DIEL HABITAT USE BY JUVENILE PACIFIC SALMON ALONG AN URBAN SHORELINE

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A thesis

submitted in partial fulfillment of the

requirements for the degree of

Master of Science

University of Washington

2021

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Program Authorized to Offer Degree:

Aquatic and Fishery Sciences

University of Washington

Abstract

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A mobile dual-frequency identification sonar (DIDSON) was used as an alternate technology to characterize diel juvenile Pacific salmon (*Oncorhynchus spp*) distributions along an armored and eco-engineered marine shoreline. Eco-engineering modifications to seawall habitat were intended to create an intertidal corridor for shore-oriented salmon and included the addition of marine mattresses (mesh bags filled with rocks) to create intertidal benches, a textured seawall to increase invertebrate colonization, and embedded glass blocks in an overhanging sidewalk to increase ambient light. Acoustic surveys conducted April to August 2019 along the Seattle, Washington, waterfront were used to compare acoustic and snorkel sampling methods and to quantify diel salmon habitat preferences. Acoustic salmon counts were at least twice as high as snorkel counts in most habitats with the difference attributed to the ability of the DIDSON to sample a greater range and volume than visual surveys. Snorkel salmon detections (i.e., salmon counts $n \geq 1$) were higher than acoustic detections at new and reference sites, apart from under pier microhabitats, with the difference attributed to the ability of snorkelers to identify lone fish or individuals within mixed groups of fish.

Results from generalized linear models and generalized linear mixed models demonstrate that the eco-engineered shoreline creates preferred diel habitat for salmon, while traditional under-pier habitats are less preferred for salmon regardless of time of day. Increased ambient light, fewer shading structures (e.g., pier pilings), and shallow water improve nearshore habitats for juvenile salmon. Pier end salmon counts exceeded under-pier counts and were similar to between pier counts, indicating that some salmon prefer to navigate around pier ends rather than under them.

Overall salmon presence was two times higher at night compared to during the day. Salmon presence in the intertidal corridor was more similar to reference sites at night than during the day. Highest salmon counts occurred in transects exposed to artificial light, indicating potential influence of light along the Seattle waterfront. The results of night sampling highlight the need for day/night fish sampling to reduce bias in habitat use and abundance estimates, and to identify diel differences in anthropogenic effects on nearshore communities. This study establishes that the DIDSON is a viable nearshore mobile sampling tool for small fishes (25 mm – 90 mm), and increases sampling efficiency, spatiotemporal knowledge of diel fish distributions, and habitat use compared to snorkel surveys.

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ACKNOWLEDGEMENTS

My foremost thanks go to my faculty advisor, John Horne, for mentoring me throughout my time at SAFS, and for spending time and energy on my behalf. I am grateful for your commitment to the success of my research project and for teaching me to be a better scientist and scientific writer.

Many thanks to Jeff Cordell and Jason Toft, for giving me the opportunity to join the WET lab and developing a research proposal to complement the lab’s work along the Seattle waterfront. Thank you for encouraging and guiding me throughout the graduate school and research process and thank you to Jeff, Jason, and Si Simenstad for imparting your wisdom regarding all things ‘juvenile salmon’. Thank you to Sarah Converse, for teaching the modeling class that opened my eyes to statistical modeling and guiding me in the proper direction in my analyses.

I would like to thank Andrew Hout for accompanying and aiding me during late nights on the water; the team at Bell Harbor Marina, for their kind assistance throughout the sampling season; and Jon Wittouck for fabricating the DIDSON mount. The assistance of Bill Hanot and Sergio Da Costa of Sound Metrics, Inc., and their patient instructions regarding DIDSON operations, is much appreciated.

I am grateful for the moral support of the members of the Fisheries Acoustic Research Lab and Wetland Ecosystem team, especially Bob Oxborrow, and the WET, for aiding me in the field as I began my project. Special thanks to Silvana Gonzales, for your willingness to answer any questions I sent your way, and Dave McGowan, for your helpful advice throughout the graduate process.

I would like to thank my precious family, who patiently supported me in this endeavor while taking on the burden of additional responsibilities.

This research was funded by Washington Sea Grant, the Seattle Department of Transportation, Aleutian Spray Fisheries and members of the Swasand family, and the H. Mason Keeler Endowment for Excellence. I am grateful for the financial investment in this research by these entities.

Chapter 1. INTRODUCTION

1.1 GENERAL INTRODUCTION

Estuaries provide crucial nursery functions for juvenile salmon, including refuge, transitional waters for saline acclimatization (Healey 1982, Simenstad et al. 1982, Thorpe 1994), and opportunities for feeding and growth (Simenstad et al. 1982). The highest proportional growth for salmon occurs in the juvenile life stage (Quinn 2018) and growth during this stage is positively linked to ocean survival and spawning (Reimers 1971, Magnusson and Hilborn 2003, Beauchamp and Duffy 2011). Adult survival in some salmonids (e.g., Chinook; *Oncorhynchus tshawytscha*) is linked to how much of the estuarine habitat has been unmodified (Magnusson and Hilborn 2003). Daily growth rates for juvenile Pacific salmon have been cited to range from 6% to 8% in Puget Sound (Simenstad et al. 1982), and high salmon densities and intensive foraging can cause nearshore habitats to become prey limited (Reimers 1971, Sibert 1979, Wissmar and Simenstad 1988). Turbid estuarine waters are used as refuge by salmon (Simenstad et al. 1982, Thorpe 1994) although piscivorous fish, marine mammals (Yurk and Trites 2000), and birds (Zamon et al. 2014, Quinn 2018) forage in turbid waters. Juvenile salmon residency within estuaries is variable by species, and within species is variable by geographic location, but modifications to estuaries can alter food, refuge, and habitat connectivity for salmonids (Quinn 2018; Simenstad et al. 1982, Toft et al. 2018).

Overwater structures and armoring (i.e., physical structures) associated with coastal urbanization (e.g., seawalls and piers) protect shorelines and urban infrastructure, yet impact nearshore marine environments. Shading from piers reduces or eliminates plant and algal growth, (Glasby 1999, Blockley 2007, Thom et al. 2008) and most associated invertebrates (Struck et al.

2004). Higher trophic levels are affected, resulting in adult and juvenile fish densities that are significantly lower under piers (Duffy-Anderson and Able 1999; Grothues et al. 2016). Seawalls impact nearshore marine environments by reducing habitat heterogeneity, prey availability, and by creating a disproportionate deep-water environment. Surface roughness and crevices are important for invertebrate colonization and refuge (Moreira et al. 2007), as is a shallow-grade slope for invertebrate production and organic material accumulation (Dethier et al. 2016, Heerhartz et al. 2016). The vertical, smooth surfaces of seawalls have less space and texture than natural shoreline, and organisms in these environments may be more susceptible to predation and competition (Klein et al. 2011). Seawalls reduce fish assemblage diversity and richness and can cause displacement of smaller fish to deeper waters, increasing their coexistence with larger predators (Munsch et al. 2015, 2017).

Ecological connectivity among estuarine habitats is crucial for anadromous species like salmon (Armstrong and Schindler 2013, Flitcroft et al. 2019), but urban structures and armoring fragment these environments. Juvenile Pacific salmon, as visual navigators and foragers are less abundant under piers than between piers during the day (Toft et al. 2007, Munsch et al. 2015, Sawyer et al. 2020) and school together in pier corners, presumably waiting for low tide to pass under piers when more ambient light penetrates (Munsch et al. 2014). Shading from piers can fragment nearshore habitats for juvenile salmon by reducing light availability for visual navigation and feeding (Munsch et al, 2015). Salmon prey availability and composition is changed by piers and seawalls, with prominent salmon prey, such as gammarid amphipods and harpacticoid copepods, being reduced or eliminated in pier habitats (Cordell et al. 2017). As a result, fewer feeding events are observed in these locations (Munsch et al. 2015). Along seawalls, salmon have fewer opportunities to feed on epibenthic invertebrates (Morley et al.

2012) and may switch to plankton as a food source (Munsch et al. 2015). Armoring also disconnects salmon from terrestrial invertebrates (Toft et al. 2007), which are preferred prey for juvenile Chinook.

Elliott Bay, located within the glacial estuary of Puget Sound in Washington, USA, is an example of urban salmonid habitat that has been altered in the past century. The bay is 90% armored and while historically the area contained mixed gravel-cobble beaches, few sloping intertidal beaches currently remain. Within Elliott Bay, Seattle's urbanized waterfront has over three kilometers of continuous seawall and numerous piers, with some extending over 160 m into the bay. The bay is a migration pathway for Pacific salmon, which are culturally, economically, and ecologically important in the Pacific Northwest (Willson and Halupka 2015, Feist et al. 2018). Three Pacific salmon species, including Chinook, pink (*O. gorbuscha*), and chum (*O. keta*) salmon occupy Seattle's waterfront during their spring/summer seaward migration (Toft et al. 2007). Their proclivity for nearshore waters (Simenstad et al. 1982) closely associates them with coastline urbanization. Along with juvenile Pacific salmon, fish species commonly found along the Seattle waterfront include shiner perch (*Cymatogaster aggregate*), herring (*Clupea pallasii*), and tubesnout (*Aulorhynchus flavidus*). Salmon are the dominant species in April and May, with a shift to perch and herring by the end of the summer.

On February 8, 2001, the Nisqually earthquake caused a 100-foot section of street along Seattle's waterfront to settle. The original, 71-year-old seawall was found to be failing due to a combination of aging and wood-boring invertebrate damage to the wood framing (Ott 2014). City planners formed a team to explore seawall designs that could enhance the nearshore environment for juvenile salmon. A pilot study by Cordell et al. (2017) demonstrated that adding texture to the seawall increased invertebrate biodiversity. The study also demonstrated that light-penetrating surfaces increased light levels below the water surface, with the potential to improve juvenile salmon access under piers (Cordell et al. 2017).

When armoring is needed to protect urban infrastructure from erosion and wave action, modifications to habitats that integrate human and natural ecosystems (i.e., ecological engineering, or eco-engineering) can mitigate some detrimental effects (Lichatowich and Williams 1995). Enhancement examples include engineered beaches (Toft et al. 2013), added slopes, texture, physical complexity (Firth 2014, Coombes et al. 2015), and structures to increase invertebrate richness (Cordell et al. 2017) and biodiversity (Browne and Chapman 2011, Chapman and Underwood 2011). To illustrate using examples, modifications to a seawall in Australia increased algae and animal species diversity by 110% (Browne and Chapman 2011). In Seattle, a nearshore restoration project that included a pocket beach and habitat bench for juvenile salmon increased fish densities in some years and increased invertebrate taxa richness (Toft et al. 2013). To mitigate shading from piers, light can be artificially or passively incorporated in under-pier habitats, although passive light integration (e.g., metal grating or glass blocks) is more economical and feasible long-term compared to artificial lighting (Ono and Simenstad 2014, Cordell et al. 2017).

In 2017, Phase I of Seattle’s new seawall was completed, stretching from Colman Dock in the south end to the Seattle Aquarium in the north. The new seawall incorporated three modifications that were designed to mitigate adverse armoring effects for juvenile salmon (Cordell et al. 2017). These included intertidal benches (i.e., mesh bags filled with rocks and placed adjacent to the seawall), a textured seawall (i.e., with added bumps, crevices, and extended shelves), and embedded glass blocks in the overhanging sidewalk (Figure 1.1). The intertidal benches increase seafloor elevation, which creates a shallow, continuous corridor for salmon to swim in and substrate for algae and invertebrates to colonize. The textured seawall and extended shelves are intended to encourage primary productivity of and colonization by salmon prey invertebrates, and to provide additional refuge for salmon. The glass blocks in the overhanging sidewalk are designed to increase daytime ambient light to aid salmon navigation and foraging. The combination of these modifications creates a shallow, continuous corridor with algae and invertebrates as food sources.

The extent to which overhead structures influence juvenile salmon distributions is unknown. Juvenile salmon may avoid overwater structures, including the overhead glass-block sidewalk, even with ambient light present. Studying juvenile salmon densities under piers with differing piling densities and light availabilities provides an opportunity to better understand the effect of structure and corresponding ambient light on juvenile salmon.

Figure 1.1. Photographs of Seattle's seawall, with enhancements, before and after reconstruction. **Left**: Original seawall. **Right**: Re-constructed seawall. Enhancements for juvenile salmon include marine mattresses to increase seafloor elevation, textured seawall and shelves for invertebrate colonization and fish refuge, and overhanging sidewalk with glass blocks for increased ambient light penetration.

Since 2003, the University of Washington Wetland Ecosystem Team (UW-WET) has used snorkel surveys to characterize salmon and fish assemblages along the Seattle waterfront and nearby reference sites (Toft et al. 2007; Munsch et al. 2014; Sawyer et al. 2020). Snorkeling as a survey method enables fish species identification, access to difficult-to-reach locations, and observation of fish behavior (e.g., feeding) in real time. The UW-WET has found that juvenile salmon are reticent to swim under piers and may wait for low tide to transit (Toft et al. 2007; Munsch et al. 2014), reducing shade under piers can improve ambient conditions for prey and increase prey densities (Cordell et al. 2017), shoreline enhancements can improve habitat for juvenile salmon and prey (Toft et al. 2013), and eco-engineered seawall modifications improve salmon habitat (Sawyer et al. 2020). However, optic sampling is impacted by ambient light intensity and water turbidity (Martignac et al. 2014).

An alternative to optic surveys is active acoustics, which are unaffected by ambient light and mostly unaffected by turbidity (Maxwell and Gove 2007; Foote 2009). The volume sampled by an acoustic camera increases efficiency and spatiotemporal coverage in narrow rivers and estuaries relative to visual surveys (Martignac et al. 2014). Sound wavelengths in water are approximately 2,000 times longer than visible light, so transmitted acoustic energy propagates much further than light energy (Simmonds and MacLennan 2005). One acoustic sampling technology is the dual-frequency identification sonar (DIDSON; Sound Metrics, Bellevue, WA, USA; www.soundmetrics.com). The DIDSON is a multibeam sonar that transmits sound pulses, with returning echoes being digitized and converted into near-video-quality images. Comparable to visual surveys, the high operating frequency, and high data resolution of the DIDSON enables observers to identify species and to ‘see’ behaviors in real time (Helfman 1999). Relative to snorkel or SCUBA surveys, DIDSON sampling can include long transects and detection of near-boundary fish, which may be difficult to spot visually (Holmes et al. 2006). A DIDSON multibeam sonar (also known as an acoustic camera) has been used to monitor fish under piers (Able et al. 2014, Grothues et al. 2016), fish near a substrate or boundary (Holmes et al. 2006), predator-prey interactions (Price et al. 2013), fish below ice (Mueller et al. 2006), and juvenile fish (Doehring et al. 2011).

Acoustic cameras are not typically used to survey juvenile fish from a mobile platform. The predominant (~95%) use in fisheries applications is a fixed deployment to enumerate migrating adult salmon in riverine environments (Bill Hanot Sound Metrics, Inc.; personal communication). Acoustic cameras have been used to survey small fish (Adams et al. 2015; Kock et al. 2016) and to identify fish assemblages along nearshore environments using a mobile sampling platform (i.e., a kayak, Able et al., 2014; Grothues et al., 2016). An advantage in acoustic data processing, compared to snorkel observations, is that raw data are recorded, and files can be reviewed in a laboratory setting at an increased or decreased frame rate to maximize efficiency of data processing. Different ranges can be viewed during playback, which permits comparison of what is happening at two ranges at the same time (e.g., juvenile salmon densities adjacent to and under a pier).

Active acoustics also enables nighttime sampling. Fish assemblages may differ at night, and diurnal-only sampling can bias abundance and habitat use estimates if inferences extend to nighttime (Rountree and Able 1993, Maes et al. 1999, Hagan and Able 2008). Sampling at night can also be used to gauge if the relative impacts of shoreline armoring differ between night and day. Knowledge of habitat influences on nocturnal fish distributions in Elliott Bay is limited because traditional sampling effort has been restricted to daytime snorkeling and net sampling (e.g., Toft et al. 2007, Beauchamp and Duffy 2011). Most nighttime research on juvenile salmon has focused on behaviors before they enter the estuary, such as their increased migration into saltwater at night (Hoar 1951, Hartman et al. 1982). In nearshore marine environments, juvenile salmon exhibit varied nocturnal dispersal patterns, swimming offshore and dispersing at some locations (Bax and Whitmus 1980; Schreiner 1977, Simenstad et al. 1982) while exhibiting no diel change in others (Cardwell et al. 1978). Although their most active feeding occurs during the day (Duffy et al. 2010) and shoreline habitat type has not been shown to influence diel diets (Munsch et al. 2015), juvenile salmon forage opportunistically on moonlit nights (Bailey et al. 1975), and they consume larger and less common prey compared to their daytime diets (Gosho 1976; Bailey et al. 1975).

The dominant factor potentially influencing diel activity in urban, nearshore aquatic fauna is ambient light. Anthropogenic-sourced artificial light at night (ALAN) creates ecological light pollution, defined as "artificial light that alters the natural patterns of light and dark in ecosystems" (Longcore and Rich 2004). At least 22% of the world's coastlines (excluding Antarctica) are exposed to light pollution (Small and Nicholls 2003). In coastal urban areas, artificial lighting from streetlights, pier lights, and buildings is prevalent along waterfronts. For most animals, vision is crucial for orientation, and those that typically navigate in a dark environment (e.g., hatchling sea turtles) can become disoriented by artificial light (Longcore and Rich 2004). ALAN can increase fish predation (Bolton et al. 2017), change fish circadian and circatidal rhythms (Pulgar et al. 2019), alter nighttime fish communities (Bolton et al. 2017), and increase small fish foraging opportunities on moonlit nights due to the attraction of their planktonic prey to artificial light (Gliwicz 1986).

Juvenile salmon may be more affected by nearshore artificial light than benthic or pelagic species due to their proclivity for shallow waters and shallower position in the water column.

Ocular adjustment can take as long as fifty minutes at twilight (Brett and Ali 1958, Fields 1966), which may explain their preference for navigating along the edges of dark spaces (Simenstad et al. 1999) and their avoidance of extreme light gradients (Salo 1979, Ono and Simenstad 2014). ALAN increases juvenile salmon predation, a main source of mortality (Yurk and Trites 2000; Brosnan et al. 2014, Beauchamp et al. 2020) and acts as an attractant that can delay juvenile salmon migration (Salo 1979, Tabor et al. 2004). For example, subyearling salmonids in two Washington lakes were up to 10 times more abundant in brightly lit areas compared to sections with reduced or no light (Tabor et al. 2017).

1.2 OBJECTIVES

The goals of this study were to test the ability of a DIDSON to quantify juvenile salmon from a mobile platform and to characterize day/night juvenile salmon distributions along an urban shoreline. Specific objectives of this study include: 1) comparing snorkel and acoustic sampling methods to detect and enumerate juvenile fish along an urbanized shoreline, 2) characterizing and comparing day and night juvenile salmon distributions among urban habitats, 3) evaluating the efficacy of the eco-engineered seawall to mitigate impacts of coastal urbanization on juvenile salmon distributions, and 4) quantifying the relative importance of environmental and physical factors influencing day and night salmonid habitat use along the Seattle waterfront.

