Preliminary evaluation of the effects of water withdrawals on estuary habitats in the Skagit River delta and their use by fishes

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Acknowledgements

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Executive Summary

While ecological effects of water scarcity and impacts of over-appropriation have long been a concern in freshwater ecosystems, evaluating these impacts in estuaries is more challenging due to the influence of tidal dynamics and corresponding gradients in habitats for native species. To address these issues in the Skagit lower river and tidal delta, the Duke Study (Duke Engineering 1999) combined analysis of tides and river flows with local study of wetlands, water levels, and scientific understanding of fish behavior to conclude that water withdrawals from the Skagit River could impact habitat function. This study informed the Skagit River Instream Flow Rule, a water right for conserving river flow when discharge at Mount Vernon WA drops 10% below its monthly average. However, the Duke Study was based on limited information, focused on linear models of flow effects, and used techniques that now, 26 years later, are outdated.

In this report, we update some of the analyses of the Duke Study, using a wealth of new data, newer techniques for projecting hydrodynamics, and more flexible modeling approaches for addressing effects of water withdrawals on fish and their habitats. The main question is the degree to which changes in water use and natural water availability affect key habitat elements for important estuarine species in the Skagit tidal delta and Skagit Bay. We consider this a "Preliminary report" because it neglects a couple of key relationships related to fish population responses and potential climate impacts, which we were not able to model under the scope of work. Nonetheless, we have produced analyses examining hydrodynamics, wetland vegetation, and habitat use by fishes for three years (low, medium low, and median flow years) and for four water use scenarios (no water use, current, and two higher water use alternatives), as well as two scenarios addressing an unregulated flow comparison.

The intersection of estuarine hydrodynamics, water "scarcity," and plant and fish biology prompted us to focus on two key time periods - late spring (increasing water use, high but waning salmon habitat use in the delta) and mid-summer (high water use, estuarine wetland stress, increasing salmon habitat use in Skagit Bay). In these two 14-day time periods, we examined hydrodynamics from the upper portion of the delta below Mt. Vernon out to the portions of Skagit Bay most directly affected by the Skagit River plume. The key hydrodynamic elements we examined were surface salinity, water surface elevation, and velocity. These characteristics in turn affected tidal marsh vegetation and fish presence and density.

Due to the breadth and variety of models we used to evaluate hydrodynamics, vegetation, and fishes, we organized this report into six chapters. ***Chapter 1*** provides an overview of the Skagit estuary, details the key elements influencing freshwater availability therein, outlines the water use scenarios we examined, and describes the key results from hydrodynamic, vegetation, and fish models. The nine combinations of model years and water use scenarios generated a broad range of effects on the Skagit River hydrograph, resulting in water withdrawals from 0% to over

20% of average daily discharge at Mount Vernon. Combinations with higher water scarcity had greater impacts on hydrodynamics, vegetation, and fishes, validating the general conclusions of the Duke Report (1999) of freshwater flow-dependent relationships. Chapter 1 also highlights several limitations of the work and lays out future research directions to address these limitations.

Chapter 2 focuses on results from hydrodynamic model development, set up, and application. Hydrodynamic model results for three water use scenarios, three model years, and two time periods reveal that salinity, one key metric influencing biological variation, is sensitive to freshwater flow, particularly in low flow years and the higher water use scenario (see Fig. E1). In contrast, only limited differences relative to baseline were observed in water surface elevation and water velocities across scenarios. Nevertheless, water surface elevation was influenced by river flow. We summarized these on a daily basis to illustrate flow-dependent inundation duration > 30 cm as in the Duke Report (1999) and observed similar patterns. However, these effects were nonlinear and varied with location in the delta channel network. We also observed flow-dependent effects on the duration of surface salinity < 5 ppt.

Chapter 3 focuses on how estuary wetland vegetation – which provides shade, cover, and insect prey for fishes – is influenced by different model years and water use scenarios. The main types of tidal wetland vegetation occur at different elevation and salinity levels, with woody vegetation such as willow and sweetgale shrubs at higher elevations and lower salinities, and herbaceous plants such as grasses, sedges in more tidally dominated areas. We observed shifts in these “niches” in the late summer for different water years and water use scenarios, particularly in the lower South Fork of the Skagit delta. Here, higher observed salinity levels in low model years and high water use scenarios were predicted to reduce shrubs by up to 10% and similarly increase herbaceous vegetation.

Much of the concern over changing river flows is related to how these patterns affect juvenile fishes using the estuary, particularly threatened Chinook salmon. In ***Chapter 4,*** we constructed models of the presence and local density of common estuarine species based on their relationships with salinity, velocity, temperature, and depth, all of which were sampled during long-term monitoring in the Skagit delta and Skagit Bay (Appendix 4.1). We applied these models using hydrodynamic outputs for the three model years, three scenarios, and two seasonal time periods to project how changes in river flow influenced juvenile salmon as well as a number of other species common in the estuarine community.

Because of strong correlations between fish presence and salinity, depth, and velocity, high levels of water use could have upwards of 8% reduction in juvenile Chinook salmon presence and abundance when present, particularly in low flow years in mid-summer (see Fig. E1). Following outcomes from the hydrodynamic and vegetation models, Chinook salmon were most strongly influenced in the lower South Fork wetlands. Other species exhibited sensitivities to these conditions, resulting in shifts in fish communities in both tidal delta and Skagit Bay in the scenario and model years when freshwater scarcity was highest.

The results of this study provide qualified support for the findings in the Duke Study and indicate that greater withdrawals (Alternative 2) during periods of lower river flows are likely to increase delta and estuary salinities to levels that are in turn likely to stress key vegetation types and reduce juvenile salmon presence and abundance, including threatened Chinook salmon. However, modeled water withdrawals under current or slightly elevated scenarios (Alternative 1) at average river flows were estimated to have minimal effects on fish use and plant communities in downstream habitats. Changes due to salinity were most pronounced in the relatively shallow South Fork region of the Skagit delta, suggesting that the effective doubling of water use under Alternative 2 in low flow years could have repercussions for habitat protection and restoration supporting recovery of Chinook salmon populations.

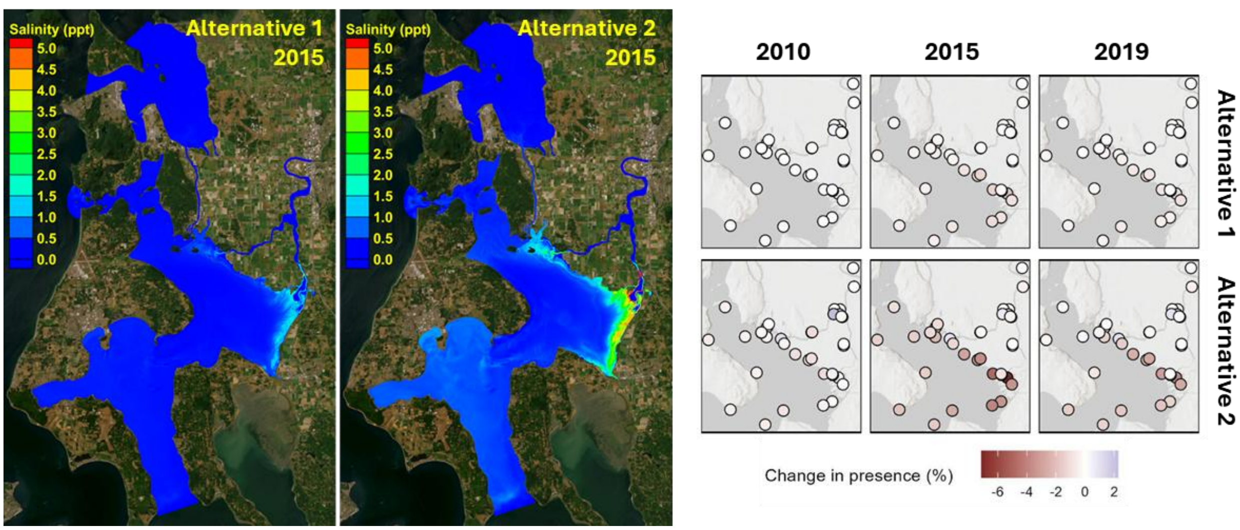


Figure E1. Left Panel: Difference in maximum salinity from May-September in Alternative 1 and 2 water use scenarios compared to current conditions as predicted by the hydrodynamic model (see Fig. 2.19). Right Panel: Projected changes in the occurrence of juvenile Chinook salmon in the three model years and in Alternative 1 and Alternative 2, compared to current conditions (see Fig. 4.3).

1. Overview of study, water use simulations, and key results

Introduction

Rivers provide essential water resources to aquatic ecosystems and human communities, yet worldwide these functions are under threat from climate change and over-appropriation (Milliman et al. 2008, Doll et al. 2009). More locally, regulatory efforts through in-stream flow rules have often been put in place to balance water needs for people and the contribution of water to ecosystem function. If these regulations can be based on sound scientific principles, communities should be able to sustainably balance water withdrawals for people and available water resources for ecosystems.

However, the central question of how much water is necessary for ecosystem function is challenging to address. Among the many dimensions of this question is how much water do fishes need to thrive in their aquatic habitats? This issue has arisen over the past 50 years in places as diverse as the Yangtze River in China (Wang et al. 2016), the US Great Plains (Perkin et al. 2015), and the Sacramento River of California (Sommer et al. 2020, Michel et al. 2021), where water scarcity has sharpened conflicts between human needs for water with ecosystem services and conservation. While instream flow requirements have often been central questions in arid portions of the western United States (Arthaud et al. 2010, Naik and Jay 2011), concerns over water scarcity have also been raised in the Coastal Pacific Northwest (Yoder et al. 2021), an ecoregion with abundant but seasonal rainfall.

In the United States, where fish populations have been listed under the Endangered Species Act, and in other countries where important fish stocks have declined due to water scarcity, questions revolving around the issue of “how much water is enough” have driven the construction of tools to determine critical aquatic resources for fishes and other species. These tools include Physical Habitat Simulation (“Phabsim”, Milhous and Waddle 2012), bioenergetic models (Rosenfeld et al. 2016), and fish life cycle models (Arthaud et al. 2010, Friedman et al. 2019, Peterson et al. 2022). These tools have helped generate science to support instream flow rules, essentially water rights defining water levels that support ecological function in rivers.

One key challenge of many tools for assessing impacts of water withdrawals is that these models can be difficult to apply in estuarine systems where a substantial component of “instream” water is due to daily tidal flux. In large river systems, the tidal component can be a significant source of variation in water levels for quite a distance upstream. For example, the Columbia River exhibits tidal variation up to Bonneville Dam, 234 km from the mouth (Jay et al. 2015).

The Skagit River Watershed, the largest river system entering Puget Sound, has been a focal point in Western Washington for water scarcity and instream flow management. Concerns over the possible over-appropriation of water to support spawning and rearing habitat for anadromous fishes prompted study of flow levels supporting fishes in the lower Skagit River and its tidal delta. These studies, collectively called “The Duke Study” (Duke Engineering 1999), formed the scientific basis for the Skagit Instream Flow Rule (IFR) (WAC-173-503-030), which identified ecological limits to withdrawals in the lower river and tidal delta. These limits were determined in-river using Phabsim and in the delta using a linear regression with river discharge and tide as predictors and water surface elevation as the main dependent variable. The analysis was used to calculate the discharge-dependent duration that channels were wetted at least one foot in depth. This analysis led to the determination that a reduction in river flow by 836 CFS would result in a 10% reduction in the duration of inundation of at least one foot depth, a conclusion that was subsequently included in the Skagit IFR.

The Duke study used methods and knowledge that were relevant at the time of the study. However, a 2021 peer review of that study by a committee of the Washington State Academy of Sciences (“the WSAS Review”) found weaknesses in the way data were collected and the methods of analysis that were used, and identified ways in which a new study might take advantage of developments since the 1990s: new data sources, better sampling designs, improved technology, better understanding of fish ecology, and new simulation models (WSAS, 2021).

To address some of the shortcomings of the Duke Report, The Joint Legislative Task Force on Water Supply provided funding for this study, which is a multi-model effort to link water withdrawals in the Lower Skagit River and tidal delta to ecological conditions related to habitat for anadromous salmonids. Through analysis of 14 scenarios modeling different levels of water scarcity (3 years x 4 water use scenarios, as well as two additional scenarios examining an unregulated hydrograph) we address the following questions:

- How do increased water withdrawals affect water surface elevation, salinity, and velocity in the Skagit tidal delta and Skagit Bay nearshore, independent of tidal fluctuations?
- How do hydrodynamic changes influence the potential maintenance of tidal delta vegetation, which provide shading, refuge, and invertebrate prey for juvenile salmon?
- How do these hydrodynamic changes influence the distribution of juvenile salmonids and other important fishes in the Skagit tidal delta and Skagit Bay, particularly during key months of water scarcity?

As noted by its title, this report should be considered a *preliminary analysis* of the influence of water withdrawals on salmonids and their habitat use, for two important reasons. First and foremost, funding and time were insufficient to address key issues of the interaction of changing climate conditions and water withdrawals on habitat impacts. Climate change impacts on hydrographs, water temperatures, and sea level all affect estuarine habitats for salmonids, as well as for people living and farming in the lower Skagit River. Climate impacts were not incorporated into hydrodynamic model scenarios, except to the extent that extreme water years and water withdrawal scenarios mimic future projections. Nevertheless, climate impacts to the Skagit's freshwater hydrograph (Hamman et al. 2016, Lee et al. 2016, Yoder et al. 2021), water temperatures (Mote et al. 2005, Bandaragoda et al. 2019), sea level (Hood et al. 2016) and cropland evapotranspiration (Yoder et al. 2021) will create novel baselines for habitat conditions in the tidal delta, and these changes deserve consideration for understanding the consequences of changing water demand on aquatic habitats and the species that depend upon them.

Secondly, analysis of impacts to fishes was limited to changes in distribution. Other models including bioenergetic models (Rosenfeld et al. 2016) and life cycle models (Friedman et al. 2019) have been used to project impacts of water use on fish growth and survival, respectively, so the models used herein to predict changes in habitat use should be treated as an initial foray into the cumulative effects of multiple impacts to salmon life cycles. Both of these issues were called out in the WSAS (2021) review and remain a priority for future research efforts (see Recommendations for future research, below).

In this chapter, we provide the key study design elements and results that address the above questions. Subsequent chapters focus on model and study subcomponents that comprise our multi-model analysis.

Conceptual model of habitat impacts of water withdrawals

We put together a set of linked models to address the challenges of modeling the multiple ways in which surface water in rivers can affect fishes in estuaries. Our conceptual framework (Fig. 1.1) illustrates multiple pathways by which a natural hydrograph can be modified by water uses and the multiple ways in which river flow can affect local estuary conditions through hydrodynamic change. The main hydrodynamic variables of interest are water surface elevation, salinity, and water temperature, although other parameters such as currents and suspended sediment concentration can be important ecological drivers. The conceptual model illustrates that elements important to wetland vegetation and estuarine fishes are influenced by both freshwater and tidal regime. As shown in our conceptual model, we expected changes in freshwater to affect persistence of marsh vegetation depending on specific elevational and salinity variation. In addition, the combination of hydrodynamic and vegetation changes could also affect juvenile fishes using tidal delta and nearshore habitats.

One effective approach for assessing the ecological consequences of altered freshwater flow regimes on estuarine fishes is to integrate hydrodynamic models with fish–environment relationship models (see Ganju et al. 2024 for a broad review). In this study, we developed hydrodynamic models and then linked them to marsh vegetation models and fish presence and

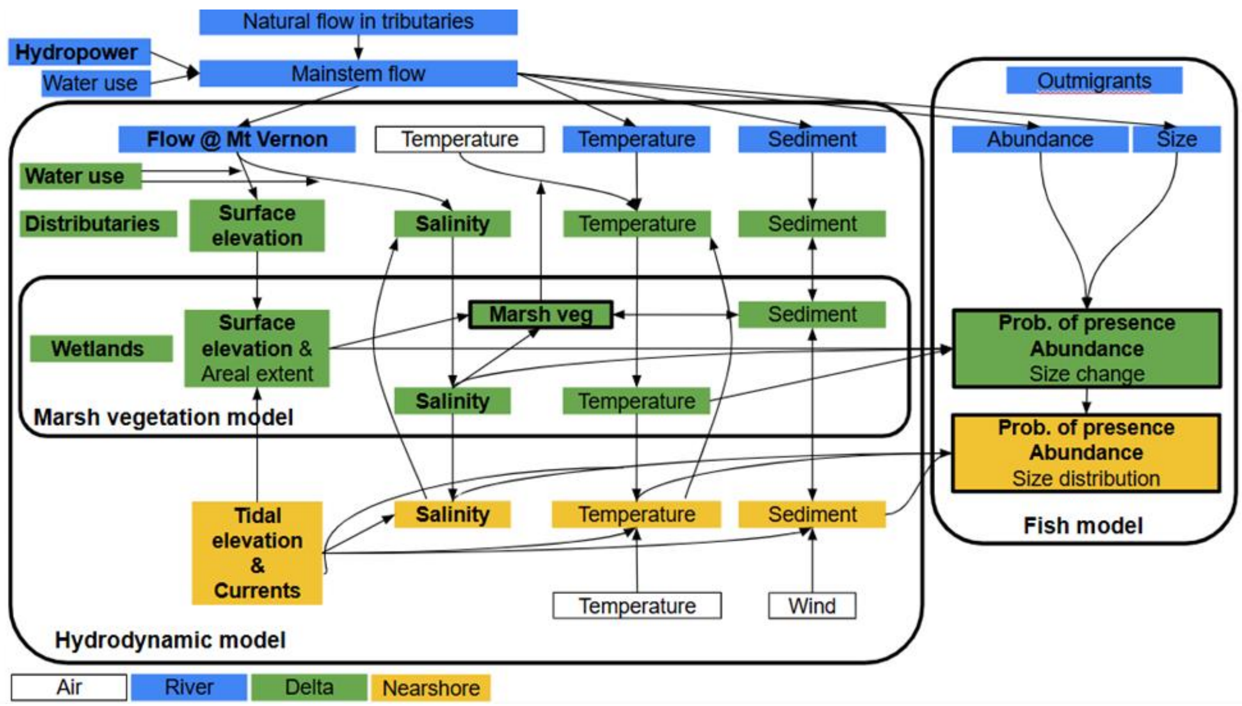


Figure 1.1. Conceptual model of how freshwater and tidal processes (including anthropogenic water uses) combine to affect habitat conditions in riverine (blue), delta (green), and nearshore (yellow), which in turn affect fishes and wetland vegetation supporting them (black-edged boxes). Arrows depict direction of effect. Boldface illustrates parameters examined in this report.

density to evaluate how a range of plausible flow regimes may influence vegetation and the fish assemblage in the Skagit River estuary (Washington, USA). Specifically, we used a hydrodynamic model to simulate how riverine and tidal flow regimes influenced local physical conditions within estuaries (e.g. depth, velocity, and salinity), and niche space and fish-environment models quantify how vegetation cover and fish presence or abundance responds to these environmental gradients. By coupling these two model types, it was possible to predict spatial and temporal variation in vegetation and fish abundance under alternative flow scenarios. Outputs of these models could be statistically evaluated for independent effects of flow and tides on hydrodynamic, vegetation, and fishes to address some of the shortcomings of the Duke study (1999), which focused solely on water surface elevation. Of course, not all pathways could be examined in the application of the models (see Fig. 1.1), and we address some of these limitations at the end of this chapter.

Freshwater flow and its uses in the Skagit delta

The Skagit River, draining areas as far north as Manning National Park in Canada, as far south as Columbia Peak in the Monte Cristo portion of the Sauk River, and as far east as the Cascade Crest, is capable of producing large flows into the tidal delta. With two Cascade volcanoes and many additional large peaks in between, this glaciated watershed produces a bimodal hydrograph

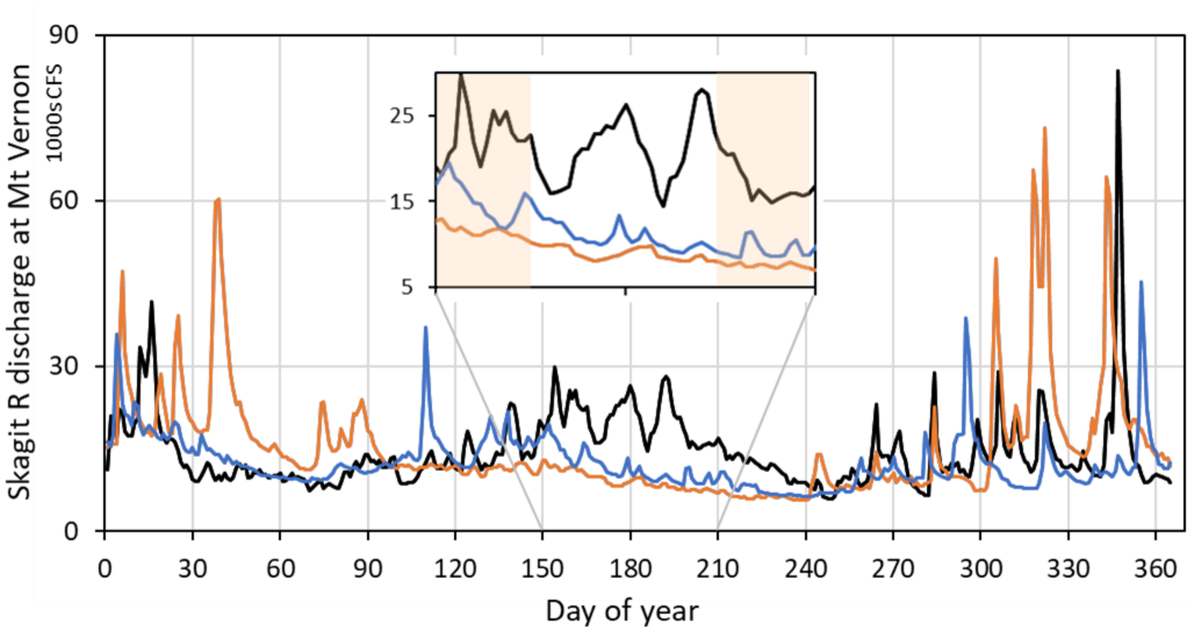


Figure 1.2. Annual hydrographs of the three model years in this study: 2010 (average year, black), 2015 (historic low, orange), and 2019 (moderately low, blue). Inset shows key spring to summer time period of snowmelt-driven hydrograph, with shaded areas denoting focal time periods of analysis of hydrodynamics and effects on fish species. Note that in lower-flow years, the snowmelt-driven increase in discharge occurred earlier in the year, outside the focal time window.

(Fig. 1.2) with precipitation-driven spikes in the late fall and winter, and snowmelt-driven freshet in the late spring. However, climate projections of the hydrograph of the Skagit River suggest that as snowfall and Cascade glaciers decline, snowmelt will decrease by 10-18% (Yoder et al. 2021, Fig. SW2). This is of particular interest for our study because the spring snowmelt coincides with increases in demand for water for agriculture and municipal uses (Yoder et al. 2021, Fig. BP5).

We used average daily discharge (cubic feet per second, CFS) data from the USGS gage at Mt Vernon in three modeled years (2010, 2015, and 2019) as the key input of freshwater water flow into the Skagit hydrodynamic model (See Chapter 2). These years correspond to average, the historical low, and a moderately low flow conditions, respectively. While none of the model years explicitly represent projected patterns of flow in a future climate, the 2015 hydrograph fits the seasonal shifts expected under climate change.

Components of the lower Skagit River water system

The Skagit River's tidal delta is the complex product of freshwater processes of the largest river in Puget Sound (Yoder et al. 2021), and people have engineered numerous changes to the way water moves in and out of the tidal delta (Fig. 1.3). Several key elements that we considered as part of conceptualization of water inputs and exports in the Skagit watershed follow.

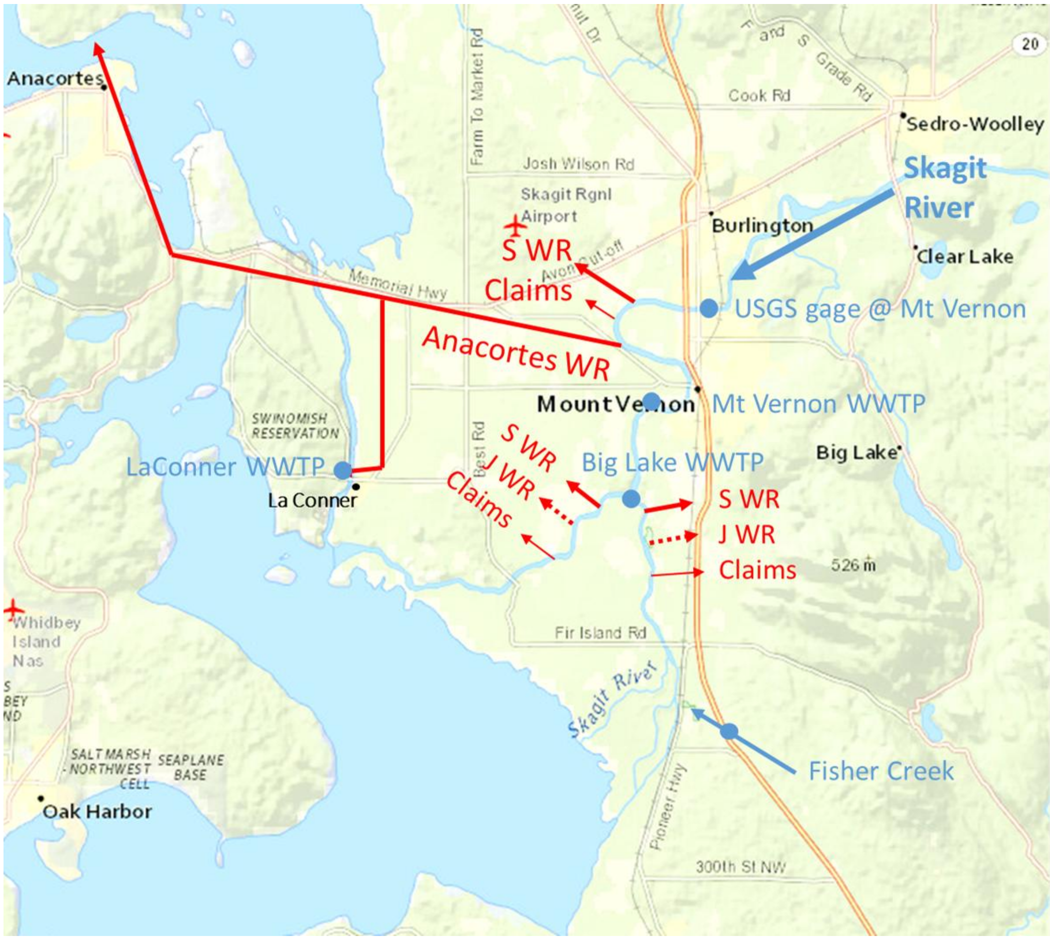


Figure 1.3. Simplified illustration of Skagit tidal delta inputs and diversions modeled in this study. Blue lines represent rivers providing flow into the delta. Blue points and labels represent locations for which both riverine and anthropogenic inputs (sewage outfalls from wastewater treatment plants (WWTP)) are monitored. Red arrows and labels depict diversions, noted as water rights senior to the Skagit Instream Flow Rule (S WR), junior water rights (J WR), and claims. While each of these diversion types have many individual points within the tidal delta, they are modeled as single diversion points. Also illustrated is the diversion for the City of Anacortes (Anacortes WR), which also provides water to the town of La Conner. (Basemap source: Google)

Skagit River hydropower

Hydropower operations on the Baker River and Upper Skagit River modify the natural hydrograph. Current operations are managed to maintain hydropower and even out some of the extreme flow patterns (Lee et al. 2016). Consequently, the observed hydrograph is different from the natural one. We used The University of Washington's Distributed Hydrology Soil Vegetation Model (DHSVM) to simulate natural flow in 2010 (see Skagit Water Story Map, Yoder et al. 2021) and compared that simulation with observed flow in the 2010 calendar year. In this simulation, the observed flow data but not the simulated hydrograph includes water use upstream of Mt. Vernon (Fig. 1.4). Both datasets ignored water inputs and exports downstream of the gage (see below).

Local natural water sources within the delta

Historically, water flowed directly into the Skagit delta not only from the Skagit River but also from local catchments surrounding the delta. In the current landscape modeled by the Skagit Delta Hydrodynamic Model (SHDM), some of these sources remain unquantified, others are small enough to be ignored, and only one additional natural source (Fisher Creek) was explicitly modeled.

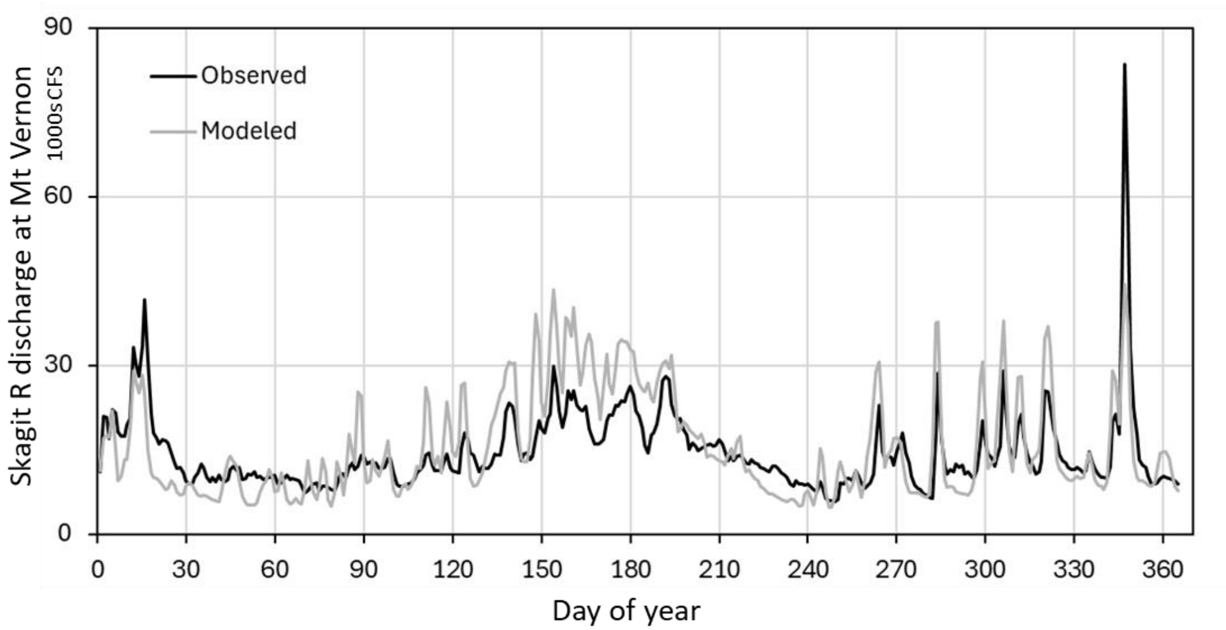


Figure 1.4. Annual hydrographs of discharge of the Skagit River at Mount Vernon in 2010: observed (black) and modeled using University of Washington's DHSVM historical reconstruction (gray).

Local precipitation. Rainwater falling within the Skagit delta comprises an input that can modulate flow levels within the delta. Incorporating precipitation would require assumptions about overland flow, amount used by locally growing plants, retention in groundwater, and release into river, distributaries, and tidal channels. As these elements are not quantified in the Skagit hydrodynamic model, we ignored local precipitation and groundwater as sources affecting hydrodynamics.

Fisher Slough. Fisher Slough is an extension of the tidal delta south of the town of Conway. It is fed by Fisher Creek and its two tributaries Big and Little Fisher Creeks. In addition, Hill Ditch, an irrigation ditch with inputs from four other foothill creeks (Carpenter, Sandy, Johnson and Bolson Creeks), joins Fisher Creek just west of I-5 (Tetratech 2007).

In water years 2007 and 2008 (10/2006 - 9/2008), USGS measured flow on Fisher Creek just east of I-5 and upstream of the Big Ditch confluence (USGS # 12200701). These data have been used to calibrate the National Weather Service's National Water Model (NWM) to produce predictions in any year of interest.

However, because the focus of the NWM is on high flows, which occur when water demand in the delta is low, we also produced a statistical model. This model related Fisher Creek flow with flow data from the EF Nookachamps River, which flows in the foothills directly to the east and is gaged by Washington State Department of Ecology (gage # 03G100) throughout the years of interest (Fisher Creek gage record as well as this report's focal years 2010, 2015, and 2019). We modeled each focal year using a generalized additive model (GAM) as follows:

log(Fisher Creek flow) ~ log(EF Nookachamps flow) + s(day of year)

where s(day of year) is the smoothing parameters of the GAM. When predictions of this model and those from the NWM were compared against the two years of Fisher Creek data, we found that the NWM better predicted high flow events while the GAM better predicted low flows (Fig. 1.5). Hence, we used a model average of the two to produce predictions of Fisher Creek flow input in the three focal years of this report.

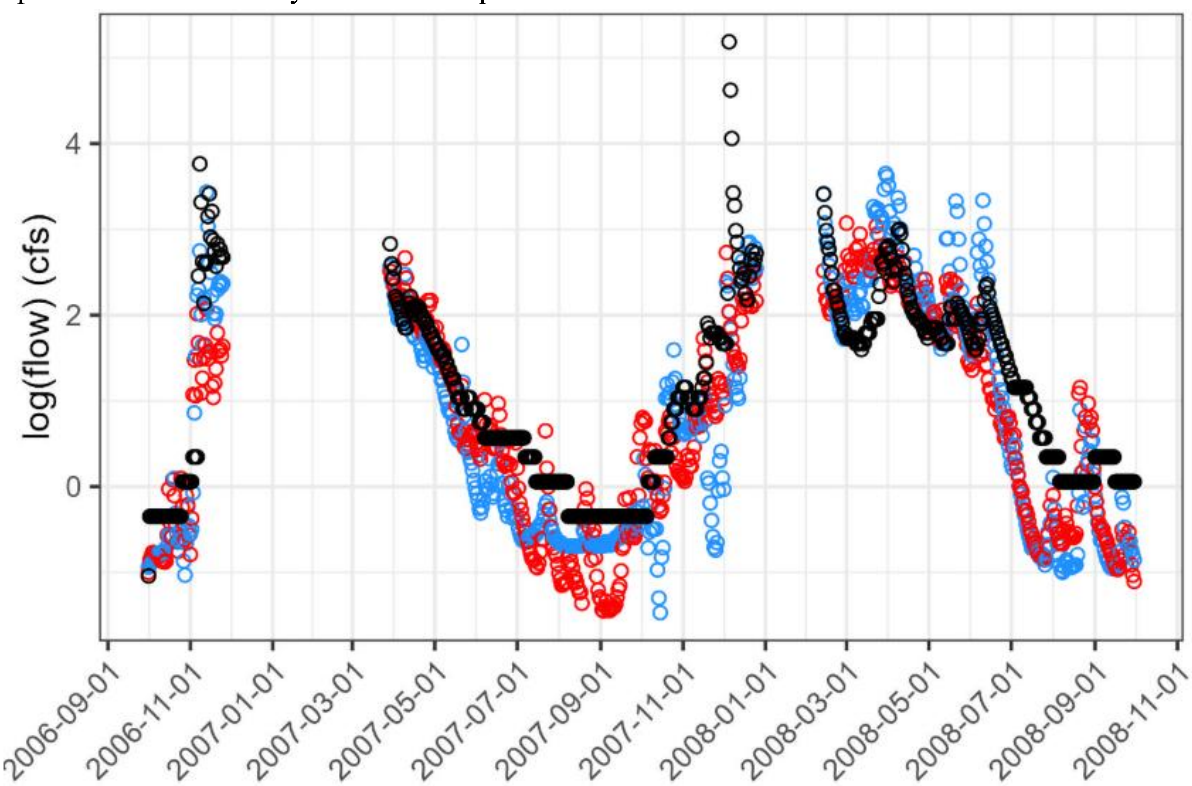


Figure 1.5. Observed flows (blue), GAM-predicted flows (red), and NWM-predicted flows (black) at the Fisher Creek gage.

Municipal water use and operations

Municipalities of Mount Vernon, Anacortes, Big Lake, and LaConner all utilize the Skagit River as water supply or release wastewater into the lower River, thereby modifying flow regimes. Each of these exports and inputs were modeled slightly differently.

Mount Vernon and connected municipalities. Skagit Public Utilities draws water from local tributaries and the Skagit River upstream of the USGS gage at Mount Vernon, so exports of these water uses are largely incorporated into any flow measure at the gage. However, the wastewater treatment plant is located on the lower mainstem and therefore acts as an additional input of water in our model. Discharge at the wastewater outflow is monitored for compliance with Department of Ecology regulations, and summaries of outflow can be found on its Water Quality Permitting and Reporting Information System (PARIS) website. For each water year, we used monthly summaries of wastewater effluent flow and applied those to each day of the relevant month.

Anacortes. The largest export of water is the City of Anacortes municipal water right, and the pumping station is located on the lower mainstem upstream of the wastewater treatment plant. Anacortes has a maximum uninterruptible water right of 85 CFS, and an additional 32.3 CFS interruptible when river flow declines below the IFR. However, the maximum water right has not been exercised due to current pumping limitations and lack of demand. However, demand does increase seasonally during the summer.

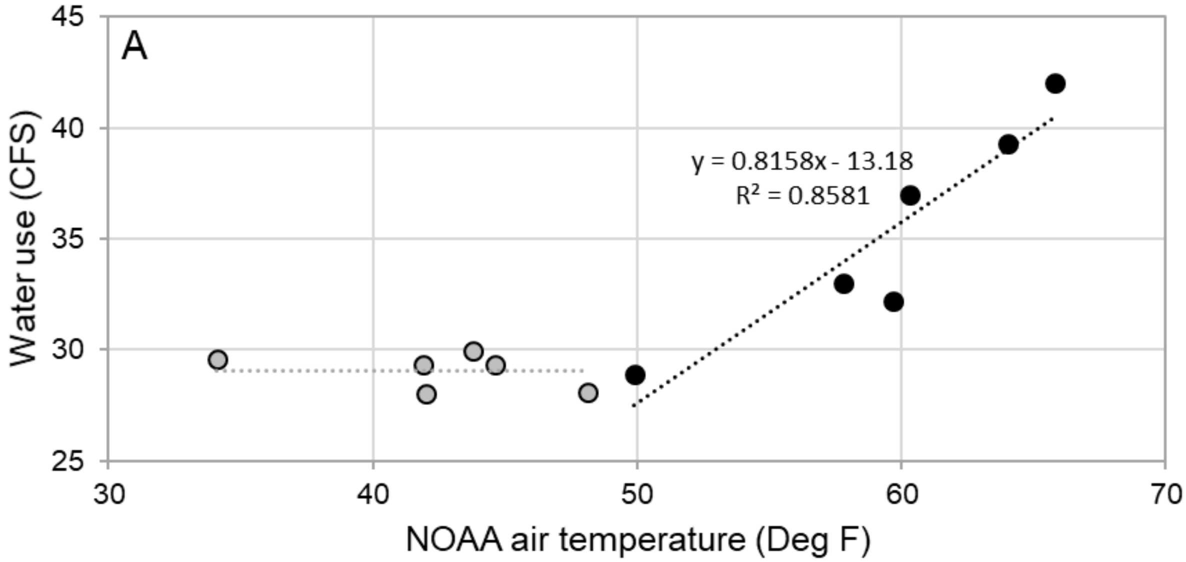
To simulate current water use, we obtained one year of data (2019) on average monthly pumping rate from the City of Anacortes. Following findings of Breyer and Heejun (2014) that water use was positively associated with air temperature in the spring and summer months, we developed an air temperature-dependent relationship with monthly 2019 values. As shown in Fig. 1.6A, water use is independent of air temperature (NOAA monthly estimate in 2019 for Puget Sound Lowland) from October - March, but increases predictably with air temperature in April-September ($R^2 = 0.86$). We applied the relationships shown in Fig. 1.6A to monthly air temperatures for the remaining two model years (2010, 2015) to obtain monthly values of current water use (Fig. 1.6B) and applied those values to each day of each month. For scenarios evaluating future water use (Alternative 2), we assumed that the full municipal water right was exercised, except to the extent that the interruptible portion of the water right could not be exercised due to low river flow.

Big Lake and LaConner wastewater effluent. Treated wastewater effluent from communities of Big Lake and LaConner are released at outfalls at the junction of the North and South Forks of the tidal delta and in Swinomish Channel directly west of Morris Road, respectively. Like Mount Vernon effluent, monthly discharge is reported through the Department of Ecology PARIS website. However, data before 2019 were not readily available for LaConner. Hence, for Big

Lake monthly values from each year were used for daily estimates of each month of the three model years, but for LaConner monthly data from 2019 were used to populate daily estimates from each model year.

Non-municipal senior water rights

When the Skagit IFR was established in 2001, a number of existing water rights in the Skagit delta were senior to the rule and were therefore not subject to interruption by the IFR. We queried the Department of Ecology's water rights dataset (in Yoder et al. 2021) and determined that these comprised a total approximately 90 CFS (Table 1.1)**.** These were a combination of both groundwater and surface water rights, and we assumed based on previous determinations (Savoca et al. 2009) that both types of water withdrawals could equally affect surface water hydrodynamics in channels of the delta. Total senior water rights were apportioned into three reaches (mainstem below Mount Vernon, North Fork, and South Fork), and amount of water use from each reach was determined from locations in the Water Rights database and assigned a specific coordinate for removal (Table 1.2). We assumed that all senior water rights were fully exercised, and we applied seasonal curves of water use from Yoder et al. 2021 (Fig. BP5). to apportion total water rights into separate months, and from there equally into a daily CFS of water use.



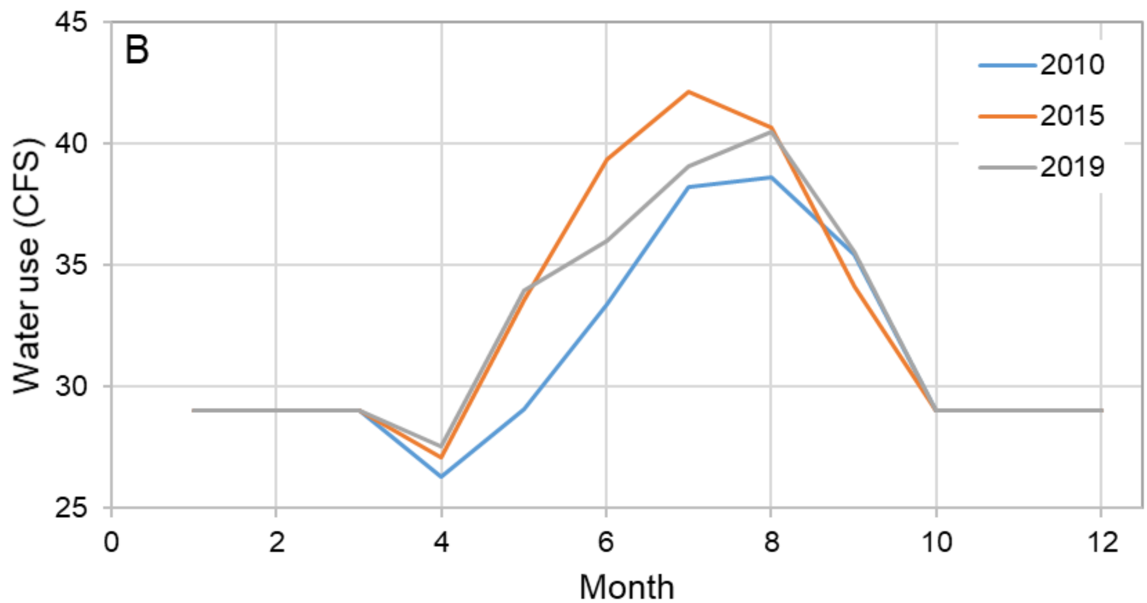


Figure 1.6. A. Monthly values of water use (in CFS) in 2019 as a function of air temperature (NOAA website). Dashed lines depict air temperature relationships used to predict water use as a function of air temperature in months of model years. B. Predicted monthly curves of water use in 2010, 2015, and 2019.

Table 1.1. Summary of the Department of Ecology's Water Rights database, ordered by status relative to the Skagit Instream Flow Rule (Senior or Junior to IFR or a Claim), Phase of right, and whether right is for ground or surface water.

		Qi (total CFS)		**Count**		
IFR Status	**Phase**	**Ground**	**Surface**	**Ground**	**Surface**	**Total records**
Senior	Certificate	64.78	4.31	77	71	148
	New Application	12.94	9.39	14	12	26
	Permit	0.89	--	1	--	1
	Anacortes	--	85.00	--	1	--
	Certificate of Change	--	0.78	--	2	2
	Change-ROE	0.06	--	3	--	3
	Superseding Permit	--	0.81	--	1	1
Junior	Certificate	--	0.98	--	5	5
	Permit	--	72.35	--	12	12
	Anacortes	--	32.30	--	1	--
	Change-ROE	3.24	--	1	--	1
Claim	With Qi	253.15	1438.28	12	20	32
	No Qi	--	--	537	275	812

Junior water rights

The Skagit IFR estimated that 200 CFS junior to the Skagit IFR (i.e., interruptible) were available in WRIA 3 in accordance with the IFR in average years. Based on a search of the Water Rights database, we estimated that 73.3 CFS (surface and groundwater combined) nonmunicipal rights were allocated in the Skagit Delta. As with senior rights, these rights were summed, apportioned into daily units following seasonal patterns of water use (see above), and applied to particular diversion points in the mainstream, North Fork, and South Fork reaches.

Because water rights junior to the IFR are interruptible, we assumed that under current conditions, these uses were "turned off" if flows at Mount Vernon declined below monthly flows specified by the IFR. However, in alternative scenarios, we assumed junior water rights were noninterruptible. The exception to this was the Anacortes junior water right, which we assumed followed the Skagit IFR as required by the regulations. We also assumed that in alternative scenarios, all 200 CFS of "available" water was exercised within the Skagit Delta, even though they were identified for the entire WRIA 3 basin.

Claims and permit-exempt water use

Claims and permit-exempt groundwater use represent two of the more challenging aspects of water use to model. By definition, claims are not recognized as official rights, and of the over 840 claims in the lower Skagit River, 32 have reported amounts of withdrawals. Furthermore, it remains unknown how many claims are currently exercised, as many claims predate the Skagit IFR. Permit-exempt groundwater uses are assumed to be relatively small, but nonetheless can add up to an uncertain cumulative groundwater removal, with uncertain effects on surface flow.

For claims, we scrutinized available data obtained by the Skagit Water Supply and Demand study (Yoder et al. 2021), and compared these data against forms reported on the Department of Ecology water rights record search to obtain amounts of water use in reported claims. In several cases, review of claim amounts revealed discrepancies with the actual claim (based on the original form in the database) reported by the claim holders. Removal of these errors substantially reduced the amount of water use for reported claims. Nonetheless, most claims (Table 1.1) did not have a water use estimate associated with them. In the absence of better information on claims, we assumed that the total water usage from claims was 200 CFS. We assumed that this amount followed seasonal patterns of water use like water rights, but did not change by model year or by scenario.

We did not separately model permit-exempt water use, although the relatively large allowance for claims provides for additional water use that could also comprise permit-exempt water use.

Water use scenarios

To obtain a range of flow conditions in the Skagit delta, we evaluated multiple scenarios of water use in three different model years: the baseline year of 2019 (moderately low flow), the lowest annual flow on record (2015), and average flow conditions (2010). In each of these years, we generated average daily flow conditions based on flow records (in CFS) from the USGS gage (Skagit River at Mt Vernon #12200500), modified by additional flow inputs and simulated exports (i.e., water use) in the Skagit River mainstem and North and South Fork of the Skagit delta.

For each flow year, we simulated four scenarios of water use (baseline: current conditions without water use, current water use, alternative 1, and alternative 2), which model increasing levels of water use within the Skagit River delta (Table 1.3). A final scenario examined an unregulated flow (no dams) scenario for 2010 only. Note that these scenarios were designed to provide a broad range of river flow in the tidal delta based on realistic estimates of water use in

Table 1.2. Modeled water inputs and removals in the Skagit delta, locations in the tidal delta network, and relevant reference for flow data.

Input or removal	Description	Latitude	Longitude	Reference
Mainstem				
Input	USGS gage at Mt Vernon	48.4450	-122.3356	USGS, avg daily flow
Input	Mt Vernon Wastewater Treatment Plant	48.4132	-122.3495	Dept. of Ecology PARIS, avg monthly flow
Removal	City of Anacortes diversion	48.4376	-122.3749	Pers. Comm., total monthly amount
Removal	Water rights senior to IFR	48.3949	-122.3635	
Removal	Water rights junior to IFR	--	--	
Removal	Claims	48.3878	-122.3722	Dept. of Ecology Water Rights, avg daily flow
North Fork				
Removal	Water rights senior to IFR	48.3689	-122.4022	
Removal	Water rights junior to IFR	48.3689	-122.4022	
Removal	Claims	48.3878	-122.3722	Dept. of Ecology Water Rights, avg daily flow
South Fork				
Input	Big Lake Wastewater Treatment Plant	48.3869	-122.3660	Dept. of Ecology PARIS, avg monthly flow
Input	Fisher Creek	48.3217	-122.3450	Statistical estimation, avg daily flow
Removal	Water rights senior to IFR	48.3530	-122.3618	
Removal	Water rights junior to IFR	48.3530	-122.3618	
Removal	Claims	48.3842	-122.3612	Dept. of Ecology Water Rights, avg daily flow
Swinomish Ch.				
Input	LaConner Wastewater Treatment Plant	48.3922	-122.4974	Dept. of Ecology PARIS, avg monthly flow

the context of the Skagit River Instream Flow Rule (WAC 173-503-030), but they were nevertheless hypothetical scenarios. Therefore, they should not be interpreted as estimates of actual water use or formal application of the Skagit's Instream Flow Rule. The combination of water use scenarios and different water years created flow conditions entering the delta ranging from 0% to over 20% of river flow, at worse doubling the level of water withdrawals limited by the Skagit Instream Flow Rule.

No water use scenario

This scenario examined current flow conditions of the Skagit River and Fisher Creek but contains no water withdrawals (or discharges) below the Mount Vernon gage. This is an unrealistic situation, so we did not extensively model consequences of this scenario on fishes and their habitat. As it does represent a logical extreme that has been assumed in previous hydrodynamic models of the Skagit estuary (e.g., Yang and Khangaonkar 2009, Khangaonkar et al. 2016), we nevertheless ran this simulation for each model year, and each scenario is available for comparison (see Appendix).

Current water use scenario

This scenario examined the current situation, in which a variety of water rights senior as well as junior to the Instream Flow Rule (IFR) modify flow on a seasonal and daily basis, with junior agricultural water rights subject to interruption when river flow drops below levels specified by the Skagit IFR.

Alternative 1 - current water use, junior water rights are noninterruptible

This scenario examined a hypothetical situation in which 200 CFS of junior water rights was rendered noninterruptible. This affected all junior water rights except the Anacortes junior water right, because seasonal patterns under current conditions did not exceed the senior water right.

Alternative 2 - future water use, water rights are noninterruptible

This scenario examined a hypothetical situation in which 200 CFS of junior water rights as well as 390 CFS of additional irrigation needs were made uninterruptible. This addition was based on projected additional water needs for agriculture due to changing precipitation and hydrograph conditions (Yoder et al. 2021). We also assumed that the maximum Anacortes water right (85 CFS) was utilized. However, Anacortes' junior water right (32.3 CFS) remained actionable, such that this water right was exercised only on days when Skagit River flows did not drop below the Skagit IFR.

Unregulated flow

This scenario evaluated an unregulated hydrograph, in which the effects of the hydropower system in the Upper Skagit and Baker Rivers were removed via University of Washington's

Distributed Hydrology Soil Vegetation Model (DHSVM, see Yoder et al. 2021). This model simulates the portion of the water cycle that encompasses precipitation, recharge into soil, evaporation/transpiration, and flow of water into rivers (Storck et al. 1998). The model has been used to project changing hydrograph due to climate impacts, and to predict flows under historical, current, or future conditions at various points in the watershed. We used a historically reconstructed run for model year 2010 and used projected average daily flow at Mt. Vernon as an input into the hydrodynamic model. We compared this to the actual 2010 daily flow data at Mt. Vernon. Both model runs assumed no water use, no discharges, nor input from Fisher Creek. As this scenario was also unrealistic, we did not predict fish and their habitat from model outputs.

Table 1.3. Water use scenarios modeled, in terms of total cubic feet per second (CFS) from agricultural sources and the City of Anacortes. Amounts for Anacortes vary in two scenarios based on monthly air temperatures. Though they were incorporated, Anacortes' interruptible water rights were not exercised in two scenarios because uninterruptible water rights were not fully exercised (noted in parentheses).

Scenario	Withdrawal source	Uninterruptible (senior to IFR) water rights (CFS)	Interruptible (junior to IFR) water rights (CFS)	Claims (CFS)
Baseline	Agriculture	0	0	0
	Anacortes	0	0	0
Current water use	Agriculture	90	73.3	200
	Anacortes	Seasonal	(32.3)	200
Alternative 1	Agriculture	290		200
	Anacortes	Seasonal	(32.3)	200
Alternative 2	Agriculture	590		200
	Anacortes	85	32.3	200

Overview of model subcomponents

Below we highlight the primary findings of the subcomponents of our modeling studies. Details on methodologies can be found in subsequent chapters.

Hydrodynamic model

The hydrodynamic model code used for this study was different from the previous SHDM effort (Whiting et al. 2017) conducted with a Finite Volume Community Ocean Model (FVCOM, (Chen et al. 2003)). During our study, the SHDM was significantly improved to accommodate the needs of this project. Specifically, the resolution of the model was greatly increased to include smaller channels in the tidal delta allowing simulation of smaller distributaries and other tidal channels utilized by juvenile salmon, a target of fish use and habitat analyses and data collection efforts in this study. This also allowed changes to the North Fork due to a natural avulsion, including the new flow pathway and shallowing of the historical North Fork pathway to be incorporated into the grid network and bathymetry. Some of these refinements had pushed the FVCOM model beyond its numerical stability limits. Subsequently, the modeling framework was migrated to a new state of the art Semi-implicit Cross-scale Hydroscience Integrated System Model (SCHISM, Zhang et al., 2016) that provided the needed robust performance. SCHISM uses a grid structure similar to FVCOM, allowing a seamless transition.

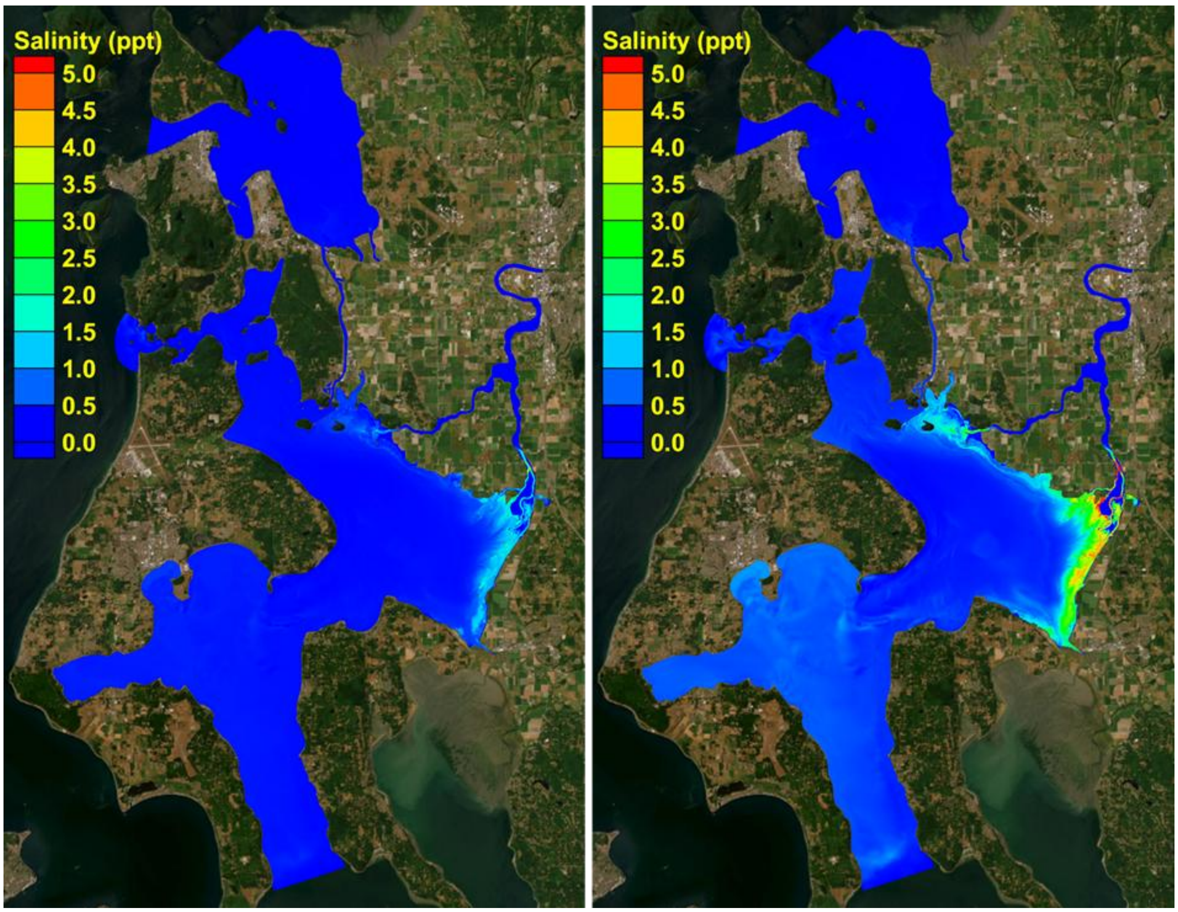


Figure 1.7 (see also Fig 2.19). Changes in maximum surface layer salinity from the current condition to two alternative water withdrawal scenarios (left: Alt1 and Right: Alt2) during May – September (2015).

Results from the model application showed some differences in hydrodynamic variables during the May-September time period in the three model years. While currents and water surface elevations were not greatly different among years, salinity varied by up to 5 ppt in an average year (2010) compared to the lowest flow on record (2015). These differences were concentrated at the North and South Fork mouths entering Skagit Bay (Fig. 1.7). Water use had compounding effects, with Alternative 2 exhibiting much greater differences than Alternative 1 in the spatial extent to which mean and maximum surface salinity levels increased.

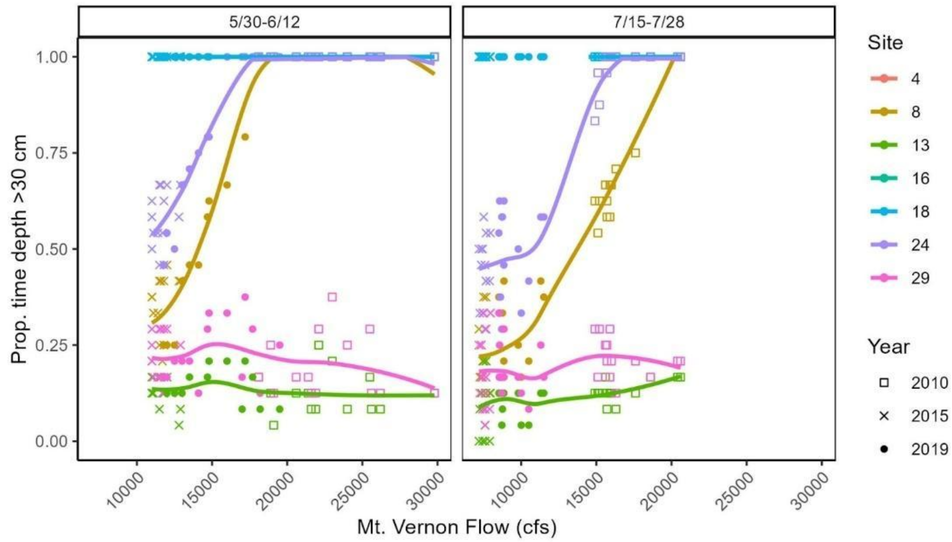


Figure 1.8 (see also Fig. 2.28). The proportion of time in a day that the depth of channel sites exceeded 30 cm compared to flow. Model outputs are based on the current scenario and include all model years. Lines show loess local regression fit to model outputs from the same site.

Despite the lack of strong effects of scenarios on water surface elevation, we nevertheless observed that in tidal channels, salinities declined and water surface elevations increased as a function of river flow (Fig. 1.8). These patterns mirrored findings from the Duke (1999) study, although this pattern was site-dependent, and exhibited nonlinear relationships.

Also of note, modeled water surface elevations and salinity values tended to underpredict observed salinities for the majority of sites. The exceptions to this pattern were those that were the lowest and highest salinities. These patterns suggest that modeled effects of salinity might underestimate the true impacts of changing salinity upon marsh vegetation and juvenile fishes including Chinook salmon.

Marsh vegetation model

Proposed changes in water management, Alternatives 1 and 2, is predicted to have direct impacts on marsh salinity by reducing freshwater delivery to the delta (see previous section). To anticipate potential impacts of water management changes on Skagit tidal marsh vegetation, we developed a predictive statistical model based on salinity and marsh elevation. These are the two most important and fundamental environmental influences on tidal marsh vegetation; they create well-known plant zonation in tidal marshes. Using field-collected data on soil salinity (salinometer), marsh surface elevation and plant species distributions (RTK-GPS with 3-cm horizontal and vertical precision) in the Skagit Delta, we used non-parametric multiplicative regression (NPMR) to model the probability of encountering a locally dominant plant species in a given combination of marsh surface elevation (from lidar data) and salinity (from the hydrodynamic model). NPMR was the chosen approach because plant species abundances generally have non-linear relationships to environmental predictors (i.e., roughly bell-shape

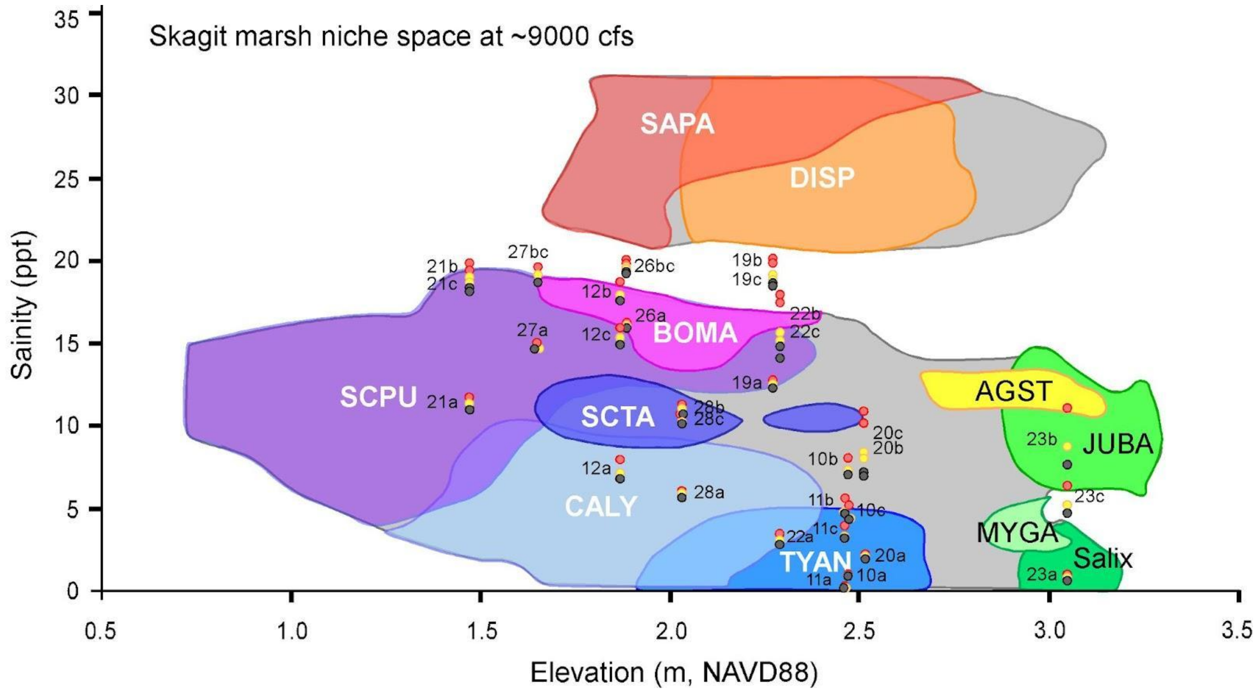


Figure 1.9 (See also Fig. 4.2). Vegetation niche space partitioning with sentinel sites (numbered as in Fig. 4-1), hydrologic year (a = 2010, b = 2015, c = 2019), and management alternatives (black = current conditions, yellow = Alternative 1, red = Alternative 2). Species polygons are bounded by the 30% frequency of occurrence isopleth, i.e., internal areas represent still higher frequencies of occurrence. Other isopleths are not shown to limit graphic complexity. AGST = *Agrostis stolonifera* (bentgrass); BOMA = *Bolboschoenus maritimus* (maritime bulrush); CALY = *Carex lyngbyei* (sedge); DISP =*Distichlis spicata* (saltgrass); JUBA = *Juncus balticus* (Baltic rush); MYGA = *Myrica gale* (sweetgale); *Salix* spp. (willow); SAPA = *Sarcocornia pacifica* (pickleweed); SCPU = *Schoenoplectus pungens* (three-square); SCTA = *S. tabernaemontani* (soft-stem bulrush); TYAN = *Typha angustifolia* (non-native cattail).

curves). Species distributions also respond to interactions between predictors, and while some predictors may be in a favorable range for a species, if even one predictor is in a mortal range (e.g., high temperature or low oxygen for fish, high salinity or low elevation for plants), then that limiting factor negates the positive effects of the other predictors–hence a multiplicative rather than additive model. NPMR is also well suited to presence/absence species data, which is the kind of data that was available.

Model quality assessment was very favorable with AUC (area under the curve) values generally in the excellent range (> 0.90, where 1.00 = perfect discrimination; Çorbacıoğlu & Aksel 2023). Differences between management scenarios were striking. Under normal flow conditions, vegetation changes were slight to undetectable for the water management alternatives (Fig. 1.9). However, under low flow conditions, Alternative 2 caused changes in vegetation composition compared to current water use, with declines in shrub cover of 15% to 27% for some shrub-dominated sites. Shrubs (willow and sweetgale) provide critical habitat for tidal beaver, whose dams provide important low-tide rearing habitat for juvenile salmon (Hood 2012). Changes in various species of herbaceous vegetation for sensitive sentinel sites ranged from 12% to 26%. In contrast, vegetation changes were relatively modest for Alternative 1; only three sensitive sentinel sites showed vegetation changes, which ranged from 3% to 9%. The vegetation change of greatest concern would be the 9% net decline in shrub cover for the Milltown reference shrub site. Summertime low flow will increasingly become normal flow as climate change progresses (Cuo et al. 2011). Thus, the low-flow scenarios are highly relevant to likely future impacts to tidal marsh vegetation.

Models of juvenile fish distributions

Quantitative relationships between species abundances and environmental conditions are often poorly characterized for estuarine fish communities (Elliott & Quintino 2007; Sheaves et al. 2016). This knowledge gap limits managers with appropriate means to balance human-driven habitat modifications (i.e. water withdrawals) with the conservation of ecosystem services and sustainable fisheries (Barbier et al. 2011; Levin & Möllmann 2015). To address this gap, we analyzed long-term monitoring data from the tidal delta and nearshore zones of the Skagit River estuary in Washington State that supports a diverse assemblage of fish species and provides critical nursery habitat for threatened Chinook salmon. We found that abundances of virtually all species were related to depth, salinity, temperature, and velocity, which are local conditions that are influenced by freshwater input. These relationships appeared to reflect species' different life histories and were often nonlinear. Chinook salmon were most abundant in waters of greater depth, lower salinity, intermediate temperature, and lower velocity; therefore, decisions that alter the freshwater flow regime may consider how these local environmental attributes will change and alter salmon access to estuarine nursery habitats.

One effective approach for assessing the ecological consequences of altered freshwater flow regimes on estuarine fishes is to integrate hydrodynamic models with fish–environment relationship models (see Ganju et al. 2024 for a broad review). Specifically, hydrodynamic models can simulate how flow regimes influence local physical conditions within estuaries (e.g. depth, velocity, and salinity), while fish–environment models quantify how fish presence or abundance responds to these environmental gradients. By coupling these two model types, it becomes possible to predict spatial and temporal variation in fish abundance under alternative flow scenarios. Broadly, this work contributes to operationalizing ecosystem-based fisheries management, a framework that strives to sustain fisheries resources by making decisions in light of understanding the ecological interactions and contexts that influence them.

