



Acoustic characterization of juvenile Pacific salmon distributions along an eco-engineered seawall

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ABSTRACT: A mobile dual-frequency identification sonar (DIDSON) was used to characterize juvenile Pacific salmon (*Oncorhynchus* spp.) daytime use of armored and eco-engineered seawall habitats along an urbanized shoreline in Puget Sound, Washington, USA. Eco-engineering included intertidal benches to elevate the seafloor, a textured seawall to provide refuge and encourage invertebrate colonization, and glass blocks in an overhanging sidewalk to increase ambient light. A DIDSON multibeam sonar gave salmon counts twice that of visual surveys, and was thus deemed effective as a mobile sampling tool for small fish (~40–90 mm) and can be advantageous relative to visual methods depending on research goals, habitats, and ambient light levels. Increased salmon presence in the eco-engineered intertidal corridor relative to traditional seawall and pier habitats showed that the combination of increased light, reduced infrastructure (e.g. pier pilings), increased texture, and a shallower seafloor improves habitat function for juvenile salmon. High densities of juvenile salmon along pier ends show that salmon also use alternative migration pathways, with unknown energetic costs and predations risks.

KEY WORDS: Acoustic · Juvenile salmon · Eco-engineering · Pier · Seawall · Shoreline armoring · Dual-frequency identification sonar · DIDSON

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1. INTRODUCTION

Armoring (i.e. physical structures) and overwater structures associated with coastal urbanization (e.g. seawalls, piers) protect urban infrastructure yet impact nearshore marine environments. The vertical, smooth surfaces of traditional seawalls have less space, texture, and refuge compared to natural shoreline (Klein et al. 2011, Heerhartz et al. 2014, Dethier et al. 2016), with reduced shallow-water habitat displacing smaller fish to deeper waters containing predators (Munsch et al. 2015). Shade from piers reduces plant and algal growth, invertebrate prey abundance and richness (Simenstad et al. 1999, Blockley 2007, Cordell et al. 2017), as well as juvenile and adult fish diversity (Duffy-Anderson & Able 1999, Munsch et al. 2015, Grothues et al. 2016). Armoring and overwater structures fragment nearshore habitats, altering juvenile fish migration abilities (Lichtatowich et al. 1995)

and impacting feeding and refuge opportunities for anadromous species (Simenstad et al. 1982, Toft et al. 2018, Flitcroft et al. 2019). These disruptions all occur during a critical life history stage characterized by high growth, predation, and osmoregulatory acclimatization to marine waters (Healey 1982, Quinn 2018).

The Seattle waterfront within Elliott Bay in Washington, USA, is a case study of an urban, altered habitat. The waterfront has over 3 km of continuous seawall to protect infrastructure and numerous piers, with some extending 160 m into the bay. Despite being 90% armored, the nearshore waterfront remains a migration corridor for juvenile Pacific salmon (Simenstad et al. 1982). Three nearshore-dependent species, namely US Endangered Species Act-listed Puget Sound chinook *Oncorhynchus tshawytscha*, pink *O. gorbuscha* (in even-numbered years), and chum salmon *O. keta* occupy Seattle's waterfront during their spring/summer seaward migration (Toft et al. 2007).

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After a 2001 regional earthquake damaged the integrity of the Elliott Bay seawall (Ott 2014), replaced seawall sections were ecologically engineered to mitigate habitat degradation (Browne & Chapman 2011, Firth et al. 2014, Morris et al. 2017) and restore some ecosystem function to benefit juvenile salmon. The new seawall habitat includes elevated marine mattresses to create intertidal benches, a textured seawall with shelves to increase surface rugosity, and an overhanging sidewalk with embedded glass blocks to compensate for light obstruction by the sidewalk (Fig. 1; Cordell et al. 2017). Marine mattresses are rock-filled mesh bags placed next to the seawall to increase shallow water habitat and to provide substrate for algae and invertebrates to colonize. The new seawall increases texture with the addition of bumps, crevices, and extended shelves, intended to encourage primary productivity and colonization by invertebrates, and to provide refuge for small fish. Glass blocks in the overhanging sidewalk increase light levels to visually aid salmon migration and foraging. These modifications in aggregate are intended to create a shallow, continuous corridor for juvenile salmon along the waterfront with invertebrates as a food source.

Since 2003, daytime snorkel surveys along Seattle's waterfront have periodically estimated juvenile salmon abundance, distributions, foraging, and schooling behaviors (Toft et al. 2007, Munsch et al. 2014, Sawyer et al. 2020). Snorkel surveys enable fish species identification and observation of behavior (e.g. feeding) in real time but can be limited by water turbidity, ambient light levels, and time in water by snorkelers. An alternative to visual surveys is active

acoustics, which are unaffected by time in water, ambient light, and water clarity (Maxwell & Gove 2007, Foote 2009). An appropriate acoustic technology to survey nearshore epipelagic fish communities is the dual-frequency identification sonar (DIDSON) (Sound Metrics). The DIDSON is a multibeam sonar that transmits sound pulses, with returning echoes digitized and converted to near video-quality images (i.e. acoustic camera). The volume sampled by an acoustic camera increases efficiency and spatiotemporal sampling in narrow rivers and estuaries relative to visual surveys (Martignac et al. 2015). The high operating frequency and high data resolution of the acoustic camera enables an analyst to identify fish and 'see' how fish are utilizing habitats in real time. Data can be reviewed later at increased or decreased frame rates, maximizing efficiency of data processing. Multiple distance ranges can be viewed during playback, which permits comparison of fish activity at 2 ranges at the same time (e.g. adjacent to and under a pier). Species identification and fish length estimations (cf. Burwen et al. 2010) are challenging in acoustics-based sampling. However, knowledge of migration behaviors, fish size, morphology, and swimming behaviors are used to differentiate species (Baumgartner et al. 2006, Doebling et al. 2011, Martignac et al. 2015). Within Elliott Bay, knowledge of fish assemblages, differing morphological and behavioral characteristics, and spawn and migration times aid in species identification.

In this study, we used a DIDSON multibeam sonar to test the efficacy of mobile sampling for juvenile salmon (40–90 mm) and to quantify daytime salmon



Fig. 1. Seattle seawall, with enhancements, before and after reconstruction. (A) Original seawall. (B) Re-constructed seawall at identical location, shown at an uncommonly low tide. Enhancements for juvenile salmon include marine mattresses (MM) to increase seafloor elevation, a textured seawall and shelves (TS) for invertebrate colonization and fish refuge, and overhanging sidewalk with glass blocks (GB) for increased ambient light

distributions among eco-engineered, traditional armored, and reference habitats. Although uncommon, acoustic cameras have been used to census small fish (Adams et al. 2015, Kock et al. 2016) and have been deployed on a mobile sampling platform to identify fish assemblages along nearshore environments (Able et al. 2014, Grothues et al. 2016). The DIDSON has also been used to monitor fish under piers (Able et al. 2014, Grothues et al. 2016), investigate predator-prey interactions (Price et al. 2013), and characterize distributions of juvenile fish (Doehring et al. 2011, Smith et al. 2021). Our study objectives were to compare snorkel and acoustic census methods to detect and enumerate juvenile fish populations along an urban coastline, quantify the relative importance of factors influencing urban habitat use by juvenile salmon, and evaluate the efficacy of eco-engineered seawall habitat to mitigate coastal urbanization impacts on juvenile salmon.

