Biological and Physical Effects of "Fish-Friendly" Tide Gates

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Prepared by
Correigh Greene and Jason Hall
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Watersheds Program, Northwest Fisheries Science Center

&

Eric Beamer, Rich Henderson, and Bruce Brown
Skagit River System Cooperative
LaConner WA

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Summary

A number of restoration techniques exist to counter widespread estuary habitat and connectivity loss across the Pacific Northwest, ranging from dike breaching and removal to installation of "fish-friendly" or self-regulating tide gates (SRTs). However, the physical and biological effects of these techniques have not been rigorously examined. In this report, we focus on the effects of SRTs, and examine their effectiveness in two different ways. First, we used a spatially extensive design to compare three site types: SRTs, flap gates, and unimpeded reference sites. The study compared ten SRT sites located from the Columbia River estuary north to Samish Bay in northern Puget Sound, five traditional flap gate sites (designed to drain freshwater but prevent tidal inundation and saltwater intrusion), and five unimpeded reference sites. Second, we used a temporally extensive design at three SRT sites to determine changes in upstream cumulative densities of Chinook salmon across the rearing season, relative to downstream values, before and after SRTs were installed.

In the spatially extensive study, we studied physical metrics upstream and downstream of tide gates and at reference sites during three visits spanning the primary spring-summer fish rearing period. We also sampled fish and invertebrates above and below tide gates and at reference sites. We found that site type appeared to affect a number of physical metrics including connectedness, water elevation, and temperature, but the degree to which each of these site types affected these physical metrics varied. In addition, densities of Chinook salmon (*Oncorhynchus tshawytscha*) and estuary rearing fish species were much greater at reference sites compared to sites with either flap gates or SRTs. For other species, overall patterns did not strongly distinguish densities between reference sites and flap gate or SRT sites.

In the temporally extensive study, the upstream/downstream ratio of Chinook salmon cumulative density at all SRTs was higher than at a traditional flap gate. The cumulative density ratio at this site increased 6-fold after a passive flap gate was replaced with an SRT, indicating that SRTs can improve habitat use by salmon. However, cumulative density ratios decreased 7-fold when a passive and manually manipulated side-hinged gate was replaced with a SRT, and this measure at all three SRT sites was an eighth to a tenth that of reference channels.

Together, these findings indicate that SRTs vary substantially based on design and operation and consequently vary in performance, depending upon the metric of interest. For estuarine-dependent species in general and juvenile Chinook salmon in particular, SRTs support habitat use above gates much less than natural channels and a little better than traditional flap gates. For other anadromous salmon species that may spawn in creeks above tide gates, SRTs do not appear to strongly inhibit passage or juvenile rearing density. These findings suggest that estuary restoration with SRTs will have limited benefits for juvenile Chinook salmon and other estuarine-dependent species, but can result in some improvement in connectivity and rearing habitat quality compared to traditional flap gate designs. SRT designs and operation standards that maximize connectivity, and site selection criteria that focus on reconnection of large amounts of habitat may overcome some of the limitations of reduced habitat use associated with SRT installation. These potential reductions can successfully be evaluated by comparing the benefits of SRT installation with those of other estuary restoration techniques (e.g., dike breaching or setback).

Introduction

Estuaries in the Pacific Northwest are important nursery areas for a number of ecologically, commercially, and culturally important species, including forage fish (e.g., herring and surf smelt), flatfish (e.g., English sole and starry flounder), and Pacific salmon (e.g., coho, chum, and Chinook salmon). These areas constitute critical habitat to chum and Chinook salmon populations listed under the Endangered Species Act (ESA), which have declined due in part to habitat loss (Hoekstra et al. 2007). Estuarine habitat loss has been implicated in variation of productivity for some fish stocks (e.g., English sole, Rooper et al. 2004; Chinook salmon, Magnuson and Hilborn 2003). Therefore, reducing estuary habitat loss is an important component for restoring ESA Threatened or Endangered populations.

Most of the original estuaries around Puget Sound, the Columbia estuary, and along the Western US coast have been surrounded by dikes that prevent tidal inundation, allowing the land to be used for agriculture, industry, or housing. In Puget Sound, it has been estimated that 80% of estuary habitat has been lost to diking and land conversion, and loss in the Columbia River estuary exceeds 65%. The typical pattern of habitat conversion is the building of dikes to prevent tidal inflow, and installation of culverts to drain rainwater and stream flow from relict sloughs. These culverts are normally fitted with traditional top hinged “flap” gates. Flap gates are designed to limit hydraulic connectivity by passive action, opening during ebbing tides to allow freshwater runoff to drain and closing on the rising tide to prevent tidal flows back into the diked area. Consequently, flap tide gates can prevent passage of adult and juvenile fishes, and change the ecological characteristics of the former estuarine wetlands into freshwater marshes and dry land.

One proposed solution to restore the loss of estuarine wetlands is the substitution of traditional tide gates with other kinds of tide gates that enable inflow of tidal water into the diked areas and (theoretically) improved connectivity for fish (Giannico and Souder 2005). This partial inflow of tidal water would ideally restore some of the wetland structure and function required by salmon and other estuarine-rearing fish, and would allow some degree of fish passage for migration and habitat use. These “fish-friendly” tide gates (also called self-regulating tide gates or SRTs) can vary greatly in design and operation. Gate doors can be a vertical flap or side-hinged, and can even include a “pet door” within the larger door that opens independently of the main door. They can be mounted to a simple headwall, or onto culverts at variable elevation. Operation of SRTs can vary in how long the gate remains open and how wide the opening is, and these automatic operations can often be manually overridden for certain time periods. Given this variation, it is unclear how well SRTs improve access and quality of estuarine habitat, compared to other restoration options (e.g., dike breaching, dike setbacks). Hence, it is unknown how well fish-friendly tide gates actually restore connectivity and rearing habitat to estuarine-dependent species (Giannico and Souder 2005).

In this report, we examine physical and ecological characteristics of ten existing SRTs. In a spatially extensive study, we compare these metrics at SRTs to the same measurements at flap gates and reference sites. In the temporally extensive study, we examine Chinook salmon density across the juvenile rearing season at three SRTs over multiple years during which tide gate design and operations changed. We compare measurements at these “treatment” sites to open-channel reference sites. In both studies, we ask: how do SRTs influence local hydrologic processes and habitat use by aquatic organisms?

Methods

We used two types of designs to examine the physical and ecological characteristics of SRTs: 1) A spatially extensive study taking place within one year, 2) A temporally extensive study occurring over several years. In the spatially extensive study, we examined physical and biological indicators at several estuaries with SRTs in comparison to those with flap gates or ungated reference sites. The benefit of a spatially extensive design is the ability to examine multiple SRTs and reference sites to one another during the same period of time to best understand the range of variation by which SRTs influence physical and biological processes. In the temporally extensive study, we monitored three SRT systems for changes that may have occurred over a period of years starting before, or just after their installation. We focused on three SRTs that have been monitored over multiple years using a post-installation or before-after-control-impact (BACI) design. These case studies provide intensive biological monitoring with some physical monitoring to better examine the success of SRT installations.

In both studies, we hypothesized that connectivity was a primary driver of the physical and biological characteristics we measured. In aquatic systems, this general concept has been used in a variety of contexts, ranging from the hydrologic connection among sites at a local level (Bottom et al. 2005) to systems at more regional scales (Beamer et al. 2005, Schick and Linley 2007,) to the interdependence of life histories of organisms that move among aquatic habitats (Able 2005). This range reflects a temporal dependency corresponding to the residence time of the water column and mobility of the organisms of interest: connectivity at the local level corresponds to movements that could occur within a day, while the larger-scale metrics might be interpreted at weekly, monthly, or annual or even longer time scales. In this report, we examine connectivity in the context of hydrologic connections at a very local level (sites within 0.5 km of each other), and use several metrics that characterize connectivity in the horizontal, vertical, and temporal dimensions (Table 1, see below for details on calculation). Horizontal metrics of connectivity focused on the degree to which two sites are connected in the horizontal plane (e.g., the proportion of the total channel width through which water can pass, or the maximum velocity of water flow, which can directly influence fish movements up and down a channel). Vertical metrics focused on connections that depended upon tidal variation in surface water elevation (e.g., maximum mean high water). Temporally-dependent metrics focused on diurnal changes in connectivity in the other dimensions (e.g., the proportion of time tide gate doors were open).

Spatially extensive study

Study design. This study was designed to determine the degree to which SRTs restored connectivity and its associated effects, compared to reference systems and systems designed to have very low connectivity (culverts with flap gates). We studied SRTs in five different systems across Washington and Oregon: Samish and Padilla Bay, Swinomish Channel, the Skagit River tidal delta, the Chehalis River, and Young's Bay (Fig. 1, Table 2). Within each system, we chose one reference site, one flap gate site, and up to three SRT sites (Table 2, Fig. 2), resulting in a total of 10 SRTs. This design allowed us to examine a wide range of SRTs and to control for local estuary conditions or distributions of particular fish species. SRT sites could vary greatly in culvert dimensions (Table 2), design (side-hinged or flap, with or without a pet door set within the larger side-hinged or flap gate), and number of passive tide gates at the site (Table 2, Fig. 2). We visited each system three times between March and July 2011 to provide replication at the site level, and these visits were timed to observe known temporal windows of rearing by salmon and other estuary-rearing fish. Visits constituted a two-week observation period during which we monitored sites with dataloggers in both spring and neap tides and sampled abundance of fish and their potential prey during spring tides.

Physical monitoring. We deployed several types of data loggers to monitor hydrologic variation over the three two-week observation periods (Fig. 3). We used Solinst® Levelloggers (3001 LTC F30/M10) to characterize water level, salinity, and water temperature upstream of each tide gate and at each reference site. A combination of Solinst® Levelloggers, Global Water® Level Logger (WL15) with an iButton® Temperature Logger (DS1922T-F5#), or Hobo® Water Level Logger (U20-001-04) sensors were deployed downstream of each tide gated site to measure water level, salinity (only at sites with Solinst® Levelloggers), and water temperature. These sensors were installed one to two days before the first biological monitoring event at each site and no later than the low tide preceding the first biological monitoring event at each site (see below). Each sensor was set to record at one-minute intervals and all sensors were retrieved 10 - 14 days after installation to capture changes in water elevation, salinity, and temperature during all fish monitoring events and both neap and spring tidal cycles.

We deployed sensors in 5 cm diameter slotted PVC monitoring well casings that were secured at each site to prevent sensor fouling and to maintain each sensor at a known elevation (see Elevation Survey methods below). Pressure data obtained from Solinst® Levellogers and Hobo® Water Level Loggers were compensated using barometric data collected with Solinst® Barologger (LT F5/M15) or Hobo® Water Level Loggers, deployed during each monitoring period to obtain water depths. Global Water® Level Logger data were compensated with the integrated barometric sensors to obtain water depth. Water elevations were calculated from the compensated water depth data series based on known elevations for each monitoring well and the depth of each sensor in each well.

We also installed Hobo® Tilt Loggers (Pendant G UA-004-64) on each tide gate to measure gate movement at one-minute intervals during each 10-14 day monitoring period. For flap gate sites and one top-hinged SRT, tilt loggers were deployed in watertight PVC canisters that were secured to the tide gate itself (Fig. 2J). On side-hinged SRTs, tilt loggers were installed on a chain that translated the horizontal movement of the gate's control arm to a vertical change in chain tension.

We also measured water velocity at each site between June and July 2011 using HACH® Ultrasonic Submerged Area Velocity Meters (Sigma 950) that were deployed for approximately 24-hours at each site. The flow sensors were secured to a telescoping pole that could be positioned in the vault or culvert at each gated site or to the monitoring well at each reference site. The telescopic pole was positioned so that the sensor was 7 cm above the ground in the middle of the culvert, vault, or channel floor. The flow sensor measured water velocity and level at 1-minute intervals during each 24-hour deployment. The flow sensor was positioned so that positive velocities indicated net flow into the channel and negative velocities indicated net flow out of the channel.

Biological monitoring. We collected biological samples three times at each of the 10 SRTs, 5 flap gate, and 5 reference sites, for a total of 60 sampling events. Biological samples consisted of 1) fish, amphibians, and large invertebrates captured in beach seines or fyke nets, and 2) samples of small invertebrates collected using a neuston net.

With the exception of two reference sites, we sampled fish and other large aquatic organisms using 3 mm mesh beach seines that were 25-31 m long by 2-3 m deep with a tightly corked float line and continuous lead line. We sampled each site during high water, focusing on a pre-determined sampling area and depth distribution that could be completely sampled by any of the nets we used. At tide gate sites, we sampled one site below and above the tide gate adjacent to the end of the culvert, but geomorphology, vegetation, and large debris required us to locate some sampling sites as much as 200 m away from the culvert. At the five reference sites, we chose two replicate sites to sample within 200 m of each other. During beach seines, we circled the predetermined sampling area with the net and hauled the net up onto a beach at which the net could be properly stacked. All fish captured in the net were counted and up to 25 of each species were measured for length. Absolute counts were converted to densities by dividing by sampling area.

Due to natural constraints, two of the five reference sites used in the temporally extensive study were sampled using fyke nets instead of beach seines. In these cases, a fyke net with a funnel trap was deployed across the channel at high tide, and fished while the tide ebbed (see below). At low tide, all captured fish were collected from the cod end of the net and counted and measured just like beach-seined samples. Fyke trap abundances were converted in a straight-forward fashion to density by expanding catch by recovery efficiency and dividing by the high-tide channel area above the trap (see below).

