Technical Report 2019-4-FINAL

POTENTIAL EFFECTS OF CHANGES TO TEMPERATURE TARGETS IN THE NORTH SANTIAM RIVER ON ADULT CHINOOK SALMON BEHAVIOR AND SURVIVAL

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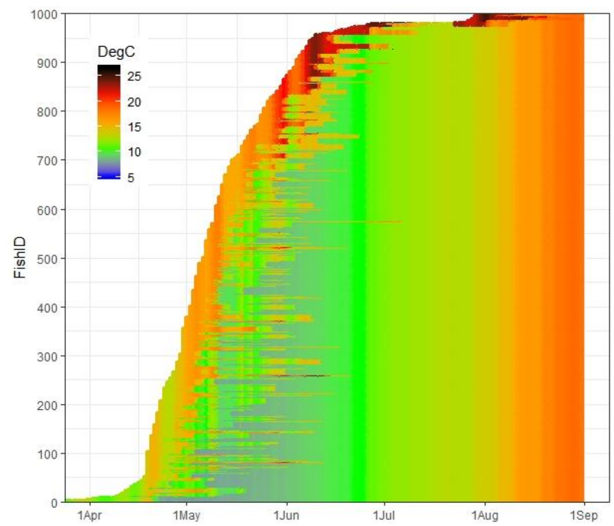

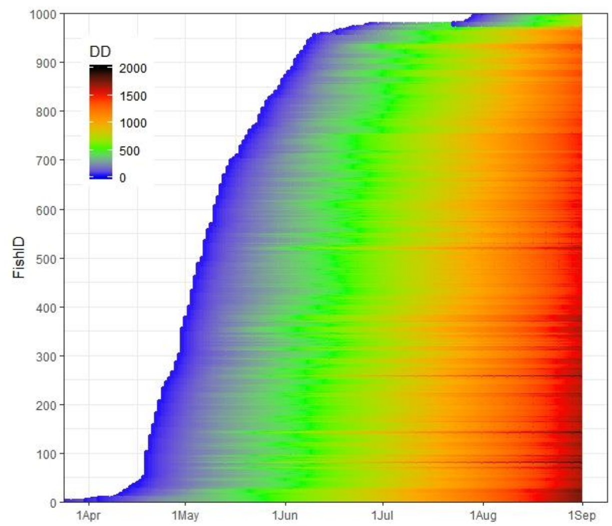

For

U.S. Army Corps of Engineers
Portland District

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Executive Summary

Report Organization

Introduction: Management assumptions and uncertainties

- Detroit and Big Cliff dams have reduced fish habitat quality in the North Santiam River
- Current dam operations negatively affect river temperature regimes downstream from the projects
- Temperature impairment alters Chinook salmon behaviors and reduces survival and fitness
- Water temperature management at the dams could partially restore historic thermal regimes downstream
- How are recent North Santiam conditions associated with Chinook salmon behavior and survival?
- Will more ‘natural’ thermal regimes improve biological outcomes for Chinook salmon?
- Which temperature metrics and biological metrics are most appropriate for assessing outcomes?
- What insights about the North Santiam Chinook population can be derived from individual experiences?

Sections 1-6: Summary and synthesis of existing data

- 1: North Santiam River environment datasets: recent conditions
- 2–3: Chinook salmon datasets: Migration timing, spawn timing
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- 8: Model development
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1. River Environment

Key results

- Data quality was generally high
- Discharge and temperatures fluctuated considerably among years
- Temperatures were highly correlated among gage sites in the North Santiam main stem
- Temperature correlations were much lower between the Willamette, North Santiam, and Little North Santiam
- 7DAMD temperatures were 22-26 °C in the Willamette River at Newberg
- 7DAMD temperatures in the North Santiam River were 20-24 °C (Greens Bridge), and ~13-17 °C (Niagara)

Relevance to temperature management for adult Chinook salmon

- Existing data provide a good baseline for temperatures generated with existing structures at Detroit Dam

2. Chinook Salmon Migration Timing

Key results

- Adult run timing at Willamette Falls was highly variable across years; median dates differed by > 30 d
- Annual run timing at the falls was earlier in warm, low-discharge years
- We estimate that mean migration time from Willamette Falls to the Bennett dams was ~30 d
- In a short time series, run timing at Bennett appeared to be later in warm, low-flow years
- In a short time series, adult collection at Minto appeared to be earlier in years with high Jun-Aug discharge
- Fin-clipped and unclipped salmon appeared to enter Minto at similar times and rates

Relevance to temperature management for adult Chinook salmon

- The proportion of the run affected by summer temperature management will vary across years
- The environmental effects on run timing in the North Santiam appear to be complex

Uncertainties

- Daily passage and collection data were not routinely reported for the Bennett dams and Minto Facility
- Counts at the two Bennett dams were highly divergent within some years, for unknown reasons
- Highly regulated conditions below Big Cliff make it difficult to interpret effects on salmon behavior

3. Chinook Salmon Redd Construction and Spawn Timing

Key results

- Most redd construction was concentrated between mid-September and mid-October
- Successfully spawned female carcasses were also concentrated from mid-September to mid-October
- Unsuccessful female carcasses were collected much earlier, starting in early summer
- Peak redd dates were somewhat earlier in warm, low-discharge years in the Willamette River
- Spawn timing was also earlier in warm, low-discharge years
- Correlations were stronger with Willamette River temperatures than North Santiam River temperatures, perhaps due to relatively lower inter-annual temperature variability in the North Santiam

Relevance to temperature management for adult Chinook salmon

- Redd construction and spawn timing may shift slightly in response to temperature management
- The environmental effects on redd construction and spawn timing in the North Santiam appear to be complex
- Discharge volume may be an important consideration in temperature management plans

Uncertainties

- Sampling protocols (e.g., timing, intensity) for redds and carcasses have unquantified biases
- Separating co-varying temperature and discharge effects on salmon behavior is difficult
- Separating exposure to stressors early in migration from proximate exposure at spawning grounds is difficult

4. Prespawn Mortality (PSM)

Key results

- Mean PSM was very high (>70%) below Bennett and high (>45%) between Bennett and Minto
- Mean PSM was high (~50%) in the Little North Santiam and low (~10%) above Detroit reservoir
- Carcasses numbers and PSM estimates varied widely among reaches and years
- Correlations between mean monthly water temperatures were mixed
- PSM tended to be positively correlated with late summer and early fall water temperatures
- PSM was negatively correlated with early summer temperatures in some cases
- PMS tended to be negatively correlated with river discharge
- PSM was positively correlated with PHOS and negatively correlated with run timing at Willamette Falls
- PSM estimates were similar among reaches within year, indicating common effects

Relevance to temperature management for adult Chinook salmon

- PSM was both positively and negatively correlated with some temperature metrics
- Warmer temperature in early summer and lower temperature in late summer may be appropriate targets
- Discharge volume may be an important consideration in temperature management plans

Uncertainties

- Carcass sampling protocols (e.g., timing, intensity) have unquantified biases
- Interpreting the apparent offsetting effects of seasonal temperatures on PSM is challenging
- Separating co-varying temperature and discharge effects on PSM is difficult
- Separating exposure to stressors early in migration from proximate exposure at spawning grounds is difficult

5. Proportion of Hatchery-Origin Spawners (PHOS)

Key results

- Hatchery-origin spawners were abundant, except between Minto and Big Cliff and in the Little N. Santiam
- PHOS was positively correlated with temperature and negatively correlated with discharge
- Environmental correlations were higher with PHOS than with PSM

Relevance to temperature management for adult Chinook salmon

- Temperature management may help reduce PHOS
- Discharge volume may be an important consideration in temperature management plans

Uncertainties

- PHOS, a relative abundance metric, is sensitive to enumeration uncertainty for fish entering the N. Santiam
- PHOS and PSM appear to be co-varying, making it difficult to separate cause and effect
- Separating co-varying temperature and discharge effects and upstream-downstream effects is difficult

6. Radiotelemetry

Key results

- Migration histories from Willamette Falls to the North Santiam River were available for ~160 adult Chinook
- Tagged salmon spent about 10 d, on average, in the Willamette River main stem
- Most tagged salmon spent a large majority of their time between Bennett and Minto (*mean* ~70 d)
- Salmon migrated faster as water temperature increased in all reaches
- Temperatures of logger-tagged salmon closely paralleled river temperature in the North Santiam
- Tagged salmon had a wide variety of behaviors near Minto, including long holding periods before entry
- Similar proportions of fin-clipped and unclipped salmon entered Minto
- Minto entry did not appear to be strongly associated with river environment

Relevance to temperature management for adult Chinook salmon

- Faster upstream movement may be achievable with warmer temperatures during migration
- Temperature management could encourage Minto entry, though uncertainty remains high

Uncertainties

- Faster migration may not result in earlier collection at Minto
- Benefits of earlier upstream migration timing may be offset by higher PSM or PHOS
- Radio-tagged salmon behaviors may differ from historical behaviors under various management scenarios

7. Temperature Management Scenarios

Key results

- Two temperature targets were used: PreDam and ODFW
- Two management scenarios were used: with and without a temperature control structure at Detroit Dam
- Simulations were run for a cool, high-flow year (2011) and a warm, low-flow year (2015)
- Temperature management effects were largest at Niagara and diminished downstream
- Temperature targets were reached for parts of the summer, but were reached less in the fall

Relevance to temperature management for adult Chinook salmon

- Warmer spring-summer migration conditions are achievable; cooler prespawn holding more difficult
- Temperature management effects on PSM and PHOS may be complex, with seasonally offsetting impacts
- A temperature control structure appears to provide important flexibility in achieving targets

Uncertainties

- Scenario testing with just two water years gives an incomplete picture of potential outcomes

8. Individual-Based Temperature Exposure Model for Chinook Salmon

Key results

- An individual-based movement model was developed for North Santiam River Chinook salmon
- Salmon movement rules were based on river temperatures and passage times of radio-tagged salmon
- The model includes 6 river reaches from Willamette Falls to the Minto Fish Collection Facility
- Salmon temperatures were derived from mean daily river temperatures at 4 USGS gages
- The model was used to generate continuous, full-migration temperature histories

Relevance to temperature management for adult Chinook salmon

- Quantitative spatial and temporal information is needed to assess potential impacts on salmon
- Ability to predict exposure and its consequences would be useful for in-season management decisions

Uncertainties

- Models inherently simplify behaviors, and important complexity may be missed
- Use of daily mean temperatures is an example of potential over-simplification
- Movement rules from salmon tagged in 2011-2014 may not be suitable under modified temperatures

9. Exposure Model: Application with Observed Temperatures

Key results

- The exposure model was run for five recent years (2010-2011, 2013-2015) with available temperature data
- Most salmon encountered their warmest temperatures in the Willamette main stem
- Late-run migrants encountered the highest temperatures; many had means >20 °C in the Willamette reach
- On average, salmon were coolest in the Bennett-Minto reach; means were ~12-14.5 °C
- Early-run migrants accumulated the most cumulative exposure (i.e., degree days)
- Annual estimates of mean cumulative exposure ranged from ~1,100 DD (2011) to ~1,500 DD (2015)

Relevance to temperature management for adult Chinook salmon

- In all years, thermal risks are likely to differ as a function of individual migration timing
- River conditions strongly influence how long salmon spend in reaches affected by temperature management

Uncertainties

- Uncertainties are similar to those described above for the exposure model

10. Exposure Model: Application Using Detroit Temperature Management Scenarios

Key results

- In the cool year, no-tower scenarios produced salmon histories that were similar to those with observed data
- In the cool year, yes-tower scenarios produced warmer salmon temperatures and higher DD accumulation
- In the warm year, no-tower scenarios produced warmer-than-observed histories, but with cooler periods
- In the warm year, yes-tower scenarios produced more complex outcomes, with some seasonal offsets
- Effects of all scenarios were most evident in the Bennett-Minto reach, where salmon spent the most time

Relevance to temperature management for adult Chinook salmon

- The tested scenarios indicate greater management flexibility with a hypothetical control tower at Detroit Dam
- Warm-year scenarios demonstrate that temperature targets will not always be achievable
- Biological tradeoffs are inevitable with regards to the timing and magnitude of temperature actions

Uncertainties

- Uncertainties are similar to those described above for the exposure model

Introduction

Adult spring Chinook salmon (*Oncorhynchus tshawytscha*) and winter steelhead (*O. mykiss*) migrating to their natal sites in the Willamette River basin pass through a diverse series of river reaches that include the lower Columbia River, portions of the Willamette River main stem, and sections of primary and secondary tributaries. Development and operation of the Willamette Valley Project (WVP) has substantially altered the river environments encountered by returning adult salmon and steelhead (NMFS 2008). The high-head, flood-control dams of the WVP regulate discharge and water temperature patterns in downstream reaches of tributaries and the main stem. The thermal effects of dams vary among rivers, among seasons, and with downstream distance, but general patterns include downstream river cooling in summer, river warming in late fall and winter, and reduced diel and within-season temperature variability (Rounds 2007, 2010). In the North Santiam River, specifically, water temperatures downstream from Big Cliff and Detroit dams average several °C cooler in summer, several °C warmer in fall and winter, and have far less daily and seasonal variation than would be expected without the dams (Figure 1).

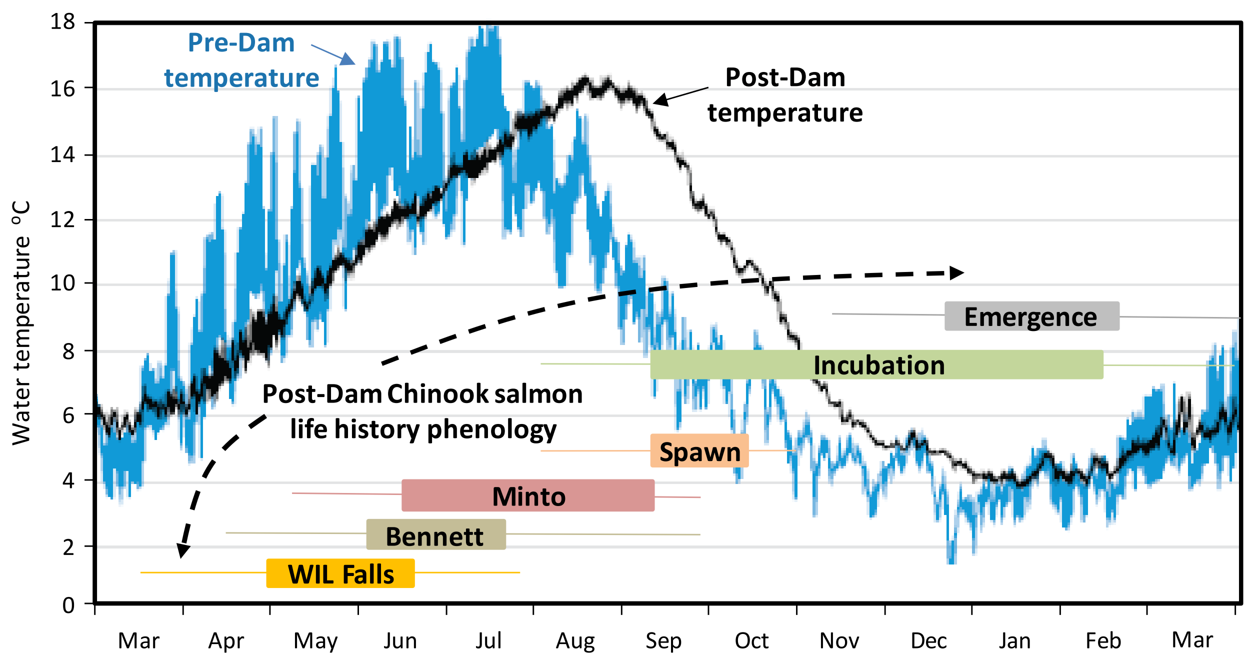


Figure 1. Conceptual diagram showing the effects of Detroit and Big Cliff dams on North Santiam River water temperature (at Niagara), in relation to spring Chinook salmon life history phenology, including adult migration timing at Willamette Falls, the Bennett dams, and at the Minto Fish Collection Facility, spawn timing, egg incubation, and fry emergence timing. Temperature data excerpted from Rounds (2010).

The 2008 Willamette Valley Project (WVP) Biological Opinion (NMFS 2008) identified water temperature management at dams as a potentially beneficial action for Chinook salmon and winter steelhead populations listed as threatened under the U.S. Endangered Species Act (NMFS 1999a, 1999b). Proposed temperature management scenarios largely seek to shift river temperatures downstream from dams towards levels that are closer to pre-dam conditions (i.e., more ‘natural’ temperatures). For example, new water temperature targets have been established for the North Santiam River to be achieved through water releases from the Detroit/Big Cliff Dam complex. The targets seek to reduce both minimum and maximum water temperatures by ~1-4 °C during summer and fall (~June-October). Targeted winter changes are smaller in magnitude (~1-2 °C) and include both slightly warmer and slightly cooler temperatures than average current conditions, depending on the season and month (see Figure 4).

The central objective of the planned changes in the North Santiam thermal regime is to shift water temperatures closer to historical seasonal levels. The biological expectation is that more natural river conditions will provide benefits to Chinook salmon and steelhead across life stages (e.g., McCullough 1999; Richter and Kolmes 2005) by matching river conditions to the adaptive traits of the populations. Potential effects of the current altered thermal and flow regime on adults include altered cues for upstream migration timing (e.g., Robards and Quinn 2002; Keefer et al. 2008a), changes in movement timing among holding and spawning habitats, changes in temperatue exposure metrics (e.g., Keefer et al. 2015), changes in adult maturation rate and timing (e.g., Sloat et al. 2014; Hearsey and Kinziger 2015), shifts in the initiation and completion of spawning, and reduced prespawn mortality (e.g., Bowerman et al. 2018). Egg development and fry emergence are likely to be accelerated under the warmer river conditions in fall and winter (e.g., Quinn 2005; Beer and Steel 2018; Fuhrman et al. 2018), perhaps resulting in poor initial rearing conditions and higher juvenile mortality (e.g., Angilletta et al. 2008). However, there is considerable uncertainty about the degree to which these biological processes may differ between the current temperature management regime and the proposed regimes. It is likely that there will be a mix of positive and negative biological outcomes, as well as some offsetting effects. For example, the fitness benefits of reduced prespawn mortality may partially offset some increased mortality associated with early fry emergence timing.

One of the primary fisheries objectives of reduced summer and fall water temperatures in the North Santiam River is to reduce premature mortality of adult Chinook salmon. The strategy has been used effectively in other salmon populations in regulated rivers (e.g., Macdonald et al. 2010, 2012). Likely PSM mechanisms associated with warm water exposure include increased disease incidence and severity (Crossin et al. 2008; Karvonen et al. 2010; Benda et al. 2015), increased stress (Cook et al. 2011; McConnachie et al. 2012; Jeffries et al. 2014), and higher energetic and cardiovascular costs (Eliason et al. 2013). Warm temperatures in the Willamette River main stem and in lower reaches of tributaries have been linked to high *en route* adult migration mortality (Schreck et al. 1994) and high prespawn mortality (PSM) of Chinook salmon in many Willamette basin tributaries (Keefer et al. 2010; Bowerman et al. 2016, 2018; Naughton et al. 2016).

Warm water exposure of adults has also been linked to lower fitness among fish that successfully spawn, primarily due to reduced gamete quality in both males and females (Schreck et al. 2001; Mann 2007; Pankhurst and King 2010; Fenkes et al. 2017). Understanding the effects of temperature exposure on adults is therefore critically important for in-season management of the adults collected at the Minto adult collection facility on the North Santiam River (and elsewhere). The timing of adult collection, fish exposure prior to collection, and conditions at holding facilities, for example, are important factors in decisions about which fish should be used for broodstock or outplanted to spawn in wild (e.g., Keefer et al. 2010; Benda et al. 2015; DeWeber et al. 2017). Currently, our understanding of the thermal exposure of adult Chinook salmon is limited, particularly in relation to questions about potential thresholds for mortality and the interactive effects of temperature and other risk factors (e.g., adult density, pathogenic processes, and/or multiple stressors). It is also probable that different subsets of the population (e.g., early- versus late-timed fish) are vulnerable to different types of risks, but within-population variation has received relatively little research attention.

Importantly, while cooler summer water temperatures resulting from operational changes at Detroit and Big Cliff dams should benefit adult Chinook salmon, the changes may also provoke some unintended negative effects. These include adult behavior changes, such as longer staging or holding periods, slowed maturation, and/or delayed spawning onset and completion. Protracted holding may increase the risk of time-sensitive PSM mechanisms such as pathogen progression (Miller et al. 2014; Benda et al. 2015) and exposure to predators or recreational fisheries. Temperature-induced behavioral changes may also reduce collection efficiency of hatchery-origin adults at the Minto Fish Collection Facility and thereby increase the proportion of hatchery-origin spawners (PHOS) and spawner density in the North Santiam River and its tributaries. Hatchery-origin Chinook salmon spawning in the wild have lower reproductive success than their wild counterparts in other Willamette River tributaries (Sard et al. 2015). High PHOS has also been associated with both elevated PSM in Chinook salmon (Bowerman et al. 2018) and reduced population-level fitness for other Pacific salmonids (Araki et al. 2008; Christie et al. 2012). A better understanding of these processes is needed.

The research objectives addressed in this report will progress from collection and assembly of existing data, to calculation of summary metrics of river environment, salmon behaviors, and survival, to analyses of the relationships among variables under recent 'baseline' temperature conditions. Subsequent report sections will summarize several proposed temperature management scenarios for the North Santiam River, using two temperature targets and the presence or absence of a hypothetical temperature control structure at Detroit Dam (details below). The final report section describes an individual-based model we developed to estimate how the various temperature management scenarios may affect acute and cumulative thermal exposure of adult Chinook salmon during migration and prespawn holding in the North Santiam River. In combination, the several report elements address adult salmon information gaps (i.e., the relationships among river environment and adult salmon behavior and survival metrics) and present an analytical tool that can be used to evaluate potential risk-benefit tradeoffs associated with the proposed temperature management strategies.

Study Objectives, Data Sources, and Methods

This section of the report provides a brief overview of how we addressed the various study components. Additional context and methodological details are provided in objective-specific sections. Importantly, this project largely entailed assembling and interpreting information from existing datasets. With the exception of the radiotelemetry projects, we did not participate in the original study designs, sampling and survey methods development, data collection, data archiving, or data quality assessment and control.

Objective 1: Assemble and summarize existing data related to adult Chinook salmon migration behaviors and survival in the North Santiam River, to address current management information gaps

The North Santiam River basin is a good location to evaluate temperature management strategies and how they may affect adult Chinook salmon behaviors and survival. The sub-basin has relatively few spawning tributaries and hatcheries and there are several high-quality existing salmon and river environment datasets. Water temperature and discharge monitoring at multiple gage sites (Figure 2) provide longitudinal characterization of the thermal environment encountered by adults during migration and main stem holding and spawning. Upstream migrants are enumerated at the Bennett dams, which are downstream from most of the high-quality salmon spawning habitat, and at the Minto Fish Collection Facility, where trapping and sorting occurs. The presence of two enumeration sites facilitates accounting for adult migrants, particularly when combined with existing spawning ground survey data. The ODFW_OWCS (Oregon Department of Fish and Wildlife, Willamette Chinook salmon) redd and carcass datasets provide baseline information on when and where Chinook salmon spawn, their demographics (e.g., hatchery-origin, natural-origin, sex ratios, etc.), and PSM and PHOS rates. The adult radiotelemetry studies conducted in 2011-2014 (e.g., Jepson et al. 2015; Keefer et al. 2015, 2017) provide another layer of information, with detailed histories of individual salmon movement rates and timing that can be used to help parameterize and validate predictive models.

River environment – The U.S. Geological Survey (USGS) has archived North Santiam River discharge and water temperature data for three main stem gage sites downstream from Big Cliff Dam: at Greens Bridge (near the confluence with the main Santiam River); near Mehama (between the Bennett dams and Minto); and near Niagara (below Big Cliff Dam, Figure 2). We also assembled data from two other USGS sites, including the Newberg site on the main stem Willamette River and the Little North Santiam River site near Mehama. We downloaded available daily data from 2000–2018 from the USGS website. Temperature time series at these sites varied in duration.

Adult Chinook salmon are present in the North Santiam River primarily from April through October and we calculated river environment metrics from this period. In initial data exploration, we found that there was considerable within-year autocorrelation among weekly, monthly, and seasonal temperature metrics and among daily minimum, mean, and maximum temperatures. To simplify, we selected two metrics for use in summaries and analysis: (1) the monthly mean of daily mean values of temperature and discharge; and (2) the annual maximum of the 7-d moving average of daily maximum temperatures (e.g., 7DADM, US EPA 2003). The latter is an indicator of the warmest conditions salmon likely encountered in a given year.

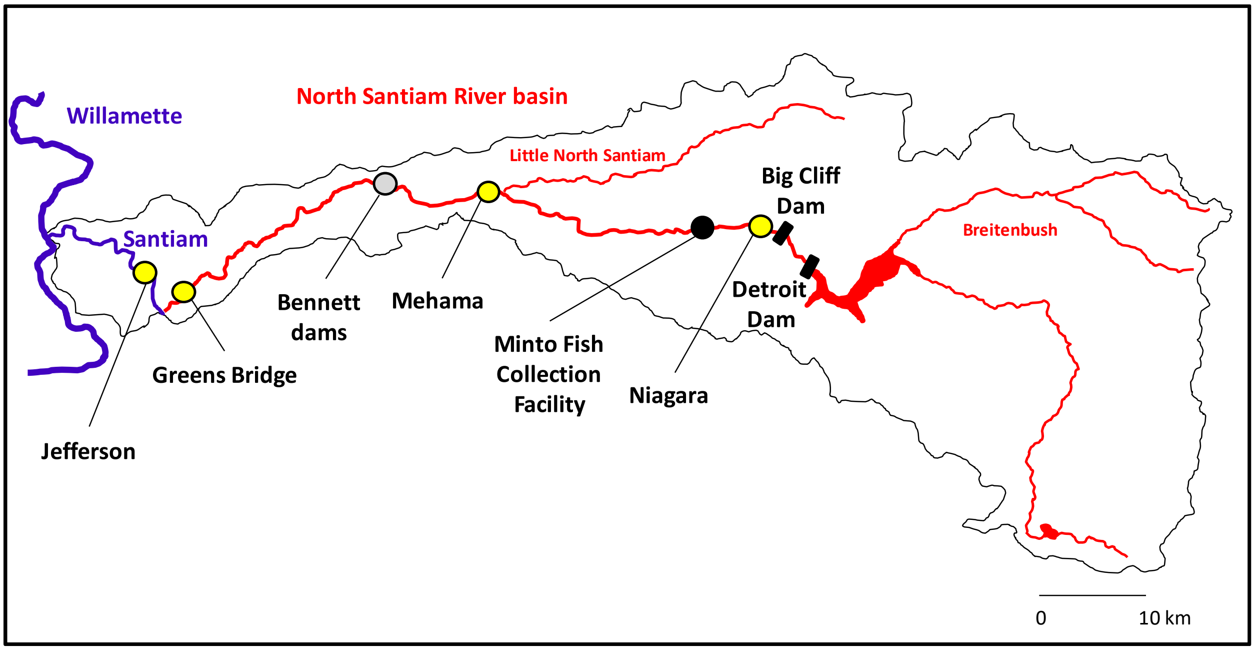


Figure 2. Map of the North Santiam River sub-basin, showing the approximate locations of four USGS gage sites (yellow circles), the Bennett dams, Minto Fish Collection Facility, and Big Cliff and Detroit dams.

Chinook salmon counts – Adult Chinook salmon and steelhead count data in the North Santiam River basin have been collected at the Bennett Dams by ODFW and at the Minto Fish Collection Facility by ODFW and USACE. We solicited existing count datasets from these agencies. Daily counts of adult Chinook salmon at Willamette Falls from 2001–2018 were downloaded from the ODFW fish counts website. The temporal resolution (e.g., daily versus weekly versus monthly counts), origin specificity (e.g., clipped versus unclipped adults), time series duration, and data quality varied considerably among locations. Time series from the Bennett dams and Minto Adult Fish Facility were shorter or less complete due to logistical challenges, such as difficulty counting adults at the Bennett dams and the intermittent (i.e., non-daily) processing of adults collected at the Minto facility. Count data were generally of lower temporal resolution and quality prior to the structural modifications at Bennett and Minto (new facility began operating in 2013).

Adult Chinook salmon count and collection data from Willamette Falls, the Bennett dams, and the Minto Fish Collection Facility were used to calculate migration timing statistics for each year with available data.

Chinook salmon redds – The ODFW_OWCS database has information on Chinook salmon redd surveys conducted in the North Santiam River basin from 2000–2016. In total, there were >3,200 reach×date redd counts collected over multiple river reaches. The dataset includes reach×date surveys where no redds were observed. Redd data have been used to assess salmon distribution, spawn timing, and escapement, but inferences can be sensitive to sample design (e.g., Hoffnagle et al. 2008; Murdoch et al. 2009, 2010; Bowerman et al. 2016). We used the redd data to calculate annual timing statistics in aggregate, including the range, relative abundance, and peak counts through time.

Chinook salmon carcasses, prespawn mortality (PSM), and PHOS – The ODFW_OWCS database has details on >4,700 adult salmon carcasses collected in the basin from 2000–2016. Each carcass was associated with a specific river reach and a variety of individual details are included in the dataset, including (when assessment was possible) date, sex, fin clip status, fork length, spawning success (females only), etc. In total, ~2,200 female Chinook salmon carcasses were in sufficiently good condition that spawning success and prespawn mortality could be assessed (assessed by percent egg retention). We used the collection dates for successfully-spawned females to calculate spawn timing metrics, recognizing that collection dates were likely 1-3 d later than spawn dates.

Prespawn mortality (PSM) was calculated as the proportion of female carcasses collected in a reach that did not successfully spawn based on established egg retention criteria. We only included females with known spawning success, as assigned in the ODFW-OWCS carcass dataset (i.e., females with 'unknown' spawning success were excluded). Annual PSM estimates were generated for specific survey reaches and for longer river sections that included multiple survey segments. We only present estimates for reaches or river sections that included at least ten female carcasses in a year.

The presence/absence of fin clips was evaluated for almost all carcasses. We estimated PHOS – the annual proportion of carcasses (males and females) collected in a reach that had a hatchery fin clip – from this dataset using only fin clip status. We calculated PHOS for specific survey reaches and for longer river sections that included multiple survey segments. Some unmarked hatchery-origin salmon were likely collected and were treated as natural-origin fish in our summaries, likely resulting in some PHOS underestimation. The ODFW_OWCS dataset does include otolith data for ~1,600 North Santiam Chinook salmon, and about a third of these were identified as hatchery-origin (presumably based on thermal marks on otoliths). We did not use otolith data in our evaluations due to uncertainties related to matching carcass data with otolith data in the dataset available.

Chinook salmon radiotelemetry – Adult Chinook salmon were collected and radio-tagged at Willamette Falls in 2011–2014 and monitored during migration (Jepson et al. 2015). About 160 of these salmon were last detected in the North Santiam River and their migration histories were used to summarize adult movements and migration timing along their route from Willamette Fall to the Minto facility. Several of the radio-tagged salmon also carried archival temperature loggers, which provided full-migration thermal exposure histories.

We calculated migration timing and reach-specific salmon passage and residence times for the tagged fish at Willamette Falls, upon Santiam and North Santiam River entry, at the Bennett dams, and upon arrival at Minto. Reach passage times were calculated between the first detection at the antenna at the reach start point to the first detection at the antenna at the reach end point.

Collection efficiency at Minto – We originally planned to estimate adult Chinook salmon collection efficiency at the Minto Fish Collection Facility by using the several adult data sources: counts at the Bennett dams, collection numbers at Minto, and redd and carcass survey data. Conceptually, collection efficiency could be calculated by dividing the number collected at Minto by the number that passed the Bennett dams, adjusted for the numbers of spawners in reaches upstream from Upper Bennett Dam (i.e., in the main stem North Santiam and in the Little North Santiam River). After preliminary analyses we rejected this approach, both because there was considerable uncertainty regarding the count data at the Bennett dams and because of the many assumptions required to estimate spawner abundance from redd and carcass data.

Objective 2: Assess relationships among Chinook salmon migration timing, PSM, PHOS, and North Santiam River environment metrics, to better quantify how recent river and dam management may influence the adult salmon metrics

Chinook salmon migration timing– We used correlation and regression analyses to evaluate the relationships among river environment metrics and adult Chinook salmon migration timing at Willamette Falls, the Bennett dams, and Minto. In these analyses, annual run timing metrics (i.e., median passage date, quartile passage dates) were the dependent variable and individual monthly mean environmental variables and/or the 7-d maximum temperature were independent variables. We also used general linear models (GLM) to evaluate the effects of multiple environmental covariates on migration timing. Our intention with both the univariate and multivariate models was to identify general relationships that might inform our evaluation of PSM rather than to build a predictive run-timing model as might be applied to manage fisheries or dam operations (e.g., Flynn et al. 2006; Peterson et al. 2018).

Chinook salmon PSM and PHOS – We evaluated relationships among Chinook salmon migration timing, river environment, PSM and PHOS using a similar correlation and regression approach. In the PSM analyses, we tested for effects of the monthly river environment metrics, the 7DADM metric, annual run timing (e.g. median migration date) at Willamette Falls, and annual PHOS. The dependent variable was annual PSM estimates, either from individual carcass survey reaches (if there was a sufficient number of years) or from several aggregated reaches. Our approach was similar for the PHOS evaluation, except 7DADM and run timing metrics were not considered as covariates. PHOS analyses were run for specific carcass survey reaches and for aggregated reaches.

Objective 3: Generate several water temperature management 'scenarios' based on proposed changes at Detroit Dam, including use of a hypothetical temperature control structure

Detroit Dam temperature management scenarios – The USGS developed a series of CE-QUAL-W2 and HEC-RAS models to predict discharge and water temperatures in the North Santiam River basin, including Detroit and Big Cliff lakes (e.g., Sullivan et al. 2007; Buccola et al. 2012, 2015; Rounds and Buccola 2015). These models have been used to evaluate the effects of a variety of potential dam operations, including the use of hypothetical temperature control structures at Detroit Dam. For our project, the models were used to generate daily river temperature data under four potential temperature management scenarios. The simulations were based on two temperature ‘targets’ (Figure 3): (1) a pre-dam target based on long-term, year-round temperature estimates for the North Santiam River prior to dam construction; and (2) a year-round target developed by ODFW in 2017 to benefit Chinook salmon. Both targets would slightly raise temperatures during spring and early summer, during the upstream migration period, and would lower temperatures during the summer and fall holding period (e.g., Figure 4). Nested within the two targets were two temperature management operations for Detroit Dam: (1, ‘yes-tower’) use of a hypothetical temperature control structure; and (2, ‘no-tower’) use of only the existing structures (i.e., regulating outlets, power outlets, and spillway) for temperature management. The ‘no-tower’ scenarios are similar, in principle, to operations at Detroit Dam since ~2007, when use of water from the spillway became more routine.

The temperature control tower scenarios assumed that a structure would intake Detroit reservoir near-surface (~2.4 m, 8 ft) water that would be mixed with water from the existing upper regulating outlet (elevation ~406 m, 1,332 ft). This scenario would have no limitations placed on power production minima because it is assumed that all outflow would be released through turbines (up to powerhouse capacity). The no-tower scenarios assume a minimum of 40% of the outflow dedicated to power outlets year-round.

Temperature simulation years – The North Santiam River temperature models were run for two example years: 2011 and 2015. These years were selected in consultation with USACE and represent a relatively cool, high-flow year (2011) and a very warm, low-flow year (2015). The model inputs included the observed environmental data from the respective years. Iterations of the model runs sought to reach the pre-dam and ODFW temperature targets downstream from Big Cliff Dam, with scenarios for each year that did or did not include the hypothetical temperature control structure. As expected, the temperature targets were achieved for only portions of each model run due to the constraints imposed by the observed hydrological and temperature input data in each year (see Results). We calculated the differences between observed mean daily and monthly temperatures in each year and river reach and the corresponding temperatures generated in the four management scenarios.

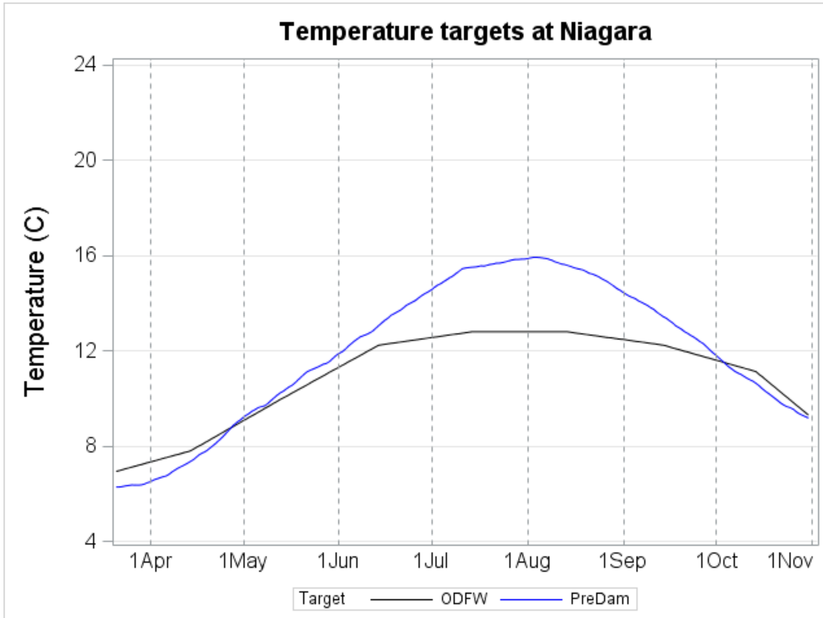


Figure 3. Mean daily water temperature (°C) targets downstream from Big Cliff Dam (at Niagara) that were applied in the temperature management scenarios. Targets were the estimated without-dams (PreDam) temperatures described by Buccola et al. (2012, 2015), and the temperature recommendations for Chinook salmon developed by ODFW in 2017 (ODFW).

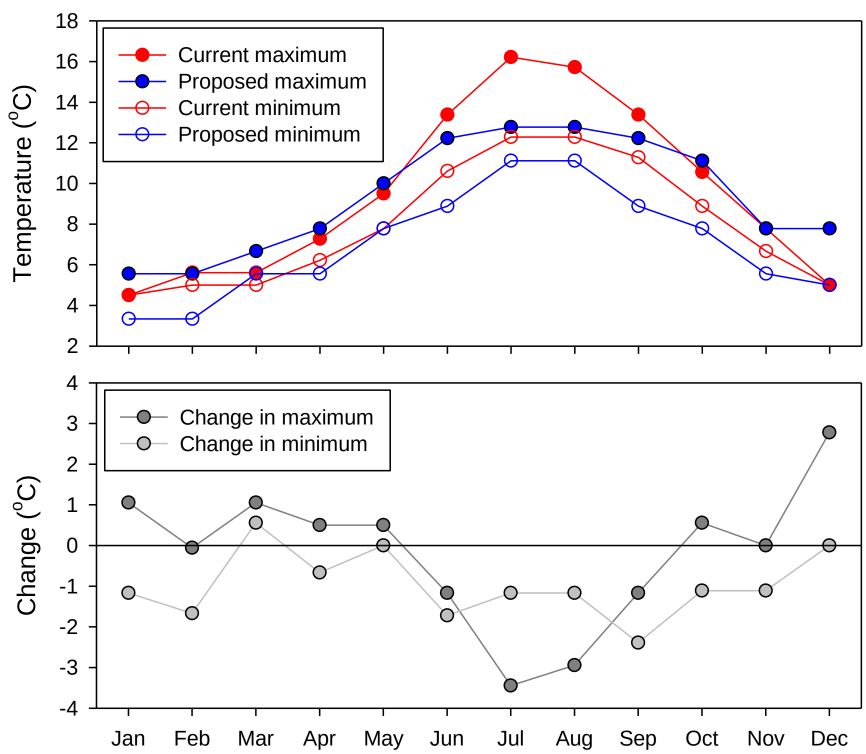


Figure 4. Current and proposed North Santiam River monthly minimum and maximum temperature targets (top) and change in temperature if the proposed ODFW targets are realized (bottom). Data source: RY18 Willamette RM&E Concept Paper APH-18-03. (Note: Oct and Nov interpolated from <10 °C).

Objective 4: Develop an individual-based model of adult Chinook salmon thermal exposure to assess potential risks and benefits of the temperature management scenarios

Chinook salmon thermal exposure model – The summaries and analyses from Objectives 1 and 2 indicated that Chinook salmon run timing, PSM, and PHOS were inter-related with Willamette River and North Santiam River temperature and discharge (see Results). The migration timing data further suggested that Chinook salmon from different portions of each annual run encountered substantially different river conditions during migration and highly variable prespawn holding duration. We hypothesize that within-run and among-year variation in the environmental experiences of individual adults is associated with a gradient in mortality risk factors (Figure 5). More specifically, we hypothesize that risks for early-run adults are related to total time in freshwater, with longer times associated with increased likelihood of disease- or senescence-related mortality. Indicators for these risks might include total residency time and cumulative degree days. Risks for late-run adults are more likely related to exposure to acutely stressful summer river temperatures, and measures of mean or maximum exposure may be more appropriate metrics.

To test these types of hypotheses, we developed an individual-based model (IBM) of thermal exposure. We parameterized the upstream movement portion of the IBM using the radiotelemetry data and the relationships between Chinook salmon migration speed and river temperature. In the model, simulated salmon movements were matched to reach-specific water temperatures using daily time steps. This allowed us to assemble continuous thermal histories from Willamette Falls to the Minto Fish Collection Facility that included active migration and prespawn holding periods.

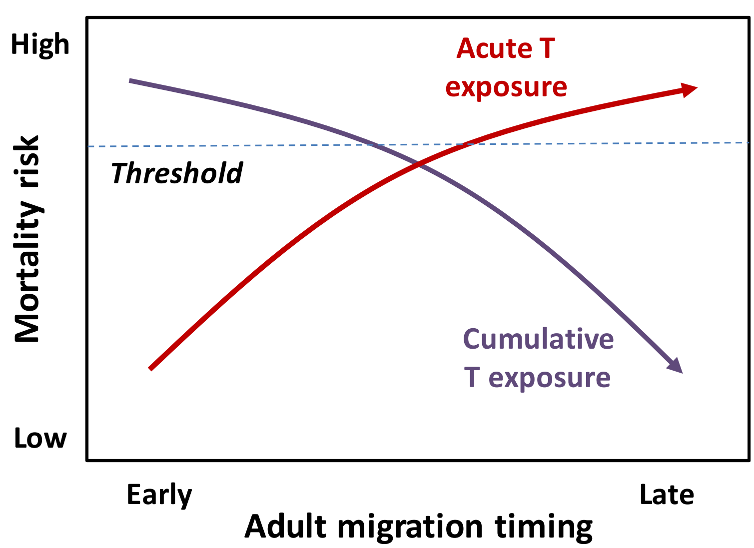


Figure 5. Hypothetical relationship between adult Chinook salmon migration timing and two temperature-mediated risk factors: exposure to acutely-stressful condition in the migration corridor versus long freshwater residence times and high cumulative exposure. The dashed horizontal line represents possible threshold values.

Exposure model applied to observed temperature data – Five recent years (2010, 2011, 2013, 2014, 2015) had near-continuous USGS temperature data at the Newberg, Greens Bridge, Mehama, and Niagara sites during the period of adult migration and holding (~March to October). In a series of model runs, we seeded the IBM with randomly-selected groups of adult Chinook salmon drawn from the five annual run-timing distributions at Willamette Falls. Results were used to characterize and compare the thermal exposure of individuals over their full migrations and in specific river reaches among years.

Exposure model applied to Detroit Dam temperature management scenarios – To assess the potential effects of temperature management at Detroit Dam, we ran the IBM using the four scenarios described previously (i.e., the pre-dam and ODFW targets, with and without a hypothetical temperature control structure). Temperature exposure metrics from the simulated 2011 and 2015 scenarios were compared to output from 'observed' temperature conditions in those years.

Results and Discussion

Summary of Existing Data

1. River environment data

Near-continuous daily time series or river discharge were available for all five USGS gage sites over the 2000–2018 evaluation period, whereas water temperature data were less consistently collected (Table 1). Of the five sites, only Niagara had temperature data in all years. Temperature monitoring at the Greens Bridge site near the North Santiam River mouth began in 2010. Temperature monitoring ended at the North Santiam gage near Mehama in September 2015 and at the Little North Santiam gage (also near Mehama) in December 2015.

Table 1. Years that mean daily water temperature data were available for USGS monitoring sites. The Greens Bridge, Mehama, and Niagara sites were on the main stem North Santiam River (for specific locations see Figure 2). Discharge data were available at all sites in all years. (Note: the Mehama temperature gage site changed to #14183010 in fall 2009.)

Year	Newberg #14197900	Greens Bridge #14184100	Mehama #14183000	Niagara #14181500	Little N. Santiam #14182500
2000				x	
2001			x	x	x
2002	x[4]		x	x	x
2003	x[5]		x	x	x
2004	x		x	x[3]	x
2005	x		x	x	x
2006	x		x	x	x
2007	x		x	x	x
2008	x[6]		x	x	x
2009	x		x[1]	x	x
2010	x	x	x	x	x
2011	x	x	x	x	x
2012	x[7]	x	x	x	x
2013	x[8]	x	x	x	x
2014	x	x	x[2]	x	x
2015	x	x	x	x	x
2016	x	x		x	
2017	x	x		x	
2018	x	x		x	

[1] June missing ≥3 days; [2] August missing ≥3 days; [3] April missing ≥3 days; [4] April missing ≥3 days
[5] June missing ≥3 days; [6] April and May missing ≥3 days; [7] April, July, August, September, and October missing ≥3 days;
[8] August missing ≥3 days

Water temperature **–** In most years, water temperatures during the adult Chinook salmon migration and holding periods were warmest in the Willamette River main stem and were progressively cooler moving upstream in the North Santiam River at the Greens Bridge, Mehama, and Niagara sites (Figures 6–8). Within-day and within-season temperature variation was generally highest at the Greens Bridge and Mehama sites (Figure 6). Low within-day variation in the Willamette River can be attributed to the stabilizing and inertial effects of high discharge. In contrast, low variation at the Niagara site was largely due to water discharge operations at Detroit and Big Cliff dams. Starting in 2007, summer operations at the dams produced somewhat higher within-day temperature variation at the Niagara site (Appendix A).

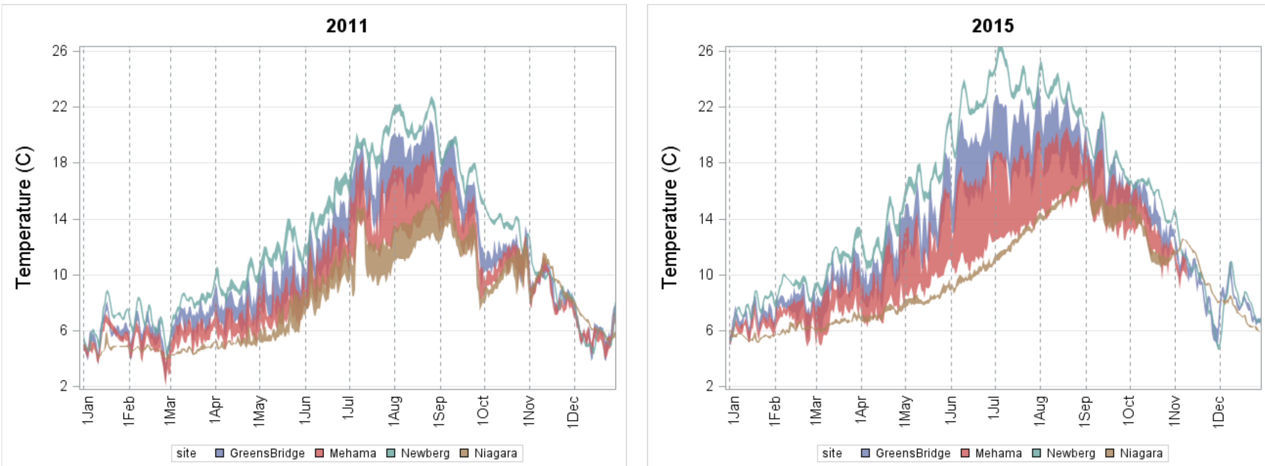


Figure 6. Annual thermographs for four USGS gage sites: Newberg (Willamette River), and Greens Bridge, Mehama, and Niagara (North Santiam River). Data are daily minimums and maximums from 2011 (recent cool year) and 2015 (recent warm year). Appendix A has annual graphs for 2000–2018.

