SHORELINE ARMORING:
IMPACTS ON NEARSHORE HABITAT
IN THE MAURY ISLAND AQUATIC RESERVE

by

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A Thesis

Submitted in partial fulfillment

of the requirements for the degree

Master of Environmental Studies

The Evergreen State College

June 2018

This Thesis for the Master of Environmental Studies Degree

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ABSTRACT

Shoreline Armoring in the Puget Sound: Impacts on nearshore habitat in the Maury Island Aquatic Reserve

Kirsten Miller

Shoreline armoring is widespread in the Puget Sound, Washington, but the impacts on the biological features of nearshore ecosystems have only recently begun to be documented. Shoreline armoring disrupts the connection between marine and terrestrial ecosystems along the shoreline and can decrease the availability of prey resources for juvenile salmon. Most previous work has been conducted in highly urban areas, and this study aims to strengthen our understanding of residentially-developed, high-bank shorelines characteristic of the central Puget Sound. Here we determine differences in shoreline vegetation, terrestrial insect assemblages, wrack coverage and composition, and fish assemblages between armored and unarmored beaches. Citizen scientists with Vashon Nature Center's BeachNET program collected data in the summer of 2017 at three beaches following protocols from the Washington Sea Grant's Shoreline Monitoring Toolbox. Results from this study determine that natural beaches have more overstory vegetation, trees, and native plant species. Terrestrial insect abundance and taxa richness was similar at armored and natural beaches, but natural shorelines host a greater percent composition of Diptera, an important prey species for juvenile salmon. Forage fish spawning occurred at armored and natural shorelines, however, natural shorelines hosted a far greater number of sand lance eggs. Natural shorelines had higher abundance and taxa richness of fish. This study suggests shoreline armoring alters shoreline conditions and decreases the availability of key habitat and prey resources for key juvenile salmon species in residentially-developed shorelines of the Puget Sound.

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Acknowledgements

I would like to thank Bianca Perla and Maria Metler at the Vashon Nature Center for working with me for the duration of this thesis. Thank you for your time, encouragement, support, and resources to make this project happen. Thanks to Vashon Island BeachNET volunteers for data collection and your interest in healthy shorelines!

I would like to thank my advisor, Erin Martin, who helped in every part of this thesis – from initial project planning to providing final edits – spending time to encourage and provide feedback along the way.

Thanks to Ben Leonard for the assisting with snorkel surveys and statistics.

I would also like to thank my parents, Burt and Debbie Miller, for always lending me an ear and providing hot showers after long days of snorkeling.

Chapter 1: Introduction

About a third of the Puget Sound's shorelines are altered with some form of shoreline armoring (Puget Sound Partnership, 2012). Shoreline armoring is put into place to prevent erosion and stabilize shorelines to allow for commercial and residential development. Armoring can include seawalls, bulkheads, and revetments constructed of large rock, concrete, wood, or steel. Although these structures are important for development along the shorelines, there is an increasing understanding that shoreline armoring may cause adverse ecological impacts along shorelines.

Armoring has been found to alter shorelines, disrupting the connectivity between marine and terrestrial ecosystems. Armoring is known to reduce shoreline vegetation, decrease terrestrial insect abundance and diversity, decrease wrack composition, and reduce egg survival rates for forage fish. Armoring can also alter diet and feeding behavior of juvenile salmon in the nearshore, as they rely on shallow, productive nearshore habitats for foraging and refuge from predators during their outmigration from natal streams to the sea (Heerhartz & Toft, 2015).

Armor removal and beach restoration is a priority in the Puget Sound region in Washington State, driven by the need to protect Pacific salmon species such as endangered populations of Chinook salmon (*Oncorhynchus tshawytscha*), an important cultural, ecological, and economic resource (Toft et al., 2014). In addition, the Puget Sound Partnership, a state agency leading the region's collective effort in Puget Sound recovery, has set targets for a net decrease of armored shorelines, or more armoring removed or restored than developed, by 2020.

Although the pace of shoreline armoring development has slowed, armoring in the Puget Sound is still increasing, as coastal habitats in the Puget Sound face unprecedented urban growth (Gittman et al., 2015). As shoreline development infringes on Puget Sound, potentially increasing the need for armoring along beaches, understanding the impacts of shoreline armoring on terrestrial and aquatic environments along the shoreline is vital for effective management.

There has been a recent momentum in the Puget Sound region to restore armored shorelines through removal of armoring structures, addition of sediments, re-planting native riparian vegetation, and addition of logs and woody debris (Toft et al., 2013, Lee et al., 2018). However, there is still little scientific information available to assess impacts of shoreline armoring and beach restoration benefits (Sobosinski, 2003). There is especially a need for highly localized studies to characterize coastal biota response to armoring across the highly diverse Puget Sound region (Lee et al., 2018).

This thesis provides a highly localized study of the effects of shoreline armoring on Vashon and Maury Islands located in the Maury Island Aquatic Reserve (MIAR), establishing a basic understanding of the physical and biological differences between armored and natural shorelines. This thesis is the first part in a longer study conducted by the Vashon Nature Center (VNC) that will assess shoreline armoring removal in the MIAR. The results presented in this thesis establish a baseline of shoreline conditions before restoration, which will occur in the summer of 2018. Post-monitoring will occur after armoring removal in 2018. The results from this study will be used as a comparison to post-restoration conditions.

This thesis specifically seeks to answer: to what extent are shoreline vegetation coverage, terrestrial insect assemblages, and wrack accumulation different between armored and natural sites in nearshore habitats in the MIAR, Puget Sound? Do these differences affect fish use of these nearshore habitats? Understanding the impacts of shoreline armoring and whether restoration has the intended benefits is essential for understanding biological recovery of shorelines and encouraging proper shoreline management (Lee et al., 2018). Understanding benefits of shoreline restoration may encourage shoreline restoration, favor alternative stabilization techniques, and reduce future shoreline armoring development in the Puget Sound region.

Background of this study

The goal of this study is to gain a solid understanding of shoreline conditions in the MIAR and how shoreline armoring impacts the local nearshore ecosystem. King County purchased three properties on Maury Island (Big Beach, Lost Lake, and Piner Point) to remove shoreline armoring and restore natural nearshore processes. An important restoration goal and project funding for King County is to improve habitat for out-migrating juvenile salmon. Shoreline armoring alters the key nearshore habitat features that juvenile salmon depend on in their early life histories. All structures and bulkheads will be removed from each restoration site and natural shoreline and hillslope processes will be restored to the maximum extent practical (Booth & Legg, 2017).

Beaches at the study sites are unique as they are steep, highly erosive shorelines. Highly localized studies are necessary to determine the impacts of armoring and how restoration benefits the nearshore along beaches in the MIAR. Bulkhead removal may allow for the return of natural erosional processes which increases habitat benefits, including increased sediment delivered to the nearshore which helps create shallow water habitat important for juvenile salmon survival (Booth & Legg, 2017).

Researchers from the VNC lead a group of citizen scientist volunteers to monitor Big Beach, Lost Lake, and Piner Point in the MIAR. Three shoreline types were monitored at each beach including a natural shoreline, an armored shoreline, and an armored shoreline where armoring will be removed in the summer of 2018. For this thesis, armoring was not removed at the “pre-restoration” site, but data was collected separately to characterize the habitat prior to the 2018 armoring removal process. This study implemented standardized monitoring protocols from the Puget Sound Partnership’s Shoreline Monitoring Toolbox (Shoreline Monitoring Toolbox, 2017). This study can be used as a model for groups to coordinate systematic studies along beaches with citizen science volunteers.

The data collected focuses on biotic parameters that serve as a metric for healthy shoreline habitat, including: shoreline vegetation, terrestrial insect assemblages, wrack coverage and composition, forage fish spawning, and fish observations from snorkel surveys. Results will be presented that demonstrate armoring reduces marine riparian vegetation, alters the composition of terrestrial invertebrates, wrack composition, and forage fish spawning, and decreases abundance and taxa richness of fish. The results also establish a pre-restoration baseline of habitat conditions at beaches targeted for restoration in the summer of 2018.

Roadmap of thesis

I will first summarize what is currently known about the effects of shoreline armoring on physical and biological shoreline conditions through a literature review including: marine riparian vegetation, terrestrial invertebrates, wrack accumulation, forage fish spawning, and fish use. The studies presented in my review focus on research assessing differences between developed and natural sites along Puget Sound shorelines. Next, methodology and statistical analysis are presented, highlighting the standardized Shoreline Monitoring Toolbox protocols. Results will be presented that show armoring reduces marine riparian vegetation, alters the composition of terrestrial invertebrates, wrack composition, and forage fish spawning, and decreases abundance and taxa richness of fish. These results will be placed into context with the current literature in the Discussion section, in addition to discussing methodology and recommendations for future VNC beach monitoring surveys. I conclude by summarizing the results and highlighting the importance of continuing long-term monitoring studies to assess the biological response to armoring and restoration and potentially encourage armoring reduction and shoreline restoration in the future.

Chapter 2: Literature Review

Worldwide, shorelines adjacent to bodies of fresh and salt waters face faster urbanization and population growth than other geographic regions (Neumann et al., 2015). Coastal regions are known to experience high immigration rates because of their ease of access to domestic and international shipping, military and defense uses, tourism, access to recreational activities, and employment opportunities (Bulleri et al., 2005; Gittman et al., 2015; Neumann et al., 2015). Coastal infrastructure and urban centers are exposed to various coastal hazards in these areas, such as storms, large waves, flooding, sea level rise, and erosion. In response, many coastal communities have established hardened structures including bulkheads, jetties, riprap revetments and seawalls, a practice commonly called “shoreline armoring” (Chapman & Underwood, 2001; Heerhartz et al., 2014; Gittman et al., 2015). In some large urban centers, such as San Diego Bay, Chesapeake Bay, Sydney Harbor, and Hong Kong’s Victoria Harbor, over 50% of the shorelines are armored (Gittman et al., 2015). In the United States alone, about 14% of the lower 48 states’ shorelines are armored, and 64% of these armored shorelines are adjacent estuaries and coastal rivers (Gittman et al., 2015). As coastal immigration and urban centers experience increased growth and development, the rate of shoreline armoring is expected to rise (Davis et al., 2002; Dugan et al., 2008; Lam et al., 2009).

Armored shorelines are associated with lower biodiversity, vegetation cover, and abundances of invertebrates and fish (Moreira et al., 2006; Dugan et al., 2008; Morley et al., 2012). Armored shorelines can increase beach erosion, as waves reflect off of armored shorelines (Heatherington & Bishop, 2012). Armoring can reduce the overall ecological health of coastal ecosystems by degrading shallow intertidal habitats that are vital to the survival of juvenile fish and aquatic invertebrates (Bilkovic & Roggero, 2008;

Seitz et al., 2006; Gittman et al., 2016). Armored shorelines disrupt the transition between aquatic and terrestrial habitats and decreases deposition of woody debris and "wrack", or organic matter deposited on shorelines (Heerhartz et al., 2014; Lee et al., 2018). This loss of organic debris affects the aquatic-terrestrial food web including fishes and macroinvertebrates associated with wrack and vegetated habitats (Bozek & Burdick, 2005; Dugan et al., 2008; Heerhartz et al., 2014; Heerhartz & Toft, 2015; Dethier et al., 2016).

In Puget Sound, Washington, the shoreline is highly valued, as it serves as a platform for recreational boating and shipping, commercial growth, and urban and suburban development (Sobosinski, 2003). Currently, about a third of Puget Sound shorelines are altered by some form of shoreline armoring (Puget Sound Partnership, 2012). Shoreline armoring in the Puget Sound is increasing. More permitted armor was gained than lost cumulatively since 2011, resulting in a net cumulative length of 0.8 miles of new armor between the years of 2011 and 2016 (Puget Sound Partnership, 2012).

Recently, there has been momentum to restore armored shorelines through removal or armoring structures, nourishment of sediments, replanting native riparian vegetation, and addition of logs and woody debris (Toft et al., 2013; Toft et al., 2014). Restoration efforts are driven by the need to protect Pacific salmon (*Oncorhynchus tshawytscha*) that are of cultural, ecological, and economic importance to the region (Rhodes et al., 2006; Munsch et al., 2016). Shallow intertidal areas are known to serve as nursery habitats for juvenile salmon, providing food and refuge from predators (Toft et al., 2016). As shoreline development infringes on Puget Sound beaches, understanding the localized impacts of armoring on terrestrial and marine environments across the Puget Sound in the is necessary for effective management and to encourage shoreline restoration and reduce the overall armoring in the Puget Sound region.

The purpose of this literature review is to summarize what is currently known about the impacts of shoreline armoring on biological shoreline conditions resulting from shoreline armoring and associated habitat alteration. Guided by regional Puget Sound recovery goals, this review focuses on the effects of shoreline armoring on shoreline habitat health, with applications for juvenile salmon. This review is organized in sections corresponding to each biotic measure included in this study: vegetation, terrestrial invertebrates, wrack accumulation, forage fish spawning, and fish use along the shoreline. These biotic parameters serve as a metric for healthy shoreline habitat, specifically focusing on features vital for juvenile salmon habitat. Throughout the paper, I will specifically highlight a case study of shoreline restoration project at the Olympic Sculpture Park, in Seattle, Washington, that looked at similar shoreline parameters before and after the removal of shoreline armoring. The restoration improved the biological function of the nearshore in a highly urbanized shoreline, mimicking natural beaches. The Olympic Sculpture Park case study demonstrates how effective management and restoration can increase natural shoreline function and increase vital habitat for juvenile salmon in Puget Sound (Toft et al., 2013).

Shoreline armoring alters physical beach processes

Shoreline armoring, including seawalls, bulkheads, and revetments, is put in place to protect shorelines from naturally eroding beaches and stabilize areas for upland commercial and residential uses (Shipman, 2010). Shoreline armoring is known to alter physical nearshore processes. The nearshore, for the purposes of this document, is defined as the physical area that extends from the far edge of the photic zone to the adjacent uplands, including the top of any associated bluffs (Guttman, 2009). Shoreline armoring replaces natural beaches with hard, vertical surfaces, acting as a physical barrier between terrestrial and aquatic ecosystems that were once connected (Sobosinski, 2003; Ecology, 2016). Physical structures cut off, or "lock up", the natural delivery of sand and gravel to the shoreline from both marine and terrestrial sources (Ecology, 2016; Dethier et al., 2016). When waves reflect off these structures, they scour away sediments which are not replaced (Shipman, 2010). This causes beaches in front of armored sites to erode slowly, leading to gradual lowering or even the disappearance of the beach (Ecology, 2016). This process is referred to as the "truncation" of the beach (Johannessen & MacLennan, 2007), where armored beaches tend to be narrower, steeper, and coarser-grained (Nordstrom, 2014).

The "coastal squeeze" is a term used to describe coastal habitat loss due rising sea levels along armored shorelines. Shoreline armoring creates a static, artificial margin between land and sea. As sea levels rise and increased storms push the coastal habitats landward, shoreline armoring prevents the upper beach from migrating inland. Beach habitats become "squeezed" into a narrowed zone (Doody, 2013; Dethier et al., 2016). The narrowing of the beach can eliminate shallow water habitat directly adjacent to shore, which is vital habitat for fish (Munsch et al., 2016). Armoring can alter physical beach conditions locally and have broader, cumulative impacts across the Puget Sound on time scales of immediate to years or decades, depending on the location in Puget Sound (Dethier et al., 2016).

Shoreline armoring alters biological function of the nearshore

Shoreline armoring is known to change the overall biological function of the nearshore ecosystem. The nearshore is highly productive. It serves as habitat for a diversity of organisms as well as a refuge and rearing ground for numerous fish species (Sobosinski, 2003). Physical and chemical processes (e.g., wind and wave energy, sediment grain size, salinity, tide height) drive biological structure and function in the intertidal zone. The physical disruption of nearshore ecosystems due to shoreline armoring can lead to altered biological response.

Shoreline armoring disrupts marine-terrestrial connectivity, alters habitat for mid-level consumers, and ultimately affects prey availability for juvenile salmon (Heerhartz et al., 2015; Toft et al., 2013). The ecology of the intertidal zone is driven by a connection between terrestrial and marine processes. Terrestrial ecosystems provide terrestrial leaf litter input, deposition of large wood, and export of organisms to the beach (Sobosinski, 2010). “Reciprocal subsidization” occurs between the terrestrial-aquatic ecosystems, where terrestrial plant matter and insects fall into the sea, and marine wrack and invertebrates are deposited onto the land (Heerhartz et al., 2015). Wrack, or the amount of seaweeds, seagrasses, and terrestrial plant debris that washes up on shore, provides nutrients and habitat for terrestrial invertebrates (Heerhartz et al., 2015). The marine-terrestrial connection plays an essential role for mid-level consumers which are important for the diets and early growth rates of juvenile salmon (Rice, 2007).

Shoreline armoring management in Washington State

Understanding the ecological effects of shoreline armoring is important for guiding policy toward reducing overall armoring and promoting the use of alternative stabilization techniques that function similarly to natural shorelines. The Puget Sound region currently has many marine species listed as threatened or endangered, caused in part by heavy development in the Puget Sound region. Widespread management efforts around the Puget Sound are focusing on restoring Puget Sound's health, with a focus on threatened and endangered species (Guttman, 2009). The listing of Puget Sound salmonids under the Endangered Species Act has prompted increased attention by managers and policy makers to focus on the impacts of shoreline armoring on natural processes that shape Puget Sound. Particular interest has been paid to the functions of beaches, especially the role of beaches in supporting organisms that occupy important niches in the food web, such as invertebrates and forage fish (Guttman, 2009).

Several planning and policy documents are designed to protect and restore Puget Sound, such as Shoreline Master Plan updates and Critical Areas Ordinances. These documents cite protecting and restoring nearshore habitat functions as an important goal in overall Puget Sound restoration efforts. Shoreline Master Programs (SMPs) guide shoreline development in local municipalities, including guidelines for shoreline armoring. SMPs are guided by state laws but tailored to the specific geographic, economic, and environmental needs of each community (Ecology, 2016). Local SMPs are currently being updated to include in-depth guidance on how to implement alternatives to bulkheads. Alternatives to bulkheading can include "soft" armoring techniques, including a variety of stabilization methods that mimic site-specific shoreline processes (Van Zwalenburg, 2016).

In addition, to build, modify, or remove armoring structures, residential and commercial contractors must obtain a Hydraulic Permit Approval through the Department of Fish and Wildlife (WDFW, 2016). This permit requires “no net loss” of ecological functions, and requires the permittee to address mitigative measures to reduce adverse impacts of the project (WDFW, 2016). Potential mitigation projects can include enhancing backshore vegetation, addition of large woody debris, and beach nourishment (Johannessen et al., 2014). Management guidelines aim to protect nearshore function, but the impacts of armor and mitigation techniques are site-specific. Providing local restoration of degraded processes, habitat, and ecological function helps maintain health of nearshore ecosystems. Thoroughly understanding local nearshore ecosystem processes and impacts of armoring can help maximize mitigation opportunities to provide the greatest benefits to nearshore systems (Johannessen et al., 2014).

The elevation at which armoring is placed on the beach can influence the scale of physical impact on nearshore ecosystems (Dethier et al., 2016). Lower elevations of shoreline armoring, or relative encroachment on the beach, have greater impacts to biological conditions on local and larger spatial scales (Dethier et al., 2016). Armoring at low elevations (threshold is approximately 1-2 vertical feet below Mean Higher High Water (MHHW)) is no longer authorized for shoreline development. However, armoring at lower elevations than 1-2 feet below MHHW is still present in the Puget Sound. Future shoreline restoration projects may have greater benefits if projects target shoreline armoring below this elevation threshold.

Although there has been an increased emphasis to address the impacts of shoreline armoring, more localized studies are needed to protect and understand critical habitat in Puget Sound. Continuing research on both localized and Puget Sound-wide scales will elucidate how shoreline armoring influences both physical and biological effects of nearshore ecosystems.

Effects of shoreline armoring on key nearshore habitat features

Shoreline armoring is known to change the overall biological function of the nearshore ecosystem. The following sections will highlight literature documenting the effects of shoreline armoring on key habitat features including: marine riparian vegetation, beach wrack, beach wrack associated species, and invertebrate abundances and composition. In addition, I will look at the effects of shoreline armoring on forage fish beach spawning. I will then demonstrate how changes in the nearshore due to armoring physically affects salmon, including their diets and feeding behavior.

Marine riparian vegetation

Shoreline armoring decreases backshore marine riparian vegetation

Riparian vegetation along marine shorelines serves a variety of critical ecological functions (Brennan, 2007). Coastal trees and other vegetation on backshore areas, banks, and bluffs help stabilize the soil, control pollution entering marine waters, provide fish and wildlife habitat. Riparian areas are transitional, providing connections between and affecting both adjacent aquatic and terrestrial systems. Marine riparian vegetation communities influence the health and integrity of marine habitats and species and are an integral part of nearshore ecosystems (Brennan, 2007).

Extensive shoreline development in the Puget Sound region has caused marine riparian habitat loss. Vegetation characterization is highly variable across the Puget Sound terrestrial shoreline, however, trees, shrubs, and other ground cover is more common at natural sites absent of shoreline armoring. The removal of vegetation is characteristic of armored shorelines in the Puget Sound. In the south-central Puget Sound, for example, trees make up 80% of the percent cover of marine riparian vegetation at natural beaches but only 46% percent cover of armored areas, where grass is more common (Heerhartz et al., 2014). In the central Puget Sound, natural beaches have over ten times more overhanging vegetation compared to armored beaches (Heerhartz et al., 2014). Gardens and lawn are more characteristic of armored sites.

Vegetation along shorelines can serve as a metric for habitat quality and a determinant of available prey resources for juvenile salmon. Over and understory vegetation is found to host a variety of insect species (Romanuk and Levings, 2003). Vegetation along the shoreline is vital habitat for terrestrial invertebrates commonly found in juvenile salmon diets. The lack of shoreline vegetation, including overhanging trees and shrubs, can affect the abundance and species of invertebrates found along the shoreline.

Vegetation along the shoreline provides habitat for terrestrial insects, such as Dipterans (flies), an important dietary component of juvenile Chinook salmon (Munsch et al., 2016). Terrestrial insects can be carried by wind onto the water surface and provide food for juvenile salmon in shallow nearshore waters. In addition, leaves, insects, and other material from overhanging terrestrial plants fall onto the backshore, forming the basis for multiple terrestrial and aquatic food webs (Guttman, 2009). Armoring can disrupt this pathway by the associated removal of backshore vegetation and causing a physical barrier between terrestrial and marine ecosystems (Toft et al., 2007). The reduction in marine riparian vegetation caused by shoreline armoring decreases available habitat for invertebrates, and in turn, may have cascading effects on juvenile salmon diets (Duffy et al., 2010).

Recognition of vegetation as a key function of the nearshore ecosystem is essential for effective shoreline management. Protecting and restoring backshore vegetation should be considered an important goal in overall Puget Sound restoration efforts. Localized studies characterizing naturally occurring vegetation compared to armored shorelines may increase the understanding of shoreline conditions and assist in decision making to preserve natural function in these areas.

Olympic Sculpture Park case study and marine riparian restoration

In 2007, the City of Seattle funded the restoration of an armored beach located at the shoreline of the Olympic Sculpture Park on Elliott Bay in the Puget Sound. This project is of great interest in the Puget Sound region as an example of habitat enhancement along urban shorelines (Toft et al., 2013). This project involves an extensive monitoring plan that is meant to inform future restoration projects in the Puget Sound.

The overall goal of this project is to support higher numbers of salmon populations and increase diversity of invertebrate assemblages as prey species for juvenile salmon. To emulate a natural shallow water habitat, a portion of a seawall was removed and replaced by a "pocket beach," and a habitat bench was placed in front of an existing seawall (Toft et al., 2013). The pocket beach replaced riprap armoring, and the habitat bench was added as shallow, low-gradient habitat at the base of an adjacent seawall. Estuarine vegetation, comprised of native plants, was planted above the pocket beach in the uplands. Biological monitoring was conducted before, during, and after this enhancement project (pre-enhancement, year 1, and year 3, respectively).

Riparian vegetation was planted in the adjacent supratidal uplands, with a focus on native species that are common in the Puget Sound coastal zone such as shore pine (*Pinus contorta*), alder (*Alnus rubra*), willows (*Salix* spp.), beach strawberry (*Fragaria chiloensis*), and dune grass (*Leymus mollis*). Restoring vegetation along the nearshore can increase habitat complexity and marine-terrestrial connectivity. In response to the increased plantings along the shoreline, some types of terrestrial insects increased in abundance and tax richness, which is further discussed in a subsequent section on terrestrial invertebrates. Other studies have shown insects to be significantly reduced on armored shorelines where vegetation was removed as well (Romanuk & Levings, 2003; Sobocinski et al., 2010). Continued development of vegetation communities along shorelines may increase the input of insects and feeding opportunities for juvenile salmon (Toft et al., 2013).

Beach wrack

Shoreline armoring and marine-terrestrial connectedness

Wrack, or seaweeds, seagrasses, and terrestrial plant debris deposited on the beach by an ebbing tide, is habitat for much of the supratidal fauna, and it serves as a basis for the nearshore detritus-based food web (Heerhartz et al., 2015). Beach wrack can be comprised of marine (e.g., Ulva spp. and Zostera spp.) or terrestrial (e.g., leaf litter and wood) sources (Sobosinski, 2003). Wrack functions as a microhabitat by providing shelter, food, and moisture necessary for many intertidal invertebrates, especially amphipods, isopods, and insects (Jedrzejczak, 2002a). These organisms are important in the biogeochemical cycling of marine material (Jedrzejczak, 2002b), and are prominent consumers in the detritus-based food web (Sobosinski, 2003). The presence of a wrack is especially important for habitat for invertebrates in areas of low primary productivity, such as sandy or gravel beaches in Puget Sound (Heerhartz et al., 2016).

