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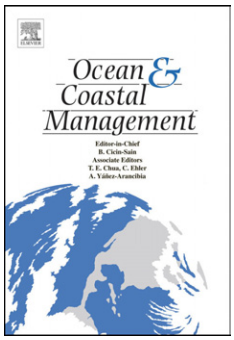

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FISH ASSEMBLAGES ON A MITIGATION BOULDER REEF AND NEIGHBORING HARDBOTTOM

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Highlights.

- Fishes on boulder reefs and natural hardbottom were counted annually for 5 years.
- Boulder mitigation reefs provided a habitat suitable for fish colonization.
- Fish richness and abundance were greater at mitigation reefs than on hardbottom.
- Fish assemblages on hardbottom and mitigation reefs had about 75% dissimilarity.
- Boulder reefs do not provide an equitable mitigation for hardbottom habitat loss.

ABSTRACT

We compared the fish assemblages on a mitigation site to neighboring natural habitat. Artificial reefs made of limestone boulders were deployed offshore Florida in August-September 2003 as mitigation for an anticipated nearshore hardbottom burial associated with a planned beach nourishment. Boulders comprising a footprint of 36,017 m^2 were deployed on sand substrate, adjacent to hardbottom, to replace an expected covering of 30,756 m^2 hardbottom. Nourishment of the beach was initiated May 2005 and completed in February 2006. Fishes on the artificial mitigation reefs and neighboring natural hardbottom were counted annually in August, 2004 through 2008, with 30-m belt transects and rover-diver surveys. Across all surveys a total of 18,313 fish of 185 species was counted. Mean species richness and abundance were typically greater on the transects at mitigation reefs than on nearshore hardbottom (NHB). MDS plots of Bray-Curtis similarity indices show a clear distinction between the mitigation reefs and NHB fish assemblages regardless if the data were, or were not, standardized to account for rugosity differences. SIMPER analysis indicated the two assemblages had, on average, 75% dissimilarity. Thus, while the mitigation boulders exhibited greater abundance and species richness than the NHB, the two assemblages differed dramatically in structure. The mitigation reefs provided a habitat suitable for fish colonization. However, this habitat differed dramatically in size and appearance from impacted NHB and created a unique environment unlike the NHB. Thus, mitigation reefs in general, and boulder reefs specifically, should not be relied upon to provide an equitable replacement to NHB habitat loss.

KEY WORDS

Artificial reef, coral-reef fishes, juvenile fishes, mitigation, beach, nourishment

1. INTRODUCTION

World-wide, beaches are important tourist destinations critical to local economies. For example, in the United States, about 75% of those with summer vacation travel plans include a visit to a beach (Houston, 2002). Beaches are constantly eroding due to poorly designed coastal defense structures (i.e. seawalls, jetties, groin fields), as well as by rising sea levels, hurricanes and other natural processes which alter the shoreline (Silberman and Klock, 1988; Wanless, 2009; Jordan et al., 2010). In Florida, beach tourism contributes more than $39 billion to the state's economy (Murley et al., 2005) and to ensure this tourism persists, an average of $20-40 million a year is spent on beach maintenance (Finkl, 1996) and larger amounts are spent periodically on beach nourishment projects. Beach nourishment (aka renourishment) is the term for the dredge-and-fill process of adding sand to a location where the natural shoreline has eroded. In Broward County, Florida, where this study was conducted, the total costs associated with the 2005-2006 beach nourishment exceeded $44 million.

Positive aspects of beach nourishment include an increase in: recreational activities and storm protection (Finkl et al., 1988; Silberman and Klock, 1988), property values, economic stimulation (Murley et al., 2005), flood control, and habitat for endangered species (Finkl, 1996). However, there are also negative aspects of beach nourishment. Sand is often collected from a marine borrow site, transported, and placed onto the recipient beach. This process has the potential to negatively impact natural hardbottom ecosystems at both sites. Nearshore habitat can become buried when additional sand is added, and increased sedimentation may occur as fill material is naturally redistributed to a more stable profile (National Research Council, 1995; Wanless and Maier, 2007; Jordan et al., 2010).

The nearshore hardbottom (NHB) habitat in much of southeast Florida is composed primarily of beachrock (Goldberg, 1973). The nearshore hardbottom ridge complex is a consistent structural feature throughout the area and is comprised of colonized pavement with rubble that contains variable sand cover dominated by encrusting zoanthids, alcyonacean corals, and macroalgae (comprising 13%, 12%, and 16% total cover, respectively) (Moyer et al., 2003; Walker et al., 2008). Although its topographical complexity is low relative to other reef habitats, this substrate contains many small holes and crevices, which are valuable habitat for cryptic species and juvenile fishes, and provides refuge and food items (Kobluk, 1988; Vare, 1991; Lindeman and Snyder, 1999; Baron et al., 2004; Banks et al., 2008).

There have been previous studies of the nearshore fishes in southeast Florida, although most of them have been unpublished (Lindeman et al., 2009). Baron et al. (2004) characterized nearshore fish assemblages in Broward County and found new settlers and early juveniles composed >84% of the nearshore fish community. Of these, >90% were grunts (Haemulidae). These juveniles are found in significantly higher abundance on nearshore hardbottom compared to other offshore reef tracts (Jordan et al., 2004; Ferro et al., 2005), further highlighting the importance of nearshore habitat. In Palm Beach County, Florida, a total of 118 fish species were observed on nearshore reefs (Vare, 1991). Once again, the most frequently occurring family was Haemulidae. Lindeman and Snyder (1999) also surveyed fish assemblages in Palm Beach County and found early-life stages (newly settled, early juvenile, and juvenile) represented >80% of individuals surveyed at three nearshore sites. Due to its proximity to shore, this NHB, which serves as

potential settlement and nursery areas for local coral-reef fishes, is vulnerable to the impact of beach nourishment (Lindeman et al., 2009). A 1995 beach restoration project in Jupiter, Florida, buried NHB habitat, reducing the number of fish species from 54 to 8 (Lindeman and Snyder, 1999).

In Broward County a beach nourishment project was initiated during 2005 to restore 11.1 km of shoreline. As in other beach nourishments, government agencies required that the adverse effects of surface water activity be mitigated (Florida Statute 373.414(1)(b)). One popular form of mitigation for the impact on NHB is the deployment of artificial reefs made of limestone boulders (Palmer-Zwahlen and Aseltine, 1994; Cummings, 1994; Yoshioka et al., 2004; Thanner et al., 2006). Yet there are few rigorous studies published on the effectiveness of this method. A study by Thanner et al. (2006), also in southeast Florida, did compare fishes on boulder reefs to neighboring hardbottom reef tracts. Although the artificial reefs were placed as mitigation for beach nourishment, their study sites were 3.1 km offshore at 20-m depth and the fish assemblages differ significantly between the NHB and the deeper offshore reef tracts (Ferro et al., 2005). Thus, the Thanner et al. (2006) study examined the compensatory attributes and not the equitability attributes of a mitigation effort.

