Changes Following Restoration to the Shoreline at Seahurst Park

February 28, 2019

PUGET SOUND ECOSYSTEM
MONITORING PROGRAM

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WASHINGTON STATE DEPT OF
NATURAL RESOURCES

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- Our collaborator Jason Toft and his colleagues at University of Washington's Wetland Ecosystem Team (WET).
- The City of Burien.
- Many field assistants.

Contents

Executive Summary

The Washington State Department of Natural Resources (DNR) is steward of 2.6 million acres of state-owned aquatic land. As part of its stewardship responsibilities, DNR monitors the condition of nearshore habitats. Monitoring results are used to guide land management decisions for the benefit of current and future citizens of Washington State. This work also supports the Puget Sound Partnership's effort to protect and restore Puget Sound.

Intertidal habitats are an important constituent of the nearshore ecosystem, and they are vulnerable to both terrestrial and aquatic stressors. One indicator of intertidal habitat health is its biotic community – the complex of flora and fauna living in and on the beach. DNR and the University of Washington (UW) have collaboratively monitored intertidal biotic communities since 1997.

Our 2018 research effort continued earlier work that examined the time-course of shoreline restoration, focusing on efforts to remove shoreline armoring at Seahurst Park, in central Puget Sound. This site has been unusually thoroughly studied, with biotic surveys done before the shoreline was armored (1974), before and after a first phase of armor removal (2005), and before and after a second phase of additional armor removal (2014). While sampling efforts have not always been consistent in terms of methods or elevations sampled, the data do suggest that some biotic and geomorphological changes can be attributed to armoring and restoration activities. Other changes likely relate to long-term increases in human usage of the Park, to development in the watershed, and/or to natural interannual variation in these dynamic habitats. The 10-year-old restored site has healthy biotic communities (i.e., similar to a nearby Reference site) at all elevations, from the backshore to the low-tide terrace. The data suggest an increase in both eelgrass cover and clam populations at this site. The 4-year-old restored site is clearly still relatively 'raw', showing substantial year to year variation in sediment and biota. The biotic communities on the mid and low-shore beach are affected both by sediment type (especially the presence of cobbles and abundance of sand), whereas insect communities on the high shore are likely dependent on the newly planted backshore vegetation, which is still becoming established. Data from this and other sites show that armor-removal efforts result in a set of changes that are likely to stabilize in the 5-10 year timeframe.

Introduction

The Washington State Department of Natural Resources (DNR) is steward of 2.6 million acres of state-owned aquatic land. The Aquatic Resources Division of DNR manages these aquatic lands for the benefit of current and future citizens of Washington State.

Program Background

The overall goal of the Intertidal Biotic Community Monitoring Project is to assess the condition of intertidal biota in greater Puget Sound. This work supports DNR's mandate to ensure environmental protection of state-owned aquatic lands that it stewards (RCW 79.105.030). Additionally, this work supports the Puget Sound Partnership's effort to protect and restore Puget Sound through tasks that are defined in the Puget Sound Action Agenda (Puget Sound Partnership 2014, 2016, 2018), and in the monitoring plans by its predecessor, the Puget Sound Action Team (Puget Sound Action Team 2007).

Intertidal and shallow subtidal habitats are an important constituent of the nearshore ecosystem. They are diverse and productive, harboring extensive populations of algae and seagrasses that contribute to food webs (both nearshore and in deeper water) and provide habitat for many other organisms (e.g., Duggins et al. 1989). Invertebrates that live in intertidal habitats are important in recycling of detritus (e.g., Urban-Malinga et al. 2008) and reducing water turbidity (e.g., Peterson and Heck 1999), as well as providing food for shorebirds, nearshore fishes, commercially important invertebrates such as crabs, and humans. Human populations use the intertidal zone for a wide range of reasons, including recreation, education, and harvesting of marine resources (Dethier et al. 2017). Intertidal and nearshore communities also serve as useful 'indicators' of ecosystem health. Because most organisms in these habitats are relatively sessile and thus unable to move away from stressors, they are vulnerable to both natural and anthropogenic changes in terrestrial and aquatic ecosystems. Demonstrated examples include sensitivity to changes in rainfall (Ford et al. 2007), ocean temperatures (Schiel et al. 2004), local pollution (Hewitt et al. 2005), sedimentation (Muth et al. 2017), and larger-scale factors such as the North Atlantic Oscillation index (Labrune et al. 2007).

DNR and the University of Washington (UW) have jointly monitored biotic communities since 1997. The intertidal biotic community sampling design and statistical analyses are described in peer-reviewed publications (Schoch and Dethier 1995, Dethier and Schoch 2005, Dethier and Schoch 2006) and multiple technical reports available through DNR at https://www.dnr.wa.gov/programs-and-services/aquatics/aquatic-science/nearshore-habitat-biotic-community-monitoring.

Intensive, relatively recent monitoring programs in Puget Sound are able to quantify changes in parameters such as eelgrass abundance or shoreline biota on the time scales of years (Gaeckle et al. 2009, Dethier and Berry 2009). However, quantifying long-term changes – for example, over the 100+ years of development along Puget Sound's

shorelines – is not possible, except on a very coarse scale, because of the absence of data. This report summarizes 2009-2018 shoreline monitoring at Seahurst Park in Seattle, where a combination of historical data and recent surveys have been used to explore long-term changes in intertidal communities in Central Puget Sound (Dethier and Berry, 2010). In Puget Sound, a human activity known to be detrimental to the health of the marine ecosystem is armoring of the shorelines. Armoring is listed as a significant "threat" by the Puget Sound Partnership (PSP 2014, 2016, 2018) and it appears as a factor disrupting natural processes in the conceptual models of the Puget Sound Nearshore Ecosystem Restoration Project (Simenstad et al. 2006). Currently, it is estimated that at least 30% of Puget Sound's shorelines are armored (Simenstad et al. 2011). The proportion for south-central Puget Sound is much higher, around 64%, and the demand for shoreline protection structures is almost certain to increase with heightened concerns about erosion caused by sea-level rise. Recent local efforts have gathered data documenting negative impacts of armoring on physical and biological features of nearshore ecosystems, especially for gravel beaches of the sort that dominate the Salish Sea (Sobocinski et al. 2010; Dethier et al. 2016; Dethier et al. 2017 and references therein). These data have helped to spur efforts to restore shoreline processes by removing armoring, and there are quantitative efforts to track new vs. removed armoring throughout the Sound (Hamel et al. 2015).

Shoreline armoring is thought to affect the nearshore environment by as many as five different mechanisms: 1) Encroachment over the upper shore, directly burying habitat ("placement loss"); 2) Disconnection of terrestrial and marine ecosystems, e.g., via loss of riparian vegetation and associated insects, and lack of recruitment of wrack and drift logs to the shore; 3) Sediment impoundment, preventing sediment eroding from banks from reaching the shore; 4) Active erosion, from reflection of waves off bulkheads (especially those built lower on the shore); and 5) Prevention of passive erosion, i.e., stopping the natural bank retreat that is occurring on many U. S. coastlines. Both active and passive erosion, in some circumstances, cause removal of fine sediments from the beach, thus steepening and coarsening the beach profile below armored portions (seen to some extent in Thurston County: Herrera 2005). These changes may make the beach less suitable for the many infaunal organisms that require finer sediments. A difficulty in assessing the impacts of armoring is that while some mechanisms (e.g., Encroachment) act immediately, others (e.g., Passive Erosion) may take decades to be visible. Another way to assess these impacts is by quantifying changes that occur following shoreline restoration that involves removal of armoring, to see the types and speed of changes that occur to shoreline shape and functions (see Lee et al. 2018).

In 2009 we conducted extensive sampling and analysis (Dethier and Berry, 2010) to quantify decadal-scale changes at beaches in Seahurst Park in south-central Puget Sound. We performed a historical comparison between surveys done in 1971, 1982-1983, and recent conditions (1999-2009). The park shoreline was extensively armored in 1974 (see Kohn et al. 1971, Dethier and Berry, 2010). In winter 2004-2005, Phase 1 of a larger shoreline restoration effort occurred; the City of Burien replaced approximately 1,100 feet of armored shoreline (rock-filled gabions on the high shore) in the south part of the park with a more natural riparian area, and regraded the beach to make a more gradually sloping intertidal zone (including beach nourishment). Phase 2, involving removal of an additional

1,800 feet of armoring in the northern half of the park, occurred in winter and spring of 2014, leaving only one section of shoreline at the north end of the park still armored. Sediment and vegetation were added to the upper shore, along with a small wetland area. The goal of both Phases was to restore nearshore processes, including the buildup and breakdown of wrack and the supply of sediments. Data from other UW researchers (Oxborrow et al. 2015, Toft 2016, Lee et al. 2018) show that both restoration phases have been successful in returning the upper beach and its biota to a more natural state, with logs, wrack, high-shore infauna, and insect communities becoming similar to those in relatively pristine areas nearby (see Discussion). Our studies lower on the shore, however, suggest much more unpredictable changes through time. In 2009 we found decade-scale declines in width of the beach, richness of intertidal biota, and abundances of clams (Dethier and Berry 2010). These substantial changes appear to relate to intensive human use of the park and to broad changes in sediment types, perhaps due to land use change in the watershed.

