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Implications of natural variation of fish assemblages to coral reef management

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Abstract In order to evaluate the impact of a planned beach renourishment project we obtained baseline data on hardbottom coral reef fish assemblages adjacent to the beach. During the summers of 2001 and 2003 we conducted 92 and 89 visual transect-counts, respectively, each 30x2x1m. Fish were recorded by species, abundance, and size class. Fish exhibited a 62.8% decrease in abundance between the 2001 and 2003 surveys. Juvenile *Haemulon* spp. (<5cm total length, TL) alone decreased 72.8% in abundance and, due to their predominance, primarily drove the decrease seen for total abundance. The second most abundant species *Halichoeres bivittatus*, also primarily juveniles (<5cm TL), contributed 8% of the total abundance and also showed a significant decrease between the 2001 and 2003 surveys. Even after removal of the two most abundant taxa, total abundance remained significantly lower the second survey year, suggesting the possibility of a community-wide decrease in abundance. However, the decrease between years was mainly due to a decrease in juveniles. When fish less than 5cm TL were removed from the dataset, no significant difference in abundance between years was detected. Species richness also declined significantly with fewer species noted in 2003 and eight fewer total species between years. Nonetheless, multivariate examination of assemblage structure did not indicate a difference between years. These results have important implications for determining potential anthropogenic change in fish assemblages (e.g. caused by beach renourishment).

Keywords coral reef, fish, beach renourishment, assessment, census, survey, environmental impact

Introduction

Detection of non-catastrophic anthropogenic change of an environmental resource requires pre-disturbance baseline data. To this end, ecologists and marine resource managers will routinely attempt to acquire baseline data on a resource (e.g. fishes, corals) prior to a project that may impact it. Typically, this acquisition is done directly before a planned impact (Lindeman and Snyder 1999). In many cases sufficient spatial replication is obtained to

insure the database represents an adequate “snapshot” of the resource of interest. However, contrary to the implicit and explicit warnings in several publications, temporal replication is often ignored, presumably due to project deadlines, funding or logistics (for references see: Wilber et al. 2003; Underwood and Chapman 2003).

We conducted a baseline survey of coral reef fishes associated with nearshore hardbottom habitat in preparation for a planned beach renourishment (the placement of sand fill, dredged offshore, on eroded beach) with the intention of examining possible change in the fish assemblage correlated with the renourishment project (Baron et al. 2004). However, the project was postponed for several years providing the opportunity to resurvey the area to obtain a measure of temporal (year-to-year) variation of the fish assemblage.

Materials and Methods

We examined variation of coral reef fish assemblages on the nearshore hardbottom of Broward County, Florida, USA twice in three years prior to an anticipated beach renourishment project. As part of a larger study, in the summer of 2001, we performed 92 (30x2x1m) visual transect-counts, at 152m intervals, along a 13-km stretch of coastline (Baron et al. 2004) (Fig. 1). Each transect ran west to east across the north-south oriented hardbottom, beginning at the nearshore edge. Differential Global Positioning System (DGPS) coordinates were documented and each species, with its abundance, and total lengths (TL), was recorded. In the summer of 2003 we returned to the sites (using the DGPS coordinates) and repeated the census using the same methods.

Since abundance data exhibited a heteroscedastic, non-normal distribution, analyses of variance (ANOVAs) were performed on $\log(x+1)$ transformed data. Species richness data were normal, thus, raw data were tested. A non-metric multi-dimensional scaling (MDS) plot was constructed of Bray-Curtis similarity indices using fourth-root transformed abundance data (PRIMER v5; Clarke and Warwick 1994). Analysis of similarity (ANOSIM) was used to test if differences in assemblage structure were present between survey years. An ANOSIM R-statistic <0.25 implies that assemblage structures are barely separable (Clarke and Gorley 2001).