The central question that our study was intended to address was: Are the mitigation reefs effective replacement for fish habitat buried by sand fill? This overarching question was parsed into multiple sub-questions: 1) Is there a difference in species richness (the number of species) between the mitigation reef and the natural hardbottom it replaces? 2) Is there a difference in specific species between the mitigation reef and the natural hardbottom it replaces? Are some species restricted to boulder reefs or to NHB? 3) Is there a difference in fish abundance (the total number of fishes, all species combined) between the boulder reef and the impacted hardbottom it is intended to mitigate? 4) Is there a difference in fish assemblage structure (a measure of the abundances of individual species) between the mitigation reef and the natural hardbottom it replaces? 5) Is the mitigation reef the correct size to replace the loss of fishes anticipated on the proposed, buried natural hardbottom? Would a larger or smaller artificial reef have been appropriate for mitigation?

2. Materials and Methods

2.1 Study Design

A series of artificial reefs made of limestone boulders were deployed offshore Hollywood Beach, Florida, USA in August-September 2003. They were placed as mitigation for anticipated nearshore hardbottom burial associated with a planned beach nourishment. Boulders comprising a footprint of 36,017 m^2 were deployed on sand substrate, adjacent to hardbottom, to replace an expected covering of a similar area of hardbottom (30,756 m^2). Nourishment of the beach was initiated in May 2005 and completed in February 2006. Fishes on the artificial, mitigation reefs and neighboring, natural hardbottom were counted annually in August, 2004 through 2008 (Fig 1).

The five questions above (see Introduction) were converted to testable hypotheses by the simple expedient of assuming no differences (questions 1-4) and that the mitigation reef is the correct size (question 5). In turn, to address these hypotheses, fish censuses were made on the mitigation reefs and natural hardbottom and the resulting data subjected to statistical analysis. Twenty-five transect counts and 25 rover-diver counts were done at the mitigation boulders and 25 transect-counts and 25 rover-diver counts were done at neighboring natural hardbottom sites. The same transect locations were used for all counts.

2.2 Transect count

The 30-m belt transects (60 m^2) were established parallel to the bottom and ignored any surface irregularities. Start and end points of the transect lines were marked with iron stakes and GPS coordinates were recorded. The topographical rugosity of each transect was determined by following the transect line from beginning to end with a fiberglass surveyor's tape and closely following the complex contours of the substrate. Comparison of the tape distance to the 30-m line yielded an estimate of gross rugosity (tape m/30 m = rugosity index) (Baron et al., 2004).

During each count a 30-m line was stretched as an orientation aid along the marked transect. The diver swam above the transect recording all fish within 1 m to either side and 1 m above the line. Species were recorded as well as abundance and total length (TL) (by size class: <2, 2-5, 5-10, 10-20, 20-30, 30-50 and >50 cm) as encountered. The diver carried a 1-m "T"-rod, with size classes demarcated, to aid in transect width and fish length estimation. Stretches of sand substrate along the transect (absence of hard substrate) greater than 3 m were also recorded. Each transect took approximately 10 minutes to complete but were not time delimited.

2.3 Rover-diver count

Rover-diver counts consisted of the diver recording all species encountered within a 30 m x 30 m quadrat during a 20-minute interval. The diver was encouraged to look wherever he or she pleased in an attempt to record the maximum number of species. No abundance or size data were recorded (Baron et al., 2004). Rover-diver counts were accomplished in an area bounded by: the transect line of the transect count, the western edge of hardbottom or the boulder tract, and a 30-m line laid directly north of the eastern end of the transect line (essentially a square, but somewhat variable depending on the hardbottom edge).

2.4 Statistical analysis

Transect and rover-diver data for total fish abundance (of each size class and all size classes combined) and total species richness per count were entered into a statistical program, Statistica (StatSoft Inc., Tulsa, OK, USA). Two-way analysis of variance (ANOVA) and a Student-

Newman-Keuls (SNK) test between means were primarily used for analyses of abundance and species richness. A nested-ANOVA was also used to examine the differences between boulders and natural hardbottom across years. The nearshore environment of Broward County provides settlement/juvenile habitat and the majority of fishes, based on size, are under a year old (Baron et al., 2003; also see below). Thus, repeated measures analyses were not used. Because abundance data exhibited a heteroscedastic, non-normal distribution, analyses of variance (ANOVAs) were performed using log(x+1) transformed data. Species richness data had normal distribution, thus, raw data were tested. A p-value <0.05 in ANOVA, and SNK tests was accepted as a significant difference. For examining assemblage structure, non-metric multi-dimensional scaling (MDS) plots were constructed using Bray-Curtis similarity indices based on log(x+1) transformed abundance data (PRIMER v6; Clarke and Warwick, 2001). Analysis of similarity (ANOSIM) was used to test if differences in assemblage structure were present between survey years and between NHB and boulders. An ANOSIM R-statistic <0.25 implies that assemblage structures are barely separable (Clarke and Warwick, 2001). The SIMPER analysis was used to identify those species contributing most to the dissimilarity between MDS clusters.

Rugosity was dramatically different between the mitigation boulder and NHB sites. The boulders had a higher rugosity index than the relatively flat natural hardbottom (mean ± SE: 1.45±0.02 versus 1.04±0.001, p<0.01, ANOVA). Thus, simply looking at areal coverage of a transect (or footprint, 60 m^2 in this case) may not provide an accurate picture of the substrate and attendant habitat available. For this reason all data for a given sampling interval were analyzed both without and with taking rugosity into account. The latter was accomplished by dividing the abundance data by the corresponding rugosity index prior to analysis.

3. RESULTS

In transect counts, a total of 17,992 fishes of 125 species was recorded. 11,592 fish of 108 species (34 families) were counted on the mitigation (boulder) reefs and 6,400 fishes of 93 species (34 families) were counted on the natural hardbottom. Of these counts, 21 species were found exclusively on natural hardbottom and 38 were found only on boulders (Table 1). Of the total fish counted, 51.0% of the boulder fishes and 77.7% of the hardbottom fishes were juveniles or small cryptic species (<5cm). Fishes on the boulders >5 cm TL comprised 49% of the total abundance compared to 22.3% on the natural hardbottom. Mean fish abundance on the transects was significantly higher on boulder than natural sites (Fig. 2). Juvenile haemulids accounted for 32.4% of total fish abundance on boulder sites and for 36.9% on natural hardbottom. The abundance data show a large increase in juveniles on both the mitigation boulders and the natural reefs in 2007 compared to other years (Table 2). This, in addition to unusually high variation in juvenile counts in 2007, likely reflects the temporal and spatial variability in recruitment. Mean species richness was greater on the 30 m transects at the boulder reefs than on the natural hardbottom (Fig. 2). The mean abundance of fishes by size classes varied considerably among years and between boulders and natural hardbottom (Table 2). Because we cannot be certain the separation between sites was sufficient to allow for fully independent replicates the results of the ANOVA and SNK analyses (Table 2) should be viewed as indicative of differences rather than absolute (Hurlbert, 1984). However, the pattern of

boulder reefs having larger total abundances and species richness values across years was consistent. Further, a nested-ANOVA of abundance, which would be less impacted by a lack of independence among samples within a treatment, likewise was significantly different between boulder reefs and NHB across years ($p < 0.01$).

The MDS plot of Bray-Curtis similarity indices shows a clear distinction between the boulder reefs and NHB assemblages (R=0.34, ANOSIM) (Fig. 3). SIMPER analysis indicated the two assemblages had average 75% dissimilarity (Table 3). Twelve species made up more than 50% of the total dissimilarity (Table 3). Juvenile haemulids contributed the most of all taxa (6.8%) to the overall dissimilarity between NHB and mitigation boulder reef (Table 3). In addition, *Haemulon aurolineatum* (>5 cm TL), *Thalassoma bifasciatum*, and *Anisotremus virginicus* were all found in higher abundances on the boulders (contributing 4.08%, 5.66%, and 4.58% to the dissimilarity, respectively) (Table 3).Thus, simply looking at areal coverage, the mitigation boulders provided more species and more fishes than the natural reef and the two assemblages differ dramatically in structure.

