

Effects of intertidal water crossing structures on estuarine fish and their habitat: a literature review and synthesis

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Executive Summary

For hundreds of years, people have built water crossing structures to enable the transportation of people, livestock, vehicles, and materials across rivers and other bodies of water. These structures have often created barriers to fish passage, an issue which has recently drawn intense scrutiny due to concerns over impacts to anadromous fish. While much work has focused on the impacts of *freshwater* crossing structures, *intertidal* structures have received less attention. This may be due to the importance of passage for adult anadromous fish in freshwater, and that bidirectional flows in intertidal environments complicate interpretation of structures as barriers. Intertidal water crossing structures likely have adverse impacts on juvenile life stages of fish due not only to impacts to passage, but also to impacts to estuarine habitats extensively used by these species as rearing environments. Examining the impacts of intertidal water crossing structures only through the lens of fish passage therefore misses key aspects to how these structures can affect fish.

In this report we review literature on intertidal water crossing structures and how they affect fish that depend on intertidal habitats for passage during migration or for extended rearing during early life stages. Our findings are important for establishing fish passage criteria, providing design guidelines, and identifying key data gaps for future research of intertidal water crossing structures.

We address the six general questions in the following sections:

What are the primary intertidal water crossing structures and how do hydraulics function at these structures? (Section 2)

Classic intertidal water crossing structures include culverts, tidegates and other tidal muting structures, and causeways and bridges. These devices allow water conveyance through channels under the structures. In addition, structures with much larger footprints such as road and railroad grades as well as dikes and levees cross intertidal wetlands, thereby obstructing tidal flow across the wetland surface as well as within channels. Hydraulic engineering of structures has often focused on 1) conveying drainage through embankments, 2) attenuating the vertical range of tidal action, 2) restricting tidal action to channels, and 4) reducing channel size.

What habitat types and natural processes are affected by structures? (Section 3)

There are four primary habitat types (shoreforms) within which structures are likely to impact intertidal physical processes and biotic communities: (1) large river deltas, (2) coastal creek confluences, (3) coastal embayments, and (4) lagoons. While other shoreforms comprise

nearshore ecosystems, these four types are most sensitive to losses in connectivity and associated impacts to habitat quantity, habitat quality, or biotic communities. These four shoreforms share landforms and vegetation types at a smaller spatial scale, which are the common habitat types that fish utilize. The occurrence and distribution of intertidal habitats are outcomes of several geomorphic processes. Historical glacial and tectonic processes have created the underlying geology, while river flow, tidal inundation, wave action, and longshore currents continue to shape habitat features through processes such as erosion and deposition. In large river deltas, riverine and tidal processes dominate, while in coastal creek confluences, fluvial processes are more offset by wave and current action, resulting in smaller intertidal footprints. Lagoons are formed behind spits and other features created primarily by sediment deposition. Embayments, a larger habitat feature that can encompass lagoons and coastal creek confluences as smaller spatial units, are maintained by erosion and scour from tides and currents. Intertidal water crossing structures have the potential to disrupt all processes influencing sediment transport.

To date, Washington State has many inventoried intertidal water crossing structures along major transportation corridors. Most of the intertidal crossings in Washington State are associated with development (either agricultural or commercial/residential) and transportation corridors (roads and railroads) in large river deltas and coastal creek confluences. Inventories and preliminary GIS analysis by WDFW indicates 872 intertidal culverts exist along state and local transportation corridors in the Puget Sound area (See Appendix 1). While many of these were not explicitly associated with specific shoreforms, 16% of the records occurred in large river deltas, 13% occurred in embayments, and 5% were associated with lagoons. Coastal creek confluences can occur in a number of different contexts, and so may account for a larger percentage of these sites than other shoreforms. It should be stressed that this inventory is incomplete: it does not include many crossing structures that are associated with private roads, dikes, and drainage district structures.

What species in Washington ecosystems are likely most impacted by intertidal water crossing structures? (Section 4)

A variety of species utilize intertidal habitat types for both passage to and from freshwater and for extended rearing. Key species which are of economic and cultural importance include many salmonids as well as some marine fishes with early rearing in intertidal environments. We highlight the species associated with each main shoreform and identify those with extended residence as well as those that are more likely to be migrants. Chinook, Coho, and Chum Salmon are three culturally important species that show evidence of extended residence in a number of these intertidal environments. Populations of all three species are federally listed as Threatened under the Endangered Species Act in many watersheds in Washington. In addition, common

members of marine species intertidal communities such as Pacific Herring, English Sole, and Shiner Surfperch also exhibit extended residence in certain habitat types. Due to their protracted residence in intertidal environments, these species might be expected to show the greatest sensitivity to impacts from intertidal water crossing structures as a consequence of both restrictions to passage and to changes in habitat quality or quantity. Other listed species such as Steelhead, Bull Trout, and Eulachon may use these systems briefly or only during migration, and so would be expected to be more sensitive to intertidal water crossing structures as barriers to movement. Conversely, nonnative species such as Smallmouth and Largemouth Bass are considered nuisance species that exploit habitats impacted by intertidal water crossing structures.

Species that use intertidal habitats naturally reside in them anywhere from hours to months. Some anadromous species like Steelhead may take advantage of the marine transition to adjust osmoregulation, and this physiological switch can be completed within a couple of days. Other species such as young-of-the-year Chinook Salmon and certain flatfish may reside in intertidal habitats for weeks to months, taking advantage of the abundant food and relative safety of these habitats to rapidly grow from small life stages. Within intertidal habitats, these species rely on riverine and tidal processes to facilitate movement at small spatial scales into areas inundated at high tide and out of areas that dry at low tide.

Movement abilities increase with size, such that individuals < 15 cm in length can be influenced by tidal currents, while species > 15 cm can readily move against some tidal and river flow velocities. Even the smaller life stages use gravity, rheotaxis, and vertical migration to find favorable microenvironments (e.g., low velocities or salinities) to adjust residence time.

How are fish affected by impacts of structures during passage during periods of residence? (Section 4)

Fundamentally, water crossing structures impede bi-directional flow of water and movements of fish in channels and other wetland environments during portions of the tidal cycle, relative to movements in natural intertidal environments. This disproportionately affects smaller fish, which use these habitats as extended rearing environments but which are more likely to be constrained by hydraulic modifications. Of particular importance for small estuarine-dependent life stages is the role of tidal and riverine flows to facilitate passive movement in rearing habitats.

Based on a broad scientific literature spanning many regions of the world, we developed conceptual models for impacts of intertidal water crossing structures upon passage and habitat quality and quantity. These structures may reduce passage as a consequence of impacts from structure dimensions (e.g., culvert length and cross sectional area), slope of structure compared

to the natural water course, tidally influenced water depth and heights of water drops (sometimes called perches), and whether the structure is gated and how it operates. Structures may also reduce habitat quality and quantity landward of water crossings by impacting salinity, inundation height and area, water residence, temperature, sedimentation, and dissolved oxygen. These processes affect both vegetation and the resulting assemblages of fish and their prey. Many studies have demonstrated large changes to fish communities associated with intertidal structures. However, isolating population impacts of these changes from other cumulative effects is challenging. Part of the challenge stems from determining the right temporal scales that matter. For example, restoration of tidal processes can readily benefit fish passage, but ameliorating legacy impacts to habitat quality and quantity may take many years.

How do scientific findings inform potential management? (Section 5)

Science-based management decisions must be made in the context of law and policy as well as multiple land use planning decisions. While these aspects are not in the scope of this scientific review, our synthesis does offer scientific guidance on management of fish and their habitats. We focus on inventory, monitoring, design and operation standards, incorporating impacts of climate change, and decision support frameworks.

Inventory – Inventories of intertidal water crossing structures have lagged behind those focused on freshwater structures. Our review suggests that inventoried intertidal water crossings will often receive higher priority for enhancement than freshwater passage barriers, primarily because of their greater potential to impact connectivity of the entire network. Greater focus on inventorying intertidal structures, with updated information that delineates a structure as a potential barrier, will help put intertidal crossings on a similar footing as freshwater structures. Multiple inventories (e.g. barrier and habitat impact inventories) can address the different impact pathways by which structures influence fish populations.

Monitoring and assessment – Monitoring and assessment are often desirable and sometimes required to evaluate the extent to which particular water crossing structures create barriers to fish passage and residence, or to verify that changes to structures for the purpose of habitat restoration actually produce the anticipated responses in fish and their habitat. Our review suggests a number of characteristics that are useful to monitor and assess for determining whether intertidal structures affect fish passage and residence, as well as habitat quantity and quality.

Design and operation standards – the general rule of thumb for designing and operating intertidal structures with the least amount of disruption to tidal and fluvial processes will likely best serve

estuarine fish populations. However, a strong tradeoff exists between societal values for habitat protections for estuarine species, and for opportunities for shoreline access and development by people. This tradeoff invariably will result in various proposed structural designs that achieve a compromise between these values. Better quantification of these trade-offs will allow for more transparent evaluation of potential impacts and therefore more informed societal decisions.

Incorporating impacts of climate change – previous research in freshwater systems indicate that water crossing structures generally need to be bigger to address impacts of changes in river flow. As yet unaddressed for intertidal structures are additional impacts of sea level rise, which would also argue for larger (and taller) water crossing structures. Design life considerations should incorporate longer-term climate projections to allow the same functionality across a structure's life span. Cumulative impacts of climate change, including impacts to river flow, sea level rise, and increased temperature should be considered when determining likely long-term impacts to habitat quantity and quality.

Decision support – In addition to the above implications, guidance for fish passage and rearing can include a range of tools from reduced complexity decision frameworks for rapid assessment to more expert-level hydraulic model analysis. These would be expected to become more complex and affect multiple metrics as guidance moves from simple determination of passability to evaluation of impacts upon habitat quality and quantity as well as long-term climate impacts. We highlight four tiers of guidance whereby different standards might be developed to advance assessments past fish passage guidance to these other aspects: inventory of structures, evaluations of fish passage, combined evaluations of fish passage and rearing, and projections of climate impacts.

What are the major information gaps? (Section 6)

Numerous information gaps exist concerning intertidal water crossing structures, and some surprisingly simple questions regarding the impacts of intertidal water crossing structures upon fish populations remain unanswered because we lack information on local movement dynamics and the population consequences of lost access and changes in habitat quality and quantity. Moreover, additional research is needed to estimate combined effects of changes in freshwater and sea level rise on impacts to hydraulics and design life of intertidal water crossing structures. Addressing these information gaps will require a combination of observations of fish movements (perhaps in experimental test beds), hydraulic modeling, population modeling to examine impacts in the context of fish life cycles, and models that better integrate multiple climate stressors.

1. Introduction

Hydromodification – the alteration of the natural flow of water (e.g., dikes, channelization, or dams) – has been a consistent endeavor of human civilization. Control of local hydrology has provided opportunity for the development of agriculture, settlements, and resource extraction. However, the benefits of hydromodifications often come at a cost to local populations of aquatic organisms that depend on aquatic habitats. Modifications in aquatic habitats have been linked to declines in populations of aquatic organisms from headwaters (Price et al. 2010) to the intertidal (e.g., Simenstad and Cordell 2000, Lotze et al. 2006). Washington state law requires that water crossing structures (e.g. bridges and culverts) be passable to fish (Barnard et al. 2013). Policies like this that drive the recovery and conservation of aquatic species and ecosystems have led to the study of impacts of hydromodifications, development of regulations and technical guidance (see Appendix 1) for assessment protocols, and design criteria for structures that may create barriers to fish passage.

Much attention has focused on small water crossing structures, principally culverts in riverine systems, and their impacts to fish passage (Dane 1978, Powers et al. 1997, Kahler and Quinn 1998, Hoffman and Dunham 2007, Price et al. 2010, Diebel et al. 2015). In freshwater drainages, the impact of hydromodifications associated with structures like culverts are traditionally interpreted as having predominantly unidirectional impacts to habitats and biota resulting from upstream to downstream gradients of flow. Fish passage technical guidelines for instream structures emphasize designs to improve the passage of fish and aquatic species through structures in both upstream (e.g., returning adult salmon migrating upriver to spawn) and downstream (e.g., juvenile salmon migrating to the ocean or overwintering habitat) directions.

In contrast to structures in riverine systems, the need for design guidelines for structures in tidally influenced areas have only recently gained traction. Design guidelines have not been developed to date for multiple reasons that make fish passage criteria for intertidal structures challenging to define. First, to determine the magnitude of a crossing structure's impact on fish passage and access to habitats, an understanding of how fish utilize the area is required. Despite broad knowledge of the importance of intertidal environments for some fishes (e.g., Hughes et al. 2014), information on habitat use, behavior, and migration timing by both juvenile and adult fish in intertidal areas is piecemeal, focused on specific species, and is often project-specific.

Second, tidal action creates bidirectional impacts that complicate the practical application of existing protocols for fish passage assessment and current fish passage design criteria focusing on matching geomorphic similarities of natural fluvial channels. Tides create daily, monthly, and seasonal variation in inundation, resulting in dynamic cycles of tidal influence. The presence

of hydraulic structures within the intertidal environment results in a local attenuation of the tidal amplitude as well as a localized shift in the timing of tidal inundation.

Third, most ideas concerning riverine water crossing structures focus on movement of water within geomorphically stable channels, not the regular inundation of a marsh surface above the elevation of the channel network. The addition of riverine flows at tidally influenced structures, both within channels and over the marsh surface at higher levels of inundation, further complicates assessment and design procedures because tidal and river forcing can act in both the same and opposing directions, and river flows can have strong seasonal patterns.

Nonetheless, the challenges of determining fish passage at intertidal structures signals a pressing need for the development of fish passage assessment protocols and design criteria for structures in coastal and estuarine systems. This information is necessary because fish passage and retention of ecological processes at these crossings may be disproportionately important given the role of estuaries in the life history of numerous culturally, recreationally, and commercially-important species (Beck et al. 2001, Minello et al. 2003, Rountree and Able 2007). However, several knowledge gaps limit the development of technical guidance for identifying the potential impacts and design criteria of tidally influenced water crossing structures: (1) knowledge of fish and other aquatic organism movements relative to tidal dynamics and the impacts of tidal muting on habitat quality is limited, and (2) improved understanding of natural physical conditions of intertidal habitats (e.g. sediment transport, channel formation) is important for establishing design standards for water crossing structures that minimize impacts to habitat forming processes.

The purpose of this document is to assemble and synthesize scientific literature from studies describing reference conditions in estuaries for fish use and evaluating potential impacts of intertidal structures on the movement, behavior, residence, abundance, and diversity of fishes as well as the ecological impacts of such structures. Although this review was geographically focused on the Pacific Northwestern United States and fishes that utilize intertidal habitats, this review is international in scope. From this synthesis, our goal is to inform the eventual development of several important products that can improve the conservation and recovery of fishes and the intertidal habitats on which they depend. These products include:

- protocols to assess the passability of existing structures in tidally influenced systems during time periods when fish are using this habitat,
- tools to prioritize the repair or restoration of fish passage in tidally influenced systems;
- design considerations and criteria for structures for fish passage in tidally influenced systems;
- design considerations and criteria for protection and restoration of habitat conditions upstream and downstream of the structures; and

- identification of knowledge gaps to inform future research needs.

Report organization

To consider the impacts of water crossing structures in intertidal environments, three fundamental questions need to be addressed:

- 1) What are the primary intertidal water crossing structures of interest and how do hydraulics and sediment transport operate at these structures (*Section 2*)?
- 2) What habitat types and natural processes are affected by structures (*Section 3*)?
- 3) How do fish use intertidal habitats where water crossing structures are built, and how are fish affected by impacts to different habitats including passage, as well as habitat quantity and quality during periods of residence (*Section 4*)?

We have organized this document to start with the main water crossing structures of interest in order to define the nature of the problem. We follow this with a description of the habitats that are impacted by intertidal water crossing structures, and then address how fish may be affected. Addressing the above questions sets the stage for addressing how better knowledge may improve management tools, including fish barrier assessment protocols and technical design guidelines (*Section 5*). The review also highlights a number of key information gaps, which we summarize in the final section (*Section 6*).

The review focuses on northern latitudes of the world where salmonids and extensive estuarine environments are prevalent, but it includes other regions with extensive study as well (see also the companion Annotated Bibliography). Hence, we refer to areas relative to their adjacent ocean, so the Pacific Northwest (British Columbia, Washington, Oregon, and California) is designated as Northeastern Pacific, coastal north Atlantic states and Canadian provinces as Atlantic Northwest, and so forth. The review addresses estuarine nursery species and anadromous species that migrate through estuaries as both juveniles – when they may be constrained by crossing structures – and as adults that are more capable of navigating structures. We do not focus on native freshwater resident species, as these are much less likely to inhabit estuarine and nearshore environments affected by marine crossing structures.

2. What are the primary intertidal water crossing structures?

People have engineered a number of hydraulic structures, including dikes and levees, to modify tidal marshes for agriculture and other development (Fig. 1). Increased pressures from farming activities, transportation development, and infrastructure have led to the construction of hundreds of intertidal water crossing structures across Washington State (Appendix 1, Fig. A1). Structures used in intertidal environments exhibit a variety of shapes and forms, depending upon the degree to which structures are designed to alter patterns of inundation and flow. Where tidal processes dominate, development has focused on reducing tidal range and channelizing tidal flow to optimize drainage. Where riverine inputs dominate, engineering alternatives have often focused on controlling and channelizing river geomorphology to allow conversions of flood plain to agricultural uses. In either case significant impacts to habitat access and quality are often the outcome.

Almost all structures located in the intertidal potentially have some effect on tidal attenuation as well as a perceptible phase shift in timing, although some structures have greater impacts than others. Water crossing structures common to intertidal areas can be grouped into several functional categories based on their design, application, and impacts to hydraulics and geomorphic processes. These functional groups, in increasing order of potential impacts to tidal connectivity, ecosystem processes, and passability, include causeways and bridges, culverts, tidal muting structures, tidegates and floodgates, road and railroad embankments, and dikes and levees (Fig. 1). We group these types into typical water crossing structures, i.e., those that cross channels of water and have been called out in previous definitions of water crossings, with other water crossing structures, which may contain typical water crossing structures but have a much longer intertidal footprint and cross entire wetlands, thereby impeding inundation of tideland surface and not just within channels.

Typical intertidal water crossing structures

Bridges and causeways

Where roads and railways cross wetlands and waterways, elevated causeways or bridges can be built to allow tidal inundation and river flow. These may be full spanning structures or may be constructed with piers, abutments, open-bottom vaults, or arches that support the structure and extend into the waterway or wetland. Bridges and causeways may also be built in conjunction with fill, dikes, and armoring that narrow or confine the waterway or wetland at the site of the crossing. In general, bridges are sometimes built over the main channel of a system and the associated elevated roadways may bisect distributaries or tidal channel networks on either side of the bridged channel.



Figure 1. Examples of intertidal water crossing structures in Puget Sound WA. A. Railroad and highway bridges, road grades, dikes on the seaward boundaries, and tidegates including a tidal muting structure (see inset of road crossing denoted by arrow) at McElroy Slough at the north end of Samish Bay. B. Dikes and road crossings surround Big Indian Slough in Padilla Bay. A set of culverts with flap-style tide gates (inset) is located at the arrow. C. Open culverts (at arrows and in inset) under road crossings drain a lagoon in Port Susan. D. Railroad grade between Everett and Mukilteo that crosses a coastal creek confluence with a culvert (inset and arrow). E. Chuckanut Bay south of Bellingham, dissected by a railroad causeway with bridge. Landscape images are from the WA Coastal Photo Atlas. Inset photo credits: A. Brittany Jones, B. Eric Beamer, C. Correigh Greene, D. Jason Hall.

Culverts

These devices comprise a number of enclosed or semi-enclosed structures conveying water under road crossings or through dikes, levees, road grades, trails, railroad grades, and other similar obstructions. Culverts may be round, oval, rectangular or bottomless arch and vary from small diameters or widths (< 0.3 meter) to very large diameters or widths (> 6 meters). Culverts can be made of a variety of materials, with the most common types consisting of cast in place or pre-cast reinforced concrete, various metals including steel and aluminum, and various plastics. Culverts may also include adaptations such as tidegates, floodgates, control structures, or tidal muting structures as outlined below.

Tidal muting structures

These structures are sometimes called “fish friendly tidegates”, “muted tidal regulators”, “self-regulating tidegates”, or “automated tide gates,” and are commonly used on culvert structures in dikes, levees, and road or railway embankments. In contrast to tidegates and floodgates, tidal muting structures are designed to allow drainage from upstream of the structure while also providing some degree of tidal flooding into the area upstream of the structure. Through a variety of designs, tidal muting structures typically remain open during some initial phase of the flooding tidal cycle and close when the flooding tide exceeds some tidal inundation threshold. The gate then opens again when upstream forcing exceeds the hydraulic threshold on the ebbing tide to allow for downstream drainage through the structure. Depending on the design and both the physical and seasonal setting, tidal muting structures may also remain open or closed for long periods of time across multiple tidal cycles. A great many designs exist for tidal muting structures (Giannico and Souder 2005). The gates can vary from top- or side-hinged barriers of varying size, to walls that rotate on a cylinder. Tidal muting structures can also be built into larger gates to provide either improved drainage, tidal flooding, or fish passage through a passive or manually controlled gate. These “pet doors” are smaller gates on larger gates that can open independently of the larger gate based on a float activated control or through negative buoyancy of materials used to construct the pet door. A larger-scale design for muting tidal inundation is the Controlled Reduced Tide (CRT) system of Belgium (Beauchard et al. 2011, Beauchard 2012), which essentially captures inundation between a high and a low dike, and allows drainage at low tides through culverts in the low dike.

Tidegates and floodgates

Tidegate and floodgate structures are often fitted to culverts that pass through dikes, levees, or road and railway embankments, and are designed to drain freshwater on ebbing tides while blocking tidal flow during flooding tides (Greene et al. 2013). These structures are often a passive top-hinged structure (a flapgate) capping the downstream end of culverts, which open when a hydraulic head differential occurs from the upstream end but closing when the weight of the gate surpasses the hydraulic head. Other types of floodgates include one-way rubber or

plastic valves that essentially act like flapgates by allowing the drainage of freshwater while preventing flooding of tidal water through passive hydraulic control. In addition to these passive structures, some floodgates built within tidal creek channels may be manually operated to open during high periods of precipitation, and then closed to prevent tidal flooding during the spring and summer growing season.

Other water crossing structures

Road and railroad embankments

Road and railroad embankments are hardened and raised substrates that are similar in design to dikes and levees, but can have a wider footprint and are primarily designed to reduce the impacts of flooding on roads through wetlands. Road and railway embankments are also commonly built along coastal bluffs or at the edges of geomorphic floodplains or tidal deltas. These embankments are typically built with culvert structures to allow for drainage from tidal flooding through existing channel networks, but are also typically built with fewer tidal or riverine connections than were present prior to construction.

Dikes and levees

Dikes are narrow earthen walls built to reduce tidal inundation, while levees are similar structures built in freshwater floodplains to impede flood flows. These structures are designed to block tidal or riverine floods onto uplands or create uplands from intertidal/floodplain areas. The construction of dikes and levees is often preceded by filling of intersecting channels or sloughs, construction of culverts, floodgates, or pump systems to drain upstream areas or concentrate flow. Dikes and levees often have culverts fitted with floodgates or pump systems to maintain drainage while eliminating tidal flooding.

What are the relative impacts of different structures?

Where undersized water crossing structures such as culverts are used, upstream impoundments often result from the reduced capability of channels to drain. In a tidal setting, undersized crossings can also restrict the flow of tidal water on the flood tide, creating impoundments of tidal water below the culvert while also reducing the extent of the tidal prism upstream.

Undersized culverts can also increase water velocities in both upstream and downstream directions in a tidal setting as the restriction of ebbing and flooding tidal water interact with freshwater drainage. The increased velocities can not only increase localized scour downstream of tidal restrictions as is common in a freshwater setting, but may also cause localized scour upstream as a result of increased water velocity through restriction of tidal flooding through undersized culverts. At some structures and settings, the tidal restrictions can be severe enough to result in overtopping of culverts, which can result in shifts in the timing and duration of tidal exchanges.

While bridges and causeways cause lower impacts to geomorphic processes, those structures that do not completely span channels or the geomorphic floodplain of the tidal wetland surfaces can nevertheless have negative impacts. Abutments for structures that do not completely span channels can impede water movement, and this can result in scouring of sediment on either side of the abutments or supports, as well as increases in water velocity or depth. In addition, large woody debris and other floating debris can become impinged upon abutments and piers, which can further impact flows as well as introduce physical barriers to the movement of aquatic organisms.

Railway and roadway embankments can restrict both overland and channelized flows of tidal and riverine water. Although these structures may include culverts or other design features to allow drainage of upland water, road and railroad embankments can also block sediment delivery to intertidal areas, resulting in erosion on the tidal side and filling on the bluff side of the embankment (Clancy et al. 2009, Schlenger et al. 2011). Dikes and levees will have the greatest impacts on tidal connectivity because these structures are designed to reduce the area inundated by tidal action. This reduction in tidal prism can facilitate sediment accretion and reduction in channel size and depth downstream of diked areas (Hood 2007).

3. What are the primary habitat types affected by intertidal water crossing structures?

Evaluating the effects of structures in the intertidal environment is predicated on understanding the natural processes, landforms, biota, and habitat-forming processes that would exist in the absence of the structure. Understanding that reference conditions and ecosystems vary among habitat types and within landscapes is necessary to appropriately evaluate the impacts of intertidal structures. Here we examine the shoreforms and associated habitat types primarily affected by tidal crossings.

Although there are many different shoreforms that comprise nearshore ecosystems (Shipman, 2008), the four most sensitive to the effects of intertidal water crossing structures are: (1) large river deltas, (2) coastal creek confluences (i.e. small river or fan deltas), (3) coastal embayments, and (4) lagoons. The first two shoreforms broadly correspond to the “river delta” geomorphic system, while the second two to the “embayment” system in the classification framework of Shipman (2008). In large river deltas, riverine and tidal processes dominate, while in coastal creek confluences, river flow processes are more offset by wave and current action, resulting in smaller intertidal footprints (Table 1). Lagoons are formed behind spits and other features created primarily by sediment deposition by littoral drift. Embayments, a larger habitat feature that can encompass lagoons and coastal creek confluences at smaller spatial scales, are maintained by a combination of continued erosion and scour by tides and currents (Shipman 2008). These four shoreforms are sensitive to changes in sediment deposition and erosion, and by extension to changes in connectivity with tidal and fluvial flow.

Within shoreforms exist a number of habitat types at smaller spatial scales (Table 1). These habitat types are composed of landforms (Shipman 2008), as well as wetland vegetation (Watson 2004, Adamus 2005), which in combination produce features used by a variety of fish and shellfish. As habitats become more marine dominated, channel landforms become less prevalent, and vegetation types shift to saltwater tolerant communities (Table 1).

Intertidal water crossing structures have the potential to disrupt all processes influencing sediment transport. The differences in habitat among large river deltas, coastal creek confluences, coastal embayments, and lagoons are primarily driven by the relative magnitude and resulting influences of riverine, tidal, and wave processes. The potential impacts of intertidal water crossing structures will vary based on both the habitat type and the setting within the intertidal habitat (Table 1). In addition, these habitat types also differ in the biotic communities that they support.

Table 1. Characteristics of shoreforms in intertidal zones and consequent sensitivity to water crossing structures.

Shoreform	Slope	Salinity range (ppt)	Dominant energy source(s)	Primary effects of structures on habitat	Intertidal habitat types impacted
Large river deltas	Low (<5%)	0-30	Rivers Tides Waves	Impounded freshwater Lower sediment accretion Reduced tidal prism	<i>Landforms:</i> Blind tidal channels Distributaries Impoundments Mudflats <i>Vegetation types:</i> Forested tidal wetlands Scrub-shrub wetlands Estuarine emergent marsh
Coastal creek confluences	High (>5%)	0-10	Rivers Tides Waves	Reduced tidal action Impounded freshwater	<i>Landforms:</i> Impoundments Distributaries Beaches <i>Vegetation types:</i> Estuarine emergent marsh Eelgrass
Embayments	Low to high	5-32	Waves Currents Tides	Reduced tidal action Impounded saline water Shoreline armoring on spits	<i>Landforms:</i> Mudflats Beaches <i>Vegetation types:</i> Estuarine emergent marsh Eelgrass Algal beds
Lagoons	Low (< 5%)	3-20	Tides Currents Waves	Reduced tidal action Impounded saline water Hardening of beaches Shoreline armoring on spits	<i>Landforms:</i> Mudflats Beaches <i>Vegetation types:</i> Estuarine emergent marsh Eelgrass Algal beds

Large river deltas

Large river deltas are the interface between large rivers and marine ecosystems (Fig. 2). This transitional environment between freshwater and marine habitat forms a highly productive ecosystem which are inhabited by a diversity of aquatic and terrestrial species. Many of these species exhibit strong geographic, seasonal, and interannual variations in the use of delta habitats (Simenstad et al. 2000, Appendix 2). Diadromous species like Pacific salmon, lamprey, and smelt rely on large river deltas as migration corridors between freshwater and marine habitats at a range of life stages including larval, juvenile, and adult (Appendix 2). In addition, numerous diadromous species and obligate estuarine species may rear extensively in river delta habitats as

juveniles or throughout their life cycle (Table 2, Appendix 2). Given the dynamic nature of salinities within large river deltas (Table 1), these habitats are less likely to support stenohaline marine species as opposed to euryhaline species that can tolerate fluctuations in salinity or that require different salinities at different life stages (Mejia et al. 2008). However, facultative use of river delta habitats by stenohaline marine species is likely where tidal prism and seasonal fluctuations facilitate saltwater intrusion within river delta habitats (Sandell et al. 2011).

The physical structure of river deltas can also be highly variable and will vary greatly based on the dominant processes that form and maintain the river delta system (Table 1). River deltas generally form the most complex systems of intertidal habitat among the different habitat types. Common features to river delta systems include distributaries, tidal channels, tidal marshes, tidal flats, and subtidal flats. The degree to which these features are formed and maintain depend on river discharge, tidal range, sediment loading, coastal geography, exposure to wave action, and geography of the lower river (Shipman 2008). Depending on the balance of these processes and attributes,

large river deltas can generally be classified as river-dominated, tide-dominated, or wave-dominated deltas. River-dominated deltas, i.e., estuaries forming at the mouth of large rivers entering marine waters, typically have extensive low gradient alluvial plains with numerous distributaries and tidal channels, and extensive upstream tidal influence. Tide-dominated deltas form at the head of bays where tidal forces typically drive the formation and characteristics of habitat within the delta (Shipman 2008). Like river-dominated deltas, tide-dominated deltas typically have numerous distributaries and tidal channels, although channel networks in tide-dominated deltas are usually less complex than in river-dominated deltas. Wave-dominated deltas occur where the coastal confluence is more exposed to wave action, which can result in the formation of barrier beaches through the longshore transport of sediment (Shipman 2008). These deltas typically have fewer distributary and tidal channels, and more restricted upriver influence of tides compared to tide-dominated or river-dominated deltas.