During each of the three visits, we also collected invertebrate samples from both above and below the 15 tide gates (180 samples total) and at one of the beach seine locations for the five reference sites (30 samples total) using two replicate tows of a 110 um neuston net towed 30 m. After each tow, the net was rinsed from the outside to remove invertebrates from the net, and the sample was sieved from the cod end of the net into a bottle with 10% buffered formalin for post-survey sorting, identification, and counting. In the lab, we focused on the larger taxa by sieving water samples through a 500 um sieve, and we categorized taxa into marine/estuarine or freshwater groups. Due to the complexities of invertebrate taxon identification, we were subsequently able to identify taxa in only one of the two replicates from the second visit, although we did compare replicates of five samples to evaluate sampling variation (40 samples total processed).

Elevation surveys. We completed high resolution Real Time Kinematic (RTK) GPS surveys to determine elevation and position data on key features at each site. At SRTs and flap gate sites, upstream and downstream monitoring wells as well as upstream and downstream invert (culvert lip) were measured using three-second occupations with a rover and onsite base station (Trimble ® R8). RTK surveys at reference sites were limited to the single monitoring well at each of these sites. At SRT sites, we also collected upstream channel thalweg measurements at accessible locations upstream of the tide gate to determine the upstream extent of tidal influence within the channel. Because of private property issues, we did not survey upstream of flap gates and reference sites, so these measurements at SRTs are reported without analysis (Table 2).

All RTK survey data were differentially corrected using two-hour Post Processed Kinematic (PPK) surveys of base stations that were established at each site. Position solutions for these two-hour PPK surveys were obtained from the Nation Oceanic and Atmospheric Administration's (NOAA) Online Positioning User Service (OPUS). All GPS data were collected using the NAD83 UTM Zone 10N coordinate system and processed using the NAVD88 (GEOID09) vertical datum from the static two-hour PPK base station observations.

Physical data processing. Tilt data were used to classify the position of each gate as open or closed during each one-minute interval. The position of the gate was then used to summarize the proportion of time each gate was open and the average, maximum, and minimum water elevation, temperature, and salinity during open and closed periods for each monitoring period above and below each tide gate. Reference sites were considered open at all times for all data summaries. Water levels occasionally dropped below the elevation of the data logger during some tidal cycles at certain sites. In these cases, data from these sensors were not used to calculate water elevation, temperature, and salinity summaries.

The maximum water elevation observed upstream of each self-regulating tide gate was compared with upstream thalweg elevation measurements to determine the channel length of tidal influence upstream at each site. We estimated tidally-influenced channel lengths based on distance measurements along the middle flow path of the primary channel, from the tide gate to the upstream-most point where thalweg elevations were less than the maximum upstream water elevation.

We also used tilt data in combination with water level to calculate connectedness, i.e., the proportion of time a given tide gate was open and when the downstream culvert invert was perched no more than 10 cm above the downstream pool surface, indicating time periods in which juvenile fish could physically enter the culvert and when water was not cascading through the channel and creating an impassible waterfall at the culvert base. At reference sites, connectedness equated with whether the site remained covered with water. Hence when water level was ≤ 0, connectedness for reference sites was also 0.

While tide gates in theory should eliminate connectedness when closed, we observed that certain tide gates could be "leaky," allowing some water flow after the gates had closed. Leakiness was detected using the velocity probes. As shown in Fig. 4A, nonleaky tide gates were characterized by a high in-flow pulse right at gate closing, followed by near-zero velocity until gates opened up, after which high outflow was observed. However, in some cases (Fig. 4B), velocity spikes could be detected after gates had closed, indicating movement of water despite gate closure. We calculated a leakiness index by dividing the cumulative velocity flux (i.e., velocity in or out, summed across recording intervals) during gate closure by the cumulative velocity flux across the entire monitored time period. For reference sites, which lacked a "closed" period, the tide is either flooding or ebbing without flux in the opposite direction. Hence leakiness for reference sites is 0.

Biological data processing. We adjusted abundance data by sampling area to calculate density of each species. We then focused on several indicator taxa (Table 3): Chinook salmon (the most ecologically sensitive estuarine-rearing species), three-spine stickleback (the most common species), total density of anadromous fish (any fish that spawns in freshwater and migrates from freshwater to the ocean during juvenile life stages), total density of estuarine-dependent species (species with key life stages in estuarine habitats), total density of nonnative species, and % of neuston invertebrates that were estuarine (i.e., do not inhabit freshwater).

We expected biological data to be highly variable and system-dependent. To reduce variation and simultaneously account for system-specific density or distribution, we transformed data using the formula:

$$I = \log_{10}[(0.001 + d_{t,s})/(0.001 + d_{r,s})]$$

where I is the relative density, d_s is the density of indicator d at site t in system s, and $x_{r,s}$ is the density of indicator x at the upstream reference site of system s. Hence, this equation indicates the orders of magnitude above (positive values) or below (negative values) which a given site differs from the upstream reference site. The addition of 0.001 in the numerator and denominator allows for situations in which a given indicator was not observed at the reference site. The equation indicates that the reference value is expected to be 0, although variation from 0 can occur because of replicate sampling at reference sites. For the % estuarine neuston index, reference site values always equal 0 because they were sampled at only one reference area.

Statistical analyses. To examine the effects of site type (flap, SRT, or reference) on physical and biological metrics measured at multiple visits, we used general linear statistical models (GLM) that focused on site as the sample unit. Site type, geographical system, sample location (upstream or downstream), and their two-way interactions were modeled as fixed effects, while visit and its interactions were modeled as repeated measures. We first performed multivariate GLMs to take advantage of covariation across multiple possible physical or biological metrics. Analysis of physical metrics included connectedness, water level relative to downstream invert (min. and max.) during periods when tide gates were open, and water surface elevation (min., mean, and max.), salinity (min., mean, and max.), and temperature (min., mean, and max.) during periods when tide gates were closed. Leakiness and velocity were not included because they were measured only once during the study. Analysis of biological metrics included all indicator groups (Chinook, stickleback, estuarine-dependent, anadromous, and nonnative), but not neuston scores because they were analyzed for only one visit. Multivariate GLMs included all data points and explicitly modeled upstream and downstream location comparisons. Thereafter, we used univariate GLMs focused on just upstream sampling locations to examine how specific metrics were influenced by site type. For metrics that were measured just once during the study, we used standard Analysis of Variance (ANOVA). We performed post-hoc comparisons of site type using Fisher's least significant difference (LSD), and set level of significance at $p < 0.05$.

Temporally extensive study

Study design. The goal of this study was to determine how well SRTs function over time as rearing habitat for juvenile Chinook salmon, relative to conditions at reference sites that were not modified during the course of this study, and (at two sites) to conditions measured before SRTs were installed. Hence, this study examined temporal aspects of tide gate function over multiple years. We focused on three SRT installations that were also included in the spatially extensive study: McElroy Slough, Fisher Slough, and Fornsby Slough (Fig. 5). The first two sites are characterized by a watershed upstream of tide gates that is large enough to support spawning by coho and chum salmon, but none of the three watersheds support spawning populations of Chinook salmon. All three sites provide estuarine habitat that can support juvenile rearing by these three salmon species as well as other estuarine-dependent species.

All sites underwent extensive restoration. At McElroy Slough, three flap gates were replaced with a side-hinged SRT and three passively opening flap gates in 2006 (Table 2, Fig. 2A). Very little habitat improvement occurred at McElroy although culvert replacement upstream of the SRT site also improved tidal connectivity. Restoration at South Fornsby Slough involved a number of phases, including replacement of a flap gate with a side hinged SRT, channel modification to allow for a marsh surface adjacent to the channel, and revegetation. Tide gate replacement at South Fornsby occurred in late summer 2005 (Table 2, Fig. 2D). Operation of both the flap gate and SRT varied over the course of the BACI study, with manual or entirely passive operation existing during our monitoring period for years 2004, 2005, and 2007. At Fisher Slough a three door, side hinged SRT system with two small submerged flap gates (one passive; one with manual capability) was installed in August 2009. This structure replaced an old set of paired wooden, side-hinged doors, hung on three separate openings with two additional submerged openings with passive flap gates. Dimension of openings and elevations remained the same (Table 2, Fig. 2E). Extensive habitat restoration (e.g., dike setback, channel relocation, revegetation) upstream of the SRT site occurred in the summer of 2010 and 2011, and SRT gates remained closed to prevent tidal inundation through this time period. However, treatment effects of habitat change (quantity or quality) did not influence this study's examination of SRT installation, because habitat restoration was designed not to affect hydrological processes until 2012. Hence, the monitoring captured changes in gate design and operation but not ongoing habitat restoration.

Restoration effectiveness of the three tide gate sites was studied using different restoration monitoring designs. Fornsby and Fisher Slough were monitored using a BACI design starting one (Fisher) or two (Fornsby) years before SRT installation and for at least two years post-SRT installation. McElroy Slough was a post-treatment design with four years of monitoring over the six years the SRT has been in operation. Tide gate operation varied by site and year. At Fisher Slough, the pre-SRT structure operated as a traditional passive side-hinged flap system most of the year with the doors held manually open during late spring through the summer. After SRT installation, the gates were set to close when tidal or Skagit River backwater flooding exceeded a set elevation which varied by management periods described in the site's Hydraulic Project Approval (HPA) (2.29 m NAVD 88 during fall and winter flood management; 2.90 m NAVD 88 during juvenile Chinook salmon migration; gates are to remain open during the summer irrigation period). The McElroy Slough SRT was set to close when tidal flooding exceeded 1.68 m NAVD 88. Reference sites for Fornsby and Fisher Sloughs were determined pre-treatment, and all three sites were compared to long-term monitoring reference sites in the region. All sites were monitored both above and below the tide gate using data loggers and a combination of beach seine and fyke trapping methods (Table 4). The sampling period was selected to coincide with the Skagit's juvenile Chinook tidal delta residence period (Beamer et al. 2005, i.e., from February to August), although this sampling period varied slightly among sites but not between strata within sites (reference, upstream, and downstream).

Monitoring locations at each tide gate were selected systematically downstream and upstream of the tide gate to represent the habitat types and spatial diversity within the project area (Table 4). Sampling sites were selected in order to compare juvenile Chinook salmon densities above the tide gate to that below the tide gate. In addition to examining reference sites associated with the Fornsby and Fisher BACI designs, we compared tide gate data with additional long-term status monitoring reference sites at a variety of locations in the Skagit River estuary. For these sites, "upstream" monitoring was the reference site itself, while "downstream" monitoring occurred in distributary channels connecting these sites to Skagit Bay. These additional sites provided better ranges for both environmental data and fish densities over time than just the two BACI reference sites could.

Environmental monitoring. We measured selected environmental variables at each site at the time of beach seining or fyke trapping to assess their potential influence on biological data across sites in the study area. We used Solinst® Levelloggers (3001 LTC F30/M10) to characterize water level, salinity, and water temperature upstream and downstream of each tide gate and at each reference site, which were set to record continuously at 15-minute intervals throughout the sampling period. Pressure data obtained from Solinst® Levellogers were compensated using barometric data collected with Solinst® Barologger (LT F5/M15) deployed in the vicinity of each site. Data loggers were placed in standpipes upstream and downstream of the tide gate, whose elevation was determined via RTK surveys (see elevation surveys, above).

In addition to data logger recordings, water temperature and salinity (ppt) were measured by the fish sampling crew at the time of the beach seine or fyke trap sample at each site using a YSI® Professional Plus Model meter. Multiple readings were taken at the surface and at the bottom of the water column within the beach seine or fyke trap set area. The values were averaged by surface or bottom for each sampling day and site. In addition, velocity was measured at each beach seine site using a Swoffer® Model 2100 flow meter. Four measurements were taken across the area seined after the set was made, and the average value of these readings was reported for each site/date combination.

Biological monitoring. We used two types of nets to sample fish at tide gate and reference sites. Most sites were sampled using a 24.4 x 1.8 m beach seine made of 3 mm knotless nylon mesh. The net was set in a "round haul" fashion by fixing one end of the net on the beach while the other end was deployed by walking or boat-towing the net "upstream" against the water current (if present), and then closing the net by returning to the shoreline in a half circle. For each beach seine set, we recorded the time and date of each set, the percent of set area (the area that the net covers compared to setting in a perfect half circle), and the maximum depth (m) of each beach seine set, taken by the fish sampling crew at the time of the sample at each beach seine site using a calibrated measuring rod. These measurements allowed catches to be converted into density estimates (see below, also Beamer et al. 2011).

For small blind channel reference sites that extensively dewater at low tide, we used a fyke trap to sample fish. The fyke trap was constructed of 3 mm knotless nylon mesh with a 2-0.6 x 2.7 m cone sewn into the opening to collect fish draining out of the blind channel site. This trap was set on posts at high tide and fished through the ebb tide, yielding a catch. Channel area measurements, water depth changes, and multiple recovery efficiency tests using groups of marked fish allowed catches to be converted into densities (see below, also Beamer et al. 2011). The maximum depth at the fyke trap site was taken at each set from the surface water relative elevation staff gage located upstream of the fyke trap at the time the trap was installed.

For all fish sampling, we identified and counted the catch by species, and measured individual fish lengths by species. We measured all fish caught by species when the catch for a specific species was 20 individuals or less. For catches larger than 20 individuals, we randomly selected 20 individuals for length samples.

