ABSTRACT

IMPACTS OF A CONSTRUCTED OYSTER BED ON INFAUNAL INVERTEBRATE COMMUNITIES IN JACK DUNSTER MARINE RESERVE

By

Terrance M. Champieux

May 2015

Oysters are important to estuarine ecosystems because of the functions they provide. Thus, oyster restoration projects are undertaken in areas where natural populations have declined. However, restoration techniques can impact sediment organic matter and benthic invertebrates that provide trophic support for important species. This study assesses the impacts of a constructed shell bed on associated sediment and invertebrate communities in a southern California bay. Within the bed site, organic matter, invertebrate abundance, and invertebrate species richness are lower only under the oyster bed. The alteration in the community under the shell is driven by a reduction in species. Tubificidae were the only remaining species under the shell. These results may be explained by the shells' action as a barrier to the mud-water interface. While significant, impacts of oyster bed construction are spatially restricted to just under the bed. Longer-term studies should be conducted to address effects of the oysters themselves.

IMPACTS OF A CONSTRUCTED OYSTER BED ON INFAUNAL INVERTEBRATE COMMUNITIES IN JACK DUNSTER MARINE RESERVE

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CHAPTER 1

INTRODUCTION

The Olympia oyster, *Ostrea lurida*, was once abundant throughout California coastal waters, including the bays and estuaries of southern California (McGraw, 2009). The Olympia oyster is the only oyster species native to the West Coast of the United States. A century ago, natural populations were depleted due to a combination of over-harvesting, dredging, pollution, and the draining and filling of wetlands (Hopkins, 1931; Barrett, 1963; Baker, 1995; Dinnel et al., 2009; McGraw, 2009). In an effort to increase population sizes of these oysters, restoration is used to recruit oyster settlers in ecosystems where they were historically present but recently greatly reduced. Since the process of oyster bed restoration is a relatively new process on the west coast of the U. S., the effects on associated sediment and sediment-dwelling invertebrates has never been studied. In order to learn what effects Olympia oysters and oyster bed restoration have on the associated invertebrate community, it is important to understand Olympia oysters' natural history and historic decline as well as ongoing restoration attempts.

The natural history of these oysters is relatively unknown. They are found in greatest numbers around the 0-meter tide level in estuarine habitats with hard substratum available for successful settlement (Brumbaugh and Coen, 2009; Wasson et al., 2014). From the plankton, veliger larvae settle on the hard substratum and metamorphose into spat. Olympia oysters are slow-growing but will grow to a shell height of 35-50mm as adults. It is unknown how long Olympia oysters will live, as their maximum age has not been reported. Their main food source is phytoplankton which they filter out of the water via large ostia openings. They are sensitive to

extremely high or low temperatures, low salinities, and pollution which make them vulnerable extreme weather events and anthropogenic-caused environmental degradation (Couch and Hassler, 1989). Their predators include many species of crab, snails, sea stars, oyster drills, and birds. In California, Olympia oysters' main predators are the stingray, *Myliobatus californica*, and the leopard shark, *Triakis semifasciata*. Little has been done to study the effects of these predators because in commercial Olympia beds they are not considered significant threats (Baker, 1995).

Olympia oysters may play an important role in habitat provisioning as many species use the oyster beds of other oyster species. Many epibionts like mussels, barnacles, sponges, polychaetes, bryozoans, and amphipods use the oysters' qualities as habitat engineers for refuge. Invasive *Crassostrea gigas* oysters have been considered competitors to the Olympia oysters, but Baker (1995) and others studies report that there is no evidence that *C. gigas* is a serious competitor because the bed structure of each species provides structural habitat for the other species (Baker, 1995).

Historically, Olympia oysters were found throughout California bays and were an important food source for Native Americans (Bonnot, 1935; Howard, 1935; Elsasser and Heizer, 1966; Kidd, 1967). As a commercially important species in the 19th century, their populations declined, partly due to over-harvesting (Hopkins, 1931; Baker, 1995; Conte, 1996). A potentially bigger impact contributing to their decline in Southern California was habitat loss. California has lost approximately 90% of its native wetlands to draining and filling for coastal development (Dahl, 1990). In recent years, there has been renewed interest among ecologists and commercial oyster farmers in restoring populations of this native Californian species, necessitating the restoration of hard substrate oyster beds for adequate settlement.

As a response to the population decline, restoration projects throughout the West Coast are being implemented to increase the number and sizes of populations of Olympia oysters. This increase in oyster number would likely also be followed by improved ecosystem function. Oysters are very important to the health and resilience of estuarine ecosystems because of the many functions they provide to these ecosystems. They build habitat for a community of animals that use the oyster bed as a refuge and a sturdy foundation (Baker, 1995; Grabowski and Powers, 2004; Rodney and Paynter, 2006; Markert et al., 2010). While cleaning and clarifying the water via filter feeding, which can improve seagrass development, they also stabilize sediment (Newell, 2004; Grabowski and Peterson, 2007; van der Heide et al., 2012).

Turbidity events resulting in low light availability can result in loss of seagrass. Since oysters' feeding activities, which normally improve water conditions, are reduced due to declines in their stock, seagrass communities could also decline as a result of oyster population declines (Newell and Koch, 2004). Oysters extract suspended particles from the water and excrete them as feces and pseudofeces which sink and become part of the sediment thus increasing the nutrient content of surrounding sediment (Newell, 2004). By decreasing turbidity and thus allowing sunlight to reach eelgrass, oysters extend ecosystem functions beyond the boundaries of the oyster bed itself (Grabowski and Peterson, 2007). Although these studies were conducted in other systems, the restoration of Olympia oysters in California has the potential to reduce turbidity and improve seagrass growth.

Based on the ecosystem functions described above, it is clear that restoration of oyster populations can be important, but the techniques and structures used for restoration (e.g. the placement of shell on a previously bare mud habitat) could impact the soft sediment mudflat community. Various techniques have been used at restoration sites in Chesapeake Bay, South

Carolina, and Washington; these include the distribution of shell with high pressure hoses, the building of reefs using shell or rock, and the laying of bags of collected shell. One method used is the laying of loose shell directly on the bare mud. This restoration seems like a viable option, but volunteers might trample the sediment during bed construction, the structure may impede water flow, or the shell may sink in the sediment under its own weight. Further, when beds are established, oyster shell may change the sediment-water interface under the shell by reducing oxygen and sediment deposition through its action as a barrier.

Despite the likelihood of impacts from the placement of shell, there is not much research on the impacts due to restoration. Whether looking at natural or restored oyster beds, most studies only consider the invertebrates associated with the bed itself or the sediment next to the bed; they do not look at the sediment and invertebrate community directly below the oyster bed. A recent literature review, which looked at published papers and environmental reports evaluating 249 restored beds, showed that none of them specifically looked at the impacts of the beds on the underlying sediment (La Peyre et al., 2014). Most studies focus on the benthic communities surrounding, within, and on top of the oyster beds (Grabowski and Powers, 2004; Rodney and Paynter, 2006; Grabowski and Peterson, 2007; Markert et al., 2010; Zacherl et al., 2011; Grabowski et al., 2012; Hollander et al., 2015). Two recent non-restoration studies in the North Atlantic looked at the macrobenthos under oyster beds of the invasive *Crassostrea gigas*. These studies found that the biodiversity of the habitat increased with the structural complexity, specifically showing an increase in infauna like oligochaetes and polychaetes, in which the spaces between the shells were micro-habitats for these infauna (Markert et al., 2010; Hollander et al., 2015).

Despite the lack of studies examining changes in macrobenthos communities under restored oyster beds, it is known that these invertebrates play an important role in the estuarine ecosystem. While macroinvertebrate densities and species richness are generally positively correlated with habitat complexity (Rodney and Paynter, 2006) what about benthic infauna under restored oyster beds? Infauna are benthic animals, such as clams, worms, and burrowing crabs, that live in the substrate of a body of water, especially in a soft sediment bottom. As an integral part of the consumer food chain, benthic infauna provide important trophic support to species of commercial and intrinsic importance like crab, fish, and birds (Sacco et al., 1994; Levin et al., 1996; Moseman et al., 2004). Infauna usually construct tubes or burrows and are commonly found in intertidal and subtidal waters. Benthic infauna are important in sediment turnover and bioturbation, activities that mix and transport particles, water, and solutes within the sediment and across the sediment water interface (Rhoads, 1974; Coen and Luckenbach, 2000; Nogaro et al., 2009; Belley et al., 2010). They reflect local environmental conditions and are used as bio-indicators for pollutant studies (Smith et al., 2001). One of the threats that infauna face is habitat loss due to development, dredging, and, possibly, oyster bed restoration. Habitat loss through destruction or fragmentation can cause infaunal abundance to decline (Micheli et al., 2008). Dredging and trawling destroy soft sediment habitat due to the impact and physical contact of the equipment to the sea floor (Thrush and Dayton, 2002; Dernie et al., 2003). Since restoration is important to the reestablishment of valuable species (i.e. Olympia oysters) it should be known if the impact of the restoration might be too high.

My project details how the underlying sediment and associated invertebrate community is impacted by shell placement in an unvegetated mudflat community in Alamitos Bay, CA.

Understanding this question helps both more completely evaluate the benefits and costs of restoration as well as help develop future selection criteria for restoration sites.

My hypotheses are:

(1) Organic matter content and flow

(1a) The oyster bed structure (specifically the presence of shell on the mudflat) results in reduced flow rates on the landward side, causing the landward side to have increased sediment deposition and sediment organic content because sediment is trapped there during outgoing water movement.

(1b) The oyster bed structure blocks sediment deposition to the underlying substratum, causing the organic matter content to be lower in the underlying sediment.

(1c) The oyster bed structure will have no effect on flow rates on the seaward side, causing no effect to the sediment and the organic matter content.

(2) Invertebrate community parameters

(2a) The oyster bed structure and subsequent reduced flow rates on the landward side causes the landward side to have higher abundance and diversity of the infauna, especially higher abundance of capitellid polychaetes relative to pre-oyster bed samples (a disturbance-tolerant and mud-loving group).

(2b) The oyster bed structure and subsequent blockage of sediment deposition and organic matter content causes the underlying sediment to have lower infaunal abundance and diversity, with lower capitellid polychaetes relative to pre-oyster bed samples.

(2c)With no change to seaward side sediment and organic matter, there is no effect on seaward side infaunal abundance and diversity.

(2d) The oyster bed establishment will not affect neighboring eelgrass infaunal abundance, diversity, and community composition before sufficient oyster settlement occurs.

CHAPTER 2

MATERIALS AND METHODS

Study Site

The study was conducted in Jack Dunster Marine Reserve (JDMR) located at the mouth of the Los Cerritos Channel in Alamitos Bay, Long Beach, California (118°7'9" N, 33°45'43" W) (Fig. 1). JDMR is a 2.7 acre site containing 1.5 acres of land and 1.2 acres of wetland and subtidal habitat, which were created in 2000 as a mitigated wetland (Apodaca, 2005). The wetlands include a restored oyster bed, mudflat, and eelgrass beds where there were four treatments: one manipulated intertidal mudflat with restored oyster bed (bed)(established June 2012—see details below), one unmanipulated intertidal mudflat (control), one subtidal eelgrass bed adjacent to the oyster bed (near), and one subtidal eelgrass bed adjacent to the control (far). Bed started at the south-west side of the eastern-most observation platform in JDMR and extended south-west as a 30m x 2m band (Fig. 2). Near ran parallel, approximately five to ten meters south-east, to bed and extended south-west as a 30m x 2m band (Fig. 2). Far continued 30 meters from the end of near and extended south-west as a 30m x 2m band (Fig. 2). Control started at the north-east side of the western-most observation platform in JDMR and extended north-east as a 30m x 2m band (Fig. 2). Control was approximately ten meters away from bed, which I hypothesized was an appropriate distance from bed to serve as a control. There was one collection treatment outside of the reserve that served as a subtidal eelgrass reference: Basin-6, which was located on the opposite side of Los Cerritos Channel (118°7'7" N, 33°45'41" W) (approx. 57m away from JDMR). There was an additional treatment along Bay Shore Avenue,

located on the opposite side of Alamitos Bay (118°7’24” N, 33°45’35” W) (approx. 400m away from JDMR) (Fig.1). Basin-6 was included as a subtidal, eelgrass-habitat control because it was likely far enough away from the restored oyster bed to eliminate possible oyster bed effects. Bayshore reference treatment had a pre-restoration collection but was not sampled in the June ’13 collection because it experienced a major decline in eelgrass and increase in cover of *Ulva* sp. Data for Bayshore are not included in this thesis.

