FISH COMMUNITIES: EVALUATING RESPONSE TO NATURAL AND ANTHOPOGENIC DISTURBANCES UNDER DIFFERENT MANAGEMENT REGIMES

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ABSTRACT

This study assessed habitat composition and fish community structure within differing management zones in the Dry Tortugas and Pulley Ridge regions of the west Florida shelf. These management zones include protected areas that are restrictive in area use (e.g., fishing). The management zones within the Dry Tortugas include ecological reserves, a marine reserve, and several marine protected areas. A portion of Pulley Ridge is also a marine protected area and is labeled as a Habitat Area of Particular Concern. I included the Pulley Ridge region in part because it was expected to be different, thus it acted as a positive control. Using a long-term data series, the habitat and fish community were analyzed within each management zone, within and among management regimes, and among management zones to detect changes in community composition as the Dry Tortugas and Pulley Ridge region were impacted by natural disturbances and anthropogenic management activities. Surveys detected significant differences within and among management zones for the reef fish community and differences among zones for habitat composition. There is evidence that differences in habitat composition were driving most of the differences seen in the fish community. Results from this study also suggest management implementation may have had an effect on the fish community within one of the ecological reserves, the Tortugas North Ecological Reserve. Due to insufficient sampling throughout time, effects from natural and anthropogenic disturbances were not clearly linked with differences observed in both the habitat and fish community. However, in general, most commercially important fish species abundances were greater in the no-take reserves when compared to open access areas.

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1. Introduction

Coral reefs are one of the most diverse ecosystems in the world, including at least one-third of described marine species but covering less than 1% of the available marine habitat (Veron et al., 2009). Coral reefs are found along the coastlines of more than 100 countries in the tropics and subtropics. Reefs are subject to many types of natural disturbances (Hu et al., 2004), and because nearly 8% of the world's population lives within 100 km of a coral reef system, anthropogenic disturbances can also have significant impacts. Both long-term (e.g., years) and short-term (e.g., months) disturbances can affect coral reef ecosystems by altering habitat composition as well as the associated reef fish communities. Changes in habitat composition can be caused by overfishing, eutrophication, bleaching, infrequent cold and warm weather events, and tropical cyclones. While the capacity to recover from disturbance varies with reef type, it is thought that communities in marine protected areas, especially marine reserves, are able to recover faster and are more resilient than unprotected areas (Game et al., 2008).

The Florida Keys coral reef ecosystem, which extends 400 km southwest of Miami, is subjected to a variety of long- and short-term natural and anthropogenic disturbances. It is also the site of several marine protected areas (MPAs) established to protect the associated coral reef communities (Ault et al., 2012). Although numerous taxa (fish, invertebrates, algae, etc.) use the reefs within MPAs in the Florida Keys for food resources, shelter, and reproduction, this study focused on changes to the habitat composition and reef fish communities of the Dry Tortugas.

1.1 Natural disturbances

Natural disturbances modify habitat composition when the physiological thresholds of benthic organisms are exceeded or when physical damage occurs. Tropical cyclones can negatively impact reefs via mechanical damage to corals and smothering of coral polyps by rubble or suspension of sediments (Jones and Syms, 1998). Typical spur-and-groove reef topography found in the Florida Keys is thought to disperse wave energy, however, severe hurricanes have broken branching corals at depths of up to 12 m (Scoffin, 1993). Fish communities are thought to recover quickly after minor storms (i.e. tropical storms) for several reasons including: 1) transient life history patterns (i.e. they seek shelter or avoid storms), 2) required resources (i.e. algae and coral) are not severely impacted, or 3) required resources recover in relatively quickly. Their responses can vary depending on the intensity of the storm, and recolonization and recovery is thought to be slowest after severe storms when the majority of the habitat is damaged (Jones and Syms, 1998). Reef fish surveys conducted in the Dry Tortugas from 2004-2006 following six hurricanes documented declines in abundances of species such as black grouper (*Mycteroperca bonaci*; Smith et al., 2011), red grouper (*Epinephelus morio*), and members of the family Labridae (Ault et al., 2012). In addition, the effects of tropical cyclones may cause a change in habitat heterogeneity which can affect the abundances of reef fish belonging to different trophic groups (Jones and Syms, 1998).

The Florida Keys and Dry Tortugas are intermittently exposed to cold water stress that has been correlated to mortality of marine benthic organisms and has the potential to impact fish communities (Colella et al., 2012). South Florida is subject to intrusions of cold air masses from the Arctic Oscillation which produces low water temperatures and can lead to fish and coral kills. Extensive coral mortality (up to 96%) and fish kills in the northern Bahamas and Florida Reef tract were documented in relation to one of these events (Roberts et al., 1982). A more recent cold water event occurred in January 2010, lasted approximately 11 days, and caused a reduction in live coral cover in the Florida Keys area (Colella et al., 2012). In events like these, protection through the creation of an MPA is questionable because political boundaries do nothing to impede natural phenomena. MPAs can, however, increase overall community resilience by reducing the amount of impacts (anthropogenic) experienced by functional groups of species within the community (Selig and Bruno, 2010).

1.2 Anthropogenic disturbances

There are a variety anthropogenic disturbances and activities that result in loss or alteration of coral reef communities. Some of the most obvious of these include overfishing and physical damage to corals due to anchoring or intense diving activity. The abundant resources provided by coral reef ecosystems support fish communities that include numerous commercially important fishery species and those resources are used for food and many other purposes (Hallac et al., 2012). The Florida Keys are exposed to heavy fishing pressure and extensive recreational use (diving, snorkeling, etc.; Bohnsack et al., 1994) which has reduced populations of the "snapper-grouper complex," including species in the genera *Lutjanus*, *Epinephelus* and *Mycteroperca* (Ault et al., 2006).

1.3 Marine protected areas

One method that has been used to address these issues is through the establishment MPAs. In the United States, there are four main types of MPAs, depending on the restrictions put in place to protect species and/or habitats. They are classified as either less restrictive marine protected areas, marine reserves, fishery reserves, or ecological reserves, which have the most stringent restrictions (Jentoft et al., 2011).

In some cases when MPAs are established, degradation of fish communities stops, or at least slows, and the community may recover as evidenced by increasing biodiversity within the protected area (Côté et al., 2001; Worm et al., 2006; Hallac et al., 2012). Studies examined by Worm et al. (2006) found increases in fish diversity in marine reserves which helped improve ecosystem stability. Ecosystem stability in that case was defined as the ability of the ecosystem to withstand reoccurring disturbances, across many trophic levels. Marine protected areas can also protect spawning habitat and thus indirectly have positive effects on the fish abundances outside the MPA, and particularly down-current of the protected area (Ault et al., 2006).

Starting in 1997, "no take" marine reserves were established in the Florida Keys National Marine Sanctuary (FKNMS) to manage declining populations of fishery species in the (Ault et al., 2006). Due to its placement in the Florida current, the Florida no-take marine reserve (NTMR) network is believed to protect important fishery spawning sites in which eggs and larvae are transported to the upper Florida Keys by the Florida Current (Ault et al., 2006). For this reason, the Florida NTMR network was expanded into the Dry Tortugas in 2001 with the establishment of ecological and marine reserve areas.

These no-take reserves strictly forbid harvesting and possessing any marine life, including living or dead coral. Management zones in the study area include three no-take areas (Fig. 1A): Tortugas North (TNER) and Tortugas South Ecological Reserves (TSER), implemented in 2001, and Dry Tortugas National Park Research Natural Area (DRTO RNA) implemented in 2007. The other management zones are Tortugas Bank

Open Access (TBO), which is open to commercial and recreational fishing, and Dry Tortugas National Park (DRTO) which is open to recreational angling only since 1960 (Ault et al., 2012).

Finally the Dry Tortugas region is generally characterized by diverse coral reef habitat which includes species such as staghorn coral (*Acropora cervicornis*), starlet coral (*Siderastrea siderea*), rough cactus coral *(Mycetophyllia ferox*), and branched finger coral (*Porites furcata*) (Porter et al., 1982). There are a few differences between the management zones. The FKNMS is open to both recreational and commercial fishing and is characterized by rocky outcrops and low-relief spur and groove formations. A few pinnacle reefs, low-relief hard bottom, and reef terrace characterize TNER (Ault et al., 2006). Both TSER and TBO have rocky outcrops and low-relief hard bottom. The DRTO RNA and DRTO regions are characterized by spur and groove formations and patch reefs.

Pulley Ridge is a unique Gulf of Mexico coral reef ecosystem which is located 66 km west of the Dry Tortugas. Pulley Ridge is considered a Habitat Area of Particular Concern (HAPC) because it contains the deepest known photosynthetic corals in the continental United States (Farrington and Reed, 2013) (Fig. 1B). These photosynthetic scleractinian corals and coralline algae are located on the southern portion of Pulley Ridge in a depth range between 60-75 m. Deeming Pulley Ridge an HAPC establishes protection from fishing practices such as trawling, traps and pots, and longline fishing but does not protect the reef from impacts caused by anchoring vessels. The advisory council for the FKNMS is currently reviewing the sanctuary regulations and are considering including Pulley Ridge within the FKNMS (Farrington and Reed, 2013).

1.4 Objectives

The Southeast Monitoring and Assessment Program (SEAMAP) reef fish video survey has sampled fish communities of hard bottom banks throughout the Gulf of Mexico since 1992. This standardized fishery-independent survey provides consistent data collection of the fish community composition through time. The goal of this study was to use a long-term data series to evaluate if community composition in the Dry Tortugas and Pulley Ridge regions changed following natural disturbances and anthropogenic management activities that impacted the region. This region included 3 out of the 4 types of MPAs found within the United States (open access MPA, marine reserve, and ecological reserve).

The objectives of this study were; 1) to determine if habitat composition and fish communities have changed over time within areas with differing management regimes (e.g., ecological reserve etc.; H_0: habitat composition and fish communities show no change in structure through time), 2) to determine if habitat composition and fish communities differ among areas with differing management regimes (H_0: habitat composition and fish communities show no change in structure given a management application of regime), and 3) to determine if habitat composition and fish communities differ among management zones (H_0: habitat composition and fish communities show no change in structure among specific management zones set up in the Dry Tortugas).

2. Materials and methods

2.1 Study area

For this study I focused on a region of the Florida Keys reef tract known as the Dry Tortugas and which is located 113 km west of Key West, Florida (Fig. 1A and 1B).

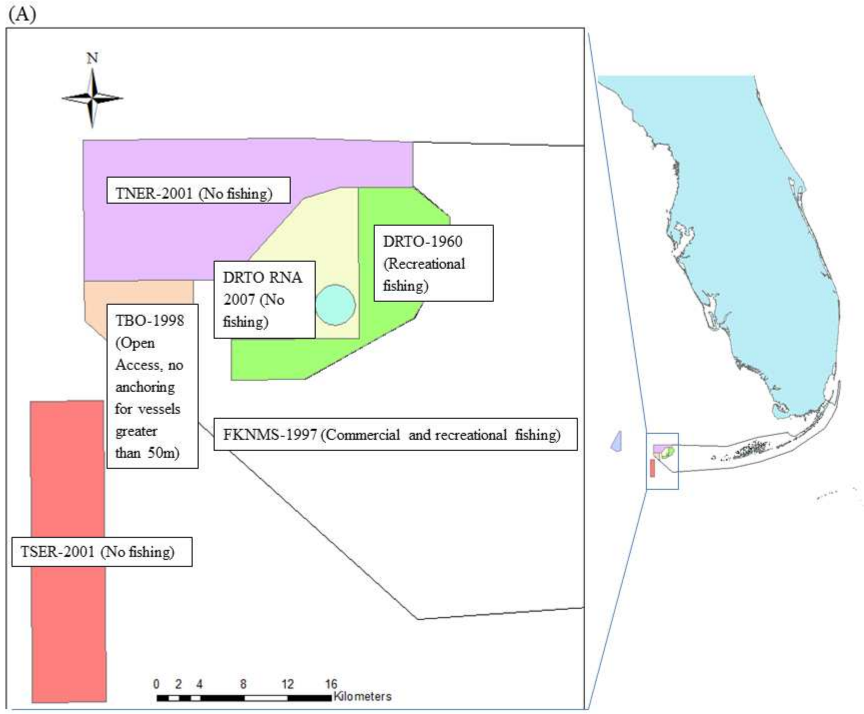
Management zones include FKNMS, DRTO RNA, DRTO, TNER, TSER, TBO, and Pulley Ridge.

2.2 Survey Data Collection

Survey sites for the SEAMAP reef fish video survey of the Gulf of Mexico (GOM) are randomly selected from available reef area in the Gulf of Mexico (~ $1771 km^2$) using a two stage random stratified sampling design (Campbell et al., 2012). The first stage, randomly selects a 10x10 min (latitude by longitude) block from a known reef universe. Blocks with greater coverage of reef habitat are more frequently selected for sampling than those containing less reef habitat. Reef areas contained within a block were previously digitized by overlaying a $0.11\ km^2$ grid on the reef area. The second stage units, which are the actual survey sites, are then randomly selected from the available points on the grid.

The array deployed contains four orthogonally mounted systems allowing for approximately 270°, non-overlapping, total field of view (Fig. 2). To keep track of which video came from the associated housing, cameras are labeled A, B, C, and D on the camera mount. A shipboard computing system (SCS) records the position (latitude and longitude), depth, and weather conditions of the site upon deployment of the video array. After the array is deployed environmental data (temperature, conductivity, depth, oxygen content, turbidity, etc.) is collected near the site using a CTD.

All four digital videos from each site are viewed to determine video quality. If only one video is of good quality (i.e., good visibility, in focus) that camera is selected for viewing. If more than one video of good quality is available, a random number is generated to determine which video will be viewed. If all four videos cannot be



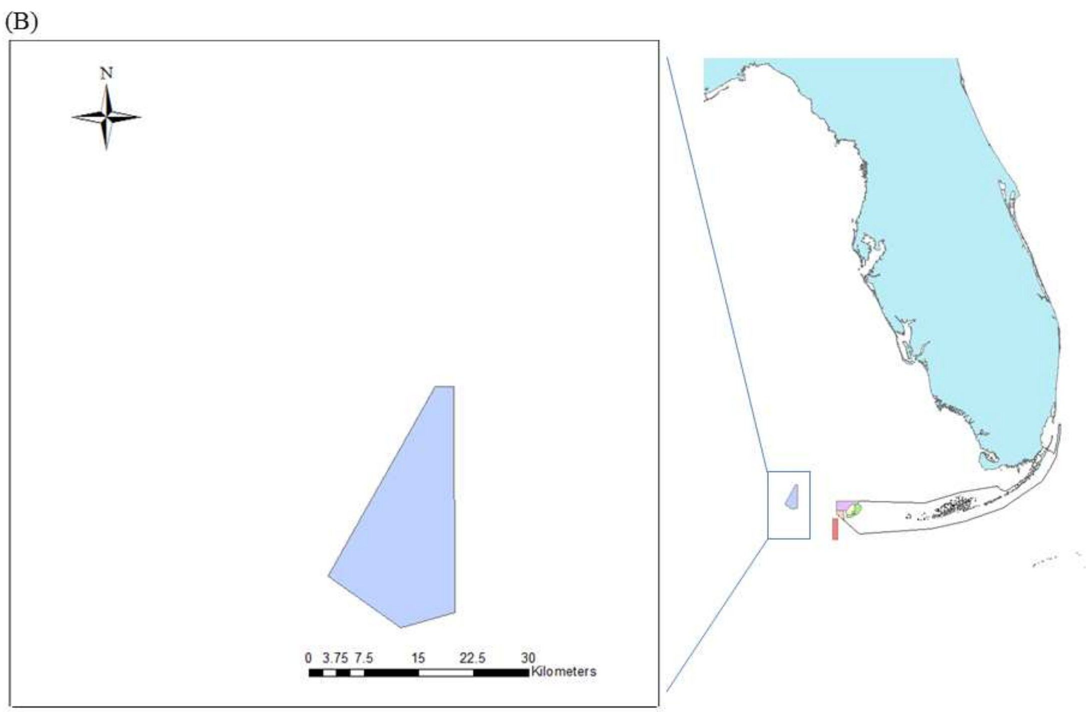


Fig.1. (A) Study area of the Florida Dry Tortugas and (B) Pulley Ridge Habitat Area of Particular Concern (HAPC). Pulley Ridge sampling sites extend further north but only part of the area has specific regulations.

viewed the station is labeled with an 'XX' operational code and dropped from the analysis. Once selected, the video is viewed for 20 minutes following the settlement of suspended material from deployment of the array. Species are identified to the lowest taxon possible and min-count abundances are determined for them (Campbell et al., 2012). Min-count is defined as the maximum number of individuals of a taxon in the field of view at the same time during a 20 min read period. The min-count value therefore represents the conservative, maximum number of individuals observed during a deployment. This method eliminates the chances of counting the same fish more than once which may occur if fish circle the camera. Min-count estimation of abundance is performed for all species that can be identified to taxon within the MPAs. Fish species that are included on the species read list were accounted for and recorded (Appendix A.1).

