

Effects of Shoreline Modifications on Supratidal Macroinvertebrate Fauna on Puget Sound, Washington Beaches

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Received: 1 August 2008 / Revised: 6 September 2008 / Accepted: 22 December 2009 / Published online: 28 January 2010
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Abstract In coastal environments, the supratidal zone bridges marine and terrestrial ecosystems and is important for energy exchange. However, it is also subject to extensive anthropogenic disturbance, such as armoring of shorelines. Shoreline armoring is extensive along many coasts, but the impacts on biota are comparatively unknown. Between 2000 and 2002, paired and synoptic sampling regimes were employed to assess armoring effects on insects and benthic macroinvertebrates in the supratidal zone of Puget Sound beaches. Paired sampling showed natural beach sites had significantly more deposited wrack. Infauna was dominated by oligochaetes and nematodes; talitrid amphipods, insects, and collembolans were significantly more numerous at natural beaches, and crustaceans were more abundant at altered beaches. Insect assemblages were diverse, with taxon richness higher at natural beach sites. In the synoptic sampling, where sites with higher elevation modifications were used, there were fewer differences in invertebrate assemblages between armored and nonarmored sites. The results show that, where shoreline

armoring lowers the land–sea interface, benthic infauna and insect assemblages are disrupted. Widespread shoreline modifications may decrease the availability of prey resources for fish and wildlife and decrease the contribution of organic material entering the nearshore ecosystem.

Keywords Supratidal · Shoreline modifications · Macroinvertebrates · Coastal development · Puget Sound · Beaches

Introduction

As human populations grow, environmental management and conservation efforts around population centers have become increasingly focused on preserving biodiversity (Savard et al. 2000). A major effect of urbanization on biodiversity results from artificial structures placed in aquatic ecotones. Along estuarine and marine shorelines, common biodiversity-altering artificial structures include docks, seawalls, and rip-rap armoring (Burdick and Short 1999; Chapman 2003). While the installation of these structures has been widespread, our understanding of how they impact shoreline habitats and fauna is relatively unknown.

Such structures may have negative effects on both resident and transient organisms: Peterson et al. (2000) found that beaches with shoreline modifications on the Gulf Coast of Mississippi harbored fewer demersal finfish and invertebrates than those with naturally vegetated shorelines; and boat docks in Massachusetts estuaries have been shown to result in depressed shoot density and canopy structure of eelgrass, *Zostera marina* L. (Burdick and Short 1999). Several recent studies of seawalls in Sydney Harbor have indicated that seawalls have fewer mobile species associat-

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ed with them compared to natural rocky shores (Chapman 2003; Chapman and Bulleri 2003). While the occurrence of these modifications is common, it is only recently that their impact on fauna has been investigated; furthermore, little attention has been paid to cumulative impacts from such structures.

Approximately one third of the Puget Sound, Washington, shoreline is anthropogenically modified, with over 70% modified in and around the greater Seattle metropolitan area (Washington Department of Natural Resources 1999), resulting in significant shoreline and intertidal beach disruption from the installation of piers, docks, shoreline revetments, and ports (Levings and Thom 1994; Nightingale and Simenstad 2001). Despite rapid growth, there has been little focus on beach ecosystem alteration and the resulting biological impacts of shoreline development (Emmett et al. 2000; Myers 2001). To understand the impacts of built structures (rip-rap revetments, bulkheads, and other types of shoreline armoring—henceforth referred to as shoreline modifications), we measured the effects of such structures on macroinvertebrates in the supratidal zone of Puget Sound, Washington.

Puget Sound is a polyhaline fjord, located in western Washington, USA. Extensive freshwater input from several large river systems creates a mixed-salinity environment. The Sound has a moderately to steeply sloping mixed-substrate shoreline, nourished primarily by sediments (e.g., glacial till) eroding from shoreline bluffs (Downing 1983). As an estuarine inland sea, it supports a wide variety of flora and fauna, including top-level predators, and numerous species of birds and finfish, including five species of commercially and culturally important Pacific salmon (*Oncorhynchus* spp.).

In Puget Sound, as in other systems, most shoreline modifications occur in or across the high intertidal or supratidal zone. The supratidal zone is defined as the area above mean higher high water (MHHW), influenced by splash and extreme high tides, generally 3.2 m above mean lower low water (MLLW) and higher in central Puget Sound (Carefoot 1977). This zone is a unique but little-studied ecotone, having largely escaped notice of both marine and terrestrial ecologists (Attrill et al. 1999). In bridging the two systems, it possesses some properties common to both and is a dynamic region of great importance for energy exchange and nutrient flux in the greater nearshore ecosystem (Polis and Hurd 1996; Zimmer et al. 2002). The ecology of the supratidal zone is driven by both terrestrial and marine processes including marine wrack deposition and decomposition, terrestrial leaf litter input, sediment deposition from bluff erosion, deposition of large wood (both from longshore currents and direct input from the upland), recycling and decomposition of organic detritus, and export of organisms to the nearshore food

web. Among the physical factors affecting supratidal ecology on Puget Sound beaches are exposure, fetch, tidal regime, drift cell dynamics, and sediment grain size. This zone also serves as a source of marine-derived nutrients that enter the terrestrial system through decomposition of deposited marine wrack.

Vegetation in and adjacent to the supratidal zone is habitat for insects (Allan et al. 2003) and other invertebrates and shades the beach, maintaining moisture and temperature thresholds essential for spawning forage fish such as surf smelt (*Hypomesus pretiosus*; Rice 2006) and marine crustaceans (Koch 1989), as well as other beach-dwelling invertebrates (Pennings et al. 2000). As secondary consumers, invertebrates are critical components of nearshore detritus-based food webs. Changes in invertebrate assemblages resulting from anthropogenic effects may have cascading impacts at higher trophic levels (e.g., finfish, mammals, and/or birds) if such food resources are limiting at critical stages in the organisms' life histories (Kitchell et al. 1994; Romanuk and Levings 2003b).