2. MATERIALS AND METHODS

2.1. Study location

The study was located along the Seattle waterfront in Elliott Bay, Puget Sound, Washington, USA. Eight sites along the waterfront were monitored through

the juvenile salmon migration, April to August 2019 (Fig. 2). Three sites ('new') contained eco-engineered seawall habitat. Three sites ('old') contained original seawall habitat. Each new and old site included an under-pier section and an open seawall section adjacent to a pier, while 2 of the 3 new and old sites included pier-end transects. Two reference sites ('reference') were chosen to represent habitats along the waterfront that had been previously restored (Toft et al. 2013) and have been colonized by aquatic biota for over a decade (Table 1). The first was along a manufactured pocket beach. The second was adjacent to an open, armored shoreline where an intertidal habitat bench had been previously created.

2.2. Sample design

Nearshore seawall sites contained 25 m strip transects that were identified using site type ('old' and 'new') and overhead structure type (between pier: 'BP'; and under pier: 'UP'). The 4 nearshore transects included old BP, new BP, old UP, and new UP seawall sites. Reference sites included 35 m transects along the entire shoreline of the pocket beach, 25 m transects along the habitat bench, and a 25 m transect ~160 m offshore parallel to pier ends. All nearshore habitats were further differentiated using relative wa-

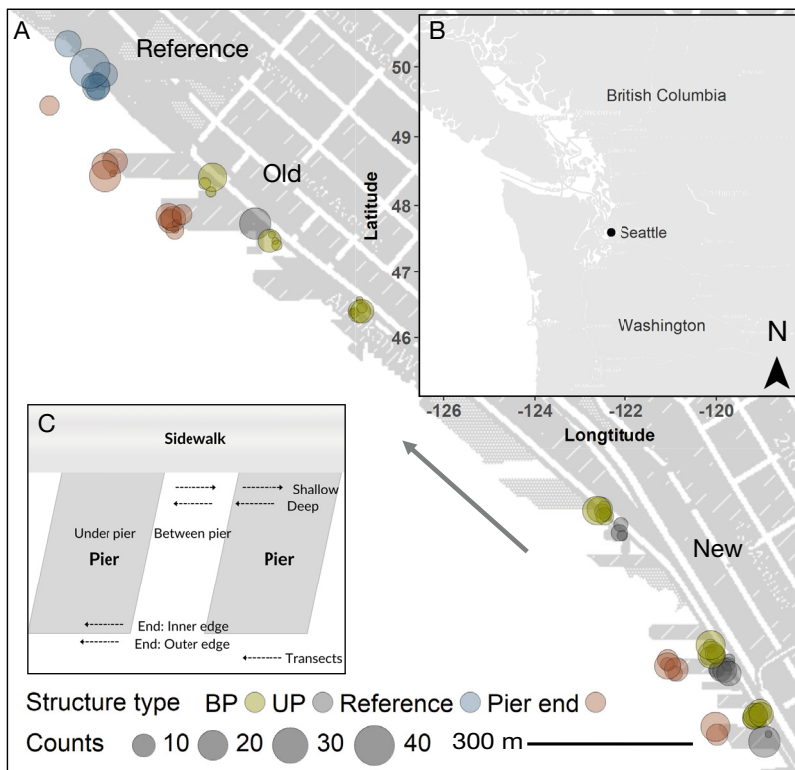


Fig. 2. Location of acoustic sampling (A) along the Seattle, WA, waterfront in (B) the Pacific Northwest, USA. (C) Seawall transects. Transects are at reference, old, and new sites. Circles are size proportional to the square root of salmon counts and are color-sorted by structure type. Counts are minimally staggered to reduce overlap. Arrow indicates the primary migration path of juvenile salmon along the Seattle waterfront. BP: between pier; UP: under pier. Map source: Google, ©2021. Map tiles by Stamen Design, under Creative Commons BY 3.0. Data by OpenStreetMap, under Open Database License

Table 1. Microhabitats defined by site type (old, new, reference) and overhead structure type (between pier [BP], under pier [UP], or reference transects); and by pier end site type (old, new) and edge type (outer, inner). Depth is defined by the distance to seawall or shoreline

Site type	Definition		
New	Seawall constructed in 2017 with textured wall and shelves, raised seafloor using 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		
BP	Adjacent to seawall with no overhead pier		
UP	Adjacent to seawall with overhead pier		
Reference	Open water with no nearby structures (reference sites only)		
Pier end – outer edge	90–160 m from seawall, dependent on pier; no overhead structure		
Pier end – inner edge	Inside edge of/shoreward of outermost piling row; under pier decking		
Microhabitats	New	Old	Reference
	BP	BP	Beach
	UP	UP	Bench
	End – outer	End – outer	End
	End – inner	End – inner	
Depth			
Shallow	3 m from seawall or shoreline; at new sites, above marine mattresses		
Deep	10 m from seawall or shoreline; at new sites, above open water		

ter depth, defined by distance to the shoreline or seawall (3 m from shoreline/seawall = 'shallow' and 10 m from shoreline/seawall = 'deep'). Although depths varied by tidal state, assigning depth categories implied relative depth and differentiated the eco-engineered corridor, which was contained within shallow depths at new BP and UP transects. Shallow transects at new seawall sites were located above marine mattresses and below the overhanging sidewalk while deep transects were located above open water and below pier decking. Four of the 6 seawall sites (2 new, 2 old) included 2 additional 25 m transects at 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 resulted in a total of 11 'microhabitats' within new, old, and reference sites (Table 1).

2.3. Data acquisition

In 2019, daytime snorkel and acoustic survey data were collected at paired reference and new transects, during high and low tides, to facilitate a comparison of survey methods. Acoustic and snorkel sampling occurred in random order predominantly within 1 wk of each other. We assumed that the same pulse of migrating juvenile salmonids was sampled by snorkel and acoustic methods based on wild chinook salmon residence time in Elliott Bay pier habitats (Ruggerone & Volk 2014). Before snorkel sampling, underwater visibility was measured with a Secchi disc. A minimum of 2.5 m visibility was required to conduct a snorkel survey. During sampling, 2 snorkelers swam parallel along shallow and deep transects (Toft et al. 2007, Sawyer et al. 2020). Each snorkeler carried a GoPro camera to record fish encounters along transect surveys, which were later used to supplement fish identification.

A DIDSON acoustic camera was used to quantify fish presence and counts along new and reference transects, which were matched to snorkel transects, and along old seawall transects, which were not sampled by snorkelers. The DIDSON was mounted beneath the center of a 3 m kayak (Fig. 3), 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 approximate to the snorkeler's field of view in non-turbid conditions. With a 14° horizontal and 29° vertical field of view, the DIDSON was tilted 12° up to ensure that the middle to upper portion of the water column that juvenile salmon occupy was ensonified. The operating frequency was set at 1.8 MHz at a sampling rate of 8–12 Hz. The DIDSON continuously sampled a 5 m range starting at 1 m in front of the kayak to a maximum distance of 6 m in front of the kayak. The maximum distance was set to maximize acquisition rate, image clarity, and to minimize fish flight responses in the data. The DIDSON transceiver was connected to a waterproof laptop running data acquisition software (DIDSON V5-26). A time-stamped file was created for each transect.