Habitat composition data are also documented to describe each site by identifying and estimating percent coverage by substrate type. While the habitat composition data does give an estimate of percent cover it is important to understand that the video is not shooting aerial type footage, so the percent coverage is a rough estimate. Furthermore the habitat composition is only being categorized from the selected camera which may or may not be absolutely representative of a region as a whole (i.e. cameras are sometimes not oriented for the best view). Substrate type is classified as silt/sand/clay, shell/gravel, rock, artificial material, and attached epifauna. The attached epifauna includes seagrass, sponge, algae, hard coral, soft coral, etc. The habitat data that will be used in this analysis includes percentages of algae, hard coral, soft coral, silt/sand/clay, shell/gravel, and rock.

Fig. 2. Camera array. NOAA Marine Operations National Oceanic and Atmospheric Administration. [modified 2012 February 09]. Available from: http://www.moc.noaa.gov/ot/visitor/reeffish.html

The data are stored as a relational data set in Access and were assembled into a "site-by-site" format in SAS (SAS Institute Inc.). The sites in the SEAMAP reef fish video data set were linked to specific MPA management areas using the spatial join function in ArcGIS. This insures the sites were labeled with the correct location and management zone. Survey blocks were categorized based on their current management regime (Table 1). Some sites that were sampled next to the border of an MPA boundary were included in the analysis as FKNMS due to the close location and similarity of regulations.

Species that were not abundant were combined at the genus or family level (Appendix A.2). For example, there were an overall total of four rock hind (*Epinephelus adscensionis*) and only six red hind (*E. guttatus*), but 194 red grouper (*E. morio*) throughout all zones and years. These species were combined into the genus *Epinephelus* sp. since they are trophically similar. However, there were instances when a rare species could not be combined at the genus level. For example, there was only one Atlantic

Table 1
MPA's located in the Dry Tortugas Florida, and with the corresponding U.S. classification and restrictions.

Tortugas MPA Designation	U.S. MPA Classification	Year	Restrictions
FKNMS	Marine Protected Area	1997	Recreational and commercial fishing, spearfishing and lobster fishing allowed. Majority of area within Gulf of Mexico Federal waters. Anchoring allowed on hard rocky bottom but not on corals if bottom can be seen.
TBO	Marine Protected Area	1998	Open access to commercial and recreational fishing, spearfishing and lobster fishing is permitted. The shallowest part of Tortugas Bank region is an area to be avoided by deep-draft vessels. Within State Waters boundary. No anchoring allowed for vessels over 50ft.
TSER, TNER	Ecological Reserve	2001	All fishing activity in this region prohibited unless permit obtained (e.g., research). TSER-Vessels have to have fishing gear stowed and be in continuous transit. TNER-Vessels have to have access permit to use a mooring buoy or stop. Vessels more than 100ft cannot use mooring buoy.
Pulley Ridge	Marine Protected Area	2005	The area in Pulley Ridge that is within the HAPC is only protected from anchoring of fishing vessels, and is not protected from anchoring of other vessel types. Bottom longline, bottom trawling, and all pots/traps are prohibited; but other types of extractive uses are not prohibited.
DRTO RNA	Marine Reserve	2007	No fishing is allowed, snorkeling and diving is allowed if commercial tour guides are used.
DRTO	Marine Protected Area	1960	Recreational fishing allowed, spearfishing prohibited.

bigeye (*Priacanthus arenatus*) and no other related taxa at the genus or family level, so this species was retained. The taxon Labroidei (suborder) was the most frequent identification for most parrotfish, with very few individuals identified to species due to the large number of phases for wrasses which makes identification difficult. For this reason all parrotfish and wrasses were aggregated into the suborder Labroidei except for the stoplight parrotfish (*Sparisoma viride*) because the species min-count was high.

Finally, video quality has improved through time, however the quality was lower in the early years of the survey which made it difficult to identify many fish to species. Finally, the intent of the SEAMAP reef fish video survey was to create indices of abundance for Fishery Management Plan (FMP) species and not necessarily to evaluate absolute abundance, hence the reduced number of species that are read at a station. The amount of labor involved precludes reading every tape for all species available.

2.3 Statistical analysis

Descriptive statistics (average abundance and standard deviation) for each taxon by year and management zone are presented along with Shannon Weiner Diversity (H', $\log_{10}$), evenness, and species richness. Community analysis to determine differences within management zones focused on non-metric multi-dimensional scaling (MDS) ordination of Bray-Curtis similarity matrices using PRIMER software (PRIMER v6, Plymouth UK: Clarke and Gorley, 2006). For each species, abundances were averaged from all sites within the management zone each year. Due to the high variability of fish abundances (e.g., schooling fish), abundance data were transformed ($\log_{10}$) prior to analysis to down-weight the most abundant species. The SIMPROF test provides a single measure of similarity for years within zones taken from a multivariate dataset (Clarke et al., 2008). The SIMPROF test also provides a means to determine if clusters of sites or variables are significantly different. This analysis tests the null hypothesis of no multivariate structure at a 95% confidence interval (Clarke et al., 2008). Therefore Iused the Bray-Curtis cluster analysis with the similarity profile test (SIMPROF) to evaluate the hypothesis that both the habitat composition and the fish community did not change over time within management zones. FKNMS, TNER, and TBO were the only management zones that contained consistent sampling throughout the years of this study and were the only zones used for this part of the analyses (Table 2). The year 2003 was omitted from the data set for this test because of extremely low samples in that year.

Hierarchical cluster analysis is calculated and displayed on a dendrogram to help illustrate differences between years within each management zone. If a non-significant result was obtained it is considered to have no internal structure and the samples of the community structure are considered homogenous (Clarke et al., 2008). Finally Iused MDS ordination as an additional method to illustrate differences between groups observed from the hierarchical cluster analysis (i.e. MDS plots represent distance between samples). Cluster analysis results were superimposed on the 2-dimensional MDS plot.

The management zones were then combined under three management regimes (marine protected area=lowest, marine reserve= medium, and ecological reserve= highest), in respect to the level of protection in which they provide. Fish abundance data for all sites were summed for each year within each management regime, making the year act as replicates for this analysis. Bray-Curtis cluster analysis, with the SIMPROF test, and MDS ordination were also used to test for differences between years among management regimes. An ANOSIM was done to determine if there were significant differences among management regimes. Finally, the similarity percentage analysis (SIMPER) was used to determine which fish taxa might be driving differences in community structure among management regimes (untransformed abundance data). SIMPER identifies the best discriminating taxa using the ratio of the mean Bray-Curtis dissimilarity to its standard deviation. A ratio ≥ 1.0 indicates that taxon consistently contributes to differences in community structure between two communities.

Bray-Curtis cluster analysis, with the SIMPROF test, and MDS ordination were also used to test for differences in habitat composition and fish communities among management zones. These analyses were augmented with analysis of similarity (ANOSIM) to determine if there were significant differences in habitat composition or fish communities among management zones. Pulley Ridge is a very different ecosystem from all the other management areas and therefore I also performed a test which excluded it from the among zone analyses so that only zones within the Tortugas management areas would be evaluated.

3. Results

I analyzed a total of 541 videos that spanned from 1997-2012 (Table 2). The sampled sites throughout the years are shown below (Fig. 3A and 3B). The survey was not conducted in the Dry Tortugas in 1998–2001 or 2006 for a variety of reasons including the random selection procedure did not select a zone that year, or because of weather, vessel and funding issues.

3.1 Habitat characterization

The major habitat composition for the FKNMS, DRTO RNA, DRTO, and TNER was silt/sand/clay and soft-coral (Table 3). The TSER was characterized by mainly algae cover along with similar percent coverages of soft-coral, silt/sand/clay, and rock. Silt/sand/clay and algae cover was the major habitat composition found in TBO and Pulley Ridge. Coverage of hard-coral and soft-coral was greatest in TNER and DRTO RNA whereas

Table 2
The locations and years sampled with the number of sites sampled for each year/location combination.

Location	1997	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012
FKNMS	14	8	5	5	10	17	5	12	14	12	10
Pulley Ridge					20	19	16	13	13	17	20
DRTO RNA	3				6	3	5	4	3	3	7
DRTO	3		3			5	1	3	3	7	3
TBO		3	8	6	4	7	4	6	3	3	6
TNER	12	18	1	14	17	23	17	19	20	19	23
TSER		4				2		1	4	5	

Pulley Ridge had the least coral coverage. DRTO had the highest silt/sand/clay bottom followed by TBO and DRTO RNA.

3.2 Fish abundance

A total of 6,130 fish were observed; yellowtail snapper (*Ocyurus chrysurus)* was the most abundant overall, constituting ~21% of observations (1,259) followed by white grunt (*Haemulon plumieri)* with ~7% of observations. Total fish abundance was highest at FKNMS and lowest at TSER (Appendix 3). On an annual basis, fish were most abundant in TNER during 2002 and 2012 followed by FKNMS in 1997 and TNER in 2011, with the fewest encountered at DRTO during 2010. Yellowtail snapper were one of the three most abundant species in all management zones sampled except Pulley Ridge and TSER. Average fish abundances within management zones when aggregating over the sampling years showed fish were extremely variable from year to year within management zones; for example, during 2011 there were an average of 5 fish per sampling area in Pulley Ridge, but only 2 fish per sampling area during 2012. In DRTO RNA an average abundance of 4 fish were observed in 2008, declining to an average of less than 1 in 2009.

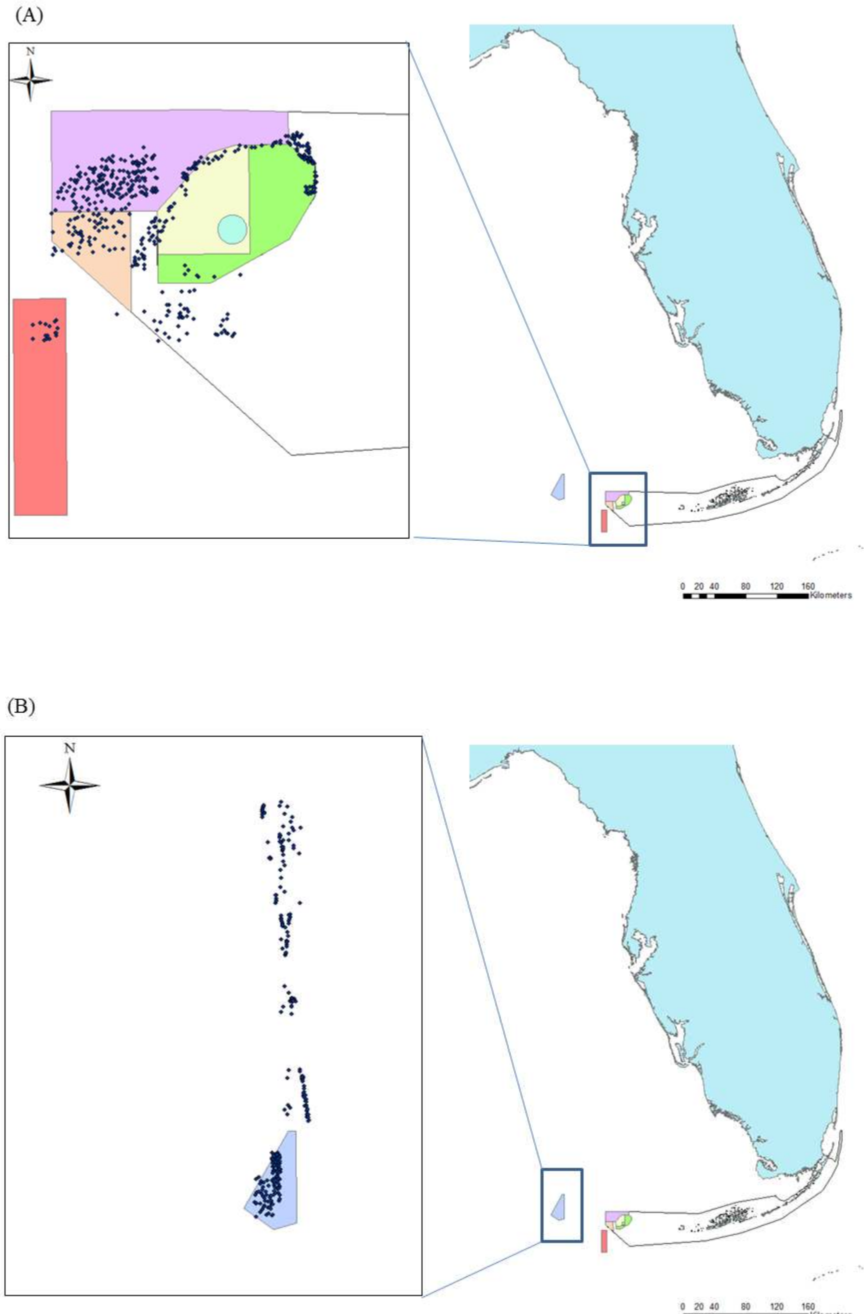


Fig. 3. (A) Sites sampled all years for this study in FKNMS, DRTO RNA, DRTO, TNER, TSER, and TBO. (B) Sites sampled inside and outside of Pulley Ridge HAPC.

Table 3
Average percent cover and standard deviation of habitat composition for the Florida Keys National Marine Sanctuary (FKNMS), Pulley Ridge, Research Natural Area (DRTO RNA), Research Natural Area (DRTO), Tortugas Bank (TBO), Tortugas North Ecological Reserve (TERN), and Tortugas South Ecological Reserve (TSER) for all years.