Water crossing structures, and tidegates in particular, are likely to be most prevalent in large river delta systems as compared to other shoreform types, as these are usually the focus of

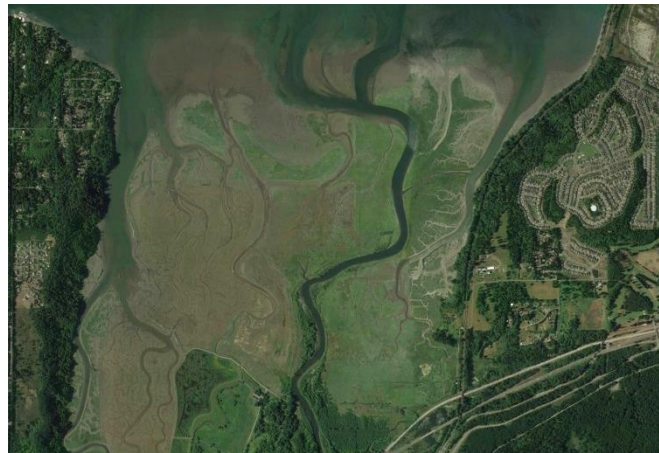


Figure 2. The tidal delta of the Nisqually River in Puget Sound, WA. Note how the highway grade acts as a major water crossing barrier to several historical channels, and both existing dikes and historical barriers removed as a result of a major restoration are apparent within the footprint of the tidal delta.

development activities (Lotze et al. 2006, Simenstad et al. 2011). Given that large river deltas can also comprise the most complex intertidal habitat systems, cumulative effects of multiple structures are also more likely within large river delta systems. In general, water crossing structures within large river delta systems will have the effect of increasing the residence time of freshwater given the presence of freshwater inputs from upstream, while reducing the intrusion of marine water and muting tidal signals (Table 1). However, the geographic position of the water crossing structure within the system and relative influence of riverine and tidal forcing will determine the range of potential impacts for a water crossing structure within large river deltas. Natural downstream transport of sediments, large woody debris, or other debris may similarly be impacted by water crossing structures in large river delta and small coastal creek confluence systems (Bowron et al. 2011). This can result in the accumulation of sediments and debris upstream of structures and the deprivation of sediments below structures (Hood 2004). Reduced transport of sediment and debris within large river deltas can impact the progradation and maintenance of tidal marshes and tidal channels (Hood 2004). Conversely, the restriction of marine influence upstream of structures can change the biotic assemblages of freshwater assemblages from salinity tolerant to freshwater conforming species; increase the duration of inundation through muting of tidal cycles while reducing the magnitude of tidal range (Greene et al. 2013); and reduce distribution of marine derived nutrients and debris upstream of the structure.

Coastal creek confluences (small river or fan deltas)

Like large river deltas, the confluence of small coastal creeks often can support species that use estuaries as migration corridors or rearing habitat (Fig. 3). However, the species assemblage can be smaller compared to large river delta systems. For example, small coastal creeks in the Pacific Northwest typically support anadromous populations of Coho, Chum, Cutthroat, and Steelhead but are less likely to support Chinook and Pink Salmon that are more common to large river systems. The presence of diadromous species in a coastal creek watershed indicates that the confluences of these systems need to support migration between freshwater and marine habitats at a range of life stages with variations in seasonal timings. In addition, juvenile salmonids migrating along the nearshore may use intertidal areas of small streams for rearing (Beamer et al. 2013). Subyearling Coho may move downstream into marine waters, follow low salinity plumes along the shore, and migrate upstream to overwinter in natal or nonnatal streams (Koski 2009, Shaul et al. 2013, Bennett et al. 2015). Like large river deltas, these small coastal creek confluences will also likely support rearing of species at a variety of life stages with variations in seasonal timing. While euryhaline species are more likely to utilize small coastal creek confluences given the presence of dynamic salinities, stenohaline marine species would also be expected given that seasonal fluctuations can maintain elevated salinities.

In contrast to large river deltas, small freshwater watersheds entering marine waters lack a consistent hydraulic effect but provide enough sediment to offset wave-based erosion, producing alluvial fans and small barrier estuaries (Shipman 2008). These systems sometimes have a high slope compared to large river deltas, which minimizes the extent of upstream tidal influence (Table 1). Likewise, the high slope of many coastal creek confluences also minimizes the relative influence of seasonal variation in river flow on the extent of tidal influence. However, low gradient streams in some coastal lowlands can support more extensive estuaries and tidal prisms.

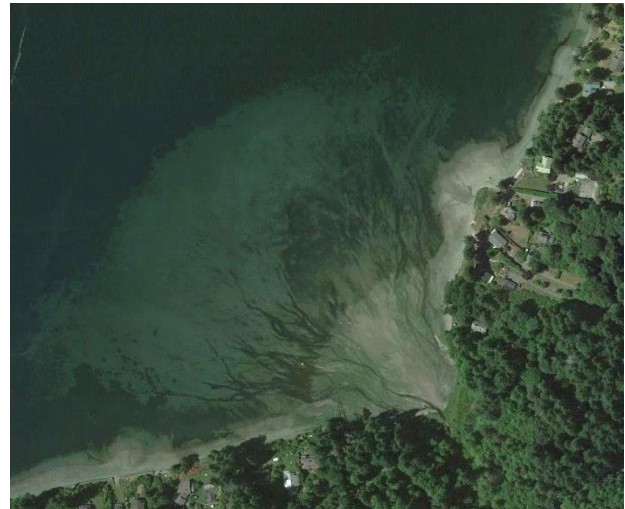


Figure 3. A coastal creek confluence on the west side of Vashon Island in Puget Sound. Note the broad alluvial fan extending into subtidal waters.

The potential impacts of water crossing structures within small coastal creek confluence systems are likely to be very similar to the potential range of impacts within large river delta systems. Like large river deltas, the presence of directional freshwater inputs will likely result in structures increasing the residence time of freshwater while muting tidal signals and decreasing the intrusion of marine water upstream of the structure (Table 1). In addition, the downstream transport of riverine derived sediment, debris, and nutrients and upstream transport of marine derived sediments, debris, and nutrients are likely to be impacted by structures in small coastal creek confluence systems as would be expected in large river delta systems. However, the extent of impacts from structures within small coastal creek confluences is likely to be smaller than in large river delta systems, especially if the small coastal creek system is a higher gradient system.

Embayments

Embayments are protected bays where wave action limits the formation of beaches, while allowing the formation of tidal channel, tidal marshes, and tidal flats (Fig. 4, Table 1) (Shipman 2008). Embayments come in a wide variety of forms and can contain many intertidal habitat types depending on the scale of consideration (Shipman 2008). In general, embayments are protected environments outside of the direct influence of river deltas, even though rivers may feed into some larger embayment habitats (Table 1). Proximity to large river delta or small coastal creek confluences will likely partially determine the relative abundance and life stages of diadromous species within embayment habitats (Beamer et al. 2005). Small streams entering embayments such as coastal inlets and barrier estuaries will also contribute to fish distribution in embayments (Beamer et al. 2013). Intertidal embayment habitats are likely to support a higher proportion of rearing juvenile life stages of anadromous fish species as opposed to adult life

stages. In addition, embayment habitats are more likely to support stenohaline marine species (e.g. jellyfish) as opposed to euryhaline species that can tolerate or require a wide range of salinities (Table 2). However, this too depends on the geographical setting and influence of large river or coastal creeks within the embayment habitat.

Although the potential for habitat impacts is likely greatest within large river delta and small coastal creek confluence habitats, water crossing structures can have similar impacts in embayment systems that are influenced primarily by marine forces. The tidal channels, marshes, and flats associated with the edges of embayment habitats are sensitive to changes in connectivity resulting from water crossing structures, although the influence of tidal action can be more confined along the shoreline as compared to the more extensive landward influence of tides in large river or coastal creek confluences.



Figure 4. Woodward Bay, an embayment extending to the west from Henderson Inlet in south Puget Sound. Note the partial blockage by a railroad crossing.

In contrast to large river delta and small coastal creek confluences, structures within embayment habitats are likely to increase the residence time of marine water (Table 1). However, structures in embayments and lagoons can restrict the extent of marine water intrusion, marine derived debris and nutrients, and tidal range upstream of structures similar to that which would occur in large river delta and small coastal creek confluences. Because marine forcing is the primary habitat forming process in embayment and lagoon habitats, restriction of tidal forcing and muting of tidal signals has the potential to influence the primary habitat forming and maintenance processes for tidal channel and marsh habitats in embayments.

Lagoons

Wave-based transport processes can create lagoons and other sheltered environments where tidal channels, tidal marshes, and tidal flats can form (Fig. 5, Table 1). Pocket estuaries are lagoons formed behind spit or barrier beaches, but they can also be formed by coastal creek confluences (Beamer et al. 2003). Lagoons are dominated by marine processes, and therefore the formation of intertidal habitat will be more strongly related to tidal range, the presence of tidal oscillations,

and geographic setting (Table 1). However, the protected environment of lagoon habitats naturally allows periodic inputs of precipitation, groundwater, or tidal trapping of nearby riverine inputs that can create a habitat with a reduced salinity or seasonal and interannual variations in salinities. In addition, the stability of barrier features that maintain the lagoon habitat can also result in changes in the structure and characteristics of intertidal habitat over time. Therefore, the formation and maintenance of intertidal habitats within lagoons are related to the maintenance of dynamic wave-based transport processes (Shipman 2008, McBride et al. 2009).

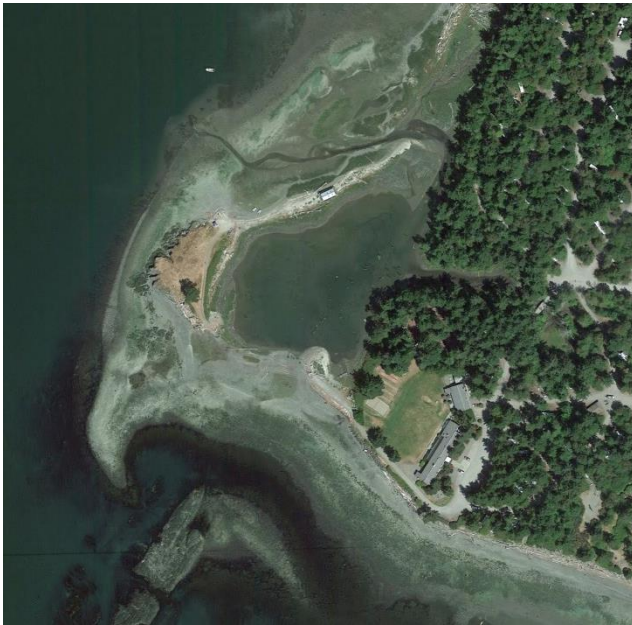


Figure 5. Lone Tree Lagoon in Skagit Bay, North Puget Sound WA. Note that this is also a coastal creek confluence entering at the middle right of the lagoon.

Like embayments, intertidal areas within lagoon habitats can support diadromous species but will tend to support juvenile life stages as opposed to adult stages. In addition, stenohaline marine species as opposed to euryhaline species are likely to be more prevalent as in embayment habitats, but this will ultimately be dependent upon the proximity and influence of large river or small coastal creeks. The potential impacts of water crossing structures in lagoons are similar to embayment habitats, with structures likely increasing the residence time of marine water (Table 1, Mejia et al. 2008, Ritter et al. 2008). Muting of landward tidal signals and marine derived sediments, nutrients, and debris may also impact habitat forming and maintenance processes in lagoon tidal channels.

4. How are fish affected by intertidal water crossing structures?

A variety of fish and shellfish (examples in Table 2) use intertidal habitats that can be affected by water crossing structures. Maintaining the ecological integrity of estuarine and nearshore environments for these species depends on better science and management regarding effects of these structures. Better science can help identify the primary mechanisms by which structures impact fish and shellfish, and can inform various options for better management in terms of structure design, maintenance, and operation.

Key species of Washington State

While many aquatic species may be sensitive to intertidal water crossing structures, this review cannot catalog them all. Instead, we focus on high profile species which are actively managed by tribal, state, and federal organizations, and species that are highly abundant and therefore may interact with managed species. Nevertheless, WAC 220-660-190 requires passage for “all species of adult and juvenile fish”, so the principles developed for high profile species apply to all others that may be affected by structures.

Species potentially affected by intertidal structures (e.g., Table 2) include those listed as Threatened or Endangered under the Endangered Species Act, whose critical habitat designations invoke protections that must be addressed in structures’ design, operation, and maintenance considerations. Key examples of these include Chinook Salmon and some Chum Salmon populations in Puget Sound and the Lower Columbia River, and Eulachon and Bull Trout throughout the state. Other fish (primarily salmon and lamprey) and shellfish (e.g., Dungeness Crab and Geoducks) not listed as Threatened or Endangered are nevertheless subject to tribal harvest rights set forth in the Stevens Treaties.

In addition to listed species, one forage fish (northern anchovy), five species of salmon, and numerous groundfish species (including estuarine dependent starry flounder and English Sole) are included in fisheries management plans of the Pacific Fisheries Management Council (PFMC), and therefore are subject to Essential Fish Habitat considerations. Under the Magnuson-Stevens Act, Essential Fish Habitat modifications resulting from placement of structures are subject to consultations with the National Marine Fisheries Service, with the intent of reducing the habitat impacts of these actions. Other species such as Surf Smelt, Pacific Herring, and sea-run Cutthroat Trout are recreationally important stocks managed by the State of Washington for fishing.

Many other species that use intertidal environments are not subject to specific regulations but are nevertheless desirable beneficiaries of intertidal restoration efforts to restore natural processes to these systems by removal or modification of structures. Some of the most abundant species are

Table 2. Species of the northeast Pacific that may be sensitive to structures used in intertidal habitats. Species are divided into those with protracted versus temporary residence in particular habitats, and by species groups with differing levels of regulatory importance. Italicized names highlight potential indicator species. Dashes = Not applicable, UNK = unknown. Subyearling and yearling migrants are denoted as (0+) and (1+), respectively.

Duration of estuarine rearing	Shoreform	ESA-listed species	Culturally, recreationally, & commercially important species	Common species of fish community	Non-native & nuisance species
Protracted residence	Large river deltas	<i>Chinook Salmon (0+)</i> <i>Coho Salmon (0+)</i> <i>Chum Salmon</i> <i>Bull Trout</i>	<i>Coho Salmon (0+)</i> <i>Cutthroat Trout</i> <i>Surf Smelt</i> <i>Starry Flounder</i>	<i>Three-spine Stickleback</i> <i>Shiner Surfperch</i> <i>Peamouth</i> <i>Northern Pikeminnow</i> <i>Largemouth Sucker</i>	<i>Smallmouth Bass</i> <i>Largemouth Bass</i> <i>Yellow Perch</i> <i>Pumpkinseed</i> <i>Bluegill</i>
	Coastal confluences	<i>Chinook Salmon (0+)</i> <i>Coho Salmon (0+)</i> <i>Chum Salmon</i>	<i>Coho Salmon (0+)</i> <i>Cutthroat Trout</i>	UNK	--
	Lagoons	<i>Chinook Salmon (0+)</i> <i>Coho Salmon (0+)</i> <i>Chum Salmon</i>	<i>Chum Salmon</i> <i>Coho Salmon (0+)</i> <i>English Sole</i> <i>Dungeness Crab</i>	<i>Staghorn Sculpin</i> <i>Snake Prickleback</i> <i>Crescent Gunnel</i>	--
	Embayments	<i>Chinook Salmon (0+)</i> <i>Chum Salmon</i> <i>Bull Trout</i>	<i>Chum Salmon</i> <i>Pink Salmon</i> <i>Cutthroat Trout</i> <i>Pacific Herring</i> <i>Surf Smelt</i> <i>N. Anchovy</i> <i>English Sole</i> <i>Dungeness Crab</i>	<i>Staghorn Sculpin</i> <i>Snake Prickleback</i> <i>Pacific Sandlance</i> <i>Crescent Gunnel</i>	<i>Lions mane Jellyfish</i> <i>Moon Jellyfish</i> <i>Fried-egg Jellyfish</i>
Brief residence or passage	Large river deltas	<i>Steelhead Trout</i> <i>Bull Trout</i> <i>Eulachon</i> <i>Green Sturgeon</i> <i>Chinook Salmon (1+)</i>	<i>Coho Salmon (1+)</i> <i>Chum Salmon</i> <i>Chinook Salmon (1+)</i> <i>Pink Salmon</i> <i>Sockeye Salmon</i> <i>Cutthroat Trout</i> <i>White Sturgeon</i> <i>Pacific Herring</i> <i>Pacific Lamprey</i>	<i>River Lamprey</i> <i>Longfin Smelt</i>	<i>American Shad</i>
	Coastal creek mouths	<i>Steelhead Trout</i>	<i>Coho Salmon (1+)</i> <i>Chum Salmon</i>	UNK	--
	Lagoons	--	--	--	--
	Embayments	<i>Steelhead Trout</i> <i>Coho Salmon (1+)</i> <i>Chinook Salmon (1+)</i>	<i>Coho Salmon (1+)</i> <i>Sockeye Salmon</i>	<i>River Lamprey</i> <i>Longfin Smelt</i>	<i>American Shad</i>

listed in Table 2, but the entire community can be much larger for particular estuary or nearshore systems. Conversely, a growing number of non-native species and some native species are considered nuisances because they prey upon or compete with native species. Numerous non-native fish introduced into rivers to enhance fishing opportunities have colonized the tidal freshwater regions of large river deltas, and structures can enhance habitat features such as scour holes, freshwater retention, and warmer temperatures for nonnative fish. In marine waters such as embayments, structures may provide substrates for jellyfish polyps (Hoover and Purcell 2009) and may increase residence time of water in the embayments (Khangaonkar and Wang 2013). Both processes may facilitate blooms of jellyfish that could compete or even prey upon desirable fish and shellfish life stages.

How do fish naturally use intertidal habitats?

The degree to which various structures can impact fish and shellfish and facilitate nuisance species will depend upon natural residence patterns of fish in intertidal habitats (Table 2). Fish may use intertidal habitats for hours to months, depending upon a number of life history characteristics and the scale at which movements are measured (Quinn 2005, Hering 2009, Levings 2016). Some anadromous species use intertidal environments for short periods of time (hours to days) during their downstream migration to facilitate changes to osmoregulation (Iwata and Komatsu 1984, Moser et al. 1991), yet others reside within intertidal habitats for extended or protracted periods of time (Levings 2016). Species with protracted residence in intertidal environments depend upon habitat features that facilitate their growth and survival (Appendix 1). Many species such as Chinook, Chum, and Coho Salmon as well as Starry Flounder and English Sole use intertidal environments as nursery environments, which they enter at early life stages (Hughes et al. 2014). These fish depend upon slow-moving and shallow (0.3-2 m) habitats such as blind tidal channels (small channels that wet and dry with tides and which are not directly connected to streams), impoundments within coastal creek confluences, and nearshore lagoons for low-energy and predator-free rearing, abundant food resources, and lower salinity for osmoregulation (Hughes et al. 2014, Levings 2016).

Because of the dynamic osmoregulatory and movement challenges inherent to intertidal habitats, most species of fish do not live their entire life cycle there (Minello et al. 2003, Able 2005). Rather, juvenile life stages colonize these habitats and reside for a period of their early life history. Some species like Chum and Coho Salmon can access coastal creek systems when spawning as adults, so coastal creek confluences and lagoons with associated creeks can be colonized by downstream movement of juveniles.

However, most species colonize these systems by facilitated movement of juveniles upstream or upchannel during flood tides (Gibson 2003). For example, subyearling Chinook Salmon utilize blind tidal channels (Congleton et al. 1981, Levy and Northcote 1982, Beamer et al. 2005,

Hering et al. 2010), nonnatal creek mouths (Beamer et al. 2013), and lagoons (Beamer et al. 2003) during downstream migrations from large rivers. Hence, they must move out of the main channel or marine water column into these smaller protected environments. Marine species such as flatfish, sculpin, and Dungeness Crab depend upon passive movement of eggs and larvae into intertidal habitats (Gibson 2003, Hughes et al. 2014). Once these smaller individuals gain access to estuarine habitats, they may inhabit these habitats for weeks to months (Hering 2009, Hughes et al. 2014), and growth may occur during these extended periods of residence before individuals migrate into more marine habitats or become adults.

Due to the influence of tides, intertidal habitats naturally wet and dry, so fish and shellfish that utilize intertidal habitats for extended periods of time must make many small-scale movements as inundated areas dry out on each low tide. This process has enabled sampling of juvenile Chinook Salmon and other species in blind tidal channels using fyke nets (Congleton et al. 1981), demonstrating extensive use of these habitats by small fish, and absence of these areas by larger fish which are potential predators (Beamer et al. 2005). Other types of capture methodologies (e.g., Bretsch and Allen 2006) confirm movement of fish into intertidal environments during flood tides, and emigration from them on ebb tides. These movements appear strongly influenced by the size of fish – smaller fish use shallower habitats, and larger fish tend to use deeper habitats but have a broader range of depths used (Bretsch and Allen 2006, Valentine-Rose and Layman 2011).

Using PIT tags in conjunction with antennas placed in a blind tidal channel, Hering et al. (2010) and McNatt et al. (2016) found tidally-associated movements of juvenile Chinook into and out of the blind tidal channel, but these were not always passive movements – for example, in the study by Hering et al. (2010) in the Salmon River estuary, as many as 25% of tagged fish moved into the channel while tide was ebbing. Conversely, juveniles may exit channels during flood tides. Most fish movements were facilitated by passive water movements, but they generally occurred at the earlier stages of flood and the later stages of ebb, suggesting swimming and other behaviors (see below) may be used to facilitate holding position within blind channels. These studies imply that as tide ebbs, small fish become more restricted to areas that are always inundated (e.g., freshwater pools, distributary channels, impoundments, and subtidal marine areas). Hence, natural tidal environments used by estuarine-dependent life stages likely require a range of water depths to accommodate fish, especially at low tides. Note that because PIT-tagged fish generally need to be at least 55 mm and juvenile Chinook Salmon enter estuaries as small as 40 mm, these studies are likely biased: smaller fish are likely more dependent upon tidal processes for passive movement into and out of estuary environments.

Observations of size-dependent habitat preferences and non-passive movements of fish in intertidal environments suggest that swimming behavior allows fish to make use of preferred

habitats for extended periods. Swimming behaviors take several forms in fish. First, fish may use directed swimming to move or maintain position in the presence of currents. A number of

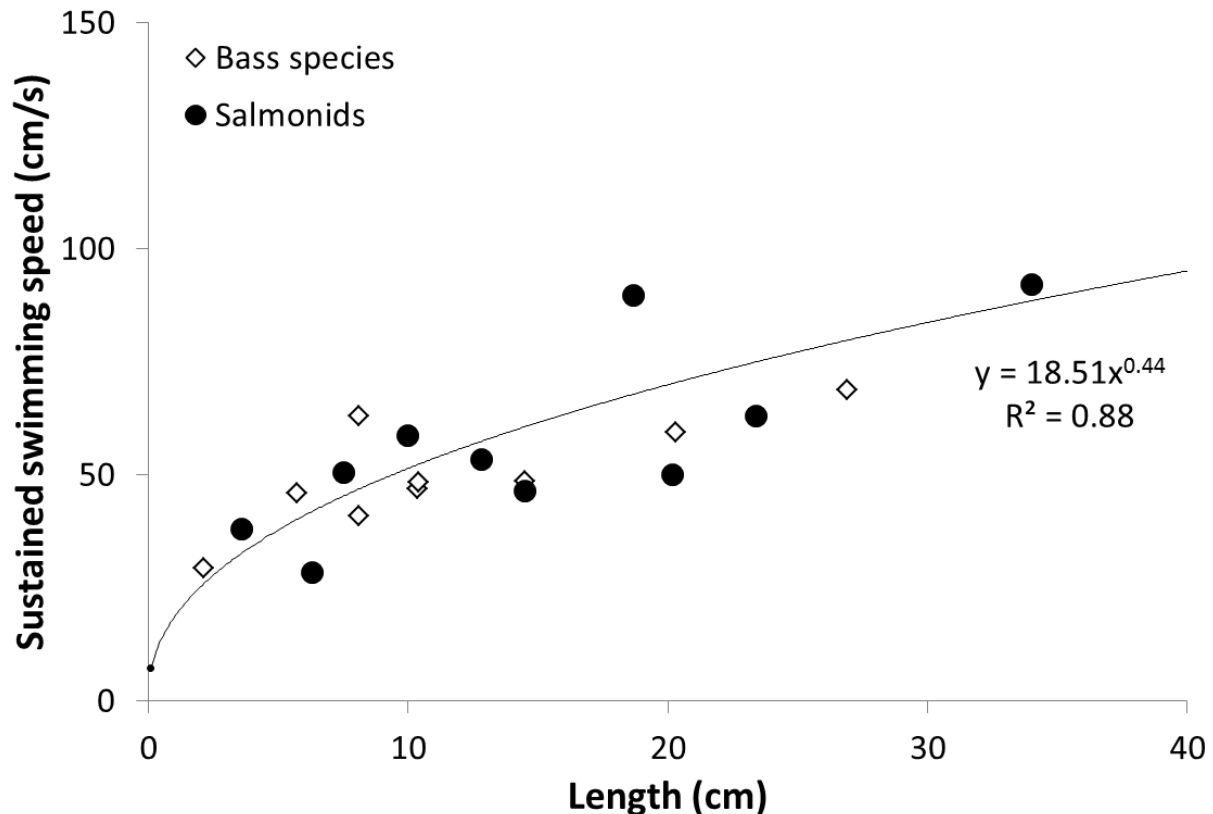


Figure 6. Sustained swimming speeds measured in the laboratory for two species of bass (*Micropterus* spp) and several salmonids (*Oncorhynchus* and *Salmo* spp). Data are taken from (Beamish 1978) and (Goolish 1991). For reference, the single small dot represents the extrapolation of the power function to the size of a small larval fish to highlight the power function relationship.

experimental studies have shown that both sustained swim speeds (velocities that fish can maintain for long periods of time without added metabolic activity) and maximum swim speeds (short, high velocity and energetically expensive bursts) are functions of body size. Smaller fish are less capable of high speeds than larger fish, and these patterns generally follow allometric power functions (e.g., Fig. 6, Beamish 1978).

The range of sustained swimming speeds indicated in Figure 6 suggests that small fish are not well equipped to actively swim against the water velocities of intertidal habitats. For example, peak velocities in embayments (e.g., NOAA Currents data for Puget Sound and Grays Harbor) and large tidal channels of river deltas (Myrick and Leopold 1963, Moser et al. 1991) range from 30-150 cm/s within distributary channels, with higher velocities occurring where there is larger tidal prism and greater geomorphic constraint (Wong 1989, Morgan-King and Schoellhamer

2013). Velocities in sheltered lagoons or small protected blind channels can be substantially less (e.g., peaks at 10-50 cm/s, (Wang et al. 1999, Behrens et al. 2016). Clearly, peak velocities in some of these habitats surpass observed sustained swimming performances shown in Figure 6 at the higher observed velocities.

Even if currents surpass sustained swimming speeds, fish may still be able to hold position in intertidal environments by virtue of several types of orientation mechanisms in response to environmental gradients. The most important of these is rheotaxis, in which fish naturally orient their body so that they directly face into currents. Other forms of taxis include phototaxis (orienting toward or away from light) and gravitaxis (orienting based on the direction of gravity). Fish may also exhibit preferences for certain current speeds, temperatures, salinities, or other environmental gradients. By a process known as selective tidal stream transport (reviewed in Boehlert and Mundy 1988), these behaviors can facilitate vertical or lateral migration in intertidal environments, thereby allowing fish to modulate how they respond to tides and currents. Selective tidal stream transport and related sensory mechanisms (reviewed in Teodosio et al. 2016) have been postulated to explain immigration into and retention within estuaries of weakly swimming larval fish spawned in marine waters. Similar mechanisms may govern behavior of juvenile anadromous fish as they migrate downstream into estuaries and other intertidal environments (Bennett and Burau 2015). Fish entering intertidal environments at relatively large sizes may depend less on these mechanisms due to their higher swimming speeds (e.g., Coho Salmon smolts, Moser et al. 1991), yet they nevertheless may make vertical or lateral movements to reduce energy expenditures. For example, some adult Chinook Salmon entering the Columbia River during upriver migration varied their vertical position, apparently to take advantage of flood tides or reduced ebb tides (Olson and Quinn 1993).

It is worth pointing out that many studies of estuarine nursery fish and their residence patterns in intertidal habitats may be subject to shifting ecological baselines (Pauly 1995) resulting because populations and their habitats have been extensively influenced by people. Movement behavior can strongly be affected by local abundance (Beamer et al. 2005), size (Beamish 1978), the presence (or absence) of predators (Gilliam and Fraser 2001), and the current amount of habitat to support these attributes (Eby and Crowder 2002, Beamer et al. 2005). People have greatly influenced all of these ecosystem components. With the possible exceptions of older studies or of stocks in remote areas, reference populations and sites are increasingly rare to study the “natural” behavioral underpinnings of movement and residence.

Conceptual model of impacts of water crossing structures

For estuary nursery species, water crossing structures have two general impacts resulting from restricted connectivity following directly from colonization (passage) and residence phases (Fig. 7). Because their ability to colonize these habitats depends upon accessibility during flood tides,

structures that limit tidal inundation will act as partial or full barriers to movement. These effects are likely to be size-dependent because smaller fish have poorer swimming capabilities than larger fish (Haro et al. 2004, Smith and Carpenter 1987), and because smaller fish rear in estuarine environments for longer periods of time (Quinn 2005). Tidal restrictions tend to limit the period of time that small fish have passage into suitable rearing habitat. As shown in Figure 7, these effects should result in the reduced likelihood for residence in more restricted environments (see “*Effects of structures upon passage*” section, below).

Individuals that are able to pass connectivity barriers nevertheless encounter a secondary effect of reduced connectivity: lower habitat quality such as elevated temperature, eutrophication, and hypoxia (Fig. 7) (see “*Effects of structures upon habitat quality and quantity*” section, below). These impacts will reduce the length of time that fish rear within areas impacted by structures. In the conceptual example provided in Figure 7, some fish are able to access estuarine areas based on their size, but because of connectivity-constrained low habitat quality, they remain in the habitat for shorter periods of time, resulting in reduced residency and growth. These individual changes would then be expressed at the local population level as reduced abundance or density.

In contrast to juvenile life stages of estuarine dependent species, large juvenile (>15 cm) and adult life stages are expected to be less sensitive to the effects of marine crossing structures. Following the conceptual graphs of Figure 7, larger fish are expected to be able to better navigate connectivity restrictions caused by structures. Furthermore, because they are larger, they are expected to rear for a shorter period of time in areas affected by loss of connectivity. The exception may be species tolerant to high temperatures and lower dissolved oxygen.

In the next sections, we review evidence for impacts of marine crossing structures upon both passage and residence stages. We ask what factors influence reduced connectivity, and to what degree reductions in connectivity influence passage and habitat quality. We first focus on impacts to juvenile life stages for which passage and rearing constraints are most pronounced, and then examine how adult life stages may be affected by losses in connectivity.