In Puget Sound, Pacific salmon (specifically, Chinook and chum salmon, *Oncorhynchus tshawytscha* and *Oncorhynchus keta*, respectively) are known to rear in estuarine and nearshore marine habitats before migrating offshore (Simenstad et al. 1982; see also Groot and Margolis 1991 for a general discussion of salmon life histories). The extent of exposed shoreline use is little studied compared to that of estuarine marsh and eelgrass ecosystems, which have been shown to be important nursery habitats for juveniles (Healey 1981; Shreffler et al. 1992; Tanner et al. 2002). Insects can be an important prey resource for juvenile salmon during their early marine life history (Brodeur 1989; Duffy et al. 2005; Romanuk and Levings 2005), including along marine shorelines. While juvenile salmon insect prey is mainly comprised of pupal and emergent chironomid flies in lower-salinity estuarine habitats (Gray et al. 2002; Tanner et al. 2002), several recent studies have found that juvenile salmon collected along marine shorelines in Puget Sound had prey comprised mostly of terrestrial insects (Toft et al. 2007; Duffy et al. 2005), suggesting that the fringing coastal terrestrial system is important in salmon early life histories in these habitats.

The extent to which shoreline modifications in Puget Sound disturb intertidal biological processes in general, and salmon habitat in particular, is not well understood. However, in many estuarine shoreline settings, modifications can result in changes to physical structure and function, including coarsening of sediments, deflection of wave energy, and increasing erosion along the unaltered shore (Inman and Brush 1973; Nordstrom 1989; Macdonald et al. 1994; Douglass and Pickel 1999). These shoreline modifications, and specifically bulkhead and rip-rap armoring, produce many physical alterations: the removal of

backshore vegetation and large wood, introduction of new material dissimilar from the natural substrate, and the replacement of beach with hard and often vertical surfaces. Consequently, when modifications are installed at or below MHHW, they intrude into the intertidal zone, shift the land/sea interface seaward to an intertidal elevation below MHHW, and radically transform much or all of the supratidal zone, potentially interrupting biological processes.

This study assesses invertebrate assemblages in the supratidal zone of Puget Sound and describes how those assemblages at sites with bulkheads and rip-rap differ from those without shoreline modifications. The purpose of this study was twofold: (1) to evaluate how the biological structure of macroinvertebrate assemblages change when the shoreline is modified by armoring and (2) to assess differences in occurrence and abundance of potential salmonid prey invertebrates in the supratidal zone at sites with and without shoreline armoring.

Methods

We used two approaches—paired and synoptic sampling—to assess invertebrate assemblages among sites with and without shoreline armoring. Paired sampling was intended to directly compare spatially discrete natural and altered beaches where shoreline modifications extended below MHHW; synoptic sampling was used to measure the indirect effects of armoring on benthic supratidal biota over a broader spatial scale. Synoptic sampling was conducted at a consistent elevation (i.e., we selected sites with shoreline modifications higher than MHHW) to allow for a standardized comparison between natural and altered sites without elevation effects.

Methodology for this study was derived from the Estuarine Habitat Assessment Protocol (Simenstad et al. 1991). By quantifying invertebrates at both armored and

natural beaches, we intended to measure and describe both direct and indirect effects of shoreline armoring on supratidal biota. Fieldwork commenced in September 2000 with a pilot study and was completed in 2002.

Paired Sampling

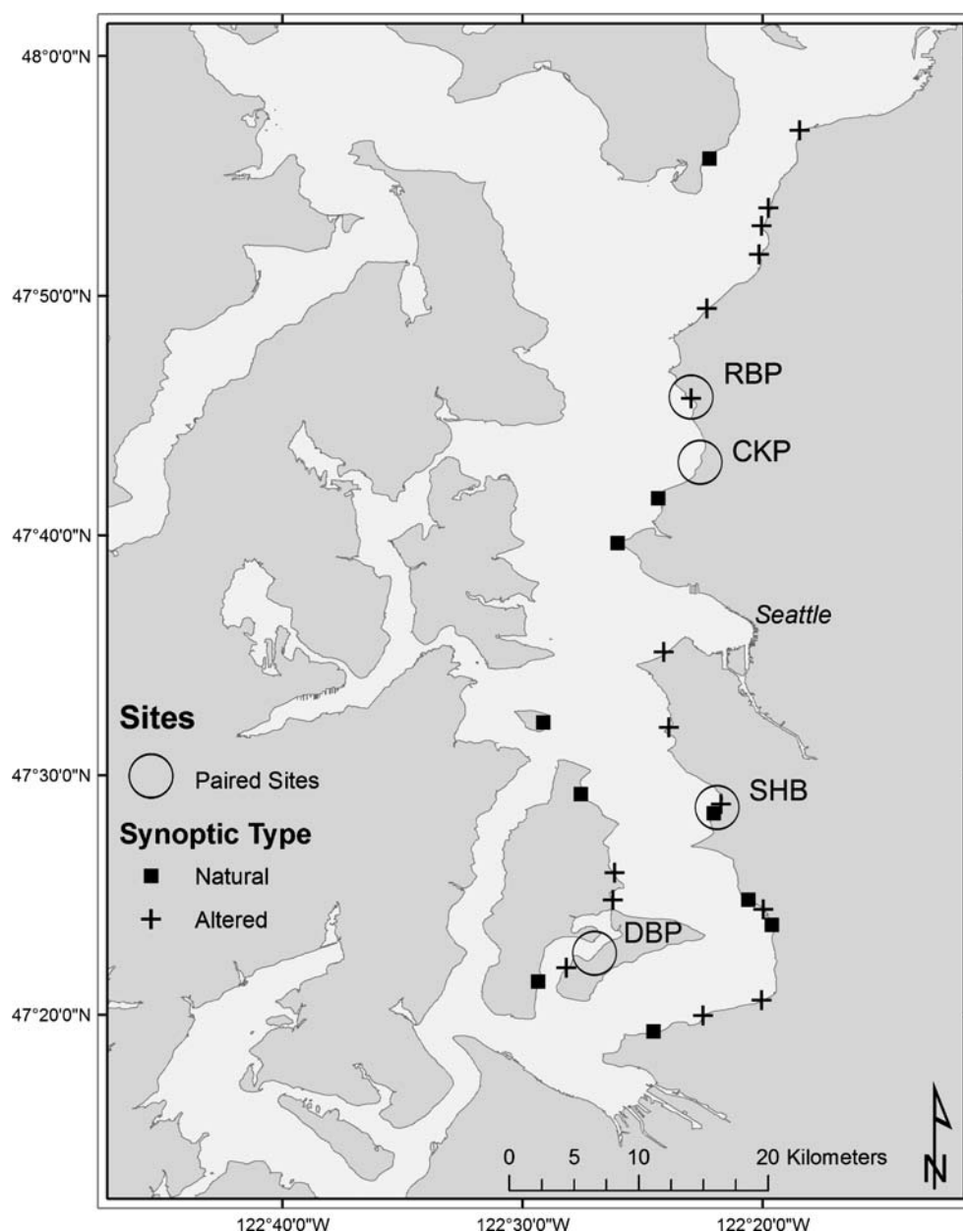
Beaches typical of those found in central Puget Sound and with public access were selected for this study. Four sites with an altered beach adjacent to a natural beach were selected for paired sampling: Carkeek Park (CKP), Dockton Beach (DBP), Richmond Beach (RBP), and Seahurst Beach (SHB; Table 1, Fig. 1). The most common beach type represented in this study was “open estuarine intertidal” beach with “mixed-coarse,” “gravel,” or “sand” substrate (sites CKP, RBP, SHB; Dethier 1990). The exception was DBP, which consisted of “partly enclosed estuarine mud.” This site was selected to represent lower-energy beaches occurring in the sampling region. Sampling was conducted from late March to early June 2001 ($n=5$) to capture spring tide events with high tides greater than +10 ft (3.05 m, MLLW). Natural and altered beaches at each site were sampled on the same day, immediately after the onset of the ebb tide.