If rugosity is taken into account, mean fish abundance and richness on boulders show the same pattern of differences from the natural hardbottom and remain significantly different. Likewise, dividing individual species abundance by the rugosity index and re-running the Bray-Curtis indices produced a near-identical MDS-plot to non-standardized data (not shown) with a clear separation between boulders and natural hardbottom.

For rover-diver counts, across years: natural hardbottom yielded 148 species from 42 families and mitigation boulders yielded 152 species from 45 families. Of these counts, 32 species were found only on natural hardbottom and 37 were found only on boulders (Table 4).

4. DISCUSSION

The NHB and mitigation boulders exhibited major differences in fish assemblage structure. The combined high species richness from transects and rover-diver counts recorded in this study (185) indicates the high diversity present in the NHB environment. On average, across years, the species composition of the NHB assemblage differed by about 30% from the boulders and the boulders differed by 45% from NHB. For the entire study the hardbottom assemblage species differed by about 18% from the boulders, and the boulders differed by 30% from hardbottom. Interestingly, the rover-diver counts accounted for 28% more species than transect counts. This clearly indicates the importance of including the rover-diver technique when attempting to compare fish assemblages.

The statistical comparison of fish assemblages on NHB and mitigation boulder transects indicated substantial differences across years. All sampling intervals showed clear differences in species and size (TL) composition, as well as differences in mean abundance. Of the total fishes surveyed, more than 64% were counted on boulder reef transects. Likewise, a higher number of species were counted on boulder transects (108) versus natural transects (93). Rover-diver counts also recorded more species on boulders than NHB (152 versus 148). There is a large

temporal and spatial variation amongst counts in both richness and abundance. The differences in abundance are primarily driven by changes in the numbers of juveniles (<5 cm TL), especially grunts. Juveniles and small cryptic species made up on average 77.7% of the fishes on hardbottom transects (range: 59-89%) and 51% of the fishes on the boulders (range: 31-71%). The causes for the temporal differences amongst richness counts are not clear (Table 2). There is a correlation between species richness and abundance for August transects ($r^2 = 0.69$) and so the differences may be caused, in part, by physical dynamics affecting multi-species recruitment or density-dependence of prey. Whatever the causes, the high variation among counts in this study clearly highlights the dangers of drawing conclusions about inshore fish assemblages from limited data. One or two "snapshot" surveys are inadequate to characterize an assemblage, especially one dominated by juvenile fishes (Jordan and Spieler, 2006).

All years showed a clear distinction between fish assemblages associated with natural hardbottom and mitigation boulders on MDS plots; boulders are less variable than natural sites both with and without rugosity standardization factored in. The physical and biological differences of these environments help to create assemblage structures which are unique to their respective areas. The natural hardbottom consists of low-relief pavement (Walker et al., 2008) and contains many small crevices and refuge spaces, providing habitat for many juvenile and small cryptic fishes. The boulders, on the other hand, contain large overhangs and void spaces that are able to provide additional refuge for larger fishes. The higher abundances of >5 cm fishes, many of which are piscivores, on boulders may indicate the lower percentages of juveniles on these reefs are due, at least in part, to predation (Table 2).

After five years the boulder assemblages retained an almost 75% dissimilarity to the natural hardbottom. Boulders showed a more compact clustering across years, which is indicative of a more homogenous environment. They offer similar refuge space and surface area throughout all transects, allowing fish assemblages to remain similar. In contrast, natural hardbottom provides a more heterogeneous and dynamic environment (Goldsmith, 1991). To some extent, fish assemblages change along with changing microhabitats. In the nearshore environment, this is especially applicable to juvenile haemulid species. Juvenile haemulids were not only the most abundant taxon but also contributed the most of all taxa to the overall dissimilarity between NHB and mitigation boulder reef (Table 3). In addition, certain fish species found on the boulders were either present in extremely low abundances or absent altogether on the natural reef, i.e. *Carangoides ruber*, *Gerres cinereus*, *Acanthurus coeruleus*, *Archosargus rhomboidalis*, and *Lutjanus griseus*. Of these, two are piscivores and important predators of juvenile fish: *C. ruber* and *L. griseus* (Randall, 1967; Froese and Pauly, 2007). In general, the boulders contained more and larger predators than the natural habitat. The increase in predators on the boulders may impact the nearby nearshore natural population, and more research is needed to determine the overall effects of the boulders on neighboring assemblages (Webster, 2002).

Relative to the NHB, the results of this study are similar to a previous survey of nearshore fish assemblages conducted in Broward County (Baron, et al., 2004). In this study, a total of 185 species, 93 on the hardbottom transects and 148 with hardbottom rover-diver counts were recorded. Baron et al. (2004) reported 164 species total, with 118 on transects and 145 with rover-diver counts. Additionally they found that juvenile fishes comprised >88% of fishes on their transect surveys. In this study, 77.7% of fishes counted on natural transects were juveniles

(≤5 cm). However, transects in this study had a lower percentage of juvenile haemulids. Approximately 55.5% of juvenile fishes (on both NHB and Boulder transects combined) were haemulids, compared to >90% found previously (Baron et al., 2004). Baron et al. (2004) recorded fishes in the months of June through August, and thus some of the differences between studies may be due to temporal variation.

Thanner et al. (2006) characterized fish assemblages at natural reef sites on the offshore reefs in Miami-Dade County, Florida. They used these data to compare assemblage structures on nearby prefabricated modules of limerock boulder artificial reefs. Despite major differences in study design, after five years of study, they, likewise to the present study, found that fish assemblages on those natural and artificial reefs did not converge in similarity. There was also higher abundance on the boulder reefs. There are, however, differences in their results from this study. They found the natural reef had higher richness than the boulders and the assemblages on both natural and artificial reef sites were dominated by gobiids, with haemulids a distant second. In addition, they found greater variability in species richness on the boulder reefs than the natural sites. These differences are likely due to differences in site selection. The previous study was conducted at 20 m depth and the offshore reef tracts have higher species richness and lower abundance of haemulids than the nearshore hardbottom (Jordan et al., 2004; Ferro, 2005).

Beach nourishment took place between May 2005 and February 2006. Fish surveys conducted after the nourishment appeared to show some impact of this activity. In August 2006 there were seven sites that contained <5 fish per transect count on the natural hardbottom versus the preceding means of approximately 35 fish per transect. The reduced abundance on August 2006 transects may be due, in part, to beach nourishment activities. Sand and other sediment placed on the beaches from May 2005 to February 2006 had already begun shifting seaward onto NHB, likely intensified by the active hurricane season that southeastern Florida experienced during the summer of 2005. Hurricane Wilma made landfall in Broward County on 24 October, 2005, with sustained winds over 159 km/h. In turn, the newly nourished beaches of Broward County experienced minor beach and dune erosion (Clark and LaGrone, 2006). This likely contributed to a larger than normal influx of sand to the nearshore hardbottom habitat. During the August 2006 survey four transects were noted to have been heavily impacted by sediment (90–100% buried) and contained between 0 and 4 fish per transect. The August 2007 survey showed that there was some potential recovery of the nearshore environment, as only three sites remained totally buried. The August 2008 data showed that one site had recovered entirely but the other two sites remained buried (100% and 83% respectively).The re-exposure of buried sites demonstrates the dynamic nature of the nearshore habitat and sand transport, as well as how some areas were able to quickly rebound, in terms of fishes, from a burial event. The ephemeral nature of this hardbottom burial may be atypical, due in part to the grain size of the nourishment sand. (Wanless and Maier, 2007; Jordan et al., 2010).