Chapter 2. DIURNAL ACOUSTIC CHARACTERIZATION OF JUVENILE PACIFIC SALMON ALONG AN ECO-ENGINEERED SEAWALL

2.1 INTRODUCTION

Overwater structures (e.g., piers) and armoring (e.g., seawalls) associated with coastal urbanization are prevalent along urban shorelines. They impact resident and transient fish who inhabit nearshore estuarine habitats during a life stage requiring acclimatization to saline waters, growth, and refuge from predation (Chapter 1, section 1.1). Piers and seawalls impact the nearshore substrate (Dethier et al. 2016), invertebrate production (Heerhartz et al. 2014), and nearshore fish communities (Toft et al. 2007). Armoring and piers fragment nearshore habitats for juvenile salmon, leading to lower salmon abundances under piers (Toft et al. 2007, Ono and Simenstad 2014). Alternatives to traditional shoreline armoring can mitigate deleterious armoring effects and restore some lost habitat function (Lichatowich et al. 1995). One example is Seattle's replacement seawall, which includes three modifications to improve nearshore habitats for juvenile salmon (for details see Chapter 1, section 1.1).

Juvenile salmon densities and habitat use have been quantified by daytime snorkel surveys as part of a long-term monitoring project along the Seattle waterfront (e.g., Toft et al. 2007). However, direct optic sampling is impacted by light availability, water turbidity, and time spent in the water. Active acoustics (e.g., DIDSON multibeam sonar) can extend the spatiotemporal coverage of snorkel surveys to include dark, turbid, and nocturnal environments (Maxwell and Gove 2007, Foote 2009). Within Elliot Bay, limited fish diversity and knowledge of salmon spawn and hatchery release timing aid in species identification, and fish size, morphology, and swimming behaviors can be used to differentiate species (Martignac et al.

2014). A limited use of DIDSON has included distribution studies of juvenile fish (Doehring et al. 2011, Kimball et al. 2010, Adams et al. 2015). However, the accuracy and precision of a mobile DIDSON survey to detect fish in this length range is unknown. Previous studies have focused on larger fish (> 250 mm), with a few exceptions (e.g., Kimball et al. 2010; Able et al. 2013). The precision of DIDSON counts has been shown to be equal to that of visual counts for adult salmon (Holmes et al. 2006; Maxwell and Gove 2007; Petreman et al. 2014) and is assumed to be consistent for the juvenile fish in this study.

This study included testing the ability of the DIDSON multibeam sonar to quantify juvenile salmon from a mobile platform, and characterizing juvenile salmon use of traditional and eco-engineered seawall habitat along the Seattle waterfront. Objectives of this study included: 1) comparing snorkel and acoustic sampling methods to detect and enumerate juvenile fish populations in an urbanized coastal environment, 2) quantifying the relative importance of factors influencing habitat use by juvenile salmon in eco-engineered environments, and 3) evaluating the efficacy of the intertidal corridor to mitigate impacts of coastal urbanization on juvenile salmon habitat use.

2.2 METHODS

2.2.1 *Study Location*

This study took place in Elliott Bay, a bay within Puget Sound, Washington, USA (47.6062° N, 122.3321° W). Puget Sound is a glacial estuary composed of sediment beaches, embayments, deltas, and rocky coastline (Finlayson 2006; Shipman 2008). Eight sampling sites along Seattle’s central waterfront were monitored throughout the juvenile salmon migration, April to August 2019 (Figure 2.1). Three sites (“new”) contained eco-engineered seawall habitat. Three sites (“old”) contained original seawall habitat. Each new and old site included a pier and

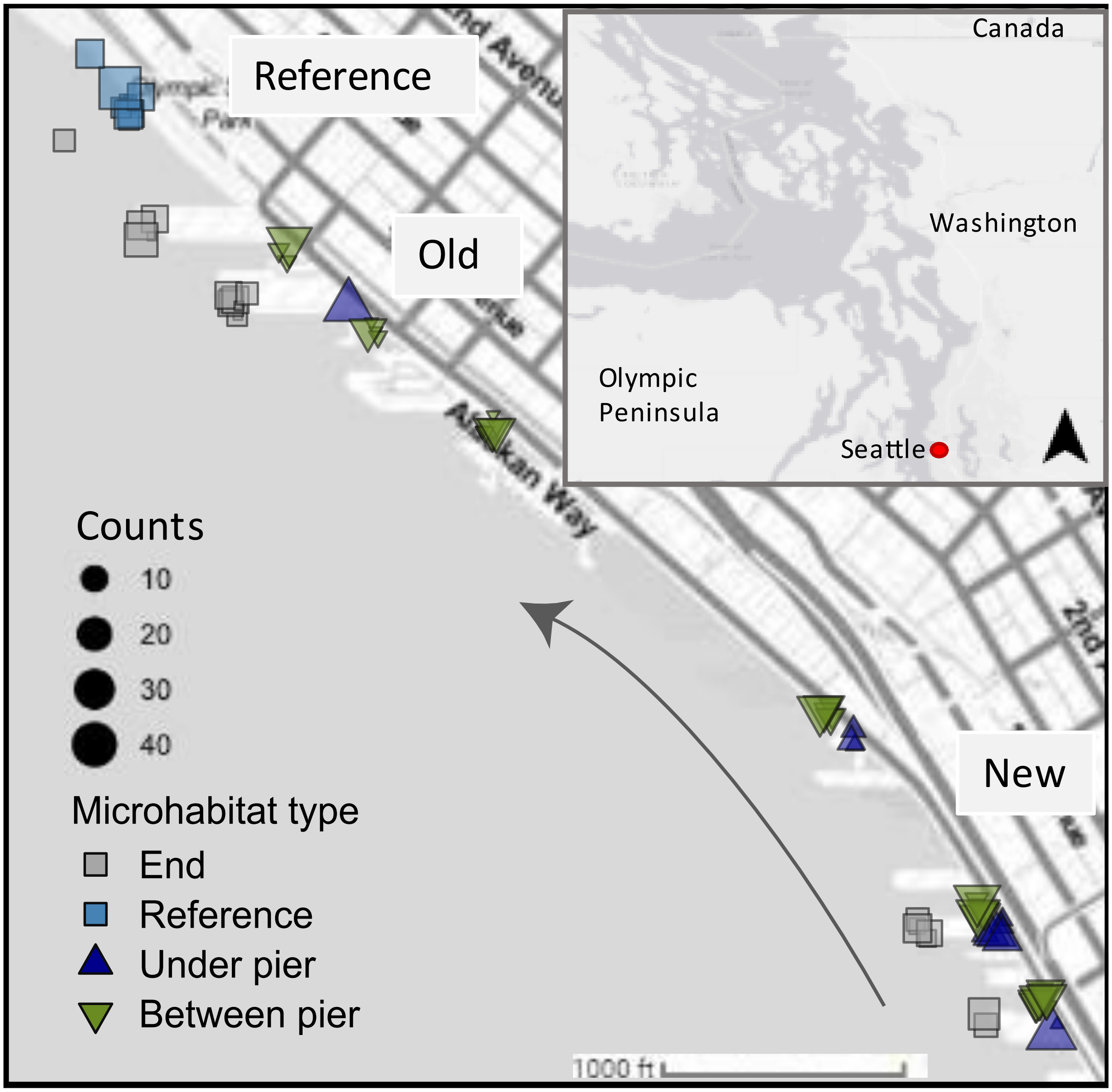

Figure 2.1. Size-proportional diurnal salmon counts along the Seattle waterfront. Shapes are size proportional to the square root of salmon counts and are shape-sorted by microhabitat. Counts are minimally jittered to reduce overlap and are not at their exact coordinates. **Inset**: Study site location in the Pacific Northwest, USA. **Main**: Study site location in Seattle, WA, USA. Transects are at reference, old, and new sites. Arrow indicates the primary migration path of juvenile salmon along the Seattle waterfront.

an open seawall section adjacent to a pier, while two of the three new sites and two of the three old sites included a pier end transect. Two reference sites ("reference") were chosen to represent habitat along the waterfront that had been previously restored and has been colonized by aquatic biota for over a decade. The first reference site was along a semi-enclosed created pocket beach. The second was adjacent to an open, armored shoreline where an intertidal habitat bench comprised of riprap had been created (Toft et al. 2013) and included a transect that was ~160 m from shore and ran parallel to the nearest pier end.

2.2.2 *Sample Design*

Nearshore seawall sites contained 25-m strip transects that were labelled by overhead structure (between pier – "BP" and under pier - "UP") and site type ("old" and "new"). As a result, the four nearshore transects at seawall sites included old BP, new BP, old UP, and new UP transects. The reference locations included 35-m (to sample the entire length of) transects along the shoreline of a semi-enclosed pocket beach, 25-m transects along a habitat bench, and one 25-m transect in the deep water transect (reference 'end'). All nearshore microhabitats were further differentiated by 'depth', defined by distance to the seawall or shoreline (3 m = "shallow" and 10 m = "deep") (e.g., new, shallow BP and new, deep BP). As a result, the eco-engineered intertidal corridor was contained within new, shallow UP and BP transects. Four of the six seawall sites (two new, two old) contained two 25-m strip transects at each of their pier ends. One transect was sampled along the outer edge of the pier end and the other transect was sampled along the inner edge of the pier end. The combination of site type and structure (BP, UP, pier end, and open transects at reference sites) configurations resulted in ten 'microhabitats' within new, old, and reference sites (Table 2.1).

Table 2.1. Microhabitats defined by site type and structure type, with transect lengths listed. Site type includes old, new, and reference sites, and overhead structure type includes between pier, BP; under pier, UP; or reference transects, Ref; and by pier end site type (old, new) and edge type (outer edge, inner edge). Nearshore microhabitats are further delineated by depth (shallow, 3 m from shoreline or seawall; or deep, 10 m from shoreline or seawall).

Site type	Definition		
New	Seawall constructed in 2017 with textured wall and shelves, raised seafloor by filled mesh bags, and overhanging glass-block sidewalk		
Old	Seawall constructed in 1936 with no texture or raised seafloor; truncated shallow-water zone, no glass blocks in overhead sidewalk		
Reference	No in-water structure, sloped gradient; cobble or riprap sediments		
Structure Type	Definition		
Between Pier “BP”	Adjacent to seawall with no overhead pier		
Under Pier “UP”	Adjacent to seawall with a pier overhead		
Reference	Open water with no nearby structures (reference sites only)		
Pier end-outer edge	Outer edge of pier end (90 m -150 m from seawall, dependent on pier) with no overhead structure		
Pier end-inner edge	Inside edge of outermost piling row, under-pier decking		
Microhabitats	New	Old	Reference
Between pier (BP) 25-m	New BP	Old BP	
Under pier (UP) 25-m	New UP	Old UP	
End – outer edge 25-m	New pier end – outer	Old pier end – outer	
End – inner edge 25-m	New pier end – inner	Old pier end – inner	
Pocket beach 35-m			Ref Open
Habitat bench 25-m			Ref Open
End - reference 25-m			Ref End

2.2.3 *Data Acquisition*

In 2019, snorkel and acoustic data were collected at matching transects to facilitate comparison of survey methods. Daytime snorkel sampling (cf. Toft et al. 2007) has been conducted for the Seattle Department of Transportation since 2007 (see also Munsch et al. 2015; Sawyer et al. 2020). Acoustic sampling was added to these sites during the juvenile salmon migration in 2019. Each transect was sampled during high and low tides. Acoustic sampling occurred predominantly within one week of snorkel surveys. It was assumed that the same pulse of migrating juvenile salmonids was sampled by snorkel and acoustic methods based on natural Chinook salmon residence time in Elliott Bay pier habitats averaging 18.4 days ± 2.3 days in June and July, with hatchery salmon residing twice as long (Ruggerone and Volk 2014). Two snorkelers swam parallel at shallow (i.e., 3 m from seawall) and deep (i.e., 10 m from seawall) transects. Before snorkel sampling, underwater visibility in the sampling locations was measured with a Secchi disc. A minimum of 2.5 m visibility was required to conduct snorkel surveys. Each snorkeler carried a GoPro camera to simultaneously record fish encounters along transect surveys which were later used to supplement fish species identification.

A DIDSON multibeam sonar was used to quantify fish presence and counts along snorkel survey transects (new and reference sites) and old seawall sites. The DIDSON was mounted beneath the center of a 3 m kayak (Figure 2.2) allowing access to shallow waters and navigation among pier pilings (cf. Able et al. 2014). The horizontal swath of the DIDSON acoustic beam was 2.5 m to each side of the kayak and equivalent to the snorkeler’s field of view in non-turbid conditions. The operating frequency was set at 1.8 MHz with a sampling rate of 8-12 Hz. The sampling range was set from 2.5 to 7.5 m, with a start range of 1.5 m from the transceiver to account for the distance between the sonar head and the kayak bow, and an end range of 7.5 m to maximize acquisition rate and image clarity, and to account for the distance before any fish flight response was observed. With a 14° horizontal and 29° vertical field of view, the DIDSON was tilted 12° to ensure that the middle to upper portion of the water column that juvenile salmon occupy was ensonified. The DIDSON transceiver was connected to a waterproof laptop running data acquisition software (DIDSON V5-26). A timestamped file was created for each transect.

A HOBO pendant Temperature/Light data logger (UA-002-08 Onset, Cape Cod, MA, USA), mounted to a PVC pipe and deployed off the front of the kayak below the water surface, measured luminous flux (lux) and temperature (°C) every ten seconds. A GoPro Hero 7 camera (San Mateo, CA, USA) mounted next to the HOBO data logger and facing forward provided a simultaneous optic record to aid in fish species identification when water clarity permitted.

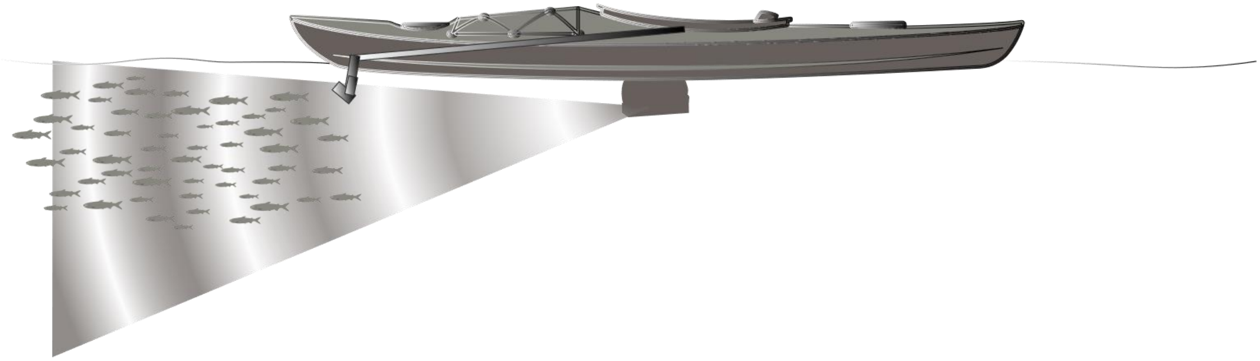

Figure 2.2. Schematic of a DIDSON camera mounted to a kayak hull, ensonifying the water at a -12º angle, 2.5 m from the center of the kayak to 7.5 m from kayak center. A GoPro camera and HOBO light pendant are mounted near the front of kayak to record optical video and light measurements.

2.2.4 *Data Processing*

Snorkel and acoustic surveys recorded fish presence, counts, and swimming behaviors. During snorkel surveys, fish lengths and feeding behaviors were recorded *in situ* on waterproof paper. To process acoustic data, DIDSON data files were played three times to: count and record fish, verify counts, and review fish identification and counts using coincident GoPro video footage (Figure 2.3). Fish that were within the range of the acoustic camera and swam into a subsequent transect from the previous transect were counted only in the first transect they were encountered. To minimize double-counting, fish that swam out of range of the camera (e.g., ahead or to the side) and could potentially be counted in a subsequent transect were subtracted from those counts. For example, if 50 salmon were detected in a shallow BP transect and swam to the side into deep water, 50 salmon were subtracted from the deep BP transect count total. Fish identifications were made to lowest possible taxonomic group. If species identification was not possible using body length, shape, swimming behavior, and/or GoPro video footage, fish were labelled “unknown” or designated as a non-labelled group (e.g., species 1). Mixed species aggregations that could not be identified to taxon with the aid of GoPro images were labelled as unknown or mixed aggregations and resulted in salmon counts set to zero for these occurrences. Relatively low species diversity coupled with staggered migration times, different body morphologies, and behaviors (e.g., schooling, swimming) facilitated identification of individual and aggregations of observed fish. The smallest salmon detected by the acoustic camera were estimated by snorkelers to be 25 mm – 50 mm. Over the sampling season, juvenile salmon mean fork length was estimated by snorkelers to range from 65 mm – 90 mm.

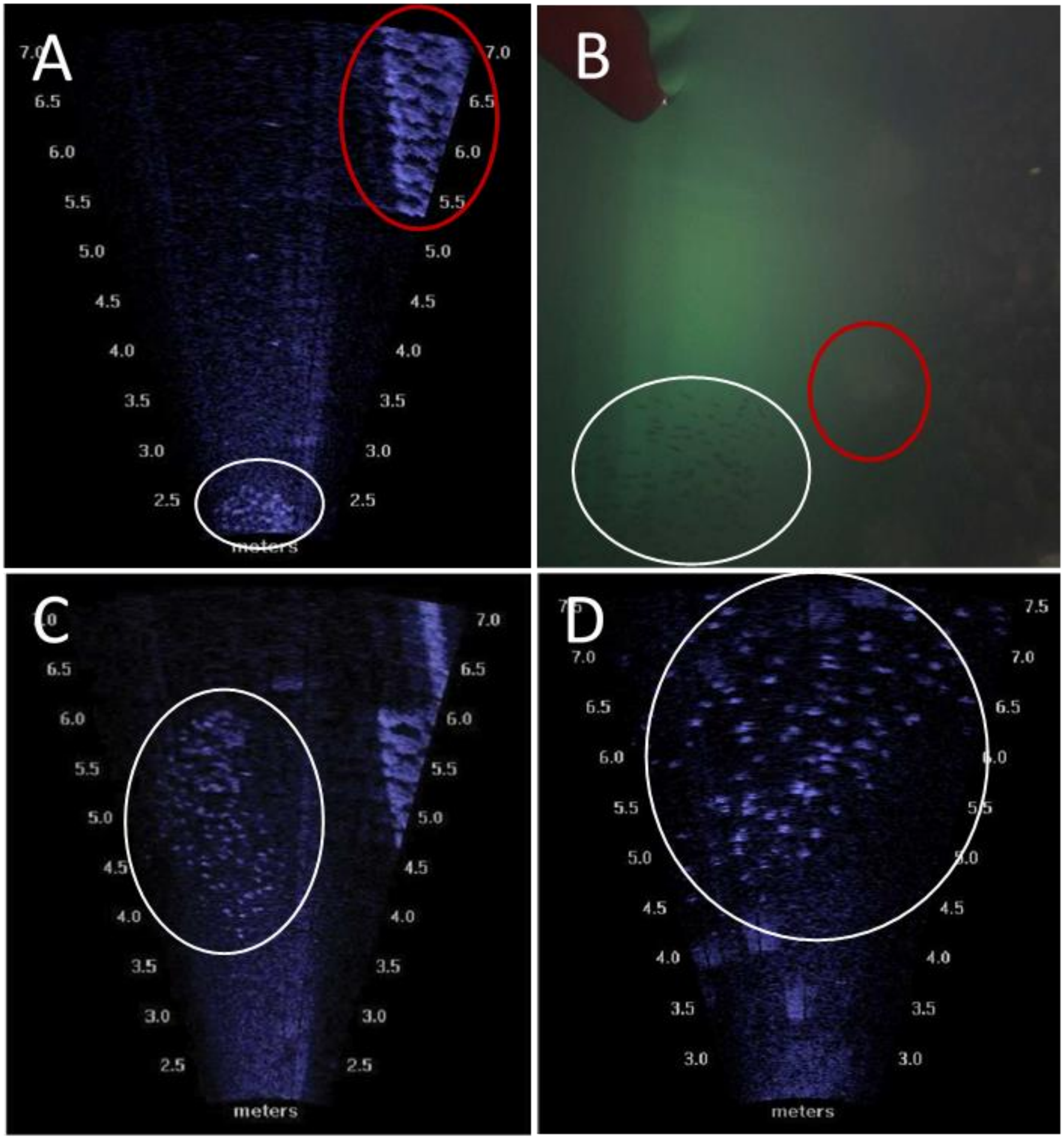

Figure 2.3. Simultaneous acoustic images and GoPro photographs of juvenile chum salmon with estimated 40 mm fork lengths, shoaling adjacent to the new seawall. White circles indicate salmon and red circles indicate a seawall shelf. C: Juvenile salmon shoaling in the intertidal corridor adjacent to a seawall shelf. D: Shiner perch group along the outside edge of a pier end.