In this study, we developed and then linked fish–environment and hydrodynamic models to evaluate how a range of plausible flow regimes may influence the fish assemblage in the Skagit River estuary. Although river flow was not specifically incorporated into fish models, projections from models were nonetheless sensitive to river flow. We found that under current conditions, the occurrence of juvenile Chinook salmon tended to increase with average daily flow of the

Skagit River, and this pattern was particularly strong in the summer. In contrast, effects of tidal range were less consistent. For both predictors, associations with daily flow were nonlinear and spatially variable.

As expected, predicted changes on fish distributions were most pronounced during drier water years and under scenarios involving increased water withdrawals, particularly in areas where environmental conditions were more sensitive to freshwater inputs. Juvenile Chinook salmon exhibited reduced predicted habitat use during dry years, with further reductions under scenarios simulating water use (Fig. 1.11). The magnitude of reduced habitat use, relative to current conditions, was generally modest (on the order of several percentage points), but potentially ecologically meaningful. Additionally, flow regime changes influenced predicted fish assemblage composition, likely reflecting species-specific responses to shifts in salinity – the primary variable that differed across scenarios in the hydrodynamic model.

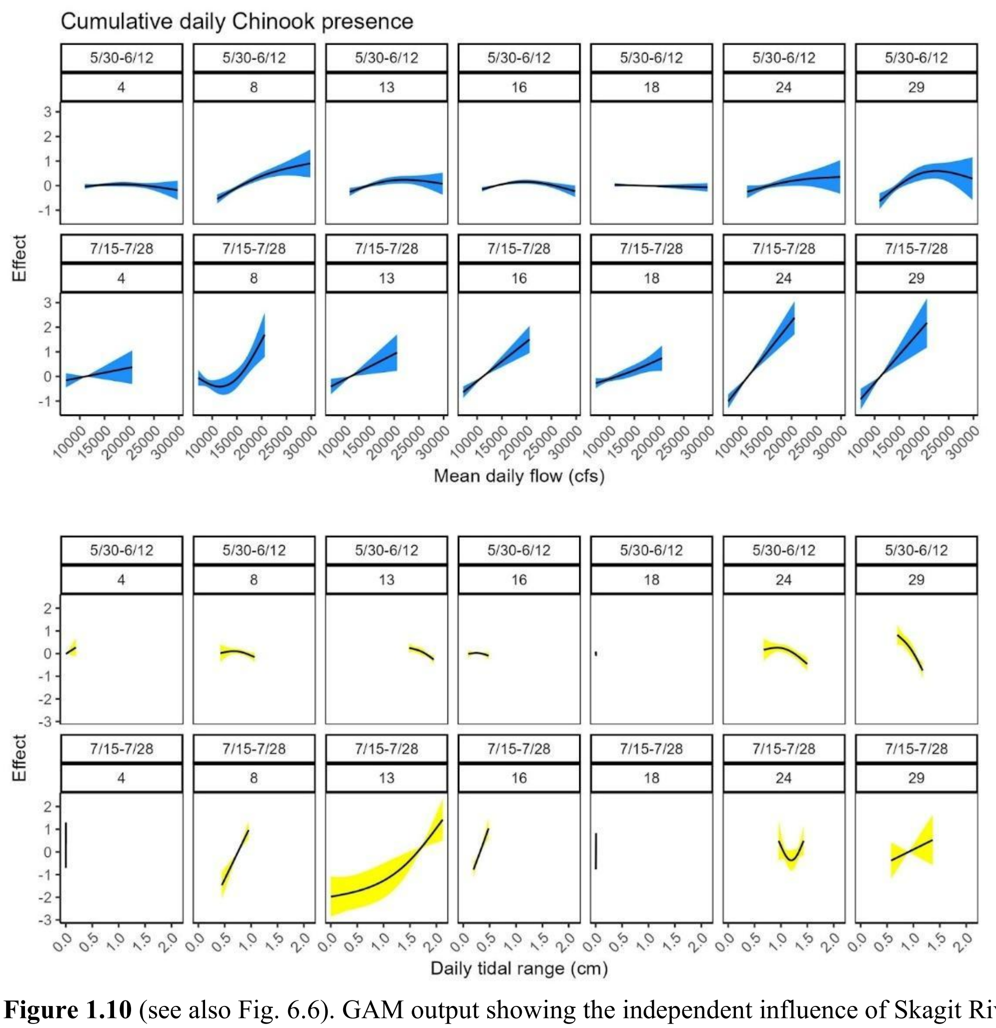


Figure 1.10 (see also Fig. 6.6). GAM output showing the independent influence of Skagit River flow (top panels) and daily tidal range (bottom panels) on Chinook salmon presence throughout a day. The first and third row of panels represent late spring (5/30-6/12), and the second and fourth rows represent mid-summer (7/15-7/28). Individual panels identify patterns at numbered sites. Shaded areas indicate 95% confidence intervals.

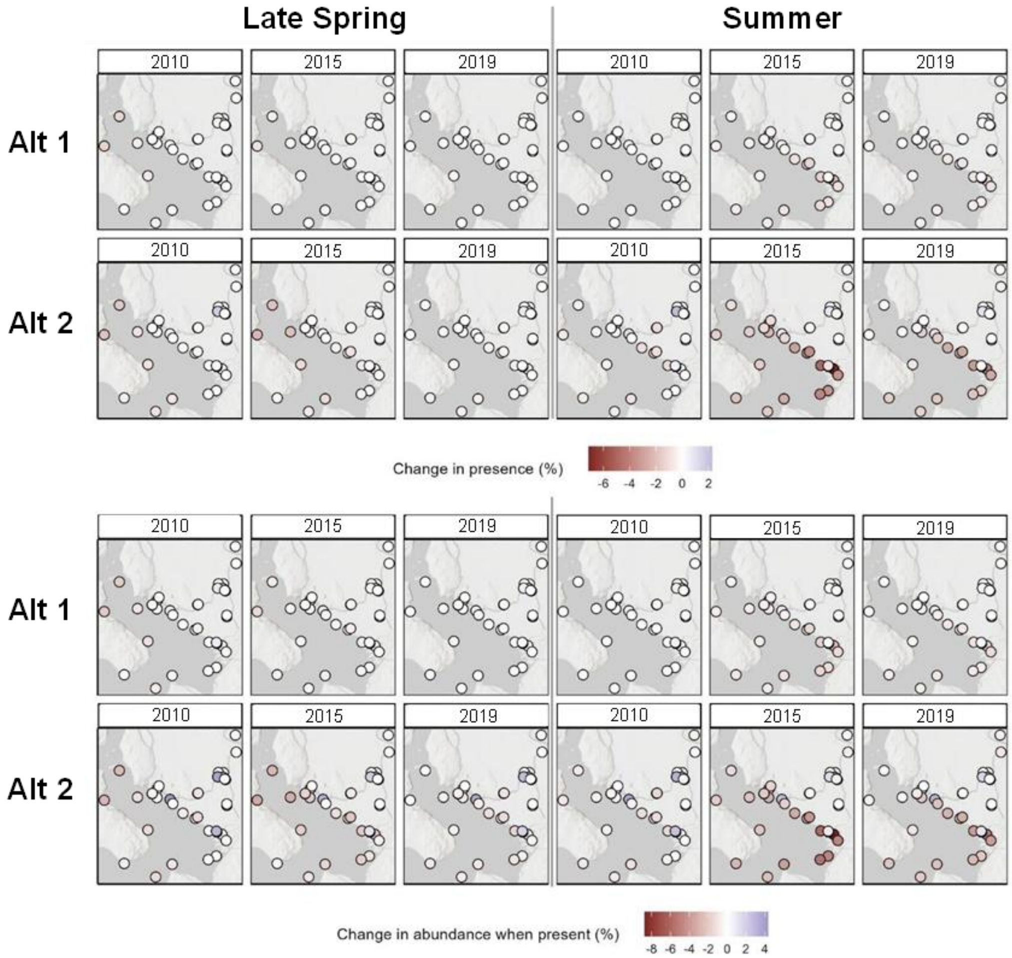


Figure 1.11 (See also Fig. 6.3). Changes in Chinook presence (top panels) and abundance when present (bottom panels) in water use scenarios Alternative 1 (Alt 1) and Alternative 2 (Alt 2) for late spring (5/30-6/12) and summer (7/15-7/28) in each of three model years. Values are from predictions of a given year and scenario minus predictions from the same year's Current Water Use scenario.

Key conclusions

Our report provides several conclusions relevant to the influence of river flows on estuarine hydrodynamics, marsh vegetation, and estuarine fishes. Water surface elevation is one facet of how freshwater influences inundation in tidal deltas. Our study provides evidence that confirms this aspect of the Duke Engineering study (1999). However, it appears from hydrodynamic model results that this general rule applies to certain locations in the tidal delta along the freshwater-marine gradient, and it also appears that the relationship between river flow and water elevation is nonlinear. In addition to water surface elevation, salinity emerged as an additional important metric influencing both the distribution of marsh vegetation and key estuary-dependent fishes including juvenile Chinook salmon. The daily duration of salinity < 5 ppt also was correlated with river flow at Mount Vernon. Of all hydrodynamic metrics, lower salinity was the best metric to distinguish higher quality sites in our simulations. Changes in currents and water surface elevation were much less apparent among scenarios.

From vegetation modeling we determined that many of the simulations resulted in relatively little change in the predicted occurrence of the main wetland species. Modeled water use scenarios resulted in changes in vegetation only under high water scarcity (low river flow, high water use). Generally, estuarine emergent marsh sites were not sensitive to changes in these scenarios, but scrub-shrub species, with much lower salinity tolerances, showed much higher sensitivity. The larger effects tended to be associated with the South Fork of the Skagit delta where the hydrodynamic model predicted greater changes in salinity.

We also found that juvenile Chinook salmon were also resilient to many of our model runs, but both presence and abundance were reduced under scenarios with high water scarcity, particularly during mid-summer. Generally, greater differences occurred among water years compared to water use scenarios, although their interaction was apparent. Like effects on vegetation, these changes were most prevalent in the South Fork of the tidal delta where a number of habitat restoration projects were implemented.

Habitat restoration in tidal deltas predictably benefits juvenile Chinook salmon (Greene et al. 2024, Greene et al. 2025). Multiple model subcomponents of the current study demonstrated that increases in salinity across water use scenarios and alternate model years had the worst effects on the South Fork area of the Skagit delta, an area that has been subject to the most restoration in the Skagit. This points to the importance of considering multiple aspects of natural resource management (Munsch et al. 2020) that could affect responses of fish populations to management actions such as habitat restoration and changes in water use.

Like the hydrodynamic modeling, we found qualified support for the Duke study's (1999) conclusions on linear effects of water use on fish habitat. Our fish-environment model predicted relatively linear reductions in daily presence by juvenile Chinook salmon during the summer as a function of daily river flow, although there was also clear evidence that tidal variation also affected these patterns. Likewise, the flow-Chinook presence relationship is nonlinear and varied among the sites examined in our study.

Uncertainty in models

The conclusions from Chapters 2-4 generally conclude that higher water scarcity (low-flow hydrograph, high water use) produces predictable changes in salinity, vegetation, and juvenile Chinook salmon and other fishes, particularly in the southern portions of the Skagit delta and bay. However, hydrodynamics, vegetation, and fish distributions were generally resilient to changes from current water use and minor increases (e.g., Alternative 1). In Table 1.4, we rank our confidence in our findings, much like the Intergovernmental Panel on Climate Change does for climate impacts (very low - very high, Masandrea et al. 2010), based on “validity of findings as determined through evaluation of evidence and agreement” (Masandrea et al. p. 3) and the potential for other factors to alter the result. While we have medium to high confidence in most of our results (See Table 1.4), the absolute effect of water scarcity was in some cases fairly subtle and location-dependent. In addition, all of our findings depend upon uncertainties in our models. Model uncertainty can occur for a number of reasons, and we divide these into uncertainties within the current model (e.g., data issues, model framework, poor assumptions), compounded across models (e.g., use of outputs from one model to project another’s), and extrinsic factors not yet considered.

As shown in Table 1.4, all of our findings include uncertainty in each of these components. Some of these issues may increase our uncertainty of conclusions, while others are likely directional, i.e., will increase our confidence one way or the other. For example, while we took pains to improve the bathymetric profile underlying the hydrodynamic model, there are likely errors across the study area, but that error is expected to be bi-directional. Similarly, changes to water management above Mt. Vernon (particularly in hydropower operations) may increase or decrease our confidence of changes in salinity, depending upon how modified hydropower operations influence the timing of flows. However, effects of sea level rise are likely to increase our confidence in impacts to changes in salinity. We likewise have medium to high confidence in changes to vegetation in portions of the tidal delta, although these conclusions depend to some extent on the limited number of variables considered when constructing niche dimensions. Incorporation of extrinsic factors such as sea level rise will likely make us more confident of salinity-based impacts. Examining effects of water scarcity on juvenile Chinook salmon and other fishes, we have medium to high confidence in the flow-dependent relationships and effects of water scarcity on presence, but lower confidence on changes in Chinook abundance due to potential density dependence and changing growth/migration rates. We have low confidence in expected changes in the entire fish community, due to the many possible species-specific responses, lack of inclusion of potential interspecific interactions, and subtle, location-specific shifts in the community.

Study limitations

This study had several limitations that may constrain the interpretation of its findings. The hydrodynamic model did not account for subsurface/groundwater processes or temperature changes, which can significantly influence vegetation or fish distributions, respectively. Additionally, it did not evaluate potential changes in hydrodynamics driven by climate change, such as altered hydrographs, increased water temperatures, or sea level rise. Despite efforts to include smaller channels in the model grid, the model nevertheless better represents larger channels and may overlook critical habitats used by species like Chinook salmon. The tidal vegetation model was not explicitly linked to the fish models and considered only elevation and salinity, omitting other important factors such as soil chemistry and substrate porosity.

Furthermore, predicted changes in vegetation or fish distributions should not be interpreted as immediate or absolute, as they can be modulated by precipitation variability, density dependence, predation, and other ecological effects not captured in the models. Other models such as bioenergetic and life cycle models might be appropriate for examining longer-term effects of flow reductions on salmon populations, although great uncertainties remain in the mechanisms linking environment to demographic parameters.

Future research

The above limitations point to a number of opportunities for future modeling that can further help address some of the uncertainties in how water use in the Skagit influences the ecology of the tidal delta and Skagit Bay nearshore. In particular, we recommend the following research directions to support environmentally sustainable water use in the Skagit estuary.

Hydrodynamics: A key component to model in future iterations of the hydrodynamic model is temperature dynamics, including how temperature is affected by freshwater inputs and the wetting and drying cycle in vegetated and unvegetated tidal delta wetlands. Once temperature is integrated, modeling the cumulative effects of climate change, water use, and habitat restoration will be valuable for understanding directional ecological change in the tidal delta and nearshore.

Marsh vegetation: We found strong delineation across the marsh as defined by elevation and salinity. Whether hydrodynamic changes resulting from lower river flow are likely to result in vegetation shifts depends greatly on the duration and extent of saltwater stress. Addressing how much stress is needed to result in directional change as predicted from our models will help improve understanding of the relationship between water use and distribution of shrubs and emergent marsh vegetation in the Skagit's estuarine wetlands.

Fish distribution, growth, and survival: Our study focused on producing models of fish distributions, and did not infer changes in growth or survival that could conceivably be outcomes of changing salinity and temperature regimes. Bioenergetics models will better incorporate temperature effects, but it is clear from our modeling results that salinity changes may also be important in understanding fish distributions. While large sudden changes in salinity can be detrimental to focal species such as juvenile salmon, it remains unclear how changes in salinity on the order of what was observed in this study (2-5 ppt) affect fish. Assuming such changes can at least have temporary effects on physiology and behavior, understanding how changes in salinity can affect bioenergetics would help improve understanding of the combined effects of flow alterations on hydrodynamic parameters to which juvenile fishes are sensitive.

Table 1.4. Confidence the major findings of this report following guidelines of the IPCC (**), and examples of sources of uncertainty related to our confidence in each finding. Within-model uncertainties include data quality, model choice and framework, and model assumptions, model-compounding uncertainties include dependencies on inputs from other models, and extrinsic factors are current or future conditions that were not considered.

		Uncertainty		
Major findings	**Confidence in result**	**Within-model**	**Model compounding**	**Extrinsic**
Hydrodynamics (Ch. 2)				
Minimal changes in WSE with increasing water scarcity	High	• Delta bathymetry	• Marine boundary conditions • Tidal simulation	• Fetch effects • Sea level rise
Increases in salinity with increasing water scarcity	High	• Delta bathymetry	• Tidal simulation	• Changing hydrograph • Water management above Mt. Vernon
Flow-dependent inundation duration	Medium	• Influence of vegetated marsh	• Tidal simulation	• Fetch effects • Sea level rise
Flow-dependent duration of low salinity	Medium	• Influence of vegetated marsh	• Tidal simulation	• Changing hydrograph • Water management above Mt. Vernon
Vegetation (Ch. 3)				
Changes in estuarine-emergent marsh with increasing water scarcity	Medium	• Species-specific sediment types • Density dependence	• Hydrodynamic salinity projection	• Fetch • Sediment porosity • Sea level rise
Changes in scrub-shrub with increasing water scarcity	High	• Density dependence • Microtopography	• Hydrodynamic salinity projection	• Sediment porosity • Sea level rise

Fishes (Ch. 4)				
Changes in Chinook presence with increasing water scarcity	High	• Density dependence • Connectivity	• Temperature dependence • Hydrodynamic projections	• Marsh vegetation, • Changing hydrograph • Sea level rise
Changes in Chinook abundance with increasing water scarcity	Medium	• Density dependence	• Temperature dependence • Hydrodynamic projections	• Marsh vegetation • Water management above Mt Vernon • Changing hydrograph • Sea level rise • Predators
Changes in fish community with increasing water scarcity	Low	• Density dependence • Autocorrelation among species	• Hydrodynamic projections	• Marsh vegetation • Changing hydrograph • Sea level rise • Predators
Flow dependent cumulative presence of Chinook salmon	High	• Connectivity	• Hydrodynamic projections	• Water management above Mt Vernon • Changing hydrograph • Sea level rise

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2. Simulating hydrodynamic changes from water withdrawals in the Skagit River

Taiping Wang and Tarang Khangaonkar

Introduction

Skagit Bay and its delta, located in northern Puget Sound, Washington State, comprise one of the most ecologically significant estuarine systems on the U.S. West Coast. The Skagit Bay and Delta system includes tidal marshes, mudflats, estuarine channels, and eelgrass beds. These diverse habitats provide essential nursery, foraging, and migratory habitat for numerous species, including endangered Chinook salmon, steelhead, migratory shorebirds, and waterfowl.

Over the past century, extensive diking, drainage, and land conversion for agriculture and development have significantly reduced the Skagit Delta's historical estuarine wetlands. In response, regional conservation and climate resilience efforts have focused on restoring tidal connectivity, sediment transport, and estuarine function to support fish and wildlife and enhance floodplain resilience. Many wetland restoration projects have been implemented or proposed, including Deepwater Slough, Fisher Slough, Rawlins Road, Wiley Slough, and the Fir Island restoration projects. These efforts aim to re-establish tidal exchange, restore distributary channels, and promote marsh formation and long-term ecosystem sustainability.

To support these restoration efforts, a series of hydrodynamic modeling studies have been undertaken over the past decades. For instance, Yang et al. (2006) introduced a three-dimensional Finite Volume Coastal Ocean Model (FVCOM) for the Skagit River Delta, focusing on Fir Island, a historically diked and subsided area. Yang and Khangaonkar (2006) also applied the model to the Rawlins Road site, examining dike modifications and channel diversions. Their results revealed that reconnecting channels, such as Hall Slough, could enhance nearshore salinity regimes and improve marsh habitat quality without compromising agriculture. This FVCOM-based model framework was later expanded to the middle Skagit floodplain to evaluate integrated flood flows and tidal hydrodynamics across the Skagit River–Bay system (Yang et al., 2012).

Scientists at the Salish Sea Modeling Center (SSMC) and the Pacific Northwest National Laboratory (PNNL) further refined the earlier Skagit hydrodynamic model in the Skagit Bay and Delta to a minimum grid resolution of ~10 m at selected places and applied to evaluate more than 20 proposed restoration projects (sites) under the Skagit Delta Hydrodynamic Modeling project (Whiting et al. 2017, Khangaonkar et al. 2017). This newly refined Skagit Delta Hydrodynamic Model (SHDM) is a three-dimensional unstructured grid coastal circulation model capable of providing dynamic oceanographic information such as water surface elevation

(WSE), currents, salinity, and bed-shear stress at a sufficiently high resolution and broad scale necessary to support assessments of interactions between river flow, tides, and different restoration actions and their cumulative effects on the nearshore habitat. This is an improved version of the hydrodynamic model of the Skagit River estuary previously developed by Yang et al. (2006) and Yang and Khangaonkar (2006) based on the same finite volume community ocean model (FVCOM) code (Chen et al. 2003), which solves the three-dimensional momentum, continuity, temperature, salinity, and density equations in an integral form by computing fluxes between non-overlapping, horizontal, and triangular control volumes.

The current study builds upon and extends the earlier SDHM by Whiting et al. (2017) in the Skagit Delta. The primary objective was to utilize the enhanced SDHM model to evaluate the impacts of water withdrawals on water levels, salinity, and velocity within salmon habitats in Skagit Bay and Delta, under various climate, hydrological, and water withdrawal scenarios. We continued using an unstructured-grid-based hydrodynamic modeling approach but transitioned the modeling framework from FVCOM to a newly developed, general-purpose hydrodynamic model, the Semi-implicit Cross-scale Hydroscience Integrated System Model (SCHISM, Zhang et al., 2016) to enable higher-resolution modeling with greater computational efficiency. The detailed modeling methodology and analysis of results are presented in the following sections.

Methods

Overview of the hydrodynamic modeling approaches

The hydrodynamic modeling task was conducted as a step-by-step process, in which we continued improving the existing SHDM summarized in the following steps.

Hydrodynamic model implementation with only the minimum necessary improvements to the prior FVCOM-based model framework.

Based on the earlier model grid in Whiting et al. (2017), the grid in the North Fork avulsion channel of Skagit River and adjacent portions of the North Fork was refined and updated using the newly surveyed channel bathymetry data collected by the U.S. Army Corps of Engineers (USACE) in 2022. Following the prior model configuration, new model simulations were set up and conducted using FVCOM for the Year of 2019, during which new field observations of water level and salinity were collected by the SRSC and made available for our use. Model predictions of water level and salinity were subsequently compared with field observations to evaluate the model performance. The results showed that the model calibration had deviated and was not able to reproduce field observations at most field observation sites. A closer examination showed that this was due to the channel migration and sedimentation near the mouth of the North Fork of Skagit River. This suggested that the model grid resolution and bathymetry needed to be systematically improved to reflect the latest topo-bathymetric changes in the lower Skagit system. Refinement was also necessary to resolve the small tidal channels in the intertidal zones to incorporate habitats and channels used by fish and where field observations were mostly collected for analyses as part of this study.

Improvements and refinements of the hydrodynamic model grid.

The earlier modeling work by Whiting et al. (2017) primarily focused on the general barotropic responses (e.g., water level, inundation depth, and bed shear stress) of the Skagit Delta under various wetland restoration scenarios. This approach did not require the model to have a highly detailed representation of small-scale features, such as narrow drainage channels only a few meters wide within the intertidal zones. However, these small channels play a critical role in maintaining hydrological connectivity between the main river, marshes, tidal flats, and Skagit Bay. This connectivity modulates salt transport and influences plant zonation, benthic habitats, and nursery grounds for fish and invertebrates, which are the focus of this study. In addition, there have been broader changes in the topo-bathymetry of the Skagit system since the 2017 study, in which the model grid was based on topo-bathymetry datasets collected prior to 2014. Therefore, it was deemed necessary to systematically refine and improve the hydrodynamic model grid over the entire model domain to improve model performance and representation of the hydrodynamic processes in the system using the available most recent topo-bathymetry datasets.

In this step, the entire model grid was significantly refined, with the finest grid resolution reaching approximately 3 meters (based on triangular grid side length) in small tidal channels. This refinement greatly improved water level simulation results at field observation sites. However, we encountered substantial challenges in using the FVCOM-based hydrodynamic model framework to simulate salinity transport in the intertidal zones with the highly refined grid. These challenges included significantly reduced computational speed due to the default explicit numerical scheme, as well as frequent model instability caused by extensive wetting and drying in the intertidal areas. Given the project requirement to efficiently complete more than 10 full-year model simulations, it was determined that a more computationally efficient and robust hydrodynamic model should be used instead for the proposed hydrodynamic modeling task.

Implementation of a SCHISM-based hydrodynamic model framework to simulate hydrodynamic responses to water withdrawal scenarios.

After a thorough review of the available hydrodynamic model options, we concluded that the SCHISM model (Zhang et al., 2016) was the best suited model for this study. Like FVCOM, SCHISM is an open-source, 3-D hydrodynamic modeling system designed for simulating barotropic and baroclinic circulation across a wide range of spatial and temporal scales. Built on an unstructured grid framework, SCHISM is highly flexible and efficient, allowing accurate representation of complex geometries such as estuaries, deltas, rivers, coastal zones, and the open ocean. It employs a semi-implicit time-stepping scheme and a hybrid vertical grid system (sigma-Z) to ensure stability and computational efficiency, even under highly dynamic conditions involving frequent wetting and drying. SCHISM is widely used in research and management applications, including hydrodynamic circulation, water quality, and ecological modeling (Ye et al., 2018 and 2020; Tian et al., 2024). Compared to FVCOM, SCHISM employs a similar unstructured-grid framework but offers greater efficiency and stability due to its semi-implicit numerical scheme. In this step, we converted the FVCOM-based Skagit Delta hydrodynamic model into SCHISM and used this new model framework to accomplish the proposed modeling work. Therefore, only the final, SCHISM-based modeling approach and results will be presented in this report.

Model grid refinement

The final model grid is shown in Figure 2.1. This grid covers similar areas as the earlier version (Whiting et al., 2017), which include Skagit Bay, Saratoga Passage, Deception Pass, Guemes Channel, Swinomish Channel, and the southern portion of Padilla Bay. The upstream river boundary ends at Skagit River at USGS Mount Vernon gage. Important topo-bathymetric features such as roads, levees, dikes, and jetties are explicitly resolved in the model grid. Compared to the earlier grid, this grid is further refined in the intertidal zones and channels, especially those narrow channels to improve interconnectivity. In addition, recently restored wetland project sites such as Fir Island Farm and Fisher Slough sites are included in the grid. However, some hydraulic structures such as tide gates could not be included in this refined model grid. These structures should be incorporated in future work to better represent realistic flow conditions in the upstream portions of distributaries (e.g., Brown Slough and Wiley Slough).

The model grid bathymetry is based on a combination of data sources. Specifically, the 1-m spatial resolution topo-bathymetric model of Puget Sound developed by the USGS (Tyler et al., 2020) was used as the primary source. This dataset combines data collected between 1887 and 2017 and provides seamless coverage of the model domain at an extremely high spatial resolution. In the intertidal zones and the floodplain, a more recent lidar DEM dataset collected by NOAA (NOAA, 2019) was used. Another major update is the avulsion channel in the North Fork of Skagit River. Although the avulsion channel was included in the earlier SDHM model grid representing 2014 conditions, it was in the early stages of development at that time and has since grown significantly. In the spring of 2022, USACE conducted a detailed bathymetric survey of the avulsion channel and portions of north fork Skagit River upstream and downstream of the opening. All these datasets clearly show that the channel has continuously migrated westward over time. Therefore, the model grid in the avulsion channel and adjacent sections of the North Fork channel was updated to reflect the latest conditions captured by the USACE survey. We recognized that these bathymetric datasets were collected in different years and contain inconsistencies due to the highly dynamic nature of the Skagit River delta; however, they represent the best available data sources for this application. The final model grid bathymetry is shown in Figure 2.2.

Figure 2.1. The refined hydrodynamic model grid used for this study.

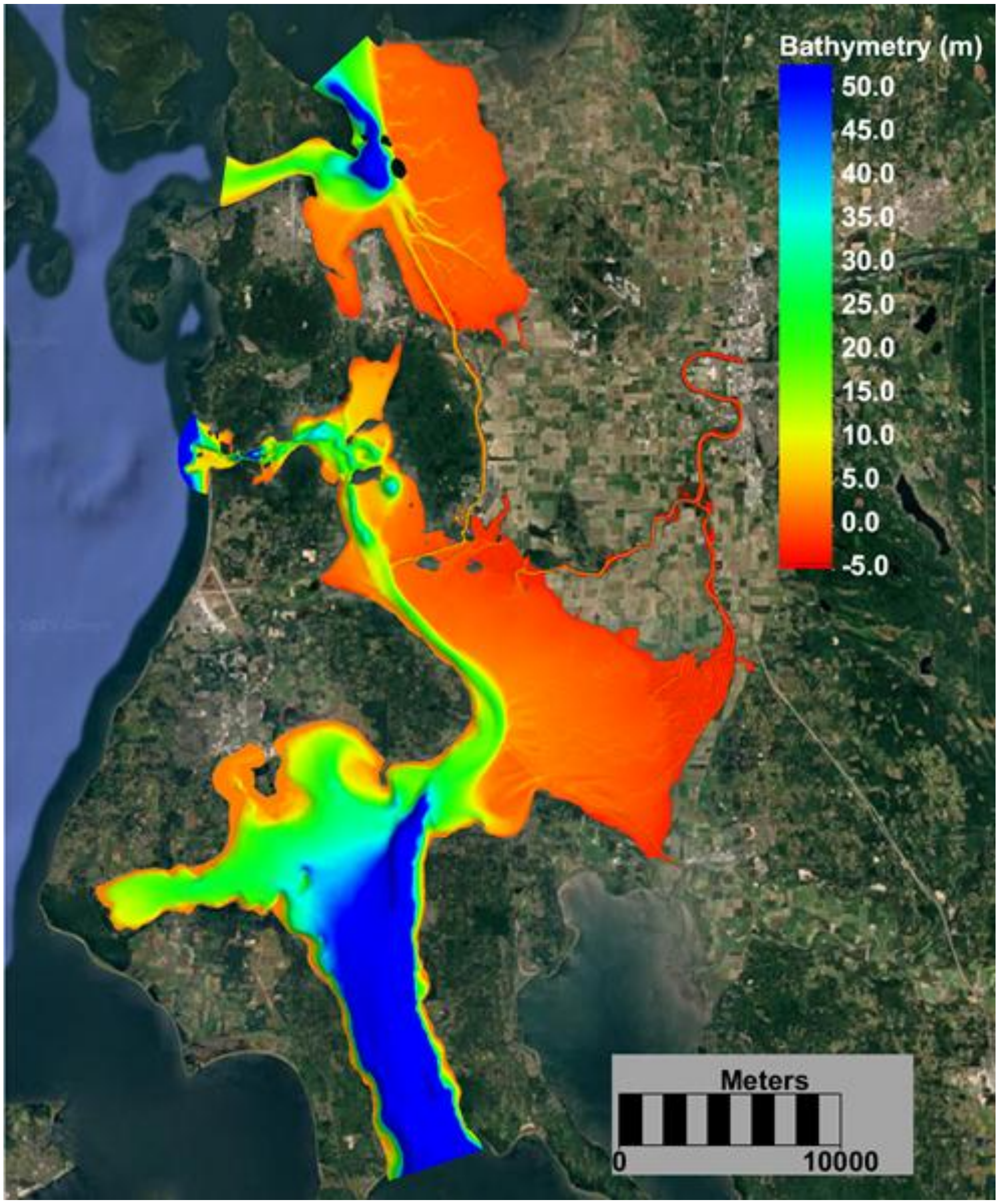

Figure 2.2. The corresponding hydrodynamic model grid bathymetry.

The model grid represents the highest resolution yet produced by the SSMC modeling team for the Skagit River delta. It consists of 334,048 triangular elements and 171,406 nodes—more than double the number in the previous version of the SDHM grid described in Whiting et al. (2017). Grid resolution (measured by the length of triangle sides) ranges from approximately 3 meters in narrow tidal channels to around 400 meters at the open boundaries. Figure 2.3 presents a comparison between the newly refined grid and the earlier SDHM grid, clearly illustrating substantial improvements in spatial resolution and topo-bathymetric representation. Figure 2.4 provides a close-up view of the North Fork avulsion channel area and the Fir Island wetland restoration project site, where narrow tidal channels are well captured by the model grid.

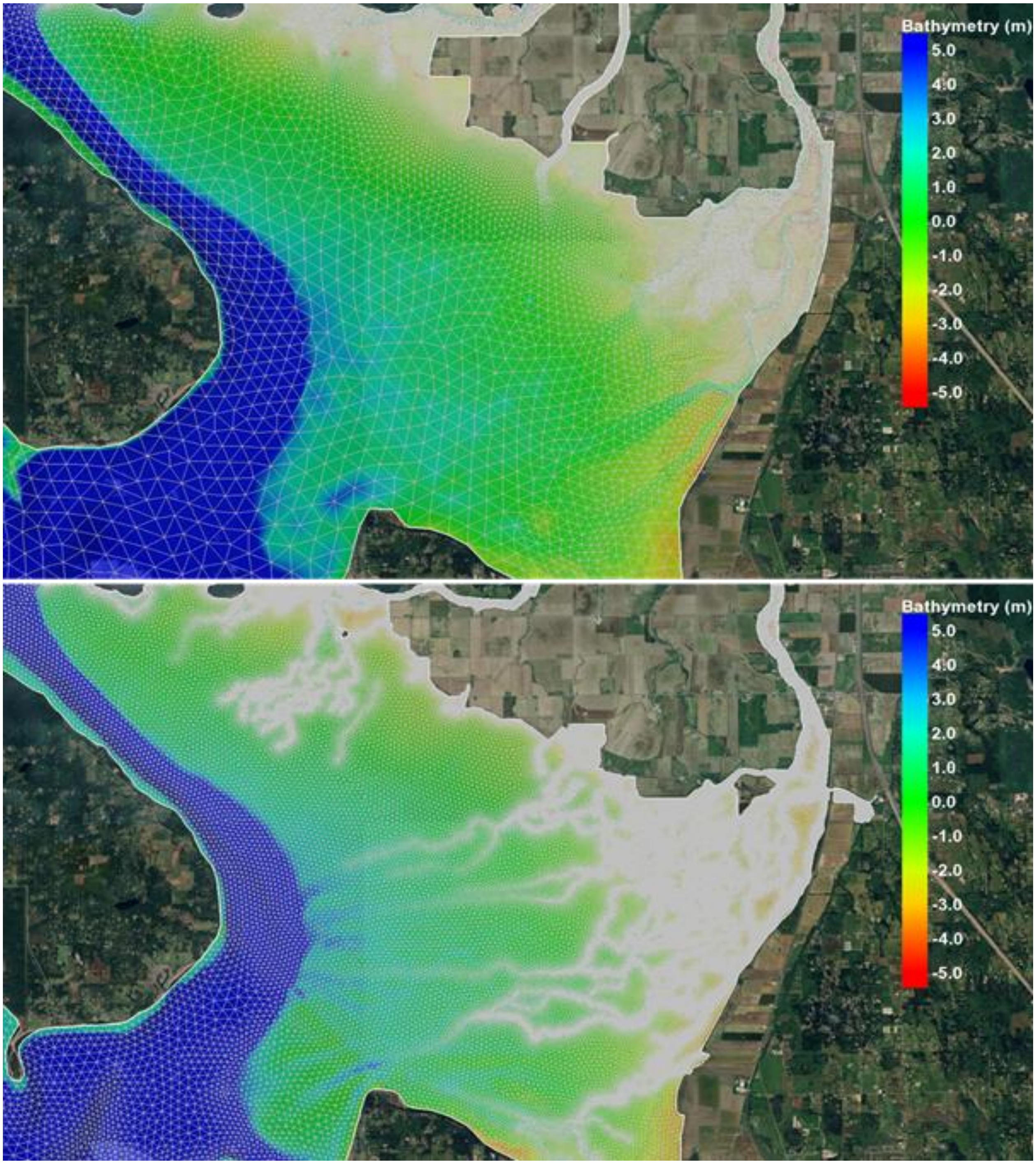

Figure 2.3. A comparison of the hydrodynamic model grid used in the earlier SDHM study (top) and in this study (bottom).

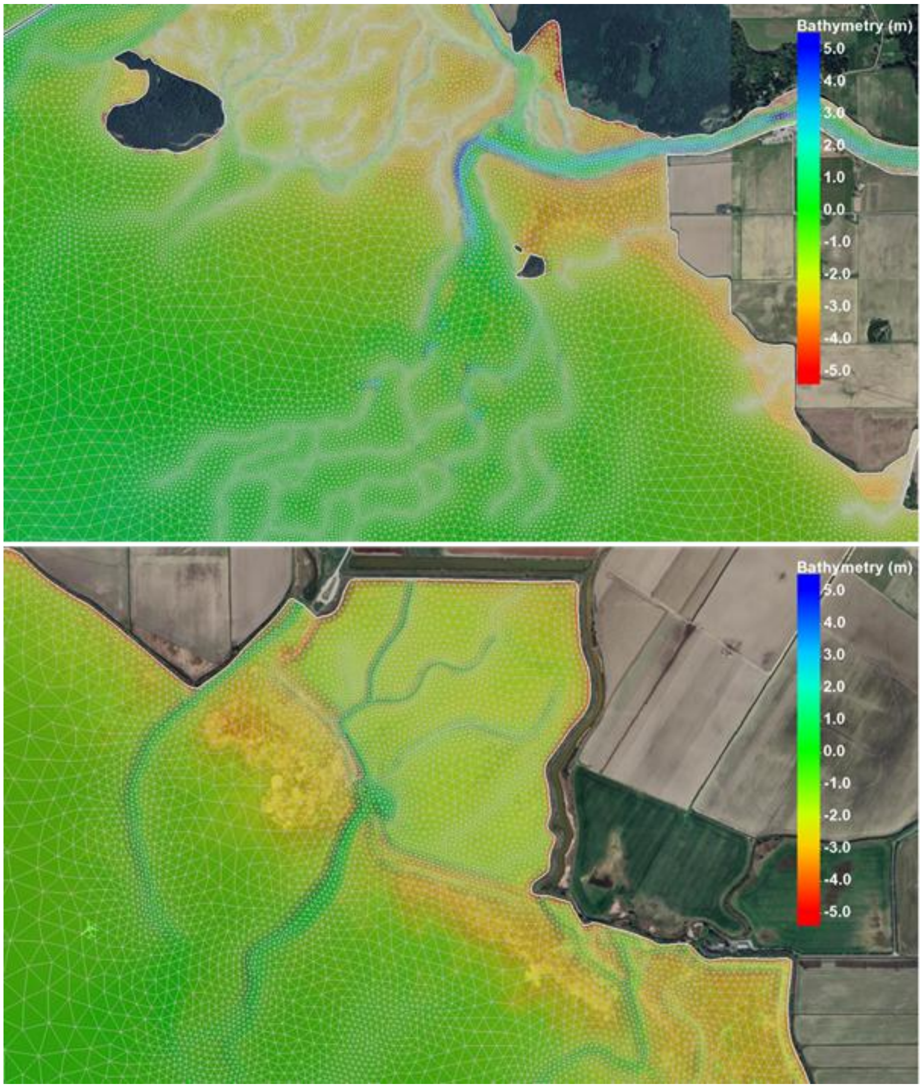

Figure 2.4. Zoom-in view of the refined hydrodynamic model grid in the North Fork Avulsion Channel (top) and Fir Island restoration project site (bottom).

Model configuration

Following the earlier model configuration, the four open boundaries were driven by tidal elevations predicted by the online XTide program (https://tide.arthroinfo.org/). Tidal elevations were specified at the following four open boundaries: (1) middle of Padilla Bay – Chuckanut Bay station, (2) Guemes Channel – Anacortes station, (3) Deception Pass – Bowman Bay station, and (4) Saratoga Passage – Greenbank station. The locations of these four XTide stations and an example plot of XTide-predicted water level time-series over a 10-day period in 2019 are shown in Figure 2.5. Tidal variation differs substantially among the four open boundary locations, with the southern boundary at Green Bank exhibiting the highest tidal range. For open boundary salinity profiles, they were interpolated onto each open boundary node and vertical level from the hourly output of the Salish Sea Hydrodynamic Model hindcasts (Khangaonkar et al., 2017 and 2019).

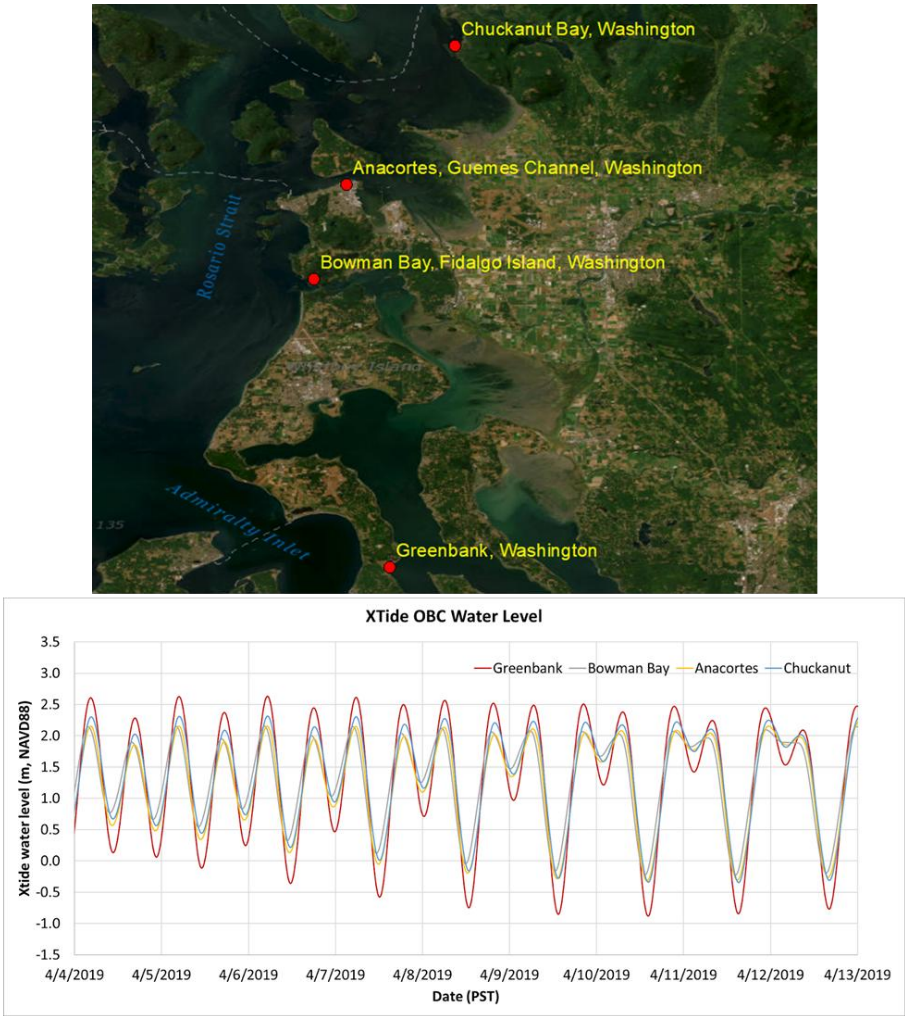

Figure 2.5. Map of the four XTide station locations used for model open boundary conditions (top panel) and example plot of water level time series (bottom panel).

In the vertical direction, a hybrid Sigma-Z coordinate system was used, which combines the flexibility of terrain-following (sigma) layers with the stability and efficiency of Z-level (fixed-depth) layers. Specifically, a total of six sigma levels were used to cover the surface layers for depths up to six meters and another six Z-levels were used to cover the bottom layers where water depths are greater than six meters. This method not only allows shallow waters and intertidal zones to be accurately represented by terrain-following sigma levels at sufficient vertical resolutions, but also enhances computational efficiency, accuracy, and stability in deeper regions using fixed-depth Z-levels. Furthermore, by using SCHISM, we were able to reduce the minimum water depth criterion for determining wetting and drying in the intertidal zones from 10 cm in FVCOM (Whiting et al., 2017) to 1 cm in this study. This significantly improved the model's performance in intertidal zones. For bottom friction, spatially varying bottom roughness heights were specified at each grid node and adjusted through model calibration. A default value of 0.001 m was found to work well for most of the domain. In areas covered by intertidal marshes, a higher value of 0.05 m was used to represent the increased drag caused by marsh vegetation.

Surface wind forcing also contributes to hydrodynamic circulation and mixing. Hourly wind forcing output obtained from the Climate Forecast System Reanalysis (CFSR; Saha et al., 2010) and the Climate Forecast System Version 2 (CFSv2, Saha et al., 2014) were used. Both the CFSR and CFSv2 products have been used by the hydrodynamic modeling team in the past for a variety of modeling applications and demonstrated a high level of accuracy (Wang et al., 2018).

Skagit River discharge is the primary freshwater source into the model domain. The flow has been continuously measured by the USGS stream gauge 12200500 near Mount Vernon. Discharge from Fisher Creek is also included in the study for the sake of completeness despite a much smaller flow rate. As described in the previous chapter, the focus of this study is the effect of water withdrawal from the Skagit River. Thus, a total of 14 water discharge and withdrawal locations were considered in the model, which include Skagit River and Fisher Creek discharge (Sites 1 and 11 in Figure 2.A1, respectively) as default. Besides, three separate flow years (2019, 2015, and 2010) corresponding to three representative flow conditions (moderately low, historic low, and average) were selected for model simulations. In conjunction with three water withdrawal conditions (Current water use condition, Alternative 1 – current water use condition with junior water rights are non-interruptible, Alternative 2 – future water use condition, in which water rights are non-interruptible), a total of 17 scenario runs were set up for the hydrodynamic model. Table 2.A1 summarizes the 17 scenario runs conducted in this study. Using Year 2019 as the example, Run 1 refers to the typical model configuration that only considers USGS gaged Skagit River discharge as the sole river input to the hydrodynamic model. By neglecting other minor freshwater input (e.g., from Fisher Creek) and any additional water withdrawals (to be determined during the course of this study), this simplified model configuration allowed us to quickly set up and calibrate the hydrodynamic model. Run 2 represents an improved baseline condition in which both Skagit River and Fisher Creek discharges are considered. Run 3 represents a more realistic baseline condition, in which additional flow input and water withdrawals are considered in the model based on the best estimates of the current water use conditions. Runs 4 and 5 are two additional sensitivity scenarios, in which two alternative water use conditions are considered. For detailed descriptions on water withdrawal estimates, please refer to Chapter 1.

For consistency, all model runs were configured in the same way and executed on the same high-performance computer using the same set of nodes. This ensures that results from each scenario run for the same year are directly comparable.

Model Validation and Result Analysis

Following the model grid update, a validation step was conducted to ensure that the model reproduced observed data in the estuary at an acceptable level of skill. The year 2019 was selected as the calibration year because it reflects more recent conditions and offers a rich set of field observations from SRSC and WDFW. These high-frequency data were collected using in situ HOBO data loggers deployed at a number of well-selected, representative monitoring sites across the Skagit Delta for periods of at least several months, depending on site-specific and sensor conditions. The sensors were regularly cleaned and inspected to ensure proper operation in the field. Specifically, they were mounted a few centimeters above the sediment bed to minimize sedimentation while remaining submerged as consistently as possible. The sensors recorded water level, salinity, and temperature at 15-minute intervals. At the end of the deployment, the data were downloaded and examined for quality assurance and control before being shared with the modeling team.

Figure 2.6 shows the field observation sites maintained by SRSC and WDFW in the Skagit Delta. All stations are in intertidal zones subject to frequent wetting and drying, which presents a significant challenge for hydrodynamic modeling, particularly in scalar transport (e.g., salinity) simulations. Example water level observations over a 10-day period in April 2019 are shown in Figure 2.7. These time-series exhibit strong spatial variability, with the lower bounds primarily governed by local topography. Model validation was carried out by iteratively tuning key parameters such as the time step and bottom friction. In addition, the model grid bathymetry was adjusted as needed around the observation sites, using observed water level data as a guide. This was necessary due to limitations in the topo-bathymetric datasets, which may not accurately capture the bottom elevations of narrow channels and often contain errors in vegetation-covered regions. Commonly used performance metrics, e.g., bias, the Pearson correlation coefficient (R), and root mean square error (RMSE), were calculated to quantify the model's skill in predicting water levels and salinity compared to field observations.

To analyze and compare model-predicted hydrodynamic responses under different water withdrawal scenarios, time series, 2D contour, and cumulative frequency plots were used. In addition, a generalized additive model (GAM), a flexible regression approach that allows for nonlinear relationships between predictors and response variables, was employed to assess the effects of flow and tidal range on the duration of inundation and threshold salinity.

Figure 2.6. Field water level and salinity observation sites used for model validation.

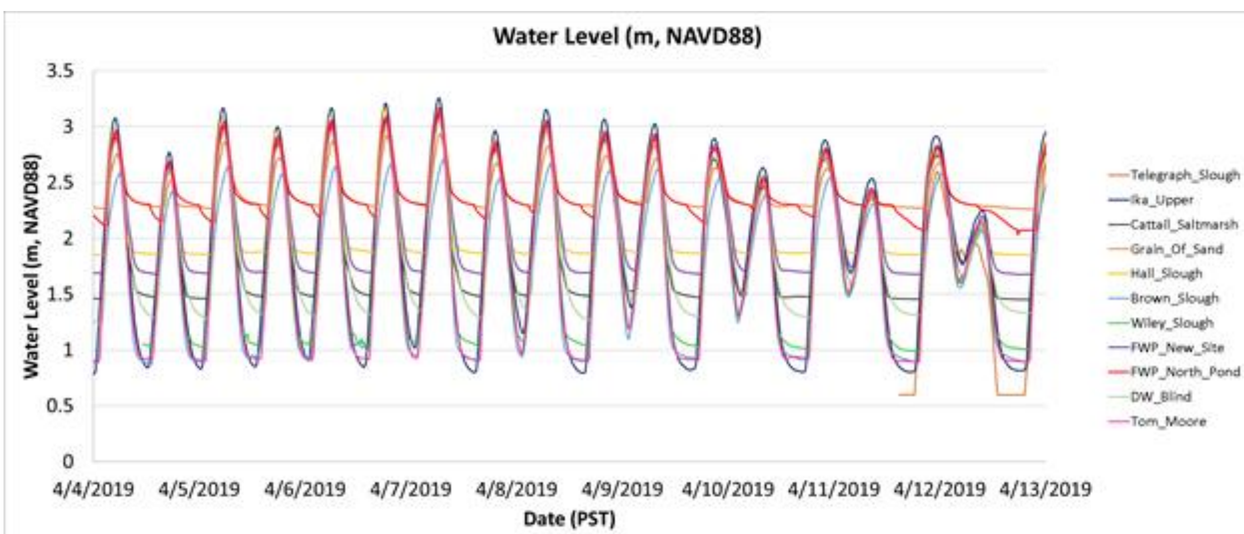

Figure 2.7. A 10-day time series plot of the water level observations in April 2019.

Results

Model validation

The model-predicted water surface elevation and salinity time series for the year 2019 were compared with field observations. Results at representative sites within the model domain are shown in Figures 2.8 and 2.9, and the associated error statistics for all sites are summarized in Table 2.1. Overall, the model performed well in simulating water level variations at most sites, as evidenced by the time series comparisons and error statistics (e.g., >0.9 correlation coefficient values). Given the large tidal range, the relative error (RMSE/Tidal Range) of 6% to 19% among

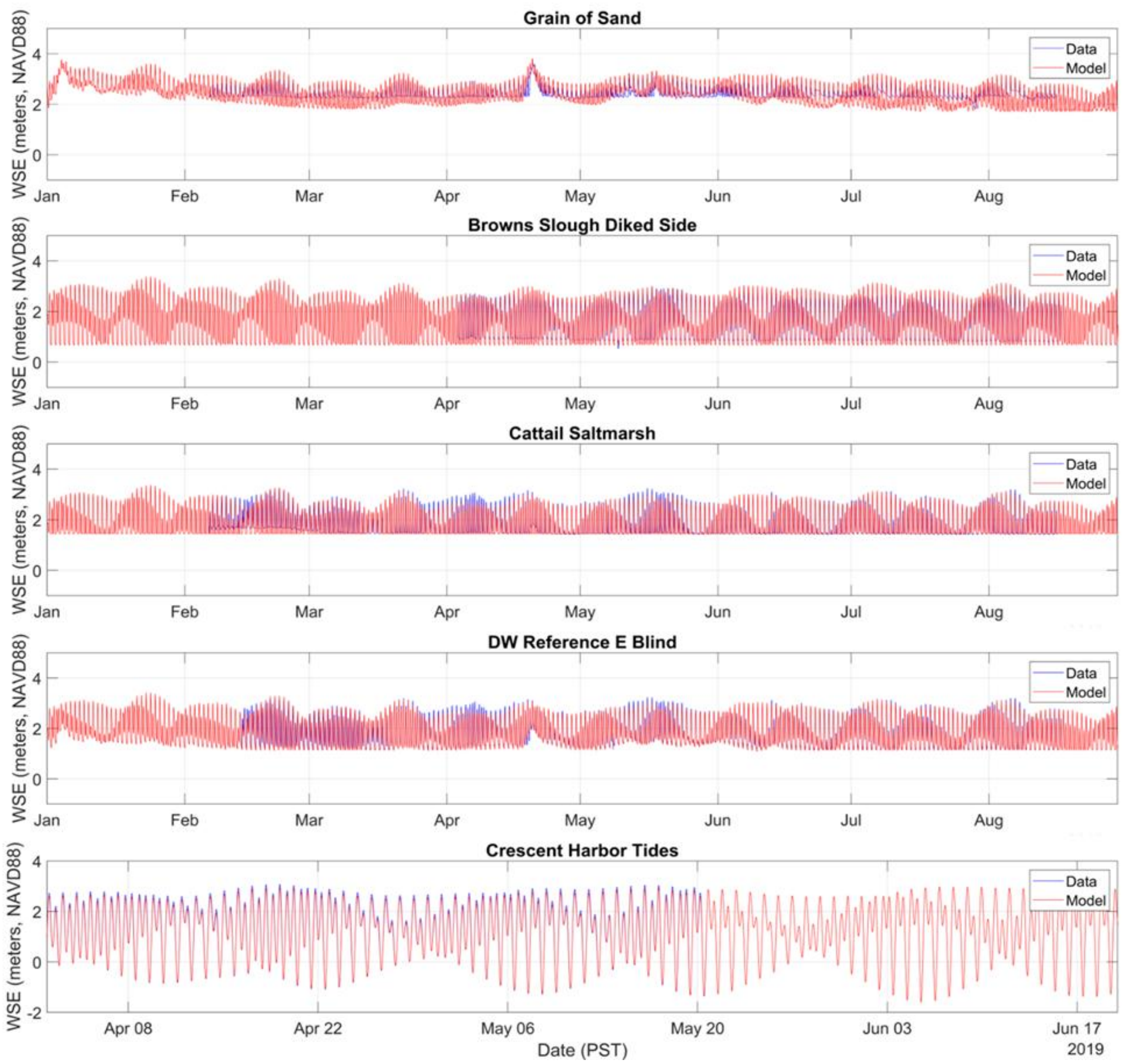

Figure 2.8. Model-data comparisons of water level time series at selected data logger sites. The last plot (Crescent Harbor) is compared against NOAA tidal predictions at Crescent Harbor station inside Skagit Bay to confirm that the model is fully capable of reproducing water levels inside Skagit Bay.

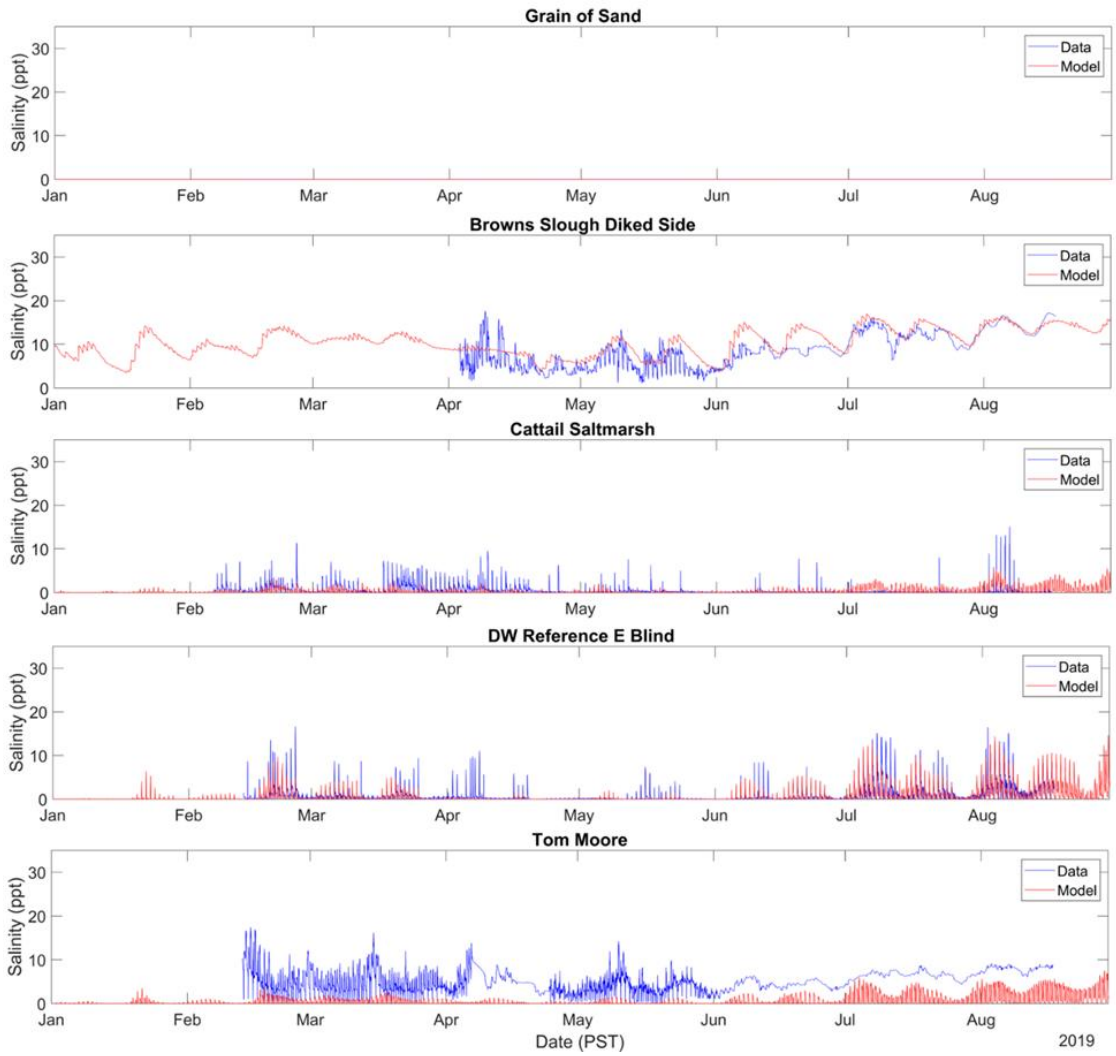

Figure 2.9. Model-data comparisons of salinity time series at selected data logger sites. Note "Grain of Sand" is an entirely freshwater site, so shown as 0 salinity in both model predictions and field observations.

all stations is considered acceptable. In contrast, accurately capturing salinity variations at these intertidal sites remains challenging, as salt transport is highly sensitive to topo-bathymetric features at both local and domain-wide scales, as well as to localized freshwater inputs from minor ditches or creeks (e.g., the Wiley Slough and Telegraph Slough sites).

Hydrodynamic effects of water withdrawals

To investigate the effects of water withdrawal on key hydrodynamic parameters in the Skagit Bay system, we analyzed changes in water surface elevation, velocity, and salinity under various water withdrawal scenarios. The analysis revealed that salinity – particularly in the surface layer – is the most sensitive parameter to reductions in freshwater input caused by water withdrawals. In contrast, the effects on water surface elevation and velocity were comparatively minor and generally negligible. This was expected, as in locations influenced by tides, reduced freshwater

Table 2.1. Model validation error statistics.

Station	Water Level (m)			Salinity (ppt)		
	Bias	RMSE	R	Bias	RMSE	R
Browns Slough Diked Side	-0.02	0.22	0.96	1.75	2.87	0.82
Cattail Saltmarsh	-0.07	0.17	0.95	-0.16	1.02	0.20
DW Reference E Blind	-0.04	0.15	0.96	-0.30	1.24	0.75
FWP N Pond	0.15	0.23	0.77	-2.59	5.46	0.53
FWP New Site	-0.08	0.16	0.95	-1.76	2.78	0.70
Grain of Sand	-0.05	0.28	0.72	-0.02	0.02	1.00
Hall Slough Tribe	-0.09	0.15	0.94	0.51	3.40	0.40
Ika Upper	-0.22	0.28	0.97	-2.41	6.58	0.42
Telegraph Slough Blind	-0.04	0.11	0.99	3.72	11.13	0.13
Tom Moore	-0.05	0.20	0.96	-4.58	5.07	0.34
Wiley Slough	-0.09	0.32	0.89	-2.31	2.74	0.50
Crescent Harbor	-0.07	0.09	1.00	NA	NA	NA

input due to water withdrawals can be quickly offset by incoming ocean water. However, since ocean water has a much higher salinity than freshwater, this results in more pronounced changes in salinity.

Figures 2.10 through 2.21 illustrate domain-wide changes in the maximum and mean surface salinity fields under two alternative water withdrawal scenarios (Alternative 1 and Alternative 2) compared with the Current Water Use scenario, across three different simulation years. In particular, the critical period from May 1 through September 30 was selected for this analysis, as it corresponds to water withdrawal activities during the dry season. The results clearly demonstrate that increases in both maximum and mean surface salinity are strongly influenced by the volume of water withdrawn and the hydrologic conditions of each year. Specifically, Alternative 1 in 2010 (an average flow year) produced the smallest increase in surface salinity, whereas Alternative 2 in 2015 (a low-flow year) led to the largest increase. The South Fork delta appears to be the most sensitive region, with maximum surface salinity increases exceeding 5 ppt during the May–September period in 2015 (Figure 2.19).

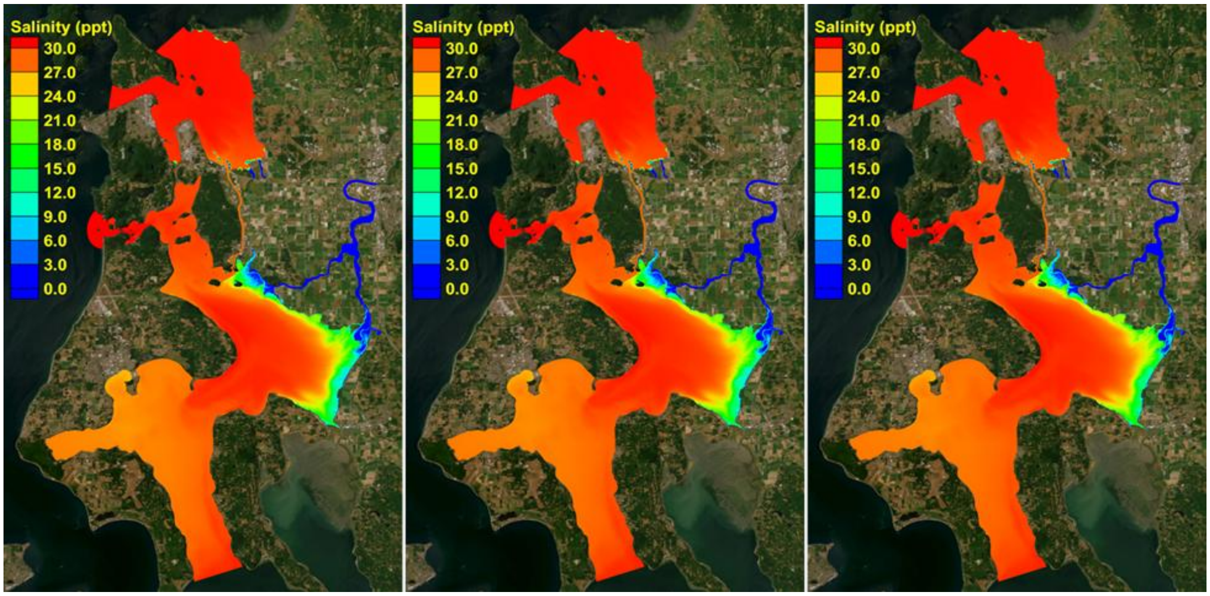

Figure 2.10. Maximum surface layer salinity during May – September (2019). From left to right (Current Water Use, Alt1, and Alt2).

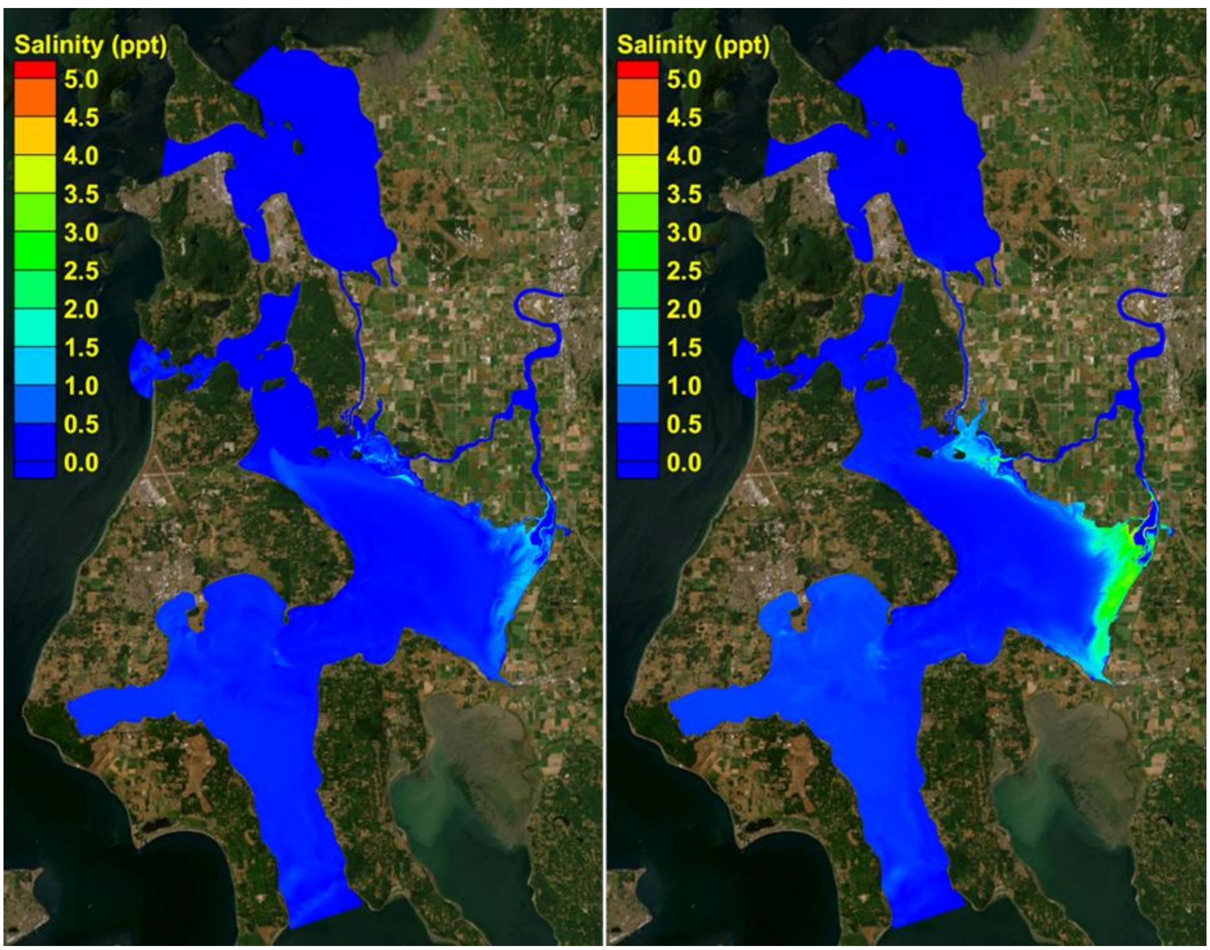

Figure 2.11. Changes in maximum surface layer salinity from the current water use condition to two alternative water withdrawal scenarios (left: Alt1 and Right: Alt2) during May – September (2019).

