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# Hydrodynamic and ecological assessment of nearshore restoration: A modeling study

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ABSTRACT

Along the Pacific Northwest coast, much of the estuarine habitat has been lost over the last century to agricultural land use, residential and commercial development, and transportation corridors. As a result, many of the ecological processes and functions have been disrupted. To protect and improve these coastal habitats that are vital to aquatic species, many projects are currently underway to restore estuarine and coastal ecosystems through dike breaches, setbacks, and removals. Understanding site-specific information on physical processes is critical for improving the success of such restoration actions. In this study, a three-dimensional hydrodynamic model was developed to simulate estuarine processes in the Stillaguamish River estuary, where restoration of a 160-acre parcel through dike setback has been proposed. The model was calibrated to observed tide, current, and salinity data for existing conditions and applied to simulate the hydrodynamic responses to two restoration alternatives. Model results were then combined with biophysical data to predict habitat responses within the restoration footprint. Results showed that the proposed dike removal would result in desired tidal flushing and conditions that would support four habitat types on the restoration footprint. At the estuary scale, restoration would substantially increase the proportion of area flushed with freshwater (<5 ppt) at flood tide. Potential implications of predicted changes in salinity and flow dynamics are discussed relative to the distribution of tidal marsh habitat.


## 1. Introduction

Human activities and economic development in the coastal region have put extreme pressure on coastal ecological systems and have resulted in significant loss of coastal habitat (Rupp-Armstrong and Nicholls, 2007; Hood, 2004a,b; Schindler et al., 2003; Platvoet and Pinkster, 1995). To protect and sustain coastal ecological resources, many restoration projects are underway in Puget Sound, Washington, USA, and other coastal areas. To evaluate ecological response under different restoration scenarios, in-depth understanding and accurate prediction of the physical processes and the implications for restoring estuarine structure and function are necessary.

Ecological processes in coastal and estuarine areas strongly depend on dynamic physical processes, such as water surface inundation, residence time, circulation and salinity variation. There have been many ecological-assessment studies for restoring coastal habitat based on observed physical parameters, such as water depth, salinity, inundation frequency and duration. Such an approach is often limited by the availability of observed data and may not have adequate temporal and spatial resolutions to forecast changes. Numerical modeling has been demonstrated as a useful and efficient tool to predict ecological change under varying physical processes and hydrological conditions and to provide guidance for estuarine and coastal restoration projects (Boumans et al., 2002; Roman et al., 1995; Silvestri et al., 2005; Yang et al., 2008). Day et al. (1999) developed a wetland elevation model designed to predict the extent to which increasing rates of sea level rise will affect wetland sustainability. The model integrates the effects of long-term processes to determine wetland sustainability in conjunction with measured short-term rates of soil elevation change. Richards et al. (2004) developed an individual-based model to evaluate the effect of hydrologic changes, such as increased water depth and salinity, on the habitat of endangered American crocodiles in the Everglades restoration. Ahn et al. (2007) developed a dynamic ecological model to predict the recruitment and seedling growth of black willow in the Illinois River in response to different flood

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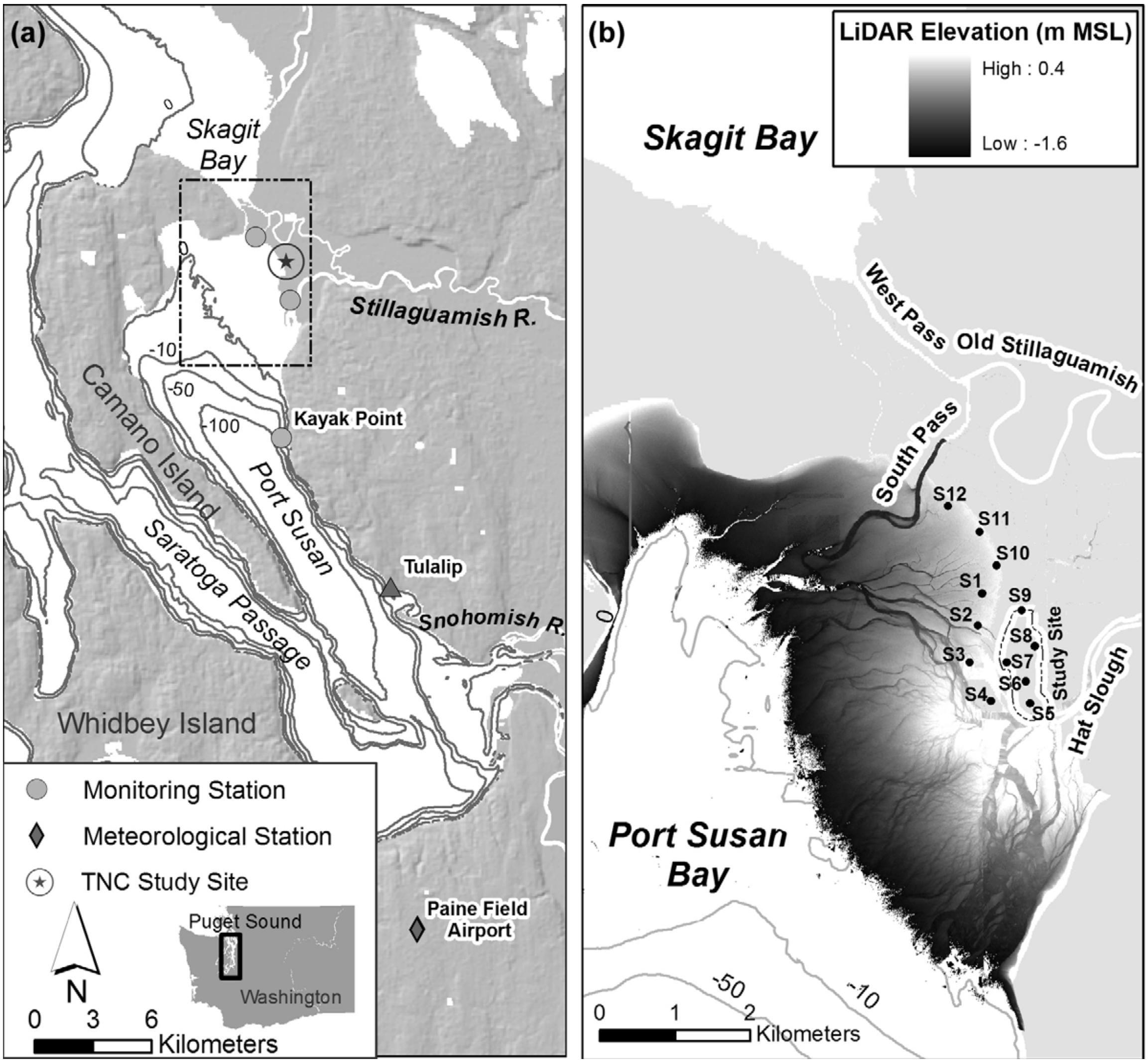


**Fig. 1.** Port Susan Bay in Puget Sound, Washington.

hydrologic conditions, such as water level, duration, and frequency. This study showed that a model is a useful tool for exploring the effects of different alternatives in ecological habitat restoration and management. By combining a model of physical processes with existing ecological data to evaluate multiple alternatives and predict ecological outcomes, one can substantially increase the power of the modeling process in informing restoration decision-making.