The physical disturbance caused by shoreline armoring can reduce the abundance and composition of wrack that accumulates on Puget Sound shorelines (Heerhartz et al., 2016). Changes in physical processes due to armoring causes the loss of high shore space, and therefore the amount of wrack that can accumulate on a beach (Heerhartz et al., 2015). Overhanging vegetation deposits terrestrial material to the beach, including leaf litter, sticks, and logs. Local, backshore vegetation is the primary source of terrestrial detritus in wrack (Heerhartz et al., 2014). Terrestrial detritus in wrack, along with marine algae, provide food and shelter to diverse communities of invertebrates. With reduced terrestrial organic debris, armored shorelines lack the resource base to support a leaf-litter invertebrate community (Heerhartz et al., 2014). Changes in composition and lack of wrack accumulation lowers taxa richness of insects and benthic macroinvertebrates associated with wrack, altering prey availability for foraging juvenile salmon (Sobocinski et al., 2010).

Shoreline armoring reduces beach wrack subsidies

Shoreline armoring can reduce the amount of high shore space on beaches, and in turn, the amount of wrack and logs that can accumulate on a beach. Reduced wrack results in significantly different, less taxa-rich and less abundant invertebrate communities (Heerhartz et al., 2015). Heerhartz et al. (2014) investigated the amount and composition of wrack and log accumulation across paired armored-unarmored beaches throughout central and south Puget Sound. They looked at the physical factors that accounted for these differences, such as beach width, elevation, slope, armoring type, and uplands vegetation. The width of the armored beaches was significantly reduced by an average of 8.9 meters, and the elevation of the beach toe was lowered by an average of 0.9 meters (Heerhartz et al., 2014).

They found there was a significant difference between natural and armored beaches, where there was 66% more total wrack cover in the spring and 76% more in the fall at natural beaches when compared to armored shorelines. The seasonal variations can be due to an increase in terrestrial inputs in the fall, such as leaves and sticks, from upland vegetation. Variations can also be due to the increase of marine algae in the summer that is susceptible to dislodgement during fall storms (Heerhartz et al., 2014).

Accumulated logs, which provide habitat for many organisms, were almost completely absent from armored beaches (Heerhartz et al., 2014). By covering the upper shore with an armoring structure, the space where logs and wrack would normally accumulate is eliminated (Heerhartz et al., 2014). This has consequences for not only for primary consumers that depend on the wrack and logs for shelter and food, but also secondary consumers that are subsidized by resources from these adjacent ecosystems (Heerhartz et al., 2015).

Shoreline armoring alters wrack composition

Shoreline armoring alters wrack composition, or type of debris deposited on the beach. Heerhartz et al. (2014), when examining the distribution of wrack on shorelines, demonstrated that there was a larger proportion of terrestrial material as compared to seagrass and algal material at natural beaches. The terrestrial component was three to seven times as abundant on unarmored beaches compared to armored beaches, depending on the season and beach location (Figure 1). In comparison, the algal proportions were much higher at armored beaches, where there was on average 74 percent algae at armored beaches versus 56 percent on natural beaches, demonstrating that there were less terrestrial inputs on armored beaches (Heerhartz et al., 2014). The clear reduction in the proportion of terrestrial material in the wrack at armored beaches demonstrates how shoreline armoring decreases marine-terrestrial connectivity (Heerhartz et al., 2014). Different types of prey species that associate with either marine or terrestrial wrack inputs can be affected by the altered composition and abundance of wrack accumulation at armored sites.

Aquatic and terrestrial invertebrates

Shoreline armoring results in less diverse communities of invertebrates available for juvenile salmon. Juvenile salmon (Juvenile Chinook and chum salmon) have eclectic diets, and may benefit from prey from diverse habitats (e.g. terrestrial vegetation, algae, soft-sediment substrates) (Brennan et al., 2004, Toft et al., 2007; Duffy et al., 2010). Chinook salmon diet analyses from Puget Sound marine beaches showed a high proportion of amphipods and insects, specifically Diptera, Homoptera, and Psocoptera, demonstrating the importance of prey from both marine and terrestrial habitats (Munsch et al., 2016). Shoreline armoring disrupts the connection between terrestrial and aquatic ecosystems, a vital function for abundant and diverse invertebrate assemblages.

Shoreline armoring constrains wrack-associated invertebrate communities

Shoreline alterations result in less diverse communities of invertebrates available for juvenile salmon. Wrack provides food and shelter for diverse communities of invertebrates, such as talitrid amphipods, isopods, and insects. Wrack invertebrates can be affected by changes in the physiological requirements of the organisms due to armoring. For example, armoring changes the sediment moisture and temperature due to the alterations in wrack cover and composition (Heerhartz et al., 2015). The effects of altered wrack cover may thus cascade, via altered food webs, to organisms such as fish (Heerhartz et al., 2015).

Heerhartz et al. (2016) measured the abundance and composition of macroinvertebrates associated with beach wrack. On average, there were twice as many terrestrial insects per sample at natural beaches than armored beaches. Insects were captured at upper shore areas at natural beaches. They also took core samples of wrack to determine what invertebrates were present. The invertebrate assemblages were found to be related to the amount and type of wrack found on beaches. Natural beaches had significantly more talitrid amphipods (sandhoppers), more insects, and fewer aquatic invertebrates in wrack samples. The talitrid genus, positively correlated with the proportion of terrestrial wrack, was on average 8.5 times more abundant on natural beaches than armored (Heerhartz et al., 2016). This result adds strong evidence for the significant reduction of terrestrial inputs due to shoreline armoring, and how this reduction can dramatically change invertebrate abundances.

Dethier et al. (2016) looked at broader scale cumulative impacts on invertebrates in wrack across the entire Puget Sound including 65 pairs or armored and unarmored beaches across north, central, and south Puget Sound regions. It is difficult to demonstrate differences attributed to armoring at this scale due to high natural variability across the Puget Sound. Beach width, riparian vegetation, numbers of accumulated logs, and amounts and type of beach wrack and associated invertebrates were consistently lower at armored beaches (Dethier et al., 2016). However, some of results were not consistent with the localized findings of Heerhartz et al. (2016). Armored beaches reduced numbers of amphipods and insects only in the central and south regions of the Puget Sound. When north beaches were included in the analysis, there were no significant differences in amphipods and insects between armored and natural sites. The exception was the talitrid amphipod genus, Megalorchestia, that showed a consistent sensitivity to armoring and where abundances were significantly reduced due to armoring encroachment on the beach.

The lack of significant difference between wrack-associated invertebrates when all beaches were compared across Puget Sound is most likely due to regional differences of shorelines. Impacts to invertebrate abundances in wrack were lower in the northern regions on Puget Sound where less armoring is present (Dethier et al., 2016). Northern shorelines also tend to have more habitat space for invertebrates. Northern shorelines contain overall more wrack due to higher algal populations and mass. In addition, there is more space for wrack accumulation, as there is lesser encroachment of armoring on the beach (Dethier et al., 2016).

Shoreline armoring alters epibenthic invertebrate communities - Olympic Sculpture Park case study

The loss of fine sediment due to shoreline armoring reduces the abundance of epibenthic invertebrates in shallow water ecosystems (Sobosinski et al., 2010; Toft et al., 2013), which can reduce epibenthic prey consumption by fish (Morley et al., 2012). Toft et al. (2013) compared epibenthic invertebrates living at the water-sediment interface between the pocket beach, habitat bench, and an adjacent armored site. The assemblages of epibenthic invertebrates became more diverse at the restored sites. Before the site enhancements, over 93% of amphipod composition consisted of one species (*Paracalliopiella pratti*), however, after the enhancement, *P. pratti* was less dominant (Toft et al. 2013). Due to the more complex habitat structure created by these enhancement projects, epibenthic assemblages became more diverse which increases the diversity of prey available for juvenile salmon (Toft et al. 2013).

Shoreline armoring alters terrestrial invertebrate communities - Olympic Sculpture Park case study

Toft et al. (2013) looked at differences of terrestrial insects in shoreline vegetation after shoreline restoration. Taxa richness and abundances of terrestrial insects generally increased post-enhancement as a result of shoreline plantings and showed higher numbers of Acari (mites), Collembola (springtails), and aphids which are important prey species for juvenile salmonids (Toft et al., 2013). Abundances of certain terrestrial insects associated with marine riparian vegetation can increase with restoration, or planting of native species, along the shoreline.

Terrestrial invertebrates can be used as a metric for habitat quality and as a determinant of available prey resources for salmon (Toft et al., 2013). Invertebrate taxa richness has been found to be greater at sites with intact shoreline vegetation than at armored sites without (Sobosinski, 2003). Invertebrates may respond to more complex habitats, as habitat complexity is known to enhance diversity (Chapman, 2003; Morley et al., 2012).

The enhanced shorelines did not always show definitive improvements over armored shorelines. For example, some salmon prey items, such as chironomids, a type of small fly, were abundant at both armored and enhanced sites (Toft et al., 2013). This could be due to the highly urban and industrialized location of this restoration project as well as lack of replication on a broader scale (Toft et al., 2013). However, the restorative plantings along the shoreline did create more complex habitats and increased the marine-terrestrial connectivity, both of which can increase inputs of terrestrial invertebrates.

Forage fish

Shoreline armoring and forage fish: an important prey species for salmon

The effects of shoreline armoring may have adverse consequences for forage fish that spawn in the intertidal zone. Forage fish, such as surf smelt, sand lance, are an important food source for juvenile and adult Pacific salmon (Rice, 2006). Surf smelt (*Hypomesus pretiosus*) and sand lance (*Ammodytes hexapterus*), utilize areas in the high intertidal zone for spawning where they deposit their eggs in gravel-sand beaches in the upper intertidal zone in the Puget Sound. Shoreline armoring may have adverse effects on egg survival rates due to the reduction of gravel-sand beach habitat and the associated vegetation loss, causing more exposure to environmental conditions such as increase sun exposure. Terrestrial vegetation provides shade and increases debris (ex. wrack and logs) in the upper intertidal zone that protects incubating embryos by providing increased shade, moisture and protection from sunlight. Removal of terrestrial vegetation that is associated with shoreline armoring can expose eggs to brighter and hotter conditions, which are less suitable environments for embryo survival (Rice, 2006).

To look at the influences that shoreline armoring has on forage fish spawning habitat, Rice (2006) compared the proportion of Surf smelt eggs containing live embryos at modified and unaltered beaches in Puget Sound, monitoring the light intensity, substrate and air temperature, and humidity at each shoreline type. The most noteworthy temperature differences were the substrate temperatures between sites. Natural beaches, associated with terrestrial vegetation and debris inputs, had a mean temperature of 14.1 degrees Celsius, whereas modified beaches had a mean daily temperature of 18.8 degrees Celsius. There was a striking difference between the proportion of smelt eggs containing live embryos between armored and natural beaches. On altered beaches, approximately half were live as compared to natural beaches (Rice, 2006). Removal of terrestrial vegetation that is associated with shoreline armoring can expose eggs to brighter and hotter conditions, which are less suitable environments for embryo survival. Although this study does document significant differences in environmental conditions between modified and natural beaches and suggests these differences affect surf smelt embryos, more detailed information on the specific environmental tolerances of smelt embryos are needed to fully understand the effects of shoreline armoring on surf smelt embryo survival (Rice, 2006).

Forage Fish - Olympic Sculpture Park Case Study

The Olympic Sculpture Park enhancement project increased the abundance of forage fish that utilized the shoreline at the habitat bench and shallow pocket beach (Toft et al., 2013). The small, pelagic schooling fish may have sought refuge from deeper waters to avoid predation that is more common in deeper waters or the use of beach sediments for spawning. Shoreline engineering, such as beach nourishment and alternative stabilization techniques, may be important for the creation of spawning habitat and egg survival (Rice, 2006). The enhancement of gravel-sand beaches in this case study proved important in preserving forage fish spawning habitat and can be used as an example for future restoration projects.

Fish assemblages

Juvenile salmon in Puget Sound

Puget Sound provides critical rearing habitat for juvenile salmon on their out-migration toward sea. In Puget Sound, several species of Pacific salmon (*Oncorhynchus spp.*) rear in nearshore marine areas, including Chinook salmon (*Oncorhynchus tshawytscha*) which are of particular concern as they are listed on the federal endangered species list. Pacific salmon are anadromous species that enter the estuarine or marine environments as juveniles and have a strong tendency to stay in shallow waters, which they use for feeding, refuge from predators, and salinity acclimation (Simenstad et al., 1982). The Puget Sound nearshore is known to be critical for early growth rates of juvenile salmon as it provides diverse sets of pelagic, benthic, and terrestrial prey resources (Simenstad et al., 1982). Abundance and quality of prey affect early marine growth which is critical survival later in their marine life (Duffy et al., 2003).

Shoreline armoring can change the structure of nearshore habitats, reducing the amount of shallow water habitat available for juvenile salmon (Munsch et al., 2016). The loss of fine sediment reduces the abundance of epibenthic invertebrates in shallow water ecosystems (Sobosinski et al., 2010; Toft et al., 2013), which can reduce epibenthic prey consumption by fish (Morley et al., 2012). Armoring that displaces backshore vegetation can reduce environmental diversity (Sobosinski et al., 2010) and fish consumption of terrestrial invertebrates (Toft et al., 2007). This reduction in availability and diversity of prey resources available for salmon is likely to be the most detrimental effect of armoring for foraging salmon species (Heerhartz & Toft, 2015).

Shorelines armoring changes juvenile salmon diets in the Puget Sound

Brennen et al. (2004) found that much of the marine mortality for Chinook salmon is determined by local conditions in the Puget Sound during their first spring and early summer. Declines in Chinook salmon marine survival since the 1980s may have been caused by reductions in the quality of feeding and growing conditions during their early life in the Puget Sound (Duffy et al., 2003). Terrestrial, shallow benthic, and pelagic habitats are the most important prey production and foraging areas for juvenile Chinook salmon in shallow marine areas of the Puget Sound. Insects specifically characteristic of terrestrial vegetated habitats, especially Hymenoptera, Homoptera, and Psocoptera, dominated the numerical composition of juvenile Chinook diets (Brennen et al., 2004). Most of the insects in the diets were fully developed winged adult forms, suggesting that they were likely wind-blown or fell from overhanging vegetation (Brennen et al., 2004). Benthic and planktonic invertebrates are also important in juvenile Chinook diets. Weight composition in Chinook salmon diets was similar between benthic, planktonic, and terrestrial prey categories. Dietary studies on Chinook, coho, and chum salmon from Hood Canal, and Commencement Bay, Duwamish Head, Skagit Bay, and Shilshole Bay, in Puget Sound, indicates that terrestrial insects and intertidal amphipods are the largest components of fish diets throughout Puget Sound. A fish diet analysis from Chinook salmon demonstrated a high proportion of terrestrial insects in this study as well, specifically Diptera, Homoptera, and Psocoptera (Rice, 2003).

There are been few studies that focused on salmon diets specifically between armored and unarmored beaches. Munsch et al. (2015) looked at diets from three species of juvenile salmon (Chinook, chum, and pink) to look at the differences in prey availability and feeding patterns among juvenile salmon between armored and unarmored beaches near the Duwamish River and restored pocket beaches along the urban shoreline along Elliott Bay. Shoreline armoring affected the composition of prey available in the environment for specifically Chum salmon along shorelines. Prey selectivity and diet composition of Chum salmon were different between armored and unarmored sites, although there was no difference in the diets or stomach fullness of other salmon groups.

Armored sites influenced the diet composition of juvenile chum salmon that select for epibenthic prey. As mentioned previously, epibenthic invertebrates living at the water-sediment interface are less diverse at armored sites where habitat structure is less complex (Toft et al., 2013). Other types of salmon which feed on plankton and invertebrates along the surface of the water were not affected at either site (Toft et al., 2013). At beaches, juvenile chum selected for epibenthic copepods, invertebrates living at the water-sediment interface. However, at seawall sites they selected for planktonic copepods (invertebrates that drift in deeper waters) (Munsch et al. 2015). This may be due to the loss of shallow, fine sediment beaches that reduces the availability of epibenthic prey (Munsch et al. 2015).

It is difficult to assess the effects of shoreline armoring on the diets of juvenile salmon due to fish mobility which could have an influence results of diet studies. However, shoreline armoring clearly affects prey resources and can change the feeding ecology of fish along developed waterfronts. Armoring can change the type of prey available, such as terrestrial insects and insects that utilize specific shallow water substrates that are altered due to shoreline armoring.

Shoreline armoring changes the distribution of juvenile salmon related to prey availability

Smaller fish less abundant along deep shorelines created by intertidal armoring

Shoreline armoring may influence juvenile salmon distribution and feeding behavior along shorelines. Juvenile salmon prefer unarmored sites that provide estuarine ecological functions, including shallow water protection and an increased diversity and abundance of prey species (Heerhartz & Toft, 2015). Heerhartz and Toft (2015) documented individual-level movement patterns and feeding behavior of juvenile salmon in shallow water along armored and unarmored shorelines. Snorkel surveys were conducted at an armored beach, a natural reference beach, and a “restored” beach with enhanced natural habitat features (at the Olympic Sculpture Park) located in the heavily armored shoreline in Elliott Bay during peak salmon outmigration periods, April-August (Heerhartz & Toft, 2015). Juvenile salmon had relatively high feeding rates along both armored and unarmored sites. However, the most important distinction between armored and unarmored shorelines for juvenile salmon may be the amount and type of prey available (Heerhartz & Toft, 2015).

Movement patterns and straightness index values were more diverse at natural beaches compared to armored shorelines (Heerhartz & Toft, 2015). Juvenile salmon move in complex paths when feeding, as they change swimming directions and dart to the surface of the water when attempting to capture prey, causing the fish to diverge from linear paths (Heerhartz & Toft, 2015). This finding suggests that fish at natural sites have increased feeding opportunities, as fish exhibited more diversity of swimming speeds and encompass a broader range of path straightness than fish at armored shorelines (Heerhartz & Toft, 2015).

Salmon demonstrated more diverse feeding behavior at natural, vegetated locations than armored sites where food sources are more limited (Toft et al. 2007). Juvenile salmon were observed darting to the surface to capture insects at more natural sites (Toft et al. 2007). Unarmored beaches also allow for more complex habitats and wider shallow intertidal zones which may enable fish to swim with greater path tortuosity while foraging while remaining in shallow water away from predators (Heerhartz & Toft, 2015).

Due to the highly mobile nature of fish, and their use of large stretches of shoreline, distinguishing population response to armoring is difficult (Dethier et al., 2016). Few studies have observed differences in fish response to armored shorelines using snorkel survey methods (Toft et al., 2013; Heerhartz & Toft, 2015). More studies using snorkel methods along Puget Sound shorelines may be beneficial to capture fish behavior along the shorelines.

Distribution of juvenile salmon - Olympic Sculpture Park case study

Toft et al. (2013) measured the effects of shoreline armoring on distributions of juvenile salmon in the light of available prey resources for salmon. Snorkel surveys were conducted in peak outmigration, April to July, before and after the site enhancement at the pocket beach, habitat bench, and adjacent riprap and seawall site for comparison. Overall, the feeding frequencies of juvenile Chinook salmon increased at the habitat bench and pocket beach dependent on the year after the site-enhancements (Table 1) (Toft et al., 2013). After the shoreline enhancement, the feeding frequencies of Chinook and chum salmon significantly increased after shoreline enhancement compared to the pre-enhanced period (Toft et al., 2013). Feeding frequencies were characterized by rapid forays to the surface to feed on neustonic prey, or terrestrial or marine organisms that float or drift near the surface of the water, and some feeding in the middle of the water column (Toft et al., 2013). The distribution of juvenile salmon changed in response to the habitat enhancement locations partly due to increased feeding opportunities (Toft et al., 2013). The more natural, enhanced beach is important for providing habitat that fosters increased diversity and abundance of prey species for juvenile salmon.

Although the percentages of salmon feeding generally increased at the enhanced sites over time, the number of salmon feeding between armored and unarmored sites did not always increase. For example, in year three, the number of chum salmon feeding at the seawall site were significantly higher (1525) at the seawall site than at the habitat bench (504) and pocket beach (163) (Toft et al., 2013). Overall, the enhancements showed improvements in salmon distribution and prey abundance as compared to heavily armored shorelines. Most importantly, shoreline restoration, especially in urban areas, may restore biological functions of the shoreline for fish.

Research needs

The effects of shoreline armoring on salmon should be fully understood when making management decision regarding the use of armoring structures on Puget Sound shorelines. This study provides more evidence of the effects of shoreline armoring on Vashon and Maury Islands, adding to the diversity of studies of armoring across the Puget Sound, focusing on less developed, residential properties on Vashon and Maury Islands.

There is widespread recognition by policy and management of potential adverse biological effects of shoreline armoring in the Puget Sound, however, there is still little empirical evidence documenting the effects, especially along the diverse regions of the Puget Sound. Many studies have been focused in highly urban areas, such as the Olympic Sculpture Park case study in Elliott Bay (near Seattle, WA). Less studies have focus in residential areas, where shoreline armoring currently makes up most of new armoring construction projects (Shipman, 2016). Puget Sound's shorelines are highly diverse, and it may be important to understand armoring impacts at specific regions across the Puget Sound. This study adds to the diversity of studies of armoring across the Puget Sound, focusing on on less developed, residential properties on Vashon and Maury Islands.

Literature regarding the biological effects of armoring is emerging, where armoring has been shown to negatively impact healthy nearshore ecosystems, including the estuary functions for juvenile salmon. Puget Sound ecosystems are regionally diverse, and subject to large differences in physical and biological conditions across space and time. Localized studies examining the biological response to armoring could benefit shoreline management policies and local decision making. This thesis is aimed to address the local impacts of shoreline armoring on key habitat features along shorelines specifically on Vashon and Maury Islands.

The results from this study are intended to increase our understanding of the physical and biological differences specifically between armored and natural shorelines in the MIAR. An important restoration goal is to improve habitat for juvenile salmon that utilize shorelines in the MIAR. This study establishes site conditions before shoreline armoring removal, scheduled to take place during the summer of 2018. The suite of environmental data analyzed in this study can be used as a metric for healthy shoreline conditions, marine-terrestrial connectivity, and juvenile salmon prey availability at beaches in the MIAR.

Chapter 3: Methods

Introduction

This thesis is a pre-restoration monitoring study that provides a baseline of shoreline conditions and how shoreline armoring affects the nearshore in the Maury Island Aquatic Reserve (MIAR). King County purchased three properties for purposes of bulkhead removal and environmental restoration at Big Beach, Lost Lake, and Piner Point. Existing structures and shoreline armoring on each property will be removed during August-September of 2018, which is after this thesis research was conducted. The natural shoreline and hillslope processes will be restored to the maximum extent practical (Booth & Legg, 2017). An important restoration goal and project funding for King County is to improve habitat for out-migrating juvenile salmon, as shoreline armoring is known to impact key functions of shoreline habitat for juvenile salmon.

Monitoring for terrestrial invertebrates, shoreline vegetation, beach wrack, forage fish spawning, and fish use occurred during the summer of 2017. At each site, monitoring occurred at three treatments: “pre-restoration” (targeted for bulkhead removal), “armored” (existing bulkhead that will not be removed), and “natural” (no bulkhead) treatment. The site selection for this project was determined by the Vashon Nature Center, King County, and the Washington State Department of Natural Resources. This project used standard field protocols adapted from the Washington Sea Grant’s Nearshore Monitoring Toolbox, a collection of simple, standardized monitoring protocols that can be used to evaluate the impacts of shoreline armoring across the Puget Sound (Shoreline Monitoring Toolbox, 2017).

Permission to access private property was obtained by the Vashon Nature Center. The sampling was overseen by Vashon Nature Center staff and conducted by trained citizen-science volunteers from the MIAR stewardship committee.

Site description

The MIAR is located on the eastern shores of Maury Island in central Puget Sound in the southwestern portion of King County. The reserve is approximately 5,530 acres of state-owned aquatic bedlands and tidelands located in Quartermaster Harbor (Perla & Metler, 2016). There are three sites in this study including Big Beach, Lost

Lake, and Piner Point. Big Beach and Lost Lake are in Quartermaster Harbor, between Vashon and Maury Islands (Appendix A). Piner Point is more exposed, as it is located just outside of the harbor on the southern tip of Maury Island.