Zone	Algae	Hardcoral	Softcoral	Silt/Sand/Clay	Shell/gravel	Rock
FKNMS	19.1	5.3	21.6	43.3	9.5	10.8
	(16.3)	(10.7)	(21.6)	(32.6)	(13.8)	(14.0)
DRTO RNA	19.4	5.8	28.0	49.2	10.7	6.7
	(14.1)	(7.0)	(23.2)	(35.2)	(15.1)	(8.4)
DRTO	13.9	6.9	18.7	60.7	12.0	5.4
	(11.0)	(12.1)	(18.5)	(30.4)	(19.8)	(8.9)
TNER	16.3	16.8	23.2	36.7	9.3	14.1
	(17.0)	(20.6)	(19.5)	(31.2)	(18.0)	(17.7)
TSER	24.5	8.6	21.0	21.6	9.1	21.4
	(23.1)	(9.4)	(19.9)	(16.9)	(8.0)	(18.3)
TBO	20.6	1.3	17.4	51.8	8.3	9.4
	(24.5)	(3.2)	(16.7)	(32.3)	(17.3)	(9.9)
Pulley Ridge	72.8	0.4	5.9	29.1	8.4	14.8
	(26.0)	(1.6)	(5.8)	(36.2)	(15.0)	(23.0)

3.2.1 FKNMS

Fish abundance within the FKNMS was relatively consistent throughout time except for 1997 and 2012 where abundance was at least twice other sampling years (Appendix Fig. 3.1). Within the FKNMS, the three most abundant taxa were yellowtail snapper, white grunt, and members of the suborder Labroidei (Appendix Fig. 4.1). Yellowtail snapper was seen in all years and abundance nearly doubled in 2008 compared with 2004 but abundance was lower from 2009-2012. The yellowtail snapper was twice as abundant as the other species during most years although Labroidei was six times more abundant in 1997 than in any other year when it was recorded. The white grunt abundance was consistently low until 2011 and 2012, when it tripled from all the previous years.

3.2.2 DRTO RNA

Species observed in the DRTO RNA management zone showed no consistent abundance patterns (Appendix Fig. 3.2). Abundance doubled and was greatest in 2008 than all other years within the DRTO RNA. *Ocyurus chrysurus*, *Haemulon* sp., and the bar jack (*Caranx ruber*) were the most abundant species in the DRTO RNA management zone (Appendix Fig. 4.2) and were at least twice as abundant as other species within the zone. Bar jacks were observed four out of the eight years sampled. Yellowtail snapper was observed in DRTO RNA during all years sampled and abundance nearly doubled in 2008 from 2007 but remained lower in abundance for the following years. *Haemulon* sp. was five times more abundant in 2007 when compared with 2008.

3.2.3 DRTO

Average fish abundances within management zones when aggregating over the sampling years in the DRTO varied large and small throughout the years (Appendix Fig. 3.3). Yellowtail snapper were the most abundant species observed in the DRTO management zone, but was not observed in all years (Appendix Fig. 4.3). The surgeonfish (*Acanthurus coeruleus*) and *Calamus* sp. were the next most abundant species, and were also only observed during some years. Surgeonfish abundance was four times that of all other species in 1997, but it was not seen again until 2011 when average abundance was 0.1.

3.2.4 TNER

Species abundances within the TNER were greatest in 2002 and was relatively consistent throughout the years of this study that were sampled (Appendix Fig. 3.4). Yellowtail snapper and white grunt were the most abundant in the TNER management zone and both were present during all years except 2003 (Appendix Fig. 4.4). Bar jacks were also one of the top three most abundant species, peaking in 2002; this species was not observed in surveys in either 2003 or 2010. Neither bar jacks nor white grunts were as abundant as yellowtail snapper.

3.2.5 TSER

Abundances of all species were at least three times higher in 2002 and 2011 than all of the other years sampled within the TSER management zone (Appendix Fig. 3.5). *Haemulon* sp., *Calamus* sp., and mutton snapper (*Lutjanus analis*) were the three most abundant assemblages in TSER (Appendix Fig. 4.5). *Haemulon* sp. was nearly twice as abundant as all the other species in 2002 but *Haemulon* sp. abundance remained low and no grunts were seen in 2010 surveys. Both *Calamus* sp. and mutton snapper were most abundant in 2010. Despite the variation in abundance of these three species, overall average fish abundances were relatively stable.

3.2.6 TBO

Within the TBO management zone, the abundances of all species were greatest in 2004 and were consistent all other years sampled in this study (Appendix Fig. 3.6). The most abundant assemblages in the TBO management zone were *Ocyurus chrysurus*, *Haemulon* sp., and *Haemulon plumieri* (Appendix Fig. 4.6). *Haemulon* sp. was present for only three years during the study and was most abundant in 2004. The abundance for yellowtail snapper varied throughout the years within TBO, with a peak abundance in 2009 which was four times higher than in 2008. Abundance for white grunts doubled in from 2004 to 2005.

3.2.7 Pulley Ridge

In 2011, species abundances were five times greater than any other year sampled within Pulley Ridge (Appendix Fig. 3.7). The most abundant species at Pulley Ridge were vermillion snapper (*Rhomboplites aurorubens*), sand tilefish (*Malacanthus plumieri*), and *Lutjanus analis* (Appendix Fig. 4.7). Abundances of both sand tilefish, and mutton snapper were lower in 2008 than the other years. Vermillion snapper were only seen in 2011 and 2012, but in 2011 its abundance was at least 30 times greater than the any other species.

3.3 Species Diversity

Highest diversity in this study was observed in the FKNMS in 2005 and 2007 (Appendix Fig. 3.1). The remaining years within the FKNMS diversity was relatively consistent. Within the DRTO RNA species diversity was greatest in 2005, and was consistent the other years sampled in this study (Appendix Fig. 3.2). Diversity was greatest in 2007 within the DRTO, but otherwise did not show consistent trends through the years (Appendix Fig. 3.3). Within the TNER management zone diversity was consistent throughout all years except 2003 (Appendix Fig. 3.4). Greatest diversity for the TNER was in 2002, 2005, and 2009. Throughout the years sampled in the TSER management zone, diversity was greatest in 2002 and was consistent all other years (Appendix Fig. 3.5). Greatest diversity was observed in 2009 within the TBO, and was consistent all other years (Appendix Fig. 3.6). Diversity throughout time was consistent in Pulley Ridge except in 2011 were it was lowest. Greatest diversity was observed in 2005 and 2010 for Pulley Ridge.

3.4 Changes in habitat composition and fish community structure within management blocks over time

For habitat composition SIMPROF tests and cluster analysis showed no significant differences within management zones FKNMS, TNER, and TBO (Appendix Fig. 5.1-5.3). There were slight differences between years that were separated such as 2012 for both FKNMS and TNER. The years 2010 and 2012 were clustered together from the other years in TBO.

The TNER management zone showed some evidence of change in fish community structure through time (Pi = 1.70, significance = 1.4%; equivalent to a p-value of 0.014, Fig. 4A and 4B). At 62% similarity to the other years, 1997 and 2002 were significantly different clusters. Black grouper (*Mycteroperca bonaci*) were not observed in this zone until 2004 and parrotfish species (*Sparisoma* sp.) were most abundant in 1997 than all other years (Fig. 5B). Jacks (*Seriola* sp.) were not seen until 2007 and mackerel (*Scomberomorus* sp.) were not seen until 2002 (Fig. 5C and 5D).

In contrast, FKNMS and TBO showed no significant changes in fish community structure over time (Appendix Fig. 6.1-6.2). There were slight differences in the FKNMS, where the year 2004 was separated from the other clusters. The two tang groups, not seen in 2004, and abundance of surgeonfish were greater than other tang species (*Acanthurus* sp.) after 2004 (Appendix Fig. 3.1). There were also slight differences observed in 2005 within TBO, grunt species (*Haemulon* sp.) were only observed in 2004 and 2005, and decreased by 60% in 2005 (Appendix Fig. 3.6).

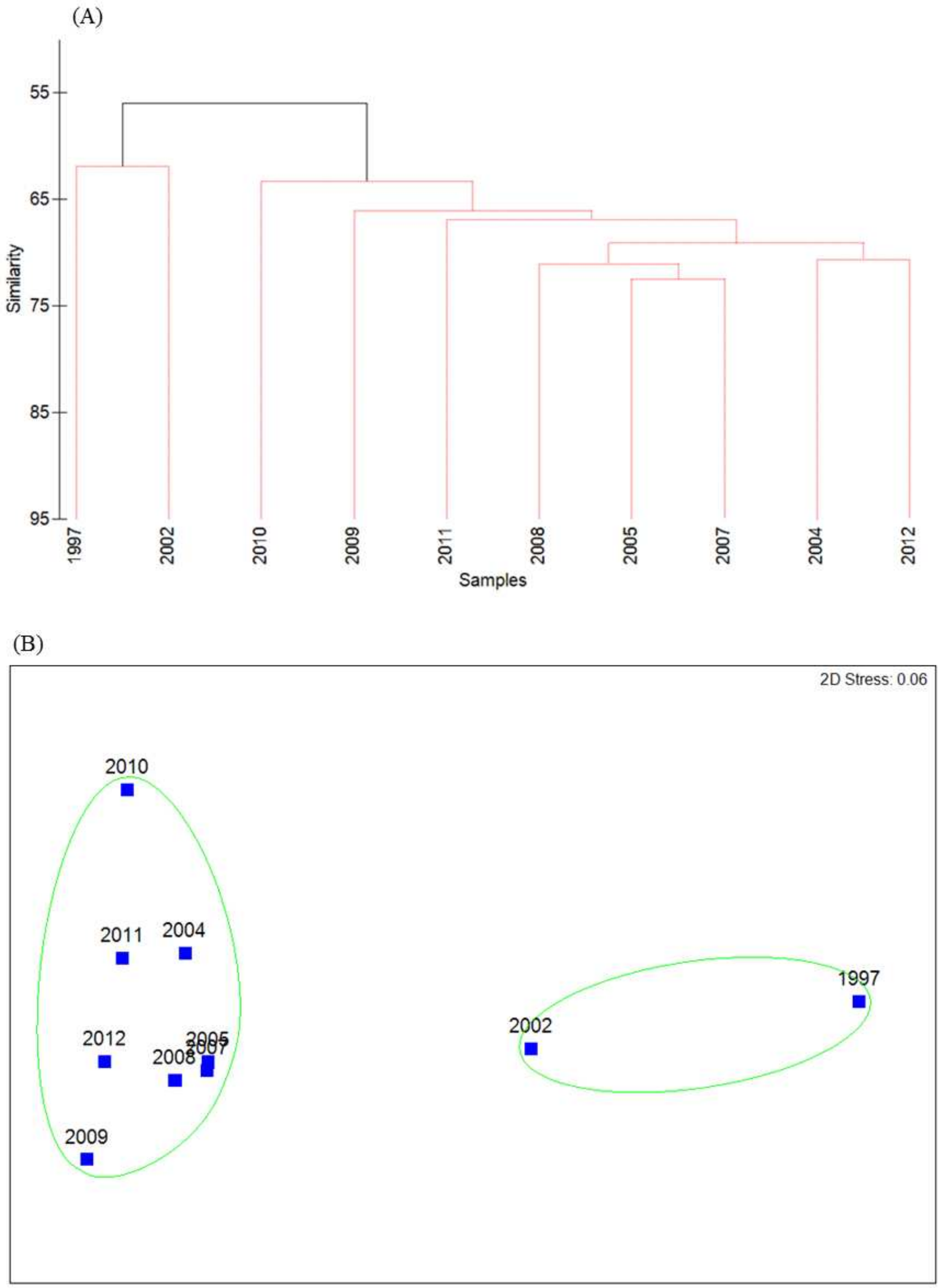


Fig. 4. Dendrogram of the Bray-Curtis cluster analysis (A) and MDS plot (B) at 62% similarity for the fish community within the Tortugas North Ecological Reserve (TNER).

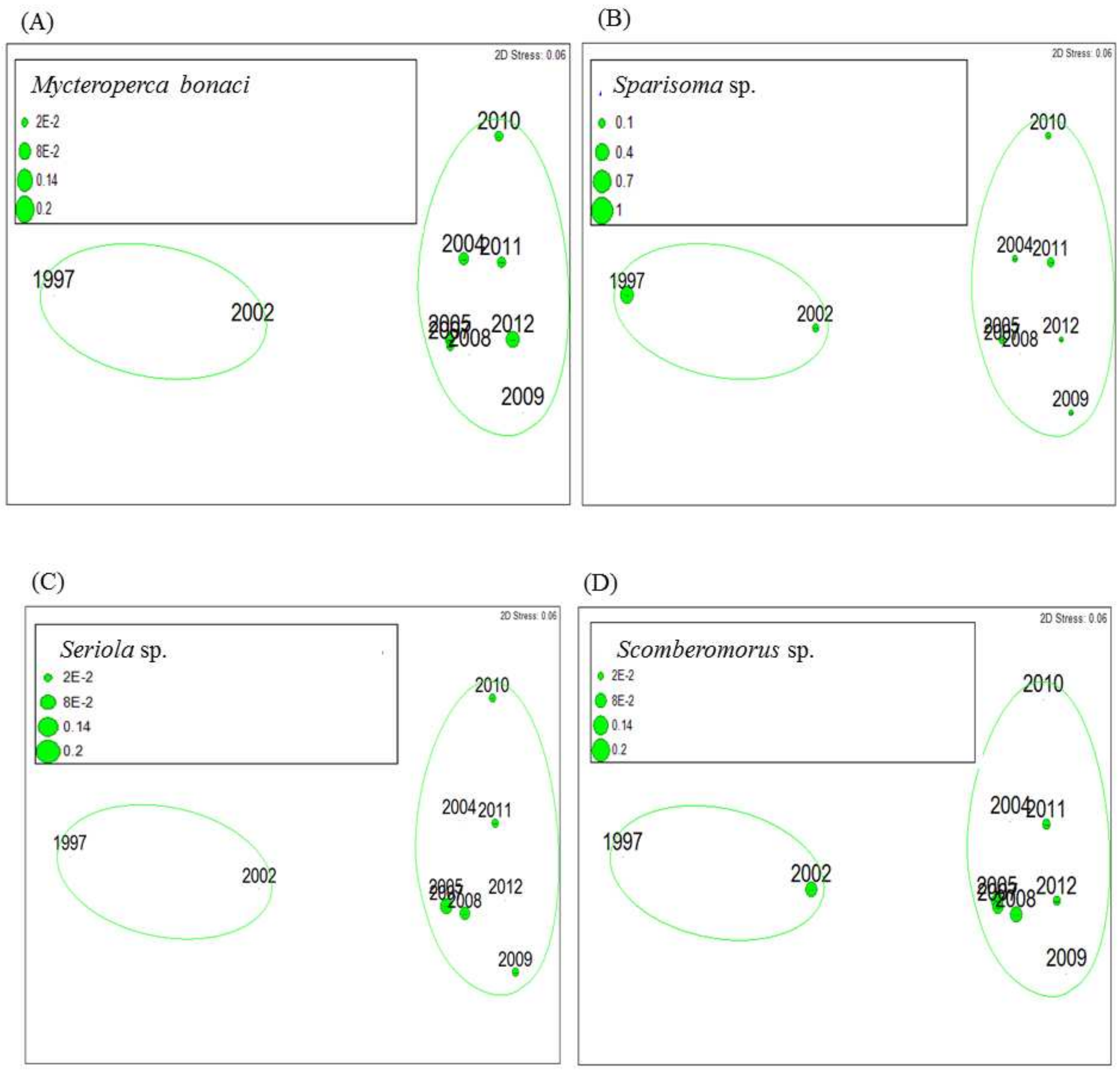


Fig. 5. Average abundances of (A) black grouper (*Mycteroperca bonaci*), (B) parrotfish (*Sparisoma* sp.), (C) jacks (*Seriola* sp.), and (D) mackerel (*Scomberomorus* sp.) are shown for the Tortugas North Ecological Reserve (TNER) management zone.

