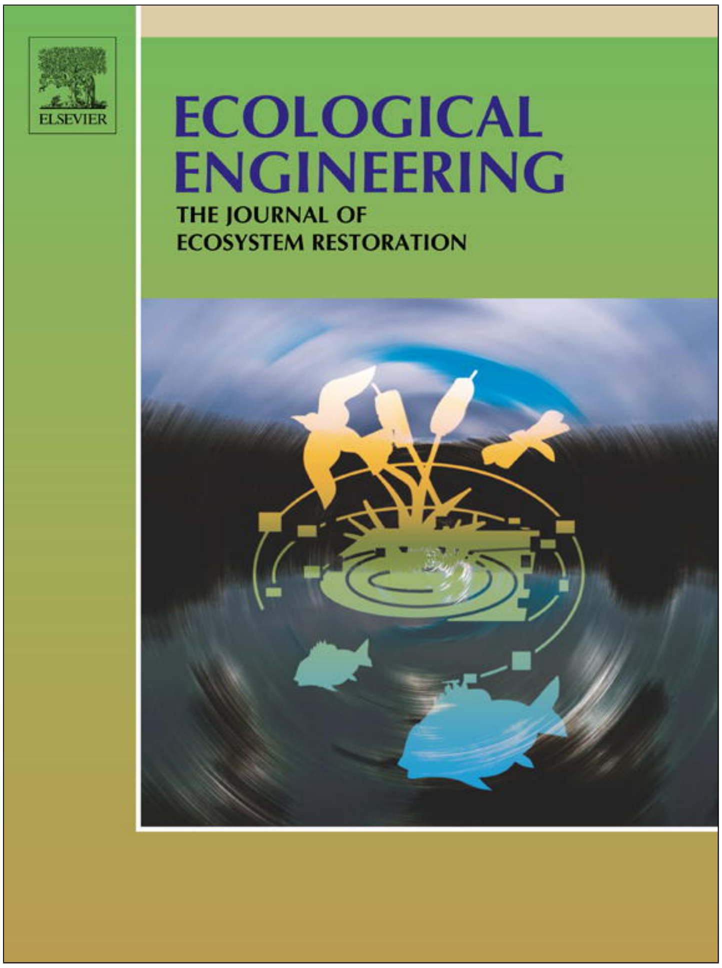
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Ecological response and physical stability of habitat enhancements along an urban armored shoreline

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ABSTRACT

Shoreline armoring is prevalent worldwide and has resulted in substantial habitat alteration in heavily urbanized areas. The biological and physical processes associated with these shorelines have in many cases been compromised, which has led to a recent focus on how to design and implement projects to restore some of the lost or impaired functions, termed enhancement. We describe a multi-year effort testing whether an enhanced site has improved conditions in Seattle, WA, USA, along urban marine shorelines of Puget Sound. The Olympic Sculpture Park opened in January 2007 and included construction of two shallow-water features: a low-terrace habitat bench placed in front of an existing seawall, and a constructed pocket beach that replaced existing riprap. Riparian vegetation was also planted in the uplands replacing impervious surfaces and manicured lawn. We measured the functions of these sites by sampling both before and after enhancements (2005, 2007, and 2009), and comparing to adjacent armored shorelines. Although we are limited in our ability to make generalizations beyond this specific site due to only having one replicate of each shoreline type, the unique aspects of this urban enhancement make it useful as a case study that can apply to other urban systems. Fishes that are dependent on shallow water habitat were a main focus of sampling, specifically outmigrating juvenile salmon (*Oncorhynchus* spp.) and larvae of other species. Terrestrial and aquatic invertebrates were also assessed, both as a metric for habitat quality and as a determinant of available prey resources for juvenile salmon. Physical features of the created habitats were monitored in post-enhancement years to measure their stability. Results showed that shoreline enhancements increased densities of larval fishes and juvenile salmon and measurements of juvenile salmon feeding behavior dependent on the year, and provided habitat for invertebrate assemblages that were different from armored shorelines and had high taxa richness. Physical resilience depended on both natural processes and human activities, demonstrating the need to incorporate anthropogenic use into the management of urban shorelines.

1. Introduction

Artificial armoring is a common feature of shorelines in many aquatic systems, especially in highly developed urban areas. Recent research has documented the detrimental effects of armoring on the ecotone between aquatic and terrestrial realms (Toft et al., 2007; Bilkovic and Roggero, 2008; Dugan et al., 2008; Defeo et al., 2009; Bulleri and Chapman, 2010; Shipman et al., 2010; Chapman and Underwood, 2011). However, there are only a few studies of habitat restoration along armored shorelines in urbanized settings where size and location of the restoration are severely limited. These studies have focused on beach nourishment (Peterson and Bishop, 2005; Defeo et al., 2009), managed realignment (French, 2006), creation of small wetlands (Grayson et al., 1999; Cordell et al., 2011), or incorporation of vegetation and ecological engineering with armoring (Chapman and Underwood, 2011), often referred to as “living shoreline” in the Gulf and Atlantic coasts of the USA (Erdle et al., 2006).

Understanding the current status of armored shorelines and potential for restoration in degraded systems is an important topic worldwide (NRC, 2007; Defeo et al., 2009). Novel designs and research pertaining to the value of benefits are needed to guide this emerging field of urban restoration. Interactions between people and nature are inevitable in urban landscapes but can lead to opportunities to improve city life (Standish et al., 2012), and principles of ecological engineering are defined to bridge human and ecological values (Mitsch, 2012). Nursery function of shorelines is one of

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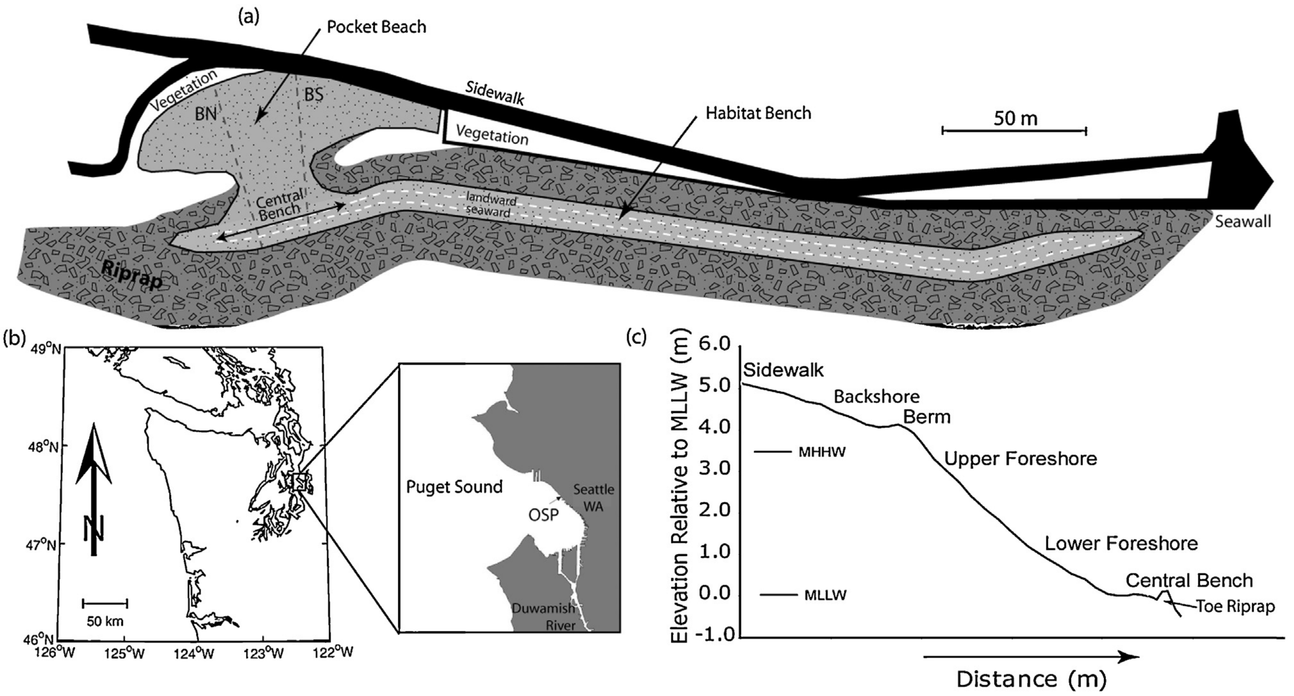

Fig. 1. (a) Plan view drawing of the Olympic Sculpture Park shoreline enhancements with approximate physical sampling locations of the beach north (BN) and beach south (BS) transect lines, seaward and landward transect lines at the habitat bench, and adjacent riprap and seawall sites, (b) location map, and (c) cross-section of the pocket beach.

many ecosystem services that can be affected by development, and habitat restoration can benefit not only fishery resources but also recreation, aesthetic quality, and other services that if engineered correctly may not impede important aspects of coastal protection (Guerry et al., 2012).

