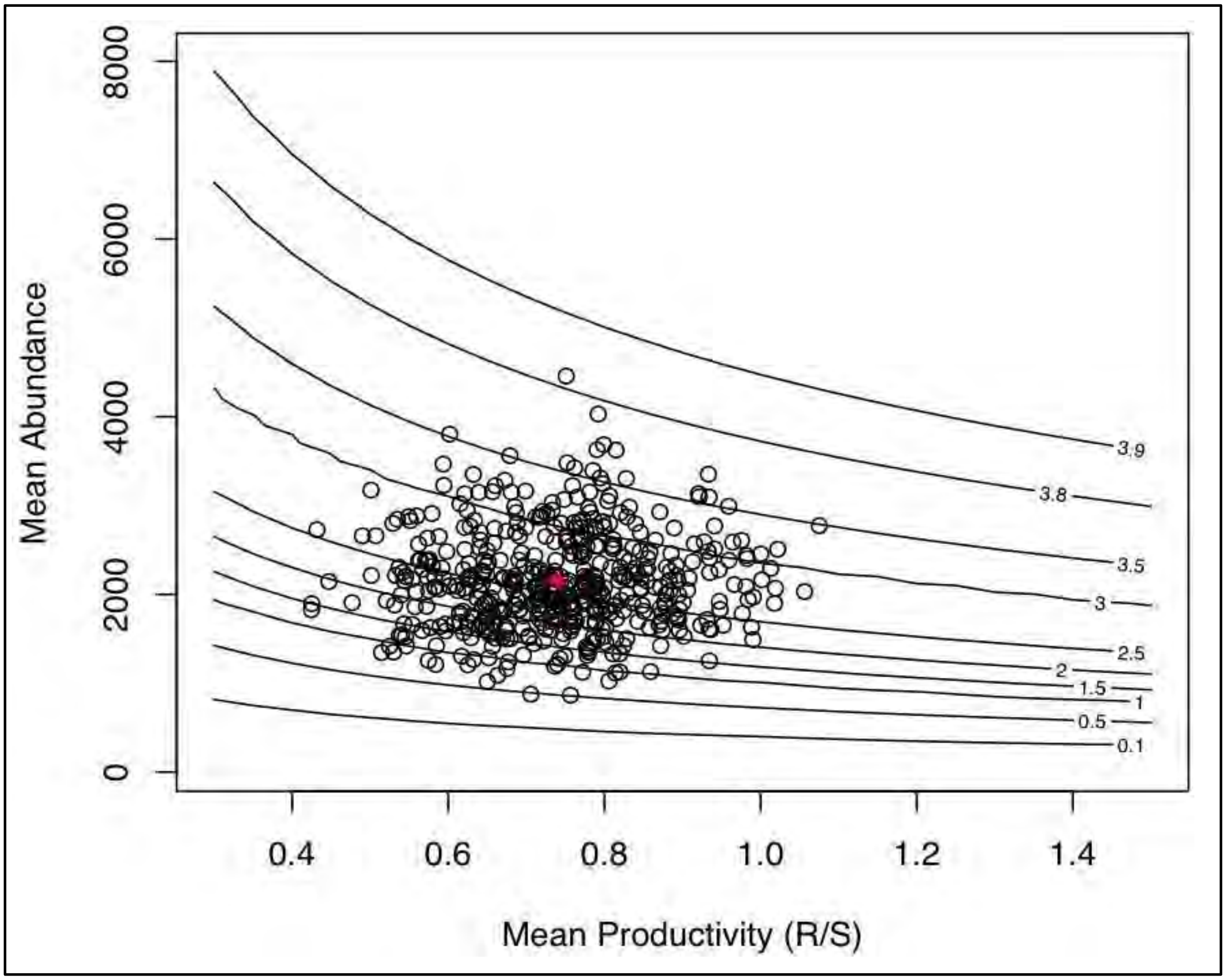


Figure 7-14. Abundance versus Productivity comparisons under baseline conditions*. Mean abundance versus mean productivity for individual model runs under baseline conditions. The solid red point is the grand mean across all runs. The isoclines represent VSP scores for productivity and abundance.*

Based on Figure 7-14, we can then determine a $VSP_{P\&A}$ score for each run, and then determine the distribution of these scores (Figure 7-15). From this distribution of scores, we can derive uncertainty measures (variance, confidence intervals) for each alternative, and we present the uncertainty in model outputs in the results presented in Section 7..

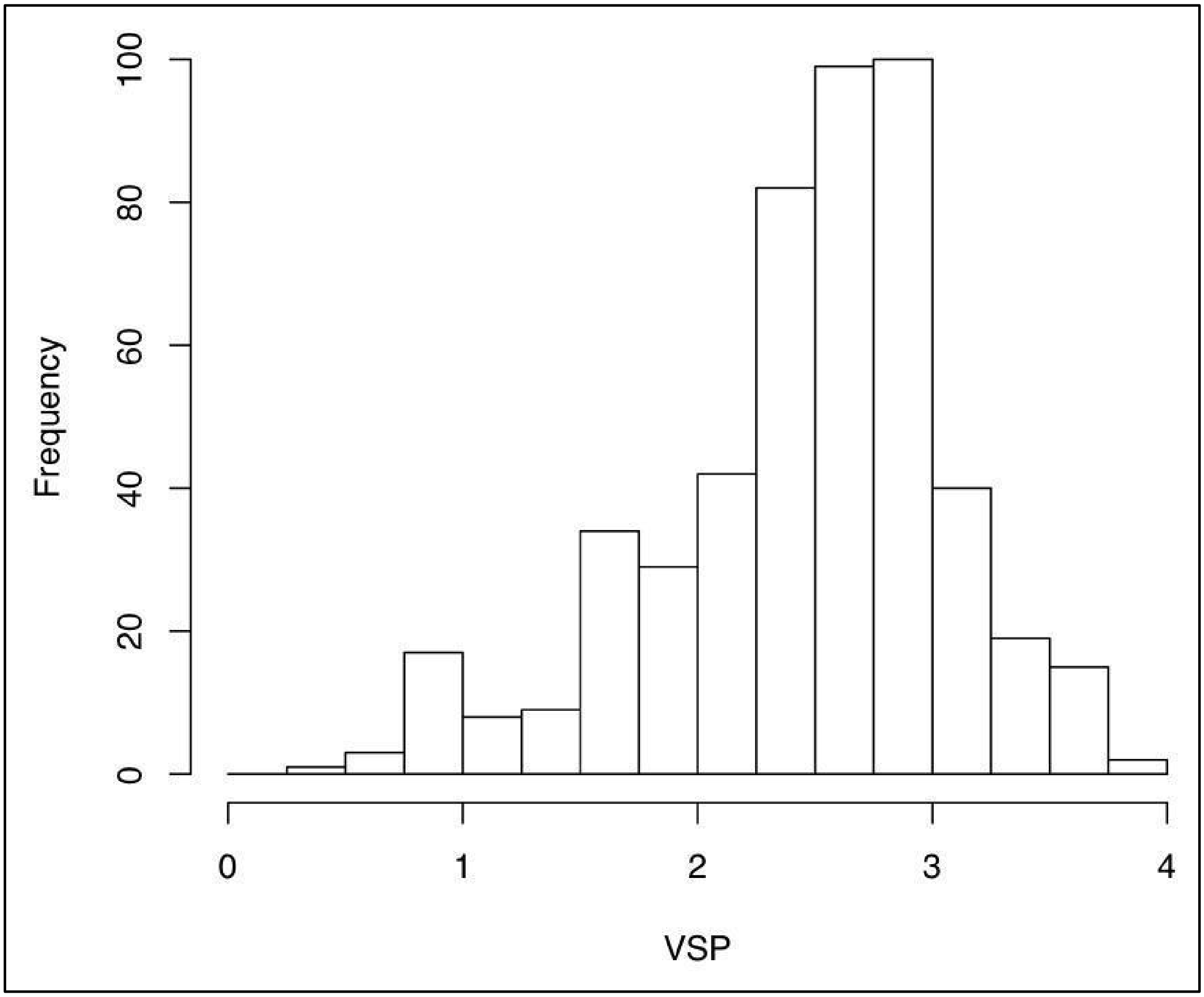


Figure 7-15. McKenzie River spring Chinook Baseline VSP productivity and Abundance Scores
Distribution of VSP scores (productivity and abundance) for McKenzie River spring Chinook under baseline conditions

7.1.9 Weighting of VSP Scores

We also calculate VSP scores for Diversity (VSP_D) and spatial structure (VSP_S) as described earlier in this chapter. To combine scores, we use the following weighting:

$$VSP_T = \frac{4 \cdot VSP_{P\&A} + VSP_D + VSP_S}{6}$$

Each VSP component has an associated variance. To estimate variance about the combined score, we use the following formula (Mood et al. 1974), and assume independence across components:

$$\mathrm{var}_T = \sum_{1}^{3} w_i^{\,2} \cdot \mathrm{var}_i$$

where varT is the total variance, wi is the weight associated with the $_i$th VSP component, and var_i is the variance associated with the $_i$th VSP component.

7.2 JUVENILE CAPACITY

7.2.1 Spawning/Incubation Habitat

In the development of the Willamette Chinook salmon and steelhead life cycle models, one of the key model parameters identified by sensitivity analysis was spawner capacity. In the original parameterization of the model (Zabel et al. 2015) we relied on reach specific estimates of spawner capacity from pre-dam habitat surveys in the 1930s and 1940s (see Parkhurst et al. 1950b, McIntosh et al. 1990). These surveys largely relied on quantification of suitable spawning gravel size, and stream depth and gradient. Although these surveys were extensive, we considered these estimates to represent a maximum number of spawners in a pre-dam stream condition and set capacities at 50% of the pre-dam estimates as a conservative estimate of habitat degradation since that time (Tables 2.1 and 2.2). More contemporary surveys by R2 Resource Consultants are considerably lower than the historical Parkhurst surveys where they overlap (R2 Resource Consultants Inc. 2009). Although we cannot quantify how much of the difference is due to habitat degradation or survey methodology, other modeling approaches have made estimates like our capacities using a blend of the historical and contemporary survey data (Bond et al. 2017, Myers et al. 2019).

7.2.2 Juvenile Rearing Habitat

Juvenile rearing capacity in the Willamette Basin is generally limited by summer rearing conditions: summer low flows and high temperatures. Bond et al. (2017) estimated Chinook salmon and winter steelhead summer parr capacity for the same stream reaches used in spawning habitat estimates. In that study reservoir rearing capacities were also estimated with a bioenergetic approach. Bond et al. estimated tributary habitat area with a model that predicted the total surface area of relevant rearing habitats in each reach and proportional area of banks, bars, and mid-channel habitats for each 200 m reach based on summer wetted widths. A similar model predicted the amount of side channel habitat available in each reach. Estimates of habitat- specific parr densities at capacity were applied to each habitat type and summed for each model reach (see tables 2.1 and 2.2), an approach which has been used in other watersheds (Bartz et al. 2006, Bond et al. 2019, Beechie et al. 2021).

Across all LCM reaches, Chinook salmon parr capacity was estimated at 15.4 million, with steelhead parr capacity being 5.9 million (Bond et al. 2017). Temperature, under current conditions, was limiting for only a few segments within the LCM reaches. Under the LCM alternatives examined, spawner abundance (both hatchery-origin and natural-origin spawners) rarely produced enough juveniles to have parr rearing capacity limit population growth.

Currently, only Chinook salmon are released from hatcheries in large numbers2[2], but these fish migrate relatively quickly out of the basin and the overall abundance is still below capacity estimates. Similarly, hatchery-reared summer-run steelhead are released in the basin, with little minimal effect on overall capacity3[3]. Only a few thousand hatchery-reared (surrogate program) winter steelhead are currently released in the Upper Willamette Basin. Based on these capacity estimates and the LCM parameters, we concluded that juvenile rearing capacity in tributaries and reservoirs was not a major limiting factor under current scenarios. Therefore, in accordance with our goal of balancing model complexity between the available data and alternative scenario evaluation, we decided not to include juvenile rearing capacity in the LCM for both Chinook salmon and steelhead.

Table 7-1. Basin and reach capacities (eggs) used in spring-run Chinook salmon models. *

Tributary	Reach	SLAM Reach	Mean Egg Capacity	Adult Spawners	R2 Adult Spawners
North Santiam River	Blw Bennett Dam	A	300,000	133	110
North Santiam River	Btwn Bennett and				1408
North Santiam River	MCF	B	5,000,000	2222	
North Santiam River	Blw Detroit	A,B,F	5,800,000	2578	1673
North Santiam River	Little North Santiam River	C	1,500,000	667	145
North Santiam River	Breitenbush River	D	6,000,000	2667	1515
North Santiam River	Upper North Santiam	E	12,500,000	5556	1313
North Santiam River	MCF to Detroit Dam	F	500,000	222	155
South Santiam River	Blw Lebanon Dam	A	150,000	67	93
South Santiam River	Thomas/Crabtree				508
South Santiam River	Creek	B	1,500,000	667	
South Santiam River	Lebanon to Foster				2120
South Santiam River	Dam	C	4,000,000	1778	
South Santiam River	Wiley Creek	D	500,000	222	58
South Santiam River	South Santiam above				
South Santiam River	Foster Dam	E	7,000,000	3111	3590
South Santiam River	South Santiam	A, B, C, D, E	12,150,000	5400	
South Santiam River	Quartzville Creek	F	2,000,000	889	368
South Santiam River	Middle Santiam River	G	4,500,000	2000	953

[2] Current production in the Upper Willamette Basin (not including the Clackamas River), 4.6 million spring-run Chinook salmon and 0.5 million summer-run steelhead.
[3] Releases of hatchery-reared juvenile Chinook salmon and steelhead in mass, may result in localized high densities, but the overall effect of these releases on productivity is likely minimal.

Tributary	Reach	SLAM Reach	Mean Egg Capacity	Adult Spawners	R2 Adult Spawners
McKenzie River	Blw Leaburg Dam	A	1,250,000	556	370
McKenzie River	Above Leaburg Dam	B	8,000,000	3556	2083
McKenzie River	Above Hendrick's Bridge	A,B	9,250,000	4111	2453
McKenzie River	Above Cougar Dam	C	17,000,000	7556	1790
Middle Fork Willamette River	Below Dexter Dam	A	150,000	67	220
Middle Fork Willamette River	Lower Fall Creek	B	500,000	222	523
Middle Fork Willamette River	Upper Fall Creek	C	2,000,000	889	1025
Middle Fork Willamette River	Fall Creek	B,C	2,500,000	1111	1548
Middle Fork Willamette River	Above Dexter/Lookout				
Middle Fork Willamette River	Pt Dams	D	20,000,000	8889	728
Middle Fork Willamette River	Above Hills Creek	E	10,000,000	4444	Na

Notes: * Egg capacity was based on approximately 50% of the estimated historical adult spawning capacity (e.g. Parkhurst et al. 1950) with 2250 eggs/adult.

R2 spawner estimates were based on habitat surveys (R2 2009).

Table 7-2. Basin and reach capacities (eggs) used in steelhead models

Tributary	Reach	SLAM Reach	Mean Capacity (eggs)	Adult Spawners (@ 2250 eggs)
North Santiam River	Blw Bennett Dam	A	50,000	22
North Santiam River	Btwn Bennett and MCF	B	3,000,000	1333
North Santiam River	Blw Detroit	A,B,F	3,300,000	1467
North Santiam River	Little North Santiam River	C	2,500,000	1111
North Santiam River	Breitenbush River	D	4,000,000	1778
North Santiam River	Upper North Santiam	E	10,000,000	4444
North Santiam River	MCF to Detroit Dam	F	250,000	111
South Santiam River	Blw Lebanon Dam	A	50,000	22
South Santiam River	Thomas/Crabtree Creek	B	1,500,000	667
South Santiam River	Lebanon to Foster Dam	C	2,500,000	1111

Tributary	Reach	SLAM Reach	Mean Capacity (eggs)	Adult Spawners (@ 2250 eggs)
South Santiam River	Wiley Creek	D	1,500,000	667
South Santiam River	S. Santiam above Foster Dam	E	5,000,000	2222
South Santiam River	South Santiam	A,B,C,D, E	11,050,000	4911
South Santiam River	Quartzville Creek	F	2,000,000	889
South Santiam River	Middle Santiam River	G	4,500,000	2000

Notes: Basin-wide egg capacity was based on Chinook reach capacity (50% of Parkhurst et al. 1950 estimates) and adjusted to a relative historical steelhead:Chinook salmon ratio (80% Smith, 1898). Within each basin allocation to specific reaches was based on observed redd distribution.

7.3 FISH BENEFIT WORKBOOK.

The Fish Benefit workbook was adapted from the Fish Benefits Workbooks User Guide (USACE 2014)

7.3.1 Fish Benefit Workbook Overview.

This section is provided to give a brief overview of the development of the Fish Benefit Workbook (FBW), and how the application of the FBW to the life cycle model influences the viability of the populations modeled. For more information, reader should refer to the User Guide (USACE 2014), the COP (USACE 2015) or the ISAB review of the FBW (Naiman 2014). Simply, the FBW is the major factor influencing the relative outcomes of each of the alternatives for each population. Other factors such as water year or ocean conditions also have a large effect on the model outcome, but these factors generally apply equally across the alternatives.

The fish benefit workbook was developed by the USACE (2014) as a method of estimating the passage route, efficiency, and survival of fish passing through a dam, weir, and/or collection structure. In developing our life cycle models, we incorporate the alternative-specific output from the FBW into the model. For each of three juvenile life-history types, FBW output provides a month-by-month summary of the proportion of fish that entered the forebay and were able to pass (Dam Passage Efficiency, DPE), and the survival of those fish that attempt to pass and were able to (Dam Passage Survival, DPS) (See example Figure 3.2). Dam structure and reservoir elevation may lead to multiple routes for passage through the dam. The effectiveness of each route depends on the amount of flow going through each route and the physical configuration of the route structure (depth from forebay surface, size of opening etc.) combined with the chance of survival with passage. Dam Passage Efficiency and DPS estimates are informed by passage studies at Upper Willamette Basin projects and surveys of passage efficiencies at other dams (e.g., Kock et al. 2019 for passage structures).

For each of the hydrologic years of record, HEC-ResSim (a hydrological model developed by the USACE[4]) calculates the elevation of the reservoir and the flows into, and out of the reservoir, and then using the conditions of the specific alternative, directs fish to the various passage routes.

Monthly DPE and DPS estimates for each life history type are weighted by the estimated monthly distribution of juveniles entering the forebay (attempting to pass) and the relative allocation to different routes (e.g., turbines, Regulating Outlets, etc.), and combined into a single DPE and DPS for each water year (Table 6-13). Where available, the distribution of juveniles at each life history stage approaching the dam is estimated from juvenile sampling (normally by rotary screw trap (RST)) at or near the head of reservoir and project tailrace. Across the passage alternatives at each project there can be significant differences in the DPE and DPS values for each juvenile life history type attempting to pass (see Appendix 1 for Alternative values).

[4] https://www.hec.usace.army.mil/software/hec-ressim/documentation.aspx

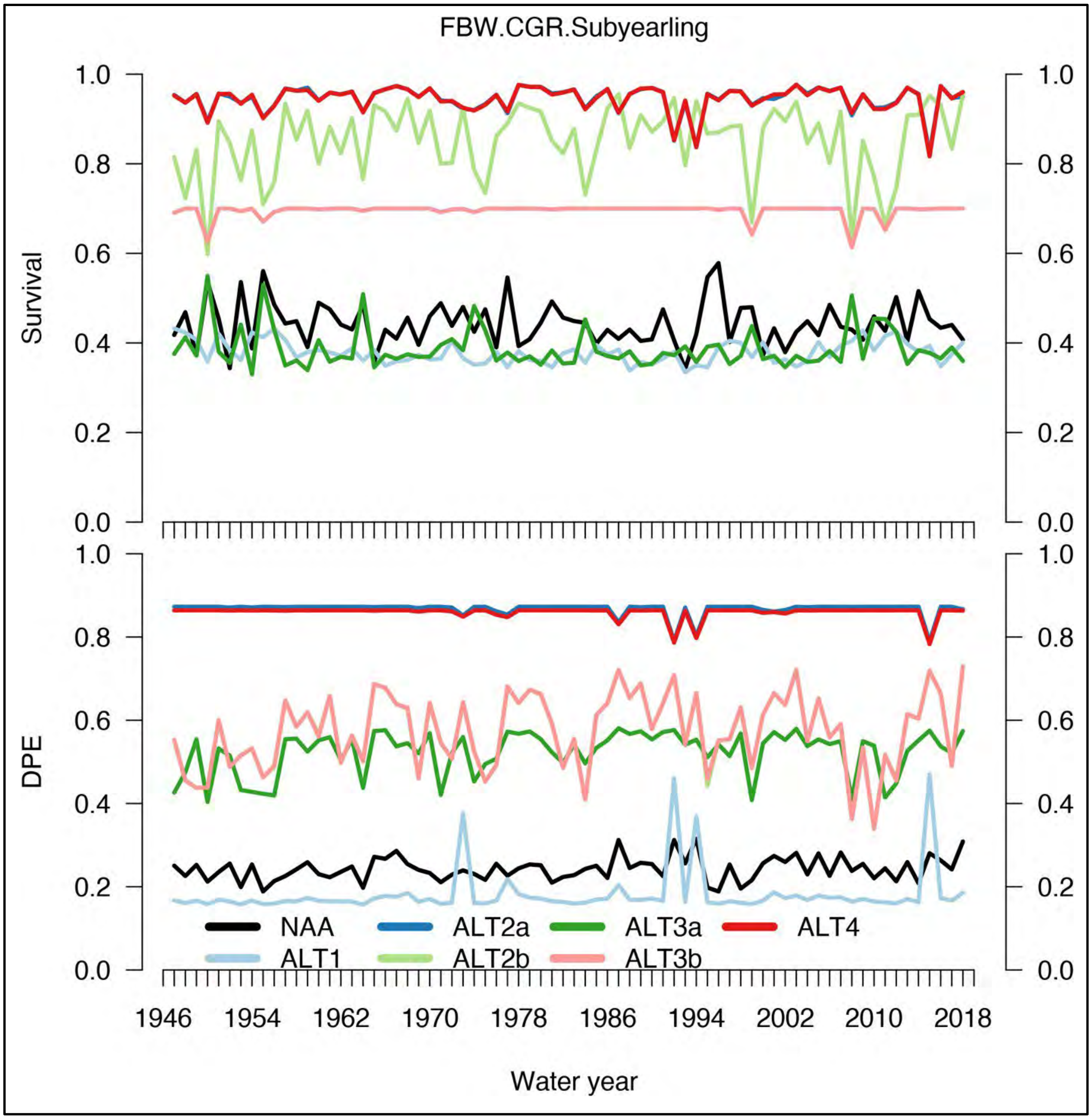


Figure 7-16. Cougar Dam Passage Survival (upper chart) and dam passage efficiency (lower chart) for fall subyearling Chinook salmon. *Dam passage survival (DPS) (above) and dam passage efficiency (DPE) (below) for fall subyearling Chinook salmon attempting to pass Cougar Dam under various passage alternatives. NAA – No action alternative (black line), Alternative 1 (Alt 1, light blue) – no passage modification, Alt 2a -floating screen structure (royal blue)_, Alt 2b – spring drawdown, fall drawdown to diversion tunnel (light green), Alt 3a – spring drawdown, fall drawdown to regulating outlet (RO, darker green), Alt 3b -- spring drawdown, fall drawdown to diversion tunne (pink), Alt 4 – floating screen structure (red).*

7.3.2 FBW and life history trajectories

The FBWs only attempt to quantify the survival rates of juveniles from the forebay to the tailrace. The tool does not account for any mortality outside this specific area (e.g., reservoir predation). For projects with re-regulating dams (Detroit, Lookout Point), the downstream extent of the FBW extended to the tailrace of the re-regulating dam. A fixed mortality rate was defined for the re-regulating dam that included the effects of re-regulation dam passage and predation in the re-regulating pool. The FBWs computed passage-route survivals, concrete survivals, and dam passage survivals for each day of the historic period of record, using simulated reservoir operations as input.

The FBWs produced survival estimates for multiple life-stages passing the dams (see example Table 3.1). The FBWs had three life stages for Chinook (fry, subyearlings, yearlings), and three life stages for steelhead (subyearlings, yearlings, age-2).

The FBWs tracked the portion of each life history cohort which are available to pass the dam, but do not pass; this is reported as “1-DPE”. The “1-DPE” value is passed to the life-cycle model for each life history cohort for incorporation into the life history outmigration patterns throughout the period of record. The life cycle models tracked the cohorts such that those not passing (for a given alternative and water year, proportion of a given life stage not passing = 1 – DPE) sustained a reservoir mortality and joined the pool of the next life history and attempt to pass in the next life history stage (Figure 3.2). For example, fry not passing and surviving to the fall subyearling stage are available to pass as fall subyearlings. The FBW routes each life stage cohort according to the month and the water year condition, resulting in a wide range of flow/operational conditions (e.g., survivals and DPEs for fall subyearlings, see Figure 6-16).

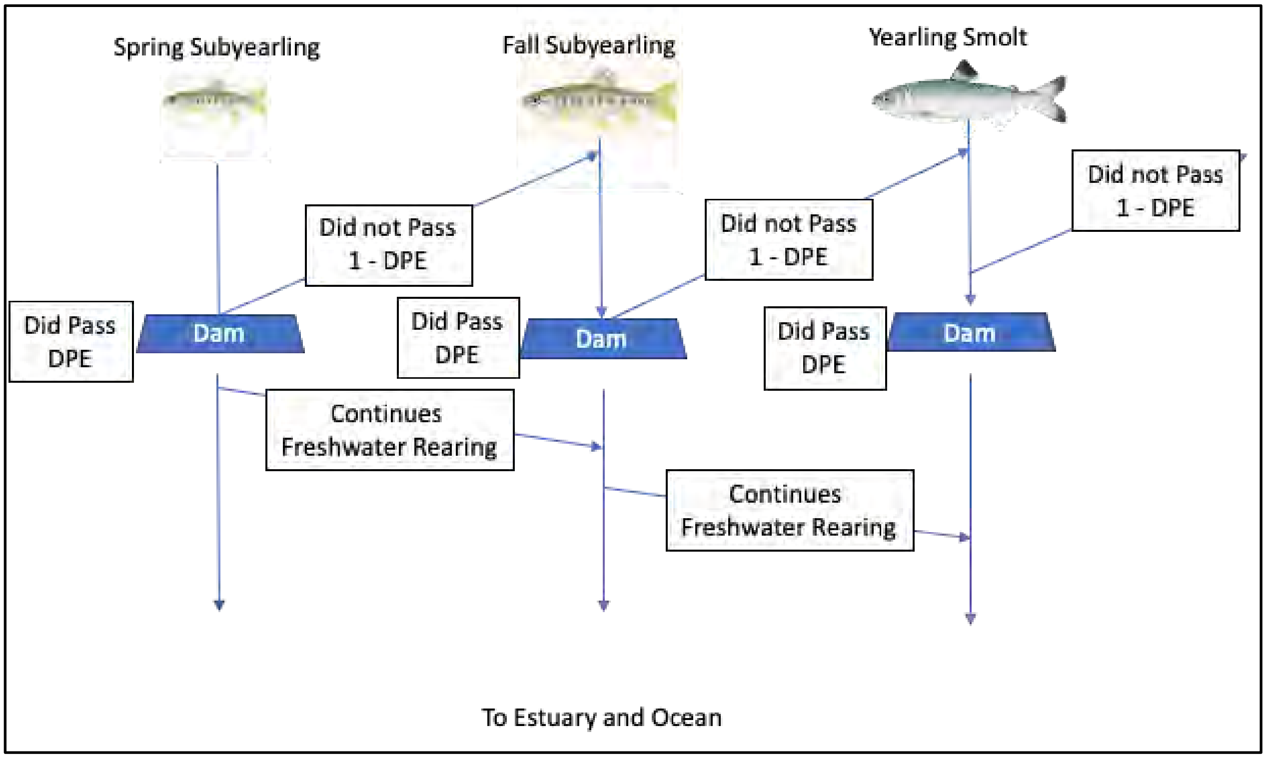


Figure 7-17. Juvenile Salmonid Downstream Emigration Through a Dam to the Ocean.
Figure 6-16 is a generalized schematic of life history trajectories for juvenile salmonids attempting to emigrate downstream through a dam and migrate to the ocean. Dam passage efficiency (DPE) represents the proportion of juveniles entering the dam forebay that successfully pass downstream. Those fish that do not emigrate at the yearling smolt stage are removed from the model.

7.3.3 FBW applied to the Life History Model

There were separate FBWs for each species (with three life histories represented in each) that were unique for each of the alternatives, making 42 FBWs in total for alternatives in all basins. The mechanics of the FBW computations work the same way for all species/ages. However, the parameters governing the FBW simulations are quite different for alternate species/ages, leading to different results (Figure 6-16).

FBW calculates its output in a seven-step process:

Step 1. Calculate fish passage structure flow
Step 2. Distribute fish temporally
Step 3. Apply Dam Passage Efficiency
Step 4. Distribute fish between conduits
Step 5. Calculate and apply passage-route survivals
Step 6. Apply re-regulating dam effects
Step 7. Roll-up daily results into summaries

FBW results were directly integrated into the life cycle model (i.e., as in Figure 3.1), and the secondary and tertiary effects for each alternative (e.g., water temperature changes as a consequence of the alternatives, and TDG effects on passing juveniles and developing alevins) were also considered. For example, operational passage alternatives (providing spill or regulating outlet flow) routed fish passing the dam to the tailrace of the project, where gas supersaturation could occur; whereas, with floating screen structures fish could be collected and transported downstream to a “safer” release location. The downstream effects of each alternative (temperature, flow, TDG) are discussed elsewhere (water temperatures in Chapter 6.8, TDG effects in Chapter 6.4)

.

Table 7-3. Yearling Chinook Salmon Detroit Dam Passage Statistics under Alt 2. *

Month	% Fish Approaching Baseline[1]	% Fish Approaching Estimated with Downstream Passage[2]	Average % Fish Distribution Lost to Forebay	Average % Fish Distribution Fish Passage Structure	Average % Fish Distribution Turbine	Average % Fish Distribution RO	Average % Fish Distribution Spillway	Average % Route Survival Fish Passage Structure	Average % Route Survival Powerhouse	Average % Route Survival RO	Average % Route Survival Spillway
September	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
October	0.1%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
November	1.9%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
December	1.1%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
January	45.9%	40.0%	6.0%	31.5%	2.0%	0.1%	0.4%	30.9%	1.0%	0.1%	0.2%
February	11.3%	30.0%	4.5%	24.1%	1.0%	0.1%	0.3%	23.6%	0.5%	0.0%	0.2%
March	8.9%	20.0%	3.0%	16.6%	0.1%	0.0%	0.3%	16.3%	0.0%	0.0%	0.2%
April	14.5%	10.0%	1.5%	8.1%	0.2%	0.0%	0.2%	7.9%	0.1%	0.0%	0.2%
May	10.9%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
June	4.8%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
July	0.4%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
August	0.2%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
Total	100.0%	100.0%	15.0%	80.4%	3.2%	0.1%	1.2%	78.7%	1.6%	0.1%	0.9%

Notes: [1]. Distribution of fish approaching the dam for passage on a monthly basis estimated from ODFW proposed timing 13SEP2012.
Resulting Average Project Survival (FBW) 85.0% x 95.7% = 81.3%

[2]. Estimated distribution of fish approaching the dam for passage on a monthly basis with effective fish passage operations or structure in place. ODFW 21AUG2012.
Average DPE = 85.0%; w/ Ave Survival of = 95.7%

Table 7-3 Fish Benefit Workbook (FBW) example shows passage efficiency, survival, and timing for yearling Chinook salmon attempting to pass Detroit Dam (North Santiam River) under Alternative 2 (floating screen structure). Similar tables exist for the spring subyearling and fall subyearling juvenile life histories. FBW elements include timing of fish entering forebay (yellow), distribution of routes taken by fish each month (orange), and survival of fish through each of those routes for each month (green). “Lost to forebay” in distribution denotes fish remaining in forebay (reservoir). Finally, a weighted passage efficiency and survival are estimated (blue), these parameters are entered into the life cycle model.

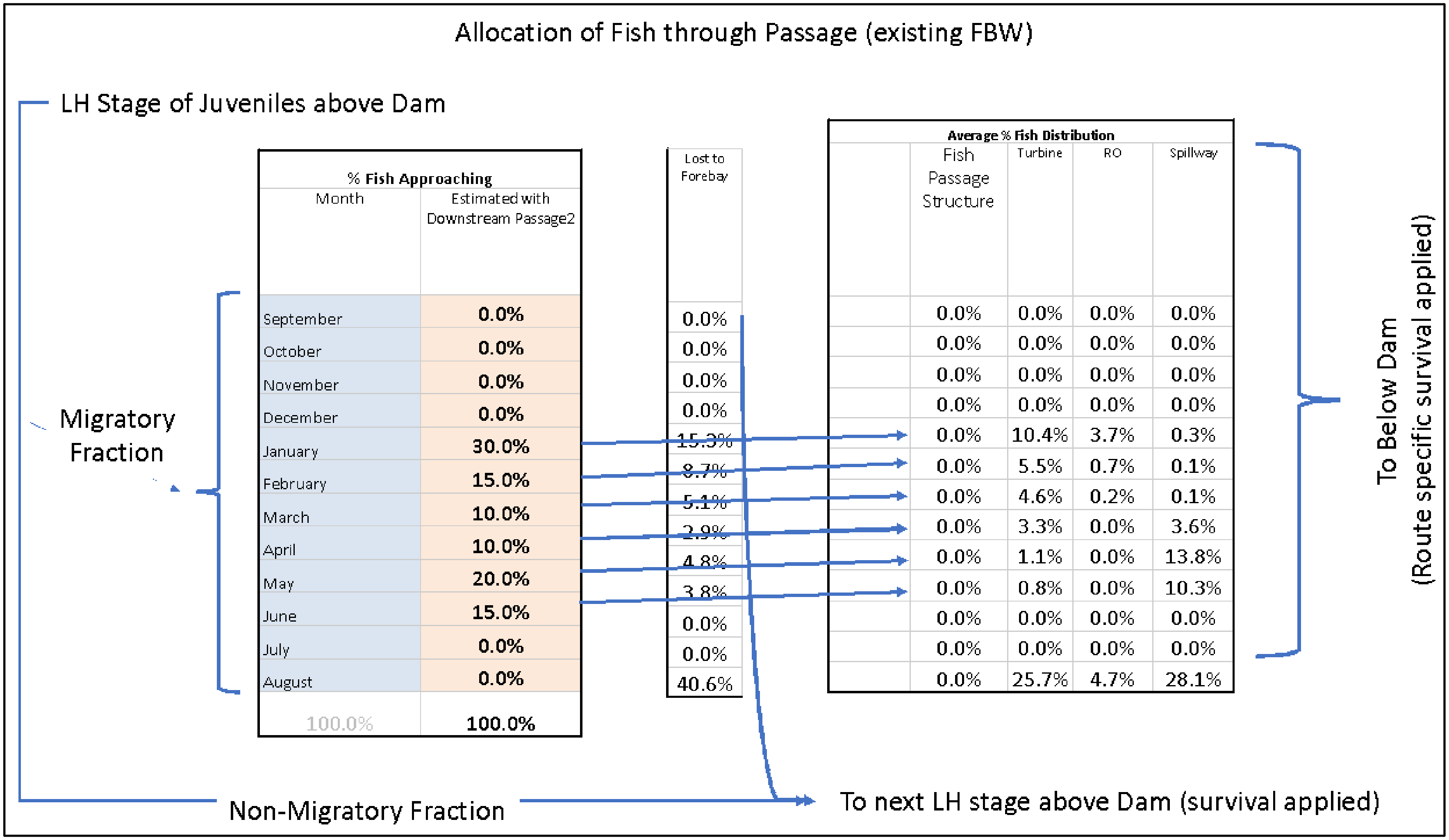
7.3.4 Fish passage routing

For any given juvenile passing the dams, the probability of passing downstream is dependent on the historical water year being considered (i.e., the amount of water available), and how that water is passed through the dam (both the timing and absolute and relative flow and route). The effectiveness of the routes, in turn, is dependent on the relative depth of the route entrance and the characteristics of the route itself (penstock, regulating outlet, fish horn, spillway, etc). These determinations are set by the USACE, based on the specific alternative.

Distribution of fish temporally and DPE

The FBW assigns each juvenile life history a temporal (monthly) distribution for entering the dam forebay. This distribution is based on juvenile trap studies (see review by Hansen et al. 2017). In general, for passage operations that implement reservoir drawdowns for passage the distribution is similar to the timing for juveniles entering the reservoir, while operations with full reservoirs (spill-based passage and passage structures) are assigned distributions more similar to juvenile capture timing at the tailrace in rotary screwtraps.

Having temporally apportioned juveniles to approach the forebay, juveniles are then apportioned to different passage routes (according to flow). At this point, the passage efficiency of each route is calculated. The collective measure of passage efficiency is expressed as dam passage efficiency (DPE). In the life cycle models, if a fish of a given life history attempts to pass but does not pass (“lost to forebay”), it returns to the reservoir and may attempt to pass in the next life history stage (Figure 3.3). Juveniles that are “lost to forebay” are also subject to a life history stage-based mortality in the LCMs before being able to attempt to pass again. A scenario not currently considered by the FBW would include a juvenile that attempted to pass in a given month, but did not, would try again within the same life history stage at the next opportunity, the next month. This would especially apply to operational passage alternatives where reservoir elevations can change dramatically from month to month. In the process of drawing down or raising the reservoir elevation to arrive at optimal passage conditions, some time is spent at elevations with poor passage conditions and the FBW did not appear to accommodate the potential for fish available to pass but in a holding pattern for the right conditions and passage options to pass. The FBW averages daily reservoir conditions to arrive at a single monthly estimate, so that the overall DPE is likely lower than optimal.



Month	% Fish Approaching: Estimated with Downstream Passage2	Lost to Forebay	Fish Passage Structure	Turbine	RO	Spillway
September	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
October	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
November	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
December	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
January	30.0%	15.3%	0.0%	10.4%	3.7%	0.3%
February	15.0%	8.7%	0.0%	5.5%	0.7%	0.1%
March	10.0%	5.1%	0.0%	4.6%	0.2%	0.1%
April	10.0%	2.9%	0.0%	3.3%	0.0%	3.6%
May	20.0%	4.8%	0.0%	1.1%	0.0%	13.8%
June	15.0%	3.8%	0.0%	0.8%	0.0%	10.3%
July	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
August	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
100.0%	100.0%	40.6%	0.0%	25.7%	4.7%	28.1%

Figure 7-18. Juvenile fish Movements over time from the forebay “ready to pass” to their relative fates.

Modeled movement of juvenile fish, showing the relative temporal distribution of fish at the forebay “ready to pass”, and the subsequent relative fate of those fish. In general, fish that are ready to pass, either pass through via a specific route (fish passage structure, turbine, regulating outlet (RO), or spillway), or do not pass (“lost to forebay”). In the life cycle models, those fish that did not pass, remained in the reservoir until the next life history stage (a proportion of which suffered a reservoir rearing loss).

Thus, the FBW allocated fish passage routes when conditions are not optimal in the operational alternatives. In another scenario, during a complete drawdown the FBW model had fish attempting to pass (and failing) during the drawdown (Alternative 2b or 3b in the McKenzie River) which could take up to two months or more. Under the existing FBW model, fish not passing would had to "wait" until the next life history stage to pass, whereas, realistically once the reservoir was at riverbed it is likely that the vast majority of fish available to pass would have passed. Similarly, if a reservoir was being filled, those fish that attempted to pass prior to the reservoir reaching the spillway crest would likely have passed at low efficiency through the RO or turbine and had to wait under the existing FBW model but would likely have passed once the spillway route was open. This DPE issue appears to disproportionately affect operational passage alternatives. Passage structures (floating screen structures) generally provided high efficiency passage on a continuous pass and were less affected by the "lost to forebay" effect.

One solution to this fixed juvenile approach would be to return those fish lost to the forebay to the pool of fish attempting to pass the next month (Figure 3.4).

Considerations in assessing LCM results and the FBW

We were not tasked with developing an alternative FBW, but it is useful to consider the LCM results in light of a potential bias in comparing structural vs operational passage alternatives. This bias is especially of concern if juveniles were effectively forced to pass through suboptimal routes, or if juveniles were artificially "delayed" by poor passage efficiency during the temporal window to which they were allocated, or if dam passage survivals were significantly different for different life history stages. A further consideration is the relative benefit of reservoir rearing compared to passage and downstream (tributary, mainstem Willamette River, Columbia River, estuary) rearing. While there is considerable uncertainty in the benefits of reservoir rearing, the LCMs contain input variables to accommodate differential survivals for different trajectories.

Alternatively, we could modify the proportion of each juvenile life history type that is available to pass (these fish subsequently attempt to pass) or explore possible life histories that remain as residents above the dams. However, this could result in a shift in juvenile life history types not observed on site. Ultimately, this and many other LCM issues relate to the behavior of juveniles in reservoirs and the initiation of downstream migratory behavior, in the absence of relevant empirical data there will be considerable uncertainty in these areas.

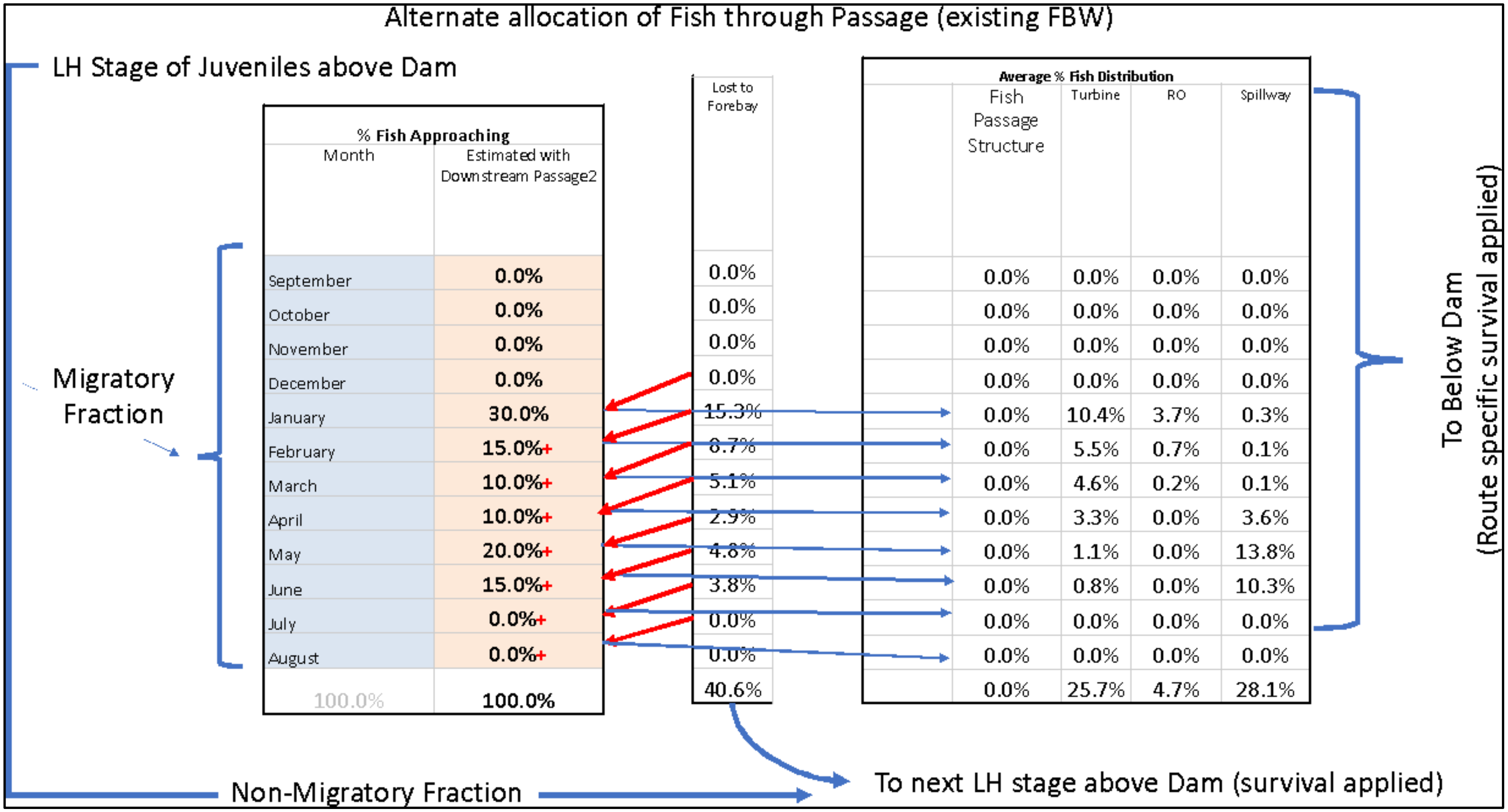


Figure 7-19. An alternative model for the movement of juvenile fish.

In Figure 6-18 case fish lost to the forebay do not wait until their next life history stage to attempt to pass downstream of the project but are returned to the pool of fish approaching the next month (red arrows) and remain as “active migrants” taking advantage of subsequent passage conditions.

7.4 JUVENILE POST-DAM PASSAGE

7.4.1 Total Dissolved Gas Related Mortalities

Both juvenile and adult Chinook salmon and steelhead are subjected to the effects of gas supersaturation. For juveniles, water conditions downstream of dams becomes supersaturated by water flowing over the spillway or water released under pressure from regulating outlets5[5].

Juveniles that pass through the dam outlets will be subjected downstream to any gas supersaturation created by dam operations. In addition, eggs, alevins, and juveniles that were incubating or rearing downstream are also subject to gas supersaturation. The downstream extent of any supersaturation created by dam operations depends on the topography of the river (rapids tend to releases gas). Similarly, adults moving upstream congregate near the base of some dams prior to adult collection (as at Foster, Cougar, Fall Creek, or Dexter Dam) or spawning. The duration of exposure varies considerably. In most cases, adults are not present during periods of high total dissolved gas (TDG).

Total dissolved gas mortalities

Estimates of total dissolved gas (TDG) levels generated by proposed passage alternatives at Upper Willamette River projects were estimated by the USACE and utilized to calculate gas bubble disease mortalities in juvenile Chinook salmon and steelhead and incorporated into life cycle models (Zabel et al. 2015). The relationship between TDG levels and mortality for different life history stages (alevin, subyearling juvenile, smolt) was initially developed by R2 Consultants (Hendrix et al. 2012) using experimental supersaturation exposure data from Rucker and Kangas (1974).

There were two applications of total dissolved gas (TDG) mortalities in the life cycle models: to developing alevins and to fishes passing the dams.

Alevins below the dams

For developing eggs and alevins in redds below dams, a mortality associated with TDG was applied that relied on prior work by R2 Consultants. Monthly mortality equation for developing alevins is

$$M_{TDGAlevin} = 1/(1 + e^{-(-26.04 + 0.058 * Days + 0.19 * TDG)}),$$

where Days is the number of days at or exceeding the standard, and TDG is the mean of daily mean TDG for a particular month. TDG estimates, for each dam and each alternative, were derived by the USACE daily through ResSim simulations for water years 1936 – 2019. The TDG

[5] During releases through the RO air is often drawn in through vents, this has the side effect of dramatically raising the percentage of total dissolved gas (pers comm Kathryn Tackley, USACE).

mortality approach consisted of counting the number of days per month at which the TDG level was at or above a standard level of supersaturation, which is set at 115%6[6]. If a month contained any days at or above the standard, a monthly mortality (using the R2 equation below) for a given month was calculated based on the entire month's average TDG levels. If no days in a month reach or exceed the standard, no monthly TDG mortality was calculated (hence, zero TDG mortality for that month). The total mortality applied to developing juveniles consisted of a weighted average of monthly mortalities (weights = 0.25 [December], 0.50 [January], 0.25 [February]) for Chinook salmon. This weighting roughly aligns with the relative emergence timing of Chinook salmon eggs.

Juveniles passing dams

Monthly mortality for juveniles passing dams is:

$$M_{TDG\,Juveniles} = 1/(1 + e^{-(-38.88 + 0.087*Days + 0.29*TDG)}).$$

The mortality applied to fishes was specific to each of the life stages that passed the dam. Also, we used as monthly weights the proportions of fishes approaching a dam each month, and they were specific for each life stage passing the dams. The proportions of fishes approaching the dams were gathered from the Fish Benefit Workbook. Also, the weights (i.e., proportions of fishes approaching) were specific for each alternative and for each dam.

Preliminary simulations using the most recent version of the life cycle model with updated TDG estimates suggested that mortalities of juveniles passing dams due to gas bubble disease could produce elevated mortalities, exceeding 40% in some cases (primarily for alternatives with high surface spill flows) using the R2-developed models. Recent evidence from screw trap monitoring below Big Cliff Dam in 2021 suggests that the observed incidence of gas bubble disease was relatively low, even during periods when levels exceeded 115-120% supersaturation (Flaherty 2021). There are a number of factors that would suggest TDG mortalities are not as severe as would be suggested by the experimental studies. Firstly, most studies were done in shallow tanks, and the potential benefits of depth depensation (roughly for each meter in depth effective TDG decreased by 10%), and the duration of exposure could vary considerably within each month. Juvenile movement and TDG mortality were both calculated on a monthly basis, but exposure would not be uniform and depending on conditions only a proportion of the fish would be exposed to hazardous conditions during that time. Previous monitoring work suggest that the median travel time for Chinook salmon and steelhead juveniles in the springtime from Detroit Dam to Minto Dam is a day (Figure 4.1).

[6] The water quality standard for the State of Oregon is 110% (OAR 340-41-0031), although this limit has been amended (125%) for mainstem Columbia River Dams during spring-spill to facilitate downstream juvenile salmon passage.

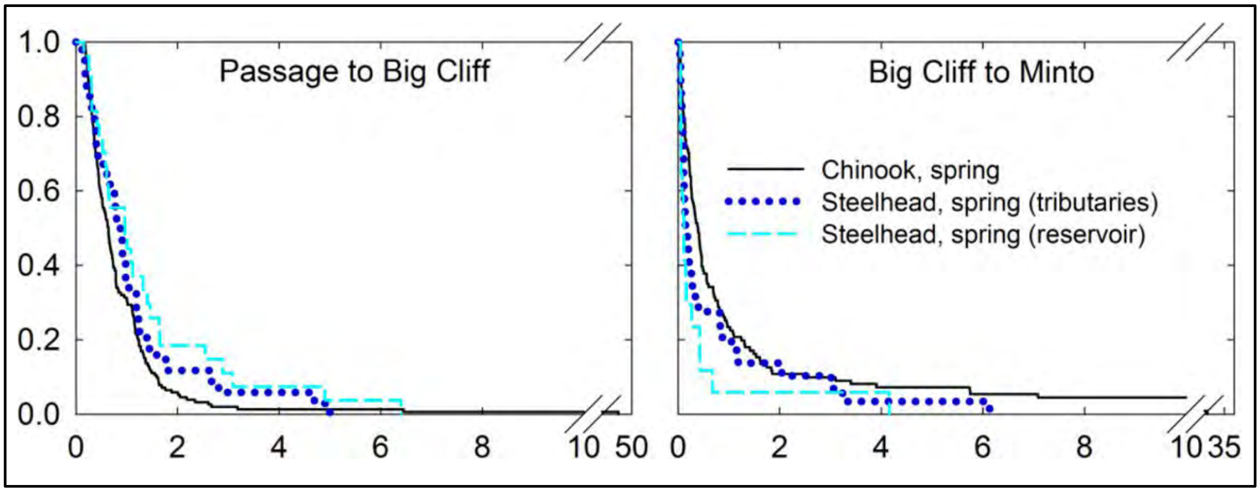


Figure 7-20. Travel days from Detroit Reservoir to Big Cliff (2013) and Big Cliff to Minto (2015)
Graphs showing travel time from the last detection at an upstream site to the first detection at the next downstream site for fish released in the spring at Detroit Reservoir, 2013. Note the different x-axis on the Passage to Big Cliff plot (From Beeman and Adams 2015, Figure 1-31).

If exposure times for fish passing through the dams is on the order of a day or two for emigrating juveniles, then the potential for mortality is greatly diminished compared to the R2 model predictions. More recent studies by Maule et al. (1995) suggest that at 120% TDG mortality doesn't increase until after 48 hours of exposure (Figure 6-20). The fish were exposed to this TDG level in relatively shallow tanks.

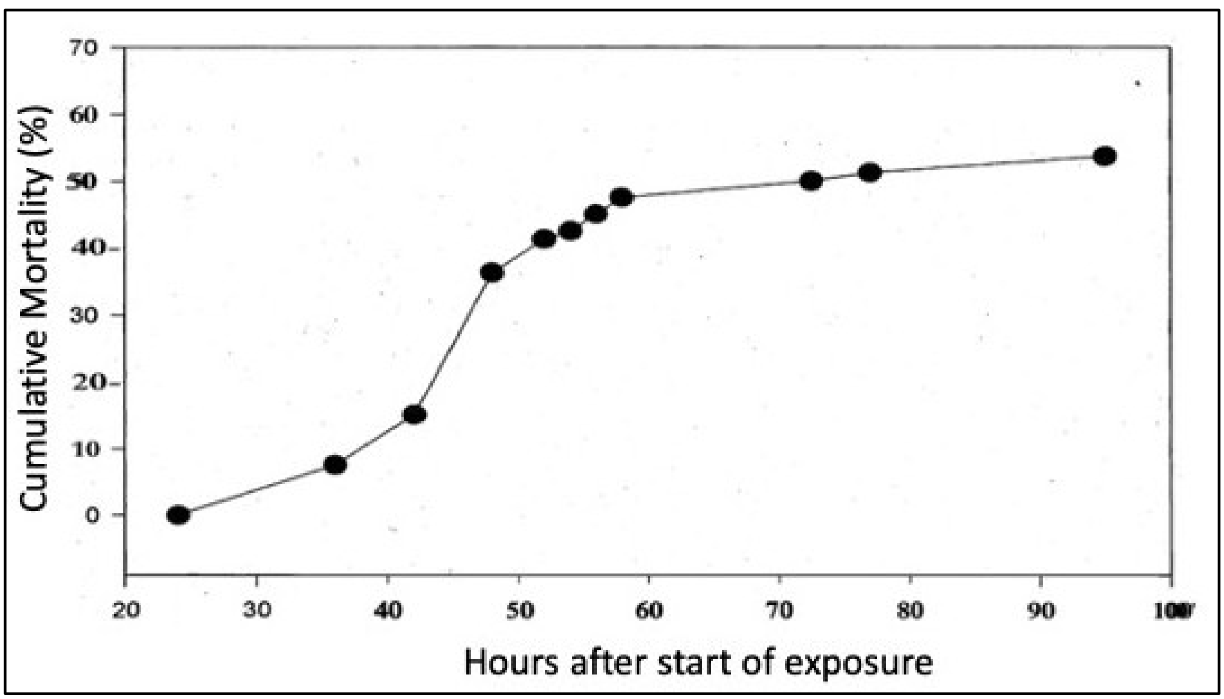


Figure 7-21. Juvenile Chinook Salmon Cumulative Mortality with 120% TDG at 12°C Exposure.
Total number of fish exposed was 75. Water depth was 28 cm. (Figure from Maule et al. 1995)

Furthermore, high mortality levels were not exhibited until TDG levels were elevated to 130% (Figure 6-21), a level very seldom reached in very limited circumstances in the alternatives (for example, below Big Cliff Dam in Alternative 3). It should be noted the precipitous increase in mortality with exposure increased from 120 to 130%.

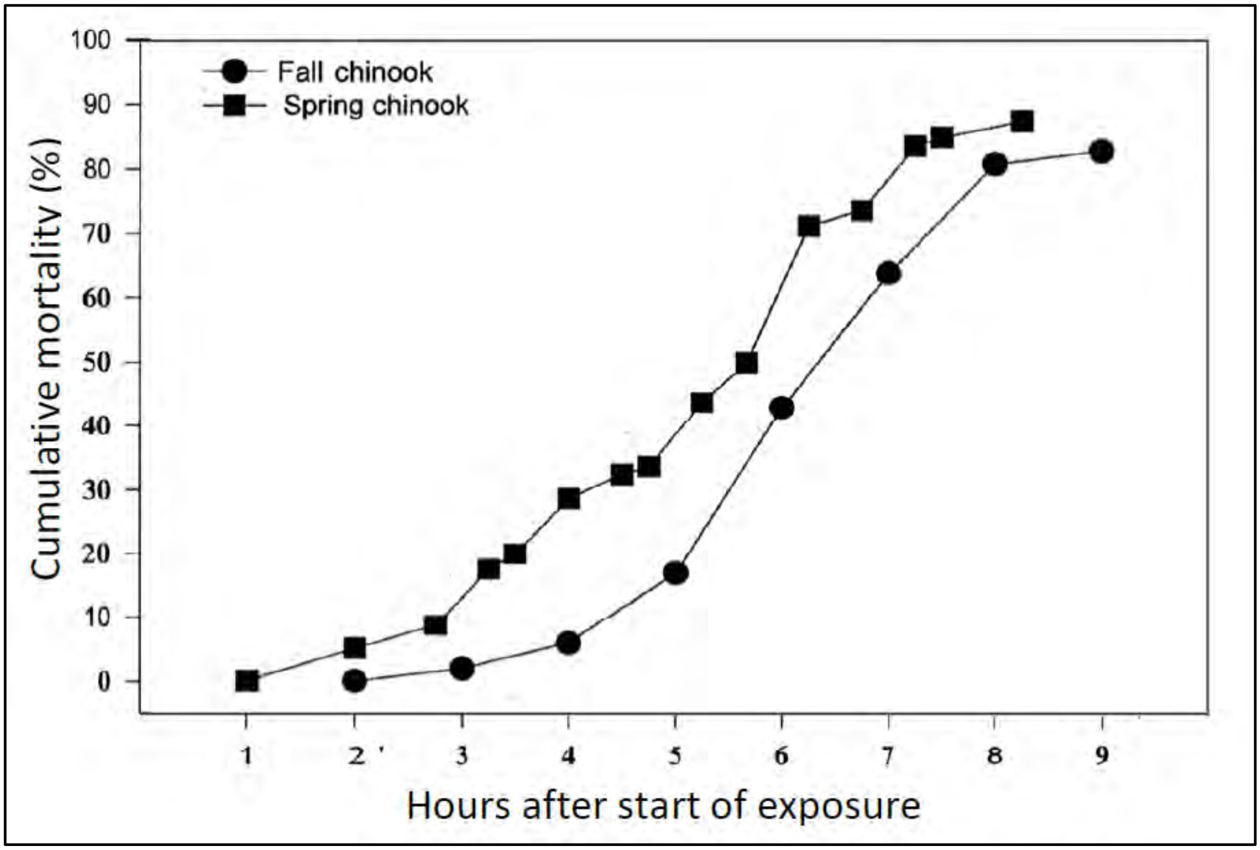
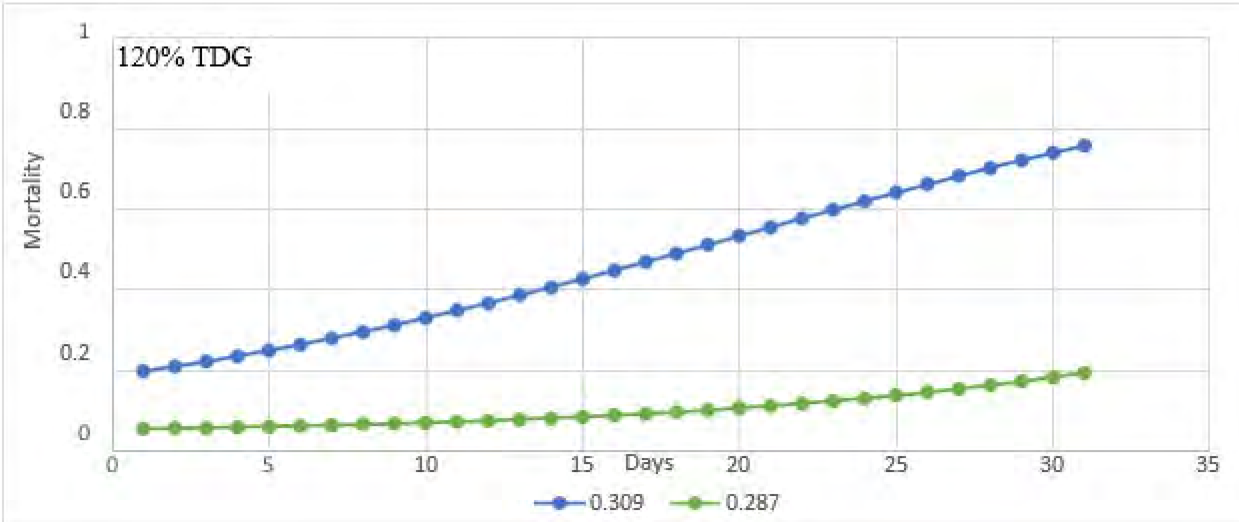


Figure 7-22. Juvenile Chinook Salmon Cumulative Mortality with 120% TDG at 12°C Exposure
Cumulative mortality of juvenile Chinook salmon during exposure to 130% TDG at 12°C. (Maule et al. 1995). Total number of fish exposed was 75 in each group. Water depth was 28 cm.

We revised the coefficient in the equation for $M_{\text{TDGJuveniles}}$ from 0.309 downward to 0.287 to reflect a decrease in the chance of an emigrating juvenile salmonid being exposed to lethal levels. This decrease was suggested by the limited time that a fish would be in a high TDG area below the dam and the likelihood that the fish would spend all or some of the time at depths greater than 1.0 m (Beeman and Maule 2006). The revised equation resulted in a significant decrease in predicted mortality levels (Figure 6-22).



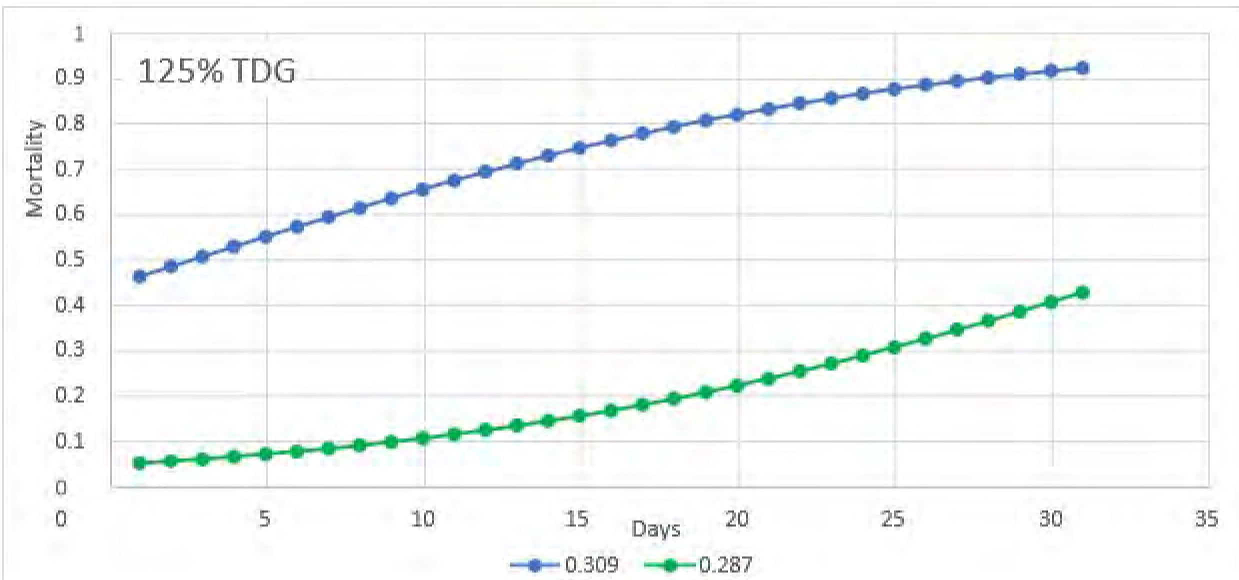


Figure 7-23. Monthly emigrating juvenile salmonid mortality by days of TDG exposure
Estimated monthly juvenile mortality as a function of days of exposure for emigrating salmonid juveniles at 120% TDG (above) and 125% TDG (below), using two different coefficients in the mortality estimate formula.

We believe that this estimate of TDG related mortality better reflects the conditions that emigrating salmonids experience following passage. This is not to diminish the potential harm that can be caused by fish holding, incubating, or residing below the projects when TDG levels are high. Hansen et al. (2017) remarked that mortality in juvenile salmonids was 92% when held in a screwtrap below the dam for periods of up to a day when TDG levels averaged 124% (117-130 %). It should also be noted that TDG levels as estimated by the USACE and made available to the fish modelers were adjusted by the USACE in some of the Alternatives (1 and 4) to account for the anticipated effect of structures to be constructed to reduce supersaturation.

7.4.2 Post Dam Passage

Juveniles can successfully emigrate downstream of the dams at any of the three juvenile life history stages. The life cycle model determines the proportion of each juvenile life history type that continues to the estuary and ocean or remains in freshwater through an additional juvenile stage. For example, a spring subyearling Chinook salmon that had successfully passed Foster Dam, could continue downstream, or remain in the reach below Foster Dam, finally emigrating as a fall subyearling or yearling (second subyearling). Currently, the proportional split in juvenile life history trajectories is predetermined by the FBW, as is the survival for each trajectory. There are a number of environmental and genetic factors that determine trajectories (Schroeder et al 2016), but it is not possible to quantify these model parameters. Similarly, Chinook salmon and winter steelhead cohorts that develop from eggs laid below the dams will exhibit multiple juvenile life history trajectories with predetermined splits and survivals.

7.5 OCEAN SURVIVAL

To tie the estuary and early ocean survival to a dynamic marine influence, we followed the framework of Zabel et al. (2006) and Zabel and Jordan (2020), which relate estuary and early ocean survival to large-scale Pacific Ocean basin level drivers and regional oceanic conditions.

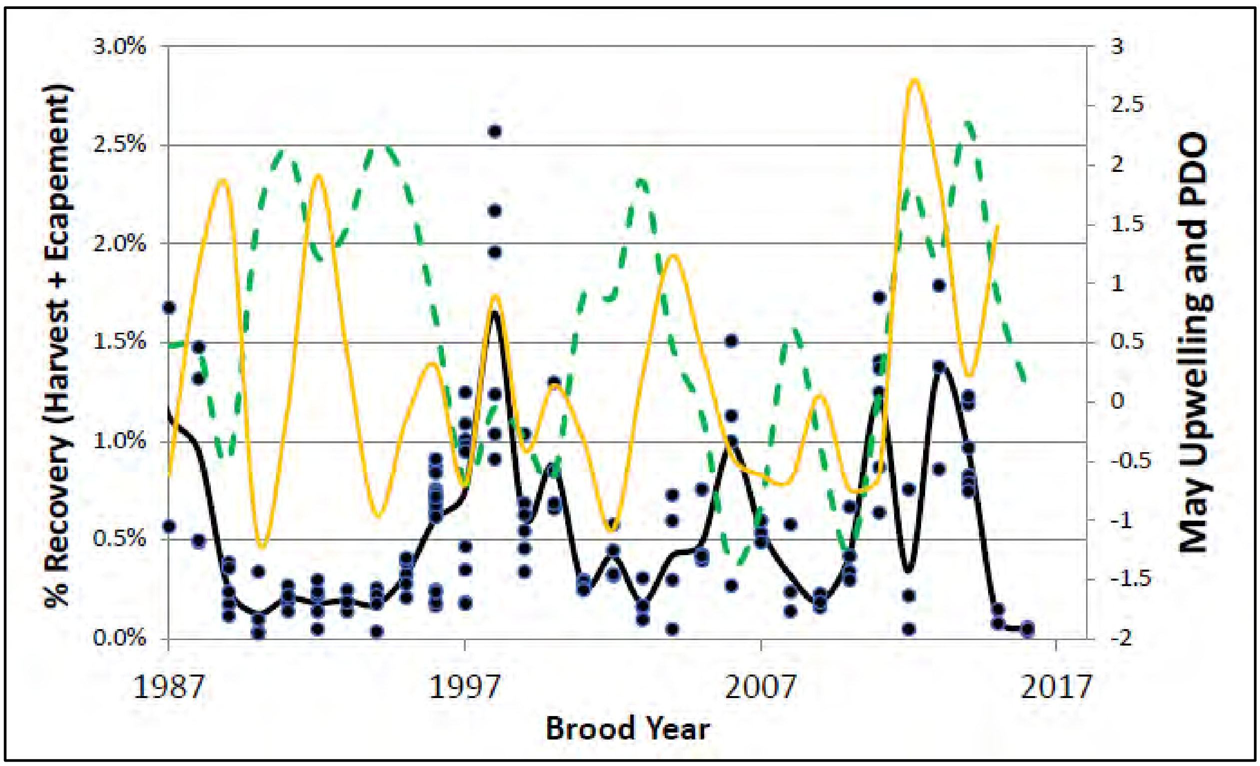


Figure 7-24. Average smolt to adult survival for hatchery reared Chinook salmon juveniles
Average smolt to adult survival for hatchery reared Chinook salmon juveniles from 142 release groups from the McKenzie Hatchery for Brood Years 1986-2016 (Blue dots and line). Standardized Upwelling Index for 48N 125W for May (Yellow), PDO – Pacific Decadal Oscillation for May (Green). Upwelling and PDO values represent the smolt year (+2) for each brood year.

First, for each basin and population we estimated estuary/early ocean survival, s3, for brood years from smolt-to-adult (SAR) survival estimates from coded-wire tagged hatchery fish (Regional Mark Processing Center, rmpc.org) by taking into account river survival through the mainstem (upstream and downstream), age-class composition of returning adults, and harvest (Zabel et al. 2006). We lacked SAR estimates specific to the two steelhead populations, and as a proxy we applied a relationship we developed from SAR estimates for Clackamas winter steelhead (Portland Gas and Electric (PGE), unpublished data). A revised estimate of PGE's Clackamas steelhead smolt counts (K. Malone, BioAnalysts, Inc., Boise, ID) applied to the steelhead SAR estimates slightly improved the estuary/ocean model fit. We assumed that subsequent yearly survival in the ocean after the first year was constant (SO, set at 0.8; Ricker 1976) due to the lack of sufficient data on marine survival of Willamette River anadromous salmonids. The equation for the estimated s3 was as follows:

$$s3 = \frac{SAR}{(p_3 + p_4 \cdot SO + p_5 \cdot SO \cdot SO + p_6 \cdot SO \cdot SO \cdot SO)}$$

where p_t is the proportion of t-year olds. These estimates of estuary/early ocean survival, s3, were related to a suite of candidate indicators, including monthly spring through fall Pacific Decadal Oscillation index (PDO; April, May, June, July, August, September; http://jisao.washington.edu/pdo/PDO.latest), the monthly Pacific coastal upwelling index (at 45° N 125° W) in the spring and summer/fall (April, May, Sept; http://www.pfeg.noaa.gov/products/PFEL/modeled/indices/upwelling/NA/data_download.html), and historical monthly and seasonal river flows in both the Willamette (at Salem, http://waterdata.usgs.gov/nwis/nwisman/?site_no=14191000&agency_cd= USGS; J. Ammann, USACE Portland District) and Columbia river mainstems (at The Dalles, http://nwis.waterdata.usgs.gov/wa/nwis/annual/?site_no=14105700&agency_cd= USGS), which influence migration timing and are representative of freshwater conditions. The choice of indicators was based on previous studies (e.g., Zabel et al. 2006, Scheuerell et al. 2009), but was constrained to those that have a long period of record to be able to encompass the ocean cycles observed over the last several decades and to cover as much as possible the water year period of record of FBW. We logit-transformed the response variable s3 to constrain back-calculated survival estimates to the (0, 1) range. The candidate model set comprised all possible combinations of predictor variables with a maximum of up to 3 predictors (Zabel et al. 2006). We used AICc as a measure of data support among the candidate models (Burnham and Anderson 2002).

For McKenzie River Chinook salmon hatchery fish, the data most supported a model that included coastal upwelling in the spring and fall, and spring PDO (Fig. 5.2). For North Santiam Chinook salmon hatchery fish, the model with the most data support included upwelling and PDO in the spring (Fig. 5.3). For South Santiam and Middle Fork Chinook salmon hatchery fish, the data supported a model that included upwelling in the fall and PDO in spring (Figs. 5.4 & 5.5). The data most supported a model fitted to Clackamas steelhead that included fall upwelling, spring PDO, and mean summer mainstem Willamette River flow (Fig. 5.6).

We used these relationships to make predictions over the entire scope of the period of record of predictors to get s3 over observed ocean cycles (Figs 5.2 – 5.6, solid lines, bottom plots). Using these estimated historical s3 time series, we integrated them into the life cycle models stochastically in two ways. First, for reach run of 100 years of the model, we chose random start locations in the historical time series and as needed we extended the time series starting with the beginning of the time series and moving forward. Second, we sampled yearly from the 95% prediction intervals about the s3 relationship (e.g., within the bounds of the dashed lines in the lower panels in Figs 5.2 – 5.6).

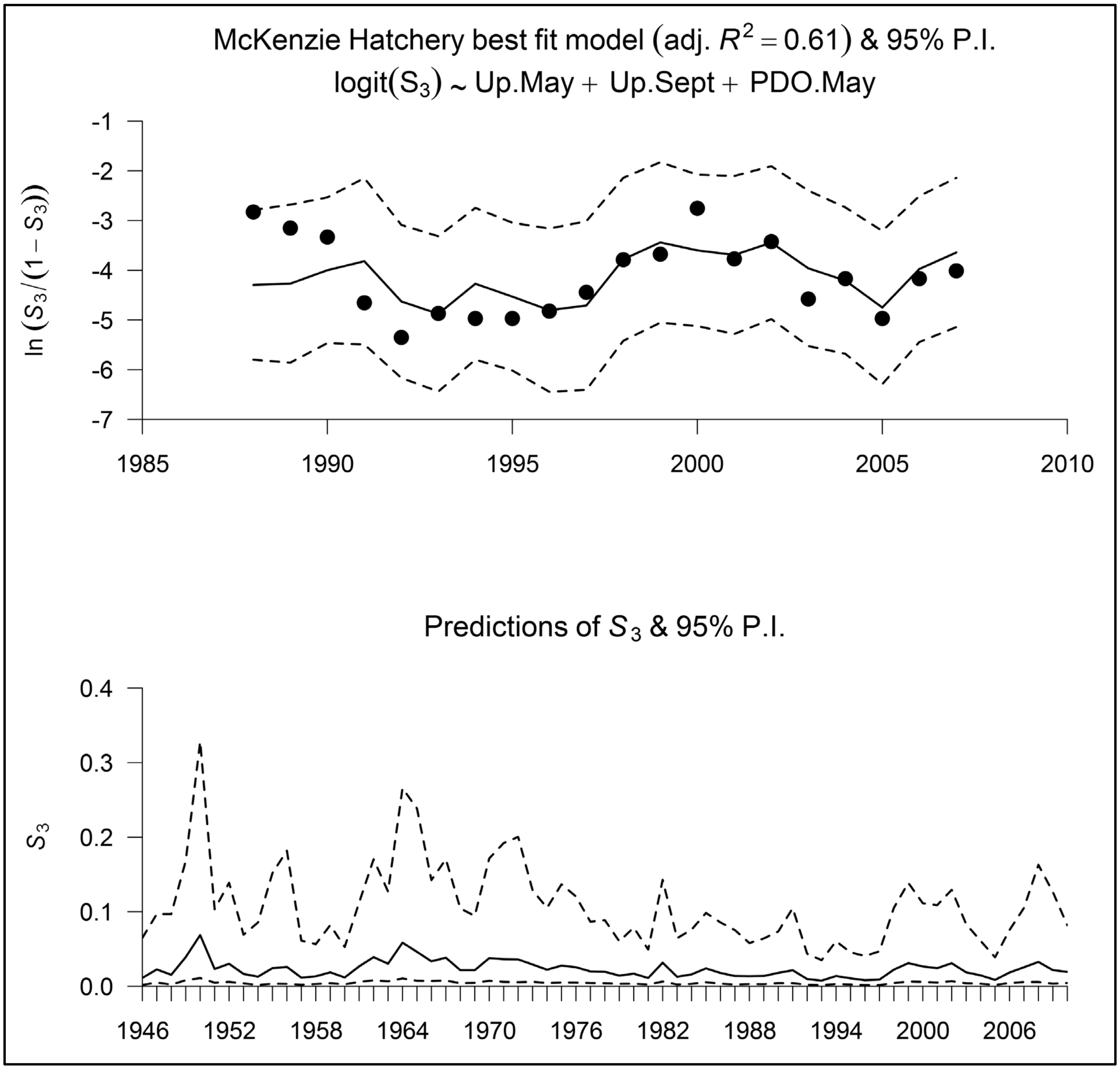


Figure 7-25. Recent McKenzie Hatchery vs all historic Chinook salmon estuary/early ocean survival. *Marine predictors (top) that were in the best fitted model to estimated estuary/early ocean survival of McKenzie hatchery Chinook salmon, and hind cast predictions (bottom) of estuary and early*

ocean (s3) survival across the full historical range of the predictors. Points are the data, solid lines are the fitted relationships, and the dashed lines are 95% prediction intervals.

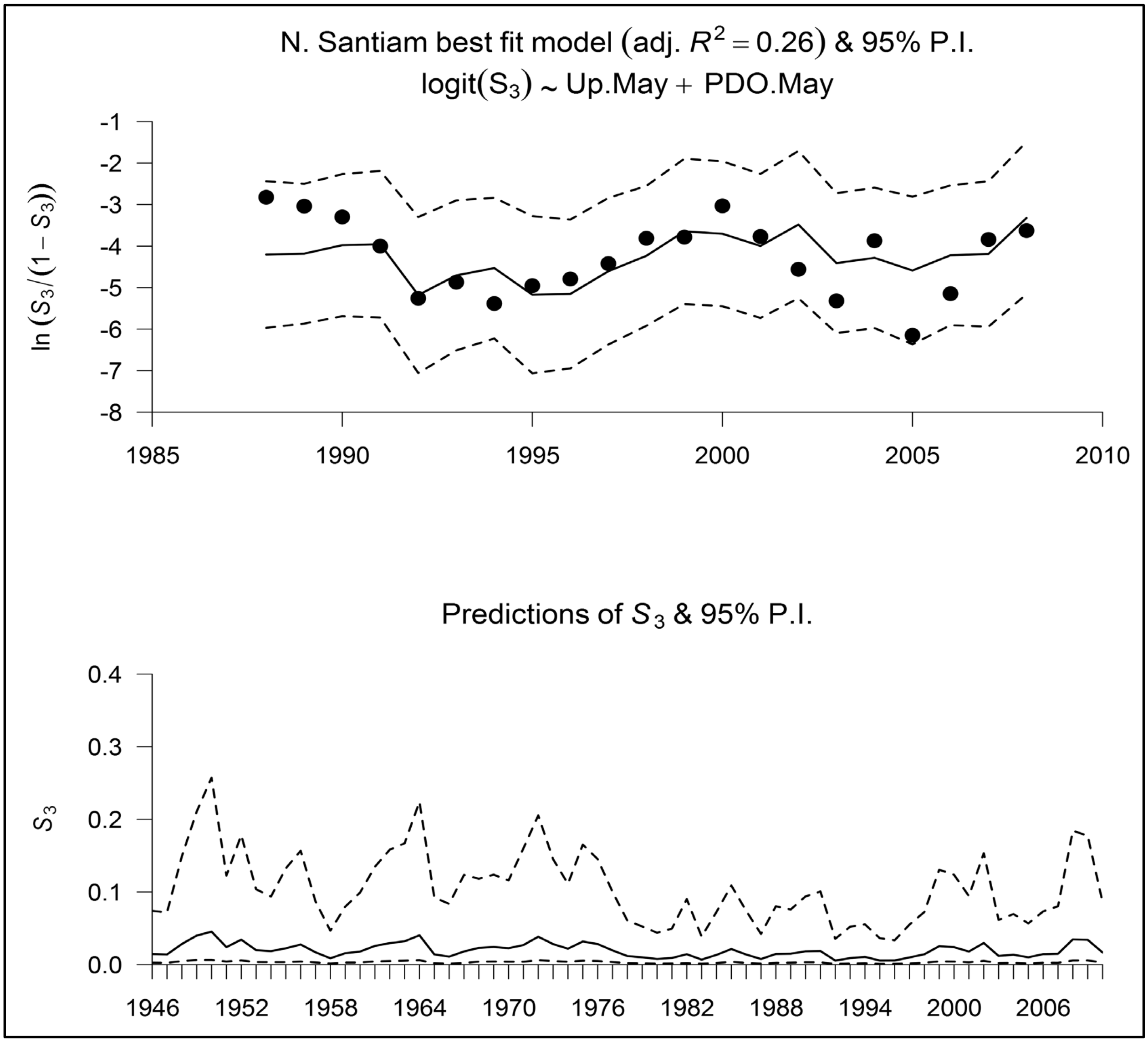


Figure 7-26. Recent North Santiam Hatchery vs all historic Chinook Salmon Estuary/Early Ocean Survival. *Marine predictors (top) that were in the best fitted model to estimated estuary/early ocean survival of North Santiam hatchery Chinook salmon, and hind cast predictions (bottom) of estuary and early ocean (s3) survival across the full historical range of the predictors. Points are the data, solid lines are the fitted relationships, and the dashed lines are 95% prediction intervals.*

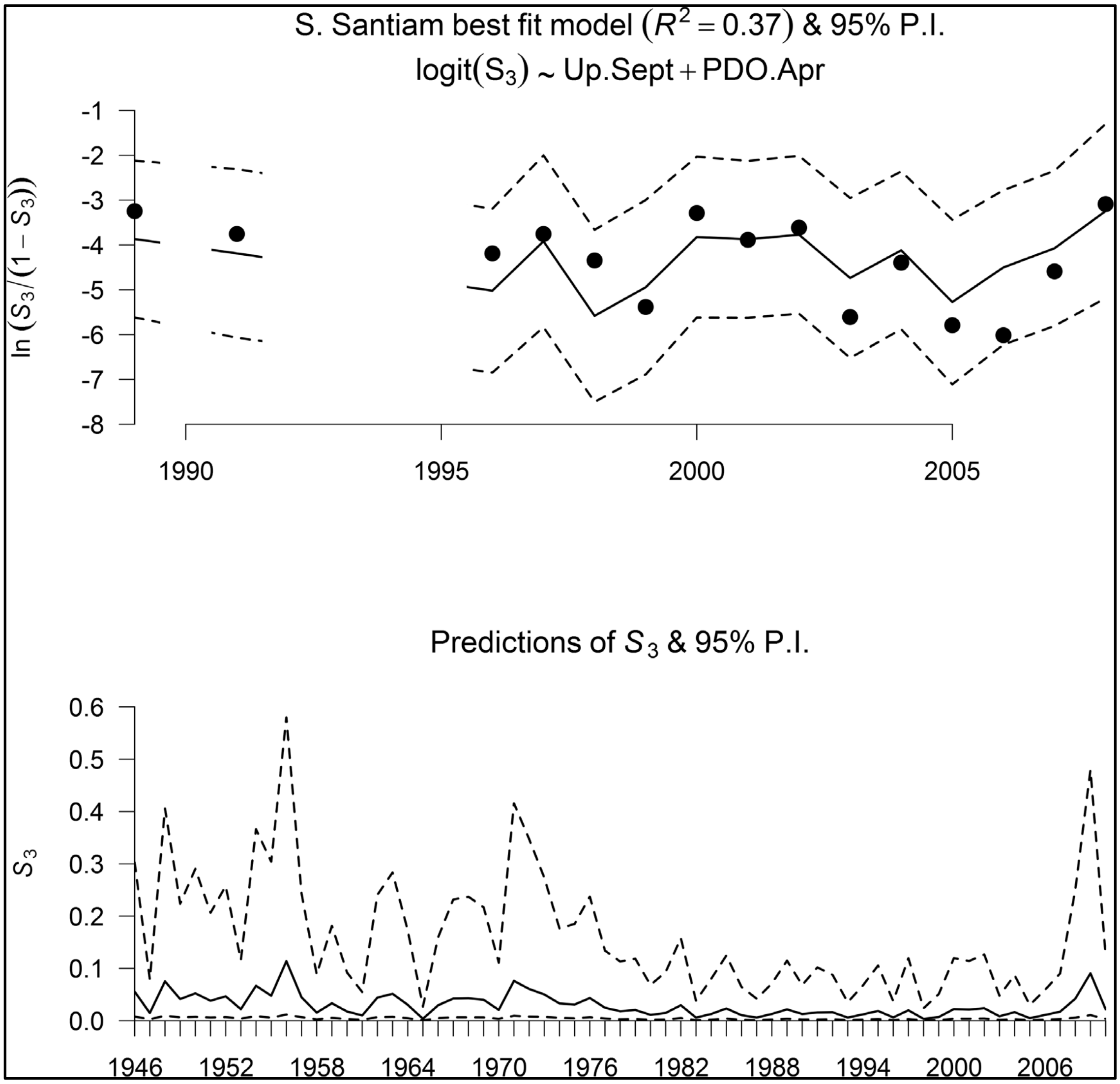


Figure 7-27. Recent South Santiam Hatchery vs all historic Chinook Salmon Estuary/Early Ocean Survival *Marine predictors (top) that were in the best fitted model to estimated estuary/early ocean survival of South Santiam hatchery Chinook salmon, and hind cast predictions (bottom) of estuary and early ocean (s3) survival across the full historical range of the predictors. Points are the data, solid lines are the fitted relationships, and the dashed lines are 95% prediction intervals.*

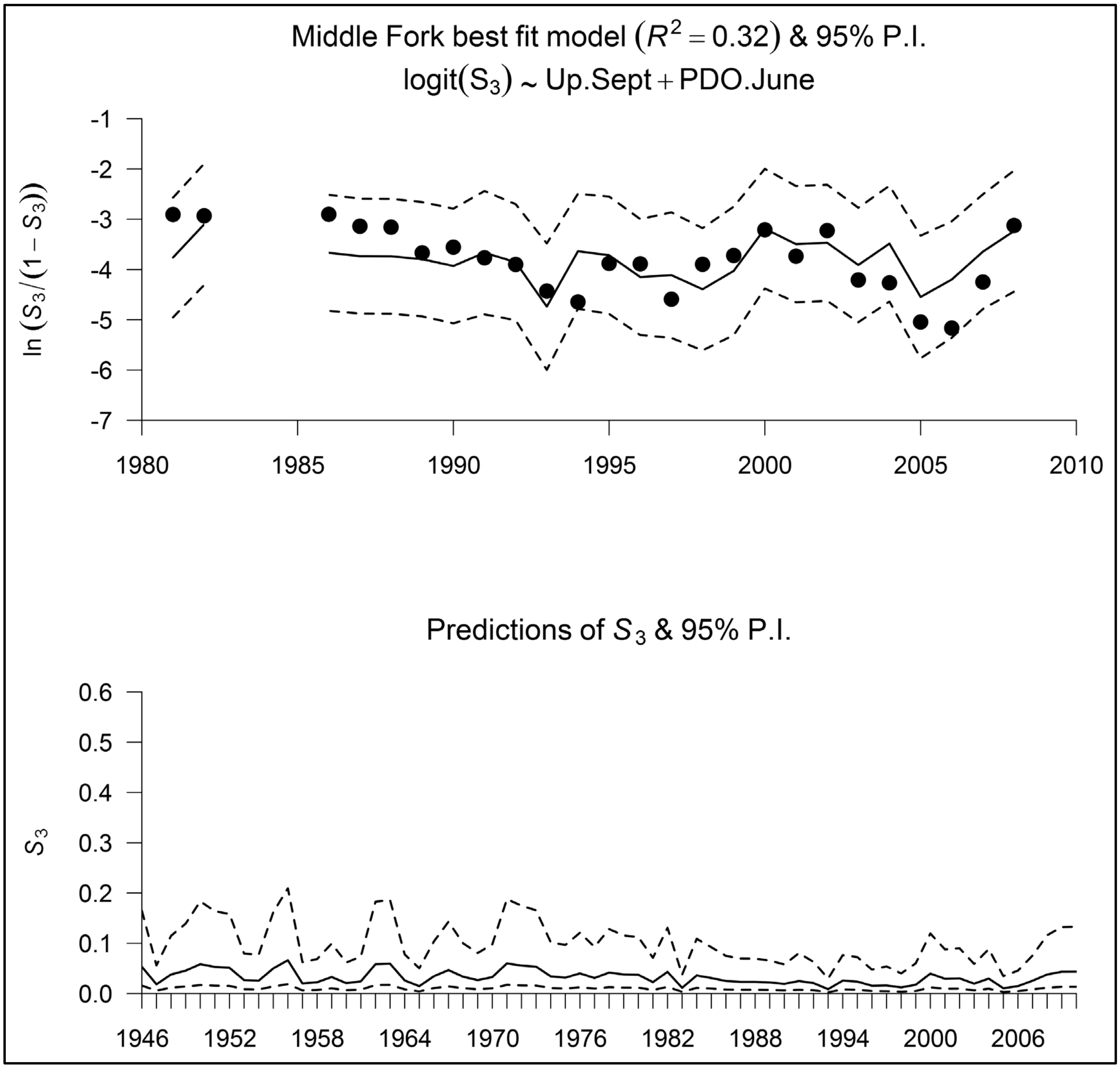
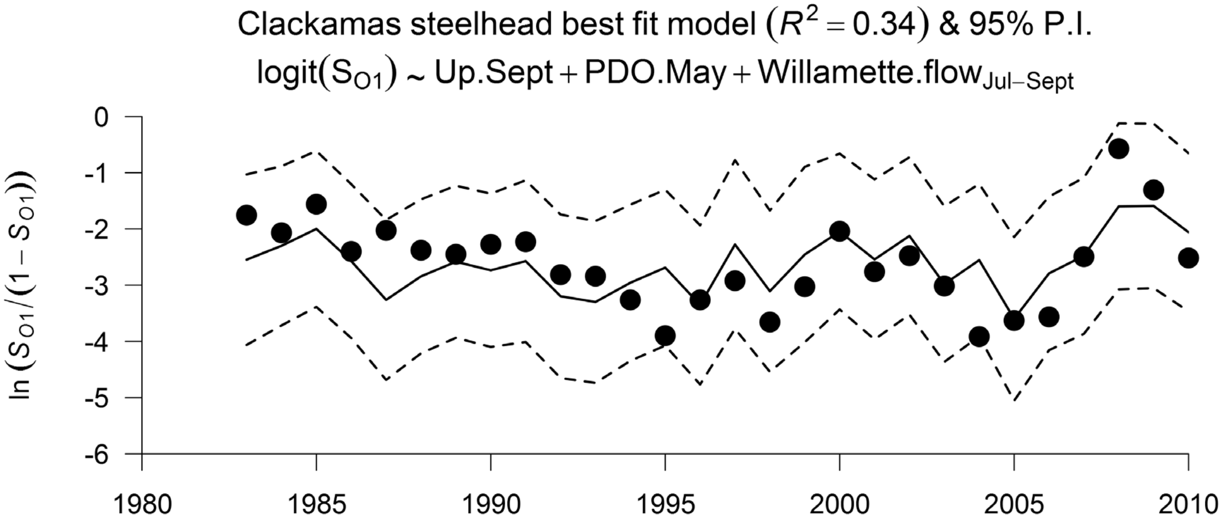


Figure 7-28. Recent Middle Fork Willamette River hatchery vs all historic Chinook Salmon Estuary/Early Ocean Survival *Marine predictors (top) that were in the best fitted model to estimated estuary/early ocean survival of Middle Fork Willamette River hatchery Chinook salmon, and hind cast predictions (bottom) of estuary and early ocean (s3) survival across the full historical range of the predictors. Points are the data, solid lines are the fitted relationships, and the dashed lines are 95% prediction intervals.*



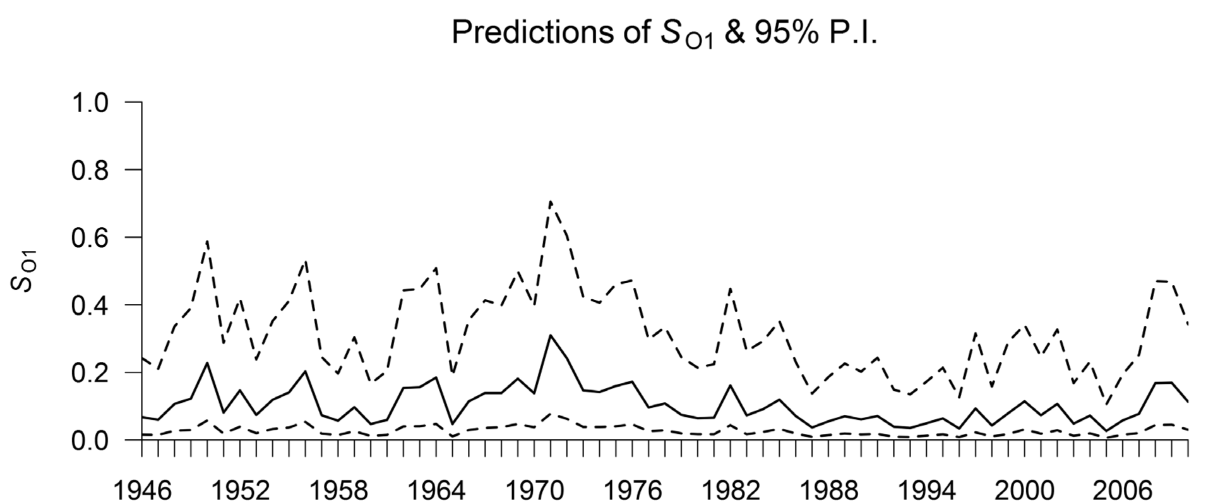


Figure 7-29. Recent Clackamas River winter steelhead vs all historic Winter Steelhead Estuary/Early Ocean Survival *Marine predictors (top) that were in the best fitted model to estimated estuary/early ocean survival of Clackamas River winter steelhead, and hind cast predictions (bottom) of estuary and early ocean (*s01*, refers to estuary and ocean entry year, similar to* s3 *for Upper Willamette River Chinook salmon) survival across the full historical range of the predictors.*

We made an adjustment to s3 to account for the data source we used for developing these relationships. In the Chinook salmon life cycle models we multiplied the yearly estimated s3 by a factor of 2 because the SAR data was from coded-wire tagged hatchery fish, and there is some evidence from other basins, such as in the Upper Columbia River, that hatchery SARs are about half those of wild fish (Zabel 2013). In the North and South Santiam winter steelhead life cycle models, we multiplied s3 by 1.5 to account for the potential for a hatchery influence on the Clackamas steelhead population and to match baseline conditions to recent spawner counts.

7.5.1 Synchronization of Freshwater and Ocean Conditions

For each of the model runs we synchronized the freshwater and ocean conditions experienced by the fishes. Broad-scale terrestrial conditions (e.g., winter snowpack, precipitation patterns) tend to be related to large-scale marine indices, such as El Nino Southern Oscillation, North Pacific Index, and Pacific Decadal Oscillation (Wallace and Gutzler 1981, Cayan 1996, Mantua et al. 1997, McCabe and Dettinger 2002). The life cycle models include a first-year ocean survival from fitted relationships to indices of ocean conditions. Through stochastic simulation, each iteration of life cycle model had a randomly selected start year, and for each model iteration, we indexed the time series of ocean conditions with the corresponding freshwater conditions of FBW, TDG mortality, and temperature and flow estimates used in the en route and pre-spawn mortality relationships.

7.6 HARVEST

7.6.1 Chinook Harvest

Chinook salmon are subject to ocean, estuary, and freshwater harvest. Changes in non-terminal ocean fisheries can influence age structure, and certain fisheries (i.e. gill nets) can also result in size/age selective harvest. The LCM utilized an overall ocean exploitation rate of 10.2% (based on the mean harvest rates between 2001 to 2018) (Ford 2022). Estuary, mainstem Columbia River, lower Willamette River, and tributary commercial and recreational fishery impacts were included into a single “In-river harvest” life stage survival (Figure 6.1). Due to their ESA listed status, natural origin Willamette River spring Chinook salmon are not directly harvested in the freshwater commercial or recreational fisheries but are subject to a net entanglement or hook mortality. The recreational catch and release fishery for unmarked fish results in an incidental mortality of 10.0% for the mainstem fishery and 12.2% for the Willamette River fishery (ODFW 2020) with encounter rates based on hatchery-origin fish harvest rate. In addition, the incidental mortality rate for unmarked spring-run Chinook salmon caught and released from the Columbia River tangle net fishery was 14.7% and 40.0% for the large-mesh gill net fishery (ODFW 2013).

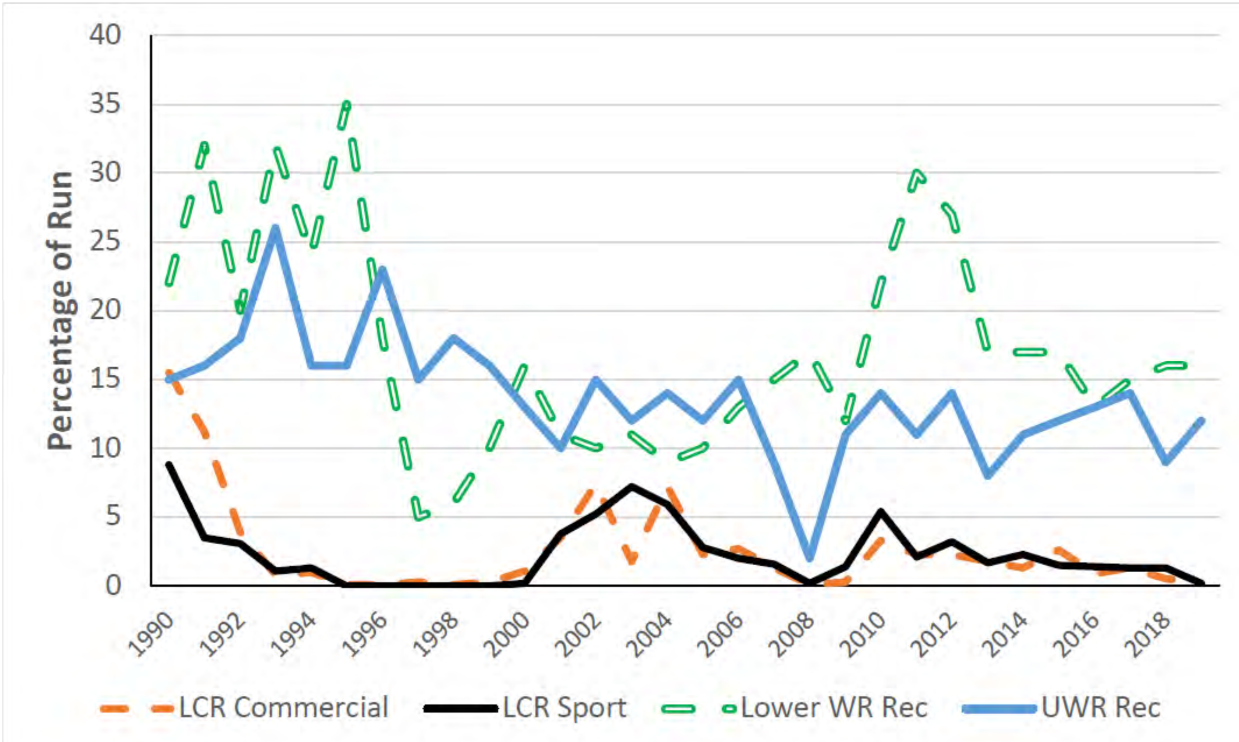


Figure 7-30. Breakdown of terminal fisheries for Upper Willamette River (UWR) Chinook salmon. *Lower Willamette River (below Willamette Falls) and UWR recreational (Rec) fisheries are mark-selective and reflect retention of clipped fish and encounter/hooking mortalities of unmarked fish; hooking mortality rates for the Willamette River are estimated at 12.2% (ODFW and WDFW 2020).*

Although there is no longer a directed fishery for natural-origin spring-run Chinook salmon, selective marking has allowed for a relatively higher harvest rate on hatchery-origin fish and therefore a higher encounter rate for natural-origin fish. From 2001-2019 an average of 11.6% of the Willamette Falls count of spring-run chinook salmon was harvested above the falls (ODFW and WDFW 2020). Population specific rates exist for some of the Upper Willamette River populations, and where these were unavailable, estimates from geographically proximate populations were used (Table 6.1). Total freshwater impacts from 2009-2019 averaged 7.87% for spring-run Chinook salmon the North Santiam River were and 8.35% for the McKenzie River (ODFW 2020) for natural-origin adults. Overall freshwater impacts for the Upper Willamette River spring-run Chinook salmon (hatchery and natural origin) average 21.2% (2001-2019).

Table 7-4. Freshwater fishery percent impact on natural-origin Willamette River spring-run Chinook, 2009-2019.

Source	Basin	2009	2010	2011	2012	2013	2014	Z	2015	2016	2017	2018	2019
FISHERY	L. Col. Commercial	1.5	6	4.7	2.5	4.6	3		3.4	1.6	1.4	1.6	1.8
FISHERY	L. Col. Recreational	0.8	1.8	0.6	1	0.5	0.9		0.7	0.3	0.1	0.5	0.2
FISHERY	Lower Willamette River	3.6	6.9	6.2	4.5	2.8	3.9		4	2.2	2.4	2.9	3
FISHERY	Lower Clackamas River	0.2	0.2	0.2	0.3	0.3	0.1		0.1	0.1	0.3	0	0
FISHERY	Upper Willamette River	0.1	0.2	0.1	0.2	0.1	0.1		0.1	0.1	0.1	0.1	0.2
FISHERY	North Santiam River	0.1	0.3	0.1	0.3	0.2	0.7		0.2	0.4	0.4	0.3	0.2
FISHERY	McKenzie River	1.3	1	1.1	1.3	0.8	0.5		0.4	0.5	0.2	0.5	1
Total Fishery	Study Area	7.6	16.4	13	10	9.3	9.2		8.9	5.2	4.9	5.9	6.4
Totals by Population	Clackamas River	6.1	14.8	11.7	8.3	8.2	7.9		8.3	4.2	4.2	5.1	5
Totals by Population	North Santiam River	6.1	15.1	11.7	8.4	8.2	8.6		8.5	4.6	4.4	5.6	5.4
Totals by Population	McKenzie River	7.3	15.8	12.8	9.5	8.7	8.4		8.6	4.6	4.2	5.7	6.2

Note: Data from ODFW (2020).

In addition to known harvest effects, the "In-river harvest" life stage also included mortalities from unknown causes between Willamette Falls and the tributaries. This mortality accounts for the shortfall in adults when comparing counts at Willamette Falls with carcass counts (including known pre-spawning mortalities), dam counts, hatchery returns, and tributary harvest. In some years the drop-out rate is considerable and may be related to mortalities prior to reach census points in the tributaries. We cover this additional en route mortality in the next chapter.

7.6.2 Winter Steelhead Harvest

Harvest rates for winter steelhead are generally thought to be modest. Estimates of harvest for natural-origin winter steelhead in the mainstem Columbia River are generally less than 1.0%, below the 2.0% ESA limit (ODFW and WDFW 2020), and were not considered in the model. Similarly, although there is a recreational fishery for hatchery-origin summer steelhead in the Willamette River, there is no retention allowed for unmarked steelhead. Because of the overlap in adult return timing of summer-run and winter-run steelhead and even spring-run Chinook salmon, there is an encounter and hooking mortality risk. Falcy (2017) suggests that winter steelhead incidental mortality in the Willamette Basin is around 5%. Based on ODFW and WDFW (2020) we set ocean harvest rates to 5% for 3 and 4 year olds and 10% for 5 year olds.

7.7 FRESHWATER ADULT MIGRATION MORTALITY

In addition to relatively reliable estimates of harvest effects, the upstream adult migration stage in the life cycle model also included mortalities from unknown causes in the mainstem Willamette River, between Willamette Falls and the terminal tributaries. This mortality accounts for the shortfall in adults when comparing counts at Willamette Falls with carcass counts (including known pre-spawning mortalities), dam counts, hatchery returns, Willamette Falls fallbacks, and tributary harvest. In some years the drop-out rate is considerable[7] (Figure 7.1).

7.7.1 Model of En Route Mortality Above Willamette Falls

To account for the annual dropout rate between Willamette Falls and tributary census points (hereafter "en-route mortality") in the life cycle model, we fit a model of en-route mortality on the logit scale as a function of environmental drivers. We used the logit transformation because it can produce threshold type behavior, and the back-transformed estimates are bounded on the range 0 to 1. Our candidate set of explanatory variables included average Willamette River discharge at Albany and Salem, and water temperature at Willamette Falls during late winter and spring through early summer (encompassing nearly all of the river migratory window). Discharge data were obtained from USGS gauges #14174000 and #14191000 using the package 'dataRetrieval' in R (Hirsh and De Cicco 2015), and water temperature at Willamette falls was measured by Oregon Department of Fish and Wildlife personnel and compiled by Stefan Talke

[7] This life stage has since been partitioned into two life history stages: estimated in-river harvest and mainstem idiopathic mortalities, to distinguish to these sources of mortality.

(Portland State University). We also included average air temperature and total precipitation in Salem and Eugene in late winter and spring through early summer, which we obtained from the National Weather Service. Models were constructed using different combinations of explanatory variables with the dredge function in the 'MuMIn' (Barton 2016), and they were ranked by AIC with a correction for small sample size.

The best model of annual en route mortality rates on the logit scale included explanatory variables, (1) average discharge at Albany from 15 April to 31 July, (2) average water temperature at Willamette Falls between 20 January and 15 March, and (3) average water temperature at Willamette Falls between 15 April and 31 July (Figure). January–March water temperatures had a positive relationship with mortality, likely because fish passed Willamette Falls earlier and therefore spent more time in the mainstem Willamette River and tributaries below census points when late-winter temperatures were warmer, increasing their time spent in the study reach (Jepson et al. 2014). We expect that en route mortality increases the longer that fish spend in the study reach (exposure), because there is more time to succumb to disease, thermal exposure, and senescence. In contrast, water temperatures during the migration from April through July were negatively correlated with mortality, likely because fish traverse the mainstem more quickly in warmer waters, decreasing their time of exposure in the study reach. Discharge during the migration also negatively correlated with mortality, which was somewhat counterintuitive because higher discharge generally slows migration and would increase exposure prior to reaching census points. However, the greater discharge may cause fish to pass Willamette Falls later in the season, and dilute harmful contaminants, thereby decreasing thermal exposure and increasing mainstem survival.

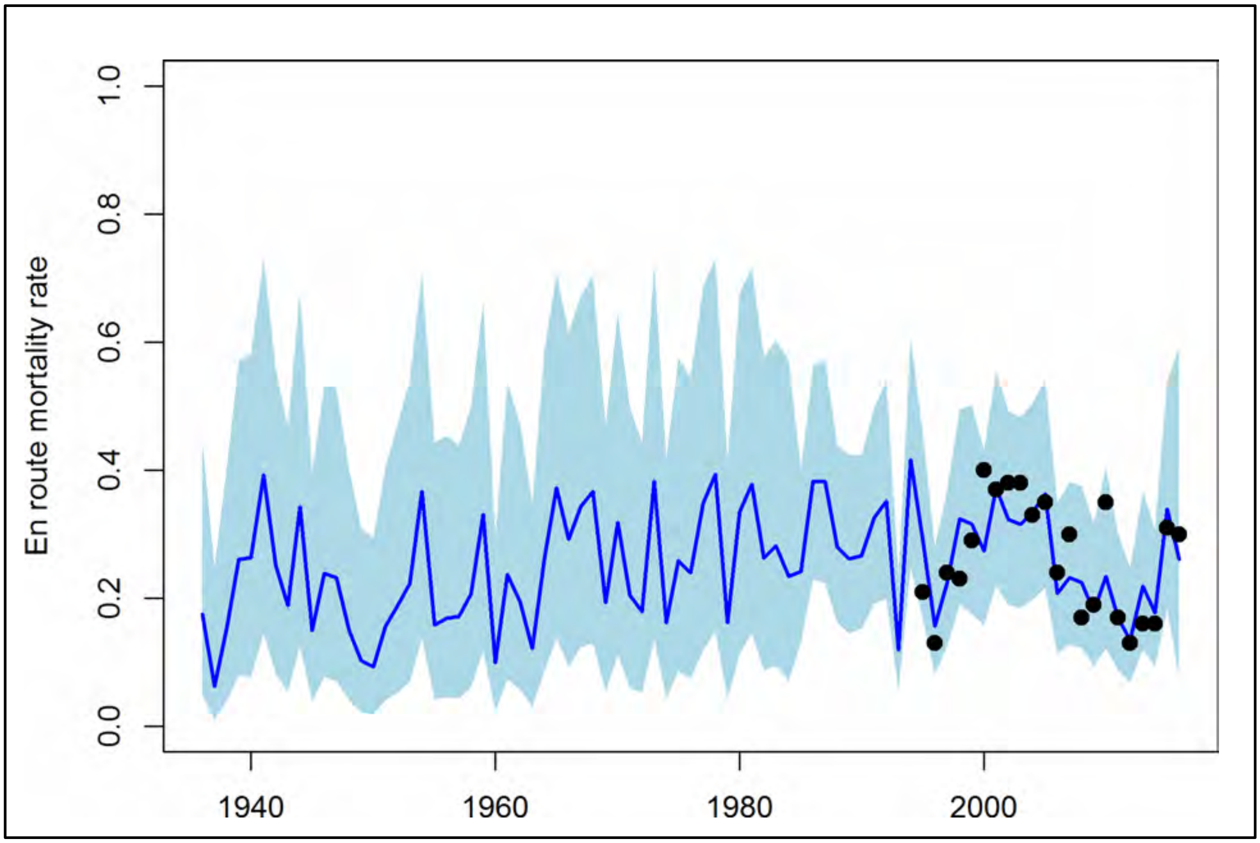


Figure 7-31. Time series of en route mortality rates and model-predicted mortality rates. *The blue shaded area represents a 95% prediction interval around the model-predictions. The greater uncertainty surrounding the estimates outside the 1985–2015 range is due to the unavailability of water temperature data at Willamette Falls in those years, necessitating the use of model-predicted temperature estimates that also had uncertainty around them.*

7.7.2 Willamette Falls temperature models

The available time series of water temperatures at Willamette Falls only spanned the period 1985 to 2015, but we wanted to hindcast historical en route mortality to 1936 for life-cycle model simulations that use Fish Benefit Workbook passage survivals and dam efficiencies reflecting waters years over that time. Therefore, we developed models to predict historical water temperatures (Moore 1967, Bottom et al. 2011, Overman 2017). Our candidate set of explanatory variables for modeling water temperature were seasonal averages of discharge, air temperature and precipitation as described above for modeling en route mortality.

The explanatory variables in the best model of average water temperature at Willamette Falls between 20 January and 15 March were (1) total precipitation in Salem from September (of the previous year) through January, (2) average air temperature in Salem from October through November of the previous fall, and (3) average air temperature at Salem from January through March. All three of the variables had positive relationships with late-winter water temperature.

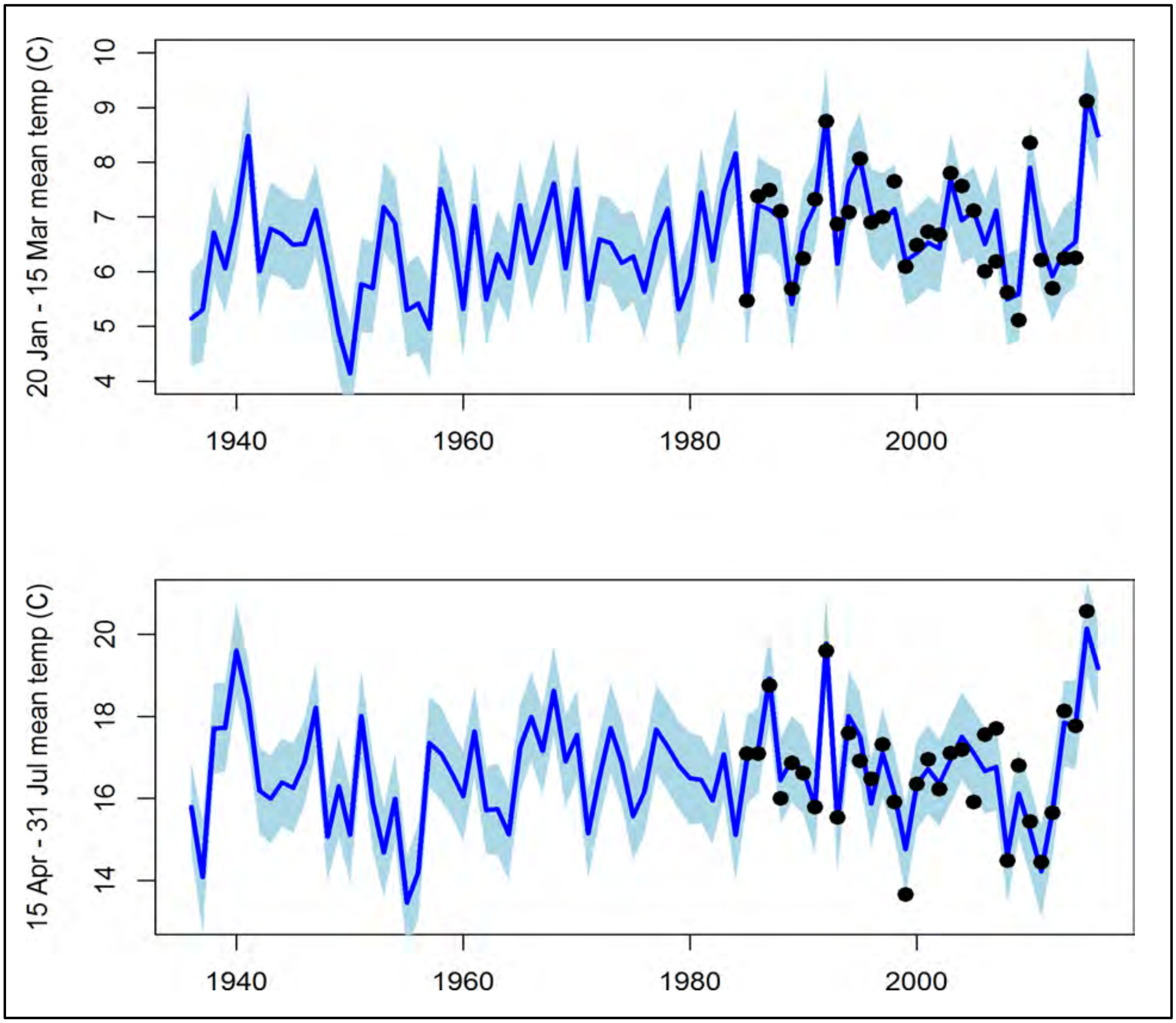


Figure 7-32. Average Water Temperatures at Willamette Falls, and Model-Predicted Water Temperature. *Time series of annual observed average water temperatures at Willamette Falls (black points), and model-predicted water temperature (blue lines). The blue shaded areas represent 95% prediction intervals around the model-predicted temperatures. The model predictions were used as inputs to the model of en route mortality in the mainstem where observed temperature values were not available.*

The best model of average water temperature at Willamette falls between 15 April and 31 July had explanatory variables, (1) average air temperature in Eugene from February through June, (2) total precipitation in Salem from March through May, and (3) average Willamette River discharge at Albany from 15 April to 31 July. Air temperature in February–June and precipitation in March–May had positive relationships with water temperature, whereas discharge in April–July had a negative relationship with water temperature (Figure 6-31).

7.8 PRE-SPAWNING MORTALITY-TRIBUTARY REACHES, WATER TEMPERATURES AND TEMPERATURE ADJUSTMENTS FOR ALTERNATIVES

Recent studies have documented high levels of pre-spawning mortality (PSM) among returning spring-run Chinook salmon adults in a number of Upper Willamette River subbasins. Schreck et al. (2013) discuss several potential causal (primarily pathogenic) factors that may be responsible for pre-spawning mortality. Bowerman et al. (2018) analyzed pre-spawning mortality in the upper Willamette River Basin and found that it positively correlated with water temperature and the proportion of spawning fish that were of hatchery origin (pHOS). We similarly related PSM to both temperature and pHOS and included additional above-dam reaches (Table 8.1). We also restricted our analysis to the years that we thought reflected contemporary fish passage operations and survey protocols (after 2011). Finally, we excluded reaches in which we believed that factors unique to those reaches, such as delayed dam passage, might be strongly affecting PSM (i.e., below Bennett, below Fall Creek, and below Dexter).

Table 7-5. Proportions of recovered female carcasses that retained ≥ 50% of their eggs *

River	Reach	2012	2013	2014	2015	2016	2017
North Santiam River	Above Bennett	0.33 (248)	0.24 (78)	0.30 (125)	-	-	0.13 (24)
North Santiam River	Minto/Big Cliff	-	-	0.14 (7)	-	-	0.13 (24)
North Santiam River	Breitenbush R.	0.00 (1)	0.05 (107)	0.03 (32)	0.12 (38)	0.05 (58)	0.13 (31)
North Santiam River	Upper N Santiam	-	0.05 (107)	0.10 (47)	0.12 (38)	-	0.13 (31)
South Santiam River	Below Foster	0.27 (337)	0.22 (206)	0.21 (161)	0.12 (290)	0.04 (530)	0.11 (213)
South Santiam River	Above Foster	0.15 (117)	0.54 (90)	0.31 (29)	0.40 (42)	0.11 (47)	0.27 (34)
McKenzie River	Below Leaburg	0.26 (285)	0.24 (25)	0.27 (95)	0.35 (40)	0.17 (60)	0.69 (16)
McKenzie River	Above Leaburg	0.02 (98)	0.03 (68)	0.02 (72)	0.05 (143)	0.00 (181)	0.01 (73)
McKenzie River	Above Cougar	0.00 (19)	0.25 (20)	0.00 (20)	0.00 (11)	0.00 (34)	0.00 (34)
Middle Fork Willamette River	Above Fall Creek	0.13 (45)	1.00 (15)	0.71 (17)	-	0.15 (20)	0.75 (8)
Middle Fork Willamette River	Mid. Fork N Fork	0.24 (284)	0.30 (152)	0.11 (94)	0.30 (82)	-	-

Notes: Proportions of recovered female carcasses that retained ≥ 50% of their eggs (PSM) used to fit model. Total numbers of carcasses examined are in parentheses.

Data were compiled from ODFW reports.

For our analysis, we used a logit transform on PSM, and assumed that the logit-transformed PSM measurements had a Gaussian error distribution. We weighted each value by the number of carcasses it represented in the model fitting. The model that was identified as providing the best fit was the one where temperature was entered additively along with pHOS (Figure 6-32).

Logit (PSM) = -13.21 + (pHOS * 2.09) + (Temp * 0.79)

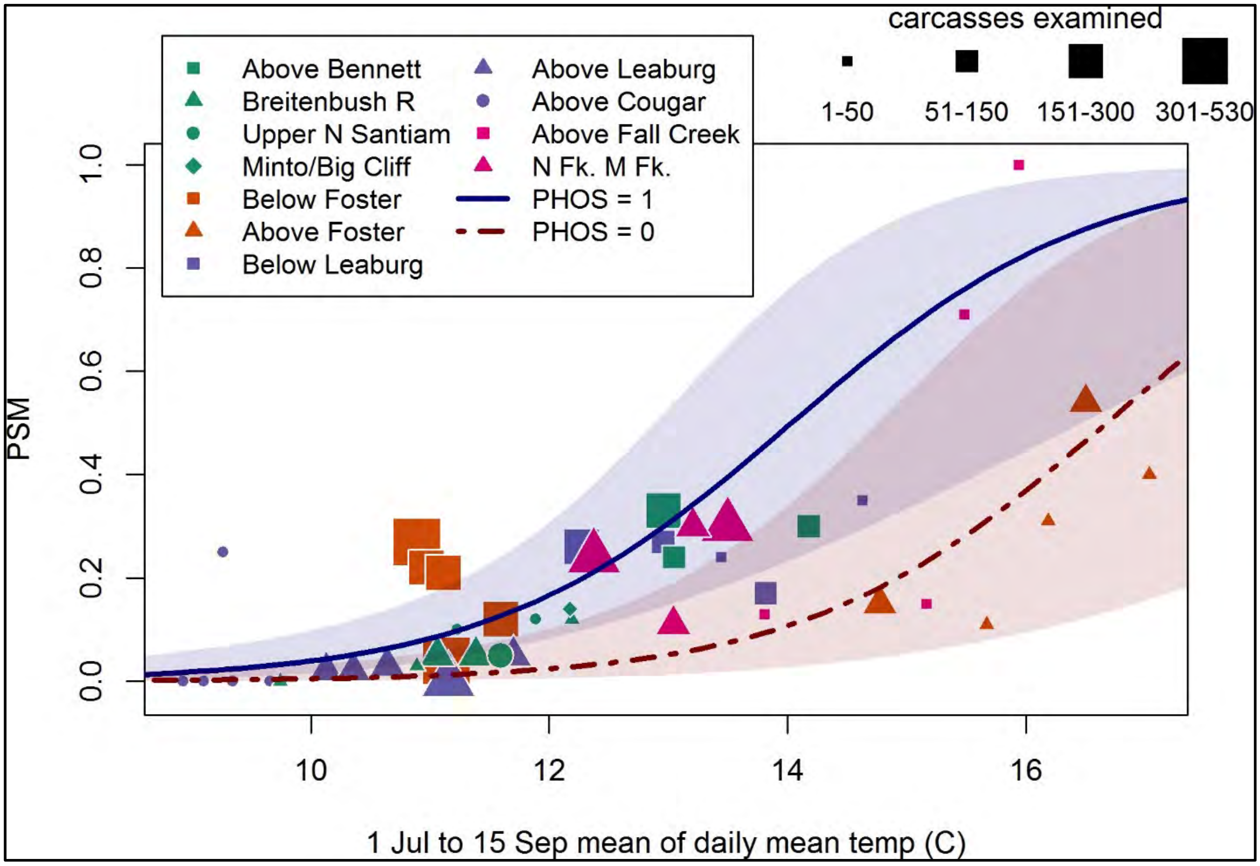


Figure 7-33. Mean stream temperatures versus female carcasses retaining ≥ 50% of egg.
Mean July–September stream temperature verses proportion of recovered female carcasses retaining ≥ 50% of eggs (PSM). The lines represent a linear model of PSM as a function of stream temperature and the proportion of hatchery-origin fish on the spawning grounds (pHOS) at 0% and 100% pHOS, while the shaded areas represent 95% confidence intervals.

Temperature– For our metric of the temperature experienced by fish during the pre-spawning and spawning period, we used the average of daily mean water temperatures from 1 July to 15 September. While spawning does not typically occur during July, most fish have reached their spawning tributary by then and hold there until spawning begins. Because the water temperatures experienced at this time could influence rates of energy- reserve depletion, and vulnerability to pathogens, we included July in our range of dates over which to average temperature as a variable in the model of PSM.

Water temperatures were obtained from USGS gauging stations throughout the basin (Figure 6-32, Table 6-5).

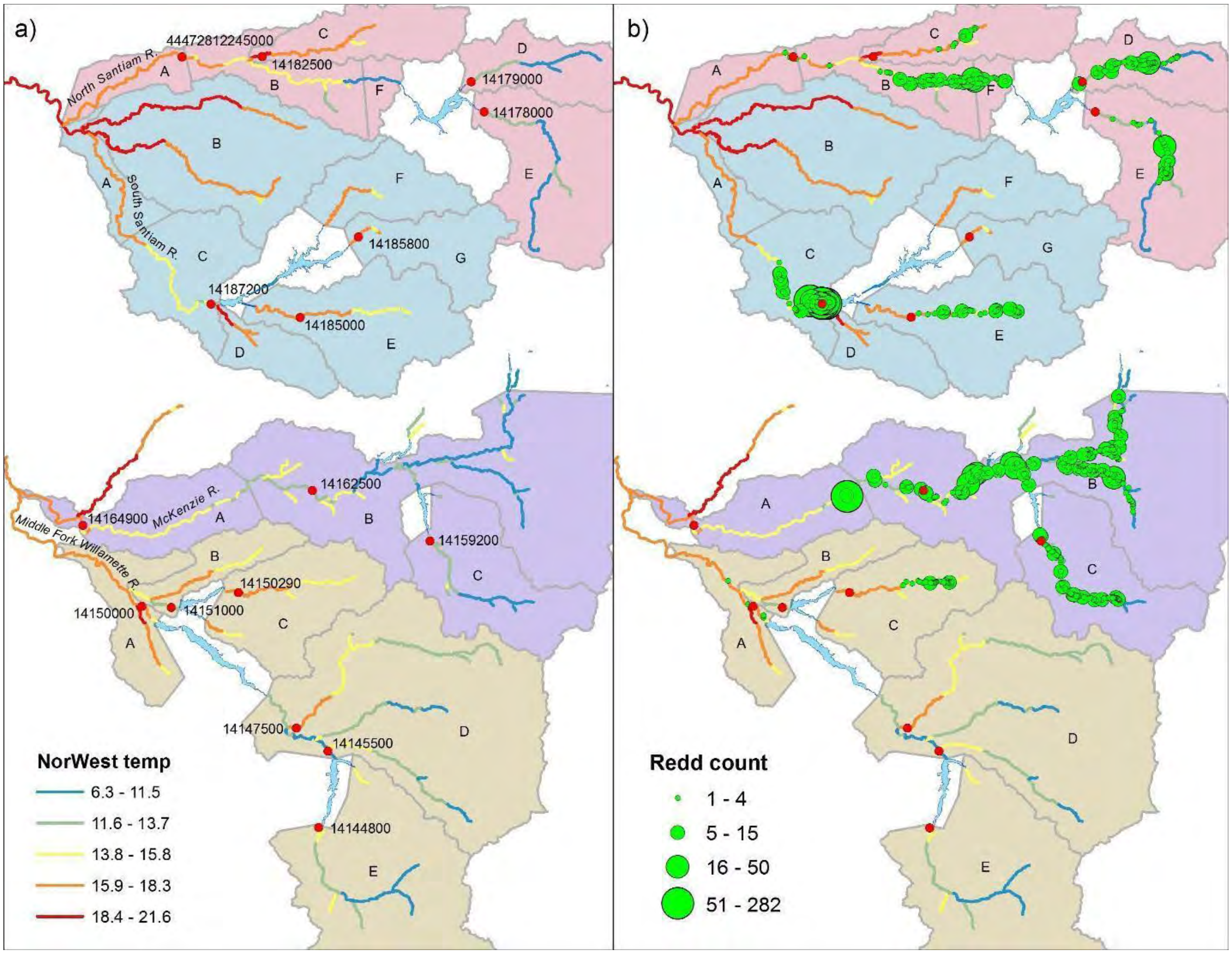


Figure 7-34. Map of the study area showing unique tributaries (shading) and reaches (letters). *The colors of the streams represent mean August water temperature (°C, 1993-2011) predicted by the NorWeST model for 1-km stream segment (Isaak et al. 2010). Red points are the locations of USGS gauges (numbered) that measure stream temperature and discharge, used in models of PSM and historical stream temperature. The green points on panel (b) show the location of redds georeferenced during spawning ground surveys in 2016.*

Temperature modeling– Water temperatures were available only from USGS gauging stations in recent years, but we wanted time series of reach-specific stream temperatures that corresponded with water years referenced in time series of other life-cycle model inputs (e.g. the time series of alternative dam passage efficiencies and dam survivals from the FBW). Thus, conditions in particular water years would be carried across to various life cycle model inputs, and covariability across inputs would account for decadal-scale fluctuations in these climate-driven processes. Therefore, we developed statistical models to predict historical water temperatures at USGS gauging sites, based on environmental variables measured since at least 1936 (Moore 1967, Bottom et al. 2011, Overman 2017). The models were fit to yearly averages of daily mean stream temperatures between 1 July and 15 September in all years in which data were available for ≥ 69 daily values out of the 77 days for a given site. Missing 8 of 77 days of data didn't appear to significantly affect averages and allowed for inclusion of more years of data with which to train models. We excluded years prior to 2007 in the training data for the model of USGS gauge #444728122450000 in the lower North Santiam, because the USACE initiated new operational temperature controls at Detroit Dam and we wanted the model to reflect current operations. We used the following candidate set of explanatory variables: average discharge from July through September at several USGS stations that had extensive historical data records, and seasonal averages of air temperature and precipitation at Eugene and Salem. These variables were standardized to have a mean of zero and a standard deviation of one. We filled an eleven- year gap in the time series of discharge at the gauge on the Breitenbush River in 1988–1998 based on its correlation with discharge at the gauge on the upper North Santiam River (correlation = 0.92). We used the 'dredge' function to assess data support for candidate models based on AIC with a correction for small sample size and limited the number of explanatory variables included in models to ≤ 2 in order to prevent overfitting. We selected the best model for each gauge based on AICc.

Our models captured a reasonable amount of interannual variability in average July–September water temperature at gauging stations (Tables 8.2 – 8.5), with the exception of two below-dam gauges. The two exceptions, gauges #14187200 below Foster Dam and #14151000 below Fall Creek Dam, were very close to dams and were presumably more affected by releases of cold water from reservoirs than the other gauges. Releases of water for temperature control, flood control, maintaining minimum flow, or hydropower (Foster Dam only) are dependent on management priorities and difficult to predict into the future. The utility of cold-water releases to provide thermal refuges for fish below dams is an area of active research (S. Rounds, Willamette Science Review 2018); however, the availability of cold water for temperature control is dependent on seasonal climatic conditions (precipitation and temperature) and operational demands.

Table 7-6. North Santiam Model Coefficients for 1 July – 15 September *

USGS Gauge	444728122450000 Below Bennet Dam	444728122450000 Above Bennet Dam	14182500 Little North Santiam	14179000 Breitenbush	14178000 Upper North Santiam	444728122450000 Minto/Big Cliff
Intercept	15.8	15.8	18.96	12.19	12.39	15.8
Little N. Santiam River near Mehama, OR flow	-	-	-	-	0.13	-
No Santiam R below Boulder Cr, near Detroit, OR flow	-	-	-	-	-	-
Breitenbush R above French Cr near Detroit, OR flow	-	-	-	-0.45	-0.6-	-

USGS Gauge	444728122450000 Below Bennet Dam	444728122450000 Above Bennet Dam	14182500 Little North Santiam	14179000 Breitenbush	14178000 Upper North Santiam	444728122450000 Minto/Big Cliff
So. Santiam River below Cascadia, OR flow	-	-	0.84	-	-	-
McKenzie River near Vida, OR flow	-	-	-	-0.42	-	-
MF Willamette River above Salt Cr, near Oakridge, OR flow	-	-	-	-	-	-
Salem air temp Mar-May	0.9	0.9	0.49	-	-	0.9
Eugene air temp Mar-May -	-	-	-	-	-	-
Salem air temp Jun-Aug -	-	-	-	-	-	-
Eugene air temp Jun-Aug -	-	-	-	-	-	-
Salem precip. Mar-May	0.41	0.41	-	-	-	0.41
Eugene precip. Nov-Feb-	-	-	-	-	-	-
Eugene precip. Jun-Aug -	-	-	-	-	-	-
Residual standard error	0.37	0.37	0.38	0.32	0.27	0.37
R^2	0.92	0.92	0.88	0.85	0.83	0.92

Note: * Coefficients for North Santiam models of reach-specific average 1 July – 15 September mean of daily means water temperatures, where flows used from the USGS gaging sites were during this same summer period.

Table 7-7. South Santiam Model Coefficients for 1 July – 15 September *

USGS Gauge	**14187200 Below Lebanon Dam**	**14182500 Thomas/Crabtree**	**14187200 Below Foster Dam**	**14182500 Wiley Creek**	**14185000 Above Foster Dam**	**14185800 Above Green Peter**	**14185800 Above Green Peter**
Intercept	12.33	18.96	12.33	18.96	16.73	16.71	16.71
Little N. Santiam River near Mehama, OR flow	-	-	-	-	-	-	
No Santiam R below Boulder Cr, near Detroit, OR flow	-0.45	-	-0.45		-	-	
Breitenbush R above French Cr near Detroit, OR flow	-	-	-	-	-	-	
So. Santiam River below Cascadia, OR flow	0.34	-0.84	0.34	-0.84	-0.71	-	
McKenzie River near Vida, OR flow	-	-	-	-	-	-	
MF Willamette River above Salt Cr, near Oakridge, OR flow	-	-	-	-	-	-	
Salem air temp Mar-May	-	0.49	-	0.49	-	-	-
Eugene air temp Mar-May -	-	-	-	-	-	-	-
Salem air temp Jun-Aug -	-	-	-	-	0.29	0.66	0.66
Eugene air temp Jun-Aug -	-	-	-	-	-	-	
Salem precip. Mar-May	-	-	-	-	-	-	
Eugene precip. Nov-Feb-	-	-	-	-	-	-	
Eugene precip. Jun-Aug -	-	-	-	-	-	-	
Residual standard error	0.45	0.38	0.45	0.38	0.28	0.39	0.39
R^2	0.34	0.88	0.34	0.88	0.9	0.86	0.86

Note: * Coefficients for South Santiam models of reach-specific average 1 July – 15 September mean of daily means water temperatures, where flows used from the USGS gaging sites were during this same summer period.

Table 7-8. Coefficients for McKenzie River Models 1 July – 15 September *

USGS gauge	14164900 Below Leaburg Dam	14162500 Above Leaburg Dam	14159200 Above Cougar Dam
Intercept	15.74	12.44	10.91
Little N. Santiam River near Mehama, OR flow	-	-	-
No Santiam R below Boulder Cr, near Detroit, OR flow	-	-	-0.29
Breitenbush R above French Cr near Detroit, OR flow	-	-	-
So. Santiam River below Cascadia, OR flow-	-	-	-
McKenzie River near Vida,OR flow	-0.84	-0.49	-
MF Willamette River above	-	-	-
Salt Cr, near Oakridge, OR flow	-	-	-
Salem air temp Mar-May	-	-	-
Eugene air temp Mar-May	-	-	-
Salem air temp Jun-Aug	-	-	-
Eugene air temp Jun-Aug	-	-	-
Salem precip. Mar-May	-	-	-
Eugene precip. Nov-Feb	-	-	0.12
Eugene precip. Jun-Aug	-	-0.33	-
Residual standard error	0.21	0.38	0.25
R^2	0.95	0.78	0.59

Note: * Coefficients for Mckenzie River models of reach-specific average 1 July – 15 September mean of daily means water temperatures, where flows used from the USGS gaging sites were during this same summer period.

Table 7-9. Middle Fork Willamette River Model Coefficients for 1 July – 15 September *

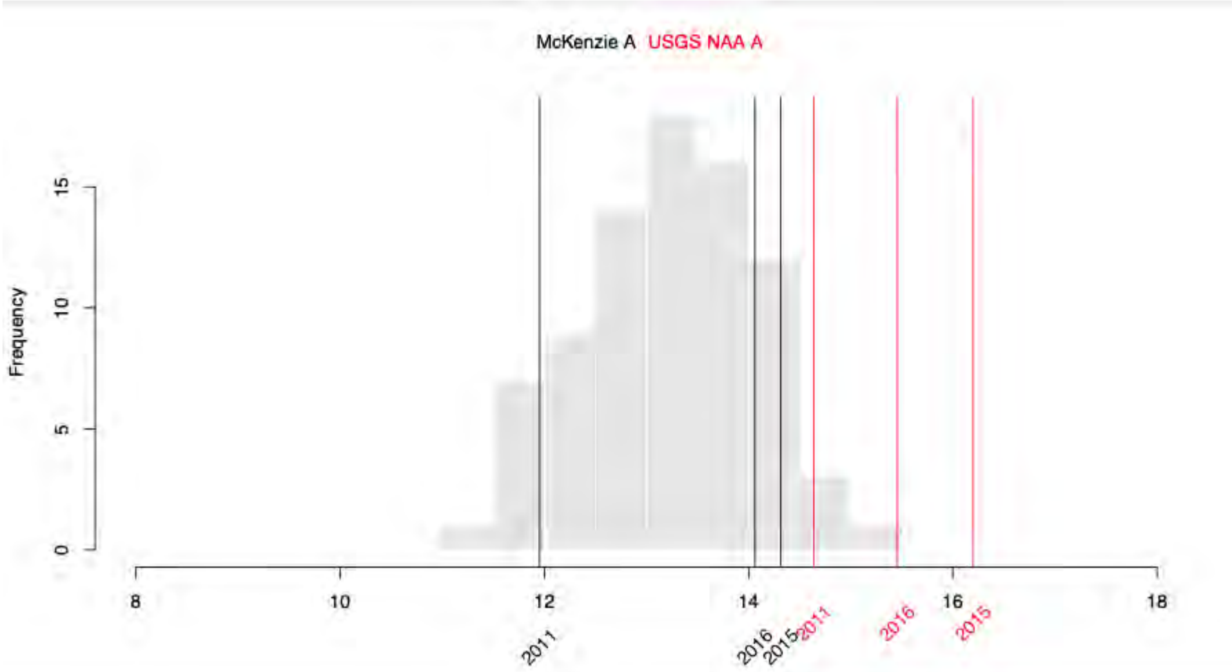
USGS gauge	14150000 Below Dexter Dam	14151000 Below Fall Creek Dam	14150290 Above Fall Creek Dam	14147500 North Fork Middle Fork	14144800 Above Hills Creek Dam
Intercept	14.93	13.02	17.38	16.57	13.97
Little N. Santiam River near Mehama, OR flow		-	-	-	-
No Santiam R below Boulder Cr, near Detroit, OR flow		-	-	-	-1.24
Breitenbush R above French Cr near Detroit, OR flow		-	-	-	-
So. Santiam River below Cascadia, OR flow		-	-	-	0.75
McKenzie River near Vida, OR flow	-	-	-	-	-
MF Willamette River above Salt Cr, near Oakridge, OR flow	-	0.55	-	-	-
Salem air temp Mar-May	0.98	-	-	-	-
Eugene air temp Mar-May	-	-0.45	-	0.4	-
Salem air temp Jun-Aug	-	-	-	-	-
Eugene air temp Jun-Aug	-	-	0.71	-	-
Salem precip. Mar-May	-	-	-	-	-
Eugene precip. Nov-Feb	-	-	-	-	-
Eugene precip. Jun-Aug	-0.86	-	-	-	-
Residual standard error	0.98	1.5	0.38	0.32	0.42
R^2	0.63	0.23	0.9	0.81	0.78

Note: * Coefficients for Middle Fork models of reach-specific average 1 July – 15 September mean of daily means water temperatures, where flows used from the USGS gaging sites were during this same summer period.

We compared the degree to which estimates of *en route* and prespawn mortalities covaried across the time window of our estimation procedure. *En route* mortality was positively correlated with prespawn mortality in all but two reaches between 1936 and 2016 in all but two reaches: below Fall Creek and below Bennett Dam (mean = 0.23, range = 0.02-0.52). Thus, these two demographic rates (PSM and en route mortality) do not generally compensate for one another in a given year. Furthermore, *en route* mortality and prespawn mortality in all reaches except below Fall Creek negatively correlated with millions of acre feet (MAF) from the Fish Benefit Workbook (-0.69). Lower MAF results in lower rates of juvenile survival and passage at dams, so it appears that climatic conditions influence multiple adult and juvenile demographic rates in the same direction in a given year.

7.8.2 Water Temperature adjustments for alternatives

Each EIS alternative was estimated Upper Willamette River Life Cycle Modeling to have some potential to alter water temperatures in reaches below dams compared to the NAA, where water temperature changes arised from either structural and/or operational actions at the dams. Water temperature changes associated with the EIS alternatives were incorporated into the Chinook salmon life cycle models in the survival relationship of adults holding in the prespawning period as described above, where prespawning survival was a function of the composition (NORs, HORs) of the spawning stock and summertime water temperatures (mean of daily means from 1 July through 15 September). For the draft EIS, the USGS estimated water temperatures from the spring – fall period for each of the EIS alternatives, including the NAA, in reaches below each of the dams considered in the EIS analysis (USACE 2022). Our working assumptions were that the water temperatures described above (both those observed and our modeled estimates) in reaches below dams, that although were cooler than the USGS estimates (e.g., Figure 8.3), were roughly analogous to stream conditions in the USGS NAA temperature modeling, and water temperatures were assumed to change in the same direction and magnitude as the change from NAA from the USGS temperature modeling as a consequence of the alternatives. Thus, we applied a 'delta' – the difference in water temperatures from the NAA and the alternatives as modeled by USGS for the draft EIS – to our observed and model-estimated water temperatures as described above.



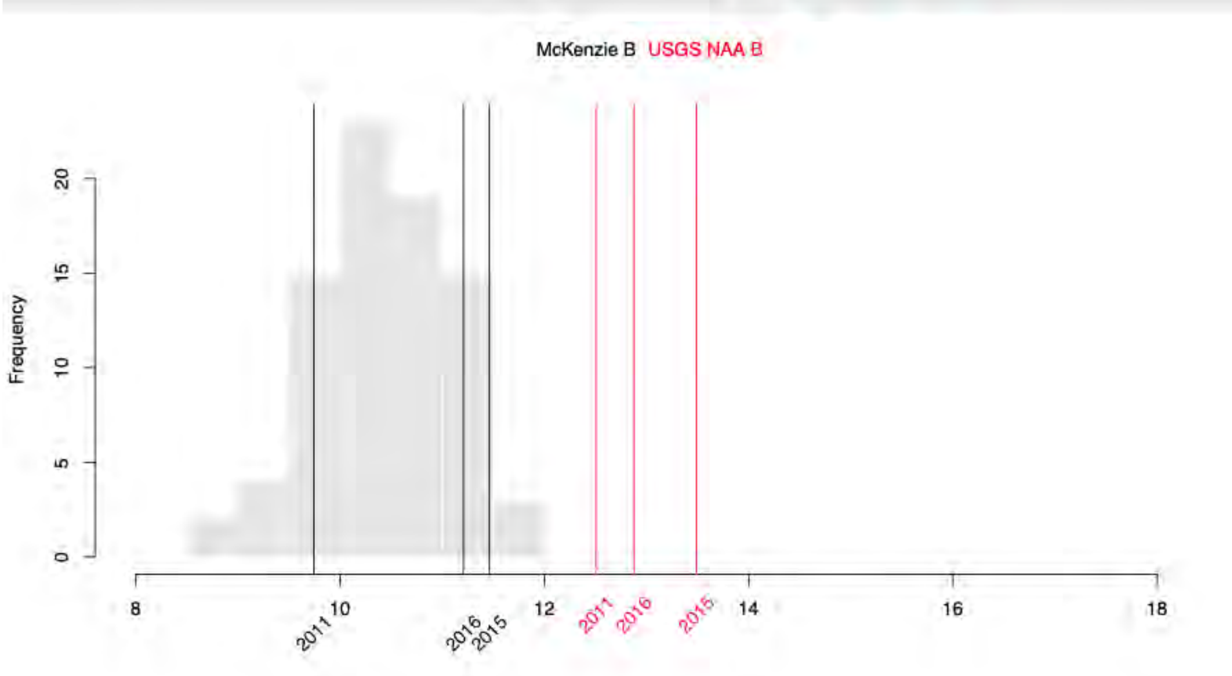


Figure 7-35. Comparison of observed July - September temperatures near McKenzie in the study area during the modeling years.

The plot shows a comparison of observed water temperatures (°C) of 1 July – 15 September mean of daily means used in the PSM modeling as described above (black lines are for the years 2011, 2015, and 2016, and the summer mean of daily mean temperature frequencies are for all water years of observations and model-estimated from our PSM modeling), with the USGS temperature modeling results (red lines) of the NAA conditions for the three water years during this PSM summer period. In the reaches below Cougar Dam (reaches 'A' and 'B'; top and bottom, respectively) in the McKenzie River Basin, observedcombined with the PSM model-estimated and the USGS-modeled temperatures were roughly similar, but the USGS-estimated temperatures were higher than the PSM modeling temperatures (observed and model-estimated) for the three water years of the USGS modeling.

Applying the alternative-specific water temperature differences (the 'delta') to the temperatures used by the life cycle models required four steps: 1) matching the USGS point estimates of temperatures to the NOAA LCM reaches; 2) calculating the prespawn period temperatures in the USGS estimates; 3) calculating the USGS temperature differences between NAA and each alternative for each of the reaches; and, 4) matching the three water years that USGS estimated to each of the years in the FBW range of water years. First, we matched the spatial extent of the NOAA LCM reaches to the stream points of the USGS locations of temperature estimates. Then, we calculated the mean of daily means for the 1 July – 15 September period for each alternative for each of the three years USGS estimated water temperatures. The temperatures estimated by USGS included daily estimates for three water years, 2011, 2015, and 2016. NAA temperature estimates were subtracted from temperatures for each alternative for each reach for each of the three water years (Table 8.6). This was used as the 'delta' to adjust the above-described water temperatures according to the corresponding EIS alternatives. The fourth step involved matching the three water years of the USGS temperature modeling to the many more water years represented in the FBW to apply the temperature adjustments to roughly similar water year types. We used the categorization of water year types found in the FBW, where there were four designations, presumably based on the type of water year and water availability in the system ('abundant'; 'adequate'; 'insufficient'; and, 'deficit'), to apply the water temperature changes, or 'delta,' from the USGS modeling. We applied the changes to FBW water years according to this key: 'abundant' were assigned the delta from water year 2011 temperature; 'adequate' water years were assigned the temperature delta from water year 2016; and, both 'insufficient' and 'deficit' were assigned the water temperate delta from water year 2015. Thus, for each alternative, we applied the changes to below-dam reach water temperatures according to the changes from NAA from the USGS water temperatures.

Table 8.6: The following are temperature differences from NAA (ALTx – NAA), where positive values represent temperature increases from NAA, and negative values represent decreases from NAA.

	McKenzie River below CGR	Middle Fork Willamette River below LOP/DEX	North Santiam River below DET/BCL	South Santiam River below FOS

Figure 7-36. Water temperatures from the USGS NAA modeling, and temperature changes ('delta') by reach for Willamette Valley System EIS alternatives (°C).

		A	A	A	B	F	A	C
	2011	14.64	14.04	11.74	11.02	10.25	16.69	14.12
NAA (USGS)	2015	16.20	19.78	15.90	14.84	13.74	21.29	19.18
	2016	15.46	17.04	13.72	12.67	11.57	17.57	14.94
	2011	0.01	2.55	0.61	0.71	0.85	2.41	3.22
ALT1 - NAA	2015	-0.45	-0.38	-0.39	-0.27	-0.13	-0.09	0.42
	2016	-0.81	1.01	1.19	1.40	1.65	2.79	4.15
	2011	-0.03	0.16	0.54	0.65	0.81	0.73	1.78
ALT2a - NAA	2015	-0.55	-0.51	-0.42	-0.30	-0.16	-3.88	-4.60
	2016	-0.01	-0.24	0.84	1.14	1.49	1.38	1.82
	2011	-0.08	0.14	0.54	0.65	0.81	0.73	1.78
ALT2b - NAA	2015	-0.49	-0.71	-0.42	-0.30	-0.16	-3.88	-4.59
	2016	-0.11	-0.19	0.84	1.14	1.49	1.15	1.88
	2011	0.17	2.76	0.75	0.82	0.90	0.96	2.08
ALT3a - NAA	2015	-0.59	0.18	1.14	1.39	1.71	-3.76	-4.41
	2016	-0.20	1.77	2.02	2.25	2.61	1.26	2.05
	2011	0.35	2.92	-0.39	-0.32	-0.18	0.64	0.97
ALT3b - NAA	2015	-0.83	-0.58	-0.88	-0.86	-0.84	1.29	2.42
	2016	-0.32	-0.41	-0.20	-0.14	-0.04	3.49	5.42
	2011	-0.04	3.30	0.54	0.65	0.81	0.28	1.06
ALT4 - NAA	2015	-0.53	-0.51	-0.42	-0.30	-0.16	-3.91	-4.63
	2016	0.00	1.59	0.84	1.14	1.49	0.45	0.86

7.9 HATCHERY PROCESSES

7.9.1 Chinook Salmon Population Hatchery Component

All four subbasins (North Santiam, South Santiam, McKenzie, and Middle Fork Willamette) in the Upper Willamette Chinook Salmon ESU being modelled in this study contain Chinook salmon hatchery programs that collect and spawn returning adults and then incubate, rear, and release juveniles back into the respective basin. Winter-run steelhead hatchery programs in the North and South Santiam were discontinued in the 1990s. Summer-run steelhead (non-native) continue to be released and collected in the Santiam Basin. The influence of summer-run steelhead on winter-run steelhead populations is not directly being modeled, but their effects would be included in the baseline survival parameterization.

Returning hatchery-origin Chinook salmon adults largely return to the hatchery rack, where they are collected, used as broodstock for future hatchery generations or in some cases transferred above projects for reintroduction studies, or disposed of. Additionally, hatchery-origin Chinook salmon adults (identified via an adipose clip) are intercepted at Minto Dam8 (North Santiam), Foster Dam (South Santiam), and Fall Creek Dam (Middle Fork Willamette) and removed, such that only unmarked "natural-origin" Chinook salmon adults are allowed upstream. Some hatchery-origin adult Chinook salmon remain in the basin and spawn naturally. For most populations studied, the proportion of natural-origin adults on the spawning grounds has decreased in recent years due to improved juvenile release strategies and the removal of hatchery-origin fish where possible (Table 6-10).

Table 7-10. Five-year mean proportion natural-origin spring-run Chinook salmon adults on spawning grounds

Population	1995-1999	2000-2004	2005-2009	2010-2014	2015-2019
Clackamas River (above dam)	0.33	0.58	0.79	0.94	0.97
Willamette Falls (Adult return)	--	--	--	--	0.22
North Santiam River (below Big Cliff Dam)	--	--	0.33	0.26	0.26
South Santiam River	--	--	0.39	0.40	0.21
McKenzie River	--	--	0.64	0.55	0.57
Middle Fork Willamette River	--	--	--	0.08	0.07

Source: Ford et al. 2012.

The presence of hatchery-origin spawners affects a population in a number of ways. Firstly, hatchery-origin spawners are considered in the estimation of pre-spawning mortality (see pre-spawning mortality section). Secondly, the total number of spawners, both hatchery- and natural

8 Natural-origin Chinook salmon adults are passed above Minto Dam to ideally spawn in the Minto Dam to Big Cliff Dam reach. Hatchery-origin adult Chinook salmon are transported above Detroit Dam. Origins are considered in the spawner capacity for each of the basin reaches. Reproductive success for naturally spawning adults and the effects of hatchery domestication are computed for both natural-origin and hatchery-origin adults (see section 9.2 below), using an estimate of the long-term influence of hatchery production in the basin. Broodstock retained in the hatchery are used to produce juveniles for release. Production levels (if sufficient broodstock are available) are set through the Hatchery Genetic Management Plans. Releases of hatchery reared juveniles generally-occur in the fall and second spring (yearlings) for Chinook salmon, and in the spring (yearlings) for summer steelhead. While some hatchery reared fish may residualize, it is believed that most emigrate relatively quickly to the ocean and do not influence freshwater rearing capacity (Schreck et al. 1994).