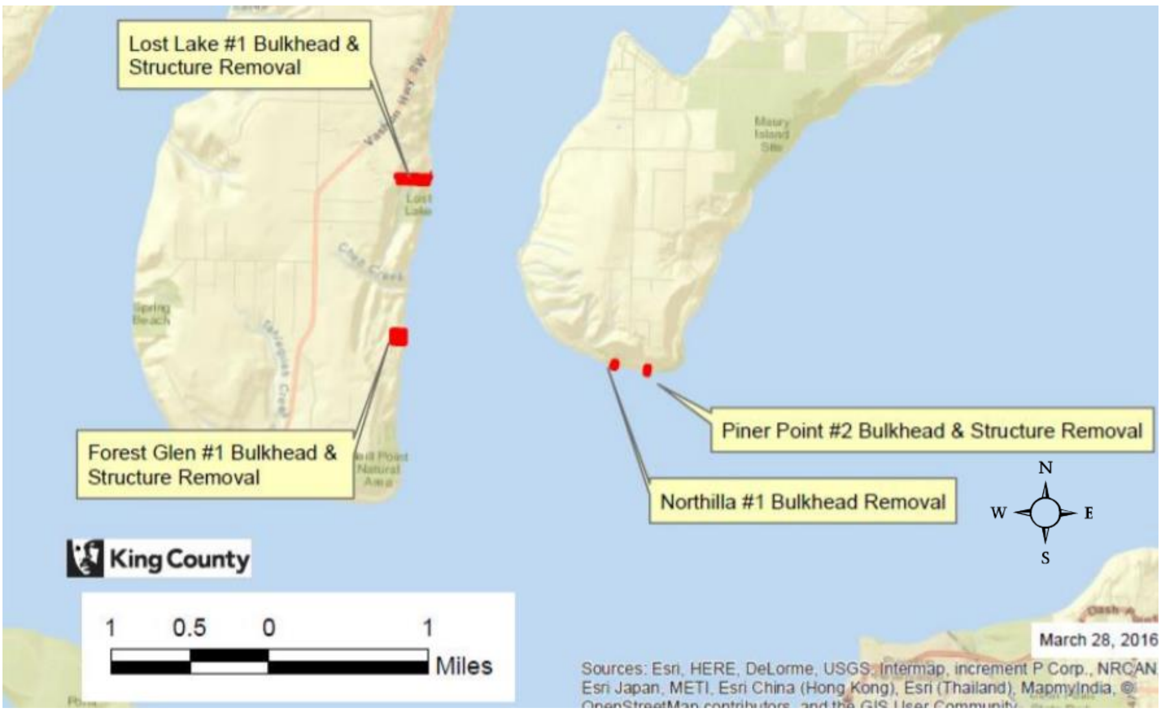


Figure 1: Vicinity map of Vashon and Maury Island showing shoreline armoring removal sites (Booth & Legg, 2017).

Puget Sound is a deep, well-mixed basin with moderately high energy, making for a unique estuary (Sobocinski, 2003). The estuarine beaches lack severe exposure, as seen on Washington’s outer coast, but are still subject to physical processes such as wind, waves, current, longshore current, and swell, not typical of more enclosed estuarine systems (Nordstrom, 1992).

The climate within the MIAR is influenced by the maritime pacific climate, which dictates the entire Puget Sound region. Annual temperatures remain mild, while precipitation levels vary greatly by location and season. Within the reserve, average rainfall measures differ by 10” from the western to the eastern extent of the reserve. According to King County records, the western region of the reserve receives an average of 46” annually, while the eastern edge at Pt. Robinson receives only 36” annually (Perla & Metler, 2016).

Tides within MIAR are large, with ranges between 3 and 4 meters (Perla & Metler, 2016). The tides are forced by the tidal variation of sea level at the mouth of the Salish Sea–the seaward end of the Strait of Juan de Fuca. The tides bring in about 8 km^3 of water each high tide, removing that water roughly 12.4 hours later (Perla & Metler, 2016).

The sites included in this study are mainly comprised of the beach type described as open estuarine intertidal habitats (Dethier, 1990). The primary sediment composition on these sites was a mix of sand and gravel derived from glacial and interglacial deposits, delivered to beaches via episodic bluff erosions, and distributed by longshore transport (Shipman, 2010). Wave energy regime and local geology are the primary drivers of beach sediment character and gradient in the Salish Sea (Dethier, 2016). Paired beach sites were within the same drift cell, or independent zone of littoral sediment transport from source to deposition area, and within the same component of that drift cell (erosional or depositional) (Dethier, 2016). This study is unique because the study sites are backed by tremendously steep and actively eroding bluffs. Slopes are very unstable at these sites and there have been recent landslides in the area (Figure 2). Changes are expected to happen quite fast after bulkhead removals are completed (Perla & Metler, 2016).

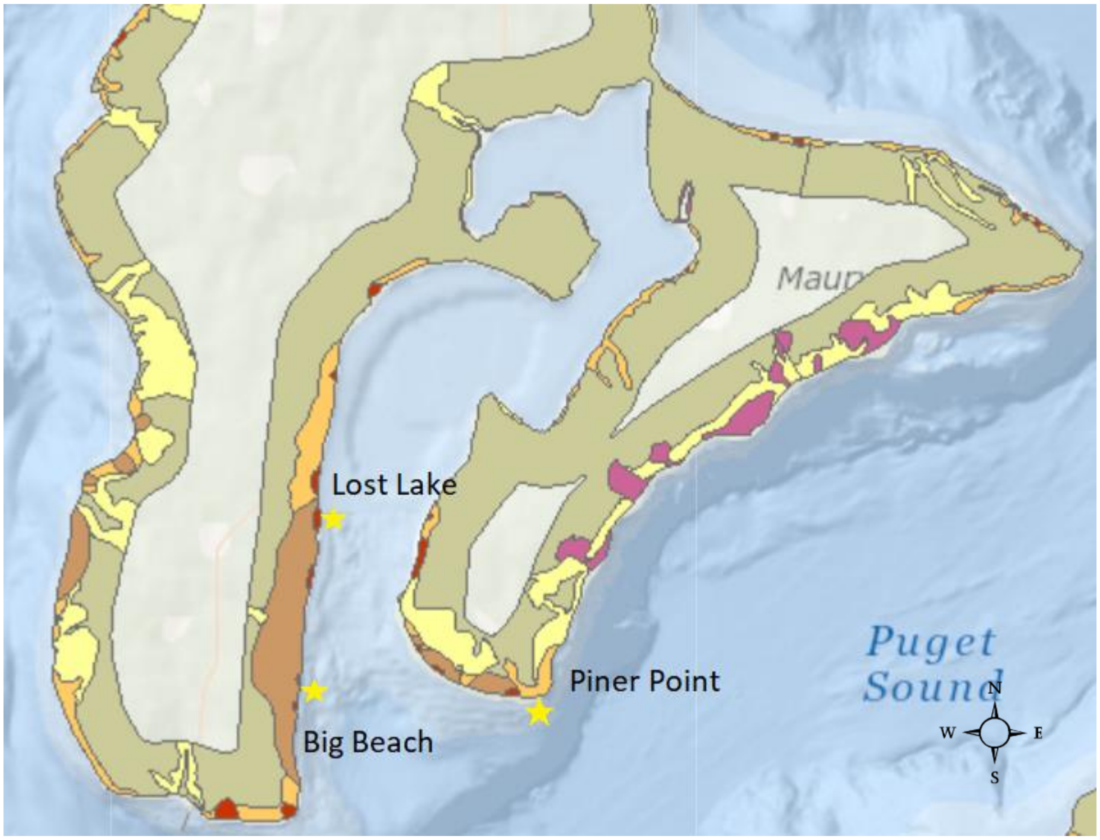


Figure 2: Slope stability in the Maury Island Aquatic Reserve. Slopes are unstable at all three sampling locations. The red indicates unstable slopes where there have been recent slides, demonstrating that each sampling location has been subjected to recent slides. The surrounding areas are also unstable, as the brown indicates unstable slopes with old slides, and the orange indicates unstable slopes in general (Washington State Coastal Atlas, 2018).

Big Beach

Big Beach (BB) is an eastward-facing, low-grade beach. This site is south of the Lost Lake site, toward the mouth of Quartermaster Harbor. This is a residential beach, however, most of the housing is set back from the beach due to the high-bank nature of this site. The pre-restoration site (-122.49166, 47.34558) is over 50 meters, and is comprised of a hodge-podge bulkhead made of wood and large boulders, or rip-rap that will be removed during the restoration. The armored site is located directly adjacent to the south of the pre-restoration site, and is characterized by a tall, concrete bulkhead. The natural site, located north of the pre-restoration and armored sites, is undeveloped and comprised of dense overhanging vegetation. The beaches in this site are approximately in the southern end of the same drift cell that drifts from south to north (Figure 3).

Lost Lake

Lost Lake (LL) is an eastward-facing, low-grade beach. This is the innermost site in Quartermaster Harbor. There is a small housing development comprised of a few houses along the beach, placed directly above the shoreline armoring. The pre-restoration site (-122.48857, 47.36060) is characterized by a 30 m wooden bulkhead with a house and a few shrubs placed directly above the armoring. The bulkhead and house will be removed during the restoration. The natural site is directly south of the pre-restoration site and has visible logs, salt grass, and overhanging trees and shrubs. The armored site is one of the most northern properties in this small housing development along the beach. The armored site has a short, wooden bulkhead with adjacent trees, shrubs, and no visible house placed in the uplands. The beaches at this site are in approximately the center of the same drift cell that drifts from approximately the south to north (Figure 3).

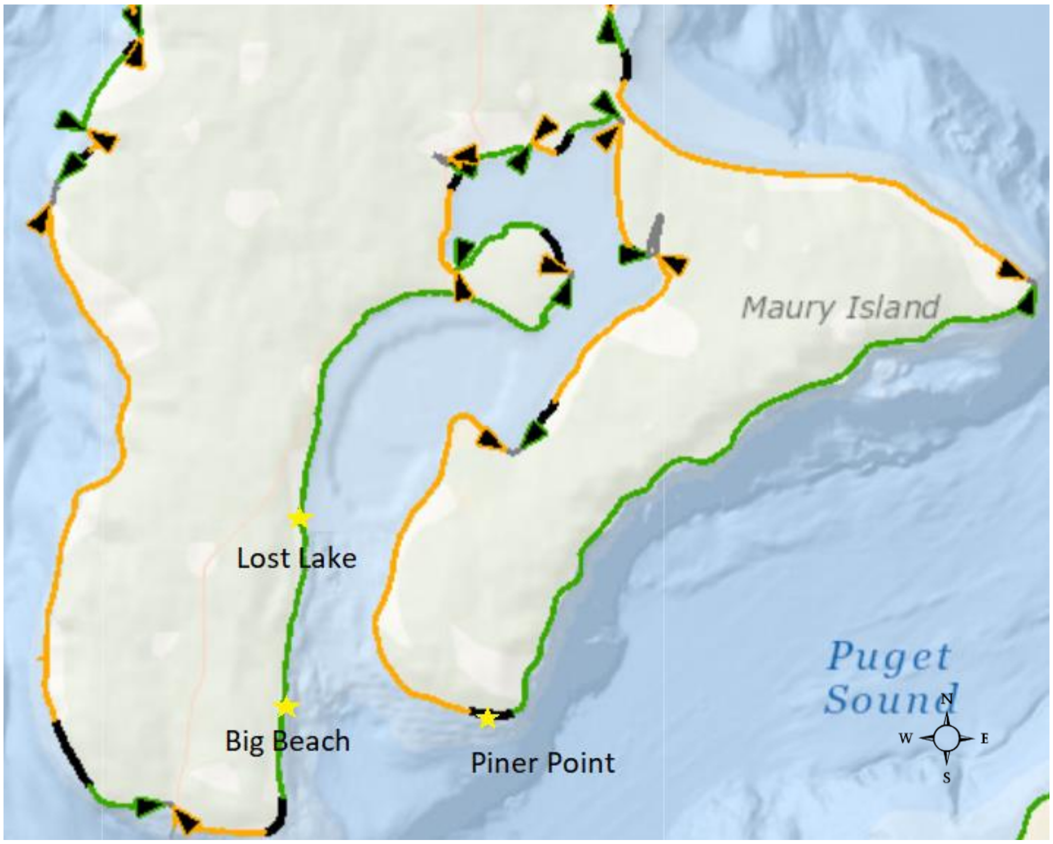


Figure 3: Drift cells in the Maury Island Aquatic Reserve. Arrows point in the direction of the drift movement. Lost Lake and Big Beach sampling locations are in the same drift cell that drifts from south toward the north (green line). Piner Point is in a divergence zone (black line), which is generally subject to more rapid erosion and significant sediment sources within littoral cells (Washington State Coastal Atlas, 2018).

Piner Point

Piner Point is a southern-facing, low-grade beach. Piner Point is the outermost site in Quartermaster Harbor, located at the south tip of Maury Island. This is a highly erosive, high-bank beach. The pre-restoration site (-122.45894, 47.34329) is a 30 meter, failing wooden bulkhead. The uplands consist of trees and shrubs that have been subjected to landsliding in the area. The armored site is located toward the west of the pre-restoration site, and is a hodge-podge of wooden structures, with a house placed almost directly above the shoreline armoring, with few overhanging trees and shrubs. The natural site is directly east of the pre-restoration site, with visibly eroding high-bank, accumulated logs, and some overhanging trees and shrubs that have be subjected to landslides. The beaches at this location are in a divergence zone, which is subject to more rapid erosion and significant sediment sources within littoral cells (Shipman, 2008) (Figure 3).

Experimental design

This study addresses how shoreline armoring can impact natural nearshore ecosystem functions by comparing paired armored and unarmored beaches at three sites. Each study site has three treatments: a pre-restoration (where the bulkhead will be removed), a natural (where no bulkhead exists), and an armored (bulkhead). The different treatment locations were chosen based on county plans to remove the bulkheads so those sites anchored subsequent location decisions. The transect lengths were consistent within each study site. Piner Point and Lost Lake transects are short (30 m) and Big Beach are long (50 m), which were determined by the length of bulkhead being removed. Transects were placed parallel to the shore on the fresh wrack line, where the most recent debris was left behind at the previous high tide. When no wrack line was present (i.e. on many bulkheaded beaches), transects were placed at the previous high tide line or toe of armoring.

The three sites at each beach location were selected in close proximity. This site selection minimizes the effects of physical properties including deposition and movement of organic debris and sediments that are largely driven by local wind, waves, and currents (Nordstrom, 1992). Treatments were close, and in some cases directly adjacent to each other, so they had similar aspects (with respect to sun, waves, and weather), bank slope, and type of sediment. When possible, armored sites with similar bulkhead structures and materials to the pre-restoration site were chosen (Perla & Metler, 2016). The three sites are contained within the same drift cell (Figure 3), reducing the variation in physical characteristics between sites (Sobocinski, 2003). This sampling regime, while beneficial in terms of minimizing spatial differences, is inherently biased due to variations between sites and treatments (Sobosinski, 2003).

Sample timing and frequency

Sampling occurred during summer low tides for maximum accessibility and to ensure all parameters were measured. Big Beach sampling occurred in June, Lost Lake in July, Piner Point in August 2017. Surveys occurred during different months due to scheduling and accessibility restrictions. Variables tested do not differ significantly between these months (Perla, 2018). Sampling was performed over two days at each site for all beach surveys. For each beach site survey, data was collected on the same day during low tide.

Snorkel surveys were conducted over a single day at each site during the high tide. Data collection occurred at Lost Lake and Big Beach in July and Piner Point in August 2017.

Forage fish spawning data is continually collected by citizen scientists every other month at these sites, except during June, July, August, when one site is sampled each month at the same time as the suite of shoreline surveys.

Surveys

Terrestrial Vegetation

Riparian vegetation provides habitat for terrestrial insects that are important prey resources for juvenile salmon. Characterizing shoreline vegetation can give valuable information about the habitat of the upper beach, marine-terrestrial connectivity, as well as habitat availability for insects (Shoreline Monitoring Toolbox, 2017).

This protocol was taken largely from the Shoreline Monitoring Toolbox (Shoreline Monitoring Toolbox, 2017) with a few modifications (Appendix B). Specifically, the overhanging riparian cover measurements were modified to better suit the conditions at these sites.

For each study site, a list was created of plants in the tree, shrub, and groundcover layers that occur in the length of the 30 or 50 m transects. Every tree that overhangs the beach and its species were counted and recorded, using the established transect as a length to sample. The width of the tree canopy that overhangs the beach was estimated. All the overhanging widths along each transect were totaled and divided by the entire transect length to get an estimate of percent cover of overhanging vegetation along the study transect.

The cover of overstory (trees) and understory (shrubs and groundcover) vegetation was estimated for three different 5x5 m plots placed along the length of the established transect at 0-5 m, 10-15 m, and 25-30 m (or if a 50 m transect at 0-5 m; 20-25 m, 45-50 m). The established transect was used as a length to sample, however, plots were placed at the bluff toe or bulkhead and extended back five meters into the terrestrial landscape. The general health of the plants was estimated for understory and overstory on a scale of 1-5 (1=barely alive; 2=major damage over more than 50 percent; 3=some damage to 30 percent; 4=minor damage less than 20 percent; 5=thriving). Reasons for low health scores were recorded in notes (i.e. drought stress, slide stress, disease). All vegetation species were categorized by native and non-native species.

Beach wrack

Beach wrack may be an important source of nutrient exchange between marine and terrestrial systems and provides shelter, food, and moisture for invertebrates. Dependent variables include the percent cover of rack and composition of marine and terrestrial organic debris. Examining composition can give information on the source material (terrestrial vs. marine sources) and the associated amounts that deposit on the beach at each site type.

Sampling was based on the Shoreline Monitoring Toolbox methods for wrack sampling (Appendix C). Briefly, two transects were established: One at the most recent high tide line with fresh wrack deposition, and a second just above MHHW where older wrack accumulates. These locations were established by visually observing wrack lines on the beach. The most recent high tide line targets mobile wrack, whereas the higher elevation sample targets the more stable wrack layer and extent that wrack is mobilized during storms. Armored beaches tended to lack wrack and log accumulation, and therefore, only the lower wrack lines were analyzed in this study. Presence of upper wrack and logs are noted at each site.

A 0.1 m^2 quadrat was placed on the beach surface at ten randomly selected points along the 30 or 50 m transects placed parallel to shore. A visual estimate of the total percent cover was taken. The percent composition of marine algae, terrestrial plant material, and eelgrass was recorded at each quadrat, and therefore, analyzed as independent of each other.

The visual assessment was based on a percentage of the quadrat, divided into 25 6x6 cm small squares, where each square equals 4 percent. Algae type (e.g., red, green, brown, or other species) was recorded. When there was less than 0.01 percent cover, we used a standardized low number (.01) to differentiate between small amounts of wrack cover to nothing at all.

This study only compares the lower wrack line. All sites had a lower wrack line, however, not all sites had an upper wrack line, creating a difficulty in comparing upper wrack lines across sites. Therefore, total percent cover number reported in this study may not be representative of the total percent cover on the actual beach.

Terrestrial insect fallout traps

Terrestrial insects are an indicator of shoreline conditions and are an important prey for juvenile salmon. Examining changes in insect assemblages due to armoring can be important, as the population stability may have cascading impacts on higher trophic levels, including fish (Shoreline Monitoring Toolbox, 2017).

Insect sampling occurred in the summer months (June-August) as juvenile Chinook salmon are feeding along the shoreline, and vegetation and insect communities are developed. The dependent variables measured include taxa richness, the number of different taxa in the sample, and composition, focusing on key salmon prey species.

Sampling was based on the Shoreline Monitoring Toolbox methods for insect sampling (Appendix D). Briefly, fallout traps were created using plastic storage bins (34.6 cm x 21 cm) filled with a weak soap-water solution. Five replicate bins were placed randomly along a 30 or 50m long study transect parallel to shore and left in place for 24 hours. After 24 hours, the contents were passed through a 0.106 mm sieve, and the material retained was preserved in 70% isopropyl alcohol. Insect samples were processed in the laboratory for numerical composition with taxonomic resolution to family.

Density was calculated by summing the number of insects found in each bin and dividing by the surface area of the bin (0.07266 m^2), for each of the five bins at each treatment. The average was then taken across the fives bins for each treatment at each site. Taxa richness was calculated by counting the number of species that occurred in each of the five bins.

Forage fish

Surf smelt (*Hypomesus pretiosus*) and Pacific sand lance (*Ammodytes hexapterus*) spawn on the beach, depositing their eggs in the sediments on the upper beach. Successful forage fish spawning can be an indicator of a healthy beach. Spawning can be impacted by changes to the nearshore due to shoreline armoring, since specific sediment sizes and tidal elevations are targeted by these fish. These fish are a vital part of the food web, being preyed upon by larger fish (e.g., salmon), marine mammals, and birds (Shoreline Monitoring Toolbox, 2017).

Bulk beach substrate samples were collected by VNC citizen scientists at each treatment at each site. Standardized forage fish beach spawning data collection methods established by Washington Department of Fish and Wildlife were used (see Appendix E). In short, a 30.48 (100 foot) transect tape was placed parallel to the shore at sandy-gravel substrates. Tidal elevation of the transect is determined by measuring the distance from the transect to an identified landmark, such as upland toe of the beach, the last high tide mark, or the water's edge. Along the established transect tape, bulk substrate samples were collected by scooping the top 5-10 cm of sediment (about two foot long scoops) at 10 evenly spaced locations.

Substrate samples were wet-screened through a through set of 4 mm, 2 mm, and 0.5 mm sieves using buckets of shore-side water. The material from the 0.5 mm sieve was placed into a rectangular dishpan with an inch of water, and winnowed into subsamples of forage fish egg-sized material. Winnowing consists of rotating or tilting the dishpan of material to cause lighter material to rise to the surface, and in short, suspend any forage fish eggs to the top of the sediment sample. Egg subsamples were collected by scooping the top layer of lighter sediment material (and any eggs) into a 16 oz jar. Sub-samples were sent to the Washington Department of Natural Resources

Aquatic Reserve Program's laboratory to be analyzed for spawning presence/absence and number of eggs.

Data was collected every other month at each site starting December 2016. During June, July, August, data was collected during the full beach surveys, and therefore only one site was sampled each month during this time.

Fish observations

The natural history of shallow-water fish communities can help identify and account for critical habitat function. This method was based on the Shoreline Monitoring Toolbox's methods for fish observation (Appendix F). During the highest tide of the day, two 50 meter transects were established parallel to the shore. One transect was established at 1.5 m depth, about 20 meters from the shore. The second transect was established at approximately 2.0 meters depth and approximately 30 meters from the shore.

Observers started by measuring underwater visibility. Ideally, surveys should only occur when visibility exceeds 2.5 m to maximize the accuracy of observations and minimize effects of observed on fish behavior (Toft et al., 2007). The shallow water depth was measured at 1.5 m using a weighted line. The second water depth was measured 10 m away (away from the beach) at the beginning of the second transect using a weighted line. The second depths varied, but were consistently around 2 m of depth. Transects ran parallel to the shore.

Observers recorded the following variables for each fish species encountered: species, a visual estimate of length to the nearest centimeter, number of individual fish, and water column position of the fish. When fish were not identifiable to the species level, names of lower taxonomic resolution were used to describe their identity (e.g. unknown forage fish). Water column positions were described in thirds: top, middle, and bottom. Feeding behavior (i.e. darting to the surface) was recorded when applicable. Number of fish and observations were averaged by treatment type (armored, natural, and pre-restoration). Taxa richness was calculated by averaging the number of species by treatment type.

Due to proximity, Big Beach and Lost Lake sites were sampled during the same day in July. Piner Point was sampled a month later in August.

Analysis

For all analyses, the independent variables were treatment type (armored, restoration, and natural sites). The following variables were compared at armored and unarmored beaches where data was collapsed across all sites and examined at each site individually as well.

- Vegetation (percent overstory, percent understory, native vs. non-native species)
- Wrack (total percent cover, percent marine, percent terrestrial, percent eelgrass)
- Insects (taxa richness, density)
- Forage fish spawning (spawning events, number of eggs)
- Fish (number of fish, taxa richness, number of observations)

All data were organized in Microsoft Excel XP®. Preliminary data exploration was performed in Excel. JMP Pro 12 was used for subsequent data analysis when sample sizes were appropriate. Statistical analysis was not performed when sample sizes or sampling frequency was low and data was non-normally distributed. All data was checked for violations of normality and skewness, and was transformed if violations were detected. Numerical data was log transformed. Arcsine transformations were applied (p'=ASIN(SQRT(p)) to proportional datasets. Transformed data was then reassessed to ensure no violations persisted. Statistical analysis was conducted on the transformed dataset. If the dataset still violated assumptions of normality and skewness upon transformation, non-parametric statistical tests were used.

One-way analysis of variance (ANOVA) was used to analyze differences between treatments (armored, natural, pre-restoration). Post-hoc tests were used to further examine differences using the Tukey test. When significant differences were found between treatments, a 2-way ANOVA was used to test for differences between groups while accounting for site and treatment (the two independent variables) as independent of each other. The critical p-value in assigning statistical significance was $\alpha=0.05$.

Kruskal-Wallace tests were used as the nonparametric equivalent to the ANOVA test. A Dunn test was used as a nonparametric post-hoc test. P-values were based the corrected alpha using the Bonferroni correction of ($p=0.0167$). The Friedman's test, or randomized-block design test, was used as a non-parametric equivalent to the mixed-design ANOVA to account for site and treatment. Post-hoc tests were not calculated for the Friedman's test, as methods for this remain relatively uncommon (Wobbrock et al., 2011). The critical p-value in assigning statistical significance was $\alpha=0.05$ for these tests.

Chapter 4: Results

The results section is broken up into sections based on the five beach parameters assessed in this study: terrestrial vegetation, wrack, insects, forage fish, and fish. Each section presents the findings from the analysis of each parameter, and assesses whether there are treatment or site effects for each parameter, when applicable. If there were no significant differences in parameters across treatments, then site interactions were not included.

Terrestrial vegetation

Treatment effects on overstory percent cover

The percent cover of overstory vegetation was averaged across all sites based on treatment (armored, natural, pre-restoration). In general, natural sites had greater average overstory vegetation coverage (83.89 ± 7.94%), compared to the armored (48.33 ± 12.25%) and pre-restoration (40.56 ± 13.86%) sites (Figure 4), an observation that was significant based on Tukey's HSD test (ANOVA $F_{(2,24)} = 3.86$, $P = 0.04$).