When The Nature Conservancy began long-term planning for managing the Port Susan Bay Preserve, which encompasses much of the Stillaguamish River estuary in Port Susan Bay, the need for a modeling assessment became critical. Modeling tools were selected to aid in understanding ecological structure and the likely response under different restoration scenarios. The Preserve encompasses a 160-acre diked area at the mouth of the river that currently provides non-tidal habitat for water birds but also constrains the distribution of freshwater, hydraulic energy, sediment, and other materials to much of the estuarine system. The 160-acre parcel exists in the fresh-to-brackish tidal marsh zone of the estuary, which is a habitat type that has experienced substantial historic losses due to diking. If restored, the diked parcel could provide productive habitat for many fishes, including threatened Puget Sound Chinook salmon, and other species that rely on tidal marshes during their life cycle. More importantly, removing the dike constraint at the mouth of the river could improve estuary-scale ecological processes.

In this paper, we present an approach for assessing ecological response to a habitat restoration project in the Stillaguamish River estuary. Water level, salinity, and surface velocity results from a hydrodynamic model are linked with locally observed ecological conditions in emergent tidal wetlands to predict the habitat types likely to occur in the restoration site. Predictions are presented for the preferred restoration alternative that was chosen, based in part, on the model results.

## 2. Methods

### 2.1. Study site description

Port Susan Bay is located in the Whidbey Basin of Puget Sound, Washington, USA. It supports one of the most productive estuaries in Puget Sound, providing important habitats for a variety of fish and wildlife. However, the Stillaguamish River has experienced significant habitat loss and degradation due to human activities in the watershed. By the mid-1900s, more than 85% of the historical emergent wetlands were converted to agricultural uses after constructing dikes and drainage systems (Collins, 1997). Estuary restoration has been identified as a priority strategy for recovering threatened fish species and improving the viability of other estuarine conservation targets in the Stillaguamish River and Port Susan Bay (SIRC, 2005).

The primary freshwater input to Port Susan Bay is the Stillaguamish River, which enters through Hat Slough and South Pass (Fig. 1). The Stillaguamish River originates on the western slope of the Cascade Mountains of Washington and has two major forks, the North and the South Forks, which merge 29 km from the mouth. The mainstem of the river splits into the Old Stillaguamish River Channel and Hat Slough, 4 km from the mouth; Hat Slough carries most of the river flow directly into Port Susan Bay. The Old Stillaguamish River Channel, which meanders to the northwest, further splits into the South Pass and West Pass branches. South Pass carries most of the flow from the Old Stillaguamish River Channel and enters at the north corner of Port Susan Bay, and West Pass connects to Skagit Bay, which is located north of Port Susan Bay. Tidal marshes and flats span the mouths of Hat Slough and South Pass, and most of the northern region of the bay is above the mean lower low water (MLLW) line. The deepest region in the bay is about 120 m below NAVD 88, located near the southern opening of the bay.

The 160-acre restoration site is located north of Hat Slough in the heart of the Stillaguamish River estuary. The site was diked and drained in the late 1950s and actively farmed through the 1990s. Following the Conservancy's acquisition of the property in 2001, water levels have been managed to provide non-tidal wetland habitat for shorebirds and waterfowl. The resultant 80-acre non-tidal wetland includes open water and vegetated habitats; major plant species include *Cotula coronopifolia* (brass buttons), *Schoenoplectus maritimus* (maritime bulrush), *Agrostis* spp. (bentgrass), and *Salicornia virginica* (pickleweed). The remaining upland acreage is dominated by *Agrostis* spp. and *Agropyron repens* (quackgrass). The ground surface within the restoration site is up to one meter lower in elevation than the adjacent intertidal marshes because of subsidence.

### 2.2. Data review

Several types of data were needed for this study, including bathymetric, hydrological, meteorological and oceanographic data for the hydrodynamic model; and topographic, environmental, and vegetation data for the habitat predictions. These data sets are reviewed in the following sub-sections.

#### 2.2.1. Bathymetric data

Bathymetry data were required to develop the hydrodynamic model grid. Bathymetry data for Port Susan Bay were obtained from the University of Washington's Puget Sound Digital Elevation Model (DEM). The DEM data have a spatial resolution of 9.1 by 9.1 m. The water depths in Port Susan Bay were extracted from the DEM data (Fig. 1a). While the DEM data were sufficient for representing the overall bay area, they did not have sufficient accuracy in the nearshore region and could not resolve the complex bathymetric features in the tideflats near the mouth of the Stillaguamish River. Thus, light detection and ranging (LIDAR) data were used to achieve the necessary resolution. Fig. 1b shows the tideflat elevations in the intertidal zone as interpolated from the LIDAR data. The multiple tidal channels near the mouths of the South Pass and Hat Slough branches of the Stillaguamish River are clearly resolved in this LIDAR data set. Channel cross-section data provided by Snohomish County Surface Water Management Division were used to define the river bathymetry in the river sections of the model. All bathymetry data used in this study were referenced to mean sea level (MSL).

#### 2.2.2. Hydrological data

The hydrological runoff data from the Stillaguamish River were obtained from the nearby United States Geological Survey (USGS) and the Washington State Department of Ecology (Ecology) gages. Total river inflow was calculated based on the sum of river flow recorded at the USGS gage in the North Fork branch and river flow recorded at the Ecology gage in the South Fork branch. Long-term USGS data show the mean annual flow rate of the Stillaguamish River is 96.6 $m^3/s$. Monthly averaged statistics indicate a low flow season occurs in August (24.5 $m^3/s$) and a high flow season in December (147.7 $m^3/s$). The mean river flow for the two-week period in October 2005 was about 74.3 $m^3/s$, approximately corresponding to the mean flow condition. A relatively high flow event occurred from October 17 to October 20, 2005, with an average flow rate of 147.9 $m^3/s$ which was comparable to the high flow condition. The minimum flow during the study period was 31.8 $m^3/s$ which was close to the low flow condition. Therefore, even with a short two-week period as considered in this study, the river flows cover a relatively wide range of hydrologic conditions and represent the normal flow conditions in the Stillaguamish River.

#### 2.2.3. Oceanographic data

Oceanographic data, including tidal elevation, velocity, and salinity, are required to specify model open boundary conditions and for model calibration. Tidal elevation for an open boundary condition was obtained from predicted tides at Tulalip Bay near the mouth of Port Susan Bay using the XTide program (Flater, 1996). A field-data collection program was conducted for model calibration from October 10 to October 25, 2005. Two mooring stations equipped with Inter Ocean S4 current meters and Hydrolab conductivity–temperature–depth (CTD) instruments were deployed in the main channels of South Pass and Hat Slough near the mouth of the Stillaguamish River (see Fig. 1a). Continuous measurements of tidal elevations, currents, salinity, and water temperature were conducted at both mooring stations. Additionally, continuous measurements of tidal elevations, salinity, and water temperature were also conducted at the Kayak Point station near the mouth of Port Susan Bay. Detailed discussion of the field data is presented in the model calibration section.

#### 2.2.4. Meteorological data

To account for wind-induced currents, wind speed and direction data were used to specify wind stress at the water surface in the model. In this study, wind data were obtained from the nearby Paine Field Airport meteorological station maintained by the National Weather Service. This airport is located in Everett, Washington, which is about 20 miles south of the study area (Fig. 1a). Average wind speed during the modeling period was about 3.4 m/s. The dominant wind direction during the data collection period was toward the north. Analysis based on 30-year wind record from a nearby meteorological station maintained by NOAA National Climate Data Center indicated that the annual average wind speed is 2.8 m/s with dominant wind direction blowing from the south. Therefore, the average wind speed and the dominant wind direction during our two-week study period are comparable to the long-term average wind characteristics in the study area.