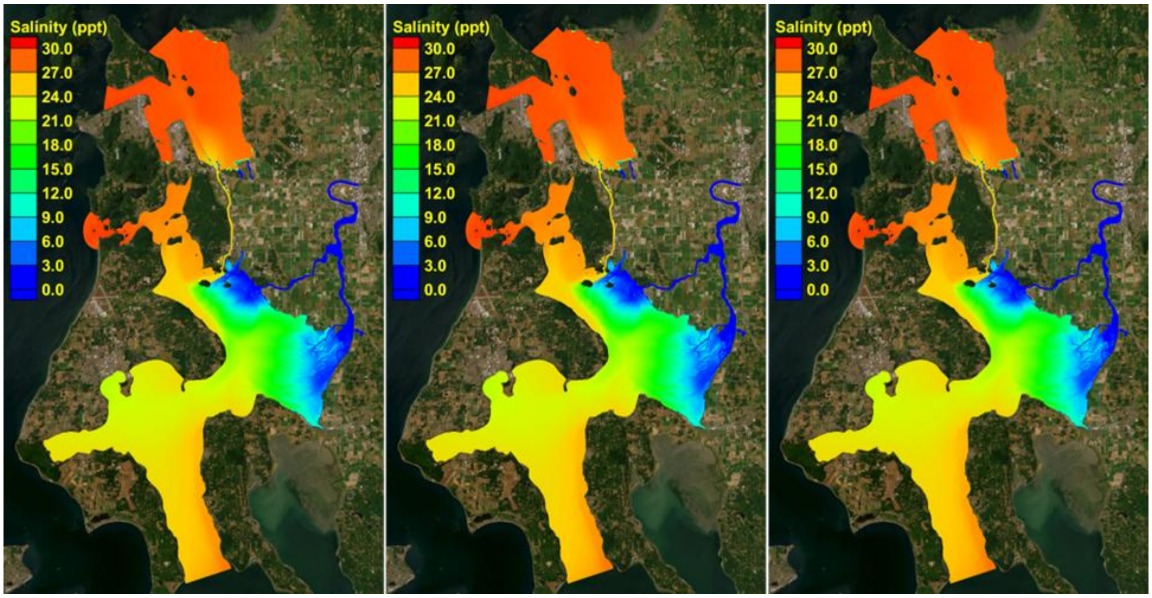

Figure 2.12. Mean surface layer salinity during May – September (2019). From left to right (Current Water Use, Alt1, and Alt2).

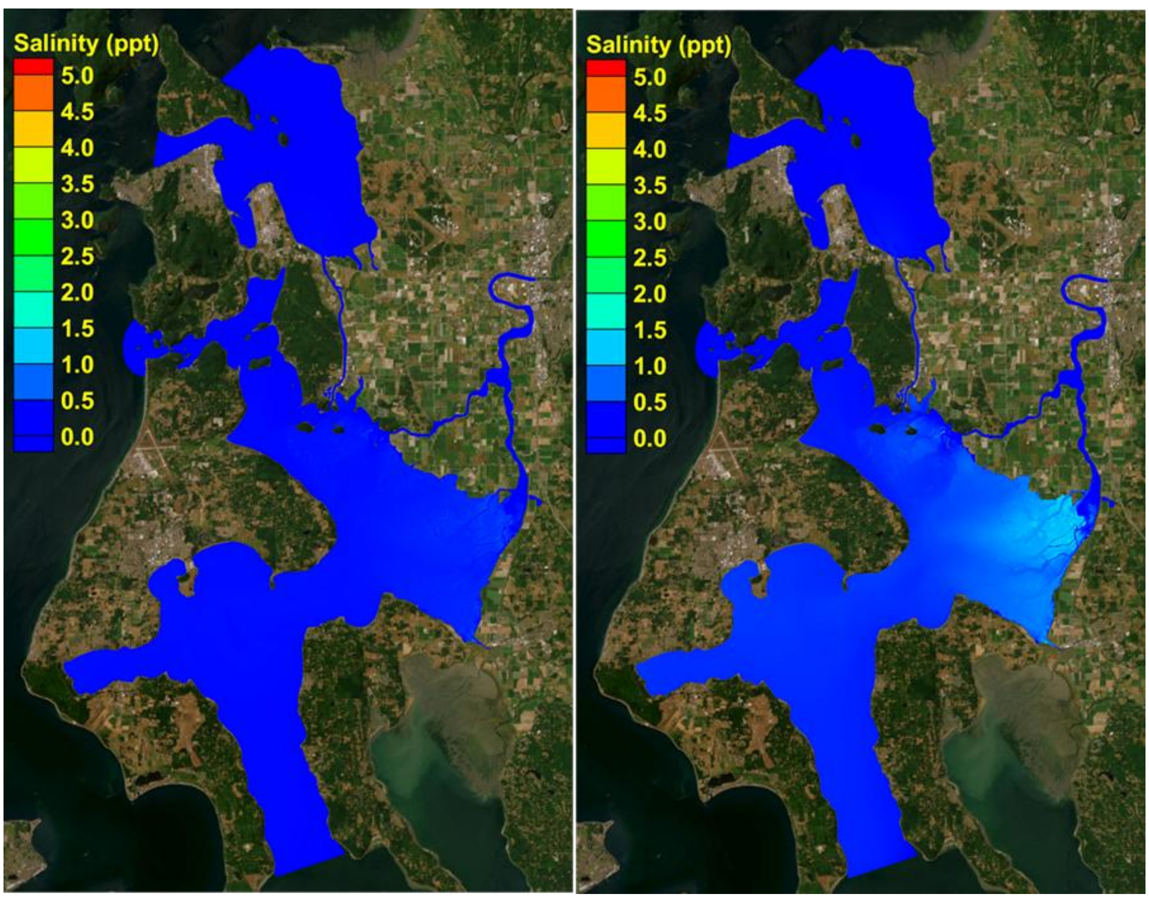

Figure 2.13. Changes in mean surface layer salinity from the current water use condition to two alternative water withdrawal scenarios (left: Alt1 and Right: Alt2) during May – September (2019).

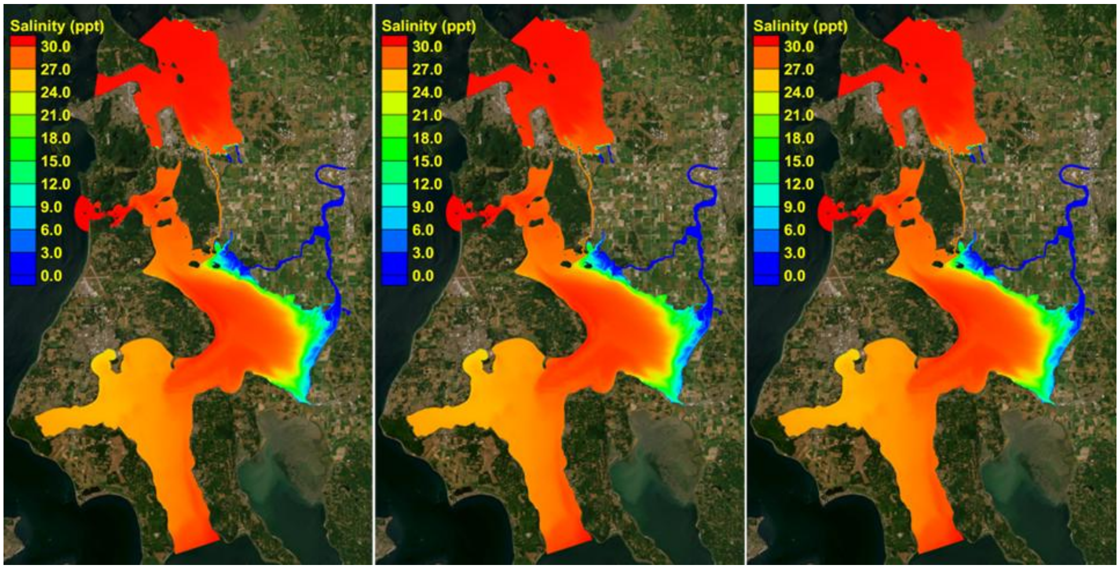

Figure 2.14. Maximum surface layer salinity during May – September (2010). From left to right (Current Water Use, Alt1, and Alt2).

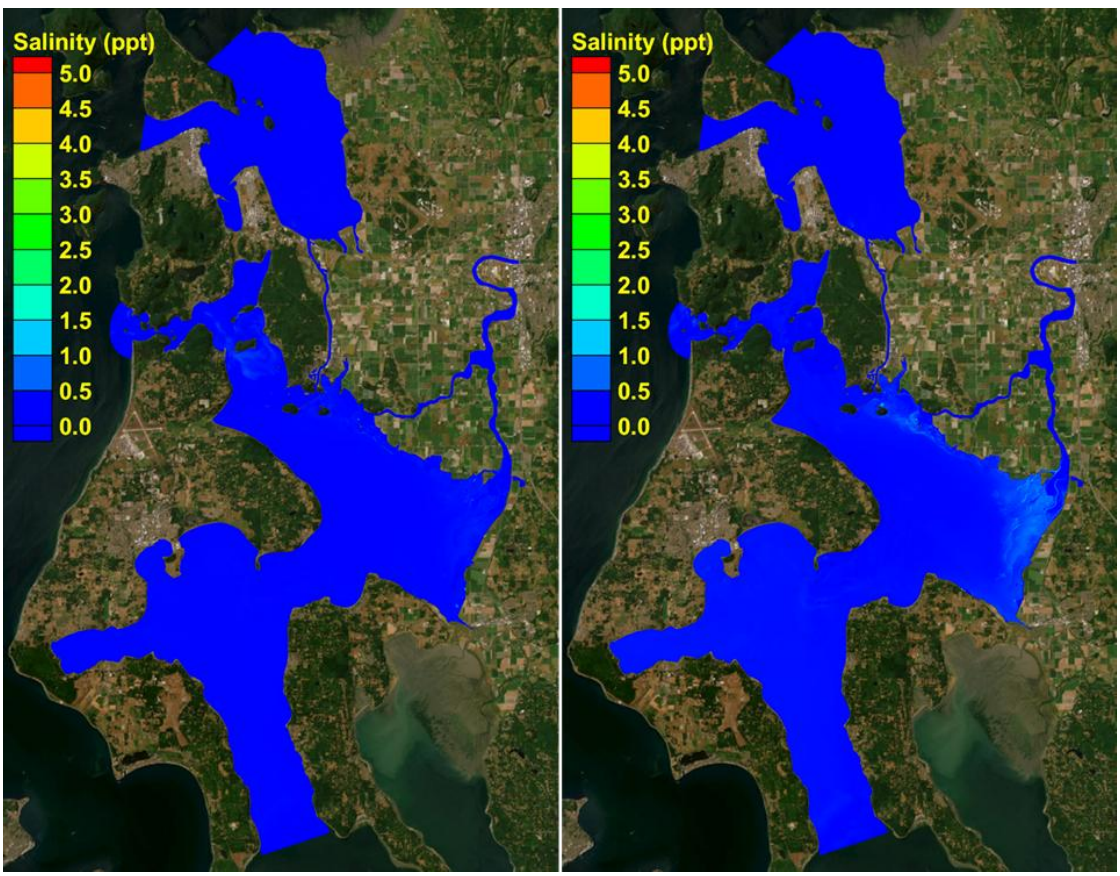

Figure 2.15. Changes in maximum surface layer salinity from the Current Water Use scenario to two alternative water withdrawal scenarios (left: Alt1 and Right: Alt2) during April – September (2010).

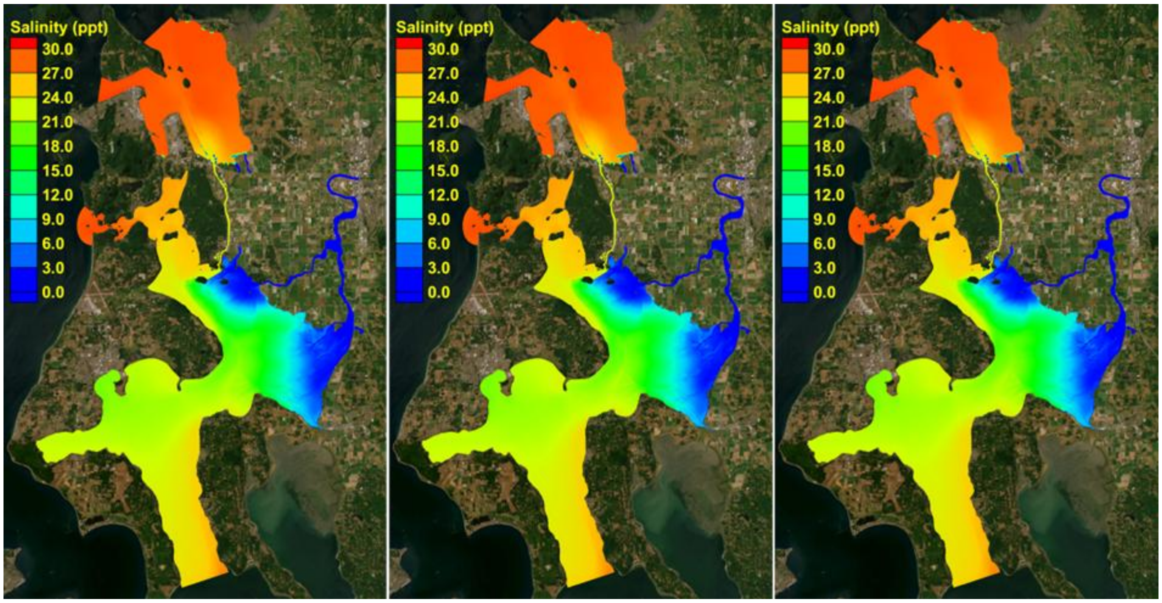

Figure 2.16. Mean surface layer salinity during May – September (2010). From left to right (Current Water Use, Alt1, and Alt2).

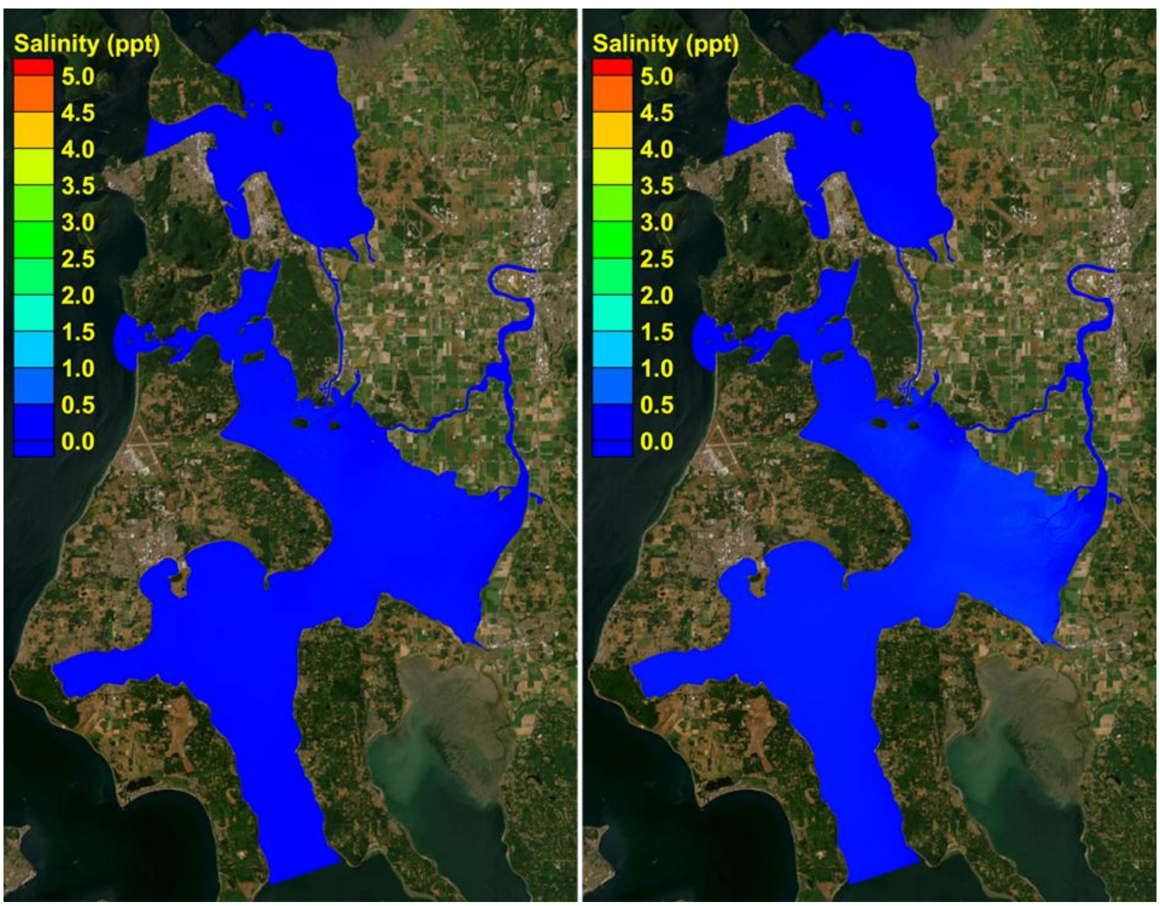

Figure 2.17. Changes in mean surface layer salinity from the Current Water Use scenario to two alternative water withdrawal scenarios (left: Alt1 and Right: Alt2) during May – September (2010).

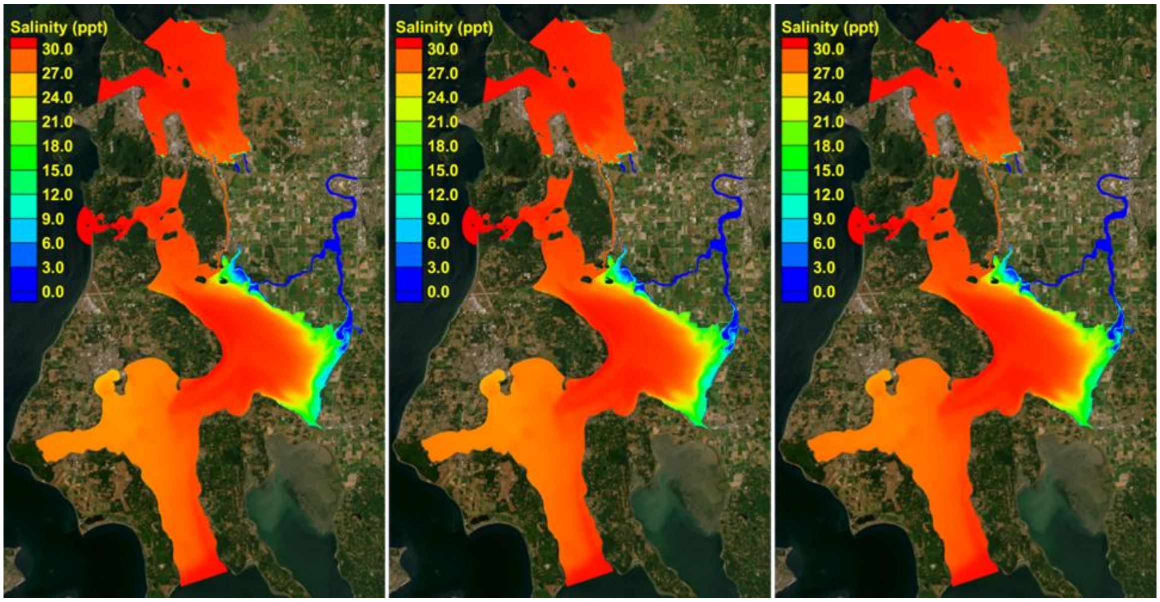

Figure 2.18. Maximum surface layer salinity during May – September (2015). From left to right (Current Water Use, Alt1, and Alt2).

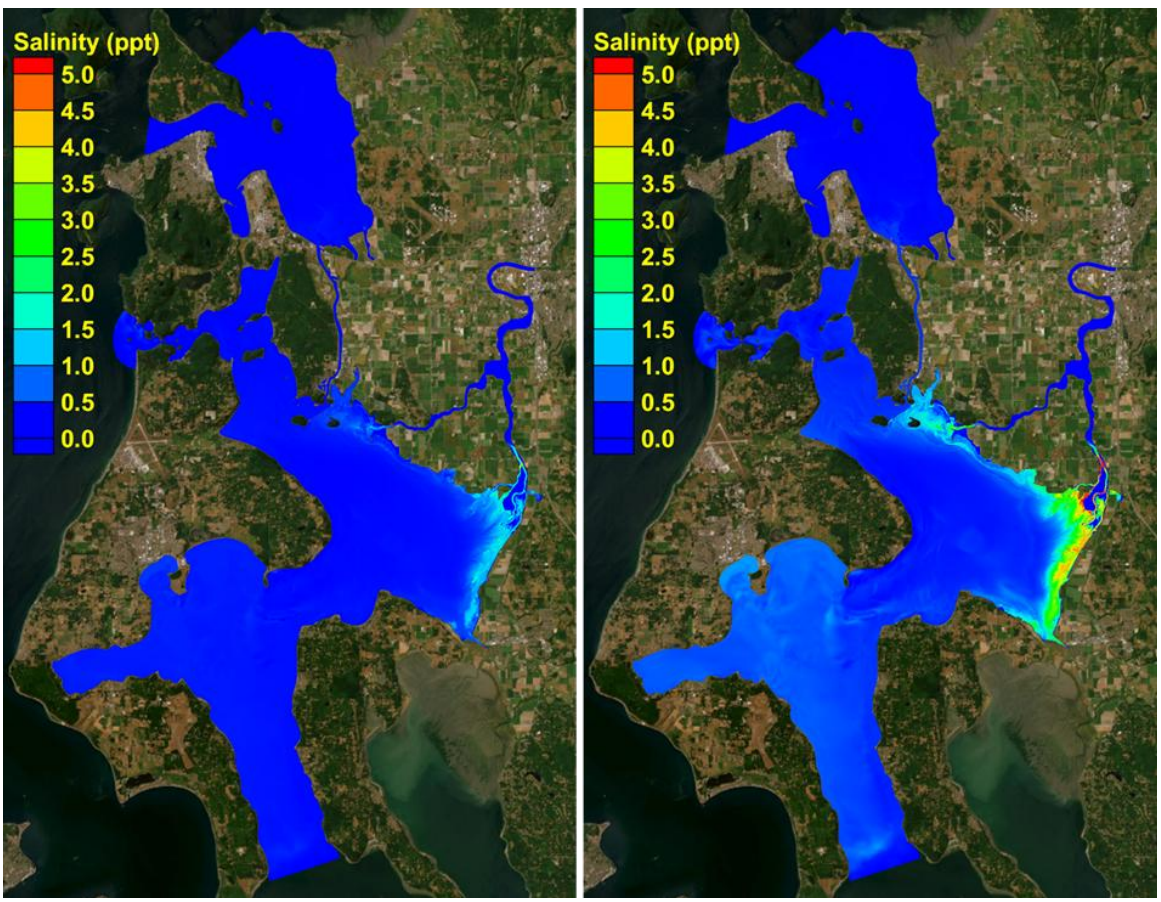

Figure 2.19. Changes in maximum surface layer salinity from the Current Water Use scenario to two alternative water withdrawal scenarios (left: Alt1 and Right: Alt2) during May – September (2015).

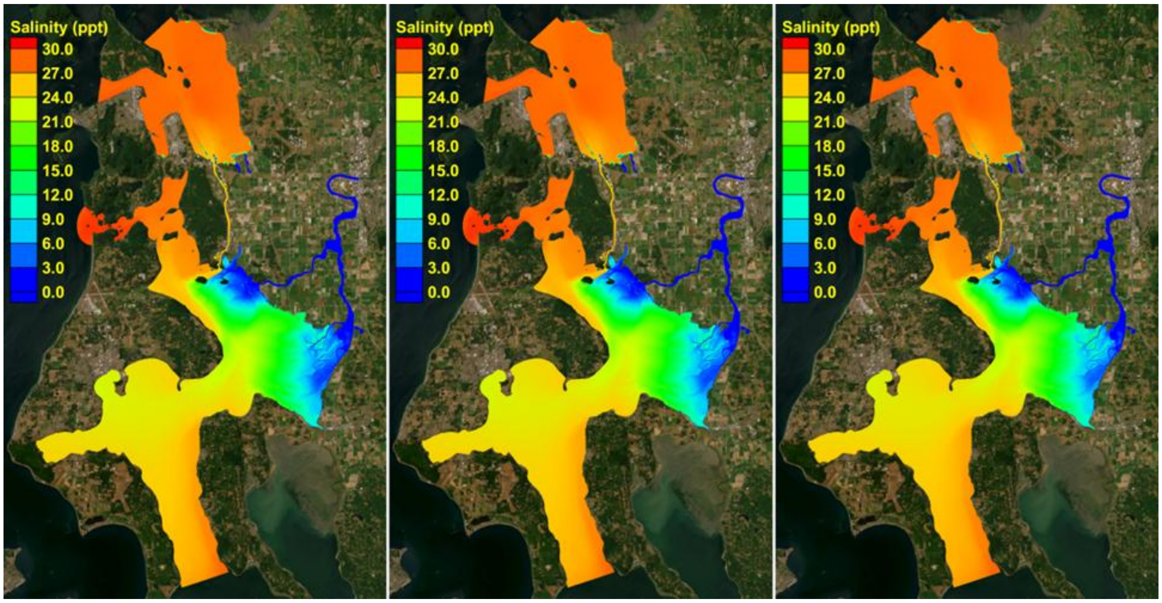

Figure 2.20. Mean surface layer salinity during May – September (2015). From left to right (Current, Alt1, and Alt2).

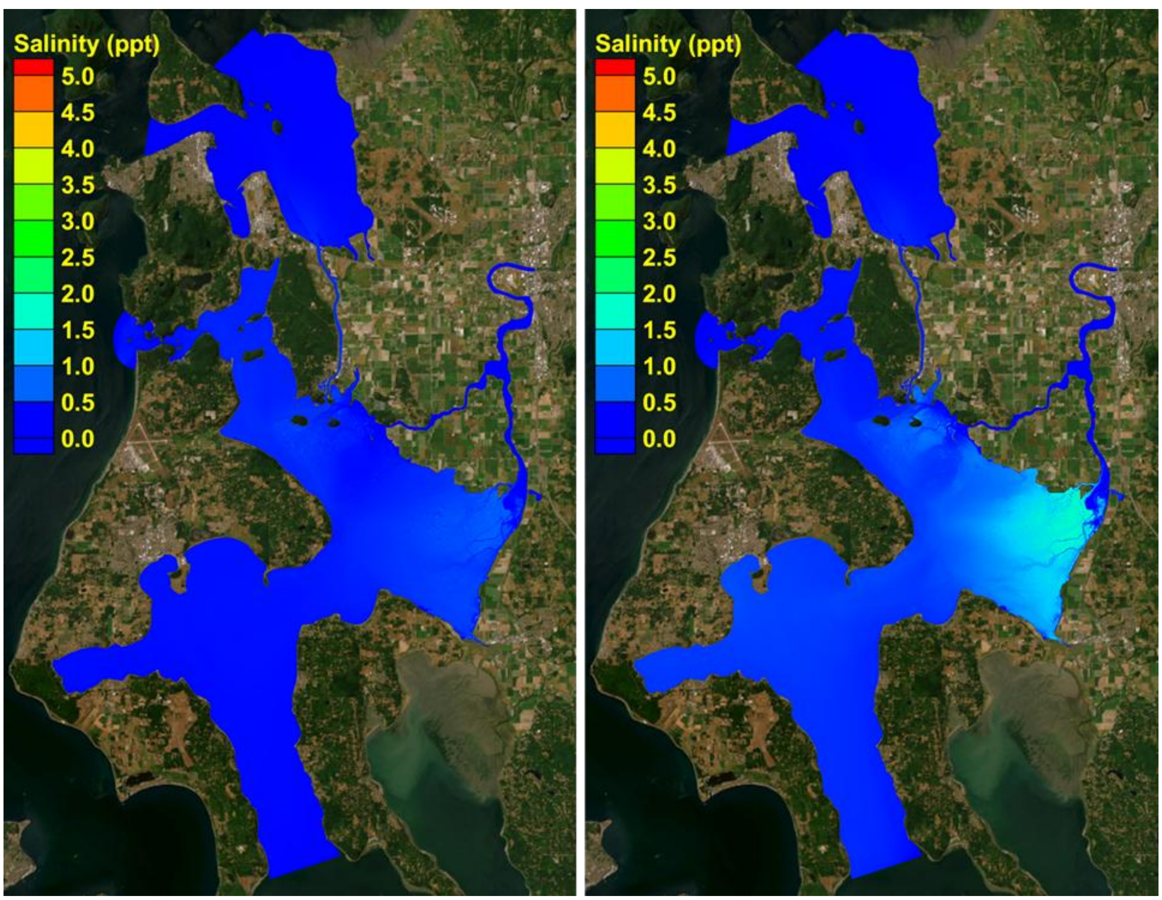

Figure 2.21. Changes in mean surface layer salinity from the Current Water Use scenario to two alternative water withdrawal scenarios (left: Alt1 and Right: Alt2) during May – September (2015).

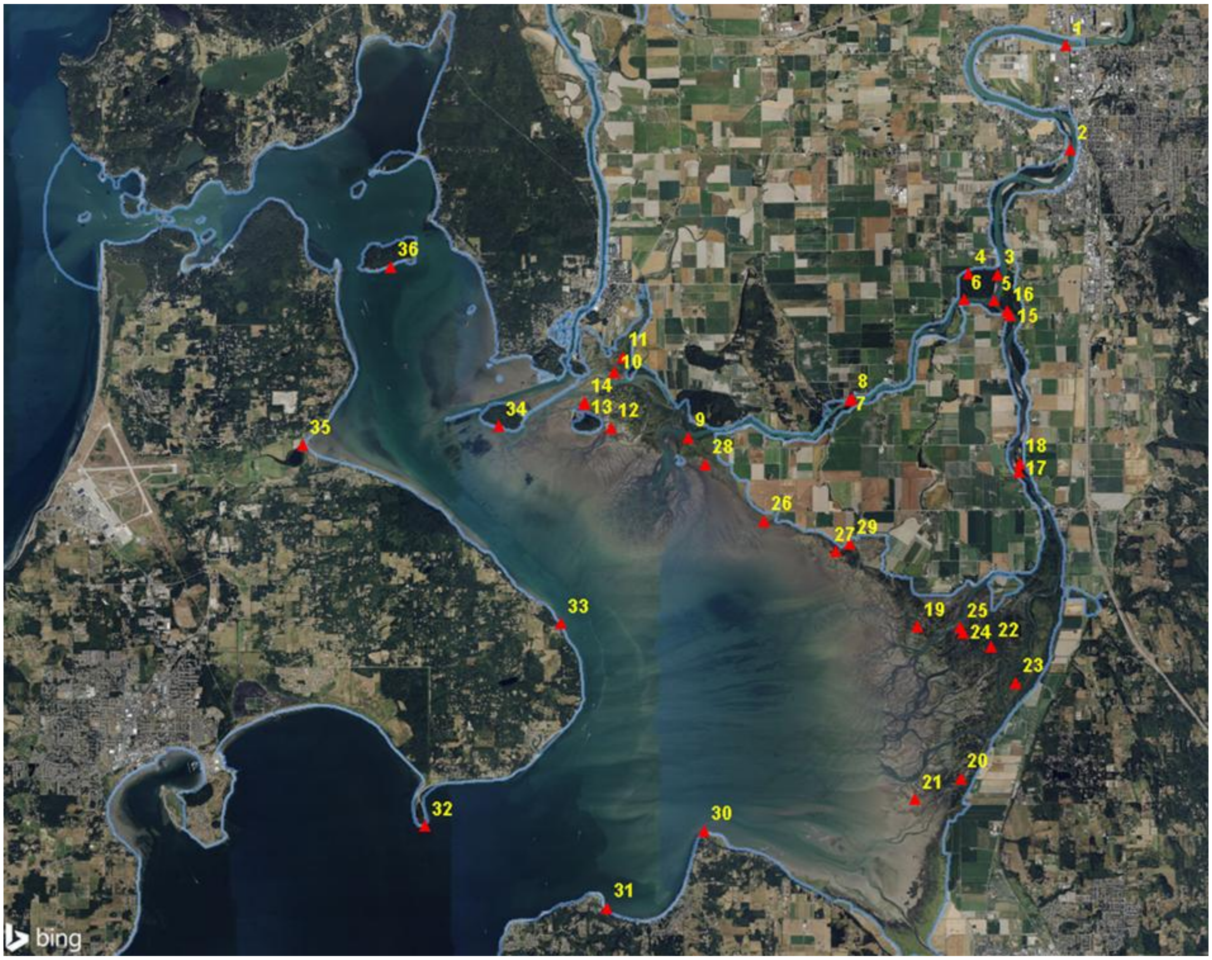

Figure 2.22. Representative model output sites (stations) used to support other research tasks in this study.

In addition to examining domain-wide changes, hourly time series of key hydrodynamic parameters were extracted and shared with the rest of the project team to support investigations into vegetation and fish dynamics. For example, a total of 36 stations, representing a range of site characteristics, were selected for time series analysis (Figure 2.22).

Figures 2.23 and 2.24 present time series and cumulative frequency comparisons of water surface elevation at four representative sites. The results indicate that, with the exception of the most upstream location (Station 5)—where the instantaneous maximum water level drop exceeds 0.15 m under the Alternative 2 water withdrawal scenario compared to the current condition—changes in water level due to withdrawal are generally negligible at the other sites.

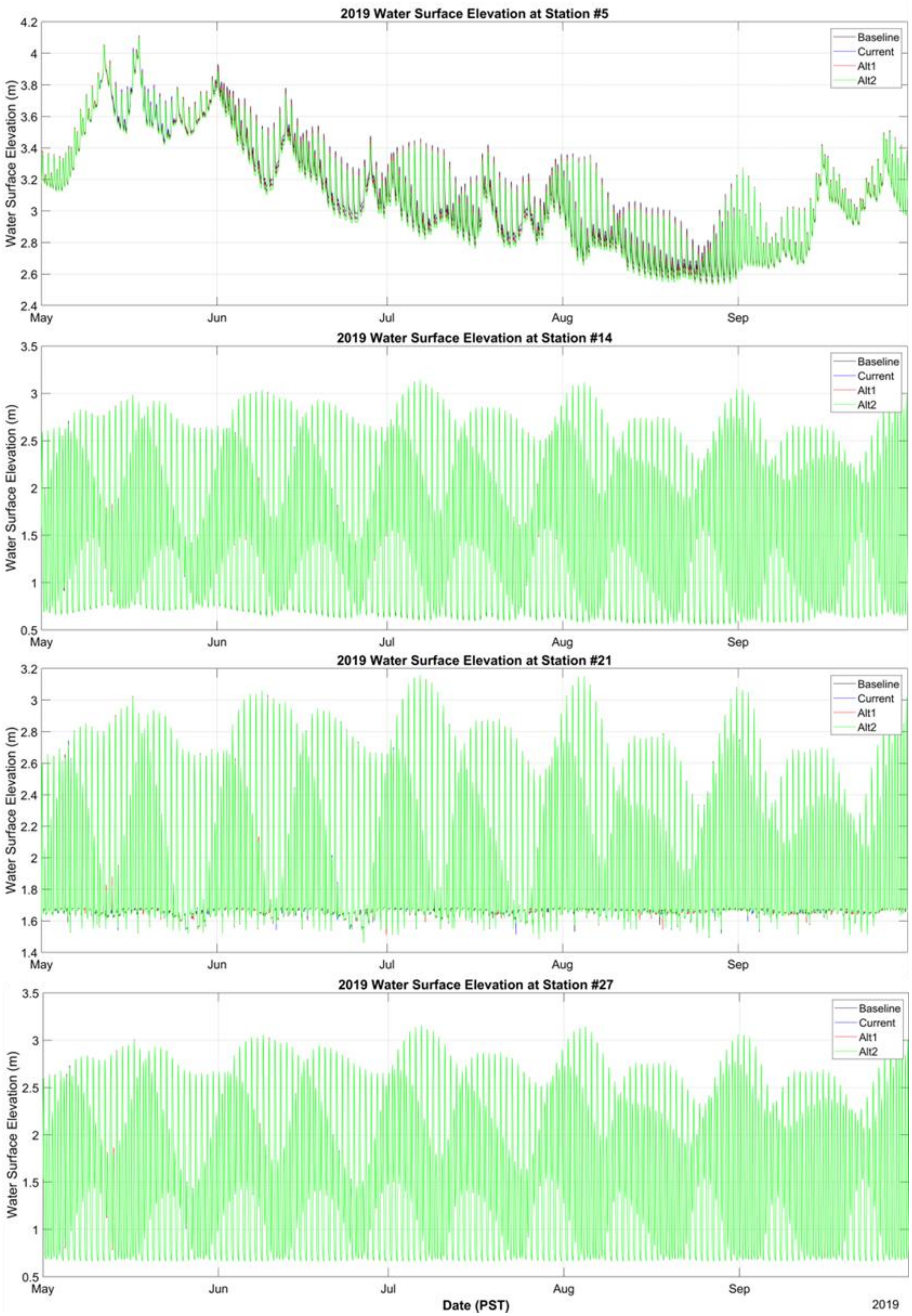

Figure 2.23. Time series comparisons of water surface elevation for the four model scenarios (No Water Use, Current water Use, Alt1, and Alt2) at Stations 5, 14, 21, and 27.

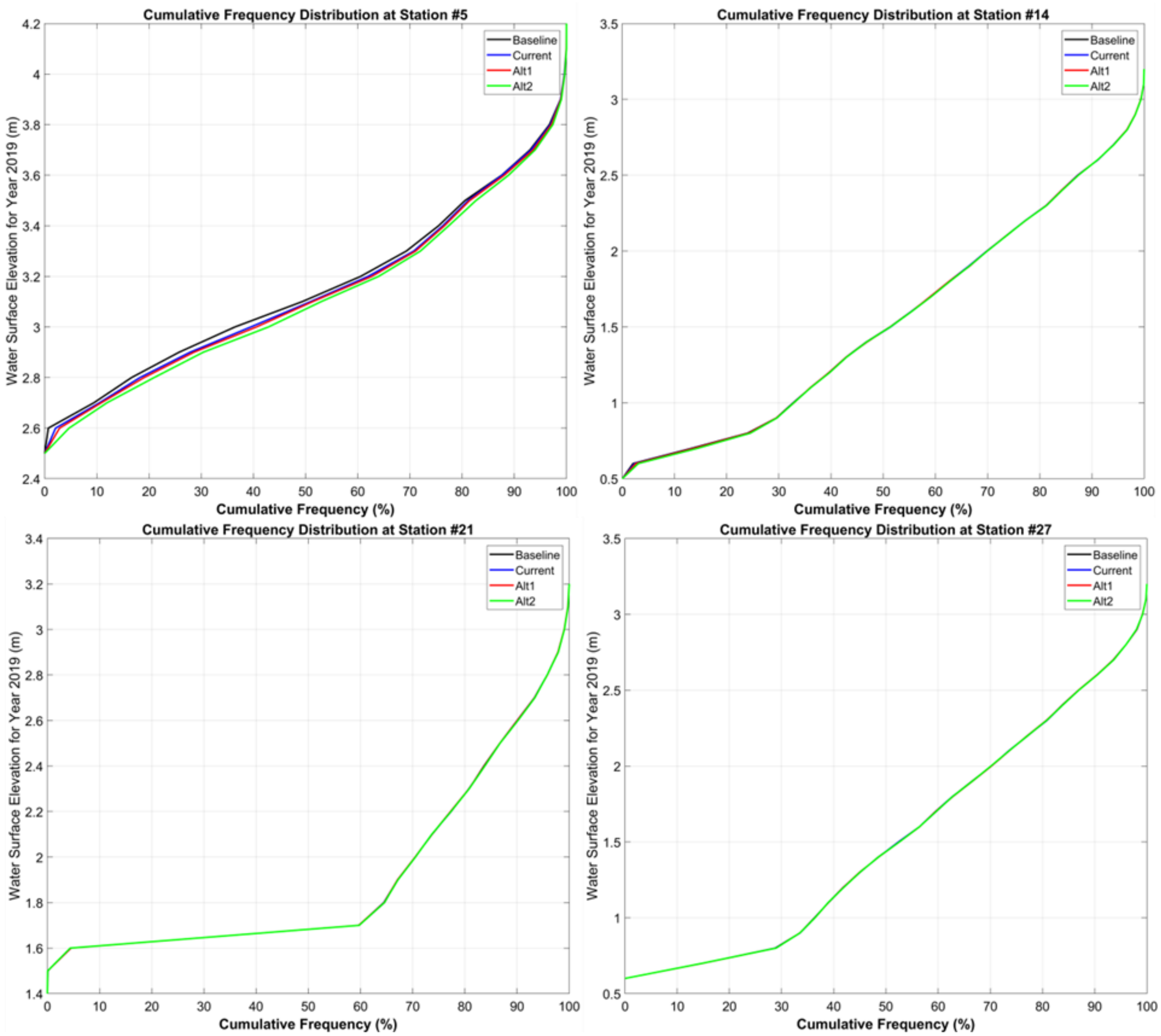

Figure 2.24. Cumulative frequency distribution comparisons of water surface elevation for the four model scenarios (No Water Use, Current Water Use, Alt1, and Alt2) at Stations 5, 14, 21, and 27.

In contrast to water level, salinity exhibits much stronger responses to water withdrawals. Figures 2.25 and 2.26 show time series and cumulative frequency comparisons of surface salinity at three downstream sites (Station 5 is excluded due to its consistently freshwater conditions). All three sites display noticeable increases in surface salinity as a result of water withdrawals. The magnitude of these changes is consistent with the patterns observed in Figures 2.10 through 2.21. The cumulative frequency plots indicate that, for the same cumulative frequency (e.g., 80%), the corresponding salinity values under the Alternative 2 scenario are approximately 1 ppt higher than those under the current condition at Stations 21 and 27.

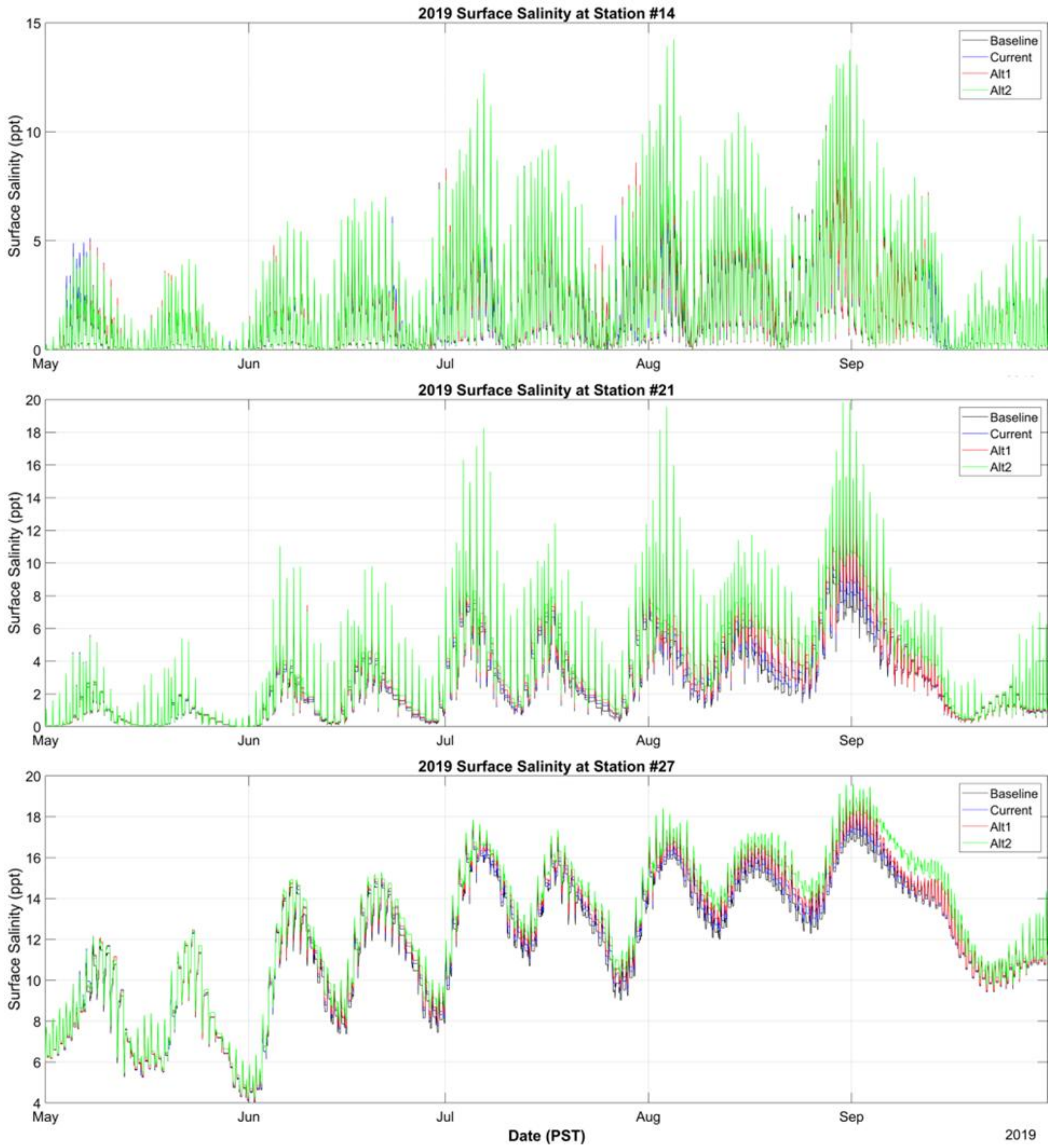

Figure 2.25. Time series comparisons of surface salinity among the four model scenarios (No Water Use, Current Water Use, Alt 1, and Alt 2) at Stations 14, 21, and 27.

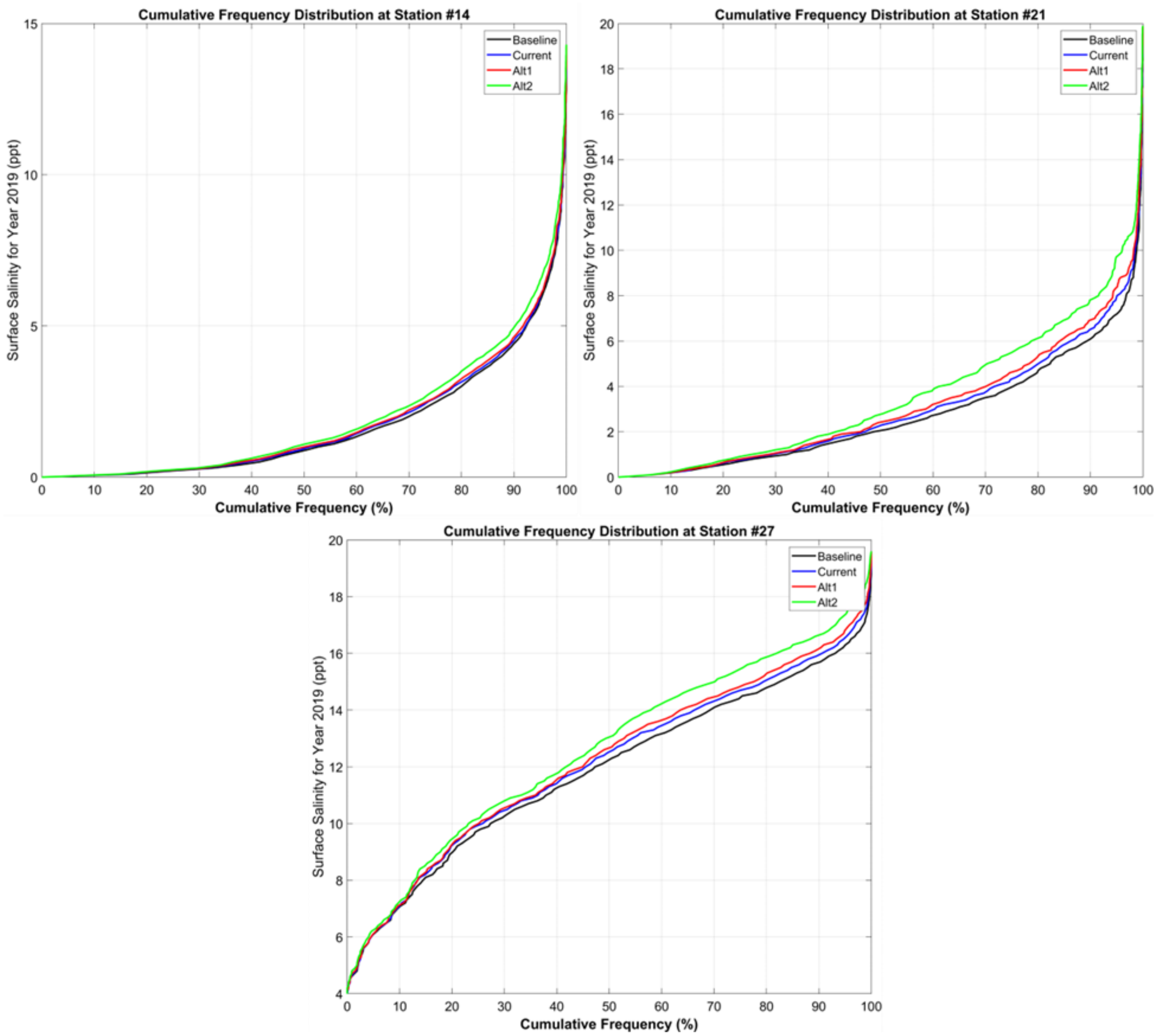

Figure 2.26. Cumulative frequency distribution comparisons of surface salinity among the four model scenarios (No Water Use, Current Water Use, Alt 1, and Alt 2) at Stations 14, 21, and 27.

To illustrate the minimal changes in current speed resulting from water withdrawals, we compared the cumulative frequency distributions of surface current speed at the same four selected stations, as shown in Figure 2.27. Compared to the changes in water level and salinity (Figures 2.24 and 2.26, respectively), the current speed exhibits only minor perturbations—even at the most upstream site (Station 5).

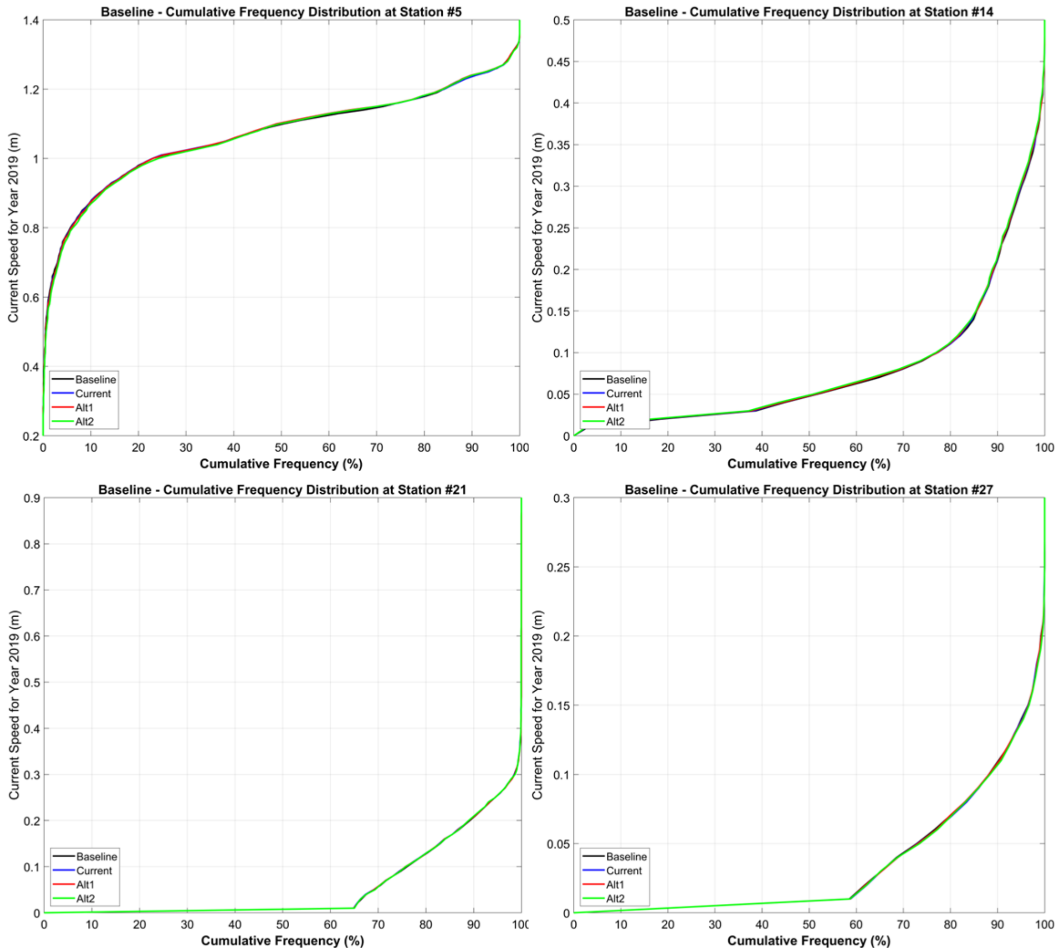

Figure 2.27. Cumulative frequency distribution comparisons of surface current speed among the four model scenarios (No Water Use, Current Water Use, Alt 1, and Alt 2) at Stations 5, 14, 21, and 27.

Effects of flow and tidal range on duration of inundation and threshold salinity

In Duke (1999), data on tidal channel depth was related to river flow and tides using linear regression, and the independent effect of flow was used to determine the impact of flow reductions. This same concept can be revisited using the hydrodynamic model by comparing daily flow with daily summaries of hydrodynamic characteristics. We focused on two patterns: the proportion of time that channel depth exceeded 30 cm (1 foot, the threshold used in the Duke study), and the proportion of time that salinity was above a threshold value. In this analysis we focus on values <5 ppt. Examination of channel sites (4, 8, 13, 16, 18, 24, and 29) – which were expected to be more sensitive to inundation than other sites – indicated that higher river flow generally increased the proportion of time that salinity remained below 5 ppt and depth was greater than 30 cm, but that the magnitude and presence of effects varied among sites (Figures 2.28 and 2.29). GAM analyses corroborated these findings and also showed that the daily tidal range influenced both proportion of time that inundation exceeded 30 cm and proportion of time that salinity was <5 ppt. For sites with adequate variation, effects of flow tended to be steeper and more linear than effects of tidal range (Figures 2.30 and 2.31).

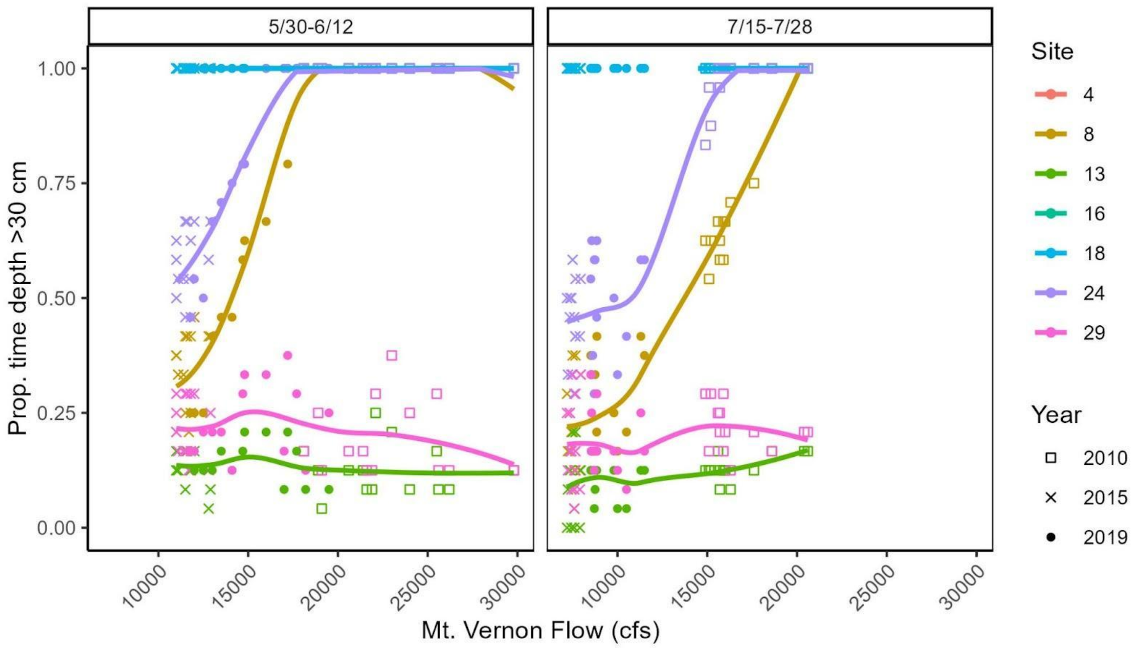

Figure 2.28. Proportion of time that depth of channel sites exceeded 30 cm compared to flow. Model outputs are based on the current scenario and include all years. Lines show loess local regression fit to model outputs from the same site.

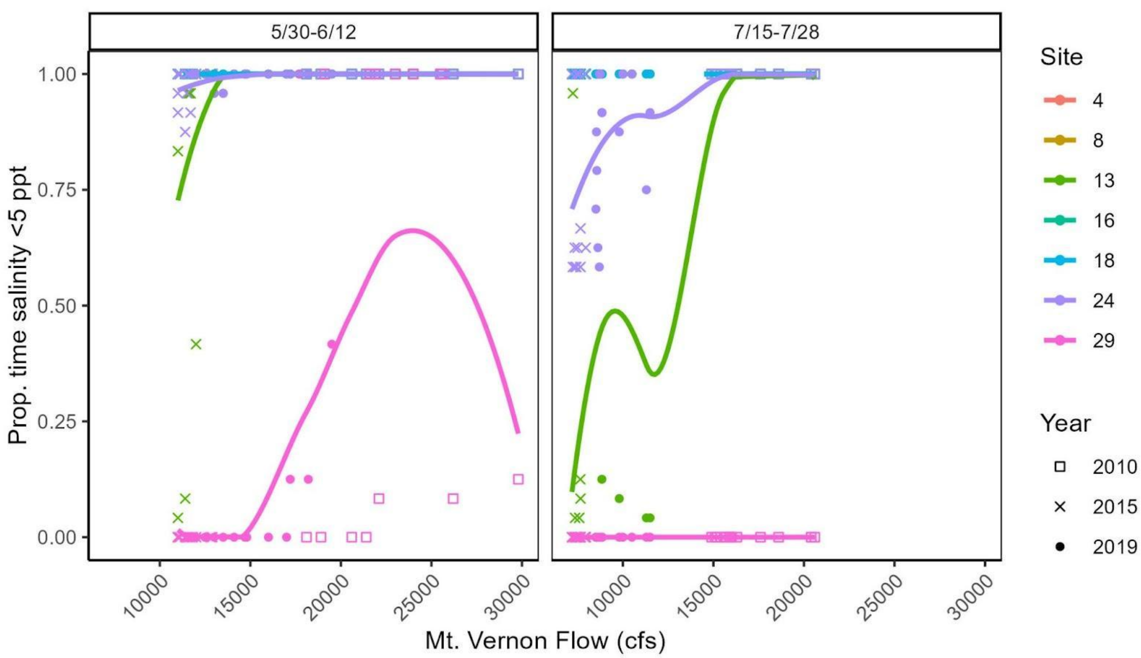

Figure 2.29. Proportion of time in a day that the salinity of channel sites was less than 5 ppt as a function of daily river flow. Model outputs are based on the current scenario and include all years. Lines show loess local regression fit to model outputs from the same site.

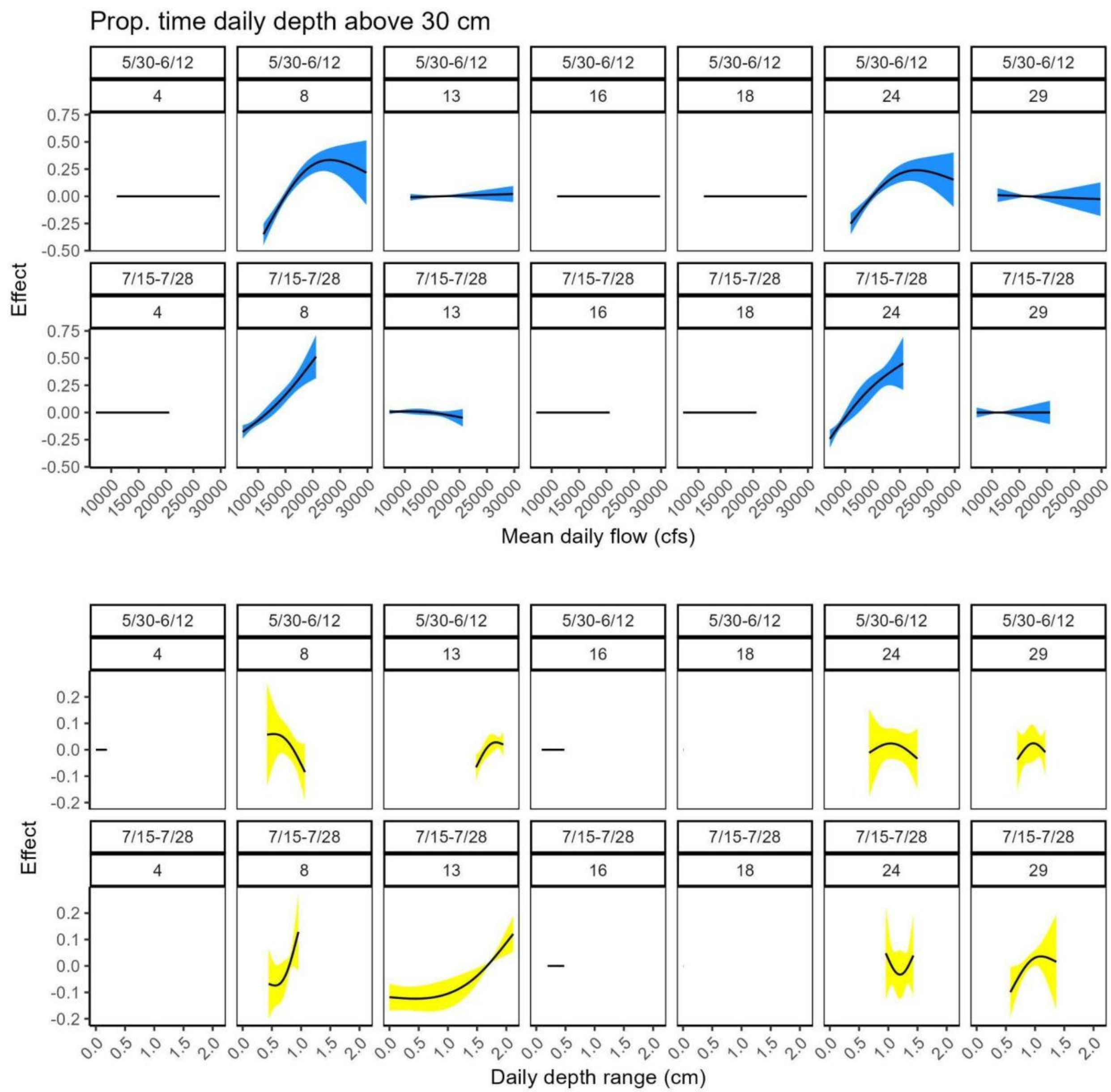

Figure 2.30. GAM model predictions showing the independent influence of flow and daily tidal range on proportion of time that channel sites' depths exceeded 30 cm. Numbers identify sites. Shaded areas indicate 95% confidence intervals.

Sensitivity analysis of water depth changes to water withdrawals

To further evaluate the impacts of water withdrawals on water depths, we calculated the daily cumulative occurrence frequency (in terms of number of hours meeting the minimum depth criterion on each day) of water depth exceeding 30 cm during the two critical periods (5/30 – 6/12 and 7/15 – 7/28) for each water withdrawal and flow year. The calculations were conducted for all the 36 sites shown in Figure 2.22. Figures 2.32 through 2.35 show the results for Year

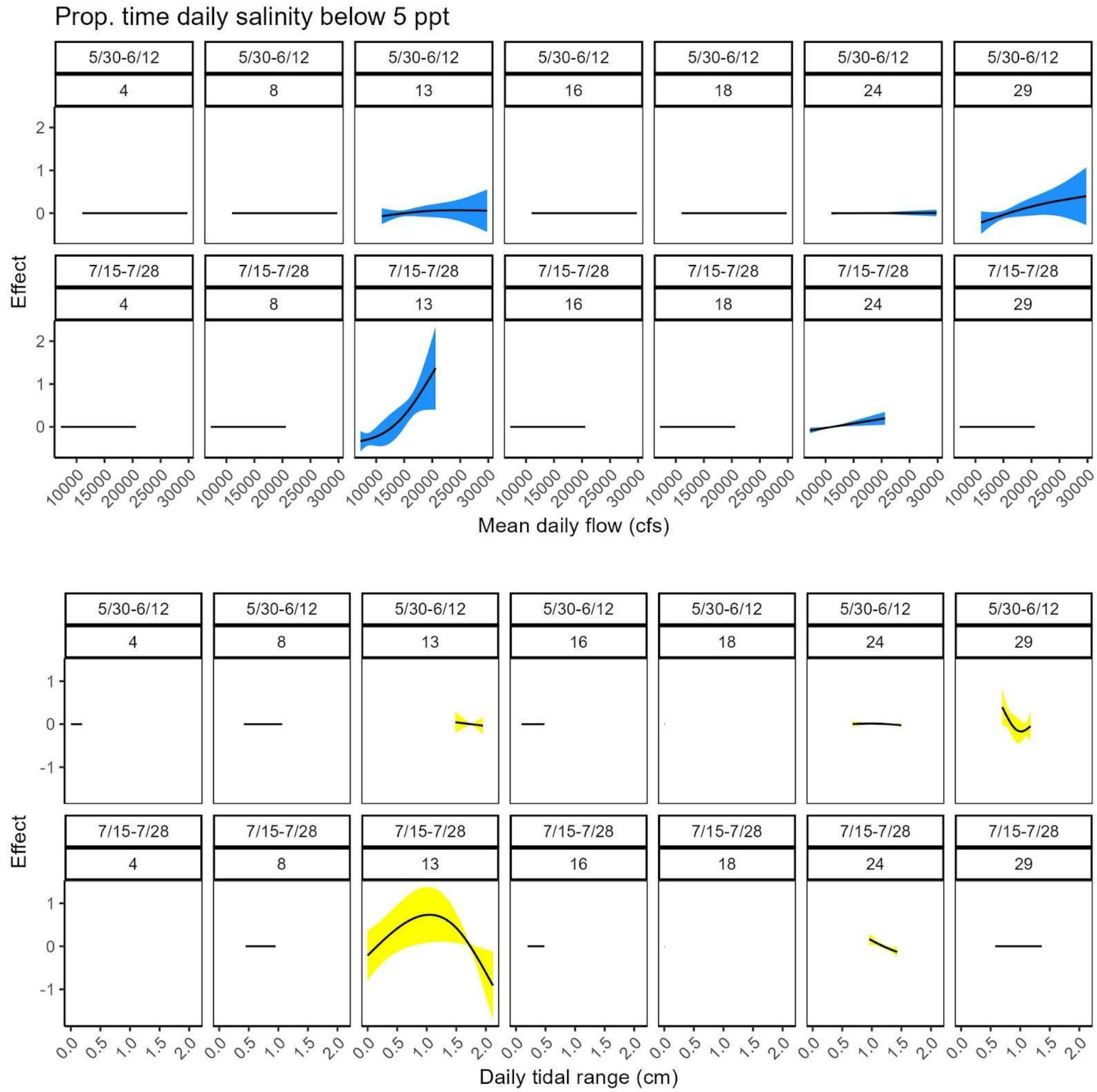

Figure 2.31. GAM output showing the influence of flow and daily depth range on proportion of time that channel sites' salinity was below ppt. Numbers identify sites. Shaded areas indicate 95% confidence intervals.

2019. Considerable changes can be seen in Figure 2.34 between the Alternative 2 water withdrawal condition and the current water use condition for the first period of 5/30 – 6/12, indicating water withdrawal does affect critical water depth occurrence frequencies on a daily basis for selected sites/regions in the Skagit Delta. The results for Years 2015 and 2010 are provided in Appendix 2.1 (Figures 2.1.4 - 2.1.21).