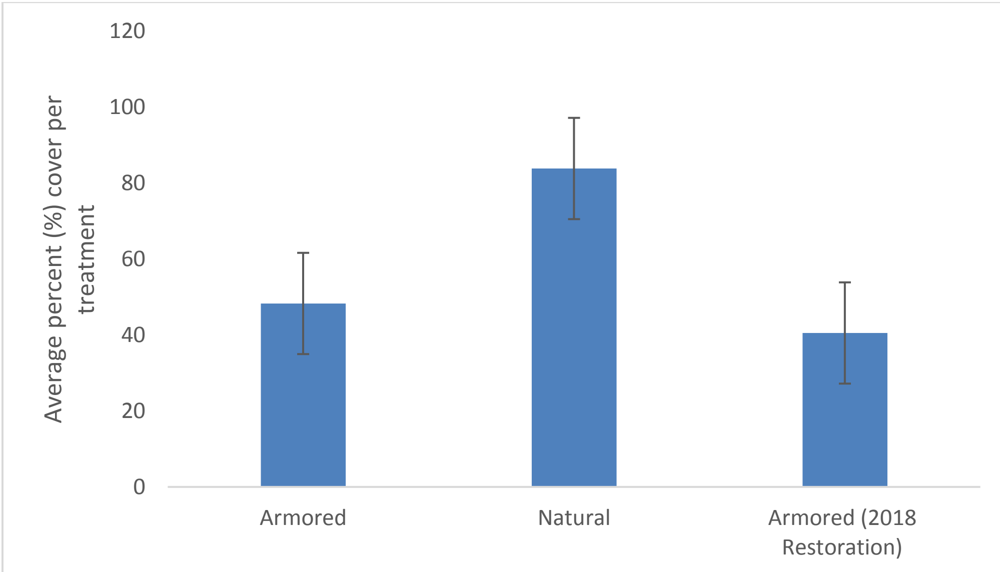


Figure 4: Average overstory vegetation percent cover per treatment.

Variations in overstory percent cover based on site

There is an interaction in that site affects overstory percent cover estimates (ANOVA $F_{(8,18)} = 1.49$, $P = 0.047$). However, the natural sites consistently have higher percent cover. Percent cover of overstory vegetation was highest at natural treatments at every study site (Figure 5). For example, the natural site at Piner Point had highest overstory percent cover (96.67 ± 3.33%), as compared to the armored (60 ± 5.77%) and pre-restoration (60 ± 15.28%). This was also true for Lost Lake and Big Beach.

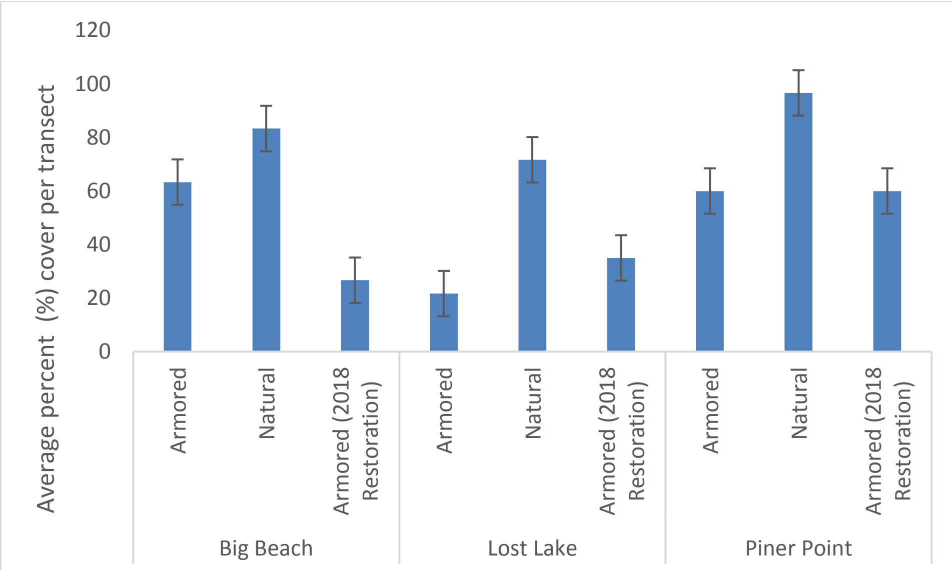


Figure 5: Percent cover of overstory vegetation per site.

Variation in understory percent cover by treatment

Understory vegetation was also averaged across all sites based on treatment (armored, natural, pre-restoration). The average understory percent cover was consistently high across treatments at all sites (averages ranged between of 73-88% across natural, armored, and pre-restoration sites) (Figure 6). When averaging across sites, the pre-restoration site had the highest average amount of understory (88.33 ± 9.98%), although there were no significant differences between treatments (Kruskal-Wallis $\chi^2_{(2)} = 2.30$, $P = 0.32$).

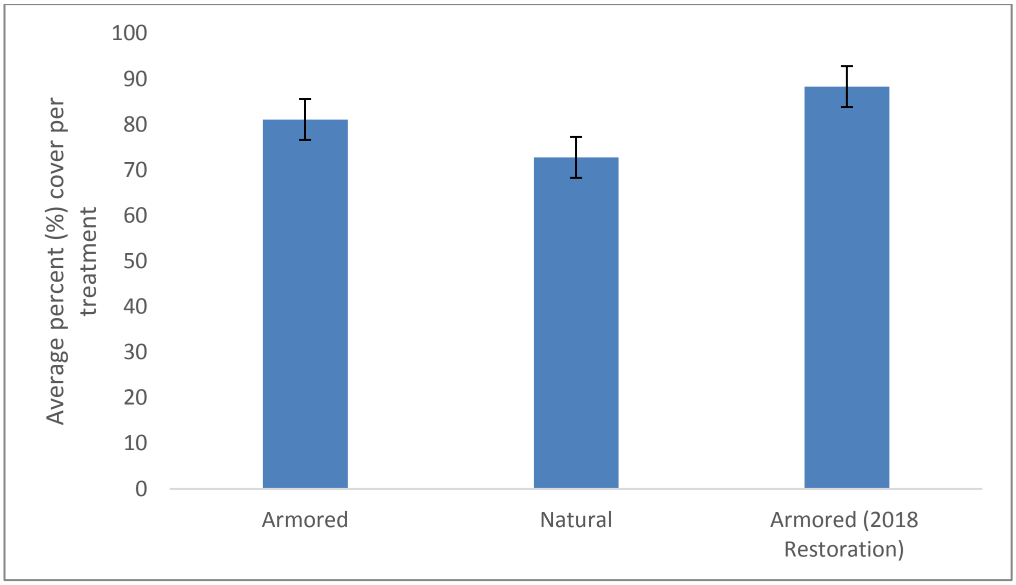


Figure 6: Percent cover of understory vegetation per treatment.

Variation in understory percent cover by site

Understory vegetation was high at all sites, ranging from approximately 40% to 100% cover across treatments at all the sites (Figure 7). Vegetation completely covered the understory at Big Beach and Piner Point pre-restoration sites, raising the average compared to the armored and natural sites. Pre-restoration sites have not been maintained for years, allowing for vegetation to grow even though there is development at the site.

At Big Beach, understory almost completely covered all sites (armored (90 ± 5.77%), natural (78.33 ± 14.24%), and pre-restoration (100 ± 0.0%)). The overall plant health at Big Beach was high (4-5). At Lost Lake, the armored and natural sites were completely covered with understory plants, and the pre-restoration site was about two-thirds covered on average (65 ± 23.61%). The overall health rating for plants at Lost Lake was high (4-5, with some lower health ratings at armored and pre-restoration sites due to development). At Piner Point, percent cover of understory was highest at the pre-restoration site (100 ± 0.0%), as compared to the armored (53.33 ± 24.04%) and natural (40 ± 5.77%). The Piner Point natural and pre-restoration beaches had lower average health ratings (3) than other sites, as invasive species and drought and salt stressed plants were present.

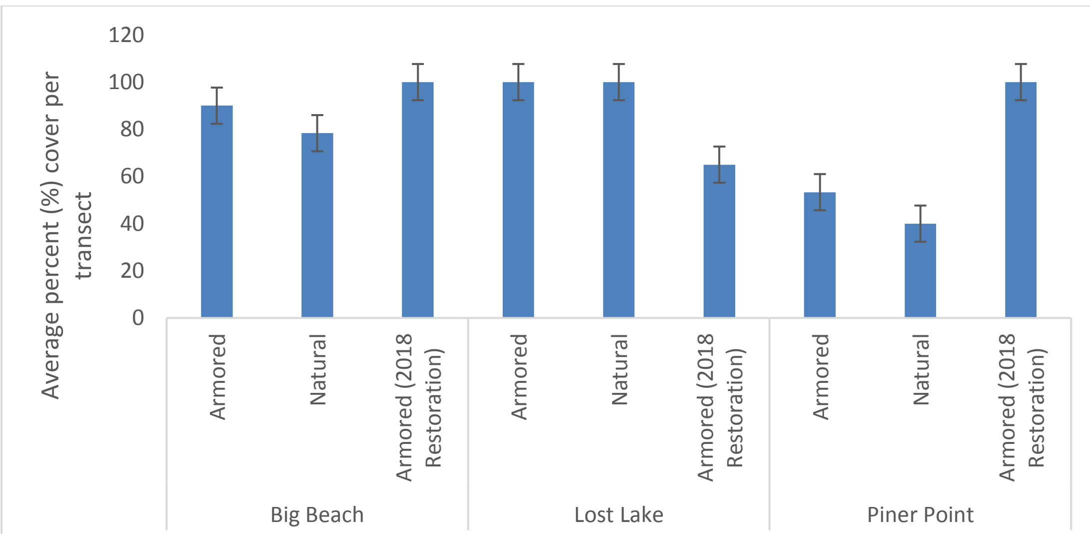


Figure 7: Percent cover of understory vegetation per site.

Overhanging trees per treatment

Overhanging trees were averaged across all sites based on treatment (armored, natural, pre-restoration). Statistical analysis was not performed due to small sample sizes. On average, natural sites had more overhanging tree cover (23.33 trees), compared to the armored (3 trees) and pre-restoration sites (3.67 trees) (Figure 8). All tree species were native at all sites in this study.

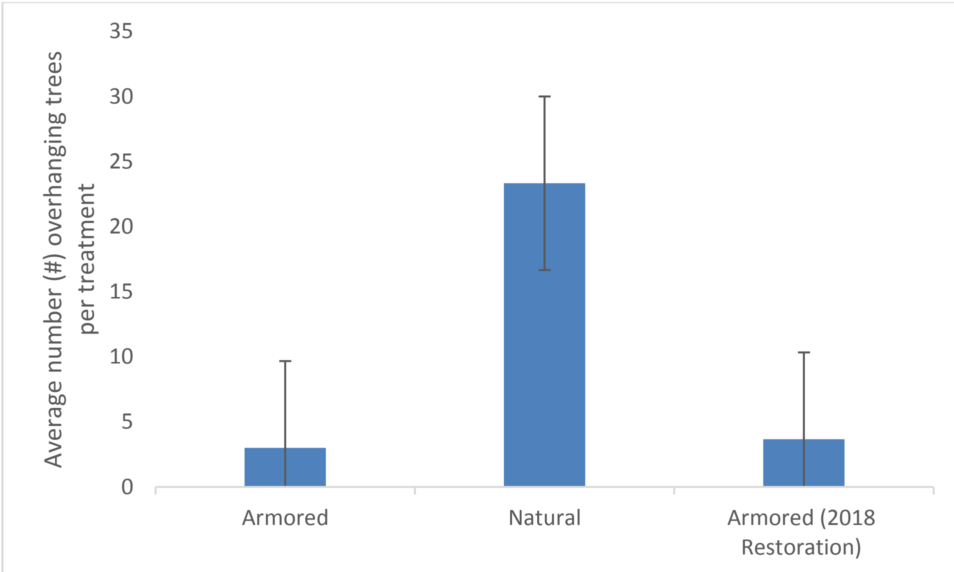


Figure 8: The number of overhanging trees per treatment.

Overhanging trees per site

Overhanging trees were more abundant at natural sites. Statistical analysis was not performed due to small sample sizes. Piner Point had the highest number of overhanging trees overall at the natural site (55 trees), compared to armored (4 trees) and pre-restoration (8 trees) (Figure 9). Tree species at Piner Point were overall more diverse. The natural site consisted of clumps of alder trees, madrone, and maple. The armored site consisted of maple and alder trees. Maple, cedar, salix, shorepine, laurel, and a few dead trees were present at the pre-restoration site.

At Lost Lake, the highest number of overhanging trees were found at the natural site (3 trees), compared to armored (1 tree) and pre-restoration (1 tree). All trees along this transect were alder.

The natural site at Big Beach had the highest number of trees (12 trees), which were all maple trees, compared to armored (4 trees) and pre-restoration (2 trees). Trees at the pre-restoration and armored sites were exclusively alder.

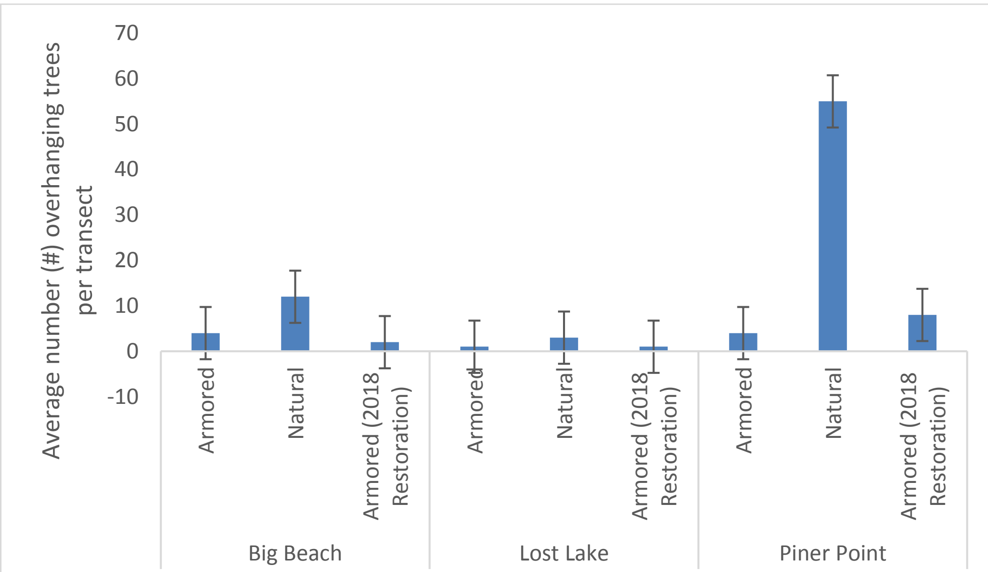


Figure 9: Number of overhanging trees per site.

Native vs. non-native species counts per treatment

Native and non-native species counts were averaged across all sites based on treatment (armored, natural, pre-restoration). Overall, natural species were more abundant at natural sites. Differences between the number of native and non-native species was the highest at the natural sites (37 native versus 7 non-native species). The number of native compared to non-native species at armored and pre-restoration shoreline types was similar, where there were 31 native compared to 30 non-native species at the armored shorelines and 20 native and 20 non-native species at the pre-restoration shorelines. Overall, the count of non-native species was highest at the pre-restoration site, potentially due to the lack of yard maintenance at pre-restoration sites (Figure 10).

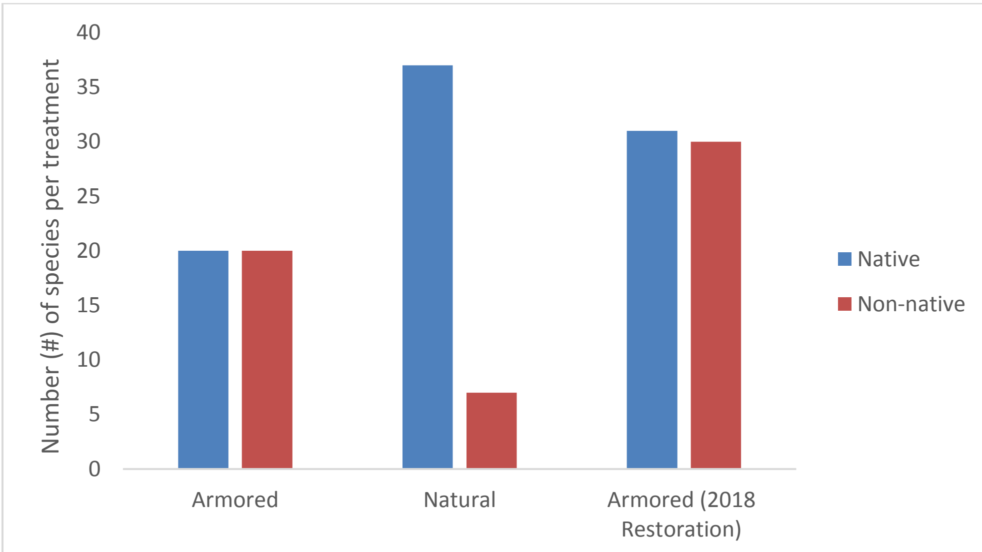


Figure 10: Number of native and non-native species per treatment.

Abundance of native vs. non-native species per site

There were no substantial trends in the number of native species compared to non-native species per transect when comparing treatments each individual site (Figure 13). However, there were a few findings of interest. There was consistency between the number of native and non-native species across all treatments at Piner Point. There were 15 native species and no non-native species at the Big Beach natural site. The highest number of non-native species were found at the Lost Lake pre-restoration site (14 non-native species).

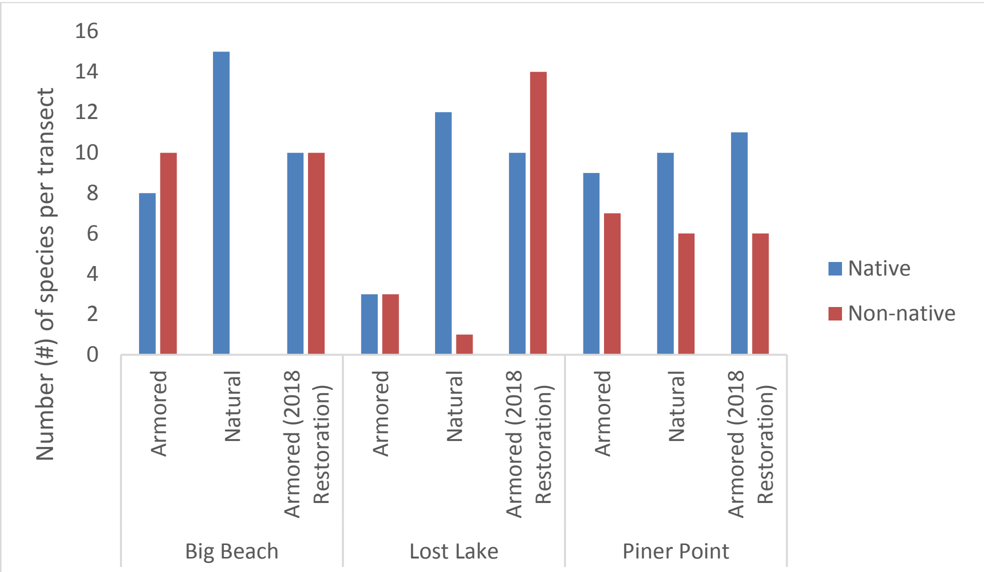


Figure 11: Number of native and non-native species at each site.

Wrack

Variation in wrack total percent cover based on treatment

Total wrack cover varied between treatments when averaged across sites (Kruskal-Wallis $\chi^2_{(2)}$ = 12.36, *P* = 0.002) (Figure 12), with more wrack found at the pre-restoration sites than the armored sites (Dunn test *Z* = 3.64, P = 0.0003). The natural (21.4 ± 5.2%) and pre-restoration sites (25.5 ± 5.25%) had similar average total percent cover, whereas average total percent cover was lower at the armored sites (7.8 ± 2.28%). A Friedman's test demonstrated that there was no interaction, meaning that treatment (armoring) has an effect on wrack total percent cover regardless of site ($\chi^2_{(2)}$ = 4.67, p = 0.097).

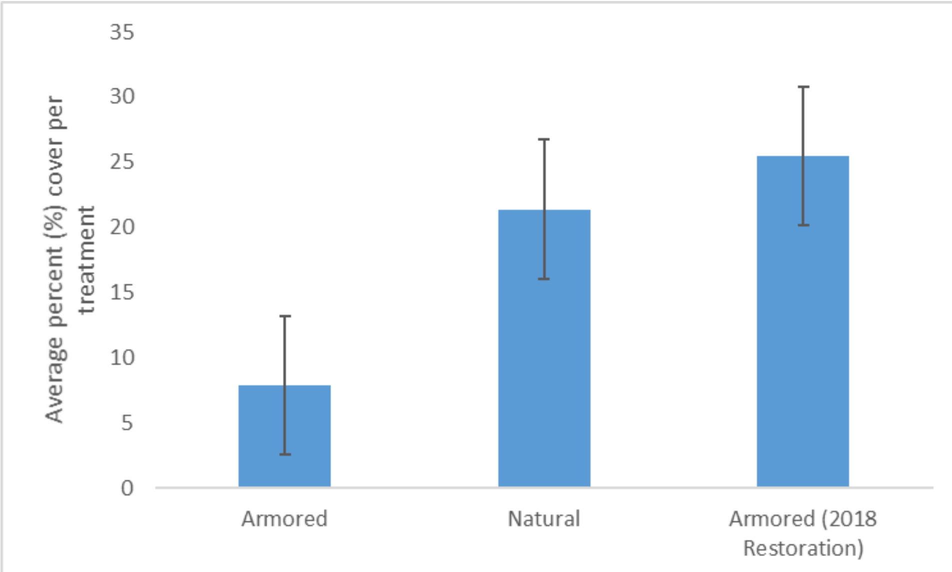


Figure 12: Percent cover of total wrack per treatment.

Site as a basis for variation in wrack total percent cover

There were no consistent trends in total wrack cover when comparing across treatments at each individual site (Figure 13). At Big Beach, there was a higher average percent of total wrack cover (40.3 ± 11.1%) at the pre-restoration site, although this difference was not statistically significant compared to the natural (19.6 ± 9.69%) and armored (16.6 ± 5.63%) sites (ANOVA $F_{(2, 27)}$ = 1.87, p=0.17).

At Lost Lake, total percent cover varied across treatments Kruskal-Wallis $\chi^2_{(2)}$ = 13.91, *P* = 0.001). Total wrack was highest at the natural site (39.4 ± 9.29%), as compared to the pre-restoration (28.5 ± 8.80%) and the armored (3.8 ± 1.67%) sites. The armored site differed significantly than the natural (Dunn test *Z* = 3.16, *P* = 0.002) and pre-restoration sites (Dunn test *Z* = 3.08, *P* = 0.002).

At Piner Point, total percent cover did not significantly vary across treatments (ANOVA $F_{(2, 27)}$ = 0.83, p = 0.45). Percent cover was highest at the pre-restoration site (7.7 ± 2.18%) compared the armored (3.2 ± 1.71%) and natural sites (5.13 ± 3.27%), although these results are not significant (Figure 13).

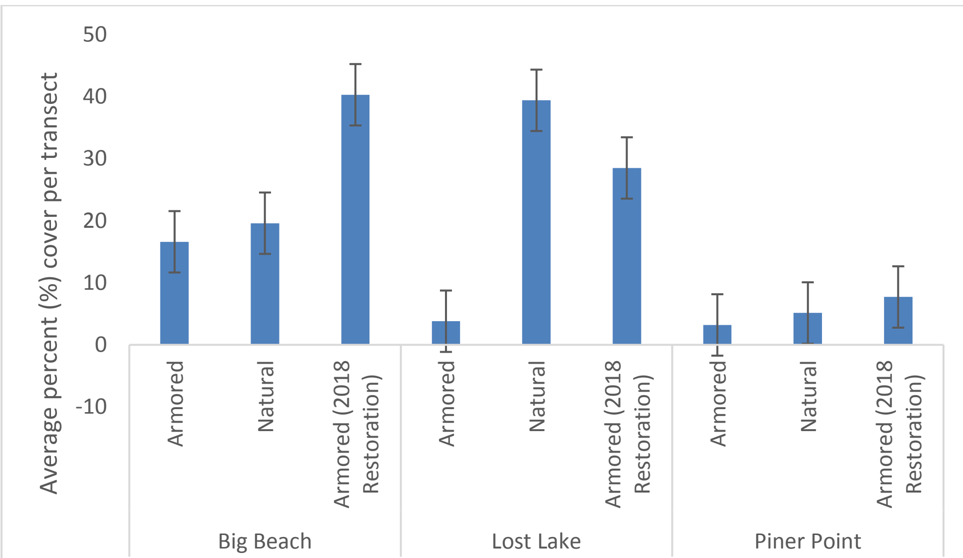


Figure 13: Percent cover of total wrack per site.

Variation in marine algae in wrack based on treatment

Percent cover of marine wrack varied across sites (Kruskal-Wallis $\chi^2_{(2)}$ = 17.003, p=0.0002) (Figure 14). The natural (24.20 ± 4.89%) and pre-restoration sites (25.10 ± 4.80%) had similar average percent cover, which was higher than the amount of marine wrack at the armored site (7.87 ± 2.28%) (Dunn test Z = 3.19, P = 0.004; Z = 3.88, P = 0.0003, respectively). A Friedman's test showed that there was no interaction between site and treatment ($\chi^2_{(2)} = 4.67$, p = 0.097). Ulvoid algae dominated the percent cover of wrack samples.