Here we explore to what extent the changes between 1971 and 2009 have been reversed by the shoreline restoration efforts in the park. The challenge is attributing changes to restoration efforts vs. to unrelated, local or regional effects. All the shorelines in the park are subject to changes occurring broadly through Puget Sound (such as in water quality and other effects of upland development) as well as to more localized effects, such as intense human use of the shoreline. As a result of our prior work, DNR-UW have data on various shoreline parameters from before Phase 1 restoration when the shoreline was largely armored (1999 to 2003 data, at 1-3 sites), to the period between the two restoration events (2009 to 2013), to after Phase 2 restoration (2015 and 2018). To accomplish our current analysis, we examine patterns in shoreline biota over the time period for which we have complete datasets from 2 elevations at 4 transect sites, and from 2009 to 2018.

Methods

We used SCALE intertidal biotic community sampling design and statistical analyses to compare shoreline conditions from 2009 – 2013 (prior to Phase 2 restoration) with those from two survey dates post-restoration, 2015 and 2018. Figure 1 shows the location of all sites where data were collected. Biological sampling was conducted each year during spring tides in late June and early July.

Prior to 2009, DNR and UW had monitored biota (1999 to 2003) at Mean Lower Low Water (MLLW) at the south end of Seahurst Park ("Park South"), as well as at sites to the north and the south of the park (named Seahurst South and Seahurst North). To focus more tightly on the locations and impacts of restoration activities, in 2009 we added 3 additional sampling locations: "Phase 1", where high-shore sampling had been done by another UW team (Toft et al. 2008) to quantify Phase 1 restoration impacts; "North Creek", to quantify changes that would occur as a result of Phase 2 restoration, and "SeaTech", at the north end of the Park, in front of the Marine Technology Lab, where armoring was to remain. We anticipated that the SeaTech site would not be directly impacted by Phase 2 restoration, but might be altered due to changes in beach morphology and creek flow just to the south. These sites can all be compared with the Reference site at the south end of the park ("Park South") where no armoring or restoration has occurred.

In 2009 we conducted SCALE sampling at each of these six sites at MLLW (described below). In addition, at the four sites within the Park (i.e., not including Seahurst South and Seahurst North), we added an additional transect at Mean Low Water (MLW: +2.8') for surface biota and infauna. Thus transects were run in the Mid zone for three years prior to Phase 2 construction (2009, 2010, 2013), and two years post-construction (2015 and 2018). Transect elevations were found using a surveying level and stadia rod, measured relative to the predicted tide at the time of the measurement.

SCALE intertidal biotic community sampling design and statistical analyses have been described in previous peer-reviewed publications (Schoch and Dethier 1995, Dethier and Schoch 2005, Dethier and Schoch 2006) and technical reports (available through DNR at https://www.dnr.wa.gov/programs-and-services/aquatics/aquatic-science/nearshore-habitat-biotic-community-monitoring). General methods are summarized here. Biotic community samples consist of mean species abundances for epibiota and infauna from 10 randomly spaced sample units along a 50 m horizontal transect. Each sample unit consists of a 0.25 m^2 quadrat to quantify abundance of surface macroflora and fauna, plus a 10 cm diameter x 15 cm deep core for macroinfauna. Percent cover is estimated for all sessile taxa in the quadrats, and all motile epifauna (organisms > ca. 3 mm) are counted. Fresh
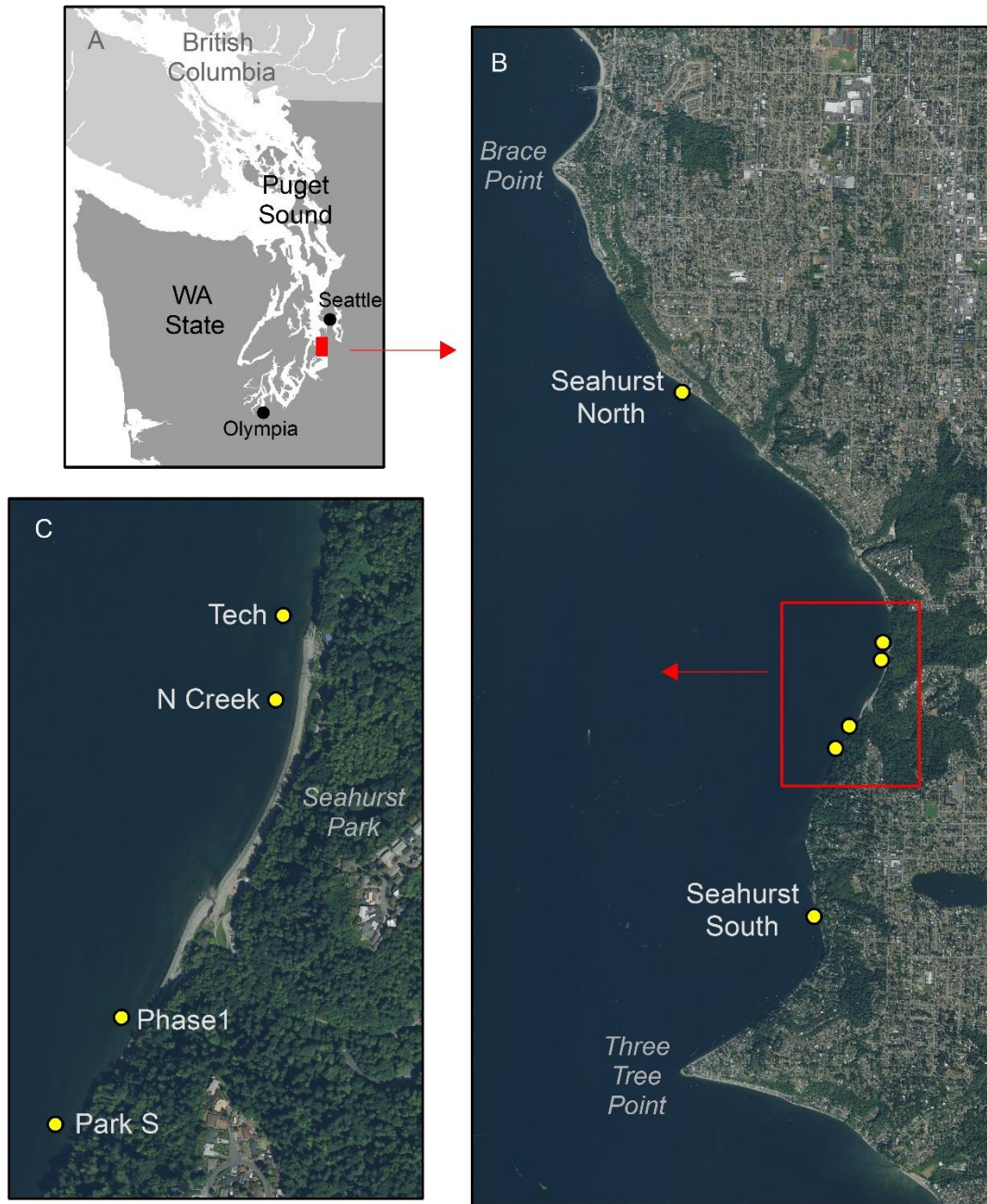
Figure 1. A. study area location in Puget Sound, Washington State; B. sites in the area; C. sites sampled in Seahurst Park.

core samples are washed in situ through 2 mm mesh sieves, thereby excluding meiofauna, juveniles of some worms, and adults of smaller crustaceans, such as cumaceans and harpacticoids. The finest taxonomic resolution used in field sampling and laboratory identification is species level, although some difficult taxa are only identified to genus or higher levels (e.g., *Pagurus* spp., Phylum Nemertea). Taxonomic references used were Kozloff (1996) for invertebrates and Gabrielson and Lindstrom (2018) for macroalgae. The multivariate analysis methods of Clarke and Warwick (1994) and PRIMER software (Clarke and Gorley 2006) were used to detect patterns in the spatial and temporal distributions of communities. The data matrix of taxon abundances was square-root transformed to reduce the contribution of highly abundant species in relation to less abundant ones in the calculation of similarity measures. We used the ordination technique of non-metric multidimensional scaling (MDS) to group communities based on the Bray-Curtis similarity metric. Graphic plots of ordination results for the two axes explaining the greatest proportion of the variance were examined for obvious sample groupings. Analysis of similarity (ANOSIM) tested the significance of hypothesized differences among sample groups. Similarity percentage (SIMPER) analyses identified the variables (species) that contributed the most to different groupings seen in the MDS plots.

