

THE EFFECTS OF PROXIMITY TO A SUBTIDAL CHANNEL ON HABITAT
UTILIZATION OF INTERTIDAL OYSTER REEFS

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ABSTRACT

Habitat use may differ with certain landscape characteristics such as patch quality and size, the presence of corridors, the connectivity of the landscape, and proximity to other habitats. In estuarine systems, intertidal oyster reefs are an important habitat whose functions may be explained by landscape theories. Proximity to other structured habitat has been shown to affect utilization of oyster reefs by both resident and transient species. Reef proximity to subtidal channels and upper intertidal areas, the source areas for transient and resident fauna, respectively, may be an important factor affecting reef utilization. This study tested whether proximity to subtidal channels influenced utilization of intertidal oyster reefs and whether microhabitat utilization varied within reef edge areas. Enhanced densities of finfish and decapods were seen around reefs at all distances compared with unstructured sandflats. Total abundances were higher over the reef interface (sand edges) compared to over shell, however smaller individuals of dominant species (juvenile *Lagodon rhomboides* and juvenile *Leiostomus xanthurus*) were found using reef interior shell to a greater extent than larger individuals, which used sand edge and sandflat areas. Reef residents showed no variation in abundance based on distance from a channel. *Fundulus heteroclitus* and larval *L. rhomboides* were the only transient species to show a distance effect, with higher abundances at reefs nearest the subtidal channel. Certain infaunal taxa were more abundant closer to channel, and total infaunal abundance was lower around reefs compared with sandflat areas in August 2005. Results indicate continuous use of reefs during tidal submergence by transients and few distance effects on associated fauna. Although patch location may not be an important

consideration for restoration and management of intertidal oyster reefs over the scale examined, the presence of reefs provides habitat for associated fauna between subtidal channels and higher intertidal areas.

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INTRODUCTION

The intertidal estuarine environment has a number of landscape characteristics that consist of spatially heterogeneous mosaics formed by habitat patches and corridors set within a background matrix of less suitable habitat (Forman 1995). Habitat loss and degradation, such as fragmentation of structured habitats (Saunders et al. 1991), have led to increased efforts to understand patch use across landscapes. Habitat selection, utilization, and community development vary with landscape characteristics, particularly with increased landscape complexity (Bell et al. 1991), and can be affected by patch quality, size, the presence of corridors and habitat connectivity. The makeup of the landscape mosaic (Gustafson and Gardner 1996), the presence of structured corridors between habitat patches (Coffman et al. 2001), and an organism's mobility (Kotliar and Wiens 1990) may affect an organism's response to spatial scale and landscape connectivity. The degree of patch usage depends on whether an organism remains at the encountered patch (Wiens 1976) or moves to new patches based on resource availability (Hanski 1994).

Large scale habitat fragmentation affects highly mobile avian species (Stephens et al. 2003) such as brown-headed cowbirds which use grasslands for foraging but lay eggs in nests of other songbirds within forest patches (Donovan et al. 2000). These songbirds, in turn, may have lower reproductive success due to increased cowbird activity (Burgham and Picman 1988), a direct result of increased forest fragmentation allowing cowbirds access to songbird habitat. Smaller fauna organisms respond to landscape changes (habitat patchiness) on smaller scales due to reduced mobility. Wetland loss limits availability of breeding pools for amphibians (DeMaynadier and Hunter 1999).

Amphipods and isopods are more abundant within smaller structured patches than larger ones (Eggleston et al. 1999). Understanding spatial patterns and the ecological effects of landscape changes, such as fragmentation and habitat complexity, are currently a main focus of landscape ecology (McGarigial and Cushman 2000) and restoration (Bell et al. 1997).

Many studies have described the importance of structured habitats in aquatic and estuarine systems on survival and population maintenance of species (Beck et al. 2001), including bluegill (Crowder and Cooper 1982), juvenile blue crabs (Orth and van Montfrans 1990), and various fisheries species (Heck and Thoman 1981) which utilize structure to avoid predators. Fragmentation can affect abundances of transient and resident species using patches of SAV and oyster reefs by increasing the edge to interior ratio (Irlandi 1997, Eggleston et al. 1999, Harwell 2004). Oyster reef morphology (Lenihan and Peterson 1998, Grabowski 2004, Harwell 2004) and/or proximity to other structured habitats (Irlandi and Crawford 1997, Rhoads 1998) can increase abundances and alter community interactions by providing refuge and foraging areas. The presence of structured corridors may reduce prey abundances and species richness within more complex habitats (oyster reefs and SAV) by providing a migration area for blue crab predators (Micheli and Peterson 1998) or corridors may facilitate migration of less vagile species in less complex habitats (oyster reefs, Brietburg et al. 2000, SAV, Darcy and Eggleston 2005).

One poorly studied aspect of estuarine systems is how organisms utilize habitat patches across shallow water flats within the intertidal zone. Most transient organisms, which do not permanently inhabit intertidal areas, must remain fully submerged and

migrate with the tides. These mobile species, particularly finfish and certain crustaceans, remain in subtidal areas or shallow pools (Kneib 1987) during low tide and utilize intertidal habitat during submergence. In southeastern North Carolina, structured habitat in the subtidal zone is minimal (Alphin et al. 1997, Posey et al. 1999); however, biogenic structure in the intertidal zone occurs in the form of reefs formed by the eastern oyster, *Crassostrea virginica*. In addition to providing many important ecosystem functions such as enhanced nutrient nutrient, particulate removal and increased water quality through high filtration and enhanced sedimentation (Newell 1988, Dame et al. 1984, Cressman et al. 2003, Nelson et al. 2004), intertidal oyster reefs provide critical habitat for many estuarine organisms.

The habitat role oyster reefs perform has become a major focus of current research, with particular concern for reef management and restoration efforts (Coen et al. 1999, Breitburg et al. 2000). Greater species abundances and diversity have been observed over oyster reefs compared to adjacent sand flats and intertidal or subtidal mud bottoms (Coen et al. 1999, Posey et al. 1999, Zimmerman et al. 1989). Oyster shell provides habitat for the recruitment of new oyster spat (Ortega and Sutherland 1992) and the 3-dimensional structure of reefs provides habitat for many species of finfish, crustaceans and molluscs (Coen et al. 1999, Posey et al. 1999). Transient species utilizing oyster reefs include top predators such as striped bass (*Morone saxatilis*) and bluefish (*Pomatomus saltatrix*), blue crabs (*Callinectes sapidus*), flatfishes, penaeid shrimp (Coen et al. 1999, Lehnert and Allen 2002, Harding and Mann 2001), and resident species including oysters (*C. virginica*), ribbed mussels (*Geukensia demissa*) and xanthid crabs (Meyer and Townsend 2000). Juvenile pinfish (*Lagodon rhomboides*), sheephead

minnows (*Cyprinodon variegatus*), and mummichogs (*Fundulus heteroclitus*) are common prey species for larger finfish associated with intertidal oyster reefs during submergence (Harwell 2004). Grass shrimp (*Palaemonetes pugio*), which are prey for many fishes, seek refuge in submerged structured habitat such as oyster reef and marsh grass (Clark et al. 2003). The presence of oyster reefs may be critical for increasing fisheries production (Peterson et al. 2003) of certain finfish and crustaceans partly due to greater abundances of prey associated with oyster reefs (Bahr and Lanier 1981) compared to unstructured areas.

One landscape aspect potentially affecting reef habitat function that has not been addressed is reef proximity to subtidal waters and how that proximity affects habitat utilization by transient and resident fauna. Subtidal habitats are a low-tide refuge for small fish and decapods (Knieb 1987), that may migrate from subtidal channels on the incoming tide and forage and/or seek refuge among structure. Accessing that structure, either salt marsh or oyster reef, is important for the foraging, growth and survival of many species (Irlandi and Crawford 1997, Madon et al. 2000). Smaller, isolated reefs located between subtidal channels and upper intertidal marsh may serve as temporary 'stepping stones' for transients migrating toward upper intertidal areas (Breitburg et al. 2000), where more structured habitat (fringing reefs, marsh grass) is present. However, spacing among patches may not be as important for highly mobile organisms that can quickly access all submerged intertidal habitat (Morrison et al. 2001). *F. heteroclitus* and juvenile *C. sapidus* prefer shallow water habitats (Ruiz et al. 1993, Dittel et al. 1995), possibly leading to higher abundances of these organisms in patches away from subtidal channels as they move with the tidal front towards higher intertidal areas. However,

these organisms may encounter and remain near intertidal structure close to the subtidal channels and not risk movement across open, unstructured flats, where predation risk may be higher (Kneib 1987). Because many transient organisms can move long distances and utilize any submerged habitat (Morrison et al. 2001), patch usage may not be affected by the migration abilities, but rather by patch position within the landscape, particularly patch proximity to the subtidal channels.

In contrast to transients, the sources for reef resident fauna may be larger oyster reefs in the mid intertidal zone. Mobile fauna may migrate from these larger patches in a manner predicted by island biogeography theory (MacArthur and Wilson 1967), where patches that are larger and closer to sources are colonized first, or metapopulation theory (Hanski and Gyllenberg 1993), where populations are sustained with movement among patches. Highest densities of mobile reef residents would be predicted to occur at reefs closer to the source (mid intertidal structures). Less vagile organisms, such as xanthid crabs or oyster toadfish (*Opsanus tau*) are found on artificial reefs within 3 months of creation (Meyer and Townsend 2000), a result of either colonization or through immigration. If migration occurs, then these organisms may be limited in movement to reefs closer to their sources in the mid intertidal zone. Many reef residents, such as *C. virginica* and *Balanus* spp., colonize through larval recruitment from a larger landscape scale (Bahr and Lanier 1981, Zimmerman et al. 1989, Ortega and Sutherland 1992); therefore, overall abundances within a small scale landscape may be controlled by post-settlement ecological interactions such as competition and predation (Abbe and Breitburg 1992, Grabowski 2004) or physical parameters of the environment, and not proximity to source areas.

In addition to the effects of distance from source on associated reef communities, there may be differences in utilization of microhabitats within reef complexes. Edge areas contain higher abundances of transients within salt marshes in the intertidal zone (Weinstein 1979) compared to interior areas. However, subtidally, there is no difference in utilization of edge over interior areas within SAV beds (Bell et al. 2001) although fragmentation and patch size variation may affect faunal communities (Hovel and Fonseca 2005). Within oyster reefs, fragmentation alters faunal assemblages (Harwell 2004), however little is known about differences in utilization between edge and interior areas.

Spatial coverage of natural oyster reefs has declined due to overharvesting, destructive fisheries methods, disease, and poor water quality, leading to increased effort in understanding the importance of oyster reefs and how to restore reefs for ecosystem functions (Coen et al. 1999, Coen and Luckenbach 2000). Landscape characteristics may influence restoration success (Bell et al. 1997), an important aspect regarding oyster management and ecosystem function. For the purposes of management and restoration of oyster reef habitat, it is important to determine which landscape characteristics influence patch use by transient and resident species and have the greatest influence on recruitment of larval oysters. This study was designed to examine the influence of reef patch proximity to a subtidal channel on the macrofaunal utilization of intertidal oyster reefs and to determine whether abundances vary between edge and interior of reefs.

METHODS

In order to assess how the distance from subtidal channels affected faunal usage of intertidal oyster reefs, artificial oyster reefs were constructed as uniform mounds using disarticulated shell on an intertidal sandflat at set distances from a subtidal channel. Sampling natural oyster reefs was not practical for this study because of non-uniform reef patch size, shape, and uncontrolled distance from subtidal channels and other structured habitats. There are drawbacks to using artificial reefs, such as development time and limited vertical complexity compared to natural reefs; however, this approach allowed for controlled placement of oyster reef patches at varying distances from a subtidal channel. Previous studies demonstrated that created reefs can provide habitat sufficient to support both resident and transient fauna (Harwell 2004) and can be colonized within three months by resident species and quickly settled by oyster spat (Meyer and Townsend 2000). Breder trap and surround net sampling were used to assess nekton abundances, shell excavations were used to measure resident epifauna, and sediment cores were taken to sample infauna adjacent to reefs. Tethering and outplant studies were conducted to test relative predation rates.

Reefs were established on an intertidal sand flat adjacent to the mouth of Hewlett's Creek, along the Atlantic Intracoastal Waterway east of Wilmington, NC (Figure 1). The study area contained natural oyster reefs and marsh vegetation in the mid and high intertidal zones. Reef construction followed that of previous studies (Harwell 2004). Twelve artificial oyster reefs were constructed in February, 2005, using clean oyster cultch. Each reef was made with 0.87-0.96 cubic meters of shell and built of a circular, uniform shape, 2.2 m in diameter and 0.2-0.25 meters in height above the sand.

2.2 m diameter reefs exceed a minimum patch size of 1.8 m diameter determined to supply sufficient habitat to support associated nekton (Harwell, 2004).

Reefs were placed into 4 transects, with each transect including reefs situated at near (5m), intermediate (30m) and far (55m) distances from the subtidal channel (Figure 2). There was an maximum 20 minute lag in submergence between each consecutive distance with a total maximum lag of 40min between 55m and 5m. Transects 1 and 2 were positioned on the eastern side of a north-south subtidal channel which runs through the study site and transects 3 and 4 were positioned on the western side of the channel. These distances are typical of intertidal sandflat areas in southeastern North Carolina and reef placement maximized distance from the channel within the study site while providing at least a 20m distance from any other structured habitat (artificial and created reefs within the study area). The distance between the subtidal channel and mid intertidal oyster reefs and marsh was greater than 75m. The sediment characteristics, salinity, tidal range and topography were similar for each transect. Three months were allowed for subsidence of the oyster shell and establishment of the initial reef communities (Meyers and Townsend 2000, Harwell 2004) before faunal sampling commenced in May, 2005.

Transient Fauna

Breder traps (Breder 1960), clear acrylic boxes measuring 31cm x 16cm x 15cm with acrylic wings creating an opening of 30cm x 15cm, were used to measure abundances of transient fauna. The traps, designed to passively sample small, epibenthic fish and decapod crustaceans, have been used effectively on oyster reefs (Harwell 2004)

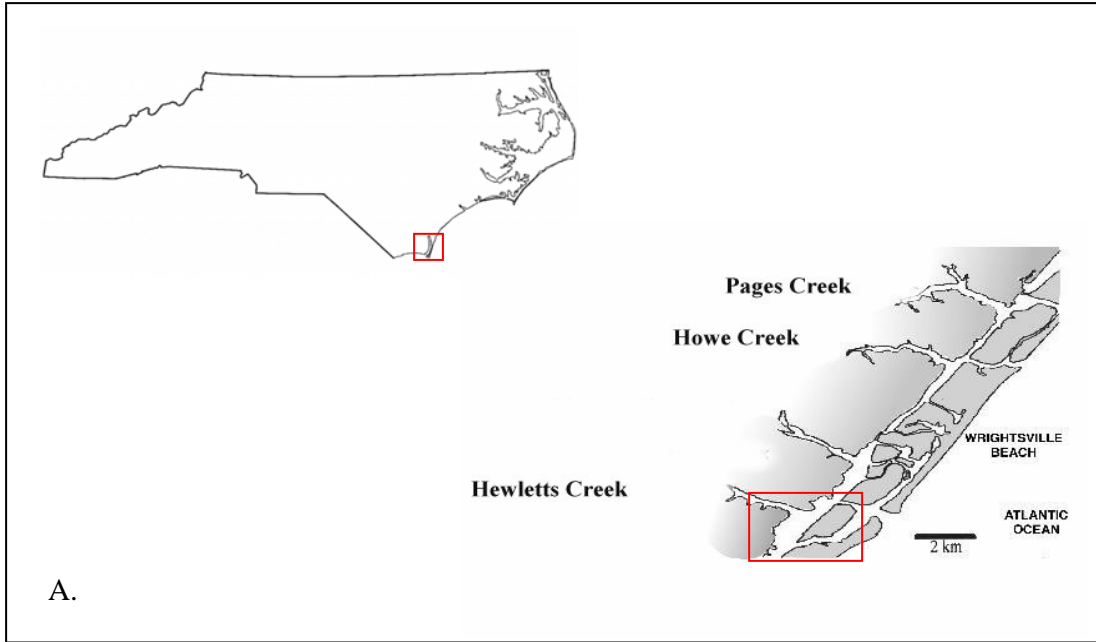


Figure 1. Study site in southeastern North Carolina ($34^{\circ}17'N$, $77^{\circ}48'W$) (A). Overhead view of study area (B). Arrow points to subtidal channel. Bars represent approximate placement of reef transects (1-4).

