

Original Articles

Shoreline armor removal can restore variability in intertidal ecosystems

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ABSTRACT

Humans have drastically modified marine nearshore ecosystems through shoreline armoring. Armor, in the form of seawalls and bulkheads, reduces the mean abundance of key ecological features of shoreline ecosystems, such as the amount of beach wrack, the number of beached logs, and the density of supratidal invertebrates. Armor also affects the physical and biological composition and diversity of these important ecological responses – altering the makeup of beach wrack and invertebrate species, for example. Less is known, however, about changes in variability – both over time and space – of ecological responses across natural, restored, and armored shores. Temporal and spatial variation in physical and biological variables can themselves be indicators of ecosystem health and effectiveness of restoration. Working alongside community (citizen) scientists, we found that beach wrack (a nutrient and habitat resource), logs (an element of habitat structure), and supratidal invertebrates (part of the consumer community) often increased following restoration. Further, not only were wrack, logs, and invertebrates on average more abundant and diverse at natural (never armored) shore types compared to armored shore types, but they also frequently had higher variance. In many cases, variance of ecological responses in restored shore types were more similar to natural shore types than armored shore types, indicating a positive effect of restoration. We found that differences among sample sites, rather than across sample years, explained more of the variation in ecological responses across all shore types. Because shoreline armoring is a pervasive human activity, public perception of this variability is key to the social context of restoration success. Participation in data collection through community science endeavors is one way to encourage an appreciation for natural variability within and across landscapes. We implore that shoreline monitoring efforts should evaluate and communicate ecosystem variability as a key indicator of restoration success.

1. Introduction

The human species is the world's greatest ecosystem engineer (Smith, 2007) and has altered habitats throughout its existence. Humans have constructed, developed, and modified their surroundings in support of society and an ever-increasing population (O'Brien and Laland, 2012). Urbanization is a quintessential manifestation of human habitat modification, not only directly affecting terrestrial systems, but also watersheds and shorelines. Whether due to urbanization, agriculture, or other human land use activities, human encroachment is strongly

associated with homogenization or “simplification” of ecosystems, affecting ecological connectivity (Bishop et al., 2017), biodiversity (Groffman et al., 2014; McKinney, 2006) and other ecological processes. Homogenization occurs both physically – through the creation of impervious and built surfaces – and biotically – through the introduction and cultivation of invasive biological communities (Groffman et al., 2014; McKinney, 2006).

One way humans have simplified marine shoreline habitats is through the construction of seawalls and the placement of riprap. Collectively referred to as *shoreline armor*, these structures prevent

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natural processes of erosion and allow building development right up to the water's edge (Cooper et al., 2020; Ma et al., 2014) – often effectively removing upper intertidal areas. Fortunately, in some cases, the construction of new shoreline armor has ebbed in recent years (Puget Sound Partnership, 2021), and in areas such as the Puget Sound region of Washington State, USA, existing armoring is also being actively removed for restoration (Dethier et al., 2017). Restoration typically involves not only armor removal, but also other living shoreline techniques such as the re-creation of a more natural sloping beach profile, beach nourishment with natural sediments, the reintroduction of native plant species, and placement of logs that naturally retain sediments and limit erosion (Dethier et al., 2017; Toft et al., 2021). Restoration practices are increasingly coupled with incentives to landowners and developers (e.g., the Puget Sound Shore Friendly initiative: <https://www.shorefriendly.org>), including the construction of new buildings farther from the natural shoreline (Johannessen and Maverick, 2020; Scyphers et al., 2020).

Studies have shown that armor reduces the mean abundance of key biological components of shoreline ecosystems – essentially driving down biomass on modified beaches (Dethier et al., 2016; Gittman et al., 2016). Decreases in the coverage and depth of beach wrack, the number of beached logs, and the density of supratidal invertebrates are just some of the effects of shoreline armoring and maintenance regimes (Dethier et al., 2017, 2016; Heerhartz et al., 2014; Lee et al., 2018; Schooler et al., 2019; Toft et al., 2021). As such, restoration success is often measured as the rebounding of these attributes: an increase in mean beach wrack cover and depth, in log number, and supratidal invertebrate density (Toft et al., 2013; Lee et al., 2018; Toft et al., 2021). In Puget Sound, where nearly a third of the shoreline is armored, removing armor has shown to be an effective strategy for restoring this biomass, along with critical ecosystem functions (Toft et al., 2021).

Human habitat alteration not only decreases the mean abundance of biological variables, but it can also suppress biological diversity and variability across scales (Shochat et al., 2010; Shochat et al., 2006). For example, in shoreline ecosystems in Puget Sound, armoring reduces the size distribution of sediments and the composition of beach wrack, which is further associated with decreases in invertebrate richness (Heerhartz et al., 2014; Lee et al., 2018; Toft et al., 2021). Simplification of the structural habitat drives homogenization of biological communities (i.e., community diversity and richness; McKinney, 2006). Structural homogenization can alter habitats to such an extent that their conditions are too severe or do not contain the necessary resources for certain species (Hobbs et al., 2009). Because diversity and variability can beget further biodiversity (Martin and Ferrer, 2015), removing variation at one level can have cascading effects at other levels: physical habitat homogenization limits the natural structure and functions that support diversity and biological heterogeneity (e.g., Romanuk and Levings, 2003).