Table 9.2. (NMFS 2019).

Table 7-11. Current releases of spring-run Chinook salmon and summer-run steelhead from hatcheries in the Upper Willamette Basin

Hatchery Program	Stocking Location	Numbers of smolts released
Spring-run Chinook Salmon	Molalla River	100,000
Spring-run Chinook Salmon	North Santiam River	704,000
Spring-run Chinook Salmon	South Santiam River	1,021,000
Spring-run Chinook Salmon	McKenzie River	605,000
Spring-run Chinook Salmon	Middle Fork Willamette River	2,200,000
Summer Steelhead	North Santiam River	121,000
Summer Steelhead	South Santiam River	121,000
Summer Steelhead	McKenzie River	108,000
Summer Steelhead	Middle Fork Willamette River	157,000

Fitness Effects from Domestication of Naturally Spawning Hatchery-Origin Salmonids

There have been a number of empirical and theoretical studies on the effects of hatchery propagation and rearing on the long-term fitness of salmonids. Domestication concerns the genetic and behavioral consequences of hatchery propagation, but not the subsequent interactions between hatchery-origin and natural-origin fish: competition, predation, disease transmission, etc. The effect on fitness from genetic changes due to domestication or outbreeding (introduction and introgression of fish from non-local populations) may persist long after hatchery programs are eliminated or modified. Most studies have attempted to understand the loss of local adaptation; however, little is known regarding the rate or potential for reestablishment of local adaptation in a population. Busack and Currens (1994) reviewed many of the potential consequences of hatchery operations on a naturally spawning population. More recently, the advent of molecular parentage techniques has enabled researchers to assess the relative reproductive fitness of hatchery-origin and naturally-produced salmon and steelhead spawning in situ.

Studies with anadromous O. mykiss have demonstrated both the short and long term effects of artificial propagation and rearing on fitness (Araki et al. 2007, Araki and Cooper 2009).

Similarly, the relative reproductive success of hatchery-origin coho salmon (O. kisutch) was generally lower for both 3-year-old males (62% of wild) and 3-year-old females (84% of wild), but higher for 2-year-old (jack) males (175% of wild) (Theriault et al. 2011). Alternatively, Ford et al. (2012) did not observe a dramatic reduction in the fitness of hatchery-reared Chinook salmon. Furthermore, Ford et al. (2012) concluded that the decrease appeared to be related to the earlier maturation of hatchery-reared males. Kostow et al. (2003) postulated that non-native summer-run steelhead in the Clackamas River exhibited a lower relative reproductive success compared to native winter-run steelhead; in addition, the presence of summer-run steelhead juveniles depressed the native steelhead populations through competition rather than genetic introgression. In contrast to these findings, (Hess et al. 2012) reported no significant difference in the reproductive success of naturally-spawning natural-origin and hatchery-origin summer-run Chinook salmon. It should be noted that the hatchery-reared fish in the Hess et al. (2012) study were the progeny of wild Chinook salmon from a stream system that has never had hatchery releases, in contrast to the Upper Willamette River Basin, where the majority of returning adults were hatchery reared for at least 50 years and some hatchery programs have been operating for over 100 years. Additionally, in the Hess study the difference between hatchery and natural- origin fish become non-significant only after excluding those fish that did not produce any returning adult progeny.

In initially developing the VSP scoring criteria, the UWLCR TRT adopted much of its quantification of hatchery influences from the All-H Hatchery Analyzer (AHA) model (HSRG 2017). The AHA model estimates the degree to which hatchery and domestication influences erode local adaptation, delay the rate of adaptation, or create a genetic load of recessive deleterious alleles, etc. It should be remembered that hatchery and domestication influences are one part of a multi-function process, where the effects (domestication, inbreeding, natural and artificial selection) are multiplicative rather than additive over time. Because the effects of domestication are retained in a fish’s genome, it is not possible to eliminate them simply by removing the fish from the hatchery environment. Instead, natural selective forces must reestablish the frequency of locally adapted genes (a process that can easily be protracted over many generations, depending on conditions).

The AHA model tries to simulate many of the effects of hatchery operations, and in doing so provides a useful indirect measure of the impact on diversity. Beyond domestication, the introduction of non-local genotypes (not addressed in the AHA model) is likely to have an erosive effect on locally-adapted life history traits and overall population fitness. In general, the AHA model concept focuses on the Proportionate Natural Influence (pNI), the relationship between the percent of hatchery origin natural spawners (pHOS) and the percent natural origin adults in the hatchery broodstock (pNOB), to determine the rate of domestication. The greater the proportion of natural-origin fish utilized as broodstock in the hatchery the slower the rate of hatchery domestication for the broodstock overall. Similarly, lowering the proportion of hatchery-origin fish that spawn in the wild decreases the frequency of “domesticated” genes entering the NOR segment of the population. pNI is calculated as pNOB/(pHOS + pNOB), with values near 1 indicating low levels of domestication and low values (near 0) indicating a strong domestication effect (Figure 9.1). pHOS is primarily a response variable dependent on the magnitude of hatchery production, differential survival of hatchery and natural-origin fish, and homing fidelity by hatchery fish to the hatchery, whereas pNOB is dependent on hatchery specific broodstock mating programs. Both pHOS and pNOB can be readily generated (in the form of a running average) from the Life Cycle model.

Table 7-12. Relative percent hatchery origin spawners (pHOS) and percent natural origin broodstock (pNOB) for Willamette River Hatcheries.

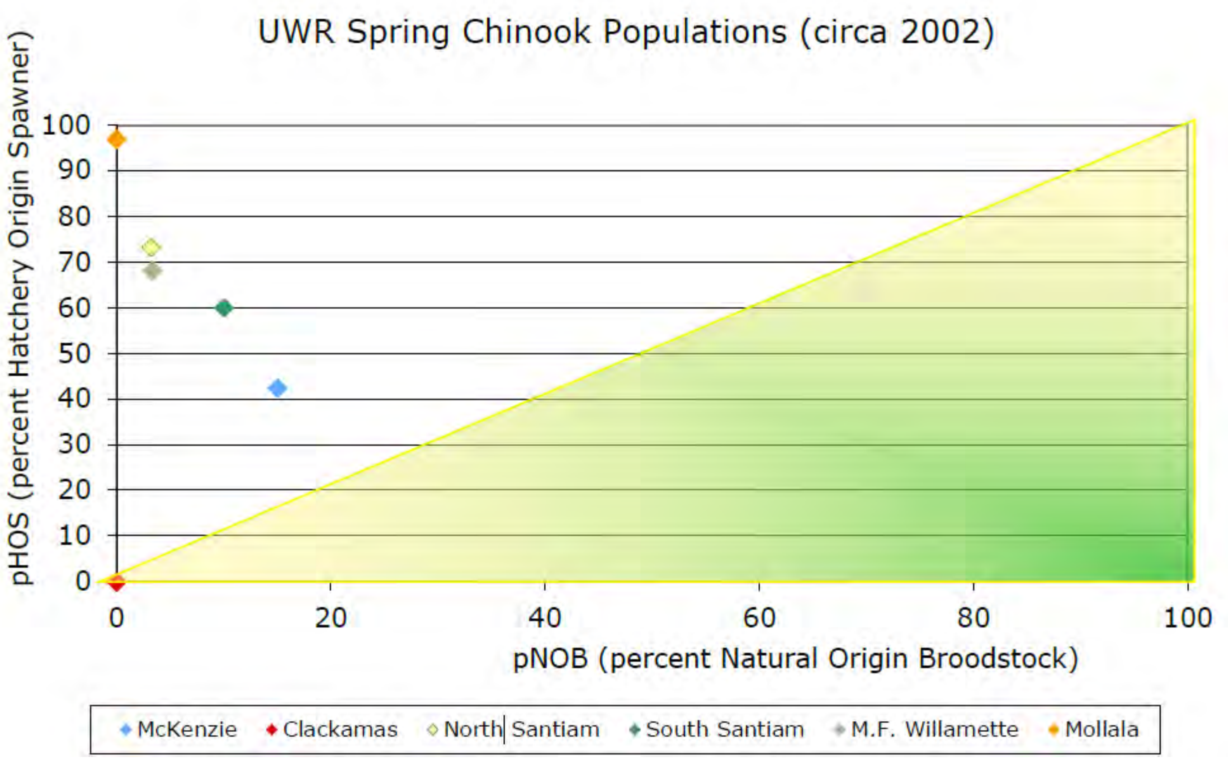


Figure 9.1: Where pHOS < pNOB, hatchery populations tend to become more natural over time (green shaded area). Where pHOS > pNOB, natural populations tend to become more domesticated.

The relative influence of pNI on diversity (expressed as a change in fitness) increases substantially as hatchery operations progress for several generations. On a single generation basis, the longer fish stay in a hatchery environment rather than in a natural environment, the greater the potential for domestication selection or the relaxation of natural selection. In general, hatchery-rearing protocols in the Upper Willamette are relatively standardized. Juveniles are released following 9 to 14 months of rearing in the autumn or second spring. Additionally, the release of fish “off station” has been minimized in basins with natural populations and some hatchery production has been transferred to Willamette basins without natural production.

Incorporating natural origin fish into hatchery broodstocks, while minimizing hatchery strays onto the natural spawning grounds provides the highest pNI. While using natural-origin fish as broodstock is desirable, it should be noted that “mining” natural populations can result in other diversity concerns: direct selection of temporal portions of the run, or severe reductions in the effective size of the natural spawning population.

We have utilized the relationship between pNI and fitness to develop a dynamic driver in the Life Cycle Model that adjusts the productivity/survival of naturally-spawning hatchery-origin fish. In the extreme case of pNI = 0, where there are no natural-origin fish in the hatchery population, hatchery fish would exhibit a 60% fitness reduction; whereas, under a no-hatchery option there would be no reduction.[9]

A linear relationship was plotted and the hatchery discount was applied at the survival from the egg to emergent fry stage. Thereafter for that brood year, naturally-produced fish from both hatchery and natural-origin parents were pooled in the model. pNI was calculated as:

$$pNI = \frac{pNOB}{pNOB + pHOS}$$

A running average for pNI was calculated to estimate the overall domestication status of the hatchery population. pNI was initialized using estimates provided by the HSRG for each hatchery program, and the initialize value was weighted to reflect 25-years of hatchery operations at the initial pNI to mimic the gradual process by which pNI would change in the population over several generations. Each year pNI was recalculated based on returning adults (Figure 9.2). This reflects that genetic selection for hatchery conditions or local adaptation is a relatively slow cumulative process and changes in hatchery practices may take several

[9] There have been a number of recent studies estimating the relative decline in fitness of artificially propagated fish. Of these, a review by Chilcote et al. (2011) estimated an 87.2% fitness reduction with a pNI=0, and provides an upper bound to the estimates. Other studies report a wide range of estimated reductions, with no consensus on the mode(s) of action or their magnitude. The 60% reduction used in the Chinook salmon Life Cycle model represents an interim estimate pending further studies.

generations to result in observable improvements in fitness. It is likely that locally-adapted gene complexes can be readily disrupted, but require considerable time to reestablish themselves.

Figure 9.2: Proportional adjustment for egg to fry survival for the progeny of hatchery-origin recruits (HOR) based on the percent natural influence (pNI) estimated for the population.

Initial simulations with the Life Cycle Model under conditions where pNI was low (< 0.25) still resulted in large numbers of NORs via the steady accumulation of hatchery-derived NORs (an outcome not observed empirically). In order to account for latent hatchery legacy effects a discount is applied to the reproductive success of NOR fish, also based on pNI (Figure 9.3). In the case of NORs, the discount is 20.0% at a maximum (pNI =0). This adjustment reflects the fact that many natural-origin Chinook salmon in the Upper Willamette River have at least one parent or grandparent that was of hatchery origin. In the North Santiam and Middle Fork Willamette River, where over 90% of the naturally-spawning fish are of hatchery-origin, it is very likely that the majority of non-marked spawning adults had at least one hatchery-origin parents.

Figure 9.3: Proportional adjustment for egg to fry survival for the progeny of natural-origin parents based on the percent natural influence (pNI) estimated for the population.

7.10 LIFE CYCLE MODEL CALIBRATIONS

7.10.1 Overview

In an effort to make the models as realistic as possible and to estimate parameters with few data available to inform them, we developed and applied a formal life cycle model calibration process. We chose a rejection sampling method that falls under the umbrella of Approximate Bayesian Computation (ABC; Beaumont 2010, Csilléry et al. 2010). Under rejection sampling, approximations of the parameters' posterior distributions and covariance structure are constructed with repeated sampling and simulation (Beaumont 2010, Hartig et al. 2011). Values of the parameters derived from ABC were drawn from these approximated multivariate posterior distributions in prospective life cycle model simulation iterations.

We were able to ABC-calibrate three of the Chinook salmon models in this report: McKenzie River and North and South Santiam Rivers. We did not have sufficient population-level spawner observations from the Middle Fork Chinook salmon or the winter steelhead populations to compare to model-generated outputs. For those life cycle models we implemented an ad hoc trial-and-error approach where we manipulated parameter values, guided where possible by the expert panel process (Zabel et al. 2015), until the models' outputs in NAA conditions fell within the ranges of observed data.

7.10.2 Methods

The ABC calibration procedure consisted of drawing a random set of parameter values from informative prior distributions (i.e., drawing parameter values from random uniform distributions, where each had specified minimums and maximums), running a single iteration (i.e., a single 100-year run) of the life cycle model with the alternative parameter sets, and comparing model outputs generated with the given parameter sets to observations for that population. Each unique parameter set was accepted (or rejected) if it was inside (outside) the deviation or difference between observed and model-generated spawners (Figure 10.1). We compared distributions of natural origin spawner abundances from the recent period annual observations with model-generated spawner abundances from 100 year simulations. We defined deviation as the Kolmogorov-Smirnoff (KS) statistic, D, which measured the degree to which the two distributions came from the same underlying distribution (Conover 1971). The KS test consisted of calculating the two-sample two-sided D statistic of a comparison of the two distributions: comparing 1) the distribution of spawning adults from each 100-year LCM iteration to 2) the distribution of recent (approximately 2005-2017 period) estimates of spawner abundance (Salmon Population Summary Database, https://www.webapps. nwfsc.noaa.gov/sps). We compared model outputs to contemporary observations so that the life cycle models would be calibrated to recent conditions and we used the NAA as our baseline environment and for dam operations (and its associated stream temperatures and dam survivals and passage efficiencies, etc.,) in the calibrations. We chose as our acceptance criterion the top 1% of KS D statistics of 50,000 iteration comparisons, which selected the parameter sets that generated model outputs that were most closely aligned to observations. The end product of the calibration process was a collection of accepted parameter sets (approximately 100-700 unique sets depending on the life cycle model that was calibrated) and each of the parameters' distributions approximated their posterior distributions. Note that because we accepted sets of parameters that produced the best fit, these sets also maintained a covariance structure among the parameters.

Despite the strict deviation criterion, which substantially restricted the number of accepted parameter sets to the top 1%, in some cases the parameters' posterior distributions appeared very similar to the prior distributions, and those parameters were dropped from the calibration sampling and their values were fixed rather than drawn from a distribution and were determined via the iterative LCM fitting procedure within ranges set according to the expert panel process (Zabel et al. 2015). After we dropped the insensitive parameters, we recalibrated with the reduced set of free parameters.

After the calibrations were completed, for prospective life cycle model iterations we randomly selected parameter sets with replacement from among the accepted sets. By sampling parameter sets (i.e., applying parameters jointly from the accepted sets), rather than sampling from each parameters' marginal distributions independently from each other, there was the potential to preserve covariance relationships, if present, among parameters.

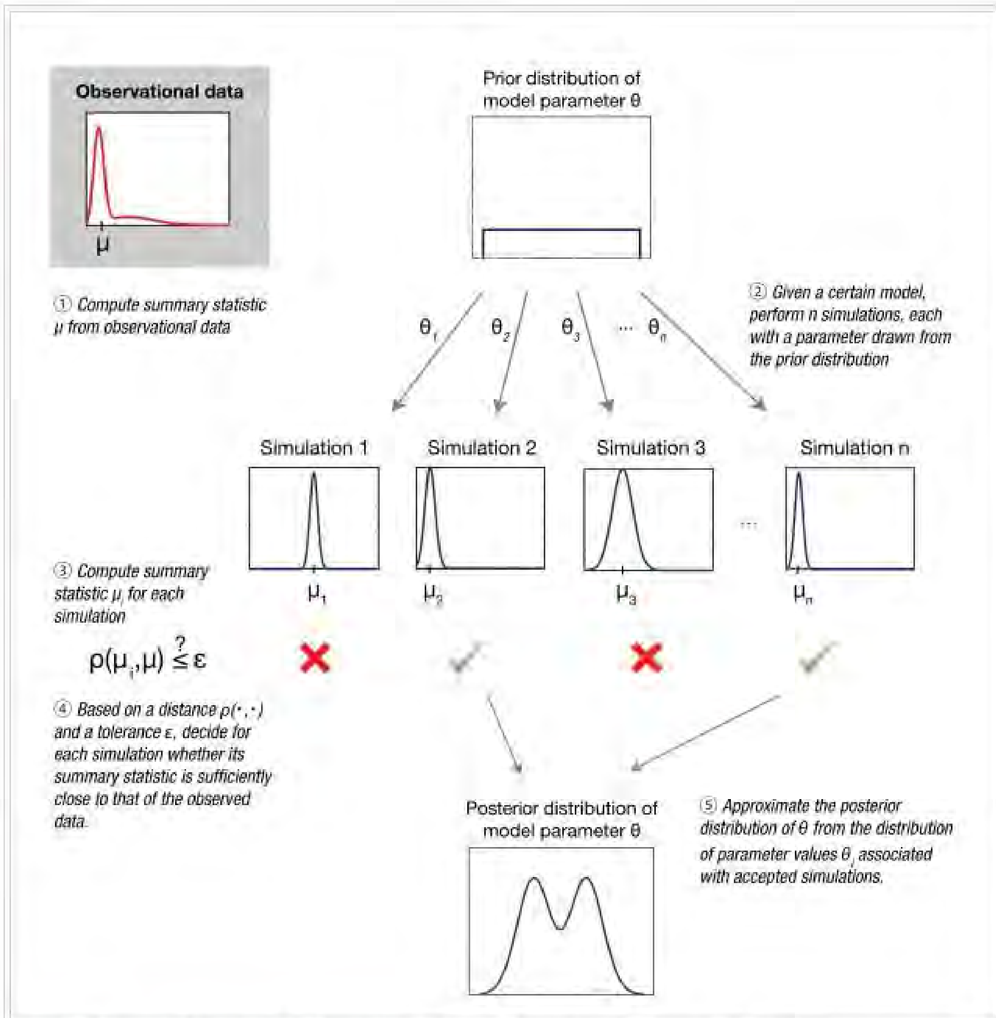


Figure 1. Parameter Estimation by Approximate Bayesian Computation: a conceptual overview

Figure 7-37. This diagram, Fig. 1 from (Sunnaker et al. 2013), illustrates the rejection sampling ABC process we followed in the life cycle model calibration.

7.10.3 Results

A check of the performance of the life cycle models as measured by the number of natural origin spawners they produced over repeated trials illustrated that median spawner abundances were very similar to medians of recent-period observed spawner abundances (Table 10.1). Post-ABC calibration, we slightly over- and under-estimated median spawner abundance for the North and South Santiam populations, respectively, and were very close to the observed number of spawners for the McKenzie Chinook salmon population.

Table 7-13. Calibrated life cycle model outputs vs observed natural origin spawners. *

Population	Observed median spawners	LCM median spawners
North Santiam Chinook salmon	498	625
South Santiam Chinook salmon	592	491
McKenzie Chinook salmon	1946	1941

Notes. * Comparisons of the life cycle models' outputs, natural origin spawners, to observed natural origin spawners after ABC calibration of parameters. Medians of observations were from approximately 2000 - 2017. Medians of life cycle models' spawner abundances, were calculated from 1000 iterations of 100 year each with ABC-calibrated parameters under NAA conditions.

Each of the three life cycle models' ABC-calibrated parameters' approximated posterior distributions are in Figures 10.2- 10.4. The ranges for the uniform priors were the minimums and maximums (x-axes) of the bars shown in each of the histograms. There appeared to be no strong relationships between parameters (Figures 10.5-10.7).

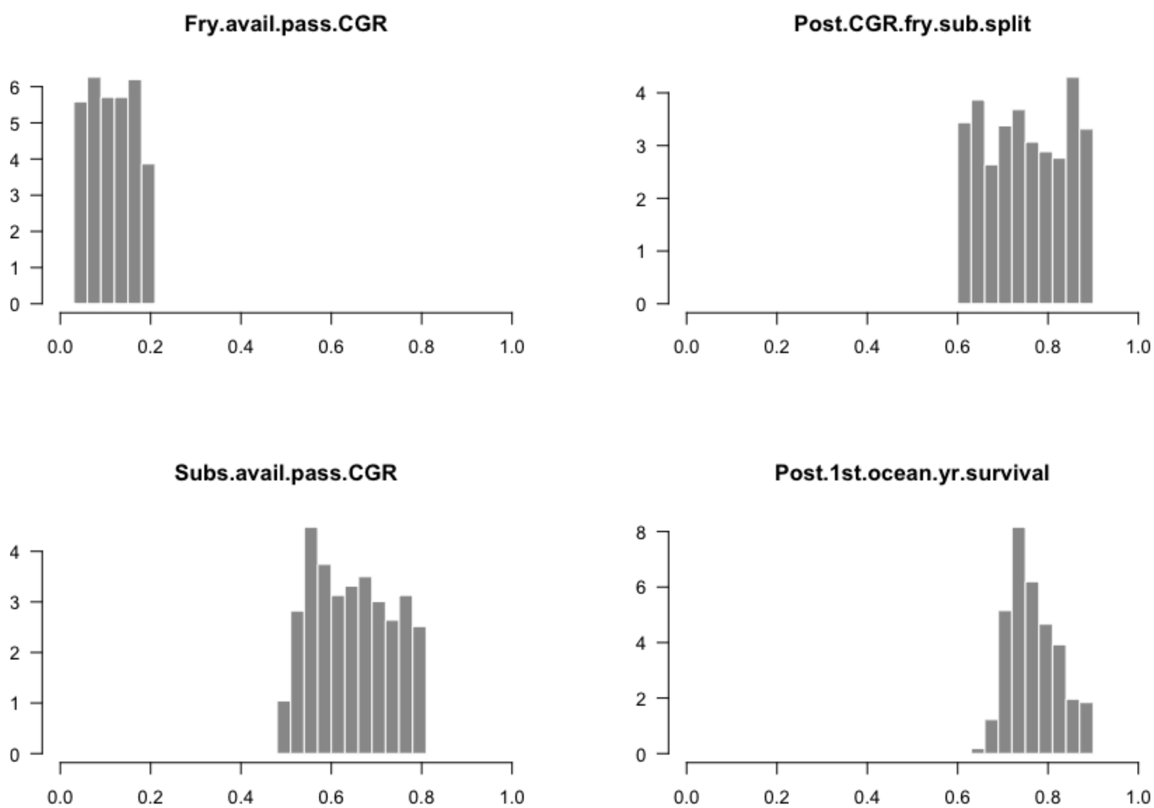


Figure 10.2: Distributions of parameters calibrated in the ABC process for the McKenzie Chinook salmon life cycle model included fry available to pass CGR Dam (top left), the split of post-CGR Dam fry remaining as fry (top right), subyearlings available to pass CGR Dam (bottom left), and annual ocean survival after the ocean entry year (bottom right).

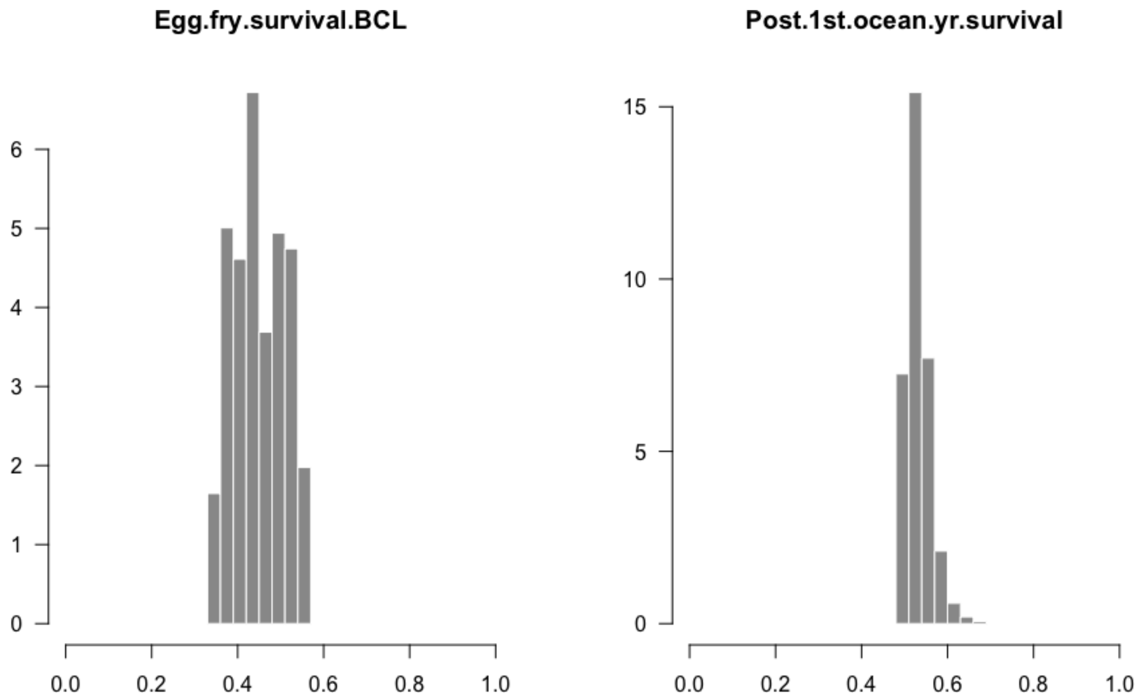


Figure 10.3: Distributions of North Santiam Chinook salmon ABC-calibrated parameters included egg-to-fry survival in the reaches below BCL Dam and annual ocean survival after the ocean entry year.

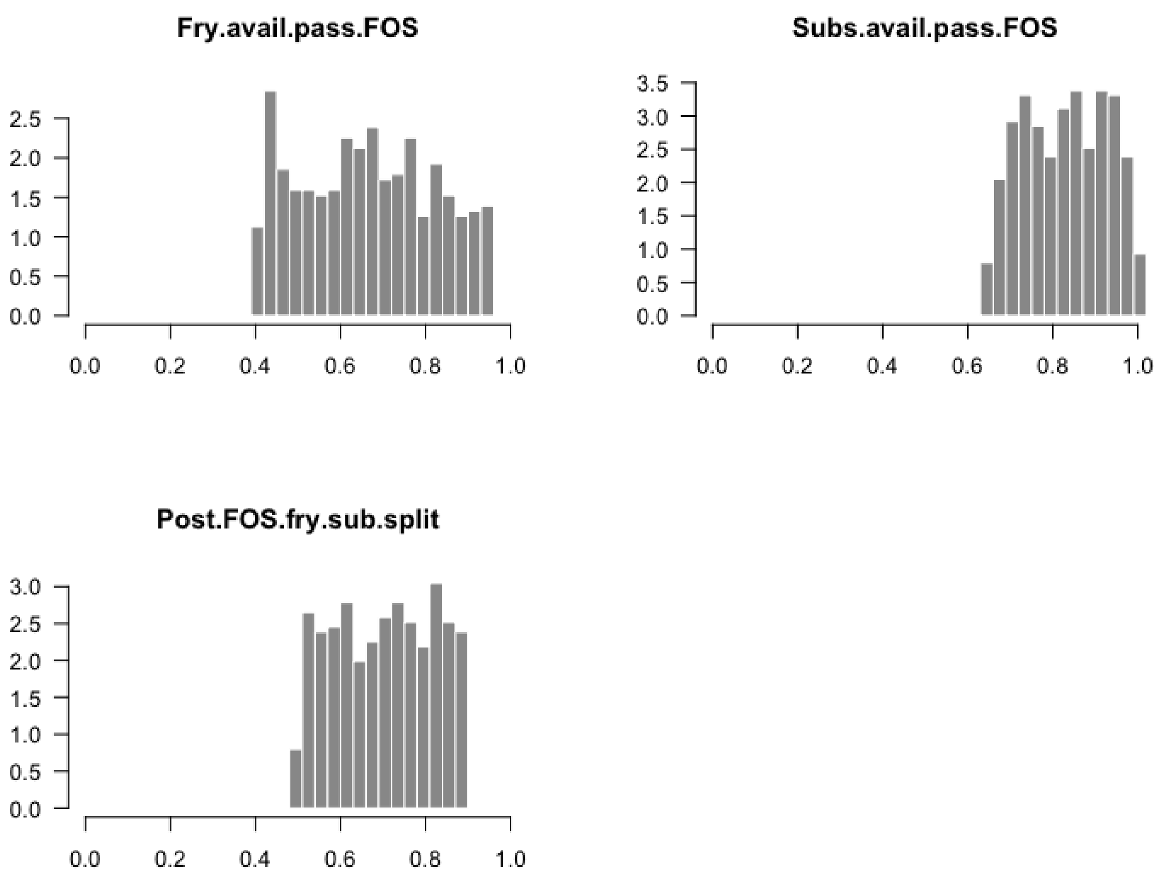


Figure 10.4: Distributions of South Santiam Chinook salmon life cycle model parameters that were ABC-calibrated included fry available to pass FOS Dam (top left), subyearlings available to pass FOS Dam (top right), and fry remaining as fry after passing FOS Dam (bottom left).

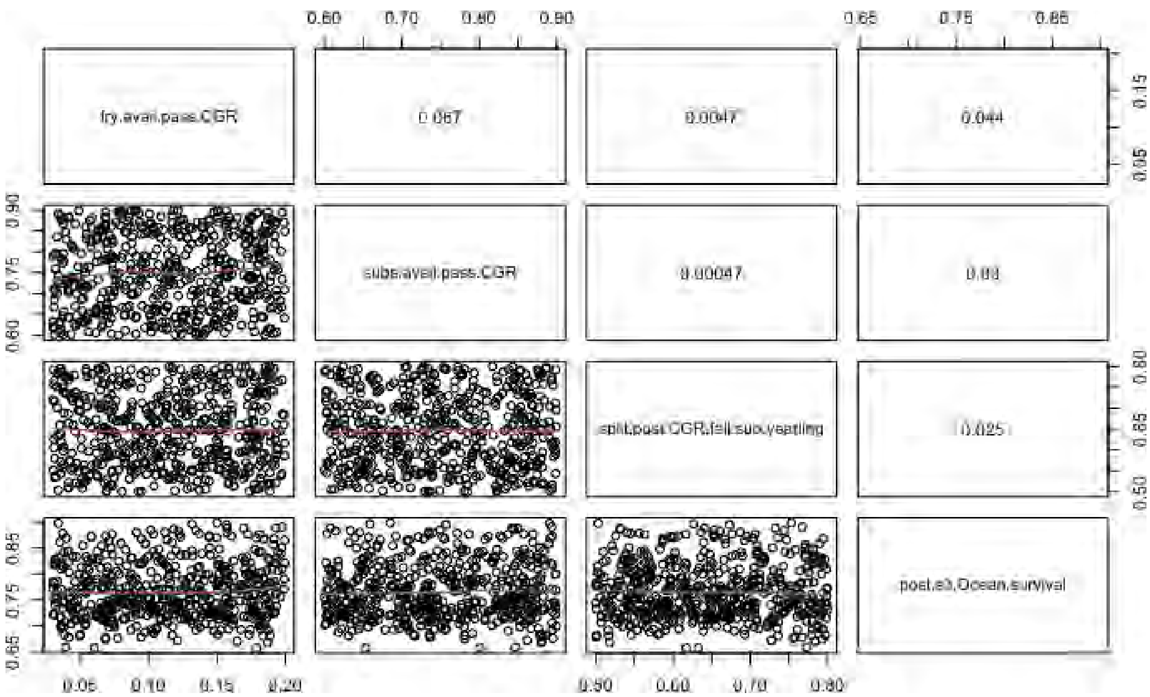


Figure 10.5: Estimated correlations between ABC-calibrated parameters for the McKenzie Chinook salmon life cycle model.

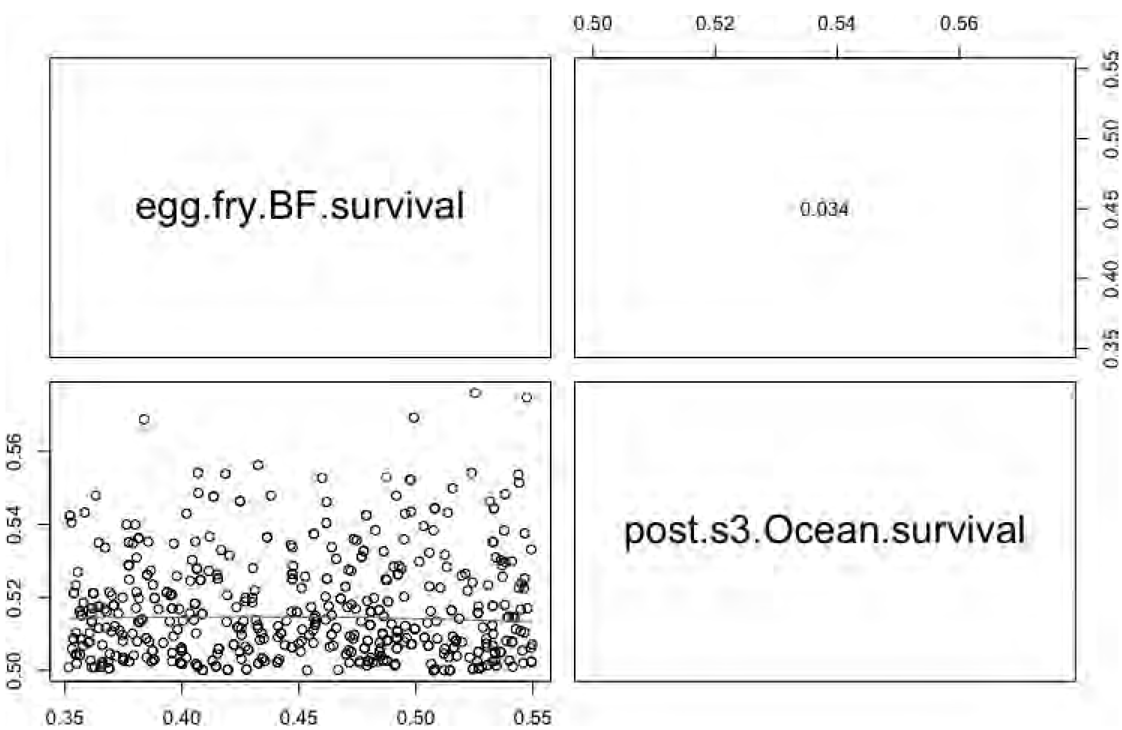


Figure 10.6: Estimated correlations of North Santiam Chinook salmon life cycle model's ABC-calibrated parameters.

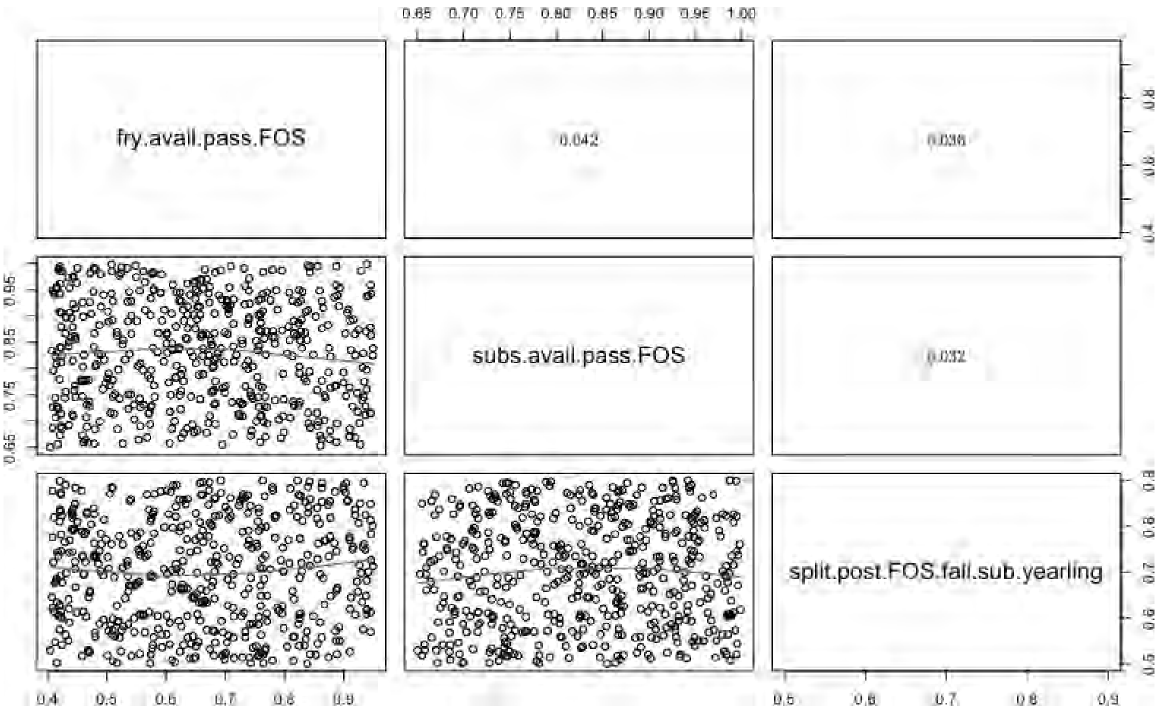


Figure 10.7: Estimated correlations of South Santiam Chinook salmon life cycle model's ABC-calibrated parameters.

7.10.4 Considerations

There were several considerations that guided our path through, and interpretation of, the calibration process. The focus of the calibrations was on parameters that were less informed by empirical data, such as those coming from the expert panel process. A larger set of parameters was included in the calibrations, and several were removed that had posterior distributions that essentially resembled their priors. We concluded that no information was gained about them through the calibrations. There is a balance between setting a sufficiently small deviation from observations such that information is gained from the ABC procedure and thereby will more closely approximate parameter posteriors, and setting too large of a deviation resulting in an approximated posterior that would resemble the prior. In our ABC process we experimented with different deviation levels (e.g., the top 1%, 5%, or 10% of the KS statistics) and compared posteriors with the priors and deduced that choosing the top 1% of the 50,000 iterations fell on the side of the potential for information gain to approximate the posteriors.

Another consideration was the amount and type of observations available for use in calibrations. Ideally, we would have benefited by having more years of spawner abundance observations over a more diverse period of record and its associated conditions. Also, it would be very helpful to include more information in the calibrations, by, for example, also including abundance estimates from another life stage, such as smolt abundances, to add an additional life cycle reference point for the calibration process.

Additionally, in the calibration process we compared model outputs generated from the FBW NAA to recent observations. However, contemporary spawner counts were a product of recent conditions and dam operations that might have deviated from NAA conditions. We felt justified in this approach because we modeled how the various alternatives varied from NAA.

7.11 LIFE CYCLE MODEL POPULATION DESCRIPTIONS

The development of the life cycle models for Willamette River populations was a two-stage process. The first stage was calibrating the model under the non-action alternative (NAA) where we estimated life stage demographic parameters for each population and fit the model to recent adult spawner abundances. A number of operational changes have been made at the dams in recent years, and adult return data more than 10 years old was thought to be largely uninformative. For passage alternatives, empirical data from previous short-term studies of operational passage or pilot structures was useful but insufficient to accurately capture the passage efficiency of dedicated, long term, operational passage or structural passage options.

For each population, we provide some general background information, with modeled reaches delineating ecological distinct areas generally separated by structures (dams), confluences, or cascades. Alternative scenarios included both FBW data for each of three juvenile life history stages for Chinook salmon and winter steelhead, but also associated files to accommodate alternate-specific changes in downstream water temperature and total dissolved gas concentration.

In reviewing the model results discussed in the next chapter, these additional considerations should be incorporated into any risk assessment. Similarly, caution should be used in analyzing the model output of population abundance as well as the probability of falling below the quasi-extinction threshold (pQET) for each of the scenarios. There is uncertainty in the estimation of model parameters and algorithms and this uncertainty can be compounded over an extended time period (100-year projection). Therefore, it may be more informative to focus on the relative outcome of each scenario, rather than focus on the absolute abundance generated.

7.11.1 North Santiam River

Chinook Salmon

The North Santiam River population of Chinook salmon presently exists below the Big Cliff/Detroit Dam complex. These dams block volitional passage to the majority of the historical spawning grounds. Currently, there is only an experimental “trap and haul” program to place predominantly hatchery origin adult Chinook salmon above Big Cliff Dam and Detroit Dam. In the absence of the trap and haul program, sufficient spawning habitat is accessible below the dam to currently sustain the population at an abundance of several hundred individuals, with the

2015-2019 geomean spawner estimate at 354 (Ford et al. 2022), a fraction of historical levels. Model Alternatives include both structural and operational passage options.

The North Santiam River is divided into six reaches: Reach A, from the confluence with the South Santiam to Bennett Dam; Reach B, from Bennett Dam to the Minto Fish Collection Facility; Reach C, the Little North Santiam River; Reach D, the Breitenbush River (above Detroit Dam); Reach E, the North Santiam River (above Detroit Dam); and Reach F, from the Minto Fish Collection Facility to Big Cliff Dam (Figure 11.1). Under current conditions (NAA) Reach A of the North Santiam River contains predominately low gradient habitat and experiences relatively high water temperatures during the adult migration and spawning period, with little potential for use as spawning habitat. Similarly, water temperatures in the Little North

Santiam River are currently near tolerance maxima for Chinook salmon adults. Most of the natural spawning occurs in Reach C and Reach F (from Bennett Dam to Big Cliff Dam). The Breitenbush River (Reach D) and North Santiam River (Reach E) are only accessible via the transportation of adults collected at the Minto Fish Collection Facility. Releases of hatchery-origin Chinook salmon juveniles into the North Santiam are currently 704,000 annually.

Currently, hatchery-origin adults comprise the majority of adults experimentally transported above Detroit Dam.

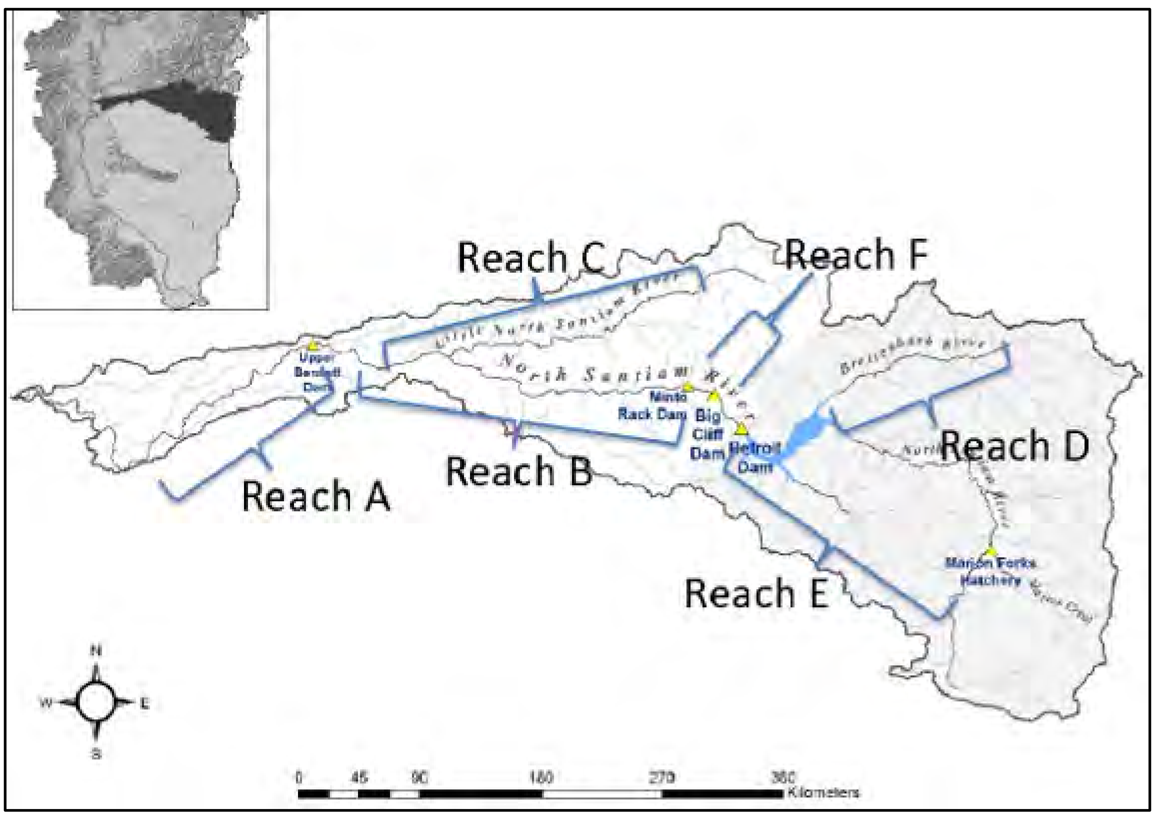


Figure 7-38. North Santiam River Basin with life cycle model reaches indicated. The shaded portion denotes habitat above a man-made barrier. Yellow triangles indicate dams.

Available historical information suggests that much of the historical spawning habitat is above the current location of Detroit Dam. Mattson (1948) estimated that 71% of the North Santiam Chinook salmon spawned above the Detroit Dam site. Similarly, Craig and Townsend (1946) reported that two-thirds of the salmon would spawn above the then proposed dam site, and only one-tenth of those spawning above the dam site utilized the area that would be inundated by the reservoir (between the mouth of the Breitenbush River and the dam site). Finally, Parkhurst et al. (1950) estimated that there was sufficient habitat in the North Santiam to accommodate at least 30,000 salmon adults, with much of this habitat below the dam site (although the preferred spawning areas were all above the dam site).

Winter Steelhead

Spawning steelhead are found throughout the basin below Big Cliff/Detroit Dam, although much of the historical spawning habitat was located above the Dams. Precise estimates of current steelhead spawning escapements are difficult to acquire given river conditions (high flows, turbid water) during March to May spawning period. Additionally, carcasses are not routinely recovered from steelhead. For this reason, the best estimate available for North Santiam steelhead is the adult count at Upper and Lower Bennett dams. It is unlikely that any significant spawning takes places below the Bennett dams. Dam counts since the late 1990s, when the influence of the terminated hatchery program was no longer a factor, have decreased overall, with a long-term (1998-2019) decline of 6.8% annually. Those steelhead that enter the Minto Fish Collection Facility are transferred above Minto Dam into the reach between Minto and Big Cliff dams, because there is no deterrent to steelhead moving downstream over Minto Dam, the counts of adults transferred may not be relevant. The short term (5-year) geometric mean for Bennett counts is 457, with a long term (1998-2019) geometric mean of 1190. A number of climatic factors (poor ocean conditions and poor freshwater conditions) in addition to marine mammal predation have been identified as potential causal factors in the overall decline of steelhead in the Upper Willamette Basin. There is currently no hatchery production for winter steelhead in the North Santiam Basin.

7.11.2 South Santiam River

Chinook Salmon

The Chinook salmon population in the South Santiam basin currently produces several hundred natural origin adults per year, which constitute about 30-40% of the total number of spawners in the basin. The South Santiam Basin contains two major impassable dams (Figure 11-2), a low-head dam (Foster Dam) that blocks volitional access to the upper South Santiam River (Reach E), but to which an existing trap and haul program provides upstream access, and a high-head dam (Green Peter Dam) that currently blocks access to Quartzville Creek (Reach F) and the Middle Santiam River (Reach G), for which there is no trap and haul program. There is limited natural spawning in the lower South Santiam River (Reach A), Thomas and Crabtree creeks (Reach B), and Wiley Creek (Reach D), with the majority of spawning occurring below Foster Dam. Reproductive success in many of the reaches below Foster Dam is likely limited by habitat degradation, although even historically the majority of the Chinook salmon spawning was above the site of Foster Dam (Mattson 1948, Parkhurst et al. 1950a). Presently, only unmarked fish are transported above Foster Dam. Chinook salmon hatchery releases into the South Santiam Basin average 1,021,000 juveniles annually.

Relatively high summer water temperatures dominate much of the South Santiam Basin; these high temperatures may negatively influence adult pre-spawn mortality, incubation success, and rearing capacity. Interestingly, the South Santiam River Reach that is least influenced by high water temperatures, Reach C, lies below Foster Dam where temperature increases are buffered by releases of deep cold water from Foster Reservoir, which in turn receives much of its colder waters from Green Peter Reservoir. In contrast to the North Santiam Basin (highly permeable geology with sustained base flows), the South Santiam Basin is of low permeability, tends to quickly transition precipitation to runoff, and is considered flashy in nature (Tague and Grant 2004). Alternatives include both structural and operation passage options at both Green Peter Dam and Foster Dam.

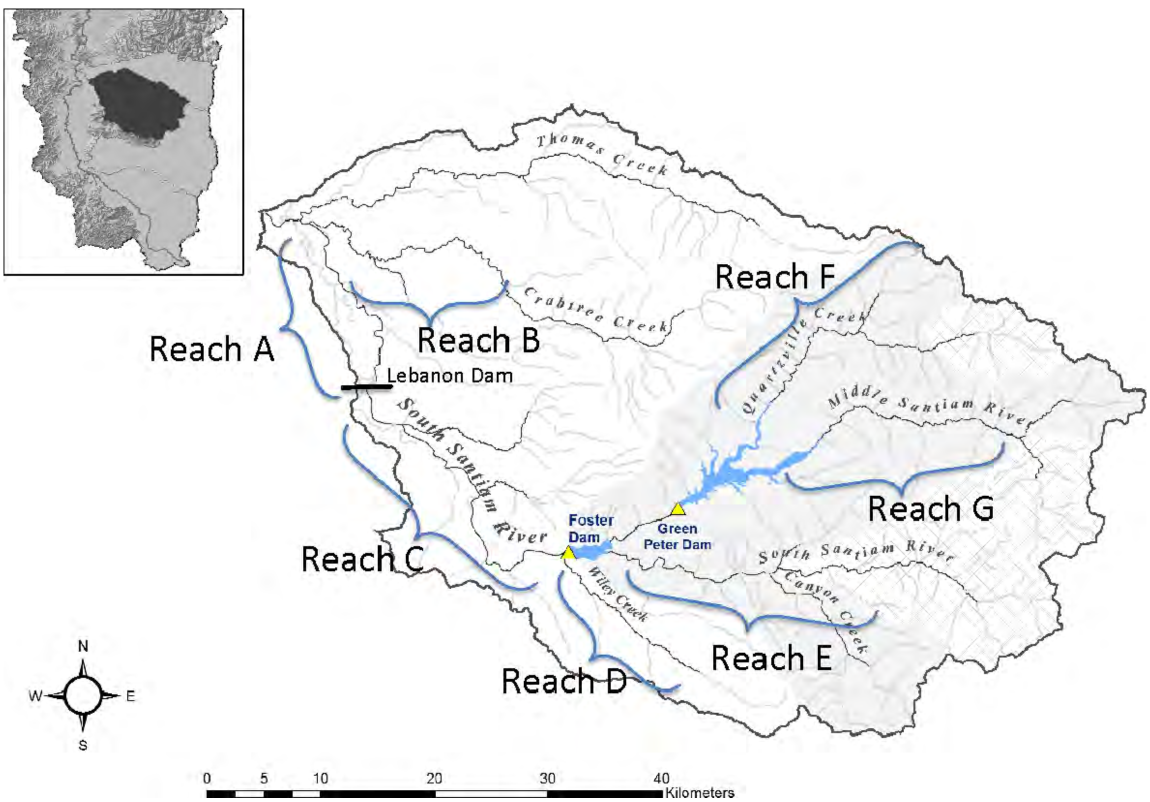


Figure 11-2. South Santiam River Basin with life cycle model reaches indicated. The shaded portion denotes habitat above a man-made barrier. Yellow triangles indicate dams.

Winter steelhead

Currently, there is natural steelhead production in all of the reaches, except the lowermost portion of the South Santiam (Reach A) and above Green Peter Dam (Reaches F and G). Reach E, the South Santiam River above Foster Dam, is accessible through a trap and haul program, with only unmarked fish being passed above. Survey data (index redd counts) are available for a number of tributaries to the South Santiam River; in addition, live counts are available for winter steelhead transported above Foster Dam. There is no basin-wide time series for South Santiam winter steelhead. Temporal differences in the index reaches surveyed and the conditions under which surveys were undertaken make the standardization of data among tributaries very difficult. For the Foster Dam time series, the most recent 10-year geometric mean (2010-2019) has been 141, with a negative trend in the abundance over those years (recognizing that the 2010 return reflected good ocean conditions and while recent years have been relatively poor). Longer time series are less meaningful, in that abundance estimates before 2009 were developed using different methodologies. Mapes et al. (2017) provides a comparison of survey methodologies.

Results suggest a population of around 1,000 spawning adults, except in 2017. Counts of winter- run steelhead at Willamette Falls indicate basin-wide depressed abundance in 2017, 2018 and 2019 (Ford et al. 2022); although the initial counts for 2020 suggest a return to more “normal” abundance. Willamette Falls winter-run steelhead counts include both early (non-native) and late (native) winter steelhead, while South Santiam River redd counts and Foster Dam counts primarily reflect native late-winter steelhead abundance. Historically, steelhead also spawned above Green Peter Dam. Wade et al. (1987) reported that in 1971 the wild winter steelhead count at Foster Dam was 4,254 fish (approximately one-quarter of the Willamette Falls count for that year). Wade et al (1987) further suggest that prior to the construction of Foster Dam, two- thirds of the steelhead passing the Foster Dam site were destined for the Middle Fork Santiam River to areas now above Green Peter Dam. On average, 2,600 steelhead passed the Foster Dam site prior to 1966. With the cessation of fish passage over Green Peter Dam in 1988, spawning is currently limited to the mainstem South Santiam, and Thomas, Crabtree, and Wiley Creeks.

There is not hatchery production of winter steelhead.

McKenzie River

Chinook Salmon

Long considered the flagship population in the Upper Willamette River Basin, the McKenzie River still provides volitional access to the majority of its historical spawning habitat.

Impassable dams in the Blue River (Blue River Dam), Upper McKenzie (Trail Bridge Dam), and South Fork McKenzie River (Cougar Dam), block historical spawning habitat for spring-run Chinook salmon. There are no plans currently to provide access to the upper Blue River, but adult “trap and haul” and juvenile bypass systems are currently being developed at the other sites. Overall, the McKenzie River population has the second largest natural origin population in the ESU (next to the Clackamas River), with a geometric mean for the 2015-2019 return years of 1,664 (Ford et al. 2022). This population has seen a decline in abundance over the last ten years, providing a further incentive to provide access to the South Fork McKenzie River.

Chinook salmon hatchery production into the McKenzie River basin is set at 605,000 juveniles, annually.

The McKenzie River is divided into three reaches for life cycle modeling purposes. Reach A includes mostly lower gradient river from the mouth of the McKenzie River to Leaburg Dam. This reach is considered marginal spawning habitat, but does account for a number of redds each year (Figure 11-3). Reach B is the most productive portion of the river, from Leaburg Dam to Trail Bridge Dam in the upper Basin, and including the lower portion of the South Fork McKenzie, below Cougar Dam. Reach C includes the South Fork McKenzie River and its tributaries above Cougar Dam.

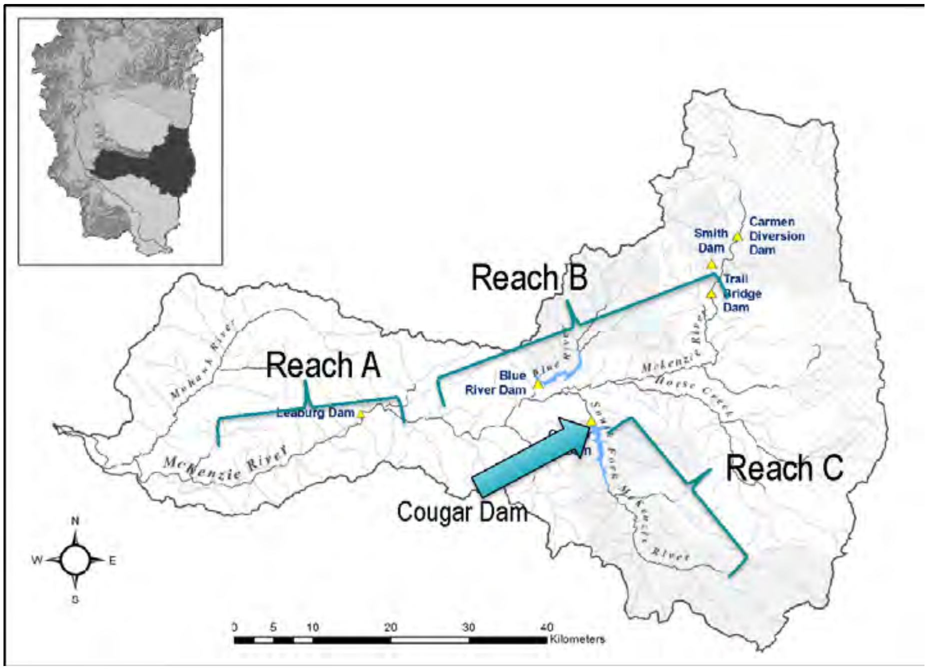


Figure 11-3. Map showing McKenzie River Basin and the three reaches (A, B, and C). Reach A corresponds to mainstem McKenzie River downstream of Leaburg Dam, Reach B corresponds to mainstem McKenzie River upstream of Leaburg Dam excluding South Fork McKenzie River upstream of Cougar Dam, and Reach C corresponds to South Fork McKenzie River upstream of Cougar Dam.

Middle Fork Willamette River

Chinook Salmon

The present day Middle Fork Willamette River population of Chinook salmon bears little resemblance to its historical counterpart in its distribution. Currently, almost all of the historical spawning and rearing habitat is volitionally inaccessible to Chinook salmon. Production by natural-origin adults is restricted to lowland low gradient unobstructed reaches and above Fall Creek Dam (where a trap and haul program currently provides access), and of which accounts for a few hundred returning adults at best. The geometric mean for the 2011-2016 period was only 351 (Fall Creek). Dexter and Lookout dams are a high head dam complex that obstructs passage beyond the lower Middle Fork Willamette River, and farther upstream beyond the North Fork Middle Fork Willamette River, Hills Creek Dam redundantly blocks access to the uppermost reach. Currently, predominately hatchery-origin adult Chinook salmon are transported above Dexter/Lookout and Hills Creek dams for research purposes. A very few juvenile Chinook salmon successfully migrate downstream over or through these high head dams. Returns of natural origin Chinook salmon to Dexter Trap have averaged 109 adults, annually (2015-2021). Although current productivity is low, historically the Middle Fork Willamette River was thought to be the major producer of Chinook salmon (Mattson 1948, Parkhurst et al. 1950). Historical and contemporary information on naturally-spawning Chinook salmon in the most productive Middle Fork Willamette River reaches (Reach D and E) is very limited, more so than for the other populations. Therefore, there was more uncertainty in the parameterization of the life cycle model for this population. Hatchery production in the MF Willamette River Basin is set at 2.2 million juveniles.

In the Middle Fork Willamette River, there were five reaches identified for life cycle modeling (Figure 11-4). Reaches A and B are the only free-flowing reaches in the basin; however, they are of limited value to salmon (low gradient with high summer temperatures). The upper watershed Reaches D and E benefit from the High Cascades geological (permeable basalt that provides ample groundwater) and snow melt (Gregory et al. 2007), but are blocked by high head dams. Similarly, Reach C (Upper Fall Creek) is blocked by Fall Creek Dam, although returning unmarked Chinook salmon are transported above the Dam. While the reservoir provides a thermal refuge from summer temperatures, Fall Creek and Winberry Creek that drain into the reservoir experience relatively high temperatures during adult pre-spawn holding. Little of the Fall Creek watershed lies in the snow zone (above 1200m), with most of this subbasin in the transitional or rain zone. Geologically it lies in the less permeable Western Cascade province (Tague and Grant 2004).

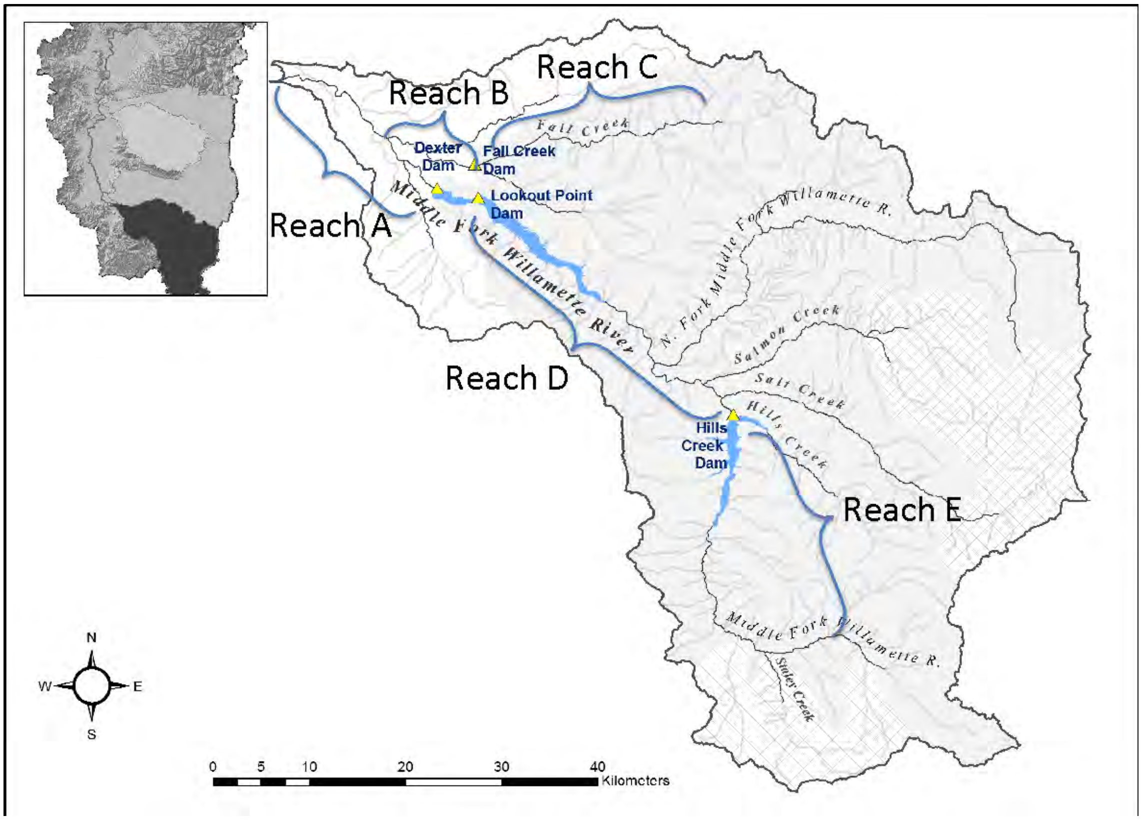


Figure 11-4. Middle Fork Willamette River reaches modeled: the mainstem Middle Fork Willamette below Dexter Dam (A), Fall Creek below Fall Creek Dam (B); Fall Creek above Fall Creek Dam (C); mainstem between Lookout Point and Hills Creek Dam (D), and mainstem above Hills Creek Dam (E). Shaded portions denote areas upstream from anthropogenic fish barriers; cross hatched areas denote areas upstream from natural fish barriers (historically inaccessible). Yellow triangles indicate dams.

7.12 LIFE CYCLE MODELING RESULTS

The following are results from the six life cycle models for each of the EIS alternatives described in the FBW. The results reflect outputs of LCMs where each was run for 1000 iterations, with 100 simulation years in each of the iterations.

The LCMs' results included total VSP scores, natural origin abundance, extinction risk measured by the probability of falling below a quasi-extinction threshold (prQET), abundance and productivity VSP scores, median replacement above dams (a measure of spatial structure), and VSP scores related to above-dam replacement. The Chinook salmon LCM outputs included additional elements: estimated proportion of hatchery-origin spawners (pHOS), pHOS VSP scores, and life history diversity as measured by smolt numbers produced from the three juvenile life history pathways and the corresponding life history VSP scores. Points show medians, and error bars show the 95% intervals around the medians.

Results Summary

The populations responded more strongly in a favorable direction to structural passage alternatives[10] compared to the NAA and to the operational alternatives, with bigger increases in natural-origin abundance, reduced extinction risk and pHOS, increasing life history diversity scores, and higher replacement above dams scores. While some of the operational alternatives generally provided better dam passage survivals and passage efficiencies than the NAA, their passage efficiencies weren't as high compared to the structural alternatives. Modest improvements in abundance due to operational passage programs still retained higher levels of prQET due to ocean and freshwater variability. In the case of North and South Santiam Chinook salmon and steelhead, estimated TDG levels below Big Cliff and Green Peter dams for the operational options were relatively high compared to the other dams and became an additional mortality factor for those populations. Water temperatures, modeled by USGS and applied as changes to observed temperatures, changed with the alternatives and had some impact on the populations in the form of prespawning mortality (see Chapter 8, Prespawning Mortality). However, this generally wasn't a major source of population response because the temperatures were altered the most in reaches with relatively small contributions to fish production. The reaches affected by water temperature changes from the alternatives were those below the dams, which generally don't represent the majority of the potential for fish production for the populations. Above dam reaches generally included higher elevation headwater areas that were less likely to have temperature-related PSM; reaches above Green Peter Dam and Fall Creek Dam were notable exceptions. In our models the above-dam reaches were not subject to water temperature changes as a consequence of the alternatives, although it is possible that some operational passage scenarios (drawdowns) may affect the use of the reservoirs as thermal refuges. The influence of secondary passage scenario effects such as temperature or TDG on population viability were minimal compared to the dam passage survival and dam passage efficiency of the different alternatives, with structural options being more successful in improving population viability relative to the NAA.

10 Refer to Table 14.1 for details on the NAA and six passage alternatives.

North Santiam Chinook salmon

Passage alternatives in the North Santiam River provided a range of total VSP scores (Figure 12.1.1). In general, the scores fell into three groups, those with structural passage options (VSP scores over 3), those with operational passage options (2.5 > VSP > 1), and the NAA with a VSP score of around 1. There was considerable variability in the Alternatives 3a and 3b, related partly to the greater influence of water year type on operational passage efficiency and survival. Where NOR abundance levels did not rise significantly, hatchery-origin adults constituted a large proportion of the naturally-spawning adults.

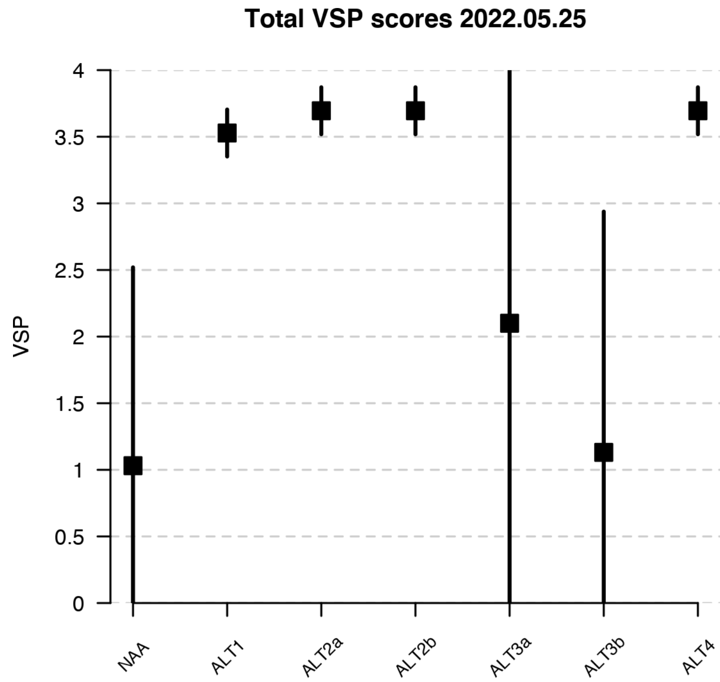


Figure 12.1.1. Total VSP score for Chinook salmon in the North Santiam River under passage alternatives. (Median VSP with 95% confidence interval from 1000 LCM runs per alternative.)

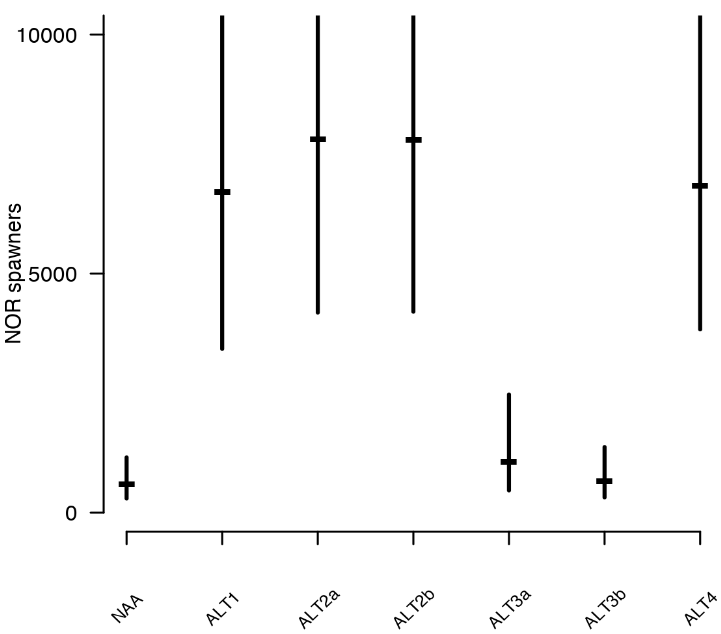


Figure 12.1.2. Estimated Chinook salmon natural origin spawner median abundance with 95% confidence interval in the North Santiam River under the passage alternatives.

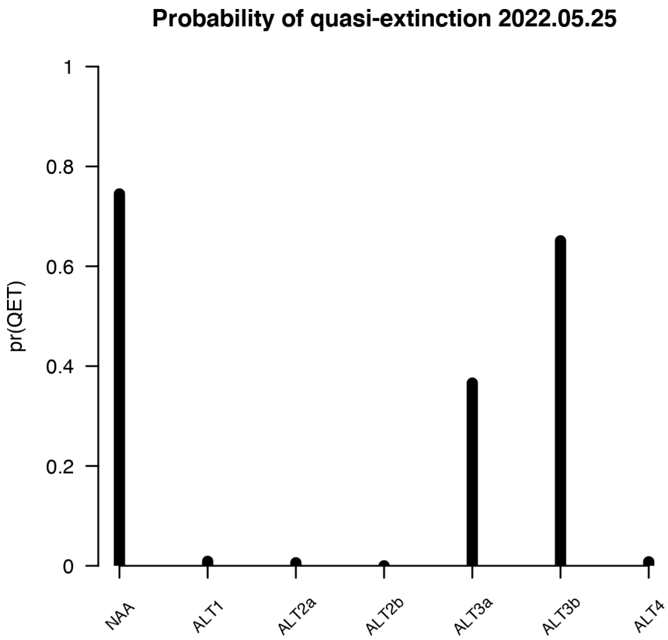


Figure 12.1.3. Probability of falling below the quasi-extinction threshold (150 adults) during the 100 year span of the model for North Santiam Chinook salmon.

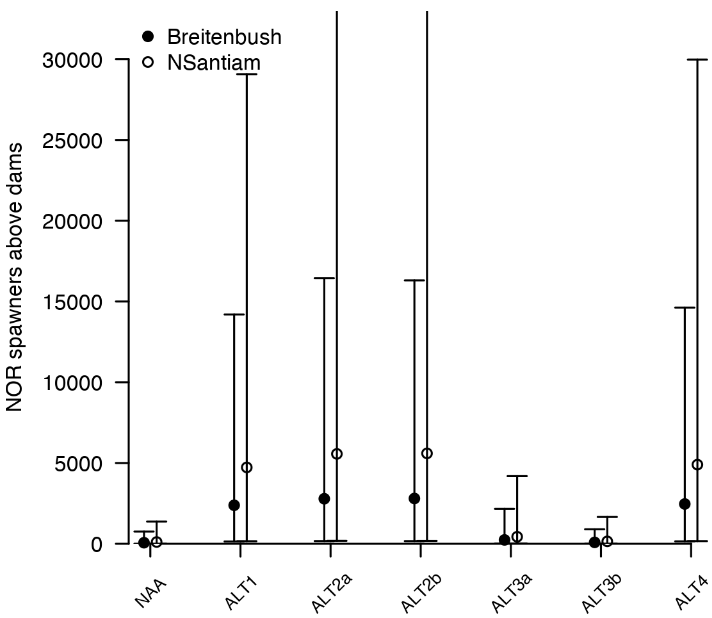


Figure 12.1.4. Median abundance (with 95% confidence interval) of natural-origin Chinook salmon spawners in above dam reaches (Breitenbush and North Santiam rivers) under different passage alternatives.

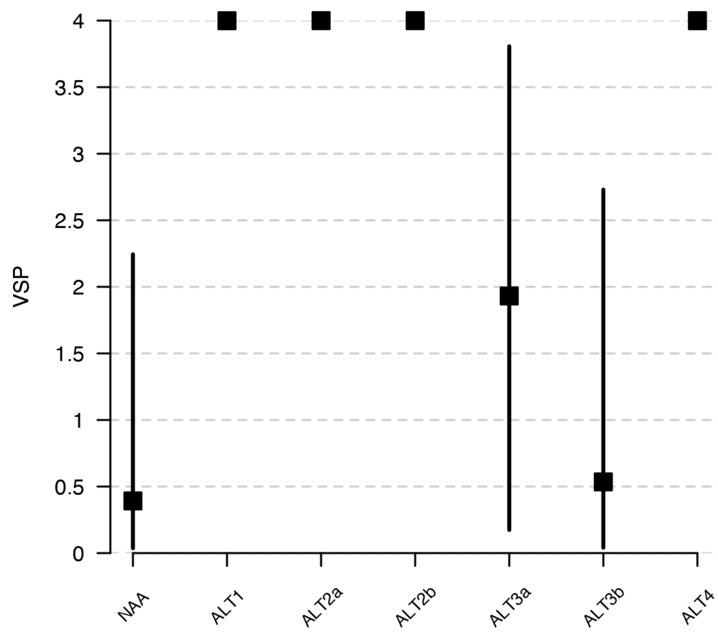


Figure 12.1.5. Median VSP score (with 95% confidence intervals) for abundance and productivity for North Santiam Chinook salmon under different passage alternatives.

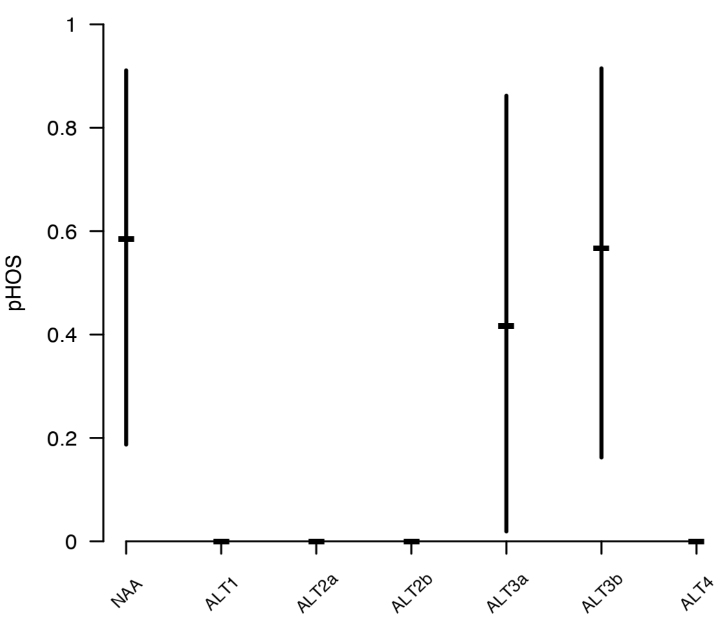


Figure 12.1.6. Median proportion (with 95% confidence interval) of hatchery-origin fish among the total number of Chinook salmon spawners in North Santiam River.

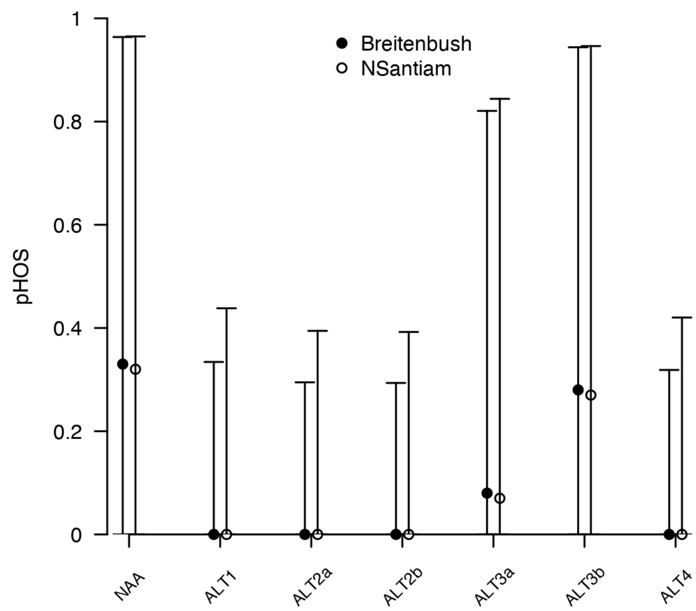


Figure 12.1.7. Median proportion of hatchery-origin Chinook salmon spawners in each of the above dam reaches (Breitenbush and North Santiam) under different passage alternatives.

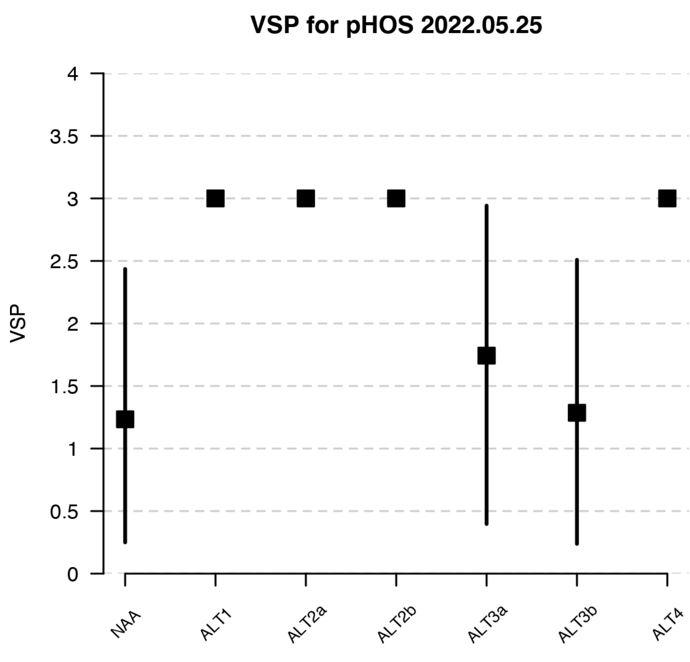


Figure 12.1.8. Median VSP scores for the proportion of hatchery-origin Chinook salmon spawners (pHOS) in the North Santiam Basin under various passage alternatives.

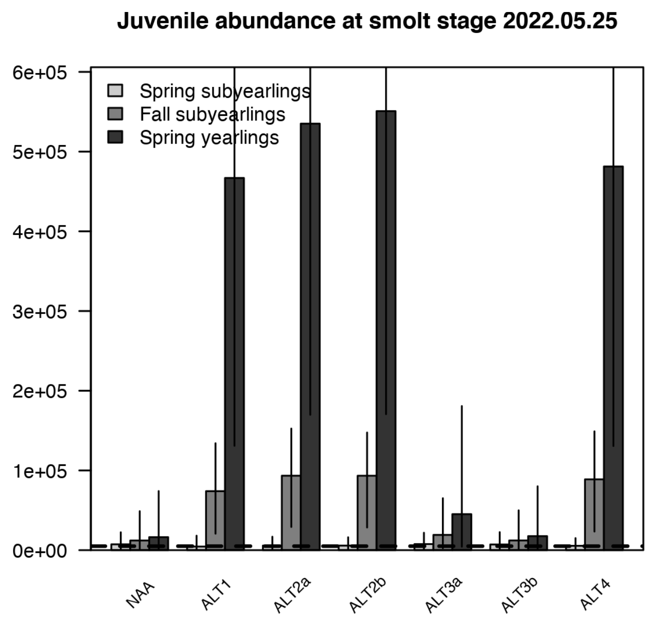


Figure 12.1.9. Median abundance for the three Chinook salmon juvenile life history stages in the North Santiam River under various passage alternatives.

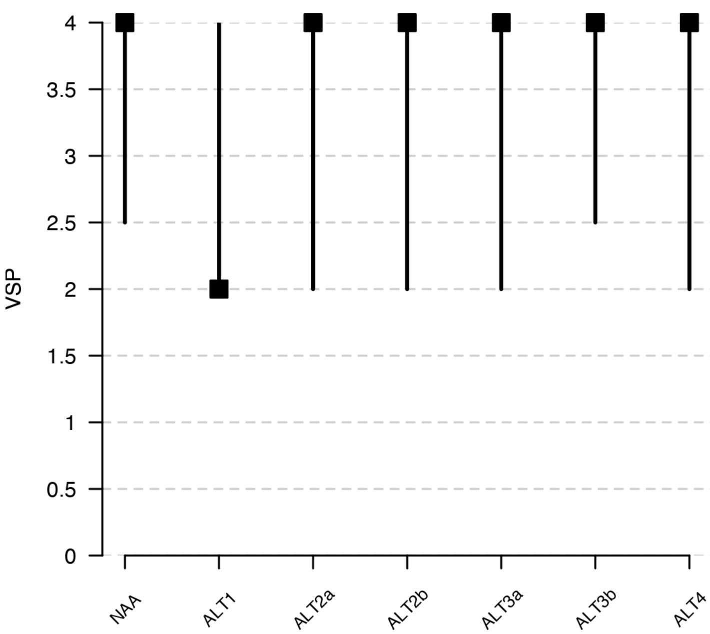


Figure 12.1.10. Median VSP score for North Santiam River Chinook salmon juvenile life history diversity under alternative passage scenarios.

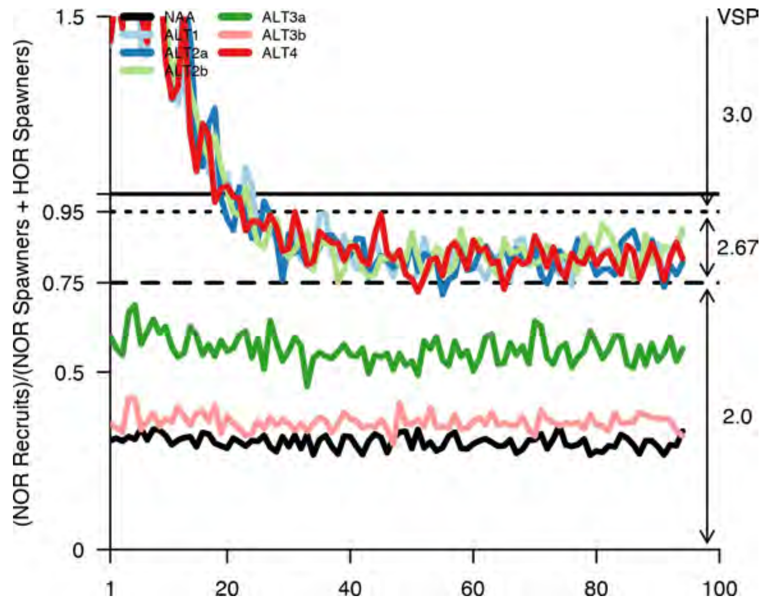


Figure 12.1.11. Chinook salmon median recruits per spawner (left axis) during 100 year model runs for North Santiam reaches above Detroit Dam under various alternative juvenile passage programs, with the corresponding VSP score (right axis).

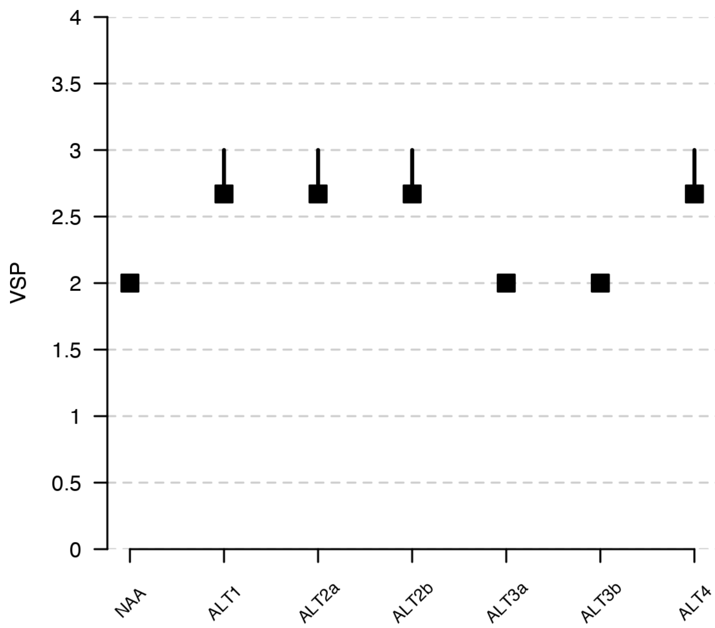


Figure 12.1.12. Median VSP score for spatial structure for Chinook salmon in the North Santiam River under various juvenile passage programs.

North Santiam Winter steelhead

All of the passage alternatives for North Santiam River winter steelhead had similar total VSP scores, although alternatives with structural passage were predicted to have higher absolute numbers of natural origin spawners. VSP scoring for abundance and productivity is based on the probability of falling below the QET.

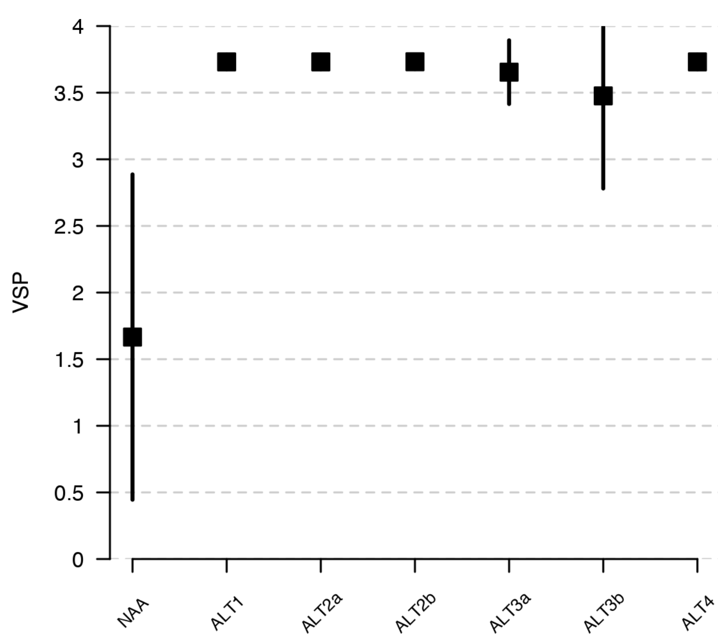


Figure 12.2.1. Total VSP score for winter steelhead in the North Santiam River under passage alternatives. (Median VSP with 95% confidence interval from 1000 LCM runs per alternative).

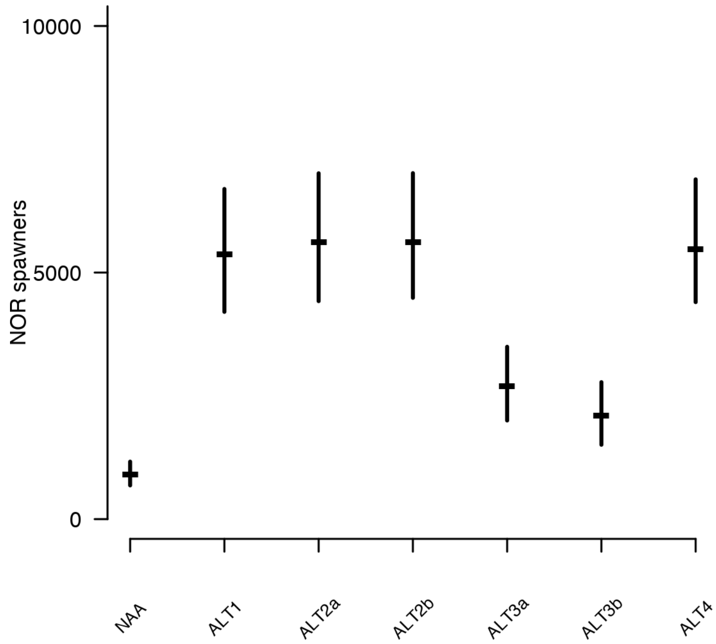


Figure 12.2.2. Estimated North Santiam River winter steelhead natural origin spawner median abundance with 95% confidence interval under passage alternatives.

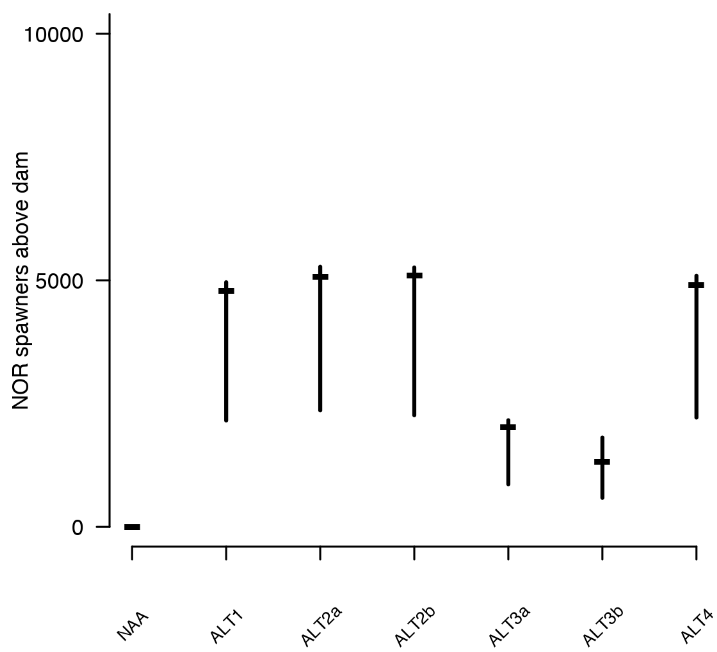


Figure 12.2.3. Median abundance of natural-origin winter steelhead spawners in above dam reaches (Breitenbush and North Santiam) under different passage alternatives.

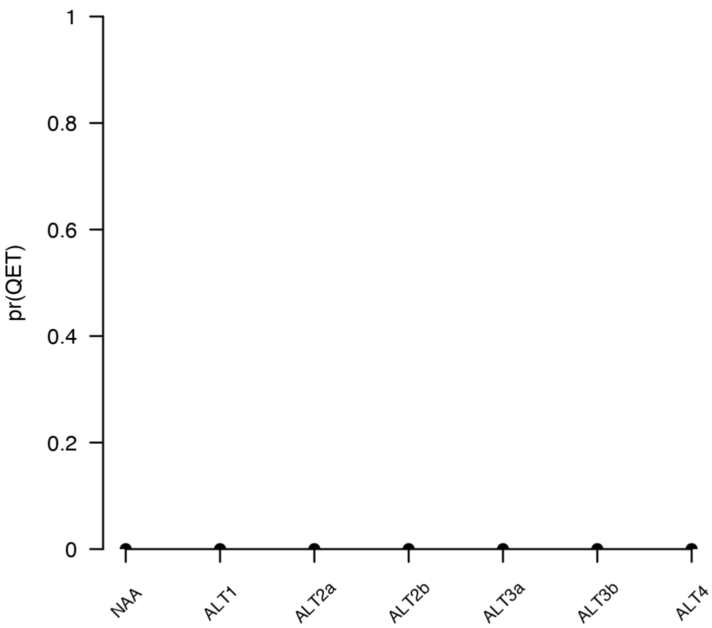


Figure 12.2.4. Probability of falling below the quasi-extinction threshold (200 adults) during the 100 year span of the model for North Santiam winter steelhead.

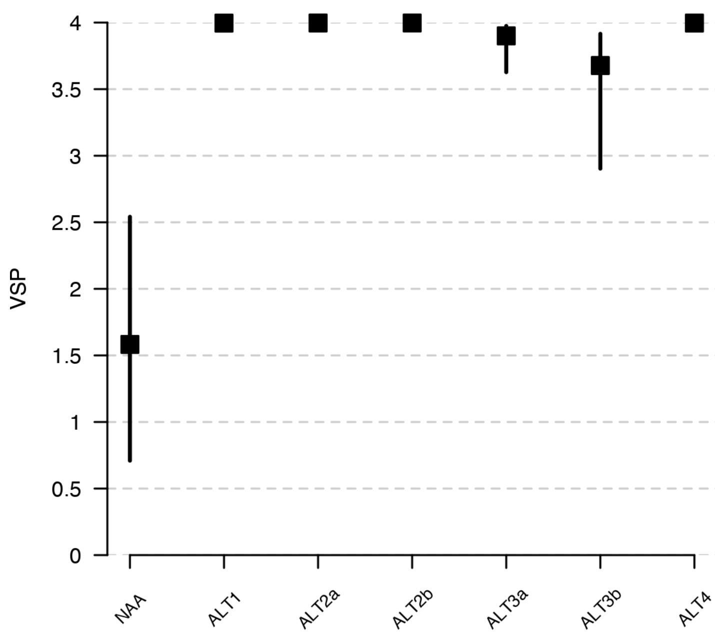


Figure 12.2.5. North Santiam winter steelhead median VSP score for abundance and productivity under different passage alternatives.

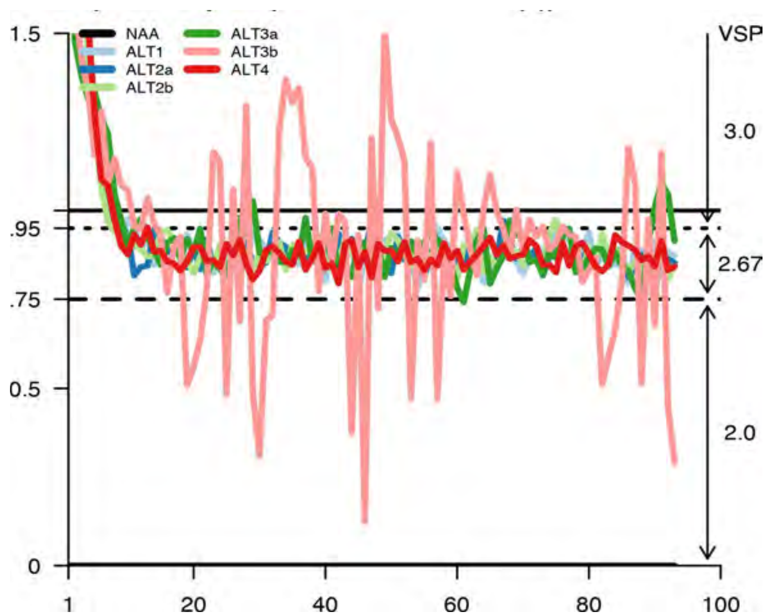


Figure 12.2.6. Winter steelhead median recruits per spawner (left axis) during 100 year model runs for North Santiam reaches above Detroit Dam under various alternative juvenile passage programs, with the corresponding VSP score (right axis).

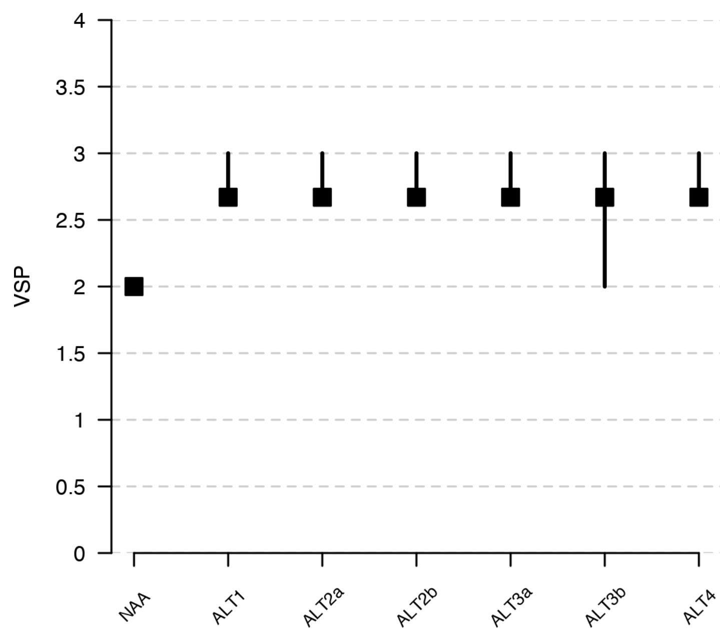


Figure 12.2.7. Median VSP score for spatial structure for winter steelhead in the North Santiam River under various juvenile passage programs.

South Santiam Chinook salmon!!!!

Among the juvenile passage alternatives modeled for south Santiam Chinook salmon, Alternative 1 (structural passage at Green Peter Dam with modified fish weir at Foster Dam) was distinctly better than the other alternatives with operational passage at Green Peter Dam.

Structural passage at Green Peter provided both relatively high efficiency passage, and by collecting juveniles they were able to bypass additional mortalities in Foster Reservoir and at Foster Dam. With the exception of Alternative 1, hatchery-origin adults continued to constitute the majority of spawners.

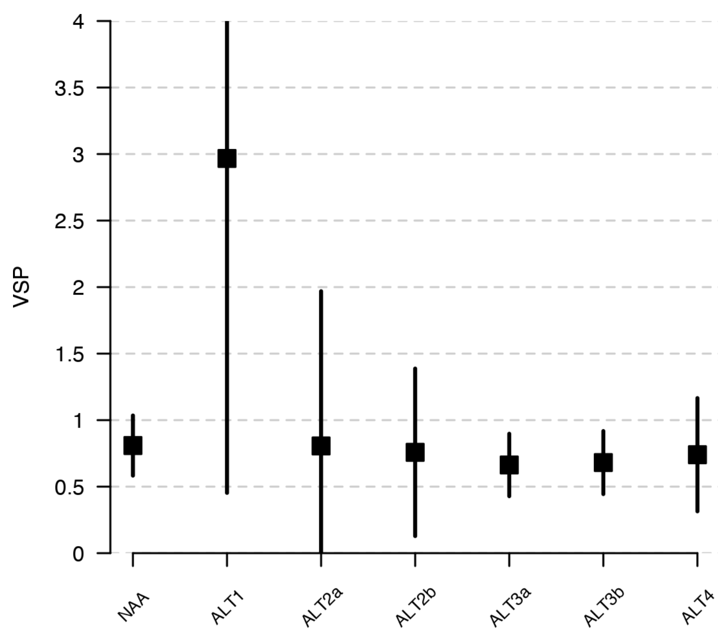


Figure 12.3.1. Total VSP score for Chinook salmon in the South Santiam River under passage alternatives. (Median VSP with 95% confidence interval from 1000 LCM runs per alternative).

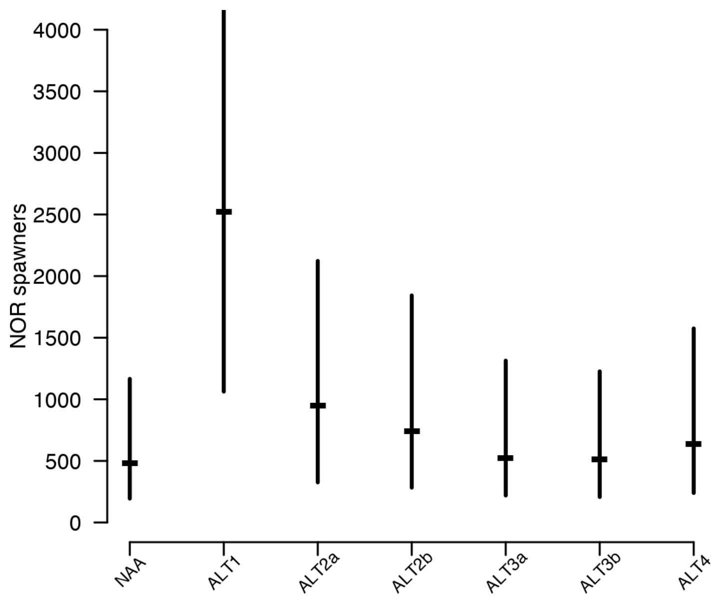


Figure 12.3.2. Estimated Chinook salmon natural origin median spawner abundance in the South Santiam River under passage alternatives.

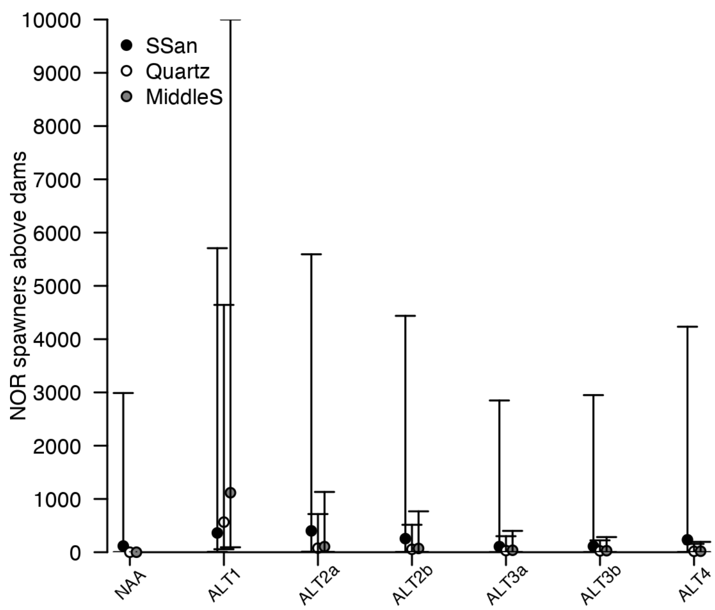


Figure 12.3.3. Median abundance with 95% confidence interval of natural-origin Chinook salmon spawners in above dam reaches (South Santiam, Quartzville Creek, and Middle Santiam) under different passage alternatives.

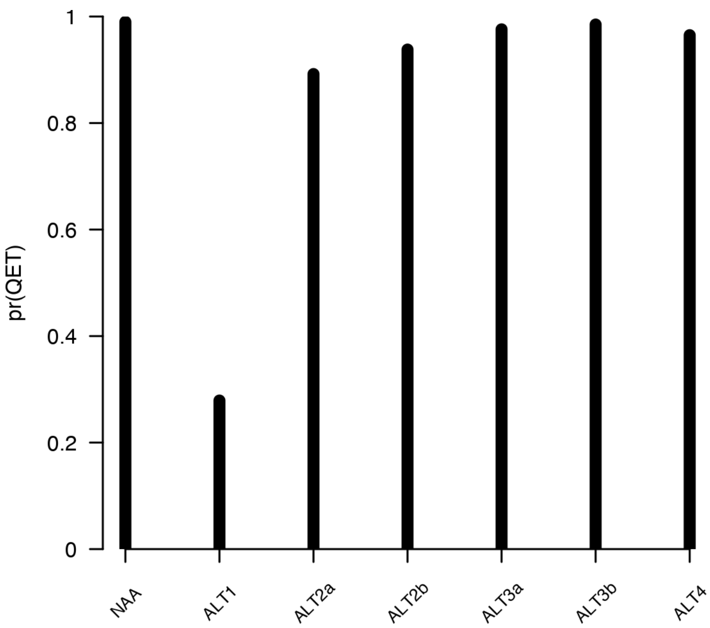


Figure 12.3.4. Probability of falling below the quasi-extinction threshold (250 adults) during the 100 year span of the model for South Santiam Chinook salmon based on 1000 runs per alternative.

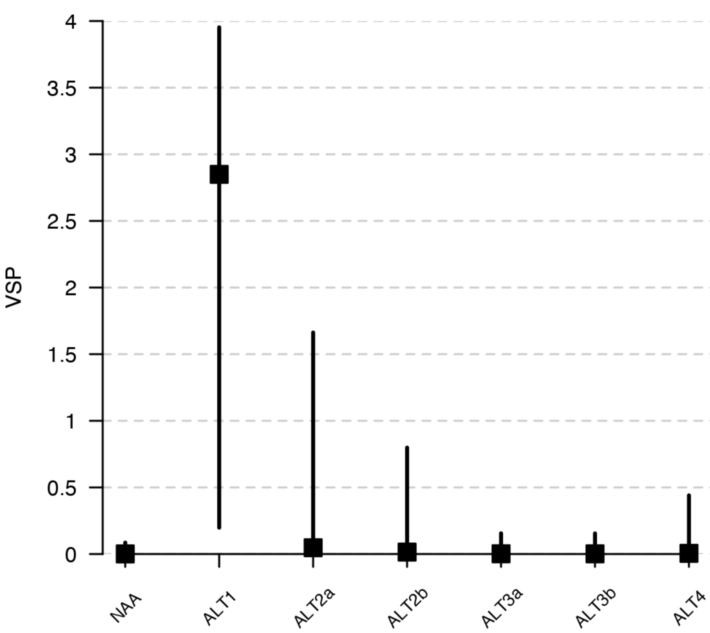


Figure 12.3.5. South Santiam Chinook salmon median VSP score (with 95% confidence interval) for abundance and productivity under different passage alternatives.

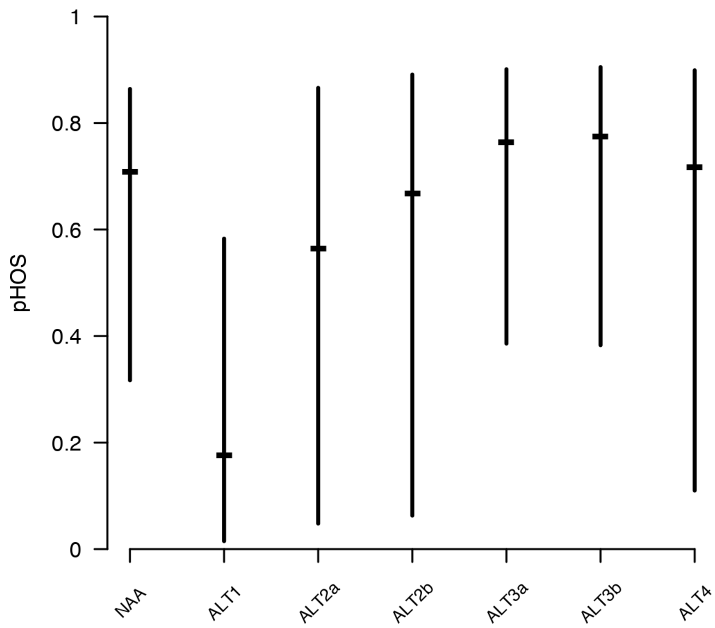


Figure 12.3.6. Median proportion (with 95% confidence interval) of hatchery-origin fish among the total number of Chinook salmon spawners in the South Santiam River.

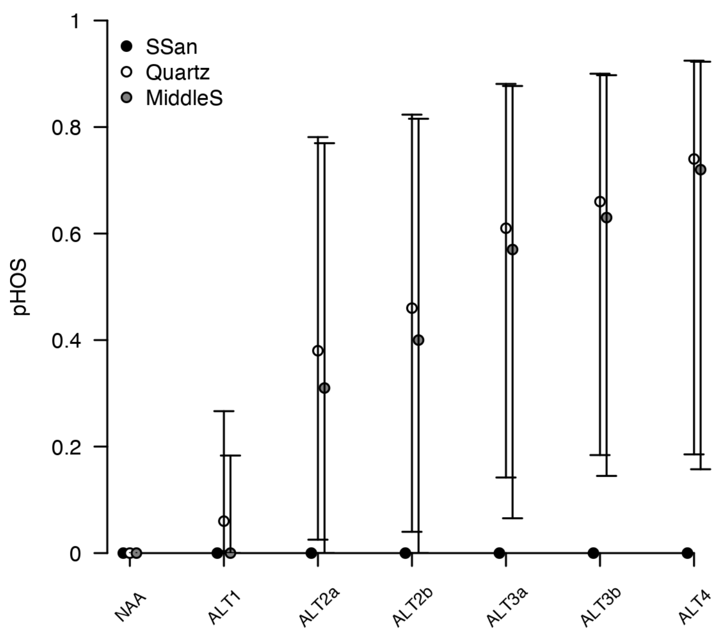


Figure 12.3.7. Median proportion of hatchery-origin Chinook salmon spawners in each of the above dam reaches (South Santiam River, Quartzville Creek, and Middle Santiam River) under different passage alternatives.

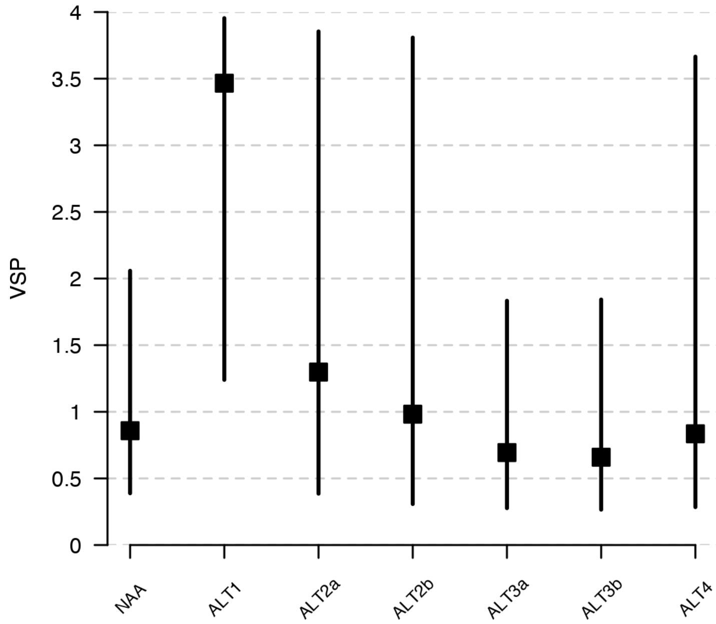


Figure 12.3.8. Median VSP score for the proportion of hatchery-origin Chinook salmon spawners (pHOS) in the South Santiam basin under various passage alternatives.

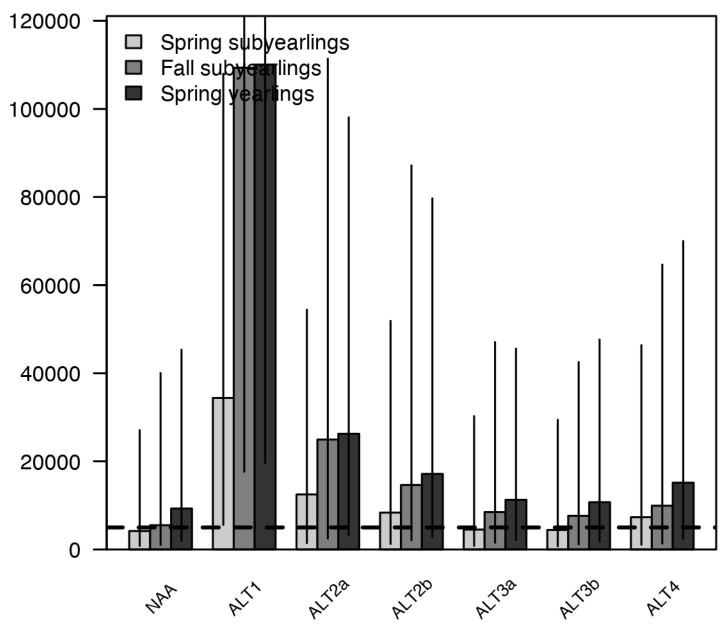


Figure 12.3.9. Median abundances for the three Chinook salmon juvenile life history stages in the South Santiam River under various passage alternatives.

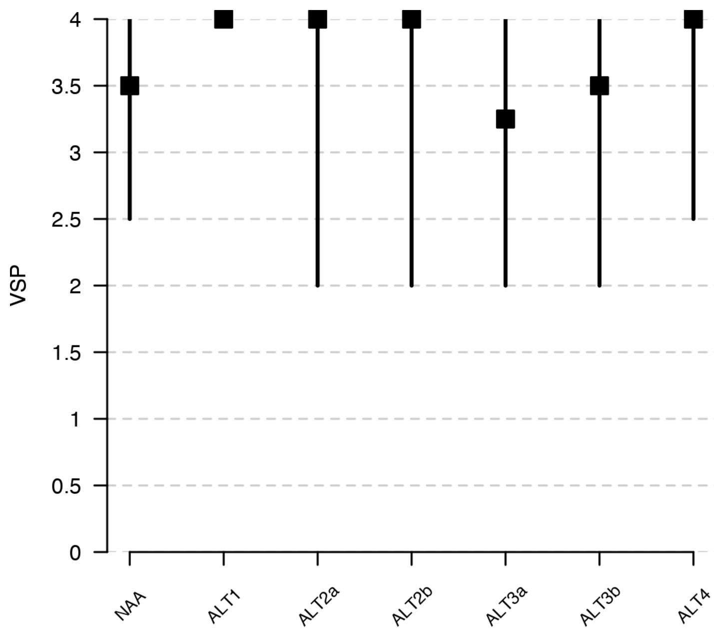


Figure 12.3.10. Median VSP score for South Santiam River Chinook salmon juvenile life history diversity under alternative passage scenarios.

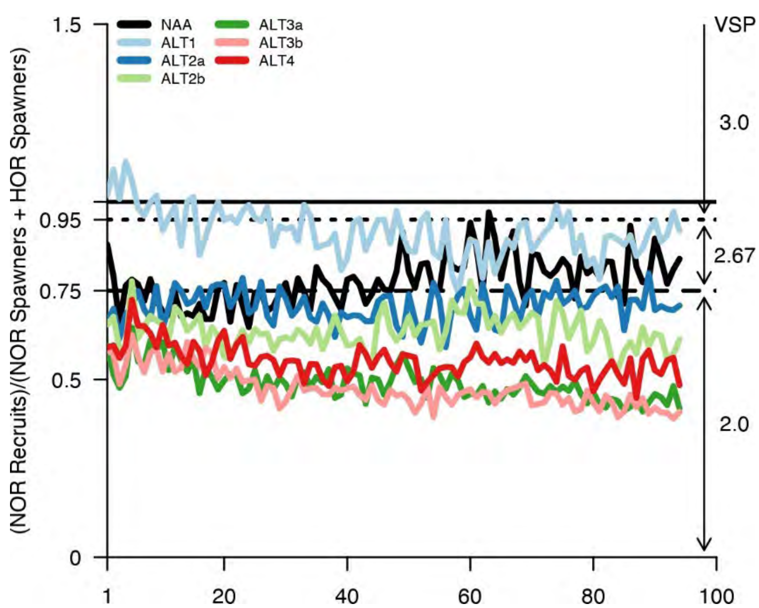


Figure 12.3.11. Chinook salmon median recruits per spawner (left axis) during 100 year model runs for South Santiam reaches above Foster Dam under various alternative juvenile passage programs, with the corresponding VSP score (right axis).

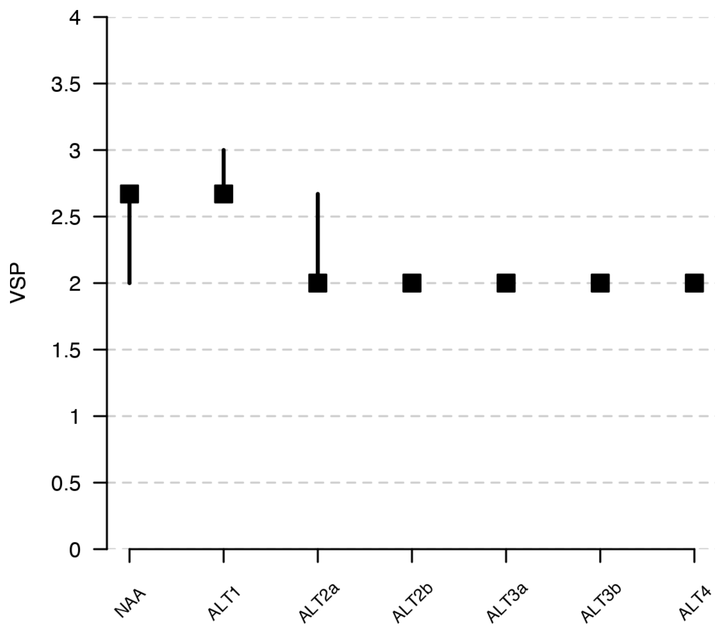


Figure 12.3.12. Median VSP score for spatial structure for Chinook salmon in the South Santiam River under various juvenile passage programs.

South Santiam Winter steelhead

For winter steelhead in the South Santiam River, Alternative 1 was estimated to maintain this population at the highest VSP level. Alternative 1 provided structural passage at Green Peter Dam with estimated high passage efficiency and survival. Additionally, the FSS allowed for the collection of juveniles and their transport below Foster Dam (eliminating additional passage mortalities at Foster Reservoir and Dam). Alternatives 2a, 2b, and 4 appeared to improve the population viability over NAA, while Alternatives 3a and 3b were indistinguishable from the NAA model scenario.

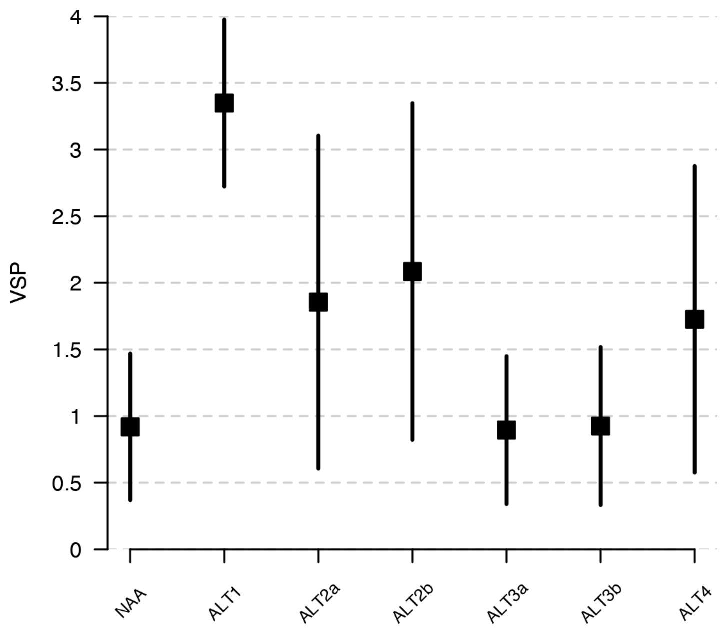


Figure 12.4.1. Median total VSP score for winter steelhead in the South Santiam River under passage alternatives. (Median VSP with 95% confidence interval from 1000 LCM runs per alternative).

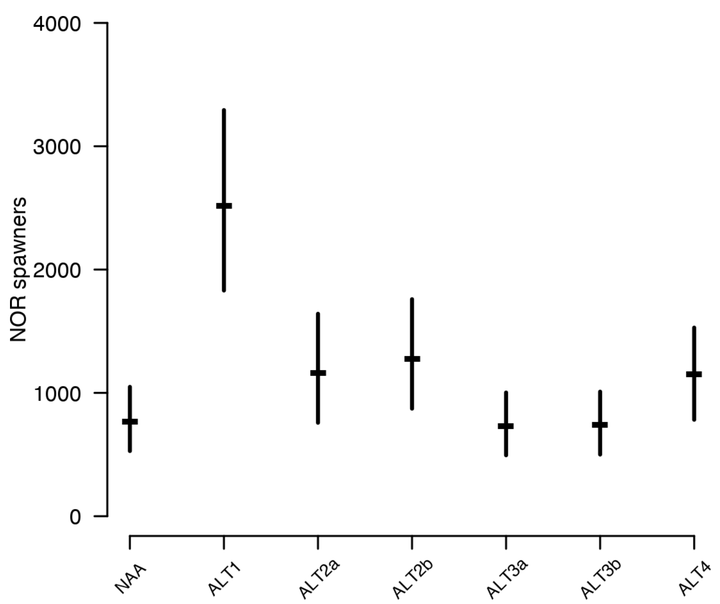


Figure 12.4.2. Estimated South Santiam River winter steelhead natural origin spawner median abundance with 95% confidence interval under passage alternatives.

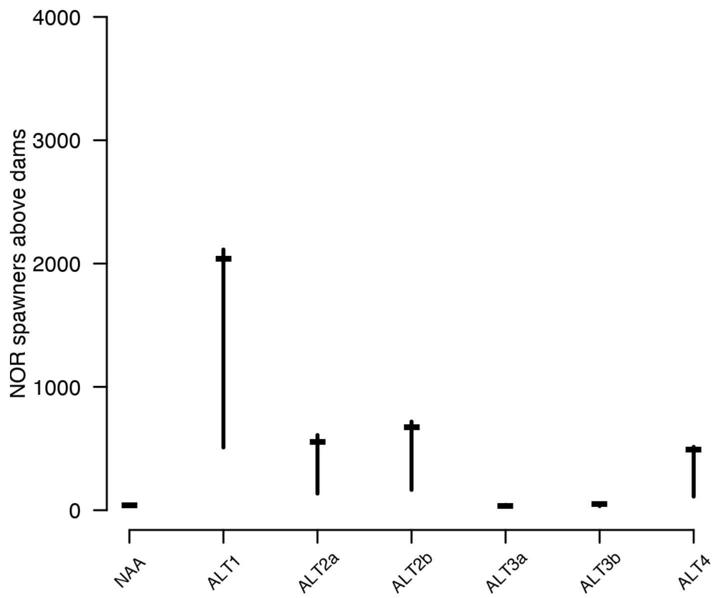


Figure 12.4.3. Median abundance of natural-origin winter steelhead spawners in above dam reaches (South Santiam River, Quartzville Creek, and Middle Santiam River) under different passage alternatives.

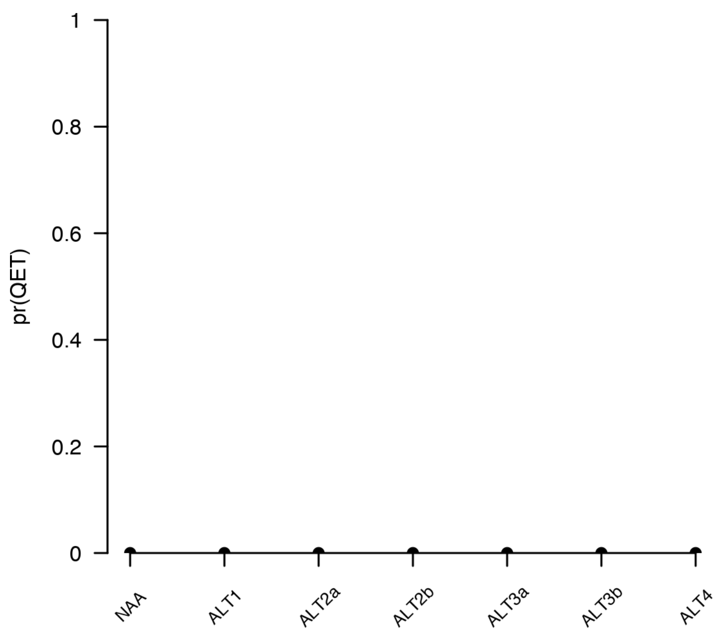


Figure 12.4.4. Probability of falling below the quasi-extinction threshold (200 adults) during the 100 year span of the model for South Santiam winter steelhead based on 1000 runs per alternative.

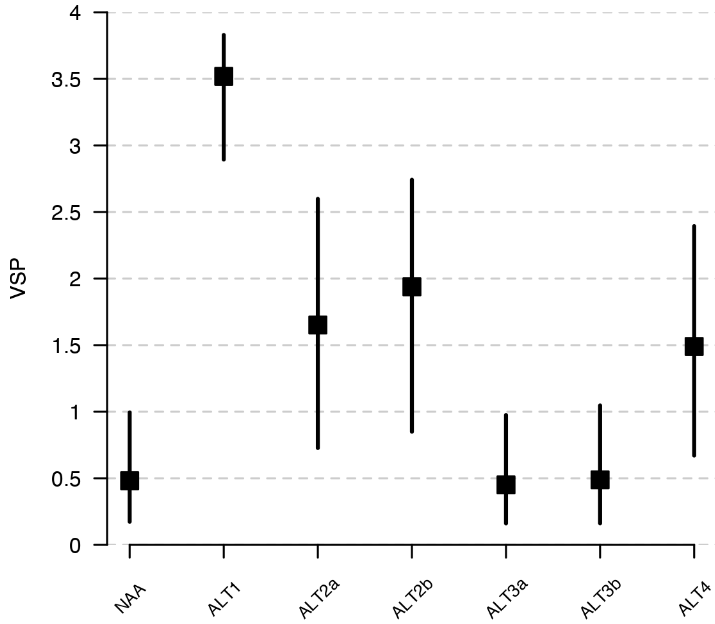


Figure 12.2.5. South Santiam winter steelhead median VSP score for abundance and productivity under different passage alternatives.

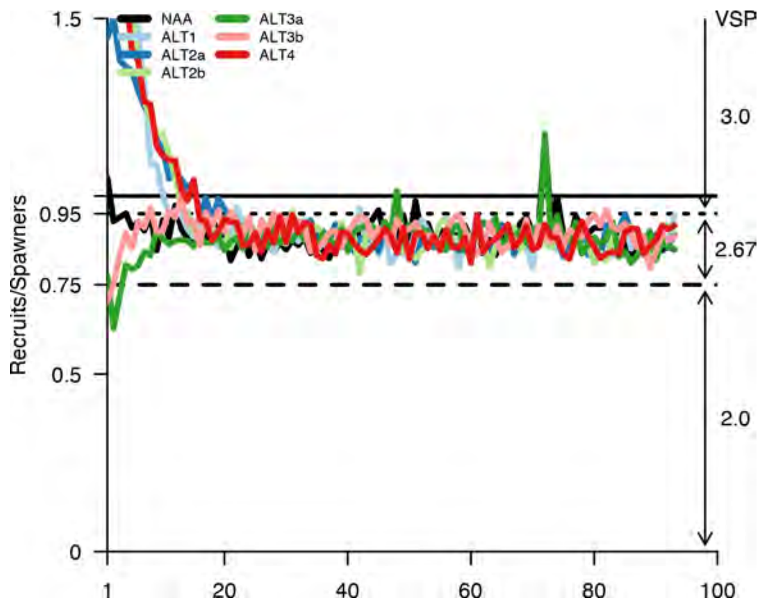


Figure 12.4.6. Winter steelhead median recruits per spawner (left axis) during 100 year model runs for South Santiam reaches above Foster Dam under various alternative juvenile passage programs, with the corresponding VSP score (right axis).

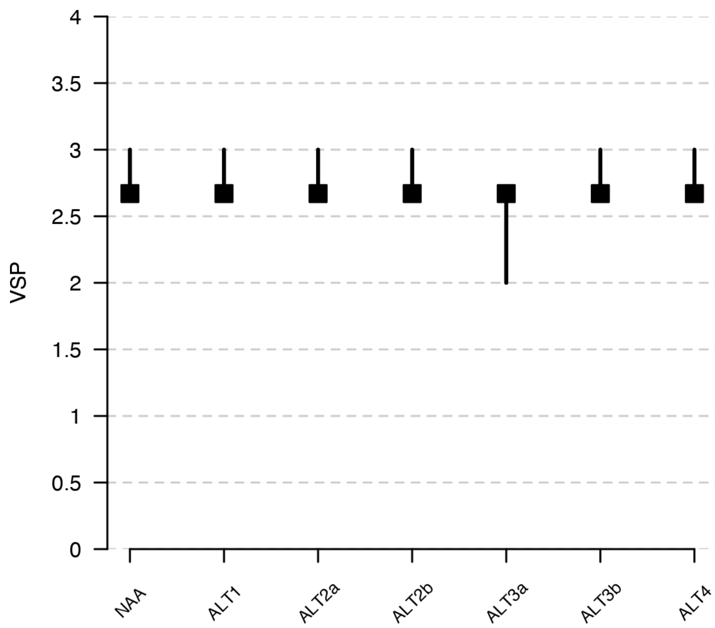


Figure 12.2.7. Median VSP score for spatial structure for winter steelhead in the South Santiam River under various juvenile passage programs.

McKenzie Chinook salmon

Model results for Chinook salmon in the McKenzie River suggested that Alternatives 2a and 4 resulted in the highest population viability. Both alternatives utilized a FSS to pass juveniles downstream at a high collection efficiency and with minimal mortality.

Alternatives 2b and 3b utilized operational passage (spring and fall draw down to the diversion tunnel) and were predicted to provide a noticeable improvement in viability of the NAA. Lastly, alternatives 1 and 3a were indistinguishable from NAA. The hatchery contribution to naturally-spawning adults was generally low for most alternatives.

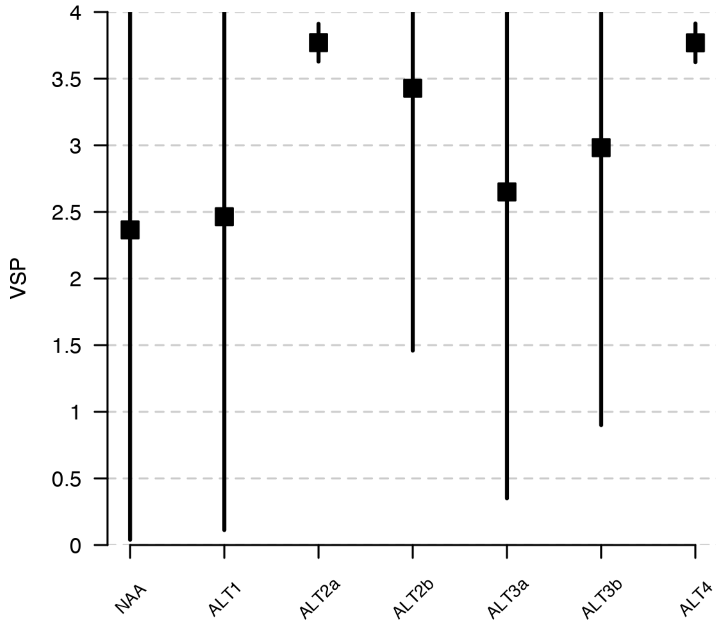


Figure 12.5.1. Total VSP score for Chinook salmon in the McKenzie River under passage alternatives. (Median VSP with 95% confidence interval from 1000 LCM runs per alternative).

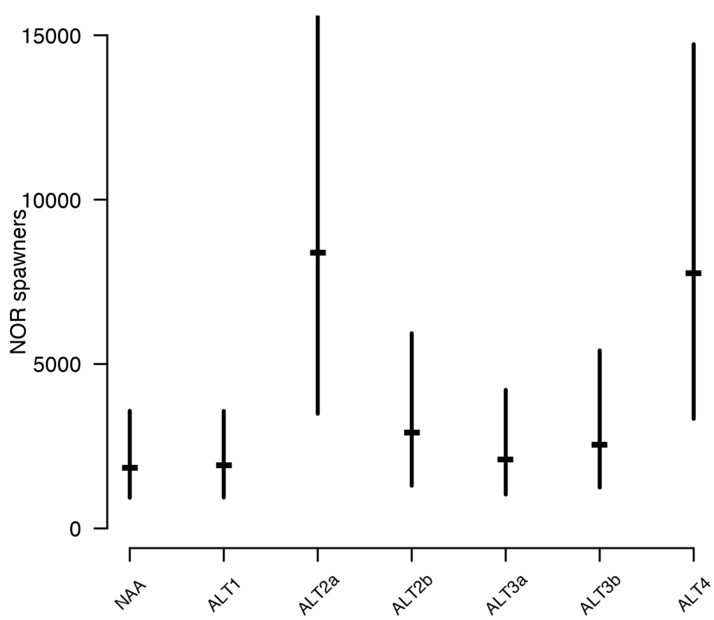


Figure 12.5.2. Estimated Chinook salmon natural origin spawner median abundance with 955 confidence interval in the McKenzie River under passage alternatives.

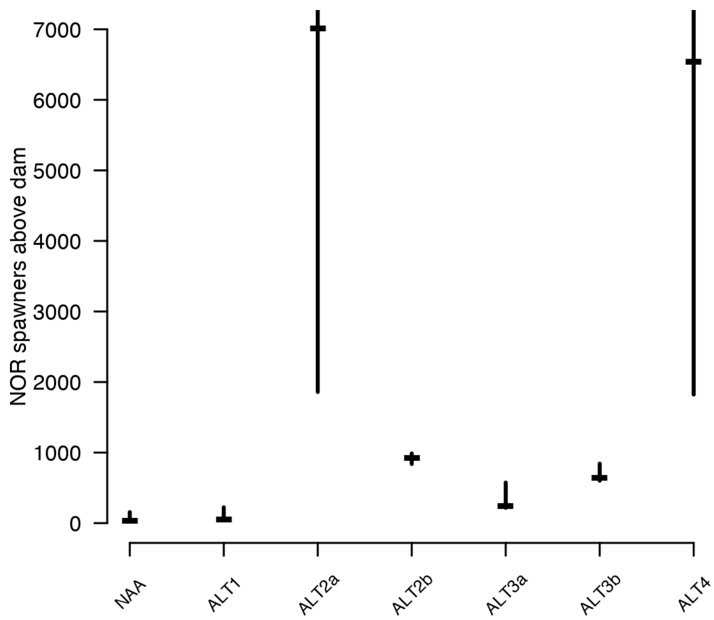


Figure 12.5.3. Median abundance of natural-origin Chinook salmon spawners above Cougar Dam (South Fork McKenzie River) under different passage alternatives.

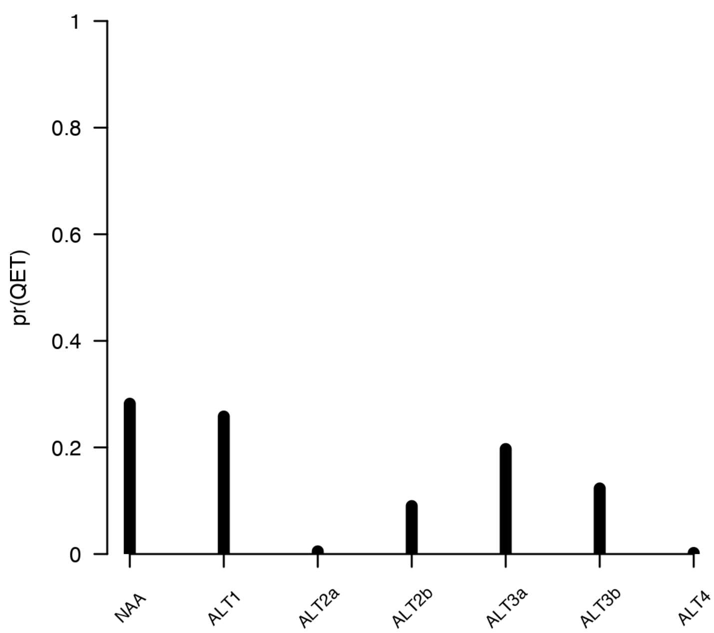


Figure 12.5.4. Probability of falling below the quasi-extinction threshold (250 adults) during the 100 year span of the model for McKenzie River Chinook salmon based on 1000 runs per alternative.

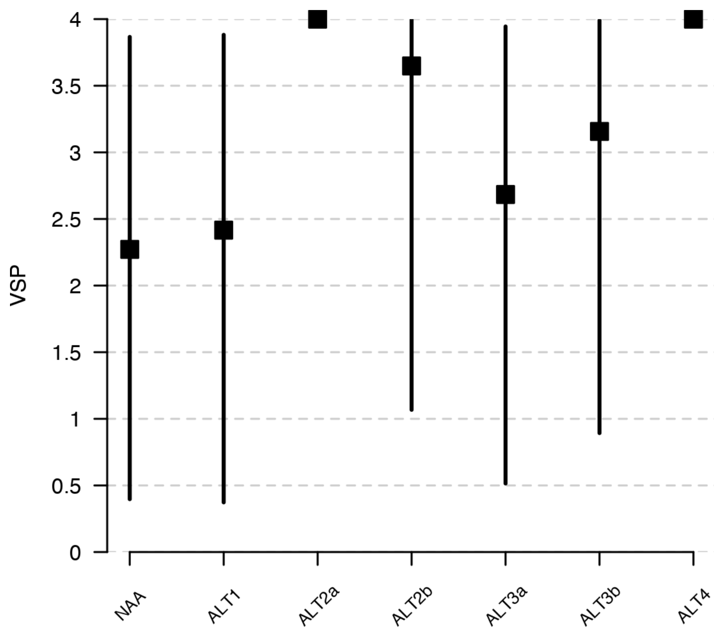


Figure 12.5.5. McKenzie River Chinook salmon median VSP score, with 95% confidence interval, for abundance and productivity under different passage alternatives.

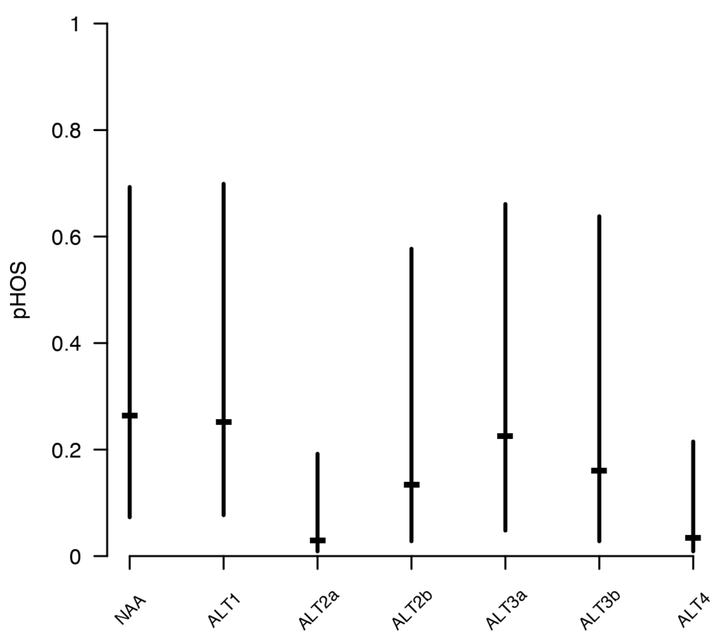


Figure 12.5.6. Median proportion of hatchery-origin fish among the total number of Chinook salmon spawners in McKenzie River.

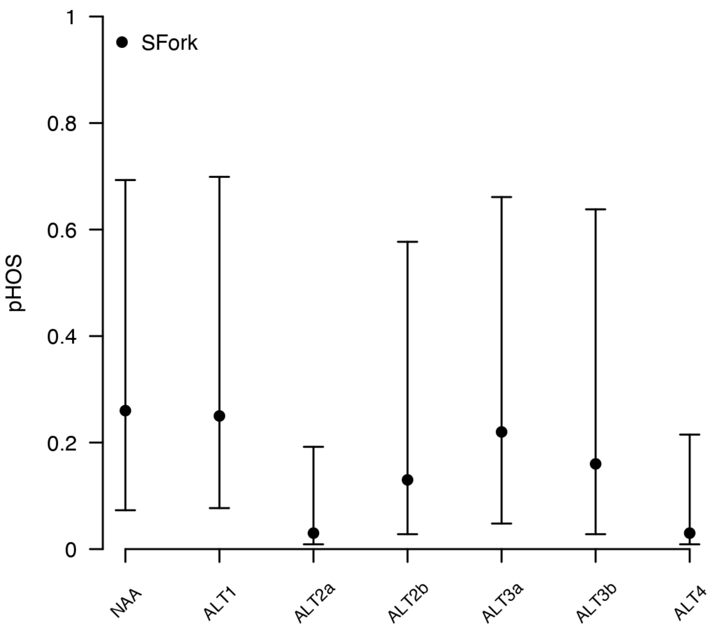


Figure 12.5.7. Median proportion of hatchery-origin Chinook salmon spawners above Cougar Dam (South Fork McKenzie River) under different passage alternatives.

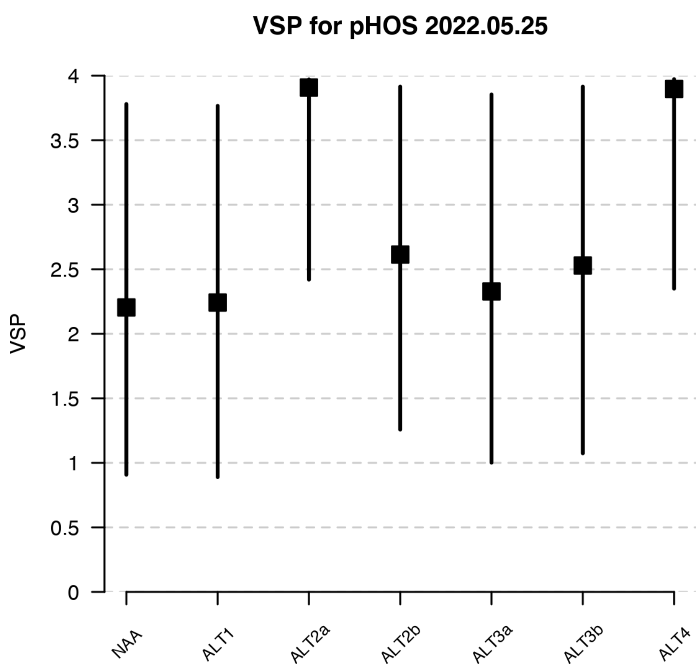


Figure 12.5.8. Median VSP score for the proportion of hatchery-origin Chinook salmon spawners (pHOS) in the McKenzie Basin under the passage alternatives.

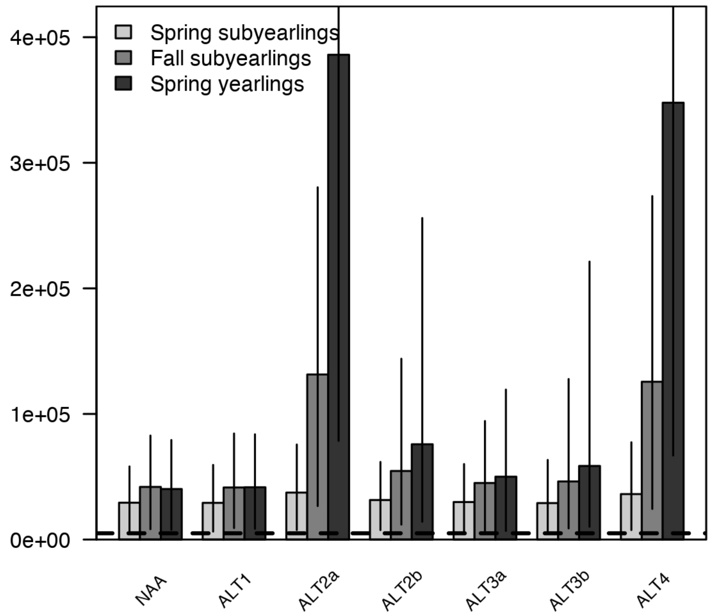


Figure 12.5.9. Median abundances for the three Chinook salmon juvenile life history stages in the McKenzie River under various passage alternatives.

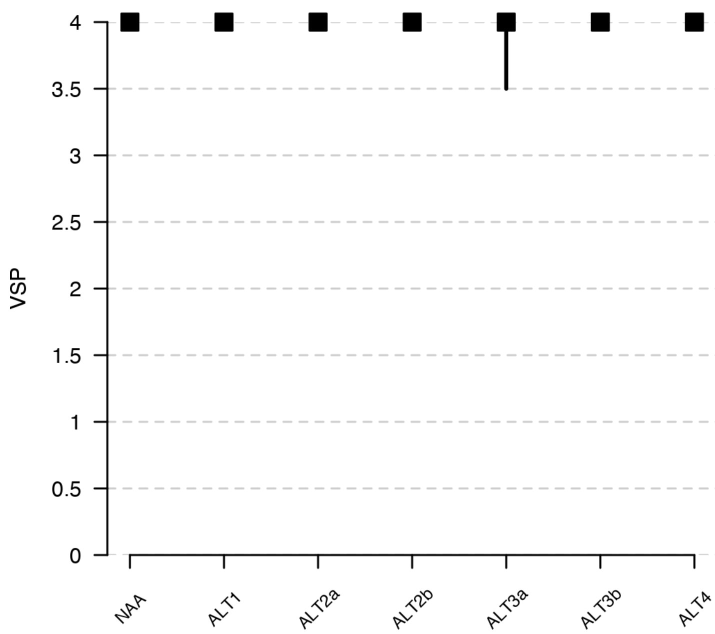


Figure 12.5.10. Median VSP score for McKenzie River Chinook salmon juvenile life history diversity under alternative passage scenarios.

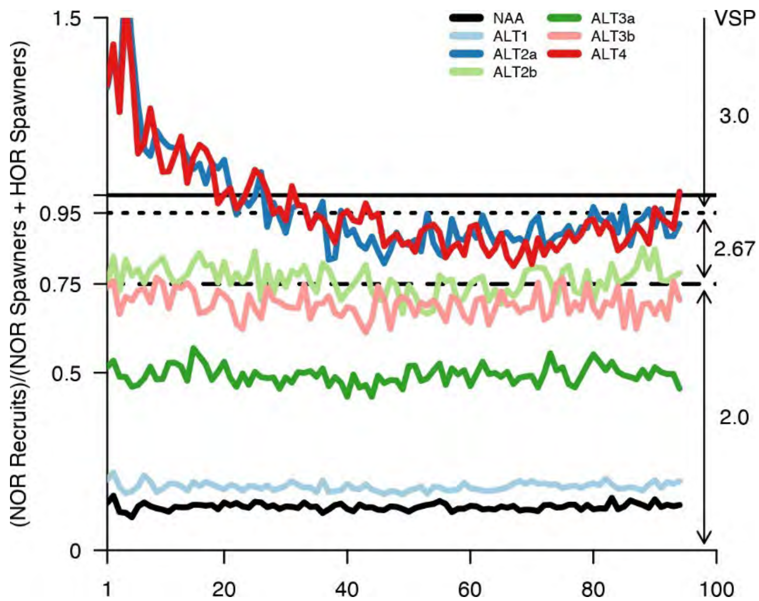


Figure 12.5.11. Chinook salmon median recruits per spawner (left axis) during 100 year model runs for McKenzie reaches above Cougar Dam under various alternative juvenile passage programs, with the corresponding VSP score (right axis).

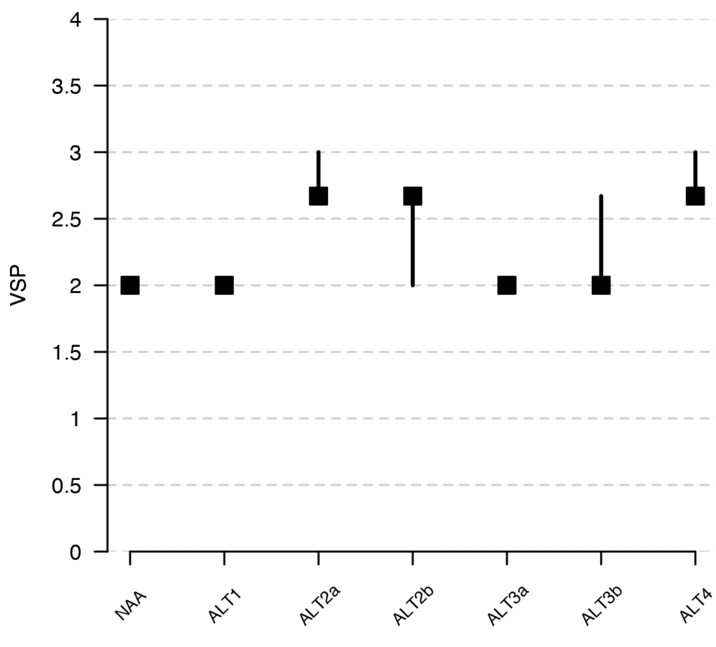


Figure 12.5.12. Median VSP score for spatial structure for Chinook salmon in the McKenzie River under the juvenile passage alternatives.

Middle Fork Chinook Salmon

Model estimates of the passage alternatives indicated that Alternative 4 provided the best improvement in Chinook salmon viability, relative to the NAA. Alternative 4 provided structural passage and juvenile collection at both Lookout Point Dam and Hills Creek Dam. Alternatives 1, 2a, and 2b all provided for a floating surface collector at Lookout Point Dam, with no additional passage operation or structure at Hills Creek. These three alternatives were estimated to improve Chinook salmon viability from the NAA, although not as much as Alternative 4. Finally, Alternatives 3a and 3b, which provided operational passage at Dexter, Lookout Point, and Hills Creek dams did not markedly improve population viability relative to the NAA.

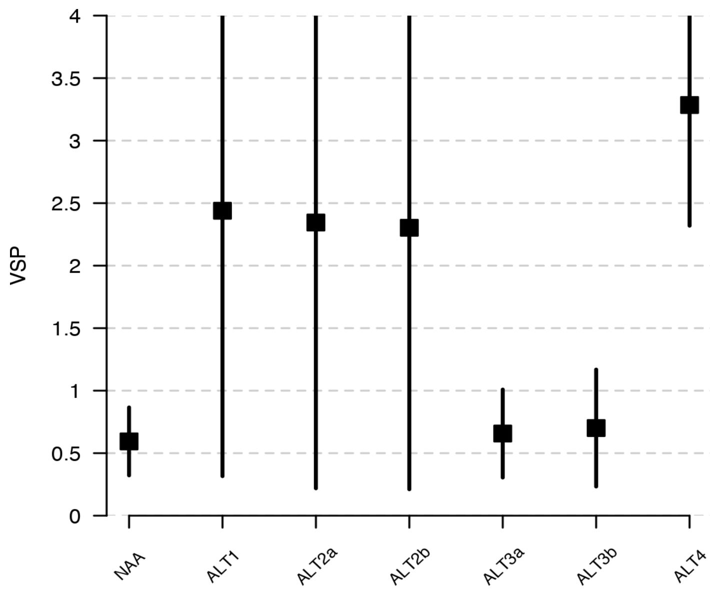


Figure 12.6.1. Total VSP score for Chinook salmon in the Middle Fork Willamette River under passage alternatives. (Median VSP with 95% confidence interval from 1000 LCM runs per alternative).