and within marsh grass (Rhoads 1998, Fell et al. 2003). Summer sampling was conducted during 4 time periods: May 27-31, June 20-21, June 27-28, and July 19-20, 2005. Winter sampling was conducted once on February 8-10, 2006. Traps were either secured to the sediment using tent stakes or anchored on reefs using steel rebar. For each distance (5m, 30, and 55m), 3 traps were placed on top of the reef (one facing the subtidal channel and the other two parallel to the channel in either direction), one trap placed along the sand edge of the reef opening towards the flooding tide (Figure 3), and 4 traps placed on the open sandflat. These treatment locations, referred to as the reef, sand edge, and sandflat treatments, respectively, were designed to determine if there are differences in microhabitat use along the reef edge. Reef traps were positioned at a lower depth than sand edge and sandflat due to vertical height of the artificial reefs (0.15m at trap opening). Beginning with the third time period (June 27-28, 2005), and with all subsequent sampling, a second Breder trap was deployed over the sand edge treatment to more accurately measure nekton abundances. The placement of traps was designed to capture transients moving along the reef edge or moving over the shell or sandflat. Paired reef transects and corresponding sandflat treatments were sampled at the same time in order to sample fish moving with the tidal front. Each pair of transects was sampled 4 times during summer 2005, twice during lunar spring tides (June 20-21 and July 19-20, 2005) and twice during lunar neap tides (May 27-31 and June 27-28, 2005) to cover the range of tidal heights and flow velocity within the study site. Organisms retained in traps were identified, measured for total length, recorded and released. On a sampling day, Breder traps were placed at each reef and sandflat location before the

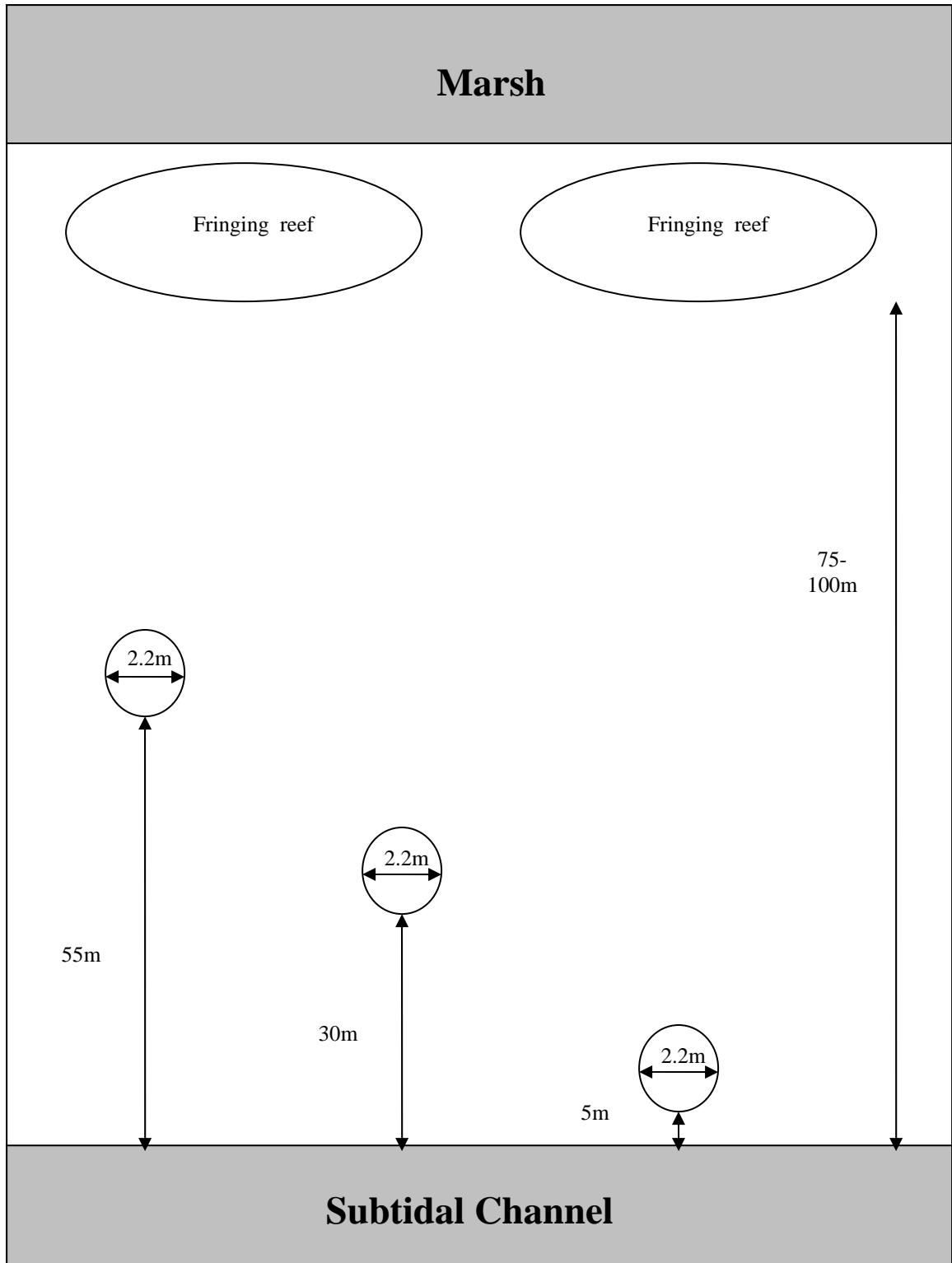


Figure 2. Location of created reef patches between subtidal channel and upper structural habitats. Drawn to scale.

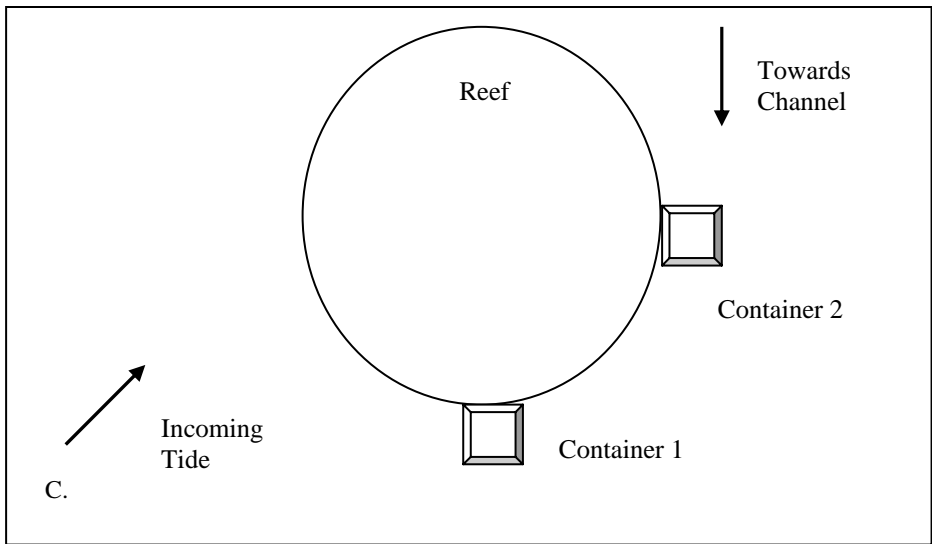
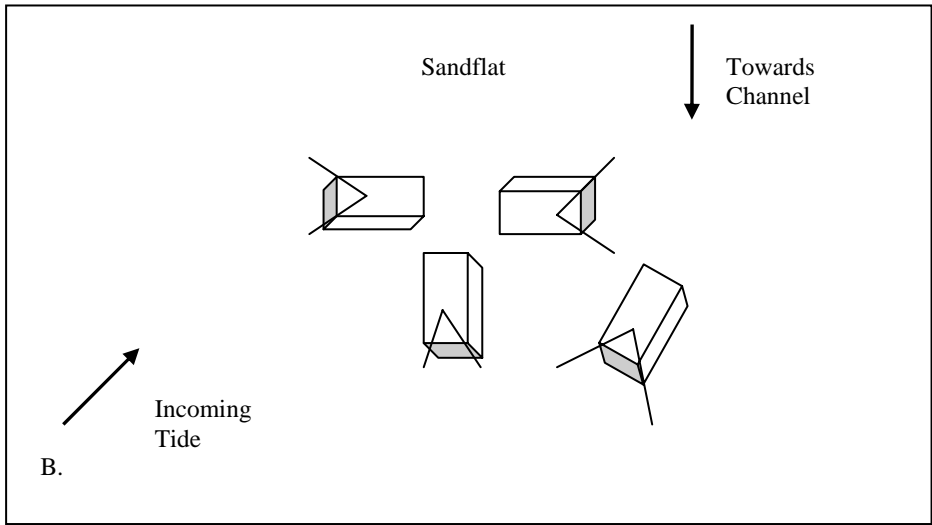
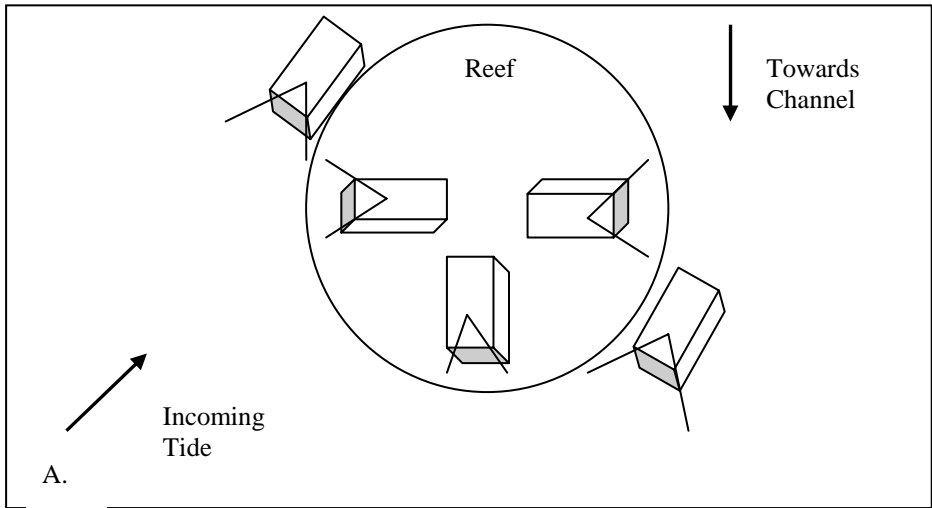


Figure 3. Breder trap configuration for reef (A) and sandflat (B) treatments. Traps are oriented to face incoming tide. Clam outplants configuration for reef treatments (C).

incoming flood tide and allowed to fish for 2 hours following submergence. As stated earlier, there was a lag between submergence for treatments at different distances. Traps were collected in order of submergence, keeping soak times constant across areas (5m first, 55m last). Preliminary data showed that sampling on the flood tide (13.1 organisms/trap) yielded higher catches than sampling during the ebb tide (3.6 organisms/trap). As a complement to Breder trap sampling, surround net sampling was conducted in August 2005 on the western transects (3 and 4). This method gave an instantaneous count of all mobile fauna around, on, or above the reef, targeting not only fauna that stay near the substrate, but larger mobile finfish and crustaceans not easily captured using Breder traps. Average water depth over treatments was 1m during sampling. A 15m seine net was unrolled around poles enclosing a circumference 3.2 meters around the reef including the entire reef and an additional 0.5m buffer (both reef and sand edge). The ends of the seine net were brought together and all organisms within were identified, measured for total length, enumerated and released. Each transect was sampled simultaneously with a surround net deployed at the three distances from the subtidal channel. Sandflat treatments were sampled in a similar manner.

Resident Fauna

Shell excavations were conducted in August 2005, following transient sampling, and in January 2006 after transients had left the system (Hoss 1974, Hales and Van den Avyle 1989). Plots sampled in August were marked to avoid re-sampling in January. During low tide, 20cm x 20cm quadrats (Harwell 2004) were placed on the reef at randomly selected positions, with the lower edge of the quadrat placed inside the reef

5cm from the reef edge. All shell within the quadrat was removed down to the shell-sediment interface and sifted using a 2mm mesh sieve and examined for organisms. Each reef and corresponding sandflat treatment was sampled with 3 quadrats. Organisms were identified to species, enumerated and released. Oyster spat was measured for total length and the total carapace width of xanthid crabs was measured. All shell was returned to the reef.

Predation

In situ tethering was conducted using the grass shrimp, *Palaemonetes pugio*, a common prey of shallow water fishes (Kneib 1987, Posey and Hines 1991, Clark et al. 2003). *P. pugio* were individually tethered (Everett and Ruiz 1993) using #7-0 surgical sutures (Ethicon, Inc.). The tail of the shrimp was pierced with a needle attached to monofilament and the end of the monofilament was knotted securely. The other end of the 30cm monofilament line was affixed to an individual lead weight. Experiments were conducted on Oct. 3 and 14, 2005, on transects 3 and 4. Ten shrimp were deployed at each of the 3 distances over two treatments: the sand edge 10cm from the reef and open sandflat at corresponding reef distances. After 14 minutes (initial trials indicated 100% predation in 30 min.), remaining shrimp were collected and the percent consumed was recorded.

Predation on a bivalve prey, the hard clam *Mercenaria mercenaria*, was assessed on November 3, 11, and 18, 2005. Hard clams were chosen because they are a common prey of blue crabs, *Callinectes sapidus* (Arnold 1984) and xanthid crabs (Whetstone and Eversole 1981). Preliminary laboratory trials were conducted to confirm blue crab

predation on hard clams (crabs consumed 100% of hard clams during trials leaving broken shell fragments). Twenty-five clams (7.7 mm avg. length) were placed inside 15cm x 15cm x 5cm deep plastic containers with defaunated sand in a tank of seawater and allowed to burrow overnight. Two containers were buried in the sediment 10cm from the reef edge and at open sandflat treatments along all four transects (Figure 3). A total of forty-eight containers were deployed during each of three experiments. Trays were retrieved after 24 hours for the November 3 deployment. Due to low predation during the November 3 deployment (79% clams remaining), 48 hour deployments were conducted on November 11 and 18. Trays were sieved through a 1mm screen and remaining clams were counted and the percent consumed was recorded.

Infauna

Invertebrate infauna was sampled to indirectly measure finfish and crustacean utilization of sediments adjacent to reefs and in unstructured areas. Infauna was sampled in August 2005 and January 2006 using sediment cores (15cm deep, 10cm diameter). Transects 3 and 4 (transects also sampled with surround nets) were each sampled during both sampling periods. Three cores were taken at each of reef edge, 0.5 m from reef edge, and corresponding unstructured sandflat. Samples were fixed in a 10% buffered formalin solution with rose bengal, then sieved through a 500 μ m mesh and stored in 70% isopropyl alcohol. Two of the triplicate samples were sorted using a dissecting microscope; the third triplicate was archived. Organisms were identified to major taxonomic group, at phylum (Hemichordata, Nematoda, Nemertea, Sipuncula), class (Mollusca, copepoda, oligochaeta, holothuroidea,), order (gammaridae), or family

(polychaeta) levels of distinction. Each sample, including all organisms and fragments that could not be identified, was analyzed for total biomass. Samples were placed in a 70° C oven for 48hours (to allow for water evaporation) and weighed (dry weight). Samples were then placed in a 500° C ashing furnace for 5h and weighed (ash weight). Total biomass was calculated as the difference between ash weigh and dry weight.

Statistical Analysis

Analysis of Variance (ANOVA) was used to compare the abundances and sizes of the dominant species by distance from subtidal channel (5m, 30, 55m) and treatment (sand edge, reef, and sandflat for Breder traps; reef and sandflat for surround nets, infauna, and predation experiments). Abundance data from Breder trap, surround net, excavation and infaunal sampling were log transformed to meet the assumptions of homogeneity of variance (F-max test, Sokal and Rohlf 1981). For Breder traps, transects sampled within each time period were treated as replicates. Mean abundance and mean fish length per Breder trap and surround net sample were used, due to non-independence of organisms within those samples. Preliminary analysis of summer 2005 Breder trap data showed no significant interaction between spring and neap tides with either distance or treatment. There was no difference in abundances between consecutive samplings (1 neap, 1 spring) in June 2005. There were significant interactions between time period and distance or treatment, regardless of tide type; therefore Breder trap data were analyzed by time period (May, June 1, June 2, and July) to assess temporal variations in distance or microhabitat use. A 3-way ANOVA was used for each dominant species using SAS PROC GLM procedure (SAS/STAT User's Guide 2003) with distance from

subtidal channel, treatment, and time period as main effects, in addition to interactive effects between distance and treatment. The May 27, 2005, Breder trap data were excluded in the analysis of *F. heteroclitus* because traps were placed after reef submergence (Ruiz et al. 1993). May 27, 2005 was the only sampling day in which no *F. heteroclitus* were caught. The Student-Newman-Keuls (SNK) test was employed to conduct pair-wise comparisons among distances and treatments when ANOVA indicated significant differences. Winter 2006 Breder trap data were analyzed separately using 2-way ANOVA with distance and treatment as main effects. For surround net data, 2-way ANOVA was run with distance from subtidal channel and treatment as main effects, in addition to interactive effects between distance and treatment.