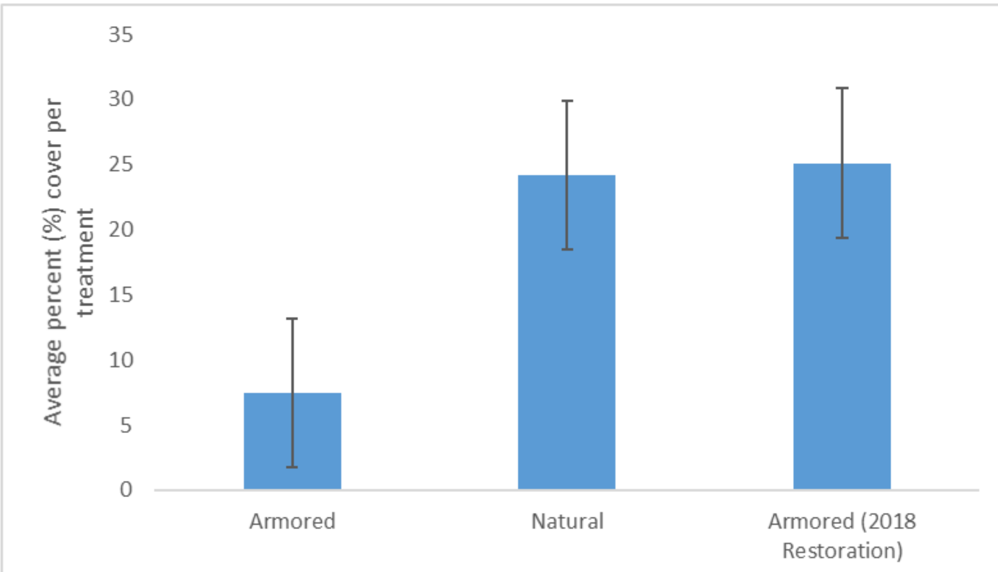


Figure 14: Percent cover of marine wrack per treatment.

Site as a basis for variation in marine wrack cover

There were no consistent trends in marine wrack cover when assessing the effects of site (Figure 15). The percent cover of marine wrack mirrors results of total wrack at each site and treatment, as total percent cover is dominated by marine components. In contrast to the total cover results, marine wrack cover was highest at the natural site (5.1 ± 3.26%) at Piner Point, although this result was not statistically significant (ANOVA $F_{(2, 27)} = 1.03$, p = 0.37)

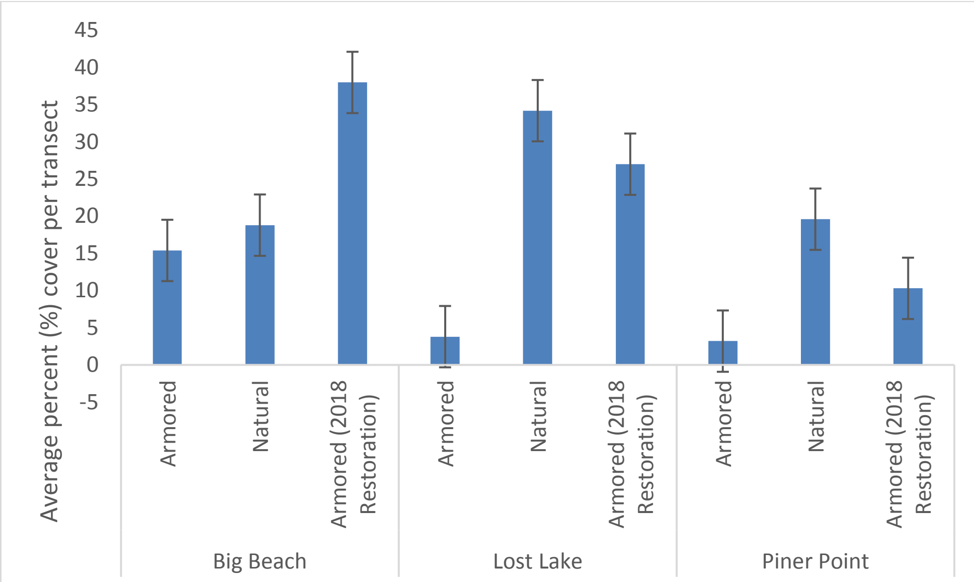


Figure 15: Percent cover of marine wrack per site.

Variation in terrestrial wrack percent cover based on treatment

When averaging across sites, the averaged terrestrial wrack percent cover varied across treatments (Kruskal-Wallis $\chi^2_{(2)}$ = 17.04, p= 0.0002) (Figure 16). The natural (5.60 ± 1.82%) had a higher average percent cover of terrestrial wrack compared to the armored (2.1 ± 1.99%) and pre-restoration (1.72 ± 0.62%) sites. The armored had significantly different medians than the pre-restoration site (Dunn test Z = 2.98, P = 0.0081) and the natural (Dunn test Z = 3.86, P = 0.0003). A Friedman's test showed no interaction between site and treatment ($\chi^2_{(2)}$ = 0.63, p = 0.73).

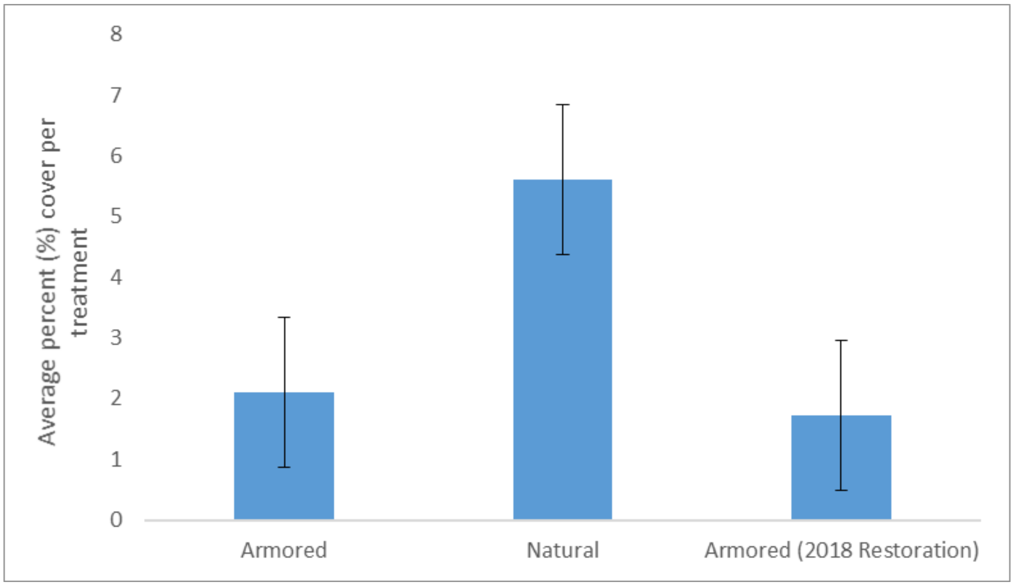


Figure 16: Percent cover of terrestrial wrack per treatment.

Site as a basis for variation in terrestrial wrack cover

Terrestrial wrack was higher at both Lost Lake and Piner Point sites (Figure 17). At Lost Lake, the terrestrial wrack was significantly different between natural (10.1 ± 4.5%) and armored (0.1 ± 0.1%) sites (Dunn test $Z = 3.19$, $P = 0.004$). At Piner Point, natural (6.2 ± 2.58%) treatments were significantly different than armored (0.2 ± 0.2%) (Dunn test $Z = 3.08$, $P = 0.006$). At Big Beach, there was a higher average of terrestrial wrack found at the armored site (6.00 ± 6.00%) as compared to the natural (0.5 ± 0.34%) and pre-restoration (0.9 ± 0.41%) sites, although there were no significant differences between treatments (Kruskal- Wallis $\chi^2_{(2)} = 2.24$, $P = 0.33$).

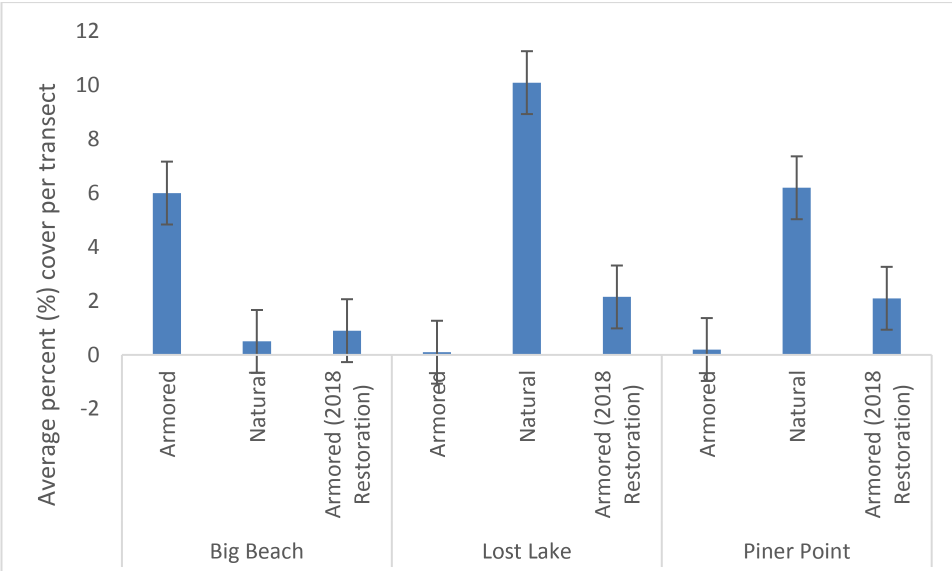


Figure 17: Percent cover terrestrial wrack per site.

Variations in eelgrass percent cover in wrack based on treatment

Eelgrass percent cover differed significantly across the three sites (Kruskal-Wallis χ^2 (2) = 35.09, p < 0.0001). Average eelgrass percent cover was highest at natural sites (3.81 ± 0.85%), compared to the pre-restoration (1.65 ± 0.45%) and armored sites (0.01 ± 0.00%). Eelgrass was highest at Lost Lake natural sites (5.81 ± 1.53%) (Figure 18). The percent cover of eelgrass differed significantly between the natural and armored treatments (Dunn test Z = 5.18, P < 0.0001). There was also a significant difference between eelgrass percent cover between the pre-restoration and armored (Dunn test Z = 5.05, P < 0.0001). A Friedman's test showed that there was no interaction between site and treatment (Friedman's test $\chi^2_{(2)}$ = 4.67, p = 0.097).

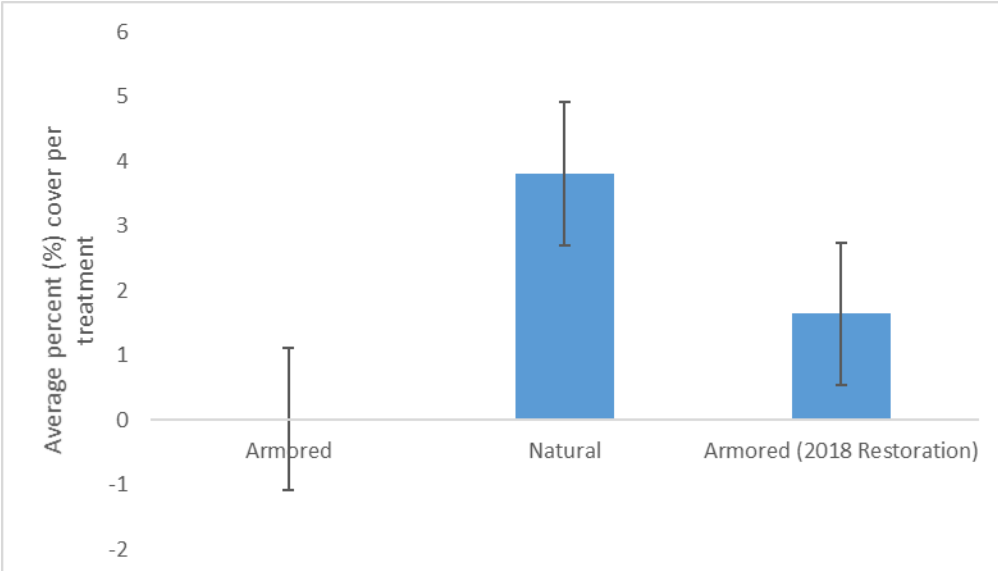


Figure 18: Percent cover of eelgrass per treatment.

Site as a basis for variation in eelgrass wrack cover

Overall, percent cover of eelgrass was higher at natural sites at both Lost Lake and Piner Point. At Lost Lake, eelgrass cover differed significantly at every treatment (Kruskal-Wallis $\chi^2_{(2)} = 17.89$, $P = 0.0001$). The natural site (5.8 ± 1.53%) had the highest average eelgrass cover, whereas the armored (0.02 ± 0.01%) pre-restoration (0.62 ± 0.26%) and sites had trace amounts (Figure 19).

At Piner Point, eelgrass cover differed significant differences between treatments (Kruskal- Wallis $\chi^2_{(2)} = 15.82$, $P = 0.0004$). Percent cover of eelgrass was significantly higher at the natural (4.8 ± 1.58%) than the armored (0.01 ± 0.01%) site (Dunn test $Z = 3.83$, $P = 0.0004$). Eelgrass at the pre-restoration site was also lower than at the natural site (2.2 ± 1.08), although results were not significant (Figure 19).

At Big Beach, eelgrass cover differed significantly between treatments (Kruskal-Wallis χ^2 (2) = 15.84, p=0.0004). The pre-restoration site had a significantly higher percent cover of eelgrass (2.1 ± 0.75%), compared to the natural (0.8 ± 0.8; Dunn test Z = 2.72, P = 0.018) and armored (0 ± 0.01; Dunn test Z = 3.63, P = 0.0008) sites (Figure 19).

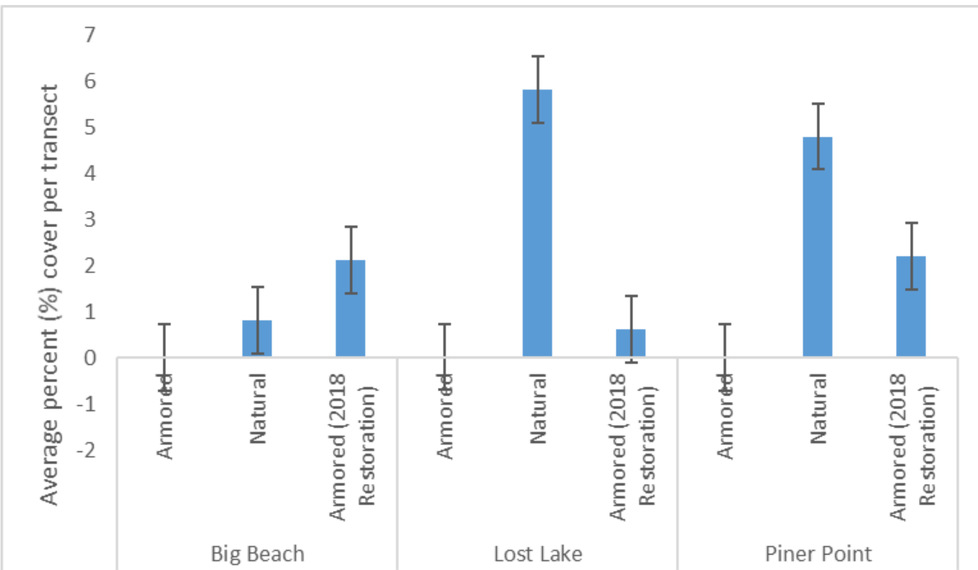


Figure 19: Percent cover of eelgrass wrack per site.

Terrestrial insects

Overall, there were no significant differences in density or taxa richness across treatments when averaged across sites. Diptera, or flies, dominated the percent composition of the samples. Natural treatments had the highest proportion of Diptera species (Table 1).

Table 1: Percent composition of terrestrial invertebrates per treatment.

Armored	**%**	**Natural**	**%**	**Armored (2018 Restoration)**	**%**
Diptera	37	Diptera	68	Diptera	43
Collembola	18	Hemiptera	9	Amphipoda	19
Hemiptera	13	Psocoptera	5	Collembola	8
Thysanoptera	10	Hymenoptera	4	Acari	6
Acari	7	Collembola	4	Psocoptera	5
Coleoptera	5	Thysanoptera	2	Hemiptera	5
Hymenoptera	4	Coleoptera	2	Coleoptera	4
Psocoptera	3	Acari	2	Hymenoptera	4
Aranae	1	Aranae	1	Thysanoptera	3
Trichoptera	1	Blattodea	1	Neuroptera	1
Lepidoptera	1	Neuroptera	1	Opiliones	1

Insect density by treatment

There were no statistically significant differences between insect density based on treatment (ANOVA $F_{(2, 42)} = 0.11$, $P = 0.89$). Average mean densities were similar across all treatment types, where natural ($345.90 \pm 108.97/m^2$) and pre-restoration ($336.73 \pm 77.76/m^2$) shorelines had similar insect densities, and densities at armored shorelines were slightly lower ($292.69 \pm 58.12/m^2$).

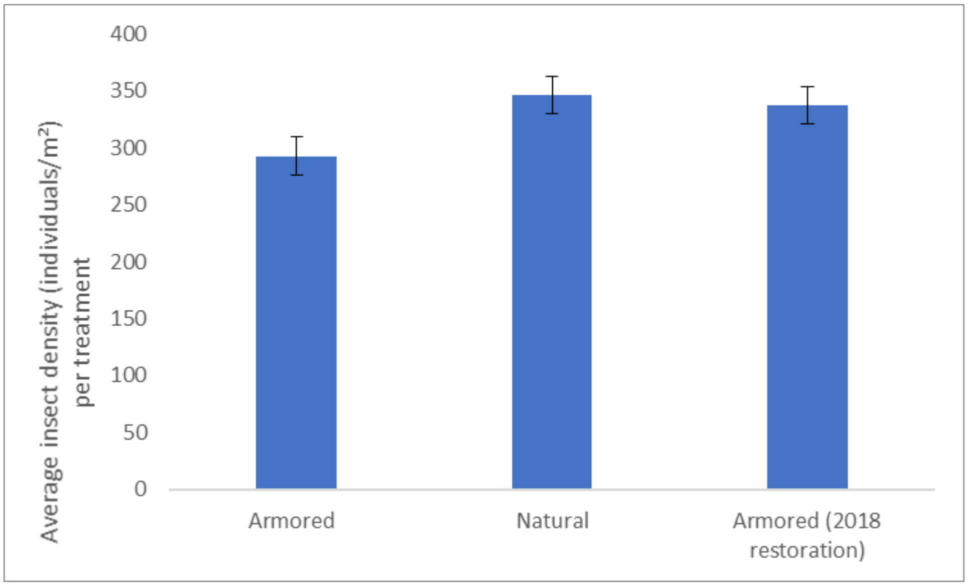


Figure 20: Average insect density (individuals/m²) per treatment.

Insect density per site

Insect density had a wide range, from 138-726 individuals/m² across all sites and treatments. The natural beach at Piner Point had the highest insect density (726.67/m²). There were no substantial trends in insect density when comparing individual sites (Figure 21).

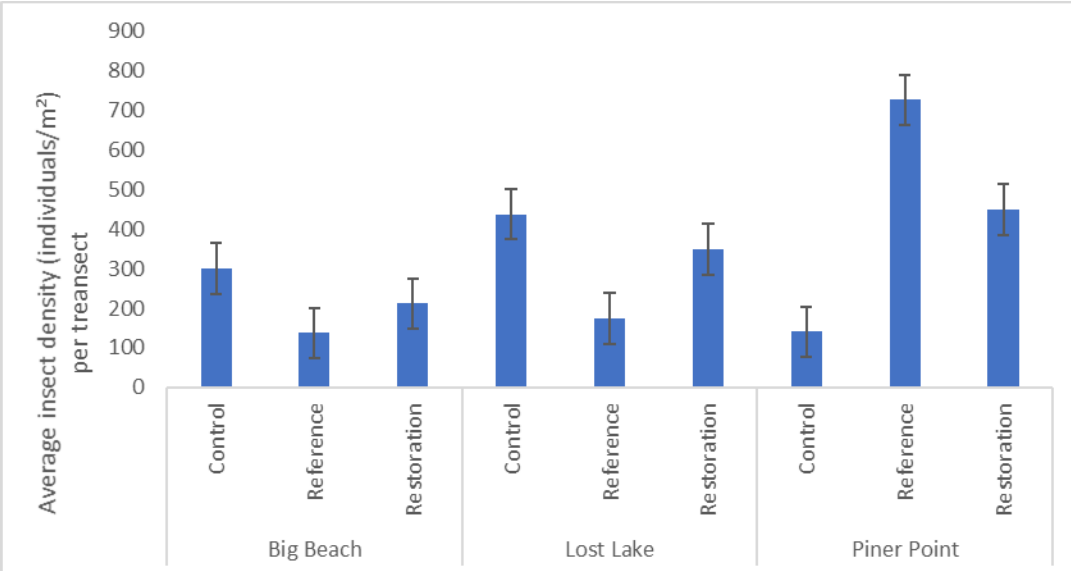


Figure 21: Average insect density (individuals/m²) per site.

Taxa richness per treatment

Taxa richness was averaged across all sites based on treatment (armored, natural, pre-restoration). Average taxa richness was similar at each treatment type (armored = 10.93 ± 1.07; natural = 9.8 ± 0.98; pre-restoration = 9.87 ± 1.02) (ANOVA $F_{(2)} = 0.38$, $P = 0.68$).

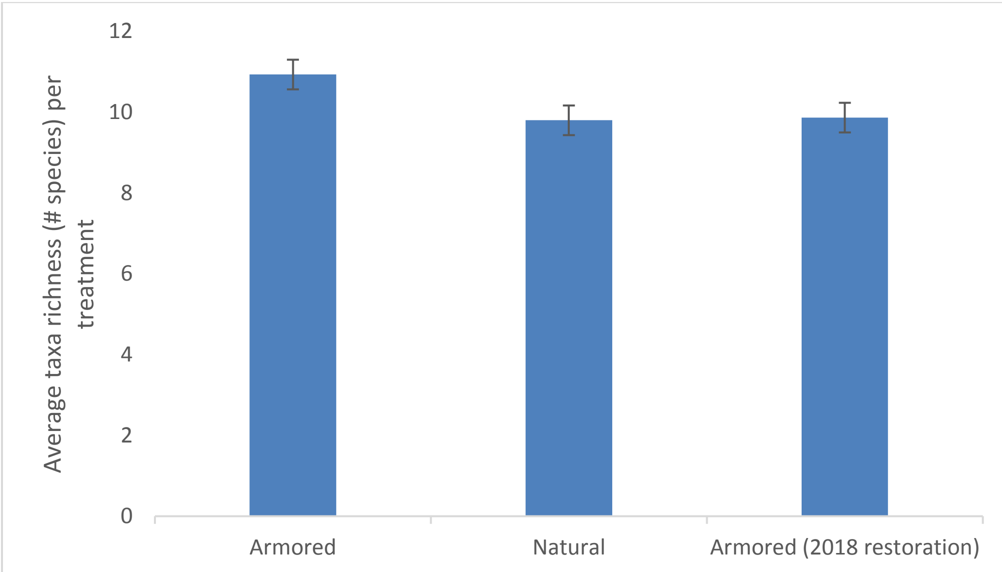


Figure 22: Average taxa richness per treatment.

Taxa richness per site

Taxa richness at each site and treatment was consistently high, ranging from approximately 30 to 70 insect species per treatment (Figure 23). There were no statistically significant patterns in taxa richness between treatments across the sites (ANOVA F(2) = 0.38, P = 0.68).

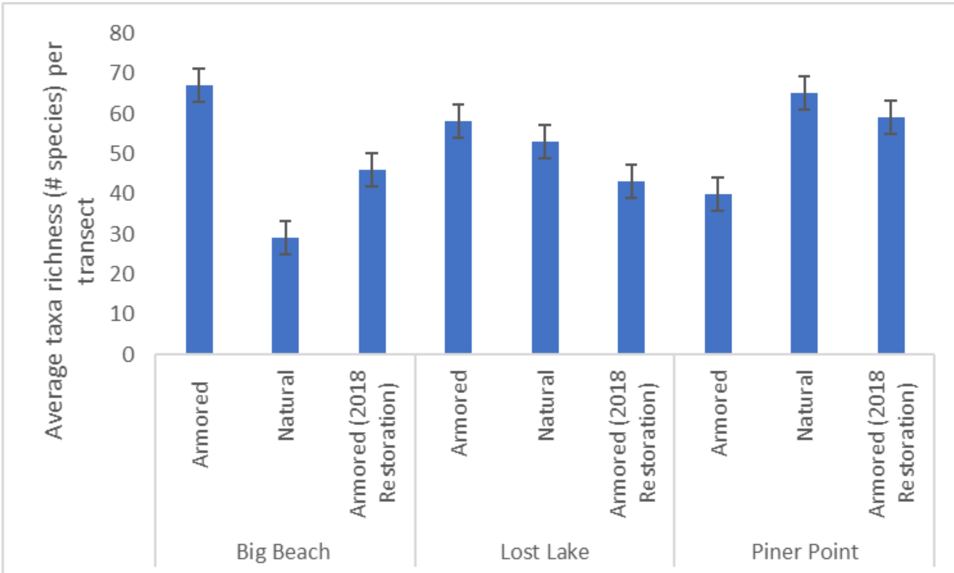


Figure 23: Average taxa richness per site.