Environmental data processing. For the temporally extensive survey, we focused on variation in measures of connectivity and surface water elevation. Metrics of connectivity included 1) the percentage of time the tide gate was opened, the 2) percentage of time that fish could move upstream, and 3) the proportion of bankfull channel that was open. The first metric was estimated either by angle logger data (described above) or by examining instantaneous changes in water level upstream compared to downstream. The second metric started with the first, but focused only on time periods in which tide was not ebbing (i.e., door is open and tide not ebbing). The third metric was based on simple measurements of open channel width. For reference sites, this value is 1 because sites are not restricted by fill, culverts, gates or other structures. For tide gate sites, the metric is the combined opening width of structures divided by channel width.

Surface water elevation metrics included % tidal muting and effective water surface elevation at mean higher high water (MHHW). To calculate the first metric, we compared water surface elevation upstream and downstream at MHHW using the NAVD88 vertical datum as a benchmark for sea level. At sites with tide gates, our calculation for percentage of tidal muting (M) was

$$M = 100*(H_d - H_u)/(H_d - E_i)$$

where H_i is the surface water height at MHHW measured either upstream (u) or downstream (d) of the tide gate, and E_i is the elevation of hydraulic control for the reference channel (r) or tide gate (t). At all reference sites, M = 0. To capture the variable nature of surface water elevation across reference sites, and directly compare this to surface water heights above the tide gates, we also examined the effective elevation at MHHW, which is simply H_u.

Biological data processing. For beach seined juvenile Chinook salmon, each set's catch was divided by set area to calculate a Chinook density for each beach seine set. For the fyke trap site, juvenile Chinook salmon catch numbers were adjusted by trap recovery efficiency (RE) estimates derived from three mark-recapture experiments using a known number of marked fish released upstream of the trap at high tide. Recovery efficiency estimates are unique to each site and are related to hydraulic characteristics of the site during trapping. We used the RE results to convert the "raw" juvenile Chinook salmon catch to an estimated population size within the channel network upstream of the fyke trap on any sampling day. The RE-adjusted Chinook salmon catch was divided by the bankfull channel area of the blind channel network upstream of the trap to calculate a juvenile Chinook salmon density. The limits of the blind channel network upstream of the trap were determined in the field. To calculate bankfull channel area, we digitized a channel polygon using high resolution orthophotos and used GIS to calculate area.

For each year of sampling, we estimated the season-long density of juvenile Chinook salmon at all monitoring sites and other long term monitoring sites located throughout the Skagit tidal delta. We term this fish density statistic *cumulative density*. Cumulative density (fish*days*ha-1) expands individual sampling efforts over the entire season using the formula

$$C = \sum_{m=F}^{L} D_m n_m$$

where C is cumulative density, m is month, D_m is the average monthly density, and n_m is the number of days in the month. The terms F and L are the first and last months sampled, respectively. Normally F = February and L = August, but these months could differ slightly depending upon the site and year (Table 4). Almost all years captured the entire pulse of estuary rearing by Chinook salmon.

We calculated separate cumulative density for upstream and downstream locations, and then divided upstream cumulative density by that downstream to obtain a cumulative density ratio that could be compared across sites, and removed any minor bias associated with variation in sampling period.

Statistical analysis. We examined Pearson correlations among the physical metrics describing variation in connectivity and water surface elevation, and determined whether these in turn correlated with the $\log_{10}$-transformed ratio (inside/outside) of cumulative density. Level of significance was $p < 0.05$

Results

Spatially extensive study

Physical effects. General patterns in physical metrics over time suggested strong differences between flap, SRT, and reference systems. The most obvious effects were the proportion of time each site type was subject to tidal flux and the resultant water levels during open and closed periods (Fig. 6). The proportion of time open was reduced approximately by half at many SRT sites relative to reference sites, and was reduced even more at flap gate sites, which opened on and ebbing tide when water had accumulated upstream of the gate (Fig. 6). These changes resulted in clear muting of tidal elevation upstream of tide gates. Temperature and salinity also exhibited apparent differences across different site types, but these metrics were more temporally variant, and systematically varied with system. At one extreme, Fisher Slough is essentially a freshwater tributary of the Skagit River subject to variation in tidal height, but little variation in salinity, and at the other extreme is Fornsby Slough, which is heavily influenced by tidal flux and consequently quite saline. Not surprisingly, the multivariate GLM revealed very strong effects of site type, upstream or downstream location, geographic system, and visit, as well as site type*location and visit*system interactions as based on Hotelling's trace statistic ($p<0.003$ for all).

Univariate analyses of physical effects focusing on upstream data revealed that physical metrics exhibited differential sensitivity to the main effects modeled (Table 5). Of the four types of metrics that focused on connectivity (connectedness, leakiness, and water elevation relative to invert and water velocity during open periods), only connectedness and leakiness exhibited systematic effects of site type or of other main effects and interactions. Connectedness increasingly differed across flap, SRT, and reference sites (Fig. 7A) as determined by post-hoc comparisons (Table 5), with SRTs exhibiting over twice the connectedness of flap gates but nearly half that of reference sites (Fig. 7A). Leakiness showed a strong opposite pattern (Fig. 7C, Table 5), with flap gates having the highest leakiness values and reference sites having the lowest, although no difference was detected between reference sites and SRTs after accounting for other effects. Both connectedness and leakiness exhibited moderate system effects as well as a site type*system interaction, indicating strong geographic variation. Although both water level relative to invert and maximum velocity into and out of channels appeared to show strong patterns across site type (Fig. 7B), these values generally exhibited little statistical difference from each other after accounting for system differences. The exception was a significant difference between flap gates and SRTs in minimum water level relative to invert when gates were open, which also varied systematically across visits.

Of the three remaining physical variables, only water surface elevation during closed periods exhibited systematic variation across site types at upstream sites (Table 5, Fig. 8A). Strong site type differences were observed in minimum, average, and maximum elevation values; post-hoc tests indicated that flap gates had 2-6 times lower water elevation than reference sites and a third to 5 times that of SRTs. In addition, SRTs had a 50% lower average water surface elevation than reference sites. Water surface elevation metrics also exhibited influences of system and visit, but no strong interactions.

In contrast, salinity and temperature exhibited weaker effects of site type relative to other main effects. Minimum and mean temperature at upstream sites during closed periods moderately differed among site types. Post-hoc tests indicated that flap gates and SRTs tended to have similar values, while reference sites were lower (Table 5, Fig. 8C). This was particularly true of minimum temperature. Salinity and temperature exhibited much stronger spatial or temporal effects, and also exhibited a number of strong interactions among main effects (Table 5).

Biological effects. The multivariate GLM revealed very strong effects of site type, upstream or downstream location, system, and visit, as well as a visit*system interaction as based on Hotelling's trace statistic ($p \leq 0.01$ for all). However, univariate analyses focused on upstream samples of biological indicators indicated that the metrics exhibited great variation with respect to the main effects modeled (Table 5). We focused on two contrasting single species indicators: Chinook salmon and three-spine stickleback. Chinook salmon are listed as Threatened under the Endangered Species Act and rely upon estuarine habitats for juvenile rearing (Healey 1980), while stickleback rear in a variety of freshwater and marine habitats and are tolerant of stressful aquatic conditions (Seilheimer and Chow-Fraser 2006). Both species were commonly encountered at sites during the course of this study (Table 3). However, each species responded differently to the study comparisons and to the range of hydrologic conditions recorded at sites. Chinook salmon densities exhibited a strong overall influence of site type, and post-hoc tests revealed that average densities were over four times greater at reference sites than at SRT or flap gate sites, which had statistically similar densities (Fig. 9A). Chinook salmon densities also exhibited strong system differences and moderate temporal differences, and a very strong system-temporal interaction. In contrast, stickleback densities did not significantly vary across site types (Fig. 9B), and were much more sensitive to visit timing. Stickleback densities exhibited great variation, and average densities were over three times higher at flap gates than at reference sites, with densities at SRT sites appearing more similar to flap gate sites.

Of the four species groups, only estuarine-dependent species exhibited strong effects of site type (Table 5, Fig. 10A). For this group, densities were over an order of magnitude greater at reference sites than at SRT and flap gate sites, which had much more similar densities. Densities for all other species groups were variable enough that they did not exhibit significant effects of site type. However, for percentage of estuarine/marine neuston (Fig. 10B), SRTs most resembled flap gates, and had a higher percentage upstream of tide gates than reference sites. For anadromous and nonnative species groups (Figs 10C-D), SRTs were more similar to reference sites, and both differed from flap gates by 3-6 times, although the very large variability in density across flap gate sites eliminated any strong significant pair-wise differences. Most species groups exhibited moderate spatial variation, and did not show strong interactions between main effects.

The combination of site type, geographic, and temporal elements to the study appeared to robustly explain connections between physical variation and biological metrics. As a simple comparison, we examined raw correlations of physical metrics with the biological metrics, and correlations of the residuals of the physical metrics (i.e., variation not explained by the statistical design) with the biological data (Table 6). Just focusing on the upstream sites, we observed a total of 15 raw correlations of physical metrics with our different biological metrics. At a significance level of 5% 4-5 of these would be expected to occur by chance, suggesting that some correlations were real patterns. After accounting for our analysis' main effects, only one correlation remained.

Temporally extensive study

Physical effects. Variation observed over time at three SRT sites, including two that were monitored before and after installation, corroborate the spatially extensive study. We focused on metrics primarily related to connectivity and water surface elevation, and how they affected cumulative densities of juvenile Chinook salmon. Both Fisher and S. Fornsby sites exhibited improvements in physical metrics in response to SRT installation. At Fisher Slough, two measures of connectivity exhibited slightly different patterns. The percentage of time that gate doors were open increased from 83% to 93%. However, the percentage of time fish could move upstream decreased slightly from 49% to 47%. The different trends at Fisher are explained by differences in gate operations between pre- and post-SRT years. In 2009 (prior to SRT), the Fisher Slough gates operated as a passive system 60% of our fish monitoring period. For the remainder (40%) of the monitoring period in 2009, the gates were manually held open. After SRT installation, the gates at Fisher were operated to close when water surface elevation caused by tidal or backwater flooding from the Skagit River was greater than a specified elevation per specific management periods described in the site's HPA (see methods above). A major difference in operation between pre- and post-SRT monitoring years at Fisher is that gates were not manually held open for the summer irrigation period in 2010 and 2011 to accommodate habitat restoration and construction occurring upstream of the Fisher SRT. At S. Fornsby, replacement of a flap gate with an SRT increased both connectivity measures by about the same amount: the percentage of time that gate doors were open increased from 28% to 40%, while the percentage of time fish could move upstream changed from 0% to 14%. Post-restoration connectivity at McElroy Slough was 67% and 39% for percentage of time doors were open and time that fish could move upstream, respectively. These connectivity parameters were only measured in 2011 at McElroy. The McElroy SRT was operated the same in each monitoring year so we assume connectivity results from 2011 reasonably represent other SRT years at McElroy.

Surface elevation metrics also exhibited improvements resulting from SRT installation. At Fisher Slough, tidal muting was reduced by roughly half (44% and 26% before and after SRT installation, respectively not including the 2009 period when the gates were manually held open), resulting in 13% increase in effective surface elevation at MHHW. At S. Fornsby, tidal muting declined from 66% to 47%, resulting in a nearly 40% increase in the effective MHHW. Across all sites, physical metrics were tightly correlated with each other (Table 7). Post SRT installation tidal muting at McElroy was 33%.

Biological effects. Increased tidal connectivity appeared to improve cumulative density of juvenile Chinook salmon rearing above tide gates at one of two BACI sites. At Fisher Slough, the replacement of manually and passively operating side-hinged gates with side hinged SRT gates was followed by a reduction in the cumulative density ratio by over 80% (Fig. 11A). This loss in cumulative density resulted in the tide gate cumulative density ratio decreasing from nearly 50% to 10% of Fisher's reference site before and after SRT installation, respectively. The story was different at S. Fornsby Slough, which exhibited systematic variation in the cumulative density ratio in relation to mode of tide gate type and operation (Fig. 11B). The greatest change was a nearly 6-fold increase in cumulative density when a passive flap gate operation was replaced with an operational side-hinged SRT. However, cumulative density ratios under all types of tide gates and operations were at least 8 times lower than cumulative density ratios at Fornsby's reference site. Although such comparisons are not possible at McElroy Slough (Fib. 11C), cumulative density ratios after SRT installation averaged 0.19 across four years of monitoring, mirroring SRT cumulative density ratios at Fisher Slough. Cumulative Chinook salmon density ratios at all SRT sites were well over an order of magnitude lower than index reference sites across the Skagit River estuary (Fig. 11D).

Across all sites and their modes of operation, cumulative density ratios exhibited strong correlations with all connectivity and surface water elevation metrics (Table 7), so it was difficult to identify a single measure that defined biological performance across all sites. Nevertheless, all variables show a strong ordering of cumulative juvenile Chinook salmon density ratio with 1) passive flap gate operations, which had the lowest levels of connectivity or surface elevation and cumulative density, 2) SRT operations, which exhibited moderate improvement in connectivity and surface water elevation but not necessarily improvement in cumulative density, and 3) reference sites with the highest levels of connectivity, surface elevation, and cumulative density (Fig. 12). Also evident is that passive flap or side hinged gates that are manually over-ridden (i.e., gates held open) during our fish monitoring period were consistently higher in cumulative Chinook salmon density ratio than purely passively operated gates.