2.3 DATA ANALYSIS

2.3.1 *Comparisons of Snorkel/Acoustic Fish Counts and Light Availability*

Detections (i.e., encounters or presence) and counts of juvenile salmon for snorkel and acoustic data were standardized to a 25 m transect length. Nonparametric Wilcoxon Mann Whitney tests (Mann and Whitney 1947; Wilcoxon 1945) were used to determine if counts differed between paired acoustic and snorkel surveys by site type and overwater structure. These tests were used to identify differences between categories rather than quantify differences among site type and overwater structure. Mann Whitney tests were used to compare light availability at the two newly created habitats that included the eco-engineered intertidal corridor (new, shallow UP and BP) and their microhabitat counterparts. Mann Whitney tests were also used to compare acoustic salmon distributions within their microhabitat by depth: shallow, new UP compared to deep, new UP.

2.3.2 *Characterizing Daytime Salmon Presence and Distributions*

To characterize salmon presence and distribution along the Seattle waterfront, generalized linear models (GLMs) and generalized linear mixed models (GLMMs) that included zero-inflated or hurdle models were considered for acoustic presence and count data (i.e., Poisson or negative binomial). Zero-inflated and hurdle models include a model for excess zeros and a model for non-zero counts. Model choice was informed by consideration of which model was more ecologically relevant to the system and to the degree in which the data met model assumptions. To avoid statistical assumption violations and to ensure appropriate use of statistical analyses, exploration of the acoustic dataset followed the protocol described by Zuur et al. (2010). Dispersion tests were used to identify the distribution of the data. Akaike Information

Criterion (AIC; Akaike 1974) and Bayesian Information Criterion (BIC; Schwarz 1978) metrics were used to compare potential count data distributions (e.g., Poisson, negative binomial) to evaluate how each distribution fit the data, and to compare potential over- or under parameterization of model fits. To test for excess zeros (i.e., more than would be expected under a Poisson or negative binomial distribution), predicted probabilities of zeros for acoustic data were summed. Due to the underfitting of zeros, zero-inflation and zero-alteration models were compared (R package 'glmmTMB' (Brooks et al. 2017 v 1.0.2.1). Using AIC and BIC values, and consideration of how the data met the assumptions of each model, the most appropriate model structure was a hurdle model using a logit link function, and a negative binomial count distribution (log link function) (Appendix A2.6). A hurdle model was an appropriate choice as it was not possible to apportion proportions of zero counts as fish absence versus lack of detection, as the hurdle model does not differentiate between sampling or real zeros (Zuur et al. 2009). To address potential spatiotemporal effects due to repeated measures at repeated sites, likelihood ratio tests were used to test random effects including 'month', 'Julian day' and 'site' against models with no random effects.

Candidate covariates in model selection included microhabitat, depth (3 m from seawall "shallow" and 10 m from seawall – "deep"), tide (< 2 m – "low") and ≥ 2 m – "high"), transect length, and *in situ* measurements of water temperature (°C) and light (lx). Light values were scaled prior to analysis. Presence and count data were scaled to a standard 25 m transect length. To avoid model overfitting, depth was included as a covariate instead of being assigned to each microhabitat category. All statistical analyses were conducted using R version 3.6.1 (R Core Team, 2019) and the R packages 'GLMMadaptive' (Rizopoulos, 2020 v 0.7-15) and 'pscl' (Zeileis et al. 2008 v 1.5.5).

Four model groups were used to characterize juvenile salmon habitat use along the Seattle shoreline: nearshore all-season (nearshore reference sites included), pier ends all-season, nearshore seawall all-season (no reference sites), and nearshore seawall peak migration (an eight-week period April 23, 2019 – June 15, 2019; 98% of total salmon counts at nearshore seawall sites occurred during this period). In each case, final models were chosen considering likelihood ratio test values, AICs, and by considering the ecological relevance of each model.

The nearshore all-season model was developed to compare juvenile salmon presence and counts among all nearshore habitats throughout the sampling season. Nearshore reference transects and 'shallow' depth transects were estimated by the intercept, and the additional effects measured the difference between the remaining microhabitats and reference transects, and the difference between 'deep' depth and shallow. Reference sites were used as a baseline because they represented the most natural habitat along the waterfront.

The pier ends model was developed to evaluate potential differences between 'old' and 'new' pier end habitats (i.e., pier ends that are connected to old or new seawall habitat), and between the outer edge of the pier end compared to the inner edge that has overhead decking. 'Outer edge' and 'old' pier ends were estimated by the intercept, such that the differences estimated were between outer and inner edge, and between old and new pier ends.

The final two models were developed to evaluate juvenile salmon differences in microhabitat utilization and schooling behaviors at seawall sites throughout the sampling season, and those during the migration peak. 'Old' seawall and 'shallow' depth transects were estimated by the intercept, and the differences estimated were between old and new seawall transects, and shallow and deep transects.

The structure for all the models is specified as:

$$f_{nb\ hurdle}(y;\pi,\mu,r)\left\{\begin{array}{c} f(binomial)(y=0;\ \pi) \\ [1-f(binomial)(y=0;\ \pi)] \end{array}\right. * \left(\frac{f_{nb}(y=0;\mu,r)}{1-f_{nb}(y=0;\ \mu,r)}\right)$$

where π = the probability of a zero (i.e., absence) for models with a random effect, or the probability of a non-zero (i.e., presence) for models with no random effect; an increase in coefficient value signifying a higher likelihood of absence or presence. The zero equation is accompanied by a negative binomial distribution to model positive counts, where μ = the mean of positive counts, and r = the scale parameter of positive counts. All covariates were incorporated in the zero portion of the models along with the count portions because understanding their effect on presence and counts was of interest. Apart from one model during peak densities that did not require a temporal random effect, final models addressed temporal autocorrelation due to repeated measures by including random intercepts that incorporated Julian day or month (Bolker et al. 2009). Individual sites were excluded as random effects due to high AIC and BIC values.

2.4 RESULTS

2.4.1 *Acoustic Salmon Counts*

Over the survey period (April 02 to August 09, 2019), 536 acoustic surveys were conducted and used in analyses. Surveys encompassed the main juvenile salmon migration period (Toft et al. 2007), and while salmon were detected less frequently at the beginning and end of the survey season, they were present throughout the sampling period. Seven percent of the acoustic transects were not used in any analysis due to an inability to complete an under-pier transect where access was blocked by high tide or obstructions, to balance samples among microhabitats, and to avoid pseudo-replication due to double sampling within a tidal cycle. A total of 8207 juvenile salmon were counted acoustically through the migration period. By site type, 2440 salmon were counted at reference sites, 2684 at old seawall sites, and 3083 at new seawall sites (Figure 2.4). By structure type, 1619 salmon were counted under piers, 1917 between piers, 2395 at nearshore reference habitats, and 2276 at pier ends. Ninety-eight percent of the salmon detected at seawall sites (i.e., not including reference and pier end transects) were counted during the eight-week peak migration period.

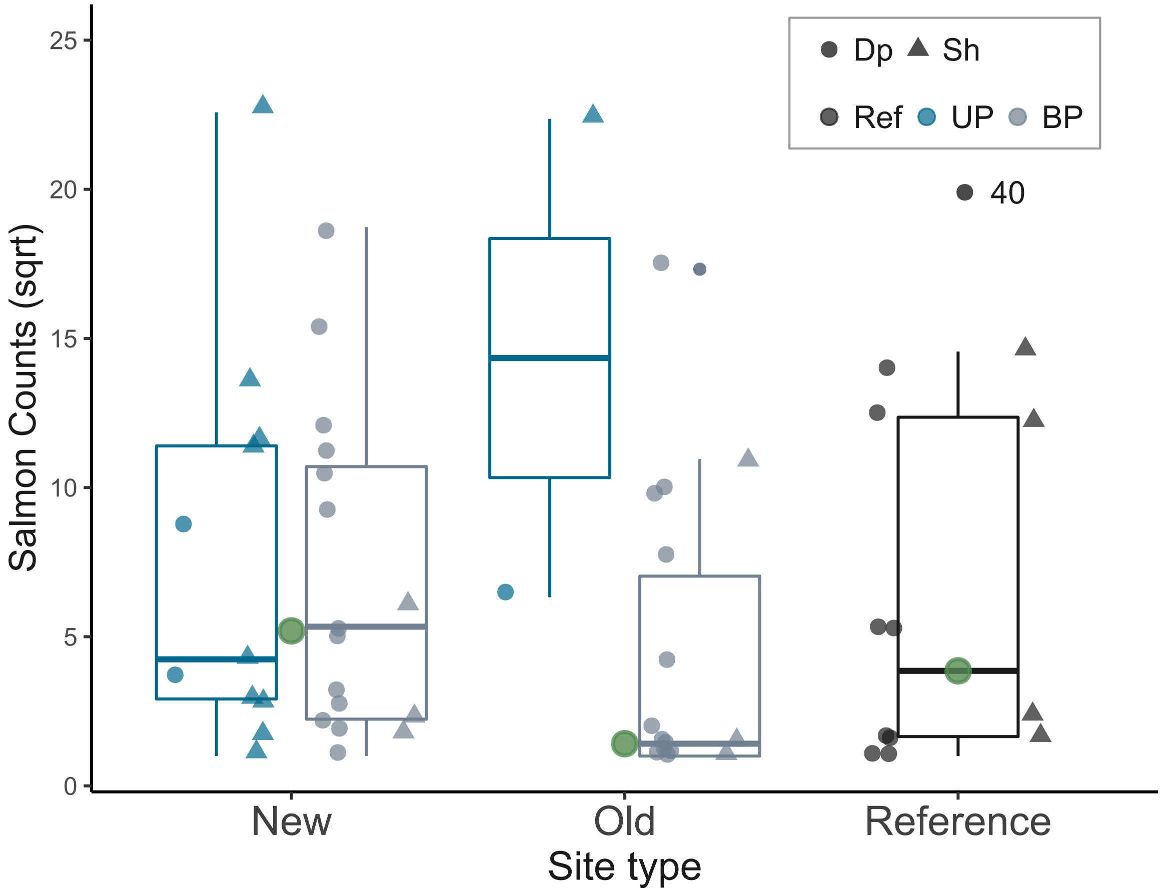

Figure 2.4. Positive salmon counts among microhabitats and within microhabitats, by depth. Boxplots depict range and medians of square root transformed juvenile salmon counts within microhabitats, by depth. Points indicate positive counts that are jittered to minimize overlap. Shapes indicate the depth at which the count occurred. An outlier count at a reference transect is labelled. Green circles indicate positive median counts by site type. Dp = Deep; Sh = Shallow; UP = under piers; BP = between piers.

2.4.2 *Acoustic and Snorkel Salmon Count Comparisons*

Within sites, acoustic counts of juvenile salmon were typically at least double that of snorkel counts. The difference is attributed to the ability of the DIDSON to sample a greater range and volume than visual surveys. Total salmon acoustic counts were 4670 compared to 1908 snorkel counts (Table 2.2). At reference sites, acoustic and snorkel counts were 2395 and 384, while at new seawall sites, acoustic and snorkel counts were 2275 and 1524. Acoustic counts were higher than snorkel counts except along shallow BP transects, possibly due to a single high salmon snorkel count in May (n = 900). Snorkel salmon detections (i.e., salmon counts $n \geq 1$) were higher than acoustic detections at new and reference sites, except for under pier microhabitats.

Table 2.2. Acoustic and snorkel presence and ranked count comparisons with count ratios. Wilcoxon Mann Whitney tests compare acoustic and snorkel positive, ranked counts by site type and within new sites by microhabitat. Count ratios and p-values are reported.

	Counts				**Presence**	
	Acoustic	Snorkel	Count Ratio	P value	Acoustic	Snorkel
Total	4670	1908	2.45	0.00003	40	70
Reference	2395	384	6.24	0.03	13	34
New sites	2275	1524	1.49	0.0006	27	36
Between Pier	1196	1463	0.82	0.008	16	29
Under Pier	1079	61	17.69	0.04	11	7

2.4.3 *Light Comparisons among Microhabitats*

Light measurements were recorded throughout the sampling season to compare light intensity among microhabitats. As expected, microhabitats with no overhead structure contained the highest median light values, while the lowest median light measurements occurred in UP locations (Table 2.3, Figure 2.5). New, shallow BP transects (i.e., the intertidal corridor between piers) had lower light values compared to old, shallow BP, likely owing to the glass-block cantilevered overhead sidewalk that blocked a portion of sunlight to the water below (see Figure 1.1; Sawyer et al., 2020). New, deep BP transects had lower light values compared to old, deep BP, likely owing to the closer spacing of piers at new sites that cast partial shadows over the open water (see Figure 2.1). Old, deep UP transects had higher median light values than other UP habitats, including the intertidal corridor, but more available light did not increase salmon presence in these transects.

Table 2.3. Median seawall light measurements and Mann Whitney light comparisons. **Left**. Median light measurements within microhabitats, by depth, collected every 10 s for each sampling event. **Right**. Mann Whitney test results comparing median light measurements between the intertidal habitats created by the new seawall (i.e., new shallow BP and UP) and their comparable habitats.

Light	Under Pier		Between Pier		New, Shallow BP p -value		
	Shallow	Deep	Shallow	Deep	Old, shallow BP	0.001	
Old	9.3	73.2	456.4	890.5	New, deep BP	0.001	
New	13.2	11.9	140.8	305.4	New, deep UP		0.77
Ref			1233.1	1040	Old, shallow UP		0.89

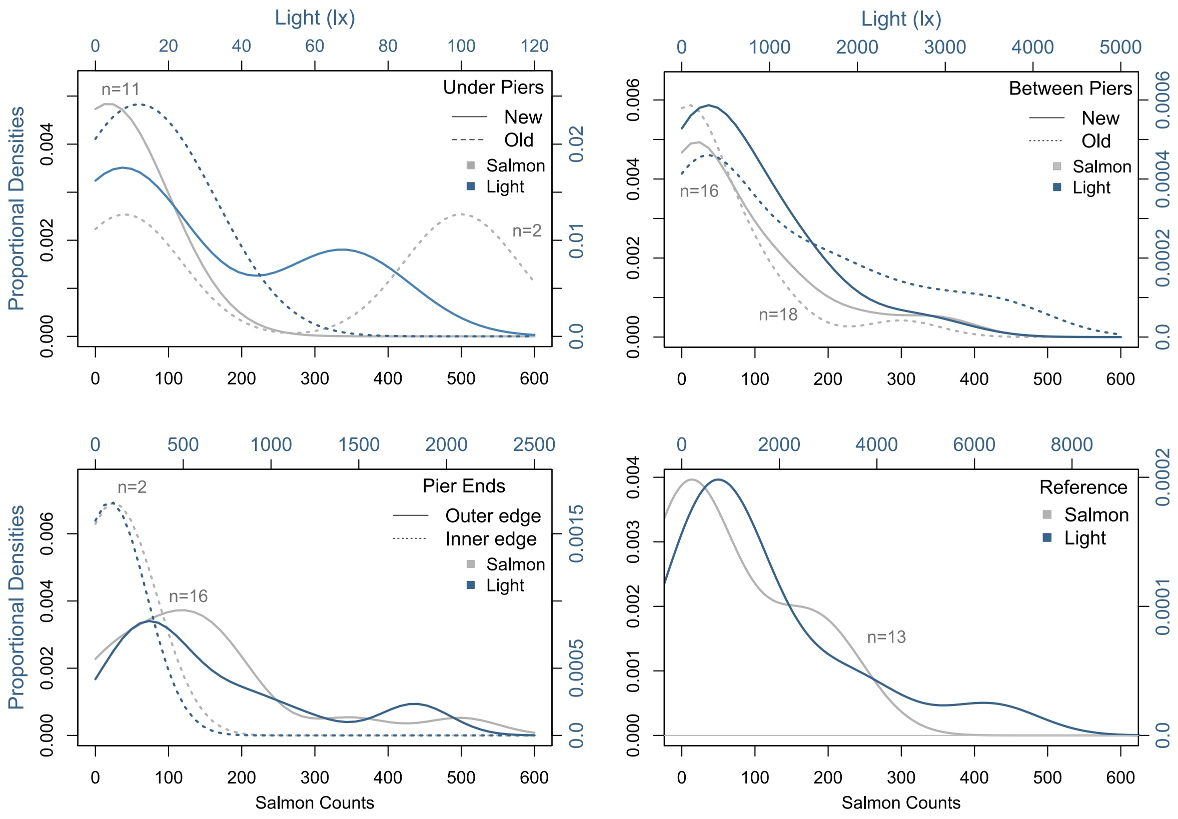

Figure 2.5. Kernel density plots of positive salmon counts and light proportional densities. Salmon count proportional densities by microhabitat are in gray, and light (lx) measurement proportional densities are in blue. Light kernel density plots are not on identical scales. The sum of salmon encounters (‘presence’) by microhabitat are labelled on the plot.

2.4.4 *Salmon Distributions by Depth*

Mann Whitney tests identified no differences in salmon counts by depth at reference locations or old UP transects (Table 2.4). At new UP transects, salmon were less evenly distributed by depth, with more salmon detected in the shallow, intertidal corridor compared to deep transects that did not have the glass block sidewalk overhead. At between pier transects, more salmon were counted in deep transects compared to those counted at shallow transects.

Table 2.4. All-season acoustic sampling presence and counts among microhabitats, by depth. Included are transects sampled, number of detections (‘presence’), counts, and median counts, by microhabitats (includes both depths). Results from Mann Whitney tests compare salmon counts within each microhabitat, by depth.

	Site	Depth	Sample	Presence	Counts	Median	P value
Between Pier	New	Shallow	46	3	43	28.5	0.002
		Deep	40	13	1153		
	Old	Shallow	45	4	124	2	0.005
		Deep	42	14	597		
Under Pier	New	Shallow	46	9	989	18	0.046
		Deep	40	2	90		
	Old	Shallow	47	1	500	270	0.95
		Deep	41	1	40		
Reference		Shallow	30	4	375	30	0.15
		Deep	30	9	2020		
Outer edge	New		26	6	808	95.5	
	Old		33	10	1372	95	
Inner edge	New		23	0	0	-	
	Old		33	2	51	25	

2.4.5 *GoPro Identification of Salmon*

GoPro camera video footage was used to supplement fish species identification in non-turbid waters and included 26 encounters for a total of 1652 salmon. As a comparison, the acoustic camera detected salmon 79 times for a total of 8207 salmon (overall median: 28.5). The GoPro detected only 0.2% of the salmon that were not detected by the acoustic camera, with 10 detections that summed to 20 salmon (overall median: 1.5). In nearly all the GoPro-only sightings, the salmon were close to the kayak and likely in the acoustic camera’s nearfield, making them acoustically undetectable (e.g., Frias-Torres and Luo 2009).