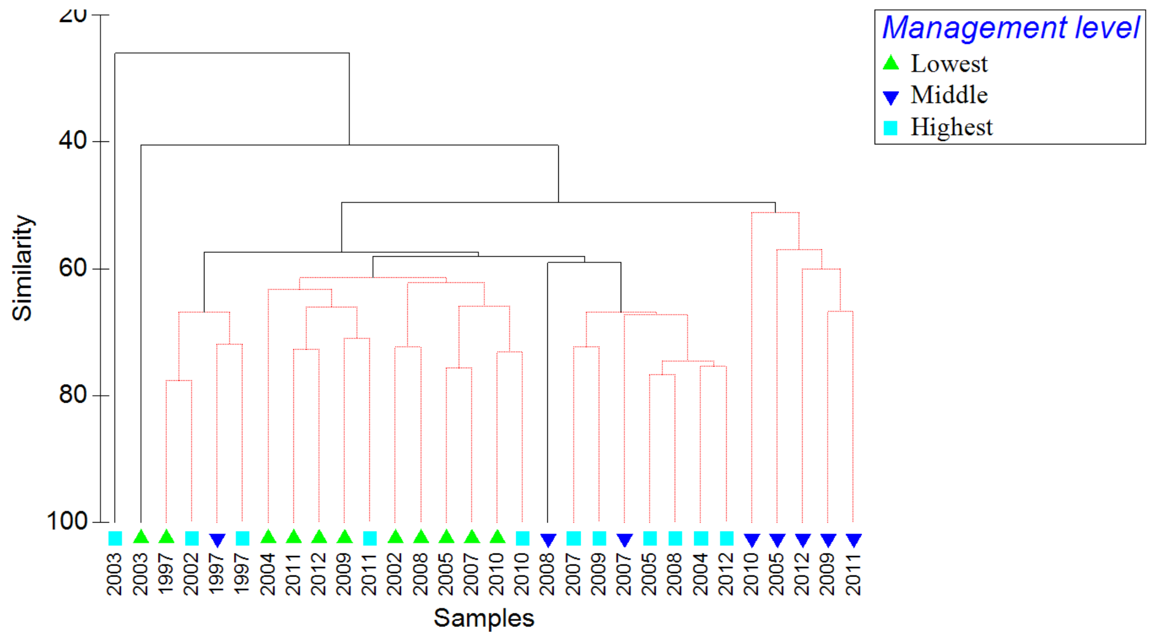
3.5 Differences in community structure among management regimes

Analysis of the SIMPROF and clusters showed that the highest level of protection (ecological reserves) in 2003 was significantly different at 26% similarity from all other years and management regimes (Pi = 4.02, significance percentage = 0.1%; equivalent to a p-value of 0.001). The lowest regime in 2003 also clustered out separately from the others (Pi = 2.79, significance percentage = 0.1%; equivalent to a p-value of 0.001) and was nearly 41% similar to all the other clusters. Dendrogram showed many significant clusters between years among management regimes (Fig. 6A). The MDS plot illustrates the clusters from the SIMPROF test for the fish community (Fig. 6B).

The analysis of similarity (ANOSIM) indicated one pairwise-comparison was significantly different among management regimes for the fish community (Global R = 0.353; significance percentage = 0.001%; equivalent to a p- value of 0.00001). The comparison between the lowest and middle regimes were significantly different (Global R = 0.546; significance percentage = 0.005%; equivalent to a p- value of 0.00005, Table 4).

The lowest and middle management regime had the largest average dissimilarity at 61.25%. Next the lowest and highest average dissimilarity was 55.27%, followed by the regime groups middle and highest at 58.16%. The SIMPER analysis demonstrated that the yellowtail snapper, white grunt, *Haemulon* sp., surgeonfish, and *Calamus* sp. best discriminated among management regimes (Appendix 7). The average abundance for yellowtail snapper in the lowest regime was nearly double the average abundance in the middle regime. The best discriminating taxa white grunt, surgeonfish, and *Calamus* sp.

(A)



(B)

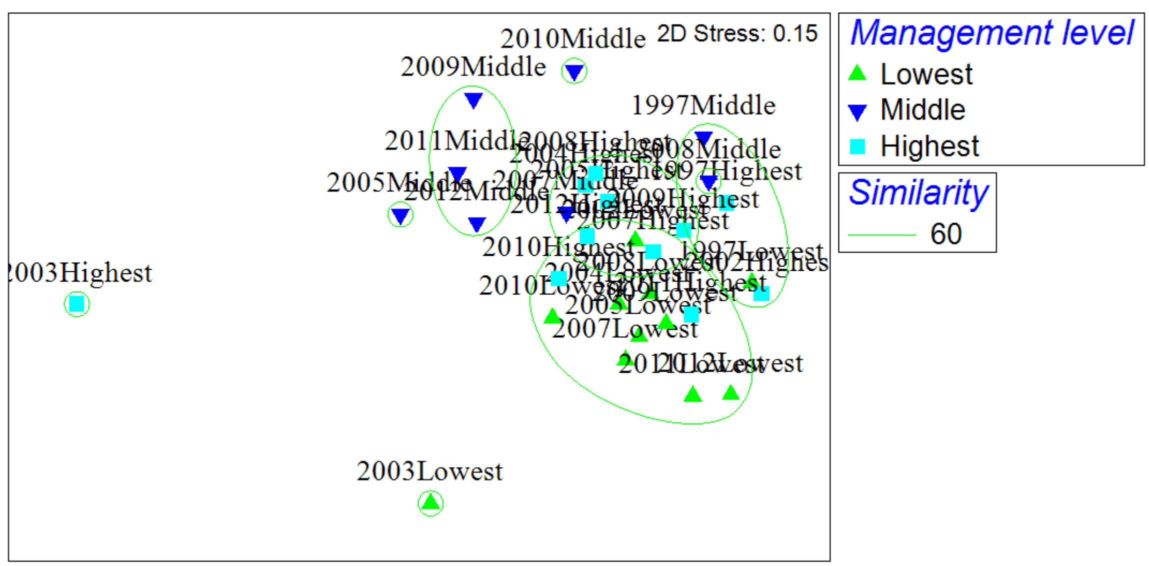


Fig. 6. (A) Dendrogram of the Bray-Curtis cluster analysis similarity and (B) MDS plot for the fish community between years among management regimes.

Table 4
Pairwise comparisons from ANOSIM analysis are shown to illustrate differences in fish community structure among management regimes in the Dry Tortugas, Florida.

Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number > = Observed
Lowest, Middle	0.546	0.005	75582	75582	4
Lowest, Highest	0.311	0.009	352716	100000	8
Middle, Highest	0.298	0.2	75582	75582	147

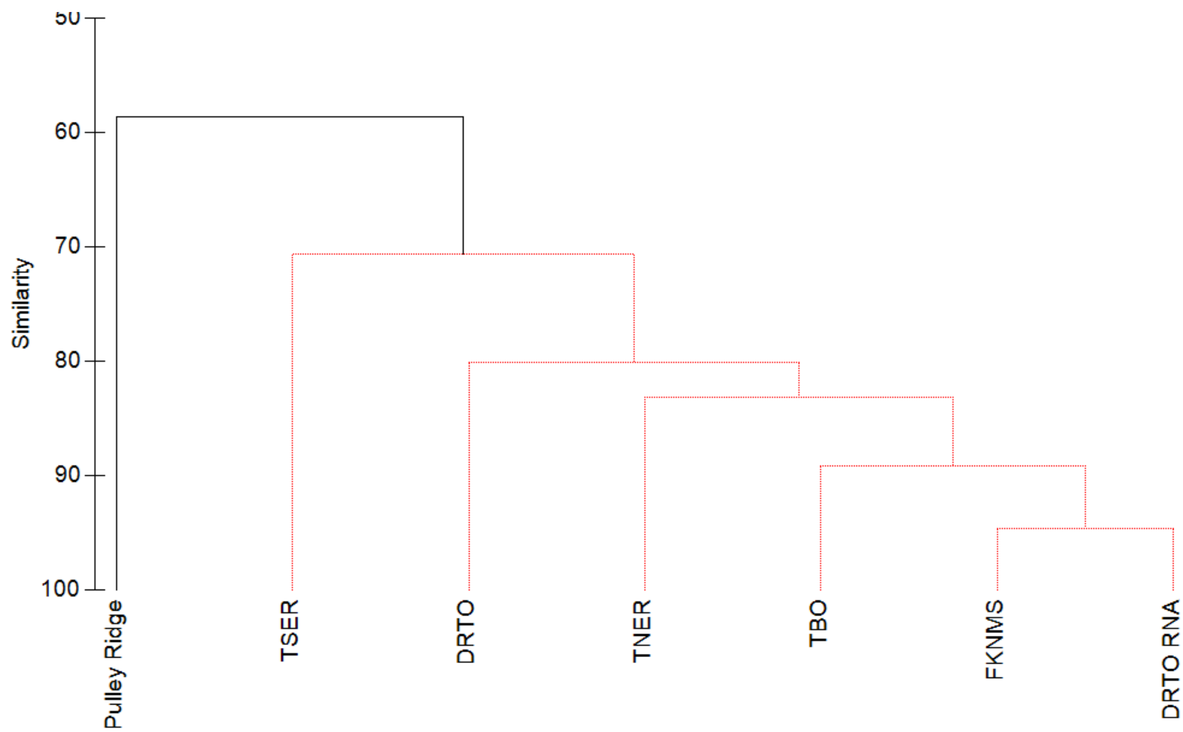
were also more abundant in the lowest protection regime. Yellowtail snapper contributed nearly 8% dissimilarity between the lowest and middle regime, and 7.6% between lowest and highest; thus also making it the most discriminating taxon for those regimes. The *Haemulon* sp. group was the most discriminating taxon for the dissimilarities between middle and highest regimes, contributing nearly 8%.

3.5 Differences in community structure among management zones

Analysis of the SIMPROF and clusters showed that Pulley Ridge was significantly different from all other management zones (Pi = 3.24, significance percentage = 3.4%; equivalent to a p-value of 0.034, Fig. 7A). The dendrogram showed that habitat composition in Pulley Ridge was 59% similar to the other management zones' habitat composition. All other zones were not statistically distinguishable from each other and the dendrogram show they were approximately 70% similar in habitat composition. When Pulley Ridge was removed from the analysis, there were no significant differences among management zones' habitat composition (Fig. 7B).

Fish community structure was significantly different from other management zones at Pulley Ridge (Pi = 11.32, significance percentage = 0.1%; equivalent to a p-value of 0.001) and TSER (Pi = 3.56, significance percentage = 0.1% equivalent to a

(A)



(B)

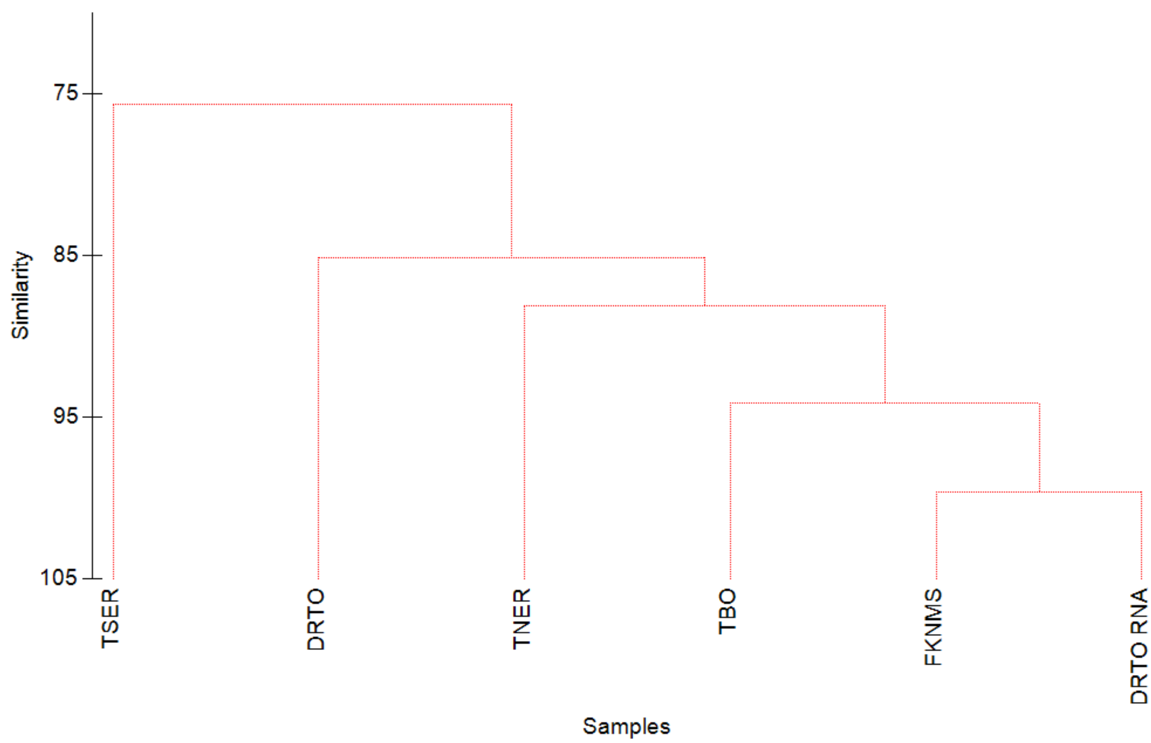
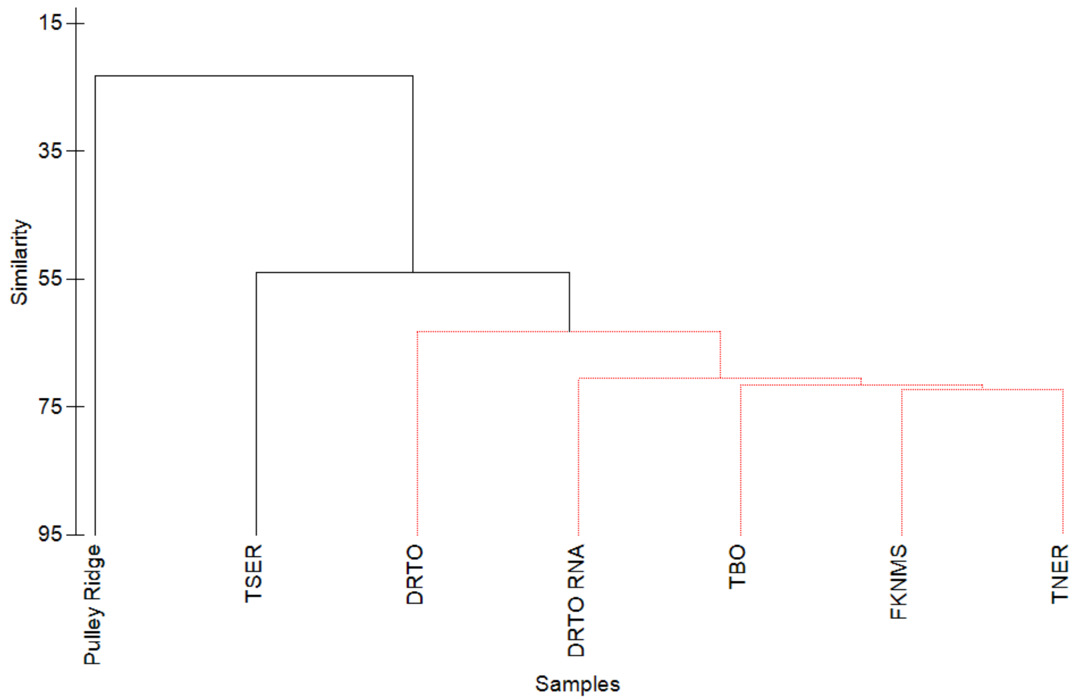


Fig. 7. (A) Dendrogram of the Bray-Curtis cluster analysis similarity for the habitat composition among management zones. (B) Dendrogram of the Bray-Curtis cluster analysis similarity for the habitat composition among management zones without Pulley Ridge.