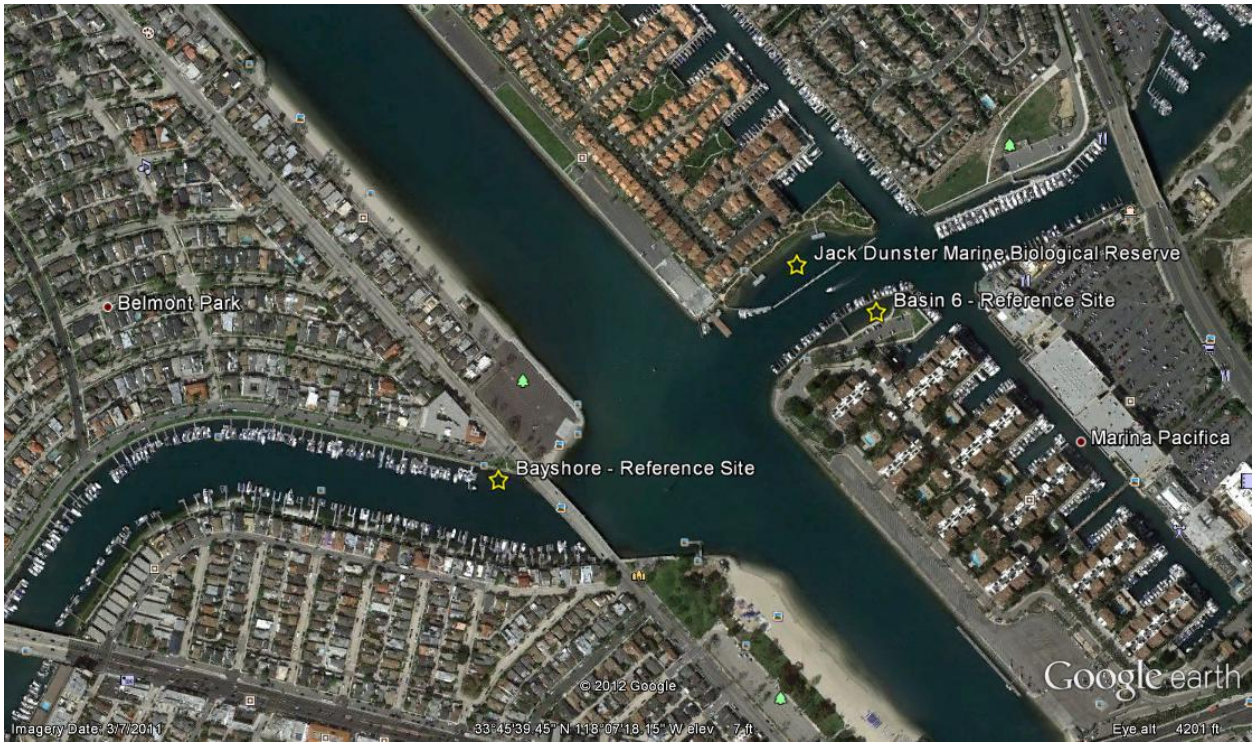

Figure 1. Alamitos Bay, Long Beach, CA showing the locations of Jack Dunster Marine Reserve, Basin6, and Bayshore.

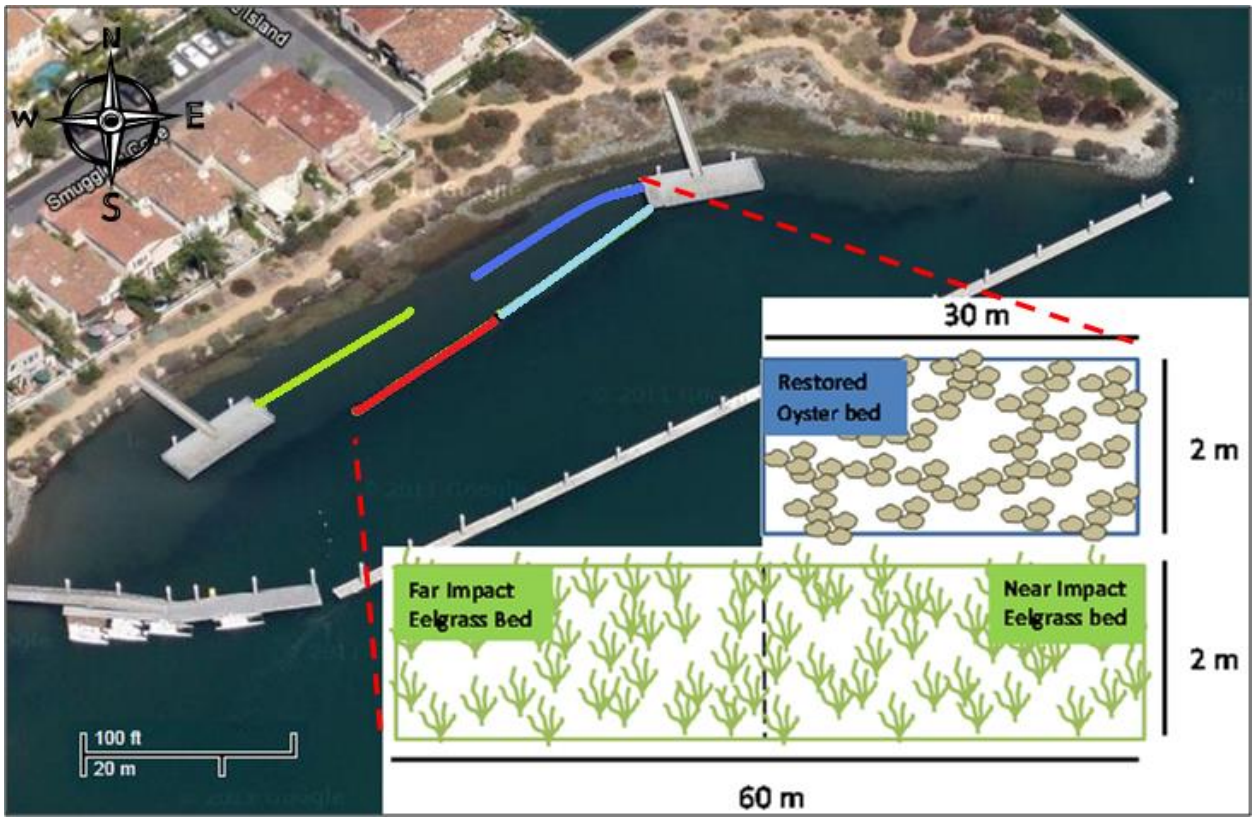

Figure 2. Schematic showing experimental design in Jack Dunster Marine Reserve. The control location (solid green line) starts at the north-east side of the western-most observation platform in JDMR and extends north-east as a 30m x 2m band. The oyster bed location (solid blue line) starts at the south-west side of the eastern-most observation platform in JDMR and extends south-west as a 30m x 2m band. There is approximately 10m between control and bed. Near and far seagrass locations (solid light blue and red lines) run parallel to bed and extend south-west as a 30m x 2m band. Far continues another 30 meters from the end of near extending south-west as a 30m x 2m band. Image credit: Google Earth.

Experimental Design

On June 20 and 21, 2012, loose *Crassostrea gigas* oyster shell (~8 cu. yards) acquired from Carlsbad Aquafarm was used to build a bed approximately 30m long by 2m wide by 8cm deep on intertidal mudflat habitat in Jack Dunster Marine Reserve (approximately 60 sq meters). The shell was laid in place using manual labor from community volunteers. The oyster bed and other study treatments were 30m x 2m bands that contained mud, oyster shell and mud, or eelgrass and mud; four of these were within the reserve, and one was outside the reserve (Fig. 1).

Sample collection occurred annually with one in June ’12 (pre-restoration) and another in June ’13 (12 months post-restoration). Additional collections occurred in Jan ’13 (seven months post-restoration) and Jan ’14, but data from these collections were not included in this thesis. In the June ’12 collection, seven infaunal cores (for invertebrate quantification) were taken (7 X 6 = 42) from each area, each from within a 0.5m x 0.5m quadrat placed at random coordinates along a 30m transect, chosen with a random number table. Pre-oyster bed samples were taken from the bed treatment sediment where the future oyster bed was to be laid. In the June ’13 collection, in order to determine the impact of flow, 21 infaunal cores (for invertebrate quantification) (7 landward from shell [In], 7 under shell [Under], and 7 seaward from shell [Out]) and 21 sediment cores (for sediment analysis) (7 landward from shell [In], 7 under shell [Under], and 7 seaward from shell [Out]) were taken from the bed treatment, each from within a 0.5m x 0.5m quadrat placed at random coordinates along the 30m transect, chosen with a random number table. For the rest of the treatments, core collection was duplicated from the June ’12 collection with the addition of seven sediment cores (for sediment analysis) per treatment. Sediment cores for organic matter and grain size analyses were not taken in the June ’12 collection. More shell was laid seaward and adjacent to the June ’12 shell at the June ’13 sampling, preventing subsequent samplings for the Out-bed portion of my thesis project.

This study uses BACI design which is used to detect nonrandom change in a series of observations made before and after manipulation of a single system. Stewart-Oaten et al. (1986) described BACI analysis, in which experimental (impact) and reference (control) ecosystems are compared before and after the treatment of the experimental system (Stewart-Oaten and Bence, 2001; Gotelli and Ellison, 2004).

Flow, Tide, and Sediment Deposition

To approximate relative flow rate, clod cards were deployed at the bed and control treatments in June 2013. Modifying methods from Muus (1968) and Doty (1971), clod cards were created using plaster of Paris poured into ice cube trays. Clod card cubes were epoxy-glued to PVC tiles (3in x 4in). They were anchored to the substrate with threaded rods drilled at two corners. The clod cards were weighed prior to deployment and after collection (after air drying for 4 days) to determine percent loss over a 24hr time period. 10 clod cards were deployed landward of the oyster bed [In], throughout the oyster bed itself [On], and seaward of the oyster bed [Out]. In addition, 10 clod cards were deployed on the mudflat in the control location. In addition, one was placed in seawater of comparable salinity in the lab to measure dissolution rate without flow (as a lab control).

To determine flow direction, approximately 5 mL of fluorescein dye was placed on the water surface at the bed and at the control treatment, using a plastic cup tethered to the end of a meter stick, and the direction of travel was recorded. This was done multiple times (during incoming and outgoing tides) to understand flow during both tidal regimes (10/02/2013 and 11/05/2014).

Organic Matter Content

Sediment cores were also collected in each treatment plot using a cylindrical push core (4.8 cm diameter, 18.1 cm^2) inserted to a depth of 3 cm. Each sediment core was placed in an individual Ziploc plastic bag on site. In lab, each core was decanted of excess water and placed in a paper bowl, then dried at 50°C, to constant weight. Prior to grain size analysis, the sediment cores were ground to a powder, and small portions were placed into pre-weighed crucibles. Crucibles with sediment were weighed, and the crucible weight was subtracted to yield pre-

combustion sediment weight. Crucibles with sediment were combusted at 500°C for two hours and left to cool. After combustion, the crucibles with sediment were weighed and the crucible weight subtracted to yield the post-combustion sediment weight. The post-combustion weight was subtracted from the pre-combustion weight to yield the organic matter weight. The organic matter weight was divided by the pre-combustion sediment weight and multiplied by 100 to yield percent organic matter.

Grain Size Analysis

The remaining ground sediment was used to determine the percent of sand, silt, and clay. Analysis of grain size used the hydrometer method (Bouyoucos, 1962).

Infaunal Sampling

Infaunal sampling protocols were based on protocols from previous studies (Levin et al., 1998; Talley and Levin, 1999; Levin and Talley, 2002; Whitcraft and Levin, 2007). On an annual sampling schedule, infaunal cores were taken in each treatment plot using a cylindrical push core (4.8 cm diameter, 18.1 cm^2) inserted to a depth of 6 cm. The size of the core was chosen in order to target infauna in a 1-2 mm size range thus excluding megafauna. Mudflat cores were placed directly into the respective containers for preserving. Scuba divers placed subtidal cores in Ziploc plastic bags, which were later decanted of excess water and transferred to containers for preserving. Infaunal cores remained unsieved while being preserved, for a minimum of 48 hours, in 8% buffered formalin with Rose Bengal. To quantify infaunal abundance, richness, and diversity, the preserved cores were sieved through a 300 µm mesh, and animals that were separated from the sediment were sorted under a dissecting microscope at 12x magnification, identified to the lowest taxonomic level possible using various taxonomic literature (Hartman, 1968; Hartman, 1969; Light et al., 2007), counted, and stored in 70%

ethanol. Abundance, species richness, Shannon-Wiener Diversity Index, and Reciprocal Simpson's Diversity Index were calculated for each core.

Statistical Analyses

To determine the effects of oyster bed restoration on associated infauna, the pre-oyster shell samples were compared to the underlying post-oyster shell samples from the bed treatment. Underlying oyster bed samples were also compared to control samples (pre- and post-restoration). To determine the effects of flow on organic matter content and infauna the landward-bed samples were compared to the seaward-bed samples. To document organic matter and infaunal changes in the mudflat that were not associated with oyster bed restoration pre-oyster shell control samples were compared to the post-oyster shell control samples from the control treatment. To determine effects of oyster bed restoration on associated eelgrass bed percent organic matter and infauna the near treatment pre-oyster shell eelgrass samples were compared to the near treatment post-oyster shell eelgrass samples. Also, far treatment pre-oyster shell eelgrass samples were compared to the far treatment post-oyster shell eelgrass samples. Additionally, near treatment post-oyster shell eelgrass samples were compared to the far treatment post-oyster shell eelgrass samples to determine effects on the eelgrass community. Basin6 pre-oyster shell samples were compared to the Basin6 post-oyster shell samples to document any changes. Bayshore reference site was not sampled because of an eelgrass reduction and *Ulva sp.* take-over.

All univariate tests were conducted with Minitab statistical software. Data were tested for normality/equal variances and transformed as needed before analysis. Due to pre-construction differences in abundance between the control and the bed, change in abundance and richness from pre-construction to post-construction sampling dates was used instead of absolute

abundance and richness. Comparisons of change in abundance, change in richness, change in diversity, sediment grain size, and percent organic matter were conducted with two-way ANOVAs (date, treatment [location and position e. g. control, bed seaward, bed under, far, near], and date x treatment) followed by Tukey's test. If data could not be normalized, comparisons were conducted with nonparametric Kruskal-Wallis tests, followed by non-parametric Tukey's Tests in R statistical package. Depending on significance of two-way ANOVAs, I split sampling dates and conducted tests within year (if sampling date was significant). Multivariate analyses were conducted on infaunal count data (square root transformed) using Primer 6 and were based on Bray-Curtis similarity indices. Pairwise comparisons of community similarity were made using analysis of similarity, ANOSIM. Similarity percentages, SIMPER, were analyzed to show which species were driving any observed community differences. Pearson's *r* or Spearman's rho correlation analysis was used to compare organic matter/grain size to biotic variables and community composition. All statistical analyses used the confidence level of $\alpha=0.05$.