2.4.6 *Nearshore Microhabitats*

Final models for the nearshore, all-season models included the fixed effects of microhabitat, light, depth, and scaled transect lengths, and the random effect of Julian day (Appendix A2.1, A2.2). Salmon were less likely to be present in non-reference transects and more likely to be present in deep transects because salmon absence was positively associated with all microhabitats compared to reference habitats, and negatively associated with deep transects compared to shallow habitats (Figure 2.6, Table 2.5). The strongest effect was at old UP transects, where every unit increase of old UP raised the odds ratio of a zero count 36 times and every increase in new UP transects raised the odds ratio of an absence 11 times (Table 2.6). This can be interpreted as salmon were 36 times less likely and 11 times less likely to be detected at old and new UP locations, respectively, compared to reference locations. Salmon were also 3 times more likely to be absent in BP transects compared to reference transects and 3 times more likely to be present in deep transects compared to shallow. Positive counts among all microhabitats resembled each other, indicating that although salmon were present less frequently in some microhabitats, they shoaled in comparable numbers among microhabitats.

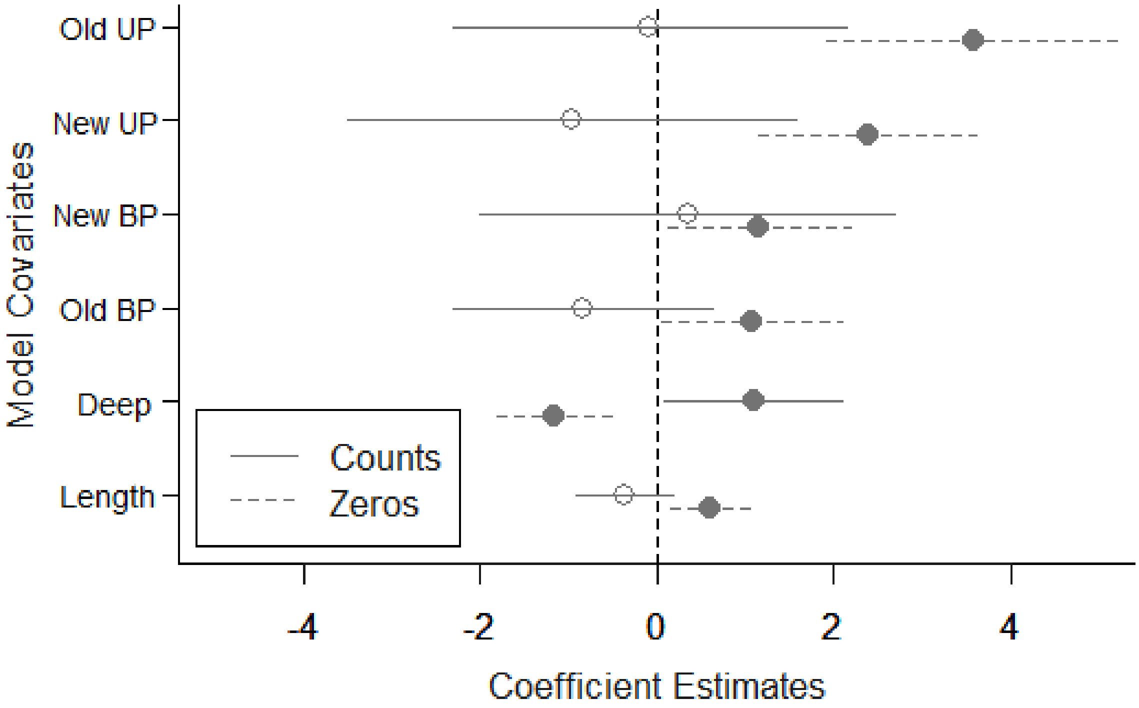

Figure 2.6. Coefficient estimates of GLMM results for the daytime all-season nearshore model. The model compares salmon presence and counts at microhabitats to reference intercepts. Nearshore reference transects and shallow depth were estimated by the intercept, such that additional effects measured the difference between the remaining microhabitats and reference transects, and the difference between depths. Horizontal lines are 95 % confidence intervals. Filled circles indicate the confidence interval width does not include zero.

2.4.7 *Pier End Microhabitats*

The best pier ends model compared pier end salmon presence and counts and included pier end microhabitat (outer edge and inner edge), site type (old and new), light, and Julian day (Appendix A2.2, A2.3). By site type, salmon were equally likely to be detected between old and new pier ends during the day. There was a strong, positive effect for salmon absence and counts at inner pier edge microhabitats compared to outer pier edges. Specifically, it was 11 times less likely for salmon to be present, and counts were 5 times lower, under inner pier edges compared to the open, outer edges (Tables 2.5, 2.6). This indicates that juvenile salmon prefer to swim in the open water around the full length of the pier rather than beneath the inside edge of the pier end, even when ambient light penetrates beneath the pier end.

2.4.8 *Seawall Habitats: All-season and Peak Densities*

The best models for salmon distributions among nearshore seawall habitats for the entire sampling season and the eight-week peak densities period included microhabitat, light, and depth (Appendix A2.2, A2.4). For all-season detections, model results indicated that old BP microhabitats had higher detections than all remaining habitats. Salmon were 13 times and 4 times more likely to be absent from old and new UP locations compared to old BP locations (Tables 2.5, 2.6). During the eight-week peak migration, model results showed no significant difference in juvenile salmon detections in all habitats compared to old BP locations. However, coefficients indicated that salmon were 4 times and 1.5 times more likely to be absent from old and new UP transects, respectively. The lower odds ratios during peak migration demonstrate that salmon utilize all habits more uniformly, even suboptimal ones, when they occur in high densities. For new BP transects, there was no all-season or peak migration difference compared to old BP transects.

All-season positive counts among seawall transects were similar apart from old UP transects, where model results indicated that counts were 3.5 higher compared to old BP transects (Table 2.6). However, this result is distorted by a high count in old UP transects (2 all-season detections, *n* = 500, 40). These two detections compare to 18 detections that summed to 721 salmon in old BP transects and 11 detections that summed to 1079 salmon in new UP transects. During peak migration, counts were alike among microhabitats except at new UP transects, where counts were 3 times lower than at old BP transects. Throughout the season and during peak migration, juvenile salmon were 2-3 times more likely to be present in deep transects, with counts around 1.5 times higher, compared to those observed in shallow transects.

Table 2.5. Final daytime GLMM equations characterizing salmon presence/absence and counts. The models characterize juvenile salmon 1) absence and positive counts, among all nearshore microhabitats, 2) absence and positive count among all pier ends microhabitats, 3) absence and positive counts among all nearshore seawall microhabitats, and 4) presence and positive counts among all nearshore seawall microhabitats during peak densities. The fixed effects fourth model during peak densities is developed using R package ‘pscl’ to characterize salmon presence, with coefficient signs opposite to all other models, which are mixed effects models developed using R package ‘GLMMadaptive’. Model parameters are defined in the main text.

Nearshore microhabitats	Zeros:	$absence \sim 1.06 + 2.38[New\ Pier] + 1.16[New\ SW] + 3.58[Old\ Pier] + 1.07[Old\ SW] + 0.19[scaled(light)] - 1.15[deep\ depth] + 0.60[scaled(transect\ length)] + [Julian\ day] + \varepsilon_i$
	Counts:	$counts \sim 2.38 - 0.96[New\ Pier] + 0.35[New\ SW] - 0.07[Old\ Pier] - 0.84[Old\ SW] - 0.39[scaled(light)] + 1.09[deep\ depth] - 0.37[scaled(transect\ length)] + [Julian\ day] + \varepsilon_i$
Pier ends	Zeros:	$absence \sim 0.64 + 2.44[\ Inside\ pier\ edge] + 0.42[New\ pier\ ends] + 0.27[scaled(light)] + \ Julian\ day] + \varepsilon_i$
	Counts:	$counts \sim 2.88 - 1.56[Inside\ pier\ edge)] + 2.0[New\ pier\ ends] + 0.09[scaled(light)] + [Julian\ day] + \varepsilon_i$
Nearshore Seawall	Zeros:	$absence \sim 1.91 + 1.42[New\ UP] + 0.10[New\ BP] + 2.59[Old\ UP] + 0.35[scaled(light)] - 1.16[deep\ depth] + [Julian\ day] + \varepsilon_i$
	Counts:	$counts \sim 1.01 + 0.10[New\ UP] + 1.26[New\ BP] + 3.46[Old\ UP] - 0.79[scaled(light)] + 1.73[deep\ depth] + [Julian\ day] + \varepsilon_i$
Nearshore Seawall: peak	Non-zero:	$presence \sim -2.35 - 0.48[New\ UP] + 0.89[New\ BP] - 1.35[Old\ UP] - 1.43[scaled(light)] + 0.91[deep\ depth] + \varepsilon_i$
	Counts:	$counts \sim 3.9 - 1.99[New\ UP] - 0.76[New\ BP] - 1.50[Old\ UP] - 0.38[scaled(light)] + 1.44[deep\ depth] + \varepsilon_i$

Table 2.6. Daytime GLMM results with estimates, standard errors, p-values, and odds ratios. The models characterize juvenile salmon 1) absence and positive counts among all nearshore microhabitats, 2) absence and positive counts among all pier ends microhabitats, 3) absence and positive counts among all nearshore seawall microhabitats, and 4) presence and positive counts among all nearshore seawall microhabitats during peak densities. The fixed effects fourth model during peak densities is developed using R package ‘pscl’ to characterize salmon presence, with coefficient signs opposite to all other models, which are mixed effects models developed using R package ‘GLMMadaptive’. Model parameters are defined in the main text. Confidence interval widths between 2.5% and 97.5% are included.

Nearshore all-season: Absence	Estimate	SE	p	Odds ratio
Intercept: Ref site, Shallow	1.06	0.44	0.83	0.91
New UP	2.38	0.63	0.0001	10.84
New BP	1.16	0.53	0.03	3.19
Old UP	3.58	0.85	< 1e -04	35.79
Old BP	1.07	0.52	0.04	2.92
Light	0.19	0.17	0.24	1.21
Depth (deep)	-1.15	0.34	0.0007	0.32
Transect length	0.60	0.23	0.008	1.82
Nearshore all-season: Count	Estimate	SE	P	2.5 % - 97.5 %
Intercept: Ref site, shallow	2.38	0.97	0.0004	1.56, 5.38
New UP	-0.96	1.29	0.46	-3.49, 1.57
New BP	0.35	1.20	0.77	-2.00, 2.70
Old UP	-0.07	1.13	0.95	-2.29, 2.15
Old BP	-0.84	0.75	0.26	-2.30, 0.63
Light	-0.39	0.29	0.18	-0.96, 0.18
Depth (deep)	1.09	0.51	0.03	0.08, 2.10
Transect length	-0.37	0.28	0.20	-0.92, 0.19
Pier Ends all season: Absence				Odds ratio
Int: Old site, Outer edge	0.64	0.39	0.098	1.90
New	0.42	0.59	0.48	1.52
Inner pier edge	2.44	0.80	0.002	11.49
Light - lx	0.27	0.31	0.38	1.31
Pier ends all season: Count				2.5% - 97.5%
Int: Old site, Outer edge	2.88	1.22	0.018	0.49, 5.27
New	1.99	1.85	0.28	-1.63, 5.62
Inside pier edge	-1.56	0.71	0.028	-2.94, -0.17
Light - lx	0.09	0.25	0.72	-0.40, 0.58
Nearshore seawall all season: Absence				Odds ratio

Int: Old BP, Shallow	1.91	0.37	< 1e -04	6.77
New UP	1.42	0.53	0.007	4.13
New BP	0.10	0.40	0.81	1.10
Old UP	2.60	0.78	0.0008	13.39
Light	0.35	0.32	0.27	1.42
Depth (Deep)	-1.16	0.38	0.002	0.31
Nearshore seawall all season: Count				2.5% - 97.5%
Int: Old BP, Shallow	1.01	0.94	0.29	-0.84, 2.85
New UP	0.10	1.29	0.94	-2.43, 2.64
New BP	1.26	1.14	0.27	-0.97, 3.48
Old UP	3.46	1.50	0.02	0.52, 6.40
Light	-0.80	0.89	0.37	-2.54, 0.95
Depth (Deep)	1.73	0.66	0.009	0.44, 3.02
Nearshore seawall peak: Presence				Odds ratio
Int: Old BP, Shallow	-2.35	0.60	< 1e-04	10.5
New UP	-0.48	0.70	0.49	1.62
New BP	0.89	0.61	0.15	0.41
Old UP	-1.35	0.90	0.13	3.87
Light	-1.43	0.92	0.12	4.19
Depth (Deep)	0.91	0.48	0.06	-0.40
Nearshore seawall peak: Count				2.5% - 97.5%
Int: Old BP, Shallow	3.90	0.60	< 1e-04	4.26, 6.43
New UP	-2.00	0.77	0.01	-3.51, -0.48
New BP	-0.76	0.67	0.25	-2.08, 0.54
Old UP	1.50	1.01	0.14	-0.48, 3.48
Light	-0.39	1.20	0.75	-2.7, 1.96
Depth (Deep)	1.44	0.58	0.01	-2.59, -0.30

2.5 DISCUSSION

Acoustic salmon counts were significantly higher than snorkel counts across most microhabitats, although ratios were not consistent. The highest ratio of acoustic to snorkel count values occurred at UP habitats, which have the lowest light levels and are more challenging for optical surveys. Given that ratio values of acoustic to snorkel counts ranged from 0.8 to 17.7 and were different across site types and microhabitats, the two sampling methods are not scalars but complement each other, where one may be preferred over the other depending on research goals, habitats, and ambient light levels. For example, snorkel methods are best used to identify fish feeding behaviors and to identify fish to species, while acoustic methods are preferred when sampling at night or in turbid waters.

Bias exists in all sampling methods that are not a complete population census (Edgar et al. 2004). This includes acoustic and snorkel sampling methods, which are both biased low (Sale and Douglas 1981; Pacific Salmon Commission 2007; Thanopoulou et al. 2018), although the DIDSON is more accurate than visual surveys at lower fish densities or higher turbidity (Martignac et al. 2014). Optical fish detectability varies by habitat, light intensity, turbidity (with the net effect of changing the field of view in visual surveys), observer experience, and animal fleeing behaviors (Stanley and Wilson 1995; Buckland et al 2015). Acoustic fish detectability is a function of fish densities, debris, and fish that are too close to the transducer to be detected (Pacific Salmon Commission 2007). An example of detection bias is that snorkel surveys in 2019 detected no salmon in deep, under-pier transects, which are low-light, piling-filled environments, while acoustic surveys detected 90 juvenile salmon in these locations. In contrast, snorkel salmon detections in 2019 (n = 70) exceeded acoustic detections (n = 40), indicating that snorkel surveys may be advantageous in identifying lone fish or individuals within mixed groups of fish. It should be noted that acoustic and snorkel surveys were not conducted in tandem and have different advantages and disadvantages, so overall fish counts and detections were not expected to be identical. Snorkel surveys are optimal in identifying fish to species in non-turbid and/or well-lit water compared to the DIDSON, which does not detect certain physical identifiers (e.g., color and distinguishing markings). Salmon could be identified to genus by size, shape, swimming behavior, and time of year during replays of the acoustic data, but species identification ultimately was based on supplemental data and knowledge of species-specific annual presence along the waterfront collected from netting and snorkel surveys (e.g., Toft et al. 2007).

Higher acoustic fish counts in low light and high turbidity waters are attributed to the acoustic camera's ability to sample over a much greater range and volume than an optical camera or snorkel surveyors (Frias-Torres and Luo 2009). Salmon were distinguishable from debris or immobile fish because they are mobile and easily startled by the kayak (see also Frias-Torres and Luo 2009). The ability of the multibeam sonar to record and digitize salmon before they fled enabled counting and species identification when reviewing the data (Stanley and Wilson 1995; Buckland et al. 2015).

Acoustic salmon counts were probably underestimated in all BP and reference microhabitats due to a mixing of conspecific, similar-sized species mid-season, which increased the numbers of mixed species groups (e.g., shiner perch and salmon) and reduced salmon counts, as salmon in these aggregations were not added to transect counts (Appendix A2.5). Fish not identified to family or species generally occurred in mixed shoals of surfperches (*Embiotocidae*) at BP and reference transects, indicating that salmon counts are probably underestimated at these locations. As a result, BP and reference habitats may be more preferred salmon habitats than model results indicate.

The optimal survey method for nearshore fish communities may include pairing optic and acoustic methods. To compare the strengths and constraints of both methods, snorkel methods are superior for species identification, length estimation, and other behavioral assessments (e.g., feeding), yet are challenged by low light levels, high turbidity, and physiological stress experienced by snorkelers. The DIDSON has a longer detection range and can sample small fish in dark or turbid environments but is limited in species identification and length estimation (Burwen et al. 2010). Used in isolation, any acoustic-based species identification is challenging (Horne 2000), but fish can be identified using supplementary information (e.g., size, shape, and swimming behavior; Baumgartner 2006; Doehring et al. 2011) and validated by visual methods.

One objective of the seawall modifications along the Seattle waterfront was to “create an effective intertidal migratory corridor for juvenile salmonids” (EBSP 2019). Post-construction monitoring in the first year has demonstrated that this objective was achieved, evidenced by increased juvenile salmon presence and abundance in previously less-occupied habitats (Sawyer et al. 2020). This achievement was likely due to a combination of minimal structure, higher light levels compared to pre-construction (Sawyer et al. 2020), and an elevated seafloor in the intertidal corridor. Although new under pier habitats had similar light levels at both depths and old under pier habitats had higher light levels than new UPs, juvenile salmon were infrequently detected in new, deep UP transects and rarely detected in old UP transects. Both transect types contained high densities of pilings and deep water. Fish can grow well in open piling fields (Grothues et al. 2016), so increased salmon presence in the intertidal corridor suggests that the lack of structure (e.g., pilings) and intertidal benches may be mitigating lower light levels from the cantilevered sidewalk and positively influencing juvenile salmon presence in the corridor.

It is hypothesized that in this study, habitat preference likely superseded the trend to navigate in nearshore, suboptimal microhabitats (i.e., old, under piers). Consistent with this supposition, the highest median salmon counts among transects were observed at pier ends (except for one outlier count at an old UP transect) and pier end counts were similar to those observed between piers, indicating that navigating in deep water around pier ends may be preferred over swimming under piers, especially for larger groups. High overall salmon counts between piers, similar to reference site counts, may indicate habitat preference but also that piers temporarily constrain salmon migration: salmon may wait for low tide to swim under piers (Munsch et al. 2014) or navigate around pier ends.

Additional monitoring is required to determine if shoreline eco-engineering mitigates the long-term effects of coastline armoring. Longer-term monitoring may also detect unintended consequences of coastal modifications (Bash and Ryan 2002; Barnas et al. 2015; Lichatowich et al. 1995). For example, in this study new, shallow BP transects had lower salmon counts compared to pre-construction years (cf. Toft et al. 2007; Munsch et al. 2014; Sawyer et al. 2020). One explanation is that juvenile salmon moved to deeper water to avoid shadows cast by the overhead cantilevered sidewalk. However, old shallow BP transects also had lower salmon counts compared to their deep counterparts, without overhead cantilevered sidewalks. With only one year of old and new site data available, the reason for this contrasting pattern cannot be explained at this time.