Effects of structures upon passage

Research in freshwater has shown that water crossing structures can affect fish passage in a number of ways, illustrated by a simple conceptual model relating various connectivity restrictions to reduced fish passage (Fig. 8). Barriers to fish movement are typically formed by vertical drops, shallow water, and increased velocity (Barnard et al. 2013). Literature on freshwater crossing structures show that structure length, roughness, slope, and cross-sectional area can strongly increase velocity and reduce water depth (Cotterell 1998, Barnard 2003, Hoffman and Dunham 2007, Barnard et al. 2013, Barnard et al. 2015). For example, material (Bates 2006) and interior design (Dane 1978) of culverts and other water crossing

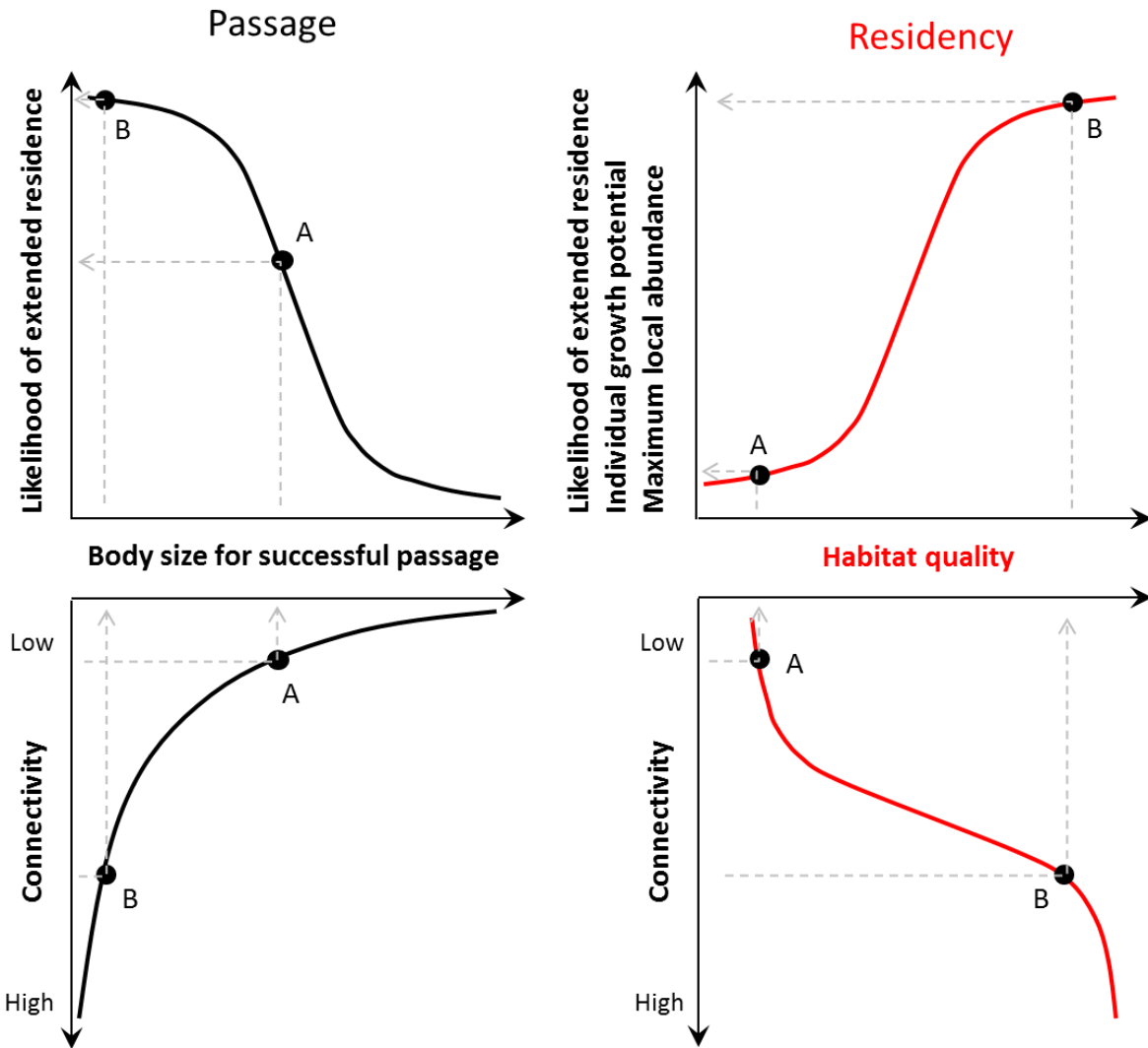


Figure 7. Conceptual effects of water crossing structures and resulting impacts of connectivity upon estuarine nursery species at two phases: passage and residency. Graphs in the lower panels illustrate impacts of changes to connectivity by water crossing structures (vertical axis) upon size-dependent passage (lower left panel) and habitat quality (lower right panel). In turn, graphs in the upper panels illustrate the effects body size at passage and habitat quality (horizontal axes) upon extended residence and other covarying characteristics (vertical axes). Examples of sites with low (A) and high connectivity (B) are noted with dots for general reference. Gray lines with arrows illustrate how different levels of connectivity have cascading consequences upon extended residence and other biological traits. Reductions in connectivity from tidal restrictions reduce the size distribution of organisms that can successfully pass structures, resulting in lost rearing opportunities for small individuals (upper left panel). For those individuals that can pass structures, reduced habitat quality resulting from disconnection also reduces rearing opportunities (upper right panel).

structures can alter channel roughness and thereby influence the ability of fish to migrate upstream. A structure's slope, roughness, and cross-sectional area combine to greatly influence velocity, which is why Washington State has proposed culvert designs using approaches such as "no slope" (Price et al. 2010) and stream simulation to improve upstream passage at water crossings (Barnard et al. 2015). Culvert length can disrupt movement if velocity characteristics throughout a culvert's length surpass the distance fish can endure while swimming into opposing currents (Smith and Carpenter 1987, Hoffman and Dunham 2007). This effect is magnified if water temperatures constrain swimming performance (Starrs et al. 2011). Culverts with vertical drops above the natural channel bed can hinder upstream migration by anadromous fish (David and Hamer 2012). Doehring et al. (2011) found that open culverts with drops in New Zealand resulted in zero upstream passage in a weakly swimming anadromous fish, compared to 65% passage at culverts without drops. Because the fish studied can move upstream by suction, installation of a ramp at these perched gates improved passage to 44%.

Intertidal water crossing structures can affect fish passage in additional ways (Fig. 8). Like barriers from freshwater structures, many of these impacts depend upon fish length (Fig. 6 & 7). Following Figure 6, the ability of fish to overcome passage restrictions through stronger swim speeds might be expected to occur between fish lengths of 7 and 10 cm, where the relationship between length and sustained swim speed starts to level off. Perhaps not coincidentally, it is within this range that many estuarine-dependent species appear to migrate to more dynamic marine waters (e.g., Chinook Salmon, Healey 1980). Because body size plays such an important role in whether structures influence passage, we discuss small and large life stages separately with this size division in mind.

Small juvenile life stages

The largest effects of intertidal water crossing structures occur for small resident fish (< 15 cm in length). Following Fig. 8, intertidal water crossing structures may result in restrictions to tidal flooding and therefore insufficient water depths for passage for a major portion of the tidal cycle. This can happen even for the relatively shallow depths (~20-30 cm) that some juvenile fish reside (Greene et al. 2013). Indeed, a key driver of loss of fish passage from water crossing structures may be tidal attenuation, whereby reduced tidal amplitudes associated with restricted tidal connectivity at water crossing structures reduces the time during which tidal channels may offer passage (Bass 2010, Greene et al. 2013). Passage may be further hampered by vertical drops created by structural sills or culvert inverts positioned at elevations higher than low tide water levels. Vertical drops eliminate passage for most small fish once tide levels fall below the elevation of the sill (Bass 2010). However, Mueller et al. (2008) showed some capability of yearling hatchery Coho Salmon to leap vertical drops no greater than 26 cm.

The more complex restrictions arise from the changes in velocity. Velocity (and its direction) changes with tides and in interaction with downstream patterns of river flow. Water crossing structures have a strong potential to increase velocities and obstruct movement. For example, using mark-recapture techniques, Eberhardt et al. (2011) found that high velocities created by culverts prevented movements of Mummichog, a common estuarine-resident fish of the Atlantic seaboard. An increase in velocity from 0 to 1 m/s created a 15% loss in passage success. Of four variables examined (channel width, culvert size, light intensity, and water velocity) water velocity had the strongest effect and had a consistent pattern across all sites examined. Furthermore, sudden changes in velocity may cause behavioral avoidance of areas with “local acceleration of flow” (Kemp et al. 2005, Kemp et al. 2005). Consistent with this idea, Bass (2010) observed greater downstream passage through tidegates when gates had a greater opening angle.

Floodgates and tidegates close off the channel when flood velocities overcome the inertia for closing the gates. Closures at tidegates persist through slack tides (when small fish can move freely) until the ebb velocity again overcomes the inertia to reopen gates. These effects have been documented to limit fish movements. Doehring et al. (2011) used sonar to observe movements of fish at culverts with and without floodgates. Greater than two times as many fish were able to pass the ungated culvert. Upstream movements at the gated channel shifted from flood tides to low tide, when the gate was open. Likewise, using PIT tagged Coho Salmon, Bass (2010) found that movement direction of yearling smolts declined from 48% upstream in nongated channels to 28% and 3% in side-hinged and top-hinged gates, respectively, and the number of movements upstream and downstream were greatly truncated. Approximately 50% of subyearlings that were tagged were observed to move upstream of tidegates, indicating that some time periods existed by which fish could move upstream. However, subyearling fish were much more sensitive to gate opening than yearlings, possibly indicating greater sensitivity to velocity changes created by the restricted gate opening.

Despite this work, there remains little information on how much restrictions influence movement patterns in intertidal environments, and how various elements contribute to losses in connectivity. This is because either individual movements are not measured, the various contributions of multiple metrics to restrictions are not measured, or both. For example, many studies simply examine abundance patterns of fish and invertebrates between reference sites and sites with water crossing structures, but any differences observed can be the outcome of both limitations on passage or on residence time (see below). Greene et al. (2013) divided the portions of time that various forces that could potentially restrict movements in reference channels, or channels with a flapgate or self-regulating tidegate (SRT). Focusing on movements of juvenile Chinook Salmon into estuarine channels, Greene et al. (2013) identified periods of base from downstream channel bottom, or blocked by high velocities (>0.25 m/sec, based on

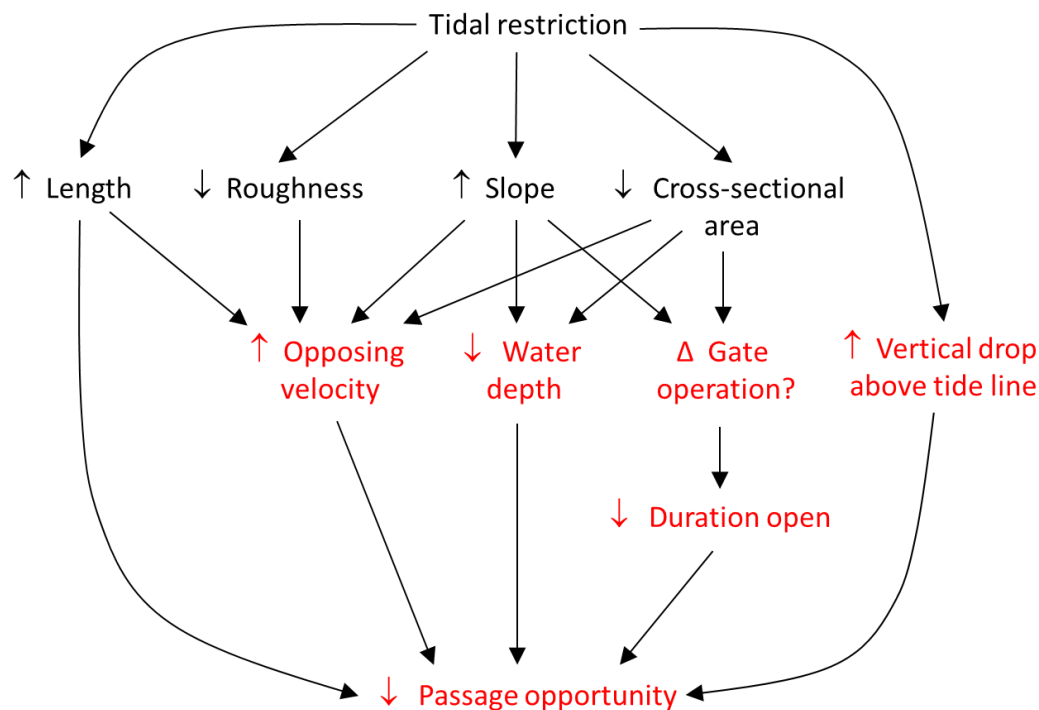


Figure 8. Conceptual model of how tidal restriction (reduced connectivity) influences passage. Arrows by each parameter indicate direction by which restrictions occur, and red text denotes parameters that are tidally influenced.

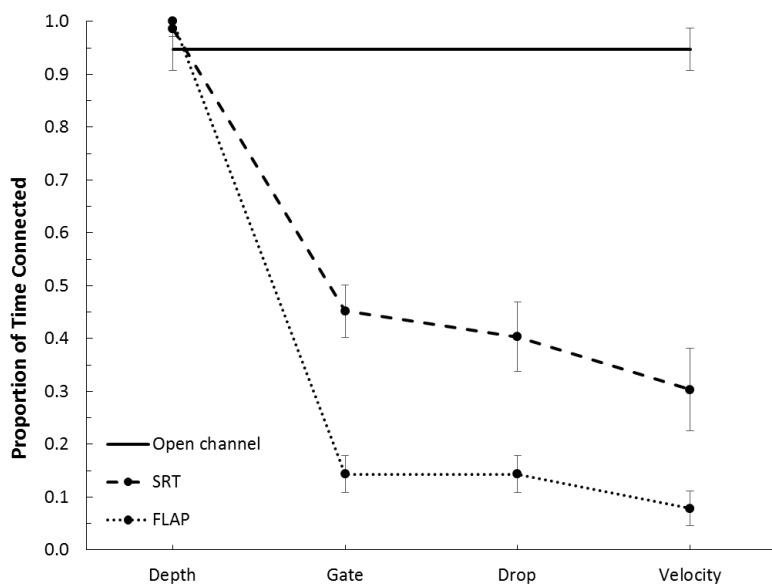


Figure 9. Influence of depth, gate opening, vertical drop, and velocity constraints on the proportion of time that sites in open reference channels, channels with self-regulating tide gates (SRT), and vertically hinged flap gates (FLAP) remained connected. These proportions are cumulative, so for example, gate opening constraints (Gate) also include the effect of depth. Data are from (Greene et al. 2013) from systems in the Oregon and Washington state. The total cumulative proportion gates were connected is shown at the right, after potential velocity barriers were taken into account.

studies of size-dependent swim rates, Fig. 7). Figure 9 shows the cumulative loss in connectivity from these potential barriers, and reveals that the presence of gates had the strongest influence on connectivity loss, followed by velocity barriers, and vertical drops. The only restriction for reference systems was water depth. In combination, these potential barriers reduced the proportion of time that fish could move upstream to approximately 30% for self-regulating tidegates (SRTs) and 10% for flapgates.

Other physical characteristics such as light levels, acoustic interference, temperature, and low dissolved oxygen could conceivably act as barriers, but these parameters have received little attention. Evidence differs for various estuarine species: low light levels from overwater structures have been proposed as barriers to movements in juvenile salmon (Kemp and Williams 2009), but explained the least amount of variation in passage success in the mark-recapture study of Mummichog (Eberhardt et al. 2011). Fish may avoid areas with high temperatures (Cherry et al. 1975, Coutant 1977) and low dissolved oxygen (Eby and Crowder 2002), but water crossing structures have not been shown to exacerbate passage restrictions due to these effects.

Large juvenile and adult life stages

Most of the studies of impacts of water crossing structures on aquatic species have focused on small estuarine resident fish, as opposed to those species that use estuary and nearshore areas largely as migratory pathways. As individuals grow, they have greater opportunities to access systems restricted by intertidal water crossing structures, because they are less affected by the physical constraints imposed by passage barriers (Fig. 7, Bass 2010). Anadromous fish that have extensively reared in freshwater tend to spend relatively short periods of time in intertidal environments (Quinn 2005). For these species (e.g., Steelhead, yearling Coho and yearling Chinook Salmon), intertidal structures may act as barriers only if they exist within the primary upstream migration pathways to spawning grounds, and not during outmigration phases.

Nevertheless, larger individuals may face challenges passing water crossing structures from the seaward side. Tidal restrictions from culverts and bridges change velocity and depth and may impact access for a greater duration of the tide cycle than in open channels. Upstream access at culverts and bridges may be limited to a narrow temporal window or result in a restricted number of migratory pathways. Consider, for example, the tidal dynamics of flap gates barring tidal inundation through culverts. These gates are designed to shut during the flood tide, effectively keeping out larger fish during the time period when tidal channels are at the depths at which larger fish might be expected to use them (Bretsch and Allen 2006, Bass 2010). Only during the initial ebb stages at high tide would landward pressure differential be sufficient to allow flap gates to open and allow upstream fish passage. As the tide ebbs, culverts positioned at higher intertidal elevations could become exposed, thereby resulting in both a vertical drop and the gate in a closed position. Culverts with flap gates positioned lower in the intertidal would likely have steep slopes, limiting upstream passage due to size-dependent velocity constraints (Haro et al.

(2004). Hence, the time window in which a flap gate was accessible to fish could be highly limited, even for fish capable of strong swim speeds. Whether these possible mechanisms greatly impact passage of adults has not been studied.

Effects of structures upon habitat quality and quantity

A large number of studies across the world have documented multiple pathways by which intertidal water crossing structures can impact the quantity and quality of fish habitat (Figure 10). The two primary habitat effects of restrictions created by these structures are changes in volume of tidal inundation and changes in salinity. Changes in the volume of tidal inundation result in reduced tidal prism, which consequently alters the balance of sedimentation and erosion, and increases water residence time (Hood 2004). Changes in sediment flux have important consequences for regulating the amount of wetland habitat, while changes in water residence and salinity result in changes in habitat quality or shifts in habitat type. All three types of water quality changes have the potential to alter the residence of estuarine-dependent species.

In general, construction of channel crossing structures tends to reduce salinity in estuarine environments, because tidal restrictions create impoundments that become more strongly influenced by freshwater inputs. Maximum in-channel salinity levels could be reduced by as much as 29 psu in estuarine systems when subject to effects of a flap gate barrier (Greene et al. 2013) and more extreme restrictions such as dikes could have even greater effects. However, alterations toward lower salinity are not universal, particularly if water crossing structures occur low in the intertidal zone. In the intertidal zone, water crossing structures do not exhibit strong swings in salinity, and increased water residence contributes to higher evaporation, further increasing the saline concentration compared to unimpeded systems. Furthermore, elevation of reference sites often differ from treatment (i.e., impounded) sites. Hence, in some cases higher salinity has been observed, while in other cases there were no discernable effects on salinity due to losses in connectivity. Nevertheless, when salinity levels upstream of tidegates were expressed as a ratio of downstream values, flap gates and muted tide gates averaged 76% and 82% the salinity of reference sites, respectively (Greene et al. 2013).

The other main driving effect of tidal restrictions upon habitat is inundation elevation. Surprisingly, elevations are not often reported in studies, probably because of the technological demands and need for tidal reference points necessary for accurate measurements. However, Greene et al. (2013) reported that maximum water surface elevations above flapgates and self-regulating tidegates on average were reduced by 50% and 65%, respectively, compared to downstream sites and reference systems.

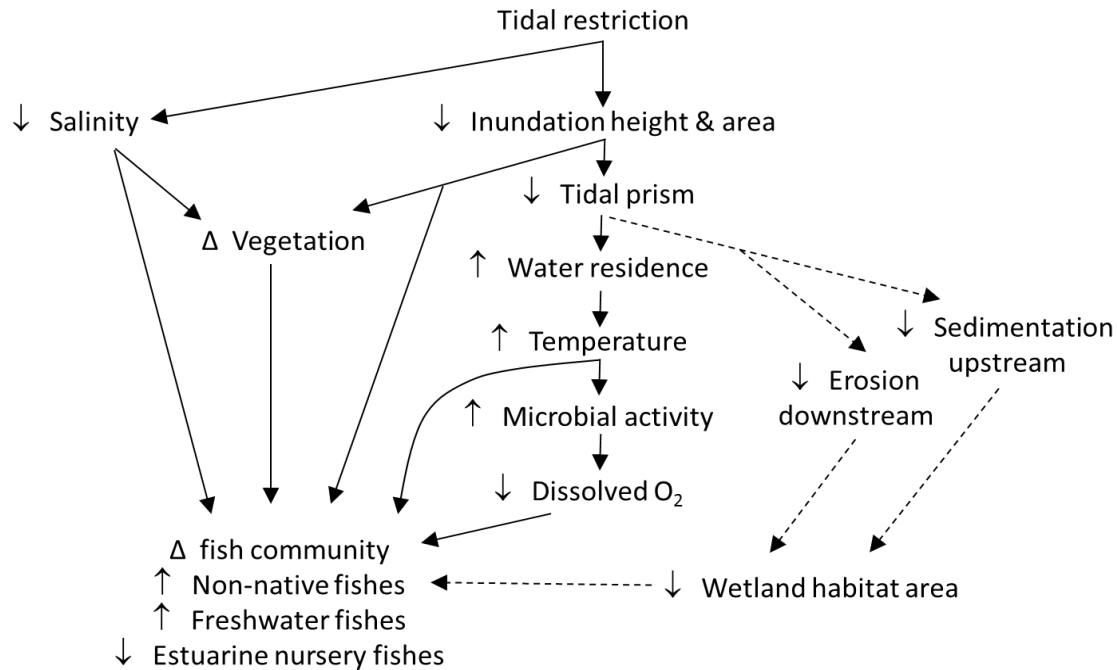


Figure 10. Conceptual model of how tidal restriction (reduced connectivity) influences changes in fish assemblages through habitat changes. Solid lines denote direct, rapid (< 5 years) changes, while dashed lines indicate longer-term (> 5 years) processes.

Through a variety of pathways, tidal restriction impacts the balance of sediment flux and decomposition, resulting in either sediment accretion or downcutting, depending upon sediment sources. Clearly, when connectivity is reduced to zero by dikes and grades, access to estuarine environments landward of these obstructions is lost. As Hood (2004) notes, further losses in estuarine habitat can occur seaward of diked wetlands. Hood (2004) offered several explanations, including 1) the loss of tidal prism which thereby promotes sedimentation downstream of dikes (see also Williams et al. 2013) and 2) increase of flow within tidal channels which results in widening and down-cutting of distributary channels. Actual documentation of the relative importance of these two processes is difficult, due to the absence of landscape-scale experiments examining changes in these processes.

Loss of wetland habitat can also occur in restricted systems landward of structures due to subsidence or altered accretion rates. Estimates of subsidence in wetlands resulting from tidal restriction in a number of estuarine systems ranged from 0.1 to 1.67 cm/year (Turner 2004). Some of this change can be explained by removal of topographic structure due to agricultural practices (Diefenderfer et al. 2008). In addition, existing channels that are landward of culverts often become shallower than more connected downstream reaches because 1) lower tidal flushing fails to remove fine sediment, and 2) elevated positions of culverts promote sedimentation up to the culvert base.

In other cases, subsidence appears to result from changes in rates of decomposition due to tidal restrictions from existing structures. Anisfeld et al. (1999) documented lower accretion rates in tidally restricted systems compared to unrestricted reference sites, and concluded based on analysis of sediment cores demonstrating similar inorganic sediment but lower organic matter that higher decomposition and sediment compaction occurred in areas that had been tidally restricted. Interestingly, Dick and Osunkoya (2000) found an opposite effect: lower decomposition rates of mangrove litter upstream of floodgates. Like many other processes, whether decomposition and accretion rates increase or decrease in response to tidal restriction is likely the consequence of geography, the habitat type, and the influence of tidal and river flow (see Turner 2004).

Changes in habitat area and depth

The consequences of changes in habitat area and depth from tidal restrictions are: 1) loss of the areal extent of tidal wetlands supporting a variety of estuarine-dependent species, 2) related to this, loss of sheet flow across the marsh surface due to reduced tidal elevation at high tide, 3) loss of complexity in the channel depth profile from channel filling and subsidence, resulting primarily in loss of habitat to fish that utilize deeper channels, and 4) shifts in the vegetation communities resulting from changes in salinity regimes. All these effects might be expected to occur at relatively slow but steady rates, thereby having longer-term biological impacts compared to more rapid effects like changes in salinity.

Changes in depth can have dramatic effects as individuals grow. While small life stages depend upon low velocity and relatively shallow environments, larger fish tend to take advantage of deeper water. Therefore, where water crossing structures create reduced depths, larger fish would be expected to be the most sensitive to changes because they use deeper water depths (Harvey and Stewart 1991). Some authors have reported the converse -- i.e. the rapid return of larger highly mobile species when impounded areas are reconnected by deepening channels (Valentine-Rose and Layman 2011, Johnson and Whitesel 2012). As noted above, water crossing structures can also result in channel deepening, particularly in areas seaward of structures (Hood 2004). Under these conditions, structures forming impoundments or scour pools (Eberhardt et al. 2011) therefore can attract resident fish predators such as native Cutthroat and Bull Trout, and nonnative fish such as bass, pickerel, and Brown Bullhead. These larger piscivorous fish are less sensitive to changes in channel velocity imposed by some structures. In addition, scour pools at low tide can concentrate and strand small fish, making them vulnerable to predators (Sommer et al. 2005). For these reasons, estuarine structures may alter opportunities for piscivorous predators compared to their prey.

The first three processes all result in loss of inundated volume (area x depth). The cumulative effects of lost wetland areas via diking can be immense in some systems. For example, Simenstad et al. (1982) estimated that over 95% of the Duwamish River estuary has been lost

due to diking and filling, and Beamer et al. (2005) estimated that over 85% cumulative estuarine habitat loss through diking in the Skagit River estuary. Areal or volumetric loss from other intertidal water crossing structures has been harder to estimate due to the fact that many are partial barriers, but the effects can be inferred from careful mapping during restoration experiments. For example, Bowron et al. (2011) estimated that wetland area increased 800% after the removal of a culvert.

The consequence of lost habitat is lower capacity, or area for rearing by native species, which would tend to increase competition among resident individuals. It is often difficult to evaluate the overall cumulative effects of losses in capacity because it occurs at the population level, and the population effects of competition can be difficult to observe when focal species are at lower abundance. Nevertheless, capacity limitations as expressed by life history variation, limited recruitment, and smaller outmigration size have been observed by Beamer et al. (2005) for the Skagit delta. Furthermore David et al. (2015) used bioenergetic models to determine that increased urbanization appeared to reduce feeding opportunity and growth potential.

Changes in vegetation

The consequences of changes in salinity as well as subsidence are alterations to both the plant communities and fish communities supported in these environments. In the Northwest Atlantic, tidal restrictions have contributed to expansion of *Phragmites australis*, an invasive freshwater wetland grass (Burdick et al. 1996, Roman et al. 2002, Buchsbaum et al. 2006). Restoration of tidal flows has often resulted in a reduction in the areal coverage or height of *Phragmites* stems (Roman et al. 2002) and other saltwater-intolerant plants, and return of halophytes such *Salicornia* and *Spartina* (Sinicrope et al. 1990, Brockmeyer et al. 1997, Konisky et al. 2006, Smith et al. 2009, Van Proosdij et al. 2010) within 3-5 years of flow restoration. Bowron et al. (2011) determined that replacement of a broken culvert that highly restricted a tidal wetland with a much larger half-culvert bridge greatly changed the wetland vegetation within two years from freshwater and terrestrial plants to brackish flora such as *Salicornia*. Success of structure removal will strongly depend upon how much subsidence has occurred and what the resultant salinity regime becomes post restoration. For example, while some of the above studies have documented big changes in just a couple of years, a review by Williams and Orr (2002) of tidal reconnections in San Francisco Bay determined that revegetation could easily take longer than 20 years to become just 50% revegetated.

The consequences of change of intertidal vegetation communities upon fish and shellfish communities has been difficult in practice to determine, especially if any vegetation improves contributions of aquatic prey or refuge from predators. Nevertheless, Warren et al. (2002) examined multiple tidal restoration events in Connecticut and found that while fish assemblages changed to reference levels of abundance within 5 years of restoration, foraging opportunities through throughout the tidal range could take as long as 15 years to recover, in part due to

recovery of different types of vegetation. Kroon and Ansell (2006) compared species associated with tidal systems and reference environments, and found that fish abundance, biomass, and species assemblages strongly depended not only on the presence of a floodgate, but also on nutrient and aquatic weed levels.

Effects on habitat quality

Loss of connectivity additionally impacts a number of water quality parameters. One of the most universal changes in loss of tidal connectivity is a change in the salinity regime (Warren et al. 2002, Raposa 2008, Bowron et al. 2011, Boys and Williams 2012, Greene et al. 2013, Mora and Burdick 2013). These changes are often toward reduced salinity in estuarine environments, where tidal restrictions create impoundments that become more strongly influenced by freshwater inputs. As noted earlier, alterations toward lower salinity are not universal, particularly if water crossing structures occur low in the intertidal zone.

A direct consequence of tidal restrictions is longer residence time of water landward of restrictions. Longer water residence time has several consequences for habitat quality. First, waters are exposed to sunlight without being mixed by tidal or river flows. As a consequence, restricted habitats become warmer. For example, Gilmore et al. (1982) found cooler maximum temperatures in areas subjected to flow through a culvert compared to areas impounded by a dike. Greene et al. (2013) found that minimum temperatures were roughly 2° C warmer upstream of flapgates and self-regulating tidegates compared to reference channels. Ennis (2009) found no significant changes in mean temperature associated with tide gate replacement, but minimum temperatures did decline by several degrees, indicating that improved circulation had some benefits on water quality conditions.

Increases in residence time imposed by tidal restrictions also impacts nutrient levels and dissolved oxygen following typical processes of eutrophication. Generally, added nutrients that are not readily flushed out of intertidal areas promote rapid algal growth and die-off, and detrital bacteria subsequently remove dissolved oxygen in the water (Cloern 2001). It follows that lengthening of water residence should decrease oxygen levels and that changes in design or operation of tidal restrictions should increase flushing, thereby reducing nutrient levels and improving dissolved oxygen levels. All predictions have been born out through various studies. Kroon and Ansell (2006) found a strong positive correlation of nutrient levels with the presence of a floodgate, and Gee et al. (2010) found that restoration of tidal flow greatly reduced nutrient loading and could improve water quality of the entire system. They also note that gate function can strongly influence water quality parameters. Gordon et al. (2015) found significant impacts to dissolved oxygen upstream of floodgates in the Fraser River: upstream concentrations of DO could drop as low as 0.08 mg/l and were less than 30% that of reference systems. Areas immediately downstream of floodgates had 88% of the DO of reference sites, indicating a minor

decrement resulting from upstream oxygen levels. Beamer et al. (2013) determined that changes in operations at a floodgate in the Skagit delta resulted in improved temperature and dissolved oxygen levels. Likewise, Johnson and Whitesel (2012) measured the effects of installation of self-regulating tidegates upon water quality properties and salmon abundance. The new structures resulted in water quality parameters that did not exceed impaired thresholds.