For excavation data, a 2-way ANOVA was used for each dominant species with distance from subtidal channel and season as main effects, in addition to interactive effects between distance and season. For the clam outplant predation experiments, the proportion of clams remaining (arc sine square root transformed) was analyzed with 2-way ANOVA with distance from subtidal channel and treatment (sand edge and sandflat) as main effects, in addition to interactive effects between distance and treatment. For infaunal abundances and biomass, 2-way ANOVA was run for each dominant group with distance from subtidal channel and treatment (reef and sandflat) as main effects, in addition to interactive effects between distance and treatment. The SNK test was employed to conduct pair-wise comparisons among distances and treatments when ANOVA indicated significant results.

RESULTS

Distance from Channel

Transient Fauna

Patterns for distance effects were mixed depending on sampling method and species. Species caught with Breder traps are listed in Table 1. For summer 2005, overall dominant species (those comprising >1% of totals) were *Fundulus heteroclitus* (mummichog), *Lagodon rhomboides* (pinfish), and *Leiostomus xanthurus* (spot). *L. rhomboides* were the most common, comprising 84.63% of total catch. Mean total abundances of all species in traps (Figure 4) did not vary across distances (Table 2) nor was there an interaction between distance and treatment or time period. For the two most abundant species sampled in 2005, *L. rhomboides* and *L. xanthurus*, there were no significant distance effects (Table 2) or interactions between distance and either treatment or time period. Total length (Figure 5) for both *L. rhomboides* and *L. xanthurus* did not vary significantly (Table 3) among distances. *F. heteroclitus* was the only common species (Figure 6) to demonstrate significant distance effects ($F=4.38$, $p=0.015$) with greater abundances at a distance of 5m from the subtidal channel compared with distances of 30m and 55m. Total length for *F. heteroclitus* did not vary significantly among distances. Abundances of other finfish combined (those comprising >1% of totals; Figure 6) and decapods did not differ across distances.

Breder trap catches in February 2006 (Figure 7) were dominated by larval *L. xanthurus* and larval *L. rhomboides*, comprising 83.97% and 14.57% respectively. Mean total abundances (Table 4), including all other finfish and decapod crustaceans, did not

Table 1. Total abundance of finfishes and decapod crustaceans sampled with Breder traps. Species listed by distance and season. S=Summer 2005, W=Winter 2006.

Species	5m		30m		55m	
	S	W	S	W	S	W
Finfish						
<i>Lagodon rhomboides</i>						
Juvenile	826		796		1064	
Larval		101		15		44
<i>Leiostomus xanthurus</i>						
Juvenile	86		56		72	
Larval		317	1	245		360
<i>Fundulus heteroclitus</i>	100		45		36	2
<i>Paralichthyes lethostigma</i>	4	1	2	1	1	
<i>Fundulus majalis</i>			2		4	2
Gobiidae spp.	1				2	1
<i>Mugil cephalus</i>	1	1				3
<i>Cyprinodon variegates</i>		1		2		2
<i>Stephanolepis hispidus</i>		2				
<i>Eucinostomus lefroyi</i>	1					
<i>Menidia menidia</i>				1		
Leptocephalus larvae	1					
Decapod Crustaceans						
<i>Farfantopenaus aztecus</i>	13		11		9	
<i>Callinectes sapidus</i>	11		9		9	
Paguroidea spp.	1				3	
<i>Paleomonetes pugio</i>	3	1				

Figure 4. Mean abundances by distance from subtidal channel and treatment for total catch, *Lagodon rhomboides*, and *Leiostomus xanthurus* caught in Breder traps during summer 2005. Bars indicate mean organisms (+SE) per trap for each treatment at each distance. Abundances in sand edge are significantly higher than other treatments for each group.

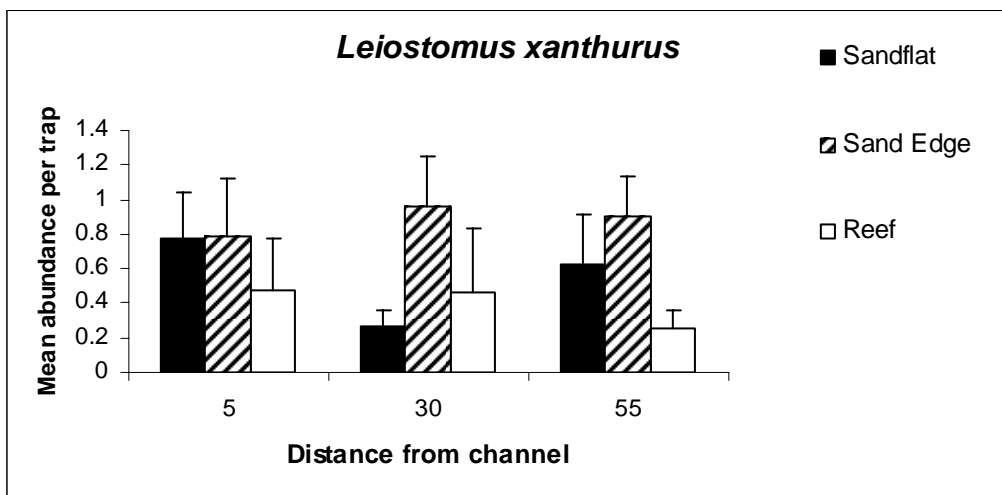
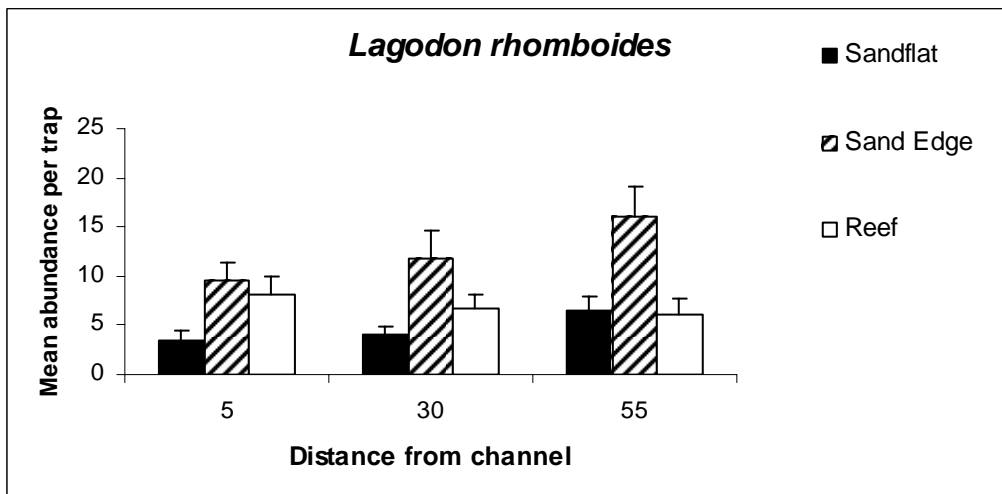
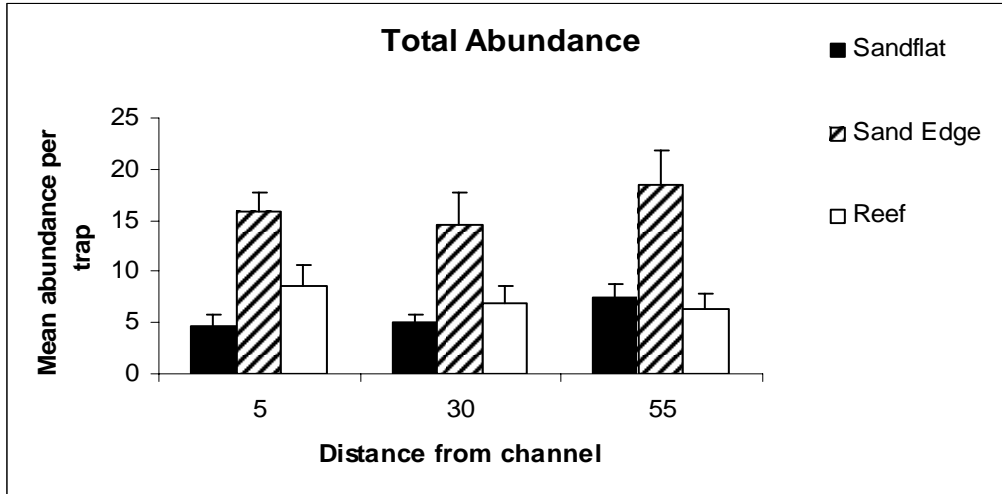


Table 2. 3-way ANOVA results for the effects of distance, treatment (on reef, sand edge, sand), time period, and 2-way interactions, on mean abundances of common species caught in Breder traps. Shown are F-values and (p-values) with significant differences indicated by an asterisk. When significant effects are present, SNK rankings are shown in decreasing order of abundance, with treatments differing indicated with different superscript letters. 5=5m, 30=30m, 55=55m, E=sand edge, R=reef, S=sandflat, May=May 27-31, Jun1=June 20-21, Jun2=June 27-28, Jul=July 19-20.

Species	Distance	Treatment	Distance*Treatment	Time Period	Distance*Time Per.	Treatment*Time Per.
<i>Lagodon rhomboides</i>	0.78 (0.4597)	10.92* (<0.0001)* E ^a , R ^b , S ^b	0.77 (0.5477)	8.72* (<0.0001)* May ^a , Jun2 ^{ab} , Jun1 ^b , Jul ^c	0.44 (0.8474)	4.38* (0.0005)*
<i>Leiostomus xanthurus</i>	0.17 (0.8473)	6.79* (0.0014)* E ^a , S ^b , R ^b	0.62 (0.6494)	10.37 (<0.0001)* May ^a , Jun2 ^b , Jun1 ^b , Jul ^b	0.81 (0.5645)	0.92 (0.4838)
<i>Fundulus heteroclitus</i>	4.38* (0.0153)* 5 ^a , 30 ^b , 55 ^b	39.84* (<0.0001)* E ^a , S ^b , R ^b	10.38* (<0.0001)*	3.90* (0.0115)* May ^a , Jul ^{ab} , Jun1 ^{ab} , Jun2 ^a	3.34* (0.0052)*	3.82* (0.0020)*
Total other finfish	0.46 (0.6319)	3.28 (0.0415)* E ^a , S ^a , R ^b	0.07 (0.9919)	1.41 (0.2429)	0.65 (0.6910)	0.83 (0.5510)
Total decapod	0.39 (0.6799)	8.68 (0.0003)* E ^a , S ^a , R ^b	1.15 (0.3380)	4.35 (0.0062)* Jun1 ^a , Jun2 ^b , Jul ^b , May ^b	0.54 (0.7781)	1.20 (0.3098)
Mean total abundance	0.73 (0.4846)	25.98* (<0.0001)* E ^a , S ^b , R ^b	1.05 (0.3855)	11.06* (<0.0001)* May ^a , Jun1 ^b , Jun2 ^b , Jul ^c	1.78 (0.1103)	6.49* (<0.0001)*

Figure 5. Mean total length (mm) for *Lagodon rhomboides*, *Leiostomus xanthurus*, and *Fundulus heteroclitus* caught in Breder traps during summer 2005. Bars indicate the mean lengths (+SE) for each treatment at each distance. Total length was significantly greater on sandflat and sand edge than reef treatments for *Lagodon rhomboides* and *Leiostomus xanthurus*.

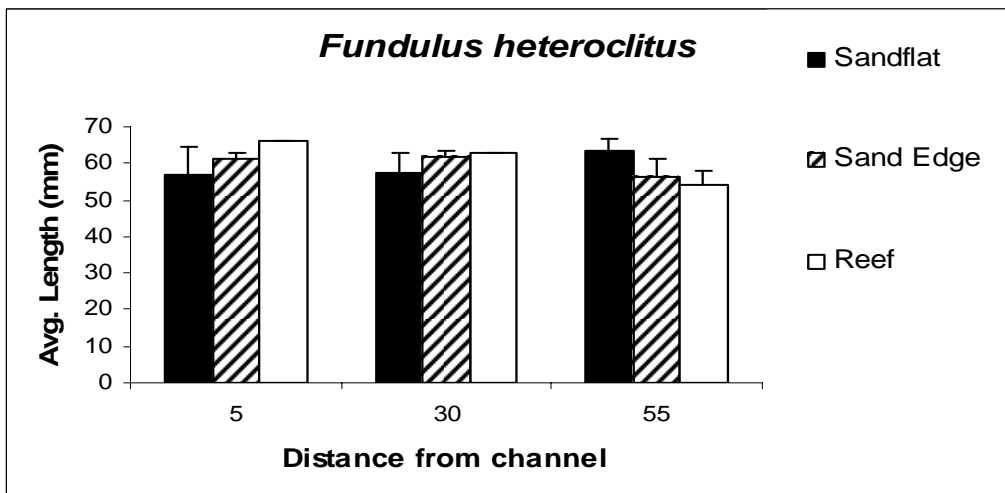
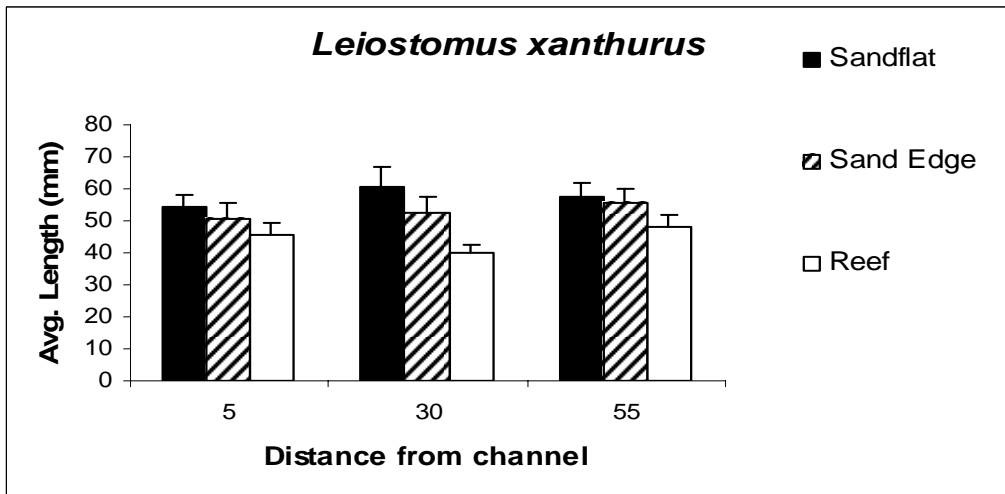
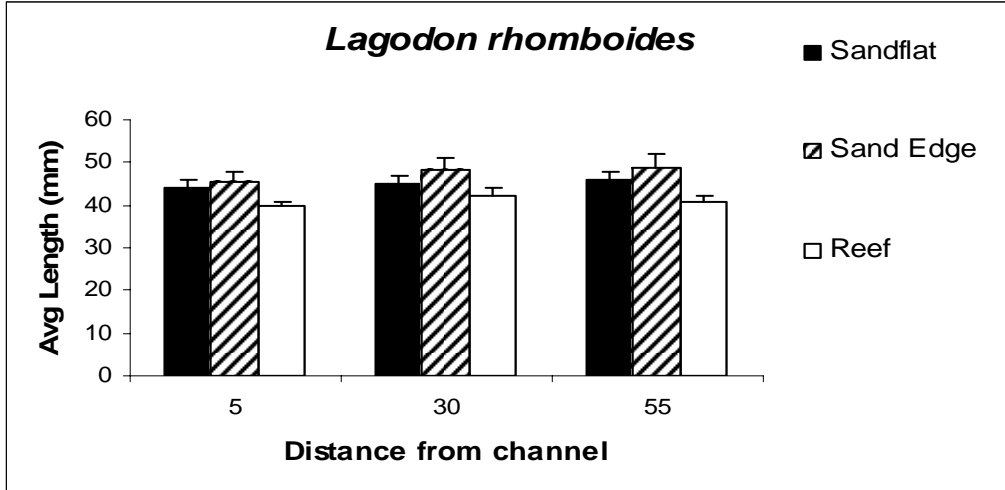


Table 3. 3-way ANOVA results for the effects of distance, treatment, time period and interactions on mean total length of dominant species caught in Breder traps. 3-way and 2-way interactions were not significant and are not shown. Shown are F-values and (p-values) with significant differences indicated by an asterisk. When significant effects are present, SNK rankings are shown in decreasing order, with differences indicated with superscript letters. 5=5m, 30=30m, 55=55m, E= sand edge, R= reef, S=sandflat, May=May 27-31, Jun1=June 20-21, Jun2=June 27-28, Jul=July 19-20.

Species	Distance	Treatment	Time Period
<i>Lagodon rhomboides</i>	3.80* (0.0263)* 55 ^a , 30 ^a , 5 ^a	7.45* (0.0010)* E ^a , S ^a , R ^b	46.99* (<0.0001)* Jul ^a , Jun2 ^b , Jun1 ^b , May ^c
<i>Leiostomus xanthurus</i>	1.32 (0.2791)	8.81* (0.0007)* S ^a , E ^a , R ^b	23.41* (<0.0001)* Jul ^a , Jun2 ^a , Jun1 ^b , May ^c
<i>Fundulus heteroclitus</i>	0.26 (0.7728)	0.07 (0.9351)	2.04 (0.1347)

Figure 6. Mean abundances by distance from subtidal channel and treatment for *Fundulus heteroclitus*, total other finfish, and total decapod crustaceans caught in Breder traps during summer 2005. Bars indicate mean (+SE) organisms per trap for each treatment at each distance. Abundances in sand edge and sandflat traps are significantly higher than reef treatments for decapods and other finfish and higher in sand edge than other treatments for *F. heteroclitus*. *F. heteroclitus* were significantly more abundant at 5m than 30m or 55m.