Samples were selected randomly along a 50-m transect tape running parallel to the shoreline. Transects remained constant throughout the study, but new random locations along the transects were selected for each sampling period. Transects for altered beach sites were located immediately seaward of the shoreline modification and those for unaltered beaches were placed along the most recent high-tide wrack line; elevations for each site were estimated using local tide prediction tables. Because of the intrusion of shoreline modifications into the intertidal zone, transects below altered beach sites were typically located 1.0 to 1.5 m lower than the natural beach transects (Table 1).

Table 1 Sampling sites for paired sampling with coordinates, dates sampled, and land/sea interface elevations. Interface elevations are in meters above mean lower low water (MLLW); note that all natural beaches were sampled at the approximate elevation of mean higher high water (MHHW)

Site location	Code	Coordinates	Dates sampled (2001)	Type	Interface elevation (MLLW)
Carkeek Park	CKP	47° 71' 19.7" N	3/25, 4/9, 4/25, 5/8, 6/8	Natural	3.5 m ~ MHHW
		122° 37' 79.3" W		Altered	2.4 m
Dockton Beach	DBP	47° 37' 16.3" N	3/27, 4/11, 4/23, 5/10, 6/4	Natural	3.8 m ~ MHHW
		122° 45' 04.9" W		Altered	2.3 m
Richmond Beach	RBP	47° 45' 41.7" N	3/24, 4/8, 4/25, 5/6, 6/7	Natural	3.5 m ~ MHHW
		122° 22' 59.4" W		Altered	2.5 m
Seahurst Beach	SHB	47° 28' 23.5" N	3/26, 4/10, 4/24, 5/9, 6/6	Natural	3.6 m ~ MHHW
		122° 22' 2.5" W		Altered	2.6 m

Fig. 1 Map of Puget Sound, WA, USA, showing sites used for paired and synoptic sampling. For synoptic sampling, beach types are indicated by representative symbols



The following data were collected along each transect: (1) visually estimated percent cover of organic debris from ten 0.3-m² quadrats; (2) five benthic infauna cores (sample area=78.5 cm²) to a depth of 15 cm; (3) five grain size cores similar to the infauna cores; and (4) five insect fallout trap samples (30-cm×20-cm×15-cm plastic tubs filled with 3–5 cm of soapy water, left out for 20 h). The benthic infauna core samples were fixed with ~10% buffered formalin. Insect fallout trap samples were collected, sieved at 0.156 mm, and preserved in 70% isopropyl alcohol in the field.

Grain size samples were processed using gravimetric dry sieve analysis (Folk 1964), with grain size classified according to the Wentworth Scale (Wentworth 1922). Uniformity coefficients were calculated for all samples to

determine grain size distribution. Animals were elutriated from the benthic infauna core sediments by agitating the sample with running water and decanting the water through a 0.5-mm sieve, and then were transferred to 70% isopropyl alcohol. Animals were sorted and identified using a dissecting microscope; insects were identified to the family level, and gammarid amphipods and other marine crustaceans were identified to species according to protocols for assessing juvenile salmon prey items (Simenstad et al. 1991).

Synoptic Sampling

To assess effects of shoreline modification at a larger spatial scale, we compared 26 sites of mixed shoreline types with

and without shoreline modifications, including bulkheads and rip-rap revetments, and with varied backshore vegetation, on both the east and west sides of the central Puget Sound Basin (Fig. 1). Sites were selected haphazardly across the study area and were staggered to spatially alternate natural and altered sites. This study component included only sites with relatively high-elevation shoreline modifications (above MHHW). Ten of the sites were natural beach stretches, and 16 had shoreline modifications. Backshore vegetation, which was not related to shoreline type, occurred at ten sites. A site was considered vegetated if trees or shrubs occurred within 50 m of the beach.

Synoptic sampling was conducted once in July (prior to seasonal wrack deposition) and once in September 2001. The latter date was selected to capture invertebrate assemblages associated with late summer deposition of green algae *Ulva* spp. (Nelson 2001) and eelgrass (*Zostera* spp.) wrack (both *Ulva* spp. and eelgrass are dislodged by early fall storms and deposited on beaches). In the September sampling, wrack deposition was categorized using a code from 1 to 5, with 5 being abundant wrack and 1 being none. All sampling was conducted at a standard elevation of approximately +3.6 m MLLW based on the strand line from the previous higher high tide. Therefore, this sampling design provided a direct comparison of benthic infauna that had colonized the same beach elevation at both altered and natural beaches, and differences in assemblage composition reflected shoreline alteration influences other than direct elevation effects.

At each site, we recorded GPS position and qualitative data about the site/beach stretch (e.g., backshore vegetation, shade, beach slope, substrate type, and wrack). Five benthic cores were collected haphazardly along the upper-strand line from the previous high tide, to a depth of 10 cm, using a 19.6-cm² PVC corer. Laboratory methods and taxonomic sorting were the same as for the paired sampling.