#### 2.2.5. Habitat data

Tidal marsh vegetation is structured by multiple environmental factors, including salinity, elevation, and tidal flooding regime (Disraeli, 1977; Hutchinson, 1982; Ewing, 1982, 1986) as well as soil aeration, texture, and organic content (Ewing, 1982). To predict vegetation response to restoration using results from the hydrodynamic model, we needed to characterize the range of environmental conditions under which emergent plant species currently occur in the estuary. Geo-referenced elevation, pore-water salinity, substrate class, and vegetation composition data were thus collected at more than 100 sampling points distributed across 4100 acres of northeastern Port Susan Bay.

**Table 1**
Habitat type and dominant plant species and substrate with mean elevation and porewater salinity.

| Habitat | Dominant species | Dominant substrate | Mean elevation (m) | Mean porewater salinity (ppt) |
|---|---|---|---|---|
| Eelgrass | *Zostera japonica* | Sand | 0.5 | 12.9 |
| Tideflat | *Zostera japonica* | Sand | 0.8 | 12.2 |
| Vegetated tideflat | *Schoenoplectus americanus* | Sand | 1.6 | 9.8 |
| Low marsh | *Schoenoplectus americanus* | Mixed fines | 1.9 | 7.8 |
| High marsh | *Agrostis* sp. | Mixed fines | 2.5 | 4.4 |
| Diked emergent wetland | *Schoenoplectus maritimus* | Mixed fines | 1.6 | 12.3 |
| Diked upland | *Agrostis* sp. | Mixed fines | 1.9 | 8.8 |

The approximate boundaries for seven habitat types were manually digitized in ArcGIS using georeferenced infrared aerial photographs from summer 2003. We use the term "habitat types" to refer to the main ecological classes, or biotopes, found in the Stillaguamish estuary, without reference to a specific organism. Habitat types were defined by apparent differences in intertidal elevation as well as vegetation density and composition, and included seagrass, tideflat, vegetated tideflat, low marsh, high marsh, diked emergent marsh, and diked upland. The habitat types were ground-truthed at 108 randomly distributed sampling points on July 11–17, 2005. At each point, a Leica SR1200 global positioning system (GPS) was used to record northing, easting, and orthometric elevation relative to NAVD 88 (±20 mm). Pore-water salinity in temporary wells was gauged using a digital salinity–conductivity–temperature instrument, and surface substrates were qualitatively classified using the general descriptions provided by McBride et al. (2005). Plant species and their relative dominance were documented within a 100-m$^2$ plot at each point from June 28 through August 18, 2005. Data from each habitat type were then summarized for all four parameters.

More than 60 plant species were recorded during vegetation sampling, with less than 20 species occurring commonly. As anticipated, species prevalence differed among habitats. To associate plant species with habitat types, data on the relative dominance of each were used. For each habitat, the number of sampling points at which a plant species was designated dominant (greater than 50% of the vegetation present, Table 1) or sub-dominant (between 25 and 50%) was tallied. This process resulted in a vegetation assemblage associated with each habitat type, to be used for predicting vegetation response to restoration. While species composition and diversity are important for other biological resources (invertebrates, birds, etc.), the resolution for the hydrodynamic model and habitat prediction as part of this study is not fine enough to determine responses of individual plant species.

For each habitat type, the dominant substrate class is presented (Table 1). Consistent with Ewing (1986), sediments toward the seaward direction tend to be coarser, dominated by sand with mixed fines and silt being the subdominant sediment type. It is likely that river and wave action re-suspend fine sediments that may be distributed in these regions. In general, grain size did not vary considerably across the system.

The habitat types ranged in mean elevation from 0.54 m to 2.46 m NAVD, with seagrass being the predominant low-elevation habitat and high marsh being the highest of the habitats (Table 1). Elevations in the diked habitats varied the least, likely due to their smaller acreage and history of agricultural use.

Mean pore-water salinity ranged from 4.4 ppt in high marsh to 12.9 ppt in seagrass (Table 1). Because of the direct influence of the river, these salinities are considerably lower than what might be measured (up to 30 ppt) in deeper portions of the bay. Pore-water salinities in the diked emergent wetland and upland were 12.3 and 8.8 ppt, respectively.

## 2.3. Hydrodynamic model development

### 2.3.1. Model description

The hydrodynamic model used in this study is the three-dimensional (3D) Finite Volume Coastal Ocean Model (FVCOM) developed by Chen et al. (2003). The model simulates water surface elevation, 3D velocity, temperature, salinity, sediment, and water quality constituents in an unstructured triangular grid in the lateral plane and a sigma-stretched coordinate in the vertical plane. FVCOM has been used to simulate circulation and transport processes in many estuaries, coastal waters, and open oceans. Zheng et al. (2003) applied the model to simulate the flooding/drying process and water exchange in Satilla River Estuary, Georgia. Zhao et al. (2006) studied the tidal flushing and eddy formation in Mount Hope Bay and Narragansett Bay using FVCOM. Yang and Khangaonkar (2009) applied FVCOM to study the mixing and salinity intrusion processes in a tide flat estuary in Puget Sound.

### 2.3.2. Model grid and forcing factors

An unstructured grid was generated for the Stillaguamish River and Port Susan Bay. The upstream river inflow boundary of the model was extended considerably beyond the split of the