The nearshore environment is an important habitat for many species of juvenile fishes that may use the nearshore environment as nursery habitat for recruitment and early development. Juvenile haemulid distribution has been extensively studied in Broward County, Florida (Jordan et al., 2004; Jordan, 2010). They exhibit both a pelagic larval stage and demersal juvenile and adult stage, and are highly abundant during the summer months (McFarland et al., 1985; Jordan et al., 2004). It is the transitional phase between their pelagic and reefal life stages, the post-

settlement phase (<2cm), in which the greatest difference in abundance is demonstrated when comparing NHB and mitigation boulder transects (Table 1).

As to questions stated in the Introduction: 1) There was a difference in species richness between the mitigation boulder reef and the NHB it was meant to replace. On transect counts, 93 species were seen on NHB compared to 108 species on boulder reef. With rover-diver counts 148 species were seen on NHB and 152 species on boulder reef. 2) There was a difference in specific species between the mitigation reef and the NHB it was meant to replace. Some species were present at one site and completely absent from the other. 3) There was a difference in fish abundance between the mitigation reef and the NHB it replaces. The boulders made up greater than 64% of the total abundance of fishes seen. 4) There was a difference in fish assemblage structure between the mitigation reef and the NHB. The two assemblages had, on average 75% dissimilarity. 5) In terms of simple richness and abundance the boulder reef was larger than habitat replacement required. The footprint, or areal coverage, of the boulder reef in this study produced almost two times the abundance and richness of fishes compared to the NHB. Clearly rugosity should be taken into account when planning mitigation reefs, simple footprint replacement can yield larger (and presumably smaller) assemblages than faunal replacement calls for.

With the substantial differences in assemblages noted here, the need for value judgment becomes apparent in evaluating boulder reef as effective mitigation. To provide a valid basis for such judgment, more research is required to obtain an understanding of the full ecosystem services provided by the natural habitat and the mitigation reef. The mitigation reef unquestionably provides a habitat that is suitable for fish colonization. However, this habitat differs dramatically in size and appearance from the area impacted and creates an environment that is not similar to that of the NHB. Different habitat characteristics produce different assemblages (Friedlander, and Parrish, 1998; Arena et al., 2007; Hackradt, 2011). Further, it is not clear what impact mitigation reefs have on the ecology of the sand habitat and what ecosystem services are altered at the site where they are deployed. It is noteworthy that the sand coverage of the nearshore hardbottom in the area of this study is ephemeral with transects being covered and uncovered. This may be due in part to the grain size of the nourishment sand (Wanless and Maier, 2007; Jordan et al., 2010). Nonetheless, when the hardbottom is buried fish species richness and abundance are reduced. However, these values are increased when the sand moves off the hardbottom and the substrate resources are once again available for colonization (i.e., refuge, invertebrate assemblage) (Spieler and Jordan, 2009). Consequently, from a fish perspective, mitigating for a seemingly transient acute impact with permanent, non-equitable artificial structure is questionable.

In sum, due to the difference in fish assemblages, the dynamic nature of nearshore sedimentation, sand transport, and a host of unknown biophysical impacts which may be associated with mitigation reefs, artificial reefs in general and boulder reefs specifically, should not be relied upon as an equitable fix to natural habitat loss. If the annual fish surveys initiated here continued over time, likely a more complete picture would emerge as to the steady-state fish assemblage and mitigative value of the boulder reef. However, at a minimum, other methods and technologies should be simultaneously pursued to find alternative approaches to hardbottom mitigation.

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Table 1. Fishes recorded on transect counts for all years with the total number of fish (T) and number of occurrences recorded (O) on both the natural hardbottom (N) and the mitigation boulders (B).

		2004		***2005***		***2006***		***2007***		***2008***	
Common Name	**Scientific Name**	**N T/O**	**B T/O**	**N T/O**	**B T/O**	**N T/O**	**B T/O**	**N T/O**	**B T/O**	**N T/O**	**B T/O**
Nurse shark	*Ginglymostoma cirratum*					1/1					
Southern stingray	*Dasyatis americana*				2/1						
Yellow stingray	*Urobatis jamaicensis*	2/2		1/1	1/1	1/1		2/2		2/2	
Tarpon	*Megalops atlanticus*			4/1	2/1						
Green moray	*Gymnothorax funebris*					1/1			1/1		
Inshore lizardfish	*Synodus foetens*							1/1			
Lizardfish species	*Synodus sp.*								1/1		
Sand diver	*Synodus intermedius*										2/2
Squirrelfish	*Holocentrus adscensionis*				1/1						1/1
Spotted scorpionfish	*Scorpaena plumieri*	2/2		2/2	1/1			1/1			1/1
Graysby	*Cephalopholis cruentata*								1/1		
Sand perch	*Diplectrum formosum*	61/22	11/7	33/11		6/5	2/1	10/4		2/1	
Red grouper	*Epinephelus morio*				1/1		1/1	1/1			
Butter hamlet	*Hypoplectrus unicolor*						1/1				
Scamp	*Mycteroperca phenax*				1/1				2/1		5/5
Greater soapfish	*Rypticus saponaceus*								1/1		1/1
Lantern bass	*Serranus baldwini*							1/1		1/1	
Harlequin bass	*Serranus tigrinus*							1/1			
Dusky jawfish	*Opistognathus whitehursti*			1/1		27/10	3/2	2/2		4/4	
Flamefish	*Apogon maculatus*	1/1									

Twospot cardinalfish	*Apogon pseudomaculatus*		1/1					1/1			
Yellow jack	*Carangoides bartholomaei*	1/1	6/2		3/2				3/1		1/1
Bar jack	*Carangoides ruber*		293/23		8/3		15/5	2/1	65/8		9/3
Blue runner	*Caranx crysos*	59/3	30/1					26/2			
Crevalle jack	*Caranx hippos*				1/1						
Greater amberjack	*Seriola dumerili*		3/2		3/2						
Mutton snapper	*Lutjanus analis*							2/2		1/1	
Schoolmaster	*Lutjanus apodus*						1/1				
Gray snapper	*Lutjanus griseus*	3/2	18/11	1/1	28/10		9/3	3/1	8/6	1/1	4/4
Mahogany snapper	*Lutjanus mahogoni*						1/1				
Lane snapper	*Lutjanus synagris*	150/22	37/15	127/23	39/15	71/18	41/20	65/11	22/12	16/10	1/1
Yellowtail snapper	*Ocyurus chrysurus*	12/6	6/6	39/16	5/5	4/3	4/4	22/13	9/4	11/7	7/5
Slender mojarra	*Eucinostomus jonesii*		5/1		15/1			3/1	3/1		
Mottled mojarra	*Eucinostomus lefroyi*	1/1									
Yellowfin mojarra	*Gerres cinereus*		50/17		7/5		14/7	1/1	75/15	7/1	70/18
Black margate	*Anisotremus surinamensis*		4/4		7/7		3/3		4/4		2/1
Porkfish	*Anisotremus virginicus*	6/2	49/22	12/2	68/22	3/3	93/23	9/3	57/23	2/2	75/19
White margate	*Haemulon album*				1/1						2/2
Tomtate	*Haemulon aurolineatum*	160/8	469/23	1/1	242/16	55/3	96/8	17/4	843/12		13/4
Caesar grunt	*Haemulon carbonarium*		1/1								
Smallmouth grunt	*Haemulon chrysargyreum*	1/1							9/1		1/1