In April, mean water temperatures at the three North Santiam sites were in a relatively narrow range between ~5–8 °C across years (Figure 7). Within-year differences among the three sites increased in May, when means ranged from ~6–13 °C. Differences among years were most evident in the summer months of June–August: means in these months were ~8 –14 °C (Niagara), ~9–16 °C (Mehama), and ~12–20 °C (Greens Bridge, Figure 6). Temperatures were lower in September, with means ranging from ~10–16 °C across sites. On average, the warmest years in the time series from April–September (the period that corresponded with upstream migration of adult Chinook salmon and initial holding) were 2015 and 2016 (Greens Bridge), 2013 and 2015 (Mehama), and 2014 and 2015 (Niagara).

In general, monthly mean water temperatures were correlated across gages sites (e.g., a warm July at one site was usually a warm July at other sites). However, there were important departures from this pattern (Figure 8). Unsurprisingly, the highest correlations ($0.94 \leq r \leq 0.99$) were between monthly values at Mehama and Greens Bridge, sites that were in close proximity and were both downstream from the Little North Santiam River. Correlations were also relatively high between the Niagara and Mehama ($0.73 \leq r \leq 0.99$) and between Niagara and Greens Bridge ($0.61 \leq r \leq 0.93$). In contrast, correlations were much lower and more variable between the North Santiam temperatures and those in the Little North Santiam and Willamette rivers. For example, correlations between Niagara and Newberg sites were fairly high in May and June ($0.53 \leq r \leq 0.78$) but were only weakly correlated in summer and fall ($-0.02 \leq r \leq 0.31$); patterns were broadly similar in comparisons between Niagara and the Little North Santiam (Figure 8).

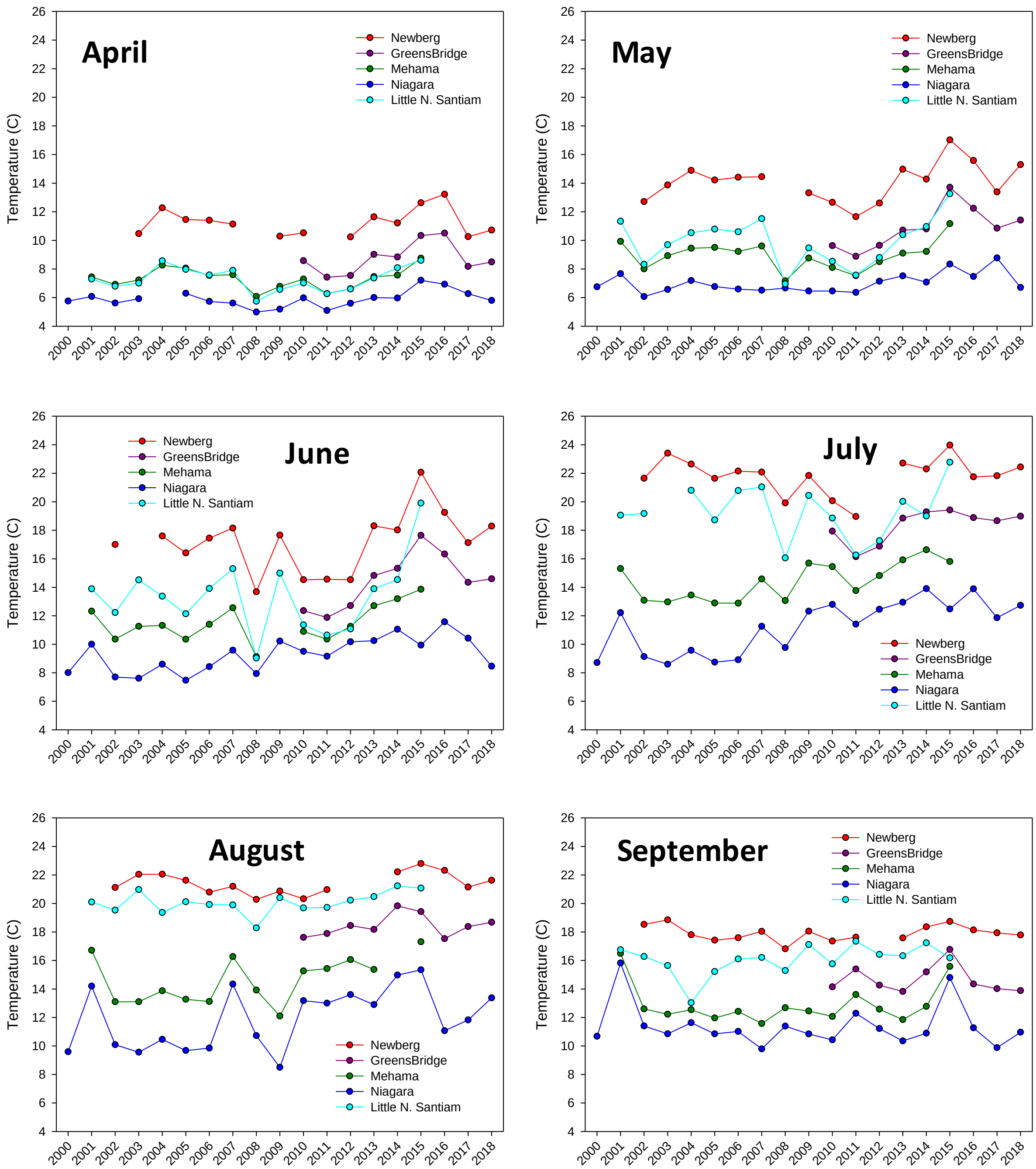


Figure 7. Mean monthly water temperature (°C) at USGS gage sites in the study area, 2000–2018.

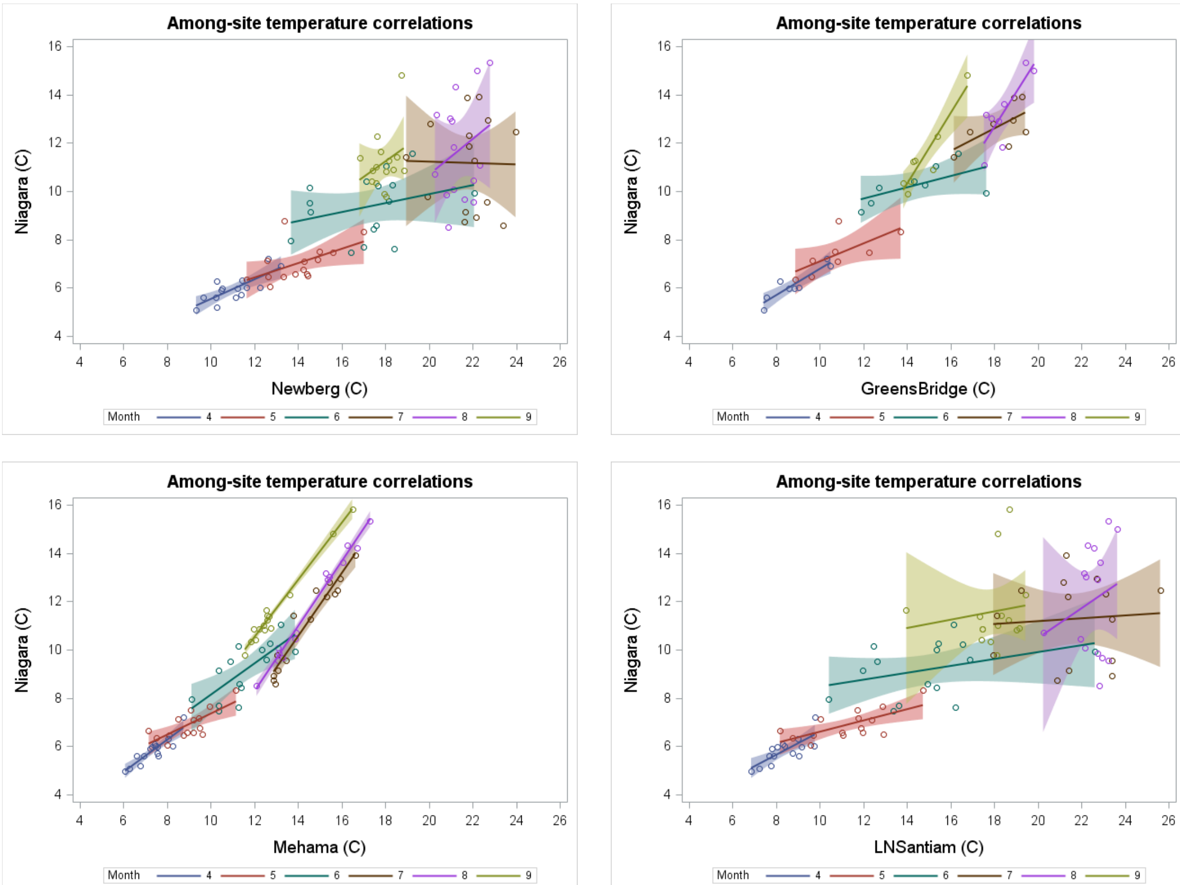


Figure 8. Linear regression relationships among mean monthly (April–September) water temperatures (°C) in the North Santiam River at the Niagara gage and the gages at Newberg (Willamette), Greens Bridge (N. Santiam), Mehama (N. Santiam), and Little N. Santiam gages. Shaded areas show 95% confidence limits for monthly regression lines. Numbers of years vary – see Table 1.

The timing and magnitude of the 7DADM values varied substantially among years and monitoring sites (Table 2). At Newberg, peak annual temperatures occurred from early July through late August and were all > 22 °C; the warmest 7DADM was 26.11 °C in 2014. In the shorter time series for Greens Bridge and Mehama, peak temperatures also occurred in July and August. All peaks were > 20 °C at Greens Bridge and were > 18 °C at Mehama. The warmest temperatures at Niagara were in September or October in 2000–2006 but were mostly in late July or August from 2007–2018, reflecting differences in dam operations early and late in the time series (Table 2; also see Figure 11). 7DADM values at Niagara were all > 13 °C and were > 16 °C in five years (2001, 2007, 2009, 2014, and 2015).

River discharge – In the Willamette River, mean monthly discharge ranged from ~7,200 cfs in August to ~31,000 cfs in April (Figure 9). Among-year variability in monthly means was lowest in August (coefficient of variation [CV] = 13.1% and highest in June (47.5%) and April (38.4%). On average, the highest April–September discharge in the time series was in 2010 and 2012; the lowest discharge was in 2007 and 2015. In all months, Willamette River discharge was negatively correlated with Willamette River temperature (Figure 10).

Table 2. Annual dates and values for maximum 7-d average daily maximum (7DADM) temperatures at four USGS gage sites.

	Newberg		GreensBridge		Mehama		Niagara	
Year	Date	T_{max}	Date	T_{max}	Date	T_{max}	Date	T_{max}
2000	-	-	-	-	-	-	17 Oct	13.46
2001	14 Jul	23.16	-	-	-	-	15 Sep	16.26
2002	22 Jul	24.64	-	-	-	-	17 Oct	13.66
2003	22 Jul	24.49	-	-	-	-	20 Oct	14.00
2004	29 Jul	23.33	-	-	-	-	16 Oct	14.00
2005	26 Jul*	24.59	-	-	-	-	25 Oct	13.67
2006	13 Jul	23.94	-	-	-	-	20 Oct	13.66
2007	16 Aug	22.64	-	-	-	-	23 Aug	16.54
2008	31 Jul	25.50	30 Jul	24.03	-	-	11 Aug	14.50
2009	16 Aug	22.09	27 Jul	21.33	26 Jul	18.96	25 Jul	16.10
2010	26 Aug	22.27	24 Aug	20.59	24 Aug	18.50	6 Aug	15.69
2011	-	-	15 Aug	21.40	14 Aug	19.24	5 Sep	15.79
2012	13 Jul	23.77	23 Jul	21.69	22 Jul	18.80	17 Aug*	15.29
2013	3 Aug*	23.79	1 Aug*	22.76	-	-	29 Jul	14.94
2014	5 Jul	26.11	3 Jul	22.44	18 Aug	20.01	5 Aug	16.71
2015	29 Jul	23.89	27 Jul	22.79	-	-	31 Aug	16.73
2016	3 Aug	23.89	1 Aug	21.69	-	-	28 Jul	15.76
2017	29 Jul	24.44	28 Jul	23.03	-	-	9 Jul	14.34
2018	-	-	-	-	-	-	30 Jul	15.80

*Maximum occurred on more than one date.

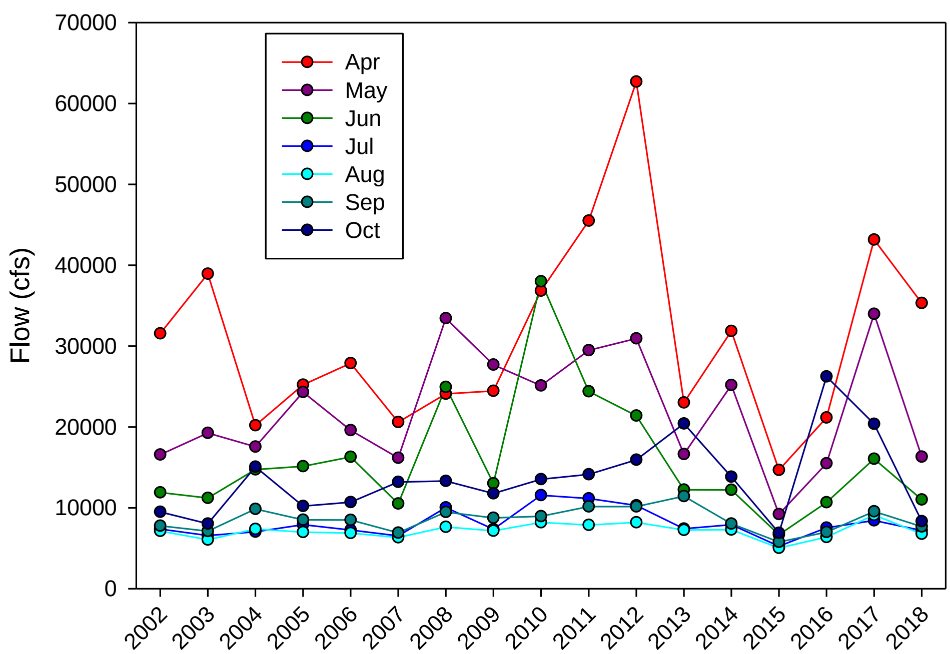


Figure 9. Mean monthly Willamette River discharge (cfs) at the USGS Newberg gage, 2002–2018.

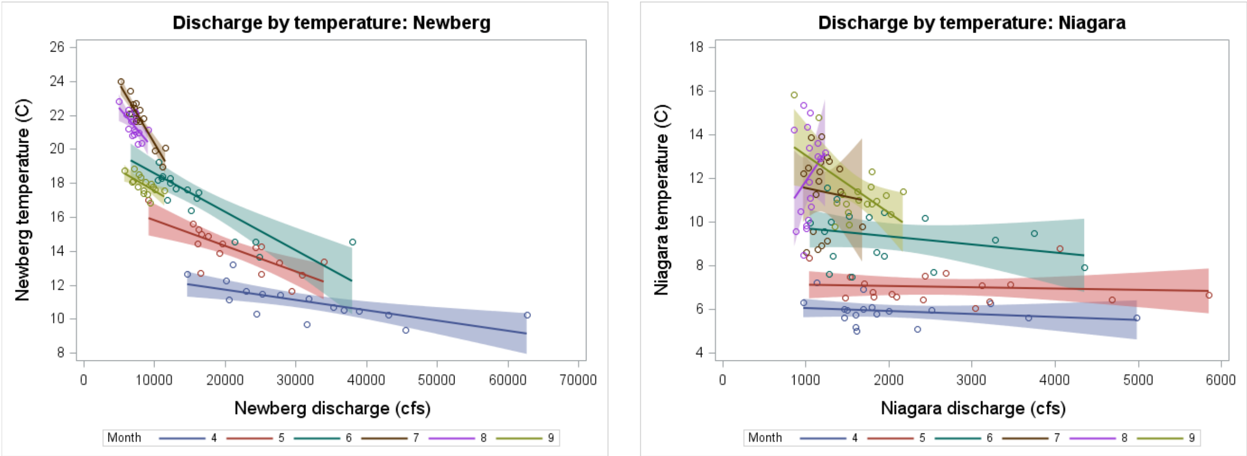


Figure 10. Linear regression relationships among mean monthly (April–September) river discharge (cfs) and water temperature (°C) in the Willamette River at the Newberg gage and in the North Santiam River at the Niagara gage. Shaded areas show 95% confidence limits for monthly regression lines. Numbers of years vary – see Table 1

In the North Santiam River, discharge was highly correlated across monitoring sites within year in most months (Figure 11). Discharge varied little among years during the low-flow period in July and August (CV = 14.6% and 9.5%, respectively). In contrast, among-year variability was relatively high in April (CV = 49.5%), May (46.3%), and June (47.3%). Some of the variation can be explained by the operational shift that occurred after 2007. The highest discharge years differed by month, but most were in 2008–2012 or 2017 (Figure 10). Unlike in the Willamette River, monthly discharge and water temperatures in the North Santiam River were inconsistently and weakly correlated across years (Figure 10), reflecting the effects of water releases at Detroit and Big Cliff dams.

Uncertainties and recommendations – The 2015 decommissioning of temperature data collection at the Mehama and Little North Santiam gages has introduced some environmental data uncertainty in the North Santiam River basin. The Little North Santiam site was the only routine monitoring in that tributary and it may be difficult to predict temperatures there given the low correlations with temperatures at the main stem North Santiam gage sites. The higher correlations between temperatures at the Niagara and Mehama gages suggest that temperatures at Mehama should be straightforward to model; however, collection of temperature data at the Mehama site would be valuable for future assessments of temperature control actions at the dams.

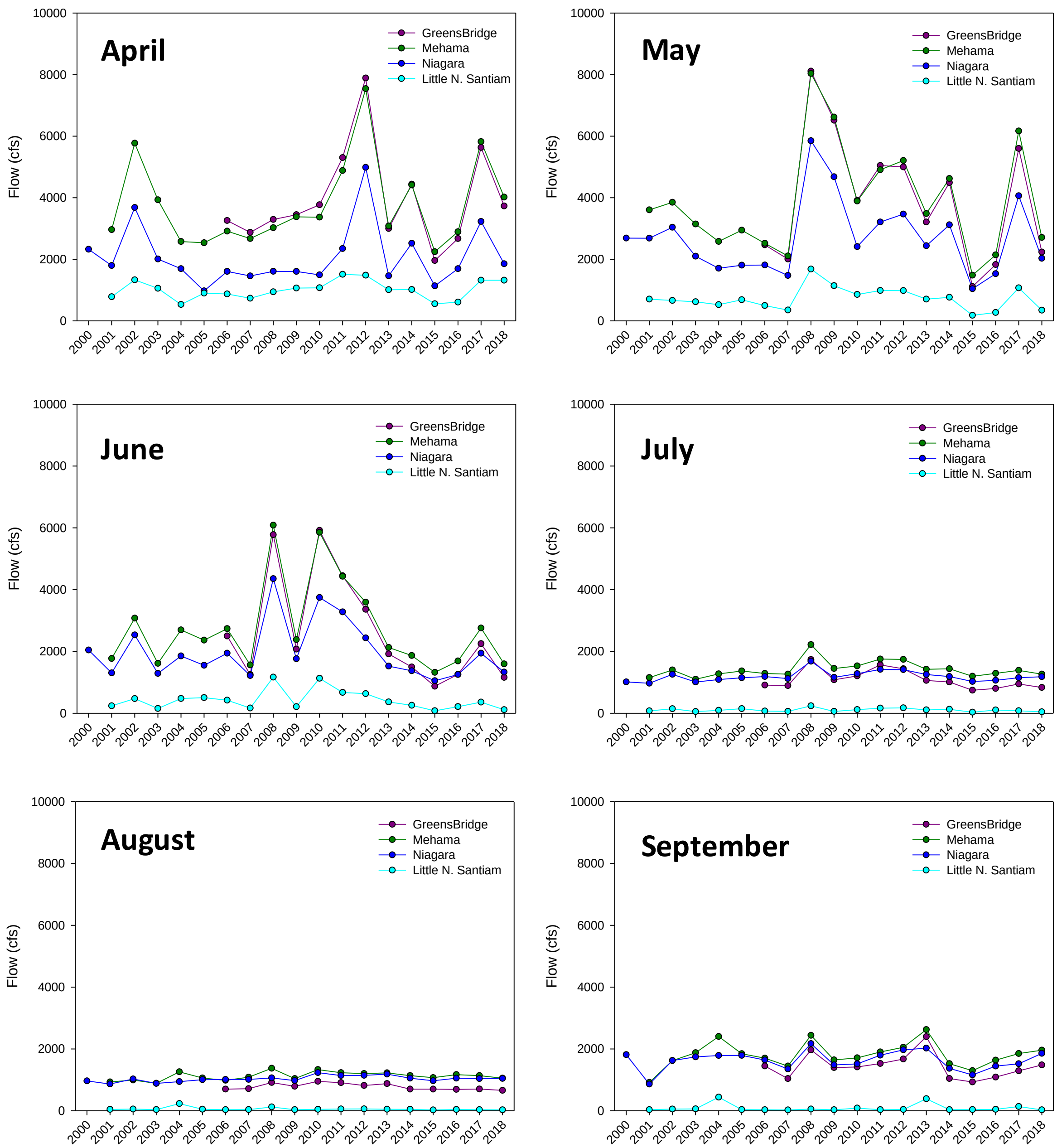


Figure 11. Mean monthly discharge (cfs) at USGS gage sites in the study area, 2000–2018. See Figure 8 for Newberg discharge.

2. Migration timing of adult spring Chinook salmon

Willamette Falls **–** The timing of annual spring Chinook salmon runs at Willamette Falls varied widely from year to year, at least partially as a function of highly variable river conditions in the spring (see Figure 9). Within year, large day-to-day changes in adult salmon counts are very common at Willamette Falls (Appendix figures B1-B3). Erratic passage at the falls is presumably related to proximate fluctuations in salmon arrival from the lower Columbia and Willamette rivers, proximate cues at the Falls (e.g., river discharge and temperature), and the effects of fisheries and pinniped activity (e.g., Wright et al. 2015).

In most years, the earliest arriving adults passed Willamette Falls in March, and the first 5% of the runs have typically passed by mid-April (Figures 16 and 17, Appendix Table B1). In contrast, quartile and median passage dates vary much more widely among years. In the 2001-2018 time series, the earliest median date was 6 May in warm, low-flow 2015 and the latest median was 11 June in cool, high-flow 2008. The average median date was about 20 May.

Annual quartile and median passage dates were negatively correlated with mean monthly water temperatures and positively correlated with mean monthly river discharge (Figure 18). With each 1 °C increase in mean March water temperature the median Chinook salmon passage date was earlier by about 6 d (Figure 19; linear regression, r^2 = 0.44, P = 0.004, n =17 years). In the mean May discharge model, each 5,000 cfs increase in discharge pushed the median passage date later by about 5 d (Figure 19; r^2 = 0.49, P = 0.002, n =17). In exploratory multiple regression models, a mixture of temperature and discharge variables generally produced the highest r^2 values for a given number of covariates.

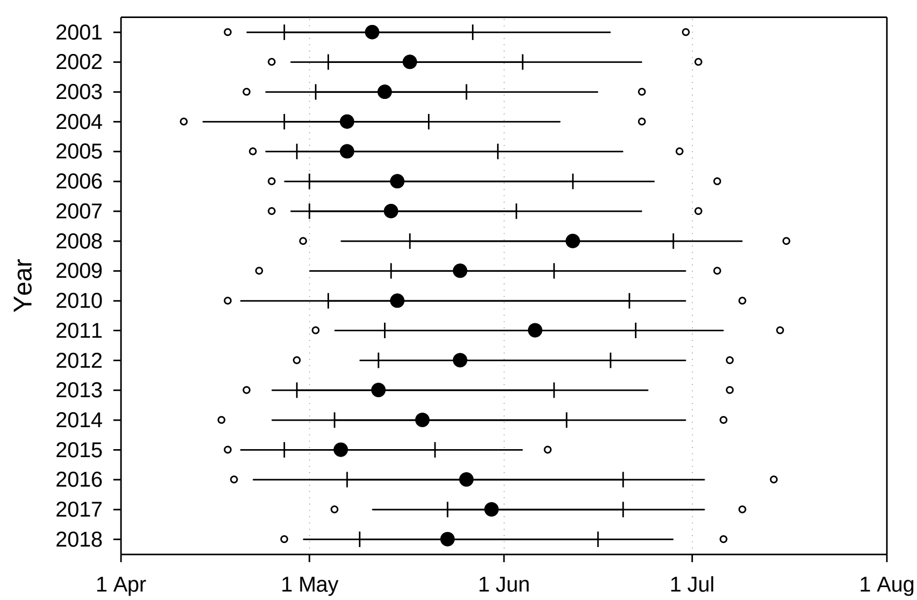


Figure 16. Annual migration timing distributions of adult spring Chinook salmon at Willamette Falls, 2001-2018. Solid circles show median date, vertical lines are the 25th and 75th percentiles, ends of horizontal lines are 10th and 90th percentiles, and open circles are 5th and 95th percentiles. Data source: *www.dfw.state.or.us/fish/fish_counts/willamette%20falls.asp*.

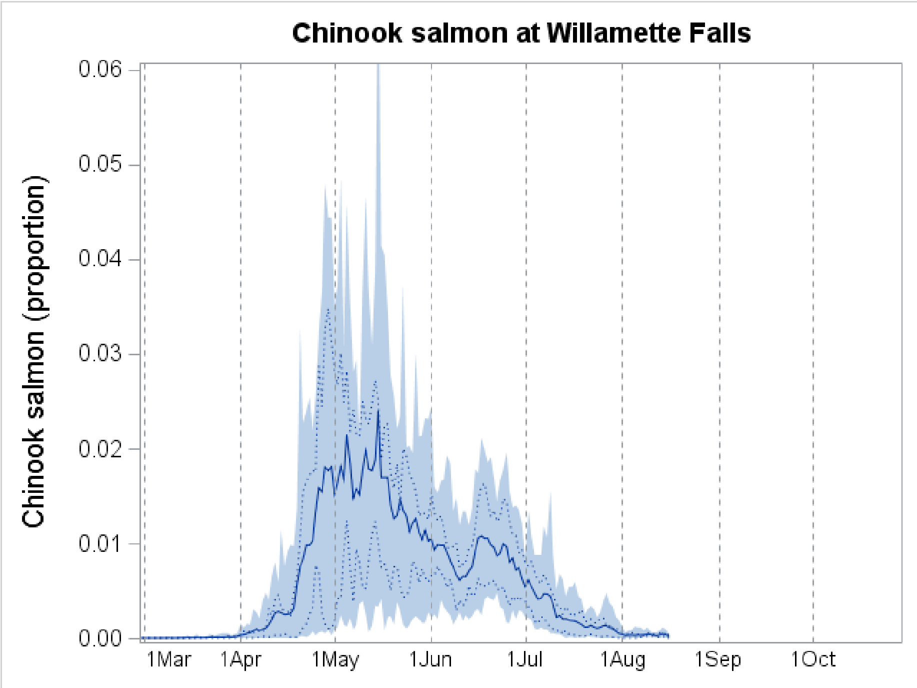


Figure 17. Migration timing of adult spring Chinook salmon at Willamette Falls, shown as the daily proportion of each annual run from 2001-2018. Solid line shows mean annual proportion on each date, dotted lines are the 25th and 75th percentiles, and shaded area covers the 10th and 90th percentiles. Data source: *www.dfw.state.or.us/fish/fish_counts/willamette%20falls.asp*. Annual run timing distributions in Appendix B.

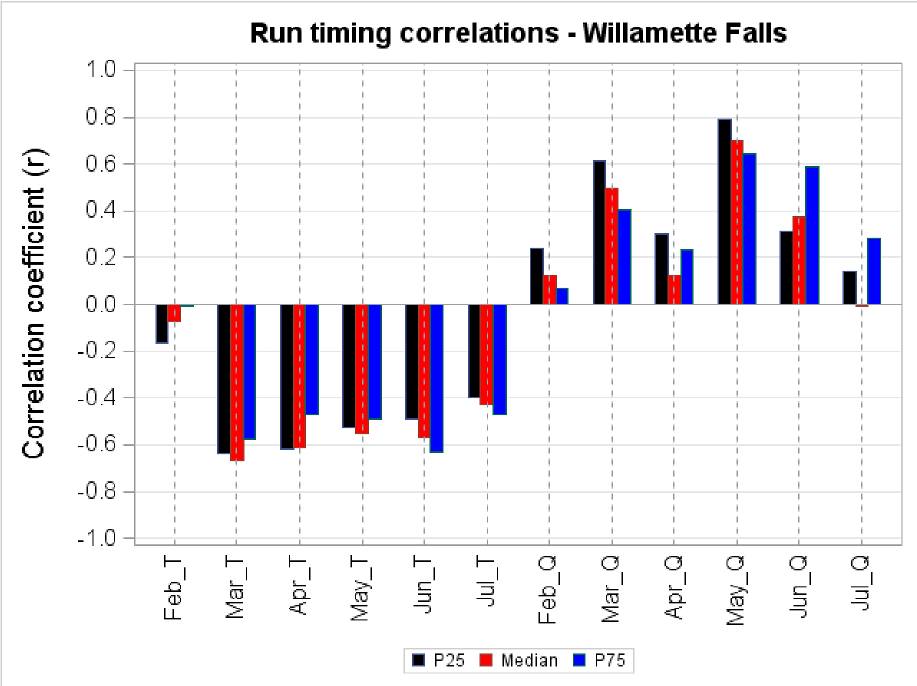


Figure 18. Summary of correlation coefficients (*r*) between annual adult Chinook salmon run timing metrics (25th, 50th, and 75th passage percentiles) and mean monthly water temperature (T) and discharge (Q) values at the Newberg USGS gage on the Willamette River, 2002-2018.

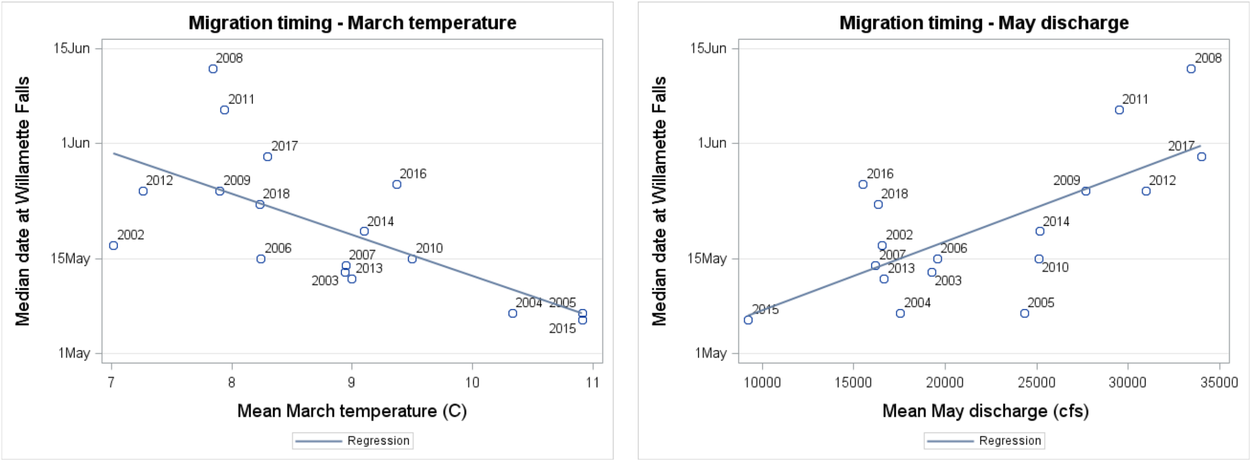


Figure 19. Liner regression relationships between the median run timing date of adult Chinook salmon at Willamette Falls and mean March water temperature (left) and mean May discharge (right), 2002-2018. River environment data from the Newberg USGS gage on the Willamette River.

Bennett dams – Enumerating adult passage at Upper and Lower Bennett dams is challenging and counts have generally been considered an index of upstream passage. For this report we reviewed summaries from video monitoring data collected by ODFW and used those summaries to tabulate daily counts. Based on our data quality assessment for a run timing evaluation, we elected to present data from five (Upper Bennett) and six (Lower Bennett) years (Figure 20).

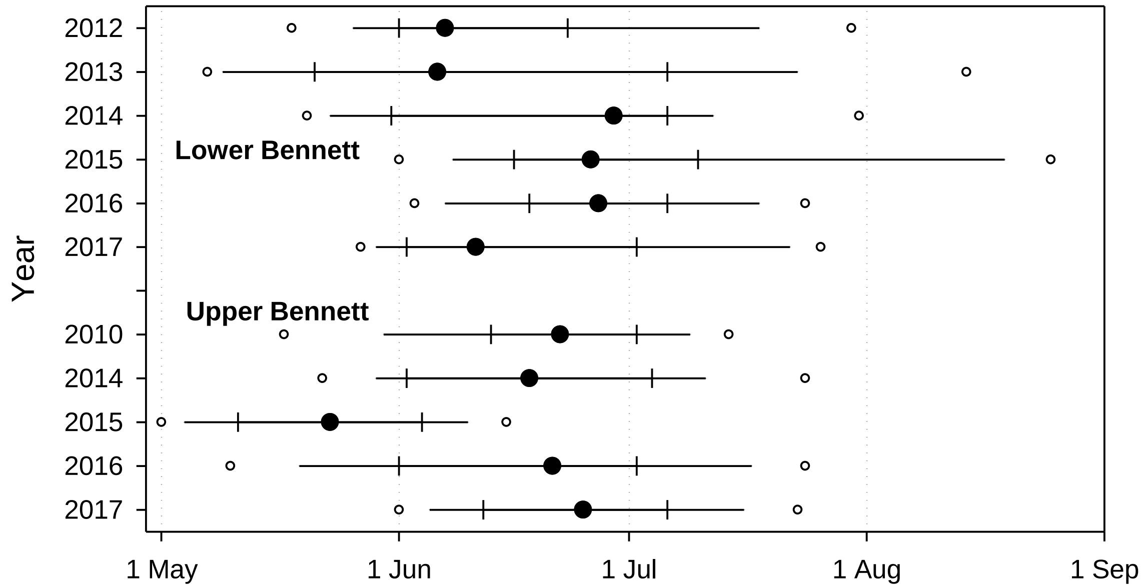


Figure 20. Annual migration timing distributions of adult spring Chinook salmon at Upper and Lower Bennett dams, 2010–2017. Solid circles show median date, vertical lines are the 25th and 75th percentiles, ends of horizontal lines are 10th and 90th percentiles, and open circles are 5th and 95th percentiles. Data source: video summaries from B. DeBow, ODFW.

There were four years (2014–2017) with daily data at both Bennett dams. Unexpectedly, adult Chinook salmon timing distributions in these years were not strongly synchronized at the two count sites. Median salmon passage dates at Upper Bennett Dam were 11 d earlier (2014), 34 d earlier (2015), 6 d earlier (2016), and 14 d later than the medians at Lower Bennett Dam (Figure 20). Across years, salmon passed Upper Bennett Dam starting in late April, with peak passage from early June to mid-July, and some passage through late August and into September (Figure 21).

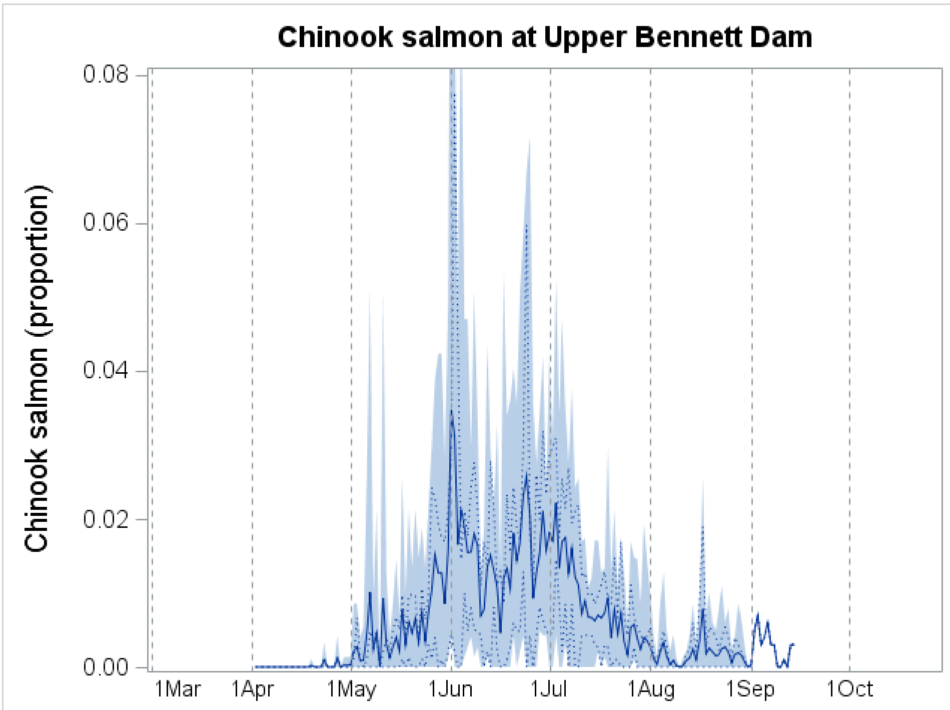


Figure 21. Migration timing of adult spring Chinook salmon at Upper Bennett Dams, shown as the daily proportion of each annual run from 2010 and 2014-2017. Solid line shows mean annual proportion on each date, dotted lines are the 25th and 75th percentiles, and shaded area covers the 10th and 90th percentiles. Data source: video summaries from B. DeBow, ODFW.

Median and quartile run timing dates at Lower Bennett Dam (n = 6 years) were positively correlated with monthly mean water temperatures at Niagara (i.e., earlier arrival in cooler years) and negatively correlated with mean river discharge (i.e., earlier arrival in high flow years). These patterns were the opposite of what was observed in the much longer time series at Willamette Falls.

Based on median run timing dates, we estimate that adult Chinook salmon passage times ranged from 12–51 d (*mean* = 29 d) from Willamette Falls to Lower Bennett Dam and from 17–38 d (*mean* =25 d) from the falls to Upper Bennett Dam. For comparison, in the radiotelemetry studies, passage times for individual adults from Willamette Falls to first detection at the Bennett dams ranged from 8–94 d (*mean* = 21 d). The telemetry-based estimates may be slightly faster than the median passage date estimate because time tagged salmon spent moving from the Bennett tailrace to the count stations was not included.

Minto Fish Collection Facility – Adult Chinook salmon and steelhead collection data at Minto was available for many years, but principally as weekly or monthly counts. Daily or near-daily data were available for five recent years (2013–2018, Figure 22). The earliest adult salmon were collected in late May or early June in most years and about a quarter of the annual runs were collected by mid-July. Median date ranged from 8–28 July. There was some evidence for pulsed trap entry, with the average trap entry metric displaying a bimodal distribution (Figure 23).

As at the Bennett dams, there was some evidence that Chinook salmon were collected at Minto somewhat earlier in years with higher discharge in June-August; correlations with mean monthly river temperatures were mixed. Regardless, we caution against over interpreting given the small number of years with daily or near-daily collection data (*n* = 6 years).

Fin clip status was assessed for all adults collected at Minto. In five of the six years, timing was very similar for fin-clipped and unclipped adults that entered the facility (Figure 24). The exception was in 2014, when cumulative entry by fin-clipped salmon lagged that of unclipped fish for most of the season.

Based on median run timing dates, we estimate that adult Chinook salmon passage times ranged from 21–36 d (*mean* = 29 d) from Lower Bennett Dam to collection at Minto and from 22–60 d (*mean* = 37 d) from Upper Bennett to Minto.

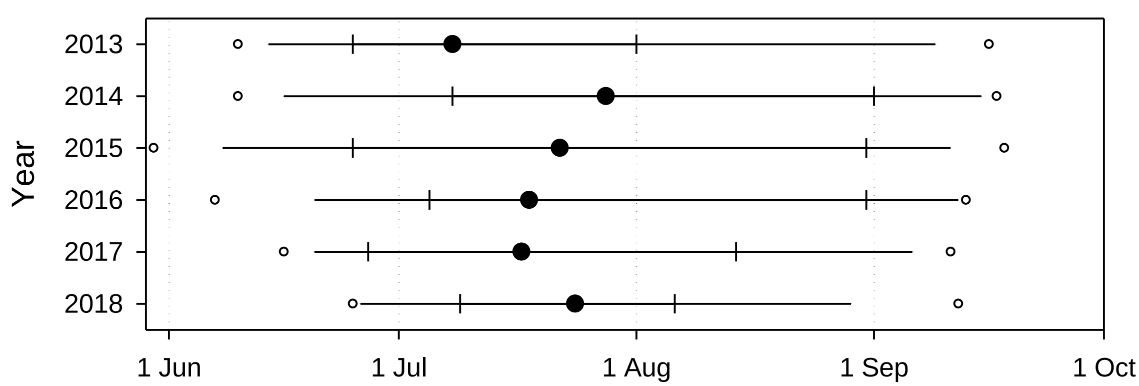


Figure 22. Annual timing distributions of adult spring Chinook salmon collection the Minto Fish Collection Facility, 2013–2018. Solid circles show median date, vertical lines are the 25th and 75th percentiles, ends of horizontal lines are 10th and 90th percentiles, and open circles are 5th and 95th percentiles. Data source: video summaries from G. Grenbemer, ODFW.

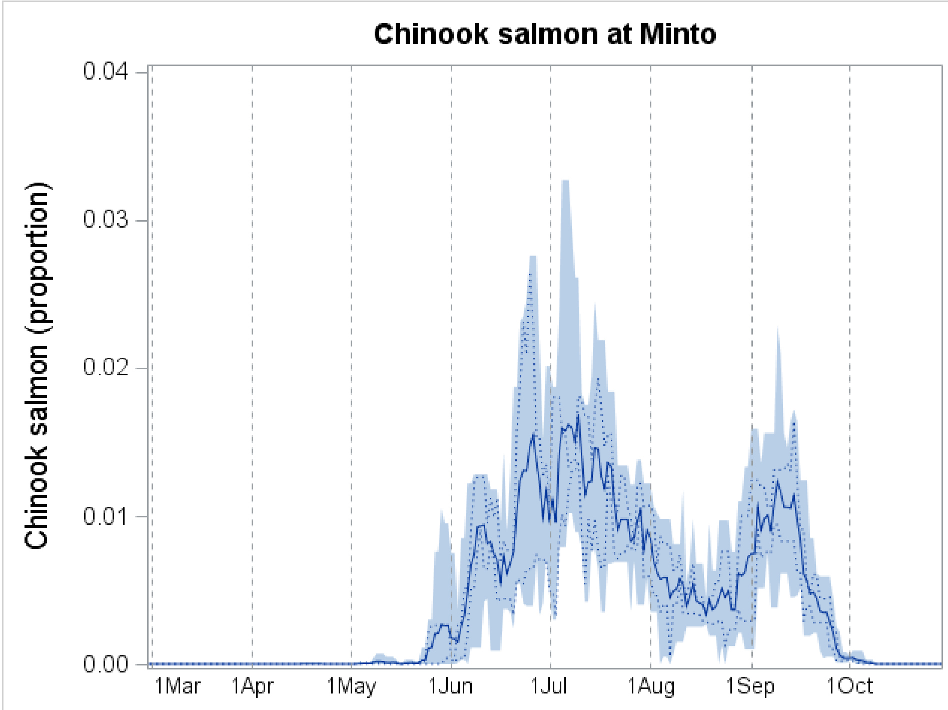


Figure 23. Collection timing of adult spring Chinook salmon at the Minto Fish Collection Facility, shown as the daily proportion of each annual run from 2013–2017. Solid line shows mean annual proportion on each date, dotted lines are the 25th and 75th percentiles, and shaded area covers the 10th and 90th percentiles; data were smoothed using a 7-d running average due to small number of years and irregular counting dates. Data source: G. Grenbemer, ODFW.

Uncertainties and recommendations – The principal uncertainties regarding Chinook salmon run timing at Willamette Falls were associated with the daily counts, which have several potential unquantified biases. These included: (1) species misidentification or enumeration errors, particularly during periods of high adult passage or poor viewing conditions (e.g., due to turbidity); (2) unknown enumeration errors related to double counting of individuals that passed the fishways, then fell back downstream and ascended one or more times (e.g., Boggs et al. 2004); and (3) potential inclusion of non-native fall Chinook salmon in the spring Chinook salmon counts, an enumeration error that would tend to skew run timing metrics somewhat later.

Daily passage data at the Bennett dams and daily adult collection data at Minto were available for fewer years than expected, despite considerable effort by ODFW personnel to assemble the datasets. The adult salmon run timing distributions at Upper and Lower Bennett dams also were not well aligned within year, and this further reduced our confidence in the data at this mid-basin enumeration site. Run timing analyses based on the short available time series were consequently less robust than desired for making management recommendations. It is possible that additional archived data could be recovered that could increase understanding of adult passage timing in the North Santiam River. Lastly, correlations between salmon run timing in the North Santiam and river environment metrics differed unexpectedly from our similar analyses at Willamette Falls and in previous studies of spring Chinook salmon at Bonneville Dam (e.g., Keefer et al. 2004, 2008). It is uncertain whether the run timing relationships with river environment at the Bennett dams was an artifact of small sample size or due to the highly-regulated environment downstream from Big Cliff Dam (i.e., the low among-year variation in temperatures during the migration period).

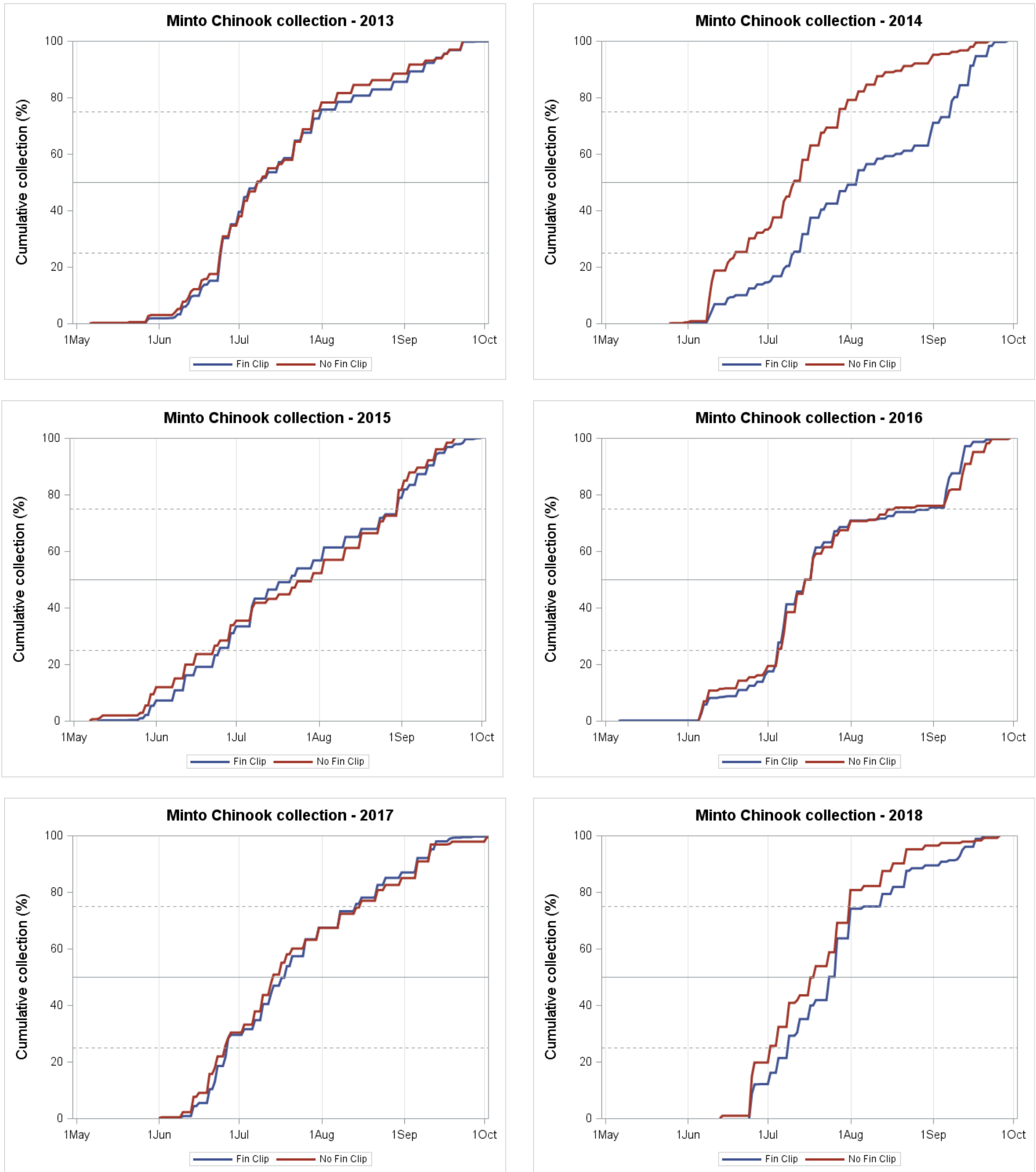


Figure 24. Cumulative distributions of the collection of fin-clipped and unclipped adult Chinook salmon at the Minto Fish Collection Facility, 2013–2018. Data source: G. Grenbemer, ODFW.

