

Biological Assessment of Three Rocky Intertidal Sites in San Diego

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Summary

Comparisons of the biodiversity and community structure of macrophytes, macroinvertebrates, and meiofauna were conducted at three rocky intertidal locations in San Diego, CA. The purpose of the study was to determine the potential impacts of a storm water drain on the community composition of an impacted site at La Jolla Cove (LJ). Comparisons of community structure were referenced to two control sites, Cabrillo National Monument (CNM) and Sunset Cliffs (SC). Results revealed that each site was significantly different from each other using numerous parameters. Biodiversity of macrophytes and macroinvertebrates was lowest at the impact site while patterns were mixed for meiofaunal diversity with diversity sometimes being higher at the impact site. Multi-variate statistical analyses of community structure of macrophytes and macroinvertebrates, as well as meiofauna, revealed that the impact site was clearly differentiated from the other two sites, but all three sites grouped separately suggesting the sites may differ due to several factors, not just the presence of storm drain. The percent cover of a list of species also differed among sites with some of these species being large contributors to the differences in overall community structure found in multi-variate analyses. There were patterns of increases in disturbance tolerant macrophytes at the impact site but patterns of responses of all disturbance indicator species were not consistent. Furthermore, meiofauna did not show any patterns even with taxa we would expect to be indicators of disturbance. Comparisons of turf depth in the field and in collected turf cores also revealed that the impact site had significantly lower turf depth and marginally lower sand accumulation. Despite some differences, evidence of obvious and consistent storm water impacts is relatively weak. However, differences among sites does warrant possible future examinations in a more robust manner using multiple impacted and non-impact sites and concentrating some sampling on non-turf habitats such as intertidal tidepools.

Background

Rocky intertidal ecosystems are located along the open coast where the sea meets the land and are composed of a highly diverse assemblage of flora and fauna. They are considered harsh ecosystems as they are exposed to a number of natural perturbations, including, but not limited to, sand burial and scour, disease, predation, competition, and daily exposure to the air during low tides that can lead to large fluctuations in temperature, salinity, and access to readily usable oxygen. During low tides, species quickly lose their ability to hold water and thus are subject to desiccation stress; to reduce loss of water, many species clamp onto rocks or close their shells thus reducing their ability to feed during periods of aerial exposure. Furthermore, rocky intertidal flora and fauna are located within the wave crashing zone and are subjected to large wave force that can potentially remove them from the rocks. In addition to

natural impacts, rocky intertidal habitats, especially in urbanized areas, are subject to a number of anthropogenic disturbances, including habitat destruction, climate change, human visitation, introductions of non-native species, overexploitation, modifications of coastal processes such as increased sedimentation, and air and water pollution (as summarized by Crowe et al., 2000; Thompson et al., 2002). When humans visit rocky intertidal habitats, for example, the subsequent trampling, handling, rock turning, and collecting can lead to reductions in abundances of numerous species (Povey and Keough, 1991; Ambrose and Smith 2005; Smith and Murray 2005; Smith et al. 2008), physiological damage (Keough and Quinn, 1998; Denis 2003), shifts in the size structure of populations towards smaller individuals (Kido and Murray, 2003; Sagarin et al., 2007; Smith et al. 2008), reductions in reproductive output (Denis 2003), and changing of ecosystem functioning (Duran and Castilla, 1989; Kingsford et al., 1991; Brown and Taylor, 1999).

Of particular concern to this study are numerous categories of pollution which can be linked to multiple sources, including terrestrial runoff, storm drain flow, sewage effluent, and oil spills and seeps. Pollution can be chronic or pulse disturbances and can be either point or non-point sources. Pollutants from any source, whether chronic or pulse, can have numerous and varying impacts on species health and normal ecosystem functioning (as summarized by Crowe et al., 2000; Thompson et al., 2002), albeit often understudied in field surveys in rocky intertidal ecosystems.

One of the more commonly studied pollutants in rocky intertidal habitats, and other coastal ecosystems, is additions of nutrients (mostly nitrogen and phosphorus) through sewage discharges and agricultural run-off. Anthropogenic inputs of nitrogen in coastal ecosystems can also be linked to atmospheric deposition (Paerl and Whittall, 1999). The general impacts of additions of nutrients are mostly observed in the flora with declines in perennial seaweeds and increases in simpler, early successional-stage, opportunistic seaweeds that utilize the nutrients to grow and bloom. For example, in the polluted inner Gullmar Fjord in Sweden, macrophyte diversity from 1941 to 1998 declined greatly (Eriksson et al. 2002), shifting from coarsely branched and perennial algae to small, thin, filamentous, and seasonal ephemerals. In the inner Oslofjord in Norway, Bokn and Lein (1978) and Bokn (1979) reported that fleshy brown fucoid seaweeds had declined compared to previous surveys; ephemeral green alga such as *Enteromorpha* (now *Ulva*) and *Ulva* became more abundant. This change in community structure was linked to the discharge of sewage; following improvements in sewage treatment, the fucoid community improved (Bokn et al. 1992). In the Ythan estuary in Scotland, Raffaelli et al (1999) reported significant increases in green macro-algal mats since the 1950s, likely related to increase in nutrients into the system. The increase in macro-algal mats had negative impacts on benthic invertebrates and the shorebirds that consume them. Increased eutrophication has been linked to the disappearance of other fleshy brown algae in other locations around the

world (Rodriguez-Prieto and Polo, 1996; Oliveira and Qi, 2003). In southern California, Littler and Murray (1975) found shifts in seaweed communities towards opportunistic flora, such as blue-green algae and *Ulva*, near a sewage outfall on San Clemente Island. Since the 1950s, addition of nutrients, mostly from sewage, was thought to have caused major shifts in southern California rocky intertidal flora (Dawson 1959, 1965) that are still present (Smith et al., unpublished data).

Sewage, or other inputs of nutrients, can also cause impacts to rocky intertidal fauna. For example, increases in some rocky intertidal filter feeders were observed near sewage outfalls in southern California possibly due to utilization of sewage as food (Littler and Murray, 1975). In Argentina, growth of a limpet was enhanced in areas affected by sewage pollution and resultant organic enrichment (Tablado et al, 1994). In this same area, sites with sewage pollution were denude of the bivalve *Brachidontes* that typically dominates the intertidal zones in that region (Lopez Gappa et al., 1990). In Australia, similar bivalves were found to have higher recruitment to a site polluted by sewage effluent coupled with lower growth and higher mortality (Hindell and Quinn, 2000). Polychaete abundances are also believed to increase near sewage outfalls and are likely good indicators of organic matter pollution (Pearson and Rosenberg, 1978; Fauchald and Jumars, 1979; Bitar 1982; Elias et al, 2003).

In addition to organic pollution, the ecological impacts of oil spills on rocky intertidal populations are also widely studied. The combined impacts of oil coating of organisms, direct lethal toxicity, and sub-lethal impacts on health, as well as the clean-up efforts using chemical dispersants and physical removal (such as using high pressure water), cause detrimental impacts to numerous rocky intertidal flora and fauna. For acute exposures due to oil spills, mass mortality of macroalgae and benthic invertebrates can occur due to chemical toxicity and smothering (Paine et al., 1996; Peterson et al., 2003). For example, following the Exxon spill in Prince William Sound, Alaska, the rockweed *Fucus*, an ecosystem engineer, was severely impact by oil, as well as by the clean up consisting of high pressure hot water (Lees et al., 1993; Paine et al., 2006). In this same location, suspension feeders, such as clams and mussels, maintained chronically elevated tissue contamination while sediments, such as that in seagrass habitats, retained oil contamination for several decades (Peterson et al., 2003). Other spills, such as that of the Torrey Canyon spill in Cornwall, resulted in losses of numerous grazers due to both oil exposure and the effects of dispersants which resulted in numerous indirect impacts including increases in ephemeral algae (Hawkins and Southward, 1992). Chronic toxicity accumulation following these spills can damage reproductive functions (summarized by Hir and Hihly, 2002), among other physiological responses. In addition to direct oil spills releasing large amount of oil in a short time period, chronic oil seepage from anthropogenic sources, an order of magnitude large in total input than spills (Foster et al., 1988; Suchanek, 1993), also leads to detrimental effects. Sublethal impacts may include physiological, carcinogenic, and cytogenic effects

resulting in population level changes (abundance, age structure, reproduction and recruitment), and community modifications (biological interactions, ecosystem functioning) (Suchanek, 1993).

Heavy metals from human sources, such as sewage, industrial discharge, and urban run-off, are also common pollutants in marine ecosystems (Crowe et al., 2000). The toxicity of heavy metals is well known as are the maximum levels of exposure before physiological stress. However, much of this work has been conducted in laboratory studies with little work examining the field effects of heavy metals on rocky intertidal ecosystems (Crowe et al., 2000). At certain levels of exposure, rocky intertidal representatives studied in the laboratory have responded in a complex manner. Some species are sensitive while others are tolerant; responses include reduced survival, increased or decreased growth (Stromgren, 1982; Munda and Hudnik, 1986), impacts on larval development (Beaumont et al., 1987; Fichet et al., 1998), and increased genetic resistance (Hoare et al., 1995). It should be noted that there are a larger studies of impacts of metals on higher trophic organisms (birds, marine mammals) due to bioaccumulation but little work is published on rocky intertidal organisms. Of the heavy metals, most of the published literature found examined the effects of anthropogenic inputs of copper through copper mining. Copper has been shown to cause marked shifts in rocky intertidal meiofaunal assemblages in Chile, including lowered densities and taxa diversities; turbellarians were often common in impacted sites while foraminiferans and harpacticoid copepods were sensitive to copper (Lee and Correa, 2005). Increases in the ephemeral alga *Enteromorpha* (now *Ulva*) were also observed as this species is resistant to high levels of copper (Castilla, 1996).

As with heavy metals, the effects of pesticides and antifouling paints, common pollutants found in marine environments, are relatively understudied for rocky intertidal systems. Tributyltin (TBT), found in antifouling paints, has been shown to negatively impact certain molluscs which appear to respond to low concentrations and result in imposex (male characteristics found in females) or female sterility (Bryan et al., 1986, 1988; Ellis and Pattisina, 1990). Developmental effects and reduced reproductive outputs, growth, and abundances can also be observed in molluscs and other marine organisms (Paul and Davies, 1986; Johansen and Mohlenburg, 1987; Gibbs et al., 1988; Salazar and Salazar, 1991).

Thermal pollution and freshwater pollution can also impact rocky intertidal populations. Thermal effluent from cooling water out-falls from power plants or other industries can cause shifts in the types of species present. For example, in Maine, thermal effluent resulted in declines in the abundances of large fleshy brown algae (Vadas et al., 1978; Bamber, 1991). Off the coast of California, thermal plumes resulted in long-term loss of foliose and encrusting

algae while grazing gastropods increased in abundance (Schiel et al., 2004; Steinbeck et al., 2005). No research was found on the effects of freshwater pollution.

Rocky intertidal locations in San Diego, CA are subjected to many of the aforementioned natural and anthropogenic disturbances. Of particular concern in this study, is pollution from terrestrial and anthropogenic sources reaching rocky intertidal zones through storm drains. At La Jolla Shores, a storm drain directly deposits a high flow of storm water directly into the rocky intertidal zone during wet weather. If the tide is high during storm water flow, dilution with ocean water appears to be quick. However, when water flow coincides with the low tides, storm water directly flows and accumulates in the intertidal zone exposing flora and fauna to freshwater containing sediments, debris, and a number of potential pollutants. The impact of the storm drain is currently not known. Therefore, the goal of this study was to assess the biological composition of the rocky intertidal habitat at the impacted site. Biodiversity and community composition and structure were compared with a reference site at Cabrillo National Monument in Point Loma where no known storm runoff occurs. A second reference site at Sunset Cliffs was also chosen for comparisons.

Methods

Site Selection. Three rocky intertidal locations were chosen in San Diego to examine differences in biodiversity and community composition. La Jolla Shore (LJ, Figures 1-4) is the furthest north location (32° 50' 55.81" N, 117° 15' 53.31" W) located in La Jolla. LJ is a NW facing intertidal zone. This site receives heavy runoff from a storm drain (#186) during wet weather and is the target site of concern. This site also receives relatively low levels of human use (personal obs.). Further south in Point Loma is Sunset Cliffs (SC, Figures 5-8, 32° 43' 06.64" N, 117° 15' 23.27" W), a W facing intertidal zone. This site does not have a large storm drain although a cement drain is present leading directly into the intertidal zone ~250 m from the sampled location. During wet weather, very minimal flow was observed. This site receives low to moderately low levels of human use (personal obs.). The site furthest south is Cabrillo National Monument (CNM, Figures 9-10, 32° 39' 57.23" N, 117° 14' 38.13" W) located on the grounds of the Cabrillo National Monument Park, managed by the National Park Services. This site, termed Zone III by CNM NPS, is WSW facing, has no direct runoff, and no human use as people are excluded from this portion of the coastline by signs and docents/park managers. All sites are gently sloping with a sandy beach above the rocky intertidal zone, a flat, long algal turf zone, and then dropping off into a low-intertidal seagrass zone. Interspersed within sites are both small and large tidepools and boulders.

Habitat Sampled. At all three locations, the flat rocky intertidal zone dominated by red algal turfs were sampled. Although both boulders and tidepools were dispersed throughout the sampled zone, a majority of the sampled area was constricted to the turf habitat.



Figure 1. Approximate sampling location at La Jolla Shores.



Figure 2. Line transects set up at La Jolla Shores.



Figure 3. View of line transects and storm drain pipe 186 at La Jolla Shores.



Figure 4. Flow of storm runoff during wet weather from storm drain pipe 186 at La Jolla Shores.



Figure 5. Approximate sampling location at Sunset Cliffs.



Figure 6. Cliff top view of sampled rocky intertidal zone at Sunset Cliffs.



Figure 7. View of baseline transect and sampled rocky intertidal zone at Sunset Cliffs.



Figure 8. View of baseline transect and sampled rocky intertidal zone at Sunset Cliffs.



Figure 9. Approximate sampling location at Cabrillo National Monument, Zone III.



Figure 10. Transect lines set up at Cabrillo National Monument, Zone III.

Measures of Percent Cover of Macrophytes and Macroinvertebrates. Percent cover of all taxa visible to the naked eye were examined using a modified point contact method (Murray et al. 2006), based loosely on method used by UC Santa Cruz's Coastal Biodiversity Survey. At each of the three sites, a 30 m length transect was placed parallel to the ocean. Eleven transects of varying length, depending on the length of the rocky bench, were laid out perpendicular to the shore, every 3 meters along the 30 m transect line. A point contact method was used whereby every species located underneath 100 evenly distributed points along the perpendicular transect lines were counted; each transect served as a replicate. For algal turfs in which several species can grow intertwined with one another, a small pinch of the turf was extracted, examined in the field, and all species identified. Each transect line consisted of 100 points in which percent cover of species was calculated based on the number of hits out of 100; given that numerous taxa were often located beneath a single point, total cover typically reached well above 100% per transect line. Sites were sampled on two occasions, once in January 2011 and again in March 2011; both sampling periods occurred during the wet part of the year with January 2011 sampling occurring during or directly after a rain event and March sampling occurring shortly after a rain event.