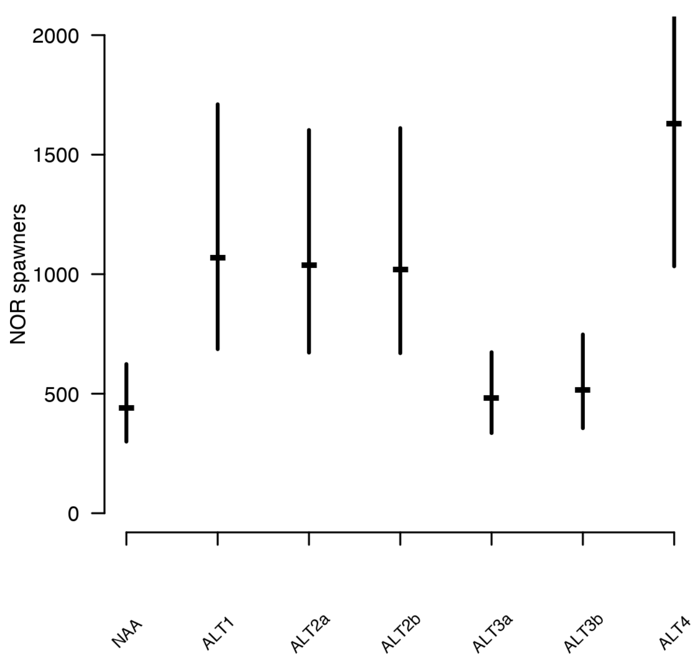


Figure 12.6.2. Estimated Chinook salmon natural origin spawner median abundance, with 95% confidence interval, in the Middle Fork Willamette River under passage alternatives.

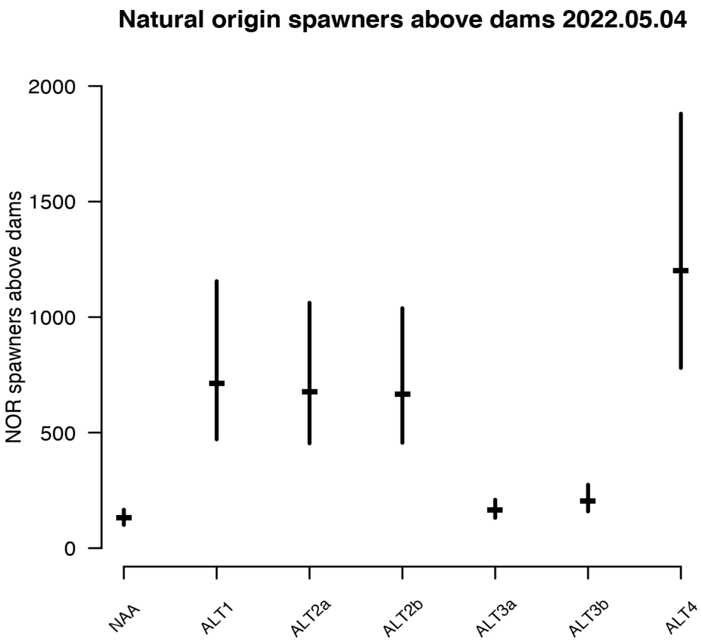


Figure 12.6.3. Median abundance of natural-origin Chinook salmon spawners above Dexter Dam (Middle Fork Willamette River and Hills Creek) under different passage alternatives.

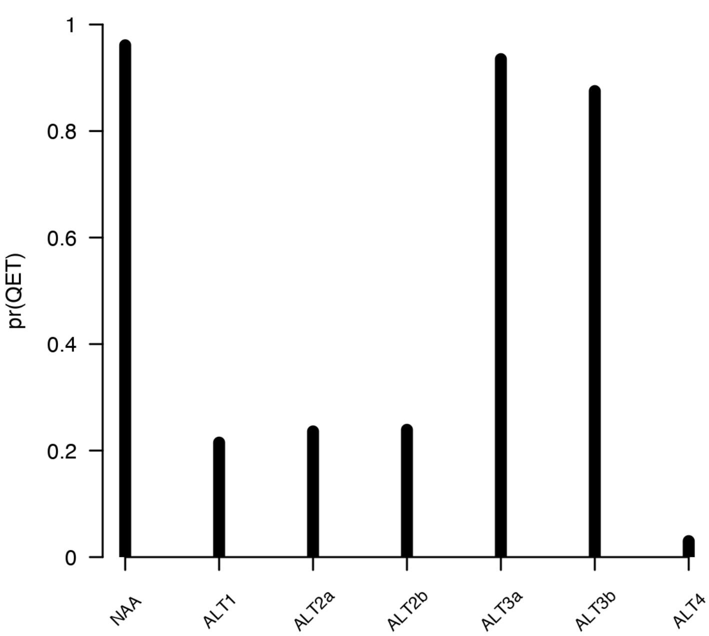


Figure 12.6.4. Probability of falling below the quasi-extinction threshold (250 adults) during the 100 year span of the model for Middle Fork Willamette River Chinook salmon based on 1000 runs per alternative.

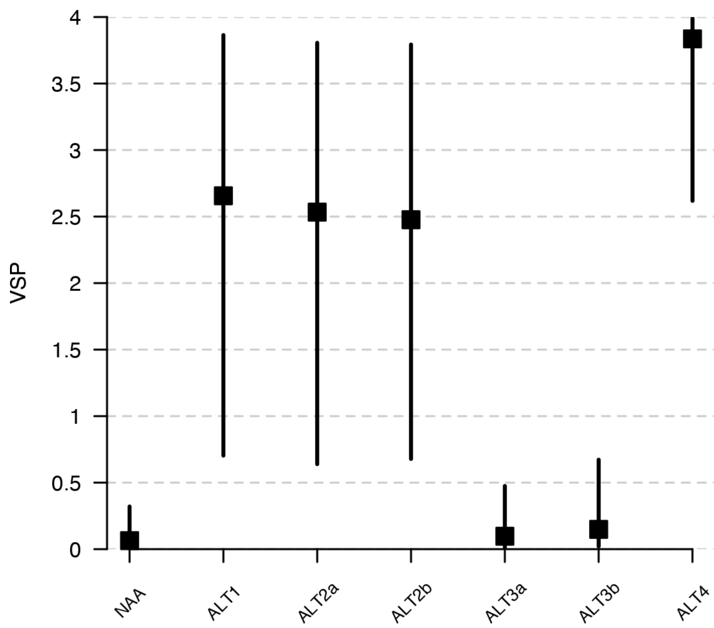


Figure 12.6.5. Middle Fork Willamette River Chinook salmon median VSP score for abundance and productivity under different passage alternatives.

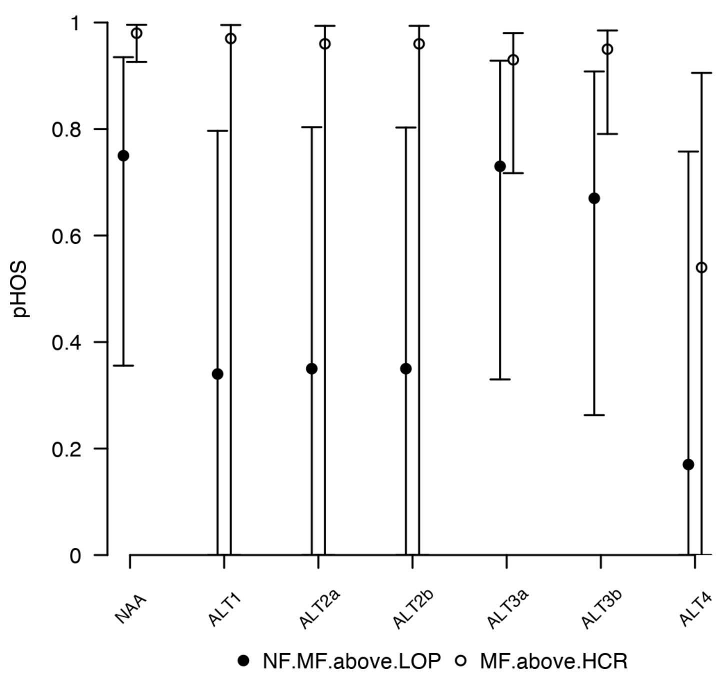


Figure 12.6.6. Median proportion of hatchery-origin Chinook salmon spawners in reaches above Dexter Dam (North Fork Middle Fork Willamette River and Hills Creek) under the passage alternatives.

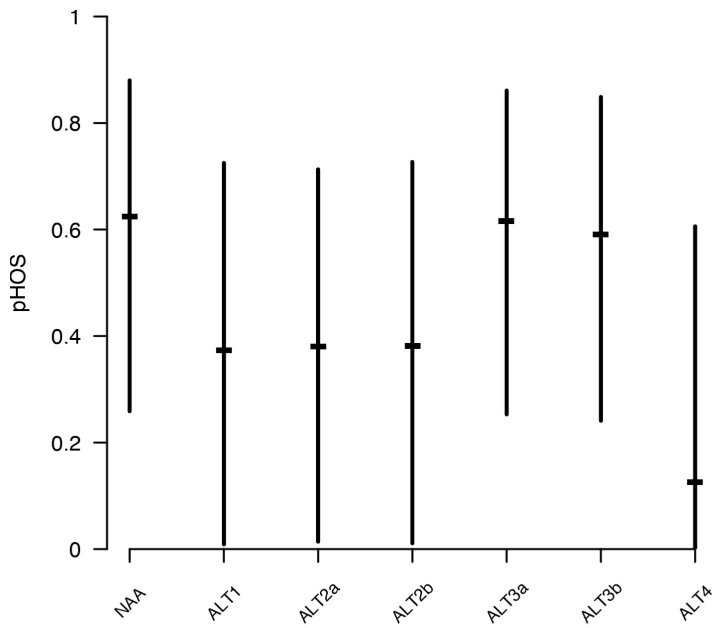


Figure 12.6.7. Median proportion (with 95% confidence interval) of hatchery-origin fish among the total number of Chinook salmon spawners in Middle Fork Willamette River.

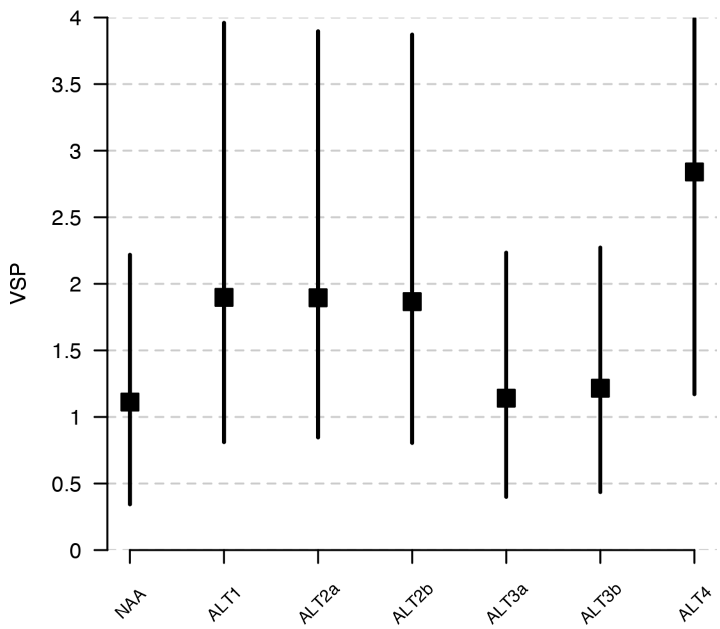


Figure 12.6.8. Median VSP score for the proportion of hatchery-origin Chinook salmon spawners (pHOS) in the Middle Fork Willamette River Basin under various passage alternatives.

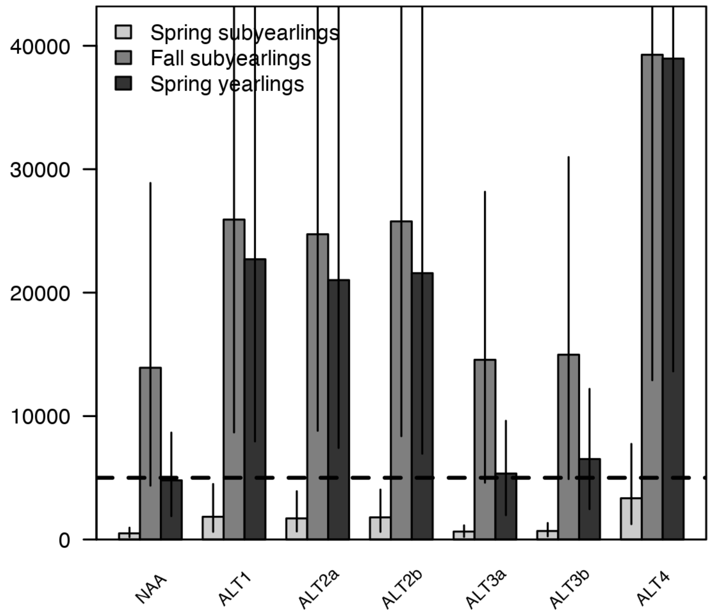


Figure 12.6.9. Median abundances for the three Chinook salmon juvenile life history stages in the Middle Fork Willamette River under various passage alternatives.

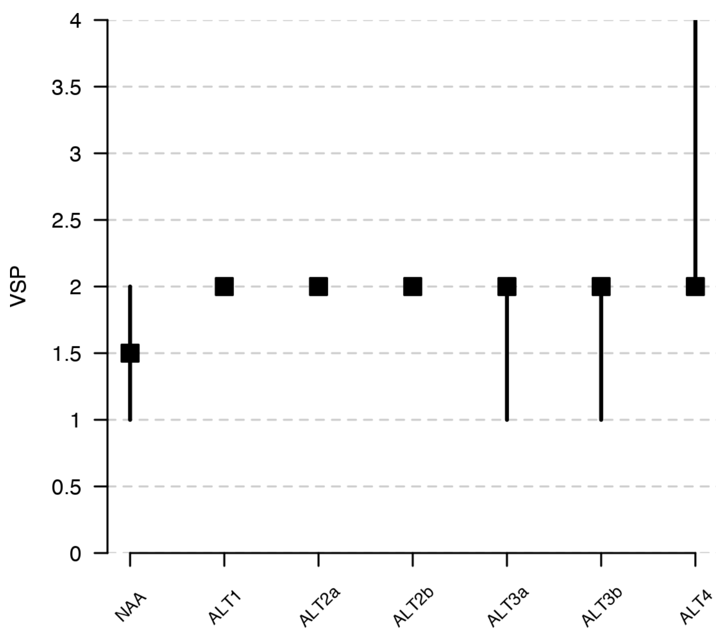


Figure 12.6.10. Median VSP score for Middle Fork Willamette River Chinook salmon juvenile life history diversity under alternative passage scenarios.

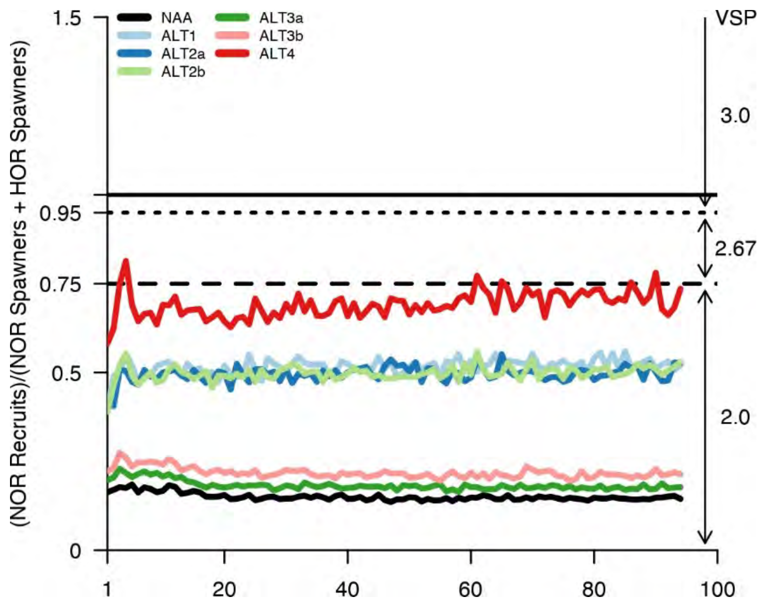


Figure 12.6.11. Chinook salmon median recruits per spawner (left axis) during 100 year model runs for Middle Fork Willamette River reaches above Dexter Dam under various alternative juvenile passage programs, with the corresponding VSP score (right axis).

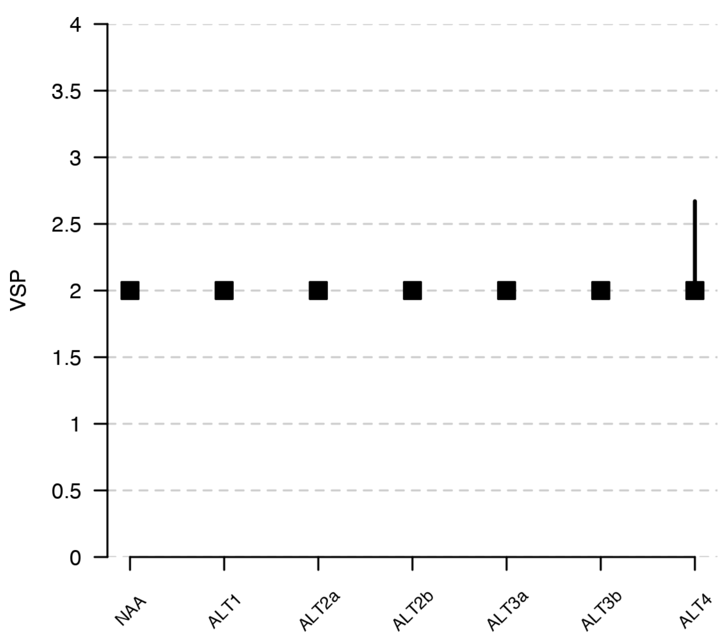


Figure 12.6.12. Median VSP score for spatial structure for Chinook salmon in the Middle Fork Willamette River under the juvenile passage alternatives.

Conclusions

The viability of modeled Chinook salmon and steelhead populations under different passage alternatives was determined by a variety of factors. The FBW was the major source of differentiation between alternatives. While a number of other factors influenced the absolute outcome of the LCMs (ocean conditions, PSM, TDG), with a few exceptions these factors were not the factors driving changes across alternatives. Passage structures (FSS and FSC) provided the highest passage survivals and dam passage efficiencies (DPE) compared to the operational alternatives, as determined by the FBW. Some of the operational passage alternatives, whether spill and/or drawdowns, were forecasted to have relatively high passage survivals, but not as high as predicted for structures. Furthermore, DPEs for most of the operational alternatives were much lower than for structures. These differences were largely due to intermediate reservoir elevations during the process of drawing down or refilling a reservoir, processes that can take several weeks or more. When the reservoirs were not at optimal passage elevations, fish attempting to pass (as determined by the FBW) were delayed and subject to reservoir mortality while waiting to attempt passage at another life history passage window. Based on the passage survivals and efficiencies in the FBW, some of the Recruits per Spawners for some of the alternatives were well below one and clearly unable to maintain self-sustaining above-dam populations. As modeled, some populations appeared to be viable (VSP ≥3) under specific passage conditions, although some populations were still below their recovery abundances (ODFW and NMFS 2011).

Alternative Performance

Comparisons across the NAA and six alternatives for all Chinook salmon and steelhead populations suggest that the highest improvement in overall VSP status would be obtained through Alternatives 1, 2a, 2b, and 4[11], with structural passage provided at least 2 projects. Results suggest that Alternative 4 would produce the most populations (4) at full viability (VSP>3), while Alternatives 1 and 2b would produce the most populations (5) at moderate viability (VSP>2). As modeled, alternatives that relied solely on operational passage, 3a and 3b, did less well compared to the other alternatives. In Chapter 3, we discussed reasons why the FBW may underestimate the performance of operational passage scenarios. Ongoing interim operational passage studies at Upper Willamette Basin dams will likely provide critical information to inform the FBW. In comparing alternatives, it is useful to consider that our LCM is population based, and that the suite of population actions described by each of the six Alternatives was developed by the USACE based on a number of considerations. Further, the combined influence of actions in each basin on downstream conditions is not reflected in any summation of population VSP scores. Therefore, we do not identify an Alternative that is “best”, rather we have focused on within population evaluations of actions for comparison and provide

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11 Results of the LCM indicated an improvement in the total VSP score for all six populations modeled of 10.96 for Alternative 1, 10.08 for Alternative 4, 9.32 for Alternative 2a, and 8.52 for Alternative 2b. Given the uncertainties in the LCMs and the non-linearity of the VSP risk assessment, these results are thus not substantially different.

these Alternative "scores" with the caveat that there is substantial uncertainty in the combined VSP score.

Fish Benefit Workbook and Alternatives

In general, predicted population abundances reflected FBW passage efficiencies. While abundances in reaches above Green Peter Dam (South Santiam River Reaches E and F) and Hills Creek Dam (Middle Fork Willamette River Reach E) with FSS and FSC structures were not as high as at other dams, they were much higher, relatively, than operational passage options.

There were a number of parameters that likely influenced the overall abundances in these reaches; these included: pre-spawning mortality (Green Peter) and spawning capacity estimates. Spawning habitat capacity was likely limiting in Quartzville Creek, (South Santiam River, Reach F), and the Minto-Big Cliff Reach (North Santiam River, Reach F). Summer parr capacity did not appear to be limiting under existing conditions. Cumulatively, uncertainty in these parameters (in addition to the FBW itself) contribute to uncertainty in the absolute abundances of spawners; rather, we feel that relative comparisons of passage scenarios are intuitively more informative than comparisons of the specific abundances of those scenarios.

The FBW assumed a proportional allocation of juvenile fish to the dam forebay on a month to month basis, with only one attempt allowed per juvenile life history stage. This passage delay effect was more pronounced with operational passage alternatives that had lower DPEs than the alternatives with structural passage (i.e., FSS, FSC). We believe that downstream migration has both genetic (Beckman and Dickhoff 1998) and environmental triggers, and that there is considerable plasticity in the initiation of downstream emigration. Ongoing operational passage studies, especially those initiated by the Federal Court Injunction, should provide further valuable information on the behavior of juvenile salmonids approaching the dam forebays that, in some cases, are similar to some of the proposed alternatives' conditions (deeper drawdowns, for example).

There were some other FBW limitations with respect to population modeling of steelhead that have the potential to be impactful. The FBW and alternatives provided no information on the collection of and passing steelhead kelts downstream of dams; therefore, we assumed a static respawn rate. Kelts are bigger and more fecund, and in some years they have the potential to make substantial reproductive contributions to the populations.

In some alternatives with juvenile collection structures, it was assumed that juveniles were collected and transported downstream of the dam lowest in the tributary. This avoided multiple passage mortalities and exposure to below-dam TDG in some cases. For the purposes of the LCM modeling, we assumed that collection, transportation, and release had a minimal effect on survival based on recent studies.

Available Data

As with any model, we relied on data inputs to inform parameters. Where there is uncertainty in the accuracy of these inputs, we attempted to be conservative in our estimates. Most of the alternatives represented novel operations or structures for which there is no empirical example in

the Willamette Basin. Similarly, each dam structure has unique characteristics, which may facilitate or detract from juvenile passage success and make it difficult to broadly apply passage data from one dam to others.

Information on the status of winter steelhead in the Upper Willamette River was very limited, especially for juvenile life history stages. And what data are available are problematic. For example, collection efficiency at the Foster Dam adult collection facility has been generally poor since it came online. Thus, returns to that facility may underestimate the productivity of the reach above Foster Dam. Spawning adults may not be able to complete their life cycle in their native reach. The LCMs run under the assumption that adult progeny of fish originating above dams find their way to their respective collection facilities for transportation to their natal reach, but clearly unsuccessful attraction of returns to adult collection facilities represents a potential for negatively affecting production in reaches above dams. Nevertheless, there is more uncertainty in the parameters and, hence, results for steelhead populations than for Chinook salmon populations.

LCM calibration was limited by available data, with temporal and spatial constraints. We were restricted to using recent adult return and spawner data for the three Chinook salmon populations' LCMs that we were able to ABC-calibrate. Furthermore, we conducted the calibration under the NAA alternative conditions which are not necessarily reflective of the conditions that generated the observed returns data.

In estimating Chinook salmon ocean survival parameters, we relied on SAR estimates based on coded wire tagged juvenile recoveries from hatchery releases specific to each of the four tributaries. SAR values were highly variable from year to year, and differences among releases and recoveries from different hatcheries for the same ocean year were evident. Additionally, SAR estimates for naturally-produced Chinook salmon in the Clackamas River were considerably higher than for hatchery fish in the four Upper Willamette tributaries, suggesting the return rates for natural-origin fish in the Upper Willamette River may be higher.

There was likely some redundancy in mortality factors inherent in hatchery-based SAR (survival from release to return to the hatchery or fishery) and en route mortality. In the calibration process we attempted to adjust the mortalities to compensate for this overlap in effects.

Other factors with the potential to impact the alternatives

This study did not include an optimization of hatchery program operations nor a full reintroduction strategy evaluation. We incorporated into the LCMs the potential for hatchery program production reduction in response to improvements in natural production and some potential domestication effects. Both hatchery program operations and outplanting strategies would have the potential to influence the outcomes of the alternatives. This could be an area of further exploration and evaluation of the alternatives.

Another potentially important contributor to population dynamics are pollutants. While some recent research has documented juvenile growth changes related to the spatial extent of legacy contaminants in the lower Willamette River (Lundin et al. 2021) more information is needed to

understand the role of pollutants and their potential for carry over and interactive effects in the FBW EIS alternatives context (also, see Lundin et al. 2019). These effects would be universal across alternatives, although changes in river flows and migration timing may have an effect on exposure.

Finally, a recent surge in pinniped predation of winter steelhead below Willamette Falls has been mitigated through pinniped control measures. During periods of low steelhead abundance at Willamette Falls, relatively modest increases in the numbers of pinnipeds had a disproportionately outsized impact on adult steelhead returns (Falcy 2017). As steelhead and Chinook salmon populations continue to persist at relatively low abundances any number of factors (for example, environmental and climatic events) can have a dramatic effect on population persistence.

Uncertainties

For above project reaches, there was considerable uncertainty in the calibration under NAA. With the exception of a few return years for some projects, we lacked the physical tag or genetic pedigree data to verify that natural origin (unmarked) adults returning to downstream collection sites are the progeny of fish that spawned above the project. Further, project operations during recent years (when adult returns were considered in the NAA calibrations) were not necessarily similar to those described in the NAA. Much of the fish migration/passage information was gathered under operational conditions that may differ significantly from those proposed in the alternatives.

We did not attempt to predict the long-term effects of operational drawdowns on reservoir conditions. We expect that these drawdowns would change the composition of fish fauna in the reservoir as well as the underlying productivity of the reservoir itself, as has been observed in Fall Creek (Murphy et al. 2019), but there is considerable uncertainty in quantifying this effect.

We did not include the future effects of climate change on freshwater or ocean survival. In most cases we assumed stationarity (e.g., values fixed) for a variety of parameters when running alternatives. Unless specifically identified, we utilized the NAA parameters to inform the models where alternative-based parameters were not provided. For most alternatives, we were provided with downstream temperature and TDG estimates. The USACE intends to include a qualitative analysis of potential climate change impacts on the LCMs' results for alternatives.

Lastly, it is difficult to predict how fish will respond to changes in reservoir elevations or outlet flow. Studies remain to be done to understand what conditions will initiate downstream migratory behavior or encourage residency.

In developing our LCMs we have attempted to account for major sources of variability in survival throughout the life cycle of Chinook salmon and winter steelhead. In modeling the attempt to reestablish self-sustaining subpopulations above the high head dams in the Willamette Basin, it became apparent that only those alternatives that provided both high passage efficiency and high survival would be successful at improving overall population viability. As determined by the FBW, these options were largely limited to structural designs, although we suggest that more empirical studies of operational passage are needed to validate the FBW. Passage options

that provided only modest improvements in overall abundance were still likely to have high probabilities of falling below the quasi-extinction thresholds, given the high variability in ocean and freshwater survivals. Finally, a model cannot capture the myriad of factors that influence survival, especially those that involve behavioral responses; therefore we encourage a more qualitative “relative” comparison of our results.

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Appendix (FBW Inputs and TDG Mortality Plots)

The following plots illustrate the Fish Benefits Workbooks (FBW) outputs for the EIS from the USACE and that were used as inputs in the NOAA Willamette life cycle models to represent alternative- and life history-specific dam survivals and dam passage efficiencies. Alternative descriptions (Table 14.1) indicate whether structural or operational passage is provided. Also included are alternative- and life stage-specific estimated mortalities associated with estimated total dissolved gas (TDG), supplied by the USACE, affecting both juveniles passing dams as well as developing alevins below dams.

In some cases, values were the same for multiple alternatives for a life stage and lines were overplotted. Additionally, some TDG mortalities are very close to zero and appear on the plots close to the x-axis

A-1

Table 14.1. Upstream and downstream passage alternatives for Upper Willamette Basin dams. FSS - floating screen structure, FSC - floating surface collector, AFF - adult fish facility, RO - regulating outlet passage, DT - diversion tunnel passage.

	Alternatives											
	1		2a		2b		3a		3b		4	
	Up	Down	Up	Down	Up	Down	Up	Down	Up	Down	Up	Down
DEX								spring spill		spring spill		
LOP		FSS		FSC		FSC		spring and fall drawdown		spring spill fall drawdown		FSS
HCR							AFF lamprey passage	spring spill fall drawdown	AFF lamprey passage	spring and fall drawdown	AFF lamprey passage	FSC
FCR								add spring spill				
CGR	add lamprey passage		add lamprey passage	FSS	add lamprey passage	spring and fall drawdown (DT)	add lamprey passage	spring and fall drawdown (RO)	add lamprey passage	spring and fall drawdown (DT)	add lamprey passage	FSS
BLU							AFF lamprey passage	fall drawdown	AFF lamprey passage	fall drawdown		
FOS		modified fish weir		modified fish weir		modified fish weir						modified fish weir
GPR	AFF	FSS	AFF lamprey passage	spring spill fall drawdown	AFF lamprey passage	spring spill fall drawdown	AFF lamprey passage	spring spill fall drawdown	AFF lamprey passage	spring and fall drawdown		
BCL		fish collected at		fish collected at DET		fish collected at		spring spill		spring spill		fish collected at

	Alternatives											
		DET				DET						DET
DET		FSS		FSS		FSS		spring and fall drawdown		spring spill fall drawdown		FSS

14.1.1. McKenzie Chinook salmon (CGR - Cougar Dam)

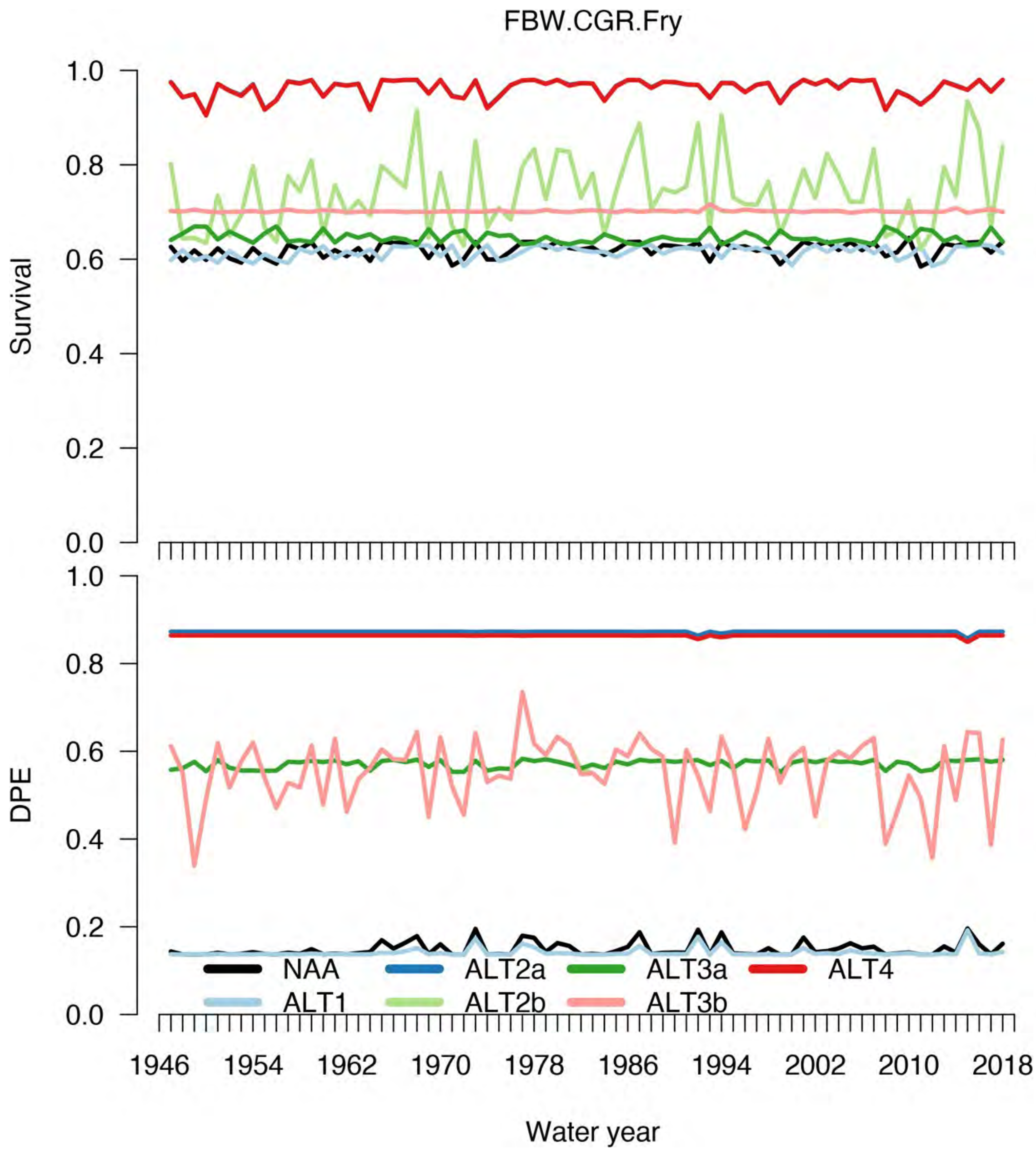


Figure 14.1.1: Cougar Dam Chinook salmon fry (spring subyearling) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

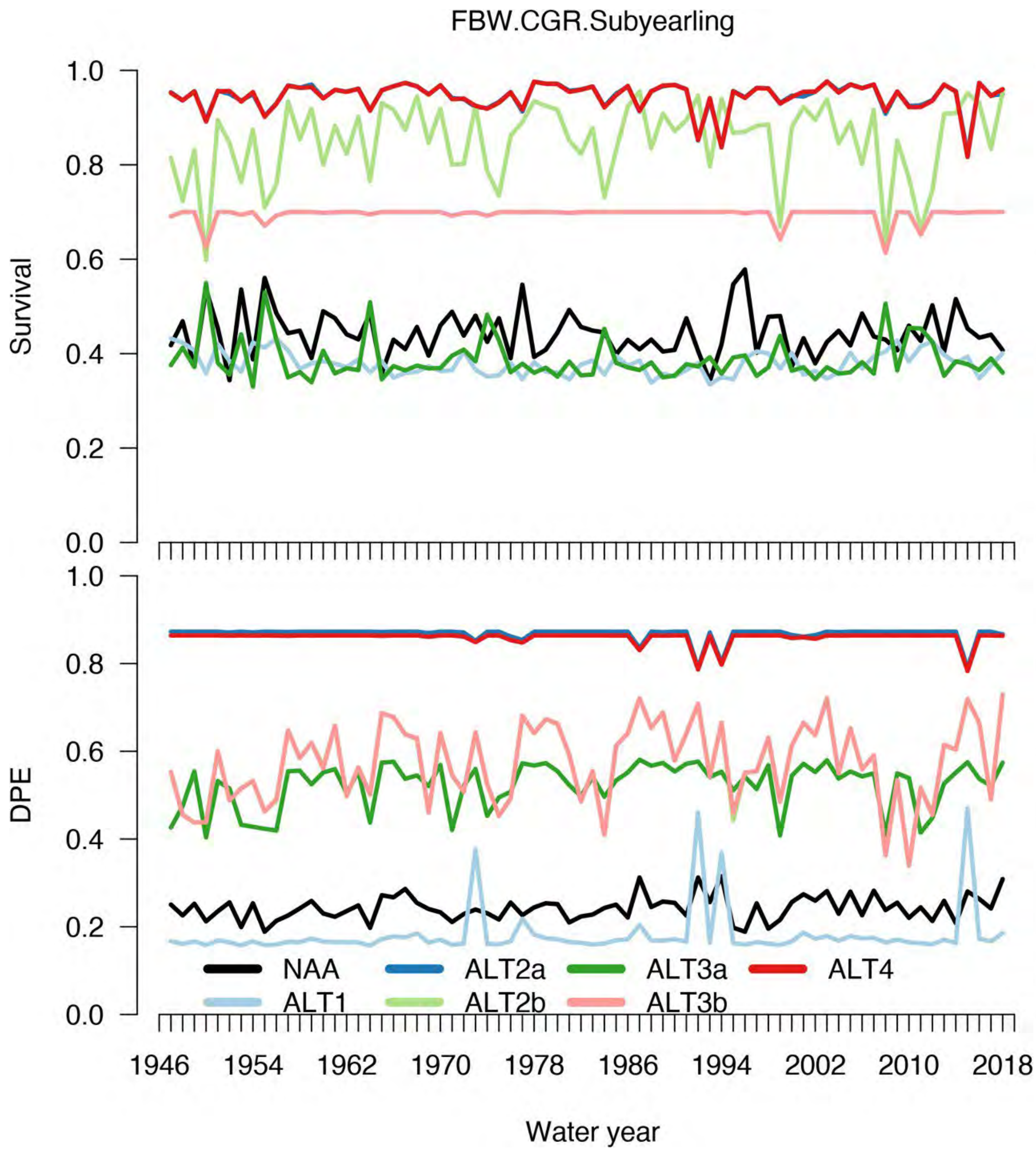


Figure 14.1.2: Cougar Dam Chinook salmon fall subyearling Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

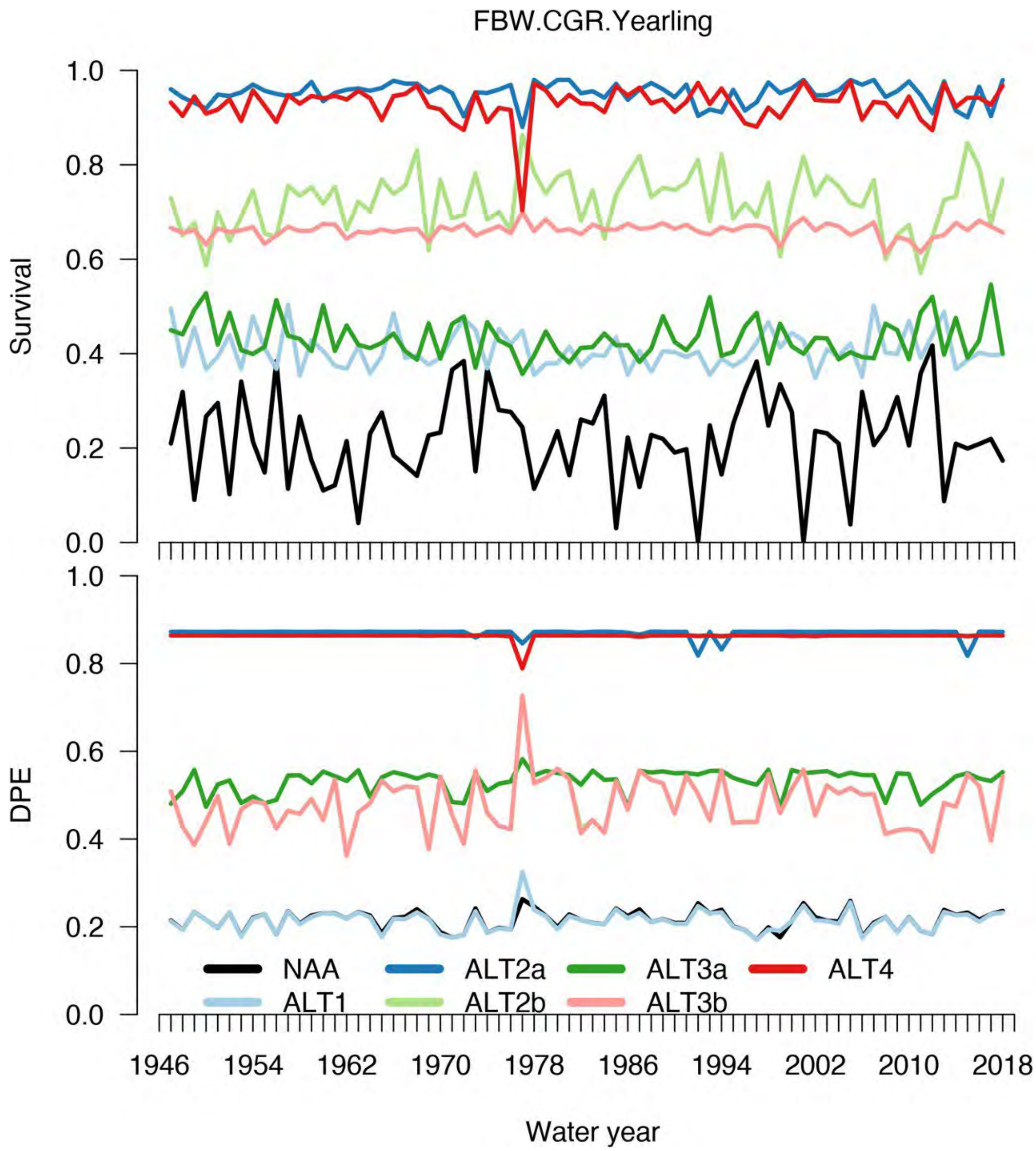


Figure 14.1.3: Cougar Dam Chinook salmon yearling Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

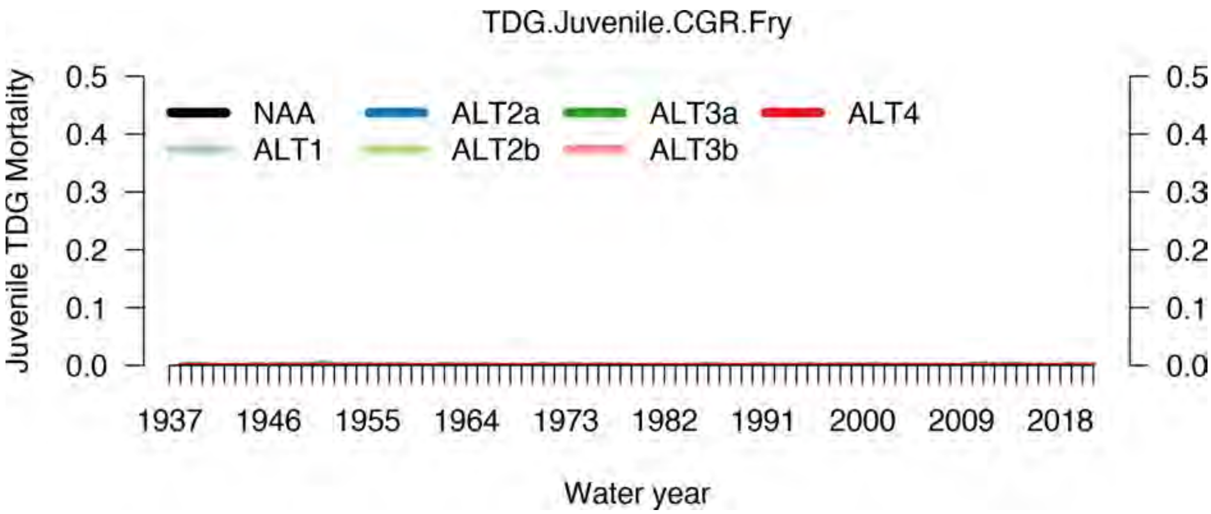


Figure 14.1.4: Cougar Dam Chinook salmon fry (spring subyearling) total dissolved gas (TDG) mortality.

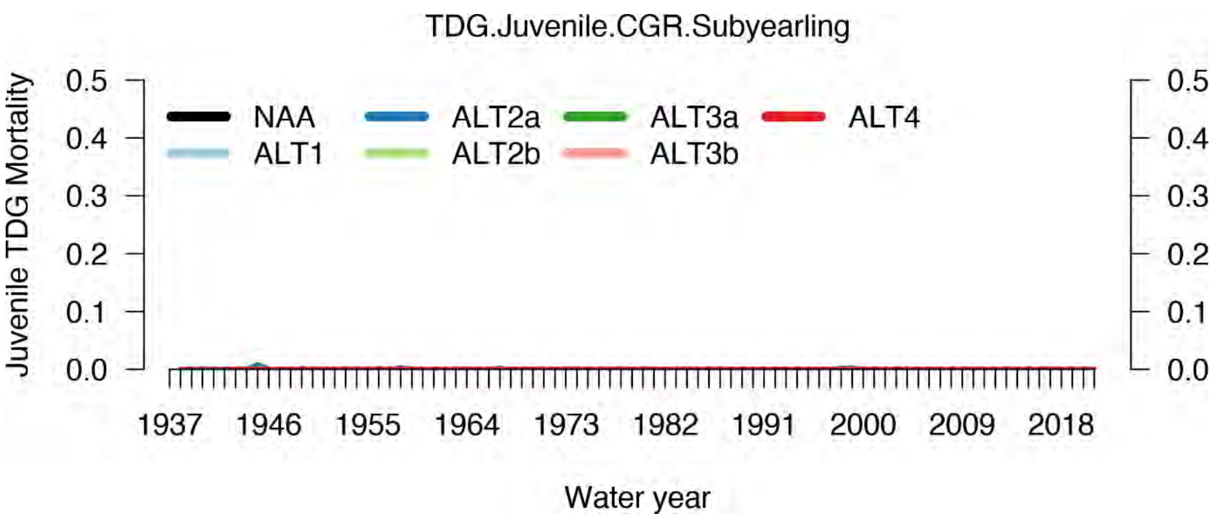


Figure 14.1.5: Cougar Dam Chinook salmon fall subyearling total dissolved gas (TDG) mortality.

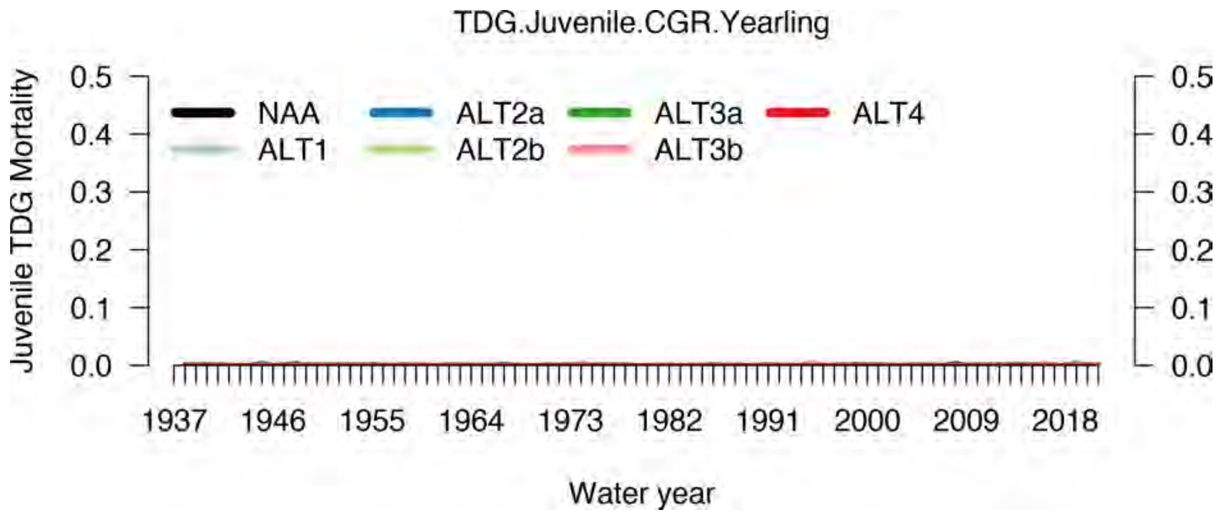


Figure 14.1.6: Cougar Dam Chinook salmon yearling total dissolved gas (TDG) mortality.

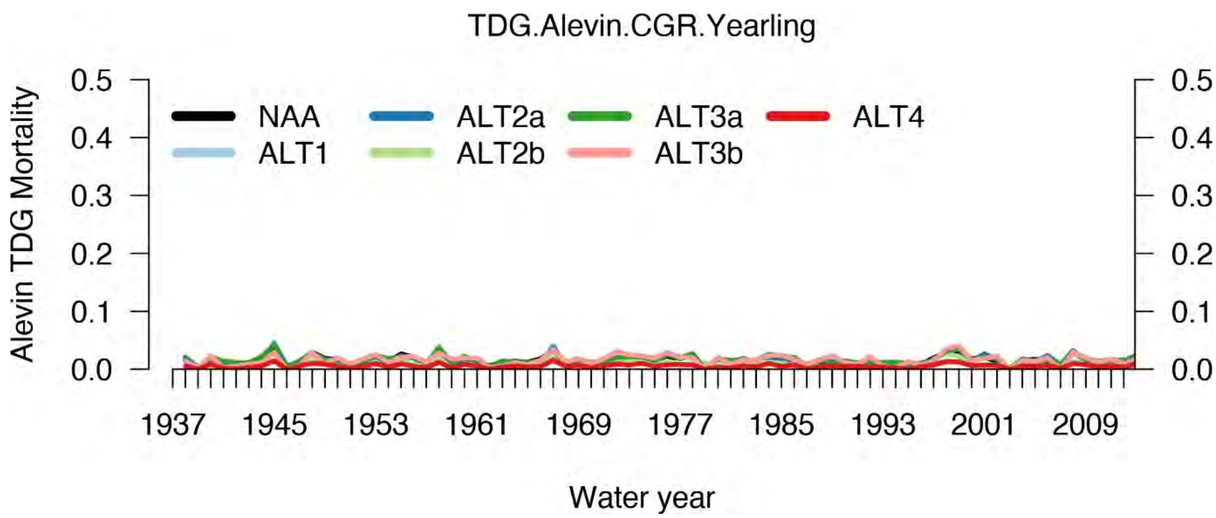


Figure 14.1.7: Cougar Dam Chinook salmon alevin total dissolved gas (TDG) mortality.

14.1.3. 14.2 Middle Fork Chinook salmon (HCR, LOP/DEX)

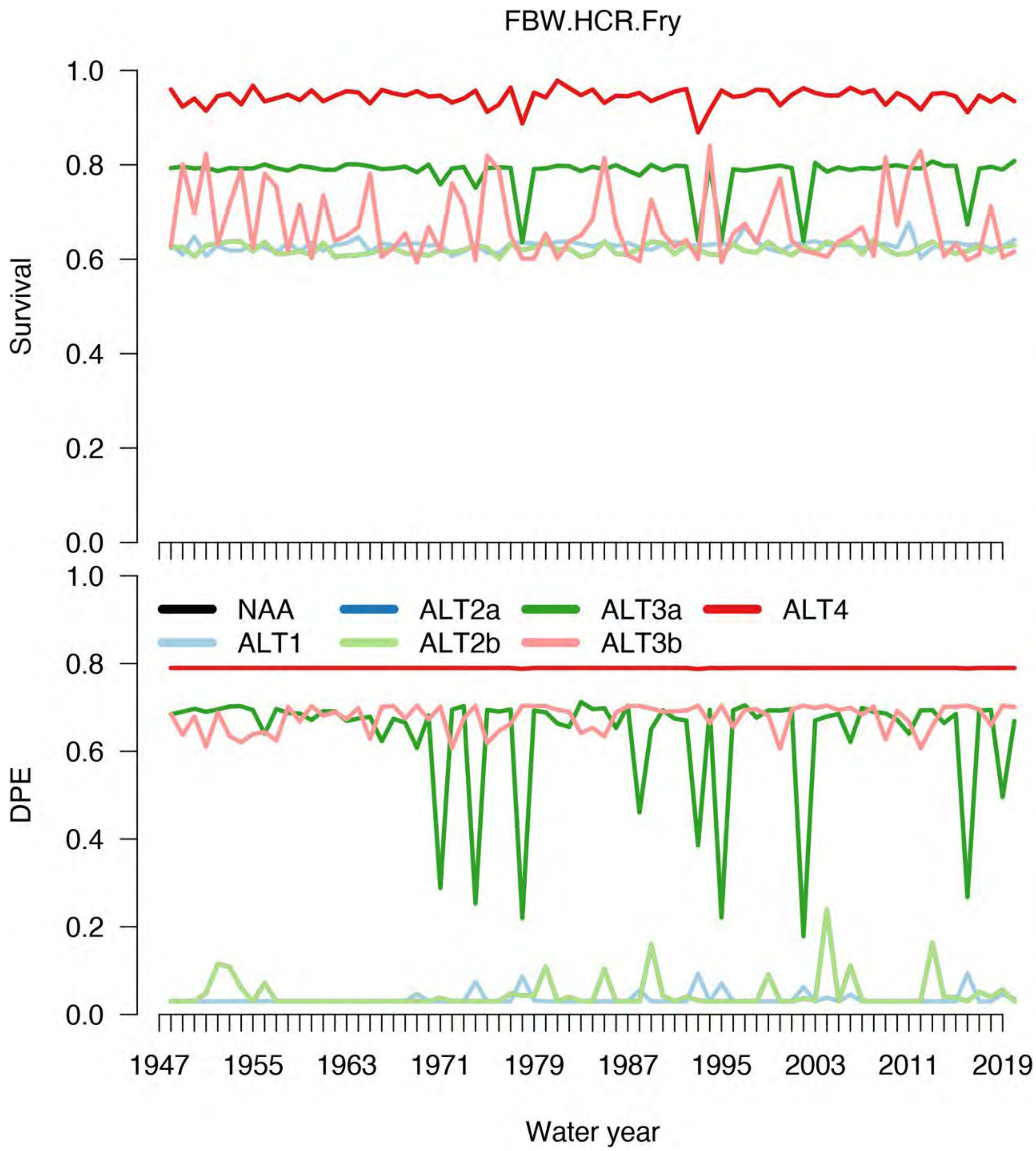
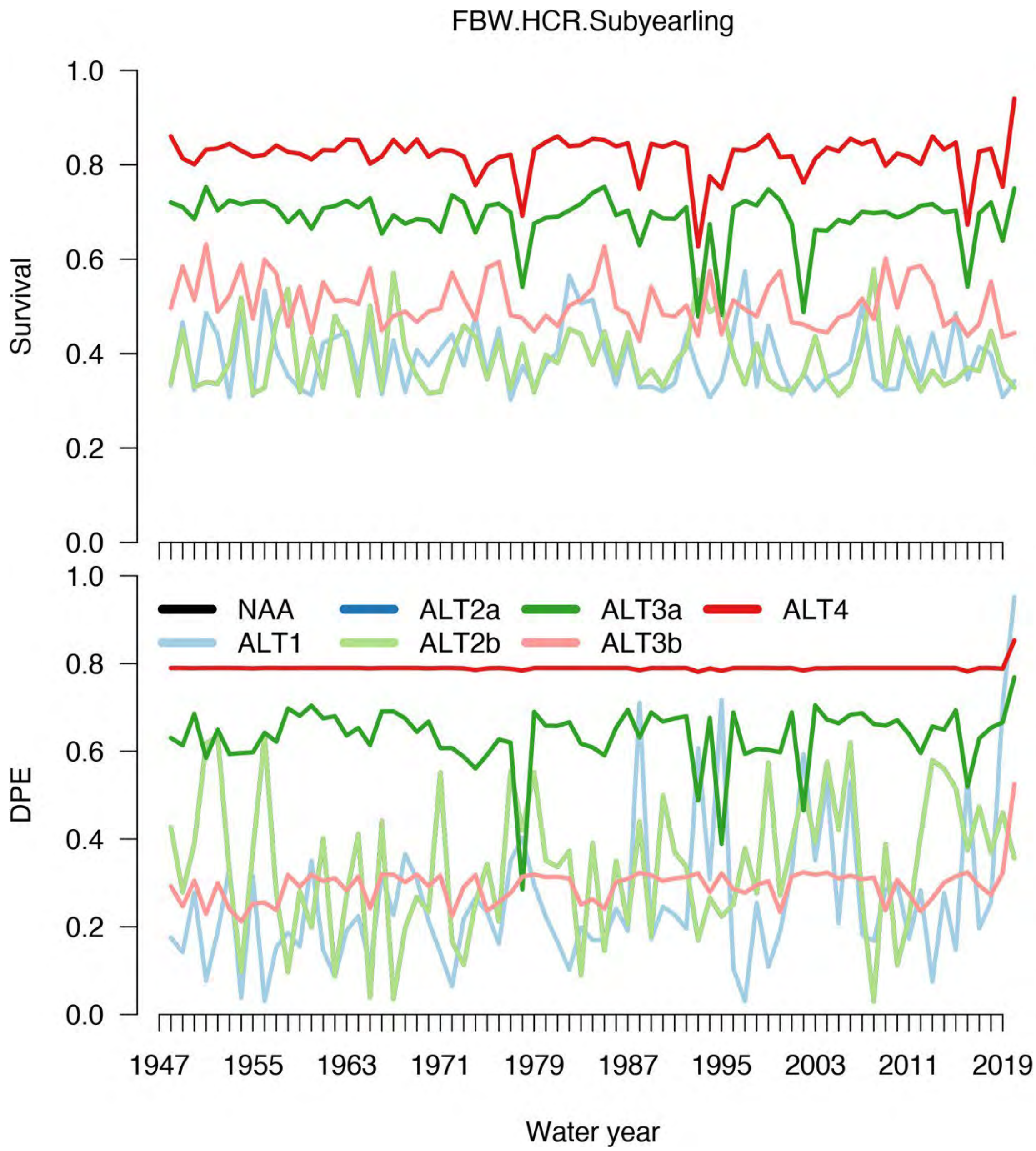


Figure 14.2.1: Hills Creek Dam Chinook salmon fry (spring subyearling) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.



14.2.2: Hills Creek Dam Chinook salmon fall subyearling Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

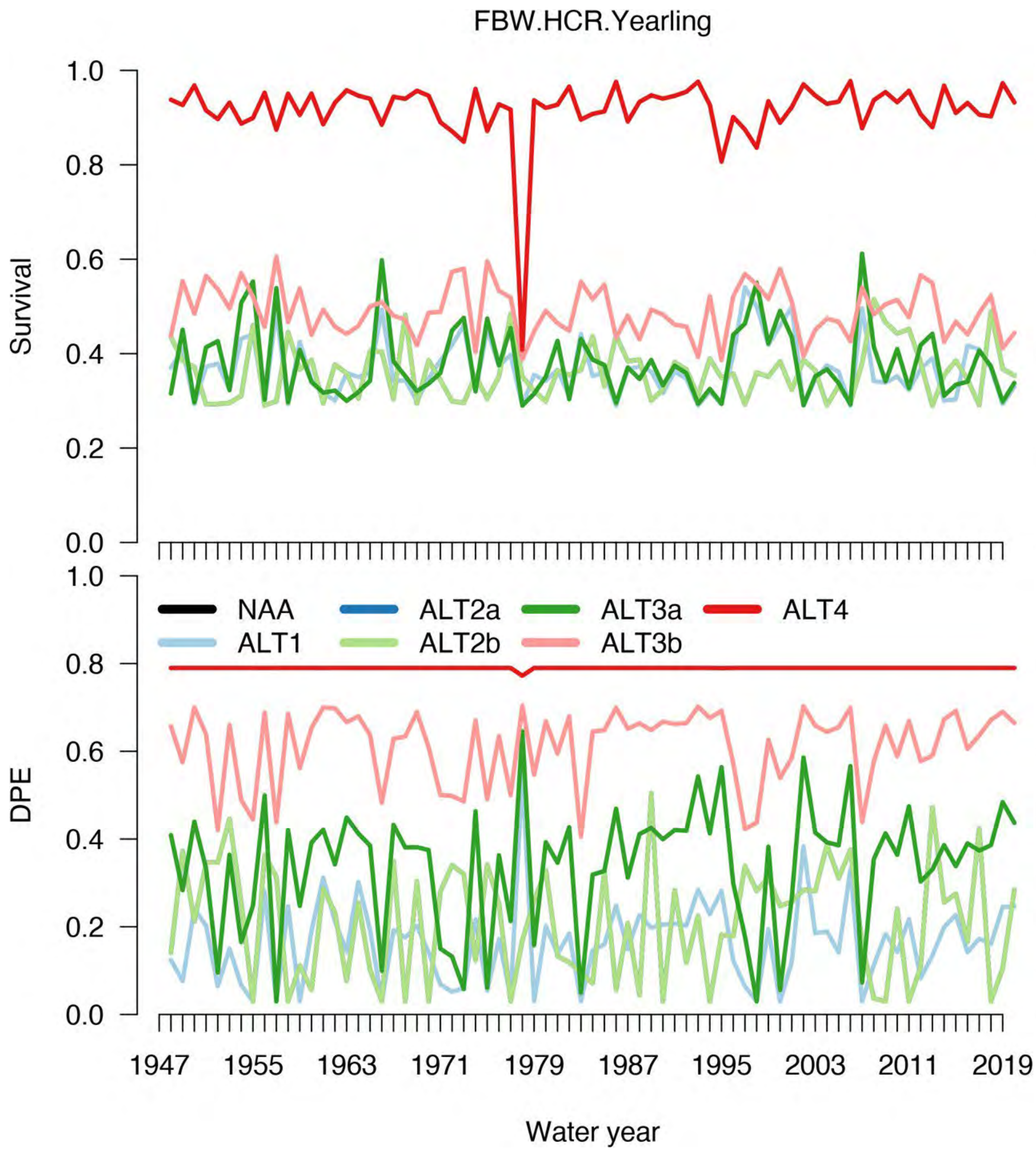


Figure 14.2.3: Hills Creek Dam Chinook salmon fall yearling Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

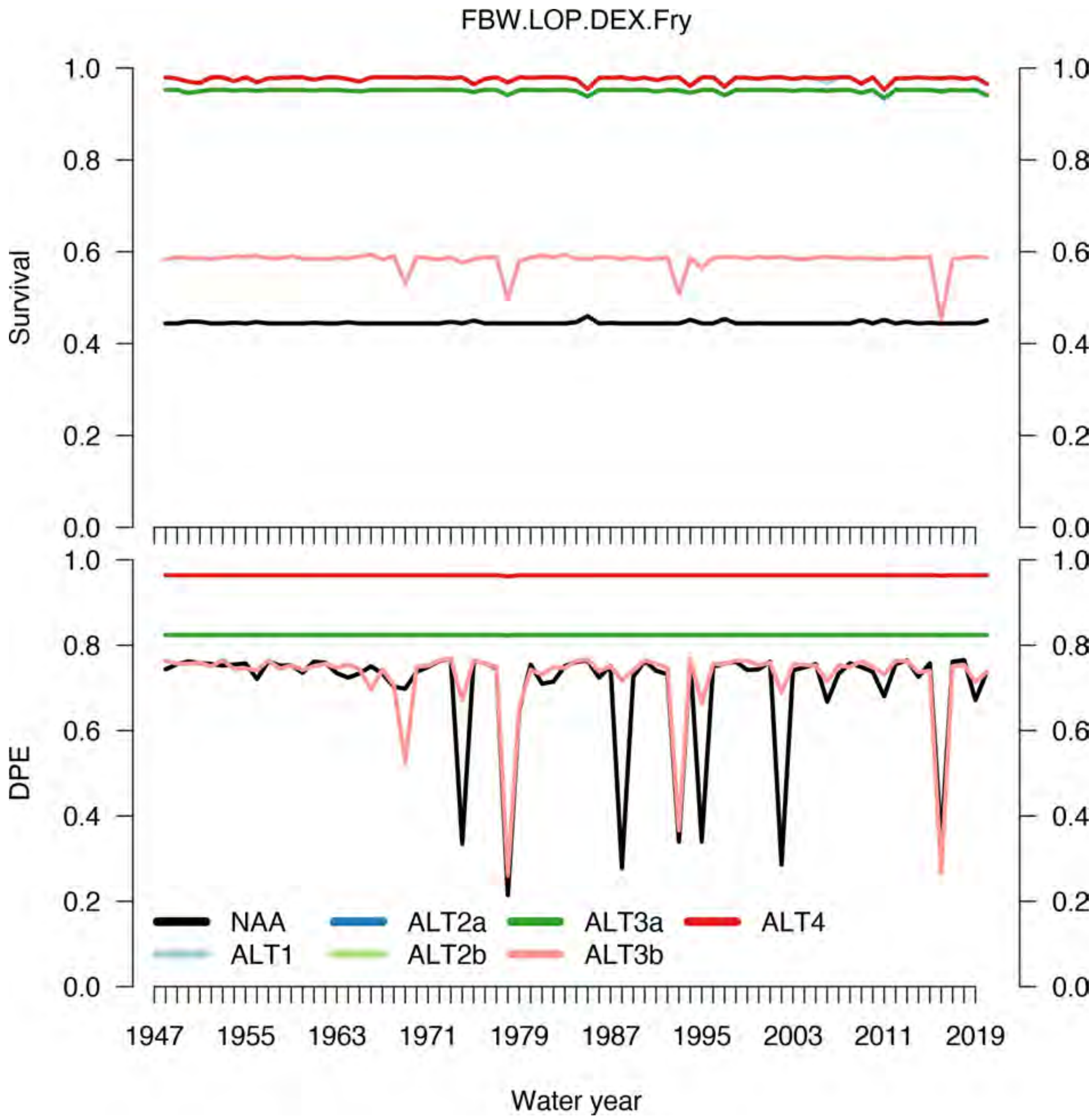


Figure 14.2.4: Lookout Point Dam and Dexter Dam Chinook salmon fry (spring subyearling) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

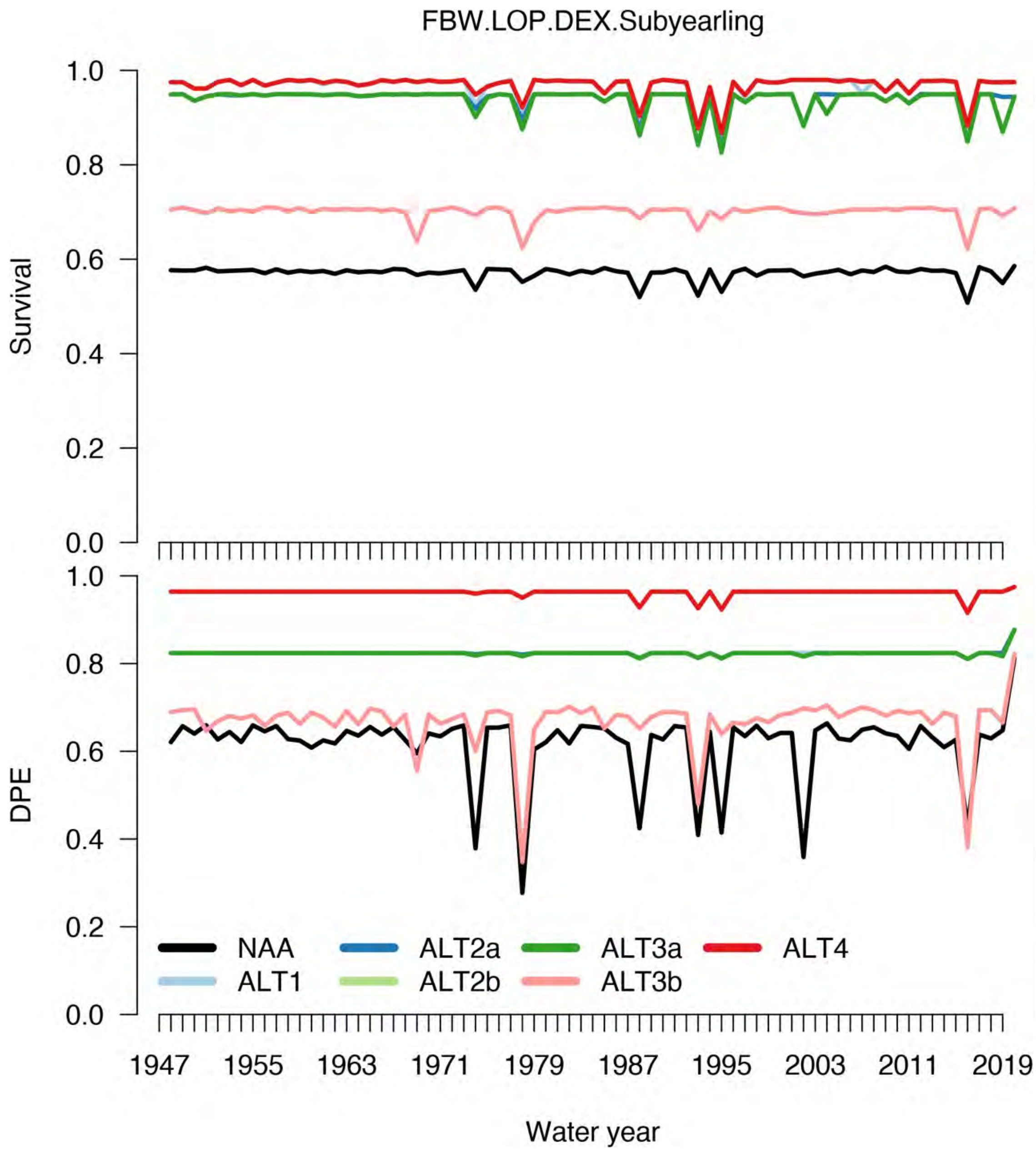


Figure 14.2.5: Lookout Point Dam and Dexter Dam Chinook salmon fall subyearling Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

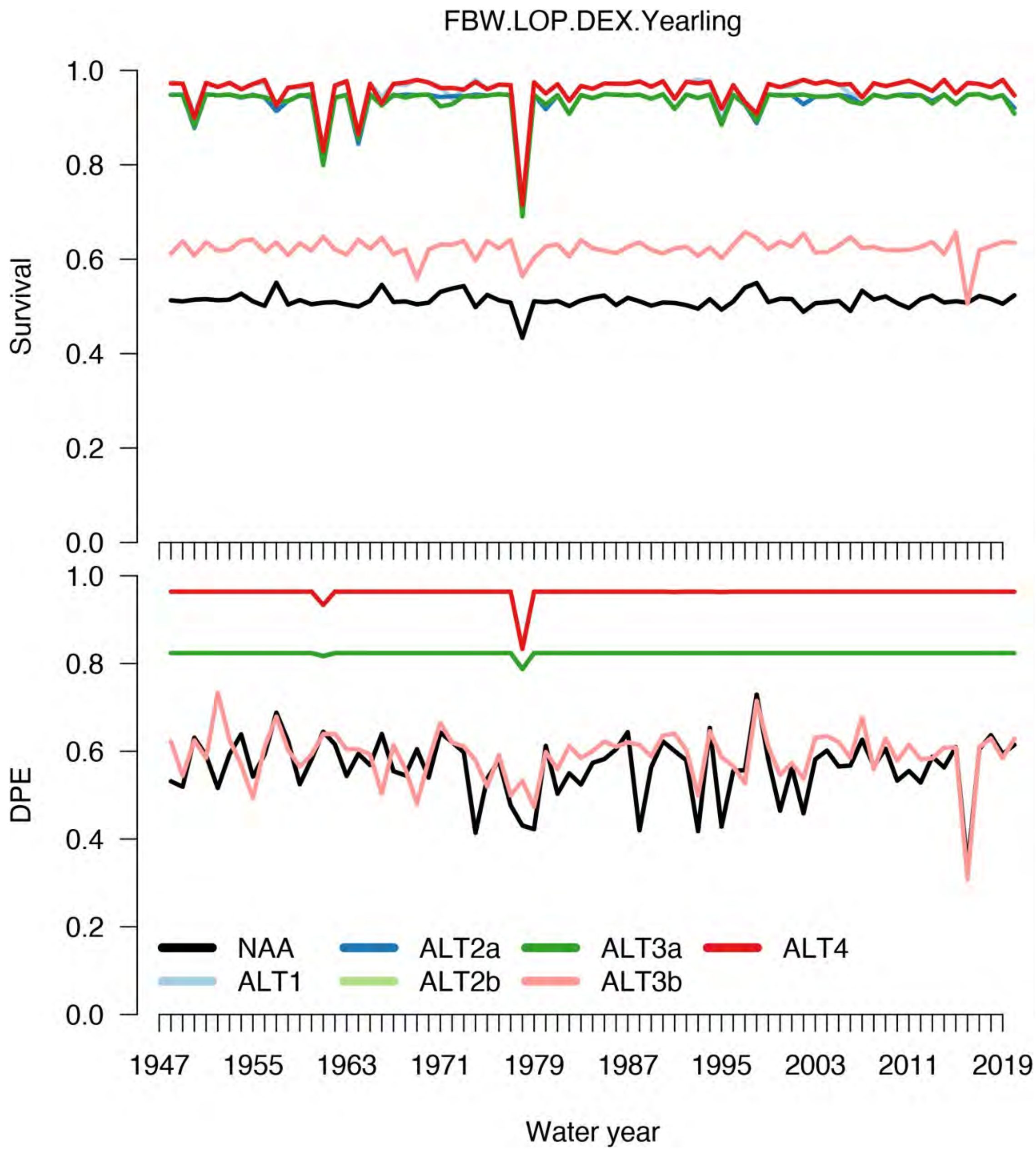


Figure 14.2.6: Lookout Point Dam and Dexter Dam Chinook salmon yearling Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

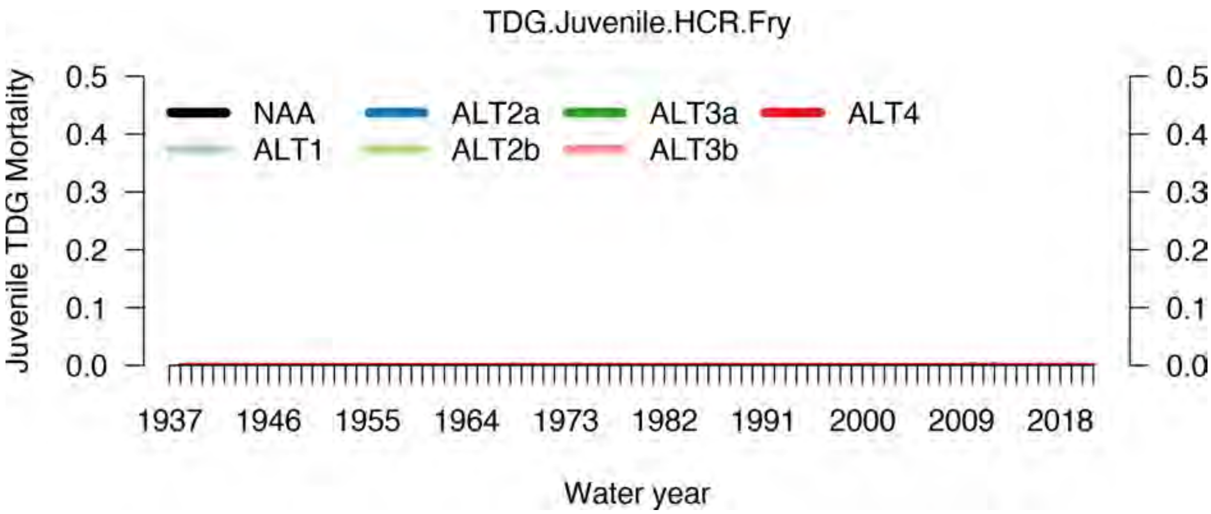


Figure 14.2.7: Hills Creek Dam fry (spring subyearling) total dissolved gas (TDG) mortality.

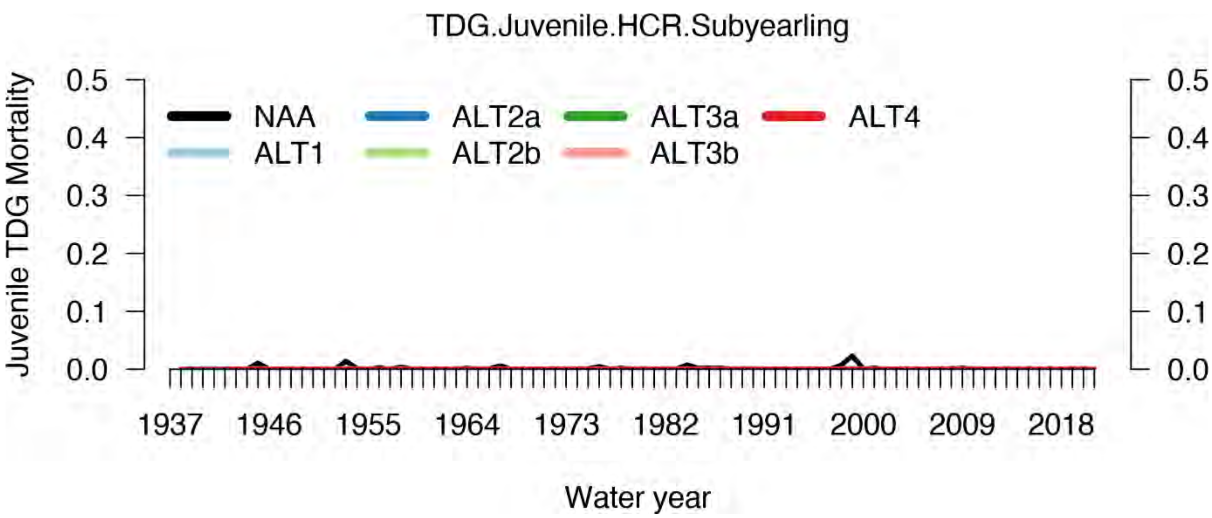


Figure 14.2.8: Hills Creek Dam fall Chinook salmon subyearling total dissolved gas (TDG) mortality.

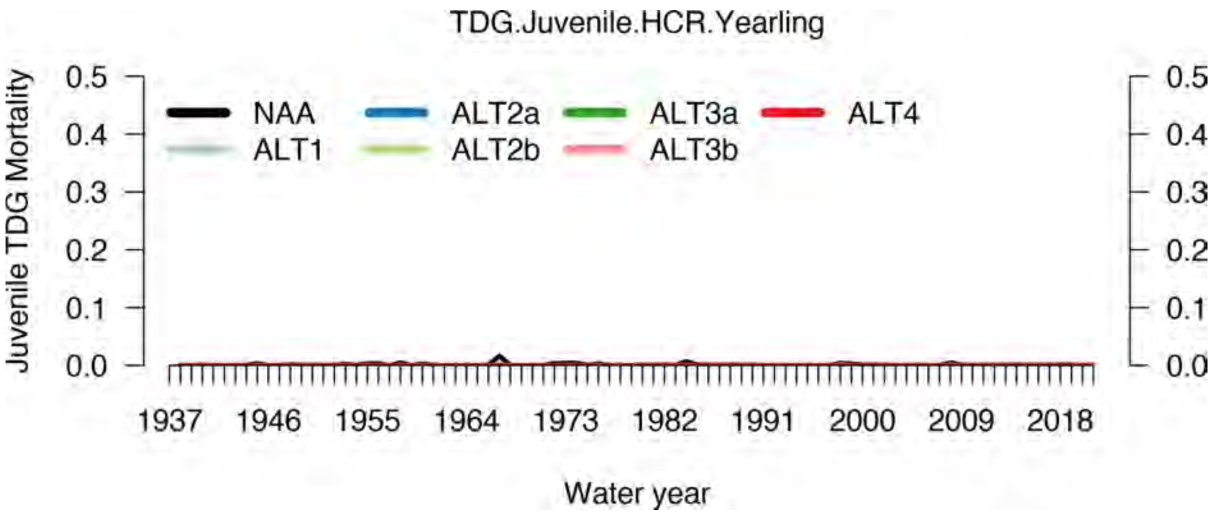


Figure 14.2.9:Hills Creek Dam Chinook salmon yearling total dissolved gas (TDG) mortality.

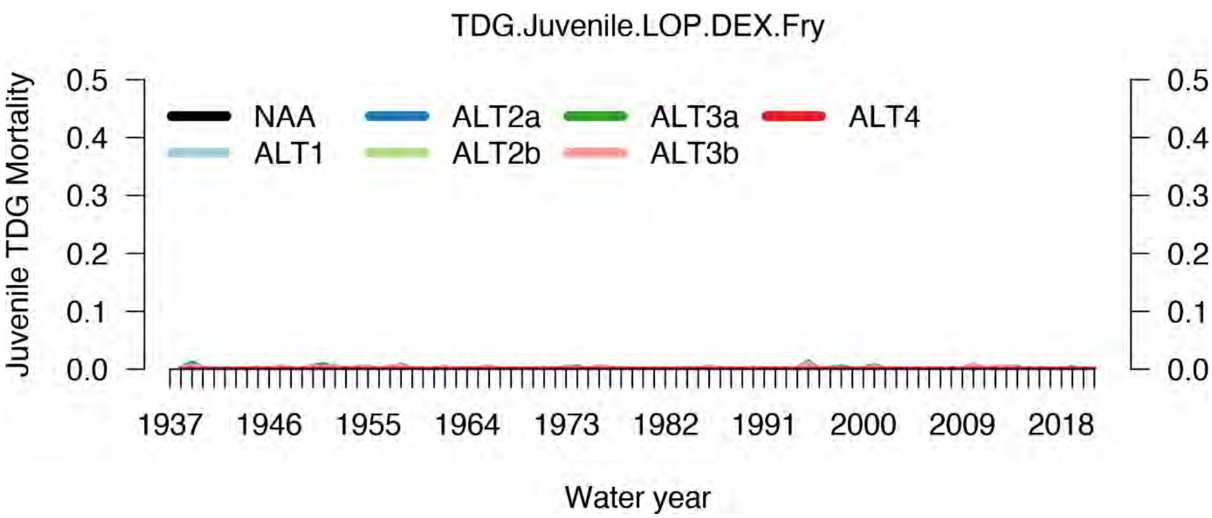


Figure 14.2.10: Lookout Point Dam and Dexter Dam Chinook salmon fry (spring subyearling) total dissolved gas (TDG) mortality.

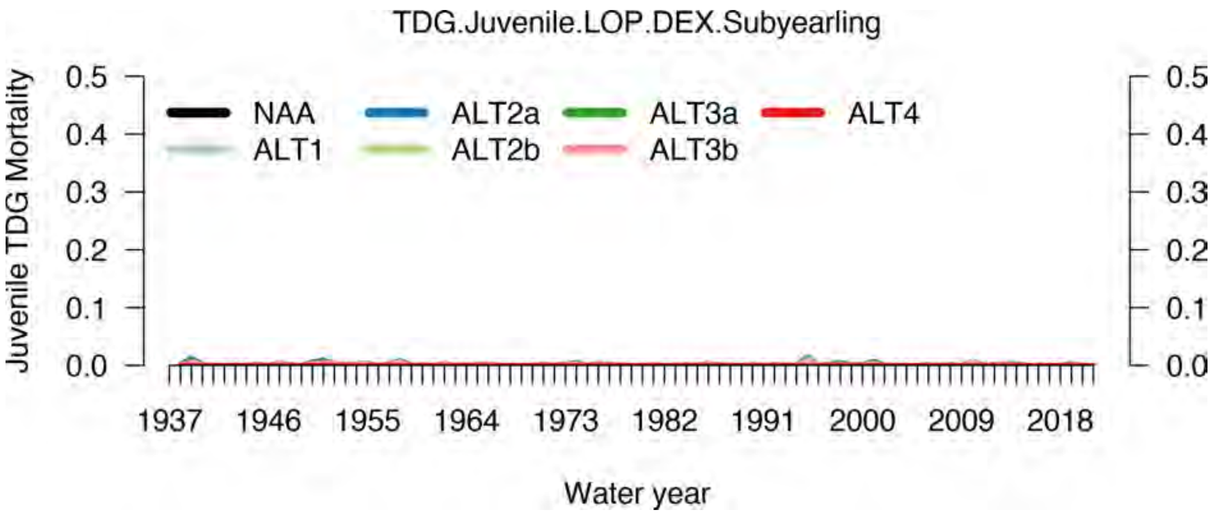


Figure 14.2.11: Lookout Point Dam and Dexter Dam Chinook salmon fall subyearling total dissolved gas (TDG) mortality.

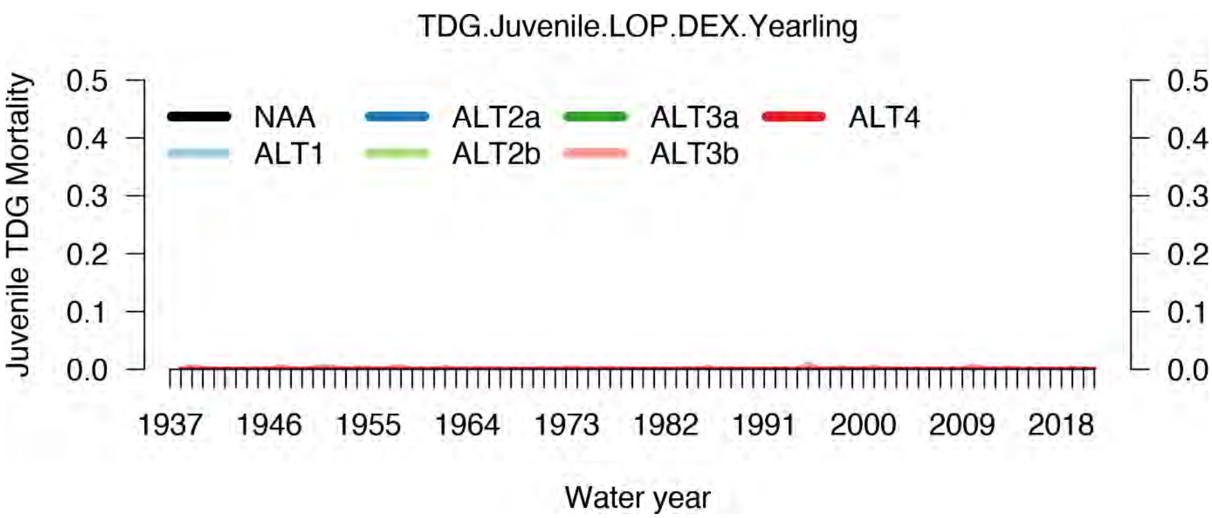


Figure 14.2.12: Lookout Point Dam and Dexter Dam Chinook salmon yearling total dissolved gas (TDG) mortality.

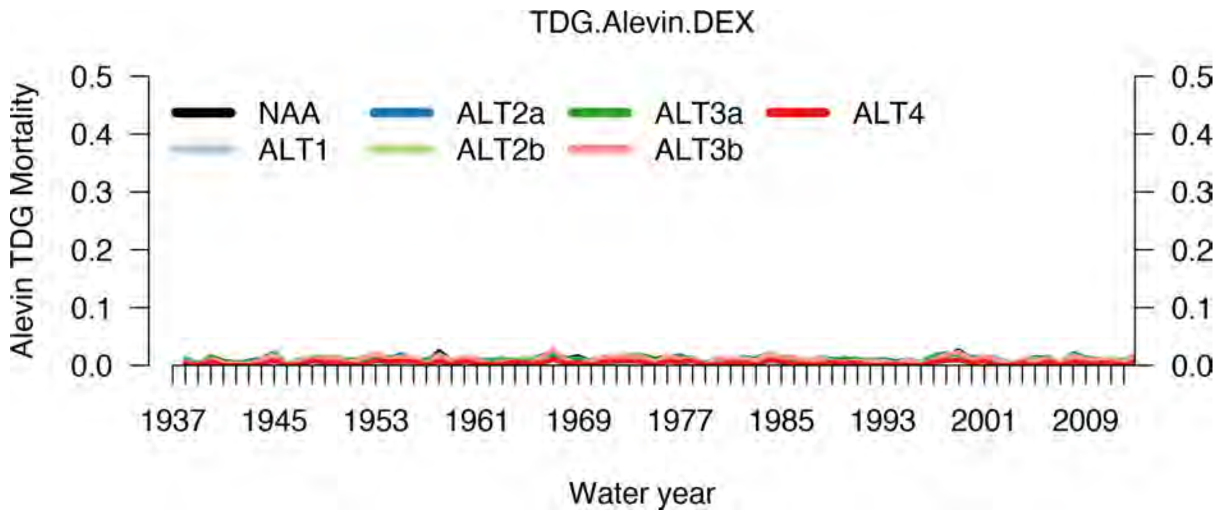


Figure 14.2.13: Dexter Dam Chinook salmon alevin total dissolved gas (TDG) mortality.

14.1.5. 14.3: South Santiam Chinook salmon (GPR, FOS)

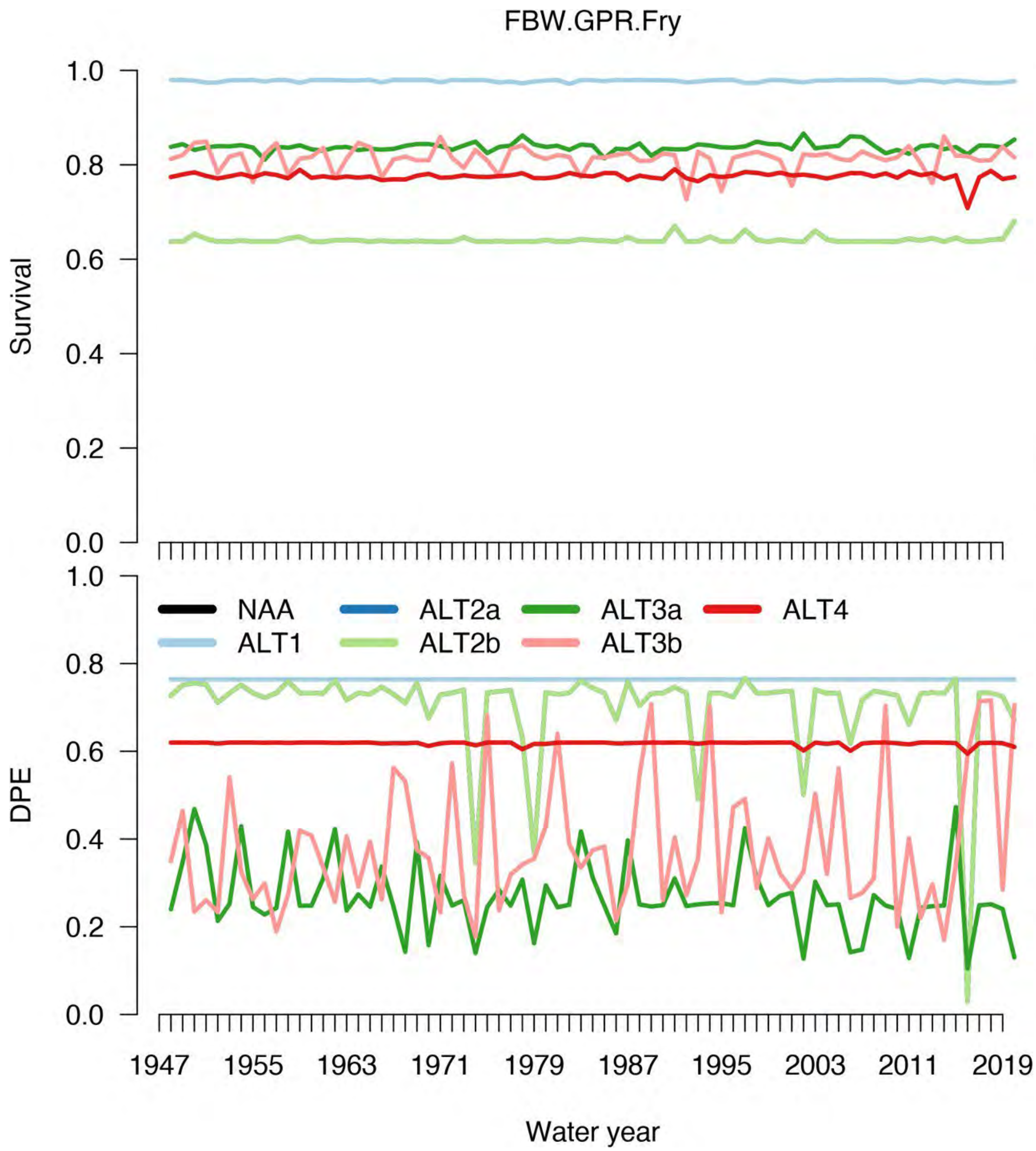


Figure: 14.3.1: Green Peter Dam Chinook salmon fry (spring subyearling) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

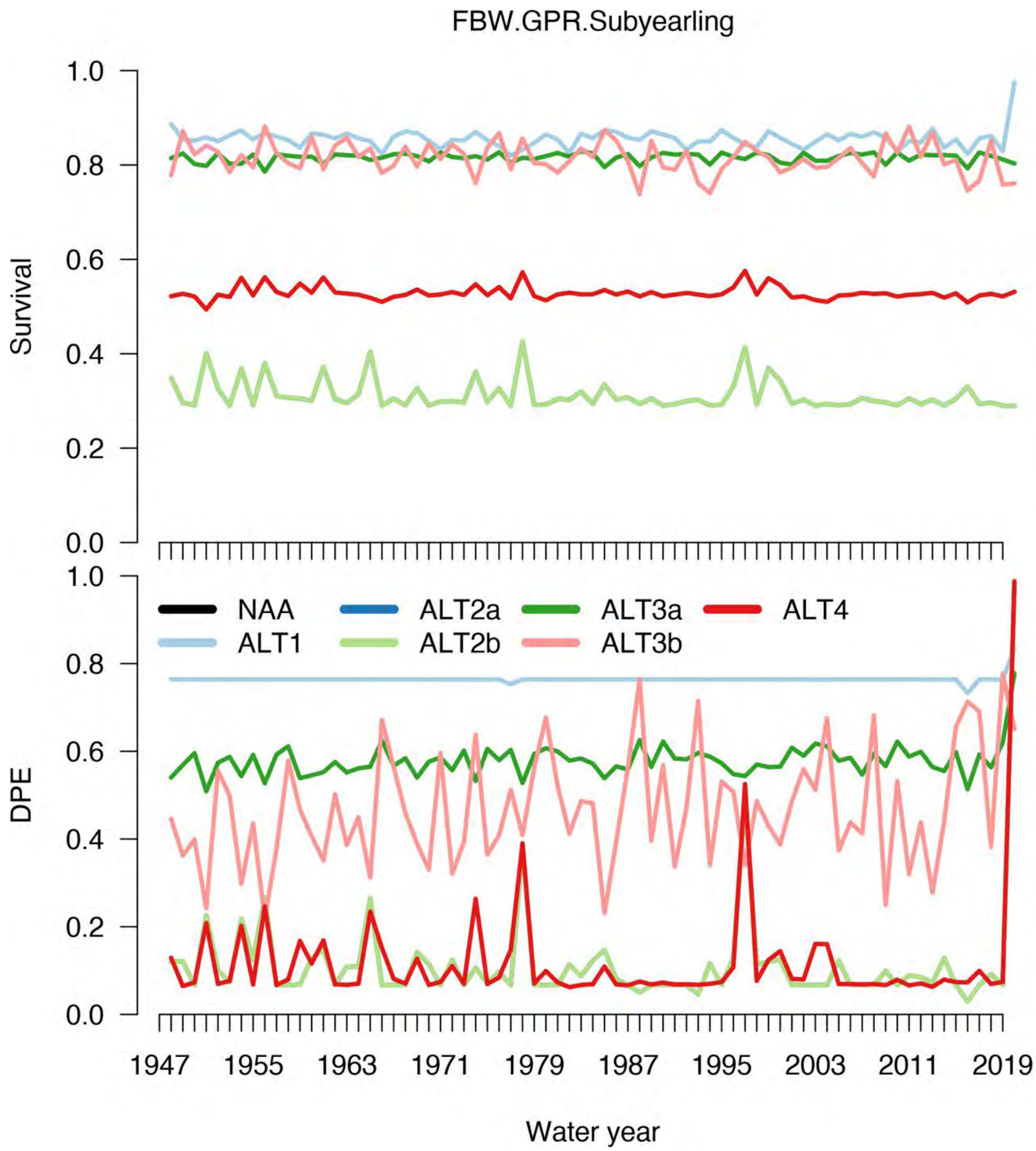


Figure 14.3.2: Green Peter Dam Chinook salmon fall subyearling Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

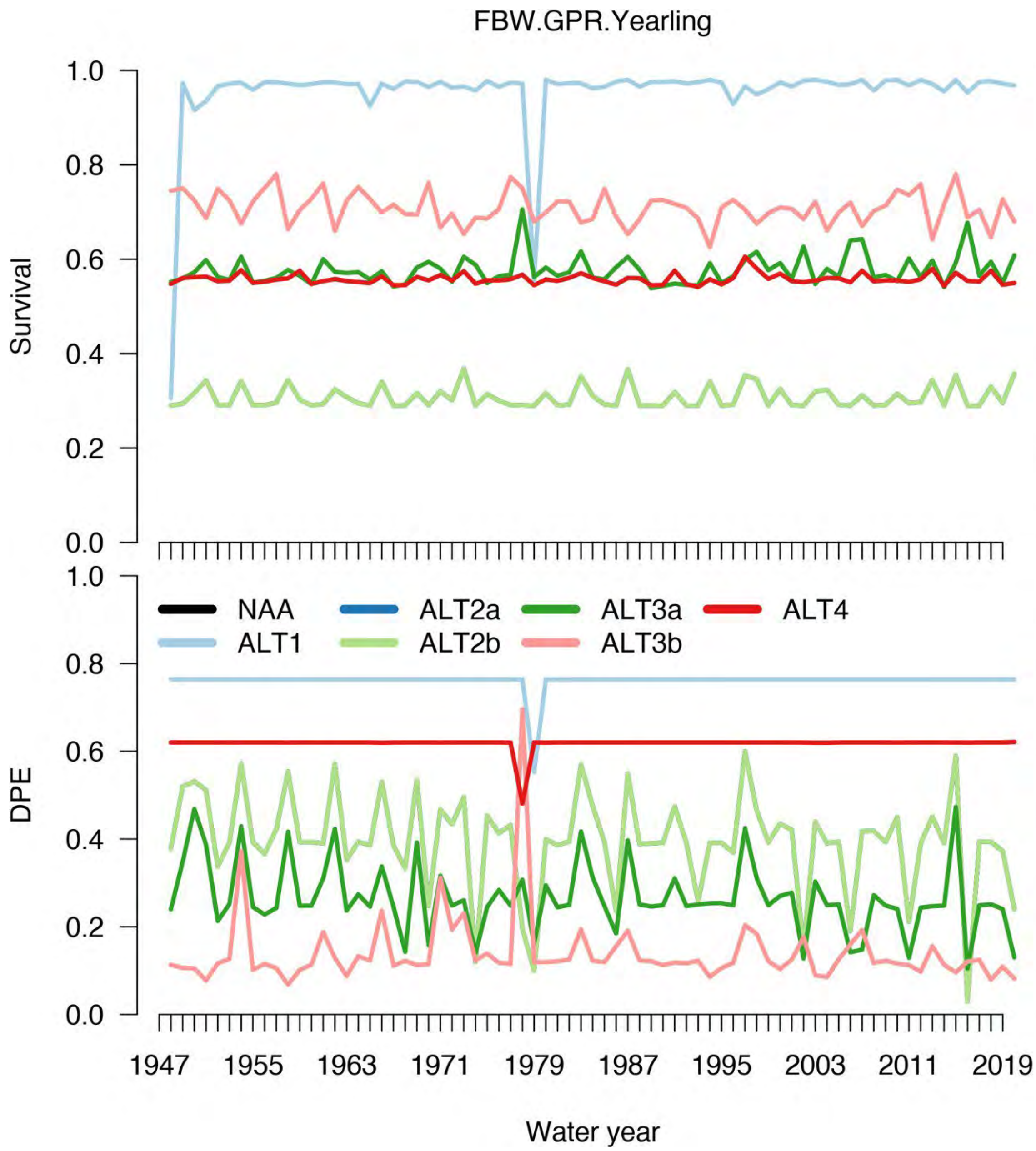


Figure 14.3.3: Green Peter Dam Chinook salmon yearling Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

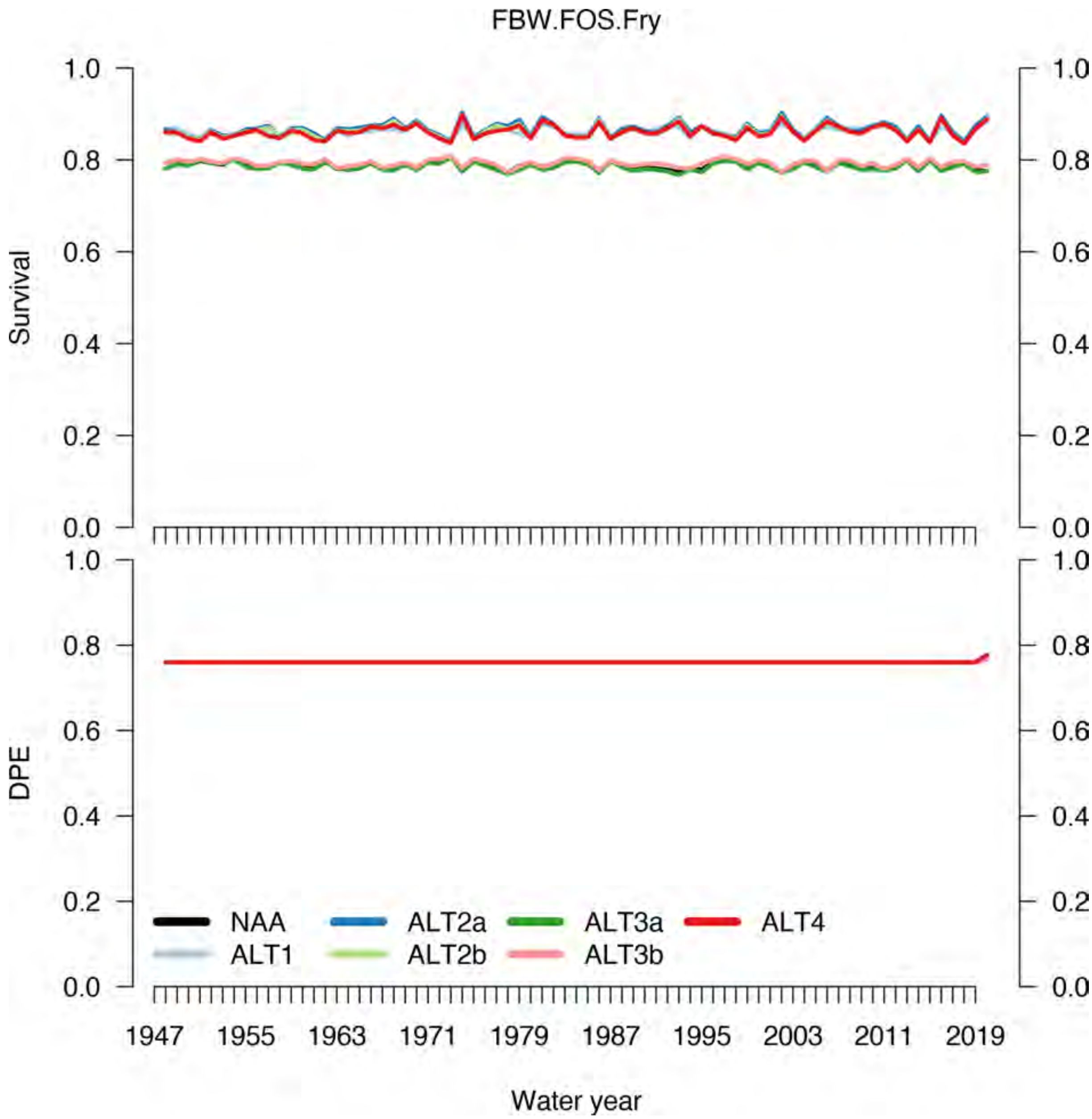


Figure 14.3.4: Foster Dam Chinook salmon fry (spring subyearling) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

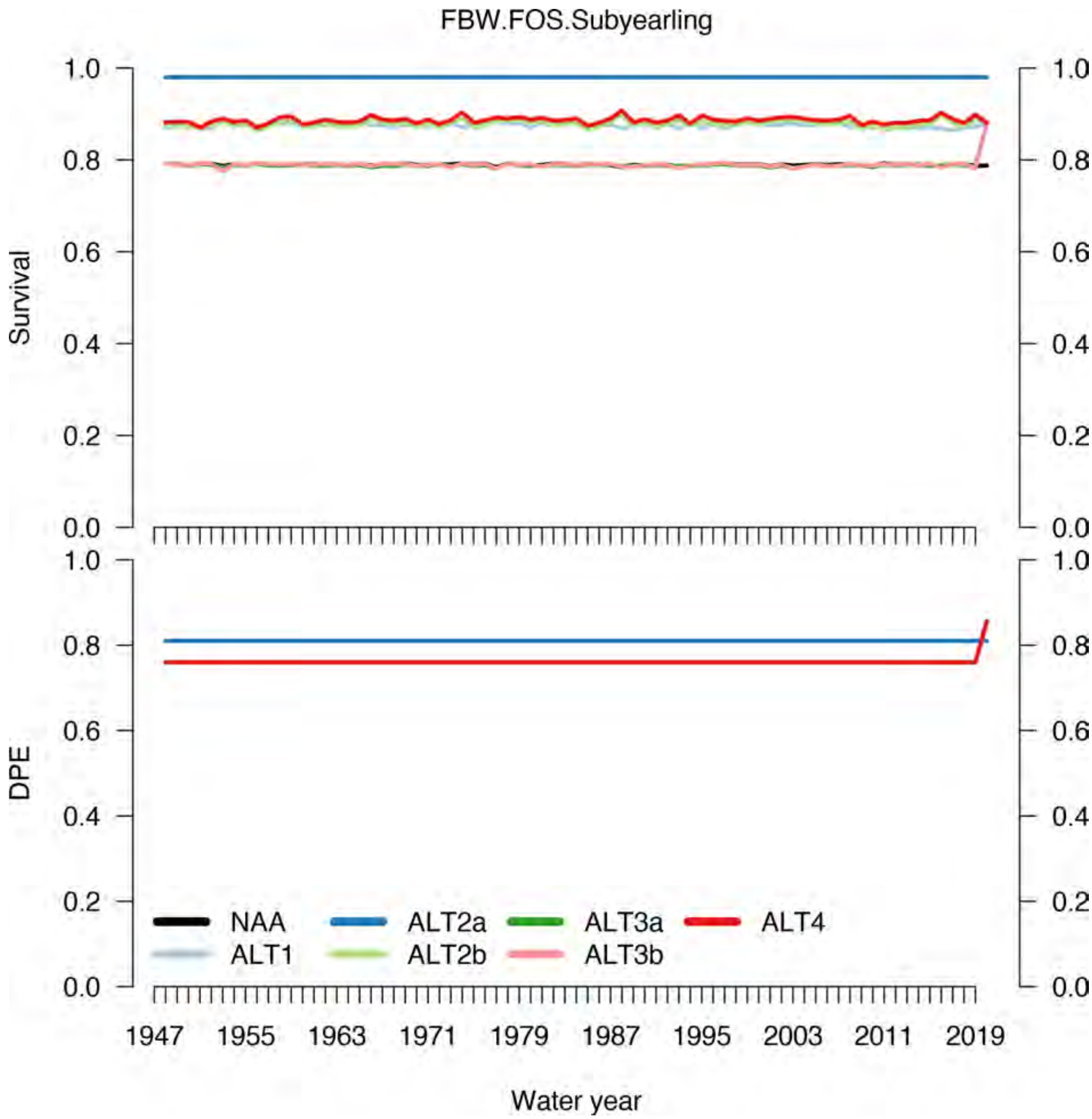


Figure 14.3.5: Foster Dam Chinook salmon fall subyearling Fish Benefit Workbook (FBW). DPE - dam passage efficiency

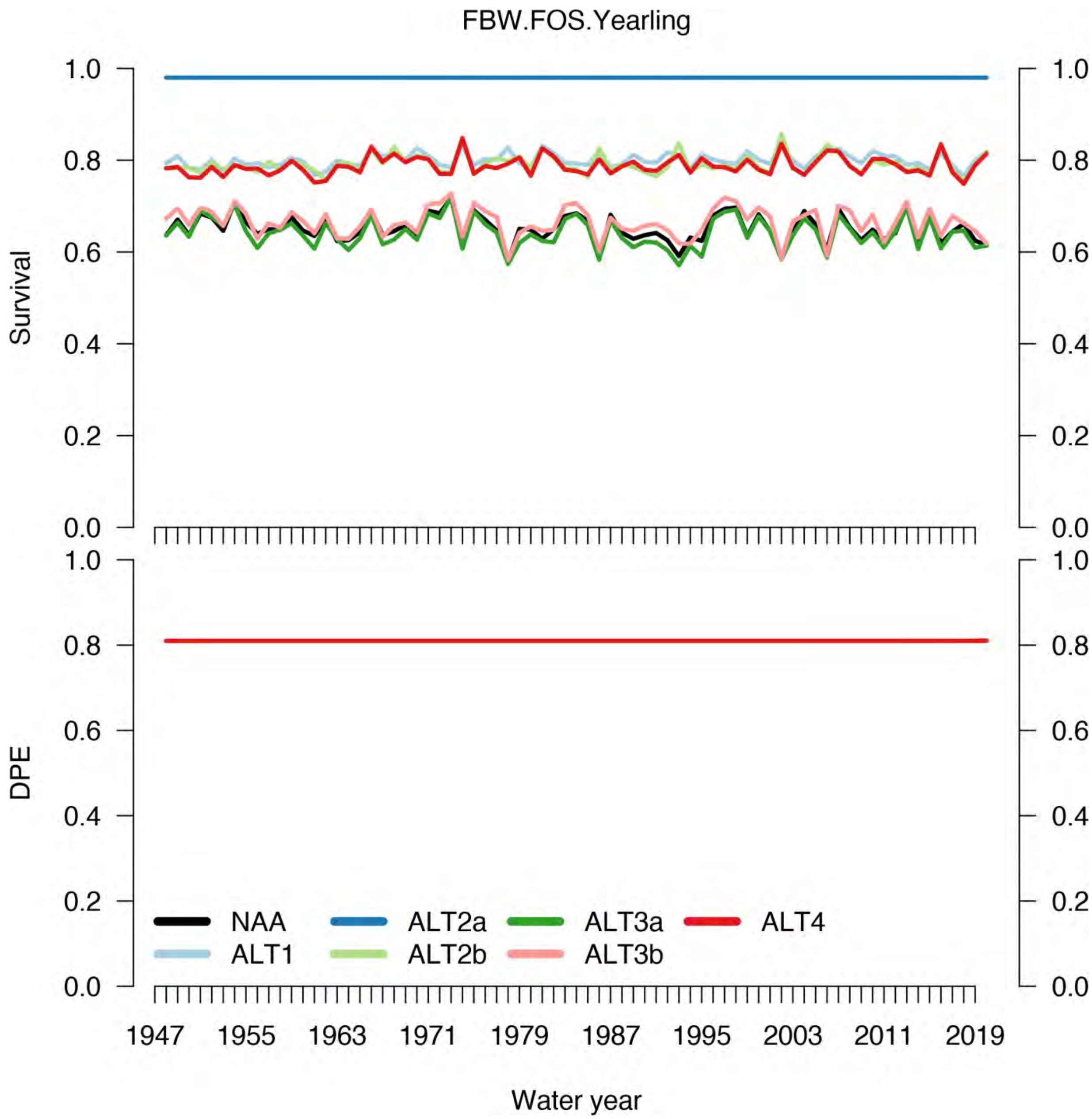


Figure 14.3.6: Foster Dam Chinook salmon yearling Fish Benefit Workbook (FBW). DPE - dam passage efficiency

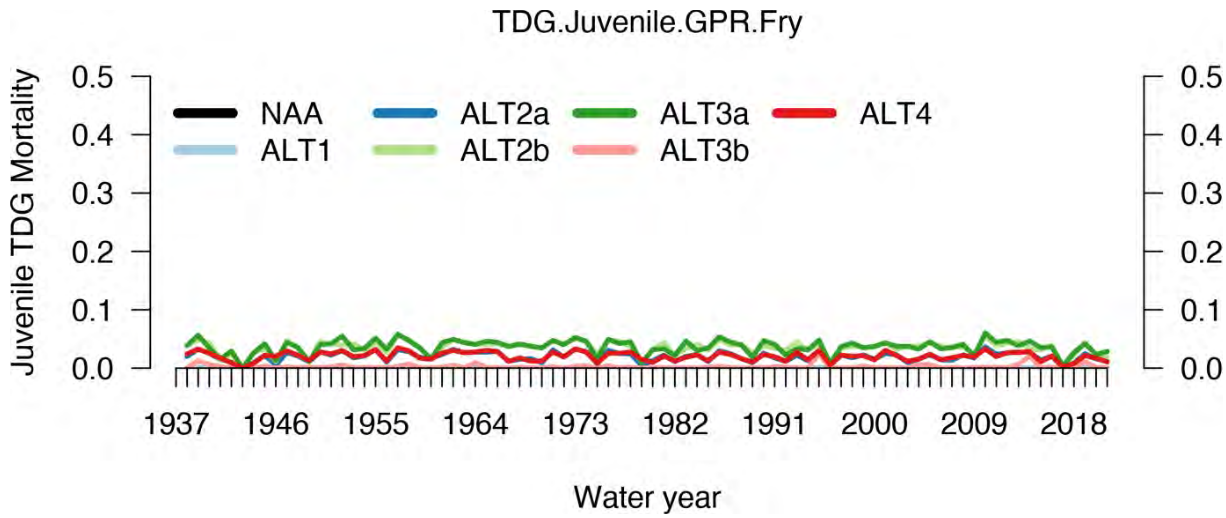


Figure 14.3.7: Green Peter Dam Chinook salmon fry (spring subyearling) total dissolved gas (TDG) mortality.

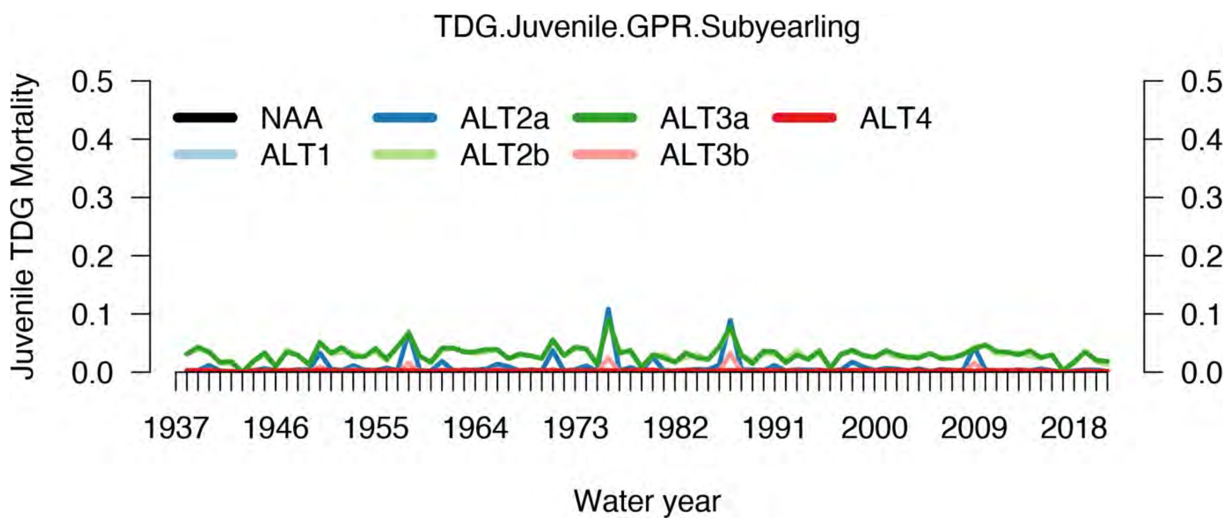


Figure 14.3.8: Green Peter Dam Chinook salmon fall subyearling total dissolved gas (TDG) mortality.

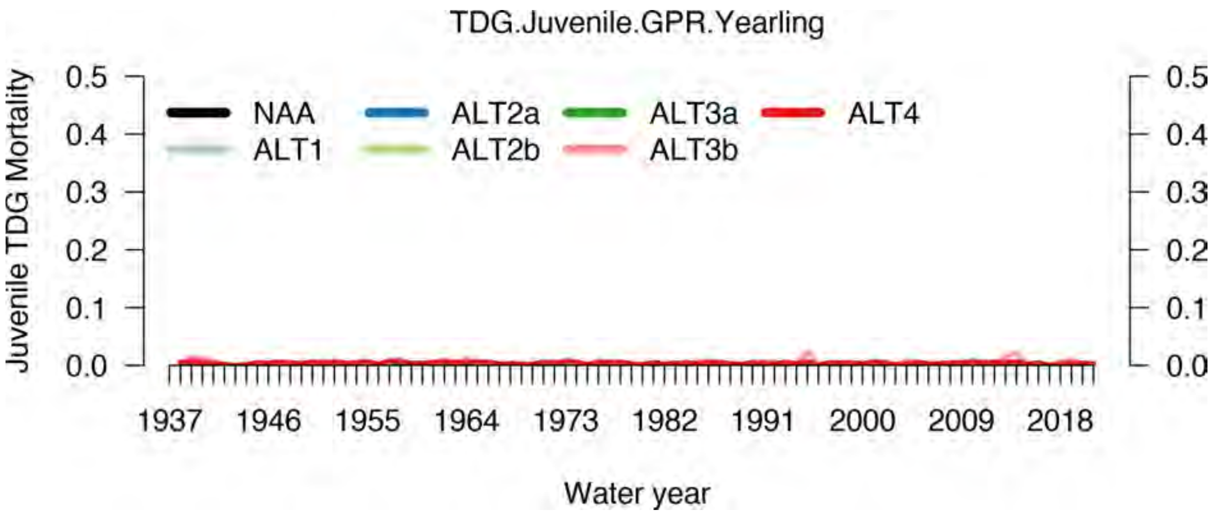


Figure 14.3.9: Green Peter Dam Chinook salmon yearling total dissolved gas (TDG) mortality.

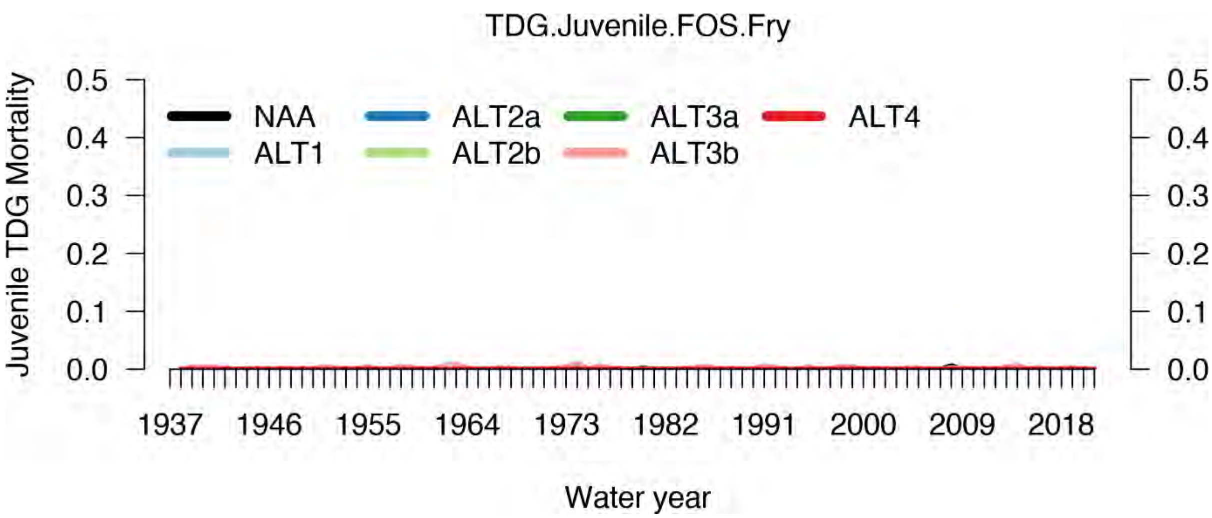


Figure 14.3.10: Foster Dam Chinook salmon fry (spring subyearling) total dissolved gas (TDG) mortality.

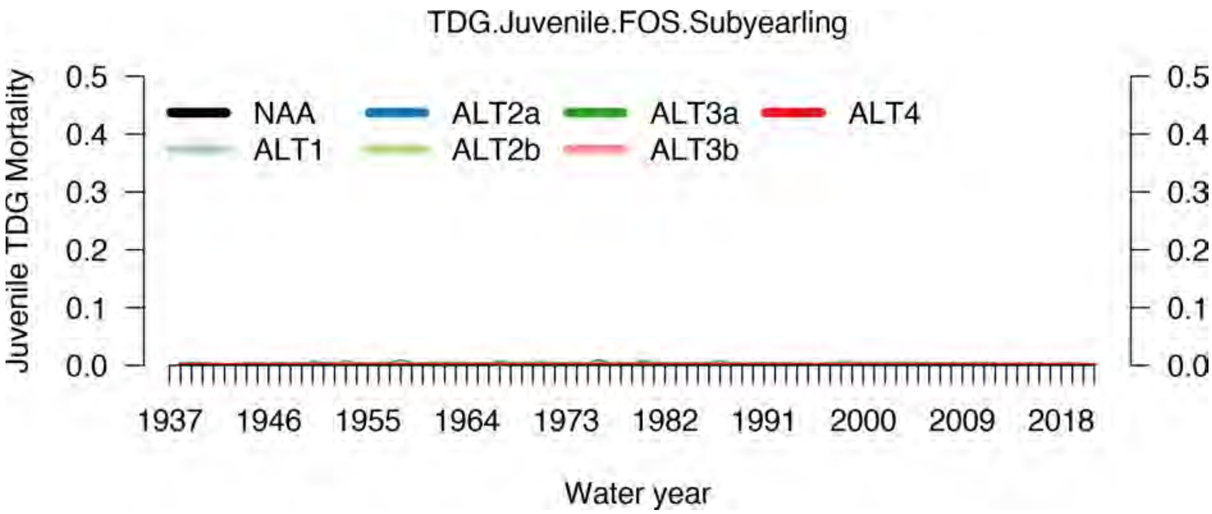


Figure 14.3.11:Foster Dam Chinook salmon fall subyearling total dissolved gas (TDG) mortality.

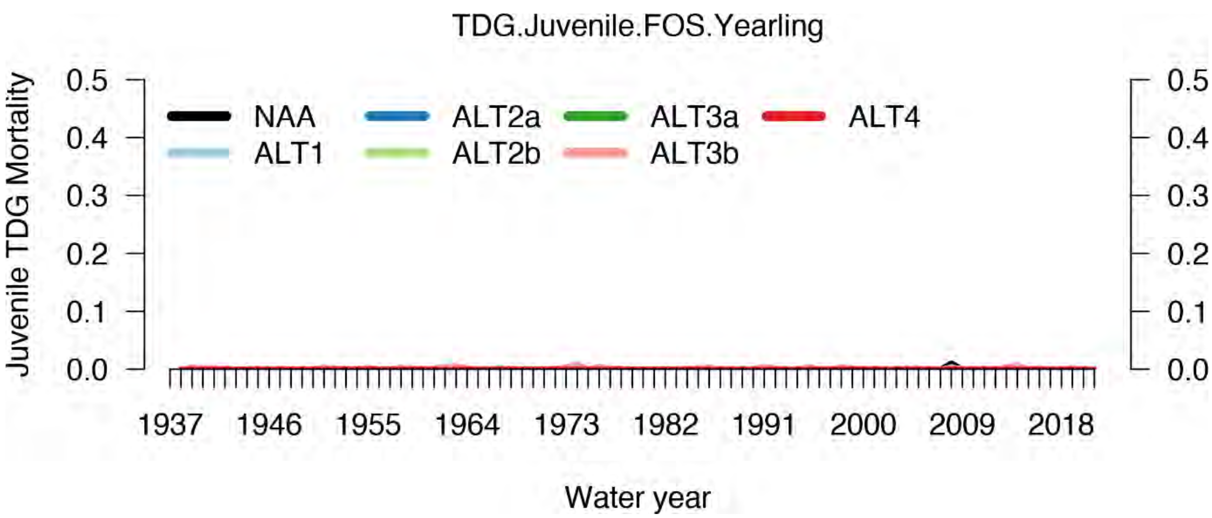


Figure 14.3.12: Foster Dam Chinook salmon yearling total dissolved gas (TDG) mortality.

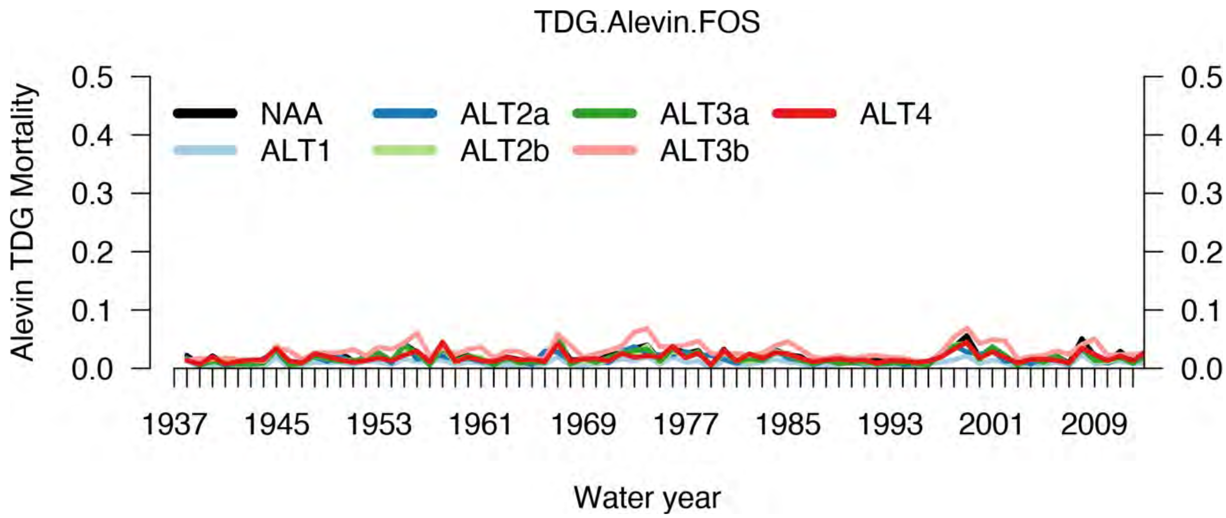


Figure 14.3.13: Foster Dam Chinook salmon alevin total dissolved gas (TDG) mortality

14.4 North Santiam Chinook salmon (GPR, FOS)

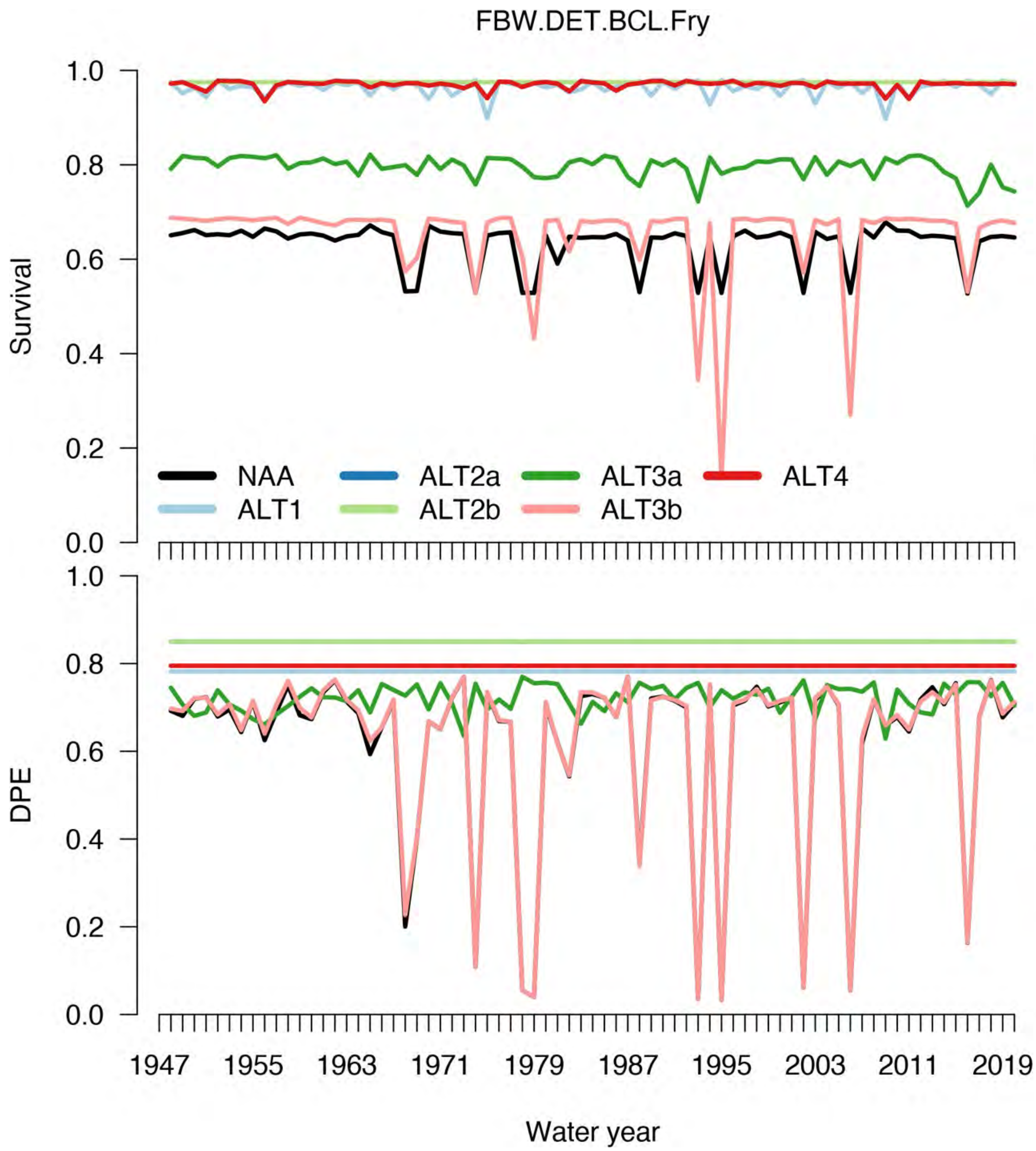


Figure 14.4.1: Detroit Dam and Big Cliff Dam Chinook salmon fry (spring subyearling) Fish Benefit Workbook (FBW). DPE - dam passage efficiency

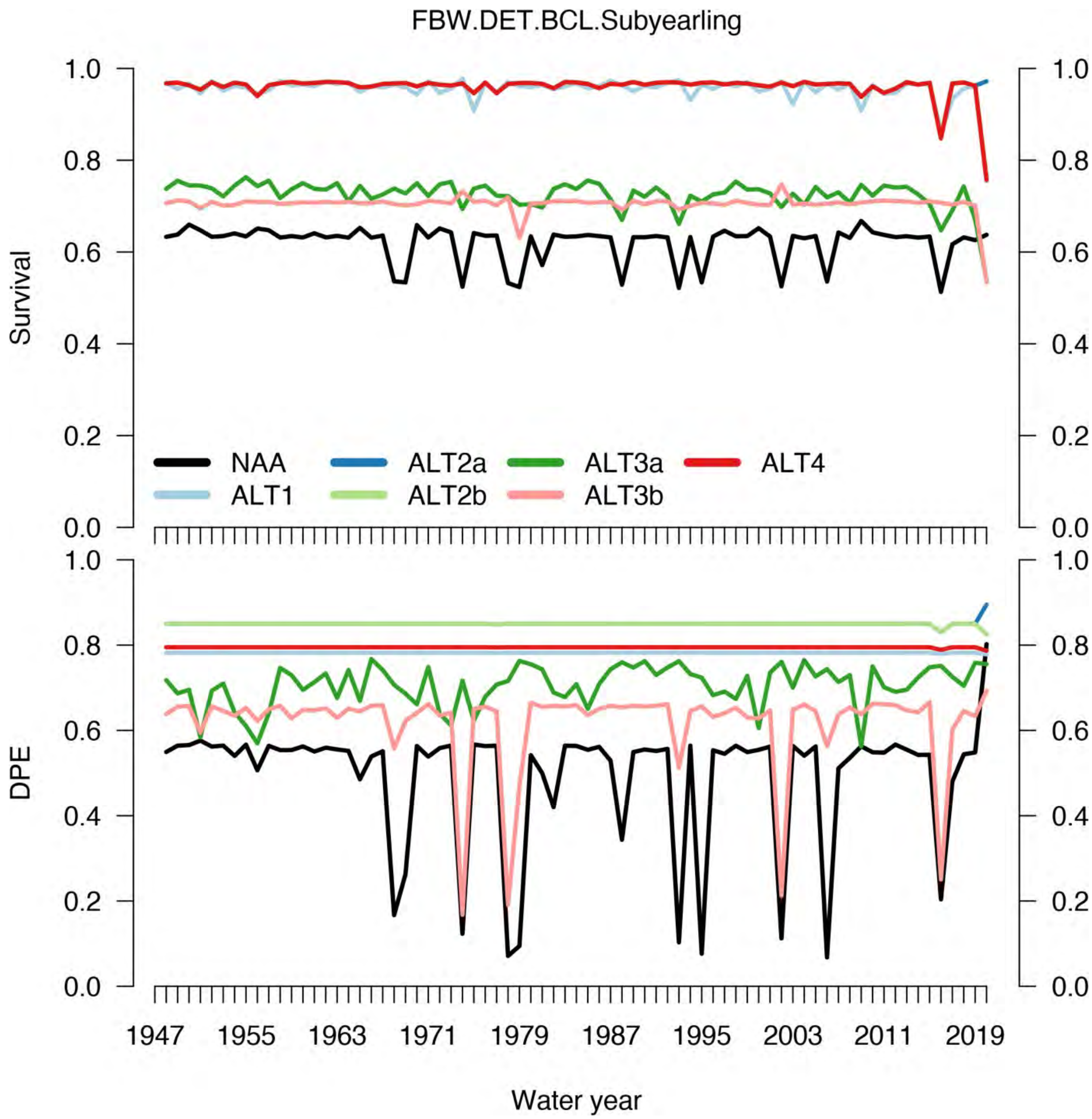


Figure 14.4.2: Detroit Dam and Big Cliff Dam Chinook salmon fall subyearling Fish Benefit Workbook (FBW). DPE - dam passage efficiency

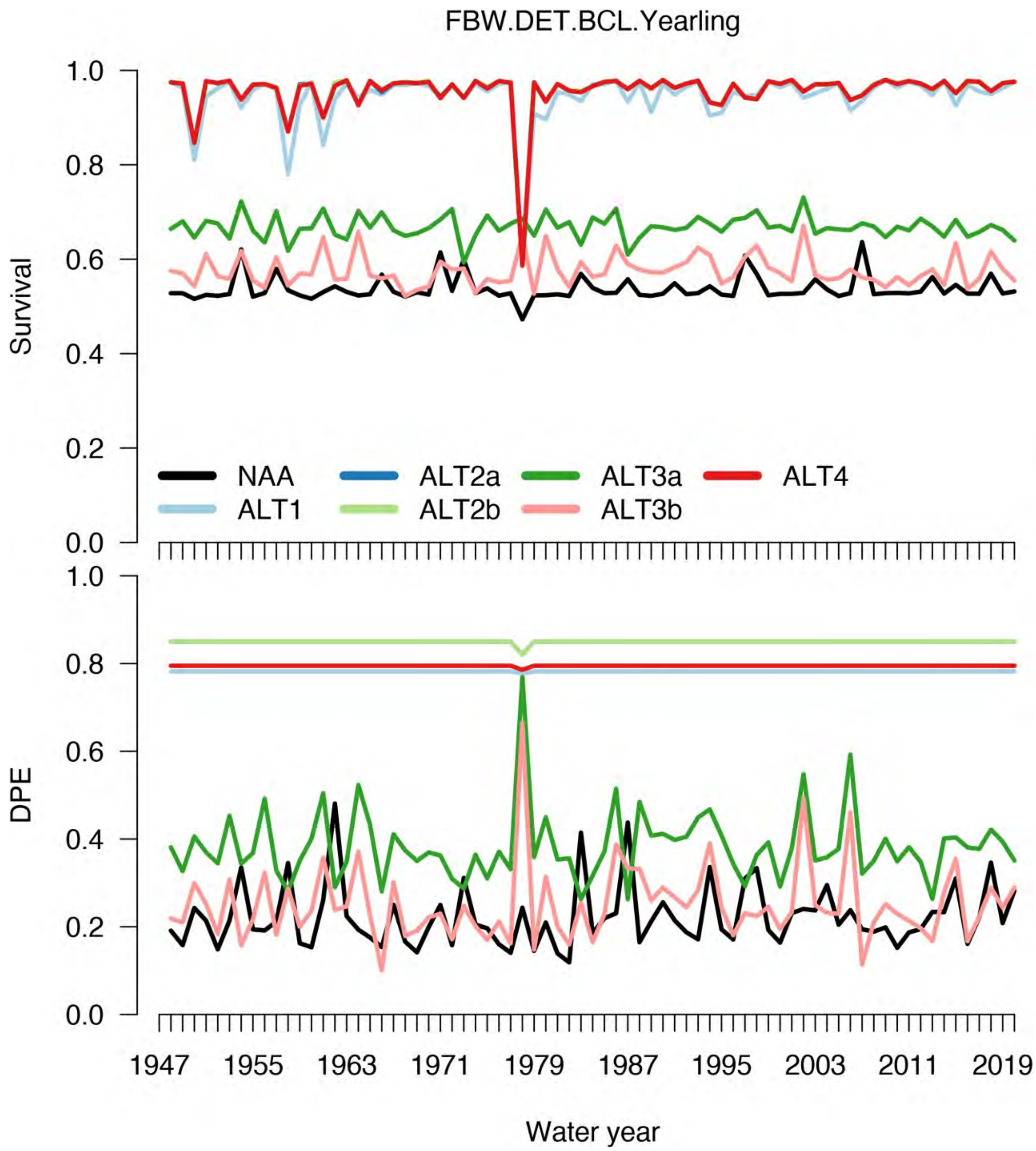


Figure 14.4.3: Detroit Dam and Big Cliff Dam Chinook salmon yearling Fish Benefit Workbook (FBW). DPE - dam passage efficiency

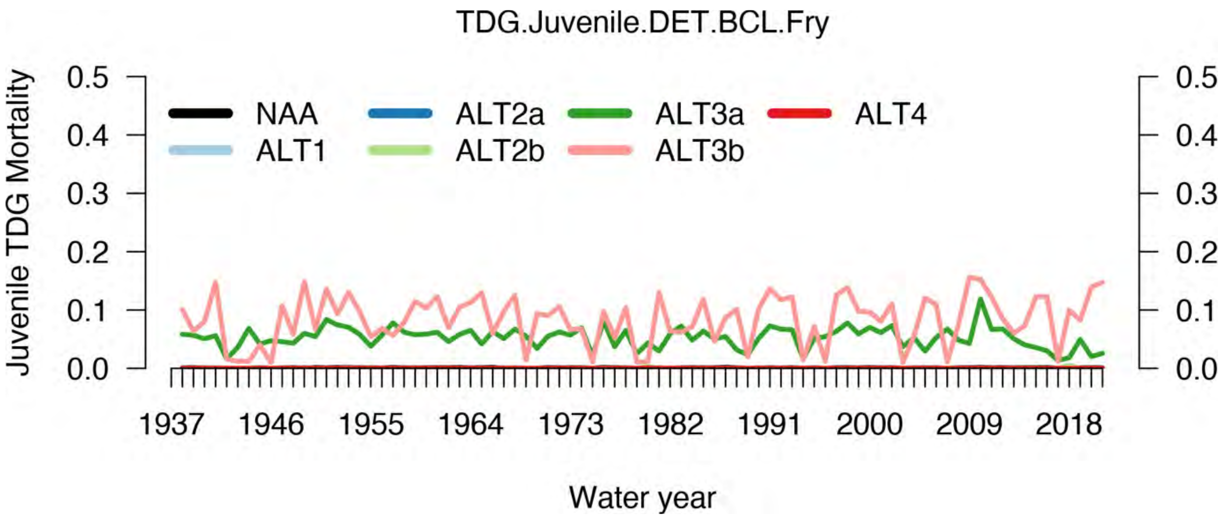


Figure 14.4.4: Detroit Dam and Big Cliff Dam Chinook salmon fry (spring subyearling) total dissolved gas (TDG) mortality.

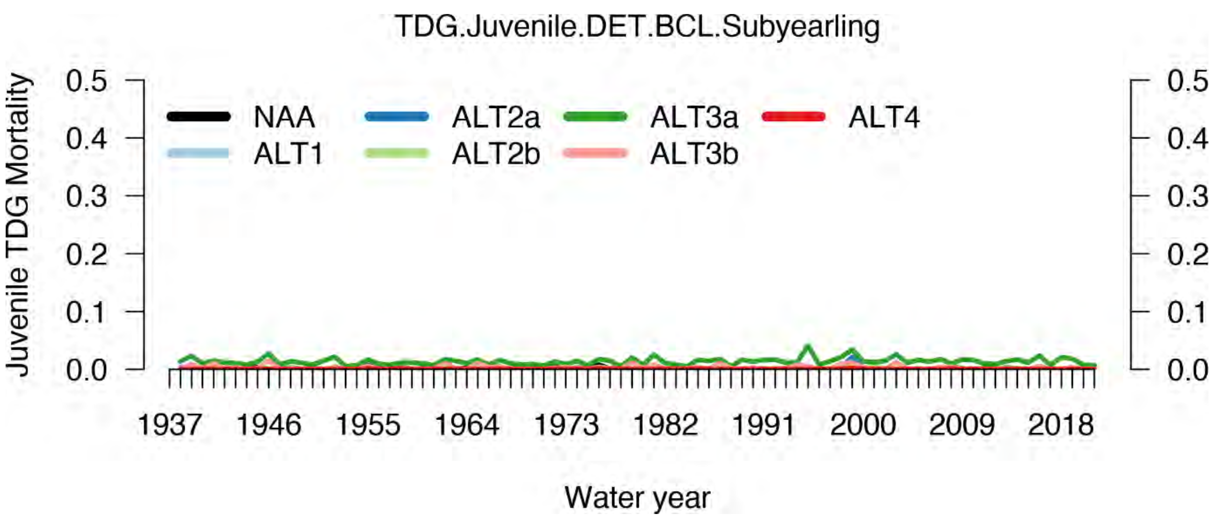


Figure 14.4.5: Detroit Dam and Big Cliff Dam Chinook salmon fall subyearling total dissolved gas (TDG) mortality.

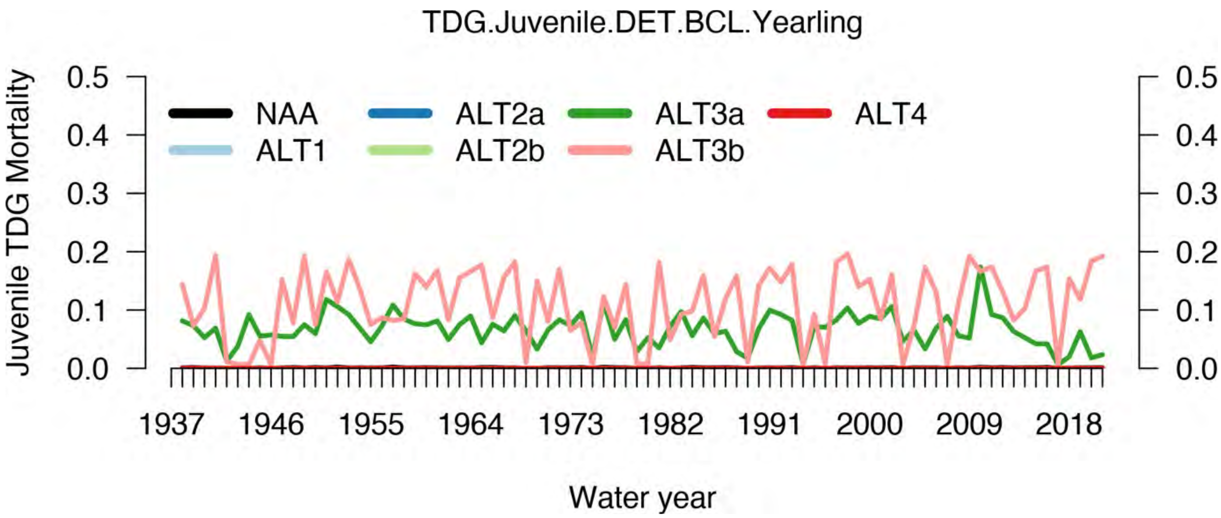


Figure 14.4.6: Detroit Dam and Big Cliff Dam Chinook salmon yearling total dissolved gas (TDG) mortality.

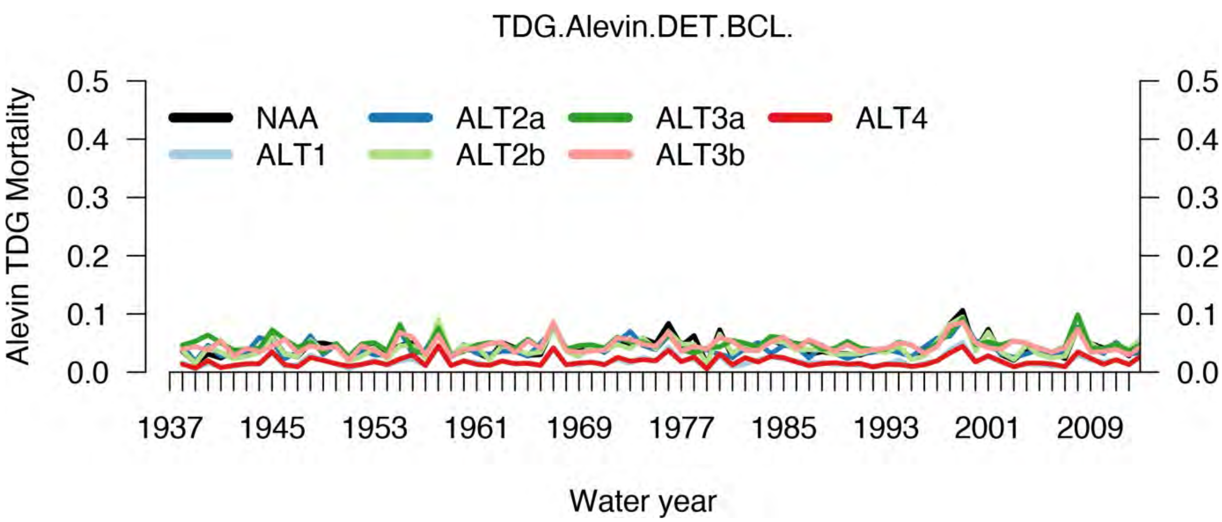


Figure 14.4.7:Detroit Dam and Big Cliff Dam Chinook salmon alevin total dissolved gas (TDG) mortality.