(A)



(B)

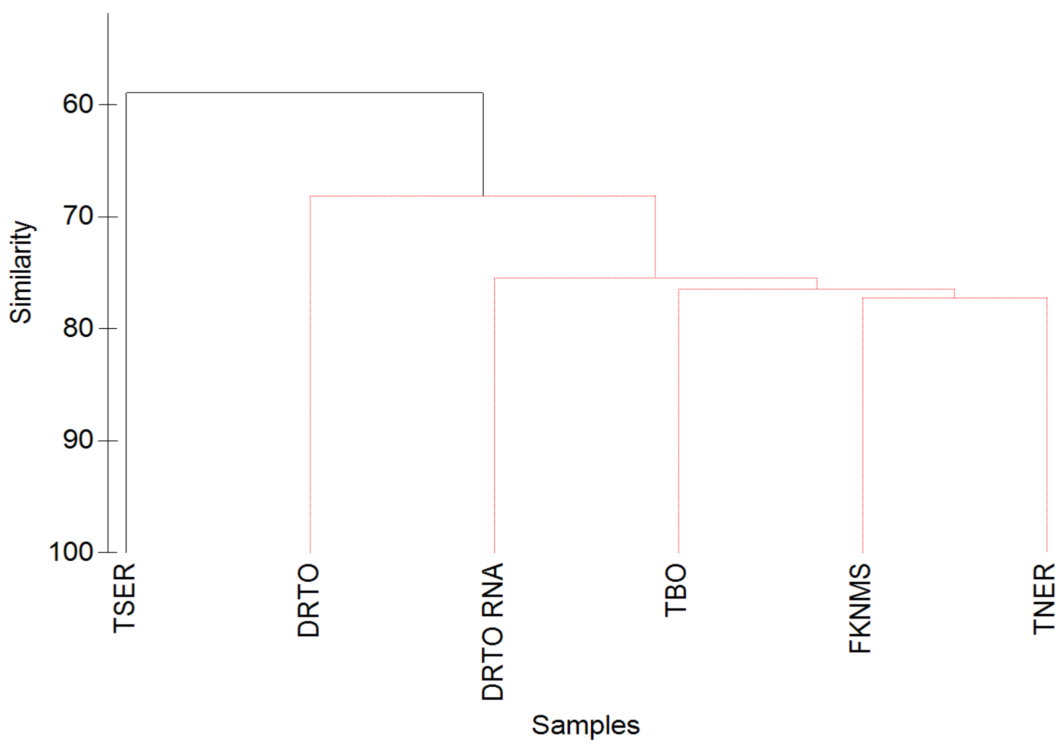


Fig. 8. (A) Dendrogram of the Bray-Curtis cluster analysis similarity for the fish community among management zones with Pulley Ridge and (B) without Pulley Ridge.

p-value of 0.001) (Fig. 8A). The dendrogram showed that fish communities at Pulley Ridge were only 28% similar to those in the other management blocks whereas TSER fish communities were 58% similar to the other management zones. When Pulley Ridge was removed from the fish community analysis, only TSER remained significantly different from the other management zones ((Pi = 3.51, significance percentage = 0.2%; equivalent to a p- value of 0.002) (Fig. 8B).

Unlike the SIMPROF and cluster analyses, the analysis of similarity (ANOSIM) showed a few more significantly different pairs among the management blocks for the habitat composition (Global R = 0.371, significance percentage = 0.1%; equivalent to a p- value of 0.001, Table 5). In this analysis, the Global R that was greater than 0.4 was considered significant, values below 0.4 were not considered significant regardless of how small the p-value. By selecting Global R values greater than 0.4 provides stronger evidence that the pairwise comparisons are actually significantly different (James Tolan, Texas Parks and Wildlife Department, personal communication). In this analysis, nearly half the pairwise-comparisons of management zones were significantly different from one another, the most significant differences were comparisons with Pulley Ridge and TSER.

Unlike the SIMPROF and cluster analyses, the analysis of similarity (ANOSIM) indicated significant differences among management blocks for the fish community (Global R = 0.38; significance percentage = 0.1%; equivalent to a p- value of 0.001, Table 6). Many pairwise-comparisons of management blocks were also significantly different from one another. Similar to the ANOSIM for habitat composition, the pairwise-comparisons that were significantly different were with Pulley Ridge and TSER.

Table 5
Matrix of pairwise comparisons from ANOSIM analysis to determine differences in habitat composition among management zones in the Dry Tortugas, Florida. Gray shading and italicized R values indicate significantly different comparisons (R > 0.40).

	FKNMS	DRTO RNA	DRTO	TNER	TBO	TSER	Pulley Ridge
FKNMS							
DRTO RNA	R = -0.013 Sig = 48.6						
DRTO	R = 0.194 Sig = 2.5	R = 0.012 Sig = 39.4					
TNER	R = 0.158 Sig = 1.7	R = 0.262 Sig = 0.2	R = *0.437* Sig = 0.1				
TBO	R = 0.081 Sig =11.2	R=-0.008 Sig =40.1	R = 0.007 Sig = 38.1	R = 0.339 Sig = 0.1			
TSER	R = *0.43* Sig =0.4	R = 0.328 Sig = 0.6	R = *0.63* Sig = 0.3	R = *0.469* Sig = 0.5	R = *0.441* Sig =0.5		
Pulley Ridge	R = *0.966* Sig = 0.1	R = *0.845* Sig = 0.2	R = *0.917* Sig =0.1	R = *0.974* Sig = 0.1	R = *0.713* Sig = 0.1	R = *0.526* Sig =0.6	

4. Discussion

The intent of this study was to evaluate the responses of fish community structure and habitat coverage following natural and anthropogenic disturbances. This broad objective was evaluated by aggregating data within individual management zone through time (i.e. years), by management regime but using years as replicate samples, and by within individual management zones and using years as replicates. The analysis of change through time showed some evidence of community change following the establishment of various MPAs but no evidence that I could detect community change following natural disturbances. The analysis of community differences by management regimes (i.e. level of MPA protection) showed significant differences between the level of protection

Table 6
Matrix of pairwise comparisons from ANOSIM analysis to determine differences in fish community structure among management blocks in the Dry Tortugas, Florida. Gray shading and italicized R values indicate significantly different comparisons (R > 0.40).

	FKNMS	DRTO RNA	DRTO	TNER	TBO	TSER	Pulley Ridge
FKNMS							
DRTO RNA	R = 0.009 Sig = 40.1						
DRTO	R = 0.306 Sig = 0.5	R = 0.215 Sig = 1.8					
TNER	R = 0.207 Sig = 0.1	R = 0.257 Sig = 0.8	R = *0.425* Sig = 0.1				
TBO	R = -0.046 Sig =77.7	R=-0.142 Sig =98.6	R = 0.23 Sig = 1.5	R = 0.203 Sig = 0.2			
TSER	R = *0.606* Sig =0.2	R = *0.549* Sig = 0.1	R = 0.301 Sig = 3.3	R = *0.56* Sig = 0.3	R = 0.399 Sig =0.5		
Pulley Ridge	R = *0.953* Sig = 0.1	R = *0.946* Sig = 0.1	R = *0.51* Sig =0.3	R = *0.798* Sig = 0.2	R = *0.878* Sig = 0.2	R = *0.729* Sig =0.1	

afforded by the MPA. The lowest of protection (i.e. open access areas) was significantly from the middle (i.e. marine reserve, DRTO-RNA) and the high (i.e.ecological reserves, TNER and TSER). The analysis of community differences by management zones showed significant differences between the specific regions and those appear to largely be driven differences in reef type. The highest protected management zone (TNER) showed significant changes following implementation. This zone also contained higher abundances of many commercially important fishery species when compared with open access areas. These abundances show that the MPA was successful for increasing abundances, maintaining a relatively consistent diversity index, as well as the appearance of black grouper, jacks, and mackerel following the creation of the TNER.

4.1 Changes in habitat and fish community structure within management zones over time

The TNER management zone showed significant differences in fish community structure through time and those differences appear after the establishment of NTMR regulations in 2001 and over the same time frame the TNER MPA showed no changes in habitat composition. Given that the only apparent change during that time was implementation of the MPA in 2001 it appears as though the implementation of the TNER had an effect on the associated fish community. Importantly the creation of the MPA effectively removed fishing pressure in this area and this coincided with the reappearance of several important recreational and commercial fisheries species (e.g. black grouper) observed in this study and in Ault et al. (2012). Both fish community structure and habitat composition in the FKNMS and TBO, which are open access areas, demonstrated some variation but did not show significant changes in community structure through time. The lack of response in the fish community in both the FKNMS and TBO management makes sense given that fishing pressure was maintained in those areas. Fishing appears to alter communities through removal of targeted individuals and at least in the TNER a change in management practice was detectable.

Other regions of the Dry Tortugas have not shown as clear a response as TNER following the establishment of an NTMR (e.g. DRTO RNA). It is difficult to determine why this region did not show an equivalent response through time however it could be the result of low sample sizes, inconsistent coverage through time, or the DRTO RNA region was simply more stable than the TNER region. Timelines of implementation of the various management strategies within the zones may also contribute to both the significant and non-significant comparisons. For example, pairwise comparison between

FKNMS and DRTO RNA was not significant probably because DRTO RNA was not implemented until 2007, prior to which DRTO RNA was included within the FKNMS boundary. There were likely too few years after implementation for significant differences to be measurable which is supported other long-term studies of MPAs (Barrett et al., 2007; Selig and Bruno, 2010; and Ault et al., 2012).

4.4 Possible effects of natural disturbances

In general it does not appear that natural disturbances altered fish community structure within management zones over time despite evidence that the area was impacted by many types of disturbances over the time span the survey was conducted. The habitat composition and fish communities in both protected and open access areas were subjected to large scale disturbances during the time period for which data were available and analyzed. These large scale disturbances were El Niño (1997–1998), dark plume (Fall 2003), hurricanes (2004–2006), and a cold water stress event (January 2010). Community structure did not show any consistent changes over management zones that could be temporally related to natural disturbances that I knew of. Some of the slight differences appear to be the result of the interaction between small sample sizes (e.g., TSER) which resulted in poor spatial and temporal coverage. For example in TBO, which is open access, there were declines in overall average abundances during years when the area was hit by hurricanes while abundances appear to increase in TERN which is a no-take reserve. Jones and Syms (1998) discussed numerous studies that document changes in trophic assemblages and abundances of reef fish species after a hurricane. In one study they reviewed, a trophic phase-shift led to increased abundances of rubble-associated species and a decline in coral-associated species such as the threespot damselfish

(*Stegastes planifrons*) and the yellow tang (*Zebrasoma flavescens*). In this study, while average abundance after a hurricane declined in TBO, diversity was relatively constant, perhaps indicating that the disturbances were not severe enough to cause a significant change in community structure over time; which was borne out by the lack of differences among years shown in the community analysis.

Unlike this analysis other studies have documented changes in species specific abundance following natural disturbances for instance after six hurricanes in the Dry Tortugas region (Charley, 2004; Ivan, 2004; Jeanne, 2004; Dennis, 2005; Wilma, 2005; Ernesto, 2006) declines in abundances of fishery species such as groupers (*Mycteroperca bonaci* [Smith et al., 2011], *Epinephelus morio* [Ault et al. 2012]), and hogfish species (Ault et al., 2012) were documented. However, Ault et al. (2012) also observed increases in yellowtail snapper and mutton snapper during visual surveys in 2008-2010. Fish mortalities have been recorded in relation to cold water intrusions (Roberts et al. 1982 and Colella et al. 2012), but the both fish communities and habitat composition seemed to be resilient. In most management zones, abundances decreased in 2010 which was the same year as the record breaking cold snap, but abundances increased in all blocks in 2011 except the DRTO RNA. There is little evidence that the fish community was affected by the natural disturbances that occurred during the years analyzed for this study.

Habitat coverage within some management zones varied greatly in percent cover of hard coral, soft coral, and algae (e.g., TERS, Pulley Ridge), while others showed a gradual increase and/or decrease in cover which may be indicative of effects from disturbances (FKNMS, TBO), but may also just be an artifact of the sampling design. The hard coral in FKNMS may have been damaged as evidenced by its absence in the areas surveyed during the three years following the hurricanes in 2004-2006. Algae and soft coral dominated at survey sites during those years and cover continued to increase up until the last year of data analyzed in this study, 2012. During 2005-2009 there were greater abundances of the herbivorous species surgeonfish and the other tang group (*Acanthurus* sp.) possibly due to the dominance of algae at sites where reef fish surveys were conducted. The regulations passed in 1997 did not protect hard coral habitat, and did not appear to promote hard coral coverage. Given the strong relationship between species assemblages and specific habitat types it will be important to stratify sampling designs by habitat to ensure that changes in population indices or community structure are not simply an artifact of small samples and poor spatial coverage.

Inability to detect community change following natural disturbances might have been the results of insufficient spatial and temporal sampling that resulted in reduced power to detect an effect if it was present. Fish communities may have recovered within a time scale that was incapable of capturing given the sampling frequency. Rapid fish species accumulation was observed in a study done by Roberts et al. (2001) and found that biomass increased faster overtime in a marine reserve when compared to an open access area. The ability of a fish community to return to its previous state following a disturbance depends on its linkages with nearby communities and larval supply from outside areas, and the Dry Tortugas is an important source of fish larvae (Bengtsson et al., 2003; Ault et al., 2012). Another possible explanation was that the video reads might not have captured the correct set of fish that were affected or the analysis required a full complement of species to detect an effect if it was present. Considering the numerous species that inhabit the reef and differ in trophic regimes the data I used is only a snapshot of the whole community. This dataset was originally designed for fishery stock management and not intended for ecosystem-based analyses which may need more species and perhaps more samples to capture community changes and diversity indices. Another possibility is that due to the mobile nature of many fish species they perhaps are unaffected by these disturbances. Large juveniles and adult fish species have been observed to move away from natural disturbances to areas not affected, including bottom dwelling species (Breitburg, 1992).

Finally, more time may be needed than the time series provided by this study for fish species to respond. Many other studies, significant differences, if any, from implementation of a marine protected area were not apparent until almost a decade later (Barrett et al., 2007; McClanahan et al., 2007; Ault et al., 2012). In one long-term study of marine reserves, Barrett et al. (2007) noted that species abundance varied greatly within and between protected areas. Variability within and among differing management regimes was observed in my study, and the temporal structure of the dataset made it difficult to associate known disturbances with differences in community structure. Lester and Halpren (2008) also noted that it is difficult to separate variations in reef fish abundances that result from disturbances, management implementation, and natural cycles. Peak biomass and densities of reef fish were not observed until seven years after implementation of the four marine national parks in Kenya (McClanahan et al. 2007). In my study, the timing of samples done in the TNER allowed me to detect changes within, but for the FKNMS and TBO this was not the case. This may be due to fluctuations in recruitment of marine fishes (Neill et al., 1994). Since there were no samples included in this study prior to implementation in 1997, and during the years 1998- 2001, it is possible the study could have missed detecting a strong recruitment year class. Groupers and Snappers are known to only produce good year classes every 5-10 years (Cowan et al. 2011).

Importantly, the relative difficulty of this analysis to detect community change through time might indicate that this survey data, or the analytical method used, might not be adequate to detecting these changes in community structure. In well sampled regions there was only one significant change in community structure (TNER). It will be important in the future to understand limitations of the survey data analyzed here, and that were intended to be used in single stock fisheries management schemes, relative to the implementation of ecosystem-based management schemes. This SEAMAP reef fish data set is based on detection of commercially and recreationally important species. The Dry Tortugas fish community is composed of many ornamental species and many of those are included in the video reads, it will be important to include different species and trophic groups in the future. These changes may help in detection of anthropogenic and/or natural disturbances if habitats are affected and different trophic groups are altered.