The biological consequences to changes in water quality parameters are likely to first be reduced residency and increased emigration by mobile sensitive species. Species with low mobility should be expected to show growth decrements, loss in condition, or increased mortality. These predictions have been verified in the field. Bass (2010) reported that emigration rates PIT-tagged Coho populations increased as a function of temperature and decreased as a function of salinity. Scott et al. (2016) documented significant changes in species composition upstream of floodgates in the Fraser River, resulting in a 2.5x decline in salmonids and increase in nonnative species. As noted above, David et al. (2015) observed losses in feeding opportunity in more modified wetland areas. In areas with greater tidal restrictions, Eberhardt et al. (2015) observed changes in stable isotopes of muscle tissues of catadromous eels, indicating shifts in typical prey consumed. Finally, Dibble and Meyerson (2012) showed that fish rearing in areas with tidal restrictions suffered greater parasitism and reduced fat content, relative to reference systems. These effects were reversed after restoration of tidal flow.

Cumulative effects from passage & habitat loss

The suite of cumulative effects mentioned above can strongly affect the assemblages of fish and shellfish using these intertidal habitats. In Northeast Pacific estuaries, numerous examples exist of tidal restrictions facilitating inhabitation by nonnative warm-water game fish (Greene et al. 2013, Sagar et al. 2013, Scott et al. 2016). Spatial comparisons of tidally muted or restricted areas with reference sites have noted greater representation by estuarine and marine fish in reference sites (Pollard and Hannan 1994, Kroon and Ansell 2006, Ritter et al. 2008, Greene et al. 2013, Moreno-Valcárcel et al. 2016). Likewise, changes in community composition back toward estuarine- and saltwater-tolerant fish fauna after restoration of tidal flows has been found in many regions, including molluscs (Thelen and Thiet 2009), crustaceans (Boys and Williams 2012), and highly mobile fishes (Brockmeyer, Rey et al. 1997, Valentine-Rose and Layman 2011, Johnson and Whitesel 2012). Ritter et al. (2008) suggested that moderate tidal restriction had equivalent effects on fish community composition as no tidal restriction, but that severe tidal restriction had large effects on fish community composition due to cumulative effects of restricted movement, water quality differences, and changes to habitat structure. Nevertheless, the effects on communities can apparently be long-lasting. While a number of these studies have documented relatively rapid shifts in species assemblages following increases in tidal connectivity, Warren et al. (2002) concluded that returns of natural assemblages could take as

long as 5 years and still be impacted by limited foraging opportunities, and Boys and Williams (2012) found delayed responses in species composition resulting from culvert removal that lasted over 10 years.

These patterns suggest that restoration of connectivity can rapidly restore passage, but that the impacts to habitat quantity and quality can linger for much longer periods of time. Consequently, designs for water crossing structures that facilitate natural processes of inundation and drainage will likely to offer habitat benefits throughout their design life. Furthermore, efforts to improve connectivity at water crossings may face the cumulative challenge of restoring both passage and habitat, and may require larger-scale reclamation efforts to “jumpstart” the restoration. For example, Beamer et al. (2013) found that simple replacement of a floodgate was insufficient to restore reference densities of rearing juvenile Chinook Salmon, but that a combination of improved floodgate operations and extensive dike setbacks landward of the floodgate did result in restoration of reference-level rearing densities.

The combined effects of reduced connectivity from intertidal water crossing structures upon biological communities will naturally vary with shoreform, since shoreforms define the degree of tidal and freshwater influence. High in the intertidal (low salinities), reduced connectivity should be expected to shift communities toward freshwater resident vegetation, invertebrates, and fish, while shoreforms occurring lower in the intertidal zone may shift communities toward eutrophic euryhaline communities.

Summary

In conclusion, intertidal structures will act as barriers to small fish, larvae, and their food, more so than to larger and transient predators. The extent to which structures reduce connectivity across tidal cycles depends upon several factors affected by the design and operation of the structure. Given the cyclicity of tidal cycles, there are opportunities for estuarine-dependent organisms to access areas landward of intertidal water crossing structures, but even structures designed to ameliorate connectivity such as muted tidal regulators can reduce connectivity by 70-80%. While there is as of yet no standard for what low level of connectivity constitutes a barrier under these conditions, these changes do appear to have strong biological effects.

In addition to reducing passage, reduced connectivity impacts the intertidal habitat quantity and quality landward and seaward of structures. These effects in turn will have additional impacts upon the community of fish residing in areas affected by water crossing structures. Current design guidelines, best management practices, and other policies do not generally take these habitat changes into account, and instead focus primarily on passage capability.

5. Implications of existing science for future work on assessment, prioritization, and design

This review has several potential implications for identifying the systems and species sensitive to intertidal water crossing structures, monitoring the effects of structures on biota, and determining the extent to which intertidal water crossing structures generate significant impacts to fish using intertidal environments. Of course, science-based management decisions must be made in the context of law and policy as well as multiple land use planning decisions, issues that are largely outside the scope of this document. Based on our review of the scientific literature, the recommendations in Table 3 are consistent with the goal of minimizing the impact of intertidal water crossing structures on fish populations.

Table 3. Summary of management implications of the literature review.

Implications
Assessment of passability & habitat monitoring
Improve inventory of culverts and tidegates
Create inventories for habitat impairment as well as passage barriers
Provide indicators for habitat monitoring as well as fish passability
Implement effectiveness monitoring for fish passability and habitat protection
Design considerations for fish passage & habitat protection
Culverts:
Identify reference conditions to develop analog to stream simulation for intertidal application
Tidegates:
Avoid wherever possible
If removal is not possible, replace/retrofit with tidal muting structures
Optimize gate opening dimensions to tidal inflow
Bridges & causeways:
Avoid muting tidal influence
Road embankments:
Cross shoreforms above the head of tide and provide sufficient connectivity upstream to reduce the impacts to fish passage and habitat forming processes
When road crossing cannot be located at head of tide, use breaches in embankments
Incorporate considerations for climate change into design criteria
Decision support
Incorporate concepts of extended residence and changes in habitat into decision frameworks
Use hydraulic models to examine likely changes in habitat as well as in passage probability
Establish tiers of fish passage guidance to better highlight state of knowledge

Assessment of fish passage and monitoring habitat conditions

Inventory

A primary challenge to identifying barriers to passage and residence is determining the distribution and footprint of intertidal water crossing structures. However, the dual threat of impacts to fish passage and to habitat impairment suggests that in some cases, identifying a structure solely as a passage barrier may be insufficient to determine its impact to fish

populations. Previous efforts to inventory impacts of intertidal water crossing structures (Barrett et al. 2006) have classified structures as tidal restrictions without careful consideration of whether they result in impacts of fish passage or habitat. Multiple inventories (passage barriers & habitat impairments), or inventories with dual attributes can address these threats.

Currently, inventories utilize reference conditions for geomorphic shoreform patterns, elevational gradients, and presence of water crossing structures (Beamer et al. 2005, McBride al. 2009) along with traditional transportation corridor inventories (WDFW Salmonscape). However, water crossing structures in large river deltas can be more difficult to inventory than other habitat types due to habitat extent, channel complexity, lack of relief, and long history of modifications. Yet large river deltas are the areas in which estuarine resident fish sensitive to losses in connectivity and habitat quality are most likely to reside. Similarly, coastal confluences and lagoon shoreforms are also likely to harbor many barriers to both passage and residence due to the many large-scale transportation features people have engineered to create access along shorelines (see Fig. 1).

Fish passage and habitat assessment

Once water crossing structures that potentially impose barriers have been identified, rapid assessment techniques are often used to determine whether a structure is a fish passage barrier. These assessments are widely used in freshwater systems, where unidirectional flow facilitate detection of fish passage barriers (Kemp and O'Hanley 2010, Price et al. 2010, Barnard et al. 2015, Diebel et al. 2015). While rapid assessment of intertidal water crossing structures has been proposed by WDFW (see Fig. 11), lack of knowledge of fish behavior over time often limits the conclusions generated by a rapid assessment, potentially resulting in water crossing structures being miscategorized or labeled “unknown” in a barrier assessment. Furthermore, assessments that do not incorporate potential habitat impairments resulting from water crossing structures are likely to ignore significant threats to estuarine-dependent fish. Some rapid assessment techniques may therefore ignore impacts to habitat use and extended residence.

An integrated suite of indicators may facilitate rapid assessment. Table 4 summarizes a number of possible indicators of impacts to passage, fish habitat use, and habitat quality in quantity highlighted in our review. A subset of these could be measured in a rapid (one-day) assessment, while others are expected to take longer. In particular, tidal fluctuations and water quality aspects related to these are expected to take at least two weeks to capture neap and spring tidal variation resulting from lunar cycles, while presence of estuarine-dependent fish and their prey should be sampled over an entire rearing season to capture the possible influence of variability in timing of immigration into intertidal habitats.

Table 4. Indicators for assessing passage, residence, and changes in habitat quality and quantity landward of intertidal water crossing structures. Unless otherwise noted, most indicators should be measured above (landward) and below (seaward) a structure to determine if structure resulted in habitat or biological changes. Amount of time required for each assessment is one day (rapid), two weeks (tidal series), and rearing period for species of interest (season).

Attribute	Fish passage	Fish residency	Habitat quality	Habitat quantity	Assessment time
Structural attributes					
Tidegate present	X	X	X	X	Rapid
Vertical drop of structure at low tide	X				Rapid
Natural bed material present at structure	X	X			Rapid
Depth within structure at low & high tide	X	X			Rapid
Width of structure compared to channel	X		X	X	Rapid
Seaward edge of structure dewatered at low tide	X				Rapid
Backwater present landward of structure	X				Rapid
Scour pool present landward or seaward of structure	X			X	Rapid
Slope of structure	X				Rapid
Dynamic environment attributes					
Tidal action present above structure	X	X	X	X	Rapid
Vegetation differences			X	X	Rapid
Head of tide above structure				X	Rapid
Tidal vertical range		X	X	X	Tidal series
Peak ebb velocity at structure duration & timing	X	X	X	X	Tidal series
Peak flood velocity at structure			X	X	Tidal series
Maximum and minimum temperature		X	X		Season
Maximum and minimum salinity		X	X		Season
Minimum dissolved oxygen		X	X		Season
Biological attributes					
Algal blooms above structure			X		Rapid
Video/sonar of movements at structure	X				Tidal series
Presence of marine zooplankton	X		X		Season
Presence of small estuarine-dependent fish	X	X			Season
Recaptures of marked mobile fish	X	X			Season

Effectiveness Monitoring for Passability and Habitat Protection

Monitoring and assessment are often required to evaluate the extent to which particular water crossing structures create barriers to fish passage and residence. Likewise, effectiveness monitoring is often used to verify that changes to structures for the purpose of habitat mitigation and restoration actually produce the anticipated responses to hydrologic processes and fish

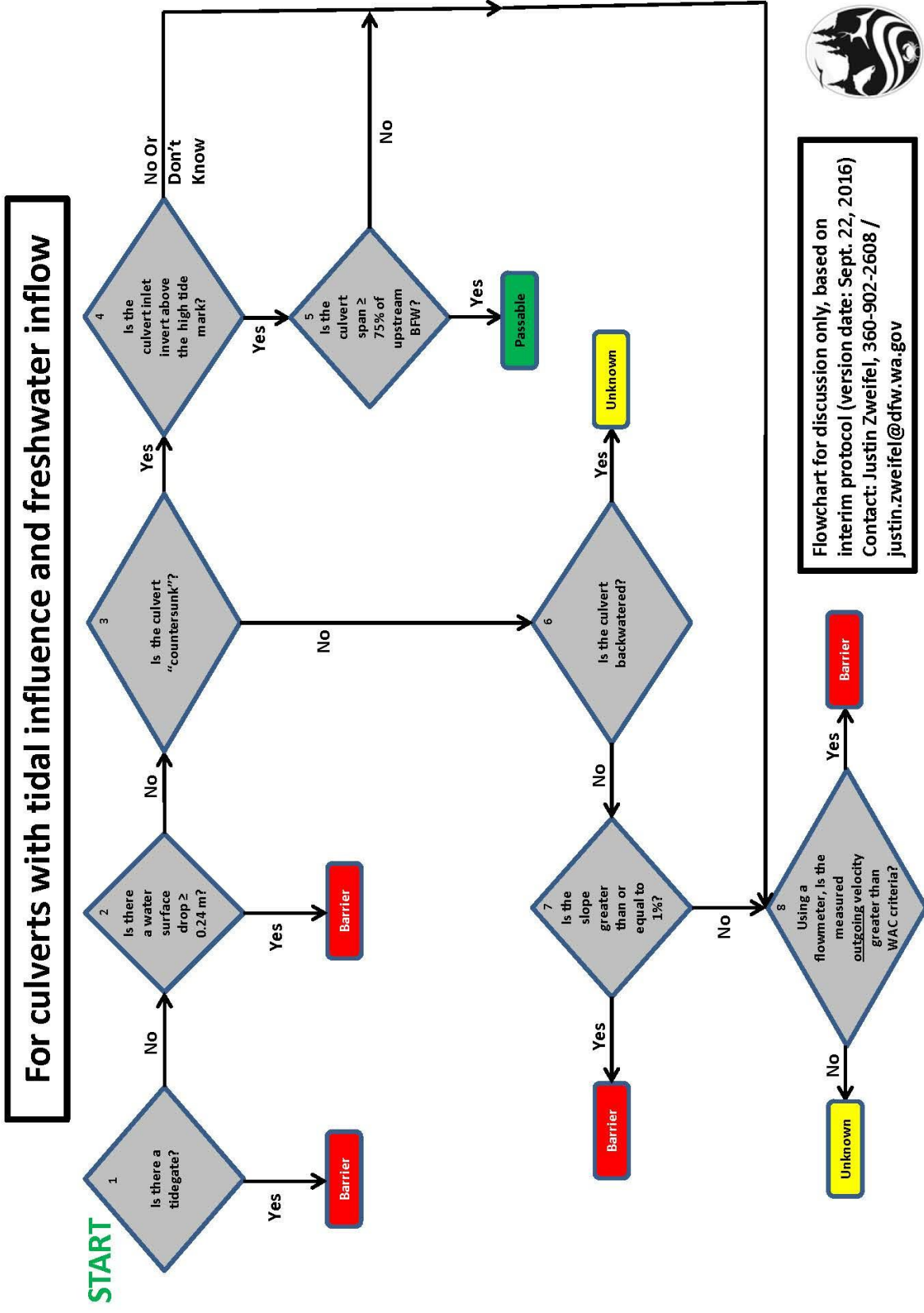


Figure 11. Draft decision framework developed by WDFW to assess whether culverts in intertidal habitats constitute fish passage barriers. This framework was adapted from WDFW’s freshwater culvert barrier assessments. It uses adult salmonids ($\geq 6''$) as the indicator species, and addresses upstream migration with assumed downstream passage. Culverts categorized as “unknown” receive additional engineering analysis for barrier determination.

habitat use (Roni et al. 2005). Following Table 4, our review suggests a number of characteristics that are useful to monitor and assess for determining whether intertidal structures or their modifications/replacements improve fish passage and residence and habitat quantity and quality.

Several techniques now exist for monitoring fish use to determine effects of structures on passage and residence. The best of these assessments take advantage of sophisticated sampling designs including seaward and landward sampling sites (Beamer et al. 2013, Greene et al. 2013, Scott et al. 2016), before and after comparisons when design or operation changes (Bowron et al. 2011), reference systems unaffected by structures (Boys et al. 2012, Beamer et al. 2013), or a combination of these. There are a variety of ways to monitor fish using these designs. The most basic include various strategies to sample fish for examining fish presence, abundance, and species assemblage (Valentine-Rose and Layman 2011, Boys et al. 2012, Beamer et al. 2013). Additional strategies include observational usage of cameras (Meynecke et al. 2008) and sonar (Doehring et al. 2011) to measure dynamic patterns of movement, and mark-recapture techniques using dyes (Eberhardt et al. 2011) and PIT tags (Meynecke et al. 2008, Bass 2010, McNatt et al. 2016). PIT tags can be especially useful because of their ability to track individuals, revealing patterns of related to fish size (Bass 2010) and other individual characteristics as well as accumulated movements of the population (Kirk and Caudill 2016).

There is an inherent trade-off in information quality versus effort for these techniques. Sonar and mark-recapture techniques often require sophisticated equipment or extensive recordings of observations. Periodic fish sampling can be highly variable depending on the timing of sampling, and metrics like total fish abundance or density or species richness differences tend to be inconsistent metrics of similarity of sites to reference conditions. While some studies have certainly showed these metrics to be positively associated with connectivity or higher quality habitat (Pollard and Hannan 1994, Taylor et al. 1998, Kroon and Ansell 2006, Boys et al. 2012, Scott et al. 2016), others have produced equivocal results (Roman et al. 2002, Raposa and Talley 2012) or even the reverse pattern (Raposa 2002, Buchsbaum et al. 2006, Van Proosdij et al. 2010, Moreno-Valcárcel et al. 2016). These ambiguous results arise in part because these summary metrics do not distinguish tolerant or nuisance species from species sensitive to passage constraints or water quality declines. Focusing on sensitive estuarine-dependent species can help lessen the potential for conflicting results (Eberhardt et al. 2011, Greene et al. 2013).

Design and operation standards for intertidal water crossing structures

In many coastal areas, there is a strong tradeoff between societal values for habitat protections for estuarine species and for opportunities for shoreline access and development by people. This tradeoff invariably will result in various proposed structural designs that attempt to achieve a compromise between these values. This review has important implications for planning, design,

and operation of water crossing structures that achieve habitat protections by improving fish passage and extended residence in intertidal environments.

Barnard et al. (2013) described several levels of analysis to evaluate potential impacts of intertidal water crossing structures: 1) qualitative assessments, 2) engineering, biological and geomorphological assessment, and 3) quantitative hydraulic modeling. From a design perspective, the authors indicated that level (1) deals primarily with the distance of a water crossing structure from the head of tide and therefore interference to the tidal prism. Level (2) is a more in-depth analysis and focused on the degree to which structures create barriers to habitat-forming processes, while level (3) quantifies and predicts system-level changes in multi-channel networks (see *Decision Support*, below). This review suggests qualitative assessments can and should go beyond characterization of impacts to tidal prism, particularly because of compounding impacts related to freshwater drainage. Furthermore, our review pointed toward several metrics that might inform a richer understanding of the multiple potential impacts of structures (Level (2)), even if only rapid assessment techniques are used. Our review also suggests that some design standards may be specific to different structures, and we highlight these below.

Culverts

Price et al. (2010) noted that many permitted freshwater culverts actually functioned as upstream passage barriers and urged better mechanisms for compliance by project sponsors. Culverts are often undersized, installed with severe slopes, or not counter-sunk to provide natural stream bed material throughout the length of the culvert. To remedy these problems, Washington State has established “stream simulation” and “No-slope” designs for culverts (Bates et al. 2003, Barnard et al. 2013) as design approaches in freshwater applications.

Neither stream simulation nor no-slope designs have been explicitly reviewed for use in intertidal environments, although both would likely result as “passable” or “unknown” in the draft decision framework proposed for rapid assessment of intertidal barriers (see Fig. 11). Most assessment techniques would likely not detect significant reductions in passability for either adults or juvenile estuarine-dependent fish. Barnard et al. (2013) point out that culverts, in combination with dikes or other embankments raising water crossings structures above intertidal wetlands, would constrict tidal flows into channels, thereby reducing sheet flow over the marsh surface at higher tides and greatly reducing the tidal prism. As noted by Hood (2004), such culvert designs would therefore act as barriers to natural habitat forming processes, and could lead to channel filling landward of structures and greater erosion on the seaward side. In addition, reduced tidal heights created by tidegates will likely result in lower sheet flow across deltas at high tides, disrupting delivery of materials such as sediment and detritus.

Of course, no-slope and stream simulation design approaches are focus on fluvial systems and are unlikely to apply directly to intertidal situations. In particular, tidal action of sheet flow over the intertidal marsh surface reduces the applicability of these techniques. Additional design approaches that incorporate overland flow in addition to within-channel flow can better evaluate effects of culverts and associated engineering upon fish and their intertidal habitats.

Tidegates and tidal muting structures

As noted above, tidegates and tidal muting structures are often used where coastal agriculture or urbanization imposes restricted tidal elevations landward of water crossing structures. Our literature review suggests that most such structures result in significant barriers to passage by juvenile fish, reduce passage opportunities for adults, impact residence patterns and restrict intertidal habitat forming processes. For pre-existing tidegate structures, research suggests that during replacement or retrofit, design features and operation guidelines can ameliorate conditions for estuarine-dependent species. For example, tidal muting structures can provide improved landward passage opportunity and tidal flushing over flapgate designs (Greene et al. 2013). Muting devices that account for larger portions of the channel width can perform well, particularly if efforts are made to optimize gate opening dimensions when they allow tidal inflow (Beamer et al. 2013). In addition, efforts to restore and reconnect channel systems landward of structures can help ameliorate habitat conditions for fish that are able to navigate the muted device.

Bridges and causeways

Barnard et al. (2013) recommended the use of bridges and elevated causeways to reduce the impacts of intertidal water crossings. Reducing the footprint of these structures (including associated road embankments) within a given shoreform will tend to maximize habitat forming processes. Bridges at lagoons and embayments often use spit features for a roadway, reducing the ability of the spit to function naturally. Even bridge designs that might be considered “unrestricted” can nevertheless result in changes to tidal dynamics. For example, Khangaonkar and Wang (2013) determined using sophisticated hydrodynamic models that construction of a floating bridge across Hood Canal, a large deep fjord of Puget Sound WA, restricted wind-based and tidal currents, thereby increasing water residence time within the entire fjord by 10%. These patterns may contribute to a large increase in apparent predator-mediated mortality in migrating juvenile Steelhead populations migrating under the bridge (Moore et al. 2013).

Roads, railroad grades, dikes and levees

All these structures create significant linear barriers to freshwater outflow and tidal inflow. Alternative designs that cross shoreforms above the head of tide and provide sufficient connectivity upstream will reduce the impacts to fish passage and habitat forming processes (Barnard et al. 2013). If upstream crossing is not possible, designs that include breaches in the structure to allow connectivity can help restore some function. However, Hood (2015) found that

even tidal restoration sites with multiple dike breaches had one fifth the number of outlets to distributaries than reference marshes, emphasizing that dike breaches likely result in much lower connectivity over time and concomitant reductions in tidal prism across marsh surfaces.

Incorporating impacts of climate change in design

Multiple climate change impacts in freshwater and marine systems will likely pose cumulative hazards in Washington State (Table 5). Freshwater impacts include increases in bank-full flows, 100-year floods, and resulting channel widths, and increased spring and summer temperatures (Snover et al. 2013, Wilhere et al. 2016). Marine impacts include sea level rise, changes in salinity, ocean acidification, and increases in marine water temperatures (Burkett 2012, Snover et al. 2013). These changes are expected to influence both passage and residence of estuary-dependent species, and also pose hazards to the structures themselves. Some of these hazards are expected to result in increased likelihood of structure failure, while others may result in long-term degradation, shortening the design life of structures. In light of these increased vulnerabilities to both fish and structures, standards of design and operation of intertidal water crossing structures deserve re-evaluation.

Wilhere et al. (2016) examined how climate impacts on 100-year floods, bank-full river discharge, and bankfull width in Washington State might affect design guidelines for culverts at stream and river water crossings. The conclusions made in their report for WA maritime ecoregions are relevant for determining how changes in freshwater systems might influence intertidal water crossing structures. They found that in these areas, bank-full widths accommodating changes in river flow will increase by 12.1% on average by the 2080s, the bank-full discharge will increase by 26.56%, and will also be subject to modest increases (2-30%) in the 100-year flood discharge. These increases would argue for concomitant increases in culvert diameter to maintain stream-simulation or no-slope guidelines. In order to reduce risk of failure due to high flows, they found that for most of the Puget Trough and the Washington Coast, stream bank-full widths of greater than 13 feet should be considered for bridge crossings rather than culverts. This follows the guidelines by Barnard et al. (2013) to use bridges when bank-full width is greater than 15 feet by applying future bankfull width projections due to climate change.

Similar arguments can be made with respect to potential catastrophic failure of intertidal water crossing structures from sea level rise and storm surge, but design guidelines for these situations have yet to be systematically modeled and need to be tailored to different levels of impact on the coast compared to within the estuaries of Puget Sound and the Columbia River. Flood risks in estuaries and other shoreforms will be exacerbated by sea level rise such that by the 2050s, the historical 100-year water level is expected to be exceeded every year. These flood events are expected to be on average 10 inches higher by the 2080s. Therefore, water-crossing structures

Table 5. Climate impacts and primary hypothesized effects upon both fish and structures in intertidal environments. Projected changes (ranges provided, or coefficient of variation in parentheses) are in terms of change in the 2080s compared to present. The URL for the NOAA climate change web portal is <https://www.esrl.noaa.gov/psd/ipcc/ocn/ccwp.html>.

System effected	Climate impact	Ecoregion	Projected change on WA coasts in 2080s	Reference	Primary effect on fish	Primary effect on structures
Freshwater	↑ Peak discharge (100-year flood)	Puget Sound WA Coast	28.4% (44.6 %) 24.1% (49.5 %)	Wilhere et al. 2016	Passage	Failure
	↑ Bankfull discharge (2-year flood)	Puget Sound WA Coast	17.3% (24.8 %) 22.6% (30.8 %)	Wilhere et al. 2016	Passage	Failure
	↑ Bankfull width during 2-year flood	Puget Sound WA Coast	8.3 % (24.2 %) 10.7% (28.4 %)	Wilhere et al. 2016	Passage	Failure
	↑ 7-day max spring/summer stream temperature	Puget Sound WA Coast	4 – 4.5 ° F 4 – 4.5 ° F	Snover et al. 2013	Residence	Degradation
Marine	↑ Sea level at high tide	Puget Sound WA Coast	4 – 56 in 4 – 56 in	Snover et al. 2013	Residence	Degradation
	↑ Coastal 7-day max summer water temperature	Puget Sound WA Coast	3.6 – 5.8° F 4.3 – 5.8° F	NOAA web portal	Residence	Degradation
	↓ average salinity	Puget Sound WA Coast	0.3 – 0.5 psu 0.1 – 0.5 psu	NOAA web portal	Residence	Degradation

would have to be designed to have higher capacity to avoid over-topping and surface erosion, along with avoidance of impacts to habitat quality.

Given projected sea level rise on the order of 4 to 56 inches by the 2080s in much of the Northeast Pacific, better design, operations, and adaptations are needed to address a serious risk of estuarine habitat loss resulting from sea level rise. In undeveloped areas, estuarine habitats can migrate landward as sea level rises, but these opportunities are cut off by tidal restrictions such as dikes (Orr et al. 2003, Torio and Chmura 2013). In large river deltas, the combined effects of sea level rise, wave generated erosion, and “coastal squeeze” by diking appears to make delta fronts vulnerable to marsh retreat without replenishment from sediments that bypass the delta when carried out in restricted channels (Hood et al. 2016). Due to fixed culvert elevations, muted channel systems are expected have reduced capability for sediment accretion relative to reference systems (Vandenbruwaene et al. 2011). Sediment accretion is the buffer for marsh resilience. Hence, if structures impact sediment transport, they may likewise impact the longevity of the marsh.

Projected changes in salinity and temperature also have important implications for water quality of landward areas impacted by structures. In Northeast Pacific coastal systems, salinity in shoreline habitats is expected to shift with freshwater hydrographs and therefore become less saline in the winter and more saline in the spring as summer as winter snow accumulation diminishes. The impacts of these changes will be most pronounced in the spring and summer, when changes in both salinity and both freshwater and marine temperatures can be further exacerbated by tidal restriction. Hence, climate impacts on salinity and temperature are projected to further reduce water quality for estuarine-dependent species in areas restricted by structures. These changes can also lead to deterioration of structures caused by increased corrosion of metals and changes in porosity of concrete.

Multiple climate impacts therefore will likely increase potential risk of incremental degradation and catastrophic failure of existing structures. In addition, multiple climate impacts could lead to reduced habitat quality for estuarine-dependent species as well as significant losses in passage if climate impacts lead to passage barriers or structure failure. In both cases, standards of design and operation need to be set to future expectations of inundation levels, flood magnitudes, poorer water quality, and frequency of extreme events. For example, to accommodate sea level rise alone, cross sectional changes in dikes (height and width) will need to be 40-86% greater than current dikes to offer the same protections in the 2080s (Jonkman et al. 2013). This change would result in an 18-35% increase in the footprint of a dike landward or seaward of its existing footprint. As noted by Reagan (2015) and Wilhere et al. (2016), the perceived societal need for these changes depend on assumptions of design life and discount rate of future impacts, which will also track uncertainty in climate change projections. Nevertheless, better guidance on improved design and operation that is resilient to climate change should offer longer term benefits for estuarine dependent fish and shellfish species as well as reduce the risks of infrastructure failure.

Decision support and tiers of guidance regarding fish passage and rearing

Improved decision support tools can help translate scientific knowledge into guidance on assessment, design, and operation of intertidal water crossing structures. Here we present some examples of potential improvements to decision support, based on review of the literature.

Decision frameworks

Decision frameworks can be used to structure the directions taken by management given existing information. For example, Figure 11 illustrates how a draft decision framework proposed for Washington State to rapidly assess whether water crossing structures act as barriers to migration by anadromous fish in intertidal areas. The literature review suggests that this type of decision framework would meet standards for fish passage for a subset of species that utilize intertidal habitats. However, the framework would probably not help determine whether particular

structures pose a barrier to extended residence or habitat forming processes. This framework would likely point to further assessment for structures that did not fall neatly into “barrier” or “passage” categories. Hence, decision frameworks for intertidal environments will likely include more steps than those for freshwater systems. A combined barrier assessment/habitat impact decision framework would better address these issues.

Assessments based on decision frameworks can help identify potential barriers, but additional effort is required to prioritize which structures need to be modified to meet passage standards (WDFW 2009). Kemp and O’Hanley (2010) and O’Hanley and Toberlin (2005) propose optimization techniques for prioritizing improvements in networks with multiple potential barriers. In river networks, downstream barriers receive higher priority because they block larger portions of watersheds. This system for prioritization could in principle be applied to intertidal areas as well, although the rules might be expected to be modified, given the existence of two-way flow network in intertidal environments.

Hydraulic models

Hydraulic models can be useful when evaluating crossing structures for passability and determining appropriate design features of structures. Modeling tools vary from simple hydraulic equations and regression models to one-, two-, and three-dimensional hydraulic models. One-dimensional (1d) models are useful for water surface profiling, determining depth averaged cross-sectional velocity normal to the direction of flow. Two-dimensional (2d) models are grid-cell based and can provide output representing depth-averaged velocity in different grid cells, generating fine-scale predictions that are independent of the cross section as a whole. Similarly, three-dimensional (3d) models add a depth component to the vector output that allow for evaluating complex situations where flow can split vertically around structures. Because of the added depth dimension, these models can address vertical stratification. For any of these models, boundary conditions (e.g., freshwater and marine inputs to the environment modeled) can have large effects on model output. In most cases, unsteady state boundary conditions are utilized at a minimum for the downstream condition, allowing the model to computationally evaluate tidal stage with time series input.

