1 **Effectiveness of Living Shorelines in the Salish Sea**

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Abstract

In human-impacted coastal ecosystems, living shorelines are becoming a common restoration

technique. However, we lack a comprehensive understanding of the ecological and physical

benefits, and how they could inform management needs. To address this, we studied

effectiveness of living shorelines at a broad spatial scale within the Washington State

boundaries of the Salish Sea, USA, with restored site ages spanning 1-11 years. We surveyed 30

beaches at ten locations, each with three strata of: (1) living shoreline beaches with armor

removed, (2) armored control beaches altered by seawalls or riprap, and (3) un-armored

reference beaches with natural conditions. We sampled eight physical and biological attributes:

beach wrack, wrack invertebrates, sediments, terrestrial insects, riparian vegetation, logs,

beach profiles, and stable isotope signatures of talitrid amphipods – generating 27 metrics

focusing on upper intertidal and supratidal elevations affected by armoring and targeted by

living shoreline actions. These metrics spanned the functions of beach stability, ecological

diversity, and food web support for juvenile salmon. Statistical tests showed that 19 of the 27

metrics had significant strata differences, indicating that some beach metrics restore quickly

(e.g., wrack accumulation), while others take longer (e.g., log accumulation). Terrestrial-

associated metrics were higher at reference beaches, but insect taxa richness and logs with

plant growth increased at beaches restored for four or more years (the average age of the living

shoreline sites). This implies that certain living shoreline functions increase through time,

providing improved food web support. Globally, trajectories of restoration have shown a range

of functional improvement with time, and will be important to monitor for nature-based

36   solutions to coastal defense given the increasing rate of shoreline stressors from global change

37   and sea level rise.

38   **Keywords:** restore, shores, armoring, invertebrates, habitat

1. Introduction

Human-associated pressures are prevalent in the land-sea ecotone of coastal systems (MEA, 2005; Neumann et al., 2015). The proliferation of infrastructure, termed ocean sprawl, has degraded natural resources along with a host of other stressors (Firth et al., 2016). With continued population growth, the struggle to balance shoreline ecological health with human needs will intensify (Arkema et al., 2013). Intertidal areas of the land-sea ecotone are a focus of human modification via practices such as land reclamation (Morris et al., 2019a; Duan et al., 2016) and use of shoreline armor to protect property and infrastructure from erosion (Ma et al., 2014). In waters of the Salish Sea within the state of Washington, USA, 29% of shorelines are armored (MacLennan et al., 2017), compared to 14% across the total US coastline (Gittman et al., 2015), but less than highly urbanized and rapidly developing coastlines, such as those along mainland China where up to 60% of the shoreline is armored (Ma et al., 2014). The coastal squeeze caused by humans building waterward, and sea levels rising landward due to global climate change, will intensify the ecological impact of future shoreline management decisions. Given that current applications may not be appropriate under future conditions, living shoreline techniques are increasingly being implemented, providing nature-based restoration solutions that address both shoreline stabilization and ecological function.

Shoreline armoring, typically vertical seawalls constructed of concrete or wood, or sloping rip-rap of placed rock, has negative ecological effects globally (Gittman et al., 2016b). This compromises the host of functions and ecosystem services that natural coasts provide (Dethier et al., 2016b). In tidal systems, armor can reduce wrack and logs on shorelines, disrupting the connections between marine and terrestrial ecosystems (Sobocinski et al., 2010;

61   Dethier et al., 2016a), a phenomenon also observed in freshwater systems (Wensink and Tiegs,

62   2016). Aquatic invertebrate and terrestrial arthropod communities are also negatively affected

63   by armoring (Chapman, 2003; Romanuk and Levings, 2003; Morley et al., 2012; Dethier et al.,

64   2016a; Dugan et al., 2017). These invertebrates are prey for small fishes, such as juvenile

65   Chinook salmon (*Oncorhynchus tshawytscha*) in the Salish Sea. Chinook salmon are listed as

66   threatened under the Endangered Species Act in Puget Sound waters of the Salish Sea, where

67   there are many co-occurring stressors including habitat loss, pollution, reduced water flows,

68   and eutrophication (Munsch et al., 2016; Toft et al., 2018). Shoreline armor can therefore

69   adversely affect foraging by these threatened fish (Toft et al., 2007; Munsch et al., 2014), and

70   may have impacts elsewhere in the food web, such as to birds, seals and orca whales that

71   depend on salmon as main food sources (Cascadia, 2018). Physical beach dynamics can also be

72   altered (Ruggiero 2010), having negative implications for beach spawning fish such as surf smelt

73   (*Hypomesus pretiosus*) that lay eggs in sediments on the upper beach (Rice, 2006), further

74   affecting the food web (Cascadia, 2018).

75   Examining the ecological value of removing or replacing shoreline armor where feasible

76   and restoring beaches is timely, given continued increases in coastal populations and the

77   already fragmented nature of coastal aquatic ecosystems (Beach, 2003). In this context we

78   often refer to 'restoration' as a vernacular phrase, whereas 'rehabilitation' is a more reasonable

79   goal (Firth et al., 2016). Although shoreline protection structures are often necessary to protect

80   people and property, there are promising "living shoreline" options that are nature-based, also

81   referred to as soft or green shorelines (Nordstrom, 2014). These options can be applied to

82   situations where complete restoration is either impractical or not feasible given human

83   constraints, and depending on site characteristics, some engineering is usually required for

84   stability. Living shoreline techniques often include a mix of design options, including armor

85   removal, sediment nourishment of beaches, log placement, planting vegetation, moving

86   seawalls further inland, incorporation of marsh sills (Johannessen et al., 2014; Gittman et al.,

87   2016a; Esteves and Williams, 2017), and lower tidal natural features such as oyster beds and

88   eelgrass (Boyer et al., 2017; Morris et al., 2019b). Extremely urbanized shorelines can also be

89   enhanced with eco-engineering to create more structurally complex armoring (Strain et al.,

90   2017; Sawyer et al., 2020), or artificial beaches (Toft et al., 2013). All of these techniques

91   contain some degree of habitat restoration, which traditionally refers to restoring habitat to a

92   natural state (Perring et al., 2015), such as in the removal of levees to restore tidal estuarine

93   marshes (Zedler, 2001). Marsh restoration has been shown to be successful even in small scale

94   industrialized settings (Cordell et al., 2011), but the full value of living shoreline implementation

95   in nearshore areas is an emerging topic (Gittman et al., 2016a), in part due to its recent

96   development as a restoration practice (Bilkovic et al., 2017).

97   In this study we evaluate the effectiveness of removing armor and restoring beaches as

98   a living shoreline technique, comparing results to reference natural-type and control armored

99   beaches. By using the same sampling methods at beaches that have had armor removed for a

100 varying number of years, we build upon previous studies to assess how living shorelines

101 function across broader spatial and temporal scales (Lee et al., 2018). Our sampling focused on

102 invertebrates and structural habitat components of beach wrack, logs, vegetation, and

103 sediments at upper intertidal and supratidal elevations most impacted by armoring (Dethier et

104 al., 2016a) and often targeted for restoration (Toft et al., 2014). Our goal is to assess the

105 effectiveness of living shorelines, and provide managers with new information on how to

106 adaptively plan for the future of developed coasts.

107

108 **2. Methods**

109 *2.1. Study Area*

110   Study sites were located in the Salish Sea, an estuarine fjord with mixed semidiurnal

111 tides (Figure 1). They were all within the Washington State, USA portion of the Salish Sea but

112 spanned a wide geographic area, including the San Juan Islands (Brown Island and Bowman

113 Bay), and multiple locations in Puget Sound including the Whidbey Basin (Cornet Bay), Hood

114 Canal (Dabob Bay), Central Basin (Powel on Bainbridge Island, Anna Smith in Dyes Inlet, two

115 locations at Seahurst Park on the eastern mainland, Dockton on Maury Island), and the South

116 Basin (Penrose in Carr Inlet). Beaches in the Salish Sea are primarily composed of sand-gravel

117 mixed sediments, originating from receding glaciers and maintained by continued erosion of

118 coastal bluffs. Sediments are distributed via longshore transport within drift cells of source to

119 deposition. Shoreline armor is often placed to prevent erosion and can impede sediment input,

120 and occurs on 29% of shorelines within our study area (MacLennan et al., 2017). Armor material

121 varies (e.g., concrete seawalls, stone riprap) and is placed at various tidal elevations above or

122 below Mean Higher High Water (MHHW).