Results

Temporal Patterns in the Mid Zone

Overall, the biota in the Mid zone in the Seahurst region is fairly depauperate, especially on the southern transects (Park South and Phase 1), where the substrate is primarily composed of unstable large pebbles. Transects in all years tended to have only 6-12 species (epibiota plus infauna) in total. MDS plots (Fig. 2) examining the biota across all 5 years and 4 sites show that in general, communities are more similar within a year among the transects than across years within transects; visually, points representing the biota on the transects group together by year rather than by site. This difference is quantified by the higher R values (denoting biotic similarity among points) for Years than for Sites: ANOSIM Years R = 0.30, p = 0.002; Sites R = 0.026, p = 0.33.
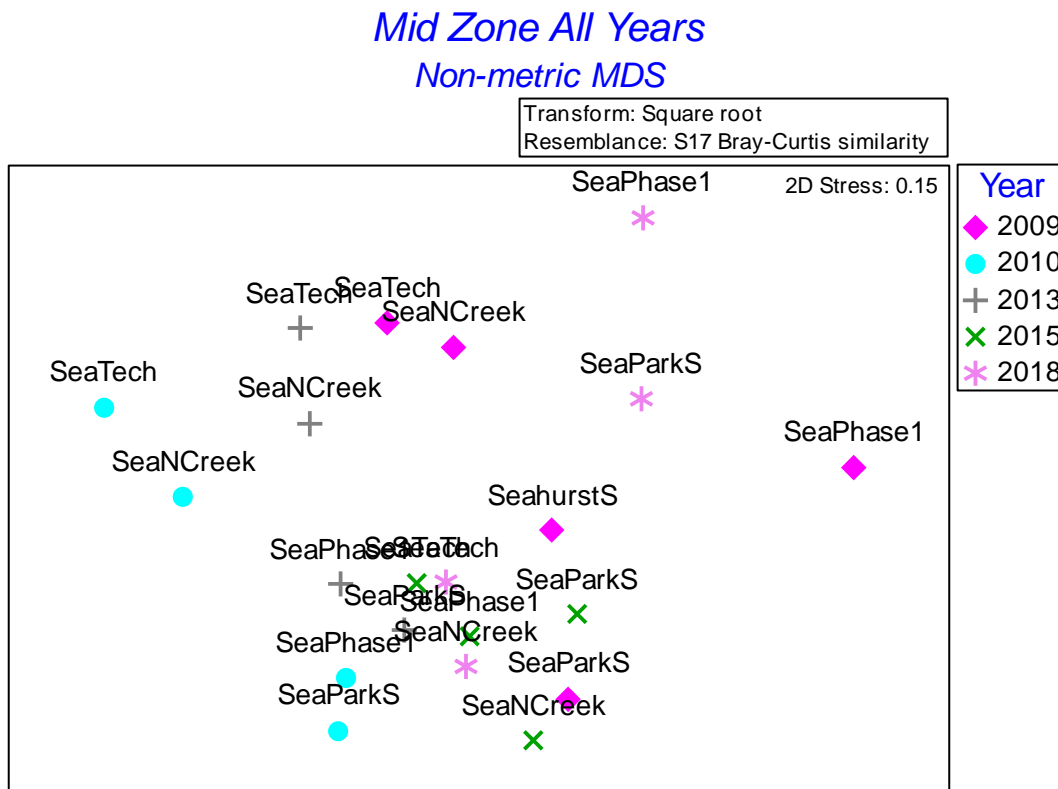

Figure 2. MDS plot of Mid-shore biota at 4 transect locations across 5 sampling dates. Points closer together indicate greater biological similarity in types and abundances of organisms.

Major biotic differences among years include: 2009 had unusually high numbers of gammarid amphipods, 2010 had more *Lottia* limpets, *Exosphaeroma* isopods, and *Mytilus* mussels; 2013 had more *Lottia, Lacuna*, and *Pagurus*; 2015 was characterized by ulvoids and gammarids; and 2018 by the carnivorous polychaete *Hemipodus*.

Biota were all quite similar (clustered green X symbols) in the Mid zone sites in 2015, a year after construction. However by 2018, North Creek and Tech still had communities similar to those in 2015, whereas the two southern beaches (Phase 1 and Park South) were quite different. Multivariate differences were clearly driven by the two southern beaches having <u>no</u> gammarids, sphaeromids, or lottiid limpets in 2018, whereas the northern beaches had many of all three groups (Fig. 3). These differences among years do not correlate with the relatively minor changes in substrates seen at those sites between 2015 and 2018 (see Fig. 4).

Figure 4 illustrates the data on surface sediments (as quantified in quadrats) seen across years in the Mid zone. Only Sand and Cobble were quantified in 2009 and 2010. The Park South (Reference) transect did not show much change in substrate types through time, with abundant pebbles and some sand. The Phase 1 restoration site got somewhat more sandy in 2013 and thereafter, but the timing (and distance from the 2014 construction site) suggest this variation is not related to restoration. Biotic changes at Phase 1 before and after this substrate change were inconsistent and relatively minor. The North Creek transect was the one sampled area whose substrate changes appeared to correspond with the restoration effort; the mid transect at this site had much less sand post-restoration (in 2015 and 2018) than pre-restoration, probably because of the diversion of the small northern creek onto a different part of the beach. With this substrate change at North Creek, there was a significant change in biota in the mid zone before (2009-13) versus after (2015-18) restoration (visible in the point locations in Figure 2). SIMPER analyses show a substantial decline in abundances of polychaetes (*Notomastus, Hemipodus, Spiochaetopterus, Glycinde),* presumably because there was less fine sediment for them to inhabit, and increases in mobile crustaceans such as sphaeromids, gammarids, and *Hemigrapsus* that can live under the pebbles.