3. Redd construction and spawn timing of adult Chinook salmon

Redd counts – Redd surveys were conducted in conjunction with carcass surveys by ODFW in reaches throughout the North Santiam River basin. Main stem reaches from the confluence with the Santiam River to Big Cliff Dam, the section affected by potential temperature management actions, are shown in Figure 25. Over the 2000–2016 time series, approximately 3,200 date×reach surveys were conducted and more than 28,000 redds were counted (including recounts when redds persisted across multiple survey dates). A majority of the effort (~64%) and the tallied redds (~80%) were in the reaches between the Bennett dams and Big Cliff Dam. Surveys upstream from Detroit Dam in the North Santiam and Brietenbush rivers and their tributaries began in 2007. About 23% of the date×reach surveys and 19% of redds counted were upstream from Detroit Dam. The remaining ~13% of date×reach surveys and ~1% of counted redds were in reaches downstream from the Bennett dams.

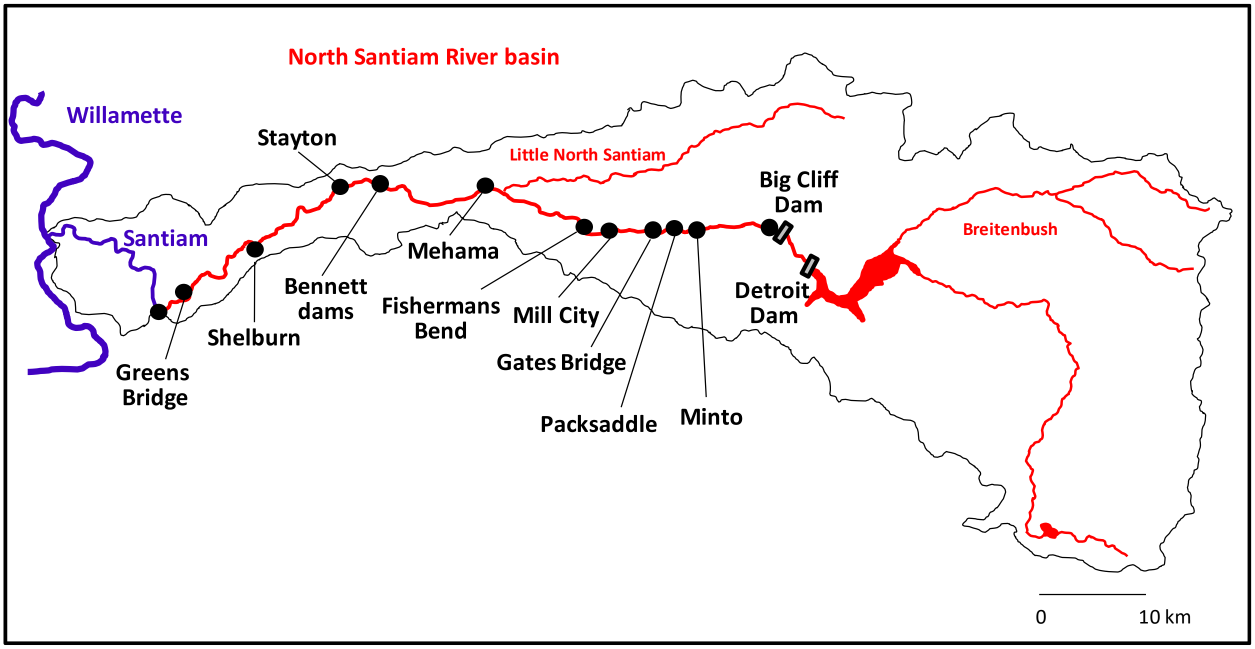


Figure 25. Map of the North Santiam River basin showing approximate start and end points of reaches where adult Chinook salmon carcasses and redds were routinely surveyed by ODFW from the mouth to Big Cliff Dam. Additional reaches were surveyed upstream from Detroit Dam and in the Little North Santiam River – see text for details regarding these reaches.

Redd construction and redd presence are indicators of spawn timing in adult salmonids (e.g., Hayes et al. 2014), but because redd longevity can persist for days to weeks beyond spawning cessation (e.g., Jones 2012), redd counts can be an imprecise spawn timing metric. Compared to the adult Chinook salmon migration timing distributions at downstream sites, redd counts were relatively compressed in time, with a large majority of counts between mid-September and mid-October (Figure 26). On average across years, distributions were fairly similar in the combined survey reaches between the Bennett dams and Big Cliff Dam and in the reaches upstream from Detroit Dam. The annual date of maximum redd abundance in the Bennett–Big Cliff reaches ranged from 14 September to 11 October (*mean* = 28 September, Table 3). Maximum abundance dates in the below-Bennett and above-Detroit river sections spanned similar ranges across years.

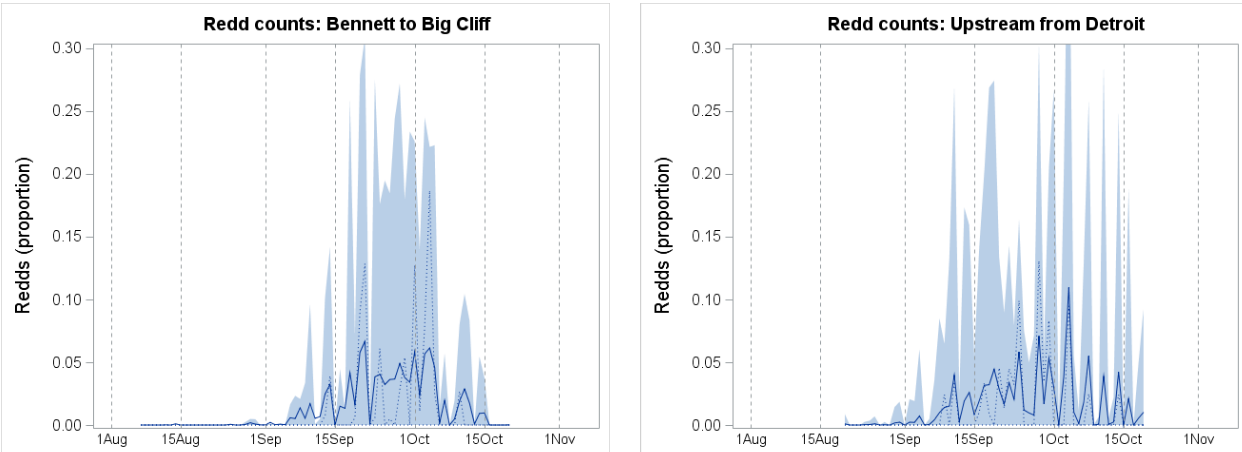


Figure 26. Distribution of Chinook salmon redd abundance through time in the reaches between the Bennett Dams and Big Cliff Dam (left) and upstream from Detroit Dam (right), 2000–2016. Solid line shows mean annual proportion on each date, dotted lines are the 25th and 75th percentiles, and shaded area covers the 10th and 90th percentiles. Data source: ODFW_OWCS.

Table 3. Maximum redd counts and dates when maximum counts were observed for three sections of the North Santiam River basin. Surveys across multiple reaches were summed by date. Data source: ODFW_OWCS.

	Below Bennett dams		Bennett dams to Big Cliff[1]		Above Detroit Dam	
Year	Max count	Max date	Max count	Max date	Max count	Max date
2000	-	-	289	14 Sep	-	-
2001	22	11 Oct	125	4 Oct	-	-
2002	14	9 Oct	151	26 Sep	-	-
2003	8	2 Oct	585	18 Sep	-	-
2004	26	15 Oct	157	1 Oct	-	-
2005	14	3 Oct	265	3 Oct	-	-
2006	13	2 Oct	128	5 Oct	-	-
2007	15	11 Oct	405	11 Oct	54	18 Sep
2008	1	-	253	-	-	-
2009	24	28 Sep	328	21 Sep	62	1 Oct
2010	41	7 Oct	455	7 Oct	289	11 Oct
2011	17	29 Sep	1,182	27 Sep	26	4 Oct
2012	5	12 Oct	734	4 Oct	92	4 Oct
2013	1	-	370	18 Sep	243	3 Oct
2014	6	16 Sep	253	23 Sep	119	30 Sep
2015	-	-	151	21 Sep	166	8 Oct
2016	4	20 Sep	262	20 Sep	141	19 Sep

[1] Includes the Little North Santiam River

In the correlation analysis, we found little evidence for strong environmental effects on the date of maximum redd abundance (Figure 27). Peak dates were somewhat earlier in years when the Willamette River was warmer ($-0.39 \leq r \leq -0.26$) and Willamette River flow was higher ($0.13 \leq r \leq 0.31$). The temperature pattern was consistent with the adult migration timing results described previously. Unexpectedly, peak counts were even less correlated ($-0.30 \leq r \leq 0.20$) with mean monthly environmental covariates in the North Santiam River, as measured at Niagara (Figure 27).

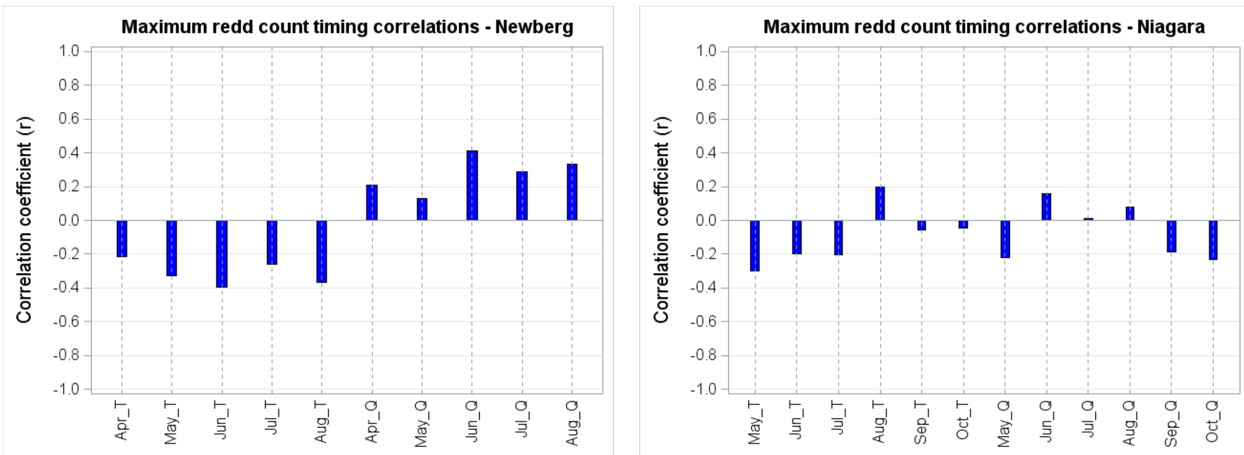


Figure 27. Summary of correlation coefficients (*r*) between annual date of the maximum number of Chinook salmon redds in the reaches between the Bennett Dams and Big Cliff Dam and mean monthly water temperature (T) and discharge (Q) values at the Newberg (Willamette River) and Niagara (North Santiam River) gages, 2000–2016. Note different months between sites.

Female carcasses **–** The ODFW_OWCS carcass dataset included 1,040 females that were considered successful spawners and another ~1,200 that were unsuccessful based on egg retention. A substantial majority of female carcasses were collected downstream from Big Cliff Dam. As with the redd counts, successfully-spawned females were mostly collected in a narrow period from mid-September to mid-October (Figures 28 and 29). In the Bennett–Big Cliff surveys, annual median carcass collection dates were even more narrowly concentrated from 23–30 September; the only exception was in 2002, when the distribution was skewed far earlier (*median* = 14 August, n = 98). It was not clear whether the 2002 data were reliable or whether surveys were limited by logistics or river conditions later in the year. We elected to exclude 2002 from correlation analyses.

Unsuccessful female carcasses were collected over a much broader and earlier series of dates than successful females in most years (Figure 28).

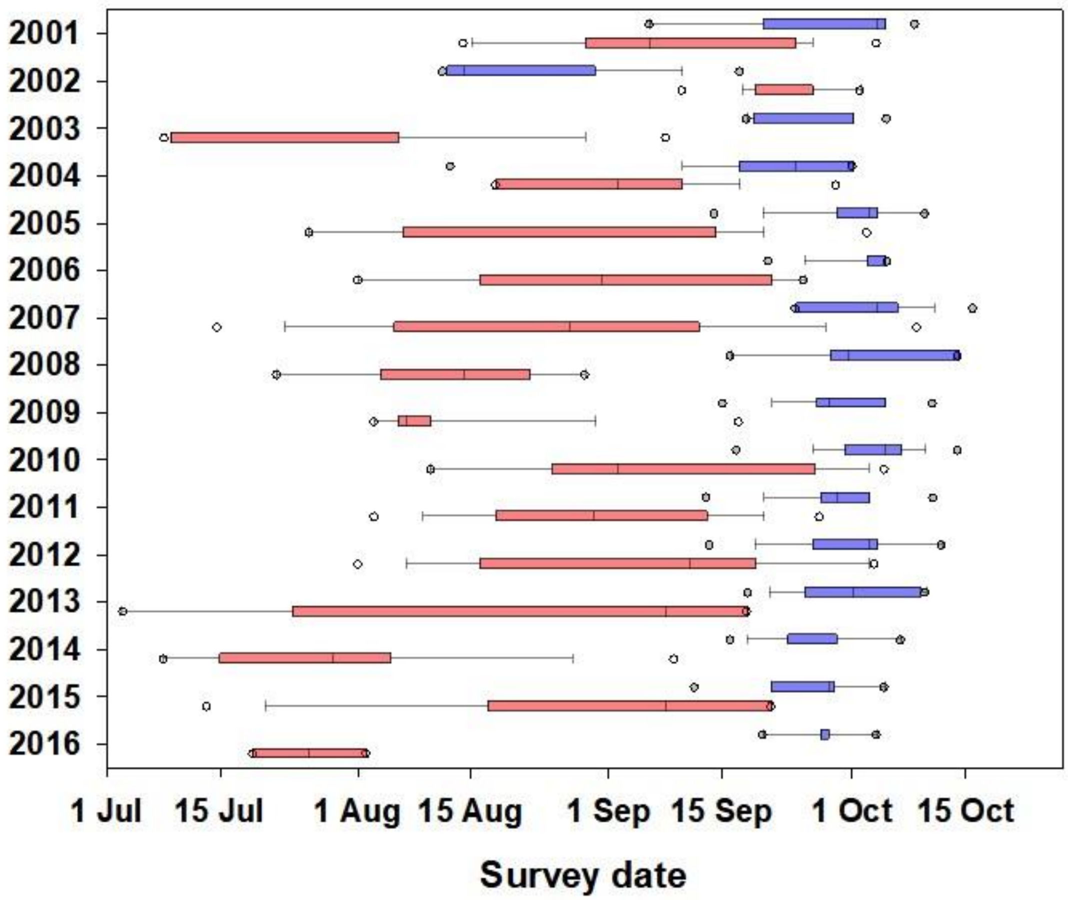


Figure 28. Annual distributions of female Chinook salmon carcass abundance through time in the reaches between the Bennett Dams and Big Cliff Dam, 2000–2016. Blue box plots are for successful spawners and red box plots are for unsuccessful spawners (i.e., prespawn mortalities), based on egg retention. Boxes show median and quartile dates, whiskers show 10^{th} and 90^{th} percentiles, and open circles show 5^{th} and 95^{th} percentiles Data source: ODFW_OWCS.

The correlations between successful female timing and river environment were consistent with the run timing results (and in contrast to the peak redd count results). Warmer Willamette River and North Santiam River temperatures and lower discharge were associated with earlier median and quartile carcass collection dates in the Bennett–Big Cliff reach (Figure 30). The highest correlation coefficients for temperature were for June-August values at Newberg ($-0.71 \leq r \leq -0.60$) and the highest coefficients for discharge were for June–August values at Niagara ($0.57 \leq r \leq 0.68$). In most comparisons, the relationships were stronger for the quartile dates than for the median dates, reflecting the very low inter-annual variation in medians.

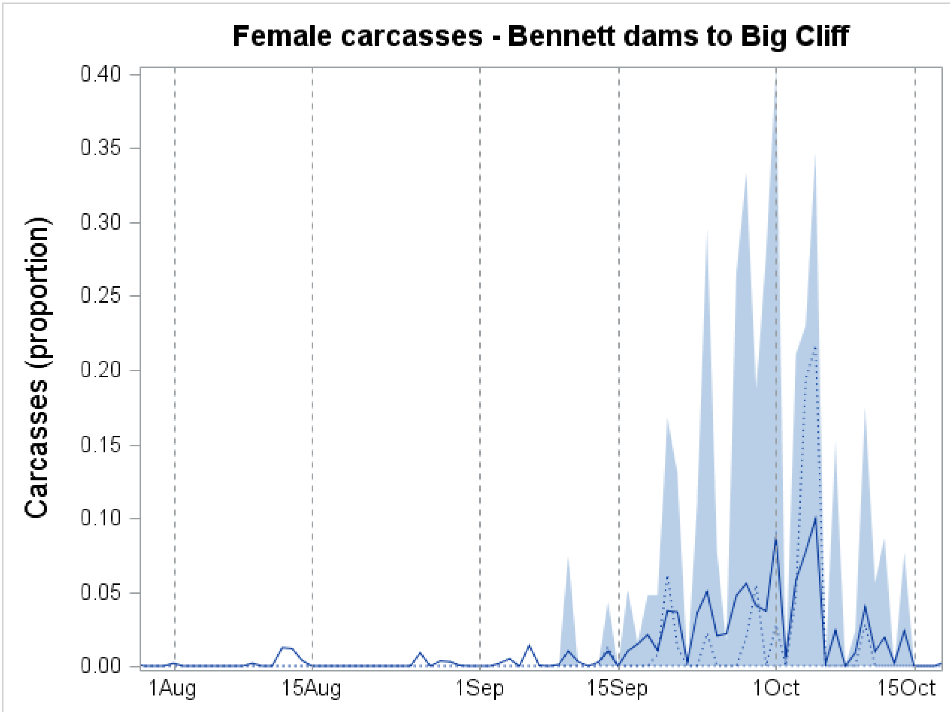


Figure 29. Distribution of female Chinook salmon carcass abundance through time in the reaches between the Bennett Dams and Big Cliff Dam, 2000–2016. Includes successful spawners only. Solid line shows mean annual proportion on each date, dotted lines are the 25th and 75th percentiles, and shaded area covers the 10th and 90th percentiles. Data source: ODFW_OWCS.

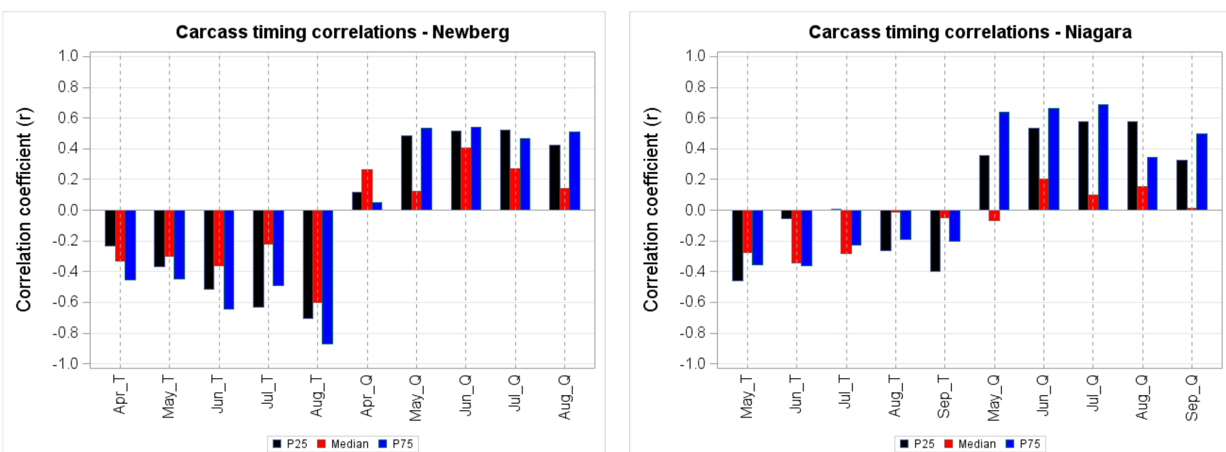


Figure 30. Summary of correlation coefficients (*r*) between collection dates for successful female adult Chinook salmon carcasses collected in the reaches between the Bennett Dams and Big Cliff Dam, 2000–2016. Collection timing metrics (25th, 50th, and 75th percentiles) were correlated mean monthly water temperature (T) and discharge (Q) values collected at the Newberg gage on the Willamette River (left) and the Niagara gage on the North Santiam River (right). (Note: 2002 excluded as apparent outlier.)

Uncertainties and recommendations – The low correlations between maximum redd date and either river temperature or discharge variables suggest that other unmeasured environmental cues influenced redd-building behaviors or that spawn timing may be controlled more by heritable traits than by proximate environmental conditions (e.g., Quinn et al. 2000). In Columbia River fall Chinook salmon population, for example, photoperiod has been a strong predictor of redd construction timing, with a small portion of the inter-annual variation explained by temperature and discharge (Hayes et al. 2014). The photoperiod association is consistent with a heritable trait hypothesis and could explain the narrow range of peak redd construction dates over multiple years.

The carcass dataset was a rich source of information, but it had several potential unquantified biases. These include: (1) biases in survey effort distribution among river reaches; (2) biases associated with survey frequency within reach (e.g., Bowerman et al. 2016); and (3) biases associated with carcass detection and recovery probabilities, including size-dependent effects and effects of survey conditions (e.g., river depth, discharge, or turbidity or weather). Survey effort and recovery biases can substantively affect carcass collection rates and subsequent assessments of biological metrics like spawn timing, sex ratios, PHOS, or prespawn mortality (e.g., Zhou 2002; Murdoch et al. 2010; Bowerman et al. 2016; DeWeber et al. 2017). We recommend that carcass survey protocols for the North Santiam River be standardized and documented insomuch as possible.

4. Prespawn mortality (PSM) of adult Chinook salmon

Female carcass data from the ODFW_OWCS database were used to estimate mortality. The total numbers of carcasses and carcasses per reach per year were highly variable (Table 4, Appendix C); consequently, some data were grouped together to increase sample size and improve confidence in PSM estimates. Results summaries focus on the reaches between Upper Bennet Dam and Big Cliff Dam, as this river section had the most redds and spawners and because it has the highest potential impact from temperature management actions at Detroit Dam. Relatively small numbers of carcasses were collected in reaches upstream from Detroit Dam (i.e., outplanted fish) and in the Little North Santiam River; PSM estimates for these areas were aggregated across reaches within year.

Survey area: multi-reach river sections – PSM estimates were very high downstream from the Bennett dams, with weighted mean values >70% in the four reaches with the most carcasses (Table 4); most annual estimates, aggregated across reaches, were > 50% (Figure 31). In the reaches between Upper Bennett and Big Cliff dams, weighted mean PSM estimates were mostly >45% (Table 4). Annual estimates in this section, with all reaches combined, ranged from ~3% in 2016 to 91% in 2001 (*mean* = 42%, Figure 31). Weighted mean estimates in the Little North Santiam River and the reaches upstream from Detroit Dam were 50% and 11%, respectively (Table 4); annual estimates were all >30% in the Little North Santiam River and were <20% above Detroit Dam (Figure 31).

Survey area: Upper Bennett Dam to Big Cliff Dam – In the reaches between Upper Bennett and Big Cliff dams, weighted mean PSM estimates ranged from 4% in the most upstream reach (Minto – Big Cliff, small *n*) to 87% in the most downstream reach just above Upper Bennet Dam

(Table 4). The largest numbers of carcasses were collected in the four reaches between Mehama and Packsaddle; weighted mean PSM estimates in these reaches ranged from 0.45–0.69.

Table 4. Numbers of female Chinook salmon carcasses collected and evaluated for prespawn mortality (PSM), numbers of years, and mean PSM estimates (weighted by numbers of carcasses per year), by North Santiam River section and survey reach, 2000–2016. Data source: ODFW_OWCS. Appendix C has annual and reach-specific estimates.

River section	Reach (up – downstream)	Female carcasses	Years	**Mean$_w$ PSM**
North Santiam mouth – Bennett dams	Upper Bennett – Stayton	121	13	**0.706**
	Stayton – Shelburn	35	8	**0.829**
	Shelburn – Greens Bridge	9	5	**0.778**
	Greens Bridge – Mouth	6	3	**0.833**
Bennett dams – Big Cliff Dam	Big Cliff – Minto Dam	24	3	**0.042**
	Minto Dam – Packsaddle	38	10	**0.632**
	Packsaddle – Gates Bridge	607	16	**0.453**
	Gates Bridge – Mill City	326	16	**0.583**
	Mill City – Fishermans Bend	254	15	**0.689**
	Fishermans Bend – Mehama	287	16	**0.572**
	Mehama – Powerlines	86	13	**0.814**
	Powerlines – Upper Bennett	78	9	**0.872**
Little North Santiam River	All reaches	167	12	**0.500**
Above Detroit Dam	All reaches	193	6	**0.109**

Annual PSM estimates in the aggregated reaches in this section were only weakly correlated with most mean monthly Willamette and North Santiam river environment metrics (Figure 32), including 7DADM. The strongest PSM correlations were with mean September water temperature at Niagara (r = 0.54) and August discharge at Niagara (r = -0.70). With each 1 °C increase in mean September water temperature Chinook salmon PSM increased by ~ 9% (Figure 33; linear regression, r^2 = 0.29, P = 0.031, n =15). In contrast to the August temperature result, PSM was negatively correlated with mean June and July temperatures at Niagara (Figure 32). In the regression for August discharge, each 50 cfs increase was associated with a ~9% reduction in PSM (r^2 = 0.49, P = 0.003, n =15).

Both run timing and PHOS were weakly associated with annual PSM estimates between Bennett and Big Cliff dams (Figure 33), but neither covariate was statistically significant. Each 10% increase in PHOS was associated with a ~4% increase in PSM (r^2 = 0.17, P = 0.140, n =15). PSM was slightly higher when run timing at Willamette Falls was early (r^2 = 0.01, P = 0.724); relatively high PSM in 2008, a high-flow year, influenced the poor regression fit (Figure 33).

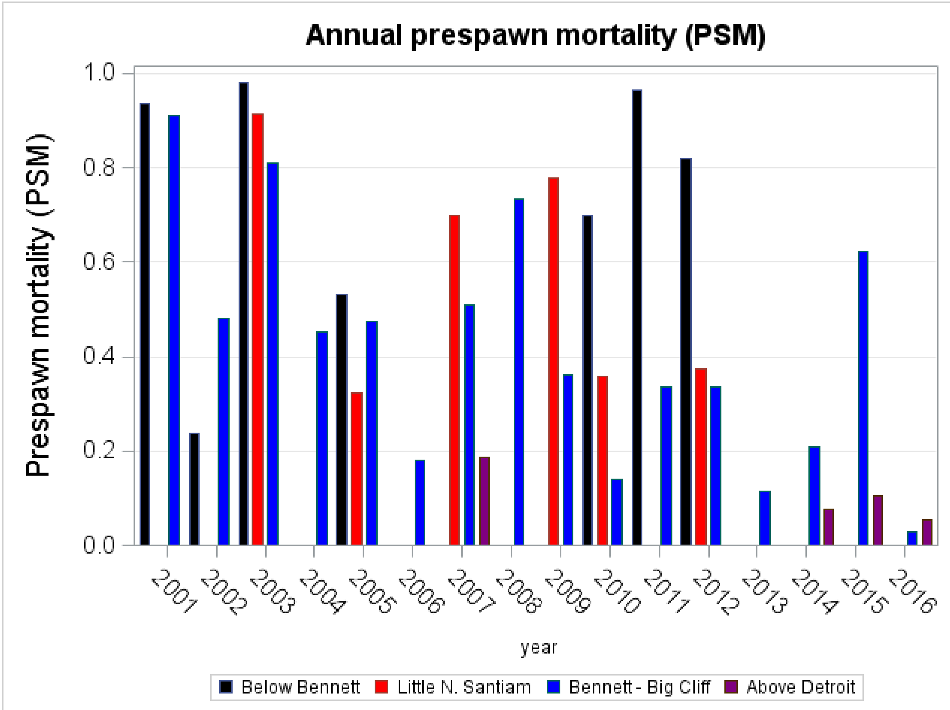
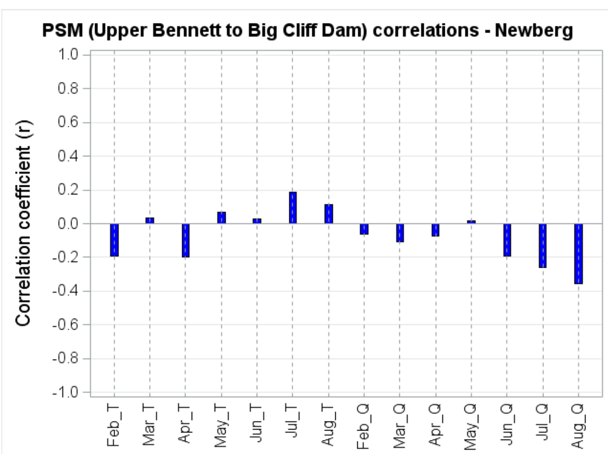


Figure 31. Annual female prespawn mortality estimates, aggregated by river section. Only includes section×year combinations with ≥ 10 carcasses. Data source: ODFW_OWCS.



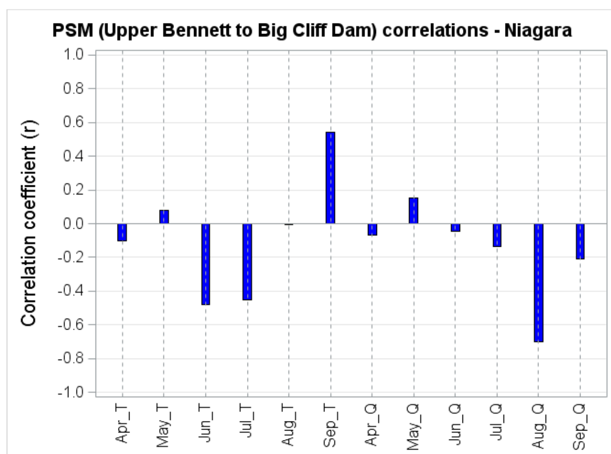


Figure 32. Summary of correlation coefficients (r) between annual prespawn mortality estimates for female Chinook salmon in the eight carcass survey reaches between Upper Bennett Dam and Big Cliff Dam and mean monthly water temperature (T) and discharge (Q) values collected at the Newberg gage on the Willamette River (left) and the Niagara gage on the North Santiam River (right), 2000–2016.

Survey area: Packsaddle to Gates Bridge and Gates Bridge to Mill City – These two survey reaches had the highest numbers of female carcasses across years and had the highest numbers of years with sufficient numbers of females to calculate annual PSM estimates. The reaches are adjacent and might be considered within-year replicates because salmon were presumably exposed to similar within-year conditions. In the correlation and regression analyses below, we evaluated the two reaches independently.

Annual PSM correlations with mean monthly environmental variables and 7DADM were very similar for the two reaches (Figures 34 and 35). The highest temperature correlations in both cases were with mean September water temperature at Niagara, with higher PSM in warmer years. An opposite effect was identified with June and July temperatures – PSM was negatively correlated with mean temperatures in those months, especially in the Gates Bridge – Mill City reach. The highest correlations with discharge were with mean August (Gates Bridge – Mill City) or September (Packsaddle – Gates Bridge) discharge at Niagara.

With each 1 °C increase in mean September water temperature Chinook salmon PSM increased by ~10% in the Packsaddle–Gates Bridge reach (Figure 36; $r^2 = 0.41$, $P = 0.013$, $n = 13$) and increased by ~9% in the Gates Bridge–Mill City reach (Figure 37; $r^2 = 0.39$, $P = 0.054$, $n = 9$). An increase of 50 cfs in August or September was associated with a ~2–3% decrease in PSM; the regression was statistically significant for the Packsaddle–Gates Bridge reach ($r^2 = 0.34$, $P = 0.030$, $n = 13$), but not for the Gates Bridge–Mill City reach ($P = 0.115$).

Annual PSM estimates increased with PHOS in both reaches, by about 5% for every 10% increase in PHOS, though the regressions were not statistically significant ($P > 0.05$; Figures 36 and 37). Similarly, higher PSM was associated with early migration timing in both reaches, with PSM increasing by ~1% for each 1 d shift earlier in the median migration date at Willamette Falls (Figures 36 and 37); the regressions were not statistically significant ($P > 0.05$).

Uncertainties and recommendations – The uncertainties and potential biases related to carcass surveys described at the end of Section 3 directly apply to the carcass-based PSM estimates. Additional PSM uncertainties include: (1) the potential for apparent offsetting effects of seasonal river temperatures on salmon mortality (i.e., warmer early summer temperatures were associated with lower PSM in some cases, whereas warmer fall temperatures were associated with higher PSM); (2) covariance in North Santiam River environment metrics made it challenging to separate their effects on PSM; (3) it is almost certain that environmental conditions in both the Willamette and North Santiam rivers affect PSM, but it is difficult to separate these effects given environmental covariance across sites within year; and (4) the influence of seasonal environmental metrics on PSM may vary across different portions of the population (e.g., on early- versus late-run fish), but PSM metrics have traditionally been based on the aggregate population. Research using individually-marked fish (e.g., Keefer et al. 2010; DeWeber et al. 2017; Naughton et al. 2018) has helped address some of these uncertainties, but principally for salmon collected at WVP adult facilities and then outplanted upstream from dams. Much less is known about the individual histories of salmon in the populations that spawn downstream from collection facilities, or about the relative influence of environmental conditions on individuals during the upstream migration vs. tributary holding periods.

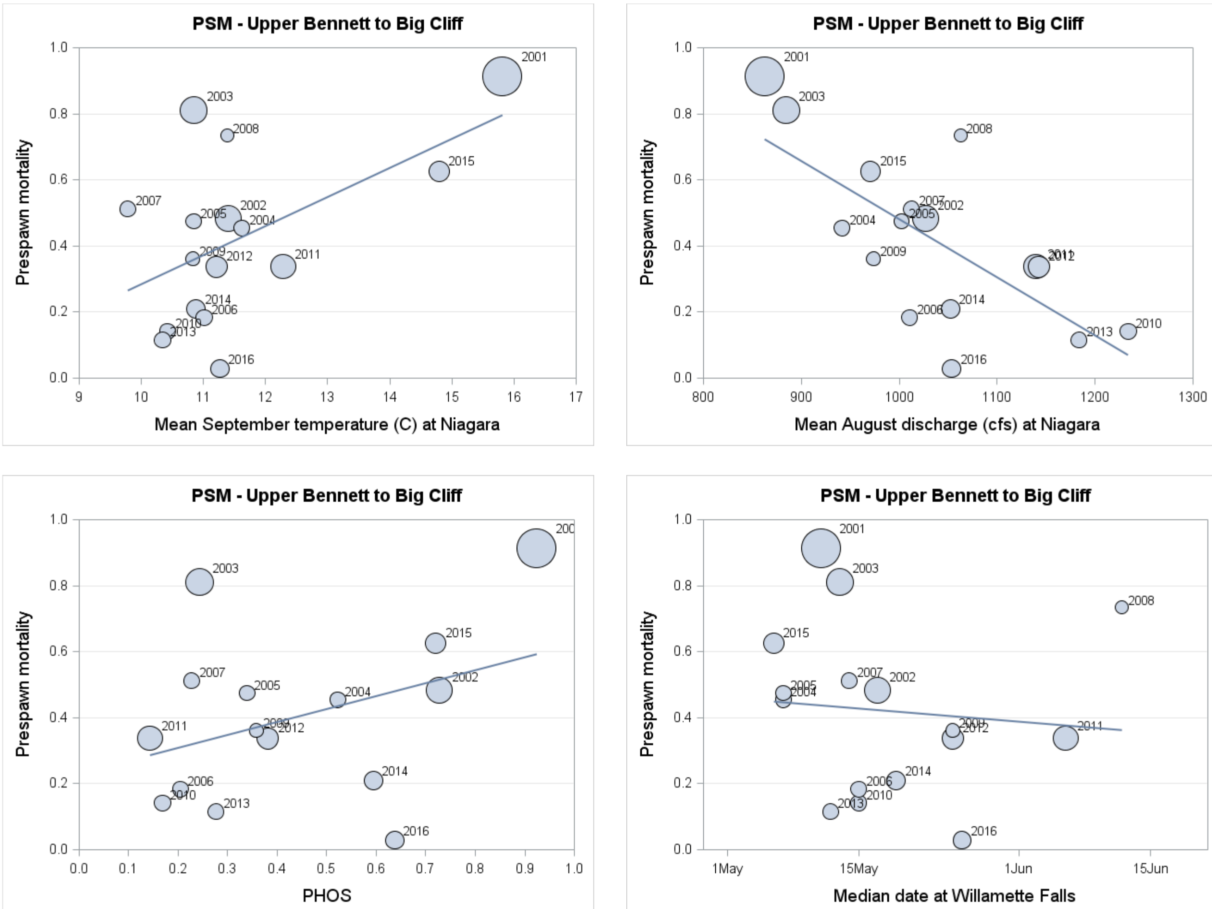


Figure 33. Linear regression relationships between annual prespawn mortality estimates for female Chinook salmon and covariates in the eight carcass survey reaches between Upper Bennett Dam and Big Cliff Dam, 2000–2016. Predictor variables are mean September water temperature at Niagara (top left), mean August discharge at Niagara (top right), proportion hatchery-origin spawners (PHOS, bottom left), and the median run timing date of adult Chinook salmon at Willamette Falls (bottom right). Circles are scaled to the number of carcasses.

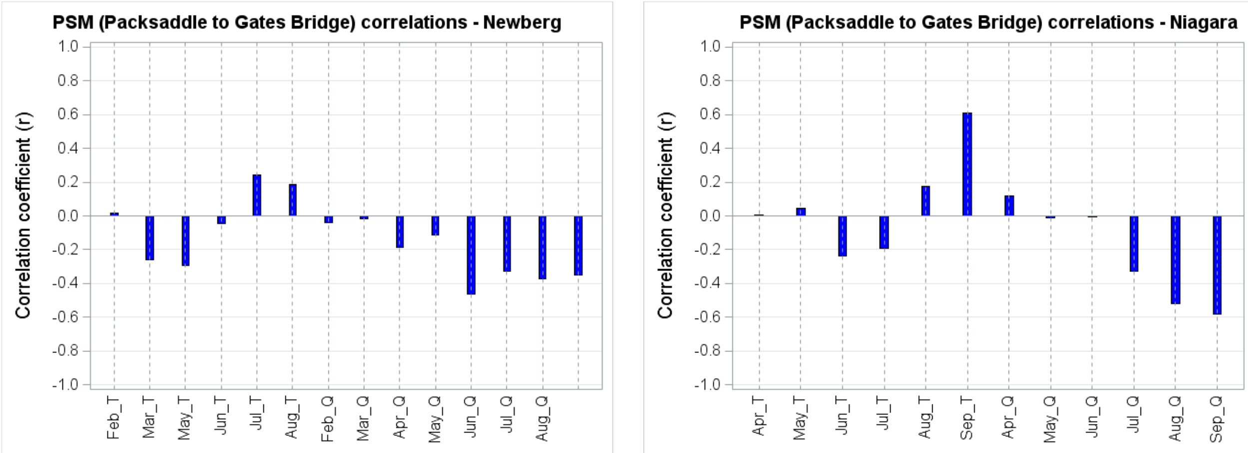


Figure 34. Summary of correlation coefficients (*r*) between annual prespawn mortality estimates for female Chinook salmon in the Packsaddle–Gates Bridge reach and mean monthly water temperature (T) and discharge (Q) values collected at the Newberg gage on the Willamette River (left) and the Niagara gage on the North Santiam River (right), 2000–2016. Only includes years with ≥10 carcasses.

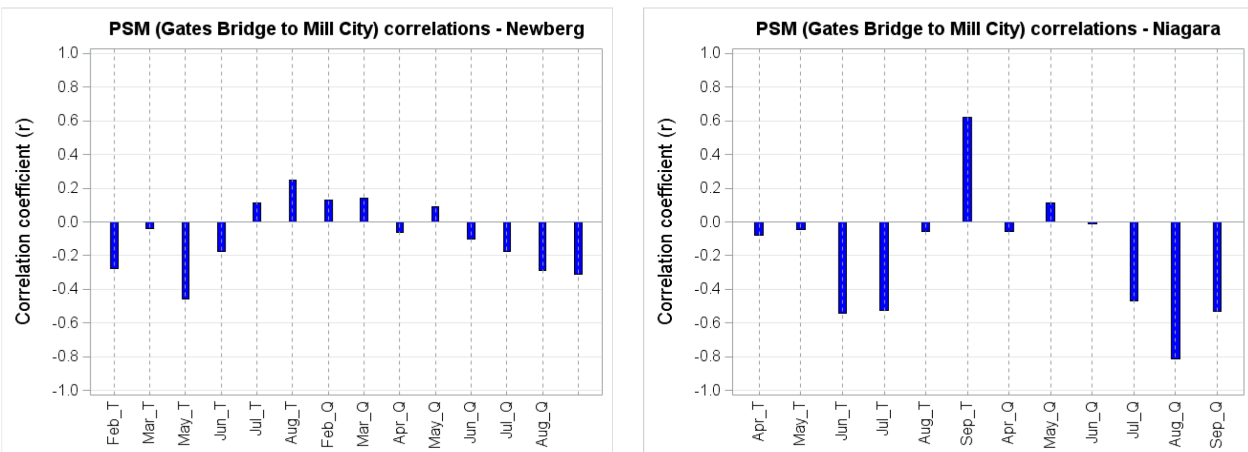


Figure 35. Summary of correlation coefficients (*r*) between annual prespawn mortality estimates for female Chinook salmon in the Gates Bridge–Mill City reach and mean monthly water temperature (T) and discharge (Q) values collected at the Newberg gage on the Willamette River (left) and the Niagara gage on the North Santiam River (right), 2000–2016. Only includes years with ≥10 carcasses.

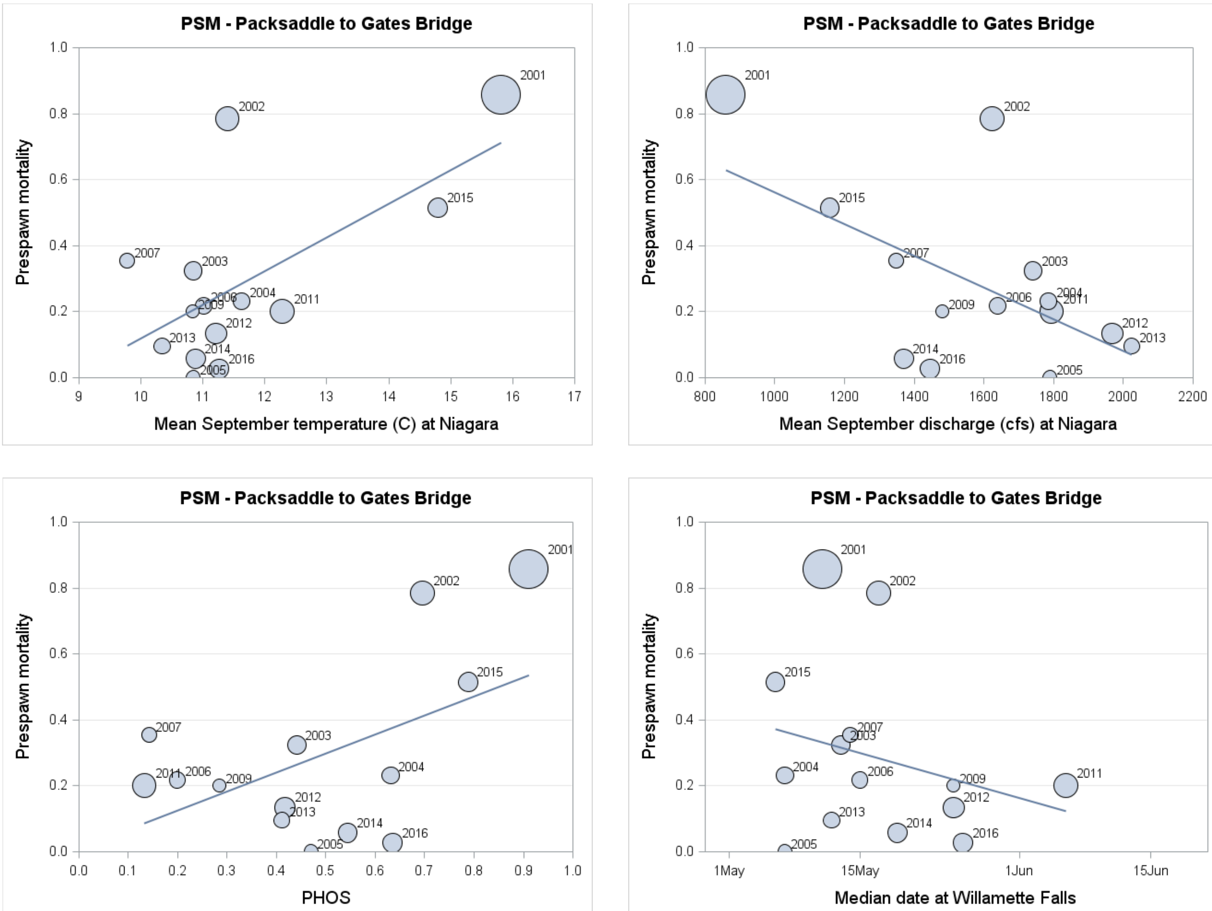


Figure 36. Linear regression relationships between annual prespawn mortality estimates for female Chinook salmon and covariates in in the Packsaddle–Gates Bridge reach, 2000–2016. Predictor variables are mean September water temperature at Niagara (top left), mean September discharge at Niagara (top right), proportion hatchery-origin spawners (PHOS, bottom left), and the median run timing date of adult Chinook salmon at Willamette Falls (bottom right). Circles are scaled to the number of carcasses. Only includes years with ≥10 carcasses.