123   We surveyed 30 beaches at ten locations, each with three strata of: (1) living shoreline

124 beaches with armor removed, (2) armored control beaches altered by seawalls or riprap, and

125 (3) un-armored reference beaches with more natural conditions. This allowed us to compare

126 metrics of living shoreline beaches (hereafter referred to as "restored") to those from adjacent

127   degraded (armored) and natural (reference) levels. The restored beaches spanned ages of 1-11

128   years, averaging four years, and a length of armor removed of 30-549 m, averaging 194 m

129   (Table 1). Living shoreline elements including sediment nourishment, log placement, and

130   vegetation planting were implemented as appropriate at each site, to mimic natural beach

131   characters. At each location, proximity, geomorphic setting, aspect to prevailing waves and

132   weather, and nearshore bathymetry among the three sampling strata were matched as closely

133   as possible as per previous studies (Dethier et al., 2016a).

Site	Year of restoration	Years restored in 2016	Length of armor removed (m)
Anna Smith	2012	4	198
Bowman Bay	2015	1	165
Brown Island	2015	1	61
Cornet Bay	2012	4	250
Dabob Bay	2009	7	30
Dockton	2013	3	107
Penrose	2013	3	213
Powel	2012	4	30
Seahurst 2005	2005	11	335
Seahurst 2014	2014	2	549
	Average:	4	194

134   **Table 1.** Year and length of armor removed at living shoreline sites.
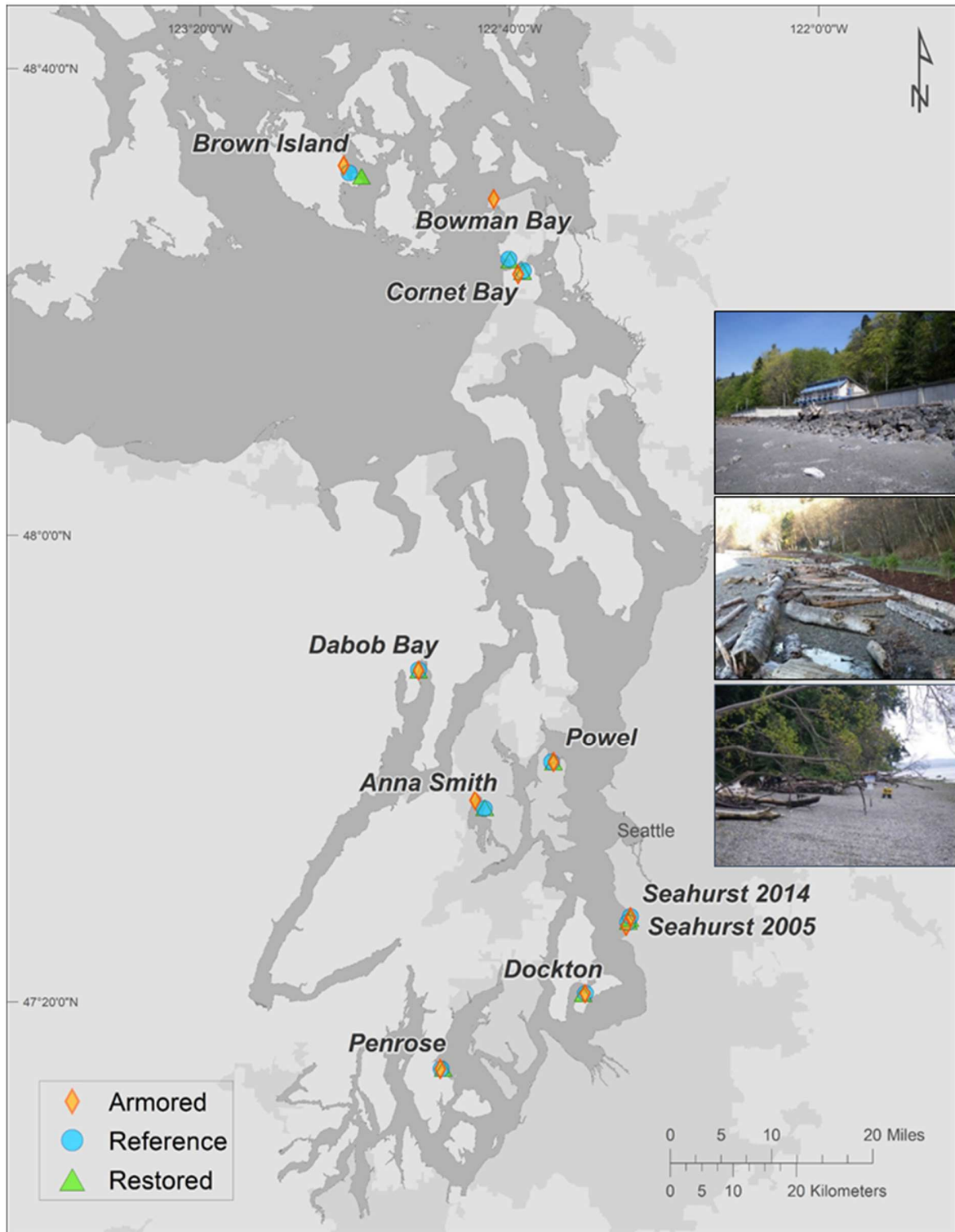
Figure 1. Map of locations and strata. Shown are representative photographs from Seahurst 2014 of armored, restored, and reference strata (top to bottom) sampled at each location.

138

2.2 Data Collection

Field data were collected June-August 2016, using methods developed during a previous

study on armoring in the region (Dethier et al., 2016a; also see protocols at the Shoreline

Monitoring Database, shoremonitoring.org). Wrack percent cover of algae, eelgrass, and

terrestrial sources deposited on the beach on an ebbing tide was quantified using a 0.1 m^2

quadrat at ten random points along a 50 m transect parallel to the beach. At two of the sites

(Dabob Bay and Powel), a 30 m transect was used due to site configurations. Expanding on

previous efforts, we focused on the most recent wrack deposition (new), and also on a higher

supratidal elevation of older wrack when present. Wrack depth and overall width of the wrack-

line were measured at each quadrat.

Invertebrates and sediments were sampled with a 15 cm diameter, 2.5 cm depth core at

five random points along the new wrack transect and frozen for later processing. Invertebrates

were separated from the samples and sorted, identified, and enumerated under dissecting

microscopes. Sediment sizes were determined by shaking dried samples in sieves in a RoTap

shaker and weighing the amount collected in each sieve. When present, talitrid amphipods

(family Talitridae, also known as "beach-hoppers") from the core samples were analyzed for

stable isotopes. Stable isotopes of carbon, nitrogen, and sulfur were processed to discriminate

between terrestrial and marine sources of food web support at five of the locations for all

strata (Bowman, Cornet, Dabob, Penrose, and Seahurst Park 2005), with up to five replicates of

pooled individuals at each strata to reduce individual variation. Three replicates of organic

material (OM) sources were collected from beach wrack and processed for stable isotopes.

160 Source materials included eelgrass (*Zostera* spp.), algae (*Ulva spp.,* filamentous green algae, and

161 *Fucus distichus*), and terrestrial alder (*Alnus rubra*) leaves. Laboratory procedures for stable

162 isotopes were similar to those in Howe and Simenstad (2015). Collected OM sources and

163 amphipods were frozen until processing in the lab. Foliage of OM sources was rinsed in dilute

164 10% HCl to remove soil carbonates and then rinsed in deionized water until neutral pH.