Substrates at the nearby Tech transect show some changes through time (Fig. 4), with a steady decrease in surface sand. Following restoration there was a drop in large cobbles (i.e., from 2013 to 2015), which may have been removed as part of the restoration effort because they were not a natural part of the beach substrate. Surface sediments post-restoration thus had more pebbles and less clean sand; this difference is visible in comparisons of quadrat photographs from 2009 and 2018 (Appendix 1). With these changes in substrate in 2013 and thereafter, the Mid-shore Tech transect had more *Lottia*, barnacles, gammarids, and sphaeromids (all on or under the cobbles), and fewer juvenile sand dollars and *Littorina* (Fig. 3).
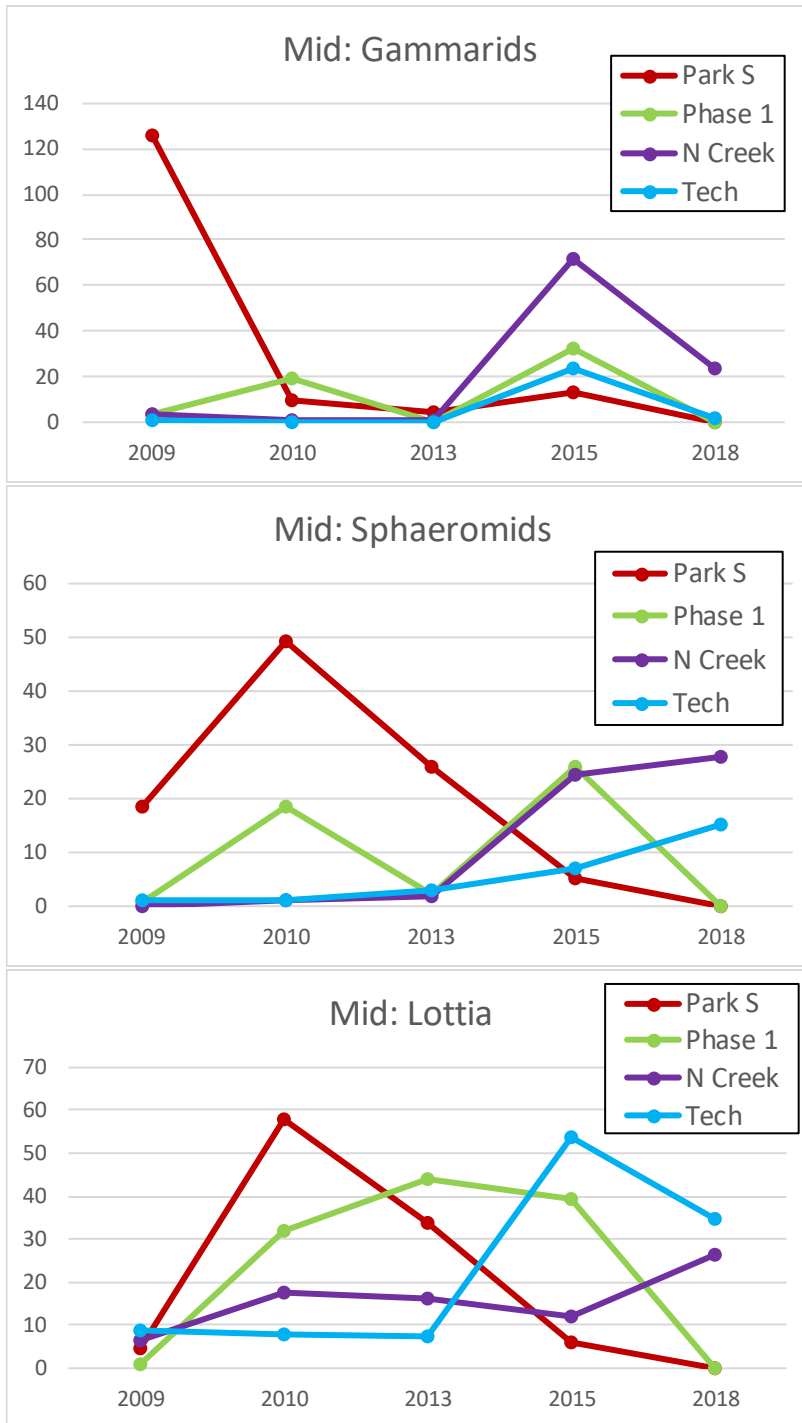
Figure 3. Abundances of some of the taxa driving year-to-year differences in the Mid zone.
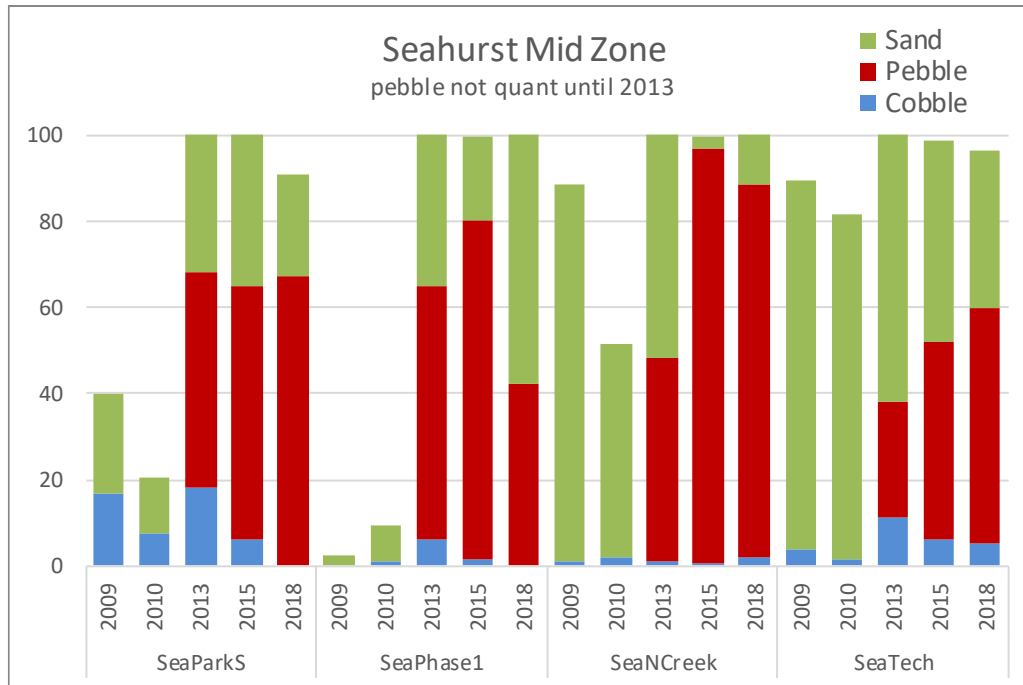
Figure 4. Surface substrates in all transects in the Mid zone. Note that pebbles (red) were not quantified until 2013. Sites are arranged from south to north.

In our 2009 surveys we quantified clam abundances not only in the regular sample units but in additional box cores; those have not been repeated since. However, some clam data come from the 10 cores per transect, since these effectively capture juvenile clams (and adults of small taxa such as *Lucina, Tellina*, and *Nutricola*). They also capture small numbers of adult clams, although many species live deeper in the sediment than our core samples and are sparse enough that small cores do not sample them effectively. Figure 5 shows the summed counts of clams in three categories: juveniles of species that are large as adults, such as *Tresus, Leukoma, Saxidomus*, and *Macoma* spp.; adults of small taxa; and occasional adults of large taxa found in the cores.
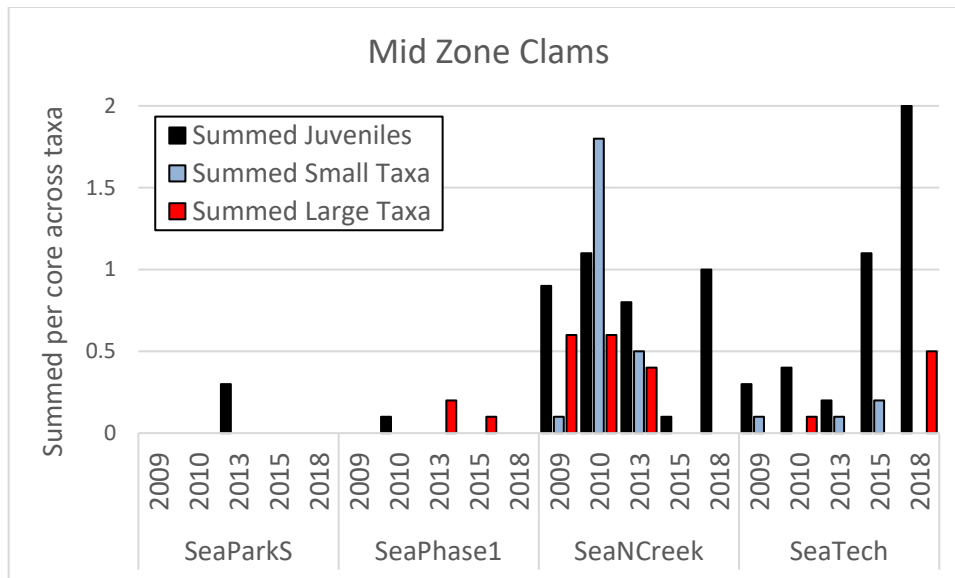
Figure 5. Summed counts per core of all the clams found per Mid-zone transect. Sites are arranged from south to north.

Few clams of any kind were found in any year at the two southern sites, where the Mid-shore is steep and unstable. In contrast, the two northern sites regularly had juvenile clams of various species, and in addition the North Creek site had adult individuals of small taxa such as *Nutricola*, which is found in clean-sand habitats. A few larger clams were found at the northern sites, especially of the invasive high-shore varnish clam, *Nuttallia*.