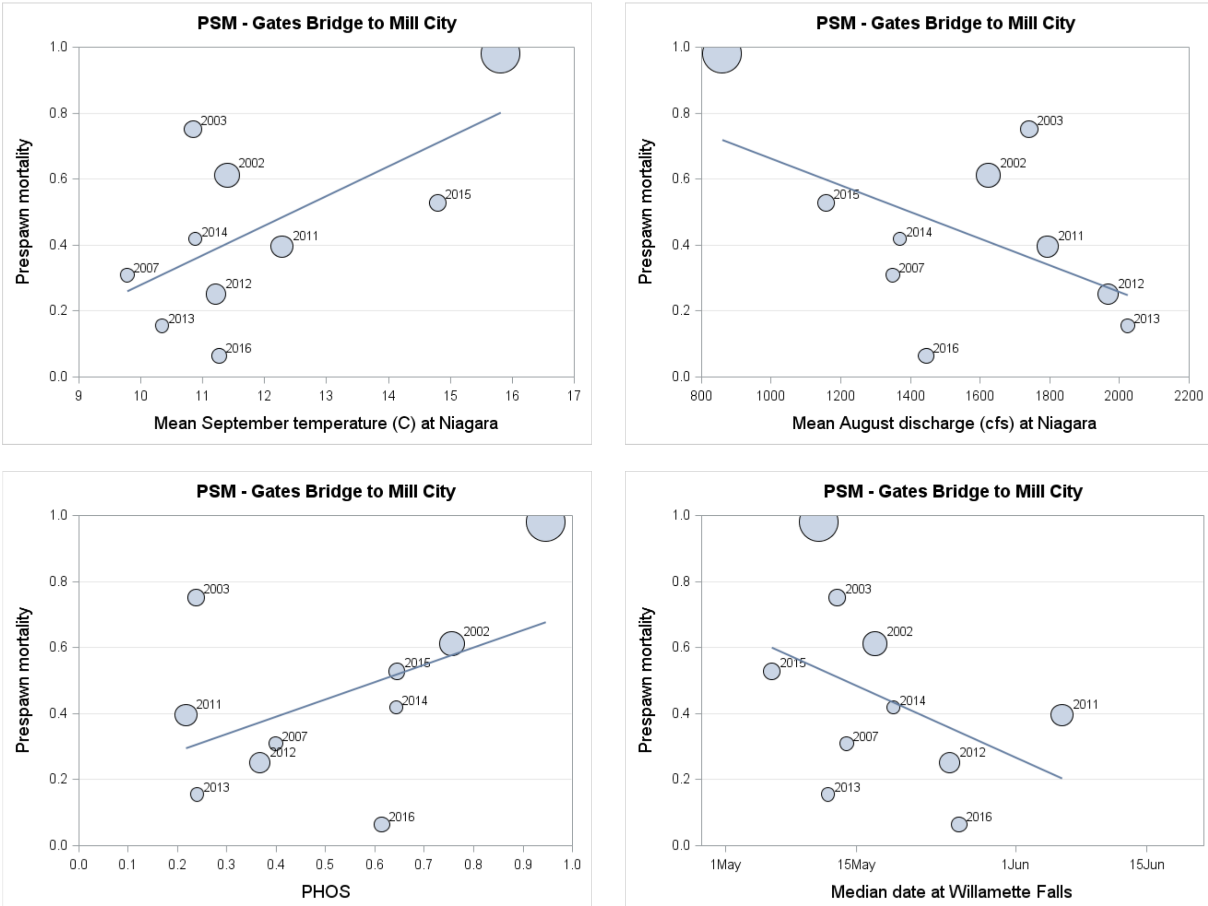


Figure 37. Linear regression relationships between annual prespawn mortality estimates for female Chinook salmon and covariates in in the Gates Bridge–Mill City reach, 2000–2016. Predictor variables are mean September water temperature at Niagara (top left), mean August discharge at Niagara (top right), proportion hatchery-origin spawners (PHOS, bottom left), and the median run timing date of adult Chinook salmon at Willamette Falls (bottom right). Circles are scaled to the number of carcasses. Only includes years with ≥10 carcasses.

5. Proportion of hatchery-origin spawners (PHOS)

Male and female carcass data from the ODFW_OWCS database were used to estimate PHOS, resulting in a substantially larger sample size than for PSM. Although there were ~4,200 carcasses with fin clip assessments, the total numbers carcasses per reach per year were highly variable (Table 5, Appendix D) and data for some reaches were aggregated for analyses. The structure of the summaries below parallels the results for PSM estimates in the previous section.

PHOS is a function of hatchery production and wild adult abundance. Consequently, some of the year-to-year variability in PHOS is related to the relative abundance of the two populations. Information on the numbers of fin-clipped and unclipped adults entering the North Santiam River is limited, with the first potential enumeration location at the Bennett dams. From daily and monthly count data provided by ODFW, we estimated the ratio of fin-clipped to unclipped Chinook salmon at the combined Bennett dams averaged 4.3 (*range* = 2.6–6.2) over the most recent six years (2013–2018). Over these same years, the ratio at the Minto Fish Collection Facility averaged 5.6 (*range* = 3.6–9.1) fin-clipped to unclipped fish. Ratios in the two time series were positively correlated (r = 0.69).

Table 5. Numbers of Chinook salmon carcasses collected and evaluated for presence of hatchery fin clips, numbers of years, and weighted mean proportion of hatchery-origin spawner (PHOS) estimates, by North Santiam River section and survey reach, 2000–2016. Data source: ODFW_OWCS. Appendix D has annual estimates.

River section	Reach (up – downstream)	Carcasses	Years	**Mean_w PHOS**
North Santiam mouth – Bennett dams	Upper Bennett – Stayton	169	14	**0.438**
	Stayton – Shelburn	79	11	**0.468**
	Shelburn – Greens Bridge	16	7	**0.250**
	Greens Bridge – Mouth	8	4	**0.250**
Bennett dams – Big Cliff Dam	Big Cliff – Minto Dam	33	3	**0.000**
	Minto Dam – Packsaddle	72	12	**0.556**
	Packsaddle – Gates Bridge	1,031	17	**0.616**
	Gates Bridge – Mill City	640	17	**0.625**
	Mill City – Fishermans Bend	417	15	**0.643**
	Fishermans Bend – Mehama	531	17	**0.490**
	Mehama – Powerlines	181	14	**0.331**
	Powerlines – Upper Bennett	136	11	**0.382**
Little North Santiam River	All reaches	347	15	**0.052**
Above Detroit Dam	All reaches	544	7	**0.888**

Survey area: multi-reach river sections – Mean PHOS estimates, weighted by the numbers of male and female carcasses assessed in each year, ranged from 25–47% in the reaches downstream from the Bennett dams (Table 5). Estimates in the reaches between Bennett and Big Cliff were slightly higher, on average, with reach-specific weighted means ranging from 33–64% in all reaches except Big Cliff–Minto, where none of 33 carcasses had fin clips (likely the result of release protocols at the Minto facility. PHOS was low in the Little North Santiam River, where the weighted mean was 5%, and high (89%) in the reaches upstream from Detroit Dam, reflecting outplant protocols that selected hatchery-origin fish in recent years.

Survey area: Upper Bennett Dam to Big Cliff Dam – Annual PHOS estimates were positively correlated with river temperatures and negatively correlated with river discharge in both the Willamette and North Santiam rivers (Figure 38). The highest correlations with temperature were with the mean August temperature at Newberg and mean September temperature at Niagara. In the regression model for August temperature at Newberg, a 1°C increase was associated with a ~20% increase in PHOS (r^2 = 0.46, P = 0.011, n = 12). In the model for September temperature at Niagara, a 1°C temperature increase translated to a ~10% increase in PHOS (Figure 39).

In the discharge regressions, a 50 cfs increase in the September mean at Niagara resulted in a ~12% reduction in PHOS (r^2 = 0.39, P =0.013, n = 14). In the Newberg model, a 5,000 cfs increase in June discharge was associated with a ~7% decrease in PHOS (r^2 = 0.30, P =0.042, n = 13; Figure 39). Median run timing date at Willamette Falls was not statistically associated with PHOS in the Upper Bennet–Big Cliff reach (P = 0.35).

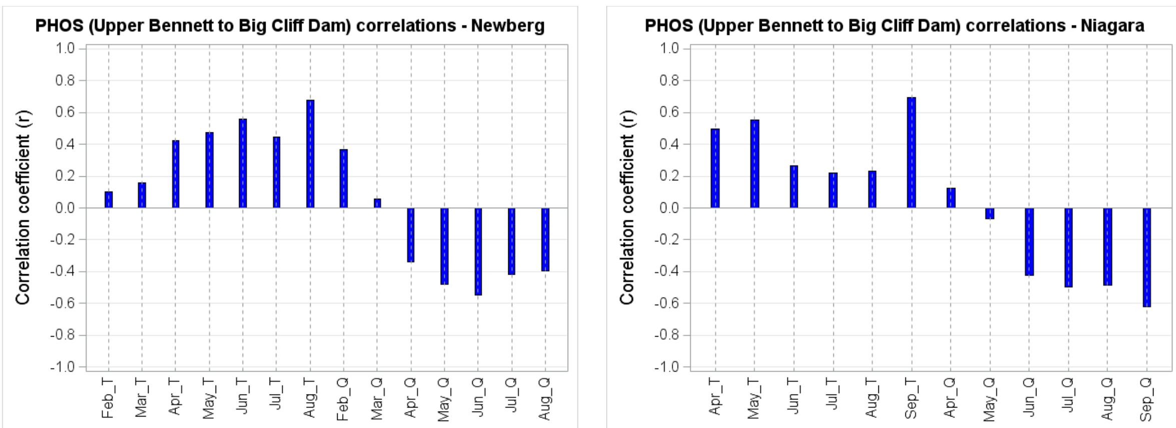


Figure 38. Summary of correlation coefficients (r) between annual Chinook salmon PHOS estimates in the eight carcass survey reaches between Upper Bennett Dam and Big Cliff Dam and mean monthly water temperature (T) and discharge (Q) values collected at the Newberg gage on the Willamette River (left) and the Niagara gage on the North Santiam River (right), 2000–2016.

***Survey area: Packsaddle to Gates Bridge and* Gates Bridge to Mill City** – As described above in the PSM section, we evaluated these two adjacent reaches independently, but they could be considered within-year replicates. Annual PHOS correlations with mean monthly environmental variables and 7DADM were similar for the two reaches, with higher PHOS associated with warmer river temperatures and lower discharge (Figures 39 and 40). The highest temperature correlations were with September means at Niagara and August means at Newberg for both reaches. The highest correlations with discharge were with July discharge at Niagara (Packsaddle – Gates Bridge), September discharge at Niagara (Gates Bridge – Mill City) and June discharge at Newberg (both reaches).

With each 1 °C increase in mean September water temperature at Niagara, Chinook salmon PHOS increased by ~10% in the Packsaddle–Gates Bridge reach (Figure 41; r^2 = 0.46, P = 0.008, n =13) and increased by ~8% in the Gates Bridge–Mill City reach (Figure 42; r^2 = 0.38, P = 0.058, n =9). A 1 °C increase in mean August temperature at Newberg translated to a ~24% increase in PHOS in the Packsadde–Gates Bridge reach (r^2 = 0.51, P = 0.005, n =11) and a ~14% increase in PHOS in the Gates Bridge–Mill City reach (r^2 = 0.21, P = 0.256, n =7).

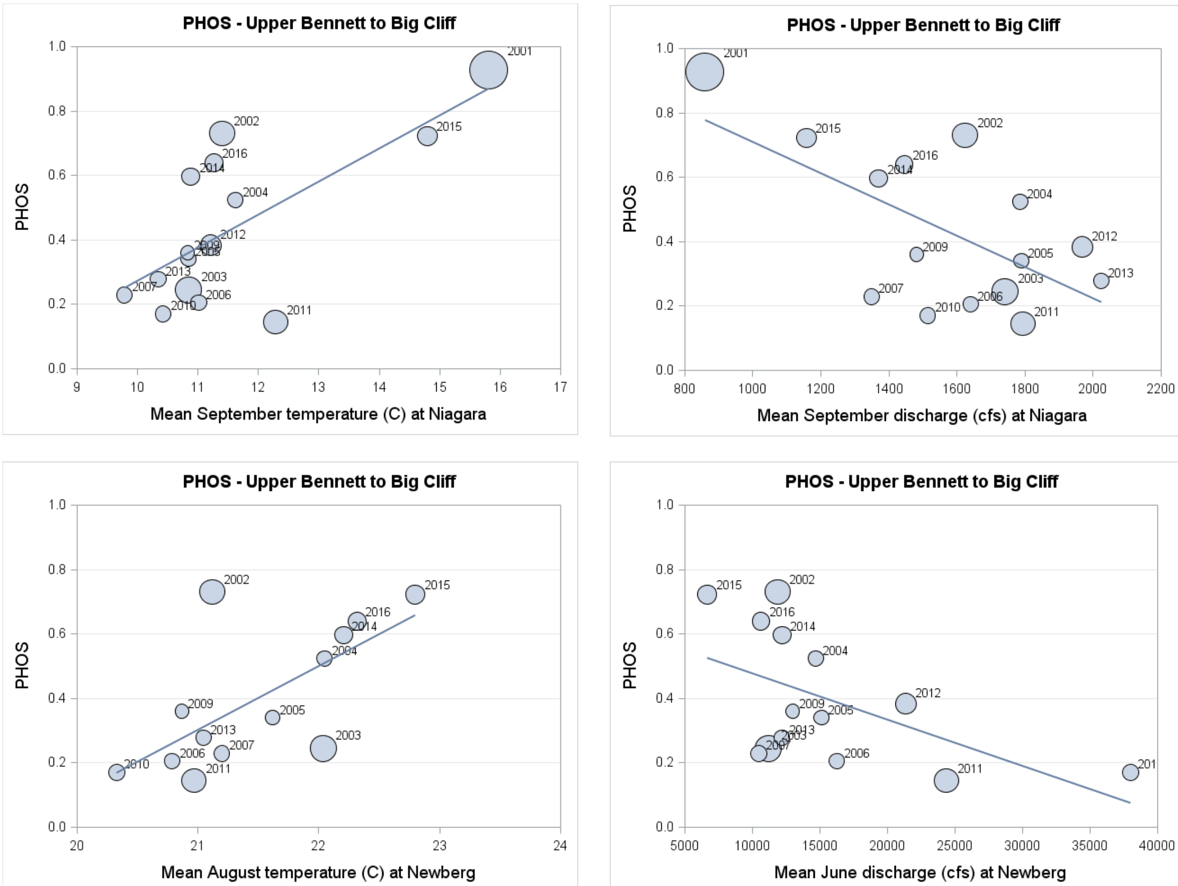


Figure 39. Linear regression relationships between annual Chinook salmon PHOS and covariates in the eight carcass survey reaches between Upper Bennett Dam and Big Cliff Dam, 2000–2016. Predictor variables are mean September water temperature at Niagara (top left), mean September discharge at Niagara (top right), mean August water temperature at Newberg (bottom left), and mean June discharge at Newberg (bottom right). Circles are scaled to the number of carcasses.

An increase of 5,000 cfs in mean June discharge at Newberg was associated with a ~12% decrease in PHOS in the Packsaddle–Gates Bridge reach (r^2 = 0.29, P = 0.059, n = 12) and a ~9% decrease in the Gates Bridge–Mill City reach (r^2 = 0.26, P = 0.161, n = 8) (Figures 41 and 42). In the regressions with Niagara discharge, PHOS decreased by ~4–5% in the Packsaddle–Gates Bridge reach with every 50 cfs increase in the July mean (r^2 = 0.29, P = 0.049, n = 13) and decreased by ~3% with every 50 cfs increase in the September mean (r^2 = 0.64, P = 0.006, n = 9). Although PHOS was higher in years with earlier migration timing at Willamette Falls, the regression were not statistically significant in either reach (P > 0.05).

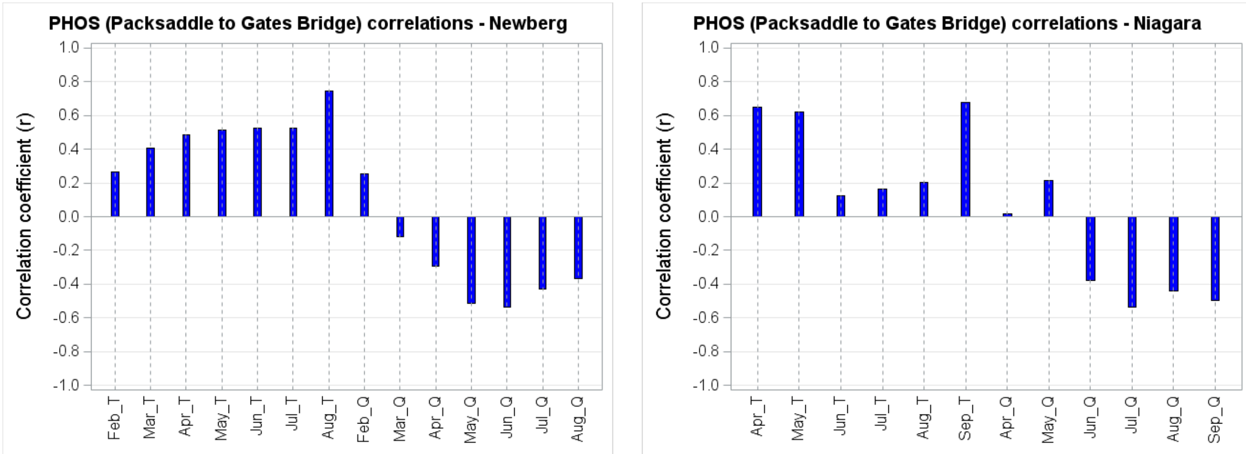


Figure 39. Summary of correlation coefficients (*r*) between annual Chinook salmon PHOS estimates in the Packsaddle–Gates Bridge reach and mean monthly water temperature (T) and discharge (Q) values collected at the Newberg gage on the Willamette River (left) and the Niagara gage on the North Santiam River (right), 2000–2016.

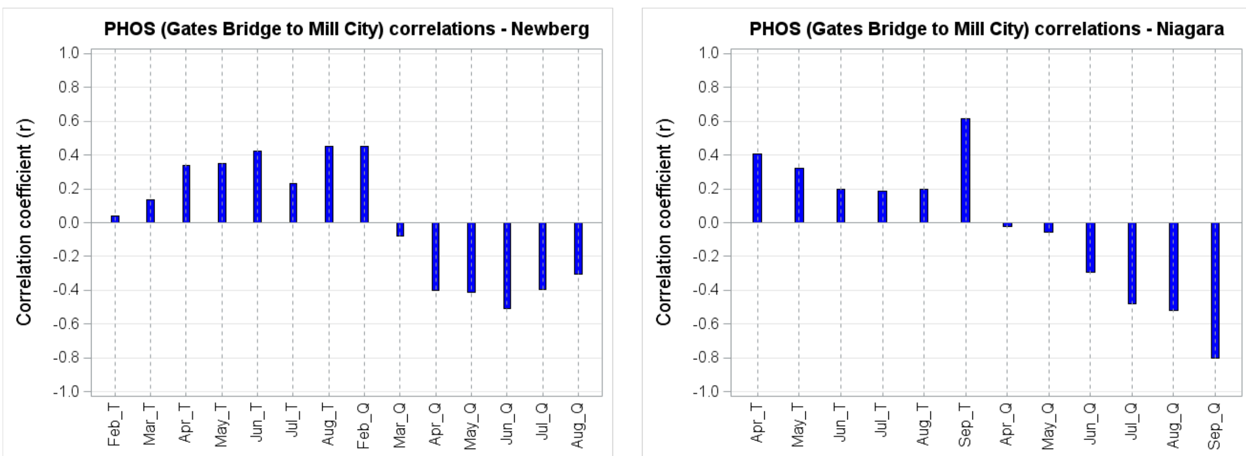


Figure 40. Summary of correlation coefficients (*r*) between annual Chinook salmon PHOS estimates in the Gates Bridge–Mill City reach and mean monthly water temperature (T) and discharge (Q) values collected at the Newberg gage on the Willamette River (left) and the Niagara gage on the North Santiam River (right), 2000–2016.

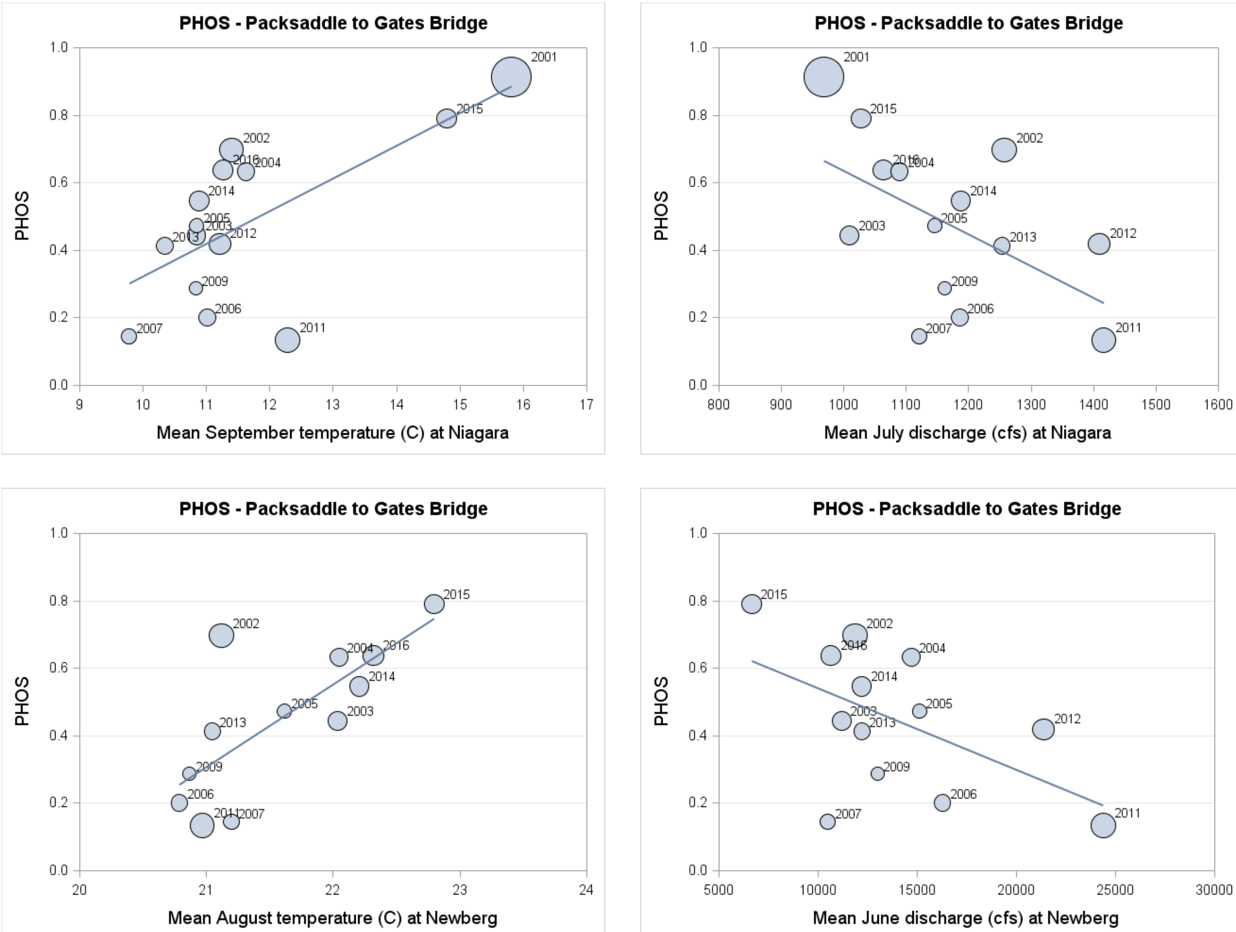


Figure 41. Linear regression relationships between annual Chinook salmon PHOS and covariates in the Packsaddle–Gates Bridge reach, 2000–2016. Predictor variables are mean September water temperature at Niagara (top left), mean July discharge at Niagara (top right), mean August water temperature at Newberg (bottom left), and mean June discharge at Newberg (bottom right). Circles are scaled to the number of carcasses.

***Uncertainties and recommendations* –** The PHOS metric is potentially sensitive to the carcass survey biases described previously, particularly because hatchery-origin carcasses concentrate in different reaches than natural-origin carcasses. Additional challenges associated with PHOS include: (1) uncertainty related to the covariance and potential cause-and-effect relationships between PSM and PHOS (e.g., density-dependent effects); (2) uncertainty regarding differential susceptibility to PSM between hatchery- and natural-origin fish independent of density effects (e.g., Bowerman et al. 2018); (3) uncertainty about year-to-year differences in the relative abundance of the two groups (i.e., how should PHOS estimates be interpreted given uncertainty about the enumeration of hatchery- and natural-origin fish that enter spawning reaches?); (4) covariance between North Santiam River temperature and discharge made it difficult to separate effects (e.g., was low discharge a cause of increased PHOS, an indirect effect associated with carcass recovery probability, or simply a secondary indicator for temperature effects?); and (5) disentangling the effects of Willamette River versus North Santiam River environment on PHOS was difficult due to the covariance among river reaches within and across years. Given the management emphasis on reducing PHOS in the

North Santiam reaches downstream from Big Cliff Dam, we recommend continued enumeration efforts and carcass surveys. A more complete accounting of adults entering the study area, including in the Little North Santiam, could help address some of the enumeration and relative abundance uncertainties.

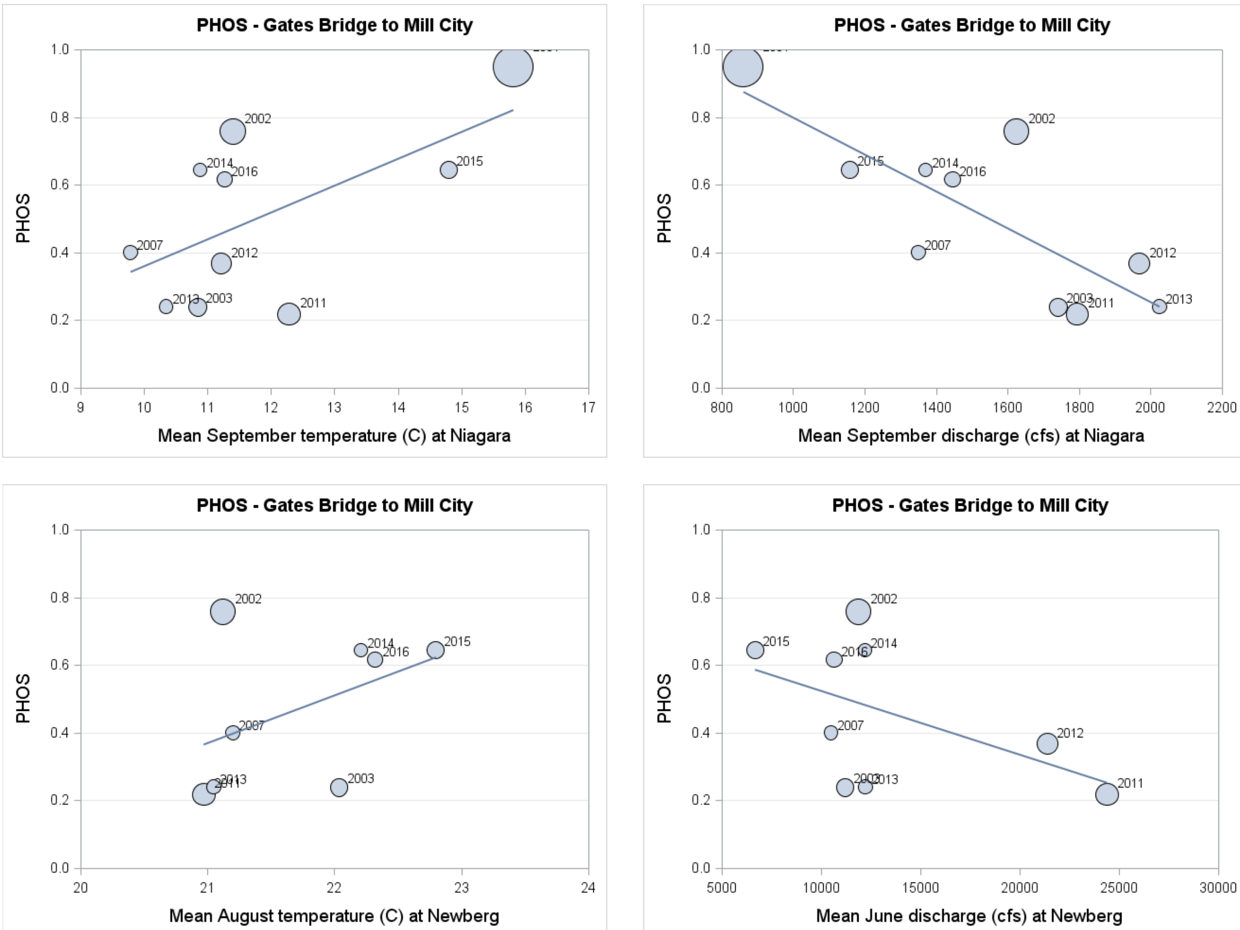


Figure 42. Linear regression relationships between annual Chinook salmon PHOS and covariates in the Gates Bridge–Mill City reach, 2000–2016. Predictor variables are mean September water temperature at Niagara (top left), mean September discharge at Niagara (top right), mean August water temperature at Newberg (bottom left), and mean June discharge at Newberg (bottom right). Circles are scaled to the number of carcasses.

6. Radio-tagged Chinook salmon

Adult Chinook salmon were collected and radio-tagged at Willamette Falls in a series of studies from 2011–2014. Details on trapping, handling, fish selection, tagging, and monitoring have been described previously (e.g., Caudill et al. 2014; Jepson et al. 2015; Keefer et al. 2015, 2017). A total of 1,350 adults were tagged in the four study years, including 831 with fin clips and 519 with no clips; unclipped fish were preferentially tagged in some years to address specific study objectives. Over the four years, 161 radio-tagged adult Chinook salmon were last detected at sites in the North Santiam River; annual sample sizes were 13 (2011), 44 (2012), 34 (2013), and 70 (2014). Migration histories for these groups were used in the upstream migration and behavioral summaries below.

Individual fish movement histories provide a useful supplement to the adult count-based summaries of migration timing and upstream movement described in previous sections. The radiotelemetry data were used to calculate migration timing distributions at locations other than the dam count sites, to assess the relationship between river temperature and fish migration rates, and to calculate fish residence times in specific river reaches. These metrics are particularly important for understanding how temperature management actions at Detroit Dam may affect the behavior and thermal exposure of adults and they were used to parameterize the thermal exposure model described in Section 8. The telemetry data were also used to summarize adult Chinook salmon behaviors near the Minto Fish Collection Facility.

***Migration timing* –** Adults radio-tagged at Willamette Falls were not collected in proportion to the annual runs for a variety of logistical reasons. However, in all four years fish were tagged from April to July. The median tagging dates for the sub-samples that returned to the North Santiam River basin were in early June in all years. For comparison, the aggregate runs at Willamette Falls (all Chinook salmon populations) had median dates that ranged from mid-May to early June (see Figure 16).

Radio-tagged salmon were detected entering the North Santiam River from April into late July or early August (Figure 43). Median dates were in the last week of June (2011 and 2012) or mid-June (2013 and 2014). Median first detection dates at the Bennett dams were 3–6 d later, on median.

***Upstream migration times* –** We calculated salmon passage and/or residence times through six consecutive study reaches: Willamette Falls – Santiam mouth, Santiam mouth – North Santiam mouth, North Santiam mouth – First detection at Bennett, First – Last detection at Bennett, Last detection at Bennett – First detection at Minto, and First – Last detection at Minto. We also calculated the cumulative time from Willamette Falls to first detection at the Bennett dams and from last detection at Bennett to the last detection at Minto (an area with highly variable behavior and long prespawn holding times). Passage times in all reaches were quite variable, but especially upstream from the Bennett dams where some salmon had long periods of prespawn holding prior to moving into the Minto trap or to presumed spawning areas (Table 6, Figure 44). Most fish spent a majority of their total time at large (i.e., migration time + prespawn holding time) in the reaches upstream from the Bennett dams.

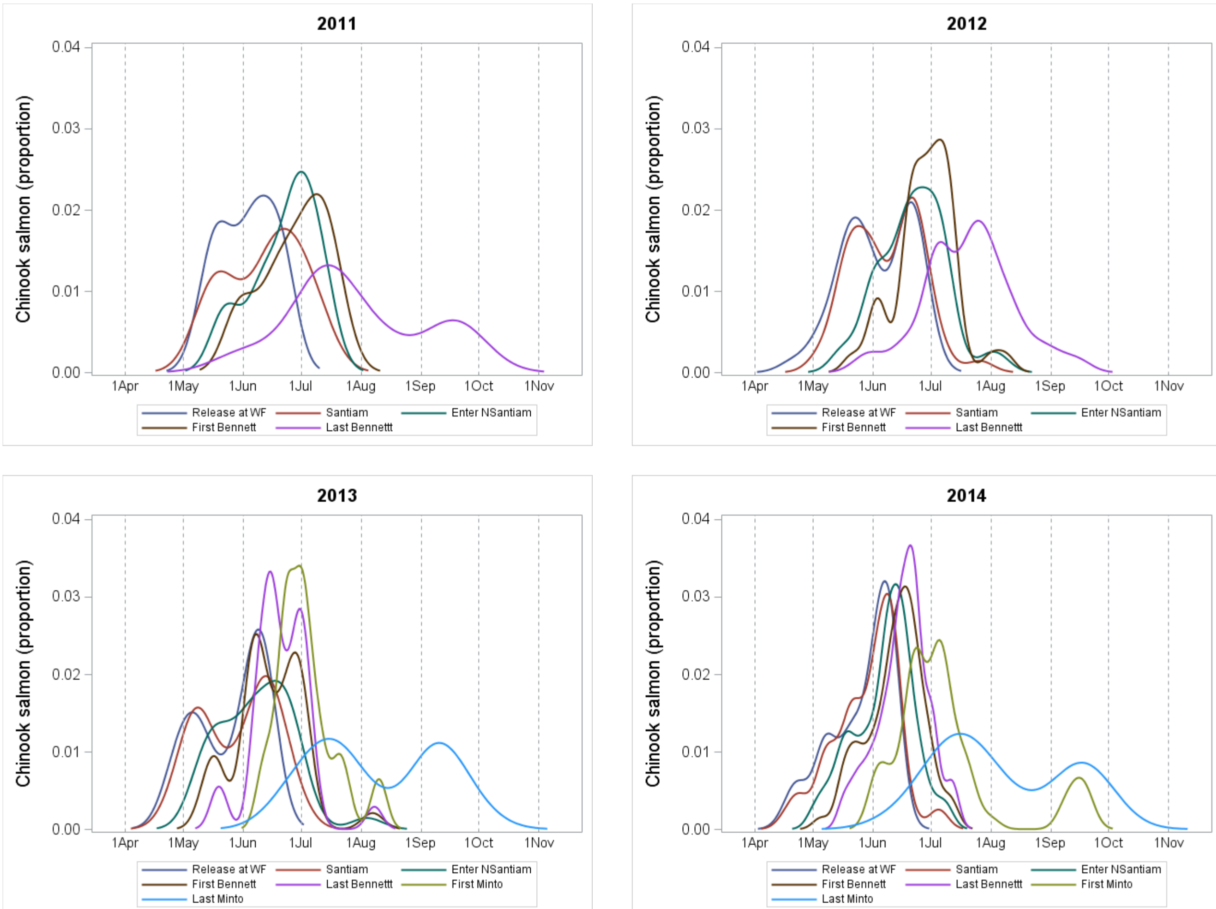


Figure 43. Annual run timing distributions of adult Chinook salmon that were radio-tagged at Willamette Falls and eventually returned to the North Santiam River in 2011–2014. Locations included Willamette Falls, Santiam River entry, North Santiam River entry, first and last detections at the Bennett dams, and first and last detections near the Minto Fish Collection Facility. Distributions were smoothed using a kernel density estimator of 0.5. There were no monitoring antennas at Minto in 2011–2012.

Table 6. Reach passage times for radio-tagged adult Chinook salmon that returned to the North Santiam River, 2011–2014.

Reach	Years	*n*	Passage / Residence time (d) Median	Mean	SD
Will Falls – Santiam mouth	2011–2014	158	7.43	10.70	8.08
Santiam mouth – North Santiam mouth	2011–2014	155	2.08	3.90	6.87
North Santiam mouth – First Bennett	2011–2014	157	4.98	6.28	4.43
First Bennett – Last Bennett	2013–2014	102	1.43	4.52	7.17
Last Bennett – First Minto	2013–2014	91	10.15	17.85	23.13
First Minto – Last Minto	2013–2014	91	24.91	32.58	30.94
Willamette Falls – First Bennett	2011–2014	158	17.77	21.09	11.83
Last Bennett – Last Minto	2013–2014	91	38.06	50.43	33.46

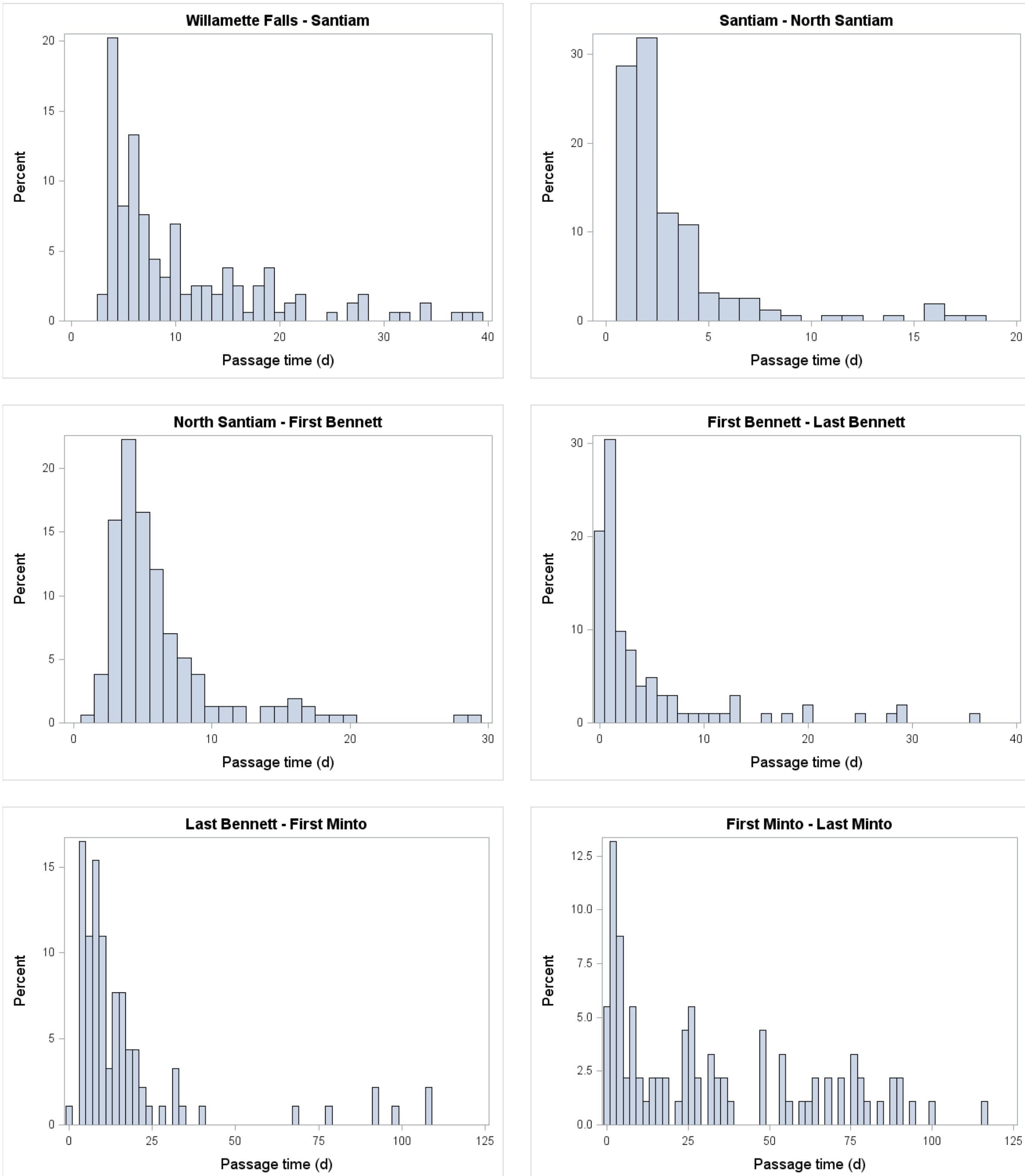


Figure 44. Distributions of radio-tagged adult Chinook salmon passage times through monitored reaches of the Willamette, Santiam, and North Santiam rivers, 2011–2014.

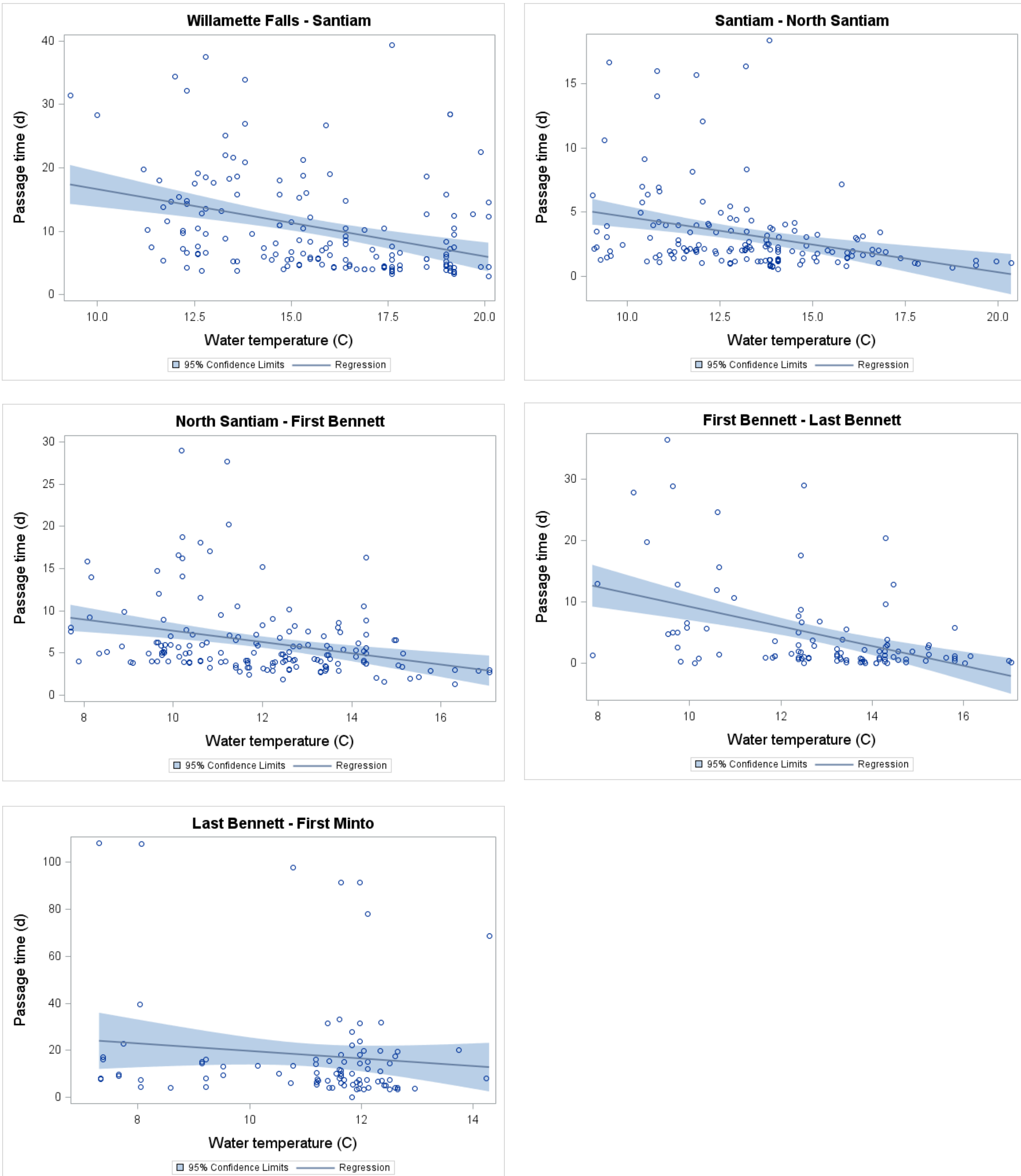


Figure 45. Liner regression relationships between the reach-specific passage times of radio-tagged adult Chinook salmon and mean daily water temperature on the date salmon entered each reach. Temperatures were from the Newberg (Willamette), Greens Bridge, Mehama, or Niagara (North Santiam) gage sites. Data from 2011–2004 were combined for the reaches below Bennett and for 2013–2014 for the at-Bennett and above Bennett reaches. Shaded areas show 95% confidence limits for regression lines.

In the downstream reaches, where salmon spent relatively little time holding, migration times were negatively correlated with water temperatures on the dates that fish entered reaches (Figure 45). Regression relationships were weak (all $r^2 < 0.20$), as some fish had long reach passage times across the encountered temperature spectrum. Nonetheless, faster passage at warmer temperatures was clearly evident. This behavioral response has been reported in several other spring Chinook salmon studies of migration rates (e.g., Keefer et al. 2004; Salinger and Anderson 2006; Strange 2012; Jepson et al. 2015). The consequences of variable and temperature-dependent migration rates include relatively slow upstream passage for early-run migrants relative to late-run migrants and within-run differences in where individuals spend their time. Such differences are potentially important for understanding and predicting effects of temperature management.

Thermal exposure histories for salmon with temperature loggers – Six of the radio-tagged North Santiam River fish also carried temperature loggers that were recovered at the Minto Fish Collection Facility (see Keefer et al. 2015 for logger details). Thermal histories for these fish indicated that mean daily temperatures that fish encountered closely tracked the fluctuations in mean daily values recorded at the gage sites (Figure 46). The four examples in Figure 46 show logger temperatures that paralleled those at the Greens Bridge and Mehama gages during the first 1–2 weeks of the time fish spent in the North Santiam River. Thereafter, logger temperature continued to track day-to-day fluctuations, and mean values fell between those recorded at the Mehama and Niagara gage sites. These histories are wholly consistent with prespawn holding between the Bennett dams and Minto, as this reach is between the two gage sites. The data also lack evidence of spatial behavioral thermoregulation (i.e., use of relatively cool- or warm-water locations).

Salmon behavior near the Minto Fish Collection Facility – The new Minto facility was operational starting in 2013 so we included only radiotelemetry data from 2013 and 2014 in this summary. Telemetry monitoring sites in the two years included two antennas near the Bennett dams, on antenna just upstream from Upper Bennett Dam, on antenna in the Little North Santiam River, and antennas in the Minto tailrace, base of the fish ladder, and at the trap weir. A single antenna was also located in the tailrace of Big Cliff Dam.

In 2013, 34 radio-tagged salmon, 26 with fin clips and 8 without clips, were detected upstream from the Bennett dams (Table 7). Similar proportions of each group (73–75%) were subsequently collected at the Minto trap (Pearson's $\chi^2 = 0.0$, $P = 0.914$). In 2014, 69 fish passed Bennett: 65% of fin-clipped fish and 78% of unclipped fish eventually entered the trap ($\chi^2 = 1.2$, $P = 0.267$). There was also no statistical difference in the percent trapped by origin when we limited the statistical tests the subset of salmon that were detected in the Minto tailrace ($n = 31$ in 2013 and 61 in 2014; Table 7).

Salmon migration times from Bennett to the Minto tailrace were highly variable, ranging from 2.7–97.7 d (*mean* = 17.8 d, *sd* = 18.5 d). Similarly, times from first detection in the Minto tailrace to first detection at the weir antenna ranged from 0.2–84.1 d (*mean* = 14.5 d, *sd* = 18.3 d). The individual detection histories indicated that some fish moved downstream after entering the Minto tailrace, and some entered and exited the fishway. Most, however, appeared to be fairly stationary, with long periods between detections before eventually entering the trap. In several exploratory analyses, trap entry did not appear to be strongly associated with North Santiam River temperature or discharge at the Niagara site.

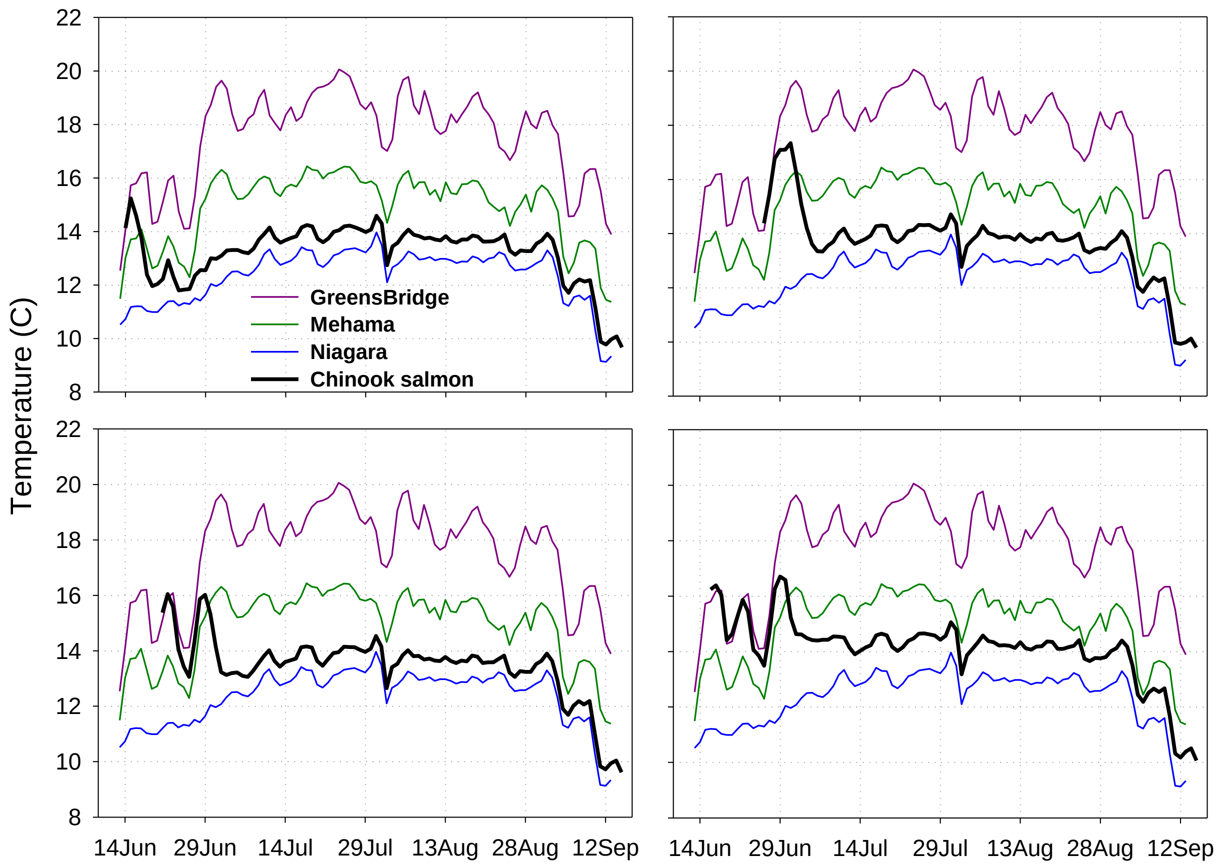


Figure 46. Four examples of mean daily adult Chinook salmon body temperatures in the North Santiam River in 2013 in relation to river temperatures at Greens Bridge, Mehama, and Niagara. Fish temperature data were collected using archival thermal loggers on salmon radio-tagged and released at Willamette Falls and then recaptured at the Minto Fish Collection Facility.