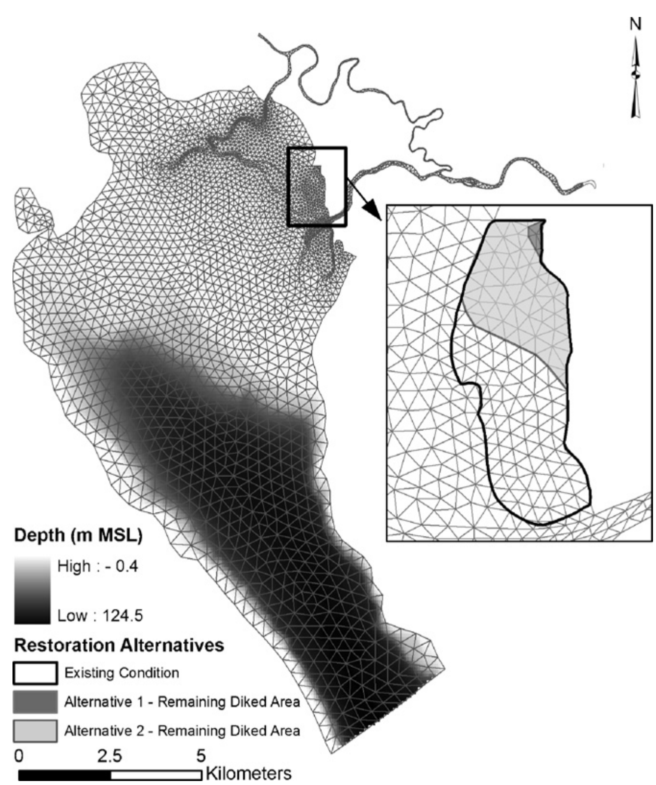


**Fig. 2.** Model grid of Port Susan Bay and restoration project site.

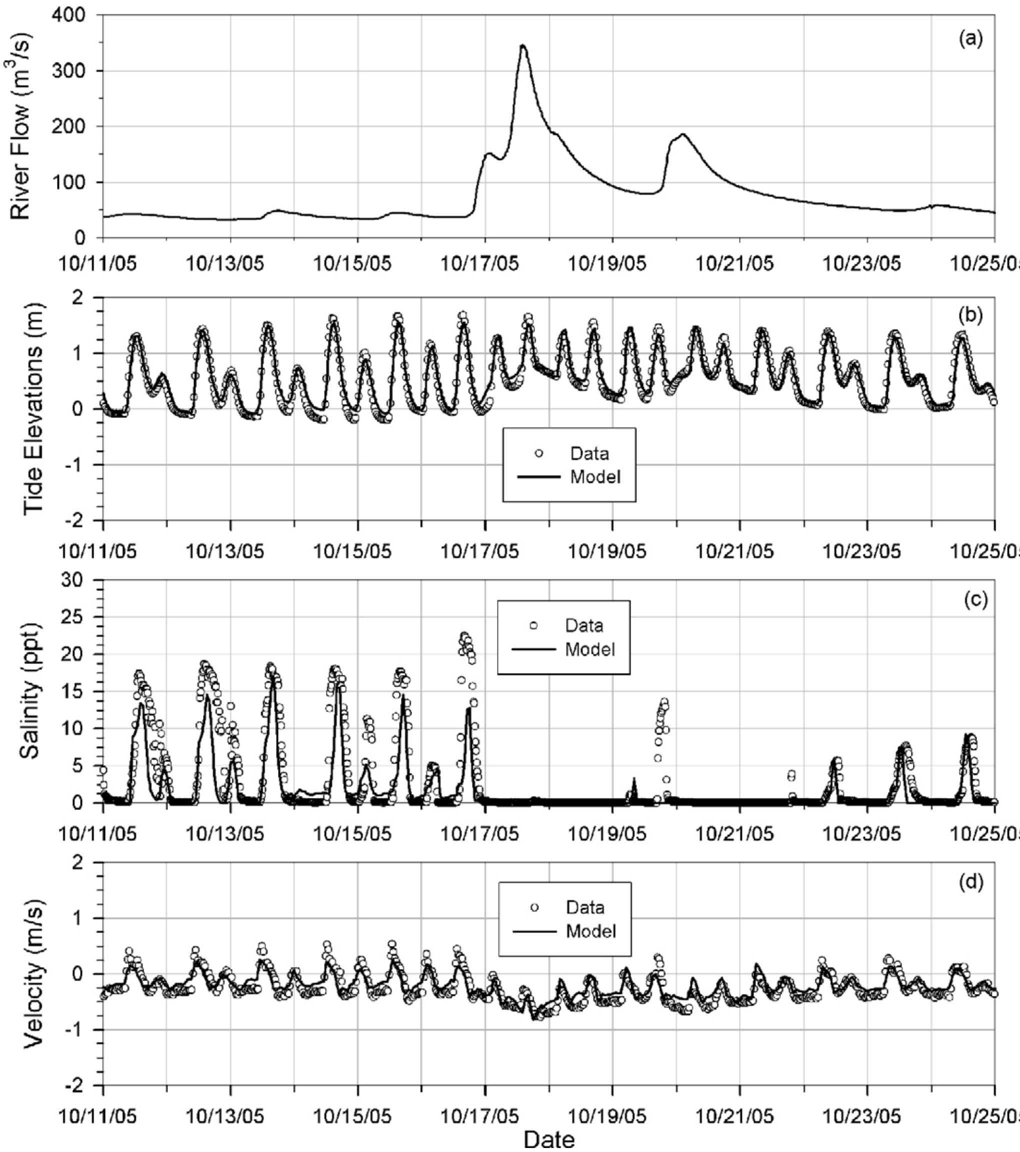


**Fig. 3.** Comparisons of predicted and observed tidal elevations, salinities, and along-channel velocities at Hat Slough Station.

Stillaguamish River into Hat Slough and the Old Stillaguamish Channel. To properly simulate the tidal wave propagation and salinity intrusion in the estuary, finer grid cells were specified in the tidal channels and intertidal zone of Port Susan Bay. The grid size varied from 30 m near the mouth of the estuary to 400 m at the entrance of Port Susan Bay. The grid resolution was gradually reduced away from the tideflats near the estuarine mouth to the open boundary to maintain the computational efficiency of the model.

The model grid consists of 5907 elements and 3352 nodes in the horizontal plane (Fig. 2). Five uniform vertical layers were specified in the water column in a sigma-stretched coordinate system. Model bathymetry was interpolated using the DEM data, LIDAR data, and river cross-section data, as described in Section 2.2.1. The model domain also included the restoration site with bed elevations along the existing dike set to 3.0 m above MSL. Water depths inside the dike were interpolated from LIDAR data, and spot measurements were recorded during habitat ground-truthing. The deepest place in the model domain is near the entrance of the Port Susan Bay, where water depth is over 120 m.

The hydrodynamic model is driven by tide, river flow, and wind stress. Tidal elevation was specified along the open boundary near the mouth of Port Susan Bay using the XTide prediction at Tulalip Bay. Salinity boundary conditions at the mouth of Port Susan Bay were initially estimated based on salinity data collected at Kayak Point. The final calibrated open boundary salinity was 30 ppt, which was consistent to the range of seasonal salinity variability in Whidbey Basin of Puget Sound (Babson et al., 2006). The Stillaguamish river inflow data were used to specify upstream model boundary conditions. At the water surface, wind stress was applied uniformly to the entire model domain.

## 3. Results

### 3.1. Model calibration

Model calibration was conducted for the period of October 10 to October 26, 2005, corresponding to the period of field-data collection. Wetting and drying processes in the intertidal zone near the mouth of the estuary were simulated in the model. Model calibration was conducted by matching model results to field data by refining bathymetry and adjusting bottom friction and open boundary salinity.

Comparisons of predicted and observed tidal elevations, along-channel velocity and salinity at Hat Slough and South Pass stations were conducted. For discussion purpose, only results at the Hat Slough station are presented because Hat Slough is the major river channel carrying freshwater to Port Susan Bay and it is closest to the restoration project site. River flow time history for the model calibration period is plotted in Fig. 3a. Comparisons of model results to observed data are shown in Fig. 3b–d. Predicted tidal elevations matched the observed data well. The absolute mean error in tidal elevation prediction at Hat Slough station is 0.0815 m. Fig. 3b shows that water surface elevations are affected by shallow water depths and river inflow as tide propagates from Port Susan Bay into the mouth of the estuary and tide flat areas. Water surface elevations during low tides at Hat Slough were elevated because of the shallow tideflat area near the river mouth. The minimum water surface elevations were about 1 m below MSL at South Pass station and about the same as MSL at Hat Slough station. Water surface elevations were further elevated during the period of high river flow from October 17 to October 21, 2005 (Fig. 3b).