Discussion

Our spatially extensive study revealed that by at least one measure (connectedness), SRTs generally reduced connectivity by over 50% relative to reference sites, and that this change appeared to have cascading effects on water surface elevation, temperature upstream of the SRT, and assemblages of estuarine fish. Flap gates reduced connectedness by another 50% relative to SRTs, or 75% relative to reference sites. Reductions in connectivity and water surface elevation should come as no surprise – that is what flap gates and SRTs are designed to do – but it is important to quantify the differences among such sites and how they affect organisms because this provides a sense of how much SRTs represent a "middle ground" in providing both benefits to adjacent land use and estuarine rearing species. Both physical and biological data indicate that whether SRTs are considered more similar to flap gates or reference sites strongly depends upon which metrics are measured. Nevertheless, the physical conditions resulting from SRT placement appear to create relatively poorly connected or unsuitable habitat for estuarine-dependent aquatic species, particularly ESA threatened Chinook salmon. For these biota, the combination of connectivity reductions and changes to habitat potential upstream of SRTs results in low enough densities that those sites more resemble flap gates than reference sites. Whereas, connectivity or physical metrics appear to vary linearly among different site types (e.g., y-axis in Fig. 7A) or tide gate operation (x-axis in Fig. 12), variation in densities of estuarine-dependent species appears exponential by site type.

These findings were further supported in the temporally intensive study and corroborated over multiple years. Our two BACI designs provided the best examples how contrasting structure designs and/or operation may influence biotic results. At S. Fornsby, top hinged flap gate replacement with a side hinged SRT was followed by a 6-fold increase in cumulative Chinook salmon densities, but these densities were still 8 times lower than its hydraulically unimpeded reference site (Fig. 11B). In contrast, the replacement of a passively and manually (depending on season) operated side hinged floodgate at Fisher Slough with a heavily engineered set of side-hinged SRTs was followed by roughly a 10-fold decline in cumulative density of Chinook salmon (Fig. 11A).

While surprising, the biotic results at Fisher Slough may best illustrate the importance of varying tide gate operations. In 2009 (before SRT installation), the passive side-hinged gates were manually held open starting in June. However, after SRT installation in 2010 and 2011, gates were closed during summer months because of habitat restoration and other construction occurring upstream. Now that the restoration has been completed, future gate operation will include the same "gates manually held open" period starting in June that occurred in 2009, and future monitoring at Fisher Slough will better determine the long term influence of SRT installation and habitat restoration at this site. Given its level of connectivity, we would expect it to perform somewhat better than other SRTs upon normal operation (Fig. 12). In any case, the temporal variation in cumulative densities at SRT sites illustrate that changes in tide gate operations can be just as important as differences in gate design.

The Fisher Slough example also under-scores the importance of monitoring. By having only one year of monitoring data before SRT installation, we have lost the opportunity to know whether 2009 was representative of all pre-SRT installation years at Fisher Slough. We strongly recommend BACI monitoring designs with multiple years for each time period. We also note slight differences in the monitoring period from 2009-2011 (Table 4), but in each year we sampled through the majority of the juvenile Chinook outmigration period, and any variation in density during the unsampled period of time is expected to be a fraction of the cumulative density estimate that was measured. Overall, our conclusion from monitoring this site in the context of other results is that various tide gate designs and their operation that reduce connectivity and mute tidal elevation appear to result in a fraction of the seasonal rearing Chinook salmon density upstream of tide gates, compared to unimpeded reference sites (Fig. 12).

The importance of connectivity. A number of studies have highlighted the importance of connectivity for habitat processes (e.g., Cloern 2007) and fish rearing (West and Zedler 2000) and migration (e.g., Gillanders et al. 2003) in aquatic systems, and we found similar patterns in this study. In theory, the primary benefit of SRTs is increased connectivity relative to traditional flap-style tide gates. Connectivity at SRTs should ameliorate flow conditions on both sides of the gate, thereby allowing muted saltwater inflow on floodtides to support estuarine-sensitive species while removing freshwater from inside dikes on ebb tides. We used several metrics to examine connectivity across horizontal, vertical and temporal dimensions (Table 1), and our analysis suggested that relatively simple metrics could robustly predict the likelihood that a channel's environmental conditions were modified by the tide gate. In the spatially extensive study, connectedness was just as powerful as site type at explaining Chinook salmon and estuarine-dependent species densities. In our temporally extensive study, a metric as simple as the percentage of the channel remaining open to water passage exhibited a surprisingly strong correlation with cumulative density.

Leakiness, while showing significant differences among site types, is not an ideal indicator since it is not a design feature of tide gates, and mostly serves to indicate that some variation in physical or biological data may be a product of design mis-function. Several flap gates appeared fairly leaky as judged from sounds of rushing water heard during at high tides (when tide gates should be fully closed), and the leakiness values confirm that. But a very leaky tide gate might not equate with high connectivity if it systematically was cracked open enough to only allow water movement at high velocity. Indeed, we did not find strong correlations of biological metrics with leakiness across the range of leakiness measured.

Other connectivity metrics did not exhibit strong variation across site types, or did not predict fish densities. Depth and velocity should in part determine upstream juvenile salmon habitat use because of physiological or behavioral limits of fish movement. In our study, water level above invert describes how well culverts are submerged during open periods when fish could have access to upstream areas. Peak velocity metrics describe the highest values of velocity in and out of the channel or culvert. Both SRTs and reference sites exhibited peak velocities that surpass exhaustion-inducing swimming velocities in juvenile salmon fry (0.27-0.43 m/s, Flagg and Smith 1979, Giannico and Souder 2005), and at SRTs, these velocities occurred during the shortened time windows in which the SRT is open to passage (Fig. 4). Ecological theory predicts that juvenile salmon would work against strong current mainly when that bioenergetic effort is worth the cost (e.g., higher growth, predator avoidance), and not simply for habitat utilization. On the other hand, extremes in both depth and velocity are not necessarily representative of the entire tidal cycle (Fig. 4, 6), and may not be predictive of fish densities because fish are likely sensitive to these cues and can adjust their behavior to move upstream under preferred conditions. This contrasts with connectedness measures, which explicitly measure the proportion of time in which fish are able to move upstream through culverts. It is likely that that velocity may have been an important constraint on upstream movements of estuarine-dependent species like juvenile Chinook salmon, but the temporal dynamics of velocity changes across the season were insufficiently measured by the study design

Roles of other environmental variables. Clearly, factors other than connectivity play a role in determining whether channels are suitable for fish and other species of interest. For example, in estuarine systems, some species inhabit specific ranges of salinity, and juvenile salmon are notably sensitive to variation in water temperature. We found that water surface elevation and temperature but not salinity were strongly influenced by site type, clearly indicating that tide gates have the potential to modify the local environment in addition to affecting accessibility. By limiting tidal elevation, tide gates also could influence the tidal prism upstream, and as a consequence could affect channel forming processes in habitat both upstream and downstream of flap gate or SRT structures (Hood 2004, 2007).

If accessibility is not constrained by tide gates, fish selecting areas to rear might cue in on physical metrics, and densities would then strongly correlate with these metrics. We found that densities of stickleback and estuarine-dependent species both correlated with a number of metrics (Table 6). However, after accounting for variance explained by site type, system and visit main effects, most of these correlations disappeared. Note that densities of stickleback and estuarine-dependent species both strongly correlated with connectedness, so the covariation of physical and biological data appear to be mediated primarily by connectivity differences among site types. Nevertheless, it is possible that key physical features influencing fish habitat preference exist at much finer scales than we measured with loggers.

Sensitivity of biological indicators. Two biological indicators appeared strongly influenced by tide gates: Chinook salmon density and more generally, density of estuary-dependent species. For both indicators, SRTs appear to function much more like flap gates than unimpeded channels, more so than would be expected by the linear reduction in connectedness. Because all estuarine-dependent species, including juvenile Chinook salmon, must swim through tide gates to access upstream areas, connectivity reductions should play a primary role in limiting rearing densities. For Chinook salmon, it is also possible that the combination of lowered connectivity and altered environmental conditions reduced utilization of channel habitat upstream of tide gates. Other studies have found that juvenile Chinook salmon in estuaries are sensitive to water depth and temperature, preferring water at least 1 m deep (Beamer et al. 2005), and less than 15° C (Brett 1952, Brett et al. 1982). Changes to these metrics do not explain why cumulative densities dropped at Fisher Slough after SRT installation (Beamer et al. 2011).

The other biological indicators – stickleback, estuarine neuston, anadromous fish, and nonnative species – were not statistically sensitive to tide gates. Even so, stickleback densities exhibited strong negative correlations with connectedness and water surface elevation and positive correlations with temperature (Table 6), which suggest that high stickleback densities might be diagnostic of poorly connected systems with water quality problems. Indeed, stickleback species are used in indices of biotic integrity to represent species that are tolerant of water quality deficits (Deegan et al. 1992, Seilheimer and Chow-Fraser 2006) in estuary and wetland environments. Furthermore, stickleback contrast with Chinook salmon in that habitat upstream of tide gates can constitute natal habitat.

In addition, anadromous and nonnative species densities exhibited relatively large differences between flap gates and the other two site types, which were swamped by variation in density at flap gates (Fig. 10). Nonnative species were quite rare, totaling of 10 species occurrences at upstream sites (6 at flap sites and 3 at SRTs), so these data should be considered largely preliminary. Anadromous species provided a much richer data set. Although Chinook salmon were included in the anadromous fish taxa, the most abundant anadromous fish were juvenile coho salmon. Across sites, coho salmon were over twice as abundant as Chinook and observed upstream of all but five tide gates (flap or SRTs), absent only in the most saline systems. For coho salmon, losses in connectivity appear only for the extreme case of flap gates. Together, data on anadromous fish and nonnative species suggest that channels upstream of SRTs are more similar to reference sites than flap gated systems.

Management applications. SRTs appear to greatly impact those fish dependent upon rearing in estuaries, but not anadromous fish that pass through them during their outmigration. The passage and upstream habitat criteria for these groups are different. For example, juvenile Chinook do not spawn in creeks upstream of the tide gates in our study. Thus, juvenile Chinook must navigate a tide gate in order to take advantage of upstream habitat. In contrast, coho salmon could originate from adults spawning in creeks upstream of the tide gate. Thus, juvenile densities upstream of tide gates likely represent rearing during outmigration (Miller and Sadro 2003) rather than upstream colonization of habitat.

Given these circumstances, it is worth asking under what conditions is installation of SRTs a prudent alternative to other potential estuary restoration techniques, and what design features should be incorporated. Even in the best of circumstances, SRTs appear to substantially hinder utilization of upstream slough habitat by Chinook salmon and other estuarine-dependent species, and as such are not likely equivalent to dike setbacks, removal, or breaching methods. As such, SRTs appear most useful for systems lacking juvenile Chinook salmon rearing potential but which still have other anadromous species such as coho salmon or steelhead spawning upstream, or for systems that are naturally disconnected (by distance or by channel network) from Chinook salmon migration pathways. In addition, SRTs may also prove useful in removing nuisance species from previously blocked slough habitats. These patterns are suggested in Fig.

10C-D, although our results were inconclusive due to high biological variation. Installation of SRTs does increase tidal elevation and salinity relative to flap gates, so these aspects may have benefits independent of whether they support Chinook salmon and other species. For example, deeper or higher saline waters may help kill noxious aquatic vegetation dependent on freshwater in channel systems.

Our primary biological metrics were fish densities, and an important additional aspect to evaluating whether SRTs might be beneficial in a particular area is the amount of rearing habitat potentially restored. It is possible that for some sites, the benefit of partial reconnection of an extensive channel area trumps the lower densities expected at these sites. In this respect, the relevant metric for calculation is the increase in rearing capacity (number of fish, i.e., density x area) expected by a tide gate installation. To evaluate the utility of a SRT, this value should be compared to the increase in rearing capacity expected by other types of restoration (e.g., breaches, dike setbacks). Because both density and rearing area are expected to be reduced at SRTs compared to other types of restoration, the expectation is that SRTs are likely *at least* an order of magnitude less effective than other type of restoration. However, other constraints may make replacement of flap gates with SRTs valuable for large channel systems.

When managers opt to utilize SRTs for restoration, three additional considerations are the tide gate design, operation, and the monitoring plan. We found that connectedness of over 60% (Table 2) of the tidal cycle was feasible for side-hinged SRTs (Table 2), and we recommend that engineers strive for this standard in design and operation, which can easily be measured using automated loggers. Likewise, SRTs can be constructed with low invert heights to increase the amount of time the SRT is not dewatered while open, and multiple gates or gates set onto a headwall instead of at the end of a culvert can be used to reduce peak velocities to <0.4 m/s (Flagg and Smith 1979, Giannico and Souder 2005) through the gate. SRTs should also have mechanisms (e.g., float settings) that can be changed as necessary in response to adaptive management. Finally, our limited evaluation of SRTs with pet doors suggest that their maintenance challenges may limit their effectiveness. Although pet doors do appear to restore connectivity for anadromous fish, we observed at one site the actual removal of the pet door by strong tidal action.

Given the large sums of money often involved with restoration projects, effectiveness monitoring of projects is often warranted to verify that SRTs function as planned (Bernhardt et al. 2005). Following our temporally extensive study, we recommend designs that incorporate sampling above and below SRTs, preferably before and after installation and in association with comparisons to sites that are not affected by the installation to enable a BACI design. Our studies also identify a number of possible metrics that could be used to monitor SRTs, and reveals that many of these are temporally sensitive (e.g., Chinook salmon densities, Table 5). When funding for monitoring is restricted, low-frequency sampling of these metrics can potentially miss important patterns. In this sense, temporally insensitive metrics such as connectedness are particularly useful for assessing tide gate function. For temporally sensitive metrics, use of automated loggers for monitoring water level and temperature can be useful to provide information over relatively long time periods as long as periodic downloading of data is feasible. Even if the use of loggers is not possible, simple measurements like changes in channel openness may provide a good index of whether the tide gate design is likely improving rearing and passage. For measurement of fish densities, monitoring the entire rearing period is warranted, especially for species like salmon that have temporally variable rearing periods.