Percent cover of species were analyzed using a One-Factor ANOVA for simplification, with sampling period ignored. Species were then grouped into ecological functions based on functional groups for macrophytes and feeding guilds for macroinvertebrates. As with species, data were analyzed with a One Factor ANOVA. Biodiversity was calculated using three measures (species richness, Pielou's Evenness, and Shannon-Wiener diversity index) and analyzed with a Two-Factor ANOVA with site and sampling period as factors. An additional One Factor ANOVA was used to examine differences among sites when both sampling periods were combined. Community data were analyzed using multi-variate analyses with the statistical program PRIMER. Community data were transformed using a square root transformation with a resemblance matrix calculated using a Bray-Curtis similarity. The similarity then was used to create multi-dimensional scaling (MDS) plots in which each transect sample was placed onto a graph with replicates that were similar in community structure plotting closer to one another. A two factor Analysis of Similarities (ANOSIM) was used to determine significant differences in community structure among sites and between sampling seasons. A Similarities Percentage (SIMPER) test was used to determine which species are contributing most to dissimilarities among sites. Community analyses were conducted on all data together as well as separating the algal turf and non-algal turf microhabitats along each transect to determine if microhabitats played a role in differences among sites.

Turf Depth. The depth of the algal turf was measured roughly every 3 points along 100 evenly distributed points along each transect. To measure depth, a pin was stuck into the turf and the distance of penetration into the turf was measured to the nearest mm. Depth of turf were

sampled in both January and March 2011. Depth data were analyzed using a Two-Factor ANOVA with site and sampling period as factors. An additional One Factor ANOVA was used to examine differences among sites when both sampling periods were combined.

Motile Invertebrate Counts. Along each transect, ~two 0.2 X 0.2 m quadrats were randomly located and all motile macroinvertebrates counted. Given the low number of motile invertebrates found during sampling in January 2011, this was not repeated during the March sampling. As with percent cover data, biodiversity and community structure analyses were conducted but with site as the only factor.

Meiofauna Abundances. To examine meiofauna located within the red algal turf, two core samples (3.75 X 3.75 cm) were collected along each transect during each of the two sampling periods. Samples were placed into containers and preserved using 90% ethanol for future sampling. In the laboratory, 15 samples per site for January 2011 and 11 samples per site in March 2011 were examined under a microscope and all meiofauna identified to the lowest taxonomic level possible, based on our expertise; this usually consisted of large scale groupings (amphipods, isopods, ostracods, etc.). Sediment within samples also was measured for volume to determine correlations between sediment and meiofauna densities and to determine site differences in sediment accumulation.

In a similar manner as the percent cover data, meiofaunal biodiversity and community structure analyses were conducted to determine differences among sites and between sampling periods in core samples. Total abundances of meiofauna and biodiversity (species richness, Pielou's evenness, and the Shannon-Wiener Index) were examined using a Two Factor ANOVA with site and sampling period as factors. An additional One Factor ANOVA was used to examine differences among sites when both sampling periods were combined. Community structure was analyzed using log (X+1) transformed counts within identified taxa for each core sample at each site per sampling period. Multi-variate analyses were similar to those of the percent cover data (MDS, ANOSIM, and SIMPER). In addition, the amount of sediment within each core was overlayed onto multi-dimensional scaling plots to determine if sediment accumulation influenced meiofaunal data. Sediment accumulation was analyzed using both a Two Factor ANOVA (site and sampling period) and One Factor ANOVA (site; sampling periods were combined).

Results

Macroinvertebrate and Macrophyte Cover, Diversity, and Community Composition:

All data combined

Transects and sites varied in the degree of algal turf and non-algal turf microhabitats. When examining the site as a whole with both microhabitats included, there were significant differences among factors for biodiversity measures, community composition, cover of some individual species, and cover of some functional groups (macroalgae) and feeding guilds (macroinvertebrates). Generally speaking, the three sites were all relatively different from one another.

Biodiversity was measured using three tools: species richness, Pielou's Evenness (J) and the Shannon-Wiener Index (H'). For species richness, there were significant differences among sites only while J and H' revealed significant differences among sites and with the site X sampling period interaction term (Two-Factor ANOVA; Table 1). In general, LJ tended to have the lowest biodiversity followed by CNM, with SC having the highest diversity (Figure 11). There were no consistent patterns among diversity measures for sampling periods, some sites increased while others decreased (hence the significant interaction term in some cases). With the two sampling periods combined, One Factor ANOVAs revealed significant differences among sites for all diversity measures (Table 1). Again, diversity was lowest at LJ and highest at SC (Figure 12).

Community composition was found to vary significantly among sites (ANOSIM Global $R=0.944$; $p<0.001$) and among sampling periods (ANOSIM Global $R=0.522$; $p<0.001$). In general, the three sites clustered separately suggesting community composition among sites is markedly different (Figure 13). Within sites, the two sampling periods tended to cluster together except for CNM where samples were mixed in one large grouping (Figure 13). The species driving differences among sites were analyzed using a Similarities Percentage (SIMPER) test. Differences between LJ and SC were driven by high *Caulacanthus*, *Ulva californica*, *Ceramium*, and bare rock and low *Jania crassa*, *Centroceras*, *Phyllospadix*, and *Lithothrix* at LJ. Differences between LJ and CNM were driven by high *Caulacanthus*, *Corallina pinnatifolia/vancouveriensis*, bare rock, and *Ceramium* and low *Phyllospadix*, bare sand, *Chondracanthus canaliculatus*, *Lithothrix*, and *Centroceras* at LJ. SC and CNM differed mostly due to high cover of *Jania crassa*, *Ulva californica*, *Centroceras*, *Lithothrix*, and *Acrosorium* and low *Phyllospadix*, bare sand, and *Ceramium* at SC.

Table 1. Statistical results of Two-Factor ANOVAs and One-Factor ANOVAs (with sampling periods combined) for three diversity measures for macrophyte and macroinvertebrate data. Bold p values denote significance.

Diversity Measure	Statistical Test	Factor	df	F stat	p value
Species Richness	Two Factor ANOVA	Site	2	32.6	<0.001
		Sampling	1	2.8	0.099
		Site X Sampling	2	2.6	0.084
	One Factor ANOVA	Site	2	30.2	<0.001
Pielou's Evenness	Two Factor ANOVA	Site	2	22.2	<0.001
		Sampling	1	0.4	0.510
		Site X Sampling	2	6.6	0.003
	One Factor ANOVA	Site	2	18.9	<0.001
Shannon-Wiener Index	Two Factor ANOVA	Site	2	47.5	<0.001
		Sampling	1	1.0	0.324
		Site X Sampling	2	6.8	0.002
	One Factor ANOVA	Site	2	40.0	<0.001

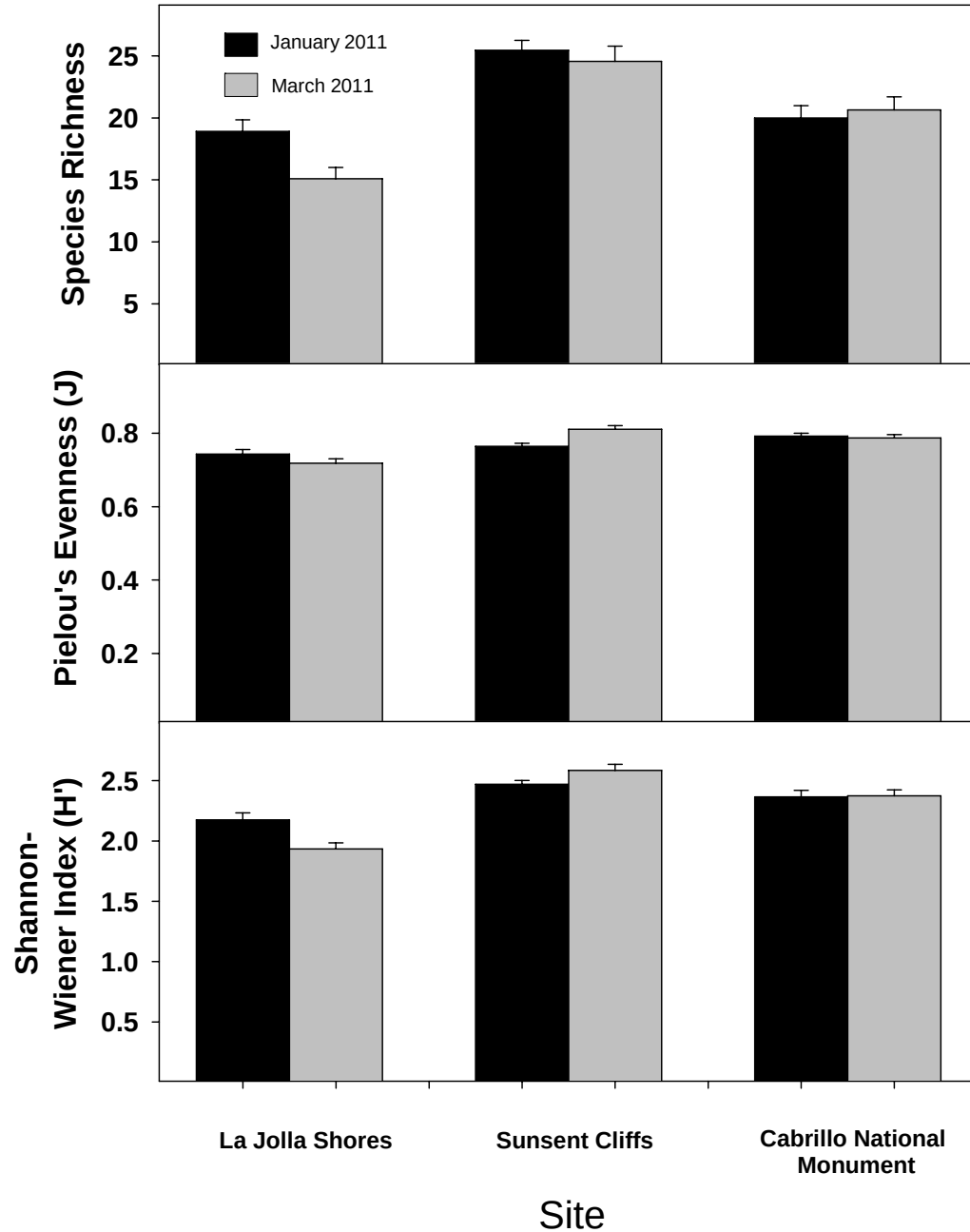


Figure 11. Diversity measures (± 1 SE), including species richness, Pielou's Evenness, and Shannon-Wiener index, of flora and fauna at three sites in San Diego during January and March 2011 sampling periods. Statistical results for these data are located in Table 2 (Two Factor ANOVA).

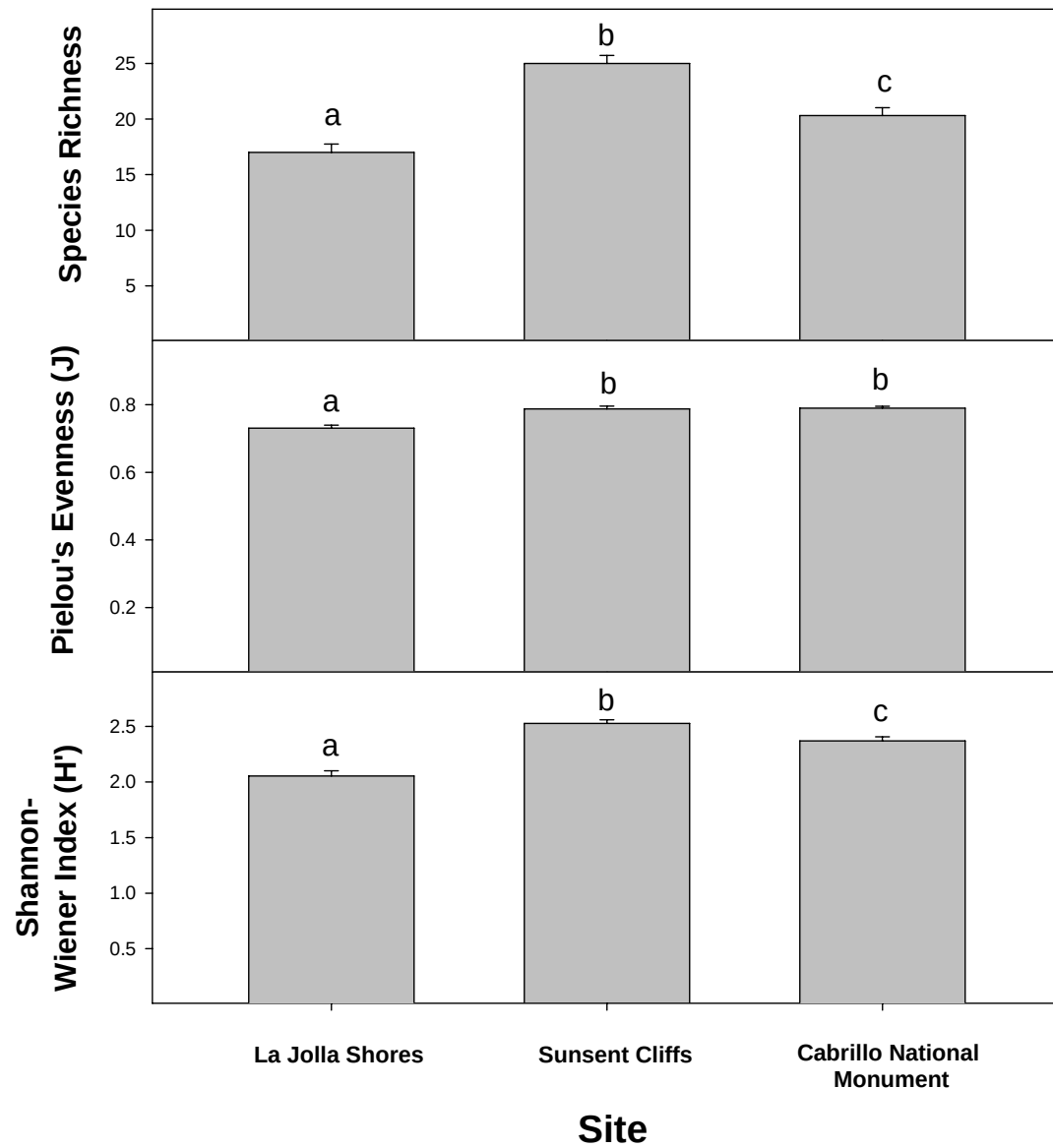


Figure 12. Diversity measures (± 1 SE), including species richness, Pielou's Evenness, and Shannon-Wiener index, of flora and fauna at three sites in San Diego with both sampling periods combined. Statistical results for these data are located in Table 2 (One Factor ANOVA). Letters above bars signify significantly similar groups using Tukey's post hoc comparisons tests.