Fig. 3c compares salinity time histories at the Hat Slough station. Salinity data were collected near the bottom. Simulated salinity time series were extracted at a particular vertical layer corresponding to CTD elevation as water depth varied during each tidal cycle. Overall, predicted salinities matched the observed data reasonably well. Sharp salinity intrusion during high tide was observed during the beginning of the study period. Salinities varied from 0 ppt during low tide to as high as 20 ppt during high tide in the observed data. Salinities became zero during the period of high flow (October 17–21, 2005), indicating that the salinity intrusion point was being pushed downstream of the Hat Slough mouth during high river flow and the observed station was located near the frontal zone of the freshwater plume during this time period. While the

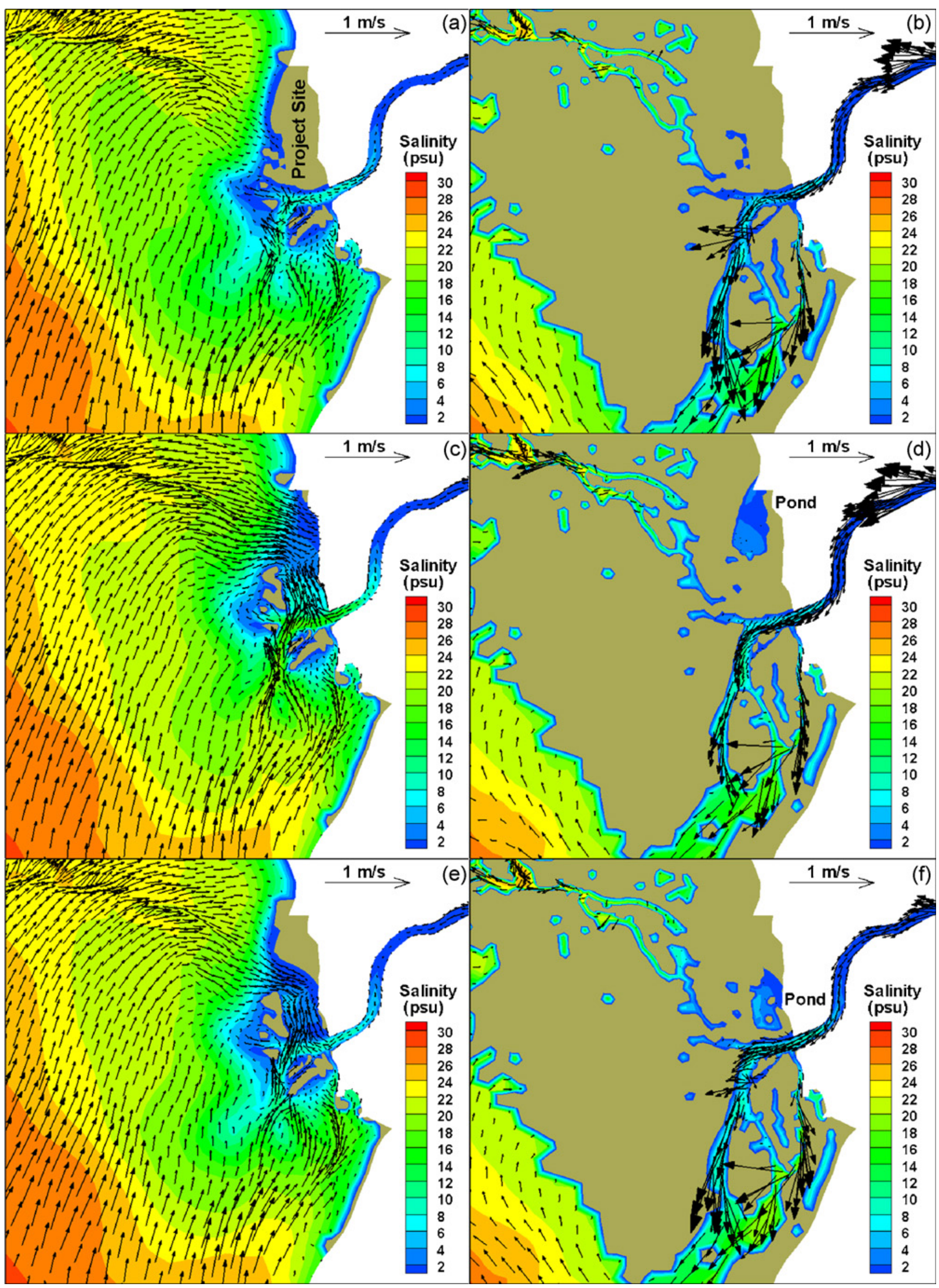


**Fig. 4.** Predicted water surface inundation, velocity and salinity distributions during flood and ebb tides under existing conditions (a and b), Alternative 1 (c and d) and Alternative 2 (e and f).

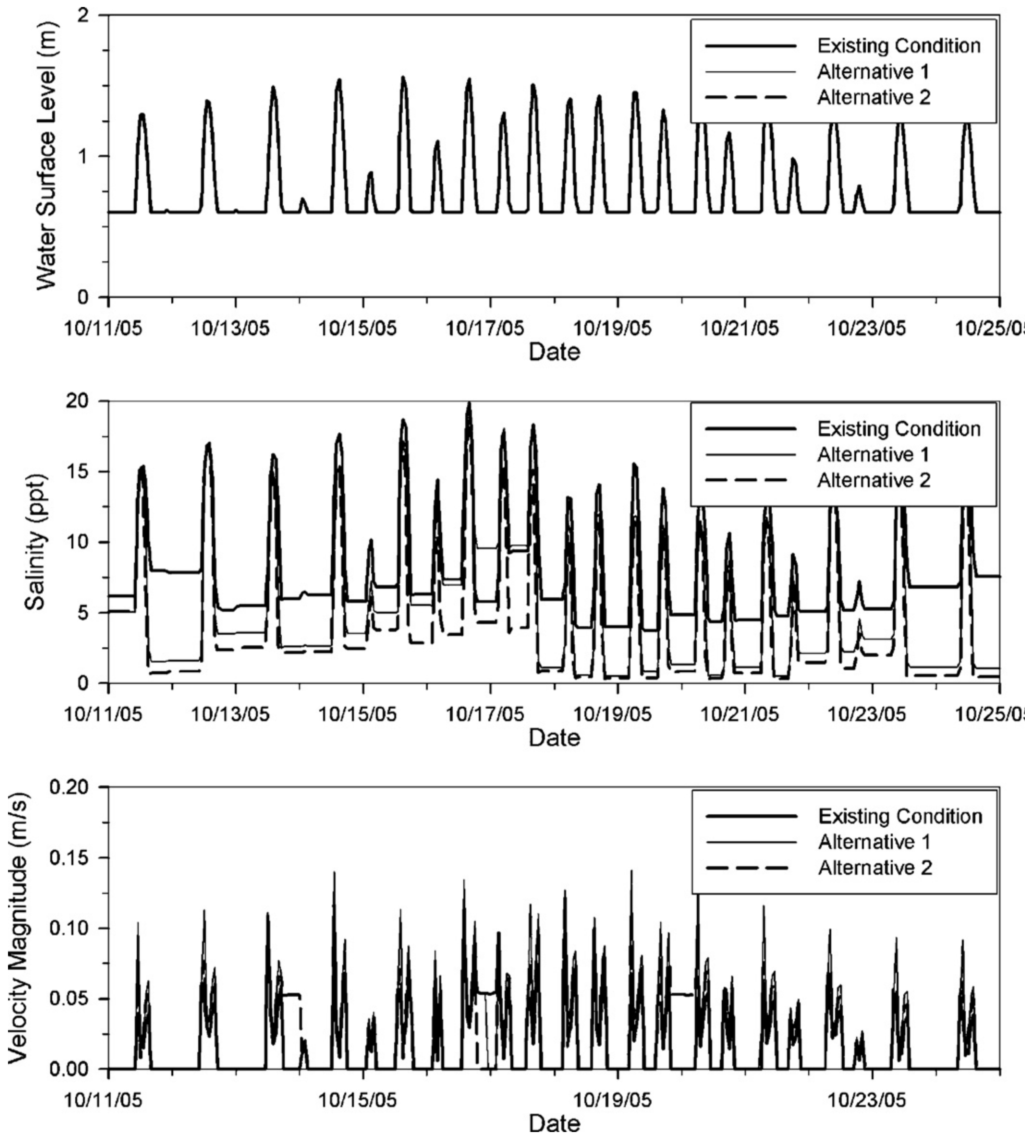


**Fig. 5.** Comparisons of water surface elevations, salinities and velocities between restoration Alternative 1 and 2 at Station S3.

model reproduced the general distribution pattern of salinity in the estuary, it tended to under-predict the high salinity intrusion during the first part of the period and failed to reproduce the single peak on October 19, 2005 as river flow dropped slightly during the four-day high flow event.