The true negative impacts of shoreline armoring, however, might not be limited to changes in mean biomass, abundances, or even the number of different species, but could also include the reduction of variance around these means. In other words, because armored shorelines are structurally more similar to one another and less complex than natural shorelines (Lawrence et al., 2021; Perkol-Finkel et al., 2018), they may also support less temporally and spatially variable ecosystems (Lawrence et al., 2021). This loss of complexity and variability itself can have negative consequences for ecosystems, affecting biodiversity and ecosystem function (Aguilera et al., 2014; Lawrence et al., 2021). Temporal and spatial variability have long been known to broadly promote increased biodiversity (Fox, 1979; Levin, 2000; Martin and Ferrer, 2015) and can also be an indicator of ecosystem health. Loss of variation through time (i.e., declines in temporal beta diversity) can reflect severely degraded systems and decreases in ecosystem function and services (e.g., lake eutrophication causes declines in species diversity, Cook et al., 2018). Although variation through time and space may be a key driver and indicator of ecosystem health and function, few

studies have focused on this variability as a measure of restoration success as a whole (but see Eviner and Hawkes, 2008), let alone in shoreline habitats.

Here we build upon previous work (cited above) documenting effects of shoreline armor and armor removal (restoration) on the mean abundance and compositional diversity of key ecological response variables in the Puget Sound region of Washington State. We extend our investigation to explore whether previously unexamined variability in these ecological responses differs among armored, restored, and natural shore types and if this variability is due to differences across sampling years or sampling sites. We expect that three key ecological response variables – beach wrack (a habitat and nutrient resource), beach logs (a component of habitat structure), and supratidal invertebrates (part of the animal consumer community) – will be more abundant and compositionally diverse at natural shore types (never armored) compared to armored shore types, as is consistent with previous studies in this area (Dethier et al., 2017, 2016; Heerhartz et al., 2014; Lee et al., 2018; 2019; Toft et al., 2021). Specifically, we hypothesize that natural shore types will have a more compositionally diverse wrack line, a greater variation of log sizes, and a higher richness of the supratidal invertebrate community compared to armored shore types. We further hypothesize natural shore types to have higher variance in ecological responses compared to shore types with armor, which we expect to have a spatially and temporally homogenizing effect on intertidal ecosystems. Depending on the success of restoration, we hypothesize restored (previously armored) shore types to have means and variances somewhere between natural and armored shore types. Further, we predict increased variances in the abundance of logs, wrack, and invertebrates at natural and successfully restored shore types to reflect increased variability across both time (sample year) and space (sample site) compared to armored shore types.

2. Material and methods

2.1. Study area

Our eighteen study sites were widely dispersed within the Washington State, USA portion of the Salish Sea, also known as the Puget Sound Region (Fig. 1), an estuarine fjord with mixed semidiurnal tides. The sites spanned a broad geographic area (over 162 km in latitude: 47.1523 to 48.6109 degrees; and 87 km in longitude: –123.0187 to –122.2399 degrees). Roughly from South to North, sites included: Edgewater Beach (Eld Inlet), Titlow Beach (Tacoma), Big Beach and Lost Lake (Vashon Island), Piner Point and Dockton Park (Maury Island), Seahurst Park (Burien), Dabob Bay (Hood Canal), Howarth Park (Everett), Waterman and Maylor Point (Whidbey Island), Dawley (Sequim Bay), Fort Townsend (Port Townsend), Cornet Bay and Bowman Bay (Deception Pass), Kukutali Preserve (Similk Bay), and Brown Island and Family Tides (San Juan Islands). Armor occurs along nearly 30% of the shoreline within our study area (MacLennan et al., 2017). Across our sample sites, armoring exists at various tidal elevations, either above or below the Mean Higher High Water line.

2.2. Study participants

Four main organizations coordinated shoreline monitoring data collection, involving a total of 312 volunteers: the University of Washington (UW: 20 volunteers), the Washington Department of Fish and Wildlife (WDFW: 12 volunteers), Vashon Nature Center (VNC: 106 volunteers and 122 high school students), and the Northwest Straits Foundation (NWSF: 52 volunteers). In total, monitoring amounted to 2023 volunteer hours over six years.