Uncertainties and questions for future study. We found substantial variation in physical and biological metrics among SRT and flap gate sites. While some of this variation can be explained by inherent geographic (system) or temporal (visit) variation that could be incorporated into the statistical design, much is likely due to variation in individual design or operation. The SRTs we examined varied from a modified flap fitted with a pet door that opened and closed with tidal fluctuations, to a highly engineered system of upstream floats connected to cams that controlled the simultaneous opening and closing of three side-hinged gates. Likewise, all tide gate sites (flap and SRT) had nonzero values of leakiness. As shown in Figures 7 through 12, individual variation did appear to affect some metrics, resulting in large effect sizes that were not significantly different between groups. This issue was particularly relevant for biological data at flap gate sites, which showed extremely high variation for some fish densities. As a consequence, this study likely is at the low end of the threshold for statistical power for some metrics, primarily due to variation around mean values. Following the rules governing statistical power (Aberson 2010), high variation can be dealt with by adding repeat visits or additional sites. For example, given the mean differences and level of significance for comparisons of flap and SRT means for anadromous and nonnative species (Fig. 10C-D), retrospective statistical power analysis suggests that the number of sites would need to have been nearly tripled or doubled, respectively, to achieve statistical significance. Although a study of that magnitude is unlikely to be done in the future, additional analysis of threshold values of connectivity that allow passage by anadromous species and utilization by nonnative species across flap gates, SRTs and reference sites is warranted.

Looking forward, these findings will likely be increasingly relevant for estuary management. Consideration of SRTs is already quite common in places like the Oregon side of the Columbia River estuary where regulations mandate fish passage, and these deliberations will likely increase as climate change impacts on sea level rise are observed. Sea level rise is predicted to reduce estuary habitat for juvenile Chinook salmon (Kennedy 1990) as slough systems become deeper, more saline, and more simplified. Restoration options will likely become fewer and SRTs will likely be increasingly seen as a compromise to provide habitat opportunity without removing dikes. Effort is needed to predict which channel systems are likely to become high-value restoration sites in the face of sea level rise, and to evaluate the alternatives to SRTs that maintain rearing habitat without compromising adjacent land use values. In addition, entrepreneurial efforts should focus on engineering or restoration designs that combine high connectivity and minimal maintenance.

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Table 1. Metrics of connectivity used in spatially and temporally extensive studies.

Connectivity metric	Definition	Dimension
Spatially extensive study		
Connectedness	% of time that the channel or tide gate door is open, and water level downstream is not more than 10 cm below the downstream invert	Temporal
Leakiness	% of total tidal flux that is opposite the direction of tidal flow when SRTs gates are closed	Horizontal
Water level	Difference of tidal height (maximum or minimum) and downstream invert	Vertical
Maximum velocity	Maximum velocity (m/s) measured at site during ebb (out) and flood (in)	Horizontal
Temporally extensive study		
Tidal muting (m)	Downstream - upstream difference in maximum higher high water (MHHW)	Vertical
Effective MHHW (m)		Vertical
Tidal muting (%)	% of downstream MHHW that is muted (downstream-upstream MHHW)	Vertical
% channel width open	% of channel width that can pass water	Horizontal
% time channel is open	% of time that the channel or tide gate door is open	Temporal
% time fish can swim upstream	% of time channel or tide gate door is open and tide is flooding	Temporal

Table 2. Sites, listed from North to South and their attributes, including site type (SRT, Flap, or Reference), the geographic system, number of SRT and passive gates (* = side-hinged gates, all others are flap gates), dimensions of the monitored culvert, total vertical cross-sectional area of gated channel (SRT and passive gated culverts combined), downstream and upstream invert elevation (NAVD88 GEOID09 Mean Low Water), mean connectedness (see text), and the upstream limit of tidal flux above SRTs. M = missing data, [1] = Side-hinged gate, [2] = Top-hinged gate, [3] = Pet door on gate, [4] = Pump house present that actively pumps water from upstream channel, and [5] = measurements are based on dimensions of the gate on its headwall, as it has no culvert.

Site name	**Site Type**	**System**	**Number of SRT Gates**	**Number of Passive Gates**	**Monitored Culvert Opening Diameter or Width x Height (m)**	**Total Area of Gated Channel (m^2)**	**Down-stream Invert Elevation (m)^2**	**Up-stream Invert Elevation (m^2)**	**Connect-edness (mean)**	**Up-stream Tidal Limit (m)**
McElroy Sl.	SRT[1]	Samish/Padilla Bay	1	3	1.8 x 1.8	13.0	0.31	0.40	0.62	1836
Edison Sl.	SRT[1]	Samish/Padilla Bay	1	6	1.84 x 1.22	20.3	0.73	0.79	0.50	3799
Edison Flap	FLAP[2,4]	Samish/Padilla Bay		3	1.21	3.4	0.15	0.20	0.05	
N. Indian Sl.	REF	Samish/Padilla Bay							0.99	
N. Fornsby	SRT[1]	Swinomish Channel	1	1	1.3 x 1.2	3.1	-0.16	-0.24	0.19	2714
Higgins Sl.	FLAP[2]	Swinomish Channel		5	1.51	9.0	-0.47	-0.20	0.14	
S. Fornsby	SRT[1]	Swinomish Channel	1	1	1.29 x 1.21	3.1	0.07	0.11	0.18	764
Old Bridge	REF	Swinomish Channel							0.80	
Fisher	REF	Skagit River							1	
Fisher Sl.	SRT[1]	Skagit River	3	2	3.35 x 2.67[5]	27.6	1.31[5]	1.31[5]	0.77	1584
Big Ditch	FLAP[2]	Skagit River		7	2.82	43.7	-0.61	-0.37	0.25	
Frye Cr.	FLAP[2,4]	Chehalis River		2	1.66	4.3	-0.24	0.52	0.07	
Alder Cr.	SRT[1]	Chehalis River	1	1*	1.88 x 1.8	6.8	-0.15	-0.15	0.51	1073
Mill Cr.	SRT[1]	Chehalis River	1	2*	1.23 x 1.12	4.1	0.12	0.12	0.52	927
Devonshire Cr.	SRT[1]	Chehalis River	1	1*	1.83 x 1.81	6.6	0.92	0.92	0.62	1690
Charley Cr.	REF	Chehalis River							0.76	
Vera Sl.	SRT[1,3]	Young's Bay	1	1	1.55 x 1.55	4.8	0.21	0.21	0.29	3667
N. Hansen Cr.	FLAP[2]	Young's Bay		1	1.23	1.2	-0.56	-0.13	0.23	
S. Hansen Cr.	SRT[2,3]	Young's Bay	1	1	1.24	2.4	-0.50	-0.07	0.14	M
S. Clatsop Sl.	REF	Young's Bay							1	

Table 3. Species captured in beach seines, their class(es) if included in a species group, and the frequency with with it was captured (at different sites, upstream or downstream sampling locations, or visits).

Species	Scientific name	Species class(es)	Frequency
Three-spine stickleback	*Gasterosteus aculeatus*		92
Staghorn sculpin	*Leptocottus armatus*	Estuarine-dependent	49
Coho salmon	*Oncorhynchus kisutch*	Anadromous	47
Chinook salmon	*O. tshawytscha*	Estuarine-dependent, Anadromous	42
Prickly sculpin	*Cottus asper*		38
Chum salmon	*O. keta*	Estuarine-dependent, Anadromous	28
Peamouth chub	*Mylocheilus caurinus*		23
Shiner perch	*Cymatogaster aggregate*	Estuarine-dependent	22
Starry flounder	*Platichthys stellatus*	Estuarine-dependent	18
Surf smelt	*Hypomesus pretiosus*	Estuarine-dependent	15
Cutthrout trout	*O. clarkia*	Anadromous	9
Snake prickleback	*Lumpenus sagitta*	Estuarine-dependent	4
Largescale sucker	*Catostomus macrocheilus*		3
Pacific herring	*Clupea pallasii*	Estuarine-dependent	3
Dungeness crab	*Metacarcinus magister*	Estuarine-dependent	3
American shad	*Alosa sapidissima*	Anadromous, Nonnative	3
Yellow perch	*Perca flavescens*	Nonnative	3
Bullfrog tadpole	*Rana catesbeiana*	Nonnative	3
Rainbow trout/steelhead	*O. mykiss*	Anadromous	2
Smallmouth bass	*Micropterus dolmieui*	Nonnative	2
Pacific sand lance	*Ammodytes hexapterus*	Estuarine-dependent	2
Pumpkinseed	*Lepomis gibbosus*	Nonnative	2
Shrimp		Estuarine-dependent	2
Eulachon	*Thaleichthys pacificus*	Anadromous	1
Crescent gunnel	*Pholis laeta*	Estuarine-dependent	1
Bluegill	*Lepomis gulosis*	Nonnative	1
Grass pickerel	*Esox americanus vermiculatus*	Nonnative	1
Rough-skin newt	*Taricha granulosa*		1
Speckled dace	*Rhinichthys osculus*		1
Unknown gadoid	*Gadidae spp.*	Estuarine-dependent	1

Table 4. Fish sampling design used in the temporally extensive study. All channels with a tide gate are denoted as treatment, while channels lacking any such structures are noted as reference. Beach seine sites are abbreviated as seines.

Area	Type	Number & type of sampling sites		# of sets per sampling site per day		Sampling period
		Upstream	Downstream	Upstream	Downstream	
Fisher	Treatment	7 seine	1 seine	1	2	February-June[1]
	Reference	1 fyke	1 seine	1	2	
S. Fornsby	Treatment	2 seine	1 seine	1	5	February-August[2]
	Reference	1 fyke	1 seine	1	3	
McElroy	Treatment	3 seine	3 seine	1	1	April-June[3]
S. Fork estuary (3 sites)	Reference	1 fyke per reference	1 seine per reference	1	2	February-August
N. Fork estuary (3 sites)	Reference	1 fyke per reference	1 seine per reference	1	2	February-August

[1]Sampling went through August in 2009 and July in 2010.
[2]Sampling periods in 2004 and 2005 were March-June and February-June, respectively.
[3]In 2011, sampling period was February-June.

Table 5. Effects of site type (Flap (F), SRT (S), or Reference(R)), system, visit or two-way interactions on physical and biological variables. Values indicate p-values (boldface indicate p< 0.05, italics indicate 0.05<p<0.1) based on a general linear model on upstream data with visit used as repeated measure. Pairwise LSD post-hoc comparisons are diagrammed in the third column, with site types ranked from lowest to highest from left to right, and connected by lines below or above letters to show types that do not significantly differ from each other, or separated by multiple spaces to indicate significant pairwise comparisons.

Variable	**Site type**	**Post-hoc**	**System**	**Visit**	**Site type* System**	**Site type* Visit**	**System* Visit**
Connectedness	**<0.001**	F S R	**0.042**	>0.1	**0.044**	>0.1	*0.082*
Leakiness	**0.009**	R S F	*0.051*	---	**0.029**	---	---
Water level (open)							
Min	>0.1	S R F	>0.1	**0.021**	>0.1	>0.1	>0.1
Max	>0.1	S F R	>0.1	>0.1	>0.1	>0.1	>0.1
Velocity (open)							
Max in	>0.1	F R S	>0.1	---	>0.1	---	---
Max out	>0.1	S R F	>0.1	---	>0.1	---	---
Surface elevation (closed)							
Min	**0.025**	F S R	**0.046**	*0.091*	>0.1	>0.1	>0.1
Mean	**0.007**	F S R	>0.1	**0.010**	>0.1	>0.1	>0.1
Max	**0.002**	F S R	>0.1	**0.028**	>0.1	>0.1	>0.1
Salinity (closed)							
Min	>0.1	R S F	**0.008**	**0.013**	**0.036**	*0.059*	**0.042**
Mean	>0.1	R F S	**<0.001**	>0.1	**0.036**	>0.1	>0.1
Max	>0.1	F R S	**<0.001**	>0.1	>0.1	>0.1	>0.1
Temperature (closed)							
Min	**0.012**	R S F	**0.034**	**<0.001**	*0.089*	**0.029**	**<0.001**
Mean	*0.062*	R F S	**0.003**	**<0.001**	>0.1	>0.1	**<0.001**
Max	>0.1	F R S	>0.1	**<0.001**	>0.1	>0.1	>0.1
Chinook	**0.028**	F S R	**0.013**	*0.068*	>0.1	>0.1	**0.001**
Stickleback	0.740	R S F	>0.1	**0.014**	>0.1	>0.1	>0.1
Estuarine spp	**0 .015**	F S R	*0.066*	>0.1	>0.1	>0.1	>0.1
Estuarine neuston	>0.1	R F S	*0.060*	---	>0.1	---	---
Anadromous spp	>0.1	F R S	>0.1	>0.1	>0.1	>0.1	>0.1
Nonnative spp	>0.1	S R F	*0.082*	>0.1	*0.063*	>0.1	>0.1

Table 6. Correlations between physical metrics and either biological metrics (Data) or their unstandardized residuals (Res.) from the GLMs. Significant correlations ($p < 0.05$) are in boldface.