4.5 Grouping management zones by regimes

The SIMPROF and ANOSIM analysis showed significant groupings by management regimes when using the years as replicate samples. The lowest level of MPA management was significantly different from both the middle and highest level of management regimes. These results make sense considering the similarities between fishing regulations between the middle and highest regimes in which both of the regimes have a no-take policy but differ only slightly in other usages. The lower R statistic between the lowest and highest was surprising. However, the middle (DRTO RNA) management regime is designated for research, and when research is being conducted many times the area is blocked off from others using the area which may make this area more protected at times than the ecological reserves. The average abundance of the yellowtail snapper contributed most toward the dissimilarity between the lowest and middle management regimes. The average dissimilarity was also the largest between the lowest and middle groups. These two groups also had the largest R statistic for the ANOSIM test. The average dissimilarity between the other comparisons was only 3%, which for the ANOSIM test; the R statistic values were very close.

The analysis of the fish community by management regime likely benefitted from the resolution gained by the increase in sample size and the subsequent increase in power to detect differences when present. The number of treatments in this portion of the analysis was narrowed down to reflect logical groupings that I anticipated might result in a change in community. In areas where fishing pressure is reduced I expected that popularly targeted commercial and recreational species might benefit. Additionally using year as a replicate allowed for more information into the model. Alternatively the difference between evaluating the data by regime rather than looking at specific zones through time was that perhaps those different zones, with slightly different habitats, might be responding over different time scales. So treating the management regimes as a group accounted for those differences. Additionally individual species are perhaps responding quicker in certain habitats than in others, or more stable habitats are responding more slowly, but on average those regime differences begin to emerge with enough time and information gathered.

4.2 Differences in community structure among management zones

The differences I detected in community structure by specific management zone appear to largely be driven by habitat type. For instance inclusion of Pulley Ridge in the cluster analysis showed that habitat composition was significantly different from all other zones. Pulley ridge being significantly different was expected due to the marked differences in its physical parameters, most notably; its substantially deeper depth compared with the other management zones. Pulley Ridge was included in the analysis in part because it was expected to be different, thus it acted as a positive control. However when Pulley Ridge was removed from the analysis, there were no differences in habitat composition, although TSER was slightly different than all other management zones which was also recognized by Ault et al. (2012).

Both TERS and Pulley Ridge were significantly different in fish community composition from all other management zones for the SIMPROF test. These differences are most likely due to different habitat composition as well. Fish tagging studies, as mentioned by Ault et al. (2012) have shown fish species migrating between open zones, DRTO, and DRTO RNA due to similar habitat composition which may be why there were high similarities between those management zones in my study.

The SIMPROF test was run on *a priori* unstructured data at a set alpha (5%) to determine if the null hypothesis would be rejected or accepted, and if rejected, more in-depth analysis would be needed. Since there were significant indications of multivariate structure within and among management zones, an ANOSIM (*a priori* structured test) was conducted to investigate differences between management zones (designated factors) for habitat composition and fish communities (Clarke et al., 2008). In fact, the results from the ANOSIM were very similar to the SIMPROF results for both habitat composition and fish communities when designating the R statistic values greater than 0.4 to be significant. Most comparisons with TSER and Pulley Ridge were significant for both habitat composition and fish communities in the ANOSIM results. Furthermore, the only difference between the ANOSIM results for habitat composition and fish communities was the comparison between DRTO RNA and TSER, which was significantly different in the fish communities for those zones. Both tests yielded similar results of TSER and Pulley Ridge being the only significantly different zones for both habitat composition and fish communities. The strong similarities between these results of the two tests strongly indicate the relationship between habitat composition and fish communities are more likely the reason for differences found between TSER and Pulley Ridge to other zones, than management implementation. The habitats within the TNER, FKNMS, DRTO, DRTO RNA, and TBO are interconnected to one another, and since fish do not know boundaries, the results seem valid. The significant difference between DRTO and TNER appear in both habitat composition and reef fish community, and may be an artifact of dissimilarity between the habitats and possibly the difference in fishing practices. The non-significant differences for the fish communities between the highest protected zone (TSER) and open access areas (DRTO and TBO) are unclear but may be an artifact of unsufficient sampling for TSER to detect differences between those zones.

The yellowtail snapper was largely the most discriminating taxa among the management zones, and was most abundant in the TNER followed by the DRTO RNA. The yellowtail snapper and white grunt average abundance in TSER were very low and were not seen in Pulley Ridge, which can also account for the significant differences from the other zones because these areas differ in habitat composition and where yellowtail snapper inhabit. The yellowtail snapper had a large contribution towards the dissimilarity between the TNER and TSER, nearly 20% and 24% toward the dissimilarity between TNER and Pulley Ridge. Grunt species were the next discriminating taxa for the management zone comparisons.

4.6 Summary and conclusions

Fish communities appear to change in structure when MPA spatial management zones are created, however the change does not appear to be consistent and always respond in sync with the application of the regulation. Fish communities may be responding over longer time periods, and differentially depending on the stability of the area of MPA establishment. Changes in fish community composition associated with natural disturbances were not apparent. The time spanned by this study may not have been long enough to capture any natural cycles in the community as a whole, whereas some cycles at the species level were apparent (e.g. yellowtail snapper; Ault et al., 2012). It is also possible that the natural disturbances that did occur were not severe enough to cause more than short-term (days, weeks, months) change thus our inability to detect community change in this data set. It is possible the within management zone analysis illustrated management effect for TNER, because this was the only zone that demonstrated significant differences after implementation. Possible explanations of non-significant differences in community composition evaluated through time include low sample sizes, reduced number of species recorded, and incomplete spatial-temporal coverage relative to either the timing of the disturbance or the speed of recovery of fish communities.

The different management zones are very close to one another and appear to have some quite a lot of overlap in community composition. The two environments that were significantly different in fish community composition, one also being significantly different in habitat community composition (Pulley Ridge), and the other (TSER), was clustered out separately. Another hypothesis is that changes to community structure within MPAs takes time (Barrett et al. 2007) and enough of it may not have elapsed to observe an effect in our data. Selig and Bruno (2010) suggest the benefits from MPAs increase as the time since the MPA was first established increase.

There is a limit to how much protection MPAs can confer on marine ecosystems and while they can help limit the occurrence of disturbances due to human activities, they cannot limit the impacts of natural disturbances. They can, however, promote resilience of fish and benthic communities so that when disturbances occur, communities are able to recover more quickly. In this study species diversity remained relatively constant from year to year at most sites, and for the most part there were no significant differences (except TNER) in community structure within management zones despite a number of large scale disturbances. This suggests that reef fish communities are resilient within the study area. The patterns of differences between management zones in pairwise comparisons did not provide a clear picture of effects of restrictions but rather suggests that habitat differences in TSER and Pulley Ridge likely resulted in differences in fish community composition. Alternatively having restricted areas in close proximity to open access areas is beneficial to both types of management zones, and probably more so to the open access area due to spill-over effects associated with the reserve (Garcia-Rubies et al., 2013). It is possible that the presence of MPAs in the Dry Tortugas promotes resilience in fish communities at a larger scale than the individual management zones. In general, the abundances of exploited species (e.g. yellowtail snapper, white grunt, members from suborder Labroidei, bar jack, red hind, etc) were greater in no-take marine reserves (TERN and TERS). The analysis for the management regimes also reflected these trends, having the marine reserve and ecological reserves less different from one another than the open access areas compared with the marine reserve which was significantly different.

Whether or not MPAs are affecting or supporting fish community resilience and function may be clearer with additional analysis of the long-term reef fish community data set that is being developed in the Dry Tortugas. However, the current sampling protocol should be modified to include more if not all species for future use in ecosystem-based modeling approaches to fisheries management.

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Appendices

Appendix.1.0 List of species included in the SEFSC-SEAMAP reef fish survey.

Groupers & Sea basses
Epinephelus adscensionis
E. drummondhayi
E. flavolimbatus
E. guttatus
E. itajara
E. morio
E. mystacinus
E. nigritus
E. niveatus
E. striatus
Mycteroperca acutirostris
M. bonaci
M. interstitialis
M. microlepis
M. phenax
M. tigris
M. venenosa
Cephalopholis cruentatus
C. fulvus
Dermatolepis inermis
Paranthias furcifer
Centropristis ocyurus
C. striata
Diplectrum formosum
Bigeyes
Cookeolus japonicus
Priacanthus arenatus
P. cruentatus
Pristigenys alta
Tilefishes
Caulolatilus chrysops
C. cyanops
C. intermedius
C. microps
Lopholatilus chamaeleonticeps
Malacanthus plumieri
Snappers
Apsilus dentatus
Etelis oculatus
Lutjanus analis
L. apodus
L. buccanella
L. campechanus
L. cyanopterus
L. griseus
L. jocu
L. mahogani
L. synagris
L. vivanus
Ocyurus chrysurus
Pristipomoides aquilonaris
Rhomboplites aurorubens
Barracudas
Sphyraena barracuda
Squirrelfishes
Holocentrus adscensionis
H. rufus
Porgys
Archosargus probatocephalus
Calamus bajonado
C. calamus
C. leucosteus
C. nodosus
C. proridens
Pagrus pagrus
Jacks
Seriola dumerili
S. fasciata
S. rivoliana
S. zonata
Elagatis bipinnulata
Caranx bartholomaei
C. crysos
C. hippos
C. latus
C. lugubris
C. ruber
Alectis ciliaris
Trachinotus carolinus
T. falcatus
Grunts
Anisotremus surinamensis
A. virginicus
Haemulon album
H. aurolineatum
H. carbonarium
H. flavolineatum
H. macrostomum
H. melanurum
H. parra
H. plumieri
H. sciurus
Orthopristis chrysoptera

Appendix 1.0 continued

Goatfishes
Mulloidichthys martinicus
Mullus auratus
Pseudupeneus maculatus
Upeneus parvus
Spadefish
Chaetodipterus faber
Surgeonfishes
Acanthurus bahianus
A. chirurgus
A. coeruleus
Sea Chubs
Kyphosus incisor
K. sectatrix
Wrasses
Lachnolamus maximus Halichoeres radiatus
Parrotfishes
Scarus coelestinus
S. coeruleus
S. guacamaia
S. taeniopterus
S. vetula
Sparisoma chrysopterum
S. rubripinne
S. viride
Triggerfishes
Balistes capriscus
B. vetula
Canthidermis maculata
C. sufflamen
Melichthys niger
Xanthichthys ringens
Boxfishes
Lactophrys bicaudalis
L. polygonia
L. quadricornis
L. trigonis
L. triqueter
Coastal Migratory Pelagics
Scomberomorus cavalla
S. maculatus
S. regalis
Rachycentron canadum
Euthynnus alletteratus
Pomatomus saltatrix

Appendix 2.0

Taxa that were combined to represent new taxonomic group for data analysis, the following species and families were combined together due to low abundance numbers. Species with * have had scientific name recently changed that may not match what was originally recorded.

Taxa combined	Common name	New taxonomic group
Calamus		*Calamus* sp.
Calamus bajonado	(Jolthead porgy)	*Calamus* sp.
Calamus leucos	(Whitebone porgy)	*Calamus* sp.
Calamus proridens	(Littlehead porgy)	*Calamus* sp.
Caranx	(Jack)	Carangidae
Haemulon sp.		*Haemulon* sp.
Haemulon album	(Margate)	*Haemulon* sp.
Haemulon aurolineatum	(Tomtate)	*Haemulon* sp.
Haemulon carbonarium	(Caesar grunt)	*Haemulon* sp.
Haemulon flavolineatum	(French grunt)	*Haemulon* sp.
Haemulon macrostomum	(Spanish grunt)	*Haemulon* sp.
Haemulon melanurum	(Cottonwick grunt)	*Haemulon* sp.
Haemulon sciurus	(Blue striped grunt)	*Haemulon* sp.
Lutjanidae	(Snapper)	*Lutjanus* sp.
Lutjanus sp.		*Lutjanus* sp.
Lutjanus apodus		*Lutjanus* sp.
Lutjanus buccanella	(blackfin snapper)	*Lutjanus* sp.
Lutjanus campechanus	(Red snapper)	*Lutjanus* sp.
Lutjanus jocu	(Dog snapper)	*Lutjanus* sp.
Scaridae		Labroidei
Scarus sp.	(Parrotfish)	Labroidei
Scarus coeruleus	(Blue parrotfish)	Labroidei
Scarus coelestinus	(Midnight parrotfish)	Labroidei
Scarus guacamaia	(Rainbow parrotfish)	Labroidei
Scarus taeniopterus	(Princess parrotfish)	Labroidei
Scarus vetula	(Queen parrotfish)	Labroidei
Labroidei		Labroidei
Labridae		Labroidei
Seriola sp.	(Jack)	*Seriola* sp.
Seriola dumerili	(Greater amberjack)	*Seriola* sp.
Seriola fasciata	(Lesser amberjack)	*Seriola* sp.
Seriola rivoliana	(Longfin yellowtail)	*Seriola* sp.
Sparisoma sp.	(Parrotfish genus)	*Sparisoma* sp.
Sparisoma chrysopterum	(Redtail parrotfish)	*Sparisoma* sp.
Sparisoma rubripinne	(Redfin parrotfish)	*Sparisoma* sp.
Acanthurus sp.		*Acanthurus* sp.
Acanthurus bahianus	(Ocean surgeon)	*Acanthurus* sp.

Appendix 2.0 continued.

Balistes sp.	(Triggerfish)	*Balistes* sp.
Balistes vetula	(Queen Triggerfish)	*Balistes* sp.
Balistes capriscus	(Grey Triggerfish)	*Balistes* sp.
Epinephelus sp.	(Grouper)	*Epinephelus* sp.
Epinephelus adscensionis	(Rock hind)	*Epinephelus* sp.
Epinephelus guttatus	(Red hind)	*Epinephelus* sp.
Epinephelus itajara	(Goliath)	*Epinephelus* sp.
Halichoeres sp.	(wrasses)	*Halichoeres* sp.
Holocentrus sp.	(Squirrelfish)	*Holocentrus* sp.
Halichoeres radiatus	(Puddingwife wrasse)	*Halichoeres* sp.
Holocentrus adscensionis	(Squirrelfish)	*Holocentrus* sp.
Holocentrus rufus	(Longspine squirrelfish)	*Holocentrus* sp.
Lactophrys sp.		*Lactophrys* sp.
Lactophrys bicaudalis	(Spotted trunkfish)	*Lactophrys* sp.
**Lactophrys quadricornis*	(Scrawled cowfish) now *Acanthostracion quadricornis*	*Lactophrys* sp.
Lactophrys trigonus	(Buffalo trunkfish)	*Lactophrys* sp.
Mycteroperca interstitialis	(Yellowmouth grouper)	*Mycteroperca* sp.
Mycteroperca microlepis	(Gag)	*Mycteroperca* sp.
Mycteroperca tigris	(Tiger grouper)	*Mycteroperca* sp.
Mycteroperca venenosa	(Yellowfin grouper)	*Mycteroperca* sp.
Scombridae		Scombridae
Scomberomorus maculatus	(Spanish mackerel)	Scombridae

Appendix 3.1. Average and standard deviation of fish species observed in the Florida Keys National Marine Sanctuary (FKNMS). * Indicates value below 0.05.