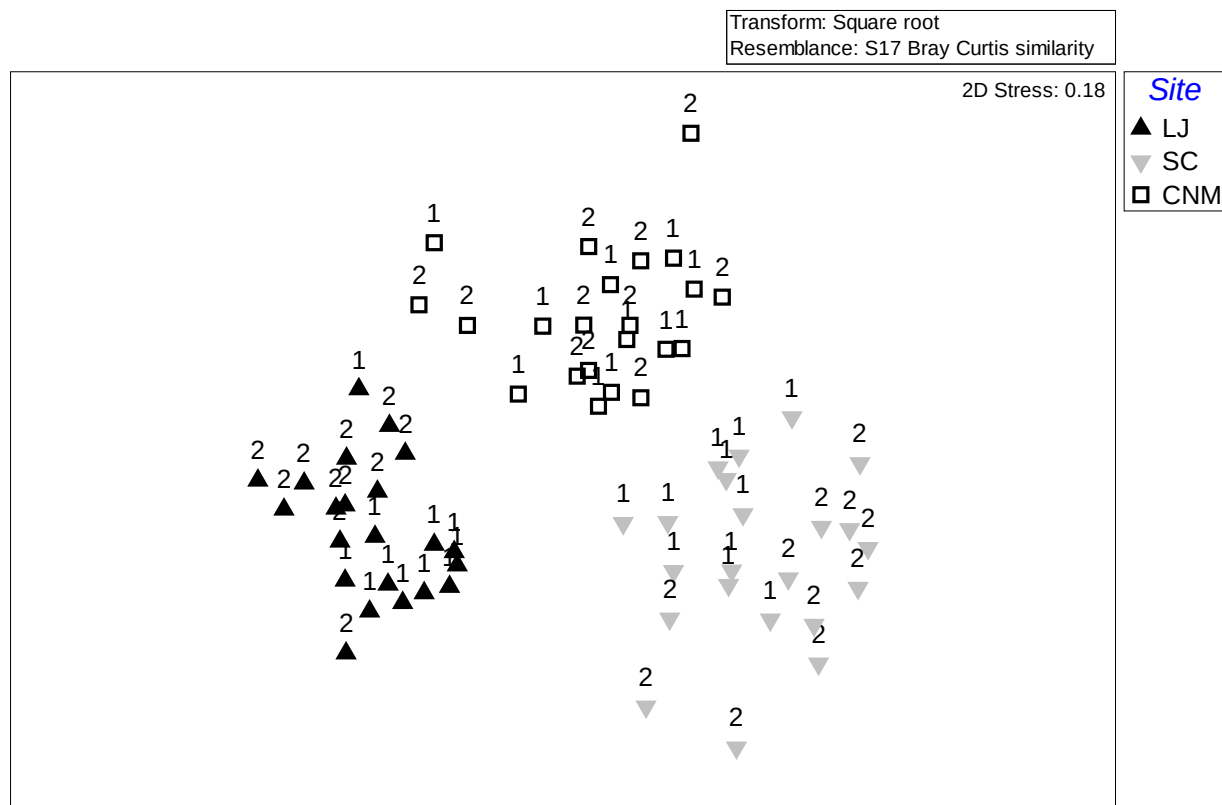


Figure 13. Multi-dimensional scaling plot of cover data from 11 transects at three sites in San Diego (LJ=La Jolla Shore, SC=Sunset Cliffs, CNM=Cabrillo National Monument) during two sampling periods (1=January 2011, 2-March 2011). Analysis of Similarities (ANOSIM) revealed significant differences among sites and between sampling periods.

In many cases, species are playing similar ecological roles. To reduce the influence of species and to instead exam shifts in taxa within different ecological roles, species are placed into functional forms (macrophytes) or feeding guilds (macroinvertebrates). Community structure using ecological roles as groupings also revealed significant differences among sites (ANOSIM Global $R=0.764$, $p<0.001$) and between sampling periods (ANOSIM Global $R=0.399$; $p<0.001$). Patterns of clustering were not quite as clear as the species data yet sites still differentiated well (Figure 14). Again, this indicates that sites were markedly different in community composition. Differences between LJ and SC were driven by low cover of seagrass, bare sand, and filamentous-like algae and high cover of blade algae, bare rock, and encrusting algae at LJ. LJ and CNM differed due to low cover of seagrass and bare sand and high cover of articulate corallines, bare rock, blade algae, and encrusting algae at LJ. Differences between SC and CNM were driven by high cover of articulated corallines, blade algae, and filamentous-like algae and low cover of seagrass and bare sand at SC.

At the individual species level, there were a number of significant differences among sites (One Factor ANOVA for simplicity; Appendix 1). Of note, LJ had significantly more bare rock, *Corallina pinnatifolia/vancouveriensis*, *Caulacanthus*, and encrusting algae than both of the other sites. SC appeared to have intermediate levels of cover in between LJ and CNM for many species, sometimes being more similar to LJ, sometimes more similar to CNM. CNM was notable for having high cover of bare sand and *Phyllospadix* and low cover of filamentous-like algae and fleshy algae.

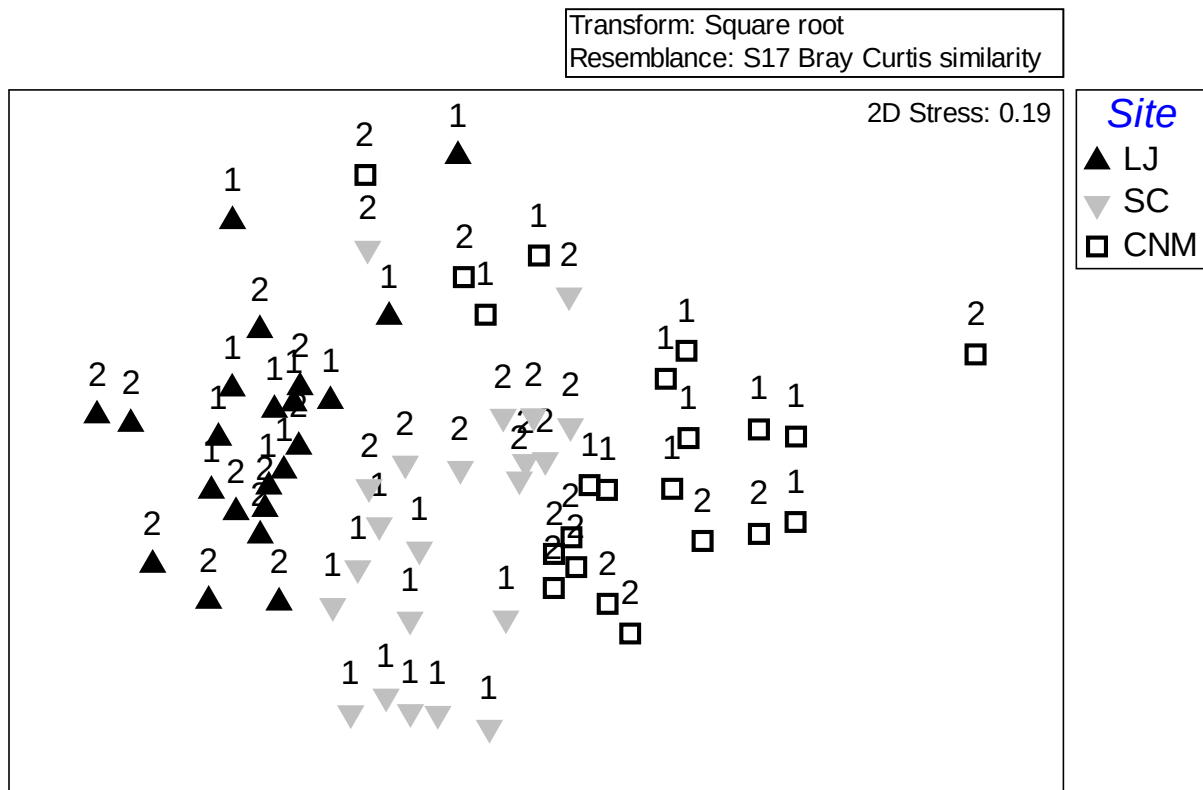


Figure 14. Multi-dimensional scaling plot of functional groups/feeding guilds cover data from 11 transects at three sites in San Diego (LJ=La Jolla Shore, SC=Sunset Cliffs, CNM=Cabrillo National Monument) during two sampling periods (1=January 2011, 2-March 2011). Analysis of Similarities (ANOSIM) revealed significant differences among sites and between sampling periods.

Algal Turf and Non-turf Habitat Comparisons

To determine if differences among sites above could be attributed to differences in the amount of available turf and non-turf microhabitats within sites, we analyzed community composition within the microhabitats separately. Using species cover data in algal turf habitats, patterns were similar to those using all data with each site grouping together and sampling periods within sites relatively grouped together (Figure 15). Analysis of Similarities (ANOSIM) found significant differences among sites (Global $R=0.861$, $p<0.001$) and sampling periods (Global $R=0.586$, $p<0.001$). Using functional groups and feeding guilds data clusters within turf microhabitats, the same patterns were observed, although not as clearly (Figure 16). Again, ANOSIM revealed significant differences among sites (Global $R=0.472$, $p<0.001$) and sampling periods (Global $R=0.543$, $p<0.001$). For the non-algal turf habitat, similar patterns were observed using species data (Figure 17; ANOSIM site Global $R=0.503$, $p<0.001$; sampling period Global $R=0.128$, $p=0.004$) and functional groups/feeding guilds data (Figure 18; ANOSIM site Global $R=0.415$, $p<0.001$; sampling period Global $R=0.122$, $p=0.004$). These patterns confirm that differences among sites using all cover data were not skewed by differences in the amount of microhabitat available at each site. Furthermore, differences appear to be driven by differences in both microhabitats. If data for all sites and sampling periods and for the two microhabitats included together, patterns suggest that there is a bigger difference among sites in the species composition of the non-algal turf habitats (Figure 19) than the algal turfs as the algal turf data clusters more closely together. Non-turf habitat was only a minor component of the overall sampled habitat thus under sampled. Therefore, patterns of more difference among non-turf habitats among sites must be interpreted with caution.

Other Site Comparisons

Multivariate analyses are very robust at detecting differences in community composition among sites. However, those differences are relative the samples analyzed. To determine the relative degree of differences among the three San Diego sites, two sites in Orange County (Crystal Cove and Corona del Mar) were also added to the multi-variate analyses for comparison. All five sites appear to group individually (Figure 20) with the two Orange County sites separating out from the three San Diego locations. Patterns suggest that although there are differences and clear differentiation of San Diego sites, these sites are, overall, group closer together in comparison to other outside locations.

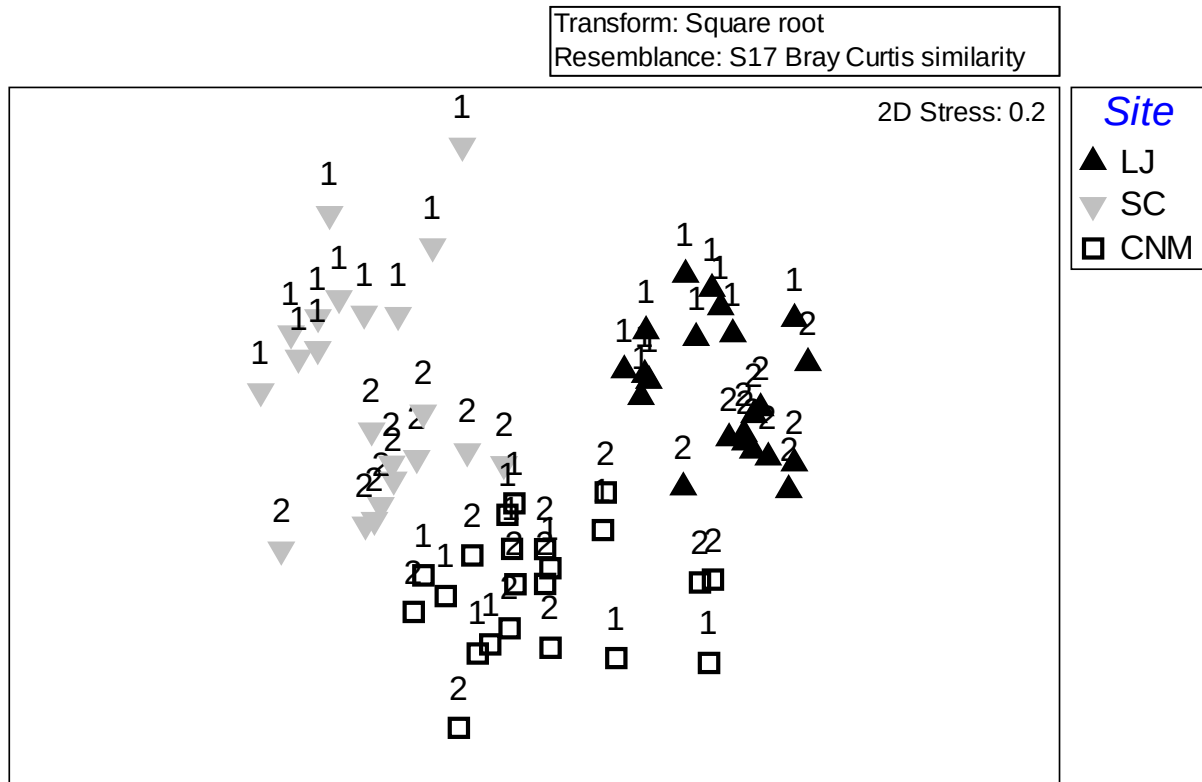


Figure 15. Multi-dimensional scaling plot of cover data with algal turf microhabitats from 11 transects at three sites in San Diego (LJ=La Jolla Shore, SC=Sunset Cliffs, CNM=Cabrillo National Monument) during two sampling periods (1=January 2011, 2-March 2011). Analysis of Similarities (ANOSIM) revealed significant differences among sites and between sampling periods.