165 Amphipods were thoroughly rinsed in deionized water to remove sediments and OM debris,

166 placed into sterile scintillation vials, and freeze-dried for 48 h, and ground to a fine powder

167 using a Wig-L-Bug® dental mill and a stainless steel vial and ball pestle. Samples were weighed

168 into tin capsules for isotope processing of $\delta^{13}C$, $\delta^{15}N$, and $\delta^{34}S$ (^{13}C and ^{15}N samples 0.4 mg, ^{34}S

169 samples 2.5 mg). Isotope analyses were performed by Washington State University's Stable

170 Isotope Core Laboratory using a Costech Analytical ECS 4010 elemental analyzer connected via

171 a gas dilution to a Thermo Finnigan Delta Plus XP mass spectrometer. Isotope ratios were

172 calculated as:

173 $\qquad \delta X\ (‰) = [(R_{sample}\ /\ R_{standard})\ \text{-}1] \times 10^3$

174 $\qquad$ where X = ^{13}C, ^{15}N, or ^{34}S, and R= $^{13}C/^{12}C$, $^{15}N/^{14}N$, or $^{34}S/^{32}S$

175 Acetanilide, yeast, and keratin were used as standards, calibrated against International Atomic

176 Energy reference materials (Vienna Canon Diablo Triolite (VCDT) ($\delta^{34}S$), Vienna Peedee

177 belemnite (VDDB) ($\delta^{13}C$), and atmospheric nitrogen ($\delta^{15}N$). Precision of replicate

178 determinations was ± 0.05‰.

179 $\qquad$ We counted the number of logs (driftwood) and the width of the log-line perpendicular

180 to shore at the same five random points that benthic cores were collected. If present,

181 secondary plant growth on the logs (e.g., grasses, weeds, moss) was noted. We also measured

182   percent of the transect that had overhanging vegetation and counted number of trees fallen

183   from the bank. Fallout traps (40 x 25 cm plastic bins with a small amount of soapy water) were

184   deployed for 24 hours to sample terrestrial insects and arthropods in supratidal habitats at five

185   random points along the wrack transect. Samples were preserved in 70% isopropanol and

186   returned to the laboratory and sorted, identified, and enumerated under dissecting

187   microscopes.

188   Beach profiles were determined during low tides using a laser level and stadia rod or

189   RTK-GPS along a transect perpendicular to shore, starting from above MHHW at top of the

190   berm, or base of armoring or bluff if below that. Beach width and slope were calculated down

191   to MLW elevation. Elevations of the toe of armor or bluff (hereafter referred to as relative

192   encroachment [RE]) and the new wrack line (hereafter referred to as WrRE) relative to MHHW

193   were noted. These were specific to the datum at each location; see Dethier et al. (2016a) and

194   supplementary materials therein for further details on geomorphic methods.

195

196   *2.3 Statistical Analyses*

197   We used a modeling analysis to test metrics with the main fixed effect of strata

198   (restored-armored-reference) as well as related parameters (% overhanging vegetation, RE,

199   WrRE, % sand sediment size, % wrack cover, and wrack depth), depending on the metric being

200   tested. The percent of each drift cell that was armored (DCA) was also included as a potential

201   parameter (as per Dethier et al., 2016a). Location was included as a random effect when there

202   were replicates of samples (Dethier et al., 2016a). Specific models depended on the metric

203   being tested and distributions therein (Zuur et al., 2009), and were based on our previous

204 analyses that tested for the effects of armoring (Dethier et al., 2016a). Percent cover or

205 proportion data were arcsine-square root transformed to improve normality and tested with a

206 linear mixed model (Dethier et al., 2016a). Measurements of wrack depth and width, number of

207 logs and width of the log-line, and invertebrate taxa richness and diversity were tested with a

208 linear mixed model. Taxa richness was the number of taxa in each sample, and diversity was

209 calculated using the Shannon-Wiener index. Counts of wrack invertebrates and terrestrial

210 insects were tested using a generalized linear mixed model, following a negative binomial

211 distribution to account for overdispersion and to adequately fit biological count data (Zuur et

212 al., 2009). Measurements of beach width and slope, RE, wrack RE, overhanging vegetation, and

213 fallen trees, that had one value for each transect, were tested for strata with a linear model.

214   For models with multiple fixed parameters, we used model selection based on *AIC*

215 criteria (Zuur et al., 2009). Residual plots and fitted values were examined for final model fits.

216 When strata were significant ($P < 0.05$), multiple comparisons were made between the three

217 restored-armored-reference strata. Restored strata were also analyzed separately, to test for

218 specific parameters of years restored and length of armor removed. Modeling analyses were

219 conducted in R version 3.5.2 (R Core Team, 2018) using the lme4 (Bates et al., 2015), lmertest

220 (Kuznetsova et al., 2016), and Multcomp (Hothorn et al., 2008) packages.

221   Multivariate analyses of wrack, wrack invertebrate, and insect assemblages were tested

222 using PERMANOVA (Anderson et al., 2008) with a Bray-Curtis resemblance matrix, again with

223 the main fixed effect of strata (restored-armored-reference) and post-hoc multiple comparison

224 tests when significant, and location as a random effect. Percent cover wrack data were sqrt-

225 transformed, and count invertebrate and insect data were log-transformed before analysis,

226   with taxa representing less than 3% of the total abundance of any one sample removed from

227   assemblage analysis (Clarke and Gorley, 2006). The two elevations of new and old wrack were

228   included as a fixed factor in tests of wrack percent composition. Covariates were included for

229   elevation (WrRE) in the wrack invertebrate assemblage analysis, and percent overhanging

230   vegetation in the insect assemblage analysis.

231   The proportional contribution of the three OM end-members (marine algae, eelgrass,

232   and tree leaves) to talitrid amphipod diets was analyzed with a Bayesian stable isotope mixing

233   model MixSIAR (Stock and Semmens, 2016) in R version 3.5.2 (R Core Team, 2018). Because

234   their isotope signatures overlapped, we pooled marine algal species (*Fucus distichus, Ulva* spp.,

235   filamentous green algae*)* into a single category (*n* = 42). We separately pooled the δ^{13}C, δ^{15}N,

236   and δ^{34}S isotope values of each OM group across sites; OM groups consisted of marine algae,

237   eelgrass (*Zostera* spp.; *n* = 18) and terrestrial alder leaf (*Alnus rubra*; *n*= 11). Summary statistics

238   of OM source isotope values were delivered to the model (Supplemental Table 1). MixSIAR

239   consequentially fits a fully Bayesian model by estimating source means and variances for each

240   tracer and assumes tracers to be independent (Stock et al., 2018).

241   Following the recommendation of Schlacher and Connolly (2014), we applied a δ^{13}C shift

242   of -1.2 +/- 0.19 ‰ (SE) to all amphipod samples in our dataset to account for inorganic carbon

243   in amphipod exoskeletons that is not reflective of diet. This value is specific to peracarids,

244   including isopods, amphipods, mysids, and tanaids. Mean trophic discrimination factors (TDF)

245   values were obtained from Mancinelli's (2012) review of amphipod feeding experiments (δ^{13}C =

246   -2.4 ± 2.0‰; and δ^{15}N = 1.6 ± 1.1‰). We used McCutchan et al.'s (2003) value for all animals

247   for δ^{34}S (δ^{34}S = 0.5 ± 1.94 ‰) due to concerns regarding small sample size for sulfur.