French grunt	*Haemulon flavolineatum*	36/5	71/14	6/2	61/16	5/1	158/23	32/4	181/20	4/2	197/23
Spanish grunt	*Haemulon macrostomum*		1/1				2/2		2/2	1/1	1/1
Sailor's choice	*Haemulon parra*	30/1	6/6	1/1	11/9	6/1	4/4		3/2		8/5
White grunt	*Haemulon plumierii*	17/7	49/22	12/6	53/22	19/9	55/18	15/4	41/12	10/6	24/14
Bluestriped grunt	*Haemulon sciurus*	8/7	14/11	6/4	8/5	1/1	9/4	4/3	9/7	5/2	13/9
Juvenile grunts	*Haemulon* spp.	201/9	119/9	364/17	659/13	147/8	317/11	1314/17	2097/7	437/13	386/9
Striped grunt	*Haemulon striatum*					13/1			1/1		
Sea bream	*Archosargus rhomboidalis*		31/15		9/8		4/3		2/1		5/3
Saucereye porgy	*Calamus calamus*						3/1				
Porgy species	*Calamus* spp.			1/1							
Silver porgy	*Diplodus argenteus*		1/1				5/3				
Spottail pinfish	*Diplodus holbrookii*				4/2						2/2
Pinfish	*Lagodon rhomboides*				2/2		1/1		2/1		3/1
Reef croaker	*Odontoscion dentex*		10/1				5/3				1/1
Highhat	*Pareques acuminatus*	5/3	5/3	11/9	1/1	4/3	1/1	14/7	7/3	13/6	5/5
Yellow goatfish	*Mulloidichthys martinicus*		3/2						5/3		1/1
Spotted goatfish	*Pseudupeneus maculatus*		2/2		2/2			2/1		7/6	5/5
Bermuda sea chub	*Kyphosus sectator*	5/1			10/4		3/3	2/1	4/1		3/3
Spotfin butterflyfish	*Chaetodon ocellatus*		1/1	1/1						2/2	2/2
Reef butterflyfish	*Chaetodon sedentarius*	1/1	1/1								1/1
Blue angelfish	*Holacanthus bermudensis*			1/1	1/1				2/2		3/3
Queen angelfish	*Holacanthus ciliaris*		1/1		5/4		3/3	1/1	8/7	1/1	10/8
Rock beauty	*Holacanthus tricolor*							2/2	1/1		

Gray angelfish	*Pomacanthus arcuatus*		8/6		8/6	1/1	7/5		6/5		8/8
French angelfish	*Pomacanthus paru*	3/3	4/4	2/2	7/4			4/4	4/4	2/2	6/6
Sergeant major	*Abudefduf saxatilis*	34/9	50/12	21/7	14/5	63/3	60/15	33/8	25/8	14/8	20/10
Blue chromis	*Chromis cyanea*		1/1								
Yellowtail damselfish	*Microspathodon chrysurus*								5/4		
Dusky damselfish	*Stegastes adustus*	6/2	21/13	4/3	20/11	7/6	20/14	22/9	27/17	15/6	65/23
Longfin damselfish	*Stegastes diencaeus*			5/2		1/1		4/1			
Beaugregory	*Stegastes leucostictus*	40/15	14/9	11/9	19/15	30/10	44/19	28/14	12/9	21/10	90/24
Bicolor damselfish	*Stegastes partitus*	1/1		1/1			2/2	10/5	5/4	6/4	2/2
Threespot damselfish	*Stegastes planifrons*				1/1		5/4		2/2		
Damselfish species	*Stegastes* sp.			1/1							
Cocoa damselfish	*Stegastes variabilis*	71/22	55/19	26/11	28/16	31/13	56/20	31/13	34/17	39/12	99/23
Spanish hogfish	*Bodianus rufus*		1/1				3/3				
Slippery dick	*Halichoeres bivittatus*	276/25	170/25	83/18	75/22	99/16	144/22	304/22	116/20	159/23	145/22
Clown wrasse	*Halichoeres maculipinna*	1/1	2/1	1/1	3/3	11/7	4/4	23/7	34/9	7/5	32/12
Blackear wrasse	*Halichoeres poeyi*		1/1	1/1		1/1		28/8	28/4	7/3	
Puddingwife	*Halichoeres radiatus*				2/2		1/1		1/1		1/1
Hogfish	*Lachnolaimus maximus*				1/1		4/3		3/1		
Bluehead	*Thalassoma bifasciatum*	8/3	92/24	7/3	52/16	7/2	80/19	43/7	212/25	7/5	228/25
Rosy razorfish	*Xyrichtys martinicensis*	1/1				2/2		1/1		1/1	
Green razorfish	*Xyrichtys splendens*	6/4		1/1		3/3					
Razorfish species	*Xyrichtys* spp.					2/1					

Parrotfish species	*Scaridae* spp.										7/4
Midnight parrotfish	*Scarus coelestinus*						1/1				
Rainbow parrotfish	*Scarus guacamaia*						2/2		1/1		8/2
Striped parrotfish	*Scarus iseri*	9/2	9/5		2/1		16/6	9/1	15/4	4/2	13/7
Princess parrotfish	*Scarus taeniopterus*				6/2		1/1	1/1			
Redband parrotfish	*Sparisoma aurofrenatum*	14/4	5/4			6/2	16/11	18/8	19/7	6/1	9/4
Bucktooth parrotfish	*Sparisoma radians*	58/16	38/17	24/14	3/2	11/7	8/7	54/11	16/10	38/13	26/11
Redfin parrotfish	*Sparisoma rubripinne*	2/2	5/3		2/2			1/1	3/1		4/4
Stoplight parrotfish	*Sparisoma viride*		6/5	1/1	4/4		26/16	1/1	17/9	1/1	25/14
Roughhead triplefin	*Enneanectes boehlkei*					1/1	1/1				
Rosy blenny	*Malacoctenus macropus*	14/9	14/8	13/8	1/1	13/6	1/1	56/17	2/2	38/14	3/2
Saddled blenny	*Malacoctenus triangulatus*			2/2				2/2		2/1	
Banded blenny	*Paraclinus fasciatus*		1/1								
Roughhead blenny	*Acanthemblemaria aspera*		6/3	2/2	3/2				6/4	2/2	5/5
Sailfin blenny	*Emblemaria pandionis*	13/5		5/3		11/8		3/1		6/4	1/1
Seaweed blenny	*Parablennius marmoreus*	7/5	14/8	13/8	3/3	7/6	10/7	19/12	5/3	16/9	6/6
Lancer dragonet	*Callionymus bairdi*					2/1					
Colon goby	*Coryphopterus dicrus*	2/2									
Bridled goby	*Coryphopterus glaucofraenum*	51/9	22/12	9/5	2/2	22/9	23/13	46/16	12/7	13/6	64/21
Masked goby	*Coryphopterus personatus*						1/1		6/1		51/8