7.12.2 14.5 South Santiam Winter Steelhead

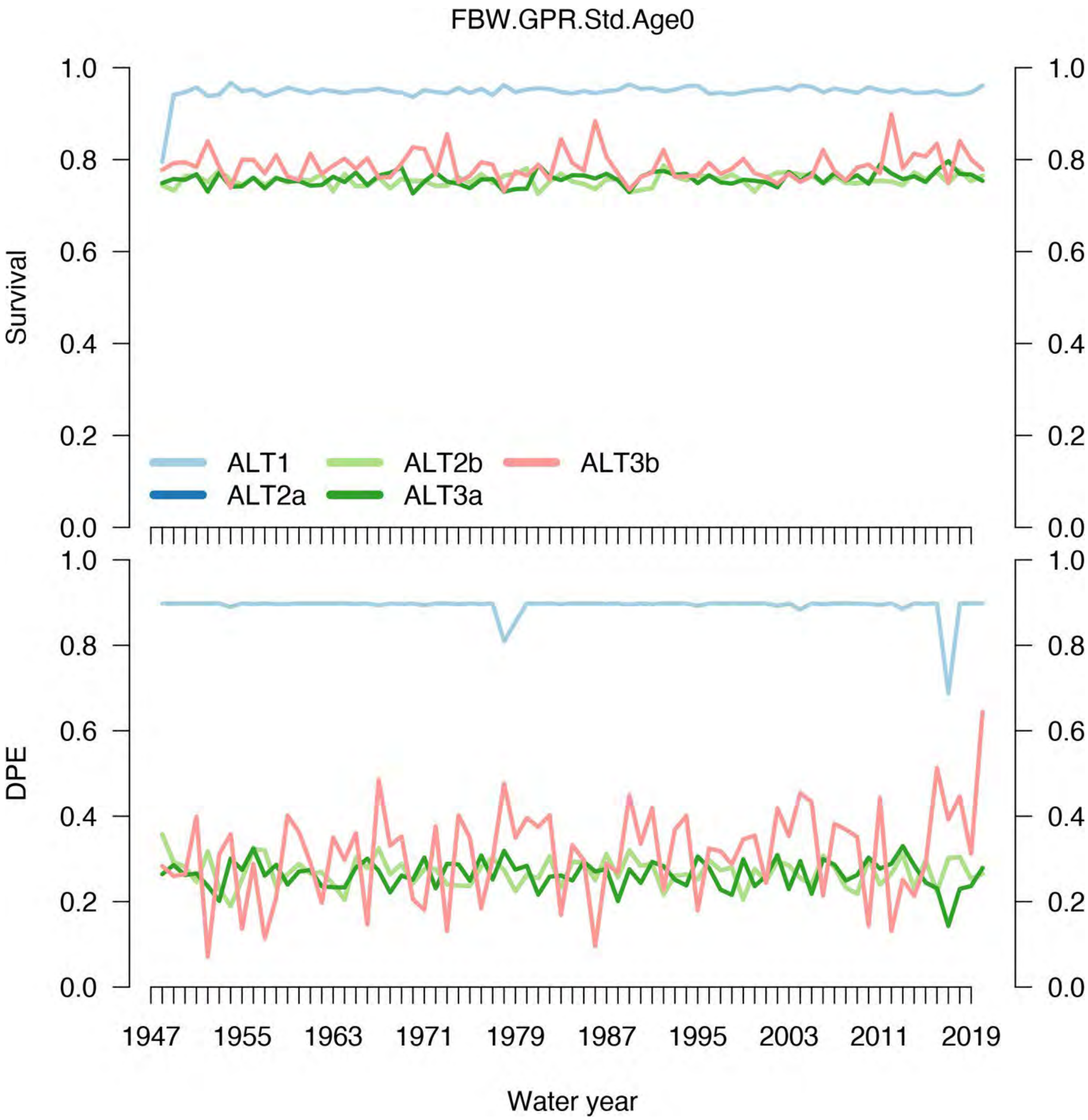
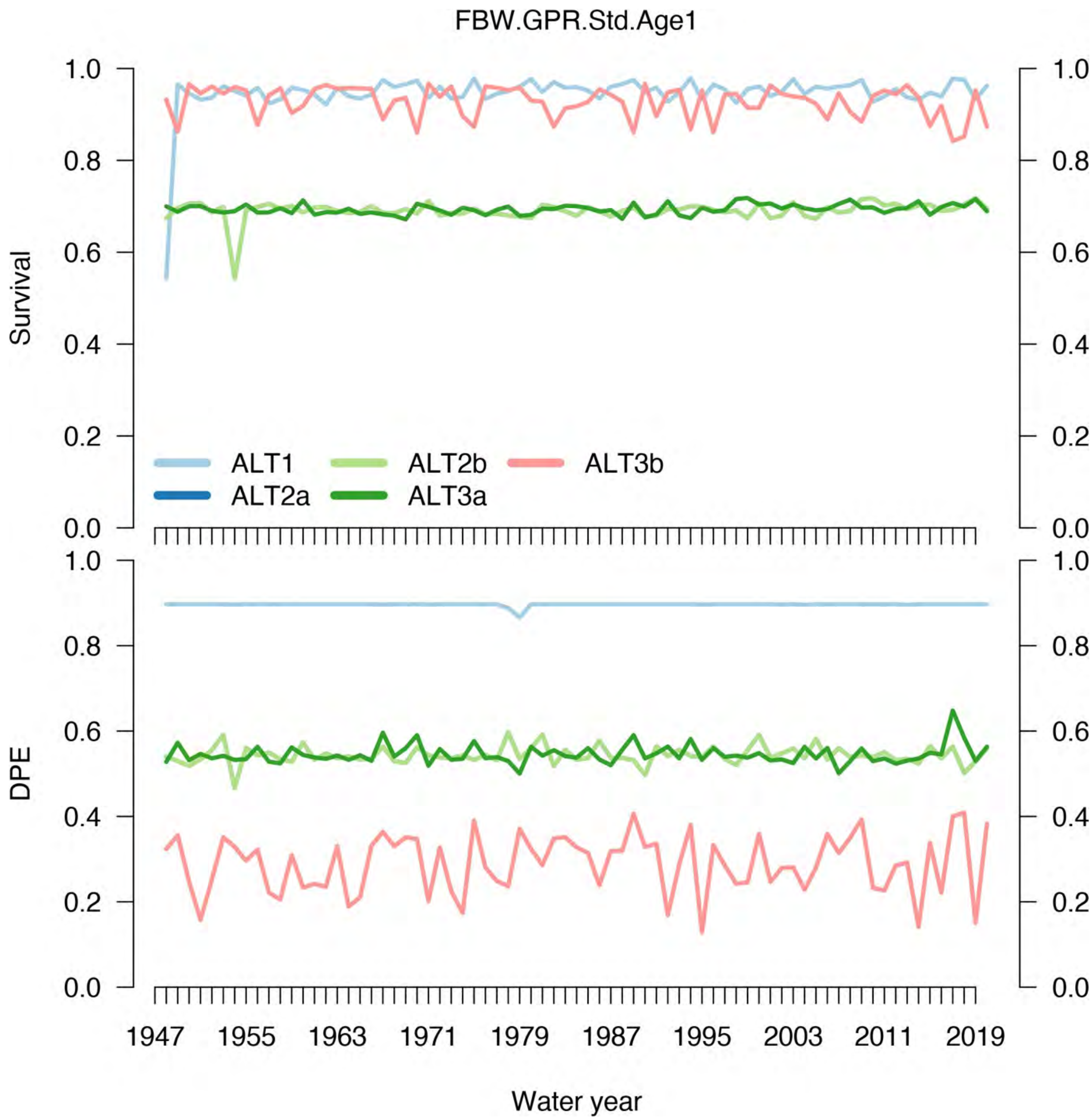
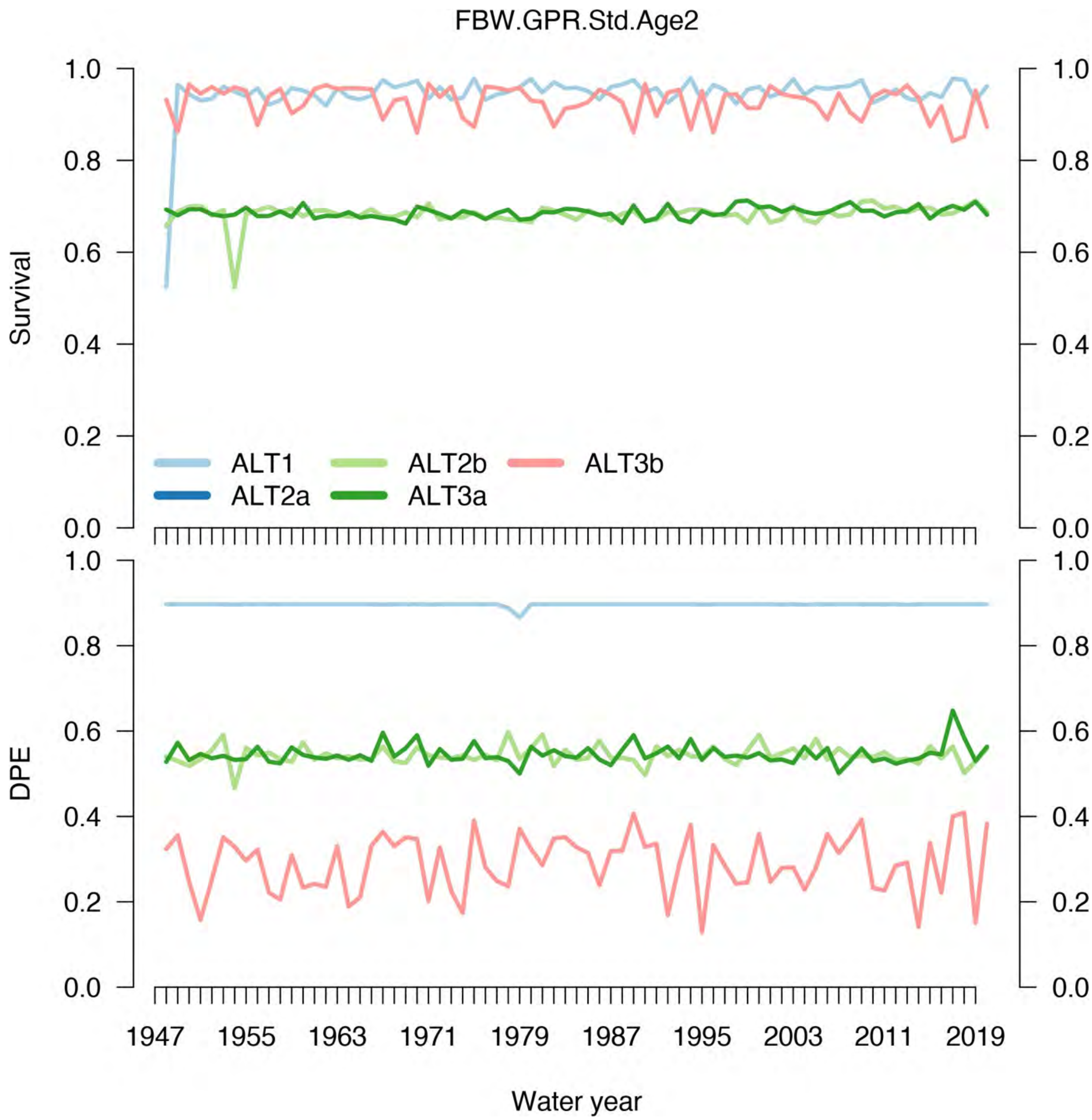


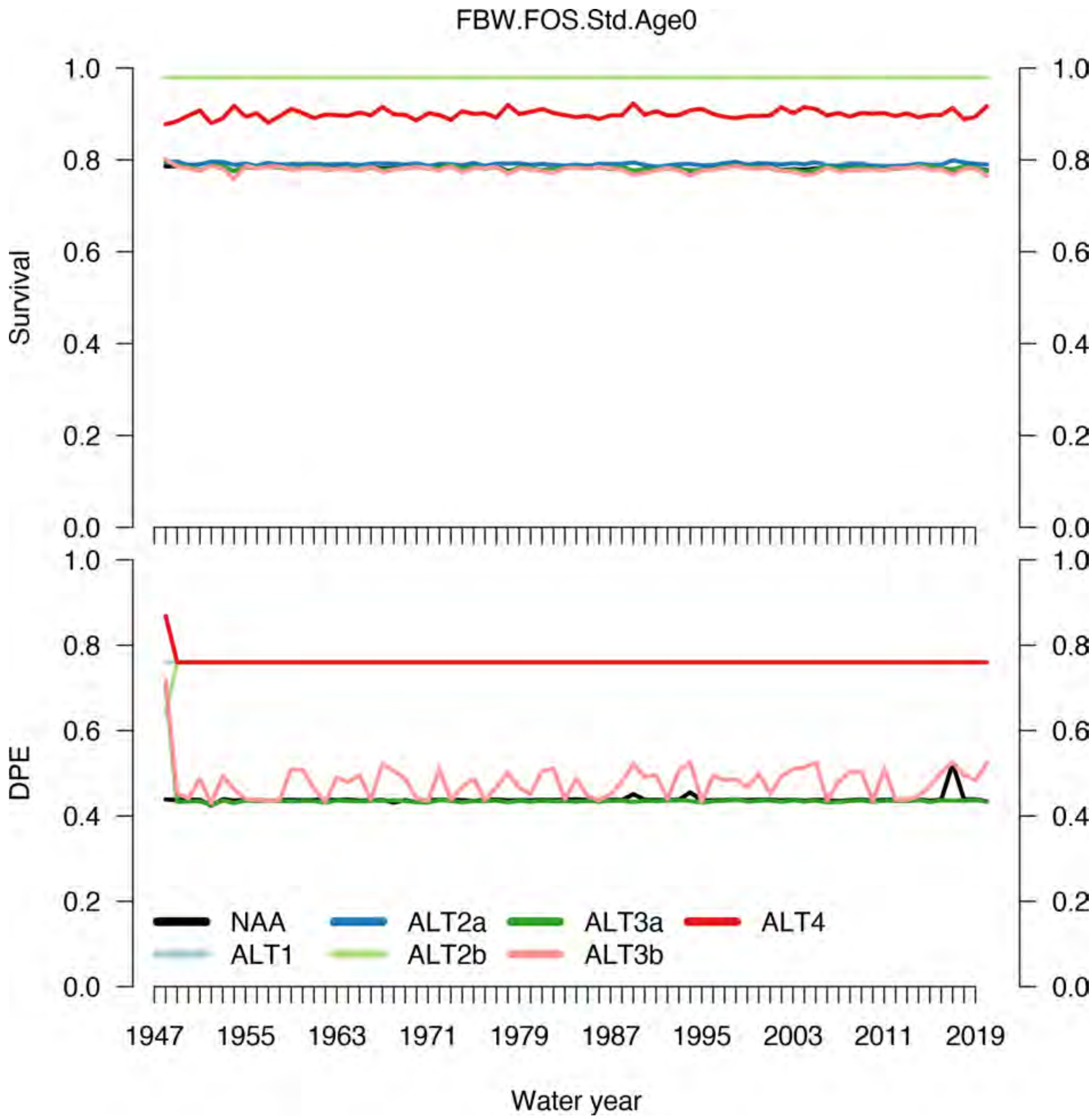
Figure 14.5.1 Green Peter Dam winter steelhead parr (Age 0) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.



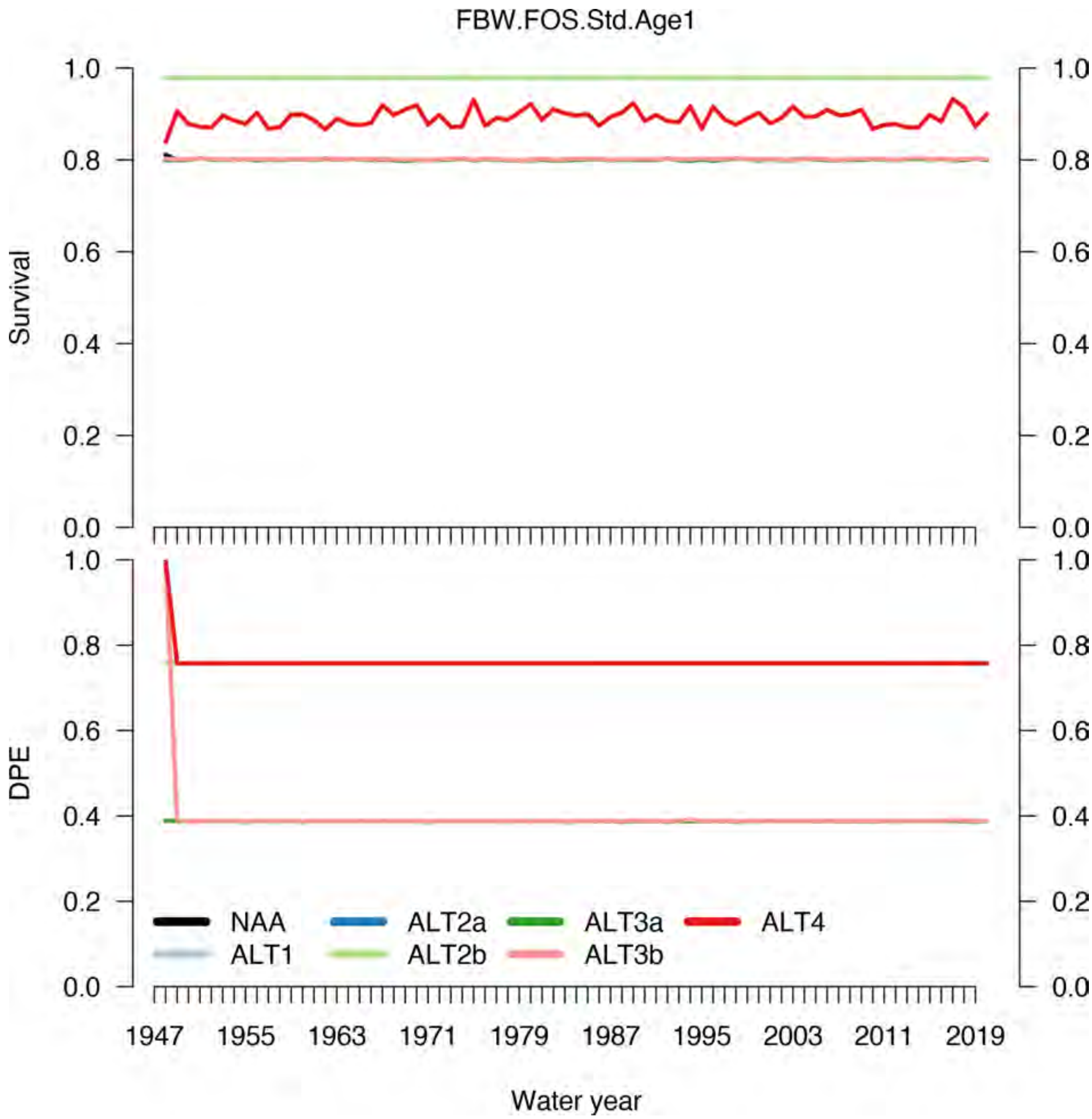
Green Peter Dam winter steelhead yearling (Age 1) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.



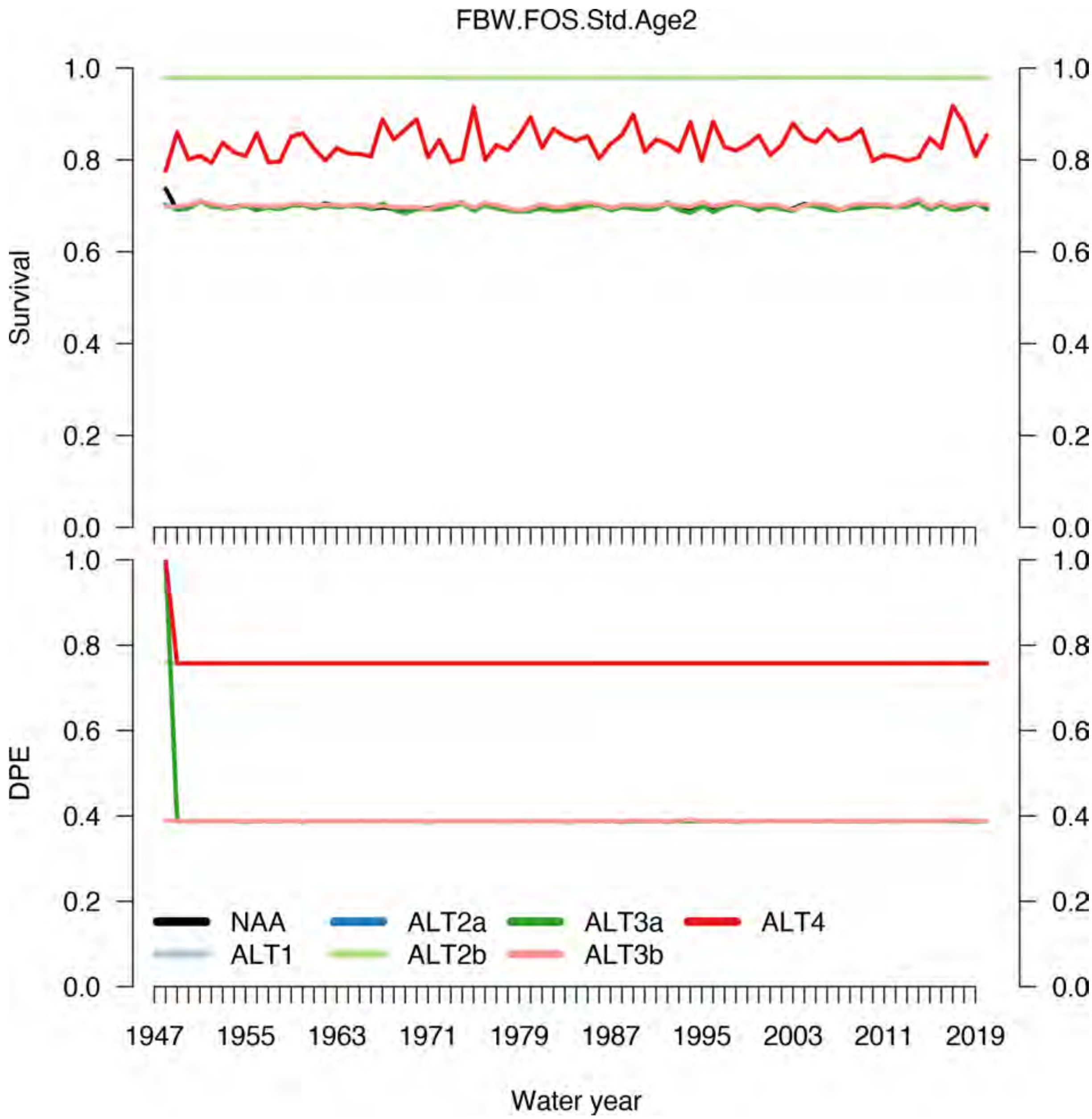
Green Peter Dam winter steelhead smolt (Age 2) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.



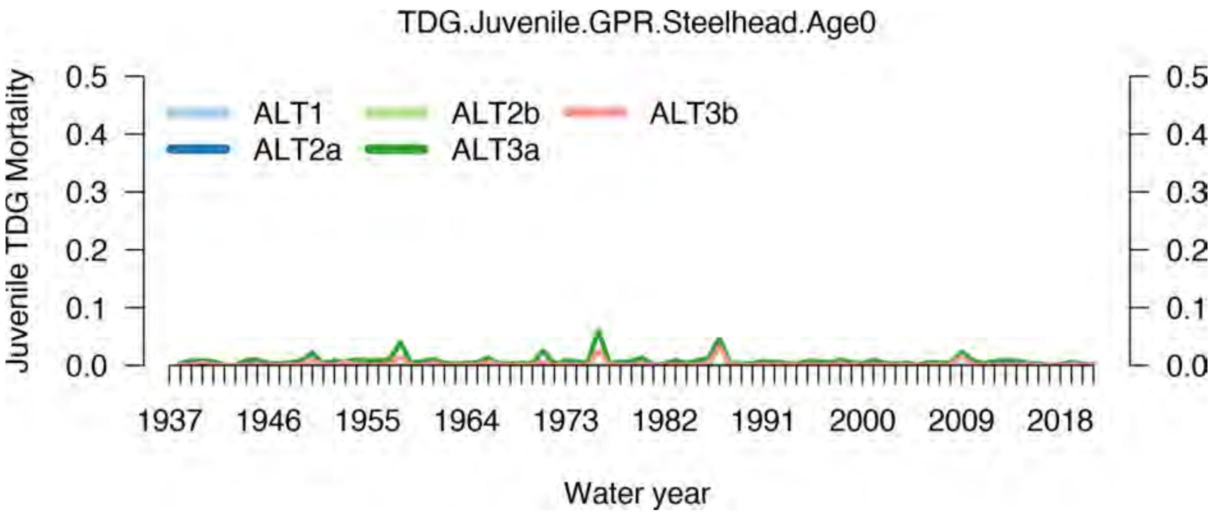
Foster Dam winter steelhead parr (Age 0) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.



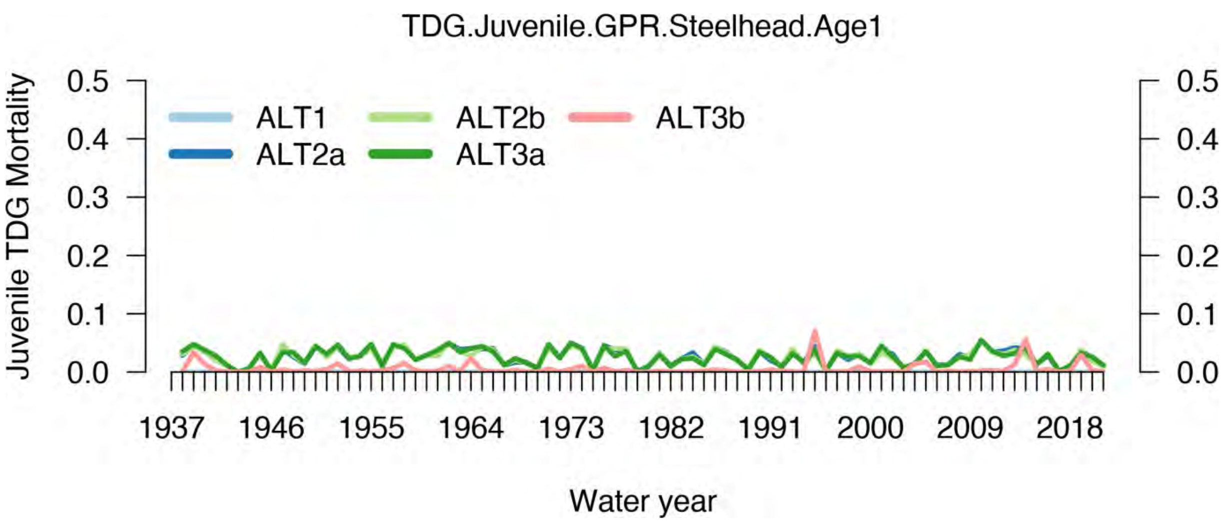
Foster Dam winter steelhead yearling (Age 1) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.



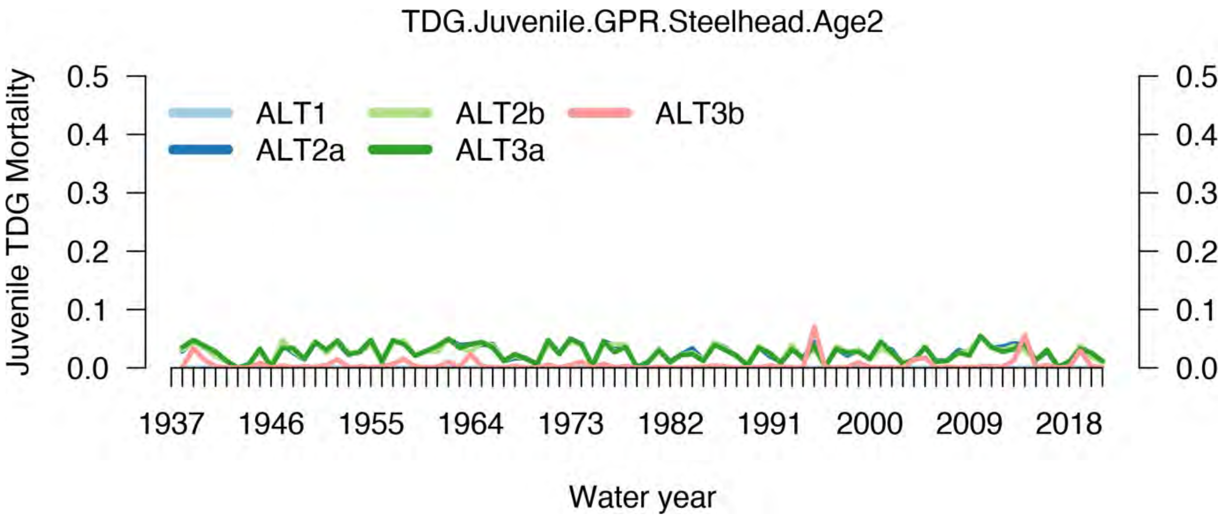
Foster Dam winter steelhead smolt (Age 2) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.



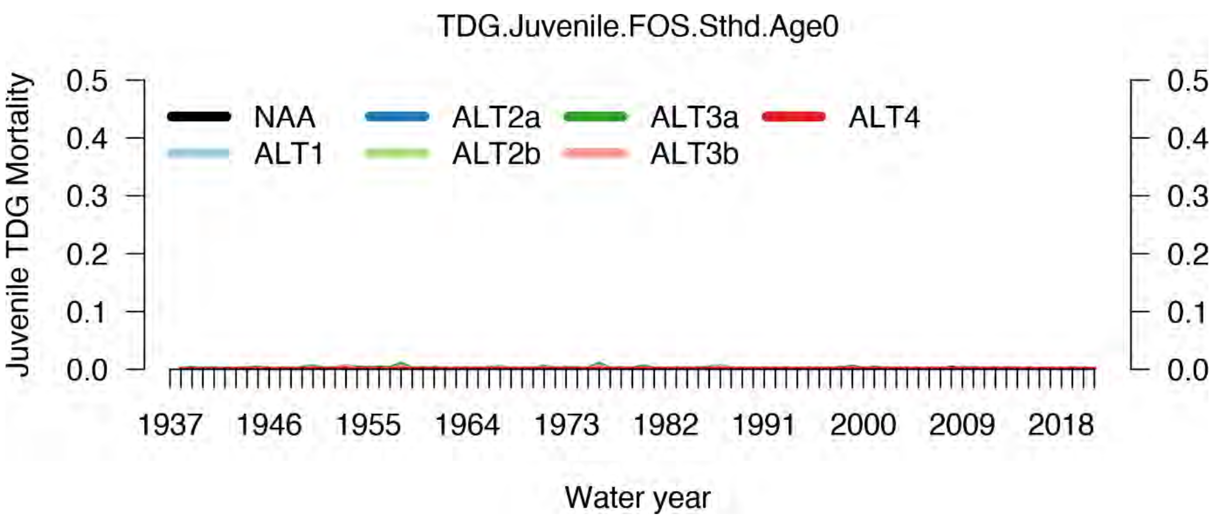
Green Peter Dam winter steelhead parr (Age 0) total dissolved gas (TDG) mortality.



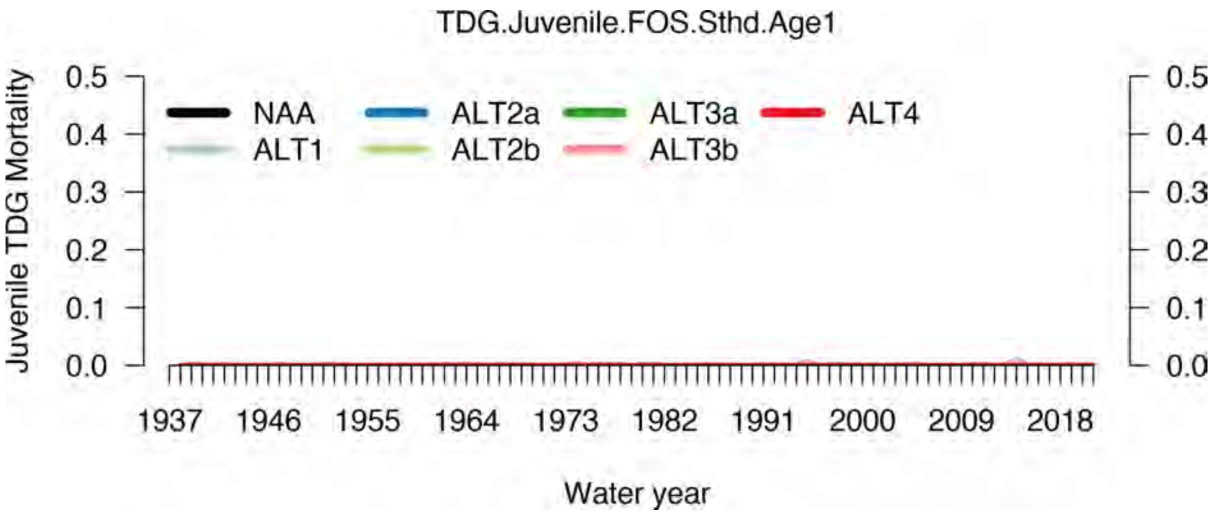
Green Peter Dam winter steelhead yearling (Age 1) total dissolved gas (TDG) mortality



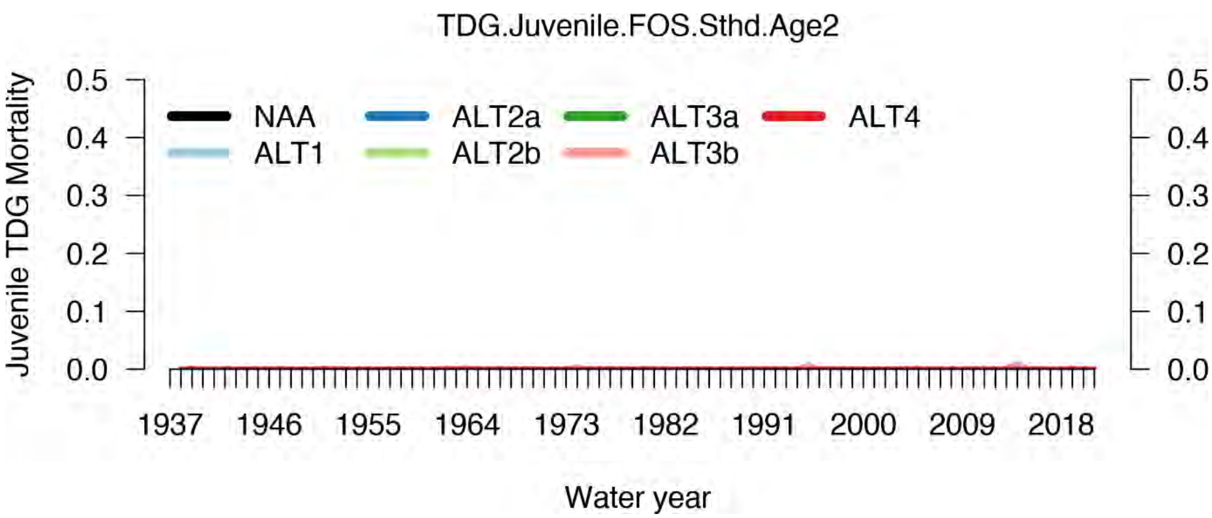
Green Peter Dam winter steelhead smolt (Age 2) total dissolved gas (TDG) mortality.



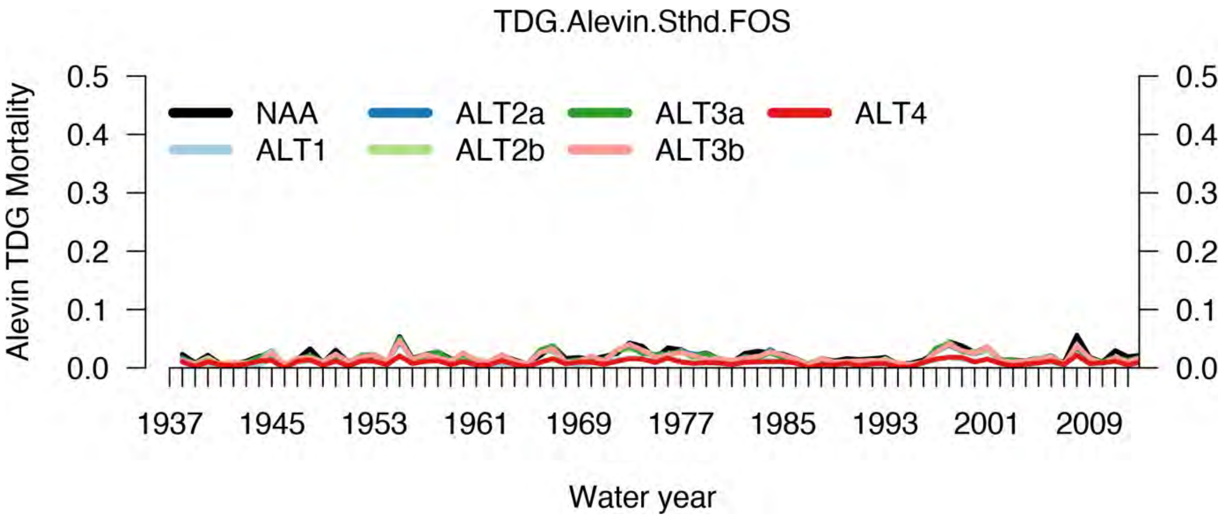
Foster Dam winter steelhead parr (Age 0) total dissolved gas (TDG) mortality.



Foster Dam winter steelhead yearling (Age 1) total dissolved gas (TDG) mortality.



Foster Dam winter steelhead smolt (Age 2) total dissolved gas (TDG) mortality.



Foster Dam winter steelhead alevin total dissolved gas (TDG) mortality.

14.1.10. 14.6 North Santiam Winter Steelhead

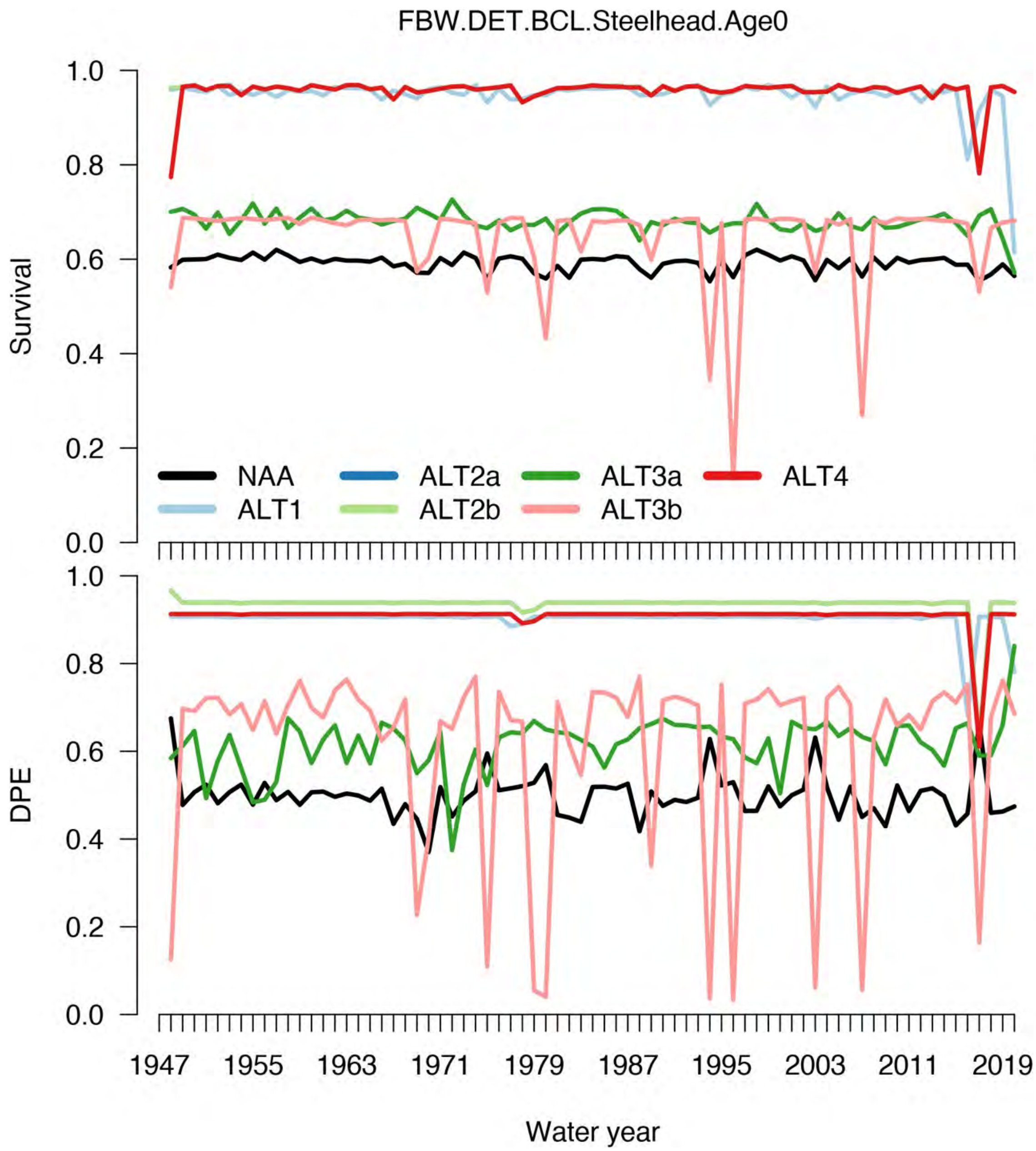


Figure 14.6.1: Detroit Dam and Big Cliff Dam winter steelhead parr (Age 0) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

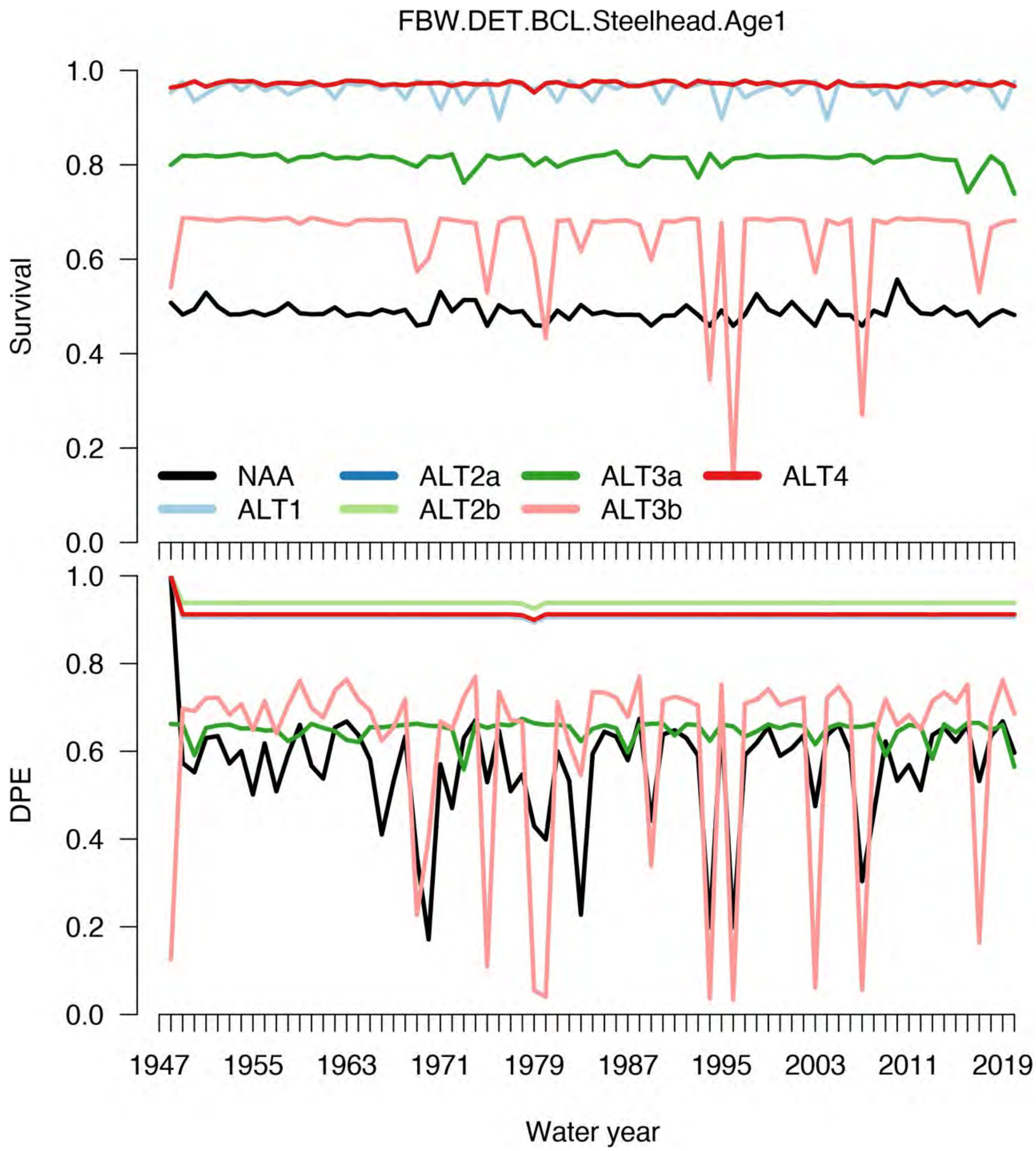


Figure 14.6.2: Detroit Dam and Big Cliff Dam winter steelhead yearling (Age 1) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

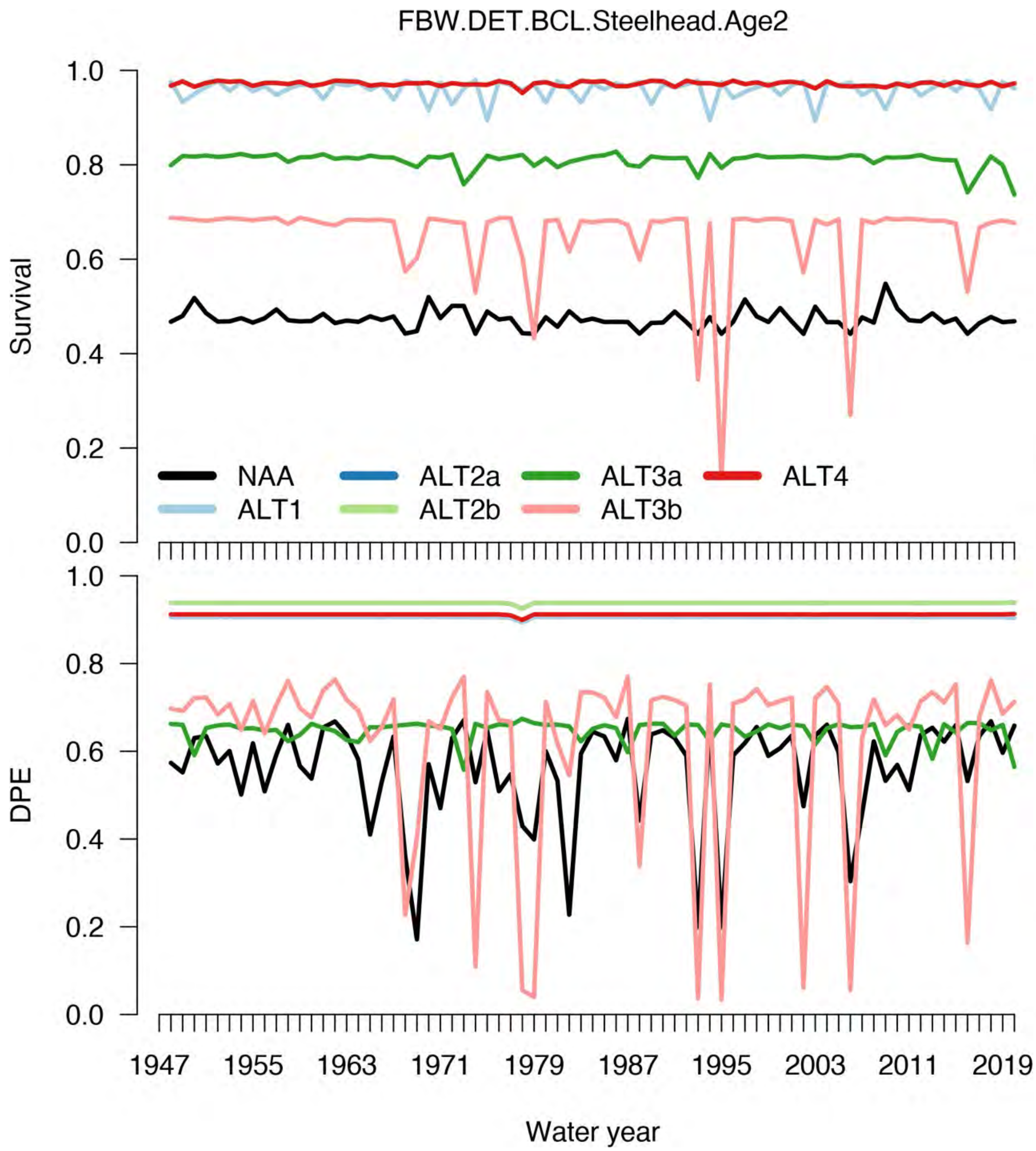


Figure 14.6.3: Detroit Dam and Big Cliff Dam winter steelhead smolt (Age 2) Fish Benefit Workbook (FBW). DPE - dam passage efficiency.

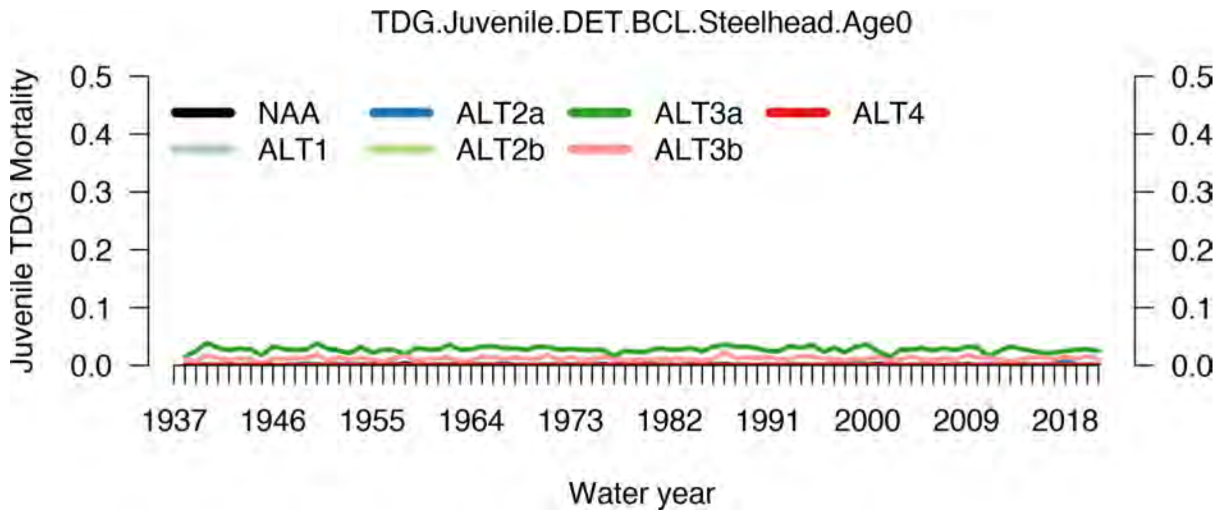


Figure 14.6.4: Detroit Dam and Big Cliff Dam winter steelhead parr (Age 0) total dissolved gas (TDG) mortality.

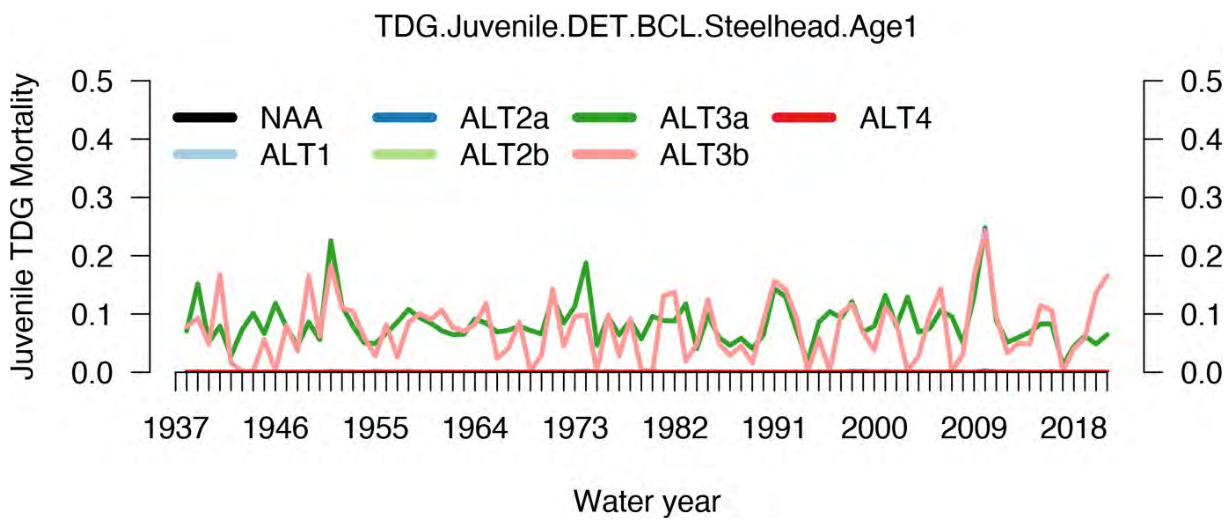


Figure 14.6.5: Detroit Dam and Big Cliff Dam winter steelhead yearling (Age 1) total dissolved gas (TDG) mortality.

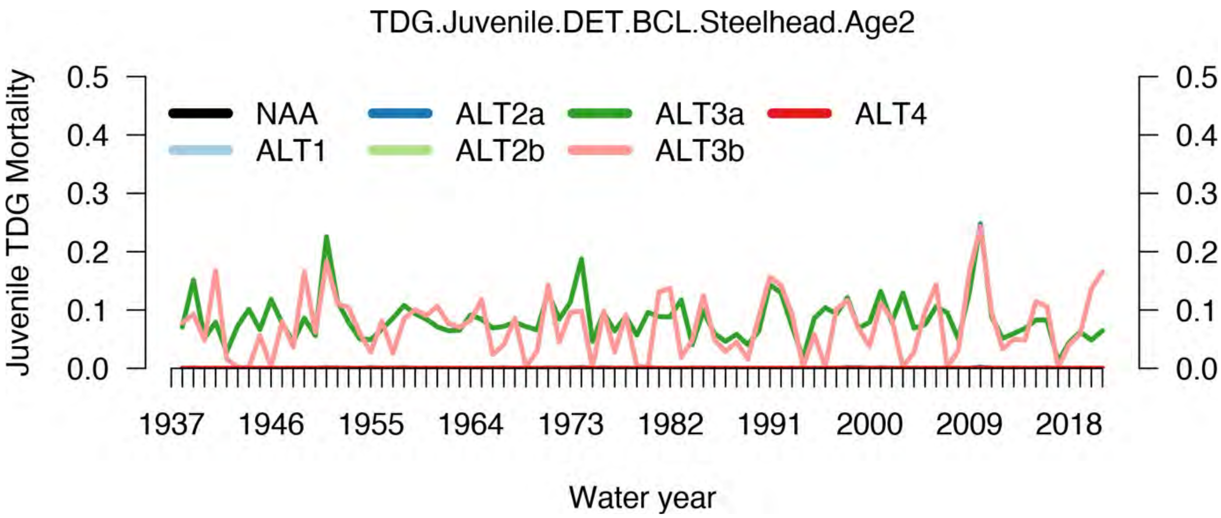


Figure 14.6.6: Detroit Dam and Big Cliff Dam winter steelhead smolt (Age 2) total dissolved gas (TDG) mortality.

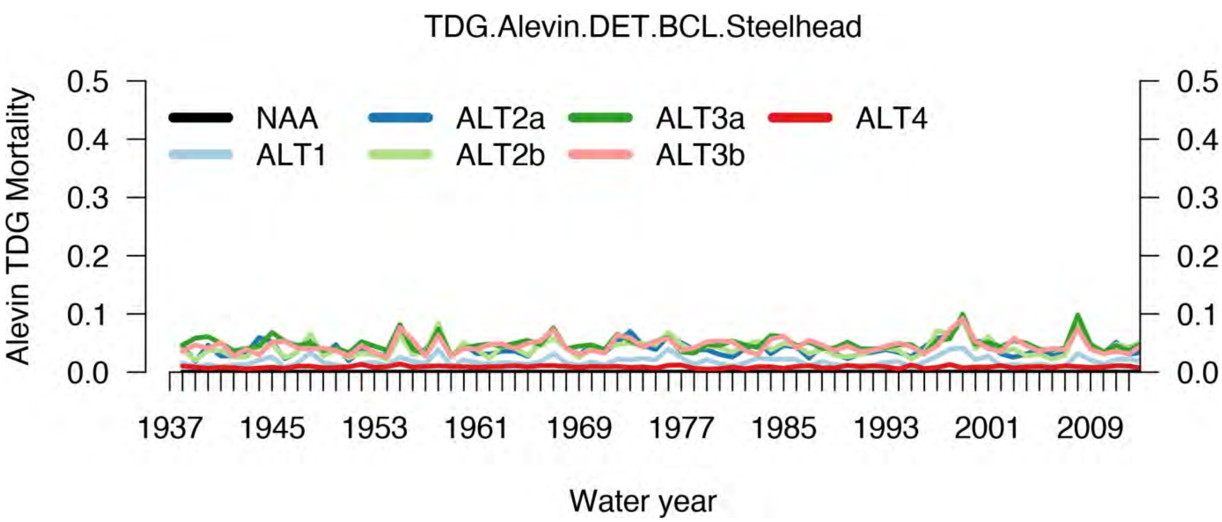


Figure 14.6.7: Detroit Dam and Big Cliff Dam winter steelhead alevin total dissolved gas (TDG) mortality.

CHAPTER 8 - TECHNICAL REPORT 2022-01

Integrated Passage Assessment (IPA) modelling to evaluate alternative US Environmental Impact Study measures for wild spring Chinook salmon (*Oncorhynchus tshawytscha*) and winter steelhead (*O. mykiss*) populations in the Upper Willamette River basin

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The authors currently live and work on unceded territories of the Coast Salish peoples– Sḵwx̱wú7mesh (Squamish), Stó:lō and Səl̓ílwətaʔ/Selilwitulh (Tsleil-Waututh) and xʷməθkʷəy̓əm (Musqueam) Nations and Ts’elxwéyeqw Tribe, and on the traditional territories of the Huron-Wendat, the Seneca, and the Mississaugas of the Credit.

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Introduction

General Introduction

This report describes the Integrated Passage Assessment (IPA) life cycle models (LCM) developed by University of British Columbia (UBC) researchers and the performance metrics computed for Environmental Impact Study (EIS) alternatives for spring run Chinook salmon (Oncorhynchus tshawytscha) and winter steelhead (O. mykiss) in the Upper Willamette sub-basins (Figure 1.1.1). The life cycle models documented in this report were intended to rank EIS alternatives, taking into account how conditions created by each alternative could impact juvenile dam passage efficiency (DPE) and dam passage survival (DPS) for each species, juvenile life stage, and dam in the Upper Willamette being assessed.

While it could be possible to evaluate the EIS alternatives using the Corps Fish Benefits Workbook software (FBW) without a life cycle model, FBW computes only the dam passage efficiency (DPE) and dam passage survival rates (DPS) for juvenile life stages of Chinook and steelhead for specific Upper Willamette dams under conditions specified in each EIS alternative. By applying some weighting to the sets of results for different juvenile life stages of the two species at each dam, EIS alternatives could be ranked but the evaluation criteria would not be adequately addressed. This is because it is of interest to rank the EIS alternatives according to how well they could meet long-term conservation objectives that can only be evaluated using LCMs. Without going into detail here (see section 1.3 on Performance Metrics), it was agreed that the EIS alternatives would be ranked according to how well each 1) prevents population extinction, 2) re-establishes abundant spawning populations above the dams, and 3) contributes to specific measures of population productivity that cannot be directly addressed using FBW outputs. And while rankings of EIS alternatives based on LCM outputs are influenced by FBW outputs, the FBW outputs by themselves would not enable evaluation of whether the EIS alternatives could satisfactorily meet any one of the above three conservation objectives. Moreover, the EIS alternatives may differentially affect other key aspects of salmon and steelhead population dynamics that are not included in FBW. For example, FBW does not account for the average flow and temperature effects of EIS alternatives on pre-spawn mortality in Chinook salmon, nor below-dam growth and survival rates of juvenile Chinook and steelhead which may contribute significantly to the EIS performance metric outcomes. In contrast to FBW, the LCMs are specifically formulated to evaluate whether objectives could be met by each of the EIS alternatives using the best available information on all key life stages while accounting for definable uncertainties in LCM components.

The Chinook and winter steelhead LCMs conform more closely to a traditional stock assessment modelling approach than the currently available approach. This is mainly due to the interest in ensuring that the LCMs applied to evaluate EIS alternatives were fitted to available time series of historical data such that the LCMs could conform to a basic level of empirical credibility, as is commonly adhered to in fisheries stock assessments. The currently available life cycle modelling approach for these fish populations has been developed and applied to these same fish populations in the Upper Willamette by the National Marine Fisheries Service (NMFS). The existing models, i.e., NMFS LCMs, are highly complex and have been under development for more than a decade (see Zabel et al. 2015). The models include detailed representations of juvenile life stages in freshwater, juvenile and sub-adult stages at sea and adult stages after returning to the river and each application pertains to a population above and below the dams in each sub-basin. The NMFS LCMs incorporate data from numerous sources and a quasi-Bayesian algorithm is applied to select and give weight to parameter combinations that align most closely with available time series of adult counts. However, the NMFS LCM departs from conventional stock assessment and fisheries model fitting approaches (see Hilborn and Walters 1992; Quinn and Deriso 1999; and more recently Edwards and Dankel 2016). This is because its quasi-Bayesian algorithm does not include formalized model fitting using a function minimizer. It can thus result in simulated distributions of population abundance that can deviate quite substantially from available time series data for example on adult counts for the fish populations of interest. In addition, it appears that results from several PIT tag studies in the past few decades in Upper Willamette sub-basins have not been incorporated in the NMFS LCMs, despite their containing information on freshwater and smolt-adult return rates.

The IPA LCMs were developed and applied to compute an agreed set of policy performance metrics (relating to, e.g., the risk of quasi extinction, stock productivity, and long-term average abundance of spawners; see Section 1.3) for a set of four structural and operational measures for Corps operated dams in the Upper Willamette that were specified in a US Environmental Impact Study. A general life cycle model for spring Chinook salmon was developed and then fitted and applied separately for the above dam populations in the Middle Fork, McKenzie, South Santiam and North Santiam sub-basins (see Section 2). A general life cycle model for winter steelhead was developed and then fitted and applied to the populations that spawn above the dams in the North Santiam and South Santiam sub-basins (see Section 3). For the projections, records of historic year flow and river water temperatures were bootstrapped to obtain plausible sets of input parameters for the IPA LCMs under each EIS alternative evaluated. To account for uncertainty in input parameter values, such as values for freshwater survival rates for juveniles, marine survival rates, and parameter for pre-spawn mortality rates were drawn from prior distributions with central tendency and prior modes based on values estimated from model fitting, PIT tag analysis or literature-based estimates.

The IPA LCMs were formulated specifically for the population components that spawn above the dams because it is of interest to evaluate the potential population dynamics responses of above dam population components to specific dam passage measures. The IPA LCMs include life cycle stages of juveniles above and below the dams, and juvenile and sub-adult stages at sea and adults below and above the dams. The IPA LCM components were formulated using information and data available from numerous studies conducted in the Upper Willamette on the fish populations of interest. For example, results from statistical analyses of PIT studies of the spring Chinook salmon and winter steelhead populations in the Upper Willamette carried out by the authors of this report (see Appendix C) and coded wire tag studies of spring Chinook salmon from hatchery production in the Upper Willamette (CTC 2021) were used to identify plausible ranges of values for IPA LCM model parameters. The IPA LCMs also used as inputs dam passage efficiencies (DPE) and dam passage survival rates (DPS) provided by the Corps' Fish Benefits Workbook software (FBW). The IPA LCMs were fitted to available time series of adult counts (both species) and age composition of spawners (Chinook salmon only; see Sections 2.5 and 3.3). Due to the large number of input parameters in the Chinook salmon IPA LCM, for example, only the early marine survival rate parameters, fraction maturing at age and annual deviates in marine survival rate (each constrained by Bayesian prior distributions) were freed up for model fitting to the adult count and spawner age composition data (see section 2.5 for details). The model fitting ensured that with a relatively small number of the model parameters freed up IPA LCMs fitted the historical time series of adult counts and spawner age composition records in adherence with commonly held standards of fisheries stock assessment.

The life cycle model and its EIS results for spring Chinook salmon are described in the first part of this report. The life cycle model and its EIS results for winter steelhead are described in the second part of this report. Details on life cycle model equations are provided in report Appendices.

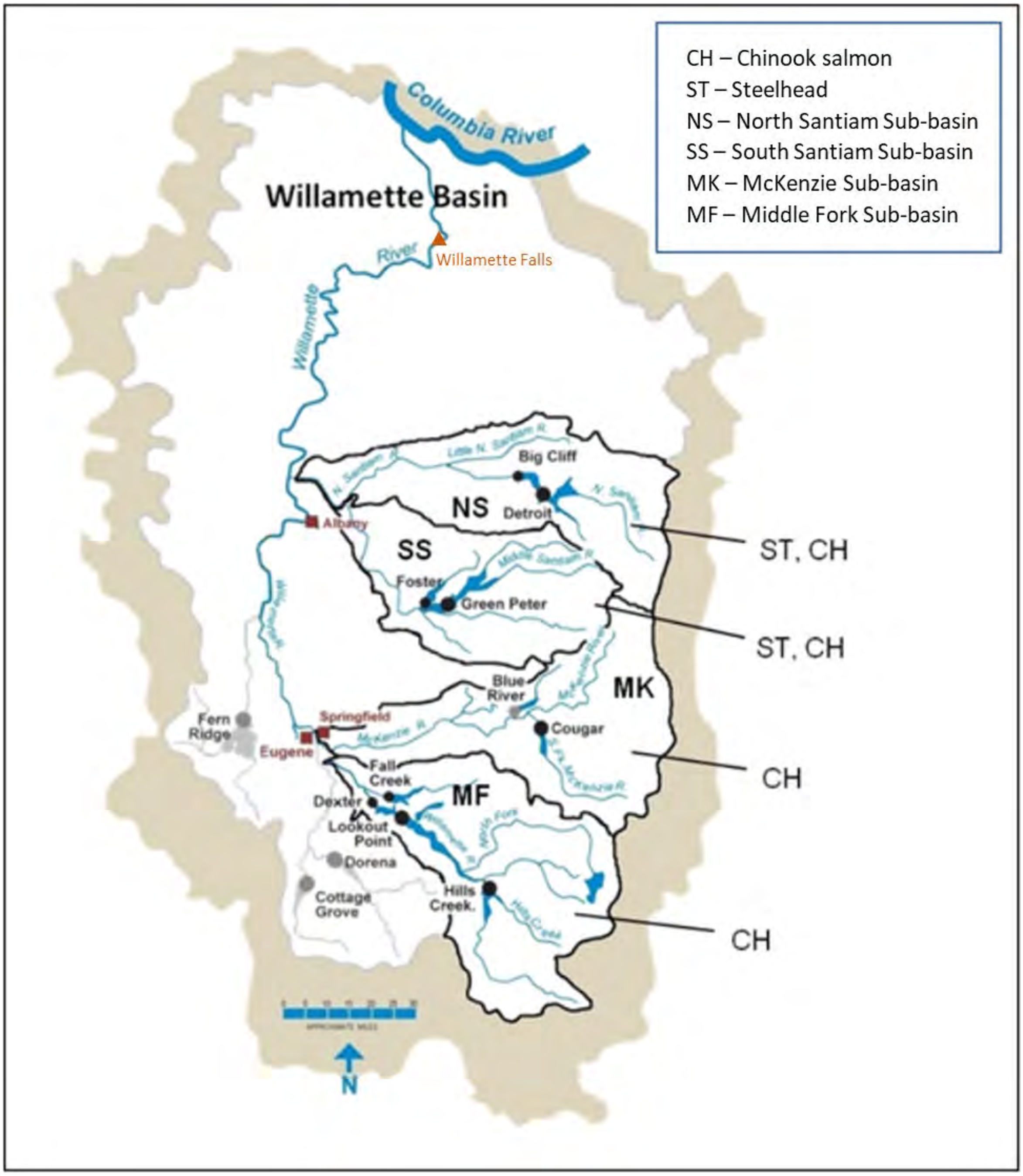

Figure 8-1. Map showing the four sub-basins of interest in the Upper Willamette River, indicating the species (spring Chinook salmon and/or winter steelhead) modelled. Also shown are the locations of the dams within each sub-basin, and Willamette Falls. Figure adapted from USACE (2015).

EIS Alternatives Evaluated

The US Army Corps of Engineers' (USACE) proposed several alternatives for downstream juvenile salmonid passage, upstream adult salmonid passage, and water temperature management for evaluation in the current Upper Willamette Valley EIS. Our modelling focussed on assessing the outcomes of implementing downstream passage alternatives, considering only the spawning population above the USACE dam projects in each of the following four sub-basins:

North Santiam – Detroit/Big Cliff (DET/BCL)

South Santiam – Foster (FOS) and Green Peter (GPR)

McKenzie – Cougar (CGR)

Middle Fork – Lookout Point/Dexter (LOP/DEX) and Hills Creek (HCR)

The downstream passage alternatives specified by USACE utilised two main categories of measures: operational and structural (USACE 2022). Operational measures included spring spill (SS) and spring drawdowns (SD) or fall drawdowns (FD) to regulating outlets (RO) or diversion tunnel (DT, where present). Structural measures included floating screen structures (FSS), floating surface collectors (FSC) or modified fish weirs. In total there were seven alternatives to evaluate, including the No Action Alternative (NAA). The measures proposed under each alternative are summarised in Table 1.2.1. Parameters determining the survivability and efficiency of dam passage under each alternative were obtained from the FBW (see Section 1.4).

Table 8-1. Summary of downstream passage measures by EIS alternative and USCAE dam project. Blank cells indicate no change to current measures. NAA=no action alternative, Alt=alternative, FSS=Floating Screen Structure, FSC=Floating Surface Collector, MW=Modified Fish Weir, SS=Spring Spill, SD=Spring Drawdown, FD=Fall Drawdown. Drawdowns to regulating outlets (RO) unless diversion tunnel (DT) specified.

		EIS alternative						
Sub-basin	Dam	NAA	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4
North Santiam	DET		FSS	FSS	FSS	SD FD	SS FD	FSS
	BCL		Collect at DET	Collect at DET	Collect at DET	SS	SS	Collect at DET
South Santiam	FOS		MW	MW	MW			MW
	GPR		FSS	SS FD	SS FD	SS FD	SD FD	
McKenzie	CGR			FSS	SD (DT) FD (DT)	SD FD	SD (DT) FD (DT)	FSS
Middle Fork	LOP		FSS	FSC	FSC	SD FD	SS FD	FSS
	DEX					SS	SS	
	HCR					SS FD	SD FD	FSC

Linked to each alternative were assumptions about the outplanting of adult returns above the dams in each sub-basin through trap and haul operations. For spring Chinook salmon, hatchery-origin returns (HOR) were available to supplement natural-origin returns (NOR) from the hatchery programs. There have been no hatchery-origin winter steelhead released in the upper Willamette since the 1990s (Myers et al. 2006), so all outplants were natural-origin winter steelhead. Hatchery summer steelhead are released below dams in the Willamette for a sport fishery (Myers et al. 2006), but these were not included in our modelling (see Section 3.1). The number and proportion of NOR and HOR Chinook salmon outplants put above dams each year varied by sub-basin, depending on the numbers of returns and hatchery program (Table 1.2.2).

The outplanting assumptions (Table 1.2.2, Table 1.2.3) were developed by consulting with the Hatchery and Genetic Management Plans for each sub-basin (ODFW & USACE 2016a, 2016b, 2018, 2019), guidance from USACE (Rachel Laird and Rich Piaskowski, pers. comm.), and numbers imposed by court injunctions (NEDC v. USACE 2021). In sub-basins where there were two dam projects (i.e., South Santiam and Middle Fork) the outplanting assumptions necessarily varied by alternative, which affected the proportions outplanted above each dam for both Chinook salmon (Table 1.2.2) and steelhead (Table 1.2.3). Under some alternatives there was no passage implemented at the upper dam (i.e., Green Peter and Hills Creek), so 100% of returning adults were assumed to be outplanted only above the lower dam. We note that under NAA there may be small numbers of juvenile steelhead passing downstream through Green Peter from a resident O. mykiss population in Green Peter reservoir that would not be excluded from the observed counts when returning to adults at Foster, but these are not modelled. Where there was passage at both dam projects within a sub-basin, we had to make assumptions about the relative proportion of adult returns that would be outplanted above each dam (Table 1.2.2, Table 1.2.3). As genetic sorting was not deemed feasible by USACE due to the mortality from holding fish for longer than necessary (Rachel Laird, pers. comm.), it was assumed that the returning adults trapped in adult collection facilities at the lower dam and released into the reservoir above would self-sort. In absence of other data, we used the relative habitat capacity of spawners (Chinook salmon, Section 2.2.1) or smolts (steelhead, Section 3.2.1) above each dam to determine the proportion of adults that would be outplanted above each dam (Table 1.2.2, Table 1.2.3). Some proportion would head to the adult collection facility at the upper dam and be outplanted above that to spawn, the remaining proportion were assumed to spawn in streams above the lower dam reservoir.

We also considered the future success of any alternative. If downstream passage was considerably improved relative to the NAA, then the numbers of natural-origin Chinook salmon adults returning could become considerably higher than current numbers. This could result in the trap and haul capacity being saturated, so we included a logistical cap on the number of fish that could be caught, held, and trucked above dams for outplanting (see Table 1.2.2). To determine a value for the outplant cap we considered the NOR spawner goals included as targets within the HGMP for each sub-basin and consulted with USACE to determine whether this value would be possible given the available truck numbers and their capacity. Studies have been undertaken in the Willamette to determine optimal hauling densities for Chinook salmon trap and haul operations (Colvin et al. 2018) and we did not want to set a cap that would exceed these. Due to the numbers of steelhead and the spawning habitat capacity being lower than for Chinook salmon, we assumed that trap and haul capacity for steelhead would not be exceeded and so did not consider implementing an outplant cap for steelhead.

Table 8-2. Chinook salmon outplanting assumptions in each sub-basin given EIS alternative. Specifications included the maximum number of hatchery-origin return (HOR) outplants, the logistical cap on the number of natural-origin return (NOR) outplants possible given population recovery, and the percentage of NOR to be outplanted above each dam in sub-basins with more than one dam project.

Sub-basin	Alternative	% NOR returns outplanted above	HOR outplants (max)	NOR outplant cap
North Santiam	NAA	0%	1,500	5,248
	Alt 1, Alt 2a, Alt 2b, Alt 3a, Alt 3b, Alt 4	100% (DET)	1,500	5,248
South Santiam	NAA, Alt4	100% (FOS)	0 (FOS) 800 (GPR)a	3,099
	Alt 1, Alt 2a, Alt 2b, Alt 3a, Alt 3b	52% (FOS) 48% (GPR)	0 (FOS) 800 (GPR)	3,099
McKenzie	NAA, Alt 1, Alt 2a, Alt 2b, Alt 3a, Alt 3b, Alt 4	100%	600	2,000
Middle Fork	NAA	100% (LOP)	1,257 (LOP) 387 (HCR)	5,000
	Alt1, Alt 3a, Alt 3b, Alt 4	58% (LOP) 42% (HCR)	1,350 (LOP) 1,100 (HCR)	5,000
	Alt 2a, Alt 2b	100% (LOP)	1,350 (LOP 387 (HCR)	5,000

aOutplanting of HOR above GPR determined by NEDC vs. USACE (2021) and is regardless of downstream passage implementation.

Table 8-3. Steelhead outplanting assumptions in each sub-basin given EIS alternative. Specifications included the percentage of NOR returns to be outplanted above each dam in sub-basins with more than one dam project.

Sub-basin	Alternative	% NOR returns outplanted above
North Santiam	NAA	0%
	Alt 1, Alt 2a, Alt 2b, Alt 3a, Alt 3b, Alt 4	100% (DET)
South Santiam	NAA, Alt4	100% (FOS)
	Alt 1, Alt 2a, Alt 2b, Alt 3a, Alt 3b	64% (FOS) 36% (GPR)

Performance metrics

We calculated performance metrics (PM) from the population trajectories output by the LCM for spring Chinook salmon and winter steelhead in each sub-basin to allow USACE to rank the EIS alternatives. The PM definitions were based upon the Viable Salmonid Population (VSP) score criteria developed by McElhany et al. (2000) and used in the 2015 Willamette Valley Projects Configuration/Operations Plan (COP, Zabel et al. 2015), but were modified following guidance from USACE (Rachel Laird and Rich Piaskowski, pers. comm.). Each PM was calculated from 10,000 simulation runs of the LCM over a 30-year management horizon prescribed by USACE, where each run was characterized by a given set of parameter values drawn from defined probability distributions. All PMs were summarised using the median across simulation runs, unless otherwise noted. The PMs were divided into four main categories:

1) Abundance

2) Productivity

3) Extinction risk

4) Diversity

The abundance PM was described as NOR spawners and aimed to evaluate the longer-term performance of each alternative by capturing population abundance at or near equilibrium. For a given simulation run, we determined the number of NOR spawners returned to each sub-basin in each year. In the Chinook salmon model, we accounted for pre-spawn mortality (PSM) above dams that would reduce the number of spawners relative to NOR returns to adult fish collection facilities, but before any logistical outplant cap was applied in a given year (see Appendix A). We assumed population abundance of Chinook salmon and winter steelhead would be lognormally distributed, with some years producing large abundances, so used the geometric mean to down-weight these larger values. The NOR spawners PM was therefore calculated in each simulation run as the geometric mean across years 16-30.

We calculated three productivity PMs: 1) recruits-per-spawner (R/S), 2) smolt-adult return rate (SAR), and 3) fry-smolt survival rate. These aimed to evaluate the maximum potential of each alternative to recover the populations from low abundance, and so were calculated over the first five years of each simulation run, i.e., during the first-generation following implementation of a given alternative. When populations are closer to or at equilibrium, productivity metrics such as R/S are close to one and so less informative about the performance of different alternatives.

Recruits were measured as returning NOR spawners (calculated post-PSM for Chinook salmon) as many as six years after spawning. We calculated the R/S performance metric in each simulation run as the geometric mean across years 1-5. We assumed that juveniles were smolts once they passed downstream of Willamette Falls, and calculated SAR as the ratio of the number of mature NOR adults returning to Willamette Falls after terminal harvest to the number of outgoing smolts at Willamette Falls five years previously. For each simulation run we took the mean over years 1-5.

Fry-smolt survival rate was calculated only for Chinook salmon as the product of all the juvenile survival rates from emergence to smolting at Willamette Falls. This included fry-migrant survival in natal stream, reservoir survival, dam passage survival, and river-smolt survival below dams (see Appendix A for values). Many of these parameters varied by juvenile age (fry, subyearling, yearling), so we calculated fry-smolt survival rates for each of the six juvenile migrant types defined (see section 2.2.3 for definitions). Similar to SAR, we took the mean over years 1-5 of each simulation run.

Extinction risk was measured by the probability of the abundance of NOR spawners falling below an extinction threshold when the population was at or near equilibrium, with the aim of identifying those passage alternatives for which the population was at more risk of being under the threshold in each sub-basin. Such estimates of extinction risk should be considered carefully as they typically have wide confidence intervals and are often optimistic due to the challenges in accounting for catastrophes that occasionally impact populations (Ludwig 1999). The extinction risk PM was calculated by determining the 4-year moving mean NOR abundance across years 16-30 of each simulation run, with the population deemed to go extinct if this mean abundance fell below a quasi-extinction threshold (QET) in those 15 years. Each simulation run thus scored either a 0 or 1, so we summarised the probability of extinction (P(NOR)<QET) as the mean across all runs rather than the median. We note that extinction risk is more typically assessed over a 100-yr period (Ludwig 1999), but we were limited by USACE tasking us to evaluate performance over the 30-yr management horizon.

We applied the QETs previously developed by the Willamette/Lower Columbia Technical Recovery Team (W/LC TRT, McElhany et al. 2007) and set by the Upper Willamette River Conservation and Recovery Plan for Chinook salmon and steelhead (ODFW and NMFS 2011, Table B.2-7) for each sub-basin, which were based on historical spawning habitat. For spring Chinook, these critical abundances were 250 NOR spawners per year in the South Santiam, McKenzie and Middle Fork Chinook, and 150 NOR spawners per year in the North Santiam. In the McKenzie sub-basin, owing to the model only accounting for populations in the North Fork of the McKenzie River, the critical abundance was pro-rated to 167 NOR spawners per year. For winter steelhead, we used the critical abundance of 200 spawners per year in the North Santiam. In the South Santiam, as the Recovery Plan did not consider habitat above Green Peter, we were instructed by USACE (R. Laird, pers. comm.) to double the critical abundance to 400 spawners per year in South Santiam, i.e., 200 above each dam. As the steelhead LCM modelled females only (see Section Section 3), these values were converted to critical abundance of females using the 58% sex ratio reported in the Willamette by Clemens (2015). This resulted in critical abundances of 116 female spawners per year in the North Santiam and 232 female spawners per year in the South Santiam.

Diversity was measured by four PMs: 1) proportion of hatchery-origin spawners (pHOS), 2) proportion of smolts of each juvenile migrant type, 3) proportion of adults of each juvenile migrant type, and 4) relative return rate of each juvenile migrant type. These PMs aimed to measure the influence of hatchery releases (with pHOS) and the life history diversity within the population (with juvenile migrant type metrics) once the population was at or near equilibrium, so were only calculated for Chinook. Due to spawners above dams being subject to PSM, which was itself a function of temperature and pHOS (Section 2.4.1 for details), for PM calculations we determined pHOS in each year as the ratio of hatchery-origin to natural-origin outplants put above each dam. The PM for pHOS was calculated as the mean pHOS across years 26-30 of each simulation run and summarised these values using the mean.

The juvenile migrant type PMs were determined by tracking the population abundance of each migrant type from dam passage through smolting at Sullivan Dam Juvenile Bypass Facility (SUJ) and return to the river as adults at Willamette Falls. For each migrant type smolt at SUJ, we calculated the geometric mean abundance across years 26-30 and then calculated the relative proportion of each migrant type for each simulation run. The same calculations were performed to determine the proportions of each migrant type at the time of adult return. The relative return rate of each migrant type was calculated as the ratio of the number of adults of each migrant type returning to Willamette Falls after terminal harvest to the number of outgoing juvenile smolts of each migrant type at Willamette Falls five years previously. This assumed that fry and subyearling smolts spent the same length of time in the ocean as yearling smolts. For each simulation run we took the mean over years 1-5 and summarised these values using the median.

For the Chinook LCM, we calculated all PMs following the removal of an initial five-year burn-in period, i.e., generation zero of a simulation run (Table 1.3.1). This was because population trajectories under some alternatives could be quite variable during this period because the model projected from the recent average annual count and initially applied average observed spawner age composition rather than model-generated spawner age composition (see Section 2 and Appendix A). Depending on the values of each alternative's dam passage parameters relative to the NAA, once the model-generated spawners into each age class started adding to the total population the abundance could become much higher or lower relative to the initial conditions. After the variability in the initial generation, the simulation model typically converged to an equilibrium within a couple of generations, which fit within the prescribed 30-year management horizon (see Section 2.7 for results from each sub-basin). The steelhead LCM was structured differently and instead projected from the final year of count data in 2021 (see Section 3 and Appendix B). We removed an initial two-year burn-in following assessment of the population trajectories following implementation of alternatives in 2022 due to increased variability in this period owing to lag effects related to the simulation of marine survival. Definitions of the PMs computed for steelhead are summarised in Table 1.3.2.

Table 8-4. Definition of performance metrics (PM) computed from the Chinook salmon life cycle model given a 30-year management horizon. PMs were calculated from 10,000 simulation runs, following removal of a 5-year burn-in. R/S = Recruits-per-spawner; SAR = smolt-adult return rate, pHOS = proportion of hatchery-origin spawners; P(NOR) < QET = probability that NOR returns are less than the Quasi-Extinction Threshold (QET). Note that the median across simulations was used as a summary statistic for each PM apart from P(NOR) < QET, which used the mean across simulations.

Performance Metric	Description	Statistic
Abundance	NOR spawners	Geometric mean of year 16-30
Productivity	R/S	Geometric mean of year 1-5
	SAR	Mean of year 1-5
	Fry-smolt survival	Mean of year 1-5
Extinction risk	P(NOR) < QET	4-yr mean, year 16-30
Diversity	pHOS	Mean of year 26-30
	% migrant type smolts	Year 26-30
	% migrant type adult returns	Year 26-30
	Migrant type SAR	Mean of year 26-30

Table 8-5. Definition of performance metrics (PM) computed from the steelhead LCM given a 30-year management horizon. PMs were calculated from 10,000 simulation runs, following removal of a 2-year burn-in. R/S = Recruits-per-spawner; SAR = smolt-adult return rate; P(NOR) < QET = probability of NOR spawners below the Quasi-Extinction Threshold (QET). R/S start year is associated with age-4 recruits, SAR start year is associated with age-2 smolts. The median across simulations was used as a summary statistic for the PM apart from P(NOR) < QET, which used the mean across simulations.

Performance Metric	Description	Statistic
Abundance	NOR spawners	Geometric mean of year 16-30
Productivity	R/S	Geometric mean of year 1-5
	SAR	Mean of year 1-5
Extinction risk	P(NOR) < QET	4-yr mean, year 16-30
Diversity	None evaluated	

Fish Benefits Workbook

Workbook overview

Juvenile downstream passage past high-head dams is modeled using the Fish Benefits Workbook (FBW), a model of downstream juvenile passage and survival informed by water flow distribution and fish responses to hydrology. This model provides two key outputs which were integrated within the LCM: dam passage efficiency (DPE), the proportion of available fish which will attempt to pass the dam under simulated conditions; and dam passage survival (DPS), the expected survival of those fish which attempt to pass. FBW was created in 2014 by USACE with the goal of ranking proposed changes to dam structure and operation based on their ability to improve juvenile fish DPE and DPS. The Corps was responsible for parameterizing and running the FBW workbooks, providing us with estimated DPE and DPS for each species, life-stage, and dam under each EIS alternative. In the current application, the model is contained in several worksheets within a Microsoft Excel (TM) workbook, using both in-cell formulae and Visual Basic for Applications; from these workbooks we extracted yearly estimates of DPE and DPS for incorporation into the LCM.

FBW was used to model spring Chinook salmon and winter steelhead as they approach and attempt to pass each of eight dams in the Willamette River Basin: Big Cliff, Detroit, Green Peter, Foster, Cougar, Dexter, Lookout Point, and Hills Creek. The model was parameterized and run for each reservoir under each alternative, with different parameterizations based on species and life stage. In the case of Chinook salmon, FBW was parameterized for fry, sub-yearlings, and yearlings; in the case of steelhead, FBW was parameterized for sub-yearlings, yearlings, and age 2 or above. FBW considers a population as the total number of same-aged juvenile fish of each species that would approach the dam in a year; as a result, FBW calculates DPE and DPS as proportions of the annually approaching population of each species and life stage.

Inputs to FBW

The FBW model was informed by two types of input parameters: hydrological and biological. Hydrological inputs to the model were generated by the Reservoir System Simulation (ResSim) software, a Corps-developed hydrological model (see USACE 2013 for details on previous ResSim modeling). Given specified fish passage and dam operation information, ResSim simulates pool elevation and flow rates through a given dam's outlets (e.g., through the spillway, turbines, regulating outlets, fish passage structures, and other outlets if present) for each day within the hydrological period of record (1934-2019, but often limited to only 1947-2019 due to data quality and logistical difficulties). ResSim estimates flow rates through each dam outlet, assuming dam operations perfectly match the management scenario without deviation. If a given structure is unavailable to fish at a given time (e.g., if pool elevation drops below the level where that outlet is available to fish), the flow rate through that outlet is 0. In some cases, management actions include temperature operations, such that simulated flow regimes and the impacts of flow on fish passage and survival can differ between water year types.

ResSim outputs serve as the foundation of the FBW model, which also includes biological parameters that inform how fish respond to hydrological conditions. FBW's biological input parameters include 1) monthly run timing for each species and life stage, 2) dam passage efficiency, 3) relative attractiveness of each outlet, and 4) route-based survival (see following section for details on how these inform the model). In 2013, to evaluate fish passage alternatives, USACE contracted TetraTech and subconsultants BioAnalysists and Alden Research Laboratory to create justifiable values for these biological input parameters, considering likely outcomes under each proposed management alternative (Alden BioAnalysts Inc. 2014). The consultants compiled gray and published literature available in 2013 for the three Chinook life stages, primarily from active and passive tagging studies at Willamette River Basin dam projects and augmented by research from other Corps projects and/or expert opinion where data gaps remained. Location-specific run timing data were available for most dams (excluding Hills Creek and Green Peter), but dam passage efficiency, route attractiveness, and route-based survival rates were primarily taken from studies of Chinook salmon at Cougar and Detroit. Full details on initial FBW parameterization are given in a comprehensive memorandum (Alden BioAnalysts Inc. 2014). Since 2014, the Corps has updated FBW input parameters as new data are made available.

Processes modelled with FBW

FBW simulates daily fish passage and survival in a series of four modelling steps. First, monthly run timing information is discretized into daily run timing. The proportion of the year's fish population expected to pass in each day is calculated based on either 1) a flat daily rate, by dividing the percent approaching in each month by the number of days in that month; or 2) in proportion to flow rates through the month (i.e., if 10% of the monthly flow is expected to pass on a certain day, 10% of the monthly fish population will try to pass on that day). All FBW runs by USACE used daily flow to adjust within-month fish distribution.

Second, approaching fish in each day are routed through the dam's outlets. When fish reach the dam's forebay, they may either attempt to pass or remain in the reservoir. The probability of the fish passing is defined by the DPE parameter, expressed either as a fixed value or as a function of pool elevation. For fish that pass the dam, they are then distributed between dam outlets based on two factors: 1) the distribution of flow through each outlet, and 2) the attractiveness of each outlet relative to alternative options. This second component, route attractiveness, is calculated as a function of proportional flow through each outlet and the outlet type (i.e., spillway, turbine, regulating outlet, or fish passage structure). Outlets that are more attractive and/or consume a higher proportion of the total flow passing through the dam are expected to have higher attractiveness to passing fish. For outlets where there are multiple sub-routes (e.g., multiple turbine gates, such that the turbine outlet encompasses several gates that serve as sub-routes), fish are also divided into sub-routes based on specified dam operations and their effect on flow distribution within sub-routes.

Third, FBW calculates survival through each outlet and sub-route. Survival estimates of fish passing through the dam were derived from various tagging experiments performed with subyearling and yearling fish at Cougar, Detroit, and other Willamette project dams. Due to limitations in tagging methods, survival estimates for fry (typically less than 60mm in length) were parameterized from 1) the upper bound of 95% confidence intervals of sub-yearling survival rate estimates where available, or 2) upwards-adjusted sub-yearling survival rates. Within FBW, passage survival through each route was parameterized either as a point value or as a function of flow through the outlet/sub-route.

Finally, daily estimates of the proportion of passing and surviving juvenile fish are then summarized into monthly and annual estimates of DPE and DPS used by the LCM. Further details of on parameterization and operation of FBW are available in Appendix K of the Willamette Valley Configuration/Operations Plan (USACE 2015).

Software Platforms

Life cycle models were developed in two main environments, Microsoft Excel and R 4.1.2 (R Core Team 2021). We first developed deterministic versions of the models in Excel, then coded probabilistic versions of the models up in R. This allowed us to cross-reference the model outputs from each and aid in debugging. We feel this is a very important quality control step to perform. Analyses undertaken to parameterize the models were performed in both Excel and R, with WinBUGS 1.4 (Spiegelhalter et al. 2007) used for some of the PIT-tag data analyses.

Spring Chinook Salmon Life Cycle Model

Introduction

LCM Modelling Approach

A sequential quasi-stock assessment modelling approach was applied for UBC LCM parameterization, fitting, and projections to evaluate alternative EIS dam passage measures. Conventional stock assessment modelling approaches and approaches taken to formulate “operating” models for management strategy evaluation (MSE) (e.g., Edwards and Dankel 2016) typically construct an age-structured population dynamics model, accounting for density dependence in the egg to juvenile phase usually with a Beverton-Holt or Ricker stock-recruit function, and then predict numbers at age in each cohort accounting for natural and fishing mortality rates at age. Some model parameters by convention, but not always, are fixed at specific values (e.g., natural mortality rates, fraction maturing at age, and somatic growth rate parameters in many stock assessments). Other parameters, e.g., ones that scale the population dynamics and abundance estimates (e.g., average unfished spawner biomass) are freed up. Freed up parameters are constrained (e.g., via Bayesian priors or minimum and maximum bounds on the “freed” up parameters) and estimated when the stock assessment or MSE operating models are fitted to time series of abundance and age composition in catches and research survey records (e.g., McAllister and Ianelli 1997; Licandeo et al. 2020). Even when Bayesian approaches are taken in stock assessment and MSE in which prior distributions are formulated for the estimated population dynamics model parameters, it remains common to keep several of the population dynamics model parameters fixed, e.g., fraction maturing at age and somatic growth rate parameters, and fit the population dynamics model to the available data using a non-linear function minimizer that minimizes the sum of the negative loglikelihood and negative log joint prior density function (see e.g. Licandeo et al. 2020 and the numerous population dynamics model fitting applications in Edwards and Dankel 2016).

Whether Bayesian or non-Bayesian approaches are taken, a key diagnostic criterion is the goodness of fit of the model to time series of records of abundance from fishery-independent surveys of abundance (e.g., Francis 2011). This is typically computed by either applying a conventional function minimizer for the model fit objective function or by applying an MCMC or SIR algorithm (e.g., McAllister and Ianelli 1997; Licandeo et al. 2020). In either case, only population dynamics models that provide good fits to the time series of abundance data are considered acceptable for projections to evaluate alternative management policy options.

Uncertainty in parameter values in stock assessments is typically accounted for by either 1) simulating plausible parameter values from prior distributions for them and bootstrapping the data, 2) by a formal Bayesian model fitting approach or 3) by testing of the sensitivity of results to applying a range of different values for parameters that are treated as fixed and given in the population dynamics model (e.g., McAllister and Ianelli 1997; see applications in Edwards and Dankel 2016; Licandeo et al. 2020). In stock assessment and MSE models, prior distributions for model parameters have been formulated by a variety of approaches ranging from Bayesian hierarchical modelling of stock-recruit data (e.g., Michielsens and McAllister 2004), statistical modelling of data from experiments with incorporation of expert judgment (e.g., McAllister et al. 1994, 2010; McAllister and Ianelli 1997) or purely from expert judgment (e.g., Uusitalo et al. 2005).

The UBC LCM conformed to conventional stock assessment and MSE approaches to model formulation and accounting for uncertainty in parameter values by 1) formulating prior distributions for many of the biological parameters in the LCM based on results and data sets published in reports and the scientific literature, 2) formulating likelihood functions for the available time series of records of adult counts at dam tailraces and spawner age composition records and then fitting the LCM, and 3) freeing up and estimating some of the population dynamics model parameters by fitting the LCM to available time series of records of adult counts at dam tailraces and spawner age compositions. Due to the large number of biological parameters in the UBC LCM (see Appendix A) and the limited amount of informative data available for model fitting, it was not possible to jointly free up all of the parameters when fitting the LCMs to the time series data. Therefore, parameters that were not freed up were set at either their most credible prior value or the mid-point of their uniform prior density function when the LCM was fitted to the data. However, when projecting the model to evaluate the EIS alternatives, the prior distributions for model parameters that were not freed up, together with the posterior distributions for the model parameters that were freed up in the model fitting were applied to simulate uncertainty in population dynamics parameter values. Further details on the model fitting and simulation methodology and results are presented in a subsection below (Section 2.5).

Overview of UBC LCM model structure and parameterization

The LCM is a time dynamic age-structured population dynamics model that represents spring Chinook salmon population dynamics in both freshwater and saltwater (Figure 2.1.1). It is composed of a set of modules each representing one of the sequential life stages from egg to spawning adult. A Beverton-Holt equation is applied to represent density dependence in egg-to-fry survival rates. Two well-known life histories are modelled (Schroeder et al. 2016), which vary in terms (i) whether the juvenile fry stay to rear in their natal spawning stream (i.e., stayers) or migrate immediately downstream to the reservoir after becoming fry (i.e., movers). Depending on their reservoir residency and life history stage (e.g., fry, subyearling or yearling) at dam passage, these groups generate six different juvenile migrant types. These six juvenile migrant types will be described in detail in a subsection below (Section 2.2.3).

The LCM took into account the juvenile passage through dams (see Section 1.4). The dam passage component of the LCM used DPE and DPS to represent the fraction of juveniles in a dam's forebay that migrate through the dam and the fraction surviving dam passage, respectively. DPE and DPS were provided separately for fry, subyearlings and yearlings, for each EIS alternative based on the Corps FBW software (Section 1.4). Because DPE was always less than 100%, and some fraction of juveniles of each migrant group at the dam forebay do not pass through the dam, the non-passing proportion were modelled to remain in the reservoir to pass at the next migration event in the year or the next spring, accounting for reservoir survival rates.

The abundances of each of the six juvenile migrant types were tracked separately through to spawning. The UBC LCM explicitly modeled for each cohort of each juvenile migrant group the numbers at age in each year in the sea, accounting for the fractions dying from natural and ocean fishing mortality and the fractions maturing at age. Fishery bycatch mortality rates were modeled for adults returning to their natal rivers. Pre-spawn mortality rates (PSM) due to e.g., heat stress and disease, were modelled for upstream migrating adults with separate components for PSM below and above dams.

After the models' modules for each life stage were formulated, plausible point estimates and ranges of parameter values were identified based on previous field studies and statistical analyses of data obtained from them. To obtain parameter estimates for downstream survival rates and total smolt-adult survival rates, a Bayesian Cormack-Jolly-Seber (CJS) estimation methodology was formulated and applied to PIT tag study release and detection records from the Upper Willamette in each of the sub-basins (Appendix C). It is important to note that these studies were performed mostly on hatchery origin spring Chinook salmon. The smolt to adult survival rate was treated as separate parameter for juveniles that enter the sea as smolts in either their first spring (i.e., soon after becoming fry), first autumn or winter (i.e., as subyearlings) or second spring (i.e., as yearlings). The CJS methodology incorporated informative priors (see Appendix H) for juvenile and adult survival rates and the detection rates of juveniles at the Sullivan Dam Juvenile Salmon Tag Detection Facility (SUJ) and the detection rates of returning adult salmon in the fishways at Willamette Falls (WFF). The use of these informative priors enabled more precise estimates of downstream juvenile survival rates and total smolt-adult survival rates. Priors for tag-induced mortality rates, tag loss rates, and the ratio of natural origin to hatchery origin survival rates of tagged fish were developed based on published findings on these parameters in the literature (see Appendix I). The adjustments for tag-induced mortality rates, tag loss rate and hatchery effect were applied to the posteriors for survival rates from the Bayesian CJS methodology to formulate adjusted posterior distributions for downstream juvenile and total smolt-adult survival rates.

The Bayesian CJS estimate of total average smolt-adult marine survival rate was reparameterized to explicitly represent the CTC (2021) derived estimates of long-term average survival rates from natural mortality and long-term average fishing mortality for each sea age of Upper Willamette hatchery spring Chinook salmon. Based on the CJS estimates of average smolt-adult survival rates, the average survival rates from both natural mortality- and fishing-at age (CTC 2021), available age composition, and the fraction maturing at age, the survival rate in the first year at sea could be solved for analytically for Chinook salmon in each sub-basin (see Appendix D). The model assumed that the cohort population dynamics are initialized when eggs are fertilized (not when the fry emerge). Salmon age was then determined by the number of years since egg fertilization. The UBC LCM allows for maturation at either age three, four, five or six years. The seven-day observed average maximum river water temperature was used to compute the pre-spawn mortality rates above the dams (see Section 2.4.1).

Based on the initial input distributions for LCM model parameters, the time series of model-projected values for historical adult abundances were markedly different from the time series of adult counts at dam tailraces. To project the population dynamics, the LCM needed to accurately represent the current abundance, and so needed calibrating. This led to a second stage of model parameterization in which a few of the LCM parameters were freed up and the LCM was fitted to the available time series of adult counts and spawner age composition records for salmon that spawned above the dams (see Section 2.5). The freed-up parameters included the average first year at sea survival rates for subyearlings and yearlings and the average fraction maturing at age. An annual deviate in first year marine survival rate was also estimated for each year for which adult count data were available. This is because in some years there were large fluctuations in counts of natural origin adults at the dam tailrace facilities that could not be explained by existing model parameters.

For the Chinook salmon operating model/calibration model, the LCM was coded in Microsoft Excel and Microsoft Excel Solver's function minimization was used to search for parameter values that minimized the objective function (i.e., the sum of the negative log likelihood of the adult count and spawner age composition and negative log prior of the freed-up parameters, see Section 2.5).

To carry out forward projections and compute output distributions of each of the performance metrics for each EIS alternative the LCM was coded in R Statistical software (R Core Team 2021). It was verified that the Excel and R models generated matching predictions of abundance at age, given the same input parameter values. For the 30-year projections, uncertainty in LCM model parameters was accounted for by drawing parameter values from probability density functions that were determined through model fitting as outlined above and in further detail in the model calibration section. The LCM for spring Chinook salmon was built to be generalizable (with modifications) to each sub-basin. This enhanced collaborative efforts between project team members, avoided duplication of model building and coding effort and made the application of model updates more efficient. Further details on the LCM simulation methodology applied to compute EIS alternative performance metrics are provided below (Section 2.5).

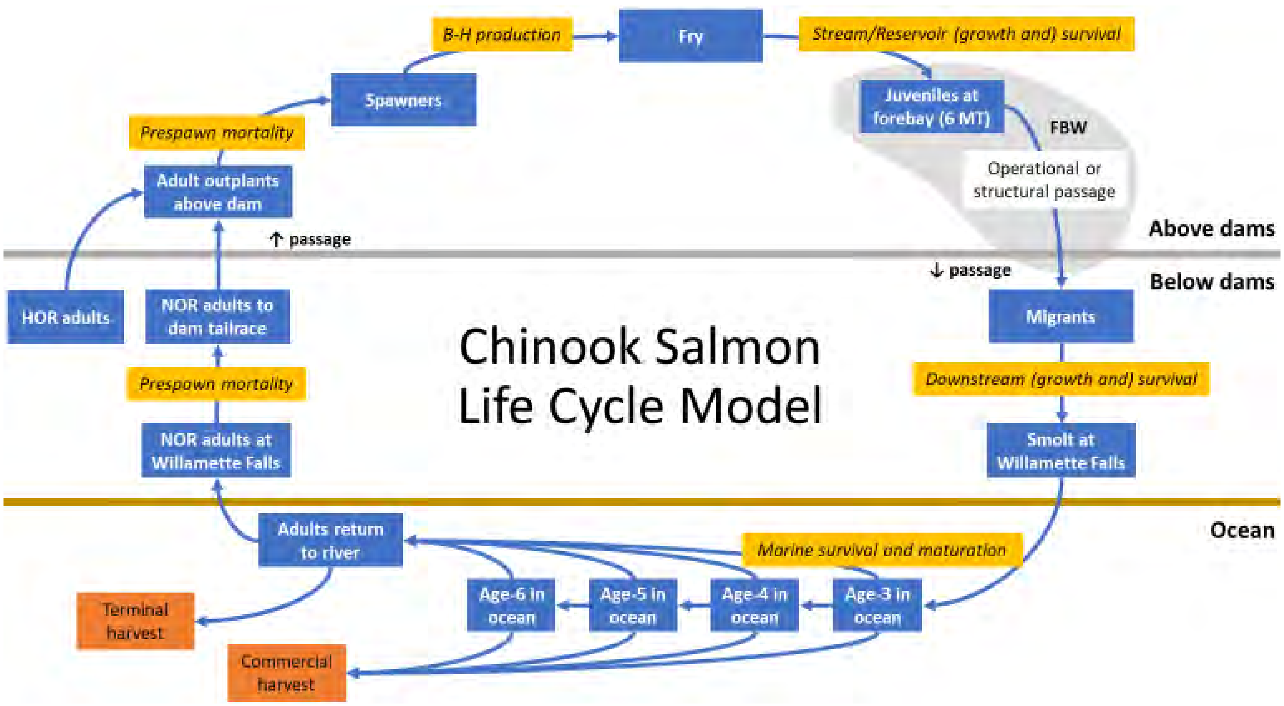

Figure 8-2. Outline of the Chinook salmon life cycle model, describing parts of the life cycle that occur above dams, below dams, and in the ocean. Blue boxes indicate life stages included in the model, yellow boxes indicate population processes included in the model, orange boxes indicate harvest rate assumptions, white boxes indicate dam passage parameters provided by Fish Benefits Workbook (FBW). Note: growth is not incorporated into the current version of the model.

Juvenile Freshwater Stage

This section describes the life cycle model components related to juvenile Chinook salmon rearing in freshwater from eggs until they were assumed to smolt and enter the estuary and marine environment after passing Willamette Falls.

Beverton-Holt density dependence in juvenile production

For all Chinook salmon populations, we assumed Beverton-Holt density dependence in egg-fry survival rate to calculate the number of emergent fry from the numbers of eggs spawned by adults in each year. We did not account for density dependence in other parts of the life cycle. As applied in this model, the Beverton-Holt productivity term (a parameter) was equal to the egg-fry survival rate in the absence of density dependence, and the spawner habitat capacity term (b parameter) was equal to the egg capacity of the spawning reaches. We chose to model density dependence as a function of spawning habitat capacity because the range of life history types would make it difficult to model it at a later juvenile stage above dams; a similar assumption has been made by previous Upper Willamette life cycle models (ODFW and NMFS 2011; Zabel et al. 2015). Using egg capacity implies that there is some limited amount of habitat available for egg deposition, meaning that as spawner density increases, redds may become superimposed or spawning will occur in less favourable habitat. We note that egg deposition from current spawner numbers is assumed to well below these egg capacity estimates, so unless there are significant improvements to downstream fish passage, we did not expect density dependence to be a major factor in our life cycle model.

For the productivity term in the Beverton-Holt model, we applied the egg-fry survival values used by Zabel et al. (2015) for the above dam reaches in each sub-basin (values in Appendix A). These values were based upon a review of studies on incubation success, with consideration for the quality of habitat in these reaches. Reaches with little or no habitat degradation had egg-fry survival rate of 50-60%, reaches with lower quality habitat had lower survival rates.

The egg capacity estimates also followed those used by Zabel et al. (2015) for the above dam reaches (values in Appendix A). These were based upon historical redd surveys prior to dam construction, e.g. Parkhurst et al. (1950). Relative to periods prior to dam construction, it is generally thought that it is mainly the quality of habitat that has changed since (i.e., affecting egg-fry survival) rather than the quantity as the amount of spawning gravels has not decreased in proportion to declines in run size. Since construction of dams there has been loss of spawning habitat to reservoir area and due to some degradation of habitat quality, so the historical data were adjusted to reflect this, typically to around 33-50% of the historical values (Zabel et al. 2015). We considered applying the spawner habitat estimates from Bond et al. (2017), but we determined that they were too large relative to Zabel et al. (2015). Bond et al. (2017) were attempting to determine the maximum number of spawners that could occupy the available and useable habitat, and recognised that their estimates were potentially too large due to their ability to determine usable habitat from stream surveys.

Fry survival and movement to reservoirs

Fry survival in natal streams depends on when they migrate downstream, which may either be as fry in the spring after emergence, as subyearlings later that summer or fall, or as yearlings by the next spring. As survival decreases with longer periods spent in the natal stream, fry-migrant survival is lower for the yearling life stage than for fry. We applied the fry-migrant survival values for each life stage used by Zabel et al. (2015) for the above dam reaches in each sub-basin (values in Appendix A).

To determine the proportion of emergent fry moving to the reservoirs at each life stage, we analysed rotary screw trap (RST) data obtained from ODFW. These data were part of a series of juvenile salmonid outmigration monitoring studies conducted in each sub-basin between 2011 and 2016 (Monzyk et al. 2011a; Romer et al. 2012, 2013a, 2014, 2015, 2016, 2017). The length of each juvenile Chinook salmon was recorded and using a combination of size- and time-based rules we determined the life stage of each juvenile. We determined that 1) juveniles migrating at <60mm were fry, 2) juveniles >60mm migrating during or after January of the year after emergence were yearlings, and 3) all other juveniles were subyearlings. An example dataset from the RST located at the head of Detroit reservoir (above Detroit) is shown in Figure 2.2.1.

Our analyses found that most of the juveniles move downstream to the reservoirs as fry (80-90%), a lower proportion move as subyearlings (10-20%), and a very low proportion move as yearlings (1-5%), e.g., Figure 2.2.2. Across sub-basins we found similar proportions to those used by Zabel et al. (2015), who performed a similar analysis using data from these RST studies 2011-2014. The values applied in each sub-basin are shown in Appendix A. We did not attempt to incorporate between year variation in the proportions moving to the reservoirs at each life stage into our models.

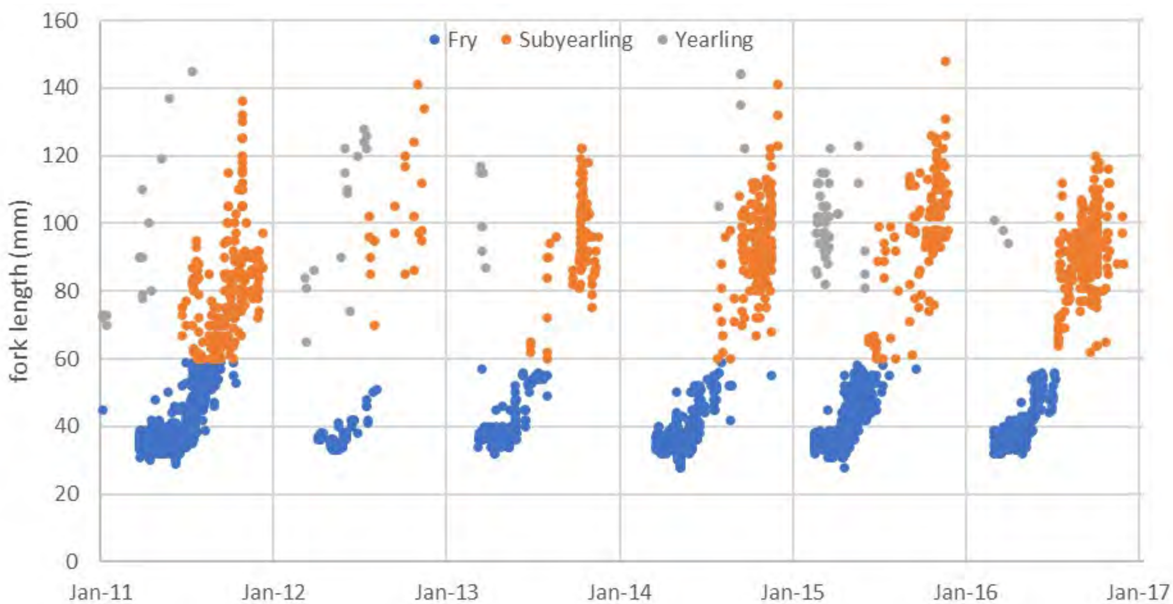

Figure 8-3. Fork length of juvenile Chinook salmon captured in rotary screw traps located at the head of Detroit reservoir by date and life stage, 2011-2016. Data from Monzyk et al. (2011a) and Romer at al. (2012, 2013a, 2014, 2015, 2016, 2017).

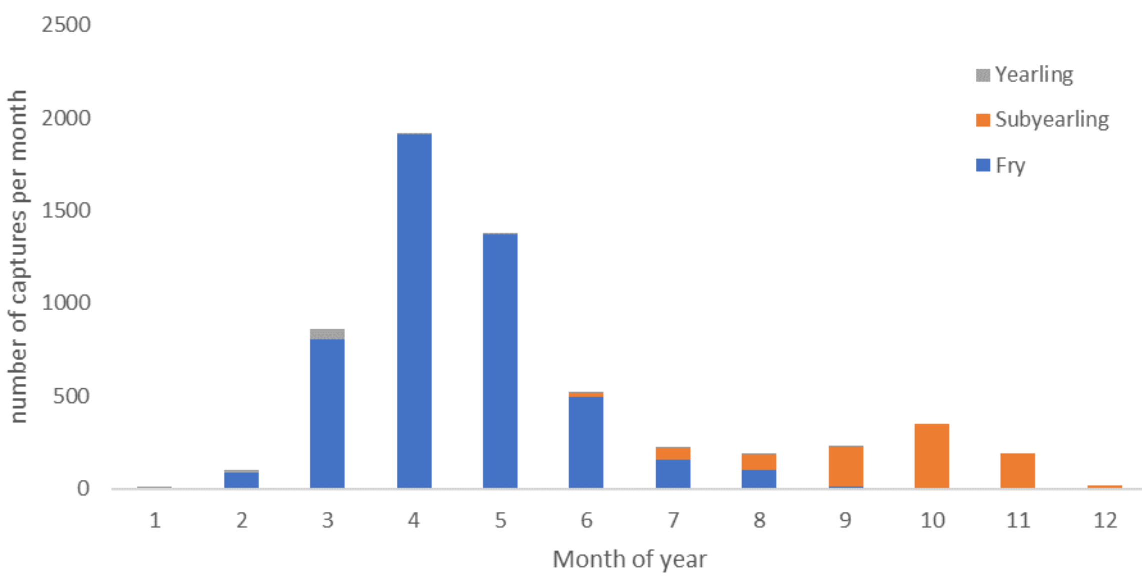

Figure 8-4. Monthly distribution of juvenile Chinook salmon captured in rotary screw traps located at the head of Detroit reservoir by life stage. Data from Monzyk et al. (2011a) and Romer at al. (2012, 2013a, 2014, 2015, 2016, 2017).

Life history pathways modelled

There is evidence for wide diversity of Chinook salmon life histories (Groot and Margolis 1991; Waples et al. 2001; Bourret et al. 2016). It is understood that maintaining this diversity is necessary for population stability as it give these populations resilience to variable and uncertain environments (Waples et al. 2009; Schroeder et al. 2016; Bourret et al. 2016). Although not completely distinct categories but a continuum, there are generally two main life history pathways discussed in the literature, defined either as stream vs. ocean (Bourret et al. 2016) or stayer vs. mover (Schroeder et al. 2016). The key difference between these is that in one pathway the fry stay and rear in the natal stream before moving to the ocean as yearlings, while in the other they move and head to the ocean at an earlier fry or subyearling life stage.

In the Willamette River, recent work by Schroeder et al. (2016) in the McKenzie sub-basin documented stayers and movers below dams. Due to timing of migration from natal streams (movers, stayer-fall migrant, stayer-spring migrant) and timing of smolting (spring subyearling, fall subyearling, spring yearling), they suggested six key life history types. Our model is for the above dam spawning population, which complicates these definitions as the dam interferes with what would occur naturally, as the reservoirs created by dams present a novel rearing habitat so the life history types present may differ from those below dams. It is difficult to determine volitional reservoir residence from impeded downstream movement, and so the influence of dams in life history diversity remains unknown (Bourret et al. 2016). Due to this, we defined diversity in terms of juvenile migrant types, rather than juvenile life history types.

We modelled six juvenile migrant types within the three main life stages (fry, subyearling, yearling), based upon when they migrate from natal streams into reservoirs and when they are able to pass the dams and are able to smolt (Figure 2.2.3). These can be related to the six life history types described by Schroeder et al. (2016), as shown in Table 2.2.1. The key distinction is that impeded passage results in some life history types not being able to smolt when they would naturally. It is not possible to distinguish between migrant types that move to reservoirs and choose to rear for a period of time from those that have their downstream migration impeded. The model tracked each of these juvenile migrant types from the reservoir, through smolting and return to the river as adults.

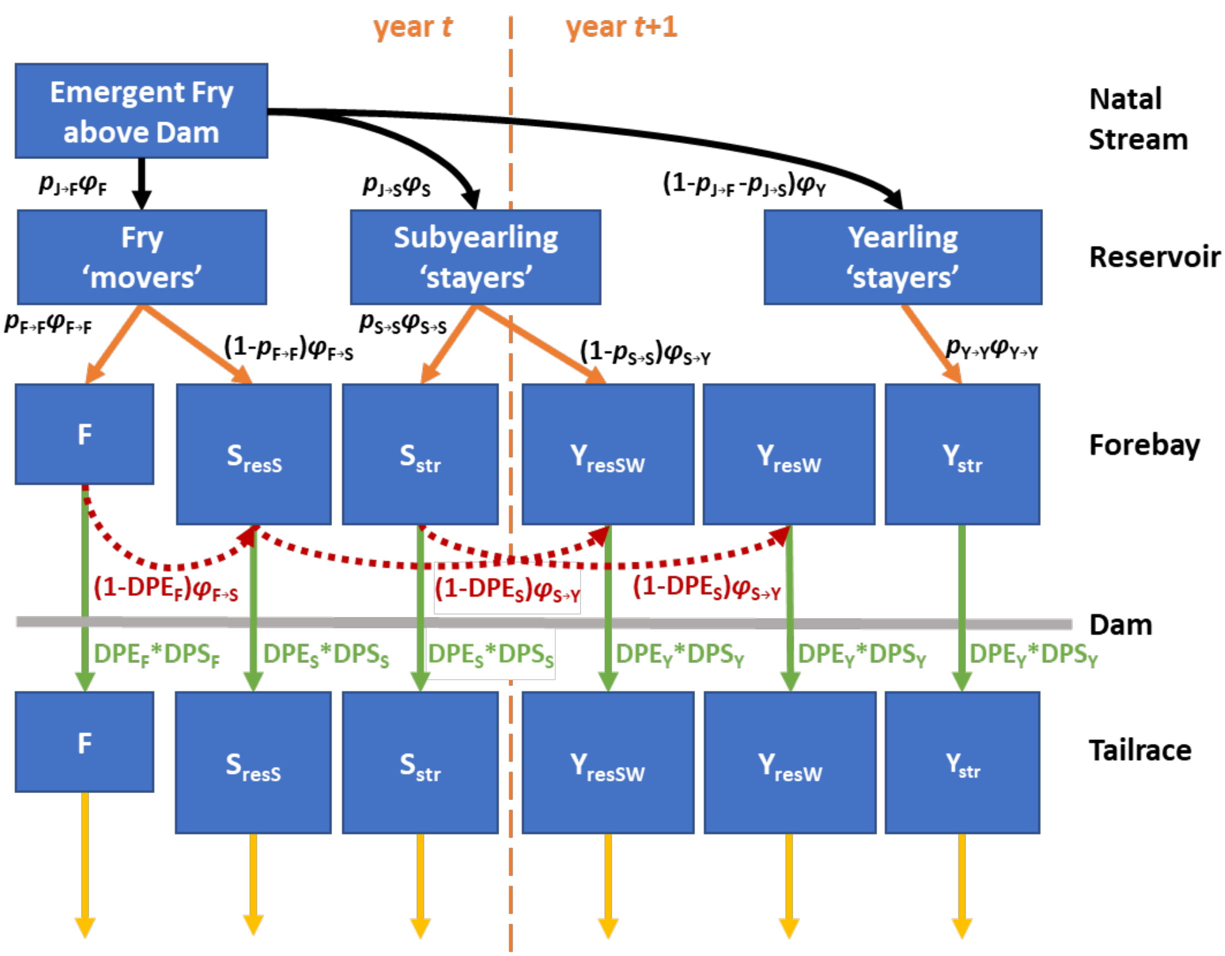

Figure 8-5. A conceptual diagram of the components included in the Chinook salmon life cycle model for above dam and dam passage processes. Diagram explains how each juvenile migrant types arises (F = fry, SresS = subyearling that spent summer in reservoir, Sstr =subyearling that spent summer in natal stream, YresSW = yearling that spent summer and winter in reservoir, YresW = yearling that spent only winter in reservoir, Ystr = yearling that spent summer and winter in natal stream). Also shown is where the model timestep changes (dashed orange line). All parameters are described in Appendix A.

Table 8-6. Definition of the juvenile migrant types applied in the Chinook life cycle model with a comparison to the life history types documented in the Willamette River by Schroeder et al. (2016).

Juvenile migrant type (life stage – rearing location before smolting)	Schroeder et al. (2016) life history type (migrant type – smolt type)
Fry	Mover – spring subyearling
Subyearling – reservoir rearing in summer	Mover – fall subyearling
Subyearling – natal stream rearing in summer	Stayer-fall migrant – autumn subyearling
Yearling – reservoir rearing in summer & winter	Mover – spring yearling

Yearling – natal stream in summer, reservoir in winter	Stayer-fall migrant – spring yearling
Yearling – natal stream rearing in summer & winter	Stayer-spring migrant – spring yearling

The contribution of each juvenile migrant type to the population may be influenced by the alternative passage measures, as changes in dam passage efficiency may allow more or less juveniles to migrate when they choose. However, this information is not available. Juveniles that rear for longer periods in reservoirs have increased growth relative to juveniles that rear in streams (Murphy et al. 2020). This may result in an advantage in smolt-to-adult survival due to increased size at smolting (Bourret et al. 2016). This has been observed in Fall Creek subyearlings that rear in the reservoir over summer but then pass the dam following a deep fall drawdown (Murphy et al. 2019). There are costs to reservoir rearing, including survival during passage at dams being lower for larger juveniles (Keefer et al. 2012) and reduced survival in the reservoir due to predation and parasitism; though some operational measures may mitigate this cost, e.g., fall drawdown may reduce predator numbers in a reservoir (Murphy et al. 2019). In the absence of specific experiments to test the effect of different dam passage measures, it is not possible to determine whether the benefits of increased smolt-adult survival outweigh the costs of increased time rearing in reservoirs.

In-reservoir survival and movement

There have been few studies on in-reservoir survival in Willamette Valley reservoirs. It is well known that the in-reservoir conditions differ between these reservoirs, not least in length but also in temperatures and predator communities. Relatively cold conditions, e.g., in Cougar and Green Peter, result in juveniles growing slower and being at risk of predation for longer (Zabel et al. 2015). Survival in these reservoirs may be similar to reservoirs where there is higher predation mortality, e.g., Detroit, but where warmer temperatures result in faster growth to lengths >100mm where predation is less of an issue (Zabel et al. 2015).

Kock et al. (2019) estimated monthly survival rates of fry in Lookout Point reservoir between April and October 2017. They found highest mortality early in summer when juvenile Chinook salmon were smaller. However, this study was not able to estimate survival of older and larger life stages, i.e., subyearlings, that moved into or remained in the reservoir in autumn and overwintered in the reservoir. More studies are needed to determine reservoir survival rates of each life stage in each reservoir.

Due to the lack of knowledge on reservoir survival, for all sub-basins we applied the distributions for reservoir survival rates from Zabel et al. (2015) for fry-subyearling and subyearling-yearling transitions. These distributions resulted from expert workshops which took place prior to the 2015 COP and considered reservoir-specific conditions. As discussed above, without empirical studies to refute the hypothesis, we assumed that reservoir survival rates of juveniles were invariant to the downstream dam passage measures applied.

We also applied the in-reservoir movement proportions for each reservoir determined during the 2015 COP expert workshops (Zabel et al. 2015). These determined the proportions of each life stage (fry, subyearling, yearling) in the reservoir that would move to the forebay and be available to pass the dam, with the remaining proportions residing in the reservoir and recruiting into the next life stage (Figure 2.2.3). The expert workshop values considered analyses of below dam RST data showing differences between the above dam RST data, which we confirmed with our own RST data analysis, but also knowledge obtained from studies of juvenile Chinook salmon distribution in some of the Willamette Valley reservoirs (Monzyk et al. 2011b, 2012, 2013, 2014, 2015). These movement proportions varied between reservoirs. We assumed that they were invariant to the downstream dam passage measures applied.

In sub-basins with two dams, i.e., South Santiam and Middle Fork, we assumed that juvenile Chinook salmon originating above and passing down through the upper dams, i.e., Green Peter and Hills Creek dams will try to pass directly through the lower dams, i.e., Foster and Lookout Point dams without stopping. This can be visualised using Figure 2.2.3, as the numbers of each juvenile migrant type in the tailrace of the upper dam would join with the same juvenile migrant type in the forebay of the lower dam (also see model equations in Appendix A). The lower dam will have its own reservoir movement proportions and survival rates that would result in the numbers of juvenile migrant types present that the upper dam populations would join. This direct movement to the lower reservoir forebay adds the assumption that smolting starts once movement downstream occurs. This assumption is supported by a previous study of gill Na+K+ ATPase activity that suggested juveniles passing the dams are undergoing smoltification and are thus more likely to emigrate than reside in lower reaches in the sub-basins (Romer et al. 2013b). In addition, it has been noted prior to dam construction that few juvenile Chinook salmon migrate in the Lower Willamette River during late spring and summer months (Dimick and Merryfield 1945). Most migration out of sub-basins would thus occur in spring and fall (see Appendix A), which we confirmed with analysis of PIT tag detections at Sullivan Dam Juvenile Bypass Facility from various studies conducted in the Willamette Valley. In the Columbia River estuary, a peak in 'subyearling' (i.e., age 0.0) Chinook salmon is observed in June (Weitkamp et al. 2012), which would represent fry migrants in our model. This supports that any fry migrants that pass dams in spring are more likely to continue downstream during the spring and smolt than reside in lower reaches over the summer, though we note due to their size there is limited empirical information from PIT tag data on Chinook fry migration.

We did not account for additional reservoir mortality of juvenile migrant types in the lower reservoir, unless they were unable to pass as determined by the dam passage efficiency and transitioned to the next life stage. In absence of data to the contrary, we assumed that juvenile migrant types from above each dam in these sub-basins will experience the same downstream migration conditions below dams and in the marine environment.

Total dissolved gas

Dam operations can lead to gas supersaturation in the river downstream of dams. Total dissolved gas (TDG) at levels that are too high can result in gas bubble trauma in both juvenile and adult salmon, which can affect survival. Gas bubble trauma occurs at TDG levels around 110%, but the effects are alleviated by the depth at which fish are found as pressure causes the bubbles to shrink. For every 1 m depth, there is around a 10% reduction in TDG compared to at the surface where it is measured, so exposure to TDG supersaturation depends on whether deeper water refuges are unavailable and if migration is delayed by passage structures to result in sustained exposure to high levels of TDG. There have been few studies examining the effect of TDG on in-river survival. Beeman and Maule (2006) performed an in-river study with tagged salmon to show that, on average, fish experienced an 18.6% reduction in TDG relative to TDG river measurements due to the average depth at which fish were found, which in the Snake River the depth during migration were >1.5m (Chinook salmon) and >2.0m (steelhead). No TDG related mortality data exists for the Willamette but previous life cycle modelling suggested that mortality from TDG was overestimated (Zabel et al. 2015). Although under some EIS alternatives, e.g., those involving spill, there were many days within each year where TDG was predicted to be >110%, there were relatively few days where TDG was >120% (under Alt 3a, mean of 39 days yr-1 at Big Cliff, zero days yr-1 at Detroit). We made the assumption that most fish would have access to water refuges of depth >1 m so that TDG effects would be minimised via depth compensation. In the absence of explicit data on the effects of TDG on fish in the Willamette sub-basins, we therefore did not incorporate it as a mortality factor in the life cycle model. Population rates were therefore assumed to be insensitive to TDG under the different EIS alternatives.

Downstream survival

The final component of juvenile Chinook salmon life stages in freshwater is survival from the dam tailraces in each sub-basin downstream to Willamette Falls, when the juveniles were assumed to smolt. Given the different distances and habitat conditions below the dams, we aimed to apply different downstream survival rates in each sub-basin. We made use of PIT tag studies conducted in some of the sub-basins that had releases of juvenile Chinook salmon into dam tailraces to estimate the survival rate between the tailrace and Sullivan Juvenile Bypass Facility (SUJ) at Willamette Falls, assuming that survival of the released fish would represent survival during downstream migration. We used a Bayesian Cormack-Jolly-Seber (CJS) model to estimate the survival parameters (Appendix C). As this model produced estimates of apparent survival rate, we applied adjustment factors to account for effects of tag loss and mortality, and also to correct for origin, as most of the studies involved releases of hatchery-origin juveniles (Appendix I). We assumed the posterior estimates obtained reflected long-term average values and applied them to all years in the model, with uncertainty accounted for by the posterior distribution.

The PIT tag studies used all involved releases of subyearling Chinook salmon, meaning the estimates we obtained related to subyearlings only. Due to their size and vulnerability to predation, it was assumed that fry would have lower downstream survival, and yearlings would have higher downstream survival. To overcome this, we examined the downstream survival rate estimates applied in each sub-basin for the 2015 COP (Zabel et al. 2015) and used the differences between fry, subyearling and yearling downstream survival to scale the subyearling estimates obtained from our analyses to determine estimates for fry and yearling downstream survival. Given the subyearling survival rate estimates, fry survival rates were thus around 0.3-0.4x of these, while yearling survival rates were around 1.3-1.4x. The studies were available for North Santiam, McKenzie and Middle Fork. While PIT tag studies were conducted in the South Santiam, the number of fish released were far fewer and downstream detections of PIT tags were minimal which led to CJS models not converging. We instead applied the North Santiam estimate, but used the relative difference between the downstream survival rate estimates applied in the 2015 COP (Zabel et al. 2015) for North Santiam and South Santiam to scale the North Santiam posterior estimate and provide an estimate for South Santiam.

Marine Stage

The evaluation of the EIS alternatives required, among other things, an evaluation of their potential effects on the long-term frequency distribution and abundance of juvenile stage life history types of spring Chinook salmon and abundance of natural origin spawners in the Upper Willamette sub-basins. Maintaining diversity in juvenile stage life history types is desired because this diversity may contain important components of a population's genetic diversity and may also contribute to population resilience when there is high interannual variability in freshwater and marine survival rates (Griffiths et al. 2014; Schroeder et al. 2016; Price et al. 2021; Welch et al. 2021). Predictions of the potential effects of EIS alternatives may depend on assumed future values for survival rates from natural mortality and fishing mortality at sea which could vary systematically in future years. Thus, predicting long-term effects of policy options on the frequency distribution of migrant types requires an age structured population dynamics model that explicitly tracks cohorts of juveniles (distinguished by freshwater life histories or migrant types) from generation to generation and predicts the abundance and frequency distribution of these migrant types in future generations under the different EIS alternatives. The model must also account for plausible future scenarios for marine survival rates, among other things. In addition, because it is known that maturation in Chinook salmon commonly occurs from ages three to six, the age-structured population dynamics model also requires estimates of fraction maturing at age to predict the abundance at age of spawners in each year.

The IPA LCM includes equations for the marine life stage of Upper Willamette Chinook salmon that predict the abundance of fish surviving and maturing at age for each of the juvenile migrant types. Figure 2.3.1 shows a representation of survival and maturation events for a cohort that becomes smolts as so-called yearlings (i.e., about one year after becoming fry in the previous spring). The model predicts surviving abundance at age using a discrete time step formulation. After egg deposition (at age A=0), smolts move into the ocean at the right age according to their life history type (e.g., A=1.5 in the case of yearling smolts). Then, the following equations predict abundance at age A+1 in a given year from smolt to spawning adults, starting with smolts:

(Equation 2.3-1) $N_{3,sea} = N_A * (1-P_2) * S_{smolt,A} * (1- U_{2,sea})$ {for smolts at age A that remain

at sea through to age 3
(Equation 2.3-2) NA+1, spawn = NA * PA * SA,T * (1- UA,T) * (1-PSM) {for fish at sea at age A that
spawn at age A+1
(Equation 2.3-3) NA+1, sea = NA, sea * (1-PA) * SA,sea * (1- UA,sea) {for fish at sea at age A that
remain at sea

where

NA,smolt is the abundance of yearling smolts that become smolts A years after their parents spawned,

Ssmolt,A is the survival rate from natural mortality for a smolt of A up to age 3,

PA is the proportion of fish maturing at age A

NA,x is the abundance at age A at stage x=sea or x=spawning,

UA,Y is the annual harvest rate on age A at stage Y=sea or Y=terminal fishery (T),

SA,Y is the annual survival rate from natural mortality, MA,Y, on age A ending at stage Y=sea or Y= the terminal fishery (T),

SA,Y = exp(-MA,Y),

MA,Y is the average instantaneous rate of natural mortality for Chinook salmon of age A at life stage Y,

PSM is the total pre-spawn mortality rate, i.e., total fraction dying from prespawn mortality below and above dams.

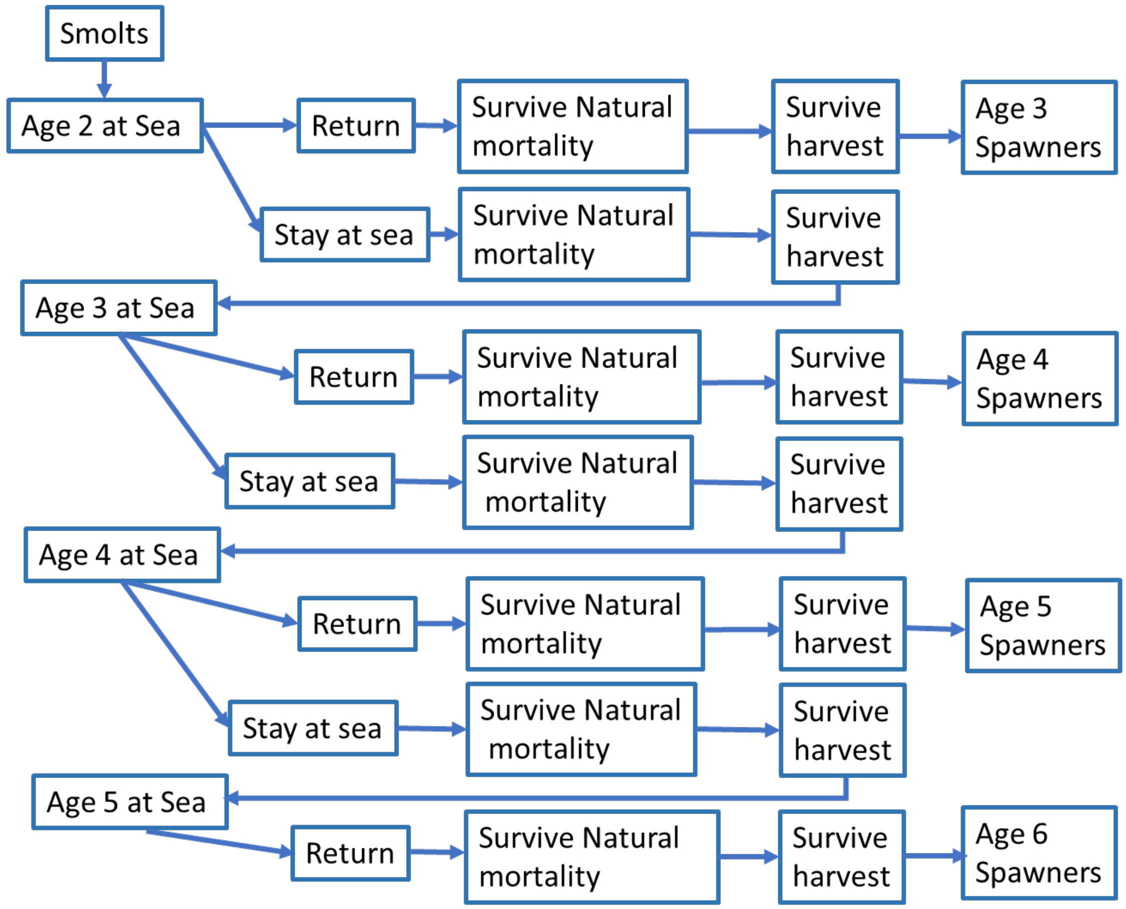

Figure 8-6. Graphical representation of the marine stages and processes for a yearling smolt (i.e., one smolting one year after hatching) to returning adult life stages of spring Chinook salmon in the Upper Willamette River

The derivation of long-term average parameter values for the marine stage used in the LCM, i.e., U, S, and P in the above equations, was not straightforward for the following reasons. While it would be desirable for parameter estimation to be able to fit the LCM to existing coded wire tag (CWT) and PIT tag data to estimate model parameters, this was not possible. Firstly, CWT data set used in Chinook Technical Committee (CTC) stock assessments (CTC 2021) to estimate stage-based fraction of hatchery produced Chinook salmon harvested at age over years were not available for LCM model fitting and estimation. The CTC has conducted stock assessments that aggregate the release and recapture records of CWT tagged fish in both sea and river fisheries from Upper Willamette Chinook hatchery releases from the different sub-basins. The CTC has provided estimates of 1) the fraction of tagged fish harvested at age over years and 2) survival rates from hatchery release to age 3 for the aggregate of hatchery released Chinook salmon in the Upper Willamette River basin. The total cumulative survival rate from age 0 to 3 estimates also provided by the CTC (2021) however include the survival rates of juveniles downstream of the hatcheries and also the sub-adult marine component. In contrast, the LCMs that we have developed are specific to natural-origin and above-dam populations of spring Chinook salmon in each sub-basin in the Upper Willamette; as such, it would be appropriate to adjust CTC parameter values to account for the different fish populations of interest.

Secondly, the CTC stock assessments apply specific assumptions about long-term average natural mortality rate at age for the marine stages of Upper Willamette hatchery chinook salmon when estimating fraction of fish harvested at age over years. To try to maintain consistency with the CTC stock assessment, we believe that it is appropriate to use the values for natural mortality rate at age (MA,Y) that the CTC has applied in its recent stock assessments of upper Willamette hatchery Chinook salmon as initial values for modelling purposes. However, the CTC does not provide direct estimates of MA,Y for the smolt to age 3 stage and the LCM thus requires derivation or estimation of values for this parameter for fry, subyearling and yearling smolts in the four Upper Willamette River sub-basins.

Thirdly, using a Bayesian Cormack-Jolly-Seber (BCJS or CJS for short) methodology we have estimated average juvenile downstream survival rates and smolt-adult survival rates for hatchery-reared spring chinook salmon in each of the four sub-basins of interest using release and detection records from PIT tag studies in each of the sub-basins (see Appendix C). The smolt-adult survival rates that we have estimated, however, are total cumulative survival rates from both natural and fishing mortality at sea and are for adults that predominantly return after a four-year life cycle (i.e., four years passing between egg to spawning adult). It is not possible to jointly estimate natural mortality rates and fishing mortality rates by year separately from the PIT study records since the PIT tag detections are not available at-sea, only from the Sullivan Juvenile (SUJ) PIT tag detection facility for juveniles going downstream and from a facility at Willamette Falls (WFF) for returning adults.

It was thus not possible to fit the LCM to the CWT and PIT tag data for parameter estimation since the CWT records were not available and the PIT tag records were not in a form that readily allowed the LCM model to be fitted to them for parameter estimation. We however used results from the CTC stock assessments, our BCJS estimations and other data to derive parameter values for the at sea life stages of the LCM (see Appendix D).

Adult Freshwater Stage

This section describes the life cycle model components related to adult Chinook salmon that have returned to the river and have passed upstream of Willamette Falls via the Willamette Falls Fishway (WFF).

Pre-spawn mortality

Mortality of adult Chinook salmon before reproduction (i.e., pre-spawn mortality) can be significant to population viability. There are two main types of pre-spawn mortality (PSM): 1) en route migration mortality, experienced when the adults return to the river and migrate upstream to spawn, and 2) onsite mortality, when adults die on the spawning grounds before reproduction (Keefer et al. 2017; Bowerman et al. 2021). We incorporated these components into the life cycle model as below dam (en route) and above dam (onsite) PSM.

In the mainstem Willamette, the en route mortality rate of adult from Willamette Falls to spawning tributaries has been estimated by radio telemetry (Keefer et al. 2017). That study found that PSM was related mainly to body condition and injuries sustained during the upstream migration, e.g., from marine mammals, than it was to stream temperature, sex or origin (natural or hatchery). We used the results from Keefer et al. (2017) to parameterize a beta distribution for below dam PSM, which had a mean of 0.165 (alpha = 130.5, beta=655). All sub-basins included the same uncertainty (see Appendix A).

Onsite PSM has been studied through carcass surveys conducted by ODFW and others in years 2010-2018 (e.g., Cannon et al. 2011; Sharpe et al. 2013, 2014, 2015, 2016, 2017; Normandeau Associates 2019). This type of PSM has been statistically modelled in relation to temperature, which causes thermal stress, and the percentage of hatchery-origin spawners (pHOS), which may cause density-related effects (Zabel et al. 2015; Bowerman et al. 2018). Accordingly, we applied the model from Bowerman et al. (2018) to predict PSM in each year from temperature (7-day maximum of average daily temperature, 7DADM) and pHOS, with random effects of year and study site. pHOS was determined from the ratio of hatchery- to natural-origin adults outplanted above each dam in each simulated year (see Appendix A).

We obtained 7DADM temperatures from USGS gages located above dams in each sub-basin. The gage numbers were 14178000 (above Detroit), 14185000 (above Foster), 14159200 (above Cougar), and 14148000 (above Lookout Point). We used the 'dataRetrieval' package in R to download the maximum daily temperatures at each gage for the period of record covered by FBW, i.e., 1946-2019 (R Core Team 2021; De Cicco et al. 2022). Temperatures in historical years typically peaked in late-July or early-August. As the LCM timestep began from 1 September, when spawning was assumed to occur, we subset these data for temperatures between 1 May (when temperatures began to warm up) and 31 August. Some gages did not operate every year, or in every day in each year, so we calculated 7DADM temperature over the May-August period in only those years for which there were no missing data. In years without complete data, we calculated the mean 7DADM at each gage for each water year type in FBW (deficit, insufficient, adequate, abundant). Then, these water-year type means were applied to the years with missing data based on those years' water year types (Figure 2.4.1). Having a complete record of annual mean 7DADM in each year in FBW's period of record (1946-2019) was necessary because of how the LCM projects into the future. FBW parameters (DPE, DPS) are specific to a given year, as are observed temperatures. The LCM simulates future years by bootstrapping year-specific sets of FBW parameters and temperatures to maintain year effects on upstream and downstream migrations. Mean observed values for above dam (i.e., onsite) PSM estimated from spawner surveys during 2010-2016 (Cannon et al. 2011; Sharpe et al. 2013, 2014, 2015, 2016, 2017; Normandeau Associates 2019) were 0.05 (above Detroit, range 0-0.12), 0.26 (above Foster, range 0.05-0.54), 0.06 (above Cougar, range 0-0.25), and 0.35 (above Lookout Point, range 0.11-0.80). Our initial predictions using the Bowerman et al. (2018) model were all above the upper ranges of these observed values. To correct this, we investigated the effect of altering the weighting on the temperature and pHOS values by including a multiplier on the Bowerman et al. (2018) coefficients to make our PSM predictions consistent with the observed above dam values. We found that temperature alone was a much more important and so down-weighted the effect of pHOS to only 1% of the value in Bowerman et al. (2018). This may reflect that the Bowerman et al. (2018) model was based upon data from below dams, where densities of hatchery-origin adults are typically much higher than above dams, and so the importance of pHOS may appear greater than above dams. We note there were no specific models available to predict above dam PSM.

In further support of the decision to down-weight the pHOS effect was that although there is evidence of a positive relationship between PSM and pHOS (Zabel et al. 2015; Bowerman et al. 2018), the mechanism for the relationship with pHOS is unknown (Bowerman et al. 2021). Compared to natural origin fish, hatchery origin adults have been observed to have higher PSM and display different spawning distributions, migration timing, and spawn timing (Bowerman et al. 2021). These differences in behavioural and physiological traits suggest that PSM may be higher where pHOS is high simply because the hatchery-origin component of the population experiences higher mortality than natural-origin fish which are better matched to the environmental conditions. To evaluate our assumptions about PSM, we included both below dam and above dam PSM rate in our sensitivity analyses.

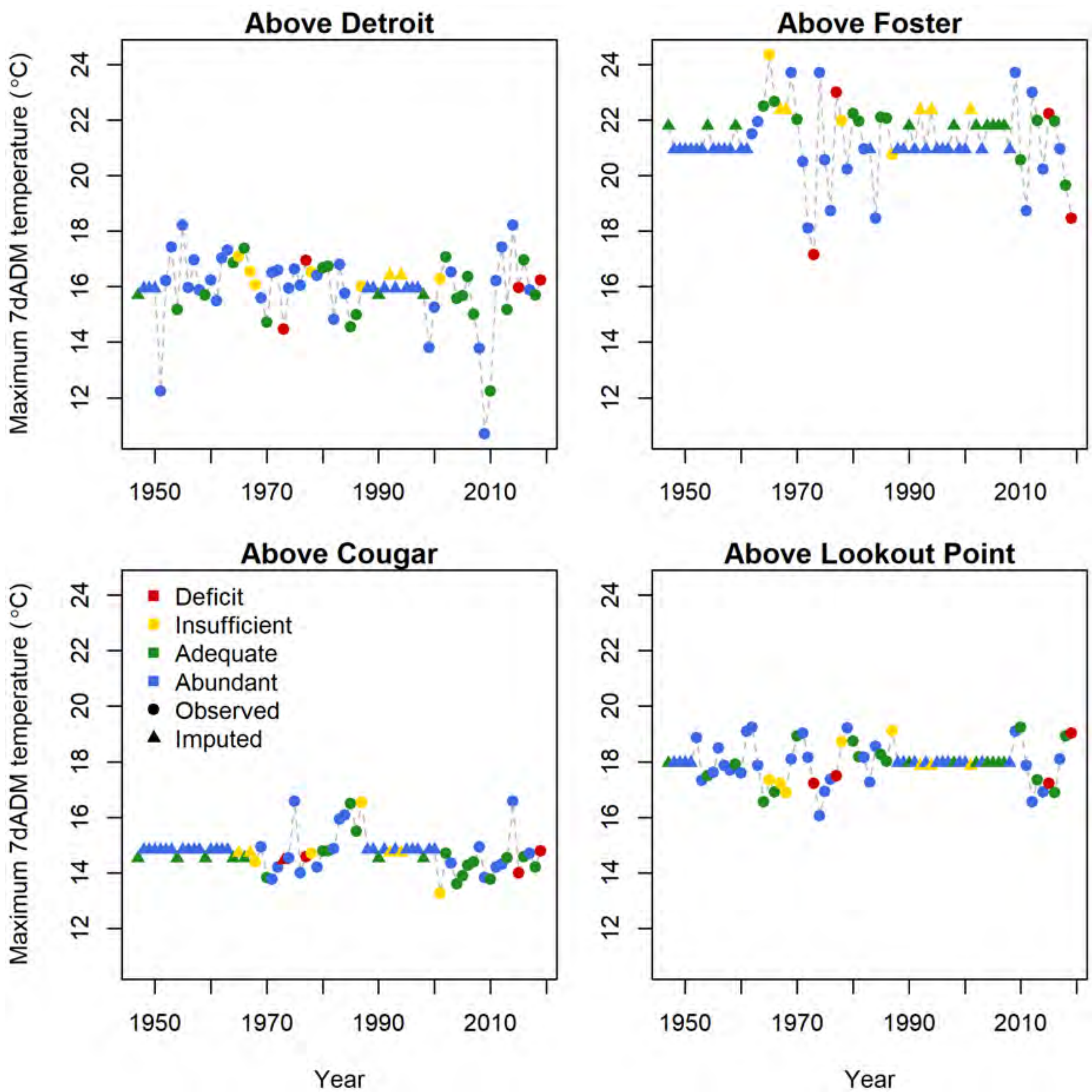

Figure 8-7. Maximum 7-day average daily maximum temperatures above dams in each Willamette sub-basin. Data obtained from USGS gages between 1947 and 2019: above Detroit (USGS14178000), above Foster (USGS14185000), above Cougar (USGS14159200), above Lookout Point (USGS14148000). 7DADM values were used to predict pre-spawn mortality above dams. Years where no data were recorded were imputed by using the mean of the water year type.

Outplanting of adults above dams

The outplanting assumptions for each sub-basin under each EIS alternative are detailed in Table 1.2.2. We assumed 100% survival during upstream passage, i.e., that there were no mortality effects associated with the trap-and-haul process. Details of outplanting locations above dams in each sub-basin were made available by USACE but were not included in our model;

outplanted adult Chinook salmon were assumed to disperse upon release throughout the available habitat prior to spawning.

Relative spawning success of HOR adults

The presence of hatchery-origin Chinook salmon has a negative effect on the reproductive performance of the spawning population, with intrinsic productivity (i.e., the α parameter in a Ricker recruitment model) of a population composed entirely of hatchery-origin fish predicted to be only 6% that of one composed entirely of wild fish (Chilcote et al. 2011, 2013). One reason for this is that hatchery-origin adults can have much lower fitness in the wild compared to natural-origin adults owing to lack of local adaptation and domestication (Araki et al. 2008). In the Willamette, hatcheries are used in each sub-basin for producing Chinook salmon for harvest and to meet conservation goals via supplementation programs, which aim to increase abundance of wild populations through outplanting of HOR adults above dams to spawn naturally. Even in 'early-generation' hatchery-origin adults, despite them coming from local- and predominantly natural-origin parents, reduced fitness effects manifest (Christie et al. 2014). As Willamette hatcheries use local-origin fish and incorporate natural-origin fish into the broodstock (ODFW & USACE 2016a, 2016b, 2018, 2019), we assumed HOR adults would have reduced fitness compared to NOR adults, and accounted for the relative reproductive success (RRS) of HOR adults in the model by penalising the number of eggs spawned by HOR adults. We assumed no interbreeding between NOR and HOR adults, such that fitness of hybrids was not accounted for in the LCM. RRS is the reproductive success of HOR adults relative to NOR adults, where NOR adults are defined as unclipped fish whose parents spawned in the wild, regardless of whether those parents are NOR or HOR.

RRS was parameterized using data from a review of studies on hatchery-origin salmon from local-origin broodstock (Christie et al. 2014). The Chinook salmon studies used were from the Wenatchee River, WA (2004-2006) and Johnson Creek, ID (2002-2005). Christie et al. (2014) used a simulation study to examine the effect of sample size on RSS, and found that estimates of RRS calculated from <250 F1 adults were not precise, as a true value of RRS=0.8 would have a 95% CI >1 with a sample size of <250 F1 adults. We therefore estimated RSS using only study years with >250 F1 adults, which provided eight male and female RRS estimates for use in the LCM. We fit a triangular distribution to the estimates from both sexes to apply in the model for RRS (see Appendix A). This had a mean of 0.53 with a range 0.39-0.84. HOR adults thus had only just over 50% of the reproductive success of NOR adults in our model. All subbasins simulated RSS using the same triangular distribution.

In applying this factor for RRS, we made the implicit assumption that HOR adults will spawn with HOR adults, leading to reduced reproductive success of the HOR population. Some studies have indicated that the negative fitness effect is reduced when HOR adults spawn with NOR adults (Janowitz-Koch et al. 2019). This might become an issue where outplanting of NOR and HOR adults occurs at the same time at the same location, in which case the RRS factor might over-penalise HOR spawning success in the LCM. However, that would suppose that spawning all occurs at the outplanting location. Assuming most disperse, it has been found that lower reproductive success of hatchery-origin fish is associated with differences in choice of spawning location (Williamson et al. 2010; Hughes and Murdoch 2017). An additional consideration for spawning success of HOR being reduced is that hatchery-origin Chinook salmon adults hauled above dams in outplanted in reservoirs did not move to the river as well as natural-origin adults (Kock et al. 2018). This would limit the spawning opportunities of hatchery-origin fish. We evaluated our assumptions about the scale of the RRS factor through sensitivity analysis.

Egg production

The numbers of eggs spawned by adults in each year depended upon the numbers of NOR and HOR adults in the spawning population, the sex ratio of the spawning population, and the origin-specific fecundity of those adults (see Appendix A). We calculated the sex ratio in each sub-basin as the mean ratio of the female returns to total returns observed at below dam adult collection facilities in each year. In each sub-basin, females comprised 40-46% of the spawning population (values in Appendix A).

In absence of Willamette-specific empirical information, we applied the natural- and hatchery-origin fecundities used by Zabel et al. (2015). Based on data available at the time of Zabel et al.'s (2015) model parameterization, there were no differences in length between NOR and HOR adults of the same age (Cannon et al. 2010). Although Zabel et al. (2015) used this observation to assume there was no difference in fecundity between similar age NOR and HOR adults, they assumed that because the average age of HOR adults returning to spawn was lower than for NOR adults, fecundities of 4,000 eggs per female for HOR adults and 4,500 eggs per female for NOR adults would be applied (Zabel et al. 2015). We assumed these fecundities were applicable in each sub-basin, except for HOR adults in the Middle Fork, for which empirical data from 2009-2015 indicated a fecundity of 3,815 eggs (R. Laird, pers. comm.). The age structure of our model determined that Chinook salmon adults returning to spawn were age-3 to age-6; although the age structure of the population differed slightly by sub-basin, we did not attempt to model differences in fecundity by age due to uncertainties in annual variability in spawner age and size.

We note that in the spawning ground surveys (Cannon et al. 2011; Sharpe et al. 2013, 2014, 2015, 2016, 2017; Normandeau Associates 2019), fish with >50% egg retention were recorded as PSM, which suggests egg deposition may result from partially spawned fish and our models would need to account for this. However, Normandeau et al. (2019) found that of 580 carcasses found in 2018, fish either retained most of their eggs or were almost fully spawned, with only 2% of the carcasses having between 30% and 70% egg retention. This indicates that partial spawning is rare, so our model assumption of no partial spawning, with fish either surviving to spawn 100% of eggs or dying before spawning, is reasonable.

Approach and methodology to fit life cycle model to data

To generate future predictions from a life cycle model, the model should be parameterized such that it can represent current conditions. For example, in management strategy evaluation (MSE; Punt et al. 2016) terminology, these are called operating models (OM). Ideally, the population model should be statistically fit to observed data, also termed conditioning the OM to the available information (De Oliveira and Butterworth 2004; Rademeyer et al. 2007). The LCM described above and detailed in Appendix A, was parameterized with specific information for each subbasin in the Willamette River (i.e., Middle Fork, McKenzie, South Santiam and North Santiam) when available. In addition, the model was informed by time series of temperature above dams and DPS and DPS estimates. To calibrate the LCM, we used records of NOR and HOR outplanted above dams and age compositions for NOR returning to each subbasin. The time series of NOR and HOR outplants were used as input data instead of predicted quantities in the LCM (i.e., the calibration model omitted the outplanting rules). The temperature above the dam predicted PSM (see Section 2.4.1), while DPS and DPE predicted the dam passage survival and efficiency for each migrating group. After fitting the model the population dynamics equations in the LCM predicted the abundance of the six migrating groups.