As coastal urbanization continues, the need for seawalls, piers, and armoring will remain. The requirement for estuary-dependent fauna to have healthy habitats also remains (Ray 2005), requiring nearshore spatial and economic planning in urban areas (Bilkovic and Roggero 2008). To benefit juvenile salmon that use estuarine corridors, future eco-engineered habitats containing piers and seawall structures should minimize pier pilings and modify passive light integration (e.g., the glass-block sidewalk) to increase light penetration. Additional monitoring efforts should be expanded to include the timing and route of individual fish trajectories (e.g., telemetry) through migratory corridors. Telemetry data can then be used to determine habitat use, residence times, energetic costs, and mortality rates of individuals and runs of specific salmonid species. Different trajectories through a migratory corridor will have different energetic costs to juvenile salmon and may influence growth or survival. Using the Seattle waterfront as an example and assuming a sustained swimming rate of one body length per second (juvenile salmon post-smolts can sustain a maximum swimming speed of around 2-4 body lengths s^{-1} for a limited time (Hvas and Oppedal 2017)), a 60 mm juvenile salmon would take an hour and fifty minutes to navigate around three sides of one of the study piers (Pier 70: length side 1 = 166m, length side 2 = 158 m, deep pier edge = 65 m, pier edge along seawall = 79m). In contrast, it would take the same fish 20 minutes to traverse the width of the pier along a straight path close to the shoreline. This difference in trajectory length around a single pier represents 5.4 times increase in swimming energy expended assuming a direct migratory path. With a total of 12 piers along the Seattle waterfront, there is up to 81 times the cost of migrating around compared to under Seattle's waterfront piers. The proportion of fish migrating along the perimeter of piers compared to those traversing under piers parallel to the shoreline is unknown, but any in-water structures along Seattle's waterfront likely increases locomotory energy expenditures, reduces feeding opportunities, and elevates predation risks for juvenile salmon. The total impact of increased energy costs navigating around pier or other overwater structures during the ocean migration phase of the juvenile salmon life history is not known for any salmonid species.

Chapter 3. NOCTURNAL HABITAT USE AND BEHAVIORS OF JUVENILE PACIFIC SALMON ALONG AN ECO-ENGINEERED SEAWALL

3.1 INTRODUCTION

Impacts of shoreline armoring (i.e., physical structures) on fish distributions have traditionally been studied during daylight hours but it is not known if these impacts are equivalent at night. Shoreline development results in estuarine degradation and loss (Cloern et al. 2016, Brophy et al. 2019). Specific impacts on nearshore aquatic communities, including juvenile salmon, are detailed in Chapter 1, section 1.1. Recognizing the effects of armoring on juvenile fish habitat, Seattle's replacement seawall was modified to mitigate some of the harmful impacts on juvenile salmon (see Chapter 1, section 1.1). The intent of the modifications was to create a shallow water, ambient-lit, continuous corridor for juvenile salmon to transit along the waterfront. Daytime ambient light was considered for replacement seawall modifications (e.g., glass block embedded sidewalk), but influences of nearshore armoring and habitat modifications may differ at night. For example, artificial light at night (ALAN) influences nocturnal juvenile salmon behaviors by attracting juvenile salmon and delaying migration (Salo 1979, Tabor et al. 2004; Chapter 1, section 1.1). Sampling at night is needed to gauge if the relative impacts of shoreline armoring and eco-engineering modifications on juvenile salmon are equivalent to those during the day.

Elliott Bay in Washington, USA, is an example of juvenile salmonid estuarine habitat with extensive shoreline modification through ongoing urbanization. Although 90% of the shoreline is armored, nearshore waters continue to be used as a migration corridor for juvenile

Pacific salmon (Toft et al. 2007) and their transit through Elliott Bay and predisposition to nearshore habitats puts them in proximity to Seattle's overwater structures and armoring. Knowledge of nocturnal fish distributions in Elliott Bay is limited because traditional sampling effort has used daytime snorkeling and net sampling (e.g., Toft et al. 2007, Beauchamp and Duffy 2011). Fish assemblages may differ at night, so limiting sampling to daylight hours can bias estimates of habitat use and fish abundances if inferences are extended to nighttime (Rountree and Able 1993, Maes et al. 1999, Hagan and Able 2008).

Active acoustics enables nighttime quantification of fish distributions due to its ability to sample fish in dark and turbid aquatic environments. A multibeam sonar, the dual-frequency identification sonar (DIDSON) can be used as a sampling tool to expand knowledge of diel fish distributions and potential influences of habitat modifications (e.g., ALAN). The intent of sampling during dark hours is to characterize juvenile salmon distributions along an urban shoreline at night, compare them to daytime distributions, and to quantify the relative importance of environmental factors influencing nighttime salmonid habitat use along an armored shoreline.

3.2 METHODS

3.2.1 *Study Location*

The study sites are located along Seattle's central waterfront within Elliott Bay, Washington, USA (47.6062° N, 122.3321° W Figure 3.1) (Chapter 2, section 2.2.1).

3.2.2 *Sample Design*

Eight sampling sites identical to the daytime sampling sites were monitored at night between May and August 2019 (Chapter 2, section 2.2.2; Figure 3.1).

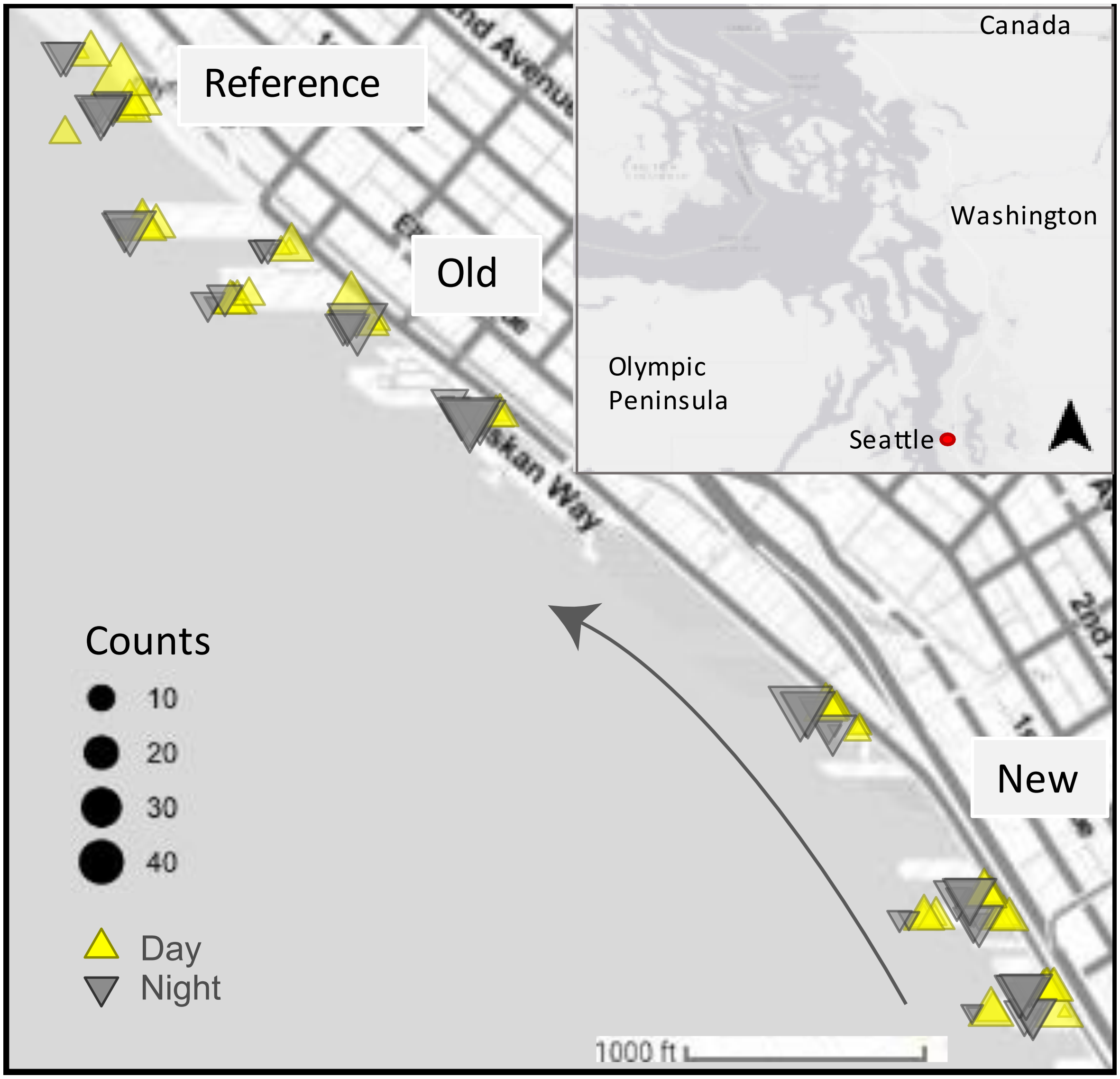

Figure 3.1. Size-proportional day/night salmon counts along the Seattle waterfront. Shapes are size proportional to the square root of salmon counts and are shape sorted by diel period (day; yellow) and (night; gray). Night counts are offset to the left of day counts to minimize overlap, although day/night sampling occurred at identical transects. Counts are minimally jittered to reduce overlap and are not at their exact coordinates. **Inset**: Study site location in the Pacific Northwest, USA. **Main**: Study site location in Seattle, WA, USA. Transects are at reference, old, and new sites. Arrow indicates the primary migration path of juvenile salmon along the Seattle waterfront.

3.2.3 *Data Acquisition*

In 2019, acoustic data were collected at night to quantify fish presence and abundance at all sites. All transects were sampled once each night to facilitate a comparison of day/night salmon distributions. Sampling began at least thirty minutes post-nautical twilight and near the time of a mean tide, when possible, to enable access beneath all piers and passage through the intertidal corridor (Figure 3.2). Night sampling occurred predominantly within one week of day sampling. It was assumed that the same pulse of migrating juvenile salmonids was sampled at day and night throughout the surveying season, based on natural Chinook salmon residence times in Elliott Bay pier habitats averaging 18.4 days ± 2.3 days, with hatchery salmon residing twice as long (Ruggerone and Volk 2014).

Figure 3.2. Artificial light present during night sampling along the Seattle waterfront. **Left**: Artificial light at night illuminating the Seattle waterfront. **Right**: Nighttime navigation in the intertidal corridor. Visible in the photo is the overhead glass block sidewalk and the textured seawall on the left.

A DIDSON multibeam sonar mounted beneath the center of a 3 m kayak was used to quantify fish presence and counts at night. A stationary DIDSON has been used at night to monitor salmon escapement and spawning (Dunbar 2010; Tiffan et al. 2005), identify fish assemblages (Becker et al. 2011, Becker and Suthers 2014), and monitor the effect of ALAN on fish interactions (Becker et al. 2013, Bolton et al. 2017). The operating frequency, sampling rate, and sampling range settings were identical to those used during day sampling (Chapter 2, section 2.2.3). The DIDSON transceiver was connected to a waterproof laptop running data acquisition software (DIDSON V5-26) and a timestamped file was created for each transect. A HOBO pendant Temperature/Light data logger (Onset, Cape Cod, MA, USA) mounted to a PVC pipe and deployed off the front of the kayak near the water surface measured luminous flux (lux) and temperature (°C) every ten seconds. Midway through the sampling season, a GoPro Hero 7 camera was deployed off the front of the kayak along the transect to provide an independent optic record to aid in fish species identification. The GoPro was mounted adjacent to two underwater lights and was deployed post-acoustic sampling to minimize fish disturbance during acoustic sampling.

3.2.4 *Data Processing*

Positive phototaxis behaviors during GoPro sampling facilitated the visual categorizing of small numbers of salmon and large numbers of other fish at night. Low fish biodiversity along the waterfront along with different body morphologies and swimming behaviors facilitated fish identifications (Baumgartner 2006, Doehring et al. 2011, Martignac et al. 2014). To process acoustic data, DIDSON data files were replayed three times to count and record fish, to verify counts, and to review fish identification and counts in conjunction with GoPro video footage. Data processing methods are fully described in Chapter 2, section 2.2.4. Detections and counts of juvenile salmon were summed by microhabitat, site type, and depth, with reference site transect counts scaled by transect length.

3.3 DATA ANALYSIS

3.3.1 *Comparisons of Diel Salmon Counts by Microhabitat, and within Night Microhabitats, by Depth*

Nonparametric Wilcoxon Mann Whitney tests (Mann and Whitney 1947; Wilcoxon 1945) were used to determine if transect counts, by microhabitat, differed between day and night. Mann Whitney tests were also used to compare salmon distributions within each nighttime microhabitat, by depth. These tests were used to identify differences between categories rather than quantify day/night and depth differences within microhabitats.

3.3.2 *Characterizing Nighttime Salmon Presence and Distributions*

To characterize nocturnal salmon presence and distribution along the Seattle waterfront, generalized linear models (GLMs) and generalized linear mixed models (GLMMs) that included zero-inflated or hurdle models were considered for acoustic presence and count data (i.e., Poisson or negative binomial). Zero-inflated and hurdle models include a model for excess zeros and a model for non-zero counts. Model choice was informed by consideration of which model was more ecologically relevant to the system and to the degree in which the data met model assumptions (see Chapter 2, section 2.3.2). To avoid statistical assumption violations and ensure appropriate use of statistical analyses, exploration of the acoustic dataset followed the protocol described by Zuur et al. (2010). Dispersion tests were used to identify the distribution of the data. Akaike Information Criterion (AIC; Akaike 1974) and Bayesian Information Criterion (BIC; Schwarz 1978) metrics were used to compare potential count data distributions (e.g., Poisson, negative binomial) to evaluate how each distribution fit the data, and to compare potential over- and under-parameterization of model fits. To test for excess zeros (i.e., more than would be expected under a Poisson or negative binomial distribution), predicted probabilities of zeros for acoustic data were summed. Due to the underfitting of zeros, zero-inflation and zero-alteration models were compared (R package ‘glmmTMB’ (Brooks et al. 2017 v 1.0.2.1). Using AIC and BIC values and consideration of how the data met the assumptions of each model, the most appropriate model structure was a hurdle model using a logit link function and a negative binomial count distribution (log link function) (Appendix A2.6). A hurdle model was an appropriate choice as it was not possible to apportion the proportions of zero counts as fish absence versus lack of detection, and the hurdle model does not differentiate between sampling or real zeros (Zuur et al. 2009). To address potential spatiotemporal effects due to repeated measures at repeated sites, likelihood ratio tests were used to test random effects including ‘month’, ‘Julian day’ and ‘site’ against models with no random effects.

Candidate covariates in model selection included microhabitat, depth (3 m from seawall “shallow” and 10 m from seawall – “deep”), transect length, and *in situ* measurements of water temperature (°C) and light (lx). Presence and count data were scaled to a standard 25 m transect length. To avoid model overfitting, depth was included as a covariate instead of being assigned to each microhabitat category. All statistical analyses were conducted using R version 3.6.1 (R Core Team, 2019) and the R packages ‘GLMMadaptive’ (Rizopoulos, 2020 v 0.7-15) and ‘pscl’ (Zeileis et al. 2008 v 1.5.5).

Three GLMM model groups were used to characterize nighttime microhabitat use by juvenile salmon along the urbanized coastline: day/night (one-hour post-sunrise and 30 minutes post-nautical twilight) all-season and peak migration (May 05, 2019 – June 15, 2019), nearshore all-season, and pier ends all-season. 97% of the total nighttime salmon counts at nearshore seawall sites, excluding reference sites and pier ends, occurred during the period of peak densities. In each case, final models were chosen considering likelihood ratio test values, AICs, and by considering the ecological relevance of the model.

The day/night all-season and peak densities model group was developed to compare overall juvenile salmon presence and counts between day and night sampling. ‘Day’ values are estimated by the intercept and the additional differences between day and night are measured.

The nearshore all-season model was developed to compare nighttime juvenile salmon presence and counts among all nearshore habitats throughout the sampling season. Nearshore reference and shallow depth transects were estimated by the intercept, and the additional effects measured the difference between the remaining microhabitats and reference transects, and the difference between shallow and deep transects. Reference transects and shallow depth transects are used as the baseline for transect comparisons.

The pier ends model was developed to evaluate potential nighttime differences between ‘old’ and ‘new’ pier end microhabitats (i.e., pier ends that are connected to old or new seawall habitat), and between the outer edge of the pier end compared to the inner edge that has overhead decking. ‘Outer edge’ and ‘old’ pier ends were estimated by the intercept, and the differences measured were between outer and inner pier end edges, and the overall difference between old and new pier ends. The model structure for final models is specified as:

$$f_{nb\ hurdle}(y;\pi,\mu,r)\begin{cases} f(binomial)(y=0;\ \pi) \\ [1-f(binomial)(y=0;\ \pi)] \end{cases} * \left(\frac{f_{nb}(y=0;\mu,r)}{1-f_{nb}(y=0;\ \mu,r)}\right)$$

where π = the probability of a zero (i.e., absence) for models with a random effect, or the probability of a non-zero (i.e., presence) for models with no random effect; an increase in coefficient value signifying a higher likelihood of absence or presence. The zero equation is accompanied by a negative binomial distribution to model positive counts, where μ = the mean of positive counts, and r = the scale parameter of positive counts. All covariates were incorporated in the zero and count portions of each model because understanding their effect on presence and abundance was of interest. Final models addressed temporal autocorrelation due to repeated measures by including random intercepts that incorporated the month of the year or Julian day (Bolker et al. 2009). The exception was a model during peak densities that required no temporal random effect. Individual sites were excluded as random effects owing to high AIC and BIC metric values. Light measurements were excluded from final analyses because they consistently measured 0 lux even in habitats illuminated by ALAN. The effect of low light conditions on salmon microhabitat use cannot be resolved in this study.

3.4 RESULTS

Over the sampling season, 412 acoustic transects were conducted at night from May 5 through August 13, 2019. Juvenile salmon were detected 79 times (*n* = 8207) during day sampling and 97 times (*n* = 21957) during night sampling (Table 3.1; see Chapter 2, section 2.2.3 for full results of day sampling). By site type, paired day/night salmon counts summed to 2440 and 2189 at reference sites, 2684 and 9582 at old sites, and 3083 and 10186 at new sites. By depth, day/night salmon counts summed to 2031 and 7785 for shallow transects, and 3900 and 12845 for deep transects. At pier ends, day/night salmon counts summed to 2231 and 1326. By seawall habitat type, day/night salmon counts summed to 1917 and 16062 between piers and 1619 and 2380 under piers. Ninety-seven percent of the salmon detected at nearshore seawall sites (i.e., not including reference and pier end transects) occurred during peak densities.

3.4.1 *Diel Salmon Count Comparisons, by Microhabitat*

Mann Whitney tests identified no day/night salmon count differences at under-pier and reference transects, indicating that day counts at these microhabitats are similar to night counts at the same microhabitats (Table 3.1, Figure 3.3). However, Mann Whitney tests identified a day/night difference in salmon counts at BP transects, which is attributed to high nighttime counts (Figure 3.4). At pier ends, Mann Whitney tests identified no day/night salmon count differences within each site type (old, new), but out of 90 day/night samples at new pier ends, there were only six daytime detections that summed to 808 salmon and two nighttime detections that summed to 9 salmon, so the test results have low power due to few positive counts.