Of course, all models depend upon their assumptions and input parameters, and few standards are currently established regarding input parameters guidelines for hydraulic models. In these contexts, the above literature review provides some starting points for model refinement. The weight of evidence suggests that reduced connectivity can arise in intertidal environments for multiple reasons (Fig. 7) and reduce opportunities for both passage and residence (Fig. 8 & 10). Hence, one-dimensional models may be limited in their application to intertidal water crossing structures, aside from informing assessment of whether structures likely constitute barriers (e.g., if average velocity through a structure is equivalent on tidal flood and ebb). Furthermore, if impacts to habitat quality and quantity are considered, models that are complex enough to

examine changes in depth as well as in-channel and within-structure velocities will provide more useful outputs than hydraulic data.

Many hydraulic assessments are performed using a “design flow” or tide, a standard used to compare different design or operation scenarios that inform whether fish passage and/or rearing is facilitated. When considering the most appropriate design tide, one option would be to use an extreme spring tide associated with an extreme flood freshwater contribution component. While this pairing would minimize failure risk of structures, this scenario is likely an unrealistic approach because the combination of the two low probability events occurring simultaneously would be a very rare occurrence.

An alternative approach could be to select the design tide similarly to selecting design flows in freshwater systems, where the 10% exceedance flow is utilized for establishing fish passage criteria for adult salmonid passage in Washington State (Barnard et al. 2013). This value was used by WDFW in recognition that passage of 100% of individuals in a population is an unrealistic goal. Although applicability of a 10% exceedance flow has not been thoroughly reviewed for intertidal systems, WDFW has completed a preliminary analysis to compare the recurrence probability of many tidal stations within Washington State (P. Smith, unpublished data). Across most tidal stations, the 10% exceedance resulted in a tidal range from MLLW to MHHW. This exceedance level could be utilized as a preferred design tide along with the addition of the 10% exceedance flow for a freshwater flow to evaluate parameters for adult passage. A freshwater contribution component for an analysis of juveniles in the intertidal would likely need to be established, and exceedance thresholds would need to consider both high and low tidal flows to realistically assess impacts at both ends of the tidal spectrum. Hence, this kind of approach may have promise but will need additional analysis.

To date, most hydraulic models have focused on the outcomes of particular design solutions (“What dynamics result from Design X?”). Efforts to improve design and operation demand hydraulic analyses of multiple design or operation scenarios, and models that can readily accommodate these scenario runs (Boumans et al. 2002). Key parameters emerging from field efforts which will help inform hydraulic models are flushing rates, scour (upstream and downstream), sedimentation and deposition, and effects of vegetation. This would help address several fundamental problems related to evaluating structure design and operation, such as optimal channel dimensions degree of gate openings for tidal muting structures, and habitat structural complexity (Bates et al. 2003, Bates 2006).

Tiers of guidance for fish passage and residence

As noted above, simple decision frameworks cannot address all questions regarding the impacts of water crossing structures simultaneously. Following from the results of this review, several tiers of guidance could be used to address inventory of potential barriers, assessment of their

impacts, and determine optimal design guidelines. These tiers should be considered increasing levels of information addressing whether existing structures constitute significant barriers or rearing habitat degradation, and whether proposed structures are designed to address these issues. As a starting point, four possible tiers of guidance addressing different levels of complexity are considered.

Tier I. The most basic level of information addresses inventorying and assessment of whether structures are potential barriers to fish upstream movement, or in a hydraulic analysis, whether particular designs are likely to be passable, based on fixed design conditions (e.g., design flows).

Tier II. A higher level of guidance can address how passage is dynamically affected, and therefore incorporates ranges of change. In the context of intertidal structures, tidal range, and its impacts upon velocity, would naturally fall into Tier II levels of information.

Tier III. This level of guidance would address potential changes in residency and therefore impacts to habitat quality and quantity. Guidance of this sort will require additional capabilities to model temporal change landward of structures.

Tier IV. This level of guidance deals with long-term changes associated with structures, including impacts of climate change. Guidance at this level would require climate projections as well as careful considerations of uncertainty.

Examples of how this might look are summarized in Table 6. For example, basic passage assessments (Tier I) might require simple decision frameworks or regression models to determine whether intertidal structures are likely passable under 10% exceedance conditions and rapid assessments of basic parameters influencing passage. These would be used to determine passability for fry-sized fish, which differentially use intertidal environments. More detailed assessments concerning fish passage (Tier II) could require two-week time courses under high and low river flow conditions to obtain temporally dynamic predictions of passage using hydraulic models. Evaluation of both fish passage and habitat limitations imposed by structures (Tier III) would require dynamic simulation in multiple seasons (e.g., winter and summer) and track multiple habitat parameters. Long-term evaluations that include climate impacts (Tier IV) would evaluate changes to fish and their habitat in the context of projections of sea level rise and high and low river flow changes. In addition to evaluating fish passage and residence, these models could be used to examine design life of structures.

Table 6. Information relevant for the four proposed tiers of guidance for intertidal water crossing structures.

Tier characteristic	Tier 1 – inventory and assessment	Tier 2 – fish passage	Tier 3 – fish passage and rearing	Tier 4 – climate impacts
Preferred type of hydraulic model	None or regression	1d or 2d	1d - 3d	1d - 3d
Design tide(s) and river flow	10% exceedance tide 10% exceedance flow	Spring and neap tide series High and low flows	Two-week continuous simulation, winter and summer High and low flows	Winter king tide projection High flow projection Low flow projection
Target fish	Salmonid fry (<50 mm in length)	Salmonid fry	Salmonid fry Larval flatfish	Salmonid fry
Parameters considered	Maximum vertical drop Minimum water depth Slope threshold Gate presence Velocity	Vertical drop range Water depth range Slope threshold Gate dynamics Velocity dynamics	Vertical drop range Water depth range Slope threshold Gate dynamics Velocity dynamics Upstream temperature Upstream oxygen Upstream depth Upstream flooded area	Same as parameters in Tier 3
Biological endpoints	Likely passage	Modeled passage probability	Modeled passage probability Modeled residence probability	Modeled passage probability Modeled residence probability
Other endpoints				Design life of structure

6. Information gaps & recommendations for research

Our knowledge of intertidal water crossing structures and their effects on estuary-dependent species is still in its infancy. The lack of knowledge is a combination of the inherent dynamics of intertidal environments, the species that use them, and the lack of information on the water crossing structures affecting them. In addition, information on impacts of climate change and socioeconomic considerations is still developing. Here we lay out what we see as the most important information gaps (Table 7), which can be addressed with additional research.

Table 7. Summary of information gaps.

Category	Data Gap
Fish species and habitats	Natural processes and reference intertidal systems Intertidal residency patterns of different fish during juvenile and adult stages Importance of distributed lagoons and coastal confluences for intertidal fish communities Use of estuarine habitats by Eulachon and lamprey
Fish movement and residency	Fish movement related to tide cycle & water column Effect of reduced fish passage and/or habitat connectivity on fish populations Impact of delayed fish movement on fish reproductive success or survival
Structures	Inventory of culverts and tidegates Dike and embankment blockages of coastal confluences Effect of turbulence, light and noise on fish passage Combined effects of structures upon passage and habitat of fish communities
Climate impacts	Climate related changes in impacts of structures upon fish passage or residency Changes in design life of structures resulting from combined climate impacts to freshwater and marine conditions

Species and habitats

Numerous challenges of studying behavioral and population related questions in tidally influenced areas where fish are dynamic and temporary residents have resulted in information gaps. For example, Simenstad et al. (1991) identified multiple challenges to evaluating whether estuarine habitat modifications impact salmonids, including lack of baseline data, habitat changes are still transitional, lack of specifically defined biological response metrics, low statistical rigor, and lack of data on net survival resulting from modifications. In these situations, isolating the population effects of water crossing structures is extremely challenging.

Furthermore, some ecological processes (e.g., changes in passage) may have rapid effects, whereas others (e.g., loss of channel depth, declines in water quality) may be incremental over many years. Hence, it is difficult to conclude from existing studies what are the relative effects of passage, and reduced habitat quality or quantity upon fish populations potentially affected by water crossing structures.

Fine scale fish movements related to tide cycle

Whereas the previous focus on freshwater barriers has dealt primarily with upstream movements of adults, assessment of whether intertidal structures constitute barriers should focus on access by juvenile fish for rearing, and the benefits of residence in intertidal environments. For these issues, understanding impacts to dynamic movements of rearing fish is critical. Key uncertainties in intertidal environments related to passage concern the benefits of passage versus the costs of holding until tidal conditions are favorable. For juveniles, a likely trade-off is the benefit of rearing in areas influenced by structures versus the risk of lost growth potential and predation risk incurred by “waiting” for favorable tides. For adults, the likely trade-off is the benefit of reproduction versus the risk of mortality from predators or local stressors (e.g. temperature spikes). A common assumption of fish passage guidelines in intertidal water crossing structures is that adult fish can wait for favorable passage conditions for days or even weeks. However, predation impacts resulting from attraction of predators to concentrated numbers of adults at a barrier have been readily observed at fish ladders in rivers and remain a policy challenge for marine mammal management, so these considerations may not be trivial.

Data gaps exist for many anadromous species, what habitats they use, and guidelines for fish passage. Important habitat-specific gaps are lagoons and small stream confluences, systems that are relatively small but which are broadly distributed on the landscape. Among culturally important species on the decline, the biggest data gaps exist for Eulachon and lamprey.

Water crossing structures

Lack of information pervades our understanding of the distribution of intertidal water crossing structures, how they impact habitat quality and quantity, and design alternatives. For example, there is as of yet no consistent inventory of intertidal culverts and tide gates, nor a firm knowledge of the number of coastal confluences blocked by roads and railroad grades.

Central questions concerning the ecological impacts of water crossing structures concern how structures perform compared to some reference condition. Due to the broad footprints of various water crossing structures, few reference systems remain. For example, in their effort to examine the effects of tide gates on fish, Greene et al. (2013) attempted to compare tide gate sites with reference systems, yet most reference sites were variously restricted by roads, dikes, and upstream tide gates. Strategies for addressing this issue include identifying and protecting these

sites, and pursuing restoration of particular sites for use as references. Of course, the latter strategy will depend upon their recovery rates from legacy disturbance.

Much of the focus of water crossing structures is the extent to which structures result in barriers to movement. There is much less information on the multiple effects of structures on habitat quality and quantity, and what designs and operations best ameliorate these longer-term impacts upon fish. Furthermore, much of the focus on water crossing structures has dealt with impacts of structures on velocity fields, and much less on how other structure-dependent attributes such as acoustic noise and lighting changes affect passage behavior.

Efforts to address many of these issues remain challenging and would be facilitated by experimental studies using mark-recapture techniques and test bed systems (Richmond et al. 2007), where the influence of connectivity could be explicitly manipulated to determine impacts to individuals using these systems. This experimental environment could facilitate tests of different structural designs, and help provide data on important hydraulic parameter values associated with different designs. Test beds also can facilitate careful study designs, so that manipulations of connectivity can be directly uncoupled from extraneous temporal or spatial noise. Without such test beds, existing structures can be used as sites to test certain questions, although spatial and temporal sources of variation become of greater concern, and conclusions may more be limited to those specific sites.

As a consequence of these information gaps, some surprisingly simple questions regarding the impacts of intertidal water crossing structures remain unanswered, such as:

- If the tide flow is going in the same direction as fish movement, do structures act as barriers?
- What are the consequences if fish are temporarily delayed in moving through culvert/tidegate?
- What are the consequences of seaward movements to sites with higher habitat quality compared to areas landward of water crossing structures?

Climate impacts

Determining how emerging effects of climate impacts will influence fish interactions with intertidal water crossing structure require site-specific projections of future climate impacts. While some of these datasets exist (e.g., maps of changes in freshwater extremes, Wilhere et al. 2016), others such as sea level rise and coastal storm projections are notably absent. Evaluating the combined influence of various climate impacts upon systems affected by intertidal water crossing structures will require tools to project multiple climate impacts in a probabilistic manner (Hamman et al. 2016, Wilhere et al. 2016).

Sea level rise is predicted to reduce estuary habitat for juvenile salmon and flatfish (Kennedy 1990) as slough systems become deeper, more saline, and more simplified. Restoration options

will likely become fewer and tidal muting devices may increasingly be seen as a necessary 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 muted tidal regulators that maintain rearing habitat without significantly compromising adjacent land use values. In addition, entrepreneurial efforts should focus on engineering or restoration designs that combine high connectivity and minimal maintenance.

Determining whether particular water crossing designs are likely cost-effective solutions for certain sites depends on our assumptions about design life (Reagan 2015), which depend not only on catastrophic events but also on rates of incremental deterioration that can be accelerated by various climate impacts (Douglass et al. 2014). Future work addressing potential rates of infrastructure deterioration in aquatic systems are necessary to better incorporate potential future deterioration-related costs into costs of structure redesign and replacement.

References

- Able, K. W. (2005). A re-examination of fish estuarine dependence: Evidence for connectivity between estuarine and ocean habitats. *Estuarine Coastal and Shelf Science* 64: 5-17.
- Adamus, P. (2005). Science review and data analysis for tidal wetlands of the Oregon Coast. Part 2 of a Hydrogeomorphic guidebook. Coos Watershed Association, US Environmental Protection Agency, Oregon Department of State Lands, Salem, OR. 216 pp.
- Anisfeld, S. C., M. J. Tobin and G. Benoit (1999). Sedimentation rates in flow-restricted and restored salt marshes in Long Island Sound. *Estuaries* 22: 231-244.
- Barnard, R. J. (2003). Evaluation of the Stream Simulation Culvert Design Method in Western Washington, a preliminary study. Washington Department of Fish and Wildlife, Olympia, WA: 25 pp.
- Barnard, R. J., J. Johnson, P. Brooks, K. M. Bates, B. Heiner, J. P. Klavas, D. C. Ponder, P. D. Smith and P. D. Powers (2013). Water Crossing Design Guidelines. Washington Department of Fish and Wildlife, Olympia, WA. 297 pp.
- Barnard, R. J., S. Yokers, A. Nagygyor and T. Quinn (2015). An Evaluation of the Stream Simulation Culvert Design Method in Washington State. *River Research and Applications* 31: 1376-1387.
- Barrett, S. B., B. C. Graves and B. Blumeris (2006). The Mount Hope Bay tidal restriction Atlas: Identifying man-made structures which potentially degrade coastal habitats in Mount Hope Bay, Massachusetts. *Northeastern Naturalist* 13: 31-46.
- Bass, A. L. (2010). Juvenile Coho Salmon movement and migration through tide gates. Masters Thesis, Oregon State University, 125 pp.
- Bates, K., B. Bernard, B. Heiner, J. Klavas and P. Powers (2003). Design of road culverts for fish passage. Washington Department of Fish and Wildlife, Olympia, WA. 112 pp.
- Bates, K. M. (2006). Culvert Design for Fish and Other Aquatic Organisms, Southeast Fish and Aquatic Species Barrier Assessment Workshop.
- Beamer, E., R. Henderson, C. Ruff and K. Wolf (2013). Juvenile Chinook Salmon utilization of habitat associated with the Fisher Slough Restoration Project, 2009 - 2013. Skagit River System Cooperative Report Prepared for the Nature Conservancy, La Conner, WA, 34 pp.
- Beamer, E., A. McBride, C. Greene, R. Henderson, G. Hood, K. Wolf, K. Larsen, C. Rice and K. Fresh (2005). Delta and nearshore restoration for the recovery of wild Skagit River Chinook Salmon: linking estuary restoration to wild Chinook Salmon populations. Supplement to Skagit Chinook Recovery Plan, Skagit River System Cooperative, LaConner, WA.

- Beamer, E., A. McBride, R. Henderson and K. Wolf (2003). The importance of non-natal pocket estuaries in Skagit Bay to wild Chinook Salmon: an emerging priority for restoration. Skagit System Cooperative Report, LaConner WA.
- Beamer, E., W. Zackey, D. Marks, D. Teel, D. Kuligowski and R. Henderson (2013). Juvenile Chinook Salmon rearing in small non-natal streams draining into the Whidbey Basin. Skagit River System Cooperative. LaConner, WA. 74 pp.
- Beamish, F. W. (1978). Swimming capacity. Pp 101-187 in W. S. Hoar and D. J. Randall eds., Fish Physiology. Academic Press, NY.
- Beauchard, O. (2012). Tidal freshwater habitat restoration through controlled reduced tide system: A multi-level assessment Ph. D. Dissertation, University of Antwerp, 97 pp.
- Beauchard, O., S. Jacobs, T. J. S. Cox, T. Maris, D. Vrebos, A. Van Braeckel and P. Meire (2011). A new technique for tidal habitat restoration: Evaluation of its hydrological potentials. Ecological Engineering 37: 1849-1858.
- Beck, M. W., K. L. Heck, K. W. Able, D. L. Childers, D. B. Eggleston, B. M. Gillanders, B. Halpern, C. G. Hays, K. Hoshino, T. J. Minello, R. J. Orth, P. F. Sheridan and M. R. Weinstein (2001). The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. Bioscience 51: 633-641.
- Behrens, D. K., F. A. Bombardelli and J. L. Largier (2016). Landward Propagation of Saline Waters Following Closure of a Bar-Built Estuary: Russian River (California, USA). Estuaries and Coasts 39: 621-638.
- Bennett, T. R., P. Roni, K. Denton, M. McHenry and R. Moses (2015). Nomads no more: early juvenile Coho Salmon migrants contribute to the adult return. Ecology of Freshwater Fish 24: 264-275.
- Bennett, W. A. and J. R. Burau (2015). Riders on the Storm: Selective Tidal Movements Facilitate the Spawning Migration of Threatened Delta Smelt in the San Francisco Estuary. Estuaries and Coasts 38: 826-835.
- Boehlert, G. W. and B. C. Mundy (1988). Roles of behavioral and physical factors in larval and juvenile fish recruitment to estuarine nursery areas. American Fisheries Society Symposium 3:51-67.
- Bottom, D. L., K. K. Jones, T. J. Cornwell, A. Gray and C. A. Simenstad (2005). Patterns of Chinook Salmon migration and residency in the Salmon River estuary (Oregon). Estuarine, Coastal and Shelf Science 64: 79-93.
- Bottom, D. L., C. A. Simenstad, J. Burke, A. M. Baptista, D. A. Jay, K. K. Jone, E. Casillas and M. H. Schiewe (2005). Salmon at river's end: the role of the estuary in the decline and recovery

of Columbia River salmon. U.S. Department of Commerce NOAA Tech. Memo NMFS-NWFSC-68. 246 pp.

Boumans, R. M., D. M. Burdick and M. Dionne (2002). Modeling Habitat Change in Salt Marshes After Tidal Restoration. *Restoration Ecology* 10: 543-555.

Bourne, C. M., D. G. Kehler, Y. F. Wiersma and D. Cote (2011). Barriers to fish passage and barriers to fish passage assessments: the impact of assessment methods and assumptions on barrier identification and quantification of watershed connectivity. *Aquatic Ecology* 45: 389-403.

Bowron, T., N. Neatt, D. van Proosdij, J. Lundholm and J. Graham (2011). Macro-Tidal Salt Marsh Ecosystem Response to Culvert Expansion. *Restoration Ecology* 19: 307-322.

Boys, C. A., F. J. Kroon, T. M. Glasby and K. Wilkinson (2012). Improved fish and crustacean passage in tidal creeks following floodgate remediation. *Journal of Applied Ecology* 49: 223-233.

Boys, C. A. and R. J. Williams (2012). Succession of fish and crustacean assemblages following reinstatement of tidal flow in a temperate coastal wetland. *Ecological Engineering* 49: 221-232.

Bretsch, K. and D. M. Allen (2006). Tidal migrations of nekton in salt marsh intertidal creeks. *Estuaries and coasts* 29: 474-486.

Brockmeyer, R. E., Jr., J. R. Rey, R. W. Virnstein, R. G. Gilmore and L. Earnest (1997). Rehabilitation of impounded estuarine wetlands by hydrologic reconnection to the Indian River Lagoon, Florida (USA). *Wetlands Ecology and Management* 4: 93-109.

Buchsbaum, R. N., J. Catena, E. Hutchins and M. James-Pirri (2006). Changes In Salt Marsh Vegetation, *Phragmites Australis*, And Nekton In Response To Increased Tidal Flushing In A New England Salt Marsh. *Wetlands* 26: 544-557.

Burdick, D., M. Dionne, R. Boumans and F. Short (1996). Ecological responses to tidal restorations of two northern New England salt marshes. *Wetlands Ecology and Management* 4: 129-144.

Burkett, V. (2012). *Coastal Impacts, Adaptation, and Vulnerabilities*. Springer, NY.

Cherry, D. S., K. L. Dickson and J. Cairns (1975). Temperatures Selected and Avoided by Fish at Various Acclimation Temperatures. *Journal of the Fisheries Research Board of Canada* 32: 485-491.

Clancy, M., I. Logan, J. Lowe, J. Johannessen, A. MacLennan, F. B. Van Cleve, J. Dillon, B. Lyons, R. Carman and P. Cereghino (2009). Management measures for protecting and restoring the Puget Sound Nearshore. Puget Sound nearshore ecosystem restoration project report, Olympia WA. 307 pp.

- Cloern, J. E. (2001). Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210: 223-253.
- Congleton, J., S. Davis and S. Foley (1981). Distribution, abundance and outmigration timing of Chum and chinook salmon fry in the Skagit salt marsh. *Proceedings of the Salmon and Trout Migratory Behavior Symposium*, University of Washington, Seattle, WA.
- Cotterell, E. (1998). Fish passage in streams: fisheries guidelines for design of stream crossings. *Fish Habitat Guidelines*. Brisbane, Australia, Queensland Department of Primary Industries. 01: 1-37.
- Coutant, C. C. (1977). Compilation of Temperature Preference Data. *Journal of the Fisheries Research Board of Canada* 34: 739-745.
- Dane, B. G. (1978). Culvert guidelines: recommendations for the design and installation of culverts in British Columbia to avoid conflict with anadromous fish. *Fisheries and Marine Service Technical Report No. 811*. Department of Fisheries and Environment, Vancouver, BC. 63 pp.
- Dane, B. G. (1978). A review and resolution of fish passage problems at culvert sites in British Columbia. *Fisheries and Marine Service Technical Report No. 810*. Department of Fisheries and Environment, Vancouver, BC. 126 pp.
- David, A., C. Simenstad, J. Cordell, J. Toft, C. Ellings, A. Gray and H. Berge (2015). Wetland Loss, Juvenile Salmon Foraging Performance, and Density Dependence in Pacific Northwest Estuaries. *Estuaries and Coasts* 39: 767-780.
- David, B. O. and M. P. Hamer (2012). Remediation of a perched stream culvert with ropes improves fish passage. *Marine & Freshwater Research* 63: 440-449.
- Dibble, K. L. and L. A. Meyerson (2012). Tidal flushing restores the physiological condition of fish residing in degraded salt marshes. *PLoS One* 7: e46161.
- Dick, T. M. and O. O. Osunkoya (2000). Influence of tidal restriction floodgates on decomposition of mangrove litter. *Aquatic Botany* 68: 273-280.
- Diebel, M. W., M. Fedora, S. Cogswell and J. R. O'Hanley (2015). Effects of Road Crossings on Habitat Connectivity for Stream-Resident Fish. *River Research and Applications* 31: 1251-1261.
- Diefenderfer, H. L., A. M. Coleman, A. B. Borde and I. A. Sinks (2008). Hydraulic geometry and microtopography of tidal freshwater forested wetlands and implications for restoration, Columbia River, USA. *Ecohydrology & Hydrobiology* 8: 339-361.
- Doehring, K., R. G. Young, J. Hay and A. J. Quarterman (2011). Suitability of Dual-frequency Identification Sonar (DIDSON) to monitor juvenile fish movement at floodgates. *New Zealand Journal of Marine and Freshwater Research* 45: 413-422.

- Doehring, K., R. G. Young and A. R. McIntosh (2011). Factors affecting juvenile galaxiid fish passage at culverts. *Marine & Freshwater Research* 62: 38-45.
- Douglass, S. L., B. M. Webb and R. Kilgore (2014). Highways in the Coastal Environment: Assessing Extreme Events. *Hydraulic Engineering Circular No. 25*. 147 pp.
- Eberhardt, A. L., D. M. Burdick and M. Dionne (2011). The Effects of Road Culverts on Nekton in New England Salt Marshes: Implications for Tidal Restoration. *Restoration Ecology* 19: 776-785.
- Eberhardt, A. L., D. M. Burdick, M. Dionne and R. E. Vincent (2015). Rethinking the Freshwater Eel: Salt Marsh Trophic Support of the American Eel, *Anguilla rostrata*. *Estuaries and Coasts* 38: 1251-1261.
- Eby, L. A. and L. B. Crowder (2002). Hypoxia-based habitat compression in the Neuse River Estuary: context-dependent shifts in behavioral avoidance thresholds. *Canadian Journal of Fisheries and Aquatic Sciences* 59: 952-965.
- Ennis, S. (2009). Effects of Tide Gate Replacement on Water Temperature in a Freshwater Slough in the Columbia River Estuary Masters thesis, Portland State University, 64 pp.
- Forest Service Stream-Simulation Working Group. (2008). Stream Simulation: An Ecological Approach to Providing Passage for Aquatic Organisms at Road-Stream Crossings. US Forest Service Report # 0877 1801-SDTDC. San Dimas CA.
- Gee, A. K., K. Wasson, S. L. Shaw and J. Haskins (2010). Signatures of Restoration and Management Changes in the Water Quality of a Central California Estuary. *Estuaries and Coasts* 33: 1004-1024.
- Giannico, G. and J. A. Souder (2005). Tide Gates in the Pacific Northwest: Operation, Types and Environmental Effects. Oregon State University, Corvallis OR. 28 pp.
- Gibson, R. N. (2003). Go with the flow: tidal migration in marine animals. *Hydrobiologia* 503 3: 153-161.
- Gilliam, J. F. and D. F. Fraser (2001). Movement in corridors: Enhancement by predation threat, disturbance, and habitat structure. *Ecology* 82: 258-273.
- Gilmore, R. G., D. W. Cooke and C. J. Donohoe (1982). A comparison of the fish populations and habitat in open and closed salt marsh impoundments in east-central Florida. *Northeast Gulf Science* 5: 25-37.
- Goolish, E. M. (1991). Aerobic and Anaerobic Scaling in Fish. *Biological Reviews* 66: 33-56.
- Gordon, J., M. Arbeider, D. Scott, S. M. Wilson and J. W. Moore (2015). When the Tides Don't Turn: Floodgates and Hypoxic Zones in the Lower Fraser River, British Columbia, Canada. *Estuaries and Coasts* 38: 2337-2344.

- Greene, C. M., J. Hall, E. Beamer, R. Henderson and B. Brown (2013). Biological and Physical Effects of 'Fish-Friendly' Tide Gates Final Report, Washington State Recreation and Conservation Office. 43 pp. URL: http://skagitcoop.org/wp-content/uploads/EB2673_Greene-et-al_2012.pdf
- Hamman, J. J., A. F. Hamlet, S.-Y. Lee, R. Fuller and E. E. Grossman (2016). Combined Effects of Projected Sea Level Rise, Storm Surge, and Peak River Flows on Water Levels in the Skagit Floodplain. *Northwest Science* 90: 57-78.
- Haro, A., T. Castro-Santos, J. Noreika and M. Odeh (2004). Swimming performance of upstream migrant fishes in open-channel flow: a new approach to predicting passage through velocity barriers. *Canadian Journal of Fisheries and Aquatic Sciences* 61: 1590-1601.
- Harvey, B. C. and A. J. Stewart (1991). Fish size and habitat depth relationships in headwater streams. *Oecologia* 87: 336-342.
- Healey, M. C. (1980). Utilization of the Nanaimo River estuary by juvenile Chinook Salmon, *Oncorhynchus tshawytscha*. *Fishery Bulletin* 77: 653-668.
- Hering, D. K. (2009). Growth, residence, and movement of juvenile Chinook Salmon within restored and reference estuarine marsh channels in Salmon River. Masters thesis, Oregon State University, 164 pp.
- Hering, D. K., D. L. Bottom, E. F. Prentice, K. K. Jones and I. A. Fleming (2010). Tidal movements and residency of subyearling Chinook Salmon (*Oncorhynchus tshawytscha*) in an Oregon salt marsh channel. *Canadian Journal of Fisheries and Aquatic Sciences* 67: 524-533.
- Hoffman, R. L. and J. Dunham (2007). Fish Movement Ecology in High Gradient Headwater Streams: Its Relevance to Fish Passage Restoration through Stream Culvert Barriers. OFR 2007-1140, U.S. Geological Survey, 37 pp.
- Hood, W. G. (2004). Indirect environmental effects of dikes on estuarine tidal channels: Thinking outside of the dike for habitat restoration and monitoring. *Estuaries* 27: 273-282.
- Hood, W. G. (2015). Predicting the number, orientation and spacing of dike breaches for tidal marsh restoration. *Ecological Engineering* 83: 319-327.
- Hood, W. G., E. E. Grossman and C. Veldhuisen (2016). Assessing Tidal Marsh Vulnerability to Sea-Level Rise in the Skagit Delta. *Northwest Science* 90: 79-93.
- Hoover, R. A. and J. E. Purcell (2009). Substrate preferences of scyphozoan *Aurelia labiata* polyps among common dock-building materials. *Hydrobiologia* 616: 259-267.
- Hughes, B. B., M. D. Levey, J. A. Brown, M. C. Fountain, A. B. Carlisle, S. Y. Litvin, C. M. Greene, W. N. Heady and M. G. Gleason (2014). Nursery Functions of U.S. West Coast Estuaries: The State of Knowledge for Juveniles of Focal Invertebrate and Fish Species. The Nature Conservancy, Arlington VA. 168 pp.

Iwata, M. and S. Komatsu (1984). Importance of Estuarine Residence for Adaptation of Chum Salmon (*Oncorhynchus-Keta*) Fry to Seawater. *Canadian Journal of Fisheries and Aquatic Sciences* 41: 744-749.

Johnson, J. and T. A. Whitesel (2012). Julia Butler Hansen National Wildlife Refuge: Post-Construction Assessment of Fishes, Habitats, and Tide Gates in Sloughs on the Mainland, 2011 Annual Report. US Fish and Wildlife Service. Columbia River Fisheries Program Office. Vancouver, WA. 30 pp.

Jones, K., T. Cornwell, D. L. Bottom, L. A. Campbell and S. Stein (2014). The contribution of estuary-resident life histories to the return of adult *Oncorhynchus kisutch*. *Journal of Fish Biology* 85: 52-80.

Jonkman, S. N., M. M. Hillen, R. J. Nicholls, W. Kanning and M. van Ledden (2013). Costs of Adapting Coastal Defences to Sea-Level Rise-New Estimates and Their Implications. *Journal of Coastal Research* 29: 1212-1226.

Kahler, T. H. and T. P. Quinn (1998). Juvenile and resident salmonid movement and passage through culverts. Washington State Transportation Center, University of Washington, Seattle WA. 38 pp.

Kemp, P. S., M. H. Gessel and J. G. Williams (2005). Fine-scale behavioral responses of Pacific salmonid smolts as they encounter divergence and acceleration of flow. *Transactions of the American Fisheries Society* 134: 390-398.

Kemp, P. S., M. H. Gessel and J. G. Williams (2005). Seaward migrating subyearling chinook salmon avoid overhead cover. *Journal of Fish Biology* 67: 1381-1391.