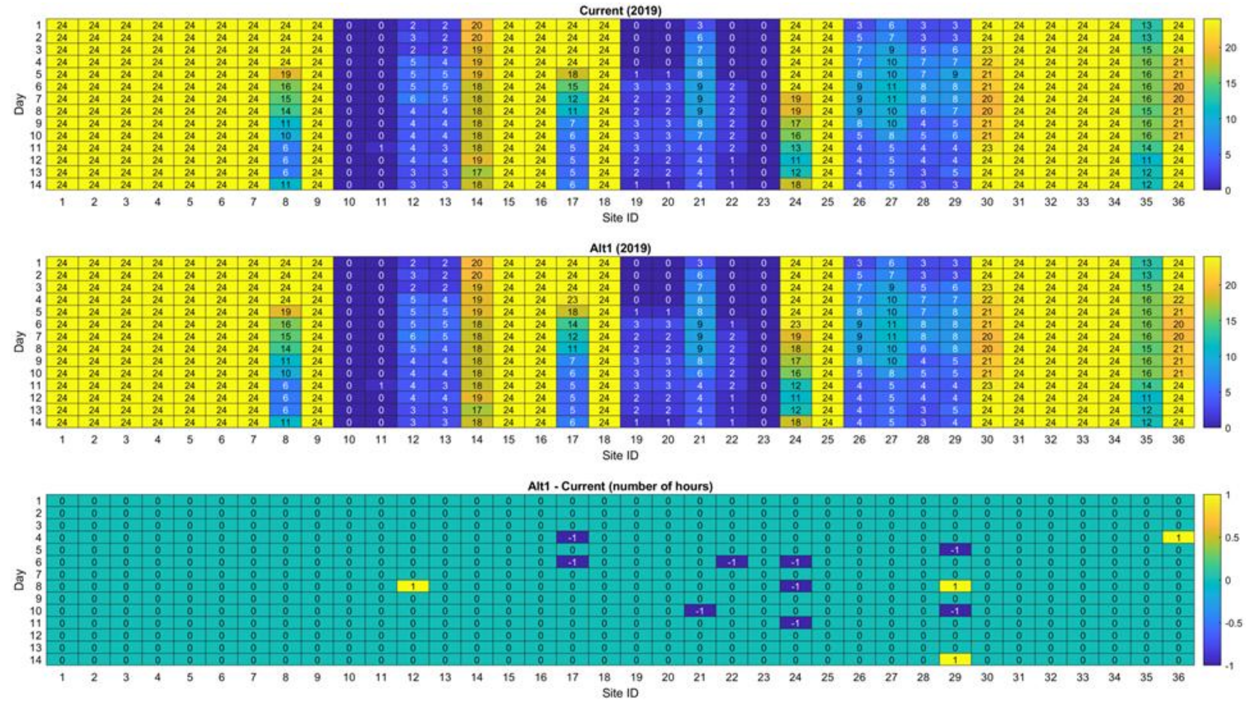

Figure 2.32. Daily cumulative occurrence frequency of water depth exceeding 30 cm during the 5/30 – 6/12 time window for Year 2019 at all 36 sites (top panel - current water use condition; middle panel - Alt 1 scenario; bottom panel - differences between Alt 1 and current water use condition).

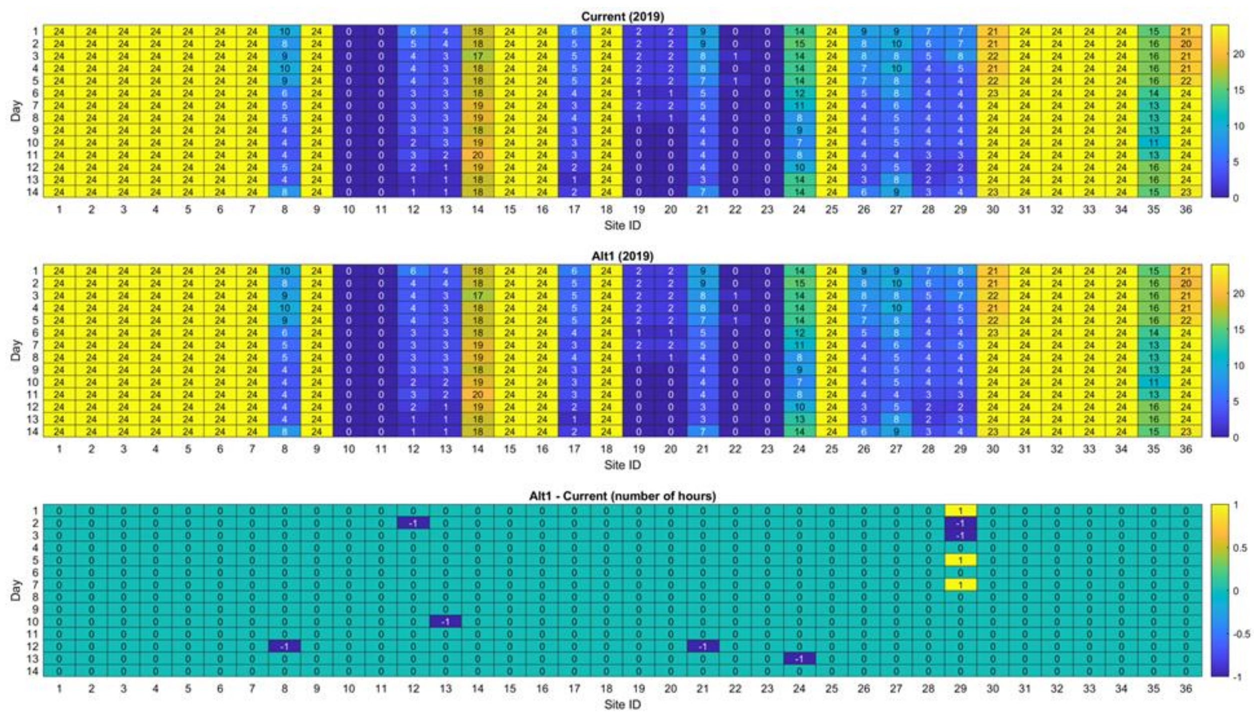

Figure 2.33. Daily cumulative occurrence frequency of water depth exceeding 30 cm during the 7/15 – 7/28 time window for Year 2019 at all 36 sites (top panel - current water use condition; middle panel - Alt 1 scenario; bottom panel - differences between Alt 1 and current water use condition).

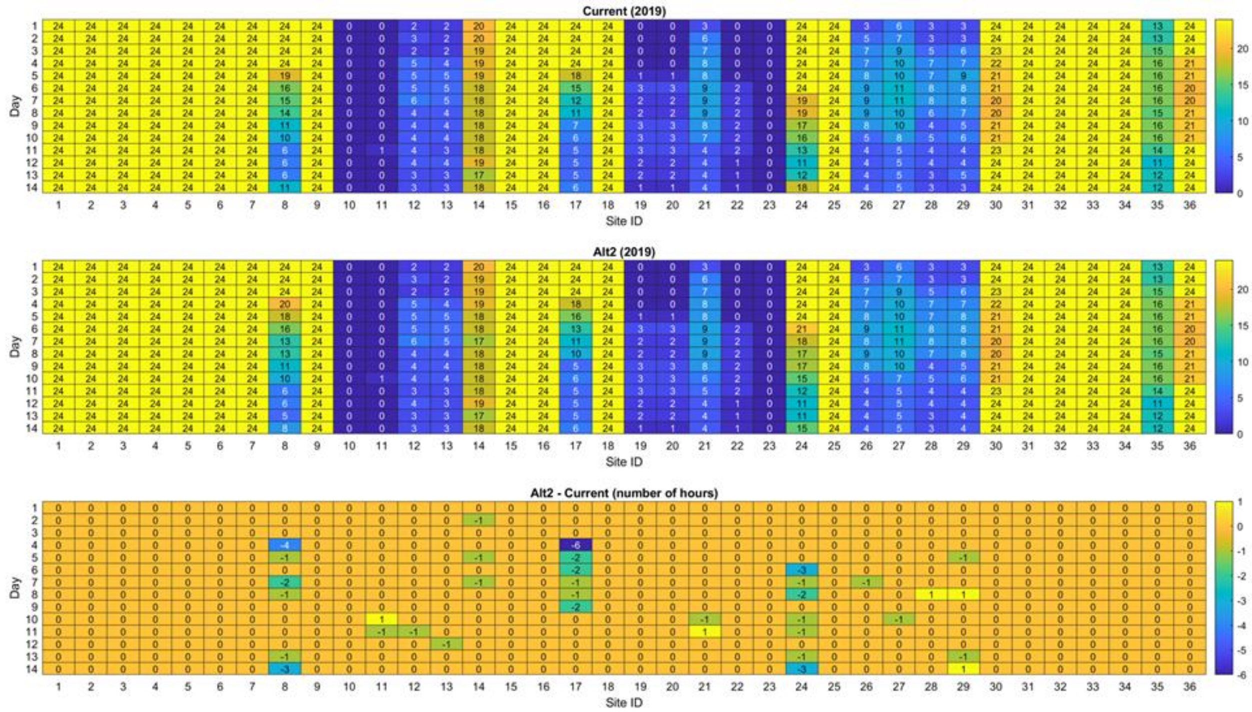

Figure 2.34. Daily cumulative occurrence frequency of water depth exceeding 30 cm during the 5/30 – 6/12 time window for Year 2019 at all 36 sites (top panel - current water use condition; middle panel - Alt 2 scenario; bottom panel - differences between Alt 2 and current water use condition).

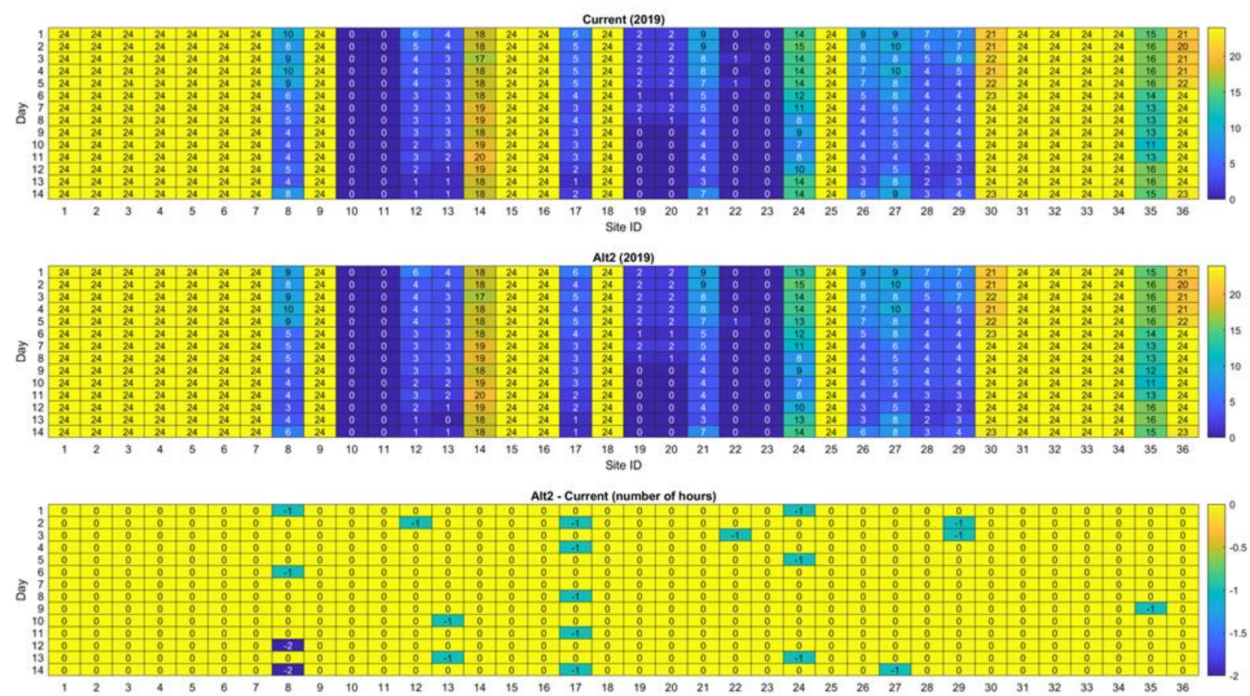

Figure 2.35. Daily cumulative occurrence frequency of water depth exceeding 30 cm during the 7/15 – 7/28 time window for Year 2019 at all 36 sites (top panel - current water use condition; middle panel - Alt 2 scenario; bottom panel - differences between Alt 2 and current water use condition).

Hydrodynamic responses to unregulated flows

We further evaluated the differences in water surface elevation and surface salinity between the unregulated flow condition and the baseline condition for model year 2010 (corresponding to Run 14 and Run 15, respectively, in Table 2.A1). Similar 2D planview and time series plots (Figures 2.36 – 2.41) were prepared to illustrate the changes. As expected, compared to the previous water withdrawal scenarios (Current water use, Alt1, and Alt2), both water level and salinity exhibit more pronounced changes under the unregulated flow condition. These changes are primarily driven by variations in Skagit River flows. During the May–September period, changes in water surface elevation are most noticeable in the mainstem of the Skagit River and the upper portions of its North and South Forks, where water levels are predominantly influenced by river discharge (Figures 2.36 and 2.37). For example, the increase in maximum water surface elevation exceeds 1 meter in the upper river reaches of the domain (Figure 2.36), while changes in mean water surface elevation are smaller (Figure 2.37). By further examining water level time series at selected locations of the model domain, we can clearly see that the most upstream station (Station 5) shows the maximum changes between the unregulated flow and baseline conditions (Figure 2.38). These changes also appear to closely follow the Skagit River hydrographs in Figure 1.4.

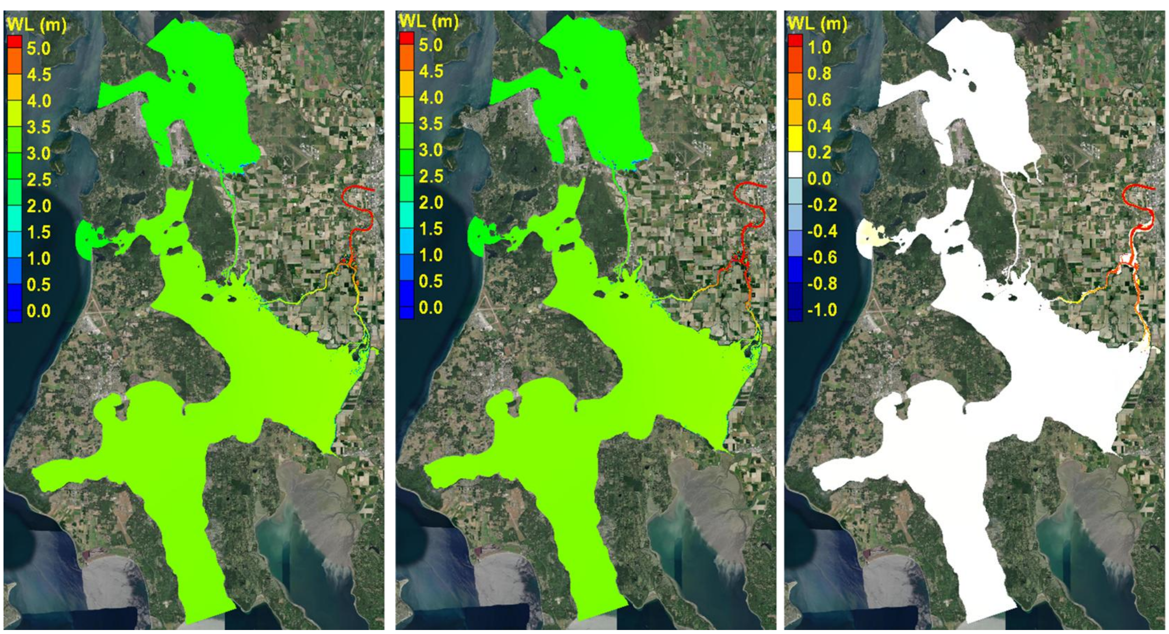

Figure 2.36. Maximum water surface elevation during May – September (2010) for the baseline (left panel) and unregulated flow (middle panel) conditions. The changes in maximum surface elevation from the baseline to the unregulated flow conditions are shown in the right panel.

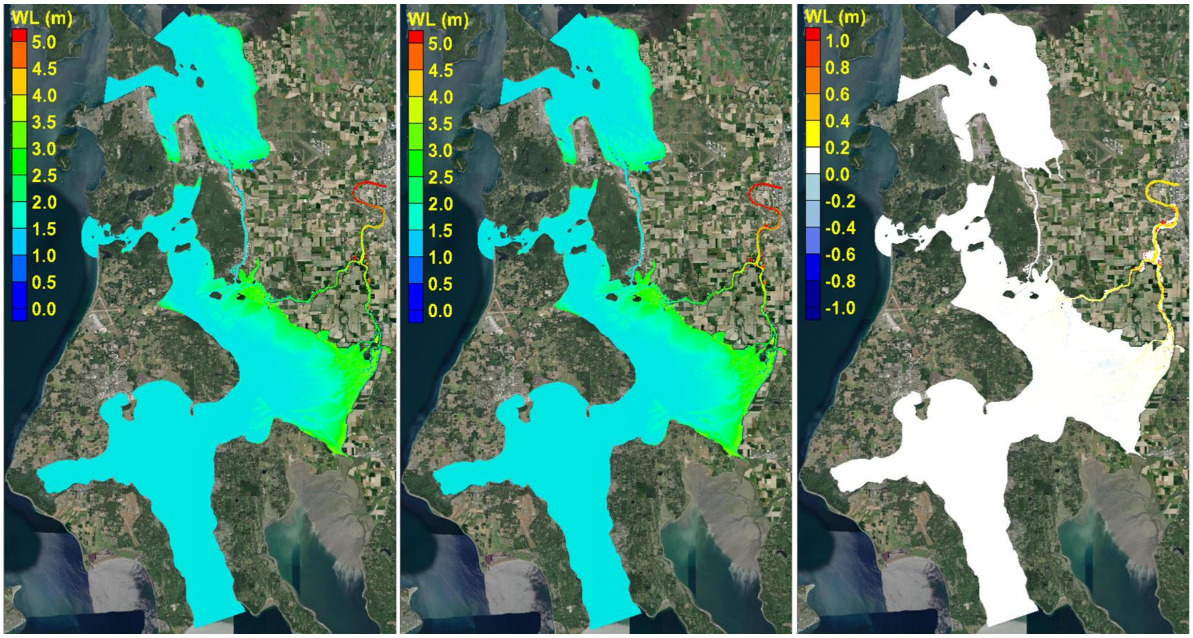

Figure 2.37. Mean water surface elevation during May – September (2010) for the baseline (left panel) and unregulated flow (middle panel) conditions. The changes in mean surface elevation from the baseline to the unregulated flow conditions are shown in the right panel.

For changes in the surface salinity field during the May–September period, the results show a more complex and mixed spatial distribution (Figures 2.39 and 2.40). Maximum surface salinity increases in the intertidal zones and Penn Cove, while generally decreasing or remaining unchanged elsewhere. In comparison, the increases in mean surface salinity within the intertidal zones of both the North and South Forks are smaller, while the rest of the domain shows a general decrease. These results indicate that the spatial variability of salinity is heavily influenced by Skagit River discharge over the model simulation period. More specifically, mean salinity in the main bay is primarily controlled by cumulative Skagit River flows, while maximum salinity in the shallower intertidal zones appears to be more influenced by short-term flow conditions. During the May–September period, unregulated flow is generally higher than baseline flow, except during the low-flow month of August. As a result, mean salinity across most of the domain – particularly in deeper regions – is generally lower (fresher) under the unregulated flow condition. In contrast, maximum surface salinity in the intertidal zones is more affected by short-term low-flow events (e.g., August) and shows significantly saltier conditions under unregulated flow. The high sensitivity of salinity to Skagit River discharge is further demonstrated by the surface salinity time-series at Stations 14, 21, and 27 (Figure 3.41), where high salinity extremes are observed at all three locations during the August low-flow period.

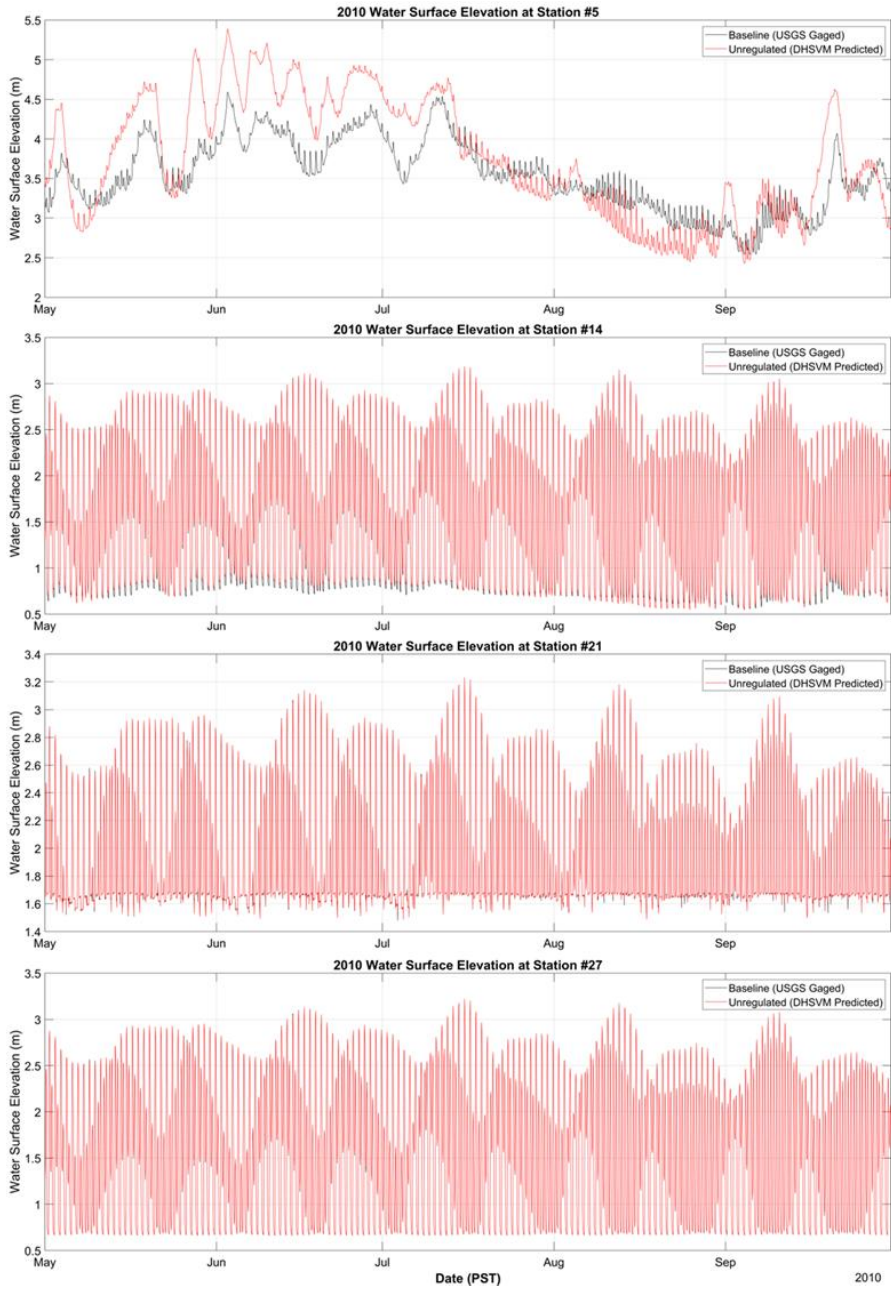

Figure 2.38. Time series comparisons of water surface elevation between the baseline and unregulated flow conditions at Stations 5, 14, 21, and 27.

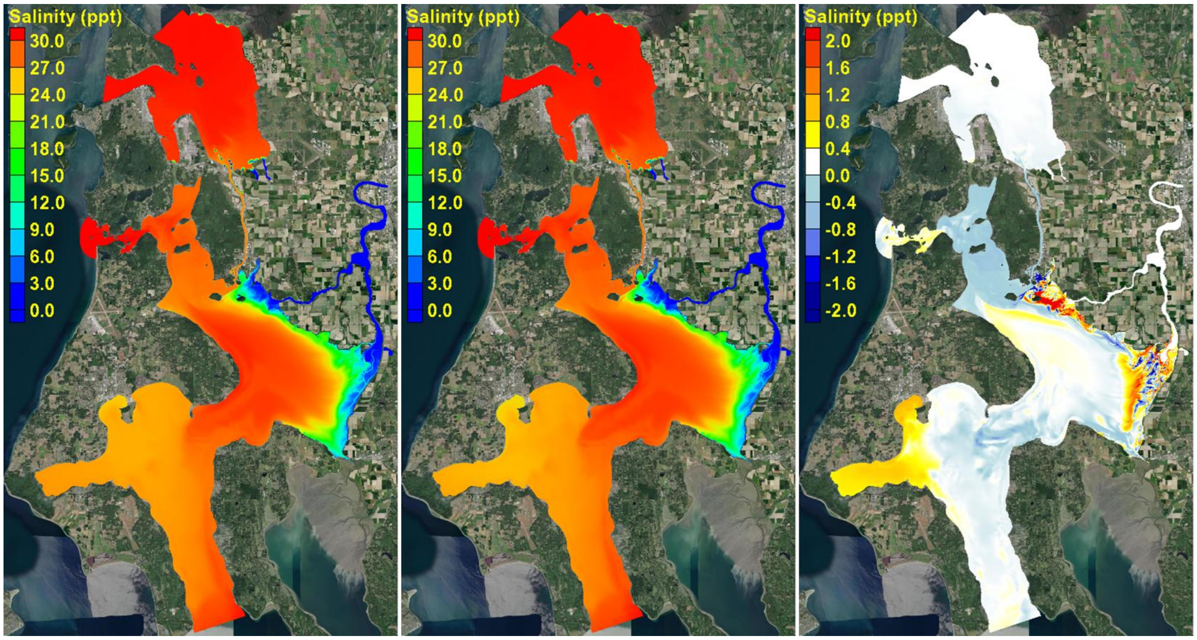

Figure 2.39. Maximum surface salinity during May – September (2010) for the baseline (left panel) and unregulated flow (middle panel) conditions. The changes in maximum surface salinity from the baseline to the unregulated flow conditions are shown in the right panel.

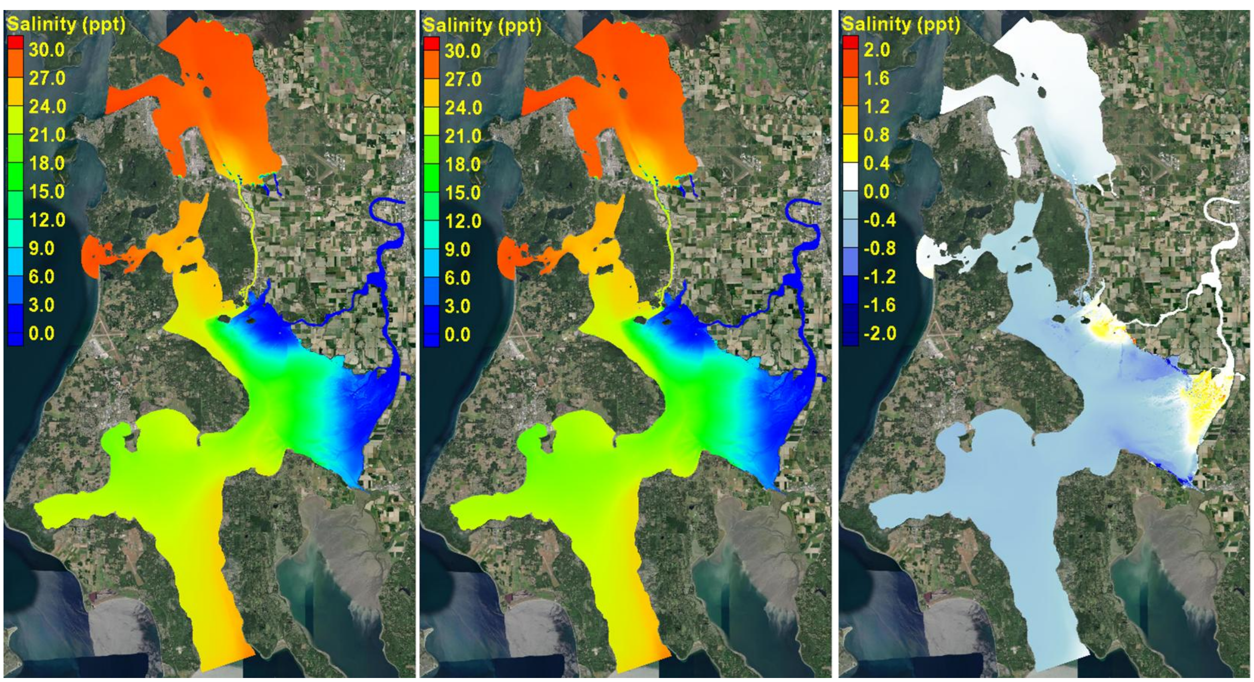

Figure 2.40. Mean surface salinity during May – September (2010) for the baseline (left panel) and unregulated flow (middle panel) conditions. The changes in mean surface salinity from the baseline to the unregulated flow conditions are shown in the right panel.

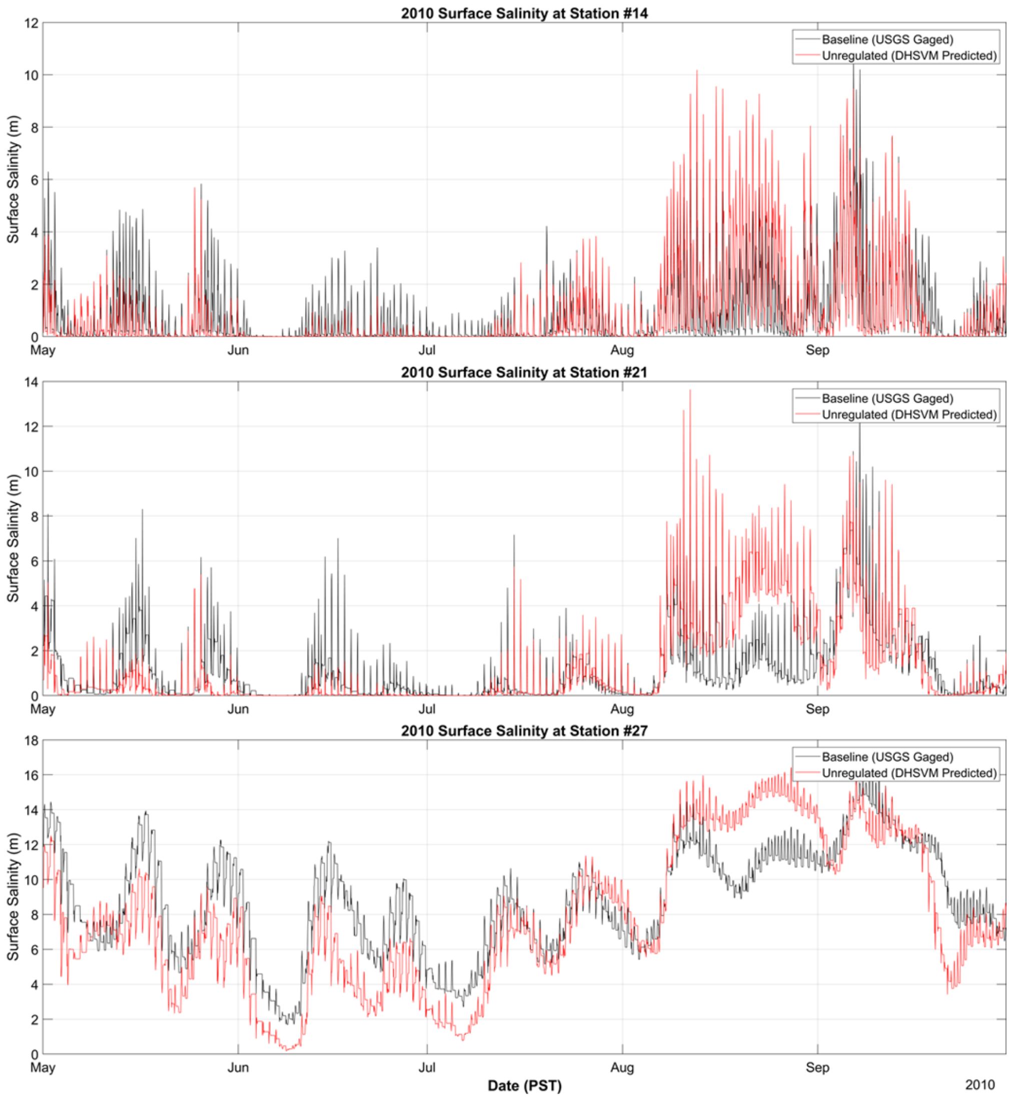

Figure 2.41. Time series comparisons of water surface elevation between the baseline and unregulated flow conditions at Stations 14, 21, and 27 (Station 5 is not shown due to its consistent freshwater condition).

Discussion

General findings from the refined hydrodynamic modeling study

We have updated the Skagit Hydrodynamic Model to a higher resolution unstructured grid, natural bathymetry, and revised channel network. A higher resolution grid was critical to match the model grid to calibration points and to this higher resolution required unsmoothed depth ranges to accurately calculate water surface elevation and depth. This combination required transitioning from FVCOM to a SCHISM model framework. While this change took some time to implement, the new framework provided much more realistic predictions in the tidal delta and nearshore, where water interacts directly with shoreline features, and where variability in substrate elevation can complicate hydrodynamics.

Nevertheless, during calibration we observed that the model can underpredict water surface elevation and especially salinity. There are a number of reasons why underpredictions might occur; a primary reason may be that tidal processes are incorporated on an hourly time step while river flow patterns are incorporated on a daily basis. Hence, fluctuations (especially increases) in flow that would be recorded by the tidal network sensors would not be reflected in flow patterns. This might explain why accuracy tended to increase over the spring and summer as the Skagit River hydrograph settled into baseflow. These patterns suggest that during the spring snowmelt, modeled effects of salinity might underestimate the true impacts of changing salinity upon marsh vegetation and juvenile fishes including Chinook salmon but would be more accurate at predicting ecological patterns later in the summer after the snowmelt.

We found differences in hydrodynamics during the May-September time period in the three model years. While currents and water surface elevations were not greatly different among years, salinity varied by up to 5 ppt in an average year (2010) compared to the lowest flow on record (2015). These differences were concentrated at the North and South Fork mouths entering Skagit Bay. Water use had compounding effects, with Alternative 2 exhibiting much greater differences than Alternative 1 in the spatial extent to which mean and maximum surface salinity increased.

Despite the lack of strong effects of scenarios on water surface elevation, we nevertheless observed that in tidal channels, salinities declined and water surface elevations increased as a function of river flow. These patterns mirrored findings from the Duke (1999) study, although this pattern was site-dependent, and exhibited nonlinear relationships. Also of note, modeled water surface elevations and salinity values tended to underpredict observed values for the majority of sites. The exceptions to this pattern were those that were the lowest and highest salinities.

In summary, the hydrodynamic model results indicate that the model is capable of reasonably capturing changes in hydrodynamics resulting from perturbations caused by water withdrawals. The magnitude of these changes varies by parameter (water level, velocity, and salinity), location, time, and flow condition, and is well correlated with the rate of water withdrawal. Salinity appears to be the most sensitive parameter, exhibiting the largest changes—for example, increases greater than 5 ppt in the South Fork intertidal zones under the high water withdrawal scenario (Alternative 2) during the lowest flow year (2015). However, the changes are comparatively smaller for the low water withdrawal scenario and during relatively high flow years. In contrast, changes in water level and velocity are much smaller and generally negligible, particularly in tidally dominated areas. Lastly, in contrast to the relatively mild flow perturbations caused by water withdrawals, the larger deviations of unregulated flow from the baseline condition result in more pronounced changes in both water level and salinity. In other words, the anticipated hydrodynamic changes due to water withdrawals are already overshadowed by those resulting from flow regulation, e.g., the regulated higher flows in August for the baseline condition effectively lead to lower salinity during that month.

Model Uncertainty

The Skagit Hydrodynamic Model developed in this study is a refined version of the original version (Whiting et al. 2017). It has been previously validated for water surface elevation, salinity, and currents using bathymetry and channel features corresponding to pre-avulsion conditions. In this study the original model was further refined in the delta region using updated bathymetry to reflect post-avulsion bathymetry and shoreline features. The model has been recalibrated using new water surface elevation and salinity data collected as part of this work. The results of model calibration presented in the form of average mean error (bias), root mean square error (RMSE), and correlation coefficient (R) establish the capability (accuracy and skill) of the model in reproducing observed data.

However, these numbers are typically larger than the model's capability of describing the change between scenarios. In other words, inherent error in the model in the inputs (river inflow, ocean boundary conditions including water level and salinity, wind field etc.) is assumed to cancel out as those remain unchanged between the scenarios. The remaining error or uncertainty is then with the model parameters such as bed friction, turbulence closure terms for horizontal and vertical diffusion. These are also unchanged between the scenarios. For this study, without considering climate change scenarios, the only planned alternative runs are with respect to change in flows entering or withdrawn from the system. Uncertainty between model scenarios is then only tied to uncertainty in estimating flows. A traditional Monte Carlo style uncertainty analysis of model calibration parameters is beyond the scope of this study. All scenario runs were conducted on the same set of HPC nodes, and repeated sensitivity tests confirmed that the results were 100% reproducible for each run—that is, identical model output was produced for repeated executions. Therefore, any differences in model results among the scenario runs for the same flow year were solely due to changes in flow inputs.

Despite considerable effort to improve the model grid representation in intertidal zones, model performance—particularly for salinity simulations in intertidal wetlands and velocity simulations in small-scale channels (on the order of several meters wide)—could be further enhanced with higher-fidelity bathymetric data. For example, the lidar datasets used to represent ground elevations in intertidal zones and floodplains tend to overestimate bare-earth elevations in vegetated areas due to inherent limitations in penetrating vegetation cover. Additionally, lidar is generally ineffective at capturing channel bottom elevations in small waterways where water is present during data acquisition. These factors result in modeled bathymetry that is often too shallow for channels and artificially high elevations for wetland areas, which are likely to reduce saltwater intrusion in the intertidal zone and lead to salinity underpredictions. They also contribute to a negative bias in water level predictions during high tides.

Lastly, all hydrodynamic model simulations conducted in this study assume a steady topo-bathymetry in the hydrodynamic model grid, which, however, can introduce significant errors and uncertainty in simulation results, particularly when assessing future conditions. In dynamic estuarine and deltaic systems like the Skagit, changes in bathymetry due to sediment deposition, erosion, sea-level rise, and restoration activities can substantially alter flow patterns, salinity distribution, and inundation extent. By neglecting these morphological changes, the model may misrepresent key physical processes and feedback, leading to reduced accuracy and confidence in long-term projections. Therefore, incorporating anticipated or adaptive bathymetric changes is critical for improving the reliability for long-term future scenario modeling.

Implications for water and salmon management

The model updates, which include changes in grid resolution, use of SCHISM, and addition of water uses, may affect some quantitative outcomes, but the overall conclusions from previous restoration analyses are likely to remain consistent. However, the improved resolution and processes are able to provide greater confidence and detail in assessing localized impacts and system responses.

Predicting hydrodynamic responses in Skagit Bay is critical for understanding and managing the health of both vegetation and salmon populations in the system, which in turn influences the broader ecological health of the Salish Sea. Water movement influences salinity, sediment transport, and inundation patterns, all of which shape the distribution and productivity of estuarine vegetation such as eelgrass and marsh plants. These habitats provide essential nursery and foraging areas for juvenile salmon, particularly Chinook, as they transition from freshwater to marine environments. Changes in tidal flows, freshwater input, or restoration efforts can alter these physical conditions, potentially impacting habitat availability and quality. Accurate hydrodynamic modeling helps guide effective restoration and management strategies that support both ecological function and species recovery.

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Appendix 2.1

This section includes additional tables and figures for this hydrodynamic modeling study.

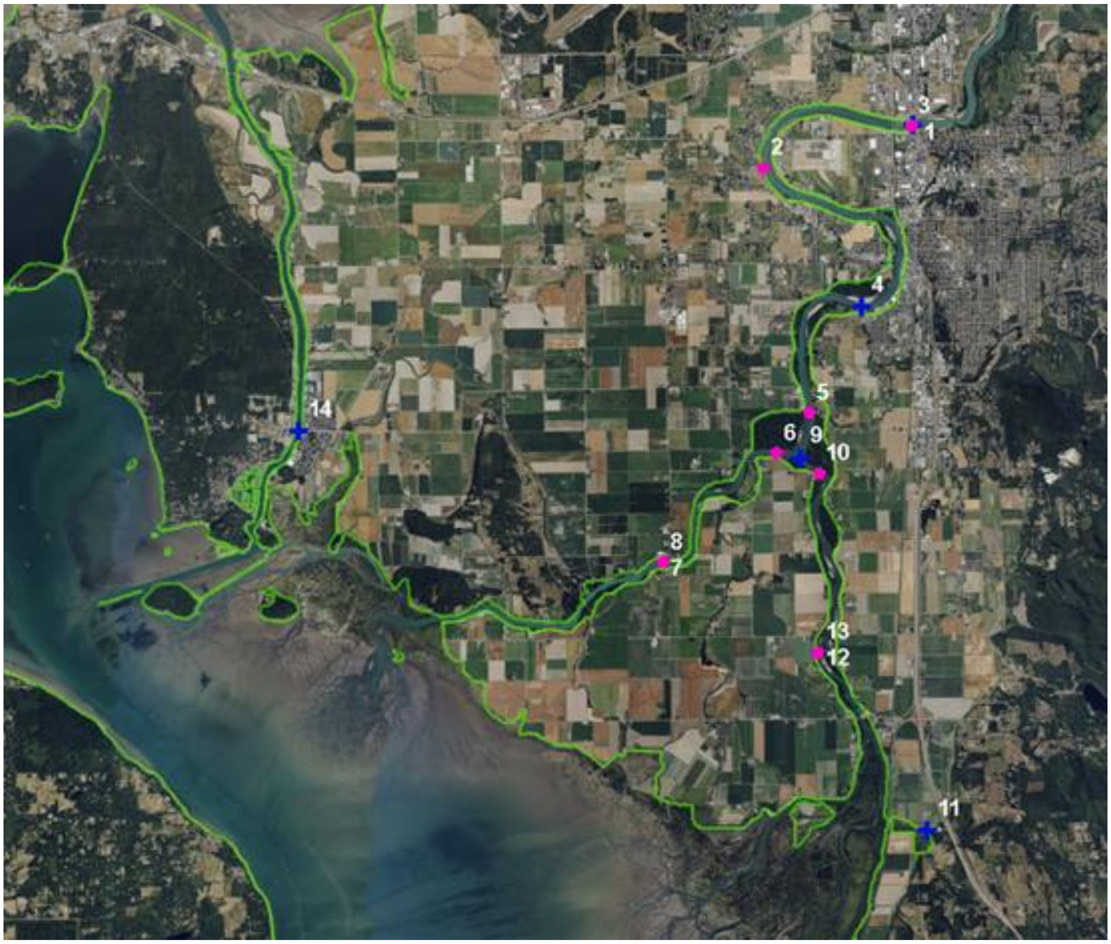

Figure 2.1.1. Flow input/withdrawal locations (blue "+" symbols denote flow input and magenta dots denote flow withdrawal). For details, please refer to Table 1.3 in Chapter 1.

Table 2.1.1. Hydrodynamic model scenario runs conducted.

ID	Scenario Name	Model Run Name	River Input/Withdrawal Description
1	Y2019 Calibration	Baseline_2019	USGS gaged flow at Mount Vernon Only
2	Y2019 Baseline	Baseline_2019_FS	USGS gaged flow at Mount Vernon + Fisher Creek
3	Y2019 Current water use	Current_2019	USGS gaged flow + Fisher Creek + current water withdrawals (Current_2019)
4	Y2019 Water use Alt 1	Current_noIFR_2019	USGS gaged flow + Fisher Creek + current water withdrawals (Current_noIFR_2019)
5	Y2019 – Water use Alt 2	Futrue_noIFR_2019	USGS gaged flow + Fisher Creek + future water withdrawals (Future_noIFR_2019)
6	Y2015 Baseline	Baseline_2015_FS	USGS gaged flow at Mount Vernon + Fisher Creek
7	Y2015 Current water use	Current_2015	USGS gaged flow + Fisher Creek + current water withdrawals (Current_2015)
8	Y2015 Water use Alt 1	Current_noIFR_2015	USGS gaged flow + Fisher Creek + current water withdrawals (Current_noIFR_2015)
9	Y2015 Water use Alt 2	Futrue_noIFR_2015	USGS gaged flow + Fisher Creek + future water withdrawals (Future_noIFR_2015)
10	Y2010 Baseline	Baseline_2010_FS	USGS gaged flow at Mount Vernon + Fisher Creek
11	Y2010 Current water use	Current_2010	USGS gaged flow + Fisher Creek + current water withdrawals (Current_2010)
12	Y2010 Water use Alt 1	Current_noIFR_2010	USGS gaged flow + Fisher Creek + current water withdrawals (Current_noIFR_2010)

13	Y2010 Water use Alt 2	Futrue_noIFR_2010	USGS gaged flow + Fisher Creek + future water withdrawals (Future_noIFR_2010)
14	Y2010 – Unregulated flow	UnRegulated_2010	DHSVM Mount Vernon flow only
15	Y2010 – Calibration	Baseline_2010	USGS gaged Mount Vernon flow only
16	Y2010 – Unregulated flow (modified)	UnRegulated_2010_FS	DHSVM Mount Vernon flow + Fisher Creek discharge
17	Y2015 Calibration	Baseline_2015	USGS gaged Mount Vernon flow only

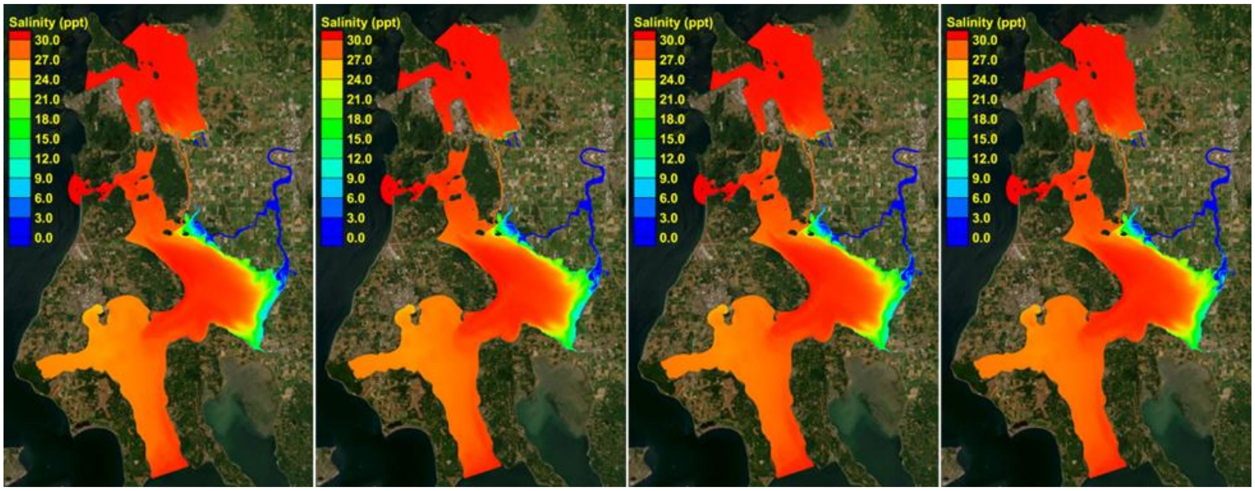

Figure 2.1.2. Maximum surface layer salinity during May – September (2019). From left to right (Baseline, Current, Alt1, and Alt2).

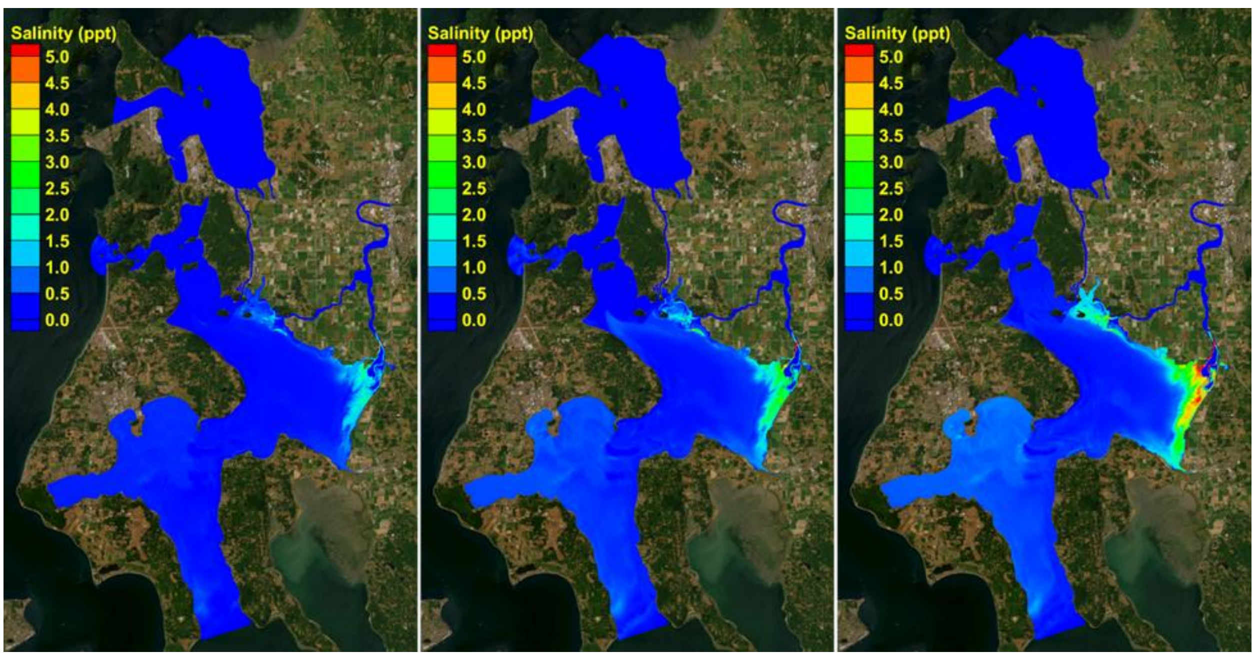

Figure 2.1.3. Changes in maximum surface layer salinity from the baseline condition to three water withdrawal scenarios during May – September (2019). From left to right (Current, Alt1, and Alt2).

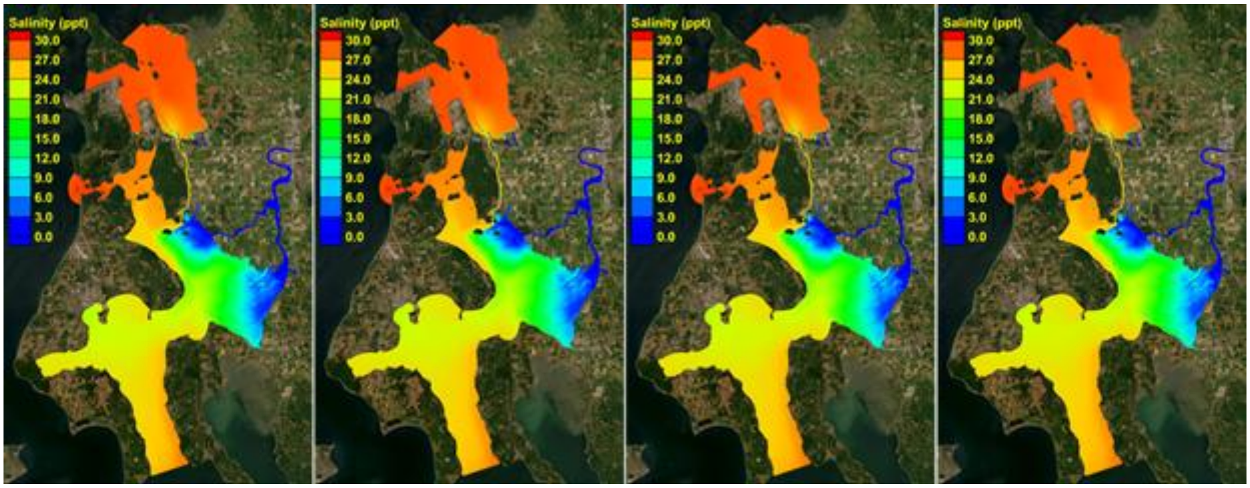

Figure 2.1.4. Mean surface layer salinity during May – September (2019). From left to right (Baseline, Current, Alt1, and Alt2).

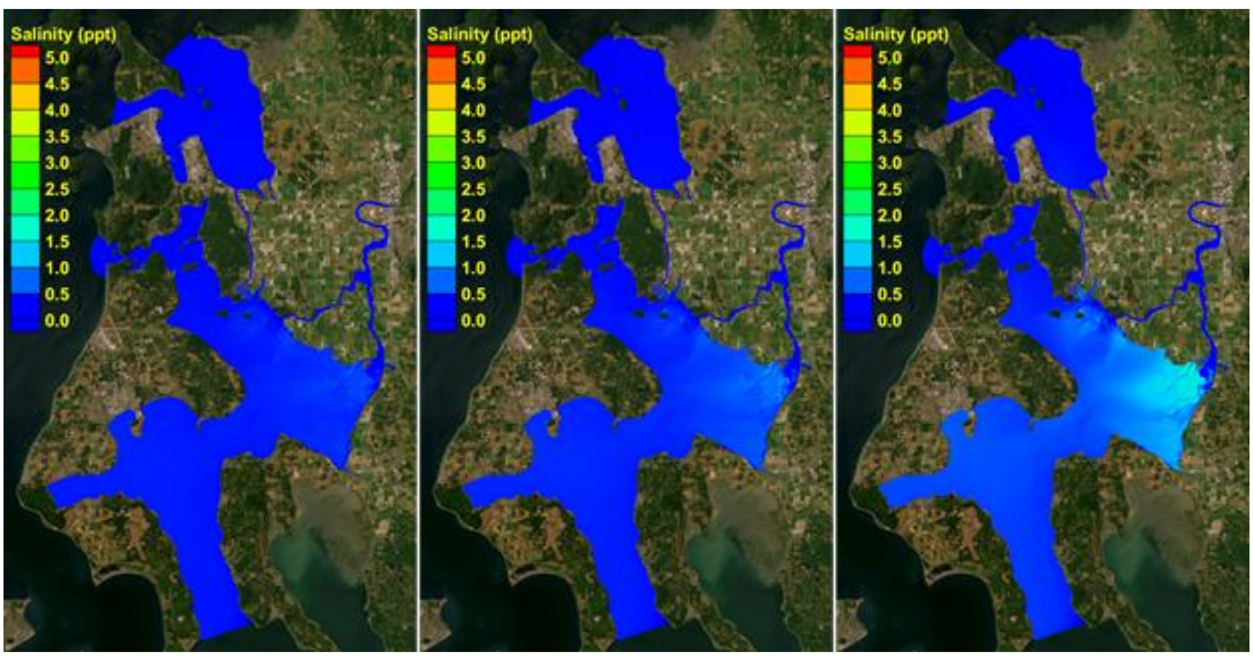

Figure 2.1.5. Changes in mean surface layer salinity from the baseline condition to three water withdrawal scenarios during May – September (2019). From left to right (Current, Alt1, and Alt2).

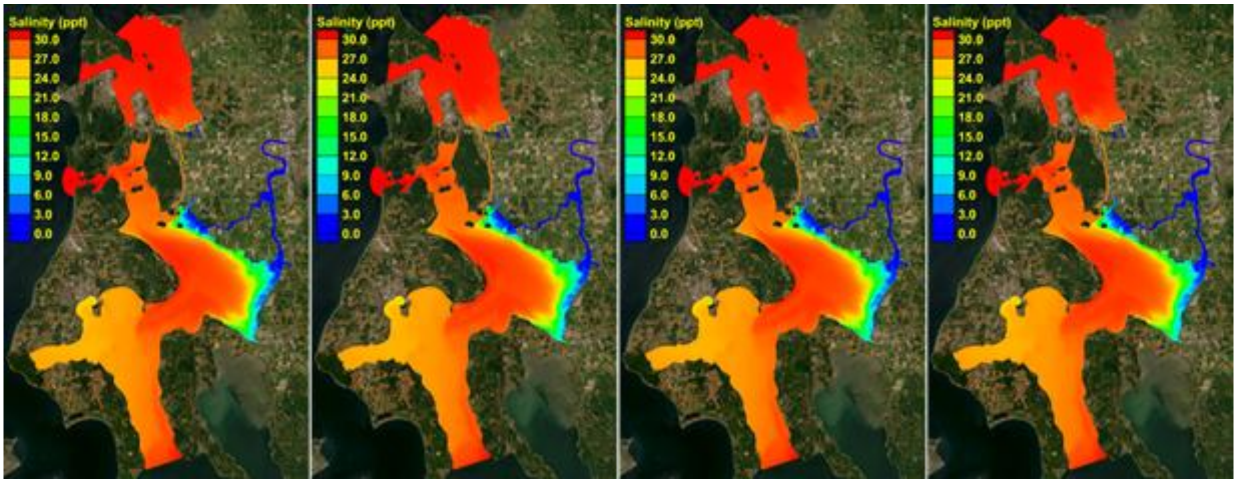

Figure 2.1.6. Maximum surface layer salinity during May – September (2010). From left to right (Baseline, Current, Alt1, and Alt2).

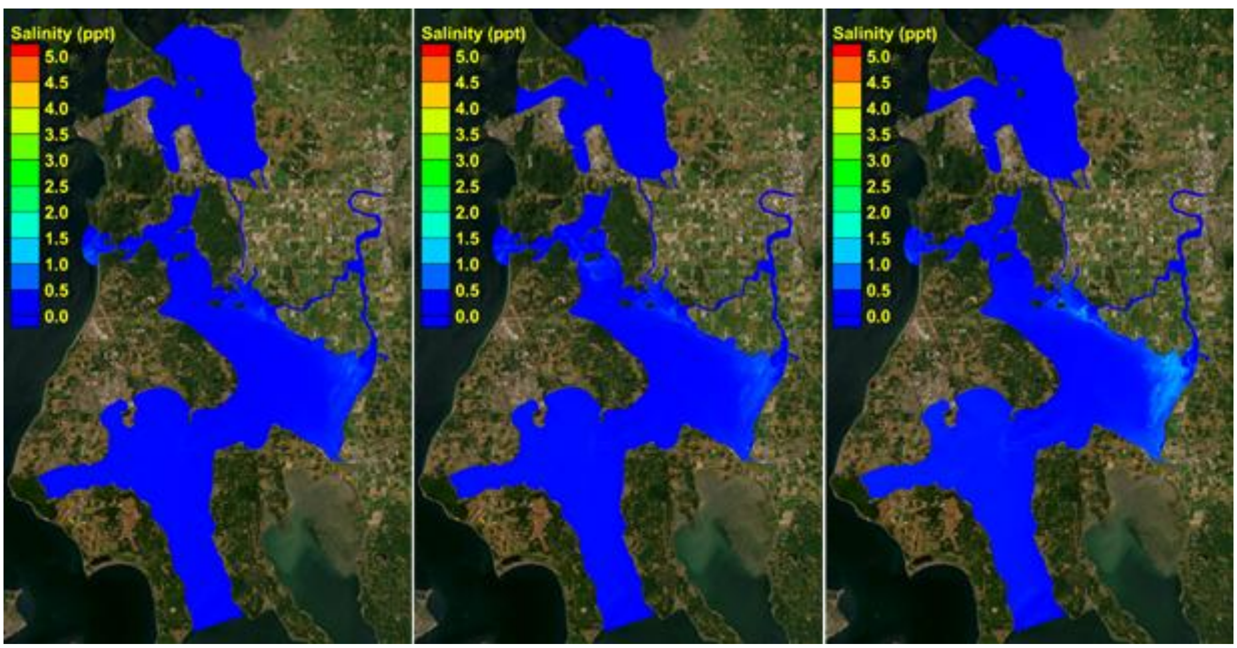

Figure 2.1.7. Changes in maximum surface layer salinity from the baseline condition to three water withdrawal scenarios during May – September (2010). From left to right (Current, Alt1, and Alt2).

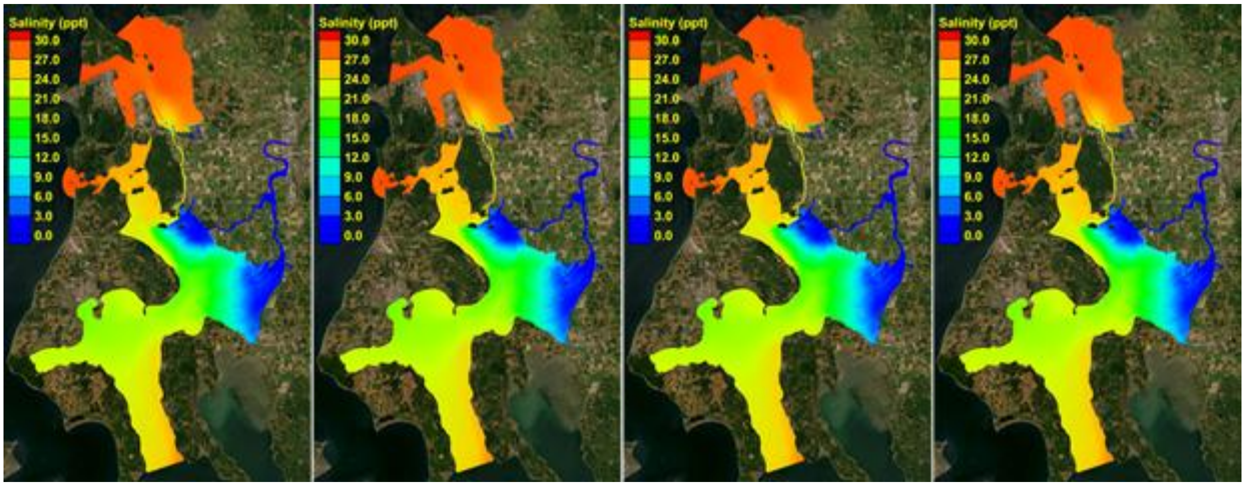

Figure 2.1.8. Mean surface layer salinity during May – September (2010). From left to right (Baseline, Current, Alt1, and Alt2).

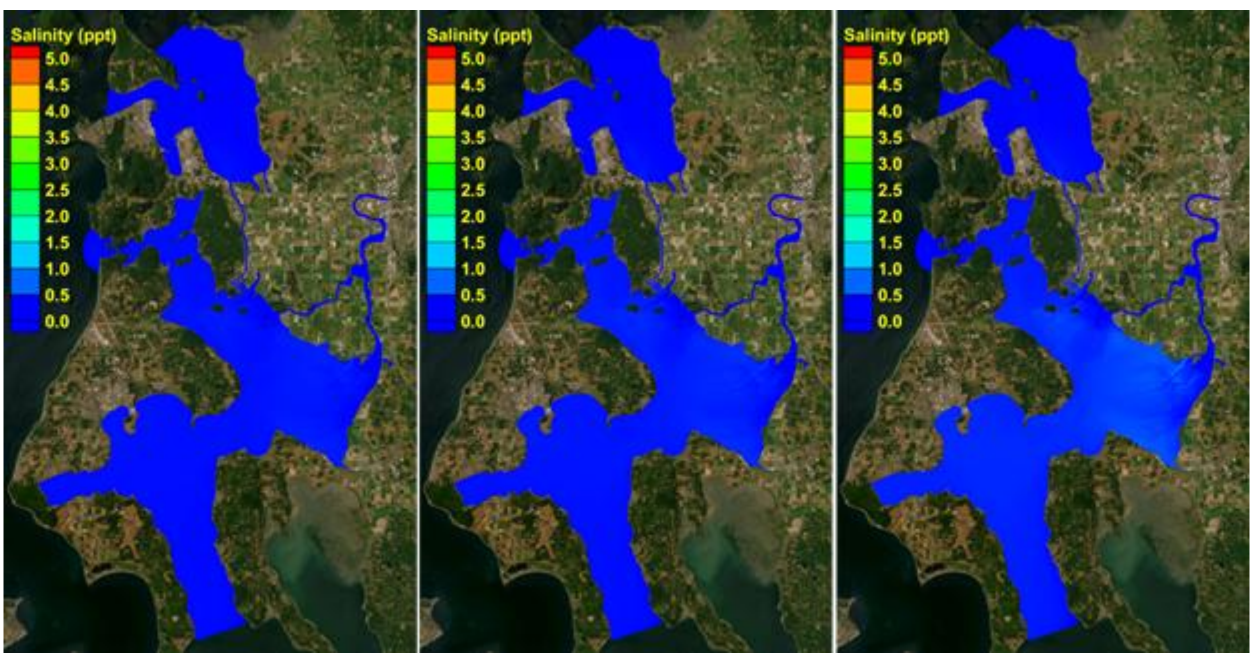

Figure 2.1.9. Changes in mean surface layer salinity from the baseline condition to three water withdrawal scenarios during May – September (2010). From left to right (Current, Alt1, and Alt2).

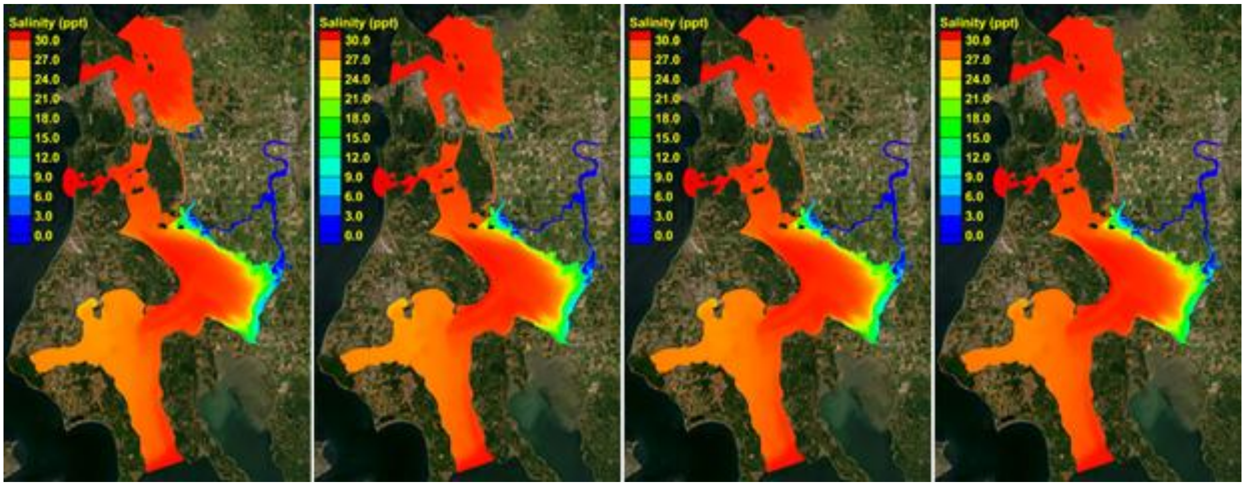

Figure 2.1.10. Maximum surface layer salinity during May – September (2015). From left to right (Baseline, Current, Alt1, and Alt2).