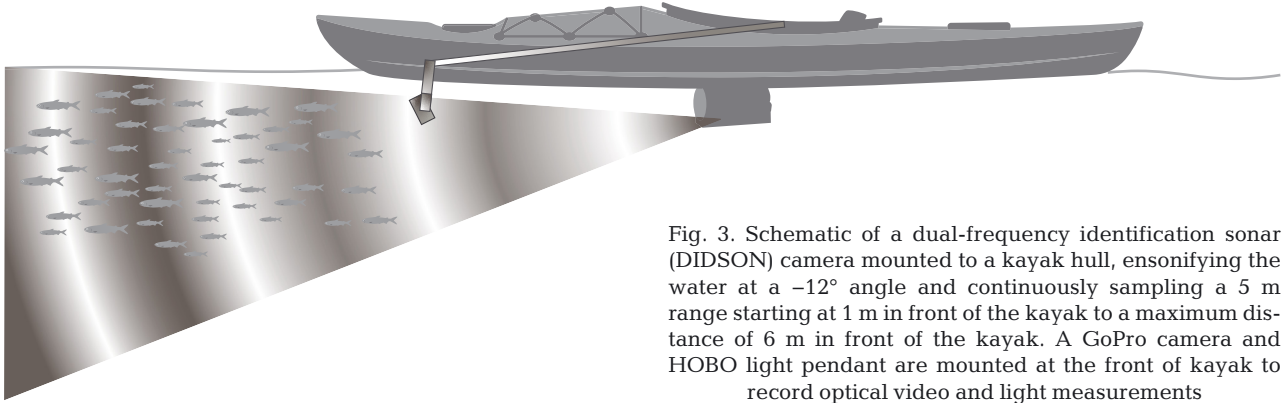


Fig. 3. Schematic of a dual-frequency identification sonar (DIDSON) camera mounted to a kayak hull, ensonifying the water at a -12° angle and continuously sampling a 5 m range starting at 1 m in front of the kayak to a maximum distance of 6 m in front of the kayak. A GoPro camera and HOBOT light pendant are mounted at the front of kayak to record optical video and light measurements

A HOBOT pendant temperature/light data logger (UA-002-08, Onset), mounted to a PVC pipe and deployed off the front of the kayak below the water surface, measured relative luminous flux (lux, 0–320 000 lux) and temperature ($^\circ\text{C}$) every 10 s. A GoPro Hero 7 camera mounted next to the HOBOT data logger and facing forward provided a simultaneous optic record to aid in fish species identification when water clarity permitted.

2.4. Data processing

Fish presence (i.e. detections), counts, and behaviors were recorded for snorkel and acoustic surveys. Fish lengths and feeding behaviors were recorded *in situ* on waterproof paper during snorkel surveys. To process acoustic data, DIDSON data files were played 3 times to count and record fish, verify counts, and review fish identification and counts using coincident GoPro video footage (Fig. 4). Fish that were encountered in-transect and swam into a subsequent transect were counted only in the first transect. Low fish species diversity coupled with staggered migration times, different morphological characteristics, and behaviors (e.g. schooling, swimming) facilitated identification of individual and aggregations of observed fish to genus or species. Fish identifications were made to lowest possible taxonomic group, but if species identification was not possible, fish were labeled 'unknown' or designated as a non-labeled group. Salmon species were grouped to genus, as species identification was not possible when only using acoustic data. Mixed species aggregations that could not be identified to taxon with the aid of GoPro images were labelled unknown or mixed aggregations and resulted in salmon counts set to zero for these occurrences. Over the sampling season, juvenile salmon mean fork length was estimated by

snorkelers to range from 65 to 90 mm. Median light measurements were calculated by microhabitat, and within nearshore microhabitats, by depth.

2.5. Data analysis

2.5.1. Characterization of acoustic and snorkel salmon sampling

Juvenile salmon acoustic and snorkel presence ($n > 0$) and counts were summed for each transect. Acoustic to snorkel count ratios for all counts were calculated by site type and within seawall sites, by structure type (BP, UP). Likelihood ratio tests identified overdispersion in count data. Generalized linear models (GLMs) were used to characterize differences in salmon counts between snorkel and acoustic sampling, dependent on model covariates, which included site type (reference, new) and overhead structure type (BP, UP) for seawall transects. Comparable snorkel/acoustic samples were too small to account for additional covariates like light and depth. Statistical analyses were conducted using R version 3.6.1 (R Core Team 2019) and the GLM was developed using the R package 'MASS' (Venables & Ripley 2002).

2.5.2. Characterizing acoustic salmon presence and distributions among microhabitats

Generalized linear mixed models (GLMMs) were developed to characterize salmon presence and abundance along the Seattle shoreline. Dispersion tests, Akaike's information criterion (AIC; Akaike 1974) metric values, and Bayesian information criterion (BIC; Schwarz 1978) metric values were used to evaluate how theoretical distribution functions (e.g. Poisson, negative binomial) fit the count data.