Dash goby	*Ctenogobius saepepallens*	2/2		2/1							
Tiger goby	*Elacatinus macrodon*			1/1	1/1		1/1		2/2		5/5
Neon goby	*Elacatinus oceanops*				2/2	2/1	1/1		1/1		1/1
Goldspot goby	*Gnatholepis thompsoni*							2/2			
Seminole goby	*Microgobius carri*			3/1	2/1						
Blue goby	*Ptereleotris calliura*	5/4		1/1	1/1			1/1		7/4	
Hovering goby	*Ptereleotris helenae*									2/1	
Atlantic spadefish	*Chaetodipterus faber*				2/1		1/1				3/2
Ocean surgeon	*Acanthurus bahianus*	4/3	81/22	22/8	50/15	2/2	54/14	29/7	95/18	24/10	184/25
Doctorfish	*Acanthurus chirurgus*	10/5	25/12	14/9	51/16	2/2	15/10	31/9	59/17	6/4	46/16
Blue tang	*Acanthurus coeruleus*	1/1	10/9		17/11		17/10	4/2	37/18		40/20
Great barracuda	*Sphyraena barracuda*								2/2		
Spanish mackerel	*Scomberomoros maculatus*								1/1		
Cero	*Scomberomorus regalis*		1/1								
Peacock flounder	*Bothus lunatus*			1/1		1/1					
Gray triggerfish	*Balistes capriscus*	6/3	21/12	10/5	19/10	6/4	15/8	3/3	7/5	6/4	15/10
Slender filefish	*Monacanthus tuckeri*							1/1			
Planehead filefish	*Stephanolepis hispidus*	1/1									
Scrawled cowfish	*Acanthostracion quadricornis*				1/1		1/1		3/3		3/3
Spotted trunkfish	*Lactophrys bicaudalis*							1/1			1/1
Smooth trunkfish	*Lactophrys triqueter*		1/1			1/1	3/2		3/2	1/1	3/3
Sharpnose puffer	*Canthigaster rostrata*			1/1	10/7	1/1	6/5	4/3	17/11	8/6	31/15
Bandtail puffer	*Sphoeroides spengleri*			2/2		1/1		1/1		1/1	2/2
Balloonfish	*Diodon holocanthus*	1/1		3/3	3/2	1/1	6/4		2/1	4/1	6/5

Porcupinefish	*Diodon hystrix*	1/1									
	Total Abundance	**1407**	**1968**	**916**	**1677**	**713**	**1510**	**2374**	**4314**	**990**	**2123**
	Total Species	**48**	**56**	**49**	**65**	**45**	**63**	**60**	**68**	**48**	**69**
	SIMPER % difference	**69.38**		**76.67**		**78.46**		**77.03**		**77.0**	

Table 2. Mean (± 1 SEM) fish abundance (by total length in cm) and species richness for counts by year on natural hardbottom (N) and mitigation boulder (B) on 25 30-m transects. Mean (± 1 SEM) total abundance and richness is provided in raw data as well as standardized for rugosity. Means in bold differ within a year (ANOVA, SNK P<0.05)

	2004		*2005*		*2006*		*2007*		*2008*	
SIZE (cm TL)	N	B	N	B	N	B	N	B	N	B
0-2	1.1±0.5	0.3±0.2	11.7±4.8	12.8±8.2	6.3±2.9	5.6±1.6	**44.2±19.6**	**3.6±1.7**	2.5±1.4	8.5±5.3
2-5	32.2±4.4	24.1±5.5	18.8±2.6	20.6±5.4	16.7±3.1	31.0±5.4	40.6±6.8	118.6±102.3	27.1±6.1	28.4±7.2
<5	33.3±4.6	24.4±5.5	30.5±6.2	33.4±9.8	23.0±4.5	36.6±6.3	84.8±23.6	122.2±102.3	29.6±6.6	36.9±8.5
5-10	**18.3±3.8**	**40.6±5.3**	**4.0±0.7**	**17.0±3.2**	**4.8±2,7**	**14.3±2.5**	**7.8±1.1**	**30.0±12.8**	**8.5±1.6**	**24.2±2.8**
10-20	**3.7±1.4**	**11.7±1.5**	**1.7±0.7**	**13.6±3.1**	**0.4±0.2**	**8.9±1.8**	**1.0±0.3**	**18.0±2.6**	**2.3±1.0**	**22.3±2.6**
20-30	1.1±0.7	2.2±0.4	**0.3±0.1**	**2.7±0.8**	0.1±0.1	0.5±0.2	1.3±1.0	1.9±0.4	**0.4±0.3**	**1.9±0.5**
30-50	0	0	0	0.2±0.1	01±0.1	0	0.1±0.1	0.3±0.1	0	0.3±0.1
50+	0	0	0.2±0.2	0.1±0.1	0	0	0	0.1±0.7	0.2±00	0.2±0.7
Total	56.4±5.6	78.9±8.4	**36.7±7.0**	**67.1±11.6**	**36.9±9.2**	**60.4±6.2**	95.0±24.2	172.0±115.2	**39.6±7.8**	**84.9±9.6**
Rug/Std	53.6±5.4	53.7±5.2	35.2±6.7	46.6±8.3	35.8±9.0	43.5±4.4	92.6±23.7	122.4±79.8	**39.0±7.7**	**60.2±6.7**
Species	**10.7±0.5**	**18.5±0.9**	**9.4±0.8**	**15.0±0.8**	**8.2±1.1**	**16.6±0.8**	**11.8±1.3**	**16.9±0.7**	**9.3±1.0**	**20.7±1.0**
Rug/Std	**10.2±0.5**	**12.07±0.6**	9.1±0.8	10.3±0.6	**7.9±1.0**	**11.9±0.5**	11.5±1.2	12.3±0.6	**9.2±1.0**	**14.8±0.7**

Table 3. SIMPER analysis of dissimilarity showing the percent contribution of each species for August 2004-2008 between the natural hardbottom (N) and the mitigation boulders (B). The average dissimilarity was 75.4%.

Species	***Group N Av.Abund***	***Group B Av.Abund***	***Contrib%***	***Cum.%***
Haemulon spp.	1.43	1.16	6.75	6.75
Thalassoma bifasciatum	0.21	1.53	5.66	12.42
Haemulon flavolineatum	0.17	1.23	4.86	17.28
Anisotremus virginicus	0.1	1.13	4.58	21.85
Acanthurus bahianus	0.26	1.22	4.51	26.36
Halichoeres bivittatus	1.7	1.58	4.3	30.66
Haemulon aurolineatum	0.3	1.08	4.08	34.74
Lutjanus synagris	1.02	0.54	3.58	38.33
Stegastes variabilis	0.71	0.95	3.3	41.62
Haemulon plumierii	0.28	0.81	3.22	44.85
Stegastes leucostictus	0.49	0.67	2.93	47.77
Acanthurus chirurgus	0.24	0.68	2.73	50.51

Table 4. Fishes recorded only on rover diver counts for all years with the number of occurrences seen on both the natural hardbottom (N) and the mitigation boulders (B).