CHAPTER 3

RESULTS

Organic Matter (June 2013)

The interaction of location and treatment (In, Out, and Under) (control versus bed-associated) was significant (F_2=5.40, P=0.009, Fig 3, Table 1). Within the bed treatment, organic matter was significantly lower underneath the bed as compared to In or Out (F_2=9.76, P=0.001, Fig 3). Within the control treatment, no significant differences existed among treatments (In, Out, and Under) (F_2=0.19, P=0.831, Fig 3) (Table 1).

Grain Size (June 2013)

There was no significant interaction. The percent sand was significantly greater at the bed location versus the control location (F_1=8.87, P=0.005, Fig 4) (Table 1). There was no significant difference among treatments.

Clod Card (proxy for flow rate) (June 2013)

The interaction of location and treatment (In, Out, and Under) (control versus bed-associated) was significant (F_2=5.19, P=0.013, Fig 5). On the bed treatment, clod card percent loss was significantly lower on the bed than landward of the bed (F_2=6.59, P=0.012, Fig 5). On the control treatment, no significant differences existed among the treatments (In, Out, and On) (F_2=1.06, P=0.377, Fig 5) (Table 1).

Qualitative Flow

During a flooding tide, fluorescein dye released at the bed traveled in a north-east direction towards the Los Cerritos Channel. A second deployment showed the dye moving north-east again, but it hugged the shore-line while moving. The dye was moving at about 20 cm / min. Wind and micro-eddies brought some dye back to the oyster bed where its travel was slowed by the oyster shell, in a south-easterly direction. Once past the oyster shell the dye continued south-east and was slowed by the eel-grass bed. The time span between entering the oyster bed and exiting the eel grass bed took about 15 minutes. At the control treatment another dye deployment showed entrainment around the dock, potentially indicating that the control treatment is in the lee of the dock during flooding tides. Traveling at about 1 meter every 5 minutes, the dye dissipated in less than thirty minutes (Fig 6).

During an ebbing tide, another dye deployment at the bed treatment traveled in a south-westerly direction towards Marine Stadium and the ocean. The dye traveled at about 1 meter every 3 minutes. The dye was purposely placed landward of the oyster bed to see if the shell would still affect the dye rate of travel. As with the flooding tide the dye was slowed by the shell and the eel grass, but not as drastically. The tide level was higher and the water column over the beds was greater. The last dye deployment at the control treatment, during an ebbing tide showed the dye moving south-westerly, under the viewing platform, and towards Marine Stadium and the ocean. The dye dissipated in about twenty minutes (Fig 7).

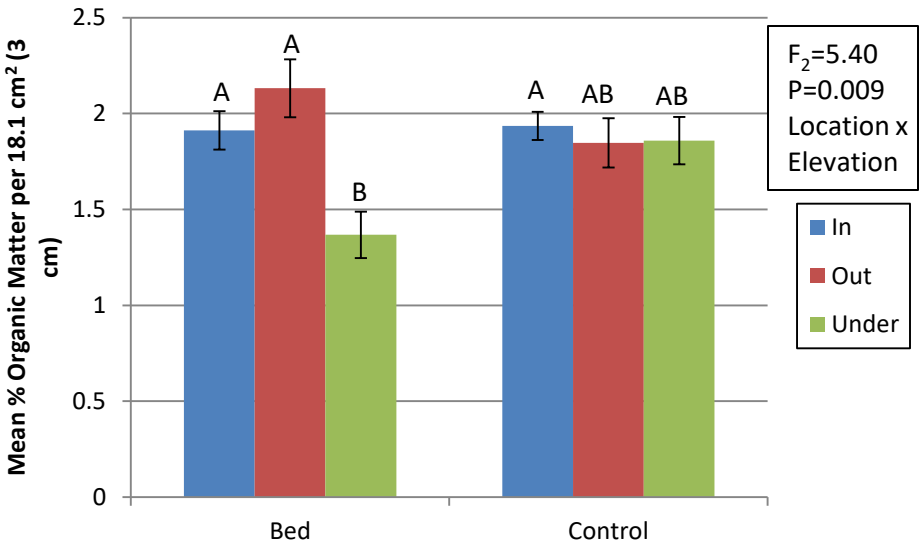

Figure 3. Mean organic matter of sediment replicates per treatment (June 2013) (Bed: In = Landward of Oyster shell, Out = Seaward of Oyster shell, Under = Under Oyster shell) (Control: In, Out, and Under are equivalent depths as Bed). Letters indicate significant differences from Tukey's test ($p < 0.05$). Standard error shown.

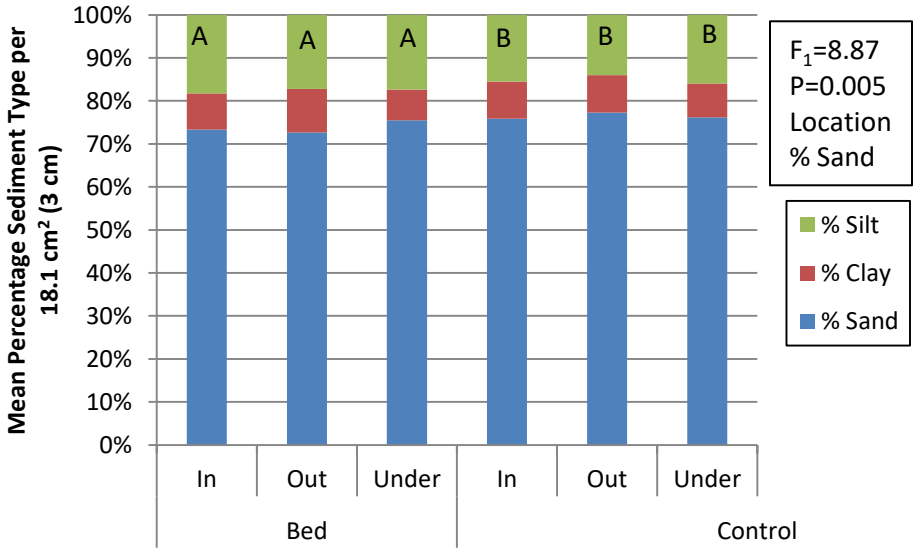

Figure 4. Mean sand, silt and clay percentages of sediment replicates per treatment (June 2013) (Bed: In = Landward of Oyster shell, Out = Seaward of Oyster shell, Under = Under Oyster shell) (Control: In, Out, and Under are equivalent depths as Bed). Letters indicate significant differences from Tukey's test ($p < 0.05$).

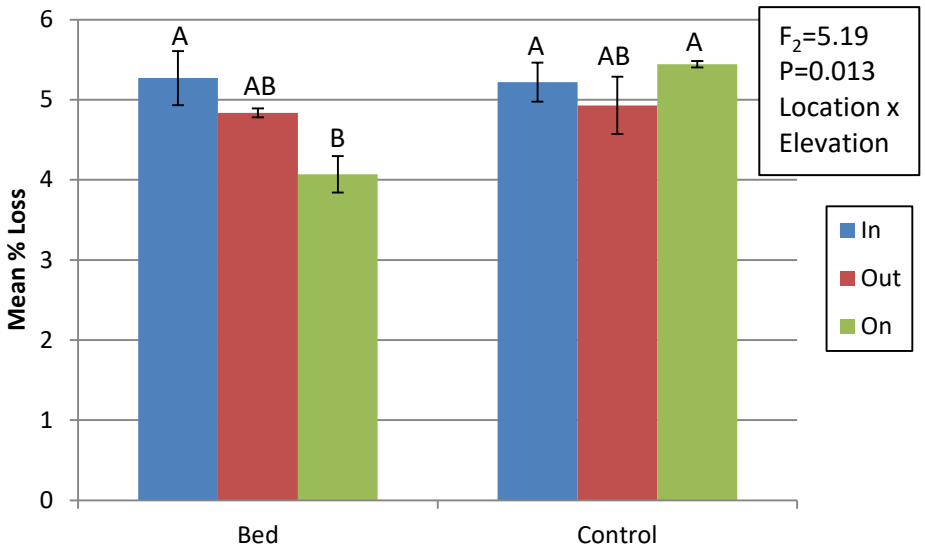

Figure 5. Mean mass loss from clod cards. (June 2013) (Bed: In = Landward of Oyster shell, Out = Seaward of Oyster shell, On = On Top of Oyster shell) (Control: In, Out, and On are equivalent depths as Bed.) Letters indicate significant differences from Tukey's test ($p < 0.05$). Standard error shown.

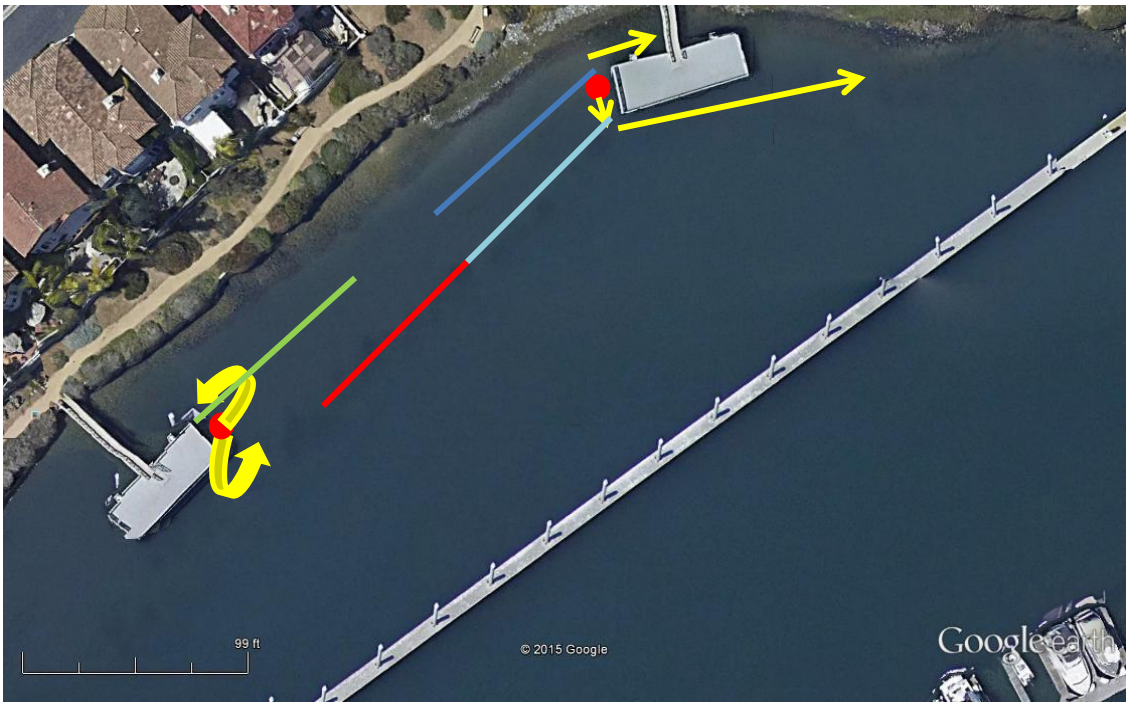

Figure 6. Fluorescein dye deployment during a flooding tide (10/02/2013). Dots indicate points of dye insertion. Arrows indicate general directional movement of the dye. Transects as previously stated.

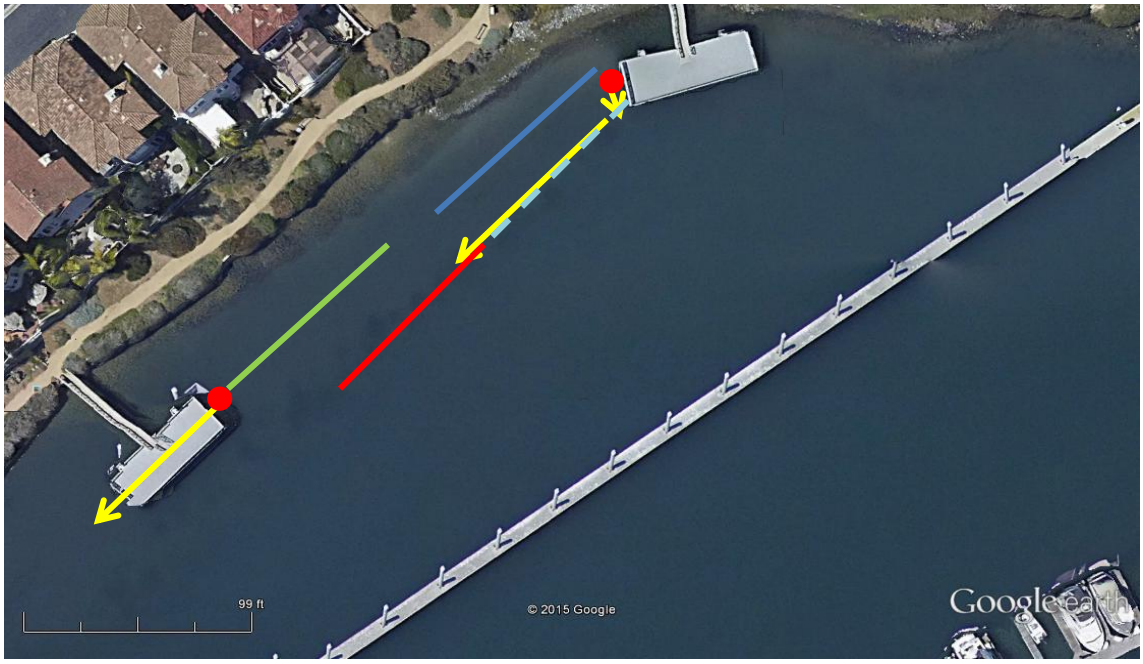

Figure 7. Fluorescein dye deployment during an ebbing tide (11/05/2014). Dots indicate points of dye insertion. Arrows indicate general directional movement of the dye. Transects as previously stated.