Table 3.1. Transects sampled, presence, and salmon counts within microhabitats, by diel period. Also included are positive median counts on all nighttime detections > 5. Mann Whitney test results compare salmon counts within each microhabitat, by diel period. UP = Under Pier, BP = Between Pier, Ref = Reference sites, Peak = 8-week period of peak salmon densities.

	Day			Night				
Nearshore	Sample	Presence	Count	Sample	Presence	Count	Median	p-value
New UP	86	11	1079	66	13	2259	140	0.24
New BP	86	16	1196	66	27	7918	8	0.004
Old UP	88	2	540	66	2	121	-	0.79
Old BP	87	18	721	68	23	8144	100	0.02
Reference	60	13	2395	46	18	2188	25.5	0.08
Ends	Sample	Presence	Count	Sample	Presence	Count	Median	Mann W
New	49	6	808	41	2	9	-	0.19
Old	66	12	1423	47	11	1317	10	0.57
Reference	14	1	45	12	1	1	-	-
Counts		79	8207		97	21957		
Median counts								
Overall – all-season			30			20		0.0004
Overall – peak			82.5			430		0.007
Between pier – peak			100			575		

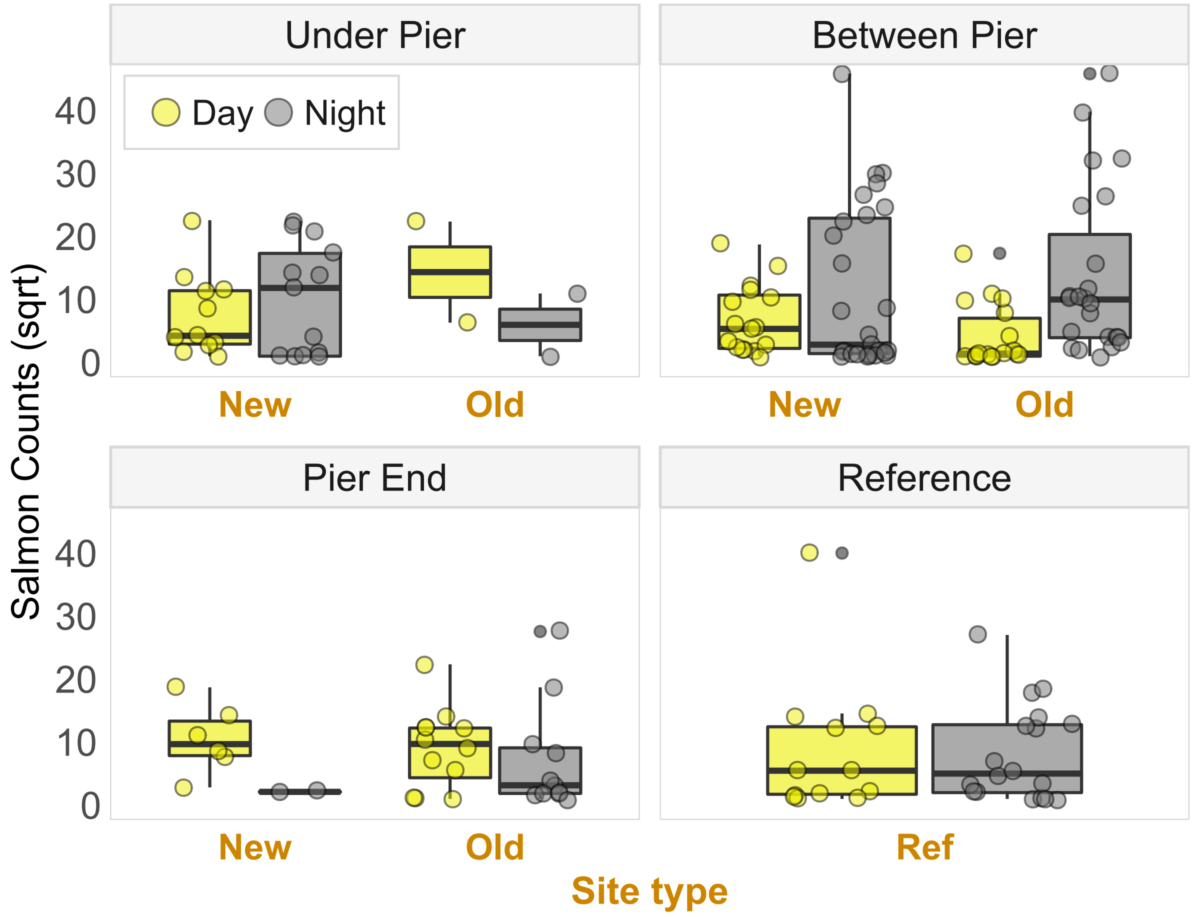

Figure 3.3. Box plots of positive, square root transformed salmon counts among microhabitats. Microhabitats include site type (new, old, reference) and under pier, between pier, and pier ends. Box plots and points are sorted by diel period (day = yellow, night = gray). Boxplots depict the interquartile range (25th and 75th quartiles), horizontal lines in boxplot show the median, and whiskers (vertical lines) show 1.5 times the interquartile range, along with outliers. Points are jittered to minimize overlap.

3.4.2 *Salmon Count Comparisons within Microhabitats, by Depth*

Mann Whitney tests identified a nighttime, within microhabitat difference in salmon counts only at old BP transects by depth category (i.e., shallow, deep) (Table 3.2). This result differs from day count results, where there was a difference between old and new BPs, by depth (Chapter 2, section 2.3.6). The lack of a statistical difference in salmon counts for new BPs at night, by depth, suggests that salmon are utilizing the new, shallow BP corridor section at night similar to the use of new, deep BP transects, where salmon are more frequently encountered.

3.4.3 *Diel All-Season and Peak Densities*

Final models comparing all-season and peak day/night detections and counts include diel period and Julian day covariates for the all-season model, and diel period for the peak densities model (Appendix A3.1, A3.2, A3.3). Throughout the sampling season, it was twice as likely to detect salmon at night compared to day because salmon absence was negatively associated with nighttime. This ratio is consistent between the full season (odds ratio = 0.56) and during peak migration (odds ratio = 0.55) (Tables 3.3, 3.4). Odds ratios are measures of association between a variable and the outcome (e.g., the odds ratio of 1 signifies that the odds of the outcome are unaffected by the variable). And although model results show day/night counts to be similar for all-season sampling, higher juvenile salmon night counts during peak densities (n = 17951) compared to daytime counts (n = 3454) corroborate model results that counts are 1.5 times higher at night during peak densities. Positive median salmon counts were five times higher at night compared to day (Med = 430; Med = 82.5) during peak densities. As an example of diel differences, 6729 salmon were counted at new sites during a May night survey. The following morning, 585 salmon were counted along the same transects.

3.4.4 *Nearshore Microhabitats*

Final nearshore, all-season model equations include microhabitat, depth, transect lengths, and month covariates (Appendix A3.3, A3.4). Salmon were less likely to be present in non-reference microhabitats and more likely to be present at deep transects (Tables 3.3, 3.4). The strongest effect was at old, under pier (UP) transects, where every unit increase of old UP raised the odds ratio of a zero count (i.e., absence) 31 times (Figure 3.5). Concurrently, every unit increase in new UP raised the odds ratio of a zero count 4 times. This difference in odds ratios illustrates that salmon were 31 (old UP) and 4 (new UP) times less likely to be present in these microhabitats compared to reference microhabitats. During the day, salmon were 36 (old UP) and 11 (new UP) times less likely to be present in these microhabitats compared to reference microhabitats (Chapter 2, section 2.4.1).

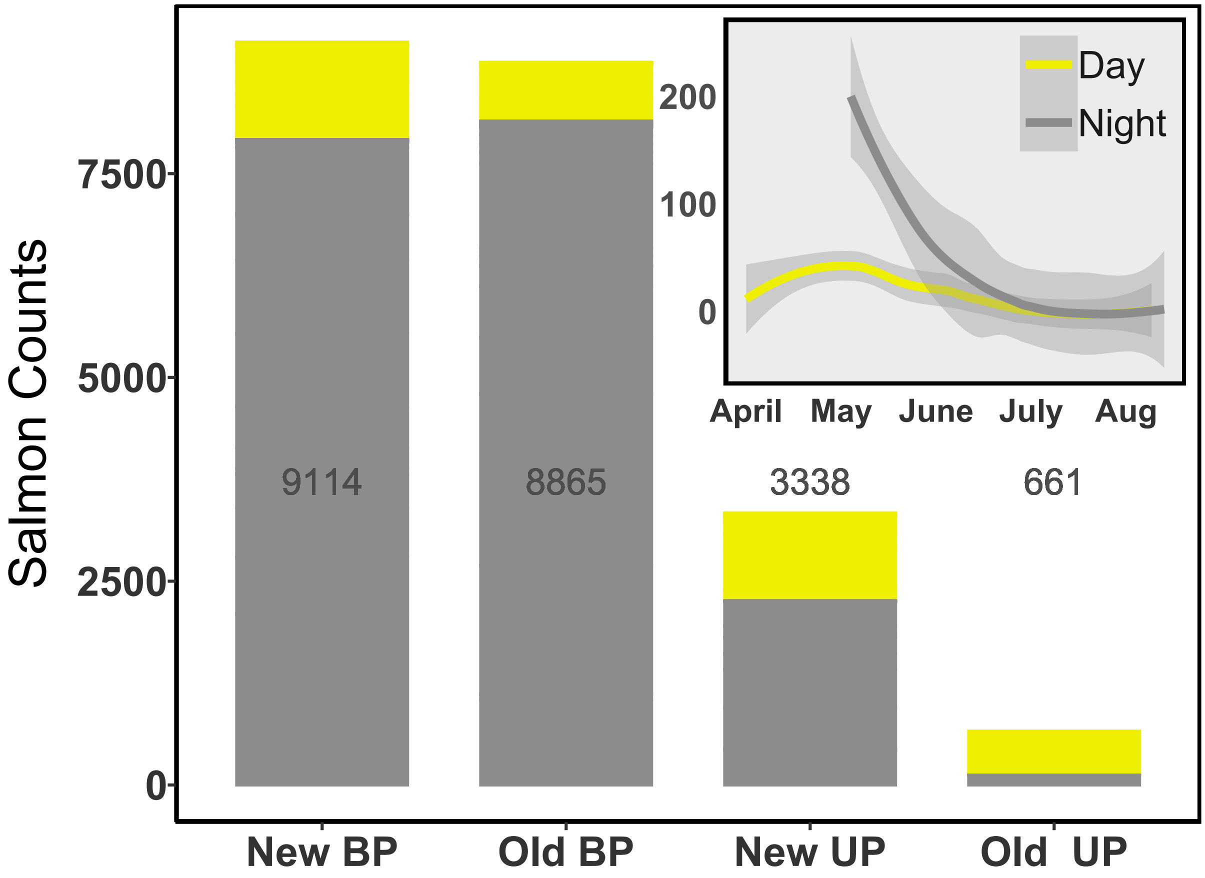

Figure 3.4. Salmon counts at seawall microhabitats and line plots of day/night salmon counts. Total counts are labelled with each bar plot. Day = yellow, night = gray. **Inset**: Day/night line plots, of salmon counts, smoothed to show overall trend. Due to logistical constraints, night sampling began one month after day sampling.

Table 3.2. All-season day/night salmon presence and counts within microhabitats, by depth. Mann Whitney tests compare salmon counts by ‘depth’ (Shallow: 3 m from shoreline; deep: 10 m from shoreline) of each nearshore transect to itself according to diel period (e.g., new shallow BP at night vs new deep BP at night). BP = Between pier; UP = Under pier.

			Day		Night		Mann Whitney p-value	
Microhabitat		Depth	Presence	Counts	Presence	Counts	Night	Day
New	BP	Shallow	3	43	11	3118	0.16	0.002
		Deep	13	1153	16	4800		
	UP	Shallow	9	989	8	1607	0.35	0.046
		Deep	2	90	5	652		
Old	BP	Shallow	4	124	5	2460	0.001	0.005
		Deep	14	597	18	5684		
	UP	Shallow	1	500	0	0	0.14	0.95
		Deep	1	40	2	121		
Reference		Shallow	4	375	9	600	0.15	0.76
		Deep	9	2020	9	1588		
Overall		Shallow	21	2031	33	7785		
		Deep	39	3900	50	12845		

Model results show that positive counts among between pier (BP) transects resemble those at reference transects, with UP salmon counts lower compared to reference transects. It should be noted that counts for old BP (*n* = 8144), new BP (*n* = 7918), and reference nearshore (*n* = 2188) sites are dissimilar, but three outlying counts during the period of peak densities at an old BP transect (*n* = 2100, *n* = 1583) and new BP (*n* = 2100), inflated overall BP counts. In old UP microhabitats, there were 2 salmon detections that summed to 121 salmon. This result differs from new UPs, where 13 salmon encounters summed to 2259, and from old BPs (transects adjacent to old UPs), where 23 encounters summed to 8144 juvenile salmon. Raw counts corroborate model results demonstrating that old UPs are suboptimal microhabitats for salmon.

To avoid model overfitting, all nearshore microhabitats incorporated both depths. This means that new UP microhabitats included the shallow intertidal corridor and deep UPs, although most detections and counts occurred in the intertidal corridor. In, new, shallow UPs, 9 daytime detections (*n* = 989) and 8 night detections (*n* = 1607) compare to new, deep UPs, with 2 daytime detections (*n* = 90) and 5 night detections (*n* = 652) (Table 3.2).

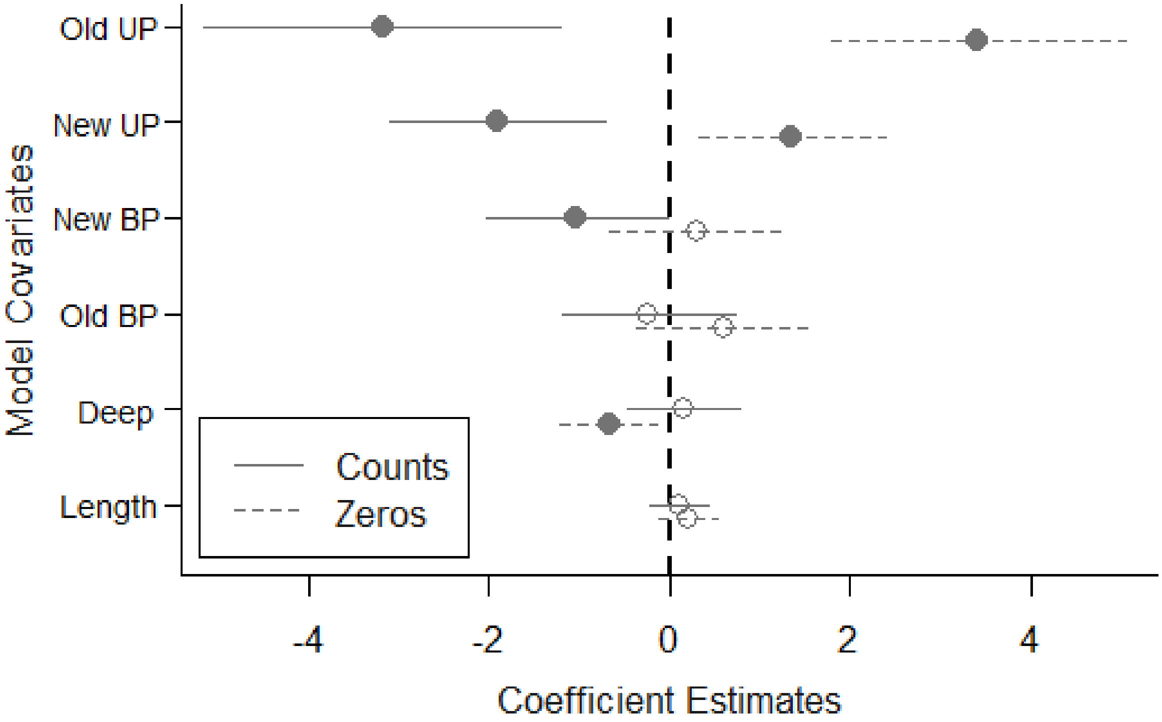

Figure 3.5. Coefficient estimates of GLMM results for the nighttime all-season nearshore model. The model compares salmon presence and counts at microhabitats to reference intercepts, which are set as Reference site transects and Shallow depth. Horizontal lines are 95 % confidence intervals. Filled circles indicate that the confidence interval width does not include zero.

3.4.5 *Pier End Microhabitats*

The pier end model equations include pier end microhabitat (outer and inner edge), site type (old and new), and month covariates (Appendix A3.3, A3.5). Juvenile salmon were 8 times more likely to be encountered at new pier ends compared to old but positive counts among pier ends were similar (Table 3.4). There is a strong, positive effect for salmon absence at inner pier edges compared to outer pier edges, where it is 18 times less likely for salmon to be present under inner pier edges compared to the open, outer edges. Model results indicate that there may be more nocturnal navigation around old pier ends compared to new pier ends, and that juvenile salmon prefer to swim around the full length of piers rather than shorten their navigation time by swimming under piers near the shoreline or along the inner edge of pier ends.

Table 3.3. Final nighttime GLMM equations characterizing salmon presence/absence and counts. The models characterize juvenile salmon 1) all-season diel absence and positive counts, 2) diel presence and positive counts during peak densities, 3) absence and positive counts among nearshore microhabitats, and 4) absence and positive counts among all pier end microhabitats. The fixed effects day/night model during peak densities is developed using R package ‘pscl’ to characterize salmon presence, with coefficient signs opposite to all other models, which are mixed effects models developed using R package ‘GLMMadaptive’.

Day versus Night: All-season	Zeros:	$absence \sim 1.76 - 0.58[night] + \ Julian\ day + \varepsilon_i$
	Counts:	$counts \sim 3.20 + 0.18[night] + \ Julian\ day + \varepsilon_i$
Day versus Night: Peak densities	Non-zeros:	$presence \sim -\ 1.55 + 0.59[night] + \varepsilon_i$
	Counts:	$counts \sim 4.72 + 1.57[night] + \varepsilon_i$
Among nearshore microhabitats	Zeros:	$absence \sim 0.47 + 1.37[New\ Pier] + \ 0.30[New\ SW] + 3.42[Old\ Pier] + 0.61[Old\ SW] - 0.67[deep\ depth] + 0.21[scaled(transect\ length)] + Julian\ day + \varepsilon_i$
	Counts:	$counts \sim 3.57 - 1.90[New\ Pier] - 1.03[New\ SW] - 3.18[Old\ Pier] - \ 0.23[Old\ SW] + 0.15[deep\ depth] + 0.10[scaled(transect\ length)] + \ Julian\ day + \varepsilon_i$
Pier ends	Zeros:	$absence \sim 0.32 \ + \ 2.88[\ Inside\ edge] + 2.04[New\ ends] + month + \varepsilon_i$
	Counts:	$counts \sim 2.48 - 0.88[Inside\ edge)] - 0.41[New\ ends] + month + \varepsilon_i$

Table 3.4. Nighttime GLMM results with estimates, standard errors, p-values, and odds ratios. The models characterize juvenile salmon 1) all-season diel absence and positive counts, 2) diel presence and positive counts during peak densities, 3) absence and positive counts among nearshore microhabitats, and 4) absence and positive counts among pier end microhabitats. The fixed effects day/night model during peak densities is developed using R package ‘pscl’ to characterize salmon presence, with coefficient signs opposite to all other models, which are mixed effects models developed using R package ‘GLMMadaptive’. Model parameters are defined in the main text. Confidence interval widths between 2.5% and 97.5% are included.