Statistical Analysis

For both study components, uncommon taxa were grouped into broader taxonomic categories for analysis. All benthic infauna taxa were analyzed in the paired sampling; however, only taxa considered important juvenile salmon prey and those considered to be characteristic of the supratidal zone were processed from insect fallout trap samples due to the high number of rare taxa. All abundance data were log-transformed [$b_{ij} = \ln(x_{ij} + 1)$] prior to statistical analyses to address the assumption of normality associated with analysis of variance (ANOVA).

Data were analyzed using a univariate two-factor ANOVA (Zar 1999) in SPSS ver.11.5 for Windows®, using the General Linear Model tool. Factors in the analysis included site (“site”), beach type (“type”), and the interac-

tion effects between them; both factors were considered fixed, as sites were not randomly selected. Post hoc tests for analysis of site differences were run in SPSS using the Tukey “honestly significant difference” test (multiple comparisons, Tukey; Zar 1999). The critical value in assigning statistical significance was $\alpha=0.05$.

For the synoptic sampling, we used ANOVA to test the effects of sampling event (July/September), shoreline type (altered/natural), vegetation (presence/absence), and wrack (presence/absence) on each taxon. To explore relationships between the invertebrate assemblage and environmental variables (shoreline type, vegetation, and wrack), multivariate analysis was conducted using the count data for benthic infauna cores (PC-ORD for Windows v.5.0, McCune and Grace 2002). Nonmetric multidimensional scaling (NMDS) was performed using the Bray–Curtis dissimilarity measure (also known as the Sorenson distance measure). We used the “slow and thorough” autopilot mode in PC-ORD.

Results

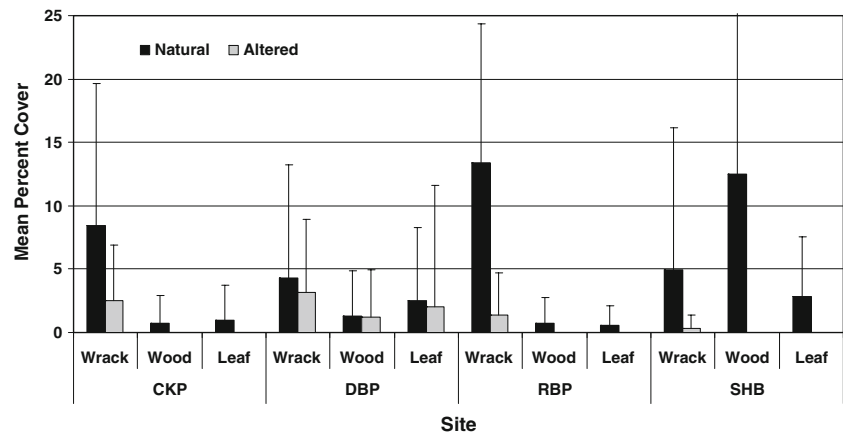
Paired Sampling

Percent Cover Quadrats

Each of the three organic debris components (marine wrack, woody debris, and leaf litter) were analyzed separately (Fig. 2), given different sources of input to the supratidal zone. Overall, organic debris was observed in greater abundance at natural beach sites, although the differences were least pronounced at site DBP. Wrack was present in small quantities in a thin strand line, never comprising more than 15% of the cover. Wrack consisted primarily of eelgrass (*Zostera* spp.) and ulvoid algae (*Ulva* spp.); other algae such as *Porphyra* spp., *Enteromorpha* spp., and unidentified red and brown species occurred in small amounts. ANOVA results showed wrack to be significantly different between altered and natural sites ($p<0.001$) with no significant interactions between factors.

Wood and leaf litter had lower percent cover than wrack, with the exception of wood at the natural portion of site SHB, where it was abundant (all woody debris accumulations were ephemeral, reflecting logs deposited on the beach during extreme high tides or those falling onto the shoreline directly from the backshore). There were significant differences between natural and altered beaches for wood ($p<0.002$), but there was also an interaction effect (type \times site) because site SHB was anomalous (Tukey HSD, abnormally high percentage of wood at the natural beach). Leaf litter was marginally nonsignificant ($p<0.054$) for beach type, with no interaction effects.

Fig. 2 Percent cover of organic material at paired sites. Error bars show one standard deviation. *Wrack* = algae and eelgrass; *Wood* = ephemeral woody debris; *Leaf* = leaf litter. *CKP* = Carkeek Park, *DBP* = Dockton Park, *RBP* = Richmond Beach, *SHB* = Seahurst Beach



Grain Size

Beach substrate grain size at all sites ranged from gravel to fines (silt and clay, with diameter $<62.5 \mu\text{m}$). Sediments were dominated by gravel ($<2,000\text{-}\mu\text{m}$ diameter) at altered beaches and medium sand ($<250\text{-}\mu\text{m}$ diameter) at natural beaches. The only site with very fine sand or silt (sediments with grain size diameter $<62.5 \mu\text{m}$) was site DBP. Beach sediments were poorly sorted; only site DBP-Natural was considered to be well sorted. Uniformity coefficients (C_u) were high at all sites. It should be noted that distribution coefficients were approximate because sediments were lumped into relatively large size range bins for analysis.

Benthic Infauna

Benthic invertebrate data were characterized by patchy distributions, many zeros, and few dominant taxa. Only two infauna taxa, oligochaetes and nematodes, appeared consistently, occurring in greater than 90% of the samples. However, the densities of these taxa were quite variable, differing by one or more orders of magnitude among cores (grand mean nematodes = 37.8 per core, $SD = 127.7$; grand mean for oligochaetes = 96.2 per core, $SD = 226.0$). Only six other taxa—amphipods of the family Talitridae (“beach hoppers” referred to here as “talitrids”), collembolans (springtails), coleopterans (beetles), acari (mites), crustaceans (crustaceans other than talitrids), and immature insects—had greater than 10% frequency of occurrence and often occurred in low densities ($75\text{--}400 \text{ m}^{-2}$). While patterns of occurrence were mostly consistent at all four paired sites, at both natural and altered stretches, site DBP often had abundances considerably different from the other three sites. For example, this site had more abundant insects, nematodes, and oligochaetes and fewer talitrids.