Total Abundance

The interaction of location and year (pre-oyster shell [2012] versus post-oyster shell [2013]) was significant (F_4=31.64, P<0.001, Fig 8, Tab 2). Within the bed treatment, total abundance was significantly lower in 2013 (post-oyster shell) as compared to 2012 (pre-oyster shell) (F_1=393.36, P<0.001, Fig 8). Within other treatments (control, near, far, basin6), no significant differences existed between years despite an increasing trend from 2012 to 2013 (Control: F_1=1.48, P=0.247, Near: F_1=2.75, P=0.123, Far: F_1=0.00, P=0.969, Basin6: F_1=1.53, P=0.240, Fig 8). The interaction of location and treatment (In, Out, and Under) (Post-oyster shell: control versus bed-associated) was significant (F_2=12.16, P<0.001, Fig 9). Within the bed treatment, total abundance was significantly lower underneath the bed as compared to inward or outward of the bed (F_2=18.00, P<0.001, Fig 9). Within the control treatment, no significant differences existed among treatments (In, Out, and Under) (F_2=0.75, P=0.486, Fig 9) (Table 2).

Tubificidae Abundance

The interaction of location and year (pre-oyster shell [2012] versus post-oyster shell [2013]) was significant (F_1=49.72, P<0.001, Fig 10). Within the bed treatment, 2013 was significantly lower in tubificid abundance than 2012 (F_1=59.27, P<0.001, Fig 10). Within the control treatment, no significant differences existed between 2012 and 2013 (F_1=1.50, P=0.245, Fig 10).

Capitellidae Abundance

The interaction of location and year (pre-oyster shell [2012] versus post-oyster shell [2013]) was significant (F_1=104.83, P<0.001, Fig 11). Within the bed treatment, 2013 was significantly lower in capitellid abundance than 2012 (F_1=61.43, P<0.001, Fig 11). Within the control treatment, 2013 was significantly higher in capitellid abundance than 2012 (F_1=54.10, P<0.001, Fig 11).

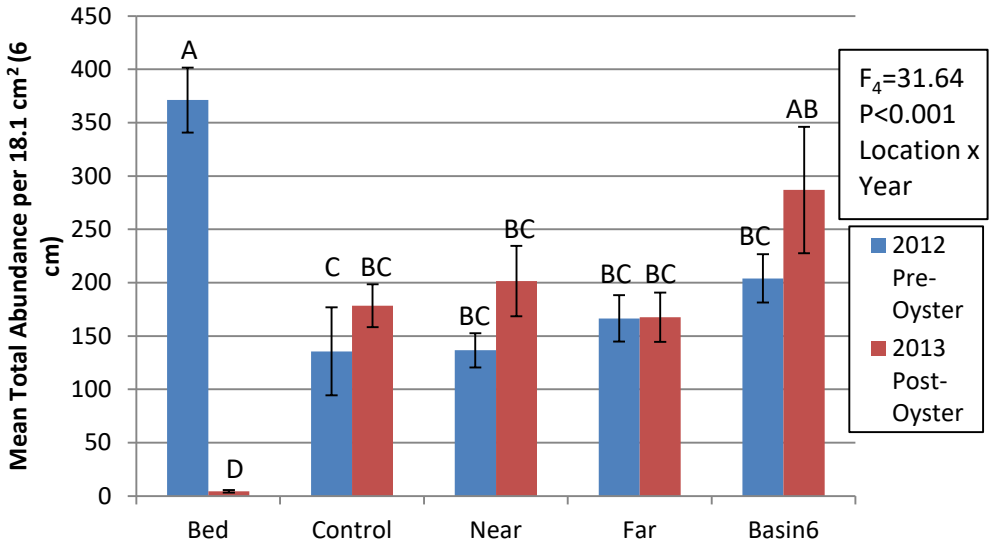

Figure 8. Mean abundance from June 2012 to June 2013 (pre- to post-restoration). Letters indicate significant differences from Tukey's test ($p < 0.05$). Standard error shown.

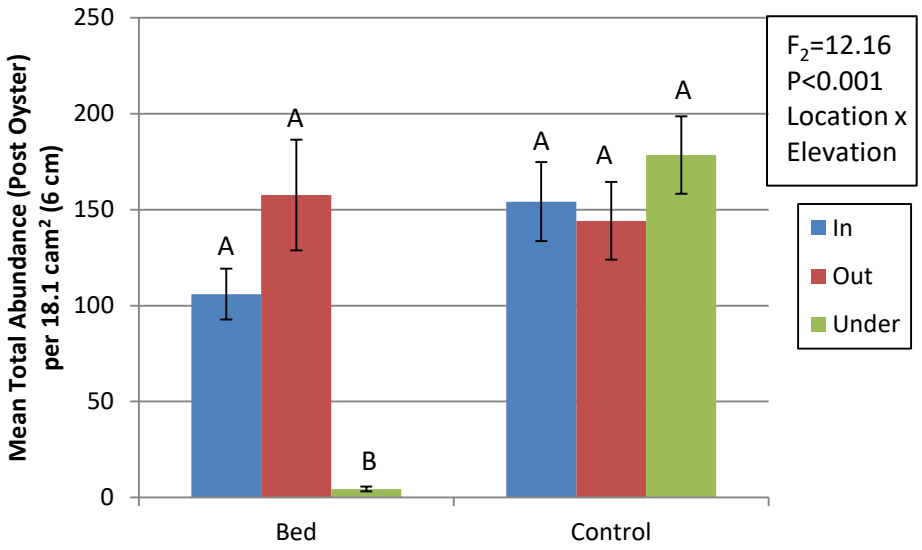

Figure 9. Mean abundance of infaunal replicates per treatment. (June 2013). Letters indicate significant differences from Tukey's test ($p < 0.05$). Standard error shown.

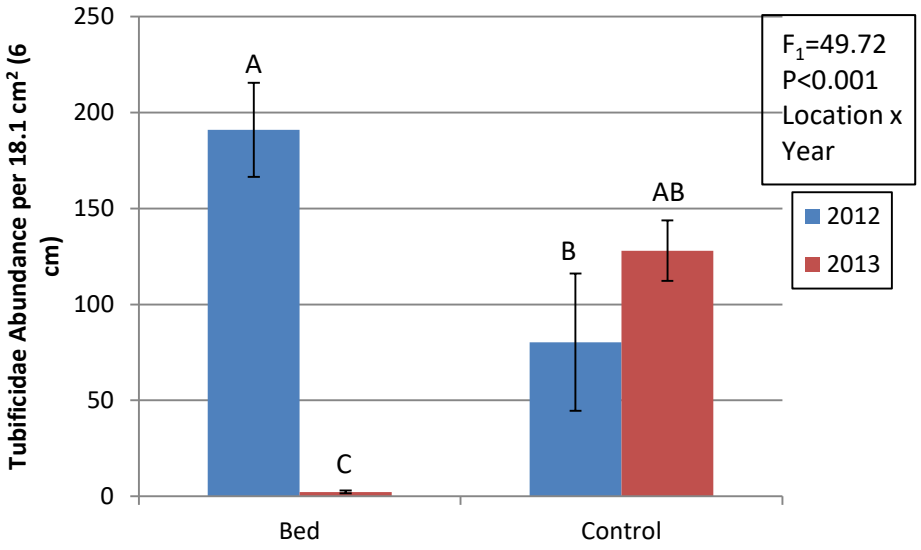

Figure 10. Mean Tubificidae abundance from June 2012 to June 2013 (pre- to post-restoration). Letters indicate significant differences from Tukey's test ($p < 0.05$). Standard error shown.

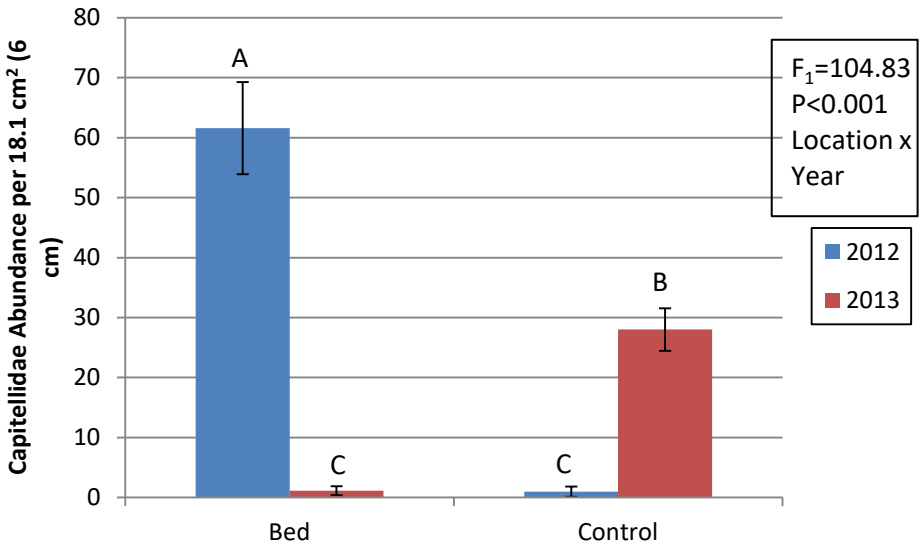

Figure 11. Mean Capitellidae abundance from June 2012 to June 2013 (pre- to post-restoration). Letters indicate significant differences from Tukey's test ($p < 0.05$). Standard error shown.

Species Richness

The interaction of location and year (pre-oyster shell [2012] versus post-oyster shell [2013]) was significant (F_4=8.59, P<0.001, Fig 12). Within the bed treatment, richness was significantly lower in 2013 (post-oyster shell) as compared to 2012 (pre-oyster shell) (F_1=169.36, P<0.001, Fig 12). Within other treatments (control, near, far, basin6), no significant differences existed between years (Control: F_1=0.89, P=0.364, Near: F_1=0.26, P=0.621, Far: F_1=0.11, P=0.751, Basin6: F_1=0.09, P=0.765, Fig 12). The interaction of location and treatment (In, Out, and Under) (Post-oyster shell [2013]: control versus bed-associated) was significant (F_2=20.95, P<0.001, Fig 13). Within the bed treatment, richness was significantly lower underneath the bed as compared to inward or outward of the bed and richness was significantly higher for outward as compared to inward or under the bed (F_2=37.75, P<0.001, Fig 13). Within the control treatment, no significant differences existed among treatments (In, Out, and Under) (F_2=0.05, P=0.953, Fig 13) (Table 2).

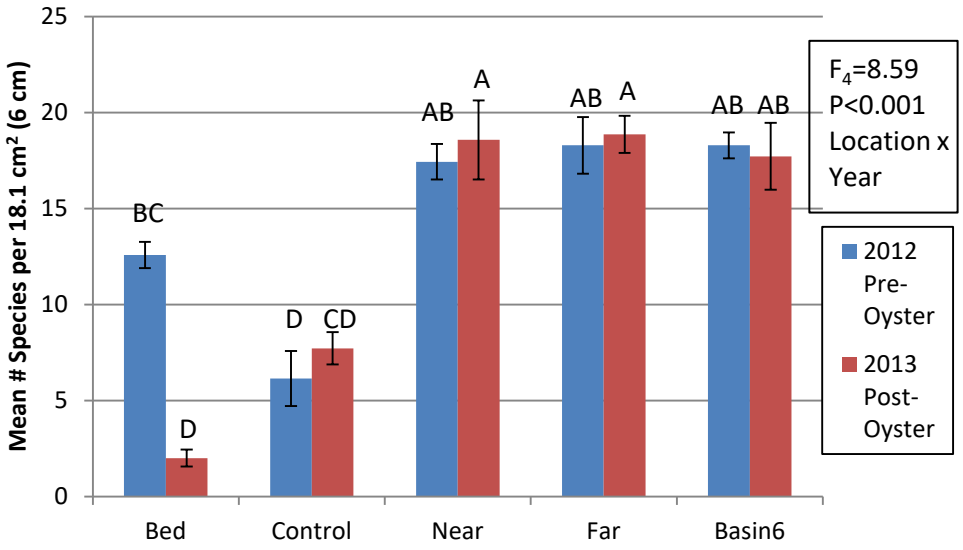

Figure 12. Mean species richness from June 2012 to June 2013 (pre- to post-restoration). Letters indicate significant differences from Tukey's test ($p < 0.05$). Standard error shown.