All-season: zeros	Estimate	SE	p-value	Odds Ratio
Intercept: Day	1.76	0.12	< 1e-04	
Night	-0.58	0.17	0.0006	0.56
All-season: counts				2.5 %, 97.5 %
Int: Day (*n* = 8207)	3.20	0.69	< 1e-04	1.84, 4.56
Night (*n* = 21957)	0.18	0.99	0.89	-1.77, 2.13
Peak: non – zeros				Odds Ratio
Intercept: Day	-1.55	0.20	< 1e -04	
Night	0.59	0.29	0.04	1.80
Peak: counts				
Intercept: Day (*n* = 3454)	4.72	0.22	< 1e -04	4.28, 5.16
Night (*n* = 17951)	1.57	0.32	< 1e -04	0.94, 2.18
Nearshore all-season: zeros				Odds ratio
Intercept: Ref transect, shallow	0.47	0.42	0.26	1.60
New UP	1.37	0.53	0.01	3.92
New BP	0.30	0.49	0.54	1.35
Old UP	3.42	0.84	< 1e-04	30.68
Old BP	0.61	0.50	0.22	1.83
Transect length	0.21	0.16	0.21	1.23
Depth (deep)	-0.67	0.28	0.02	0.51
Nearshore all-season: counts	Estimate	SE	p-value	2.5% - 97.5%
Intercept: Ref transect, shallow	3.57	1.30	0.006	1.02, 6.11
New UP	-1.90	0.61	0.002	-3.10, -0.71
New BP	-1.03	0.51	0.04	-2.03, -0.02
Old UP	-3.18	1.00	0.001	-5.15, -1.22
Old BP	-0.23	0.49	0.64	-1.18, 0.73
Transect length	0.10	0.17	0.53	-0.22, 0.43
Depth (deep)	0.15	0.32	0.63	-0.47, 0.78
Pier ends: zeros				Odds ratio
Int: Old sites, Outer pier edge	0.32	0.41	0.44	

New	2.04	0.84	0.01	7.69
Inside pier edge	2.88	1.09	0.008	17.84
Pier ends: counts	Estimate	SE	p-value	2.5% - 97.5%
Int: Old sites, Outer pier edge	2.48	1.13	0.03	0.27, 4.69
New	-0.41	1.23	0.74	-2.82, 2.00
Inside pier edge	-0.88	1.37	0.52	-3.57, 1.81

3.5 DISCUSSION

Although juvenile Pacific salmon distributions differ from each other at night throughout the Pacific Northwest (e.g., dispersal vs non-dispersal; Bax and Whitmus 1980; Cardwell et al. 1978), this study reveals a pattern of increased nocturnal juvenile salmon presence and counts along the Seattle waterfront compared to those observed during the day. Diel distribution differences may arise from increased seaward movement from freshwater (Ali and Hoar, 1959, Simenstad et al. 1982) and/or nocturnal onshore movement from offshore, possibly influenced by ALAN (Tabor et al. 2017). The eco-engineered seawall positively influenced night salmon distributions, similar to observed day distributions (Sawyer et al. 2020; Chapter 2, section 2.4.1), with increased nighttime presence and counts in the intertidal corridor (new, shallow UP and BP) compared to day presence and counts. Model results demonstrate that salmon use eco-engineered UP transects more similarly to reference transects than old BP transects, and more so at night.

Each sampled microhabitat incorporated shallow and deep transects. Fish detections and counts indicate that the new, shallow UP corridor positively influenced salmon presence and densities compared to the new, deep UP transects. New UP transects were the only microhabitats at night with higher juvenile salmon counts in shallow transects (i.e., the intertidal corridor) compared to deep transects, and juvenile salmon were never encountered along old, shallow UP transects at night. The data also indicates increased nocturnal use of the new, deep UP transects parallel to the intertidal corridor, which did not occur in old UP transects. The combination of fewer structures (e.g., pilings) associated with the overhead, cantilevered sidewalk and possible increased artificial light penetration through the glass blocks may have enhanced juvenile salmon habitat in the intertidal corridor. ALAN penetrating through the glass block sidewalks at night enabled kayak navigation in the intertidal corridor with no additional lighting. Ambient light navigation was not possible under old piers, which have an increased number of pilings compared to the intertidal corridor. Since juvenile chum salmon shoals do not disperse until a light intensity of 0.001 lux (Ali 1959), and chum were the principal salmon species present in this study, their eyes may adjust sufficiently to navigate in the dimly lit corridor, but this hypothesis has not been evaluated.

Old, under-pier microhabitats are suboptimal for salmon regardless of diel period and may be less preferred at night compared to swimming around pier ends. Old UPs were the only seawall microhabitats where nighttime salmon presence and abundance did not increase compared to day counts. Salmon may swim along pier edges to avoid the sharp light gradient required to enter shaded UP microhabitats from artificial-lit BP locations (Salo 1979, Simenstad et al. 1999; Ono and Simenstad 2014). This hypothesis is corroborated by overall lower juvenile salmon counts in old UPs compared to new UPs, and the eight times greater likelihood of detecting salmon at old pier ends compared to new pier ends. Collectively, the results suggest that nocturnal avoidance of suboptimal habitat causes some salmon to remain between piers at nearshore locations, while others override their proclivity for nearshore waters and navigate around piers. Navigating around piers potentially increases exposure to predation (Simenstad et al. 1999) and delays migration, which may impact both growth and survival.

An additional influence on nocturnal juvenile salmon distributions along Seattle's waterfront may be ALAN. ALAN functions as a fish attractant that can nearly halt juvenile salmon migration (Tabor et al. 2017) and increases their predation risk (Beauchamp et al. 2020). Eliminating ALAN has been shown to reduce predation on juvenile salmon (e.g., by immature pinnipeds (Yurk and Tites 2000)). The influence of ALAN is suggested by the two times higher likelihood of detecting salmon at night compared to day and Mann Whitney test results, in which BP transects subject to ALAN were the only microhabitats with significantly different day/night counts. This result is also consistent with the notable difference in day/night positive median counts observed at BP transects during the period of peak salmon densities (day: med = 100; night: med = 575) (Chapter 2, section 2.4.1). This contrast does not occur in reference sites which have limited ALAN exposure (day: med = 177.5; night: med = 164). In the presence of ALAN, salmon smolts may opportunistically forage (Fraser and Metcalfe 1997), potentially delaying migration. Although light measurements in all transects measured 0 lux, salmon in artificial lit transects exhibited some swimming behaviors consistent with daytime feeding (e.g., darting and swimming to the surface). The DIDSON consistently detected large zooplankton at night in unlit and artificial-lit locations, indicating the presence of potential prey for juvenile salmon. ALAN coinciding with increased planktonic presence could initiate foraging during a critical growing period in locations containing fewer high-quality prey due to coastal armoring (Duffy et al. 2010).

High BP salmon counts suggest habitat preference and positive phototaxic behavior, but also indicate a temporary constraint on salmon from continuing their seaward migration as they avoid extreme light gradients to minimize ocular adjustment (Ali and Hoar 1959). It is noteworthy that GLMM results indicate similar detections and counts between BP and ALAN-limited reference transects, suggesting that additional factors may contribute to increased nocturnal salmon abundances, including movement to shallow waters for real or perceived refuge, and/or increased seaward movement through Elliott Bay.

Similar to day sampling, night salmon counts were likely underestimated in BP and reference microhabitats due to increased occurrence of mixed similar-sized fish shoals during mid-season sampling (Appendix A2.5, A3.6). Mixed fish shoals consisted predominantly of surfperches with few salmon (verified by nighttime GoPro sampling), but mixed species aggregations were labelled as mixed groups rather than classified to family or species and were not included in juvenile salmon transect totals.

Fish habitat use may change between day and night (Helfman 1986, Gray et al. 1998) due to diel period, anthropogenic influences, or differing effects of habitat modifications (Davies et al. 2014). For that reason, day and night sampling can be used to identify diel fish distribution and behavioral differences (Ryer and Olla 1998), habitat use changes (Diogo and Pereira 2016), and may reveal anthropogenic impacts within urbanized juvenile fish habitats that would not be apparent in day-only sampling.

Chapter 4. SUMMARY AND CONSIDERATIONS FOR FUTURE RESEARCH

4.1 ACOUSTIC METHODOLOGY: THE DIDSON AS A MOBILE SAMPLING TOOL OF SMALL FISH

Results of this study demonstrate that the DIDSON multibeam sonar is an effective mobile sampling tool of small fishes (25 mm – 90 mm), and increases sampling efficiency, spatiotemporal knowledge of diel fish distributions, and habitat use compared to snorkel surveys. A review of the DIDSON's applicability for fish sampling concluded that acoustic cameras appear to be "the most efficient monitoring method" in narrow rivers and estuaries (Martignac et al. 2014). Acoustic salmon counts were comparable or higher than snorkel counts except in shallow BP transects, where mixed shoals of similar-sized fish were challenging to identify to species. Accuracy and precision tests have not been conducted for snorkel or mobile DIDSON sampling of salmon in the 25 mm – 90 mm length range, so counts for this study are relative abundances. Acoustic detection can change based on shoal densities (i.e., high densities can block individual targets), turbidity, and fish angle relative to the incident acoustic beam (Simmonds and MacLennan 2005). Multibeam sonar accuracy tests for adult salmon include tethering fish (Burwen et al. 2010) and constructing fishing weirs for a complete census (Holmes et al. 2006). Tethering small salmon and constructing weirs is not logistically feasible along the Seattle waterfront. However, detection accuracy and precision tests may be feasible using hatchery salmon in enumeration holding areas at Green-Duwamish hatcheries (average lengths at release time for pink and chum: 50 mm; Chinook: 80 mm; Duwamish-Green Hatcheries EIS, 2017). For this study, one reviewer examined files three times after data acquisition, but precision tests can be conducted using multiple reviewers to review the same acoustic files.

4.1.1 *DIDSON Tips to Maximize Sampling Effectivenesss*

To maximize juvenile salmon detections, the DIDSON multibeam sonar mounting angle should be carefully set to maximize ensonification of the water column section that contains the target fish species. For this study, the final camera angle was 12° from the kayak hull to avoid surface noise while ensuring that the mid- to upper portion of the water column, where salmon reside, was ensonified. During data acquisition using a mobile platform, water surface conditions should be considered, as wave action can blur acoustic images and prohibit identification of individual fish, rendering the resulting data invalid.

The multibeam sonar recorded unique events and behaviors, demonstrating its utility for behavioral studies and its ability to capture real-time events in dark and turbid conditions. As behavioral examples, juvenile salmon were recorded seeking shelter under bumps and shelves of the eco-engineered seawall and fleeing from beneath bumps and shelves to open water after being disturbed. The multibeam sonar also recorded active pier avoidance by juvenile salmon, as they fled to deep water to avoid being 'pushed' under piers by the kayak. The DIDSON also recorded a large spawning event at reference transects, where 19000 small forage fish were counted over two nights, and a potential predation event, where a herring school was recorded fleeing from a juvenile pinniped. The ability to record real-time fish behaviors in varied habitats and light availabilities is key for behavioral studies and for informing conservation decisions (Sutherland 1998, Helfman 1999).

4.2 CONSIDERATIONS FOR UNDER PIER HABITATS

The results of this study corroborate and build on findings of Toft et al. (2007, 2013), Munsch et al. (2014), and Sawyer et al. (2020), who demonstrated that traditional UPs are suboptimal microhabitats for juvenile salmon. Although GLMM results showed that salmon were more evenly distributed among microhabitats during the eight-week period of peak densities during the day, there were few detections of salmon in old UPs (*n* = 2) compared to new UP (*n* = 11), old BP (*n* = 5), new BP (*n* = 12) and reference transects (*n* = 13) and the only old UP salmon detections occurred during the period of peak densities. There is a need for more comprehensive day/night sampling during peak salmon densities.

It was 2-3 times more likely for salmon to be present in deep (i.e., 10 m from seawall/shoreline) transects compared to shallow (i.e., 3 m from seawall/shoreline), and with counts 1.5 times higher, similar to the findings of Sawyer et al. (2020). This result differs from results reported by Toft et al. (2007) and Munsch et al. (2014), who counted more salmon in shallow BP transects compared to deep. The change at new seawall sites is attributed to the presence of the overhead eco-engineered sidewalk and the associated reduced lighting that salmon may avoid during the day. Inconsistent with this hypothesis is the observation that more salmon were detected at old, deep BP transects compared to shallow BP transects even though the old, shallow transects did not have overhead sidewalks. Reasons for this inconsistency are unknown. However, while Mann Whitney test results show a nighttime salmon count difference at old BP transects, by depth, test results show no nighttime difference at new BPs between depths, indicating that salmon counts in shallow and deep transects are more similar at night due to increased presence in the intertidal corridor.

This study included characterization of juvenile salmon at pier ends along the Seattle waterfront. The key result is that significant numbers of salmon (*n* = 2231, day; *n* = 1326, night) are overriding their predisposition for the nearshore habitats that provide refuge and foraging opportunities to navigate around piers. This behavior is likely to avoid the dark environments under piers and the extreme light gradients adjacent to piers (Ono and Simenstad 2014). The highest daytime median salmon counts for detections > 5 were observed at pier ends, and the magnitude of these counts were similar to those observed in BP transects. Higher abundances in open water at pier ends demonstrate that juvenile salmon prefer to swim around rather than under piers to maximize ambient light levels, especially if it is possible to travel in deeper waters within larger shoals.

Piling spacing may be important to consider in pier construction. Salmon were only found beneath the inner edge of the end of Pier 69. Pier 69 was unique at the time of sampling because it is constructed with widely spaced concrete and metal pilings, while the remaining piers along the waterfront have densely spaced wood and concrete pilings. However, many more salmon chose to swim around the pier ends (n = 2180, day; n = 1229, night) rather than beneath the inner edge of pier ends (n = 51, day; n = 97, night), so the best construction cost/benefit investment from a juvenile salmon perspective may be to modify nearshore pier habitats sufficiently to reduce or eliminate salmon navigation around pier ends. Furthermore, a deep reference site transect at an equivalent distance from the shoreline as the nearest and northernmost pier end was sampled to determine if salmon remain in deep water after passing the final pier. This transect had only two salmon detection during the sampling season (n = 45, day; n = 1, night), which supports the hypothesis that early migrating juvenile salmon prefer nearshore, shallow waters.

Future studies should include sampling between pier ends to quantify potential salmon migration between pier ends parallel to the shoreline. Instead of following pier outlines, some salmon may conserve energy and shorten the overall migration distance by swimming between pier ends. Along the Seattle waterfront, piers are unevenly spaced, and salmon pier end navigation may differ based on pier proximity and other factors including ambient light levels, time of day, and salmon densities. Nocturnal pier end difference by site type, in which salmon were 8 times more likely to be found at old pier ends compared to new pier ends, may indicate increased nocturnal navigation around old pier ends, but this hypothesis is unverified.

An objective of the seawall modifications along the Seattle waterfront was to “create an effective intertidal migratory corridor for juvenile salmonids” (EBSP 2019). Results from this study demonstrating increased juvenile salmon presence and abundance in previously low occupied, under-pier habitats, confirmed initial post-construction observations that this objective was achieved (Sawyer et al. 2020). During the day, salmon were 36 times and 11 times less likely to be detected at old and new pier habitats compared to reference locations, demonstrating that eco-engineered pier microhabitats more closely mimic reference microhabitats compared to traditional pier microhabitats. This achievement was likely due to a combination of minimal structure, higher light levels compared to pre-construction (Sawyer et al. 2020), and an elevated seafloor in the intertidal corridor. Preliminary transect sampling under piers detected juvenile salmon aggregations beyond the 25 m transect length, so future studies should include longer under-pier transects to quantify juvenile salmon navigation under piers, especially during peak migration. It is noted that salmon were only detected in old, under-pier transects during peak migration. The entire intertidal corridor should be sampled bi-monthly as a single transect throughout the season and weekly during peak salmon densities to further resolve patterns of intertidal corridor use, including increased presence or avoidance of seawall microhabitats.

Finally, the focus of this study is salmon habitat use rather than salmon abundance, so sampling effort should focus on the period of peak salmon densities. This sampling strategy will reduce zeros and increase positive counts in the resulting data, enabling models to better characterize day/night salmon distributions by microhabitat and differentiate salmon distributions in microhabitats by depth when salmon are present.

4.3 DIEL JUVENILE SALMON DISTRIBUTIONS

This was the first acoustic nighttime study of juvenile salmon distributions along Seattle’s waterfront, and it was designed to permit a comparison to daytime salmon distributions in traditional, eco-engineered, and reference microhabitats. Throughout the sampling season, it was twice as likely to encounter salmon at night compared to day and during the eight-week peak migration, juvenile salmon counts were 1.5 times higher at night compared to day. Increased nighttime salmon presence could be explained by a combination of increased nocturnal seaward migration, increased onshore movement from deeper waters, and/or an attraction to artificial light. Increased onshore movement seems less likely given comparable day/night pier end counts and the all-season consistency of higher nighttime salmon presence. Larger juvenile salmon move offshore as the season progresses, but the increased nighttime presence compared to day occurred throughout the season. It is hypothesized that increased seaward migration occurs at night similar to riverine environments (Ali and Hoar 1959) given the 2 times higher salmon presence at night throughout the season and the proximity of the study sites to the mouth of the Duwamish waterway. Salmon may subsequently or independently become delayed by ALAN attraction. This delay is indicated by significantly higher counts at ALAN-exposed BP microhabitats and contrasts with lower counts at reference sites, which have limited ALAN exposure. During the day, salmon were 11 times less likely to be present in new UP habitats compared to reference sites and at night, salmon were 4 times less likely to be present under new piers compared to reference sites. The reduced nighttime contrast suggests that eco-engineered habitat positively influences salmon more at night than during the day.

Additional studies are needed to elucidate reasons for increased nighttime salmon presence along the waterfront. Continuous day and night monitoring of at least one site along the intertidal corridor may provide data to address this challenge. The use of a stereo camera system (e.g., Williams et al. 2014) is one technology that may aid in the recognition of differing diel behaviors (e.g., unidirectional nighttime movement). Sampling the length of the intertidal corridor more frequently may provide data that can be used to identify and differentiate day/night salmon corridor use. Paired day/night sampling can be used to ascertain changes in salmon distribution differences (e.g., daytime offshore movement) by comparing counts from night transects to those sampled the following day.

Next steps should ascertain reasons for high BP salmon counts, which support conclusions of Salo (1979), Tabor et al. (2019f), and Beauchamp et al. (2020) that ALAN influences juvenile salmon behaviors and may delay migration. Juvenile salmon vision is not adapted for low light feeding (Ali 1959, Fraser and Metcalfe 1997). If juvenile salmon are foraging nocturnally, then foraging activity is likely facilitated by ALAN. Diet sampling can be used to determine if salmon are using ALAN to forage (Bailey et al. 1975). Plankton should be sampled under and between piers to identify potential salmon prey and to determine how the species composition differs by microhabitat and exposure to ALAN. Coincident studies should quantify piscivorous bird foraging as a potential source of juvenile salmon mortality (Quinn 2018) given the documented piscivorous predation by birds on juvenile salmon in estuaries (Zamon et al. 2014, Phillips et al. 2017). Although not quantified for this study, actively feeding piscivorous birds were consistently observed adjacent to surface waters illuminated by ALAN, preying on herring and salmon shoals.