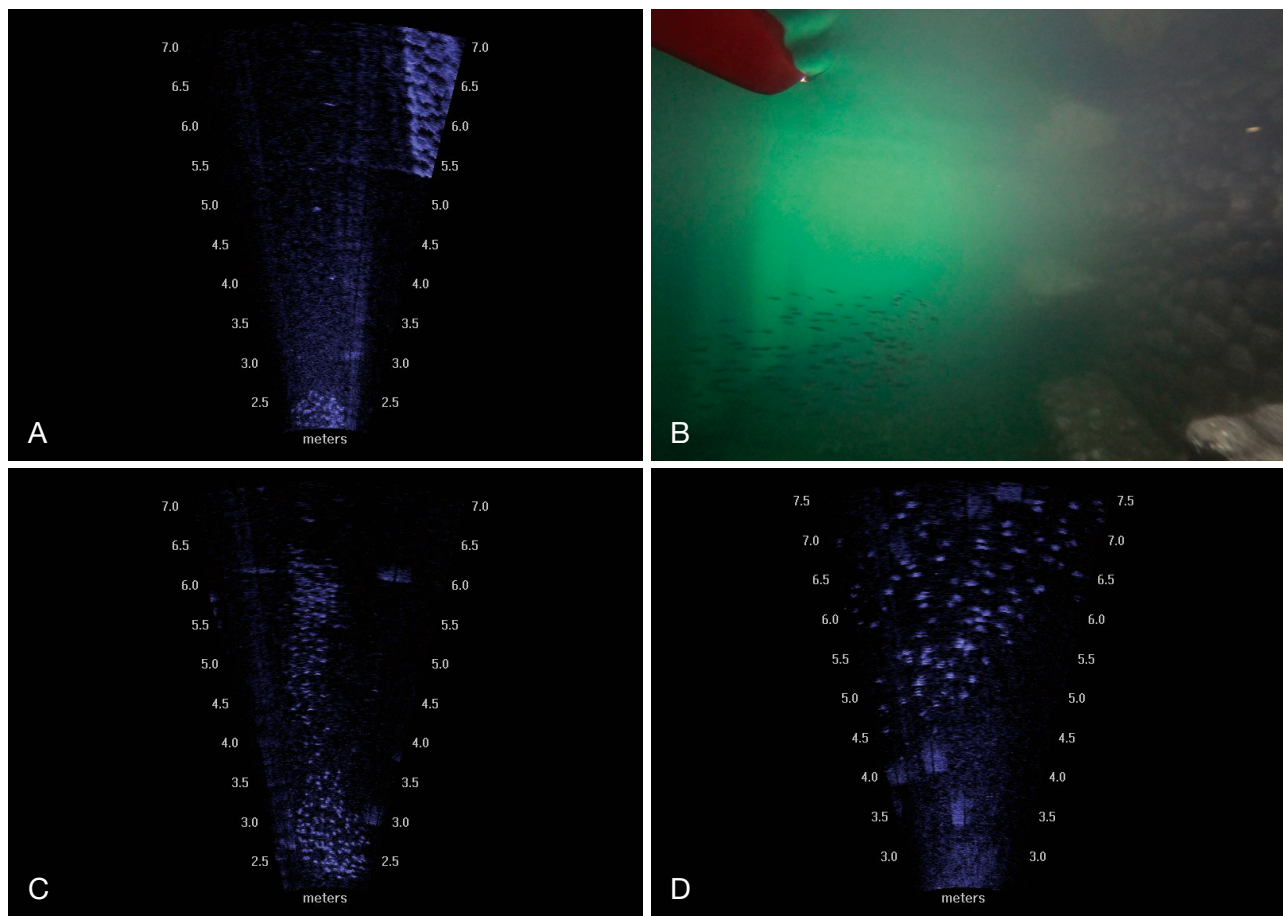


Fig. 4. Acoustic and GoPro images of juvenile fish. (A,B) Simultaneous acoustic and GoPro images of juvenile chum salmon (estimated 40 mm fork lengths) adjacent to the new seawall in the intertidal corridor. A textured seawall shelf is visible in the images. (C) Juvenile salmon shoaling in the intertidal corridor. (D) Shiner perch along an outer pier end

GLMMs with random effects including 'month', 'day of the year', and 'site', and 'no random effect' were built using the R package 'glmmTMB' v. 1.0.2.1 (Brooks et al. 2017) due to its capability to include a null random effect. Likelihood ratio tests compared models to identify potential spatiotemporal effects due to repeated measures at sites over time. 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 (m) (to account for a 35 m reference site transect), water temperature ($^{\circ}\text{C}$), and light (lx). Light values ranging from 0 to 62 000 lux were centered prior to analysis to reduce the orders of magnitude difference among values. To avoid model overfitting, depth was included as a covariate instead of being assigned to each microhabitat. Final models were built using the R package 'GLMMadaptive' v. 0.7-15 (Rizopoulos 2020) and were chosen by

considering likelihood ratio test values and AIC values, and by accounting for the ecological relevance of each model.

3. RESULTS

3.1. Acoustic salmon counts

During the juvenile salmon sampling season (2 April to 9 August 2019), 536 acoustic transects were sampled, predominantly on a weekly basis, and used in analyses. Sampling encompassed most of the juvenile salmon migration period (Toft et al. 2007), and while salmon were detected less frequently at the beginning and end of the sampling season, they were present throughout the sampling period. A total of 8207 juvenile salmon were identified and counted in the data. By site type, salmon counts were highest

Table 2. Acoustic sampling presence and counts of juvenile salmon among microhabitats, by depth. Included are transects sampled, number of detections ('presence'), counts, and non-zero median counts (presence $n > 5$) by microhabitat (includes both depths to increase sample size). Medians not calculated for presence $n \leq 5$ are represented by '–'

Microhabitat	Depth	Sample	Presence	Counts	Median
New	Between pier	Shallow	46	3	43
		Deep	40	13	1153
New	Under pier	Shallow	46	9	989
		Deep	40	2	90
Old	Between pier	Shallow	45	4	124
		Deep	42	14	597
Old	Under pier	Shallow	47	1	500
		Deep	41	1	40
Reference		Shallow	30	4	375
		Deep	30	9	2020
		End	14	1	45
New	Outer edge		26	6	808
New	Inner edge		23	0	0
Old	Outer edge		33	10	1372
Old	Inner edge		33	2	51
Overall	Shallow		21	2031	
	Deep		39	3900	

at new seawall sites ($n = 3083$), followed by old seawall sites ($n = 2684$) and reference sites ($n = 2440$). By structure type, salmon counts were highest at pier ends ($n = 2231$) followed by nearshore BP transects ($n = 1917$), and nearshore UP transects ($n = 1619$) (Table 2, Fig. 5).

3.2. Acoustic and snorkel salmon count comparisons

Acoustic salmon counts were typically more than double snorkel counts at the same transect locations. Total acoustic salmon counts for comparable transects were 4670 compared to 1908 for snorkel surveys (Table 3). The ratio between acoustic and snorkel salmon counts was not consistent among site types. As an example, reference transect counts had a higher ratio (6.24) than new seawall transects (1.49). 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, apart from UP transects, which are low-light environments. Acoustic salmon counts were likely underestimated in all BP and reference transects due to a mid-season mixing of similar-sized species, which increased the

numbers of mixed species groups (e.g. shiner perch and salmon) and reduced salmon presence and counts, as salmon in these aggregations were not added to transect counts (Table A1 in the Appendix). As a result, BP and reference habitats may be more preferred salmon habitats than study results indicate.

Snorkel and acoustic count data followed a negative binomial distribution. Final generalized linear regression models comparing acoustic and snorkel counts incorporated a negative binomial distribution (log link) and included site type and overhead structure type within new seawall sites. Acoustic counts were set as the baseline and compared to snorkel counts. Coefficient estimates for snorkel counts were generally lower than acoustic counts overall (estimate = -0.96 ; SE = 0.33), including at reference sites (estimate = -1.25 ; SE = 0.59).

The strongest difference between the 2 methods was at low-light, UP transects (estimate = -2.42 ; SE = 0.65), where snorkel counts under piers were lower compared to acoustic counts. There was minimal snorkel/acoustic count difference at BP transects (estimate = -0.45 ; SE = 0.49) (Table 3).

3.3. Light comparisons among microhabitats

As expected, daytime microhabitats with no overhead structure had the highest median light values, while UP transects had the lowest median light measurements (Fig. 5). New, shallow BP transects (i.e. the intertidal corridor between piers) had lower light values compared to old, shallow BP transects, likely owing to the glass-block overhead sidewalk that blocked a portion of sunlight to the water below, depending on time of day, cloud cover, and date (see Fig. 1; Sawyer et al. 2020). Old, deep UP transects had higher median light values than other UP habitats, including the intertidal corridor, but this did not correspond to higher salmon counts.