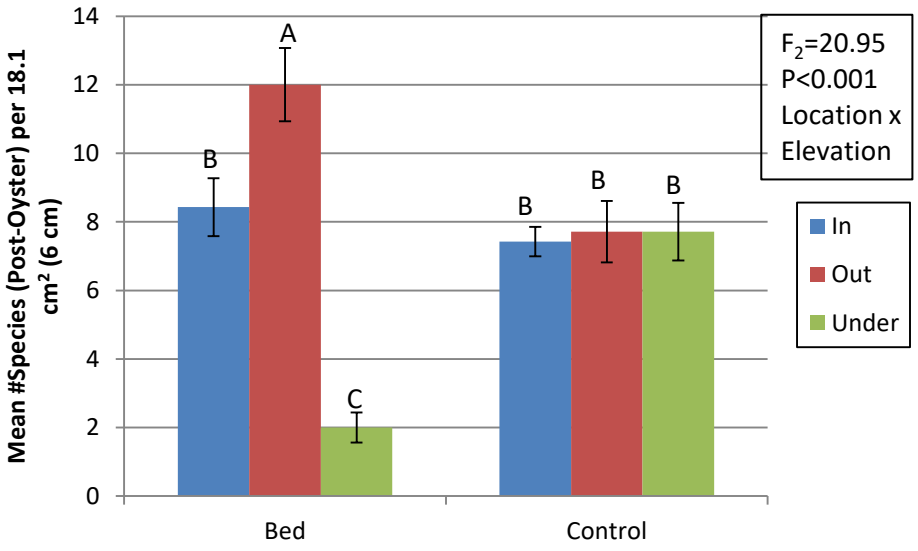

Figure 13. Mean species richness of infaunal replicates per treatment (June 2013). Letters indicate significant differences from Tukey's test ($p < 0.05$). Standard error shown

Shannon-Weiner Diversity (H')

The interaction of location and year (pre-oyster shell [2012] versus post-oyster shell [2013]) was significant (F_4=5.44, P=0.001, Fig 14). Within the bed treatment, H' was significantly lower in 2013 (post-oyster shell) as compared to 2012 (pre-oyster shell) (F_1=14.07, P=0.003, Fig 12). Within the control treatment, H' was significantly higher in 2013 (post-oyster shell) as compared to 2012 (pre-oyster shell) (F_1=7.16, P=0.02, Fig 14). Within other treatments (near, far, basin6), no significant differences existed between years (Near: F_1=0.11, P=0.751, Far: F_1=1.51, P=0.243, Basin6: F_1=0.02, P=0.896, Fig 14). The interaction of location and treatment (In, Out, and Under) (Post-oyster shell [2013]: control versus bed-associated) was significant (F_2=10.92, P=0.004, Fig 15). Within the bed treatment, H' was significantly lower underneath the bed as compared to inward or outward of the bed (F_2=9.14, P=0.002, Fig 15). Within the control treatment, no significant differences existed among treatments (In, Out, and Under) (F_2=1.62, P=0.225, Fig 15) (Table 3).

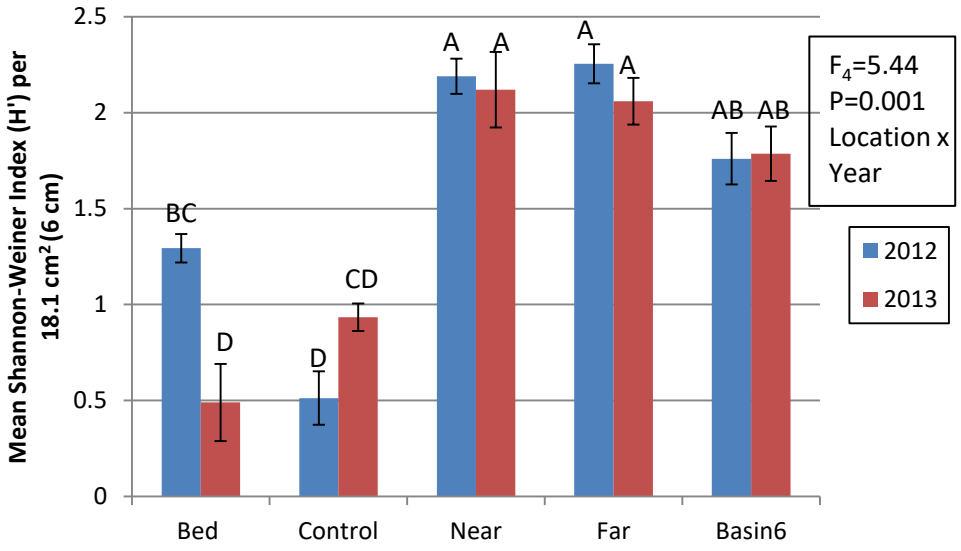

Figure 14. Mean Shannon-Weiner Index (H') from June 2012 to June 2013 (pre- to post-restoration). Letters indicate significant differences from Tukey's test ($p < 0.05$). Standard error shown.

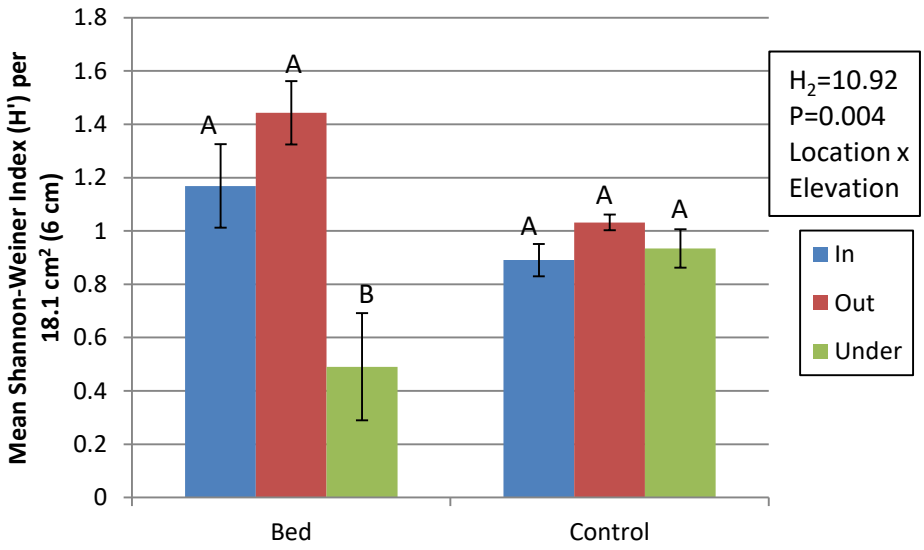

Figure 15. Mean Shannon-Weiner Index (H') of infaunal replicates per treatment (June 2013). Letters indicate significant differences from Tukey's test ($p < 0.05$). Standard error shown.

Reciprocal Simpson's Diversity (1/D)

No significant interaction existed between location and year (Table 3). Simpson's diversity was also not significantly different between years. However, the near and far treatments were significantly higher than all other locations and Basin 6 was significantly higher than control, (F_4=27.17, P<0.001, Fig 16). Post-oyster shell [2013], within the bed treatment, no significant differences existed among treatments (In, Out, and Under) (F_2=1.82, P=0.190, Fig 17). Within the control treatment, no significant differences existed among treatments (In, Out, and Under) (F_2=1.22, P=0.318, Fig 17) (Table 3).

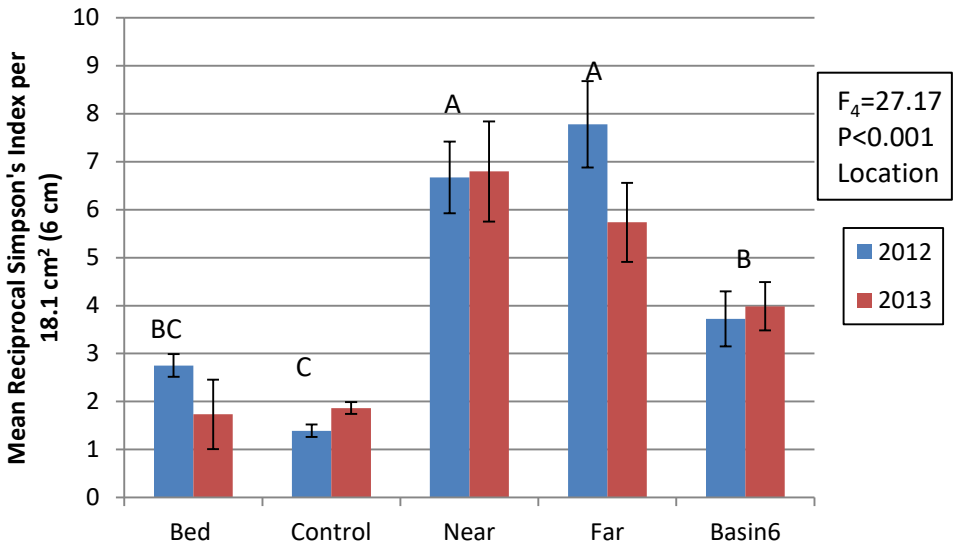

Figure 16. Mean Reciprocal Simpson's Index (1/D) from June 2012 to June 2013 (pre- to post-restoration). Letters indicate significant differences from Tukey's test ($p < 0.05$). Standard error shown.

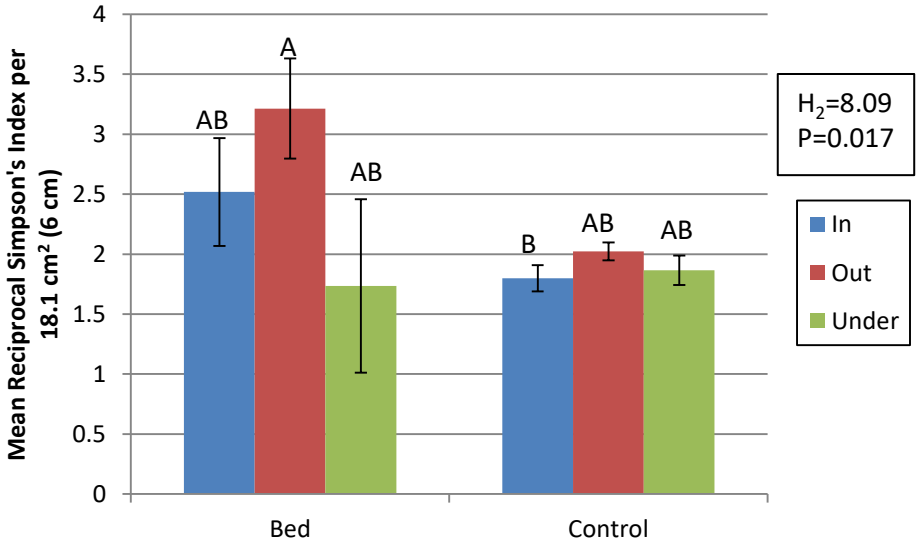

Figure 17. Mean Reciprocal Simpson's Index (1/D) of infaunal replicates per treatment (June 2013). Letters indicate significant differences from Tukey's test ($p < 0.05$). Standard error shown.

Community Structure

Average non-metric multi-dimensional scaling plot (MDS) of taxa found shows community change for all treatments from 2012 to 2013 (pre- to post-restoration) (ANOSIM, R=0.448, P=0.001, Fig 18). SIMPER analysis shows that Tubificidae oligochaetes are driving 27.99% of the difference within the bed treatment which had an average dissimilarity of 91.48% from 2012 to 2013 (pre- to post-restoration) (Table 4). Tubificidae oligochaetes are driving 19.74% of the difference within the control treatment which had an average dissimilarity of 65.16% from 2012 to 2013 (pre- to post-restoration) (Table 4). The qualitative trajectory of change from 2012 to 2013 (pre- to post-restoration) for bed treatment is remarkably different than trajectories of other treatments (Fig 18). Oligochaetes and Polychaetes made up the highest percentage of found species (Figs 22 and 23). Lists of specific species' abundances can be found in the appendix.

Explanatory Relationships

Spearman's rho correlation showed that there was a positive correlation between percent mud and percent organic matter ($\rho=0.709$, $p<0.001$, Fig 19) and between percent mud and Tubificidae abundance from all treatments in 2013 ($\rho=0.467$, $p=0.011$, Fig 20). Pearson's *r* correlation showed that there was a positive correlation between percent mud and Capitellidae abundance from all treatments in 2013 ($r=0.346$, $p=0.042$, Fig 21).

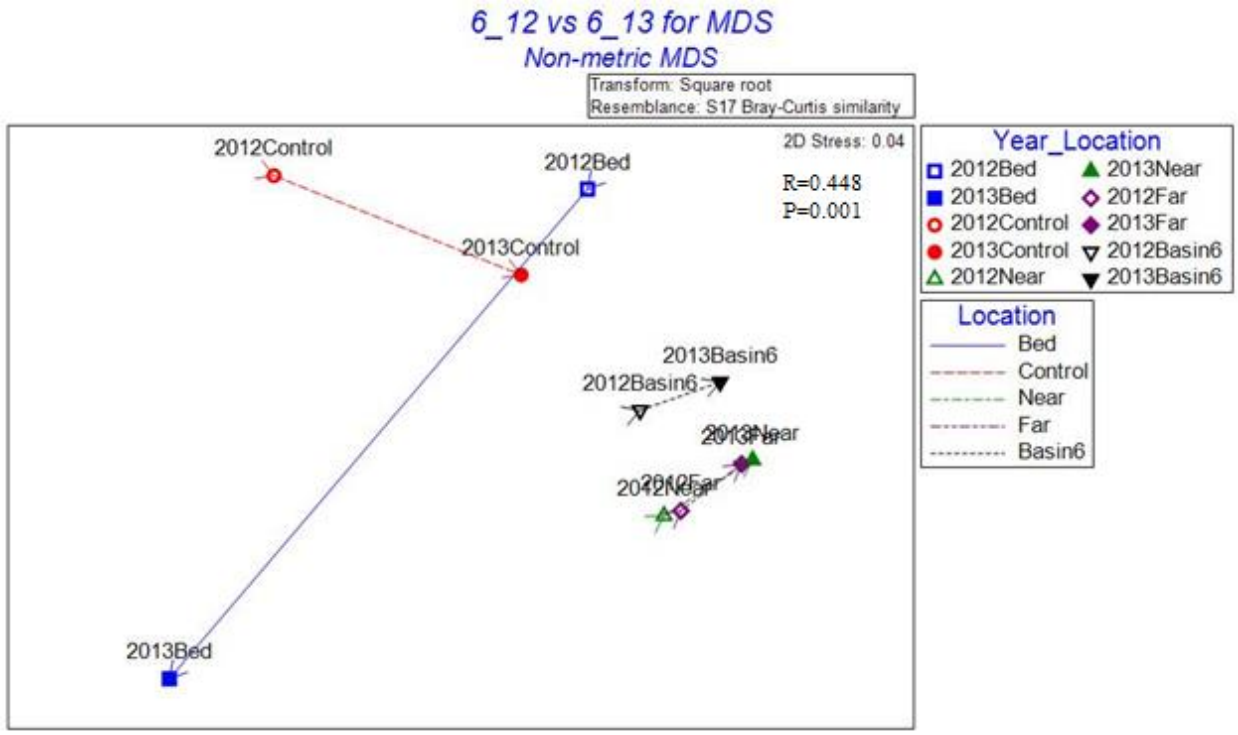

Figure 18. Average MDS plot showing change in community structure in all treatments from June 2012 to June 2013 (pre- to post-restoration) ($p < 0.05$).