2.3. Surveys & data collection

Each study site had at least one “natural reference” shore type

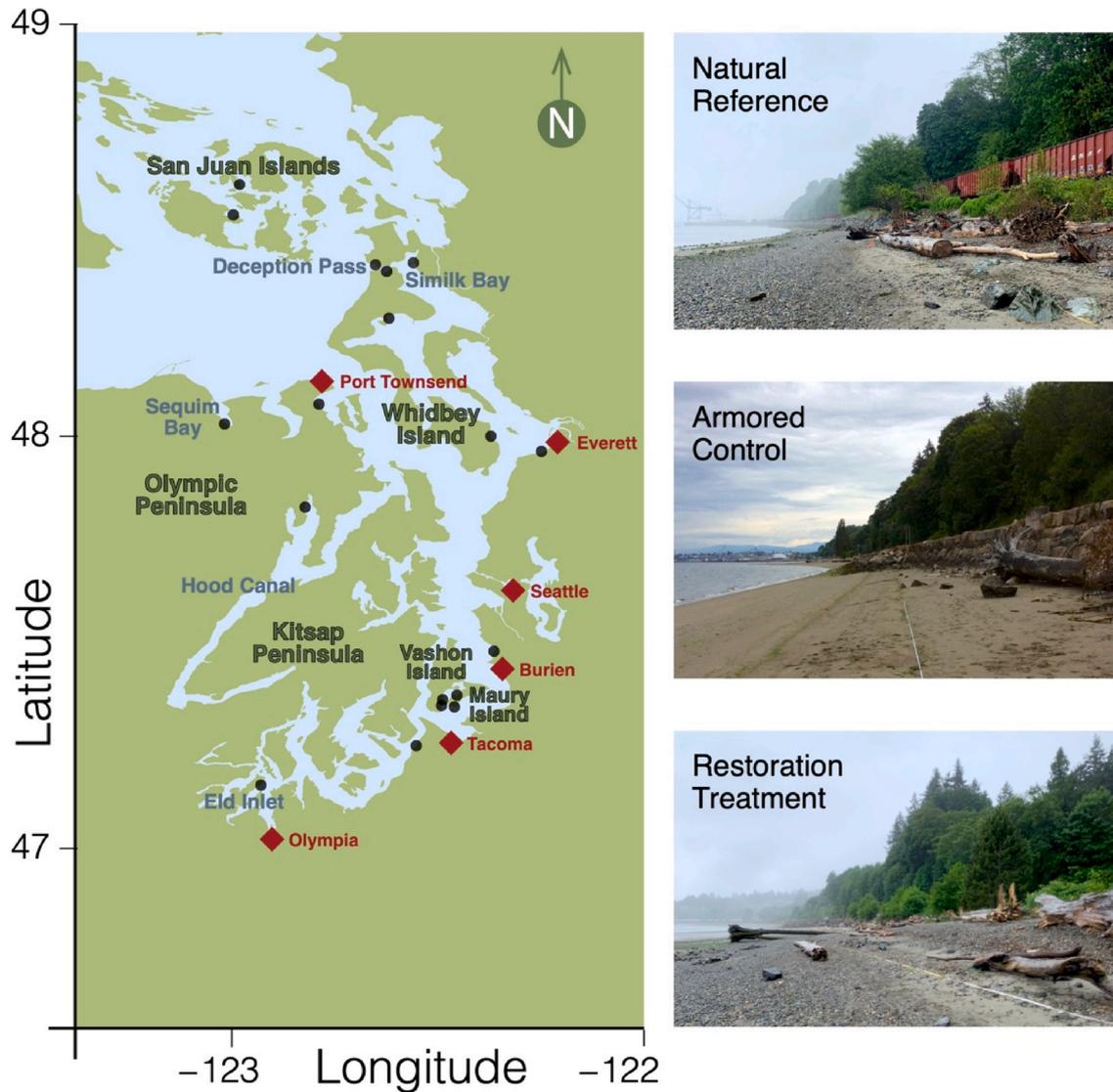


Fig. 1. Map of 18 study sites in the Puget Sound region of Washington State, USA with cities shown with red diamonds, water bodies labeled in blue, and land masses labeled in green (left) and representative photos of the three shore types taken at Howarth Park (Everett): natural reference, armored control, and restoration treatment. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

(natural shore types had never been armored), and one or both of an adjacent “armored control” shore type (armored shore types currently armored and unrestored) and a “restoration treatment” (restored shore types with armor removed). Though all restored sites had armor removed, restoration also often included sediment nourishment, beach profile changes, log placement, and planting of native vegetation to mimic the ecological conditions of natural (unaltered) shorelines. We collected field data and samples in collaboration with volunteer community scientists (guided by coordinators) between June and August from 2015 to 2020. All data collectors used methods developed in Dethier et al. (2016), Toft et al. (2021), and described in standard protocols available open access online as part of the Shoreline Monitoring Database (<https://shoremonitoring.org>). We collected all data and samples along 50-meter transects placed parallel to the shoreline at each of the three shore types on an ebbing tide when the upper beach + 6' Mean Lower Low Water and above was exposed. Specific protocols are described below.

Beach Wrack: We (the authors, coordinators, and volunteers) measured three different ecological response variables for beach wrack at the majority of sites surveyed – two reflecting relative wrack *abundance*: (1) wrack percent cover; (2) wrack depth; and (3) one reflecting

beach wrack *diversity* (only calculated for sites where wrack was present) using the percent cover of three main types of wrack: marine algae, eelgrass, and terrestrial-derived matter (e.g., leaves, branches). We surveyed wrack along the most recent ebbing tide deposition (“wet wrack line”) at ten random points (generated in advance either through “random.org”, in Microsoft Excel, or in base R; version 4.1.1; R Core Team 2021) along the 50-meter transect. We used a 0.1-meter-squared quadrat divided into 25 equal squares (5 by 5) placed in the center of the wrack line to aid in measurements. We approximated the percent total cover of wrack within the quadrat, wrack depth at the deepest point using a clear plastic ruler, and percent cover of the three beach wrack categories. We calculated wrack diversity using the Shannon Diversity Index on proportions of algae, eelgrass, and terrestrial matter using the R package “vegan” version 2.5.7 (Oksanen et al., 2020).

Beach Logs: We measured four different ecological response variables for beach logs – two reflecting relative log *abundance*: (1) number of logs; (2) log line width; and two reflecting log *diversity* (only calculated for sites where logs were present): (3) representation of log sizes – large and small logs; (4) richness of growth on logs – marine, terrestrial, or no growth. We measured the number of logs (greater than 1 m in length and excluding fallen trees) and the width of the log line at five random points

along the 50-meter transect. We measured the width of the log line perpendicular to the transect as the distance from the seaward-most edge of logs to the landward-most edge of logs. We then counted the number of large (greater than 2 m long) and small (between 1 and 2 m long) logs intersecting the perpendicular line. We scored log size diversity as “1” if both log sizes (small and large) were present and as “0” if only one log size was present (either small or large). We recorded the number of logs with marine (e.g., barnacles) and/or terrestrial growth (e.g., grasses). We scored log growth richness from 0 to 2 corresponding to no growth, one growth type, or both growth types present.

Supratidal Invertebrates: We measured three different ecological response variables for supratidal invertebrates – one reflecting invertebrate *abundance*: (1) total invertebrate density; and two reflecting invertebrate *diversity*: (2) family richness – number of different invertebrate families; and (3) order richness – number of different invertebrate orders. We sampled invertebrates using fallout traps (40 × 25 cm plastic bins with a small amount of soapy water) deployed for 24 h at five random points along the 50-meter transect in the supratidal habitat. We filtered samples through a 106 µm sieve and preserved them in 70% isopropanol prior to sorting, identifying to family and order, and counting under dissecting microscopes in the laboratory. We calculated invertebrate density as the total number of individuals per meter-squared per day (24 h) and invertebrate richness as the total number of different families and orders per fallout trap.