248   Non-informative priors were used for each OM source. Mixing models can be sensitive

249   to priors (Brown et al. 2018); with only three non-overlapping sources, the use of three tracer

250   isotopes, and a tight consumer isotope cloud that falls within the source geometry polygon, our

251   data are likely less susceptible to model bias produced as a consequence of prior choice. We

252   incorporated fixed (strata) and random (location) effects as covariates to explain variability in

253   mixture proportions as described in Stock et al. (2018). We selected "normal" for the model

254   runtime. Gibbs sampling was performed for each model using three parallel chains in JAGS

255   (Plummer, 2003). Gelman-Rubin and Geweke diagnostics were performed as part of the

256   MixSIAR model package and showed good convergence. We used PERMANOVA (Anderson et

257   al., 2008) to test for differences in talitrid amphipod isotope values across strata of restored,

258   reference, and armored beaches using a Euclidean distance resemblance matrix as

259   recommended for data with negative values (Clarke and Gorley, 2006), similar to above

260   analyses.

261

262   **3. Results**

263   Most metrics tested showed significant differences among restored, reference, and

264   armored strata, determined by final models based on *AIC* selection or PERMANOVA analyses

265   (19 of 27; Table 2). Total percent cover of new wrack was higher at restored and reference than

266   at armored strata (Table 2, Figure 2). Percent of terrestrial components in new wrack was

267   highest at reference strata, algae was highest at restored strata, and eelgrass components

268   showed no differences. In aggregate, these results were also significantly different for overall

269   composition (PERMANOVA). For old wrack, restored and reference strata were similar to each

270 other in composition but both differed from armored strata composition when it occurred

271 (three, seven, and ten locations at armored, reference, and restored strata, respectively). There

272 were no differences in among-strata wrack width; new wrack depth was highest at restored

273 strata (Table 2, Figure 3). Number of logs and log-line width were higher at reference than

274 restored and armored strata. The percent of logs with plant growth was higher at reference

275 than armored strata (average of 11% and 0%, respectively), and neither were different from

276 restored strata (9%).

277   Three taxa—nematode and oligochaete worms and talitrid amphipods—made up 96%

278 of invertebrate densities in the wrack core samples. Nematode and oligochaete worms

279 dominated wrack invertebrate densities collected at armored strata and were significantly

280 higher at armored and reference than at restored strata (Table 2, Figure 2). Talitrid amphipods

281 (*Traskorchestia* and *Megalorchestia* spp.) had higher densities at reference than at restored and

282 armored strata. Shannon-Wiener diversity index values were higher at restored and reference

283 than at armored strata, but there were no differences in taxa richness among the strata (Table

284 2, Figure 3). Wrack invertebrate assemblages were overall significantly different among strata

285 (PERMANOVA), with all strata being different from each other. Total terrestrial insect and

286 arthropod densities collected in fallout traps were higher at reference than at restored and

287 armored strata (Table 2, Figure 2). Shannon-Wiener diversity index values were higher at

288 reference than at restored strata, and taxa richness showed no significant strata differences

289 (Table 2, Figure 3). Overall insect and arthropod assemblages were significantly different among

290 strata (PERMANOVA), with restored and reference strata being similar and both different from

291 armored strata. Some of these assemblage similarities at restored and reference strata were

292   due to non-flying arthropods such as acari (mites) and collembola (springtails) that may exhibit

293   more localized site fidelity than flying insects.

294   Of the physical and vegetative structural metrics that were measured, two were

295   significantly related to strata – the relative encroachment (RE) of the toe of armor or bluff

296   below the MHHW tidal elevation was higher at armored than at restored and reference strata,

297   and percent of overhanging vegetation was higher at reference than at restored and armored

298   strata (Table 2, Table 3). Percent sand collected in sediment samples, beach width, beach slope,

299   Wrack RE, and number of fallen trees did not differ among strata.

300   Several other parameters were significant in most final models based on *AIC* selection

301   (14 of 19 metrics with additional parameters included; Table 2). Of these, beaches extending

302   above MHHW (i.e., negative values of RE) had more eelgrass in the wrack, wider wrack-lines,

303   and more logs (Figure 4 A-C); wrack invertebrate diversity and wrack depth had similar but

304   more subtle patterns (Supplemental Figure 1 A-B). Conversely, percent sand in the cores was

305   higher when armor or bluff descended below MHHW (positive values of RE) (Figure 4 D).

306   Wrack-line values of total and algae percent cover, and amphipod densities were generally

307   highest around MHHW (WrRE values of 0; WrRE varied due to high tide levels on a given day of

308   sampling; Figure 4 F-G; Supplemental Figure 4 C). Other metrics were higher when wrack was

309   centered either below MHHW (terrestrial percent cover; Supplemental Figure 1 D), at or above

310   MHHW (wrack invertebrate taxa richness; Supplemental Figure 1 E), or were variable in their

311   responses (wrack width and wrack invertebrate diversity; Supplemental Figure 1 F and 1 G).

312   Density of amphipods and taxa richness of wrack invertebrates showed a positive response

313   with percent total wrack cover, especially increasing from 0 to 50% wrack cover (Figure 4 H-I).

314   Diversity and taxa richness of wrack invertebrates generally increased with wrack depth,

315   especially between 0 to 1 cm (Supplemental Figure 1 H-I). Density of nematode and oligochaete

316   worms in the wrack had a positive relationship with percent sand (Figure 4 J), while taxa

317   richness and diversity of wrack invertebrates were highest at middle values of percent sand

318   (Supplemental Figure 1 J-K). Taxa richness of terrestrial insects and arthropods had a slight

319   positive relationship with percent of overhanging vegetation (Figure 4 E), which was also a

320   significant covariate in the insect assemblage analysis; the response of wrack total percent

321   cover was variable (Supplemental Figure 1 L). The DCA parameter (percent of drift cell armored)

322   was not selected in any of the final models.

323   Specific to analyses of the restoration strata, year since restoration was the only

324   parameter that showed significance in final models based on *AIC* selection (Table 4). The

325   terrestrial-associated metrics of logs with plant growth and insect taxa richness showed a

326   positive increase with number of years restored (Figure 5). Linear length of restoration, and

327   percent overhanging vegetation were not significant parameters.

328   We observed strong differences in the $\delta^{13}C$ and $\delta^{34}S$ values among OM sources, and less

329   distinction in $\delta^{15}N$ values (Figure 6, Supplemental Table 1, Supplemental Figure 3). Marine algae

330   and eelgrass were more enriched in all three isotopes compared to terrestrial OM, which

331   displayed negative ^{15}N values and variable $\delta^{34}S$ values. Isotope signatures of talitrid amphipods

332   significantly differed across strata of restored, reference, and armored (Table 2). Across all

333   strata, talitrid amphipod diets were dominated by marine algae, followed by eelgrass, then

334   terrestrial OM (Table 5). Talitrid amphipod diets reflected the strata in which they occurred: the

335   proportion of marine algae contributing to diets was highest at reference strata, followed by

336   restoration strata, and then armored strata.

337

| | Potential Parameters | | | | | | | |
Metric and Model Type	Strata	%Veg	RE	WrRE	%Sand	%Wrack	Depth	Strata Differences
Wrack Total % [a]	**2.2E-16**	0.012		0.0002				Rest&Ref>Arm
Wrack Terrestrial % [a]	**5.7E-16**			0.002				Ref>Rest>Arm
Wrack Algae % [a]	**2.0E-16**			0.017				Rest>Ref>Arm
Wrack Eelgrass % [a]	0.98		7.2E-06					NA
Wrack % Composition [f]	**0.0001**							New: Ref≠Rest≠Arm
(New and Old Wrack)								Old: Ref&Rest≠Arm
Wrack Depth [b]	**0.0006**		0.023					Rest>Arm&Ref
Wrack Width [b]	0.89		2.2E-16	1.4E-10				NA
Number of Logs [b]	**1.1E-09**		0.004					Ref>Rest&Arm
Log-line Width [b]	**2.5E-07**							Ref>Rest&Arm
Log % Plant Growth [a]	**0.036**							Ref>Arm
Wrack Worms [c]	**0.001**				1.3E-15			Ref&Arm>Rest
Wrack Amphipods [c]	**0.008**			0.003		5.2E-09		Ref>Rest&Arm
Wrack Taxa Richness [b]	**0.043**			0.011	0.0002	0.019	0.011	NA
Wrack Diversity [b]	**0.0004**		0.003	0.034	0.009		0.005	Rest&Ref>Arm
Wrack Invert. Assemblage [f]	**0.0001**							Ref≠Rest≠Arm
Talitrid Isotope Signature [f]	**0.0007**							Ref≠Rest≠Arm
Insect Total [c]	**1.1E-05**							Ref>Rest&Arm
Insect Taxa Richness [b]	0.32	**0.022**						NA
Insect Diversity [b]	**0.045**							Ref>Rest
Insect Assemblage [f]	**0.0001**	0.0001						Ref&Rest≠Arm
Sediment Sand % [a]	0.17		6.0E-07					NA
Beach Width (m) [d]	0.68							NA
Beach Slope [d]	0.26							NA
RE [d]	**0.010**							Arm>Rest&Ref
Wrack RE [d]	0.90							NA
Overhanging Vegetation % [e]	**0.001**							Ref>Rest&Arm
Fallen Tree Number [d]	0.23							NA