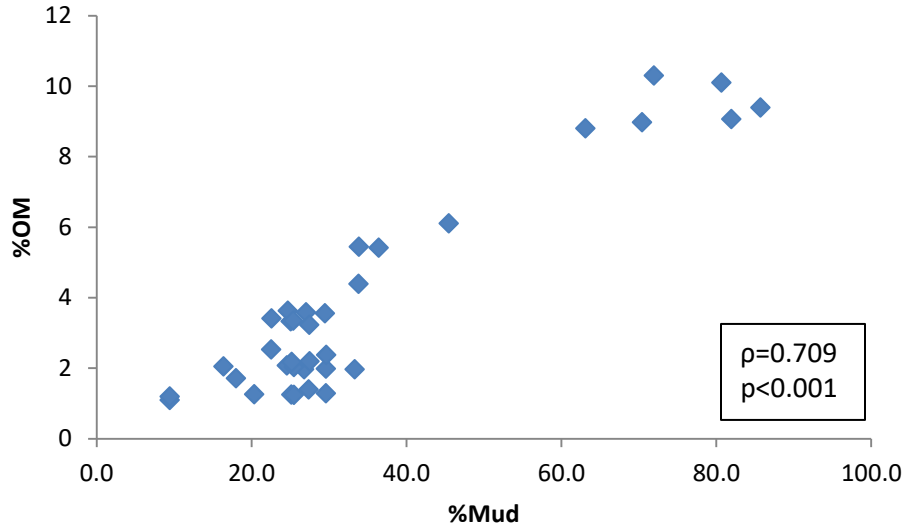

Figure 19. Spearman's rho correlation plot of percent mud and percent organic matter from all treatments in 2013 ($p < 0.05$).

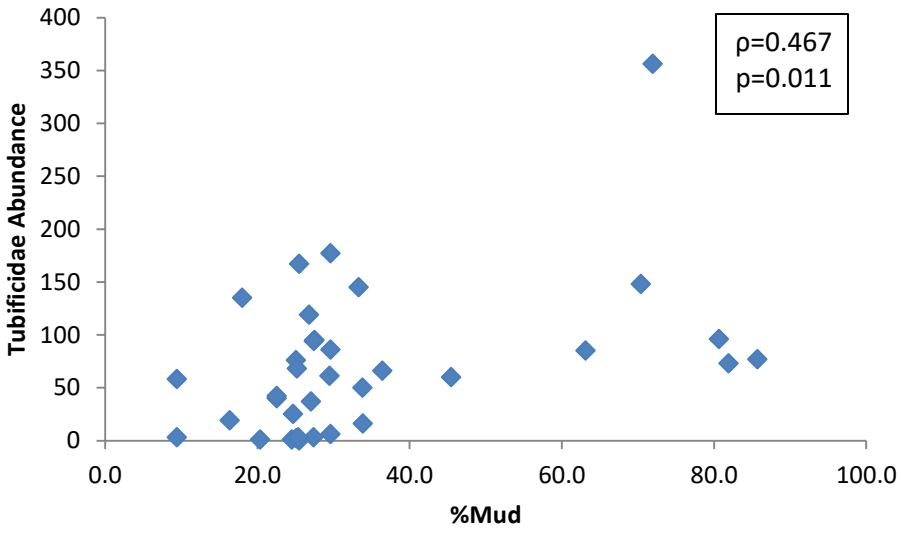

Figure 20. Spearman's rho correlation plot of percent mud and Tubificidae abundance from all treatments in 2013 ($p < 0.05$).

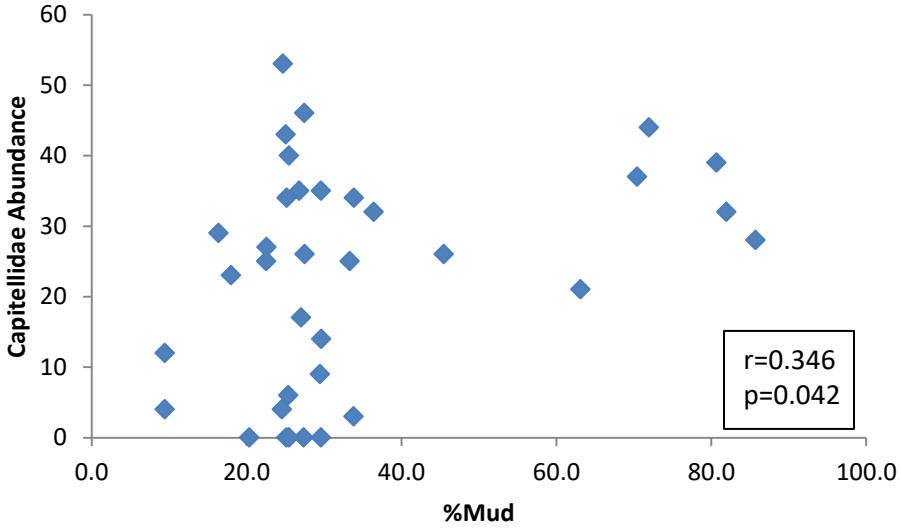

Figure 21. Pearson's *r* correlation plot of percent mud and Capitellidae abundance from all treatments in 2013 ($p < 0.05$).

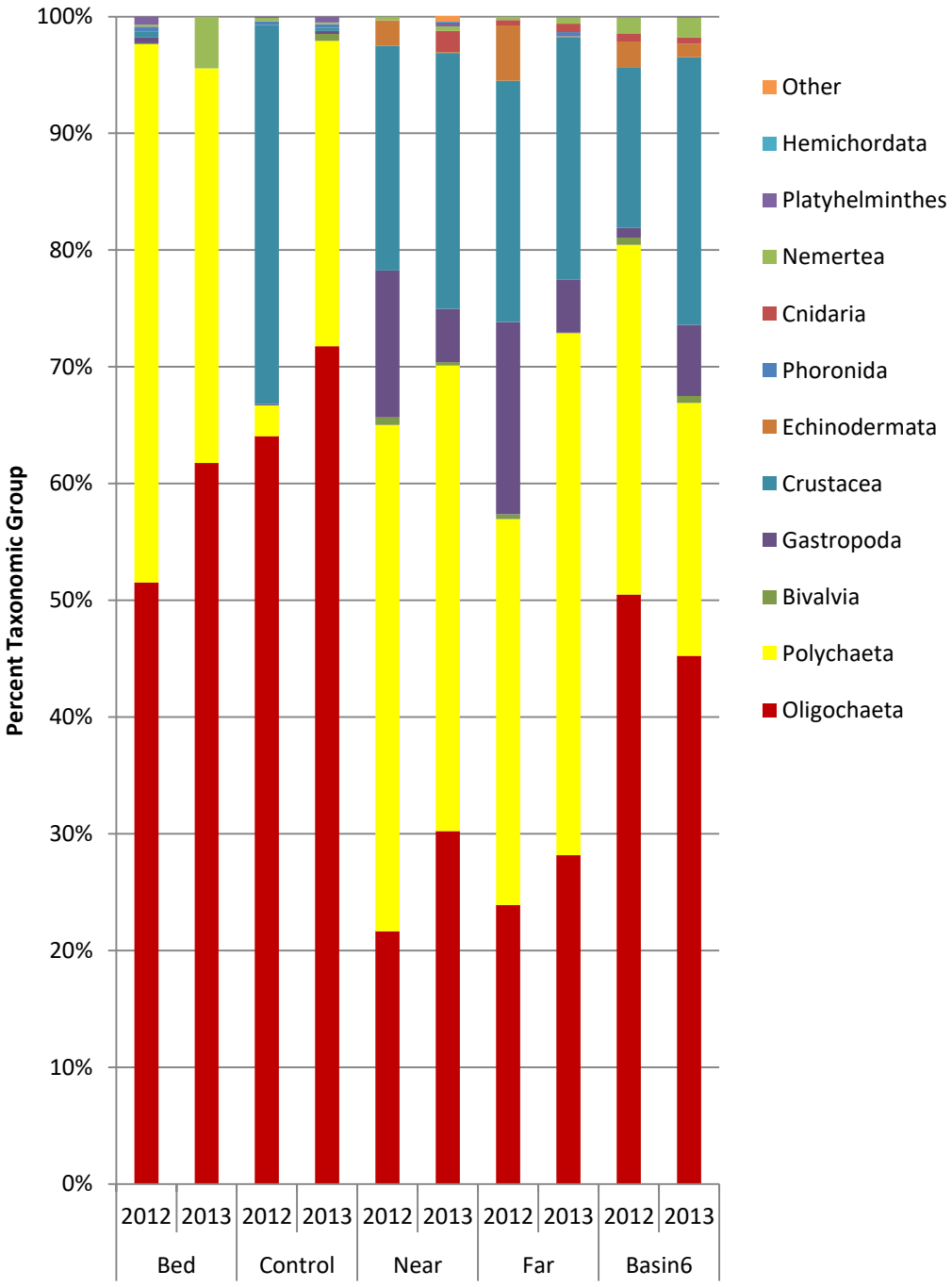

Figure 22. Percent of total macrofauna by larger taxonomic groups from June 2012 to June 2013 (pre- to post-restoration).

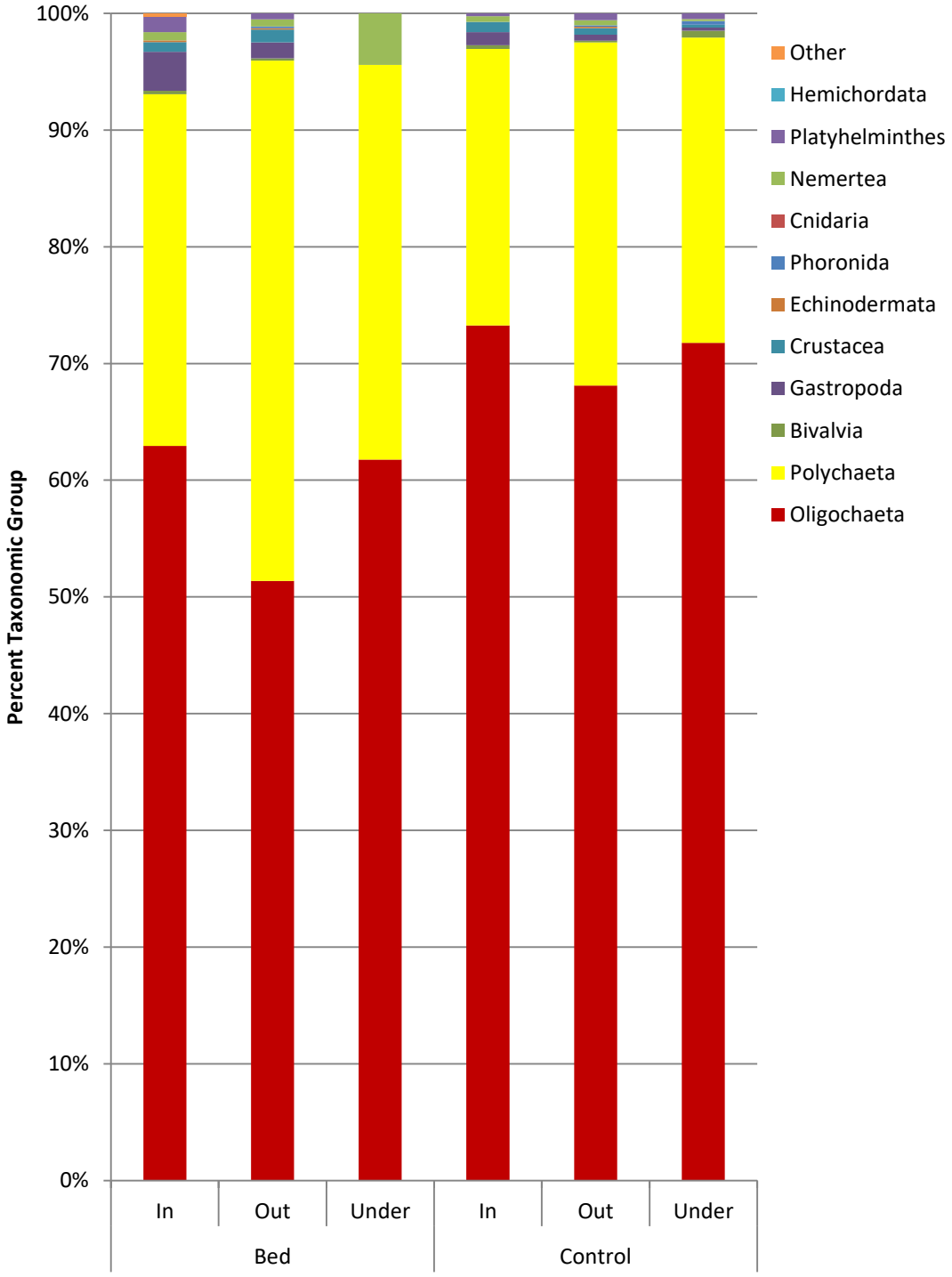

Figure 23. Percent of total macrofauna by larger taxonomic groups of infaunal replicates per treatment (June 2013).