Although it is usually impossible to restore original conditions to extremely modified shorelines, it can be feasible to enhance or rehabilitate shorelines within urban constraints (Simenstad et al., 2005), often using principles of ecological engineering (Chapman and Underwood, 2011; Mitsch, 2012). We therefore use the term *enhancement* for actions that are intended to make progress toward the goal of *restoration*. Along a gradient of modified to unmodified shorelines, there is a progression from fully armored to historic natural conditions, with varying degrees of enhancement, restoration, and current natural conditions in between. The Seattle Art Museum incorporated enhancements along the shoreline of the Olympic Sculpture Park, which opened in January 2007 along a highly urbanized marine shoreline of Puget Sound in Seattle, WA, USA (Fig. 1). In order to provide both public access and ecological benefits, a pocket beach and habitat bench were created in shallow nearshore waters with plantings of vegetation and placement of sediments and driftwood on the beach (Fig. 2). These features replaced seawall and riprap armoring, with an overall goal of supporting higher diversity and numbers of fishes and invertebrates.

This project is of great interest in the Puget Sound region as an example of habitat enhancement along urban shorelines, and has therefore been a focus for an extensive monitoring plan that is meant to inform future projects. Similar projects have been either planned or recently implemented along stretches of Seattle shoreline, in other cities such as Bellingham, Olympia, and Everett, and also along non-urban shorelines such as in park settings. The Park is in a key area for rearing and migration of juvenile salmonids (Toft et al., 2007), especially Chinook salmon (*Oncorhynchus tshawytscha*) that are listed as threatened under the Endangered Species Act and chum salmon (*O. keta*), both of which use nearshore habitats more than other species of salmon. Larval fishes are also a focus of shallow water enhancements, particularly those of forage fishes that are nearshore spawners (Penttila, 2007).

Research worldwide has started to document how estuarine and coastal shoreline habitats can affect nearshore fish distribution, abundance, and nursery functions. For example, fish assemblages have been found to differ due to environmental variables of habitat types in the west coast of Australia (Valesini et al., 2004), Chesapeake Bay, USA (Bilkovic and Roggero, 2008), and Puget Sound, USA (Toft et al., 2007). Shoreline armoring can negatively impact fish prey such as terrestrial insects (Toft et al., 2007) and aquatic invertebrates to varying degrees, depending on how low in tidal elevation the armoring encroaches (Peterson et al., 2000; Chapman, 2003; Cruz Motta et al., 2003; Romanuk and Levings, 2003; Moschella et al., 2005; Sobocinski et al., 2010). These types of impacts can limit the opportunity for shoreline-oriented fishes such as juvenile salmon to feed and benefit from a site (Simenstad and Cordell, 2000). Shoreline modifications can also add hard structures uncharacteristic of the habitat, which in certain cases attract atypical and sometimes non-indigenous organisms (Glasby, 1998; Davis et al., 2002; Glasby et al., 2007). Overall, enhancement of armored shorelines seeks to improve nearshore conditions for native fishes and invertebrates.

Artificial beaches are becoming increasingly popular for shore protection worldwide. Where shoreline erosion has traditionally been controlled by hard structures such as seawalls, creating coarse clastic beaches (i.e., nourished with sediments that range from sand to boulders) can provide an alternative solution, as well as improve habitat. Coarse clastic beaches can presumably decrease potential adverse impacts of harder shoreline stabilization such as increased wave energy, scour, and interruption of sediment supply to coastal systems (Shipman et al., 2010). This can restore or enhance natural beach processes and habitats, and create recreational and ecological opportunities that did not exist before. However, many of the impacts of establishing nourished beaches are unknown with

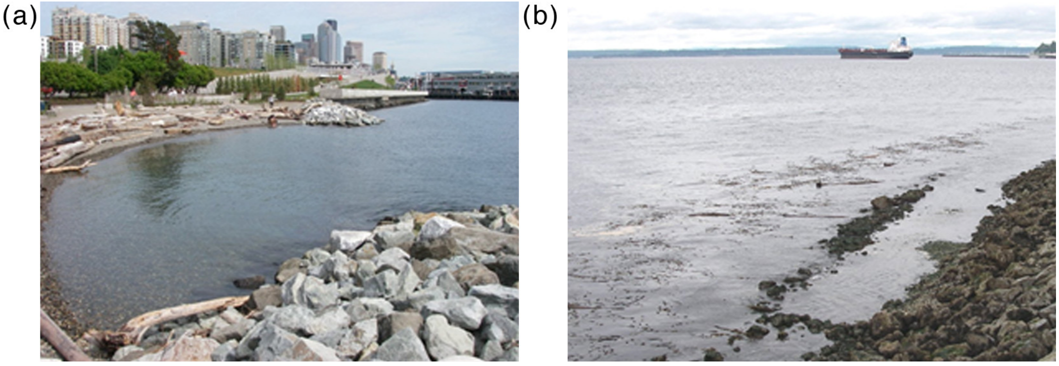

Fig. 2. Photographs of the Olympic Sculpture Park at (a) the pocket beach, and (b) the habitat bench. The pocket beach replaced riprap armoring, and the habitat bench was added as a shelf to the base of the seawall. The habitat bench is not visible in (a) as it is inundated at high tide along the seawall; (b) shows the habitat bench at low tide with kelp beds on the seaward side.

regard to their geomorphic stability and specific effects on biota (Speybroeck et al., 2006; Defeo et al., 2009). For this reason, it is useful to incorporate both physical and biological elements when evaluating overall performance of shoreline enhancements.

Many marine shorelines have a variety of natural and modified segments depending on the location, and it is important to address how a small-scale enhancement in an urban area can benefit the design and implementation of future small-scale enhancements in other locations. The objectives of this study which was performed prior to and following an enhancement project are to: (1) determine if incorporation of the habitat bench and pocket beach along the urban waterfront provide a step towards a return to natural conditions for nearshore fishes and invertebrates, and (2) evaluate whether the physical structures remain intact with minimal beach re-nourishment and/or stabilization. Although we are limited in our ability to make generalizations beyond this specific site due to only having one replicate of each shoreline type, this research is applicable to shoreline design scenarios worldwide that have similar placement of armoring at the ecotone between terrestrial and marine realms.

2. Study area and project design

In Puget Sound an average of 27% of the shoreline is armored, increasing to ~65% near urban centers (Simenstad et al., 2011). These structures are usually composed of vertical seawalls and riprap boulder fields. The Olympic Sculpture Park is located on the shores of Elliott Bay near where the Duwamish River flows into Puget Sound, within an area of extensive seawall and riprap armoring (Fig. 1). Tides in Puget Sound are mixed semidiurnal (average diurnal tidal range between MLLW and MHHW of 3.46 m), with two nearly equal high water levels and two unequal low water levels each day. Natural intertidal beaches in this part of Puget Sound are characterized by mixed sand and gravel sediments, with eroding coastal bluffs providing the primary source of sediments (Shipman et al., 2010). Puget Sound is considered a fjordal estuary, with cold temperate waters and salinities typically above 25 psu when not directly adjacent to river input.