Predicted along-channel velocities matched the observed data well at Hat Slough station (Fig. 3d). Positive velocities indicated landward flow direction. Velocities in Hat Slough did not show a clear spring-neap tidal cycle because of the damping effects of river inflow and the shallow tide flats. At Hat Slough station, dominant seaward (downstream) flow was observed during the high river flow period (October 17–21, 2005), even though tidal influence was still present.

Fig. 3b–d shows that the responses of hydrodynamics to hydrological forcing (river flow) are quite fast in Port Susan Bay because of the relative small spatial scale and shallow water depths. The effect of high flow event was clearly shown in the modeled and observed water surface elevation (Fig. 3b), salinity (Fig. 3c) and velocity (Fig. 3d). Therefore, even though the modeling period was relative short in this study, the hydrological forcing actually corresponded to a wider range of conditions and the model seems to respond to such a large variation reasonably well.

Overall, the hydrodynamic model for the Stillaguamish River estuary and Port Susan Bay was calibrated satisfactorily. The model successfully reproduced many important features observed in the unique and complex estuarine system, including wetting and drying process in the intertidal zone, strong salinity intrusion in the estuary, and backwater effects in the river channels due to the shallow tide flats and high river-flow. Model calibration results indicated that the model was capable of reproducing the physical processes in the estuary and could be used to evaluate the restoration alternatives in Port Susan Bay.

### *3.2. Model simulations of restoration alternatives*

The calibrated hydrodynamic model of Port Susan Bay was applied to simulate the hydrodynamic responses of the estuary to potential restoration alternatives. Predicted hydrodynamic changes, including water surface elevations, inundation time, salinity, and surface velocities were then used to project vegetation response and habitat evolution within the restoration site.

In this study, two restoration alternatives were assessed and compared with a no-action alternative represented by the existing-condition model (Fig. 2). In Alternative 1, the dike was set all the way back to the Conservancy's eastern property line, except for a small area in the northeast corner of the existing diked area. The total area restored to river and tidal influence was 150 acres. In Alternative 2, a region of wetland in the northern portion of the site was diked off and maintained for waterbird use while the dike in the southern portion was set back to the eastern property line. The total area restored was approximately 80 acres. Bed elevations along the removed sections of the dike were reduced down to grade in both alternatives.

Predicted water surface inundation, salinity distribution, and surface velocities corresponding to high and low tides on October 15, 2006, for the existing conditions and both alternative conditions are shown in Fig. 4. Under the existing conditions, the entire estuary and bay were inundated during high tide, except for the diked restoration site north of Hat Slough (Fig. 4a). Salinity intrusion in the South Pass channel was much further upstream than in Hat Slough because most of the Stillaguamish River water flows into Hat Slough and to the bay. During low tide, a large portion of the bay becomes dry (light brown area) and the ebb flows are confined to the tidal channels (Fig. 4b). In Alternative 1, the restored 150 acres was completely inundated during high tide with salinities mostly less than 8 ppt (Fig. 4c). During low tide, some of the tidal water was trapped in the low-elevation region in the northern portion of the site and resulted in a small pond (Fig. 4d). In Alternative 2, the restored portion of the site (80 acres) was fully inundated during high tide while the diked portion was not inundated (Fig. 4e). Similar to Alternative 1, a freshwater pond resulted in Alternative 2 during low tide because of lower bed elevations due to subsidence (Fig. 4f).

To evaluate the hydrodynamic changes and the effect on estuarine habitat in response to the restoration alternatives, time histories of water surface elevations, salinities, and surface velocities near the restoration site were compared among model runs for the existing condition, Alternative 1, and Alternative 2. The 12 nodes (S1 to S12) where model results were extracted and compared are shown in Fig. 1b. These nodes fall in three regions: (1) the area just outside the restoration site (S1 to S4); (2) the area within the restoration site (S5 to S9); and (3) the area along the shoreline between the mouth of South Pass and the north end of the restoration site (S10 to S12). In this paper, only comparisons of model results at nodes S3 and S7 are presented and discussed.

Fig. 5 shows the comparisons of water surface elevations, salinities and velocities at S3 for the existing conditions, Alternative 1, and Alternative 2. The flat elevations shown in the plots during low tides correspond to a dry period. Because S3 is located outside the restoration site, water surface elevations are identical to each other for all the conditions. Salinity comparisons indicate that dike removal results in salinity reduction during ebb tide at S3 because additional freshwater is diverted to the north from Hat Slough. Velocities are small (<0.15 m/s) at S3 in the tideflat areas and quite similar under all conditions. It is important to note that model results at S1, S2, and S4 are generally very similar to S3.

Under existing conditions, all locations within the restoration site are not tidally inundated and modeled salinities are assumed to be zero. Under alternative conditions, S7 was always inundated (Fig. 6). However, water surface elevations in Alternative 2 were higher than in Alternative 1 during low tides. This is because the ponded water in Alternative 2 was retained at higher bed elevations than that in Alternative 1 (Fig. 4f) and because the marsh outside of the restoration footprint is higher in elevation. On most days, salinity in the restored site is very low (<2.5 ppt), and Alternative 1 is less than 1 ppt higher than Alternative 2, likely attributable to increased saltwater intrusion, though not likely ecologically significant. Salinity is increased during the period from October 14 to 17, 2006 due to the effect of spring tide and low river flow. Diurnal variation in salinity during this period is muted, which is likely caused by the ponding effect at this site. For reference, distributions of water surface elevations, salinities and velocities at S5 and S6 within the restored region are similar to those at node S7.

Nodes S8 and S9 were only affected in Alternative 1 because in Alternative 2 they remained within the diked area. S9 was always inundated due to the ponding effect at low tides. Unlike other locations, salinities at S9 were strongly influenced by tides propagating from the northern tidal channels connected to South Pass. Salinities varied from zero during low tides to as high as 10 ppt during high tides.

Compared to the existing conditions, model results in both alternative conditions showed no changes in water surface elevations at S10 to S12, which were located along the shoreline between the restoration site and South Pass. Model results for restoration alternatives also showed little influence on salinity distributions, which were highly controlled by tides and river flow.

### 3.3. Ecological assessment for the preferred alternative

Based largely on ecological dynamics at a larger spatial and temporal scale than is described here, Alternative 1 was selected as the preferred alternative. This alternative removed most of the existing dike and opened up a greater area to tidal inundation. Site-scale ecological change associated with the preferred alternative was then evaluated.