Table 1. ANOVA Table with Sediment Composition and Flow Rate Data

Metric	Source	DF	adj SS	adj MS	F	P
Percent Organic Matter	Location	1	0.06186	0.06186	0.63	0.434
	Treatment	2	1.13234	0.56617	5.73	0.007
	Loc*Treat	2	1.06771	0.53386	5.40	0.009
	Residual	36	3.55768	0.09882		
Percent Sand	Location	1	238.74	238.74	8.87	0.005
	Treatment	2	10.84	5.42	0.20	0.819
	Loc*Treat	2	23.99	11.99	0.45	0.644
	Residual	36	969.47	26.93		
Percent Clod Card Loss	Location	1	1.6750	1.6750	5.63	0.026
	Treatment	2	1.2904	0.6452	2.17	0.136
	Loc*Treat	2	3.0889	1.5445	5.19	0.013
	Residual	24	7.1377	0.2974		

Table 2. ANOVA Table with Infaunal Abundance and Richness Data

Metric	Source	DF	adj SS	adj MS	F	P
Total Abundance (under bed: 2012 and 2013)	Location	4	70323	17581	2.67	0.041
	Year	1	21437	21437	3.25	0.076
	Loc*Year	4	494425	123606	18.74	<0.001
	Residual	60	395686	6595		
Total Abundance (In, Out, Under: 2013 only)	Location	1	50822	50822	19.35	0.000
	Treatment	2	25466	12733	4.85	0.014
	Loc*Treat	2	63887	31944	12.16	<0.001
	Residual	36	94569	2627		
Species Richness (under bed: 2012 and 2013)	Location	4	2067.66	516.91	48.88	<0.001
	Year	1	43.21	43.21	4.09	0.048
	Loc*Year	4	363.43	90.86	8.59	<0.001
	Residual	60	634.57	10.58		
Species Richness (under bed: 2012 and 2013)	Location	1	0.214	0.214	0.05	0.825
	Treatment	2	178.048	89.024	20.51	<0.001
	Loc*Treat	2	181.857	90.929	20.95	<0.001
	Residual	24	156.286	4.341		

Table 3. ANOVA Table with Infaunal Diversity Data

Metric	Source	DF	adj SS	adj MS	F	P
H' (under bed: 2012 and 2013)	Location	4	26.5725	6.6431	54.42	<0.001
	Year	1	0.2710	0.2710	2.14	0.149
	Loc*Year	4	2.7582	0.6896	5.44	0.001
	Residual	60	7.6036	0.1267		
H' (In, Out, Under: 2013 only)	Location	1	0.0706	0.0706	0.68	0.414
	Treatment	2	1.9606	0.9803	9.49	<0.001
	Loc*Treat	2	1.4787	0.7394	7.16	0.002
	Residual	36	3.7194	0.1033		
1/D (under bed: 2012 and 2013)	Location	4	328.892	82.223	27.17	<0.001
	Year	1	3.390	3.390	1.12	0.294
	Loc*Year	4	15.893	3.973	1.31	0.276
	Residual	60	181.605	3.027		
1/D (under bed: 2012 and 2013)	Location	1	3.694	3.694	3.40	0.073
	Treatment	2	4.697	2.348	2.16	0.130
	Loc*Treat	2	3.129	1.565	1.44	0.250
	Residual	24	39.117	1.087		

Table 4. SIMPER Analyses for Community Differences

Treatment	**Avg. Dissimilarity (%)**	**Lead Driver**	**Contribution (%)**	**Tubificidae Contribution (%)**	**Capitellidae Contribution (%)**
Bed	**91.48**	**Tubificidae**	**27.99**	**---**	**22.94**
Control	**65.16**	**Tubificidae**	**19.74**	**---**	**20.5**
Near	**49.72**	***Fabricia stellaris***	**5.88**	**4.9**	**7.42**
Far	**45.91**	***Barleeia sp***	**7.29**	**6.56**	**5.54**
Basin6	**36.13**	**Tubificidae**	**9.6**	**---**	**7.59**

CHAPTER 4

DISSCUSSION

Oyster Bed Impacts

The first goal of this study was to assess if oyster bed restoration would affect the surrounding sediment and organic matter of the restoration site mudflat (hypotheses 1 a-c). I hypothesized that this could occur uniformly across the site or due to alterations in the flow regime by the oyster shells themselves, it could occur differently on the landward versus seaward sides of the bed. I found that the sediment on the bed treatment did not differ among landward, seaward or under bed locations, but was higher in sand compared to the control. This difference between the locations could be indicative of the differing flow patterns between the bed and control treatments as observed at flooding tide; the water flowed past the bed treatment while remaining entrained around the control treatment. Faster moving water can deposit larger grain size particles (e.g. sand) while large and smaller grain size particles can settle out of slower moving water. The bed itself did not appear to affect how sediment was deposited within the bed site. However, future studies should include fine scale flow meters and sediment accumulation tiles placed at different locations within the study area.

The organic matter content of the sediment showed no significant differences based solely on location. Differences among treatment (In, Out, and Under) were significant, but so was the interaction of location and treatment. The sediment under the bed structure showed a significantly lower organic matter content than the landward and seaward areas on the bed. This difference is likely due to the shell blocking sediment deposition onto the mud or potentially due

to loss of invertebrates and their contribution to organic matter under the bed. The clod cards showed that there was no significant difference in flow between bed and control or among the treatments (In, Out, and On) at control. However flow was significantly higher on the bed compared among the treatments. This corresponds to the significantly lower organic matter under the bed; on face value there would seem to be some correlation. I hypothesized that flow was the mechanism for organic matter content differences, but there may be some other influence on organic matter because the clod cards were placed on top of the shell bed while the organic matter was collected from under it. Higher flow may have been an artifact of clod card placement; since they were elevated in the water column they may have had higher flow exposure.

The second goal of this study was to assess if the oyster bed restoration would affect the infaunal invertebrate communities of the restoration site mudflat directly under and surrounding the constructed oyster bed. The impact of the shell on the mudflat is limited to directly under the shell: infaunal abundance, richness, and diversity was lower directly under the shell, but not in the areas surrounding the shell. In addition, because there was no difference between sampling years at either the control or eelgrass treatments, I can attribute this change to the presence of the oyster bed. It is likely that the under bed samples were lower in invertebrates than the surrounding sediment because of the physical barrier the shell creates between the water and the sediment. Other studies have demonstrated that a barrier blocks the transfer of oxygen, causing hypoxia or an anoxic environment, which is a limiting condition for infaunal invertebrates (Reidenbach et al., 2013). Another future direction could be to make oxygen measurements in and around an Olympia oyster bed to further test this hypothesized mechanism.

In assessing community dynamics it was important to understand what species were behind any observed changes in the community; therefore I looked at capitellid polychaetes specifically because of their ability to better handle high organic-pollutant impacted environments and their affinity for fine grained sediment (Pearson and Rosenberg, 1978; Giangrande et al., 2005; Markert et al., 2010). I found that capitellid polychaetes disappeared from the under-bed community from 2012 to 2013 (pre- to post-restoration). Tubificid oligochaetes also declined in the under bed samples; however, while at low abundances, they were the only species present in this location. Like capitellids, tubificids are better adapted to environments with high organic pollutants and fine grained sediments (Brinkhurst and Kennedy, 1965; Leynen et al., 1999; Markert et al., 2010). The decrease seen under the bed may show the movement of these groups to interstitial space on the shell bed, not a localized extinction of these groups (Markert et al., 2010) (Zacherl unpub.). This can be tested by examining sorted invertebrates collected from the shells themselves; this work is underway.

Importance of Site Selection

As part of this experimental design, a control treatment (only 10m away, constrained to be within the same reserve) was selected without exploratory invertebrate sampling. My invertebrate samples were collected immediately pre-construction. Once my samples were sorted my data showed less sand at the bed, different water flow, and differences in the invertebrate community between the control and pre-construction mudflat, indicating that the control treatment selection did not have equivalent starting conditions to the bed site. However, because of my use of classic BACI design (Before-After Control-Impact), I was able to determine the change at a specific impact site (e.g. constructed oyster bed), without equivalent starting conditions at the experimental control. I was able to use the control treatment as a metric of

annual sampling variability (i.e. did the control vary from 2012 to 2013) (pre- to post-restoration). Since no significant change occurred in the control, I can attribute the change seen in the bed to the impact of the oyster shell and not random change. BACI only allows me to make inferences about this location, but my findings may be relevant to Olympia restoration in general. However, the pre-construction differences emphasize the importance of careful treatment selection, beyond just proximity and appearance, in both experimental design and in restoration projects.

My study demonstrates the importance of sediment and community conditions of the restoration habitat in interpreting oyster restoration programs. Eelgrass beds and invertebrate populations are spatially and temporally heterogeneous within treatments, as demonstrated within JDMR. The dynamics of invertebrates are heavily influenced by sediment parameters and other local environmental conditions; yet many oyster restoration programs do not consider or quantify these parameters during planning. A comprehensive understanding of site conditions prior to construction planning would improve ability to develop site-specific restoration plans.

Ecosystem Impacts

Other studies show that oysters can improve seagrass development through the act of cleaning and clarifying the water. (Newell, 2004; Grabowski and Peterson, 2007; van der Heide et al., 2012). Recent research supports the idea that oysters can enhance seagrass development outbound of the actual oyster bed. My study included invertebrate community measurements within an eelgrass bed just outward of the created oyster bed (hypothesis 3). While no significant increases in either eelgrass density (Briley et al. unpub.) or in the invertebrate abundances were observed, it is worth noting that no significant declines or community composition changes in the invertebrate community occurred either. This is an excellent process control for construction

impacts. Based on my results, oyster bed restoration does not negatively impact eelgrass beds that are as close as two meters away from the restoration site within Jack Dunster Marine Reserve. This implies likelihood that the same would apply to other oyster restorations near eelgrass beds. With time, I expect to see increased Olympia oyster settlement. With increased settlement, the parameters of my study could be evaluated again to determine the actual impact of these oysters.

Management Implications

In this study, I had the opportunity to evaluate the impacts of oyster bed creation on the infaunal invertebrate community on an actual restoration project using BACI design. This repeated measures design is a good way to assess the impact of an oyster bed restoration in a small study site. Because I was evaluating a real world restoration project, I had only one site within one bay along one oyster bed. This lack of spatial replication restricts any inferences made about the effects of oyster restoration to my study site. Future studies should look at multiple Olympia restorations to add spatial replication. Doing this could allow inferences to be made on the effects of oyster restoration in general. However, pairing science with ongoing restoration offers a unique opportunity to evaluate realistic projects and influence future projects.

In a site evaluation, a parameter should be added that describes the invertebrate community. The creation of an oyster bed on a mud flat could lead an invertebrate community to transition from one of infauna to epifauna, the invertebrates that live on top of a substratum. Managers should incorporate pre-site evaluation of infaunal invertebrate communities because they may be a valuable food source to species of concern. If potential sites are foraging grounds for sensitive or endangered species (e.g. Ridgeway's rail) that rely on infaunal resources, placement of a bed on their food source might cause species of concern to become more

threatened then they already are and contribute to species loss. Oyster beds do supply habitat to epifauna, but transitions to epifauna are not the same as being infauna. Different consumers prey on epifauna than the ones that prey on infauna because the accessibility for food to different. Infauna in my study may not shift to epifauna because of the presence of the oyster shell; infauna may however shift locations and become inaccessible to different consumers because of micro-habitats created between shells, as seen in Markert's study where *Crassostrea* shells formed stable micro-habitats in the shell-space matrix (Markert et al., 2010). This is not necessarily negative for the infauna but may be negative for the consumers that rely on infaunal sources, due to restricted access.

In conclusion, it was unique to evaluate impacts of Olympia oyster bed restoration on the associated sediment and infauna especially underneath the constructed shell bed. My study has demonstrated that the impact is limited to just under the constructed bed. Yes, this was the first time this question was answered for Olympia oysters, but it is important to understand these impacts, not only for this study but for potential other restorations because of potential impacts to species of concern.

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APPENDIX

Table 5. June 2012 Infaunal abundances (mean ± 1 S. E.).

June 2012									
Lowest Taxon Identified	Bed			Control			Near	Far	Basin6
	In	Out	(Pre)Under	In	Out	(Pre)Under			
Tubificidae			190.86 ± 24.48			80.29 ± 35.72	28.71 ± 5.76	40.43 ± 6.89	108.57 ± 21.74
Spionidae									2.86 ± 2.86
Pseudopolydora paucibranchiata			1.57 ± 0.43				3.00 ± 0.72	5.29 ± 1.76	9.43 ± 3.15
Polydora ligni								0.29 ± 0.29	
Polydora nuchalis			1.43 ± 0.37			0.14 ± 0.14			0.14 ± 0.14
Polydora sp.									
Streblospio benedicti			99.71 ± 17.56			1.14 ± 0.99	0.29 ± 0.29	0.29 ± 0.18	0.29 ± 0.29
Spiophanes sp.			0.14 ± 0.14						
Spiophanes duplex								0.14 ± 0.14	
Prionospio lighti			0.43 ± 0.30			0.14 ± 0.14	0.43 ± 0.43		0.57 ± 0.20
Prionospio steenstupi								0.57 ± 0.43	0.14 ± 0.14
Apoprionospio pygmaea							1.29 ± 0.61	0.86 ± 0.46	
Mediomastus ambiseta			21.29 ± 5.06				31.43 ± 8.72	24.57 ± 4.67	24.86 ± 2.52
Mediomastus californiensis								0.14 ± 0.14	
Capitella capitata			39.86 ± 6.46			1.00 ± 0.85	4.57 ± 3.75	0.29 ± 0.18	4.86 ± 1.20
Notomastus tenuis			0.43 ± 0.43					0.43 ± 0.43	0.29 ± 0.18
Exogone spp.			1.86 ± 0.59			0.29 ± 0.18	8.86 ± 1.94	10.43 ± 1.56	1.29 ± 0.42
Sphaerosyllis spp.							1.14 ± 0.70	3.71 ± 0.75	2.86 ± 1.06
Typosyllis alternata						0.86 ± 0.55	0.14 ± 0.14		
Typosyllis sp.							1.57 ± 0.72	0.29 ± 0.29	0.14 ± 0.14
Eteone californica			0.86 ± 0.46			0.29 ± 0.18			
Phyllodocidae			0.14 ± 0.14						
Fabricia stellaris			2.14 ± 0.74				2.86±1.61		
Euchone limnicola									0.43 ± 0.20
Pista pacifica							1.00 ± 0.53	0.14 ± 0.14	0.14 ± 0.14

Table 6. June 2012 Infaunal abundances continued (mean ± 1 S. E.).