Forage Fish

When all sites were averaged by treatment, surf smelt spawning occurred at each treatment type during the sampling window of December 2016-May 2017. On average, there were more surf smelt eggs at the pre-restoration treatment (110 eggs) than the armored (66 eggs) and natural (76 eggs) treatments (Figure 24). Sand lance spawning occurred at natural (350 eggs) and pre-restoration treatments (5 eggs), and was not present at armored treatments during this time frame (Figure 24).

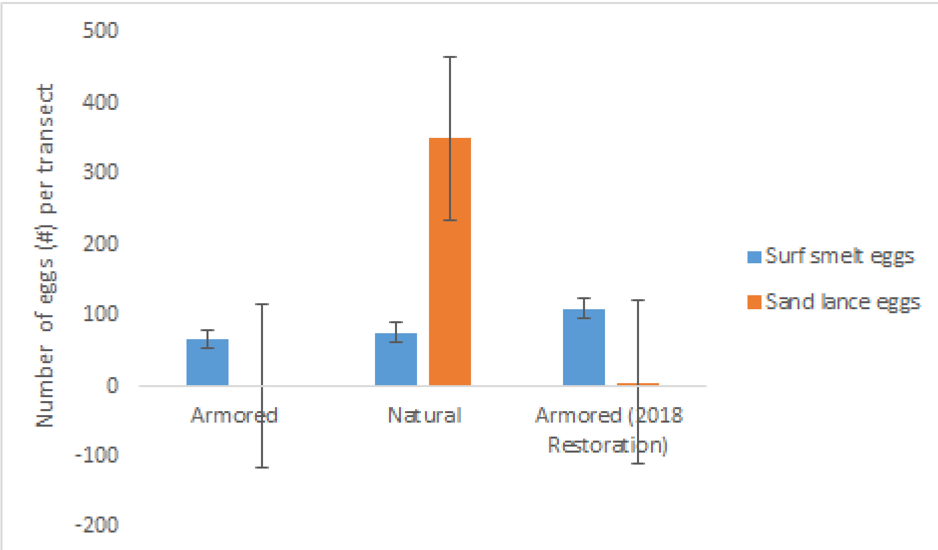


Figure 24: Average number of surf smelt and sand lance eggs per treatment.

Overall, there were more surf smelt spawning events than sand lance at all treatments, with the highest number of events at the natural sites (8 events). An event is defined as any egg found at the site (range from 1 to 350). There were more sand lance spawning events at the pre-restoration sites (3 events), although there were more sand lance eggs overall at the natural site.

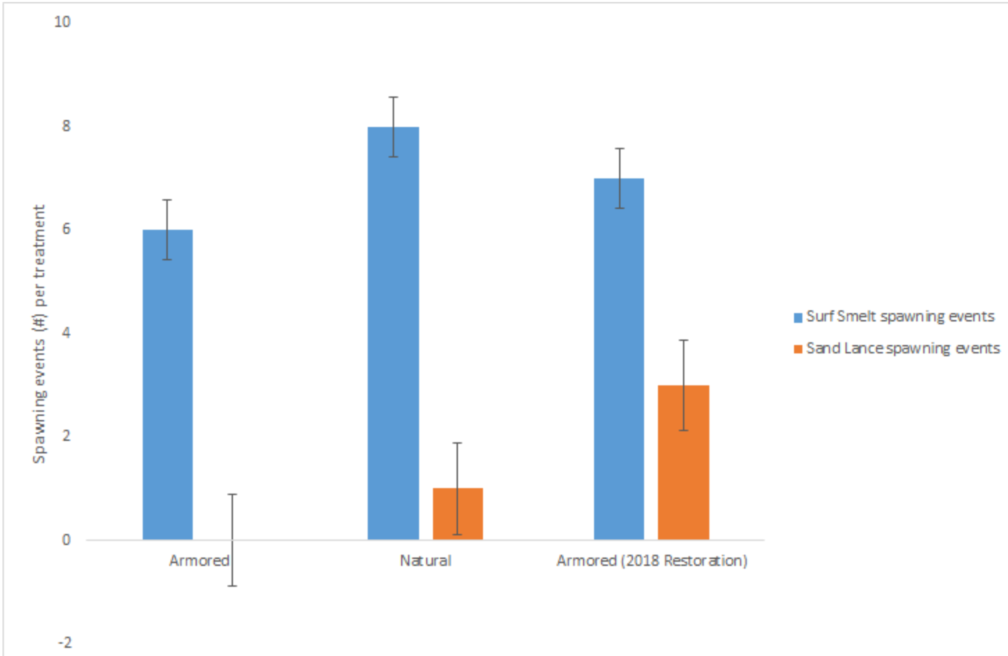


Figure 25: Number of surf smelt and sand lance spawning events per treatment during sampling timeframe (December 2016-May 2017).

When comparing each site and treatment individually, surf smelt spawning was more common than sand lance spawning across sites and treatments. Lost Lake had an overall higher number of surf smelt spawning events across all sites during this time. There was one large sand lance spawning event at the Piner Point natural site in December where around 350 eggs were found (Figure 26).

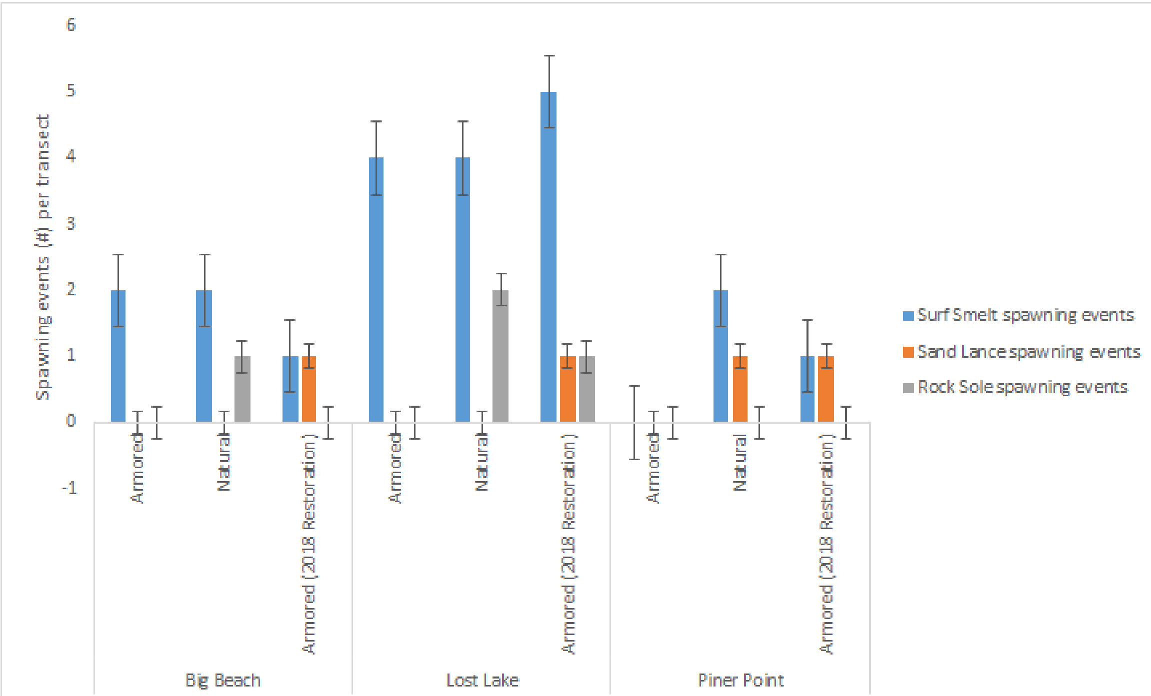


Figure 26: Number of surf smelt and sand lance spawning events per site during sampling timeframe (December 2016-May 2017).

Forage fish spawning at Big Beach

There were very few eggs found overall at treatments at the Big Beach site. In December, there were two surf smelt eggs found at the natural site. No eggs were found at other sites. There were slightly more surf smelt eggs found at the armored site (4 eggs) than the natural (1 egg) and pre-restoration (2 eggs) sites in February 2017. There were no eggs found at any of the treatments in April 2017.

Overall, there were few sand lance eggs found at Big Beach treatments. There were three sand lance egg found at the pre-restoration site in December 2016. There were no sand lance eggs found in February or April 2017 at any of the treatments (Figure 28).

Forage fish spawning at Lost Lake

Surf smelt spawned at all three treatments at Lost Lake in both January and May 2017. The only treatment without surf smelt spawning during these sampling events was the pre-restoration site in March 2017. The highest number of surf smelt eggs were found at the pre-restoration site in both January (64 eggs) and May (35 eggs) 2017 (Figure 29).

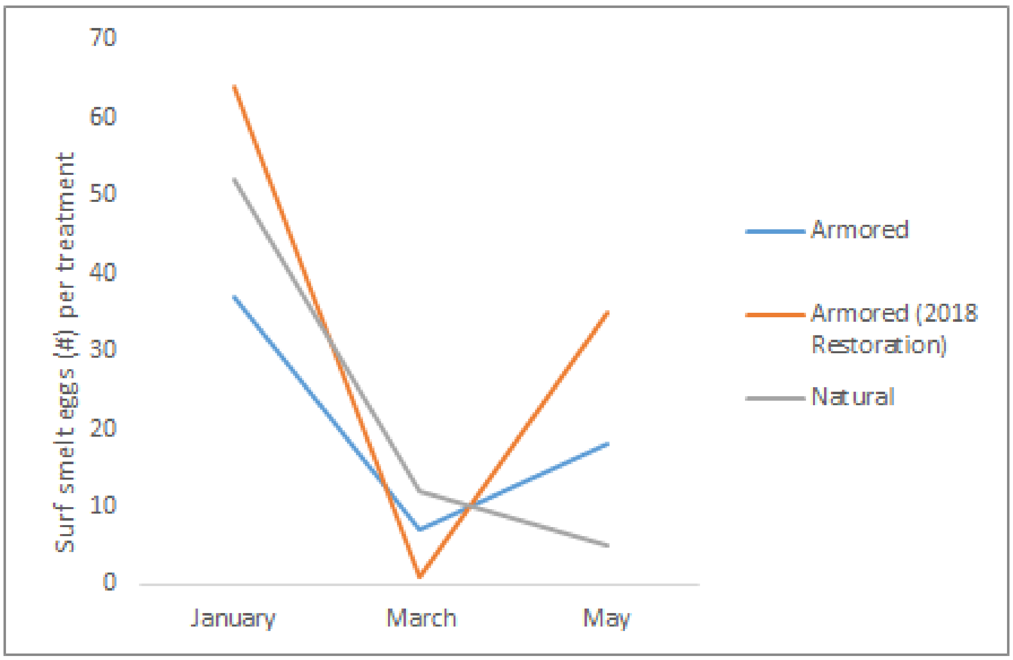


Figure 27: Number of surf smelt eggs per treatment at Lost Lake between January-May 2017.

Overall, there were few sand lance eggs found at Lost Lake treatments. There was one sand lance egg found at the pre-restoration site in January 2017. There were no eggs found in March or May 2017 at any of the treatments.

Forage fish spawning at Piner Point

At Piner Point, surf smelt spawning was only present in the winter (December 2016). In December, there were more surf smelt eggs found at the pre-restoration site (8 eggs) than at the natural site (4 eggs) and the armored site (none) in December 2016. There were no eggs found at any of the treatments in February or April 2017.

There were few sand lance spawning events across treatments at Piner Point. However, there were approximately 350 sand lance egg found at the natural site in December 2016. There were no eggs found at any of the treatments in February or April 2017.

Fish assemblages

The number of observations (how many individual times fish were spotted) were averaged across treatments. In general, there were more observations of fish on average at natural sites (3 fish) compared to the armored (1.33 fish) and pre-restoration (0.33 fish) sites (Figure 33).

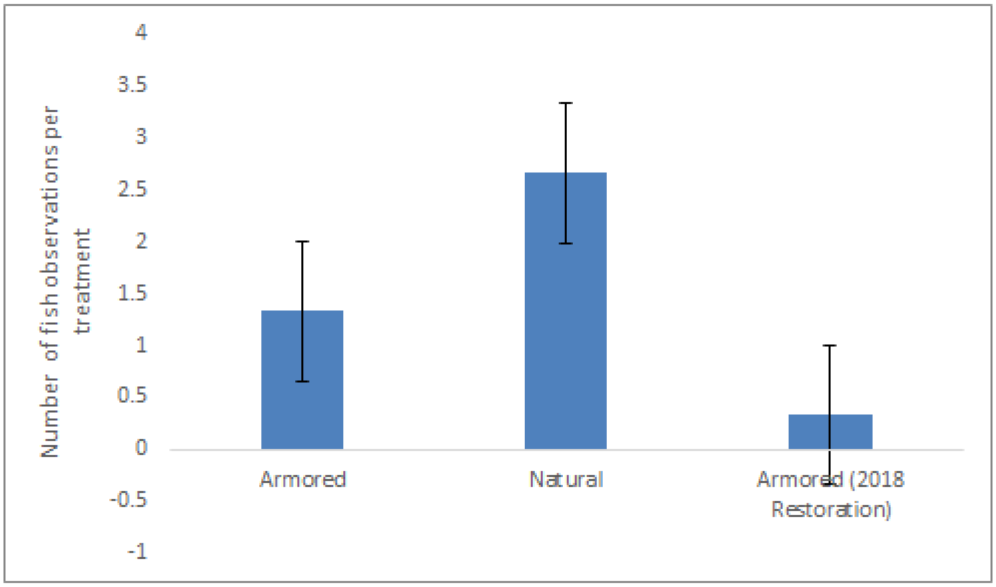


Figure 28: Number of fish observations per treatment.

Taxa richness of fish was averaged across treatments for all sites. Taxa richness was higher at natural sites (2.67 species) compared to armored (1.33 species) and pre-restoration sites (0.33 species) (Figure 34).

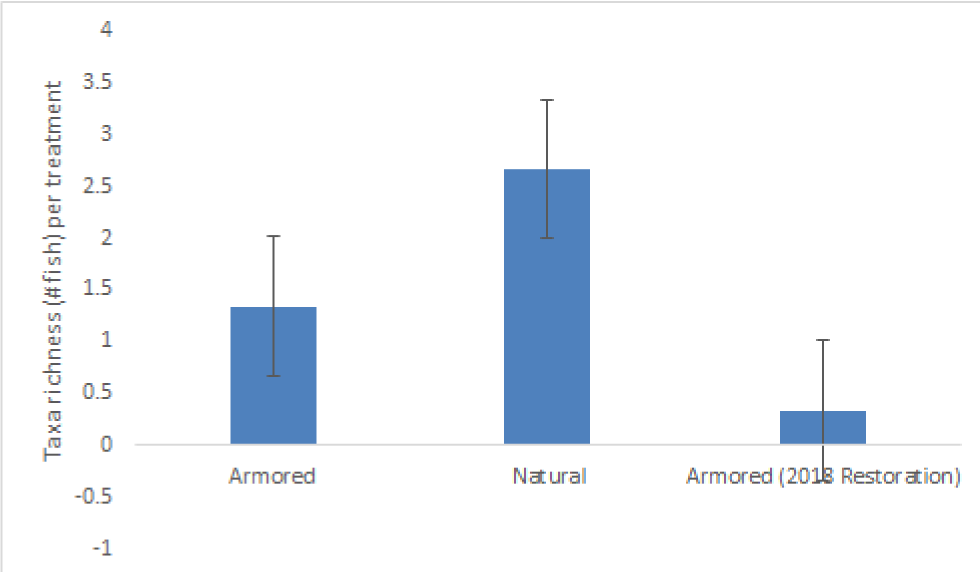


Figure 29: Taxa richness of fish per treatment.

The total number of fish were averaged across treatments for all sites. There was a higher average number of fish at natural sites (92.33 fish) compared to the armored (68.33 fish) and pre-restoration (0.33 fish) sites (Figure 35).

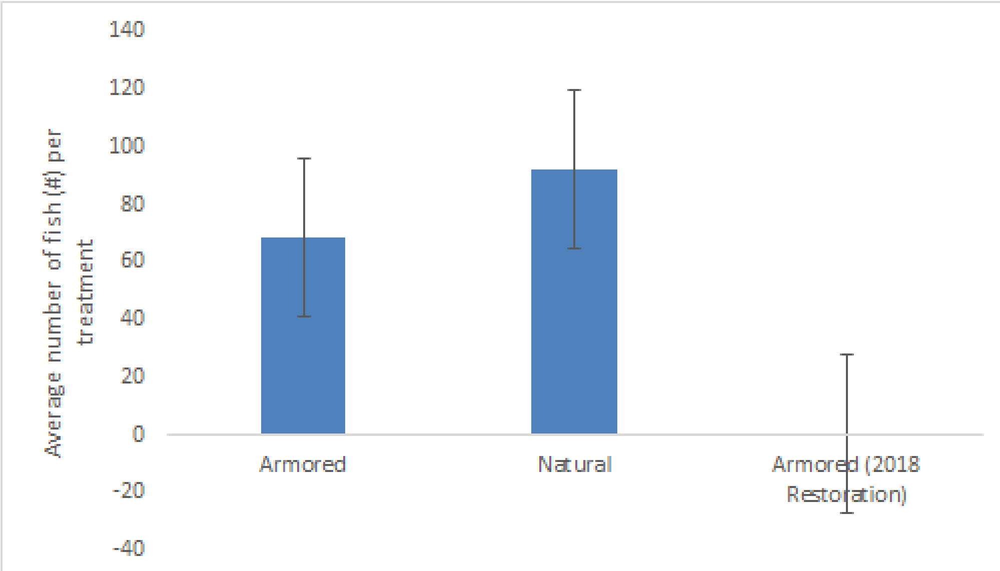


Figure 30: Total number of fish observations per treatment.

Fish Observations

At Big Beach, there were two species of fish observed at the natural site: unknown forage fish and anchovy (approximately 200 fish). There were no observations at the armored or pre-restoration sites (Table 2). At Lost Lake, there were more total observations, total fish, and number of species at the natural site (Table 3). At Piner Point, there were more total observations and number of fish species at the natural site. There were more overall fish (200 unknown species of forage fish) observed at the armored site (Table 4).

Table 2: Fish observations.

Site	Treatment	Total observations	Total fish	Number of species	Species type
Big Beach	Armored	0	0	0	N/A
Big Beach	Natural	2	101	2	Forage fish (unknown), anchovy
Big Beach	Armored (2018 Restoration)	0	0	0	N/A
Lost Lake	Armored	2	4	2	Sculpin, rock sol
Lost Lake	Natural	4	6	3	Shiner perch, sculpin, trout (unknown)
Lost Lake	Armored (2018 Restoration)	1	1	1	Saddleback gunnel
Piner Point	Armored	2	201	2	Sculpin, forage fish (unknown)
Piner Point	Natural	3	170	3	Surf smelt, salmon (unknown), forage fish (unknown)
Piner Point	Armored (2018 Restoration)	0	0	0	N/A

Chapter 5: Discussion

Shoreline armoring reduced complexity of the nearshore. Beaches with development had altered nearshore habitats compared to natural beaches. Changes in the nearshore can lead to altered biological response for fish by reducing vital habitat and prey availability for fish. The following discussion summarizes and provides context to the results found in this study.

Marine riparian vegetation

Overstory vegetation was higher at natural treatments compared to armored treatments. In addition, natural treatments had a higher average number of trees. Most of the trees at all sites were native. The percent of understory cover was similarly high at each shoreline type, around 70-90% cover across the treatment types.

The study sites in the MIAR had a relatively high amount of over and understory vegetation cover compared to studies in the literature. Vegetation conditions found in this study are not always typical of developed shorelines in the Puget Sound. Overall, the shorelines at these sites are relatively healthy compared to highly urban shorelines. The average plant health ratings across sites was four, and above three at all sites, where one is dead and five shows vigorous growth. Shoreline armoring is usually associated with the removal of over and understory vegetation and the replacement with maintained yards where grass lawns are more common (Heerhartz et al., 2014). This was not always the case at armored treatments, and especially the restoration sites where vegetation was overgrown.

The vegetation present at these sites, and in general, supports the vital connection between terrestrial and aquatic ecosystems. Terrestrial vegetation fosters habitat for insects and provides natural beach function, such as shading and moisture retention. More riparian vegetation contributes to the input of terrestrial insects in the nearshore (Toft et al., 2013). Terrestrial insects, such as dipterans (flies), can be carried by wind from terrestrial ecosystems onto the water surface and provide food for juvenile Chinook salmon (Munsch et al., 2016).

Maintaining shoreline vegetation and an intact upper-beach is necessary for full function of the supratidal zone. Previous studies have shown that vegetation removal, which is common at armored treatments, results in significant differences between backshore invertebrate and insect assemblages (Toft et al., 2014; Heerhartz et al., 2014). Introducing native riparian vegetation at armored shorelines or after armoring removal can improve the marine-terrestrial connectivity and may facilitate a rapid response from terrestrial macroinvertebrate assemblages, a vital part of Chinook diets and coastal food webs (Toft et al., 2014; Lee et al., 2018).

Wrack cover

Shoreline armoring is known to reduce the accumulation of wrack and logs on Puget Sound shorelines and reduces the relative proportion of terrestrial wrack (Heerhartz et al., 2014). A significantly lower amount of wrack was found at armored treatments compared to the pre-restoration treatments. This study demonstrated significant differences in overall wrack cover when comparing across treatments, although small sample sizes made for weaker statistical inferences. Similar amounts of wrack were found at the natural and pre-restoration treatments in this study.

Wrack was also analyzed by its composition (i.e. marine vs. terrestrial). Natural treatments had a higher proportion of terrestrial wrack (5.60 ± 1.82), when compared to armored (2.1 ± 1.99) and pre-restoration (1.72 ± 0.62) treatments, although this finding is not statistically significantly. Terrestrial inputs are important for nearshore ecosystems in providing nutrients and habitat for invertebrates in nearshore ecosystems (Heerhartz et al., 2014). The decrease of vegetation inputs from the uplands due to shoreline development can decrease the amount of terrestrial organic material that accumulates on the upper shore (Heerhartz et al., 2014). Terrestrial inputs are known to influence the abundance and composition of invertebrates in nearshore ecosystems (Heerhartz et al., 2015). Higher percent cover of terrestrial vegetation cover may be due to the increased terrestrial vegetation present at natural treatments, although further analysis is needed to assess this correlation.

Natural sites had a higher proportion of eelgrass wrack (3.81 ± 0.85). Eelgrass wrack differed significantly across all treatments. A Friedman's test showed that there was no interaction between site and treatment, meaning the significant difference between natural and armored treatments can be accepted regardless of site effects. Eelgrass was highest at Lost Lake natural treatments (5.8 ± 1.53), potentially reflecting the local abundance of seagrass meadows in the region (Perla, 2018). Further analysis could look at site as a factor, as local eelgrass beds and drift cell location may have an influence on eelgrass cover across sites.

The placement in the drift cell can affect the direction of drift (e.g. longshore currents, estuarine outflow) and can affect biological response variables. For example, drift cell placement of sites included in this study (Big Beach, Lost Lake, and Piner Point) may have caused some of the variation in accumulation of wrack. Total percent cover of wrack was higher at both Big Beach and Lost Lake, located in the same drift cell, compared to Piner Point, which is located within a diverging drift cell where net shore drift goes in either direction and less accumulation occurs (Washington State Coastal Atlas, 2018).

The overall accumulation and wrack composition measured in this study can be used as an indicator for healthy habitat for invertebrate populations. Wrack total percent cover was similar at natural sites (approximately 21% cover) and pre-restoration sites (approximately 25% cover) and lower at the armored sites (approximately 8% cover). This is comparable to findings from Heerhartz et al. (2014), where wrack composition was more diverse at natural sites and amounted to approximately 20% cover at natural sites compared to 10% percent cover at armored beaches during the summer months. Diverse organisms take advantage of the shelter and moisture provided by wrack and logs that accumulate on the beach (Heerhartz et al., 2014). Wrack provides habitat (moisture and shelter) for talitrid species, which are mobile scavengers that feed on wrack—especially algae—and other detritus (Heerhartz et al., 2015).

This study compared lower wrack lines, although upper wrack lines and accumulated logs were present at most natural treatments. As a result, the total percent cover of wrack reported in this study is likely not representative of the total percent cover on the actual beach. Future studies should measure total surface area of wrack in the field (combining upper and lower wrack lines) to get a more accurate representation of wrack cover on the beach and a better understanding of differences in wrack cover between armored and unarmored shorelines.

The results from this study represents a snapshot in time, as samples were collected during one tide at the site scale. Wrack is delivered to beaches after almost every high tide, and therefore increased sampling frequency and replication may illustrate temporal effects more thoroughly. In addition, sampling across seasons may show trends not apparent in this study. For example, increased wrack is common in the autumn as deciduous trees lose their leaves (Sobocinski, 2003).

Terrestrial insects

This study specifically assessed the differences in abundance and taxa richness of terrestrial insects, as terrestrial insects may be essential dietary components of fish throughout the Puget Sound (Lee et al., 2016). Terrestrial invertebrates can be used as a metric for habitat quality and as an indicator of available prey resources for salmon (Toft et al., 2013). Natural sites had a slightly higher insect density ($345.90 \pm 108.97/m^2$), although the results were not statistically significant compared to restoration ($336.73 \pm 77.76/m^2$) and armored ($292.69 \pm 58.12/m^2$) sites. However, when assessing each individual site, the natural treatment at Piner Point had a much higher insect density ($726.67\ m^2$) than all treatments at all sites. The abundance of insects seemed to show no correlation with the amount of overhanging vegetation. For example, overstory vegetation was highest at natural treatments, but insect abundance was similar across sites.