		2004		2005		2006		2007		2008	
Common Name	**Scientific Name**	**N**	**B**	**N**	**B**	**N**	**B**	**N**	**B**	**N**	**B**
Lesser electric ray	*Narcine brasiliensis*							2			
Spotted eagle ray	*Aetobatus narinari*	1									
Goldentail moray	*Gymnothorax miliaris*							1			
Goldspotted eel	*Myrichthys ocellatus*				1		1				
Snakefish	*Trachinocephalus myops*									2	
Blackbar soldierfish	*Myripristis jacobus*	1									
Trumpetfish	*Aulostomus maculatus*				1						1
Barbfish	*Scorpaena brasiliensis*			1							
Plumed scorpionfish	*Scorpaena grandicornis*		1								
Common snook	*Centropomus undecimalis*		1		9		4		5		5
Coney	*Cephalopholis fulva*	1									
Rock hind	*Epinephelus adscensionis*						1		1	1	1
Red hind	*Epinephelus guttatus*							1			
Gag	*Mycteroperca microlepis*				2		3				
Whitespotted soapfish	*Rypticus maculatus*									1	2
Banded jawfish	*Opistognathus macrognathus*					1					
Barred cardinalfish	*Apogon binotatus*					1					
Conchfish	*Astrapogon stellatus*		1								
Sharksucker	*Echeneis naucrates*						1		1		
Mackerel scad	*Decapterus macarellus*								1		
Round scad	*Decapterus punctatus*		3						1		
Leatherjack	*Oligoplites saurus*			1							
Almaco jack	*Seriola rivoliana*				2		1				2
Blackfin snapper	*Lutjanus buccanella*			1			1				
Yellowtail snapper	*Ocyurus chrysurus*	17	11	22	13	10	10	16	9	20	15
Vermilion snapper	*Rhomboplites aurorubens*						3				
Tripletail	*Lobotes surinamensis*					1					
Flagfin mojarra	*Eucinostomus melanopterus*							1			
Cottonwick	*Haemulon melanurum*	2					1			1	
Pigfish	*Orthopristis chrysoptera*		7		2				2		
Boga	*Inermia vittata*								1		
Sheepshead porgy	*Calamus penna*							1	1	1	
Foureye butterflyfish	*Chaetodon capistratus*						1	2			1
Banded butterflyfish	*Chaetodon striatus*	1						1			
Townsend angelfish	*Holacanthus* sp.						1				

Brown chromis	*Chromis multilineata*								1		
Dwarf wrasse	*Doratonotus megalepis*			1							
Bluelip parrotfish	*Cryptotomus roseus*							1	1	1	
Greenblotch parrotfish	*Sparisoma atomarium*			2	4	1	1	1	2	2	
Redtail parrotfish	*Sparisoma chrysopterum*		4			1	4	1	5		4
Lofty triplefin	*Enneanectes altivelis*								1		
Downy blenny	*Labrisomus kalisherae*		1					1			
Hairy blenny	*Labrisomus nuchipinnis*										2
Molly miller	*Scartella cristata*					1					
Pallid goby	*Coryphopterus eidolon*			1	1						
Banner goby	*Microgobius microlepis*			2					1		
Orangespotted goby	*Nes longus*						1				
Spanish mackerel	*Scomberomorus maculatus*	1					1			1	2
Peacock flounder	*Bothus lunatus*					1		2			
Ocean triggerfish	*Canthidermis sufflamen*					1					
Orange filefish	*Aluterus schoepfii*	1									
Orangespotted filefish	*Cantherhines pullus*							1		2	
Slender filefish	*Monacanthus tuckeri*					1					
Honeycomb cowfish	*Acanthostracion polygonius*		1				1			1	
Spotted trunkfish	*Lactophrys bicaudalis*		2				1		2		1
Trunkfish	*Lactophrys trigonus*				2		1	1			1
Striped burrfish	*Chilomycterus schoepfii*			1							

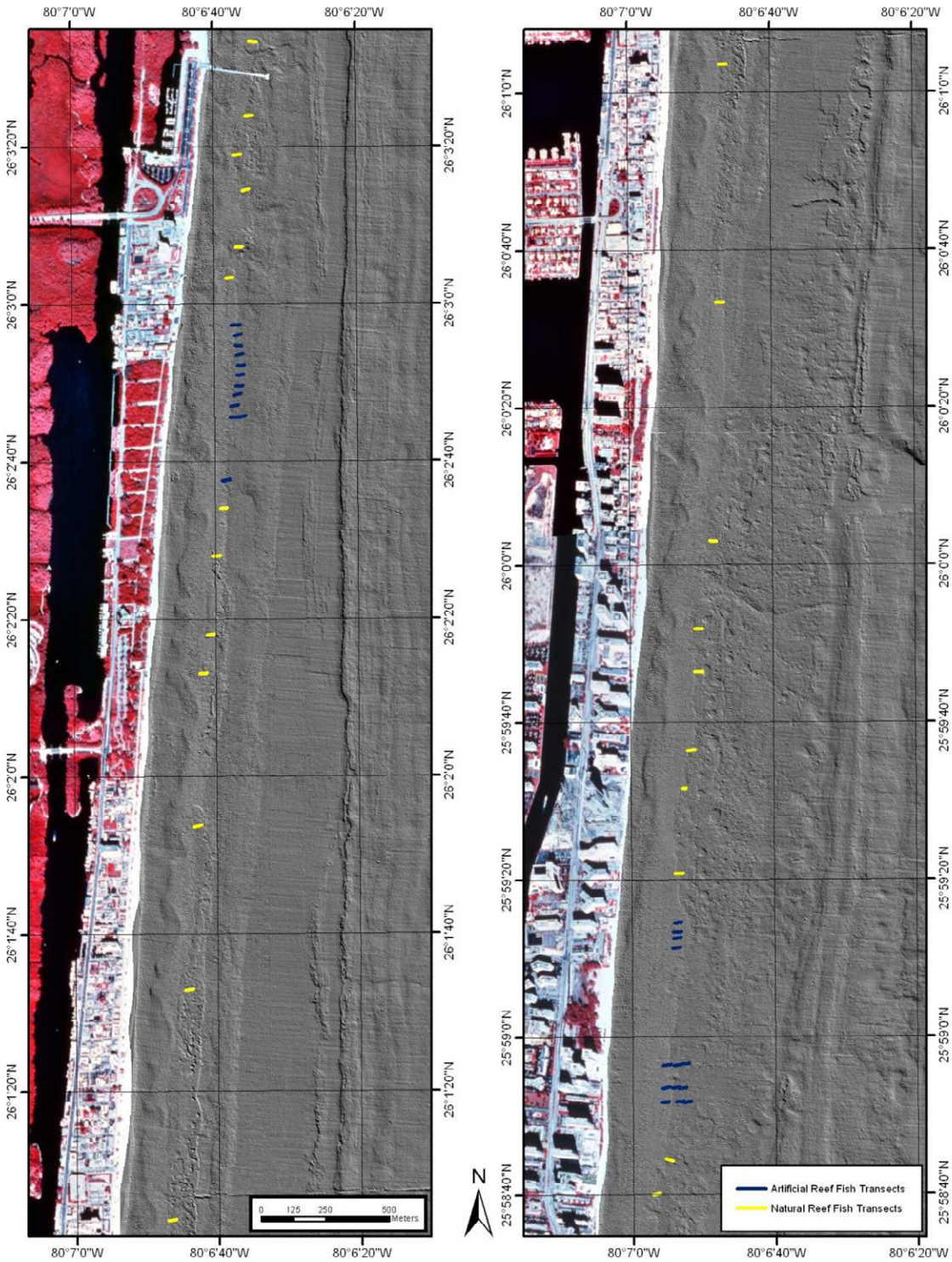

Figure 1 Laser Airborne Depth Sounding (LADS) image showing the 25 artificial reef transects (blue) and 23 of 25 (2 transects are outside the range of these photos) natural reef transects (yellow) surveyed.