3.4. GoPro identification of salmon

GoPro video supplemented fish species identification with 26 encounters that summed to 1652

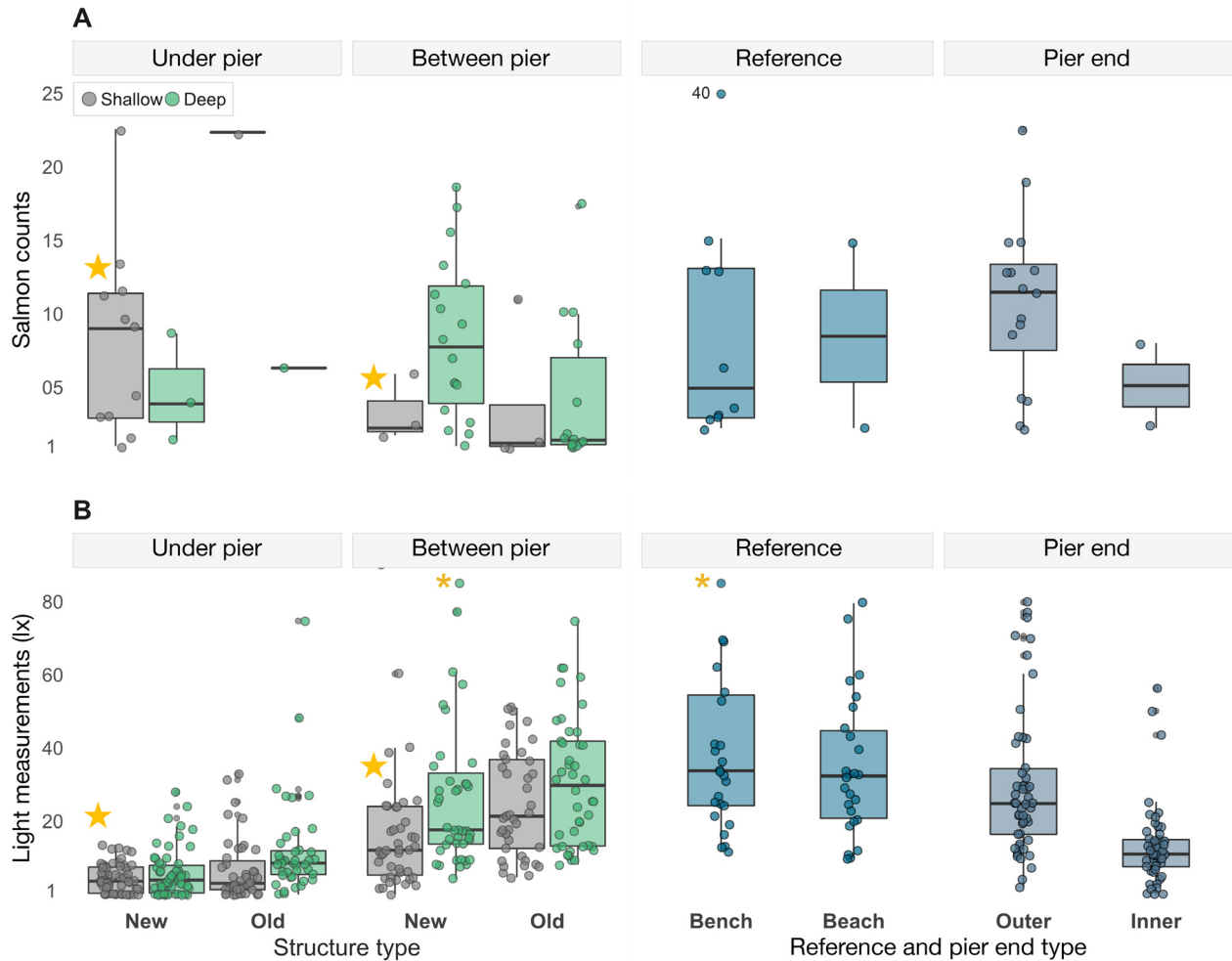


Fig. 5. (A) Non-zero salmon counts and (B) light measurements among microhabitats and within seawall transects, by depth. Salmon counts and light measurements are square root transformed to reduce data spread and points are staggered to minimize overlap. Light measurements were collected every 10 s for each sampling event. Boxplots depict median count values within the box and display maximum and minimum count values with vertical lines. Stars indicate the eco-engineered intertidal corridor (new, shallow under-pier and beside-pier transects). Asterisks indicate high light values (square-root lx): between pier ($n = 8$; $\max = 159$); Reference ($n = 3$; $\max = 250$). A high salmon count is labeled

Table 3. Acoustic and snorkel salmon presence and count comparisons, count ratios, and negative binomial generalized linear model results with estimates and standard errors. Models compare acoustic and snorkel juvenile salmon counts, with acoustic counts set as the baseline. The models were developed using R package 'MASS' (Venables & Ripley 2002). Model parameters are defined in Section 3.2. BP: between pier; UP: under pier

	— Presence —		— Counts —		Acoustic:snorkel Count ratio	Estimate	SE
	Acoustic	Snorkel	Acoustic	Snorkel			
Total	40	61	4670	1908	5:2	-0.96	0.33
Reference	13	21	2395	384	6:1	-1.25	0.59
New sites	27	40	2275	1524	3:2	-0.72	0.39
BP	16	33	1196	1463	1:1	-0.45	0.49
UP	11	7	1079	61	18:1	-2.42 ^a	0.65

^aSmall sample size for regression analysis comparing snorkel/acoustic UP salmon counts

salmon. As a comparison, the acoustic camera detected salmon 79 times for a total of 8207 salmon. The GoPro detected only 0.2% of the salmon that were missed by the acoustic camera, with 10 detections that summed to 20 salmon. In nearly all GoPro-only sightings, the salmon were close to the kayak and likely undetectable by the acoustic camera (e.g. Frias-Torres & Luo 2009).

3.5. Acoustic GLMM results

All acoustic count data were best fit with a negative binomial distribution. Testing for excess zeros (i.e. more than would be expected under a Poisson or negative binomial distribution) demonstrated an underfitting of zeros by a Poisson or negative binomial distribution. Inconsistency of salmon presence in transects (i.e. frequent zeros), consideration of AIC and BIC values, and model assumptions demonstrated that the most appropriate structure for all models was a hurdle model (logit link) with a negative binomial count distribution (log link). The resulting structure for GLMMs was specified as:

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

where π = the probability of a zero (i.e. absence) for models. An increase in a coefficient value signifies a higher likelihood of an absence. 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 the model because understanding their effect on presence and abundance of juvenile salmon was of interest. The full model equation and covariate testing are presented in the Supplement (www.int-res.com/articles/suppl/m682p207_supp.pdf; see Equation S1 and Table S1). Final models addressed temporal autocorrelation due to repeated measures by including temporal random intercepts (Bolker et al. 2009). Individual sites were excluded as random effects due to the low number ($n = 8$) of sampling sites and high AIC/BIC values.