338 Notes: [a] linear mixed model-sqrt arcsin transformed, [b] linear mixed model, [c] generalized linear mixed model-
339 negative binomial, [d] linear model, [e] linear model-sqrt arcsin transformed, [f] Permanova. RE (relative encroachment
340 of armor or bluff to MHHW elevation), WrRE (wrack line RE), DCA (% drift cell armored), %Veg (overhanging
341 vegetation), %Sand (sediments at each sample), %Wrack (Total at each sample), Depth (Wrack depth at each
342 sample), Wrack Worms (oligochaetes and nematodes), Rest (restored), Arm (armored), Ref (reference).

343

344 **Table 2**. Summary of overall statistical tests. Parameters in potential models are highlighted in

345 grey for each metric, *P*-values are shown for parameters selected in final models based on *AIC*

346 criteria, and are highlighted in bold if significant. The strata *P*-value is shown for all models.

Strata	Beach width	Beach slope	%Sand	RE	WrRE	%Veg	Fallen trees
Armored	25.0	0.09	0.43	**0.28**	0.38	24	0.7
Restored	26.6	0.12	0.30	-0.67	0.20	32	0.6
Reference	29.3	0.10	0.30	-0.41	0.29	**90**	1.9

Table 3. Average physical and vegetative characteristics at armored, restored, and reference strata. Column headings are same as defined in Table 2, significant differences highlighted in bold.

Metric and Model Type	Potential Parameters		
	Year	Length	%Veg
Wrack Total % [a]	0.19		
Wrack Terrestrial % [a]	0.29		
Wrack Algae % [a]	0.32		
Wrack Eelgrass % [a]		0.25	
Wrack Depth [b]		0.11	
Wrack Width [b]	0.40		
Number of logs [b]	0.29		
Log-line Width [b]	0.33		
Log % Plant Growth [a]	**0.026**	0.09	
Wrack Worms [c]		0.24	
Wrack Amphipods [c]		0.09	
Wrack Taxa Richness [b]	0.19		
Wrack Diversity [b]	0.35		
Insect Total [c]	0.07		
Insect Taxa Richness [b]	**0.007**		
Insect Diversity [b]	0.18		

Notes: [a] linear mixed model-sqrt arcsin transformed, [b] linear mixed model, [c] generalized linear mixed model-negative binomial. Year (of restoration), Length (armoring removed), %Veg (overhanging vegetation), Wrack Worms (oligochaetes and nematodes).

Table 4. Summary of statistical tests specific to restored strata. Parameters in potential models are highlighted in grey for each metric, *P*-values are shown for parameters selected in final models based on *AIC* criteria, and are highlighted in bold if significant.

360

|	Marine Algae		Eelgrass		Terrestrial	
Strata	Mean	Std. Dev.	Mean	Std. Dev.	Mean	Std. Dev.
Armored	0.503	0.130	0.455	0.131	0.043	0.037
Restored	0.558	0.127	0.389	0.126	0.053	0.045
Reference	0.608	0.125	0.369	0.123	0.023	0.024

361

362 **Table 5.** Summary statistics of MixSIAR posterior probability distributions estimating the mean

363 (±SD) diet contribution of OM sources to talitrid amphipods collected at armored, restored, and

364 reference strata. Posterior probability distribution plots are available in Supplemental Figure 2.