The mean number of taxa per sample for all sites combined was less than 8, with all natural and altered beaches combined averaging 8.6 and 6.9 taxa, respectively

(Fig. 3, standard deviations 1.4 and 1.8, respectively). Only site CKP had a greater mean number of taxa at the altered beach site. ANOVA showed differences in taxon richness to be marginally nonsignificant ($p < 0.058$) between natural and altered sites; neither site nor the interaction factor was significant. When sites were analyzed individually with paired sample t tests, only two natural sites—RBP and SHB—had significantly different (higher) taxon richness compared to their corresponding altered sites (one-tailed test, RBP, $p < 0.003$ and SHB, $p < 0.02$).

Mean abundance data were lumped into five broad taxonomic groups (crustaceans, insects, nematodes, oligochaetes, and talitrids) for analysis (Fig. 4). Talitrids, a known upper beach fauna, were more abundant at natural beach sites than at altered sites. While beach type was a significant factor, site and the interaction factor (type \times site) were also significant; post hoc tests for site showed sites DBP and CKP to be different from SHB and RBP. Densities of three of the five analyzed taxa (talitrids, insects, and crustaceans) were significantly different between altered and natural beaches (i.e., “type” result was significant); however, in all cases, the site factor and the interaction factor (type \times site) were also significant, indicating high between-site variability. There were no

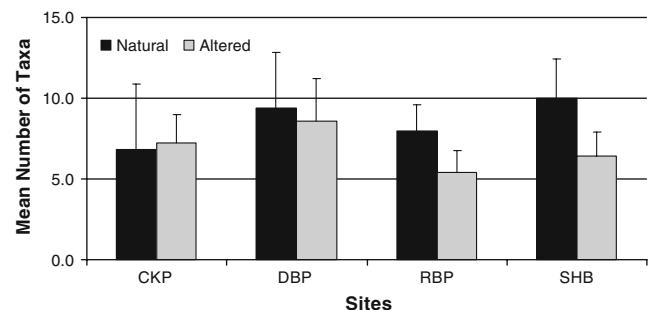
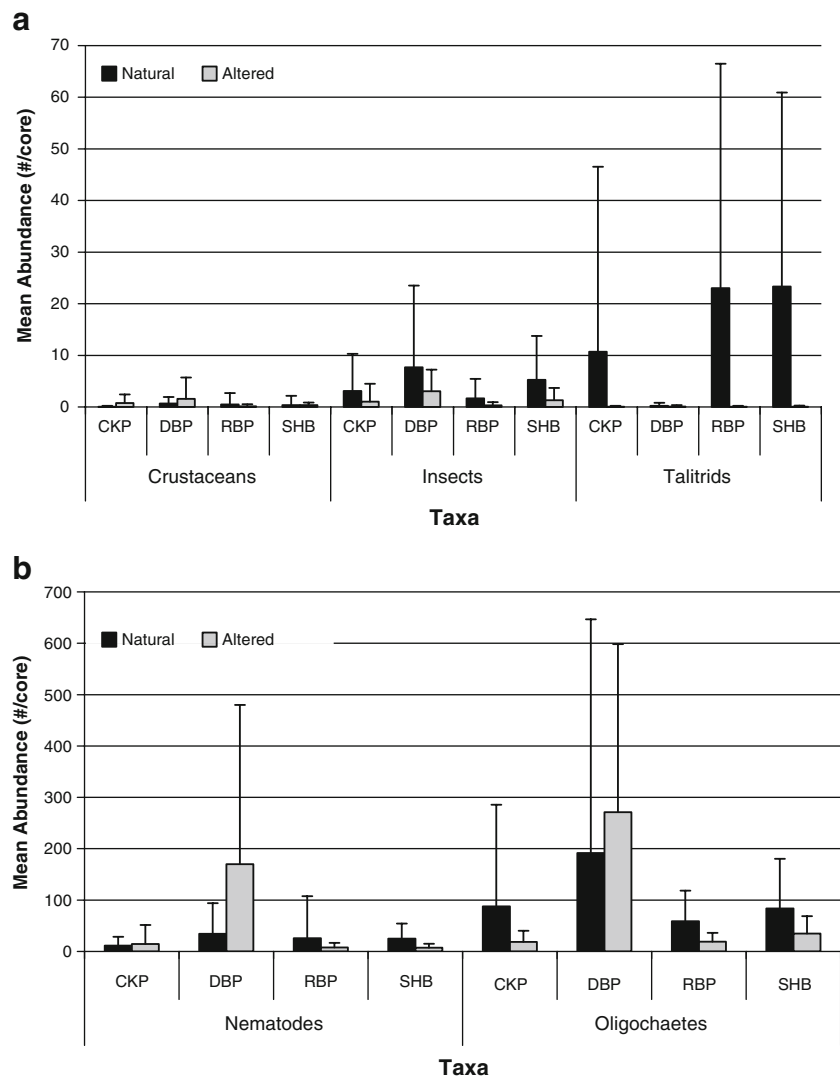


Fig. 3 Mean number of benthic infauna taxa at paired sampling sites. Error bars represent one standard deviation

Fig. 4 Mean abundances of benthic invertebrates from paired sampling benthic cores for all sites. Crustaceans, insects, and talitrids (**a**) have abundances an order of magnitude lower than nematodes and oligochaetes (**b**). Error bars show standard deviations



statistically significant differences for beach type for oligochaetes and nematodes; however, as with the other taxa, there were site and interaction effects. In all cases, site DBP proved to be different from the others in post hoc tests. For the taxa with significant beach type differences, natural beaches had higher densities of insects and talitrids, and altered beaches had higher densities of crustaceans.

Insect Fallout Traps

Insect fallout traps captured 105 taxa, although only 21 taxa (19%) occurred in more than 10% of the samples (Table 2). Organisms ranged from crustaceans such as talitrids and oniscoid (terrestrial) isopods, to 13 orders of insects, including winged and crawling forms. Many taxa were rare, with 45 taxa occurring in less than 2% of the samples. Site taxon richness ranged from 22 (SHB-Altered) to 61 (SHB-Natural), with an average of 39.9 taxa per site

(Fig. 5). Site SHB-Natural had the greatest taxon richness and the greatest difference in density and diversity between altered and natural beaches. Mean taxon richness at natural beaches (50, SD=10.2) was higher than at altered beaches (29.8, SD=5.9); ANOVA showed both type and site to be significant ($p < 0.002$), and post hoc tests indicated that site CKP was most different from the other sites. When sites were analyzed individually, taxon richness was significantly higher at the natural beach only at site SHB.