3.5.1. Nearshore microhabitats

The nearshore regression model was developed to compare juvenile salmon presence and abundance among nearshore habitats. Coefficient values for

nearshore reference transects and shallow depth transects were estimated by the intercept, and the remaining coefficient values represent the difference between each microhabitat and reference transects and the difference between deep and shallow depths. Reference sites were used as a baseline as they represent the most natural habitat along the waterfront. The final GLMM included fixed effects of microhabitat, light, depth, transect length, and the random effect of day of the year (Tables A2 & A3).

The odds of salmon being absent was 36 times higher at old UP microhabitats compared to reference microhabitats and 11 times higher at new UP microhabitats (Table 4). This can be interpreted as salmon being 36 and 11 times less likely to be present at old and new UP transects compared to reference transects, respectively. Salmon were 3 times more likely to be absent in BP transects compared to reference transects and 3 times more likely to be

Table 4. Daytime generalized linear mixed model results with estimates, standard errors, p-values, odds ratios, and confidence intervals. The models characterize juvenile salmon absence and positive counts among all nearshore microhabitats and absence and positive counts among all pier end microhabitats. Model parameters are defined in Section 3.5. Confidence interval widths between 2.5 and 97.5 % are included. BP: between pier; UP: under pier

	Estimate	SE	p	Odds ratio/CI
Nearshore all-season: absence				
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	$<1 \times 10^{-4}$	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				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				
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
Light	-0.80	0.89	0.37	-2.54, 0.95
Depth (Deep)	1.73	0.66	0.009	0.44, 3.02

present in deep transects compared to shallow. Positive counts among all microhabitats resembled each other, indicating that although salmon were less frequent in some microhabitats, they aggregated in comparable numbers among microhabitats.

3.5.2. Pier end microhabitats

The pier ends regression model was developed to compare juvenile salmon presence and abundance 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 and the inner edge that has overhead decking. Coefficient values for 'outer edge' and 'old' pier end values were estimated by the intercept, and the remaining coefficient values measured the differences between edge types and between overall site type (new, old).

The final pier end model included pier end microhabitat (outer edge and inner edge), site type (old and new), light, and day of the year (Tables A3 & A4). Salmon were equally likely to be detected between old and new pier ends. There was a strong effect for salmon absence and lower counts at inner pier edges 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 outer edges (Table 4). Juvenile salmon preferred to swim in open water around the full perimeter of the pier rather than under pier ends, even though ambient light levels under the inner edge of pier ends were higher than those along UP transects adjacent to the seawall.

Juvenile salmon have been documented transitioning to offshore in marine waters as they grow (Beauchamp & Duffy 2011). Acoustic measurements along a reference transect that was located adjacent to, and at an equivalent distance from the shore as the northernmost pier end (Fig. 2), only detected salmon once ($n = 45$) during the sampling season, which supports the supposition that the juvenile salmon in this study were migrating around piers rather than transitioning to offshore waters.

4. DISCUSSION

The DIDSON multibeam sonar is an effective mobile sampling tool to detect, and with supplementary information, identify small (40–90 mm) fish in coastal habitats. The DIDSON works independent of ambient light and can increase sampling efficiency

and spatiotemporal knowledge of fish distributions relative to visual 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. 2015; p. 504). Acoustic and snorkel methods complement each other. Acoustic methods are preferred when sampling extensively, or in dark or turbid waters, but are limited in species identifications and length estimations compared to optics (Burwen et al. 2010). The highest acoustic-to-snorkel count ratios occurred in UP habitats. Snorkel surveys detected no salmon in paired deep, UP transects, which are low-light, piling-filled environments, while acoustic surveys detected low numbers ($n = 90$) of juvenile salmon in these transects. Higher overall acoustic fish counts are attributed to the acoustic camera's ability to sample over a much greater range and volume than an optical camera or snorkel surveyor, and the independence of available light (Frias-Torres & Luo 2009). As a primary constraint, the DIDSON does not discern physical identifiers (e.g. color and distinguishing markings) among fish targets, and any acoustic-based species identification is challenging when used in isolation (Horne 2000). However, fish can be identified using size, shape, and swimming behavior (Baumgartner et al. 2006, Doebling et al. 2011) and validated by knowledge of species-specific annual presence along the waterfront collected from netting and snorkel surveys (e.g. Toft et al. 2007).

Snorkel methods are superior for species identification, length estimations, and other behavioral assessments (e.g. feeding), but are constrained by low light levels, high turbidity, and heat loss and exertion due to time spent in the water. With the exception of UP transects, snorkel salmon presence exceeded acoustic salmon presence in all habitats, indicating that with sufficient ambient light, snorkel surveys are advantageous in identifying individual fish or individuals within mixed groups relative to acoustic methods. The lowest acoustic to snorkel count ratios were in shallow BP transects, where mixed species shoals of similar-sized fish were challenging to identify within the acoustic data record. Acoustic and snorkel surveys were not conducted simultaneously, and overall detections or fish counts were not expected to match. 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 2 sampling methods should not be treated as equivalents that can be scaled but can be used to complement each other depending on the environment(s) sampled.

This was the first study comparing old and new pier habitats in a single season, with results showing that juvenile salmon presence and counts were higher along nearshore, eco-engineered UP habitats than along conventional UP habitats. These results indicate that progress has been made towards fulfilling the seawall eco-engineering objective to create an intertidal migratory corridor for juvenile salmonids (EBSP 2019). Eco-engineered pier microhabitats more closely mimic reference habitats compared to traditional UP habitats, where the likelihood of salmon presence/absence in eco-engineered habitats was more similar to reference transects than traditional pier habitats. These results corroborate and build on findings of Toft et al. (2007, 2013), Munsch et al. (2014), and Sawyer et al. (2020), who concluded that traditional UP microhabitats are suboptimal for juvenile salmon.

Although new, shallow UP habitats (i.e. the intertidal corridor) had higher light levels compared to pre-construction (Sawyer et al. 2020), light measurements along these transects had comparable intensities to new, deep UPs and comparable or lower median light availability than old UPs. Despite the comparable light levels, juvenile salmon were infrequently encountered in new, deep UPs and old UPs, both of which contain deep water and dense pilings. The combination of a textured seawall, shallow water habitat from the intertidal benches, and minimal pilings likely mitigates lower light intensities caused by the overhead sidewalk, and explains increased salmon presence in the intertidal corridor relative to traditional pier habitats. High overall salmon counts between piers, similar to reference site counts, indicate preferred habitat but also suggest that piers temporarily delay juvenile salmon. Juvenile salmon may wait for lower water levels associated with an ebbing tide to swim under piers (Munsch et al. 2014) or navigate around pier ends.