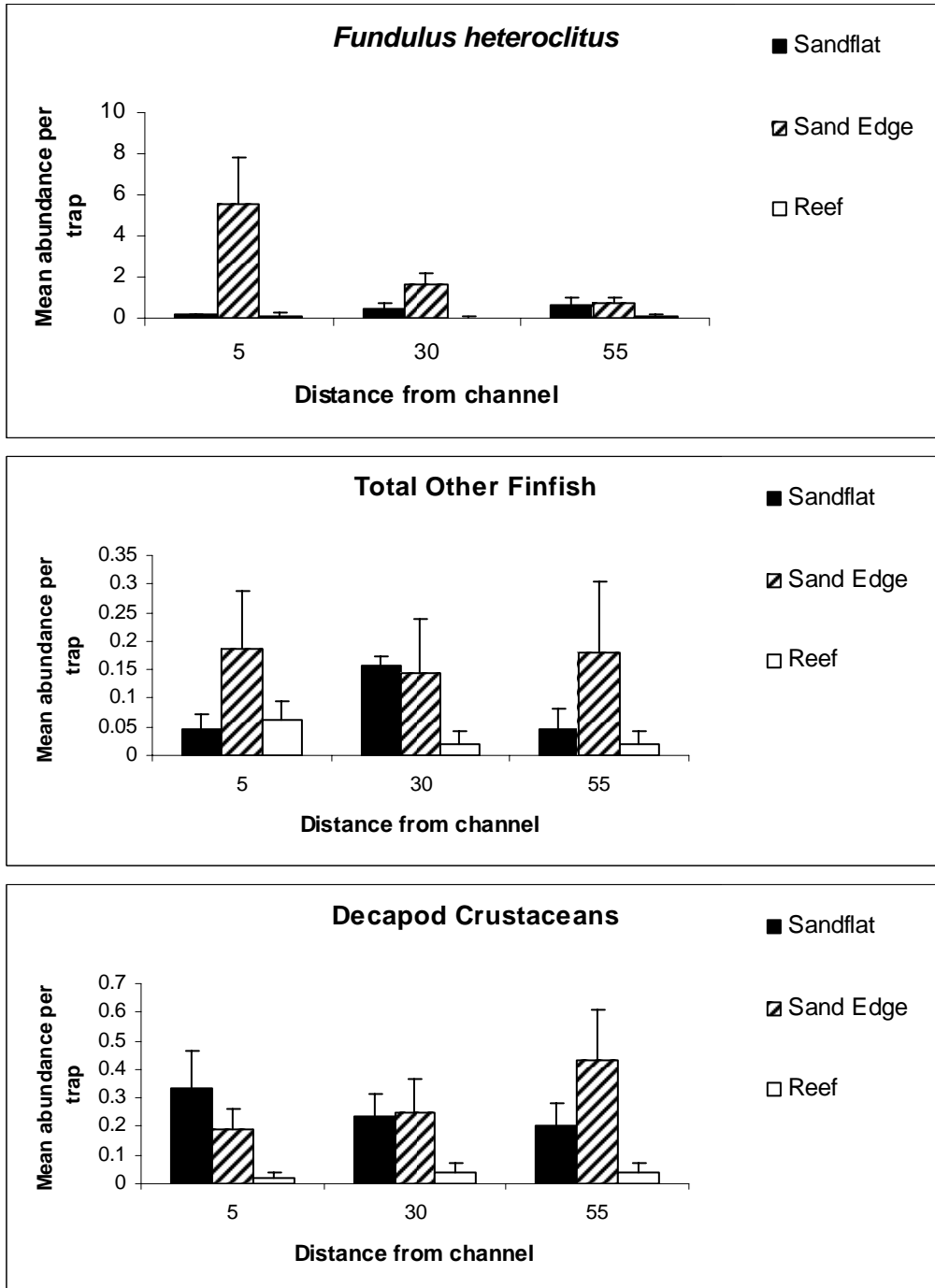


Figure 7. Mean abundances by distance from subtidal channel and treatment for larval *Leiostomus xanthurus* and larval *Lagodon rhomboides* caught in Breder trap during winter 2006. Bars indicate mean abundance (+SE) per trap for each treatment at each distance. Abundances in sand edge and sandflat are significantly higher than reef treatments for *Leiostomus xanthurus*, and significantly higher in sand edge than reef for *Lagodon rhomboides*. Abundances are significantly higher at 5m than 30m for *L. rhomboides*.

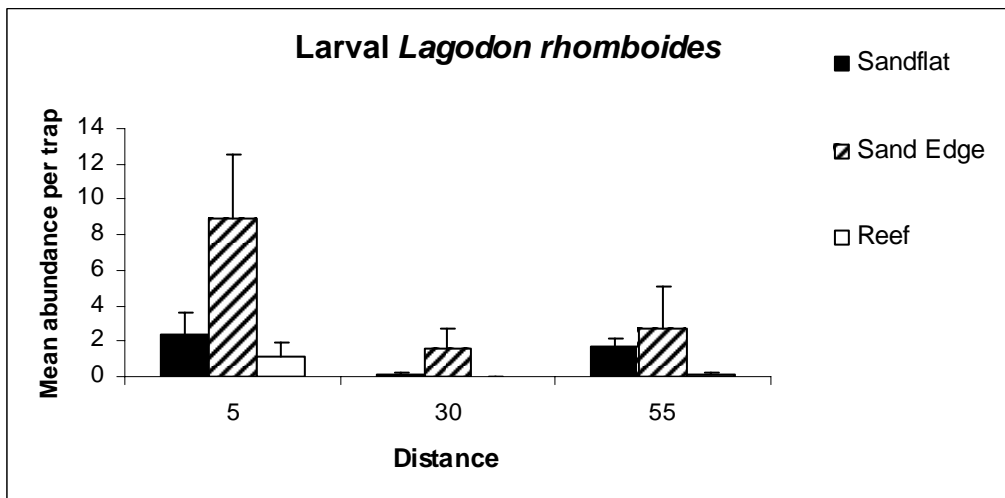
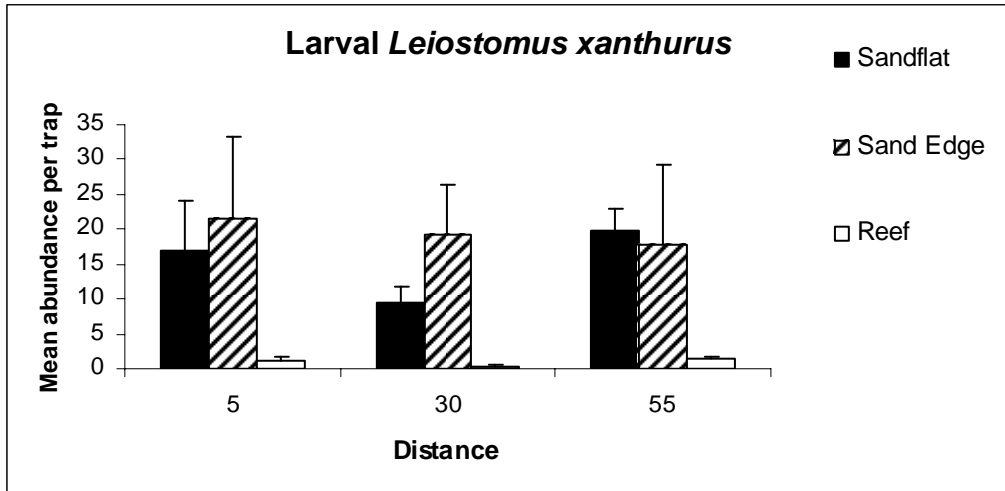


Table 4. 2-Way ANOVA for the effects of distance, treatment and interactions on larval pinfish and spot sampled with Breder traps in February 2006. When significant effects are present, SNK rankings are shown in decreasing order, with differences indicated with superscript letters. 5=5m, 30=30m, 55=55m, E=sand edge, R=reef, S=sandflat.

Species	Distance	Treatment	Distance*Treatment
Larval <i>Leistomus xanthurus</i>	0.30 (0.7424)	29.07 (<0.0001)* E ^a , S ^a , R ^b	0.38 (0.8212)
Larval <i>Lagodon rhomboides</i>	4.18 (0.0276)* 5 ^a , 55 ^{ab} , 30 ^b	5.37 (0.0111)* E ^a , S ^{ab} , R ^b	0.73 (0.5813)
Mean total abundance	0.92 (0.4098)	25.23* (<0.0001)* E ^a , S ^a , R ^b	0.49 (0.7418)

vary among distances nor was there an interaction with treatment. For larval *L. xanthurus*, there were no significant distance or interactive effects. Larval *L. rhomboides* were significantly more abundant at a distance of 5m from the subtidal channel than a distance of 30m ($F=4.18$, $p=0.028$), however, there was no significant interaction between distance and treatment. Surround net catches were dominated (Table 5) by *Anchoa hepsetus* (striped anchovy), juvenile *L. rhomboides*, and juvenile *L. xanthurus*, comprising 66.40%, 29.83%, and 2.10% of the total catch, respectively. There were no significant differences (Table 6) between distances for juvenile *L. rhomboides*, juvenile *L. xanthurus*, and total other reef using organisms (Figure 8). There were no significant differences (Table 7) in mean fish length for juvenile *L. rhomboides* or juvenile *L. xanthurus* (Figure 9) among distances. There was a marginally significant difference in abundances of non-reef finfish (dominated by *A. hepsetus*) with higher catches at a distance of 55m from the channel than 30m and no fish at 5m (Figure 10).

Resident Fauna

Abundances of resident epifauna from excavations were analyzed separately by sampling period: August 2005 or January 2006. Dominant species (comprising >1% of totals) included: *Crassostrea virginica* (eastern oyster), *Panopeus herbstii* (black-fingered mud crab), and *Balanus* spp (barnacles). *C. virginica* accounted for 88.08% of all organisms counted. Excluding oyster spat, *Crepidula* spp. (slipper shell) and *Guekensia demissa* (ribbed mussel) comprised >1% of the remaining organisms, and thus were also analyzed. Spat shells and scars from dead oysters were also recorded, but followed trends similar to live spat (Figure 11) and were not analyzed statistically.

Table 5. Total abundance of finfishes and decapod crustaceans sampled in summer 2005 with surround nets by distance from subtidal channel.

Species	5m	30m	55m
Finfish			
<i>Anchoa hepsetus</i>		771	620
<i>Lagodon rhomboides</i>	221	182	222
<i>Leiostomus xanthurus</i>	17	18	9
<i>Eucinostomus lefroyi</i>	2	3	5
<i>Mugil curema</i>		3	1
<i>Menidia menidia</i>		4	
<i>Selene vomer</i>	2		
<i>Eucinostomus argenteus</i>			1
<i>Paralichthys lethostigma</i>	1		
<i>Synodus foetens</i>			1
Crustaceans			
<i>Callinectes sapidus</i>	3	1	
<i>Farfantopenaus aztecus</i>	3		1
<i>Hippolyte</i> sp.			1

Table 6. 2-way ANOVA for the effects of distance, treatment and interactions on mean abundances of dominant species sampled with surround nets. Shown are F-values and (p-values) with significant differences indicated by an asterisk.

Species	Distance	Treatment	Distance*Treatment
<i>Lagodon rhomboides</i>	0.09 (0.9162)	0.80 (0.3791)	0.53 (0.5912)
<i>Leiostomus xanthurus</i>	1.93 (0.1633)	0.10 (0.7563)	2.55 (0.0945)
Total other reef users	0.04 (0.9578)	1.51 (0.2280)	0.40 (0.6736)
Total other non-reef users	2.87 (0.0723)	1.20 (0.2820)	0.33 (0.7241)

Figure 8. Mean abundances by distance from subtidal channel and treatment for *Lagodon rhomboides* and *Leiostomus xanthurus* caught in surround nets. Bars indicate mean abundance (+SE) per sample for each treatment at each distance. There were no significant differences.

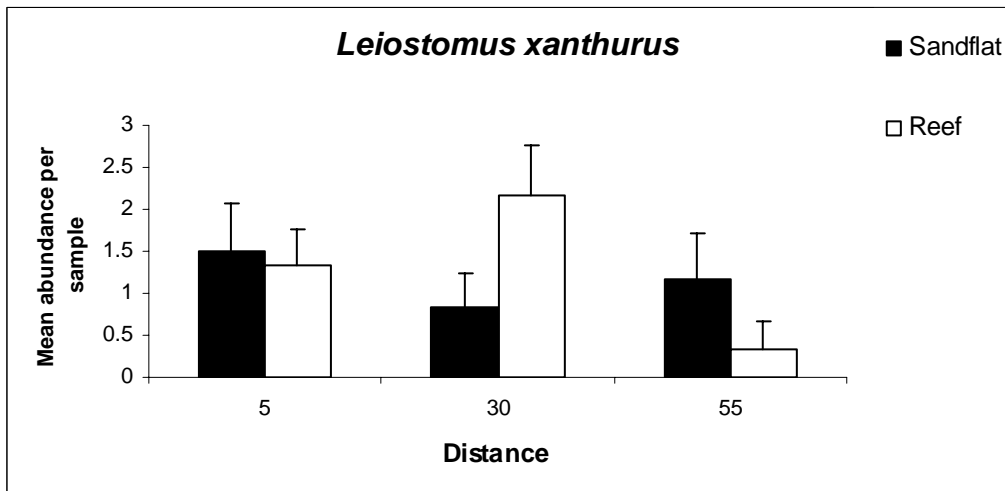
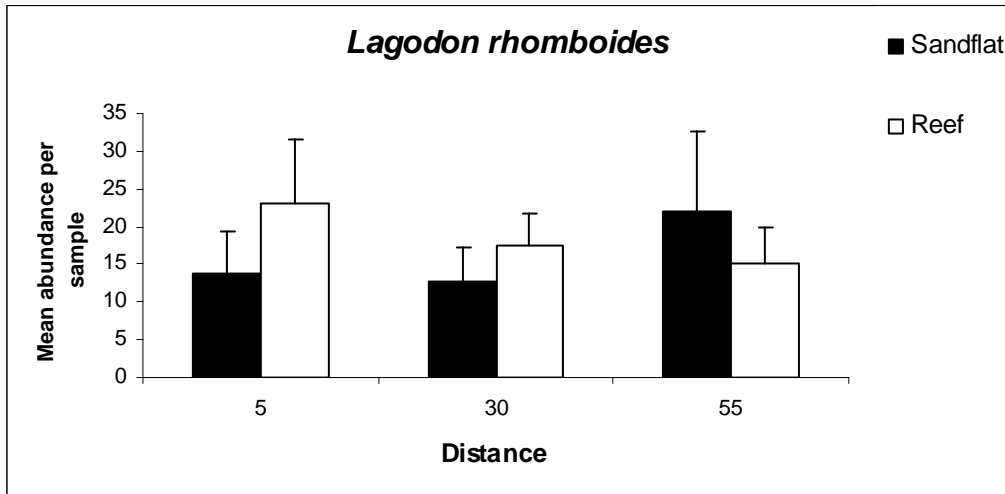


Table 7. 2-way ANOVA for mean total length of *Lagodon rhomboides* and *Leiostomus xanthurus* sampled with surround nets. Shown are F-values and (p-values) with significant differences indicated by an asterisk. When significant effects are present, SNK rankings are shown in decreasing order, with differences indicated with superscript letters. R=Reef treatment, S=sandflat treatment.

Species	Distance	Treatment	Distance*Treatment
<i>Lagodon rhomboides</i>	0.24 (0.7905)	8.12 (0.0080)* S ^a , R ^b	0.55 (0.5846)
<i>Leiostomus xanthurus</i>	0.39 (0.6850)	0.00 (0.9524)	0.51 (0.6080)

Figure 9. Mean total fish length (mm) for *Lagodon rhomboides* and *Leiostomus xanthurus* caught with surround nets. Bars indicate mean lengths (+SE) for each treatment at each distance. Total length was significantly greater for sandflat than reef treatment for *Lagodon rhomboides*.