The Seattle Art Museum purchased the main parcels of land with help from the Trust for Public Land, and Weiss/Manfredi was hired as the overall design lead for the Park. The main shoreline habitat elements were designed by Anchor QEA. Funding for the project was provided by widespread community support, with a combination of private, state, city, county, and federal sources. Reinforcements along the Park's seawall segment were spurred by concerns about the long-term seismic stability of the existing structure. The seawall along Seattle's waterfront was in need of replacement but at the time the City of Seattle did not have plans to replace the northern section for some years. The Seattle Art Museum chose to address the seawall during the Park's construction rather than experience disruption after the Park was created, with support from the City of Seattle in developing habitat improvements along the waterfront. Reinforcement of some of the existing seawall along with the associated habitat enhancements cost $5.5 million, which was cost-effective compared to the initial estimate of $50–80 million to completely replace that portion of seawall.

Two main shoreline enhancements were constructed at the Park, a habitat bench and a pocket beach (Fig. 2). The ~290 m long habitat bench was created out of compacted coarse-grained, angular sediment to simulate natural shallow water habitat in the low intertidal along the existing face of the Seattle seawall. Along the remaining stretch of shoreline north of the existing seawall, a ~100 m long pocket beach was excavated from a stretch of riprap armoring and surfaced with pebbles and cobbles. All sediments were locally sourced by Glacier NW, placed with a barge and crane, and spread with equipment on the beach. Riparian vegetation was planted in the adjacent supratidal uplands, with a focus on native species that are common in the Puget Sound coastal zone such as shore pine (*Pinus contorta*), alder (*Alnus rubra*), willows (*Salix* spp.), beach strawberry (*Fragaria chiloensis*), and also dune grass (*Leymus mollis*) at the pocket beach. Public use of these features has been extensive in this urban environment. Since construction, it is estimated that 400,000 visitors/year visit the Park with at least 1/3 of those using the pocket beach for recreational purposes, including sitting on the driftwood located at the berm, walking to the water line, and throwing stones into the water.

3. Methods

3.1. Spatial and temporal sampling

Biological sampling sites were located at the pocket beach, habitat bench and at adjacent riprap and seawall armoring for comparison as appropriate (only the riprap site was long enough for fish sampling). These adjacent armored sites were used as comparative un-restored sites, as there were no natural reference sites nearby along the city shoreline (Cordell et al., 2011). Biological monitoring was conducted during spring and summer in pre-enhancement (2005) and years 1 and 3 post-enhancement (2007 and 2009), to measure effects of the enhancements on biota compared to pre-enhanced conditions. Physical monitoring focused on the stability of the habitat bench and pocket beach in years 0 to 3 post-enhancement.

3.2. Beach profiles and sediment grain size

Two across-beach transects were established for monthly monitoring in all post-enhancement years (Fig. 1a). Profile surveys were conducted at low spring tides to capture the complete beach profile. Elevations were determined using a laser leveler and direct rod measurements with a stable point on the habitat bench as a reference (vertical precision ±4 cm). Mean Lower-Low Water (MLLW) was used as the vertical datum, and survey data were converted to MLLW using the measured water-surface elevation and NOAA tidal observations. Change in area on transects was calculated as the cross-sectional area of sediment between the beach profile and the −2 m MLLW elevation line. The change in area between surveys indicates the volume change of sediment on the beach. The entire habitat bench was surveyed in March 2007, April 2008, April 2009 and April 2010. Two transect lines were laid out along the bench and elevations were measured as for the beach profiles.

Characteristic sediment samples were obtained once in winter and summer in years 0, 1 and 3 from the beach shore face and berm, sampling both the surface and subsurface. At each sampling site, the surface sample was scraped to a depth exceeding approximately one diameter of the cobble surface material (∼5 cm). After initial characterization of subsurface material (collected ∼10–15 cm within the bed), additional samples of the subsurface were only collected if it was visually noted to be dissimilar to the surface sediment. Sediment samples were analyzed with standard grain-size analysis methods for coarse sediment, using the Wentworth (1922) grain-size classification scale. The fraction of sediment smaller than −4 phi was sieved through progressively finer sieves, and the coarser grains were individually measured on the intermediate axis. The median grain size (D_{50}) was obtained from the grain-size distribution, and sorting estimated from the width of the grain-size distribution histograms.

3.3. Fish

Fish and invertebrate sampling spanned the peak juvenile salmonid outmigration period from April to July, in pre-enhancement (2005) and years 1 and 3 post-enhancement (2007 and 2009). Twelve weekly snorkel surveys were used to observe fish in each year, a method that allows for high replication of effort across variable habitat types (Toft et al., 2007). Transects at the pocket beach spanned the 35 m length of the inundated beach, other transects were 75 m in length. Each site was characterized at shallow and deep water depths at both high and low tides (3 m and 10 m from shore; water depths varied between 1.3 and 3.9 m depending on site and tide), with two additional shallow transects in the inundated portion of the pocket beach at high tides to account for the intertidal gradient not present at other sites. This allowed surveys of the entire intertidal zone at each sampling site. Fish numbers were standardized by transect length and water visibility as fish number/[transect length (m) × horizontal secchi (m)]. We quantified fish species and number, approximate body lengths (2.5 cm increments), water column position (surface, mid-water, bottom), and occurrence of feeding behavior. Behavior measurements are rare in assessing restoration success, and have mostly been applied to studies of birds (Lindell, 2008). Such behavioral indicators can become evident before that of broader population responses, and are important as applied to the conservation efforts for endangered salmon species (Berger-Tal et al., 2011).

3.4. Invertebrates

Epibenthic invertebrates living at the water-sediment interface at ∼0 m MLLW, and terrestrial insects and other arthropods in shoreline vegetation were sampled April to July in pre-enhancement and years 1 and 3 post-enhancement. During each year five dates were sampled for epibenthic invertebrates and seven dates were sampled for insects. At each date seven random samples were collected at each site. A 16 cm diameter pump was used to sample epibenthic invertebrates, which were then preserved in 10% buffered formalin and sieved at 0.106 mm in the laboratory. Passive fallout traps were used to sample insects and other terrestrial arthropods (plastic storage bins 40 cm × 25 cm). The bottom of the traps was covered with a weak soap-water solution, and after 24 h the contents were sieved at 0.106 mm and preserved in 70% isopropanol. Invertebrate samples were processed in the laboratory for numerical composition, with taxonomic resolution to species for epibenthic crustaceans and family for insects.

3.5. Statistical analysis

Data were analyzed using Before-After Control-Impact (BACI) techniques as outlined in Underwood (1991, 1992), with the existence of an impact shown by a significant statistical interaction between Before-After and Control-Impact (Underwood, 1991). To address the question of shifts in invertebrates, measurements of taxa richness and assemblage structure of epibenthic invertebrates and insects were compared with the fixed factors site, year, and the interaction term, and the random factor date nested within each year, using an ANOVA test on taxa richness in the SPSS program and a multivariate PERMANOVA test (Anderson et al., 2008) on assemblages in the PRIMER program (Clarke and Gorley, 2006) (alpha = 0.05) using a Bray–Curtis resemblance matrix. Assemblage data were log-transformed before analysis with those taxa representing less than 3% of the total abundance of any one sample removed (Clarke and Gorley, 2006). When factors were significant, post hoc tests were used to further examine differences using the Tukey test on taxa richness and pairwise tests on assemblages, with a multivariate SIMPER analysis on assemblages to determine taxa that most contributed to the differences.

Similarly, to address the question of changes in juvenile salmon and larvae of other fishes (the two fish groups for which shallow water enhancements were designed to benefit), densities were compared with the fixed factors site, year, water depth, and the interaction terms, and the random factor date nested within each year, using an ANOVA test on log-transformed densities in the SPSS program. When factors were significant, post hoc tests were used to further examine differences using the Tukey test. To address changes in feeding behavior of juvenile salmon, frequencies of feeding versus not feeding for juvenile Chinook and chum salmon were compared using chi-square tests in the program SPSS.