2.4. Statistical analysis

Before-After Control Impact (BACI) Analysis: We used generalized linear mixed-effects models to compare differences in the ten ecological responses across the three shore types before and after a restoration event, which only occurred in the “restoration treatment” shore type. All models were of the form:

$$\text{response} \sim \text{shore_type} * \text{before_after} + (1|\text{sample_year}) + (1|\text{sample_site}/\text{transect}).$$

That is, random intercept models with interacting fixed effects between shore type and before vs. after restoration ($\text{shore_type} * \text{before_after}$) and crossed random effects of sample year ($1|\text{sample_year}$) and transect nested within sample site ($1|\text{sample_site}/\text{transect}$). We numbered transects uniquely within, but not across, each site. Therefore, at a minimum, each site had transects numbered between 1 and 3 (one per shore type) with some sites having up to six transects if there were multiple of each shore type (please see the example [supplemental data file](#)). Since it was possible to include transect as a random effect, we performed BACI models on all replicates within transect rather than on transect means (as with following analyses in this study). The BACI model requires at least two “treatments”; one where an impact takes place (here, the restored shore type) and one “control” where it does not (here, both the armored shore type and the natural reference shore type [Pardini et al., 2018; Smith, 2002](#)). Having two types of control allowed us to determine whether changes in the response were consistent with the impact (restoration) or occurred across all shore types. BACI models showing significant interactions between shore type and before vs after restoration indicated a potential effect of restoration on a given response variable. In these cases, we performed post-hoc pairwise comparisons between before and after values for each shore type. Details regarding transformation of the response variable and specific model families employed can be found in [Supplemental Table 1](#).

Variability Among Shore types Analysis: We used Bartlett’s Tests to evaluate heterogeneity in variances among the three shore types (natural reference, restored treatment, and armored control) for each of the ten response variables. We then used F-tests to determine equality of variances between pairs of shore types. F-tests were one-sided to test the hypothesis that natural and/or restored shore types had higher variances than armored shore types. We performed these analyses on mean values taken over all replicates for each transect.

Variation by Sample Year and Site: We used linear and generalized

linear mixed-effects models to test whether shore type, sample site, and sample year affected each of eight response variables (note we excluded log size or growth richness due to lack of statistical power) and whether the effect of sample site and year depended on shore type. All models were in the form:

$$\text{response} \sim \text{shore_type} + \text{sample_year} + \text{sample_site} + \text{shore_type} * \text{sample_year} + \text{shore_type} * \text{sample_site} + (1|\text{site_transect}).$$

That is, random intercept models with main fixed effects of shore type, sample year, sample site ($\text{shore_type} + \text{sample_year} + \text{sample_site}$), interacting fixed effects of shore type and sample year ($\text{shore_type} * \text{sample_year}$) and shore type and sample site ($\text{shore_type} * \text{sample_site}$), and random effects of uniquely identified site transects ($1|\text{site_transect}$). If interactions between shore type and sample year or site were significant, we performed post-hoc tests to determine which shore types varied by year or sample site, respectively. We performed these analyses on mean values taken over all replicates for each transect since including nested random effects of transect within sample site resulted in model non-convergence. Details regarding transformation of the response variable and specific model families employed can be found in [Supplemental Table 1](#). We used the R package “glmmTMB” (version 1.1.2.2; [Brooks et al., 2017](#)) to run generalized linear models, “emmeans” (version 1.6.0, [Lenth, 2021](#)) for post hoc marginal mean comparison, and “dplyr” (version 1.0.5, [Wickham et al., 2021](#)) for data manipulation and visualization within R software (version 4.1.1., [R Core Team, 2019](#)).

3. Results

3.1. Before-After control impact

Overall, restoration had significant effects on beach wrack, logs, and invertebrates compared to natural and armored sites ([Fig. 2, Table 1](#)). Specifically, the interaction between shore type and sample time (before/after restoration) was significant for beach wrack cover and diversity. Beach wrack percent cover significantly increased over time in restored and natural reference treatments relative to armored treatments, whereas wrack depth increased similarly for all three treatments, and beach wrack diversity only increased significantly for the restored treatment.

The interaction between shore type and sample time (before/after) was significant for number of logs and width of the log line. The number of logs and length of the log line increased significantly for only the restored treatment, but did not change significantly for either the natural reference or armored treatments. Natural shore types were more likely to have both small and large sized logs, but there was no interaction between shore type and sample time.

Interactions were not significant between shore type and sample time for invertebrate density and family richness, though both were significantly different among shore types. Natural shore types had higher invertebrate density and family richness than both restored and armored shore types. The interaction between shore type and sample time was significant for invertebrate order richness, which increased significantly for the restored treatment but did not change significantly for the natural or armored shore types.

3.2. Variance among shore types

There were significant differences in variances among the three shore types – armored control, natural reference, and restoration treatment – depending on the response variable. In general, ecological variables for natural reference shore types had the largest variances and armored control shore types had the lowest variances. Restored shore types typically had variances more similar to natural shore types for wrack and invertebrates, but more similar to armored shore types for beach logs ([Fig. 3](#)).