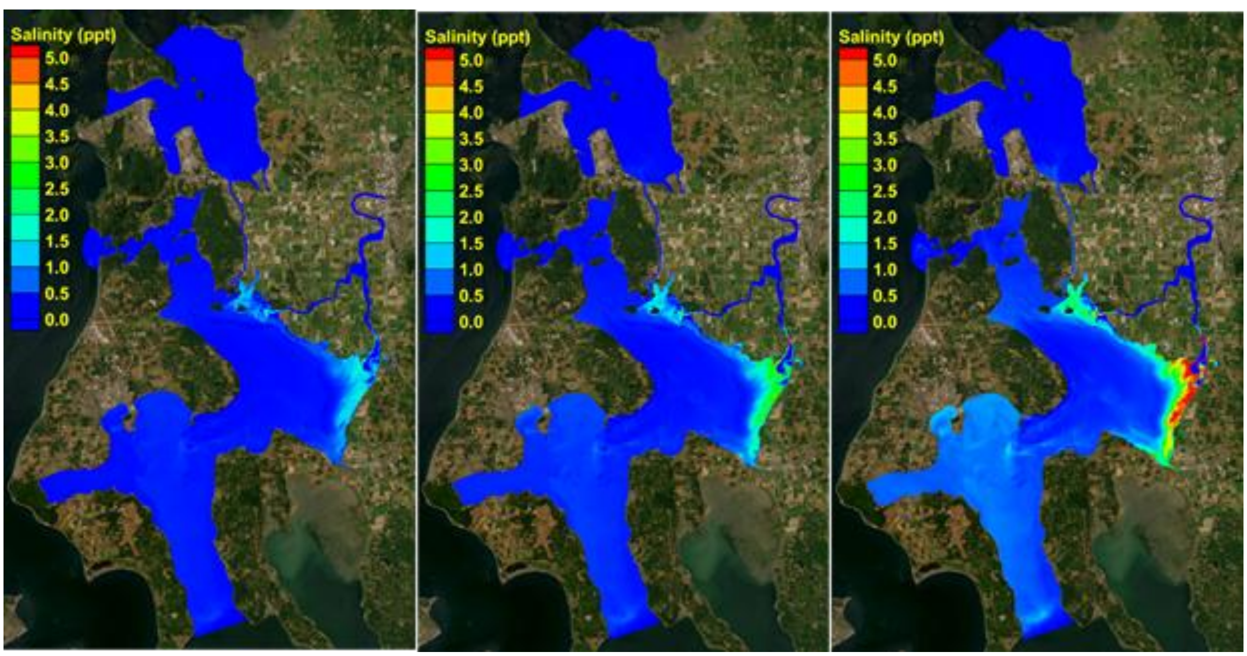

Figure 2.1.11. Changes in maximum surface layer salinity from the baseline condition to three water withdrawal scenarios during May – September (2015). From left to right (Current, Alt1, and Alt2).

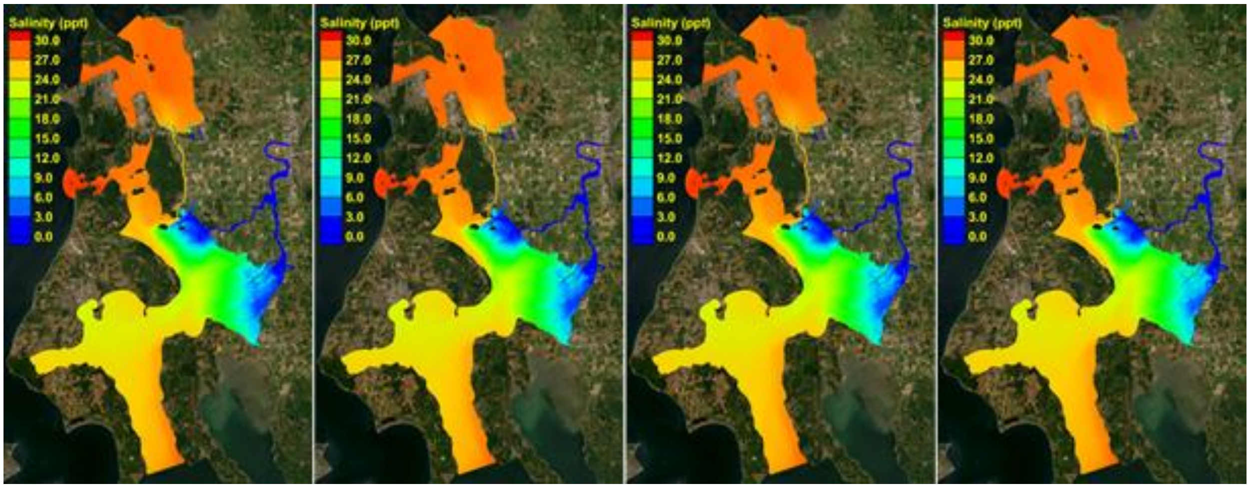

Figure 2.1.12. Mean surface layer salinity during May – September (2015). From left to right (Baseline, Current, Alt1, and Alt2).

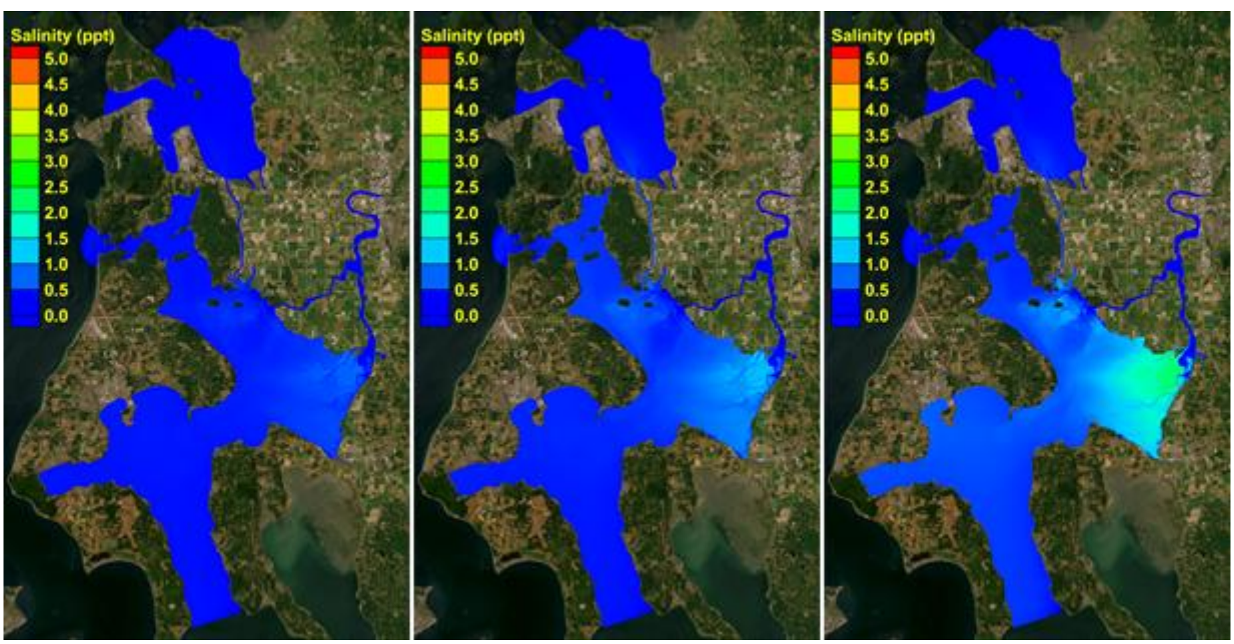

Figure 2.1.13. Changes in mean surface layer salinity from the baseline condition to three water withdrawal scenarios during May – September (2015). From left to right (Current, Alt1, and Alt2).

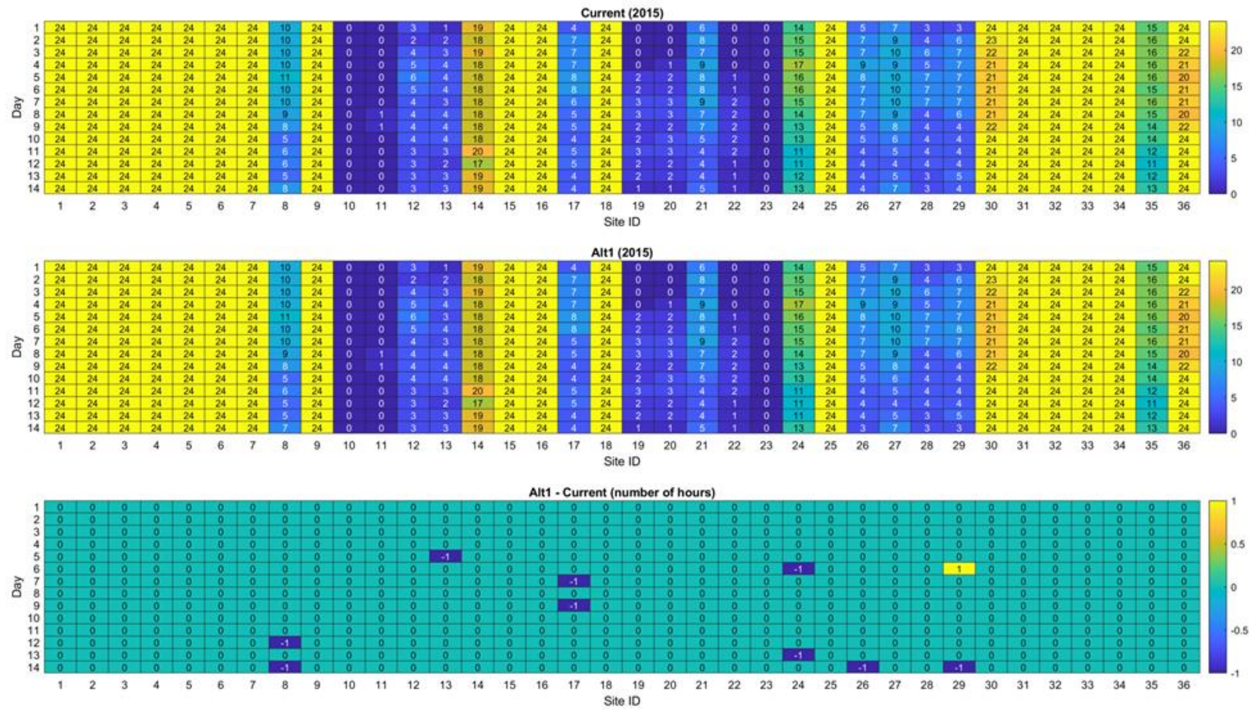

Figure 2.1.14. Daily cumulative occurrence frequency of water depth exceeding 30 cm during the 5/30 – 6/12 time window for Year 2015 at all 36 sites (top panel - current condition; middle panel - Alternative 1 scenario; bottom panel - differences between Alt 1 and current condition).

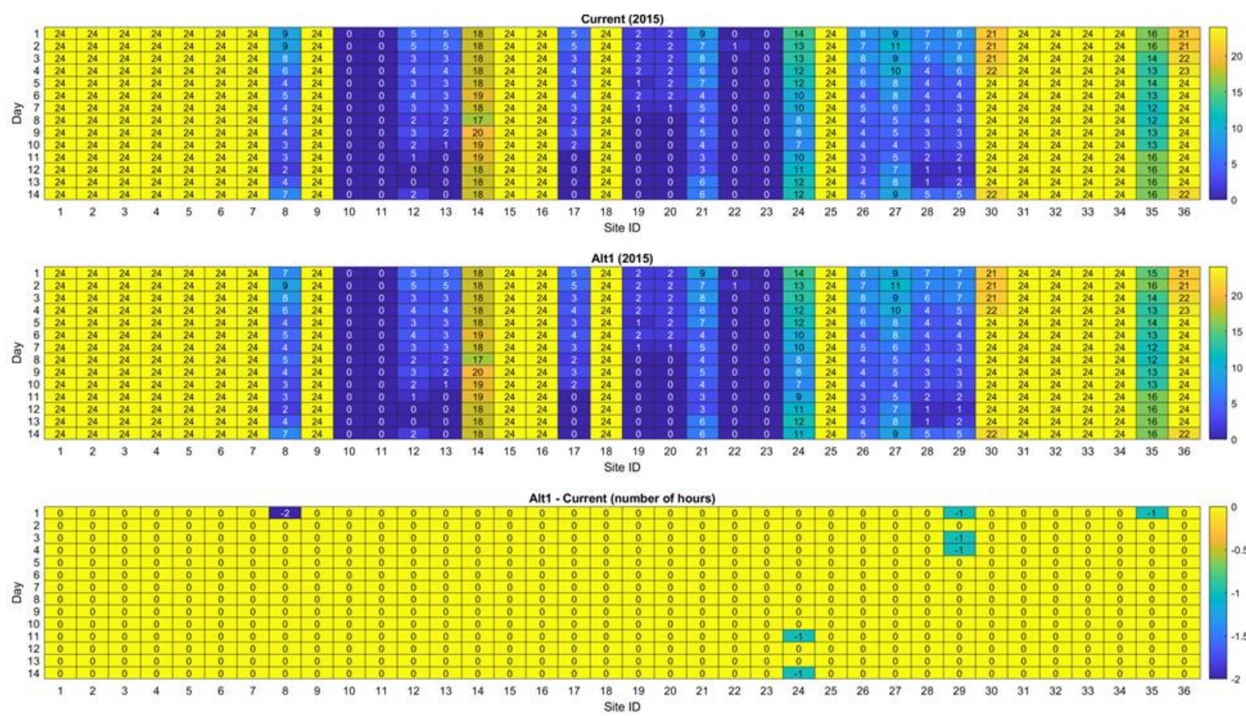

Figure 2.1.15. Daily cumulative occurrence frequency of water depth exceeding 30 cm during the 7/15 – 7/28 time window for Year 2015 at all 36 sites (top panel - current condition; middle panel - Alternative 1 scenario; bottom panel - differences between Alt 1 and current condition).

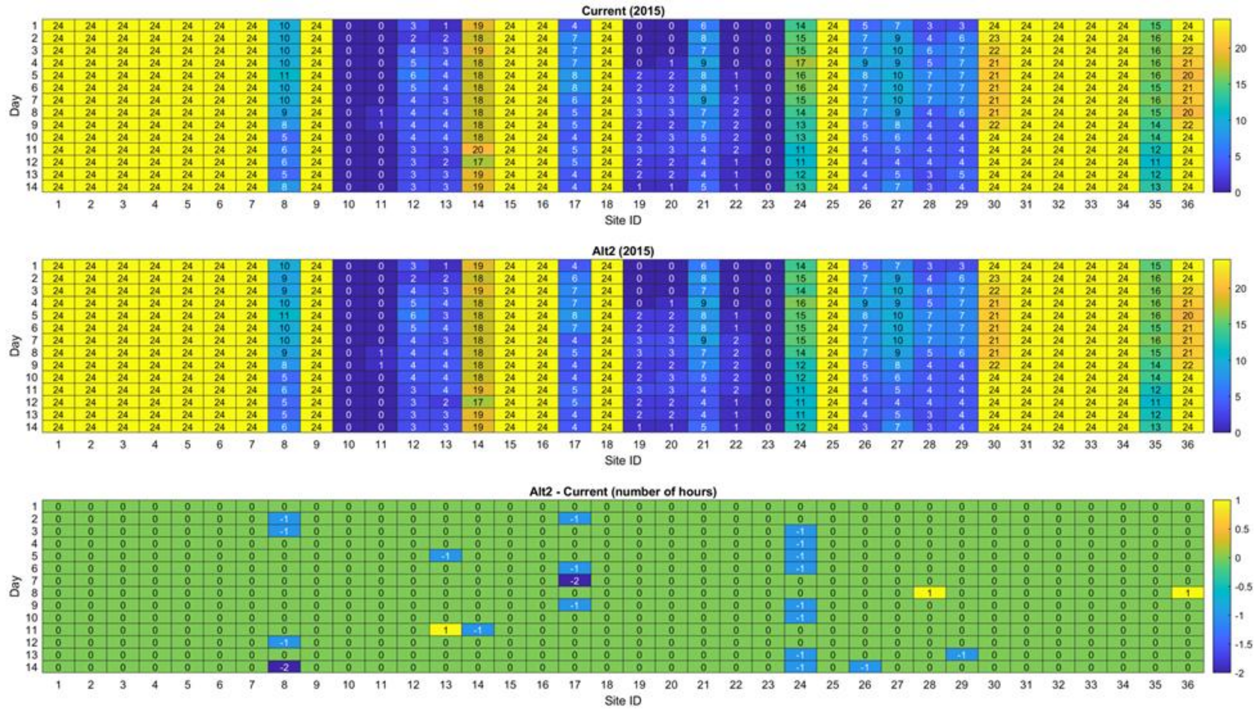

Figure 2.1.16. Daily cumulative occurrence frequency of water depth exceeding 30 cm during the 5/30 – 6/12 time window for Year 2015 at all 36 sites (top panel - current condition; middle panel - Alternative 2 scenario; bottom panel - differences between Alt 2 and current condition).

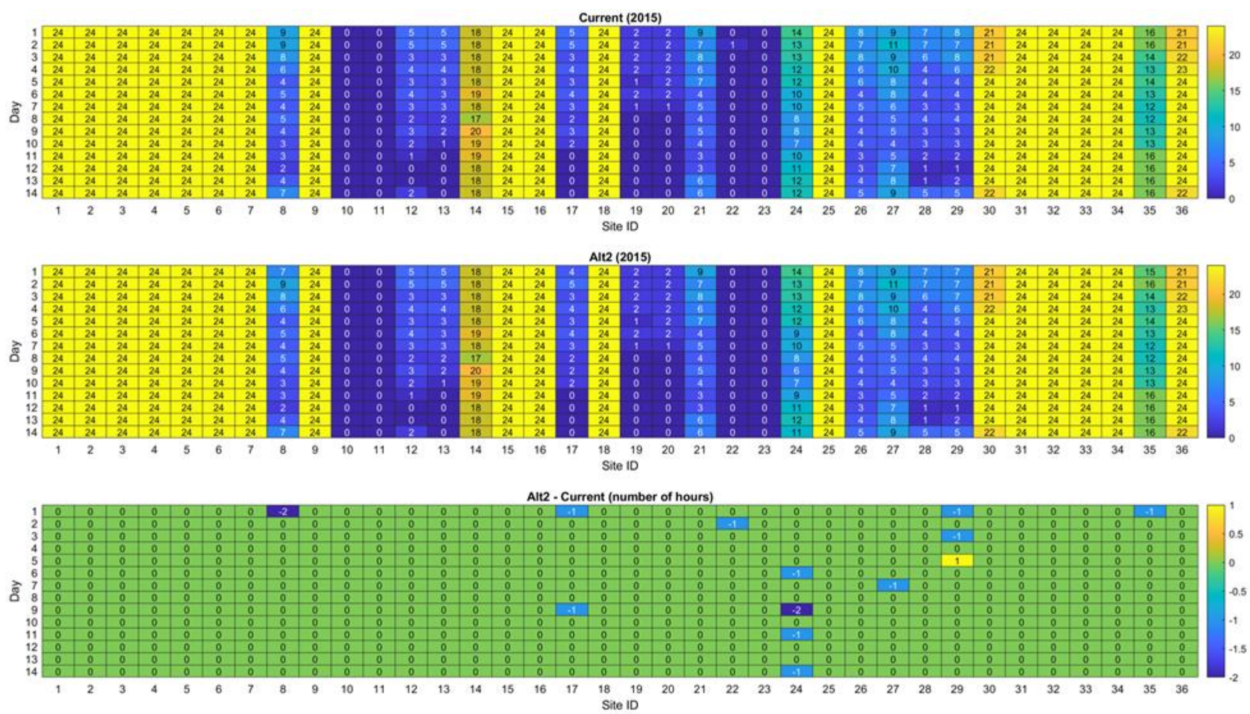

Figure 2.1.17. Daily cumulative occurrence frequency of water depth exceeding 30 cm during the 7/15 – 7/28 time window for Year 2015 at all 36 sites (top panel - current condition; middle panel - Alternative 2 scenario; bottom panel - differences between Alt 2 and current condition).

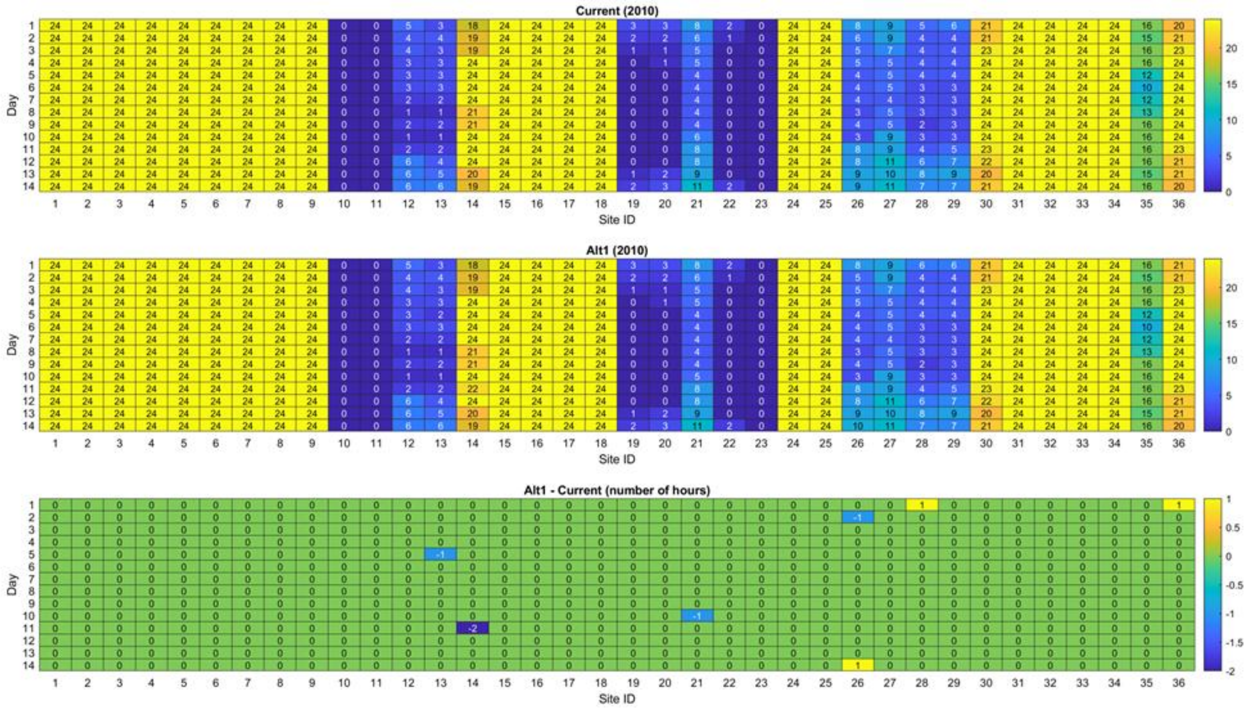

Figure 2.1.18. Daily cumulative occurrence frequency of water depth exceeding 30 cm during the 5/30 – 6/12 time window for Year 2015 at all 36 sites (top panel - current condition; middle panel - Alternative 1 scenario; bottom panel - differences between Alt 1 and current condition).

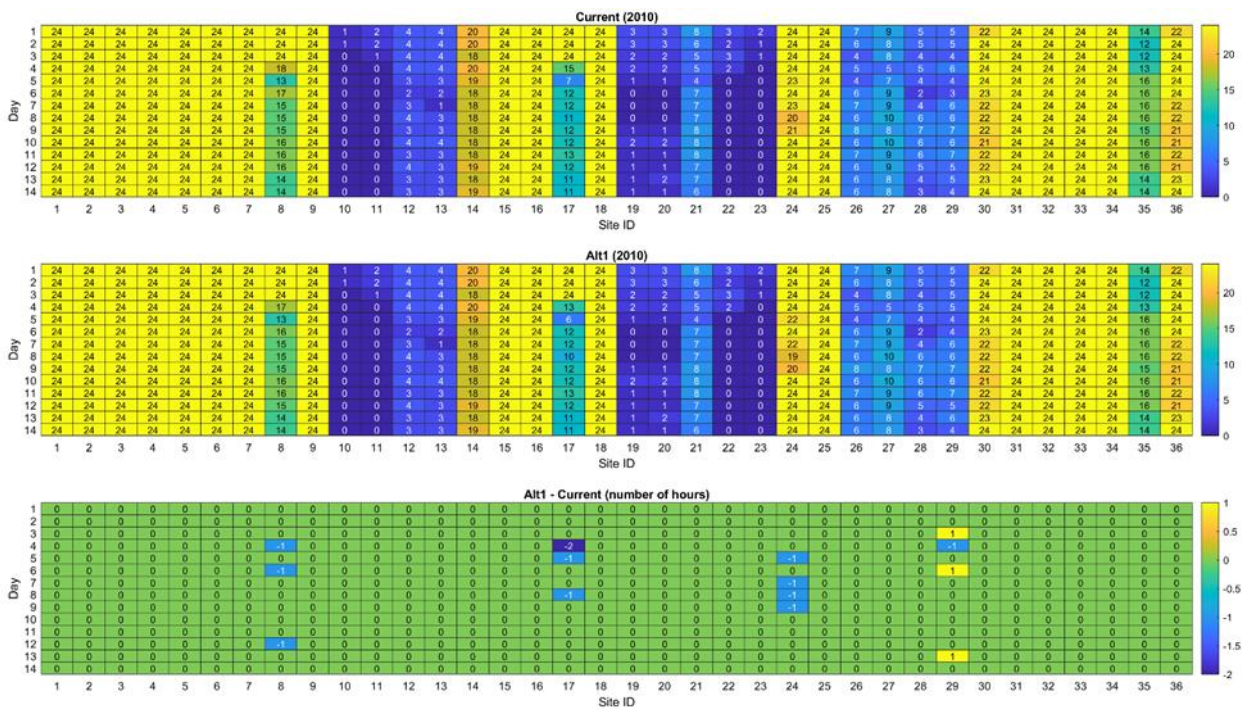

Figure 2.1.19. Daily cumulative occurrence frequency of water depth exceeding 30 cm during the 7/15 – 7/28 time window for Year 2015 at all 36 sites (top panel - current condition; middle panel - Alternative 1 scenario; bottom panel - differences between Alt 1 and current condition).

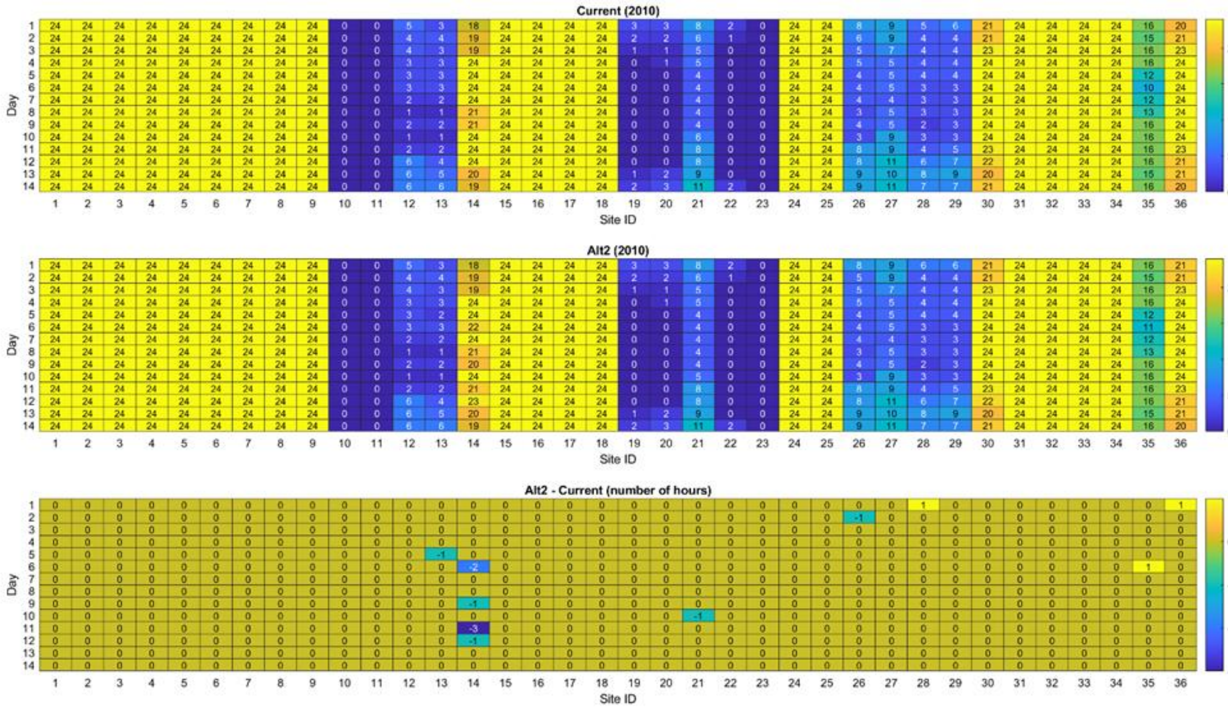

Figure 2.1.20. Daily cumulative occurrence frequency of water depth exceeding 30 cm during the 5/30 – 6/12 time window for Year 2015 at all 36 sites (top panel - current condition; middle panel - Alternative 2 scenario; bottom panel - differences between Alt 2 and current condition).

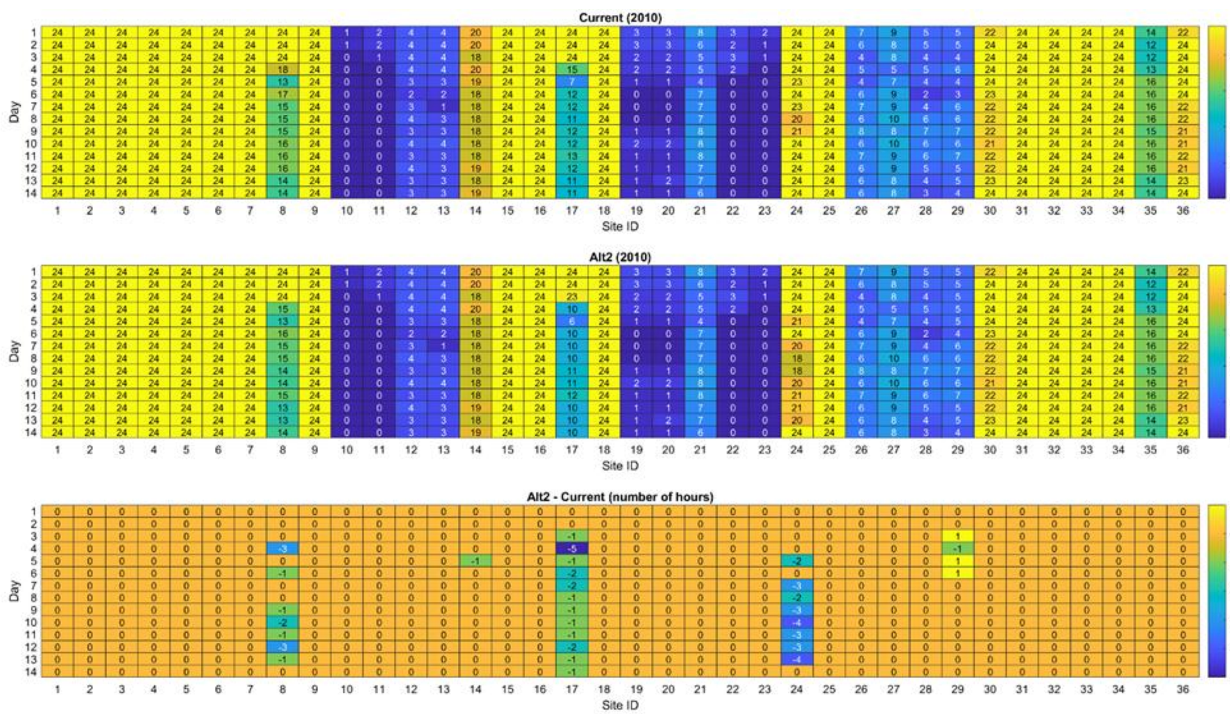

Figure 2.1.21. Daily cumulative occurrence frequency of water depth exceeding 30 cm during the 7/15 – 7/28 time window for Year 2015 at all 36 sites (top panel - current condition; middle panel - Alternative 2 scenario; bottom panel - differences between Alt 2 and current condition).

3. Predicted vegetation responses to changes in tidal delta water levels and salinities

Greg Hood

Introduction

Coastal river deltas are areas of high agricultural and fisheries production throughout the world, but because of their low elevation and flat relief they are also highly vulnerable to direct and indirect impacts of management (Overeem & Syvitski 2009; Loucks 2019). One often highly impactful form of management is water diversion for human needs, an extreme example of this is the Colorado River and its nearly extinct delta from overallocation of river flows (Glenn et al. 1996; Pitt 2001). Even modest reductions in freshwater inflows can alter salinity gradients, sediment delivery, and nutrient fluxes, thereby influencing the distribution and productivity of tidal marsh vegetation (Day et al. 2000; Barendregt & Swarth 2013). For example, low-volume irrigation or municipal withdrawals upstream can reduce peak flows, extending periods of saline intrusion into marsh channels and shifting plant community composition from freshwater- to salt-tolerant species (Williams & Orr 2002; Craft et al. 2009).

Vegetation plays a fundamental role in the ecology and geomorphology of tidal marshes. It traps suspended sediments facilitating marsh accretion; binds soils in its roots, stabilizing shorelines and tidal channel cross-sections; dampens storm waves to protect shorelines; filters and transforms nutrients; and sustains a marsh foodweb for herbivores (e.g., insects, ducks, geese, beaver, deer) and detritivores (polychaetes, arthropods, and other invertebrates), and through them important predators, such as shorebirds and fish, including threatened Chinook salmon.

The question motivating this work is whether seemingly modest water diversions can nevertheless produce significant ecological impacts in river delta marshes. We examine three water use scenarios in the Skagit Delta: [a] Current water use, which modifies flow on a seasonal and daily basis, with junior agricultural water rights subject to interruption when river flow drops below levels specified by the Skagit Instream Flow Rule; [b] Alternative 1, in which 200 CFS (1.2% of mean annual flow) of junior water rights was rendered non-interruptible; and [c] Alternative 2, in which 200 CFS of junior water rights as well as 390 CFS of additional irrigation needs (a total of 3.6% of mean annual flow) were made uninterruptible based on projected additional water needs for local agriculture. We also assumed that the city of Anacortes' maximum water right (85 CFS) was utilized.

Soil porewater salinity and marsh surface elevation have been frequently shown to control tidal marsh vegetation distributions, creating well-known zonation patterns in tidal marsh species distributions (Adams 1963; Ewing 1983; Snow and Vince 1984; Bertness and Ellison 1987;

Pennings and Callaway 1992; Crain et al. 2004). The influence of elevation is mediated by the frequency and duration of tidal inundation which creates oxygen stress on root respiration. Other factors can also influence tidal marsh vegetation, including soil types, physical disturbance by logs, physical disturbance by plant or algal wrack, wave energy, herbivory, competition, and facilitation. Nevertheless, salinity and elevation are the two fundamental and most important influences on tidal marsh vegetation distributions. These two factors can also be more easily measured or estimated from remote imagery (elevation, via lidar) or field sampling (salinity) than can many of the other influences. Furthermore, the proposed changes in water management, Alternatives 1 and 2, may have direct impacts on marsh salinity by reducing freshwater delivery to the delta, but they are unlikely to affect soil type, physical disturbance, wave energy, etc. Thus, to anticipate potential impacts of water management changes on Skagit tidal marsh vegetation we focused on marsh elevation and salinity as the most relevant predictors of species distributions. Consequently, data were collected on these two environmental factors and on plant species distributions to develop a predictive statistical model of tidal marsh plant species distributions in the Skagit Delta. This model was then applied to the management alternatives to evaluate potential changes to the vegetation communities of the Skagit Delta.

Methods

Data Collection

Soil porewater salinity data was collected throughout the Skagit marshes in a 160-m sampling grid by digging soil pits to a depth of 50 cm, collecting the seepage water in glass vials, allowing suspended sediment to settle, and then measuring salinity with an optical refractometer. All samples were collected at low tide, and river discharge was noted during the collection period. Of 529 sampling points in the delta (Fig. 3.1), 248 (47%) were sampled repeatedly at different river discharges to try to control for the effects of river discharge on soil porewater salinity. River discharges ranged from 4,000 to 30,000 CFS during salinity sampling over several different dates. Most sampling took place around flows of 9,000; 14,000; and 20,000 CFS.

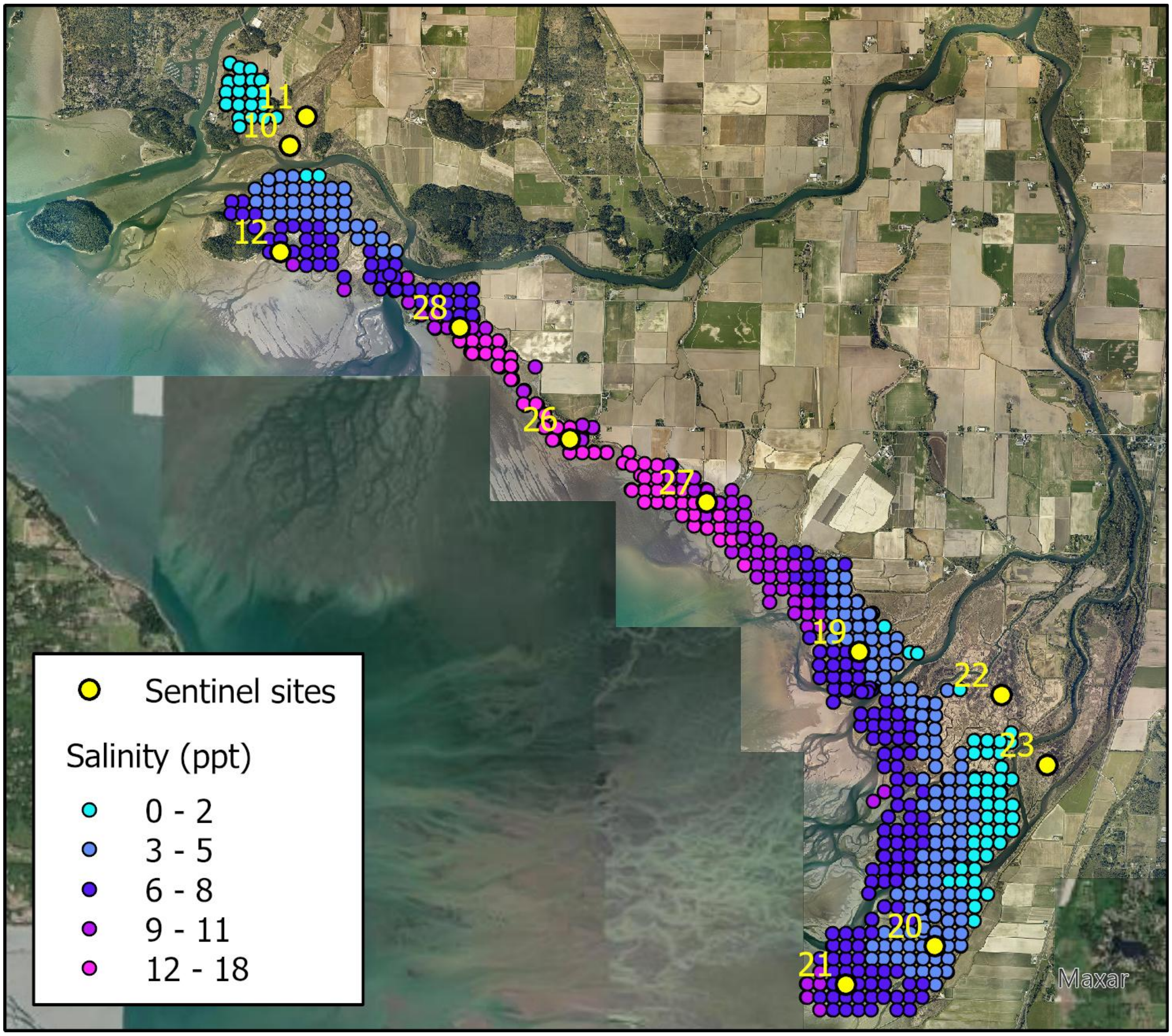

Figure 3.1. Locations of soil porewater salinity sampling points and sentinel sites used for tidal marsh vegetation prediction. 10 = Sullivan Slough shrubs, lower; 11 = Sullivan Slough shrubs, upper; 12 = Ika Island low marsh; 19 = Old Wiley Slough; 20 = Tom Moore Slough, upper; 21 = Tom Moore Slough, lower; 22 = middle South Fork shrubs; 23 = Milltown reference shrubs; 26 = Hall Slough; 27 = Browns Slough; 28 = Rawlins Road low marsh.

Porewater salinity was standardized to 9,000 CFS river discharge, which is characteristic of normal late summer low flows, to align the vegetation model and the hydrodynamic model (Ch. 2) with regard to seasonal flows. Local linear regression models of salinity versus flow were generated for sub-regions of the tidal marshes where salinity was variable, and these were used to estimate porewater salinity at 9,000 CFS unless those points had been sampled at 9,000 CFS. The hydrodynamic model (Ch. 2) predicted late summer flow conditions for each water management alternative by sentinel site, because this time period, due to its typical low flows, was considered the most stressful for vegetation and thus likely the most determinative of species distributions.

Vegetation and elevation data were collected simultaneously by walking marsh transects with a real-time kinetic global positioning system (RTK-GPS, 3-cm vertical and horizontal accuracy). Survey points were spaced 35 paces (approximately 20 m) along each transect. At each survey point the dominant plant species was noted. However, only every other point was used to parameterize the NPMR model, so that data point spacing was approximately 40 m. This spacing reduced the possibility of autocorrelation due to clonal plant growth. These data (Fig. 3.2; N = 3750) have been collected over several years for a variety of marsh monitoring and assessment projects. Thus, they were not tailored to this particular modeling effort. GIS was used to create a 50-m fishnet grid that covered the Skagit marshes, and each grid cell was assigned a porewater salinity value according to the nearest salinity sampling point. The

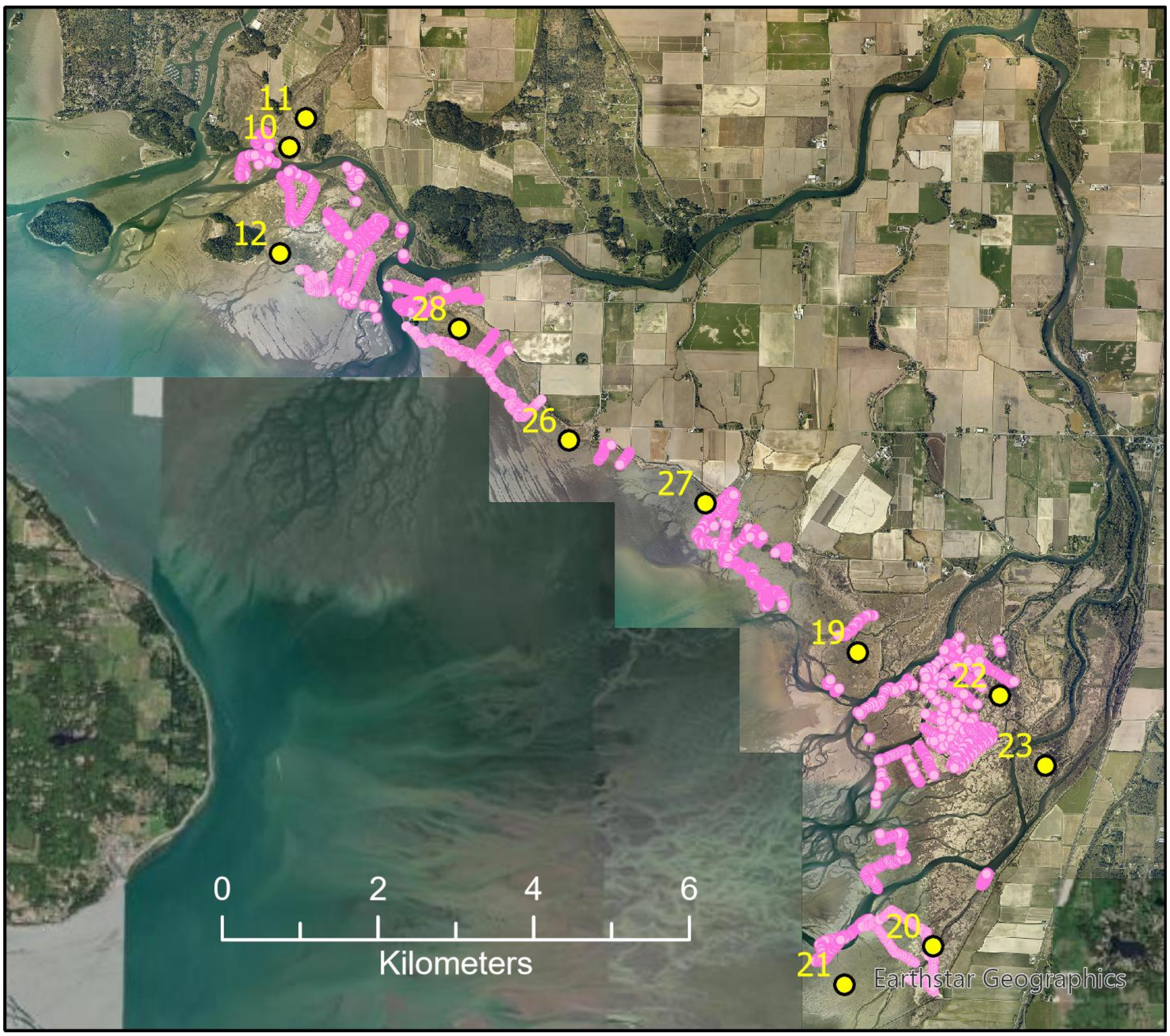

Figure 3.2. Locations of vegetation/elevation sampling points (pink) in the focal area of the Skagit Delta (N = 3400). Additional sampling occurred in peripheral areas (Swinomish Channel, Telegraph Slough, Padilla Bay) where salinities were higher (N = 350). Sentinel sites for vegetation prediction (yellow) are shown for reference.

result was a collection of 1538 sampling units (50-m grid cells) that each contained a value for dominant plant species, elevation, and porewater salinity. Twenty-eight plant species occurred as dominants in the data set, but only eleven occurred with sufficient frequency ($n > 30$) to allow their distributions to be modeled. These consisted of *Agrostis stolonifera* (AGST, creeping bentgrass [a naturalized species], $n = 63$); *Bolboschoenus maritimus* (BOMA, maritime bulrush, $n = 42$); *Carex lyngbyei* (CALY, marsh sedge, $n = 251$); *Distichlis spicata* (DISP, saltgrass, $n = 186$); *Juncus balticus* (JUBA, Baltic rush, $n = 31$); *Myrica gale* (MYGA, sweetgale, $n = 64$); *Salix* spp. (willow, $n = 47$); *Sarcocornia pacifica* (SAPA, pickleweed, $n = 103$); *Schoenoplectus pungens* (SCPU, three-square bulrush, $n = 235$); *S. tabernaemontani* (SCTA, soft-stem bulrush, $n = 122$); and *Typha angustifolia* (TYAN, narrow-leaf cattail [an invasive non-native species], $n = 234$).

Statistical Analysis

We chose to model plant distributions by non-parametric multiplicative regression (NPMR) because of the flexibility to characterize interacting factors unbounded by the simplified assumptions of parametric and linear models (McCune, 2006), and its better performance in tests with common statistical techniques, such as generalized linear models (GLMs) and generalized additive models (GAMs) (McCune 2011).

For each species of interest, NPMR binomial (presence/absence as a dominant) models were fitted and applied to predictor variables using the HyperNiche 2 software package (McCune & Mefford 2004; McCune 2006; Yost 2008) and its default settings (i.e., improvement criterion = 0, step size = 5% of predictor range, maximum allowable missing estimates = 10%, data/predictor ratio = 10, and minimum neighborhood size for acceptable model = *n* plots x 0.05). Because NPMR is non-parametric, it requires no assumptions regarding the shape of species response to environmental gradients. Predictor variables are considered multiplicatively, allowing the effect of one predictor to covary in complex ways with other predictors. NPMR is a local mean estimator, i.e., the proportion of a species occurrence in a locally defined environmental neighborhood is used to estimate the probability of occurrence. The environmental neighborhood consists of plots that lie close to the target site in multidimensional predictor space. The size of the neighborhood is defined by a tolerance range around the target site. The shape of the neighborhood diminishes gradually from the target point using weights based on a smoothing parameter, i.e., standard deviation of the Gaussian distribution. For each species, models were fitted through a leave-one-out cross-validation process, which guards against model overfitting and allows the error rate of the training data set to approximate the error rate of predictions.

For each modeled species, a stepwise free-search was used to seek a range of models with different combinations of predictors, neighborhood size, and tolerances. An optimum model was selected for each species, using log likelihood ratios (log B) which expresses the relative performance of a fitted model versus a “naïve” model. The naïve model is the species average frequency of occurrence in the dataset. Log B is an unbounded measure, so it can become very large when strong relationships are modeled with large data sets. Its utility as a descriptive statistic lies in the fact that it increases as the weight of evidence for the model increases.

Selected models were assessed using a Monte Carlo randomization test with 1000 runs to evaluate model fit. Evaluation metrics included log B, the area-under the curve (AUC) statistic, and the improvement %. AUC provides a threshold independent measure of presence–absence model quality (Hanley & McNeil 1982). An AUC of 0.5 indicates no model discrimination, 0.7 to 0.8 is considered acceptable discrimination, 0.8 to 0.9 is considered very good, and more than 0.9 is considered outstanding (1.0 = impossibly perfect). Improvement % is the ratio of plots that receive probability estimates considered improvements over the naïve model. Improvements are considered as the percentage of presence plots that have a higher probability estimate than observed prevalence, or plots with species absence having a lower probability estimate than observed prevalence.

Statistical Prediction

The NPMR model was applied to lidar-derived elevation and hydrodynamic model-derived (see Ch. 2 for details) salinity values for the three previously described water management alternatives and eleven sentinel sites distributed across the active Skagit delta (Figs. 3.1 and 3.2). Sentinel sites were chosen as likely to be among the most responsive to alterations in river hydrology. They consisted principally of areas that were either relatively low elevation, high salinity, or dominated by tidal shrub vegetation. Elevation values were generated from lidar data (Washington Geological Survey 2020), adjusted by shrub cover and change in water surface elevations predicted by the hydrodynamic model. Because water surface elevation change was predicted to be minimal, the lidar data was not adjusted for this factor, but shrubs can cause error in bare ground elevation estimates. To address this issue for shrub-dominated sites, existing RTK-GPS measurements in shrub areas were compared to lidar estimates at the same locations; a sample size of 86 paired comparisons showed that lidar overestimated bare ground elevation in shrub-dominated locations by an average of 25 cm. Thus, lidar data for shrub-dominated sentinel sites were adjusted by this amount to get an unbiased estimate of ground elevation. Salinity values for each sentinel site and management scenario were acquired from the hydrodynamic model and not adjusted.

Table 3.1. Summary of model evaluation metrics for each modeled species, ordered by AUC. Parameter tolerances reflect the resolution of local smoothing parameters (small values are good).

	N[1]	AUC	% Improvement vs. null model	logB	ChiSq	p	Elev. (m) tolerance	Salinity (ppt) tolerance
Schoenoplectus pungens	235	0.98	92%	204.2	940.2	<<0.0001	0.13	1.6
Distichlis spicata	186	0.96	88%	141.2	650.0	<<0.0001	0.13	1.6
Bolboschoenus maritimus	42	0.95	85%	35.0	161.3	<<0.0001	0.13	1.6
Juncus balticus	31	0.95	85%	29.9	137.89	<<0.0001	0.13	3.1
Sarcocornia pacifica	103	0.95	85%	82.1	378.25	<<0.0001	0.26	1.6
Agrostis stolonifera	63	0.91	80%	37.5	172.9	<<0.0001	0.13	1.6
Myrica gale	64	0.91	76%	37.2	171.1	<<0.0001	0.13	1.6
Salix spp.	47	0.90	73%	28.2	129.9	<<0.0001	0.13	1.6
Carex lyngbyei	251	0.90	75%	111.4	512.8	<<0.0001	0.13	1.6
Typha angustifolia	234	0.88	70%	94.3	434.1	<<0.0001	0.13	1.6
S. tabernaemontani	122	0.84	70%	45.0	207.3	<<0.0001	0.13	1.6

[1] N = number of points where the species was dominant out of 1,538 samples

Results

Model quality was generally outstanding with AUC values typically > 0.90 (Table 3.1). As expected, tidal marsh elevation and salinity were very important predictors of vegetation presence/absence, with the finest possible model tolerance (resolution) of 5% of the data range achieved for most of the predicted species. The data range was 2.6 m for elevation and 32 ppt for salinity. With local mean models, tolerance is inversely related to the importance of a variable.

Model-represented niche partitioning was consistent with many decades of field experience in the Skagit Delta (Fig. 3.3). However, the niche space had a distinct gap, in the vicinity of 20 ppt, between high salinity and lower salinity marshes. The lower salinity space represents data collected from the primary Skagit Delta marshes, i.e., at the mouths of the North and South Fork Skagit River distributaries and the bayfront between. This space is relatively speciose. The higher salinity marsh space was dominated by *Sarcocornia pacifica* (SAPA) and *Distichlis spicata* (DISP). This niche space represents data collected from higher salinity, peripheral parts

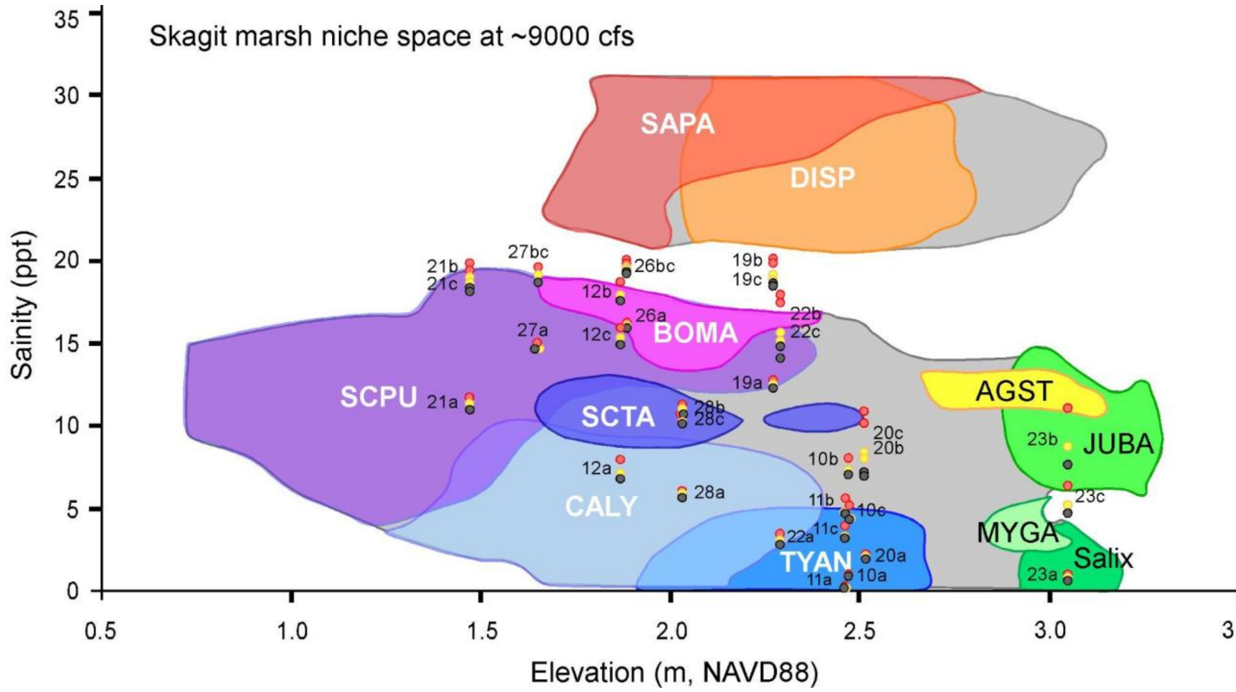

Figure 3.3. Vegetation niche space partitioning with sentinel sites (numbered as in Fig. 4.1), hydrologic year (a = 2010, b = 2015, c = 2019), and management alternatives (black = current conditions, yellow = Alternative 1, red = Alternative 2). Species polygons are bounded by the 30% frequency of occurrence isopleth, i.e., internal areas represent still higher frequencies of occurrence. Other isopleths are not shown to limit graphic complexity. AGST = *Agrostis stolonifera* (bentgrass); BOMA = *Bolboschoenus maritimus* (maritime bulrush); CALY = *Carex lyngbyei* (sedge); DISP =*Distichlis spicata* (saltgrass); JUBA = *Juncus balticus* (Baltic rush); MYGA = *Myrica gale* (sweetgale); *Salix* spp. (willow); SAPA = *Sarcocornia pacifica* (pickleweed); SCPU = *Schoenoplectus pungens* (three-square); SCTA = *S. tabernaemontani* (soft-stem bulrush); TYAN = *Typha angustifolia* (non-native cattail).

of the Skagit Delta, i.e., marshes along the Swinomish Channel, remnant tidal portions of Telegraph Slough, and northeastern Padilla Bay saltmarshes, areas that are spatially distinct and disjunct from the termini of the Skagit River and their direct freshwater influence.

The gap in niche space likely reflects real constraints on habitat expression resulting from interactions between Skagit Delta geomorphology and hydrology. The tight correlation with distinct geography suggests this is so. Areas near the river have low to moderate salinity, depending on their connectivity to river distributaries and proximity to Skagit Bay, while high salinity areas are not possible except in areas like the Swinomish Channel, Telegraph Slough, and northeastern Padilla Bay that are distant from freshwater river input. It is unlikely that it reflects an unfortunate gap in sampling effort that simply missed areas with intermediate salinity, because sampling was extensive (3400 points) and broadly distributed throughout the delta.

The sentinel sites and their associated management alternatives are plotted in the niche space in Figure 3.3. For each sentinel site Alternative 2 is always more distant from current conditions than Alternative 1, sometimes by a small amount and sometimes by a large amount. This is consistent with the model predictions summarized in Fig. 3.4 and Supplementary Table 3.2.

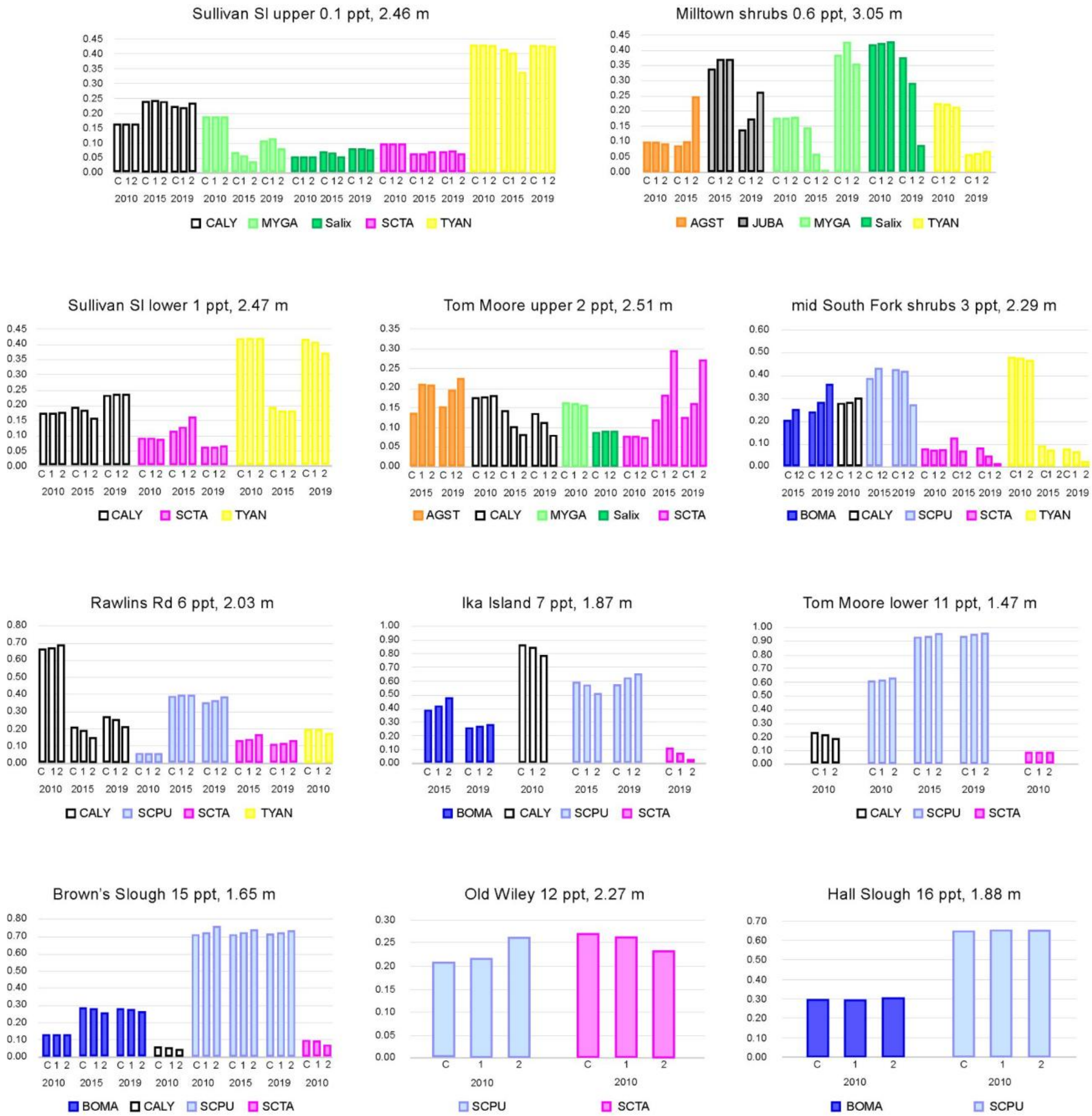

Fig. 3.4. Summary of predicted vegetation composition at eleven sentinel sites in response to current (C), alternative 1 (1), and alternative 2 (2) water management modeled for normal (2010) and low-flow (2015 and 2019) years.. Sentinel sites are arranged in order of low salinity and high elevation (top graphs) to high salinity and low elevation (bottom graphs). Only vegetation that occurred at > 7% frequency were plotted. AGST = *Agrostis stolonifera* (bentgrass); BOMA = *Bolboschoenus maritimus* (maritime bulrush); CALY = *Carex lyngbyei* (sedge); DISP =*Distichlis spicata* (saltgrass); JUBA = *Juncus balticus* (Baltic rush); MYGA = *Myrica gale* (sweetgale); *Salix* spp. (willow); SAPA = *Sarcocornia pacifica* (pickleweed); SCPU = *Schoenoplectus pungens* (three-square); SCTA = *S. tabernaemontani* (soft-stem bulrush); TYAN = *Typha angustifolia* (non-native cattail).

Normal flow, water year 2010

For all sentinel sites, there was generally little vegetation response to differences in water management under normal flow conditions (water year 2010). There were only three modest exceptions involving comparisons between current management and Alt 2. At the Browns Slough marsh site, there was an increase in SCPU (*Schoenoplectus pungens*) cover from 71% to 76%, with marginal change in other species. At the Old Wiley Slough site, there was an increase in SCPU cover from 21% to 26%, which is a more substantial proportional change, while SCTA (*S. tabernaemontani*) declined from 27% to 23%. At the Ika Island low marsh site, CALY (*Carex lyngbyei*) declined from 86% to 79% cover, a difference of 7%, but modest in proportion to original cover. Other species showed negligible change. Finally, when current management was compared to Alt 1, for normal flow conditions, there were negligible (0-1%) differences in predicted vegetation cover for all sites.

Low flow, Current Water Use vs. Alternative 1

For low-flow years (2015 and 2019) comparisons between current management and Alternative 1 were notable only for three sentinel sites. At the upper Tom Moore Slough site, creeping bentgrass (AGST) increased from 14% to 21% cover in 2015, and from 15% to 20% in 2019, and bulrush (SCTA) increased similarly from 12% to 18% and from 13% to 16% during both years. Meanwhile CALY declined from 14% to 10% and from 14% to 11% in those two years. The mid-South Fork shrubs site saw an increase in maritime bulrush (BOMA) from 20% to 25% and from 24% to 28% for both low-flow years, while SCTA decreased from 13% to 7% in 2015 and DISP increased from 2% to 7% in 2019. For the Milltown reference shrub site, there was a net decrease in shrub cover (willows [Salix] and sweetgale [MYGA]) ranging from 4% to 9%, depending on water year. Baltic rush (JUBA) showed a marginal 3-4% increase.

Low flow, Current Water Use vs. Alternative 2

These three sensitive sentinel sites showed even greater contrast between current management and Alternative 2, with some striking changes in vegetation cover. The Milltown Island reference shrub site was the site that showed greatest response with a strong decrease in shrubs, either sweetgale (MYGA; -15%) or willow (*Salix* spp.; -27%), depending on low-water year. Decreases in shrub cover resulted in increases in high elevation herbaceous marsh vegetation in the form of AGST (+16%) or JUBA (+12%), again depending on water year. For the mid-South Fork shrub site comparison between current management and Alt 2 was not possible for the 2015 water year, likely because there was not sufficient data for shrubs in this combination of elevation and high salinity. For the 2019 low-flow water year, saltgrass (DISP) increased from 2% to 28% cover, BOMA increased from 24% to 36%, and three-square (SCPU) decreased from 43% to 27%. At the upper Tom Moore Slough site, SCTA increased in cover from 12% to 30% for 2015 and from 13% to 27% for 2019, AGST increased from 14% to 21% in 2015 and 15% to 23% in 2019, while CALY declined from 14% to 8% in both years.

North Fork sentinel sites

The North Fork sentinel site, Ika Island, Sullivan Slough upper and lower, and Rawlins Road, were relatively resilient to management alternatives under low flow conditions, because salinity changes were relatively modest. Consequently, the biggest change at Rawlins Road was a decline of 6% in CALY (from 21% to 15% and 27% to 21%) during both water years with marginal increases in SCPU, SCTA, and BOMA. At Ika, BOMA increased by 8% while SCPU decreased by the same percentage in 2015; in 2019 SCPU increased by 8% while SCTA decreased by the same. However, the decrease in SCTA was proportionally severe as it was from 11% to 3%. The Sullivan Slough shrub sites, upper and lower, had a modest increase in AGST of 5% and modest declines in cattail (TYAN) of 5% (lower, 2019) and 8% (upper, 2015).

High salinity sentinel sites

The high salinity sentinel sites consisted of Halls Slough, Browns Slough, Old Wiley Slough, and lower Tom Moore Slough. Model predictions could not be made for the Halls Slough or Old Wiley Slough sites at low flows because the predicted salinities fell in the niche space data gap between the active and peripheral parts of the Skagit Delta (see Fig. 3.3). The combination of high elevation and high salinity is rare in the active Skagit Delta; much more common is low elevation and high salinity—areas closer to the bay are lower and saltier. For this reason, Browns Slough, which is lower, could be modeled and may provide some insight into Halls Slough and Old Wiley Slough, because the predicted salinity differences between Browns and Halls Sloughs are < 0.7 ppt, and between Browns and Old Wiley Sloughs they are < 0.4 ppt. Browns Slough is at lower elevation (1.65 m) than the other two sites (1.88 m and 2.27 m), so Browns Slough should be particularly vulnerable to the combined stresses of high salinity and longer inundation. Nevertheless, modeling shows that Browns Slough is very refractory to the modeled management changes. There is virtually no vegetation change there under the various management alternatives. Lower Tom Moore Slough is also an area of low elevation (1.47 m) and high predicted salinities (similar to Old Wiley Slough predictions). Here also there is essentially no vegetation change; SCPU remains dominant at >93% cover for all low flow management scenarios.

The refractory nature of the model predictions for the North Fork and high salinity sentinel sites paralleled empirical observations of refractory salinity variation in these areas in response to observed variation in river flow. Temporal variability in observed soil porewater salinity was very low in tidal freshwater areas such as marsh near the Deepwater Slough restoration site, Milltown Island, and the Dunlap Bay/Sullivan Slough portion of the North Fork marsh. It was also relatively low in the high salinity bay front areas near the Brown and Hall Sloughs outlets. In contrast, the seaward portion of Tom Moore Slough and other nearby distributaries was one of the most variable regions.

Discussion

Vegetation type exhibited no to little response to differences in water management (Alternative 1 and Alternative 2) under normal flow conditions (water year 2010). During low flow conditions (water years 2015 & 2019), there were measurable changes at three sentinel sites to changes from Current Water Use to Alternative 1. The three sites were dominated by estuarine shrub vegetation types (upper Tom Moore, Milltown reference shrub, and mid-South Fork shrub) and the vegetation changes ranged from 3% to 9%. The vegetation change of greatest concern would be the 9% net decline in shrub cover for the Milltown reference shrub site. Shrubs provide important habitat for intertidal beaver whose dams create low-tide pools that support high densities of juvenile Chinook salmon (Hood 2012). Shrubs compose estuarine forested transition habitats that have been documented to provide increased foraging opportunities (Greene et al. 2020, Davis et al. 2018). The eight other sentinel sites had little to no response to Alternative 1.