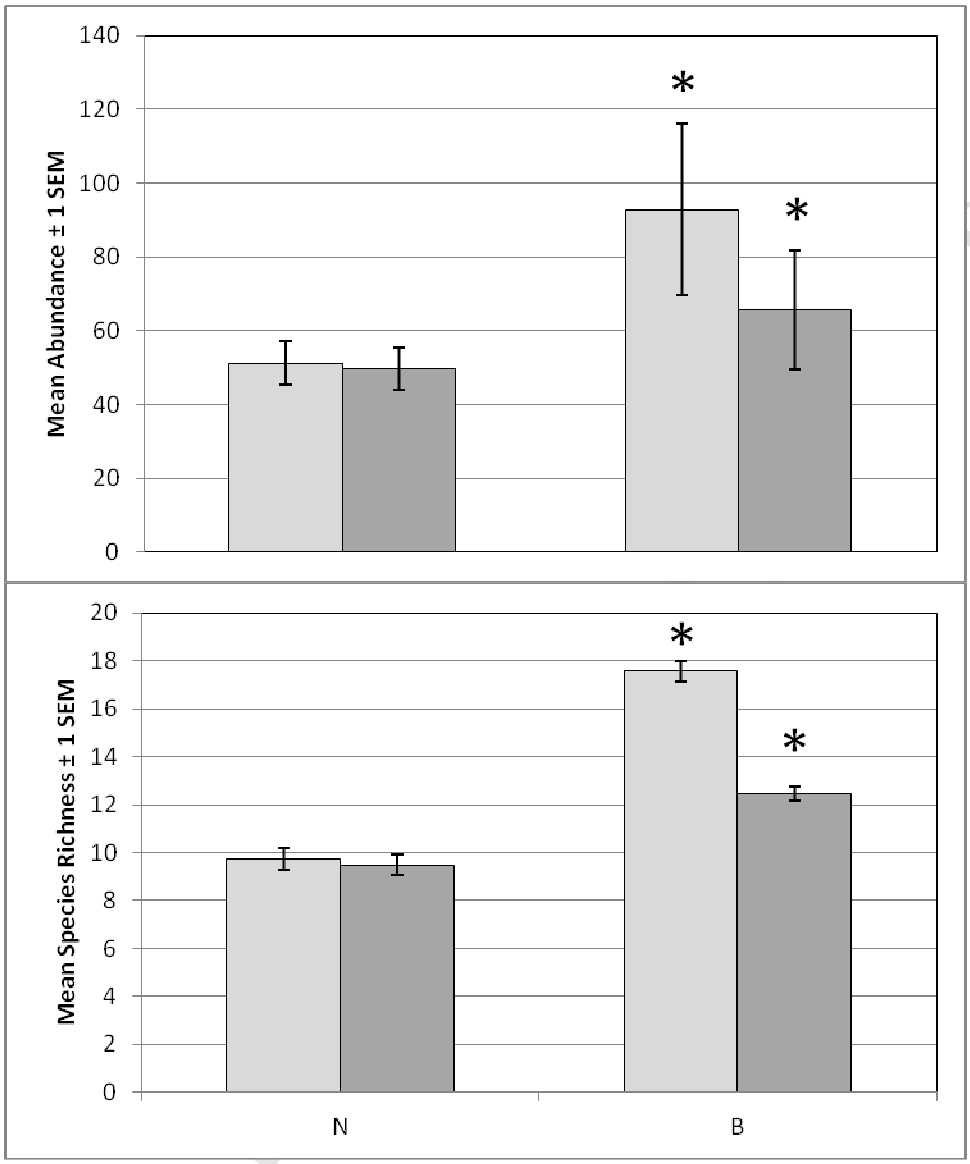

Figure 2. Mean total abundance and species richness (± 1 SEM) of fishes (August 2004-2008) on 25 transects of natural hardbottom (N) and mitigation boulders (B) without (light grey) and with rugosity standardization (dark grey). The asterisks indicate significant differences ($p<0.05$: ANOVA; SNK) in species richness between bars of the same color.

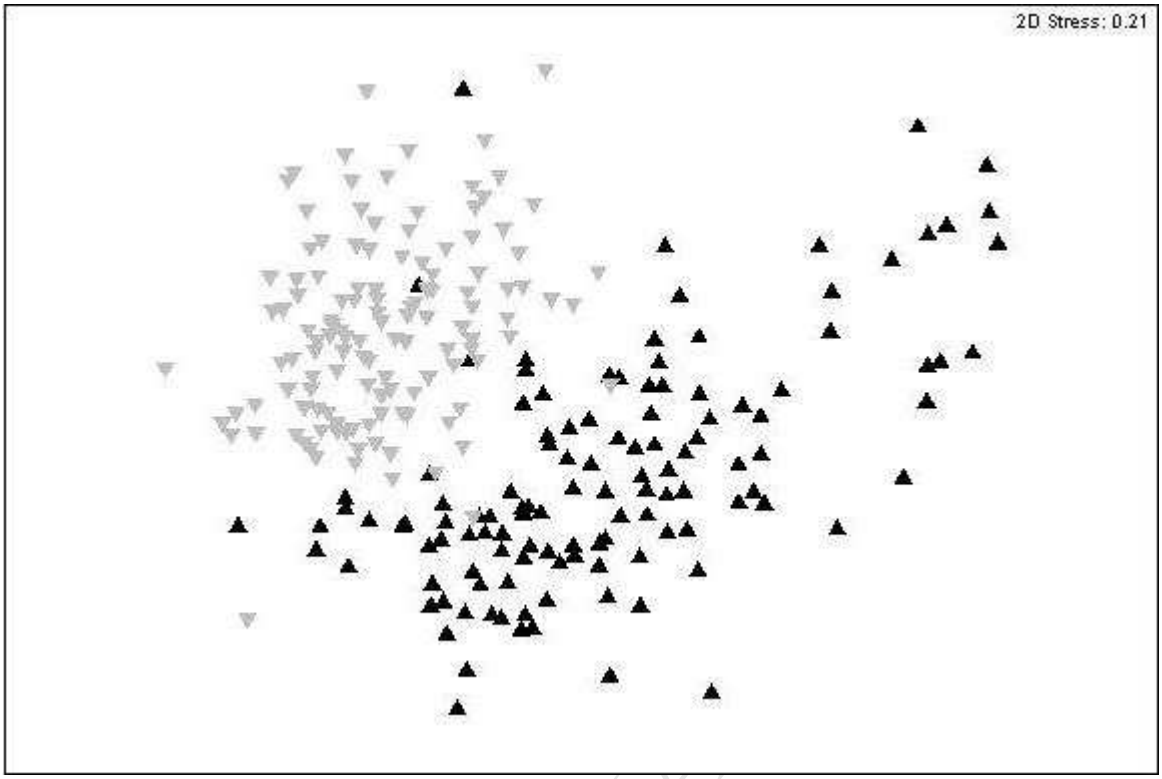

Figure 3. MDS plot (across all surveys) of Bray-Curtis similarity indices for the natural hardbottom (dark triangle) and the mitigation boulders (light triangle) not standardized for rugosity. ANOSIM between N and B: Global R=0.491