4. Results

4.1. Beach profiles and sediment grain size

The beach profile consisted of a small storm berm that exhibited variability in height (+3.9 to 4.4 m MLLW) and location between surveys, and a foreshore with average slope ∼0.17 that led to the habitat bench at ∼0.0 MLLW with a riprap toe at the seaward edge. The overall effect of changes to the beach profile between construction and year 3 was a slight reduction in surface elevations over the upper and lower foreshore on the north side of the beach (BN) and a flattening of the beach slope on the south side of the beach (BS) due to sediment loss from the upper foreshore (Fig. 3a and b). Both beach transects lost sediment over the monitoring period from the

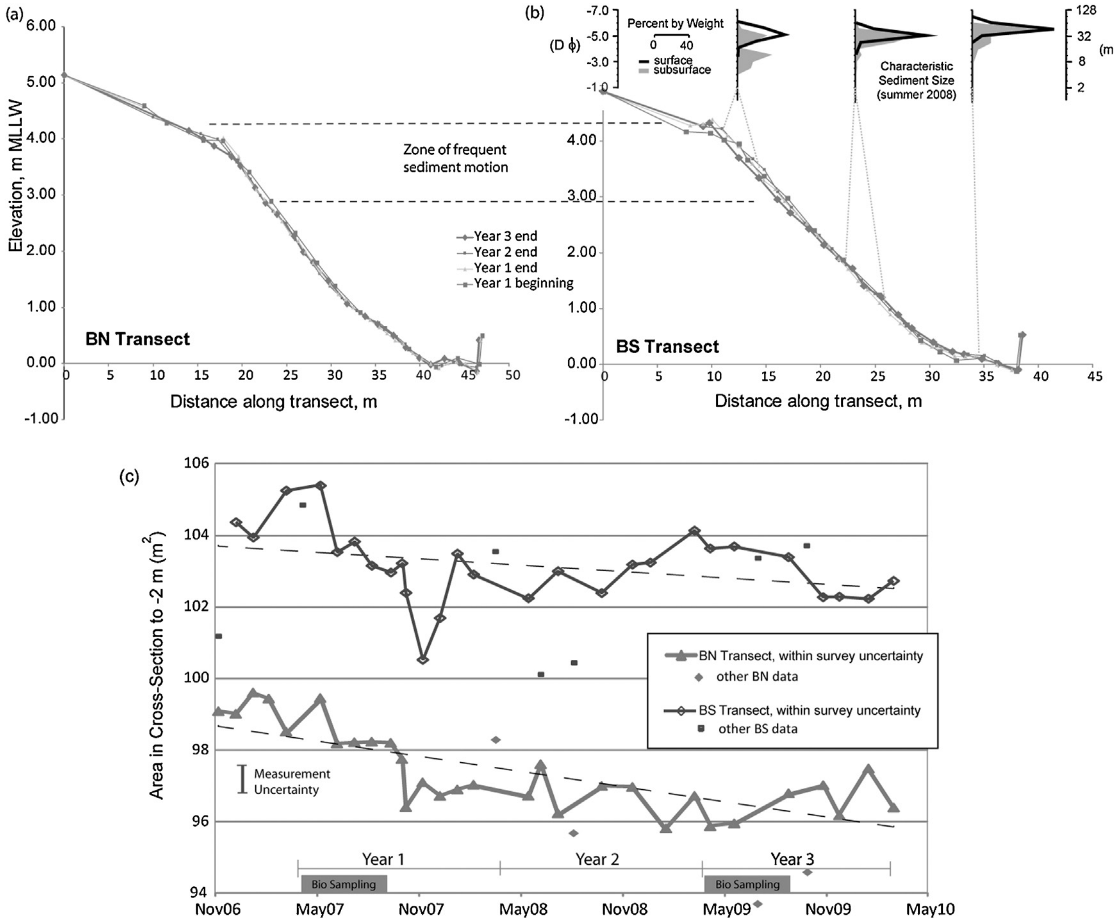

Fig. 3. Selected profiles across the pocket beach to illustrate the changes in beach profiles observed over the monitoring period for (a) BN transect and (b) BS transect. Characteristic sediment grain-size histograms are shown in (b) for the zones across the beach. Grain size histograms on the BN transect are similar. Temporal evolution of the cross-sectional area between the −2 m contour and the beach surface is shown in (c) and represents the volume change of beach material over time. An error estimate of volume based on measurement error is located in the bottom left corner.

original ~100 m^3/m that was placed (2.7 and 1.9 m^3/m shoreline length for BN and BS transects, respectively, between construction and year 3). The data trend suggests that much of this sediment loss occurred in 2007–2008 (year 1 and 2) and change stabilized or slowed in 2009–2010 (year 3; Fig. 3c). The range of elevations on the habitat bench (between 0.0 and −1.0 m MLLW) was similar for all years (±~10 cm except in one area of early riprap failure), indicating relatively little change in elevation.

The upper foreshore of the pocket beach had a layer of relatively well sorted surface sediments that ranged from −6.5 to −4.25 phi ($D_{50} \sim 20$–90 mm) (Fig. 3b). The surface sediment size at lower elevations was on the larger end of the size range. There was a subsurface layer on the upper foreshore that was finer with median diameter of −3.75 to −2.25 phi ($D_{50} \sim 5$–14 mm). At elevations below approximately +2.0 m MLLW, the subsurface and surface sediments were equivalent in size. The ranges of upper foreshore sediment size and profile slope are plotted together along with other datasets described in the Discussion (Fig. 4). Over time, grain size on the upper foreshore generally became more variable (less well sorted) and smaller in median grain size, associated with shifts in the coarser surface layer along the beach and limited mixing of subsurface and surface sediment. In addition, visual observations on the lower foreshore and habitat bench of clasts with no biological coating and smaller grain size indicates movement of individual clasts from the berm and upper foreshore to lower elevations.

4.2. *Fish composition, abundance, and foraging*

Twenty-three species of fishes and crabs were counted during snorkel surveys, including three species of juvenile salmon (chum, Chinook, and coho; Table 1). The larval fish category was abundant dependent on the year (Fig. 5a), and included all individuals too small to identify to species by snorkel observations (average total length 19 mm). They formed large schools often over 1000 fish, and occurred mainly in the middle of the water column (89% of numbers). Two main types of larval fish were observed: (1) post-larval forage fish, a sub-sample of which were identified as smelt (Osmeridae), and (2) demersal-type larval fish, a sub-sample of which were identified as sculpin (Cottidae). Of the larval fishes that could be categorized, 85% were of the forage fish morphology. Larger smelt (average total length 63 mm) were also observed only at the habitat bench. Of the juvenile salmon, chum were the most abundant, followed by Chinook, both of which were typically more abundant in shallow transects (Fig. 5b). Potential juvenile salmonid predators such as larger lingcod and sculpin were rare (Table 1). Water salinity and temperature varied little with water depth, averaging 24.8 and 12.2 °C at the surface, and 25.9 and 11.9 °C at the bottom, respectively.

Densities of juvenile salmon were significantly different for site, depth, and the interaction terms with year (Table 2). Their densities in shallow depths at the habitat bench and pocket beach were significantly higher than at the riprap site in year 1, but not in year

Table 1
Species list and average length estimates of fish and crabs from snorkel surveys in 2009, with total numbers (not standardized by transect length or visibility) and number of observations. Length estimates of fish are based on total length, and crab on carapace width.