Variable	**Chinook**		**Stickleback**		**Estuarine**		**Neuston**		**Anadromous**		**Nonnative**	
	Data	**Res.**	**Data**	**Res.**	**Data**	**Res.**	**Data**	**Res.**	**Data**	**Res.**	**Data**	**Res.**
Connectedness	0.21	-0.02	**-0.28**	-0.01	**0.39**	-0.01	-0.19	0.07	0.23	-0.03	-0.06	0.04
Leakiness	0.03	-0.03	-0.01	0.14	**-0.30**	0.01	-0.22	-0.04	-0.10	-0.09	-0.06	0.08
Water level (open)												
Min	0.10	-0.06	-0.14	-0.01	0.05	-0.06	-0.17	-0.12	0.19	0.20	0.18	-0.03
Max	-0.09	0.11	**-0.44**	**-0.28**	**0.28**	-0.04	0.15	-0.02	0.09	0.11	0.07	-0.16
Velocity (open)												
Max in	0.20	0.06	-0.06	0.03	0.09	0.03	-0.22	-0.16	0.25	0.21	-0.02	-0.20
Max out	0.05	-0.17	0.27	0.02	0.13	-0.16	0.01	0.24	-0.28	-0.27	0.27	0.10
Surface elevation (closed)												
Min	0.16	-0.12	-0.18	0.11	0.13	-0.03	-0.19	0.02	0.13	-0.06	-0.13	0.11
Mean	0.20	-0.13	**-0.29**	0.10	**0.31**	-0.06	-0.11	0.08	0.16	-0.08	-0.15	0.08
Max	0.17	-0.15	**-0.31**	0.11	**0.36**	-0.07	-0.13	0.06	0.16	-0.06	-0.14	0.05
Salinity (closed)												
Min	-0.11	-0.05	**0.30**	0.12	-0.18	-0.02	0.22	-0.05	-0.24	-0.05	0.04	-0.01
Mean	0.11	-0.05	**0.27**	-0.02	-0.10	-0.05	0.02	0.20	-0.08	-0.06	-0.10	-0.03
Max	0.16	-0.05	0.23	-0.06	-0.12	-0.01	-0.02	0.24	0.01	-0.01	-0.21	-0.01
Temperature (closed)												
Min	-0.08	0.04	**0.26**	0.03	-0.07	0.02	0.19	-0.22	0.08	-0.04	0.12	-0.05
Mean	0.03	-0.02	**0.28**	0.02	-0.05	0.00	-0.05	0.09	0.15	-0.02	-0.05	-0.02
Max	0.11	-0.08	**0.30**	0.05	-0.03	-0.06	-0.10	0.26	0.17	-0.06	-0.14	0.02

Table 7. Cross correlations of physical variables measured in the temporally extensive study. All correlations are significant (p<0.05).

	Tidal muting (m)	**Tidal muting (%)**	**Effective MHHW (m)**	**Channel wid. open (%)**	**Time doors open (%)**	**Cum. density ratio**
Tidal muting (%)	0.96					-0.79
Effective MHHW (m)	-0.95	-0.93				0.75
% channel width open	-0.87	-0.88	0.93			0.87
% Time doors open	-0.98	-0.92	0.93	0.84		0.65
% Time fish can swim upstream	-0.95	-0.85	0.86	0.75	0.96	0.56

Figure 1. Location of study sites by type (circle = flap, square = SRT, and triangle = reference) in five systems in estuaries across the Pacific Northwest.

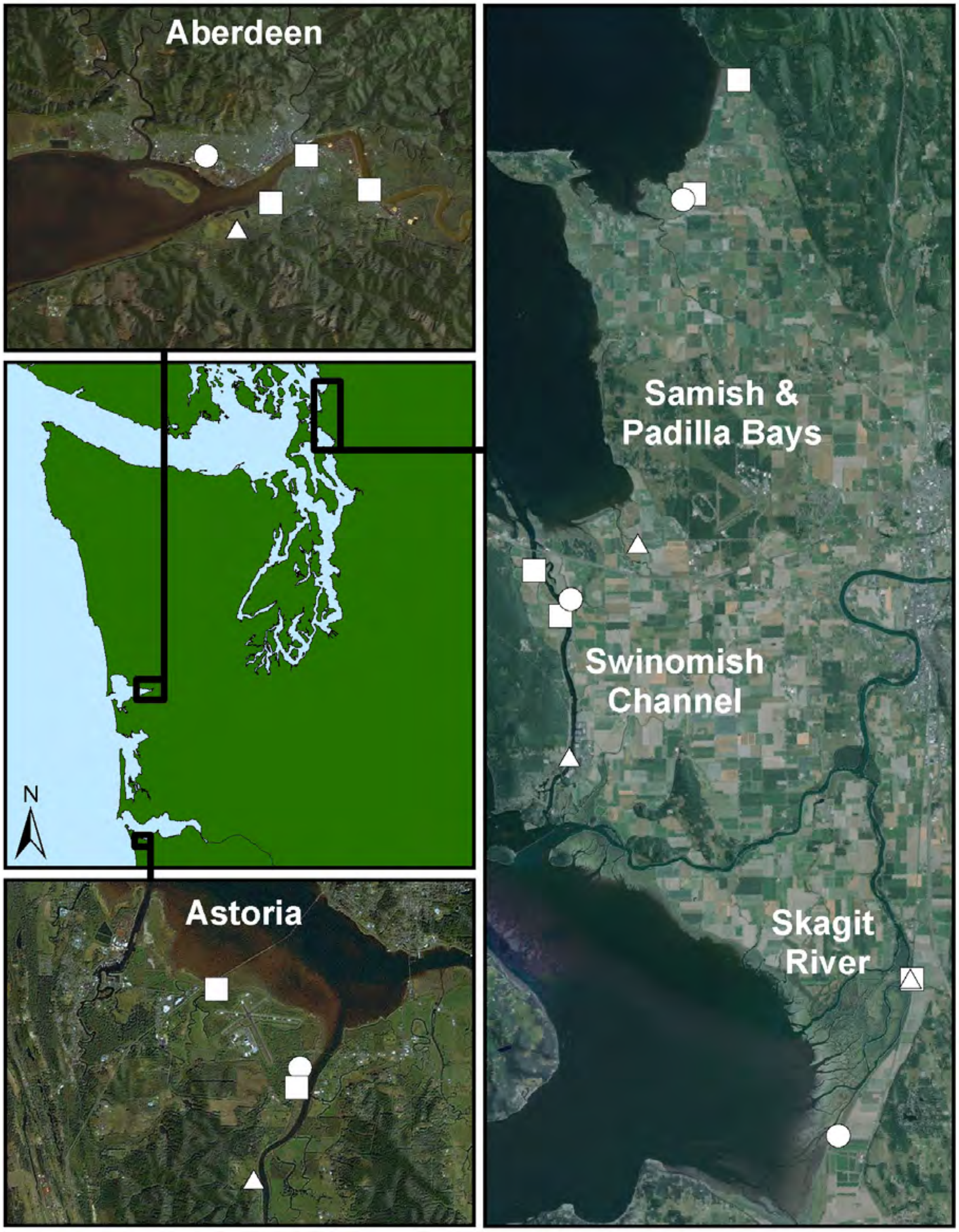

Figure 2. Self-regulating tide gates (A-J) and examples of a flap gate (K, North Hansen) and a reference site at low tide (L, Charlie Creek) used in the study design. SRTs are ordered from North to South as in Table 2. Photo credits to Jason Hall, except A (Brittany Jones), B (Oscar Bunting), C and E (Dana Rudy), D (Todd Mitchell), H (Kirsten Weinmeister), and I (Curtis Roegnor).

Figure 3. Physical measurements collected at sites. This figure depicts a cross section of an SRT site with gate and rectangular culvert (with supporting dike fill material or bridge construction removed) connecting an "upstream" channel with its estuary "downstream". Measurements are: 1) angle logger recording whether tide gate was open or closed, 2) and 3) stand pipes for loggers collecting downstream and upstream water level, salinity, and temperature data, respectively, 4) invert elevation of culvert measured at low tide, and 5) velocity probe that measured water flow and direction as well as culvert water level (dotted line). Flap gates and reference sites were also measured for these attributes as appropriate.

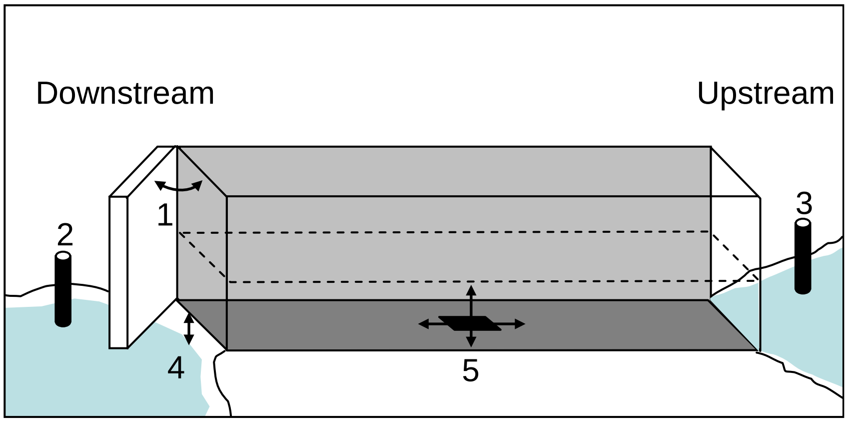

Figure 4. Water velocity (thin line, right axis) and culvert water level (thick line, left axis) within a nonleaky (A) or leaky (B) SRT over the course of a tidal cycle, including the period of time the gate was closed (gray).

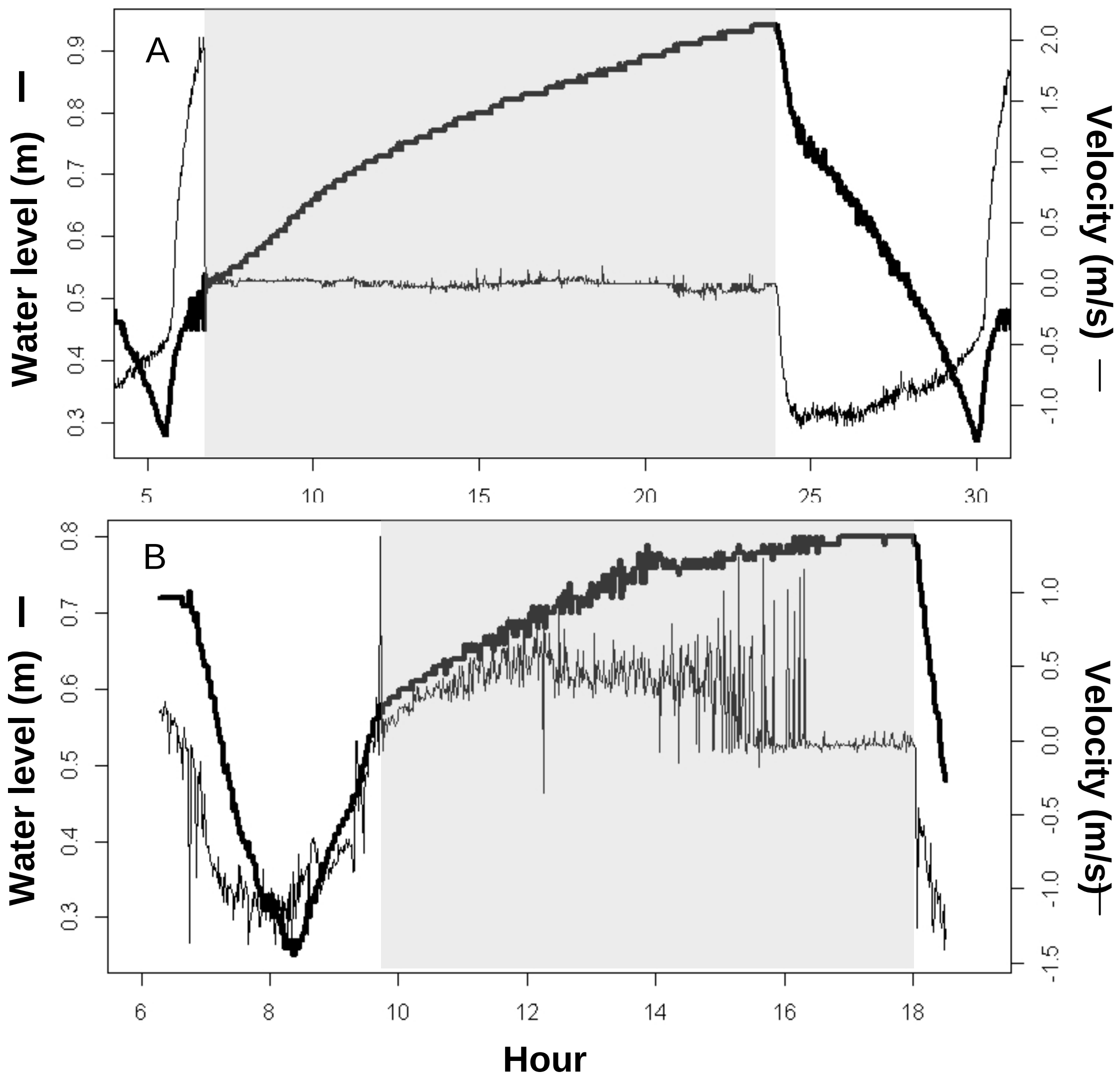

Figure 5. Sites used in the temporally extensive study. Boxes are SRT sites and triangles are reference sites.