Finally, sampling in even years will enable inclusion of biennially spawning pink salmon. The regional pattern of pink salmon bi-annual migration during even-numbered years did not align with the 2019 sampling season. Given that habitat utilization differs among salmon species (Groot and Margolis 1991; Flitcroft et al. 2019), habitat preferences quantified for salmon in this study may not provide information on habitat preferences of pink salmon, which have an earlier seaward migration and shorter estuarine residence than other salmonids (Quinn 2018). For example, ESA-listed Puget Sound Chinook salmon reside in the Elliot Bay estuary for approximately four weeks during peak densities (Ruggerone and Volk 2014), so they may depend more on estuarine feeding and refuge compared to species with shorter estuarine residencies (e.g., pink salmon).

Understanding behaviors of migratory animals enables prediction of how they will respond to modified or novel environments (Sutherland 1998). Migratory fish can be an indication of the biological health of a habitat (Karr 1991), and the contiguity of their habitat, including migratory delays, may be used as metrics of habitat quality. As coastal urbanization continues, eco-engineering measures may be an avenue to simultaneously protect urban infrastructure and mitigate some ecological effects of coastline armoring.

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APPENDIX A2

Appendix A2.1: Daytime characterization of juvenile salmon among nearshore transects throughout the sampling season. Potential count distributions are compared using AIC and BIC values. Likelihood ratio tests are used to identify potential spatial or temporal random effects with R package 'glmmTMB' due to its ability to incorporate no random effect when comparing random effects in global models. R package 'GLMMadaptive' is used to develop mixed models with potential covariates and a hurdle negative binomial distribution. Models are ordered in the table by lowest AIC value. Covariates are included in zero and count portions because their influence on salmon presence and counts are of interest. Specific sites are excluded from fixed covariates due to low numbers of sites. The final model is based on likelihood ratio test values and consideration of the model's ecological relevancy.

Global Model	DF	AIC	ΔAIC	BIC	ΔBIC
Hurdle negative binomial	17	788.58	0.00	855.53	6.49
Zero-inflated negative binomial	17	790.51	1.93	857.45	8.42
Negative binomial	9	813.59	25.01	849.03	0.00
Linear	9	4505	3716.4	4540.4	3691
Zero inflated	16	10247	9458.3	10310	9461
Poisson	8	24019	23231	24051	23202
Random Effects w/ LRT tests		Chi-square	p-value		
Full model + no random effect					
Full model + (1|month)		22.27	2.4e-06		
Full model + (1|site)		0	0.9998		
Full model + (1|Julian day)		29.99	4.3e-08		
Potential Covariates (with Julian day)	DF	AIC	ΔAIC	BIC	ΔBIC
Microhabitat + light + depth + transect length	18	767.7	0.0	776.4	0.0
Microhabitat + light + depth + transect length + tide	20	771.4	3.7	781.1	4.7
Microhabitat + light + depth	16	774.1	6.4	781.9	5.5
Microhabitat + light + length	16	780.3	12.6	788.0	11.6
Microhabitat + light + trans length + tide	18	783.4	15.8	792.2	15.8
Microhabitat + light	14	785.2	17.5	791.9	15.6
Microhabitat + light + tide	16	788.3	20.6	796.0	19.6
Light	6	800.3	32.6	803.2	26.8
Microhabitat	12	895.1	127.4	902.8	126.4
Null	4	905.1	137.4	907.6	131.2

Model in R: salmon ~ microhabitat + scale (light) + depth + scale(length) + Julian day

Appendix A2.2: Likelihood Ratio Test results for models. Includes degrees of freedom, chi-square test statistics, and p-values. Final models are italicized. ‘Light’ was included as a covariate in final models due to its high influence on AIC values. ‘Site type’ was included in final pier ends model to balance comparisons to nighttime pier ends models.

Likelihood Ratio Test values for nested Nearshore models (excludes pier ends)	DF	Chisq	p-value
Salmon ~ 1	4		
Salmon ~ microhabitat	12	30.54	0.0002
Salmon ~ microhabitat + light	14	1.64	0.44
Salmon ~ microhabitat + light + length	16	8.89	0.012
Salmon ~ microhabitat + light + length + depth	*18*	*16.60*	*0.0002*
Salmon ~ microhabitat + light + length + depth + tide	20	0.25	0.88
Likelihood Ratio Test values for nested Pier ends models			
Salmon ~ 1	4		
Salmon ~ Pier edge	6	15.79	0.0004
Salmon ~ Pier edge + site type	8	1.81	0.40
Salmon ~ Pier edge + site type + light	*10*	*1.17*	*0.56*
Likelihood Ratio Test values for nested Nearshore Seawall models (excludes reference transects and pier ends)			
Salmon ~ 1	4		
Salmon ~ microhabitat	10	28.18	0.00009
Salmon ~ microhabitat + light	12	1.65	0.44
Salmon ~ microhabitat + light + depth	*14*	*16.12*	*0.0003*
Salmon ~ microhabitat + light + depth + tide	16	0.72	0.70
Likelihood Ratio Test values for nested Nearshore Seawall models during peak densities			
Salmon ~ 1	4		
Salmon ~ microhabitat	10	15.68	0.016
Salmon ~ microhabitat + light	12	4.88	0.087
Salmon ~ microhabitat + light + depth	*14*	*8.92*	*0.012*
Salmon ~ microhabitat + light + depth + tide	16	0.36	0.834

Appendix A2.3: Daytime, all-season characterization of juvenile salmon among pier end transects. Potential count distributions are compared using AIC and BIC values. Global models with random effects are compared against a global model with no random effect. Likelihood ratio tests are used to identify potential spatial or temporal random effects with R package ‘glmmTMB’ due to its ability to incorporate no random effect when comparing random effects in global models. R package ‘GLMMadaptive’ is used to develop mixed models with potential covariates and a hurdle negative binomial distribution. Models are ordered in the table by lowest AIC value. Covariates are included in zero and count portions because their influence on salmon presence and counts are of interest. Specific sites are excluded from fixed covariates due to low numbers of sites and to avoid overfitting. The final model is based on likelihood ratio tests values and consideration of the model’s ecological relevancy.

Global Modal	DF	AIC	ΔAIC	BIC	ΔBIC
Zero-inflated negative binomial	9	288.16	0.00	311.52	0.00
Hurdle negative binomial	9	288.18	0.02	311.52	0.02
Negative binomial	4	309.96	21.80	320.94	9.42
Linear	5	1128.15	839.99	1141.12	829.61
Zero inflated	8	1911.03	1622.9	1931.79	1620.28
Poisson	4	7204.71	6916.6	7215.09	6903.58

Random Effects w/ LRT tests	Chi-square	p-value
Full model + no random effect		
Full model + (1|month)	4.22	0.04
Full model + (1|site)	0	0.99
Full model + (1|Julian day)	5.06	0.02

Potential Covariates (with Julian day)	DF	AIC	ΔAIC	BIC	ΔBIC
Site type + pier edge type + light	10	283.7	0.0	288.6	0.0
Light	6	294.0	10.3	297.0	8.4
Pier edge type	6	296.7	13.0	300.5	12.0
Site type + pier edge type	8	298.4	14.7	303.5	15.0
Null	4	309.8	26.1	312.3	23.8
Site type	6	311.7	28.0	315.6	27.0

Model in R: salmon ~ site type + pier edge type + scale (light) + Julian day

Appendix A2.4: Daytime characterization of juvenile salmon among nearshore seawall transects throughout the sampling season and during peak densities. Count distribution assessments are not included in table because the datasets are a subset from Table A2.1 and follow negative binomial count distributions. Likelihood ratio tests are used to identify potential spatial or temporal random effects with R package 'glmmTMB' due to its ability to incorporate no random effect when comparing random effects in global models. R package 'GLMMadaptive' is used to develop mixed models with potential covariates with a hurdle negative binomial (HNB) distribution for the all-season nearshore seawall model. Potential models for the nearshore seawall model are ordered in the table by lowest AIC value. The global nearshore seawall model during peak densities required no random effects and R package 'pscl' is used to develop models with potential covariates. Potential models for the nearshore seawall model during peak densities are identical to the all-season nearshore seawall model and are not included. Covariates are included in zero and count portions because their influence on salmon presence and counts are of interest. Specific sites are excluded from fixed covariates due to low numbers of sites. The final model is based on likelihood ratio tests and consideration of the model's ecological relevancy.

Random Effects testing: all-season		Chi-square	p-value		
Full HNB model + no RE					
Full HNB model + (1|month)		6.93	0.008		
Full HNB model + (1|site)		0	1		
Full HNB model + (1|Julian day)		12.23	0.00046		
Potential Covariates (with Julian day)	DF	AIC	ΔAIC	BIC	ΔBIC
Microhabitat + light + depth	14	602.32	0.00	609.11	0.00
Microhabitat + light + depth + tide	16	605.60	3.28	613.62	4.25
Microhabitat + light	12	614.44	12.12	620.26	11.15
Microhabitat + light + tide	14	616.15	13.83	622.94	13.83
Light	6	631.56	29.24	634.47	25.36
Microhabitat	10	707.56	105.23	713.95	104.83
Null	4	720.29	117.97	722.85	113.74
Model in R: salmon ~ microhabitat + scale(light) + depth + Julian day					
Random Effects tests: peak densities		Chi-square	p-value		
Full HNB model + no RE					
Full HNB model + (1|month)		0	0.999		
Full HNB model + (1|site)		0	0.999		
Full HNB model + (1|Julian day)		0	0.999		
Model in R: salmon ~ microhabitat + scale(light) + depth					

Appendix A2.5: Daytime acoustic counts, by microhabitat, of fish species/groups throughout the sampling season. Surfperch groups are predominantly comprised of shiner perch.

	Unknown	Surfperch/salmon Mixed shoals	Herring	Tubesnout	Surfperch	Unknown forage fish
New BP	272	1683	0	0	1257	6
Old BP	56	413	2371	519	1226	0
New UP	98	8	0	251	95	0
Old UP	96	11	200	757	0	201
Ref	118	259	5601	0	434	280
All ends	36	99	6510	130	965	20

Appendix A2.6: Negative Binomial Hurdle Model Equations

$$f_{nb\ hurdle}(y;\pi,\mu,r)\begin{cases} f(binomial)(y=0;\ \pi) \\ [1-f(binomial)(y=0;\ \pi)] * \left(\dfrac{f_{nb}(y=0;\mu,r)}{1-f_{nb}(y=0;\ \mu,r)}\right) \end{cases}$$

$$\pi = probability\ of\ a\ positive\ count$$
$$\mu = mean\ of\ positive\ counts$$
$$r = scale\ of\ positive\ counts$$

Probability of a zero/non-zero count

$$z_i \sim Bernoulli(\pi_i)$$
$$logit(\pi) = X_d\beta_d$$

Counts

$$c_{i\ \sim} Negbin(\mu_i)$$
$$\log(\mu) = X_c\beta_c$$

APPENDIX A3

Table A3.1: Overall, all-season characterization of juvenile salmon day/night presence and count. Potential count distributions are compared using AIC and BIC values. Likelihood ratio tests are used to identify potential spatial or temporal random effects with R package ‘glmmTMB’ due to its ability to incorporate no random effect when comparing random effects in global models. R package ‘GLMMadaptive’ is used to develop mixed models with potential covariates and a hurdle negative binomial distribution. Models are ordered in the table by lowest AIC value. Covariates are included in zero and count portions because their influence on salmon presence and counts are of interest. Specific sites are excluded from fixed covariates due to low numbers of sites. The final model is based on likelihood ratio test values and consideration of the model’s ecological relevancy.

	DF	AIC	ΔAIC	BIC	ΔBIC
Hurdle negative binomial	5	2771.24	0.00	2795.51	0.00
Zero-inflated negative binomial	5	2771.25	0.01	2795.52	7.88
Negative binomial	3	2773.07	1.83	2787.63	7.89
Linear	3	12300.3	9529.06	12314.9	9527.2
Zero inflated	4	66593.5	63822.2	66612.9	63825
Poisson	2	160632.5	157861	160642	157854

Random Effects w/ Likelihood Ratio tests	Chi-square	p-value
Full model + no RE		
Full model + (1|month)	96.81	2.2e-16
Full model + (1|site)	0	0.997
Full model + (1|Julian day)	112.42	2.2e-16

Testing potential covariates	DF	AIC	ΔAIC	BIC	ΔBIC
Day/Night + Julian day	6	2660.63	0.00	2668.18	0.00
Null + Julian day	4	2668.51	7.88	2673.54	5.36

Models in R: salmon ~ (day/night) + Julian day

Table A3.2: Overall characterization of juvenile salmon presence and counts between day and night sampling during peak densities (April 23 – June 15). This is a temporal subset of the dataset in Table A3.1. Potential count distributions are compared using AIC and BIC values. Likelihood ratio tests are used to identify potential spatial or temporal random effects with R package ‘glmmTMB’ due to its ability to incorporate no random effect when comparing random effects in global models. The global day/night peak densities model required no random effects, and R package ‘pscl’ is used to develop models with potential covariates following a hurdle negative binomial distribution. Models are ordered in the table by lowest AIC value. Covariates are included in zero and count portions because their influence on salmon presence and counts are of interest. Specific sites are excluded from fixed covariates due to low numbers of sites and to avoid overfitting. The final model is based on likelihood ratio tests values and consideration of the model’s ecological relevancy.

	DF	AIC	ΔAIC	BIC	ΔBIC
Zero-inflated negative binomial	5	1131.67	0.00	1150.02	0.00
Hurdle negative binomial	5	1131.67	0.00	1150.02	0.00
Linear	3	4020.19	2888.53	4031.20	2881.2
Zero inflated	4	20379.4	19247.7	20394.1	19244.1
Negative binomial	3	78112.4	76980.7	78123.4	76973.4
Poisson	2	78147.7	77016.0	78155.0	77005.0

Random Effects w/ Likelihood Ratio tests		Chi-square	p-value
Full model + no random effect			
Full model + (1|month)		0	0.997
Full model + (1|site)		0	0.997
Full model + (1|Julian day)		0	0.997

Testing potential covariates	DF	AIC	ΔAIC	BIC	ΔBIC
Day/Night	6	1133.84	0.00	1136.75	0.00
Null	4	1146.23	12.39	1148.17	11.42

Model in R syntax: salmon ~ (day vs night)

Table A3.3: Likelihood Ratio Test results for nested nighttime models. Includes degrees of freedom, chi-square test statistics, and associated p-values. Final models are italicized. Final nested nearshore model was chosen to account for differing transect lengths.

Likelihood Ratio Test values for day/night all-season model	DF	Chisq	p-value
Salmon ~ 1			
Salmon ~ day vs night	*2*	*11.88*	*0.0026*
Likelihood Ratio Test values for day/night peak period model			
Salmon ~ 1			
Salmon ~ day vs night	*2*	*16.39*	*0.0003*
Likelihood Ratio Test values for nested nearshore model			
Salmon ~ 1			
Salmon ~ microhabitat	8	58.73	8.3e-10
Salmon ~ microhabitat + depth	10	6.03	0.049
Salmon ~ microhabitat + depth + length	*12*	*2.02*	*0.365*
Likelihood Ratio Test values for nested pier ends model			
Salmon ~ 1			
Salmon ~ Site type	2	6.68	0.035
Salmon ~ Site type + pier edge type	*4*	*13.00*	*0.002*

Table A3.4: Nighttime characterization of juvenile salmon among nearshore transects throughout the sampling season. Potential count distributions are compared using AIC and BIC values. Likelihood ratio tests are used to identify potential spatial or temporal random effects with R package ‘glmmTMB’ due to its ability to incorporate no random effect when comparing random effects in global models. R package ‘GLMMadaptive’ is used to develop mixed models with potential covariates and a hurdle negative binomial distribution. Models are ordered in the table by lowest AIC value. Covariates are included in zero and count portions because their influence on salmon presence and counts are of interest. Specific sites are excluded from fixed covariates due to low numbers of sites. The final model is based on likelihood ratio test values and consideration of the model’s ecological relevancy.

All-season nearshore					
	DF	AIC	ΔAIC	BIC	ΔBIC
Hurdle negative binomial	15	1275.91	0.00	1332.06	0.00
Zero-inflated negative binomial	15	1276.09	0.18	1332.23	11.35
Negative binomial	8	1290.77	14.85	1320.71	11.52
Linear	8	4320.91	3045.0	4350.85	3030.14
Zero inflated	14	40214.0	38938	40266.4	38945.7
Poisson	7	82342.7	81067	82368.9	81048.2

Random Effects w/ Likelihood Ratio tests	Chi-square	p-value
Full model + no RE		
Full model + (1|month)	80.3	2.2e-16
Full model + (1|site)	0	1
Full model + (1|Julian day)	67.5	2.2e-16

Potential Covariates (with ‘month’)	DF	AIC	ΔAIC	BIC	ΔBIC
Microhabitat + depth	14	1195.30	0.00	1186.71	0.00
Microhabitat + length +depth	16	1197.29	1.99	1187.47	0.76
Microhabitat	12	1197.33	2.03	1189.97	3.26
Microhabitat + length	14	1199.53	4.23	1190.94	4.23
Null	4	1240.06	44.76	1237.61	50.89

Model in R: salmon ~ microhabitat + scale(length) + depth + month

Table A3.5: Nighttime characterization of juvenile salmon among pier end transects throughout the sampling season. Potential count distributions are compared using AIC and BIC values. Likelihood ratio tests are used to identify potential spatial or temporal random effects with R package ‘glmmTMB’ due to its ability to incorporate no random effect when comparing random effects in global models. R package ‘GLMMadaptive’ is used to develop mixed models with potential covariates and a hurdle negative binomial distribution. Models are ordered in the table by lowest AIC value. Covariates are included in zero and count portions because their influence on salmon presence and counts are of interest. Specific sites are excluded from fixed covariates due to low numbers of sites and to avoid overfitting. The final model is based on likelihood ratio tests values and consideration of the model’s ecological relevancy.

All-season pier ends					
	DF	AIC	ΔAIC	AICc	BIC
Hurdle negative binomial	9	177.9	0.0	180.2	200.2
Zero-inflated negative binomial	9	178.0	0.1	180.3	200.3
Negative binomial	5	188.4	10.5	189.2	200.8
Linear	5	955.1	777.2	955.8	967.5
Zero inflated	8	1525.7	1347.8	1527.5	1545.5
Poisson	4	2822.5	2644.6	2823.0	2832.4

Random Effects w/ Likelihood Ratio tests	Chi-square	p-value
full model + no random effect		
full model + (1|month)	3.58	0.05
full model + (1|site)		1
full model + (1|Julian day)		1

Potential Covariates (with ‘month’)	DF	AIC	ΔAIC	BIC	ΔBIC
Site type + pier edge type	8	184.75	0.00	179.84	0.00
Pier edge type	6	188.66	3.91	184.98	5.14
Site type	6	193.74	9.00	190.06	10.23
Null	4	196.42	11.68	193.97	14.13

Model in R: salmon ~ site type + pier edge type + month

Appendix A3.6: Nighttime acoustic counts, by microhabitat, of fish species/groups throughout the sampling season. Surfperch groups are predominantly comprised of shiner perch.

	Unknown fish	Surfperch/salmon Mixed shoals	Herring	Tubesnout	Surfperch	Unknown forage fish
New BP	71	2471	0	0	2668	10
Old BP	188	2541	2901	2540	3128	1393
New UP	143	850	0	40	312	50
Old UP	439	107	576	964	26	0
Reference	76	2923	300	0	807	28652
All ends	95	253	550	25	182	0