Taxon	1997	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Acanthurus coeruleus	1.1	0.4	0.0	0.0	0.2	0.9	0.8	0.8	0.1	0.8	0.5	0.6
	(1.2)	(0.7)			(0.6)	(2.1)	(1.1)	(1.0)	(0.4)	(0.9)	(1.0)	(1.1)
Acanthurus sp.	0.9	0.3	0.0	0.0	0.2	0.6	1.0	0.2	0.0	0.1	0.0	0.3
	(1.3)	(0.5)			(0.6)	(1.2)	(1.2)	(0.6)		(0.3)		(0.8)
Anisotremus virginicus	0	0	0	0	0	0	0	0	0	0.1	0.1	0.0
										(0.3)	(0.3)	(0.1)
Balistes sp.	0	0	0	0	0	0.1	0.2	0.0	0.0	0.1	0.0	0.0
						(0.2)	(0.5)			(0.3)		(0.2)
Calamus sp.	1.7	0.3	0.0	1.6	0.5	0.5	1.0	0.1	0.8	0.6	0.7	0.7
	(2.1)	(0.5)		(2.2)	(1.3)	(0.9)	(2.2)	(0.3)	(1.0)	(1.4)	(1.1)	(1.3)
Calamus calamus	0.6	0.1	0.0	0.0	0.6	0.8	0.6	0.5	0.1	0.8	0.2	0.5
	(0.8)	(0.4)			(0.8)	(2.0)	(0.6)	(0.7)	(0.4)	(1.2)	(0.6)	(1.0)
Carangidae	0.5	0	0.4	0	0.5	0.1	0	0	0	0	0	0
	(0.9)		(0.6)		(1.6)	(0.5)						
Caranx bartholomaei	0	0	0.8	1.4	0.6	0.6	0.0	0.0	0.4	0.6	1.8	0.5
			(1.8)	(3.1)	(1.6)	(1.7)			(1.2)	(2.0)	(3.7)	(1.8)
Caranx crysos	0.3	0.0	2.4	0.0	0.7	0.4	0.0	2.0	0.2	0.0	0.4	0.5
	(0.8)		(5.4)		(1.3)	(1.0)		(4.1)	(0.8)		(0.7)	(1.9)
Caranx ruber	0.4	0.3	0.4	0.0	0.1	0.0	0.0	0.4	0.1	0.3	4.3	0.6
	(0.9)	(0.7)	(0.9)		(0.3)			(1.0)	(0.3)	(0.9)	(12.6)	(3.8)
Centropristis ocyurus	0	0	0	0	0	0	0	0	0	0	0	0
Cephalopholis cruentata	0.4	0.1	0.2	0.0	0.3	0.0	0.0	0.0	0.0	0.2	0.2	0.1
	(0.5)	(0.4)	(0.5)		(0.7)					(0.4)	(0.4)	(0.4)

Appendix 3.1 continued.

Taxon	1997	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Cephalopholis fulva	0.1											0.0
	(0.3)											(0.1)
Diplectrum formosum	0.1	0.3	0.0	0.0	1.3	0.5	0.0	0.2	0.6	0.4	0.3	0.4
	(0.3)	(0.7)			(1.4)	(0.9)		(0.4)	(1.2)	(1.0)	(0.7)	(0.9)
Diplectrum sp.	0	0.1	0	0	0	0	0	0	0	0	0	0.0
		(0.4)										(0.1)
Epinephelus sp.	0.1	0.3	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
	(0.5)	(0.7)		(1.3)								(0.4)
Epinephelus morio	0.4	0.5	0.2	0.4	0.2	0.1	0.0	0.2	0.3	0.8	0.4	0.3
	(0.5)	(0.5)	(0.5)	(0.6)	(0.4)	(0.3)		(0.4)	(0.5)	(0.8)	(0.5)	(0.5)
Euthynnus alletteratus	0	0	0	0	0	0.1	0	0	0	0	0	0.0
						(0.2)						(0.1)
Haemulon sp.	0.2	0.0	0.0	0.2	0.0	0.4	0.0	0.0	0.0	0.0	0.2	0.1
	(0.4)			(0.5)		(1.7)					(0.4)	(0.7)
Haemulon plumieri	1.1	0.6	0.0	0.0	0.7	0.4	0.6	0.7	0.4	2.4	3.1	1.0
	(1.0)	(1.2)			(1.1)	(0.8)	(0.6)	(0.9)	(0.5)	(3.7)	(7.8)	(2.8)
Holocentrus sp.	0.1	0.1	0	0	0	0	0	0	0	0	0	0.0
	(0.4)	(0.4)										(0.2)
Halichoeres sp.	1.1	0.6	0.0	0.0	0.0	0.0	0.0	0.2	0.1	0.1	0.1	0.2
	(1.6)	(0.7)						(0.4)	(0.3)	(0.3)	(0.3)	(0.7)
Kyphosus sp.	0	0	0	0	0	0	0	0	0	0	0	0
Labroidei	6.5	1.1	0.0	0.0	0.6	0.0	0.2	0.0	0.0	0.3	0.0	1.0
	(5.6)	(2.1)			(1.3)		(0.5)			(0.9)		(2.9)
Lachnolaimus maximus	0.4	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.3	0.5	0.2
	(0.7)	(0.4)			(0.3)	(0.3)				(0.7)	(1.0)	(0.5)

Appendix 3.1. Continued.

Taxon	1997	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Lactophrys sp.	0.5	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.1	0.1	0.0	0.1
	(1.6)				(0.4)				(0.3)	(0.3)		(0.6)
Lutjanus analis	0.4	0.9	0.0	0.4	1.1	0.4	0.4	0.1	0.3	1.1	1.8	0.6
	(0.9)	(1.4)		(0.6)	(1.9)	(0.5)	(0.9)	(0.3)	(0.5)	(1.5)	(4.0)	(1.5)
Lutjanus griseus	0.1	0.0	2.4	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.5	0.2
	(0.4)		(5.4)		(0.3)						(1.3)	(1.2)
Lutjanus sp.	0.1	0	0	0	0	0	0	0	0	0	0	*
	(0.4)											(0.1)
Lutjanus synagris	0	0	0	0	0	0.2	0	0.3	0	0	1.2	0.2
						(0.8)		(0.7)			(3.8)	(1.2)
Malacanthus plumieri	0.1	0.1	0.0	0.0	0.1	0.2	0.0	0.0	0.1	0.0	0.0	0.1
	(0.3)	(0.4)			(0.3)	(0.6)			(0.3)			(0.3)
Mulloidichthys martinicus	0	0	0	0	0	0	0	0	0	0	0	0
Mycteroperca bonaci	0.1	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.1	0.1	0.1	0.1
	(0.3)	(0.4)			(0.3)				(0.3)	(0.3)	(0.3)	(0.2)
Mycteroperca sp.	0	0	0	0	0	0	0	0	0	0	0.3	*
											(0.7)	(0.2)
Mycteroperca phenax	0	0	0	0	0	0	0	0	0	0	0	0
Ocyurus chrysurus	1.6	2.9	2.6	2.6	1.4	1.0	4.8	0.8	1.4	2.2	4.3	2.0
	(2.8)	(4.6)	(3.7)	(3.7)	(2.8)	(1.6)	(6.6)	(1.3)	(2.1)	(2.4)	(11.6)	(4.4)
Pagrus pagrus	0	0	0	0	0	0	0	0	0	0	0	0
Priacanthus arenatus	0	0	0	0	0	0	0	0	0	0	0	0
Pseudupeneus maculatus	1.6	1.0	0.0	0.2	0.3	0.9	0.6	0.8	0.1	0.2	0.1	0.6
	(2.3)	(0.8)		(0.5)	(0.7)	(1.5)	(0.9)	(1.1)	(0.4)	(0.4)	(0.3)	(1.2)

Appendix 3.1. Continued.

Taxon	1997	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Rhomboplites aurorubens	0	0	0	0	0	0	0	0	0	0	0	0
Scomberomorus sp.	0.1	0.0	0.2	0.8	0.3	0.1	0.0	0.0	0.1	0.3	0.1	0.2
	(0.4)		(0.5)	(1.8)	(1.0)	(0.3)			(0.3)	(0.5)	(0.3)	(0.5)
Scomberomorus regalis	0	0	0	0	0.2	0	0	0	0.1	0.3	0.0	0.1
					(0.4)				(0.4)	(1.2)		(0.4)
Scombridae	0	0	0.2	0	0	0	0	0	0	0	0	0.01
			(0.5)									(0.1)
Seriola sp.	0	0.1	0.0	0.0	0.0	0.2	0.0	0.0	0.1	0.3	0.1	0.1
		(0.4)				(0.7)			(0.3)	(0.6)	(0.3)	(0.4)
Serranidae	0.4	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
	(0.7)	(0.7)										(0.3)
Sparidae	0	0.1	0	0	0	0	0	0	0	0	0	0.0
		(0.4)										(0.1)
Sparisoma sp.	1.1	0.3	0.0	0.0	0.3	0.4	0.0	0.1	0.0	0.7	0.2	0.4
	(1.8)	(0.7)			(1.0)	(1.1)		(0.3)		(1.2)	(0.4)	(1.0)
Sparisoma viride	0.7	0.0	0.0	0.2	0.8	0.2	0.6	0.1	0.1	0.3	0.5	0.3
	(1.0)			(0.5)	(1.2)	(0.5)	(0.9)	(0.3)	(0.3)	(0.5)	(1.0)	(0.7)
Sphyraena barracuda	0.1	0.1	0.0	0.4	0.0	0.0	0.2	0.1	0.1	0.3	0.3	0.1
	(0.3)	(0.4)		(0.9)			(0.5)	(0.3)	(0.3)	(0.5)	(0.5)	(0.4)
Trachinotus falcatus	0	0	0	0	0	0	0	0	0	0	0.5	1.6
											*	(0.5)

Appendix 3.1. Continued.

Taxon	1997	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Diversity H' log10	1.2	1.2	0.8	0.9	1.3	1.3	0.9	1.0	1.1	1.2	1.1	
Species richness d	31.0	25.0	8.0	11.0	24.0	23.0	12.0	17.0	21.0	26.0	27.0	
Evenness J	0.8	0.8	0.8	0.9	0.9	0.9	0.80	0.8	0.8	0.9	0.8	
Average abundance of species per year	6.5	1.8	1.0	0.9	2.3	3.1	1.1	1.8	1.5	3.2	4.6	
SD regardless of species	(13.9)	(3.7)	(3.0)	(2.4)	(3.5)	(4.7)	(3.6)	(4.2)	(3.4)	(5.9)	(9.8)	
Number of sites read for	14	8	5	5	10	17	5	12	14	12	10	

Appendix.3.2 Average and standard deviation of fish species observed in the Dry Tortugas Research Natural Area (DRTO RNA). * Indicates value below 0.05.

Taxon	1997	2005	2007	2008	2009	2010	2011	2012	Total
Acanthurus coeruleus	0.7	0.7	0	2	0.3	1.0	0	0.3	0.7
	(0.6)	(1.2)		(2.1)	(0.5)	(1.7)		(0.8)	(1.2)
Acanthurus sp.	1.3	0.5	0.7	0.8	0.5	0	1.0	0	0.5
	(0.6)	(1.2)	(0.6)	(1.3)	(1.0)		(1.0)		(0.9)
Anisotremus virginicus	0	0	0	0	0	0	0	0	0
Balistes sp.	0	0	0	0	0	0	0	0	0
Calamus sp.	0.7	0	0	0	0	2.0	0	0.3	0.3
	(1.2)					(2.0)		(0.5)	(0.8)
Calamus calamus	0.3	0.3	0.3	0.6	0.5	0	0.3	0.1	0.3
	(0.6)	(0.8)	(0.6)	(0.6)	(0.6)		(0.6)	(0.4)	(0.5)
Carangidae	0	0	0	0	0	0	0	0	0
Caranx bartholomaei	0	0.2	0	0	0	0	0	1.3	0.3
		(0.4)						(2.0)	(1.0)
Caranx crysos	0	0.3	0	1.8	0	0	0	0.4	0.4
		(0.8)		(4.0)				(1.1)	(1.6)
Caranx ruber	7.0	0	1.0	5.4	0	0	0	0.3	1.6
	(12.1)		(1.7)	(3.6)				(0.5)	(4.1)
Centropristis ocyurus	0	0	0	0	0	0	0	0	0
Cephalopholis cruentata	0	0.3	0.3	0	0	0	0	0	0.1
		(0.8)	(0.6)					0	(0.4)
Cephalopholis fulva	0	0	0	0	0	0	0	0	0
Diplectrum formosum	0	1.0	0	0	0	0	0.7	0	0.2
		(1.6)					(1.2)		(0.8)
Diplectrum sp.	0	0.8	0	0	0	0	0	0	0.2
		(1.3)							(0.6)
Epinephelus sp.	0	0	0	0	0	0	0	0	0
Epinephelus morio	0.7	0.2	1.0	0	0	0.3	0.3	0.4	0.3
	(0.6)	(0.4)				(0.6)	(0.6)	(0.8)	(0.5)
Euthynnus alletteratus	0	0	0	0	0	0	0	0	0
Haemulon sp.	0	0	3.0	15.4	0	0	0	0	2.5
			(4.4)	(34.4)					(13.2)
Haemulon plumieri	0.3	0.2	1.3	0.8	0.3	0.3	1.0	0.7	0.6
	(0.6)	(0.4)	(1.2)	(0.8)	(0.5)	(0.6)	(1.0)	(1.5)	(0.9)
Holocentrus sp.	0	0	0	0	0	0	0	0	0

Appendix 3.2 continued

Taxon	1997	2005	2007	2008	2009	2010	2011	2012	Total
Halichoeres sp.	1.3 (0.6)	0	0	0.2 (0.5)	0	0	0	0.1 (0.4)	0.2 (0.5)
Kyphosus sp.	0	0	0	0	0	0	0	0	0
Labroidei	11.3 (2.5)	0.2 (0.4)	0	0	0	1.3 (2.3)	0	0	1.2 (3.4)
Lachnolaimus maximus	0	0	0.3 (0.6)	0.2 (0.5)	0.3 (0.5)	0.3 (0.6)	0	0	0.1 (0.3)
Lactophrys sp.	0	0	0	0	0	0	0	0	0
Lutjanus analis	1.0 (1.0)	0	0.7 (0.6)	0.4 (0.6)	0.3 (0.5)	0.7 (0.6)	0.6 (0.6)	0.4 (0.5)	0.4 (0.6)
Lutjanus griseus	0	0	0	0	0	0	0	0	0
Lutjanus sp.	0	0	0	0	0	0	0	0	0
Lutjanus synagris	0	0	0	0	0	0	0	0	0
Malacanthus plumieri	0	0	0	0	0	0	0	0.1 (0.4)	* (0.2)
Mulloidichthys martinicus	0	0	0	0	0	0	0	0	0
Mycteroperca bonaci	0.3 (0.6)	0	0	0	0	0	0	0	* (0.2)
Mycteroperca sp.	0	0	0	0	0	0	0	0	0
Mycteroperca phenax	0	0.2 (0.4)	0	0	0	0	0	0	* (0.2)
Ocyurus chrysurus	1.7 (0.6)	1.7 (4.1)	3.7 (4.7)	7.0 (6.6)	1.8 (2.4)	4.0 (3.6)	2.7 (3.8)	1.9 (2.8)	3.0 (4.0)
Pagrus pagrus	0	0	0	0	0	0	0	0	0
Priacanthus arenatus	0	0	0	0	0	0	0	0	0
Pseudupeneus maculatus	1.0 (1.0)	0.2 (0.4)	2.0 (1.7)	0.4 (0.6)	0.8 (1.0)	0	0.3 (0.6)	0.4 (0.8)	0.6 (0.9)
Rhomboplites aurorubens	0	0	0	0	0	0	0	0	0
Scomberomorus sp.	0	0.8 (1.6)	1.0 (1.0)	0	0.3 (0.5)	0	0.3 (0.6)	0.1 (0.4)	0.3 (0.8)
Scomberomorus regalis	0	0	0	1.0 (1.0)	0	0.3 (0.6)	0	0	0.2 (0.5)
Scombridae	0	0.3 (0.8)	0	0	0	0	0	0	* (0.3)

Seriola sp.	0	0	0	0	0	0	1.7 (2.1)	0	0.2 (0.7)
Serranidae	0	0	0	0	0	0	0	0	0
Sparidae	0	0	0	0	0	0	0	0	0
Sparisoma sp.	0.7 (0.6)	0	0.3 (0.6)	0.8 (1.8)	0	0	0	0.1 (0.4)	0.2 (0.7)
Sparisoma viride	1.0 (0)	0.3 (0.8)	0	0.8 (0.8)	0	0	0	0.1 (0.4)	0.3 (0.6)
Sphyraena barracuda	0	0.5 (1.2)	0	0.2 (0.5)	0.3 (0.5)	0	0	0	0.2 (0.6)
Trachinotus falcatus	0	0	0	0	0	0	0	0	0
Diversity H' log10	0.9	1.1	1.0	0.8	0.9	0.8	0.9	1.0	-
Species richness d	15.0	18.0	12.0	16.0	10.0	9.0	10.0	16.0	-
Evenness J	0.7	0.9	0.9	0.7	0.9	0.8	0.9	0.9	-
Average abundance of species per year	1.8	1.0	0.9	3.8	0.4	0.6	0.5	1.0	-
SD regardless of sp	(5.6)	(2.0)	(2.3)	(12.3)	(1.1)	(2.0)	(1.5)	(2.4)	-
Number of sites read for	3	6	3	5	4	3	3	7	-

A.3.3 Average and standard deviation of fish species observed in the Dry Tortugas (DRTO). * Indicates value below 0.05. # Indicates one sighting therefore no standard deviation can be calculated.