Common name	Scientific name	Average length (cm)	Total number of fish	Number of observations
Chum	*Oncorhynchus keta*	7.0	3978	91
Chinook	*Oncorhynchus tshawytscha*	9.5	272	48
Coho	*Oncorhynchus kisutch*	8.8	6	6
Chinook/Coho	*O. tshawytscha/kisutch*	10.3	19	13
Juvenile Salmon, unk.	*Oncorhynchus spp.*	8.8	244	6
Bull Trout	*Salvelinus confluentus*	25.0	1	1
Larval Fish	–	1.9	265,657	54
Herring	*Clupea harengus pallasi*	6.3	5000	1
Smelt	Osmeridae	6.3	254	4
Pacific Sand Lance	*Ammodytes hexapterus*	11.3	4	3
Tubesnout	*Aulorhynchus flavidus*	7.7	5054	67
Shiner Perch	*Cymatogaster aggregata*	8.9	11,256	104
Striped Seaperch	*Embiotoca lateralis*	14.7	613	344
Pile Perch	*Rhacochilus vacca*	14.4	315	169
Kelp Perch	*Brachyistius frenatus*	9.5	54	24
Three-Spined Stickleback	*Gasterosteus aculeatus*	5.0	1	1
Lingcod	*Ophiodon elongatus*	45.3	11	11
Kelp Greenling	*Hexagrammos decagrammus*	20.0	1	1
Greenling	Hexagrammidae	26.3	3	1
Sculpin	Cottidae	20.0	1	1
Goby	Gobiidae	6.3	3	3
Crescent Gunnel	*Pholis laeta*	20.0	1	1
Penpoint Gunnel	*Apodichthys flavidus*	12.5	1	1
Fish, unk.	-	8.1	5	4
Kelp Crab	*Pugettia* spp.	8.2	46	42
Red Rock Crab	*Cancer productus*	13.0	76	19
Dungeness Crab	*Cancer magister*	11.3	2	1
Cheiragonid Crab	*Telmessus cheiragonus*	10.0	1	1

3 (additional post hoc Tukey tests). Densities of larval fishes were significantly different in site, year, and the interaction term, but not for depth (Table 2). Their densities were significantly higher in year 3, and in that year the habitat bench and pocket beach were significantly higher compared to the riprap site (additional post hoc tukey tests).

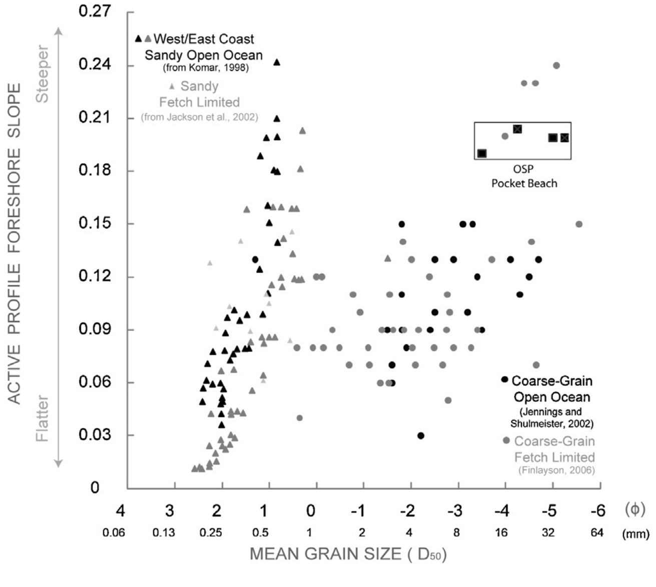

Fig. 4. Median grain size of beach sediment at an elevation of ∼+3 m MLLW as a function of active beach profile slope for nourished and natural beaches in Puget Sound and coastlines around the world. Multiple points for a specific beach reflect different sampling periods. Data for sandy beaches obtained from Komar (1998), Bascom (1953), Weigel (1964) and Jackson et al. (2002). Data for coarse-grained beaches obtained from Jennings and Schulmeister (2002) and Finlayson (2006). The Finlayson (2006) data set is a comprehensive survey of beaches around Puget Sound.

Feeding frequencies were analyzed using data from shallow transects where most of the observations on juvenile salmon occurred (Chi-square tests). For Chinook salmon in year 3, feeding frequencies at the habitat bench and pocket beach were higher than at the riprap site (significant only for the habitat bench, $p = 0.003$; Table 3). In year 1, there tended to be higher feeding frequency at the habitat bench and pocket beach, but these results were not significant. Year 3 feeding frequency was significantly higher at the habitat bench ($p = 0.0006$) and pocket beach ($p = 0.006$) as compared to the pre-enhanced period. For chum salmon in year 3, feeding frequencies at the habitat bench and pocket beach were significantly lower than at the riprap ($p < 0.0001$), which contrasted with the numbers in year 1 when the habitat bench and pocket beach had significantly higher feeding frequencies compared to the riprap ($p < 0.0001$ and $p = 0.04$, respectively; Table 3). Chum feeding frequencies were significantly higher at the habitat

Table 2
Results of statistical tests on epibenthic invertebrate and insect taxa richness (ANOVA) and assemblages (PERMANOVA), and juvenile salmon and larval fish densities (ANOVA). *p*-values for fixed factors are stated to four decimal places, and are bold if significant ($p < 0.05$).

	Insects		Epibenthic invertebrates	
	Taxa richness	Assemblages	Taxa richness	Assemblages
Site	**0.0001**	**0.0001**	**0.0001**	**0.0001**
Year	0.14	**0.0003**	**0.0001**	**0.0002**
Site × Year	**0.0001**	**0.0001**	**0.0001**	**0.0001**

	Juvenile Salmon densities	Larval fish densities
Site	**0.0003**	**0.0007**
Year	0.09	**0.0001**
Depth	**0.0001**	0.63
Site × year	**0.017**	**0.0001**
Site × depth	**0.0002**	0.39
Year × depth	**0.021**	0.92
Site × year × depth	0.08	**0.018**