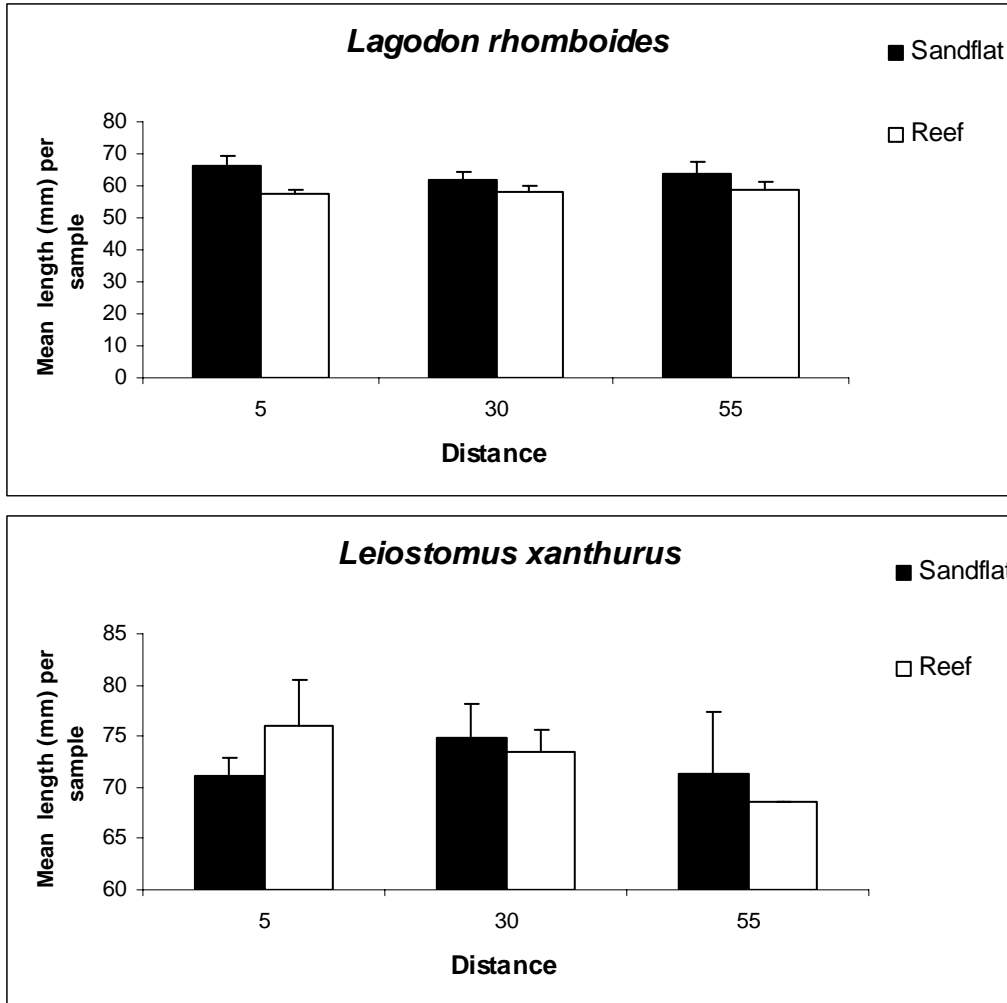


Figure 10. Mean abundances by distance from subtidal channel and treatment for total non-reef using fish and total other reef users caught in surround nets. Bars indicate mean organisms (+SE) per trap for each treatment at each distance. There were no significant differences.

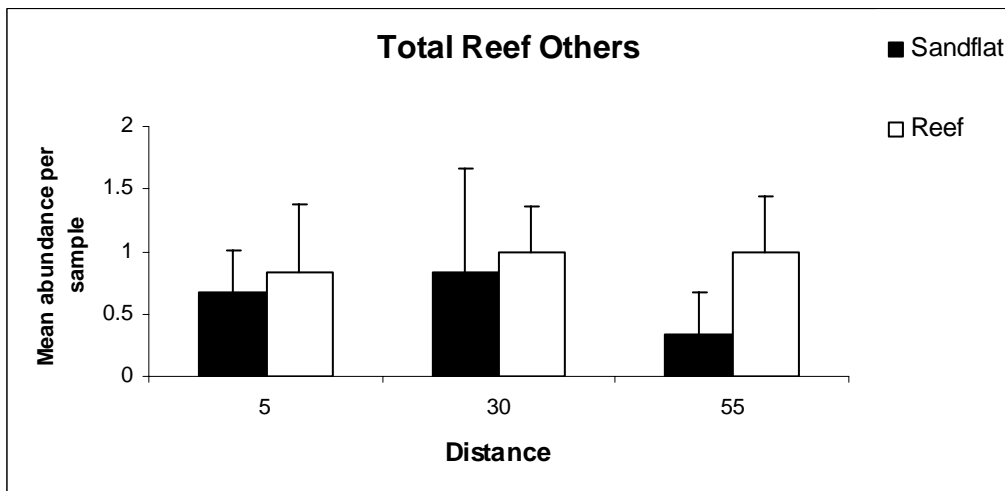
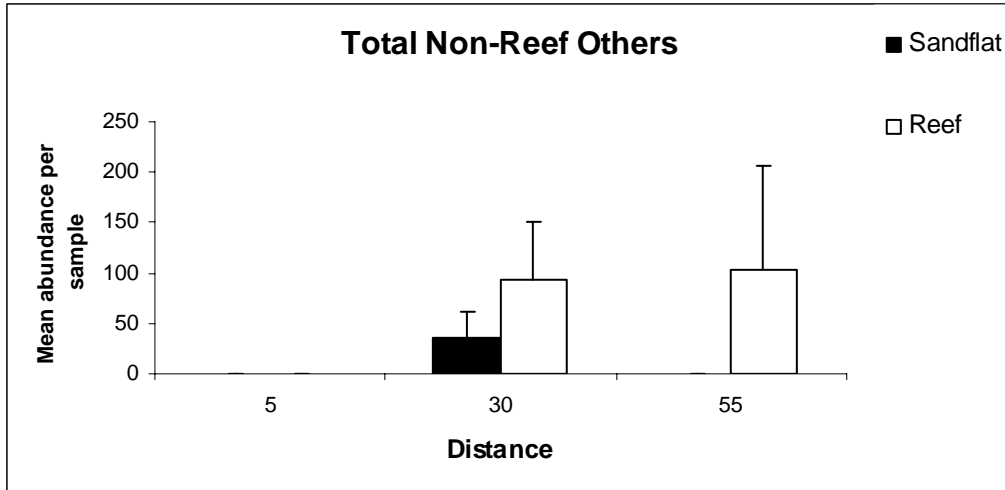
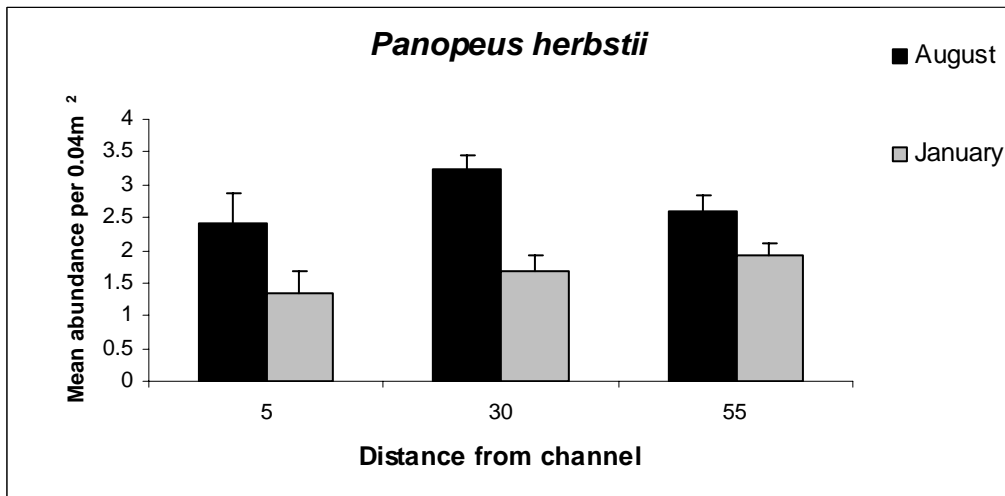
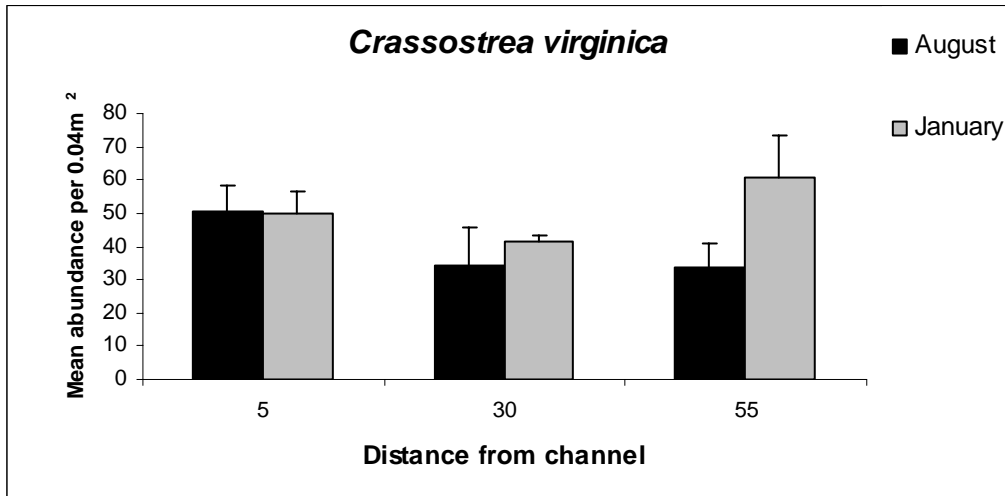


Figure 11. Mean abundances for *Crassostrea virginica* and *Panopeus herbstii* in excavations for August 2005 and January 2006 samplings. Bars indicate mean (+SE) organisms per quadrat per reef for each treatment at each distance. Abundances in August are significantly higher than January for *P. herbstii*.



Other non-dominant organisms sampled representing <1% of fauna collected from excavations include: *Alpheus heterochaelis* (pistol shrimp), Acmaeidae spp. (limpets), *Eurypanopeus depressus* (flat mud crab), Holothuroidea sp. (sea cucumber), *Petrolisthes armatus* (porcelain crab), unidentified clams, and an unidentified shrimp.

Illyanassa obsoleta (mud snail) was the only epifaunal species found in sandflat treatments. For barnacles (Figure 12), there was a marginally non-significant distance effect ($F=3.10$, $p=0.070$), with higher abundances on reefs at a distance of 5m than 30m or 55m. No other dominant species (Table 8) exhibited distance effects or interaction between distance and season. There were no significant differences (Table 8) in total length for oyster spat (Figure 13) and carapace width for mud crabs among distances.

Predation Experiments

There were no distance effects for predation on *Paleomonetes pugio* during the tethering experiments. Predation was 100% at a distance of 5m from subtidal channel, 89% at 30m, and 100% at 55m. Clam outplant experiments yielded high variability between samples (0-100%). Trials conducted on 3 November (24 hr.) and 18 November (48 hr.) yielded no distance effects (Table 9), however the 11 November (48 hr.) deployment showed an interaction between distance and treatment ($F=3.88$, $p=0.040$). At a distance of 5m from the subtidal channel, more clams survived in the reef edge treatment than the sandflat treatment (Figure 14), though there was no distance effect overall.

Treatment Effects

Figure 12. Mean abundances by distance for *Balanus* spp, *Geukensia demissa*, and *Crepidula* spp. in excavations for August 2005 and January 2006 samplings. Bars indicate the mean abundance (+SE) per quadrat. Abundances in January are significantly higher than August for *G. demissa*.

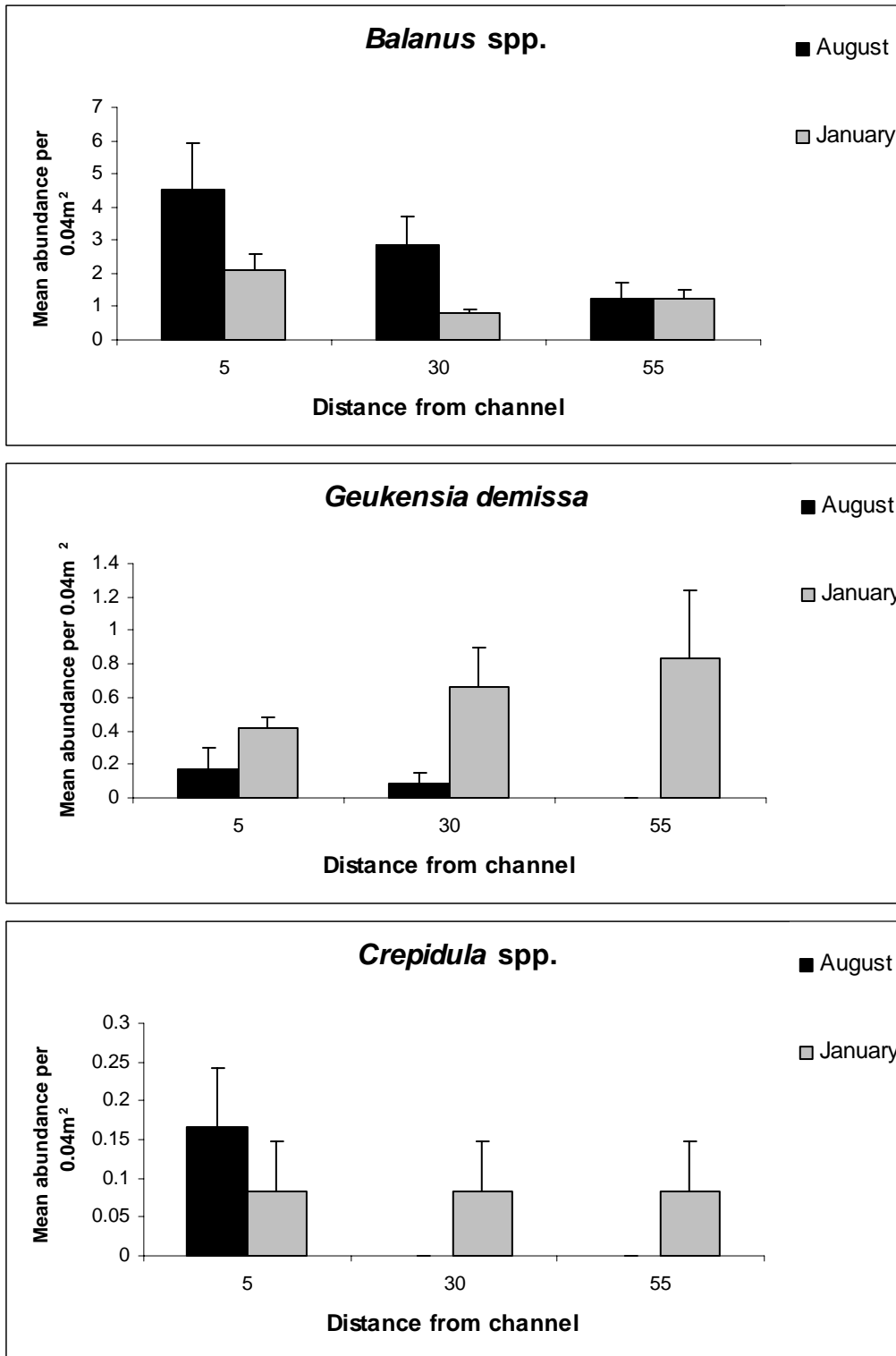


Table 8. 2-way ANOVA results for mean abundance of common species and mean total length of *Crassostrea virginica* and mean carapace length of *Panopeus herbstii* from excavations. Shown are F-values and (p-values) with significant differences indicated by an asterisk. When significant effects are present, SNK rankings are shown in decreasing order, with differences indicated with superscript letters. 5=5m, 30=30m, 55=55m, SUM=August 2005, WIN=January 2006.

Species	Distance	Season	Distance*Season
Abundance			
<i>Crassostrea virginica</i>	0.75 (0.4849)	3.19 (0.0910)	0.53 (0.5996)
<i>Balanus</i> spp.	3.10 (0.0698)	3.17 (0.0917)	1.41 (0.2696)
<i>Panopeus herbstii</i>	1.50 (0.2493)	9.54* (0.0063)* SUM ^a , WIN ^b	0.42 (0.6655)
<i>Guekensia demissa</i>	0.05 (0.9523)	8.45 (0.0094)* WIN ^a , SUM ^b	0.48 (0.6257)
<i>Crepidula</i> spp.	0.92 (0.4153)	0.23 (0.6367)	0.92 (0.4153)
Length			
<i>Crassostrea virginica</i>	0.68 (0.5168)	101.80* (<0.0001)* WIN ^a , SUM ^b	0.31 (0.7360)
<i>Panopeus herbstii</i>	0.38 (0.6897)	0.01 (0.9383)	0.13 (0.8762)

Figure 13. Mean total length (mm) by distance for *Crassostrea virginica* and mean carapace width for *Panopeus herbistii* in excavations from August 2005 and January 2006 samplings. Bars indicate the mean length (+SE) per quadrat. Total length was significantly greater in January than August for *C. virginica*.