Kemp, P. S. and J. R. O'Hanley (2010). Procedures for evaluating and prioritising the removal of fish passage barriers: a synthesis. *Fisheries Management and Ecology* 17: 297-322.

Kemp, P. S. and J. G. Williams (2009). Illumination influences the ability of migrating juvenile salmonids to pass a submerged experimental weir. *Ecology of Freshwater Fish* 18: 297-304.

Kennedy, V. S. (1990). Anticipated Effects of Climate Change on Estuarine and Coastal Fisheries. *Fisheries* 15: 16-24.

Khangaonkar, T. and T. P. Wang (2013). Potential alteration of fjordal circulation due to a large floating structure-Numerical investigation with application to Hood Canal basin in Puget Sound. *Applied Ocean Research* 39: 146-157.

Kirk, M. A. and C. C. Caudill (2016). Network analyses reveal intra-and interspecific differences in behaviour when passing a complex migration obstacle. *Journal of Applied Ecology* 2016: 1-10. 10.1111/1365-2664.12786

Konisky, R. A., D. M. Burdick, M. Dionne and H. A. Neckles (2006). A regional assessment of salt marsh restoration and monitoring in the Gulf of Maine. *Restoration Ecology* 14: 516-525.

- Koski, K. (2009). The fate of Coho Salmon nomads: the story of an estuarine-rearing strategy promoting resilience. *Ecology and Society* 14: 4. URL: <http://www.ecologyandsociety.org/vol14/iss1/art4/>
- Kroon, F. J. and D. H. Ansell (2006). A comparison of species assemblages between drainage systems with and without floodgates: implications for coastal floodplain management. *Canadian Journal of Fisheries and Aquatic Sciences* 63: 2400-2417.
- Levings, C. (2016). *Ecology of Salmonids in Estuaries around the World: Adaptations, Habitats, and Conservation*. UBC Press, Vancouver BC.
- Levy, D. A. and T. G. Northcote (1982). Juvenile Salmon Residency in a Marsh Area of the Fraser River Estuary. *Canadian Journal of Fisheries and Aquatic Sciences* 39: 270-276.
- Lotze, H. K., H. S. Lenihan, B. J. Bourque, R. H. Bradbury, R. G. Cooke, M. C. Kay, S. M. Kidwell, M. X. Kirby, C. H. Peterson and J. B. Jackson (2006). Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* 312: 1806-1809.
- McBride, A., S. Todd, O. Odum, M. Koschak and E. Beamer (2009). *Developing a Geomorphic Model for Nearshore Habitat Mapping and Analysis*. Skagit River System Cooperative, LaConner WA. 62 pp.
- McNatt, R. A., D. A. Bottom and S. A. Hinton (2016). Residency and Movement of Juvenile Chinook Salmon at Multiple Spatial Scales in a Tidal Marsh of the Columbia River Estuary. *Transactions of the American Fisheries Society* 145: 774-785.
- Mejia, F., M. K. Saiki and J. Y. Takekawa (2008). Relation Between Species Assemblages Of Fishes and Water Quality In Salt Ponds and Sloughs In South San Francisco Bay. *Southwestern Naturalist* 53: 335-345.
- Meynecke, J.-O., G. C. Poole, J. Werry and S. Y. Lee (2008). Use of PIT tag and underwater video recording in assessing estuarine fish movement in a high intertidal mangrove and salt marsh creek. *Estuarine, Coastal and Shelf Science* 79: 168-178.
- Minello, T. J., K. W. Able, M. P. Weinstein and C. G. Hays (2003). Salt marshes as nurseries for nekton: testing hypotheses on density, growth and survival through meta-analysis. *Marine Ecology Progress Series* 246: 39-59.
- Moore, M., B. A. Berejikian and E. P. Tezak (2013). A floating bridge disrupts seaward migration and increases mortality of Steelhead smolts in Hood Canal, Washington state. *PLoS One* 8: e73427.
- Mora, J. W. and D. M. Burdick (2013). Effects of man-made berms upon plant communities in New England salt marshes. *Wetlands Ecology and Management* 21: 131-145.

Moreno-Valcárcel, R., F. Oliva-Paterna, S. Bevilacqua, A. Terlizzi and C. Fernández-Delgado (2016). Long-term effects of tidal restriction on fish assemblages in east Atlantic coastal marshlands. *Marine Ecology Progress Series* 543: 209-222.

Morgan-King, T. L. and D. H. Schoellhamer (2013). Suspended-Sediment Flux and Retention in a Backwater Tidal Slough Complex near the Landward Boundary of an Estuary. *Estuaries and Coasts* 36: 300-318.

Moser, M. L., A. F. Olson and T. P. Quinn (1991). Riverine and Estuarine Migratory Behavior of Coho Salmon (*Oncorhynchus kisutch*) Smolts. *Canadian Journal of Fisheries and Aquatic Sciences* 48: 1670-1678.

Mueller, R. P., S. S. Southard, C. W. May, W. H. Pearson and V. I. Cullinan (2008). Juvenile Coho Salmon leaping ability and behavior in an experimental culvert test bed. *Transactions of the American Fisheries Society* 137: 941-950.

Myrick, R. M. and L. B. Leopold (1963). Hydraulic geometry of a small tidal estuary. US Government Printing Office, Washington DC.

Olson, A. F. and T. P. Quinn (1993). Vertical and Horizontal Movements of Adult Chinook Salmon *Oncorhynchus-Tshawytscha* in the Columbia River Estuary. *Fishery Bulletin* 91: 171-178.

Orr, M., S. Crooks and P. B. Williams (2003). Will restored tidal marshes be sustainable? *San Francisco Estuary and Watershed Science* 1: 1-33.

Pess, G., S. Morley and P. Roni (2005). Chapter 10: Evaluating Fish Response to Culvert Replacement and Other Methods for Reconnecting Isolated Aquatic Habitats. *Monitoring stream and watershed restoration*. P. Roni. Bethesda, MD, American Fisheries Society: 267-276.

Pauly, D. (1995). Anecdotes and the shifting baseline syndrome of fisheries. *Trends in ecology & evolution*, 10: 430.

Pollard, D. A. and J. C. Hannan (1994). The ecological effects of structural flood mitigation works on fish habitats and fish communities in the lower Clarence River system of south-eastern Australia. *Estuaries* 17: 427-461.

Powers, P. D., K. M. Bates, T. Burns, R. Gowen and R. Whitney (1997). Culvert Hydraulics Related to Upstream Juvenile Salmonid Passage. Washington State Department of Transportation Project No. 982740. Washington Department of Fish and Wildlife, Olympia WA, 21 pp.

Price, D. M., T. Quinn and R. J. Barnard (2010). Fish passage effectiveness of recently constructed road crossing culverts in the Puget Sound region of Washington State. *North American Journal of Fisheries Management* 30: 1110-1125.

- Quinn, T. P. (2005). *The Behavior and Ecology of Pacific Salmon and Trout*. Bethesda MD, American Fisheries Society.
- Raposa, K. (2002). Early responses of fishes and crustaceans to restoration of a tidally restricted New England salt marsh. *Restoration Ecology* 10: 665-676.
- Raposa, K. B. (2008). Early Ecological Responses to Hydrologic Restoration of a Tidal Pond and Salt Marsh Complex in Narragansett Bay, Rhode Island. *Journal of Coastal Research*, Special Issue 55: 180 – 192. doi: <http://dx.doi.org/10.2112/SI55-015>
- Raposa, K. B. and D. M. Talley (2012). A meta-analysis of nekton responses to restoration of tide-restricted New England salt marshes. Pp 97-118 in C.T. Roman and D.M. Burdick, eds. *Tidal Marsh Restoration: A Synthesis of Science and Management*. Island Press, Washington DC.
- Reagan, R. (2015). *Where the Stream Meets the Road: Prioritizing Culvert Replacement for Fish Passage*. Masters thesis, Oregon State University, 55 pp.
- Richmond, M. C., Z. Deng, G. R. Guensch, H. Tritico and W. H. Pearson (2007). Mean flow and turbulence characteristics of a full-scale spiral corrugated culvert with implications for fish passage. *Ecological Engineering* 30: 333-340.
- Ritter, A. F., K. Wasson, S. I. Lonhart, R. K. Preisler, A. Woolfolk, K. A. Griffith, S. Connors and K. W. Heiman (2008). Ecological signatures of anthropogenically altered tidal exchange in estuarine ecosystems. *Estuaries and Coasts* 31: 554-571.
- Roman, C. T., K. B. Raposa, S. C. Adamowicz, M. J. James-Pirri and J. G. Catena (2002). Quantifying vegetation and nekton response to tidal restoration of a New England salt marsh. *Restoration Ecology* 10: 450-460.
- Roni, P., M. C. Liermann, C. Jordan and E. A. Steel (2005). Steps for designing a monitoring and evaluation program for aquatic restoration. Pp. 13-34 in P. Roni, ed. *Monitoring stream and watershed restoration*. American Fisheries Society, Bethesda MD.
- Rountree, R. and K. Able (2007). Spatial and temporal habitat use patterns for salt marsh nekton: implications for ecological functions. *Aquatic Ecology* 41: 25-45.
- Sagar, J., A. Borde, L. Johnson, C. Corbett, J. Morace, K. Macneale, W. Temple, J. Mason, R. Kaufmann and V. Cullinan (2013). *Juvenile salmon ecology in tidal freshwater wetlands of the lower Columbia River and Estuary: Synthesis of the ecosystem monitoring program, 2005-2010*. Prepared by the Lower Columbia Estuary Partnership for the Bonneville Power Administration, Portland OR. Available from the Lower Columbia Estuary Partnership, 196 pp.
- Sandell, T., J. Fletcher, T. Buehrens, A. McAninch and M. Wait (2011). *Grays Harbor Juvenile Fish Use Assessment: 2011 Annual Report*. Prepared for the Chehalis Basin Habitat Working Group, Wild Fish Conservancy, 117 pp.

Schlenger, P., A. MacLennan, E. Iverson, K. Fresh, C. Tanner, B. Lyons, S. Todd, R. Carman, D. Myers, S. Campbell and A. Wick (2011). Strategic Needs Assessment: Analysis of Nearshore Ecosystem Process Degradation in Puget Sound. Puget Sound Estuary and Nearshore Restoration Partnership. Olympia, WA.

Scott, D. C., M. Arbeider, J. Gordon and J. W. Moore (2016). Flood control structures in tidal creeks associated with reduction in nursery potential for native fishes and creation of hotspots for invasive species. *Canadian Journal of Fisheries and Aquatic Sciences* 73: 1-11.

Shaul, L. D., R. Ericksen, K. Crabtree and J. Lum (2013). Beyond the Estuary: An Extension of the Nomad Life History Strategy in Coho Salmon. Technical Report No. 9, North Pacific Anadromous Fish Commission: 5.

Shipman, H. (2008). A Geomorphic Classification of Puget Sound Nearshore Landforms. Puget Sound Nearshore and Estuary Restoration Partnership, Seattle WA.

Simenstad, C., W. G. Hood, R. Thom, D. Levy and D. Bottom (2000). Landscape Structure and Scale Constraints on Restoring Estuarine Wetlands for Pacific Coast Juvenile Fishes. Concepts and Controversies in Tidal Marsh Ecology, Vineland, NJ, Springer Netherlands.

Simenstad, C., M. Ramirez, J. Burke, M. Logsdon, H. Shipman, C. Tanner, J. Toft, B. Craig, C. Davis, J. Fung, P. Bloch, K. C. Fresh, S. D. Myers, E. Iverson, B. A. P. Schlenger, C. Kiblinger, P. Myre, W. Gertsel and A. MacLennan (2011). Historical Change of Puget Sound Shorelines: Puget Sound Nearshore Ecosystem Project Change Analysis. Puget Sound Estuary and Nearshore Restoration Partnership. Olympia, WA.

Simenstad, C. A. and J. R. Cordell (2000). Ecological assessment criteria for restoring anadromous salmonid habitat in Pacific Northwest estuaries. *Ecological Engineering* 15: 283-302.

Simenstad, C. A., K. Fresh, J. Flemma and D. Clarke (1991). Effects of estuarine habitat modifications on anadromous salmonids: A literature survey. Report # FRI-UW-9123. Fisheries Research Institute, University of Washington, Seattle WA, 47 pp.

Simenstad, C. A., K. L. Fresh and E. O. Salo (1982). The role of Puget Sound and Washington coastal estuaries in the life history of Pacific salmon: an unappreciated function. Pp 343-364 in V.S. Kennedy, ed. *Estuarine Comparisons*. Academic Press, NY.

Sinicrope, T. L., P. G. Hine, R. S. Warren and W. A. Niering (1990). Restoration of an Impounded Salt-Marsh in New-England. *Estuaries* 13: 25-30.

Smith, L. S. and L. T. Carpenter (1987). Salmonid fry swimming stamina data for diversion screen criteria. Seattle, WA, Fisheries Research Institute, University of Washington, Seattle WA.

- Smith, S. M., C. T. Roman, M.-J. James-Pirri, K. Chapman, J. Portnoy and E. Gwilliam (2009). Responses of Plant Communities to Incremental Hydrologic Restoration of a Tide-Restricted Salt Marsh in Southern New England (Massachusetts, U.S.A.). *Restoration Ecology* 17: 606-618.
- Snover, A., G. Mauger, L. W. Binder, M. Krosby and I. Tohver (2013). Climate change impacts and adaptation in Washington State: Technical summaries for decision makers. State of Knowledge Report prepared for the Washington State Department of Ecology. Climate Impacts Group, University of Washington, Seattle WA.
- Sommer, T. R., W. C. Harrell and M. L. Nobriga (2005). Habitat use and stranding risk of juvenile Chinook Salmon on a seasonal floodplain. *North American Journal of Fisheries Management* 25: 1493-1504.
- Starrs, D., B. C. Ebner, M. Lintermans and C. J. Fulton (2011). Using sprint swimming performance to predict upstream passage of the endangered Macquarie perch in a highly regulated river. *Fisheries Management and Ecology* 18: 360-374.
- Taylor, D. S., G. R. Poulakis, S. R. Kupschus and C. H. Faunce (1998). Estuarine reconnection of an impounded mangrove salt marsh in the Indian River Lagoon, Florida: short-term changes in fish fauna. *Mangroves and Salt Marshes* 2: 29-36.
- Teodosio, M. A., C. B. Paris, E. Wolanski and P. Morais (2016). Biophysical processes leading to the ingress of temperate fish larvae into estuarine nursery areas: A review. *Estuarine Coastal and Shelf Science* 183: 187-202.
- Thelen, B. A. and R. K. Thiet (2009). Molluscan Community Recovery Following Partial Tidal Restoration of a New England Estuary, USA. *Restoration Ecology* 17: 695-703.
- Torio, D. D. and G. L. Chmura (2013). Impacts of Sea Level Rise on Marsh as Fish Habitat. *Estuaries and Coasts* 38: 1288-1303.
- Turner, R. E. (2004). Coastal wetland subsidence arising from local hydrologic manipulations. *Estuaries* 27: 265-272.
- Valentine-Rose, L. and C. A. Layman (2011). Response of Fish Assemblage Structure and Function Following Restoration of Two Small Bahamian Tidal Creeks. *Restoration Ecology* 19: 205-215.
- Van Proosdij, D., J. Lundholm, N. Neatt, T. Bowron and J. Graham (2010). Ecological re-engineering of a freshwater impoundment for salt marsh restoration in a hypertidal system. *Ecological Engineering* 36: 1314-1332.
- Vandenbruwaene, W., T. Maris, T. J. S. Cox, D. R. Cahoon, P. Meire and S. Temmerman (2011). Sedimentation and response to sea-level rise of a restored marsh with reduced tidal exchange: Comparison with a natural tidal marsh. *Geomorphology* 130: 115-126.

Wang, Y. P., R. S. Zhang and S. Gao (1999). Velocity variations in salt marsh creeks, Jiangsu, China. *Journal of Coastal Research* 15: 471-477.

Warren, R. S., P. E. Fell, R. Rozsa, A. H. Brawley, A. C. Orsted, E. T. Olson, V. Swamy and W. A. Niering (2002). Salt marsh restoration in Connecticut: 20 years of science and management. *Restoration Ecology* 10: 497-513.

Watson, E. B. (2004). Changing elevation, accretion, and tidal marsh plant assemblages in a South San Francisco Bay tidal marsh. *Estuaries* 27: 684-698.

Washington Department of Fish and Wildlife (WDFW) (2009). Fish Passage Barrier and Surface Water Diversion Screening Assessment and Prioritization Manual. Washington Department of Fish and Wildlife, Olympia WA, 240 pp.

Wilhere, G., J. Atha, T. Quinn, L. Helbrecht and I. Tohver (2016). Incorporating Climate Change into the Design of Water Crossing Structures, Washington Department of Fish and Wildlife, Olympia WA, 51 pp.

Williams, J. R., T. M. Dellapenna and G.-h. Lee (2013). Shifts in depositional environments as a natural response to anthropogenic alterations: Nakdong Estuary, South Korea. *Marine Geology* 343: 47-61.

Williams, P. B. and M. K. Orr (2002). Physical evolution of restored breached levee salt marshes in the San Francisco Bay estuary. *Restoration Ecology* 10: 527-542.

Wong, C. R. (1989). Observations of tides and tidal currents in Elkhorn Slough. Masters thesis, San Jose State University, 108 pp.

Appendix 1. History of Washington State Fish Passage Technical Guidance

Washington state law has required fish passage for dams and obstructions since 1890 and for all water crossing structures since 1949, although early application was restricted to highway culverts and larger stream and river systems (Barnard et al. 2013). As the understanding of fish passage needs evolved, the criteria and design for water crossing structures to meet fish passage requirements also evolved. Historically, fish passage rules and guidance were based on the hydraulic design approach to provide fish passage. This methodology involved applying design flow hydrology to the geometry of the road crossing structure and evaluating the resulting outfall drop, flow depth and velocity based on the swimming capabilities of the “target fish.” The “target fish” defined by Washington state law are adult salmonids and a 6-inch trout, and were primarily focused on providing upstream movement to spawning grounds based on swimming ability criteria for adult salmonids. These early hydraulic design approaches emphasizing adult salmon swimming ability criteria ultimately resulted in fish passage and accessibility problems for weaker swimming aquatic organisms and fish, including juvenile salmonids.

In 2001, the U.S. District Court ruled that the State of Washington has a duty to preserve fisheries secured to the tribes in the Stevens Treaties. The court declared that the right of taking fish imposes a duty upon the state to refrain from building or operating culverts under state-maintained roads that hinder fish passage and thereby reduce the number of fish that would otherwise be available for tribal harvest. The court further declared that the State of Washington currently owns and operates culverts that violate this duty. A federal court injunction, issued March 2013, requires the state to significantly increase the effort to remove or upgrade state-owned culverts that block habitat for salmon and Steelhead by 2030.

Recognizing the need for juvenile and sub-adult fish to migrate both downstream and upstream, water crossing structure design guidelines were further modified to reflect an improved understanding of migratory and resident fish ecology (Bates et al. 2003, WDFW 2009, Barnard et al. 2013). In addition, Washington State rules were updated in 2015 to require that water crossing structures on fish-bearing waters provide unimpeded passage of adult and juvenile fishes ([WAC 220-660-190](#)). Passage is assumed for bridges and culverts if there are no barriers due to behavioral impediments such as excessive slope, vertical drop, high velocity, lack of surface flow, uncharacteristically coarse bed material, and other related conditions (WAC 220-660-190).

Since the late 1990’s, design and assessment criteria to minimize the impacts of water crossing structures have been improved by implementation of geomorphic-based designs such as the “no-slope method” and stream simulation (Bates et al. 2003, Forest Service Stream-Simulation Working Group 2008, Barnard et al. 2013). The U.S. Forest Service implemented these improvements based on the consideration of movements of both adult and juvenile life stages of

migratory and resident aquatic species. Additional considerations for geomorphic-based designs were due to failure of some structures designed using hydraulic methods to comply with fish passage criteria over time using existing design methods. The no-slope and stream simulation methods focus on maintaining natural elevation gradients and structure of the natural streambed rather than designing to fish swimming abilities (Bates et al. 2003, Forest Service Stream-Simulation Working Group 2008, Barnard et al. 2013). By maintaining natural stream characteristics and fluvial processes at the crossing (channel width, slope, velocity, streambed composition, sediment transport, and channel complexity), these approaches improve the passage for all aquatic organisms by providing no net loss of habitat characteristics.

Table A1. Examples of components utilized to determine the effectiveness of riverine stream simulation designs.

Component	Criterion for stream simulation
Bankfull/bed width	Bankfull width is similar to upstream and downstream reaches.
Culvert/channel gradient	The slope of the bed inside the culvert is similar to the slope of the upstream channel.
Countersink/scour	Culvert invert or footings do not become exposed for life of structure.
Cross section/channel shape	Adjacent channel shape is continuous through the crossing.
Floodplain continuity	Constructed channel mimics adjacent channel habitat and flow conditions.
Sized to pass	Crossing provides unimpeded passage of fish, 100-year flood (Q100) events, large woody debris, and sediment.
Substrate	Channel substrate mimics reference reach gradation curve.
Structure span	Crossing width (span) allows for geomorphic processes to occur, including Q100 year flood.
Crossing length	Confined channel length is minimal, long crossings should have wider channels.
Bank hardening	Bank armoring should be minimized and use bio-engineering techniques if necessary.
Channel stability	Constructed channel morphology mimics adjacent channel habitat conditions.
Hydrology	Water crossing should be consistent with watershed conditions and land use.
Meander Bends	Channel planform is consistent with prevailing adjacent planform.

The improvements to water crossing structure design and evaluation criteria were developed primarily for riverine crossings. Water crossing structures in tidally influenced areas such as bridges, causeways, culverts, and tidegates (see *Section 2*) present unique challenges for design and assessment criteria. Target fish species, including all species of Pacific salmon, depend on tidally influenced channels for migration corridors between marine and riverine environments (e.g., juvenile migration to the ocean and adult migration to riverine spawning grounds). In addition, many species including some salmon depend on access to tidally influenced rearing

habitat during juvenile life stages (Levy and Northcote 1982, Bottom et al. 2005, Bottom et al. 2005b, Jones et al. 2014). The WDFW's most recent Water Crossing Design Guidelines provided some initial technical guidance for tidally influenced crossings in an appendix (Barnard et al. 2013, Appendix D), and recognized the importance of further developing these guidelines. Similar metrics to those presented in Table A1 could be developed specifically for intertidal crossings, but these principles need further biological support.

Providing access to upstream habitat blocked to fish passage has long been recognized as an effective restoration action for salmon recovery (Dane 1978, Powers et al. 1997, Bates et al. 2003, Pess et al. 2005). Assessment of existing structures is a focus for many watershed groups, Tribes, and government agencies in order to identify and prioritize restoration actions. The WDFW has had an active inventory and assessment program since the late 1990's, identifying over 47,000 sites statewide in the centralized WDFW Fish Passage and Screening Diversion Inventory database. WDFW has identified a minimum of 872 intertidal water crossing structures through existing inventory and GIS (Fig. A1). However, lack of fully developed fish passage assessment protocols in intertidal areas results in the majority of tidally influenced crossings designated as "unknown" for passability. Some tidal water crossing structures are currently being passed over for improvements due to lack of clear assessment and design criteria, missing the opportunity to address arguably the most important portions of the watershed and the most productive type of ecosystem. Some intertidal water crossings are being replaced out of necessity (e.g., infrastructure failure) but are designed with incomplete knowledge of design considerations that will benefit fish. These water crossing structures have an expected life of 30-100 years, so it is critical that their design is appropriate to support long-term fish passage and habitat forming processes, and be resilient to impacts of climate change.

Fish passage regulations for all culverts and bridges are described in [WAC 220-660-190](#), and regulations for tidegates and floodgates are specified in [RCW 77.55.281](#) and [WAC 220-660-430](#). By statute, WDFW cannot require "fishways" (currently undefined in the regulations), for tidegates associated with agricultural drainage systems that were in existence prior to May 20, 2003, unless fishways were already incorporated into structure design. For existing fishways, provisions requiring self-regulating tidegates to achieve fish passage were not enforced as required by statute RCW 77.55.281. For tidegates constructed or repaired/replaced for non-agricultural drainage systems or on agricultural drainage systems after 2003, WDFW can require

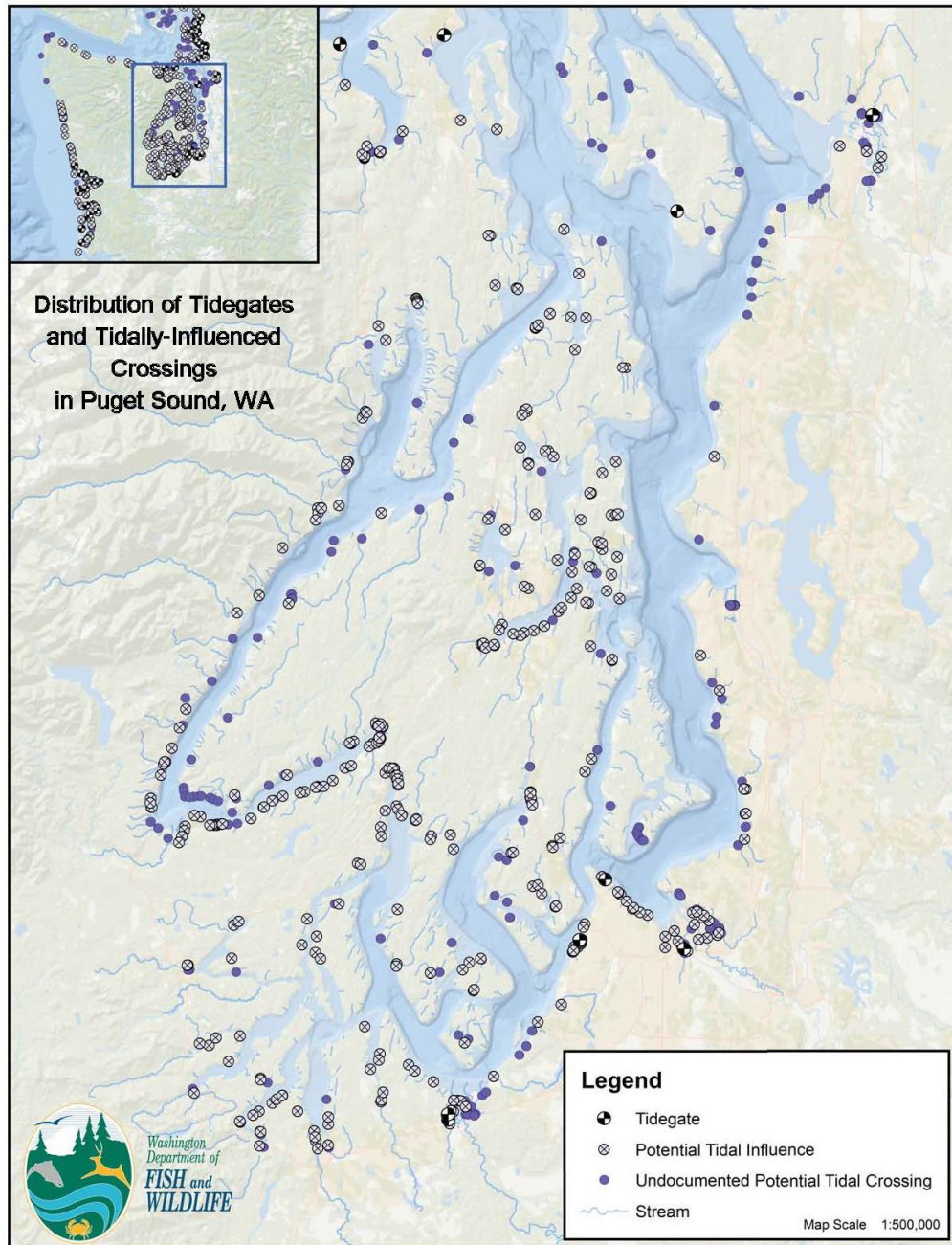


Figure A1. WDFW’s inventory of tidegates and intertidal water crossing structures along transportation corridors in the Puget Sound region. Crossings occurred within 1000 feet of a DNR shorezone feature. If the site had a tide gate or had tidal influence in the WDFW fish passage database comments, the site was assumed to be tidally influenced. Those sites greater than 26 feet were removed on the assumption they were above head of tide. Note that this inventory is incomplete because other tidegates and water crossing structures exist but are not associated with transportation corridors.

fish passage as part of the structure (WAC 220-660-430). In addition, local and federal regulations for tidegates can apply, but in practice, tidegate regulation has been transferred locally via use of formal agreements between stakeholders and regulatory agencies to allow replacement of structures and to promote fish benefits.

In Washington State, fish passage has been defined in practice as providing passage during the 90% of time that fish are present. In freshwater, this has largely been applied either by using a geomorphically based design approach or by analysis of riverine hydrology during the time periods (seasons) where juvenile fish targeted for protection are present. In tidal applications, the analysis would not only include a seasonal basis, but also would be need to be evaluated in the context of daily tidal cycles. Given the tidal range in Washington State, the velocities during tidal exchange in natural channels often exceed criteria that have been used for fish passage. At some crossings without limited freshwater flow, the flood tide does not reach the outlet until well into the tidal cycle. Therefore the tidal application of this type of criteria would need special consideration to avoid requiring conditions for passage which exceed those found in natural conditions. In addition, knowledge of fish behavior related to tide cycles is critical to development of this assessment criteria in understanding when, how, where in the water column and in what tidal conditions fish move into and out of intertidal habitat.

Appendix 2. Profiles of key species of Washington potentially affected by intertidal water crossing structures

The purpose of this appendix is to provide general information for key species that occur in intertidal habitats of Puget Sound and Washington's coastal estuaries. The potential impacts of water crossing structures depend upon species' natural tolerances as well as the degree to which they rely on intertidal habitats for rearing habitat and migration corridors, the timing and duration of rearing and migrations, and the life stages during which rearing and migrations occur. Therefore, this appendix provides general information on species life history variation, population distribution, and migration and rearing patterns. Information from these species profiles are also summarized in Tables A2 and A3 for species that rely on intertidal habitats for prolonged rearing or for migration corridors, respectively.

Relevant information on behavioral patterns and swimming speeds are also provided in these species profiles. However, behavioral and swimming speed information are not known or readily available for all species and life stages, and information is particularly sparse for juvenile life stages of species that rely on intertidal habitats. Furthermore, variation in methodologies and definitions for determining swimming speed thresholds vary greatly and are often derived from artificial lab based trials. Therefore, the swimming speed information presented in this appendix are intended as a general reference to inform the development of criteria for assessing or designing water crossing structures that consider variations in swimming speeds among different species and life stages. The swimming speeds reported here are generally lumped into short duration burst speed as well as sustained and prolonged swimming speeds as described in (Beamish 1978), with sustained swimming speeds being those that fish can be maintained for long periods of time (>200 minutes) and prolonged swimming speeds are speeds that can be maintained for 20 seconds to 200 minutes. A compilation of swimming speed data used in these species profiles are provided in Table A4.