Variance in wrack cover and depth differed significantly among

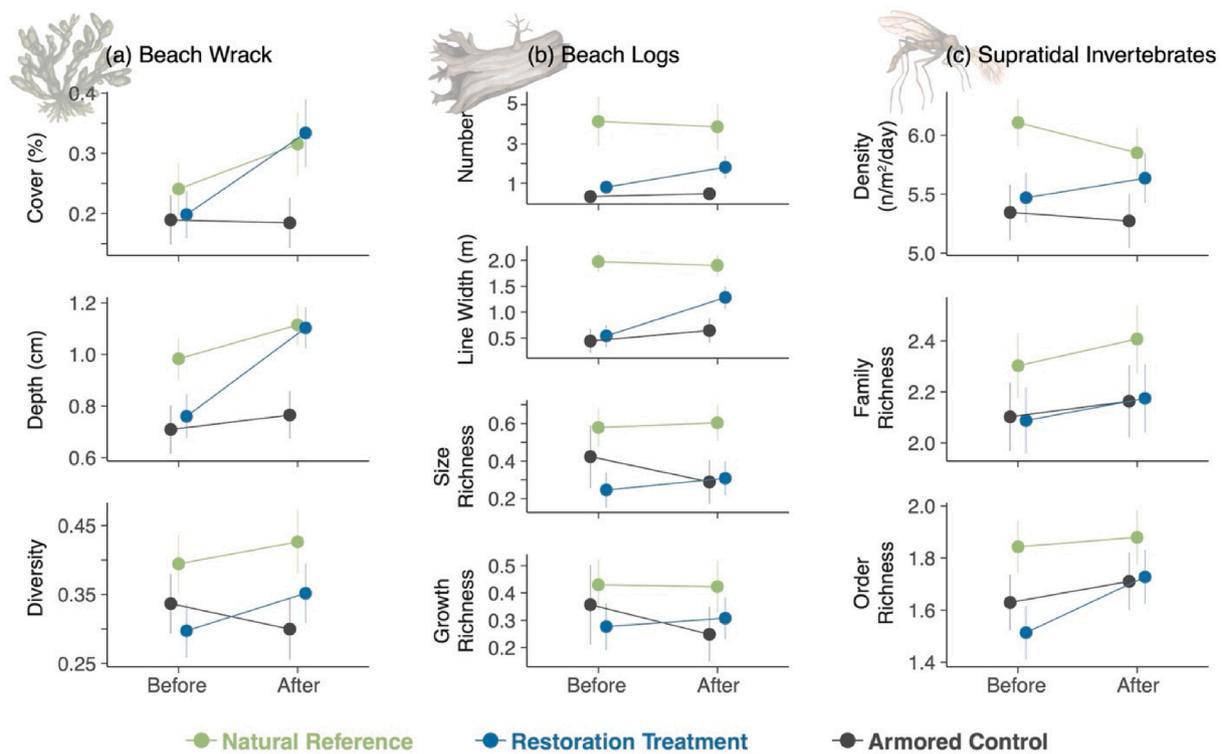


Fig. 2. Before After Impact Control (BACI) Generalized Linear Mixed Effects model outputs showing marginal means and 95% confidence intervals for ten ecological responses before and after restoration for three shore types: natural reference (green), restoration treatment (blue), and armored control (grey), and including (a) three beach wrack variables: percent cover, depth, and diversity of types (algae, eelgrass, terrestrial); (b) four beach log variables: total log number, width of the log line, presence of one or both sizes (small and large), and the richness of growth on logs (none, marine, and/or terrestrial); (c) three supratidal invertebrate variables: total density (number/m²/day), family richness, and order richness. Please see Table S1 for transformations and model families. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 1

Summary ANOVA results of main ecological response variables (wrack, logs, and invertebrates) for Before After Impact Control (BACI) generalized linear mixed-effects models and post-hoc pairwise tests where the interaction term (before/after*shore type) was significant. Significance of pairwise test indicates response changed significantly before and after the restoration for that shore type. (+) denotes an increase and (-) denotes a decrease in the response variable. Significance is denoted by asterisks: * p < 0.05; ** p < 0.01; *** p < 0.001. No asterisk denotes a non-significant comparison. Please see Table S1 for transformations and model families.

		Main Effects (X ² -values)		Interaction (X ² -values)	Post-hoc Pairwise Tests (F-ratios)		
		Before-after	Shore type		Armored control	Natural reference	Restoration treatment
Beach Wrack	Cover (%)	0.06	1.76	25.9***	0.81	12.8(+)**	36.9(+)**
	Depth (mm)	0.78	12.6**	19.7***	0.78	7.72(+)**	42.1(+)**
	Diversity (3 types)	1.37	9.53**	7.35*	1.37	1.41	4.05(+)*
Beach Logs	Number	4.16**	78.9***	51.5***	4.16	0.62	47.6(+)**
	Line Width (m)	2.78	69.1***	54.7***	2.78	0.70	53.6(+)**
	Growth (0-2 types)	0.60	1.89	0.81	-	-	-
	Sizes (1-2 sizes)	0.73	10.8**	1.42	-	-	-
Supratidal Invertebrates	Density (n/m ² /day)	0.146	11.5**	4.38	-	-	-
	Family Richness	0.481	6.03*	0.20	-	-	-
	Order Richness	1.42	14.1***	6.52*	1.42	0.38	13.1(+)**

shore types (Bartlett Test: wrack cover $K^2 = 7.2$, $p = 0.03$, wrack depth $K^2 = 6.8$, $p = 0.03$, Fig. 3 (a)). Natural and restored shore types did not differ significantly in variance, but both had significantly higher variance in wrack cover compared to armored shore types (F-Test: natural > armored $F = 1.7$, $p = 0.02$; restoration > armored $F = 2.2$, $p = 0.005$). Natural shore types had significantly higher variance in wrack depth compared to armored shore types (F-Test: natural > armored $F = 2.0$, $p = 0.005$); however, pairwise differences in variance between natural and restored, and between restored and armored shore types were not significantly different. Finally, wrack diversity variance did not differ significantly among the three shore types.