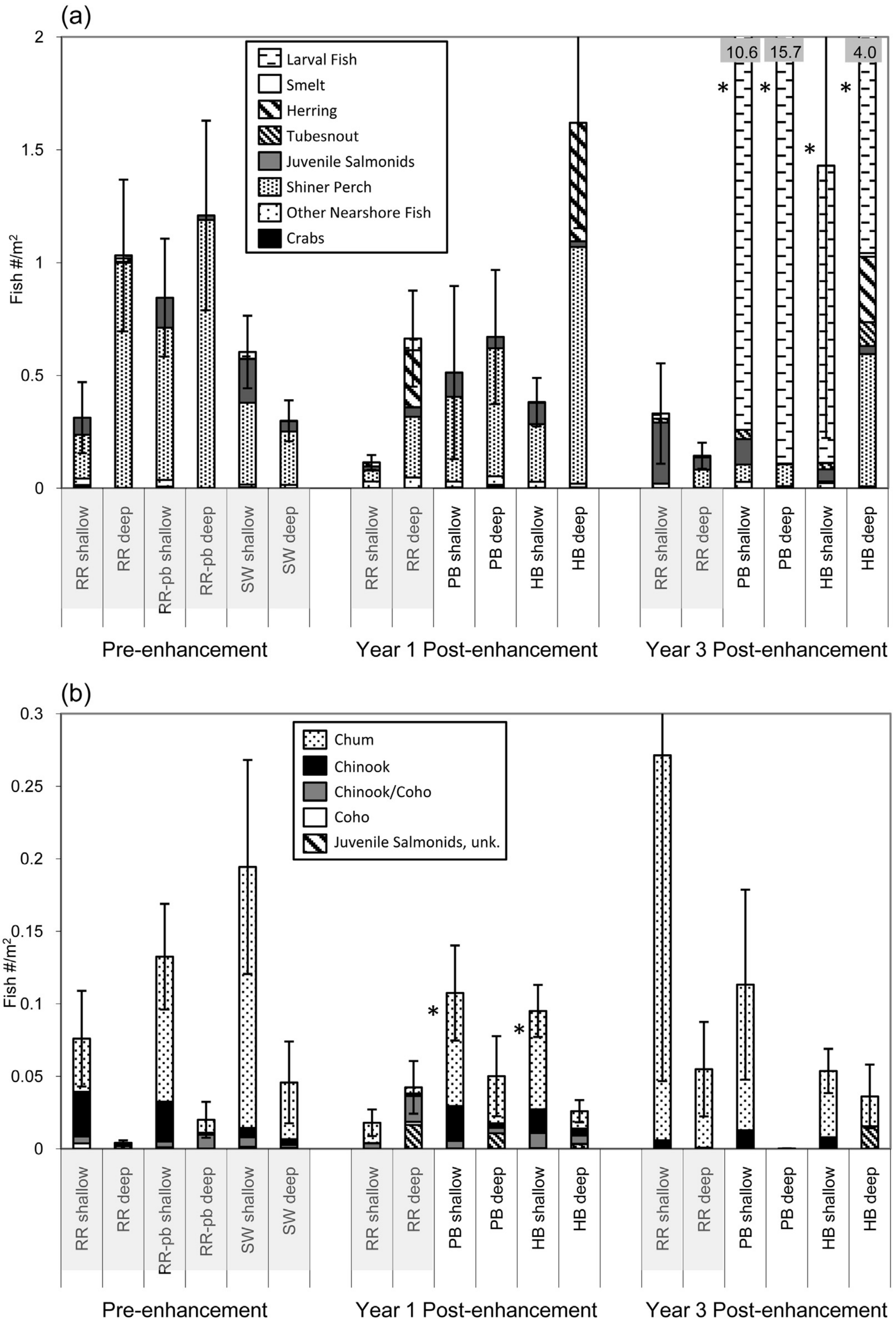

Fig. 5. Average total fish (a) and juvenile salmon densities (b) at shallow and deep transects by site and year. RR = Riprap, RR-pb = Riprap where pocket beach was later created, SW = Seawall where habitat bench was created, PB = Pocket Beach, and HB = Habitat Bench. Shaded site names are armored, error bars are ±SE, and asterisks depict significant differences detailed in Table 2. The *Y*-axis is truncated to show detail, total numbers are highlighted when above the scale.

bench in years 1 and 3 ($p < 0.0001$) and the pocket beach in year 3 ($p < 0.0001$) than they were pre-enhancement. Feeding events of juvenile salmon typically consisted of rapid forays to the surface to feed on neustonic prey, as well as some feeding in the middle of the water column; feeding directly on substrata was not observed. This reflected the location of the fish in the water column: Chinook salmon occurred mainly in the middle to upper part of the water column (67% of all observations in middle, 33% at surface), and chum salmon occurred mostly near the surface (85% surface, 15% middle).

Table 3
Frequencies of feeding observed for Chinook and chum salmon (N = number of fish), with summaries of Chi-square analysis (bold text indicates $p < 0.05$). Separate Chi-square tests were conducted comparing feeding at the enhanced sites to the riprap site (shaded) for each post-enhancement year ("S" indicates site significance), and comparing feeding at the enhanced sites to their pre-enhanced frequencies (shaded; "Y" indicates year significance).

Chinook	Pre-enhancement		Year 1		Year 3	
	N feeding (%)	N not feeding	N feeding (%)	N not feeding	N feeding (%)	N not feeding
Habitat Bench Shallow	201 (67%)	97	331 (64%)	189	**141 (82%) S,Y**	30
Pocket Beach Shallow	194 (57%)	147	91 (59%)	62	**48 (76%) Y**	15
Rip-Rap shallow			*20 (53%)*	*18*	*27 (60%)*	*18*

Chum	Pre-enhancement		Year 1		Year 3	
	N feeding (%)	N not feeding	N feeding (%)	N not feeding	N feeding (%)	N not feeding
Habitat Bench Shallow	467 (10%)	4290	**827 (55%) S,Y**	678	**504 (58%) Y**	365
Pocket Beach Shallow	160 (11%)	1260	**35 (8%) S**	395	**163 (36%) Y**	288
Rip-Rap shallow			4 (3%)	142	**1525 (81%) S**	353

4.3. Invertebrate assemblages

Taxa richness of epibenthic invertebrates was significantly different for site, year, and the interaction term (Table 2), with subsequent Tukey post hoc tests showing values at the habitat bench and pocket beach were higher than the riprap and seawall sites in years 1 and 3, and were higher in years 1 and 3 than pre-enhancement (Fig. 6a). Assemblages were also significantly different for site, year, and the interaction term (Table 2), with subsequent pairwise tests showing that the assemblages at the habitat bench and pocket beach were different than the riprap and seawall sites in years 1 and 3, and were different in years 1 and 3 than pre-enhancement (Fig. 6a). Assemblages at the habitat bench and pocket beach were dominated by harpacticoid copepods and amphipods, with especially high numbers of the harpacticoids *Tisbe* sp., *Harpacticus* spp., and *Heterolaophonte longisetigera* (Fig. 6a; SIMPER analysis). Composition of amphipods changed after site enhancement. Pre-enhancement, over 93% of amphipod composition consisted of one species, *Paracalliopiella pratti*. In years 1 and 3, *P. pratti* was less dominant at the habitat bench and pocket beach, both of which had greater amphipod diversity due to other taxa such as *Calliopius* sp. and *Desdimelita californica*, while the armored sites continued to be dominated by *P. pratti* (Fig. 6a).

Taxa richness in fallout traps was significantly different for site, and the site interaction term with year (Table 2), with subsequent Tukey post hoc tests showing values at the habitat bench and pocket beach were higher than the riprap and seawall sites in years 1 and 3 (Fig. 6b). Assemblages were also significantly different for site, year, and the interaction term (Table 2), with subsequent pairwise tests showing that the assemblages at the habitat bench and pocket beach were different than the riprap and seawall sites in years 1 and 3, and were different in years 1 and 3 than pre-enhancement (Fig. 6b). Assemblages at the habitat bench and pocket beach had high numbers of Acari (mites) and Collembola (springtails), also with more aphids than riprap and seawall (Fig. 6b; SIMPER analysis). Dipterans in the family Chironomidae typified the armored sites, and chironomids were also present at the habitat bench and pocket beach and relatively abundant at the habitat bench in year 1, along with contributions of other dipterans such as Sciaridae and Cecidomyiidae.

5. Discussion

Despite stresses induced by public use and other constraints on size and location imposed by the urban setting (e.g., foot traffic, vessel wakes, restricted boundaries), the habitat bench and pocket beach were relatively stable and there was a rapid and fairly consistent development of aquatic and terrestrial biota. The combined benefits of the pocket beach, habitat bench, and surrounding planted vegetation have created a mosaic of new habitats, providing more natural conditions for biota compared to riprap and seawall armored shorelines. Placing our results on the physical and biological aspects of the habitat bench and pocket beach along with results from other regional and worldwide datasets will help develop the role of enhanced shorelines in restoration theory.

5.1. Physical stability

Overall, the constructed coarse-clastic beach was relatively stable over the study period, as is typically the case with natural beaches of this type (Carter and Orford, 1984). Although stable in form, sediment was frequently in motion, particularly on the upper foreshore, the elevation zone where swash zone processes are focused on Puget Sound shorelines exposed to mixed semidiurnal tides. The greatest changes in the profiles at the pocket beach occurred as sediment moved from the middle to upper foreshore into the berm under energetic winter conditions, consistent with other studies on coarse-grained beaches (e.g., Everts et al., 2002). The driftwood on the berm appears to help stabilize local areas and acts as a trap for sediment, as also has been described for similar beach types (e.g., Finlayson, 2006).

The Park setting within a downtown public space brings challenges not typically considered in restoration implementation. Shifting of sediments during calm wind and wave conditions indicates that there may be forcing mechanisms not associated with natural causes that are important to the movement of sediment on this urban beach. During summer and generally calm natural conditions, lower low tides occur during daylight, and the public actively use the foreshore of the beach, and we infer that increased foot traffic caused the observed decreases in berm elevation and flattening of the upper foreshore slope. On the lower foreshore individual grains of upper foreshore surface sediment could be clearly identified due to the lack of biota growing on them and their size. These were likely brought to the lower foreshore through human intervention (e.g., throwing of rocks, often observed). Particles moved beyond the riprap toe of the beach and bench in this manner have no natural mechanism to bring them back over this barrier to the foreshore. Humans are drivers of change on urban beaches, and knowledge of anthropogenic use is important for understanding beach processes in public parks.