Chinook Salmon (Oncorhynchus tshawytscha)

Chinook Salmon are semelparous anadromous salmon that primarily spawn in large river systems along the entire extent of Washington's coastline, although populations may occur in smaller streams as well (Wydoski and Whitney 2003). Upriver migrations of adults can occur at almost any month of the year, but typically occur in one to three peaks in the spring, summer, or fall months (Healey 1991). In Washington State's rivers, these upriver migrations typically occur between March and November (Healey 1991, Quinn 2005), during which adults may encounter intertidal water crossings as they migrate from the marine environment to their riverine spawning habitats. However, Chinook primarily spawn in larger river systems and therefore upriver migrations are not likely directly impacted by water crossing structures in either distributary channels or smaller blind tidal channels. Adult Chinook can attain the largest body sizes among

the Pacific Salmonids, with adults ranging in size from 40 to 152 cm and an average body size of 91 cm (Wydoski and Whitney 2003). Prolonged swimming speeds for adult Chinook are 1.25-1.96 m/s, with sustained speeds 1.21-3.3 m/s and burst speeds of up to 6.71 m/s (Table A4).

Juvenile Chinook Salmon are capable of expressing a wide range of variations in both the timing of juvenile migrations and duration of residency in the estuaries of their natal rivers. Coastal Washington Chinook populations predominantly express subyearling migrant timings, with juveniles migrating downstream soon after emergence with smaller fry migrants (< 4.5 cm) that enter the estuary between February and May and larger parr migrants (4.5 – 10 cm) that migrate between May and August (Healey 1991, Wydoski and Whitney 2003, Quinn 2005). The predominance and timing of migrant fry and parr can vary between systems and years. While parr migrants generally exhibit extended rearing in river systems, fry rear extensively (weeks to months) in estuaries prior to migrating to marine habitats (Congleton 1981, Beamer et al. 2005). Subyearling migrants primarily rear in blind channels of the river deltas to their natal rivers (Congleton 1981, Beamer et al. 2005), but may also move into nearshore lagoons or pocket estuaries (Beamer et al. 2003), or small coastal creek confluences (Beamer et al. 2013). From these rearing habitats, all subyearling migrants migrate into marine waters, rearing in embayments for extended periods of time through the summer and into fall.

Yearling migrants are also present among Washington's Chinook populations, with larger juveniles (10.0 - 16.0 cm) typically migrating downstream from April through June after having spent over a year rearing in mainstem and tributary habitats (Quinn 2005). Although yearling migrants do not typically rear in estuaries for extended periods of time, they can occur in estuaries through November (Quinn 2005) and may remain from days to weeks in large river deltas and embayments to complete the physiological adaptations necessary to transition from fresh to saltwater (Beamer et al. 2005).

Subyearling juvenile Chinook fry may avoid water velocities that exceed 0.3 m/s (Quinn 2005), prefer depths that are at least 1 meter (Beamer et al. 2005), and water temperatures cooler than 15°C (Brett 1952, Brett et al. 1982). Water velocities of up to 0.1 m/s will generally not impede juvenile salmonid movements (Wightman and Tayler 1976), although there are considerable variations in body sizes of juveniles due to the large variations in life history. Juvenile swimming speeds will vary greatly due to variations in size at migration among Chinook. Prolonged swimming speeds range from 0.14-0.50 m/s (Table A4), with speeds for fry migrants being 0.1-0.2 m/s, 0.2-0.5 for parr migrants, and 0.4-0.5 m/s for yearling migrants (Wightman and Tayler 1976, Smith and Carpenter 1987).

Table A2. Typical rearing periods are listed by species, life stage (juvenile and adult), and shoreform habitat types (LRD = large river delta, CCC = coastal creek confluence, LAG = lagoon, and EMB = embayments). See species profile descriptions for references used to construct this table. Where juvenile or adult life stages are known to not use a shoreform for rearing (e.g., typically spend less than a week in that habitat type) or are likely a rare or minor occurrence, values are listed as --. UNK indicates that rearing information was not found for that species, life stage, and shoreform type but rearing may occur.

Species	Shoreform Habitats	Typical Rearing Period in intertidal habitats		Typical Size Range (cm)	
		Juvenile	Adult	Juvenile	Adult
Chinook	LRD & CCC	Feb-Aug	--	3-9	--
	LAG	Feb-Jun	--	3-9	--
	EMB	May-Sep	--	7-20	--
Coho	LRD & CCC	Feb-Dec	--	3-19	--
	LAG	Feb-Jun	--	3-9	--
	EMB	Apr-Jul	--	90-20	--
Pink	LRD & CCC	--	--	--	--
	LAG	Feb-Jul	--	2.8-12.5	--
	EMB	Feb-Aug	--	2.8-12.5	--
Chum	LRD & CCC	Feb-May	--	3.5-10.7	--
	LAG	Feb-Aug	--	3.5-24.9	--
	EMB	Feb-Aug	--	3.5-24.9	--
Cutthroat	LRD & CCC	Jan-Sep	Jan-Sep	20-30	20-48
Steelhead	LRD & CCC	UNK	--	--	--
	LAG & EMB	UNK	--	--	--
Bull Trout	LRD & CCC	Mar-Jul	Mar-Sep	9-29	25-89
	LAG	--	--	--	--
	EMB	Apr-Jul	Mar-Sep	9-29	25-89
Eulachon	LRD	Mar-Aug	NA	0.5-3.5	--
Pacific Herring	LRD, CCC, LAG, & EMB	Feb-Aug	Jan-Jun	3.5-11	11-46
Surf Smelt	LRD, CCC, LAG, & EMB	Year-round	Year-round	4-10	10-21
English Sole	LRD	May-Aug	--	5.5-12.5	--
Dungeness Crab	LRD, CCC, LAG, & EMB	Year-round	--	0.6-10	--
Three-spine Stickleback	LRD, CCC, LAG, & EMB	Year-round	Year-round	2-4	4-9.5
Shiner Perch	LRD, CCC, LAG, & EMB	Jul-Sep	Jun-Sep	2-8	8-20
Staghorn Sculpin	LRD, CCC, LAG, & EMB	Mar-Oct	Year-round	0.5-5	5-34
River Lamprey	LRD & CCC	Apr-Jul	Jul-Sep/Apr	4-19	19-30
Smallmouth & Largemouth Bass	LRD & CCC	May-Oct	--	3.5-20.5	--

Table A3. Species-specific migration information of species potentially affected by intertidal water crossing structures as barriers to movement. See species profile descriptions for references used to construct this table. Typical rearing periods are listed by species, life stage (juvenile and adult), and shoreform habitat types (LRD = large river delta, CCC = coastal creek confluence, LAG = lagoon, and EMB = embayments). Where juvenile or adult life stages are known to not use a shoreform for migration or are likely a rare or minor occurrence, values are listed as --. UNK indicates that migration information was not found for that species, life stage, and shoreform type but migration may occur.

Species	Habitat	Migration Period		Size Range (cm)	
		Juvenile	Adult	Juvenile	Adult
Chinook	LRD	Feb-Aug	Mar-Nov	3-16	40-152
Coho	LRD & CCC	Feb-Dec	Aug-Jan	3.3-19	40-90
Pink	LRD & CCC	Feb-May	Aug-Nov	2.8-3.5	36-76
Chum	LRD & CCC	Feb-Jun	Aug-Nov	3.5-6.6	43-108
Cutthroat	LRD & CCC	Jan-Sep	Jul-Jan	20-30	20-48
Steelhead	LRD & CCC	Sep-Jun	Nov-May	7.0-23	45-123
Bull Trout	LRD & CCC	Mar-Jul	May-Aug	9-29	25-89
Eulachon	LRD	Mar-Apr	Feb-Mar	0.5-3.5	13-30
Pacific Herring	LRD, CCC, LAG, & EMB	Feb-Aug	Jan-Jun	3.5-11	11-46
Surf Smelt	LRD, CCC, LAG, & EMB	Year-round	Year-round	4-10	10-21
English Sole	LRD	May-Aug	--	5.5-12.5	--
Dungeness Crab	LRD, CCC, LAG, & EMB	Apr-Jun & Aug-Oct	--	0.6-10	--
Three-spine Stickleback	LRD, CCC, LAG, & EMB	Year-round	May-Jul	1-3	3.8-9.5
Shiner Perch	LRD, CCC, LAG, & EMB	Jul-Sep	Jun-Sep	2-8	8-20
Staghorn Sculpin	LRD, CCC, LAG, & EMB	Mar-Oct	--	0.5-5	--
River Lamprey	LRD & CCC	Apr-Jul	Jul-Sep	4-19	19-30
Pacific Lamprey	LRD & CCC	Nov-Jun	Apr-Sep	10.2-15.7	19.4-74.5
Smallmouth & Largemouth Bass	LRD & CCC	May-Oct	--	3.5-20.5	--

Coho Salmon (Oncorhynchus kisutch)

Coho Salmon are semelparous anadromous salmon that spawn in a wide range of small to large river systems throughout Washington State (Wydoski and Whitney 2003). Among Washington Coho populations, mature adult Coho typically migrate upstream between August and January, peaking between August and November (Wydoski and Whitney 2003). Given that Coho may spawn in smaller river systems, upriver migrations of Coho are potentially impacted by water crossing structures in large river distributaries as well as tidal channels connected with distributary networks for smaller coastal creeks and tributaries in larger river deltas systems. Furthermore, Coho tend to migrate upstream after freshets, during which high flows could exacerbate the impacts of water crossings structures that increase water velocities when flows peak during a freshet event (Quinn 2005). Returning adults typically range in size from 40 - 90

cm (Sandercock 1991). Prolonged swimming speeds for Coho adults are 0.97-1.13 m/s, with sustained speeds of 0.3-3.2 m/s and burst speeds of up to 6.4 m/s (Table A4). Adult Coho can navigate through water depths as shallow as 5 cm during upriver migrations, and can leap over 2 meters (Sandercock 1991).

Juvenile Coho can express a range of variations in the timing of downstream migrations and the duration of residency in intertidal habitats. Among Washington State Coho populations, downstream migration of subyearlings can occur within weeks of emergence although these migrations typically occur after rearing in freshwater for up to two years or more (Wydoski and Whitney 2003, Bennett et al. 2011, Bennett et al. 2014). Downstream migrations of juvenile Coho can migrate downstream from February to December with peaks that typically occur in the spring, summer, and fall months with sizes ranging from 3.3 to 19.3 cm depending on the duration of freshwater rearing prior to migration (Sandercock 1991). Similar to Chinook Salmon, subyearling migrants are thought to rear longer in estuarine habitats (Jones et al. 2014), whereas yearlings typically move through their natal estuaries relatively quickly, spending days or up to a week rearing in estuaries before migrating out to marine habitats (Quinn 2005, Clements 2012). However, recent studies have demonstrated that some Coho life histories may utilize tidal wetland habitats extensively with rearing periods of up to eight months (Eaton 2010). In addition, juvenile Coho have also been observed moving to non-natal freshwater rearing habitats (Bennett et al. 2011, Beamer et al. 2013), which would indicate that juvenile Coho may rely on intertidal channels associated with large river deltas and small coastal creek confluences as they move to non-natal freshwater rearing habitats. Therefore, juvenile Coho may be found in systems that do not support spawning adult Coho and these upriver migrating juveniles would be more sensitive to upriver migration barriers than larger adult migrants. Like Chinook Salmon, Coho Salmon will use lagoons as fry, and while many smolts migrate through estuarine habitats relatively quickly, they do exhibit longer residence in embayments.

Water velocities of up to 0.1 m/s will generally not impede juvenile salmonid movements (Wightman and Tayler 1976), but Coho juveniles can vary greatly in size like juvenile Chinook Salmon. Prolonged swimming speeds increase by about 0.03 m/s for each 1 cm increase in body length (Wightman and Tayler 1976), with prolonged swimming speeds ranging from 0.07-0.58 m/s and sustained swimming speeds of 0.06-0.64 m/s (Table A4).

Pink Salmon (Oncorhynchus gorbuscha)

Pink Salmon are a semelparous anadromous salmon, and only 16 major populations of Pink Salmon occur in Washington waters (Wydoski and Whitney 2003). All of these major populations occur in rivers that drain into the Puget Sound or Strait of Juan de Fuca (Wydoski and Whitney 2003), although other river systems, including the Columbia River, support small runs (Heard 1991). The major Pink Salmon runs in Washington are almost all odd year runs, with even runs being restricted primarily to the Snohomish River watershed (Wydoski and

Whitney 2003). Spawning primarily occurs in upriver habitats, but spawning within tidally influenced reaches has been observed (Heard 1991, Wydoski and Whitney 2003). Adults typically begin to appear in the bays and estuaries of their natal rivers in June to September in the year after they entered marine waters, and migrate upriver to spawn in August to November (Heard 1991). Adults are typically small compared to other Pacific Salmon, ranging from 36 - 76 cm in fork length at maturity (Heard 1991, Wydoski and Whitney 2003). Adult pink typically spawn closer to the sea than other Pacific salmon, and are not known to be good at navigating high-velocity barriers or vertical barriers (Heard 1991), with adults having a maximum sustained swimming speed of 0.61-1.07 m/s and prolonged swimming speed of up to 1.73 m/s (Table A4).

Pink Salmon spend about 15 months maturing in the ocean before returning to spawn as two-year-old adults. This highly specialized life history structure is the least variable among Pacific salmon species, and results in the odd or even year run structures (Heard 1991). Likewise, juvenile Pink Salmon migrants will only be present in alternating years depending on the run timing of adults. Juvenile Pink Salmon migrate very soon after emergence and have the smallest body sizes (2.8 - 3.5 cm) as they migrate quickly through the estuaries of their natal rivers (Congleton 1981, Heard 1991, Quinn 2005). Downstream migrations typically occur from early February to late May, and peak in late March to early May (Quinn 2005). Juvenile Pink Salmon move quickly to marine habitats where they may spend an extended period of time in embayments and other shallow habitats, where they can reach 2.8 - 12.5 cm before moving offshore by June. Juvenile Pink Salmon avoid shadows from overwater structures like docks as well as overwater structures that lack shadows and are also known to avoid bulkhead and marinas (Wydoski and Whitney 2003, Ono et al. 2010). These behaviors may predispose Pink Salmon to avoid culverts and other water crossing structures. Water velocities of up to 0.1 m/s will generally not impede juvenile salmonid movements (Wightman and Tayler 1976), although Pink Salmon are small weak swimmers that primarily migrate downstream rapidly. Prolonged swimming speeds for juvenile Pink Salmon range from 0.14-0.16 and may fall below 0.1 m/s (Table A4, Wightman and Tayler 1976).

Chum salmon (Oncorhynchus keta)

Chum salmon are a semelparous anadromous Pacific Salmon species, with a similar distribution in Washington State to Chinook and Coho Salmon (Wydoski and Whitney 2003). Adult Chum can range in size from 43 – 108 cm, and may enter their spawning rivers in early September and continuing into March in some systems (Salo 1991, Wydoski and Whitney 2003). Hood Canal and Strait of Juan de Fuca summer Chum are distinct from other Puget Sound and Washington Chum populations, with adult migrations occurring earlier from August to September (Salo 1991, Tynan 1997, Wydoski and Whitney 2003). Most Chum Salmon spawn in the larger coastal rivers of Washington. Nevertheless, adults may also spawn in small coastal creeks and intertidal habitats, although this behavior may be more common in their northern ranges (Salo 1991,

Wydoski and Whitney 2003). Sustained swim speeds for adult Chum are 0.54 m/s with prolonged swim speeds of 0.52 - 3.10 m/s with a burst speeds of 1.82 - 4.60 m/s (Table A4). Although Chum adults are strong swimmers and can navigate relatively high velocities when sufficient water depth is present, they are not generally good leapers and will typically stop at vertical barriers during their migration upriver (Salo 1991).

Juvenile Chum migrate downstream soon after emergence, much like Pink Salmon, with downstream migrations typically occurring from early February to late May, and peak in late March to early May (Congleton 1981, Salo 1991, Quinn 2005). Downstream migrations of summer Chum occurring in Hood Canal and the Strait of Juan de Fuca typically peak in mid to late March as compared to later fall or winter run timings that peak in late April (Tynan 1997). Chum migrate downstream at 3.5 – 6.6 cm, and can spend up to 23 weeks rearing in their natal estuaries or nearby tidal channel habitats (Congleton 1981). During early marine entry, they may rear in lagoons and embayments for extended periods of time. Juvenile Chum move in and out of tidal channel habitats with tidal cycles during this rearing period, and can grow quickly to 24.9 cm by July or August before migrating to sea to mature (Congleton 1981, Salo 1991, Wydoski and Whitney 2003). Juvenile Chum are known to avoid shadows from over water structures like docks as well as overwater structures that lack shadows, which can delay migration and increase risk to predation (Ono et al. 2010), but can be attracted to shadows in natural environments from overhanging vegetation during their downstream migration (Salo 1991). Water velocities of up to 0.1 m/s will generally not impede juvenile salmonid movements (Whitman and Taylor 1976), and prolong swimming speeds range from 0.13-0.18 m/s (Table A4).

Cutthroat Trout (*Oncorhynchus clarki*)

Coastal Cutthroat Trout is perhaps the member of *Oncorhynchus* that is most tolerant to watershed impacts, and the species occurs in nearly all drainages (small and large) of Puget Sound and coastal Washington (Wydoski and Whitney 2003). Coastal Cutthroat are capable of expressing anadromous life histories and life histories that remain in freshwater throughout their entire life cycle and may make multiple spawning migrations, much like Steelhead (Wydoski and Whitney 2003). Anadromous adults typically return to freshwater spawning habitats after about three months, with some spending as little as a week in the ocean while others may spend up to five months in the ocean (Wydoski and Whitney 2003). This results in adults entering stream and river estuaries in July through January, with timings typically being later among Puget Sound populations. Adults are not known to overwinter in the ocean, but immature sub-adults may use non-natal river and streams to overwinter prior to maturing in the ocean before a spawning migration. However, coastal Cutthroat typically have high fidelity to their natal streams and will usually return to their natal river to spawn. Anadromous adults can reach sizes of up to 48 cm but may be as small as 20 cm.

Anadromous coastal Cutthroat Trout, juveniles typically migrate to the ocean after one to four years of rearing in freshwater, with most migrating after two or three years, and some evidence for migrations occurring after up to six years of rearing in freshwater (Wydoski and Whitney 2003). Downstream migrations 20 – 30 cm long juveniles typically occur from January through September, with peaks in April through May. Juveniles may use estuaries and shallow intertidal habitats extensively, with sub-adults spending remaining in estuarine habitats throughout their entire first migration to feed upon other fish.

Steelhead Trout (Oncorhynchus mykiss)

Steelhead trout are capable of expressing a resident Rainbow Trout life history that remains in freshwater throughout the life cycle or anadromous Steelhead life histories that migrate to the ocean to mature. The expression of resident and anadromous life histories is conditional, and all individuals have the capacity to express either life history trajectory regardless of parentage or previous selective pressures (Thrower and Joyce 2004, McPhee et al. 2007, Kendall et al. 2014, Phillis et al. 2015). Given that Steelhead can be found in rivers, streams, and creeks of almost all sizes in Washington state's coastal watersheds (Wydoski and Whitney 2003), anadromous Steelhead may be produced throughout Washington's coastal watersheds. Adult Steelhead typically return to spawning habitats in either the winter or summer, and adults can range in size from about 45 cm for 123 cm depending on the number of years spent in the ocean (Wydoski and Whitney 2003). Iteroparity is common among Steelhead, and therefore adults can utilize intertidal coastal creek confluence and large river deltas as both upstream and downstream migration corridors during adult life stages.

Downstream migrations of juvenile Steelhead can occur at a wide range of ages with highly variable and potentially extended freshwater rearing times, and migrants can be encountered throughout the year with peaks typically occurring in the fall, winter, and spring months (Wydoski and Whitney 2003, Hall et al. 2016). These migrants can range in size from young of the year that are less than 7 cm in length to under 18 cm for age-1 to age-3 juvenile migrants and as large as 23 cm for an age-7 juvenile migrant (Wydoski and Whitney 2003, Hall et al. 2016). Once in the intertidal estuarine habitats of small coastal creek confluences and large river deltas, anadromous juvenile Steelhead are thought to move through estuaries relatively quickly with little to no rearing within estuaries being common (Quinn 2005, Clements 2012). However, juvenile anadromous Steelhead, and potentially resident Rainbow Trout, have been observed moving between river systems through marine habitats (Hall et al. 2016). Therefore, juvenile *O. mykiss* have the potential to utilize river deltas and coastal creek confluences as downstream and upstream migration corridors. Within large river systems, juvenile Steelhead generally migrate downstream and through estuarine habitats very quickly. However, on the Pacific coast, juvenile Steelhead are known to make use of coastal confluences and lagoons with barrier beaches that

seasonally close, thereby providing juveniles highly productive in-river foraging areas until when these areas re-open to marine input (Bond 2006, Bond et al. 2008).

Adult Steelhead are capable of perhaps the greatest maximum swimming speeds among the salmonids, with burst speeds of 4.27-8.22 m/s, prolonged speeds of 1.5-4.4 m/s, and sustained speeds of 0.28-2.11 m/s (Table A4). Water velocities of up to 0.1 m/s will generally not impede juvenile salmonid movements, and the large range of size at migration among juvenile Steelhead suggests that swimming speeds will likely vary greatly with body size (Wightman and Tayler 1976). Sustained swimming speeds for juveniles range from 0.32-0.66 m/s with prolonged swimming speeds of 0.47-0.98 m/s and burst speeds of 0.73-2.2 m/s (Table A4).

Bull Trout (*Salvelinus confluentus*)

Like Steelhead, Bull Trout can express anadromous and resident life histories and it is likely that expression of these variations are conditional as in Steelhead given that all life history types can occur in the same systems (Wydoski and Whitney 2003, Hayes et al. 2011). Bull Trout can be found in small to large river systems throughout Washington's Puget Sound and coastal watersheds (Wydoski and Whitney 2003), Skagit River produces the most abundant population of anadromous Bull Trout (Hayes et al. 2011). Juvenile Bull Trout typically migrate to marine habitats from March through July at 9 – 29 cm depending the duration of freshwater rearing, but migrations to the ocean may also continue through December (Goetz 2004, Brenkman and Corbett 2005, Zimmerman and Kinsel 2010, Hayes et al. 2011). Sub-adults and adults typically rear extensively in shallow intertidal and subtidal habitats no more than 2-4 meters depth from April to July, although older individuals may move to deep areas of embayments in the summer to find the cold temperatures they prefer (Goetz 2004, Goetz 2007). Returns to freshwater occur primarily in May through August (Goetz 2004, Goetz 2007, Hayes et al. 2011). Mature adults can range reach sizes of up to 89 cm in size (Wydoski and Whitney 2003), with anadromous sub adults being 25.0 - 40.0 cm (Goetz 2004, Goetz 2007). Critical swimming speeds, which are a subcategory of prolonged swimming speed, for Bull Trout are an average of 0.5 m/s for 11-23 cm fish and 0.7 m/s for 32-42 cm fish (Table A4, Mesa et al. 2004). Juvenile Bull Trout generally prefer slower velocity habitat with instream cover (Mesa et al. 2004).

Table A4. Sustained, prolonged, and burst swimming speeds of adult and juvenile life stages for selected Pacific Salmonid and trout species. UNK indicates that migration information was not found for that species, life stage, and shoreform type but migration may occur.

Species	Life Stage	Sustained (>200 min, m/s)	Prolonged (20 sec - 200 min, m/s)	Burst (< 20 sec, m/s)	Sources
Chinook	Adult	1.22-3.3	1.25-1.96	1.00-6.71	Wightman and Tayler 1976, Beamish 1978, Bell 1991, Brown and Geist 2002, Geist et al. 2003, Wydoski and Whitney 2003, Trudel et al. 2004
Chinook	Juvenile	UNK	0.14-0.50	0.57-0.70	Kerr 1953, Weaver 1963, Wightman and Tayler 1976, Randall et al 1987, Smith & Carpenter 1987, Sambilay 1990
Coho	Adult	0.31-3.20	0.97-1.13	1.61-6.40	Weaver 1963, Paulik & DeLacey 1957, Beamish 1978, Bell 1991, Webb 1971, Ellis 1966, Bainbridge 1960, Paulik et al. 1957, Lee et al. 2003, Glova & McInerney 1977, Sandercock 1991, Farrell et al 2003
Coho	Juvenile	0.06-0.64	0.07-0.58	UNK	Beamish 1978, Smith & Carpenter 1987, Glova & McInerney 1977, Bainbridge 1960, Brett et al. 1958, Bell 1991
Pink	Adult	0.61-1.07	0.89-1.73	UNK	Sambilay 1990, Ware 1978, Farrell et al 2003, Brett 1982
Pink	Juvenile	UNK	0.14-0.16	UNK	Smith & Carpenter 1987, Brett 1982
Chum	Adult	0.54	0.52-3.10	1.82-4.60	Powers & Orsborn 1985, Brett 1982, Aaserude & Orsborn 1986, Houston 1959, Beamish 1978, Salo 1991
Chum	Juvenile	UNK	0.13-0.18	UNK	Smith & Carpenter 1987, Houston 1959, Wightman and Tayler 1976
Steelhead	Adult	0.28-2.11	1.5-4.40	4.27-8.22	Webb 1971, Bainbridge 1960, Bell 1991, Clay 1995
Steelhead	Juvenile	0.32-0.66	0.47-0.98	0.73-2.2	Bainbridge 1960, Beamish 1978, Jones et al. 1974
Bull Trout	Adult	UNK	0.5-0.7	UNK	Mesa et al. 2004
Bull Trout	Juvenile	UNK	UNK	UNK	--

Eulachon (Thaleichthys pacificus)

Eulachon are anadromous smelt that are primarily found in Columbia River and north Puget Sound rivers (Wydoski and Whitney 2003). Adults typically spawn upriver of salt influence from February to March and may range in size 13 - 30 cm depending on the age of return (Wydoski and Whitney 2003, Clarke et al. 2007). Larval fish are 0.5 cm in length at emergence and drift passively to estuarine habitats from March through April, where they may rear for up to a year before moving to deeper offshore waters, but the duration and size distributions of juveniles during estuary residency is poorly understood (Hay and McCarter 2000, Wydoski and Whitney 2003, Clarke et al. 2007). Eulachon are weak swimmers, prefer slower waters, and juveniles are considered passive drifters (Clarke et al. 2007).

Pacific Herring (Clupea pallasii)

Pacific Herring are one of the three most common forage fish species in Washington's nearshore waters and utilize every marine and estuarine habitat (Penttila 2007). Although Pacific Herring are primarily marine species, they extensively use intertidal habitats in embayments for spawning and rearing (Norcross et al. 2001, Penttila 2007, Small et al. 2011). Mature adults aggregate in schools near inshore spawning habitats from January to June in preparation for spawning. Spawning typically occurs from February through May (Small et al. 2011). Pacific Herring typically spawn from January through April in south-central Puget Sound and the Strait of Juan de Fuca, and January through June in North Puget Sound (Penttila 2007), and similar time frames on the Washington coast. Juveniles emerge a few weeks after spawning, and the larval fish passively drift in the water column 2-3 months and then move to deeper offshore, shallow inshore estuaries, or inland waters to mature (Small et al. 2011, Norcross et al. 2001). Juvenile herring that use shallow inshore estuaries for rearing may grow from 3.5 to 15 cm in length before moving out of shallow nursery bay and estuary habitats by their first August following emergence (Small et al. 2011, Norcross et al. 2001). Mature adult Pacific Herring are 11-46 cm in length (Small et al. 2011, Norcross et al. 2001).

Surf Smelt (Hypomesus pretiosus)

Surf Smelt are also one of the most common forage fish species in Washington's coastal waters, and can be found in all marine and estuarine habitats in Puget Sound and coastal waters of Washington State (Loosanoff 1938, Penttila 2007). Although Surf Smelt are primarily a marine species, they extensively use intertidal and shallow subtidal habitats in embayments for spawning and during rearing in their first year of life (Penttila 2007). Surf Smelt spawn in upper intertidal ranges of gravel-sand beaches throughout Puget Sound and coastal Washington nearshore areas, and spawning can occur at all seasons or with peaks in the fall/winter (September – March) or summer months (May – August) (Rice 2006, Penttila 2007). Adults and their eggs are tolerant to a wide range of salinities, and thus are common in estuaries with strong freshwater influence as well as marine dominated embayments (Quinn et al. 2012, Weitkamp et

al. 2012). Larval Surf Smelt are about 4 cm in length and are passive drifters much like Pacific Herring during their first phase of life, and adults can attain a size of about 21 cm (Loosanoff 1938, Quinn et al. 2012, Weitkamp et al. 2012). Surf Smelt are not known to make migrations to the open ocean, and may spend their entire lives in the shoreline or estuarine habitats near their spawning sites (Penttila 2007).

English Sole (Parophrys vetulus)

English Sole rely extensively on estuaries as juveniles (Gunderson et al. 1990). Adults can reach a size of 62 cm, but larger adults are typically only found in deep-water habitats. Adult English sole spawn off-shore from September through April, but their early pelagic larvae move to coastal and estuarine areas for rearing. Settling larval fish can be as small as 1.5 cm, are strongly associated with substrate, and the timing of their settlement into estuarine rearing habitats can be variable depending on spawning timing (Gunderson et al. 1990). Settlement in estuarine habitats typically occurs between May and August, with most juveniles settling in estuarine habitats by 5.5 cm (Gunderson et al. 1990). Migration out of estuarine rearing habitats typically occurs at about 8 cm with most fish larger than 12.5 cm having left the estuaries to continue rearing in coastal waters.

Dungeness Crab (Cancer magister)

Dungeness Crab rely extensively on estuaries as juveniles (Gunderson et al. 1990). Dungeness Crab adults typically release larvae between January and February, and the pelagic larvae can settle on substrate at a size of 0.6 cm total carapace width beginning in April through June (Gunderson et al. 1990). Larval crab can settle on a variety of substrates, and will typically remain within the habitat in which they settled until they molt (Gunderson et al. 1990).

Therefore, juveniles can initially rear in subtidal to intertidal habitats in a range of conditions. Those settling in estuarine habitats have faster growth rates than the cohorts settling in subtidal habitats, attaining sizes of up to 5.0 cm total carapace width in contrast to 1.8 cm by September. Subtidal rearing juveniles may also enter the estuary in the April after their first year at 5.6-8.7 cm. Most juveniles migrate from their estuarine nurseries after their first year of rearing (~ 10 cm in width) to embayments or the ocean (Gunderson et al. 1990).

Three-spine Stickleback (Gasterosteus aculeatus)

Three-spine Stickleback are one of the most tolerant and widespread among Washington's native fishes, and can be found in completely freshwater to marine habitats at almost any time of the year throughout the Washington coastline (Wydoski and Whitney 2003). The marine form is anadromous and typically migrates to freshwater around June, and can be 3.8 - 9.5 cm total length (Wydoski and Whitney 2003), and newly emerged broods of juveniles can be smaller than 2 cm in total length. The small body size and weak swimming abilities of Three-spine Stickleback make this species sensitive to impacts from high water velocities or vertical drops (Wydoski and Whitney 2003). However, this species can often be found as the most common

species above and below water crossing structures that completely disconnect hydrology for long periods of time, e.g., a flap gated structure that remains closed for months (Greene et al. 2013).