Table 7. Numbers of radio-tagged adult Chinook salmon that passed the Bennett dams in 2013–2014 and their subsequent detection in the Minto tailrace and Minto Fish Collection Facility. 'Fate' refers to whether or not salmon were collected at the trap.

	Passed Bennett dams				Detected in Minto tailrace			
Year	*n*	Fate	Fin clip	No clip	*n*	Fate	Fin clip	No clip
2013	34	Minto	19	6	31	Minto	19	6
		No Minto	7	2		No Minto	6	0
2014	69	Minto	30	18	61	Minto	30	18
		No Minto	16	5		No Minto	10	3

***Uncertainties and recommendations* –** The radiotelemetry study, though not designed to address the current study objectives, did provide useful data for individual adult salmon behaviors in the North Santiam River, including near the Minto collection facility. Uncertainties about the dataset include: (1) the potential effects on non-representative sampling of the adult runs at Willamette Falls (e.g., Keefer et al. 2017); (2) the potential for handling and/or radio-tagging (e.g., Caudill et al. 2014) to affect subsequent fish behaviors in the North Santiam River; (3) the geographically-limited monitoring effort upstream from the Bennett dams and at Minto made it difficult to infer fine-scale fish movements, holding locations, or potential mechanisms that affected Minto entry/non-entry; and (4) it is uncertain whether the behaviors of radio-tagged salmon in 2011–2014 reasonably reflect behaviors of untagged salmon, or behaviors in years with substantively different environmental conditions. Improved understanding about Minto collection effectiveness may require more targeted research, potentially including tagging studies or passive monitoring such as sonar or video (e.g., Clabough et al. 2017; Keefer et al. 2018a).

Summary of Proposed Water Temperature Management Scenarios

7. Simulated temperature results for 2011 and 2015

The four scenarios – Daily water temperature data for the North Santiam River downstream from Big Cliff Dam were generated using the CE-QUAL-W2 and HEC-RAS models and four scenarios in each year. The scenarios were: (1) pre-dam temperature target with a hypothetical control tower at Detroit Dam; (2) pre-dam temperature target with no control tower; (3) ODFW temperature target with a hypothetical control tower at Detroit Dam; and (4) ODFW temperature target with no control tower. In all simulations, downstream target temperatures were achieved on some – but not all – dates due to the constraints imposed by model inputs (i.e., observed hydrological and meteorological data).

Niagara 2011 **–** We compared the data generated with the four temperature management scenarios to the observed data. In the scenario using the pre-dam target, the modeled no-tower temperatures closely tracked the observed temperatures (Figure 12). Difference between mean monthly observed and modeled temperatures were ±0.50 °C in all months except October, when the modeled mean was 1.02 °C warmer than the observed mean (Figure 13). In the simulated pre-dam target, yes-tower scenario, the modeled temperatures were considerably warmer than the observed means in May and June (+1.2–1.3 °C) and July and August (+2.5–3.5 °C). The pre-dam target, yes-tower scenario results largely achieved the temperature management objectives (see Figure 3).

The ODFW target, no-tower scenario produced temperatures that were slightly (< 0.5 °C) cooler than the observed temperatures in five of seven months (Figures 12 and 13). The modeled October mean was ~1.2 °C warmer than observed, similar to in the pre-dam, no-tower model. In the ODFW target, yes-tower scenario, modeled temperatures were warmer than observed in May through July (+1.2–1.3 °C) and were slightly cooler than observed in August and September (-0.2–0.3 °C).

Niagara 2015 **–** The warm, low-flow river conditions in 2015 made it difficult to reach either the pre-dam or ODFW target temperatures without a temperature control structure. Compared to the observed water temperatures, in both no-tower scenarios the modeled mean temperatures were near observed temperatures in April through July and were considerably warmer (+2.6–2.7 °C) than observed in August and September (Figures 12 and 13).

Modeled temperatures were closer to target values in the yes-tower scenarios. With the pre-dam target, monthly means were warmer than observed in May through July (+2.1–3.2 °C) and were cooler than observed in August and (-1.5 °C) and September (-0.6 °C). With the ODFW target, monthly means were warmer in April through June (+0.8–2.1 °C) and were cooler in July and August (-2.7 °C); October modeled temperatures were warm (Figures 12 and 13).

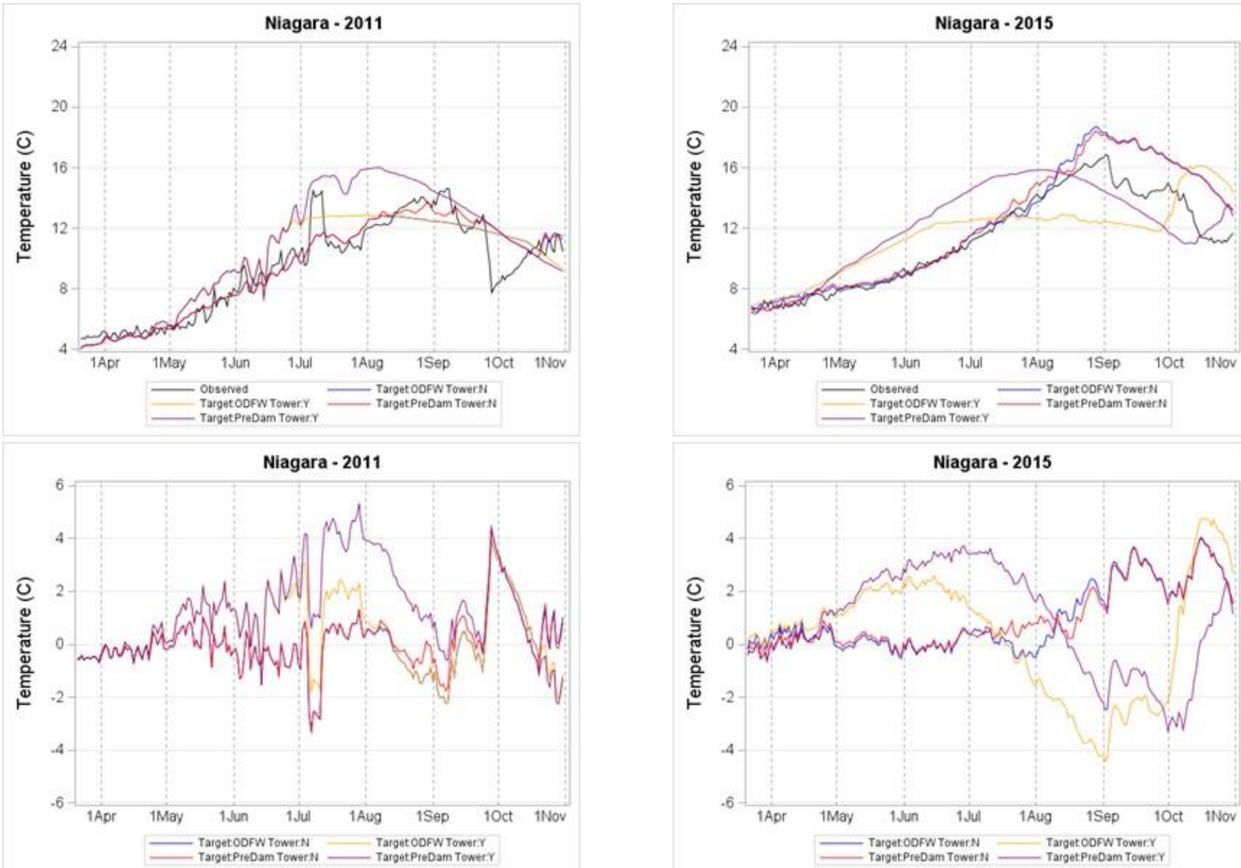


Figure 12. Top panels: Observed and simulated mean daily water temperatures (°C) in the North Santiam River at Niagara in 2011 and 2015. Bottom panels: Differences (°C) between observed and simulated daily water temperatures. The four simulated scenarios were based on PreDam or ODFW targets, with and without a hypothetical temperature control structure at Detroit Dam.

Mehama 2011 **–** As expected, the model scenarios produced temperature patterns at Mehama that closely paralleled those at Niagara, though the magnitude of the temperature differences from observed values was generally lower at Mehama (Figures 13 and 14). In the 2011 no-tower scenarios, mean monthly differences from observed were all ±0.6 °C; the largest differences were cooler modeled temperatures in June and warmer temperatures in October. In the 2011 yes-tower scenarios, differences from observed were ±0.7 °C in all months except July, when difference were+2.4 °C (pre-dam target) and +1.0 °C (ODFW target).

Mehama 2015 **–** In 2015, modeled temperatures were warmer than observed in the no-tower scenarios in all months except April (Figures 13 and 14). No-tower monthly means were warmer by 0.7–1.1 °C in June through August and by 1.7–1.8 °C in September and October. In contrast, the yes-tower scenarios resulted in some late summer cooling, particularly with the ODFW target, where modeled means were 1.6–2.1 °C cooler than observed in August and September (Figures 13 and 14). In the yes-tower scenarios, modeled means were higher than observed in May through July; the largest differences were in the pre-dam target model in June and July (+2.7 °C).

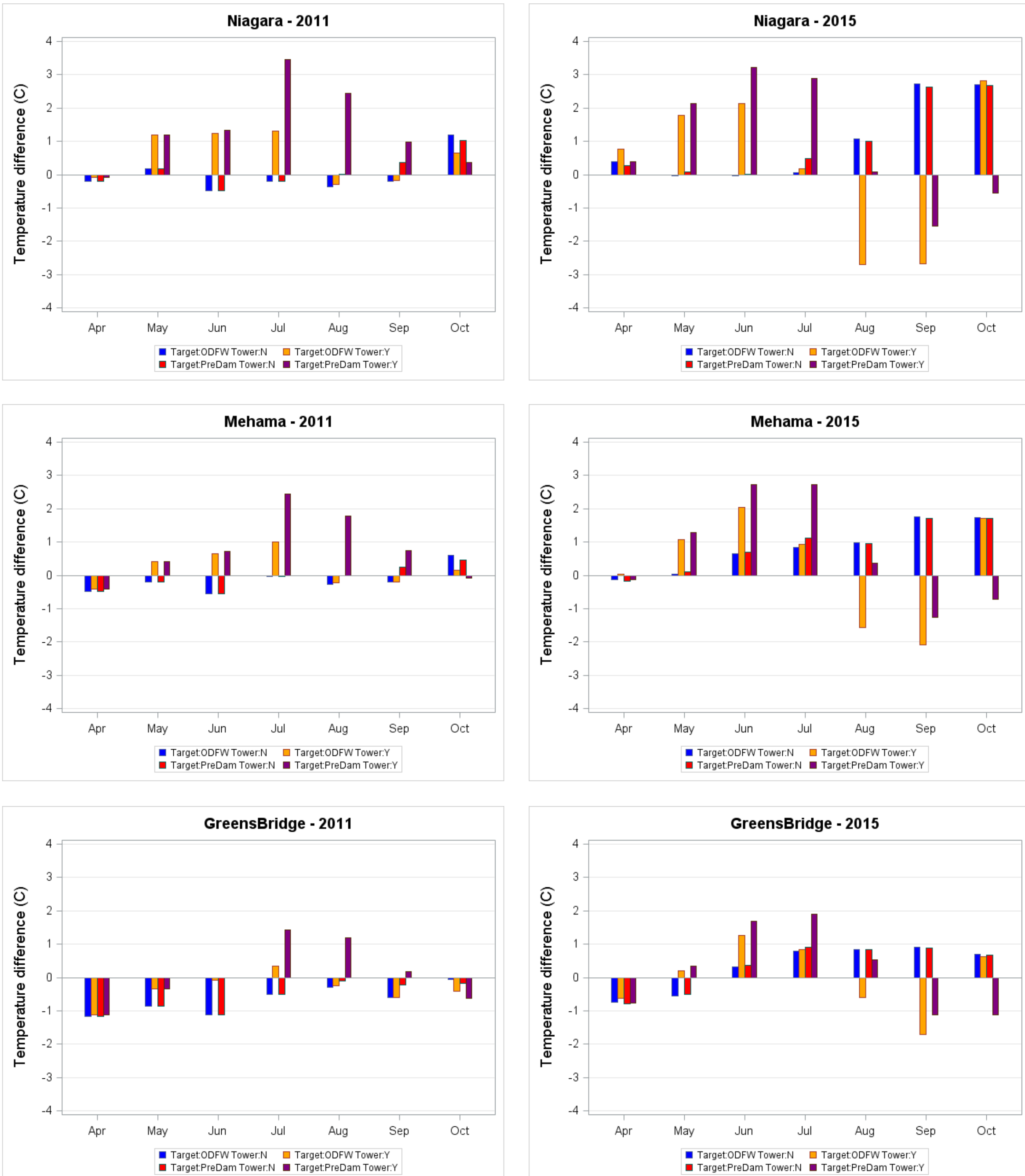


Figure 13. Mean monthly differences (°C) between observed water temperatures and the four simulated temperature scenarios at Niagara, Mehama, and Greens Bridge sites in 2011 and 2015.

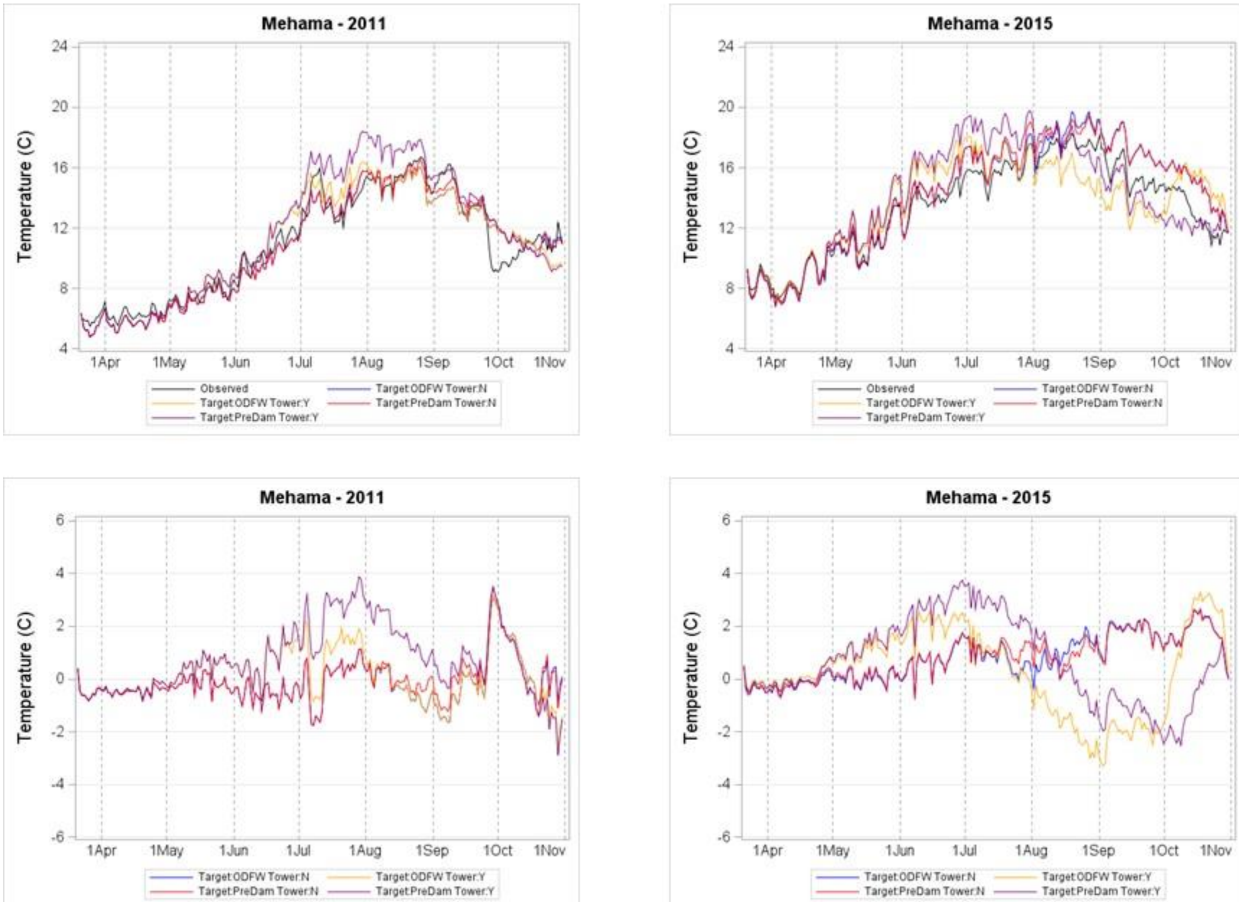


Figure 14. Top panels: Observed and simulated mean daily water temperatures (°C) in the North Santiam River at Mehama in 2011 and 2015. Bottom panels: Differences (°C) between observed and simulated daily water temperatures. The four simulated scenarios were based on PreDam or ODFW targets, with and without a hypothetical temperature control structure at Detroit Dam.

***Greens Bridge 2011 and 2015* –** In general, the temperature scenarios at Greens Bridge paralleled those at the upstream sites (Figures 13 and 15). However, a notable difference was that modeled temperatures were somewhat cooler (-0.6–1.1 °C) than observed in April and May of both years and in June (-1.1 °C) of 2011. The yes-tower scenarios produced cooler August and September temperatures (-0.6–1.7 °C) with the ODFW target and cooler September and October temperatures (-1.1 °C) with the pre-dam target. Summer and fall temperatures were 0.3–0.9 °C warmer in the no-tower scenarios.

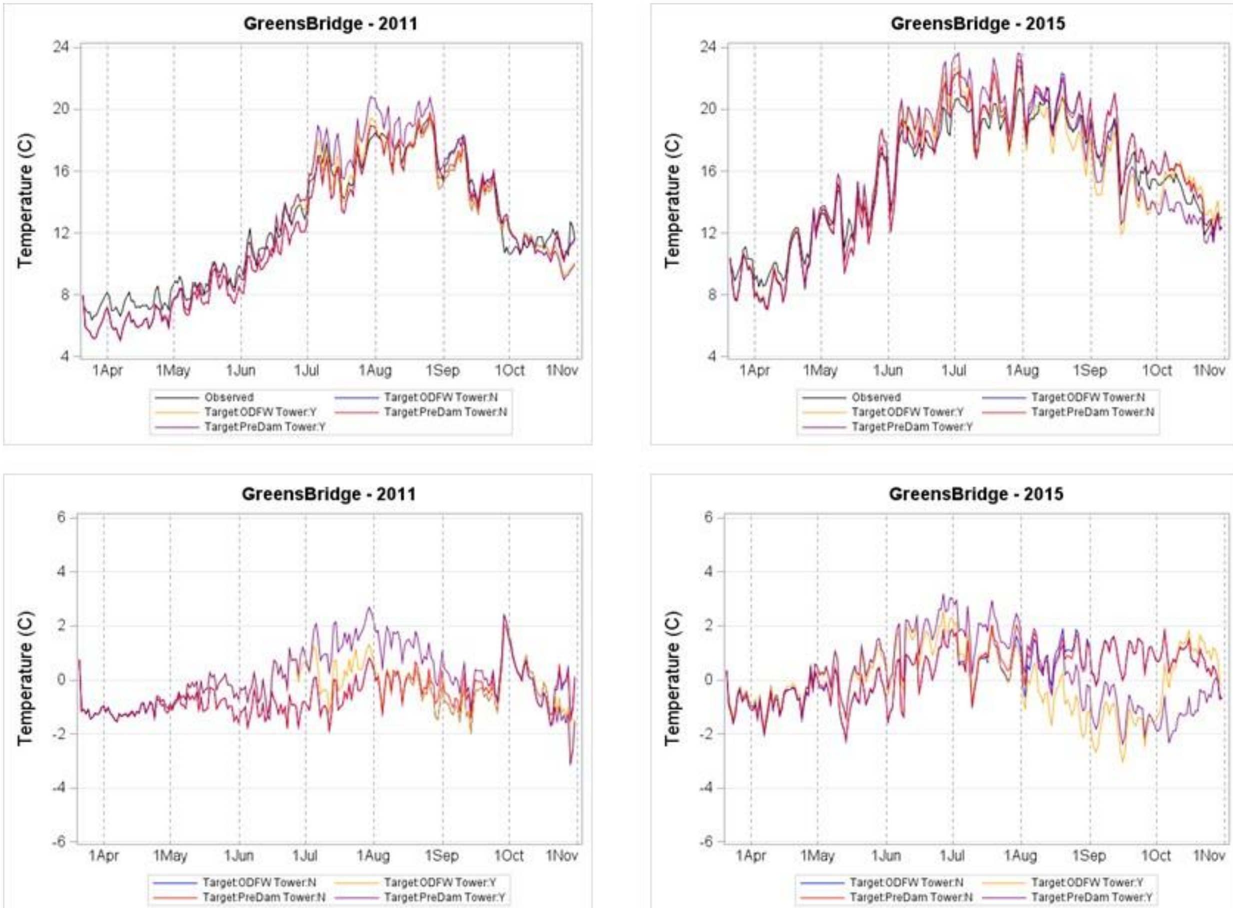


Figure 15. Top panels: Observed and simulated mean daily water temperatures (°C) in the North Santiam River at Greens Bridge in 2011 and 2015. Bottom panels: Differences (°C) between observed and simulated daily water temperatures. The four simulated scenarios were based on PreDam or ODFW targets, with and without a hypothetical temperature control structure at Detroit Dam.

Effects of simulated Niagara temperatures on Chinook salmon PSM and PHOS **–** In a simple application, we used the monthly mean temperature×PSM and temperature×PHOS correlations to predict these salmon metrics under the four simulated temperature scenarios for 2011 and 2015. The selected reach was from Upper Bennett Dam to Big Cliff Dam, the section with the largest carcass sample sizes, and the selected temperature site was Niagara, because temperatures there were most responsive to the Detroit management scenarios (see Figures 32 and 38 for the empirical correlation results).

In the 2011 simulations, the warmer mean temperatures for June and July resulted in notably lower predicted annual PSM for the two yes-tower scenarios, as temperature in these months was negatively correlated with PSM in the empirical data (Figure 32). Estimates were 31–33% (ODFW target, yes-tower) and 20–30% (pre-dam target, yes-tower) versus 41–44% using the observed temperatures in the correlation model (Table 8). Conversely, warmer September temperatures in the pre-dam target, yes-tower scenario resulted in higher annual PSM (57%) compared to the correlation based on the observed September temperature results (48%), as temperature in September was positively correlated with PSM in the empirical data. None of the simulated temperatures resulted in decreased PHOS of >5% (Table 8). However, in the correlation-based predictions, the yes-tower scenarios were associated with considerably higher PHOS for several months (Table 8).

The simulated temperatures in 2015, when used in the correlation analysis, resulted in mixed predicted effects on Chinook salmon (Table 9). In the two yes-tower scenarios, higher PSM was predicted from the simulated May water temperatures, whereas lower PSM was predicted from the June, July, and September temperatures. The no-tower predictions varied little from the observed temperature predicted PSM, except for the September data, where warmer simulated temperatures resulted in higher predicted PSM. Predicted PHOS also had mixed results in both yes-tower scenarios, with warmer simulated temperatures in early summer resulting in higher PHOS, while cooler August and September temperatures resulted in lower PHOS (Table 9).

***Uncertainties and recommendations* –** We did not have any specific concerns about the models used to simulate the North Santiam River temperatures, which were previously vetted (e.g., Sullivan et al. 2007; Buccola et al. 2012, 2015; Rounds and Buccola 2015). However, we recommend that additional management scenarios and additional example years should be explored. The demonstration years we selected (2011, 2015) provided just two points along a continuum of North Santiam water years. The 2011 scenarios were probably reasonably representative of cool, high-flow years. In contrast, conditions in 2015 may have been appropriate for a relatively narrow range of years near the upper end of the warm-dry spectrum (and perhaps of future conditions). It would be instructive to run scenarios for near-average water years and less extreme warm-dry years and to better identify the limits of temperature management (i.e., the range of conditions under which temperature targets cannot be reached).

Table 8. Mean observed and simulated daily water temperatures in the North Santiam River at Niagara in 2011, with predicted Chinook salmon PSM and PHOS estimates in the combined reaches from upper Bennett Dam to Big Cliff Dam. Estimates were predicted from the monthly correlations with temperature shown in Figures 32 (PSM) and 38 (PHOS); note that September water temperature had the highest correlations with PSM and PHOS. Blue-shaded cells indicate estimates were ≥5% lower than and red-shade cells indicate estimates were ≥5% higher than under the observed temperature correlation.

Month	Observed	Target:ODFW Tower:No	Target:ODFW Tower:Yes	Target:PreDam Tower:No	Target:PreDam Tower:Yes
	Observed and simulated temperatures (°C) at Niagara				
April	5.10	4.88	5.01	4.88	5.01
May	6.36	6.54	7.56	6.54	7.56
June	9.16	8.67	10.40	8.67	10.49
July	11.41	11.22	12.72	11.21	14.87
August	13.00	12.64	12.71	13.01	15.44
September	12.29	12.10	12.11	12.66	13.27
	Predicted annual PSM (%)				
April	45.5	46.4	45.9	46.4	45.9
May	39.9	40.5	44.1	40.5	44.1
June	43.5	48.3	31.4	48.3	30.4
July	41.1	42.2	33.0	42.3	19.6
August	41.9	41.9	41.9	41.9	41.7
September	48.4	46.7	46.8	51.7	57.0
	Predicted annual PHOS (%)				
April	25.1	20.5	23.1	20.5	23.1
May	30.5	34.5	56.4	34.5	56.4
June	41.9	39.4	48.2	39.4	48.7
July	43.3	42.7	47.0	42.7	53.1
August	45.5	44.6	44.8	45.6	51.6
September	50.6	48.7	48.8	54.5	60.7

Table 9. Mean observed and simulated daily water temperatures in the North Santiam River at Niagara in 2015, with predicted Chinook salmon PSM and PHOS estimates in the combined reaches from upper Bennett Dam to Big Cliff Dam. Estimates were predicted from the monthly correlations with temperature shown in Figures 32 (PSM) and 38 (PHOS); note that September water temperature had the highest correlations with PSM and PHOS. Blue-shaded cells indicate estimates were ≥5% lower than and red-shade cells indicate estimates were ≥5% higher than under the observed temperature correlation.

	Observed and simulated temperatures (°C) at Niagara				
Month	Observed	Target:ODFW Tower:No	Target:ODFW Tower:Yes	Target:PreDam Tower:No	Target:PreDam Tower:Yes
April	7.21	7.61	7.97	7.48	7.60
May	8.34	8.31	10.13	8.44	10.47
June	9.94	9.90	12.08	9.96	13.147
July	12.47	12.54	12.64	12.95	15.36
August	15.34	16.37	12.59	16.31	15.39
September	14.80	17.52	12.13	17.44	13.26
	Predicted annual PSM (%)				
April	36.0	34.2	32.6	34.8	34.2
May	46.8	46.7	53.1	47.2	54.3
June	35.9	36.2	14.8	35.7	4.1
July	34.5	34.1	33.4	31.5	16.6
August	41.7	41.7	41.9	41.7	41.7
September	70.5	94.4	47.0	93.7	56.9
	Predicted annual PHOS (%)				
April	70.0	78.5	86.2	75.8	78.3
May	73.4	72.8	out of range	75.6	out of range
June	45.9	45.7	56.8	46.0	62.4
July	46.3	46.5	46.8	47.7	54.5
August	51.3	53.9	44.5	53.7	51.4
September	76.4	n/a	49.1	out of range	60.6

8. Exposure model development

Model overview – It is clear that the thermal exposure of North Santiam River Chinook salmon varies widely among years. Perhaps just as importantly, their exposure can vary substantially within migration years, with potentially very different experiences for early-run versus late-run migrants (Figure 47). The total freshwater residence times of early-run spring Chinook salmon may be 90 d or more longer than residence times of late-run fish. The early group may be disproportionately vulnerable to the proliferative pathogens that have been associated with prespawn mortality in the Willamette population and elsewhere (e.g., Kent et al. 2013; Benda et al. 2015). Early-run fish may also be susceptible to accelerated senescent processes, particularly if water temperatures are warm or other stressors are present (e.g., Morbey et al. 2005; Quinn et al. 2016). Risks specific to late-run fish include exposure to much warmer conditions in the migration corridor, with elevated physiological demands (e.g., Hague et al. 2011; Eliason et al. 2013), increased stress response (McCullough et al. 1999, 2009; Hinch et al. 2012), and higher potential en route and prespawn mortality (e.g., Naughton et al. 2005; Keefer et al. 2008b, 2010; Bowerman et al. 2016, 2018). It is also possible that the warm conditions encountered by late-run fish may exacerbate some pathogens and parasites (e.g., Bradford et al. 2010).

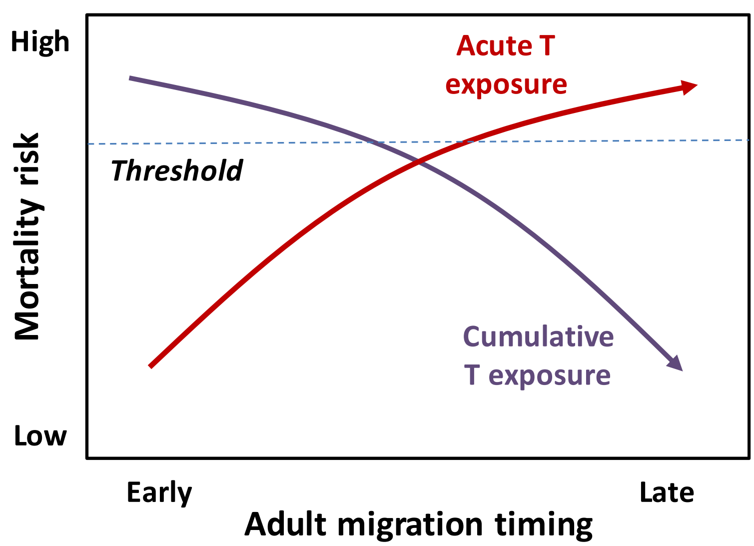


Figure 47 (repeat of Figure 5). Hypothetical relationship between adult Chinook salmon migration timing and two temperature-mediated risk factors: exposure to acutely-stressful condition in the migration corridor versus long freshwater residence times and high cumulative exposure.

The individual-based model we developed for North Santiam Chinook salmon was structured to capture the among-year and within-run variation in thermal exposure. The model was constructed in SAS version 9.4 (SAS Institute, Cary, NC). It was built on a daily time step and includes fish movements through multiple river reaches, starting at Willamette Falls on the

Willamette River and ending at or downstream from the Minto Fish Collection Facility on the North Santiam River. Individual thermal histories were generated by pairing salmon location with the nearest river temperature data from USGS gages along the migration route. For this report, we ran the model with observed river temperatures (Section 9) and the simulated temperatures from the Detroit Dam management scenarios (Section 10).

Model step 1: Virtual sample of adult Chinook salmon **–** One of the conclusions from the 2011–2014 radiotelemetry study was that adults from the upper Willamette River sub-basin populations were well-mixed throughout their migrations at Willamette Falls (Jepson et al. 2015). We therefore seeded the thermal exposure model with randomly-selected samples from the adult migration timing distributions at the falls (e.g., Figure 48, also see Appendix B). Random draws ensured that fish in the model were reasonably representative of the run in the modeled year.

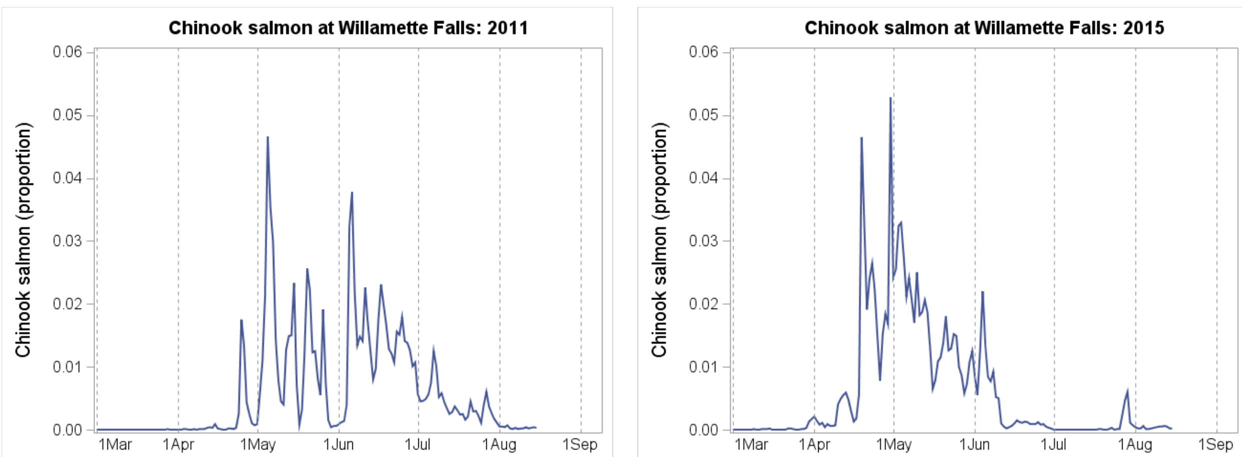


Figure 48. Daily counts of adult Chinook salmon at Willamette Falls, expressed as a proportion of the total runs in a relatively cool, high-flow year (2011, left) and a warm, low-flow year (2015, right). Random samples drawn from the annual run-timing distributions were used to seed the temperature exposure simulations. Data source: *www.dfw.state.or.us/fish/fish_counts/willamette%20falls.asp*.

Model step 2: Upstream movement rules **–** We simulated salmon movement upstream by using the reach-specific passage time data from the radiotelemetry study. The five model reaches are shown in Table 10 and include reaches for the main stem Willamette and lower Santiam rivers and three sections of the North Santiam River. We fit the radiotelemetry data to a series of Weibull distributions for each reach based on the water temperature salmon encountered on their reach entry dates. Temperature bins were 2 °C, except at the upper and lower distributions, where we grouped data due to small sample sizes. Bootstrapping was used to estimate the three Weibull parameters (theta, sigma, shape) using the random sampling with replacement method in PROC SURVEYSELECT in SAS.

Model step 3: Matching individuals to river temperatures **–** The start dates for the model seed fish were matched with the respective mean daily water temperatures at the Newberg gage site on the Willamette River. Based on those temperatures, a reach passage time was generated for each fish by randomly drawing from the temperature-appropriate Weibull distribution for the first reach in Table 10. Individual thermal histories for the reach were built by matching the daily Willamette River temperature data to each date that each fish remained in the reach.

The model was run iteratively for each successive upstream reach. For example, the dates that individuals exited the Willamette–Santiam mouth reach were used to initiate the sub-model for estimating passage times for the short reach from the Santiam mouth to the North Santiam mouth. Daily temperatures for this second reach were drawn from the Greens Bridge gage, located near the North Santiam River mouth. Temperature data from the lower Santiam River would have been more appropriate for this reach, but the Jefferson gage site (#14189000) stopped collecting temperature data in 1987. Regardless, radio-tagged salmon spent < 4 d in this reach, on average, and temperatures at the Greens Bridge gage were a reasonable proxy. Temperature data from the Mehama site were used while model salmon moved from the lower North Santiam to the Bennett dams and Niagara temperature data were used while salmon were upstream from the Bennett dams.

Table 10. River reach- and water temperature-specific Weibull parameters estimated using passage time data from radio-tagged Chinook salmon in 2011–2014 (reaches from Willamette Falls to first detection at Bennett Dams) or in 2013–2014 (at and upstream from Bennett dams[1]). Parameter estimates theta (threshold), sigma (scale), and shape were estimated by bootstrapping using 1,000 iterations (random sampling with replacement).

Reach	Temperature (°C)	Salmon (*n*)	Theta	Sigma	C
Release – Santiam mouth[2]	< 12	35	4.244	8.099	1.245
	12 to 14	95	3.662	10.991	1.339
	14 to 16	95	3.540	5.777	1.006
	16 to 18	91	3.218	4.715	1.000
	> 18	59	3.002	5.127	1.000
Santiam mouth – N. Santiam mouth	< 10	13	1.410	2.980	1.010
	10 to 12	37	1.202	3.211	1.004
	12 to 14	54	0.789	2.500	1.004
	14 to 16	31	0.650	1.537	1.262
	> 16	20	0.724	1.081	1.269
N. Santiam mouth – First Bennett	< 10	30	3.918	2.999	1.006
	10 to 12	47	2.653	5.399	1.001
	12 to 14	51	2.12	3.259	1.349
	> 14	29	1.418	3.527	1.243
First Bennett – Last Bennett	< 10	14	0.355	10.841	1.076
	10 to 12	13	0.226	5.653	1.015
	12 to 14	39	0.069	3.123	1.000
	> 14	36	0.039	2.406	1.001
Last Bennett – First Minto[3]	< 12	59	3.674	8.850	1.059
	> 12	24	3.345	7.272	1.026

[1] Telemetry data from 2010–2011 at the Bennett dams were excluded due to adult trapping and other modifications
[2] Includes fish last recorded anywhere in the Santiam River basin
[3] Excludes fish with residence times > 90 d (<12 °C) and > 50 d (>12 °C) to achieve convergence

***Model step 4: Complete individual exposure histories* –** We selected 1 September as the terminus for each model run because this date approximated the timing of the earliest successful female spawners (see Figures 28 and 29). In addition, almost all salmon had reached the most upstream reach by 1 September, so extending the model later into the fall resulted in almost all individual experiencing the same temperatures. This had little heuristic value and tended to reduce differences among individuals.

In all of the model summaries presented in Sections 9 and 10, we seeded the model with 1,000 individuals at Willamette Falls. The output included a continuous, spatially-referenced history of mean daily water temperatures for each fish from the falls through 1 September. Some of the model fish encountered water temperatures that were stressful (i.e., > 23 °C), but we did not attempt to incorporate premature mortality into the model structure.

In the summaries below, we present a mix of reach-specific and full-migration estimates of mean and cumulative exposure for the model fish. Because the largest impacts from the temperature management scenarios were realized at the Niagara and Mehama sites (see Figure 13), we emphasize model results from the most upstream reaches where prespawn holding by Chinook salmon was extensive.

***Uncertainties and recommendations* –** The individual-based model we developed was parameterized using relationships derived from the radiotelemetry dataset and river temperatures. The model was intended to identify broad differences in exposure for the North Santiam River population across a variety of temperature scenarios in addition to estimating individual experiences. There are several uncertainties associated with the model, including: (1) the Weibull distributions used to drive upstream movements by salmon were a substantially simplified representation of the observed behavioral data; (2) behaviors of the radio-tagged fish may not have reasonably represented behaviors in other years with different environmental or operational conditions; (3) use of mean daily water temperatures was a simplification of the variation of daily exposure that fish experienced; (4) all model fish ended their migrations near Minto and on 1 September, two artificial constraints intended to simplify among-scenario comparisons; and (5) use of temperature data from four gage sites along the migration route similarly reduced individual variation in exposure that may have been biologically important, particularly for fish with long holding periods.

Behavioral models with finer spatial and temporal resolution (e.g., Snyder et al. 2019) and/or more complex but realistic underlying movement distributions (e.g., Crozier et al. 2017) could be developed for North Santiam River Chinook salmon, but would require better spatially-referenced temperature data and several assumptions about fish behaviors (e.g., movement versus holding) within the study reaches and about final distributions (e.g., among spawning reaches, including the Little North Santiam and downstream from the Bennett dams). Collection of additional individual thermal histories like those in Figure 46 would help validate that salmon are not actively thermoregulating in warmer or cooler patches during migration and holding.

9. Simulation model using the observed temperature data

Reach-specific passage times and mean exposure – We ran the exposure model using observed water temperature data from the five years when there was nearly complete daily data available from April–September at the Newberg, Greens Bridge, Mehama, and Niagara gages (see Table 1). The years were 2010, 2011, 2013, 2014, and 2015. Mean modeled passage times for the Willamette River reach ranged from 6.5–11.6 d (Figure 49). The fastest passage through this reach was in 2013, when modeled mean temperatures were highest. Mean modeled passage times through the three reaches from the Santiam River mouth to pass the Bennett dams ranged from 2.9–3.5 d, 6.1–6.9 d, and 5.4–7.8 d, respectively, with relatively little among-year variability. Mean temperature exposures in these reaches were in a fairly narrow range (Figure 49).

Modeled passage times from the Bennett dams to first detection at Minto ranged from 11.4–12.2 d. Very low among-year variation in this reach reflected the similarity in the Weibull distributions for the narrow range of temperatures encountered by model fish (see Table 10).

Time fish spent near Minto (Reach 6) was the most variable among reaches (Figure 49). The shortest modeled residence time in this reach was in 2011 (51 d), when migration timing was relatively late and river temperatures were relatively cool. The longest mean time (78.7 d) was in 2015 when migration timing was early and river temperatures were warm. The distributions for mean individual exposure in Reach 6 showed that fish experienced notably different conditions in the five years (Figure 50). The coolest overall distribution was in 2011, the warmest – by a considerable margin – was in 2014. The widest distribution was in 2015 and the narrowest was in 2013.

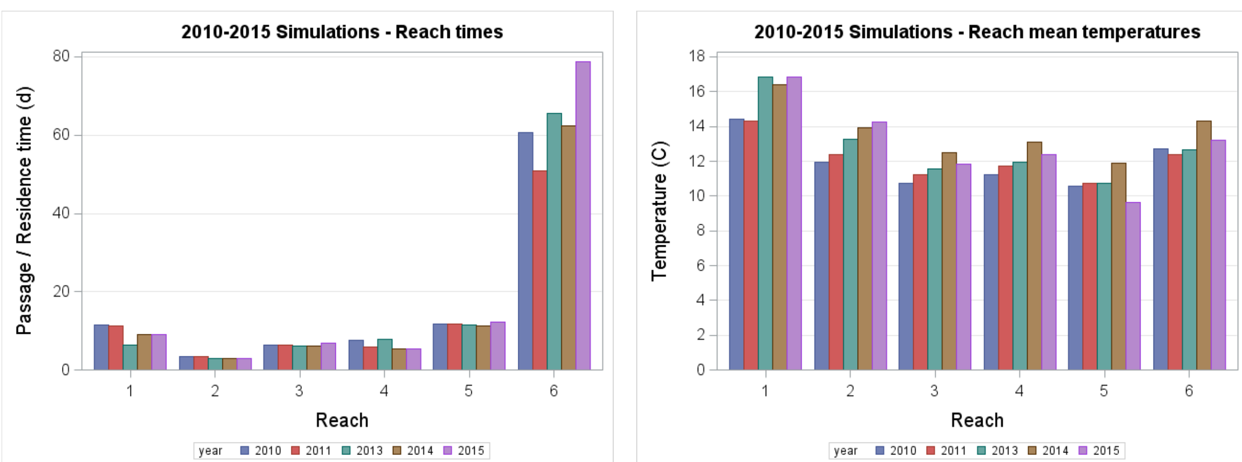


Figure 49. Mean modeled Chinook salmon passage times (d, left) and mean temperature exposure (°C, right) through six Willamette and North Santiam River reaches in 2010–2011 and 2013–2015.

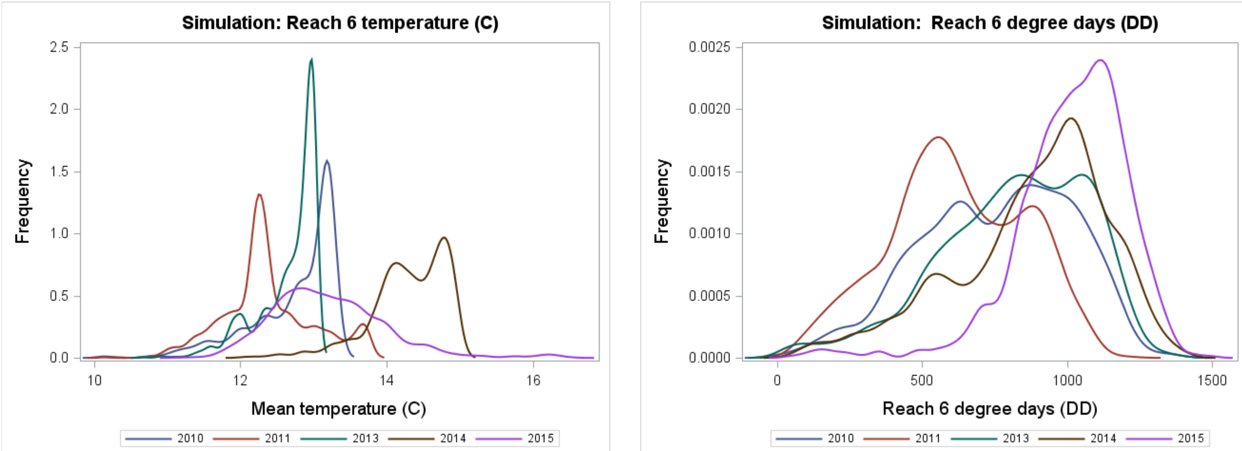


Figure 50. Annual distributions of mean water temperature exposure (left) and degree day accumulation (right) for adult Chinook salmon during the time from in Reach 6 (i.e., near the Minto Fish Collection Facility) through 31 August in the simulations for 2010–2011 and 2013–2015. Distributions were smoothed using a kernel density estimator of 0.5.

The combined differences in Reach 6 residence times and mean individual exposures also produced varied distributions of cumulative exposure (e.g., degree days [DD]; Figure 50). In all years, some fish had Reach 6 DD that were near zero, indicating that they arrived at Minto in late August. However, some fish in all years also accumulated > 1,000 DD. Mean Reach 6 DD accumulations were: 756 DD (2010), 619 DD (2011), 817 DD (2013), 877 DD (2014), and 1,018 DD (2015).

Full migration histories – Full migration times from Willamette Falls to 1 September reflected the among-year differences in run timing. Mean times range from 90 d in 2011 to 115 d in 2015. Across model years, estimates of individual mean salmon exposure ranged from near 10 °C to > 18 °C (Figure 51). The distributions of means varied from essentially unimodal in 2015, when fish spent extended periods in the relatively stable temperatures near Minto, to clearly bimodal in 2011 and 2013, when run timing at Willamette Falls was also bimodal (Appendix Figure B3).

Degree day accumulations over the full migrations ranged from about 500 DD to nearly 2,000 DD (Figure 51). Annual means were 1,256 DD (2010), 1,095 DD (2011), 1,298 DD (2013), 1,366 DD (2014), and 1,482 DD (2015).

The 1,000 thermal histories in each year, when plotted together, clearly demonstrate the tradeoffs in thermal exposure for early-run versus late-run migrants (Figures 52–56). In all years, some of the latest migrants encountered temperatures in the 20–22 °C range in the Willamette River main stem. A number of fish experienced days when mean temperatures were 22–25 °C in 2013, 2014, and especially 2015. In contrast, early-run fish experienced general cool conditions in the Willamette main stem but had very high DD accumulations. The earliest fish in 2010 and 2011 had total migration accumulations of ~1,300–1,600 DD, whereas DD totals of ~1,600–1,900 were common in early-run fish in 2014 and 2015 (Figures 52–56).