Variance in log number and the width of the log line differed significantly among shore types (Bartlett Test: log number $K^2 = 23.3$, $p < 0.001$, log line width $K^2 = 8.9$, $p = 0.01$, Fig. 3 (b)). Variance in log number was significantly higher in natural shore types compared to restored and armored shore types (F-Test: natural > restored $F = 3.6$, $p < 0.001$; natural > armored $F = 3.1$, $p < 0.0001$), but did not differ between restored and armored shore types. Natural and restored shore types did not differ significantly in variance, but both had significantly higher variance in log line width compared to armored shore types (F-Test: natural > armored $F = 2.2$, $p = 0.002$; restored > armored $F = 2.0$, $p = 0.01$). Variance in the presence of both log sizes and in growth

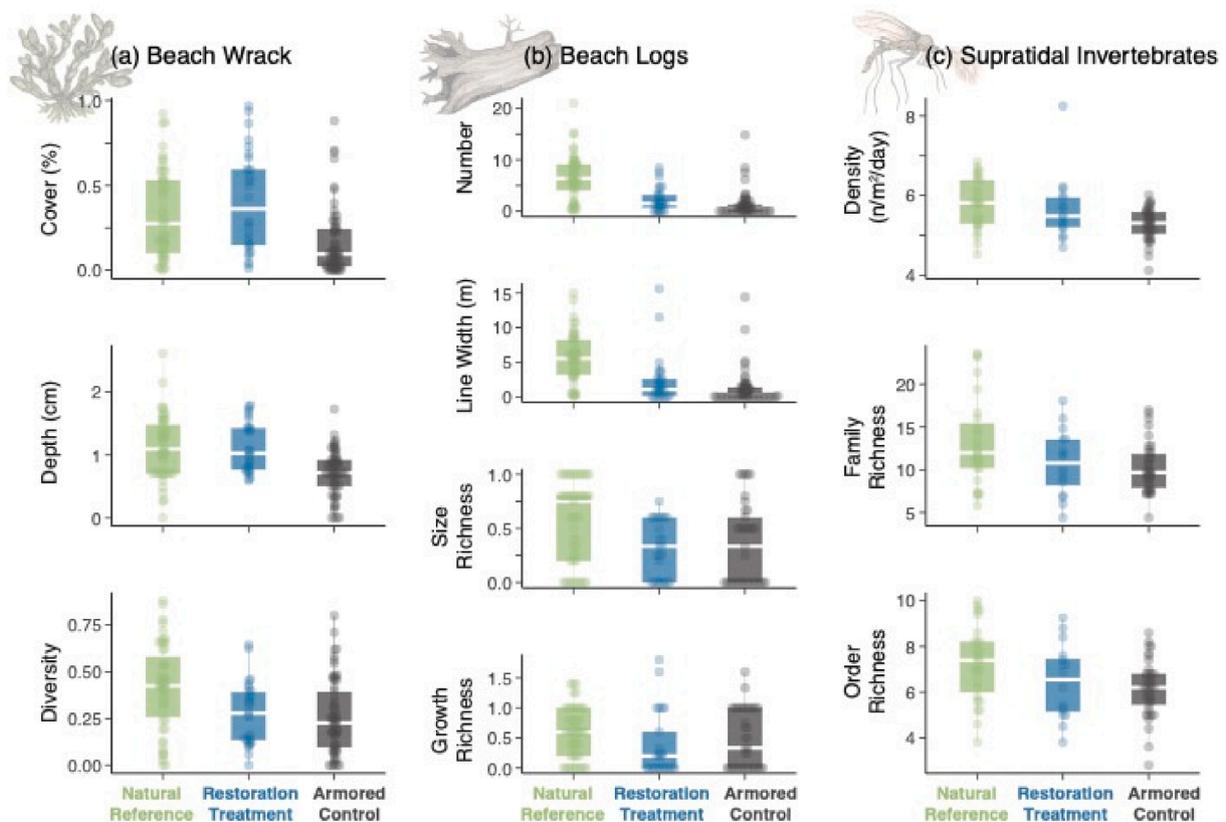


Fig. 3. Boxplots showing differences in variability among three shore types: natural reference (green), restoration treatment (blue), and armored control (grey) for ten ecological responses including (a) three beach wrack variables: percent cover, depth, and diversity of types (algae, eelgrass, terrestrial); (b) four beach log variables: total log number, width of the log line, presence of one (=0) or both (=1) sizes (small and large), and the richness of growth on logs (none, marine, and/or terrestrial); (c) three supratidal invertebrate variables: total density (number/m²/day), family richness, and order richness. Please see Table S1 for transformations. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

richness did not differ among shore types.

Variance in invertebrate density and family diversity, but not order diversity, differed significantly among shore types (Bartlett Test: invertebrate density $K^2 = 12.1$, $p < 0.01$, invertebrate family richness $K^2 = 10.6$, $p < 0.01$, Fig. 3 (c)). Natural and restored shore types had similar variances in invertebrate density, and both had larger variances compared to armored shore types (F-Test: natural > armored $F = 2.4$, $p = 0.007$; restored > armored $F = 4.0$, $p = 0.0003$). Finally, natural shore types had higher variance in invertebrate family richness than armored shore types (F-Test: natural > armored $F = 3.3$, $p = 0.0005$), but there were no other significant pairwise differences.

3.3. Variability across sample year and sample site

Shore type had significant interactive effects with sample site on all seven of the eight ecological response variables tested (Table 2, Fig. 4). In other words, all ecological responses except for wrack depth in armored, natural, and restored shore types varied differently depending on sample site. Only for log line width and invertebrate order richness was there also a significant interaction between shore type and sample year. Post-hoc tests showed that there were significant differences among sample sites for natural shore types for all response variables except wrack depth. Restored shore types were significantly variable among sites for most response variables, except for wrack and diversity. Finally, armored shore types were significantly variable among sites for response variables except invertebrate density.