Species richness in these Mid zone transects varied slightly among sites and years (Fig. 6) but with so few species inhabiting this elevation, it is difficult to discern any real patterns. No sites or years showed substantially higher or lower richness than others.
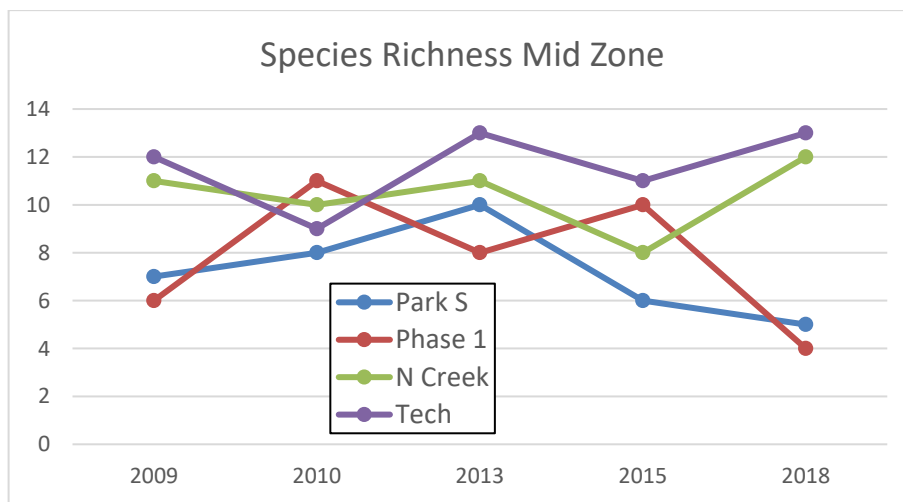

Figure 6. Species richness (number of taxa per transect) in the Mid zone at the 4 locations in the park.

Temporal Patterns in the Low Zone

SCALE surveys in the Low zone (MLLW) were begun at two sites outside the park (Seahurst S and Seahurst N) in 1999 (Fig. 1); these data are included in Figure 7, which shows similarity of biotic communities at all sites and years in the Low zone. In contrast to the Mid zone where Years were more different than Sites, in the Low zone the Sites tend to remain distinct from each other, varying less among Years. Park South and the two older sites, all of which were farther from the restoration activities, were less variable among years than the other three Seahurst sites (Fig. 7).
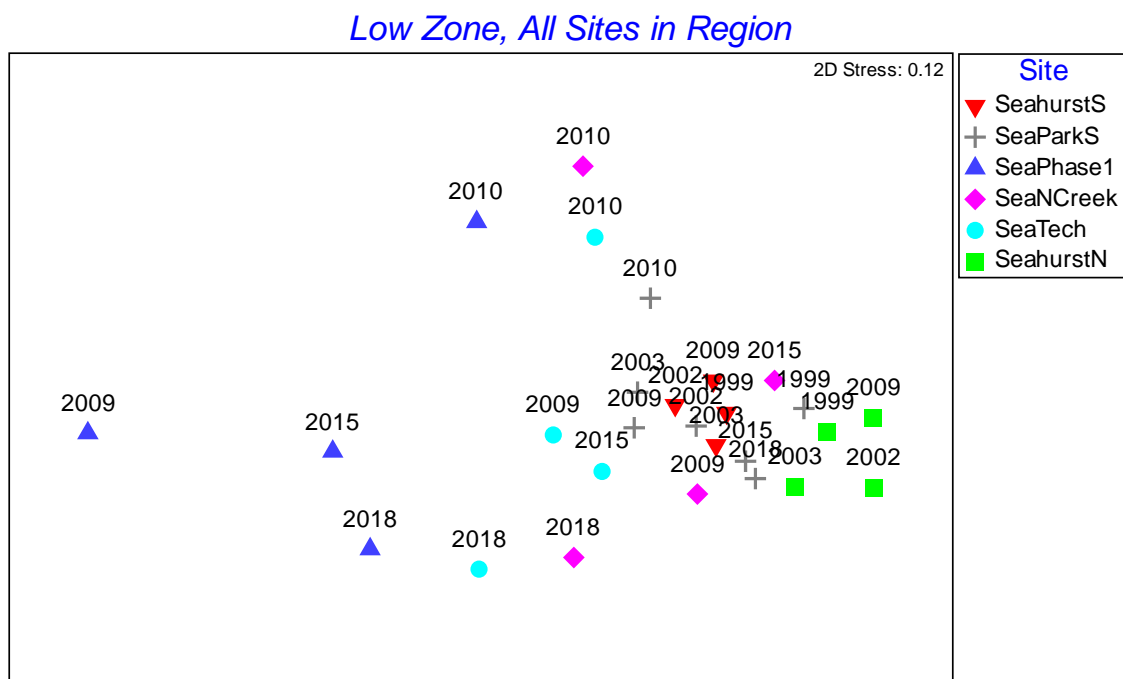

Figure 7. MDS plot of biota at all sites within and near Seahurst Park sampled at MLLW in all years.

When the older sites (which were not sampled after 2009) are omitted and only those in Seahurst Park are included, a few new patterns emerge (Fig. 8). The biota still tend to cluster (with a higher R value) primarily by Site rather than by Year (ANOSIM Sites R =0.535, p = 0.001; Years R = 0.236, p = 0.0025), but Years are more distinct than when all the Sites are included (as in Figure 7). Figure 8 implies that the data from 2010, which cluster in lower right of the plot, might be driving these year-to-year differences.
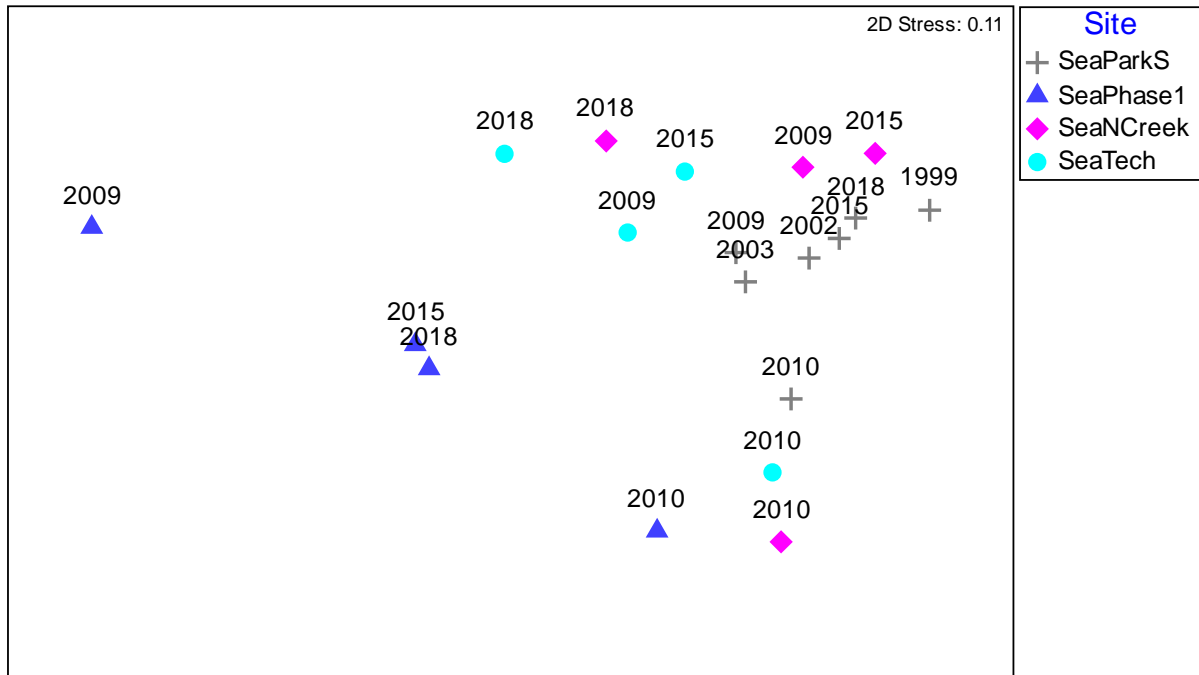
Figure 8. MDS plot of Low zone biota at just the four sites within Seahurst Park.

When 2010 data are removed from the analysis, differences among years in fact do disappear (ANOSIM Site R = 0.715, p = 0.001; Year R = 0 (NS)). Thus other than 2010, differences among biota sampled are almost entirely Site-effects rather than Year-effects. The 2010 differences are due largely to unusually high numbers of sphaeromid isopods (Fig. 9) in surface quadrats, especially *Exosphaeroma inornata* (identified in cores: not illustrated). The Mid zone transects saw a similar pulse in isopods that year (Fig. 3). Low numbers of gammarid amphipods in 2010 at all sites (Fig. 9) also contributed to the year being anomalous. Sediment data through time (Fig. 10) do not show 2010 surface sediment types to be different from years before or after, at least not in a consistent manner.