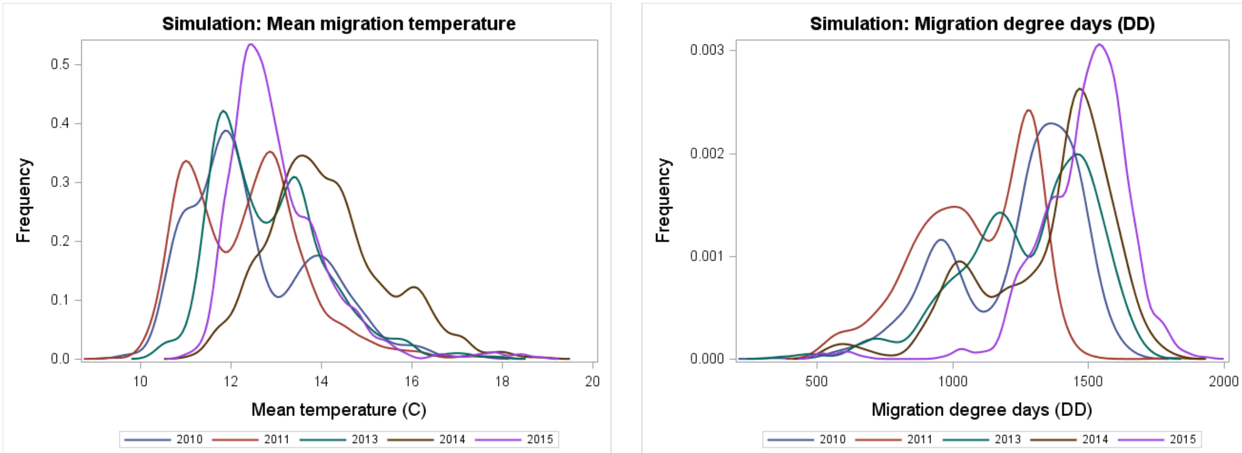


Figure 51. Annual distributions of mean water temperature exposure (left) and total degree day accumulation (right) for adult Chinook salmon during their full migration from Willamette Falls to the Minto Fish Collection Facility in the simulations for 2010–2011 and 2013–2015. Distributions were smoothed using a kernel density estimator of 0.5.

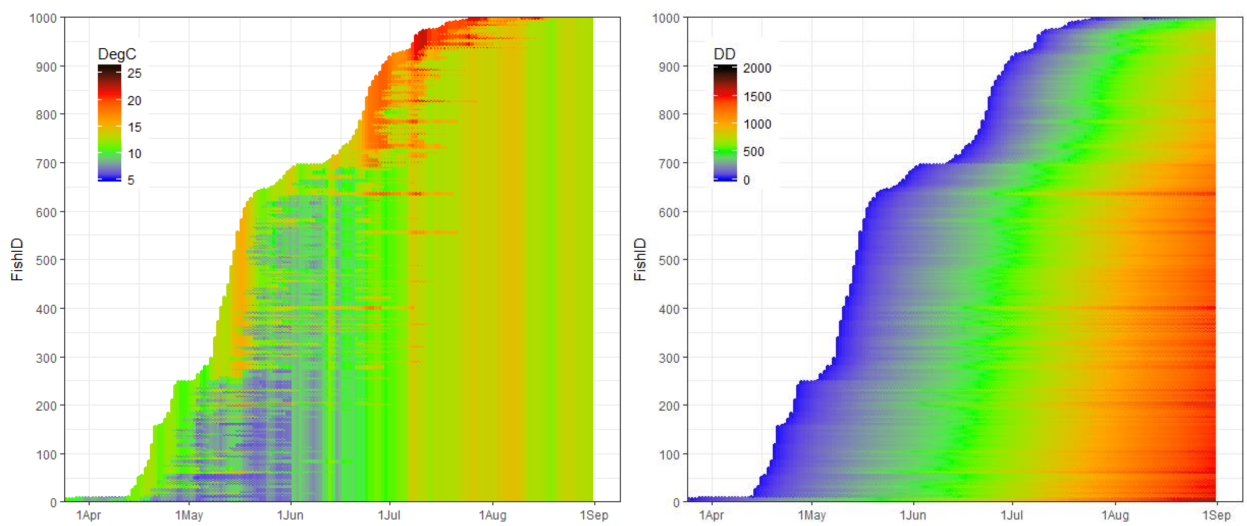


Figure 52. Simulated mean daily water temperature (°C, left) and cumulative degree day accumulation (DD, right) for 1,000 adult Chinook salmon randomly sampled from the run timing distribution at Willamette Falls in 2010. Each horizontal line represents one salmon history. Salmon progressed upstream to the Minto Fish Collection Facility using the model movement parameters and individuals were matched each day to the 2010 river temperature at the nearest USGS gage site. All histories ended on 31 August.

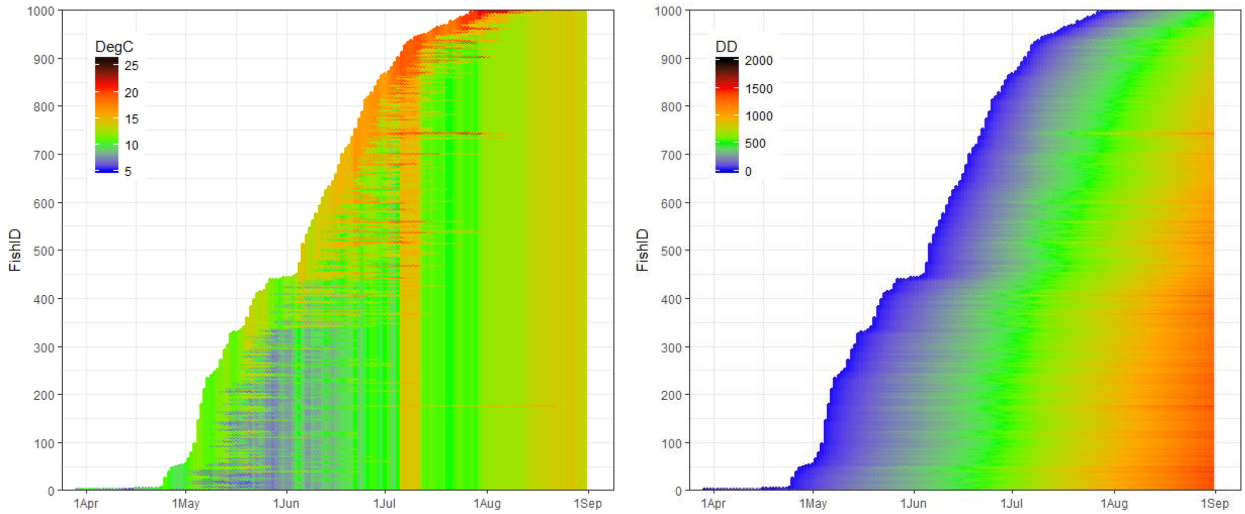


Figure 53. Simulated mean daily water temperature (°C, left) and cumulative degree day accumulation (DD, right) for 1,000 adult Chinook salmon randomly sampled from the run timing distribution at Willamette Falls in 2011. Each horizontal line represents one salmon history. Salmon progressed upstream to the Minto Fish Collection Facility using the model movement parameters and individuals were matched each day to the 2011 river temperature at the nearest USGS gage site. All histories ended on 31 August.

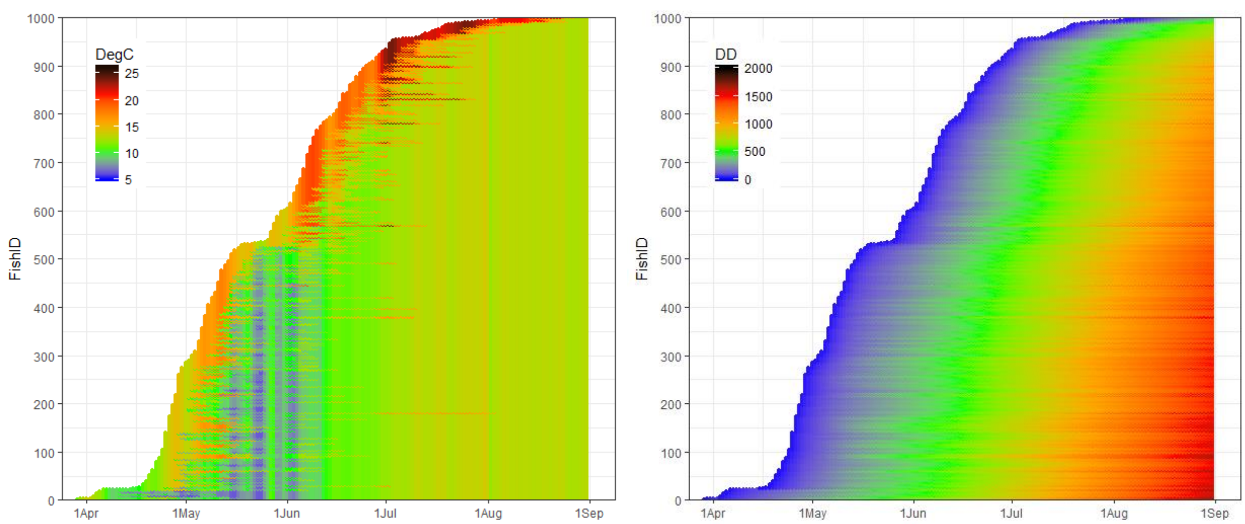


Figure 54. Simulated mean daily water temperature (°C, left) and cumulative degree day accumulation (DD, right) for 1,000 adult Chinook salmon randomly sampled from the run timing distribution at Willamette Falls in 2013. Each horizontal line represents one salmon history. Salmon progressed upstream to the Minto Fish Collection Facility using the model movement parameters and individuals were matched each day to the 2013 river temperature at the nearest USGS gage site. All histories ended on 31 August.

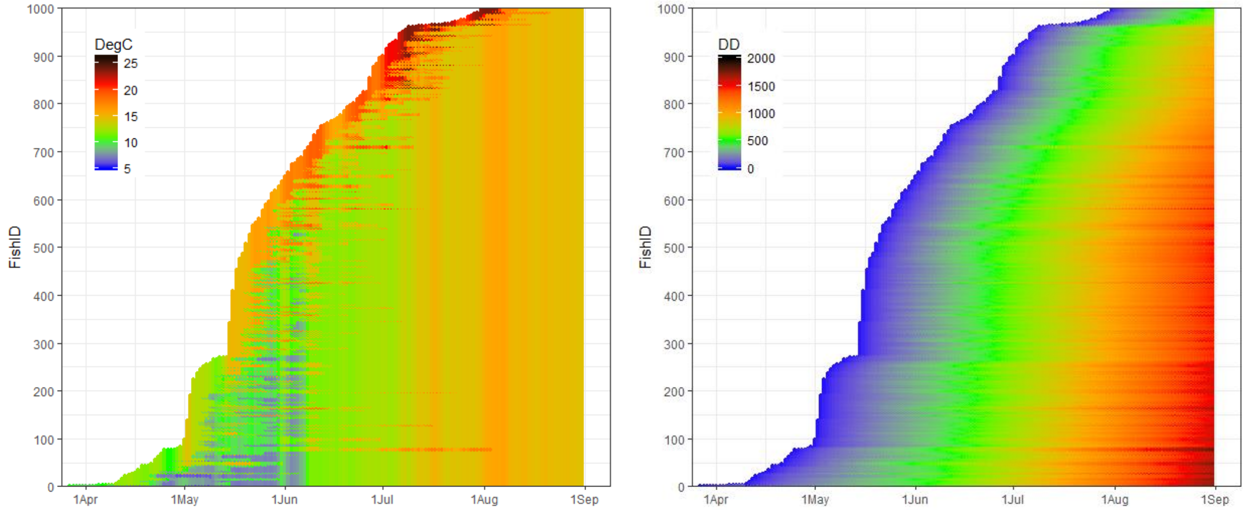


Figure 55. Simulated mean daily water temperature (°C, left) and cumulative degree day accumulation (DD, right) for 1,000 adult Chinook salmon randomly sampled from the run timing distribution at Willamette Falls in 2014. Each horizontal line represents one salmon history. Salmon progressed upstream to the Minto Fish Collection Facility using the model movement parameters and individuals were matched each day to the 2014 river temperature at the nearest USGS gage site. All histories ended on 31 August.

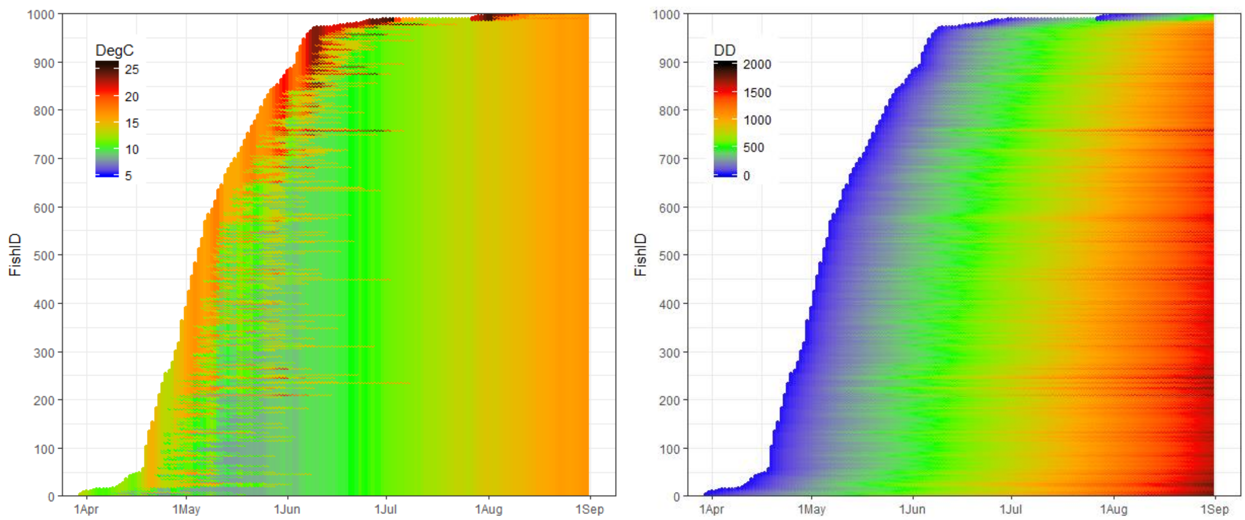


Figure 56. Simulated mean daily water temperature (°C, left) and cumulative degree day accumulation (DD, right) for 1,000 adult Chinook salmon randomly sampled from the run timing distribution at Willamette Falls in 2015. Each horizontal line represents one salmon history. Salmon progressed upstream to the Minto Fish Collection Facility using the model movement parameters and individuals were matched each day to the 2015 river temperature at the nearest USGS gage site. All histories ended on 31 August.

In the five model years, just two (0.04%) simulated salmon – both in 2015 – exceeded the acute and cumulative thresholds (Figures 57–61). The acute threshold was exceeded by 3.2% (2010), 0.7% (2011), 10.1% (2013), 11.2% (2014), and 11.4% (2015) of the runs. There was more among-year variation in the percent that exceeded the cumulative threshold, with <1% (2010 and 2011), 4.1% (2013), 10.5% (2014), and 22.5% (2015). In this hypothetical scenario, the percent of the runs that exceeded one or both exposure thresholds were: 4.1% (2010), 0.8% (2011), 14.2% (2013), 21.7% (2014), and 33.7% (2015).

***Uncertainties and recommendations* –** It is likely that there are thermal exposure thresholds for adult Chinook salmon that, when reached, result in substantial mortality or fitness costs. However, identifying these values has remained elusive for several reasons, including expected differences in thermal tolerance within and among conspecific populations (e.g., Crozier et al. 2008; Farrell et al. 2008; Eliason et al. 2011; Strange 2012; Keefer et al. 2018b) and the logistical challenges of linking exposure along complete migration routes with individual survival and fitness outcomes (e.g., Minke-Martin et al. 2017; Keefer et al. 2018b). Previous efforts to estimate total exposure of Willamette River basin Chinook salmon using individual temperature-logging tags (e.g., Keefer et al. 2015; Naughton et al. 2018b) have shown that total accumulation varies widely among individuals, largely as a function of migration timing but also depending on prespawn holding location. An important limitation in these studies was that acute and cumulative exposure could only be calculated for fish that were recovered (i.e., the survivors), making it impossible to identify mortality thresholds.

The current thermal exposure model should be useful for identifying how many North Santiam Chinook salmon might encounter potential acute or cumulative exposure thresholds (Figures 57–61). The hypothetical thresholds in these figures were a mean exposure of 21 °C in the Willamette River reach for acute exposure and 1,600 DD for cumulative exposure, but other thresholds could be easily quantified. The 21 °C acute threshold was likely conservative, because some fish encountered very warm temperatures in Reach 1 but did not necessarily have a mean temperature >21 °C. The cumulative threshold may also be conservative, as premature mortality in other adult salmon populations has been observed at much lower DD accumulations (e.g., Wagner et al. 2005).

We recommend that future studies directly address the mortality threshold questions for North Santiam River Chinook salmon, or for other Willamette River populations given their shared genetic history (e.g., Johnson and Friesen 2014). Laboratory or hatchery raceway experiments could manipulate holding duration, seasonally vary water temperatures, and perhaps include risk covariates like pathogen infections (e.g., Benda et al. 2015), injuries (e.g., Keefer et al. 2017), or run-timing cohort. Such experiments would provide important information about threshold values and mortality mechanisms.

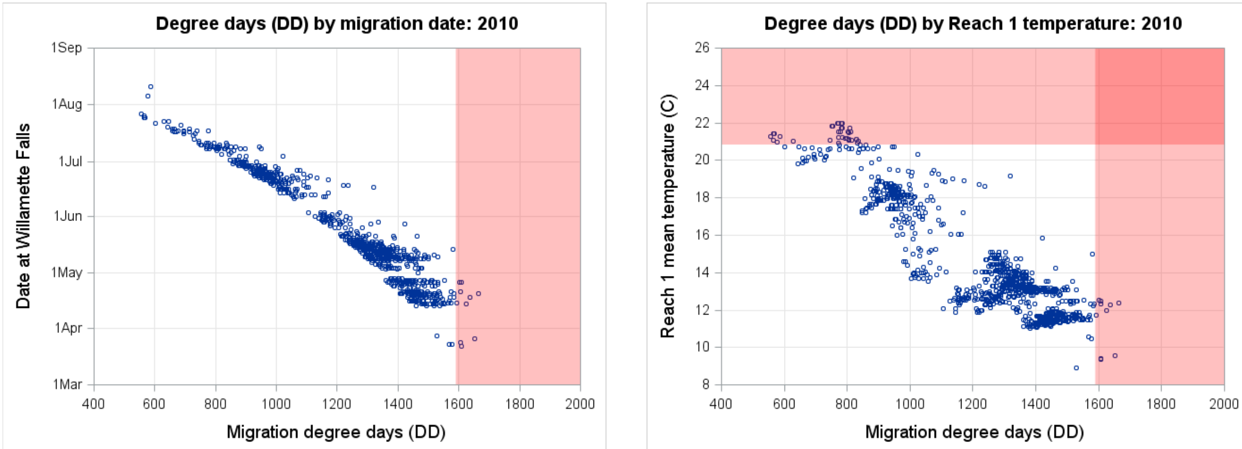


Figure 57. Scatterplots showing the relationships between total accumulated degree days and migration start date (left) and mean Chinook salmon temperature exposure in the Willamette Falls–Santiam River reach (right) in the 2010 simulation. Red shaded areas represent hypothetical acute stress (21 °C) and cumulative exposure (1,600 DD) thresholds for adult Chinook salmon.

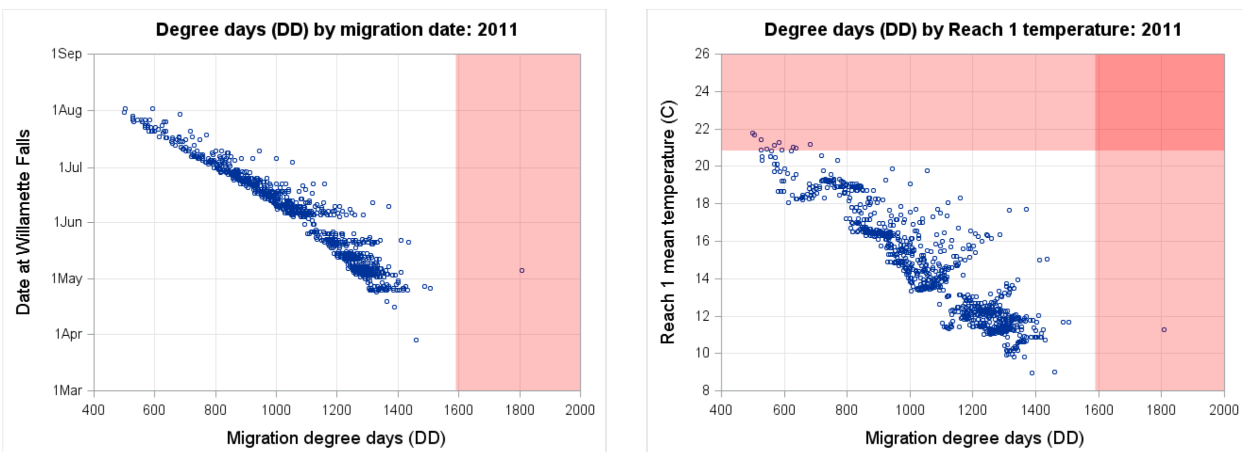


Figure 58. Scatterplots showing the relationships between total accumulated degree days and migration start date (left) and mean Chinook salmon temperature exposure in the Willamette Falls–Santiam River reach (right) in the 2011 simulation. Red shaded areas represent hypothetical acute stress (21 °C) and cumulative exposure (1,600 DD) thresholds for adult Chinook salmon.

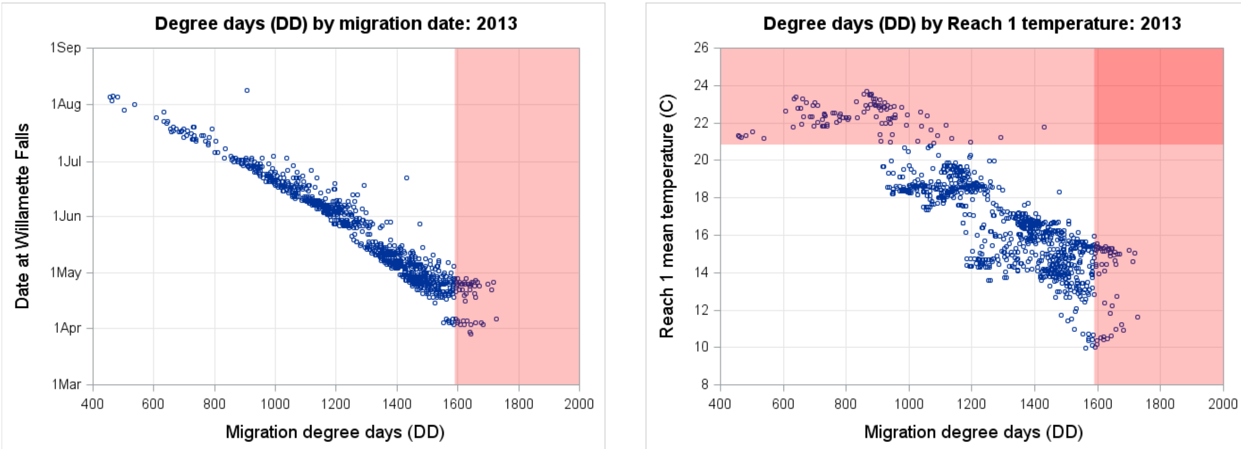


Figure 59. Scatterplots showing the relationships between total accumulated degree days and migration start date (left) and mean Chinook salmon temperature exposure in the Willamette Falls–Santiam River reach (right) in the 2013 simulation. Red shaded areas represent hypothetical acute stress (21 °C) and cumulative exposure (1,600 DD) thresholds for adult Chinook salmon.

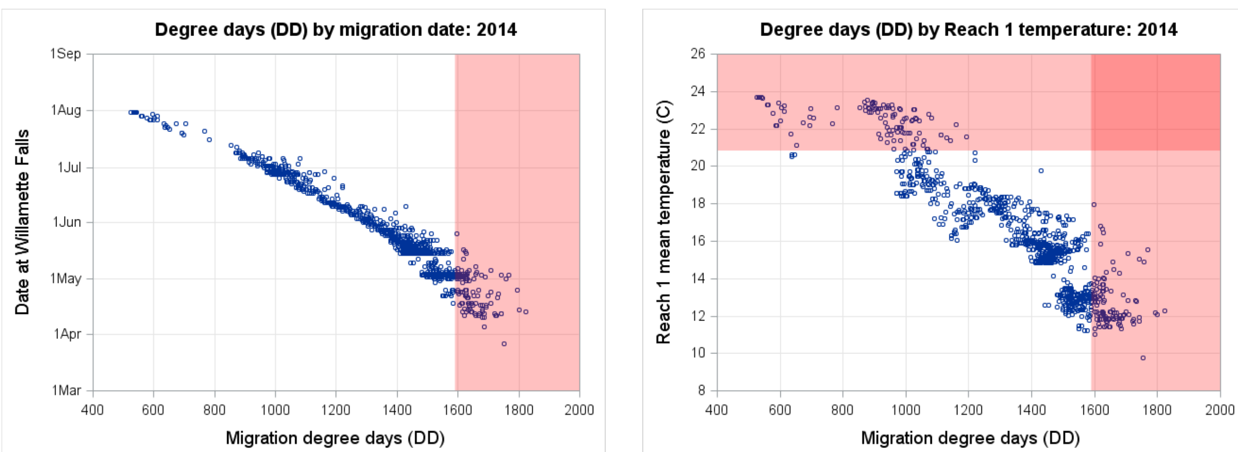


Figure 60. Scatterplots showing the relationships between total accumulated degree days and migration start date (left) and mean Chinook salmon temperature exposure in the Willamette Falls–Santiam River reach (right) in the 2014 simulation. Red shaded areas represent hypothetical acute stress (21 °C) and cumulative exposure (1,600 DD) thresholds for adult Chinook salmon.

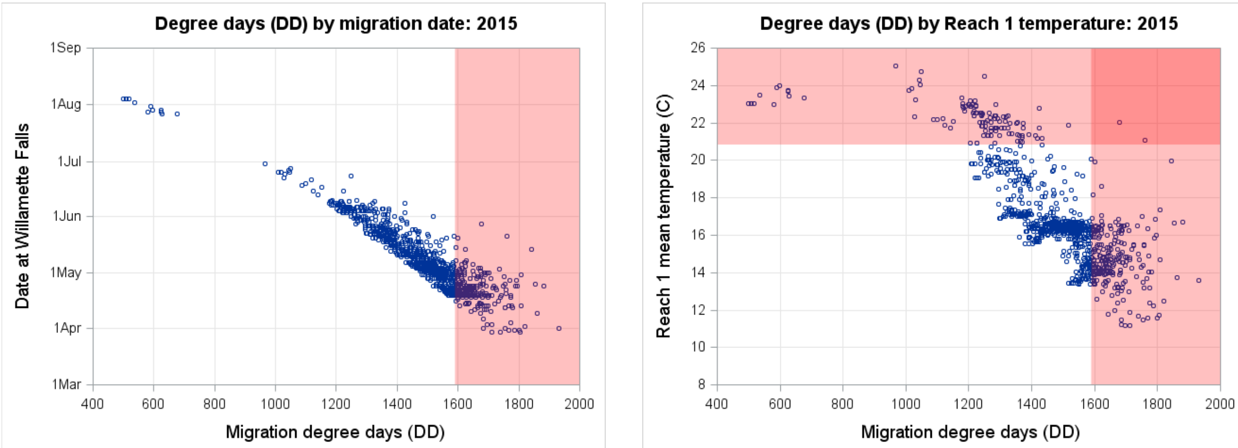


Figure 61. Scatterplots showing the relationships between total accumulated degree days and migration start date (left) and mean Chinook salmon temperature exposure in the Willamette Falls–Santiam River reach (right) in the 2015 simulation. Red shaded areas represent hypothetical acute stress (21 °C) and cumulative exposure (1,600 DD) thresholds for adult Chinook salmon.

10. Simulation model and the Detroit temperature management scenarios

Scenario recap **–** In 2011, the example year with relatively cool river temperatures and late adult Chinook salmon run timing, the four temperature management scenarios produced temperatures in the North Santiam River that were either similar to or warmer than the observed 2011 temperatures (Figure 62). In the scenario with the ODFW target temperature and a hypothetical tower, mean monthly temperatures were warmer by a little more than 1 °C from May–July, the period when most upstream migration occurred. In the scenario with the pre-dam target and tower=yes, temperatures were warmer from May–August, but especially in July (~3.5 °C) and August (~2.5 °C).

In 2015, the example year with very warm river conditions, all four management scenarios produced warmer-than-observed temperatures in April–July (Figure 62). The no-tower scenarios also resulted in warmer conditions in September and October, whereas the yes-tower produced cooler September conditions (both targets) and cooler August conditions (ODFW target).

2011, pre-dam target **–** Results from the pre-dam, no-tower model produced Chinook salmon mean daily and cumulative thermal histories that closely approximated those created using the observed data (Figure 63). The pre-dam target, yes-tower model resulted in warmer daily temperatures for most salmon (compare top and bottom left panels in Figure 63) and higher cumulative exposure, particularly among early-run fish that were exposed to warmer conditions for most of their migration and prespawn holding periods (compare top and bottom right panels in Figure 63).

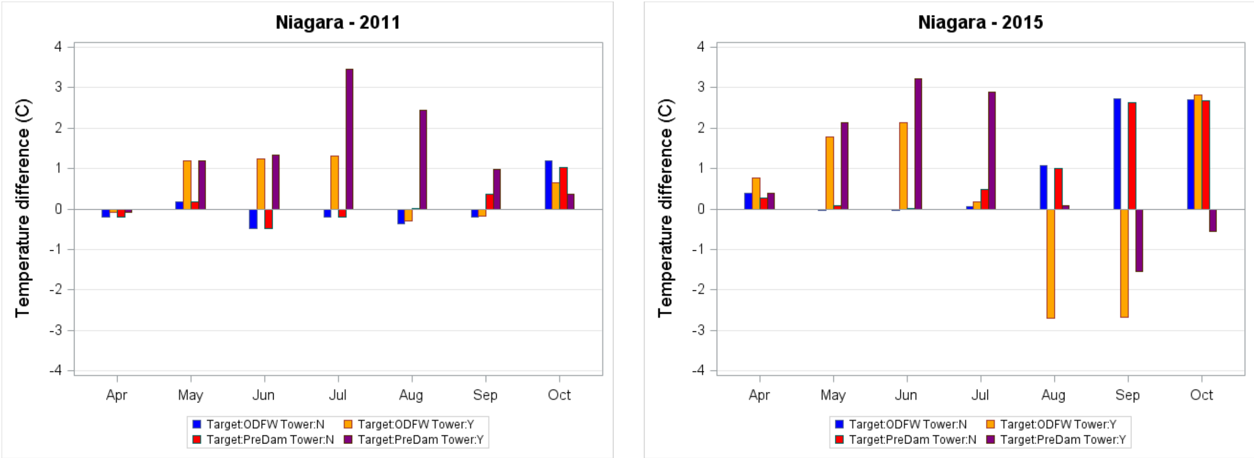


Figure 62. Mean monthly differences (°C) between observed water temperatures and the four simulated temperature scenarios at Niagara in 2011 and 2015. See Figure 13 for Mehama and Greens Bridge summaries.

2011, ODFW target – The ODFW target, no-tower model produced salmon exposure histories that were very slightly cooler than the histories with the observed temperature data (Figure 64). In contrast, the ODFW target, yes-tower model produced warmer histories through the middle of summer, and very slightly cooler conditions in August. These two effects partially offset, resulting in cumulative histories that were similar to those with the observed data (compare top and bottom right panels in Figure 64).

2011, reach-specific exposure metrics – We selected two outputs from the exposure model to demonstrate how salmon exposure differed among the temperature management scenarios: mean fish temperatures (Figure 65) and mean DD accumulations (Figure 66) in the four North Santiam River reaches. The exposure model results showed that mean fish temperatures were most variable in Reach 3 (North Santiam mouth to Bennett), and became progressively less variable through reaches 4 (near Bennett) and 5 (Bennett to first Minto). Among-fish variability was very low in Reach 6 (holding near Minto) in all scenarios in 2011. A second general pattern with the mean exposure metric was that mean exposures were higher in the yes-tower scenarios than in the no-tower scenarios. Mean exposure was highest in the pre-dam target, yes-tower scenario in reaches 5 and 6 (Figure 66).

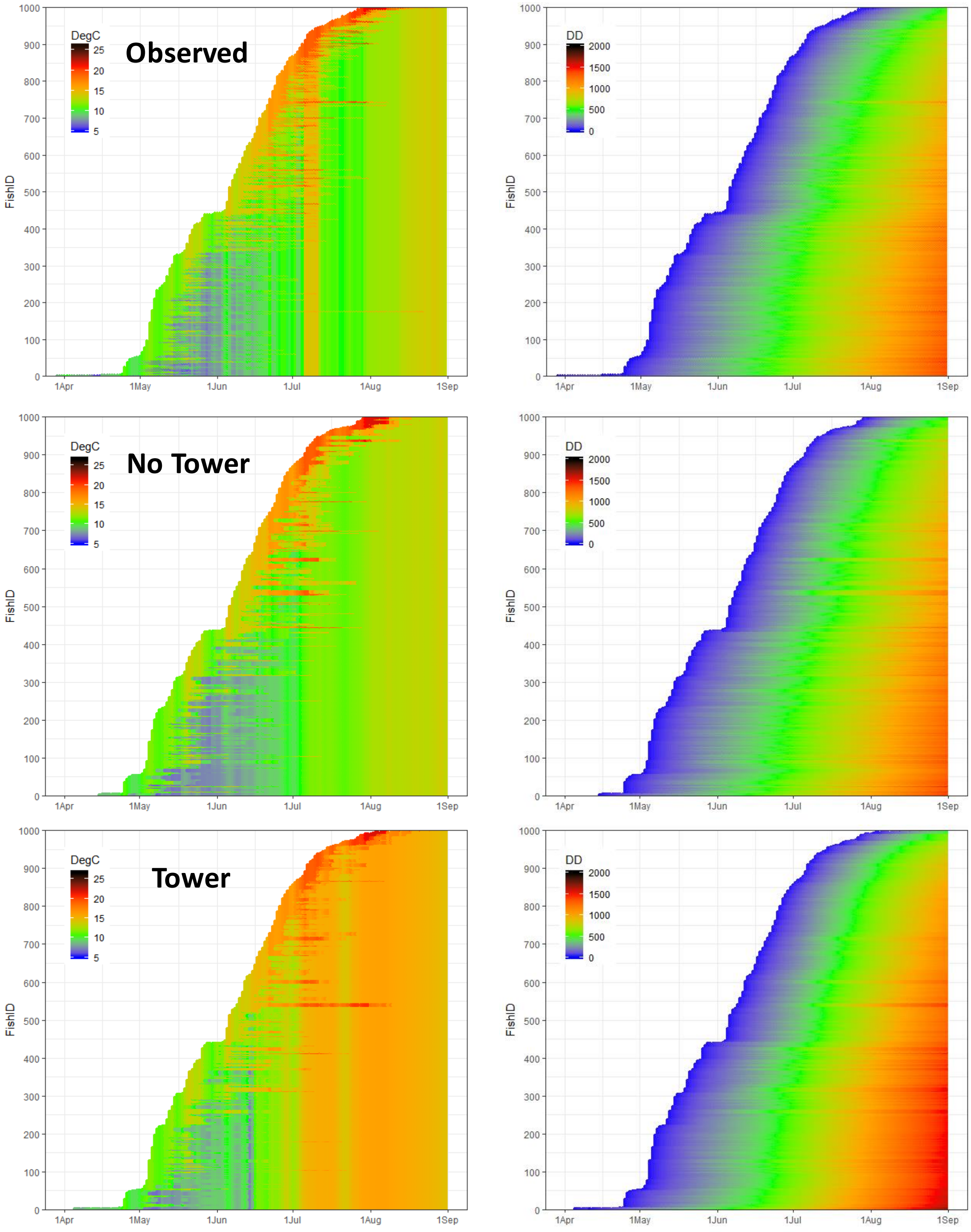


Figure 63. Temperature management scenario testing. Simulated results for **2011** using the **PreDam temperature targets**. Top panels show results with data from the 'Tower = No' scenario. Bottom panels show results with the 'Tower = Yes' scenario.

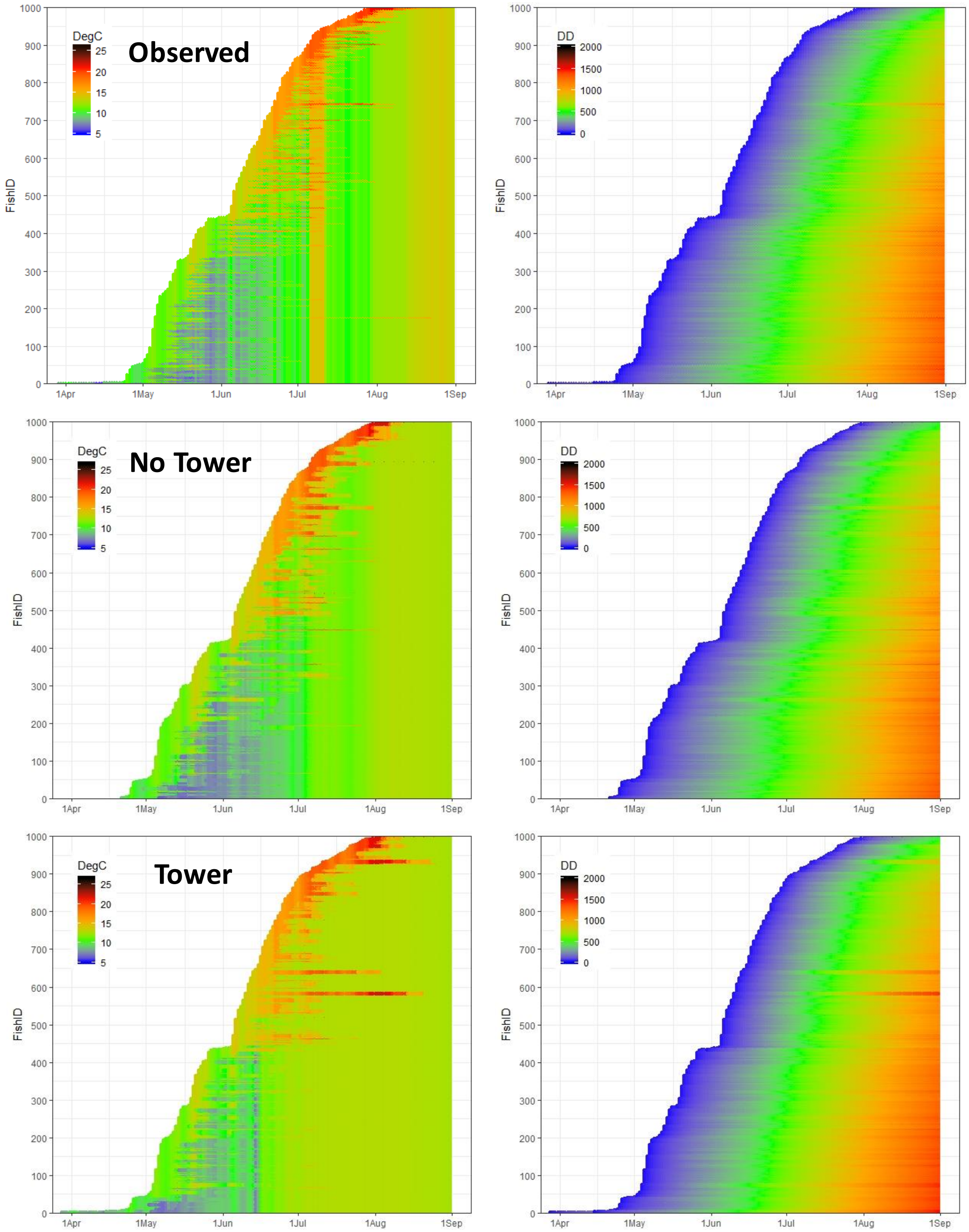


Figure 64. Temperature management scenario testing. Simulated results for **2011** using the **ODFW temperature targets**. Top panels show results with data from the 'Tower = No' scenario. Bottom panels show results with the 'Tower = Yes' scenario.

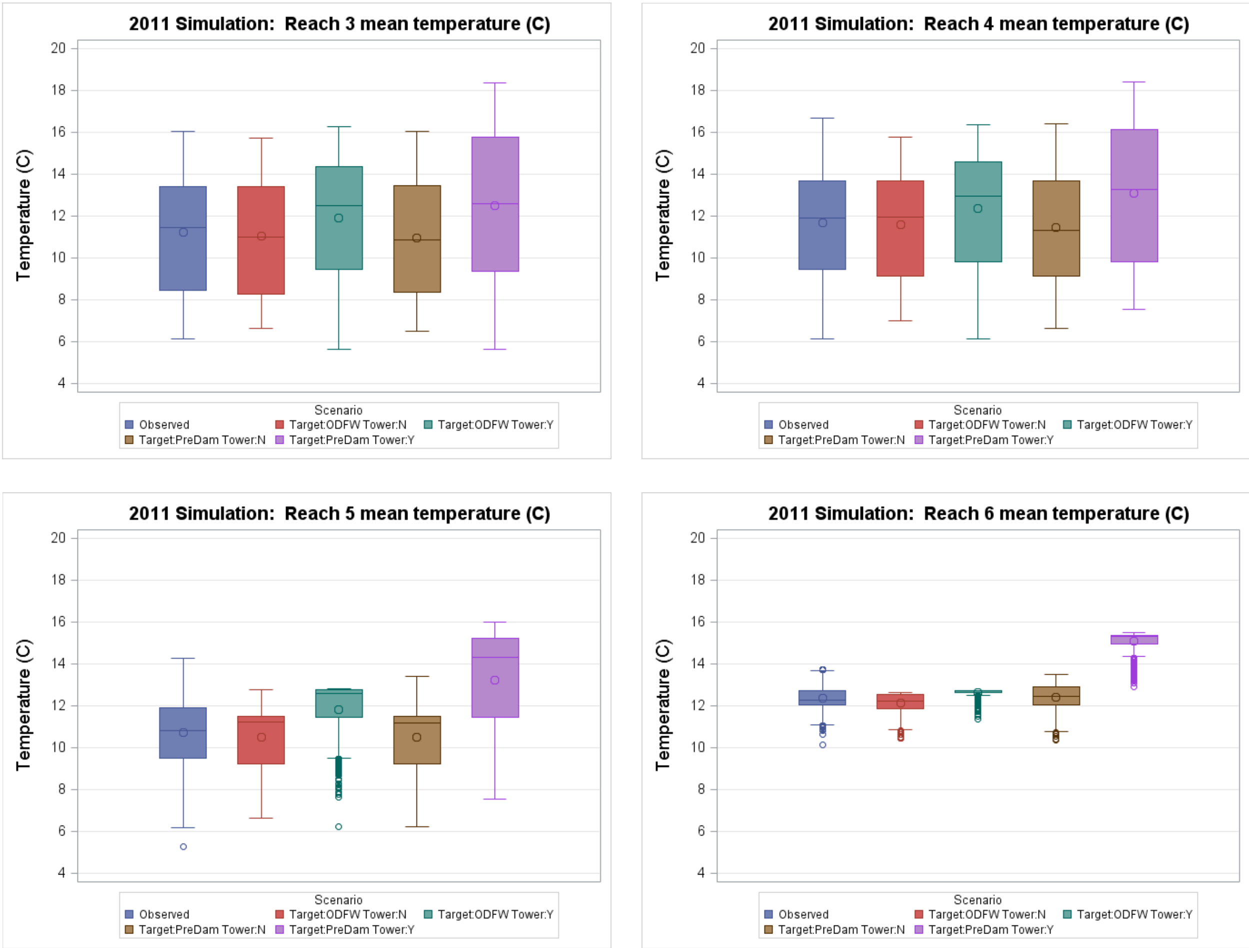


Figure 65. Comparison of simulated mean temperature exposure for individual Chinook salmon in four North Santiam River reaches: mouth–First Bennett (Reach 3), first Bennett–last Bennett (Reach 4), last Bennett–first Minto (Reach 5), and first Minto–last Minto (Reach 6) in **2011**. Box plots show the **observed** temperatures, and the four temperature scenarios: **PreDam temperature targets** (‘**Tower=No**’ and ‘**Tower=Yes**’) and **ODFW temperature targets** ‘**Tower=No’ and ‘Tower=Yes**’.

Among-scenario differences in DD accumulation were very slight in reaches 3–5 because most salmon spent just a few days in each reach (Figure 66). DD distributions were right-skewed as some fish had longer residence times, irrespective of scenario. In Reach 6, differences in total DD accumulation were more evident. The yes-tower scenarios had higher totals than the no-tower scenarios. The highest accumulation, on average, was in the pre-dam target, yes-tower scenario (Figure 66).

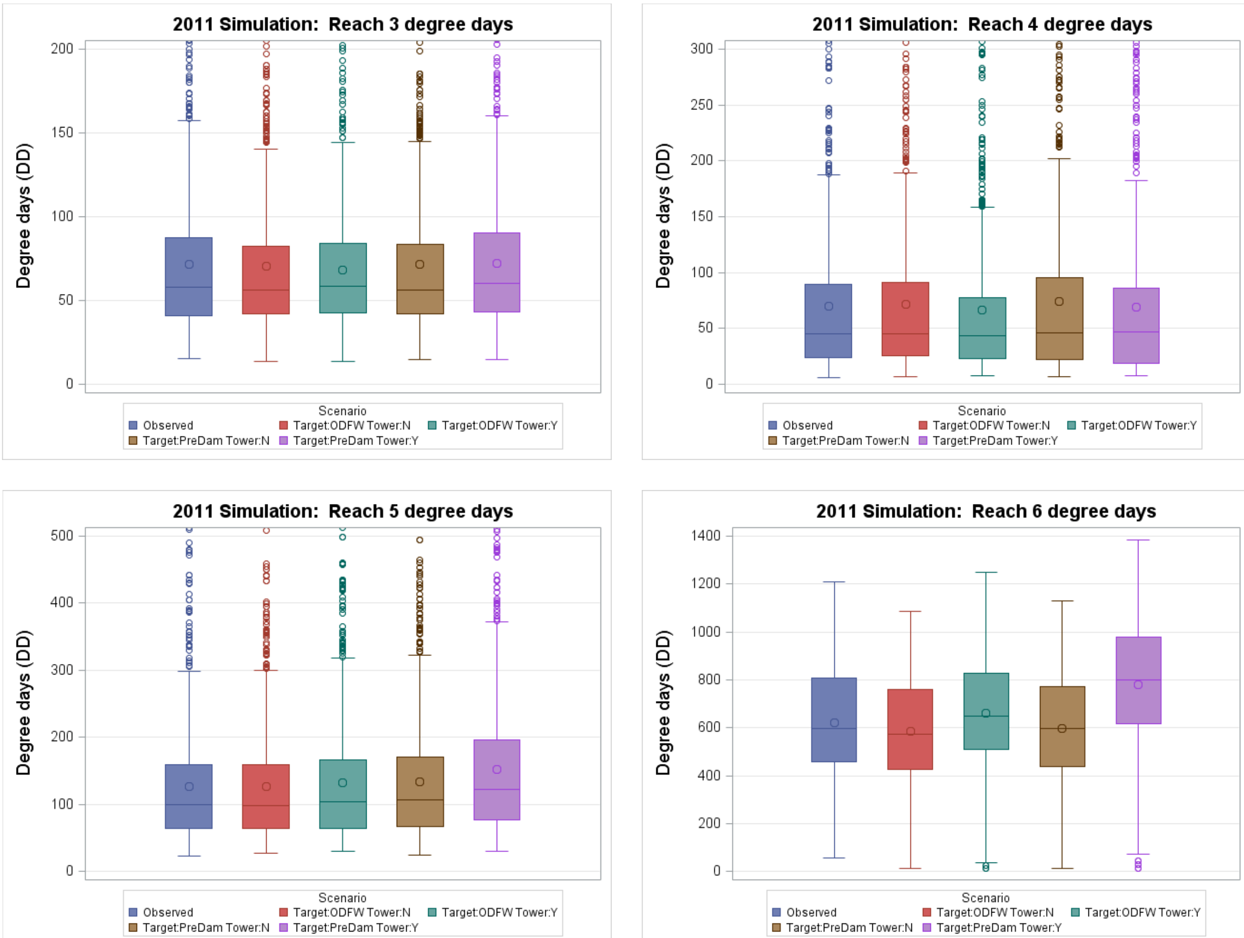


Figure 66. Comparison of simulated degree day accumulation for individual Chinook salmon in four North Santiam River reaches: mouth–First Bennett (Reach 3), first Bennett–last Bennett (Reach 4), last Bennett–first Minto (Reach 5), and first Minto–last Minto (Reach 6) in **2011**. Box plots show the **observed** temperatures, and the four temperature scenarios: **PreDam temperature targets** (‘**Tower=No**’ and ‘**Tower=Yes**’) and **ODFW temperature targets** and ‘**Tower=No’ and ‘Tower=Yes**’. Some outliers not shown to facilitate comparisons.