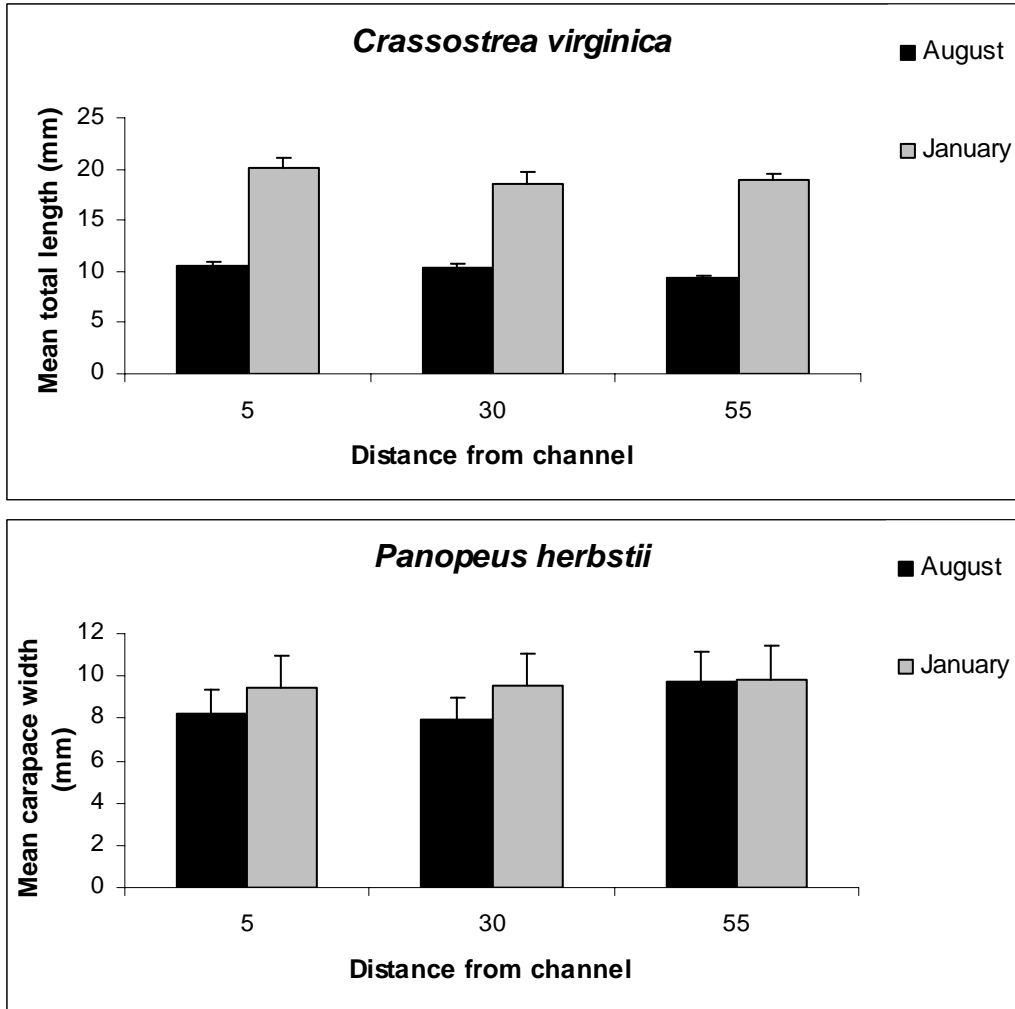
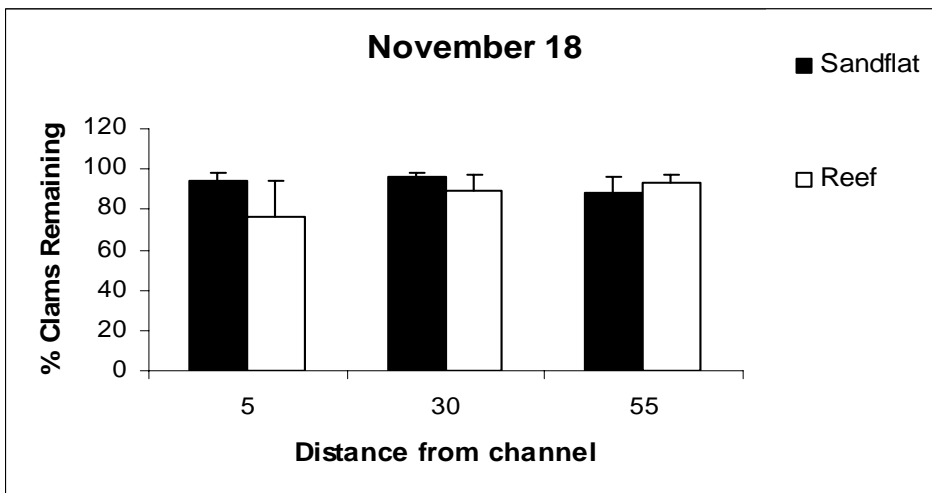
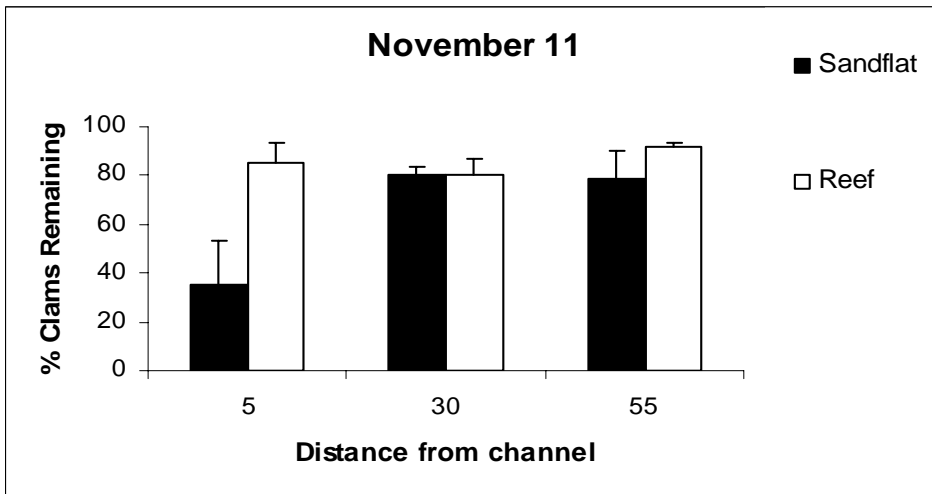
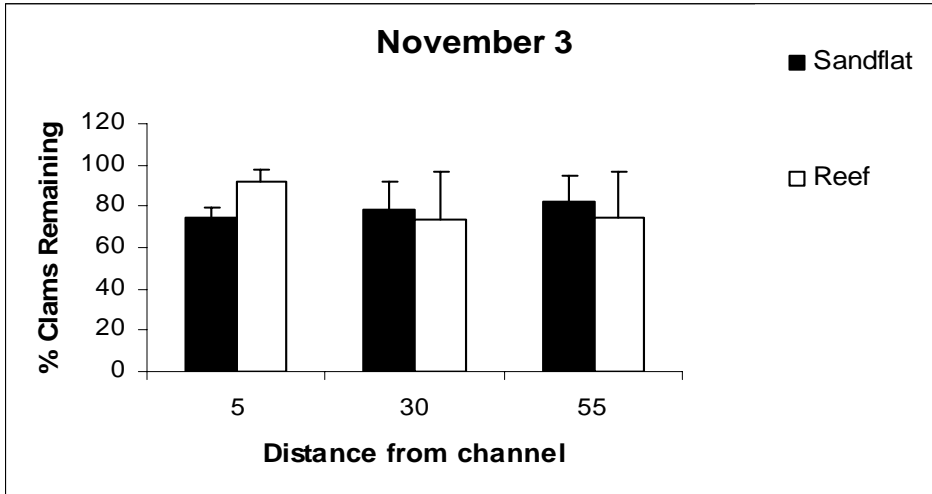


Table 9. 2-way ANOVA results for the effects of distance, treatment and possible interactions from clam predation experiments. Shown are F-values and (p-values) with significant differences indicated by an asterisk. When significant effects are present, SNK rankings are shown in decreasing order, with differences indicated with superscript letters. 5=5m, 30=30m, 55=55m, R=Reef, S=sandflat.

Date	Distance	Treatment	Distance*Treatment
Nov. 3, 2005	0.11 (0.7466)	0.12 (0.8919)	0.47 (0.6346)
Nov. 11, 2005	2.78 (0.0885)	7.82* (0.0119)* R ^a , S ^b	3.88* (0.0396)*
Nov. 18, 2005	0.39 (0.6855)	0.78 (0.3855)	0.61 (0.5537)

Figure 14. Mean percent clams, *Mercenaria mercenaria*, remaining from November 2005 outplant experiments. Bars indicate (+SE) the mean percentage of clams remaining per treatment for each distance. Percent clams remaining was significantly higher overall for reef than sandflat on November 11.



Transient Fauna

There were significant treatment effects for species sampled with Breder traps (sand edge, reef, sandflat) and surround nets (reef, sandflat). For summer 2005 (Figure 4), mean total abundances caught in Breder traps (Table 2) were higher for the sand edge treatment than reef or sandflat treatments. *L. rhomboides* and *L. xanthurus* were significantly more abundant on the sand edge treatment than either reef or sandflat treatments. For *L. rhomboides*, there was an interaction between treatment and time period (Table 10) with higher abundances on reef treatments compared with sandflat in May and the first June sampling, and higher abundances for sandflat treatment than reef treatment for July. There were significant differences in total length (Table 5) for *L. rhomboides* and *L. xanthurus*, with longer pinfish (Figure 6) caught on sand edge (average total fish length 47.5mm) and sandflat treatments (44.9mm) than reef treatments (41.0mm). Longer spot were caught over sandflat (average length 57.2mm) and sand edge (53.3mm) treatments than on reef (44.4mm) treatments.

F. heteroclitus abundances showed a significant 3-way interaction between distance, treatment, and time period ($F=2.62$, $p=0.005$). During all time periods (Table 11), *F. heteroclitus* was more common on sand edge treatments than reef treatments. Only the July sampling yielded no significant differences between sand edge and sandflat treatments. In May, there was an interaction between distance and treatment ($F=6.36$, $p=0.010$) with higher abundances of *F. heteroclitus* found on sand edge treatments at a distance of 30m from subtidal channel ($F=11.04$, $p=0.041$) than sandflat and reef treatments, and no interaction with distances of 5m and 55m. For the June 20-21 time

Table 10. By time period treatment effects for *Lagodon rhomboides* and mean total abundance for summer 2005 Breder trap sampling. Shown are F-values and (p-values) with significant differences indicated by an asterisk. When significant effects are present, SNK rankings are shown in decreasing order, with differences indicated with superscript letters. E=sand edge, R= reef, S=sandflat.

Time Period	Treatment
<i>Lagodon rhomboides</i>	
May 27-31	6.78* (0.0034)* E ^a , R ^a , S ^b
June 20-21	4.50* (0.0195)* R ^a , E ^{ab} , S ^b
June 27-28	5.73* (0.0073)* E ^a , R ^b , S ^b
July 19-20	9.55* (0.0005)* E ^a , S ^a , R ^b
Mean Total Abundance	
May 27-31	13.23* (<0.0001)* E ^a , R ^b , S ^c
June 20-21	5.58* (0.0089)* E ^a , R ^a , S ^b
June 27-28	9.41* (0.0006)* E ^a , S ^b , R ^b
July 19-20	22.68* (<0.0001)* E ^a , S ^a , R ^b

Table 11. Effects of distance, treatment, and interactions, by time period, on mean abundances of *Fundulus heteroclitus* caught in Breder traps. Shown are F-values and (p-values) with significant differences indicated by an asterisk. When significant effects are present, SNK rankings are shown in decreasing order, with differences indicated with superscript letters. 5=5m, 30=30m, 55=55m, E=sand edge, R=reef, S=sandflat, May=May 27-31, Jun1=June 20-21, Jun2=June 27-28, Jul=July 19-20.

Time Period	Distance	Treatment	Distance*Treatment
May 27-31	2.94 (0.1041)	12.92* (0.0023)* E ^a , R ^b , S ^b	6.36* (0.0103)*
June 20-21	7.13* (0.0037) 5 ^a , 30 ^b , 55 ^b	14.40* <0.0001)* E ^a , R ^b , S ^b	4.91* (0.0049)*
June 27-28	0.17 (0.8429)	9.29* (0.0009)* E ^a , S ^b , R ^b	0.48 (0.7481)
July 19-20	1.01 (0.3760)	5.85* (0.0078)* E ^a , S ^a , R ^b	1.07 (0.3895)

period (Figure 15), there was an interaction between distance and treatment ($F=4.91$, $p=0.005$) with higher abundances of *F. heteroclitus* found on sand edge treatments at a distance of 5m from subtidal channel ($F=13.61$, $p=0.002$) than sandflat and reef treatments, and no significant interactions with distances of 30m and 55m. Total other finfish and decapod crustaceans (Figure 6) were far more common on sand edge and sandflat treatments than on the reef.

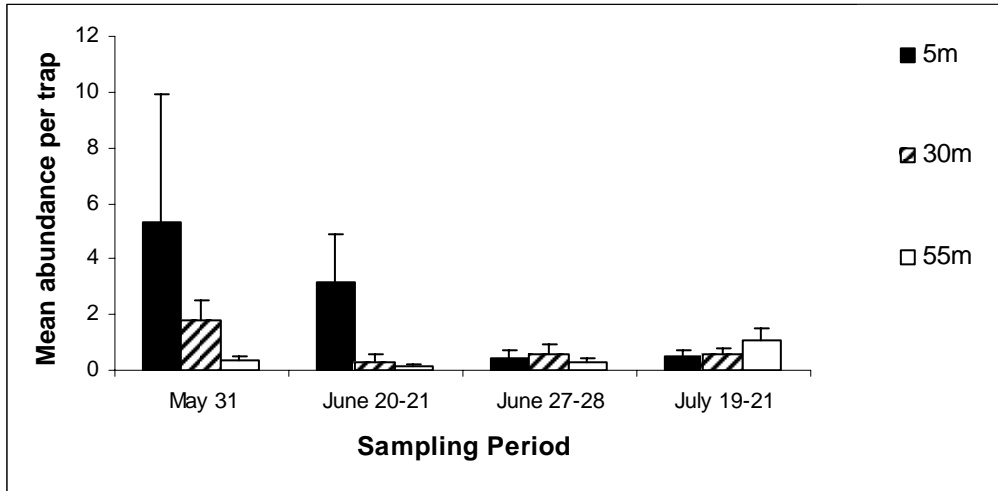
For Breder trap catches in winter 2006, mean total abundances (Table 4), including less abundant finfish and decapod crustaceans, were higher on sand edge and sandflat treatments than on reef treatments. Larval *L. xanthurus* showed a significant treatment effect (Figure 7), with abundances higher on sand edge and sandflat treatments than reef treatment. Larval *L. rhomboides* were more common on sand edge treatments than reef treatments.

Surround net catches indicated no significant differences (Table 6) in abundance between treatments (reef and sandflat) for juvenile *L. rhomboides*, juvenile *L. xanthurus*, total non-reef finfish, and total other reef users. *L. rhomboides* were significantly longer (Table 7) on sandflat than reef treatments (mean total lengths: 64.2mm and 58.1mm respectively, Figure 9). For *L. xanthurus*, there were no significant total length differences between distances.

Predation Experiments

Predation upon *Paleomonetes pugio* was 100% for tethers deployed on sandflat treatments and 93.6% for reef treatments. For the clam outplant experiments there were

Predation upon *Paleomonetes pugio* was 100% for tethers deployed on sandflat treatments and 93.6% for reef treatments. For the clam outplant experiments there were Figure 15. Mean abundances by distance from subtidal channel and sampling period for *Fundulus heteroclitus* caught in Breder traps during summer 2005. Bars indicate mean organisms (+SE) per trap for each distance during each time period. Abundances were significantly higher at a distance of 5m than 30m or 55m during June 20-21.



no treatment effects for the November 3 and 18 deployments. There was an interaction between distance and treatment mentioned above on November 11, 2005.

Seasonal Effects

Transient Fauna

For summer 2005 Breder traps, highest total abundance occurred during May sampling, with intermediate catches in June, and lower catches in July (Table 2). There was an interaction between time period and treatment with higher abundances on sand edge compared with sandflat treatments in May and the first June sampling, and higher abundances for the sandflat treatment than reef treatment for July.

L. rhomboides total length differed between summer 2005 time periods (Table 3) with longer fish caught in July than in May and intermediate lengths caught in June. Time period differences occurred for *L. xanthurus*, with significantly higher abundances (Table 2) during the May 27-31 sampling than June 20-21, June 27-28, and July 19-20 samplings. *L. xanthurus* total length was different between sampling time periods ($F=23.41$, $p<0.001$) with longer fish caught in July than in May and intermediate lengths caught in the June periods. *F. heteroclitus* were more abundant in May than in the second June sampling period. There was no significant difference in *F. heteroclitus* total length between time periods. Abundances of other finfish did not differ across time periods, however, there were significantly more decapods ($F=4.35$, $p=0.006$) caught during the June 20-21 sampling than the remaining time periods.