4. Discussion

We found that armor has negative and homogenizing effects on many

key ecological parameters in shoreline ecosystems, often affecting the mean and variance of beach wrack, beach logs, and supratidal invertebrates. As with previous studies, our results from 18 sites across Puget Sound, WA, confirmed that the mean values of key ecological response variables were usually higher for natural compared to armored shore types (Dethier et al., 2016; Heerhartz et al., 2014; Lee et al., 2018; Sobocinski et al., 2010; Toft et al., 2021). In the majority of responses, there was an increase in these variables at restoration sites after armor was removed. Previous studies, however, have focused on these changes in mean values, but have not measured changes in variance around these means. Although it is not typically considered a measure of restoration success, we also found that the variance in these parameters in restored shore types was often more similar to variance in natural shore types, but was lower in armored shore types. Further, variance in different responses across all shore types was mostly driven by variability among sample sites, and only occasionally driven by variability through time.

Our results also showed that mean compositional diversity – in terms of beach wrack diversity, the presence of different sizes of logs, and the richness of supratidal invertebrates – was higher for natural shore types compared to armored shore types. Restoration was typically associated with relative increases in beach wrack diversity and invertebrate order richness, displaying a transition to levels more like those of natural shore types – a positive indication of restoration effectiveness. This finding is consistent with previous studies showing changes in wrack composition (Heerhartz et al., 2014) and increases in invertebrate richness with restoration (Toft et al., 2021). Restoration was not, however, significantly associated with changes in the presence of both log size classes, though natural shore types were more likely to have both large and small logs. Because logs are often deliberately placed on beaches during the restoration process (Johannessen et al., 2014; Toft et al., 2021),

Table 2 Summary ANOVA results of main ecological response variables (beach wrack, beach logs, and supratidal invertebrates) for spatial (sample site) and temporal (sample year) generalized linear mixed-effects models and post-hoc pairwise tests where the interaction term (sample year × shore type × sample site) was significant. Because only transects with logs present were included in the analysis of log size and growth richness, these could not be tested due to limited data across sites and years. Significance of pairwise test indicates the response was significantly different among sample years or sample sites for that shore type. Significance is denoted by asterisks: * p < 0.05; ** p < 0.01; *** p < 0.001. No asterisk denotes a non-significant comparison. Please see Table S1 for transformations and model families.

	Main Effects (X ² -values)			Interactions (X ² -values)			Post-hoc Pairwise Tests (F-ratios)						
	Sample year	Sample site	Shore type	Shore type × Sample year	Shore type × Sample site	Shore type × Sample year × Sample site	Armored control		Restoration treatment		Sample Site		
							Armored control	Natural reference	Armored control	Natural reference	Armored control	Restoration treatment	
Beach Wrack	Cover (%)	3.52	56.2***	2.07	5.92	42.2**	-	-	-	-	5.41***	6.24***	5.37***
	Depth (mm)	11.35*	25.50**	4.71	8.34	26.8	-	-	-	-	-	-	-
	Diversity (3 types)	8.20	38.8***	1.11	10.0	32.8*	-	-	-	-	3.60**	4.31***	1.75
Beach Logs	Number	10.3	171.5***	32.3***	12.0	141.0***	-	-	-	-	10.6***	19.0***	10.3***
	Line Width (m)	7.78	150.0***	34.3***	21.5*	109.3***	0.18	0.17	0.20	16.7***	12.5***	16.7***	7.10***
Supratidal Invertebrates	Density (n/m ² /day)	12.9*	36.2***	0.96	9.25	51.1***	-	-	-	-	1.40	7.24***	4.81**
	Family Richness	18.3**	118.4***	4.32	2.74	50.6***	-	-	-	-	5.63***	23.7***	4.56**
	Order Richness	46.9***	52.0***	10.7**	26.8*	43.1***	6.26***	9.38***	2.11	10.4***	8.85***	10.4***	5.90***

additional care should be taken to ensure representation of different size classes, which can contribute to structural heterogeneity and therefore more diverse nearshore communities (Heilmann-Clausen and Christensen, 2004). Additionally, more time may be needed post-restoration to see the full recovery of log size diversity (Toft et al., 2021). Further research should explore different elements of log structural diversity and whether it, as well as wrack compositional diversity, are positively associated with shoreline community biodiversity at higher trophic levels.

In addition to dissimilarities in mean values, shore types displayed differences in variance, which were likely driven by variability among sample sites. In particular, ecological variables measured in natural shore types consistently had higher variance than those in armored shore types. For beach wrack and supratidal invertebrates especially, restored shore types had variances more similar to natural than to armored shore types, indicating that restoration might recover some natural variation. Within shore type, variance in ecological responses was primarily driven by variation among sample sites rather than across sample years. This potentially indicates high habitat heterogeneity across the study region, which could reflect physical and biological variation in the characteristics of different shoreforms, such as wave energy regime and local geology (Dethier et al., 2016). All responses showed significant interactions between site and shore type, whereas only log line width and invertebrate order richness showed significant interactions between year and shore type. Within natural shore types, there was significant variation among sites for all responses. On the other hand invertebrate density did not differ significantly among armored sites, and wrack diversity did not differ among restored sites.

Researchers and practitioners are beginning to document and acknowledge the negative impacts of shoreline modification on ecological variability (Aguilera et al., 2014; Lawrence et al., 2021) and work in other systems has shown that anthropogenically modified ecosystems can be less variable across time and space (Buyantuyev and Wu, 2009; Gittman et al., 2016; Groffman et al., 2014; McKinney, 2006; Wu et al., 2011). For example, primary productivity is less variable through time in urbanized areas of Phoenix, AZ compared to areas with natural land cover (Buyantuyev and Wu, 2009). Numerous studies have cited the spatial homogenization of biological communities across urban and modified landscapes (Groffman et al., 2014; McKinney, 2006), including shorelines (Gittman et al., 2016). Spatial variation in beach wrack, logs, and invertebrate communities is common across natural coastlines and can be driven by interacting differences in shore slope, shore type, aspect, landcover, latitude, and sediment type, among other characteristics (Dethier et al., 2016; Heerhartz et al., 2014; Reimer et al., 2018; Romanuk and Levings, 2003). Armoring hampers this natural variation, homogenizing the shore structure and creating uniformity across sites (Dethier et al., 2016; Gittman et al., 2016). This has cascading effects on the ecological community. For example, regardless of location, armored shore types have more uniform sediment sizes, which are known to affect wrack composition (Heerhartz et al., 2014), nutrient ratios, and wrack invertebrate diversity and composition (Toft et al., 2021). Thus, we suggest that increases in the temporal and spatial variability can be a valuable indicator of effective restoration. Monitoring, and recovering this variability among sites, should be a key restoration objective.