2015, pre-dam target **–** Modeled salmon exposure results in 2015 differed more among scenarios than in 2011. In 2015, the pre-dam target, no-tower model, Chinook salmon were warmer than when using the observed data, especially in August. In the pre-dam target, yes-tower model, salmon temperatures were warmer in June, July and early August, with modestly cooler conditions in late August (compared three left panels in Figure 67). Cumulative exposure was highest in the yes-tower scenario, reflecting the effects of the prolonged period of warmer water in summer. Early- and mid-run salmon had very high DD totals.

2015, ODFW target**–** The ODFW target, no-tower model produced warmer salmon exposure histories than the model using observed temperatures, primarily in August (Figure 68). The ODFW target, yes-tower model produced histories that were slightly warmer in June but substantially cooler in August. These effects partially offset, resulting in cumulative histories that were similar to those with the observed data (compare the three right panels in Figure 68).

2015, reach-specific exposure metrics **–** The distributions of reach-specific mean fish temperatures were more compressed in the 2015 scenarios (Figure 69) than in the 2011 scenarios. Among-fish variability in mean exposure was higher in reaches 3 and 4, with interquartile ranges that were ~2–3 °C, than in reaches 5 and 6 above the Bennett dams where interquartile ranges were ~1–1.5 °C. The yes-tower scenarios produced warmer mean salmon temperatures than the no-tower scenarios, except the ODFW target, yes-tower mean was lower in reach 6 – there was exceptionally low variation in this estimate (Figure 69). Mean exposure was highest in the pre-dam target, yes-tower scenario in reaches 5 and 6.

Among-scenario differences in DD accumulation were very slight in Reach 3 and were low in reaches 4–5 (Figure 70). The yes-tower scenarios produce slightly higher mean DD accumulations than the no-tower scenario in Reach 5. Results were mixed in Reach 6: the highest mean accumulations were in the ODFW target, yes-tower scenario and the lowest DD totals were in the pre-dam target, yes-tower scenario.

Uncertainties and recommendations **–** Results from the thermal exposure model, run with the different temperature scenario inputs, demonstrate the complex interplay among Chinook salmon run timing, reach-specific residence times, and offsetting seasonal temperature changes. Predicting potential biological outcomes (e.g., PSM, spawning success, fitness) from the modeled scenarios is constrained by our lack of understanding about some of the key mechanisms affecting adult salmon behaviors and survival (see previous recommendations). Some key uncertainties include: (1) what are the biological effects of exposure to warmer water during upstream migration versus during prespawn holding?; (2) are there tradeoffs between faster migration under the warmer proposed migration conditions and the resulting longer prespawn holding times?; (3) what is the 'best' operational scenario to maximize survival for all portions of an annual run, and should some run-timing groups be prioritized?; (4) will earlier arrival and/or warmer early-season conditions at Minto affect collection efficiency at the facility?; and (5) did the 2011 and 2015 example years sufficiently capture the diversity of potential temperature management outcomes (see comments in Section 7)?

We conclude by suggesting that models like the exposure model are useful for generating quantitative thermal histories and demonstrating how thermal experiences are likely to differ for adult salmon migrating at different times of the migration season, in different types of water years and under different management scenarios. The structure of the current model is simple, with no triggers for significant behavioral changes (e.g., migration cessation by fish that encounter acutely stressful temperatures) or mortality events (e.g., *en route* mortality due to injury, disease, or energetic exhaustion). We recommend that future model-based evaluations of temperature management options in the North Santiam River incorporate mortality mechanisms or other threshold elements. For example, we used the exposure model histories in a Wisconsin bioenergetics model (e.g., Stewart and Ibarra 1991; Keefer et al. 2019) to produce the energy histories in Figure 71. If PSM has an energetic exhaustion component, this type of model would be effective for predicting the proportion of a run that would die prematurely under various management scenarios. There is currently no similar mortality model for temperature-mediated disease processes for Chinook salmon, despite growing evidence that pathogens are strongly associated with PSM in the Willamette River basin and elsewhere (e.g., Kent et al. 2013; Miller et al. 2014; Benda et al. 2015).

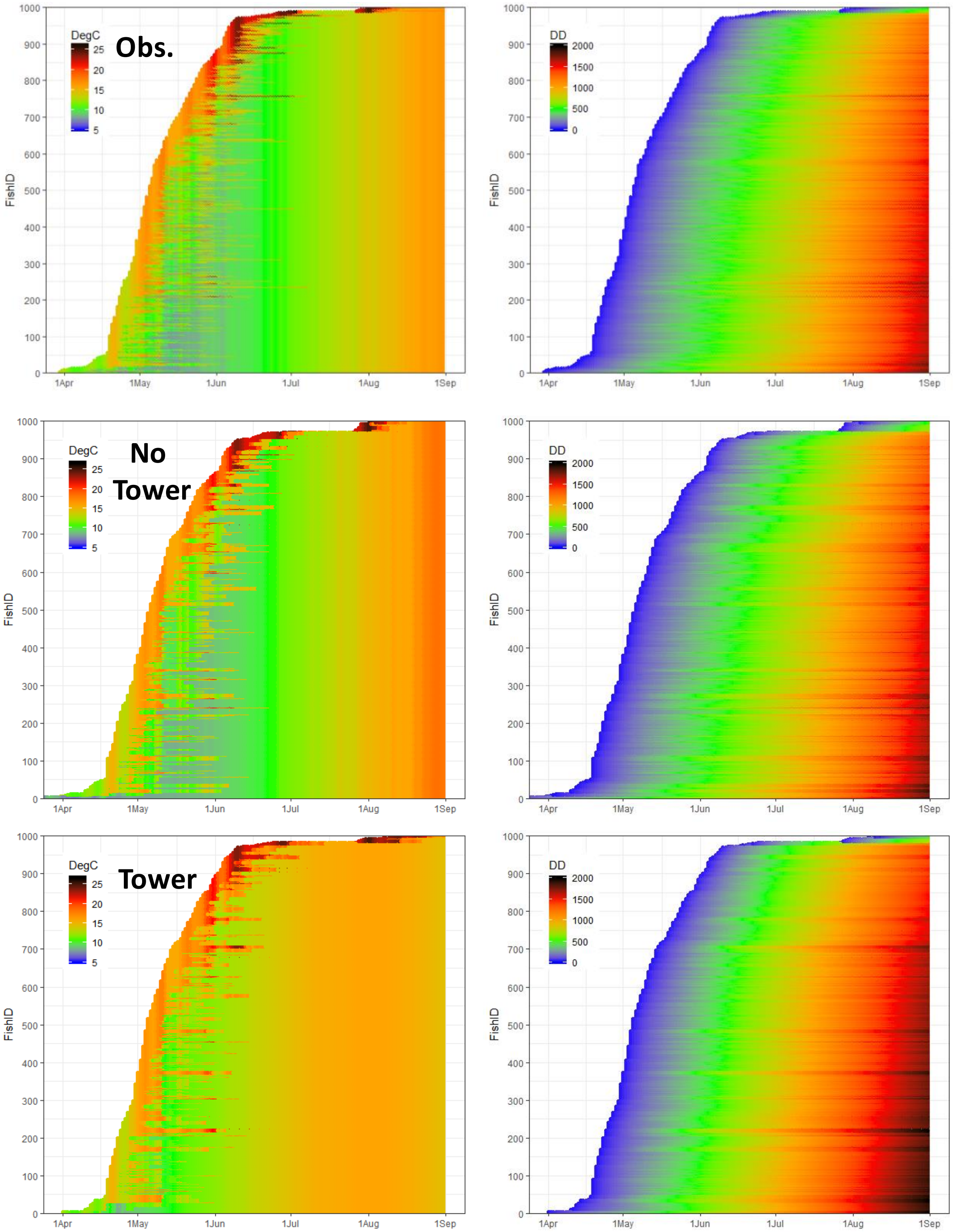


Figure 67. Temperature management scenario testing. Simulated results for **2015** using the **PreDam temperature targets**. Top panels show results with data from the observed temperature data. Middle panels show results with data from the 'Tower = No' scenario. Bottom panels show results with the 'Tower = Yes' scenario.

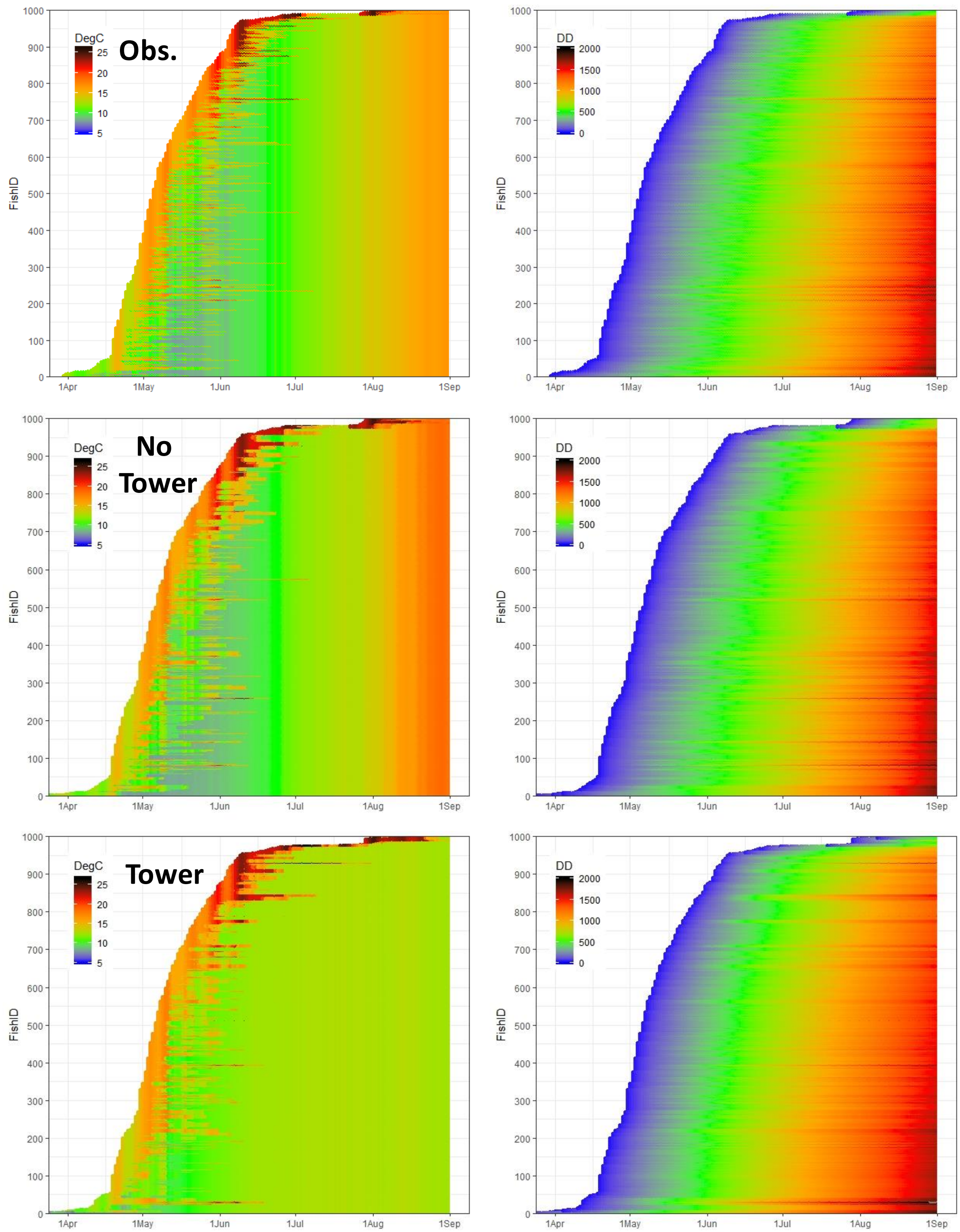


Figure 68. Temperature management scenario testing. Simulated results for **2015** using the **ODFW temperature targets**. Top panels show results with data from the observed temperature data. Middle panels show results with data from the 'Tower = No' scenario. Bottom panels show results with the 'Tower = Yes' scenario.

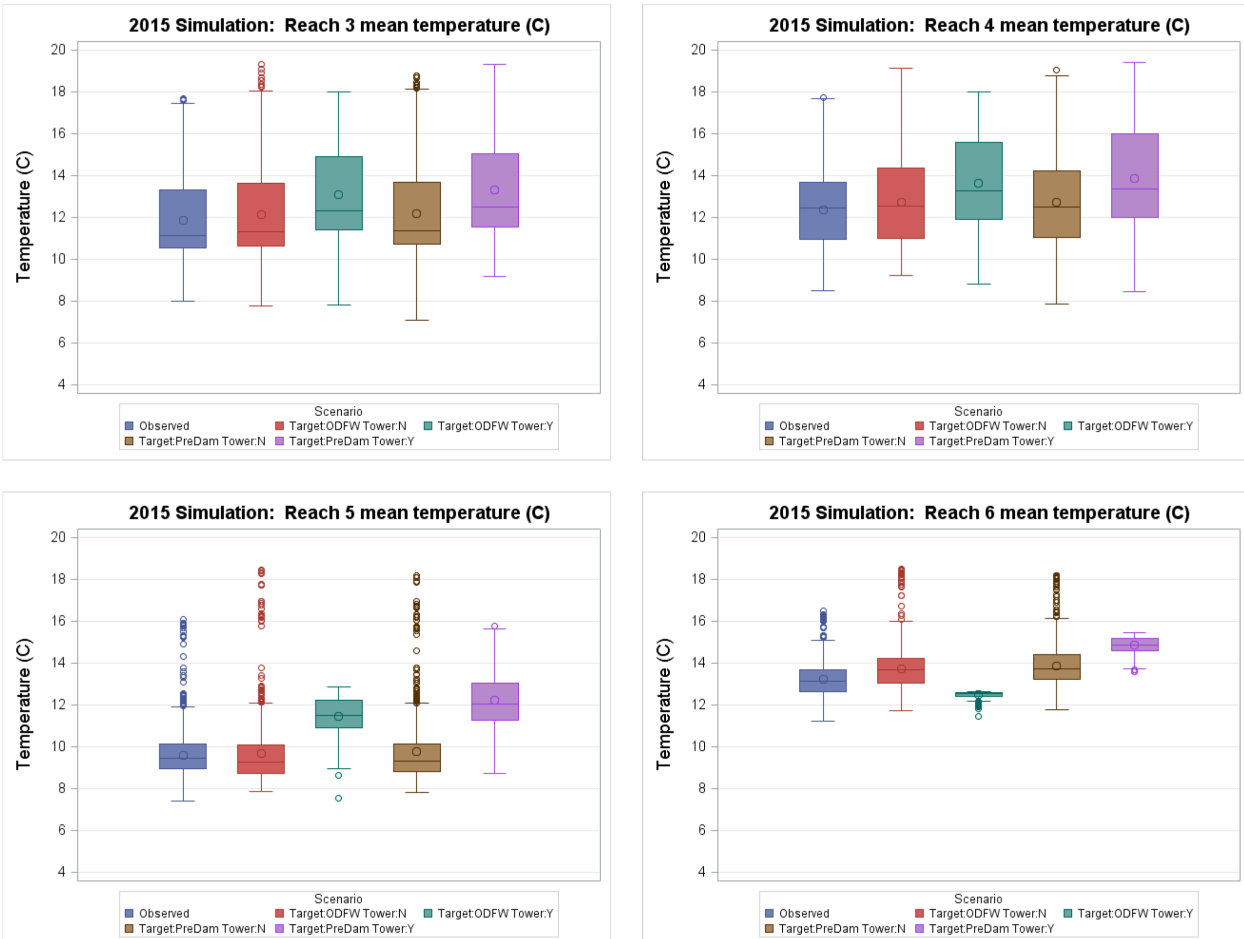


Figure 69. Comparison of simulated mean temperature exposure for individual Chinook salmon in four North Santiam River reaches: mouth–First Bennett (Reach 3), first Bennett–last Bennett (Reach 4), last Bennett–first Minto (Reach 5), and first Minto–last Minto (Reach 6) in **2015**. Box plots show the **observed** temperatures, and the four temperature scenarios: **PreDam temperature targets** (‘**Tower=No**’ and ‘**Tower=Yes**’) and **ODFW temperature targets ‘Tower=No’ and ‘Tower=Yes**’.

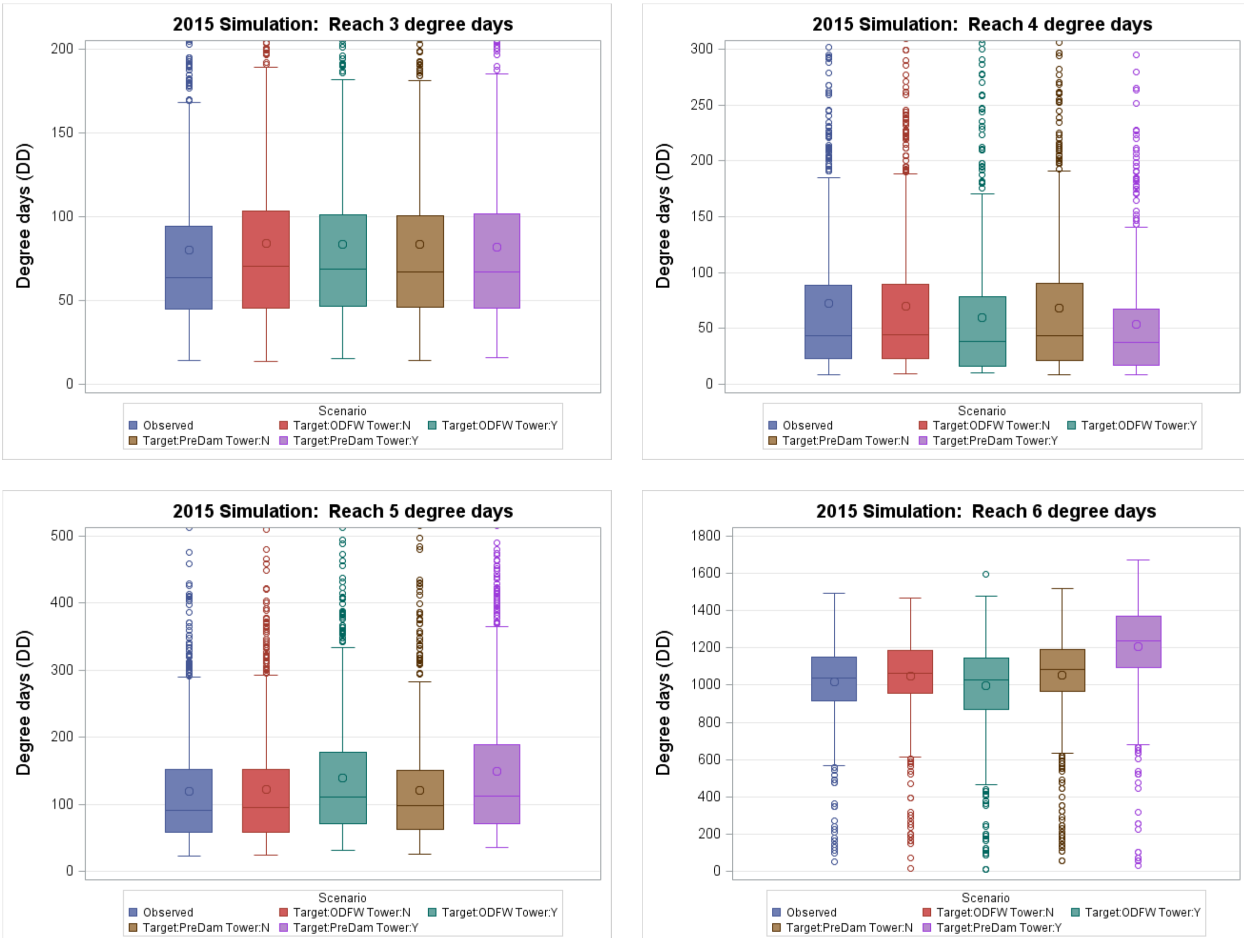


Figure 70. Comparison of simulated degree day accumulation for individual Chinook salmon in four North Santiam River reaches: mouth–First Bennett (Reach 3), first Bennett–last Bennett (Reach 4), last Bennett–first Minto (Reach 5), and first Minto–last Minto (Reach 6) in **2015**. Box plots show the **observed** temperatures, and the four temperature scenarios: **PreDam temperature targets** (‘**Tower=No**’ and ‘**Tower=Yes**’) and **ODFW temperature targets** and ‘**Tower=No’ and ‘Tower=Yes**’. Some outliers not shown to facilitate comparisons.

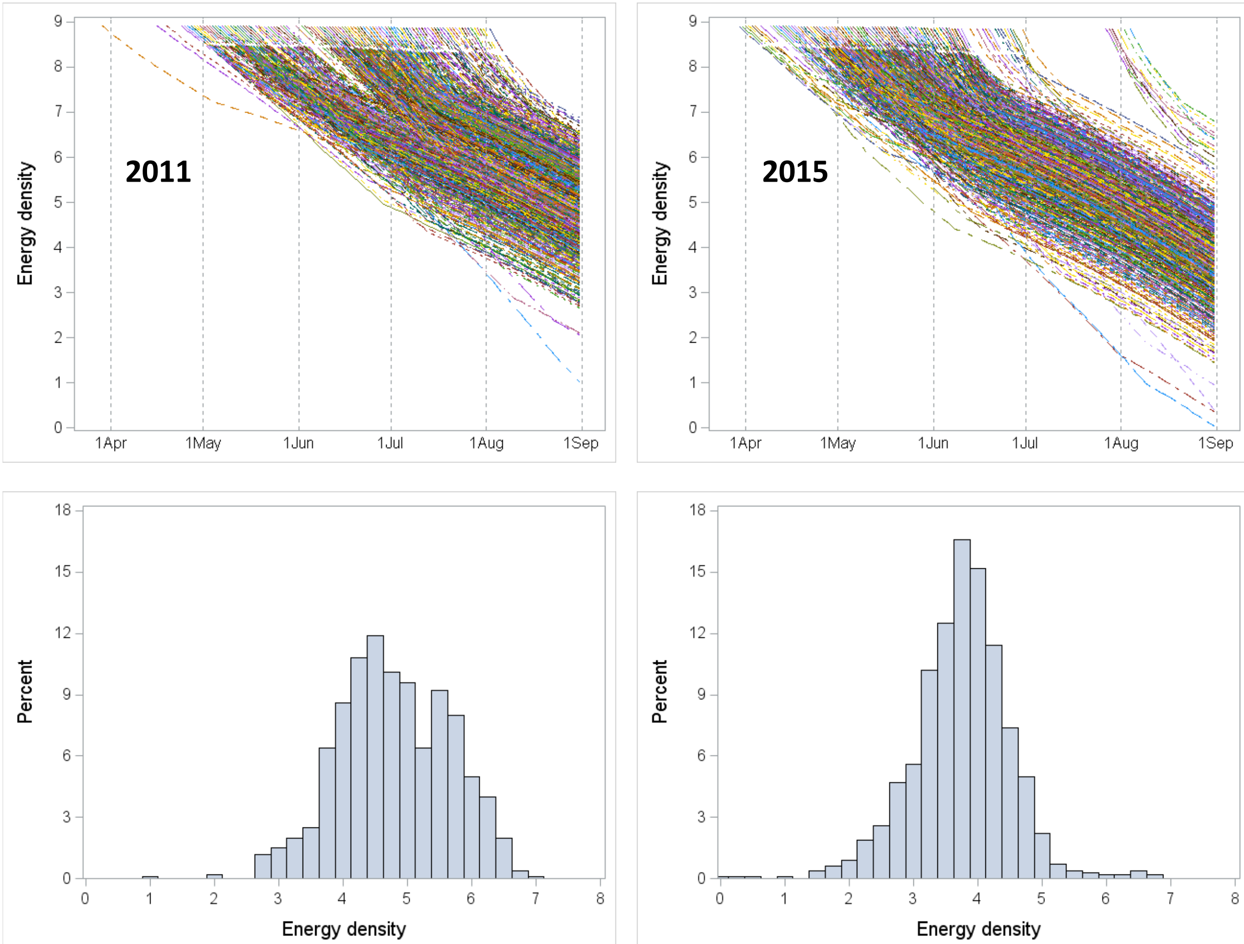


Figure 71. Examples of a bioenergetics application of the thermal exposure model showing individual energy density histories from Willamette Falls to Minto (top) and distributions of the densities on 1 September (bottom). Data inputs were the daily temperature histories from the exposure model (observed temperatures), the bioenergetics parameters for Chinook salmon from Stewart and Ibarra (1991), and swim speeds of 1BL/s during migration and 0.25 BL/s during holding; each salmon had an initial mass of 6.3 kg and density of 9 J/g at Willamette Falls. The figure shows the substantially higher energetic costs of migrating in a warm year and of early migration timing.

References

Angilletta, M. J. J., E. A. Steel, K. K. Bartz, J. G. Kingsolver, M. D. Scheuerell, B. R. Beckman, and L. G. Crozier. 2008. Big dams and salmon evolution: changes in thermal regimes and their potential evolutionary consequences. Evolutionary Applications 1(2):286-299.

Araki, H., B. A. Berejikian, M. J. Ford, and M. S. Blouin. 2008. Fitness of hatchery-reared salmonids in the wild. Evolutionary Applications 1(2):342-355.

Beer, W. N., and E. A. Steel. 2018. Impacts and implications of temperature variability on Chinook Salmon egg development and emergence phenology. Transactions of the American Fisheries Society 147:3-15.

Benda, S. E., G. P. Naughton, C. C. Caudill, M. L. Kent, and C. B. Schreck. 2015. Cool, pathogen-free refuge lowers pathogen-associated prespawn mortality of Willamette River Chinook salmon. Transactions of the American Fisheries Society 144:1159-1172.

Boggs, C. T., M. L. Keefer, C. A. Peery, T. C. Bjornn, and L. C. Stuehrenberg. 2004. Fallback, reascension and adjusted fishway escapement estimates for adult Chinook salmon and steelhead at Columbia and Snake River dams. Transactions of the American Fisheries Society 133:932-949.

Bowerman, T., M. L. Keefer, and C. C. Caudill. 2016. Pacific salmon prespawn mortality: patterns, methods, and study design considerations. Fisheries 41:738-749.

Bowerman, T., A. Roumasset, M. L. Keefer, C. S. Sharpe, and C. C. Caudill. 2018. Prespawn mortality of female Chinook Salmon increases with water temperature and percent hatchery origin. Transactions of the American Fisheries Society 147:31-42.

Bradford, M. J., J. Lovy, D. A. Patterson, D. J. Speare, W. R. Bennett, A. R. Stobbart, and C. P. Tovey. 2010. *Parvicapsula minibicornis* infections in gill and kidney and the premature mortality of adult sockeye salmon (*Oncorhynchus nerka*) from Cultus Lake, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 67(4):673-683.

Buccola, N. L., S. A. Rounds, A. B. Sullivan, and J. C. Risley. 2012. Simulating potential structural and operational changes for Detroit Dam on the North Santiam River, Oregon, for downstream temperature management. US Geological Survey, Scientific Investigations Report 2012-5231.

Buccola, N. L., A. J. Stonewall, and S. A. Rounds. 2015. Simulations of a hypothetical temperature control structure at Detroit Dam on the North Santiam River, Northwestern Oregon. US Geological Survey, Open-File Report 2015-102.

Caudill, C. C., M. A. Jepson, S. R. Lee, T. L. Dick, G. P. Naughton, and M. L. Keefer. 2014. A field test of Eugenol-based anesthesia versus fish restraint in migrating adult Chinook salmon and steelhead. Transactions of the American Fisheries Society 143(4):856-863.

Christie, M. R., M. L. Marine, R. A. French, and M. S. Blouin. 2012. Genetic adaptation to captivity can occur in a single generation. Proceedings of the National Academy of Sciences of the United States of America 109(1):238-242.

Clabough, T. S., M. L. Keefer, T. Blubaugh, G. P. Naughton, C. C. Caudill, and D. Thompson. 2017. Evaluation of adult Chinook salmon behavior at the Foster Dam adult fish facility – 2016. Technical Report 2017-3, University of Idaho to U.S. Army Corps of Engineers, Portland District.

Cook, K. V., S. H. McConnachie, K. M. Gilmour, S. G. Hinch, and S. J. Cooke. 2011. Fitness and behavioral correlates of pre-stress and stress-induced plasma cortisol titers in pink salmon (*Oncorhynchus gorbuscha*) upon arrival at spawning grounds. Hormones and Behavior 60:489-497.

Crozier, L.G., T. Bowerman, B.J. Burke, M.L. Keefer, and C.C. Caudill. 2017. High stakes steeplechase: a behavior-based model to predict individual travel time through diverse migration segments. Ecosphere 8(10): e01965.

Crozier, L. G., R. W. Zabel, and A. F. Hamlet. 2008. Predicting differential effects of climate change at the population level with life-cycle models of spring Chinook salmon. Global Change Biology 14:236-249.

DeWeber, J. T., J. T. Peterson, C. Sharpe, M. L. Kent, M. E. Colvin, and C. B. Schreck. 2017. A hidden-process model for estimating prespawn mortality using carcass survey data. North American Journal of Fisheries Management 37:162-175.

Eliason, E. J., and coauthors. 2011. Differences in thermal tolerances among sockeye salmon populations. Science 332:109-112.

Eliason, E. J., T. D. Clark, S. G. Hinch, and A. P. Farrell. 2013. Cardiorespiratory collapse at high temperature in swimming adult sockeye salmon. Conservation Physiology 1:cot008.

Farrell, A. P., S. G. Hinch, S. J. Cooke, D. A. Patterson, G. T. Crossin, M. Lapointe, and M. T. Mathes. 2008. Pacific salmon in hot water: applying aerobic scope models and biotelemetry to predict the success of spawning migrations. Physiological and Biochemical Zoology 81:697-708.

Fenkes, M., J. L. Fitzpatrick, K. Ozolina, H. A. Shiels, R. L. Nudds. 2017. Sperm in hot water: direct and indirect thermal challenges interact to impact on brown trout sperm quality. Journal of Experimental Biology 220:2513-2520.

Flynn, L., A. E. Punt, and R. Hilborn. 2006. A hierarchical model for salmon run reconstruction and application to the Bristol Bay sockeye salmon (*Oncorhynchus nerka*) fishery. Canadian Journal of Fisheries and Aquatic Sciences 63:1564-1577.

Fuhrman, A. E., D. A. Larsen, E. A. Steel, G. Young, and B. R. Beckman. 2018. Chinook salmon emergence phenotypes: describing the relationships between temperature, emergence timing and condition factor in a reaction norm framework. Ecology of Freshwater Fish 27:350-362.

Hague, M. J., M. R. Ferrari, J. R. Miller, D. A. Patterson, G. L. Russell, A. P. Farrell, and S. G. Hinch. 2011. Modelling the future hydroclimatology of the lower Fraser River and its impacts on the spawning migration survival of sockeye salmn. Global Change Biology 17:87-98.

Hayes, D. B., B. J. Bellgraph, B. M. Roth, D. D. Dauble, and R. P. Mueller. 2014. Timing of redd construction by fall Chinook salmon in the Hanford Reach of the Columbia River. River Research and Applications 30(9):1110-1119.

Hearsey, J. W., and A. P. Kinziger. 2015. Diversity in sympatric Chinook salmon runs: timing, relative fat content and maturation. Environmental Biology of Fishes 98:413-423.

Hinch, S. G., S. J. Cooke, A. P. Farrell, K. M. Miller, M. Lapointe, and D. A. Patterson. 2012. Dead fish swimming: a review of research on the early migration and high premature mortality in adult Fraser River sockeye salmon *Oncorhynchus nerka*. Journal of Fish Biology 81:576-599.

Hoffnagle, T. L., R. W. Carmichael, K. A. Frenyea, and P. J. Keniry. 2008. Run timing, spawn timing, and spawning distribution of hatchery- and natural-origin spring Chinook Salmon in the Imnaha River, Oregon. North American Journal of Fisheries Management 28:148-164.

Jeffries, K. M., S. G. Hinch, T. Sierocinski, P. Pavlidis, and K. M. Miller. 2014. Transcriptomic responses to high water temperature in two species of Pacific salmon. Evolutionary Applications 7:286-300.

Jepson, M.A., M.L. Keefer, C.C. Caudill, T.S. Clabough, C.S. Erdman, T.J. Blubaugh, and C.S. Sharpe. 2015. Migratory behavior, run timing, and distribution of radio-tagged adult winter steelhead, summer steelhead, spring Chinook Salmon, and Coho Salmon in the Willamette River: 2011-2014. Technical Report 2015-1, University of Idaho to the US Army Corps of Engineers, Portland District.

Johnson, M. A., and T. A. Friesen. 2014. Genetic diversity and population structure of spring Chinook salmon from the upper Willamette River, Oregon. North American Journal of Fisheries Management 34:853-862.

Jones, E. J. 2012. Environmental stochasticity and the reliability of redd count data: a simulation study of redd construction, redd loss, and survey frequency in a small coastal stream in northern California. Masters Thesis, Humboldt State University.

Karvonen, A., P. Rintamäki, J. Jokela, and E. T. Valtonen. 2010. Increasing water temperature and disease risks in aquatic systems: climate change increases the risk of some, but not all, diseases. International Journal for Parasitology 40:1483-1488.

Keefer, M. L., T. S. Clabough, M. A. Jepson, T. Bowerman, and C. C. Caudill. 2019. Temperature and depth profiles of Chinook salmon and the energetic costs of their long-distance migrations. Journal of Thermal Biology 79:155-165.

Keefer, M. L., T. S. Clabough, M. A. Jepson, T. Blubaugh, G. Brink, G. P. Naughton, C. T. Boggs, and C. C. Caudill. 2018a. Evaluation of adult Chinook salmon behavior at the Foster Dam adult fish facility and in Foster Dam reservoir on the South Santiam River, 2017. Technical Report 2018, University of Idaho to the US Army Corps of Engineers, Portland District.

Keefer, M. L., T. S. Clabough, M. A. Jepson, E. L. Johnson, C. A. Peery, and C. C. Caudill. 2018b. Thermal exposure of adult Chinook salmon and steelhead: diverse behavioral strategies in a large and warming river system. PLOS ONE 13(9):e0204274.

Keefer, M. L., T. S. Clabough, M. A. Jepson, G. P. Naughton, T. J. Blubaugh, D. C. Joosten, and C. C. Caudill. 2015. Thermal exposure of adult Chinook salmon in the Willamette River basin. Journal of Thermal Biology 48:11-20.

Keefer, M. L., M. A. Jepson, G. P. Naughton, T. J. Blubaugh, T. S. Clabough, and C. C. Caudill. 2017. Condition-dependent en route migration mortality of adult Chinook salmon in the Willamette River main stem. North American Journal of Fisheries Management 37(2):370-379.

Keefer, M. L., C. A. Peery, T. C. Bjornn, M. A. Jepson, and L. C. Stuehrenberg. 2004. Hydrosystem, dam, and reservoir passage rates of adult Chinook salmon and Steelhead in the Columbia and Snake Rivers. Transactions of the American Fisheries Society 133:1413-1439.

Keefer, M. L., C. A. Peery, and C. C. Caudill. 2008a. Migration timing of Columbia River spring Chinook salmon: effects of temperature, river discharge, and ocean environment. Transactions of the American Fisheries Society 137:1120-1133.

Keefer, M. L., C. A. Peery, and M. J. Heinrich. 2008b. Temperature-mediated en route migration mortality and travel rates of endangered Snake River sockeye salmon. Ecology of Freshwater Fish 17:136-145.

Keefer, M. L., G. A. Taylor, D. F. Garletts, G. A. Gauthier, T. M. Pierce, and C. C. Caudill. 2010. Prespawn mortality in adult spring Chinook salmon outplanted above barrier dams. Ecology of Freshwater Fish 19:361-372.

Kent, M. L., S. Benda, S. St-Hilaire, and C. B. Schreck. 2013. Sensitivity and specificity of histology for diagnoses of four common pathogens and detection of nontarget pathogens in adult Chinook salmon (*Oncorhynchus tshawytscha*) in fresh water. Journal of Veterinary Diagnostic Investigation 25:341-351.

Macdonald, J. S., J. Morrison, and D. A. Patterson. 2012. The efficacy of reservoir flow regulation for cooling migration temperature for sockeye salmon in the Nechako River watershed of British Columbia. North American Journal of Fisheries Management 32:415-427.

Macdonald, J. S., D. A. Patterson, M. J. Hague, and I. C. Guthrie. 2010. Modeling the influence of environmental factors on spawning migration mortality for sockeye salmon fisheries management in the Fraser River, British Columbia. Transactions of the American Fisheries Society 139:768-782.

Mann, R. D. 2007. The effects of high temperature exposures on migration success and embryo quality of Snake River adult Chinook salmon and steelhead. MS Thesis. University of Idaho, Moscow, ID.

McConnachie, S. H., K. V. Cook, D. A. Patterson, K. M. Gilmour, S. G. Hinch, A. P. Farrell, and S. J. Cooke. 2012. Consequences of acute stress and cortisol manipulation on the physiology, behavior, and reproductive outcome of female Pacific salmon on spawning groundsq. Hormones and Behavior 62:67-76.

McCullough, D. A. 1999. A review and synthesis of effects of alterations to the water temperature regime on freshwater life stages of salmonids, with special reference to Chinook salmon. EPA Report 910-R-99-010.

McCullough, D. A., and coauthors. 2009. Research in thermal biology: burning questions for coldwater stream fishes. Reviews in Fisheries Science 17:90-115.

Miller, K. M., and coauthors. 2014. Infectious disease, shifting climates, and opportunistic predators: cumulatie factors potentially impacting wild salmon declines. Evolutionary Applications 7:812-855.

Minke-Martin, V., S. G. Hinch, D. C. Braun, N. J. Burnett, M. T. Casselman, E. J. Eliason, and C. T. Middleton. 2018. Physiological condition and migratory experience affect fitness-related outcomes in female sockeye salmon. Ecology of Freshwater Fish 27:296-309.

Morbey, Y. E., C. E. Brassil, and A. P. Hendry. 2005. Rapid senescence in Pacific salmon. The American Naturalist 166:556-568.

Murdoch, A. R., T. N. Pearsons, and T. W. Maitland. 2009. Use of carcass recovery data in evaluating the spawning distribution and timing of spring Chinook Salmon in the Chiwawa River, Washington. North American Journal of Fisheries Management 29:1206-1213.

Murdoch, A. R., T. N. Pearsons, and T. W. Maitland. 2010. Estimating the spawning escapement of hatchery- and natural-origin spring Chinook salmon using redd and carcass data. North American Journal of Fisheries Management 30:361-375.

National Marine Fisheries Service (NMFS). 1999a. Endangered and threatened species; threatened status for three Chinook salmon evolutionarily significant units (ESUs) in Washington and Oregon, and endangered status for one Chinook salmon ESU in Washington. Federal Register 64:14308–14328.

National Marine Fisheries Service (NMFS). 1999b. Endangered and threatened species; threatened status for two ESUs of steelhead in Washington and Oregon. Federal Register 64(57):14517–14528.

National Marine Fisheries Service (NMFS). 2008. Endangered Species Act section 9(a)(2) consultation biological opinion and Magnuson–Stevens Fishery Conservation and Management Act essential fish habitat consultation: consultation on the "Willamette River Basin Flood Control Project". National Oceanic and Atmospheric Administration, NMFS, NorthwestRegion, F/NWR/2000/02117, Seattle.

Naughton, G. P., C. C. Caudill, T. S. Clabough, M. L. Keefer, M. R. Morasch, T. J. Blubuagh, and M. A. Jepson. 2016. Migration behavior and spawning success of spring Chinook salmon in Fall Creek, the North Fork Middle Fork Willamette, and the Santiam rivers: relationships among fate, fish condition, and environmental factors, 2015. Technical Report 2016-2, University of Idaho to U.S. Army Corps of Engineers.

Naughton, G. P., C. C. Caudill, M. L. Keefer, T. C. Bjornn, L. C. Stuehrenberg, and C. A. Peery. 2005. Late-season mortality during migration of radio-tagged sockeye salmon (*Oncorhynchus nerka*) in the Columbia River. Canadian Journal of Fisheries and Aquatic Sciences 62:30-47.

Naughton, G. P., M. L. Keefer, T. S. Clabough, M. J. Knoff, T. J. Blubaugh, and C. C. Caudill. 2018a. Tag effects on prespawn mortality of Chinook Salmon: a field experiment using PIT tags, radio transmitters, and untagged controls. North American Journal of Fisheries Management 38:96-103.

Naughton, G. P., M. L. Keefer, T. S. Clabough, M. J. Knoff, T. J. Blubaugh, C. Sharpe, and C. C. Caudill. 2018b. Reservoir provides cool-water refuge for adult Chinook salmon in a trap-and-haul reintroduction program. Marine and Freshwater Research 69(12):1995-2007.

Pankhurst, N. W., and H. R. King. 2010. Temperature and salmonid reproduction: implications for aquaculture. Journal of Fish Biology 76:69-85.

Peterson, M. L., A. N. Fuller, and D. Demko. 2017. Environmental factors associated with the upstream migration of fall-run Chinook salmon in a regulated river. North American Journal of Fisheries Management 37:78-93.

Quinn, T. P. 2005. The behavior and ecology of Pacific salmon and trout. American Fisheries Society, Bethesda, MD.

Quinn, T. P., P. McGinnity, and T. E. Reed. 2016. The paradox of "premature migration" by adult anadromous fishes: patterns and hypotheses. Canadian Journal of Fisheries and Aquatic Sciences 73(7):1015-1030.

Quinn, T. P., M. J. Unwin, and M. T. Kinnison. 2000. Evolution of temporal isolation in the wild: Genetic divergence in timing of migration and breeding by introduced chinook salmon populations. Evolution 54(4):1372-1385.

Richter, A., and S. A. Kolmes. 2005. Maximum temperature limits for Chinook, coho, and chum salmon, and steelhead trout in the Pacific Northwest. Reviews in Fisheries Science 13:23-49.

Robards, M. D., and T. P. Quinn. 2002. The migratory timing of adult summer-run steelhead in the Columbia River over six decades of environmental change. Transactions of the American Fisheries Society 131:523-536.

Rounds, S. A. 2007. Temperature effects of point sources, riparian shading, and dam operations on the Willamette River, Oregon. US Geological Survey, Scientific Investigations Report 2007-5185.

Rounds, S. A. 2010. Thermal effects of dams in the Willamette River Basin, Oregon. US Geological Survey, Scientific Investigations Report 2010-5153.

Rounds, S. A, and N. L. Buccola. 2015. Improved algorithms in the CE-QUAL-W2 water-quality monitoring for blending dam releases to meet downstream water-temperature targets. US Geological Survey, Open-File Report 2015-1027.

Salinger, D. H., and J. J. Anderson. 2006. Effects of water temperature and flow on adult salmon migration swim speed and delay. Transactions of the American Fisheries Society 135:188-199.

Sard, N. M., K. G. O’Malley, D. P. Jacobson, M. J. Hogansen, M. A. Johnson, M. A. Banks, and M. Krkošek. 2015. Factors influencing spawner success in a spring Chinook salmon (*Oncorhynchus tshawytscha*) reintroduction program. Canadian Journal of Fisheries and Aquatic Sciences **72**:1390-1397.

Schreck, C. B., W. Contreras-Sanchez, and M. S. Fitzpatrick. 2001. Effects of stress on fish reproduction, gamete quality, and progeny. Aquaculture 2001(1-4):3-24.

Schreck, C. B., J. C. Snelling, R. D. Ewing, C. S. Bradford, L. E. Davis, and C. H. Slater. 1994. Migratory behavior of adult spring Chinook salmon in the Willamette River and its tributaries. Project #88-160-3, Oregon Cooperative Fishery Research Unit, Oregon State University, Corvallis. Available: docs.streamnetlibrary.org/BPA_Fish_and_Wildlife/92818-7.pdf

Sloat, M. R., D. J. Fraser, J. B. Dunham, J. A. Falke, C. E. Jordan, J. R. McMillan, and H. A. Ohms. 2014. Ecological and evolutionary patterns of freshwater maturation in Pacific and Atlantic salmonines. Reviews in Fish Biology and Fisheries 24:689-707.

Snyder, M. N., and coauthors. 2019. Individual based modelling of fish migration in a 2-D river system: model description and case study. Landscape Ecology 34:737-754.

Stewart, D. J., and M. Ibarra. 1991. Predation and production by salmonine fishes in Lake Michigan, 1978-1988. Canadian Journal of Fisheries and Aquatic Sciences 48:909-922.

Strange, J. S. 2012. Migration strategies of adult Chinook salmon runs in response to diverse environmental conditions in the Klamath River basin. Transactions of the American Fisheries Society 141:1622-1636.

Sullivan, A. B., S. A. Rounds, S. Sobieszczyzk, and H. M. Bragg. 2007. Modeling hydrodynamics, water temperature, and suspended sediment in Detroit Lake, Oregon. US Geological Survey, Scientific Investigations Report 2007-5008.

U.S. EPA. 2003. EPA region 10 guidance for Pacific Northwest State and tribal temperature water quality standards. Region 10 Office of Water, EPA 910-B-03-02.

Wagner, G. N., and coauthors. 2005. Metabolic rates and swimming performance of adult Fraser River sockeye salmon (*Oncorhynchus nerka*) after a controlled infection with *Parvicapsula minibicornis*. Canadian Journal of Fisheries and Aquatic Sciences 62:2124-2133.

Wright, B., T. Murtaugh, and R. Brown. 2015. Willamette Falls pinniped monitoring project, 2015. Report by Oregon Department of Fish and Wildlife, Corvallis and Clackamas, OR.

Zhou, S. 2002. Size-dependent recovery of Chinook salmon in carcass surveys. Transactions of the American Fisheries Society 131(6):1194-1202.

APPENDIX A. Mean daily water temperature data from four USGS gages from 2001–2018: Newberg (Willamette River, #14197900), Greens Bridge (N. Santiam River, #14184100), Mehama (N. Santiam River, #14183000), and Niagara (N. Santiam River, #14181500).