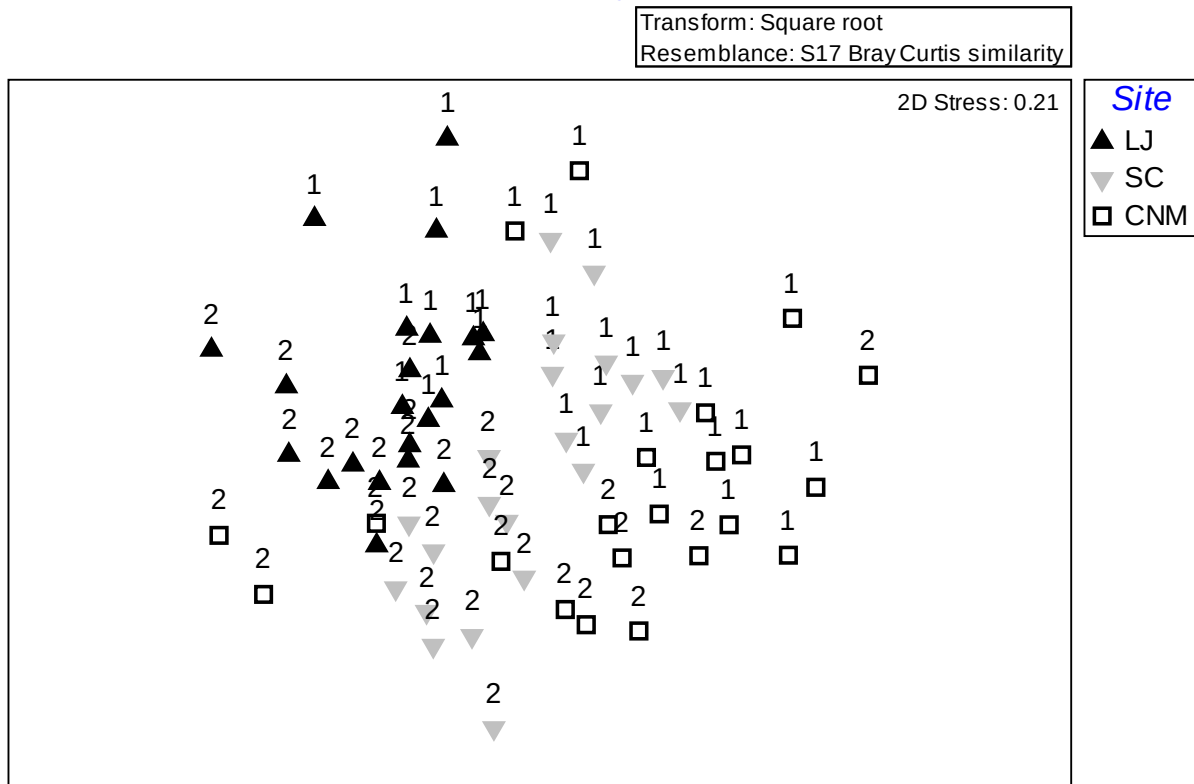


Figure 16. Multi-dimensional scaling plot of functional groups/feeding guilds cover data within turf microhabitats from 11 transects at three sites in San Diego (LJ=La Jolla Shore, SC=Sunset Cliffs, CNM=Cabrillo National Monument) during two sampling periods (1=January 2011, 2-March 2011). Analysis of Similarities (ANOSIM) revealed significant differences among sites and between sampling periods.

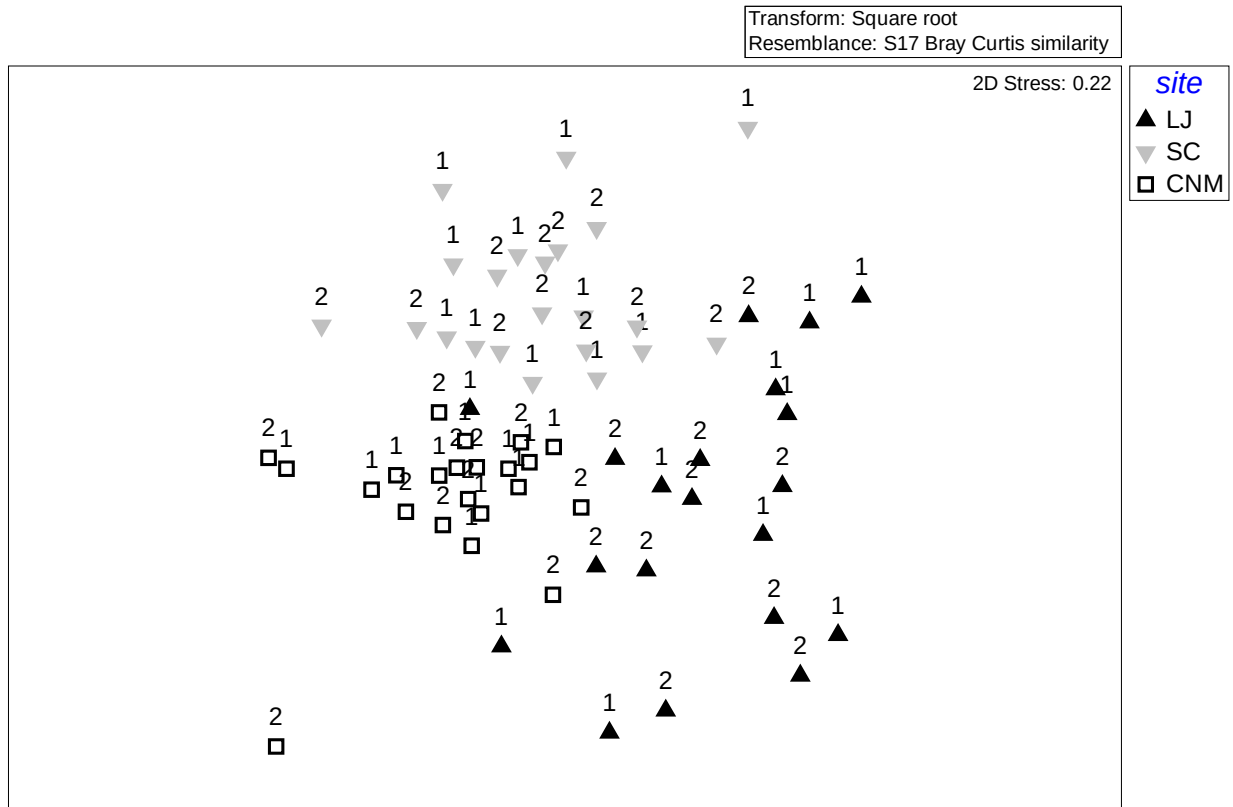


Figure 17. Multi-dimensional scaling plot of cover data with non-algal turf microhabitats from 11 transects at three sites in San Diego (LJ=La Jolla Shore, SC=Sunset Cliffs, CNM=Cabrillo National Monument) during two sampling periods (1=January 2011, 2=March 2011). Analysis of Similarities (ANOSIM) revealed significant differences among sites and between sampling periods.

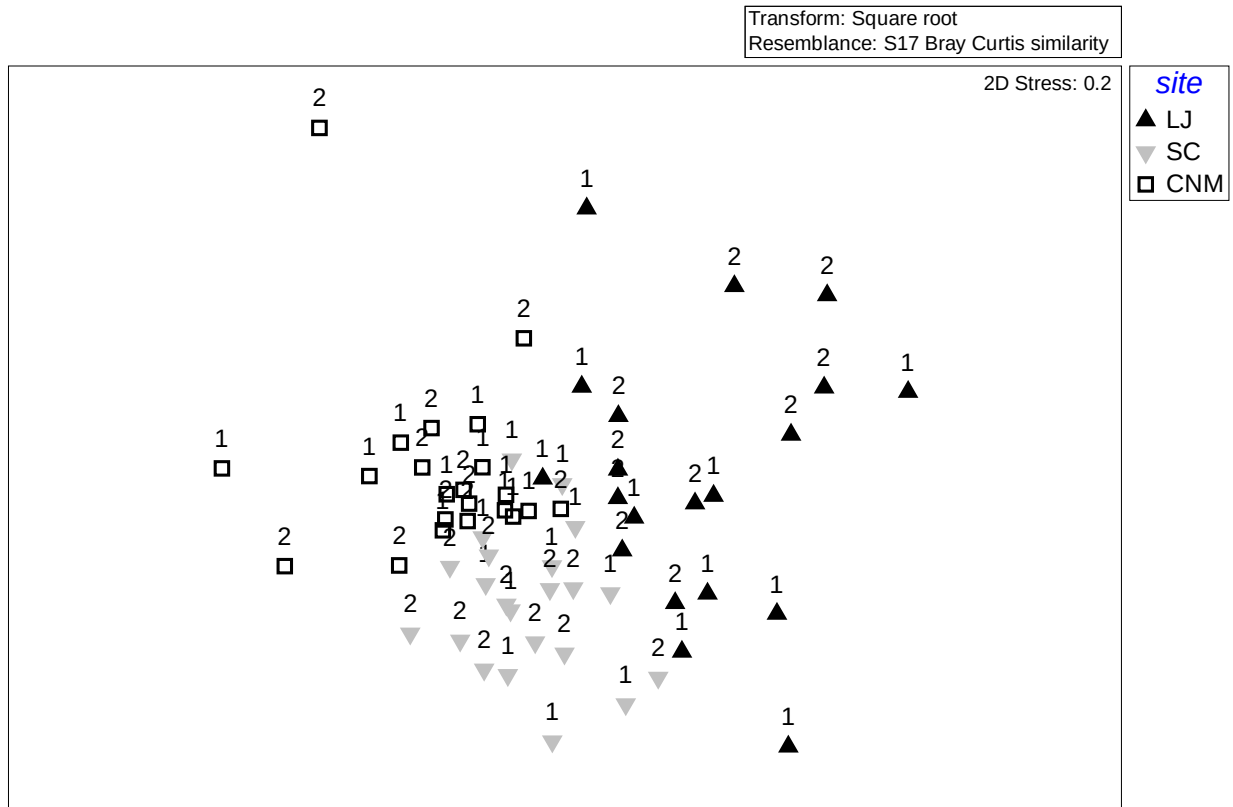


Figure 18. Multi-dimensional scaling plot of functional groups/feeding guilds cover data within non-algal turf microhabitats from 11 transects at three sites in San Diego (LJ=La Jolla Shore, SC=Sunset Cliffs, CNM=Cabrillo National Monument) during two sampling periods (1=January 2011, 2-March 2011). Analysis of Similarities (ANOSIM) revealed significant differences among sites and between sampling periods.

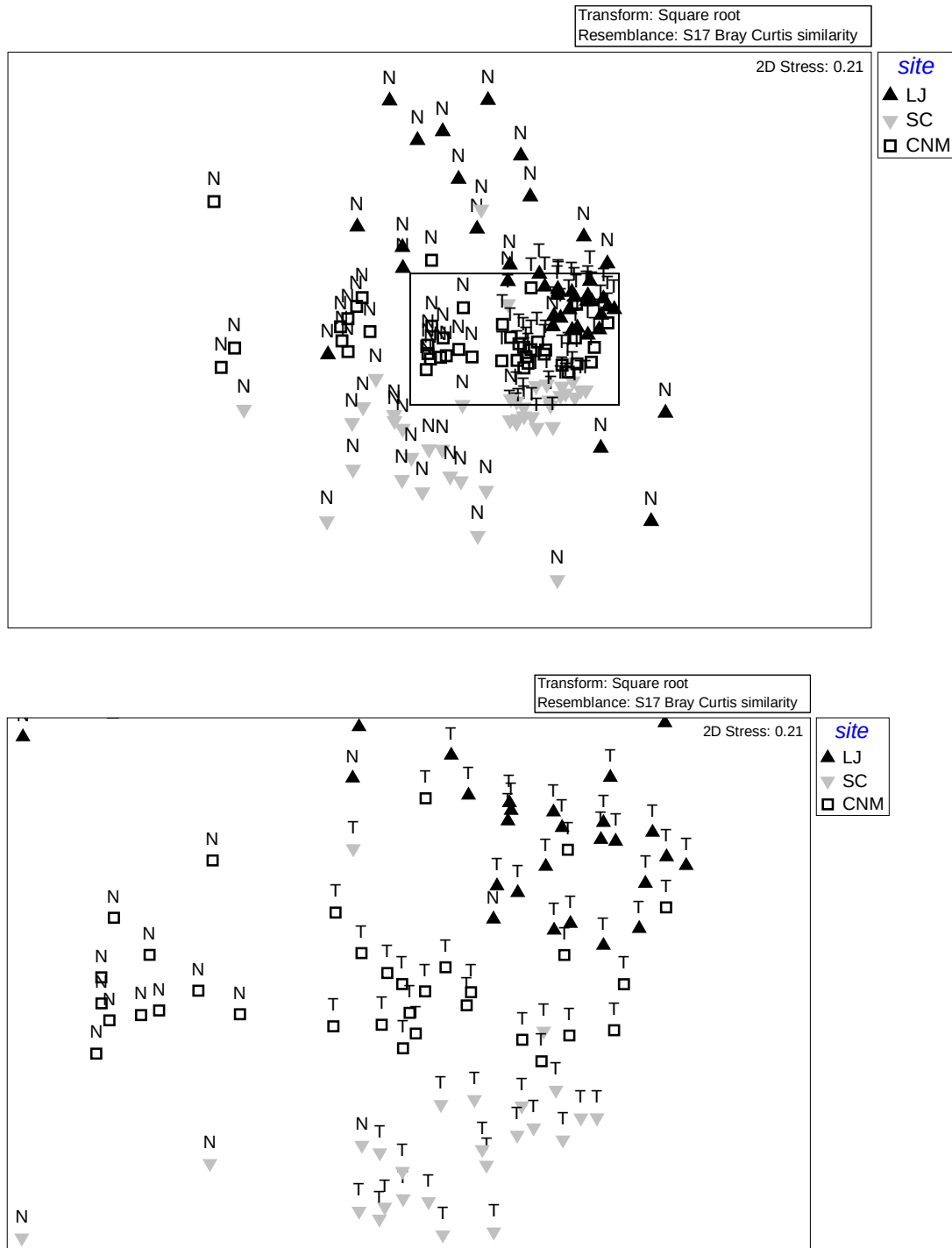


Figure 19. Multi-dimensional scaling plot of cover data with algal turf (T) AND non-algal turf (N) microhabitats from ~11 transects at three sites in San Diego (LJ=La Jolla Shore, SC=Sunset Cliffs, CNM=Cabrillo National Monument) during two sampling periods. Figure below is enhanced view of square in above figure to focus on the turf habitat cluster.