Our work not only has implications for human-driven habitat modification and restoration, but also sets an important precedence for the study of public perception in socio-ecological relationships. Private single-family homeowners are responsible for the majority of recent shoreline armoring in Puget Sound (Puget Sound Partnership, 2021). The popularity stems at least partially from the implication of “protection” – defense against the combined threats of erosion, sea level rise, and storm surges (Cooper et al., 2020). As with much human modification, armoring also gives the appearance of a more “controlled”, “orderly”, and “cleaner” shoreline (Gittman et al., 2021). Sociological studies show that the relationships between modified landscapes and human perception are complex, with a mixed appreciation for “messy”

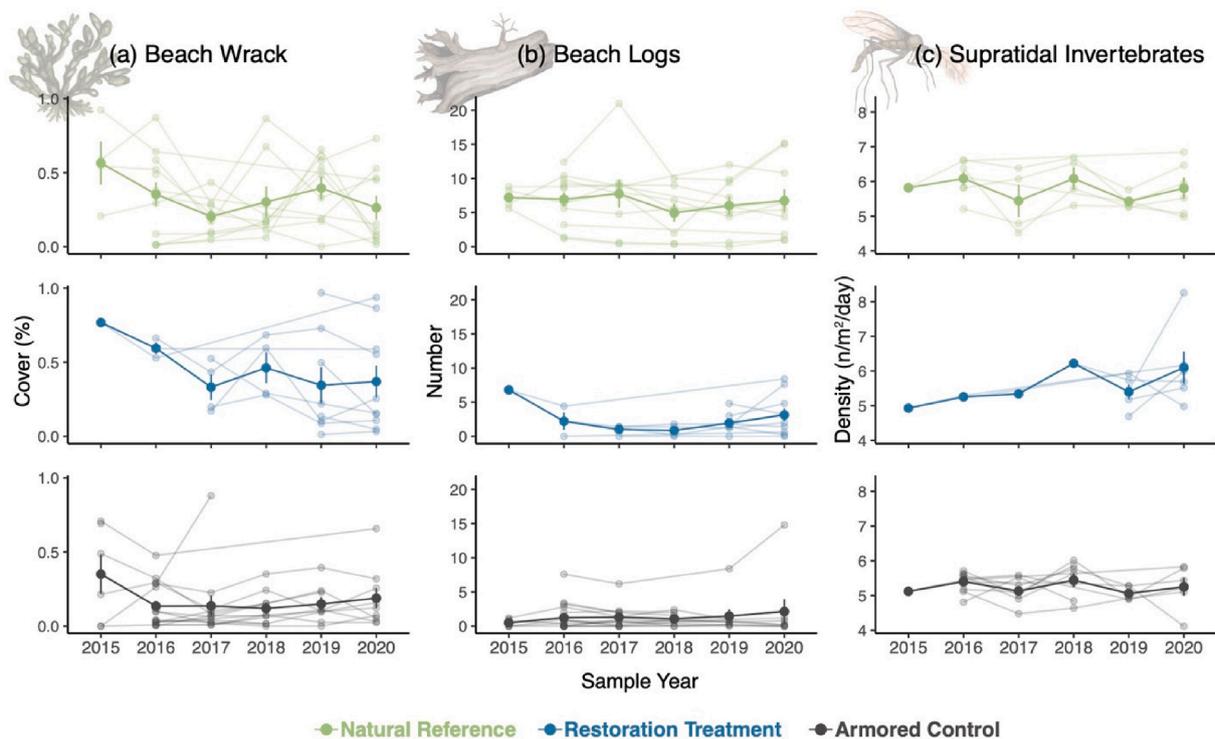


Fig. 4. Generalized Linear Mixed Effects model outputs showing marginal means and 95% confidence intervals for three representative response variables (a) percent wrack cover; (b) total number of logs; (c) total invertebrate density (number/m²/day) across sample years (x-axis) and sample sites (faded points and lines) for the three shore types: natural reference (green), restoration treatment (blue), and armored control (grey). Please see [Table S1](#) for transformations and model families. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

or more variable and diverse landscapes (Fairchild et al., 2018; Nassauer, 1995). For example, homeowners in the Seattle, WA, USA area often show a strong preference for properties with low-diversity monospecific lawns – a proclivity that is influenced by neighborhood norms, aesthetics, and property value (Fuentes, 2021). These sentiments can be partially extended to shoreline property owners (Gittman et al., 2021). Still, studies have shown that many residents of the Puget Sound area also emphasize “naturalness” as central to their sense of place (Trimbach and Biedenweg, 2021).

The success of shoreline restoration is multi-tiered, with both ecological and social implications. Though armor removal can help create more heterogeneous, variable, and healthy ecosystems, people have mixed perceptions of the value of this variability. A community, “citizen” science framework – such as that employed by our study – can help promote the value of these “messier” natural or restored systems through involving community members’ firsthand participation in monitoring surveys (Toft et al., 2017). Future work should more directly examine the impact of shoreline armoring and subsequent restoration on human perception (McAfee et al., 2021) – by surveying community scientist volunteers before and after participation in monitoring projects. We encourage researchers to broadly engage community members in restoration monitoring to increase awareness about the value and advantages of naturally diverse and variable ecosystems.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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