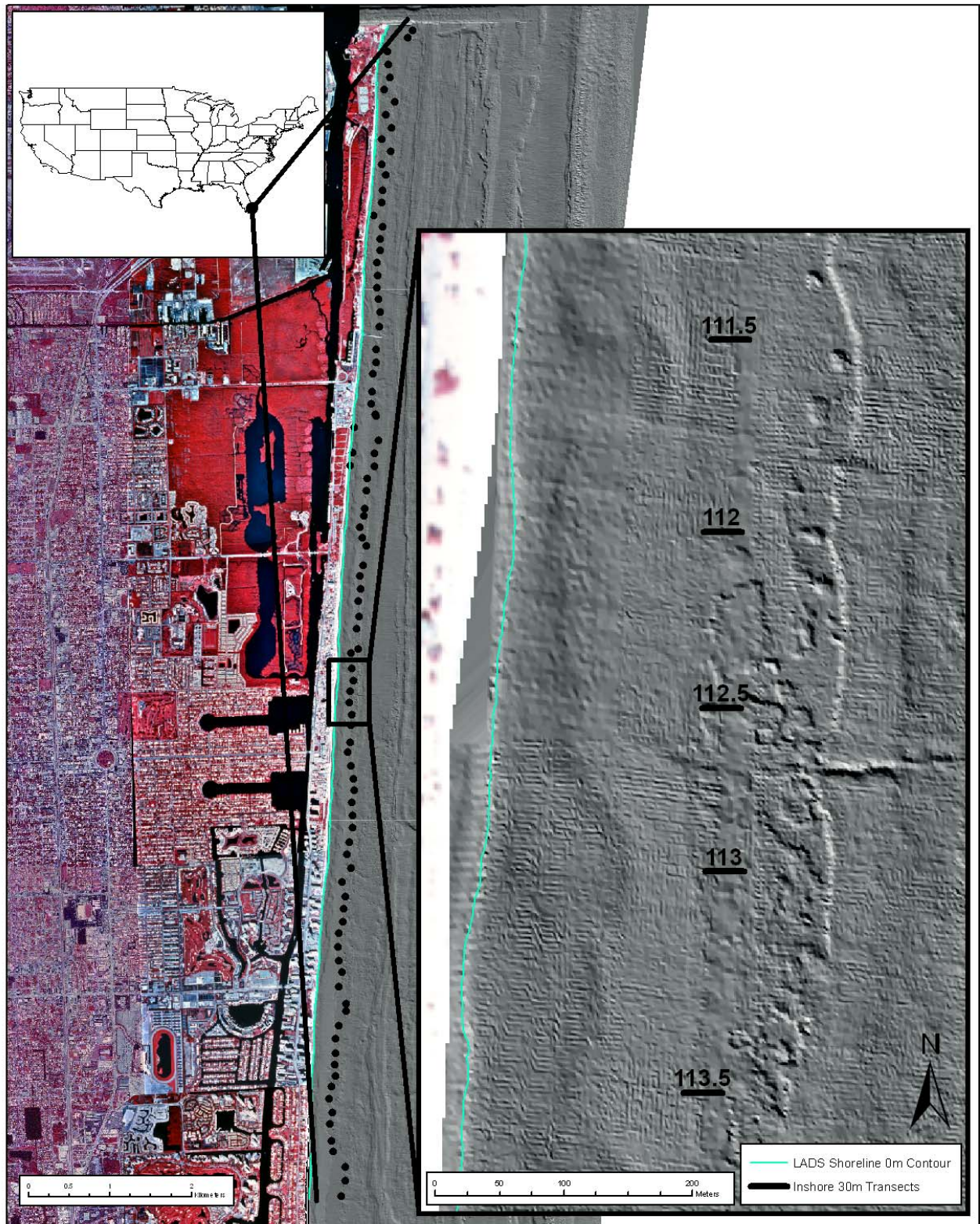


Figure 1. Laser Airborne Depth Sounding (LADS) image of study area and transect sites, southern Broward County, Florida, USA.