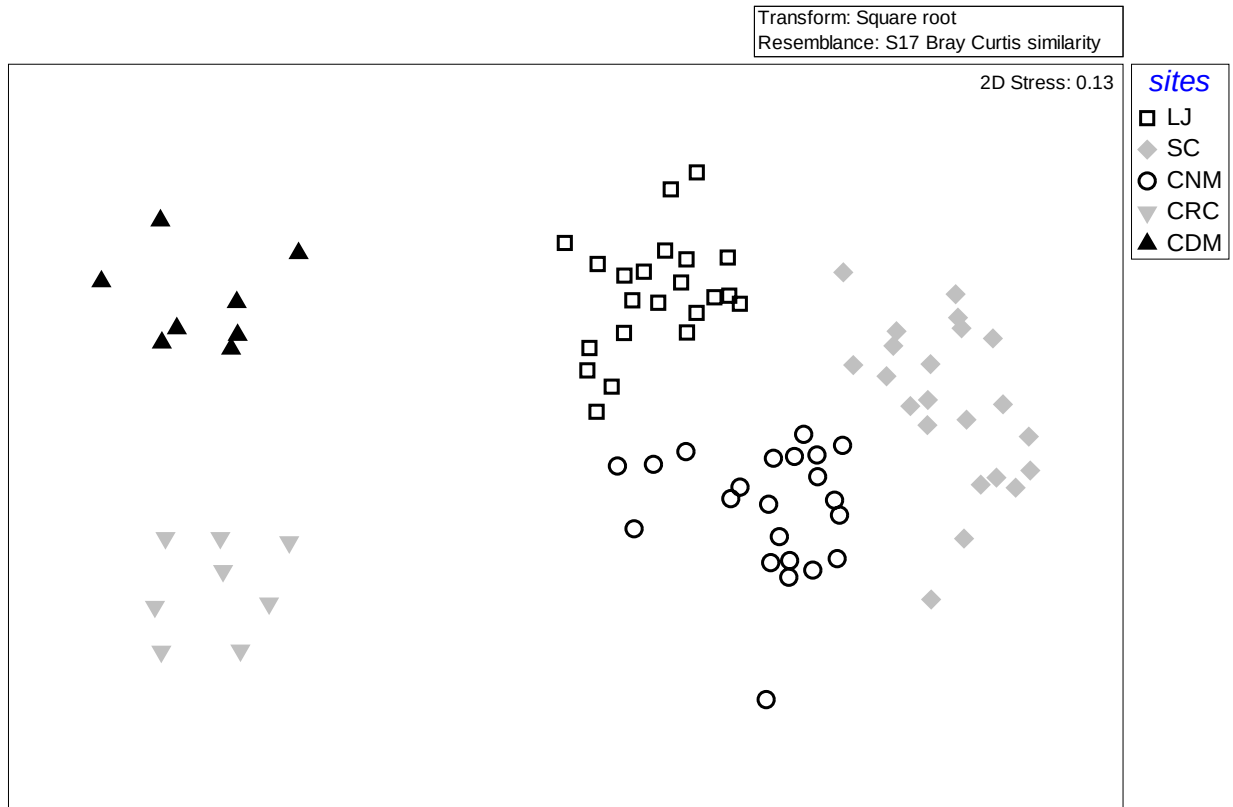


Figure 20. Multi-dimensional scaling plot of species percent cover data from 11 transects at three sites in San Diego (LJ=La Jolla Shore, SC=Sunset Cliffs, CNM=Cabrillo National Monument) and two sites in Orange County for reference (CRC=Crystal Cove, CDM=Corona del Mar).

Turf Depth:

The depth of the algal turf was found to vary among sites and among sampling seasons; no significant interaction term was observed (Two Factor ANOVA; Table 1; Figure 21a). Turf depth was lowest during the March sampling at La Jolla Shores (LJ) and highest in January at Cabrillo National Monument (CNM). In general, LJ had the lowest turf depth during each of the two sampling periods followed by Sunset Cliffs (SC) and CNM. This pattern was confirmed with a One-Factor ANOVA with the two sampling periods combined (Table 1). Here, LJ algal turf depth was significantly lower than CNM but similar to SC; SC was the intermediate and was similar to both LJ and CNM (Figure 21b).

Table 2. Statistical results of Two-Factor ANOVA and One-Factor ANOVA (sampling periods combined) for algal turf depth. Bold p values denote significance.

Test	Factor	df	F stat	p Value
Two Factor ANOVA	Site	2	10.8	<0.001
	Sampling Period	1	13.3	<0.001
	Site X Sampling	2	2.2	0.115
One Factor ANOVA	Site	2	9.4	<0.001

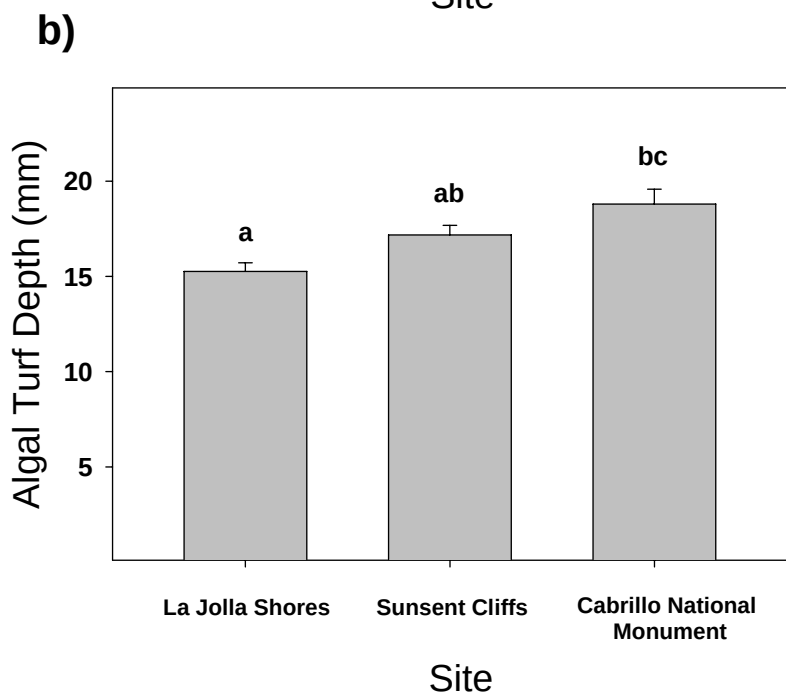
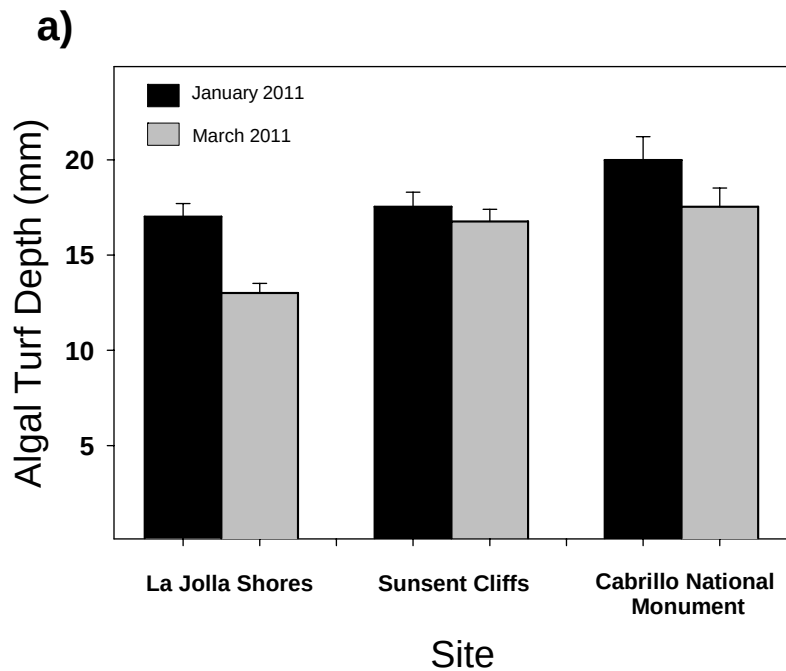


Figure 21. Depth of algal turf (± 1 SE) for three sites in San Diego in a) two sampling periods in January and March 2011 and b) both sampling periods combined. Two Factor ANOVA revealed significant differences among sites and sampling periods (a) while a One-Factor ANOVA revealed significant differences among sites with both sampling periods combined (b). Letters above bars in b signify significantly similar groups using Tukey's post hoc comparisons tests.

Motile Macroinvertebrate Counts

The number of motile invertebrates were counted in quadrats sampled along transect lines during the first sampling season. Because there were very few motile invertebrates, this sampling was not conducted during the second sampling period. Species richness of motile invertebrates was low ranging from 0.78 species per plot at LJ to 1.09 species at SC (Figure 22a) and was not found to be significantly different (One-Factor ANOVA, $df=2$, $F\text{-stat}=0.84$, $p=0.435$). Shannon-Wiener index also varied little among sites ranging from 0.075 at CNM to 0.230 at SC (Figure 22b); no significant difference was detected (One-Factor ANOVA, $df=2$, $F\text{-stat}=1.78$, $p=0.177$).

Community composition of motile invertebrates was found to vary among sites (Figure 23). Analysis of Similarities (ANOSIM) revealed significant differences among sites (Global $R=0.137$, $p<0.001$). Given the low numbers of invertebrates, differences among sites are likely meaningless.

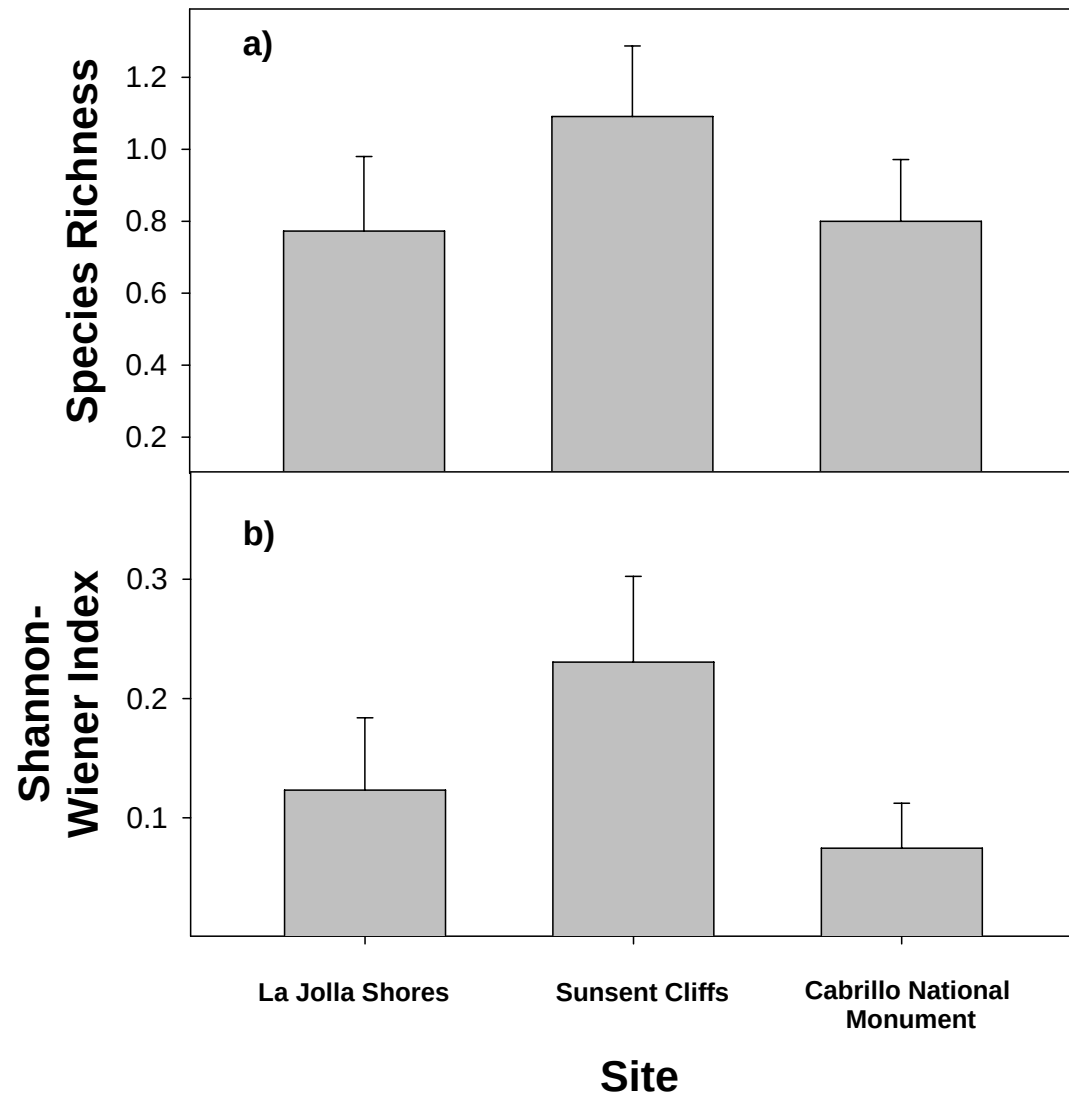


Figure 22. Species richness (a) and Shannon-Wiener diversity index (b) of motile invertebrate counts at three sites in San Diego in January 2011. Note $n=22$ for LJ and SC and 20 for CNM.

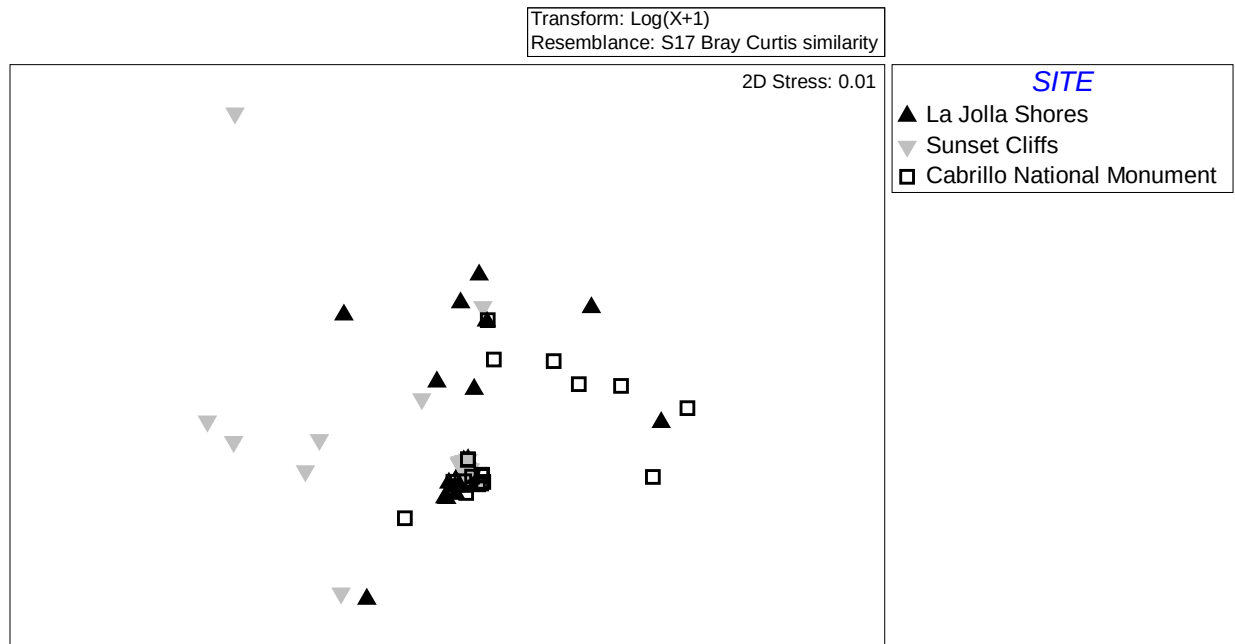


Figure 23. Multi-dimensional scaling plot of motile invertebrate counts data from 20-22 plots at three sites in San Diego (LJ=La Jolla Shore, SC=Sunset Cliffs, CNM=Cabrillo National Monument) during January 2011 sampling. Analysis of Similarities (ANOSIM) revealed significant differences among sites.