Figure 10 illustrates trends in surface substrates in the Low transects. Only Sand and Cobble were quantified in 2009 and 2010. The North Creek transect was quite a bit less sandy in 2015 than the prior two years, which may have been an effect of the restoration, especially of channeling the creek (and its sediment load) to a different location. But this transect returned to being sand-dominated in 2018. The adjacent Tech transect had a wave of pebble (and less sand) in 2015 but then returned to sand dominance in 2018. Thus the two sites closest to the Phase 2 restoration showed a decline but then recovery of sand quantities, likely related to what was coming down the small local stream and/or the routing of the stream.
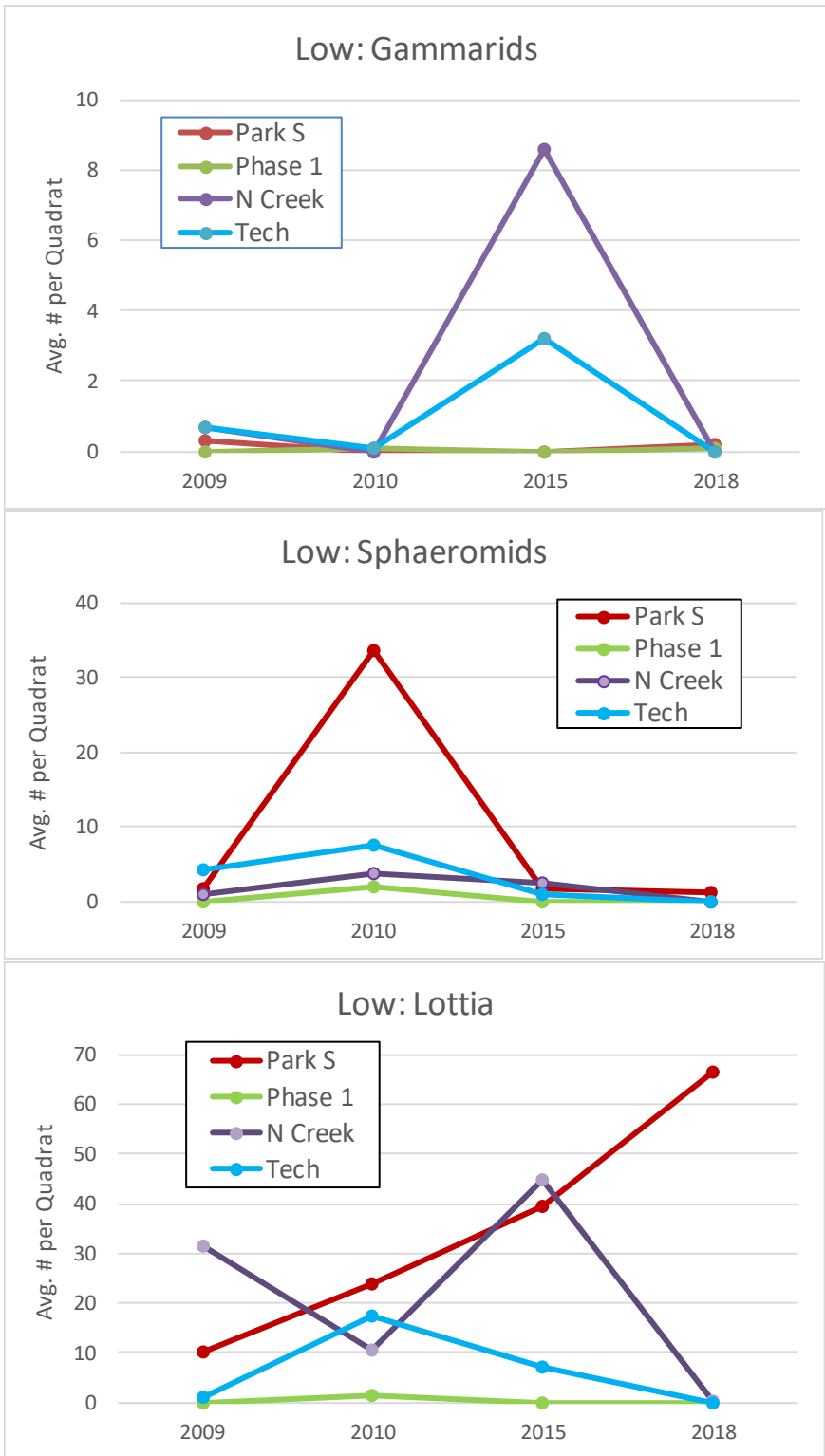
Figure 9. Abundances through time of three key taxa driving differences among years and sites in the Low zone.
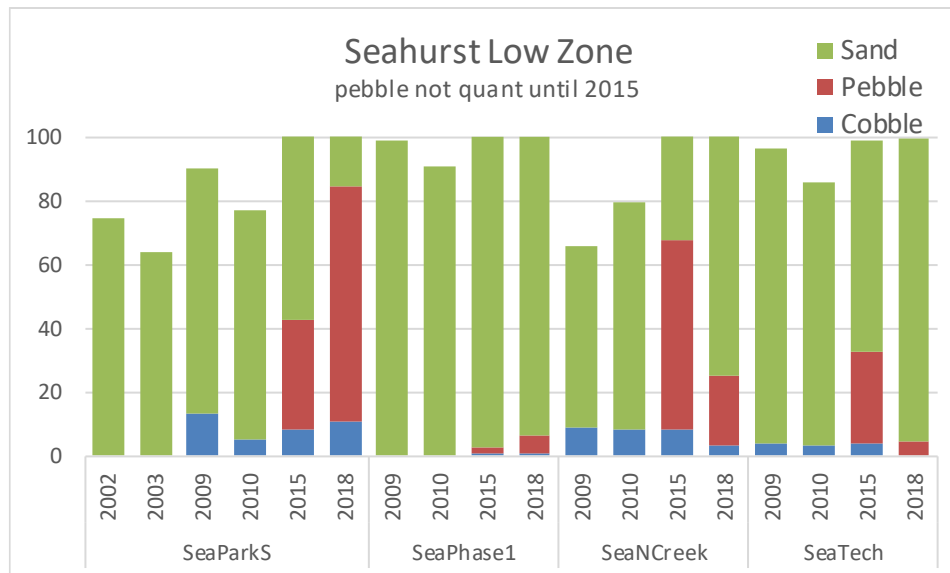
Figure 10. Surface substrates in all transects in the Low zone. Note that pebbles (red) were not quantified until 2015. Sites are arranged from south to north.

Unlike the two northern locations, the Park South Low transect has been getting steadily less sandy and more pebbly from 2009 onwards. It will be interesting to keep tracking the substrates at this location; is it really losing its fine sediment long-term? Might there be shoreline armoring updrift causing sediment starvation? The Phase 1 site, in contrast, has been consistently sand dominated, and has the most eelgrass of all the sites; this is the primary reason for its biota separating clearly from the other sites' in Figure 8. The eelgrass cover at this site has been steadily increasing across the years (2009 to 2018 = mean of 5, 58, 87, and 80% cover). Accompanying this change has been an increase in species richness (Fig. 11); in 2009 when the substrate was mostly plain sand (Fig. 10), there were only 3 species along the transect including sand dollars; by 2018, with stable eelgrass cover for at least 5 years, there were 18 species found, with the increases coming in infauna (diverse worms and small clams) and epibiota on the eelgrass blades including *Lacuna* snails.

Overall, the Low zone transects were somewhat more diverse than the Mid zone, with an average of 11.5 species per transect vs. 9.25 in the Mid zone. In contrast, the average species per transect at three regularly sampled beaches at the south end of Whidbey Island (Possession Point) is 56. This substantial difference follows two trends discussed elsewhere (e.g., Dethier and Schoch 2005); species richness is higher further to the north in Puget Sound (perhaps related in part to reduced temperature and salinity stresses there) and is predictably higher with more surface cobble and less sand. Long-term averages of substrates in the Low zone at Possession are 24% cobble, 67% pebble, and only 9% sand.
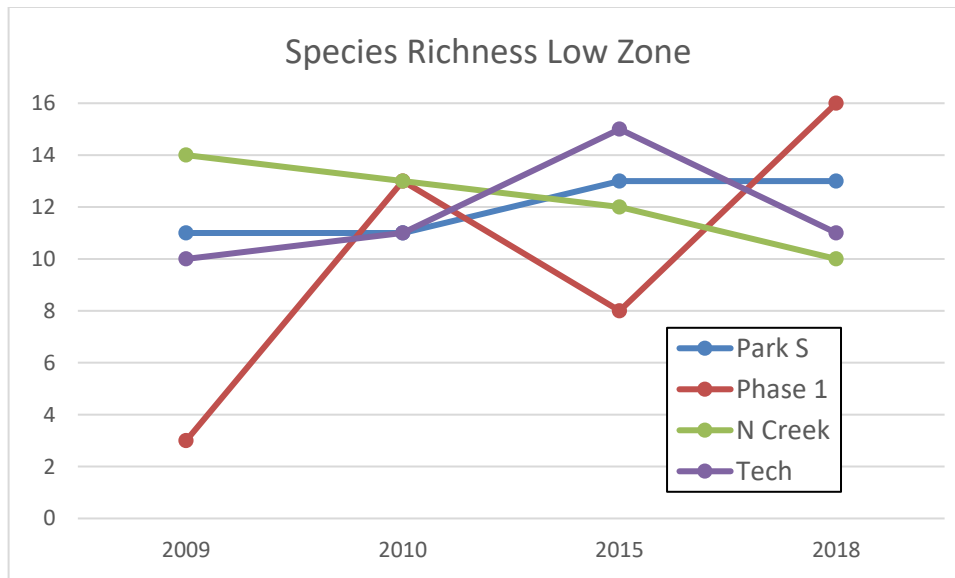
Figure 11. Species richness in the low zone across time at all 4 sites in the park.