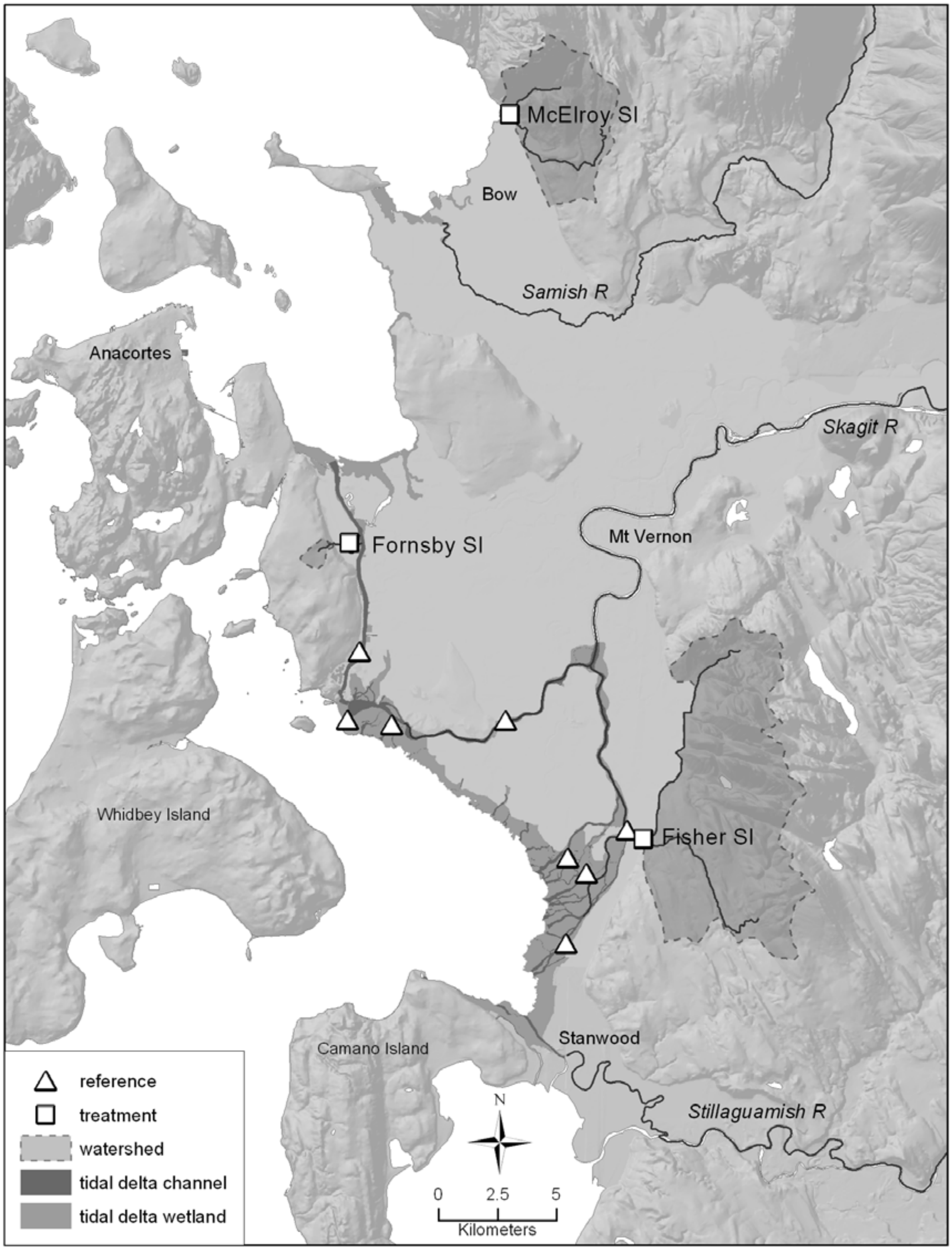

Figure 6. Examples of how water level, temperature, and salinity vary below (thin lines) and above (thick lines) across a single day at three different site types within one system (Samish/Padilla Bay). Grayed areas indicate periods of time when tide gates were closed.

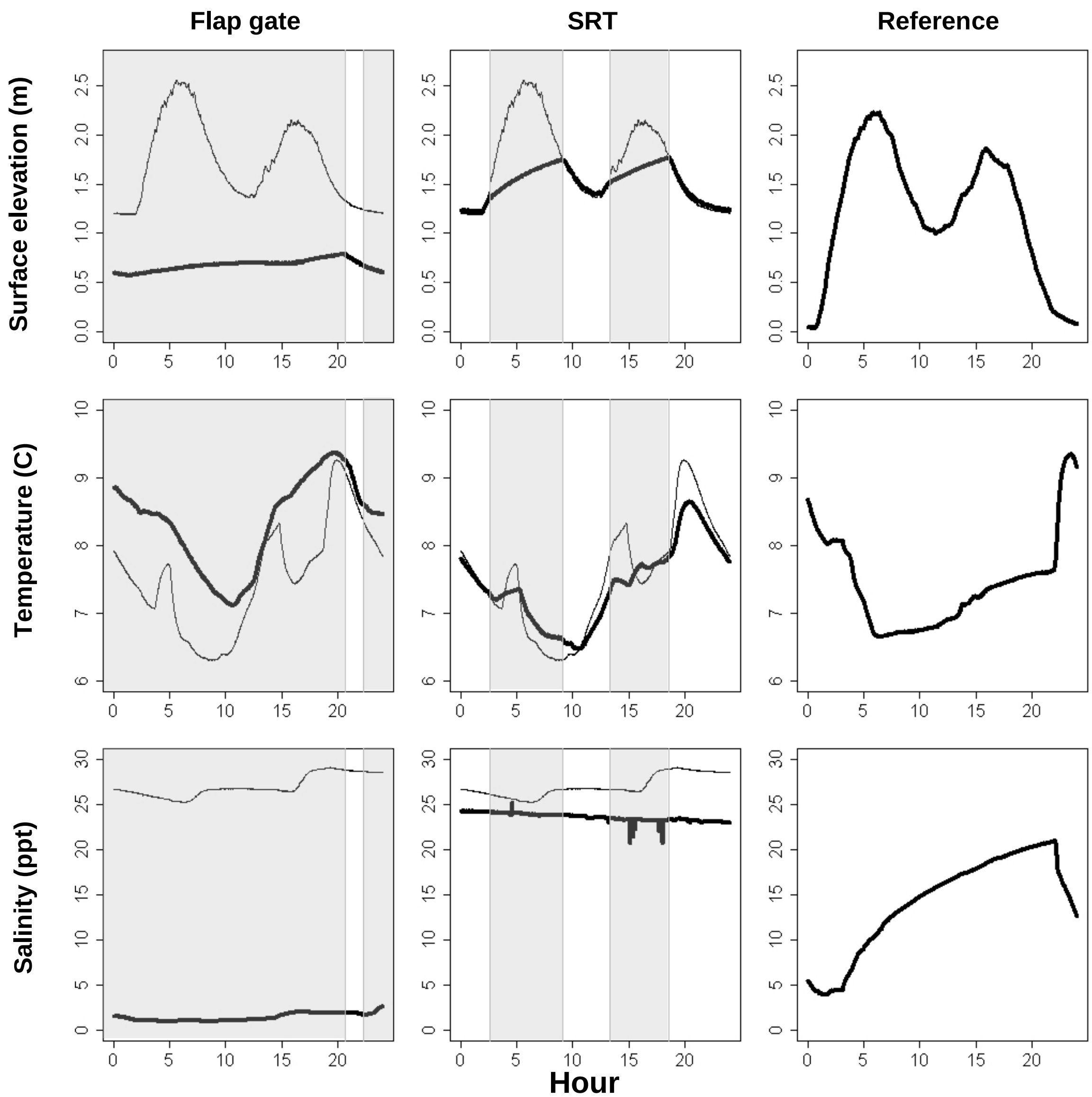

Figure 7. Metrics influencing connectivity at each site type. A) connectedness (proportion of time site is accessible and inside-outside water level < 10 cm), B) maximum (filled diamonds) and minimum (filled circles) water level relative to invert height downstream, C) leakiness (see text for definition), and D) maximum velocity of water moving out of (positive scores, filled diamonds) or into (negative scores, open diamonds channel. Error bars indicate ± 2 standard errors.

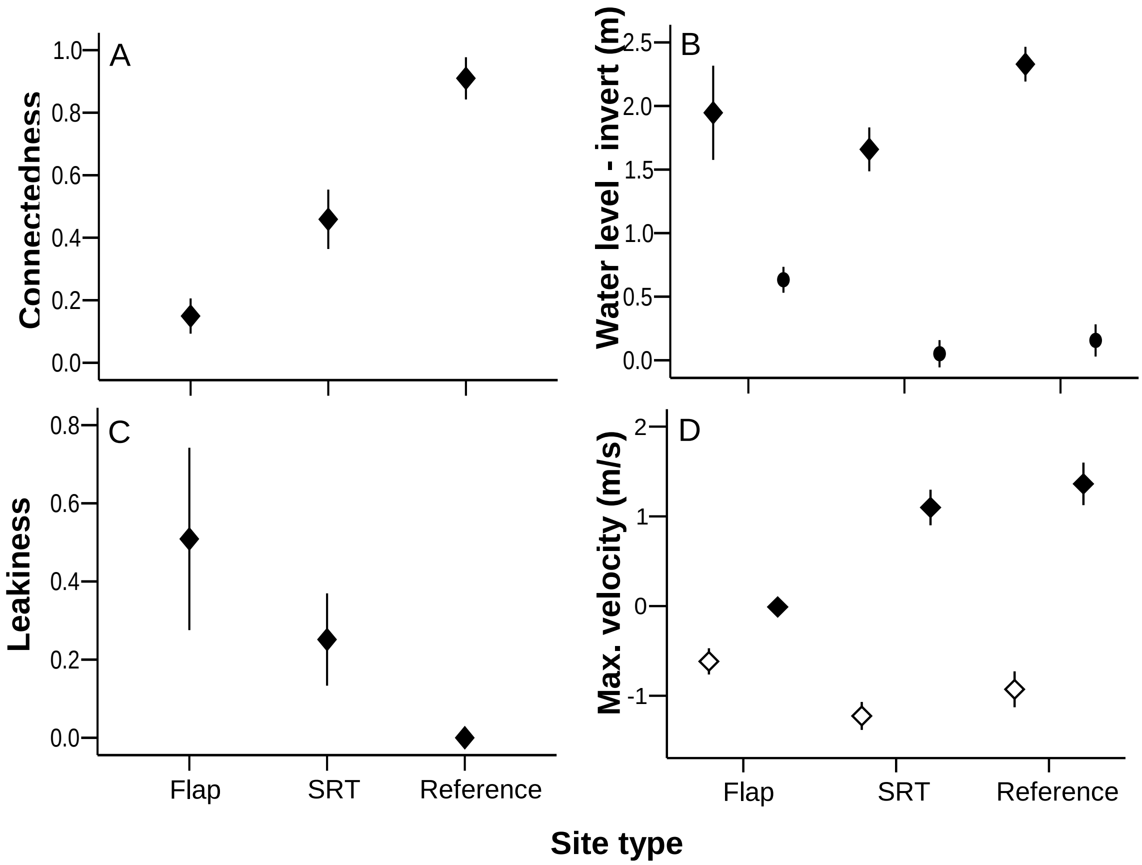

Figure 8. Mean values of average, maximum, and minimum water level, salinity, and temperature by site type downstream (open diamonds) and upstream (filled diamonds) of tide gates. Reference sites were measured at only one place, and indicated as upstream. Average values are represented by the diamonds, and maximum and minimum are indicated by the endpoint of the error bar.

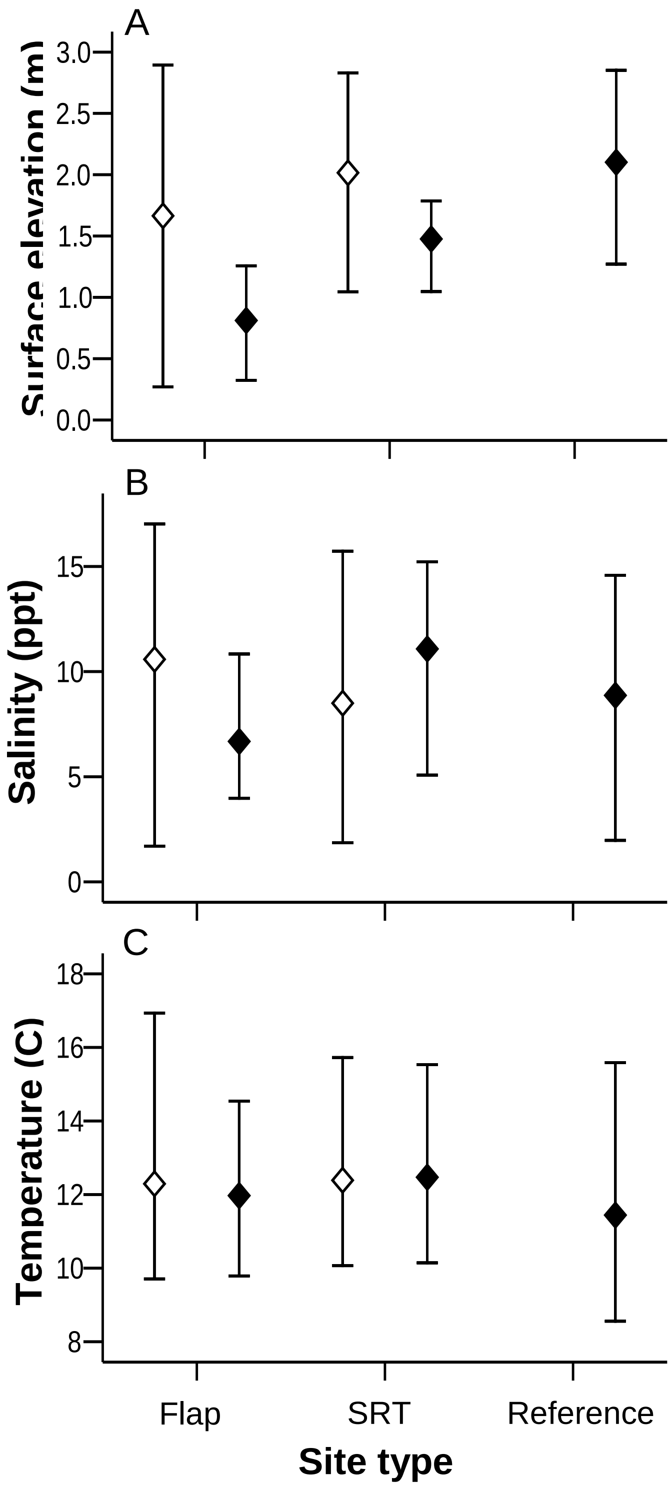

Figure 9. Mean differences (± 2 standard errors) in relative density of A) Chinook salmon and B) three-spine stickleback upstream (filled diamonds) and downstream (open diamonds) of flap gates, self-regulating tide gates (SRT), and reference sites. Densities for a given site are represented relative to its reference site and are log base-10 transformed (see methods).