Algal Turf Meiofauna Diversity and Community Composition

A total of 21 different taxa levels were examined in algal turf cores, including bivalves, arachnids, insects, amphipods, ostracods, and others (see Appendix II). The total number of meiofauna within core samples varied from 57 to over 1000 individuals. In general, there were no patterns of total abundances except that SC and LJ decreased in their abundances from January to March 2011 but CNM increased markedly during that time period (Figure 24). Two Factor ANOVA resulted in the interaction term being significant (Table 3). No difference was found even if the two sampling periods were combined (Table 3, Figure 25). Species richness was found to be significantly higher at the impact site than the reference sites in both Two Factor and One Factor ANOVAs (Table 3, Figures 24-25). Pielou's evenness did not differ among sites but was elevated during the March sampling (Table 3, Figures 24-25). In general, the Shannon-Wiener diversity index was higher at the impact site and higher at all sites during the March sampling (Table 3, Figures 24-25).

Community composition of meiofauna in algal turf cores differed among sites and between sampling periods (Sites ANOSIM Global $R=0.586$, $p<0.001$; Sampling period ANOSIM Global $R=0.331$, $p<0.001$). All three sites grouped individually with sampling season grouping out within a site as well (Figure 26). The differentiation between SC and CNM was less clear than the differentiation between LJ and both SC and CNM. Similarities Percentage (SIMPER) test showed that LJ differed from SC with LJ having higher abundances of bivalves, cumaceans, copepods, arachnids, and isopods and lower abundances of insects and amphipods. LJ differed from CNM with more bivalves, cumaceans, arachnids, copepods, isopods, and ostracods and fewer insects. SC differed from CNM, mostly driven by fewer insects, and copepods and more amphipods and nematodes. Overlaying of sediment accumulation for each sample revealed no correlation between meiofauna assemblages and the amount of sediment (Figure 27).

Abundances of individual taxa varied greatly among sites and sampling seasons (Appendix II). For simplicity, taxa differences among sites were examined using a One Factor ANOVA (Appendix II). Of note, the impacted site had higher numbers of arachnids (mites), copepods, cumaceans (hooded shrimp), isopods, and bivalves and low numbers of insects.

Algal Turf Sediment Accumulation

Sediment volume in collected cores ranked lower at the impacted site but no difference was found using Two Factor and One Factor ANOVAs (Table 4, Figure 28a,b).

Table 3. Statistical results of Two-Factor ANOVAs and One-Factor ANOVAs (with sampling periods combined) for three diversity measures for meiofaunal data. Bold p values denote significance.

Diversity Measure	Statistical Test	Factor	df	F stat	p value
Total Number of Individuals	Two Factor ANOVA	Site	2	0.7	0.502
		Sampling	1	0.2	0.693
		Site X Sampling	2	10.5	<0.001
Species Richness	Two Factor ANOVA	Site	2	16.5	<0.001
		Sampling	1	0.6	0.430
		Site X Sampling	2	0.75	0.474
	One Factor ANOVA	Site	2	18.2	<0.001
Pielou's Evenness	Two Factor ANOVA	Site	2	0.52	0.598
		Sampling	1	7.08	0.010
		Site X Sampling	2	0.8	0.457
	One Factor ANOVA	Site	2	0.7	0.501
Shannon-Wiener Index	Two Factor ANOVA	Site	2	7.18	0.001
		Sampling	1	4.0	0.049
		Site X Sampling	2	0.8	0.462
	One Factor ANOVA	Site	2	7.78	0.001

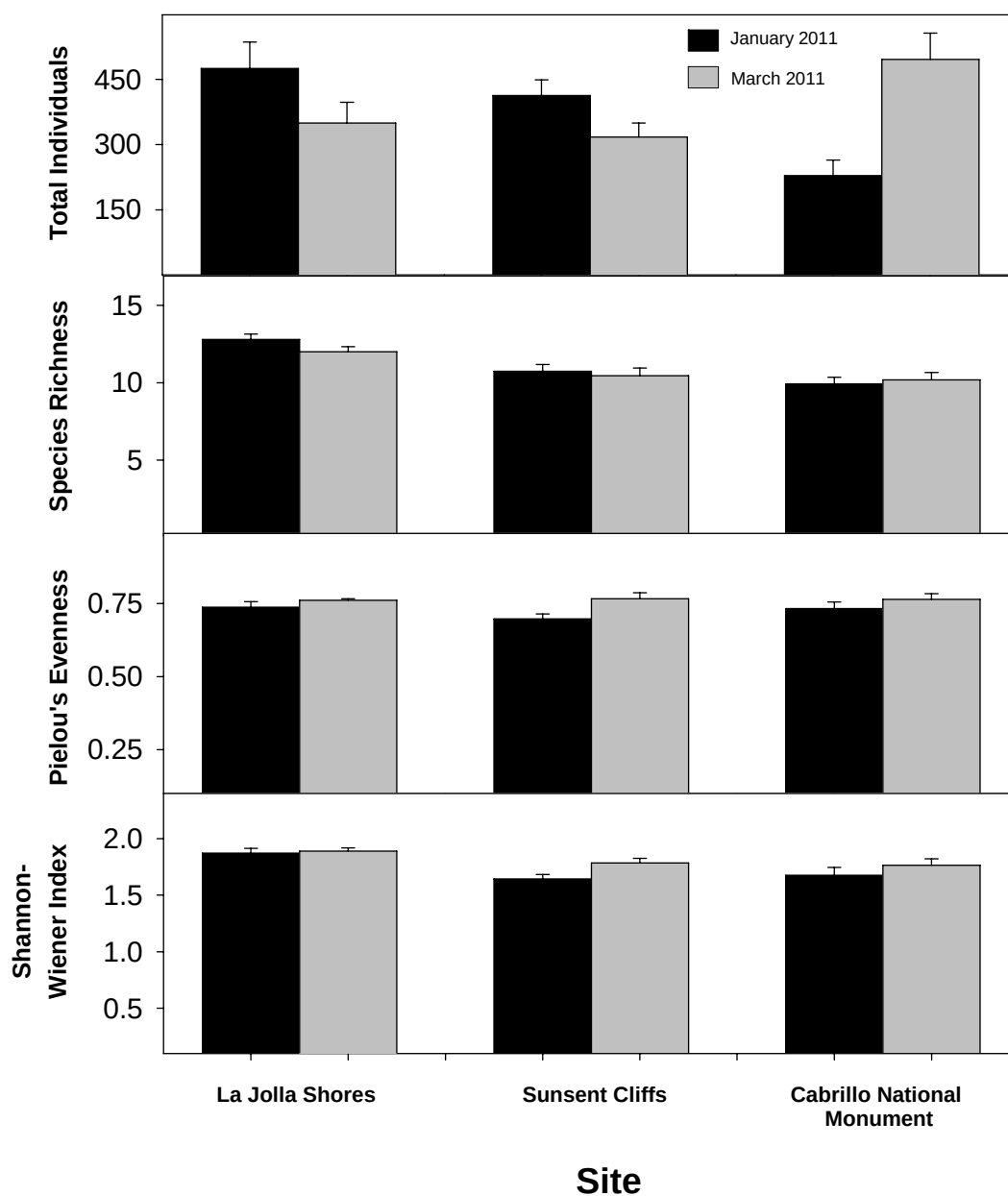


Figure 24. Total number of individuals and diversity measures (± 1 SE), including species richness, Pielou's Evenness, and Shannon-Wiener index, of meiofauna from algal turf cores at three sites in San Diego during January and March 2011 sampling periods. Statistical results for these data are located in Table 3 (Two Factor ANOVA).

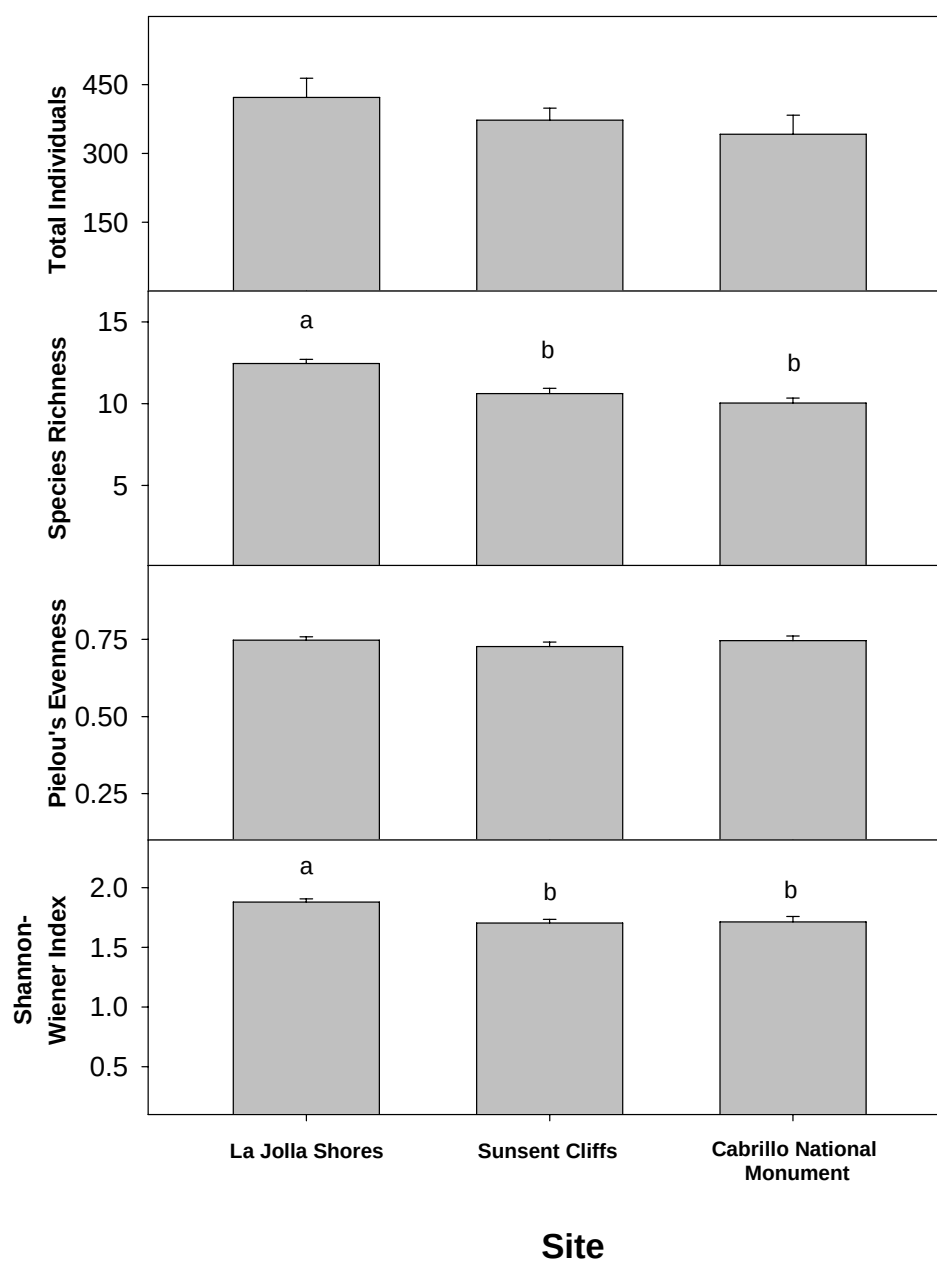


Figure 25. Total number of individuals and diversity measures (± 1 SE), including species richness, Pielou's Evenness, and Shannon-Wiener index, of meiofauna from algal turf cores at three sites in San Diego with both sampling periods combined. Statistical results for these data are located in Table 3 (One Factor ANOVA). Letters above bars signify significantly similar groups using Tukey's post hoc comparisons tests when differences were found.

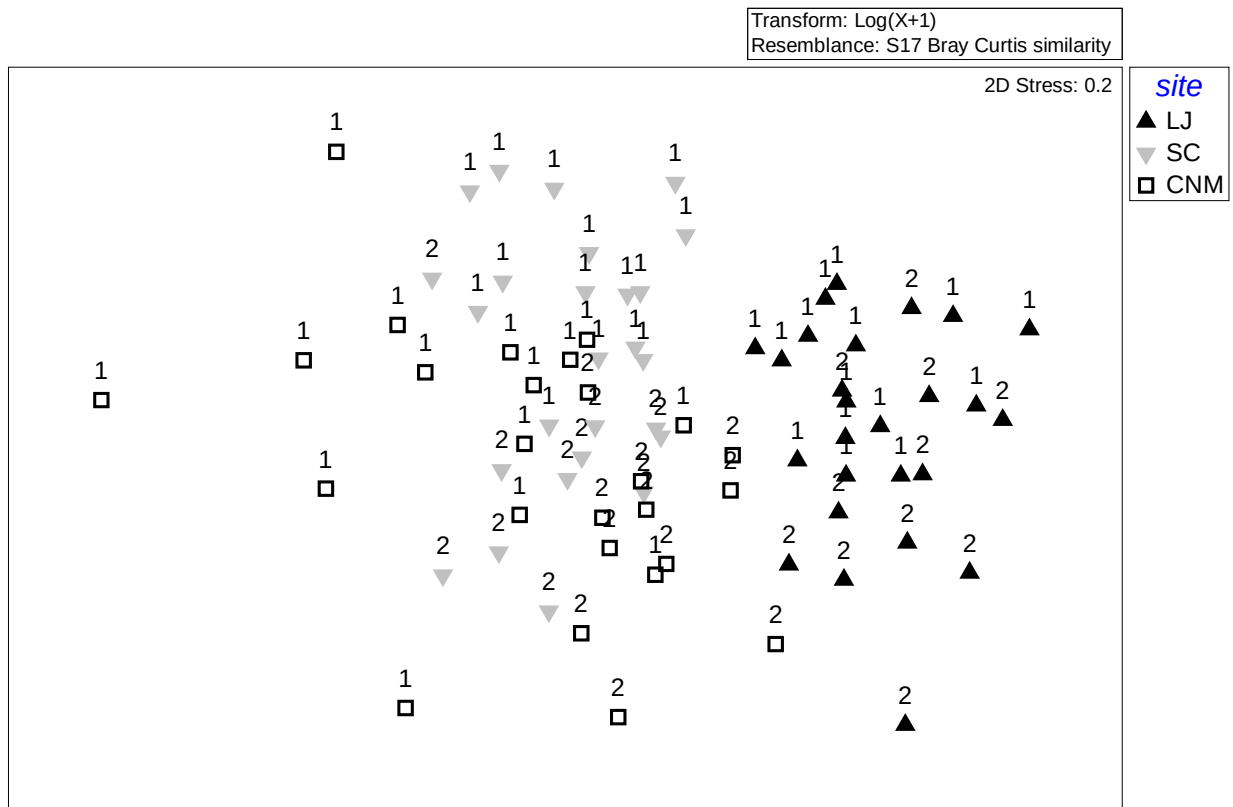


Figure 26. Multi-dimensional scaling plot of meiofaunal counts from 15 cores in January 2011 (1) and 11 cores in March 2011 (2) at three sites in San Diego (LJ=La Jolla Shore, SC=Sunset Cliffs, CNM=Cabrillo National Monument). Analysis of Similarities (ANOSIM) revealed significant differences among sites and between sampling periods.