Clams in the low-zone cores were more abundant and diverse than those in the Mid transects (Fig. 12); *Tresus* (horse clams) were abundant enough to be plotted separately (Fig. 13). In several years, especially 2015 and 2018, we saw strong recruitment of juvenile clams of large species, especially *Macoma inquinata*. "Small taxa" (clams with adults <15mm length) at this elevation are mostly *Lucina tenuisculpta*, which are often associated with eelgrass, and *Tellina modesta* which tend to be found in clean sand. The complete absence of clams at the Phase 1 site in 2009 probably relates to the substrate in that year being entirely bare sand, with only 3 taxa found on the whole transect. Clams overall are not abundant enough, however, to distinguish any changes in abundance that might have resulted from the Phase 2 restoration work.
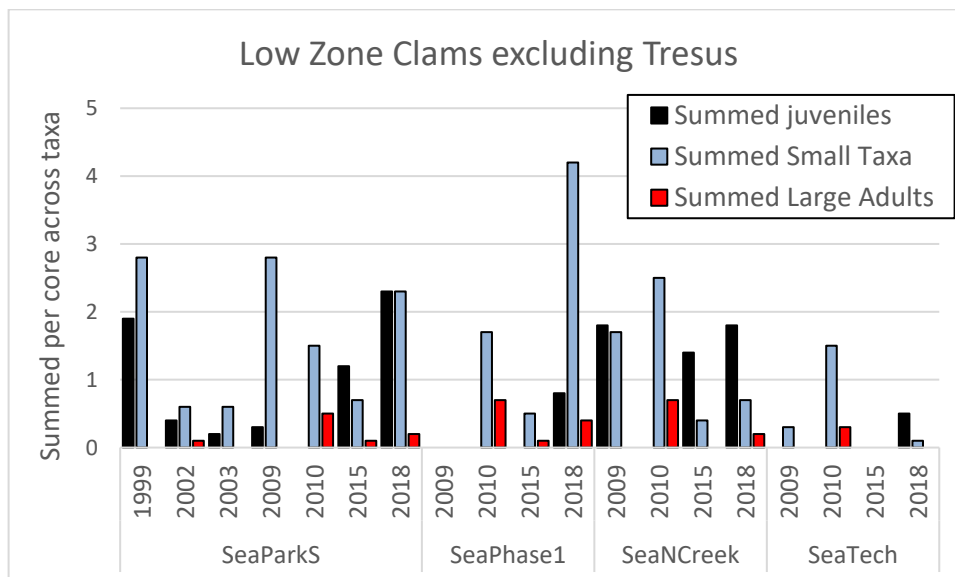

Figure 12. Summed counts per core of all the clams found per Low-zone transect, excluding *Tresus* (graphed separately).

Tresus capax clams, which were very abundant throughout the park in 1971 (Kohn et al. 1971), are now quite rare at least as adults. We did observe a number of recruitment events of *Tresus* into the Low transect areas, especially at the Park South site where large numbers of juveniles were seen in 1999 (Fig. 13) and smaller numbers in 2010 and 2018; many small individuals were seen in the other sites in 2010 but not 2018. Holes of adult clams are counted in surface quadrats; it may be a positive sign that there are significant numbers of these now seen in the Phase 1 site where none were seen in the first few sampling years, and there is a suggestion that numbers may be on the rise at the Tech site as well (Fig. 13).
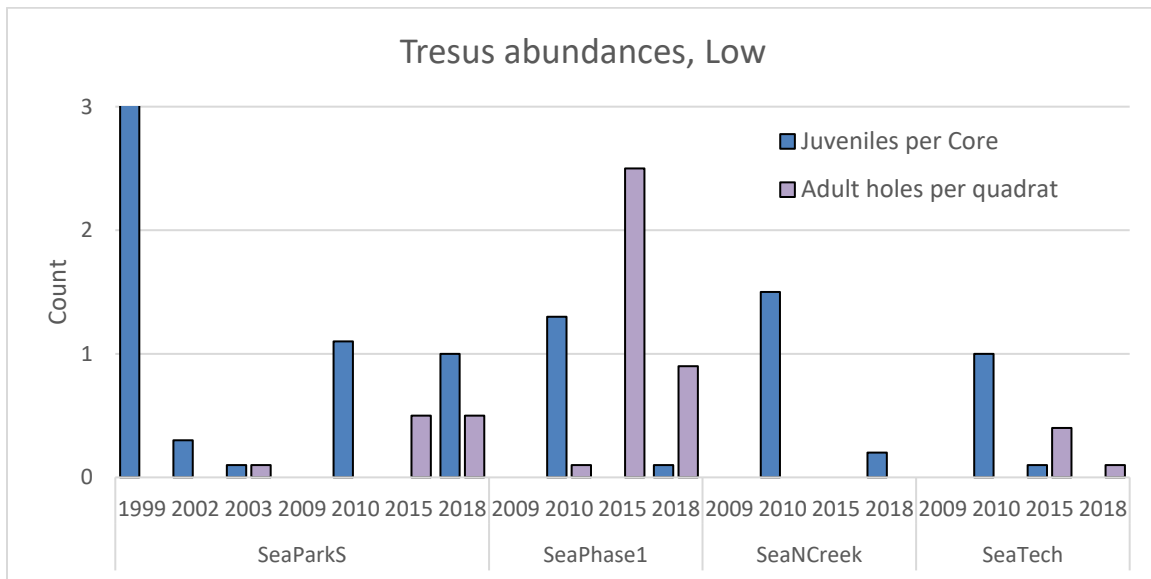

Figure 13. Numbers of juvenile *Tresus* per core and adult siphon holes per quadrat at the four sites in the park. The bar for Park South 1999 is cut off to make other data more visible; the average juvenile count per core on that date was 6.2 at that site.

Figure 14 compares the beach profiles in 2016 among the four sites (data from J. Toft). It shows virtually identical beach morphologies (slope, and beach width from the bank down to MLLW) at the Reference (Park South) and Phase 1 beaches, although there is a wider backshore area at the Reference beach. The Tech site has its backshore cut off by armoring but has a substantial low-tide terrace like those at the Reference and Phase 1 sites. The recently-restored North Creek site has a similar upper limit to Phase 1 (similar engineered shoreline shape for restoration), but had (at least in 2016) an abbreviated low-tide terrace. During our 2018 sampling it appeared that this terrace might be building out to look like the other sites, but we do not have profiles from this date.
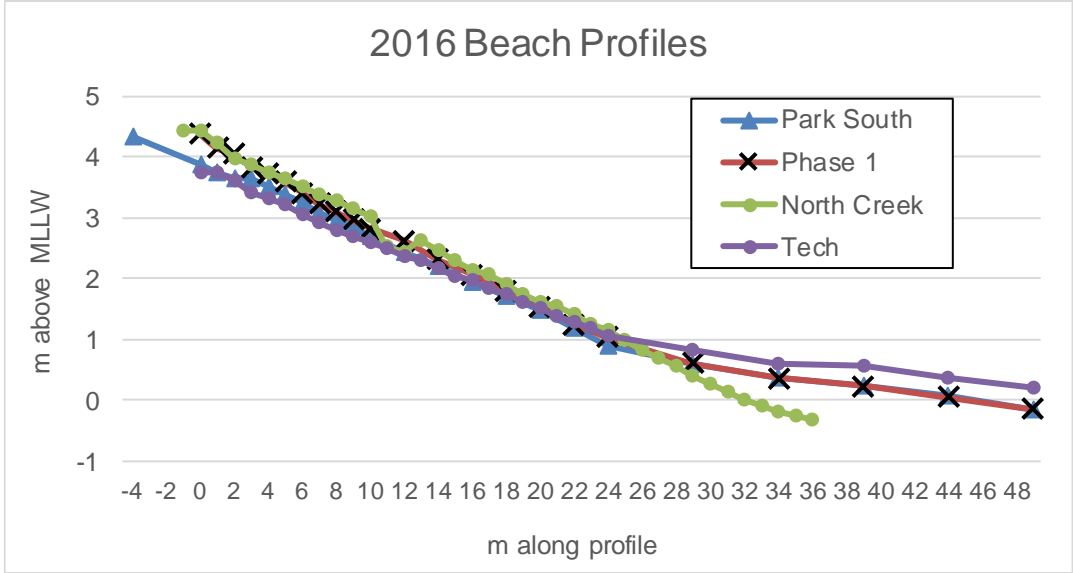
Figure 14. Beach profiles in summer 2016 from the four sites in Seahurst Park. Data from J. Toft.