Dipterans (midges and other true flies), collembolans, and talitrids were the most abundant taxa (Fig. 6). Collembolans were the most common taxon, occurring in 67% of the samples, with an average density of 132 m⁻² (7.9 animals per sample, SD=13.3). They were more dense at natural than altered beach sites ($p < 0.024$). Dipterans, mainly consisting of chironomids (midges), appeared in 98% of the samples analyzed. Dipterans were the most taxon-rich order in the samples, with 30 different families. However, half of these families had fewer than five

Table 2 Taxa occurring in greater than 10% of paired insect fallout trap samples, with taxonomic group used for analysis in the first column

Taxonomic group (for analysis)	Taxon
Other	Acari Arachnid
Diptera	Cecidomyiidae Ceratopogonidae Chironomidae Dolichopodidae Empididae Muscidae Phoridae Psychodidae Psyllidae Sphaerospheridae Tethinidae Tipulidae
Collembola	Sminthuridae Isotomidae
Coleoptera	Ptiliidae Staphylinidae
Thysanoptera	Thysanoptera
Talitridae	<i>Traskorchestia</i> spp.

occurrences when all samples were combined; other common dipteran families are listed in Table 2.

ANOVA results for chironomids and other dipterans (excluding chironomids) showed significant type and site differences (as well as the site \times type interaction), suggesting that the distribution of these taxa was variable between beaches. Post hoc tests showed site CKP to be different from the other sites; it had higher densities of both dipterans and chironomids at the altered site, which was not the case at any of the other sites where these taxa were more numerous at the natural beach.

We specifically analyzed the insects of the order Homoptera further because they were relatively abundant in Chinook salmon diets collected at these and other similar sample sites in Puget Sound (Brennan et al. 2004). Homopterans occurred in 35% of the fallout trap samples. Eight homopteran families were represented, although 78% were from three families: Aphididae, Psyllidae, and Cicadellidae. Homopteran abundance was significant with respect to beach type ($p < 0.03$; site was not significant, $p < 0.1$).

Talitrid amphipods of several species—*Megalorchestia pugettensis* and *Traskorchestia* spp.—were caught in the fallout traps. Differences in densities of talitrids were significant ($p < 0.005$) for site and type, as well as the interaction factor. In post hoc tests, site SHB was significantly different from other the sites, due to a much

higher abundance of talitrid amphipods. In all cases, natural beaches had significantly higher densities than altered beaches, but high variances suggested that talitrids were patchy in both space and time.

Synoptic Sampling

Similar to results from the paired samples, the synoptic benthic invertebrate samples were sparsely populated, with few dominant taxa, low taxon richness, and high variances around the density estimates (Table 3). Oligochaetes were ubiquitous and numerically dominant but variable in abundance, ranging from 510 to 400,000 m^{-2} , with an average of 18,000 m^{-2} (36.4 per core). Nematodes were also ubiquitous, but less abundant, averaging 2,500 m^{-2} (5.4 per core); mean abundances for both taxa were lower than those in the paired sampling due to low abundances in the July sampling. Adult insects, including dipterans, coleopterans, collembolans, and other orders, were found in more than 90% of the samples with a mean of 6.5 insects per sample overall (3,315 m^{-2} , $SD=9.25$); in July and September, the mean was 3.1 and 9.9 insects per sample, respectively. Immature insects (mostly dipterans) occurred in over 60% of the samples on each of the sampling dates with mean abundances of 255 m^{-2} (July) and 367 m^{-2} (September).

Overall, 3.5 times more organisms were collected in September samples than in July samples. Densities of talitrids, oligochaetes, and adult insects were significantly higher in September (Table 4); mean abundance of immature insects and nematodes showed no difference between sampling dates, although in both cases mean abundance and total numbers of animals collected were higher in the September samples. Frequency of occurrence of taxa was mostly consistent between sampling events, although talitrid amphipods and some insects (coleopterans and dipterans) occurred more frequently on the second sampling date.

Shoreline type (altered/natural), the main treatment effect, was significant only for nematodes ($p < 0.034$) and

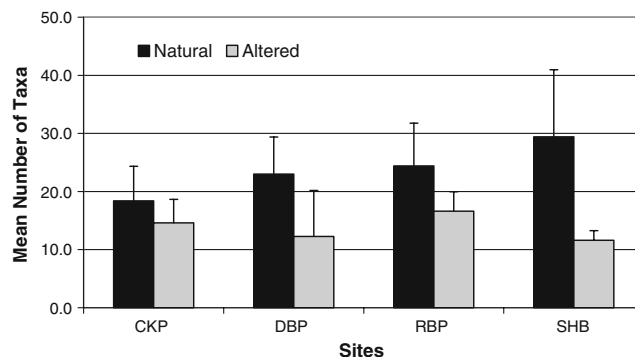
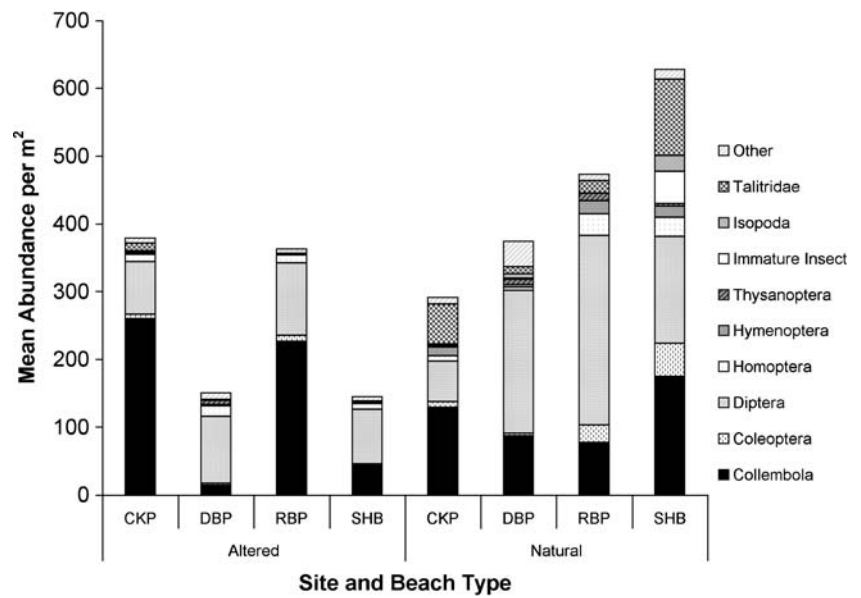