One concern for management is the stability of beaches that have been constructed with defined grain size and slope as part of their ecological designs. This can be assessed by comparing the sediment size and profile slope relationship of the pocket beach with other sand and coarse-grained beaches (natural and

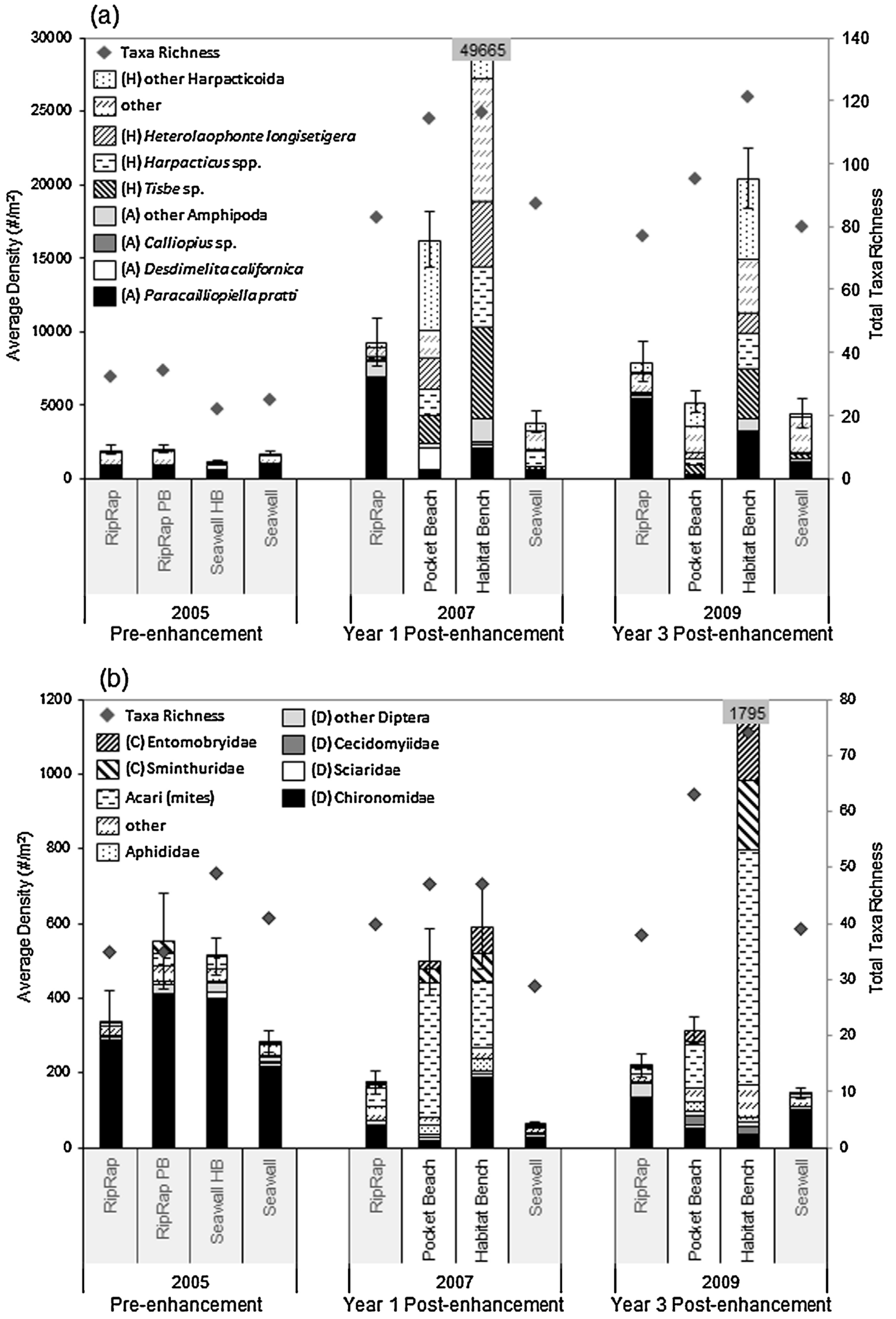

Fig. 6. Average epibenthic invertebrate (a) and fallout trap (b) densities and taxa richness by site and year. RipRap PB = Riprap where pocket beach was later created, and Seawall HB = Seawall where habitat bench was created. Shaded site names are armored, and error bars are ±SE. Amphipods (A) are shaded and Harpacticoid copepods (H) and other taxa are patterned in (a), and Diptera (D) are shaded and Collembola (C) and other taxa are patterned in (b). There were significant differences in assemblages and taxa richness (see statistical details in Table 2). The *Y*-axis is truncated to show detail, total numbers are highlighted when above the scale.

nourished) from Puget Sound and worldwide (Fig. 4). The data from sandy beaches provide a relatively consistent relationship between sediment size and beach slope, and the slope of that relationship has been found to change with wave energy (e.g., Bascom, 1953; Weigel, 1964). In contrast, coarse clastic beaches, although distinct from sandy beaches, show a wide spread in the relationship between a characteristic sediment size and beach slope, which cannot be simply attributed to wave energy. Antecedent morphology of the source area and the subsequent wide distributions of grain sizes in the beach material play a complex role in defining the morphology of the beaches (McLean and Kirk, 1969; Carter and Orford, 1984; Finlayson, 2006). The size/slope relationship for the pocket beach falls slightly outside the envelope of data from natural beaches in Puget Sound (Finlayson, 2006), and is generally steeper and slightly coarser. In Puget Sound and in other areas at mid to northern latitudes much of the beach substrate comes directly from the glacial tills incorporated in nearby bluffs. In contrast to the unimodal surface sediment at the pocket beach, natural beaches have an abundant source of sands, pebbles, and cobbles for subsequent reworking (Terich, 1987; Mason and Coates, 2001). In comparison to coarse-grained beaches throughout the world in both open ocean and fetch-limited environments, the pocket beach falls within the range, but groups with only a few other steep beaches. The implication of this comparison is that the pocket beach is likely to be relatively stable for the conditions it exists in, but natural processes will likely reduce the beach slope over time.

5.2. Fish use and behavior

Incorporation of the habitat bench and pocket beach have provided refuge for larval fishes and juvenile salmon and increased feeding opportunities for juvenile salmon dependent on the year, all benefits of creating shallow water areas in an urban setting where the majority of the shoreline has an armored steep intertidal zone. The increased feeding frequency of juvenile Chinook salmon at the habitat bench and pocket beach further show that behavioral response is a useful indicator of small-scale enhancements where space and replication are limited, and other studies would benefit by employing measurements of animal behavior. Increased feeding behavior at the habitat bench and pocket beach represents a potential for improved growth (Cordell et al., 2011), although we would need more measures of specific diet components in order to model possible growth benefits.