Resident Fauna

Abundances of resident epifauna varied between sampling seasons. For *Crassostrea virginica* spat January 2006 abundances included oysters that set in summer 2005, and fall 2005, though there was no difference in the overall abundances. In August, oyster density (live) was 39.58 individuals per 0.04m². In January, density was 51.17 individuals per 0.04m². *Panopeus herbstii* showed strong seasonal differences (Table 8), with higher abundances in August 2005. *Geukensia demissa* were more abundant in January than August.

Mean total length (Table 9) of live oyster spat was significantly higher (Figure 13) in January 2006 (19.2mm) than August 2005 (10.9mm); however, there were no significant variations in oyster length across distances. Mean carapace width of *P. herbstii* did not vary between seasons. Overall, the carapace width of mud crabs in August ranged from 2.5mm-40.3mm and in January ranged from 3.1mm-21.8mm.

Infauna

Sediment cores were analyzed by major taxa and by season: August 2005 and January 2006. Dominant taxa found in cores (those comprising >1% of total) include: polychaete (Capitellidae, Cirratulidae, Glyceridae, Orbiniidae, Paraonidae, Spionidae, and Syllidae), oligochaetes, juvenile bivalves, and amphipods (Gammaridae). Mean total abundance (Figure 16) differed by season, with more organisms found in January. Mean total abundances (Table 12) did not differ across distances in August, however higher abundances occurred in sandflats than sand edge treatments. For January cores (Table 13), there were no significant differences in abundance across distances or between treatments.

Figure 16. Mean abundances for total infauna per 0.01m² for August 2005 and January 2006. Bars indicate (+SE) mean organisms per cores each treatment at each distance. Abundances were significantly higher for sandflat than reef treatments in August.

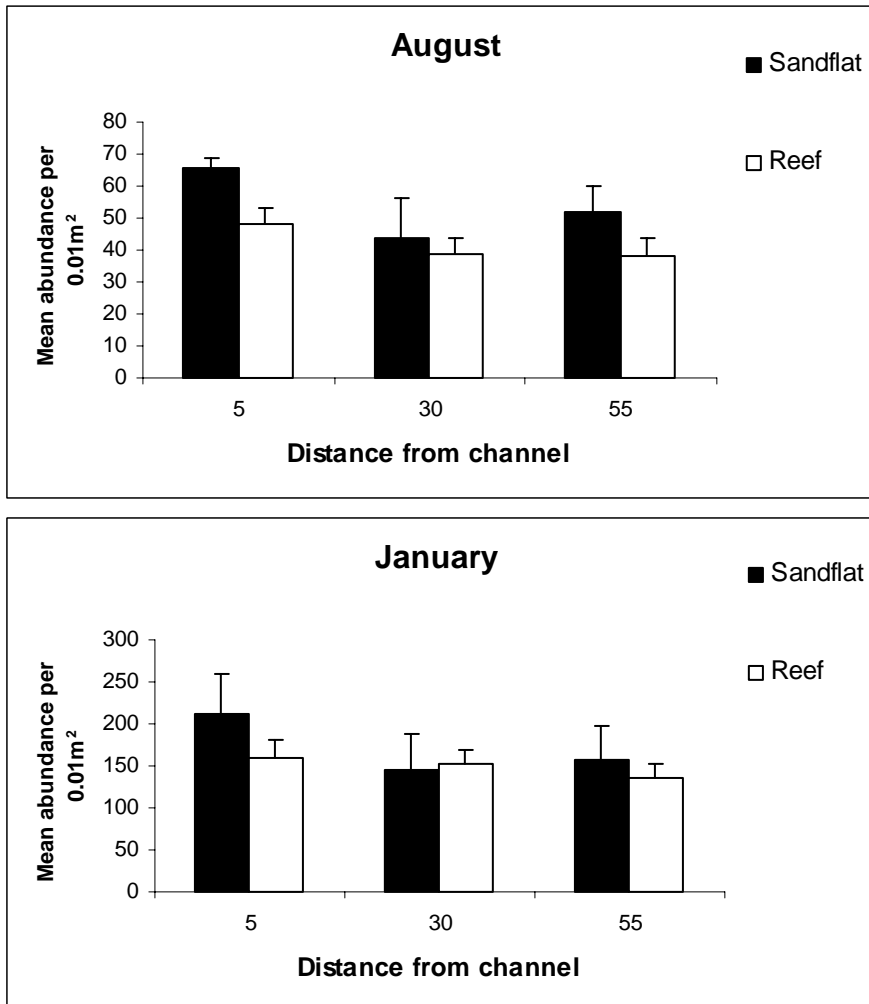


Table 12. 2-way ANOVA results for the effects of distance, treatment and possible interactions on mean abundance of dominant taxa in sediment cores from August 2005. Shown are F-values and (p-values) with significant differences indicated by an asterisk. When significant effects are present, SNK rankings are shown in decreasing order, with differences indicated with superscript letters. 5=5m, 30=30m, 55=55m, E=sand edge, S=sandflat.

Species	Distance	Treatment	Distance*Treatment
Mean total abundance	4.54 (0.0631)	6.37* (0.0451)* S ^a , E ^b	0.66 (0.5487)
Total polychaetes	3.54 (0.0964)	1.70 (0.2403)	1.36 (0.3265)
Capitellidae	0.18 (0.8413)	0.09 (0.7778)	0.16 (0.8545)
Cirratulidae	1.72 (0.2574)	0.03 (0.8701)	3.30 (0.1078)
Glyceridae	1.47 (0.3031)	3.57 (0.1076)	0.46 (0.6518)
Paraonidae	2.90 (0.1318)	5.10 (0.0648)	0.03 (0.9727)
Orbiniidae	0.96 (0.4332)	0.31 (0.5999)	0.26 (0.7781)
Spionidae	7.56* (0.0229)* 5 ^a , 55 ^a , 30 ^b	3.07 (0.1303)	2.43 (0.1685)
Syllidae	2.39 (0.1723)	2.86 (0.1417)	1.17 (0.3729)
Oligochaete	0.02 (0.9770)	0.40 (0.5497)	0.34 (0.7215)
Juvenile bivalve	4.42 (0.0661)	16.62* (0.0065)* S ^a , E ^b	0.06 (0.9405)
Gammaridae	1.56 (0.2856)	11.59* (0.0144)* S ^a , E ^b	3.16 (0.1154)
Total infaunal biomass	0.38 (0.6985)	1.76 (0.2329)	2.51 (0.1610)

Table 13. 2-way ANOVA results for the effects of distance, treatment and possible interactions on mean abundance of dominant taxa in sediment cores from January 2006. Shown are F-values and (p-values) with significant differences indicated by an asterisk. When significant effects are present, SNK rankings are shown in decreasing order, with differences indicated with superscript letters. 5=5m, 30=30m, 55=55m, E=sand Edge, S=sandflat.

Species	Distance	Treatment	Distance*Treatment
Mean total abundance	0.58 (0.5891)	0.13 (0.7354)	0.29 (0.7558)
Total polychaetes	3.46 (0.1003)	2.99 (0.1343)	1.58 (0.2803)
Capitellidae	0.66 (0.5515)	0.01 (0.9381)	2.11 (0.2020)
Cirratulidae	7.15* (0.0258)* 5 ^a , 30 ^{ab} , 55 ^b	0.61 (0.4633)	0.22 (0.8057)
Glyceridae	0.12 (0.8848)	0.13 (0.7319)	0.47 (0.6482)
Orbiniidae	6.80* (0.0287)* 30 ^a , 5 ^a , 55 ^b	2.85 (0.1426)	1.25 (0.3519)
Paraonidae	0.49 (0.6337)	1.13 (0.3288)	0.14 (0.8694)
Spionidae	0.63 (0.5640)	0.02 (0.9017)	2.53 (0.1599)
Syllidae	0.89 (0.4590)	3.85 (0.0974)	0.58 (0.5860)
Oligochaete	0.06 (0.9388)	0.71 (0.4315)	0.01 (0.9883)
Juvenile bivalve	13.48* (0.0060)* 5 ^a , 30 ^b , 55 ^b	0.07 (0.8002)	5.03 (0.0521)
Gammaridae	1.25 (0.3519)	0.11 (0.7519)	0.47 (0.6465)
Total infaunal biomass	0.63 (0.5622)	0.45 (0.5285)	0.77 (0.5038)

The total number of polychaetes (Figure 17) did not differ across distances or treatments in either August or January, however family level responses were detected. In August, Spionidae were found in greater abundances at distances of 5m and 55m from the subtidal channel compared with distances of 30m ($F=7.56$, $p=0.023$). Also in August, Paraonidae showed marginally non-significant greater abundances ($F=5.10$, $p=0.065$) in sandflat treatments. In January, Cirratulidae and Orbiniidae showed significant distance effects, with cirratulids more common at 5m from subtidal channel than 55m ($F=7.15$, $p=0.026$), and orbiniids more common at distances of 30m and 5m compared with 55m ($F=6.80$, $p=0.029$).

Juvenile bivalves (Figure 18) were more common in sandflat cores than sand edge cores in August (Table 12). There was a marginally non-significant distance effect, with more bivalves at a distance of 5m from subtidal channel. In January, there were no treatment differences; however there was a distance effect. Bivalves were more abundant in cores taken at a distance of 5m from the subtidal channel than distances of 30m or 55m. Gammarid amphipods (Figure 19) had higher abundances in sandflat treatments in August ($F=11.59$, $p=0.014$), but there were no distance effects and no significant distance or treatment effects in cores taken in January. Measurements of biomass from sediments cores showed no significant distance or treatment effects for both August and January cores. There was a significantly higher infaunal biomass ($F=6.46$, $p=0.026$) in January (40.18mg) than August (22.54mg).

Figure 17. Mean abundances for total polychaetes per 0.01m^2 for August 2005 and January 2006. Bars indicate (+SE) mean organisms per cores each treatment at each distance.

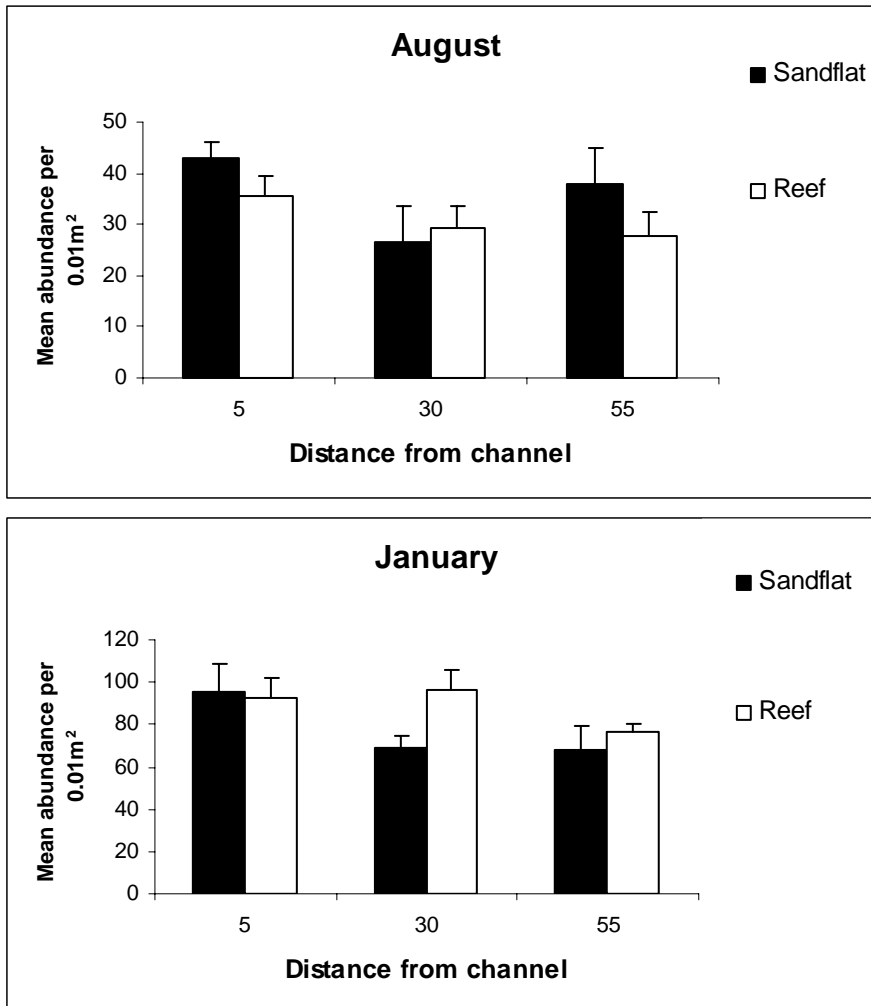


Figure 18. Mean abundances for juvenile bivalves per 0.01m² for August 2005 and January 2006. Bars indicate (+SE) mean organisms per cores each treatment at each distance. Abundances were significantly higher for sandflat than reef treatments in August. Abundances were significantly higher at 5m than 30m or 55m in January.

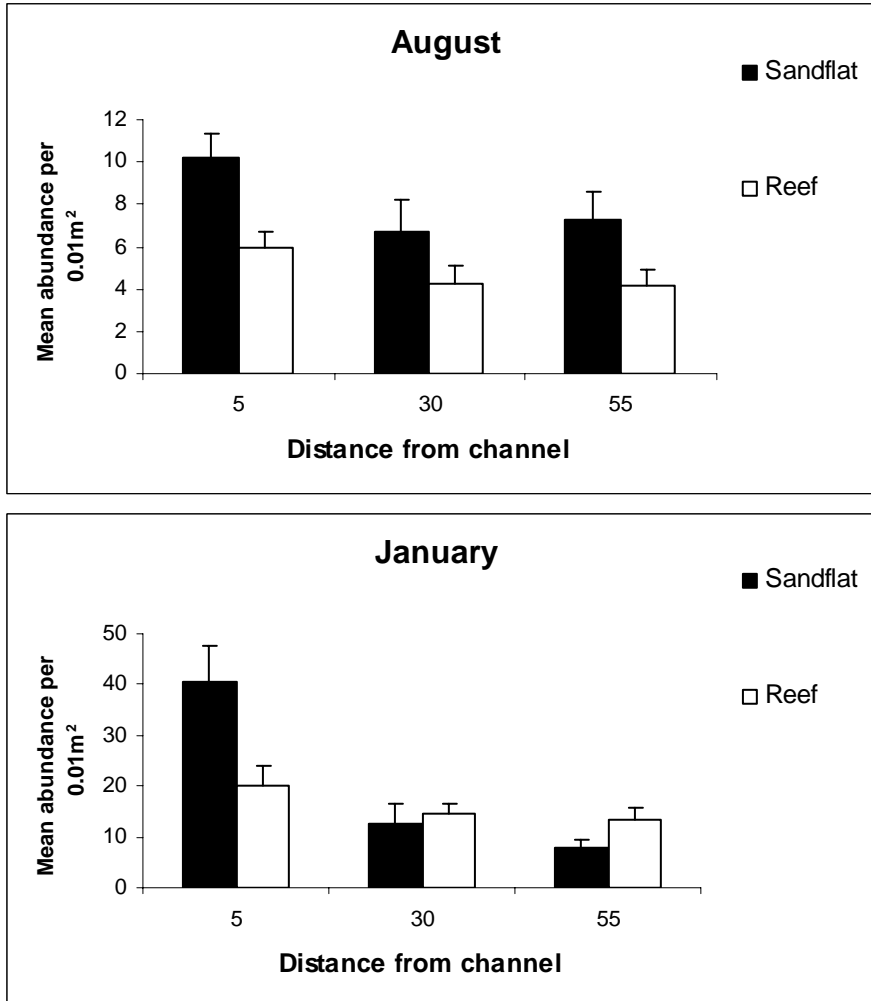
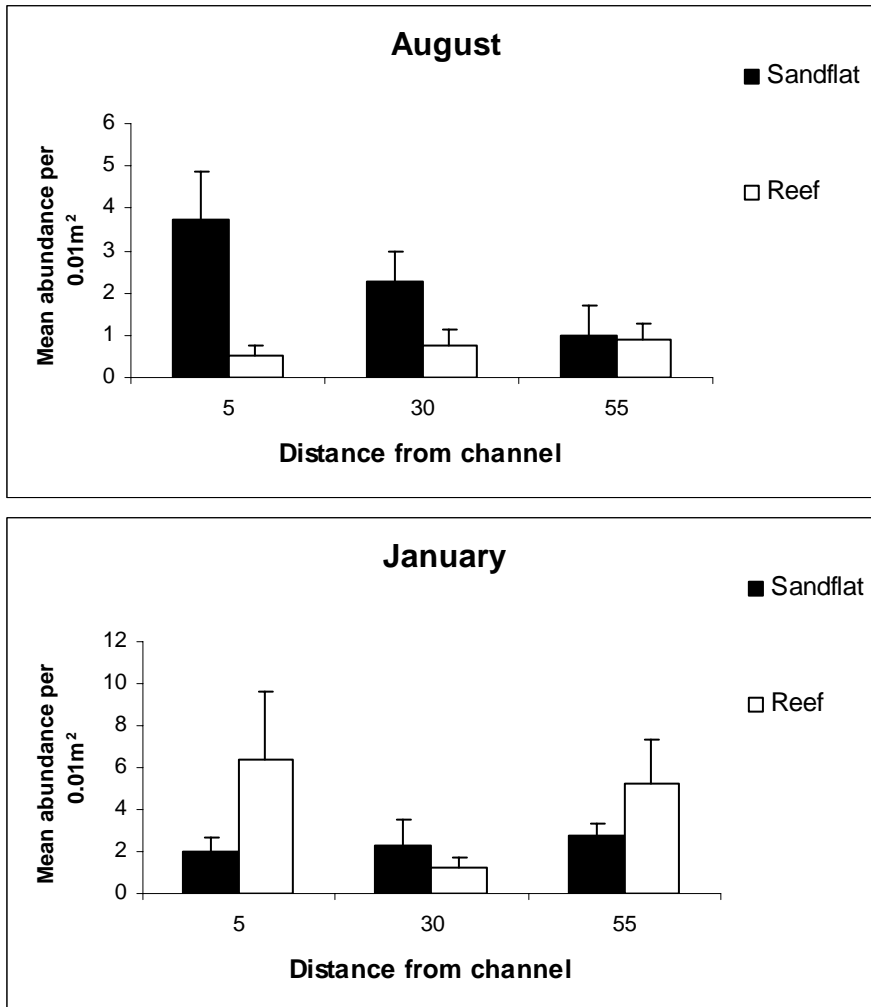


Figure 19. Mean abundances for gammarid amphipods per 0.01m² for August 2005 and January 2006. Bars indicate (+SE) mean organisms per cores each treatment at each distance. Abundances were significantly higher for sandflat than reef treatments in August.