Results and Discussion

Coral reef fish assemblages exhibit spatial and temporal fluctuations of abundance and species occurrence. The variability is principally attributed to

immigration and emigration and post-recruitment processes. Although only two annual surveys (summer 2001 and 2003) of the nearshore hardbottom were conducted, total fish abundance showed a 62.8%

decrease (Table 1). Juvenile (<5cm TL) *Haemulon* spp. (Haemulidae) comprised over 75% of the total abundance in 2001 and 55% in 2003. The 72.8% decrease in abundance demonstrated by this genus (Table 1) primarily drove the decrease seen for total abundance. The second most abundant species *Halichoeres bivittatus* (Labridae), also primarily juveniles (<5cm TL), contributed 8% of the total abundance and also showed a significant decrease between the 2001 and 2003 surveys (Table 1). The third and fourth most abundant species were adult (>5cm TL) *Haemulon plumierii* and *Haemulon aurolineatum*, contributing 3% and 2% to the total abundance, respectively. Neither species exhibited significant declines in abundance between years. Even after removal of the two most abundant taxa, total abundance remained significant lower the second survey year (2003) (Table 1), suggesting the possibility of a community-wide decrease in abundance. However, the decrease between years was mainly due to a decrease in juveniles. When fish less than 5cm TL were removed from the analysis, no significant difference in abundance between years was detected (Table 1). There was also a significant decrease in the mean species richness per count (Table 1), with slightly fewer species noted in 2003 (eight fewer total species). Despite the apparent changes suggested by univariate analyses, multivariate examination of assemblage structure of the entire dataset (MDS plot of Bray-Curtis similarity indices and ANOSIM) did not indicate a difference between years (Fig. 2).

Because the inter-year variation in abundance noted here was chiefly due to a difference in juvenile fish abundance, a simple solution to reducing natural variations in pre- and post-impact comparisons would be to eliminate juveniles from analysis. However, in many cases, including the nearshore hardbottom in this study, the area of impact may be juvenile or nursery habitat and removing juveniles from the dataset would eliminate the principal resource of concern (Baron et al. 2004).

Fish ecologists are well aware of the effects of both stochastic settlement and post-settlement processes on

fish abundance (Doherty 1991; Jones 1991). However, parties interested in resource management are typically not trained ecologists. Resource management can be a contentious process with scientists and special interest groups (e.g. conservation groups, developers) defending highly polarized positions.

This is certainly the case with beach renourishment, when the economic interests of city managers, beach residents, and urban developers are opposed by conservation groups concerned with ecological damage (Lindeman 1997a,b; Pilkey 1997; Lindeman and Snyder 1999; Huston 2002). It is critical in such a highly charged political setting that environmental assessment is as accurate as possible, with a full measure of natural variation, to insure any correlation between environmental change and anthropogenic activity has a high probability of reflecting causality.

Our data highlight the importance of understanding potential variation in a baseline dataset. Hypothetically, in this study, if the project had gone forward as scheduled, the natural decrease seen in fish abundance between the 2001 and 2003 surveys could have been attributed to the negative effects of beach renourishment, by groups opposing beach renourishment. In the same hypothetical situation, if the natural inter-year variation had been reversed (i.e. fish abundance and species richness increased from 2001 to 2003), the increase could have been hailed as the positive ecological effects of beach renourishment by its proponents. In either case, interpretation of the results would have been spurious.

At this point, when dealing with nursery habitat, there does not appear to be a single pre/post-assessment approach that avoids the problems of the extreme variations found in some assemblages primarily comprised of juvenile fishes. Obviously, a temporally deep baseline dataset would be ideal. Such datasets would provide a better measure of variation than a single "snapshot" survey. This would insure that pre/post-impact statistical difference does not simply reflect natural variation and would allow for realistic power analysis to determine the sample size required to avoid a type II statistical error. However, cost and time

Table 1. Means, standard error (\pm SE), and ANOVA test statistics (ANOVAs for abundances were run using $\log[x+1]$ transformed data).