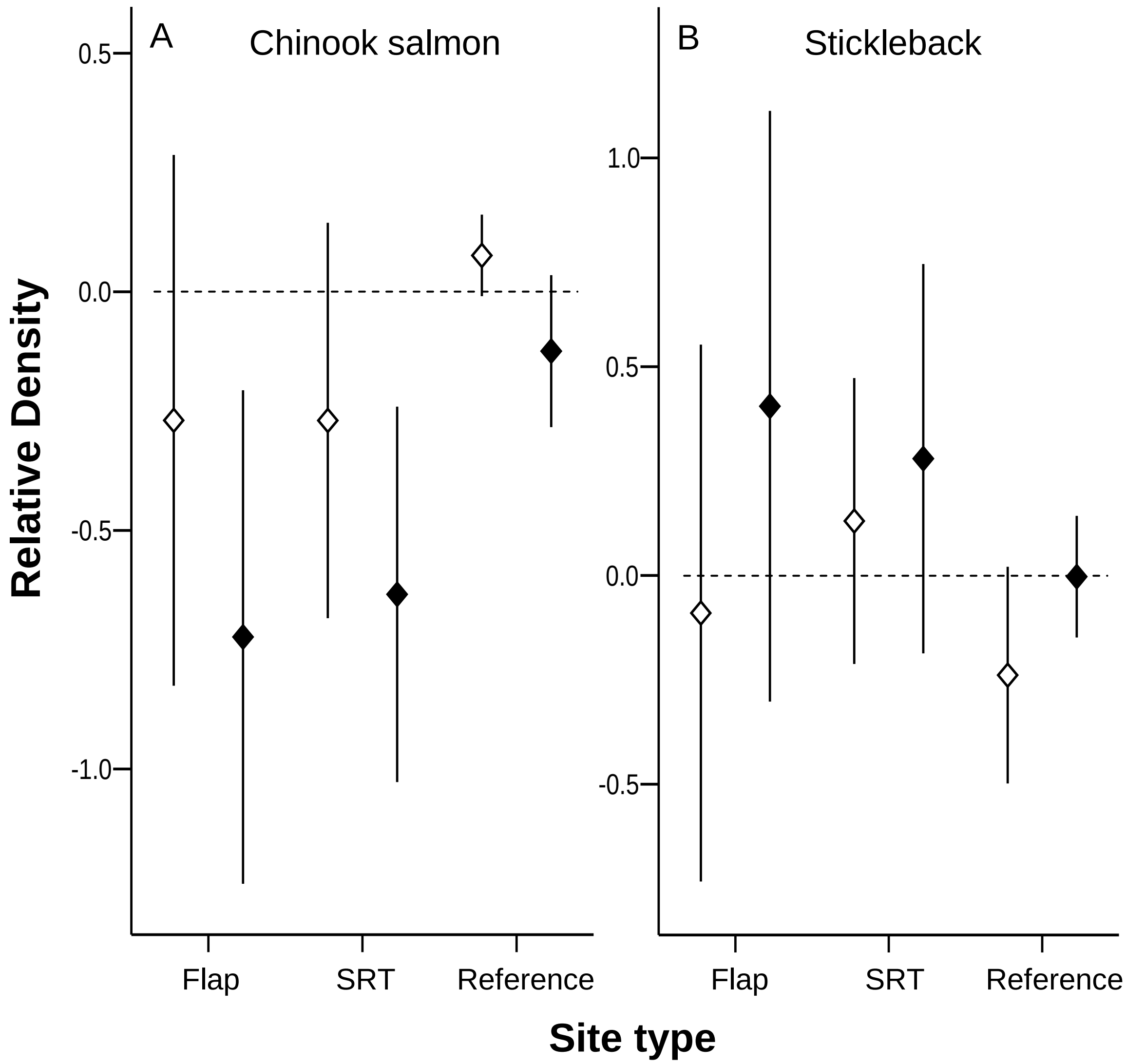

Figure 10. Mean differences (± 2 standard errors) in relative density of A) estuarine dependent species B) estuarine neuston, C) anadromous species, and D) nonnative species upstream (filled diamonds) and downstream (open diamonds) of flap gates, self-regulating tide gates (SRT), and reference sites. Densities for a given site are represented relative to its reference site and are log base-10 transformed (see methods).

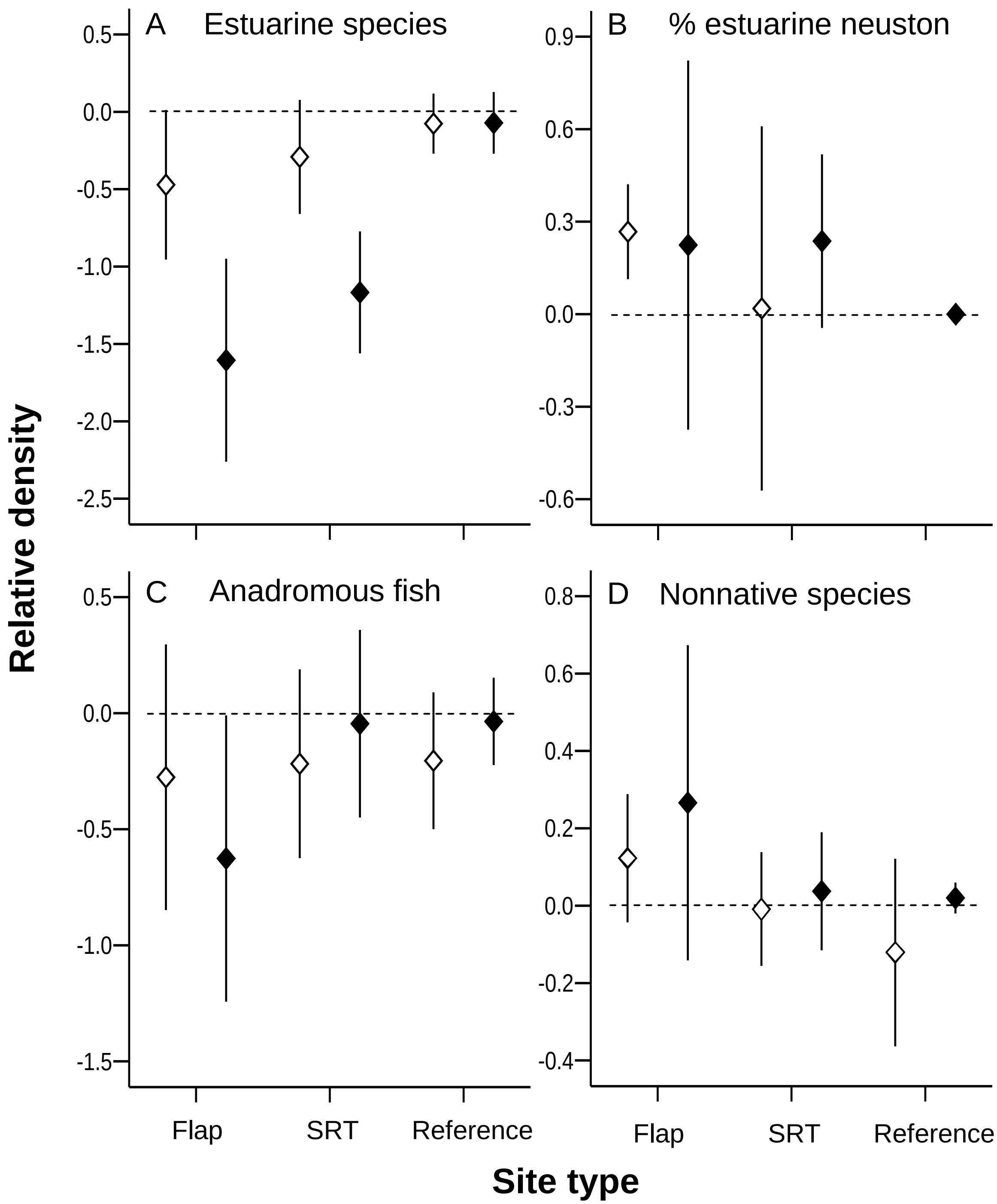

Figure 11. Ratio of cumulative Chinook salmon density (inside/outside) at A) Fisher Slough monitoring sites, B) S. Fornsby monitoring sites, C) McElroy Slough, and D) Skagit estuary refrence sites (SF = South Fork, NF = North Fork) , referenced by year and/or variation in operation. Dark gray and white bars are treatment and reference sites, respectively. In A and B, Man. = Manual, S-hinge = Side-hinge, and Ref. = Reference. Error bars are ± 1 standard error. Note that the range changes on the y-axis among panels.

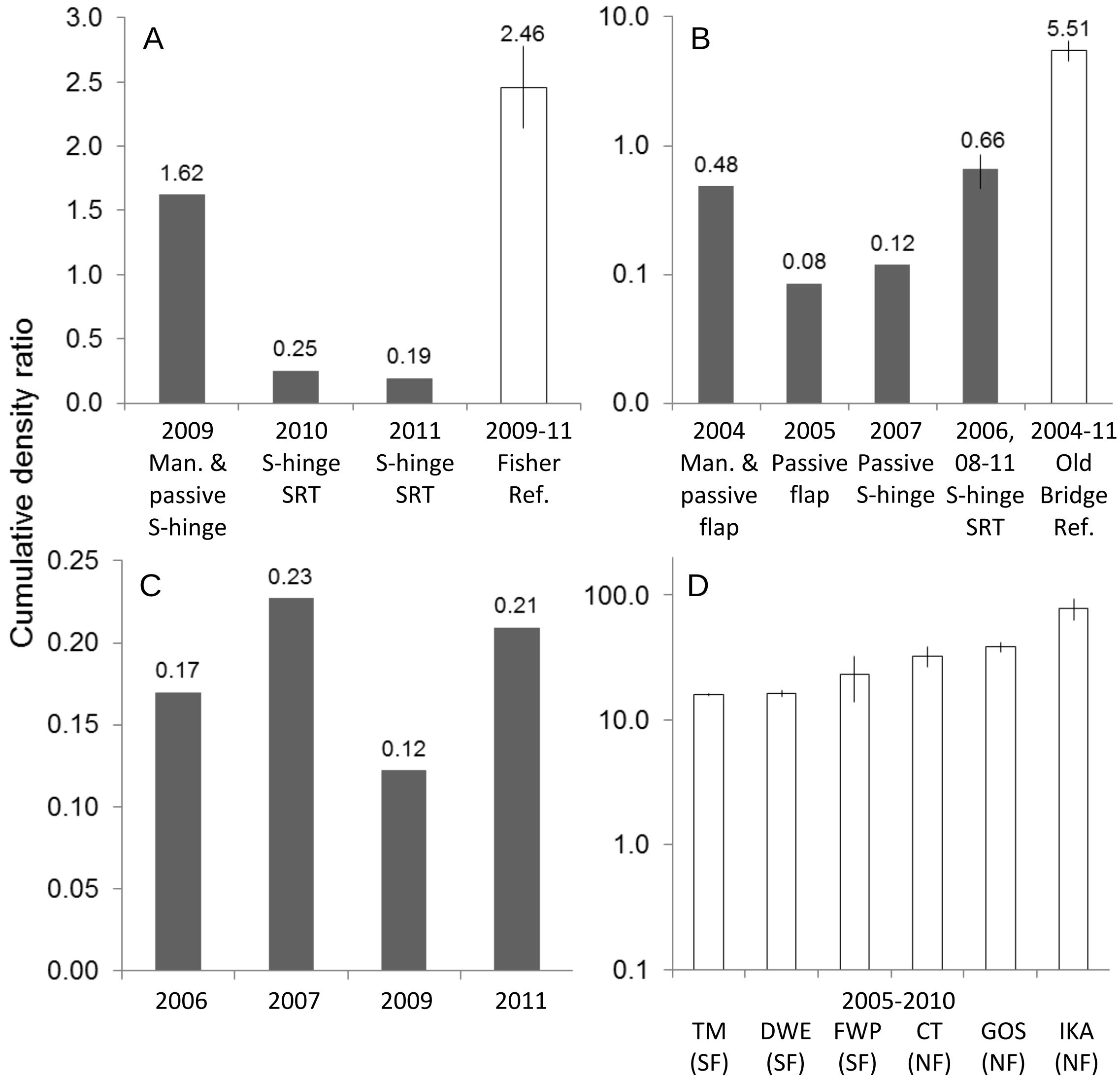

Figure 12. Relationships between cumulative Chinook salmon density (inside/outside) and A) proportion of the channel that is open B)effective mean high high water (MHHL) above sea level (NAV88), for sites varying in operation or type (treatment versus reference).