Discussion

The major challenge in analysis of any ecological change is determining whether it results from a long-term trend, from recent or local events such as a restoration effort, or is simply indicative of normal temporal variation (e.g., interannual or seasonal). The data from the last 11 years at Seahurst Park suggest that we are observing changes of all three types. In the absence of clear spatial or temporal correlates of change, the default (most parsimonious) explanation is generally 'normal variation', which we know is very high in marine nearshore communities. This is perhaps especially true for the beaches of the Salish Sea, where annual variation in storm intensity (and thus beach erosion and alongshore drift) and sediment supply to beaches (via streams or bluff failure) cause the shoreline morphology and abundance of fine sediments to change substantially from year to year. Beach biota clearly vary in concert with such sediment changes.

The patterns of biotic variation analyzed here suggest one change that is likely due to the Phase 2 armoring removal at the north end of Seahurst Park. In the Mid zone at the North Creek site (which was physically closest to the Phase 2 work), there was a clear decline in the abundance of sand during the two sampling dates following the restoration work, and corresponding changes in the biota from sand-loving to cobble-associated species. The lack of sand at this site could also have contributed to the "missing" low-tide terrace (Fig. 14). While sediment types varied at other sites and in other years, none of these changes corresponded temporally to the Phase 2 restoration effort. It is possible that some of the sediment changes at the other sites were indirectly a result of the restoration work, but lagged in time because they were farther from the impacted area, but such a connection would be very hard to prove.

Two other patterns of variation suggests gradual long-term changes in sediment or biota. First, the Low shore transect at Park South (the Reference, relatively pristine area) has gradually become less sand-dominated and with more pebbles and cobbles on the surface (Fig. 10). To date, the biota along this transect do not show any corresponding unidirectional change, although continuing to track this will be of interest. Second, the adjacent Low transect at Phase 1, which has been sand-dominated in all our survey years, has gradually experienced an increase in seagrass cover and changes in associated biota. In this case, the substrate appears relatively constant but the community seems to be evolving into a different, perhaps stable state (clean mobile sand vs. sand stabilized by eelgrass rhizomes). It is perhaps a coincidence, but nonetheless encouraging, to see this healthy and valued habitat type thriving following shoreline restoration. The numbers of clams including *Tresus* in the low zone also appear to be increasing.

Additional information on changes following restoration come from sampling done at the same locations within the park but at different elevations on the beach, by the UW Wetland Ecosystem Team (Toft 2016; Oxborrow et al. 2015). The purpose was to compare the biota at higher elevations at the Reference site (Park South), with those from the beach restored in 2005 (Phase 1), and the beach restored in 2014 (North Creek). Benthic core samples were taken at three elevations farther up the beach than our sampling, at +5, +8, and +12' above MLLW, and they also quantified abundance of wrack, logs, and insects high on the shore. The lowest of these

elevations was at approximately the lower edge of the beach face that was re-graded as part of the restoration, and the +8' elevation was at the lower edge of the North Creek armoring before it was removed. This UW team found that the Phase 1 site, restored ca. 10 years ago, had biota in the benthic cores very similar to that at the Reference site. In the newly restored (1 year) site, they found beach wrack and beach hopper amphipods already increasing, with wrack accumulation on the high shore similar at all three sites. The high elevation (+12') cores were also becoming biologically similar to the Reference and Phase 1 cores. The cores at lower elevations at the newly restored site were more variable, and tended to have more polychaetes whereas the other sites had more amphipods. Only insect density and richness were clearly still reduced at the newly restored site versus the other two sites; the beach restored in 2014 had vegetation planted at that time, but a year later this riparian vegetation was likely still developing relative to vegetation at the Reference and older restoration sites. Many of the insect groups trapped in this sampling effort are associated with riparian vegetation, so it is not surprising that this part of the fauna was not yet similar to the 'older' beaches. Overall, a meta-analysis of restoration impacts following armor-removal on shorelines including at Seahurst suggests a 5-10 year time frame for relatively complete return of high-shore biotic communities to a reference condition (Lee et al. 2018).

There is considerable irony in looking back at the effort and funding spent over 45 years at this site to create an enormously altered shoreline (1974: armoring, fill, walkways, playgrounds, lawn) and to perform biological surveys prior to this construction (1971) -- and then to undo this construction in phases (2005, 2014) with abundant biological survey work before, during, and after. The most recent set of restoration actions was based on intervening decades of research into the importance of 'natural' shorelines to nearshore processes, and into determining what key elements of 'natural' are critical (e.g., sediment supply, riparian vegetation). We have to hope that the restored shoreline configuration will remain in line with our understanding of 'best practices' based on future research on the shoreline ecology of the Salish Sea. At this time, there is evidence of some positive changes that can be attributed to restoration (both older and more recent), especially in high-shore habitats and perhaps in eelgrass and clam populations. Sediment types and beach morphologies are likely to continue to evolve as fine sediments are delivered to the shore and winnowed, and changes in sediments strongly affect the biota of the beach. Human use (trampling, poaching) is likely to remain intense at this highly-visited shoreline (Fig. 15), altering the changes that restoration efforts can produce.

Figure 15. School groups at the SeaTech MLLW transect in June, 2018.

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Appendix 1

Seahurst photos: SeaTech transect, +2.8
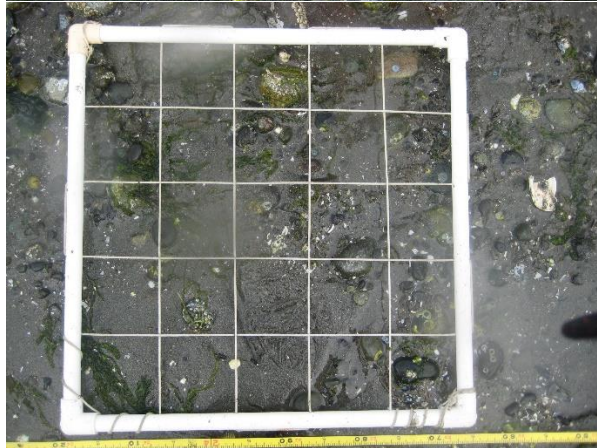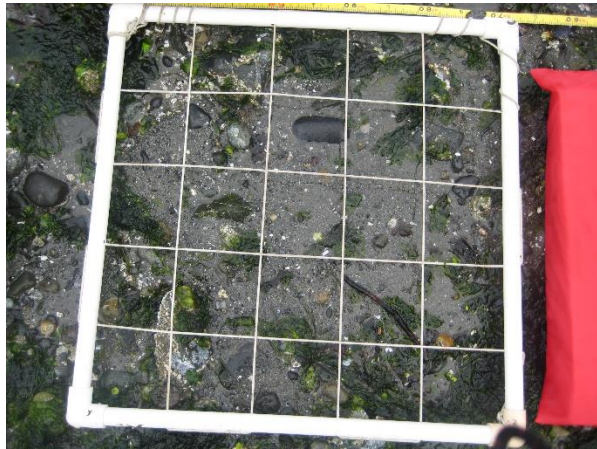

2009 – Transect with more large cobbles.

2018 – transect with less clean sand.

Southern end of SeaTech +2.8 transect (2009) = much sandier than northern end

Below: SeaTech +2.8 quads B, E, I

2009   2018