	2001		2003		Mean Square	F	p
	Mean	\pm SE	Mean	\pm SE			
Total abundance	116.88	20.29	43.51	7.48	6.36	28.73	<0.000001
Species richness	9.28	0.44	7.76	0.44	104.32	5.87	0.016390
Juvenile <i>Haemulon</i> spp.	88.58	19.79	24.20	6.81	10.48	15.47	0.000120
<i>Halichoeres bivittatus</i>	7.98	0.78	4.71	0.45	1.28	8.93	0.003193
Total abundance excluding juvenile <i>Haemulon</i> and <i>Halichoeres bivittatus</i>	20.61	2.08	14.81	1.97	1.01	8.32	0.004395
Total abundance excluding fishes <5cm TL	10.90	1.90	11.30	1.70	0.048	0.245	0.622

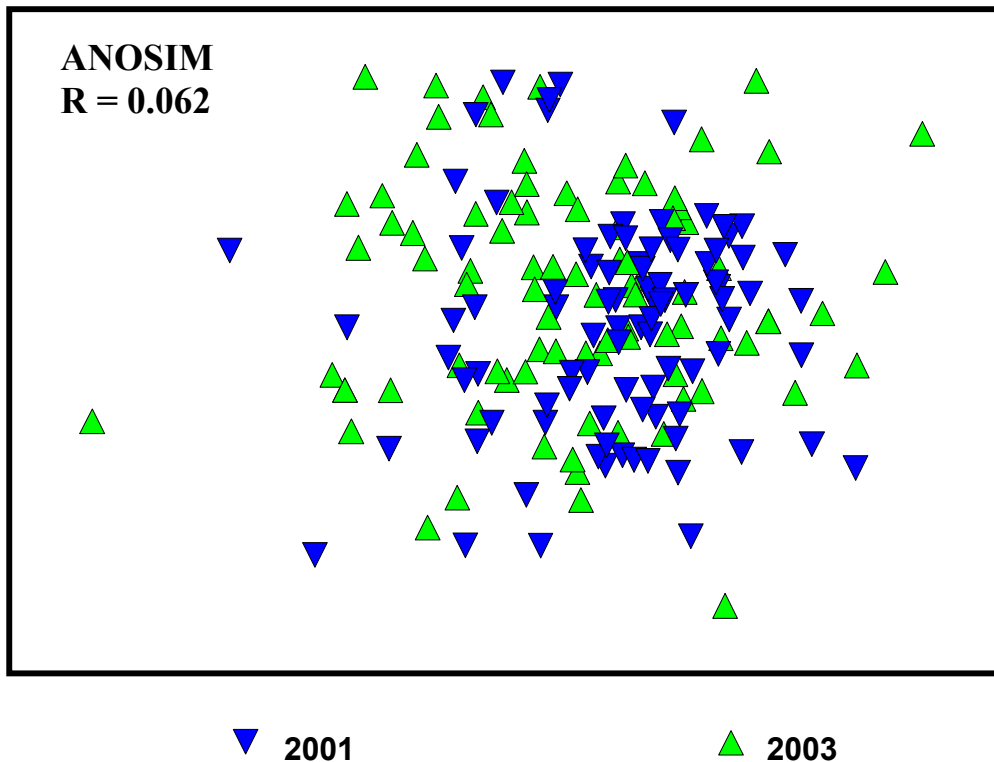


Figure 2. MDS plot of Bray-Curtis similarity indices by survey year (distance between triangles is proportional to similarity, e.g. closer proximity = greater similarity).

considerations make it unlikely that temporally deep datasets will be available for most impact assessments. The use of non-impacted control sites for comparison (e.g. BACI design) is a potentially useful approach but, in many cases, such a sampling design may be limited by the spatially stochastic variation found in larval fish settlement (Doherty 1991; Osenberg and Schmitt 1996). If available, the use of an indicator species might provide insight into environmental impact on juvenile habitat (Courtenay et al. 1980; Jones and Kaly 1996) but such an indicator, will, presumably, be location-specific. Most likely, impact appraisal, for other than catastrophic change, will depend on the interpretation of multiple assessment approaches rather than a single unambiguous answer.

In summary, our data suggest that juvenile fishes may not provide reliable indicators of anthropogenic change in fish communities. Likewise, a single snapshot baseline database may miss significant temporal variation in abundance, thus, providing misleading information for examining anthropogenic change.

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