The model fitted likelihood components for the NOR returns and age compositions using a least-squares approach:

$$\boldsymbol{ll_{NOR}} = \frac{\sum_t^n (\ln(NOR_t^{obs}/NOR_t^{pred}))^2}{2(SD_{NOR})^2} \quad \text{Equation 2.5-4}$$

$$\boldsymbol{ll_{pa}} = \frac{\sum_y^n \sum_a^A \left(p_{a,t}^{obs} - p_{a,t}^{pred}\right)^2}{2(SD_{pa})^2} \quad \text{Equation 2.5-5}$$

where NOR_y^{pred} are the predicted NORs after en route PSM and SD_{NOR} is the SD assumed (usually 1.0) for the NOR_y^{obs} time-series. The $p_{a,y}^{obs}$ and $p_{a,y}^{pred}$ terms are observed and predicted proportion-at-age for NOR returns at year t, respectively, after en route PSM. The SD_{pa} term is the assumed SD for the proportion-at-age time series (usually 1.0).

We fitted the early marine survival parameters ($\widehat{\varphi_{0+\rightarrow 3}^{o}}$ and $\widehat{\varphi_{1+\rightarrow 3}^{o}}$), the proportion-at-age for the age structure ($\widehat{p_{0+\rightarrow 3}}$, $\widehat{p_{3\rightarrow 4}}$, and $\widehat{p_{4\rightarrow 5}}$), and annual deviation terms (e^{t-2}) for the early marine survival parameters (see Appendix A). The number of e^{t-2} estimated depended on how many years of NOR_y^{obs} time-series data were available for each subbasin.

Bayesian priors were included for the early marine survival parameters, the annual deviation terms, and $p_{0+\rightarrow 3}$ (when the initial model fit produced a zero value for this parameter which is inconsistent with the observed data).

$$\boldsymbol{p(\varphi_{0+\rightarrow 3}^{o})} = \frac{\left[\ln(\widehat{\varphi_{0+\rightarrow 3}^{o}}) - \ln(\mu_{\varphi_{0+\rightarrow 3}^{o}})\right]^2}{2(SD_{\varphi_{0+\rightarrow 3}^{o}})^2} \quad \text{Equation 2.5-6}$$

$$\boldsymbol{p(\varphi_{1+\rightarrow 3}^{o})} = \frac{\left[\ln(\widehat{\varphi_{1+\rightarrow 3}^{o}}) - \ln(\mu_{\varphi_{1+\rightarrow 3}^{o}})\right]^2}{2(SD_{\varphi_{1+\rightarrow 3}^{o}})^2} \quad \text{Equation 2.5-7}$$

$$\boldsymbol{p(e^t)} = \sum_t^n \frac{\left[\ln\left(\widehat{e^{t-2}}\right)\right]^2}{2(SD_{e^t})^2} \quad \text{Equation 2.5-8}$$

$$p(p_{0+\to 3}) = \frac{\left[\ln(\widehat{p_{0+\to 3}}) - \ln(\mu_{p_{0+\to 3}})\right]^2}{2(SD_{p_{0+\to 3}})^2}$$ Equation 2.5-9

The prior mean ($\mu_{\varphi^o_{0+\to 3}}$) and prior SD ($SD_{\varphi^o_{1+\to 3}}$) for the early marine survival parameters were subbasin-specific and described in Appendix D.

For the first year, the age structure was initialized using the observed proportion of returning adults spawning at age above dams ($s_3, \ldots, s_6$; e.g., see Appendix A). The following objective function was minimized in Microsoft Excel using SOLVER:

$$ObjFun = ll_{NOR} + ll_{pa} + p(\varphi^o_{0+\to 3}) + p(\varphi^o_{1+\to 3}) + p(e^t) + p(p_{0+\to 3})$$
Equation 2.5-10

Overall, the time series of NOR and HOR and age composition were short in all sub-basins (see results sections of each subbasin for the NOR and HOR time series fitted to the data). The prior mean and prior SD values are found in Table 2.5.1.

Table 8-7. Bayesian prior mean and prior SD used in the calibration model for each subbasin in the Willamette River.

Parameter	Value (Middle Fork, McKenzie, South and North Santiam)
SD_{NOR}	1.0; 0.4; 1.0; 1.0
SD_{pa}	1.0; 1.0, 1.0; 1.0
$\mu_{\varphi^o_{0+\to 3}}$	0.047; 0.0161; 0.010; 0.0132
$\mu_{\varphi^o_{1+\to 3}}$	0.399; 0.138; 0.087; 0.1131
$SD_{\varphi^o_{0+\to 3}}$	1.0; 0.59; 0.59; 0.59
$SD_{\varphi^o_{1+\to 3}}$	1.0; 0.59; 0.59; 0.59
SD_{e^t}	2.0; 2; 2.0; 2.0
$\mu_{p_{0+\to 3}}$	-; 0.027; -; -
$SD_{p_{0+\to 3}}$	-; 0.012; -; -

Assumptions made and uncertainties modelled in the life cycle model

Juvenile freshwater stage	Assumption made	Assessment of assumption
Egg-fry stage follows a Beverton-Holt function	The Beverton-Holt productivity term (for egg-fry survival rates) comes from a meta-analysis by Zabel et al. (2015), as there are no direct estimates for the Willamette River. The capacity term (b parameter) was equal to the number of eggs and is a function of spawning habitat capacity estimates (Zabel et al. 2015). Using eggs implies density dependence as a limited amount of habitat is available for egg deposition. As spawner density increases, redds will become superimposed, or spawning will occur in a less favourable habitat. When available, parameter values are basin-specific.	Probability distribution on Beverton-Holt parameters. Sensitivity analysis on egg-fry survival rates.
There are six juvenile migrant types above dams within three main groups (fry, fall subyearlings, yearlings)	We assumed that there are six juvenile migrant types within the three main life stages (fry, subyearling, and yearling) based on when they migrate from natal streams into reservoirs and when they can pass the dams and are able to smolt (Figure 2.2.3). These are fry, SresS (subyearling that spent summer in the reservoir), Sstr (subyearling that spent summer in the natal stream), YresSW (yearling that spent summer and winter in the reservoir), YresW (yearling that spent only winter in the reservoir), and Ystr (yearling that spent summer and winter in the natal stream). Schroeder et al. (2016) study in the McKenzie sub-basin identifies six analogous juvenile Chinook salmon life histories based upon movers and stayers below dams. There are no direct estimates of the proportions of different migrant groups in fry produced above dams. We adopt the split proportions from Zabel et al. (2015). Splits for juvenile migration groups above dams are invariant to downstream dam passage measure (there are no studies on how a downstream dam passage measure can modify migrant types/migrant proportions). When available, parameter values are basin-specific.	Sensitivity analysis of splits in different migrant groups

Survival rates of juvenile migrant types above dams	There have been few studies on in-reservoir survival in Willamette reservoirs. Kock et al. (2019) estimated monthly survival rates of fry in Lookout Point reservoir between April and October 2017. Thus, we assumed juvenile survival rates from Zabel et al. (2015). Also, we assumed that the reservoir survival rate of juveniles is invariant to the downstream dam passage measure applied. There are no studies on how a dam passage measure can modify juvenile survival rates. When available, parameter values are basin-specific.	Probability distribution on reservoir survival rates for migrant groups. Sensitivity analysis on survival rates for more relevant juvenile migrant types
In two-dam models, juveniles originating above and passing down through Hills Creek and Green Peter will try to pass directly through downstream dams without stopping	Assumes smolting starts once movement downstream. It assumes that the numbers of each juvenile migrant type in the tailrace of the upper dam (e.g., Hills Creek) will join with the same juvenile migrant type in the forebay of the lower dam (e.g., LOP). Thus, juvenile migrant groups above each dam will experience the same conditions in downstream migration below dams and in marine environments. Splits between migration groups were taken from Zabel et al. (2015). When available, parameter values are basin-specific.	None.
DPS and DPE	The DPS and DPE values for fry, subyearling and yearling come from FBW outputs. Bootstrapped time-series of DPS and DPE (historical years) were drawn paired with above dam temperature. DPS and DPE values are basin-specific.	Sensitivity analysis for more relevant juvenile migrant types
Downstream survival below dam to Willamette Falls	There are no specific studies to estimate downstream survival by migrant types, particularly for fry (too small for tagging). Therefore, when PIT tag data were available, Cormack-Jolly-Seber methods were used to estimate release-smolt survival. Most data releases were subyearlings. For fry and yearlings, downstream survival estimates were scaled (from subyearling estimates) based upon [survival] differences assumed in Zabel et al. (2015). PIT tag-based estimates were adjusted for tagging loss and tagging-induced mortality. We assumed that survival estimates from fits to PIT tag data reflect long-term average values. Parameter values are basin-specific.	Probability distribution for downstream survival of each migrant group. Sensitivity analysis for more relevant juvenile migrant types
Adult marine stage	[…]	[…]

Adult marine survival in early ages (to age-3) and older ages	The LCM was run using historical outplant (NO+HO) data and fitted to observed NOR return data to obtain early age marine survival estimates and annual deviates. We bootstrapped time-series of early marine survival estimates using annual deviates estimated during calibration for future early marine survivals. Parameter values are basin-specific. Age compositions, mean marine survivals for older ages (i.e., age-3+), and PIT-tag based estimates of smolt-to-adult survival rate were used to construct priors for the early marine survivals in the model calibration. We assumed that the early marine survivals reflect long-term average values (see Appendix D for details). Marine survivals for older ages were taken from CTC reports (CTC 2021) and were assumed to be fixed values and time-invariant in the LCM.	Bootstrapped time-series of early marine survival estimates. Sensitivity analysis for early ages (to age-3).
At sea fishing mortality rates and incidental mortality rates in terminal fisheries	We assumed that the at-sea fishing mortality rates derived from CTC (2021) reflect the long-term average mortality rates. These were used to represent the present and future fishing mortality rates in the LCM, as there is no understanding of how harvest rates may change. Fishing mortality rates were fixed and time-invariant in the LCM	None
Model age structure	LCM was fitted to observed age composition. We assumed that the observed spawner age compositions from carcass surveys reflect the population's age structure. The data are sparse, but the model fit relative supports the empirical data. Model age structure is time-invariant.	None
Adult freshwater stage	[…]	[…]
Straying	We assume no straying of returning adults. Fish return to their natal spawning habitats with no straying to the hatchery, other subbasins, or other spawning areas within each subbasin.	None
En route PSM	En route mortality depended more on body condition/injuries than stream temperature and sex or origin (natural or hatchery). We used Keefer et al. (2017) results to parameterize a beta distribution for below dam PSM, which had a mean of 0.16. All sub-basins used the beta parameter distributions	Probability distribution for en-route PSM. Sensitivity analysis from 0 to 1.

Onsite PSM	We used the Bowerman et al. (2018) model to predict PSM from temperature (7-day maximum of average daily temperature, 7DADM) and pHOS. pHOS was determined from the hatchery ratio to natural-origin adults outplanted above each dam in each simulated year. Parameter values are basin-specific (i.e., 7DADM).	Probability distribution for onsite PSM. Sensitivity analysis from 0 to 1.
Relative spawning success (RSS) of HOR adults	It was assumed that hatchery-origin adults have much lower fitness in the wild than natural-origin adults due to a lack of local adaptation and domestication (Araki et al. 2008). RRS was parameterized using data from a review of studies on hatchery-origin salmon from local-origin broodstock (Christie et al. 2014). Relative spawning success of HOR adults relative to natural origin adults follows a distribution ranging between 0.4 and 0.8 (mean= 0.59). All sub-basins used the beta parameter distributions.	Probability distribution for RSS. Sensitivity analysis from 0 to 1.
Egg production/sex ratio of the spawning population	We assumed the natural- and hatchery-origin fecundities and sex ratio of Zabel et al. (2015) or empirical data from sub-basins. Parameter values are basin-specific, fixed and time-invariant.	None
Hatchery and natural-origin outplanting above dams	Outplanting specifications varied by sub-basin, which EIS delined. We assumed no mortality effects associated with the trap-and-haul process or temperature stress on adults from holding. Also, we assumed that outplanted adult Chinook salmon disperse upon release throughout the available habitat before spawning.	None
Availability of hatchery origin adults for outplanting	We do not model hatchery production in this LCM. Instead, we assume that if hatchery origin adults are required to supplement outplanting (e.g., at Cougar Dam in the McKenzie subbasin), sufficient hatchery-origin returns are available according to outplanting rules specified in the HGMPs for each subbasin (ODFW & USACE 2016b, 2016a, 2018, 2019).	None
Model age structure	LCM was fitted to observed age comps. We assumed that the observed spawner age compositions from carcass surveys reflect the population's age structure. The data are sparse, but the model fit relative supported the empirical data. Model age structure is time-invariant.	None

Annual time step	We assumed the annual time step is sufficient to represent the life history processes. There is insufficient data to describe juvenile migrant types or to fit a calibration model at finer timescales. The annual timestep begins in September of the year, to represent peak spawning and to align with FBW outputs.	None

Results from each sub-basin

North Santiam

Model calibration

We initialised the model with data on the number of natural-origin return (NOR) and hatchery-origin return (HOR) adults outplanted above Detroit between 2000-2021 (Table 2.7.1). The NAA dam passage parameters obtained from FBW were applied for those specific years, together with the above dam temperature data for determining PSM. The model was then fitted to available data on NOR to Minto (2015-2021) and age composition from spawner surveys above Detroit reservoir (2011-2016).

Table 8-8. Numbers of natural-origin return (NOR) and hatchery-origin return (HOR) Chinook salmon outplanted above Detroit reservoir and the numbers of NOR to Minto adult collection facility by year. Data obtained from ODFW HRME reports (Cannon et al. 2010, 2011; Sharpe et al. 2013, 2014, 2015, 2016, 2017) and Greg Grenbemer (ODFW, pers. comm.).

Year	NOR outplants above Detroit	HOR outplants above Detroit	NOR to Minto
2000	0	933	no data
2001	0	1068	no data
2002	0	2677	no data
2003	0	2884	no data
2004	0	2475	no data
2005	0	599	no data
2006	0	1843	no data
2007	0	967	no data
2008	0	218	no data
2009	0	900	no data
2010	49	2435	no data
2011	0	148	no data
2012	0	253	no data
2013	0	1103	no data
2014	0	872	no data
2015	474	1044	647
2016	0	1238	521
2017	0	1615	502
2018	0	1004	245
2019	0	1030	788
2020	0	2614	1609
2021	0	1314	464

We note that as prior to 2015 the numbers of NOR and HOR were not reported separately, the time series of NOR was relatively short (2015-2021). To obtain a suitably large pool for bootstrapping from, we thus decided to combine the annual deviates in early-age survival rates across both the North Santiam and South Santiam sub-basins (see Appendix A for values) for the projections under EIS alternatives, as the South Santiam had a longer time series of NOR. Our decision to combine them was supported by the means and CVs in the annual deviates being similar between these two sub-basins, with a mean 0.211 (CV=2.794) from the North Santiam calibration and 0.288 (CV=2.849) from South Santiam calibration. Some annual deviates in both sub-basins were large (i.e., >1).

The calibrated model fit both NOR and age compositions reasonably well (Figure 2.7.1), except to the age-4 and age-5 spawner composition in 2015 and the large spike in NOR observed in 2020, which were almost twice that in the surrounding years. The calibrated parameter values for early-age marine survivals ($\varphi^{o}_{0+\rightarrow 3}$, $\varphi^{o}_{1+\rightarrow 3}$) and proportions maturing at age ($p_{0+\rightarrow 3}$, $p_{3\rightarrow 4}$, $p_{4\rightarrow 5}$) are shown in Table A.2 (Appendix A).

Performance Metrics for EIS alternatives

Under the NAA, the model reached an equilibrium within five years. The projected NOR were higher than recent observed NOR (Figure 2.7.2) because to maintain consistent outplanting assumptions across alternatives, the model assumed that there were 1,500 HOR adults available to outplant each year. The mean number of HOR adults outplanted over the last 10 years was actually only 1,208; if the model instead assumed this observed HOR outplant value rather than the assumed maximum 1,500 value, the projected NOR at equilibrium were similar to the mean observed NOR (results not shown).

Under NAA, where only HOR were outplanted above Detroit, due to the lower relative spawning success of hatchery-origin adults, productivity (i.e., R/S) is lower than if the natural-origin adults that returned to Minto were outplanted above Detroit (Table 2.7.2, Figure 2.7.3). Despite this, the outplanting of HOR adults appears to remove any risk of extinction under the NAA (P<QET=0), and the R/S value of 1.057 indicates the population above the dam is above replacement. There is zero natural-origin spawning occurring above Detroit under the NAA, shown by pHOS being equal to 1.0.

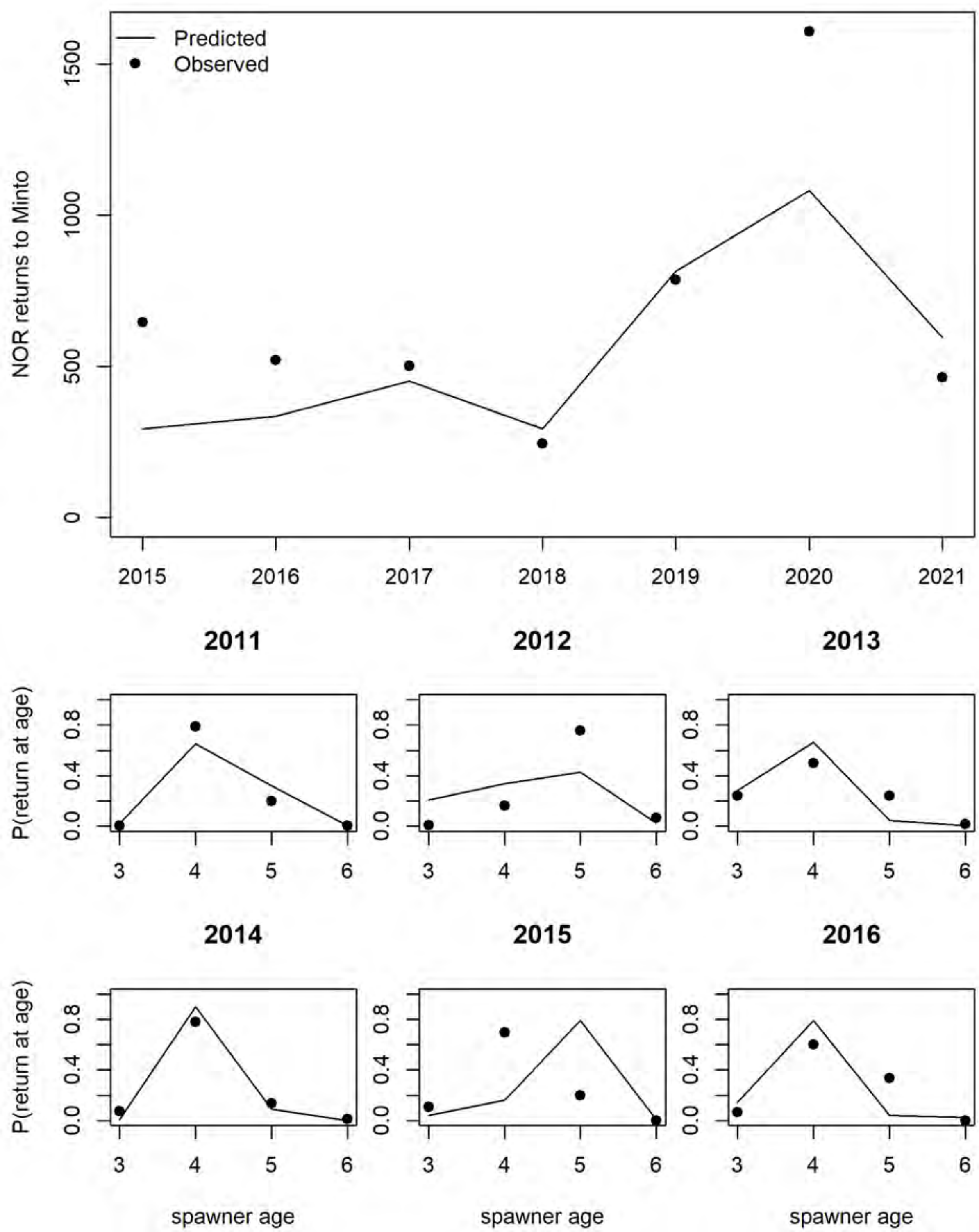

Figure 8-8. Predicted number of natural-origin returns (NOR) to Minto compared to the observed number of NOR in 2015-2021 (upper panel). Predicted spawner age composition after model calibration compared to the observed spawner age composition from spawner surveys above Detroit (lower panel).

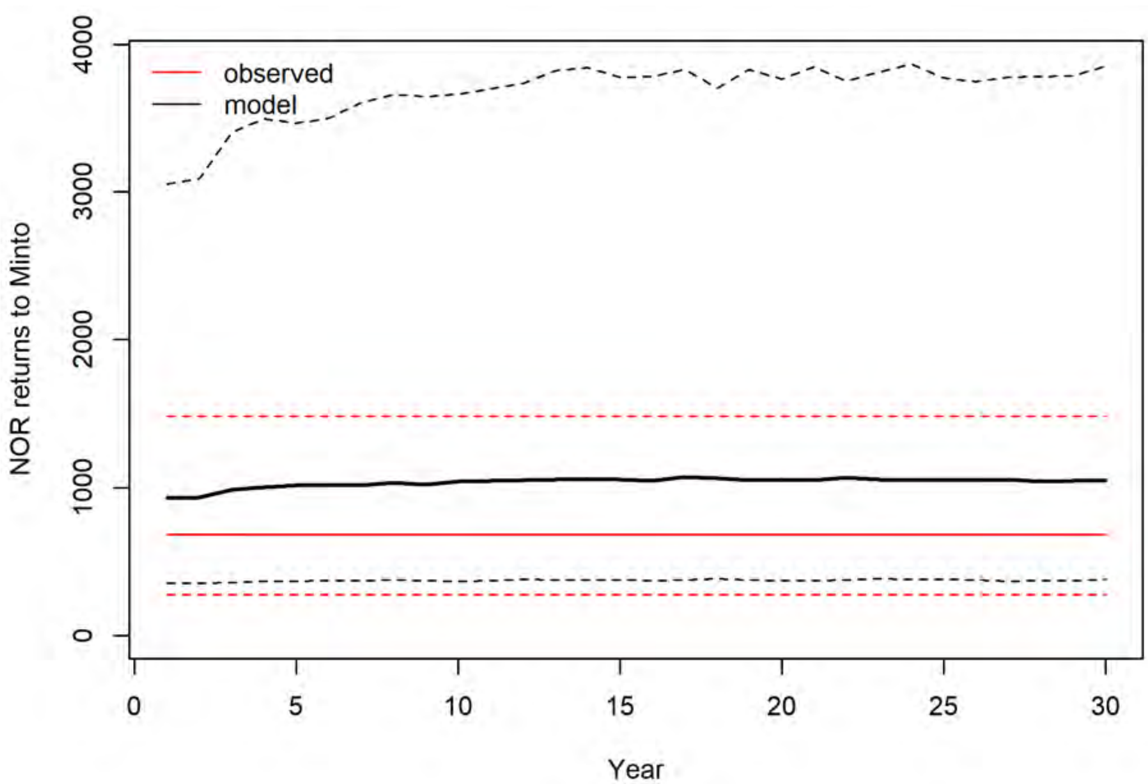

Figure 8-9. NOR returns in the North Santiam projected under the NAA, showing the median (solid black line) and 95% confidence interval (dashed black lines) from 10,000 simulation runs. The median (solid red line) and 95% percentiles (dashed red lines) of the observed NOR data (2015-2021) are shown for comparison.

Table 8-9. North Santiam Chinook salmon performance metrics under each EIS alternative. Summary statistics are medians from 10,000 simulation runs (mean for P<QET). Definitions for each performance metric are found in Section 1.3.

	EIS alternative						
Performance metric	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
NOR spawners	963	12,530	13,083	13,016	7,710	5,923	12,720
R/S	1.057	2.057	2.047	2.072	1.855	1.655	2.050
SAR	0.068	0.072	0.07	0.072	0.078	0.078	0.071
pHOS	1.000	0.000	0.000	0.000	0.000	0.000	0.000
P<QET	0.000	0.000	0.000	0.000	0.000	0.000	0.000

Under all other alternatives, due to the addition of NOR outplants permitted due to implementation of improved passage, productivity is much higher because of the relatively higher reproductive success of natural-origin spawners in addition to the improved passage survival. This means that the population increases and reaches an equilibrium determined by the logistical outplant cap of 5,428 adults within a couple of generations and there is zero risk of extinction. Due to the swift recovery of NOR spawners to numbers above 1,500, the outplanting of hatchery-origin adults is no longer required under non-NAA alternatives, shown by pHOS being equal to 0.0. NOR spawner numbers shown are post-PSM and before the outplant cap, so are higher numbers than the logistical cap, and closer to the maximum expected given spawner habitat capacity above Detroit. These numbers highlight the potential for larger natural-origin spawning populations to be achieved if the logistical capacity for outplanting of NOR adults was increased, e.g., through addition of extra staff, holding tanks, and trucks.

Population recovery is faster under Alt1/Alt2a/Alt2b/Alt4 because the structural measure applied under these alternatives (FSS) has much higher DPS. Although dam passage survival is lower under Alt3a/Alt3b that involve spill and drawdown operations, those alternatives still lead to population recovery. It is important to highlight the uncertainty associated with the performance metrics (Figure 2.7.3 and Figure 2.7.4), as under all alternatives the 95% confidence interval for R/S extends down to or below 1.0, i.e., population replacement. This indicates that despite there being minimal probability of population extinction, any future changes to the system not incorporated in this model, e.g. due to climate change, could result in poorer population recovery.

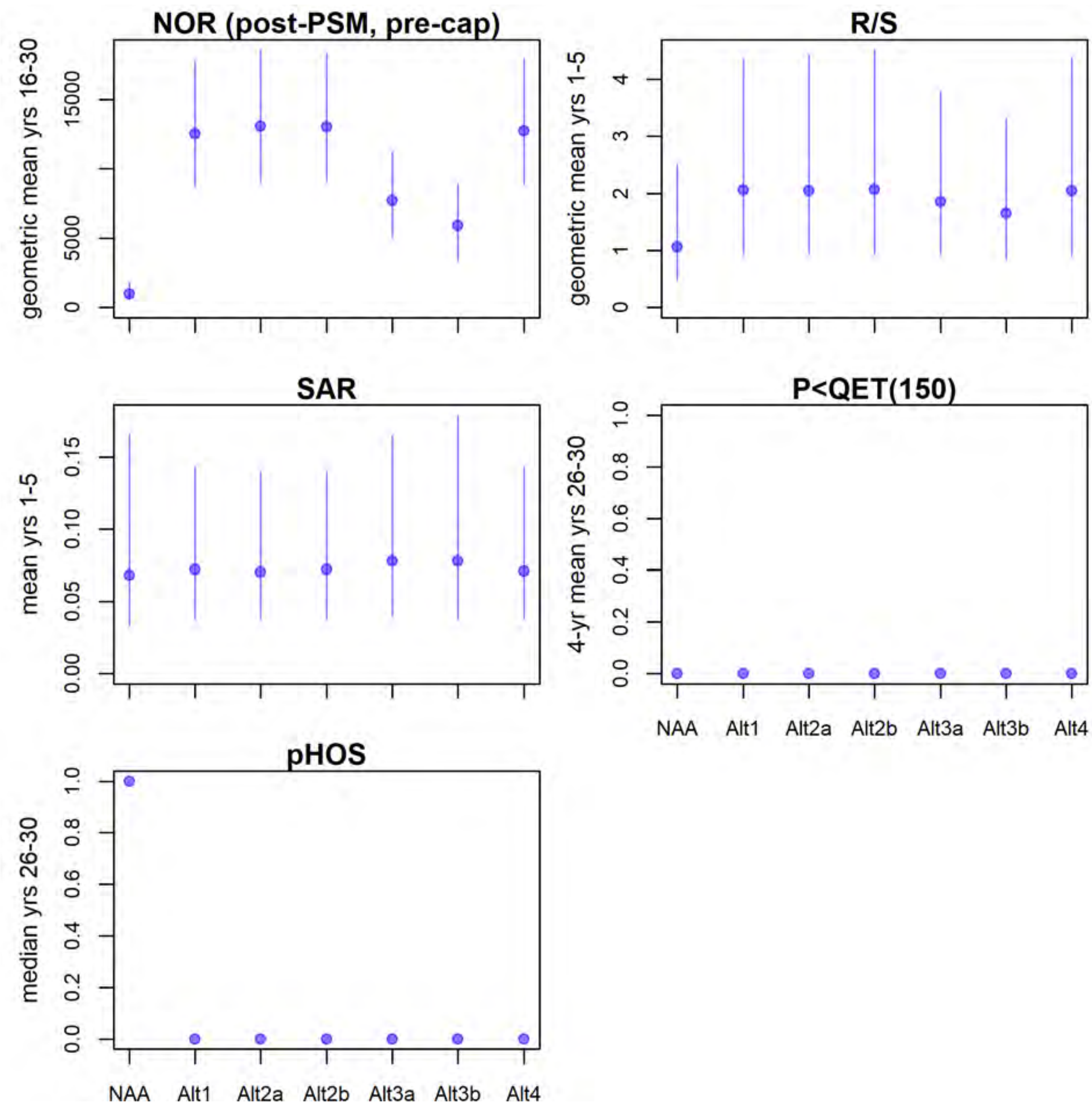

Figure 8-10. Uncertainty in performance metrics under each alternative in the North Santiam: 1) NOR spawners (post-PSM and pre outplant cap), 2) recruits-per-spawner (R/S), 3) smolt-adult return rate (SAR), 4) probability of extinction given a quasi-extinction threshold of 150 NOR spawners, 5) proportion of hatchery-origin spawners (pHOS). Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs. Error bars are not shown for P<QET, owing to its binary outcomes.

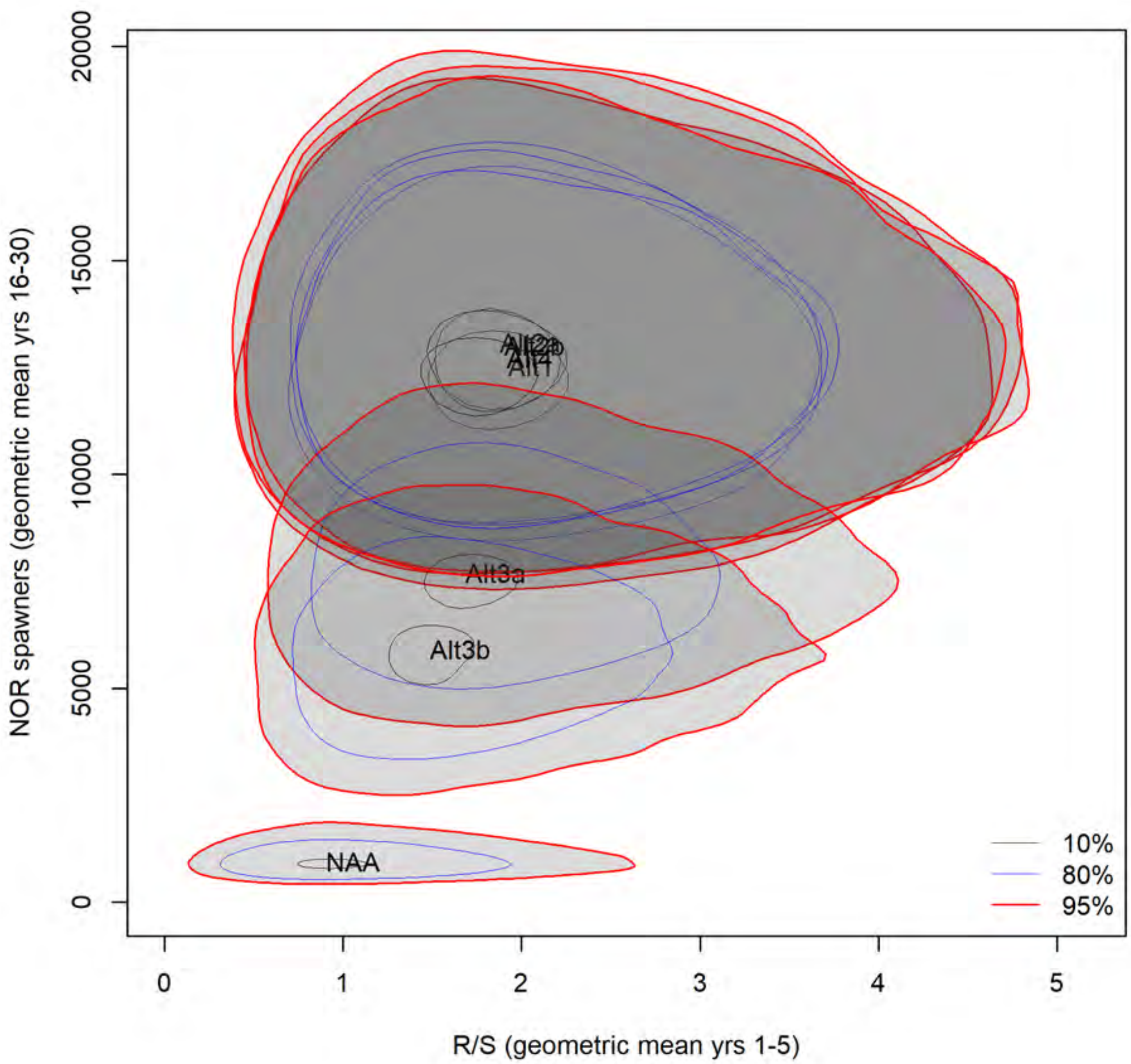

Figure 8-11. 2-D kernel density estimates (bandwidth=30) from 10,000 simulation runs to show the trade offs between NOR spawner and R/S performance metrics under each EIS alternative in the North Santiam.

The diversity of the population, as determined by relative proportions of different juvenile migrant types, was not markedly different under different alternatives (Figure 2.7.5). There were slightly fewer numbers of yearling Chinook migrating under alternative 3a and 3b, which involved operational passage measures. These results highlight that under our assumptions of river juvenile splits (Appendix A), fry migrants do not contribute to the population, as very low numbers of these returned as adults due to their much lower marine survival.

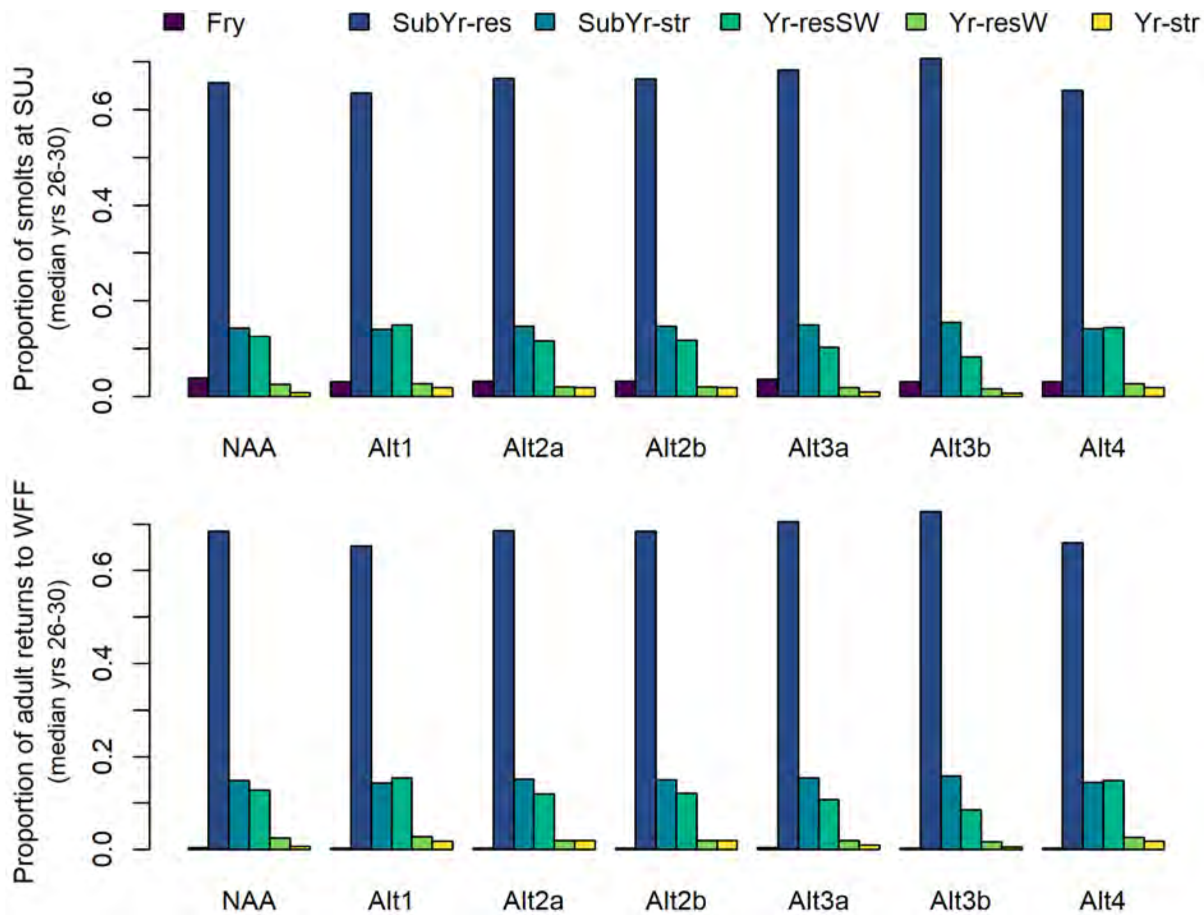

Figure 8-12. Relative proportions of different juvenile migrant types passing Detroit-Big Cliff under each EIS alternative when counted as smolts at Sullivan Juvenile Facility (SUJ) or as returning adults at Willamette Falls (WFF). The migrant types modelled were Fry, Subyearlings that spent the summer in the reservoir (SubYr-res), Subyearlings that spent the summer in the natal stream (SubYr-str), Yearlings that spent summer and winter in the reservoir (Yr-resSW), Yearlings that spent only winter in the reservoir (Yr-resW), and Yearlings that spent summer and winter in the natal stream (Yr-str).

Fry to smolt survival varied between alternatives and juvenile migrant type (Figure 2.7.6). Fry to smolt survival was lower for those migrant types which spent longer periods in the reservoir prior to passage, i.e., subyearlings that spent the summer in the reservoir and yearlings that had spent both summer and winter in the reservoir. Yearlings that only spent winter in the reservoir had comparable survival to yearlings that reared in the natal stream. Fry to smolt survival was lowest under the NAA and Alts 3a/3b, which had operational passage measures with lower dam passage survival than the alternatives with structural passage measures. The relative adult return rates were lowest for the fry migrant type, but comparable across the other juvenile migrant types, and there was minimal difference between the alternatives (Figure 2.7.7).

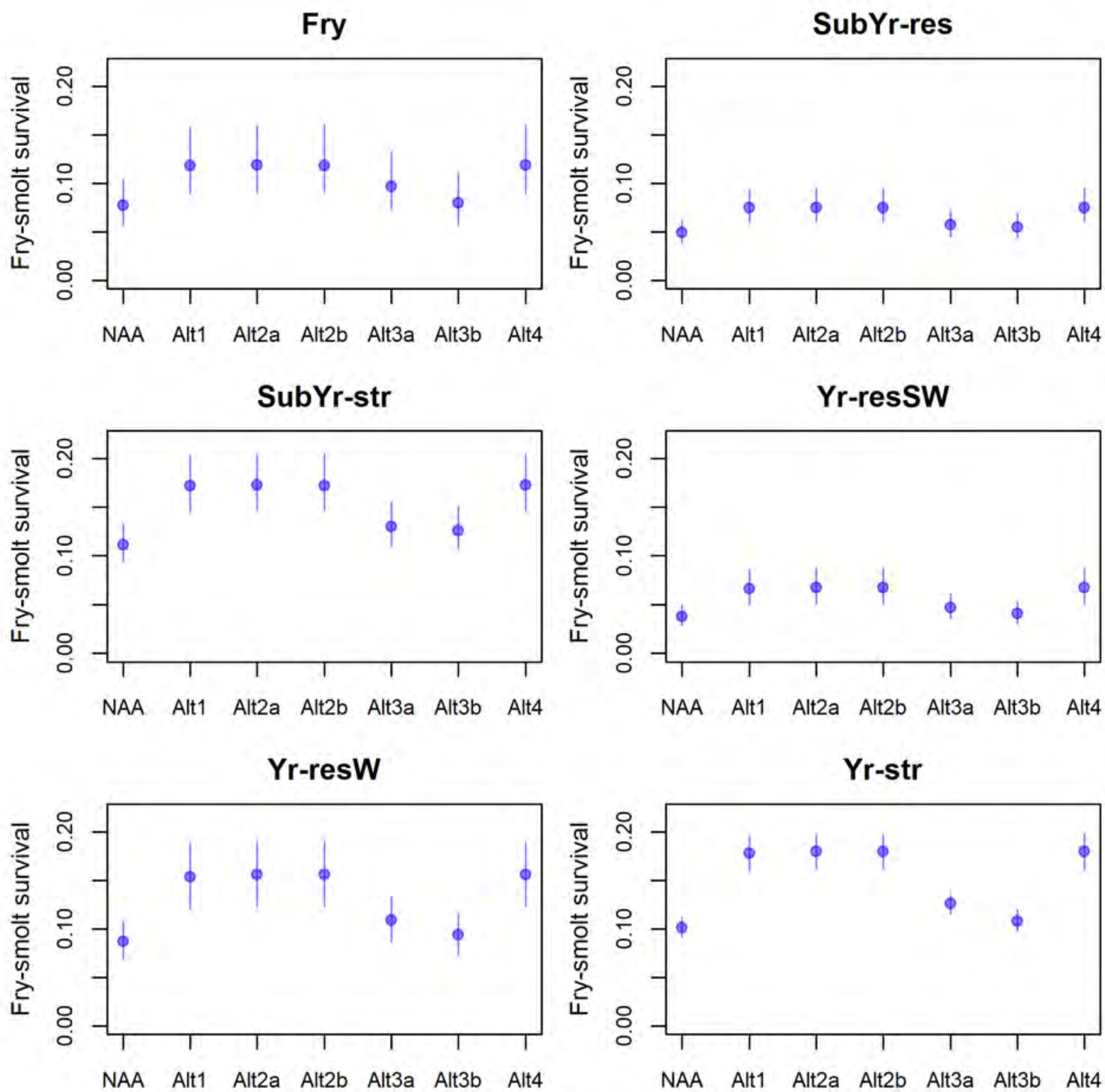

Figure 8-13. Fry to smolt survival for each juvenile migrant type under each EIS alternative. Calculated as the mean over years 1-5 of each simulation run. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs.

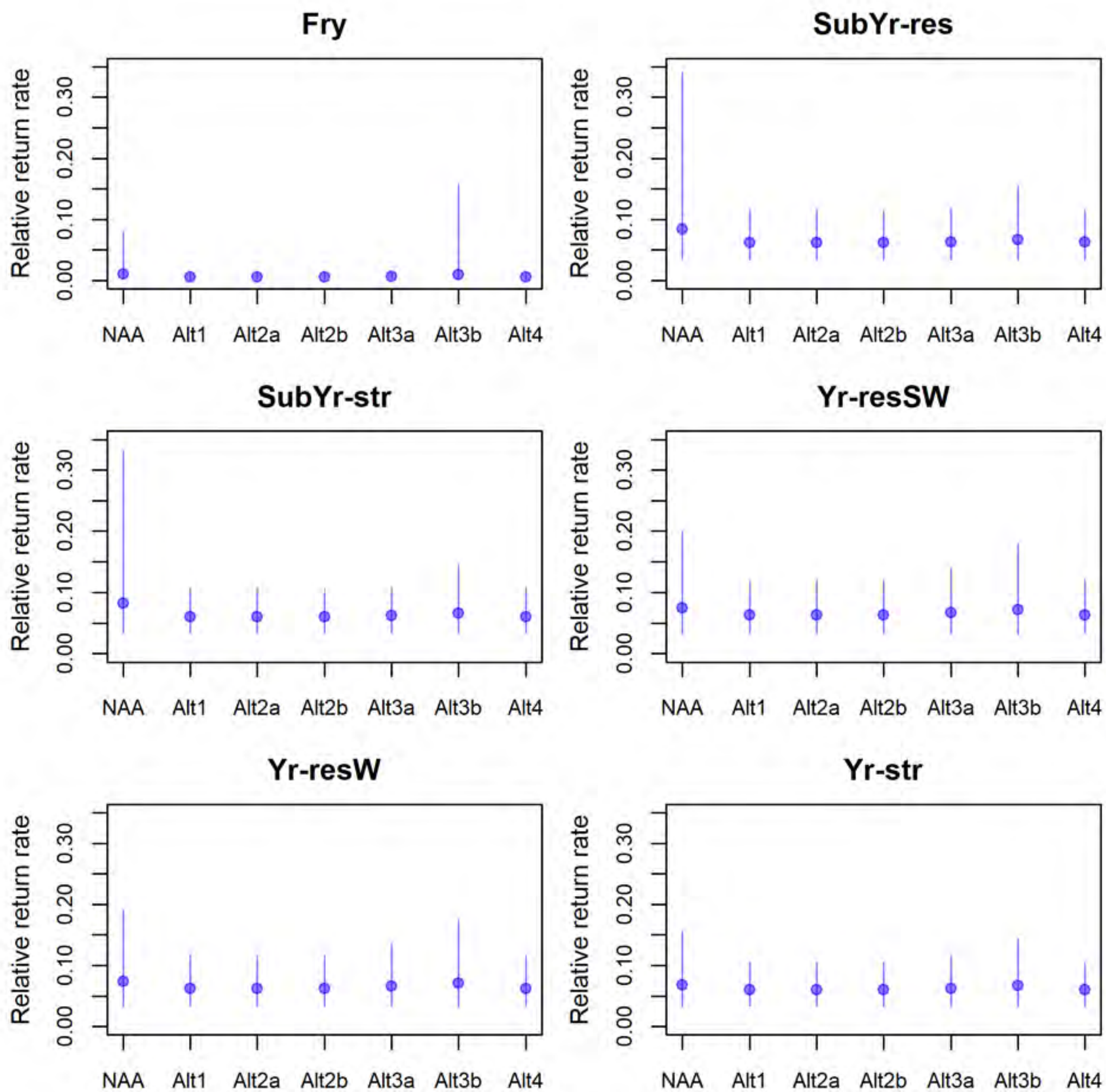

Figure 8-14. Relative return rates of each juvenile migrant type under each EIS alternative. Calculated as the mean over years 26-30 of each simulation run. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs.

Parameter Uncertainties

The LCM was probabilistic and the distributions of those parameters that had uncertainty are shown in Figure 2.7.8. The distributions for early-age marine survivals were determined by the annual deviates obtained from the model calibration, these do not appear as smooth distributions because of the relatively low number of deviates (n=23).

The uncertainty in dam passage parameters, DPE and DPS, provided by FBW varied by alternative (Figure 2.7.9). For those alternatives with structural passage measures (Alt 1, Alt 2a, Alt 2b, Alt 4), there was almost no uncertainty in DPE, which was >0.8 in each age class under each of these alternatives. Dam passage survival was close to 1 for structural alternatives. The differences in NOR spawners between alternatives are closely related to the differences between dam passage parameters.

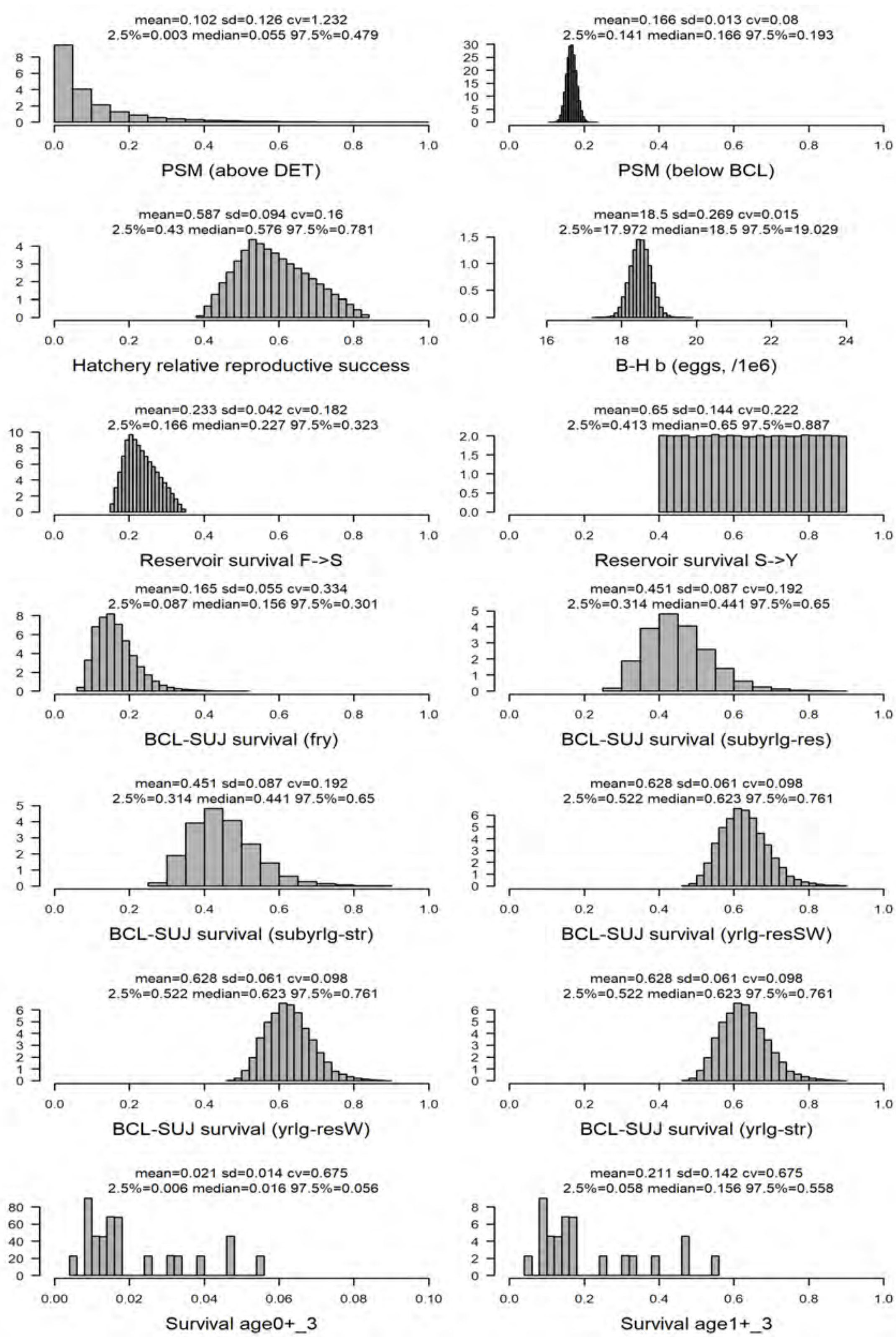
mean=0.102 sd=0.126 cv=1.232
2.5%=0.003 median=0.055 97.5%=0.479
PSM (above DET)
mean=0.166 sd=0.013 cv=0.08
2.5%=0.141 median=0.166 97.5%=0.193
PSM (below BCL)
mean=0.587 sd=0.094 cv=0.16
2.5%=0.43 median=0.576 97.5%=0.781
Hatchery relative reproductive success
mean=18.5 sd=0.269 cv=0.015
2.5%=17.972 median=18.5 97.5%=19.029
B-H b (eggs, /1e6)
mean=0.233 sd=0.042 cv=0.182
2.5%=0.166 median=0.227 97.5%=0.323
Reservoir survival F->S
mean=0.65 sd=0.144 cv=0.222
2.5%=0.413 median=0.65 97.5%=0.887
Reservoir survival S->Y
mean=0.165 sd=0.055 cv=0.334
2.5%=0.087 median=0.156 97.5%=0.301
BCL-SUJ survival (fry)
mean=0.451 sd=0.087 cv=0.192
2.5%=0.314 median=0.441 97.5%=0.65
BCL-SUJ survival (subyrlg-res)
mean=0.451 sd=0.087 cv=0.192
2.5%=0.314 median=0.441 97.5%=0.65
BCL-SUJ survival (subyrlg-str)
mean=0.628 sd=0.061 cv=0.098
2.5%=0.522 median=0.623 97.5%=0.761
BCL-SUJ survival (yrlg-resSW)
mean=0.628 sd=0.061 cv=0.098
2.5%=0.522 median=0.623 97.5%=0.761
BCL-SUJ survival (yrlg-resW)
mean=0.628 sd=0.061 cv=0.098
2.5%=0.522 median=0.623 97.5%=0.761
BCL-SUJ survival (yrlg-str)
mean=0.021 sd=0.014 cv=0.675
2.5%=0.006 median=0.016 97.5%=0.056
Survival age0+_3
mean=0.211 sd=0.142 cv=0.675
2.5%=0.058 median=0.156 97.5%=0.558
Survival age1+_3

Figure 8-15. Histograms of parameter values used in the North Santiam Chinook salmon model used to evaluate EIS alternative, showing summary statistics for each parameter. Values are from 10,000 simulation runs.

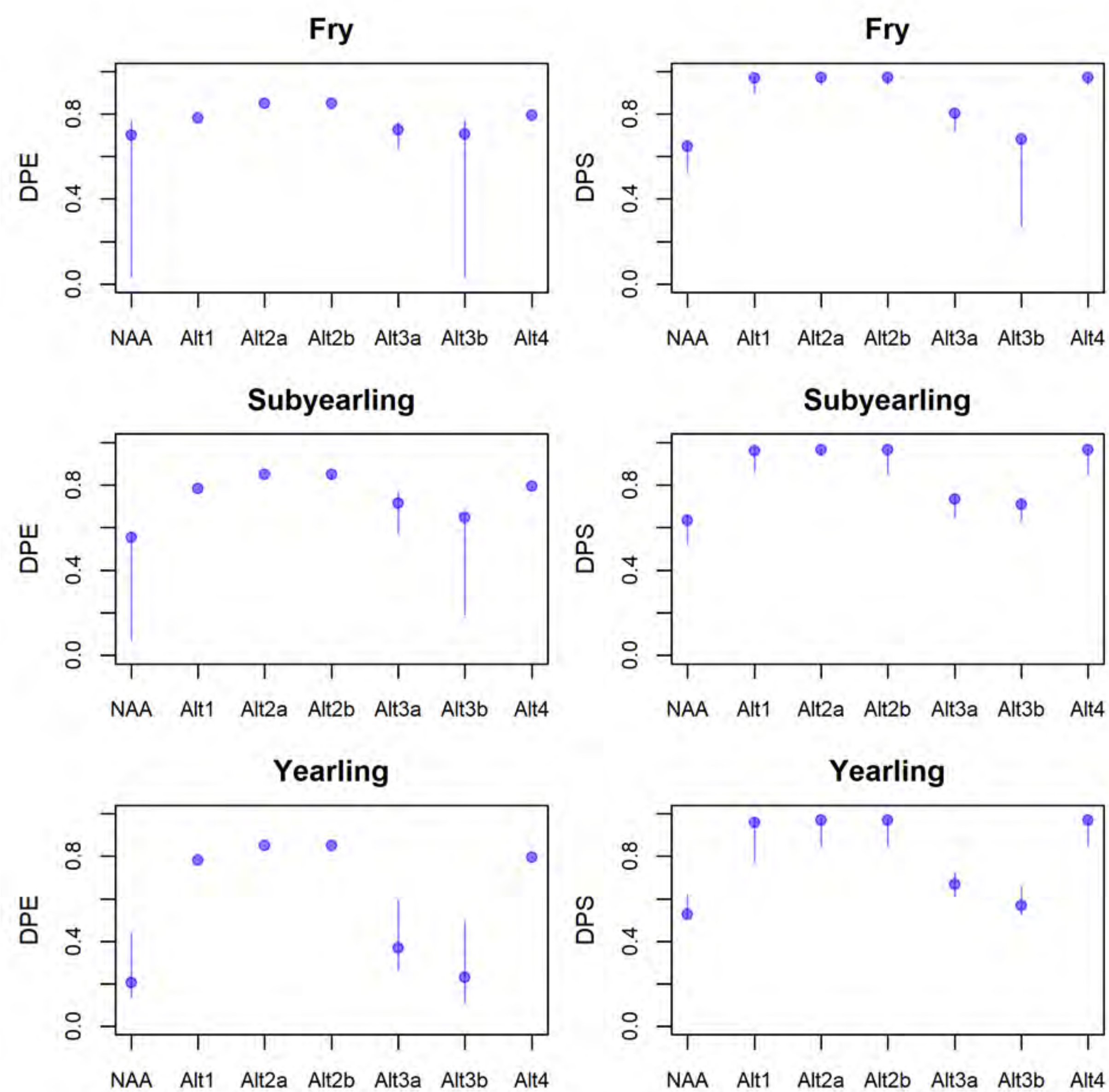

Figure 8-16. Uncertainty in dam passage efficiency (DPE) and dam passage survival (DPS) provided by FBW for each age class of Chinook salmon (Fry, Subyearling, Yearling) passing Detroit dam under each EIS alternative. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs over a 30-year time period.

South Santiam

Model calibration

We initialised the model with data on the number of natural-origin return (NOR) and hatchery-origin return (HOR) adults outplanted above Foster between 2002-2021 (Table 2.7.3). A change in policy to maintain a 100% natural-origin spawning population above Foster meant that no HOR were outplanted above Foster in the years after 2008. The above dam temperature data were applied for those specific years for determining PSM. There were modifications to the fish weir at Foster in spring 2018, which resulted in changes to the dam passage efficiency and survival (Liss et al. 2020). These changes were reflected in the most recent FBW runs under NAA, so to ensure that the returning adults predicted for model calibration reflected contemporary juvenile downstream passage conditions we applied the NAA dam passage parameters obtained from FBW during the 2015 COP evaluations (Zabel et al. 2015). We used the outplanting data from 2002-onwards to initialise the model and then fitted the model to data on NORs to Foster (2006-2021) and age composition from spawner surveys above Foster reservoir (2011-2016). Although the time series of NOR used from the South Santiam (2006-2021) was longer than that from the North Santiam (2015-2021), we followed the North Santiam justification of similar means and CVs (Section 2.7.1.1) to combine the annual deviates in early-age survival rates across both the Santiam sub-basins. This provided a larger pool of annual deviates to bootstrap from for the projections under EIS alternatives (see Appendix A for values).

Table 8-10. Numbers of natural-origin return (NOR) and hatchery-origin return (HOR) Chinook salmon outplanted above Foster reservoir and the numbers of NOR to Foster adult collection facility by year. Data obtained from ODFW HRME reports (Cannon et al. 2010, 2011; Sharpe et al. 2013, 2014, 2015, 2016, 2017) and Brett Boyd (ODFW, pers. comm.).

Year	NOR outplants above Foster	HOR outplants above Foster	NOR to Foster
2002	0	771	795
2003	0	548	451
2004	0	1850	1855
2005	0	936	972
2006	75	857	278
2007	18	385	146
2008	163	521	478
2009	445	0	447
2010	720	0	718
2011	1215	0	1215
2012	962	0	1058
2013	904	0	904
2014	380	0	414
2015	617	0	632
2016	277	0	277

2017	255	0	255
2018	87	0	87
2019	136	0	136
2020	353	0	353
2021	179	0	179

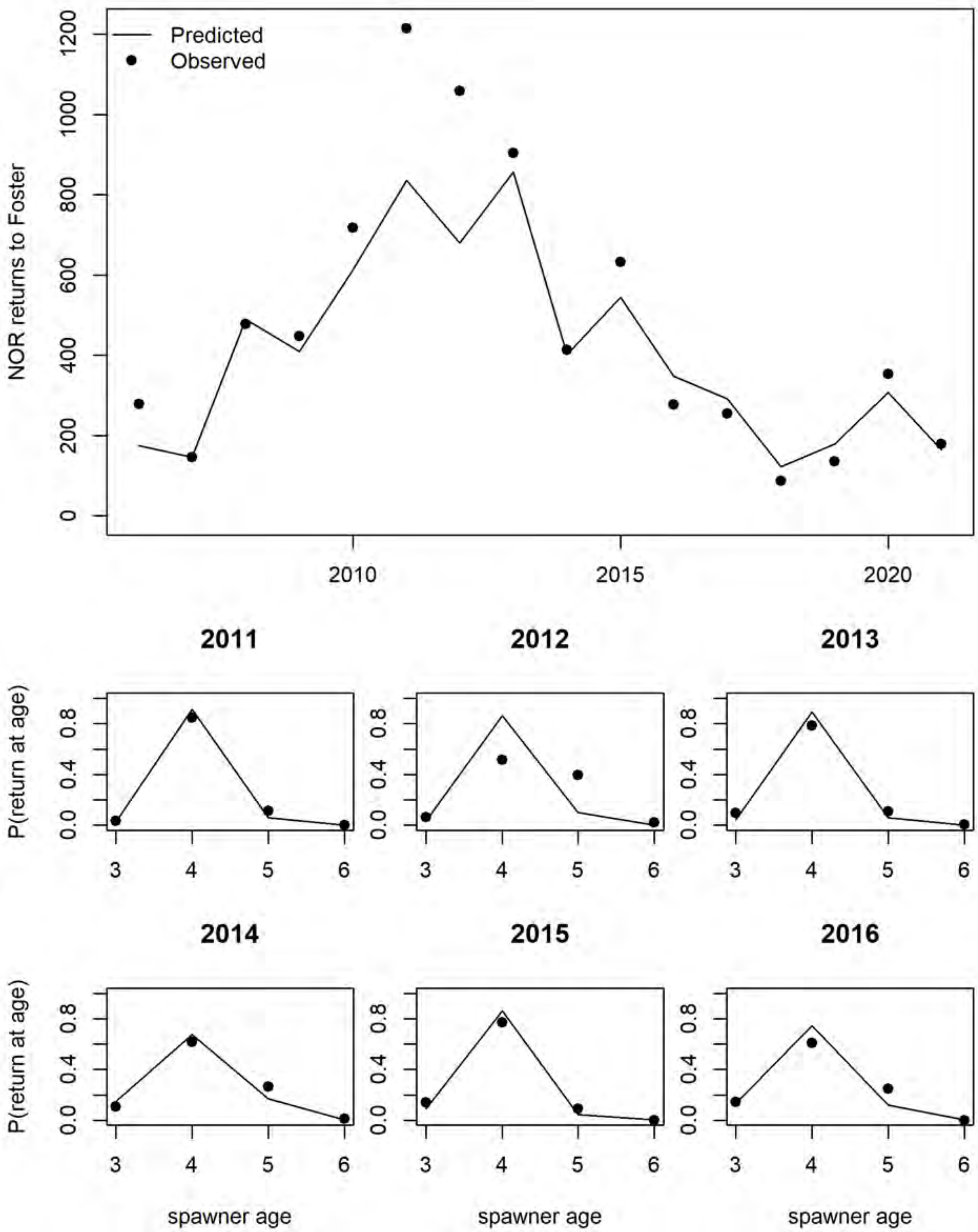

Figure 8-17. Predicted number of natural-origin returns (NOR) to Foster compared to the observed number of NOR in 2006-2021 (upper panel). Predicted spawner age composition after model calibration compared to the observed spawner age composition from spawner surveys above Detroit (lower panel).

The calibrated model fit both NOR and age compositions reasonably well (Figure 2.7.10), except to the age-4 and age-5 spawner composition in 2012 and the large values for NOR observed in 2011-2012. The calibrated parameter values for early-age marine survivals ($\varphi^{o}_{0+\to 3}$, $\varphi^{o}_{1+\to 3}$) and proportions maturing at age ($p_{0+\to 3}$, $p_{3\to 4}$, $p_{4\to 5}$) are shown in Table A.4 (Appendix A).

Performance Metrics for EIS alternatives

Under the NAA, the model projected NOR declined towards extinction during the 30-year time frame (Figure 2.7.11). This decline from the initial value of 430 NORs began immediately during the 5-year burn in period, which is not shown. The initial value was the mean across the last 10 years of observed NORs (i.e., 2012-2021), but since 2012 the returns have been declining to around only 20% of the 2012 value in 2021 (Figure 2.7.10). This decline followed the end of HOR outplanting above Foster in 2008 (Table 2.7.3). The projected South Santiam population thus appears to be continuing this decline without implementation of alternative passage measures or the restoration of HOR outplanting.

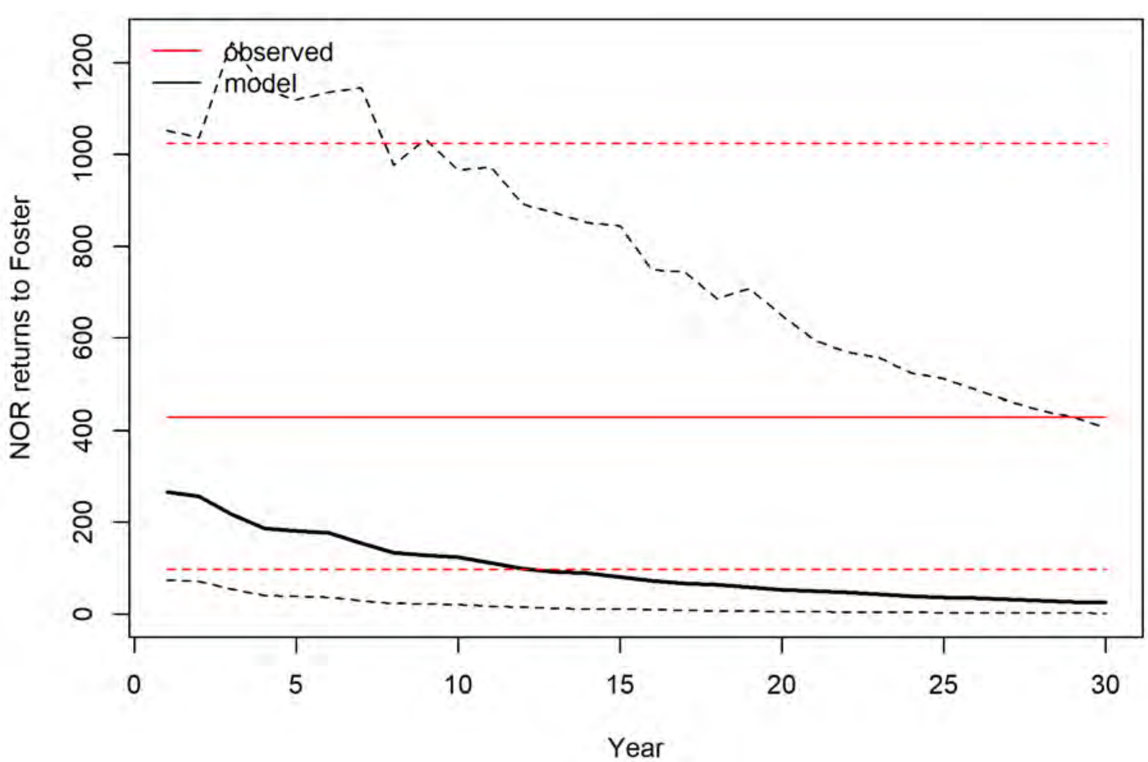

Figure 8-18. NOR returns in the South Santiam projected under the NAA, showing the median (solid black line) and 95% confidence interval (dashed black lines) from 10,000 simulation runs.

The median (solid red line) and 95% percentiles (dashed red lines) of the observed NOR data (2012-2021) are shown for comparison.

Under both NAA and Alt 4, there is no passage through Green Peter and thus no outplanting of adults above Green Peter (see Section 1.2). The recruits per spawner in the South Santiam population is well below 1.0, i.e., population replacement, and thus the population is at very high risk of extinction (Table 2.7.4, Figure 2.7.12). SAR is lower under these alternatives because without production above Green Peter, a higher proportion of the juvenile population is assumed to smolt from Foster as fry which have poor marine survival (see Table A.4 in Appendix A).

Table 8-11. South Santiam Chinook salmon performance metrics under each EIS alternative. Performance metrics are specified for the spawning population above Foster (FOS) and Green Peter (GPR) where appropriate. Summary statistics are medians from 10,000 simulation runs (mean for P<QET). Definitions for each performance metric are found in Section 1.3.

	EIS alternative						
Performance metric	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
NOR spawners (FOS)	25	1046	772	590	433	313	57
NOR spawners (GPR)	NA	1295	963	728	535	386	NA
R/S	0.643	1.564	1.5	1.411	1.32	1.243	0.755
SAR	0.039	0.056	0.056	0.055	0.053	0.052	0.042
pHOS (FOS)	0	0	0	0	0	0	0
pHOS (GPR)	NA	0.385	0.427	0.474	0.536	0.607	NA
P<QET	0.993	0.004	0.028	0.101	0.322	0.609	0.956

The addition of downstream passage at Green Peter, and the incorporation of additional HOR adults for outplanting above Green Peter, reduces the risk of extinction. Alt 1, the only alternative with structural passage at Green Peter, results in the highest NORs, with spawning populations of >1,000 adults above each dam (FOS and GPR). Although the risk of extinction was reduced under Alt 3a and Alt 3b, these results indicate the importance of further modifications to the fish weir at Foster in recovering the population. Despite the recovery potential under Alt 1 being much improved, with a near zero risk of extinction, NOR do not reach the logistical outplant cap indicating there is potential to recover the population further. Another indication of this is that the outplanting of HOR adults above Green Peter under all alternatives with passage there remained relatively important, as pHOS did not fall much below 40% (Table 2.7.4).

We note that although the downstream passage measures are the same for Alt 2a and Alt 2b, the outcomes in terms of NOR spawners and extinction risk are not the same. This is because the input DPE and DPS values from FBW are different, especially at Foster, due to RES-SIM hydrological data.

It is important to highlight the uncertainty associated with the performance metrics (Figure 2.7.12 and Figure 2.7.13), as under all alternatives the 95% confidence interval for R/S extends down to or below 1.0, i.e., population replacement. This indicates that despite there being minimal probability of population extinction under Alt 1, any future changes to the system not incorporated in this model, e.g. due to climate change, could result in poorer population recovery.

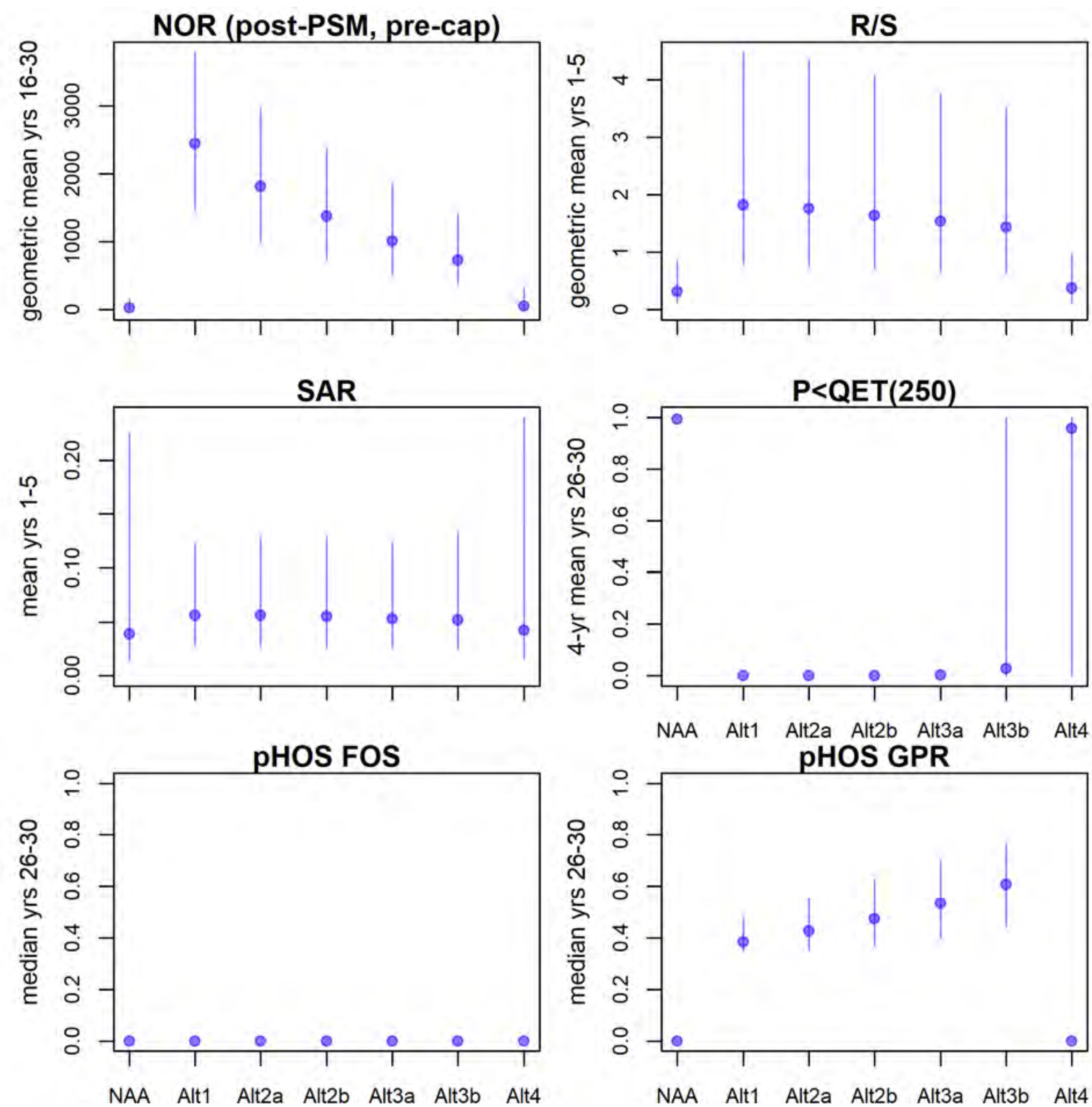

Figure 8-19. Uncertainty in performance metrics under each alternative in the South Santiam: 1) NOR spawners (Foster, FOS, + Green Peter, GPR, post-PSM and pre outplant cap), 2) recruits-per-spawner (R/S), 3) smolt-adult return rate (SAR), 4) probability of extinction given a quasi-extinction threshold of 250 NOR spawners (P<QET(250)), 5) proportion of hatchery-origin spawners (pHOS). Median (circles) and 95% confidence intervals (lines) are from 10,000

simulation runs. Error bars are not shown for P<QET, owing to its binary outcomes. Note that there is no adult outplanting at GPR under the NAA or Alt4, such that under these alternatives, pHOS at GPR is 0% of a non-existent population.

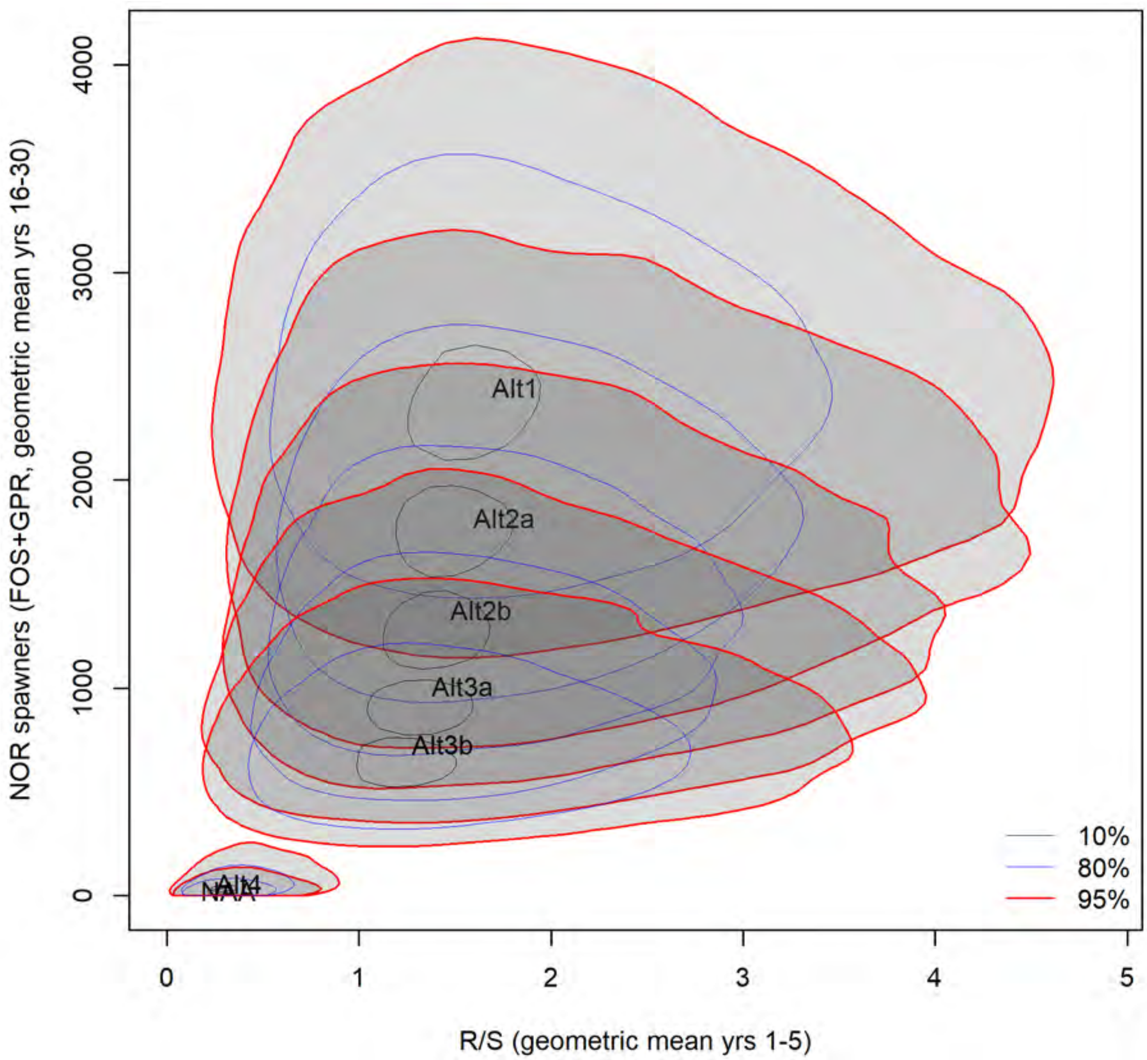

Figure 8-20. 2-D kernel density estimates (bandwidth=40) from 10,000 simulation runs to show the trade offs between NOR spawner and R/S performance metrics under each EIS alternative in the South Santiam.

The diversity of the population, as determined by relative proportions of different juvenile migrant types, was quite different between the NAA and Alt 4 alternatives and all the other alternatives to a larger proportion of smolts being fry under NAA and Alt 4 (Figure 2.7.14). The reason for this difference is that under these two alternatives, all juveniles that move out of natal streams move into Foster reservoir, where under our assumptions about river juvenile splits (see Table A.4 in Appendix A) there is a high proportion of fry (75%) being available to pass Foster dam as fry, rather than remaining in the reservoir to attempt passage as subyearlings or yearlings as is assumed for Green Peter. There was a higher proportion of yearling migrant types as both smolts and returning adults under Alt 1.

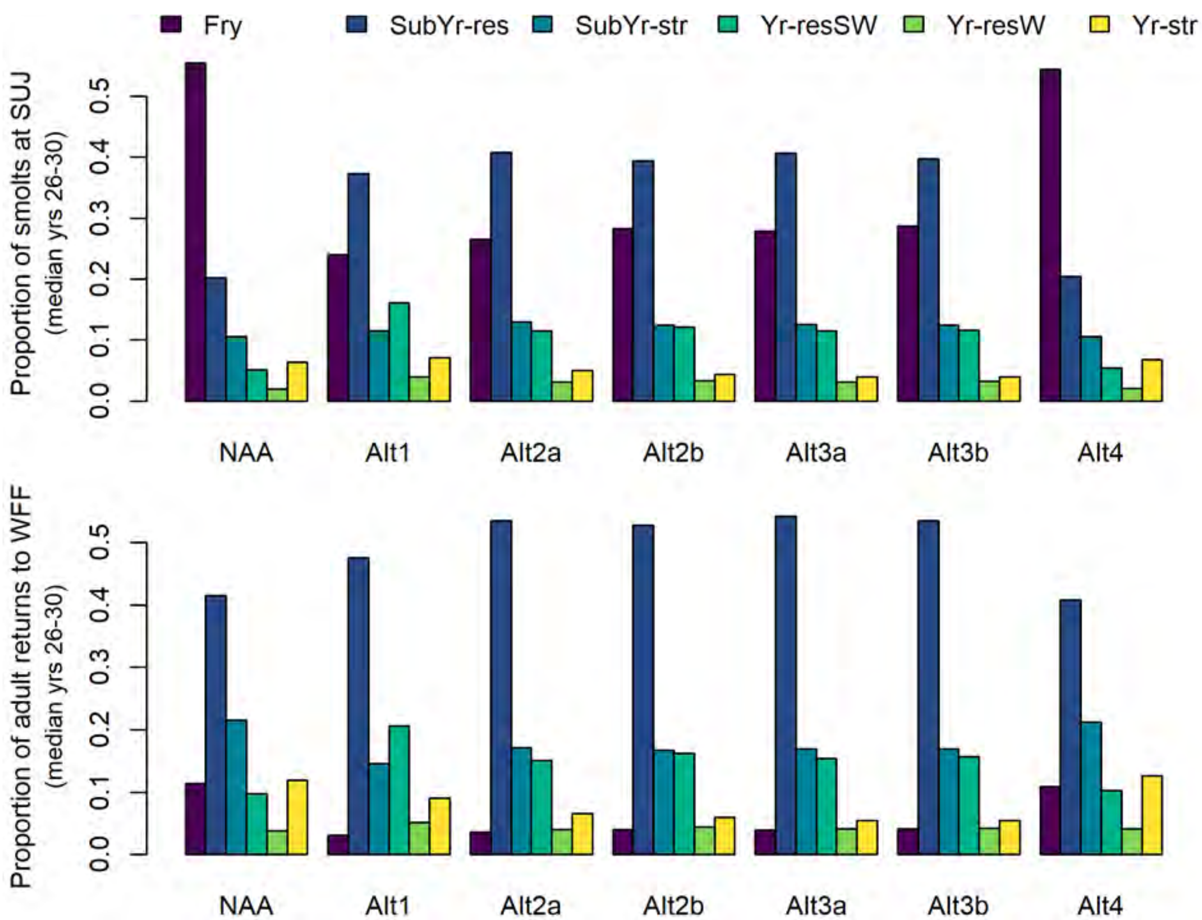

Figure 8-21. Relative proportions of different juvenile migrant types passing Foster under each EIS alternative when counted as smolts at Sullivan Juvenile Facility (SUJ) or as returning adults at Willamette Falls (WFF). The migrant types modelled were Fry, Subyearlings that spent the summer in the reservoir (SubYr-res), Subyearlings that spent the summer in the natal stream (SubYr-str), Yearlings that spent summer and winter in the reservoir (Yr-resSW), Yearlings that spent only winter in the reservoir (Yr-resW), and Yearlings that spent summer and winter in the natal stream (Yr-str).

Fry to smolt survival varied between alternatives and juvenile migrant type (Figure 2.7.15 and Figure 2.7.16). Fry to smolt survival was lower for those migrant types which spent longer periods in the reservoir prior to passage, i.e., subyearlings that spent the summer in the reservoir and yearlings that had spent both summer and winter in the reservoir. Yearlings that only spent winter in the reservoir had comparable survival to yearlings that reared in the natal stream. Fry to smolt survival was lowest under Alt 3a, which had spring spill at Green Peter and no modified weir at Foster. The relative adult return rates were lowest for the fry migrant type, but comparable across the other juvenile migrant types, and there was minimal difference between the alternatives (Figure 2.7.17).

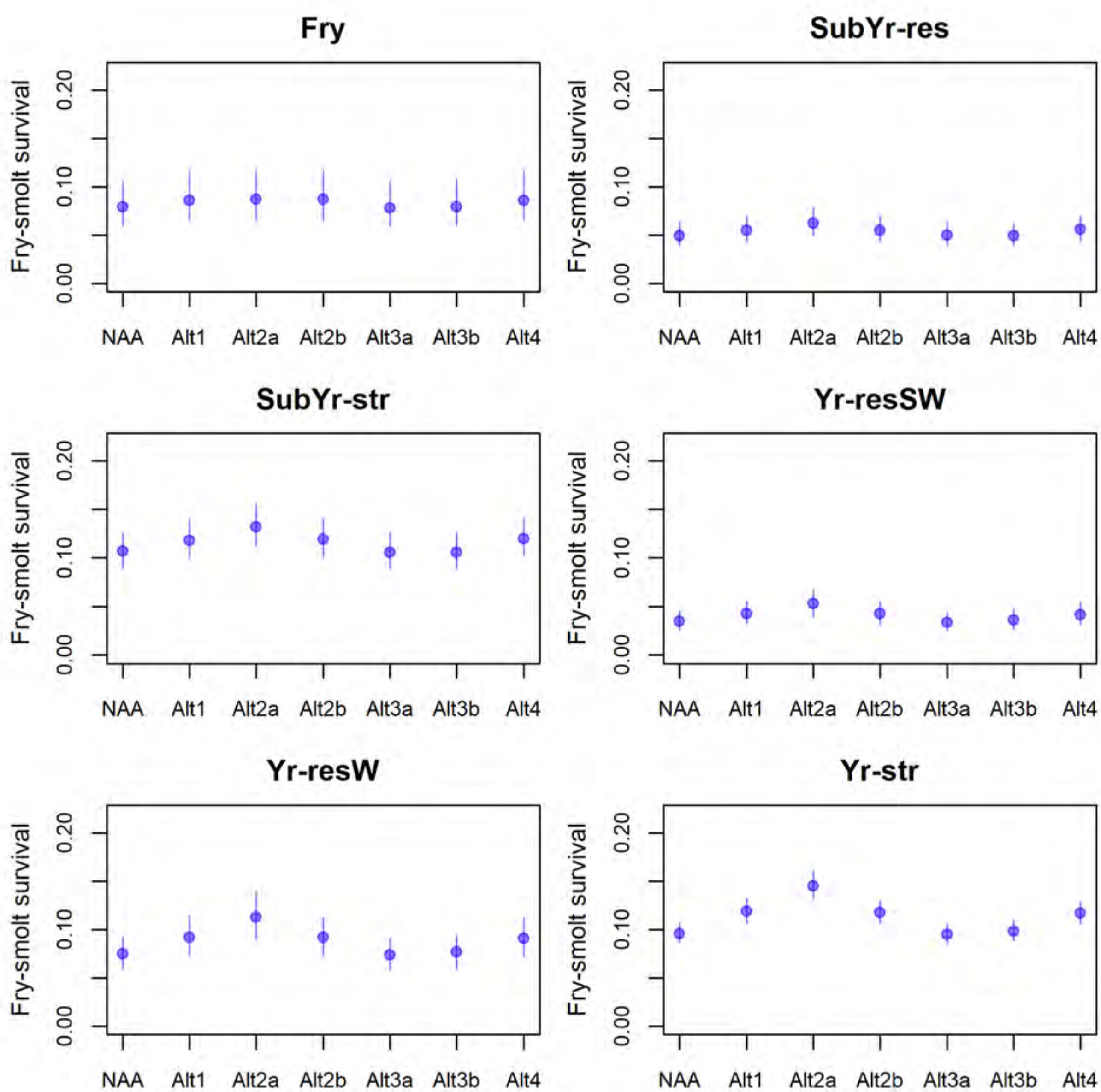

Figure 8-22. Fry to smolt survival for each juvenile migrant type that emerged above Foster under each EIS alternative. Calculated as the mean over years 1-5 of each simulation run. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs.

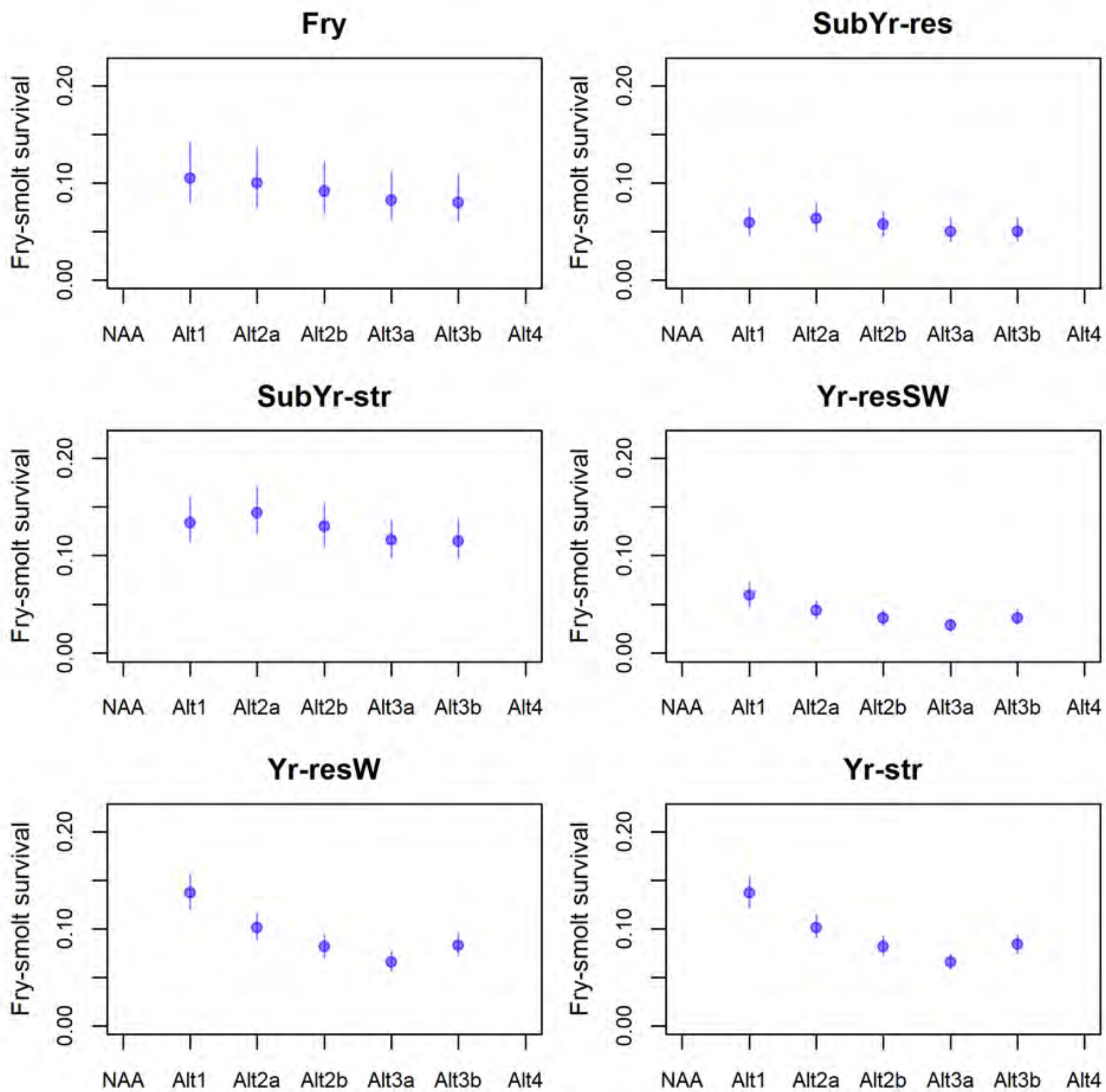

Figure 8-23. Fry to smolt survival for each juvenile migrant type that emerged above Green Peter under each EIS alternative. Calculated as the mean over years 1-5 of each simulation run. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs.

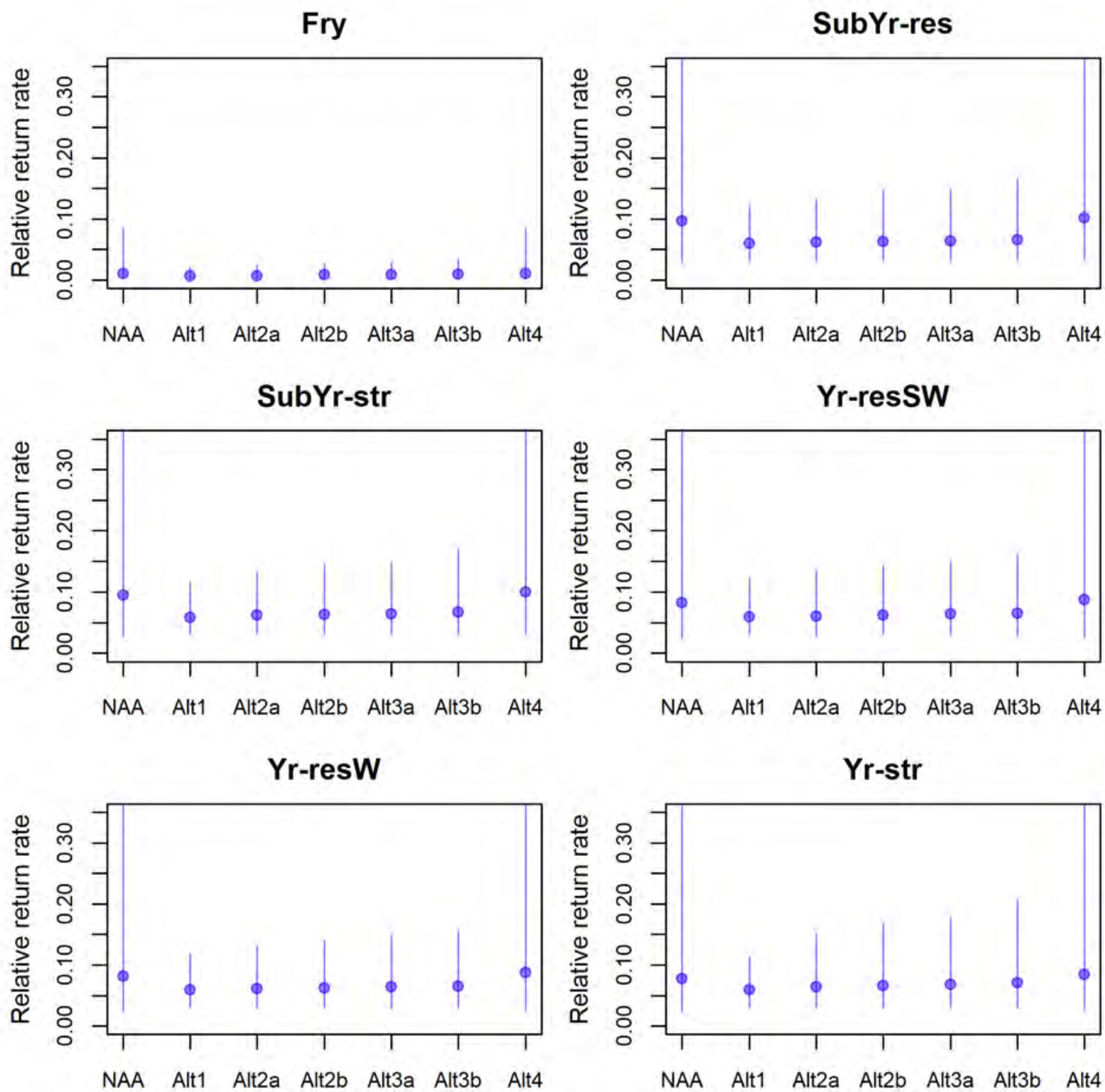

Figure 8-24. Relative return rates of each juvenile migrant type under each EIS alternative. Calculated as the mean over years 26-30 of each simulation run. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs.

Parameter Uncertainties

The population dynamics model was probabilistic and the distributions of those parameters that had uncertainty are shown in Figure 2.7.18. The distributions for early-age marine survivals were determined by the annual deviates obtained from the model calibration, these do not appear as smooth distributions because of the relatively low number of deviates (n=23).

The uncertainty in dam passage parameters, DPE and DPS, provided by FBW varied by dam and alternative (Figure 2.7.19 and Figure 2.7.20). DPE through Foster had almost no uncertainty and had very similar values among alternatives. DPS through showed more differences between alternatives, being lowest under NAA, Alt 3a and Alt 3b where there was no modified fish weir. However, there was only uncertainty in DPS for yearlings. DPE through Green Peter showed large differences between the alternatives, and there was some uncertainty for each juvenile migrant age. DPE was highest for Alt 1, which was the only structural passage measure evaluated. DPE was lowest for Alt 3b, which evaluated spring drawdown, but this was also the alternative with the greatest uncertainty in DPE. DPS through Green Peter also showed large differences between the alternatives. It was highest for Alt 1, and lowest for those alternatives with spring spill (Alts 2a/2b/3a), though for yearlings the uncertainty in the DPS values suggested that Alt 3b could perform as poorly.

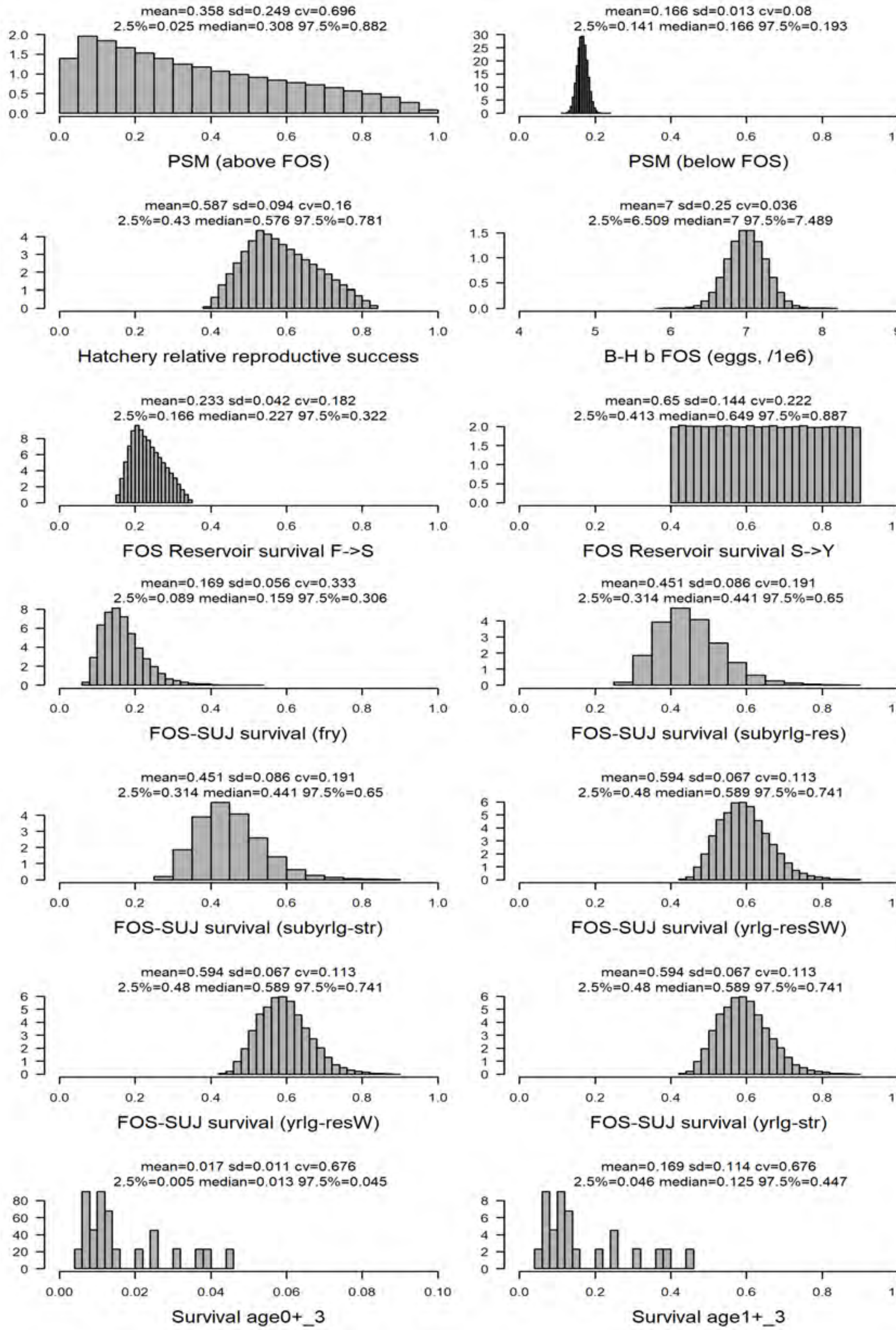

Figure 8-25. Histograms of parameter values used in the South Santiam Chinook salmon model used to evaluate EIS alternative, showing summary statistics for each parameter. Values are from 10,000 simulation runs.

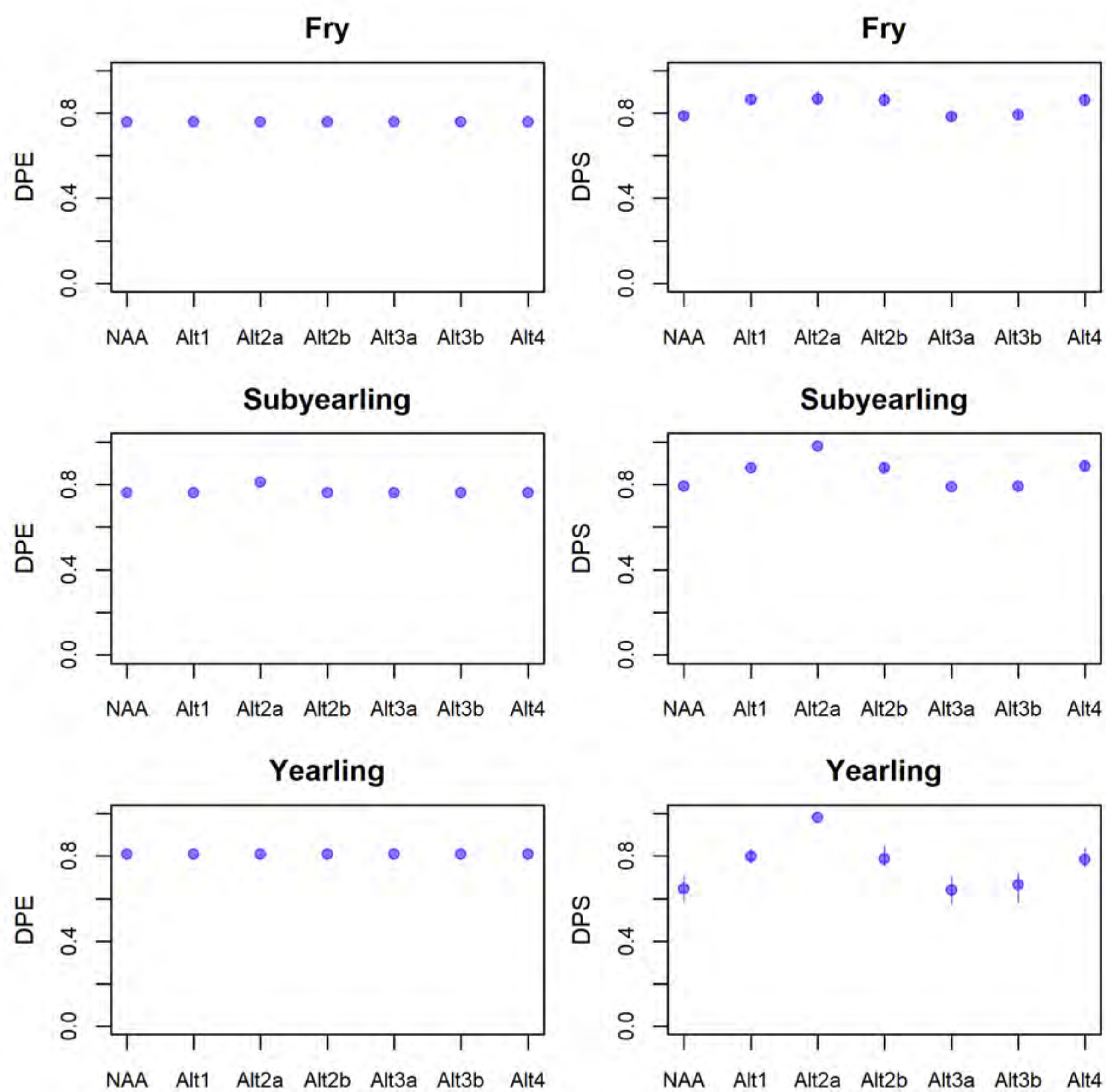

Figure 8-26. Uncertainty in dam passage efficiency (DPE) and dam passage survival (DPS) provided by FBW for each age class of Chinook salmon (Fry, Subyearling, Yearling) passing Foster dam under each EIS alternative. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs over a 30-year time period.

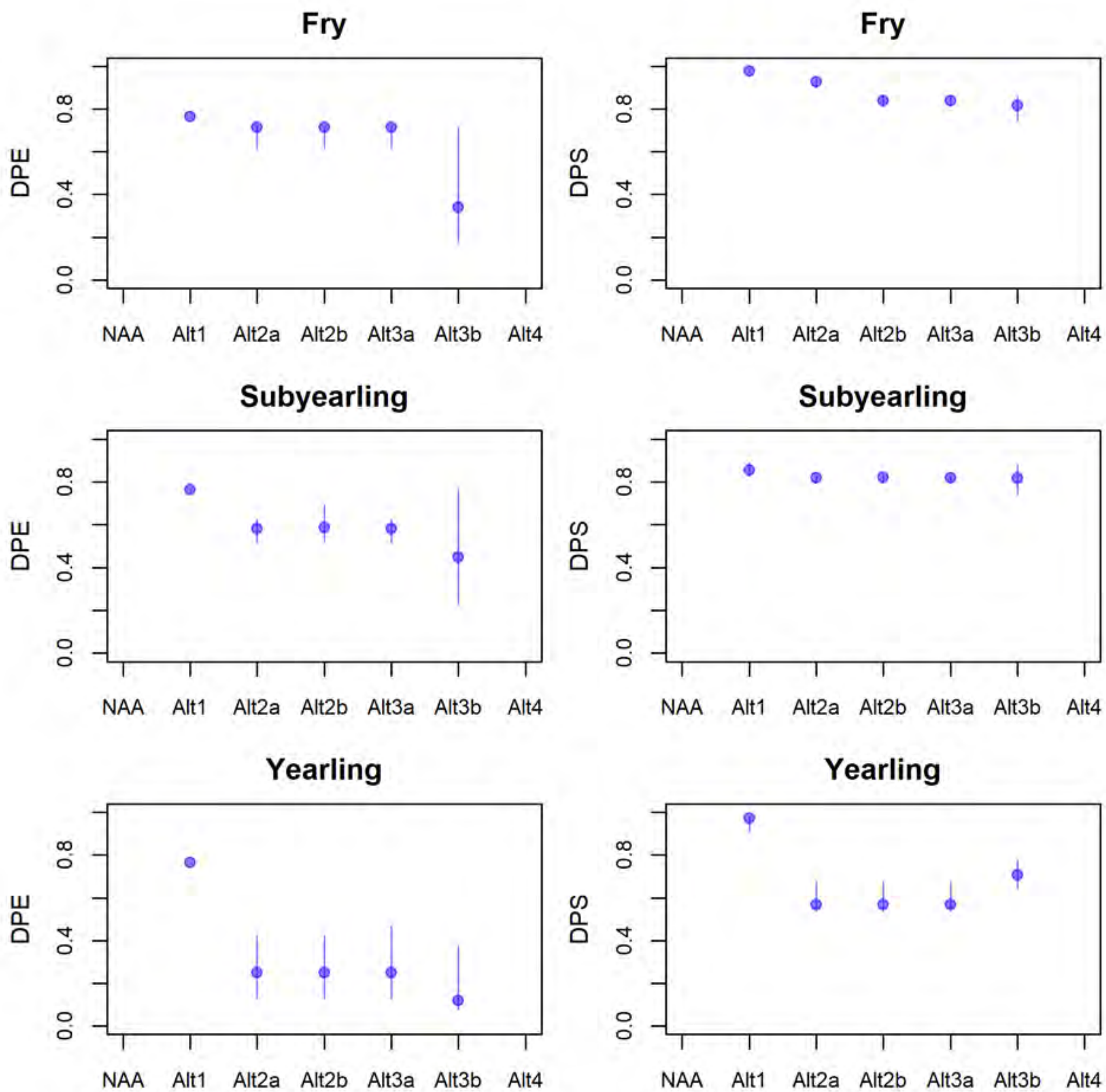

Figure 8-27. Uncertainty in dam passage efficiency (DPE) and dam passage survival (DPS) provided by FBW for each age class of Chinook salmon (Fry, Subyearling, Yearling) passing Green Peter dam under each EIS alternative. Note there is no downstream passage under NAA and Alt 4. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs over a 30-year time period.

McKenzie

Model calibration

To initialize the model, we compiled data on the number of NOR and HOR adults outplanted above Cougar dam between 2010-2021 (Table 2.7.5). For the purpose of calibrating the model, we applied FBW's estimated dam passage survival and efficiency for these years—using simulations under the NAA alternative—and estimated PSM using above-Cougar temperature data. Then, we fit the model to available data on NOR to the Cougar adult fish collection facility and age composition from spawner surveys (2011-2016).

Table 8-12. Numbers of natural origin and hatchery-origin Chinook salmon returns (NOR and HOR, respectively) outplanted above Cougar reservoir and of NOR returns to the Cougar adult fish collection facility. Data obtained from ODFW & USACE (2018).

Year	NOR outplants above Cougar	HOR outplants above Cougar	NOR returns to Cougar fish collection facility
2010	252	510	252
2011	385	345	385
2012	522	429	522
2013	191	441	191
2014	155	542	155
2015	157	600	157
2016	244	475	244
2017	165	446	165
2018	68	548	68
2019	78	381	78
2020	95	311	95

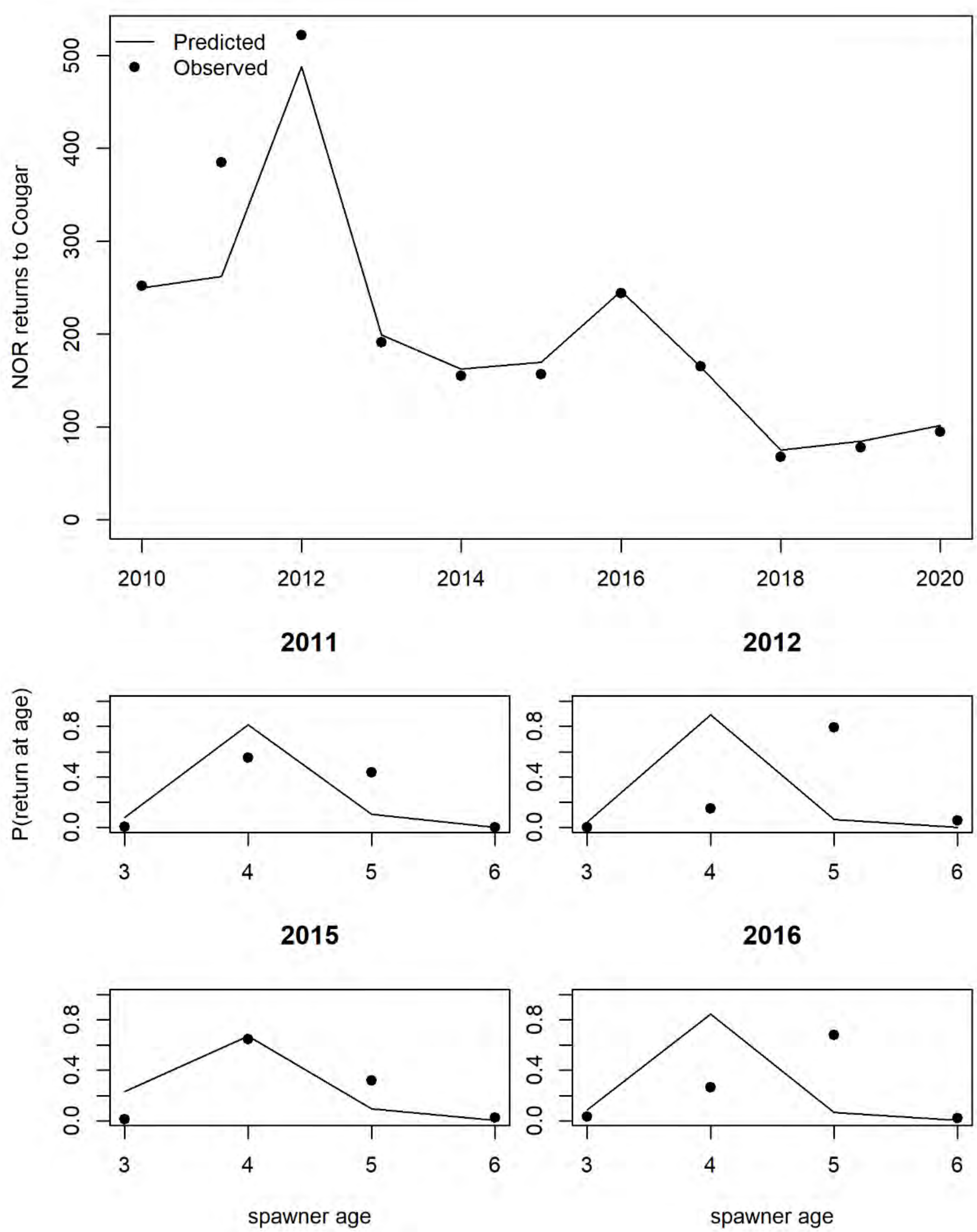

Figure 8-28. Predicted number of natural-origin returns (NOR) to Cougar adult fish collection facility compared to the observed number of NOR in 2015-2020 (upper panel). Predicted spawner age composition (after model calibration) compared to the observed spawner age composition from spawner surveys above Cougar (lower panel). Note that age composition data were not available for 2013 and 2014.

The calibrated model fit showed NOR abundance decreasing since 2011, indicating the current outplanting plan (median=446; range 311-600; mean=210) cannot compensate for freshwater and marine survival of NOR returns. Model-estimated NOR abundances are low (median=165;

range 95-522), with a peak in 2012 followed by a punctuated decline through the rest of the time series. The model was difficult to fit, likely because of the short time and the lack of contrast in abundance between years (i.e., a “one-way-trip time series” c.f. Hilborn and Walters 1992). The calibrated model fit the NOR returns well (Figure 2.7.21), but age compositions did not. For 2012 and 2015, age-4 returns were overestimated by the model, and age-5 returns underestimated. The calibrated parameter values for early-age marine survivals ($\varphi^{o}_{0+\to 3}$, $\varphi^{o}_{1+\to 3}$) and proportions maturing at age ($p_{0+\to 3}$, $p_{3\to 4}$, $p_{4\to 5}$) are shown in Table A.6 (Appendix A).

Performance Metrics for EIS alternatives

Under the NAA, the model reached an equilibrium in less than five years, after a modest decline from starting NOR returns. This trend matches observed data in recent years, which show declining natural origin returns to Cougar over the years 2010 to 2020. Despite this, model projections were lower than the mean historic returns from 2010-2020 (Figure 2.7.22). Expected NOR were well under the outplanting NOR target of 600.

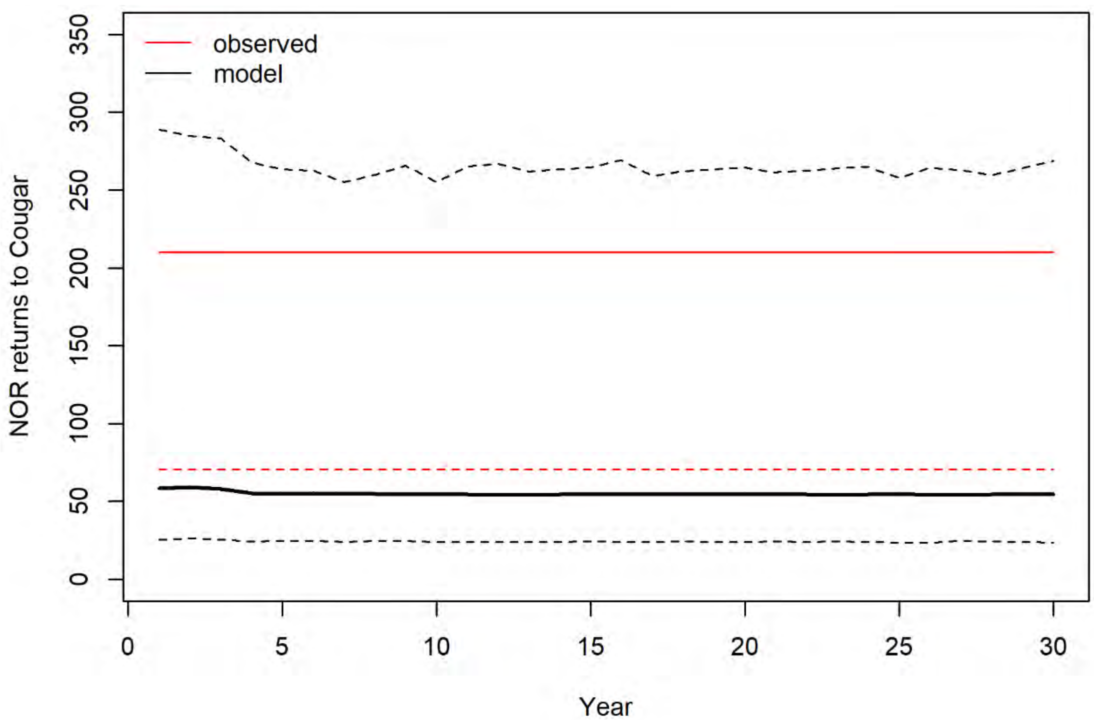

Figure 8-29. NOR returns in the McKenzie projected under the NAA, showing the median (solid black line) and 95% confidence interval (dashed black lines) from 10,000 simulation runs. The median (solid red line) and 95% percentiles (dashed red lines) of the observed NOR data (2010-2020) are shown for comparison.

Table 8-13. McKenzie Chinook salmon performance metrics under each EIS alternative Summary statistics are medians from 10,000 simulation runs (mean for P<QET). Definitions for each performance metric are found in Section 1.3.

	EIS alternative						
Performance metric	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
NOR spawners	56	44	590	291	108	220	582
R/S	1.05	1.039	1.352	1.206	1.086	1.163	1.333
SAR	0.028	0.026	0.03	0.029	0.025	0.027	0.03
pHOS	0.879	0.905	0.193	0.475	0.771	0.577	0.197
P<QET	1	1	0.005	0.376	0.999	0.739	0.004

Under all alternatives that implemented improvements to juvenile downstream passage (i.e., all but Alt1), performance of the chinook populations was improved compared to performance under the NAA. Under no alternative was the logistical outplanting cap reached, and supplementation of NOR adults with HORs above Cougar was necessary regardless of alternative (indicated by pHOS > 0).

Population recovery was highest under Alt2a and Alt4, which both implemented a floating surface structure to improve downstream passage of juvenile salmon. Under these alternatives, DPE and DPS were approximately 3-fold higher than under the NAA (Figure 2.7.29). Other alternatives that incorporated operational management changes (i.e., spring and/or fall drawdown) instead of structural changes had lower expected DPE (and, to a lesser extent, DPS) for all life history stages (Figure 2.7.29). This trend is also reflected in expected pHOS and probability of quasi-extinction. Operational alternatives required more supplementation of adult outplants with HOR and population recovery was low with high probability of the population falling below the quasi-extinction threshold (Table 2.7.6 and Figure 2.7.23).

Despite this, the mean R/S was above replacement under all alternatives (Table 2.7.6), suggesting that on average, population recovery is likely. Mirroring other performance metrics, Alts 2a, 2b, and 4 were most supportive of high R/S. However, the 95% confidence intervals of outcomes under all alternatives included R/S values less than 1 (Figure 2.7.23 and Figure 2.7.24). Especially if there are future changes not incorporated in the model (e.g., oceanic regime shifts), there may be a non-negligible risk that the population may not be able to replace itself (i.e., if R/S falls < 1). Importantly, due to the high uncertainty of model results—indicated by the wide confidence intervals for each performance metric—there is some risk of population quasi-extinction even under the most optimistic alternatives.

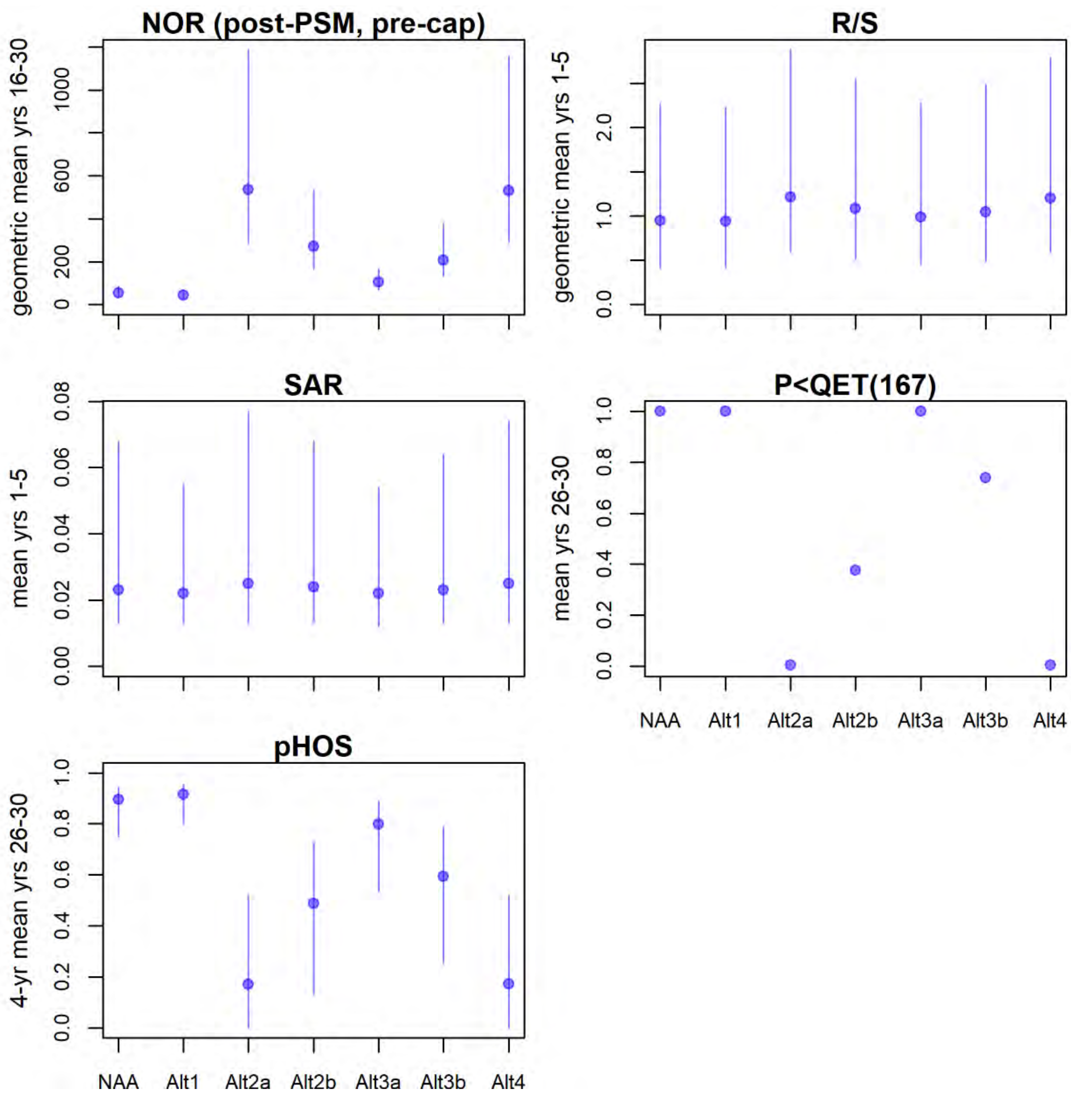

Figure 8-30. Uncertainty in performance metrics under each alternative for the McKenzie subbasin: 1) NOR spawners (post-PSM and pre outplant cap), 2) recruits-per-spawner (R/S), 3) smolt-adult return rate (SAR), 4) probability of extinction given a quasi-extinction threshold of 167 NOR spawners, and 5) proportion of hatchery-origin spawners (pHOS). Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs. Error bars are not shown for P<QET, owing to its binary outcomes.

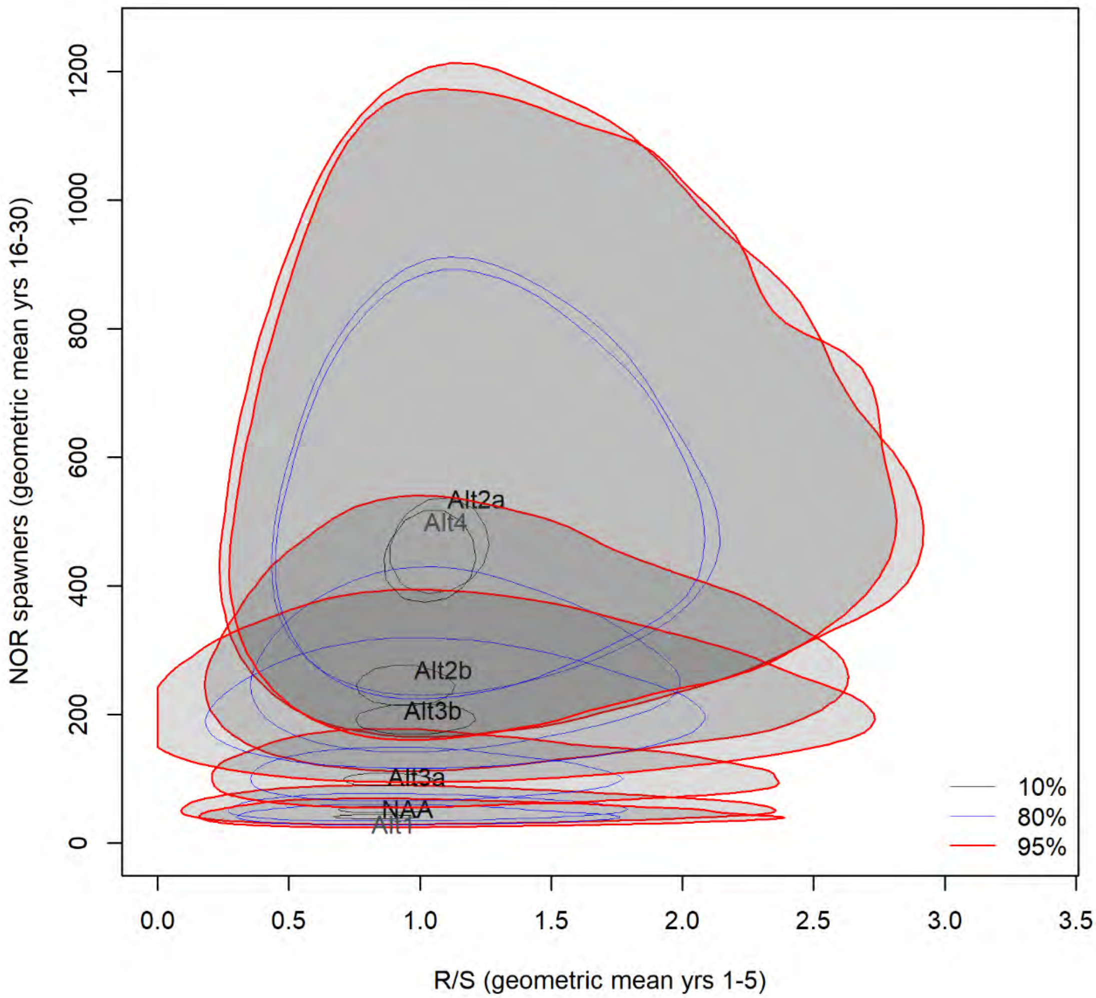

Figure 8-31. 2-D kernel density estimates (bandwidth=30) from 10,000 simulation runs to show the trade offs between NOR spawner and R/S performance metrics under each EIS alternative in the McKenzie. Note that results from Alt2a and Alt4 are overlapping in the uppermost kernel. Due to overlap in performance between NAA and Alt1 and Alt2a and Alt4, labels for Alt1 and Alt4 have been displaced below and to the left of their respective kernel centroids. Note that while the kernel density estimate for R/S values under Alt3b appear to include negative values; this is an artefact of the plotting method as the model does not generate R/S < 0.

In all alternatives with dam passage improvements (i.e., except NAA or Alt1), diversity of juvenile migrant types shifted to have greater representation of sub-yearlings that remain in-reservoir during their first summer (Sub Yr-res) and lower representation by yearlings spend both summer and winter in-reservoir (Figure 2.7.25). This is due to increased DPE for these juvenile stages, especially for alternatives 2a and 4 which implemented structural passage improvements. These results highlight that fry migrants do not significantly contribute to the adult population compared to those that migrate at older life stages, due to their lower marine survival rates.

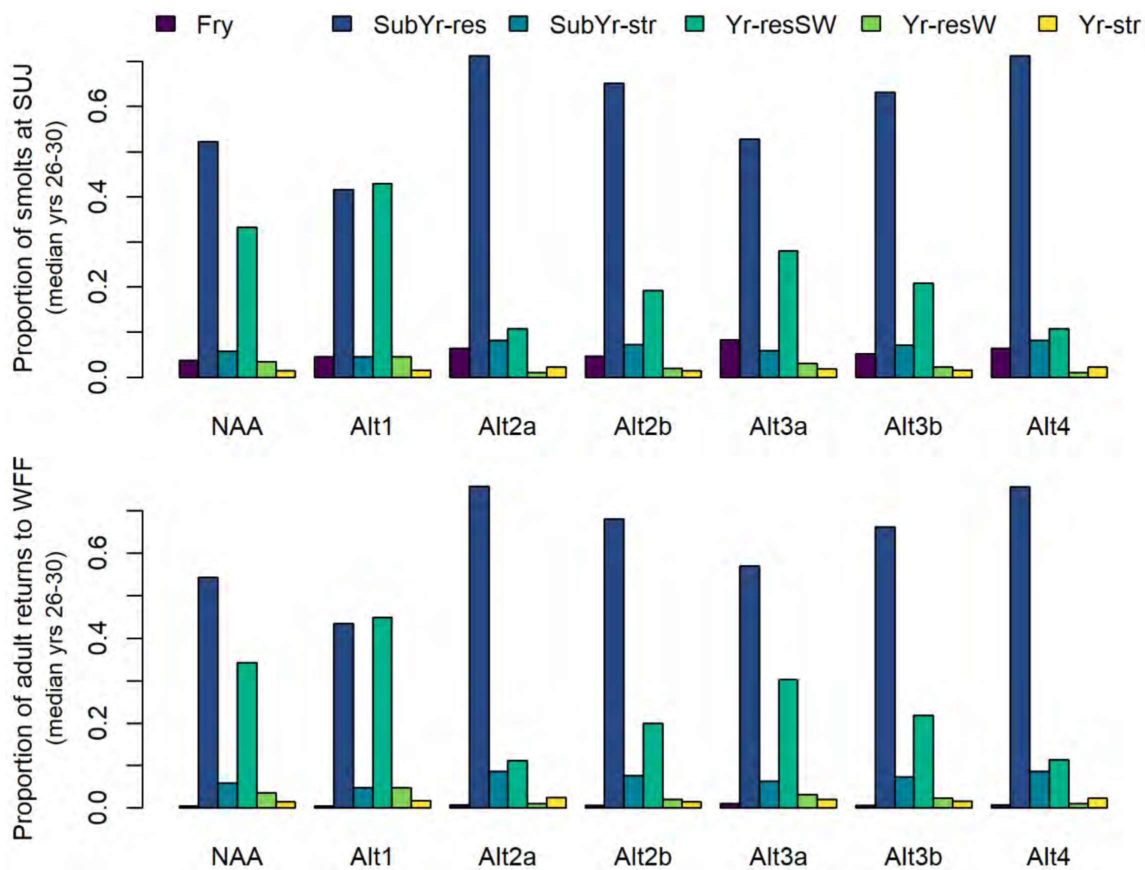

Figure 8-32. Relative proportions of different juvenile migrant types passing Cougar dam under each EIS alternative when counted as smolts at Sullivan Juvenile Facility (SUJ) or as returning adults at Willamette Falls (WFF). The migrant types modelled were Fry, Subyearlings that spent the summer in the reservoir (SubYr-res), Subyearlings that spent the summer in the natal stream (SubYr-str), Yearlings that spent summer and winter in the reservoir (Yr-resSW), Yearlings that spent only winter in the reservoir (Yr-resW), and Yearlings that spent summer and winter in the natal stream (Yr-str).

Fry to smolt survival varied somewhat between alternatives and migrant types (Figure 2.7.26). Survival of fry migrants was generally low and insensitive to which alternative is implemented. Subyearlings remaining in their natal stream over summer (SubYr-str) and yearlings that either remained in their natal stream through winter and summer (Yr-str) or spent only winter in-reservoir (Yr-resW) demonstrated the most varied survival estimates between alternatives, with survival rates maximized under alternatives 2a/b and 4. Fry to smolt survival rates were lowest under the NAA and Alt1, but similarly low under Alt3a for all juvenile migrant types. Considering relative return rates (Figure 2.7.27), fry and Yr-str types had the lowest expected returns under all alternatives. Alternatives had minimal influence on relative return rates of these juvenile migrant types.

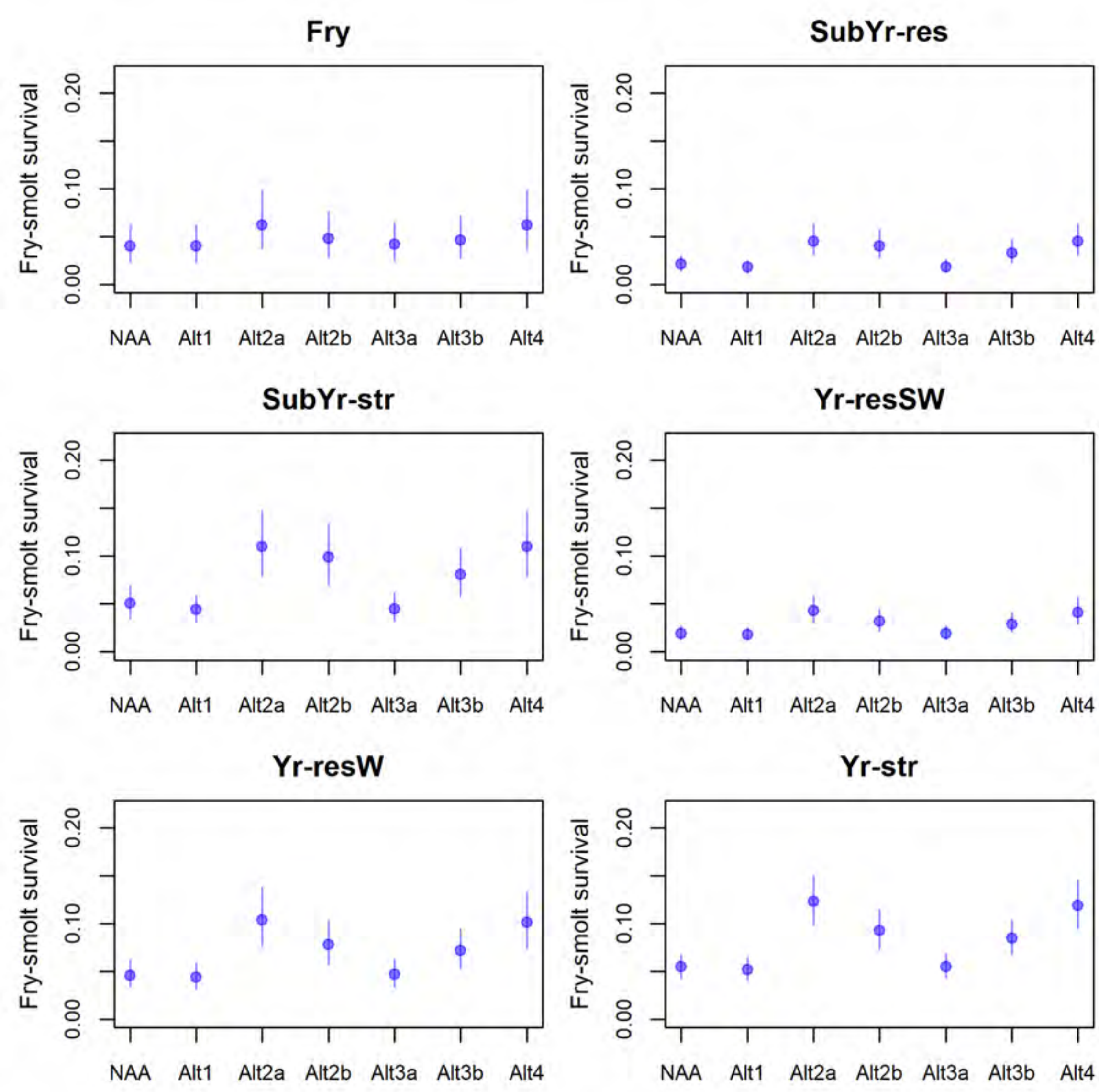

Figure 8-33. Fry to smolt survival for each juvenile migrant type under each EIS alternative. Calculated as the mean over years 1-5 of each simulation run. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs.

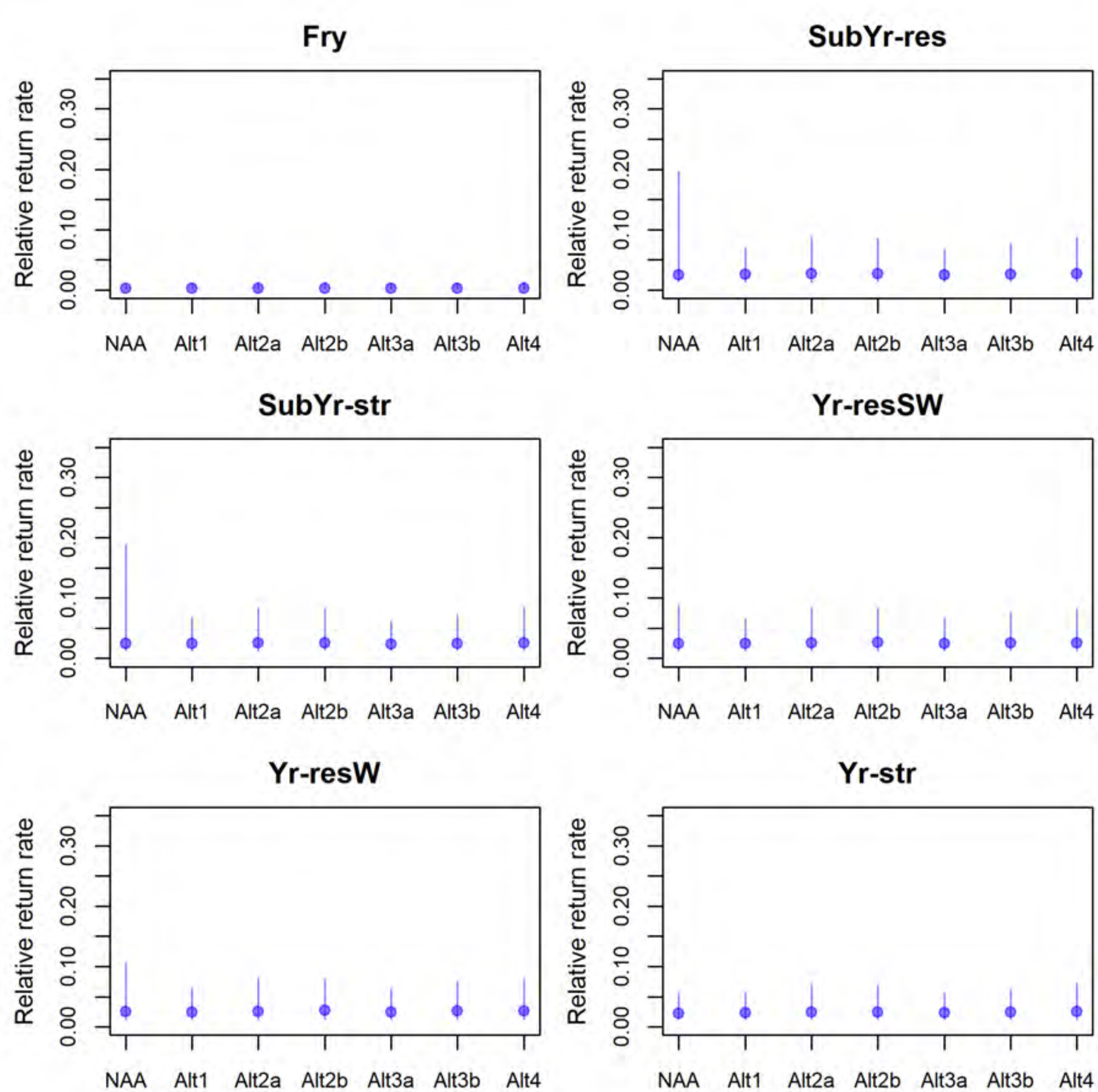

Figure 8-34. Relative return rates of each juvenile migrant type under each EIS alternative. Calculated as the mean over years 26-30 of each simulation run. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs.

Parameter Uncertainties

The population dynamics model was probabilistic and the distributions of those parameters that had uncertainty are shown in (Figure 2.7.28). Sampling distributions used to describe early-age marine survivals were determined by the annual deviates obtained from the model calibration—these do not appear as smooth distributions because of the few deviates (n=9).

The uncertainty in dam passage parameters, DPE and DPS, provided by FBW varied by alternative (Figure 2.7.29). For those alternatives with structural passage measures (i.e., Alts 2a and 4), DPE was high (>0.8) and highly certain. Here, DPS was also high and highly certain, nearly 100% for all life history stages with minimal uncertainty. In comparison, for alternatives with operational passage only (i.e., Alts 2b and 3a/b) both DPE and DPS were lower than structural passage alternatives. Notably, uncertainty in DPE and DPS were highest for the subyearling and yearling components of the population and were the most variable in response to different alternatives.

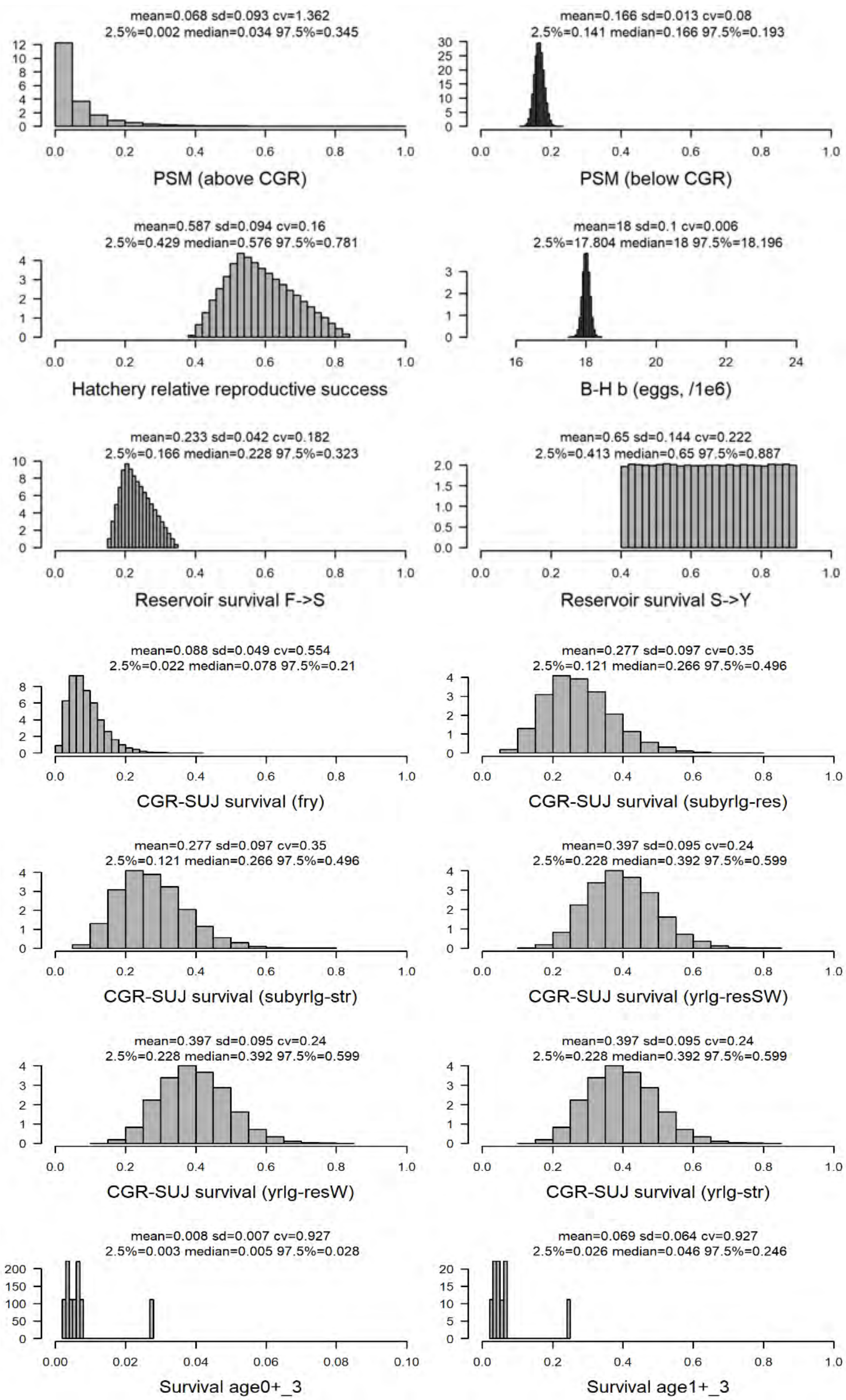

Figure 8-35. Histograms of parameter values used in the McKenzie Chinook salmon model used to evaluate EIS alternative, showing summary statistics for each parameter. Values are from 10,000 simulation runs.

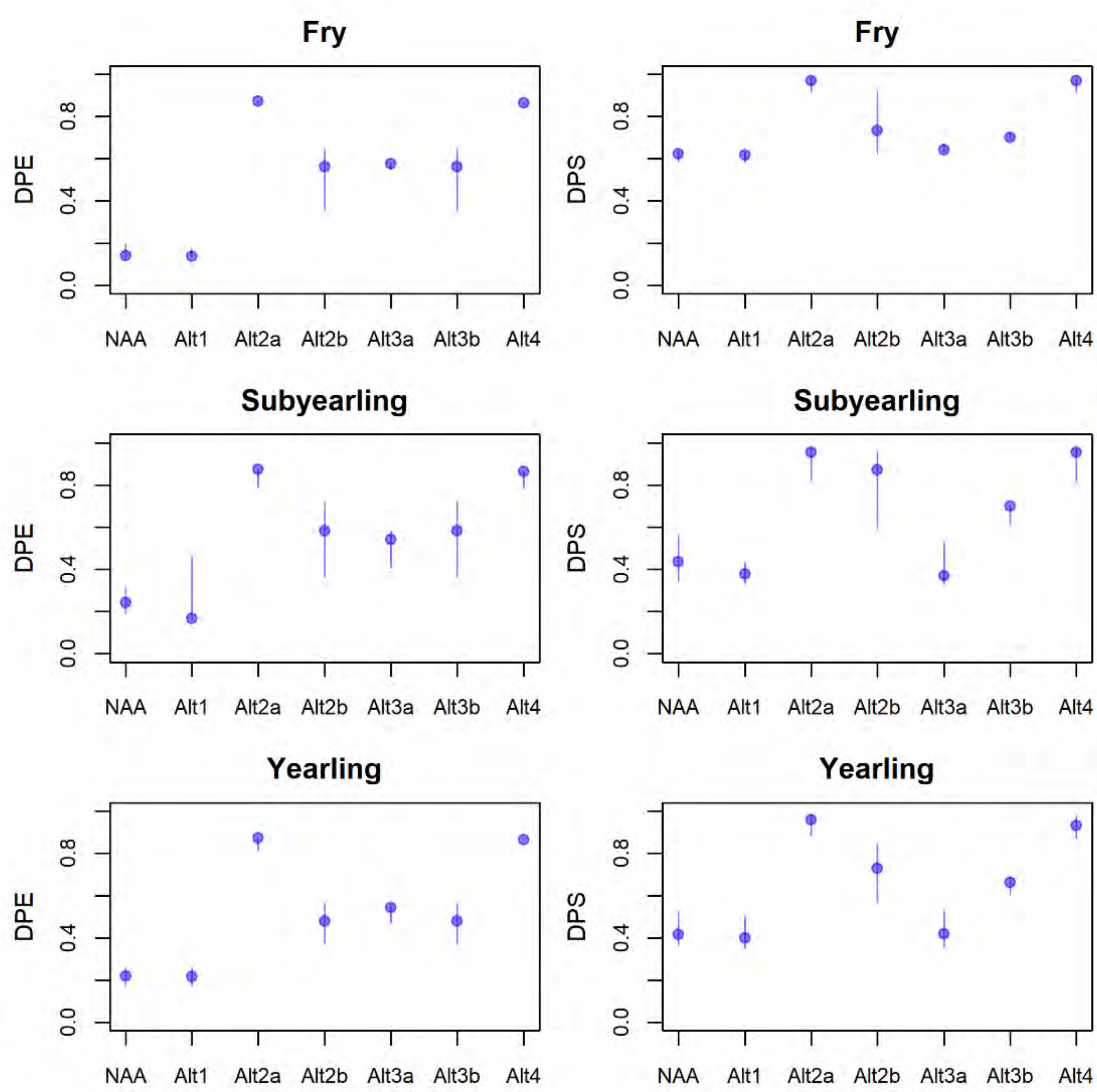

Figure 8-36. Uncertainty in dam passage efficiency (DPE) and dam passage survival (DPS) parameters provided by FBW for each age class of Chinook salmon (Fry, Subyearling, Yearling) passing Cougar dam under each EIS alternative. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs over a 30-year time period.

Middle Fork

Model calibration

We initialized the model with data on the number of NOR and HOR adults outplanted above Lookout Point in years 2002-2021 (Table 2.7.7). The model used dam passage parameters from FBW simulations of the NAA alternative for those specific years, together with the above dam temperature data, to determine PSM. The model was then fitted to available data on NOR returns to Dexter (2005-2021) and age composition from spawner surveys above Dexter reservoir (2008-2015). Middle Fork data shows very low NOR returns of over the last two decades (average 94, range 14-259, n=19; Table 2.7.7).