365
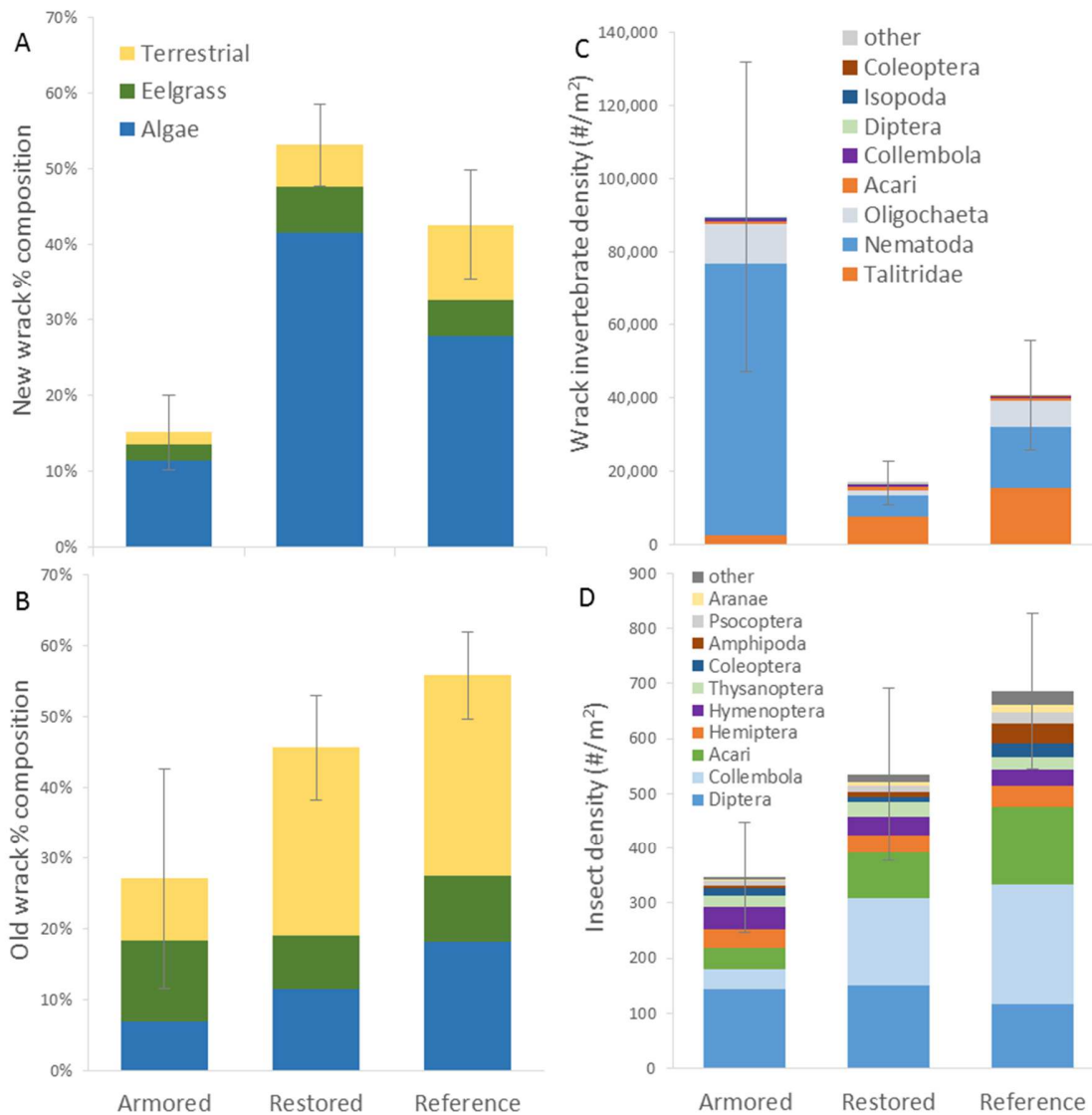
Figure 2. Percent composition at armored, restored, and reference strata for (A) new wrack measured in quadrats; (B) old wrack measured in quadrats; (C) wrack invertebrate densities sampled in cores; and (D) insect and other arthropod densities sampled with fallout traps. Error bars are standard error.
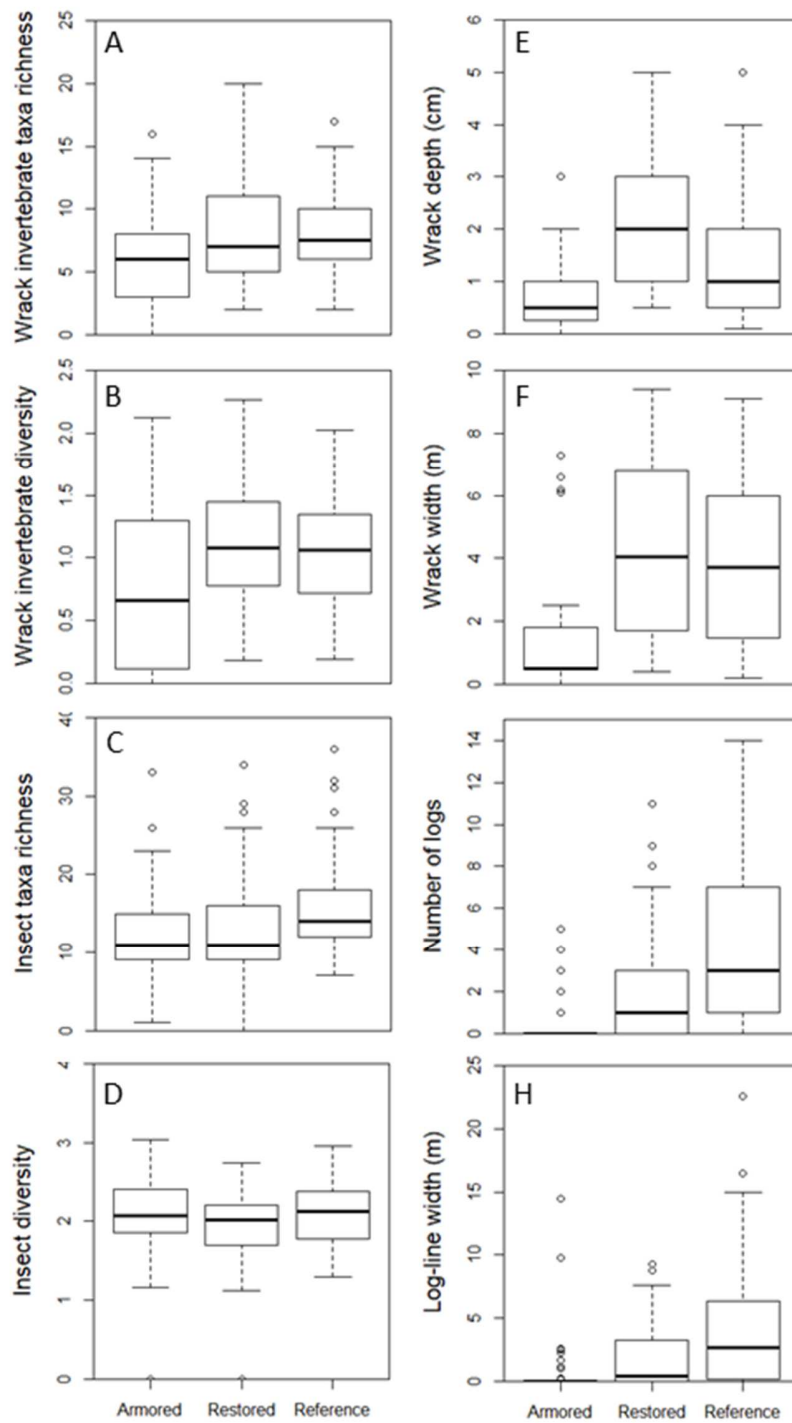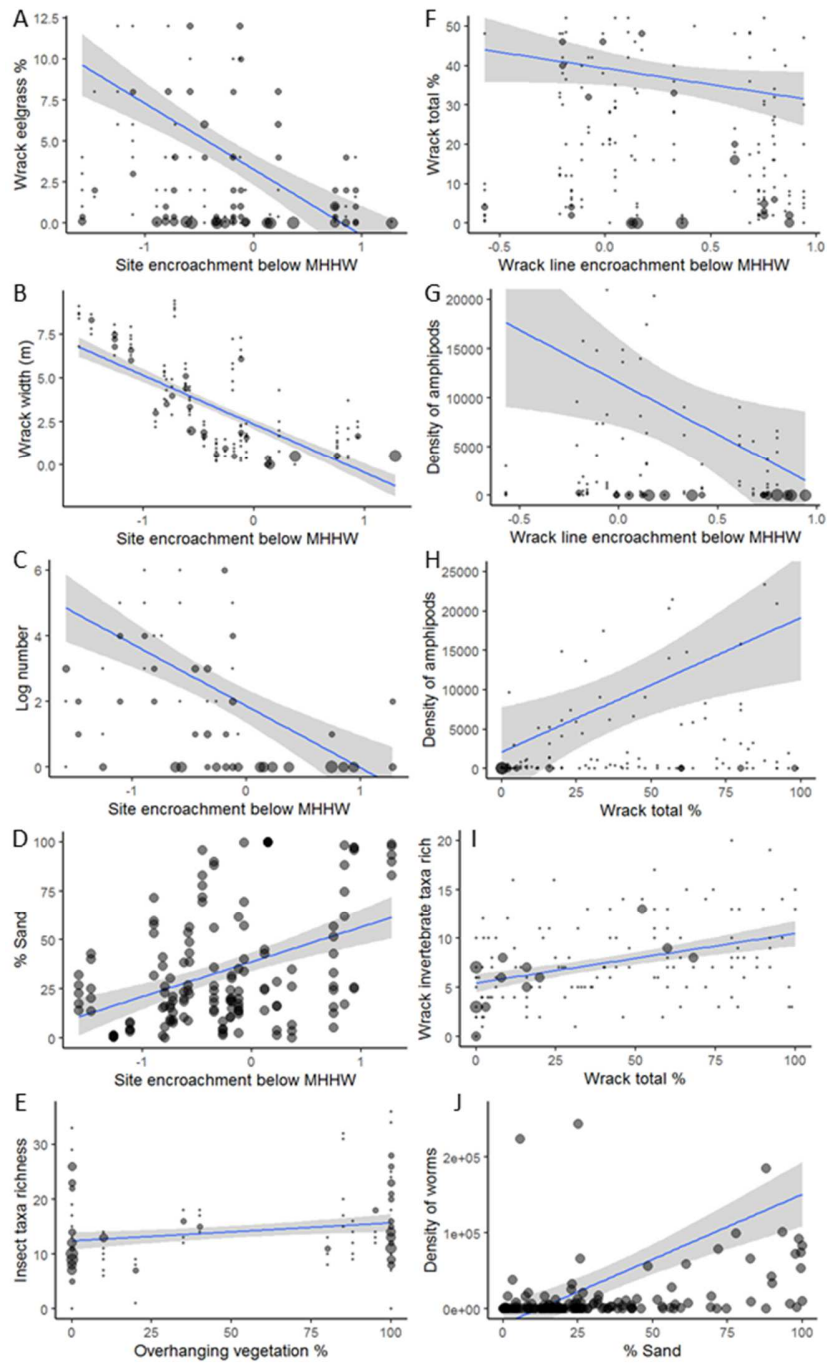
Figure 3. Strata boxplots of (A) wrack invertebrate taxa richness; (B) wrack invertebrate diversity; (C) insect taxa richness; (D) insect diversity; (E) wrack depth; (F) wrack width; (G) number of logs; and (H) width of the log-line.
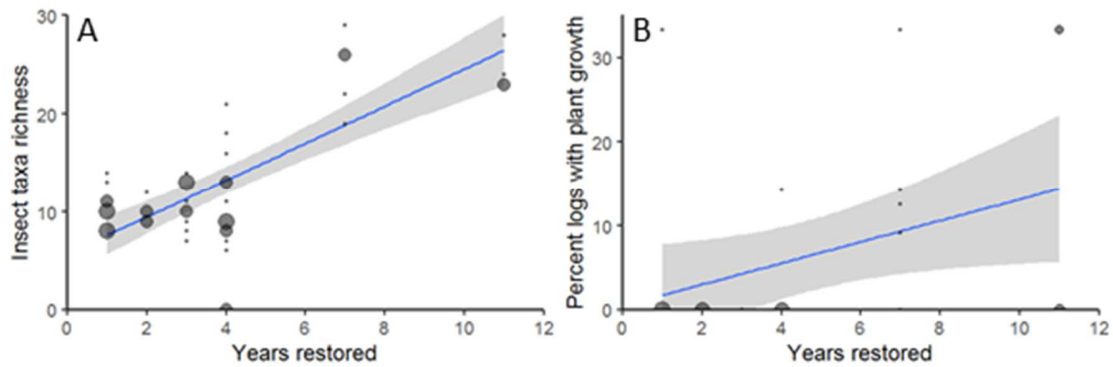
Figure 4. Statistically significant relationships between measured parameters and response