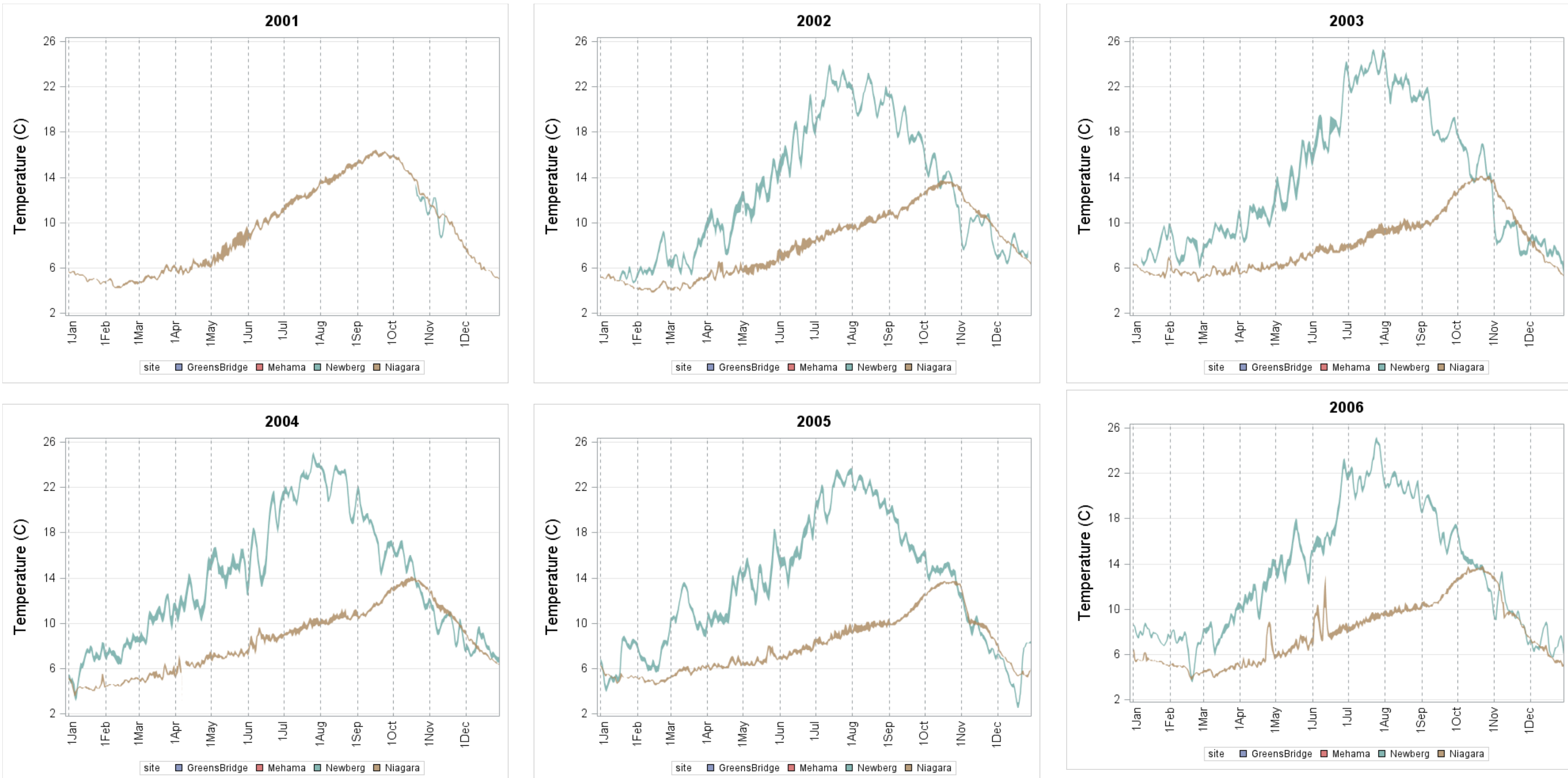


Figure A1. Mean daily water temperature data, 2001-2006.

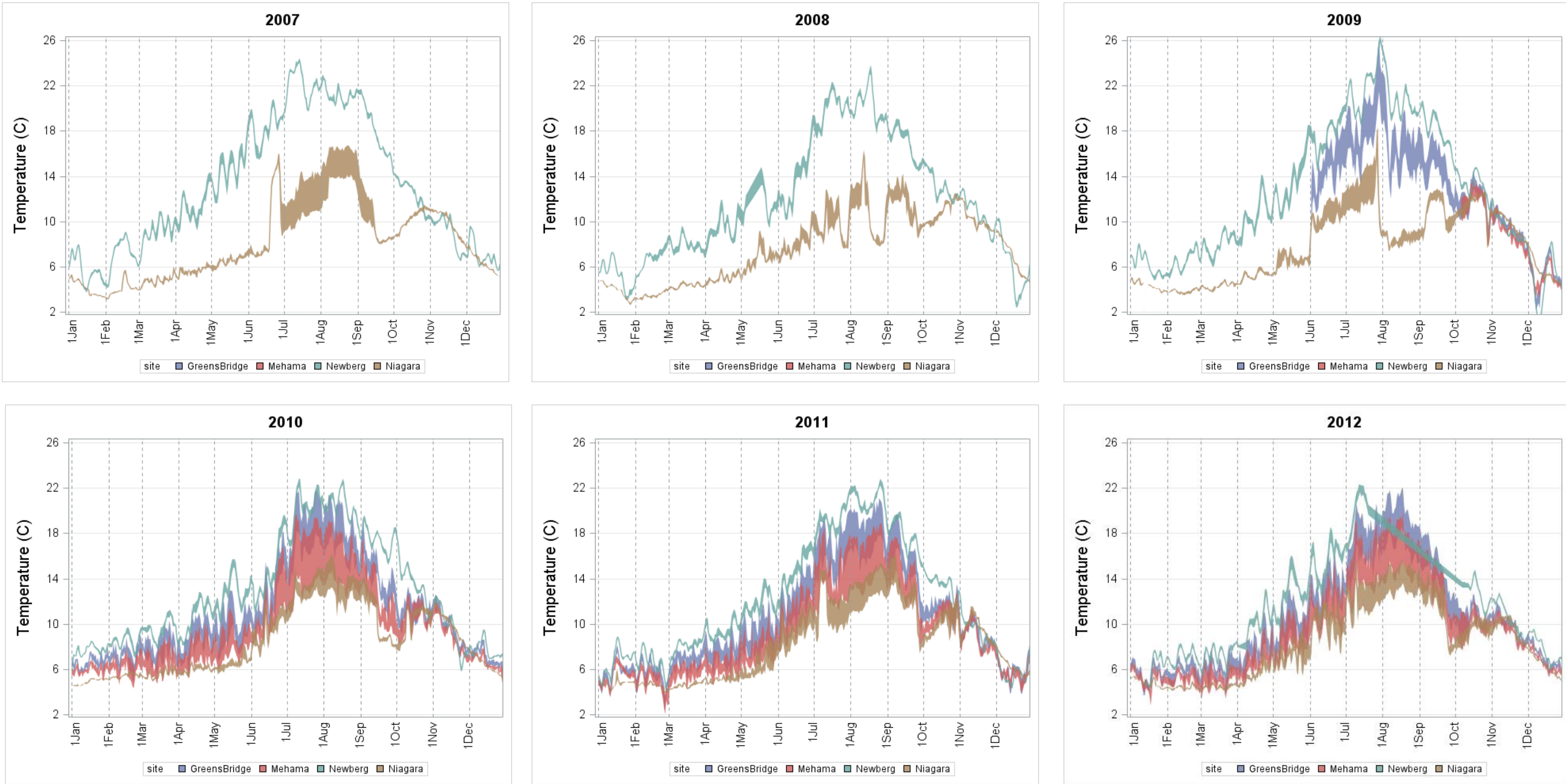


Figure A2. Mean daily water temperature data, 2007-2012.

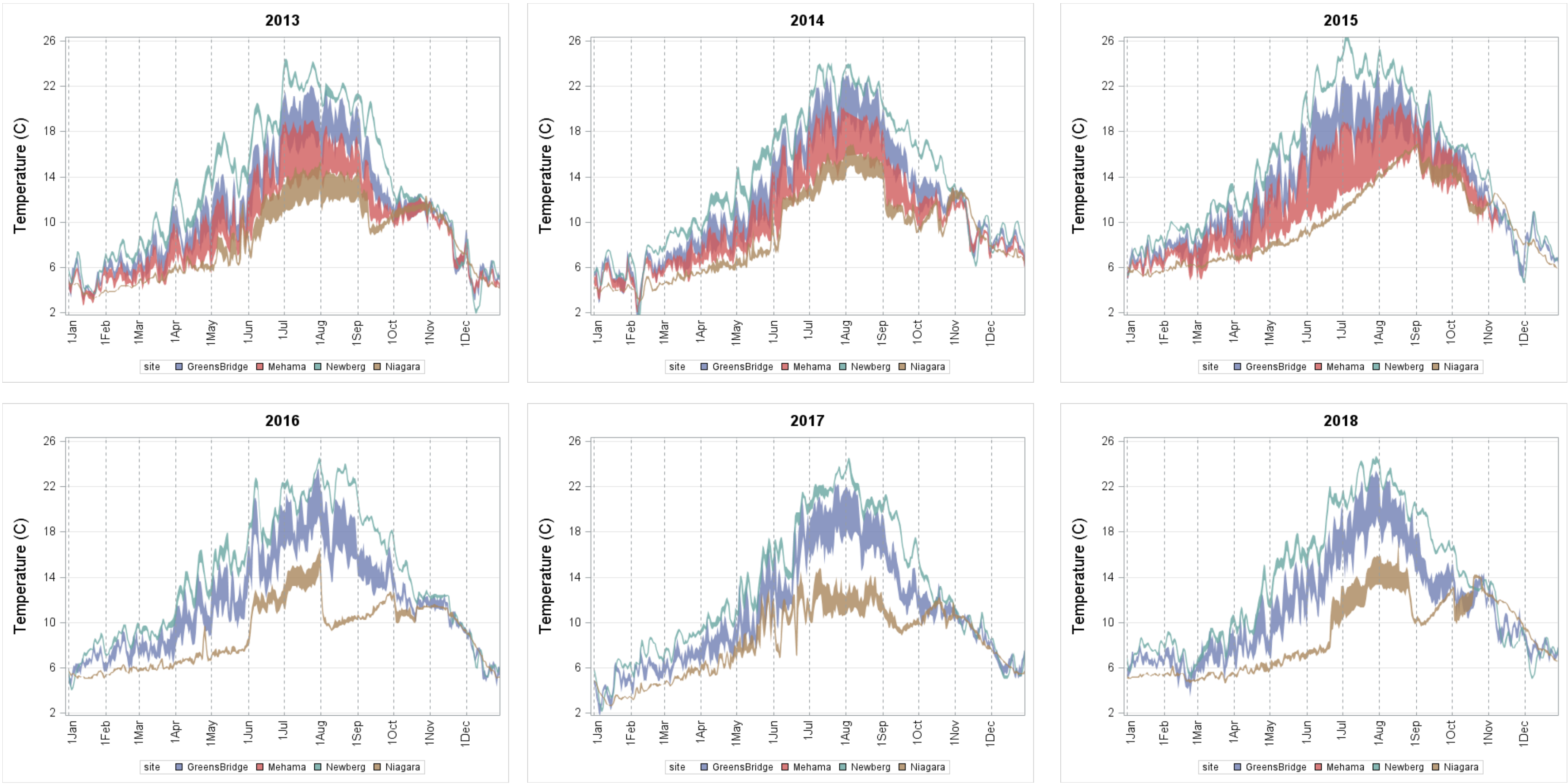


Figure A3. Mean daily water temperature data, 2013-2018.

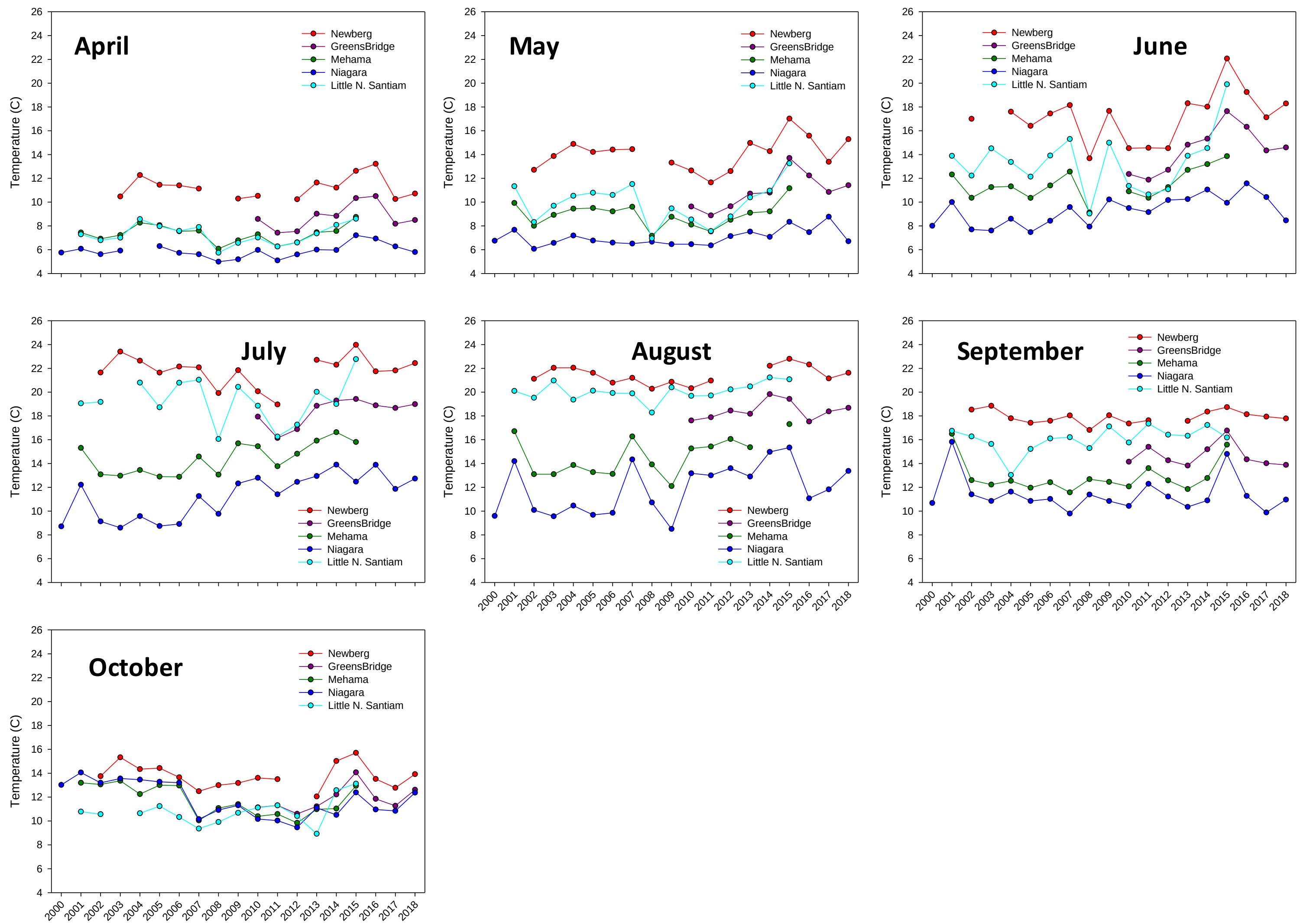


Figure A4. Mean monthly mean water temperature data from five USGS gages from 2000–2018: Newberg (Willamette River, #14197900), Greens Bridge (N. Santiam River, #14184100), Mehama (N. Santiam River, #14183000), Niagara (N. Santiam River, #14181500), and Little North Santiam River (LNS, #14182500).

APPENDIX B. Adult Chinook salmon run timing distributions at Willamette Falls, 2001–2018 and summary tables.

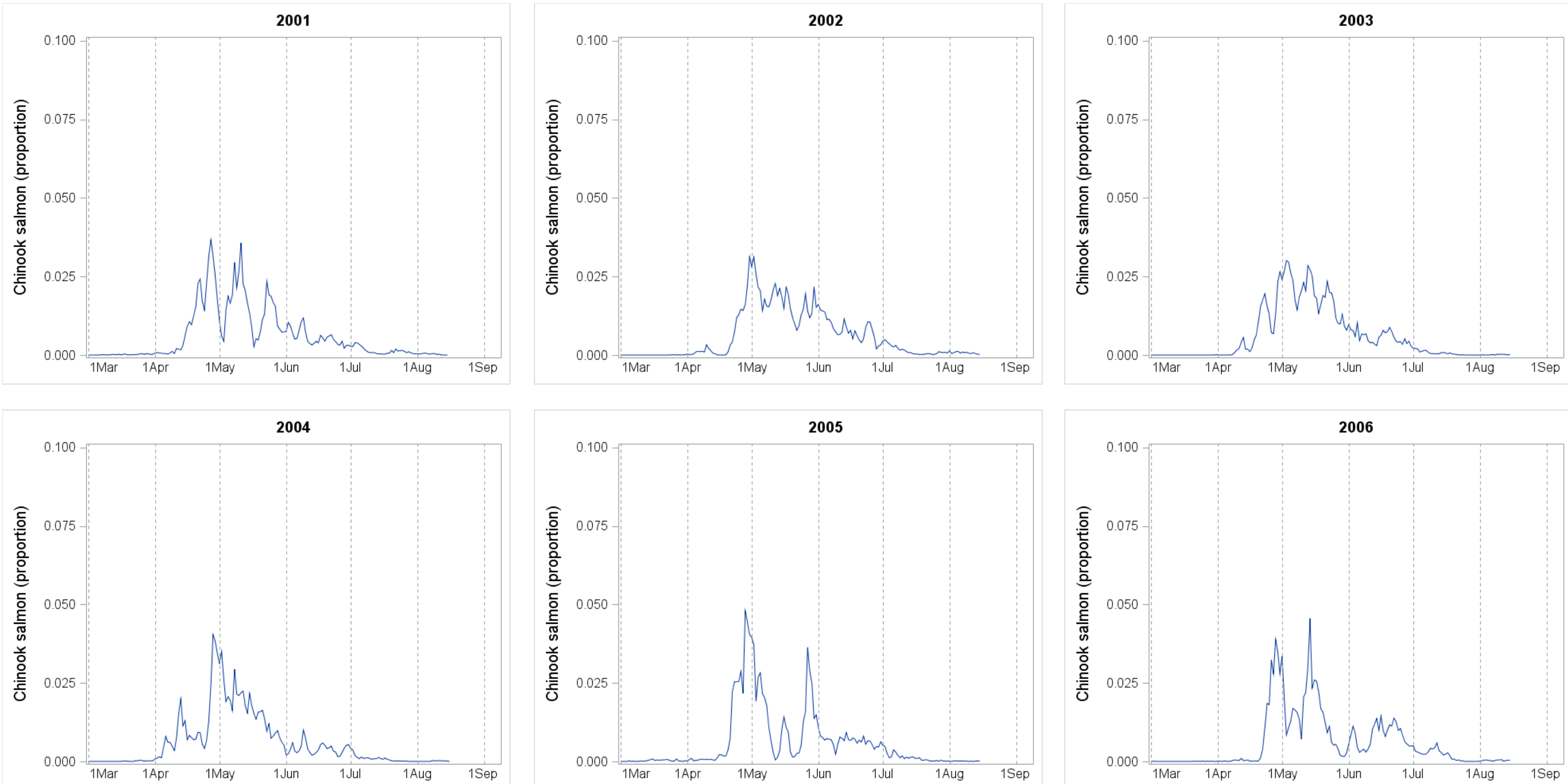


Figure B1. Daily counts of adult Chinook salmon at Willamette Falls, 2001-2006. Source: ODFW.

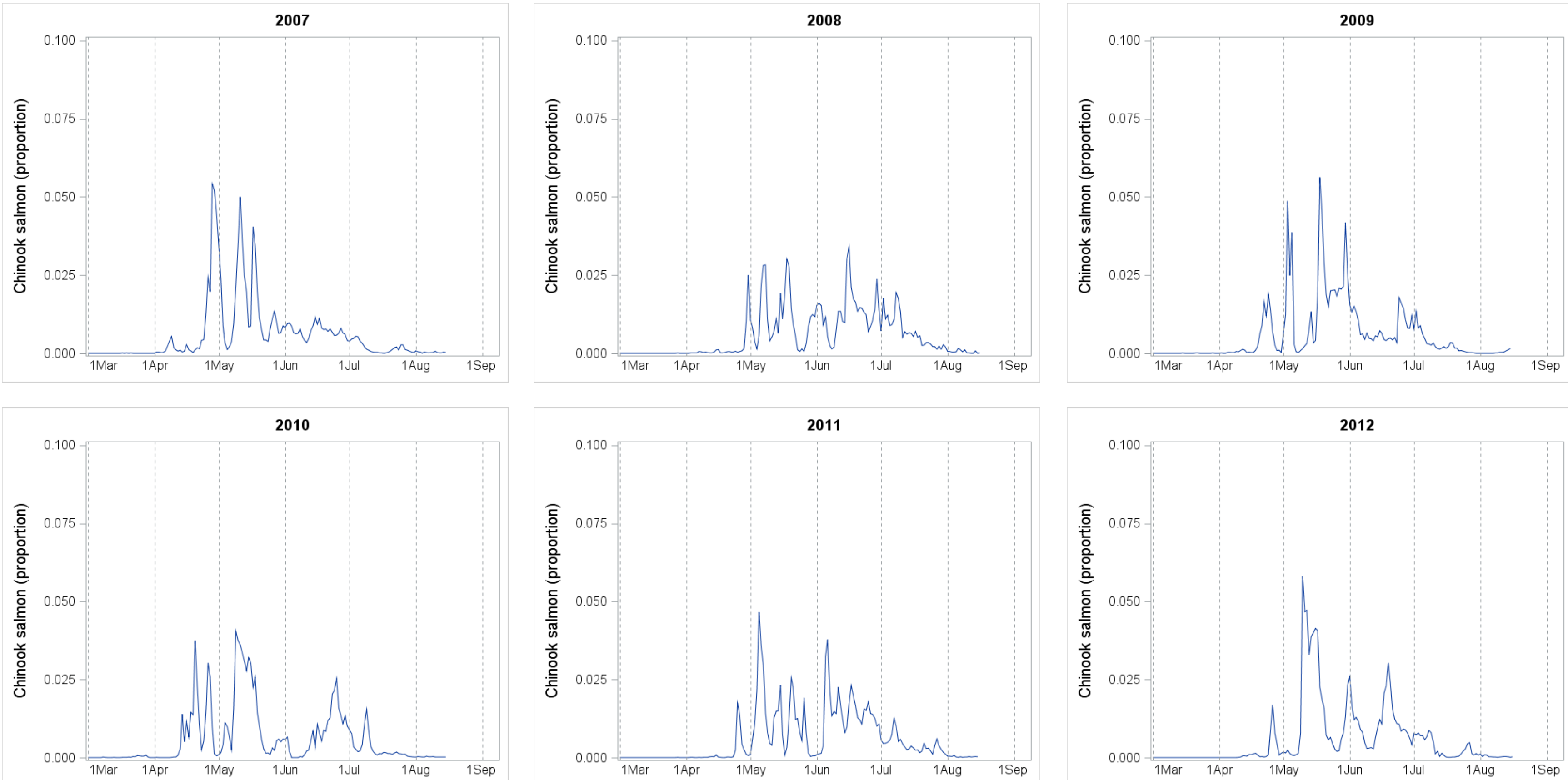


Figure B2. Daily counts of adult Chinook salmon at Willamette Falls, 2007-2012. Source: ODFW.

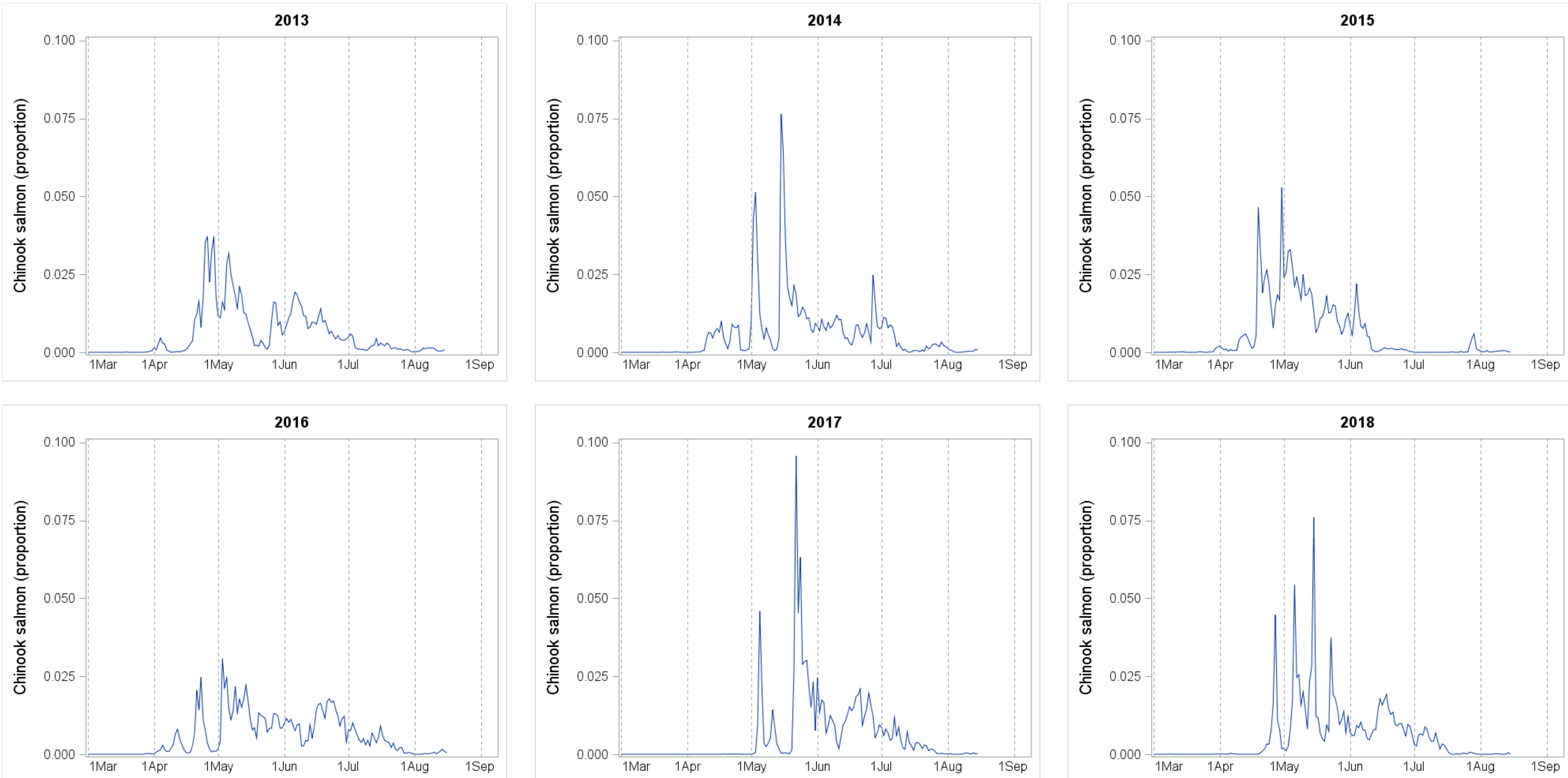


Figure B3. Daily counts of adult Chinook salmon at Willamette Falls, 2013-2018. Source: ODFW.

Table B1. Adult Chinook salmon run timing percentile dates at Willamette Falls, 2001–2018. Source: ODFW

Year	5%	10%	25%	**50%**	75%	90%	95%	Rank
2001	18 Apr	21 Apr	27 Apr	**11 May**	27 May	18 Jun	30 Jun	4
2002	25 Apr	28 Apr	4 May	**17 May**	4 Jun	23 Jun	2 Jul	10
2003	21 Apr	24 Apr	2 May	**13 May**	26 May	16 Jun	23 Jun	6
2004	10 Apr	13 Apr	26 Apr	**6 May**	19 May	9 Jun	22 Jun	2
2005	22 Apr	24 apr	29 Apr	**7 May**	31 May	20 Jun	29 Jun	2
2006	25 Apr	27 Apr	1 May	**15 May**	12 Jun	25 Jun	5 Jul	8
2007	25 Apr	28 Apr	1 May	**14 May**	3 Jun	23 Jun	2 Jul	7
2008	29 Apr	5 May	16 May	**11 Jun**	27 Jun	8 Jul	15 Jul	18
2009	23 Apr	1 May	14 May	**25 May**	9 Jun	30 Jun	5 Jul	13
2010	18 Apr	20 Apr	4 May	**15 May**	21 Jun	30 Jun	9 Jul	8
2011	2 May	5 May	13 May	**6 Jun**	22 Jun	6 Jul	15 Jul	17
2012	28 Apr	8 May	11 May	**24 May**	17 Jun	29 Jun	6 Jul	13
2013	21 Apr	25 Apr	29 Apr	**12 May**	9 Jun	24 Jun	7 Jul	5
2014	17 Apr	25 Apr	5 May	**19 May**	11 Jun	30 Jun	6 Jul	11
2015	18 Apr	20 Apr	27 Apr	**6 May**	21 May	4 Jun	8 Jun	1
2016	18 Apr	21 Apr	6 May	**25 May**	19 Jun	2 Jul	13 Jul	15
2017	5 May	11 May	23 May	**30 May**	20 Jun	3 Jul	9 Jul	16
2018	27 Apr	30 Apr	9 May	**23 May**	16 Jun	28 Jun	6 Jul	12

Table B2. Lower and Upper Bennett run timing dates based on daily data and Minto collection dates. Source: ODFW

Site	Year	5%	10%	25%	**50%**	75%	90%	95%
Lower Bennett	2010	17 May	30 May	13 Jun	**22 Jun**	2 Jul	9 Jul	14 Jul
	2014	22 May	29 May	2 Jun	**18 Jun**	4 Jul	11 Jul	24 Jul
	2015	1 May	4 May	11 May	**23 May**	4 Jun	10 Jun	15 Jun
	2016	9 May	18 May	31 May	**20 Jun**	1 Jul	16 Jul	23 Jul
	2017	1 Jun	5 Jun	12 Jun	**25 Jun**	6 Jul	16 Jul	23 Jul
Upper Bennett	2012	17 May	25 May	31 May	**6 Jun**	22 Jun	17 Jul	29 Jul
	2013	7 May	9 May	21 May	**6 Jun**	6 Jul	23 Jul	14 Aug
	2014	20 May	23 May	31 May	**29 Jun**	6 Jul	12 Jul	31 Jul
	2015	1 Jun	8 Jun	16 Jun	**26 Jun**	10 Jul	19 Aug	25 Aug
	2016	2 Jun	6 Jun	17 Jun	**26 Jun**	5 Jul	17 Jul	23 Jul
	2017	27 May	29 May	2 Jun	**11 Jun**	2 Jul	22 Jul	26 Jul
Minto	2013	10 Jun	14 Jun	25 Jun	**8 Jul**	1 Aug	9 Sep	16 Sep
	2014	10 Jun	16 Jun	8 Jul	**28 Jul**	1 Sep	15 Sep	17 Sep
	2015	30 May	8 Jun	25 Jun	**22 Jul**	31 Aug	11 Sep	18 Sep
	2016	6 Jun	19 Jun	4 Jul	**17 Jul**	30 Aug	11 Sep	12 Sep
	2017	6 Jun	20 Jun	27 Jun	**17 Jul**	14 Aug	6 Sep	11 Sep

APPENDIX C: Annual numbers of female Chinook salmon carcasses that were collected and assessed for prespawn mortality (PSM) in the North Santiam River basin, 1998–2016. Source: ODFW_OWCS database and summaries from C. Sharpe (ODFW).

Table C1. Adult female PSM estimates in the Big Cliff – Minto Dam and Minto Dam – Packsaddle reaches, 1998–2016.

	Big Cliff – Minto Dam				Minto Dam – Packsaddle			
	ODFW_OWCS		Sharpe		ODFW_OWCS		Sharpe	
Year	Females	PSM	Females	PSM	Females	PSM	Females	PSM
1998	-	-	-	-	-	-	-	-
2001	-	-	-	-	8	1.000	-	-
2002	-	-	-	-	5	1.000	*9*	*0.000*
2003	-	-	-	-	-	-	-	-
2004	-	-	-	-	3	0.667	*7*	*0.429*
2005	-	-	-	-	5	0.400	*2*	*0.500*
2006	-	-	-	-	-	-	-	-
2007	-	-	-	-	2	0.500	*7*	*0.286*
2008	-	-	-	-	1	1.000	*1*	*1.000*
2009	-	-	-	-	-	-	-	-
2010	-	-	-	-	-	-	*2*	*0.500*
2011	-	-	-	-	3	0.000	*6*	*0.167*
2012	-	-	-	-	5	0.000	*4*	*0.000*
2013	-	-	*1*	*1.000*	1	0.000	*4*	*0.250*
2014	*8*	*0.000*	*8*	*0.125*	-	-	*2*	*1.000*
2015	*4*	*0.250*	*4*	*0.250*	5	1.000	*5*	*1.000*
2016	12	0.000	-	-	-	-	-	-
Mean PSM	24	0.083	13	0.458	38	0.557	49	0.467
$Mean_w$ PSM	24	0.042	13	0.231	38	0.632	49	0.347

Table C2. Adult female PSM estimates in the Packsaddle – Gates Bridge and Gates Bridge – Mill City reaches, 1998–2016.

	Packsaddle – Gates Bridge[1]				Gates Bridge – Mill City			
	ODFW_OWCS		Sharpe		ODFW_OWCS		Sharpe	
Year	Females	PSM	Females	PSM	Females	PSM	Females	PSM
1998	-	-	49	0.122	-	-	*9*	*0.222*
2001	179	0.855	-	-	101	0.980	-	-
2002	60	0.783	45	0.333	41	0.610	40	0.375
2003	31	0.323	196	0.235	20	0.750	69	0.551
2004	26	0.231	67	0.507	*1*	*1.000*	47	0.574
2005	13	0.000	64	0.219	*3*	*0.333*	22	0.182
2006	23	0.217	46	0.174	*7*	*0.000*	17	0.059
2007	17	0.353	49	0.327	13	0.308	31	0.323
2008	*5*	*1.000*	*8*	*0.000*	*7*	*0.429*	*8*	*0.375*
2009	10	0.200	17	0.118	*4*	*0.750*	*6*	*0.333*
2010	*9*	*0.000*	54	0.241	*8*	*0.125*	37	0.243
2011	60	0.200	151	0.192	33	0.394	91	0.275
2012	45	0.133	36	0.111	28	0.250	21	0.238
2013	21	0.095	37	0.189	13	0.154	20	0.100
2014	35	0.057	62	0.161	12	0.417	32	0.281
2015	35	0.514	35	0.514	19	0.526	19	0.526
2016	38	0.026	-	-	16	0.063	-	-
Mean PSM	607	0.312	916	0.230	326	0.443	469	0.310
Mean$_w$ PSM	607	0.453	916	0.242	326	0.583	469	0.345

[1] Includes 1 fish from Packsaddle–Mill City

Table C3. Adult female PSM estimates in the Mill City – Fishermans Bend and Fishermans Bend – Mehama reaches, 1998–2016.

	Mill City – Fishermans Bend[1]				Fishermans Bend – Mehama			
	ODFW_OWCS		Sharpe		ODFW_OWCS		Sharpe	
Year	Females	PSM	Females	PSM	Females	PSM	Females	PSM
1998	-	-	*2*	*0.000*	-	-	16	0.688
2001	119	0.950	-	-	61	0.869	-	-
2002	23	0.217	20	0.800	40	0.125	37	0.892
2003	17	0.824	54	0.815	58	0.828	139	0.906
2004	*3*	*1.000*	24	0.875	*7*	*0.857*	31	0.935
2005	*7*	*0.857*	16	0.875	*2*	*0.500*	18	0.889
2006	*5*	*0.200*	15	0.083	*4*	*0.250*	21	0.190
2007	*1*	*0.000*	13	0.462	*6*	*1.000*	22	0.591
2008	-	-	*2*	*0.500*	*2*	*1.000*	*1*	*1.000*
2009	*1*	*0.000*	*6*	*0.333*	*8*	*0.250*	14	0.286
2010	27	0.000	*8*	*0.750*	*3*	*1.000*	15	0.400
2011	*5*	*1.000*	77	0.143	56	0.286	109	0.321
2012	16	0.750	12	0.667	11	0.545	10	0.500
2013	*1*	*0.000*	10	0.400	7	0.000	*6*	*0.167*
2014	*7*	*0.429*	12	0.417	*8*	*0.625*	12	0.667
2015	18	0.722	18	0.722	12	0.833	13	0.846
2016	*4*	*0.000*	-	-	*2*	*0.000*	-	-
Mean PSM	254	0.463	289	0.523	287	0.561	464	0.619
$Mean_w$ PSM	254	0.689	289	0.527	287	0.572	464	0.653

[1] Includes 3 fish from Mill City–Mehama

Table C4. Adult female PSM estimates in the Meham – Powerlines and Powerlines – Upper Bennett reaches, 1998–2016.

	Mehama – Powerlines[1]				Powerlines – Upper Bennett			
	ODFW_OWCS		Sharpe		ODFW_OWCS		Sharpe	
Year	Females	PSM	Females	PSM	Females	PSM	Females	PSM
1998	-	-	*2*	*0.500*	-	-	*1*	*1.000*
2001	*4*	*1.000*	-	-	*5*	*1.000*	-	-
2002	*8*	*0.000*	*9*	*1.000*	12	0.333	*9*	*0.667*
2003	43	0.977	74	0.946	40	1.000	53	0.981
2004	*2*	*0.500*	40	0.975	-	-	35	1.000
2005	*2*	*1.000*	*7*	*1.00*	*6*	*1.000*	*9*	*0.778*
2006	*3*	*0.000*	*6*	*0.333*	*2*	*0.500*	*3*	*0.333*
2007	*6*	*1.000*	*2*	*1.000*	-	-	*1*	*1.000*
2008	-	-	-	-	-	-	-	-
2009	*2*	*1.000*	*3*	*1.000*	-	-	*1*	*1.000*
2010	*3*	*1.000*	*6*	*1.000*	-	-	*4*	*0.500*
2011	*6*	*0.833*	12	0.833	*6*	*1.000*	12	0.917
2012	*3*	*1.000*	*2*	*1.000*	*5*	*0.800*	*4*	*0.750*
2013	-	-	-	-	*1*	*1.000*	*1*	*1.000*
2014	*2*	*0.000*	*5*	*0.600*	-	-	-	-
2015	*2*	*1.000*	*2*	*1.000*	*1*	*1.000*	*1*	*1.000*
2016								
Mean PSM	86	0.716	170	0.861	78	0.848	134	0.840
$Mean_w$ PSM	76	0.814	170	0.918	78	0.872	134	0.910

[1] Includes 7 fish from Mehama–Stayton (South Channel) and 1 fish from Mehama–Upper Bennett reaches

Table C5. Adult female PSM estimates in the Upper Bennett – Stayton and Stayton – Shelburn reaches, 1998–2016.

	Upper Bennett – Stayton (both channels)				Stayton – Shelburn			
	ODFW_OWCS		Sharpe		ODFW_OWCS		Sharpe	
Year	Females	PSM	Females	PSM	Females	PSM	Females	PSM
1998	-	-	*1*	*1.000*	-	-	-	-
2001	14	0.929	-	-	*2*	*1.000*	-	-
2002	30	0.267	29	0.759	*7*	*0.143*	11	0.818
2003	43	0.977	96	0.990	*3*	*1.000*	29	1.000
2004	*4*	*0.750*	10	0.800	*2*	*1.000*	21	1.000
2005	*9*	*0.375*	27	0.667	*1*	*1.000*	*3*	*1.000*
2006	*1*	*0.000*	*2*	*0.500*	-	-	-	-
2007	-	-	-	-	*1*	*1.000*	*2*	*1.000*
2008	-	-	*1*	*1.000*	-	-	-	-
2009	*1*	*1.000*	*2*	*1.000*	-	-	*1*	*1.000*
2010	*2*	*1.000*	*4*	*1.000*	-	-	*9*	*1.000*
2011	*8*	*0.875*	19	0.947	16	1.000	16	1.000
2012	*6*	*0.667*	*4*	*0.750*	*3*	*1.000*	*3*	*1.000*
2013	*1*	*1.000*	*1*	*1.000*	-	-	-	-
2014	*1*	*1.000*	*3*	*1.000*	-	-	2	1.000
2015	-	-	-	-	-	-	1	1.000
2016	*1*	*0.000*	-	-	-	-	-	-
Mean PSM	121	0.680	199	0.878	35	0.893	98	0.983
Mean_w PSM	121	0.706	199	0.890	35	0.829	98	0.980

Table C6. Adult female PSM estimates in the Shelburn – Greens Bridge and Greens Bridge – mouth reaches, 1998–2016.

	Shelburn – Greens Bridge				Greens Bridge – Mouth			
	ODFW_OWCS		Sharpe		ODFW_OWCS		Sharpe	
Year	Females	PSM	Females	PSM	Females	PSM	Females	PSM
1998	-	-	*7*	*0.000*	-	-	*1*	*0.000*
2001	-	-	-	-	-	-	-	-
2002	*1*	*0.000*	*2*	*1.000*	-	-	-	-
2003	-	-	28	1.000	*4*	*1.000*	*2*	*1.000*
2004	-	-	*4*	*1.000*	-	-	*1*	*1.000*
2005	*1*	*1.000*	*2*	*1.000*	*1*	*1.000*	*3*	*1.000*
2006	-	-	-	-	-	-	-	-
2007	-	-	-	-	-	-	-	-
2008	-	-	*1*	*0.000*	-	-	-	-
2009	-	-	*2*	*0.000*	*1*	*0.000*	*1*	*0.000*
2010	*1*	*0.000*	*1*	*0.000*	-	-	-	-
2011	*4*	*1.000*	*7*	*1.000*	-	-	-	-
2012	*2*	*1.000*	*2*	*1.000*	-	-	-	-
2013	-	-	-	-	-	-	-	-
2014	-	-	-	-	-	-	-	-
2015	-	-	-	-	-	-	-	-
2016	-	-	-	-	-	-	-	-
Mean PSM	9	0.600	56	0.600	*6*	*0.667*	*8*	*0.600*
$Mean_w$ PSM	9	0.778	56	0.804	*6*	*0.833*	*8*	*0.750*

Table C7. Adult female PSM estimates in all reaches of the Little North Santiam River and all reaches upstream from Detroit Dam, 1998–2016.

	Little North Santiam (all reaches)				Above Detroit Dam (all reaches)			
	ODFW_OWCS		Sharpe		ODFW_OWCS		Sharpe	
Year	Females	PSM	Females	PSM	Females	PSM	Females	PSM
1998	-	-	-	-	-	-	-	-
2001	4	0.500	-	-	-	-	-	-
2002	-	-	7	0.714	-	-	-	-
2003	23	0.913	26	0.808	-	-	-	-
2004	3	0.333	8	0.500	-	-	-	-
2005	48	0.354	43	0.395	-	-	-	-
2006	5	0.600	6	0.333	-	-	-	-
2007	10	0.700	9	0.667	16	0.188	16	0.188
2008	9	0.000	9	0.000	-	-	-	-
2009	27	0.778	27	0.815	-	-	21	0.143
2010	14	0.357	19	0.316	3	0.667	206	0.039
2011	4	0.250	16	0.313	-	-	1	0.000
2012	16	0.375	16	0.375	-	-	5	0.000
2013	4	0.250	5	0.600	2	1.000	115	0.052
2014	-	-	-	-	39	0.077	84	0.083
2015	-	-	-	-	76	0.105	79	0.101
2016	-	-	-	-	57	0.053	-	-
Mean PSM	167	0.448	191	0.486	193	0.348	527	0.076
Mean_w PSM	167	0.500	191	0.508	193	0.109	527	0.066

APPENDIX D: Annual numbers of male and female Chinook salmon carcasses that were collected and assessed for fin clips to assess proportion of hatchery-origin spawners (PHOS) in the North Santiam River basin, 2000–2016. Source: ODFW_OWCS database.

Table D1. Adult PHOS estimates in the four reaches between Big Cliff and Mill City, 2000–2016. Note: in 2000, an additional 505 carcasses were assessed from the combined Minto–Fishermans Bend reach (PHOS = 0.519).

	Big Cliff – Minto Dam ODFW_OWCS		Minto Dam – Packsaddle ODFW_OWCS		Packsaddle – Gates Bridge[1] ODFW_OWCS		Gates Bridge – Mill City ODFW_OWCS	
Year	Carcasses	PHOS	Carcasses	PHOS	Carcasses	PHOS	Carcasses	PHOS
2000	-	-	-	-	*2*	*1.000*	*2*	*1.000*
2001	-	-	14	1.000	364	0.912	226	0.947
2002	-	-	10	0.400	79	0.696	74	0.757
2003	-	-	-	-	43	0.442	42	0.238
2004	-	-	*7*	*0.571*	38	0.632	11	0.727
2005	-	-	*5*	*0.400*	17	0.471	*7*	*0.000*
2006	-	-	-	-	40	0.200	*9*	*0.222*
2007	-	-	*3*	*0.000*	21	0.143	15	0.100
2008	-	-	*1*	*0.000*	*5*	*0.000*	*9*	*0.000*
2009	-	-	*2*	*1.000*	14	0.286	*5*	*0.400*
2010	-	-	-	-	20	0.150	20	0.150
2011	11	0.000	*5*	*0.000*	98	0.133	55	0.218
2012	*7*	*0.000*	*6*	*0.333*	67	0.418	38	0.368
2013	15	0.000	*5*	*0.400*	34	0.412	25	0.240
2014	-	-	*6*	*0.667*	77	0.545	46	0.644
2015	-	-	*8*	*0.750*	57	0.789	31	0.645
2016	-	-	-	-	55	0.636	26	0.615
Mean PSM	33	0.000	72	0.460	1,031	0.463	640	0.445
Mean_w PSM	33	0.000	72	0.556	1,031	0.616	640	0.625

[1] Includes 6 fish from Packsaddle–Mill City reach

Table D2. Adult PHOS estimates in the four reaches between Mill City and Upper Bennett, 2000–2016.

	Mill City – Fishermans Bend[1] ODFW_OWCS		Fishermans Bend – Mehama ODFW_OWCS		Mehama – Powerlines[2] ODFW_OWCS		Powerlines – Upper Bennett[3] ODFW_OWCS	
Year	Carcasses	PHOS	Carcasses	PHOS	Carcasses	PHOS	Carcasses	PHOS
2000	-	-	63	0.492	21	0.571	-	-
2001	166	0.940	114	0.912	*5*	*0.800*	*8*	*0.600*
2002	33	0.788	47	0.809	12	0.667	18	0.667
2003	24	0.417	82	0.268	67	0.030	58	0.241
2004	13	0.154	16	0.500	19	0.421	26	0.538
2005	12	0.500	*3*	*0.000*	*3*	*0.000*	*6*	*0.333*
2006	*7*	*0.429*	*8*	*0.250*	*7*	*0.000*	*2*	*0.000*
2007	*2*	*0.000*	*9*	*0.000*	*7*	*0.571*	-	-
2008	-	-	*2*	*0.000*	-	-	-	-
2009	*4*	*0.500*	12	0.167	*2*	*1.000*	-	-
2010	*3*	*0.000*	11	0.182	*5*	*0.400*	-	-
2011	63	0.143	81	0.086	12	0.417	*9*	*0.222*
2012	24	0.500	17	0.235	*8*	*0.500*	*6*	*0.333*
2013	*1*	*0.000*	13	0.308	-	-	*1*	*0.000*
2014	24	0.583	28	0.643	*9*	*0.667*	*1*	*0.000*
2015	33	0.667	20	0.750	*4*	*0.750*	*1*	*0.000*
2016	*8*	*0.750*	*5*	*0.600*	-	-	-	-
Mean PSM	417	0.425	531	0.365	181	0.485	136	0.280
Mean_w PSM	417	0.643	531	0.490	181	0.331	136	0.382

[1] Includes 10 fish from Mill City–Mehama reach

[2] Includes 36 fish from other reaches starting at Mehama

[3] Includes 26 fish from Powerlines–Stayton reach

Table D3. Adult PHOS estimates in the four reaches between Upper Bennett and the North Santiam River mouth, 2000–2016.

	Upper Bennett – Stayton ODFW_OWCS		Stayton – Shelburn ODFW_OWCS		Shelburn – Greens Bridge ODFW_OWCS		Greens Bridge – Mouth ODFW_OWCS	
Year	Carcasses	PHOS	Carcasses	PHOS	Carcasses	PHOS	Carcasses	PHOS
2000	-	-	-	-	-	-	-	-
2001	16	0.813	*3*	*0.667*	-	-	-	-
2002	48	0.750	14	0.571	*1*	*0.000*	*2*	*1.000*
2003	52	0.192	*6*	*0.667*	-	-	*4*	*0.000*
2004	*5*	*0.400*	12	0.833	-	-	-	-
2005	12	0.167	*1*	*0.000*	*2*	*0.000*	*1*	*0.000*
2006	*2*	*0.000*	-	-	-	-	-	-
2007	-	-	*1*	*0.000*	-	-	-	-
2008	-	-	-	-	*2*	*1.000*	-	-
2009	*2*	*0.000*	-	-	-	-	*1*	*0.000*
2010	*2*	*0.000*	*2*	*0.000*	*1*	*1.000*	-	-
2011	14	0.143	28	0.143	*6*	*0.000*	-	-
2012	*9*	*0.444*	*5*	*0.400*	*3*	*0.000*	-	-
2013	*1*	*0.000*	-	-	-	-	-	-
2014	*4*	*0.750*	*4*	*1.000*	-	-	-	-
2015	*1*	*1.000*	*3*	*1.000*	-	-	-	-
2016	*1*	*1.000*	-	-	*1*	*1.000*	-	-
Mean PSM	169	0.404	79	0.480	16	0.429	8	0.250
Mean_w PSM	169	0.438	79	0.468	16	0.250	8	0.250