Taxa richness was slightly higher at armored treatments, although this result was not statistically significant. Natural sites were the least diverse in taxa richness. Diptera dominated samples at all shoreline types (armored, natural, and restoration). Diptera, Psocoptera, and Homoptera terrestrial invertebrate species have been found to dominate proportions of Chinook salmon diets (Munsch et al., 2016).

Higher insect abundance and taxa richness between natural versus armored treatments have been documented in previous studies (Sobosinski, 2003, Romanuk & Levings, 2003, Sobocinski et al., 2010, Toft et al., 2013, Lee at al., 2018). Insect abundance and taxa richness is known to be greater at sites with intact shoreline vegetation than at sites lacking vegetation (Sobosinski, 2003). The Olympic Sculpture Park case study demonstrated clearer results where both density and taxa richness were higher in areas where shoreline vegetation had been planted than areas that had none (Toft et al., 2013). The high amount of vegetation at all treatment types in the MIAR could be a factor in the similarities of taxa richness and density across shoreline types. Each treatment was 70-90 percent covered with understory vegetation. Overstory was highest at the natural treatments (about 80 percent), but was still relatively high (approximately 40 percent cover) at both the armored and restoration sites, so perhaps the differences in over and understory vegetation were not substantial enough to affect insect taxa richness and density.

This data represents a snapshot in time of shoreline conditions in the MIAR during the summer months. Further research with more data collection over a longer time may elucidate insect community response to treatment type and vegetation differences more clearly (Sobocinski, 2003).

Forage fish

Forage fish (surf smelt and sand lance) spawning occurred at each beach location across the sampling window of December 2016 to May 2017. Surf smelt spawning events were consistently high across treatment types, where the number of spawning events ranges from 6 to 8 across treatments (armored, natural, pre-restoration), where the most spawning events were found at natural sites (8 events). The highest amount of sand lance spawning events occurred at the pre-restoration treatments (3 events), compared to the natural (1 event) and the armored (no spawning events). When comparing the number of eggs across treatments, natural treatments had a higher average number of sand lance eggs, and pre-restoration treatments had the highest number of surf smelt eggs.

The effects of shoreline armoring on forage fish spawning is unclear from this study. Further analysis should continue to monitor forage fish spawning over longer periods of time. This may indicate long trends in preferential spawning locations. Comparing over long time scales would eliminate biases due to spawning seasonality, as preliminary results in this study suggest that sand lance tend to spawn in the winter, whereas surf smelt spawn all year round.

The effects of shoreline armoring may be clearer if the health of egg embryos, or the number of dead versus healthy eggs, is examined between armored and natural treatments. Shoreline armoring is known to decrease the overall health and survival of egg embryos (Rice, 2006). Forage fish egg survival depends on sediments, shade, wrack cover, and other debris characteristic of natural shorelines (Rice, 2006). Accounting for these site characteristics in future analyses may elucidate the effects of shoreline armoring on forage fish spawning and embryo survival as well.

Fish assemblages

Fish data collected from snorkel surveys was limited, as only a single survey was successful at each site. However, the study did indicate preliminary trends in fish use of the nearshore, where more observations of fish occurred at natural treatments. Taxa richness of fish was higher at natural treatments compared to the armored and pre-restoration treatments as well. Due to limited data, statistically significant trends of fish use or feeding behaviors were not established. Previous findings from snorkel surveys in the Puget Sound demonstrate that shoreline armoring alters fish distribution and prey availability in shallow water habitats (Toft et al., 2007). Development along the shoreline is likely to change the character of overhanging vegetation and insects, as demonstrated in this study, which can have cascading effects on fish habitat and prey resource availability.

Understanding of fish behavior and habitat use in the nearshore is limited (Munsch et al., 2016). Nearshore fish communities are typically studied using physical capture (e.g. netting) rather than observing behavior directly. As a consequence, it is difficult to connect their fine-scale habitat use and behavior to basic ecological theory (Munsch et al., 2016). Snorkel surveys allow an observational where fish can be observed behaving more naturally compared to beach sein and netting techniques (Munsch et al., 2016).

Previous studies using these snorkel survey methods have demonstrated clearer results between armoring and natural shorelines, where natural shorelines demonstrated an increase in fish abundance and feeding behavior (Toft et al., 2007; Toft et al., 2013, Heerhartz et al., 2015; Munsch et al., 2016). These studies were conducted over longer periods of time and were located in highly urban environments where an intermittent natural beach may have a greater impact.

Overall, this study provides insight into the localized effects of armoring on fish communities in the MIAR. This study fine-tuned snorkel survey methods that can be used by the VNC to continue to monitor fish across time in the MIAR.

Suggestions for future fish surveys

Fish observations occurred at different sites at different months (Big Beach and Lost Lake in July, Piner Point in August) and captures only a snapshot in time. Fish data is not directly comparable between sites due to differences in fish migration patterns across time. The peak out-migration of juvenile salmon is around June and July, although juvenile Chinook are found along Puget Sound shorelines from late January through September (Shoreline Monitoring Toolbox, 2012). Only one school of salmon was observed over the three surveys, where a school of about 20 juvenile pink salmon were spotted at Piner Point in August 2017. Forage fish (either surf smelt or unknown species) dominated the observations (both individually and in large schools) at Big Beach in July 2017 and Piner Point in August 2017. A few observations of large schools of forage fish resulted in the large number of fish observations at these sites in particular. Surf smelt are known to spawn at various times of the year, whereas Pacific sand lance spawn during winter months (Rice, 2006).

The differences in physical space between sites could account for some of the differences in fish use. For example, Big Beach and Lost Lake are located within Quartermaster Harbor, whereas Piner Point is more exposed to deeper waters and faster currents found in Dalco and East Passages (Washington State Coastal Atlas, 2018). Many factors can affect the distribution of salmon and other fishes including, but not limited to, proximity to freshwater and out-migration corridors, predation risk, and water depth, all of which may differ between sites included in this study (Toft et al., 2007).

Due to differences in fish use of nearshore habitats across space and time, future studies should survey all three sites in the same month, and preferably in the same day. Again, the peak out-migration of juvenile salmon is around June and July, although juvenile Chinook are found along Puget Sound shorelines from late January through September (Shoreline Monitoring Toolbox, 2012). Future studies may focus on sampling between June and July to better document the effects of armoring on juvenile salmon use in the nearshore. Lastly, surveys were conducted during daylight hours only, which may not account for all fish use and behavior, although night surveys are not recommended with volunteers due to safety concerns.

The most favorable conditions for snorkel surveys occurred during the highest, high tide of the day, during the evening, where high tides reached the bulkhead and calmer conditions were more common. Fish are known to be more active during the evening time, according to local fishers in the area. The highest, high tide occurred in the evening occasionally during the summer, which made for more ideal sampling conditions. Conditions were not favorable for sampling in the middle of the day (between 10am and 4pm) during the lower-high tide. Wind, increased turbidity, and sunlight decreased visibility during the mid-day. In addition, the lower-high tide did not always reach the bulkhead, which could potentially lessen the effect of armoring on fish because the tide does not reach the bulkhead during this time.

Snorkelers experienced very limited visibility in the water column during all surveys, where visibility ranged from 1.5 to 2 meters. The Shoreline Monitoring Toolbox's fish survey methods advise a 2.5 meter visibility, which was never achieved during these surveys. Surveys scheduled during earlier months (May-June) were not conducted due to low visibility (less than one meter). Future studies should ensure at least a 1.5 to 2 meter visibility before proceeding to sample.

Lastly, future analysis should compare fish data to nearshore habitat parameters measured in this study. As more fish data is collected at these sites, comparing fish data to habitat parameters (i.e. canopy cover and terrestrial insect data) may increase the understanding of fish response to nearshore habitat conditions.

Study design and suggestions for future research

Beaches in the MIAR are unique

Sampling occurred over a small geographic region in Quartermaster Harbor between Vashon Island and Maury Island in the Puget Sound, and therefore, the results may not be directly comparable to shorelines across the Puget Sound. Shorelines in the Maury Island Aquatic Reserve tend have intermittent development, but have some natural beaches along with armored beaches that may still have beneficial qualities of a natural shoreline (i.e. over and understory vegetation, wrack accumulation). Studies have focused on shorelines that are significantly more modified than elsewhere in Puget Sound, such as in Elliott Bay in Seattle, where about 90% of the shoreline is modified by retaining structures (Weitkamp et al. 2000) compared to one-third for the rest of the Puget Sound (Puget Sound Partnership, 2012). The results of this study may not show as drastic results as shorelines in highly developed areas. That said, the results of this study demonstrate that shoreline development disrupts marine-terrestrial connectivity and alters important habitat for fish, which can be widely applied to shorelines across the Puget Sound.

The paired design in this study helps control for variability in environmental parameters and provides the ability to test for armoring-related differences. However, differences between armored and unarmored shorelines were not always clear. Armored (both the armored and pre-restoration) treatments have different levels of development and yard maintenance (i.e. grass vs. overgrown understory vegetation) that could cause differences in response variables. For example, the armored treatments are owned and maintained by property owners, whereas restoration sites that have been purchased by King County have not been maintained for years (Perla, 2018). This may suggest that the condition of the habitat (presence of over and understory vegetation) along shorelines may provide habitat benefits, even when shoreline armoring is present.

Localized geomorphology in the MIAR

Unlike many shoreline restoration removal projects in the literature, the bulkheads being removed in MIAR are along actively eroding, high-bank shorelines. The geomorphology at these sites is distinct from other studies and bulkhead removals that have been done on low-bank waterfront. More immediate changes are expected after bulkhead removals are completed (Perla & Metler, 2016). The placement of each treatment in the drift cell should be considered when examining response variables, especially after restoration. The placement in the drift cell can affect the direction of drift (e.g. longshore currents, estuarine outflow) and can affect biological response variables. For example, drift cell placement of sites included in this study (Big Beach, Lost Lake, and Piner Point) may have caused some of the variation in accumulation of wrack. Total percent cover of wrack was higher at both Big Beach and Lost Lake, located in the same drift cell, compared to Piner Point, which is located within a diverging drift cell where net shore drift goes in either direction and less accumulation occurs (Washington State Coastal Atlas, 2018).

In contrast to many shoreline armoring removal studies in highly urban areas with severe loss of beach, sites in the MIAR are characteristic of shallow, low-grade beaches that provide shallow water habitat for fish. Fish are known to prefer shallow water areas as a nursey habitat and refuge from prey during juvenile life stages (Ruiz et al., 1993). Larval forage fish are known to use shallow water and beaches as nursey grounds as well (Pentilla, 2007).

Analysis did not include beach profile data that was collected in 2016 and 2017. Future studies can look at the changes in beach profile over the different summers to analyze the effects of the shoreline armoring and shoreline restoration. In addition, shoreline armoring elevation on the beach was not examined in this study. Lower elevations of shoreline armoring negatively affect beach parameters on local and larger spatial scales (Dethier et al., 2016). Armoring placed below MHHW has substantially more effect on parameters than armoring higher on the beach (Dethier et al., 2016). Future analysis in the MIAR should account for shoreline armoring elevation, as restoring sites with armoring lower on the beach may be more beneficial to restoring natural physical and biological conditions.

Citizen science: a challenge and a resource

This study was conducted by the VNC with the help of citizen-science volunteers. This study highlights the importance of citizen science monitoring and the standardized protocols using Puget Sound Partnership's Shoreline Monitoring Toolbox. Citizen science projects can enhance scientific research and assist government agencies with restoration monitoring for minimized costs and effort (Perla, 2018). Results from this project established a baseline dataset of pre-restoration shoreline conditions that will assist King County in understanding the benefits of shoreline restoration. Citizen science research can also help to meet community involvement and outreach goals established for many county and state governments. Participation from citizen science volunteers is beneficial for many reasons, including the understanding of shoreline issues by local citizen scientists, fostering community ambassadors for shoreline health, and increasing community buy-in (Perla, 2018). Monitoring in the MIAR is an effective way of involving and educating the local community in restoration projects along the Vashon and Maury Island shorelines.

Citizen science projects are often organized by volunteer groups or nonprofits that often lack funding and resources for long term monitoring. The design of this study is limited due to the nature of citizen science involvement. The study lacks replication across time and sample sizes are low. Each beach was sampled one time during the summer, as it is difficult to train and organize large groups of volunteers with limited staff capacity at the VNC. Measurements may be inherently biased between individuals for each study. Different volunteers were involved across the duration of this project, and therefore some inaccuracy and lack of objectivity may be an issue.

That said, this project demonstrates a strong example of citizen science involvement in shoreline restoration monitoring projects across the Puget Sound. The data collected in this study would not have existed without the help and support of citizen science volunteers. The standardized methods from the Shoreline Monitoring Toolbox are intended to be used by nonprofits and citizen science groups across the Puget Sound.

Working with small sample sizes

One of the challenges in this analysis is small sample sizes for vegetation, wrack, and fish parameters. Small sample sizes can increase error rate and potentially distort the response interpretations, so the results cannot necessarily be generalized to other shorelines across the Puget Sound (Lee et al., 2018). Ideally, future studies would sample more often over the summer months and eventually across time to look at long term trends with larger sample sizes. Increased sample sizes over time is essential for fully understanding the effects of armoring (Lee et al., 2018). Coordinating monitoring projects with citizen scientists can be difficult due to lack of time and resources from small nonprofits and local volunteers. If the VNC wants more conclusive answers about a certain beach parameter, the non-profit may choose to focus on one beach parameter and sample more often. As this study evolves, the VNC may consider coordinating with a small group of volunteers to take responsibility for sampling a certain beach parameter throughout the year. For example, forage fish beach spawning surveys are currently monitored throughout the year by trained citizen scientists with little coordination needs from the VNC.

Suggestions for planning, management, and restoration

Existing and new shoreline management policies should encourage homeowners and stakeholders to protect natural shorelines and embrace shoreline restoration when it can simultaneously protect properties and biodiversity. It is critical for policymakers to consider the benefits of shoreline armoring removal before undertaking new shoreline development (Lee et al., 2018). When artificial barriers are removed and aquatic habitats merged with terrestrial habitats, biological and physical processes may be reconnected and allowed to function more naturally (Toft et al., 2013).

The impacts of shoreline armoring may be reversible, as seen in previous studies of shoreline restoration (Toft et al., 2014; Dethier et al., 2016). Although few studies have assessed the effectiveness of armoring removal on restoring coastal ecosystems, studies generally demonstrate that shorelines without armoring can host higher abundance and diversity (Toft et al., 2014; Heerhartz et al., 2014; Dethier et al., 2016; Lee et al., 2018). Maintaining marine-terrestrial linkages should be a top priority for shoreline restoration, current homeowners, and future development where armoring is necessary (Heerhartz et al., 2015).

Removal of armoring is not always feasible. Recently, alternatives to shoreline armoring, including armoring removal, have emerged that can both protect coastal infrastructure and restore ecological health (Gittman et al., 2016). Shorelines without armoring can provide the same function as natural erosional barriers (Lee et al., 2018). If engineered correctly, employing techniques that can stabilize the shoreline by mimicking site-specific shoreline processes, restoration may still provide the benefits of coastal protection (Guerry et al., 2012; Toft et al., 2013). For example, large woody debris protects from beach erosion, but also enhances wrack accumulation and improves aquatic-terrestrial connectivity (Heerhartz et al., 2014).

Continuing to monitor post-restoration in the MIAR is essential to assess the recovery of restored coastal ecosystems. Few studies have assessed the effectiveness of armoring removal on restoring coastal ecosystems (Lee et al., 2018). Increasing the geographical scope and number of studies of these coastal biota types can increase knowledge of restoration effectiveness to help inform management policies.

Chapter 6: Conclusion

The ecological impact of large-scale shoreline armoring on nearshore ecosystems is still largely unknown, though this study demonstrated some proximal effects of shoreline armoring in the MIAR. Overall, armoring can affect biological conditions of shorelines on a local scale. The results of this research demonstrate a baseline of environmental conditions in the MIAR and the localized effects of shoreline armoring. Understanding the current condition of these beaches is necessary to determine how coastal biota will respond to shoreline armoring removal and restoration occurring in the summer of 2018.

This study showed results from studies from five different shoreline parameters including vegetation cover, wrack cover, insect density and taxa richness, forage fish beach spawning, and fish observational studies. This study indicated noticeable differences between shoreline types (armored, natural, and pre-restoration treatments). The following is a summary of results:

1. Vegetation: The presence of armoring decreased the percent cover of overstory vegetation and trees compared to natural beaches. Similar percent cover of understory vegetation was found across treatments (armored, natural, pre-restoration). Natural beaches had more native species and a higher plant health index.
2. Wrack: Pre-restoration sites had a higher abundance of total wrack cover. Natural beaches had a higher abundance of terrestrial and eelgrass wrack cover. Upper wrack lines and logs were found at natural beaches.

3. Insects: Insect diversity and taxa richness were similar across treatments. Percent composition of samples were dominated by Diptera, or flies, at all sites, but was greatest at natural beaches.
4. Forage fish: Surf smelt and sand lance spawning occurred at all treatment types. Natural beaches had a higher number of sand lance eggs. Sand lance tended to spawn in the winter, whereas surf smelt spawning was present throughout the sampling time frame (December 2016 to May 2017).
5. Fish: Natural sites had a higher taxa richness, number of individual fish sightings, and number of individual fish.

Shoreline armoring clearly disrupts the connection between terrestrial-aquatic ecosystems, as demonstrated by the reduction in backshore vegetation and terrestrial-associated wrack cover. Terrestrial-aquatic connection along the shoreline provides vital habitat functions for invertebrates and fish that utilize the shorelines. Overall, this study demonstrates that shoreline development changes the biological response to nearshore habitats by reducing the marine-terrestrial connectivity which provides vital habitat for invertebrates and fish. The effects of armoring may be minimized if shorelines mimic natural conditions, as seen in the Olympic Sculpture Park example (Toft et al., 2013), by ensuring vital nearshore parameters are maintained and marine-terrestrial ecosystems remain connected. Shoreline restoration and shoreline armoring removal is effective in improving the health and productivity of coastal ecosystems, and will continue to be monitored at the three sites in the MIAR.

References

Bilkovic, D.M, Roggero, M.M. (2008). Effects of coastal development on nearshore estuarine nekton communities. *Marine Ecology Progress Series*, 358:27-39.

Booth, B., Legg, N. (2017). Technical Memorandum: Technical Review Memorandum, Maury Island Aquatic Reserve Armoring Removal Project (E00376-20). Prepared by Cardno, Seattle, WA for the Watershed Stewardship Program, King County, WA, USA.

Bozek C.M., Burdick D.M. (2005). Impacts of seawalls on saltmarsh plant communities in the Great Bay Estuary, New Hampshire USA. *Wetlands Ecology and Management*, 13(5):553-568

Brennan, J.S., and H. Culverwell. (2004). Marine Riparian: An Assessment of Riparian Functions in Marine Ecosystems. Published by Washington Sea Grant Program, UW Board of Regents, Seattle, WA.

Brennan, J.S. (2007). Marine Riparian Vegetation Communities of Puget Sound. Puget Sound Nearshore Partnership Report No. 2007-02. Published by Seattle District, U.S. Army Corps of Engineers, Seattle, Washington.

Brennan, J., Culverwell, H., Gregg, R., Granger, P. (2009). Protection of Marine Riparian Functions in Puget Sound, Washington. Published by Washington Sea Grant Program, UW Board of Regents Seattle. 139 pp.

Brennan, J.S., Higgins, K.F., Cordell, J.R., Stamatiou, V.A. (2004). Juvenile Salmon Composition, Timing, Distribution, and Diet in Marine Nearshore Waters of Central Puget Sound in 2001-2002. King County Department of Natural Resources and Parks, Seattle, WA. 164 pp.

Bulleri F., Chapman M.G., Underwood A.J. (2005). Intertidal assemblages on seawalls and vertical rocky shores in Sydney Harbour, Australia. *Austral Ecology*, 30(6):655-667.

Chapman, M.G. (2003). Paucity of mobile species on constructed seawalls: effects of urbanization on biodiversity. *Marine Ecology Progress Series*, 264, 21-29.

Chapman, M.G., Underwood, A.J. (2011). Evaluation of ecological engineering of ‘armoured’ shorelines to improve their value as habitat. *Journal of Experimental Marine Biology and Ecology*, 400(1–2):302-313.

Conrad, C.C., Hilchey, K.G., (2011). A review of citizen science and community-based environmental monitoring: Issues and opportunities. *Environmental Monitoring and Assessment*, 176(1-4): 273-91.

Coyle, J.M., M.N. Dethier (2010). Review of shoreline armoring literature, in Shipman, H, Dethier, M.N., Gelfenbaum, G., Fresh, K.L., Dinicola, R.S. (2010). Puget

Sound Shorelines and the Impacts of Armoring - Proceedings of a State of the Science Workshop, May 2009: U.S. Geological Survey Scientific Investigations Report 2010-5254, pp. 245-265.

Davis J.L.D., Levin L.A., Walther S.M. (2002). Artificial armored shorelines: sites for open-coast species in a southern California Bay. *Marine Biology,* 140(6):1249-1262

Desbonnet, A., P. Pogue, V. Lee, and N. Wolff. 1994. Vegetated buffers in the coastal Zone – A summary review and bibliography. Coastal Resources Center Technical Report No. 2064. University of Rhode Island Graduate school of Oceanography. Narragansett, R I 02822. 72 pp.

Dethier, M.N., Raymond, W.W., McBride, A.N., Toft, J.D., Cordell, J.R., Ogston, A.S., Berry, H.D. (2016). Multiscale impacts of armoring on Salish Sea shorelines: Evidence for cumulative and threshold effects. *Estuarine, Coastal and Shelf Science*, 175, 106–117.

Department of Ecology (Ecology, 2016). Shoreline Management: Introduction to the Shoreline Management Act. Accessed February 24, 2016 from http://www.ecy.wa.gov/programs/sea/sma/st_guide/intro.html

Doody, J.P., 2013. Coastal squeeze and managed realignment in southeast England, does it tell us anything about the future? *Ocean and Coastal Management*, 79, 34e41.

Duffy, E.J. (2003). Early marine distribution and trophic interactions of juvenile salmon in Puget Sound (Doctoral dissertation, University of Washington).

Duffy, E.J., Beauchamp, D.A., Sweeting R.M., Beamish, R.J., Brennan, J.S. (2010). Ontogenetic diet shifts of juvenile Chinook salmon in nearshore and offshore habitats of Puget Sound. *Trans. Am. Fish Soc.*, 139: 803-823.

Duffy E. J., Beauchamp D.A. (2011). Rapid growth in the early marine period improves the marine survival of Chinook salmon (*Oncorhynchus tshawytscha*) in Puget Sound, WA. *Can. J. Fish Aquat. Sci.*, 68:232-240.

Dugan, J.E., Hubbard, D.M., Rodil, I.F., Revell, D.L., Schroeter, S. (2008). Ecological effects of coastal armoring on sandy beaches. *Marine Ecology*, 29(s1):160-170.

Dunagan, C. (2017, October 22). With sea-level rise, waterfront owners confront their options. Retrieved March 21, 2018, from https://www.eopugetsound.org/magazine/is/slr.

Fresh, K.L. (2006). Juvenile Pacific Salmon in Puget Sound. *Puget Sound Nearshore Partnership Report No. 2006-06*. Published by Seattle District, U.S. Army Corps of Engineers, Seattle, Washington.

Guerry, A.D., Ruckelshaus, M.H., Arkema, K.K., Bernhardt, J.R., Guannel, G., Kim, C.-K., Marsik, M., Papenfus, M., Toft, J.E., Verutes, G., et al. (2012). Modeling benefits from nature: using ecosystem services to inform coastal and marine spatial planning. *Int. J. Bio. Sc., Eco. Serv. Man*., 1–15 (19 more authors).

Gittman, R.K., Fodrie, F.J., Popowich, A.M., Keller, D.A., Bruno, J.F., Currin, C.A., Peterson, C.H., Piehler, M.F. (2015). Engineering away our natural defenses: an analysis of shoreline hardening in the US. *Frontiers in Ecology and Environment*, 13(6):301-307.

Gittman, R.K., Peterson, C.H., Currin, C.A., Fodrie, F.J., Piehler, M.F., Bruno, J.F. (2016). Living shorelines can enhance the nursery role of threatened estuarine habitats. *Ecological Applications*, 26(1):249-263.

Guttman, E.S. (2009). Characterizing the Backshore Vegetation of Puget Sound (Unpublished master's thesis). Prescott College.

Hart, C. (2010). Marine Shoreline Armoring and Puget Sound (10-06-003). Washington State Department of Ecology, Shorelands and Environmental Assistance Program.

Heatherington, C., Bishop, M.J. (2012). Spatial variation in the structure of mangrove forests with respect to seawalls. *Marine and Freshwater Research*, 63(10):926-933.

Heerhartz, S.M., Toft, J.D., Cordell, J.R., Dethier, M.N., & Ogston, A.S. (2016). Shoreline Armoring in an Estuary Constrains Wrack-Associated Invertebrate Communities. *Estuaries and Coasts*, 39(1), 171–188.

Heerhartz, S.M., & Toft, J.D. (2015). Movement patterns and feeding behavior of juvenile salmon (Oncorhynchus spp.) along armored and unarmored estuarine shorelines. *Environmental Biology of Fishe*s, 98(6), 1501–1511.