A new result from characterizing daytime juvenile salmon habitat use at pier ends is that many salmon are leaving the shallow, nearshore habitat that provides refuge and foraging opportunities (Simenstad et al. 1982) to navigate around piers. This behavior allows avoidance of dark environments under piers and strong light gradients adjacent to piers (Ono & Simenstad 2014). The highest median salmon counts were observed at pier ends, and the magnitude of these counts was similar to those observed in BP transects. The highest pier end salmon counts, which occurred in the first month of the sampling period and were likely chum salmon, indicate that salmon at pier ends were avoiding UP habitats rather than tran-

sitioning to deeper waters as is typical later in the season (Simenstad et al. 1982). This behavioral choice results in a migratory trajectory around piers to maximize ambient light levels, especially if it is possible to travel within larger shoals. Almost 43 times more salmon were observed swimming around the pier ends rather than under the inner edge of pier ends, even with elevated light levels at inner pier end transects compared to some nearshore microhabitats. 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 waterfronts likely increases locomotory energy expenditures, reduces feeding opportunities, and elevates predation risks for juvenile salmon (cf. Munsch et al. 2015, Cordell et al. 2017). Eco-engineering that increases refuge, connectivity, and prey availability for juvenile salmon in nearshore urban habitats may reduce salmon navigation around pier ends and is expected to result in increased juvenile salmon growth and survival.

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), necessitating nearshore spatial and economic planning in urban areas (Bilkovic & Roggero 2008). To benefit juvenile salmon that use urban corridors, future eco-engineering along urban waterfronts can minimize pier pilings, incorporate shallow water habitat, and enhance passive light integration. These eco-engineering solutions are best applied when armor removal and incorporation of other living shoreline elements are not possible (Toft et al. 2021). Migratory fish presence and abundance can be used as indicators of biological health for specific habitats (Karr 1991). The contiguity of their habitat, including obstructions that may delay migration, may be used as metrics of habitat quality. As the need for coastal armoring continues, eco-engineering measures may be an avenue to simultaneously protect urban infrastructure and mitigate coastal urbanization impacts.

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LITERATURE CITED

- ✦ Able KW, Grothues TM, Rackovan JL, Buderman FE (2014) Application of mobile dual-frequency identification sonar (DIDSON) to fish in estuarine habitats. *Northeast Nat* 21: 192–209
- Adams NS, Smith C, Plumb JM, Hansen GS, Beeman JW (2015) An evaluation of fish behavior upstream of the water temperature control tower at Cougar Dam, Oregon, using acoustic cameras, 2013. Open-File Rep 2015-1124. US Geological Survey, Reston, VA
- ✦ Akaike H (1974) A new look at the statistical model identification. *IEEE Trans Automat Contr* 19:716–723
- Baumgartner LJ, Reynoldson N, Cameron L, Stanger J (2006) Assessment of a dual-frequency identification sonar (DIDSON) for application in fish migration studies. Fisheries Final Report Series No. 84. NSW Department of Primary Industries, Cronulla
- Beauchamp D, Duffy E (2011) Stage-specific growth and survival during early marine life of Puget Sound chinook salmon in the context of temporal-spatial environmental conditions and trophic interactions. Final report to the Pacific Salmon Commission. Rep WACFWRU-11-01. Washington Cooperative Fish and Wildlife Research Unit, Seattle, WA
- ✦ Bilkovic DM, Roggero MM (2008) Effects of coastal development on nearshore estuarine nekton communities. *Mar Ecol Prog Ser* 358:27–39
- ✦ Blockley DJ (2007) Effect of wharves on intertidal assemblages on seawalls in Sydney Harbour, Australia. *Mar Environ Res* 63:409–427
- ✦ Bolker BM, Brooks ME, Clark CJ, Geange SW, Poulsen JR, Stevens MH, White JS (2009) Generalized linear mixed models: a practical guide for ecology and evolution. *Trends Ecol Evol* 24:127–135
- ✦ Brooks ME, Kristensen K, van Benthem KJ, Magnusson A and others (2017) GlmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. *R J* 9:378–400
- ✦ Browne MA, Chapman MG (2011) Ecologically informed engineering reduces loss of intertidal biodiversity on artificial shorelines. *Environ Sci Technol* 45:8204–8207
- ✦ Burwen DL, Fleischman SJ, Miller JD (2010) Accuracy and precision of salmon length estimates taken from DIDSON sonar images. *Trans Am Fish Soc* 139:1306–1314
- ✦ Cordell JR, Munsch SH, Shelton ME, Toft JD (2017) Effects of piers on assemblage composition, abundance, and taxa richness of small epibenthic invertebrates. *Hydrobiologia* 802:211–220
- ✦ Dethier MN, Raymond WW, McBride AN, Toft JD and others (2016) Multiscale impacts of armoring on Salish Sea shorelines: evidence for cumulative and threshold effects. *Estuar Coast Shelf Sci* 175:106–117
- ✦ Doehring K, Young R, Hay J, Quarterman A (2011) Suitability of dual-frequency identification sonar (DIDSON) to monitor juvenile fish movement at floodgates. *NZ J Mar Freshw Res* 45:413–422
- ✦ Duffy-Anderson JT, Able KW (1999) Effects of municipal piers on the growth of juvenile fishes in the Hudson River estuary: a study across a pier edge. *Mar Biol* 133:409–418
- EBSP (Elliott Bay Seawall Project) (2019) 2018 Post Construction Monitoring Report (Year 1). Anchor QEA and University of Washington, School of Aquatic and Fishery Sciences, Seattle, WA
- ✦ Firth LB, Thompson RC, Bohn K, Abbiati M and others (2014) Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures. *Coast Eng* 87:122–135
- ✦ Flitcroft RL, Arismendi I, Santelmann MV (2019) A review of habitat connectivity research for Pacific salmon in marine, estuary, and freshwater environments. *J Am Water Resour Assoc* 55:430–441
- Foote K (2009) Acoustic methods: brief review and prospects for advancing fisheries research. In: Beamish RJ, Rothschild BJ (eds) *The future of fisheries science in North America*. Fish & Fisheries Series, Vol 31. Springer, Dordrecht, p 313–343
- ✦ Frias-Torres S, Luo J (2009) Using dual-frequency sonar to detect juvenile goliath grouper *Epinephelus itajara* in mangrove habitat. *Endang Species Res* 7:237–242
- ✦ Grothues TM, Rackovan JL, Able KW (2016) Modification of nektonic fish distribution by piers and pile fields in an urban estuary. *J Exp Mar Biol Ecol* 485:47–56
- ✦ Healey MC (1982) Timing and relative intensity of size-selective mortality of juvenile chum salmon (*Oncorhynchus keta*) during early sea life. *Can J Fish Aquat Sci* 39:952–957
- ✦ Heerhartz SM, Dethier MN, Toft JD, Cordell JR, Ogston AS (2014) Effects of shoreline armoring on beach wrack subsidies to the nearshore ecotone in an estuarine fjord. *Estuar Coast* 37:1256–1268
- ✦ Horne JK (2000) Acoustic approaches to remote species identification: a review. *Fish Oceanogr* 9:356–371
- ✦ Karr JR (1991) Biological integrity: a long-neglected aspect of water resource management. *Ecol Appl* 1:66–84
- ✦ Klein JC, Underwood AJ, Chapman MG (2011) Urban structures provide new insights into interactions among grazers and habitat. *Ecol Appl* 21:427–438
- Kock TJ, Perry RW, Hansen AC (2016) Survival of juvenile chinook salmon and coho salmon in the Roza Dam fish bypass and in downstream reaches of the Yakima River, Washington, 2016. Open-File Report 2016–1210. Version 1.0: Originally posted December 22, 2016; Version 1.1: April 10, 2017. US Geological Survey, Reston, VA
- ✦ Lichatowich J, Mobrand L, Lestelle L, Vogel T (1995) An approach to the diagnosis and treatment of depleted Pacific salmon populations in Pacific Northwest watersheds. *Fisheries* 20:10–18
- ✦ Martignac F, Daroux A, Bagliniere JL, Ombredane D, Guillard J (2015) The use of acoustic cameras in shallow waters: new hydroacoustic tools for monitoring migratory fish population. A review of DIDSON technology. *Fish Fish* 16:486–510
- ✦ Maxwell SL, Gove NE (2007) Assessing a dual-frequency identification sonar's fish-counting accuracy, precision, and turbid river range capability. *J Acoust Soc Am* 122:3364–3377
- ✦ Morris RL, Chapman MG, Firth LB, Coleman RA (2017) Increasing habitat complexity on seawalls: investigating large- and small-scale effects on fish assemblages. *Ecol Evol* 7:9567–9579
- ✦ Munsch SH, Cordell JR, Toft JD, Morgan EE (2014) Effects of seawalls and piers on fish assemblages and juvenile salmon feeding behavior. *North Am J Fish Manag* 34:814–827
- ✦ Munsch SH, Cordell JR, Toft JD (2015) Effects of seawall armoring on juvenile Pacific salmon diets in an urban estuarine embayment. *Mar Ecol Prog Ser* 535:213–229
- ✦ Ono K, Simenstad CA (2014) Reducing the effect of overwater structures on migrating juvenile salmon: an experiment with light. *Ecol Eng* 71:180–189
- ✦ Ott J (2014) Alaskan Way Seawall (Seattle). <https://www.historylink.org/File/10807>
- ✦ Price V, Auster P, Kracker L (2013) Use of high-resolution DIDSON sonar to quantify attributes of predation at ecologically relevant space and time scales. *Mar Technol Soc J* 47:33–46
- Quinn T (2018) *The behavior and ecology of Pacific salmon and trout*, 2nd edn. University of Washington Press, Seattle, WA
- R Core Team (2019) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna
- ✦ Ray GC (2005) Connectivities of estuarine fishes to the coastal realm. *Estuar Coast Shelf Sci* 64:18–32
- Rizopoulos D (2020) GLMMadaptive: generalized linear mixed models using adaptive Gaussian quadrature (v 0.7-15). <https://cloud.r-project.org/web/packages/GLMMadaptive/GLMMadaptive.pdf>
- Ruggerone G, Volk E (2014) Residence time and growth of natural and hatchery chinook salmon in the Duwamish Estuary