Fig. 5 Mean number of insect taxa from fallout traps at paired sampling sites. Error bars represent one standard deviation. Only at site SHB was taxon richness significantly different

Fig. 6 Density (abundance per square meter) of the most common taxa from paired insect fallout traps, per site and treatment. Collembolans and dipterans are the dominant taxa



oligochaetes ($p < 0.046$); all other taxa (e.g., talitrids, oligochaetes, immature insects, and adult insects) showed no significant difference in mean abundance between shoreline types (Fig. 7). Because so few crustaceans appeared in these samples, this taxon was omitted from the analysis. No type \times time interaction effects were observed in the models.

To evaluate the community as a whole, samples were plotted in multivariate space using NMDS. NMDS results for all samples showed that the best solution for analysis, based on Bray–Curtis dissimilarity, was a three-dimensional model, with a final stress of 9.08 and a final instability of < 0.0001 , achieved after 102 iterations. Three factors were overlaid on the NMDS plot: shoreline type (altered/natural), vegetation (ranked from 1 to 5, based on estimated percent cover), and wrack (ranked 1–5, based on estimated percent cover). Despite a reasonably good model fit, there was little correlation among these factors and the position of samples in the

model plot. For example, shoreline type resulted in a Pearson’s correlation coefficient for axis one of $r^2 = 0.025$; these results indicate a lack of correlation between the variable and the arrangement of the samples in multivariate space. The wrack and vegetation cover variables were similarly nonsignificant ($r^2 = 0.119$ and $r^2 = 0.179$ for the primary axis, respectively).

Discussion

Results from the paired sampling demonstrated an impact on invertebrate communities where shoreline modifications intruded into the upper intertidal and supratidal zones; the results from the synoptic sampling, conducted where shoreline modifications infringed less on the intertidal zone, were less conclusive. Combined, these findings suggest that effects to the invertebrate communities occur as a result of shoreline modifications (in this study, mostly

Table 3 Comparison of abundances for common taxa from both paired and synoptic sampling. For the synoptic sampling, mean abundances are for all sites, by beach type, for both sampling dates. For the paired sampling, mean abundances are for all four sites by beach type. All abundances are mean number per square meter

Sampling regime	Talitrids	Crustaceans	Oligochaetes	Nematodes	Collembolans	Insects
Synoptic						
Natural	1,785	0	9,833	3,060	2,315	337
Altered	1,014	6	6,515	803	1,039	281
Paired						
Natural	1,819	47	15,086	3,274	238	314
Altered	7	91	11,236	6,608	79	103

Table 4 ANOVA results of selected taxa from synoptic sampling and association with sampling event and shoreline type

Taxon	Sampling event (July/September)	Shoreline type (Natural/armored)
Talitridae	<i>0.017</i>	0.256
Crustacea	–	–
Oligochaeta	<i>0.009</i>	<i>0.046</i>
Nematoda	0.408	<i>0.031</i>
Immature Insecta	0.629	0.588
Insecta	<i>0.020</i>	0.307

Significant results are in italics. Where results are significant for sampling event, the September event had higher abundances and for shoreline type; the natural beaches had greater abundance. Crustaceans were omitted from this analysis due to infrequent occurrence

rip-rap and bulkheads), especially when they displace and degrade the supratidal zone by coarsening sediments and shifting the land–sea interface lower in the intertidal. This results in changes to supratidal invertebrate assemblages, significant decreases of some ecologically important taxa, and a reduction in deposited marine wrack. Although the results of our paired and synoptic sampling were not directly comparable, they suggest that where shoreline modifications occur above MHHW, effects on the supratidal fauna are less pronounced.

The armored beaches in our paired study all had beach–interface elevations below mean higher high water, eliminating the beach and turning the supratidal zone into a steep or vertical hard substrate. This displacement of the supratidal zone was reflected in a shift in the invertebrate assemblages: fewer talitrid amphipods (“beach hoppers”) and insects and more marine crustaceans occurred at altered beaches. While our study did not assess the mechanisms for the change in assemblage, it is likely due to a combination of: (1) elevation differences—altered sites were lower in the intertidal zone, resulting in a different fauna; (2) replacement of sediments typical of supratidal beaches (sand and gravel) with large rip-rap boulders and other coarser substrates not suitable for the characteristic beach fauna of this zone; and (3) lack of areas available for wrack buildup at the altered beaches, largely eliminating invertebrates associated with wrack.

In the synoptic sampling, we found that many of the major invertebrate groups had higher overall densities at the natural beaches as compared to the altered beaches. However, except for oligochaetes and nematodes, none of these differences were statistically significant. One explanation for this is that all of the sites selected for synoptic sampling had intact shore to MHHW elevation, resulting in more intact beach habitat than was the case at paired sample sites. This suggests that there is an elevation threshold

below MHHW at which shoreline armoring effectively eliminates supratidal beach habitat by shifting the beach–water interface to an area of higher wave energy and longer submergence. Our results suggest that shoreline modifications established above MHHW may have fewer direct impacts on supratidal benthic organisms, especially talitrid amphipods. However, armoring at any elevation can have other physical impacts on the intertidal zone such as decreasing sediment deposition, changing substrate grain size, removing vegetation (NRC 2007), increasing temperature (Rice 2006), altering the microhabitats that the animals in this zone use, and interrupting connectivity between the aquatic and terrestrial ecosystem.