Lowest Taxon Identified	Bed			Control			Near	Far	Basin6
	In	Out	(Pre)Under	In	Out	(Pre)Under			
Amaeana occidentalis								0.14 ± 0.14	
Aphelochaeta sp.								0.14 ± 0.14	0.14 ± 0.14
Protocirrineris socialis									0.29 ± 0.29
Polycirrus sp.							0.14 ± 0.14		
Dorvillea sp.							3.00 ± 1.40	1.29 ± 0.64	0.14 ± 0.14
Scoletoma zonata								0.29 ± 0.18	
Scoletoma sp.							0.14 ± 0.14		0.14 ± 0.14
Nephtys caecoides									0.14 ± 0.14
Nereis procera			2.00 ± 0.93			0.43 ± 0.20	0.29 ± 0.29	0.43 ± 0.30	3.57 ± 1.07
Glycera nana			0.29 ± 0.18						
Goniada brunnea							0.43 ± 0.30	0.86 ± 0.55	
Leitoscoloplos pugettensis								0.71 ± 0.71	0.43 ± 0.30
Cossura pygodactylata							0.14 ± 0.14	0.71 ± 0.47	0.71 ± 0.57
Clymenella californica								0.43 ± 0.43	2.14 ± 0.51
Bivalvia			0.14 ± 0.14				0.71 ± 0.29	0.57 ± 0.20	1.29 ± 0.52
Myidae			0.14 ± 0.14						
Gastropoda									0.29 ± 0.18
Barleeia sp.						0.14 ± 0.14	17.29 ± 4.58	28.43 ± 6.47	1.14 ± 0.40
Acteocina sp.			1.71 ± 0.78				0.14 ± 0.14		0.14 ± 0.14
Bulla gouldiana						0.14 ± 0.14			
Lottia paleacea								0.14 ± 0.14	
Hemigrapsus oregonensis			0.29 ± 0.18			0.57 ± 0.57			
Hippolyte californiensis							0.29 ± 0.18	0.14 ± 0.14	0.14 ± 0.14
Heptacarpus sp.								0.14 ± 0.14	
Grandidierella japonica			0.71 ± 0.47			0.29 ± 0.18	10.29 ± 2.44	17.00 ± 1.91	16.00 ± 5.21
Protohyale frequens			0.57 ± 0.43			45.29 ± 23.36	0.71 ± 0.71		

Table 7. June 2012 Infaunal abundances continued (mean ± 1 S. E.).

Lowest Taxon Identified	Bed			Control			Near	Far	Basin6
	In	Out	(Pre)Under	In	Out	(Pre)Under			
Eusiroidea						0.14 ± 0.14			
Deutella californica							0.14 ± 0.14	0.57 ± 0.20	
Hemiproto sp.							0.14 ± 0.14		0.14 ± 0.14
Mayerella banksia			0.14 ± 0.14				0.29 ± 0.29	0.14 ± 0.14	
Leptochelia dubia			0.14 ± 0.14				2.14 ± 1.10	0.71 ± 0.36	
Zeuxo normani						0.14 ± 0.14	5.71 ± 0.94	6.14 ± 2.74	0.43 ± 0.20
Paracerceis sculpta							0.14 ± 0.14	0.86 ± 0.40	
Uromunna ubiquita								0.14 ± 0.14	0.14 ± 0.14
Paranthura elegans									0.57 ± 0.30
Heteroserolis carinata							0.14 ± 0.14		
Califanthura squamosissima							0.14 ± 0.14		
Nebalia sp.						3.29 ± 1.58			
Ophiuroidea							2.86 ± 1.14	9.00 ± 3.21	4.00 ± 1.05
Phoronida			1.14 ± 1.14			0.29 ± 0.29			
Anthozoa								0.14 ± 0.14	1.57 ± 0.69
Cladonema sp.								0.43 ± 0.30	
Copepoda								0.14 ± 0.14	
Ostracoda							5.57 ± 1.09	8.43 ± 1.62	10.86 ± 1.81
Nemertea			0.57 ± 0.30			0.43 ± 0.20	0.43 ± 0.20	0.43 ± 0.20	2.57 ± 0.97
Platyhelminthes			2.57 ± 1.31			0.29 ± 0.18			0.14 ± 0.14

Table 8. June 2013 Infaunal abundances (mean ± 1 S. E.).

June 2013									
Lowest Taxon Identified	Bed			Control			Near	Far	Basin6
	In	Out	Under	In	Out	Under			
Tubificidae	67.71 ± 12.57	87.86 ± 25.93	2.29 ± 0.75	115.71 ± 19.50	98.86 ± 15.23	128.00 ± 15.68	49.57 ± 6.92	48.00 ± 13.67	127.86 ± 39.49
Pseudopolydora paucibranchiata	0.29 ± 0.18	0.43 ± 0.30		0.14 ± 0.14	0.57 ± 0.30	0.43 ± 0.20	6.29 ± 1.69	3.57 ± 1.07	1.0 ± 0.44
Polydora ligni							0.14 ± 0.14		
Polydora nuchalis	0.14 ± 0.14								
Polydora sp.				0.14 ± 0.14					
Streblospio benedicti	7.00 ± 2.85	7.00 ± 3.46	0.14 ± 0.14	11.43 ± 3.08	10.43 ± 3.41	12.57 ± 2.05	0.29 ± 0.29	1.00 ± 0.22	
Prionospio lighti		0.29 ± 0.29			0.43 ± 0.43	0.14 ± 0.14	0.14 ± 0.14	0.43 ± 0.30	
Prionospio steenstupi		0.14 ± 0.14		0.14 ± 0.14			1.86 ± 0.74	2.29 ± 0.81	0.14 ± 0.14
Spio maculata						0.43 ± 0.43			
Spio sp.					0.14 ± 0.14				
Microspio sp.						0.14 ± 0.14			
Scolelepis squamata					0.14 ± 0.14				
Scolelepis cf. *tridentata*		0.14 ± 0.14			0.14 ± 0.14		0.57 ± 0.43		
Boccardiella hamata	0.14 ± 0.14					0.14 ± 0.14			
Mediomastus ambiseta	15.71 ± 3.38	27.86 ± 3.33	0.57 ± 0.57	19.29 ± 2.31	20.43 ± 2.72	27.71 ± 3.54	29.43 ± 5.66	23.43 ± 6.12	20.43 ± 2.43
Mediomastus californiensis			0.14 ± 0.14						0.14 ± 0.14
Capitella capitata						0.14 ± 0.14			11.43 ± 2.20
Notomastus tenuis			0.43 ± 0.43			0.14 ± 0.14		0.29 ± 0.18	0.43 ± 0.20
Exogone spp.	3.71 ± 1.43	12.00 ± 1.11	0.43 ± 0.43	2.29 ± 0.87	8.71 ± 2.31	4.14 ± 1.56	19.86 ± 8.53	20.71 ± 4.45	2.57 ± 1.13
Exogone molesta							0.57 ± 0.57		
Sphaerosyllis spp.					0.14 ± 0.14		2.00 ± 0.98	4.14 ± 1.37	1.00 ± 0.49
Typosyllis alternata							0.71 ± 0.36	0.57 ± 0.43	
Typosyllis sp.			0.14 ± 0.14						
Eteone californica	0.14 ± 0.14	0.71 ± 0.42		0.14 ± 0.14	0.29 ± 0.18	0.14 ± 0.14			
Fabricia stellaris	2.86 ± 1.32	10.29 ± 6.25		0.29 ± 0.18	0.14 ± 0.14		16.86 ± 8.32	1.14 ± 0.74	

Table 9. June 2013 Infaunal abundances continued (mean ± 1 S. E.).

Lowest Taxon Identified	Bed			Control			Near	Far	Basin6
	In	Out	Under	In	Out	Under			
Euchone limnicola		1.43 ± 0.69				0.57 ± 0.37	4.86 ± 1.96	2.14 ± 0.55	
Pista pacifica		0.14 ± 0.14					0.14 ± 0.14	0.29 ± 0.18	
Armandia brevis		0.14 ± 0.14			0.14 ± 0.14		0.14 ± 0.14	0.71 ± 0.36	
Ophelia limnacina					0.14 ± 0.14				
Ophelia sp.							0.14 ± 0.14		
Dorvillea longicornis							0.43 ± 0.30	6.14 ± 5.07	0.14 ± 0.14
Dorvillea sp.		0.43 ± 0.43					0.29 ± 0.18	1.57 ± 1.41	
Scoletoma zonata		0.14 ± 0.14					0.29 ± 0.18	0.86 ± 0.26	0.14 ± 0.14
Scoletoma sp.		0.29 ± 0.18							
Nereis procera							0.57 ± 0.43	0.71 ± 0.36	12.71 ± 2.49
Glycera nana								0.29 ± 0.18	
Goniada brunnea	0.57 ± 0.43	0.71 ± 0.18		0.14 ± 0.14			0.43 ± 0.20		
Marphysa stylobranchiata								0.14 ± 0.14	
Leitoscoloplos pugettensis							0.86 ± 0.40	0.29 ± 0.18	0.86 ± 0.26
Cossura pygodactylata							0.86 ± 0.70	0.86 ± 0.34	0.29 ± 0.18
Ctenodrilus serratus	0.29 ± 0.29	2.00 ± 0.87			0.14 ± 0.14		0.29 ± 0.18	0.29 ± 0.29	
Clymenella californica							0.29 ± 0.29	0.14 ± 0.14	1.29 ± 0.71
Bivalvia	0.29 ± 0.18	0.29 ± 0.18		0.43 ± 0.30	0.29 ± 0.18	1.14 ± 0.70	0.57 ± 0.43		1.57 ± 0.65
Mytilus galloprovincialis								0.14 ± 0.14	
Cerithidea californica	0.14 ± 0.14					0.14 ± 0.14			
Caesia fossatus						0.14 ± 0.14			
Barleeia sp.							5.43 ± 1.59	7.00 ± 2.51	19.43 ± 10.20
Acteocina sp.	3.14 ± 2.16	1.43 ± 0.61		1.29 ± 0.57	0.57 ± 0.57	0.29 ± 0.18	0.43 ± 0.43		0.29 ± 0.18
Bulla gouldiana							0.14 ± 0.14		
Lottia paleacea									0.29 ± 0.29
Lottia sp.									0.14 ± 0.14

Table 10. June 2013 Infaunal abundances continued (mean ± 1 S. E.).

Lowest Taxon Identified	Bed			Control			Near	Far	Basin6
	In	Out	Under	In	Out	Under			
Olea hansineensis					0.14 ± 0.14		0.14 ± 0.14		
Hemigrapsus oregonensis								0.14 ± 0.14	0.14 ± 0.14
Neotrypaea californiensis		0.14 ± 0.14		0.14 ± 0.14		0.14 ± 0.14			
Upogebia sp.	0.14 ± 0.14								
Hippolyte californiensis								0.14 ± 0.14	
Grandidierella japonica		1.00 ± 0.38		0.57 ± 0.30	0.43 ± 0.30		20.14 ± 4.81	17.43 ± 5.28	43.57 ± 24.69
Protohyale frequens							0.14 ± 0.14		
Mayerella banksia							12.86 ± 5.54	6.57 ± 2.13	4.86 ± 3.56
Leptochelia dubia	0.29 ± 0.29				0.14 ± 0.14		4.71 ± 1.76	0.71 ± 0.42	1.29 ± 0.81
Zeuxo normani							0.29 ± 0.29		3.00 ± 1.60
Isopoda	0.29 ± 0.29								
Paracerceis sculpta								0.29 ± 0.29	0.57 ± 0.30
Uromunna ubiquita								0.14 ± 0.14	
Paranthura elegans									0.71 ± 0.42
Holothuroidea	0.14 ± 0.14	0.14 ± 0.14							
Ophiuroidea							0.14 ± 0.14	0.14 ± 0.14	4.29 ± 2.83
Leptosynapta albicans		0.14 ± 0.14			0.14 ± 0.14				
Phoronida		0.43 ± 0.43			0.14 ± 0.14	0.57 ± 0.43		0.43 ± 0.43	
Cnidaria							0.14 ± 0.14	0.43 ± 0.43	
Anthozoa							3.71 ± 2.52		1.71 ± 0.71
Cladonema sp.							0.43 ± 0.20	0.71 ± 0.36	
Copepoda	0.29 ± 0.29			0.71 ± 0.71		0.29 ± 0.29			0.14 ± 0.14
Ostracoda		0.43 ± 0.30		0.14 ± 0.14			12.29 ± 3.64	12.29 ± 6.51	19.00 ± 5.36
Nemertea	0.71 ± 0.29	0.71 ± 0.29	0.29 ± 0.18	0.86 ± 0.34	0.57 ± 0.30	0.29 ± 0.18	0.71 ± 0.29	0.86 ± 0.34	5.29 ± 2.25
Platyhelminthes	1.71 ± 1.04	0.86 ± 0.70		0.29 ± 0.18	0.86 ± 0.46	0.71 ± 0.42	0.71 ± 0.71	0.14 ± 0.14	0.14 ± 0.14
Hemichordata							0.14 ± 0.14		