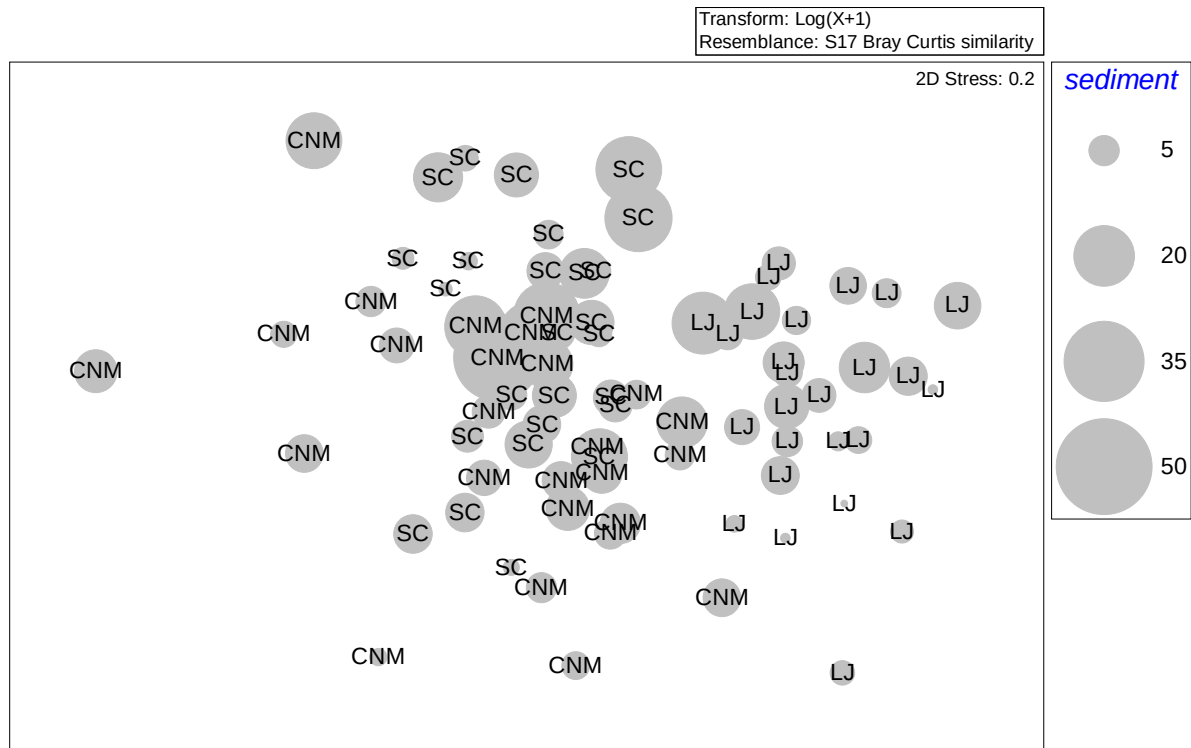


Figure 27. Multi-dimensional scaling plot of meiofaunal counts from 15 cores in January 2011 and 11 cores in March 2011 at three sites in San Diego (LJ=La Jolla Shore, SC=Sunset Cliffs, CNM=Cabrillo National Monument). Overlaid on individual points is the relative sediment accumulation in core samples.

Table 4. Statistical results of Two-Factor ANOVA and One-Factor ANOVA (sampling periods combined) for sediment accumulation in algal turf cores.

Test	Factor	df	F stat	p Value
Two Factor ANOVA	Site	2	2.3	0.110
	Sampling Period	1	3.1	0.084
	Site X Sampling	2	0.3	0.757
One Factor ANOVA	Site	2	2.4	0.100

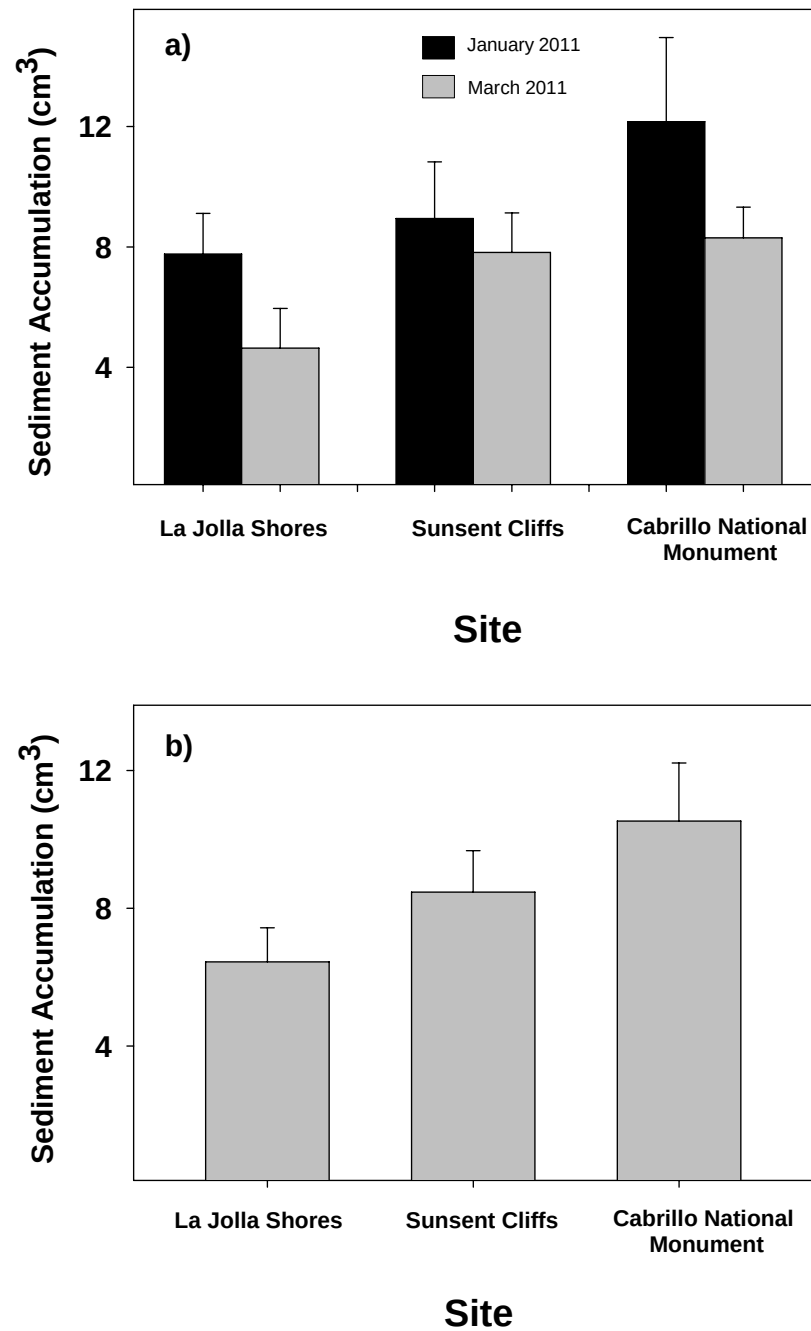


Figure 28. Accumulation of sediment (± 1 SE) in cores collected in algal turfs at three sites in San Diego in a) two sampling periods in January and March 2011 and b) both sampling periods combined. Neither data sets yielded significant differences among factors (Table 4).

Discussion

The composition of flora and fauna of rocky intertidal habitats in southern California can be influenced by a large suite of natural factors, including biotic interactions, oceanographic patterns, larval supply, sand burial and scour, wave action, nutrients, desiccation, temperature and salinity fluctuations, substrate type, bench slope, and others (see Denny and Gaines, 2007). Equally important in heavily urbanized areas, such as that of southern California, are the impacts of anthropogenic disturbances from trampling, overexploitation, and pollution, among other impacts (see Crowe et al., 2000; Thompson et al., 2002). Comparing the biological communities among sites can elucidate differences in community structure that can be speculatively attributed to any number of influencing factors. When trying to examine the impacts of a specific factor, attempts are made to reduce the influence of other conditions by choosing sites that are similar (Murray et al., 2006). Typically, this is difficult as factors can vary over a very small spatial scale, thus patterns must be interpreted with caution. In many cases, a factor influencing community structure and functioning is examined by comparing a number of sites with and without the impact factor thus providing multiple replicates of impacted and non-impacted communities (Murray et al., 2006). In this study, we attempted to examine the influence of a storm drain on the biodiversity and community composition of a rocky intertidal ecosystem at La Jolla Shores (LJ). However, given that only one site with storm water impacts was chosen and compared to reference sites (Cabrillo National Monument (CNM) and Sunset Cliffs (SC)), no causation of differences of the biological assessments can be made. In simplest terms, this study examines differences in the biological communities of three sites; any shifts in community structure at the impacted site may be attributed to multiple factors. Speculatively, some patterns will be explained by possible influences of disturbances, and may or may not be disturbances associated with the storm drain, including exposure to freshwater, sediment, debris, and other pollutants. The lack of understanding of the impacts storm water runoff on rocky intertidal species also makes correlation with storm drain runoff at the impacted site elusive.

Comparisons of the biodiversity and community structure of macrophytes, macroinvertebrates, and meiofauna at three locations in San Diego revealed that each site was significantly different from each other using numerous parameters. Biodiversity of macrophytes and macroinvertebrates was lowest at the impact site (LJ) while biodiversity of meiofauna varied among sampling periods among sites but with some patterns of higher diversity at the impact site. Multi-variate statistical analyses of community structure of 1) macrophytes and macroinvertebrates and 2) meiofauna revealed that the impact site was clearly differentiated from the other two sites. The percent cover of a list of macrophytes species also differed among sites with some of these species being large contributors to the differences in overall community structure found in multi-variate analyses. There were patterns of decreases in seagrass and increases in disturbance tolerant species at the impact site, although not consistent across all disturbance indicators. Meiofaunal taxa counts revealed that a number of taxa were higher at the impact site, including organism that we would expect to be negatively impacted by storm runoff. Comparisons of turf depth in the field and in collected turf cores also revealed that the impact site had significantly lower depth and marginally lower sand accumulation. Despite some clear differences, evidence of obvious and consistent storm water impacts are relatively weak. However,

differences among sites does warrant possible future examinations in a more robust manner using multiple impacted and non-impact sites.

Community composition of the impacted site was significantly different than the reference sites suggesting shifts in the abundances and/or species make-up. This was also observed in the community composition of the meiofauna associated with the algal turf. However, the two reference sites also grouped separately indicating that community composition for both macroflora/fauna and meiofauna varied among all three sites and that differences may not be attributable solely to the influence of storm drain runoff. If storm drain runoff were the driving factor for the shift in the composition of the impacted site, we would loosely expect that the two reference sites would group together. Furthermore, although differences in community structure were determined, these differences may not be overwhelmingly large or ecologically important. Multi-variate statistics compares each site in relation to each other; although there was little variation within a site, there were enough differences in assemblages among sites to note significance. To examine the overall scale of difference in community structure for macrophytes and macroinvertebrates, two additional rocky intertidal sites from previously sampled sites in Orange County were added. These sites are not, in our opinion, remarkably different from the intertidal sites in San Diego; however, when all five sites were analyzed together, the San Diego sites tended to clump more closely together (albeit still significantly different from one another). This suggests, in our opinion, that San Diego sites differ in only a small overall degree.

The three sites sampled were relatively similar with a dominance of red algal turf, which was the main focus of our sampling. However, interspersed within the site were a number of small and large tidepools and, to a lesser degree, boulders that fell within our sampling scope. Given that different sets of species live in algal turfs as compared to tidepools, we conducted additional analyses of our data to determine if assemblage differences among sites were attributable to differences in the availability of different habitat types at each site. Comparisons of turf and non-turf (mostly tidepools) habitat types revealed similar patterns to the full data set suggesting availability of habitats was not an important driver. However, analyses suggested that community composition differences within non-turf habitats were much larger than community composition differences within algal turfs. Given that sampling was somewhat limited in tidepools, patterns are somewhat inconclusive due to lack of large replication. However, patterns do warrant further research that targets robust sampling of tidepool habitats. Tidepools may be better indicators of storm water runoff, for example, as storm water is more likely to accumulate in these pools during low tides. In addition, large declines in seagrass, a species typical in tidepools, were observed at the impact site and may be a good indicator of disturbance thus warranting specific study.

The types of macro-species present at each site and their relative abundances varied and are the basis for differences in community composition. However, many species play similar ecological roles and differences in the species present at different sites may not be ecologically important if the different species are playing the same ecological role or are indicators of the same processes that control those groups of similar species. To examine this, macrophytes were placed in functional groups based on established characteristics (Steneck and Dethier, 1994). For example, LJ was dominated by the red algal species *Corallina pinnatifolia/vancouveriensis* while SC was dominated by *Jania crass*. Given that both

species are articulated corallines playing the same ecological role, differences in sites at the species level may not be apparent at the ecological level. Similarly, macroinvertebrates were placed into feeding guilds (Murray et al. 2006) whereby a list of species that are filter feeders, for example, are placed into the same category. Analyses of community structure based on functional groups and feeding guilds yielded the same patterns as species data suggesting that differences among sites do relate to potential differences at the ecological level.