The negative impact of shoreline modifications on the deposition of organic debris, especially wrack, is noteworthy because it is habitat for much of the supratidal fauna that contributes to the nearshore detritus-based food web (Dugan et al. 2003; Romanuk and Levings 2003a). Higher densities of invertebrates in synoptic samples taken in autumn may have been due to increased algae and eelgrass deposition typically occurring at this time in Puget Sound (Nelson 2001). Adult and immature insects were collected in the synoptic sampling infauna cores in moderate abundances; these organisms are associated with wrack, which serves as habitat and a food source. While no seasonal difference was noted in the synoptic sampling for talitrids, possibly due to their low abundance, these amphipods are mobile scavengers and are known to be especially dependent on wrack, which provides habitat (moisture and shelter) and food for them and other invertebrates (Bousfield 1981; Koch 1989; Pank 1997; Colombini et al. 2000; Jedrzejczak 2002).

In addition to decreasing wrack, armored sites change the supratidal ecotone into steeper hard substrate, a suboptimal habitat for semiterrestrial talitrids and other

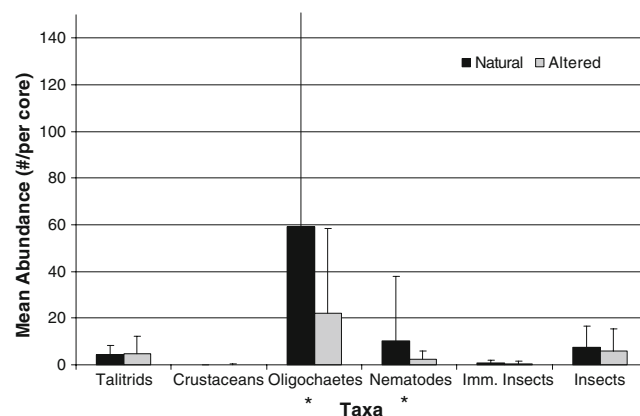


Fig. 7 Mean abundance of prominent taxa found in benthic core samples from synoptic sampling of Puget Sound beaches at sites with (armored) and without (natural) shoreline modifications. Error bars represent standard deviations from the mean, and asterisks indicate statistical significance ($\alpha=0.05$)

supratidal fauna. At altered beaches, talitrids and other supratidal invertebrates appear to be at least partially replaced by sediment-dwelling crustaceans. Although semi-terrestrial talitrids are not commonly found in the diets of juvenile Puget Sound salmon (Simenstad et al. 1979; Brennan et al. 2004), they are prey for a number of bird species, staphylinid beetles, and small mammals (Bousfield 1981; Pank 1997; Hubbard and Dugan 2003). The connection between talitrids and the nearshore food web is still relatively unknown and should be explored further because these organisms can be abundant at natural beaches and are thought to be important to marine nutrient cycling in the supratidal zone (Polis and Hurd 1996). A recent study on Vancouver Island, British Columbia, by Lewis et al. (2007) showed shore crabs, *Hemigrapsus nudus*, to forage extensively on talitrids and to use wrack mats for refuge. In turn, *H. nudus*, which is common on Pacific Northwest beaches, is consumed by birds and other animals (Toweil 1974; Vermeer 1982). Thus, shoreline armoring impacts on talitrids may affect higher trophic levels as well.

The results from our paired sampling insect fallout trap samples showing lower taxonomic diversity at armored beaches may be a result of proximal effects of shoreline modifications. For example, the elimination of shoreline vegetation and decrease in the deposition of wrack and its associated insects may explain the differences between altered and natural beach sites. Studies show that marine backshore vegetation hosts a variety of insect species (Romanuk and Levings 2003a), and vegetation is likely to alter the physical dynamics of beaches through input of organic debris and thermal regulation of the substrate, via shading (Jedrzejczak 2002). Although we did not explicitly measure insects associated with vegetation (the inclusion of backshore vegetation in the synoptic analysis was opportunistic), our data from insect fallout traps at paired armored and nonarmored Puget Sound beaches suggest that backshore vegetation increases insect diversity and abundance: the armored site with less vegetation (e.g., CBP) had consistently lower taxon richness than the natural beaches with extensive vegetation (e.g., SHB). As with the benthic core results, this result may indicate that armoring is accompanied by a significant qualitative change in the nearshore habitat for many species and loss of some biological function.

Our paired sampling sites were intended to be representative of beach types present within the central Puget Sound basin, and there were some differences among the sites with regard to energy regime (and thus, sediment grain size, turbulence, and other physical factors) that impacted invertebrate distribution and abundance. Dockton Beach (Site DBP) was different from the other sites in being located in a protected embayment. In many cases, this site was an outlier, with different invertebrate assemblages at

both altered and natural sites as compared to the other three sites. The inclusion of this site led to increased between-site variance resulting in many interaction factors in the statistical models, often reducing our ability to observe a main treatment effect (beach type). Focusing on one beach type (such as that represented by CKP, RBP, and SHB) and adding additional sites similar to those may have resulted in a more robust statistical design, albeit one limited to a specific beach type. A recent summary by Schlacher et al. (2008) addresses many of the sampling issues surrounding sandy beach surveys and provides guidance on how best to sample these habitats, with regard to fauna.

The supratidal zone is an ecotone possessing attributes of an ecological edge between terrestrial and marine ecosystems (Carefoot 1977; Naiman and Decamps 1990; Simenstad et al. 1997). While this study indicates that shoreline armoring has localized impacts on supratidal biota, the cumulative impacts of large-scale shoreline armoring on the ecological functions of this zone are still largely unknown. This study illustrated both direct and proximal effects of armoring and, along with other recent studies, begins to describe linkages between the terrestrial and marine systems via the supratidal zone, e.g., through foraging of terrestrial and marine animals in the supratidal and transfer of terrestrial detritus from upland sources. However, many of these linkages are still poorly understood, and continuing to quantify the mechanisms and magnitude of ecological contributions to the supratidal from shoreline vegetation, deposited wrack and invertebrates are important in determining large-scale biological impacts of shoreline armoring.

Acknowledgements We thank C. Rice, J. Gregg, M. Nelson, and the Wetland Ecosystem Team for field and lab support. M. Dethier and B. Miller provided comments on early drafts of the manuscript. R. Latour provided help with statistical analysis, and three anonymous reviewers provided suggestions that strengthened the final version. Support for this project was provided by King County, Washington Department of Natural Resources, US Army Corps of Engineers Seattle District, NOAA Fisheries NWFSC, and the University of Washington School of Aquatic and Fishery Sciences.

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