metrics, arranged by parameters RE (A-D), overhanging vegetation (E), WrRE (F-G), wrack total

% (H-I), and % sand (J). Additional plots are in Supplemental Figure 1. Lines represent linear

relationships with 95% confidence intervals, size of data points are scaled to count of values.

381

Figure 5. Statistically significant temporal trajectories at restored strata of (A) insect taxa

richness, and (B) percent of logs with plant growth. Lines represent linear relationships with

95% confidence intervals, size of data points are scaled to count of values.

386

387
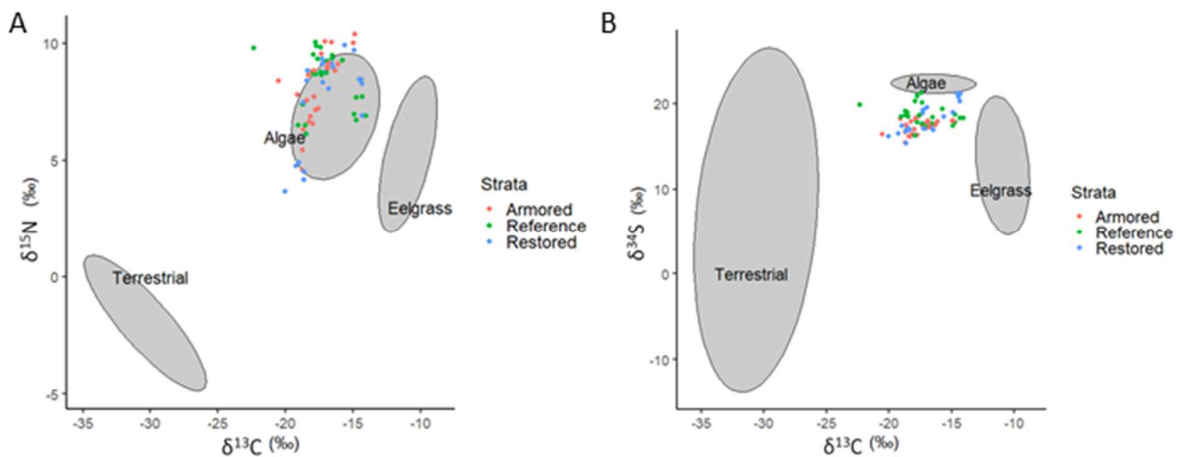
Figure 6. Talitrid amphipod δ^{15}N vs. δ^{13}C (A) and δ^{13}C vs. δ^{34}S (B) isotopic signatures by strata;

Armored = red diamonds, Reference = green diamonds, Restored = blue diamonds. Grey

ellipses represent the 95% CI of algae, eelgrass, and terrestrial OM source isotope values; raw

amphipod values are not corrected for trophic discrimination, but are corrected for inorganic

carbon removal per Schlacher and Connolly (2014).

393

4. Discussion

Our analysis of 27 measurements across eight metrics of physical and biological conditions found that values for living shoreline restored strata were often higher than armored strata but less than reference strata. Results show that restoring intertidal beach slopes increases the accumulation of beach wrack, a result also documented at eco-engineered sites in Australia (Strain et al., 2018). In new wrack lines, measurements of wrack depth, total percent cover, and algal percent cover were highest at restored strata. Algae were the dominant wrack source (primarily composed of green algae and *F. distichus*) likely because of adjacent source habitat, and our Salish Sea sites were more protected than coastal beaches where more kelp and seagrasses occur in the wrack (Reimer et al., 2018). Terrestrial percent cover in the new wrack was intermediate at restored strata – reference conditions had a higher input of leaves, sticks, and seeds, and both reference and restored strata had higher values than armored strata. Importantly, a line of older wrack accumulated in supratidal areas at all restored strata, and the composition of this old wrack was equal to that of reference and different from armored strata, primarily composed of terrestrial sources. Thus, removing armor can restore beach wrack zones, especially given limited maintenance regimes that can be common at urban beaches such as grooming and sediment filling, as recently evaluated in southern California (Schooler et al., 2019). There are important implications of these findings, as the beach wrack zone provides significant marine subsidies to intertidal and supratidal herbivore and decomposer communities, which research in the US and Canadian west coast has shown is patchy by nature (Orr et al., 2005), and can vary with beach morphology (Reimer et al., 2018).

415   Logs deposited as driftwood on beaches were most numerous at reference strata. Log

416   movement in the Salish Sea is greatest during winter months, when high river flows and storm

417   surges coupled with high waves and tides can move logs into the supratidal (Simenstad et al.,

418   2003). Since this process follows an annual cycle, it is reasonable that accumulation of

419   driftwood from external sources takes time to reach reference conditions, in addition to

420   gradual local input of fallen trees that take time to grow. Given that reference beaches also had

421   more overhanging vegetation that had a positive effect on insect taxa richness and

422   assemblages, the connection that these logs provide to the marine riparian zone warrants

423   further study. These marine riparian and driftwood zones may be important both to fish (e.g.,

424   providing shade and prey resources) and to birds (Dugan et al., 2003). Riparian and log

425   functions are better understood in freshwater and estuarine wetland systems (Simenstad et al.,

426   2003), although there are documented positive relationships such as better egg survival of

427   beach-spawning surf smelt on shaded shorelines (Rice, 2006). Despite some knowledge gaps,

428   amount of marine riparian vegetation is being used as an indicator for Chinook salmon recovery

429   (PSRITT, 2015). A more informed understanding would allow us to better protect currently

430   forested shorelines, and inform recommendations for vegetation plantings and log placement

431   at living shoreline sites, similar to what exists in freshwater systems. Since these natural

432   functions have implications for Chinook salmon recovery, they also apply to imperiled Southern

433   Resident orca whales, for which the primary recommended goal of recovery by a recent task

434   force is to increase Chinook salmon abundance (Cascadia, 2018).