Low-flow vegetation response was much greater for comparisons between current management and Alternative 2. The most significant change was a decline in willow or sweetgale cover of 27% and 15%, respectively at the Milltown reference shrub site, depending on water year. This large decline would likely significantly impact beaver and the habitat that they provide juvenile Chinook salmon. At the mid-South Fork shrub site and the upper Tom Moore Slough site there were large swings in herbaceous cover, e.g., DISP +26%, SCTA +18%, SCPU -16%, BOMA +12%, among others, which clearly indicate high sensitivity and vulnerability. The North Fork sentinel sites had no to moderate responses to Alternative 2, while the high salinity sentinel sites had or are likely to have virtually no response.

While current modeling provides clear predictions of vegetation change, it is not clear at what rate the ecosystem could respond to management changes. Tidal marsh vegetation change can occur at a wide range of rates, depending on rates of environmental change (sea level rise, warming, nutrient pollution, disease, grazing intensity, changes in river hydrographs) and species-specific responses to environmental changes. Change can be very rapid, within one growing season, as in the case of sudden vegetation dieback (SVD), which may be caused by eutrophication, fungal disease, drought, or a combination of many stressors (Elmer et al. 2013). Change can also occur at a decadal pace in response to climatic changes, as has happened in Tasmanian marshes in response to changes in rainfall, wind, and temperature over 30 years (Prahalad et al. 2012). Similarly, invasion by non-native species, such as narrow-leaf cattail (Typha angustifolia) in the Skagit marshes, can cause gradual displacement of native species over many decades (Hood, unpublished data). Given the complexity of the phenomenon, predicting rates of vegetation community change in response to environmental change or management change is very challenging.

Another difficulty with interpreting model results is that it is sometimes unclear how the ecological or geomorphological roles of different plant species vary. While it is known that cattail (TYAN) provides nesting habitat for marsh wrens and redwing blackbirds; that sedge (CALY) provides forage for ducks, geese, beaver, and deer; that the rhizomes of bulrushes (SCPU, BOMA) are grubbed by geese and swans; and that tidal shrubs provide forage and building material for beaver, who in turn provide important low-tide pool habitat for juvenile salmon; it is unclear how these species (especially the herbaceous species) may differ in their ability to trap and accrete sediment, baffle storm waves, or produce invertebrate prey for fish (including threatened juvenile salmon). Thus, it is unclear whether it matters if at the mid-South Fork sentinel site SCPU declines from 43% to 27% occurrence, while BOMA increases from 24% to 36% and DISP increases from 2% to 28%. On the other hand, the importance of tidal shrub vegetation in providing rearing habitat for juvenile salmon is better documented (Greene et al. 2020, Davis et al. 2018, 2024; Hood 2012). Tidal shrub vegetation provides sizable supplies of insect prey to juvenile salmon, while beaver dams in tidal shrub habitat quadruple the amount of low-tide habitat available to fish such that juvenile salmon densities (by volume) are three times higher in low-tide beaver pools than in channel shallows. These low-tide beaver pools are full of organic detritus that likely supports high invertebrate prey production. Low-tide beaver pools also allow small fish to avoid being flushed into large distributary channels; greater residence time in small blind tidal channels likely leads to better growth and survival of juvenile salmon during estuarine residence. Finally, tidal shrub vegetation shades small tidal channels, providing some thermal refuge for heat-sensitive fish such as juvenile salmon. Thus, the significant impacts to tidal shrub vegetation resulting from Alternative 2 have a high likelihood of impacting salmon productivity in the Skagit Delta.

Model uncertainty

Aspects of climate change, such as sea level rise, temperature increases and precipitation rate changes, were not directly evaluated within this evaluation. Sea level rise is another ongoing climate change stress that was not considered in this modeling effort, but which could add additional stress to the system, if sediment supply is locally limited and compounds the stress of increased salinity caused by increased water withdrawals from the river. Sediment supply is likely abundant in the North and South Fork sub-deltas, but it is likely insufficient in the bay fringe between the sub-deltas where historical distributaries across Fir Island were blocked in the 1950s and where marsh erosion has since been significant (Hood et al. 2016). Increases in air and water temperature and changes in precipitation rates driven by climate change are known to affect the presence and density of submerged and non-submerged tidal vegetation (see review Short et al. 2016). The Pacific Northwest is projected to have increases in annual air temperature of 3.0 C by 2080 and significant changes in seasonal precipitation (Mote et al. 2010).

Another extrinsic uncertainty in evaluating the potential impacts of different water use scenarios is the unknown outcome of the Seattle City Light relicensing process. Ongoing negotiations between Seattle City Light, the State of Washington, federal agencies, and Tribes could result in significant changes to water management. Because these outcomes remain unresolved, we were unable to incorporate scenarios reflecting potential changes arising from the relicensing process.

Some uncertainties within the analytical technique mainly related to leveraging previous data collection efforts. This study only had access to Skagit estuary observations that are constrained by available reference sites. With >70% of the Skagit estuary lost to diking and drainage over the last century (Collins 2000), there are limitations to describing the full suite of sites that vegetation will occupy. Our results are consistent, however, with published literature. Overall, there were 1,538 observations taken to describe vegetation occupancy across salinity and elevation gradients. These observations were unequally distributed across each vegetation class with some having < 80 observations which can impact correct evaluation of constraints on occupancy (Mackenzie et al. 2017). Model validation techniques suggest predictions are valid; however, with some sensitivity in results if additional data are included.

Implications for water and salmon management

While uncertainties exist about the rate of vegetation change and of some of the ecological consequences of changes in species composition, it is also clear that significant areas of tidal shrub vegetation and their ecological function for beaver and juvenile salmon appear sensitive to both water use (modeled) and sea level rise (not modeled). Scrub-shrub communities not only contribute to habitat formation through their influence on the distribution of dam-building beavers, they also can contribute to reduced water temperatures and added insect prey communities. These latter elements are important and (in this report) unmodeled components of habitat that can affect the distribution and growth of juvenile salmon and other fishes. Given ongoing stresses from non-native species invasions, especially cattail (TYAN) (Clifton et al. 2018), there are multiple dimensions to the continued resilience of scrub-shrub communities.

For other vegetation communities, it remains unclear what the effects of a reduction in occurrence of one species or another means for juvenile Chinook salmon and other species, other than concluding that the occurrence of some vegetative cover provides more ecological function than unvegetated sand and mudflat (Davis et al. 2018). In sum, this work supports the idea that vegetation response is resilient to current water management, but that additional water uses will likely increase cumulative stress on scrub-shrub communities during low-flow years.

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Table S1. Summary of vegetation model predictions (frequency of species occurrence as dominant) for sentinel sites by water scenarios. Frequencies of occurrence that are > 0.05 are in bold. AGST = Agrostis stolonifera; BOMA = Bolboschoenus maritimus; CALY = Carex lyngbyei; DISP = Distichlis spicata; JUBA = Juncus balticus; MYGA = Myrica gale; SCPU = Schoenoplectus pungens; SCTA = S. tabernaemontani; TYAN = Typha angustifolia.

Site	Scenario	Salin	Elev	AGST	BOMA	CALY	DISP	JUBA	MYGA	Salix	SAPA	SCPU	SCTA	TYAN	sum	Predicted dominants	Largest Δ between scenarios	Interpretation
Hall Sl marsh	2010Cur	16.0	1.88	0.00	**0.30**	0.01	0.00	0.00	0.00	0.00	0.00	**0.65**	0.03	0.00	0.99	1st SCPU 2nd BOMA	BOMA 0.01	Little to no veg change
	2010A1	16.1	1.88	0.00	**0.30**	0.01	0.00	0.00	0.00	0.00	0.00	**0.65**	0.03	0.00	0.99			
	2010A2	16.2	1.88	0.00	**0.31**	0.00	0.00	0.00	0.00	0.00	0.00	**0.65**	0.02	0.00	0.99			
Hall Sl marsh	**2015B1**	19.2	1.88	0.00	**0.50**	0.00	0.00	0.00	0.00	0.00	0.01	**0.48**	0.00	0.00	0.99	Prediction limited by salin. > 19.2 BOMA, SCPU likely co-dominant	NA	Given small salinity change, veg change is likely negligible
	2015Cur	19.4	1.88					0.00			0.01				0.01			
	2015A1	19.6	1.88					0.00			0.02				0.02			
	2015A2	19.9	1.88					0.00							0.00			
Hall Sl marsh	2019Cur	19.6	1.88					0.00							0.00	Prediction limited by salin. > 19.2 BOMA, SCPU likely co-dominant	NA	Given small salinity change, veg change is likely negligible
	2019A1	19.7	1.88					0.00							0.00			
	2019A2	20.2	1.88					0.00							0.00			
Browns Sl marsh	2010Cur	14.7	1.65	0.00	**0.13**	**0.06**	0.00	0.00	0.00	0.00	0.00	**0.71**	**0.10**	0.00	1.00	1st SCPU 2nd BOMA 3rd SCTA	SCPU 0.05 SCTA -0.03 CALY 0.02	Biggest Δ is Cur vs Alt2. SCPU up; CALY & SCTA down.
	2010A1	14.8	1.65	0.00	**0.13**	**0.05**	0.00	0.00	0.00	0.00	0.00	**0.72**	**0.09**	0.00	1.00			
	2010A2	15.1	1.65	0.00	**0.13**	**0.04**	0.00	0.00	0.00	0.00	0.00	**0.76**	**0.07**	0.00	1.00			
Browns Sl marsh	2015Cur	18.7	1.65	0.00	**0.28**	0.00	0.00	0.00	0.00	0.00	0.00	**0.71**	0.00	0.00	1.00	1st SCPU 2nd BOMA	SCPU 0.03 BOMA -0.03	Little to no veg change
	2015A1	19.1	1.65	0.00	**0.28**	0.00	0.00	0.00	0.00	0.00	0.00	**0.72**	0.00	0.00	1.00			
	2015A2	19.7	1.65	0.00	**0.25**	0.00	0.00	0.00	0.00	0.00	0.01	**0.74**	0.00	0.00	1.00			
Browns Sl marsh	2019Cur	19.0	1.65	0.00	**0.28**	0.00	0.00	0.00	0.00	0.00	0.00	**0.72**	0.00	0.00	1.00	1st SCPU 2nd BOMA	SCPU 0.02 BOMA -0.02	Little to no veg change
	2019A1	19.2	1.65	0.00	**0.27**	0.00	0.00	0.00	0.00	0.00	0.00	**0.72**	0.00	0.00	1.00			
	2019A2	19.6	1.65	0.00	**0.26**	0.00	0.00	0.00	0.00	0.00	0.01	**0.74**	0.00	0.00	1.00			
Old Wiley marsh	2010Cur	12.3	2.27	**0.09**	**0.14**	**0.08**	0.00	0.04	0.00	0.00	0.00	**0.21**	**0.27**	**0.12**	0.95	1st SCTA 2nd SCPU 3rd BOMA	SCPU 0.05 SCTA -0.04 BOMA 0.02	Biggest Δ is current v Alt2. SCPU up; SCTA down.
	2010A1	12.4	2.27	**0.09**	**0.14**	**0.08**	0.00	0.04	0.00	0.00	0.00	**0.22**	**0.26**	**0.12**	0.96			
	2010A2	12.8	2.27	**0.07**	**0.16**	**0.07**	0.00	0.04	0.00	0.00	0.00	**0.26**	**0.23**	**0.11**	0.96			

Site	Scenario	Salin	Elev	AGST	BOMA	CALY	DISP	JUBA	MYGA	Salix	SAPA	SCPU	SCTA	TYAN	sum	Predicted dominants	Largest Δ between scenarios	Interpretation
Old Wiley marsh	2015Cur	18.9	2.27					0.03							0.03	hole in the niche space	NA	NA
	2015A1	19.2	2.27					0.02							0.02			
	2015A2	20.1	2.27					0.01							0.01			
Old Wiley marsh	2019Cur	18.6	2.27					0.03							0.03	hole in the niche space	NA	NA
	2019A1	18.9	2.27					0.03							0.03			
	2019A2	20.0	2.27					0.01							0.01			
Rawlins Rd low marsh	2010Cur	5.7	2.03	0.00	0.00	**0.67**	0.00	0.00	0.00	0.01	0.00	0.00	**0.06**	**0.20**	0.94	1st CALY 2nd TYAN	TYAN -0.02 CALY 0.02	Little to no veg change
	2010A1	5.8	2.03	0.00	0.00	**0.67**	0.00	0.00	0.00	0.01	0.00	0.00	**0.06**	**0.19**	0.94			
	2010A2	6.1	2.03	0.00	0.00	**0.69**	0.00	0.00	0.00	0.01	0.00	0.00	**0.06**	**0.18**	0.94			
Rawlins Rd low marsh	2015Cur	10.6	2.03	0.04	**0.14**	**0.21**	0.00	0.00	0.00	0.00	0.00	0.13	**0.39**	0.04	0.95	1st SCTA 2nd CALY 3rd BOMA	CALY -0.06 SCPU 0.04 BOMA 0.03	Biggest Δ is current vs Alt2. SCPU up; CALY down
	2015A1	10.8	2.03	0.04	**0.15**	**0.19**	0.00	0.00	0.00	0.00	0.00	0.14	**0.39**	0.04	0.95			
	2015A2	11.3	2.03	0.04	**0.17**	**0.15**	0.00	0.00	0.00	0.00	0.00	0.17	**0.40**	0.03	0.96			
Rawlins Rd low marsh	2019Cur	10.2	2.03	0.04	**0.12**	**0.27**	0.00	0.00	0.00	0.00	0.00	**0.11**	**0.35**	0.04	0.95	1st SCTA 2nd CALY 3rd BOMA	CALY -0.06 SCTA 0.04 BOMA 0.02	Biggest Δ is current vs Alt2. SCTA up; CALY down.
	2019A1	10.3	2.03	0.04	**0.13**	**0.26**	0.00	0.00	0.00	0.00	0.00	**0.12**	**0.36**	0.04	0.95			
	2019A2	10.6	2.03	0.04	**0.14**	**0.21**	0.00	0.00	0.00	0.00	0.00	**0.13**	**0.39**	0.04	0.95			
Tom Moore upper	2010Cur	1.9	2.51	0.01	0.00	**0.18**	0.00	0.00	**0.17**	**0.09**	0.00	0.00	**0.08**	**0.40**	0.93	1st TYAN 2nd CALY 3rd MYGA	MYGA -0.01	Little to no veg change
	2010A1	2.0	2.51	0.01	0.00	**0.18**	0.00	0.00	**0.16**	**0.09**	0.00	0.00	**0.08**	**0.40**	0.93			
	2010A2	2.2	2.51	0.01	0.00	**0.18**	0.00	0.00	**0.16**	**0.09**	0.00	0.00	**0.08**	**0.40**	0.92			
Tom Moore upper	2015Cur	7.0	2.51	**0.14**	0.00	**0.14**	0.00	0.02	0.03	0.03	0.00	0.01	**0.12**	**0.20**	0.68	1st TYAN 2nd SCTA 3rd CALY	SCTA 0.18 AGST 0.07 CALY -0.06	Biggest Δ is current vs Alt2. SCTA, AGST up; CALY down
	2015A1	8.4	2.51	**0.21**	0.01	**0.10**	0.00	0.03	0.02	0.01	0.00	0.02	**0.18**	**0.21**	0.79			
	2015A25	10.9	2.51	**0.21**	0.03	**0.08**	0.00	0.05	0.00	0.00	0.00	0.05	**0.30**	**0.22**	0.94			
Tom Moore upper	2019Cur	7.2	2.51	**0.15**	0.00	**0.14**	0.00	0.02	0.03	0.03	0.00	0.01	**0.13**	**0.19**	0.69	1st TYAN 2nd SCTA 3rd AGST	SCTA 0.14 AGST 0.08 CALY -0.06	Biggest Δ is current vs Alt2. SCTA, AGST up; CALY down
	2019A1	8.1	2.51	**0.20**	0.00	**0.11**	0.00	0.02	0.02	0.02	0.00	0.02	**0.16**	**0.20**	0.75			
	2019A2	10.1	2.51	**0.23**	0.02	**0.08**	0.00	0.04	0.01	0.00	0.00	0.04	**0.27**	**0.23**	0.92			
Tom Moore lower	2010Cur	11.0	1.47	0.00	**0.06**	**0.23**	0.00	0.00	0.00	0.00	0.00	**0.61**	**0.09**	0.00	0.99	1st SCPU 2nd CALY 3rd SCTA	CALY -0.04	Little to no vegetation change
	2010A1	11.2	1.47	0.00	**0.06**	**0.22**	0.00	0.00	0.00	0.00	0.00	**0.62**	**0.09**	0.00	0.99			
	2010A2	11.7	1.47	0.00	**0.08**	**0.19**	0.00	0.00	0.00	0.00	0.00	**0.63**	**0.09**	0.00	1.00			

Site	Scenario	Salin	Elev	AGST	BOMA	CALY	DISP	JUBA	MYGA	Salix	SAPA	SCPU	SCTA	TYAN	sum	Predicted dominants	Largest Δ between scenarios	Interpretation
Tom Moore lower	2015Cur	18.2	1.47	0.00	**0.07**	0.00	0.00	0.00	0.00	0.00	0.00	**0.93**	0.00	0.00	1.00	1st SCPU 2nd BOMA	SCPU 0.03 BOMA -0.03	Little to no vegetation change
	2015A1	18.5	1.47	0.00	**0.06**	0.00	0.00	0.00	0.00	0.00	0.00	**0.94**	0.00	0.00	1.00			
	2015A2	19.5	1.47	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	**0.96**	0.00	0.00	1.00			
Tom Moore lower	2019Cur	18.5	1.47	0.00	**0.06**	0.00	0.00	0.00	0.00	0.00	0.00	**0.94**	0.00	0.00	1.00	1st SCPU	BOMA -0.02 SCPU 0.02	Little to no vegetation change
	2019A1	19.0	1.47	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	**0.95**	0.00	0.00	1.00			
	2019A2	19.9	1.47	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.01	**0.96**	0.00	0.00	1.01			
mid South Fork shrub	2010Cur	2.9	2.29	0.01	0.00	**0.28**	0.00	0.00	**0.07**	**0.06**	0.00	0.00	**0.08**	**0.48**	0.97	1st TYAN 2nd CALY 3rd SCTA	CALY 0.02 TYAN -0.01	Little to no vegetation change
	2010A1	3.0	2.29	0.01	0.00	**0.28**	0.00	0.00	**0.07**	**0.06**	0.00	0.00	**0.08**	**0.48**	0.97			
	2010A2	3.5	2.29	0.01	0.00	**0.30**	0.00	0.00	0.05	**0.06**	0.00	0.00	**0.08**	**0.47**	0.96			
mid South Fork shrub	2015Cur	14.1	2.29	0.04	**0.20**	**0.06**	0.01	**0.06**	0.00	0.00	0.00	**0.39**	**0.13**	**0.09**	0.97	1st SCPU 2nd BOMA 3rd SCTA	SCPU 0.04 SCTA -0.06 BOMA 0.05	Alt2 could not be evaluated. SCPU & BOMA up; SCTA down.
	2015A1	15.1	2.29	0.03	**0.25**	0.04	0.03	**0.06**	0.00	0.00	0.00	**0.43**	**0.07**	**0.07**	0.99			
	2015A2	17.9	2.29					0.04			0.00				0.04			
mid South Fork shrub	2019Cur	14.8	2.29	0.03	**0.24**	0.04	0.02	**0.06**	0.00	0.00	0.00	**0.43**	**0.08**	**0.08**	0.98	1st SCPU 2nd BOMA 3rd DISP	DISP 0.26 SCPU -0.16 BOMA 0.12	Biggest Δ is current vs Alt2. DISP, BOMA way up; SCPU, SCTA, TYAN down.
	2019A1	15.7	2.29	0.02	**0.28**	0.03	**0.07**	**0.06**	0.00	0.00	0.00	**0.42**	0.05	**0.07**	1.00			
	2019A2	17.6	2.29	0.01	**0.36**	0.01	**0.28**	0.05	0.00	0.00	0.00	**0.27**	0.01	0.02	1.03			
Milltown ref shrub	2010Cur	0.6	3.05	**0.10**	0.00	0.00	0.00	0.01	**0.17**	**0.42**	0.00	0.00	0.00	**0.22**	0.93	1st Salix 2nd TYAN 3rd MYGA	Salix 0.01 MYGA 0.01 TYAN -0.01	Little to no vegetation change
	2010A1	0.7	3.05	**0.10**	0.00	0.00	0.00	0.01	**0.18**	**0.42**	0.00	0.00	0.00	**0.22**	0.93			
	2010A2	0.8	3.05	**0.10**	0.00	0.00	0.00	0.01	**0.18**	**0.43**	0.00	0.00	0.00	**0.21**	0.93			
Milltown ref shrub	2015Cur	7.7	3.05	**0.09**	0.00	0.00	**0.09**	**0.34**	**0.15**	0.01	0.00	0.00	0.02	0.04	0.74	1st JUBA 2nd AGST 3rd DISP	AGST 0.16 MYGA -0.15 DISP 0.06	Biggest Δ is current vs Alt2. AGST, DISP up; MYGA gone
	2015A1	8.7	3.05	**0.10**	0.00	0.01	**0.13**	**0.37**	**0.06**	0.00	0.00	0.00	0.02	0.02	0.72			
	2015A2	11.1	3.05	**0.25**	0.00	0.01	**0.15**	**0.37**	0.00	0.00	0.00	0.00	0.02	0.01	0.81			
Milltown ref shrub	2019Cur	4.7	3.05	0.02	0.00	0.00	0.00	**0.14**	**0.38**	**0.38**	0.00	0.00	0.01	**0.06**	0.99	1st MYGA 2nd Salix 3rd JUBA	Salix -0.27 JUBA 0.12 MYGA -0.03	Biggest Δ is currrent vs Alt2. JUBA up Salix way down
	2019A1	5.2	3.05	0.02	0.00	0.00	0.00	**0.18**	**0.43**	**0.29**	0.00	0.00	0.01	**0.06**	0.99			
	2019A2	6.4	3.05	0.06	0.00	0.00	0.03	**0.26**	**0.35**	**0.09**	0.00	0.00	0.01	**0.07**	0.88			
	2010Cur	0.9	2.47	0.01	0.00	**0.18**	0.00	0.00	**0.18**	**0.07**	0.00	0.00	**0.09**	**0.42**	0.95		MYGA 0.01	

Site	Scenario	Salin	Elev	AGST	BOMA	CALY	DISP	JUBA	MYGA	Salix	SAPA	SCPU	SCTA	TYAN	sum	Predicted dominants	Largest Δ between scenarios	Interpretation
Sullivan Sl shrub lower	2010A1	1.0	2.47	0.01	0.00	**0.18**	0.00	0.00	**0.18**	**0.07**	0.00	0.00	**0.09**	**0.42**	0.95	1st TYAN 2nd CALY 3rd MYGA		Little to no veg change
	2010A2	1.1	2.47	0.01	0.00	**0.18**	0.00	0.00	**0.17**	**0.07**	0.00	0.00	**0.09**	**0.42**	0.95			
Sullivan Sl shrub lower	2015Cur	7.0	2.47	**0.14**	0.00	**0.19**	0.00	0.01	0.02	0.03	0.00	0.01	**0.12**	**0.19**	0.71	1st TYAN 2nd CALY 3rd AGST	AGST 0.05 SCTA 0.04 CALY -0.03	Modest veg change
	2015A1	7.3	2.47	**0.16**	0.00	**0.18**	0.00	0.02	0.02	0.02	0.00	0.01	**0.13**	**0.18**	0.72			
	2015A2	8.0	2.47	**0.19**	0.00	**0.16**	0.00	0.02	0.02	0.01	0.00	0.02	**0.16**	**0.18**	0.77			
Sullivan Sl shrub lower	2019Cur	4.4	2.47	0.02	0.00	**0.24**	0.00	0.00	**0.07**	**0.08**	0.00	0.00	**0.06**	**0.42**	0.89	1st TYAN 2nd CALY 3rd Salix	TYAN -0.05 Salix -0.02 MYGA -0.02	Modest veg change
	2019A1	4.7	2.47	0.02	0.00	**0.24**	0.00	0.00	**0.07**	**0.07**	0.00	0.00	**0.06**	**0.41**	0.87			
	2019A2	5.2	2.47	0.04	0.00	**0.24**	0.00	0.01	0.05	**0.06**	0.00	0.00	**0.07**	**0.37**	0.83			
Sullivan Sl shrub upper	2010Cur1	0.1	2.46	0.01	0.00	**0.17**	0.00	0.00	**0.19**	**0.06**	0.00	0.00	**0.10**	**0.43**	0.95	1st TYAN 2nd MYGA 3rd CALY	no Δ	Little to no veg change
	2010A1	0.1	2.46	0.01	0.00	**0.17**	0.00	0.00	**0.19**	**0.06**	0.00	0.00	**0.10**	**0.43**	0.95			
	2010A2	0.2	2.46	0.01	0.00	**0.17**	0.00	0.00	**0.19**	**0.06**	0.00	0.00	**0.10**	**0.43**	0.95			
Sullivan Sl shrub upper	015Cur	4.5	2.46	0.02	0.00	**0.24**	0.00	0.00	**0.07**	**0.07**	0.00	0.00	**0.06**	**0.42**	0.89	1st TYAN 2nd CALY 3rd SCTA	TYAN -0.08 MYGA -0.03 Salix -0.02	Biggest Δ is current vs Alt2. TYAN & shrubs down.
	2015A1	4.8	2.46	0.02	0.00	**0.24**	0.00	0.01	**0.06**	**0.07**	0.00	0.00	**0.07**	**0.40**	0.87			
	2015A2	5.6	2.46	0.05	0.00	**0.24**	0.00	0.01	0.04	0.05	0.00	0.00	**0.07**	**0.34**	0.80			
Sullivan shrub upper	019Cur	3.2	2.46	0.01	0.00	**0.22**	0.00	0.00	**0.11**	**0.08**	0.00	0.00	**0.07**	**0.43**	0.93	1st TYAN 2nd CALY 3rd MYGA	MYGA -0.03	Little to no veg change
	2019A1	3.1	2.46	0.01	0.00	**0.22**	0.00	0.00	**0.11**	**0.08**	0.00	0.00	**0.07**	**0.43**	0.93			
	2019A2	4.0	2.46	0.01	0.00	**0.24**	0.00	0.00	**0.08**	**0.08**	0.00	0.00	**0.07**	**0.43**	0.91			
Ika Island low marsh	010Cur	6.7	1.87	0.00	0.00	**0.86**	0.00	0.00	0.00	0.00	0.00	0.03	0.04	0.03	0.97	1st CALY	CALY -0.07 SCTA 0.04 SCPU 0.02	Biggest Δ is current vs Alt2. CALY down; others modest.
	2010A1	7.0	1.87	0.00	0.00	**0.85**	0.00	0.00	0.00	0.00	0.00	0.04	0.04	0.03	0.96			
	2010A2	7.9	1.87	0.00	0.01	**0.79**	0.00	0.00	0.00	0.00	0.00	0.05	0.08	0.02	0.96			
Ika Island low marsh	015Cur	17.6	1.87	0.00	**0.40**	0.00	0.00	0.00	0.00	0.00	0.00	**0.59**	0.00	0.00	1.00	1st SCPU 2nd BOMA	BOMA 0.08 SCPU -0.08	Biggest Δ is current vs Alt2; SCPU down, BOMA up.
	2015A1	17.9	1.87	0.00	**0.42**	0.00	0.00	0.00	0.00	0.00	0.00	**0.57**	0.00	0.00	1.00			
	2015A2	18.8	1.87	0.00	**0.48**	0.00	0.00	0.00	0.00	0.00	0.00	**0.51**	0.00	0.00	1.00			
Ika Isl. low marsh	019Cur	14.9	1.87	0.00	**0.27**	0.02	0.00	0.00	0.00	0.00	0.00	**0.58**	**0.11**	0.00	0.98	1st SCPU 2nd BOMA 3rd SCTA	SCPU 0.08 SCTA -0.08 BOMA 0.02	Biggest Δ is current vs Alt2; SCPU, BOMA up; SCTA down
	2019A1	15.4	1.87	0.00	**0.27**	0.01	0.00	0.00	0.00	0.00	0.00	**0.62**	**0.07**	0.00	0.98			
	2019A2	15.9	1.87	0.00	**0.29**	0.01	0.00	0.00	0.00	0.00	0.00	**0.66**	0.03	0.00	0.99			

4. Effects of water withdrawals for fishes in the Skagit River delta

Stuart Munsch, Correigh Greene, and Michael LeMoine

Introduction

Estuaries provide important habitats for fishes, but they are often modified by human activities (Beck et al. 2001, Lotze et al. 2006, Sheaves et al. 2015). Many fishes inhabit estuaries where they benefit from features such as abundant prey, predator refuge, and diverse habitat options (Beck et al. 2001, Sheaves et al. 2015, Nagelkerken et al. 2015). However, the utility of estuarine habitats depends on physical conditions that connect habitats and promote habitat use, which are often impaired in human-dominated landscapes (Simenstad and Cordell 2000).

Novel flow regimes are a product of human activities that can reduce habitat accessibility by altering the physical environment. Water regulation and extraction for various purposes (e.g., agricultural, municipal) commonly change the magnitude and timing of freshwater input, which can reorganize estuarine ecosystems downstream (Alber 2002, Fan and Huang 2008, Cloern and Jassby 2012, Greene et al. 2015). Freshwater input drives the arrangement of basic water quality attributes such as salinity and velocity, which can alter the estuarine fish assemblage due to species' habitat preferences and tolerances (Ferguson et al. 2013, Colombano et al. 2022). Decisions to modify flow regimes must often balance the needs of multiple constituents and mandates including those concerned with fishes; scientific research that elucidates potential ecological changes can help inform these decisions (Alber 2002).

One way to understand effects of modified flow regimes on fishes is to link outputs of fish-environment models and hydrodynamic models (Ganju et al. 2016) (Fig. 1.1). That is, statistical models based on fish monitoring data collected in estuaries can quantify relationships between fish presence or abundance and local physical environments (see Appendix 4.1). Simultaneously, hydrodynamic models can predict how changes to freshwater input will rearrange the physical environment across estuaries (e.g., salinity, water surface elevation) (see Chapter 2). Then, the fish-environment models can use water quality attributes predicted by hydrodynamic models under different water use scenarios to make predictions about fish responses to modified flow regimes.

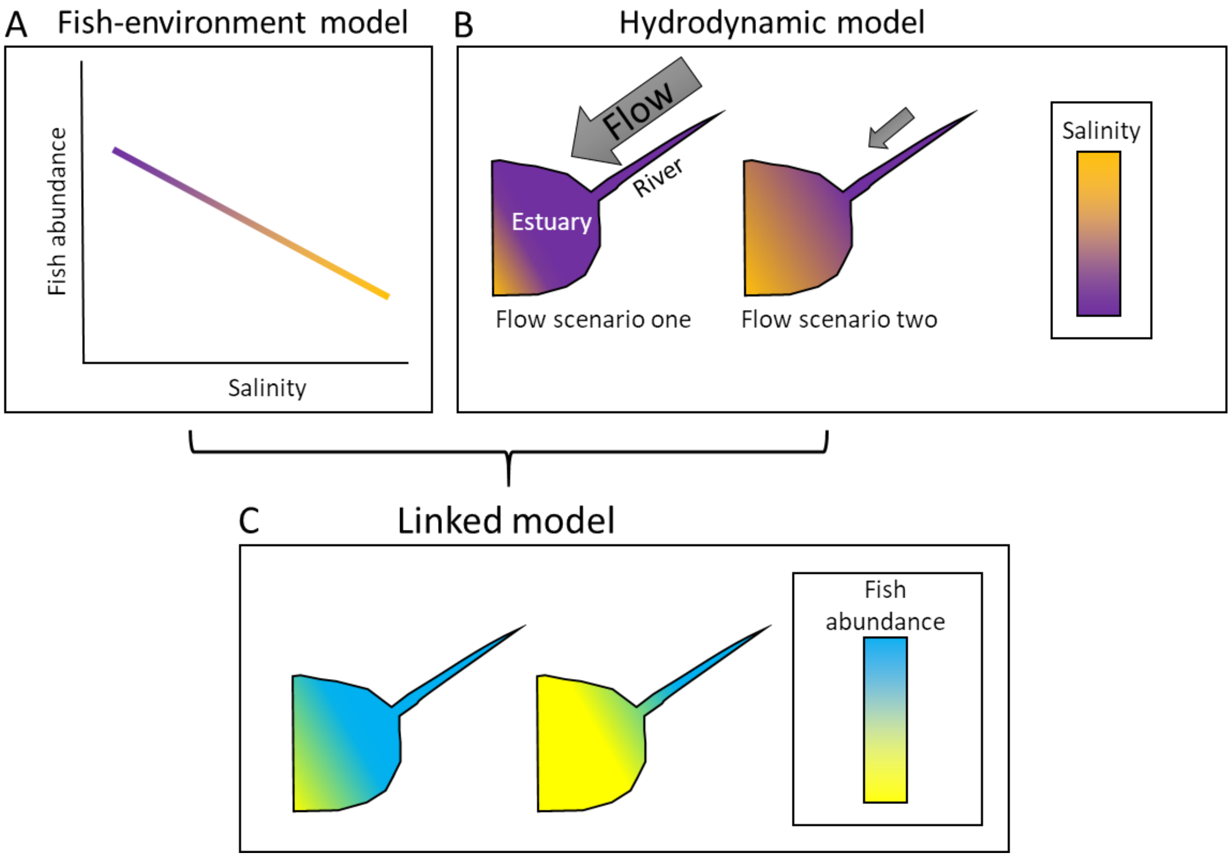

Figure 4.1. Conceptual diagram that shows how fish-environment models can be linked with hydrodynamic models to infer fish responses to modified flow regimes. The fish-environment model (A) predicts that a hypothetical fish species is more abundant in fresher waters. The hydrodynamic model (B) predicts that decreased flows will shift waters of greater salinity toward the river mouth. Then, salinity values predicted by the hydrodynamic model under different flow scenarios can be used as predictors in the fish-environment model to generate a linked model (C). The linked model predicts that scenario two's reduced flows will reduce abundance of the fish, especially in areas that are farther toward sea.

The Skagit River estuary is inhabited by a diversity of fish that include Chinook salmon (*Oncorhynchus tshawytscha*), an iconic species that is listed as threatened under the U.S. Endangered Species Act (ESA), central to indigenous tribal cultures, and supports a vibrant sport fishing community. In addition, the Skagit River estuary supports a number of other marine, freshwater and anadromous fish species, some commercially important (e.g. chum salmon, coho salmon and others (staghorn sculpin and starry flounder, Ruckelshaus and McClure 2007) that provide for ecosystem resilience. In addition, the river also provides a critical source of water for cities, residences, and specifically, irrigation for economically important agriculture industries whose water use could change over time.

Our goal was to provide a decision support tool for managers and constituents in the Skagit estuary who seek to understand potential effects of modified flow regimes on fishes. In this chapter, we link fish-environment statistical relationships (Appendix 4.1) and hydrodynamic models describing environmental conditions (Chapter 2) to infer how potential changes to flow from water use may alter Skagit River discharge and its estuarine fish assemblage, particularly juvenile rearing of ESA listed Chinook Salmon.

Methods

Study system

The Skagit River estuary is formed by north and south forks that drain into Skagit Bay (Fig. 4.2). The Skagit River expresses a transitional hydrologic regime, with peak flows in the lower elevations occurring in fall and early winter and higher elevations dominated by peak flows in spring and summer (Beechie 1992, Beechie et al. 2006). The natural flow regime is modified by regulation from dams and water extraction to support various human uses. The Skagit River delta is in an agricultural landscape that includes some natural and restored wetlands and channels (Simenstad et al. 2011, Chamberlin et al. In Review). The nearshore areas of the surrounding landscape include beaches and mudflats, which are modified by residential development (Simenstad et al. 2011).

The Skagit River estuary is inhabited by many fishes typical of shallow areas in Puget Sound (e.g., Pentilla 2007, Toft et al. 2007, Munsch et al. 2016). These species include a breadth of life histories and morphologies (e.g., anadromous salmonids, flatfish, forage fish) whose abundances are influenced by water conditions (e.g., salinity, water depth) (Appendix 4.1). Many species in Puget Sound use shallow areas in and around estuaries as juveniles to forage and avoid predators (Simenstad et al. 1982, Healey 1982, Simenstad and Cordell 2000, Munsch et al. 2016).

Summary of previous work

This study uses preexisting fish-environment statistical relationships (see Appendix 4.1) and hydrodynamic models (See Chapter 2). To synopsize key details:
Fish-environment models were GAMMs that quantified relationships between fish abundance and the environment (GAMMs; Wood 2004, Zuur et al. 2009). GAMMs were informed by ~8,000 beach seine samples from Feb.-Aug. during 2015-2023 in delta and nearshore waters of the Skagit River system (Fig. 4.2). Beach seine samples were opportunistically selected based on sites that were within restored estuarine marsh or marsh that had no record of being modified. In 2007, North Fork Skagit River avulsed resulting in a major geomorphic and fish pathway change (Beamer and Wolfe 2015). Channel migration slowed in 2015. They examined presence (1 or 0) and abundance when present of species present in at least 3% and 10% of samples, respectively. Explanatory variables included smooth effects of depth, salinity, temperature, velocity, and day of year; linear quantitative effects of set area (delta only), categorical effects of gear type (small or large beach seine), tide stage (ebb, flow, high, or low), and channel type (blind or distributary; delta only), and random intercepts of year and station.

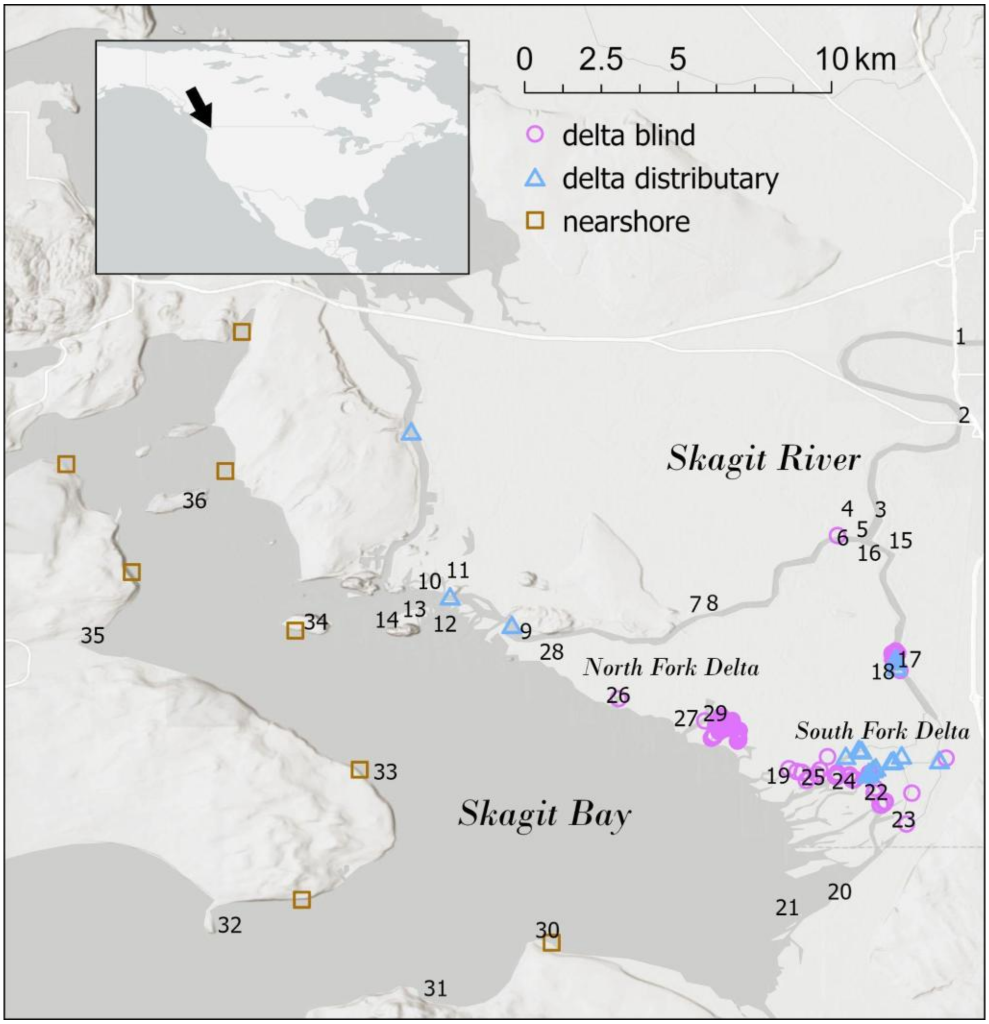

Figure 4.2. Map of relevant locations in the Skagit River estuary. Shapes indicate sampling locations in delta and nearshore waters. Delta sampling was split between blind and distributary channels. Numbers identify locations where we predicted fish responses to water conditions predicted by the hydrodynamic model, which made water quality predictions across the entire estuary.

The hydrodynamic model generated hourly predictions of local water quality across the study region as a product of different flow regimes. It considered nine different flow regimes, which included combinations of three different water use scenarios (current, alternative one, and alternative two) and three different years with different levels of water availability (2010, 2015, and 2019) (Table 4.1). The Current Water Use scenario refers to the Skagit River's present arrangement, whereby junior agricultural water rights can be interrupted when river flow drops below levels specified by the instream flow rule. Alternative 1 represents a hypothetical situation in which 200 CFS of junior water rights are not interruptible. Alternative 2 represents a hypothetical situation in which 390 CFS of irrigation needs are made uninterruptible in addition to 200 CFS of junior water rights. In addition, water year 2010 had average flow, water year 2019 had moderately low flow, and 2015 had the lowest flow on record. Detailed explanations of these scenarios and the underlying water rights are available in Chapter 1. Altogether, multiple versions of the hydrodynamic model forced by variable levels of natural water availability and water use enabled us to explore fish responses across a realistic range of flow conditions and contextualize effects of water withdrawals relative to natural variation in water scarcity.

Table 6.1. Factors used to simulate flow regimes. Nine flow regimes were simulated from each combination of the three water years and three water use scenarios.

Term	**Definition**	**Levels**	**Summary**
Water year	Real-world water availability in the Skagit River during specified water year	2010	Average water availability
		2015	Lowest water availability on record
		2019	Moderately low water availability
Water use scenario	Water withdrawals allowed based on instream flow rules and water rights	Current Water Use	Status quo water use
		Alternative 1	Higher water use
		Alternative 2	Highest water use

Novel analyses

We used predictions from the fish-environment and hydrodynamic models to make predictions of fish responses in specific locations and times. In these linked models, the predictor variables of interest were depth, salinity, and velocity, which were predicted by the hydrodynamic model and were also used to inform fish-environment models. The hydrodynamic model did not predict temperature, but this was a predictor variable in fish-environment models. In linked models, we therefore used average temperature values measured during monitoring within relevant time periods (see below) to represent predictors with neutral effects on fish abundances. Likewise, a parameter in fish-environment models in the delta was set area and we used set area averaged across all samples as a neutral predictor. Another parameter in fish-environment models was whether a delta site was a blind or distributary channel and we used the real-world sites' designation as blind or distributary channel to make predictions. Additionally, we set effects of all random intercept parameters to zero. A day-of-year parameter that accounted for seasonal phenology was also in fish-environment models and we made predictions corresponding to the days of year of relevant time periods (see below).

The remaining tide (e.g., ebb, flow) and gear (e.g., large beach seine, small beach seine) parameters in fish-environment models were categorical, meaning that models must be arbitrarily informed by one level of the categorical variable to make predictions, which raise and lower predictions based on whether (1) or not (0) the level is specified. We arbitrarily made predictions that specified an ebb tide and using the large beach seine method so that predictions were uniformly raised or lowered. A nuance in such predictions is that these categorical variable effects, while uniform, occurred on logit (fish presence) and log (fish abundance when present) scale and then predictions were back-transformed to the arithmetic scale for presentation purposes. Moreover, there was some variation among species in response to the tide state (the effect of gear was comparatively uniform, with more fish caught using the large beach seine). Thus, the absolute numbers of fish abundances that models predicted should be interpreted cautiously because they depended to some degree on the selection of categorical variable levels. The qualitative (e.g., increase or decrease in fish abundance) and collective (e.g., sites and species that responded more or less to changes in flow regimes) properties of predicted responses to changes in flow regime were not affected by this nuance.

A challenge in this analysis was to distill important patterns from an enormous amount of information. The hydrodynamic model generated predictions of water quality at relatively fine temporal and spatial resolution, constituting an immense amount of predictions across time and space throughout the estuary. Moreover, we previously built fish-environment models that could predict responses to different flow regimes for numerous fish species. To produce a manageable number of fish response predictions, we therefore focused on select species, locations, and time periods.

The impetus of this study was to understand responses in Chinook salmon, so we first examined univariate responses of this species in 36 sites intended to represent the breadth of habitats across the estuary (Fig. 4.2). The 36 sites were selected for their representativeness of the estuary and that the sites were sampled for at least a full season for model validation purposes. We focused on two time periods when Chinook salmon were relatively abundant and when water withdrawals were ongoing: May 30 June 12, when water withdrawals were moderate and salmon abundance was high and July 15 - July 28, when water withdrawals were high and salmon abundance was moderate. These periods were two weeks long to capture one complete spring and neap tidal cycle and thus avoid bias that could be imposed by making comparisons during periods that experienced differential tidal effects on water quality.

To summarize habitat use, we summed hourly predictions across the two-week period. This was necessary because the hourly time step of the hydrodynamic model required us to calculate predictions of habitat use by fish on an hourly time step. The two-week sums of habitat use were the values that we presented directly in maps of Chinook salmon abundance and that we input into multivariate analyses (see below) of assemblage composition.

To corroborate the results of the linked fish-environment and hydrodynamic models, we wanted to explore a more direct fish-river flow relationship akin to river flow-dependent inundation reported by Duke Engineering (1999). In a separate analysis, we used general additive models (GAMs, without mixed effects) to explore daily Chinook salmon presence in relation to daily mean flow and daily range in channel sites. We focused on channel sites because they were shallower and therefore logically more sensitive to tidal inundation levels than deeper sites. GAMs are useful to explore nonlinear relationships, as they can be used to fit multiple parameters without explicitly representing the "shape" of the relationship between dependent and independent variables. Like other statistical models, they can also be used to control the effect of one variable (e.g., tides), to evaluate the independent effect of another (e.g., river flow). In these GAMs, the response variable was daily Chinook salmon presence summed across 24 hours and the explanatory variables were sites daily depth range (i.e., maximum minus minimum values), and daily mean flow as measured at USGS gage 12200500 in Mt. Vernon. We fit separate GAMs for each site so that we could explore patterns among sites depending on their characteristics such as distance to the river mainstem. We used all model years' predictions but only those of the current water use scenario to fit GAMs to flow and tidal patterns.

After examining the responses of Chinook salmon to different flow regimes and in light of the changes in water quality predicted by the hydrodynamic model (Wang et al. 2025), we examined fish community structure in light of the different water use scenarios for each water year. We selected fish species that were known for their cultural and economic importance (i.e. salmon) and for their abundance and importance to estuarine ecosystems in Puget Sound (Pietsch and Orr 2015). We used the previous Chinook salmon results and hydrodynamic results to guide our analysis. We focused on 18 sites that contrasted in sensitivity of water quality to the flow regime and examined a time period when effects were potentially greatest. This included the South Fork of the delta where water quality was sensitive to the flow regime (sites 19-25; Fig. 4.1), and sites in the North Fork and areas upstream (sites 1-6, 15-16; Fig. 4.1) where water quality was comparatively insensitive to the flow regime. In particular, salinity increased in the south fork of the delta but less so in upriver sites under low flow conditions. We examined the July 15 - July 28 period, during which simulated water withdrawals were greater than May 30 - June 12.

Using presence predictions over the time period from GAMM approach (Appendix 4.1), we evaluated fish community change with nonmetric multidimensional scaling (NMDS) on dissimilarity quantified by a Bray-Curtis matrix to visualize predicted effects of different flow regimes on assemblage composition. We divided presence predictions by the total time of present to attain proportional composition at a site. We then have site, year, and water use scenario classifiers for each site. We generated species vectors showing gradients in ordination space of the NMDS plot using the vegan function envfit (Oksanen et al. 2025).

Our analysis did not include tests of statistical significance because the number of data points – thus statistical power – used to make various comparisons was arbitrarily determined by the number of data points generated by predictive models and because predicted responses to habitat use were inevitable given the sensitivity of fish to water quality parameterized in fish-environment models (Munsch et al. 2025) and the changes to water quality predicted by the hydrodynamic model (Wang et al. 2015).

Analyses were performed in R version 4.3.1 using the packages mgcv and vegan (Wood 2004, R Core Team 2025, Oksanen et al. 2025).

Results

Chinook salmon abundance

Several patterns emerged from predicted responses of Chinook salmon to different simulated flow regimes (Figs. 4.3, 4.4). Overall, presence in alternative water use scenarios compared to the current scenario ranged at the site level from ~7% loss to 2% gain, with most sites losing presence. Changes to abundance when present in alternative water use scenarios compared to the current scenario ranged at the site level from ~8% loss to 4% gain, with most sites losing abundance when present. Changes in salmon habitat use were greatest in the south fork of the delta, during 2015, during the July 15-28 time period, and under water scenario alternative two. Areas where abundance did not change or slightly increased relative to current water use scenarios were closer to freshwater input. In contrast, areas in the delta – particularly the south fork – and greater Skagit Bay generally declined during alternative use scenarios. Uncertainty in model predictions was relatively constant among water use scenarios although there were slight discrepancies in the delta South Fork (See Appendix 4.2).

Predicted Chinook salmon presence increased when flows were higher (Fig. 4.5), particularly during the summer period. This pattern was generally true among sites, although the shape and magnitude of the relationship varied among sites (Fig. 4.6). The influence of daily tidal range also varied among sites (Fig. 4.6). Note that in both Figure 4.5 and 4.6, the range of both river flows and tidal range differs between time periods. Flows of less than 12,000 CFS were not observed in the three model years in the spring time period, and flows greater than 23,000 were not observed in the summer time period. This might complicate predictions of the overall shape of the relationship between, say, river flow and occurrence of Chinook salmon. Nevertheless, for most sites and time periods, the relationship between flow and fish was positive below 20,000 CFS.

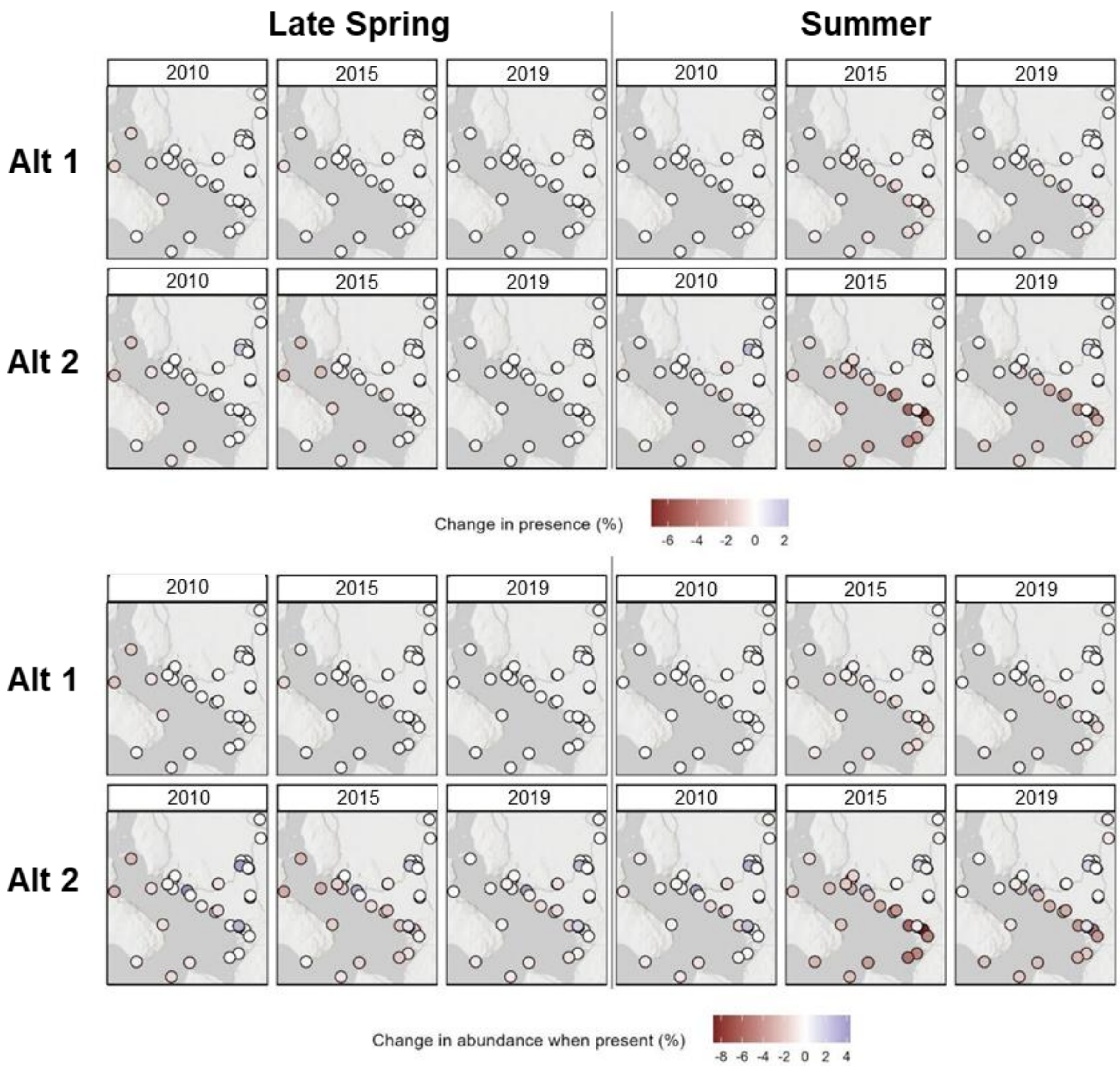

Figure 4.3. Changes in Chinook presence (top panels) and abundance when present (bottom panels) in water use scenarios Alternative 1 (Alt 1) and Alternative 2 (Alt 2) for late spring (5/30-6/12) and summer (7/15-7/28) in each of three model years. Values are from predictions of a given year and water use scenario minus predictions from the same year's Current Water Use scenario.

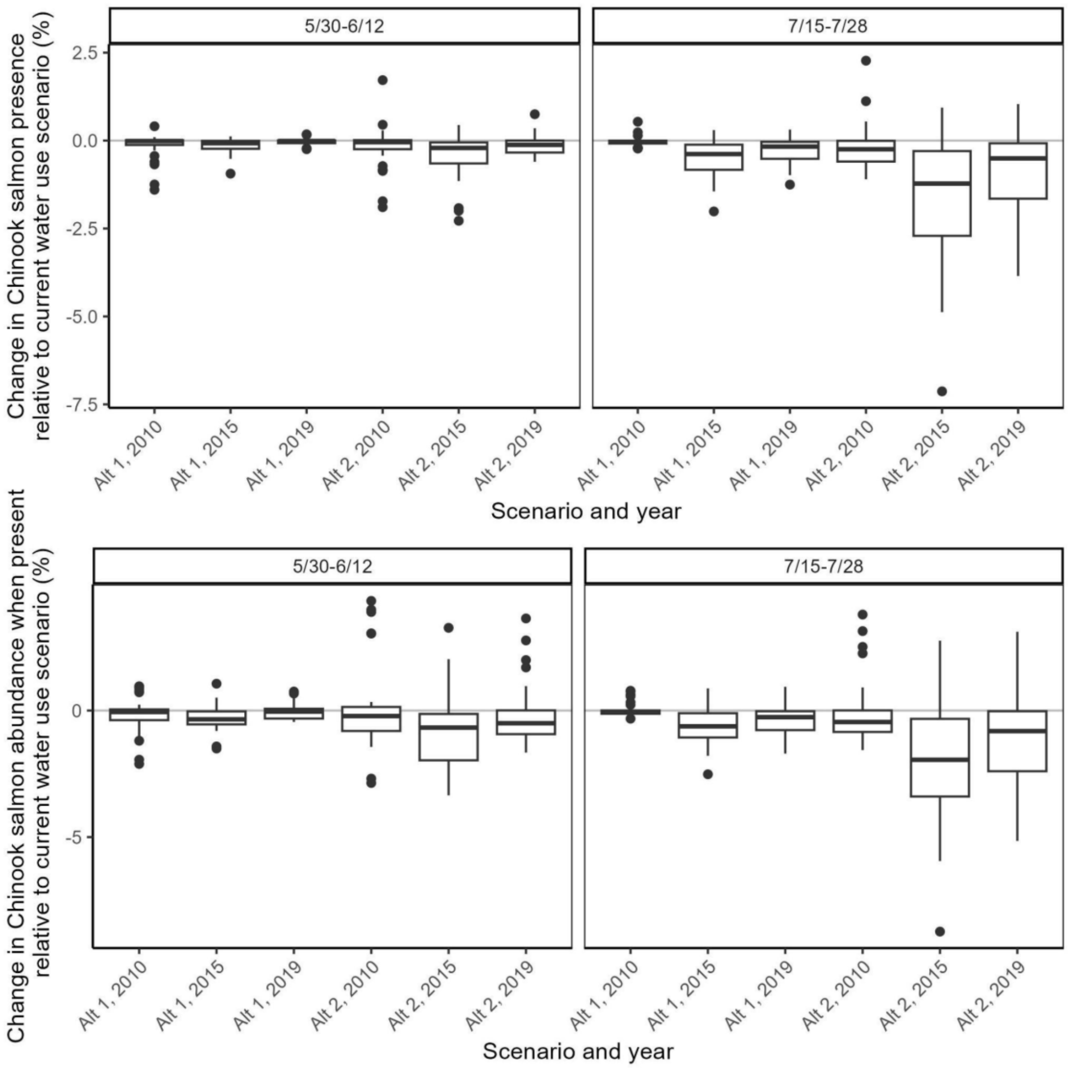

Figure 4.4. Boxplots summarizing changes across 36 sites in Chinook salmon abundance in two alternative water use scenarios relative to present scenarios. The grey line indicates zero change.

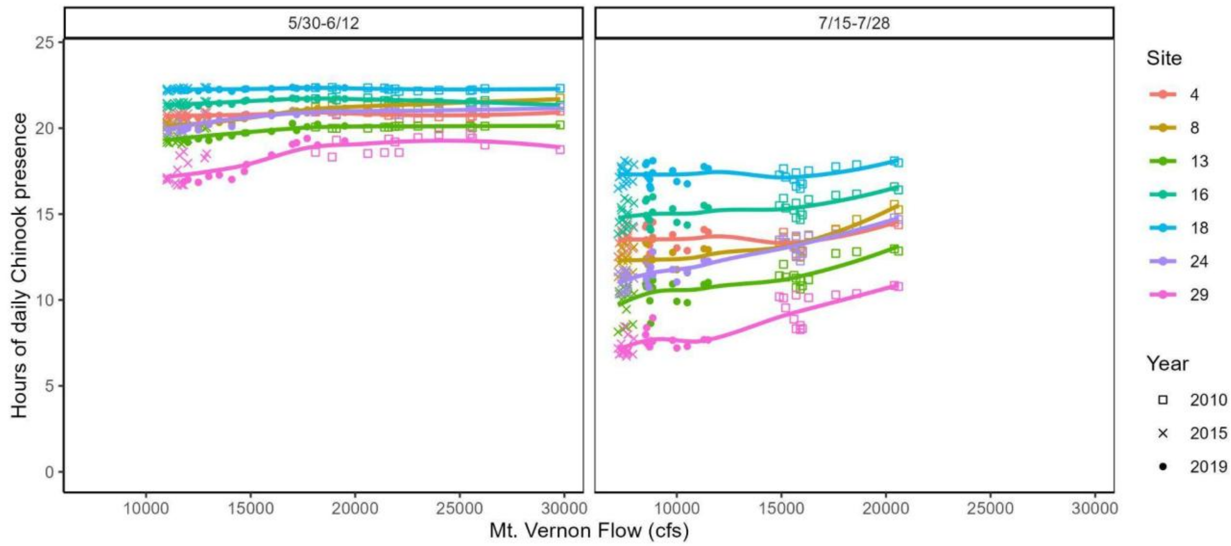

Figure 4.5. Daily Chinook salmon presence compared to flow. Model outputs are based on the current scenario and include all years. Lines show loess local regression fit to model outputs from the same site.

Assemblage composition

NMDS plots indicated that predicted assemblage composition varied among flow regimes and sites were differentially sensitive to changes (Fig. 4.7). Changes to the assemblage were most apparent in water year 2015 and in the south fork of the delta versus areas upstream. Site effects predominated over year effects, which predominated over scenario effects. This was evidenced by sites clustering together regardless of year or scenario, the year 2015 and to a lesser extent 2019 producing a point spread toward the bottom right of ordination space, and different scenarios within the same year producing negligible to slight spreads toward the bottom right of ordination space. These slight differences are within areas where we believe the largest differences are likely occurring. We believe these predicted differences are associated with changes in the salinity at the site and the saltwater tolerance of some fish species. Namely, the hydrodynamic model suggests in 2015 low river flow conditions that additional water withdrawals will increase salinities in the South Fork Skagit (Chapter 2). Fish and environment relationships (Appendix 4.1) exhibit that increase salinities result in a decline of anadromous Chinook salmon and coho salmon and freshwater sculpin, but more euryhaline species (e.g. staghorn sculpin and threespine stickleback) are less affected.

Salinity is not the only drive, hence the importance of the NMDS. Water depth was also important for some fishes (e.g. shiner perch) and that presence from the fish-environment models is likely an interplay of multiple site-specific factors. More importantly for the South Fork, we do observe a shift in species community structure from additional water withdrawals during the lowest river discharges.

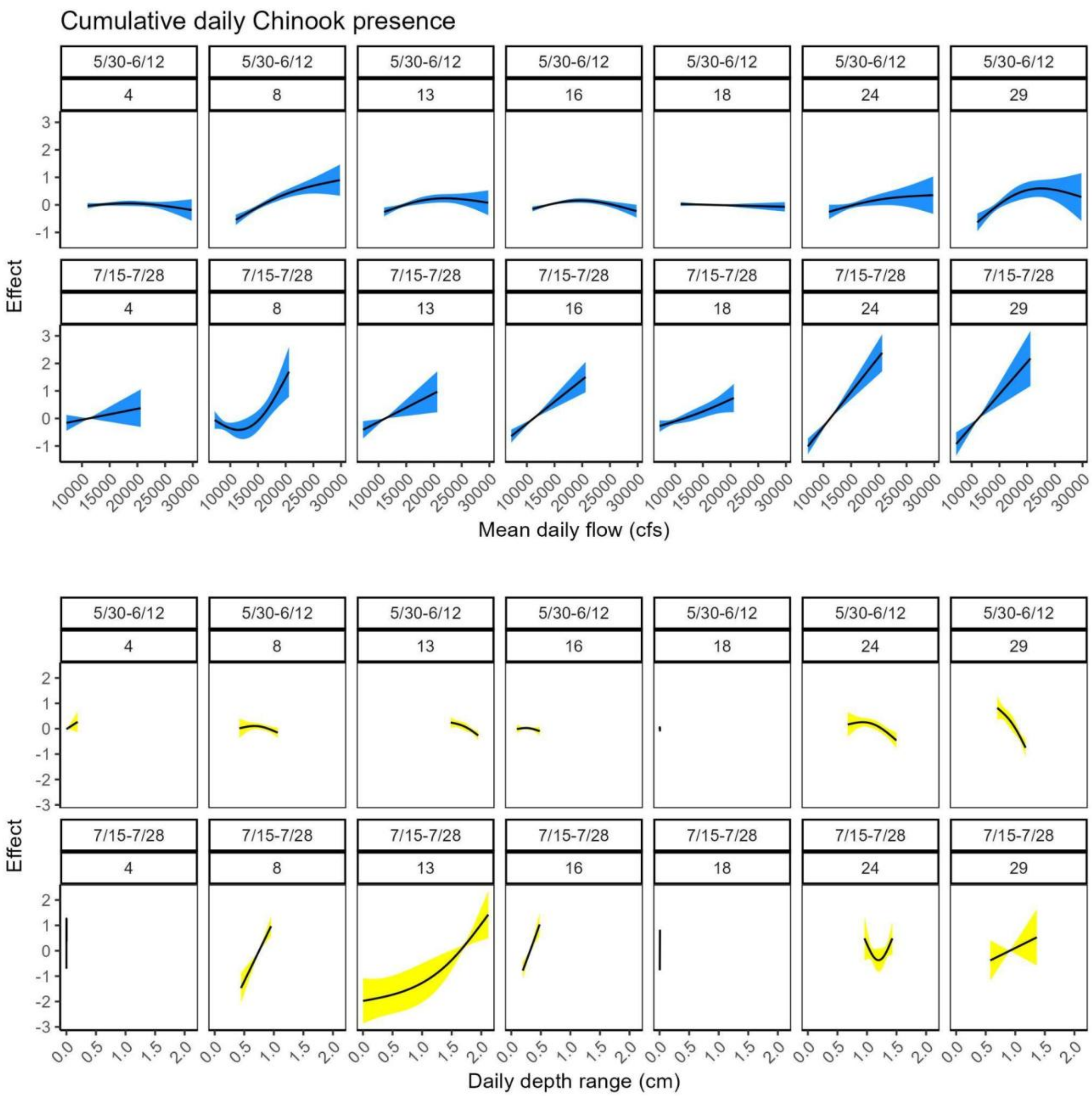

Figure 4.6. GAM output showing the influence of flow and daily depth range on daily Chinook salmon presence. Numbers identify sites. Shaded areas indicate 95% confidence intervals.

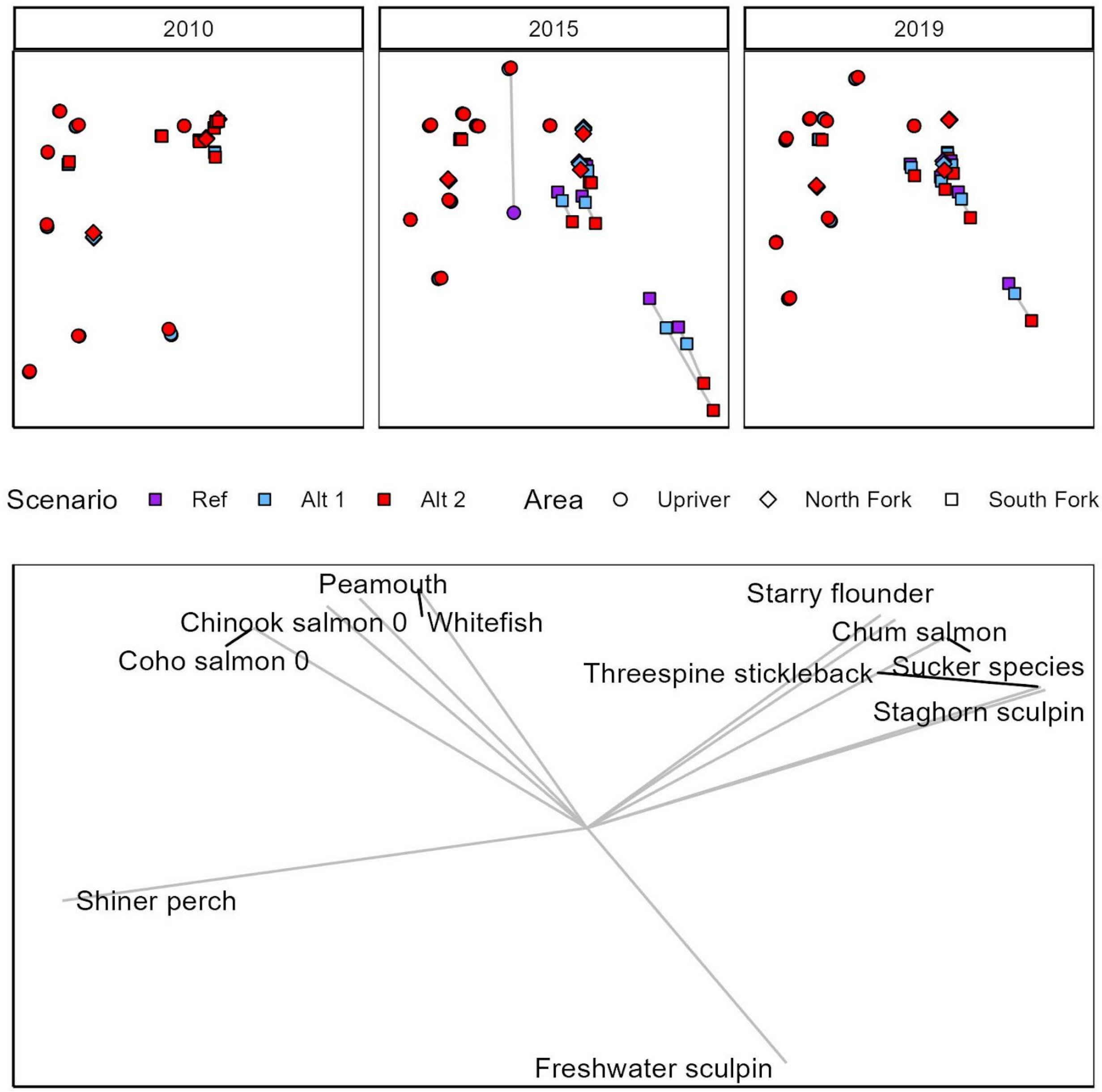

Figure 4.7. Nonmetric multidimensional scaling (NMDS) plots that compare assemblage composition in mainstem, North Fork, and South Fork delta across different years and water use scenarios. One ordination is shown, with a stress level of 0.07, points faceted by year, and vectors shown separately on the bottom panel to enhance visibility. Gray lines in the top figure row connect observations from the same site. The same number of points are in each top panel, but a lack of effects among scenarios can cause points from the same site to be superimposed.