To develop vegetation response predictions, we related model results for the 12 nodes described in the previous section to field-collected elevation and habitat data from 43 nearby sampling points. As described previously, hourly water surface elevations were modeled for October 10–25, 2005, and time series data were provided for the 12 nodes under current and restored conditions. Water surface elevation time series and sampling point elevations were entered into Tide Miner 3.0 (Warren, 1999) to calculate current and restored inundation times for each sampling point. Note that because there was slight variation in water surface elevations across nodes, sampling point inundation was calculated using output from the closest node. Also, the hourly water surface elevations were not true tidal heights because a minimum water depth of 20 cm was specified in the model for wetting and drying simulations. To compensate for this condition and ensure dewatering at low tide, ground elevations of the 43 nearby sampling points were increased by 22 cm. Inundation times were also calculated for the modeled stations and used to determine the predicted habitat types (Table 2). Using the Tide Miner results, we constructed a predictive habitat inundation model depicting the minimum and maximum inundation times for each intertidal habitat under existing conditions. Other studies have developed vegetation models directly from elevation (e.g., Hood, 2004a,b); however, due to the geographic extent of the model and the altered hydrologic conditions (i.e., ponding) expected to occur within the restoration site, direct predictions from elevation seemed inappropriate here.

We then used our habitat inundation model to predict the types and locations of habitats within the diked area after restoration. Post-restoration inundation times for sampling points and model nodes within the diked area were compared to the model and the appropriate habitat was assigned to each. Habitat coverages were delineated around these points in ArcGIS, using 2003 LIDAR elevations to guide the final polygon shapes (Fig. 7). Note that the small elevation range (0.6 m) of sampling points inside the dike and the considerable overlap between the ranges of candidate habitat types hinder the ability to make definitive delineations. The delineations show generally where the habitat types may occur, but the actual boundaries are likely to be ill-defined. Based on the delineated polygons, we predict that high and low marsh habitats will develop on more than half of the site, where inundation times range from about 0–35%. These habitats are likely to be vegetated with the species

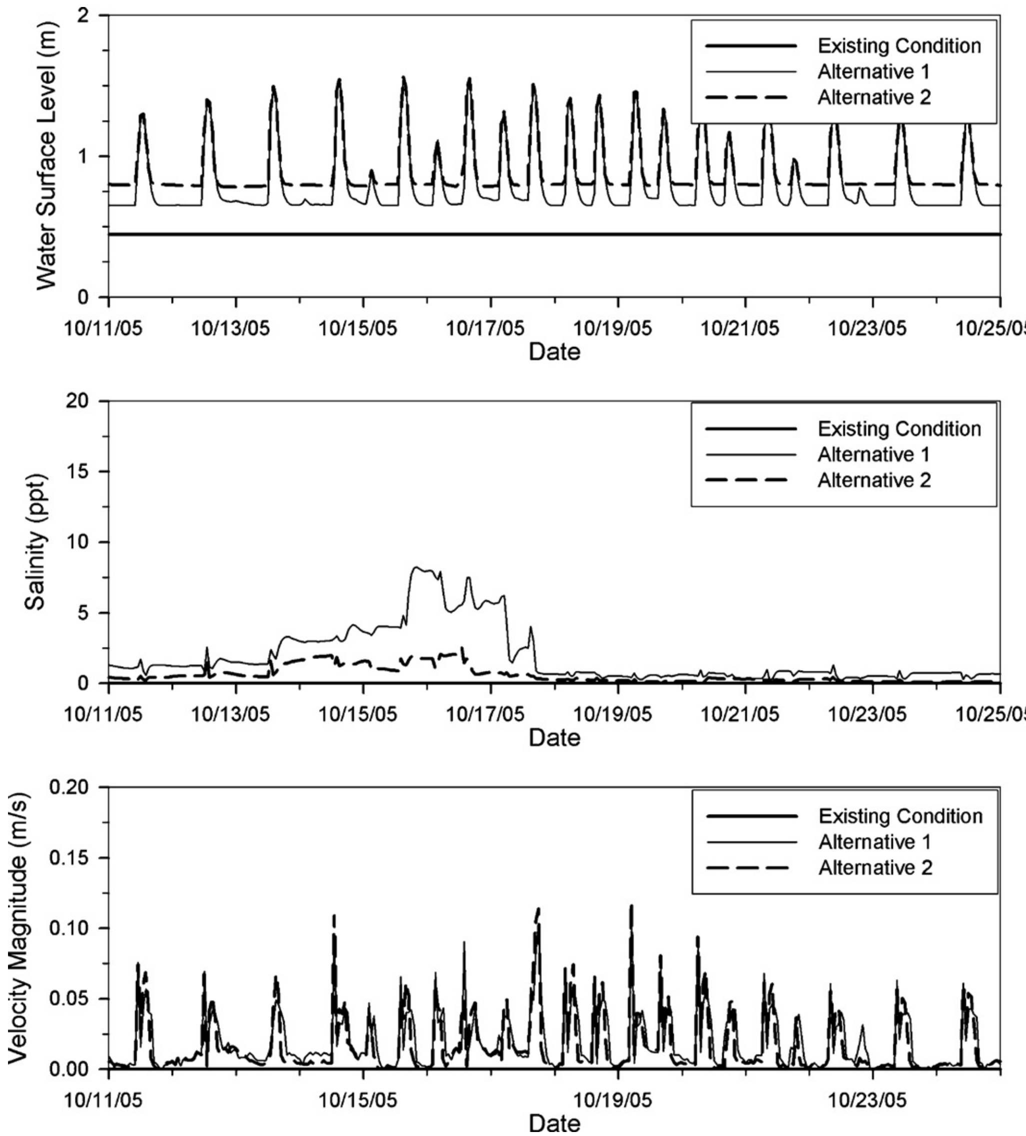


**Fig. 6.** Comparisons of water surface elevations, salinities, and velocities between restoration Alternative 1 and 2 at Station S7.

listed in Table 1, primarily *S. americanus* and *S. maritimus*, with *Carex lyngbyei*, *Agrostis* sp., and the invasive non-native *Spartina anglica* also likely components of the community. The permanently ponded area is expected to cover nearly 40% of the restored site, and it may be colonized by the non-native seagrass, *Zostera japonica*.

According to the hydrodynamic model, there will be only minimal changes in ambient-water salinity following restoration. This suggests that there will likely also be minor changes in pore-water salinity. However, the relationship between ambient-water salinity and pore-water salinity is not straightforward. Pore-water salinities can be affected by sediment type, frequency of tidal inundation,

**Table 2**
Model output stations and attributes.

| Station | Habitat type | Elevation (m, NAVD88) | Inundation time-existing (%) | Inundation time-predicted (%) | Median salinity-existing | Median salinity-predicted |
|---|---|---|---|---|---|---|
| S1 | Vegetated tideflat | 1.49 | 42 | 42 | 12.03 | 8.63 |
| S2 | Vegetated tideflat | 1.61 | 37 | 39 | 4.48 | 1.47 |
| S3 | Low marsh | 1.72 | 33 | 33 | 6.81 | 3.14 |
| S4 | High marsh | 2.24 | 14 | 14 | 1.19 | 1.75 |
| S5 | *Low marsh* | 1.92 | 0 | 26 | 0.00 | 1.03 |
| S6 | *Low marsh* | 1.85 | 0 | 33 | 0.00 | 0.78 |
| S7 | *Ponded* | 1.57 | 0 | 100 | 0.00 | 0.86 |
| S8 | *Ponded* | 1.74 | 0 | 100 | 0.00 | 0.42 |
| S9 | *Ponded* | 1.50 | 0 | 100 | 0.00 | 0.57 |
| S10 | Low marsh | 1.67 | 34 | 34 | 9.21 | 8.70 |
| S11 | Vegetated tideflat | 1.52 | 41 | 41 | 10.15 | 9.98 |
| S12 | Vegetated tideflat | 1.51 | 41 | 41 | 12.07 | 11.88 |

The italicized habitat type refers to predicted habitats inside the existing dike.