Shiner Surfperch (Cymatogaster aggregata)

Shiner Surfperch can be found throughout the tidal range of large river deltas, coastal creek confluences, embayments, and lagoons throughout Washington State's coastline and can tolerate a wide range of salinities and temperatures (Wydoski and Whitney 2003). Shiner Surfperch are born live typically between July and August, and range in size from 2.2 - 4.3 cm (Wydoski and Whitney 2003). Juveniles will typically remain in estuarine habitats until about 8 cm total length by late fall, when they move to deeper water habitats where they can reach a sizes of 9.4 - 20 cm total length.

Staghorn Sculpin (Leptocottus armatus)

Staghorn Sculpin can be found in subtidal to tidal habitats throughout Washington state's shoreline and estuary habitats, and can tolerate a wide range of salinities and temperatures (Wydoski and Whitney 2003). The adults can reach sizes of up to 34 cm and are typically found where salinities are higher, e.g., in the lower extent of estuaries or in intertidal and subtidal areas of other shore form habitats, as they require higher salinities for spawning. Adults spawn from October through March, with hatching larvae being 0.5 cm in length (Wydoski and Whitney 2003). The juveniles have a high tolerance for a wide range of salinities, and will commonly move into freshwater and estuarine habitats in the spring (Wydoski and Whitney 2003). As they grow larger, they move down to the lower estuary or higher salinity marine habitats in the summer through fall (Wydoski and Whitney 2003).

River Lamprey (Lampetra ayresii)

River Lamprey are anadromous and spawn in rivers throughout Washington State's shoreline (Wydoski and Whitney 2003). Juveniles spend a variable amount of time rearing in freshwater before they migrate downstream. Juvenile migrations to marine habitats occur from April to July (Beamish and Youson 1987, Wydoski and Whitney 2003), and these migrants range in size from 4-19 cm in length (Wydoski and Whitney 2003). Juvenile River Lamprey found in the Columbia River estuary from April through September, and individuals were actively feeding and rearing in estuarine habitats. (Weitkamp et al. 2015). Some individuals may remain in brackish waters until they mature (Beamish and Youson 1987), while others rear in surface waters of reduced salinity waters from July through September (Beamish and Youson 1987, Wydoski and Whitney 2003). Returning adults are 20-30 cm in length, and spawn in the rivers from April through June (Beamish and Youson 1987, Wydoski and Whitney 2003, Weitkamp et al. 2015). However, little is known about where these returning adults rear until they spawn. Some research suggests that they could remain in saltwater, in estuaries and intertidal habitats of their spawning rivers or in riverine habitats until they spawn (Beamish 1980). Spawning appears to be restricted to freshwater habitats (Beamish 1980). As with Pacific Lamprey, adult River Lamprey are capable

of passing barriers like boulder cascades, waterfalls, and dam walls by clinging to wetted surfaces and slowly climbing up barriers (Wydoski and Whitney 2003). This capability should be considered in addition to swimming speeds when evaluating barriers to adult migration.

Pacific Lamprey (Entosphenus tridentatus)

Pacific Lamprey are anadromous and spawn in river systems throughout Puget Sound and Washington's coasts (Wydoski and Whitney 2003, Hayes et al. 2013). The juveniles rear in freshwater for 4-7 years before metamorphosing and migrating to the ocean where they will rear for 1-2 years before returning to freshwater rivers to spawn (Hayes et al. 2013). Among Puget Sound rivers and the Columbia River estuary, adults range in size from 19.4 – 74.5 cm, and juveniles are 10.2 – 15.7 cm at the time of migration to salt water (Hayes et al. 2013, Weitkamp et al. 2015). Juvenile migrations to saltwater may generally occur from November through June (Beamish 1980, Hayes et al. 2013, Weitkamp et al. 2015), although there is considerable variation in peak timing among Washington's river systems (Hayes et al. 2013). Adults generally return from deep-water ocean rearing habitats in April – September, after which they overwinter in freshwater in preparation for spawning in the spring (Beamish 1980, Wydoski and Whitney 2003). Research suggests that Pacific Lamprey juveniles may need to move to saltwater relatively quickly after metamorphosis, and therefore do not likely rear in estuaries for long periods of time (Beamish 1980). As with River Lamprey, adult Pacific Lamprey are capable of passing barriers like boulder cascades, waterfalls, and dam walls by clinging to wetted surfaces and slowly climbing up barriers (Wydoski and Whitney 2003). This unique capability should be considered in addition to swimming speeds when considering barriers to adult lamprey migration. Juveniles ranging in size from 13.6 – 16.5 cm have burst speeds that range from 0.6-0.9 m/s, with sustained swimming speeds of 0-0.5 m/s (Dauble et al. 2006).

Smallmouth Bass (Micropterus dolomieu) and Largemouth Bass (Micropterus salmoides)

Smallmouth and Largemouth Bass are an introduced species that can be found in watersheds that drain into Puget Sound and Columbia River estuaries (Wydoski and Whitney 2003). Both bass species are generally confined to freshwater habitats, but can be found in tidally influenced and low salinity habitats in the upper reaches of estuary systems (Schmidt and Stillman 1998, Brown et al. 2009, Greene et al. 2013). However, residence in low salinity habitats is likely short in duration (Brown et al. 2009). Greene et al. (2012) found juvenile bass ranging in size 4-20 cm in tidally influenced sites in June – July, and Schmidt and Stillman (1998) observed Smallmouth Bass moving into estuarine habitats from June – July at 0.8 – 2.8 cm. Unpublished data for the Snohomish River estuary indicates bass were present in intertidal and estuarine habitats from May through October and ranged in size from 3.5 – 20.5 cm. Smallmouth Bass fry can maintain position in water velocities up to 0.14-0.16 m/s (Schmidt and Stillman 1998), with adult prolonged swimming speeds that range from 0.5 m/s to 0.6 m/s for fish ranging in size from 24 to 45 cm (Peake 2004).

Appendix 2. References

- Beamer, E., A. McBride, C. Greene., R. Henderson, G. Hood, K. Wold, K. Larsen, C. Rice and K. Fresh (2005). Delta and Nearshore Restoration for the Recovery of Wild Skagit River Chinook Salmon: Linking Estuary Restoration to Wild Chinook Salmon Populations. Appendix D, Skagit Chinook Recovery Plan 2005, Skagit River System Cooperative and the Washington Department of Fish and Wildlife.
- Beamer, E., A. McBride, R. Henderson and K. Wolf (2003). The importance of non-natal pocket estuaries in Skagit Bay to wild Chinook Salmon: an emerging priority for restoration, Skagit System Cooperative Research Report.
- Beamer, E., W. Zackey, D. Marks, D. Teel, D. Kuligowski and R. Henderson . (2013). Juvenile Chinook Salmon rearing in small non-natal streams draining into the Whidbey Basin. Skagit River System Cooperative. LaConner, WA.
- Beamish, R. J. (1980). Adult biology of the River Lamprey (*Lampetra ayresi*) and the Pacific Lamprey (*Lampetra tridentata*) from the Pacific coast of Canada. Canadian Journal of Fisheries and Aquatic Sciences 37: 1906-1923.
- Beamish, R. J., and C. E. M. Neville (1995). Pacific salmon and Pacific Herring mortalities in the Fraser River plume caused by River Lamprey (*Lampetra ayresi*). Canadian Journal of Fisheries and Aquatic Sciences 52: 644-650.
- Bennett, T. R., P. Roni, K. Denton, M. McHenry and R. Moses (2014). Nomads no more: early juvenile Coho Salmon migrants contribute to the adult return. Ecology of Freshwater Fish 24: 264-275.
- Bennett, T. R., R. C. Wissmar and P. Roni (2011). Fall and spring emigration timing of juvenile Coho Salmon from the East Twin River, Washington. Northwest Science 85: 562-570.
- Bond, M. H. (2006). Importance of estuarine rearing to central California Steelhead (*Oncorhynchus mykiss*) growth and marine survival . Masters thesis, University of California Santa Cruz, 59 pp.
- Bond, M. H., S. A. Hayes, C. V. Hanson and R. B. MacFarlane (2008). Marine survival of Steelhead (*Oncorhynchus mykiss*) enhanced by a seasonally closed estuary. Canadian Journal of Fisheries and Aquatic Sciences 65: 2242-2252.
- Brenkman, S. J., and S. C. Corbett (2005). Extent of anadromy in Bull Trout and implications for conservation of a threatened species. North American Journal of Fisheries Management 25: 1073–1081.

- Brett, J. R. (1952). Temperature tolerance in young Pacific salmon, genus *Oncorhynchus*. Journal of the Fisheries Research Board. Canada 9: 265-323.
- Brett, J. R., (1982). The swimming speed of adult Pink Salmon, (*Oncorhynchus gorbuscha*), at 20° C and a comparison with sockeye salmon (*O. nerka*). Canadian Technical Report of Fisheries and Aquatic Sciences, Report 1143.
- Brown, T. G., B. Runciman, S. Pollard, A. D. A. Grant and M. J. Bradford (2009). Biological synopsis of Smallmouth Bass (*Micropterus dolomieu*). Canadian Manuscript Report of Fisheries and Aquatic Sciences 2887.
- Clarke, A. D., A. Lewis, K. H. Telmer and J. M. Shrimpton (2007). Life history and age at maturity of an anadromous smelt, the eulachon *Thaleichthys pacificus* (Richardson). Journal of Fish Biology 71: 1479-1493.
- Clay, C.H. (1995). Design of fishways and other fish facilities. CRC Press, Boca Raton, Florida.
- Clements, S., T. Stahl and C. B. Schreck (2012). A comparison of the behavior and survival of juvenile Coho Salmon (*Oncorhynchus kisutch*) and Steelhead trout (*O. mykiss*) in a small estuary system. Aquaculture 362-363: 148-157.
- Congleton, J. L., S. K. Davis and S. R. Foley (1981), June. Distribution, abundance and outmigration timing of Chum and chinook salmon fry in the Skagit salt marsh. In Proceedings of the Salmon and Trout Migratory Behavior Symposium, pp. 153-163.
- Dauble, D. D., R. A. Moursund and M. D. Bleich (2006). Swimming behavior of juvenile Pacific Lamprey, *Lampetra tridentata*. Environmental Biology of Fishes. 75: 167-171.
- Goetz, F. A., E. Jeanes and E. Beamer. (2004). Bull Trout in the nearshore. U.S. Army Corps of Engineers, Preliminary Draft, Seattle.
- Goetz, F., E. Jeanes, and C. Morello. (2007). Puget Sound Bull Trout migration and habitat use study, Nooksack River and estuary and northeast Puget Sound nearshore. Progress report prepared for the U.S. Fish and Wildlife Service, Interagency Agreement 13410-6-H001, by the U.S. Army Corps of Engineers, Seattle, and R2 Resource Consultants, Redmond, Washington.
- Greene, C., J. Hall, E. Beamer, R. Henderson and B. Brown (2012). Biological and physical effects of “fish-friendly” tide gates. Report prepared for the Estuary and Salmon Restoration Program.
- Gunderson, D. R., D. A. Armstrong, Y. B. Shi and R. A. McConnaughey (1990). Patterns of estuarine use by juvenile English Sole (*Parophrys vetulus*) and Dungeness Crab (*Cancer magister*). Estuaries 13: 59-71.

- Hall, J., P. Roni, T. Bennett, J. McMillan, K. Hanson, R. Moses, M. McHenry, G. Pess and W. Ehinger (2016). Life history diversity of Steelhead in two coastal Washington watersheds. *Transactions of the American Fisheries Society* 145: 990-1005.
- Hay, D., and P.B. McCarter (2000). Status of the eulachon *Thaleichthys pacificus* in Canada. Canadian Stock Assessment Secretariat, Research Document 2000/145.
- Hayes, M. C., R. Hays, S. P. Rubin, D. M. Chase, M. Hallock, C. Cook-Tabor, C. W. Luzier and M. L. Moser (2013). Distribution of Pacific Lamprey *Entosphenus tridentatus* in watersheds of Puget Sound based on smolt monitoring data. *Northwest Science* 87: 95-105.
- Hayes, M.C., S. P. Rubin , R. R. Reisenbichler , F. A. Goetz , E. Jeanes and A. McBride (2011). Marine Habitat Use by Anadromous Bull Trout from the Skagit River, Washington, Marine and Coastal Fisheries 3: 394-410.
- Healey, M.C. (1991). Life history of Chinook Salmon (*Oncorhynchus tshawytscha*). In Pacific Salmon Life Histories. C. Groot and L. Margolis, editors. University of British Columbia Press, Vancouver, BC, Canada.
- Heard, W.R. (1991). Life history of Pink Salmon (*Oncorhynchus gorbuscha*). In Pacific Salmon Life Histories. C. Groot and L. Margolis, editors. University of British Columbia Press, Vancouver, BC, Canada.
- Jones, K., T. Cornwell, D. L. Bottom, L. A. Campbell and S. Stein (2014). The contribution of estuary-resident life histories to the return of adult *Oncorhynchus kisutch*. *Journal of Fish Biology* 85: 52-80.
- Kendall, N. W., J. R. McMillan, M. R. Sloat, T. W. Buehrens, T. P. Quinn, G. R. Pess, K. V. Kuzishchin, M. M. McClure, and R. W. Zabel (2014). Anadromy and residency in Steelhead and rainbow trout *Oncorhynchus mykiss*: a review of the processes and patterns. *Canadian Journal of Fisheries and Aquatic Sciences* 72: 319–342.
- Kuzishchin, K. V., A. Y. Mal'tsev, M. A. Gruzdeva, K. A. Savvaitova, D. S. Pavlov and J. A. Stanford (2007). On joint spawning of anadromous and resident mykiss *Parasalmo mykiss* in rivers of western Kamchatka. *Journal of Ichthyology* 47: 348–352.
- Loosanoff, V.L. (1938). The spawning run of the pacific Surf Smelt *Hypomesus pretiosus*. *International Review of Hydrobiology*. 36: 170-183.
- McPhee, M. V., F. Utter, J. A. Stanford, K. V. Kuzishchin, K. A. Savvaitova, D. S. Pavlov and F. W. Allendorf (2007). Population structure and partial anadromy in *Oncorhynchus mykiss* from Kamchatka: relevance for conservation strategies around the Pacific Rim. *Ecology of Freshwater Fish* 16: 539–547.

- Mesa, M. G., L. K. Weiland and G. B. Zydlewski (2004). Critical swimming speeds of wild Bull Trout. Northwest Science. 78: 59-65.
- Moore, J. W., J. D. Yeakel, D. Peard, J. Lough and M. Beere (2014). Life-history diversity and its importance to population stability and persistence of a migratory fish: Steelhead in two large North American watersheds. Journal of Animal Ecology 83: 1035–1046.
- Norcross, B. L., E. D. Brown, R. J. Foy, M. Frandsen, S. M. Gay, T. C. Kline Jr., D. M. Manson, E. V. Patrick, A. J. Paul and K. D. E. Stokesbury (2001). A synthesis of the life history and ecology of juvenile Pacific Herring in Prince William Sound, Alaska. Fisheries Oceanography 10: 42-57.
- Ono, K., C. Simenstad, J. Toft, S. Southard, K. Sobocinski and A. Borde (2010). Assessing and mitigating dock shading impacts on the behavior of juvenile Pacific salmon (*Oncorhynchus* spp.): Can artificial light mitigate the effects. Washington State Department of Transportation Research Report. WA-RD, 755 pp.
- Peake, S. (2004). An evaluation of the use of critical swimming speed for determination of culvert water velocity criteria for Smallmouth Bass. Transactions of the American Fisheries Society 133: 1472-1479.
- Penttila, D. (2007). Marine forage fishes in Puget Sound. Puget Sound Nearshore Partnership Report No. 2007-03, Seattle District, U.W. Army Corps of Engineers.
- Phillis, C. C., J. W. Moore, M. Buoro, S. A. Hayes, J. C. Garza and D. E. Pearse (2015). Shifting thresholds: rapid evolution of migratory life histories in Steelhead/rainbow trout, *Oncorhynchus mykiss*. Journal of Heredity 107: 51-60.
- Quinn, T., K. Krueger, K. Pierce, D. Penttila, K. Perry, T. Hicks and D. Lowry (2012). Patterns of Surf Smelt, *Hypomesus pretiosus*, intertidal spawning habitat use in Puget Sound, Washington State. Estuaries and Coasts 35: 1214-1228.
- Quinn, T.P. (2005). The behavior and ecology of Pacific Salmon and Trout. American Fisheries Society, Bethesda, Maryland.
- Rice, C.A. (2006). Effects of shoreline modification on a northern Puget Sound beach: microclimate and embryo mortality in Surf Smelt (*Hypomesus pretiosus*). Coastal and Estuarine Research Federation 29: 63-71.
- Salo, E.O. (1991). Life history of Chum Salmon (*Oncorhynchus keta*). In Pacific Salmon Life Histories. C. Groot and L. Margolis, editors. University of British Columbia Press, Vancouver, BC, Canada.
- Savvaitova, K. A., K. V. Kuzishchin, M. A. Gruzdeva, D. S. Pavlov, J. A. Stanford and B. K. Ellis (2003). Long-term and short-term variation in the population structure of Kamchatka

- Steelhead *Parasalmo mykiss* from rivers of western Kamchatka. *Journal of Ichthyology* 43: 757–768.
- Schmidt, R. E., and T. Stillman (1998). Evidence of potamodromy in an estuarine population of Smallmouth Bass (*Micropterus dolomieu*). *Journal of Freshwater Ecology* 13: 155-163.
- Simenstad, C. A., and J. R. Cordell (2000). Ecological assessment criteria for restoring anadromous salmon habitat in Pacific Northwest estuaries. *Ecological Engineering* 15: 283-302.
- Small, M. P., J. L. Loxterman, A. E. Frye, J. F. Von Bargen, C. Bowman and S. F. Young (2005). Temporal and spatial genetic structure among some Pacific Herring populations in Puget Sound and the Southern Strait of Georgia. *Transactions of the American Fisheries Society* 13: 1329-1341.
- Thorpe, J. E. (1994). Salmonid fishes and the estuarine environment. *Estuaries* 17: 76–93.
- Thrower, F. P., and J. E. Joyce. (2004). Effects of 70 years of freshwater residency on survival, growth, early maturation, and smolting in a stock of anadromous rainbow trout from southeast Alaska. *American Fisheries Society Symposium* 44: 485–496.
- Tynan, T. (1997). Life history characterization of summer Chum Salmon population in the Hood Canal and eastern Strait of Juan de Fuca Regions. Washington Department of Fish and Wildlife Report 00153.
- Weitkamp, L. A., P. J. Bentley and M. N. Litz (2012). Seasonal and interannual variation in juvenile salmonids and associated fish assemblage in open waters of the lower Columbia River estuary, USA. *U.S. National Marine Fisheries Service Fishery Bulletin* 110:426–450.
- Weitkamp, L. A., S. A. Hinton and P. J. Bentley (2015). Seasonal abundance, size, and host selection of western river (*Lampetra ayresii*) and Pacific (*Entosphenus tridentatus*) lampreys in the Columbia River estuary. *Fishery Bulletin* 113: 213-226.
- Wightman, J. C. and C. D. Taylor (1976). Salmonid swimming performance in relation to passage through culverts. Fish and Wildlife Branch, Ministry of Recreation and Conservation, Victoria, B.C.
- Wydoski, R. S. and R. R. Whitney (2003). *Inland Fishes of Washington*. American Fisheries Society, Bethesda, Maryland.
- Zimmerman, M. and C. Kinsel (2010). Migration of anadromous juvenile Bull Trout in the Skagit River, 1990-2009. Washington Department of Fish and Wildlife Report #01173.

Appendix 3. Glossary

Anadromous: A diadromous life history strategy whereby adults spawn in freshwater but mature in estuarine and marine environments.

Bridge: A structure that spans a body of water, water-way, or valley to provide passage over the obstacle. Bridge designs vary but generally do not have a constructed floor that fully encloses the water way or valley that it spans. However, piers, open-bottom vaults, or other structural arches may be used to support the spanning structure across the span of the structure.

Blind tidal channel: Tidally influenced channels that primarily drain tidally flooded water, as opposed to distributaries that convey riverine or upland runoff to the marine environment. Blind tidal channels may have multiple connections to distributaries or the marine environment, but hydrological processes are dominated by tidal flooding and drainage from tidal forcing as opposed to riverine forcing.

Catadromous: A diadromous life history strategy whereby adults migrate to marine environments to spawn and juveniles mature in freshwater.

Causeway: An elevated structure over wet or low ground, typically to convey road, railway, or foot traffic.

Coastal creek confluence: The confluence of small coastal creek or river systems with the marine environment, which usually have a higher slope and reduced upstream tidal influence as compared to large river delta systems.

Connectivity: The degree to organisms or hydrogeomorphic processes can move or flow between habitats. Connectivity can be reduced by natural processes (e.g., a seasonal barrier beach at a small coastal creek confluence) or by water crossing structures. Although connectivity impacts can be similar in riverine and tidally dominated environments, tidal forcing requires consideration of extend landward impacts through restriction of tidal flooding in contrast to environments dominated by the predominately unidirectional downstream forcing of riverine dominated environments.

Critical Habitat: Specific geographic areas defined for species listed as Threatened or Endangered under the Endangered Species Act that contain features essential to the conservation of the species. Federal agencies are required to avoid significant land use adverse modifications of designated Critical Habitat.

Cross-sectional area: The two-dimensional area of a water crossing structure or channel that is measured perpendicular to the direction of flow along the plane of the water crossing structure.

Culvert: Typically, a structure that allows water to flow under or through another structure (e.g., embankment or dike), which is typically embedded in soil or concrete. A culvert is often cylinder like in shape, but can also be used to generally refer to any shaped passage conveying water through another structure.

Decision framework: A tool that supports the application of impact assessment of water crossing structures, which can be used to direct the application of management actions based on best practices (WDFW 2009).

Design flow: A designated upper limit of flow through which upstream fish-passage criteria are satisfied based on flow exceedance probabilities. Design flows consider water crossing structure dimensions, materials, elevation, and slope as well as flow exceedance probabilities.

Design life: The anticipated life of a structure that considers construction materials, deterioration rates under the designed conditions, and catastrophic events of a certain exceedance probability.

Design tide: An upper tidal limit or tidal regime that is considered in combination with design flows to determine fish passage criteria are satisfied. Design tides consider water crossing structure dimensions, materials, elevation, and slope as well as tide and flow exceedance probabilities.

Diadromous: A suite of life history strategies where life cycles are spent in freshwater and saltwater environments.

Dike: An elongated elevated wall or berm designed to prevent landward tidal flooding.

Embayment: Protected bays, which are generally outside of the direct influence of large river deltas, where wave action limits the formation of beaches, while allowing the formation of tidal channel, tidal marshes, and tidal flats (Shipman 2008).

Essential Fish Habitat: Essential Fish Habitat (EFH) is defined in the Magnuson-Stevens Fishery Conservation and Management Act as ‘those waters and substrates necessary to fish for spawning, breeding, feeding or growth to maturity.’ Essential Fish Habitat is defined for all

stocks with commercial fishery management plans determined by the National Marine Fisheries Services and approved by regional fishery management councils. .

Euryhaline: Organisms that are able to adapt to a wide range of salinities, which may include transitions between different salinities at different life stages or ability to move between different salinities within and between different life stages.

Eutrophication: A process by which excess nutrients cause dense growths of primary producers, e.g., aquatic plants or phytoplankton, and the death of animal life from reduced oxygen as the excessive biomass of primary producers is decomposed by oxygen consuming bacteria.

Facultative: An organism that is not constrained to one habitat type and is capable of utilizing a range of habitat types within or across life stages.

Flapgate: A passively controlled structure that caps the downstream end of culverts or vaults, which open when a hydraulic head differential occurs from the upstream end but closes when the weight of water exceeds the hydraulic head from the upstream side. Flapgates are typically top hinged, but may also be side hinged.

Floodgate: One-way valves, e.g., rubberized conical pinch valves or doors, fitted to the downstream end of culverts or vaults that allow drainage of upstream waters when a hydraulic head differential occurs from the upstream end but closes when the weight of water exceeds the hydraulic head from the upstream side. During periods of high precipitation and river flows, floodgates are normally used to prevent inundation by high riverine flood waters and to facilitate drainage of areas upland of floodgates. Some floodgates are manually operated, and left open when precipitation and river flows subside in the late spring or summer.

Fyke net: A funnel-shaped net set across a channel used to sample small juvenile fish moving seaward on ebbing tides. Fish leave the narrow end of fyke into a collection box where they are enumerated.

Habitat: “the physical, biological, and chemical characteristics of specific unit of the environment occupied by a specific plant or animal”. (Fresh et al. 2004)

Habitat quality: Measures of habitat parameters that determine the suitability of the habitat to aquatic organisms, which can include a number of water quality parameters (e.g., temperature, salinity, turbidity, dissolved oxygen, or pH) as well as abiotic parameters (e.g, invasive species, prey availability).

Habitat quantity: A physical measure of habitat quantity, usually by surface area or volume for specific habitat units.

Hydraulic model: Models that can be used to describe flows through a water-crossing structure. These models are typically constructed as either 1d, 2d, or 3d models. One-dimensional (1d) models are useful for water surface profiling, determining depth averaged cross-sectional velocity normal to the direction of flow. Two-dimensional (2d) models are grid-cell based and can provide output representing depth-averaged velocity in different grid cells, generating fine-scale predictions that are independent of the cross section as a whole. Similarly, three-dimensional (3d) models add a z component to the vector output and can include structured and unstructured grid assemblies that allow for evaluating complex situations where flow can split vertically around structures.

Hydrogeomorphic processes: The interaction and linkage of hydrological (e.g., river flows and tidal flows) and landform processes (e.g., sediment transport) across spatial or temporal dimensions.

Hydromodification: Alterations of natural water flow through a landscape that often include channel modifications and channelization, but also include changes in land cover and land use changes that impact water flow through a landscape.

Hypoxia: A state in which dissolved oxygen concentrations have decreased below a threshold at which aquatic animals can obtain sufficient oxygen to support metabolic activity. The threshold for hypoxia varies by species, species assemblages, and ecosystems.

Invert: The bottom of a culvert or vault.

Iteroparous: A species that is capable of spawning multiple times before dying.

Lagoon: A subset of the embayment shoreform habitat type, which are protected or sheltered habitats that are dominated by marine processes. Lagoons are more dynamic than embayments as these habitats are created by wave-based transport of materials which can change the structure of the habitat over time more so than larger embayment features (Shipman 2008).

Large river delta: The interface between large rivers and marine ecosystems. The physical structure of river deltas can also be highly variable and will vary greatly based on the dominant processes that form and maintain the river delta system. River deltas generally form the most

complex systems of intertidal habitat among the different habitat types. Common features to river delta systems include distributaries, tidal channels, tidal marshes, tidal flats, and subtidal flats (Shipman 2008).

Levee: Similar in construction to a dike, but designed to prevent upland flooding from riverine floods.

Natal: Pertaining to the area in which an organism was born, usually in reference to an anadromous species and their spawning river.

Non-natal: Pertaining to the areas where the individual was not born, but which may be used for spawning or rearing.

Obligate: An organism that is constrained to a specific habitats or life histories based on physiological requirements or tolerances, which may change across life stages.

Passability: The capability or quantification of passage through a structure or obstacle.

Perch: The height above the culvert or vault invert (floor) and the downstream or upstream water level. A culvert or vault that is perched has a floor that is elevated above the downstream or upstream water level, which can create a passage barrier due to plunging water or disconnected flow between upstream and downstream habitats.

Pet door: A small hole in a larger gate that can open independently of the larger door, either through passive or active controls. Pet doors are designed to allow for either fish passage or drainage.

PIT tag: A passive integrated transponder that can be used to tag small and large organisms, like juvenile and adult salmon. The tag can be decoded at stationary readers that span channels, or through mobile scanning, fish collections, or smolt traps to identify individual fish. This individual based detection of fish can be used to determine run sizes and survival parameters (e.g., outmigrant production and smolt to adult return rates), movement patterns (e.g., the timing and duration of movements related to migration or rearing), and growth rates.

Prolonged swimming speed: A swimming speed that can be maintained for 20 seconds to 200 minutes without fatigue (Beamish 1978).

Railroad embankment: A raised earthen berm that serves as the base for a railroad grade.

Rearing: An extended period of time spent in a particular habitat during which important growth or physiological adaptations or life stage transitions are made.

Residence: The portion of a fish's or other aquatic organism's life cycle spent in one habitat type. For example, migratory fish might have a very short (hours to days) residence in an estuary, while estuarine nursery species exhibit prolonged or protracted (weeks to months) estuarine residence. In the context of salmonids, this term is sometimes used to denote life histories that remain in one habitat type for the entire life cycle. For example, resident Rainbow Trout stay in freshwater rivers throughout its entire life cycle while Steelhead migrate to the ocean to mature before returning to spawn.

Rheotaxis: A form of behavioral attraction that causes fish to turn and face into currents.

Road embankment: An elevated earthen berm that serves as the base for a road or causeway.

Roughness: Quantification of the degree of drag on flowing water by a surface. Most commonly expressed as a dimensionless Manning's number.

Sea level rise: An increase in the volume of water in the ocean, which would result in an increase in the global mean sea level, local sea levels, and an upward shift in tidal datums.

Semelparous: A species that spawns only once before dying.

Self-regulating tide gate: Tidal gating structures that are designed to allow drainage from upstream of the structure while also providing some degree of tidal flooding into the area upstream of the structure. Some operate based on pressure differentials produced within a resistance mechanism, while others rely upon the vertical movements of floats to trigger opening and closing of gates. The gates themselves can vary from top- or side-hinged barriers of varying size, to walls that rotate on a cylinder.

Shoreform: Habitat types that are differentiated based on the predominance of physical processes that shape the habitats.

Slope: Change in vertical elevation with respect to horizontal distance.

Stenohaline: An organism that cannot tolerate a wide fluctuation in the salinity of water, which can be used for an exclusively freshwater or marine species.

Sustained swimming speed: Speeds that fish can maintain for long periods (>200 minutes) without muscular fatigue (Beamish 1978). Where cruising speeds are reported, these were considered a subcategory of sustained swimming speeds in this report.

Tidal attenuation: A decrease or muting of tidal forcing that results in a reduced tidal prism, tidal range, or intrusion of marine waters.

Tidal flux: Measurements of water volume, salt, or temperature exchange at the mouth of an estuary, delta, tidal channel, or coastal creek confluence through a tidal cycle.

Tidal muting structure: These structures are sometimes called “fish friendly tidegates”, “muted tidal regulators”, “self-regulating tidegates”, or “automated tide gates,” and are commonly used on culvert structures in dikes, levees, and road or railway embankments. In contrast to tidegates and floodgates, tidal muting structures are designed to allow drainage from upstream of the structure while also providing some degree of tidal flooding into the area upstream of the structure.

Tidal prism: The volume of water between mean high tide and mean low tide, which can also be expressed as an extent of tidal flooding within a system.

Tidegate: A culvert or vault that is fitted with a structure that restricts upstream flow of tidal waters while allowing downstream flow of upland runoff, which is usually passively controlled by differences in hydraulic head or manually controlled by valves or gates.

Water crossing structures: Structures that facilitate the transport or conveyance of people or materials over a body of water or waterway, which can include bridges, causeways, and culverts, vaults.

Water residence: The residence time of water is calculated based on the volume of water and the rates of export from or addition to the system. Systems or habitat units with longer residence times are more vulnerable to water quality issues.