435   Responses of arthropods to restored living shorelines were varied. Although there were

436   some positive responses, notably in diversity of wrack invertebrates and assemblages of insects

437 being equal at restored and reference strata, densities at restored strata were lower. There was

438 a non-significant trend of talitrid amphipods in the wrack at restored strata approaching

439 reference densities, a metric that may be worthy of future monitoring because talitrids are

440 important beach detritivores and prey for birds (Dugan et al., 2003), and occupy a more varied

441 niche than the oligochaete and nematode worms that were dominant at the low-diversity

442 armored beaches. Regarding insects, their assemblages are important food web components of

443 the nearshore. For example, insects can be important prey for juvenile Chinook salmon in the

444 Salish Sea (Duffy et al., 2010), and these insects have been shown to depend on the

445 supralittoral ecotone in British Columbia, Canada (Romanuk and Levings, 2003). Accounting for

446 insect response is especially important given that 40% of insect species worldwide are

447 threatened by extinction mainly driven by habitat loss (Sánchez-Bayo and Wyckhuys, 2019), and

448 more robust analyses and interpretation of trends in insect populations are needed (Thomas et

449 al., 2019).

450   In addition to habitation, stable isotope analysis showed that talitrid amphipods depend

451 on marine algae and eelgrass sources in the beach wrack for feeding. The finding that diet

452 contribution of marine algae to talitrids was lowest at armored strata emphasizes the

453 importance of restoring wrack lines when armoring truncates their deposition. Though there

454 was more terrestrial accumulation at reference beaches in the wrack, diet contribution of algae

455 remained the highest there. It is likely that terrestrial material needs more bacterial and fungal

456 conditioning before entry into talitrid amphipod food webs. Amphipods similarly show stronger

457 linkages to algal than seagrass trophic signatures in Australia (Poore and Gallagher, 2013).

458 However, they can be opportunistic in their feeding in wrack lines on the Portuguese coast

459   (Bessa et al., 2014), and studies from Canada and Spain highlight their need for habitat

460   heterogeneity (Pelletier et al., 2011; Ruiz-Delgado et al., 2015). Habitat heterogeneity may be a

461   useful target for restoration and conservation, for example by increasing and/or diversifying

462   algal subsidies to enhance invertebrate prey for threatened shorebirds, as exhibited by beach-

463   nesting plovers in Australia (Schlacher et al., 2017). Our data show that denser and thicker

464   wrack accumulation increases arthropod response, especially up to 50% wrack cover and 1cm

465   wrack depth, and these levels could be used as guidelines for management.

466   Even though the placement of armor on our control beaches encroached below MHHW

467   and impacted the metrics referenced above, this did not correspond with any significant

468   differences in overall beach width, slope, percent sand, or elevation of the wrack line. Thus, our

469   study sites represent situations where placement of armor covers the supratidal and parts of

470   the upper intertidal zone, but does not necessarily prevent the occurrence of beach habitat

471   features at lower intertidal elevations. This is important to consider, because the more armor

472   encroaches into the intertidal zone, the greater the impacts, and current management practices

473   are tailored to prevent this (Dethier et al., 2016b). Interestingly, the wrack line was sandier

474   when armor or natural bluff descended just below MHHW, a trend opposite to the typical view

475   that increased wave action coarsens sediments (Ruggiero, 2010). This can likely be explained by

476   the natural bluffs at our study locations eroding and depositing sand, and the armored strata at

477   our study locations were likely not placed low enough to dramatically affect wave action.

478   Our study builds on similar living shoreline work in context with armored and natural

479   beaches (Gittman et al., 2016a), and with case studies that have longer before-and-after

480   effectiveness monitoring (Lee et al., 2018). The space-for-time framework of our living

481   shoreline sites of different ages allowed us to assess if any beach functions increased through

482   time across spatial scales (Morgan and Short, 2002), and shows that there could be optimal

483   timelines for obtaining the best precision for these measurements. Two of the terrestrial-based

484   metrics – percent of logs with plant growth, and insect taxa richness – showed significant

485   increases four to seven years after living shoreline implementation. These metrics take time to

486   develop, as driftwood is often deposited on beaches during annual events of winter storms, and

487   an increase in different insect taxa likely depends on plant growth and other habitat features.

488   Although log placement is often used as a living shoreline technique, results such as ours could

489   suggest better site designs. For example, to encourage more rapid development of terrestrial

490   functions, logs with secondary vegetative growth already established could be used, along with

491   planting native vegetation of different sizes and structures that increase the complexity of the

492   canopy. Transplanting of a few established plants from nearby natural sites could also help

493   connect shoreline processes.

494   Before and after case studies in our study area show similar temporal increases in

495   measured metrics, from five to ten years after site restoration (Lee et al., 2018). A study of six

496   sites, ranging 0-8 years old in North Carolina found the effectiveness of living shorelines to be

497   evident for nekton after three years (Gittman et al., 2016a). Biotic responses in estuarine

498   restored wetlands show similar increases in some measurements three to five years after

499   restoration (Simenstad and Thom, 1996; Morgan and Short, 2002), although there is high

500   variability in type and timing, with some responses taking longer than 10 years (Zedler and

501   Calloway, 1999; Morgan and Short, 2002). Some hydrologic measurements from constructed

502   *Spartina alterniflora* marshes in North Carolina reached equivalency rapidly in 1-3 years, while

503   most ecological attributes took 5-15 years (Craft et al., 2003). Seagrasses in Florida showed a

504   greater response at seven versus three years after restoration (Bell et al., 2014). Oysters recruit

505   rapidly following oyster reef restoration in coastal Louisiana, but ecosystem services such as

506   fish response and shoreline stabilization take longer (La Peyre et al., 2014). Thus, there is no

507   single time period it takes for a site to reach functional effectiveness post-restoration. Although

508   it can be difficult to generalize across aquatic studies, many of which are relatively short-term

509   (Verdonschot et al., 2013), these comparisons among living shoreline and restoration actions

510   display some commonalities, that certain functions respond positively in short-term site

511   development, and some long-term trajectories exist, while others are variable and follow

512   uncertain trajectories. Globally, rates of recovery for coastal systems can be <5 years for high-

513   turnover biological components, and a minimum of 15-25 years for higher biotic functions

514   (Borja et al., 2010). Therefore, recovering ecosystems often have a recovery debt compared to

515   reference levels (Moreno-Mateos et al., 2017), emphasizing the need for ecosystem restoration

516   paired with protection efforts. Long-term datasets are rare, and benefit from input from

517   scientists, managers, and the broader community, so that trends in the data apply to multiple

518   objectives and conservation goals (Toft et al., 2017b).

519   With increased understanding of the growing impacts of shoreline armoring on the

520   environment, removing armor where feasible is an attractive option, and incorporating nature-

521   based alternatives to hard armor for coastal protection is increasing globally (Bilkovic et al.,

522   2017). As an example of living shoreline applications gaining momentum in the Salish Sea, since

523   our 2016 fieldwork data collection through 2020 we know of 19 new armor removal sites within

524   our study area in Washington State. The drivers for armor removal at these sites are a mix of

525 habitat restoration, reducing maintenance and repair costs, and changes in aesthetic or

526 environmental values of property owners. Our assessment of the effects of armor removal will

527 help to guide the expectations of restoration practitioners and property owners for future

528 projects.

529   In the face of increasing levels of coastal urban growth and sea level rise (Sutton-Grier et

530 al., 2015), there is great potential for living shorelines to both enhance shoreline health and

531 protect people using more natural approaches (Arkema et al., 2013; Toft et al., 2017a).

532 Examples of this include reducing wave heights (Narayan et al., 2016), and maintaining coastal

533 resilience in the face of hurricanes (Smith et al., 2018). To accomplish this, knowledge must be

534 accessible not only to restoration practitioners and scientists, but also to waterfront

535 landowners and government officials (Scyphers et al., 2015). This will account for diverse social

536 perspectives on shoreline systems, and aid in decision making on managing these systems. In

537 our study region, the Puget Sound Partnership prioritizes the use and evaluation of living

538 shorelines (PSP, 2018), nationally NOAA has guiding principles that encourage the use of living

539 shorelines (NOAA, 2015), and globally artificial ocean sprawl of infrastructure has stimulated

540 the development of new nature-based solutions to coastal protection (Firth et al., 2016). Given

541 these management concerns of coastal communities from local to global levels and across

542 multiple audiences, it is important to continually refine the implementation of living shorelines

543 in order to improve habitat for both people and nature.

544

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