Discussion

We used fish-environment (Appendix 4.1) and hydrodynamic models (Chapter 2) to predict changes in Chinook salmon and fish community structure in the Skagit River estuary depending on different flow regimes. These flow regimes represented natural variation in annual water availability and human use scenarios. Simulated water years and water use scenarios that decreased freshwater input to the estuary reduced predicted Chinook salmon habitat use and changed predicted assemblage composition. The effect of natural water scarcity as simulated by the water availability of the lowest flow year on record was greater than the effect of the water use scenarios. Decreases in Chinook salmon habitat use were on the order of single digit percentages and most apparent in the water year with lowest water availability and the water use scenario with greatest water extraction. Given that the hydrodynamic model primarily affected salinity and species' differential relationships with salinity, the predicted changes to habitat use were probably driven by changes in predicted salinity. Indeed, in response to different flow regimes, areas such as the South Fork experienced the greatest changes in predicted salinity and the greatest changes in fish habitat use while the opposite occurred in more riverine areas. Moreover, the direction of these changes was often consistent with salinity preferences. For example, lower flow regimes were predicted by the hydrodynamic model to increase salinity while Chinook salmon were predicted by the fish-environment models to be more abundant in lower salinity waters; thus, lower flow regimes reduced Chinook salmon abundance via increases in salinity. This research provides a decision-support tool for people tasked with balancing water use for human needs and fish conservation and may provide a framework for integrating hydrodynamic and fish-environment models to understand potential ecological impacts of human-influenced flow regimes in other estuaries.

We found that correlations between river flow and juvenile Chinook salmon emerged from predictions from hydrodynamics. At the broadest level, the relationship between river flow and fish abundance should be a unimodal pattern, i.e., the highest abundance at moderate flows and low abundance at both very low and very high flow levels. These patterns were reflected in the emergent nonlinear relationship between daily Chinook occurrence and Skagit River flow in late spring: an increasing relationship up to approximately 20,000 CFS, above which flows had less effect on distribution or even a negative effect depending upon site. However, in the summer, where the higher levels of river flow were much restricted and lower flows were more common, a strongly positive relationship existed for almost all sites. The time period from July to August when these relationships exist coincides with juvenile Chinook salmon migration out of the delta, as well as when water needs in the lower Skagit by people greatly increase. Hence, in average years, water scarcity likely restricts the tail end of the juvenile Chinook salmon rearing period. In lower flow years when snowmelt is much less robust, these effects likely shift into earlier time periods as well.

Given these relationships, it may seem paradoxical that we observed relatively subtle differences in changes in occurrence of juvenile Chinook salmon and other estuarine species across scenarios, years, and time periods when river flow levels were reduced due to natural variation (model years) and water use (scenarios). It is worth noting that even in situations with greatest water scarcity (low flow years, high water use scenario, summer time period), water use comprised no more than 20% of the mainstem flow, a relatively modest total withdrawal in comparison to other systems which can be over 50% appropriation. Our interpretation is that even in the periods of greatest water scarcity, river flows in the larger North Fork were sufficient to support fish but approached conditions impacting occurrence and abundance in the shallower South Fork.

In this context, it is worth considering the potential for future water use needs to conflict with habitat restoration goals. Restoration projects will not function as planned if the modified flow regime prevents fish from using these areas (Munsch et al. 2020). We observed the highest impacts to fish habitat occurrence in the South Fork of the Skagit delta, a subunit that includes the majority of habitat restoration projects to improve habitat conditions for ESA listed Chinook salmon. It is therefore worth keeping in mind that any changes to water use in the Skagit should be considered in light of the benefits to salmon populations (Greene et al. 2024, Greene et al. 2025) afforded by cumulative restoration efforts.

Model Uncertainty

There are several levels of uncertainty in our analysis. First, our use of GAMMs (Appendix 4. 1) used to predict fish-environment relationships includes statistical uncertainty from the fitting exercise including parameter covariance. We tried to minimize the effects of parameter covariance by including random effects and categorical variables in the GAMM that helped partition variance, but they nevertheless exist to some extent. One important variable influencing fish densities in the Skagit delta is competition with other individuals (Greene et al. 2025). We included this element, which was not the central focus of this study, only through site and year random effects, but as shown in Greene et al. (2025), the abundance of migrants entering the delta is important for predicting presence and local density. In addition, we have opted to use these models that focus on distribution and abundance instead of other possible model frameworks that could examine growth or survival because there is more certainty from direct be measurement. However, future efforts could examine how flow-related changes in presence and abundance might translate to changes in individual residence and movement, and thereby influence changes in growth and survival.

A second level of uncertainty is the potential compounding effect of uncertainty from use of hydrodynamic output (Chapter 2). Of the ways in which effects might be compounded, we note that the hydrodynamic model tends to underestimate salinity levels at some locations (Fig 2.9), thereby potentially reducing the expected effect on juvenile Chinook salmon. Hence, with better hydrodynamic predictions of salinity, we might expect projected Chinook presence and abundance to be even lower. In addition, one important hydrodynamic output that wasn't produced by the model was expected changes in water temperature. We used seasonal patterns observed for each model year instead. If lower flows increase temperature, the effect on juvenile Chinook will likely be worse.

A third level of uncertainty is a number of extrinsic factors that were not incorporated into the modeling effort. Notably, we were unable to combine vegetation model predictions with juvenile Chinook salmon response. While some of these relationships are complex and would be hard to predict (e.g., how does fish presence change with dominance of different emergent marsh species?), one considerably important one is the effect of scrub-shrub vegetation on temperature and prey (Greene et al. 2020). Other interactions of interest include the influence of beaver to form deeper pool habitat and the potential for scrub-shrub habitat to provide cover from aerial predators.

Independent of marsh vegetation, several important other extrinsic factors that may influence the distribution and abundance of juvenile Chinook salmon and other fishes in the Skagit delta and bay include upriver timing of flows through hydropower modification, and climate impacts such as changes in flow, temperature, and sea level. These are expected to have cumulative effects on salmon populations (Crozier et al. 2019), and all likely will influence residence in the tidal delta.

Implications for water and salmon management

Our combination of hydrodynamic, vegetation, and fish distribution modeling has identified several important implications for both water and management of threatened salmon populations. First, the existing basis for the Skagit Instream Flow Rule is generally supported – lower river flows generally translate to reduced inundation and higher salinity, the combination of which has predictable effects on wetland vegetation and juvenile Chinook salmon, particularly in the summer. However, due to nonlinearities of the flow-habitat relationship as well as spatial differences in depth and proximity to the river and tides, hydrodynamics, vegetation, and juvenile Chinook salmon may be more resilient minor changes in water use than the Duke study (1999) concluded. Whether this statement stands up to shifting baselines in the Skagit hydrograph and sea level resulting from climate change is an important question worth further examination.

However, an effective doubling of water use as modeled in our scenarios will likely have impacts to residence of juvenile Chinook salmon, particularly in low-flow years. This finding is especially important in light of the finding that areas targeted for habitat restoration for juvenile Chinook salmon (Greene et al. 2025) appear to be at greater risk to changes in salinity.

This work therefore highlights the importance of considering salmon management in the context of multiple cumulative effects (Munsch et al. 2020). In the example above, we highlight the potential combined influence of water use and restoration; other important cumulative effects include changes in flow regimes from hydropower operations as well as climate impacts as noted above. Hence, we see this report as a preliminary foray into a broader examination of how multiple changes in the Skagit Watershed will influence estuarine habitat and the fishes that use it.

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Appendix 4.1. Effects of local environmental variation on occurrence and abundance of estuarine fishes

Stuart Munsch, Correigh Greene, and Michael LeMoine

Introduction

Ecological research seeks to understand interactions that shape species abundances and distributions (Krebs 1972). A key but often deficient component of this effort is the study of natural history (Able 2016). Natural history can be defined as "the observation and description of the natural world, with the study of organisms and their linkages to the environment being central" (Tewksbury et al. 2014). Our ability to anticipate ecosystem dynamics, particularly in environments that humans modify, relies on such knowledge (Tewksbury et al. 2014, Able 2016). One important application of natural history knowledge is ecosystem-based fisheries management, a framework that strives to sustain fisheries resources by making decisions in light of understanding the ecological interactions and contexts that influence them (Link 2010). Indeed, many ecological crises including those that impact fisheries can be attributable to past human stressors that were imposed on ecosystems with little consideration for, or awareness of, the natural history of commercially and culturally valuable species (Yoshiyama et al. 1998, Lichatowich 1999). Research that enhances knowledge of species' natural histories, such as relationships with habitat attributes, can provide managers and constituents with information about potential consequences of human modifications to ecosystems.

Natural history knowledge is critical for understanding biological dynamics in estuaries. Areas where rivers drain into the sea support shifting physical environments that drive pervasive responses in taxa like fish (Martinho et al. 2007, Columbano et al. 2022). Many estuaries draw research and management attention because they support fish nurseries and other productive habitats (Beck et al. 2001, Sheaves et al. 2015) that can be stressed by human activities (Greene et al. 2015, Munsch et al. 2017, Toft et al. 2018, Hogson et al. 2020). Indeed, much of the human population lives near estuaries and coasts (Small and Nicholls 2003), and natural resources in these areas have declined globally (Lotze et al. 2006). Additionally, estuarine species express diverse life histories and habitat preferences that can generate differential responses to environmental attributes (Love 2011, Hughes et al. 2014, Williams et al. 2017, Columbano et al. 2022). Thus, it is beneficial to observe a diversity of species to understand how estuarine fish assemblages may respond to shifts in the physical environment. Overall, given that estuaries support many species but their environments are often modified by human activities, it is important to understand linkages between species like fish and local environments to infer how they will respond to natural and anthropogenic changes to the environment.

Estuaries in Puget Sound (WA, USA) epitomize this need. Puget Sound is a temperate fjord estuary complex where numerous rivers drain into the Salish Sea and the Pacific Ocean beyond. Puget Sound is inhabited by a diversity of fishes such as Chinook salmon (*Oncorhynchus tshawytscha*) that use estuaries as nurseries where small prey and predator refuge are abundant (Simenstad et al. 1982). Many of Puget Sound's estuaries have been converted to urban and agricultural landscapes (Simenstad et al. 2011). For example, the urban cities of Everett, Seattle, and Tacoma lie on the historical footprint of the Snohomish, Duwamish, and Puyallup River estuaries, respectively, and farmland lies on the historical footprint of the Skagit and Stillaguamish River estuaries (Simenstad et al. 2011). Development of these areas has imposed human stressors on estuaries, including modification to flow regimes that influence environmental attributes such as salinity and temperature (Greene et al. 2015). Puget Sound's human population continues to grow (PSRC 2024) yet its fishes are integral to the culture, which has prompted efforts to understand links between its environment and fishes so that managers can sustain ecosystem services amidst development (Ruckelshaus et al. 2009).

Here we leverage nine years of monitoring data to quantify links between fish abundance and water quality variables in delta and nearshore waters of the Skagit River estuary. The Skagit River is Puget Sound's largest river and produces most of its Chinook salmon that are listed as threatened under the U.S. Endangered Species Act (NOAA 2007). Its estuary is extensively inhabited by salmon and other fish taxa that express a diversity of life histories. Our goal was to understand how local attributes of water quality including depth, salinity, temperature, and velocity influence habitat use and thus better understand the natural history of this system and the potential for fish to respond to natural and anthropogenic disturbances to its physical environment. More generally, estuaries are poised to benefit from ecosystem-based management approaches because their fish assemblages are often valued by people and strongly linked to dynamic physical environments altered by human activities (e.g., Martinho et al. 2007, Williams et al. 2017, Columbano et al. 2022). We used the relationships documented in this appendix to predict the distribution and abundance of fishes in water use scenarios and model years (Chapter 4) using hydrodynamic model outputs (Chapter 2).

Methods

Study system

The Skagit River drains 6,900 km^2 of forested, mountainous land primarily in Washington (USA) but also British Columbia (CAN) (Fig. 4.1.1). It expresses a transitional hydrologic regime whereby flow peaks in winter and summer due to precipitation concentrated around winter months followed by freshets (Beechie et al. 2006). Its flow regime is modified by regulation from dams and water extraction to support agriculture, hydropower, and various municipal activities. The Skagit River enters Puget Sound via two forks that flow through an agricultural landscape with some natural areas that form a modified delta complex (Simenstad et al. 2011). The natural state of this estuary included a large tidal delta with numerous blind and distributary channels within a wetland landscape (Collins 2000). Diking, dredging, and filling mostly in the mid-nineteenth century greatly reduced the delta's footprint, and ongoing restoration efforts have begun to offset some losses (Beamer et al. 2005, Chamberlin et al. In Review). The delta opens into Skagit Bay, which is semi-enclosed by various land masses but is connected to the rest of Puget Sound to the South and the more oceanic Strait of Juan de Fuca to the west. Nearshore areas of Skagit Bay include sandy beaches, mudflats, and modified residential shorelines typical of developed but not highly urban Puget Sound (Simenstad et al. 2011). At the seascape scale, Puget Sound provides a mosaic of delta and nearshore ecosystem types including marshes, sandy beaches, mudflats, kelp forests, eelgrass meadows, and rocky intertidal zones.

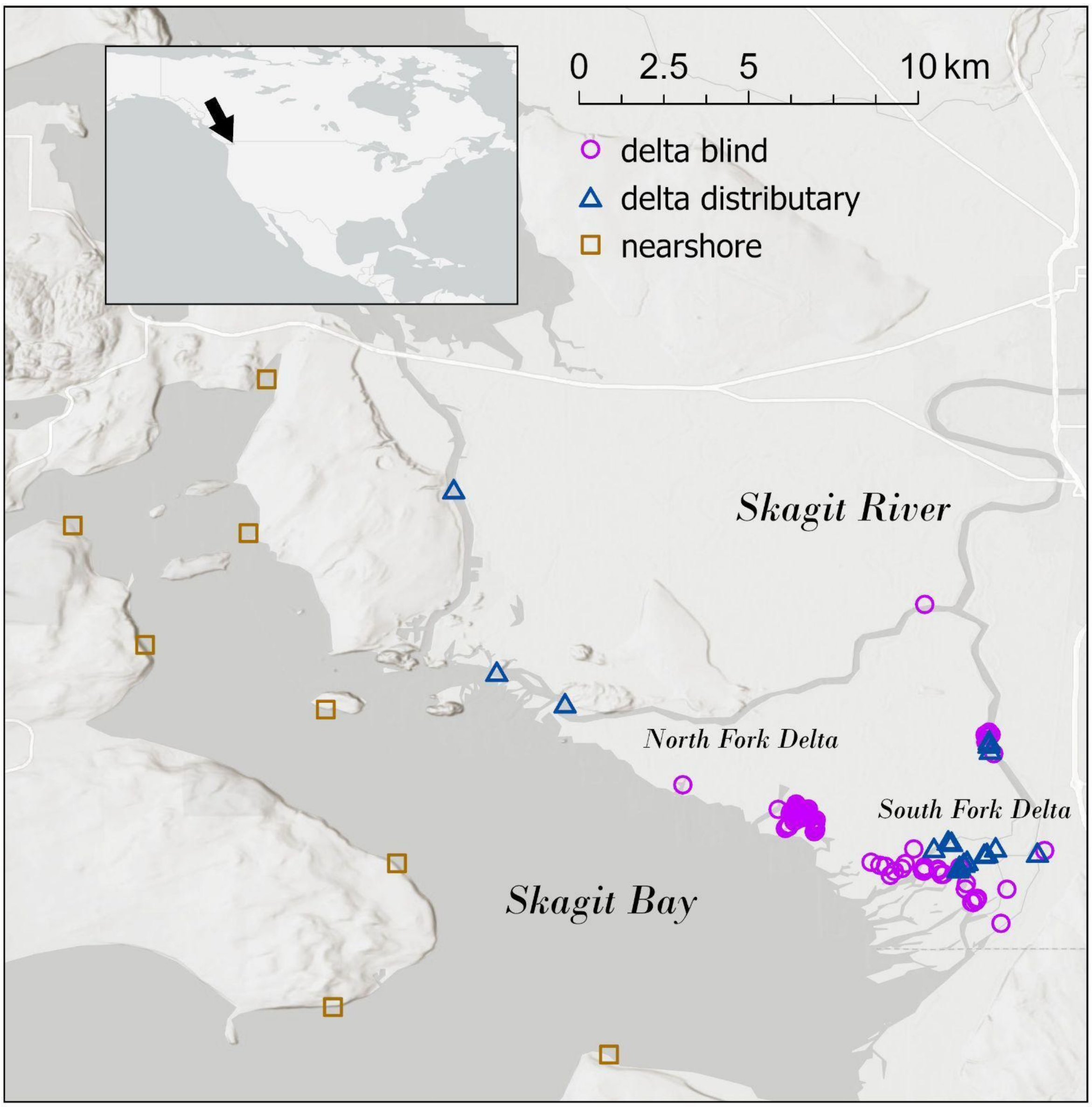

Figure 4.1.1. Map of the study region and sampling sites in delta and nearshore waters. Within delta waters, sampling took place in blind and distributary channels.

Puget Sound is inhabited by many fish species that aggregate in delta and nearshore areas primarily in spring and summer (e.g., Pentilla 2007, Toft et al. 2007, Munsch et al. 2016). These fishes express a breadth of life histories (e.g., anadromy, beach spawning), morphologies (e.g., flatfish, “silvery, spindle-shaped” forage fish), and habitat preferences (e.g., constant movement in the water column, punctuated movement along the benthos) (Love 2011, Hughes et al. 2014, Munsch et al. 2016). Its deltas and nearshore areas are especially important for small and juvenile fish because they provide predator refuge and abundant prey from terrestrial, benthic, and planktonic realms (Simenstad et al. 1982, Healey 1982, Simenstad and Cordell 2000, Munsch et al. 2016). Puget Sound’s estuaries are requisite habitats for anadromous species like salmon because they are on migratory paths between watersheds and oceans (Simenstad and Cordell 2000). Juvenile salmon use estuaries as stopover habitats to grow before they enter marine systems where they experience size-selective predation (Duffy and Beauchamp 2011, Sawyer et al. 2023). Indeed, biologists have long recognized that estuaries are important components in Puget Sound’s seascape salmon nursery (Nagelkerken et al. 2015, Simenstad et al. 1982).

Fish and environmental monitoring

We sampled fish biweekly from Feb. - Aug. in 2015-2023 (Beamer et al. 2024). This entailed 3,294 and 5,069 beach seine hauls in delta and nearshore waters, respectively. Beach seines are nets deployed along shore that enclose shallow areas via float and lead lines that span the water column. Fish are captured as the ends of the net are drawn together and pulled landward.

We used two types of beach seines in delta and nearshore waters: small and large. Small beach seines were designed to sample shallow intertidal areas 1-2 m deep. Large beach seines were designed to sample intertidal-subtidal areas 2-5 m deep. The small beach was 80-ft (24.4 m) by 6-ft (1.8 m) by 1/8-in (0.3 cm) mesh knotless nylon net and the large beach seine was 37 m by 3.7 m by 0.3 cm mesh knotless nylon net. Both nets were set in “round haul” fashion by fixing one end of the net on the beach, while the other end is deployed by setting the net “upstream” against the water current, if present, and then returning to the shoreline in a half circle. Small beach seines were deployed and fished via wading out in the channel. Large beach seine sets were deployed by boat and retrieved by hand on shore. We primarily used large beach seines to sample nearshore waters so that we could reach intertidal and subtidal areas. We primarily used small beach seines to sample delta waters because these areas were often shallower, more constricted channels. However, we exclusively used the small beach seine during 2020-2021 due to restrictions during the COVID-19 pandemic.

Sampling regimes differed slightly between delta and nearshore waters. Due to the more constrained shoreforms of delta waters, we consistently sampled the same amount of area in nearshore waters but variable amounts of areas in delta waters. We therefore estimated a set area for each delta sample that we could account for as a proxy for fishing effort in analyses below, in addition to the influence of fishing with small versus large beach seines (details on set area calculations: Beamer et al. 2024, their Supplemental Information). Another nuance was that we divided sampling effort in the delta between blind (56% of obs.) and distributary channels (46% of obs.) whereas this distinction did not apply in nearshore waters, which were not channelized.

Captured fish were identified and enumerated. The identity of natural versus hatchery-origin salmon was inferred from the state of individuals' adipose fins that are clipped at hatcheries as well as detectable wire tags implanted by hatcheries in salmon nasal cartilage. We excluded hatchery salmon from our analyses to focus on the behavior of natural-origin salmon. Additionally, age-0 and age-1 Chinook and coho salmon inhabit this system and age classes can be readily distinguished via bimodal length distributions in delta, but not nearshore waters. Where possible (i.e., in the delta), we analyzed the two age-classes separately because they may use habitats differently. In practice, this led to us excluding observations of the rarer age-1 fish (see below). Chum and pink salmon were entirely age-0. Also, we lumped the identities of Prickly sculpin (*Cottus asper*) and coastrange sculpin (*Cottus aleuticus*) into one category called "freshwater sculpin" because these species were difficult to distinguish.

At the time of sampling, we directly measured or associated observations via time and date with environmental variables. Local variables included salinity, temperature, velocity, and depth. Salinity and temperatures were measured 0.1 m below the surface and 0.1 m above the bottom with a YSI multiparameter sonde. Measures were taken at the center of the small beach seine set and at 1 meter water depth for large beach seine sets. Water velocity was measured 0.1 m below the surface at the center of a small beach seine set. For large beach seine sets, water velocities were measured 1 m, 5 m and 10 m from the wetted edge of the shore. Water velocities were measured using either a Swoffer 3000 velocity meter or JDC Flowatch flowmeter. Water depth was measured with a stadia rod in the middle of each set and by boat if necessary in deeper waters. Additionally, we noted whether observations happened during ebb, flood, high, or low tides.

Statistical analyses

We used generalized additive mixed models (GAMMs; Wood 2004, Zuur et al. 2009) to quantify relationships between fish and local environmental variables. Fish counts were zero-inflated and we therefore separately modeled presence (0 or 1) and abundance when present, which we fit with binomial and negative binomial distributions, respectively. We modeled presence and abundance when present of species caught in >3% and 10% of samples, respectively, calculated separately for delta and nearshore waters. For brevity, we use the term "abundance" hereafter when we refer to both presence and abundance when present.

In these models, the explanatory variables were smooth effects of temperature, salinity, velocity, depth, and day of year; linear quantitative effects of set area (delta only), categorical effects of gear type (small or large beach seine), tide stage (ebb, flow, high, or low), and channel type (blind or distributary; delta only), and random intercepts of year and station. The purpose of the random intercepts was to account for variation among years and sites attributable to factors that we did not measure directly (e.g., annual recruitment, additional habitat attributes). We constrained smoothers to three knots to avoid excessive flexibility (e.g., sine-like waves) in their parameter estimates that were biologically implausible. Overall, we were primarily interested in understanding the potentially nonlinear relationships between fishes and local environments. We accounted for other factors (e.g., random effects, gear type, tide stage) so that models could better quantify the environmental effects. We did not consider alternative model parameterizations because we had strong prior reason to expect each explanatory variable to be influential to most species regardless of model evaluation criteria (Zuur et al. 2017).

We presented model output grouped by species with similar life histories to reduce clutter in figures and examine for patterns in habitat use among groups of species. We assigned species to groups following the categories and guilds proposed by Elliott et al. (2007) and refined by Potter et al. (2015), shown below in the Results. Notably, Pacific sandlance (*Ammodytes hexapterus*) can burrow in the benthos, but would have been in the water column when captured by nets.

Analyses were conducted in R version 4.3.1 (R Core Team 2025) using the package mgcv (Wood 2004).

Results

We focused our analyses on 11 species in delta waters and 20 species in nearshore waters that were captured in >3% of nets (Table 4.1.1). Among the more common species overall were threespine stickleback, Chinook salmon, staghorn sculpin, chum salmon, and starry flounder. Some species were more common in the delta than nearshore waters (e.g., freshwater sculpin, peamouth, whitefish) while others were more common in nearshore waters (e.g., surf smelt, snake prickleback, bay pipefish).

We observed fish across a range of environmental conditions (Fig. 4.1.2). The distributions of depth, temperature, and velocity were similar and skewed in delta and nearshore waters. In contrast, waters were much fresher in the delta and the distributions of salinity values were left and right skewed in delta and nearshore waters, respectively.

Table 4.1.1. Identification and occurrence of fishes in this study shown in descending order of occurrence. Occurrence is the number of times that a species was present divided by the number of samples. Species were modeled in terms of presence and abundance when present if they exceeded occurrences of 3% and 10%, respectively (excluded species not shown).

Region	Common name	Scientific name	Occurrence	Guild	Estuary use	Water column use	Figure group
Delta	Threespine stickleback	*Gasterosteus aculeatus*	0.527	Anadromous	Facultative	Midwater	1
	Chinook salmon (age-0)	*Oncorhynchus tshawytscha*	0.376	Anadromous	Facultative	Midwater	1
	Starry flounder	*Platichthys stellatus*	0.355	Marine estuarine-opportunist	Facultative	Benthic	2
	Chum salmon	*Oncorhynchus keta*	0.235	Anadromous	Facultative	Midwater	1
	Freshwater sculpin	*Cottus sp.*	0.154	Semi-Catadromous	Facultative	Benthic	3
	Staghorn sculpin	*Leptocottus armatus*	0.133	Marine estuarine-opportunist	Facultative	Benthic	2
	Peamouth	*Mylocheilus caurinus*	0.1	Freshwater straggler	Facultative	Midwater	3
	Shiner perch	*Cymatogaster aggregata*	0.089	Marine estuarine-opportunist	Facultative	Midwater	2
	Whitefish	*Prosopium williamsoni*	0.079	Freshwater straggler	Facultative	Midwater	3
	Coho salmon (age-0)	*Oncorhynchus kisutch*	0.077	Anadromous	Facultative	Midwater	1
	Sucker species	*Catostomus* sp.	0.032	Freshwater straggler	Facultative	Midwater	3
Nearshore	Shiner perch	*Cymatogaster aggregata*	0.498	Marine estuarine-opportunist	Facultative	Midwater	2

	Staghorn sculpin	*Leptocottus armatus*	0.445	Marine estuarine-opportunist	Facultative	Benthic	3
	Starry flounder	*Platichthys stellatus*	0.387	Marine estuarine-opportunist	Facultative	Benthic	3
	Saddleback gunnel	*Pholis ornata*	0.313	Marine straggler	Obligate	Benthic	4
	Surf smelt	*Hypomesus pretiosus*	0.289	Marine estuarine-opportunist	Obligate	Midwater	2
	Chinook salmon	*Oncorhynchus tshawytscha*	0.237	Anadromous	Facultative	Midwater	1
	Chum salmon	*Oncorhynchus keta*	0.223	Anadromous	Facultative	Midwater	1
	Threespine stickleback	*Gasterosteus aculeatus*	0.204	Anadromous	Facultative	Midwater	1
	Snake prickleback	*Lumpenus sagitta*	0.189	Marine straggler	Obligate	Benthic	2
	English sole	*Parophrys vetulus*	0.176	Marine estuarine-opportunist	Obligate	Benthic	3
	Bay pipefish	*Syngnathus californiensis*	0.134	Marine straggler	Obligate	Benthic	4
	Pacific sandlance	*Ammodytes hexapterus*	0.093	Marine estuarine-opportunist	Obligate	Midwater	2
	Pacific herring	*Clupea pallasi*	0.091	Marine straggler	Obligate	Midwater	2
	Pink salmon	*Oncorhynchus gorbuscha*	0.065	Anadromous	Facultative	Midwater	1
	Pile perch	*Phanerodon vacca*	0.049	Marine straggler	Obligate	Benthic	4
	Great sculpin	*Myoxocephalus polyacanthocephalus*	0.049	Marine straggler	Obligate	Benthic	5
	Padded sculpin	*Artedius fenestralis*	0.046	Marine straggler	Obligate	Benthic	5

	Crescent gunnel	*Pholis laeta*	0.042	Marine straggler	Obligate	Benthic	4
	Coho salmon	*Oncorhynchus kisutch*	0.038	Anadromous	Facultative	Midwater	1
	Sharpnose sculpin	*Clinocottus acuticeps*	0.033	Marine straggler	Obligate	Benthic	5

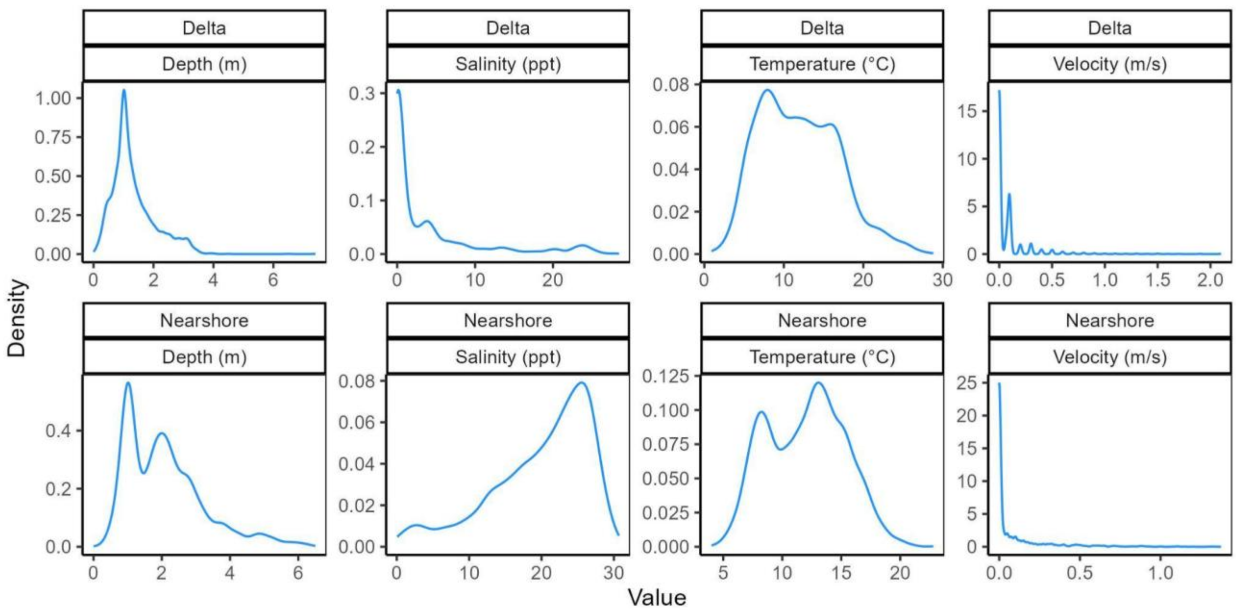

Figure 4.1.2. Distribution of local environmental variables observed concurrently with fish. Shown are kernel density estimates, which are smoothed representations of histograms.

Delta

The abundances of species observed in at least 3% of samples in delta waters varied with environmental conditions (Figs. 4.1.3 – 4.1.5). Some relationships were consistent among species. In general, abundance declined with increasing salinity and velocity. Indeed, only whitefish were more likely to be present in higher-velocity waters. Another pattern was that species expressed different, often-nonlinear relationships with temperature. Abundances of Chinook, chum, and coho salmon peaked at 11-12 °C and staghorn sculpin and shiner perch peaked at 14-17 °C. The abundance of other species plateaued with temperature, such as whitefish below 10 °C or stickleback and freshwater sculpin above 17 °C.

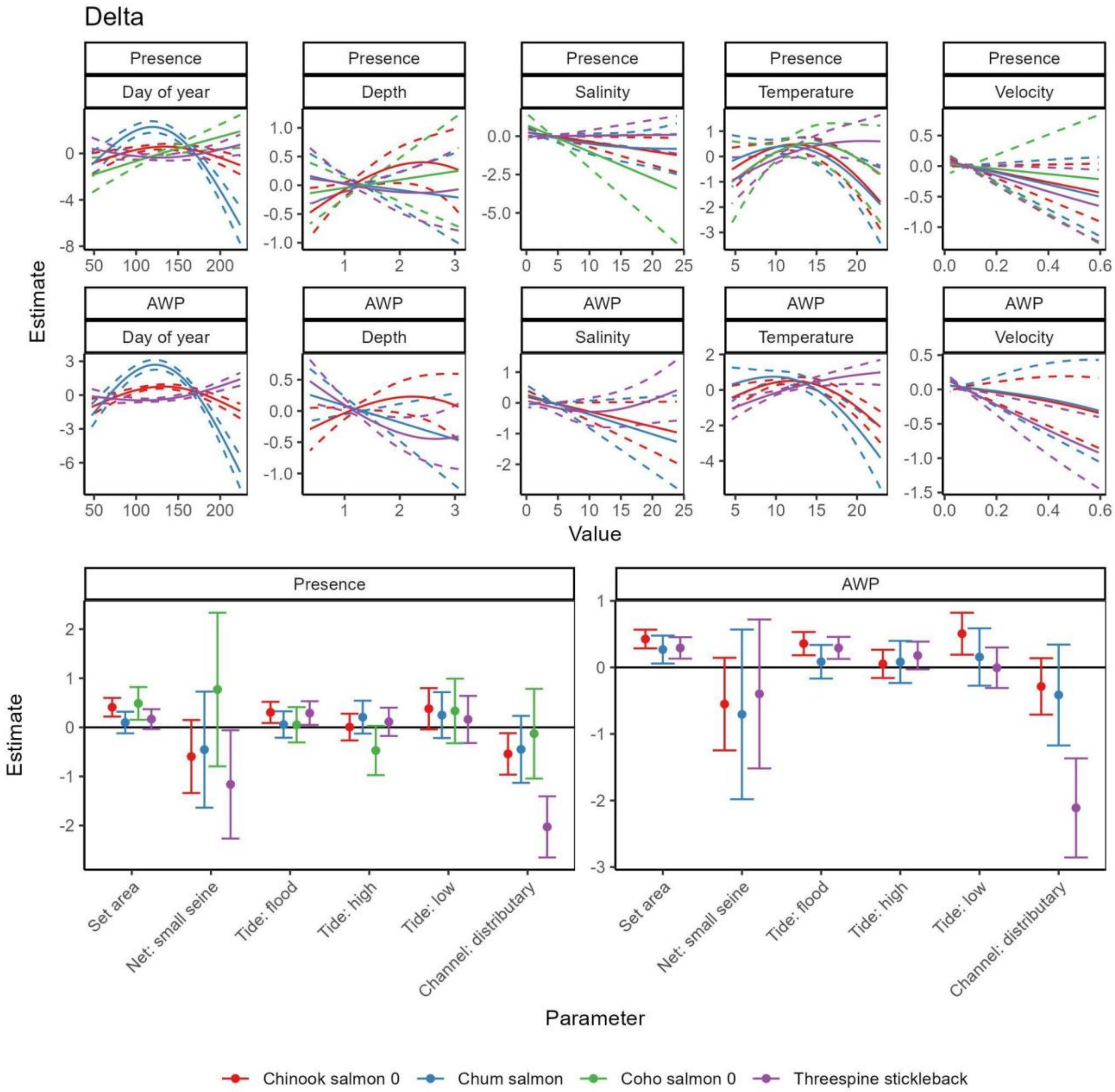

Figure 4.1.3. Parameter estimates of models that described fish abundance in delta waters. These species are anadromous, use estuaries facultatively, and inhabit the water column (Table 4.1.1). In these and similar figures, dashed lines in the top panel and whiskers in the bottom panel show 95% confidence intervals; the bottom panel omits parameter estimates of global intercepts to better visualize the other parameter estimates; the environmental variable units are depth (m), salinity (practical salinity units per thousand), temperature (°C), velocity (m/s); effects of the small seine are contrasted with the large seine, the effect of tides are contrasted with ebb tides, and the effect of distributary channels are contrasted with blind channels; AWP stands for abundance when present; 0 indicates age-0.

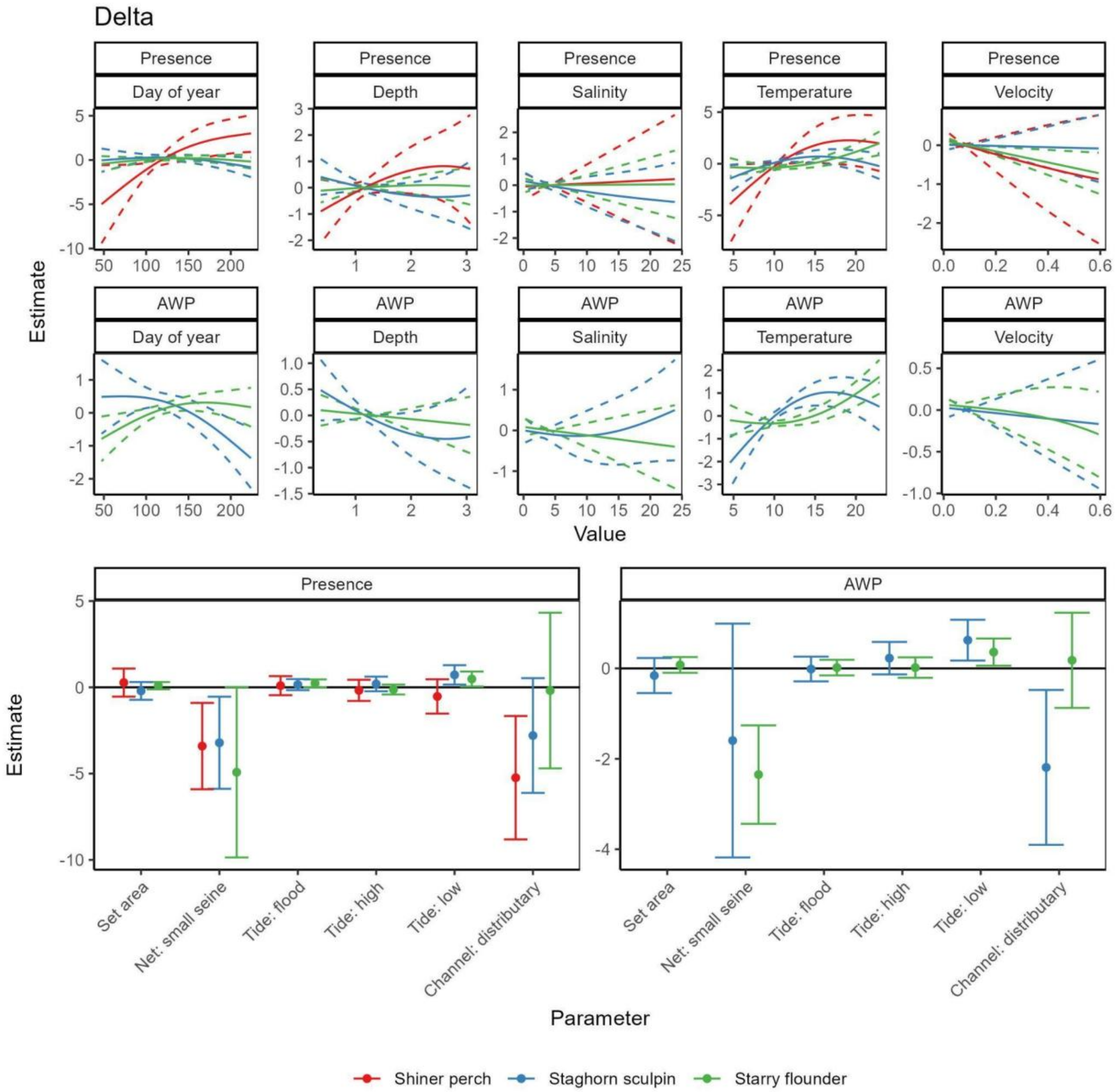

Figure 4.1.4. Parameter estimates of models that described fish abundance in delta waters. These species are marine estuarine-opportunists and use estuaries facultatively (Table 4.1.1). See Fig. 4.1.3 for additional details.

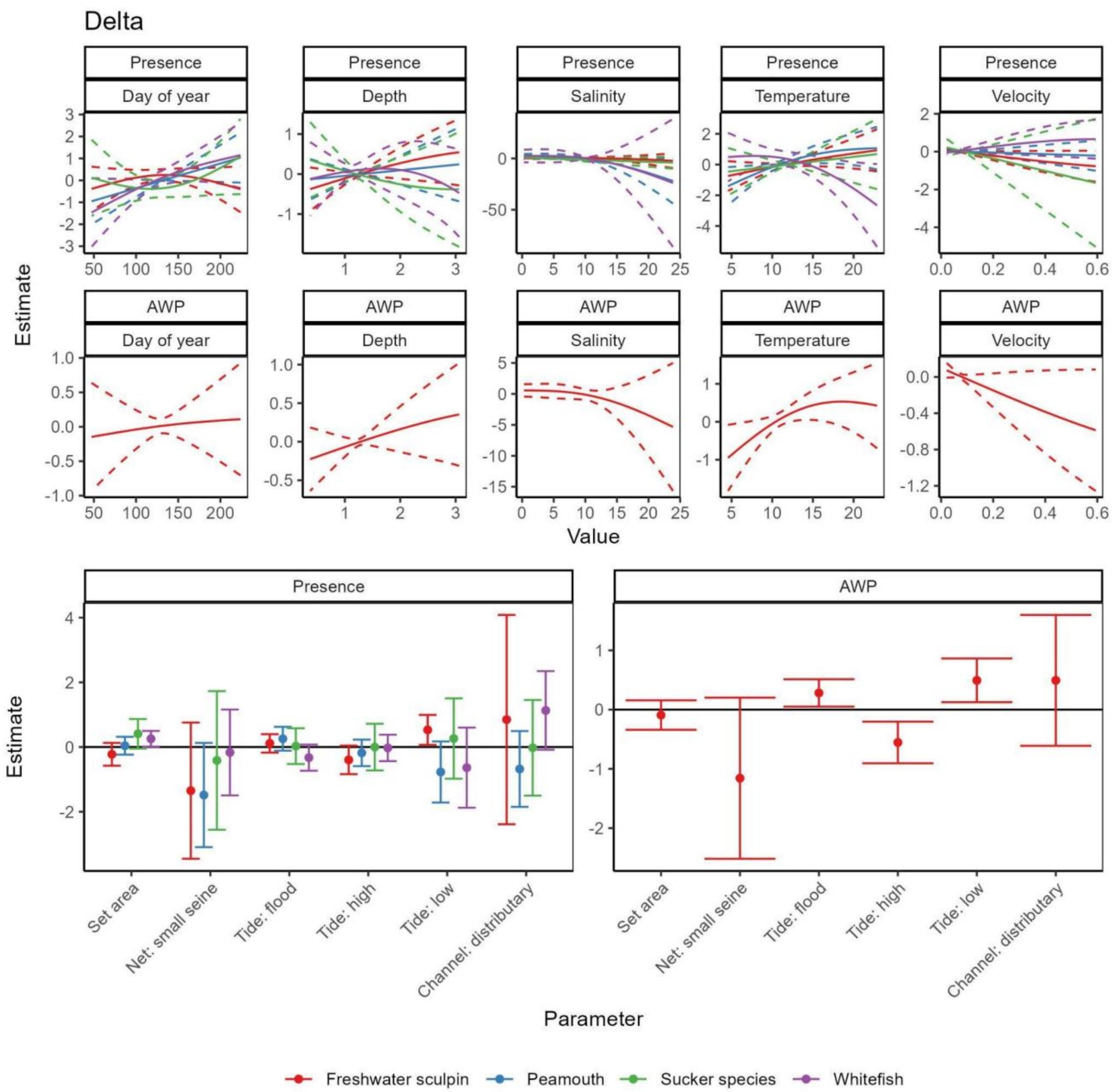

Figure 4.1.5. Parameter estimates of models that described fish abundance in delta waters. These species include semi-catadromous, freshwater straggler, and marine estuarine-opportunists, and inhabit estuaries facultatively (Table 4.1.1). See Fig. 4.1.3 for additional details.

The effect of depth varied among species (Figs. 4.1.3 – 4.1.5). Chum salmon, threespine stickleback, staghorn sculpin, and sucker species were more abundant in shallower waters, while Chinook salmon, coho salmon, and freshwater sculpin were more abundant in deeper waters. Abundances of other species such as peamouth and starry flounder did not vary as much with depth.

Various other factors influenced fish abundance (Figs. 4.1.3 – 4.1.5). Abundances were generally greater when the large – rather than small – beach seine was deployed. Anadromous species were more abundant with increasing set area, in blind – rather than distributary – channels, and at low tide (Fig. 4.1.3). Species' abundances rose as spring shifted into summer, with variable timing among species. Those that used habitats earlier in the year often became less abundant by the end of the annual monitoring period.

Nearshore

The abundances of the 20 species observed in at least 3% of samples in nearshore waters varied with environmental conditions (Figs. 4.1.6 - 4.1.10). As in delta waters, fishes were generally more abundant in lower velocity waters, although this relationship was weaker in anadromous species (Fig. 4.1.6). Other relationships between abundance and the environment varied among species.

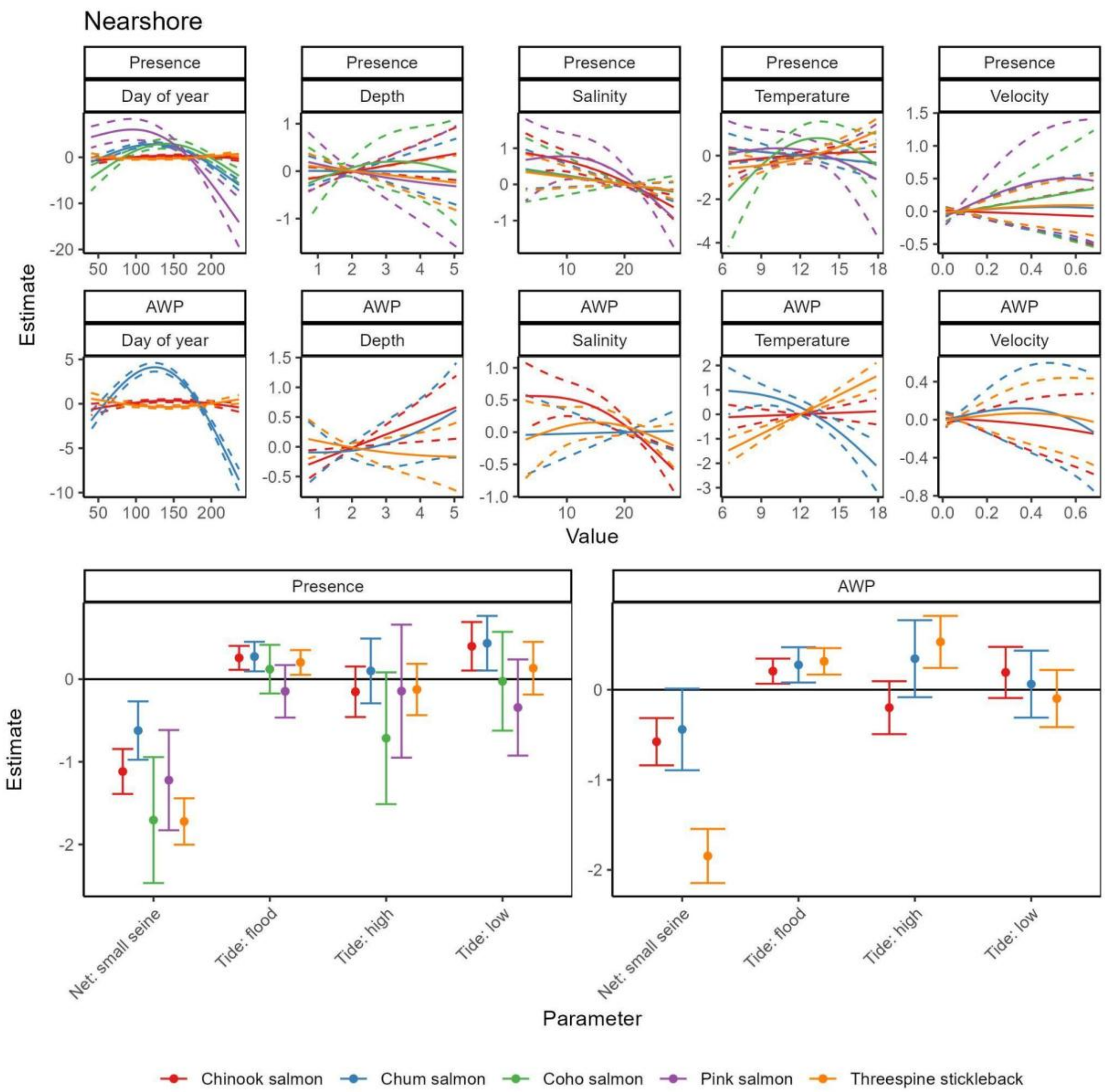

Figure 4.1.6. Parameter estimates of models that described fish abundance in nearshore waters. These species are anadromous, use estuaries facultatively, and inhabit the water column (Table 4.1.1). See Fig. 4.1.3 for additional details.

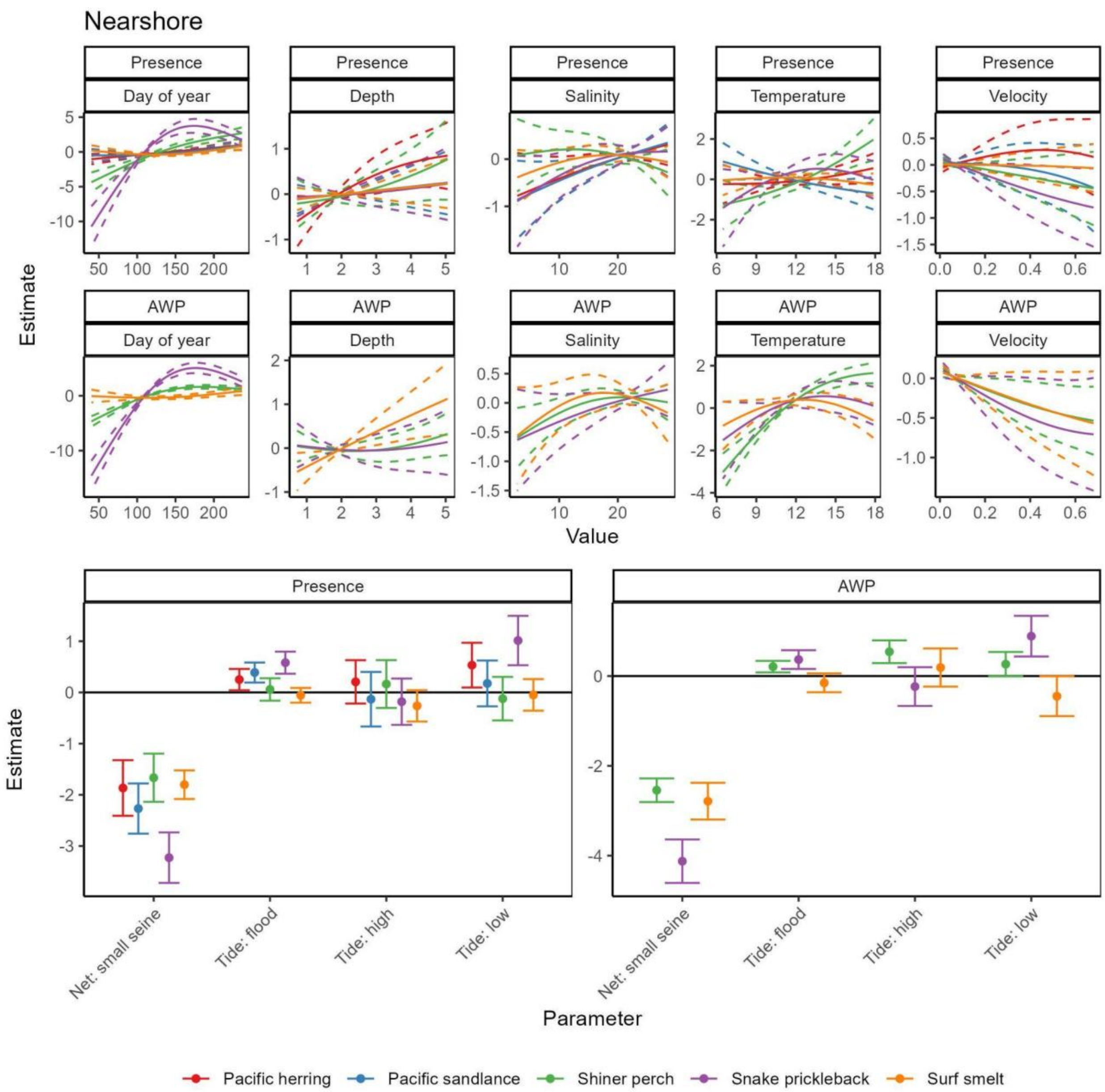

Figure 4.1.7. Parameter estimates of models that described fish abundance in nearshore waters. These species include marine stragglers and marine estuarine-opposists and inhabit the water column (Table 4.1.1). See Fig. 4.1.3 for additional details.

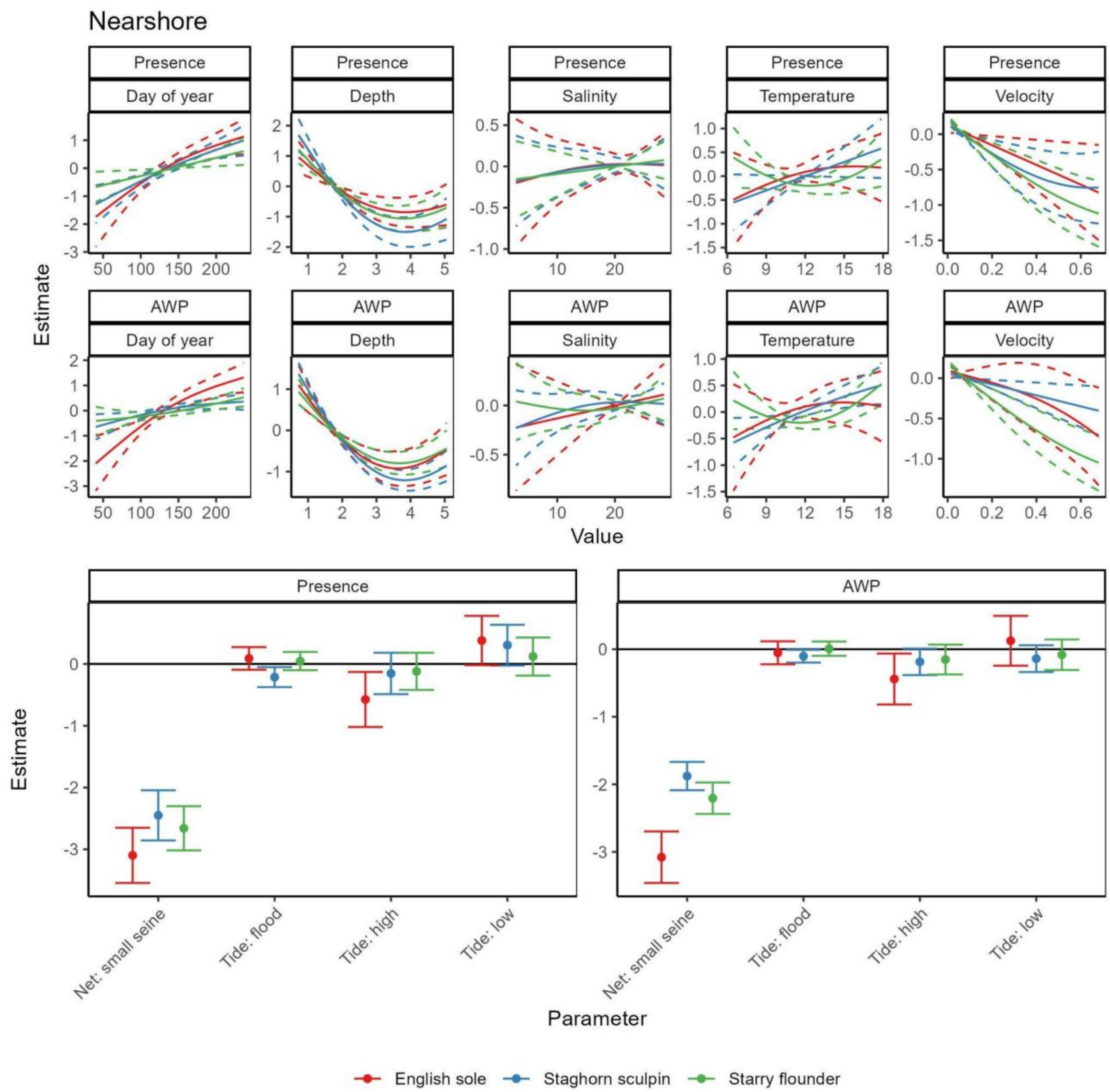

Figure 4.1.8. Parameter estimates of models that described fish abundance in nearshore waters. These species are marine estuarine-opposists (Table 4.1.1). See Fig. 4.1.3 for additional details.

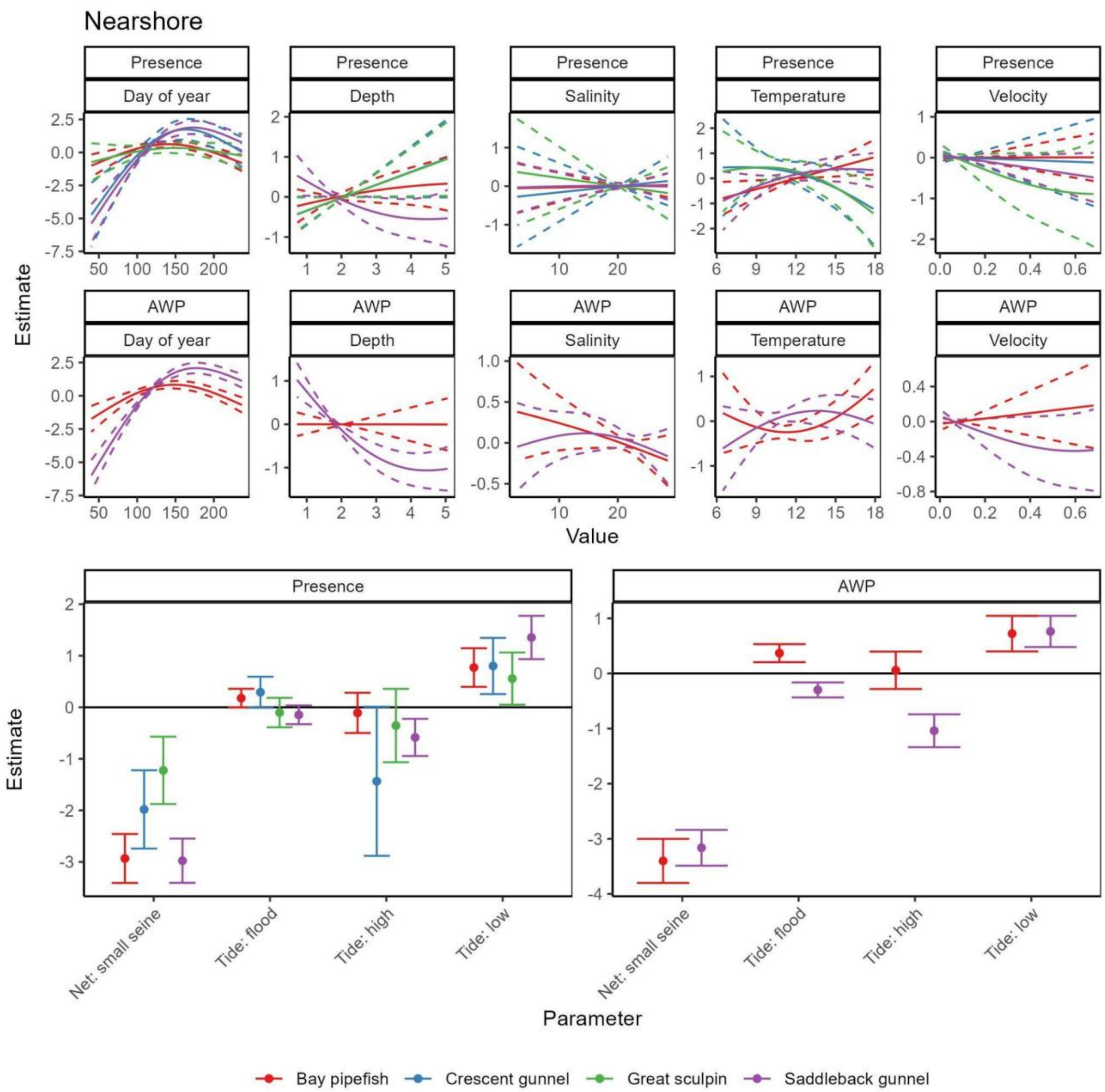

Figure 4.1.9. Parameter estimates of models that described fish abundance in nearshore waters. These species are marine stragglers and use estuaries obligately (Table 4.1.1). See Fig. 4.1.3 for additional details.

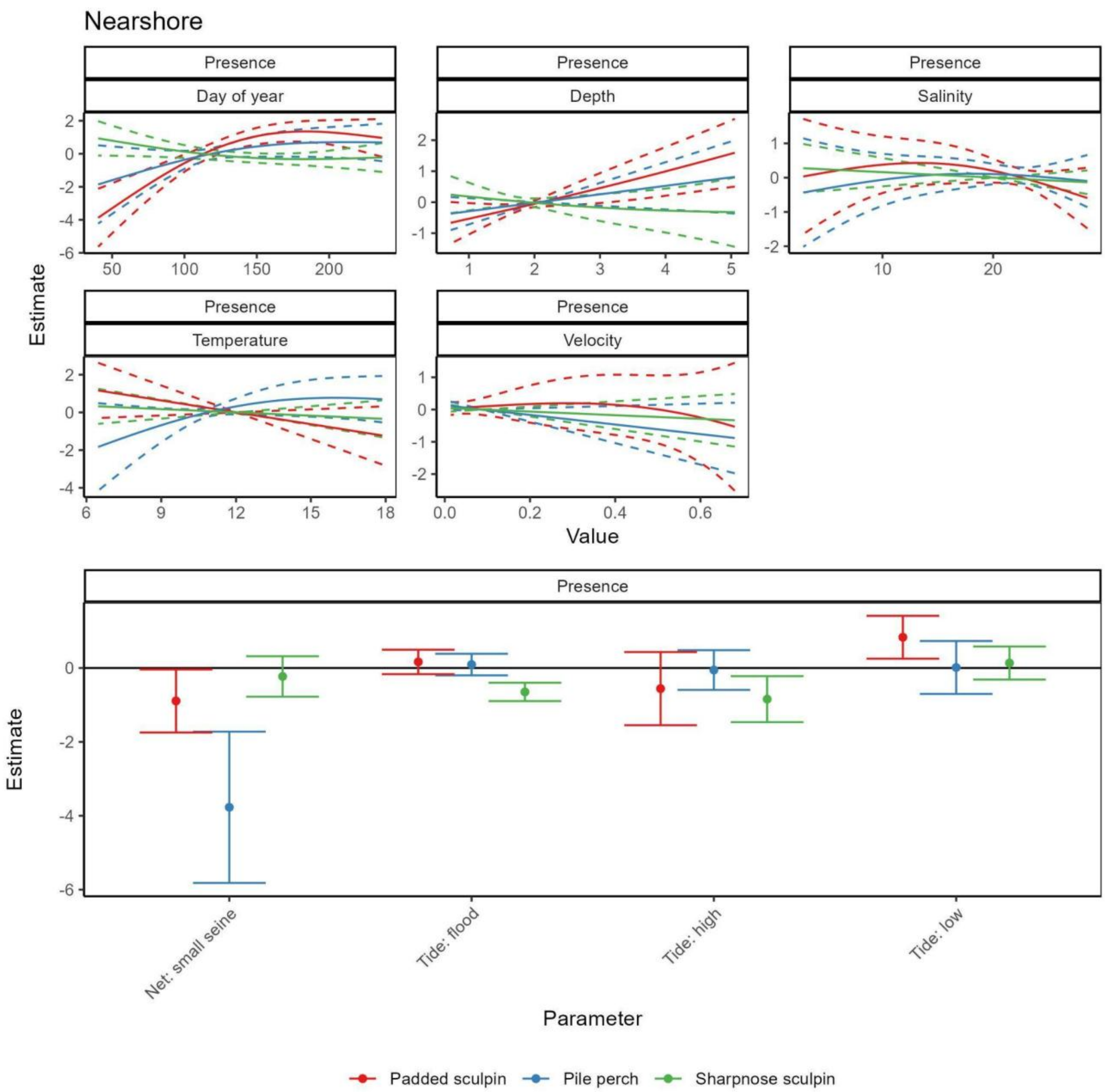

Figure 4.1.10. Parameter estimates of models that described fish abundance in nearshore waters. These species are marine stragglers and use estuaries obligately (Table 4.1.1). See Fig. 4.1.3 for additional details.

Anadromous species' relationships with the local environment were often similar to those observed in delta waters (Figs. 4.1.3, 4.1.6). With some exceptions, their abundances increased with increasing depth, decreased with increasing salinity, and were maximized at intermediate temperatures. In contrast to responses in delta waters, the effects of salinity and velocity on anadromous species were more nonlinear and abundances were maximized at slightly higher temperatures.

Other species expressed different relationships with the environment (Figs. 4.1.7 – 4.1.10). Abundances of marine stragglers and marine-estuarine-opportunists often increased with salinity or were maximized at intermediate salinity levels. Abundances of species that inhabit the water column such as Pacific herring and surf smelt increased with depth (Fig. 4.1.7) while the opposite was often true for species such as English sole, staghorn sculpin, and starry flounder that rest on the bottom of the water column (Figs. 4.1.8, 4.1.9).

Examining fish abundances related to other factors, salmon generally inhabited nearshore waters earlier in the year than the other species (Fig. 4.1.6), and fewer fish of all species were captured in the large – rather than small – beach seine. In addition, abundances of species that inhabit the water column were not strongly or coherently associated with tidal cycles (Figs. 4.1.6, 4.1.7). In contrast, species that inhabit the bottom of the water column were generally more abundant at low tide and less abundant at high tide (Figs. 4.1.6, 4.1.7).

Discussion

We quantified multiple fishes' relationships with local environmental conditions in the Skagit River estuary's delta and nearshore waters. We sampled fish with beach seines and analyzed the data with GAMMs to quantify fishes' relationships with environmental conditions while accounting for other factors of the sampling regime that influence abundance. Abundances of virtually all species were related to depth, salinity, temperature, and velocity. Relationships between abundances and the environment were often nonlinear, which included both dome and asymptotically-shaped curves. Some relationships appeared to reflect species' habitat preferences. This information fills a basic knowledge gap that seeks to understand species' natural histories so that we may better anticipate their dynamics amidst natural and human-caused environmental changes (Tewksbury et al. 2014, Able 2016).

Many of our findings were intuitive. For example, whitefish were uniquely more abundant in higher velocity waters, which was consistent with their physiology that is adapted to swim in swiftly moving streams (Taylor et al. 2012). Also, salmon were more abundant in fresher waters, which may reflect a residual preference for fresh water similar to river habitats, and the sequential arrival timing of pink, chum, coho, and Chinook salmon was also consistent with these species' typical migration phenologies (Quinn 2018). Additionally, species such as shiner perch and English sole with natural ranges as far south as Baja California were more abundant in warmer waters, and species such as Pacific herring and surf smelt that actively swim in the water column and away from extreme shallows were more abundant in deeper waters (Love 2011, Munsch et al. 2016).