The calibrated model fitted the NOR returns reasonably well (Figure 2.7.30). The model was relatively difficult to fit due to the large spikes in NOR returns observed in 2007-2008 and in 2019-2020, which were almost four times the NOR returns observed in surrounding years. The calibrated parameter values for early-age marine survivals ($\varphi^{o}_{0+\rightarrow 3}$, $\varphi^{o}_{1+\rightarrow 3}$) and proportions maturing at age ($p_{0+\rightarrow 3}$, $p_{3\rightarrow 4}$, $p_{4\rightarrow 5}$) are shown in Table A.8 (Appendix A).

Table 8-14. Numbers of NOR and HOR Chinook salmon outplanted above Lookout point reservoir and the numbers of NOR returns to Dexter adult collection facility by year. Data obtained from ODFW & USACE (2016a).

Year	NOR outplants above LOP	HOR outplants above LOP	NOR returns to Dexter
2002	82	3683	82
2003	14	1683	14
2004	57	2646	57
2005	50	748	50
2006	78	749	78
2007	251	304	251
2008	259	254	259
2009	27	1094	27
2010	14	1408	14
2011	82	1659	82
2012	52	2468	52
2013	67	1899	67
2014	no data	1065	no data
2015	56	1030	56
2016	no data	687	no data
2017	34	707	34
2018	27	308	27
2019	221	338	221
2020	239	412	239
2021	80	316	80

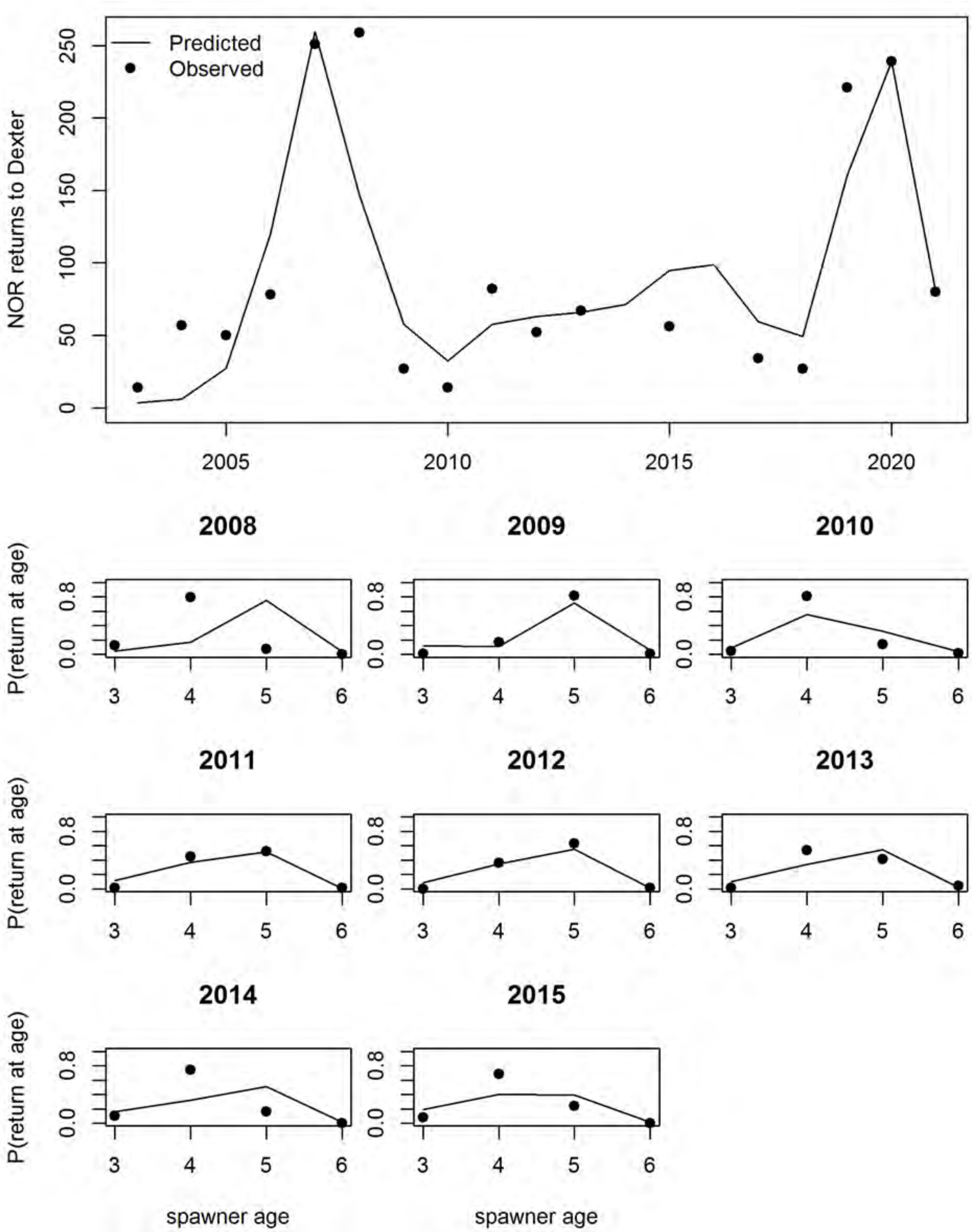

Figure 8-37. Predicted number of NOR returns to Dexter after model calibration compared to the observed number of NOR returns in 2002-2021 (upper panel). Predicted spawner age

composition after model calibration compared to the observed spawner age composition from spawner surveys above Lookout point (lower panel).

Performance Metrics for EIS alternatives

The LCM reached equilibrium within five years. Under the NAA, the projected NOR returns at equilibrium were similar to the mean observed NOR returns (Figure 2.7.31). Thus, the NAA represents the current conditions at Middle Fork. The NAA assumed that 1,257 and 387 HOR adults were available to outplant each year above Lookout point and Hill Creek, respectively. Alt2a and Alt2b assume similar outplants (1,350 and 387 HOR outplants above Lookout point and Hill Creek, respectively). While Alt1, Alt3a, Alt3b, and Alt4 follow a different outplant plan. Thus, these are not comparable with NAA, Alt2a, and Alt2b (see Table 1.2.2).

Under Alt2a and Alt2b, the increase in NOR returns (Figure 2.7.32; Table 2.7.8) compared with NAA is explained by the increase in DPS (e.g., from 0.4 to 0.95, Figure 2.7.38). The increase in DPS for Alt2a and Alt2b also reduced the risk of extinction (Figure 2.7.32 and Table 2.7.8).

Table 8-15. Middle Fork Chinook salmon performance metrics under each EIS alternative. Summary statistics are medians from 10,000 simulation runs (mean for P<QET). Definitions for each performance metric are found in Section 1.3.

	EIS alternative						
Performance metric	NAA	Alt1	Alt2a	Alt2b	Alt3a	Alt3b	Alt4
NOR LOP spawners	118	182	366	350	95	107	336
NOR HC spawners	NA	121	NA	NA	63	72	224
R/S	1.09	1.149	1.237	1.247	1.076	1.108	1.245
SAR	0.011	0.011	0.012	0.012	0.011	0.011	0.011
pHOS LOP	0.889	0.848	0.742	0.746	0.911	0.903	0.754
pHOS HC	NA	0.864	NA	NA	0.922	0.914	0.778
P<QET	1.0	0.98	0.56	0.56	0.99	0.99	0.65

Alt1, Alt3a, and Alt3b did not increase the expected NOR returns compared with NAA despite the outplanting NOR above Hill Creek (e.g., 1,100 NOR) (Figure 2.7.32 and Table 2.7.8). For subyearlings and yearlings, Alt3a and Alt3b had DPS and DPE values similar to NAA (Figure 2.7.38), which explains the lower NOR returns as this migrating group is the one that contributes most to the NOR (Figure 2.7.36). As a result, overall, the risk of extinction is close to 1.0 (Figure 2.7.32). Alt1 had higher NOR returns because the DPS and DPE were lightly higher than those for Alt3a and Alt3b (Figure 2.7.38). Alt 4 had the highest NOR returns because all migrating groups had DPS and DPE close to 1 (Figure 2.7.38). As a result, this reduced the risk of extinction to 0.1 (Figure 2.7.32 and Table 2.7.8).

Under all alternatives, the R/S indicates the population above the dam is above replacement but primarily based on HOR (Figure 2.7.32 and Table 2.7.8), so outplanting is required to maintain those levels. All alternatives had similar SAR (Figure 2.7.32 and Table 2.7.8). It is important to highlight the uncertainty associated with the performance metrics (Figure 2.7.32, Figure 2.7.33 and Figure 2.7.37), as under all alternatives, the 95% confidence interval for R/S extends down to or below 1.0, i.e., population replacement. However, all alternatives, including NAA have R/S have values >2, indicating that MF Chinook salmon could increase to twice the level of population replacement if conditions improve.

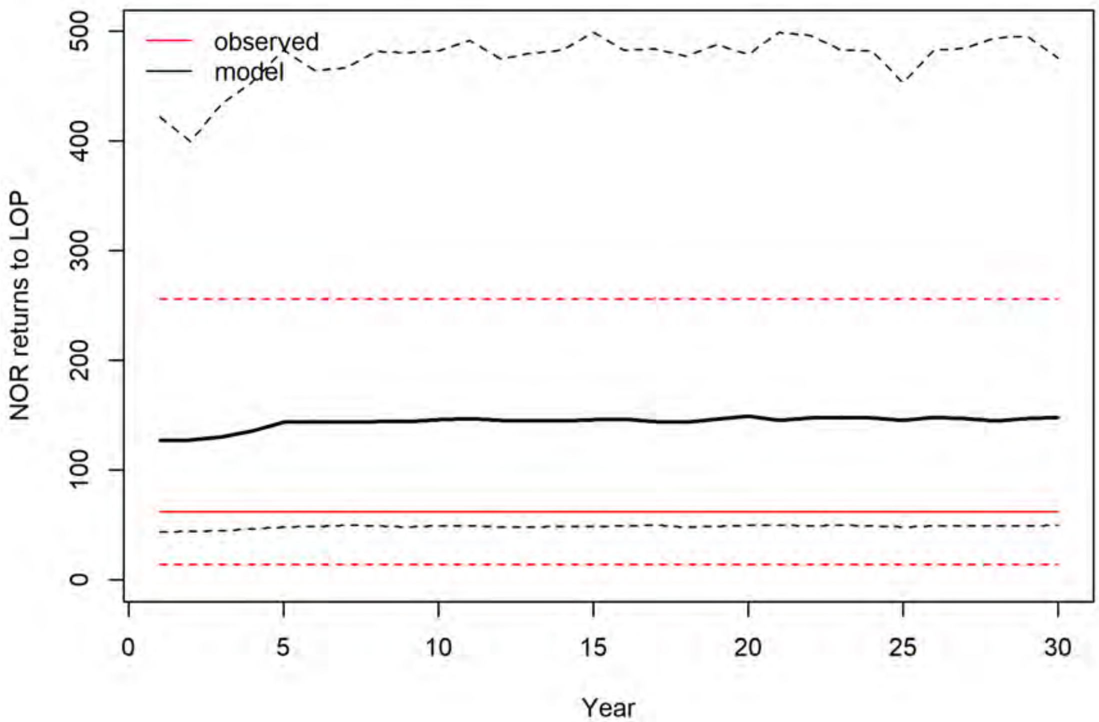

Figure 8-38. NOR returns to the Dexter-LOP projected under the NAA, showing the median (solid black line) and 95% confidence interval (dashed black lines) from 10,000 simulation runs. The median (solid red line) and 95% percentiles (dashed red lines) of the observed NOR return data (2002-2021) are shown for comparison.

The diversity of the population, as determined by relative proportions of different juvenile migrant types, was not markedly different from the alternatives (Figure 2.7.34). Subyearlings are the migrant type that contributes most to the population increase (Figure 2.7.34). These results highlight that fry migrants do not contribute to the population (Figure 2.7.36). Very few of these return as adults due to their much lower marine survival than the subyearlings group (Figure 2.7.37) (see Appendix A).

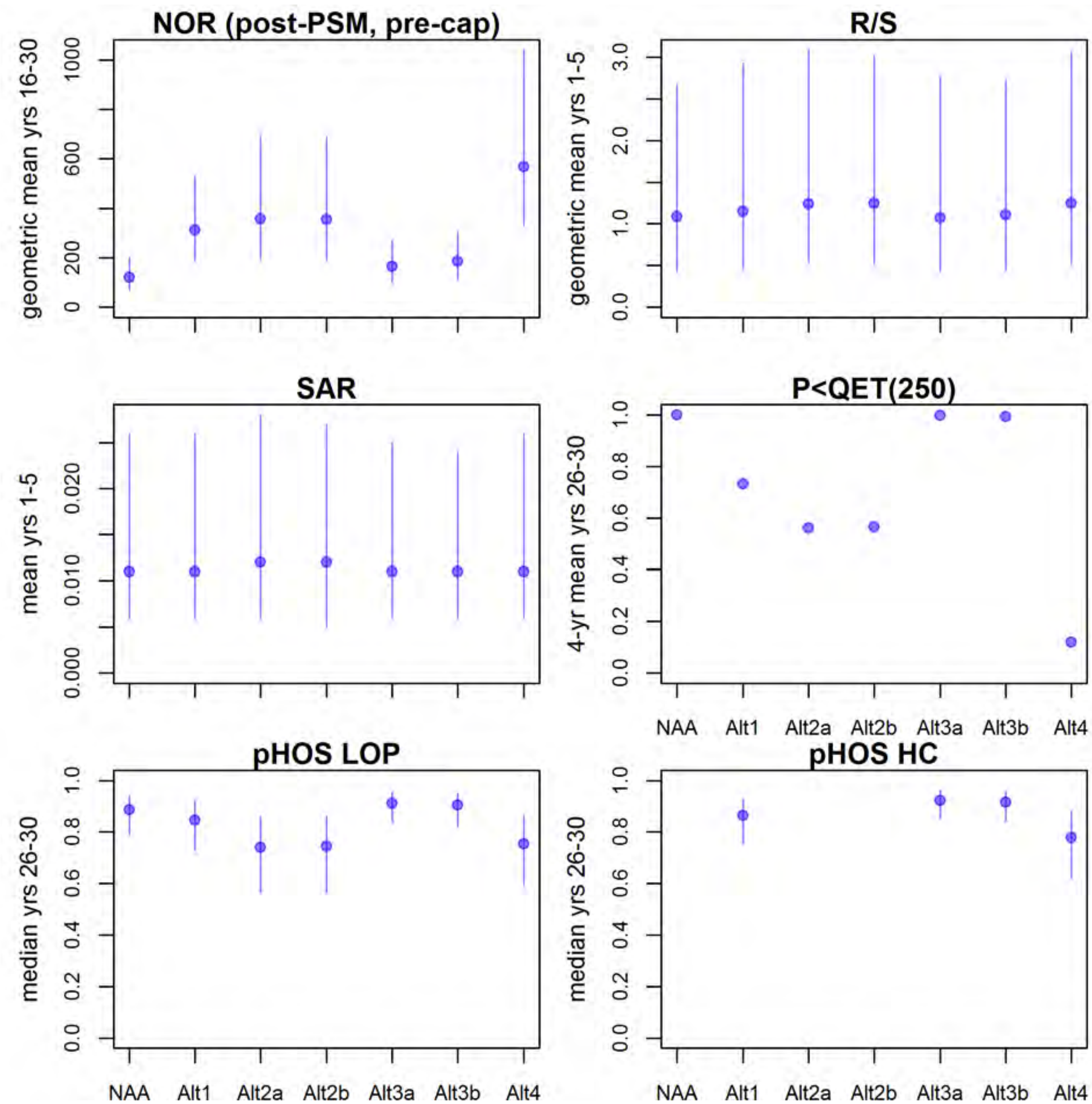

Figure 8-39. Uncertainty in performance metrics under each alternative: 1) NOR spawners (post-PSM and pre outplant cap), 2) recruits-per-spawner (R/S), 3) smolt-adult return rate (SAR), 4) probability of extinction given a quasi-extinction threshold of 250 NOR spawners, 5) proportion of hatchery-origin spawners (pHOS). Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs. Error bars are not shown for P<QET, owing to its binary outcomes.

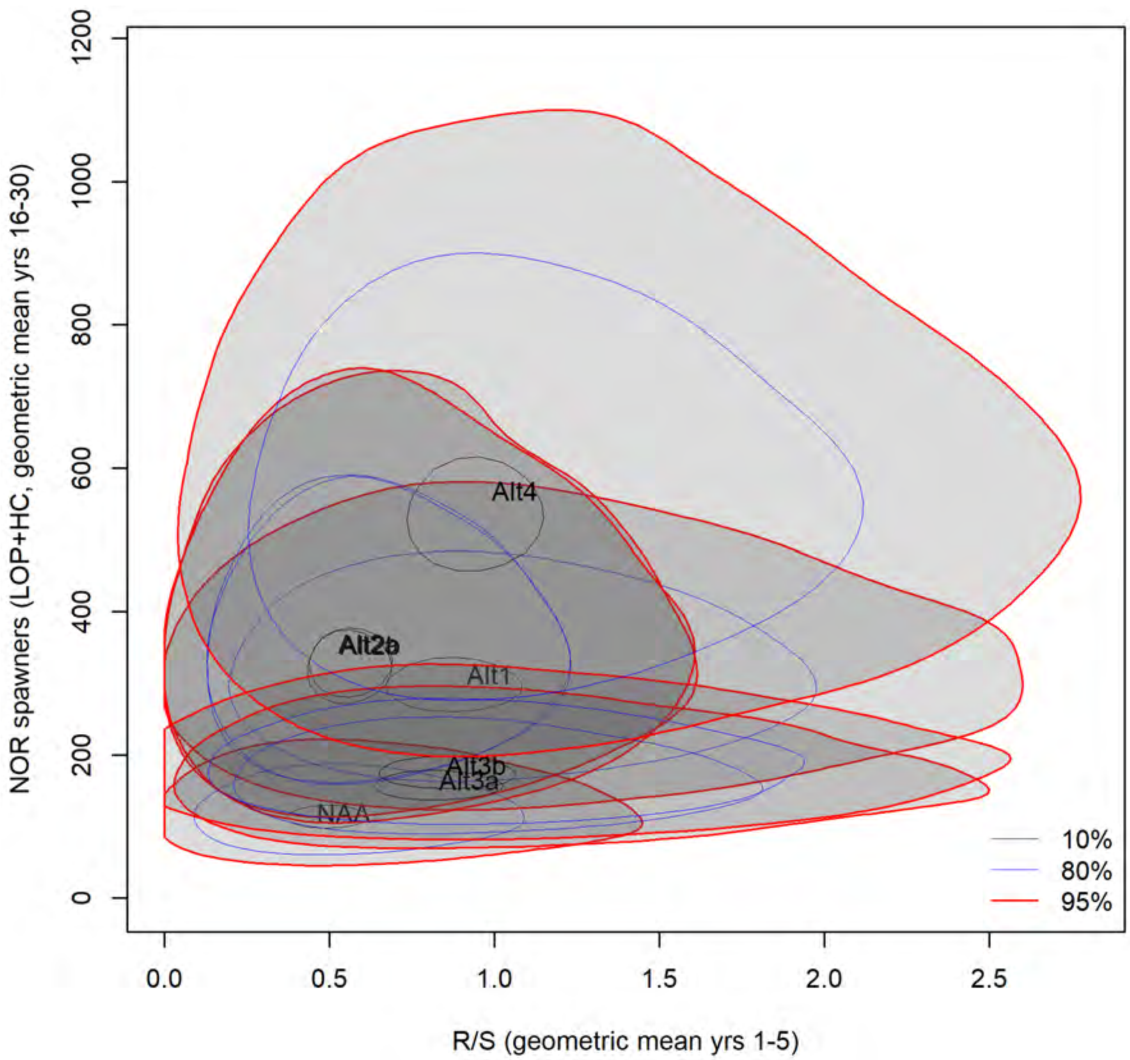

Figure 8-40. 2-D kernel density estimates (bandwidth=30) from 10,000 simulation runs to show the trade-offs between NOR spawner and R/S performance metrics under each EIS alternative in the Middle Fork.

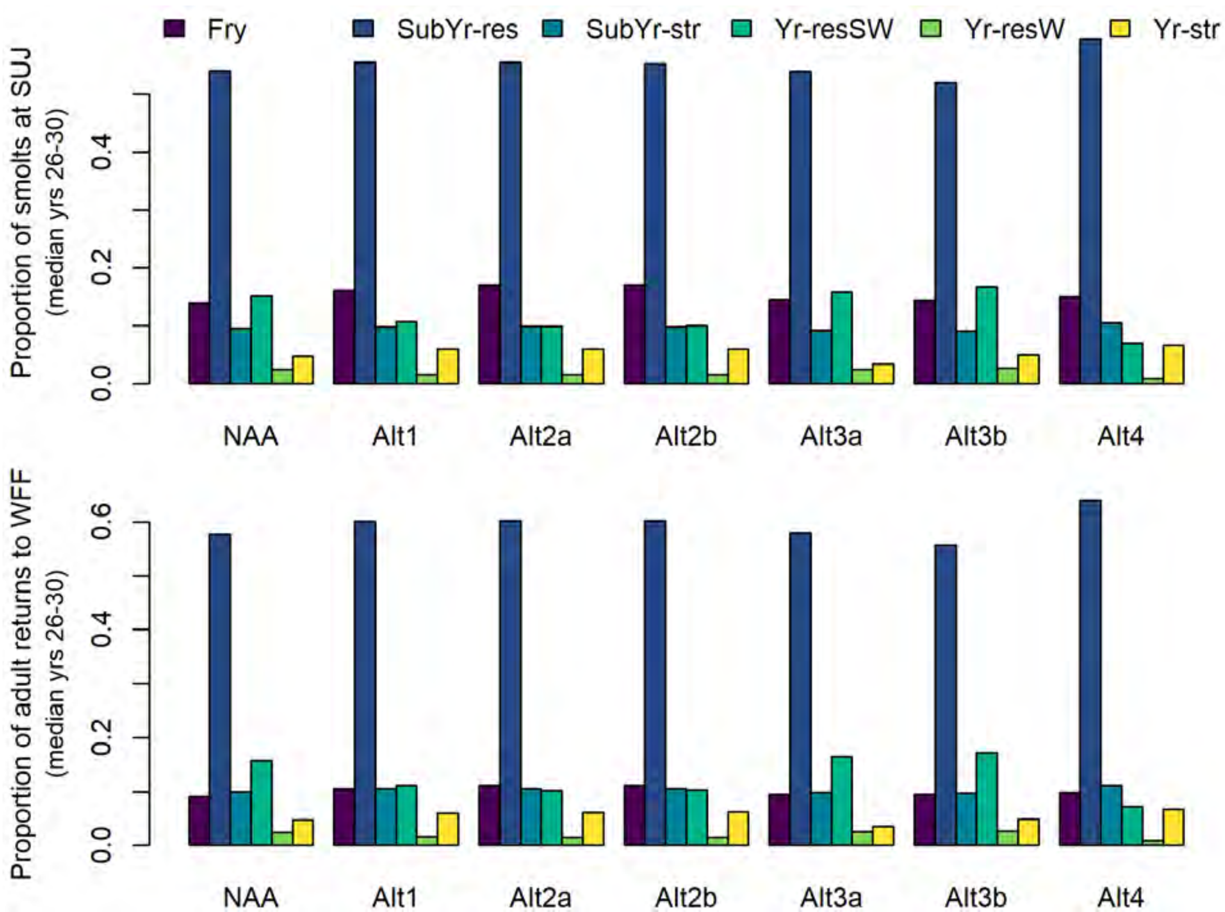

Figure 8-41. Relative proportions of different juvenile migrant types passing Dexter-Lookout point under each EIS alternative when counted as smolts at Sullivan Juvenile Facility (SUJ) or as returning adults at Willamette Falls (WFF). The migrant types modelled were Fry, Subyearlings that spent the summer in the reservoir (SubYr-res), Subyearlings that spent the summer in the natal stream (SubYr-str), Yearlings that spent summer and winter in the reservoir (Yr-resSW), Yearlings that spent only winter in the reservoir (Yr-resW), and Yearlings that spent summer and winter in the natal stream (Yr-str).

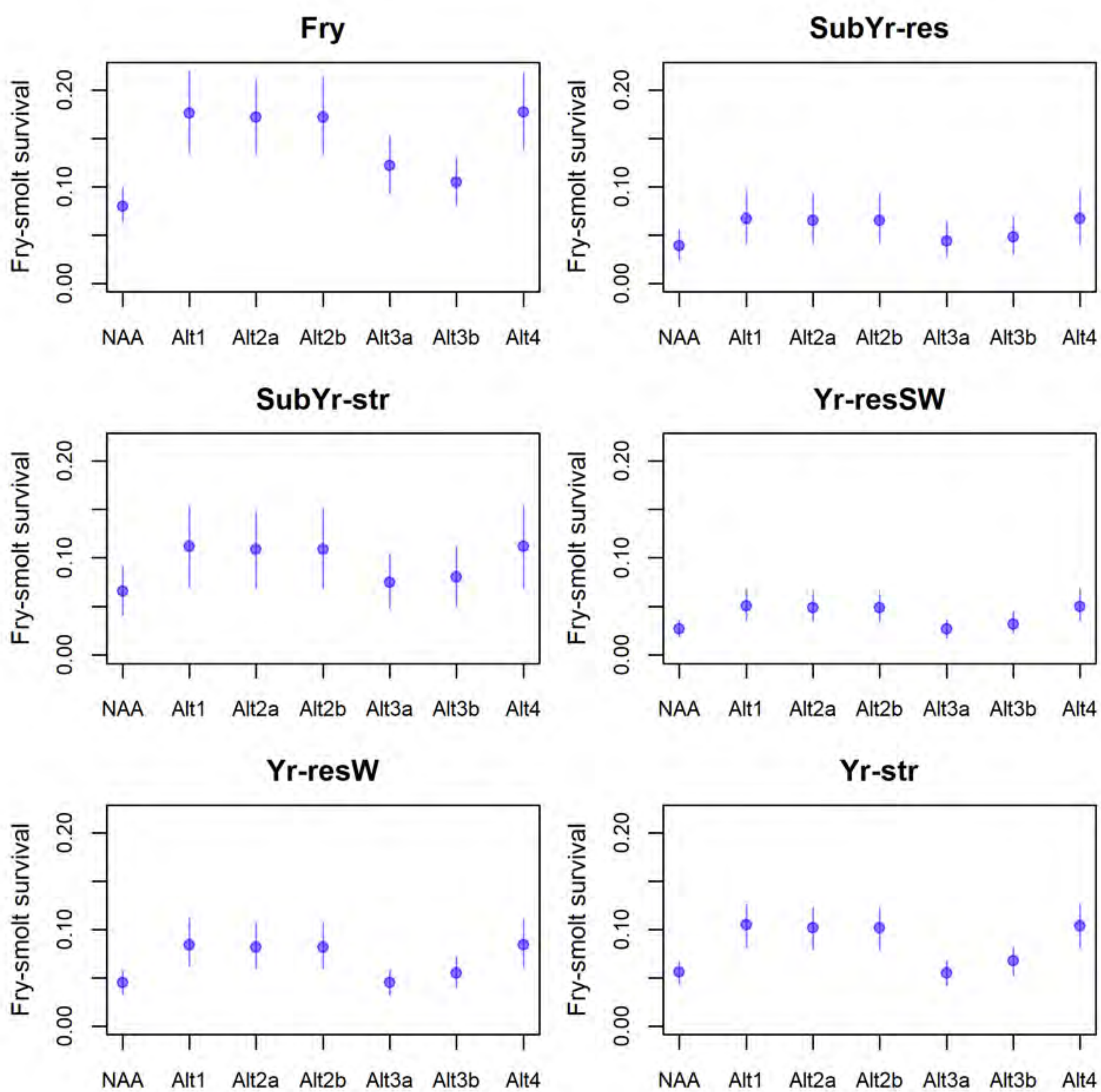

Figure 8-42. Fry to smolt survival for each juvenile migrant type passing Dexter-Lookout point under each EIS alternative. Calculated as the mean over years 1-5 of each simulation run. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs.

Fry-to-smolt survival varied between alternatives and juvenile migrant types (Figure 2.7.35). Fry had higher freshwater survival rates than other migrant groups because they spent shorter periods in the reservoir before passage. Subyearlings (SubYr-res) spent the summer in the reservoir before passing dams, while yearlings (Yr-resSW) had spent both summer and winter in the reservoir. Yearlings that only spent winter in the reservoir had comparable survival to yearlings that reared in the natal stream.

Across alternatives, fry-to-smolt survival was lowest under the NAA, Alt3a and Alt3b, which had operational passage measures with lower dam passage survival than the structural passage measures.

The relative adult return rates were lowest for the fry migrant type, but comparable across the other juvenile migrant types, and there was minimal difference between the alternatives (Figure 2.7.36).

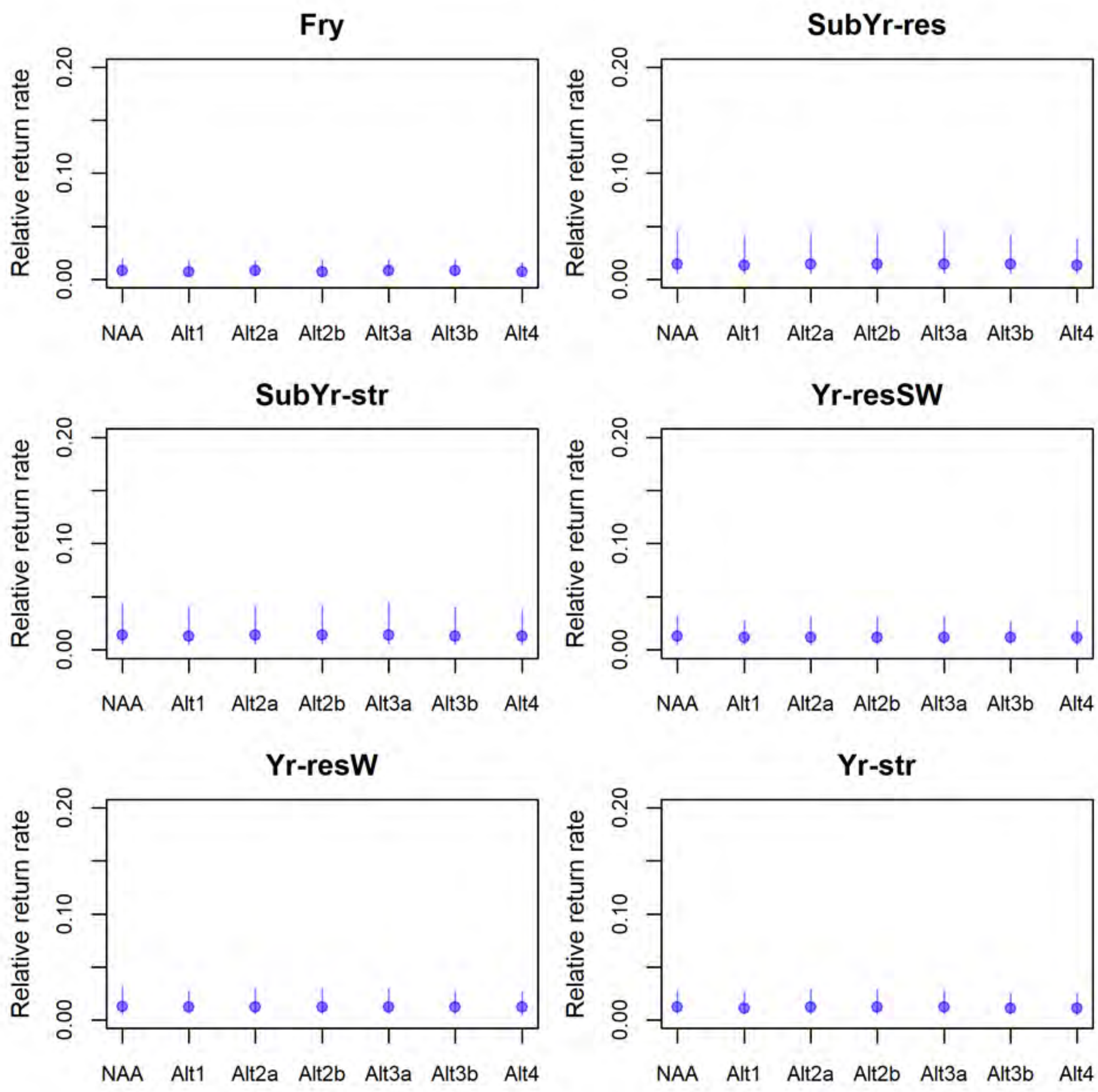

Figure 8-43. Relative return rates of each juvenile migrant type passing Dexter-Lookout under each EIS alternative. Calculated as the mean over years 26-30 of each simulation run. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs.

Parameter Uncertainties

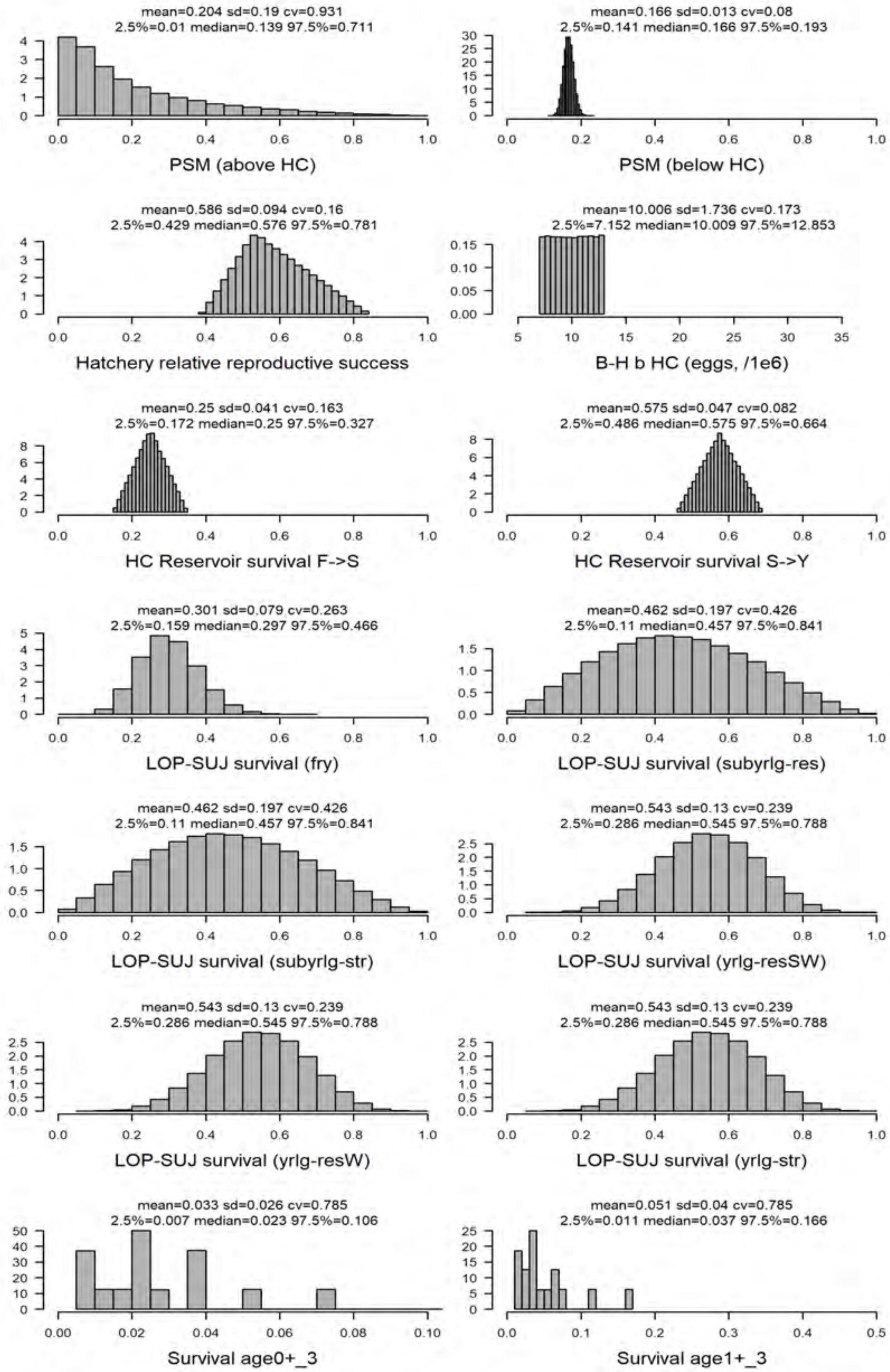
mean=0.204 sd=0.19 cv=0.931
2.5%=0.01 median=0.139 97.5%=0.711
PSM (above HC)
mean=0.166 sd=0.013 cv=0.08
2.5%=0.141 median=0.166 97.5%=0.193
PSM (below HC)
mean=0.586 sd=0.094 cv=0.16
2.5%=0.429 median=0.576 97.5%=0.781
Hatchery relative reproductive success
mean=10.006 sd=1.736 cv=0.173
2.5%=7.152 median=10.009 97.5%=12.853
B-H b HC (eggs, /1e6)
mean=0.25 sd=0.041 cv=0.163
2.5%=0.172 median=0.25 97.5%=0.327
HC Reservoir survival F->S
mean=0.575 sd=0.047 cv=0.082
2.5%=0.486 median=0.575 97.5%=0.664
HC Reservoir survival S->Y
mean=0.301 sd=0.079 cv=0.263
2.5%=0.159 median=0.297 97.5%=0.466
LOP-SUJ survival (fry)
mean=0.462 sd=0.197 cv=0.426
2.5%=0.11 median=0.457 97.5%=0.841
LOP-SUJ survival (subyrlg-res)
mean=0.462 sd=0.197 cv=0.426
2.5%=0.11 median=0.457 97.5%=0.841
LOP-SUJ survival (subyrlg-str)
mean=0.543 sd=0.13 cv=0.239
2.5%=0.286 median=0.545 97.5%=0.788
LOP-SUJ survival (yrlg-resSW)
mean=0.543 sd=0.13 cv=0.239
2.5%=0.286 median=0.545 97.5%=0.788
LOP-SUJ survival (yrlg-resW)
mean=0.543 sd=0.13 cv=0.239
2.5%=0.286 median=0.545 97.5%=0.788
LOP-SUJ survival (yrlg-str)
mean=0.033 sd=0.026 cv=0.785
2.5%=0.007 median=0.023 97.5%=0.106
Survival age0+_3
mean=0.051 sd=0.04 cv=0.785
2.5%=0.011 median=0.037 97.5%=0.166
Survival age1+_3

Figure 8-44. Histograms of parameter values used in the Middle Fork Chinook salmon model to evaluate EIS alternative, showing summary statistics for each parameter. Values are from 10,000 simulation runs.

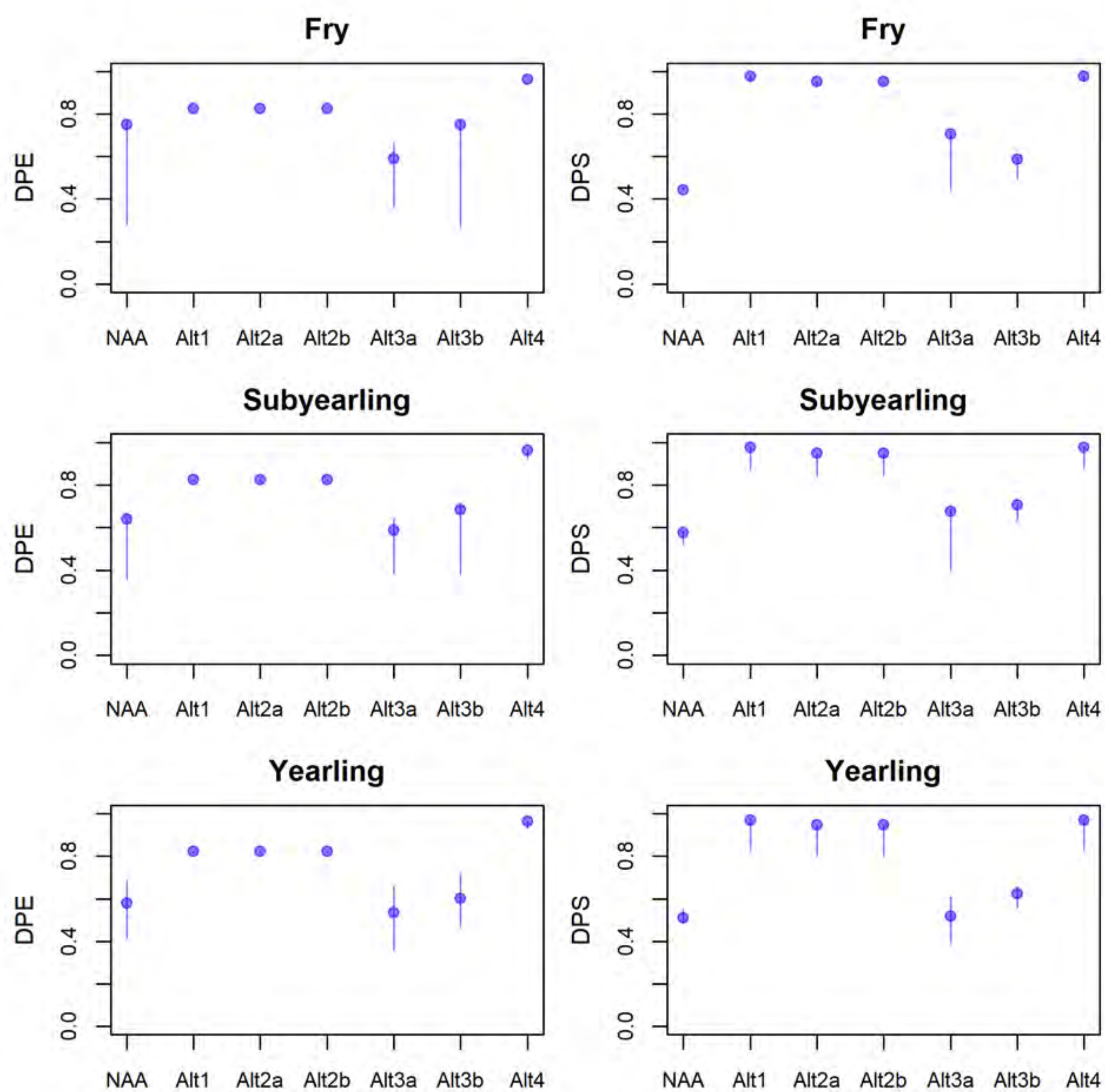

Figure 8-45. Uncertainty in dam passage efficiency (DPE) and dam passage survival (DPS) provided by FBW for each age class of Chinook salmon (Fry, Subyearling, Yearling) passing Lookout Point dam in the Middle Fork under each EIS alternative. Median (circles) and 95% confidence intervals (lines) are from 10,000 simulation runs over a 30-year time period.

Sensitivity analysis of the recovery potential of spring Chinook salmon

Input parameters for the different groups of migrant spring Chinook salmon, when possible, were estimated from PIT tag data or the multi-stage LCM fitted to data. Otherwise, they were taken from Willamette-specific reports (e.g., Romer et al. 2017; Sharpe et al. 2017) or published literature (e.g., Zabel et al. 2015). However, most of the input parameters were from reports containing estimates generated from expert opinion. The parameter estimate distributions for the many LCM parameters ranged from broad to very narrow, but overall most parameter distributions were broad and uncertain for all sub-basins. We performed a sensitivity analysis for some key model parameters to assess their effect on the predicted NOR abundance by projecting the LCM for 30 years for the NAA scenario (i.e., current conditions) for all sub-basins. The key model output we tracked as a performance metric was the geometric mean post-PSM NOR abundance for simulation years 26-30. We used the deterministic model to vary one parameter (e.g., survival or mortality rates and proportion) over a range of its plausible values (e.g., 0,…,1) while fixing the other parameters to their mean estimated value (or best guess estimate). The sensitivity analysis included juveniles survival rates for some specific migration groups such as egg-to-fry survival (α), reservoir survivals ($\varphi_{F\rightarrow S}^{resv}$ and $\varphi_{S\rightarrow Y}^{resv}$), river-smolt survival (φ_{F}^{rss} and φ_{SresS}^{rss}), proportion for migration groups ($p_{F\rightarrow F}$ and $p_{J\rightarrow F}$), and DPE and DPS (for fry, subyearling and yearling). In addition, we assessed sensitivity to survival rates for adult stages such as early adult marine survivals ($\varphi_{0^{+}\rightarrow 3}^{o}$ and $\varphi_{1^{+}\rightarrow 3}^{o}$) and PSM (below and above dams) (see Appendix A for NAA parameters).

Note it is not possible to determine how much a parameter can increase or decrease from its estimated value under different alternatives. Still, sensitivity analysis shows the relative importance of each parameter on the NOR population outcomes while the other parameters are fixed. Thus, we expanded their range to reasonable values. For example, the estimated early adult marine survivals $\varphi_{1^{+}\rightarrow 3}^{o}$ ranged from 0.039 to 0.06 (Appendix A for NAA parameter values) but we assumed that this value could increase to values < 0.3. Similarly, we assumed that other parameters could have values ranging from 0 to 1 (e.g., PSM, DPE, and DPS). In the sensitivity analysis plots below, we show the estimated value used in the LCM. Note that the ranges of values for the parameters are also uncertain.

For all sub-basins, the results show that early adult marine survivals are the most influential parameter affecting the predicted NOR abundance, in particular, the $\varphi_{1^{+}\rightarrow 3}^{o}$ (Figure 2.8.3, Figure 2.8.6, Figure 2.8.9, Figure 2.8.12). In addition, the $\varphi_{F\rightarrow S}^{resv}$ (Figure 2.8.1, Figure 2.8.4, Figure 2.8.7, Figure 2.8.10) seems to be relevant for all sub basins. DPE and DPS need to increase significantly to produce higher NORs from the reference mean estimated values (Figure 2.8.2, Figure 2.8.5, Figure 2.8.8, Figure 2.8.11). PSM (above and below dams) are also important such as higher values could generate very low NOR and even extinction, in particular, for the South Santiam (e.g., PSM > 0.4) (Figure 2.8.3). Similarly, low values of α and φ_{F}^{rss} (e.g., <0.3) could lead the South Santiam population to extinction (Figure 2.8.1). High $p_{J\rightarrow F}$ seems relevant to increase NOR at very low abundance levels (i.e., South Santiam); more emergent fry need to move and stay in the reservoir (Figure 2.8.1).

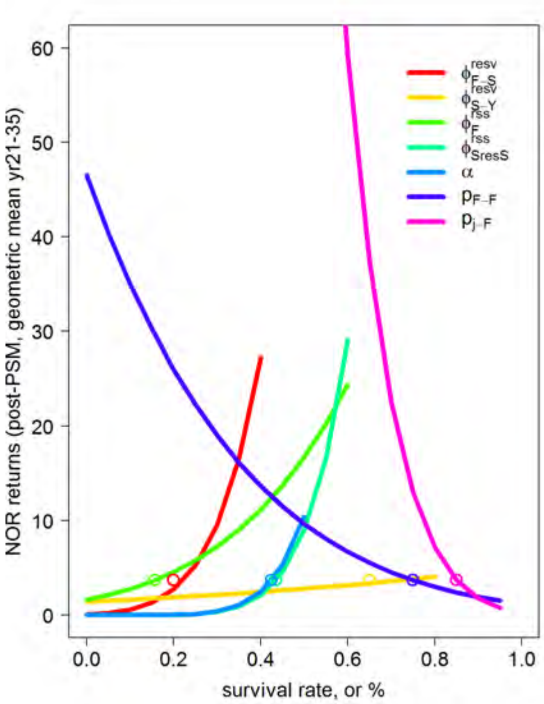

Figure 8-46. Sensitivity results for South Santiam under the NAA for juvenile stages. Dots indicate the mean estimated value.

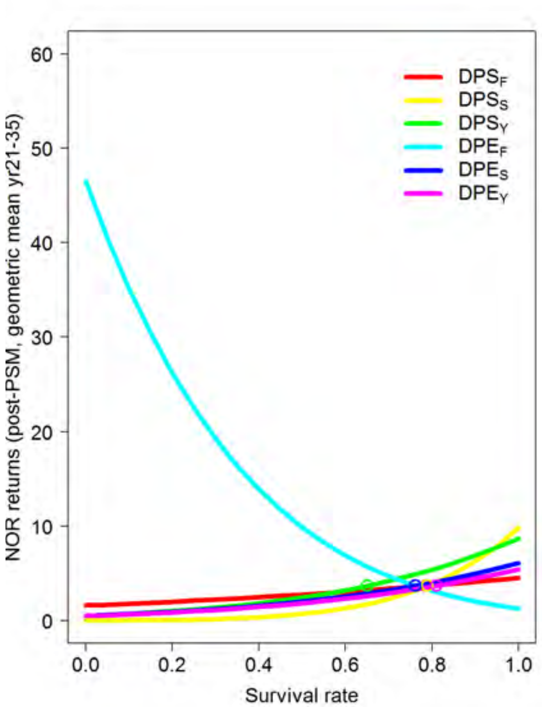

Figure 8-47. Sensitivity results for South Santiam under the NAA for juvenile stages. Dots indicate the mean estimated value.

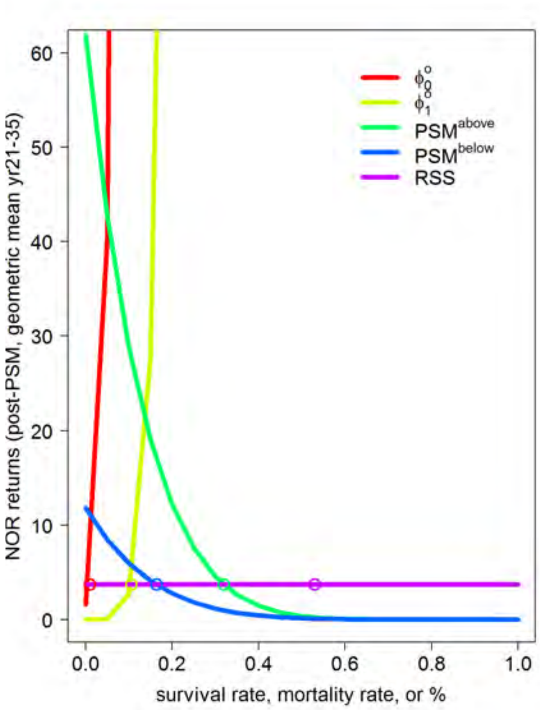

Figure 8-48. Sensitivity results for South Santiam under the NAA for adult stages. Dots indicate the mean estimated value

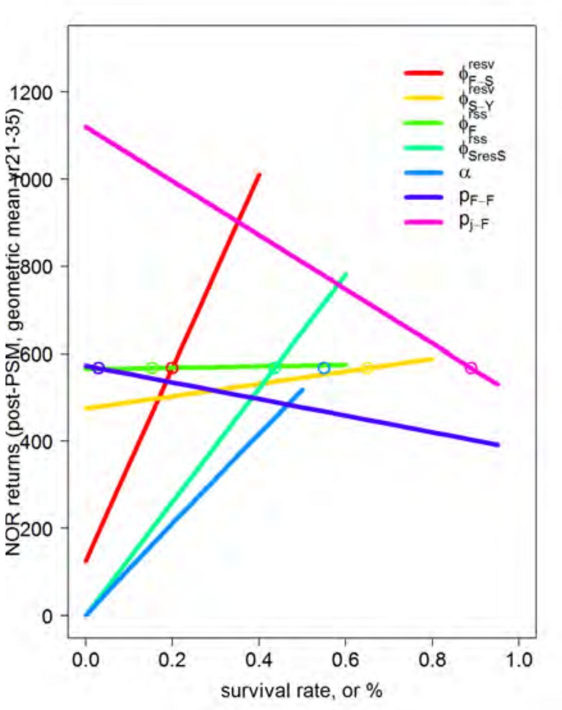

Figure 8-49. Sensitivity results for North Santiam under the NAA for juvenile stages. Dots indicate the mean estimated value.

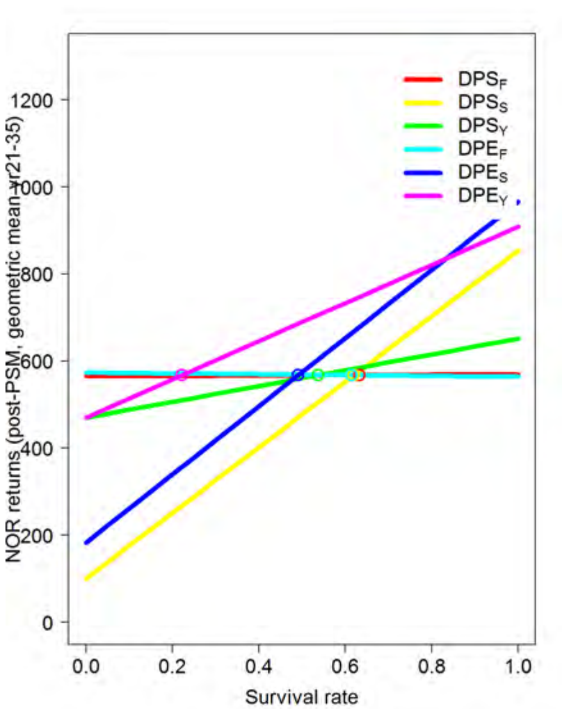

Figure 8-50. Sensitivity results for North Santiam under the NAA for juvenile stages. Dots indicate the mean estimated value.

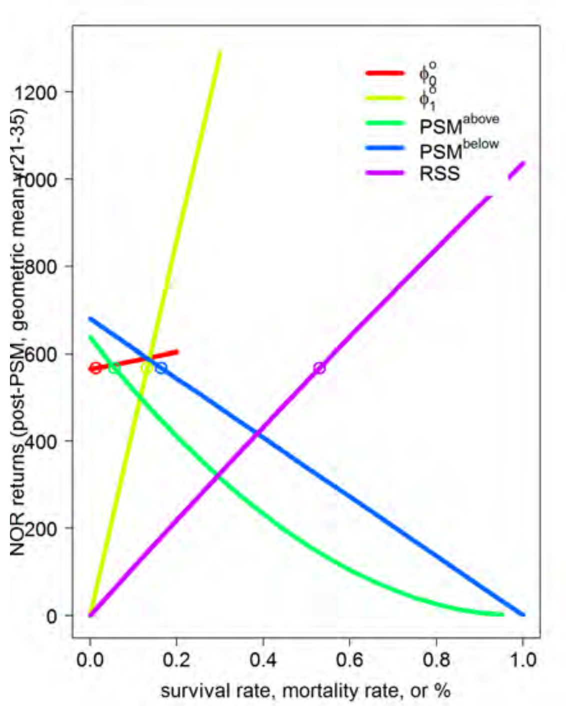

Figure 8-51. Sensitivity results for North Santiam under the NAA for adult stages. Dots indicate the mean estimated value.

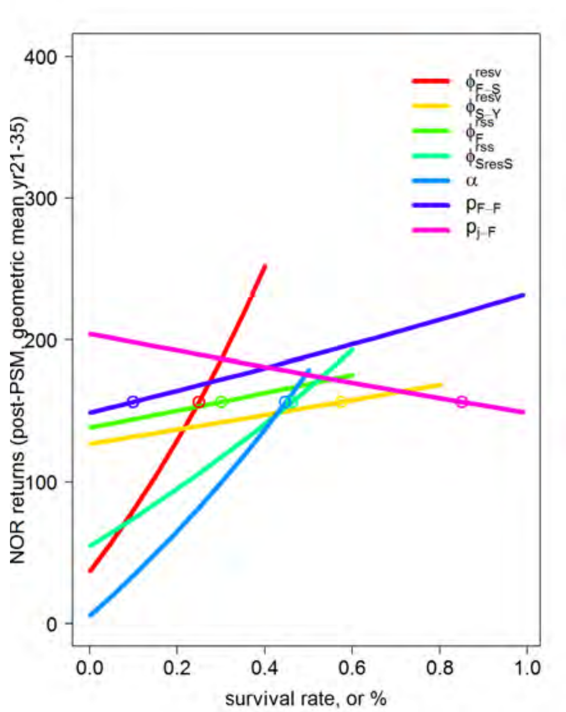

Figure 8-52. Sensitivity results for Middle Fork under the NAA for juvenile stages. Dots indicate the mean estimated value.

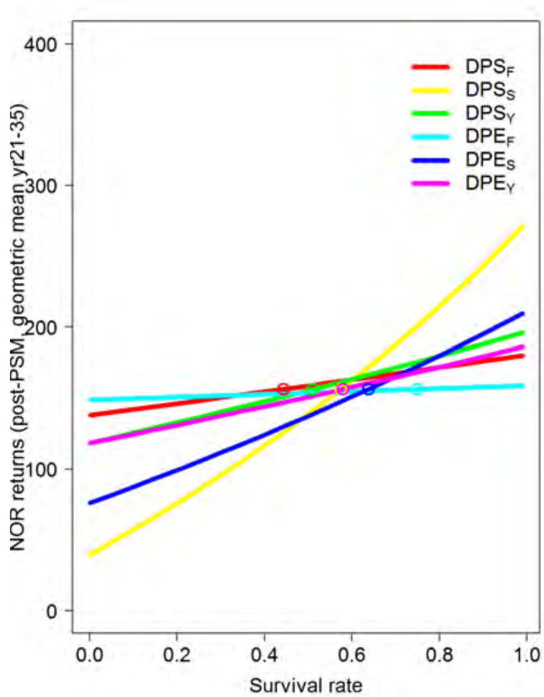

Figure 8-53. Sensitivity results for Middle Fork under the NAA for juvenile stages. Dots indicate the mean estimated value.

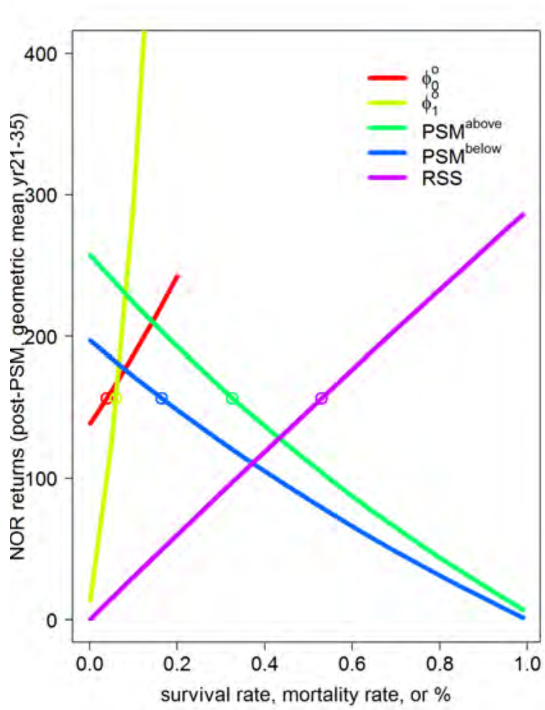

Figure 8-54. Sensitivity results for Middle Fork under the NAA for adult stages. Dots indicate the mean estimated value

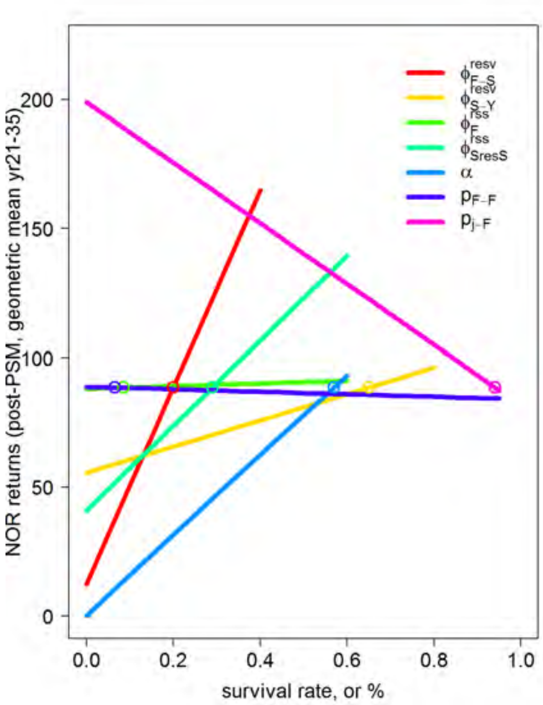

Figure 8-55. Sensitivity results for McKenzie under the NAA for juvenile stages. Dots indicate the mean estimated value.

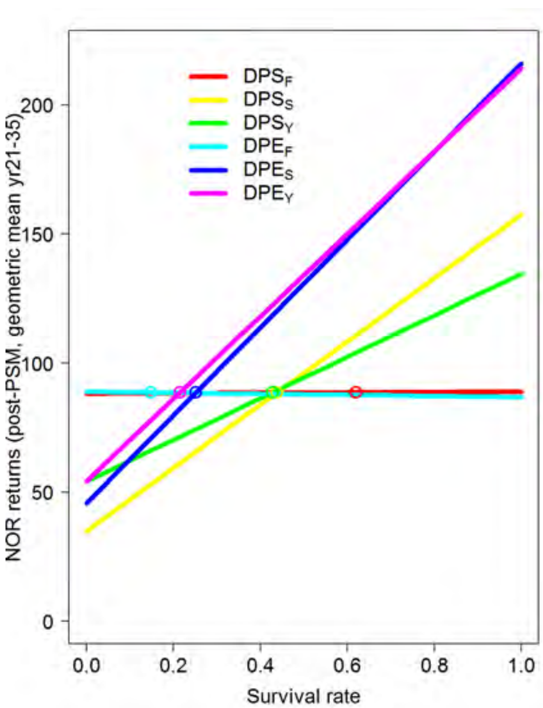

Figure 8-56. Sensitivity results for McKenzie under the NAA for juvenile stages. Dots indicate the mean estimated value.

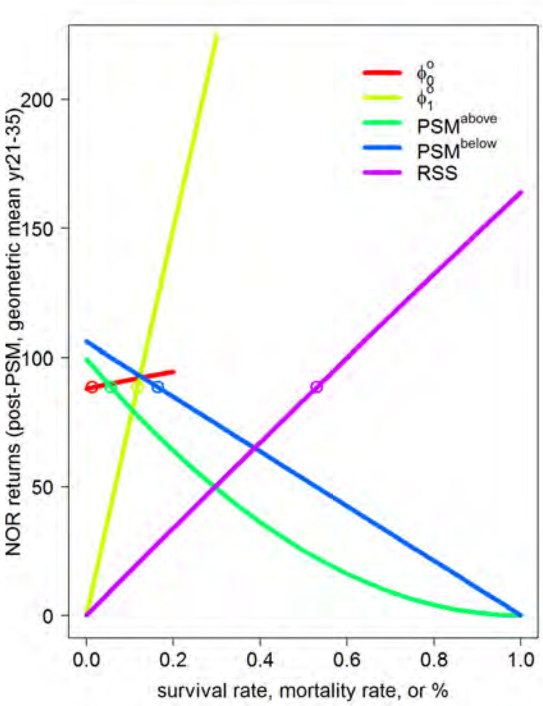

Figure 8-57. Sensitivity results for McKenzie under the NAA for adult stages. Dots indicate the mean estimated value.

Winter Steelhead Life Cycle Model

Introduction

Upper Willamette River (UWR) Oncorhynchus mykiss can be segregated into four population groups based on run type and genetic markers (Van Doornik et al. 2015). Anadromous late winter-run steelhead and non-anadromous rainbow trout are native to the Willamette, whereas early winter-run and summer-run steelhead are introduced. Winter-run steelhead are spatially segregated with early and late run fish in west and east tributaries, respectively. The UWR late winter-run steelhead population was listed as threatened in 1999 under the Endangered Species Act (NMFS 1999). Four sub-basin populations comprise the Upper Willamette River Distinct Population Segment for wild winter steelhead: Molalla, Calapooia, North Santiam, and South Santiam (Figure 3.1.1). Abundance for the four sub-basin populations (Figure 3.1.2) is estimated by proportionally allocating counts of returning winter-run adults at Willamette Falls, either through radio-telemetry (Jepson et al. 2015) or based upon redd surveys (Falcy 2017).

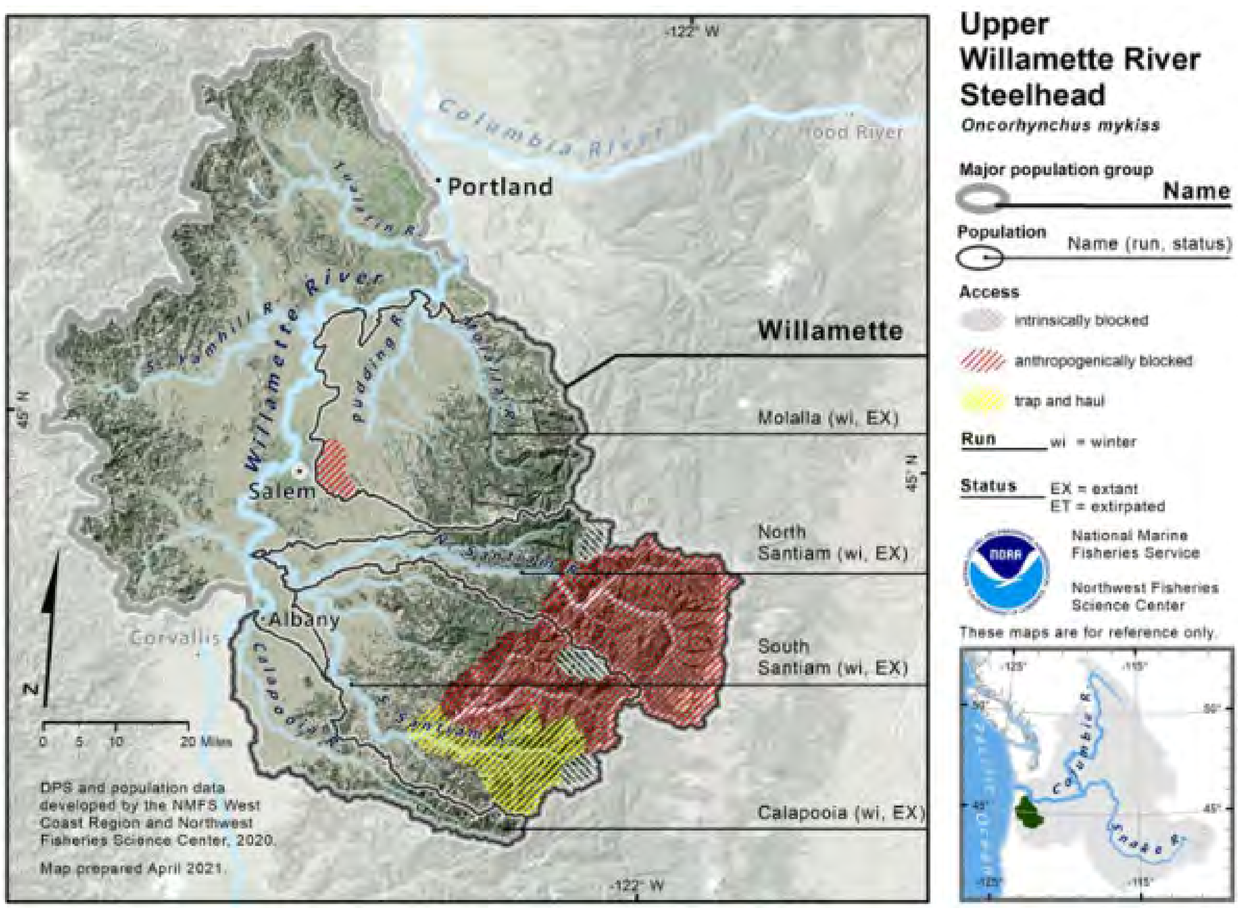

Figure 8-58. Map of the four demographically independent populations in the Upper Willamette River steelhead Distinct Population Segment (figure reproduced from Ford 2022).

The Molalla and Calapooia populations are not impacted by high-head dams but since their construction by USACE in the 1960s, the Detroit-Big Cliff dam project (North Santiam) and the Green Peter and Foster dam projects (South Santiam) have blocked migration to spawning grounds above the dams in these sub-basins (see Figure 1.1.1). In the North Santiam, an estimated 39% of spawning habitat is above Detroit dam, while in the South Santiam an estimated 63% of spawning habitat is above Foster dam (R2 Resource Consultants 2009). This has been cited as a major cause of winter steelhead population decline in these Upper Willamette sub-basins (NMFS 2008; ODFW and NMFS 2011). A summer steelhead hatchery continues for a recreational fishery, but late winter steelhead stocking ended in the 1990s (Myers et al. 2006). The percentage of hatchery-origin adults returning to North Santiam and South Santiam has been zero since 2000 and 1990, respectively (ODFW 2005). Although hatchery winter steelhead were outplanted above dams until this time, summer steelhead have never been outplanted above dams. The South Santiam wild winter steelhead population has been designated as a core population needed for population recovery (NMFS 2008; ODFW and NMFS 2011), and wild winter steelhead returns are outplanted above Foster dam. There is no outplanting of adult returns above Green Peter dam as there is currently no downstream passage implemented, and there is no outplanting above Detroit dam in the North Santiam as downstream passage is not effective.

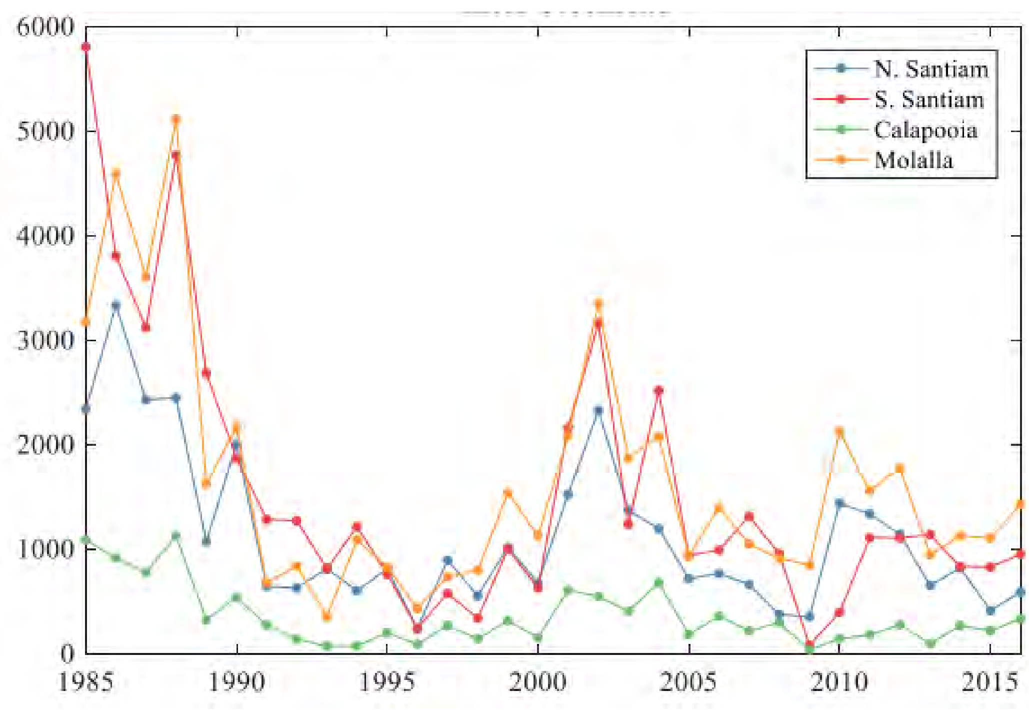

Figure 8-59. Estimated abundances of wild winter steelhead based on proportionally allocating Willamette Falls counts using redd surveys within each population of the Distinct Population Segment.

Although Willamette Falls counts are available from 1967, lack of redd surveys prevents sub-basin abundance estimation prior to 1985 (figure reproduced from Falcy 2017).

The wild winter steelhead population above Foster dam in the South Santiam is monitored by counts of returning adults at Foster adult collection facility (Figure 3.1.3). These fish can be assumed to be recruits from spawners outplanted above the dam in previous years, as all returning wild adults to Foster were outplanted above the dam. No winter steelhead were outplanted above Green Peter. Since 1990, when the last hatchery-origin winter steelhead were recorded in the South Santiam (ODFW 2005), the wild population has fluctuated by up to 1,000 fish but is currently at very low numbers following a decline that began in 2010. Only 10 wild females were recorded in 2017, with 25 recorded in 2021. A potential factor in the decline is density-independent predation mortality by sea lions below Willamette Falls, which has increased since the 1990s and has a large negative effect on the viability of UWR winter steelhead (Falcy 2017; Wright et al. 2020). In 2018, lethal pinniped management began at Willamette Falls in an attempt to reduce pinniped predation mortality (Steingass et al. 2019).

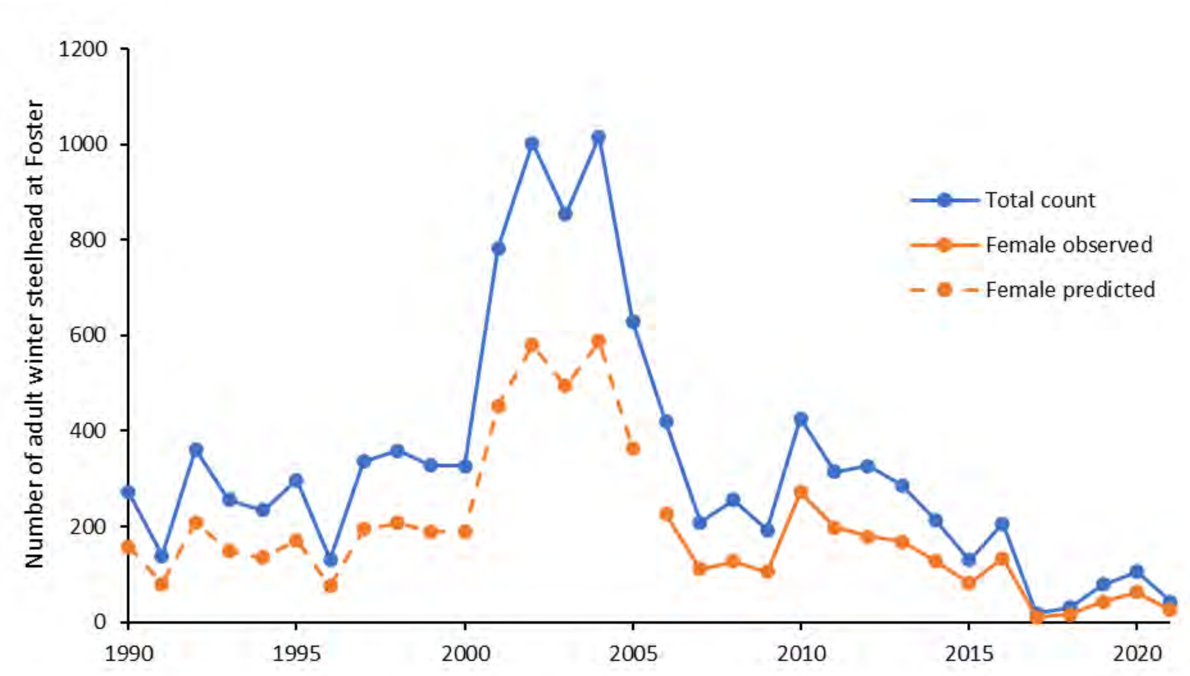

Figure 8-60. Annual count of wild winter steelhead returning to Foster adult collection facility, 1990-2021. Data from Mapes et al. (2017) and ODFW (Brett Boyd, pers. comm.).

Males and females were recorded separately from 2006 onwards. The number of females prior to 2006 was predicted using the mean sex ratio from 2006-2021, which equalled the 58% females reported by Clemens (2015) for the Willamette over the 1981-1994 period. 1990 was the last year in which hatchery-origin winter steelhead were recorded at Foster (ODFW 2005).

Winter-run steelhead are categorised as ocean-maturing steelhead, as they return to the river sexually mature and spawn shortly thereafter. In the Willamette, returning winter steelhead pass Willamette Falls from mid-February to mid-May to spawn in the headwaters in March through June, with peak spawning in late April and early May (ODFW and NMFS 2011). Juvenile steelhead emerge 8-9 weeks after spawning and rear in the headwaters for 1-4 years, though most smolt in April-May as two-year olds (ODFW and NMFS 2011). Most Willamette steelhead spent two years in the ocean, where they are thought to migrate north to Canada and Alaska and into the North Pacific (Myers et al. 2006). Winter steelhead can show a wide range of life history strategies. They are iteroparous and so can return to the ocean as kelts and subsequently return to spawn multiple times. The repeat spawning rate of wild winter steelhead in the Willamette is around 10% (Clemens 2015; Jepson et al. 2015), with repeat spawners predominantly being females (>80%, Clemens 2015) that usually spend one year post spawning in the ocean before returning to spawn (ODFW and NMFS 2011). As the anadromous form of rainbow trout, steelhead can also residualise and remain in the reservoir depending on rearing conditions (Kendall et al. 2014). Little is known about rates of residualism in UWR wild winter steelhead, particularly given the complication that dams without fish passage provide (Zabel et al. 2015).

The diversity in life history that results from different durations of freshwater and ocean rearing, age at maturation, incidence of repeat spawning, and nonanadromy can buffer fluctuations in population abundance via portfolio effects (Moore et al. 2014; Hodge et al. 2016). In order to account for most or all these life history strategies, population dynamics models for winter steelhead can become very complex but difficult to parameterize given lack of available data (e.g., Zabel et al. 2015). Our task was to evaluate downstream passage measures at all three USACE dam projects (Foster, Green Peter, Detroit) specified by the EIS alternatives (Section 1.2). Due to absence of downstream passage, there is currently outplanting only into the South Santiam above Foster, with no outplanting above Green Peter, or above Detroit in the North Santiam. Due to this, there are minimal data on winter steelhead production above dams other than for the population that spawns above Foster. In addition, it is generally recognised that there are very few studies on stage-specific survival during winter steelhead juvenile stages, with none conducted in the Willamette (Zabel et al. 2015), meaning data availability to parameterize models is poor. This lack of sub-basin specific data made construction of a life cycle model, similar to that we constructed for Chinook salmon (Section 2), very difficult.

Among the performance metrics (Section 1.3), the potential for population recovery is measured by the productivity metric, recruits per spawner (R/S). The evaluation of passage alternatives can be framed as a series of questions about R/S:

1) Is the population minimally viable under current conditions, i.e., is R/S > 1 at low spawner density?

2) If not viable, can passage mitigation measures raise R/S > 1?

3) If R/S < 1 under all passage alternatives, can additional measures, e.g., additional harvest restrictions, predator control, be used to raise R/S > 1?

These questions can be evaluated using a relatively simple survival model because maximum R/S at low density for each generation is the product of survivals for each life history stage (Moussalli and Hilborn 1986). We constructed a multistage life cycle model for winter steelhead and fitted it to spawner-recruit data from the available Foster counts, making life history assumptions about the freshwater and marine stages of the life cycle given the available data on winter steelhead in the Willamette and geographically local drainages. The fitted model was then used to project the populations in the South Santiam and North Santiam forward given the dam passage measures determined by each EIS alternative.

Model structure

We followed the approach of Moussalli & Hilborn (1986) to construct a multistage life cycle model for wild winter steelhead in the Upper Willamette River. This assumes that the life cycle consists of a sequence of contiguous, non-overlapping life stages that are linked by density-independent survival rates, and that density-dependence in the stages can be described by a single Beverton-Holt recruitment function. Although winter steelhead numbers are currently low such that density-dependence is likely not a factor, the model had to account for density-dependence should the passage alternatives under evaluation result in population recovery and increased numbers. Using the available spawner and recruit data, we developed the model for the South Santiam population above Foster, and then extended it to the populations above Green Peter in the South Santiam and Detroit in the North Santiam.

The integrated passage assessment model for winter steelhead (WS-IPA LCM) accounts for survival through three stages:

1) Freshwater survival, ϕ_F. Survival of the juvenile population from egg deposition to arrival at the Foster dam forebay, i.e., egg-smolt survival. We assumed density-dependence in this stage given freshwater capacity for smolts, B_F.

2) Passage survival, ϕ_P. Survival of the smolting population through Foster dam and downstream to Willamette Falls.

3) Marine survival, ϕ_M. Survival of the adult population in the ocean from downstream migration past Willamette Falls to just prior to egg deposition on the spawning grounds above Foster dam.

Given the product of the survival rates and fecundity, recruits per spawner (R/S) becomes:

$$\boldsymbol{R}/\boldsymbol{S} = \boldsymbol{f\phi_F\phi_P\phi_M}(\mathbf{1}+\boldsymbol{r})$$ Equation 3.2-11

where f is the number of female eggs per female, r is the repeat spawning or iteroparity ratio, and R is the number of female spawners produced by S female spawners.

The corresponding Beverton-Holt recruitment function is:

$$\boldsymbol{R} = \boldsymbol{aS}\Big/\left(\mathbf{1}+\frac{\boldsymbol{aS}}{\boldsymbol{B}}\right)$$ Equation 3.2-12

where $a = f\phi_F\phi_P\ \phi_M(1+r)$ and $B = B_F\phi_P\phi_M(1+r)$ are the stock-recruit parameters of the Beverton-Holt model.

We simplified the demography by modelling only female eggs, which we assume are half the total egg production. This decision was supported by data from the Willamette as more females than males are present in the returning adults to Foster (Figure 3.1.3) and >80% of repeat spawners are female (Clemens 2015; Jepson et al. 2015). The sex ratio discrepancy is assumed to result from high non-anadromy rates in males and poor survival of male kelts that spend more time pursuing reproductive opportunities (Fleming 1998; Keefer et al. 2018). Ignoring the males meant that we could avoid the complexity of modelling anadromous and non-anadromous males.

Both theory and observation indicate that abundance in steelhead is regulated both by density dependent survival in the freshwater stage (Keeley 2003; ISAB 2015) and by density independent survival in the marine stage, with both affected by extrinsic factors such as seasonal environmental conditions and interspecific interactions (Bailey et al. 2018; Scheuerell et al. 2021). Thus, of the three stages described above, only ϕ_F is assumed to be density dependent and the number of smolts (NS) produced by NE eggs follows a Beverton-Holt recruitment function:

$$\boldsymbol{N_S = a_F N_E/(1 + a_F N_E/B_F)}$$

Equation 3.2-13

where aF is the freshwater productivity parameter at low density and BF is the freshwater capacity parameter. This relationship is assumed to be the result of a sequence of egg-fry, fry-parr and parr-smolt Beverton-Holt recruitment functions, with independent productivity and capacity parameters, that can be combined into a single composite Beverton-Holt recruitment function (Moussalli and Hilborn 1986). The productivity parameter aF is the product of the independent productivity parameters ai across all juvenile stages, where i is life stage. However, the capacity parameter BF is a complex function of both Bi and ai, which makes the estimation of BF as a composite parameter problematic. However, if we assume that only ϕ_F is density dependent, then the capacity parameter over the entire life cycle in Equation 3.2-2 will be the product of BF and the subsequent survival terms:

$$\boldsymbol{B = B_F\phi_P\ \phi_M(1 + r)}$$

Equation 3.2-14

We used Equation 3.2-2 to estimate ϕ_F as a free parameter under current conditions, assuming the other parameters are known (see Sections 3.2.1-3.2.3 for stage-specific details on parameter specifications and assumptions, Section 3.3 for details on model fitting). Given density-independence in the passage survival and marine survival stages, by assuming these are known we can remove their effects, meaning that the freshwater survival stage, i.e., ϕ_F, will account for all other mortality effects. This is complicated by two main factors: age structure of the recruit generation and variable marine survival.

To consider age structure, we first assumed that all juveniles smolt at age-2 (see Section 3.2.1 for discussion of data to support this). We then assumed that female adults spent at least two years in the ocean (Clemens 2015) so that the virgin spawners would first return at age-4. Considering repeat spawners as old as age-7, the R/S equation (Equation 3.2-1) can be rewritten as:

$$\boldsymbol{R_{y+j}} \Big/ \boldsymbol{S_y} = \boldsymbol{f\phi_F(1+r)\sum_{j=4}^{6} c_j\ \phi_{P(y+2)}\ \phi_{M(y+2)}}$$ Equation 3.2-15

where y is the brood year, Ri+j is the number of recruits of different ages from the yth brood year that return in year y+j. Each cj is the proportion of age j virgin female spawners in each cohort. We assumed that the values of c4, c5 and c6 are 58.6%, 24.6% and 2.5%, respectively, with repeat spawners aged 5-7 making up the remaining r=13.8% of recruits from each brood year (see Section 3.2.3 for spawner age structure details). Cohorts are all assumed to have identical age structures.

We incorporated variable marine survival by using an index of marine survival rate, which varies with an apparent approximate decadal periodicity (see Appendix E for construction of the marine survival rate time series).

Freshwater Stage

The freshwater stage of the winter steelhead IPA LCM begins with egg deposition and ends with smolts reaching the dam forebay. It therefore includes parameters for fecundity, freshwater survival, and freshwater capacity.

The fecundity of virgin and repeat spawners is expected to be different due to the larger size of repeat spawners that have spent additional time growing in the ocean. There is limited data on relationships between age or size and fecundity, which may differ between virgin and repeat spawners. A recent Snake River study found that growth between spawning events was less for females that were large when virgin spawners, indicating that larger females did not accrue fecundity on the second spawn as much as did smaller females (Copeland et al. 2019). An additional factor is egg size, which increases with female age at spawning and in repeat spawners (Quinn et al. 2011). Repeat spawners contribute more to the population than virgin spawners (Seamons and Quinn 2010); because egg size is positively correlated with early growth and survival (Einum and Fleming 1999) this contribution may be greater than expected based upon fecundity alone.

In absence of Willamette-specific data on age-related fecundity, we assume the fecundities for virgin and repeat spawners are equivalent to age-weighted average fecundities. Based on the fecundity values reviewed by and used in Zabel et al. (2015), we set virgin spawner fecundity at 4,000 eggs/female (η_{virgin}) (assuming 50% of virgin spawners are female) and repeat spawner fecundity at 5,200 eggs/female (η_{repeat}) (assuming 80% of repeat spawners are female). The average number of female eggs per female will be a function of virgin and repeat fecundity and their ratio in the spawning population (see 3.2.3 for determination of iteroparity rates). Assuming that 50% of eggs will be female, the average number of female eggs per female will be:

$$\boldsymbol{f} = \left(\%_{\boldsymbol{virgin}} * \boldsymbol{\eta}_{\boldsymbol{virgin}} + \%_{\boldsymbol{repeat}} * \boldsymbol{\eta}_{\boldsymbol{repeat}}\right)/\mathbf{2}$$

Equation 3.2-16

Given the above fecundities η for virgin and repeat spawners, and that 13.8% of female spawners are repeat spawners (Section 3.2.3; Clemens 2015), f is equal to 2,083 female eggs per female.

One model uncertainty in the winter steelhead IPA LCM concerns the role of non-anadromous females in maintaining egg production under poor smolt to adult survival. Almost all the O. mykiss juveniles from the South Santiam River upstream of Foster Dam were either native winter steelhead (82%) or winter steelhead x rainbow trout cross (12%, Johnson et al. 2021). Since the majority of hybrids involve a small male and a large female, egg production from genetically independent resident rainbow trout females can be ignored. However, steelhead of both sexes may adopt a non-anadromous life history, which leaves open the possibility that female non-anadromous steelhead may contribute to egg supply. Progeny of non-anadromous and anadromous steelhead can be distinguished by trace element analysis of the otolith nucleus (Zimmerman and Reeves 2002; Courter et al. 2013; Kendall et al. 2015), but this information is not currently available for Willamette steelhead populations. Until further information becomes available, we assume that egg production depends entirely on anadromous females that are passed upstream of dams and note that Monzyk et al. (2017 p. 4) make a similar assumption.

There are several survival components to ϕ_F, namely egg-fry, fry-parr, and parr-smolt survival. These processes may occur in the headwater streams where spawning occurs, or after downstream movement into the reservoir. There is very little information about these survival rates. In the 2015 COP Appendix, the life cycle model of Zabel et al. (2015) applied values for egg-fry survival, fry-migrant survival (of age-0, age-1, and age-2 fish), and reservoir survival (of age-0, age-1 and age-2 fish) obtained from the literature and parameter workshops involving experts on steelhead. Our simpler model structure combines these survivals into one survival rate parameter ϕ_F, which we estimate as the probability density function for a free parameter during model fitting.

An assumption about ϕ_F is that all winter steelhead smolts are age-2 and migrate downstream in the spring, which we support with data from the literature as detailed below. We note that data on juvenile migration timing comes only from the South Santiam above Foster dam, given the lack of winter steelhead production above other dams. Captures in rotary screw traps (RST) between 2011 and 2016 indicated that of juvenile steelhead entering Foster reservoir, 75.5% were age-0 fish, 20.5% were age-1 fish, and 4% were age-2 fish (Romer et al. 2012, 2013a, 2014, 2015, 2016, 2017). In these RST studies, captures of age-0 fish above Foster reservoir began in late June, shortly after emergence, and continued throughout the year. Age-1 fish were captured in spring and fall, with a variable peak in movement timing between years. Age-2 fish were only captured above Foster in the spring. This indicates that most juvenile steelhead have moved into the reservoir prior to age-2, with those few that migrate as age-2 assumed to be smolts.

The RST located below Foster, in the turbine tailrace, typically captured very few age-2 smolts, but consistently caught fish in the fall, mostly age-0 and some age-1 (Romer et al. 2012, 2013a, 2014, 2015, 2016, 2017). This is in contrast to PIT-tag data from a study on juvenile winter steelhead tagged after capture in the above Foster reservoir RST. No age-0 fish were detected passing the Foster weir antenna and <3% of tagged fish passed the dam in the fall (Monzyk et al. 2017), similar to radio-telemetry results from Hughes et al. (2016). Nearly all (97%) of passage at Foster dam occurred from March through June (Figure 3.2.1), with a peak in May (Monzyk et al. 2017). Of the PIT-tagged fish that passed the dam, 84% were age-2, with most tagged age-0 and age-1 fish rearing for at least one additional year before migrating past the dam (Monzyk et al. 2017). The 84% age-2 smolts compares well to the 81% of wild Willamette winter steelhead that scale analysis revealed had spent two years in freshwater (Clemens 2015). Other than differences in survey design (i.e., RST located in turbine tailrace, PIT antenna located on fish weir), a reasonable explanation for screw traps catching higher numbers of age-0 in the fall is that age-0 fish are more likely to be entrained by the turbine penstocks than older juveniles, i.e., unlike the PIT-tagged fish, these age-0 fish do not represent downstream migrants.

These results from screw trap and tagging studies indicate that many juvenile winter steelhead enter Foster reservoir at age-0, but do not pass Foster dam until age-2. Thus, although clearly not 100% of passage through the dam is by age-2 fish in the spring, we can justify the assumption that all smolt production takes place above Foster dam before they exit as age-2 smolts. This means we can avoid modelling the younger age classes and their passage survival. Any juveniles that do not smolt at age-2 are assumed to either be lost to the forebay or residualize in the reservoir.

Under the assumption of smolting at age-2, the passage of non-smolting juvenile steelhead of younger ages would represent losses to the population rather than potential for additional smolt production downstream of the dam. We therefore assumed that smolt production depends only on the habitat capacity above dams to specify parameter BF in the Beverton-Holt recruitment function in the winter steelhead IPA LCM. We used estimates of winter steelhead parr capacities by reach (Bond et al. 2017, Table 3.6) and set the above dam smolt capacities at 50% of parr capacity. Capacities included only mainstem and current side channel habitat and, for the capacity above Foster, did not include habitat upstream of Green Peter. The number of smolts in a reach following the winter will necessarily be lower than the number of parr in that reach the previous summer, so we assumed that smolt capacity was 50% of parr capacity based upon parr-to-smolt survival being 0.5. There is no estimate of this for Willamette steelhead so we based it upon the value of 0.49 estimated for steelhead in the Keogh River (Tautz et al. 1992). We note this is more conservative than the value of 0.3 used by the Puget Sound Steelhead Technical Recovery Team (Hard et al. 2015). Given our assumption and the values in

Bond et al. (2017), the habitat capacity above Foster was 76,473 smolts, above Green Peter was 42,596 smolts, and above Detroit was 112,833 smolts.

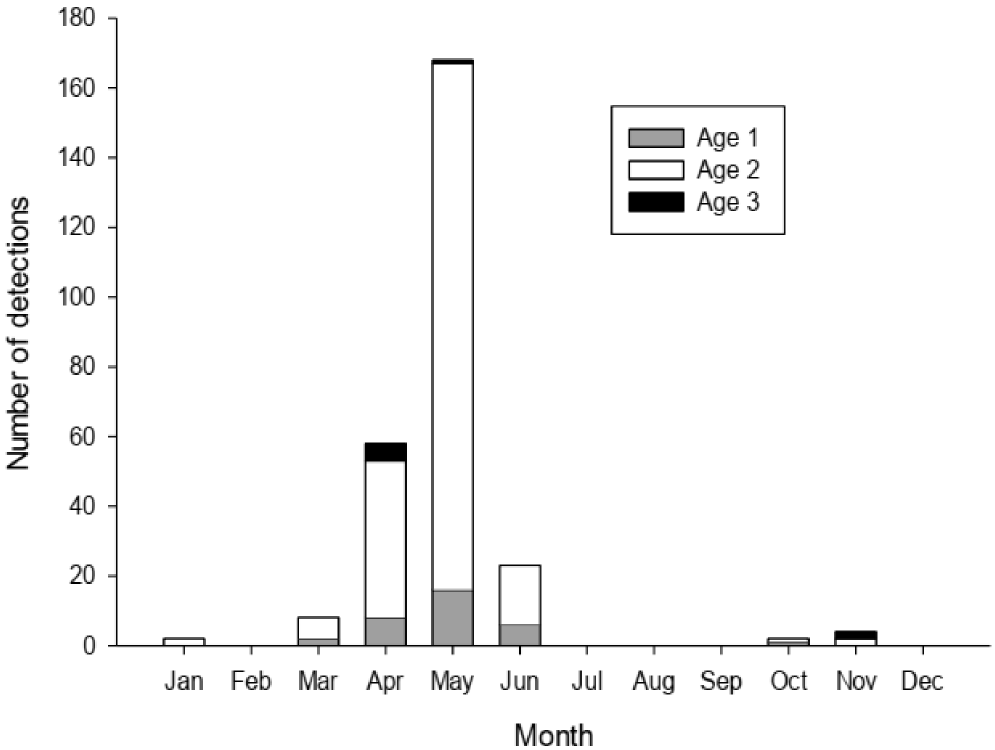

Figure 8-61. Monthly detections by age of PIT-tagged juvenile winter steelhead passing Foster dam, 2014-2017 (figure reproduced from Monzyk et al. 2017).

Passage Stage

The passage stage of the WS-IPA LCM includes components related to dam passage and survival of smolts during downstream migration to Willamette Falls. Age-2 steelhead smolts in the dam forebay must first find a route through the dam, then survive passage through that route. The first process is determined by DPE, the proportion of fish detected in the forebay near the dam face that are subsequently detected downstream of the dam. The second process is determined by DPS, the proportion of fish that survive dam passage, weighted by the proportion of fish using each route (i.e., turbine, spillway, etc.) and the survival for each route. These dam passage parameters were provided for age-2 steelhead by the FBW (Section 1.4). DPE and DPS were not required for age-0 and age-1 fish as these age classes were not modelled.

Similar to freshwater survival above dams, no estimates of downstream migration survival from dam tailraces to Willamette Falls were available for winter steelhead, with few studies on downstream migration survival available from elsewhere. The survival rate for this below-dam stage of the life cycle is likely to be relatively high. Given the distance the travel time is short, of PIT-tagged steelhead smolts passing Foster dam the mean travel time to Willamette Falls was only 5.9 days (Monzyk et al. 2017). Short travel time is also indicated by comparison of the monthly distribution of detections of steelhead smolts at Willamette Falls (Figure 15, Romer et al. 2016) to the distribution of age-2 smolts passing Foster dam (Figure 3.2.1), which show an almost identical distribution March-June. Data on date of ocean entry derived from trawl samples of steelhead in the Columbia River estuary and ocean plume, which use genetic information to identify Willamette River steelhead, show a similar mean ocean entry date of 13 May (Weitkamp et al. 2015) to the timing of steelhead smolts passing Foster dam (Monzyk et al. 2017). It therefore appears that steelhead smolts that pass Foster dam do not reside for any length of time above or below Willamette Falls, and once migration is started above the dam it continues to the ocean. We note this supports our assumption that few steelhead produced above dams spent much time rearing downstream.

In addition to short travel times indicating high downstream migrant survival rates, i.e., due to lowered predation risk, age-2 smolts are large enough to avoid significant instream predation. Data from trawl samples in the Columbia River estuary suggest that wild Willamette steelhead smolts have a mean length of 158 mm (Weitkamp et al. 2012, 2015; Daly et al. 2014), which compares to size of steelhead migrants from Wind River, located just above Bonneville dam, having a mean length of 161 mm (Wilson et al. 2021). Steelhead smolts from Keogh River on Vancouver Island averaged 171 mm (Friedland et al. 2014).

The age-2 river migrant survivals used in the 2015 COP ranged from 0.6 for smolts produced above Foster and above Detroit, to 0.65 for smolts produced above Green Peter (Zabel et al. 2015 p. 8.12 & 9.16). These values apparently come from expert workshops rather than tagging studies. The only empirical estimate available comes from analysis of PIT-tag data from a 2014 release of hatchery summer steelhead in Foster dam tailrace (ODFW, unpublished results). We used the Cormack-Jolly-Seber (CJS) model to estimate survival from release to Willamette Falls (Appendix C). The posterior mean estimate of ϕRSS for summer steelhead was 0.705 (95% CI: 0.491-0.954). Although the timing of migration was similar to the age-2 winter steelhead smolts observed in South Santiam (Romer et al. 2016; Monzyk et al. 2017), this is likely an underestimate of the age-2 survival rate as the release was of age-1 fish that had a mean length of 85 mm. Downstream survival of wild Snake River steelhead smolts has been well-studied, with the mean in-river survival between 1997 and 2018 estimated at 48% (McCann et al. 2022). Smolts from Snake River must travel a longer distance and pass multiple dams during migration from Snake River to Bonneville Dam compared to smolts migrating out of Willamette River sub-basins. If we make a conservative assumption that Willamette smolt migration mortality is 50% of smolts from Snake River, we calculate a downstream migration survival rate of 74%. Considering all these values and the uncertainty in the summer steelhead estimate, we assumed a downstream migration survival rate of age-2 winter steelhead smolts of 0.74. We note that as ϕ_P is part of the product of survivals with ϕ_F, if the downstream migrant survival is very different to our assumed value, this will be reflected in the estimate of ϕ_F.

Marine Stage

The marine stage of the WS-IPA LCM includes several survival components: 1) survival from Willamette Falls to the Columbia estuary, 2) survival while in the ocean, 3) survival during upstream migration to the dam (Foster or Detroit) tailrace, and 4) survival until spawning once outplanted above dams. The last two components are both considered pre-spawn mortality (PSM) but given the timing of the late winter steelhead adult run, i.e., when river temperatures are low, PSM can be assumed to be very low and as such are not included in ϕ_M. We note that in the model fitting process, non-negligible PSM would be incorporated in the estimate of ϕ_F, as that is the only free survival parameter being estimated. As the estimation model predicts the numbers of female adults returning to Foster, ϕ_M also implicitly includes failure of fish to enter the adult collection facility, for which straying rates are unknown.

Although annually variable, marine survival of most Pacific Northwest steelhead populations declined during the 1980s and has remained at low levels since (Kendall et al. 2017). Marine survival among Columbia River populations appears to covary (Figure 3.2.2), indicating that factors affecting marine survival influence all populations similarly across time. The most important processes related to smolt-adult survival of steelhead appear to occur early in the marine stage (Weitkamp et al. 2015; Kendall et al. 2017). Annual variation in marine survival is linked via sea surface temperature to the timing of high-quality food availability in the nearshore coastal environment, with individual survival also related to fish size and outmigration timing (Friedland et al. 2014; Wilson et al. 2021). The importance of fish size at outmigration as a predictor of survival indicates freshwater growing conditions can have carryover effects on marine survival (Wilson et al. 2021).

We assume the annual counts at Foster (Figure 3.1.3) reflect the spawning population above Foster dam. Counts by sex were only recorded in the Foster count from 2006 onwards (Mapes et al. 2017; ODFW, Brett Boyd, pers. comm.). For fitting the model to data prior to 2006, we used an estimate of the mean sex ratio from between 2006 and 2021 to apportion the total adult count prior to 2006 into males and females. Over this 15-year period, the mean female spawner percentage was 58% (range 50-65%), which equalled the 58% females reported by Clemens (2015) for the Willamette over the 1981-1994 period. There was no relationship between the number of returning spawners and the sex ratio.

Adult steelhead returning to spawn can either be virgin spawners or repeat spawners and may have spent different numbers of years in the ocean. The balance of factors affecting survival between spawns and fecundity of repeat spawners will determine the proportion of repeat spawners, or iteroparity rate, in a population (Copeland et al. 2019). Assignment of recruits to spawner cohorts thus depends on the age structure of the spawning population and the proportion of repeat spawners. Given the lack of available data, it appears that scales are not typically removed from adult steelhead returning to Foster adult collection facility, so age composition and repeat status is unknown. Therefore, we use more general information from scale analyses conducted in the Willamette, including from an angler survey between 1981 and 1994 that reports spawner age composition and iteroparity rates (Clemens 2015) and from a radio-telemetry study between 2012-2013 that reports iteroparity rates (Jepson et al. 2015).

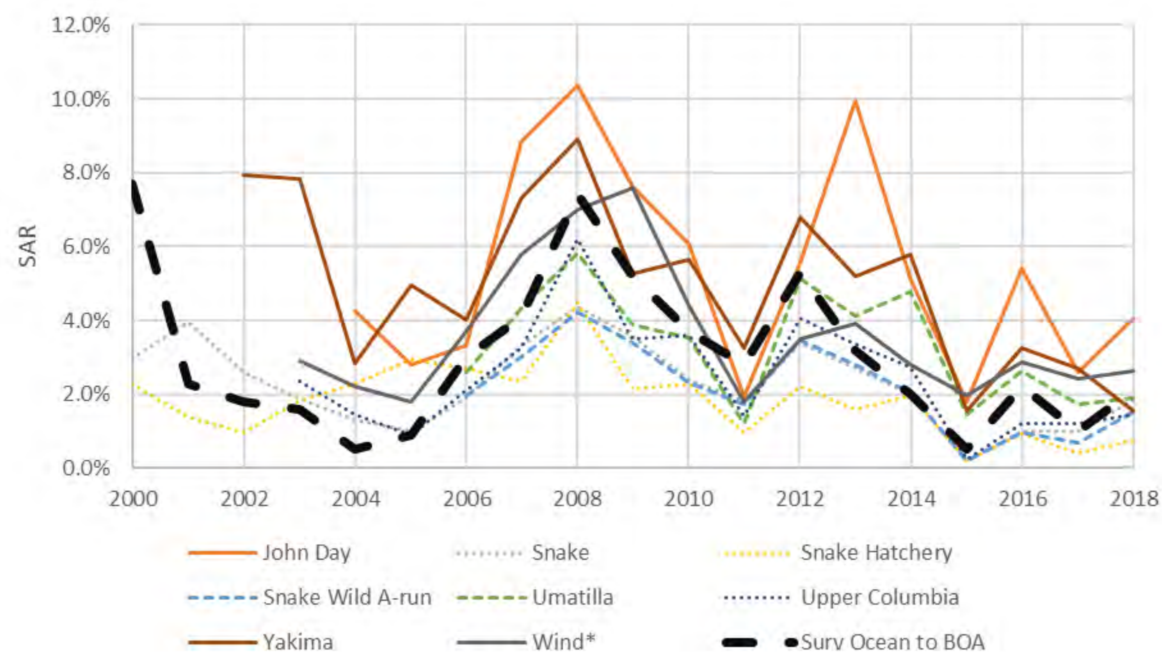

Figure 8-62. Smolt-adult return rates (SAR) estimated by McCann et al. (2022) using PIT-tag data.

Estimates are from various Middle and Upper Columbia River wild steelhead populations and reflect the survival as smolts detected at upper dams to return as adults to Bonneville Dam (BOA). Wind River (Lower Columbia, asterisked) estimates were obtained from Wilson et al. (2021). Thick dashed line shows the estimated ocean survival rate for wild Snake River steelhead from estuary entry as smolts to adult return (McCann et al. 2022).

Estimates of iteroparity rates come from analysis of scales collected during an angler survey in the Willamette sub-basins between 1981 and 1994 (Clemens 2015). For both winter- and summer- run steelhead, the data were sorted by sex, spawner status (virgin or repeat spawner), and origin (hatchery or wild). We summarised data for wild winter steelhead to determine the proportion of female spawners that were virgin or repeat spawners. Out of the 181 female wild winter steelhead, 86.2% of female spawners were virgin spawners and 13.8% were repeat spawners (Table 3.2.1). While this iteroparity rate may appear high in comparison to some populations (Clemens 2015), we note it is for females only rather than both sexes combined and the iteroparity rate of female and male winter steelhead combined was 9.6%. A much higher proportion (83.3%) of the repeat spawners sampled by Clemens (2015) were female, with similarly biased sex ratios of repeat spawners observed in other steelhead populations (Keefer et al. 2008; Christie et al. 2018; Copeland et al. 2019). We also performed similar calculations using two years of recent data from the South Santiam (Jepson et al. 2015) and found a similar iteroparity rate value, with 14.5% of female spawners being repeat spawners.

Table 8-16. Virgin and repeat spawner counts for wild Upper Willamette River winter steelhead (data from Table 2 in Clemens 2015). The ratio of % of female repeats to % of female virgin spawners is interpreted as the survival from first spawning to subsequent spawning events.

	Counts by Status			Proportions by Status		
Sex	Virgin	Repeat	Total	Virgin	Repeat	Total
Female	156	25	181	0.498	0.080	0.578
Male	127	5	132	0.406	0.016	0.422
Total	283	30	313	0.904	0.096	1.000
	% female by status			55.1%	83.3%	
	% of females with status			86.2%	13.8%	
	Female kelt to repeat survival (%repeat / %virgin)					16.0%

We then used the estimated iteroparity rate to calculate the female kelt to repeat survival rate (Table 3.2.1), which were interpreted from the ratio of % female repeat spawners to % female virgin spawners, which equalled 16.0% (=13.8/86.2). Use of the kelt to repeat survival rate meant that we could model the repeat spawners without needing to separately model 1) kelt migration back downstream through the dam, for which there were no estimates of dam passage efficiency of dam passage survival from FBW; and 2) survival of kelts in the ocean. For the latter, there is a general lack of knowledge about this survival rate.

Limited spawner age composition data exist for Willamette winter steelhead, with the same age composition previously assumed for each of the four sub-basins in the Distinct Population Segment (ODFW and NMFS 2011). An important consideration for our model was that this reported age composition did not consider the virgin or repeat spawner status. We instead constructed a spawner age composition by spawner status, i.e., virgin or repeat spawner, using the angler survey data from Clemens (2015). In that study, scales were analysed to determine freshwater age and saltwater age, with the frequency of freshwater and saltwater ages reported. These ages were not reported by sex, so we assumed the same frequencies for both male and female winter steelhead. Then, we used these frequencies to construct a table of the frequency of each freshwater and saltwater age combination, e.g., freshwater age-2 and saltwater age-2 means a spawner of age-4 (Table 3.2.2). Given our freshwater stage assumption that all smolts are age-2, we do not treat age-5 spawners that are age-2.3 or age-3.2 differently, i.e., two years in freshwater and three in saltwater vs. three years in freshwater and two years in saltwater. To maintain simplicity, we also did not consider combinations involving fish that had spent four years in freshwater or five years in the ocean, as sample size of observed returns was very low for these ages, i.e. one fish. Next, we assumed that spawners that had spent only two years in saltwater were virgin spawners, i.e., 100% of age-2.2 and age-3.2 spawners were virgin spawners. In order for iteroparity rate of female spawners (13.8%) to equal that in the constructed age composition table when summed across ages, we assumed that the ratio of repeat to virgin spawners by age-2.3 and age-3.3 spawners was the same and then used Solver in Excel to adjust this ratio so that the iteroparity rates matched (Table 3.2.2).

Table 8-17. Constructed spawner age composition by spawner status of wild Upper Willamette River winter steelhead. Percentage by freshwater (FW) age and saltwater (Salt) age were combined to provide the % composition for each age combination (data from Table 4 in Clemens 2015). The ratio of Repeat to Virgin by age (shown in bold) was assumed to be the same for age 2.3 and 3.3 spawners, and this ratio was adjusted to make the % Repeat by Status implied by the age structure in this table equal to the observed % Repeat in Table 3.2.1 (13.8%). Note that Clemens (2015) does not provide age breakdown by sex, and while hatchery freshwater ages were excluded, saltwater age was not broken out by origin.

			% Status by Age		% Composition by Status	
Age (FW.Salt)	Spawner Age	Total %	Virgin	Repeat	Virgin	Repeat
2.2	4	58.6%	100%	0.0%	58.6%	0.0%
2.3	5	20.6%	52.7%	47.3%	10.8%	9.8%
3.2	5	13.8%	100%	0.0%	13.8%	0.0%
3.3	6	4.8%	52.7%	47.3%	2.5%	2.3%
2.4	6	1.5%	0.2%	99.8%	0.0%	1.5%
3.4	7	0.3%	0.0%	100%	0.0%	0.3%
	Total	99.6a%		Total	85.8%	13.8%

a Total % does not sum to 100% as for simplicity we did not include the fish sampled that was determined to have spent four years in freshwater (1/233 fish) and the fish that spent one year or five years in the ocean (3/1091 and 1/1091 fish, respectively). This means that the % Virgin by Status does not equal 86.2% determined in Table 3.2.1.

The total female age structure calculated was 58.6% at age-4 (all virgin spawners), 34.4% at age-5 (virgin + repeat spawners), 6.3% at age-6 (virgin + repeat spawners), and 0.3% at age-7 (all repeat spawners). From the percent composition by status, we thus determined that 68.4% of the virgin female spawners would be age-4 (=58.6/85.8, Table 3.2.2), 28.7% age-5, and 3.0% age-6. These proportions were used to determine the numbers from a given smolt year that would return as virgin age-4, age-5, or age-6 spawners. Assuming kelts spend one year in the ocean to recondition, the kelt-to-spawner survival rate (16%) was then applied to the numbers of virgin age-4, age-5, and age-6 spawners to determine the numbers of repeat-5, repeat-6, and repeat-7 spawners, respectively, in future years. We did not account for kelts that may return to the ocean multiple times, i.e., repeat-repeat spawners. To determine the model predicted number of spawners in a given year, we lastly took the sum of virgin and repeat spawners of all ages in that year. We illustrate the construction of a steelhead spawner age structure using these values in Table 3.2.3.

Table 8-18. Illustration of construction of a wild winter steelhead spawner age composition by spawner status (virgin or repeat), given that 1000 adults survive in the ocean from each smolt year. Cells outlined in green are the cohort from brood year 0. Note the total number of spawners from any cohort will exceed the number of individuals due to repeat spawners.

Brood year	Smolt year	Adults after marine survival	Virgin age-4	Virgin age-5	Virgin age-6	Repeat age-5	Repeat age-6	Repeat age-7	Total spawners
0									
1									
2	2	1000							
3	3	1000							
4	4	1000	684						
5	5	1000	684	287		110			
6	6	…	684	287	30	110	46		
7	7	…	684	287	30	110	46	5	1162
8	8	…	…	287	30	110	46	5	…
9	9	…	…	…	30	…	46	5	…
10	10	…	…	…	…	…	…	5	…

Model fitting and projections

Model fitting

The WS-IPA LCM for the above Foster population predicts the annual abundance of female winter steelhead arriving at Foster dam tailrace. Most of the parameters in the model are assumed to be fixed and known, but we freed up some parameters to be estimated by fitting the model to count data from Foster adult collection facility. The fixed parameters by life cycle stage were:

Freshwater stage: fecundities for virgin and repeat spawners, and freshwater smolt capacity (see Section 3.2.1).

Passage stage: successful passage rate of age-2 smolts as the product of dam passage efficiency (DPE) and dam passage survival (DPS) in each year from FBW, and downstream migration survival rate (see Section 3.2.2).

Marine stage: mean marine survival rate (see Section 3.2.3 and Appendix E).

The estimated parameters were ϕ_F, mean freshwater survival rate of juvenile steelhead from egg to age-2 above the dam, and dy, annual deviates in marine survival from 1993-2019. To fit the model, the first year of count data used was 1991, as this was the first year in which no hatchery winter steelhead were outplanted above Foster (see Section 3.1). To initialise the model, we therefore also had to estimate N1991, the abundance of female spawners in the initial year of count data, i.e., female adult winter steelhead at Foster tailrace in 1991.

Female steelhead were recorded separately at Foster from 2006-2021 (Mapes et al. 2017; ODFW, Brett Boyd, pers. comm.), but prior to 2006 only total counts were available (ODFW, Brett Boyd, pers. comm.). The number of females prior to 2006 was predicted using the mean sex ratio from the 2006-2021 counts, which was similar to the 58% females reported by Clemens (2015) for the Willamette over the 1981-1994 period (Figure 3.1.3). ϕ_F and dy, were estimated by fitting model predictions of annual female spawner abundance starting in 1991 and running through to 2021 to the observed and imputed female counts at Foster from 1991-2021. We assume that the counts reflect spawners produced above the dam, and that all returning spawners enter the collection facility.

Model prediction and fitting under the NAA alternative used a time series of DPE and DPS from 1991 to 2019 to reflect historical dam passage conditions at Foster. However, recent modifications to the fish weir at Foster in spring 2018 resulted in changes to DPE and DPS (Liss et al. 2020), and therefore we used the NAA dam passage parameters from the FBW run for the 2015 COP evaluations (Zabel et al. 2015) to model NAA downstream dam passage for smolt years 1991-2017. For the 2018-2019 smolt years, DPE and DPS values were obtained from the January 2022 FBW run of the NAA Alternative, as these reflect the post-2018 changes in the fish weir. This ensured that the returning adults predicted for model fitting reflected both historical and contemporary juvenile downstream passage conditions.

We note that radio-telemetry studies were conducted at Foster in 2015, 2016 and 2018, which had the aim of estimating dam passage efficiency and dam passage survival of age-2 steelhead at high and low pool (Hughes et al. 2016, 2017; Liss et al. 2020). The estimates of DPE and DPS from these studies went into the parameterization of FBW (see Section 1.4), but the annual estimates in the time series obtained from FBW may not exactly reflect the values from those specific years due to hydrological conditions outside of the study periods in each year.

To initialise the age structure to provide a predicted count of virgin and repeat female spawners across all ages in 1991 we assumed that N1991 was constant from 1984-1991. The time series of marine survival rate starting in 1986 was then used to generate returning female spawners of virgin age-4, virgin age-5, virgin age-6, repeat age-5, repeat age-6 and repeat age-7 from 1988-onwards. This resulted in estimates of annual deviates in marine survival from 1986 onwards. Owing to the lag in adult returns, the last year that marine survival rate deviates are determined from data is 2019.

We constrained the deviates in marine survival using a prior distribution on the natural logarithm of each annual deviate in marine survival rate where the standard deviation was 1.0 (Equation 3.3-1). An informative prior distribution was developed and applied for N1991 (see Appendix G for details). ϕ_F was assumed to have a flat lognormal prior, i.e., with large variance. The sum of the logarithm of the priors for the estimated parameters was given by:

$$\boldsymbol{Logprior = constp - \frac{\left(log\left(\frac{S_F}{\mu_{S_F}}\right)\right)^2}{2\sigma_{S_F}^2} - \frac{\left(log\left(\frac{N_{1991}}{\mu_{N_{1991}}}\right)\right)^2}{2\sigma_N^2} - \sum_{y=1993}^{2019}\frac{(d_y - 0)^2}{2\sigma_d^2}}$$

Equation 3.3-17

where $constp$ is the sum of constants in the prior density functions that remained constant, μ_{S_F} is the prior median value for freshwater survival rate, $\sigma_{S_F}^2$ is the prior variance for freshwater survival rate, $\mu_{N_{1991}}$ is the prior median value for the abundance of adult female steelhead at the Foster tailrace in 1991, σ_N^2 is the prior variance for the abundance of female steelhead, and σ_d^2 is the prior variance for deviates from the estimated mean marine survival rate for Wind River steelhead (see Appendix E for details on marine survival rate time series, Appendix G for prior on spawner abundance).

A lognormal likelihood function was applied in fitting the WS-IPA LCM to the time series of adult female counts at the Foster tailrace.

$$\boldsymbol{LogLik = constL - \sum_{y=1991}^{2021}\frac{\left(\log(N_{y,obs}) - log(N_{y,pred})\right)^2}{2\sigma_{SHL}^2}}$$

Equation 3.3-18

where constL is a component of the lognormal likelihood function that remained constant in parameter estimation, $N_{y,obs}$ is the observed or approximated number of female adult steelhead counted, $N_{y,pred}$ is the model-predicted number of female adult steelhead at the Foster Dam tailrace in year y and σ_{SHL}^2 is the variance term in the lognormal likelihood function (see Appendix B).

Model projections

Fitting the winter steelhead IPA LCM to Foster count data resulted in estimates of mean freshwater survival and annual deviates in marine survival, and the model predicted age structure in 2021, the final year of count data. We added the estimated marine survival deviates to the mean marine survival rates to obtain a time series of estimated marine survival rates for the winter steelhead population above Foster. The time series of estimated marine survival rates suggested periodic cycling with an apparent periodicity of about 10 years. To account for this, we simulated a positive lag 1 autocorrelation in future marine survival rates (see Appendix F for details). We then projected the model forwards from 2021 given the freshwater survival estimate, the lag 1 autocorrelated marine survival rate estimates, the predicted age structure in 2021, and the dam passage parameters obtained from FBW under the different EIS alternatives for downstream dam passage at Foster.

Currently there is only winter steelhead production above Foster, and not above Green Peter or above Detroit, so it was only possible to fit the model to data from Foster. For projections under the EIS alternatives for the above Green Peter and above Detroit populations, we assumed that the marine survival rate time series would be the same as that estimated for the above Foster population. This was reasonable given the close geographic location of these populations. As the last year in which marine survival deviates were estimated was 2019, the first year of modelling lag 1 autocorrelation in marine survival rate was 2020.

Given the freshwater survival estimate was for the above Foster population, we had to make assumptions about what values to apply for freshwater survival in populations above the other two dams, where rearing conditions in the headwaters and reservoirs might be quite different to those above Foster. Without knowledge of the age of juvenile movements in the above Green Peter and above Detroit populations, we used the values for egg-fry survival, fry-migrant survival of age-2 fish, and reservoir survival of age-2 fish from the 2015 COP Appendix (Zabel et al. 2015) to construct a value for survival from egg to dam forebay for each population (Table 3.3.1). We used the differences between the above Foster and Green Peter and above Foster and Detroit values to determine a scalar to apply to the above Foster estimate of freshwater survival obtained from our estimation model. The scaled values were used in Green Peter and Detroit projections.

In the South Santiam projections, the above Foster and above Green Peter populations were modelled independently. These populations were then combined to enable evaluation of performance metrics under each EIS alternative in the South Santiam. This assumed that winter steelhead outplanted above Foster would self-sort and those adults produced above Green Peter would move to the tailrace of Green Peter dam and be outplanted above Green Peter. Survival of adults from Foster to outplanting into spawning habitat above Green Peter was assumed to be 100%.

Table 8-19. Calculation of implied freshwater survival rates and scalars relative to Foster from the values for above dam survival rate components in Zabel et al. (2015), p9.8-9.12 for South Santiam, p8.7-8.10 for North Santiam.

Parameter	Foster	Green Peter	Detroit
Egg-fry survival	0.425	0.450	0.435
Fry-migrant survival (age-0)	0.525	0.600	0.600
Fry-migrant survival (age-0)	0.325	0.350	0.350
Fry-migrant survival (age-0)	0.250	0.275	0.275
Reservoir survival (age-0 to age-0)	0.400	0.300	0.400
Reservoir survival (age-0 to age-1)	0.150	0.200	0.200
Reservoir survival (age-2 to age-2)	0.500	0.500	0.500
Reservoir survival (age-2 to age-2)	0.500	0.450	0.500
Reservoir survival (age-2 to age-2)	0.850	0.850	0.850
Calculated values			
Survival to forebay (age-0 reservoir entry)	0.007	0.010	0.011
Survival to forebay (age-1 reservoir entry)	0.029	0.030	0.032

Survival to forebay (age-2 reservoir entry)	0.090	0.105	0.102
Freshwater survival scalar applied	1.000	1.165	1.126

Uncertainties and Assumptions

Juvenile freshwater stage	Assumption made	Assessment of assumption
Abundance is regulated by density-dependence only in the freshwater (egg-smolt) stage and follows a Beverton-Holt function	The life cycle consists of a sequence of contiguous, non-overlapping life stages (i.e., egg-to-above dam smolt, above dam smolt-to-below dam smolt, smolt-to-adult) that are linked by density-independent survival rates, as densities are currently low. Density-dependence in the egg-smolt stages (i.e., egg-to-fry, fry-to-parr, parr-to-smolt) can be described by a single Beverton-Holt recruitment function, as described by (Moussalli and Hilborn 1986). The Beverton-Holt smolt capacity term is calculated from parr rearing habitat capacity estimates above dams (Bond et al. 2017) adjusted for parr-smolt survival rate. Using smolt capacity implies that if density increases then density dependence would be due to limited amount of habitat being available for juvenile rearing above dams.	Sensitivity analysis on Beverton-Holt capacity parameter by scaling base case β by 0.75 and 1.5
Survival rates of juvenile steelhead above dams	We assumed that the freshwater survival rate estimated for the above Foster juvenile population is invariant to the downstream dam passage measure applied. There are no studies on how a dam passage measure can modify juvenile survival rates in reservoirs above dams. Freshwater survival was constant over time. In absence of other information on juvenile survival of steelhead above Green Peter and Detroit, survival rate components for each of the above dam reaches from Zabel et al. (2015) were multiplied. The relative egg-to-age2 survival rates for above Green Peter and Detroit were compared to that for above Foster to produce a scalar for use with the Foster freshwater survival estimate.	Probability distribution for ϕ_F from model fitting. Sensitivity analysis by scaling base caseϕ_F by 0.75 and 1.5.

In the two-dam model, juveniles originating above and passing down through Green Peter will try to pass directly through downstream dams without stopping	Assumes smolting starts once movement downstream. It assumes that the numbers of age-2 smolts in the tailrace of GPR will join with the age-2 smolts in the forebay of FOS. Smolts above each dam will experience the same conditions in downstream migration below dams and in marine environments.	None.
DPS and DPE	The DPS and DPE values for age-2 come from FBW outputs. Bootstrapped time-series of DPS and DPE (historical years) were drawn. DPS and DPE values are sub-basin specific.	Bootstrap of historical time series. Sensitivity of PMs to DPE*DPE over [0,1] range.
Downstream migrant survival below dam to Willamette Falls	Telemetry data indicates steelhead smolts produced above dams do not spend much time rearing in reaches downstream of dams before passing Willamette Falls (Monzyk et al. 2017), suggesting high migrant survival rates. There are no specific studies to estimate downstream migrant survival of winter steelhead smolts in the Willamette so our assumed value considers several sources of data (see Section 3.2.2). Same value assumed for each sub-basin.	None.
All juveniles smolt at age-2 and migrate downstream in spring, individuals that do not smolt are lost to forebay or residualise	We only model one life history type, age-2 smolts (see Section 3.2.1). All smolts rear until age-2 above dams, with all female juveniles attempting to smolt at age-2, i.e., no residualism. Rearing can take place in the stream or the reservoir, the model does not account for fry-smolt survival being higher in the stream. Younger (age-0 and age-1) and older (age-3) smolts are observed in tagging data but have very low frequency of occurrence (Monzyk et al. 2017). Smolts of these other life history types entrained at Foster do not contribute to the adult returns to Foster.	None.

Only females modelled	We assume that demography of winter steelhead is less affected by male abundance. Each male can mate with multiple females, males are prone to a non-anadromous life history which makes them less vulnerable to low marine survival, male kelts have poor survival as they spend more time pursuing reproductive opportunities before heading downstream (Fleming 1998; Keefer et al. 2018). This is supported by data indicating more females than males are present in the returning adults to Foster and >80% of repeat spawners are female (Clemens 2015; Jepson et al. 2015).	None.
Adult marine stage	[…]	[...]
Adult marine survival	We assume that the smolt-adult survival of Willamette steelhead is similar to that of the geographically local Wind River population (Wilson et al. 2021). The marine survival rate estimates used in the LCM are assumed to include any upstream migration mortality, e.g., from pinniped predation or PSM. In the LCM, all mortality is assumed to occur on entry to the ocean, with female steelhead spending at least two years in the ocean before maturing. No mortality between older age classes remaining in the ocean was modelled. Only single repeat spawners were modelled, with all kelts spending one year in the ocean to recondition.	Estimated annual deviates on marine survival. Lag-1 autocorrelation included in projections. Sensitivity analysis by 1) scaling the base case mean survival rate by 0.0 to 2.0; 2) setting ρ at 0 (i.e., no autocorrelation) or 0.9; 3) setting repeat spawner survival at 0% and 32% (i.e., 2x base case).
Model age structure	All cohorts have identical an age structure by spawner status, i.e., virgin or repeat spawner, which is time invariant. See Section 3.2.3 for age structure derivation using data from Clemens (2015)	None.

At sea fishing mortality rates and incidental mortality rates in terminal fisheries	We assumed that mortality of winter steelhead from fisheries was negligible and would be included in the marine survival estimate.	None.
Adult freshwater stage	[...]	[…]
Returning adults to Foster are recruits from the spawners outplanted above dam	The time series of adult counts from Foster adult collection facility reflects all of the female steelhead arriving at, and then spawning above, Foster.	None.
Straying	We assume no straying of returning adults. Fish return to their natal spawning habitats with no straying other subbasins, or other spawning areas within each subbasin.	None.
PSM (en route and on site) is negligible	We assumed that PSM (both en route and on spawning grounds) is negligible due to the late winter timing of the run meaning temperature stress will be minimal. Bycatch mortality from in-river summer steelhead fishery was not considered.	None.
Outplanting above dams	Outplanting specifications varied by sub-basin and EIS alternative, which USACE defined. We assumed no mortality effects associated with the trap-and-haul process or temperature stress on adults from holding. Also, we assumed that outplanted adult winter steelhead disperse upon release throughout the available habitat before spawning.	None.
Egg production/sex ratio of the spawning population	Fecundity does not vary by age, with values used equal to age-weighted fecundities. Egg production above dams is entirely from anadromous females. Female eggs are 50% of egg production.	None.
Annual time step	We assumed the annual time step is sufficient to represent the life history processes. There is insufficient data to describe the juvenile freshwater stage or data to fit the model at finer timescales.	None.

Results under base case parameterization

Fitting of the LCM to data

The model was initialized from 1984 through 1990, and then fitted to observed counts of adults at Foster dam tailrace from 1991 through 2021 (Figure 3.5.1). Each year a marine survival rate deviate was estimated. The number of spawners to initialize the fits was estimated to be 119.77 with a standard error of 46.35. The estimated freshwater survival was 0.037 with a standard error of 0.006. As described, we did not have age composition data for these fits.

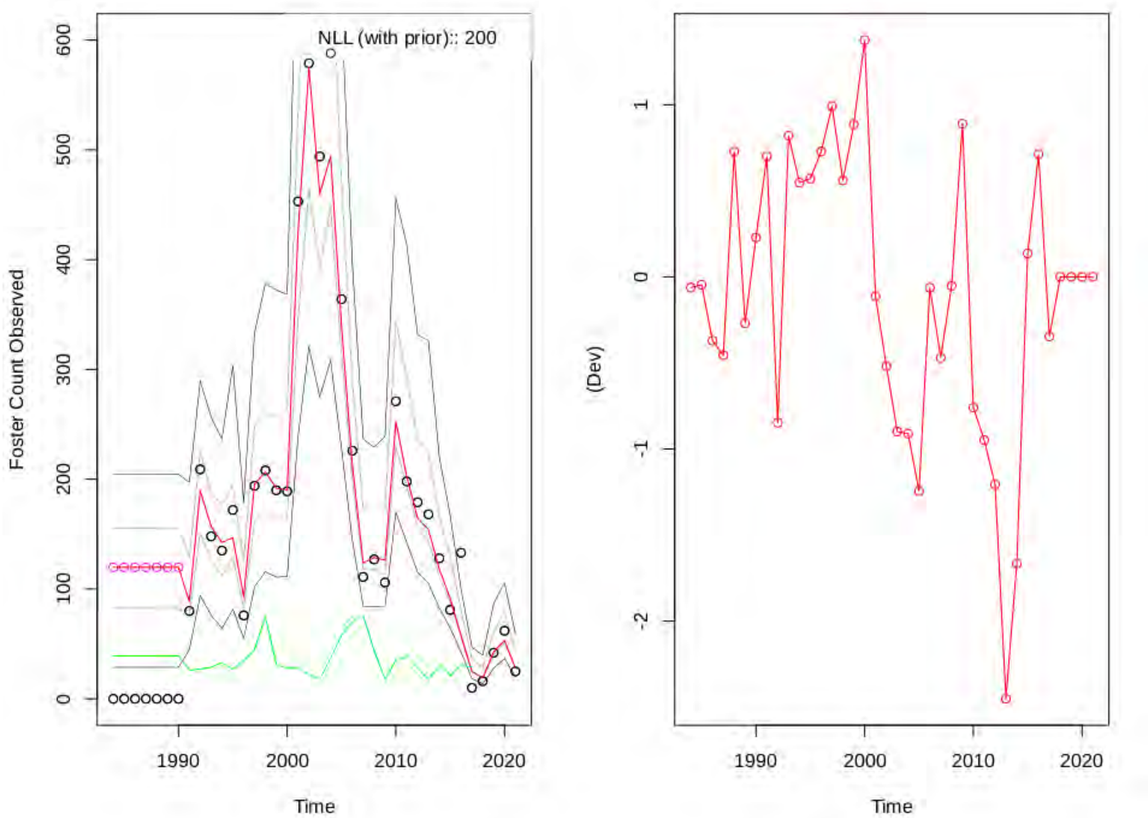

Figure 8-63. Fit of the winter steelhead LCM to data. Left panel shows the observed adult female counts as open black circles (1991-2021), the fitted counts as a red line and the rescaled marine survival rate as a green line. Given no observed count data prior to 1991, the open pink circles reflect the prior mean abundance in 1991, used to initialize the age structure. The confidence intervals of the fitted counts are shown as black lines above and below the fit. The deviates in marine survival rate from the prior mean values are shown in the right panel.

Performance Metrics for EIS alternatives

Performance metrics were computed under base case parameterization (see Appendix B) for the three dam projects that impact winter steelhead, i.e., Foster (FOS), Green Peter (GPR), Detroit (DET), and for the entire South Santiam (SS) sub-basin (i.e., FOS and GPR combined) for all of the different EIS alternatives that were represented in FBW outputs (see Table 3.5.1-Table

3.5.4). Results are shown from 10,000 simulations. Extinction risk (P<QET) was quite sensitive to the average DPS*DPE and the variability in DPS*DPE. For example, Green Peter Alt 1 and Detroit Alt 1 both had similar DPS*DPE (0.64 and 0.65). However, the P<QET were quite different. This can be explained by the fact that that the standard deviation of DPS*DPE in Alt 1 of Detroit was 0.25 (CV=0.39) compared with 0.28 (CV=0.45) for Green Peter.

Table 8-20 Performance metrics of the different EIS alternatives for Foster dam in the base case. Marine survival denotes the long-term average projected marine survival rates. Initial marine survival denotes the average marine survival rates in the first four years of implementation. P<QET=probability of population being less than the quasi-extinction threshold, SAR=smolt-adult return rate, DPS=dam passage survival, DPE=dam passage efficiency. Definitions for each summary performance metric are found in Section 1.3.

FOS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.326	1.326	1.326	0.504	0.509	1.144	0.506
Geomean Spawners	249.8	249.8	249.8	8.5	8.8	159.3	8.6
P<QET Threshold	0.720	0.720	0.720	0.996	0.996	0.811	0.996
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422
Marine Survival	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422
Initial Marine Survival	0.0421	0.0421	0.0421	0.0421	0.0421	0.0421	0.0421
DPS	0.875	0.875	0.875	0.680	0.684	0.777	0.682
DPE	0.723	0.723	0.723	0.470	0.470	0.723	0.470
DPS*DPE	0.641	0.641	0.641	0.318	0.320	0.567	0.319

Table 8-21. Performance metrics of the different EIS alternatives for Green Peter dam in the base case. Marine survival denotes the long-term average projected marine survival rates. Initial marine survival denotes the average marine survival rates in the first four years of implementation. P<QET=probability of population being less than the quasi-extinction threshold, SAR=smolt-adult return rate, DPS=dam passage survival, DPE=dam passage efficiency. N.B. There is no downstream passage at GPR under NAA or Alt 4. Definitions for each summary performance metric are found in Section 1.3.

GPR	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b
Recruits Per Spawner (R/S)	1.617	0.771	0.763	0.770	0.541
Geomean Spawners	315.5	33.5	32.4	33.4	9.9

P<QET Threshold	0.588	0.974	0.976	0.975	0.997
SAR	0.0422	0.0422	0.0422	0.0422	0.0422
Marine Survival	0.0422	0.0422	0.0422	0.0422	0.0422
Initial Marine Survival	0.0421	0.0421	0.0421	0.0421	0.0421
DPS	0.854	0.675	0.672	0.674	0.834
DPE	0.817	0.578	0.576	0.578	0.401
DPS*DPE	0.716	0.390	0.387	0.390	0.314

Table 8-22. Performance metrics of the different EIS alternatives for the entire South Santiam sub-basin in the base case. Marine survival denotes the long-term average projected marine survival rates. Initial marine survival denotes the average marine survival rates in the first four years of implementation. P<QET=probability of population being less than the quasi-extinction threshold, SAR=smolt-adult return rate, DPS=dam passage survival, DPE=dam passage efficiency. Definitions for each summary performance metric are found in Section 1.3.

SS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.496	1.110	1.108	0.667	0.523	1.144	0.506
Geomean Spawners	567.6	284.1	283.0	42.1	18.7	159.3	8.6
P<QET Threshold	0.653	0.826	0.827	0.989	0.996	0.811	0.996
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422
Marine Survival	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422
Initial Marine Surival	0.0421	0.0421	0.0421	0.0421	0.0421	0.0421	0.0421
DPS	0.864	0.775	0.773	0.677	0.761	0.777	0.682
DPE	0.770	0.650	0.649	0.524	0.436	0.723	0.470
DPS*DPE	0.679	0.505	0.503	0.354	0.321	0.567	0.319

Table 8-23. Performance metrics of the different EIS alternatives for Detroit dam in the base case. Marine survival denotes the long-term average projected marine survival rates. Initial marine survival denotes the average marine survival rates in the first four years of implementation. P<QET=probability of population being less than the quasi-extinction threshold, SAR=smolt-adult return rate, DPS=dam passage survival, DPE=dam passage efficiency. Definitions for each summary performance metric are found in Section 1.3.

DET	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.648	1.721	1.721	1.042	0.725	1.678	0.534

Geomean Spawners	780.1	872.6	872.6	208.9	96.0	817.7	21.7
P<QET Threshold	0.385	0.352	0.352	0.744	0.880	0.371	0.981
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422
Marine Survival	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422
Initial Marine Survival	0.0421	0.0421	0.0421	0.0421	0.0421	0.0421	0.0421
DPS	0.860	0.869	0.869	0.758	0.649	0.869	0.528
DPE	0.824	0.846	0.846	0.647	0.627	0.828	0.587
DPS*DPE	0.727	0.756	0.756	0.492	0.419	0.739	0.315

Summary Figures for Performance Metrics

To make it easier to compare the different performance metrics across different EIS alternatives, we produced several summary plots for each dam and sub-basin for which passage was evaluated 1) Foster (Figure 3.5.2-Figure 3.5.5), 2) Green Peter (Figure 3.5.6-Figure 3.5.9), 3) South Santiam sub-basin (Figure 3.5.10-Figure 3.5.13), 4) Detroit (Figure 3.5.14-Figure 3.5.17). While some EIS alternatives performed more poorly than others, there was considerable overlap between the different distributions. Note that several of the y-axes are on log scales because of long tails in the performance metrics. Still, the substantial overlap in performance metrics between the different EIS alternatives suggests that many of the EIS alternatives do not result in widely differing outcomes for winter steelhead. This does not imply that ranking the different EIS alternatives is impossible or even undesirable; but that small differences in the means of the simulations should be discounted when considering the relative high variance in the PM results.

Foster

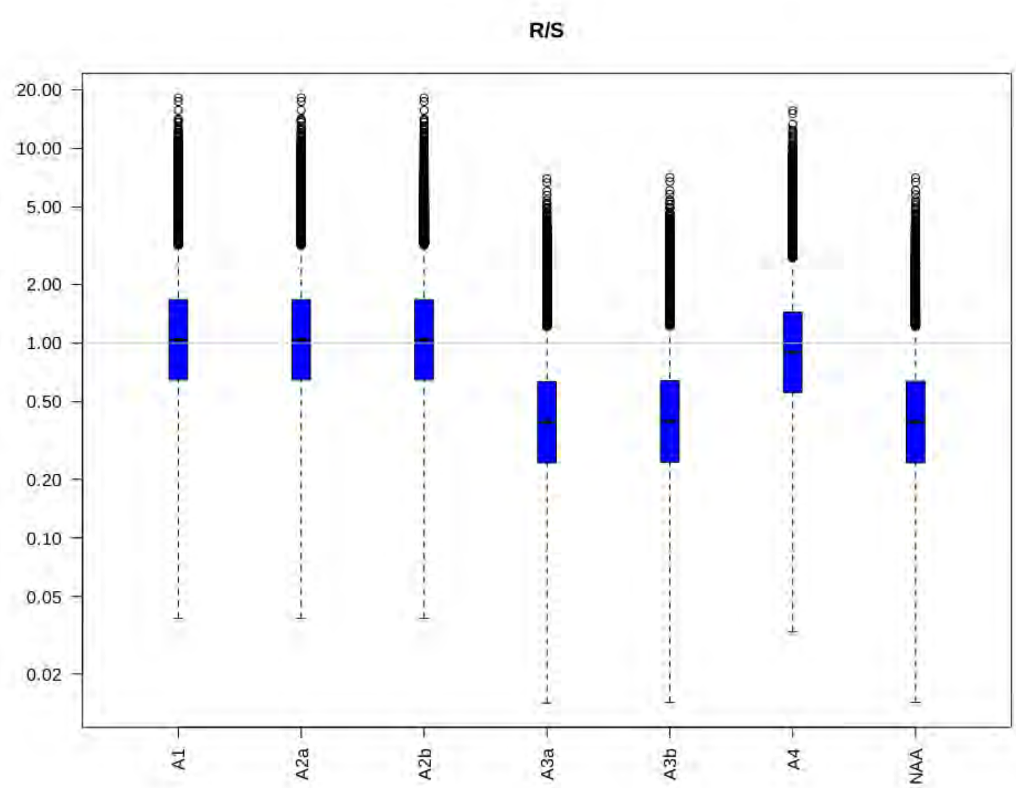

Figure 8-64. Distribution of Foster recruits per spawner (R/S) for each of the different EIS alternatives from 10,000 simulation runs.

The horizontal grey line is at 1 for R/S. The y-axis is on a log distribution because there are long upper tails. Note that many of the EIS alternatives have distributions that appear quite similar. This is not surprising since DPS*DPE is the same for a number of the EIS alternatives.

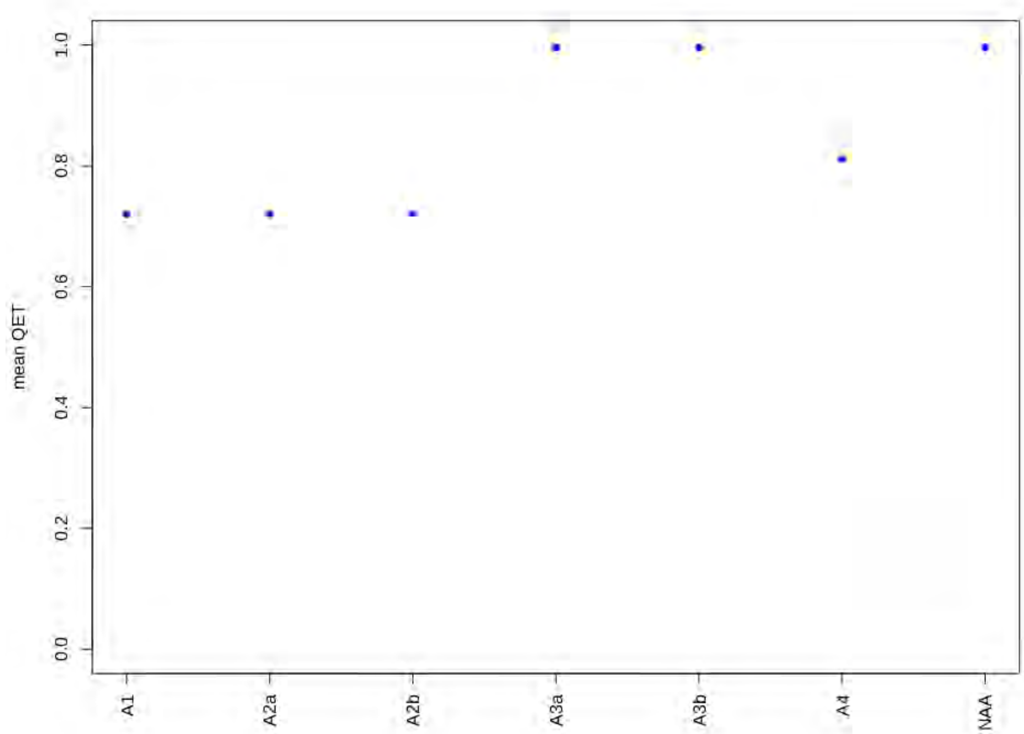

Figure 8-65. Mean of Foster P<QET for each of the different EIS alternatives from 10,000 simulation runs.

Note that many of the EIS alternatives do not perform well with probabilities of extinction close to 1, but some have relatively lower extinction risk.

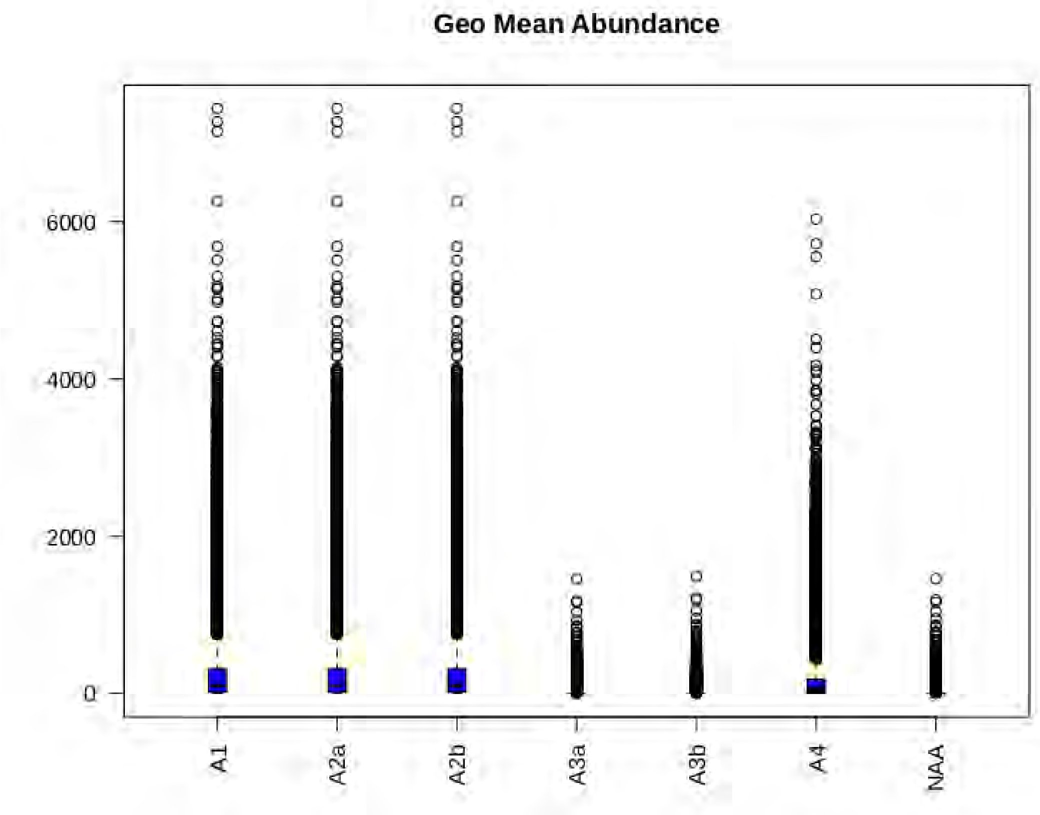

Figure 8-66. Distribution of Foster spawner abundance for each of the different EIS alternatives from 10,000 simulation runs.

The y-axis is on a log distribution because there are long upper tails. Note that many of the EIS alternatives have distributions that appear quite similar. This is not surprising since DPS*DPE is similar for many EIS alternatives.

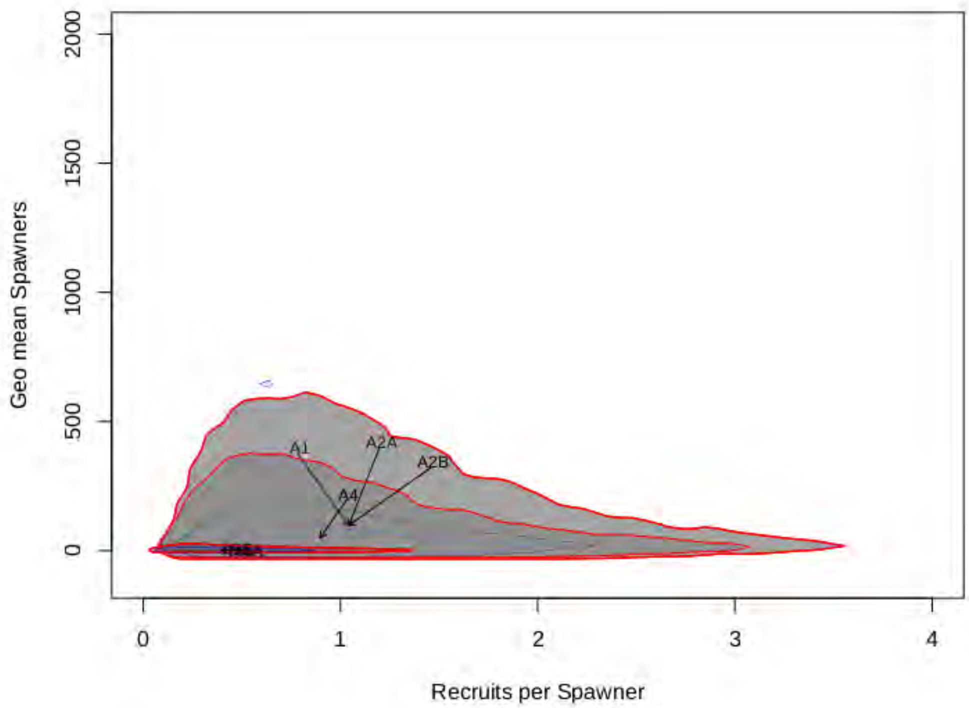

Figure 8-67. Joint Distribution of Foster recruits per spawner (R/S) and spawner abundance for each of the different EIS alternatives.

Note that many of the EIS alternatives have distributions that appear to overlap. The label is at the mean and the red line includes 95% of the 10,000 simulations.

Green Peter

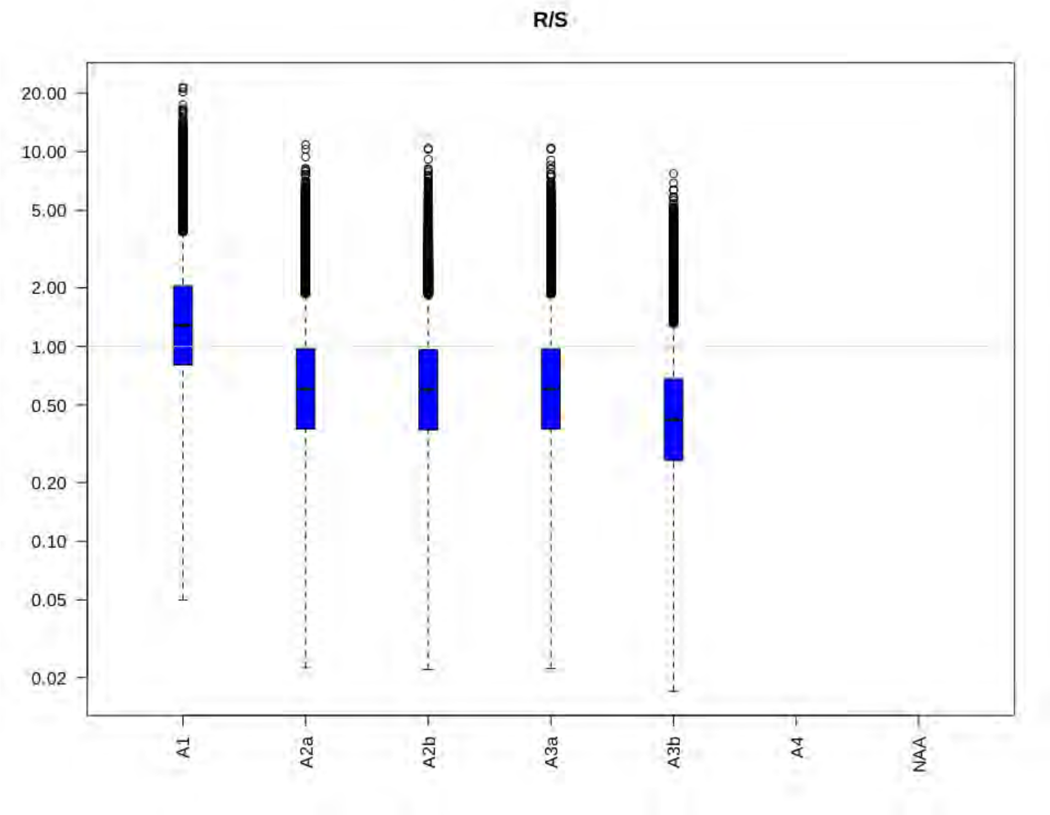

Figure 8-68. Distribution of Green Peter recruits per spawner (R/S) for each of the different EIS alternatives from 10,000 simulation runs.

The horizontal grey line is at 1 for R/S. The y-axis is on a log distribution because there are long upper tails. Note that many of the EIS alternatives have distributions that appear quite similar. This is not surprising since DPS*DPE is the same for many EIS alternatives.