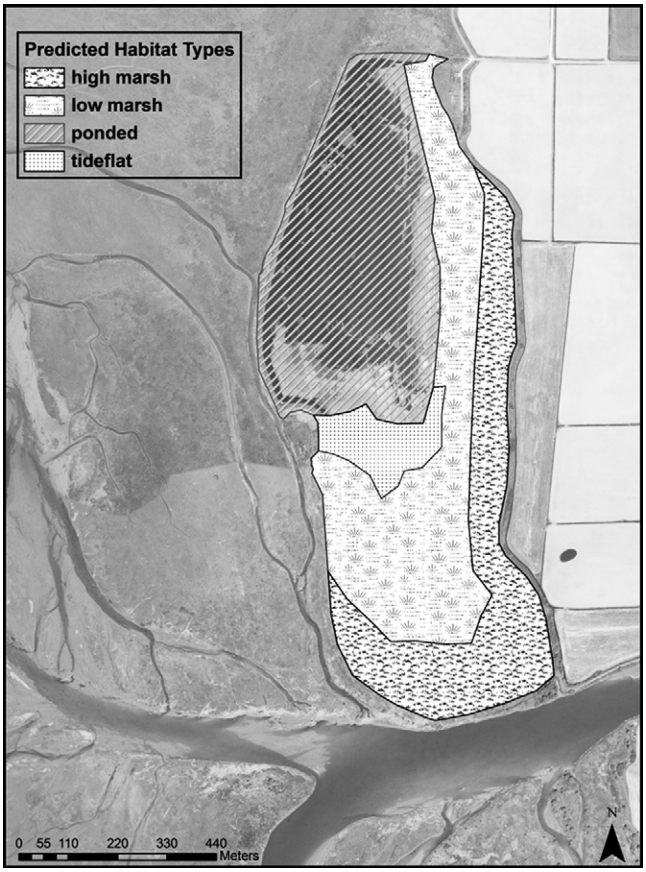


**Fig. 7.** Predicted habitats within the restoration area at Port Susan Bay.

sediment composition, and type of vegetation. Eilers (1975) documented seasonal variability due to rainfall and evaporation rates, with higher pore-water salinities occurring in late summer. Ewing (1986) showed salinity to be a stressor on marsh plant species in Skagit Bay, with plants in plots with pore-water salinity around 4 ppt producing more above-ground biomass than those in more saline plots (8–12 ppt). In addition, pore-water salinity tended to be lower than ambient salinity in that study, with the maximum pore-water salinity around 15 ppt and ambient water at 22 ppt (Ewing, 1986).

**Table 3**
Salinity tolerance for predicted species in the low marsh habitat type (Hutchinson, 1989).

| | Salinity tolerance (ppt) | |
|---|---|---|
| | Min | Max |
| Low marsh | | |
| *Schoenoplectus americanus* | 0 | 15 |
| *Schoenoplectus maritimus* | 3 | 15 |
| *Carex lyngbyei* | 0 | 22 |
| *Triglochin maritimum* | 0 | 22 |
| *Eleocharis palustris* | 0 | 12 |
| *Distichlis spicata* | 3 | 20 |
| *Atriplex patula* | 3 | 21 |
| *Spartina anglica* | No data | No data |
| High marsh | | |
| *Agrostis* sp. | 0 | 12 |
| *Carex lyngbyei* | 0 | 22 |
| *Schoenoplectus americanus* | 0 | 15 |
| *Salicornia virginica* | 6 | 20 |
| *Juncus balticus* | 1 | 10 |

Table 3 shows the estimated minimum and maximum pore-water salinity tolerances for the dominant tidal marsh species, as summarized by Hutchinson (1989) for Puget Sound and the Georgia Basin region. Ambient salinity, with model results lower than 10 ppt (and means less than 5 ppt), should not exclude any of the potential dominant tidal marsh plant species (Table 3) within the restoration site, although low salinity may be a limiting factor for establishment and survival of the non-native seagrass *Z. japonica* in the ponded portion of the restored area (S7 and S9) (Kaldy, 2006).

## 4. Discussion and summary

As part of the Port Susan Bay restoration assessment and design study, a three-dimensional hydrodynamic model was developed for the Stillaguamish River estuary and Port Susan Bay in Puget Sound using FVCOM. The model was calibrated to observed tide, current, and salinity data collected for a neap-spring tidal cycle. The model successfully reproduced the wetting and drying process in the inter-tidal zone, river plume dispersion, tidal dynamics, and backwater effect of freshwater inflow. Although the accuracy of the model results can be further improved through refinement of the model grid and increase in vertical resolution, the level of model calibration achieved was considered sufficient for the purpose of this study.

Site-scale responses to two different restoration alternatives were evaluated and based, in part, on the model results for estuary-scale processes and long-term dynamics external to the model, a preferred restoration alternative (Alternative 1) was selected. Further analysis and assessment of ecological processes and habitat evolution at the restoration site were conducted for the preferred alternative.

Based upon data collected on the existing conditions at Port Susan Bay and the predicted hydrodynamic changes for the selected restoration alternative, we have determined likely habitats and plant species occurrence for the proposed area of restoration. However, stochastic processes (e.g., flooding or redistribution of sediments) that have not been accounted for in the model are likely to determine the actual restoration outcome in the proposed area. Habitats can be subject to significant changes under extreme events, for example due to sediment transport and trapping during storms and drying of tide flats/shallow channels and increased salinities during droughts. Although wind-induced waves were not considered in this study, because wave energy generally is not strong in Port Susan Bay, wave action could result in significant inundation, salinity change, and sediment transport during storm events, especially in the tideflat region. However, the modeling of extreme events and wave action was beyond the scope of this study.

The existing dike may influence hydrogeomorphic dynamics outside the diked area, and once the dike is removed, this may lead to redevelopment of tidal channels (Hood, 2004a,b). Furthermore, the hydrodynamic simulations suggest that velocities could change in some parts of the estuary as a result of dike removal. This change in the estuary could also contribute to changes in the channel system and the distribution of sediment. Although we did not analyze the resulting marsh plain for long-term morphological changes, information on channel formation in newly restored tidal freshwater marshes indicates that remnant channels in diked marshes may proceed to function normally following tidal reconnection (Diefenderfer et al., 2008).

The proposed restoration is anticipated to result in four habitat types on the project footprint: pond, tideflat, low marsh, and high marsh. Dominant species would include *Schoenoplectus americanus* and *S. maritimus*, *C. lyngbyei*, *Eleocharis palustris*, *Distichlis spicata*, and *Atriplex patula*. The exact patterns of development will be largely dependent on flood conditions after the dike is removed, which will aid in structuring the marsh and tideflat areas. Additionally, considerable inter-annual variability is common at both restoring and reference marshes and the plant community at the restoration site would likely evolve rapidly in the first few years after the dike removal (Thom et al., 2002). It is unlikely that the project site will revert to its historical condition due to subsidence. However, the predicted habitats will continue to support birds and will also expand foraging opportunities for juvenile salmonids and other fishes in Port Susan Bay.

In summary, the use of the hydrodynamic model in combination with ecological predictions added to the understanding of how the Port Susan Bay Preserve would develop with the proposed restoration project. Because vegetation communities are structured in large part by hydrology (elevation) using existing data on vegetation occurrence in combination with the hydrodynamic model produced a more comprehensive picture of the selected restoration scenario in terms of habitat development. While a robust data collection effort is necessary for this approach, the information gained can be useful, especially if deciding between multiple alternatives or designing a restoration project with specific habitat goals in mind.

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