Heerhartz, S.M., Dethier, M.N., Toft, J.D., Cordell, J.R., & Ogston, A.S. (2014). Effects of Shoreline Armoring on Beach Wrack Subsidies to the Nearshore Ecotone in an Estuarine Fjord. *Estuaries and Coasts*, 37(5), 1256–1268.

Higgins, K., Schlenger P., Small, J., Hennessy, D., and Hall, J. (2005). Spatial relationships between beneficial and detrimental nearshore habitat parameters in WRIA 9 and the City of Seattle. Proceedings of the 2005 Puget Sound Georgia Basin Research Conference.

Jedrzejczak, M.F. (2002a). Spatio-temporal decay 'hot spots' of stranded wrack in a Baltic sandy coastal system. Part I. Comparative study of the pattern: 1 type of wrack vs beach sites. *Oceanologia* 44:491-512.

Jedrzejczak, M.F. (2002b). Stranded Zostera marina L. vs wrack fauna community interactions on a Baltic sandy beach (Hel Peninsula, Poland): a short-term pilot study. Part II. Driftline effects of succession changes and colonisation of beach fauna. *Oceanologia* 44:367-387.

Lam N.W.Y., Huang R., Chan B.K.K. (2009). Variations in intertidal assemblages and zonation patterns between vertical artificial seawalls and natural rocky shores: a case study from Victoria Harbour, Hong Kong. *Zoological Studies* 48(2):184-195.

Lee T.S., Toft J.D., Cordell J.R., Dethier M.N., Adams J.W., Kelly R.P. (2018) Quantifying the effectiveness of shoreline armoring removal on coastal biota of Puget Sound. *PeerJ* 6:e4275.

Moreira, J., Chapman, M.G., Underwood, A.J. (2006). Seawalls do not sustain viable populations of limpets. *Marine Ecology Progress Series*, 322:179-188.

Morley, S.A., Toft, J.D., Hanson, K.M. (2012). Ecological effects of shoreline armoring on intertidal habitats of a Puget Sound urban estuary. Estuaries and Coasts 35(3):774-784.

Munsch, S., Cordell, J., & Toft, J. (2015). Effects of seawall armoring on juvenile Pacific salmon diets in an urban estuarine embayment. *Marine Ecology Progress Series*, 535, 213–229.

Munsch, S., Cordell, J., & Toft, J. (2016). Fine-scale habitat use and behavior of a nearshore fish community: nursery functions, predation avoidance, and spatiotemporal habitat partitioning. *Marine Ecology Progress Series*, 557, 1–15.

Munsch, S.H., Cordell, J.R., & Toft, J.D. (2015). Effects of shoreline engineering on shallow subtidal fish and crab communities in an urban estuary: A comparison of armored shorelines and nourished beaches. *Ecological Engineering*, 81, 312–320.

Munsch, S. H., Cordell, J.R., Toft, J.D., & Morgan, E.E. (2014). Effects of Seawalls and Piers on Fish Assemblages and Juvenile Salmon Feeding Behavior. *North American Journal of Fisheries Management*, 34(4), 814–827.

National Research Council (NRC) 2000. Clean Coastal Waters: Understanding and Redu cing the Effects of Nutrient Pollution. National Academy Press, Washington, D.C.

Neumann, B., Vafeidis, A.T., Zimmermann, J., Nicholls, R.J. (2015). Future coastal population growth and exposure to sea-level rise and coastal flooding—a global assessment. *PLOS ONE*, 10(3): e0118571.

Nordstrom, K.F. (1992). Estuarine Beaches. *Elsevier Applied Science*, New York.

Nordstrom, K.F. (2014). Living with shore protection structures: A review. *Estuarine, Coastal and Shelf Science*, 150, 11–23.

Penttila, D.E. (2007). Marine Forage Fishes in Puget Sound. Puget Sound Nearshore Partnership Report U.S. Army Corps of Engineers, Seattle, Washington, pp. 23 Report No. 2007-03.

Pontee, N. (2013). Defining coastal squeeze: A discussion. *Ocean & Coastal Management*, 84, 204–207.

Perla, B. (2018). Personal communication. Vashon Nature Center, Vashon, Washington.

Perla, B.S., Metler, M.A. (2016). Bulkhead removal monitoring study in Maury Island Aquatic Reserve. Quality Assurance Project Plan. Vashon Nature Center, Vashon, WA. 52 pp.

Puget Sound Partnership. (2012). 2012 State of the Sound: A Biennial Report on the Recovery of Puget Sound. Puget Sound Partnership, Tacoma, Washington.

Rhodes, L.D., Durkin, C., Nance, S.L., Rice, C.A. (2006). Prevalence and analysis of Renibacterium salmoninarum infection among juvenile Chinook salmon *Oncorhynchus tshawytscha* in North Puget Sound. *Diseases of Aquatic Organisms,* 71:179-190.

Rice, C.A. (2006). Effects of shoreline modification on a northern Puget Sound beach: microclimate and embryo mortality in surf smelt (*Hypomesus pretiosus*). *Estuaries and Coasts* 29: 63-71.

Rice, C.A. (2003, April). Unpublished data from the NOAA/NMFS Northwest Fisheries Science Center and Skagit System Cooperative. Presented at the Georgia Basin/Puget Sound Research Conference, Vancouver, BC, Canada.

Romanuk, T.N., Levings, C.D. (2003). Associates between arthropods and the supralittoral ecotone: dependence of aquatic and terrestrial taxa on riparian vegetation. *Environmental Entomology*, 32: 1343-1353.

Ruiz, G.M., Hines, A.H., Posey, M.H. (1993). Shallow water as a refuge habitat for fish and crustaceans in non-vegetated estuaries: an example from Chesapeake Bay. *Mar. Ecol. Prog. Ser.* 99, 1-16.

Seitz, R.D., Lipcius, R.N., Olmstead, N.H., Seebo, M.S., Lambert, D.M. (2006). Influence of shallow-water habitats and shoreline development on abundance, biomass, and diversity of benthic prey and predators in Chesapeake Bay. *Marine Ecology Progress Series*, 326:11-27.

Shipman, H. (2008). A Geomorphic Classification of Puget Sound Nearshore Landforms. Puget Sound Nearshore Partnership Report No. 2008-01. Published by Seattle District, U.S. Army Corps of Engineers, Seattle, Washington.

Shipman, H. (2010). *The geomorphic setting of Puget Sound: implications for shoreline erosion and the impacts of erosion control structures*, in Shipman, H., Dethier, M.N., Gelfenbaum, G., Fresh, K.L., and Dinicola, R.S., eds., 2010, Puget Sound Shorelines and the Impacts of Armoring— Proceedings of a State of the Science

Workshop, May 2009: U.S. Geological Survey Scientific Investigations Report 2010-5254, p. 19-34.

Shoreline Monitoring Toolbox, Washington Sea Grant (2017). https://sites.google.com/a/uw.edu/toolbox/

Simenstad, C.A., Fresh, K.L., Salo, E.O. (1982). The role of Puget Sound and Washington coastal estuaries in the life history of Pacific salmon: an unappreciated function. Pages 343-364 in V.S. Kennedy, editors. *Estuarine comparisons.* Academic Press, New York.

Sobocinski, K.L. (2003). The Impact of Shoreline Armoring on Supratidal Beach Fauna of Central Puget Sound. University of Washington School of Aquatic and Fishery Sciences.

Sobocinski, K.L., Cordell, J.R., & Simenstad, C.A. (2010). Effects of Shoreline Modifications on Supratidal Macroinvertebrate Fauna on Puget Sound, Washington Beaches. *Estuaries and Coasts*, 33(3), 699–711. http://doi.org/10.1007/s12237-009-9262-9.

Toft, J.D., Cordell, J.R., Heerhartz, S.M., Armbrust, E.A., Simenstad, C.A. (2010). Fish and invertebrate response to shoreline armoring and restoration in Puget Sound, in Shipman, H., Dethier, M.N., Gelfenbaum, G., Fresh, K.L., and Dinicola, R.S., eds. (2010). *Puget Sound Shorelines and the Impacts of Armoring—Proceedings of a State of the Science Workshop*, May 2009: U.S. Geological Survey Scientific Investigations Report 2010-5254, p. 161-170.

Toft, J.D., Cordell, J.R., Armbrust, E.A. (2014). Shoreline Armoring Impacts and Beach Restoration Effectiveness Vary with Elevation. *Northwest Science*, 88(4), 367–375.

Toft, J.D., Cordell, J.R., Simenstad, C.A., Stamatiou, L.A. (2007). Fish Distribution, Abundance, and Behavior along City Shoreline Types in Puget Sound. *North American Journal of Fisheries Management*, 27(2), 465–480.

Toft, J.D., Ogston, A.S., Heerhartz, S.M., Cordell, J.R., Flemer, E.E. (2013). Ecological response and physical stability of habitat enhancements along an urban armored shoreline. *Ecological Engineering*, 57, 97–108.

Van Zwalenburg, Kim (2016, February 1). Personal interview on shoreline armoring. Department of Ecology, Olympia, Washington.

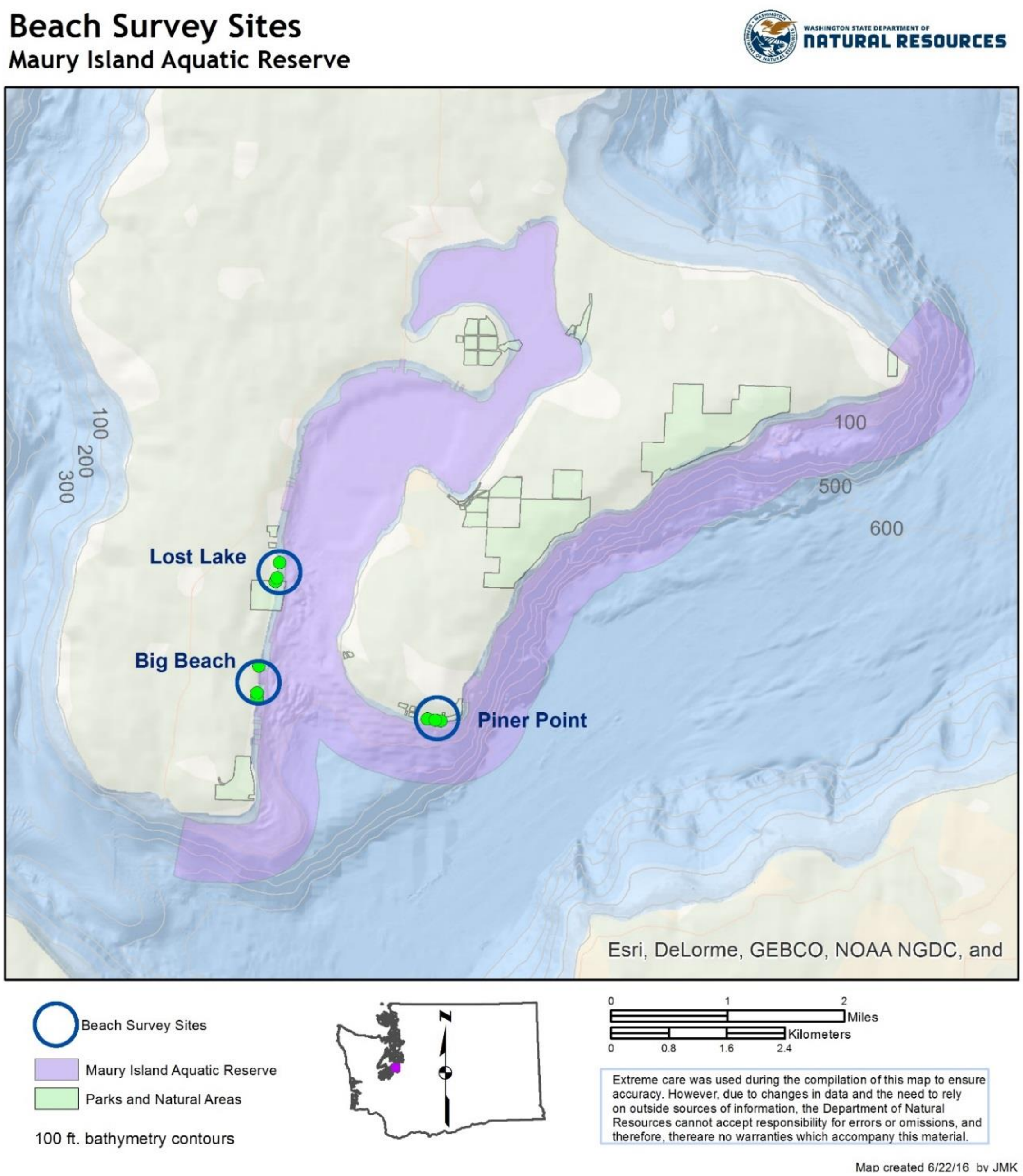
Washington State Department of Fish and Wildlife (WDFW, 2016). Hydraulic Project Approval (HPA): Frequently Asked Questions. Accessed February 20, 2016 at http://wdfw.wa.gov/licensing/hpa/faq.html.

Weitkamp, D.E., G.R. Ruggerone, L. Sacha, J. Howell, B. Bachen. (2000). Factors affecting Chinook populations. Parametrix, Natural Resource Consultants, and Cedar River Associates, Report to City of Seattle, Washington.

Wobbrock, J.O., Findlater, L., Gergle, D., Higgins, J.J. (2011). The Aligned Rank Transform for nonparametric factorial analyses using only ANOVA procedures. Proceedings of the ACM Conference on Human Factors in Computing Systems (CHI '11). Vancouver, British Columbia (May 7-12, 2011). New York: ACM Press, pp. 143-146. Honorable Mention Paper.

Appendices

Appendix A. Map of Maury Island Aquatic Reserve



Appendix B. Shoreline Monitoring Toolbox vegetation sampling protocol

Vegetation

Characterizing shoreline vegetation such as dunegrass and willows can give valuable information on the habitat of the upper beach and marine-terrestrial connectivity. This may change depending on shoreline armoring, development in the uplands, and new plantings of vegetation at restoration sites. Vegetation stabilizes the shoreline and provides habitat for terrestrial insects that are prey resources for juvenile salmon.

Materials

- Two 50 m measuring tapes, one for the transect and one for vegetation measurements
- 0.25 m^2 pvc quadrat (0.5 x 0.5 m)

Sampling Summary

- Generate a plant species list
- Percent cover of over and understory vegetation
- Canopy diameter of trees
- Health ratings of vegetation
- Dunegrass: 50 m transect parallel to shore. N=5 measurements for patch width and 0.25 m^2 quadrats for shoot density and percent cover

Scale of Effort

$ Cost – low, simple materials and data are all field-based

$ People – low, 2-3 people can establish transects and record vegetation data

$ Fieldwork time – low, 1 day, once a year in July when vegetation is lush

$ Processing time – low, entering field data into computer format

$ Technical expertise – medium, identification of plant species

Additional Resources

Reports that have used this method: Toft et al. 2012

Also see Chappell 2006 for species information of vegetation in the Puget Sound region

Suggested citation: Shoreline Monitoring Toolbox. Washington Sea Grant.
Website: wsg.washington.edu/toolbox

Methods

Start by generating a plant species list for the site, noting native and introduced species. Estimate the percent cover of over (trees) and understory (e.g., dunegrass, salal) vegetation in increments of 5% at different areas; this is best done in ~5 x 5 m patches, choose a subset depending on the size of the site and location of key vegetation features. Measure the canopy diameter of trees at their widest point by using a transect tape. Give each vegetation area a health rating between 1 (dead) and 5 (vigorous growth), noting specific plants/trees that are characteristic of the rating. At patches of dunegrass establish a transect parallel to shore along its length, or for 50 m if the patch is very long. At five random points along the transect measure the width of the dunegrass patch, and use a 0.25 m^2 quadrat to estimate shoot density and percent cover in increments of 1%. Sample in a summer month such as July when vegetation is lush.

Data to record in the field

Date, time, site name, sample numbers, vegetation data. It is advisable to take a digital photo of the transect and specific vegetation types for documentation.

Processing

Enter the field data into computer spreadsheets. Monitoring over time can generate growth parameters for different vegetation types and detail any changes in over and understory structure. Vegetation data can be used as causal factors for other data types such as insects and shorebirds.

Appendix C. Shoreline Monitoring Toolbox wrack sampling protocol

Beach Wrack

Characterizing beach wrack provides valuable information on the habitat of the upper beach and marine-terrestrial connectivity. This may change depending on shoreline armoring, source material alterations, and winter storms. Beach wrack provides food and shelter for many invertebrates, and foraging habitat for shorebirds.

Materials

- 50 m transect tape
- 32 x 32 cm pvc quadrat, subdivided with string into 25 6 x 6 cm small squares

Sampling Summary

- 50 m transect parallel to shore
- 0.1 m^2 quadrat (32 x 32 cm)
- N=10 random quadrats per transect
- Transects at most recent wrack line and higher elevation older wrack line
- Measure % cover of algae, eelgrass, terrestrial plants, and trash

Scale of Effort

$ Cost – low, simple materials and data are all field-based

$ People – low, 2-3 people can establish transects and record quadrat data

$ Fieldwork time – low, 1 day, once a year in September when wrack lines are exposed

$ Processing time – low, entering field data into computer format

$ Technical expertise – low, identification of major wrack types

Additional Resources

Reports that have used this method:
Dethier et al. 2016
Heerhartz et al. 2014
Sobocinski et al. 2010

Other methods that require a larger scale of effort and more technical expertise: methods in Heerhartz et al. 2014 that measure biomass of wrack

Suggested citation: Shoreline Monitoring Toolbox. Washington Sea Grant.
Website: wsg.washington.edu/toolbox

Methods

At ten random points along a 50 m transect parallel to shore, place a 0.1 m^2 quadrat on the beach surface and conduct a visual estimate of the percent composition of algae, eelgrass, terrestrial plant material, and trash. Divide the quadrat with string into 25 6 x 6 cm small squares to facilitate these estimates – each square equals 4%. If possible, specify the algae type (e.g., red, green, brown, or species). Establish two transects: (1) at the most recent high tide line that has fresh wrack deposition, and (2) just above MHHW in older wrack. The most recent high tide line will target mobile wrack, whereas the higher elevation sample will target the more stable wrack layer. If there is a bluff or shoreline armoring, sample the elevation at the base. Sample in September as it is typically a period of high wrack accumulation, and on an ebbing tide when the upper beach +6' MLLW and above is exposed.

Data to record in the field

Date, time, site name, transect elevation, sample number, beach wrack data. It is advisable to take a digital photo of the transect and of some example quadrats for documentation.

Processing

Enter the field data into computer spreadsheets. The percentages for each wrack type can be analyzed separately, or combined for a percentage of total wrack cover. The different wrack types give information on the source material available (e.g., riparian vegetation for terrestrial sources), and the amounts that deposit on the beach.

Appendix D. Shoreline Monitoring Toolbox insect sampling protocol

Insects

Terrestrial insects are a good indicator of shoreline conditions and an important prey component for juvenile salmon. Using passive fallout traps to characterize the insect community simulates insects that could fall on the surface of the water and be available as fish prey. Insect communities may vary depending on the amount of riparian vegetation, shoreline armoring, and other habitat features.

Materials

- Plastic storage bins 40 x 25 cm (0.1 m^2), 10 cm high
- Natural dishwashing soap (biodegradable, odorless)
- 0.106 mm sieve
- Water sprayer, two buckets for collecting and sieving water
- Jars and labels, 70% isopropyl alcohol
- Microscope

Sampling Summary

- 50 m transect parallel to shore above tidal inundation
- Place bins with a few drops of soap and ~5 cm of sieved water
- N=5 random bins per transect
- Leave for 24 hours, preserve in 70% isopropyl alcohol
- SAFETY: isopropyl alcohol is flammable, store carefully and avoid skin contact

Scale of Effort

$$$ Cost – high, field and laboratory supplies can be expensive (e.g., alcohol, microscopes)

$ People – low, 2-3 people can deploy and collect bins

$$ Fieldwork time – medium, once a month June and July, two days in a row for deployment and collection

$$$ Processing time – high, analyzing insect samples in the laboratory

$$ Technical expertise – medium, depending on insect ID level

Additional Resources

Reports that have used this method:
Toft et al. 2013
Sobocinski et al. 2010

Suggested citation: Shoreline Monitoring Toolbox. Washington Sea Grant.
Website: wsg.washington.edu/toolbox

Methods

Use plastic storage bins (preferably 40 x 25 cm) filled with 5 cm of soapy water as fallout traps. Make sure to measure the surface area of the bins to standardize counts. Place five replicate bins randomly along a 50 m transect parallel to shore. Pour a few drops of natural odorless dishwashing soap in the bottom, and fill with about 5 cm of sieved water. The dishwashing soap relieves surface tension so that insects will remain trapped, and sieving the water ensures that there are no invertebrates that could contaminate your sample. Leave the bins in place for 24 hours. To collect the insects, drain each bin through a 106 micron mesh sieve, and spray the insects into a sample jar (fill a spray bottle or weed sprayer with sieved water for this). Fix the sample in 70% isopropyl alcohol and label the jar. Sample in June-July when juvenile Chinook salmon are feeding along the shoreline, and vegetation and insect communities are developed.

Data to record in the field

Date, site name, time of deployment and collection, sample number (also include these on the jar label). It is advisable to take a digital photo of the transect for documentation.

Processing

Microscope identification of insects requires some skill and time. Chironomidae flies and aphids are two key juvenile salmon prey items that should be identified at the Family taxonomic level. Other insects such as Hymenoptera and Lepidoptera can be identified at the Order level if taxonomic expertise is limited. Processing at a consistent taxonomic level allows calculation of diversity measurements (e.g., taxa richness, the number of different taxa in the sample). Convert counts to density (#/m^2) based on the surface area of the bin.

Appendix E. Shoreline Monitoring Toolbox fish protocol

Fish

Improving habitat for out-migrating juvenile salmon is often a goal of nearshore restoration efforts. Direct observation of fish use of a site is desirable to assess function of the site. Surface snorkel surveys are recommended as an observational method that can generate data without handling fish. Observations are focused on juvenile salmon abundance, feeding behaviors, and records of other nearshore fishes.

Materials

- Snorkel gear – drysuit or wetsuit, mask, snorkel, fins, ankle weights
- 50 m or longer transect tape
- Underwater writing tablet, or clipboard with datasheet printed on waterproof paper

Sampling Summary

- 75 m transect parallel to shore
- 3 m and 10 m from shore for deep sites, 1.5 m water depth if shallow
- Need at least 2.5 m water visibility
- SAFETY: Highly advised to be a skilled swimmer and have snorkel or SCUBA dive experience. Always stay at the surface, be aware of any boat traffic or hazards, and have a shore-based observer

Scale of Effort

$$$ Cost – high, snorkel gear is expensive, SCUBA divers may already have gear which would greatly reduce costs

$ People – low, 2 snorkelers and 1 shore observer can establish transects and record data

$$$ Fieldwork time – high, base effort 2x/month at high tides May-July

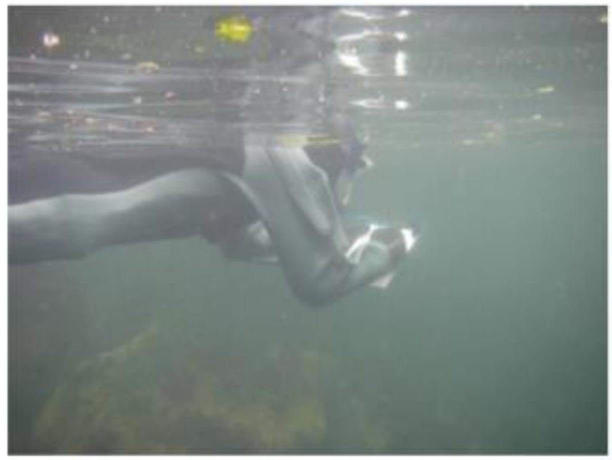
$$ Processing time – medium, entering field data into computer format, possible verification of fish ids

$$$ Technical expertise – high, snorkel surveys and fish identifications both require background knowledge

Additional Resources

Reports that have used this method: Toft et al. 2007, 2013

Suggested citation: Shoreline Monitoring Toolbox. Washington Sea Grant.
Website: wsg.washington.edu/toolbox



Methods

Conduct surface snorkel surveys parallel to shore along a 75 m transect at high tide. Have two snorkelers in the water and a shore-based observer. The water depth and distance from shore may vary with the site – for deep sites target 3 m and 10 m from shore, for shallower sites target 1.5 m water depth. These are good ranges for juvenile Chinook salmon. Smaller juvenile chum and pink salmon may be in shallower water. Record observations of fish species, number (approximate if over 20), length range (2.5 cm increments), water column position (surface, mid-water, bottom), and feeding behavior. Swim slowly and consistently, scanning the water column with a focus near the water's surface where juvenile salmon are likely to be (tilt your head sideways for this). Pause to record data as appropriate. Data can be written on either an underwater writing tablet or clipboard with datasheet printed on waterproof paper. Use the transect tape to measure the transect length, water depth, and underwater visibility (horizontal distance that you can see the writing tablet underwater – needs to be at least 2.5 m). May is a good month to target the peak outmigration of juvenile chum and pink salmon, June and July are good peak months for Chinook.

Data to record in the field

Date, time, site name, transect length, water depth, distance from shore, underwater visibility, fish data. An underwater digital camera can help document fish presence.

Processing

Enter the field data into computer spreadsheets. Fish counts are standardized by numbers/m^2 as: fish number/(transect length x underwater visibility).