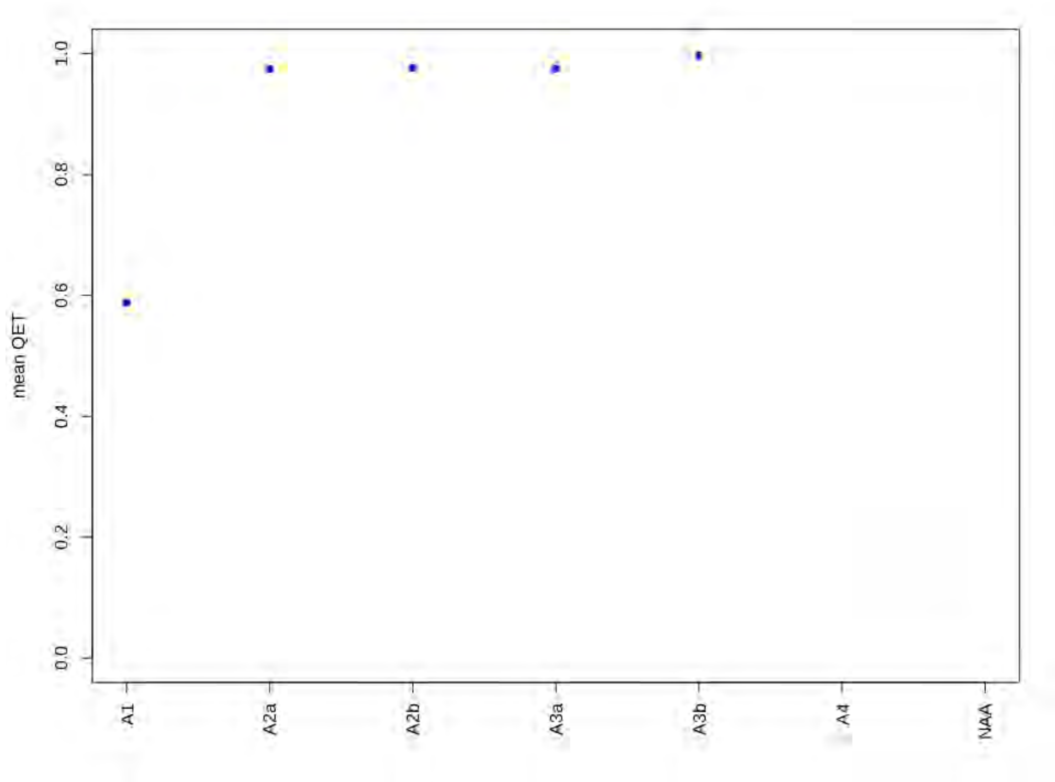

Figure 8-69. Mean of Green Peter P<QET for each of the different EIS alternatives from 10,000 simulation runs.

Note that many of the EIS alternatives do not perform too well, with only Alt 1 having relatively low extinction risk.

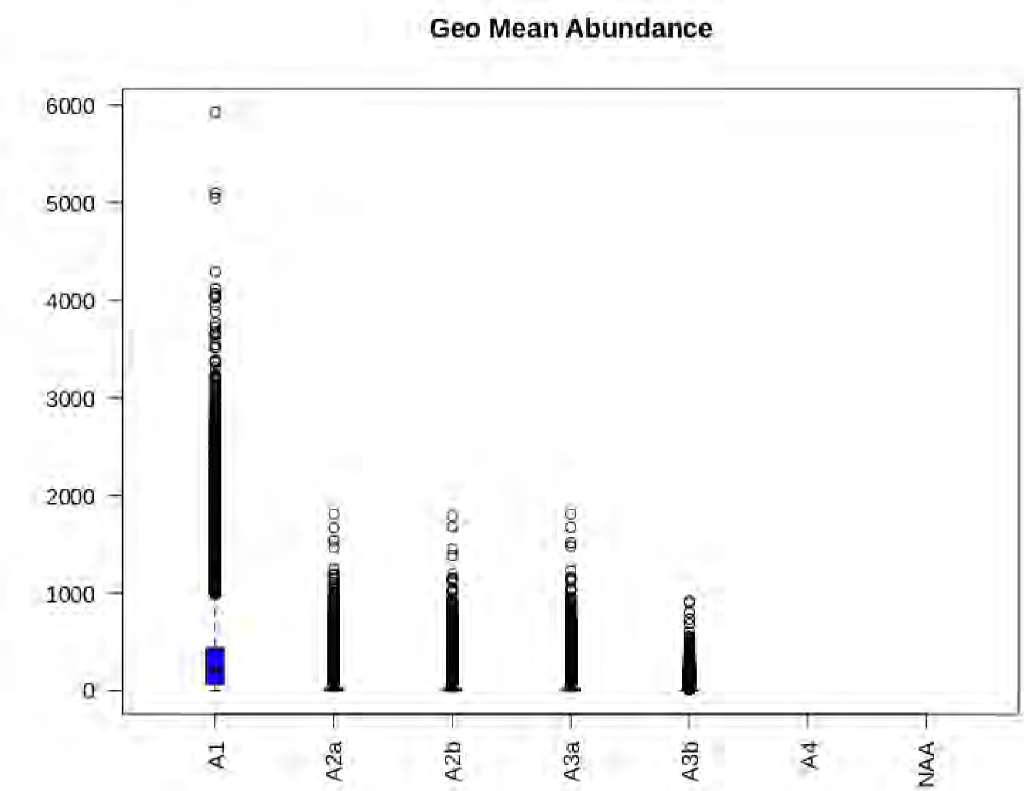

Figure 8-70. Distribution of Green Peter spawner abundance for each of the different EIS alternatives from 10,000 simulation runs.

The y-axis is on a log distribution because there are long upper tails. Note that many of the EIS alternatives have distributions that appear quite similar. This is not surprising since DPS*DPE is similar for many EIS alternatives.

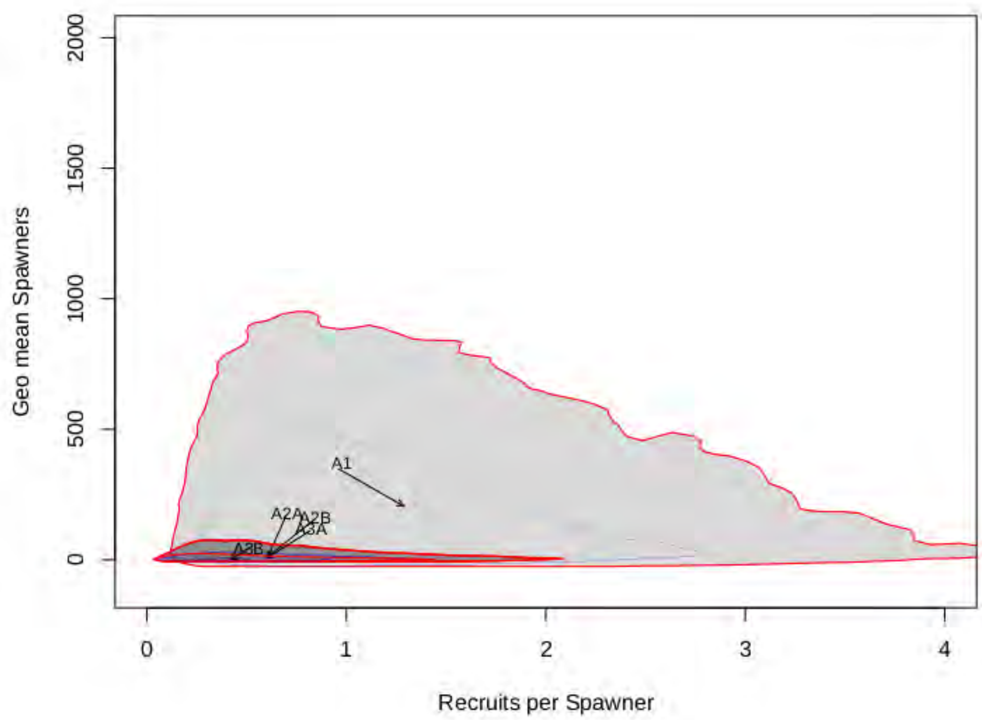

Figure 8-71. Joint Distribution of Green Peter recruits per spawner (R/S) and spawner abundance for each of the different EIS alternatives.

Note that many of the EIS alternatives have distributions that appear to overlap. The label is at the mean and the red line includes 95 % of the 10,000 simulations.

South Santiam sub-basin

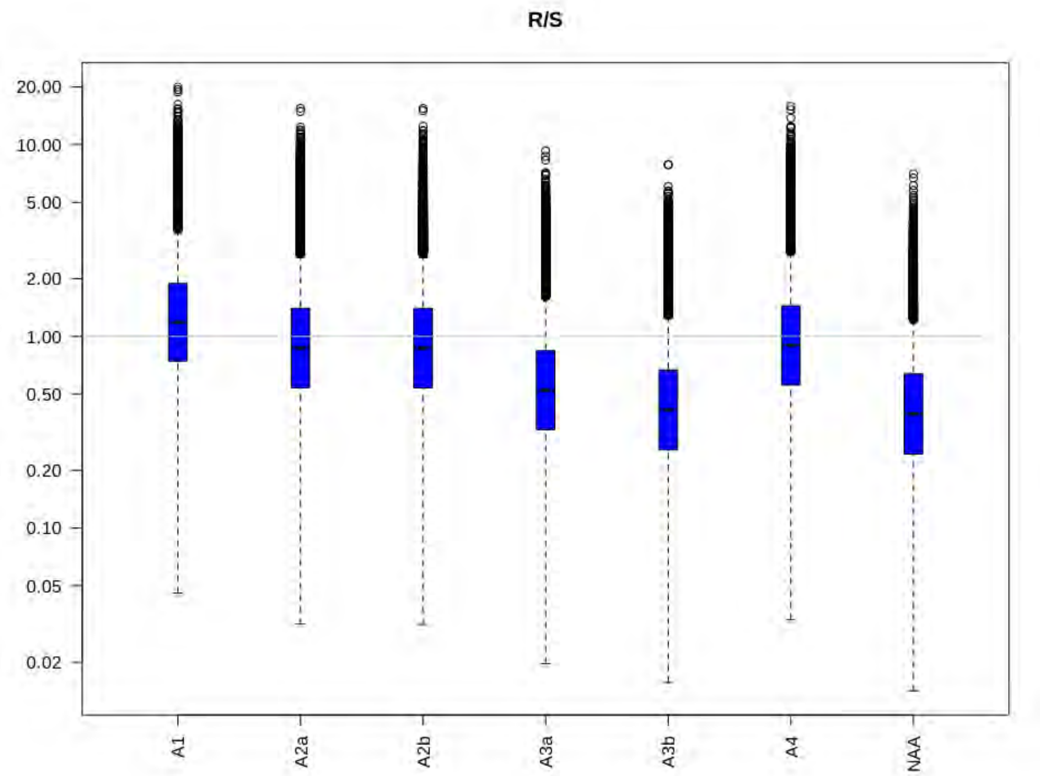

Figure 8-72. Distribution of South Santiam recruits per spawner (R/S) for each of the different EIS alternatives from 10,000 simulation runs.

The horizontal grey line is at 1 for R/S. The y axis is on a log distribution because there are long upper tails.

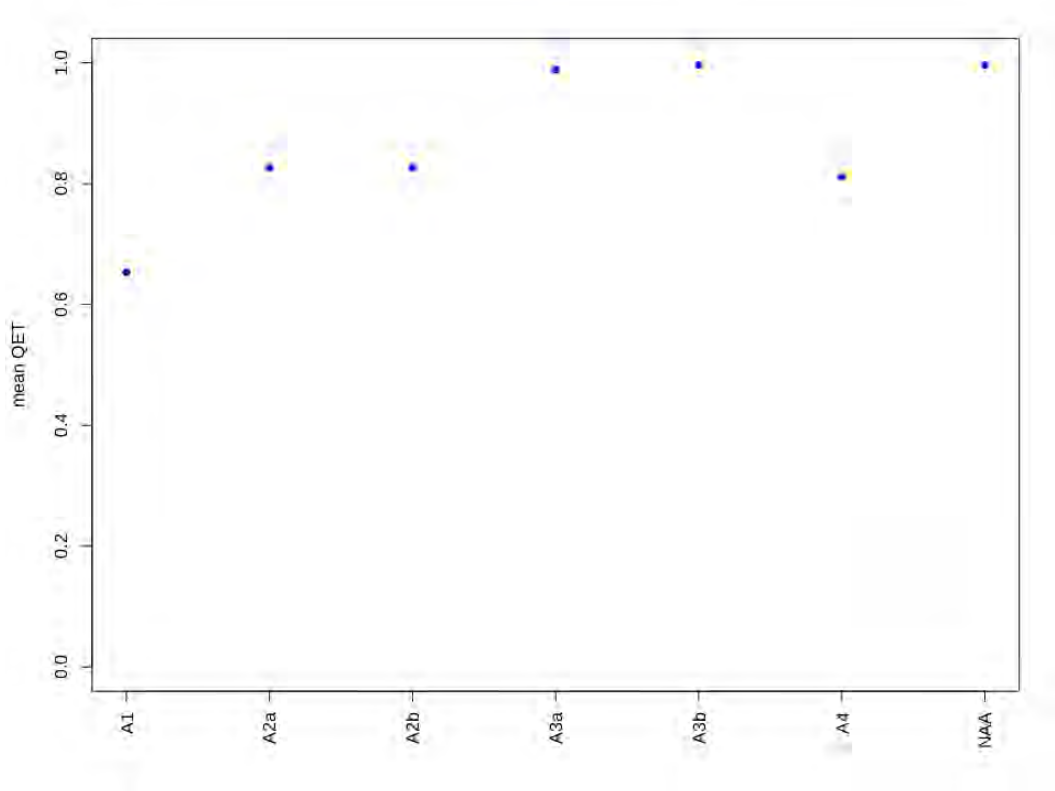

Figure 8-73. Mean of South Santiam P<QET for each of the different EIS alternatives from 10,000 simulation runs.

Note that many of the EIS alternatives do not perform well. The threshold was twice as large as for Foster and Green Peter (232 rather than 116 per dam), since the performance metrics are the sum of the two rivers.

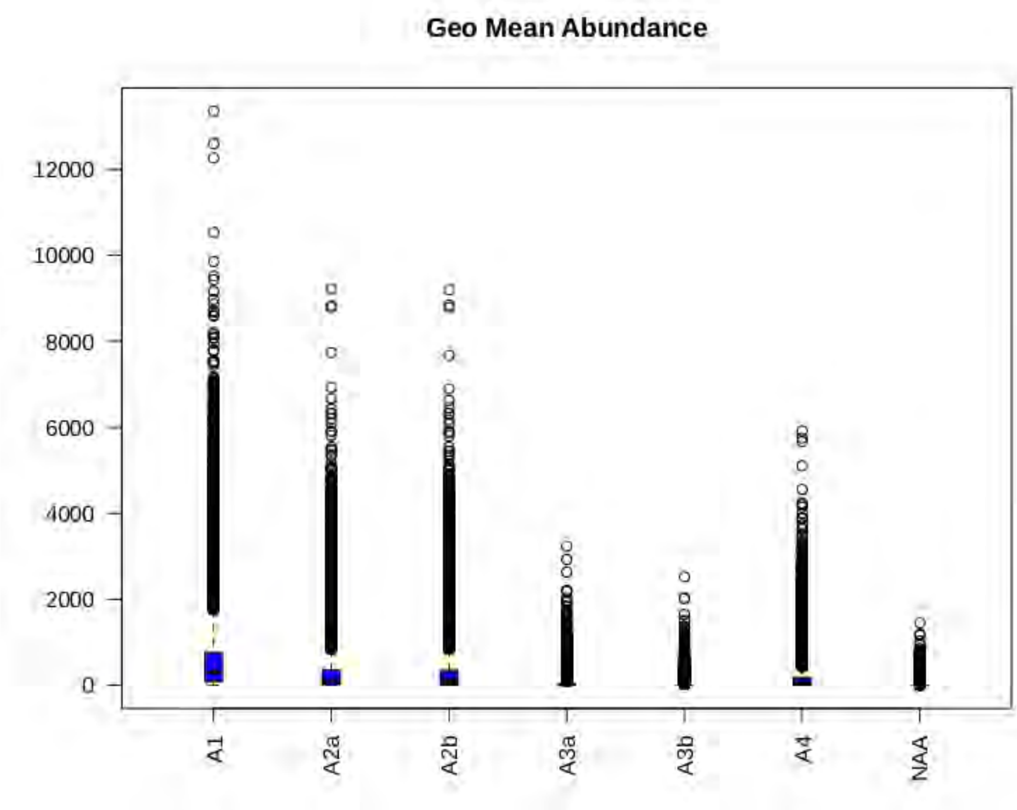

Figure 8-74. Distribution of South Santiam spawner abundance for each of the different EIS alternatives from 10,000 simulation runs.

The y axis is on a log distribution because there are long upper tails. Note that many of the EIS alternatives have distributions that appear quite similar. This is not surprising since DPS*DPE is similar for many EIS alternatives.

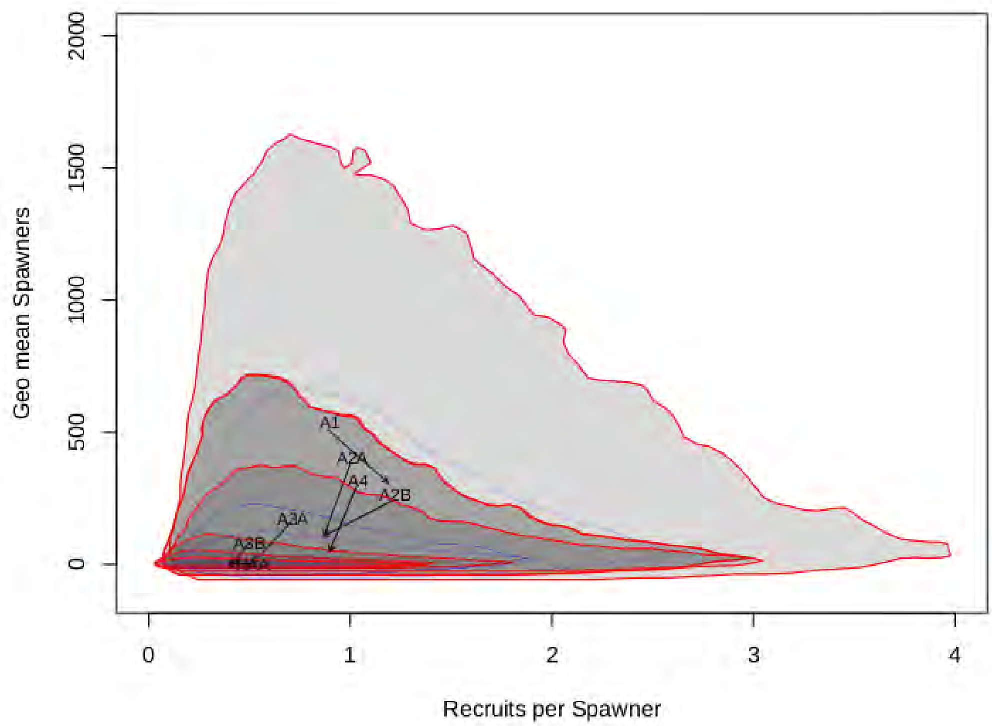

Figure 8-75. Joint Distribution of South Santiam recruits per spawner (R/S) and spawner abundance for each of the different EIS alternatives.

Note that many of the EIS alternatives have distributions that appear overlap. The label is at the mean and the red line includes 95 % of the 10,000 simulations.

Detroit

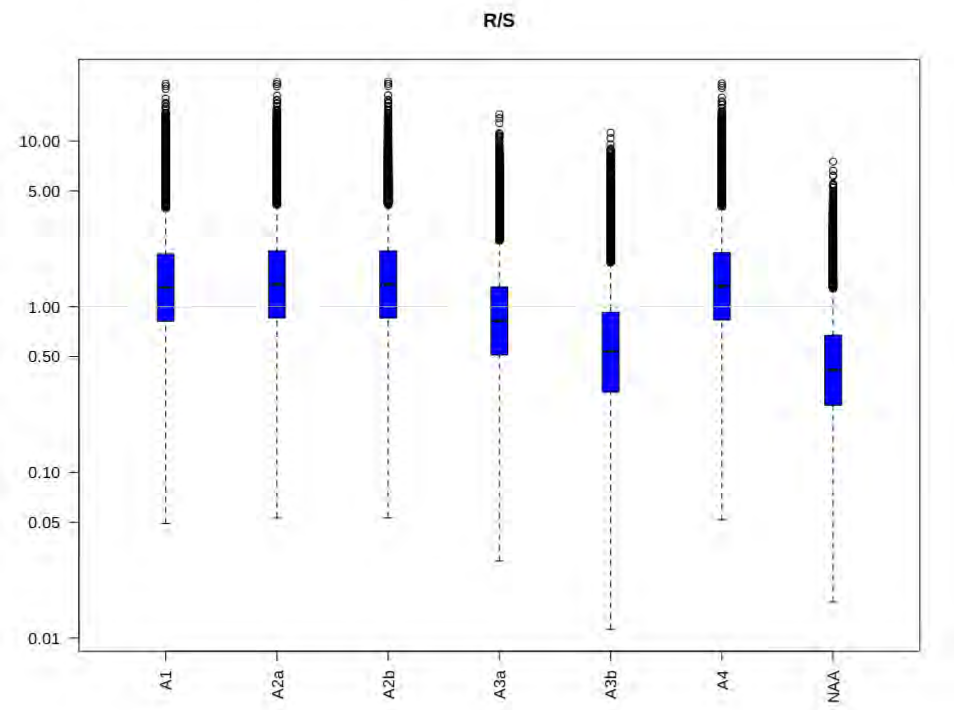

Figure 8-76. Distribution of Detroit recruits per spawner (R/S) for each of the different EIS alternatives from 10,000 simulation runs.

The horizontal grey line is at 1 for R/S. The y-axis is on a log distribution because there are long upper tails. Note that many of the EIS alternatives have distributions that appear quite similar. This is not surprising since DPS*DPE is similar for many EIS alternatives.

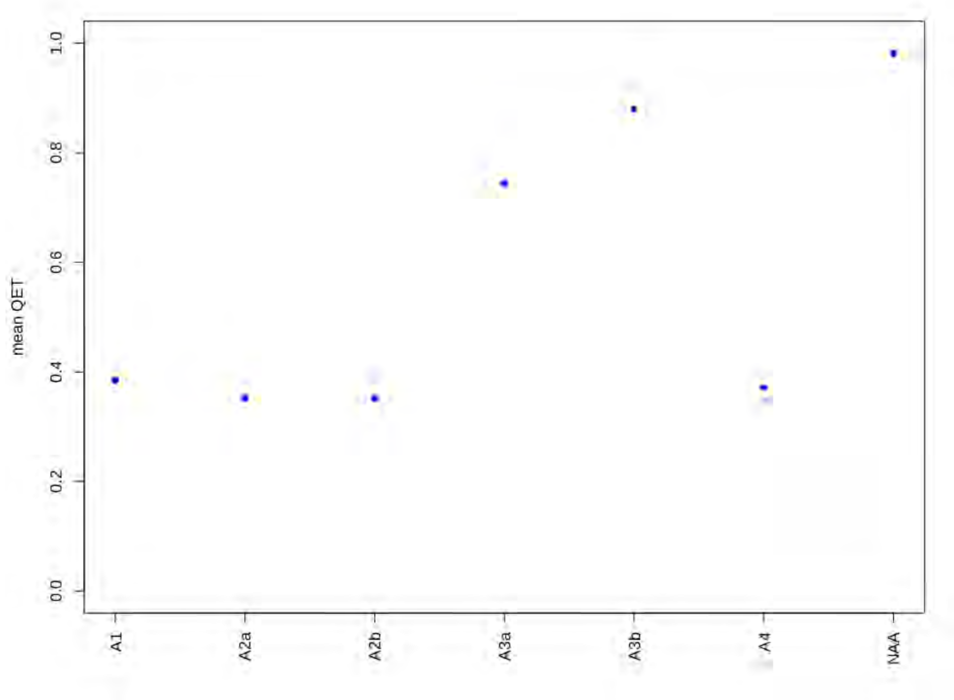

Figure 8-77. Mean of Detroit P<QET for each of the different EIS alternatives from 10,000 simulation runs.

Note that while some of the EIS alternatives do not perform too well, some have less than 50% risk of extinction.

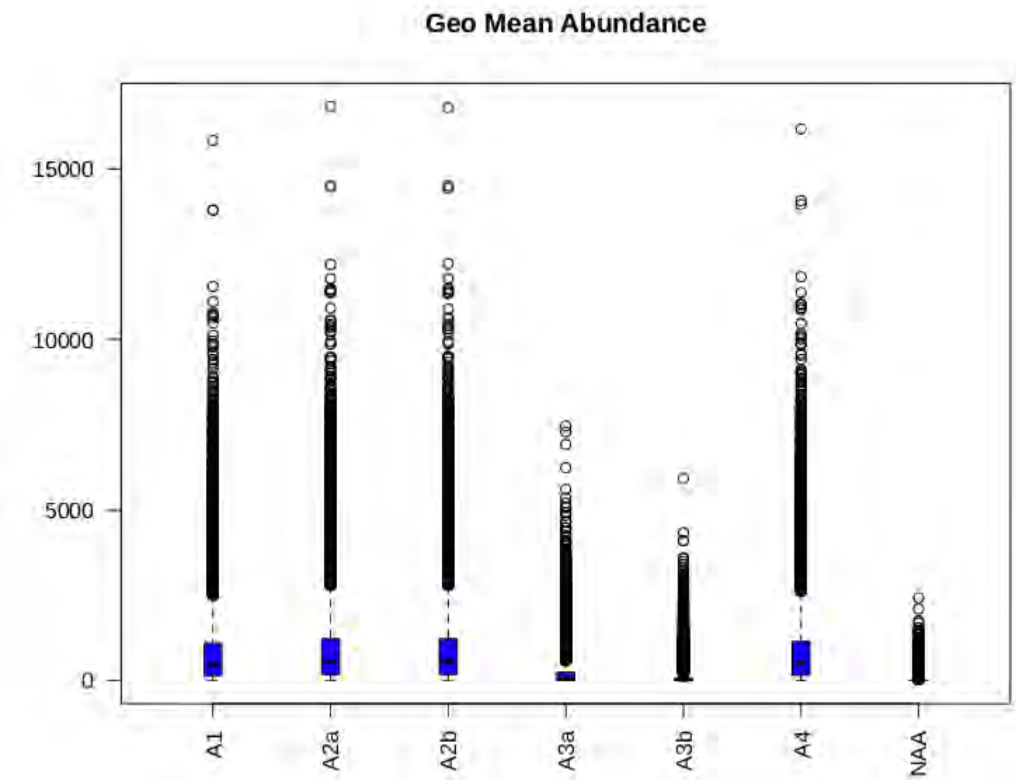

Figure 8-78. Distribution of Detroit spawner abundance for each of the different EIS alternatives from 10,000 simulation runs.

The y-axis is on a log distribution because there are long upper tails. Note that many of the EIS alternatives have distributions that appear quite similar. This is not surprising since DPS*DPE is similar for many EIS alternatives.

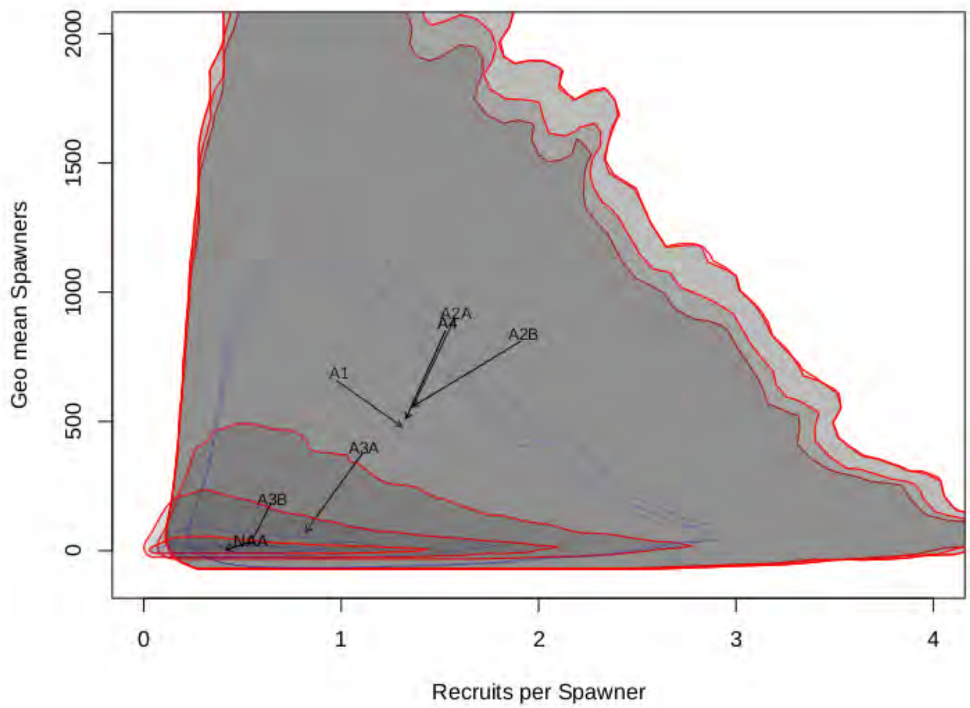

Figure 8-79. Joint Distribution of Detroit recruits per spawner (R/S) and spawner abundance for each of the different EIS alternatives.

Note that many of the EIS alternatives have distributions that appear overlap. The label is at the mean and the red line includes 95 % of the 10,000 simulations.

Sensitivity analysis of the recovery potential of winter steelhead

Isopleth Figures Comparing Sensitivity to Marine Survival and Passage Rate

A major issue in evaluating these results is how sensitive they are to marine survival. Marine survival in the 2030s may be meaningfully different than it was between 2000 and 2020. If marine survival increases over the next decade the performance metrics could be different enough to change which EIS alternatives are acceptable. These figures show the mean performance metric of 5,000 trajectories for different DPS*DPE, marine survival scenarios (Figure 3.6.1-Figure 3.6.6). The marine survival scaling just multiplies the marine survival that was simulated, so a factor of 1.5 multiplies the simulated marine survival by a factor of 1.5. The horizontal lines show the mean DPS*DPE of the different EIS alternatives. It should be noted that the mean DPS*DPE somewhat obscures the annual variability in DPE*DPE.

Since DPS*DPE is directly proportional to the product of all the freshwater survivals these figures show how freshwater survival and marine survival together determine some key performance metrics. Management procedures that only improve freshwater survival may not always be sufficient to ensure population persistence. Since the EIS alternatives only consider dam passage there may be some limitations to their efficacy. The nonlinearity is due to the population approaching carrying capacity.

Foster

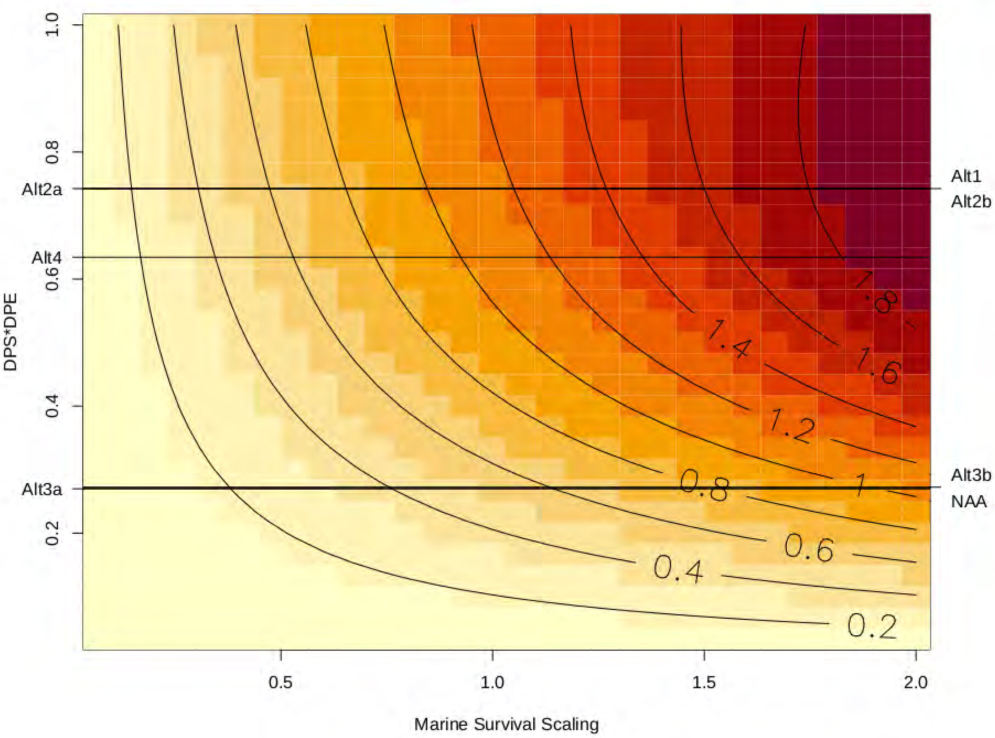

Figure 8-80. Isopleth of Foster recruits per spawner (R/S) across a range of marine survival and dam passage rates.

The horizontal lines show the mean DPE*DPS for the different EIS alternatives.

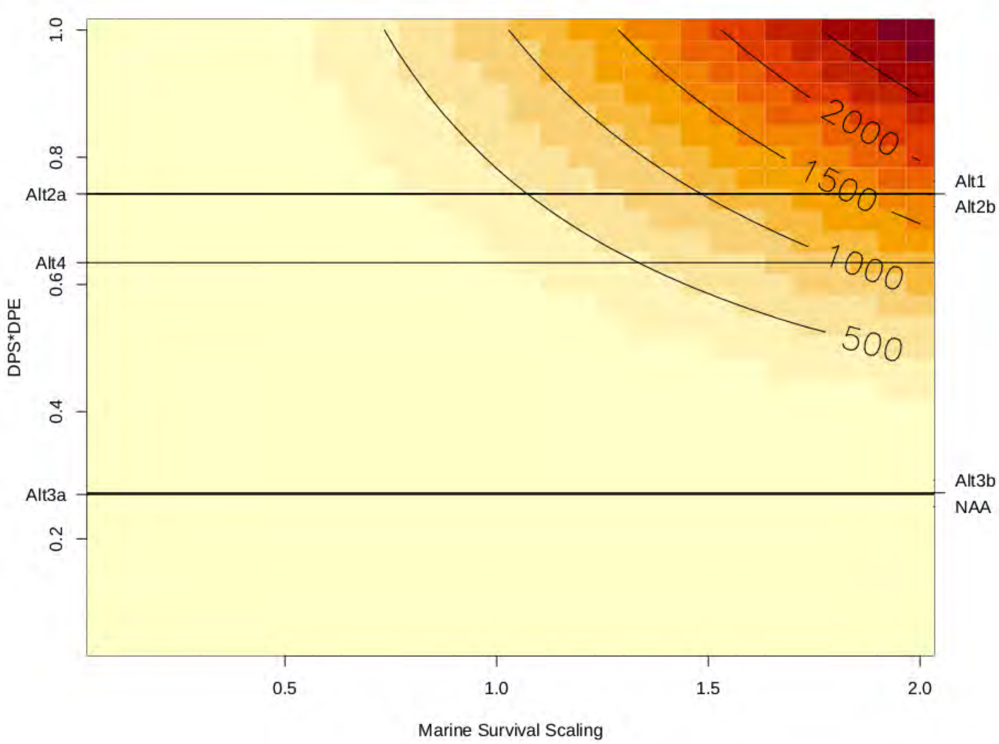

Figure 8-81. Isopleth of Foster spawner abundance across a range of marine survival and dam passage rates.

The horizontal lines show the mean DPE*DPS for the different EIS alternatives.

Green Peter

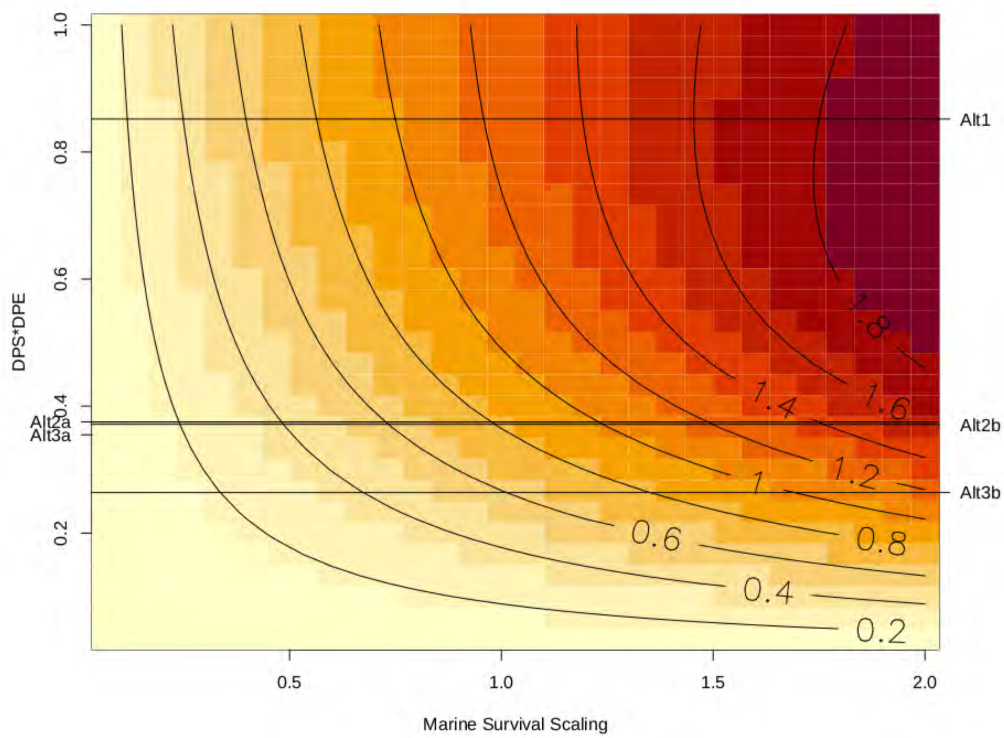

Figure 8-82. Isopleth of Green Peter recruits per spawner (R/S) across a range of marine survival and dam passage rates.

The horizontal lines show the mean DPE*DPS for the different EIS alternatives.

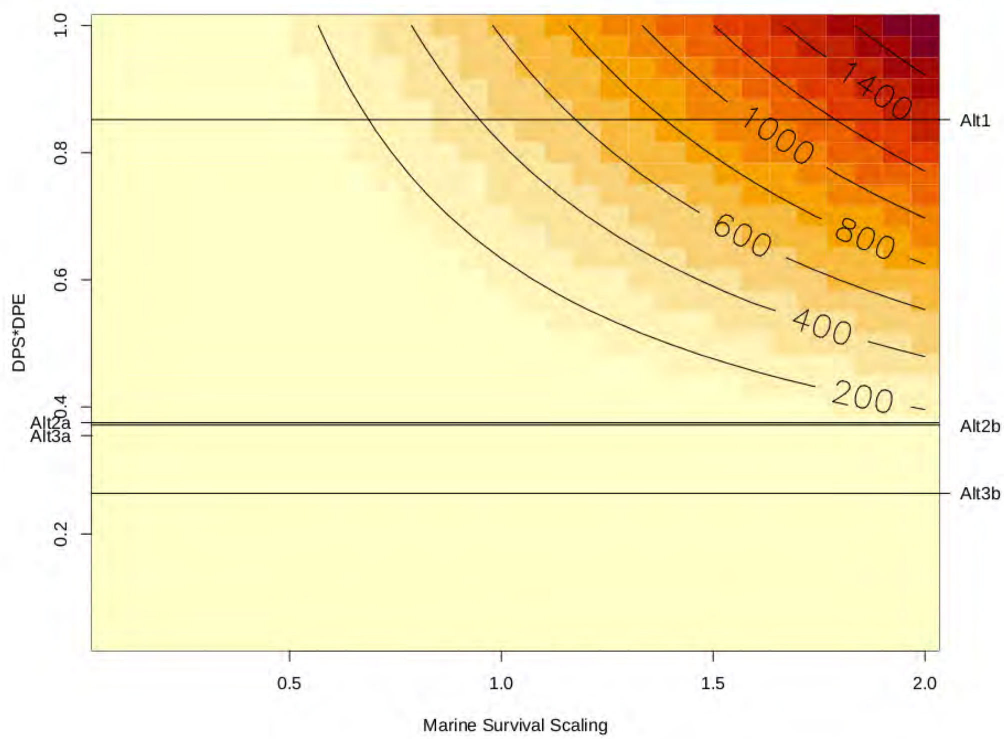

Figure 8-83. Isopleth of Green Peter spawner abundance across a range of marine survival and dam passage rates.

The horizontal lines show the mean DPE*DPS for the different EIS alternatives.

Detroit

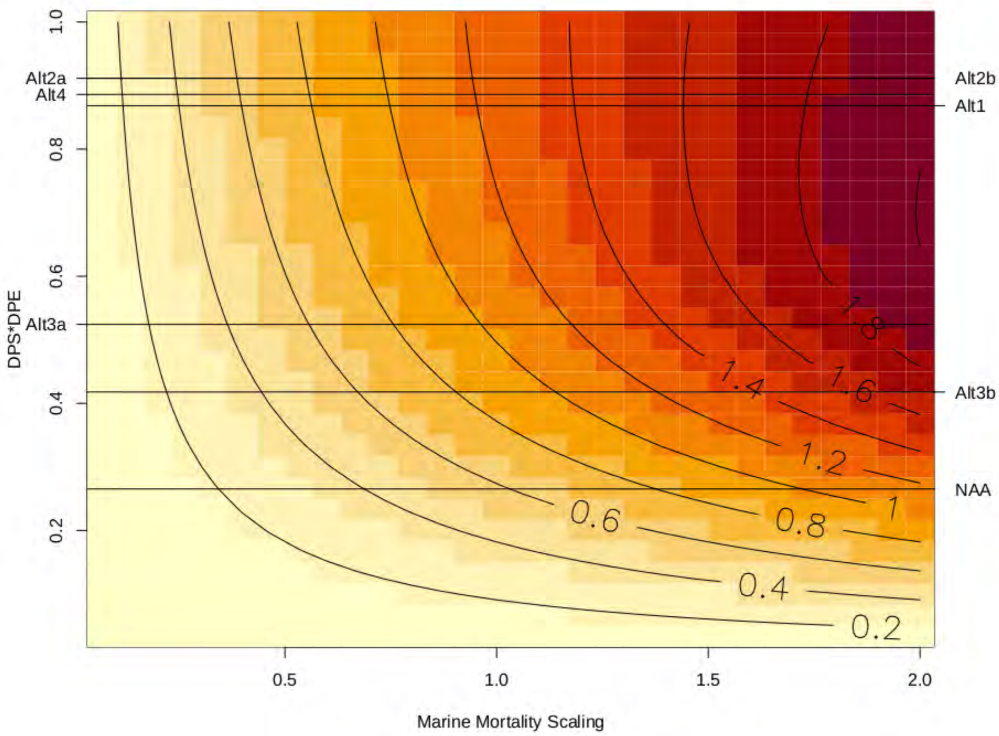

Figure 8-84. Isopleth of Detroit recruits per spawner (R/S) across a range of marine survival and dam passage rates.

The horizontal lines show the mean DPE*DPS for the different EIS alternatives.

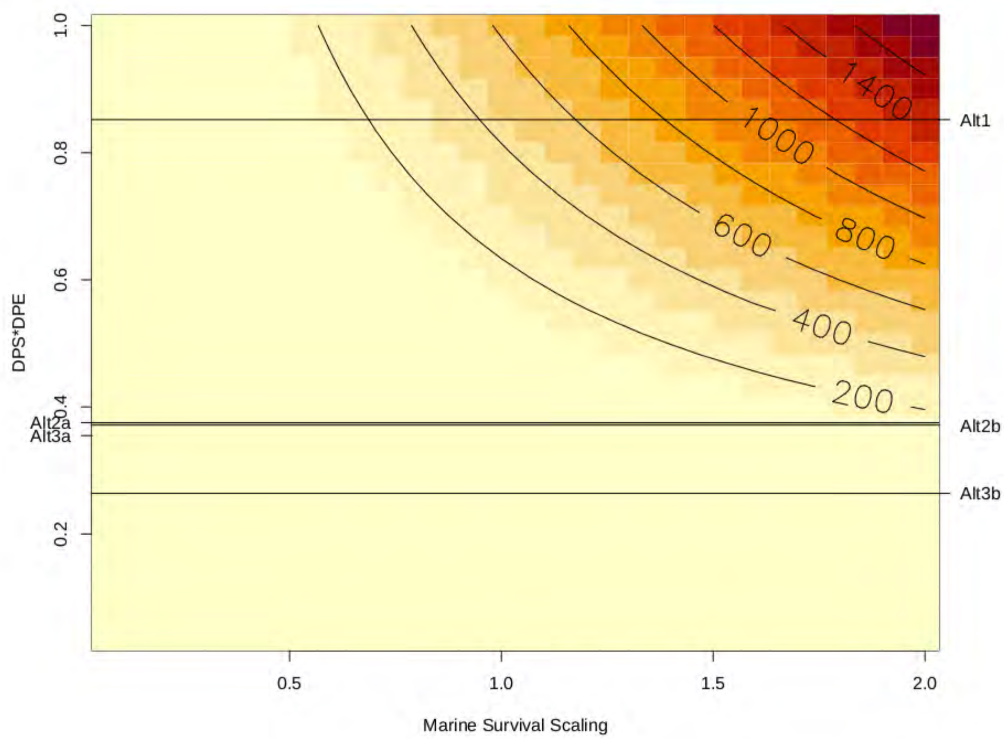

Figure 8-85. Isopleth of Detroit spawner abundance across a range of marine survival and dam passage rates.

The horizontal lines show the mean DPE*DPS for the different EIS alternatives.

Sensitivity to Specific Parameters

The projections were carried out 1) without any, and also with higher (ρ=0.9), autocorrelation in marine survival, 2) with either zero or higher kelt-return spawner survival rates (twice base case, i.e., 32%), 3) with reduced (0.75 times base case) and also increased (1.5 times base case) freshwater survival rate, and 4) with lower (0.75 times base case) and also higher (1.5 times base case) smolt capacity. Performance metrics were sensitive to all of these modifications.

With no autocorrelation in marine survival rate the proportion of trajectories that fall below the QET is lower than under the base case scenario with autocorrelation (Table 3.6.1-Table 3.6.4). This is because with autocorrelated marine survival trajectories that experience low marine survival are likely to have multiple consecutive years of low marine survival. However, without autocorrelated marine survival trajectories are unlikely to be exposed to multiple consecutive years of low marine survivals. The scenario with stronger autocorrelation than the base case allowed for some trajectories to build up high abundances when marine survival rate was high for several years (Table 3.6.5-Table 3.6.8). Note that only the marine survival rate was assumed to be autocorrelated, not DPS or DPE.

Repeat spawners are important. Removing repeat spawners led to much lower spawner abundances (Table 3.6.9-Table 3.6.12). Doubling the kelt survival rate led to much higher spawner abundances (Table 3.6.13-Table 3.6.16). Changing freshwater survival rate by a factor of 0.75 (Table 3.6.17-Table 3.6.20) and 1.5 (Table 3.6.21-Table 3.6.24) had predictable effects on the performance metrics. Changing capacity (Beverton-Holt B parameter) also just had modest effects on the performance metrics (Table 3.6.25-Table 3.6.32).

Sensitivity to Autocorrelation in Marine Survival

No autocorrelation

Table 8-24. Performance metrics of the different EIS alternatives for Foster dam in the no (ρ=0) autocorrelation case. Projected mean marine survival rates shown for comparison to base case.

FOS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.153	1.153	1.153	0.436	0.440	0.993	0.438
Geomean Spawners	288.1	288.1	288.1	7.7	8.0	179.2	7.7
P<QET Threshold	0.569	0.569	0.569	0.999	0.999	0.710	0.999
SAR	0.0423	0.0423	0.0423	0.0423	0.0423	0.0423	0.0423
Marine Survival	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422
Initial Marine Survival	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-25. Performance metrics of the different EIS alternatives for Green Peter dam in the no (ρ=0) autocorrelation case.

GP	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b
Recruits Per Spawner (R/S)	1.411	0.670	0.663	0.670	0.470
Geomean Spawners	375.3	36.0	34.7	35.9	9.7
P<QET Threshold	0.355	0.981	0.982	0.981	0.999
SAR	0.0423	0.0423	0.0423	0.0423	0.0423
Marine Survival	0.0422	0.0422	0.0422	0.0422	0.0422
Initial Marine Survival	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-26. Performance metrics of the different EIS alternatives for the entire South Santiam sub-basin in the no (ρ=0) autocorrelation case.

SS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.303	0.964	0.962	0.579	0.459	0.993	0.438
Geomean Spawners	665.5	325.0	323.8	43.8	17.7	179.2	7.7
P<QET Threshold	0.454	0.732	0.733	0.995	0.999	0.709	0.999
SAR	0.0423	0.0423	0.0423	0.0423	0.0423	0.0423	0.0423
Marine Survival	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422
Initial Marine Survival	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-27. Performance metrics of the different EIS alternatives for Detroit dam in the no (ρ=0) autocorrelation case.

DET	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.436	1.500	1.500	0.907	0.630	1.463	0.463
Geomean Spawners	926.3	1037.0	1037.0	236.9	102.2	971.3	20.8
P<QET Threshold	0.183	0.160	0.160	0.585	0.828	0.173	0.988
SAR	0.0423	0.0423	0.0423	0.0423	0.0423	0.0423	0.0423
Marine Survival	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422
Initial Marine Survival	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

High autocorrelation

Table 8-28. Performance metrics of the different EIS alternatives for Foster dam in the high (ρ=0.9) autocorrelation case.

FOS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.477	1.477	1.477	0.569	0.574	1.277	0.571
Geomean Spawners	438.9	438.9	438.9	31.8	32.7	307.3	32.1
P<QET Threshold	0.718	0.718	0.718	0.972	0.971	0.783	0.972

SAR	0.0413	0.0413	0.0413	0.0413	0.0413	0.0413	0.0413
Marine Survival	0.0424	0.0424	0.0424	0.0424	0.0424	0.0424	0.0424
Initial Marine Survival	0.0406	0.0406	0.0406	0.0406	0.0406	0.0406	0.0406

Table 8-29. Performance metrics of the different EIS alternatives for Green Peter dam in the high (ρ=0.9) autocorrelation case.

GP	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b
Recruits Per Spawner (R/S)	1.763	0.857	0.849	0.857	0.605
Geomean Spawners	453.5	75.9	73.9	75.7	29.5
P<QET Threshold	0.624	0.924	0.926	0.924	0.973
SAR	0.0413	0.0413	0.0413	0.0413	0.0413
Marine Survival	0.0424	0.0424	0.0424	0.0424	0.0424
Initial Marine Survival	0.0406	0.0406	0.0406	0.0406	0.0406

Table 8-30. Performance metrics of the different EIS alternatives for the entire South Santiam sub-basin in the high (ρ=0.9) autocorrelation case.

SS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.646	1.237	1.235	0.746	0.595	1.277	0.571
Geomean Spawners	897.3	515.6	513.7	108.2	62.5	307.2	32.1
P<QET Threshold	0.668	0.793	0.794	0.949	0.971	0.783	0.972
SAR	0.0413	0.0413	0.0413	0.0413	0.0413	0.0413	0.0413
Marine Survival	0.0424	0.0424	0.0424	0.0424	0.0424	0.0424	0.0424
Initial Marine Survival	0.0406	0.0406	0.0406	0.0406	0.0406	0.0406	0.0406

Table 8-31. Performance metrics of the different EIS alternatives for Detroit dam in the high (ρ=0.9) autocorrelation case.

DET	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.808	1.885	1.885	1.157	0.808	1.840	0.599

Geomean Spawners	1146.1	1258.2	1258.2	395.9	213.4	1191.8	68.2
P<QET Threshold	0.484	0.459	0.459	0.733	0.831	0.473	0.939
SAR	0.0413	0.0413	0.0413	0.0413	0.0413	0.0413	0.0413
Marine Survival	0.0424	0.0424	0.0424	0.0424	0.0424	0.0424	0.0424
Initial Marine Survival	0.0406	0.0406	0.0406	0.0406	0.0406	0.0406	0.0406

Sensitivity to Repeat Spawner Survival

No Repeat Spawners

Table 8-32. Performance metrics of the different EIS alternatives for Foster dam in the no repeat spawner case.

FOS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.333	1.333	1.333	0.506	0.510	1.149	0.507
Geomean Spawners	154.6	154.6	154.6	3.9	4.0	94.7	3.9
P<QET Threshold	0.811	0.811	0.811	0.999	0.999	0.884	0.999
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-33. Performance metrics of the different EIS alternatives for Green Peter dam in the no repeat spawner case.

GP	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b
Recruits Per Spawner (R/S)	1.633	0.776	0.768	0.775	0.544
Geomean Spawners	209.8	18.0	17.3	17.9	4.7
P<QET Threshold	0.710	0.991	0.991	0.991	0.999
SAR	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-34. Performance metrics of the different EIS alternatives for the entire South Santiam sub-basin in the no repeat spawner case.

SS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.509	1.118	1.117	0.672	0.532	1.149	0.507
Geomean Spawners	366.1	173.1	172.5	21.9	8.7	94.7	3.9
P<QET Threshold	0.762	0.897	0.898	0.996	0.999	0.884	0.999
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-35. Performance metrics of the different EIS alternatives for Detroit dam in the no repeat spawner case.

DET	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.662	1.735	1.735	1.049	0.729	1.692	0.536
Geomean Spawners	516.9	584.1	584.1	122.5	49.3	544.2	10.2
P<QET Threshold	0.497	0.460	0.460	0.840	0.939	0.481	0.993
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

High Repeat Spawner Survival

Table 8-36. Performance metrics of the different EIS alternatives for Foster dam in the high repeat spawner case.

FOS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.245	1.245	1.245	0.489	0.493	1.081	0.490
Geomean Spawners	458.3	458.3	458.3	22.6	23.4	308.6	22.9
P<QET Threshold	0.507	0.507	0.507	0.982	0.982	0.625	0.982
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-37. Performance metrics of the different EIS alternatives for Green Peter dam in the high repeat spawner case.

GP	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b
Recruits Per Spawner (R/S)	1.467	0.728	0.720	0.727	0.516
Geomean Spawners	488.3	67.1	65.1	66.9	22.5
P<QET Threshold	0.386	0.926	0.929	0.926	0.987
SAR	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-38. Performance metrics of the different EIS alternatives for the entire South Santiam sub-basin in the high repeat spawner case.

SS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.367	1.058	1.057	0.626	0.508	1.081	0.490
Geomean Spawners	949.3	526.4	524.4	89.9	46.0	308.6	22.9
P<QET Threshold	0.442	0.653	0.654	0.960	0.985	0.625	0.982
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-39. Performance metrics of the different EIS alternatives for Detroit dam in the high repeat spawner case.

DET	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.503	1.564	1.564	0.975	0.685	1.528	0.512
Geomean Spawners	1246.2	1375.3	1375.3	390.7	197.9	1298.8	51.9
P<QET Threshold	0.199	0.176	0.176	0.540	0.737	0.189	0.939
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Sensitivity to Freshwater Survival

Reduced Freshwater Survival

Table 8-40. Performance metrics of the different EIS alternatives for Foster dam in the reduced freshwater survival case.

FOS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.057	1.057	1.057	0.388	0.392	0.905	0.389
Geomean Spawners	60.5	60.5	60.5	1.1	1.1	34.3	1.1
P<QET Threshold	0.946	0.946	0.946	0.999	0.999	0.971	0.999
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-41. Performance metrics of the different EIS alternatives for Green Peter dam in the reduced freshwater survival case.

GP	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b
Recruits Per Spawner (R/S)	1.360	0.615	0.608	0.614	0.425
Geomean Spawners	126.5	7.8	7.5	7.8	1.9
P<QET Threshold	0.863	0.997	0.996	0.997	0.999
SAR	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-42. Performance metrics of the different EIS alternatives for the entire South Santiam sub-basin in the reduced freshwater case.

SS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.257	0.848	0.846	0.544	0.415	0.904	0.389
Geomean Spawners	188.0	68.8	68.4	8.9	3.0	34.3	1.1
P<QET Threshold	0.908	0.973	0.973	0.999	0.999	0.971	0.999
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-43. Performance metrics of the different EIS alternatives for Detroit dam in the reduced freshwater case.

DET	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.371	1.437	1.437	0.838	0.576	1.398	0.417
Geomean Spawners	268.7	310.9	310.9	50.2	19.9	285.6	3.4
P<QET Threshold	0.767	0.742	0.741	0.949	0.983	0.757	0.999
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Increased Freshwater Survival

Table 8-44. Performance metrics of the different EIS alternatives for Foster dam in the increased freshwater survival case.

FOS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.577	1.577	1.577	0.674	0.679	1.392	0.675
Geomean Spawners	681.2	681.2	681.2	53.6	55.2	493.1	54.1
P<QET Threshold	0.273	0.273	0.273	0.942	0.939	0.389	0.941
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-45. Performance metrics of the different EIS alternatives for Green Peter dam in the increased freshwater survival case.

GP	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b
Recruits Per Spawner (R/S)	1.768	0.954	0.946	0.954	0.696
Geomean Spawners	585.3	111.1	108.2	110.8	42.9
P<QET Threshold	0.228	0.851	0.856	0.851	0.969
SAR	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-46. Performance metrics of the different EIS alternatives for the entire South Santiam sub-basin in the increased freshwater case.

SS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.673	1.382	1.381	0.822	0.691	1.391	0.675
Geomean Spawners	1268.6	793.2	790.4	165.1	98.3	493.0	54.1
P<QET Threshold	0.249	0.437	0.438	0.903	0.954	0.388	0.941
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-47. Performance metrics of the different EIS alternatives for Detroit dam in the increased freshwater case.

DET	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.802	1.865	1.865	1.240	0.891	1.828	0.690
Geomean Spawners	1552.7	1684.3	1684.3	596.3	327.6	1606.7	105.9
P<QET Threshold	0.072	0.061	0.061	0.319	0.545	0.067	0.850
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Sensitivity to Smolt Capacity

Low Capacity

Table 8-48. Performance metrics of the different EIS alternatives for Foster dam in the low freshwater capacity case.

FOS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.322	1.322	1.322	0.503	0.508	1.140	0.505
Geomean Spawners	192.4	192.4	192.4	6.6	6.8	122.9	6.7
P<QET Threshold	0.771	0.770	0.771	0.998	0.998	0.851	0.998
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-49. Performance metrics of the different EIS alternatives for Green Peter dam in the low freshwater capacity case.

GP	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b
Recruits Per Spawner (R/S)	1.613	0.770	0.762	0.769	0.540
Geomean Spawners	238.5	25.4	24.6	25.3	7.5
P<QET Threshold	0.670	0.986	0.987	0.986	0.999
SAR	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-50. Performance metrics of the different EIS alternatives for the entire South Santiam sub-basin in the low freshwater capacity case.

SS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.489	1.113	1.111	0.664	0.529	1.140	0.505
Geomean Spawners	432.6	218.4	217.6	32.1	14.4	122.9	6.7
P<QET Threshold	0.721	0.871	0.871	0.994	0.998	0.851	0.998
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-51. Performance metrics of the different EIS alternatives for Detroit dam in the low freshwater capacity case.

DET	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.639	1.711	1.711	1.039	0.722	1.669	0.533
Geomean Spawners	596.1	666.5	666.5	160.5	73.9	624.7	16.8
P<QET Threshold	0.439	0.404	0.404	0.794	0.911	0.424	0.989
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

High Capacity

Table 8-52. Performance metrics of the different EIS alternatives for Foster dam in the high freshwater capacity case.

FOS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.334	1.334	1.334	0.505	0.510	1.149	0.507
Geomean Spawners	357.7	357.7	357.7	11.9	12.4	227.3	12.1
P<QET Threshold	0.653	0.653	0.653	0.992	0.992	0.753	0.992
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-53. Performance metrics of the different EIS alternatives for Green Peter dam in the high freshwater capacity case.

GP	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b
Recruits Per Spawner (R/S)	1.625	0.773	0.765	0.772	0.542
Geomean Spawners	466.3	49.2	47.6	49.1	14.5
P<QET Threshold	0.484	0.948	0.950	0.949	0.992
SAR	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-54. Performance metrics of the different EIS alternatives for the entire South Santiam sub-basin in the high freshwater capacity case.

SS	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.510	1.106	1.104	0.674	0.532	1.149	0.507
Geomean Spawners	827.6	408.1	406.5	61.2	26.9	227.3	12.1
P<QET Threshold	0.561	0.762	0.764	0.976	0.992	0.753	0.992
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Table 8-55. Performance metrics of the different EIS alternatives for Detroit dam in the high freshwater capacity case.

DET	Alt 1	Alt 2a	Alt 2b	Alt 3a	Alt 3b	Alt 4	NAA
Recruits Per Spawner (R/S)	1.662	1.736	1.736	1.048	0.728	1.693	0.536
Geomean Spawners	1132.5	1268.0	1268.0	300.5	137.6	1187.5	30.9
P<QET Threshold	0.323	0.295	0.295	0.675	0.832	0.312	0.968
SAR	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422	0.0422

Discussion

The aim of the analysis reported here was to compute and report performance metrics for the evaluation of several EIS measures for dams in the four Upper Willamette Basin. The performance metrics of interest were specified by USACE and were based on consultations with government agencies and stakeholder organizations. The performance metrics included measures of long-term average abundance, population productivity levels in the years following initial implementation, and the long-term risk of quasi-extinction of the above-dam population. In order to compute these performance metrics for each of the EIS alternatives, population dynamics models for the above-dam populations were constructed and parameterized using results from previous studies and available datasets from each of the sub-basins. Outputs of USACE's Fish Benefits Workbook software package (FBW) provided the dam passage measures (passage efficiency and survival) in each EIS alternative for each of the dams. FBW gave outputs for dam passage efficiency and dam passage survival rates that often differed considerably between fry, sub-yearling and yearling juvenile stages. However, FBW software was not designed to compute the performance metrics required of the EIS. These could only be computed with the application of full life cycle models for the above-dam natural origin populations that could be projected for several generations. Also, the ranking and relative performance of the EIS alternatives based on the performance metrics were not deterministically linked to the FBW outputs. FBW provided outputs only with regards to juvenile salmon and steelhead. Density dependence in egg-adult survival, and pre-spawn mortality processes which are also density dependent and can also be important determinants of fish population responses to alterations to dam passage structures and operations not accounted for in FBW calculations. Performance metrics on extinction thresholds, i.e., P<QET, were sensitive to population responses to variability in FBW outputs, not just their mean values. In some instances, the probability of exceeding QETs for different EIS alternatives were markedly different when in contrast the mean values for the FBW outputs were practically the same between the EIS alternatives; here the larger variances in FBW outputs, despite having the same mean values, for some of the alternatives, led to a higher frequency of occurrence of QET being exceeded.

Even for relatively short-lived species such as salmon and steelhead, life cycle models can contain numerous structural components and may require a large body of data and information to make them functional and credible for the purpose of evaluating fish population responses to different sets of dam passage structures and operations. This was the case for the LCMs that we developed and applied in this report. Due to limitations in time and the availability of data for model fitting, the preferred approach of statistically fitting the LCM to all available data to estimate its parameters could not be implemented. Instead, we identified from available literature sources and data sets plausible parameter values and implemented these as prior distributions for LCM parameters. To ensure that the LCMs were consistent with time series of historical abundance and spawner age-composition records, the LCMs were fitted to these time series by freeing up a few key parameters and fixing the remainder of the LCM parameters at their prior central tendencies or prior modal values. For the fits of the LCM to the data for each sub-basin, it was necessary to free up a time series of annual process error deviates. As the adult counts were obtained at the dam tailrace, these counts were deemed to be highly accurate and not due to observation error. Larger discrepancies between deterministic model predictions and adult counts were thus accommodated mainly in the process error terms. The process error deviates were then bootstrapped to further account for uncertainty in model predictions when the fitted LCMs were projected to compute the PMs for the EIS alternatives. In all four sub-basins, the fitted LCMs produced close approximations of the observed time series of adult counts. This feature of the approach thus conformed to the conventional standards of model fitting in stock assessment which typically require a close fit between model predictions and time series of abundance (Francis 2011).

LCM output of PM values facilitated ranking of EIS alternatives in terms of how well key conservation objectives could be met rather than the passage performance values provided by FBW. Rankings of the EIS alternatives tended to be similar across the three categories of PMs and also between the different sub-basins, though with some exceptions. For example, for winter steelhead, Alternatives 1, 2a, and 2b ranked nearly the same and the highest for the Foster dam but Alternative 1 ranked higher for the Green Peter dam. And for the South Santiam sub-basin, Alternative 1 ranked overall highest while for the North Santiam sub-basin, Alternatives 2a and 2b tied for the highest ranking. There was still a fairly wide band of uncertainty in key performance metrics such as the ones for productivity and abundance and there remains considerable uncertainty for many LCM parameters.

The analysis also included tests of sensitivity of model predictions of key performance metrics to ranges of values for LCM input parameters. PM calculations were sensitive to values for smolt-adult survival rates, pre-spawn mortality rates, juvenile freshwater survival rates above dams, and dam passage survival rates. For winter steelhead, PMs were also sensitive to assumptions about smolt-adult survival rates, future survival rates of kelts, mean freshwater survival rates for juveniles, the freshwater capacity parameter for juveniles, and the level of autocorrelation at lag 1 in marine survival rate. There remains considerable uncertainty in all of these parameters. Should the priors formulated for them poorly represent the true values, the PMs computed for the EIS alternatives could deviate considerably from what they should be and even the actual rankings of the EIS alternatives in terms of the PMs could be quite different from results found in this report.

Potential Extensions to the LCMs

Some potential extensions to the LCMs include the following:

1. Move model onto a monthly timestep

The LCMs are currently run on an annual time step with within-year processes modeled sequentially. However, FBW operates at a daily time step and DPE and DPS outputs can be calculated at weekly, monthly or annual time steps. As there is considerable seasonality on a month-by-month basis, especially at the juvenile stages in freshwater, monthly time steps could offer a more accurate representation of freshwater survival and growth processes. However, a monthly time step model would have far greater data requirements than models at coarser time steps and this trade-off would need to be assessed carefully before a decision was made to adopt a monthly time step model.

2. Implement full Bayesian MCMC versions in ADMB/TMB (Fournier et al. 2012; Kristensen et al. 2016) to improve calibration and account for parameter correlation and uncertainty

A relatively simple model fitting approach was applied that freed up a few model parameters and a time series of annual process error deviates to fit the available time series of adult counts and spawner age composition data. This required fixing the other parameters at their prior median or modal values. A more refined approach that would better account for posterior correlation between, and uncertainty in, model parameters would be to apply a Bayesian MCMC approach. It is uncertain whether this latter approach could be got to work numerically and it could take considerable effort to find this out and get the MCMC algorithm working as intended. It may also be better to fit the LCM also to the release and recapture records for the coded wire tag releases, time permitting, to allow more consistent estimation of survival rate parameters.

3. Model reservoir and downstream growth for prediction of size-based SAS

Peterson et al. (2022) incorporated a model of downstream growth and used smolt size and season to predict smolt-adult survival rates in spring Chinook salmon in the Upper Willamette River basin. A maximum likelihood approach was taken for this which may not necessarily appropriately represent uncertainties in parameter estimates for simulation modelling. We intend to reformulate a downstream growth model and also use smolt size and season to predict SAS and would do this in a Bayesian modelling framework of PIT tag data sets to allow uncertainties to be more appropriately represented when model projections are carried out.

4. Develop new scenarios for freshwater and marine survival rate responses to climate change scenarios

Given the progressive and credible future changes to freshwater and marine conditions predicted under climate change, it would be appropriate to develop extensions to represent potential freshwater and marine survival rate responses to a carefully formulated set of climate change scenarios. Dam passage measures and dam operations that could meet conservation objectives when climate change scenarios are considered could be somewhat different from those that could be found to do so under the current set of scenarios which ignore climate change.

5. Evaluate sensitivity of results to different hatchery mitigation options

Hatchery production has historically been required to mitigate for blocked access to above-dam habitat. Should the implementation of new dam passage measures successfully rebuild above dam salmon and steelhead populations, the hatchery mitigation measures could eventually be adjusted. The UBC research group is currently extending its LCMs to enable evaluation of alternative approaches to adjust production and transport of hatchery fish in response to observed changes in salmon population abundance above dams. This model extension will include plausible interactions between hatchery and natural origin fish, and will be used to evaluate changes to the hatchery program according to proposed measures. Results from these new extended LCMs are expected to be available within the next year.

6. Refine modelling of reservoir survival rates with further studies on ecological responses to spring/fall drawdowns

Fairly rapid and well-pronounced ecological responses were observed after fall drawdowns in the Fall Creek Reservoir starting in 2011 (Murphy et al. 2019). While it could be expected that analogous ecological responses could occur with the implementation of fall and spring drawdowns in other Willamette reservoirs, it could be expected that the ecological responses eventually realized could be considerably different than those that have been observed in the Fall Creek Reservoir. This is because the extent of drawdown could be less, and there exist considerable differences in fish species composition, hydrology, geomorphology and bathymetry between reservoirs in the Upper Willamette Basin. Therefore, it would appear that understanding of potential ecological responses to spring and fall drawdowns could only be obtained with carefully designed monitoring before and after the implementation of spring and fall drawdowns on a reservoir-by-reservoir basis.

7. Refine modelling of below-dam PSM as function of accumulated temperature

Some studies have shown that below-dam PSM can be predicted based on accumulated temperature (Peterson et al. 2022). The UBC team's investigations of this in the South Santiam sub-basin have been consistent with these findings. However, there wasn't time to apply this investigation to other sub-basins and implement it as part of the EIS.

8. Develop new model components for rearing habitat capacities for juveniles

The LCMs represent density dependent survival using Beverton-Holt functions of fry abundance versus egg deposition in Chinook salmon and smolt abundance versus egg deposition in steelhead. However, density dependent growth and survival during other life history phases may be important, particularly at higher abundances. It may thus be appropriate to review available literature (e.g., ISAB 2015) and recent studies by the USGS to formulate new model components for rearing capacities for juvenile salmon and steelhead.

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Appendices

Spring Chinook Salmon Model Specifications

Model Definitions applied in all four sub-basins

Table 8-56. Index and location definitions used in the Chinook salmon life cycle model.

Symbol	Description
a	Adult age (a=2, …, 6)
t	Brood year starting in September (t=0, …, 30). Note this is not a calendar year timestep, September is assumed to be when eggs go into gravel.
N	Natural-origin Chinook salmon
H	Hatchery-origin Chinook salmon
NOR	Natural-origin adult returns
HOR	Hatchery-origin returns
NO	Natural-origin adult outplants
HO	Hatchery-origin adult outplants
NS	Natural-origin spawners
HS	Hatchery-origin spawners
E	Eggs
J	Emergent fry
F	Fry life stage (see Section 2.2.3 for life stage definitions)
S	Subyearling life stage
Y	Yearling life stage
str	Juvenile migrant type rears only in natal stream before attempted dam passage (see Section 2.2.3 for juvenile migrant type definitions)
resS	Juvenile migrant type rears in reservoir over summer before attempted dam passage
resW	Juvenile migrant type rears in reservoir over winter before attempted dam passage
resSW	Juvenile migrant type rears in reservoir over both summer and winter before attempted dam passage
nat	Natal stream
res	Head of reservoir
fby	Reservoir forebay
BCLTR	Big Cliff dam tailrace
GPRTR	Green Peter dam tailrace
FOSTR	Foster dam tailrace
CGRTR	Cougar dam tailrace
HCTR	Hills Creek dam tailrace
LOPTR	Lookout Point dam tailrace

Model Parameters and Equations for each sub-basin

North Santiam

Table 8-57. Parameter and other input variable values for the North Santiam Chinook salmon life cycle model.

aValues for parameters without a time subscript drawn from a distribution vary between simulations.
bα and β parameters of a beta distribution are derived from a mean and CI.

Symbol	Description	Valuea	Reference
Model parameters			
NOR^{init}	Initial number of natural-origin returns to Minto adult collection facility	682	Mean number of NOR to Minto, 2015-2021 (ODFW count data).
HO^{max}	Maximum hatchery-origin outplants	1,500	HGMP (section 15.2) states that up to 1,500 HOR spring Chinook may be outplanted above DET until long-term passage solution is approved (ODFW & USACE 2016b). N.B. Mean number of outplants 2011-2021 was only 1,112 (ODFW count data).
κ	Logistical cap on total outplant numbers (natural-origin + hatchery-origin)	5,428	ODFW & USACE (2016b), Columbia River Partnership Task Force (2020)
PSM_t^{below}	Pre-spawn mortality (PSM) en route to dam	Beta(130.543, 654.937)b; mean=0.165 (95% CI 0.141-0.193)	Based on a 2011-2014 radio-telemetry study (Keefer et al. 2017)
PSM_t^{above}	Pre-spawn mortality (PSM) on spawning grounds above dam	$f(7DADM_t, pHO_t)$	Adjusted function from Bowerman et al. (2018), see Section 2.4.1.
$7DADM_t$	7-day average of daily maximum temperature (oC) above Detroit reservoir	Bootstrap 1947-2019	Data from USGS gage 14178000. Years with missing data were back-filled using the

			water-year type mean value. See Section 2.4.1.
η^N	Fecundity (natural origin adults)	4,500	Zabel et al. (2015), p5.7
η^H	Fecundity (hatchery origin adults)	4,000	Zabel et al. (2015), p5.7
ν	Sex ratio (female spawners to total spawners) for natural- and hatchery-origin adults	0.44	Mean from observed returns to Minto (ODFW count data, 2009-2021)
RRS	Relative reproductive success of HO to NO adults	Triang(0.39, 0.53, 0.84)	Based on analysis of Christie et al. (2014), see Section 2.4.3.
α	Beverton-Holt egg-fry survival	0.55	Zabel et al. (2015), p5.9
β	Beverton-Holt egg capacity	Norm(6E+06, 1E+05) + Norm(1.25E+07, 2.5E+05)	Zabel et al. (2015), p5.7 (reach D+E)
$\varphi_{t,\{F,S,Y\}}^{nat}$	Fry-migrant survival in natal stream (fry mover; subyearling stayer; yearling stayer)	0.75; 0.4; 0.3	Zabel et al. (2015), p5.9
$\varphi_{t,F\to F}^{DET}$	Detroit reservoir survival (fry to fry)	1	Zabel et al. (2015), p5.10
$\varphi_{t,F\to S}^{DET}$	Detroit reservoir survival (fry to subyearling)	Triang(0.15, 0.2, 0.35)	Zabel et al. (2015), p5.10
$\varphi_{t,S\to S}^{DET}$	Detroit reservoir survival (subyearling to subyearling)	1	Zabel et al. (2015), p5.10
$\varphi_{t,S\to Y}^{DET}$	Detroit reservoir survival (subyearling to yearling)	Unif(0.4, 0.9)	Zabel et al. (2015), p5.10
$\varphi_{t,Y\to Y}^{DET}$	Detroit reservoir survival (yearling to yearling)	1	Zabel et al. (2015), p5.10
$\varphi_{t,F}^{rss}$	River-smolt survival for fry (BCL-SUJ)	Median = 0.156 (CV = 0.334)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p5.11 (see Section 2.2.6 and Appendix C)
$\varphi_{t,S_{resS}}^{rss}$	River-smolt survival for subyearlings that spent	Median = 0.441 (CV = 0.192)	CJS model posterior output adjusted for

	summer in reservoir (BCL-SUJ)		tagging/hatchery effects (see Appendix C)
$\varphi_{t,S_{str}}^{rss}$	River-smolt survival for subyearlings that spent summer in natal stream (BCL-SUJ)	Median = 0.441 (CV = 0.192)	CJS model posterior output adjusted for tagging/hatchery effects (see Appendix C)
$\varphi_{t,Y_{resSW}}^{rss}$	River-smolt survival for yearlings that spent summer and winter in reservoir (BCL-SUJ)	Median = 0.623 (CV = 0.098)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p5.11 (see Section 2.2.6 and Appendix C)
$\varphi_{t,Y_{resW}}^{rss}$	River-smolt survival for yearlings that spent only winter in reservoir (BCL-SUJ)	Median = 0.623 (CV = 0.098)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p5.11
$\varphi_{t,Y_{str}}^{rss}$	River-smolt survival for yearlings that reared in natal stream (BCL-SUJ)	Median = 0.623 (CV = 0.098)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p5.11
$\varphi_{0+\rightarrow 3}^{o}$	Survival from natural mortality in the ocean (0+ to age-3 adult)	0.013	Estimated during model calibration, using CJS model posterior output as prior (see Appendix C and Appendix D). N.B. 0+ pass SUJ before 1st birthday.
$\varphi_{1+\rightarrow 3}^{o}$	Survival from natural mortality in the ocean (1+ to age-3 adult)	0.132	Estimated during model calibration, using CJS model posterior output as

			prior (see Appendix C and Appendix D)
$\varphi^{o}_{a\rightarrow a+1}$	Survival from natural mortality in the ocean (age a≥3 and a≤5)	0.397; 0.500; 0.630	Assumed by CTC (2021), see Appendix D for details.
ε_t	Annual deviates in early ocean survival	{1.197, 0.082, -0.244, 0.171, 0.784, -0.511}	Estimated during model calibration for 2013-2018 (see Section 2.5)
u^{o}_{3}	Harvest rate of age-3 adults in ocean (commercial fishery)	0.0545	Calculated from CTC (2021) (see Appendix D for details)
u^{o}_{4+}	Harvest rate of age-4 and older adults in ocean (commercial fishery)	0.109	Calculated from CTC (2021) (see Appendix D for details)
u^{r}_{3+}	Harvest rate of adults when going from ocean to WFF (terminal fishery)	0.052	Calculated from CTC (2021) (see Appendix D for details)
$s_{\{3,4,5,6\}}$	Observed proportion of returning adults to North Santiam spawning at age-3 to age-6	0.050; 0.638; 0.296; 0.016	Model used average from ODFW reports (Sharpe et al. 2013, 2014, 2015, 2016, 2017).
Fish Benefits Workbook Output			
$DPE_{t,\{F,S,Y\}}$	Dam Passage Efficiency for fry, subyearling, yearling	Bootstrap 1947-2019	FBW output (USACE 2022), see Section 1.4.
$DPS_{t,\{F,S,Y\}}$	Dam Passage Survival for fry, subyearling, yearling	Bootstrap 1947-2019	FBW output (USACE 2022), see Section 1.4.
River juvenile splits			
$p_{\{J\rightarrow F,J\rightarrow S,J\rightarrow Y\}}$	Proportion of emergent fry leaving natal stream as fry movers; subyearling stayers; and yearling stayers, respectively	0.89; 0.10, 0.01	Zabel et al. (2015), p5.20, ODFW RST data analysis (2011-2016, see Section 2.2.2)
$p_{\{F\rightarrow F,F\rightarrow S\}}$	Proportion of fry in reservoir moving to forebay as fry; and subyearlings	0.03; $1-p_{F\rightarrow F}$	Zabel et al. (2015), p5.20 (mean value)
$p_{\{S\rightarrow S,S\rightarrow Y\}}$	Proportion of subyearlings in reservoir moving to forebay as subyearlings; and yearlings	0.825; $1-p_{S\rightarrow S}$	Zabel et al. (2015), p5.20 (mean value)

$p_{Y \to Y}$	Proportion of yearlings in reservoir moving to forebay as yearlings	1	Assumed value
Ocean adult splits			
$p_{\{0+\to 3, a\to a+1\}}$	Proportion of adults in ocean returning to river to spawn (a≥3 and a≤5)	0.032; 0.595; 0.943; 1	Estimated during model calibration (see Section 2.5). Assumes all age-6 adults return to spawn.

Table 8-58. North Santiam Chinook salmon life cycle model equations. Unless otherwise indicated, equations are specified for t ≥ 1.
aAssumes return to sub-basin and PSM occurs prior to September in each year, so spawning occurs in the model year after return.
bAssumes that subyearling migrant types pass dams before first birthday but pass Willamette Falls as age-1+ smolts.
cWhere a=4, (1 – a) indicates fish in the age-0+ or age-1+ to age-3 stage groups, rather than age-3 only fish.

#	Description	Equation
Adults in freshwater		
1	Natural-origin adults returning to Big Cliff dam tailrace (Minto Collection Facility)a	$NOR_t^{BCLTR} = \begin{cases} NOR^{init} \; t = 0 \\ NOR_t^{WFF} * \left(1 - PSM_t^{below}\right) \; t \geq 1 \end{cases}$
2	Natural-origin adult outplants above Detroit (t ≥ 0)	$NO_t = \min(NOR_t^{BCLTR}, \kappa)$
3	Hatchery-origin adult outplants above Detroit (t ≥ 0)	$HO_t = \min(HO^{max}, \kappa - NOR_t^{BCL})$
4	Percentage of hatchery-origin spawners (t ≥ 0)	$pHO_t = HO_t/(NO_t + HO_t)$
5	Natural-origin spawnersa	$NS_t = NO_{t-1} * \left(1 - PSM_{t-1}^{above}\right)$
6	Hatchery-origin spawnersa	$HS_t = HO_{t-1} * \left(1 - PSM_{t-1}^{above}\right)$
7	Percentage of hatchery-origin spawners after PSM^{above}	$pHOS_t = HS_t/(NS_t + HS_t)$
Juveniles in freshwater		
8	Natural-origin eggs	$E_t^N = NS_t * \nu * \eta^N$
9	Hatchery-origin eggs	$E_t^H = HS_t * \nu * \eta^H * RRS$

10	Emergent fry in natal stream	$J_t = \dfrac{\alpha * (E_t^N + E_t^H)}{\left(1 + \dfrac{\alpha * (E_t^N + E_t^H)}{\beta}\right)}$
	Movement to Detroit reservoir	
11	Number of fry movers migrating to Detroit reservoir from natal stream in spring	$N_{t,F}^{res} = J_t * p_{J \to F} * \varphi_{t,F}^{nat}$
12	Number of subyearling stayers migrating to Detroit reservoir from natal stream in fall	$N_{t,S}^{res} = F_t * p_{J \to S} * \varphi_{t,S}^{nat}$
13	Number of yearling stayers migrating to Detroit reservoir from natal stream in spring (t ≥ 2)	$N_{t,Y}^{res} = J_{t-1} * p_{J \to Y} * \varphi_{t,Y}^{nat}$
	Movement to Detroit forebay	
14	Number of fry in Detroit reservoir moving to forebay in spring	$N_{t,F}^{fby} = N_{t,F}^{res} * p_{F \to F} * \varphi_{t,F \to F}^{DET}$
15	Number of subyearlings that spent summer in reservoir moving to Detroit forebay	$N_{t,S_{resS}}^{fby} = N_{t,F}^{res} * p_{F \to S} * \varphi_{t,F \to S}^{DET} + N_{t,F}^{fby} * \left(1 - DPE_{t,F}\right) * \varphi_{t,F \to S}^{DET}$
16	Number of subyearlings that spent summer in natal stream moving to Detroit forebay	$N_{t,S_{str}}^{fby} = N_{t,S}^{res} * p_{S \to S} * \varphi_{t,S \to S}^{DET}$
17	Number of yearlings that spent summer and winter in reservoir moving to Detroit forebay (t ≥ 2)	$N_{t,F_{resSW}}^{fby} = N_{t-1,S}^{res} * p_{S \to Y} * \varphi_{t-1,S \to Y}^{DET} + N_{t-1,S_{resS}}^{fby} * \left(1 - DPE_{t-1,S}\right) * \varphi_{t-1,S \to Y}^{DET}$
18	Number of yearlings that spent only winter in reservoir moving to Detroit forebay (t ≥ 2)	$N_{t,Y_{resW}}^{fby} = N_{t-1,S_{str}}^{fby} * \left(1 - DPE_{t-1,S}\right) * \varphi_{t-1,S \to Y}^{DET}$
19	Number of yearlings that reared in natal stream moving to Detroit forebay (t ≥ 2)	$N_{t,Y_{str}}^{fby} = N_{t,Y}^{res} * p_{Y \to Y} * \varphi_{t,Y \to Y}^{DET}$
	Movement to Big Cliff tailrace	
20	Number of different migrant types passing Detroit dam to	$N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{BCLTR} = N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{fby} * DPE_{t,\{F,S,S,Y,Y,Y\}} * DPS_{t,\{F,S,S,Y,Y,Y\}}$

	Big Cliff tailrace. For yearlings t ≥ 2	
21	Number of yearlings that did not pass Detroit dam and assumed dead in summer (t ≥ 2)	$N_{t,Y}^{dead} = \left(N_{t,Y_{resSW}}^{fby} + N_{t,Y_{resW}}^{fby} + N_{t,Y_{str}}^{fby}\right) * \left(1 - DPE_{t,Y}\right)$
Smolts at Willamette Falls		
22	Number of different migrant types reaching SUJ. For yearlings t ≥ 2	$N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{SUJ} = N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{BCLTR} * \varphi_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{rss}$
Age structure of adults remaining in ocean		
23	Number of adults (age-3) in the ocean of fry migrant type (t ≥ 3)	$NA_{t,0+\to 3}^{F} = N_{t-2,F}^{SUJ} * (1 - p_{0+\to 3}) * \varphi_{0+\to 3}^{o} * e^{\varepsilon_{t-2}} * (1 - u_3^o)$
24	Number of adults (age-3) in the ocean of subyearling migrant type that spent summer in reservoir (t ≥ 3)b	$NA_{t,1+\to 3}^{S_{resS}} = N_{t-2,S_{resS}}^{SUJ} * (1 - p_{0+\to 3}) * \varphi_{1+\to 3}^{o} * e^{\varepsilon_{t-1}} * (1 - u_3^o)$
25	Number of adults (age-3) in the ocean of subyearling migrant type that spent summer in natal stream (t ≥ 3)b	$NA_{t,1+\to 3}^{S_{str}} = N_{t-2,S_{str}}^{SUJ} * (1 - p_{0+\to 3}) * \varphi_{1+\to 3}^{o} * e^{\varepsilon_{t-1}} * (1 - u_3^o)$
26	Number of adults (age-3) in the ocean of yearling migrant type that spent summer and winter in reservoir (t ≥ 3)	$NA_{t,1+\to 3}^{Y_{resSW}} = N_{t-1,Y_{resSW}}^{SUJ} * (1 - p_{0+\to 3}) * \varphi_{1+\to 3}^{o} * e^{\varepsilon_{t-1}} * (1 - u_3^o)$
27	Number of adults (age-3) in the ocean of yearling migrant type that spent only winter in reservoir (t ≥ 3)	$NA_{t,1+\to 3}^{Y_{resW}} = N_{t-1,Y_{resW}}^{SUJ} * (1 - p_{0+\to 3}) * \varphi_{1+\to 3}^{o} * e^{\varepsilon_{t-1}} * (1 - u_3^o)$
28	Number of adults (age-3) in the ocean of yearling migrant type that reared in natal stream (t ≥ 3)	$NA_{t,1+\to 3}^{Y_{str}} = N_{t-1,Y_{str}}^{SUJ} * (1 - p_{0+\to 3}) * \varphi_{1+\to 3}^{o} * e^{\varepsilon_{t-1}} * (1 - u_3^o)$
29	Number of adults (age ≥ 4 and age ≤ 6) in the ocean of each migrant type (t ≥ 4)c	$NA_{t,a}^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}} = N_{t-1,a-1}^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}} * (1 - p_{a-1\to a}) * \varphi_{a-1\to a}^{o} * (1 - u_{4+}^{o})$
Age structure of adults returning to river		

30	Number of adults (age-3) returning from the ocean of fry migrant type (t ≥ 3)	$NOR_{t,3}^{F} = N_{t-2,F}^{SUJ} * p_{0+\to 3} * \varphi_{0+\to 3}^{o} * (1 - u_3^r)$
31	Number of adults (age-3) returning from the ocean of subyearling migrant type that spent summer in reservoir (t ≥ 3)	$NOR_{t,3}^{S_{resS}} = N_{t-2,S_{resS}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
32	Number of adults (age-3) returning from the ocean of subyearling migrant type that spent summer in natal stream (t ≥ 3)	$NOR_{t,3}^{S_{str}} = N_{t-2,S_{str}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
33	Number of adults (age-3) returning from the ocean of yearling migrant type that spent summer and winter in reservoir (t ≥ 3)	$NOR_{t,3}^{Y_{resSW}} = N_{t-1,Y_{resSW}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
34	Number of adults (age-3) returning from the ocean of yearling migrant type that spent only winter in reservoir (t ≥ 3)	$NOR_{t,3}^{Y_{resW}} = N_{t-1,Y_{resW}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
35	Number of adults (age-3) returning from the ocean of yearling migrant type that reared in natal stream (t ≥ 3)	$NOR_{t,3}^{Y_{str}} = N_{t-1,Y_{str}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
36	Number of adults (age ≥ 4 and age ≤ 6) returning from the ocean of each migrant type (t ≥ 4)c	$NOR_{t,a}^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}} = N_{t-1,a-1}^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}} * p_{a-1\to a} * \varphi_{a-1\to a}^{o} * (1 - u_{4+}^r)$
Age structure of adults returned to Willamette Falls		
37	Number of adults from age a≥3 and a≤ 6 returned to WFF	$NOR_{t,a}^{WFF} = \begin{cases} s_a * \left(NOR^{init}/\left(1 - PSM_t^{below}\right)\right) & t < a \\ NOR_{t,a}^{F} + NOR_{t,a}^{S_{resS}} + NOR_{t,a}^{S_{str}} + NOR_{t,a}^{Y_{resSW}} + NOR_{t,a}^{Y_{resW}} + NOR_{t,a}^{Y_{str}} & t \geq a \end{cases}$
38	Total number of natural-origin adults returning to WFF	$NOR_{t}^{WFF} = \sum_{a=3}^{6} NOR_{t,a}^{WFF}$

South Santiam

Table 8-59. Parameter and other input variable values for the South Santiam Chinook salmon life cycle model.
aValues for parameters without a time subscript drawn from a distribution vary between simulations.
bα and β parameters of a beta distribution are derived from a mean and CI.

Symbol	Description	Valuea	Reference
Model parameters			
NOR^{init}	Initial number of natural-origin returns to Foster Adult Collection Facility	430	Mean number of NOR to Foster, 2012-2021 (ODFW count data)
HO^{max}	Maximum hatchery-origin outplants	0 (above Foster) 800 (above Green Peter)	USACE (Rachel Laird & Rich Piaskowski, pers. comm.), NEDC v. USACE (2021)
κ	Logistical cap on total outplant numbers (natural-origin + hatchery-origin)	3,099	ODFW & USACE (2019), Columbia River Partnership Task Force (2020)
p_{FOS}	Outplant ratio Foster:Green Peter	1.0 (NAA, Alt4); 0.52 (otherwise)	Assumed from relative habitat capacity in above dam reaches (Zabel et al. 2015)
PSM_t^{below}	Pre-spawn mortality (PSM) en route to dam	Beta(130.543, 654.937)b; mean=0.165 (95% CI 0.141-0.193)	Based on a 2011-2014 radio-telemetry study (Keefer et al. 2017)
PSM_t^{above}	Pre-spawn mortality (PSM) on spawning grounds above dam	$f(7DADM_t, pHO_t)$	Adjusted function from Bowerman et al. (2018), see Section 2.4.1.
$7DADM_t$	7-day average of daily maximum temperature (oC) above Foster reservoir	Bootstrap 1947-2019	Data from USGS gage 14185000. Years with missing data were back-filled using the water-year type mean value. See Section 2.4.1.
η^N	Fecundity (natural origin adults)	4,500	Zabel et al. (2015), p7.7
η^H	Fecundity (hatchery origin adults)	4,000	Zabel et al. (2015), p7.7

ν	Sex ratio (female spawners to total spawners) for natural- and hatchery-origin adults	0.44	Mean from observed returns to and outplants above Foster dam (ODFW count data, 2009-2021)
RRS	Relative reproductive success of HO to NO adults	Triang(0.39, 0.53, 0.84)	Based on analysis of Christie et al. (2014), see Section 2.4.3.
α^{GPR}	Beverton-Holt NO egg-fry survival (above Green Peter reservoir)	0.525	Zabel et al. (2015), p7.8
α^{FOS}	Beverton-Holt NO egg-fry survival (above Foster reservoir)	0.425	Zabel et al. (2015), p7.8
β^{GPR}	Beverton-Holt egg capacity (above Green Peter reservoir)	Norm(2E+06, 2.5E+05) + Norm(4.5E+06, 2.5E+05	Zabel et al. (2015), p7.7 (reach F+G)
β^{FOS}	Beverton-Holt egg capacity (above Foster reservoir)	Norm(7E+06, 2.5E+05)	Zabel et al. (2015), p7.7 (reach E)
$\varphi_{t,\{F,S,Y\}}^{GPR_{nat}}$	Fry-migrant survival in natal stream above Green Peter (fry mover; subyearling stayer; yearling stayer)	0.75; 0.4; 0.3	Zabel et al. (2015), p7.9
$\varphi_{t,\{F,S,Y\}}^{FOS_{nat}}$	Fry-migrant survival in natal stream above Foster (fry mover; subyearling stayer; yearling stayer)	0.6; 0.3; 0.25	Zabel et al. (2015), p7.9
$\varphi_{t,F\to F}^{GPR}$	Green Peter reservoir survival (fry to fry)	1	Zabel et al. (2015), p7.10
$\varphi_{t,F\to S}^{GPR}$	Green Peter reservoir survival (fry to subyearling)	Triang(0.15, 0.2, 0.35)	Zabel et al. (2015), p7.10
$\varphi_{t,S\to S}^{GPR}$	Green Peter reservoir survival (subyearling to subyearling)	1	Zabel et al. (2015), p7.10
$\varphi_{t,S\to Y}^{GPR}$	Green Peter reservoir survival (subyearling to yearling)	Triang(0.6, 0.75, 0.9)	Zabel et al. (2015), p7.10
$\varphi_{t,Y\to Y}^{GPR}$	Green Peter reservoir survival (yearling to yearling)	1	Zabel et al. (2015), p7.10

$\varphi^{FOS}_{t,F\rightarrow F}$	Foster reservoir survival (fry to fry)	1	Zabel et al. (2015), p7.10
$\varphi^{FOS}_{t,F\rightarrow S}$	Foster reservoir survival (fry to subyearling)	Triang(0.15, 0.2, 0.35)	Zabel et al. (2015), p7.10
$\varphi^{FOS}_{t,S\rightarrow S}$	Foster reservoir survival (subyearling to subyearling)	1	Zabel et al. (2015), p7.10
$\varphi^{FOS}_{t,S\rightarrow Y}$	Foster reservoir survival (subyearling to yearling)	Unif(0.4, 0.9)	Zabel et al. (2015), p7.10
$\varphi^{FOS}_{t,Y\rightarrow Y}$	Foster reservoir survival (yearling to yearling)	1	Zabel et al. (2015), p7.10
$\varphi^{rss}_{t,F}$	River smolt survival for fry (FOSTR-SUJ)	Median = 0.159 (CV = 0.333)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p7.15 (see Section 2.2.6 and Appendix C).
$\varphi^{rss}_{t,S_{resS}}$	River smolt survival for subyearlings that spent summer in reservoir (FOSTR-SUJ)	Median = 0.441 (CV = 0.191)	CJS model posterior output adjusted for tagging/hatchery effects (see Appendix C)
$\varphi^{rss}_{t,S_{str}}$	River smolt survival for subyearlings that spent summer in natal stream (FOSTR-SUJ)	Median = 0.441 (CV = 0.191)	CJS model posterior output adjusted for tagging/hatchery effects (see Appendix C)
$\varphi^{rss}_{t,Y_{resSW}}$	River smolt survival for yearlings that spent summer and winter in reservoir (FOSTR-SUJ)	Median = 0.589 (CV = 0.113)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p7.15 (see Section 2.2.6 and Appendix C).

$\varphi_{t,Y_{resW}}^{rss}$	River smolt survival for yearlings that spent only winter in reservoir (FOSTR-SUJ)	Median = 0.589 (CV = 0.113)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p7.15
$\varphi_{t,Y_{str}}^{rss}$	River smolt survival for yearlings that reared in natal stream (FOSTR-SUJ)	Median = 0.589 (CV = 0.113)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p7.15
$\varphi_{0+\to 3}^{o}$	Survival from natural mortality in the ocean (0+ to age-3 adult)	0.011	Estimated during model calibration, using CJS model posterior output as prior (see Appendix C and Appendix D). 0+ pass SUJ before 1st birthday.
$\varphi_{1+\to 3}^{o}$	Survival from natural mortality in the ocean (1+ to age-3 adult)	0.105	Estimated during model calibration, using CJS model posterior output as prior (see Appendix C and Appendix D)
$\varphi_{a\to a+1}^{o}$	Survival from natural mortality in the ocean (age a≥3 and a≤5)	0.397; 0.500; 0.630	Assumed by CTC (2021), see Appendix D for details.
ε_t	Annual deviates in early ocean survival	{-0.263, 0.206, 0.564, 0.998, 2.134, 1.176, 1.357, -0.139, -0.075, -0.454, -0.442, -0.907, -0.541, 0.811, 0.182}	Estimated during model calibration for 2004-2018 (see Section 2.5)
u_3^o	Harvest rate of age-3 adults in ocean (commercial fishery)	0.0545	Calculated from CTC (2021) (see Appendix D for details)
u_{4+}^o	Harvest rate of age-4 and older adults in ocean (commercial fishery)	0.109	Calculated from CTC (2021) (see Appendix D for details)

u_{3+}^{r}	Harvest rate of age-3 and older adults when going from ocean to WFF (terminal fishery)	0.052	Calculated from CTC (2021) (see Appendix D for details)
$s_{\{3,4,5,6\}}$	Observed proportion of returning adults to South Santiam spawning at age-3 to age-6	0.089; 0.715; 0.189; 0.008	Model used average from ODFW reports (Sharpe et al. 2013, 2014, 2015, 2016, 2017).
Fish Benefits Workbook Output			
$DPE_{t,\{F,S,Y\}}^{GRP}$	Green Peter Dam Passage Efficiency for fry, subyearling, and yearling	Bootstrap 1947-2019	FBW output (USACE 2022), see Section 1.4.
$DPS_{t,\{F,S,Y\}}^{GRP}$	Green Peter Dam Passage Survival for fry, subyearling, and yearling	Bootstrap 1947-2019	FBW output (USACE 2022), see Section 1.4.
$DPE_{t,\{F,S,Y\}}^{FOS}$	Foster Dam Passage Efficiency for fry, subyearling, yearling	Bootstrap 1947-2019	FBW output (USACE 2022), see Section 1.4.
$DPS_{t,\{F,S,Y\}}^{FOS}$	Foster Dam Passage Survival for fry, subyearling, yearling	Bootstrap 1947-2019	FBW output (USACE 2022), see Section 1.4.
River juvenile splits			
$p_{\{J\to F,J\to S,J\to Y\}}^{GPR}$	Proportion of emergent fry leaving natal stream above Green Peter as fry movers; subyearling stayers; and yearling stayers, respectively	0.85; 0.10, 0.05	Zabel et al. (2015), p7.24
$p_{\{J\to F,J\to S,J\to Y\}}^{FOS}$	Proportion of emergent fry leaving natal stream above Foster as fry movers; subyearling stayers; and yearling stayers, respectively	0.85; 0.10; 0.05	Zabel et al. (2015), p7.24, ODFW RST data analysis (2011-2016, see Section 2.2.2)
$p_{\{F\to F,F\to S\}}^{GPR}$	Proportion of fry in Green Peter reservoir moving to forebay as fry; and subyearlings	0.03; $1-p_{F\to F}^{GPR}$	Zabel et al. (2015), p7.25 (mean value)
$p_{\{S\to S,S\to Y\}}^{GPR}$	Proportion of subyearlings in Green Peter reservoir moving to forebay as subyearlings; and yearlings	0.88; $1-p_{S\to S}^{GPR}$	Zabel et al. (2015), p7.25 (mean value)

$p_{Y\to Y}^{GPR}$	Proportion of yearlings in Green Peter reservoir moving to forebay as yearlings	1	Assumed value
$p_{\{F\to F, F\to S\}}^{FOS}$	Proportion of fry in Foster reservoir moving to forebay as fry; and subyearlings	0.75; $1-p_{F\to F}^{FOS}$	Zabel et al. (2015), p7.25 (mean value)
$p_{S\to S}^{FOS}$	Proportion of subyearlings in Foster reservoir moving to forebay as subyearlings; and yearlings	0.875; $1-p_{S\to S}^{FOS}$	Zabel et al. (2015), p7.25 (mean value)
$p_{Y\to Y}^{FOS}$	Proportion of yearlings in Foster reservoir moving to forebay as yearlings	1	Assumed value
Ocean adult splits			
$p_{\{0+\to 3, a\to a+1\}}$	Proportion of adults in ocean returning to river to spawn (a≥3 and a≤5)	0.034; 0.807; 0.950; 1	Estimated during model calibration (see Section 2.5). Assumes all age-6 adults return to spawn.

Table 8-60. South Santiam Chinook salmon life cycle model equations. Unless otherwise indicated, equations are specified for t ≥ 1.
aAssumes return to sub-basin and PSM occurs prior to September in each year, so spawning occurs in the model year after return.
bAssumes that all fish coming from Green Peter head direct to Foster forebay and survive to attempt dam passage through Foster.
cAssumes that subyearling migrant types pass dams before first birthday but pass Willamette Falls as age-1+ smolts.
dWhere a=4, (1 – a) indicates fish in the age-0+ or age-1+ to age-3 stage groups, rather than age-3 only fish.

#	Description	Equation
		Adults in freshwater
1	Natural-origin adults returning to Foster dam tailrace (Foster Adult Collection Facility)a	$NOR_t^{FOSTR} = \begin{cases} NOR^{init} \; t = 0 \\ NOR_t^{WFF} * \left(1 - PSM_t^{below}\right) \; t \geq 1 \end{cases}$
2	Natural-origin adult outplants above Foster (t ≥ 0)	$NO_t^{FOS} = \min\begin{pmatrix} p_{Fos} * NOR_t^{FOSTR}, \\ p_{FOS} * \kappa \end{pmatrix}$
3	Natural-origin adult outplants above Green Peter (t ≥ 0)	$NO_t^{GPR} = \min\begin{pmatrix} (1 - p_{FOS}) * NOR_t^{FOSTR}, \\ (1 - p_{FOS}) * \kappa \end{pmatrix}$
4	Hatchery-origin adult outplants above Foster (t ≥ 0)	$HO_t^{FOS} = \min\begin{pmatrix} p_{FOS} * HO^{max}, \\ p_{FOS} * \kappa - NOR_t^{FOSTR} \end{pmatrix}$
5	Hatchery-origin adult outplants above Green Peter (t ≥ 0)	$HO_t^{GPR} = \min\begin{pmatrix} (1 - p_{FOS}) * HO^{max}, \\ (1 - p_{FOS}) * \kappa - NOR_t^{FOSTR} \end{pmatrix}$
6	Percentage of hatchery-origin outplants (t ≥ 0)	$pHO_t = HO_t^{FOS} / (NO_t^{FOS} + HO_t^{FOS})$
7	Natural-origin spawners above Fostera	$NS_t^{FOS} = NO_{t-1}^{FOS} * \left(1 - PSM_{t-1}^{above}\right)$
8	Natural-origin spawners above Green Petera	$NS_t^{GPR} = NO_{t-1}^{GPR} * \left(1 - PSM_{t-1}^{above}\right)$
9	Hatchery-origin spawners above Fostera	$HS_t^{FOS} = HO_{t-1}^{FOS} * \left(1 - PSM_{t-1}^{above}\right)$

10	Hatchery-origin spawners above Green Petera	$HS_t^{GPR} = HO_{t-1}^{GPR} * \left(1 - PSM_{t-1}^{above}\right)$
11	Percentage of hatchery-origin spawners above Foster after PSM^{above}	$pHOS_t^{FOS} = HS_t^{FOS} / (NS_t^{FOS} + HS_t^{FOS})$
12	Percentage of hatchery-origin spawners above Green Peter after PSM^{above}	$pHOS_t^{GPR} = HS_t^{GPR} / (NS_t^{GPR} + HS_t^{GPR})$
	Juveniles in freshwater	
13	Natural-origin eggs above Foster	$E_t^{N,FOS} = NS_t^{FOS} * \nu * \eta^N$
14	Natural-origin eggs above Green Peter	$E_t^{N,GPR} = NS_t^{GPR} * \nu * \eta^N$
15	Hatchery-origin eggs above Foster	$E_t^{H,FOS} = HS_t^{FOS} * \nu * \eta^H * RRS$
16	Hatchery-origin eggs above Green Peter	$E_t^{H,GPR} = HS_t^{GPR} * \nu * \eta^H * RRS$
17	Emergent fry in natal stream above Foster	$J_t^{FOS} = \dfrac{\alpha^{FOS} * \left(E_t^{N,FOS} + E_t^{H,FOS}\right)}{\left(1 + \dfrac{\alpha^{FOS} * \left(E_t^{N,FOS} + E_t^{H,FOS}\right)}{\beta^{FOS}}\right)}$
18	Emergent fry in natal stream above Green Peter	$J_t^{GPR} = \dfrac{\alpha^{GPR} * \left(E_t^{N,GPR} + E_t^{H,GPR}\right)}{\left(1 + \dfrac{\alpha^{GPR} * \left(E_t^{N,GPR} + E_t^{H,GPR}\right)}{\beta^{GPR}}\right)}$
	Movement to Green Peter reservoir	
19	Number of fry movers migrating to Green Peter reservoir from natal stream in spring	$N_{t,F}^{GPR_{res}} = J_t^{GPR} * p_{J\to F}^{GPR} * \varphi_{t,F}^{GPR_{nat}}$
20	Number of subyearling stayers migrating to Green Peter reservoir from natal stream in fall	$N_{t,S}^{GPR_{res}} = J_t^{GPR} * p_{J\to S}^{GPR} * \varphi_{t,S}^{GPR_{nat}}$
21	Number of yearling stayers migrating to Green Peter reservoir from natal stream in spring (t ≥ 2)	$N_{t,Y}^{GPR_{res}} = J_{t-1}^{GPR} * p_{J\to Y}^{GPR} * \varphi_{t,Y}^{GPR_{nat}}$
	Movement to Foster reservoir	

22	Number of fry movers migrating to Foster reservoir from natal stream in spring	$N_{t,F}^{FOS_{res}} = J_t^{FOS} * p_{J\rightarrow F}^{FOS} * \varphi_{t,F}^{FOS_{nat}}$
23	Number of subyearling stayers migrating to Foster reservoir from natal stream in fall	$N_{t,S}^{FOS_{res}} = J_t^{FOS} * p_{J\rightarrow S}^{FOS} * \varphi_{t,S}^{FOS_{nat}}$
24	Number of yearling stayers migrating to Foster reservoir from natal stream in spring (t ≥ 2)	$N_{t,Y}^{FOS_{res}} = J_{t-1}^{FOS} * p_{J\rightarrow Y}^{FOS} * \varphi_{t,Y}^{FOS_{nat}}$
Movement to Green Peter forebay		
25	Number of fry in Green Peter reservoir moving to forebay	$N_{t,F}^{GPR_{fby}} = N_{t,F}^{GPR_{res}} * p_{F\rightarrow F}^{GPR} * \varphi_{t,F\rightarrow F}^{GPR}$
26	Number of subyearlings that spent summer in reservoir moving to Green Peter forebay	$N_{t,S_{resS}}^{GPR_{fby}} = N_{t,F}^{GPR_{res}} * p_{F\rightarrow S}^{GPR} * \varphi_{t,F\rightarrow S}^{GPR} + N_{t,F}^{GPR_{fby}} * \left(1 - DPE_{t,F}^{GPR}\right) * \varphi_{t,F\rightarrow S}^{GPR}$
27	Number of subyearlings that spent summer in natal stream moving to Green Peter forebay	$N_{t,S_{str}}^{GPR_{fby}} = N_{t,S}^{GPR_{res}} * p_{S\rightarrow S}^{GPR} * \varphi_{t,S\rightarrow S}^{GPR}$
28	Number of yearlings that spent summer and winter in reservoir moving to Green Peter forebay (t ≥ 2)	$N_{t,Y_{resSW}}^{GPR_{fby}} = N_{t-1,S}^{GPR_{res}} * p_{S\rightarrow Y}^{GPR} * \varphi_{t-1,S\rightarrow Y}^{GPR} + N_{t-1,S_{resS}}^{GPR_{fby}} * \left(1 - DPE_{t-1,S}^{GPR}\right) * \varphi_{t-1,S\rightarrow Y}^{GPR}$
29	Number of yearlings that spent only winter in reservoir moving to Green Peter forebay (t ≥ 2)	$N_{t,Y_{resW}}^{GPR_{fby}} = N_{t-1,S_{str}}^{GPR_{fby}} * \left(1 - DPE_{t-1,S}^{GPR}\right) * \varphi_{t-1,S\rightarrow Y}^{GPR}$
30	Number of yearlings that reared in natal stream moving to Green Peter forebay (t ≥ 2)	$N_{t,Y_{str}}^{GPR_{fby}} = N_{t,Y}^{GPR_{res}} * p_{Y\rightarrow Y}^{GPR} * \varphi_{t,Y\rightarrow Y}^{GPR}$
Movement to Green Peter tailrace		

31	Number of different migrant types passing Green Peter dam to Green Peter tailrace. For yearlings t ≥ 2	$N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{GPRTR} = N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{GPR_{fby}} * DPE_{t,\{F,S,S,Y,Y,Y\}}^{GPR} * DPS_{t,\{F,S,S,Y,Y,Y\}}^{GPR}$
32	Number of yearlings that did not pass Green Peter dam and assumed dead in summer (t ≥ 2)	$N_{t,Y}^{GPR_{dead}} = \left(N_{t,Y_{resSW}}^{GPR_{fby}} + N_{t,Y_{resW}}^{GPR_{fby}} + N_{t,Y_{str}}^{GPR_{fby}}\right) * \left(1 - DPE_{t,Y}^{GPR}\right)$
Movement to Foster forebay		
33	Number of fry in Foster reservoir moving to forebayb	$N_{t,F}^{FOS_{fby}} = N_{t,F}^{FOS_{res}} * p_{F\to F}^{FOS} * \varphi_{t,F\to F}^{FOS} + N_{t,F}^{GPRTR}$
34	Number of subyearlings that spent summer in reservoir moving to Foster forebayb	$N_{t,S_{resS}}^{FOS_{fby}} = N_{t,F}^{FOS_{res}} * p_{F\to S}^{FOS} * \varphi_{t,F\to S}^{FOS} + N_{t,F}^{FOS_{fby}} * \left(1 - DPE_{t,F}^{FOS}\right) * \varphi_{t,F\to S}^{FOS} + N_{t,S_{resS}}^{GPRTR}$
35	Number of subyearlings that spent summer in natal stream moving to Foster forebayb	$N_{t,S_{str}}^{FOS_{fby}} = N_{t,S}^{FOS_{res}} * p_{S\to S}^{FOS} * \varphi_{t,S\to S}^{FOS} + N_{t,S_{str}}^{GPRTR}$
36	Number of yearlings that spent summer and winter in reservoir moving to Foster forebay (t ≥ 2)b	$N_{t,Y_{resSW}}^{FOS_{fby}} = N_{t-1,S}^{FOS_{res}} * p_{S\to Y}^{FOS} * \varphi_{t-1,S\to Y}^{FOS} + N_{t-1,S_{resS}}^{FOS_{fby}} * \left(1 - DPE_{t-1,S}^{FOS}\right) * \varphi_{t-1,S\to Y}^{FOS} + N_{t,Y_{resSW}}^{GPRTR}$
37	Number of yearlings that spent only winter in reservoir moving to Foster forebay (t ≥ 2)b	$N_{t,Y_{resW}}^{FOS_{fby}} = N_{t-1,S_{str}}^{FOS_{fby}} * \left(1 - DPE_{t-1,S}^{FOS}\right) * \varphi_{t-1,S\to Y}^{FOS} + N_{t,Y_{resW}}^{GPRTR}$
38	Number of yearlings that reared in natal stream moving to Foster forebay (t ≥ 2)b	$N_{t,Y_{str}}^{FOS_{fby}} = N_{t,Y}^{FOS_{res}} * p_{Y\to Y}^{FOS} * \varphi_{t,Y\to Y}^{FOS} + N_{t,Y_{str}}^{GPRTR}$
Movement to Foster tailrace		
39	Number of different migrant types passing Foster dam to Foster tailrace. For yearlings t ≥ 2	$N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{FOSTR} = N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{FOS_{fby}} * DPE_{t,\{F,S,S,Y,Y,Y\}}^{FOS} * DPS_{t,\{F,S,S,Y,Y,Y\}}^{FOS}$

40	Number of yearlings that did not pass Foster dam and assumed dead in summer (t ≥ 2)	$N_{t,Y}^{FOS_{dead}} = \left(N_{t,Y_{resSW}}^{FOS_{fby}} + N_{t,Y_{resW}}^{FOS_{fby}} + N_{t,Y_{str}}^{FOS_{fby}}\right) * \left(1 - DPE_{t,Y}^{FOS}\right)$
Smolts at Willamette Falls		
41	Number of different migrant types reaching SUJ. For yearlings t ≥ 2	$N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{SUJ} = N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{FOSTR} * \varphi_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{rss}$
Age structure of adults remaining in ocean		
42	Number of adults (age-3) in the ocean of fry migrant type (t ≥ 3)	$NA_{t,0+\to 3}^{F} = N_{t-2,F}^{SUJ} * (1 - p_{0+\to 3}) * \varphi_{0+\to 3}^{o} * e^{\varepsilon_{t-2}} * (1 - u_3^o)$
43	Number of adults (age-3) in the ocean of subyearling migrant type that spent summer in reservoir (t ≥ 3)c	$NA_{t,1+\to 3}^{S_{resS}} = N_{t-2,S_{resS}}^{SUJ} * (1 - p_{0+\to 3}) * \varphi_{1+\to 3}^{o} * e^{\varepsilon_{t-1}} * (1 - u_3^o)$
44	Number of adults (age-3) in the ocean of subyearling migrant type that spent summer in natal stream (t ≥ 3)c	$NA_{t,1+\to 3}^{S_{str}} = N_{t-2,S_{str}}^{SUJ} * (1 - p_{0+\to 3}) * \varphi_{1+\to 3}^{o} * e^{\varepsilon_{t-1}} * (1 - u_3^o)$
45	Number of adults (age-3) in the ocean of yearling migrant type that spent summer and winter in reservoir (t ≥ 3)	$NA_{t,1+\to 3}^{Y_{resSW}} = N_{t-1,Y_{resSW}}^{SUJ} * (1 - p_{0+\to 3}) * \varphi_{1+\to 3}^{o} * e^{\varepsilon_{t-1}} * (1 - u_3^o)$
46	Number of adults (age-3) in the ocean of yearling migrant type that spent only winter in reservoir (t ≥ 3)	$NA_{t,1+\to 3}^{Y_{resW}} = N_{t-1,Y_{resW}}^{SUJ} * (1 - p_{0+\to 3}) * \varphi_{1+\to 3}^{o} * e^{\varepsilon_{t-1}} * (1 - u_3^o)$
47	Number of adults (age-3) in the ocean of yearling migrant type that reared in natal stream (t ≥ 3)	$NA_{t,1+\to 3}^{Y_{str}} = N_{t-1,Y_{str}}^{SUJ} * (1 - p_{0+\to 3}) * \varphi_{1+\to 3}^{o} * e^{\varepsilon_{t-1}} * (1 - u_3^o)$

48	Number of adults (age ≥ 4 and age ≤ 6) in the ocean of each migrant type (t ≥ 4)d	$NA_{t,a}^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}} = N_{t-1,a-1}^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}} * (1 - p_{a-1\to a}) * \varphi_{a-1\to a}^{o} * (1 - u_{4+}^{o})$
Age structure of adults returning to river		
49	Number of adults (age-3) returning from the ocean of fry migrant type (t ≥ 3)	$NOR_{t,3}^{F} = N_{t-2,F}^{SUJ} * p_{0+\to 3} * \varphi_{0+\to 3}^{o} * (1 - u_3^r)$
50	Number of adults (age-3) returning from the ocean of subyearling migrant type that spent summer in reservoir (t ≥ 3)	$NOR_{t,3}^{S_{resS}} = N_{t-2,S_{resS}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
51	Number of adults (age-3) returning from the ocean of subyearling migrant type that spent summer in natal stream (t ≥ 3)	$NOR_{t,3}^{S_{str}} = N_{t-2,S_{str}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
52	Number of adults (age-3) returning from the ocean of yearling migrant type that spent summer and winter in reservoir (t ≥ 3)	$NOR_{t,3}^{Y_{resSW}} = N_{t-1,Y_{resSW}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
53	Number of adults (age-3) returning from the ocean of yearling migrant type that spent only winter in reservoir (t ≥ 3)	$NOR_{t,3}^{Y_{resW}} = N_{t-1,Y_{resW}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
54	Number of adults (age-3) returning from the ocean of yearling migrant type that reared in natal stream (t ≥ 3)	$NOR_{t,3}^{Y_{str}} = N_{t-1,Y_{str}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
55	Number of adults (age ≥ 4 and age ≤ 6) returning from the	$NOR_{t,a}^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}} = N_{t-1,a-1}^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}} * p_{a-1\to a} * \varphi_{a-1\to a}^{o} * (1 - u_{4+}^{r})$

	ocean of each migrant type (t ≥ 4)d	
Age structure of adults returned to Willamette Falls		
56	Number of adults from age a≥3 and a≤ 6 returned to WFF	$NOR_{t,a}^{WFF} = \begin{cases} s_a * \left(NOR^{init} / \left(1 - PSM_t^{below}\right)\right) & t < a \\ NOR_{t,a}^{F} + NOR_{t,a}^{S_{resS}} + NOR_{t,a}^{S_{str}} + & t \geq a \\ NOR_{t,a}^{Y_{resSW}} + NOR_{t,a}^{Y_{resW}} + NOR_{t,a}^{Y_{str}} & \end{cases}$
57	Total number of natural-origin adults returning to WFF	$NOR_t^{WFF} = \sum_{a=3}^{6} NOR_{t,a}^{WFF}$

McKenzie

Table 8-61. Parameter and other input variable values for the McKenzie Chinook salmon life cycle model.
aValues for parameters without a time subscript drawn from a distribution vary between simulations.
bα and β parameters of a beta distribution are derived from a mean and CI.

Symbol	Description	Valuea	Reference
Model parameters			
NOR^{init}	Initial number of natural-origin returns to Cougar Dam Adult Fish Collection Facility	130	Mean number of NOR to Cougar, 2016-2021 (ODFW count data).
HO^{max}	Maximum hatchery-origin outplants	600	HGMP (section 15.2) states that up to 600 HOR spring Chinook (400 female, 200 male) may be outplanted above Cougar Dam to evaluate the potential for above-dam restoration of self-sustaining populations (ODFW & USACE 2018). N.B. Mean total number of outplants 2016-2021 was 432 (ODFW count data).
κ	Logistical cap on total outplant numbers (natural-origin + hatchery-origin)	4,861	Maximum number of outplanted adults from Cougar Trap and McKenzie Hatchery from years 1993-2020 (ODFW & USACE 2018, Appendix D).
PSM_t^{below}	Pre-spawn mortality (PSM) en route to dam	Beta(130.543, 654.937)b; mean=0.165 (95% CI 0.141-0.193)	Based on a 2011-2014 radio-telemetry study (Keefer et al. 2017)
PSM_t^{above}	Pre-spawn mortality (PSM) on spawning grounds above dam	$f(7DADM_t, pHO_t)$	Adjusted function from Bowerman et al. (2018), see Section 2.4.1.

$7DADM_t$	7-day average of daily maximum temperature (oC) above Cougar reservoir	Bootstrap 1947-2019	Data from USGS gage 14159200. Years with missing data were back-filled using the water-year type mean value. See Section 2.4.1.
η^N	Fecundity (natural origin adults)	4,500	Zabel et al. (2015), p5.7
η^H	Fecundity (hatchery origin adults)	4,000	Zabel et al. (2015), p5.7
ν	Sex ratio (female spawners to total spawners) for natural- and hatchery-origin adults	0.5	
RRS	Relative reproductive success of HO to NO adults	Triang(0.39, 0.53, 0.84)	Based on analysis of Christie et al. (2014), see Section 2.4.3.
α	Beverton-Holt egg-fry survival	0.575	Zabel et al. (2015), p4.7 (reach C)
β	Beverton-Holt egg capacity	Norm(1.7E+07, 1E+05)	Zabel et al. (2015), p4.6 (reach C)
$\varphi_{t,F}^{nat}$; $\varphi_{t,S}^{nat}$; $\varphi_{t,Y}^{nat}$	Fry-migrant survival in natal stream (fry mover; subyearling stayer; yearling stayer)	0.75; 0.425; 0.325	Zabel et al. (2015), p4.8
$\varphi_{t,F\to F}^{CGR}$	Cougar reservoir survival (fry to fry)	1	Zabel et al. (2015), p4.8
$\varphi_{t,F\to F}^{CGR}$	Cougar reservoir survival (fry to subyearling)	Triang(0.15, 0.2, 0.35)	Zabel et al. (2015), p4.8
$\varphi_{t,S\to S}^{CGR}$	Cougar reservoir survival (subyearling to subyearling)	1	Zabel et al. (2015), p4.8
$\varphi_{t,S\to Y}^{CGR}$	Cougar reservoir survival (subyearling to yearling)	Unif(0.4, 0.9)	Zabel et al. (2015), p4.8
$\varphi_{t,Y\to Y}^{CGR}$	Cougar reservoir survival (yearling to yearling)	1	Zabel et al. (2015), p4.8
$\varphi_{t,F}^{rss}$	River-smolt survival for fry (CGR-SUJ)	Median = 0.0784 (CV = 0.555)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p4.10 (see Section 2.2.6 and Appendix C)

$\varphi_{t,S_{resS}}^{rss}$	River-smolt survival for subyearlings that spent summer in reservoir (CGR-SUJ)	Median = 0.267 (CV = 0.350)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p4.10 (see Section 2.2.6 and Appendix C)
$\varphi_{t,S_{str}}^{rss}$	River-smolt survival for subyearlings that spent summer in natal stream (CGR-SUJ)	Median = 0.267 (CV = 0.350)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p4.10 (see Section 2.2.6 and Appendix C)
$\varphi_{t,Y_{resSW}}^{rss}$	River-smolt survival for yearlings that spent summer and winter in reservoir (CGR-SUJ)	Median = 0.392 (CV = 0.240)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p5.11 (see Section 2.2.6 and Appendix C)
$\varphi_{t,Y_{resW}}^{rss}$	River-smolt survival for yearlings that spent only winter in reservoir (CGR-SUJ)	Median = 0. 392 (CV = 0. 240)	CJS model posterior output adjusted for tagging/hatchery effects, scaled by relative values from Zabel et al. (2015), p5.11 (see Section 2.2.6 and Appendix C)
$\varphi_{t,Y_{str}}^{rss}$	River-smolt survival for yearlings that reared in natal stream (CGR-SUJ)	Median = 0. 392 (CV = 0. 240)	CJS model posterior output adjusted for tagging effects, scaled by relative values from Zabel et al. (2015), p5.11
$\varphi_{0+\rightarrow 3}^{o}$	Survival from natural mortality in the ocean (0+ to age-3 adult)	0.013	Estimated during model calibration, using CJS model posterior output as prior (see Appendix C

			and Appendix D). N.B. 0+ pass SUJ before 1st birthday.
$\varphi^{o}_{1+\to 3}$	Survival from natural mortality in the ocean (1+ to age-3 adult)	0.18	Estimated during model calibration, using CJS model output as prior (see Appendix C and Appendix D)
$\varphi^{o}_{a\to a+1}$	Survival from natural mortality in the ocean (a≥3 and a≤5)	0.397; 0.500; 0.630	CTC (2021)
ε_t	Annual deviates in early ocean survival	{0.734, -0.681, -0.940, -0.981, -0.644, -0.557, -1.216, -1.506, -1.238}	Estimated during model calibration for 2009-2017 (see Section 2.5)
u^{o}_{3}	Harvest rate of age-3 adults when in ocean (commercial fishery)	0.0545	Calculated from CTC (2021) (see Appendix D for details)
u^{o}_{4+}	Harvest rate of age-4 and older adults when in ocean (commercial fishery)	0.109	Calculated from CTC (2021) (see Appendix D for details)
u^{r}	Harvest rate of adults when going from ocean to WFF (terminal fishery)	0.052	Calculated from CTC (2021) (see Appendix D for details)
s_3; s_4; s_5; s_6	Observed proportion of returning adults to McKenzie spawning at age-3 to age-6	0.013; 0.404; 0.558; 0.025	Model used average from ODFW reports (Sharpe et al. 2013, 2014, 2015, 2016, 2017)
Fish Benefits Workbook Output			
$DPE_{t,F}$; $DPE_{t,S}$; $DPE_{t,Y}$	Dam Passage Efficiency for fry; subyearling, yearling stages	Bootstrap 1947-2019	FBW output (USACE 2022), see Section 1.4.
$DPS_{t,F}$; $DPS_{t,S}$; $DPS_{t,Y}$	Dam Passage Survival for fry; subyearling; yearling stages	Bootstrap 1947-2019	FBW output (USACE 2022), see Section 1.4.
River juvenile splits			
$p_{J\to F}$; $p_{J\to S}$; $p_{J\to Y}$	Proportion of emergent fry leaving natal stream as fry movers; subyearling stayers; and yearling stayers	0.94; 0.05; 0,01	Zabel et al. (2015), p4.17, ODFW RST data analysis (2011-2016, see Section 2.2.2)

$p_{F\rightarrow F}$; $p_{F\rightarrow S}$	Proportion of fry in reservoir moving to forebay as fry; subyearlings	0.065; 0.935	Zabel et al. (2015), p4.17 (mean value)
$p_{S\rightarrow S}$; $p_{S\rightarrow Y}$	Proportion of subyearlings in reservoir moving to forebay as subyearlings; yearlings	0.825; 0.175	Zabel et al. (2015), p4.17 (mean value)
$p_{Y\rightarrow Y}$	Proportion of yearlings in reservoir moving to forebay as yearlings	1	Assumed value
Ocean adult splits			
$p_{0+\rightarrow 3}$	Proportion of adults in ocean returning to river to spawn (a≥3 and a≤5)	0.027; 0.750; 0.950	Estimated during model calibration (Section 2.5). Assumes all age-6 adults return to spawn.

Table 8-62. Life cycle model equations. Unless otherwise indicated, equations are specified for t ≥ 1.
aAssumes return to sub-basin and PSM occurs prior to September in each year, so spawning occurs in the model year after return.
bAssumes that subyearling migrant types pass dams before first birthday but pass Willamette Falls as age-1+ smolts.
cWhere a=4, (1 – a) indicates fish in the age-0+ or age-1+ to age-3 stage groups, rather than age-3 only fish.

#	Description	Equation
Adults in freshwater		
1	Natural-origin adults returning to Cougar dam tailrace (Cougar Collection Facility)a	$NOR_t^{CGRTR} = \begin{cases} NOR^{init} \; t = 0 \\ NOR_t^{WFF} * \left(1 - PSM_t^{below}\right) \; t \geq 1 \end{cases}$
2	Natural-origin adult outplants above Cougar (t ≥ 0)	$NO_t = \min(NOR_t^{CGRTR}, \kappa)$
3	Hatchery-origin adult outplants above Cougar (t ≥ 0)	$HO_t = \min(HO^{max}, \kappa - NOR_t^{CGR})$
4	Percentage of hatchery-origin spawners (t ≥ 0)	$pHO_t = HO_t/(NO_t + HO_t)$
5	Natural-origin spawnersa	$NS_t = NO_{t-1} * \left(1 - PSM_{t-1}^{above}\right)$
6	Hatchery-origin spawnersa	$HS_t = HO_{t-1} * \left(1 - PSM_{t-1}^{above}\right)$
7	Percentage of hatchery-origin spawners after PSM^{above}	$pHOS_t = HS_t/(NS_t + HS_t)$
Juveniles in freshwater		
8	Natural-origin eggs	$E_t^N = NS_t * \nu * \eta^N$
9	Hatchery-origin eggs	$E_t^H = HS_t * \nu * \eta^H * RRS$
10	Emergent fry in natal stream	$J_t = \dfrac{\alpha * (E_t^N + E_t^H)}{\left(1 + \dfrac{\alpha * (E_t^N + E_t^H)}{\beta}\right)}$
Movement to Cougar reservoir		
11	Number of fry movers migrating to Cougar reservoir from natal stream in spring	$N_{t,F}^{res} = J_t * p_{J \to F} * \varphi_{t,F}^{nat}$
12	Number of subyearling stayers migrating to Cougar reservoir from natal stream in fall	$N_{t,S}^{res} = F_t * p_{J \to S} * \varphi_{t,S}^{nat}$

13	Number of yearling stayers migrating to Cougar reservoir from natal stream in spring (t ≥ 2)	$N_{t,Y}^{res} = J_{t-1} * p_{J\to Y} * \varphi_{t,Y}^{nat}$
Movement to Cougar forebay		
14	Number of fry in Cougar reservoir moving to forebay in spring	$N_{t,F}^{fby} = N_{t,F}^{res} * p_{F\to F} * \varphi_{t,F\to F}^{CGR}$
15	Number of subyearlings that spent summer in reservoir moving to Cougar forebay	$N_{t,S_{resS}}^{fby} = N_{t,F}^{res} * p_{F\to S} * \varphi_{t,F\to S}^{CGR} + N_{t,F}^{fby} * (1 - DPE_{t,F}) * \varphi_{t,F\to S}^{CGR}$
16	Number of subyearlings that spent summer in natal stream moving to Cougar forebay	$N_{t,S_{str}}^{fby} = N_{t,S}^{res} * p_{S\to S} * \varphi_{t,S\to S}^{CGR}$
17	Number of yearlings that spent summer and winter in reservoir moving to Cougar forebay (t ≥ 2)	$N_{t,F_{resSW}}^{fby} = N_{t-1,S}^{res} * p_{S\to Y} * \varphi_{t-1,S\to Y}^{CGR} + N_{t-1,S_{resS}}^{fby} * (1 - DPE_{t-1,S}) * \varphi_{t-1,S\to Y}^{CGR}$
18	Number of yearlings that spent only winter in reservoir moving to Cougarforebay (t ≥ 2)	$N_{t,Y_{resW}}^{fby} = N_{t-1,S_{str}}^{fby} * (1 - DPE_{t-1,S}) * \varphi_{t-1,S\to Y}^{CGR}$
19	Number of yearlings that reared in natal stream moving to Cougar forebay (t ≥ 2)	$N_{t,Y_{str}}^{fby} = N_{t,Y}^{res} * p_{Y\to Y} * \varphi_{t,Y\to Y}^{CGR}$
Movement to tailrace		
20	Number of different migrant groups passing Cougar dam to tailrace. For yearlings t ≥ 2.	$N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{CGRTR} = N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{fby} * DPE_{t,\{F,S,S,Y,Y,Y\}} * DPS_{t,\{F,S,S,Y,Y,Y\}}$
21	Number of yearlings that did not pass Cougar dam and assumed dead in summer (t ≥ 2)	$N_{t,Y}^{dead} = (N_{t,Y_{resSW}}^{fby} + N_{t,Y_{resW}}^{fby} + N_{t,Y_{str}}^{fby}) * (1 - DPE_{t,Y})$
Smolts at Willamette Falls		
22	Number of different migrant types reaching SUJ. For yearlings t ≥ 2	$N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{SUJ} = N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{BCLTR} * \varphi_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{rss}$
Age structure of adults remaining in ocean		

23	Number of adults (age-3) in the ocean of fry migrant type (t ≥ 3)	$NA^{F}_{t,0+\rightarrow 3} = N^{SUJ}_{t-2,F} * (1 - p_{0+\rightarrow 3}) * \varphi^{o}_{0+\rightarrow 3} * e^{\varepsilon_{t-2}} * (1 - u^{o}_{3})$
24	Number of adults (age-3) in the ocean of subyearling migrant type that spent summer in reservoir (t ≥ 3)b	$NA^{S_{resS}}_{t,1+\rightarrow 3} = N^{SUJ}_{t-2,S_{resS}} * (1 - p_{0+\rightarrow 3}) * \varphi^{o}_{1+\rightarrow 3} * e^{\varepsilon_{t-1}} * (1 - u^{o}_{3})$
25	Number of adults (age-3) in the ocean of subyearling migrant type that spent summer in natal stream (t ≥ 3)b	$NA^{S_{str}}_{t,1+\rightarrow 3} = N^{SUJ}_{t-2,S_{str}} * (1 - p_{0+\rightarrow 3}) * \varphi^{o}_{1+\rightarrow 3} * e^{\varepsilon_{t-1}} * (1 - u^{o}_{3})$
26	Number of adults (age-3) in the ocean of yearling migrant type that spent summer and winter in reservoir (t ≥ 3)	$NA^{Y_{resSW}}_{t,1+\rightarrow 3} = N^{SUJ}_{t-1,Y_{resSW}} * (1 - p_{0+\rightarrow 3}) * \varphi^{o}_{1+\rightarrow 3} * e^{\varepsilon_{t-1}} * (1 - u^{o}_{3})$
27	Number of adults (age-3) in the ocean of yearling migrant type that spent only winter in reservoir (t ≥ 3)	$NA^{Y_{resW}}_{t,1+\rightarrow 3} = N^{SUJ}_{t-1,Y_{resW}} * (1 - p_{0+\rightarrow 3}) * \varphi^{o}_{1+\rightarrow 3} * e^{\varepsilon_{t-1}} * (1 - u^{o}_{3})$
28	Number of adults (age-3) in the ocean of yearling migrant type that reared in natal stream (t ≥ 3)	$NA^{Y_{str}}_{t,1+\rightarrow 3} = N^{SUJ}_{t-1,Y_{str}} * (1 - p_{0+\rightarrow 3}) * \varphi^{o}_{1+\rightarrow 3} * e^{\varepsilon_{t-1}} * (1 - u^{o}_{3})$
29	Number of adults (age ≥ 4 andage ≤ 6) in the ocean of each migrant type (t ≥ 4)c	$NA^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}_{t,a} = N^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}_{t-1,a-1} * (1 - p_{a-1\rightarrow a}) * \varphi^{o}_{a-1\rightarrow a} * (1 - u^{o}_{4+})$
Age structure of adults returning to river		
30	Number of adults (age-3) returning from the ocean of fry migrant type (t ≥ 3)	$NOR^{F}_{t,3} = N^{SUJ}_{t-2,F} * p_{0+\rightarrow 3} * \varphi^{o}_{0+\rightarrow 3} * (1 - u^{r}_{3})$
31	Number of adults (age-3) returning from the ocean of subyearling migrant type that spent summer in reservoir (t ≥ 3)	$NOR^{S_{resS}}_{t,3} = N^{SUJ}_{t-2,S_{resS}} * p_{0+\rightarrow 3} * \varphi^{o}_{1+\rightarrow 3} * (1 - u^{r}_{3})$
32	Number of adults (age-3) returning from the ocean of subyearling migrant type that spent summer in natal stream (t ≥ 3)	$NOR^{S_{str}}_{t,3} = N^{SUJ}_{t-2,S_{str}} * p_{0+\rightarrow 3} * \varphi^{o}_{1+\rightarrow 3} * (1 - u^{r}_{3})$

33	Number of adults (age-3) returning from the ocean of yearling migrant type that spent summer and winter in reservoir (t ≥ 3)	$NOR_{t,3}^{Y_{resSW}} = N_{t-1,Y_{resSW}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
34	Number of adults (age-3) returning from the ocean of yearling migrant type that spent only winter in reservoir (t ≥ 3)	$NOR_{t,3}^{Y_{resW}} = N_{t-1,Y_{resW}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
35	Number of adults (age-3) returning from the ocean of yearling migrant type that reared in natal stream (t ≥ 3)	$NOR_{t,3}^{Y_{str}} = N_{t-1,Y_{str}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1 - u_3^r)$
36	Number of adults (age ≥ 4 andage ≤ 6) returning from the ocean of each migrant type (t ≥ 4)c	$NOR_{t,a}^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}} = N_{t-1,a-1}^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}} * p_{a-1\to a} * \varphi_{a-1\to a}^{o} * (1 - u_{4+}^r)$
Age structure of adults returned to Willamette Falls		
37	Number of adults from age a≥3 and a≤ 6 returned to WFF	$NOR_{t,a}^{WFF} = \begin{cases} s_3 * \left(NOR^{init} / \left(1 - PSM_t^{below}\right)\right) & t < a \\ NOR_{t,a}^{F} + NOR_{t,a}^{S_{resS}} + NOR_{t,a}^{S_{str}} + NOR_{t,a}^{Y_{resSW}} + NOR_{t,a}^{Y_{resW}} + NOR_{t,a}^{Y_{str}} & t \geq a \end{cases}$
38	Total number of natural-origin adults returning to WFF	$NOR_{t}^{WFF} = \sum_{a=3}^{6} NOR_{t,a}^{WFF}$

Middle Fork

Table 8-63. Parameter and other input variable values for the Middle Fork Chinook salmon life cycle model.
aValues for parameters without a time subscript drawn from a distribution vary between simulations.
bα and β parameters of a beta distribution are derived from a mean and CI.

Symbol	Description	Value	Reference
Model parameters			
NOR^{init}	Initial number of natural-origin returns to Dexter Adult Collection Facility	62	Median number of NOR to Dexter from 2002 to 2021 (ODFW count data)
HO^{max}	Maximum hatchery-origin outplants	387 (above HC) 1,257 (above LOP)	Average HO adults put above HC to support bull trout (R. Laird, pers. comm.) See Table 1.9-2 in MFW HGMP (ODFW & USACE 2016a)
κ	Logistical cap on total outplant numbers (natural-origin + hatchery-origin)	5,000	5,000 was assumed to be reasonable based on estimates from outplanting facility capacities (ODFW & USACE 2016a) for LOP/HC during the spawning period
p_{LOP}	Outplant ratio LOP:HC	1.0 (NAA, Alt2a, Alt2b); 0.42 (otherwise)	Assumed from relative habitat capacity in above dam reaches (Zabel et al. 2015)
PSM_t^{below}	Pre-spawn mortality (PSM) en route to dam	Beta(130.543, 654.937)b; mean=0.165 (95% CI 0.141-0.193)	Based on a 2011-2014 radio-telemetry study

			(Keefer et al. 2017)
PSM_t^{above}	Pre-spawn mortality (PSM) on spawning grounds above dam	$f(7DADM_t, pHO_t)$	Adjusted function from Bowerman et al. (2018), see Section 2.4.1
$7DADM_t$	7-day average of daily maximum temperature (oC) above LOP reservoir	Bootstrap 1947-2019	Data from USGS gage 14185000 (see freshwater adult section). Years with missing data were back-filled using the water-year type mean value (see Section 2.4.1).
η^N	Fecundity (natural origin adults)	4,500	(ODFW & USACE 2016a, p61)
η^H	Fecundity (hatchery origin adults)	3,815	Empirical data from 2009-2015 (R. Laird, pers. comm.). N.B. data not included in MFW HGMP (ODFW & USACE 2016a)
ν	Sex ratio (female spawners to total spawners) for natural- and hatchery-origin adults	0.45	Empirical data from 2009-2015 (R. Laird, pers. comm.). N.B. data not included in MFW HGMP (ODFW & USACE 2016)
RRS	Relative reproductive success of HO to NO adults	Triang(0.39, 0.53, 0.84)	Based on analysis of Christie et al. (2014), see Section 2.4.3.
α^{HC}	Beverton-Holt NO egg-fry survival (above HC reservoir)	0.448	Zabel et al. (2015), p6.9
α^{LOP}	Beverton-Holt NO egg-fry survival (above LOP reservoir)	0.448	Zabel et al. (2015), p6.9

β^{HC}	Beverton-Holt egg capacity (above HC reservoir)	Unif(7E+06, 13E+06)	Zabel et al. (2015), Table 10.4
β^{LOP}	Beverton-Holt egg capacity (above LOP reservoir)	Unif(17E+06, 25.5E+06)	Zabel et al. (2015), Table 10.4
$\varphi_{t,\{F,S,Y\}}^{HC_{nat}}$	Fry-migrant survival in natal stream above HC for fry mover; subyearling stayer; and yearling stayer	0.6; 0.25; 0.2	Zabel et al. (2015), p6.10
$\varphi_{t,\{F,S,Y\}}^{LOP_{nat}}$	Fry-migrant survival in natal stream above LOP for fry mover; subyearling stayer; and yearling stayer	0.6; 0.25; 0.2	Zabel et al. (2015), p6.10
$\varphi_{t,F\to F}^{HC}$	HC reservoir survival (fry to fry)	1	Zabel et al. (2015), p3.96
$\varphi_{t,F\to S}^{HC}$	HC reservoir survival (fry to subyearling)	Triang(0.15, 0.25, 0.35)	Zabel et al. (2015), Table 10.4
$\varphi_{t,S\to S}^{HC}$	HC reservoir survival (subyearling to subyearling)	1	Zabel et al. (2015), p3.96
$\varphi_{t,S\to Y}^{HC}$	HC reservoir survival (subyearling to yearling)	Unif(0.4, 0.9)	Zabel et al. (2015), Table 10.4
$\varphi_{t,Y\to Y}^{HC}$	HC reservoir survival (yearling to yearling)	1	Zabel et al. (2015), p3.96
$\varphi_{t,F\to F}^{LOP}$	LOP reservoir survival (fry to fry)	1	Zabel et al. (2015), p3.96
$\varphi_{t,F\to S}^{LOP}$	LOP reservoir survival (fry to subyearling)	Triang(0.15, 0.25, 0.35)	Zabel et al. (2015), Table 10.4
$\varphi_{t,S\to S}^{LOP}$	LOP reservoir survival (subyearling to subyearling)	1	Zabel et al. (2015), p3.96
$\varphi_{t,S\to Y}^{LOP}$	LOP reservoir survival (subyearling to yearling)	Unif(0.4, 0.9)	Zabel et al. (2015), Table 10.4
$\varphi_{t,Y\to Y}^{LOP}$	LOP reservoir survival (yearling to yearling)	1	Zabel et al. (2015), p3.96
$\varphi_{t,F}^{rss}$	River smolt survival for fry (DEXTR-SUJ)	Beta(9.823, 22.753)b; mean= 0.301, SD = 0.079	CJS model posterior output adjusted for tagging/hatchery effects (see Appendix C).
$\varphi_{t,S_{resS}}^{rss}$	River smolt survival for subyearlings that spent summer in reservoir (DEXTR-SUJ)	Beta(2.499, 2.905)b; mean= 0.462, SD = 0.197	CJS model posterior output adjusted for tagging/hatchery

			effects (see Appendix C).
$\varphi_{t,S_{str}}^{rss}$	River smolt survival for subyearlings that spent summer in natal stream (DEXTR-SUJ)	Beta(2.499, 2.905)	CJS model posterior output adjusted for tagging/hatchery effects (see Appendix C).
$\varphi_{t,Y_{resSW}}^{rss}$	River smolt survival for yearlings that spent summer and winter in reservoir (DEXTR-SUJ)	Beta(7.452, 6.248)b; mean= 0.543, SD = 0.129	CJS model posterior output adjusted for tagging/hatchery effects (see Appendix C).
$\varphi_{t,Y_{resW}}^{rss}$	River smolt survival for yearlings that spent only winter in reservoir (DEXTR-SUJ)	Beta(7.452, 6.248)	CJS model posterior output adjusted for tagging/hatchery effects (see Appendix C).
$\varphi_{t,Y_{str}}^{rss}$	River smolt survival for yearlings that reared in natal stream (DEXTR-SUJ)	Beta(7.452, 6.248)	CJS model posterior output adjusted for tagging/hatchery effects (see Appendix C).
$\varphi_{0+\to 3}^{o}$	Survival from natural mortality in the ocean (0+ to age-3 adult)	0.0389	Estimated during model calibration, using CJS model posterior output as prior (see Appendix C and Appendix D). N.B. 0+ pass SUJ before 1st birthday.
$\varphi_{1+\to 3}^{o}$	Survival from natural mortality in the ocean (1+ to age-3 adult)	0.0609	Estimated during model calibration, using CJS model posterior output as prior (see

			Appendix C and Appendix D)
$\varphi^{o}_{\{a\to a+1\}}$	Survival from natural mortality in the ocean (age a≥3 and a≤5)	0.397; 0.500; 0.630	Assumed by CTC(2021), see Appendix D for details.
ε_t	Annual deviates in early ocean survival	{0.642, -1.093, -1.733, 0.781, 0.269, -0.518, -0.318, -0.51, -0.512, -1.481, -1.586, -0.64, 1.002, -0.002, -0.015, -0.019}	Estimated during model calibration for 2005-2020 (See Section 2.5)
u^{o}_{3}	Harvest rate of age-3 adults in ocean (commercial fishery)	0.0545	Calculated from CTC (2021) (see Appendix D for details)
u^{o}_{4+}	Harvest rate of age-4 and older adults in ocean (commercial fishery)	0.109	Calculated from CTC (2021) (see Appendix D for details)
u^{r}_{3+}	Harvest rate of age-3 and older adults when going from ocean to WFF (terminal fishery)	0.052	Calculated from CTC (2021) (see Appendix D for details)
$s_{\{3,4,5,6\}}$	Observed proportion of returning adults to South Santiam spawning at age-3 to age-6	0.089; 0.715; 0.189; 0.008	Model used average from ODFW reports (Sharpe et al. 2013, 2014, 2015, 2016, 2017). N. B. Values from South Santiam due to low return numbers to MF
Fish Benefits Workbook Output			
$DPE^{HC}_{t,\{F,S,Y\}}$	HC Dam Passage Efficiency for fry, subyearling, and yearlings	Bootstrap 1947-2019	FBW output (USACE 2022), see Section 1.4.
$DPS^{HC}_{t,\{F,S,Y\}}$	HC Dam Passage Survival for fry, subyearling, and yearlings)	Bootstrap 1947-2019	FBW output (USACE 2022), see Section 1.4.

$DPE^{LOP}_{t,\{F,S,Y)}$	LOP Dam Passage Efficiency for fry, subyearling, and yearlings	Bootstrap 1947-2019	FBW output (USACE 2022), see Section 1.4.
$DPS^{LOP}_{t,\{F,S,Y\}}$	LOP Dam Passage Survival for fry, subyearling, and yearlings	Bootstrap 1947-2019	FBW output (USACE 2022), see Section 1.4.
River juvenile splits			
$p^{HC}_{\{J\to F,J\to S,J\to Y\}}$	Proportion of emergent fry leaving natal stream above HC as fry movers; subyearling stayers; and yearling stayers, respectively	0.85; 0.10; 0.05	Zabel et al. (2015), p6.25
$p^{LOP}_{\{J\to F,J\to S,J\to Y\}}$	Proportion of emergent fry leaving natal stream above LOP as fry movers; subyearling stayers; and yearling stayers, respectively	0.85; 0.10; 0.05	Zabel et al. (2015), p6.25, ODFW RST data analysis (2011-2016, see Section 2.2.2)
$p^{HC}_{\{F\to F,F\to S\}}$	Proportion of fry in HC reservoir moving to forebay as fry; and subyearlings	0.04; $1-p^{HC}_{F\to F}$	Zabel et al. (2015), Table 10.4 (mean value)
$p^{HC}_{\{S\to S,S\to Y\}}$	Proportion of subyearlings in HC reservoir moving to forebay as subyearlings; and yearlings	0.825; $1-p^{HC}_{S\to S}$	Zabel et al. (2015), Table 10.4 (mean value)
$p^{HC}_{Y\to Y}$	Proportion of yearlings in HC reservoir moving to forebay as yearlings	1	Assumed value
$p^{LOP}_{\{F\to F,F\to S\}}$	Proportion of fry in LOP reservoir moving to forebay as fry; and subyearlings	0.1; $1-p^{LOP}_{F\to F}$	Zabel et al. (2015), Table 10.4 (mean value)
$p^{LOP}_{\{S\to S,S\to Y\}}$	Proportion of subyearlings in LOP reservoir moving to forebay as subyearlings; and yearlings	0.875; $1-p^{LOP}_{S\to S}$	Zabel et al. (2015), Table 10.4 (mean value)
$p^{LOP}_{Y\to Y}$	Proportion of yearlings in LOP reservoir moving to forebay as yearlings	1	Assumed value
Ocean adult splits			
$p_{\{0+\to 3;a\to a+1\}}$	Proportion of adults in ocean returning to river to spawn (a≥3 and a≤5)	0.0135; 0.2055; 0.950; 1	Estimated during model calibration (see Section 2.5). Assumes all age-6

			adults return to spawn.

Table 8-64. Middle Fork Chinook salmon life cycle model equations. Unless otherwise indicated, equations are specified for t ≥ 1.
aAssumes return to sub-basin and PSM occurs prior to September in each year, so spawning occurs in the model year after return.
bAssumes that all fish coming from HC head direct to LOP forebay and survive to attempt dam passage through LOP.
cAssumes that subyearling migrant types pass dams before first birthday but pass Willamette Falls as age-1+ smolts.
dWhere a=4, (1 – a) indicates fish in the age-0+ or age-1+ to age-3 stage groups, rather than age-3 only fish.