- and Elliott Bay, Washington, based on otolith chemical and structural attributes. Prepared for the US Army Corps of Engineers, Contract no. DACW67-02-C-0009. Natural Resources Consultants, Inc., Seattle, WA
- Sawyer AC, Toft JD, Cordell JR (2020) Seawall as salmon habitat: Eco-engineering improves the distribution and foraging of juvenile Pacific salmon. *Ecol Eng* 151:105856
- Schwarz G (1978) Estimating the dimension of a model. *Ann Stat* 6:461–464
- Simenstad CA, Fresh KL, Salo EO (1982) The role of Puget Sound and Washington coastal estuaries in the life history of Pacific salmon; an unappreciated function. In: Kennedy VS (ed) *Estuarine comparisons*. Academic Press, Gleneden Beach, OR, p 343–364
- Simenstad C, Nightingale B, Thom R, Shreffler DK (1999) Impacts of ferry terminals on juvenile salmon migrating along Puget Sound shorelines. Phase 1: synthesis of state of knowledge. Washington State Transportation Commission, Department of Transportation, Seattle, WA
- Smith CS, Paxton AB, Donaher SE, Kochan DP and others (2021) Acoustic camera and net surveys reveal that nursery enhancement at living shorelines may be restricted to the marsh platform. *Ecol Eng* 166:106232
- Toft JD, Cordell JR, Simenstad CA, Stamatou LA (2007) Fish distribution, abundance, and behavior along city shoreline types in Puget Sound. *N Am J Fish Manag* 27:465–480
- Toft JD, Ogston AS, Heerhartz SM, Cordell JR, Flemer EE (2013) Ecological response and physical stability of habitat enhancements along an urban armored shoreline. *Ecol Eng* 57:97–108
- Toft JD, Munsch SH, Cordell JR, Siitari K and others (2018) Impact of multiple stressors on juvenile fish in estuaries of the northeast Pacific. *Glob Change Biol* 24:2008–2020
- Toft JD, Dethier MN, Howe ER, Buckner EV, Cordell JR (2021) Effectiveness of living shorelines in the Salish Sea. *Ecol Eng* 167:106255
- Venables WN, Ripley BD (2002) *Modern applied statistics with S*, 4th edn. Springer, New York, NY

Appendix: Additional data

Table A1. Daytime acoustic counts, by microhabitat, of fish species/groups through the sampling season. Surfperch groups are predominantly comprised of shiner perch. Herring are summed separate from forage fish; forage fish include surf smelt and sand lance. BP: between pier; UP: under pier

	Unknown	Surfperch/salmon	Herring	Tubesnout	Surfperch	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
Reference	118	259	5601	0	434	280
All ends	36	99	6510	130	965	20

Table A2. Characterization of juvenile salmon among nearshore transects through the sampling season. The final model is based on likelihood ratio test (LRT) values and consideration of the model's ecological relevancy. AIC: Akaike's information criterion; BIC: Bayesian information criterion. Note that 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. LRTs are used to identify potential spatial or temporal random effects using the R package 'glmmTMB' (Brooks et al. 2017) due to its ability to incorporate a null random effect

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 with LRT tests		Chi-sq		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 day of the year)		29.99		4.3e-08	

Table A3. Likelihood ratio test (LRT) 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 strong influence on values of Akaike's information criterion. 'Site type' was included in final pier ends model to balance comparisons to future pier-end model analyses

	df	Chi-sq	p
LRT values for nested nearshore models			
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
<i>Salmon ~ microhabitat + light + length + depth</i>	18	16.60	0.0002
Salmon ~ microhabitat + light + length + depth + tide	20	0.25	0.88
LRT values for nested pier-end models			
Salmon ~ 1	4		
Salmon ~ Pier edge	6	15.79	0.0004
Salmon ~ Pier edge + site type	8	1.81	0.40
<i>Salmon ~ Pier edge + site type + light</i>	10	1.17	0.56

Table A4. Daytime, all-season characterization of juvenile salmon among pier-end transects. The final model is based on likelihood ratio test (LRT) values and consideration of the model's ecological relevancy. Potential count distributions are compared using Akaike's information criterion (AIC) and Bayesian information criterion (BIC) values. Global models with random effects are compared against a global model with no random effect. LRTs are used to identify potential spatial or temporal random effects using the R package 'glmmTMB' (Brooks et al. 2017) due to its ability to incorporate a null random effect

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 with LRTs	Chi-square			p	
Full model + no random effect					
Full model + (1 month)		4.22		0.04	
Full model + (1 site)		0		0.99	
Full model + (1 day of the year)		5.06		0.02	

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