Although there is a large list of macro-species present at some or all sites (Appendix I), with each varying among sites in abundance to different degrees, there were some notable patterns. At the impacted site, there were relatively higher abundances of bare rock, *Corallina pinnatifolia/vancouveriensis*, articulated corallines, *Caulacanthus*, encrusting algae and *Ulva*. These species/taxa are species that tend to be described as opportunistic and/or disturbance tolerant species suggesting the LJ is more disturbed than the reference locations. For example, encrusting algae, although not considered an opportunistic species because of slow growth, is common in habitats with frequent and intense disturbance (Steneck, 1986; Steneck and Martone, 2007). Since encrusting algae are quickly overgrown by more erect algal forms, they are often found when other fleshy and erect algae are not present due to consistent disturbance. *Ulva* is considered an opportunistic species that takes advantage of newly opened space in a space-limited habitat (Sousa, 1979); as with encrusting algae, *Ulva* is not a good competitor against slow growing, late-succession species. Disturbance may also be indicated by the high amount of bare rock at LJ; given space limitations, it is infrequent that bare space is available unless disturbance has occurred recently and species have yet to colonize the newly opened space. The presence of disturbance tolerant species and bare rock suggest that slower growing, late-successional species have had inadequate time to recolonize the habitat or disturbance occurs frequently enough that recolonization never initiates. The low levels of biodiversity at the impacted site may also be further evidence of disturbance, although diversity as an indicator of disturbance is complex as low diversity can occur at both high and low levels of disturbance, while often highest during intermediate levels of disturbance (Sousa, 1979). The increased presence of the non-native red alga *Caulacanthus ustulatus* may also speculatively be an indicator of disturbance. It has been well documented that exotic seaweeds become established in disturbed habitats (see review Williams and Smith, 2007) and observations of the distribution of *Caulacanthus* at other sites in southern California suggest similar correlations. In addition to being disturbance tolerant species, *Ulva* (see summary Teichberg et al., 2010), *Caulacanthus* (Diez et al. 1999) and some *Coralline* spp. (Littler and Murray, 1975; Diez et al. 1999) are known to be enhanced by nutrient enrichment and that higher cover of these species at the impacted site may be due to increased nutrients in the storm water flow.

In addition to those species discussed with higher abundances, there were a number of species that were lower in abundance at the impacted site, including seagrass (*Phyllospadix*) and grazers. Seagrass was markedly lower at LJ, several magnitudes less than the reference sites (Appendix I). In addition, if non-turf habitat type only is examined (seagrass is more common in tidepools than turfs), differences in seagrass are even larger with only 5% of non-turf habitat at the impacted site consisting of seagrass while seagrass covered 40-55% of non-turf habitats at the reference sites. Although variable in responses, seagrasses, including *Phyllospadix*, are susceptible to nutrients and sedimentation (see

summary Bull et al., 2004). The percent cover of sand also was lower at the impacted site, nearly a third less than CNM. Sand may be transported out of the rocky intertidal zone by storm water runoff flow. Transport of sediment out the rocky intertidal zone may also be indicated by a lower depth of the algal turf at the impacted site; speculatively, sand accumulation in the turfs leads to increase turf depth. The ecological impacts of loss of sand accumulation can have different impacts on different species; however, sediment accumulation appeared to have no impact on the meiofaunal assemblages, the taxa most likely to be influenced by sediments.

Although the above examples indicate more disturbance tolerant species are present at the impacted sites, there are some inconsistent patterns. For example, decreases in fleshy algae and tough and leathery algae would be expected in disturbed habitats while increases in filamentous algae and blade-like sheet algae would be concurrently observed. However, percent cover of these seaweed types was similar among all three sites. In addition, the numbers and diversity of meiofauna taxa did not match expectations for disturbance. Meiofauna, generally speaking, were more diverse at the impact site. In addition, the arachnids, copepods, cumaceans, and isopods were higher at the impacted site; although little research has been conducted on pollutant impacts of these type of meiofauna, we would expect that they would be negatively impacted.

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Appendix I: Species Cover and ANOVA Results

Appendix I. Mean percent cover of individual species, functional groups (macrophytes), and feeding guilds (macroinvertebrates) at three sites during two sampling periods and with all samples combine (All). Included are p value results from One-Factor ANOVA comparing sites using all data and the patterns of Tukey's multiple comparisons post hoc tests. Bold p values indicate significance.

	LJ	LJ	LJ	SC	SC	SC	CNM	CNM	CNM	ANOVA p value	Tukey's Multiple comparisons
	Jan 11	Mar 11	All	Jan 11	Mar 11	All	Jan 11	Mar 11	All		
Bare Rock	12.1	11.8	12.0	3.1	3.6	3.4	3.5	5.8	4.7	<0.001	CNM=SC<LJ
Bare Sand	8.4	3.1	5.7	5.3	14.5	9.9	21.3	15.0	18.1	<0.001	LJ=SC<CNM
Seaweeds:											
<i>Corallina pinnatifolia/vancouveriensis</i>	61.6	62.7	62.2	44.4	34.1	39.2	32.2	25.2	28.7	<0.001	CNM<SC<LJ
<i>Caulacanthus ustulatus</i>	21.5	15.5	18.5	0.1	0.5	0.3	2.7	2.6	2.7	<0.001	CNM=SC<LJ
<i>Ulva californica</i>	16.3	39.2	27.7	44.4	9.8	27.1	12.4	34.6	23.5	0.635	
<i>Ceramium</i> spp.	12.3	21.0	16.6	5.9	1.7	3.8	9.3	9.6	9.5	<0.001	SC<CNM<LJ
<i>Chondracanthus canaliculatus</i>	2.6	5.2	3.9	12.7	11.3	12.0	14.2	13.9	14.0	<0.001	LJ<SC=CNM
<i>Lomentaria hakodatensis</i>	7.5	2.1	4.8	8.1	4.4	6.2	2.8	2.5	2.6	0.012	CNM=LJ, LJ=SC, CNM<SC
<i>Lithothrix aspergillum</i>	4.3	1.2	2.7	12.5	6.1	9.3	2.5	5.0	3.8	<0.001	CNM=LJ<SC
<i>Centroceras clavulatum</i>	3.7	0.3	2.0	4.5	29.0	16.8	5.8	3.8	4.8	<0.001	CNM=LJ<SC
Crustose corallinaceae	3.4	1.5	2.5	0.5	2.8	1.7	0.7	0.3	0.5	0.017	CNM=SC, SC=LJ, CNM<LJ
<i>Jania crassa</i>	0.7	0.6	0.7	10.7	20.5	15.6	0.3	0.2	0.2	<0.001	CNM=LJ<SC
<i>Phyllospadix</i> spp.	0.6	0.7	0.7	12.8	10.6	11.7	24.1	20.3	22.2	<0.001	LJ<SC<CNM
<i>Gelidium pusillum</i>	2.6	6.5	4.5	3.9	4.3	4.1	4.7	6.1	5.4	0.534	
<i>Chaetomorpha aerea</i>	1.6	2.6	2.1	1.7	2.9	2.3	2.1	0.5	1.3	0.193	
<i>Acrosorium ciliatum</i>	1.5	0.0	0.8	2.6	4.6	3.6	0.1	0.0	0.0	<0.001	CNM=LJ<SC
Psuedolithoderma/Ralfsiaceae	1.5	1.4	1.5	0.4	0.4	0.4	0.6	0.7	0.7	0.053	
<i>Sargassum muticum</i>	1.0	0.3	0.6	0.7	0.4	0.5	0.6	1.8	1.2	0.176	
<i>Cryptopleura crispa/corallinara</i>	0.9	0.6	0.8	1.5	3.1	2.3	0.5	0.2	0.3	<0.001	CNM=LJ<SC
<i>Pterocladia capillacea</i>	0.8	0.5	0.7	2.0	1.6	1.8	0.3	0.2	0.2	<0.001	CNM=LJ<SC
<i>Gastroclonium parvum</i>	0.6	0.0	0.3	0.5	1.0	0.7	0.8	0.5	0.7	0.432	
<i>Lithothamium pacificum</i>	0.5	0.5	0.5	0.0	0.0	0.0	0.0	0.1	0.0	<0.001	CNM=SC<LJ
<i>Egregia menziesii</i>	0.5	0.3	0.4	0.3	0.0	0.1	0.4	0.2	0.3	0.663	
<i>Laurencia pacifica</i>	0.5	0.0	0.2	0.1	0.0	0.0	0.1	0.2	0.1	0.588	
<i>Silvetia compressa</i>	0.5	0.2	0.3	0.0	0.0	0.0	0.2	0.4	0.3	0.126	
<i>Ulva intestinalis</i>	0.5	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.004	CNM=SC<LJ
<i>Jania tenella</i>	0.3	0.0	0.1	0.1	0.1	0.1	0.0	0.2	0.1	0.858	
<i>Gelidium coulteri</i>	0.2	0.0	0.1	0.1	0.3	0.2	0.0	0.0	0.0	0.207	

[illegible]

UNIDed Amphipod	0.2	0.2	0.2	0.5	1.3	0.9	0.3	1.5	0.9	0.076	
<i>Balanus glandula</i>	0.0	0.0	0.0	0.0	0.2	0.1	0.0	0.1	0.0	0.522	
<i>Agnathostoma eisini</i>	0.0	0.0	0.0	0.2	0.5	0.3	0.2	0.0	0.1	0.017	LJ=CNM, CNM=SC, LJ<SC
UNIDed Polychaeta	0.0	0.0	0.0	0.4	0.1	0.2	0.0	0.0	0.0	0.004	CNM=LJ<SC
<i>Pagurus samuelis</i>	0.0	0.1	0.0	0.0	0.2	0.1	0.3	0.0	0.1	0.778	
<i>Pachygrapsus crassipes</i>	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.374	
<i>Megastrea undosa</i>	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.374	
<i>Epiactis prolifera</i>	0.0	0.0	0.0	0.0	0.3	0.1	0.0	0.0	0.0	0.374	
<i>Chlorostoma aureotincta</i>	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.374	
<i>Pugettia producta</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.1	0.131	
<i>Littorina</i> spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.1	0.374	
<i>Nuttalina</i> spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.2	0.1	0.043	LJ=SC<CNM
<i>Serpulorbis squamigerus</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.374	
<i>Tetraclita rubescens</i>	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.374	
<i>Stenoplax conspicua</i>	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.374	
<i>Notoplana acticola</i>	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.374	
<i>Navanax inermus</i>	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.1	0.0	0.609	
Functional Groups Seaweeds											
Articulated Corallines	66.9	64.5	65.7	68.1	62.3	65.2	35.1	30.5	32.8	<0.001	CNM<LJ=SC
Encrusting Algae	5.5	4.0	4.7	1.2	3.3	2.2	1.5	1.2	1.3	0.001	CNM=SC<LJ
Filamentous-like Algae	17.9	24.2	21.0	16.5	36.8	26.6	19.4	17.1	18.2	0.032	CNM=LJ, LJ=SC, CNM<SC
Fleshy Algae	38.0	30.0	34.0	39.4	34.0	36.7	28.7	29.5	29.1	0.036	CNM=LJ, LJ=SC, CNM<SC
Blade Algae	17.6	39.8	28.7	46.0	13.1	29.5	12.8	35.6	24.2	0.482	
Tough and Leathery Algae	2.0	0.7	1.4	1.4	0.9	1.1	1.2	2.4	1.8	0.534	
Feeding Guild Macroinvertebrates											
Scavengers	0.2	0.3	0.2	0.0	0.5	0.2	1.7	1.6	1.7	<0.001	LJ=SC<CNM
Filter Feeders	0.5	0.5	0.5	0.8	0.6	0.7	0.6	1.2	0.9	0.557	
Grazers	0.5	0.2	0.3	0.9	2.1	1.5	0.9	1.7	1.3	0.012	LJ<CNM=SC
Carnivores	0.0	0.0	0.0	0.2	0.0	0.1	0.0	0.1	0.0	0.522	

Appendix II: Meiofaunal Counts and ANOVA Results

Appendix II. Mean counts of grouped taxa from 3.75 X 3.75 cm cores of algal turfs from at three sites during two sampling periods and with all samples combine (All). Included are p value results from One-Factor ANOVA comparing sites using all data and the patterns of Tukey's multiple comparisons post hoc tests. Bold p values indicate significance.

	LJ	LJ	LJ	SC	SC	SC	CNM	CNM	CNM	ANOVA p value	Tukey's Multiple comparisons
	Jan 11	Mar 11	All	Jan 11	Mar 11	All	Jan 11	Mar 11	All		
Annelids, Polychaetes	11.5	21.5	15.8	18.7	14.5	16.9	9.3	9.5	9.4	0.305	
Arthropods, Amphipods	14.1	23.4	18.0	74.5	32.2	56.6	14.1	44.8	27.1	P<0.001	LJ=CNM<SC
Arthropods, Arachnids	8.5	8.4	8.4	0.0	0.1	0.0	0.1	0.8	0.4	P<0.001	CNM=SC<LJ
Arthropods, Copepods	62.6	61.9	62.3	5.1	19.0	11.0	8.0	67.1	33.0	P<0.001	SC=CNM, CNM=LJ, SC<LJ
Arthropods, Cumaceans	19.6	8.5	14.9	0.1	0.0	0.0	0.1	0.0	0.0	P<0.001	CNM=SC<LJ
Arthropods, Decapods	0.1	0.0	0.0	0.1	0.1	0.1	0.1	0.1	0.1	0.592	
Arthropods, Insects	3.1	1.6	2.5	2.2	31.3	14.5	7.6	44.5	23.2	0.004	LJ=SC, SC=CNM, LJ<CNM
Arthropods, Isopods	6.5	6.3	6.4	1.2	0.5	0.9	0.6	2.4	1.3	P<0.001	CNM=SC<LJ
Arthropods, Pycnogonids	1.6	0.1	1.0	0.7	0.4	0.6	0.1	0.0	0.0	0.009	CNM=SC, SC=LJ, CNM<LJ
Arthropods, Ostracods	79.7	70.6	75.8	49.3	43.4	46.8	27.0	152.6	80.2	0.089	
Cnidarians, Anthozoans	0.0	0.2	0.1	0.2	0.0	0.1	0.1	0.1	0.1	0.9	
Cnidarians, Colonial Hydroids	0.7	0.1	0.4	0.3	0.0	0.2	0.3	0.0	0.2	0.515	
Echinoderms, Ophiuroids	0.1	0.3	0.2	0.7	0.3	0.5	0.0	0.0	0.0	0.010	CNM=LJ, LJ=SC, CNM<SC
Echinoderms, Holourthroids	0.0	0.0	0.0	0.0	0.2	0.1	0.0	0.0	0.0	0.132	
Foraminifera	41.1	13.0	29.2	90.5	32.6	66.0	60.7	40.4	52.1	0.005	LJ=CNM, CNM=SC, LJ<SC
Molluscs, Bivalves	31.1	28.1	29.8	3.2	1.8	2.6	2.7	3.8	3.2	P<0.001	CNM=SC<LJ
Molluscs, Gastropods	142.3	85.1	118.1	114.2	113.8	114.0	82.4	105.1	92.0	0.448	
Molluscs, Polyplacophora	0.1	0.0	0.0	0.1	0.3	0.2	0.2	0.4	0.3	0.152	
Nematodes	52.6	20.4	39.0	51.4	25.7	40.5	14.5	24.1	18.5	0.051	
Platyhelminthes, Turbellids	0.0	0.0	0.0	0.1	0.3	0.2	0.2	0.1	0.2	0.246	
Sipunculids	0.2	0.4	0.3	0.7	1.0	0.8	0.6	0.5	0.5	0.123	