Our findings were consistent with previous research that showed variation in the estuarine environment influences fish abundances and that effects differ among groups of species. For example, droughts in California (USA), Portugal, and South Australia changed salinity and temperature environments across estuaries, and effects on abundance and stress levels varied among species (Martinho et al. 2007, Ferguson et al. 2013, Jeffries et al. 2016, Colombano et al. 2022). Collectively, the literature and our study suggest a "winners and losers" scenario whereby shifts in water quality will alternatively favorably or unfavorably change conditions for species depending on their habitat preferences and tolerances (e.g., freshwater, estuarine, marine, opportunist, generalist, specialist). An example of this in our study was that salmon were uniquely less abundant in higher salinity waters. Also uniquely, salmon enter the estuary from freshwater habitats and perhaps they deliberately avoid more saline waters that are characteristic of marine waters until they have sufficiently developed (Simenstad et al. 1982, Sawyer et al. 2023, Quinn 2018). Indeed, similar to our study, anadromous fishes in the San Francisco Bay estuary were less abundant during drought that reduced salinity across its seascape whereas other groups of species responded unevenly or positively to drought (Colombano et al. 2022). Given the dynamic nature of estuaries and their location at the intersection of variable watershed and oceanic regimes, we may expect a general pattern that 1) a diversity of species are capable of inhabiting various areas depending on prevalent conditions, 2) species adapted to different specific attributes or ranges of conditions will respond differentially to variation in the physical environment, and 3) we must examine fish-environment relationships across a breadth of species to more fully anticipate how the fish assemblage will respond to disturbances that shape the physical environment.

Variation in some of these relationships among species may reflect fishes' natural histories. For example, bottom-oriented fishes uniquely tended to be most abundant at low tide and in shallower waters, whereas fish that used the water column were not strongly related to tide and were more abundant in deeper waters. Perhaps bottom-oriented fish can maneuver more effectively at low water levels than fish with morphologies adapted to the water column and can thus exploit shallow conditions for favorable scenarios. These could include limited escape responses of prey, limited attack capabilities of predators, or more abundant prey amidst fewer competitors (sensu Boswell et al. 2019, Colombano et al. 2020, 2021). Indeed, the tidal cycle overlaid on heterogenous (e.g., salinity, shoreform, temperature) estuarine seascapes is a fundamental process in estuaries and may generally support habitat partitioning across time in species with different life histories (Colombano et al. 2020).

Our findings may inform management. Specifically, Chinook salmon in Puget Sound are protected by the U.S. Endangered Species Act (NOAA 2007). They were most abundant in waters of greater depth, lower salinity, intermediate temperature, and lower velocity; therefore, decisions that alter the freshwater flow regime may consider how these local environmental attributes will change and ultimately alter access to Chinook salmon estuarine nursery habitats. Notably, considerable resources have been invested to restore habitats for Chinook salmon in the Skagit River estuary (Chamberlin et al. 2025) and it is critical to consider that human activities (e.g., fishing, water regulation, water extraction) will influence the ability of salmon to access restored habitats and thus assimilate the benefits of these investments (Simenstad and Cordell 2000, Munsch et al. 2020). In Chapter 4, we used a hydrodynamic model to rigorously quantify how changes to the flow regime can alter local environmental conditions, and we used the fish-environment models presented here to project the responses of fish across a range of water availability and extraction scenarios.

An important nuance to this study is that it was scaled to populations and annual rearing windows. Our models essentially quantified the "average" states of relationships between fish and local environments at these scales. However, it is important to understand that additional, important processes unfold at finer scales (e.g., individual, diel) such as ontogenetic habitat shifts (Nagelkerken et al. 2015 Sheaves et al. 2015, Munsch et al. 2016) to deeper or more saline waters generally and in salmon, respectively, and movements across heterogeneous environments to make the most of unevenly distributed conditions (Armstrong et al. 2013). Important processes also unfold at larger scales. For example, the differential responses to physical conditions among species probably generated asynchronous changes in abundance across time among groups of species in response to variation in the environment, thus stabilizing total abundances and maximizing overall use of the estuary at the fish assemblage scale (Colombano et al. 2022). A comprehensive assessment of estuarine habitats and their sensitivity to human influences should consider relationships between fishes and the environment across such additional scales.

Another nuance is that our study was scoped to direct relationships between fish and the local environment, but other indirect effects of the environment on fish are important. For example, vegetation including species that form habitats, produce prey, and recruit symbiotic species, are also sensitive to changes in salinity and water levels (e.g., Silvestri et al. 2005). Specifically in the Skagit River estuary, invasive cattail (*Typha angustifolia and T. x glauca*) occupy slightly lower elevation and saltier waters than native Pacific Willow (*Salix* Spp.) and may thus displace willow if freshwater inputs are chronically low, but only willow shades waters and recruits beavers (*Castor canadensis*) that generate wetlands and juvenile salmon habitats (LeMoine 2021).

Estuaries are dynamic environments that provide vital habitats to many species but are often modified by human stressors (Beck et al. 2001, Lotze et al. 2006, Sheaves et al. 2014, Greene et al. 2015). Many studies have shown that variation in estuaries' physical environments – including water quality attributes that human activities modify – provokes changes to the fish assemblage (e.g., Martinho et al. 2007, Ferguson et al. 2013, Jeffries et al. 2016, Colombano et al. 2022). Within this very report, we have observed hydrodynamic changes related to water withdrawals in low-flow years that can have cascading effects both directly on fish species (Chapter 4) but also on habitat structure (Chapter 3). These changes, which often emerge via differential responses among species, can be intuitive when information on species' natural histories (e.g., habitat preferences) are available (Tewksbury et al. 2014, Able 2016). Our study bolsters natural history knowledge via quantitative descriptions of relationships between fish abundance and local environmental variation. This information may guide decisions in this and estuary or others that seek to leverage ecosystem-based perspectives to better anticipate the potential for natural and human disturbances to influence natural resource dynamics (Link 2010).

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Appendix 4.2. Prediction errors for water scenarios

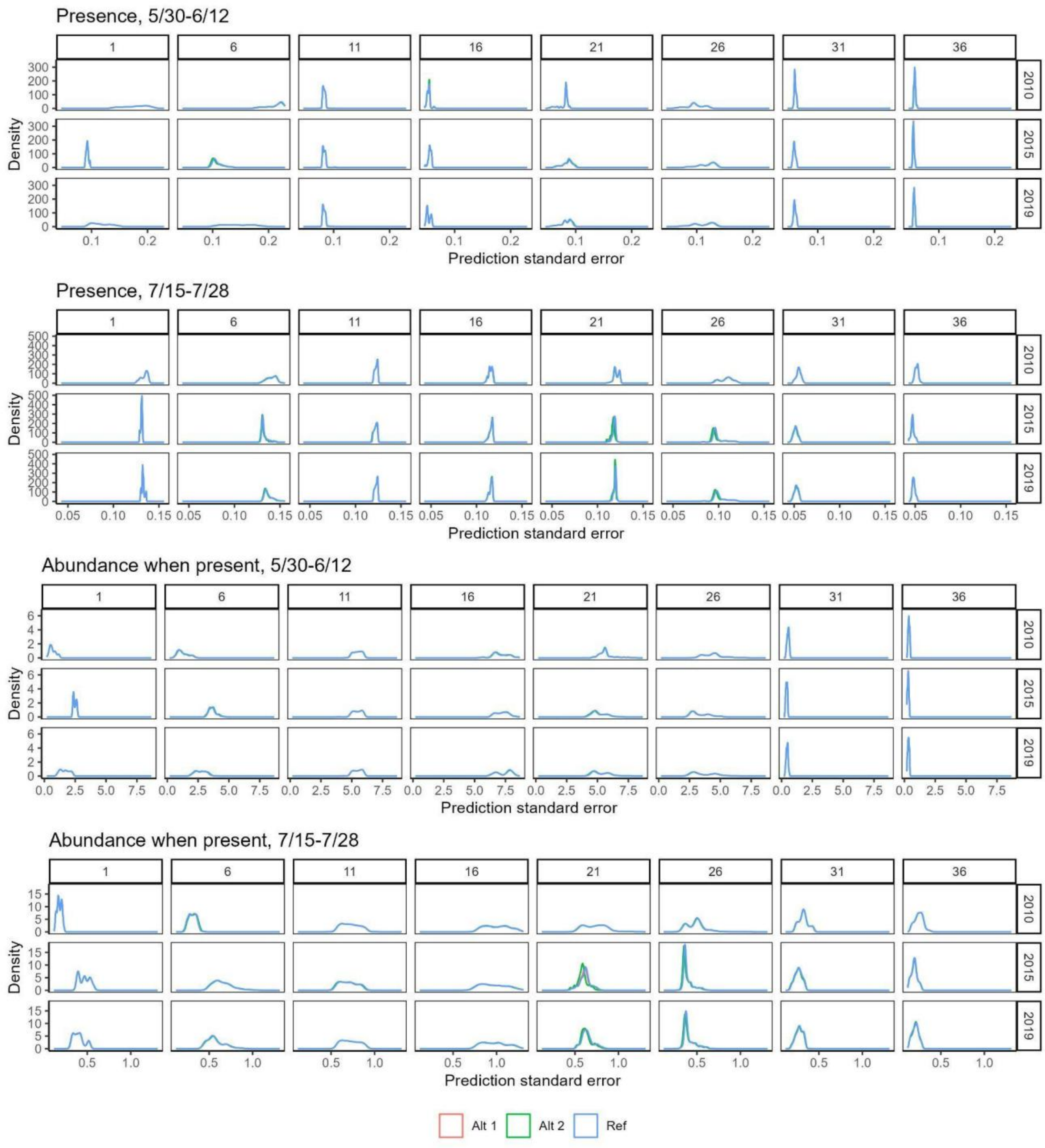

Figure 4.2.1. Density (smoothed histogram) plots comparing sites' standard errors in Chinook salmon abundance predictions among water use scenarios. Numbers indicate sites and every fifth site is shown to improve plot visibility.

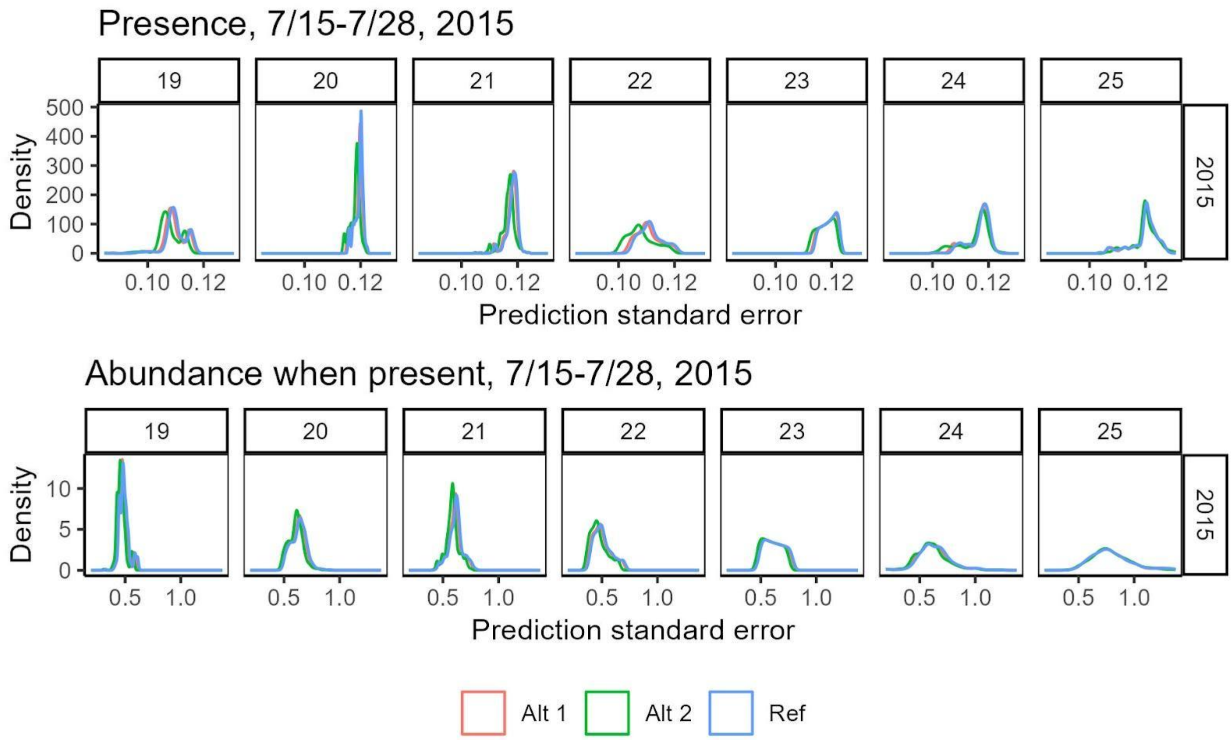

Figure 4.2.2. Density (smoothed histogram) plots comparing south fork delta sites' standard errors in Chinook salmon abundance predictions among water use scenarios. Numbers indicate sites.

Appendix 4.3 Research Team response to Washington State Academy of Sciences Review

The Research Team thanks WSAS for providing helpful reviews by three anonymous reviewers. We appreciated the constructive nature of these comments and have made an effort to address every one. However, in some cases there was insufficient time during the revision period to address every comment to our liking. Nevertheless, these comments will shape our final products for any additional publication effort.

Reviewer	Comment	Final response
Summary	First, the reviewers were impressed by the overall quality of the work, the detail that was included in the report, and the quality of the writing. The report will be a valuable contribution to our scientific knowledge of the Skagit.	Thank you! We were unable to address every single comments in the short time available for revisions, but the review is very helpful, and we will be addressing reviewer comments in any publications that come out of this report.
	Second, it would be helpful if the report were better integrated, in two senses. There are many detailed comments from the reviewers on the need for better integration of the various chapters of this report, so that the reader is given a better sense of the report as a whole. Moreover it would be good if the report could comment on its contribution to the overall objective of the three projects, as efforts to improve our scientific knowledge of the Skagit and to support decisions regarding its management.	While integration can be challenging when multiple authors are completing separate chapters, we have made an effort to address this comment by including common sections (uncertainty, implications for water and salmon management) in each chapter and summarized these sections in Chapter 1. We also made an effort to build concepts from Chapter 1 to 4. However, we were unable to address the second point (contribution of this project in reference to the other two Task Force Phase II projects) because we have not had access to those findings. Perhaps this suggestion would be better addressed in summaries provided by WSU.
	Third, the reviewers found a general lack of concern for uncertainty, and felt that more attention to uncertainty was needed in the various measurements and observations, in their integration into comprehensive perspectives, and in the conclusions and predictions that emerged from the modeling efforts. It is important to know how measurement uncertainties propagate into the predictions that may be used in future decision making, such as those concerning the magnitude of landward advances of salinity.	We have added a section to address uncertainty in the hydrodynamic modeling, vegetation, and fish modeling chapters. We also added in a summary of these in Chapter 1
	Fourth, while the reviewers appreciated the value of the alternative scenarios that were examined in the report, they felt that it would have been useful to have included some discussion of more extreme scenarios, such as those that might reflect dramatic climate-change scenarios, sea-level rise, or tectonic events.	We have added discussion in each chapter to discuss how additional climate scenarios might further modify observed patterns.
Reviewer 1	**General**	
	Does the work reflect the best of current scientific methods? Each section does a nice job of using current scientific methods and approaches. The set-up of the hydrodynamic models was good in Chapter 1, and the use of different water years and hydrodynamic modeling was appropriate in Chapter 2. Chapter 3 was the least robust but has great potential to provide more detailed results regarding effects of altered freshwater input levels as they affect salinity in small tidal channels. Chapter 4 was a nice description of vegetation work and modeling. Chapter 5 offered a robust fish study. And Chapter 6	Since Chapter 3 was not actually integrated into the other studies, we removed it from the report. We too are intrigued by the analysis and will aim to publish this elsewhere. Given that the original Chapter 5 did not have any water scenarios included, we made that an appendix to the following Chapters. As a consequence, we shortened the number of numbered chapters to 4.

	integrated aspects of chapters 1 and 2 with chapter 5 in an effective way.	
	Does the work help to fill some of the gaps in our knowledge of the Skagit Estuary? Yes, this word definitely fills many of the gaps in our knowledge of the Skagit Estuary and of fish habitats and uses. This work points to the importance of salinity as a strong driver of biological patterns in vegetation and fish (particularly salmonids) in the Skagit Delta and nearshore habitats. This was out of the scope of the original Duke study and points to a key metric for monitoring and investigation. This study also points to the influence that changing the amount of freshwater available at the Delta, particularly in the summer, has on salinity levels and available fish habitat. This is a key finding from this work.	Thank you
	Do the conclusions follow from the evidence? Yes, the conclusions do follow from the evidence. However, the different sections of this report are not well integrated, and gleaning conclusions is challenging. See more comments on this in the next section.	While integration can be challenging when multiple authors are completing separate chapters, we have made an effort to address this comment by including common sections (uncertainty, implications for water and salmon management) in each chapter and summarized these sections in Chapter 1. We also made an effort to build concepts from Chapter 1 to 4.
	Each of the sections offers insightful results that are relevant to the broader topic of the effect of water withdrawal in the Skagit mainstem. However, for some sections, the implications for salmon or management are buried in their individual discussion sections and are not well integrated or obvious.	Each chapter now has an "Implications for salmon and water management" section, which we summarize in Chapter 1.
	How well do the 6 parts hold together? The 6 parts are generally stand alone chapters of the report. Chapter 6 offers some integration between the hydrodynamic modeling and the fish study. Oddly enough, in Chapter 5, the authors say they plan on doing hydrodynamic modeling in a later study, and I don't think they meant Chapter 6? Perhaps so. If the chapters could link to one another, and point out how they support one another within the text, that would make this report more integrated. Results do demonstrate consistent findings, particularly related to the role of salinity in defining the available habitat for plants and for fish, but the chapters do not try and make those connections for the reader. It is very hard to make a report written by different authors in each chapter to really be inter-related. However, the sections of this report are so well written it does not see that it would take a lot of additional effort to link results and to make sure that the focus of findings in the individual discussion sections ties chapters together.	We corrected the statement in Chapter 5 (now Appendix 4.1). Basically, subsequent chapters build on previous ones, so we have tried to built the linkages in that direction, and then summarize some of the common themes in Chapter 1.
	Comments on the Sections of the report:	

	Executive Summary: The executive summary picks out some of the key findings from the chapters, particularly as they relate to salmon broadly, and Chinook in particular. One question: in the short summary for Chapter 5, they say 30 years of fish data. I thought the chapter said it was for 8 years of fish data, from 2015-2023?	While the dataset is 30 years, we focused on post-avulsion data (2015-23). We revised this statement to indicated it was "long-term", not 30 years.
	Chapter 1: Overview of study, water use simulations, and key results: The first section of the report does a nice job of laying out the different simulations that were completed. The types of simulations were appropriate and demonstrated a thoughtful approach to the selection of models that would demonstrate a variety of potential model futures. The models did a nice job of representing the inputs and outputs into the Skagit delta.	Thank you
	Chapter 2: Simulating hydrodynamic changes from water withdrawals in the Skagit River: Did the "model grid refinement" include tide gates? They are not specifically listed. Tide gates could affect pathways of flow and inundation relevant for salmon habitat in the delta distributaries.	We fully agree that hydraulic structures such as tide gates influence flow pathways in the delta distributaries. Unfortunately, tide gates are not currently supported as a built-in capability in the SCHISM model, so we did not include them in this study. We believe this is an important feature that should be added to SCHISM to improve the representation of flow and salt exchange. Thank you for pointing out this limitation.
	How is it that "… analysis revealed that salinity – particularly in the surface layer – is the most sensitive parameter to reductions in freshwater input caused by water withdrawals. In contrast, the effects on water surface elevation and velocity are comparatively minor and generally negligible." Are the models not sensitive to the sometimes high amount of freshwater withdrawal? Or does the marine water essentially compensate for less freshwater inputs, thus not changing water levels? The figures that showed how alternative scenarios different from the current scenario were particularly helpful at showing differences in salinity values among scenarios.	The model is capable of accurately capturing the hydrodynamic responses caused by water withdrawals. However, the magnitude of the response varies across different variables. For water level and velocity, the changes are relatively minor, as the reduction in freshwater input (due to withdrawals) can be quickly offset by the influx of ocean water. However, since ocean water has a significantly higher salinity than freshwater, this results in much larger changes in salinity. Therefore, salinity—particularly in the surface layer, where freshwater tends to remain—is the most sensitive indicator of the effects of water withdrawals.
	Chapter 3. Extending hydrodynamics into small tidal channels and wetlands: This provocative section is highly relevant to the question of effects on salmon spawning and rearing habitat. An expansion of this section with an additional research project would be warranted. Additionally, the question posed by the authors related to whether salinity scales with the hydrodynamic model is an important one that could be explored in more detail along with a finer-scale velocity and discharge study.	Since Chapter 3 was not actually integrated into the other studies, we removed it from the report. We too are intrigued by the analysis and will aim to examine this in more detail in the futre.

	Chapter 4. Predicted vegetation responses to changes in tidal delta water levels and salinities: This very thoughtful section provides a nice assessment of potential changes to vegetation, mostly aligned with alterations in patterns of salinity. However, the focus of this larger project is the effect on fish habitat. In the discussion, the links among shrub cover (which as the largest modeled change under alternate flow scenarios), beaver, beaver dams, and juvenile Chinook rearing was made. I suggest this section could be improved by expanding the introductory section to clearly frame the important role vegetation plays in creating complex habitat for Chinook. This is relevant beyond beaver and includes the allochthonous inputs of insects that are associated with vegetation cover, and that vegetation cover can allow us to track salinity levels that indirectly allow us to track availability of Chinook habitats. The link to insects is mentioned in the Executive Summary, but not in the chapter.	We added a paragraph to the introduction summarizing the centrality of vegetation in geomorphic processess and foodwebs.
	Chapter 5. Effects of local environmental variation on occurrence and abundance of estuarine fishes: It is unclear how often sampling occurred. The authors mention sampling from Feb-August over an 8 year period, and describe differences between delta and nearshore areas, but it's not clear if sampling was done once a year in this window of Feb-August, or more frequently. There was no reference in the analysis to time. The February to August timeline includes periods of higher freshwater discharge (winter and spring), and lower discharge (summer). It would be helpful for the authors to specifically discuss fish presence and associations with salinity and discharge based on these seasons. This would help set up the link between hydrodynamic models and fish in Chapter 6. The authors point out their observation of the link between salmon and low salinity which is a critical observation for the relevance of the finding that salinity is likely to change the most under scenarios of freshwater use in the Skagit River system. On page 123, the authors say they are going to develop a hydrodynamic model to quantify how changes in flow regime alter local environmental conditions relevant to fish. Isn't that Chapter 6? Perhaps the authors could refer to Chapter 6 here?	Sampling occurred biweekly starting in February. We did not break the presence/absence and density when present models into separate time periods, instead using the entire range and incorporating day of year explicitly as a variable. Using the entire time period allowed us to pull out any time frame of interest. We focused on late spring and summer because of the combination of fish presence and increasing water scarcity. Fish are abundant in the tidal delta in later winter and early spring, but water demand is minimal then.
	Could results from Chapter 4 be integrated into the discussion of vegetation on page 123 (paragraph 2)?	We added a sentence addressing this point in this chapter (now appendix 4.1).
	Chapter 6. Effects of water withdrawals for fishes in the Skagit River delta: The statement "Predicted Chinook salmon presence increased when flows were higher (Fig. 6.5), particularly during the summer period", is confusing. Isn't summer the low-flow period? Or is this meant to be that within a season, Chinook salmon presence was higher when freshwater flows were higher?	We have corrected the writing error and replace "summer" with "late spring"

	This chapter is critical in that it integrates other chapters in a meaningful way. Since this chapter seems to function in this way, would it be possible to also refer to results from other chapters here? The preliminary expansion of hydrodynamic modeling from Chapter 3 wasn't part of this, but it could be referred to here as a topic in the discussion as another aspect of future work. Likewise, more connection between the parallel findings of changes in vegetation composition (Chapter 4) and salmon habitat (this chapter) as they relate to salinity could be made in the discussion.	We have added discussion in Chapter 4 (previously Chapter 6) that builds on results from previous chapters.
Reviewer 2	**General**	
	Does the work reflect the best of current scientific methods? The hydrodynamic work is highly competent in addressing the questions it addresses. However, the authors describe the report as preliminary analysis. They correctly note that several neglected factors might be important (item 1), to which I'd add that simulation of flow within marshes could be improved (item 2). Therefore, the hydrodynamic effort seems within the range typical of expert scientists studying estuarine flows, but improved predictions would be possible.	We fully agree with your suggestions. Accurately simulating flow and scalar transport within marshes is indeed highly challenging and requires ongoing improvement. There are noticeable discrepancies between the model-predicted salinity and observed values at several monitoring sites, which will need to be addressed in future work.
	Does the work help to fill some of the gaps in our knowledge of the Skagit Estuary?. Although I am not expert in such coupled models, I believe the coupling of biology with hydrodynamics is a step forward for the Skagit. I believe the hydrodynamic model is the best so far for this purpose in the Skagit, and the examination of water-use scenarios is interesting.	Thank you!
	Do the conclusions follow from the evidence? There is a transition from offshore salty water to onshore freshwater. For me, a key prediction of the report is this: other things being equal, this salinity transition would move onshore if more water were withdrawn upstream, with implications for plants and fish. This seems very reasonable. However, one difficulty is that other things are unlikely to remain equal (item 1 below). Additionally, I am unsure how much skill the models have in predicting how far onshore the salinity transition would move, and how big the biological implications would be (item 3). In the limited time before the final report is submitted, I suggest prioritizing clarification of this uncertainty.	Thank you for the insightful comments. We are confident in the model's ability to predict the temporal and spatial distributions of salinity in Skagit Bay. However, we acknowledge that it is significantly more challenging to accurately capture salinity patterns in intertidal zones, which are dominated by small-scale topobathymetric features and vegetation.It is true that salinity would move onshore and intrude further inland if more river flow were withdrawn upstream. In the study report, we presented the relative changes in salinity distribution compared to the baseline condition for each water withdrawal scenario, with a general focus on the overall spatial patterns. Although these changes are most evident in the shallow intertidal zones, we believe the model predictions are reasonable and reflect the best possible representation given the model's resolution and capabilities.Due to time constraints, we were unable to further quantify the changes in the saltwater intrusion limit in this report. However, we plan to address this in greater detail in the forthcoming manuscript.
	More detailed comments	

	1. Concerning scope: Water use is most important in late summer, when flows decline as snowmelt fades. I suggest the report should state for context whether these flows are expected to change with warming climate and reduced snowpack. If ballpark estimates of likely changes in late-summer low flows have been published for the Skagit or similar systems, I suggest they should be stated for context. This might only require a few sentences and references in chapter 1. I also wonder about local relative sea level rise, and whether saltmarsh sedimentation could keep pace. In know these topics have been studied, and a few sentences and references summarizing key facts in the introduction would be very useful. The report clearly excludes these questions from its scope, which might have been a reasonable choice, depending on project budget details that I know nothing about. But I do wonder if neglected effects might be large. It is helpful that the report compared the scenario 2 flow reduction with baseline flow (discharge reduction of up to 20% - this important fact definitely should be retained, and I think stated in chapter 1, where I found it hard to figure out how large a relative discharge reduction was being considered among the alternatives). But it would also be useful to understand the size of this reduction relative to other, omitted effects.	We have included statements related to climate impacts on hydrograph, temperature, and sea level.
	2. Friction dramatically slows flows through marsh vegetation, but this is often neglected in estuarine circulation models, and the report does not mention simulation of this effect. In Figure 2.A5, I think some of the biggest predicted withdrawal effects on salinity might occur in marshy areas[1]. I suggest the report should clarify whether vegetation drag was simulated, and if not, note this as a possible source of error.	Thank you for the insights. We have added a description of the bottom roughness values in the Methods section. Spatially varying bottom roughness height values (Z_0) were used in the model. Specifically, a default value of 0.001 m was found to perform well for most of the domain based on our prior experience in this region. For areas covered by marshes, a higher value of 0.05 m was applied to represent the increased drag caused by marsh vegetation. In the SCHISM model, there is an additional vegetation module that can explicitly simulate the form drag caused by plants in the water column. However, using this module requires more field data and considerable effort to configure and calibrate. Moreover, the module assumes constant plant density and height over time, which does not reflect natural variability. Therefore, we chose a more practical and simplified approach to represent the increased friction and drag associated with marsh vegetation.

	3. In the limited time remaining, I suggest clarifying the author's confidence in model predictions. Loosely, I wonder if language like high, medium, and low confidence, as used in IPCC reports, might provide be a useful model for communicating uncertainty. For the hydrodynamic model, correlations between predictions and raw time series measurements are not high (Table 2.1). AUC for vegetation looks good. For the fish models, I did not see model skill in predicting raw measurements at all[2] (this should be addressed, e.g. reporting bias, RMSE, R, as is done in Table 2.1, or Brier Skill Score, or similar). However, while these statistics should be reported, they may provide an unduly pessimistic view; just as it may be possible to predict averaged climate years in advance without predicting daily weather fluctuations, it may be possible to predict onshore movement of averaged salinity in response to water withdrawals without predicting every short-term variation. But can averaged predictions be tested? If usefully located measurements were available during already simulated intervals in years other than 2019, it would be very interesting to see whether the model has skill predicting the changes in 2-week-averaged salinities between years. This might more directly test a key prediction (how far salinity moves onshore when discharge declines). Regardless, it would be very useful to discuss more prominently the limits of model skill with regard to key predictions. For example, I think the model often predicts that Alternative-2 moves the salinity transition onshore 100m-1km, with considerable variability depending on local channels, but it would be valuable to have guidance on what confidence should be placed such predictions.	We provide a discussion of uncertainty in each chapter and we address our confidence in model predictions therein. We also followed the suggestion to include an IPCC-style level of confidence for the of the main findings reported from each chapter. Howver, we were unable to address in our limited revision time the suggestion of utilizing data to validate model predictions in Chapters 3 and 4 (this was done in Chapter 2).
	4. I didn't understand how, or whether, the channel flow estimates of Chapter 3 were used for fish or vegetation modeling (channel width doesn't really change with Alternatives 1 and 2). This should be clarified.	Since Chapter 3 was not actually integrated into the other studies, we removed it from the report.
	5. A very minor point: Many plots showed surface layer salinity. I wonder if bottom layer salinity is more related to sediment salinity, which I think was used to estimate salinity effects on vegetation.	We focused on surface salinity because surface waters are likely to most impact vegetation (that layer spills over into marshes on the flood), and salmon (very surface oriented).
	6. I found the modeling of vegetation very interesting, since this would likely be a significant habitat change.	Thank you
	There are numerous cases where refinement of presentation and correction of typos would be valuable. I didn't write most down, but e.g. Fig.2.28: caption "exceeded 10 ppt" contradicts y axis label.	We have made an effort to fix minor spelling or presentation errors.
	Vegetation sampling map should be shown before conclusions drawn from sampling.	A veg sampling map was added (Fig. 3.2)
	p.55: "the proportion of time that salinity and depth"	We have fixed this grammatical error.
	p.79: "four our more "	This chapter was removed (see above).
	Figures 5.3-5.10: all axes lack units.	We have added units in the first figure and made reference to this one.

Reviewer 3	**Executive Summary**	
	The executive summary describes a study that is cohesive and makes logical sense. Each chapter appears to build on the knowledge summarized by the previous one(s) in a way that is intuitive for the reader. I noticed some minor grammatical and editorial errors; e.g., concepts (like the "Duke Study") being introduced with little to no context, confusing description of model scenarios, misuse of en dash where em dash is needed. Due to the length of this report, I am choosing to focus my comments on the "meat" of its content, but the authors and report recipients should be aware that these editorial errors exist.	We have made an effort to correct grammatical, organizational, and other editorial issues.
	Chapter 1	
	The first chapter provides a broad overview of the study and its purpose, a description of the water use simulations, and some key results. Overall, I found this section to be robust; however, I have some concerns regarding chapter content. Namely, I would like more of an explanation as to why the water use and streamflow models are being described in Chapter 1, as opposed to Chapter 2 (the hydrologic model). I am also concerned that it's redundant to describe the results in this chapter when they are already described in their respective chapters and the executive summary. It seems like this chapter could be streamlined by focusing on:	We included many of the details of the simulations as a separate Chapter 1 because they are referred to in not only Chapter 2 but also in Chapters 4 and 6 (now Chapters 3 and 4, respectively). Given that the other reviewers call for more integration, it made sense to us to provide results in multiple places. This may seem a bit redundant at times, but we hope there's value (for different readers) in having a very brief, high level overview for the Executive summary, more depth summarizing results in Chapter 1, and all the details in the following chapters. While we did not remove the results and other summary at the end, we did address the reviewer's points below.
	1) Background to the system and context for the study design and study purpose.	We have provided more backgroun on the study system and laid out the context for the study.
	2) Providing more context for the conceptual model in Fig. 1.1. How are these components derived?	We have added in more details regarding the conceptual model.
	3) Historical discharge, water source, and water usage patterns and trends, including potentially some more context for what the system was like prior to human development.	We have added additional context and reordered portions of the text to address this point.
	4) Explanation of water use scenarios (I think what the authors have now is fairly robust).	Thank you
	Specific comments	
	1) What is Phabsim? (page 5)	It is the name of a model. We added that term, and citation
	2) Swap the order of your study questions on page 6. Veg should come before fish to coincide with the chapter order.	Done
	3) I found quite a few editorial errors in this chapter. An example: "analysis of impacts to fishes WAS limited to…" I would suggest the authors comb the report for minor errors.	We have made an effort to correct grammatical, organizational, and other editorial issues.
	4) On page 8, this is where I am a little concerned about re-hashing results. Also, see comment above about vegetation coming before fish.	We have rewritten this to focus more on the transition from conceptual models to coupled fish-environment models.
	5) What is the imagery source for Figure 1.2?	We have added in the reference
	6) For Figure 1.3, the inset is not that much larger than the plot itself. Also, hydrographs are usually shown for the water year, October – September.	We have used the calendar year because that is the way the hydrodynamic model is run and because it is more intuitive to most readers. For Fig. 1.3 we investigated other possible graphical configurations but felt this slight slize increase from larger figure to inset conveyed the infromation well.

	7) What does DHSVM stand for on page 10? The authors need to be careful about defining their acronyms at first use.	We have made an effort to define all acronynms at first use.
	8) On page 12, should be "generalized additive model."	Done
	9) The model procedure for Figure 1.6 is not explained in enough detail to be repeatable.	We added text to clarify how we estimated water use for years in which we lacked water use data.
	10) Table 1.1 is hard to read due to double spacing.	We have removed the double spacing.
	11) On page 18, it's not clear how the unregulated flow scenario was calculated. Where does that information come from?	We added a note about the source of the DHSVM and include the reference to the Skagit Story Map, where unregulated flow runs are described.
	12) On page 20, what is FVCOM? What is SCHISM?	These are now defined in the text
	13) For Figure 1.9, wouldn't local precipitation affect vegetation growth and salinity regimes? As an aside, I feel that the lack of inclusion of local precipitation as a whole is a shortcoming to the model.	Local precipitation will not likely not influence Fig. 1.9 because measurements are made at the end of summer when precipitation effects are generally very minimal. While local precipitation would have been desirable to include, it would have required a different model to incorporate both groundwater recharge and evapotranspiration in cropland and tidal wetlands.
	14) Figures 1.10 and 1.11 should be more clearly labeled (June/July; Alt 1/Alt 2).	We have provided additional labels and/or description in the figure caption for these two figures
	15) On page 26, why are the Key Conclusions and Study Limitations sections numbered when everything else is in paragraph form?	We have made these originally-outlined sections into full text.
	16) On page 29, references are out of alphabetical order.	We have fixed this.
	Chapter 2	
	The second chapter outlines the SCHISM hydrodynamic model used to derive surface level, salinity, and flow estimates for the Skagit River Delta, model output, and validation. As for Chapter 1, I noticed numerous minor grammatical and editorial mistakes that warrant attention from the authors (for example, switching back and forth between past and present tense in the same sentence). I will not list them all here.	We have made an effort to correct grammatical, organizational, and other editorial issues.
	Major comments	
	1) I thought the approach and reasoning were explained well; however, the context for the study and background information was glossed over. Case in point, there were only 10 citations for this entire chapter, and most of them were related to documentation about the various hydrodynamic models.	We added more text and citations on the background of this study as suggested. More background about this study could be found in Chapter 1.
	2) On page 34, I would like to hear more about the SCHISM limitations. I don't think these are particularly well addressed in the Introduction or Discussion. Also, out of curiosity, why is SCHISM more computationally efficient than FVCOM?	SCHISM is a very robust, computationally efficient hydrodynamic model. Based on our experience, its performance and accuracy are highly sensitive to the quality and resolution of the unstructured grid as well as the timestep. It requires the authors to have a good knowledge of grid generation and hydrodynamic modeling. Because it also uses a semi-implicit time integration numerical scheme, which allows it to use larger timesteps than FVCOM (which typically uses explicit time-stepping constrained by strict CFL condition), it is generally more computational efficient.

	3) I found the description of the validation procedure to be lacking. If continuous water level and salinity data loggers were used to validate the model, the authors need to include a description of how those loggers were deployed, a map of their location(s), how often data were collected, the QA/QC procedures, etc. It is unacceptable that this information isn't provided.	We added additional descriptions on the field data collection. The data locations are also indicated in Figure 2.6. in the "Model Validation" subsection of the Methods section.
	4) In Table 2.1, none of the error terms (bias, RMSE, R) are described or defined in the methods, and a description of the statistical model validation procedure itself is almost entirely absent. This needs to be addressed in detail.	We added definitions of these statistical methods in the same "Model Validation" subsection of the Methods section.
	5) On page 55, the authors reference a GAM analysis, but this analysis is not mentioned in the methods section. If the authors conducted a GAM, they need to explain their procedure in full in the methods.	We added more description on the GAM analysis method in the "Methods" section.
	6) I found that the discussion section did a weak job of contextualizing the findings in terms of broader impacts in Puget Sound and the Pacific Northwest (see comment above about only 10 references). At the very least, the authors should provide context for why these findings are important re: the vegetation and salmon analyses conducted in the following chapters.	We added more discussion on why the hyrdroynamic responses are potential important to Pugest Sound and especially on vegetation and salmon populations.
	7) I think the Appendix could be done away with altogether. If this was a publication and not a report, I could see the need for including this as supplementary info. For example, the table with the hydrodynamic model runs is helpful and could be tidied up and included in the main text.	We are including the appendix in case there is additional interest in particular model runs.
	There are a few tables/figures that I feel could be done away with altogether. Figures 2.A14-2.A21 are not particularly informative. I would just get rid of them.	We are including the appendix in case there is additional interest in particular model runs.
	Minor comments	
	1) On page 32, SCHISM should be defined at first mention.	Both SCHISM and FVCOM models are now defined in the Introduction Section where they are first mentioned.
	2) On page 38, would like more information on what XTide is and how it derives predictions for these sites. Also, there is no map up to this point as to where these four sites are located.	XTide (https://tide.arthroinfo.org/) is a free, open-source software application that provides tide predictions for over 9,500 locations worldwide. It uses a database of harmonic constituents to compute accurate tide and current predictions, and it's been widely used by mariners and researchers. We have been using XTIDE predictions for many model applications in Puget Sound. As suggested, we added more description in the methods section and also updated Figure 2.5 to include the location information.
	3) On page 39, would like more background on the sigma-Z coordinate system. What does this mean?	The Sigma-Z vertical coordinate is a hybrid vertical grid that combines the strengths of terrain-following (sigma) and fixed-depth (Z-level) coordinates to improve model accuracy and efficiency by using sigma coordinates in shallow regions and Z-levels in deeper offshore areas. This allows SCHISM to better resolve vertical processes across varying depths and minimize numerical errors (e.g., pressure graident errors associated with sigma coordinates). We added additional description in the report as suggested.

	4) On page 40 and throughout the report, I found the use of “Alt 1” and “Alt 2” to be confusing. I had to repeatedly flip back to Chapter 1 to get a definition for these scenarios. Is there a way to describe these more effectively for readers so they don’t have to flip around in the report?	We improved the text in the "Methods" section to provide a better description on these scenarios.
	5) On page 40, the list of open boundary sites is redundant with the list on page 38.	Thanks for pointing this out. We also noticed this error after sending out the draft report for review. We mistakenly copied the wrong text when formating the report into the Word format. We have fixed this error using the correct text during the revision.
	6) Figure 2.6, here is the study area map. I find it difficult to see some of the site points. The four “open boundary” locations still are not shown.	Two sites (FWP N Pond and FWP New Site) are next to each other and are indeed very hard to differentiate from the map. We have enlarged the figure during the revision to give a better view. We also added open boundary stations as a new panel in Figure 2.5.
	7) On page 42, how did the authors define “acceptability” for the water level predictions?	This is based on the typical accepted level of model performance (in terms of error statistical parameters) in simulating water levels in estuarine waters.
	8) On page 42, there is absolutely no description of salinity error and how it was quantified (see major comment above). In Figure 2.9 and Table 2.1 we can see that model predictions aren’t great and that the RMSE and R values (R 2 ?) are very poor. This warrants more detail in the methods and results.	We concur that compared to water level predictions, the model's performance in simulating salinity is not very satisfying. We totally agree there is certainly a lot of room for improvement but we also have to acknowledge it is still too challenging to accurate predict salinity (or scalar transport in general) at these intertidal sites, even using the best available state-of-the-art hydrodynamic models. We feel with better bathymetry data and models, we could do a better job in predicting salinity. On the other hand, despite of the relatively big errors in salinity predictions, we still believe the model results reasonably capture the spatial and temporal distributions in the Skagit system.
	9) On page 43, I find it concerning that the authors state that salinity is sensitive to reductions in freshwater inputs when their model performance is so poor. How confident can they be about these findings given model error?	The high sensitivity of salinity response to freshwater inputs and water withdrawals are driven by the nature of physics, i.e., salt transport is heavily influenced by both oceanic and freshwater exchanges. We believe the relative changes in salinity distributions between different water withdrawal scenarios are physically correct and indicative, despite that the model's performance in predicting absolute salinity values warrant further improvement.
	10) In Table 2.1, how is an RMSE >5 ppt acceptable when salinity rarely exceeds 5–10 ppt at some sites?	Given the challenges in simulating salinity in intertidal marshes due to complex wetting and drying dynamics and numerous small-scale topographic features that could not be sufficiently resolved by the bathymetry datasets, we feel our model results are still reasonable judged from our years of experience in this field.
	11) Figure 2.10 and throughout, only one legend is needed for a multi-panel figure unless the scales change from panel to panel.	Thanks and we totally agree. We chose to keep the legends in each figure of this report, just in case each figure may be used separately. In the forthcoming manuscript, we will remove redundant legends as recommended.
	12) Figure 2.10 and throughout, the labels “Current,” ”Alt 1,” and “Alt 2” would be better placed above each panel. Also see comment about usage of Alt 1 and Alt 2 throughout document.	We totally agree. We plan to improve these in the manuscript. For the study report, since we are not restricted by space, we decided to leave them as they are.
	13) On page 51, the statement “For example, a total of 36 sites…” belongs in the methods.	We corrected this as suggested.
	14) On page 51, “representative sites” representative of what?	We feel they represent different geographic locations in the Skagit Delta.
	15) On page 51, what does “instantaneous maximum water level drop” mean?	It means these differences are calculated at the exact same timestep from the model output, e.g., not the difference averaged over certain time window.

	16) Figures 2.23 and 2.25, it’s hard to see the different scenarios due to overlap.	Indeed they are, because the results of these scenarios almost overlay with each other so could not be differentiated from the line plots. We had tried to use different line types to improve the visibility. However, we found they may turn out to be more misleading. So we decided to keep the same line types but use different colors to give a better comparisons.
	17) Page 55, “that salinity and depth” what?	The text was correct as "...indicated that higher river flow generally increased the proportion of time that salinity remained below 5 ppt and depth was greater than 30 cm,...".
	18) Figure 2.28, the y axis says “<5 ppt” and this makes more logical sense than >10 ppt.	It was a mistake in the earlier text and has been corrected in the revision.
	19) Figure 2.30, ditto above.	The same error was corrected.
	20) Figures 2.31-2.34, see comment above about removing these figures/tables.	We agree with your recommendations, but feel better to keep them in the report to provide additional information given we are not restricted by space. We totally agree these figures are not needed for journal manuscripts.
	21) On page 62, the authors reference AME, RMSE, and WS, but none of these metrics of model performance are described in the methods or results.	We apologize for the inconsistency here. We have revised the text as suggested.
	22) On page 62, is the bathymetric data really the problem as to why model output for salinity was poor?	Yes, bathymetry data is one major source of error for model's poor performance in predicting salinity. The current bathymetry dataset (lidar survey) could not accurate capture the true microtopography (e.g., small drainage channels) due to the presence of vegetation and water. There are additional reasons beyond the limitation of the model itself, such as the numerical errors in simulating wetting and drying, over-simplication in simulating the vegetation effects
	23) Two citations, Whiting et al. 2017 and Duke 1999 are missing from the lit cited section.	They are corrected in the revision.
	Chapter 3	
	I found this chapter to be underdeveloped and a bit confusing. In Chapter 2, the authors used their hydrodynamic model to estimate water surface elevation and salinity. This chapter appears to be an extension of that, using a completely different modeling exercise to model the relationship between channel morphology and flow. I think that needs to be explained more clearly to start with, because my first thought upon reading this was “why wasn’t this chapter combined with Chapter 2?”	Since Chapter 3 was not actually integrated into the other studies, we removed it from the report.
	Minor Comments	See above
	1) On page 77, “this lower resolution limit is about 20 meters.” I thought the SDHM grid in Chapter 2 was as small as 3 meters for smaller tidal channels? What model are the authors referring to?	See above
	2) Again, the justification to use “hydraulic geometry theory to ‘downscale’ the model results” would benefit for more context about what model the authors are trying to improve.	See above
	3) In Figure 3.1, an inset would be helpful to show these channels’ locations within the Skagit River Delta. Also, it would be helpful to label Channels 1 and 4, which are referred to in other figures, but their location is not described.	See above

	4) On page 79, the passage "One possible explanation for the deviation…" would fit better in the discussion.	See above
	5) In Figure 3.2, it is clear that channel width and area covary strongly (R 2 = 0.996), so why examine both of them in Figure 3.3?	See above
	6) Also in figures 3.2 and 3.3, it would be helpful to label points so we know where the outliers are (mostly in Channel 4) spatially.	See above
	7) In the results, since you measured flow every 15 minutes, it would be interesting to analyze those data through time at each site, so we can see how flow changes with respect to the tide and/or water depth. This would help address the caveat you mention later in your discussion; that data were collected on different days, and thus different tidal cycles.	See above
	8) On page 81, what about the tradeoff between velocity and water depth? (See above comment)	See above
	9) On page 81, "it seems likely that water surface elevation is unlikely to scale with channel cross section," but isn't it driven by the tide?	See above
	10) I'm not sure why the field methods are included in the appendix. Seems like you should just move the relevant material to the main body of the text and omit the rest.	See above
	11) In the appendix, include lat/long values in main body of text or omit. No need to include sampling figure twice.	See above
	12) I found the description of the GAM in the appendix to be unhelpful and lacking. I would suggest the authors provide a full justification for this procedure, the response variable, the predictor variables, and more details on how the analysis was conducted, and include it in the main text.	See above
	Chapter 4	
	This chapter describes the use of a non-parametric multiplicative regression technique to model vegetation response to flow and water use scenarios. I found the predicted vegetation responses to changing environmental conditions (salinity) to be compelling, but I think the way the model output is described here is inefficient. The author(s) collected data from more than 500 sampling sites (elevation, salinity, river discharge, dominant vegetation species), but they only modeled output for eleven sentinel sites, which were supposed to be representative of delta conditions. Nevertheless, in Figure 4.2 I see that there are gaps in the range of elevational/salinity gradients that were captured by these sentinel sites (e.g., sites <1.5 m NAVD88, sites with salinity >20 ppt). Furthermore, I found the presentation of these results as a narrative of percent changes and a very large table (Table 4.2) to be clunky. I think it would be more helpful to present these data as spatial output, so that readers can clearly see where vegetation changes are occurring on the delta. Points could be color coded or scaled to represent magnitude or percent change. I do like Figure 4.2, but it is mentally challenging to place these points in space. Even better, if the	We did not have time to present the data as suggested, but this is a good suggestion for publication. We now present conclusions using suggesting bar graphs of change for each sentinel site.

	author(s) could find a way to produce a spatial model of salinity using their 500 sampling sites and spatial interpolation methods, maps of predicted dominant species and how they respond to the scenarios would be a very cool and helpful addition.	
	Major comments	
	1) I think the author(s) should do more digging in the data they used to parametrize the model to determine why the data gap at 20 ppt occurred. I think this would be straightforward to do by creating histograms of elevation and salinity using the sampling data. In general, more information about these sampling sites and how they were selected (random?) and the range of conditions they encompass would be helpful.	This issue is adequately discussed in the text. "The gap in niche space likely reflects real constraints on habitat expression limitations resulting from interactions between Skagit Delta geomorphology and hydrology. The tight correlation with distinct geography suggests this is so. Areas near the river have low to moderate salinity, depending on their connectivity to river distributaries and proximity to Skagit Bay, while high salinity areas are not possible except in areas like the Swinomish Channel, Telegraph Slough, and northeastern Padilla Bay that are distant from freshwater river input. It is unlikely that it reflects an unfortunate gap in sampling effort that simply missed areas with intermediate salinity, because sampling was extensive (3400 points) and broadly distributed throughout the delta." The niche space gap is not surprising; not all possible combinations of salinity and marsh elevation need to occur on a particular landscape. The additional information requested in the last sentence of the comment is being provided by the two new maps of salinty and vegetation sampling points (see comment 3, below).
	2) I think there’s an issue with using interpolated salinity values at 9,000 cfs in a modeling exercise that encompasses different flow conditions in 2010, 2015, and 2019. My guess is that flow varied substantially among these years, and thus the “starting point” for salinity likely varied. If I remember correctly, this was estimated by the hydrodynamic model in Chapter 2. The author(s)’njustification for using 9,000 cfs to parametrize and run the model needs to be clearer.	Text was modified and added to provide the requested justification
	Minor comments	
	1) On page 88, would be helpful for the reader to re-hash what Alt 1 and Alt 2 are.	This was done by adding a paragraph on this topic to the Introduction.

	2) On page 88, I would not say salinity is easily estimated from remote imagery.	This issue was clarified with parentheticals as follows: "These two factors can also be more easily measured or estimated from remote imagery (elevation, via lidar) or field sampling (salinity)..."
	3) It would be helpful to have a sampling map of how the >500 sampling points that were used to parameterize the model were distributed across the delta.	This map was added as was an additional, similar map showing > 3400 vegetation sampling points.
	4) On page 88, 50 cm deep? Are you sure it wasn't 5 cm?	It was 50 cm. Not 5 cm.
	5) In the methods, it would be helpful to re-hash all the model scenarios (flow, water usage).	It made more sense to do this in the Introduction; see response to comment 1 above.
	6) On page 89, I think the passage "Variability was very low in tidal freshwater areas…" belongs in the results.	This paragraph was moved to the end of the results, with additional modifications to relate to the preceding paragaphs. In this location it also makes a good transition to the Discussion.
	7) On page 89, define RTK-GPS abbreviation.	Done
	8) On page 89, specify that TYAN is non-native.	TYAN was specified as "invasive non-native" , while AGST was specified as "naturalized"
	9) On page 90, is the NPMR procedure accounting for spatial autocorrelation at all?	Autocorrelation was minimized as much as possible by parameterizing the NPMR model with relatively widely spaced vegetation data, on the order of 40-m spacing. This was thought likely to minimize the effect of clonal growth on autocorrelation. This is stated in the methods section.
	10) On page 90, how was the NPMR model "applied" to elevation and salinity values? Do you mean these values were used to parameterize the model?	This was clarified by specifying that the NPMR model was applied to model-derived elevation (from lidar) and salinity (from the hydrodynamic model) for the sentinel sites.
	11) On page 90, you need to clearly specify where the elevation and salinity values for the sentinel sites are coming from. I'm assuming a lidar DEM and the hydrodynamic model output from Chapter 2?	To clarifiy, the word "sentinel" was added to the following sentence. "Salinity values for each sentinel site and management scenario were acquired from the hydrodynamic model and not adjusted." The text already states that "...for eleven sentinel sites... Elevation values were generated from lidar data, adjusted by shrub cover and change in water surface elevations predicted by the hydrodynamic model." This seems pretty clear.
	12) On page 90, I'm concerned about the limitations of generating predictions for only 11 sentinel sites. See comment above. If there's away to generate spatially explicit output, that would be ideal over picking 11 sites.	We are exploring mechanisms for generating spatially explicit output on the scale suggested, but in the meantime we have chosen a reasonably manageable number of sentinel sites which are representative of environmental variation in the delta and of likely sensitive and responsive locations based on our understanding of the system.
	13) On page 91, what was the source of your lidar data?	The lidar data source was cited in the text and added to the references.
	14) On page 91, this vegetation bias can be accounted for using more specific statistical models like LEAN (see Buffington et al. 2016).	Buffington et al. (2016) state that, "We used a site-specific, multivariate approach to model the relationship between lidar error, determined by subtracting the lidar DEM from the RTK-GPS data, NAIP-derived vegetation indices, and lidar elevation. Specifically, the model was defined as: E = l + v + v2 + l*v + l*v2 + v2*v + l*v*v2, where, E is the error (lidar elevation minus RTK-GPS elevation), l is the uncorrected lidar DEM elevation, and v is the NDVI." Their goal was to develop a way of estimating vegetation caused error (E) from remote sensing, but to do so they had to determin that error in the field with RTK-GPS. I skipped the model building and went straight to directly measuring the error (E) with my available RTK-GPS data, i.e., Buffington et al. (2016) were not needed.

	15) A table of sentinel site numbers, names, elevations, and salinity values would be more useful than cramming this information into the figure caption of Figure 4.1.	In response to comment 3, this figure was altered to include salinity sampling points (and their values). Additionally, another similar figure shows vegetation sampling points that also references the sentinel sites. Thus, the sentinel sites are spatially located along with salinity and vegetation. The spatial representation in these figures seems more informative than a table.
	16) In Table 4.1, what does Elevation tolerance and Salinity tolerance mean? Column names need to be clearly explained in the table caption.	Done
	17) On page 92, the sentence "With local mean models…" should be explained in more detail and moved to the methods.	Model parameter tolerance is a result. The brief explanation of a possibly unfamiliar term for many readers seems necessary at this point where the result on tolerance is first mentioned.
	18) On page 92, specify that this niche space where there was a gap is ~20 ppt.	Done
	19) On page 93, the paragraph "The gap in niche space may reflect…" should go in the discussion. Also, it's easy to determine the reason this gap occurred. Can't you just check your data and see what dominant vegetation occurred at 20 ppt sites, or if there was a dearth of 20 ppt sites?	We understand the reviewer's concern, but this a result-relevant comment and was only of concern to one of three reviewers. We focued the Discussion focus on the model predictions relative to the 3 water management scenarios, their interpretations, implications, and uncertainties and caveats. This short paragraph on niche partitioning observed in current vegetation communities (not predicted ones) seems more appropriately addressed immediately following Fig. 3.3, so that the focus of the Discussion remains on predictions related to management alternatives. Bending the "rules" a little for the sake of readability can be appropriate, as in this case.
	20) On page 93, would be helpful to know if the sentinel sites were sampled.	We assume the reviewer is asking if the sentinel sites were sampled for porewater salinity and vegetation/elevation. The two new maps (response to comment 3) show the sentinel sites superimposed on locations of soil porewater salinity and vegetation sampling, thereby addressing this comment.
	21) See comment above. It would be helpful to see sentinel sites mapped with predictions. Even better, it would be cool to have spatially explicit output for dominant vegetation.	We had a hard time envisioning how mapping predictions for each alternative would be done concisely; it would seem to require at least two maps, one showing differences between current conditions vs Alt 1 and current vs. Alt 2, and this would have to be done for each species showing significant change for a total of 2 x n maps. Instead we followed the suggestion of creating bar plots to show differences (see next comment).
	22) On page 94, at the very least, it would be more helpful for readers to visualize changes using a bar plot, rather than writing them out as a laundry list of percent changes.	We have added a collection of bar plots (one for each sentinel site) and moved the table to a supplementary appendix.
	23) On page 101 (discussion), I'm curious about the timescale of vegetation change with respect to the water management scenarios. How long does it take vegetation to shift following changing environmental conditions like salinity? At what temporal scale are we managing here?	A paragraph on this topic was added to the Discussion. The bottom line is that rates of vegetation change can vary from sudden dieback (within one growing season) to decadal scales (e.g., invasion by non-native species) and depends on the natural history of the ecosystem and its species.

	24) On page 102, watch subjective language like "huge."	I think this language is appropriate, especially in a Discussion section, where my judgement can be expressed. Should I blandly state only the % change in vegetation without commenting on whether this change was small or large, or very small or very large? These are all subjective evaluations, but such evaluations are relevant in a Discussion and relevant to assessing the impact of management alternatives. My professional judgement about 26%, 18%, etc. changes being "huge" does not seem unreasonable. I think these are huge and concerning changes. If I said "mind-blowing" or "incredible" that would be overly dramatic and inappropriate.
	Chapters 5 and 6	
	I am reviewing these chapters together because I feel there is room for significant restructuring. Chapter 5 was structured like a stand-alone publication, with little contextualization or reference to the overall technical document. The only place where the hydrodynamic modeling exercise was mentioned was on page 123 in the discussion. The authors used a burdensome approach with a GAMM model for every single species observed above a certain abundance threshold. There is no discussion as to how the alternative water management scenarios will affect fish assemblages. Instead, this is addressed in Chapter 6 using an NMDS procedure. Chapter 6 itself is where the overall theme of the report is better incorporated, integrating the hydrodynamic model from Chapter 2 and the GAMM output for Chinook salmon in Chapter 5.	We have reorganized the report so that the fish/habitat models (formerly Chapter 5) is now the first appendix of the fish - flow simulations (formerly Chapter 6, now Chapter 4). This addresses the organizational concerns highlighted in this comment and others.
	My suggestion is to better integrate the hydrodynamic model output in both chapters to frame community shifts and Chinook salmon abundance in the context of the overall goal of the report. In Chapter 5, use a constrained ordination approach like a partial CCA or distance-based RDA with a Bray-Curtis dissimilarity matrix to directly link fish assemblages to environmental variables. This will help mitigate some of the major issues with running individual GAMMs for each species and with substantial covariance among explanatory variables (see comments below). Variance partitioning can be used to determine the relative importance of these variables in driving fish assemblages. The "predict" function can also be used with these multivariate approaches to determine how assemblages will change in the delta and nearshore intertidal zone as a result of the hydrodynamic model output.	These ideas are constructive, but we had limited opportunity to implement them in our rather brief period for revision. Nevertheless, these suggestions will be useful for revising chapters for publication.
	In Chapter 6, the authors can use the GAMM procedure described in current Chapter 5 to predict Chinook salmon responses to predicted hydrologic change, as they have already done. The difference is that the GAMM procedure methods are now moved to Chapter 6, and the NMDS is omitted. In this case, the GAMM procedure is only used to predict shifts in abundance for Chinook salmon and other major species of interest.	We have reorganized the report so that the fish/habitat models (formerly Chapter 5) is now the first appendix of the fish - flow simulations (formerly Chapter 6, now Chapter 4). This helps improve organization and partially addresses the concern raised in this comment.
	My other concern is that, similar to Chapter 4, the authors selected "sentinel" sites in Chapter 6 on which to predict model output when spatially explicit data are available from the hydrodynamic model. It	We considered producing maps as in Chapter 2. Howver, the sheer amount of data (4 flow simulations x 3 model years x 2 time periods x 2 fish metrics x multiple days and hours) motivated us to simplify summaries using sentinel sites.

	would be interesting to see a spatially explicit map, as opposed to the points shown in Figure 6.3.	
	Major Comments	
	1) The introduction of Chapter 5 is very broad, and is clearly structured for submission to a scientific journal. There is no reference to how this part of the study fits in with the broader report. The introduction hardly talks about the importance of this research for the Skagit in particular until the final paragraphs, where it frames the Skagit as a model study system.	We have reorganized the report so that the fish/habitat models (formerly Chapter 5) is now the first appendix of the fish - flow simulations (formerly Chapter 6, now Chapter 4).
	2) On page 105, it would be helpful to have more specific study objectives, and to frame those objectives in terms of the broader report.	We have reorganized the report so that the fish/habitat models (formerly Chapter 5) is now the first appendix of the fish - flow simulations (formerly Chapter 6, now Chapter 4). In this organization, it becomes more clear that the previous Chapter 5 (now Appendix 4.1) is mostly about fish-environment models, and the previous Chapter 6 (now Chapter 4) is about examining water scarcity simulations.
	3) As a result of the structuring this chapter, a lot of the information ends up being redundant with Chapter 1. For example, most of the material in the study system section is already presented in Chapter 1, or could be moved there.	While true, we decided to retain the information so that each chapter could function on its own. If readers are interested in getting a high-level view, they can read the Executive Summary, If they want get more details without reading everything, they can read Chapter 1. If they want the specifics to each modeling exercise (perhaps independently of the other modeling studies), they can turn to Chapters 2-4 and still get sufficient background information.
	4) On page 109, I'm seeing environmental predictors that likely covary substantially. For example, temperature and day of year almost certainly covary. I imagine water depth and tide stage do too. It is inadvisable to include covarying explanatory variables, even in a robust and flexible modeling framework like a GAMM. I suggest completing a complete evaluation of the degree of covariance among predictors (using Pearson's R or similar), and omitting some of these predictors.	We examined a number of these cross-correlations, and determined that their correlations are not as strong as predicted, particularly in relation to fish abundance or presence absence. For example, salmon presence increases and then declines in the delta over the spring and summer, while temperature continues to increase through August. Likewise, tide stage and water depth do tend to covary, but their relationships with indvidual fish can be different, especially in comparison to the other tide stages such as ebb and flood. We have also removed one variable that did show a strong correlation - dissolved oxygen was tightly correlated with temperature, so we removed dissolved oxygetn.
	5) In your GAMM model, how did you account for the fact that the delta sites were highly clustered, and thus may have exhibited substantial spatial autocorrelation? Was this accounted for by the sampling station random effect?	Yes, we incorporated site/stratum as a random effect.
	6) On page 109, what model evaluation criteria were used and why didn't you report them?	Given that all these species rely on intertidal environments, we hypothesized that the suite of variables we examined were important in predicting presence or abundance. Hence, we focused on a common model as opposed to many (possibly different) optimized models.
	7) On page 109, see comment above. To assess community assemblage and predict community change, it would have been more advisable to run a multivariate statistical analysis like a partial CCA or distance-based RDA. Then a GAMM could be used to parse out environmental relationships for species of interest, like in Figure 5.3. Running things	While the suggestion is valid, particularly where there is interest in understanding community compostion in light of multiple predictors, our greater interest was not the individual effects but more on the larger controls of these variables - river flow and tides. We also were not able to re-evaluate the modeling in the time that we had for completion of the report. As we

	separately as the authors have done makes it very challenging to get at "the meat" of how these assemblages are responding to environmental conditions.	consider publishing the chapters from this report, we will consider this comment again.
	8) As mentioned above, in the discussion section of Chapter 5, this work's connection with the hydrodynamic model and broader report themes is unclear. There is absolutely no mention of the water management scenarios. The only time integration with the hydrodynamic model is mentioned is on page 123, "In a forthcoming effort, the authors will use a hydrodynamic model to…"	This chapter is primarily about building models to investing the sensitivity of a variety of fish species to hydrodynamic parameters, not about the water simulations. Given the concern expressed by the reviewer, we decided to reorganize this chapter as an appendix of the final chapter.
	9) The statistical effect of time (which is likely confounded by water temperature) is hardly addressed in the methods/results or discussed. We know that salmon use the delta and nearshore intertidal zone differently between February and August (the study period), and we know that densities of fish likely vary widely. The authors need to better address seasonal shifts in community assemblages to capture the dynamics of this system and the way that environmental factors are actually affecting habitat use.	Fishes generally have seasonal peaks, and these can sometimes be partially explained by other factors like temperature (see previous comment). However, migrations and other phenological aspects are an important aspect of each fish's life history independent of temperature or other variables. Hence we included time of year in each model. This also allowed us to report predictions for specific time periods.
	Minor Comments	
	1) Figure 5.1, it would be helpful to have another inset for the South Fork Delta, where sampling stations are highly clustered together.	While we were unable to incorporate this comment by the revision deadline, this is a good suggestion for when we turn this Appendix into a publication.
	2) On page 108, the authors used a hurdle model structure to first model presence/absence and then abundance at sites where fish were present. I am not familiar with this outside of a Bayesian context, but it seems to me that the hurdle model could be better integrated, rather than running two separate models. I'm unsure whether this is possible in the 'mgcv' package for GAMMs in R. Something the authors might want to look into a little more.	We kept model results separate because it is difficult to account for the combined variance of each submodel, outside a Bayesian context.
	3) On page 108, if the authors had approximate sampling area measurements for the large and small seines, why not use density as a response variable and omit the categorical seine variable?	Regardless of the net dimensions, there is likely some selectivity in the efficiency of these two nets to catch certain fish. Hence we maintained this parameter in the model.
	4) On page 109, what does "We accounted for other factors…" mean?	These are random effects, gear types, and tide stages that provide noise to the fish-environment signal. We clarified this statment by providing these examples.
	5) On page 109, the sentence "We focused our analyses on…" belongs in the methods.	We felt this was a good way to provide a results-oriented foray into species composition.
	6) I would like to see more justification as to why the prediction sites shown in Figure 6.2 were chosen.	The 36 sites were selected for their representativeness of the estuary and that the sites were sampled for at least a full season for model validation purposes.
	7) On page 133, thank you for explaining the different water usage scenarios. I think a brief description like this would be helpful in the other data chapters as well.	Thank you

	8) On page 134, if you wanted to run spatially explicit model output, the difference between blind and distributary channels could be indicated as a digitized polygon.	Yes this is true. We had a number of technical hurdles unifying the HDM with the fish model that produce information at different temporal and spatial scales. We decided to predict to descreet locations for it intuativeness and avoid additional error structure from a spatial model
	9) On page 135, why was the two-week Chinook abundance value calculated as a sum and not an average?	River flow was a the resolution of each day, so we summed hourly predictions to obtain a daily effect.
	10) On page 135, how is this supplementary GAM procedure different from the GAMM described in Chapter 5, and why is it necessary? I think the authors need more justification.	The GAM analysis is used as a corroborating approach to the fish-environment and hydrodynamic model approach to examine independent effects of river flow (never modeled in GAMMS) and tide.
	11) On page 136, "scenario to" what?	We have fixed this error
	12) On page 136, it's very confusing where the NMDS data are coming from. Are these abundance values from 2010, 2015, and 2019 or are they generated from the GAMM predictions? Also, see my comments above about using a constrained ordination procedure instead.	The description was revised for clarity. The predictions are for presence over the time period at the one hour intervals, which is divided by total time to attain a proportion of presence.
	13) In Figure 6.3, the panels need to be labeled more clearly; presence vs. abundance (top/bottom), two-week period, water usage scenario, flow scenario.	We have revised the figure heading to make it more clear
	14) In Figure 6.5, isn't flow confounded by the effect of day of year? See comments about covariance above.	Not entirely, in as much as the prediction explicitly incorporates time of year and hydrodynamic elements that flow might effect. Furthermore, river flow doesn't necessarily increase or decrease over the course of the year.
	15) For Figure 6.6, ditto the comment above about panels being labeled more clearly.	We have corrected this as we did in Chapter 1.
	16) For Figure 6.7, see comments above about using a constrained ordination. There's no way to clearly see how environmental variables (in particular, salinity) are driving community assemblages.	We used salinity differences as one example of why we focused the NMDS to the South Fork. We decided to try to refine the evalution to this area since it was the only area with significant differences amonth the water use alternatives. We worked on provided language that no differences were expected in other portions of the estuary to balance the focuse on where change is happening.
	[1] Are those large marshes that I see where the Southern Skagit channel empties into bay?	Yes those are large marshes, and much of the restoration effort in the Skagit delta has been focused in South Skagit Delta.
	[2] There are confidence intervals on trends, but for many fitting techniques this is a different thing; with increasing sample size, confidence intervals will narrow down, whereas model skill for predicting individual measurements will not improve.	This is true, and a good suggestion for completing this work for publication.