DISCUSSION

This study indicated mixed patterns of reef utilization based on distance from subtidal channel. Overall abundances of transient fauna sampled in summer 2005 with both Breder traps and surround nets did not differ among the three distances. During February 2006, the high abundances of larval *Leiostomus xanthurus* coincide with early spawning and recruitment to the estuary (Hales and Van Den Avyle 1989) followed by a later recruitment by *Lagodon rhomboides* (Hoss 1974). A weak distance effect was found for larval pinfish, with higher abundances nearer the subtidal channel (5m) than at an intermediate distance (30m). This may be a consequence of an initial encounter with structure near the subtidal channel resulting from initial movement during the beginning of the flood tide. No distance effects were found for dominant species, total non-reef users and total other reef users sampled with surround nets. There were also no differences in total fish length for dominants across distances for either transient sampling method.

In addition to larval *L. rhomboides*, *Fundulus heteroclitus* also showed distance effects. This study period coincided with the seasonal spawning period for *F. heteroclitus* (Abraham 1985). *F. heteroclitus* are less vagile than juvenile *L. rhomboides* and *L. xanthurus*, prefer shallow water (Kneib 1987) and migrate along the tidal front (Abraham 1985). Since surround net sampling occurred in deeper water, these data offer additional evidence for tidal front migration of *F. heteroclitus* because no fish were caught using this method. Breder trap data show significantly higher mummichog abundances at the sand edge treatment at a distance of 5m (June 20-21) from the subtidal channel over distances of 30m and 55m. This pattern is repeated on May 31, but was not

significant, most likely due to low replication with only 1 day included in this sampling time. Catches from later sampling periods (June 27-28, July 19-20) were too low to detect distance patterns (approximately 0.5 mummichogs per trap). Abraham (1985) suggests that the marsh edge is the preferred habitat for *F. heteroclitus*, which seek refuge and forage in the tall grass. This might lead to a prediction that *F. heteroclitus* abundances would be highest at a distance 55m from the subtidal channel, which is the closest to marsh grass. Breder traps were placed before submergence and as the tide rises, the traps fill with water and transients were passively caught in the traps. If *F. heteroclitus* stay with the tidal front, those data should reflect similar abundances of fish caught between distances. A mesocosm experiment (McDonald 2003) found *F. heteroclitus* orienting with oyster shell in deeper water compared with reefs and unstructured bottom in shallower water. During the May 31 and June 20-21 sampling periods, mummichogs may have encountered structure nearer the subtidal channels and remained there after the tidal front passed and the water became deeper, and this pattern was not detectable in later time periods due to lower catches. Since there is little structure in subtidal areas (low tide refuges for mummichogs, Kneib 1987), the reefs closest to the channel may be more important to migrating *F. heteroclitus*. There may be a tradeoff between staying in shallow water and staying near structure. However, using surround net data, once the depth of water increases, then *F. heteroclitus* are no longer in the mid intertidal areas.

Resident epifauna, those organisms residing permanently within the oyster reefs, showed no difference in abundances with respect to proximity to mid intertidal areas for mobile species such as xanthid crabs. There was a trend towards higher *Balanus* spp.

abundances among reefs at a distance of 5m from the subtidal channel compared with reefs at 30m and 55m. This may be a result of greater recruitment or survival due to greater submergence of reefs nearer subtidal waters.

Panopeus herbstii, showed evidence of both larval recruitment and migration to the newly created reefs. In August 2005, less than six months after creation, fully grown adult mud crabs were present on reefs (carapace width min 2.5mm, max 40.3mm), suggesting immigration between patches by adult crabs. This migration may follow a pattern of island biogeography (MacArthur and Wilson 1967) with crabs moving from established habitat to nearby patches, although the distances used in this study may be small enough (Kotlair and Weins 1990) to allow for migration of xanthid crabs. The crabs may be using reefs as stepping stones (Breitburg et al. 2000) for migration across the intertidal area. Further work is needed to determine if migration thresholds exist for adult xanthid crabs.

Predation experiments yielded mixed results. Predation on *Paleomonetes pugio* was too high to detect a pattern, possibly an artifact tethering and resulting limited mobility of the prey (Kneib and Steele 2000). Clam outplants were conducted late in the season (November 2005), possibly after many predators migrated from the estuary (Hoss 1974, Hales and Van Den Avyle 1989). Clam outplants from 3 November, and 18 November showed no difference between distances, however the 11 November trial indicated increased predation in sandflat treatments at a distance of 5m from the subtidal channel compared with other treatments and distances. This is probably a result of a single predator finding a particular sampling tray and consuming most of the clams inside. There was high variability in prey consumption and during preliminary laboratory

trials, a blue crab (*Callinectes sapidus*) consumed the majority of clams within certain trays but few clams within others. Shell fragments from the clam outplant trials were consistent with shell fragments remaining from crab predation in those laboratory trials. These data suggest that only a few predators may have been responsible for clam predation in the field.

An important result of this study is that there are microhabitat utilization differences occurring on the reef edge. The size of the reefs (2.2m diameter) created a high edge to interior ratio (Forman 1995). Increase edge habitat may be utilized by certain transient species in intertidal areas (Harwell 2004) as long as patch size is above a minimum threshold. Understanding edge effects may be important in understanding variation in patch usage (Saunders et al. 1991), particularly with a goal of restoration and management. The interface of reef and sand could be an important microhabitat for transients migrating between subtidal channels and the high intertidal zone because this habitat may provide a combination of foraging area and protective structure for refuge. In fragmented forests, brown-headed cowbirds are more abundant than continuous forests, because smaller patches allow access to foraging within grasslands and reproduction within forest patches (Burgham and Picman 1998).

For Breder trap summer 2005 data, *L. rhomboides*, *L. xanthurus*, and *F. heteroclitus* were more abundant in sand edge habitat compared with on reef and sandflat treatments. *F. heteroclitus*, as previously stated, were more abundant on the sand edge nearer the channel (5m) during the May 31 and June 20-21 periods. These fish may be encountering the edge microhabitat and remaining there to forage and seek refuge until the tidal front moves higher into the intertidal zone, and more structured habitat such as

marsh grass becomes available (Abraham 1985). Also, higher abundances of *L. rhomboides* and *L. xanthurus* (juveniles and larval) were found on the sand edge than the on reef or sandflat treatments.

Smaller *L. rhomboides* (mean total length 41.0mm) were caught in Breder traps located over oyster shell than in either sand edge (47.5mm) or sandflat treatments (44.93mm) contradicting previous evidence that smaller fish were found in greater abundances over unstructured sand than a restored reef (Harding and Mann 2001). *L. xanthurus* captured in reef treatments (44.4mm) were 22% smaller than those captured over sandflat treatments (57.2mm) and 17% smaller than those individuals found in sand edge treatments (53.3mm). The presence of shell may have provided a refuge for smaller fish to escape predation. Larger fish may be exposed to a lower risk of predation and therefore are more commonly found using unstructured sandflats and sand borders of intertidal reefs, habitats that may be better for foraging due to higher abundances of infaunal prey in adjacent sediments (Powell 1994). Breder traps catch fish migrating during the first 2 hours of the flood tide and may select for smaller, juvenile fish (Rozas and Minello 1997), however surround net data support these results for *L. rhomboides*. Surround nets were not as selective for fish size, with *L. rhomboides* length ranging from 45.0-147.0mm. Abundances of *L. rhomboides* and *L. xanthurus* taken in surround nets treatments did not vary significantly between reef and sandflat. Significantly smaller *L. rhomboides* were captured around the reefs (58.1mm) than on unstructured sandflats (64.2mm). This pattern is similar to Breder trap data for *L. xanthurus* (larger fish on sandflat) but since surround nets combine both edge and interior, this data seems to contradict Breder trap data for *L. rhomboides*, where larger fish were found in the sand

edge and smaller fish on the reef. During mid tide, the reef complex (reef and sand edge) may be act as a refuge for smaller individuals. No significant size differences in spot length were found from surround net catches, probably a result of low abundances.

Catches of transient fauna from Breder traps were higher in May than in July, with intermediate abundances in June. Although this abundance data may only reflect seasonal migration or mortality of juvenile fish away from the estuary, the patterns observed highlight the microhabitat utilization of the reef edge. Smaller fish utilize reef shell habitat to a greater extent than larger fish of the same species, which are found in greater abundances on the sand edge and sandflats. For *L. rhomboides*, the shift in microhabitat utilization that occurred during the summer may be a direct result of the increase in total length. *L. rhomboides* caught earlier were significantly smaller and were using the reef shell as well as the sand edge over sandflat. As the fish grew, the reef habitat may become less important, perhaps because of a reduced risk of predation. The larger, late season fish may then use sand edge and sandflat areas with high abundances of infauna to forage. August surround net data confirm that smaller *L. rhomboides* orient to reefs compared with open sandflat during mid tide, possibly for the mix of both refuge potential from the structure and foraging area.

In winter 2006, both larval *L. xanthurus* and larval *L. rhomboides* were significantly more abundant in sand edge and sandflat treatments than the reef treatment. This result may be a behavioral mechanism keeping larval fish in shallow water as the tide rises. Due to the 3-dimensional structure of the reefs, these fish may have migrated past reefs, or been moved by tidal flow, by the time the tide had fully submerged the reefs. For larval fish, shallow water may offer a better refuge from predation than

structured habitat with vertical relief. Mid-tide sampling would be necessary to relate larval fish abundances to those found with Breder traps.

This study supports data from previous work (Meyer 1994, Meyer and Townsend 2000, Harwell 2004) that oyster reefs support enhanced densities of resident organisms than unstructured habitat. Although reef proximity to upper intertidal source areas or subtidal channels resulted in few significant patterns, patch isolation (>20m to other structured habitat in this study) may be the reason similar colonization rates of resident fauna occurred among reef patches (Virnstein and Curren 1986, Roberts and Poore 2006).

There was no variation in the recruitment of *Crassostrea virginica* between the two sampling periods. There were two spawning peaks, represented by one size class in August 2005, and two size classes in January 2006. Ortega and Sutherland (1992) discovered regional variation in oyster spawning ranging from a single seasonal peak to continuous spawning. Because two size classes were sampled in January 2006 (summer and fall recruitments), the mean total length of the oysters was significantly longer than those sampled earlier. There were significant seasonal differences for *Panopeus herbstii* (black-fingered mud crab) and *Geukensia demissa* (ribbed mussel). Mud crabs were more common in August 2005 than January 2006. Mussels, particularly *G. demissa*, are an important food source for mud crabs (Seed 1980) and the increase in abundance in January 2006 could be a result of a decline in mud crab abundances and activity. Overall, resident data suggest quick colonization and recruitment of new reef patches.

Infaunal abundances varied greatly between seasons, most likely due to reduced fish and decapod abundances during winter sampling and fall recruitment of some infaunal organisms. There were higher abundances of all major infaunal groups in

January 2006. The polychaete family Spionidae exhibited highest densities at 5m and 55m in August 2005. There is no obvious spatial correlation to nekton abundances that explains the Spionidae pattern. In January 2006, two polychaete families, Cirratulidae and Orbiniidae, and juvenile bivalves were found in higher abundances in treatments closer to the subtidal channels. This may be a result of recruitment patterns and survivorship nearer the subtidal channel where submergence time is greater (approximately 20 min. in this study). The pattern was not seen in the summer, most likely due to predation by seasonal transients.

In August 2005, there was a weak subhabitat effect for mean total infaunal abundances, juvenile bivalves and gammarid amphipods, with higher abundances away from reefs. The lower infaunal abundances around oyster reefs may be the result of increased nekton utilization of reefs (Powell 1994). A halo effect, where infaunal abundances are lower around structure as a result of predation by epifauna, has been found in infaunal communities associated with intertidal oyster reefs (Powell 1994) and nearshore hard bottom communities (Posey and Ambrose 1994). In January 2006, there was no difference in infaunal abundances between reef edge and sandflat treatments; a possible result of reduced numbers of reef-associated predators present in the estuary.

The results of this study provide additional support for the importance of oyster reefs as habitat for transient nekton and resident epifauna as well as evidence for variation in microhabitat use along the reef edge. There were some significant differences in epifaunal and nekton abundances related to distance from subtidal channel, particularly with *F. heteroclitus*, larval *L. rhomboides*, and *Balanus* spp. However, for the majority of species sampled, there was similar use of all distances sampled. This

study showed that patch oyster reefs at varying distances from subtidal channels provide sufficient recruitment habitat for oyster larvae, as well as habitat for other reef residents. Patch reefs are also used during flood tide by transient finfish and crustaceans in greater abundances than unstructured sandflats. Reefs at all distances provided habitat for transient juveniles and larval fishes. During mid-tide, while high intertidal habitat such as marsh grass and fringing oyster reefs are available, patch reefs between the high intertidal zone and subtidal channel are still utilized by transient species.

The specific distances (5m, 30, 55m) used in this study for the creation of artificial reefs may not necessarily be perceived by transient and resident nekton, which have access to all areas of the intertidal zone through mobility and recruitment. Jordan et al. (2005) found increased fish abundances in marine artificial reef habitats with the greatest separation (25m, similar to present study) between patches. Patch isolation may be the reason that abundances were similar among reef patches used in this study. Evidence from this study may not approximate patch usage within larger intertidal systems (>75m between channels and high intertidal zone), though the distances between channels and higher marsh used in this study are typical of areas in southeastern North Carolina.

The loss of oyster reef habitat in recent years requires better understanding of the ecological function of oysters (Coen et al. 1999, Breitburg 2000), as well as an increased effort towards conserving and restoring oyster reefs. A decline in oyster reefs has led to fragmentation of patch reefs across the landscape which results in the increase of edge to interior habitat among reefs. This study emphasizes the importance of the edge habitat for motile species, including differences in microhabitat utilization along the shell/sand

interface by different sizes of transient fishes, giving further evidence that edge habitat may be beneficial to enhancing abundances of certain transient species (Harwell 2004). Creating additional patch reefs in a restoration area will lead to increased habitat heterogeneity, an important component of estuarine nursery habitat (Beck et al. 2001). Breitburg et al. (2000) suggest that created reefs where no structured habitat exists could serve as 'stepping stones' for migrating organisms. These patch reefs may provide a corridor between subtidal channels and high intertidal structure, reducing predation on fish utilizing shallow areas. Abundance data suggest little preference by most transients and xanthid crabs for habitat at particular distances from subtidal channels. Most mobile organisms used reefs at all distances; perhaps a result of the distances between patches being close (Gustafson and Gardner 1996) or the movement ability of the organisms overcoming the spatial scale and connectivity of the landscape (Kotliar and Weins 1990).

Patch reef usage gives evidence that transients may use patches as stepping stones during daily migration through the intertidal zone and that patchiness within a larger landscape may be more important than larger continuous habitat (Hovel and Fonseca 2005). Although the specific distances used in this study may not be limiting for most species, the presence of patch reefs between subtidal channels and high intertidal areas (such as marshes and larger fringing reefs) may provide suitable habitat and act as stepping stones for migration. These patch reefs supply edge habitat which may be crucial for growth and survival of transient species because this microhabitat provides access to refuge and foraging areas. A multi-species restoration effort may be enhanced by providing structured habitat, such as oyster reef patches, to organisms over the entirety of the intertidal range.

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