Enhancements could improve shoreline habitat elsewhere along armored shorelines, especially for fish and crustaceans that are known to prefer shallow water areas as nursery habitat during juvenile life stages (Ruiz et al., 1993), of which there are many examples of worldwide (Beck et al., 2001). Anadromous fish such as salmon that are transitioning through the estuary are but one type that utilizes estuaries as nurseries, there are a variety of life history strategies that also make use of estuarine nursery habitat during juvenile stages (Whitfield, 1999). Benefits for fish have been demonstrated at other types of enhancements along armored shorelines, such as managed realignment of shoreline edges and creation of small wetlands (Simenstad and Thom, 1996; French, 2006; Cordell et al., 2011), and incorporation of vegetation and ecological engineering with armoring (Chapman and Underwood, 2011) sometimes termed “living shoreline” (Erdle et al., 2006). Another common enhancement option is sediment nourishment of beaches, which remains controversial as to its perceived benefit and possible detrimental effect on fish (Peterson and Bishop, 2005; Defeo et al., 2009) although when applied to suitable habitats such as dunes is more apt to be successful (Nordstrom et al., 2011). Incorporation of the results from our study with these other examples of fish enhancement point to confirmed use of nekton if a suitable design is employed that matches the local site conditions, therefore it will be useful to have a variety of proven designs that can be applied to different scenarios. A potential modifier to our own results is the scale, or toe elevation of armoring. The lower the elevation the greater the impacts on fish distributions, shifting locations from steepened intertidal waters into subtidal waters (Toft et al., 2007), and in the case of the Olympic Sculpture Park where armoring extended into subtidal waters the subsequent change into a lower gradient shoreline has improved both fish habitat and public access. This response might be more subtle for locations where armoring only encroaches into upper intertidal areas.

The significantly higher numbers of larval fishes that we observed at the pocket beach and habitat bench in year 3 compared to the riprap and to pre-enhancement conditions suggests that the enhancements provided more natural habitat for these fishes. Previous snorkel surveys in the area have not documented larval fishes in such consistently high numbers (Toft et al., 2007), and netting surveys have not captured larval fishes because mesh sizes were too large (Brennan et al., 2004; Nelson et al., 2004). The larval fishes we observed may have been hatched in substrata provided by the sites, using the sites as refuge, or transported there by water currents. The habitat bench and pocket beach represent the first shallow water areas north of the downtown Seattle seawall, and shallow water provides refuge habitat for small fish in other human-impacted systems. For example, in Chesapeake Bay Ruiz et al. (1993) found that several species of small fishes and mobile invertebrates shifted their distributions from deeper to shallower water to avoid predation after the human-induced demise of submerged aquatic vegetation. In the Puget Sound region, larval forage fishes are known to use shallow water and beaches as nursery grounds (Penttila, 2007). The smelt, herring, and tubesnout that were abundant at the habitat bench and pocket beach may have been linked to the abundant larval fishes, either because they all sought the refuge attributes of the enhanced habitats or because the adult fishes used the shallow water and beach sediments there for spawning. Altered beaches have been shown to have lower proportions of live embryos of surf smelt (*Hypomesus pretiosus*) eggs than more natural beaches (Rice, 2006), so enhancement of urban beaches has the potential to both provide new spawning habitat and improve egg survival in existing habitat.

5.3. Invertebrates

The new, more complex habitats at the pocket beach and habitat bench had high epibenthic invertebrate taxa richness compared to less complex armored shorelines. Complexity is known to enhance diversity (Kovalenko et al., 2012), and this response to structural complexity has been documented elsewhere in surveys of armored seawalls and un-armored shorelines (Chapman, 2003; Morley et al., 2012). Our research adds to this by showing a similar response to complexity created by enhancement of a shoreline. Association of epibenthic invertebrates with the enhanced shorelines may be similar to signatures from natural structural complexity, where for example epibenthic invertebrate densities have been found to be higher in oyster and eelgrass plots than in bare mudflats (Hosack et al., 2006), and invertebrate diversity has been found to increase with algae complexity (Dean and Connell, 1987). Overall, the rapid development of epibenthic assemblages at the habitat bench and pocket beach suggests that even small-scale enhancement projects can increase biological function along urbanized shorelines, although more studies are needed to determine the associations between sediment size, invertebrate presence and response of fish feeding. Juvenile Chinook and chum salmon have eclectic diets, and providing prey from a number of different habitats (e.g., terrestrial vegetation, algae, soft-sediment substrates) will be beneficial to them (Brennan et al., 2004; Nelson et al., 2004; Toft et al., 2007; Duffy et al., 2010).

Abundances of certain insects that are associated with vegetation have increased as a result of shoreline plantings. These included aphids that are known to be juvenile salmonid prey items (Brennan et al., 2004), whereas other prey items such as chironomids are abundant at both armored and enhanced sites. A study along an estuary at the Campbell River in BC, Canada, also documented observations of juvenile salmonids darting to the surface to feed (MacDonald et al., 1987), further showing the importance of surface-feeding at various points in the estuary. Other studies have shown insects to be significantly reduced on armored shorelines where vegetation was removed (Romanuk and Levings, 2003; Sobocinski et al., 2010), and continued development of the vegetation communities may increase the input of insects and feeding opportunities for juvenile salmon.

6. Restoration implications, and conclusions

Management of armored shorelines will be an increasing issue due to the conflicting forces of sea level rise and shoreline development, termed “coastal squeeze” (NRC, 2007; Defeo et al., 2009). More information about the effects of coastal squeeze is available for sandy beaches than for mixed sediment beaches characteristic of Puget Sound (Nordstrom, 2000; Defeo et al., 2009), and rigorous studies are needed across different beach morphologies to fully understand these effects. Although the habitat bench and pocket beach could not address restoration of beach processes such as sediment supply, the structural aspects have been shown to be stable along an armored setting, and should remain viable with ongoing coastal squeeze. Along a developed shoreline, the ecosystem goods and services provided by a mosaic of engineered and natural conditions may be more resilient under current processes than those of an unrealistic historic goal (Jackson and Hobbs, 2009), especially in cases of extreme urbanization where the original morphology of the shoreline has been completely altered such as along the downtown Seattle waterfront. Novel ecosystems that have formed under altered conditions require creative management solutions for restoration goals, and collaborations between managers and scientists are necessary to understand the full usefulness and application of shoreline enhancements (Seastedt et al., 2008). Our study has shown that the habitat bench and pocket beach have beneficial aspects for nearshore fishes and invertebrates, and not only did these design features develop by way of meetings with many agencies such as the City of Seattle, Seattle Art Museum, and design consultants, but are currently being proposed for possible use along the future rebuild of the Seattle seawall, a structure that throughout its existence has been ignored for any habitat improvements.

It is important to acknowledge that there was some annual variation in our results, and that the enhanced shorelines did not always show definitive improvements over armored shorelines. As mentioned previously, wide interpretation of our results must be done with caution as we are limited by our ability to generalize with only one replicate of each shoreline type. Since our study has focused on the first three years post-enhancement, long-term conditions of the enhanced sites will not be known until they become more stable in ecological and physical structure, depending on site-specific processes (Simenstad and Thom, 1996; Dethier and Schoch, 2005).

Following seawall removal, enhancing armored shorelines in order to approach restored conditions while still providing shoreline stability is relatively new to both design and science. Given the caveats on the limited replication and associated analysis and interpretation of these unique sites, we offer several conclusions from our study that should be reinforced or reputed from future studies as they emerge:

(1) Nursery area for fish: Nearshore fish used shallow-water enhancements along armored shorelines. More fish (juvenile salmon and larvae of other species, dependent on the year) were observed along enhanced shorelines, often with higher feeding activity.
(2) Foraging opportunities: Invertebrates that are prey for juvenile salmon and other fish colonized the low gradient, finer-grained intertidal habitats that were incorporated into the armored shorelines. Most of these aquatic invertebrates increased in taxa richness and numbers with enhancements in place.
(3) Riparian value: Some types of terrestrial insects increased in abundance and taxa richness where patches of shoreline vegetation were planted. Vegetation does require routine maintenance in an urban park setting, so progression to a full natural riparian zone will be somewhat limited.
(4) Connectivity: Linkages between aquatic and terrestrial zones are broken on heavily armored shorelines. When artificial barriers are removed and aquatic habitats merged with terrestrial habitats, this provides the opportunity for biological and physical processes to reconnect across the ecotone.
(5) Physical resilience: In heavily urbanized settings, habitat enhancements have limited ability to change larger scale processes such as sediment supply, and this may lead to the need for maintenance. Human use of urban beaches can create a signal of surface sediment loss such that occasional nourishment may be needed.

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