#	Description	Equation
Adults in freshwater		
1	Natural-origin adults returning to LOP dam tailrace (LOP Adult Collection Facility)a	$NOR_t^{LOPTR} = \begin{cases} NOR^{init} \; t = 0 \\ NOR_t^{WFF} * \left(1 - PSM_t^{below}\right) \; t \geq 1 \end{cases}$
2	Natural-origin adult outplants above LOP (t ≥ 0)	$NO_t^{LOP} = \min\begin{pmatrix} p_{LOP} * NOR_t^{LOPTR}, \\ p_{LOP} * \kappa \end{pmatrix}$
3	Natural-origin adult outplants above HC (t ≥ 0)	$NO_t^{HC} = \min\begin{pmatrix} (1 - p_{LOP}) * NOR_t^{LOPTR}, \\ (1 - p_{LOP}) * \kappa \end{pmatrix}$
4	Hatchery-origin adult outplants above LOP (t ≥ 0)	$HO_t^{LOP} = \min\begin{pmatrix} p_{LOP} * HO^{max}, \\ p_{LOP} * \kappa - NOR_t^{LOPTR} \end{pmatrix}$
5	Hatchery-origin adult outplants above HC (t ≥ 0)	$HO_t^{HC} = \min\begin{pmatrix} (1 - p_{LOP}) * HO^{max}, \\ (1 - p_{LOP}) * \kappa - NOR_t^{LOPTR} \end{pmatrix}$
6	Percentage of hatchery-origin outplants (t ≥ 0)	$pHO_t = HO_t^{LOP} / (NO_t^{LOP} + HO_t^{LOP})$
7	Natural-origin spawners above LOPa	$NS_t^{LOP} = NO_{t-1}^{LOP} * \left(1 - PSM_{t-1}^{above}\right)$
8	Natural-origin spawners above HCa	$NS_t^{HC} = NO_{t-1}^{HC} * \left(1 - PSM_{t-1}^{above}\right)$
9	Hatchery-origin spawners above LOPa	$HS_t^{LOP} = HO_{t-1}^{LOP} * \left(1 - PSM_{t-1}^{above}\right)$
10	Hatchery-origin spawners above HCa	$HS_t^{HC} = HO_{t-1}^{HC} * \left(1 - PSM_{t-1}^{above}\right)$
11	Percentage of hatchery-origin spawners above LOP after PSM^{above}	$pHOS_t^{LOP} = HS_t^{LOP} / (NS_t^{LOP} + HS_t^{LOP})$

12	Percentage of hatchery-origin spawners above HC after PSM^{above}	$pHOS_t^{HC} = HS_t^{HC}/(NS_t^{HC} + HS_t^{HC})$
Juveniles in freshwater		
13	Natural-origin eggs above LOP	$E_t^{N,LOP} = NS_t^{LOP} * \nu * \eta^N$
14	Natural-origin eggs above HC	$E_t^{N,HC} = NS_t^{HC} * \nu * \eta^N$
15	Hatchery-origin eggs above LOP	$E_t^{H,LOP} = HS_t^{LOP} * \nu * \eta^H * RRS$
16	Hatchery-origin eggs above HC	$E_t^{H,HC} = HS_t^{HC} * \nu * \eta^H * RRS$
17	Emergent fry in natal stream above LOP	$J_t^{Fos} = \dfrac{\alpha^{LOP} * \left(E_t^{N,LOP} + E_t^{H,LOP}\right)}{\left(1 + \dfrac{\alpha^{LOP} * \left(E_t^{N,LOP} + E_t^{H,LOP}\right)}{\beta^{LOP}}\right)}$
18	Emergent fry in natal stream above HC	$J_t^{HC} = \dfrac{\alpha^{HC} * \left(E_t^{N,HC} + E_t^{H,HC}\right)}{\left(1 + \dfrac{\alpha^{HC} * \left(E_t^{N,HC} + E_t^{H,HC}\right)}{\beta^{HC}}\right)}$
Movement to HC reservoir		
19	Number of fry movers migrating to HC reservoir from natal stream in spring	$N_{t,F}^{HC_{res}} = J_t^{HC} * p_{J\to F}^{HC} * \varphi_{t,F}^{HC_{nat}}$
20	Number of subyearling stayers migrating to HC reservoir from natal stream in fall	$N_{t,S}^{HC_{res}} = J_t^{HC} * p_{J\to S}^{HC} * \varphi_{t,S}^{HC_{nat}}$
21	Number of yearling stayers migrating to HC reservoir from natal stream in spring (t ≥ 2)	$N_{t,Y}^{HC_{res}} = J_{t-1}^{HC} * p_{J\to Y}^{HC} * \varphi_{t,Y}^{HC_{nat}}$
Movement to LOP reservoir		
22	Number of fry movers migrating to LOP reservoir from natal stream in spring	$N_{t,F}^{LOP_{res}} = J_t^{LOP} * p_{J\to F}^{LOP} * \varphi_{t,F}^{LOP_{nat}}$
23	Number of subyearling stayers migrating to LOP reservoir from natal stream in fall	$N_{t,S}^{LOP_{res}} = J_t^{LOP} * p_{J\to S}^{LOP} * \varphi_{t,S}^{LOP_{nat}}$

24	Number of yearling stayers migrating to LOP reservoir from natal stream in spring (t ≥ 2)	$N_{t,Y}^{LOP_{res}} = J_{t-1}^{LOP} * p_{J\to Y}^{LOP} * \varphi_{t,Y}^{LOP_{nat}}$
Movement to HC forebay		
25	Number of fry in HC reservoir moving to forebay	$N_{t,F}^{HC_{fby}} = N_{t,F}^{HC_{res}} * p_{F\to F}^{HC} * \varphi_{t,F\to F}^{HC}$
26	Number of subyearlings that spent summer in reservoir moving to HC forebay	$N_{t,S_{resS}}^{HC_{fby}} = N_{t,F}^{HC_{res}} * p_{F\to S}^{HC} * \varphi_{t,F\to S}^{HC} + N_{t,F}^{HC_{fby}} * \left(1 - DPE_{t,F}^{HC}\right) * \varphi_{t,F\to S}^{HC}$
27	Number of subyearlings that spent summer in natal stream moving to HC forebay	$N_{t,S_{str}}^{HC_{fby}} = N_{t,S}^{HC_{res}} * p_{S\to S}^{HC} * \varphi_{t,S\to S}^{HC}$
28	Number of yearlings that spent summer and winter in reservoir moving to HC forebay (t ≥ 2)	$N_{t,Y_{resSW}}^{HC_{fby}} = N_{t-1,S}^{HC_{res}} * p_{S\to Y}^{HC} * \varphi_{t-1,S\to Y}^{HC} + N_{t-1,S_{resS}}^{HC_{fby}} * \left(1 - DPE_{t-1,S}^{HC}\right) * \varphi_{t-1,S\to Y}^{HC}$
29	Number of yearlings that spent only winter in reservoir moving to HC forebay (t ≥ 2)	$N_{t,Y_{resW}}^{HC_{fby}} = N_{t-1,S_{str}}^{HC_{fby}} * \left(1 - DPE_{t-1,S}^{HC}\right) * \varphi_{t-1,S\to Y}^{HC}$
30	Number of yearlings that reared in natal stream moving to HC forebay (t ≥ 2)	$N_{t,Y_{str}}^{HC_{fby}} = N_{t,Y}^{HC_{res}} * p_{Y\to Y}^{HC} * \varphi_{t,Y\to Y}^{HC}$
Movement to HC tailrace		
31	Number of different migrant types passing HC dam to HC tailrace. For yearlings t ≥ 2	$N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{HCTR} = N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{HC_{fby}} * DPE_{t,\{F,S,S,Y,Y,Y\}}^{HC} * DPS_{\{t,F,S,S,Y,Y,Y\}}^{HC}$
32	Number of yearlings that did not pass HC dam and assumed dead in summer (t ≥ 2)	$N_{t,Y}^{HC_{dead}} = \left(N_{t,Y_{resSW}}^{HC_{fby}} + N_{t,Y_{resW}}^{HC_{fby}} + N_{t,Y_{str}}^{HC_{fby}}\right) * \left(1 - DPE_{t,Y}^{HC}\right)$
Movement to LOP forebay		

33	Number of fry in LOP reservoir moving to forebayb	$N_{t,F}^{LOP_{fby}} = N_{t,F}^{LOP_{res}} * p_{F\to F}^{LOP} * \varphi_{t,F\to F}^{LOP} + N_{t,F}^{HCTR}$
34	Number of subyearlings that spent summer in reservoir moving to LOP forebayb	$N_{t,S_{resS}}^{LOP_{fby}} = N_{t,F}^{LOP_{res}} * p_{F\to S}^{LOP} * \varphi_{t,F\to S}^{LOP} + N_{t,F}^{LOP_{fby}} * \left(1 - DPE_{t,F}^{LOP}\right) * \varphi_{t,F\to S}^{LOP} + N_{t,S_{resS}}^{HCTR}$
35	Number of subyearlings that spent summer in natal stream moving to LOP forebayb	$N_{t,S_{str}}^{LOP_{fby}} = N_{t,S}^{LOP_{res}} * p_{S\to S}^{LOP} * \varphi_{t,S\to S}^{LOP} + N_{t,S_{str}}^{HCTR}$
36	Number of yearlings that spent summer and winter in reservoir moving to LOP forebay (t ≥ 2)b	$N_{t,Y_{resSW}}^{LOP_{fby}} = N_{t-1,S}^{LOP_{res}} * p_{S\to Y}^{LOP} * \varphi_{t-1,S\to Y}^{LOP} + N_{t-1,S_{resS}}^{LOP_{fby}} * \left(1 - DPE_{t-1,S}^{LOP}\right) * \varphi_{t-1,S\to Y}^{LOP} + N_{t,Y_{resSW}}^{HCTR}$
37	Number of yearlings that spent only winter in reservoir moving to LOP forebay (t ≥ 2)b	$N_{t,Y_{resW}}^{LOP_{fby}} = N_{t-1,S_{str}}^{LOP_{fby}} * \left(1 - DPE_{t-1,S}^{LOP}\right) * \varphi_{t-1,S\to Y}^{LOP} + N_{t,Y_{resW}}^{HCTR}$
38	Number of yearlings that reared in natal stream moving to LOP forebay (t ≥ 2)b	$N_{t,Y_{str}}^{LOP_{fby}} = N_{t,Y}^{LOP_{res}} * p_{Y\to Y}^{LOP} * \varphi_{t,Y\to Y}^{LOP} + N_{t,Y_{str}}^{HCTR}$
Movement to LOP tailrace		
39	Number of different migrant types passing LOP dam to LOP tailrace. For yearlings t ≥ 2	$N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{LOPTR} = N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{LOP_{fby}} * DPE_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{LOP} * DPS_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{LOP}$
40	Number of yearlings that did not pass LOP dam and assumed dead in summer (t ≥ 2)	$N_{t,Y}^{LOP_{dead}} = \left(N_{t,Y_{resSW}}^{LOP_{fby}} + N_{t,Y_{resW}}^{LOP_{fby}} + N_{t,Y_{str}}^{LOP_{fby}}\right) * \left(1 - DPE_{t,Y}^{LOP}\right)$
Smolts at Willamette Falls		
41	Number of different migrant types reaching SUJ. For yearlings t ≥ 2	$N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{SUJ} = N_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{LOPTR} * \varphi_{t,\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}^{rss}$
Age structure of adults remaining in ocean		

42	Number of adults (age-3) in the ocean of fry migrant type (t ≥ 3)	$NA^{F}_{t,0+\rightarrow 3} = N^{SUJ}_{t-2,F} * (1 - p_{0+\rightarrow 3}) * \varphi^{o}_{0+\rightarrow 3} * e^{\varepsilon_{t-2}} * (1 - u^{o}_{3})$
43	Number of adults (age-3) in the ocean of subyearling migrant type that spent summer in reservoir (t ≥ 3)c	$NA^{S_{resS}}_{t,1+\rightarrow 3} = N^{SUJ}_{t-2,S_{resS}} * (1 - p_{0+\rightarrow 3}) * \varphi^{o}_{1+\rightarrow 3} * e^{\varepsilon_{t-1}} * (1 - u^{o}_{3})$
44	Number of adults (age-3) in the ocean of subyearling migrant type that spent summer in natal stream (t ≥ 3)c	$NA^{S_{str}}_{t,1+\rightarrow 3} = N^{SUJ}_{t-2,S_{str}} * (1 - p_{0+\rightarrow 3}) * \varphi^{o}_{1+\rightarrow 3} * e^{\varepsilon_{t-1}} * (1 - u^{o}_{3})$
45	Number of adults (age-3) in the ocean of yearling migrant type that spent summer and winter in reservoir (t ≥ 3)	$NA^{Y_{resSW}}_{t,1+\rightarrow 3} = N^{SUJ}_{t-1,Y_{resSW}} * (1 - p_{0+\rightarrow 3}) * \varphi^{o}_{1+\rightarrow 3} * e^{\varepsilon_{t-1}} * (1 - u^{o}_{3})$
46	Number of adults (age-3) in the ocean of yearling migrant type that spent only winter in reservoir (t ≥ 3)	$NA^{Y_{resW}}_{t,1+\rightarrow 3} = N^{SUJ}_{t-1,Y_{resW}} * (1 - p_{0+\rightarrow 3}) * \varphi^{o}_{1+\rightarrow 3} * e^{\varepsilon_{t-1}} * (1 - u^{o}_{3})$
47	Number of adults (age-3) in the ocean of yearling migrant type that reared in natal stream (t ≥ 3)	$NA^{Y_{str}}_{t,1+\rightarrow 3} = N^{SUJ}_{t-1,Y_{str}} * (1 - p_{0+\rightarrow 3}) * \varphi^{o}_{1+\rightarrow 3} * e^{\varepsilon_{t-1}} * (1 - u^{o}_{3})$
48	Number of adults (age ≥ 4 and age < 6) in the ocean of each migrant type (t ≥ 4)d	$NA^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}_{t,a} = N^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}}_{t-1,a-1} * (1 - p_{a-1\rightarrow a}) * \varphi^{o}_{a-1\rightarrow a} * (1 - u^{o}_{4+})$
Age structure of adults returning to river		
49	Number of adults (age-3) returning from the ocean of fry migrant type (t ≥ 3)	$NOR^{F}_{t,3} = N^{SUJ}_{t-2,F} * p_{0+\rightarrow 3} * \varphi^{o}_{0+\rightarrow 3} * (1 - u^{r}_{3})$
50	Number of adults (age-3) returning from the ocean of subyearling	$NOR^{S_{resS}}_{t,3} = N^{SUJ}_{t-2,S_{resS}} * p_{0+\rightarrow 3} * \varphi^{o}_{1+\rightarrow 3} * (1 - u^{r}_{3})$

	migrant type that spent summer in reservoir (t ≥ 3)	
51	Number of adults (age-3) returning from the ocean of subyearling migrant type that spent summer in natal stream (t ≥ 3)	$NOR_{t,3}^{S_{str}} = N_{t-2,S_{str}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1-u_3^r)$
52	Number of adults (age-3) returning from the ocean of yearling migrant type that spent summer and winter in reservoir (t ≥ 3)	$NOR_{t,3}^{Y_{resSW}} = N_{t-1,Y_{resSW}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1-u_3^r)$
53	Number of adults (age-3) returning from the ocean of yearling migrant type that spent only winter in reservoir (t ≥ 3)	$NOR_{t,3}^{Y_{resW}} = N_{t-1,Y_{resW}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1-u_3^r)$
54	Number of adults (age-3) returning from the ocean of yearling migrant type that reared in natal stream (t ≥ 3)	$NOR_{t,3}^{Y_{str}} = N_{t-1,Y_{str}}^{SUJ} * p_{0+\to 3} * \varphi_{1+\to 3}^{o} * (1-u_3^r)$
55	Number of adults (age ≥ 4 and age ≤ 6) returning from the ocean of each migrant type (t ≥ 4)d	$NOR_{t,a}^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}} = N_{t-1,a-1}^{\{F,S_{resS},S_{str},Y_{resSW},Y_{resW},Y_{str}\}} * p_{a-1\to a} * \varphi_{a-1\to a}^{o} * (1-u_{4+}^r)$
Age structure of adults returned to Willamette Falls		
56	Number of adults from age a≥3 and a≤ 6 that returned to WFF	$NOR_{t,a}^{WFF} = \begin{cases} s_a * \left(NOR^{init}/\left(1-PSM_t^{below}\right)\right) & t < a \\ NOR_{t,a}^{F} + NOR_{t,a}^{S_{resS}} + NOR_{t,a}^{S_{str}} + NOR_{t,a}^{Y_{resSW}} + NOR_{t,a}^{Y_{resW}} + NOR_{t,a}^{Y_{str}} & t \geq a \end{cases}$
57	Total number of natural-origin adults returning to WFF	$NOR_t^{WFF} = \sum_{a=3}^{6} NOR_{t,a}^{WFF}$

Additional notes

$\varphi_{t,F}^{rss}$: Empirical estimates derived from fish released (May/June) at DEX tailrace and detected between May-September at SUJ for all years combined (2011-2014).

$\varphi_{t,S_{resS}}^{rss}$: Empirical estimate was derived by subtracting an estimate of the reservoir-dam-passage survival (from paired releases for LOP-HOR and DEX tailrace) to the posterior release (above dam)-to-SUJ survival estimate for Fall Creek

$\varphi_{t,Y_{resSW}}^{rss}$: Estimates derived averaging estimates from beach seine data (i.e., NO) from 2014 and 2016. These fish were released between April and June and the detections at SUJ occurred

Winter Steelhead Model Specifications

Model definitions

Table 8-65. Index, parameter, and other state variable definitions used in fitting the winter steelhead IPA LCM and making projections under different dam passage alternatives.

Symbol	Description	Value	Reference
		Index	
t	Time step of population from initial period for model fitting to end of 30-year EIS management horizon for projections	t=1984, …, 2054	N.B. time period included age structure initialisation (1984-1990), model fitting (1991-2021), and projections following a two-year burn-in.
		Freshwater stage	
η_v	Fecundity (virgin spawners)	4,000	Zabel et al. (2015), Section 3.2.1
η_r	Fecundity (repeat spawners)	5,200	Zabel et al. (2015), Section 3.2.1
p_{vs}	Proportion of virgin spawners	$1 - p_{rs}$	
p_{rs}	Proportion of repeat spawners	0.138	Clemens (2015), Section 3.2.3
f	Female eggs per female spawner $\eta = (p_{vs} * f_v + p_{rs} * f_r)/2$	2,078	Calculated, see Section 3.2.1
ϕ^F	Freshwater survival above Foster	Estimated parameter	Estimated during model fitting, see Section 3.5
δ_{GPR}; δ_{DET}	Scalar on ϕ^F for freshwater survival above Green Peter and above Detroit	1.165; 1.126	Calculated using relative survival estimates in Zabel et al. (2015), see Table 3.3.1
β_{FOS}; β_{GPR}; β_{DET}	Beverton-Holt smolt carrying capacity. Number of smolts above Foster; above Green Peter; and above Detroit	76,473;, 42,596; 112,833;	Bond et al. (2017), Section 3.2.1 (mainstem + current side channel)
		Passage stage	
DPE_t	Dam Passage Efficiency for age-2 smolts (for Foster, Green Peter, Detroit)	Bootstrap 1947-2019	FBW output (USACE 2022), Section1.4.

DPS_t	Dam Passage Survival for age-2 smolts (for Foster, Green Peter, Detroit)	Bootstrap 1947-2019	FBW output (USACE 2022), Section1.4.
ϕ^{RSS}	Downstream smolt migration survival from dam tailrace to Willamette Falls	0.74	Assumed value, see Section 3.2.2
Marine stage			
ϕ_t^{SAS}	Smolt-adult (marine) survival rate in year t	Time series 1991-2018	Based upon Wind River (Wilson et al. 2021) and Snake River (McCann et al. 2022) estimates, see Appendix E for details
p_{v4}	Proportion of virgin spawners that return at age 4	0.684	Clemens (2015), Section 3.2.3
p_{v5}	Proportion of virgin spawners that return at age 5	0.287	Clemens (2015), Section 3.2.3
p_{v6}	Proportion of virgin spawners that return at age 6	0.030	Clemens (2015), Section 3.2.3
ϕ^K	Kelt to repeat spawner survival rate	0.160	Clemens (2015), Section 3.2.3
Model fitting and projection			
N_t^{obs}	Annual count of adult female steelhead at Foster tailrace in year t	1991-2021	Reconstructed from Mapes et al. (2017) and ODFW (Brett Boyd, pers. comm.), see Section 3.1 for details (Figure 3.1.3)
d_t	Annual deviates in smolt-adult (marine) survival rate	Estimated parameters	Estimated during model fitting for years 1993-2019, see Section 3.3.1
σ_d	Prior standard deviation for the deviates from the estimated mean marine survival rate	1.0	Assumed value
N_{1991}	Initial abundance of adult female steelhead at Foster tailrace in 1991	Estimated parameter	See Section 3.3.1. N.B. assumed to be the abundance 1984-1990 for initialisation of model age structure in 1991.

$\mu_{N_{1991}}$	Prior median value for the abundance of adult female steelhead at Foster tailrace in 1991	142	See Section 3.3.1
σ_N	Prior standard deviation in the natural logarithm of abundance of female steelhead at Foster tailrace	0.53	See Section 3.3.1
σ_{SHL}^2	Variance in the lognormal likelihood function	Estimated parameter	See Section 3.3.1
μ_{SAS}	Long-term average of estimated marine survival rate 1986-2018	0.0423	See Appendix F
σ_{SAS}	Standard deviation in the natural logarithm of estimates of marine survival rate 1986-2018	0.963	See Appendix F
ρ	Lag-1 autocorrelation coefficient in estimated marine survival rate 1986-2018	0.601	See Appendix F
ϕ_t^M	Simulated future marine survival rate under lag-1 autocorrelation	Time series 2019-2054	See Appendix F

Model equations applied in model fitting and projections

Table 8-66. Steelhead life cycle model equations as applied in model fitting and projections. For projections in years t=2022-2054, DPE and DPS are bootstrapped from the FBW values in the 1947-2019 period of record (Section 1.4). Marine survival rates $\boldsymbol{\phi_t^M}$ for projections were simulated assuming lag-1 autocorrelation following the equations in Appendix F.
aFor Green Peter and Detroit juvenile production, smolt capacity will be β_{GPR} and β_{DET}, with ϕ^F scaled by δ_{GPR} and δ_{DET}, respectively.
bFor Green Peter and Detroit dam passage, DPE and DPS values relate to FBW output for these dams.

	Description	Equation
1	Number of age-2 female juveniles above Foster dam in year ta	$N_t^{juv} = \begin{cases} f * \phi^F * N_{1991} / \left(1 + \dfrac{\eta * \phi^F * N_{1991}}{\beta_{FOS}}\right); \ t < 1991 \\ f * \phi^F * N_t^{spwn} / \left(1 + \dfrac{\eta * \phi^F * N_t^{spwn}}{\beta_{FOS}}\right); \ t \geq 1991 \end{cases}$
2	Number of age-2 female smolts at Willamette Fallsb	$N_t^{smolt} = N_t^{juv} * DPE_t * DPS_t * \phi^{RSS}$
3	Number of adults in the ocean	$N_t^{ad} = \begin{cases} N_t^{smolt} * \phi_t^{SAS} * e^{d_t}; \ t \leq 2019 \\ N_t^{smolt} * \phi_t^{M}; \ t > 2019 \end{cases}$
4	Number of virgin spawners at age-4	$N_t^{v4} = N_t^{ad} * p_{v4}$
5	Number of virgin spawners at age-5	$N_t^{v5} = N_t^{ad} * p_{v5}$
6	Number of virgin spawners at age-6	$N_t^{v6} = N_t^{ad} * p_{v6}$
7	Number of repeat spawners at age-5	$N_t^{r5} = N_{t-1}^{v4} * \phi^K$
8	Number of repeat spawners at age-6	$N_t^{r6} = N_{t-1}^{v5} * \phi^K$
9	Number of repeat spawners at age-7	$N_t^{r7} = N_{t-1}^{v6} * \phi^K$
10	Number of spawners in year t	$N_t^{spwn} = \begin{cases} N_{1991}; \ t < 1991 \\ N_{t-4}^{v4} + N_{t-5}^{v5} + N_{t-6}^{v6} + N_{t-5}^{r5} + N_{t-6}^{r6} + N_{t-7}^{r7}; \ t \geq 1991 \end{cases}$

Cormack-Jolly-Seber (CJS) analysis of PIT tag data

Survival and detection estimates for wild Chinook salmon in the Willamette River basin

Survivorship is likely one of the most critical life-history parameters for understanding fish population dynamics (Hilborn and Walters 1992). Still, there is very limited information with which to estimate survival rates for wild Chinook salmon in the Willamette River basin. The LCM used PIT-tagged hatchery-reared juvenile Chinook salmon released below project dams in the Willamette River basin to estimate survival rates. The standard statistical framework used to estimate survival rates is the Cormack-Jolly-Seber model (CJS model; Cormack 1964; Jolly 1965; Seber 1965), which has been applied routinely to study survival in salmonids (Skalski et al. 1998; Thorley and Andrusak 2017). The procedure for estimating the survival parameters is typically carried out using maximum likelihood estimation (MLE) (Skalski et al. 1998; White and Burnham 1999), but recently the estimation procedure has also been performed using a Bayesian framework (Royle 2008; Thorley and Andrusak 2017). Bayesian analysis has the advantage of "borrowing" information from previous similar studies to inform current research. This information is incorporated in the form of a prior probability distribution for a parameter which is being estimated (e.g., detection rates). Prior probability distributions are particularly relevant in data-limited situations where data is not informative or lacking. Moreover, well-informed priors help to reduce uncertainty in estimated parameters as in the case of building a LCM, in which the model parameters usually have high uncertainty (e.g., survival rates).

Data manipulation and Statistical analysis

We obtained tag detection data from the PIT Tag Information System (PTAGIS; http://www.ptagis.org/). PIT-tagged juvenile fish were detected at Juvenile Fish Bypass Facility (SUJ) and the Estuary Towed Array (TWX). Adult fish were detected at Willamette Falls Fishway (WFF). Specific locations (e.g., WFF) can be found through the PTAGIS website (e.g., see Figure C.1). The raw data included detailed information for each PIT-tagged fish (tag code, release site and date, detection site and date, release size, etc.). For the CJS model, each individual PIT-tagged fish was tabulated according to its "detection history" from its release site (e.g., Dam-tailrace) using a code of 1 and 0 (e.g., see Cooch and White 2013). Then, we combined the individual records to calculate the total number of PIT-tagged fish arriving at each location tabulated as an observed frequency for each possible detection history.

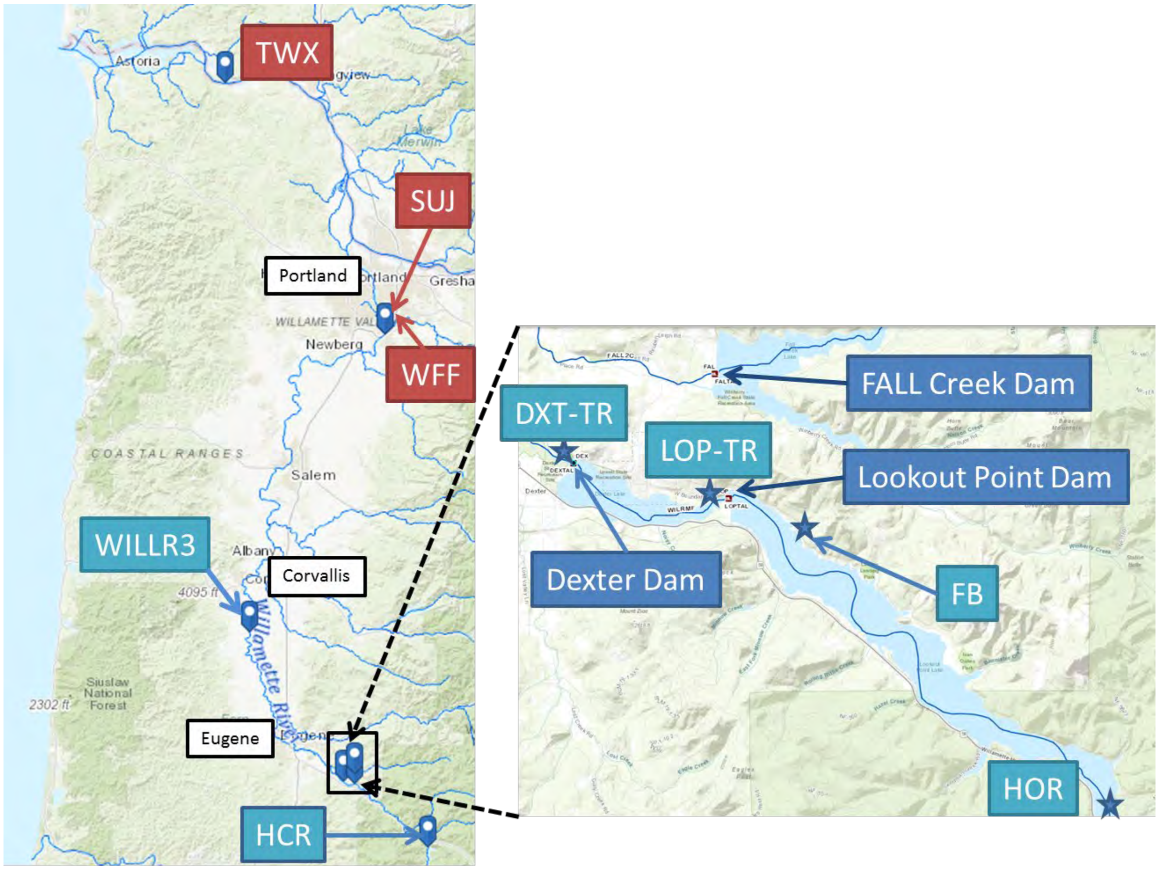

Figure 8-86. Dams and reservoirs in the Middle Fork Willamette River. Stars indicate approximate release sites for juvenile hatchery-raised Chinook salmon during 2011-2014: Lookout Point head of reservoir (LOP-HOR), Lookout Point Dam forebay (LOP-FB), Lookout Point Dam tailrace (LOP-TR), Dexter Dam tailrace (DXT-TR), Fall Creek reservoir (FCR), Hills Creek Reservoir (HCR).

The detection sites were the Sullivan Juvenile Fish Bypass Facility (SUJ), Estuary Towed Array (TWX), and Willamette Falls Fishway (WFF). The release site for juvenile natural-origin Chinook salmon was the Willamette River (WILLR3) from 2014 to 2016. Source (https://psmfc.maps.arcgis.com/apps/webappviewer/index.html)

For each of the six migration groups, the chinook salmon LCM requires time/season specific survival rates (e.g., freshwater survival during spring, fall, next spring) but there was limited PTAGIS data with which to estimate these group-specific survival rates. Paired release experiments conducted in the Willamette River basin are not designed to develop LCM parameters but to test hypotheses about migration and survival through dams, and/or survival in the reservoir dam survival (e.g., Brandt et al. 2016). In the case of the fry movers group, for example, these fish are too small to be PIT-tagged (Schroeder et al. 2016). Similarly, for Middle Fork, there are no PTAGIS data with which to estimate survival rates for the subyearling migration group. This is because only hatchery raised fish are PIT-tagged, and the data are concentrated during the spring release period, inconsistent with migration timing of natural origin subyearlings. Therefore, it was not possible to derive direct estimates of each migrant group's survival rate, and the LCM did not apply specific freshwater survival rate estimates (i.e., S1, see below) for each migration group. Instead, the LCM used adjusted CJS estimates for subyearling migrants (when available) scaled by relative values from Zabel et al. (2015) (see 2.2.6 for derive the survival rates for fry, φ_F^{rss}). Note that the LCM did not use from the CJS model estimation. Instead, these s were $\varphi_{0^+\to 3}^{o}$ and $\varphi_{1^+\to 3}^{o}$) in the calibration model (details in Section 2.5 and Appendix D).

We used the minimum configuration for the CJS : one release site (i.e., below-dam) and two detection sites (i.e., SUJ and WFF; represented as a diagram in Figure C.2). This model had four parameters (assuming survival and detection rates are different for each location). The observed detection history for a single-release detection model followed a multinomial probability distribution. The statistical likelihood is the product of the observed detection histories (i.e., 111, 110, 101, 100) and the expected probabilities for each possible detection history. Here, the detection history 111 indicates cases where fish were detected at both detection sites after release; 110, detected at only site 1; 101, detected at site 2; or 100, released and not detected again. Expected detection histories are determined by the parameter vector, ɵ = $\{S_1, P_2, S_2, P_3\}$. The log-likelihood can be written as (Cooch and White 2013):

$$\ln \boldsymbol{L}(\boldsymbol{S_1}, \boldsymbol{P_2}, \boldsymbol{S_2}, \boldsymbol{P_3}) =$$

Equation C-19

$$n_{111} \ln(S_1 P_2 S_2 P_3) +$$
$$n_{110} \ln(S_1 P_2 [1 - S_2 P_3]) +$$
$$n_{101} \ln(S_1 [1 - P_2] S_2 P_3) +$$
$$n_{100} \ln(1 - S_1 P_2 - S_1 [1 - P_2] S_2 P_3)$$

where $n_{111}, .., n_{100}$ are the observed frequency of PIT-tagged fish that have the expected probability detection histories of $111, .., 100$, respectively. S1 is the probability that a fish tagged in release location 1 will survive until it just reaches detection site 2, P2 is the probability that a fish will be detected in detection site 2, given the fish is alive, and so on for each detection site. Note that for the last survival and detection parameters, S2 and P3, the parameters are not identifiable (i.e., they cannot be estimated separately, but the product S2P3=λ can be estimated). Typically, most CJS studies estimate the product of these two parameters but not the unique values because these parameters are not identifiable (Lebreton et al. 1992; Cooch and White 2013; Powell and Gale 2015). However, because PIT-tagged adult fish arriving through the fish ladder facility at WFF are believed to have a probability of detection close to 100% (Pease et al. 2020), this information was included to inform a prior distribution for detection site 3, e.g., p(P3). Under this assumption, S2 and P3 become identifiable and it was possible to estimate S2. For S1, P2, and S2, we assigned reasonable prior probability distributions based on previous studies (see Appendix H for details). Prior distributions for S1 and S2 were derived from wild PIT-tagged juvenile Chinook salmon in the Willamette River basin (Appendix H for details). P2 depended upon flow (Schroeder et al. 2016), so, it was necessary to develop separate prior distributions for each water year type, given the influence of water type on detection rate (see Appendix H for details).

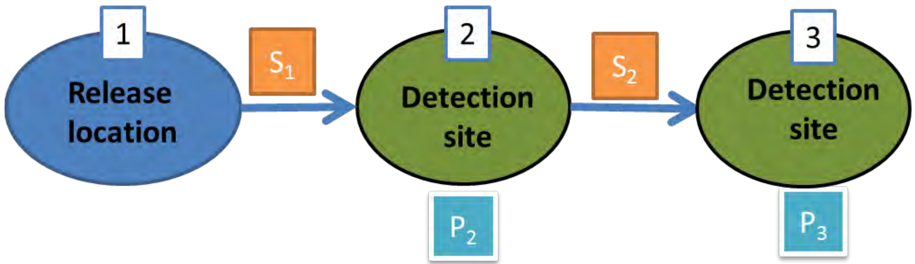

Figure 8-87. Diagram for a Cormack-Jolly-Seber (CJS) model with three sampling locations. S1, S2, P2, and P3 are the model parameters.

In this diagram, 1, 2, and 3 indicate the three sampling events; a site where the single release occurred and two detection sites. S{1,2} and P{2,3} are the survival and detection rate parameters, respectively.

To obtain posterior distributions for the model parameters ɵ, we used a random walk Metropolis algorithm with a multivariate normal proposal distribution implemented in R (R Core Team 2021), using the function "MCMCmetrop1R" in the "MCMCpack" package (Martin et al. 2011). The algorithm MCMC was initialized at the MLE for the model parameters and ran for four million iterations. The first two million iterations were discarded, after that, each of every 100 iterations were stored for further analysis. Convergence was evaluated by visually inspecting trace plots and posterior distributions to ensure that the posterior distributions for the model parameters were unimodal. Also, the model was tested at different initial starting values.

We tested the CJS model under simulated data. Overall, the model "recovered" the true values for the survival and detection rates (a simulation-estimation experiment is described in detail in Appendix J). Additionally, we tested a state-space model in JAGS (Plummer 2003) and WinBUGS (Spiegelhalter et al. 2007). After testing, the state-space model produced similar estimates to the multinomial model but was prohibitively slow for large releases (e.g., >100,000, see below). We opted for the multinomial model for simplicity; results of the JAGS/WinBUGS model are not shown here.

Parameter estimation, model outputs, and diagnostics

Freshwater survival for fry the Middle Fork Willamette River (φ_F^{rss}) was derived from paired experiments for hatchery fish released above and below Dexter dam (Table C.1). Due to few detections at SUJ, detection data from years 2011 to 2014 (inclusive) were combined, using detections between May and September to mimic migration timing for fry movers. The detection histories were: n100= 120,999; n110=4,894; n101=13; and n111=4. The following prior distributions were incorporated in the objective function as:

$$\boldsymbol{p}(\theta) = \boldsymbol{Beta}(\boldsymbol{S_2}|\boldsymbol{2.44, 3.66}) + \boldsymbol{Beta}(\boldsymbol{S_3}|\boldsymbol{3.19, 277.7}) + \boldsymbol{Beta}(\boldsymbol{P_2}|\boldsymbol{0.45, 2.5})$$
Equation C-20
$$+Beta(P_2|191.1,3.9)$$

CJS model parameters (θ) converged to their posterior estimates (Figure C.3 and Figure C.4). The mean posterior estimate for φ_F^{rss} was 0.192 (median=0.079), and the parameter estimates had considerable variability (CV=0.413, Table C.2). Note that the prior distribution for the WFF detection, pP_2), allows us to separate the SAS from the detection rate parameter (P3φ_F^{rss} estimate did not account for tagging loss, tagging induced mortality rates, or the effect of hatchery origin. Rather, survival rate estimates were later adjusted to account for these effects (see Appendix I).

Table 8-67. Release location/year and month of juvenile hatchery Chinook salmon arriving at SUJ in the Middle Fork Willamette River

Release location/year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
FCR_2013										3	136	11
DXT_TR_2011						4	476	22	1			
DXT_TR_2012	1	2	8	1		158	2338	150	30	43	2	
DXT_TR_2013		2	1			22	1210	44	32	6	2	1
DXT_TR_2014	1	4	4			26	359	17	9	36	4	
LOP_FB_2013	59	6				2	143	104	79	9	40	25
LOP_HOR_2011						5	174	6	10	5		1
LOP_HOR_2012	1	1		3	5	21	281	133	49	35	2	1
LOP_HOR_2013	42	5	1		3	1	124	80	47	8	50	22
HCR_2012	1			1	8						1	4
HCR_2013	2	1			8	1						1

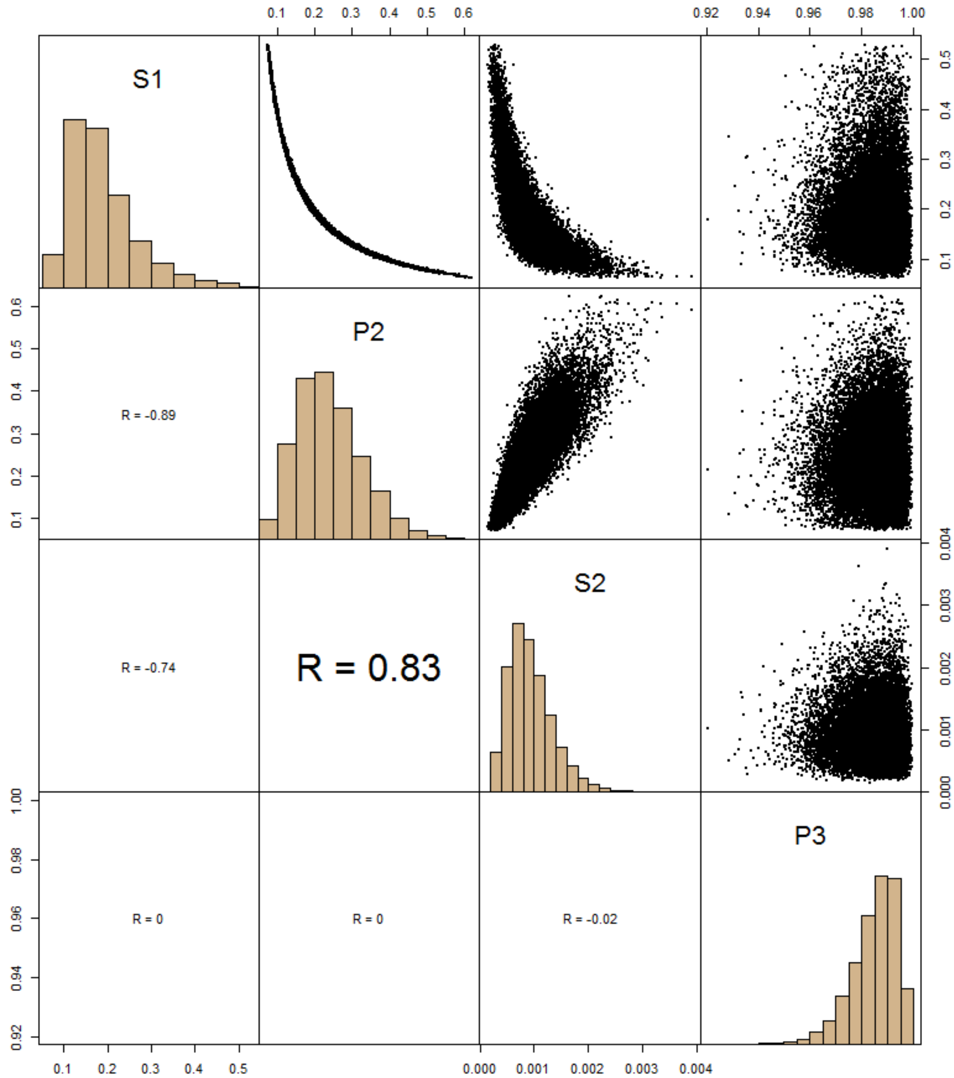

Figure 8-88. Posterior survival and detection rate distributions for juvenile Chinook salmon released at Dexter tailrace, detected at SUJ during May-September as juveniles, and later detected as adults at WFF in the Middle Fork Willamette River.

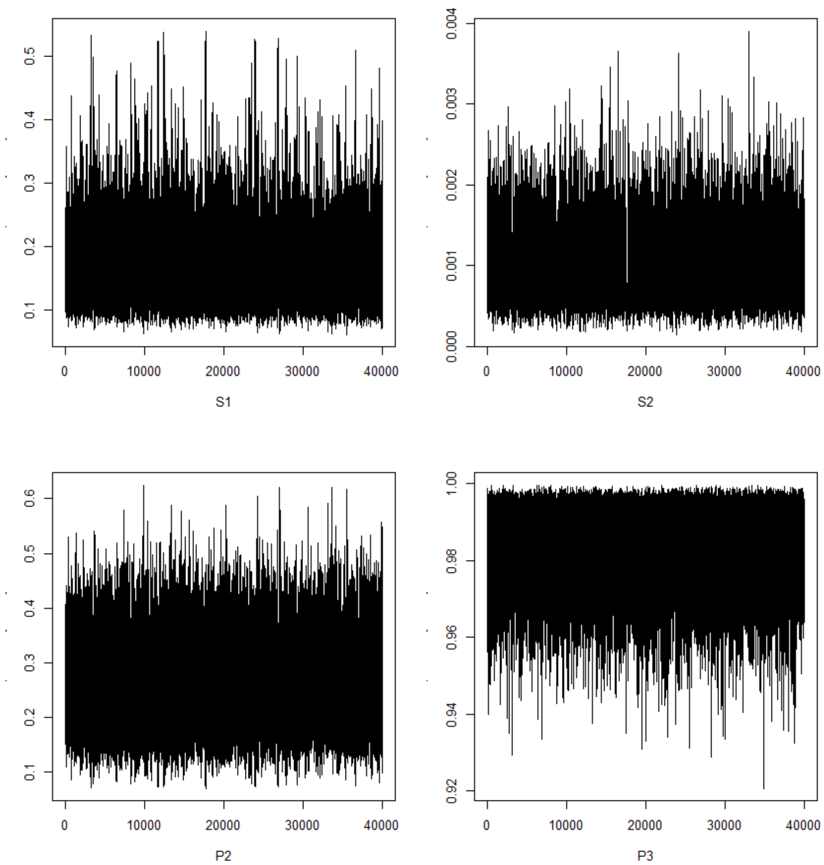

Figure 8-89. MCMC traceplots for posterior survival (S1 and S2) and detection rates (P2 and P3) for juvenile Chinook salmon released at Dexter tailrace in the Middle Fork Willamette River. Then, the juveniles were detected at SUJ (during May-September) and the adults at WFF.

Table 8-68. Posterior survival (S1 and S2) and detection rates (P2 and P3) for juvenile Chinook salmon released at Dexter tailrace in the Middle Fork Willamette River, detected at SUJ during May-September as juvenile and as adults at WFF. LB and UB are the 0.05 and 0.95 quantiles of the probability distribution.

parameter	mean	SD	CV	median	LB	UB
S1 (φ_F^{rss})	0.192	0.079	0.413	0.173	0.098	0.354
P2	0.235	0.088	0.374	0.225	0.110	0.394
S2	0.00092	0.00041	0.44143	0.00086	0.00039	0.00168
P3	0.985	0.009	0.009	0.986	0.968	0.996

Reparameterization of CJS survival rate components for use in the life cycle model

Parameterization of the LCM's at-sea component for Chinook salmon was done in four stages. In the first three stages, we approximated initial values for LCM parameters for upper Willamette Chinook salmon stocks without specificity to each sub-basin. In the fourth stage, the initial year at sea natural mortality rate parameters and fraction maturing at age were freed up from their initial parameter values to fit to data for each sub-basin.

Stage 1 of LCM parameterization: obtaining initial parameter values for the at sea LCM component

In the first stage, a set of parameter values for harvest rates (U in Equations 2.3-1 to 2.3-3) was obtained based on Chinook Technical Committee stock assessment results (CTC 2021). Based on the total number of CWTs accounted for in each year, the CTC (2021) provided estimates of average fraction of accounted-for CTWs that were 1) harvested in sea and terminal fisheries, or 2) escaping and aggregated for ages 3-6 for brood years 2009-2018. As the CTC results were based on release and recapture records of CWTs from several different hatcheries in the Upper Willamette, these did not account for potential significant differences in marine survival rates for spring Chinook stocks from the different sub-basins. In addition, the CTC survival rate estimates for age 0 to age 3 survival rate describe survival from release points at hatcheries to age 3 fish at sea, and thus included a downstream freshwater survival component in addition to marine survival. In contrast, the BCJS estimates of marine survival rates estimated the juvenile downstream survival from release points to SUJ and the smolt-adult survival rates separately.

The information in Appendix Table A70 of CTC (2021) (Table D.1) was first translated into fraction surviving from harvest at sea in each year. Table A70 provided the sum of tags accounted for in each year (Ty) and the percentage of these tags that were captured in the different fish fisheries and that were tallied in the escapement. The harvest rate at sea, Usea,y, was computed by dividing the sum of the tags caught at sea Cy,sea by Ty. The abundance Nsea,y after harvest at sea was computed as Ty(1- Usea,y). The harvest rate from terminal net harvest (UTN,y) was computed by dividing the tags caught in the terminal net fishery, CTN,y, by Nsea,y. The abundance after terminal net harvest NTN,y was computed from Nsea,y (1- UTN,y). The harvest rate from terminal recreational harvest (UTR,y) was computed by dividing the tags caught in the terminal net fishery CTR,y by NTN,y. The abundance after terminal recreational harvest NRN,y was computed from NTN,y (1- UTR,y). The total terminal harvest rate on hatchery Chinook salmon was computed from (CTN,y + CTR,y)/ Nsea,y. The incidental mortality from catch and release of wild salmon in net fisheries, Dy, was approximated with the calculation, IN * CTN,y where IN, the incidental mortality rate of Chinook salmon caught in a terminal net fishery, was obtained from ODFW (2009). The fraction of wild fish dying from incidental mortality in net fisheries was given by Dy/ Nsea,y. The predicted number of wild fish escaping from terminal net fisheries, NTWN,y, was approximated by Nsea,y (1- Dy/ Nsea,y). The incidental mortality Ry from catch and release of wild salmon in terminal recreational fisheries was approximated from IR * CTR,y where IR , the incidental mortality rate of Chinook salmon released after capture from recreational gear was obtained from (ODFW 2009). The fraction of wild fish dying from incidental mortality in terminal recreational fisheries was approximated by Ry/ NTWN,y. The predicted number of wild fish escaping from terminal recreational fisheries was given by NTWN,y (1- Ry/ NTWN,y). The fraction of wild fish dying from incidental mortality in terminal fisheries was obtained from (Ry + Dy)/ Nsea,y. See Table D.2 for approximations of 1) abundance before and after harvest, and 2) harvest rates in the at sea, and terminal fisheries for hatchery Chinook salmon. See Table D.3 for approximations of incidental mortality rates terminal fisheries for hatchery Chinook salmon.

Table 8-69. Records of tags accounted for, ages, catch in at sea fisheries, terminal net fishery catch, terminal recreational fishery catch, and escapement by brood year from Table A70 in CTC (2021) for Upper Willamette Hatchery Chinook salmon.

Brood Year	Total number of tags accounted for (Ty)	Ages	Sea Catch CTC estimated (Usea,y)	Terminal Net Catch CTC estimated (CTN,y)	Terminal Sport Catch CTC estimated (CTR,y)	Total catch CTC (Ty)	CTC Surviving (Escapement) (Ey)
2009	3845	3,4,5,6	323	311	777	1411	2434
2010	11269	3,4,5,6	755	428	3662	4846	6423
2011	7649	3,4,5,6	627	398	3190	4215	3434
2012	5869	3,4,5,6	810	288	2189	3287	2582
2013	6289	3,4,5,6	421	283	1874	2578	3711
2014	14285	3,4,5,6	1628	429	3114	5171	9114
2015	16822	3,4,5,6	2103	824	4306	7233	9589
2016	5819	3,4,5,6	1315	250	1315	2880	2939
2017	6455	3,4,5,6	800	161	1485	2446	4009
2018	5430	3,4,5,6	353	98	1330	1781	3649

Table 8-70. Approximations of accounted for tagged fish and harvest rates in at sea fisheries, terminal net fishery catch, terminal recreational fishery catch, and escapement by brood year from Table A70 in CTC (2021) for Upper Willamette Hatchery Chinook salmon.

Brood Year	Abundance after natural mortality	Harvest rate at sea	Abundance after harvest at sea	Terminal net harvest rate	Hatchery fish escaping terminal net fishery	Terminal sports harvest rate	Terminal U (combined)
2009	3845	0.084	3522	0.088	3211	0.242	0.309
2010	11269	0.067	10514	0.041	10086	0.363	0.389
2011	7649	0.082	7022	0.057	6624	0.482	0.511
2012	5869	0.138	5059	0.057	4771	0.459	0.490
2013	6289	0.067	5868	0.048	5585	0.336	0.368
2014	14285	0.114	12657	0.034	12228	0.255	0.280
2015	16822	0.125	14719	0.056	13895	0.310	0.349
2016	5819	0.226	4504	0.056	4254	0.309	0.348

2017	6455	0.124	5655	0.029	5493	0.270	0.291
2018	5430	0.065	5077	0.019	4979	0.267	0.281

Table 8-71. Approximations of Incidental mortalities on wild Chinook salmon caught in terminal net and recreational fisheries, assuming an average incidental mortality rate of 0.293 for net caught and released fish and 0.122 for recreationally caught and released fish using tag abundance values from Table D.2 (Source: ODFW 2009).

Brood Year	Incidental mortalities on wild in terminal net fisheries	Incidental mortality of wild fish dying in terminal net fisheries	Wild fish escaping alive terminal net fisheries	Number of incidental mortalities of wild fish in terminal sport fishery	Incidental mortality of wild fish in terminal sport	Total wild fish dying in terminal fisheries	Incidental mortality on wild fish killed in terminal fisheries
2009	91	0.0259	3431	95	0.028	186	0.053
2010	125	0.0119	10389	447	0.043	572	0.054
2011	116	0.0166	6905	389	0.056	505	0.072
2012	84	0.0166	4975	267	0.054	351	0.069
2013	83	0.0141	5785	229	0.040	311	0.053
2014	125	0.0099	12531	380	0.030	505	0.040
2015	241	0.0164	14478	525	0.036	766	0.052
2016	73	0.0162	4431	160	0.036	234	0.052
2017	47	0.0083	5607	181	0.032	228	0.040
2018	29	0.0056	5048	162	0.032	191	0.038

Stage 2 of LCM parameterization: refining parameter values for the at sea LCM component

The aim in this second stage was to formulate parameter values based on available information obtained from CTC (2021), our BCJS estimates of smolt-adult survival rates for upper Willamette Chinook salmon stocks and other sources that could serve as initial place-holder values for the parameters in Equations 2.3-1 to 2.3-3. The CTC had provided (S. Hawkshaw, DFO pers. comm.) assumed values for survival rate from natural mortality for Upper Willamette Chinook salmon in their 4rd-6th ocean years of 0.3969, 0.5000, and 0.6300. There were taken as initial values for the parameter S in Equations 2.3-1 to 2.3-3. Initial values for proportion maturing at age for fish in their 3rd -6th years were derived from Zabel et al. (2015) (i.e., 0.005, 0.4, 0.97 and 1, respectively) and it was assumed that all fish in their 6th year matured. Approximations of average annual harvest rates, U, at age for fish either staying at sea or in terminal fisheries that were obtained stage 1 are show in Table D.4.

We sought also to derive an initial value for M for fish in their first sea year since CTC stock assessments provided only an estimate of release to age 3 survival that included also the juvenile freshwater (FW) survival component. For yearling smolts, we started with the CTC (2021) estimate of the average release to age 3 survival rate, SCTC,0-3, of 0.0286. As noted above, this value included downstream survival from the point of release. To obtain a value with the FW juvenile component removed, we therefore divided this survival rate by our CJS estimate of downstream survival rates for PIT tagged juvenile Chinook salmon in the Upper Willamette that were released from November to January, averaged from PIT tag studies in the four sub-basins. The estimate of the average CJS estimated survival rate from release to SUJ for PIT tagged hatchery produced Chinook salmon, SCJS,0-SUJ, was 0.654. We applied a literature-based estimate of the ratio of survival rates of natural origin to hatchery origin juvenile chinook salmon of 1.379 (Appendix I) to adjust the hatchery-based estimate to one that could represent a survival rate for natural origin fish, ANOR/HOR. The CTC-based approximation of average survival rate, for example yearling smolt to age 3 survival, was thus obtained from Ssmolt,1.5 = SCTC,0-3 / SCJS,0-SUJ * ANOR/HOR (Table D.4).

Table 8-72. Initial placeholder parameter values for LCM parameters in Equations 2.3-1 to 2.3-3. Values for ages 4-6 in the 2nd column are values that have been applied in CTC stock assessments. See text for how the initial value for smolt to age 4 survival rates was derived. Values in the 3rd column were obtained from fitting to available spawner age composition records from the Upper Willamette watershed (Sharpe et al. 2017). Values in the two columns on the right were averaged from estimates from 2009-2018 obtained from stage 1 above and CTC (2021) and shown in Table D.2 and Table D.3.

Ending Age (at spawning)	Initial S at age used by the CTC	Proportion returning to spawn at beginning of year	Proportion of fish staying at sea that were harvested in the year, 0.109 comes from CWT studies for hatchery salmon CTC (2021)	Proportion of fish that matured that died from fishing mortality when going from the sea to WFF in the year: CTC (2021)
Age 3	0.0603	0.0374	0.0545	0.0520
Age 4	0.3969	0.6482	0.1090	0.0520
Age 5	0.5000	0.9338	0.1090	0.0520
Age 6	0.6300	1.0000	0.1090	0.0520

Using the initial approximation of Ssmolt,1.5 and other assembled parameter estimates, the yearling smolt to age 4 adult survival rate (SASCTC,1.5-4) based on CTC derived inputs could be computed from Equations 2.3-1 to 2.3-3:

SASCTC,1.5-4 = (1-P2) * SCTC,smolt,1.5 * (1- U2,sea) P3 * S3,T * (1- U3,T)

It could be expected that the value computed for SAS1.5-4 based on the CTC (2021) derived parameter values would be different from the averaged CJS estimate of SAS2-4. This could be due to the CJS estimate using a smaller subset of years than the CTC-based estimate, for example, among other things. It was of interest, however, to solve for a value of Ssmolt that is consistent with the averaged CJS estimate of SAS2-4. This was obtained by applying the ratio:

SCJS,smolt,1.5 = SCTC,smolt,1.5 * SASCJS,1.5-4 / SASCTC,1.5-4

Thus, as a starting point, the parameter values for the LCM Equations 2.3-1 to 2.3-3 used CTC derived inputs and were adjusted to become more consistent with the average of SAS estimates obtained from our CJS estimations. The values for PA, however, were obtained from Zabel et al. (2015), when in fact records of spawner age composition were available from the four Upper Willamette sub-basins. See Table D.4 for example input values for the calculation of SAS based on CTC inputs.

Stage 3 of LCM parameterization: initial fitting to averaged spawner age composition records

In the third stage of refining initial parameter values, we freed up proportion spawning at age parameters PA and used Equations 2.3-1 to 2.3-3 to estimate survival at sea, fitting to averaged spawner age composition records for natural origin spring chinook salmon from the four Upper Willamette sub-basins.

The long-term average spawner composition GA for A from 3 to 6 years was computed using the following equations (for e.g., yearling smolts):

F3 = P2 * Ssmolt, 1.5 * (1- U2,T) * (1-PSM)

F4 = Ssmolt, 1.5 * (1- U2,sea) * (1-P2) * P3 * (1-U3,T) * (1-PSM)

F5 = Ssmolt, 1.5 * (1- U2,sea) * (1-P2) *(1- U3,sea) * (1-P3) * P4 * (1-U4,T) * (1-PSM)

F6 = Ssmolt, 1.5 * (1- U2,sea) * (1-P2) *(1- U3,sea) * (1-P3) *(1- U4,sea) * (1-P4)* P5 * (1-U5,T) * (1-PSM)

GA = FA / (F3+F4+F5+F6)

As PSM was treated as a constant in the above equations, the term with it was left out of the calculations. The objective function that was minimized to fit the PA parameters was:

$$obj = 1000 * \sum_{A=3}^{6} \left(\ln\left(G_A^{obs}\right) - \ln\left(G_A^{pred}\right)\right)^2$$

where obs denotes observed, and pred denotes predicted. Values of parameters obtained from Stages 1-3 computations with inputs from CTC (2021), Zabel et al. (2015) and our Bayesian CJS analysis of records from PIT tag studies in the Upper Willamette are shown in Table D.5. Observed age composition records for spawners above dams in the four Willamette sub-basins are shown in Table D.6.

See Section 2.5 for details on how the LCMs were calibrated to the time series of natural origin counts at dam tailraces and spawner age composition records from above dams in the four sub-basins.

Table 8-73. Example input values and results obtained from Stages 1-3 computations with inputs from CTC (2021), Zabel et al. (2015) and our Bayesian CJS analysis of records from PIT tag studies in the Upper Willamette are shown in Table D.5, also using an input value of 0.02 for SAS1.5-4 from CJS estimation. Table D.6 shows the averaged spawner age composition of wild Chinook salmon recorded in the four sub-basins of interest in 2015.

Starting Age based on September birth date	S from natural mortality adjusted using a scaling factor that adjusts initial stage survival so that the CJS SAS is achieved.	Adjusted SAS annual terms for computing #of fish remaining in the sea at the end of the year	Adjusted SAS annual terms for computing the number of fish that show up at WFF after migrating from the sea	Proportion returning to spawn at beginning of year fitted to observed spawner age compositions
Age 1.5	0.0901	0.0820	0.0032	0.0374
Age 3	0.3969	0.1244	0.2438	0.6482
Age 4	0.5000	0.0295	0.4426	0.9338
Age 5	0.6300	0.0000	0.5972	1.0000

Table 8-74. Observed counts of wild (W) and hatchery (H) spawners at age in the North Santiam (NSNT), South Santiam (SSNT), McKenzie (McK), Middle Fork (MFW) sub-basins in 2015. Source ODFW HRME Annual Report for 2015-2016 (Sharpe et al. 2017).

2015 Age	NSNT SGS W	NSNT SGS H	SSNT SGS W	SSNT SGS H	McK SGS W	McK SGS H	MFW SGS W	MFW SGS H	Sum of W counts	Proportion
3	7	0	28	0	3	0	7	1	45	0.0835
4	46	12	155	71	145	47	35	35	381	0.7069
5	13	40	18	10	72	20	5	17	108	0.2004
6	0	0	0	0	5	0	0	0	5	0.0093

Time series of steelhead marine survival rate

The steelhead IPA estimation model treats mean annual marine survival rate as a known parameter and estimates annual deviates from this. To fit the model to the Foster count data, we used a time series of marine survival from 1986 to 2019 (see Section 3.3). To our knowledge, apart from the Clackamas population in recent years (2015-2018), marine survival rate has not been estimated specifically for any Willamette winter steelhead populations. We constructed an index of marine survival using data from the literature for geographically local populations (Wilson et al. 2021; McCann et al. 2022), assuming that once steelhead smolts reach the estuary, then survival is similar, or at least correlated, among local populations. This assumption is supported by Kendall et al. (2017), who found smolt survival rates were more positively correlated for proximate populations, i.e., those whose river mouths were close together.

The population of wild steelhead closest to the Willamette River for which a relatively long time series of marine survival data was available is Wind River (Wilson et al. 2021). Wind River enters the Columbia River at river kilometre (rkm) 251, just upstream of Bonneville Dam (rkm 234), meaning it is only 71 rkm above the confluence of the Willamette with the Columbia at rkm 163. Wilson et al. (2021) used PIT-tag data and detections of smolts and adults at Bonneville Dam to estimate the smolt-adult survival rate for the years 2003-2014. We assumed this Wind River time series would be similar to smolt-adult survival rate for the Willamette population in these years.

A longer time series of PIT-tag data-based steelhead marine survival from 1993-2018 is available for the Snake River (Table B.130 & Figure 4.39 in McCann et al. 2022). Unlike Wind River, Snake River is not in the Lower Columbia, as it enters the Columbia at rkm 522, but the Snake River marine survival estimates were calculated as the survival from Bonneville Dam as smolts (i.e., on entry to the Columbia estuary) to return to Bonneville as adults, so are comparable. We examined the correlation between the Wind River and Snake River smolt-adult survival rates, which were positively correlated with an R2 of 0.667 (Figure E.1). Given the degree of correlation, we assumed that the annual variation would be similar between the populations and used the relationship (slope = 0.793, intercept = 0.013) to predict Wind River smolt-adult survival from the Snake River data for those years without observed data (1993-2002 and 2015-2018, Figure E.1). The absolute prediction error was 0.7%, though most values were <±0.5% (Figure E.1). For predictions outside of the Snake River data we used mean values from the neighbouring ranges, given recent understanding of regime changes in steelhead marine survival (Wilson et al. 2022). For 1986-1992, we used the mean from 1993-2009 (reflecting the “compensatory” regime identified by Wilson et al. 2022), and for 2019, we used the mean from 2010-2018 (reflecting the “declining” regime). This resulted in a Wind River marine survival time series from 1986-2019 comprised of predictions 1986-2002, observations 2003-2014, and predictions 2015-2019 (Figure E.2).

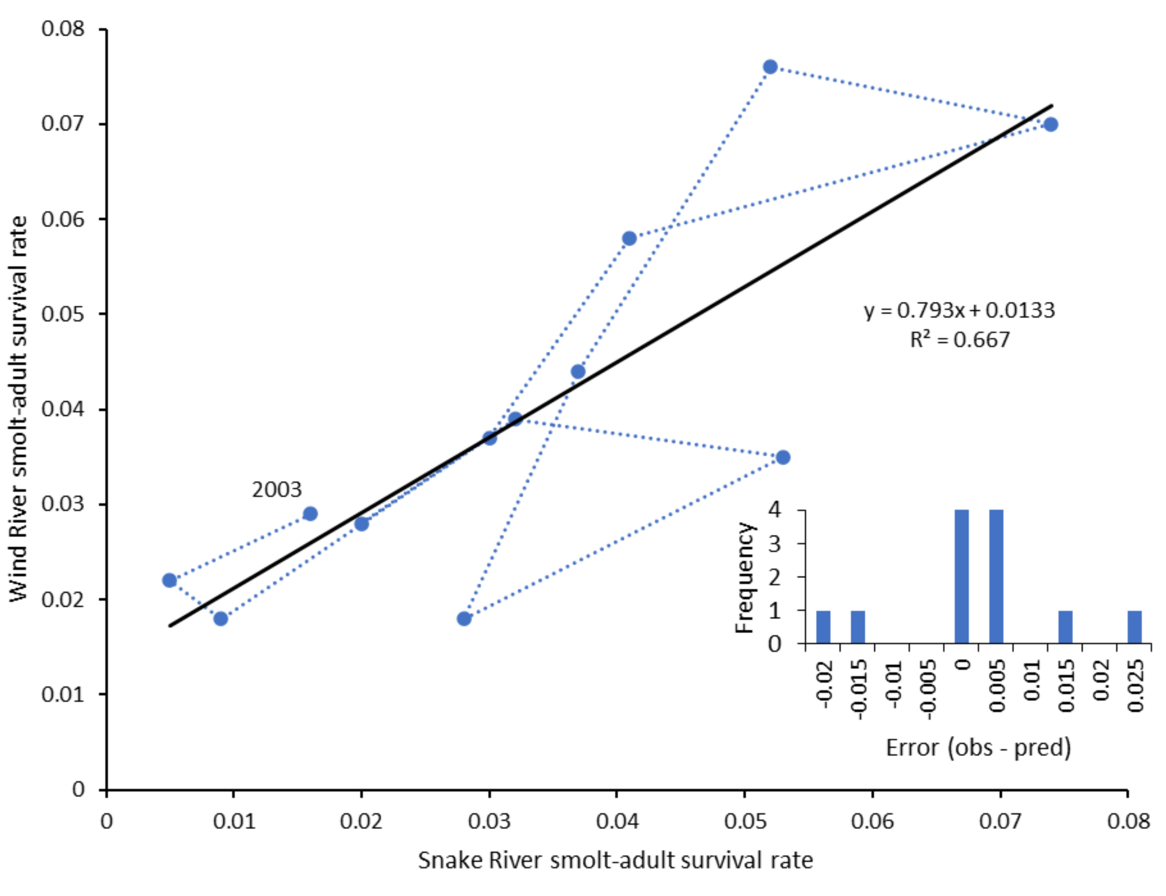

Figure 8-90. Relationship between estimates of steelhead smolt-adult survival rates for the Wind River and the Snake River populations, 2003-2014.

The blue dotted line shows the relationship over time from the first year (2003). Black line shows a linear model fit to the data with regression parameters and R2. Histogram (inset) shows the distribution of prediction error when this model was used to predict Wind River survival rates from the Snake River data.

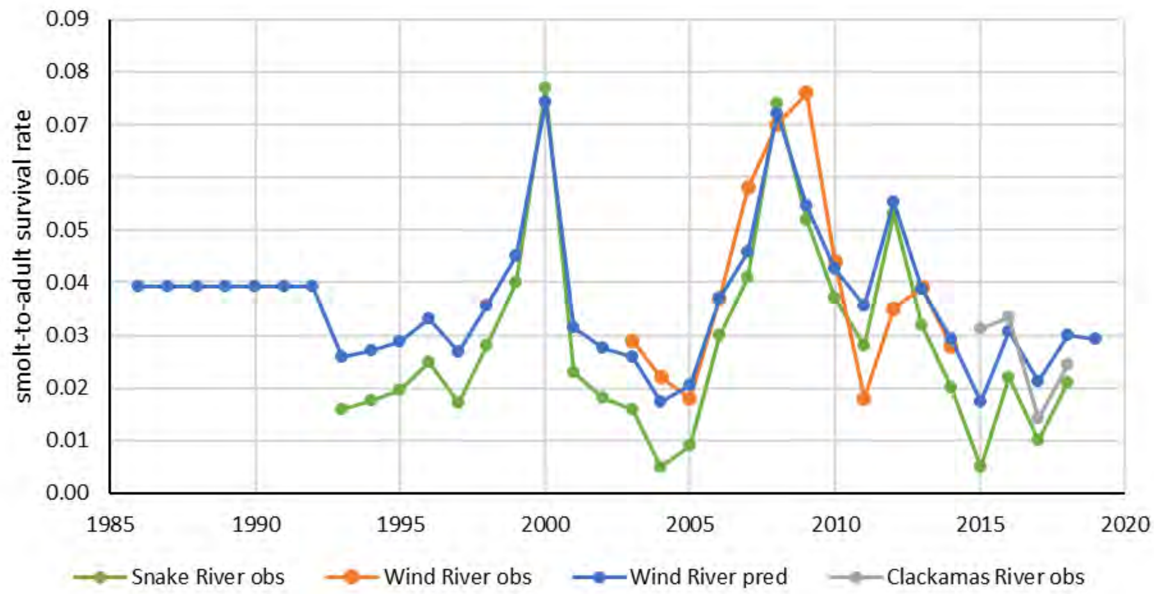

Figure 8-91. Time series of wild steelhead marine survival rate from 1986-2019 using observed data for the Wind River 2003-2014 and predictions from the relationship of these estimates with observed data from the Snake River 1993-2018.

Observed data from the Clackamas River are shown for 2015-2018.

The McCann et al. (2022) Snake River data were from analysis of wild summer steelhead, not wild late-winter steelhead. Also, the Wilson et al. (2021) Wind River data were comprised of both wild summer and wild winter steelhead, though were predominantly summer-run steelhead. We recognise that summer- and winter-run steelhead will likely have different smolt-adult survival rates given the relative timing of return to the river, though note that the difference between late-winter returns and summer returns may only be a couple of months given that time of return varies substantially among and within stocks of summer and winter steelhead (Hess et al. 2016; Copeland et al. 2017; McMillan et al. 2022). To ensure the scale of the predicted time series from summer-run data was relevant to winter-run steelhead, we examined the PIT-data based Clackamas River smolt-adult survival rate estimates for wild winter steelhead obtained from PGE (Garth Wyatt, pers. comm.). Migration timing of native Clackamas winter steelhead parallels those at Willamette Falls (Keefer and Caudill 2010), so survival should be comparable. The Clackamas values were within the range of the predicted time series for 2015-2018, and shared similar annual variation (Figure E.2). Therefore, this justifies our use of this constructed time series as a relative index for mean marine survival in our estimation model, as the estimated deviates will account for any unknown bias between the index and 'true' wild late-winter Willamette steelhead marine survival.

Modelling Lag 1 Autocorrelation in Marine Survival Rates for Winter Steelhead in the Upper Willamette River Basin

Estimates of smolt to adult marine survival rates for steelhead stocks in the Columbia River have shown time varying trends with apparent approximate decadal periodicity (McCann et al. 2022). Fairly well-pronounced covariation between steelhead stocks in annual patterns in estimates of marine survival rates of steelhead cohorts in stocks from different tributaries of the Columbia River has also been identified (McCann et al. 2022). This covariation suggests that marine survival rates of Upper Willamette steelhead stocks could be approximated by the average of estimated marine survival rates in Columbia River steelhead stocks. Given the apparent covariation between Columbia River steelhead stocks in estimates of smolt-adult, i.e., marine, survival rates, we thus assumed that the prior mean value for annual marine survival rates of Upper Willamette steelhead stocks could be approximated by the average of estimated marine survival rates from estimates available for some of the steelhead stocks in Columbia River (see Appendix E for the formulation of the prior mean values for marine survival rate for Upper Willamette winter steelhead stocks). Figure F.1 shows the time series of prior mean values for marine survival rate for Upper Willamette winter steelhead stocks.

Time series of adult female steelhead counts at the Foster Dam tailrace were available to fit a winter steelhead LCM (the full IPA LCM for steelhead is described in Section 3). The average annual freshwater survival rate of juvenile steelhead above the dam, Sfw, and initial abundance of adult female steelhead in 1991, Nf1991, were fitted to the data. In addition, annual deviates in smolt-adult marine survival rates from 1993 to 2019, $d_{y,S}$, were estimated but constrained by a prior distribution on the natural logarithm of each annual deviate (Equation F-1). The LCM predicted the annual abundance of female winter steelhead arriving at the Foster Dam tailrace starting in 1991 and running through to 2018 (see Figure 3.5.1). From 1991-2006 only total counts were available for adult steelhead at the Foster Tailrace. For 1991-2006, the average of the annual fraction of female adult steelhead was approximated using counts from 2007-2018 (see Section 3.2.3). Informative prior distributions were also formulated and applied for Sfw and Nf,1991 (see Appendix G for details). The sum of the logarithm of the priors for the freed up parameters was given by:

$$\boldsymbol{Logprior = constp - \frac{\left(log\left(\frac{S_{fw}}{\mu_{Sfw}}\right)\right)^2}{2\sigma^2_{Sfw}} - \frac{\left(log\left(\frac{N_{f1991,}}{\mu_{Nf1991}}\right)\right)^2}{2\sigma^2_{Nf1991w}} - \sum_{y=1993}^{2019}\frac{(d_{y,S}-0)^2}{2\sigma^2_{dS}}} \qquad \text{Equation F-21}$$

where $constp$ is the sum of constants in the prior density functions that remained constant, μ_{Sfw} and σ^2_{Sfw} are the prior median and prior variance for Sfw; μ_{Nf1991} and σ^2_{Nf1991} are the prior median and prior variance for the Nf1991, and σ^2_{dS} is the prior variance for $d_{y,S}$. σ^2_{dS} was assigned a value of 1.0.

A lognormal likelihood function was applied in fitting the LCM to the time series of adult female counts and approximated counts at the Foster tailrace.

$$\boldsymbol{LogLik = constL - \sum_{y=1991}^{2021} \frac{\left(\log(N_{y,fobs}) - log(N_{y,fpred})\right)^2}{2\sigma_{SHL}^2} - n * log(\sigma_{SHL})}$$

Equation F-22

where constL is a component of the lognormal likelihood function that remained constant in parameter estimation; $N_{y,fobs}$ and $N_{y,fpred}$ are the observed/approximated and model-predicted number of female adult steelhead at the Foster Dam tailrace in year y, respectively, n is the number of years, and σ_{SHL}^2 is the variance term in the lognormal likelihood function, with σ_{SHL} estimated at 0.244.

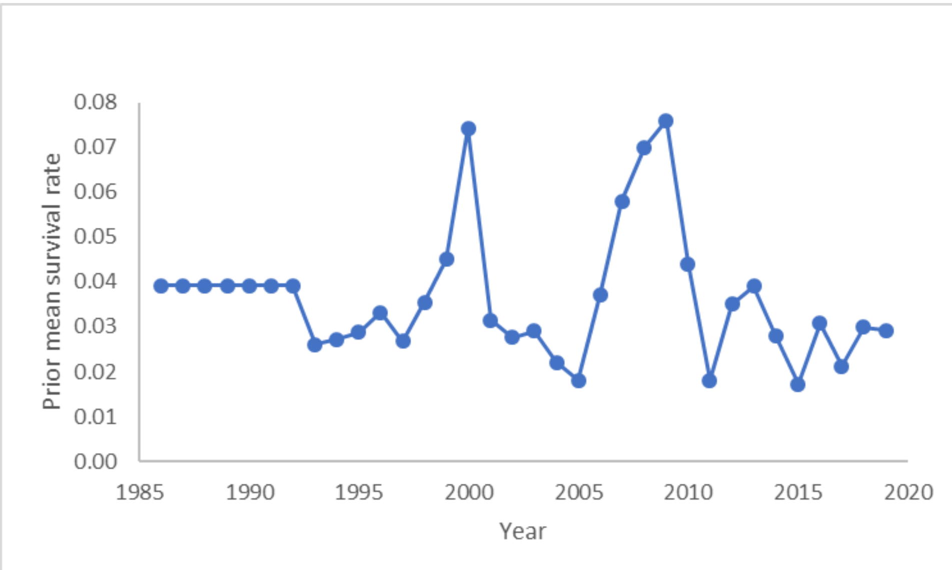

Figure 8-92. Time series of prior mean marine survival rates for upper Willamette River steelhead stocks based on estimates for other steelhead stocks in the Columbia River watershed (see Appendix E).

The estimated deviates from the average of Columbia River steelhead stock marine survival rate show a negative trend after about 2000 but an increase from 2005-2010, followed by lower values since then (Figure F.2). This suggests that since about 2000 marine survival rate in the Foster Reservoir population has declined as with other Columbia River steelhead stocks but it increased from 2015-2018.

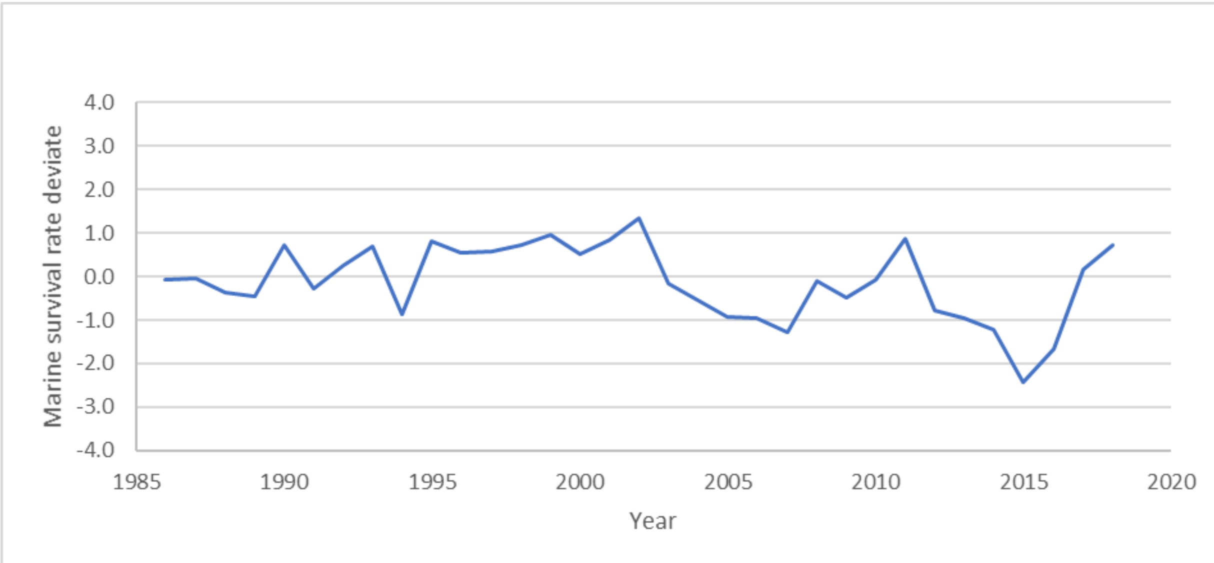

Figure 8-93. Estimated deviates between marine survival rate for the Foster Reservoir winter steelhead stock and the prior mean marine survival rate for upper Willamette River Basin Winter steelhead stocks (Figure F.1).

The estimated marine survival rate deviates, $d_{y,S}$, were applied to the average marine survival rates, μ_{MS}, to obtain a time series of estimated marine survival rates for the Foster Reservoir steelhead stock (Figure F.3). For additional details on the model fits to the adult female count data and other parameter estimates see Section 3.5.

The time series of estimated marine survival rates suggested periodic cycling with an apparent periodicity of about 10 years (Figure F.3). The lag 1 autocorrelation coefficient, , in the estimated marine survival rate was 0.601 (p-value = 0.0002). The long-term average of the estimated marine survival rate was 0.0423. The standard deviation in the natural logarithm of the estimates of marine survival rate, MS was 0.963.

For the winter steelhead IPA LCM projections under the different EIS alternatives for the North Santiam and South Santiam sub-basins, we needed to simulate future marine survival. We assumed that future marine survival rates had a and MS as estimated in the marine survival time series to be consistent with the historical time series of marine survival rate estimates (Figure F.3). Future marine survival rates were simulated as follows:

$$\boldsymbol{S_{M,y+1} = \mu_{MS} exp\left(\epsilon_y \rho + \delta_{y+1}\sqrt{1-\rho^2} - \frac{\sigma_S^2}{2}\right)}$$

Equation F-23

$$\boldsymbol{\epsilon_y = \epsilon_{y-1}\rho + \delta_y\sqrt{1-\rho^2}}$$ Equation F-24

$$\boldsymbol{\delta_y \sim Normal(0, \sigma_S^2)}$$ Equation F-25

Test simulations in Excel and R confirmed that the mean of the simulated marine survival rates was equal to μ_{MS} , the variance in the natural logarithm of the simulated marine survival rates was σ_S^2 , and the correlation coefficient between the simulated marine survival rates and survival rate at lag 1 year was equal to ▯.

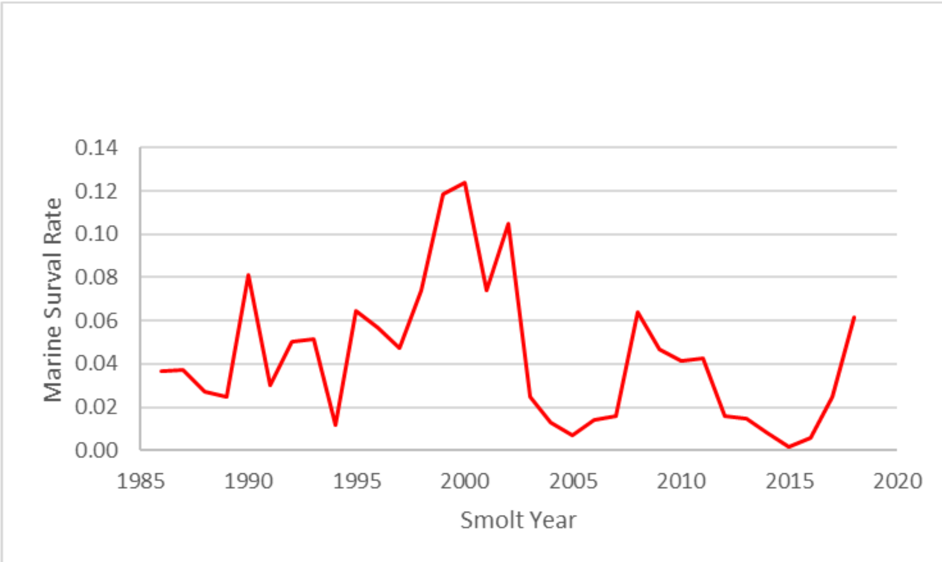

Figure 8-94. Time series of estimated marine survival rates for the Foster Reservoir steelhead stock

Prior for initial steelhead spawner abundance

Preliminary WS-IPA LCM estimation revealed that there was large negative correlation between the estimates of freshwater survival, ϕ_F, and the estimate of initial abundance of female spawners in 1991. This meant that the count data at Foster were equally well predicted by low initial abundance and high freshwater survival, or by high initial abundance and low freshwater survival. Some of the higher initial abundances with similar likelihood were beyond the historical range of the Foster counts, and so were not deemed credible. We therefore developed an informative prior to put on the initial abundance in 1991 to ensure that the estimate was credible and that a reliable estimate of freshwater survival would be obtained. The prior would have central tendency at the value most likely for 1991 abundance, but incorporate uncertainty related to the observed variability in abundance in the years around 1991.

There are various time series of winter steelhead in the South Santiam that are available or can be constructed for the period around 1991. ODFW & NMFS (2011, p13 of Appendix B) provide a time series of South Santiam spawner abundance, which together with the presented fraction of wild spawners can be used to determine wild spawners. Falcy (2017) estimated abundance in the South Santiam based on proportionally allocating Willamette Falls counts using redd surveys within each population of the steelhead Distinct Population Segment (Figure 3.1.2). Mapes (2017) estimated abundance from annual index surveys. Radio-telemetry studies by Jepson et al. (2013, 2014, 2015) estimated the escapement of radio-tagged adults to the South Santiam, the mean of these values can be used to generate a time series of South Santiam abundance from the Willamette Falls counts (Figure G.1).

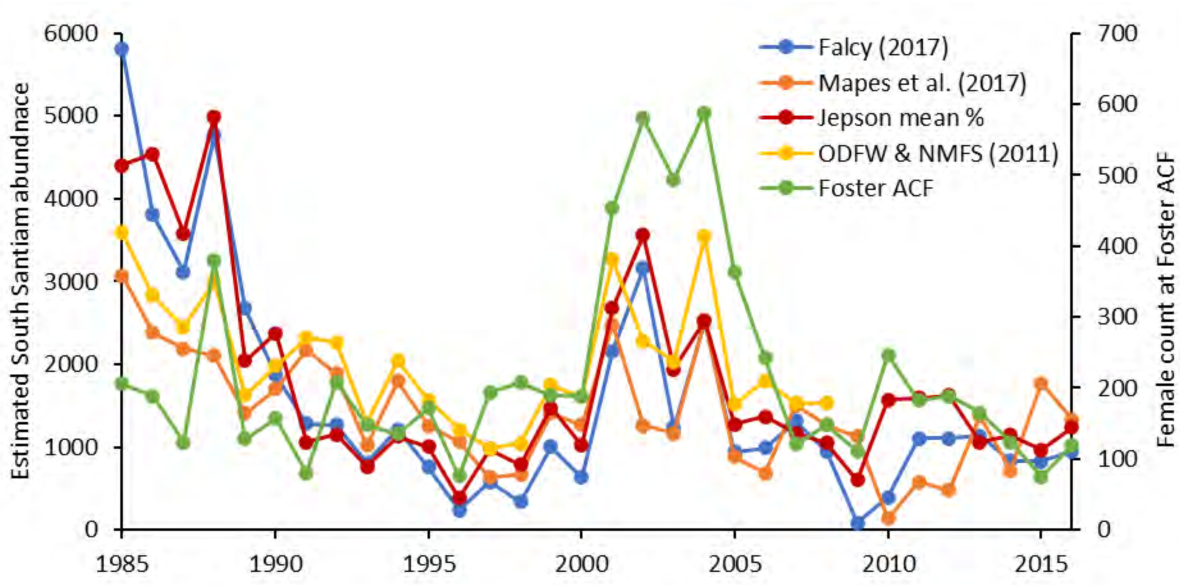

Figure 8-95. Time series of estimated South Santiam spawner abundance from various studies and female counts at Foster adult collection facility, 1985-2016.

See Figure 3.1.3 for details of the Foster count data.

These abundance time series all follow a similar pattern and highlight the annual variability in spawner abundance. However, using them to specify abundance of female spawners above Foster requires assumptions about both the sex ratio of the adults that return to Foster, and the proportion of returns to the South Santiam that return to Foster. We instead decided to develop the prior for initial abundance directly using the count data at Foster dam (Figure 3.1.3).

Due to the approximate decadal periodicity in marine survival rates, we chose to place a 10-year window around 1991 and estimated summary statistics for female spawner abundance in this period. For 1987-1996, the mean abundance was 161, median abundance was 142, and CV was 0.537. We used these values to specify a log-normally distributed prior with a mean equal to ln(median) and a standard deviation in log values equal to the CV (Figure G.2).

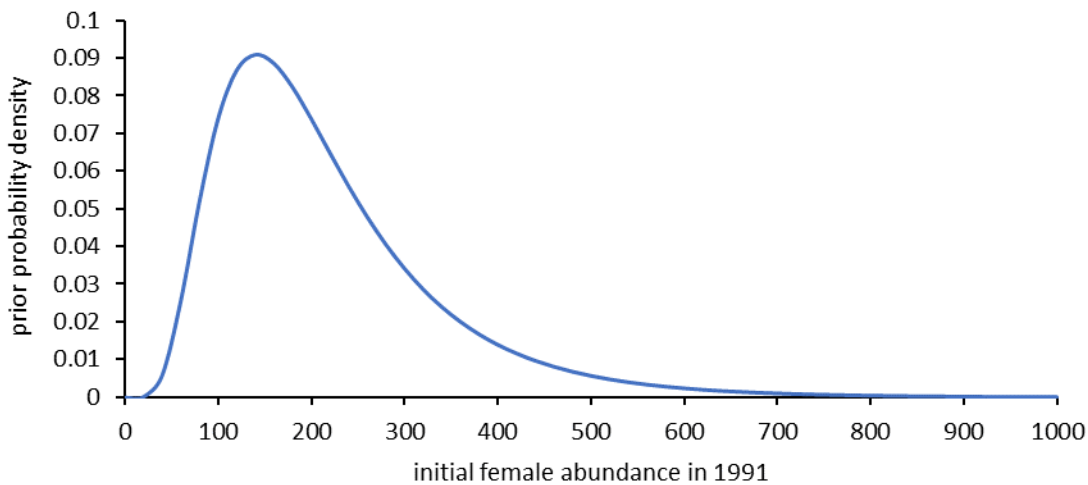

Figure 8-96. Prior probability distribution for initial abundance of female spawners at Foster in 1991.

Development of Bayesian priors for Cormack-Jolly-Seber model parameters

Apparent survival rates can be estimated from PIT tag data using the Cormack-Jolly-Seber (CJS) model (Lebreton et al. 1992; Cooch and White 2013), which models the survival rate between discrete release and detection locations by adjusting the numbers detected by the detection probability at each location. Using releases of PIT-tagged juvenile fish into dam tailraces and the PIT tag detection arrays at the Sullivan Dam juvenile bypass facility (SUJ) located at Willamette Falls and at the Willamette Falls adult ladder (WFF) (Figure C.1), river-smolt survival rate (ϕRSS) and smolt-adult survival rate (ϕSAS) can be estimated as the survival of juveniles from release to SUJ and the survival from SUJ to WFF, respectively. Unfortunately, data from PIT tag studies can be sparse, with few fish detected at some sites providing a challenge to understand whether low numbers of detections are due to low survival or low detection probability. Within a Bayesian modelling framework, uncertainty in model parameter estimates can be reduced by incorporating prior knowledge via informative prior probability distributions. Priors can be derived from expert knowledge, published data, or other analytical methods (McAllister et al. 2001, 2010; Porteus et al. 2019). To reduce uncertainty in CJS model parameter estimates, we developed informative priors specific to the Willamette for ϕRSS, ϕSAS, detection probability at SUJ (pSUJ), and detection probability at WFF (pWFF). These priors were applied in analyses of data from PIT tag studies in the North Santiam, McKenzie and Middle Fork Willamette sub-basins.

Prior for release-smolt survival probability

The ϕRSS of juvenile Chinook salmon migrating from dam tailraces to Willamette Falls is a parameter in the IPA model, with juveniles passing Willamette Falls assumed to be smolts. Estimates of ϕRSS in the sub-basins were obtained from CJS models of PIT-tagged fish released into the tailraces and detected at SUJ and used as downstream river-smolt survival in IPA models.

Various radio telemetry studies have been undertaken by USGS in the North Santiam to evaluate in-reservoir behaviour, dam passage, and downstream migration of juvenile Chinook salmon released in Detroit reservoir (Beeman and Adams 2015; Kock et al. 2015). Using data from radio telemetry arrays in reaches below Minto dam where the detection probability is assumed to be 1.0, it was possible to calculate survival probabilities from Minto dam to Portland. The distributions of these survival probabilities were used to develop a prior for ϕRSS, assuming that given the similar distance between them, survival between these two locations would be similar to that from below Big Cliff to Willamette Falls.

Results from three USGS releases of PIT-tagged juvenile Chinook salmon (Beeman and Adams 2015; Kock et al. 2015) were used to calculate the mean and 95% confidence interval (CI) for the survival probability of each release from Minto to Portland:

1) 2013 spring yearling release, mean survival probability = 0.388 [0.234, 0.546]

2) 2013 fall subyearling release, mean survival probability = 0.650 [0.124, 0.925]

3) 2014 fall subyearling release, mean survival probability = 0.236 [0.197, 0.275]

Beta distributions were fitted to the 95% CI using the 'beta.parms.from.quantiles' function in R (R Core Team 2021) (Figure H.1). As there was no consistent difference between estimates from the one year of spring yearling releases and those from the two years of fall subyearling releases, a beta distribution was fitted to all three empirical distributions combined to construct a prior distribution, ϕRSS ~ Beta(2.440, 3.665). The Beta prior distribution had a mean of 0.399 and CV of 0.46 (Figure H.1).

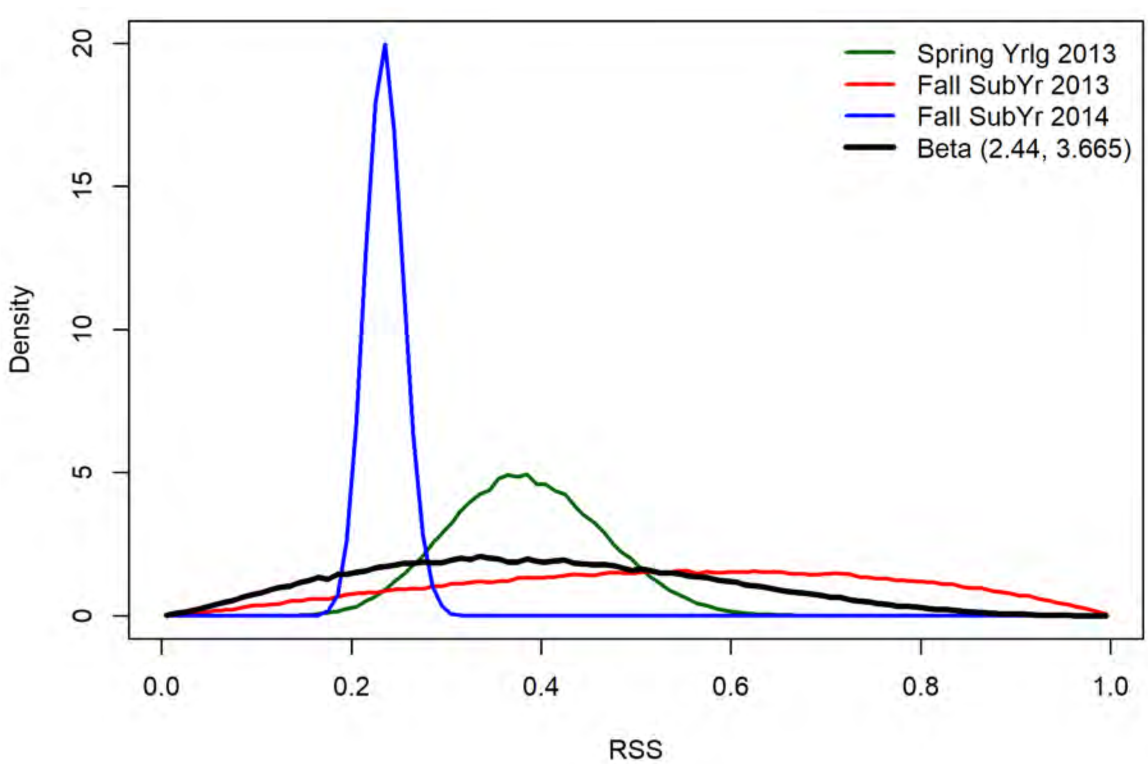

Figure 8-97. Empirical distribution of juvenile Chinook salmon release-to-smolt survival probability from three USGS telemetry studies conducted in the North Santiam in 2013 and 2014.

A Beta distribution (black line) was fitted to the empirical distributions to construct an informative prior for ϕRSS.

Prior for smolt-adult survival probability

Estimates of Chinook salmon smolt-adult survival (ϕSAS) from the analysis of PIT tag studies in each sub-basin fed into the IPA model parameters for early ocean survival via the reparameterization of CJS survival rate components and model calibration process (Section 2.5 and Appendix D). We used estimates of smolt-adult return rates (SAR) for the Willamette hatchery stock from 1977-2013 (Welch et al. 2021) to parameterize a prior distribution for ϕSAS. These estimates of SAR were based upon analysis of coded-wire tags obtained by Welch et al. (2021) from the Chinook Technical Committee of the Pacific Salmon Commission.

The empirical distribution of SAR estimates (Figure H.2) had a mean of 0.011 and a CV of 0.55. Although SAR estimates appear log-normally distributed, as they are a probability and thus bounded at 1, we assumed a Beta distribution to construct a prior for ϕSAS. This assumption fit well with the data as a Beta distribution fit to these estimates using the R 'fitdistrplus' package (Delignette-Muller and Dutang 2015) had the same mean (0.011) and CV (0.55) as the empirical distribution (Figure H.2). This resulted in a prior distribution, ϕSAS ~ Beta(3.193, 277.761).

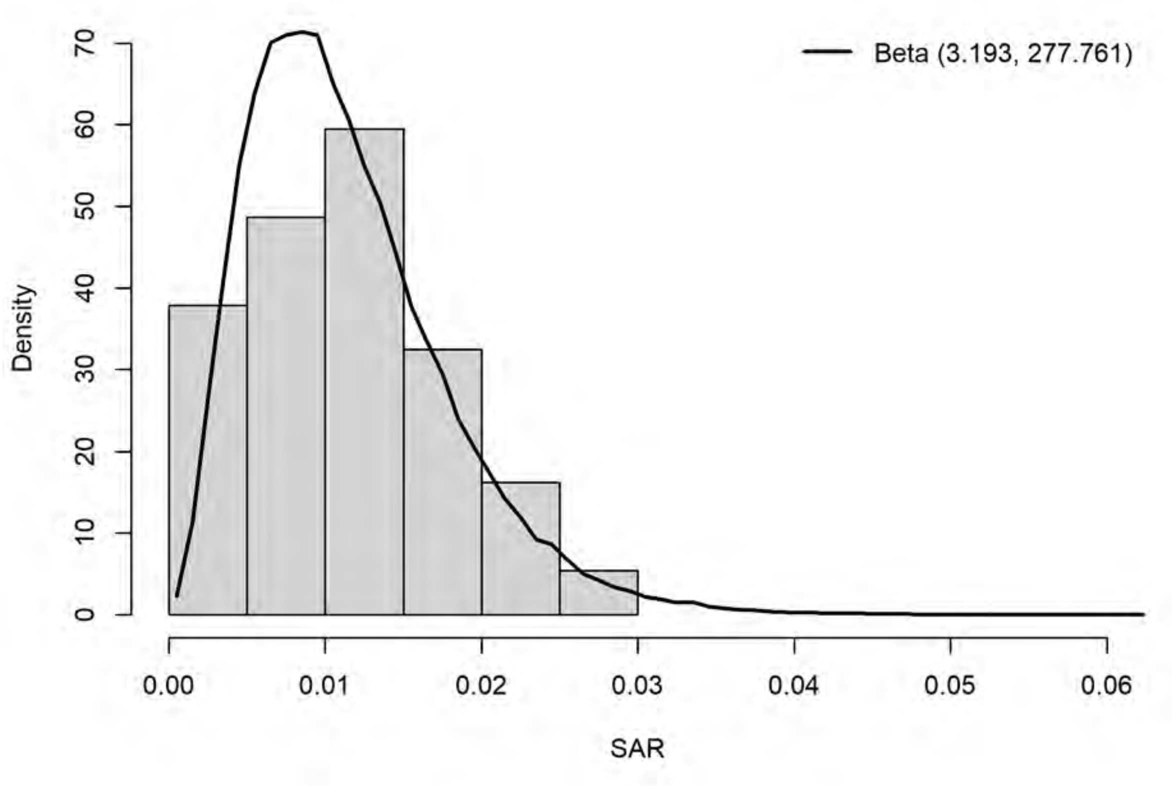

Figure 8-98. Distribution of smolt-adult return rate (SAR) provided by Welch et al. (2021) from coded wire tag (CWT) data for the Willamette stock.

Solid black line shows a beta distribution fitted to the empirical estimates to be used as an informative prior for ϕSAS.

Prior for Sullivan Juvenile Fish Bypass Facility detection probability

Uncertainty in detection probability at detection sites affects estimation of survival probability, so we developed a prior for the PIT tag detection probability at SUJ, pSUJ. The SUJ PIT tag detection array site was operational between 1999 and 2018, and consisted of two antennae, one located in the North Fish Bypass and one in the Unit 13 Bypass at Sullivan hydroelectric plant (PTAGIS; http://www.ptagis.org/).

The development of the prior was a two-step process. First, we specified a mechanistic prior for detection probability at SUJ given knowledge of the detection components. We then used empirical estimates of detection probability from telemetry studies to update this mechanistic prior.

Specifying a mechanistic prior

The probability of a PIT-tagged fish being detected at SUJ can be specified as the product of three components. A smolt migrating downstream and arriving at Willamette Falls can either go directly down the Falls undetected, or pass through Sullivan hydroelectric plant with proportion h. If a fish enters the plant, it can either go through the turbines (Units 1-13) or be guided towards the bypass facilities (North Fish Bypass and Unit 13 Bypass) with fish guidance efficiency g. PIT antennae are located in the bypass facilities and detect the PIT-tagged fish with efficiency a. Combining these parameters results in p'SUJ = hga.

Data from telemetry studies that utilised double-tagged fish (radio- and PIT-tags; citations in Table H.1) released immediately upstream of Willamette Falls were used to parameterize p'SUJ by assuming uniform distributions for each component parameter using the observed minimum and maximum values (Table H.1).

Table 8-75. Uniform distribution range values for component parameters of detection probability at SUJ (p'SUJ = hga) that were used to construct the mechanistic prior for p'SUJ.

Component parameter description	Symbol	Min	Max
Proportiona of smolts passing through the powerplant	h	0.01d	0.37
Fish Guidance Efficiencyb for smolts to pass through bypasses	g	0.767	1
Bypass antenna detection efficiencyc for PIT tags	a	0.7	0.93

a Karchesky et al. (2010)

b Karchesky & Pyper (2009); Skalski et al. (2000); Karchesky et al. (2010); Schroeder et al. (2016)

c Karchesky & Pyper (2009); Schroeder et al. (2016)

d assumed value

The mechanistic prior distribution for p'SUJ was obtained as the product of components h, g, and a using a Monte Carlo simulation method. A single simulation involved drawing random numbers from each component's parameter distribution (Table H.1) to generate a value for p'SUJ. We had no a priori knowledge of any associations between h, g, and a, but note that uncorrelated random draws will underestimate precision of the prior for p'SUJ if such associations exist. The mechanistic prior distribution for p'SUJ was determined from a total of 5,000 simulations run in WinBUGS 1.4 (Spiegelhalter et al. 2007), implemented from within R using the R2WinBUGS package (Sturtz et al. 2005).

Updating the mechanistic prior with empirical data

The data used to establish the mechanistic prior were obtained during a relatively narrow range of discharge values (19,636-35,048 cubic feet per second, cfs). It is known that the discharge at Willamette Falls, as well as fish size, affects the proportion of smolts passing through the Sullivan plant, as high discharges result in more fish going directly over the Falls (Karchesky et al. 2010). The discharge at Willamette Falls varies seasonally (Figure H.3), being low in summer and higher over winter. The volume of river flow diverted through the Sullivan powerhouse is relatively constant at 6,000 cfs (Schroeder et al. 2016). As the proportion of total discharge diverted towards SUJ varies, so the detection probability of smolts passing SUJ varies due to changes in component h. The empirical updates to the mechanistic prior thus needed to consider different flow regimes.

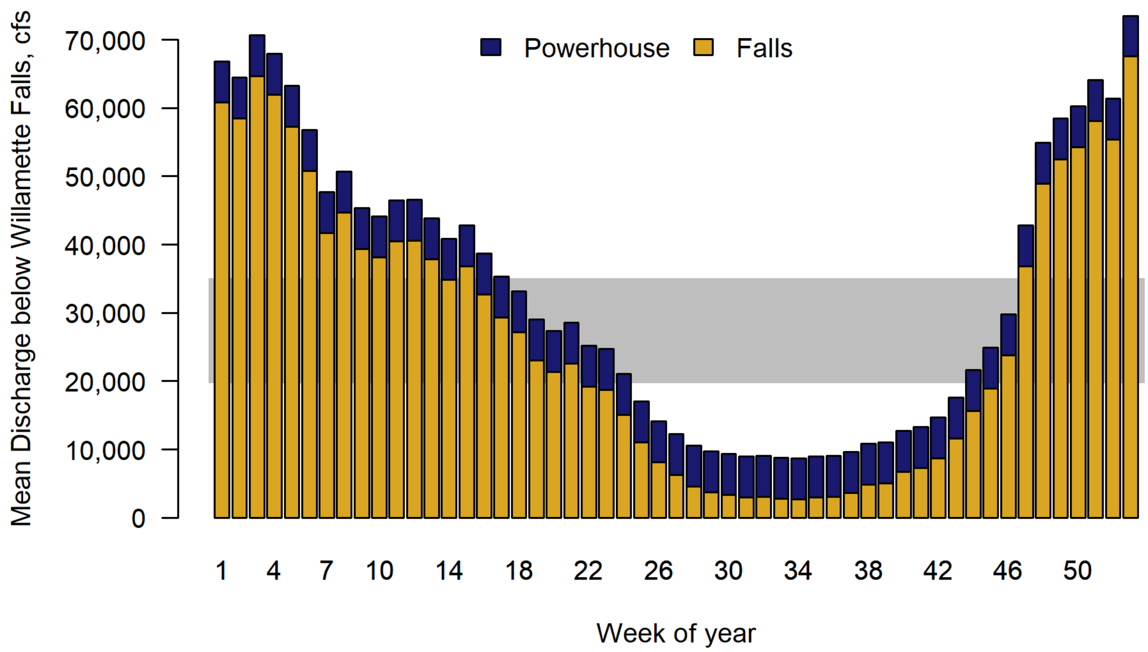

Figure 8-99. Mean discharge below Willamette Falls by week of year for the period 1990-2021.

Discharge measured in cubic feet per second (cfs) at the USGS gage in Portland (USGS 14211720). The volume of river flow diverted through the Sullivan powerhouse is relatively constant at 6,000 cfs (Schroeder et al. 2016). White and grey horizontal bars indicate flow rates considered low (<20,000 cfs, lower white bar), mid (grey bar), and high (>35,000 cfs, upper white bar).

We obtained data from juvenile Chinook salmon telemetry studies conducted in North Santiam by USGS during fall of 2014 (Kock et al. 2015) and in South Santiam by PNNL during spring and fall of 2015, 2016 and 2018 (Hughes et al. 2016, 2017; Liss et al. 2020) that were primarily aimed at determining juvenile passage survival at Detroit and Foster dams, respectively. These studies also had radio-telemetry arrays at Willamette Falls, so we were able to determine the number of double-tagged (radio- and PIT-tagged) Chinook salmon (subyearling or yearling) and steelhead (age 2) smolts released above the dams that were subsequently available for detection by the PIT-detection array at SUJ. Using the known PIT tag codes from the double-tagged smolts, we could then query the PIT Tag Information System (PTAGIS) to determine which of those smolts were detected at SUJ.

The USGS radio-telemetry array was directly downstream of Willamette Falls so we assumed the number of detections was the number passing Willamette Falls. The PNNL radio-telemetry array was 6 km upstream of Willamette Falls, which meant we had to correct the number of detections at the PNNL array for mortality incurred between these locations. We obtained an estimate of this mortality using data from a radio-telemetry study on passage route selection through the Willamette Hydroelectric Project (Karchesky et al. 2010), where Chinook salmon smolts were released at approximately the same location above Willamette Falls the as the PNNL array. Although the mean time from the radio-array to detection in the PIT-array at SUJ was only 14 hr, across 17 releases in March and April 2010, 232 of 267 released smolts were detected by the radio-telemetry array below Willamette Falls, i.e., 13% did not survive. We adjusted the PNNL radio-telemetry detection numbers for this 13% mortality between the upstream release location and Willamette Falls.

We summarised the data into monthly numbers of smolts available for detection (i.e., had survived downstream migration from release to Willamette Falls as determined by radio-telemetry) and those detected at SUJ for those months where there was at least one smolt available for detection at SUJ. This resulted in 22 'experiment' months with enough data. We then calculated the mean monthly discharge below Willamette Falls for those months to determine the frequency of detection events that occurred under low flow (<20,000 cfs), mid flow (20,000 – 35,000 cfs) or high flow (>35,000 cfs) conditions. The empirical estimates of detection probability, calculated as the number of detections at SUJ divided by the number of detections as Willamette Falls, showed an expected negative relationship with mean monthly discharge (

Figure H.4).

We constructed a hierarchical Bayesian model to account for the effects of low, mid, and high flow levels on the probability of detection. The number of detections at SUJ in each month (N_i^{det}) was assumed to be binomially distributed where the number of experiments was the number of smolts available for detection at Willamette Falls in each month (N_i^{avail}) and the probability of success was the flow-specific detection probability at SUJ in each month (p_{flow_i}):

$$\boldsymbol{N_i^{det}} \sim \text{Binomial}(\boldsymbol{N_i^{avail}}, \boldsymbol{p_{flow_i}}) \quad \text{Equation H-26}$$

To prevent estimated values outside of the [0,1] range, the priors for flow-specific detection probabilities were assumed to be normally distributed in logit space, with hyperprior means by flow level (μ_{flow}) and a singular hyperprior standard deviation (σ):

$$\text{logit}(\boldsymbol{p_{flow}}) \sim \text{Normal}\left(\text{logit}(\boldsymbol{\mu_{flow}}), \text{logit}(\boldsymbol{\sigma})\right)$$

Equation H-27

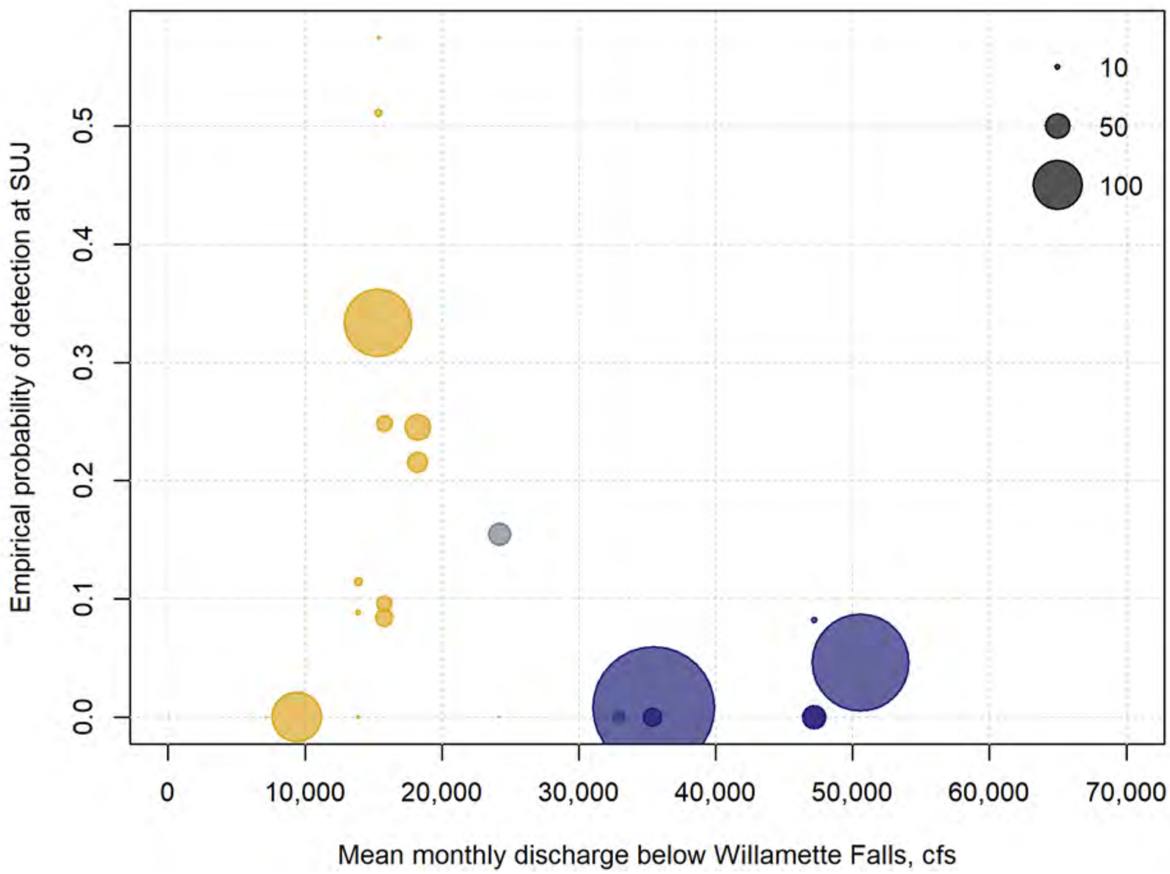

Figure 8-100. Empirical estimates of detection probability of salmonid smolts (Chinook salmon and steelhead) at Sullivan Juvenile Bypass Facility (SUJ) in relation to the mean monthly discharge below Willamette Falls.

The empirical detection probability was estimated in each month as the number of PIT- and radio-tagged fish which were detected by the PIT-detection array in SUJ divided by the number determined to have reached the Falls by detection in radio-telemetry arrays. Data are categorised by low flow (<20,000 cubic feet per second, gold), medium flow (20,000-35,000 cfs, grey), and high flow (>35,000 cfs, blue). Symbol size indicates the number of fish reaching the Falls in each month.

The hyperprior means (μ_{flow}) were obtained from the mechanistic prior for detection probability by drawing from the uniform distribution of each component parameter $(h_{flow}, g_{flow}, a_{flow})$ given the minimum and maximum values in Table H.1. The prior on hyperprior standard deviation (s.d.) was assumed to be uniformly distributed with the upper value set at the maximum possible for a uniform(0,1) variable:

$$\boldsymbol{\sigma} \sim \text{Uniform}\left(\mathbf{0}, \sqrt{(\mathbf{1/12})}\right) \qquad \text{Equation H-28}$$

We then obtained a posterior predictive distribution for each flow level in logit space given hyperparameters.

Samples from the joint posterior probability distribution of the unknown parameters p($h_{flow}, g_{flow}, a_{flow}, \sigma$ | data) were simulated by MCMC integration using WinBUGS 1.4 implemented from within R. The joint posterior was estimated from two independent MCMC chains run in parallel with initial values chosen randomly from the priors. We recorded 25,000 iterations from each chain after removing the first 5,000 as burn-in, and derived inferences from a sample of 50,000 iterations. We assessed convergence of the Markov chains to the posterior distribution by visual inspection of parameter trace plots and use of the R coda package (Plummer et al. 2006). Gelman-Rubin convergence statistics (Gelman et al. 2004) were <1.01 for all parameters, suggesting the chains had fully converged.

Results and summary

The mechanistic prior distribution for p'SUJ had a mean of 0.137 (s.d. 0.076) and a CV of 0.562, characterized by a beta distribution, p'SUJ ~ Beta(5.76, 35.28). The mean value for the mechanistic p'SUJ prior was consistent with the data from PNNL double-tagged fish studies, where only 10-15% of fish were detected at SUJ (Hughes et al. 2016, 2017; Liss et al. 2020).

Empirical estimates of pSUJ under low flow conditions from telemetry studies (n=13) had a mean of 0.193 (s.d. = 0.188). There was only a 0.024 difference in mean estimate of pSUJ for those data from releases of juvenile chinook (subyearling and yearlings), where the mean was 0.182 (range 0 – 0.511), compared to data from releases of juvenile steelhead (age 2), where the mean was 0.206 (range 0 – 0.574). This indicates the two species had similar detectability and thus supports our construction of a generic prior that used data from both Chinook salmon and steelhead smolts. The back-transformed mechanistic-empirical posterior predictive distribution for low flow conditions had a median of 0.149 and a CV of 0.939 (Figure H.5).

The mean of empirical estimates of pSUJ under mid flow conditions (n=3) was 0.052 (s.d. = 0.089). The back-transformed mechanistic-empirical posterior predictive distribution for mid flow conditions had a median of 0.080 and a CV of 1.192 (Figure H.5). The mean of empirical estimates of pSUJ under high flow conditions (n=6) was 0.023 (s.d. = 0.034). The back-transformed mechanistic-empirical posterior predictive distribution for high flow conditions had a median of 0.019 and a CV of 1.882 (Figure H.5).

We used the mean and variance of these back-transformed posterior predictive distribution to determine Beta prior distributions for each flow level:

pSUJ-low ~ Beta(0.672, 2.430)

pSUJ-mid ~ Beta(0.446, 2.500)

pSUJ-high ~ Beta(0.215, 3.915)

These flow-specific pSUJ prior distributions were used in the CJS models (see Appendix C). When necessary, we determined which flow-specific prior to use for each particular release group of juvenile Chinook or steelhead by taking the weighted mean of monthly discharge by the number of detections observed at SUJ in each month.

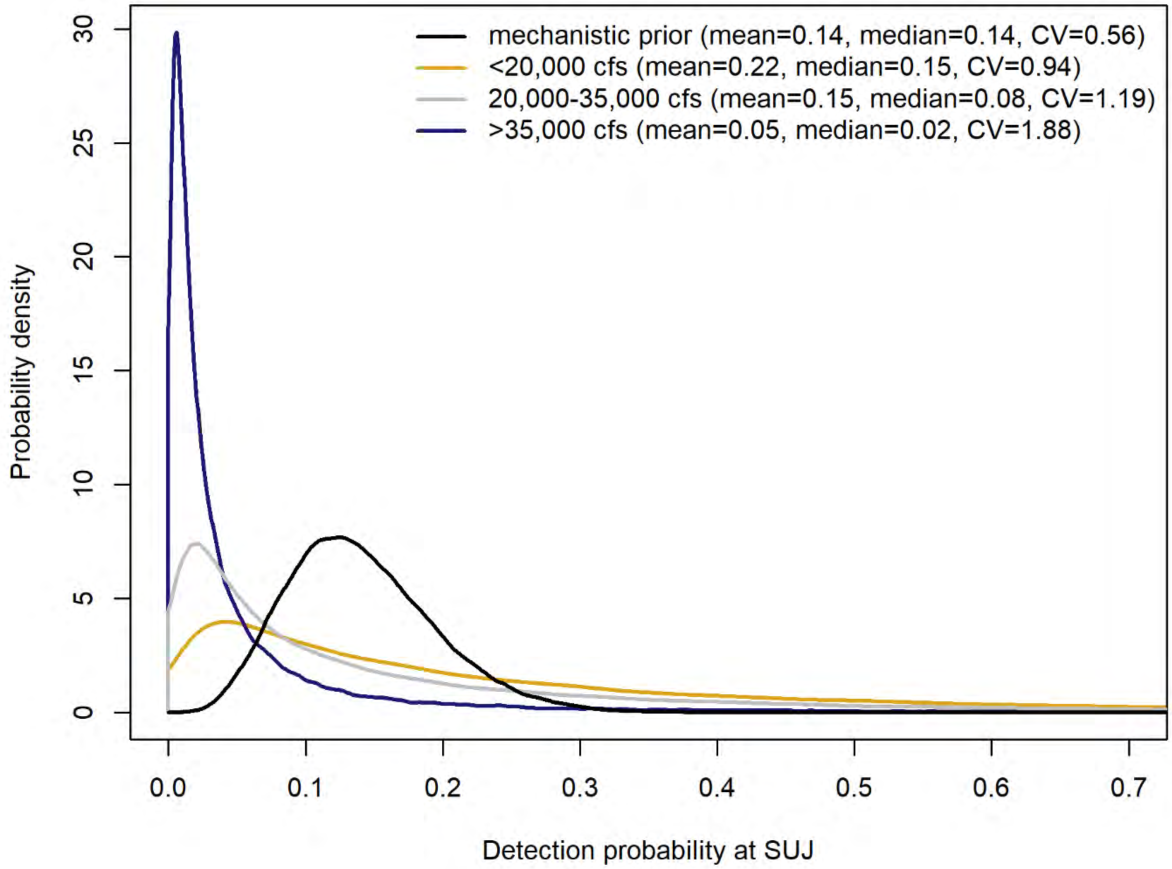

Figure 8-101. Prior probability distributions for detection probability at Sullivan Juvenile Fish Bypass Facility (pSUJ).

The informative mechanistic prior specified using data obtained during medium flow (20,000-35,000 cfs) was updated with empirical estimates of detection probability from telemetry studies, resulting in informative mechanistic-empirical priors for use in low flow (<20,000 cfs), medium flow (20,000-35,000 cfs) or high flow (>35,000 cfs) situations.

8.1.2 WinBUGS code for hierarchical Bayesian model for SUJ detection probability prior specification

```
model{
 for(j in 1:3){
 #specify components under 1=low/2=mid/3=high flow
 h[j] ~ dunif(0.01,0.37) #proportion of smolts passing through the powerhouse
 g[j] ~ dunif(0.767,1.0) #fish guidance efficiency for smolts to pass through bypass
 a[j] ~ dunif(0.7,0.93) #bypass antenna detection efficiency for PIT tags
 #mechanistic prior
 ppred[j] <- h[j]*g[j]*a[j] #predicted detection probability at SUJ under low/med/high flow
 pglmean[j] <- logit(ppred[j]) #logit transform
 postpredlpd[j] ~ dnorm(pglmean[j], pgltau) #produce a posterior predictive value for logit
prob
 postpredpd[j] <- exp(postpredlpd[j])/ (1+exp(postpredlpd[j])) #back transform posterior pred
 p
 }
 maxvar <- 1/12 #compute max possible variance for U(0,1)
 maxsd <- sqrt(maxvar) #transform to SD
 sd ~ dunif(0,maxsd) #keep sd constant across flow levels
 pglsd <- logit(sd) #logit transform the sd
 pgltau <- 1/(pglsd*pglsd) #compute a hyper prior precision for p
 hprior ~ dunif(0.01,0.37) #dummy prior for h
 gprior ~ dunif(0.767,1.0) #dummy prior for g
 aprior ~ dunif(0.7,0.93) #dummy prior for a
 ppredp <-hprior*gprior*aprior #non-updated hyper prior for p
 #update with empirical estimates of detection probability by flow level
 #p detect per experiment is exchangeable within each of the 3 levels
 #p detect is hierarchical
 for(i in 1:nobs){
 ndets[i] ~ dbin(pd[i],ntags[i]) #compute prob. of each set of detections
 pd[i] <- exp(pdlogit[i])/ (1+exp(pdlogit[i])) #back transform logit p to p
 pdlogit[i]~ dnorm(pglmean[grp[i]], pgltau) #hyperprior for logit p detect
 }
}
```

8.1.3 Willamette Falls Fishway detection probability

Adult Chinook salmon and steelhead returning to the Willamette River must ascend the Willamette Falls Fishway (WFF). The WFF interrogation site was operational between 2005 and 2019, and consisted of four PIT-detection antennae, two located on the weir downstream of the fish counting window, and two on the weir upstream of the fish counting window (PTAGIS). Given the number of antennae, and that all fish passing the ladder are counted by ODFW staff, we assumed that the detection probability at WFF (pWFF) was very close to 1. Others have analysed PIT-tag data from the Willamette and made a simplifying assumption that detection probability at WFF is equal to 1 (Pease et al. 2020). Rather than remove all uncertainty associated with this parameter, we instead used a highly informative Beta prior, pWFF ~ Beta(191.1,3.9). This had a mean of 0.98 and a CV of 0.01. This enabled both pWFF and ϕSAS to be identifiable in the CJS model, rather than as their product.

8.2 ATTACHMENT I: DEVELOPMENT OF BAYESIAN PRIOR DISTRIBUTIONS ACCOUNTING FOR TAG LOSS, TAGGING MORTALITY AND DIFFERENCES IN SURVIVAL RATES BETWEEN THE TAGGED AND TARGET POPULATIONS

To obtain credible estimates of survival rates for populations of spring Chinook salmon in the Upper Willamette River, a Bayesian Cormack-Jolly-Seber (CJS) model was fitted to data from paired release and other studies that applied PIT tags to in most instances hatchery-raised juvenile spring Chinook salmon. These juvenile PIT tagged fish were then released into different river reaches in the sub-basins of the Upper Willamette River Valley. The CJS model applications provided estimates of apparent survival rates of juvenile salmon from the release points to the Sullivan Dam tag detection facility above Willamette Falls and apparent smolt-adult survival rates also (see Appendix C for details).

It is well known that fish tagged with PIT and other types of tags may have higher mortality rates than untagged fish from the same population. Tagging a fish may cause some injuries that may make it more susceptible to disease or predation (see e.g., Melnychuk et al. 2014). Tags may also fall off of the fish or malfunction such that they cannot be detected by PIT tag detection devices (e.g., Arnason and Mills 1981; Knudsen et al. 2009; McCann et al. 2018). In addition, tags are often put on hatchery-raised (i.e., hatchery origin, HOR) fish to estimate survival rates for wild-type or natural origin (NOR) fish of the same species that were born in the same tributary or river. However, it has been found that the survival rates of fish in their natural habitats can differ between HOR fish versus NOR conspecifics (see e.g., Waples 1991; Araki and Schmid 2010; Williamson et al. 2010; Hagen et al. 2020; HRSG 2020a, 2020b).

Statistical methods to estimate survival rates from mark and recapture records typically provide, among other things, estimates of apparent survival rates of the tagged population of fish. Yet apparent survival rate estimates from the tagging studies could thus be different, on average, from fish in the natural origin population that were not tagged. Some published studies have quantified the above-mentioned three factors, i.e., the magnitude of tag-induced mortality rates (e.g., Melnychuk et al. 2014), the rate of loss of tags (e.g., from malfunction, falling off or ejection from the fish) (e.g., McCann et al. 2018) and average differences in survival rates between HOR and NOR populations (Araki and Schmid 2010; Williamson et al. 2010; Hagen et al. 2020; HRSG 2020a, 2020b). To account for these three factors we formulated a prior distribution for an adjustment factor for tagging-based estimates of survival rates for different salmon life stages that incorporated plausible ranges of values for tagging mortality, tag loss and differential survival based on fish origin. We did not include this prior in the parameter estimation. Instead, the apparent survivals estimates were adjusted after parameter estimation.

According to Arnason and Mills (1981) apparent survival probability estimates are confounded with the probability of tag loss. The estimate of survival rate ($\hat{S}$) corrected for imperfect tag retention ($\hat{\tau}$) is given by:

$$\hat{\boldsymbol{S}} = \frac{\hat{\boldsymbol{\phi}}}{\hat{\boldsymbol{\tau}}} \qquad \text{Equation I-1}$$

where $\hat{\tau}$ is 1 – probability of tag loss and $\hat{\phi}$ is the unadjusted estimate of [apparent] survival rate, e.g., derived from CJS analysis.

This adjustment has been applied in the context of estimating survival and capture probabilities of New Zealand fur seals (Arctocephalus forsteri, Bradshaw et al. 2003). This form of the adjustment has also been applied to account for residualization in steelhead (Oncorhynchus mykiss, Melnychuk et al. 2014). Following Melnychuk et al. (2014) and extending the Arnason and Mills (1981) adjustment to account for all three sources of average differences between tagged and untagged populations of interest, we applied the following equation:

$$\hat{\boldsymbol{S}} = \frac{\hat{\boldsymbol{\phi}}}{\hat{\boldsymbol{\tau}}_{\boldsymbol{l}}\hat{\boldsymbol{\tau}}_{\boldsymbol{M}}\hat{\boldsymbol{\tau}}_{\boldsymbol{H}}} \qquad \text{Equation I-2}$$

Where $\hat{\tau}_l$ is the tag retention rate accounting for tag loss, $\hat{\tau}_M$ is equal to 1 – tag induced mortality, $\hat{\tau}_H$ is the ratio of the survival rate of the study population (i.e., HOR) to the population of interest (i.e., NOR).We assumed independence across the three sources of adjustments and that correlation was zero between each of the pairs of adjustment factors.

Evidence of tag Loss, tag induced mortality, and potential differences between hatchery and wild populations

8.2.1 Tag Loss

PIT tag loss rates can depend on the life history type, e.g., fry that stay in natal spawning areas versus immediately migrate from these areas, and on life stage. We have identified two main stages for tag loss adjustments based on life stage:

1) 0+ to 6 months following release. i.e., from the release site (e.g., above dam) to their last in river detection site (e.g., Sullivan Juvenile Bypass Facility (SUJ))

2) 6 months – 4/5 years, e.g., passage at SUJ as smolts to return at Willamette Falls Fishway.

Foldvik and Kvingedal (2018) found that Atlantic salmon had a 91% tag retention rate over a 533 day period and that the majority of tag loss occurred within the first six months, followed by the next 1 -1.5 years. For Chinook salmon, Knudsen (2009) found that retention was 98% after 1-2 months, and 80.5-82.8% between six months and four years.

In a comparative survival study on Chinook salmon, the Fish Passage Centre (McCann et al. 2018) found PIT tag retention rates 92.3-96.6%. Higher loss rates similar to Knudsen of at least 15% have been less frequently observed. Numerous variables to be considered have included species, size of tag relative to that of the fish, laboratory study versus in-situ double tag experiment, recovery time after tagging, skill of tagger, method of tag implantation, methods and quality of analysis, and population specific stressors.

8.2.2 Tag Induced Mortality

In a four year study, the Fish Passage Center (McCann et al. 2018) tagged 75,000 fish per year using a double tagging experiment design, and found no impact of tagging on mortality. Knudsen (2009) in a 5-year study tagged 40,000 fish per year in a double tag experiment and found tag-induced mortality rates 4-33% with an average of 10%. Camacho et al. (2018), examining IDFG run reconstructions from 1996-2014 found smolt-adult return rates (SARs) were 46% greater than PIT tagging-based estimates. Schaller and Petrosky (2007) found in NOAA run reconstructions from 1994-2004 SARs were 19% greater than PIT tag-based estimates. The Fish Passage Centre had noted about Chinook salmon studies, the efficiency of hand detectors in Knudson (2009) had not been factored into tag induced mortality rates and run reconstructions themselves could also contain bias in estimates of SARs. Vollset et al. (2020) conducted a systematic literature review of 100 reports to investigate tag-induced mortality. He consolidated results from 18 publications with PIT experiments to generate a relationship between tag-induced mortality and the ratio of tag length to fish length. Vollset et al. (2020) found consistent evidence of tag induced mortality across a wide range of literature and experimental settings.

Our baseline estimate of tag induced mortality is based on the application of the relationship derived in Vollset et al. (2020). Vollset et al.'s relationship was based on studies from 7, 14, and 21 days from the date of tagging. These studies did not observe a high degree of time-dependent mortality, but assessing the credibility of the relationship would require comparing the calculated cumulative mortality with estimates in the Chinook salmon literature on longer time horizons. As this information/data were not available, we calculated a representative tag length: fish length ratio for the release group. We then obtained a 95% credible interval from the Vollset et al. relationship (Figure I.1).

Based on the computed ratio of tag length to tagged fish length, we then generated a distribution of cumulative mortality assuming a uniform error distribution based on the 95% interval.

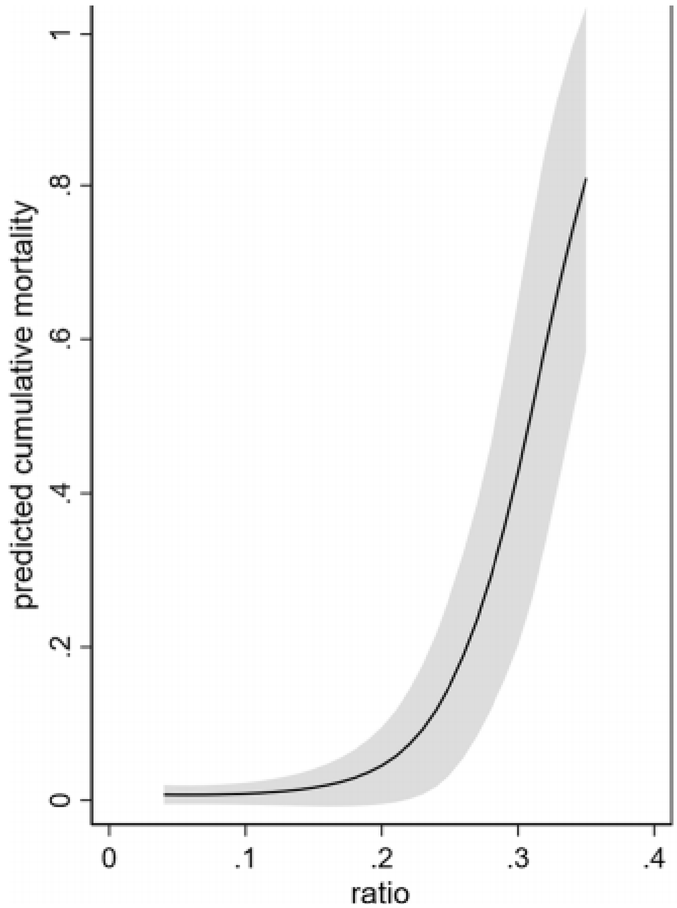

Figure 8-102. Mean (solid line) and 95% (grey shaded area) predicted cumulative tag induced mortality rate based on the ratio of tag length to fish length (obtained from Vollset et al. 2020).

8.2.3 Adjustments to represent potential differences in survival rates between hatchery and wild populations

The largest PIT tag studies in the Willamette River basin have been conducted on hatchery reared salmon and steelhead as opposed to NOR fish. However, it has been commonly found that there can exist differences in stage-based survival rates between HOR and NOR salmon (and steelhead) from the same river system (see, e.g., Waples 1991; Araki and Schmid 2010; Williamson et al. 2010; Hagen et al. 2020; HRSG 2020a, 2020b). Potential reasons for these differences may include differences in exposure to stressors such as parasites and pathogens, differences in predation rates, growth rates, and associated foraging and predator avoidance behaviours between HOR and NOR populations. Additionally, there may also be origin-specific physical characteristics including fork length, morphology and age and/or developmental status at release. A large number of studies have found two-to-three-fold differences in stage-specific survival rates between HOR and NOR salmon and steelhead. However, some reports have found no difference in stage-specific survival rates between HOR and NOR fish. Melnychuk et al. (2014) and Moore et al. (2012) have provided estimated relationships between apparent Cormac-Jolly-Seber survival rate estimates and explanatory variables. Melnychuk et al. (2014)

model included as covariates, fork length, release date, hatchery versus wild, reach specific parameters, and initial mortality parameters.

8.2.4 Methodology

Formulating a prior density function to adjust apparent survival accounting for all three factors

We formed prior density functions for tag retention rates, survival rates from tag induced mortality, and the ratio of survival rate between HOR and NOR salmonid populations for the denominator terms in eq 2, based on the above cited literature. A Monte Carlo simulation approach was applied to compute a probability density function for the adjustment factor (A), such that it accounts for apparent survival rate estimates for 1) in-river release to smolt survival, and 2) smolt to adult survival rates. The adjustment factor was first drawn from the uniform density function for each individual factor and the combined adjustment factor was computed from:

$$\boldsymbol{A} = \frac{\mathbf{1}}{\bar{\boldsymbol{\tau}}_{l,s}, \bar{\boldsymbol{\tau}}_{M,s}, \bar{\boldsymbol{\tau}}_{H,s}} \qquad \text{Equation I-1}$$

where $\bar{\tau}_{l,s}$ denotes population mean tag retention rate of $\hat{\tau}_l$, $\hat{\tau}_{M,s}$ denotes population mean survival rate from tag-induced mortality, and $\hat{\tau}_{H,s}$ denotes the population mean ratio of survival rate between HOR and NOR fish. The subscript s denotes stage, i.e., either in-river release to smolt or smolt-adult. The specifications for the minimum and maximum values for the adjustment factors are provided in Table 6-88.

Table 8-76. Minimum And Maximum Specification for Survival Rate Adjustment Factors. *

Stage (s)	τ_(l,s,min)	τ_(l,s,max)	τ_(M,s,min)	τ_(M,s,max)	τ_(H,s,min)	τ_(H,s,max)
River smolt	0.97	0.99	0.85	0.95	0.5	0.95
Smolt-Adult	0.85	0.98	0.67	0.96	0.5	0.95

Note: * Specifications for the minimum and maximum values in the prior uniform distributions for the apparent survival rate adjustment factors.

Figure I.2 shows example Monte Carlo frequency distributions from 1000 simulations of the combined adjustment factors for the river smolt and smolt-adult stages.

Table I.2 shows 95% lower and upper bounds and medians for unadjusted and adjusted survival rate estimates from paired release experiments on hatchery raised Chinook salmon in the McKenzie sub-basin. Adjustments can have a considerable impact on downstream passage survival, as average increases to lower bounds, median, and upper bounds under the applied priors are nearly a factor of 2.

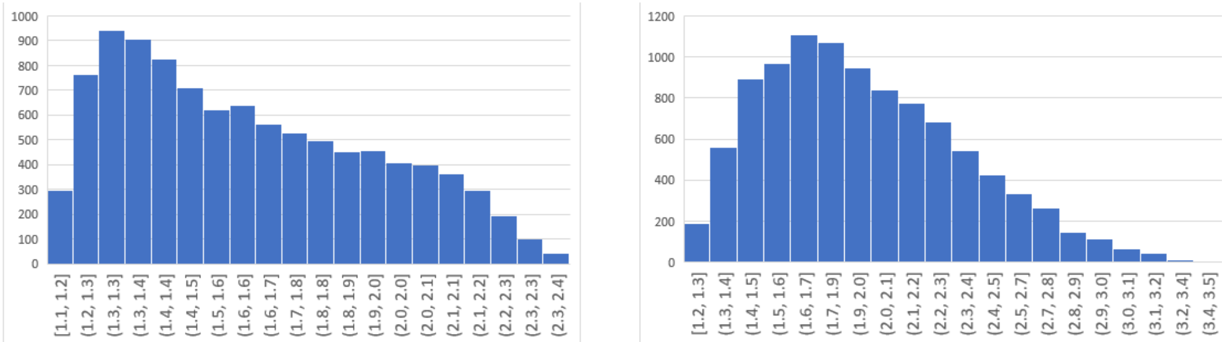

Figure 8-103. Example Monte Carlo frequency distributions from 10,000 simulations of the combined adjustment factors for the river smolt and smolt-adult stages. *Example Monte Carlo frequency distributions from 10,000 simulations of the combined adjustment factors for the river smolt and smolt-adult stages. Bin ranges for adjustment factors are shown on the horizontal axis and frequency of simulated occurrence is shown on the vertical axis.*

Table 8-77. 95% lower and upper bounds and medians for unadjusted and adjusted survival rate estimates from paired release experiments on hatchery raised Chinook salmon in the Middle Fork sub-basin in years 2011-2014. RSS means release to SUJ survival rates, SAS means smolt-adult survival rate, the year indicates the release year, and condition indicates the water flow conditions in the year of release.

Apparent and Adjusted Survival Estimates									
				Dexter Paired Release			Lookout Point Head of Reservoir		
Phase	Year	Condition	Estimate Type	Lower Bound	Median	Upper Bound	Lower Bound	Median	Upper Bound
RSS	2011	Good Spring Spill	Unadjusted	27%	59%	97%	10%	22%	68%
			Adjusted	43%	92%	100%	15%	34%	100%
SAS	2011	Poor Marine	Unadjusted	0.02%	0.08%	0.29%	0.04%	0.20%	0.81%
			Adjusted	0.03%	0.15%	0.54%	0.07%	0.38%	1.52%
RSS	2012	Some Spring Spill	Unadjusted	16%	27%	40%	4%	8%	18%
			Adjusted	25%	42%	63%	7%	13%	28%
SAS	2012	Good Marine	Unadjusted	0.05%	0.10%	0.21%	0.3%	0.7%	1.5%
			Adjusted	0.09%	0.19%	0.39%	0.6%	1.4%	2.9%
RSS	2013	Some Spring Spill	Unadjusted	11%	23%	54%	5%	13%	33%
			Adjusted	17%	36%	84%	8%	20%	53%
SAS	2013	Poor Marine	Unadjusted	0.01%	0.06%	0.18%	0.08%	0.25%	0.76%
			Adjusted	0.03%	0.10%	0.33%	0.15%	0.46%	1.42%
RSS	2014	Some Spring Spill	Unadjusted	4%	9%	27%	0.3%	0.7%	3.6%
			Adjusted	6%	13%	43%	0.5%	1.1%	5.7%
SAS	2014	Good Marine	Unadjusted	0.005%	0.055%	0.272%	0.06%	0.66%	3.55%
			Adjusted	0.009%	0.103%	0.507%	0.12%	1.24%	6.62%

8.3 ATTACHMENT J: SIMULATION-ESTIMATION ANALYSIS OF THE BAYESIAN CORMACK-JOLLY-SEBER MODEL

Model performance can be examined using simulation-estimation analysis, in which models are fitted to simulated data where the true values of the parameters are known. Simulation-estimation analysis can help understand what features make data informative or uninformative and enable identification of datasets for which a model is likely to produce biased or imprecise estimates. The approach is commonly used in fisheries science (e.g., Magnusson and Hilborn 2007; Robert et al. 2010). We used simulation-estimation analysis to evaluate estimation performance of the Cormack-Jolly-Seber (CJS) model within a Bayesian framework. Given the range in release sizes and low numbers of adult returns to Willamette Falls observed in Upper Willamette PIT tagging studies, we sought to determine how reliable the estimates of survival were under a range of release sizes and detection histories. Release sizes of natural-origin juveniles were typically in the order of 1,000, while releases of hatchery-origin juveniles were an order of magnitude greater. We therefore examined bias and precision from releases of 1,000 and 10,000 fish. In addition, it was common for studies to result in mark-recapture datasets that had zero fish that were detected at all sites, so we examined the effect of having a zero count in this detection history. The simulation-estimation analysis consisted of separate steps as detailed below.

8.3.1 Simulation of detection histories

We simulated a study system with three sites j, a release location x (j=1) and two detection sites, at Sullivan Dam Juvenile Bypass Facility (SUJ, j=2) and Willamette Falls Adult Fishway (WFF, j=3). To generate mark-recapture data for released juvenile fish we used known ('true') parameter values for survival ϕ_j to each site j and detection probabilities p_j. We first simulated the latent alive or dead state (zi,j) for each individual fish i as smolts at SUJ and as returning adults at WFF given the initial release location (x) and release-to-smolt (RSS) and smolt-to-adult (SAS) survival probabilities (ϕ_{SUJ} = 0.3, ϕ_{WFF} = 0.01, respectively):

$$\boldsymbol{z_{i,j}} \sim \begin{cases} \text{Bernoulli}(\boldsymbol{\phi_j}), \boldsymbol{z_{i,j-1} = 1} \\ \boldsymbol{0},\ \boldsymbol{z_{i,j-1} = 0} \end{cases} \text{, for } \boldsymbol{j} = \text{SUJ, WFF.} \qquad \text{Equation J-1}$$

All fish were assumed to be alive on release, i.e. zi,x = 1. We then simulated the detection histories of each individual (yi,j) given its current latent state (zi,j) and site-specific detection probabilities (p_{SUJ} = 0.1, p_{WFF} = 0.97, respectively):

$$\boldsymbol{y_{i,j}} \sim \begin{cases} \text{Bernoulli}(\boldsymbol{p_j}), \boldsymbol{z_{i,j} = 1} \\ \boldsymbol{0},\ \boldsymbol{z_{i,j} = 0} \end{cases} \text{, for } \boldsymbol{j} = \text{SUJ, WFF.} \qquad \text{Equation J-2}$$

We used R (R Core Team 2021) to generate 100 mark-recapture datasets for each release size of 1,000 or 10,000 individual fish. This resulted in datasets where individual fish had either the n111, n110, n101 or n100 detection history, where n reflects the frequency of detection ("1") or lack thereof ("0") at the release location, SUJ and WFF, respectively (e.g., n111 = 10; 10 released fish were detected at both SUJ and WFF).

8.3.2 Bayesian estimation

For each simulated dataset, we built a Bayesian CJS model that was identical to the model used to simulate the data, meaning that any differences between true and estimated parameters were due to the performance of the estimation method and not to mis-specification of the model. In this evaluation we fitted the data using the state-space model, but note that the multinomial model used in estimating survival parameters from Willamette sub-basin releases of PIT-tagged juvenile Chinook salmon (Appendix C) produced similar results to the state-space model in preliminary analyses using both models.

We first modelled the latent state of each individual i, given Equation J-1, and then modelled each individual's detection history as a function of its latent state, given Equation J-2. After preliminary analyses indicated poor estimation performance using vague priors, we specified informative priors for the survival and detection probabilities (Appendix H). For the probability of detection at SUJ, we assumed that all releases were undertaken during medium flow conditions. The priors used were:

φ_{SUJ} ~ "Beta" (2.440,3.665) Equation J 3

φ_{WFF} ~ "Beta" (3.193,277.76)

p_{SUJ} ~ "Beta" (0.446,2.500)

p_{WFF} ~ "Beta" (40.00,1.667)

Samples from the joint posterior probability distribution of the unknown parameters and latent states p(ϕ_{SUJ}, ϕ_{WFF}, p_{SUJ}, p_{WFF}, zi,j | data) were simulated by MCMC integration using WinBUGS 1.4 (Spiegelhalter et al. 2007) implemented from within R using the R2WinBUGS package (Sturtz et al. 2005). The joint posterior was estimated from three independent MCMC chains run in parallel with initial values chosen randomly from the prior. To conserve computer memory, we recorded 1 in 10 iterations after the first 1,000 iterations were removed as the burn-in, and we derived inferences from a sample of 15,000 iterations from three chains of 5,000 iterations. We assessed convergence of the Markov chains to the posterior distribution by visual inspection of parameter trace plots and use of the R coda package (Plummer et al. 2006). Gelman-Rubin convergence statistics (Gelman et al. 2004) were <1.01 for all parameters, suggesting the chains had fully converged.

Measurement of estimation bias

The performance of the estimation method was evaluated by how accurately the marginal posterior probability distribution of each parameter estimated the true value used to simulate the dataset. For each parameter θ, this accuracy was measured using the median of the marginal posterior compared to the true value, as summarised by the percent relative bias (PRB):

PRB = 100[(estimated θ – true θ) / true θ] Equation J-1

For each release size, boxplots were used to summarise the PRB for [ϕ_{SUJ}, ϕ_{WFF}, p_{SUJ}, p_{WFF}] across the 100 simulated datasets. We note that 100 is relatively low for a simulation-estimation study, but computation time limited analysis of further additional simulated datasets. As some of the observed mark-recapture datasets from the Upper Willamette had low numbers of adult returns to Willamette Falls, with some zero counts in the n111 detection history, we also summarised bias with similar simulated datasets removed.

8.3.3 Results and Summary

Estimation using the Bayesian CJS model resulted in reliable estimates of all parameters. The mean percent relative bias across all parameters was ≤15% (Table J.1, Figure J.1), with the number of fish released not having a large effect on the PRB. This finding was not unexpected due to the use of informative priors for the model parameters. For those simulated datasets where the bias in particular parameters was greater than the mean PRB, the true value was always found within the 95% credible interval of the marginal posterior for those parameters. We therefore concluded that the model results in accurate estimates of all model parameters.

The removal of simulated mark-recapture datasets that had a zero count in the n111 detection history resulted in small reductions in the mean bias (Table J.1). The range of PRB values for ϕ_{SUJ}, ϕ_{WFF} and p_{SUJ} was >20% smaller under this summary, particularly for the smaller release size. This indicated that estimates from datasets with non-zero counts in the n111 detection history were more accurate than those with zero counts in this detection history. Given this finding we did not analyse observed datasets that had zero n111 counts, with the implication that parameter estimates would be further biased without the use of informative priors.

Table 8-78. Table 6 90. PRB in from simulated datasets of 1,000 or 10,000 released fish. *

N released	Detection histories	N simulations	ϕ_SUJ	ϕ_WFF	p_SUJ	p_WFF
1,000	All	100	10.02	-7.56	-4.05	-0.33
1,000	n111>0 only	29	1.67	11.19	4.45	-0.28
10,000	All	100	15.44	-9.26	-9.42	-0.33
10,000	n111>0 only	92	12.16	-6.67	-7.06	-0.32

Notes: * Mean percent relative bias (PRB) in estimated parameters φ_SUJ, φ_WFF, p_SUJ, and p_WFF from simulated datasets of 1,000 or 10,000 released fish. We compare PRB for all parameters considering all simulated datasets, and when filtering to simulations with non-zero counts in the n111 detection history (n111 > 0).

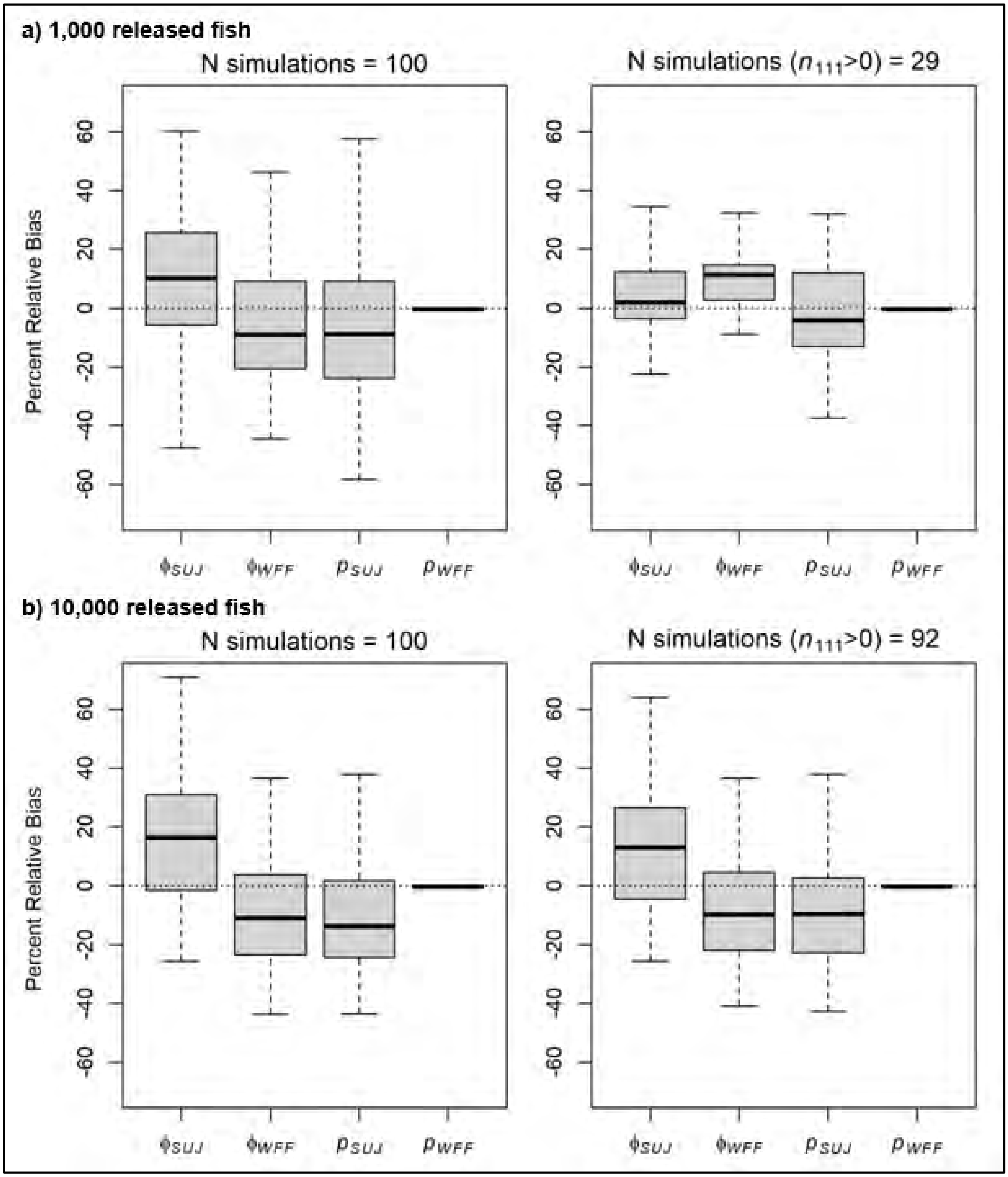

Figure 8-104. Percent relative bias (PRB) Distributions from modeled fish sets. *Distributions of percent relative bias (PRB) in the median of the marginal posterior probability distributions from 100 generated mark-recapture datasets resulting The median PRB for each parameter is shown as a black bar, boxes represent the interquartile range (IQR), and whiskers represent the range of PRB. We compare PRB for all parameters considering all simulated datasets, and when filtering to those with non-zero n111 detection histories.*