Taxon	1997	2003	2007	2008	2009	2010	2011	2012	Total
Acanthurus coeruleus	13.0	0	0	0	0	0	0.1	0	1.4
	(21.7)						(0.4)		(7.2)
Acanthurus sp.	2.3	0	0.2	0	0	0	0.1	0	0.3
	(2.5)		(0.5)				(0.4)		(1.0)
Anisotremus virginicus	0	0	0	0	0	0	0.14	0	0.04
							(0.4)		(0.2)
Balistes sp.	0	0	0	0	0	0	0	0	0
Calamus sp.	0.7	0	0	0	1.3	0	2.7	2.3	1.1
	(1.2)				(2.3)		(3.6)	(3.2)	(2.4)
Calamus calamus	0	0	0	0	0.3	0	0.4	0	0.1
					(0.6)		(0.8)		(0.5)
Carangidae	0	1.0	0	0	0	0	0	0	0.1
		(1.7)							(0.6)
Caranx bartholomaei	0	4.3	1.4	0	0	0	0.1	0.3	0.8
		(0.6)	(3.1)				(0.4)	(0.6)	(1.8)
Caranx crysos	0	2.3	1.2	5.0	0	0	0	0.7	0.7
		(2.1)	(2.2)	#				(0.6)	(1.5)
Caranx ruber	0	0	0	0	0	0	0	0.33	0.04
								(0.6)	(0.2)
Centropristis ocyurus	0	0	0	0	0	0	0	0	0
Cephalopholis cruentata	0.3	0	0.2	0	0	0	0	0.3	0.1
	(0.6)		(0.5)					(0.6)	(0.3)
Cephalopholis fulva	0	0	0	0	0	0	0	0	0
Diplectrum formosum	0	0.7	0.6	0	0.3	0.3	1.1	1.3	0.7
		(1.2)	(0.9)		(0.6)	(0.6)	(3.0)	(1.2)	(1.6)
Diplectrum sp.	0	0	0	0	0	0	0	0	0
Epinephelus sp.	0	0	0	0	0	0	0	0	0
Epinephelus morio	0.7	0.3	0.2	0	0	0	0.3	0	0.2
	(0.6)	(0.6)	(0.5)				(0.5)		(0.4)
Euthynnus alletteratus	0	0	0	0	0	0	0	0	0
Haemulon sp.	0	0	0.6	0	0	0	0	9.0	1.1
			(1.3)					(15.6)	(5.1)
Haemulon plumieri	2.3	0	1.2	0	0	0	1.3	0.3	0.8
	(2.1)		(2.7)				(1.6)	(0.6)	(1.6)

Appendix 3.3 continued

Taxon	1997	2003	2007	2008	2009	2010	2011	2012	Total
Holocentrus sp.	0	0	0	0	0	0	0	0	0
	(0.6)								(0.2)
Kyphosus sp.	0	0	0	0	0	0	0	0	0
Labroidei	4.7	0	0	0	0	0	0	0.3	0.5
	(2.3)							(0.58)	(1.60)
Lachnolaimus maximus	0.3	0	0	0	0	0	0.1	0	0.1
	(0.6)						(0.4)		(0.3)
Lactophrys sp.	0	0	0	0	0	0	0	0	0
Lutjanus analis	0.3	0.3	0	0	0	0	0.4	0	0.2
	(0.6)	(0.6)					(0.5)		(0.4)
Lutjanus griseus	0	0	0	0	0	0	0.7	0.7	0.3
							(1.9)	(1.2)	(1.0)
Lutjanus sp.	0	0	0	0	0	0	0	0	0
Lutjanus synagris	0	0.3	0	0	0	0	0	5.3	0.6
		(0.6)						(9.2)	(3.0)
Malacanthus plumieri	0	0	0	0	0	0	0	0	0
Mulloidichthys martinicus	0	0	0	0	0	0	0	0	0
Mycteroperca bonaci	0	0.3	0.2	0	0	0	0	0	0.1
		(0.6)	(0.5)						(0.3)
Mycteroperca sp.	0	0	0	0	0	0	0	0	0
Mycteroperca phenax	0	0	0	0	0	0	0	0	0
Ocyurus chrysurus	1.3	0.3	0	0	6.3	0	6.1	6.3	3.1
	(2.3)	(0.6)			(11.0)		(10.8)	(10.1)	(7.2)
Pagrus pagrus	0	0	0	0	0	0	0	0	0
Priacanthus arenatus	0	0	0	0	0	0	0	0	0
Pseudupeneus maculatus	0	0	0.4	0	0	0	0	0	0.1
			(0.9)						(0.4)
Rhomboplites aurorubens	0	0	0	0	0	0	0	0	0
Scomberomorus sp.	0	0.3	0	0	0	0	0.4	0	0.1
		(0.6)					(0.8)		(0.5)
Scomberomorus regalis	0	0	0	0	0	0	0	0	0
Scombridae	0	0	0	0	0	0	0	0	0
Seriola sp.	0	0	1.6	0	0	0	0	0	0.3
			(3.6)						(1.5)
Serranidae	0	0	0.2	0	0	0	0	0	*
			(0.5)						(0.2)

Appendix 3.3 continued

Taxon	1997	2003	2007	2008	2009	2010	2011	2012	Total
Sparidae	0	0	0	0	0	0	0 (0.38)	0	0 (0.19)
Sparisoma viride	0.3 (0.6)	0	0	0	0	0	0	0.3 (0.6)	0.1 (0.3)
Sphyraena barracuda	0.7 (1.1)	0	0	0	0.3 (0.6)	0	0	0	0.1 (0.4)
Trachinotus falcatus	0	0	0	0	0	0	0	0	0
Diversity H' log10	0.8	0.8	1.0	0	0.4	0	0.8	0.8	-
Species richness d	12.0	10.0	12.0	2.0	5.0	1.0	15.0	13.0	-
Evenness J	0.7	0.8	0.9	*	0.6	*	0.7	0.7	-
Average abundance of species per year	1.6	0.6	0.8	0.1	0.5	0.02	2.0	1.7	-
SD regardless of species	5.9	2.1	1.9	0.7	2.7	0.1	6.7	5.1	-
Number of sites read for	3	3	5	1	3	3	7	3	-

Appendix.3.4. Average and standard deviation of fish species observed in the Tortugas North Ecological Reserve (TNER). * Indicates value below 0.05. # Indicates one sighting therefore no standard deviation can be calculated.

Taxon	1997	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Acanthurus coeruleus	3.6	1.2	0	0.6	0.8	0.4	0.7	0.8	0.1	1.2	0.7	0.9
	(4.8)	(1.3)		(0.8)	(1.1)	(0.7)	(1.8)	(1.2)	(0.5)	(1.8)	(1.4)	(1.9)
Acanthurus sp.	1.4	1.0	0	0.1	0.7	0.7	0.3	0.4	0.3	0.5	0.1	0.5
	(1.4)	(2.0)		(0.3)	(0.9)	(0.8)	(0.6)	(1.0)	(0.7)	(1.2)	(0.5)	(1.0)
Anisotremus virginicus	0.1	0.1	0	0.1	0	*	0.1	0.1	0	0.1	0.1	0.1
	(0.3)	(0.3)		(0.3)		(0.2)	(0.2)	(0.3)		(0.2)	(0.3)	(0.2)
Balistes sp.	0.1	0	0	0	0	0	0	0	0.1	0.1	0	*
	(0.3)								(0.2)	(0.2)		(0.1)
Calamus sp.	0.6	0.4	0	0.4	0.3	0.4	0.4	0	0.8	1.1	0.3	0.4
	(0.7)	(0.6)		(0.6)	(0.5)	(0.9)	(0.5)		(0.8)	(2.3)	(0.9)	(1.0)
Calamus calamus	0.3	0.5	0	0.1	0.4	0.4	0.3	1.0	0.6	0.9	0.4	0.5
	(0.5)	(0.7)		(0.3)	(0.6)	(0.7)	(0.6)	(0.9)	(0.8)	(0.9)	(1.0)	(0.8)
Carangidae	0.2	1.5	0	0.2	0	0	0.1	0.1	0	0	0	0.2
	(0.6)	(3.0)		(0.6)			(0.5)	(0.2)				(1.0)
Caranx bartholomaei	0	0.2	1.0	0.1	0	0.1	0	0	0.2	0.1	0.2	0.1
		(0.7)	#	(0.3)		(0.4)			(0.4)	(0.3)	(0.6)	(0.4)
Caranx crysos	0	0.1	0	0	0	0	0	0.1	0	0.1	0.1	*
		(0.5)						(0.2)		(0.2)	(0.4)	(0.2)
Caranx ruber	0.9	3.8	0	0.2	0.4	1.1	0.8	0.9	0	0.4	0.7	0.9
	(2.3)	(11.2)		(0.4)	(1.0)	(3.5)	(1.3)	(1.4)		(1.0)	(1.4)	(4.0)
Centropristis ocyurus	0	0	0	0	0.2	0	0	0	0	0	0	*
					(0.7)							(0.2)

Appendix 3.4 continued.

Taxon	1997	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Cephaloph-olis cruentata	0.8	0.4	0	0.6	0.4	0.3	0.5	0.3	0.5	0.4	0.3	0.4
	(1.0)	(0.7)		(1.1)	(0.6)	(0.6)	(0.7)	(0.6)	(0.9)	(0.8)	(0.6)	(0.7)
Cephalopholis fulva	0	0	0	0	0	*	0	0.1	0	0	0	*
						(0.2)		(0.2)				(0.1)
Diplectrum formosum	0	0	0	0	0	0.3	0.1	0.1	0	0.1	0.1	0.1
						(0.7)	(0.2)	(0.3)		(0.2)	(0.6)	(0.4)
Diplectrum sp.	0	0	0	0	0	0	0	0	0	0	0	0
Epinephelus sp.	0	0	0	0	0.06	0	0	0	0	0	0	0.01
					(0.2)							(0.1)
Epinephelus morio	0.3	0.7	3.0	0.6	0.7	0.4	0.5	0.4	0.5	0.4	0.8	0.5
	(0.7)	(0.8)	#	(1.1)	(0.8)	(0.8)	(0.6)	(0.5)	(0.7)	(0.7)	(0.8)	(0.8)
Euthynnus alletteratus	0	0	0	0	0.1	0	0	0	0	0	0.1	*
					(0.2)						(0.4)	(0.2)
Haemulon sp.	2.9	0.3	0	0.7	0.8	0.7	0.1	0.3	0.1	1.0	0.5	0.7
	(7.1)	(0.6)		(1.6)	(2.6)	(2.9)	(0.5)	(0.7)	(0.2)	(3.0)	(1.9)	(2.6)
Haemulon plumieri	0.8	0.9	0	1.6	0.7	1.5	0.5	1.1	1.0	1.2	3.0	1.3
	(0.6)	(0.9)		(4.2)	(1.3)	(3.6)	(0.7)	(1.2)	(1.3)	(1.9)	(8.2)	(3.5)
Holocentrus sp.	0	0	0	0.4	0	0.1	0	0.1	0	0.1	0	0.1
				(0.7)		(0.4)		(0.3)		(0.2)		(0.3)
Halichoeres sp.	1.2	0.5	0	0	0.1	0	0	0.1	0.1	0	0	0.1
	(1.2)	(0.7)			(0.2)			(0.2)	(0.2)			(0.5)
Kyphosus sp.	0	0.1	0	0	0	0	0	0.2	0	1.0	0.3	0.2
		(0.5)						(0.9)		(4.1)	(1.1)	(1.4)

Appendix 3.4 continued.

Taxon	1997	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Labroidei	9.5	1.9	0	0.1	0.1	0	0.3	0.1	0.1	0.2	0	0.9
	(8.0)	(1.8)		(0.3)	(0.2)		(1.0)	(0.3)	(0.3)	(0.7)		(3.1)
Lachnolaimus maximus	0.3	0.2	0	0.2	0.4	0.3	0.1	0.1	0.3	0.3	0.2	0.2
	(0.5)	(0.4)		(0.6)	(0.6)	(0.5)	(0.5)	(0.2)	(0.6)	(0.6)	(0.5)	(0.5)
Lactophrys sp.	0.1	0.1	0	0	0.1	0	0	0	0.1	0	0	*
	(0.3)	(0.5)			(0.2)				(0.2)			(0.2)
Lutjanus analis	0.2	0.4	0	0.3	0.2	0.3	0.2	0.3	0.4	0.3	0.6	0.3
	(0.4)	(1.2)		(0.5)	(0.6)	(0.5)	(0.4)	(0.6)	(0.6)	(0.6)	(1.0)	(0.7)
Lutjanus griseus	0.9	0.2	0	0	0	0	0.1	0.1	0.1	0.5	0.4	0.2
	(2.6)	(0.5)					(0.2)	(0.5)				
Lutjanus sp.	0.3	0.2	0	0.3	0.1	0.4	0.5	0.2	0.1	0.3	0.1	0.2
	(0.9)	(0.6)		(0.7)	(0.5)	(0.8)	(0.9)	(0.5)	(0.2)	(0.6)	(0.3)	(0.6)
Lutjanus synagris	0	0	0	0	0	0	0	1.4	0	0	0.1	0.2
								(5.5)			(0.4)	(1.8)
Malacanthus plumieri	0	0.1	0	0.1	0.1	0	0	0.2	0.4	0.2	0	0.1
		(0.2)		(0.5)	(0.5)			(0.5)	(0.8)	(0.4)		(0.4)
Mulloidichthys martinicus	0	0	0	0	0	0	0.2	0.3	0	0	0	*
							(0.5)	(0.8)				(0.3)
Mycteroperca bonaci	0	0	0	0.1	0.1	*	0	0	0.1	0.1	0.1	*
				(0.3)	(0.2)	(0.2)			(0.2)	(0.2)	(0.3)	(0.2)
Mycteroperca sp.	0	0.1	0	0.1	0.1	0.1	0	0	0.1	0	0	0.1
		(0.3)		(0.3)	(0.2)	(0.3)			(0.5)			(0.2)
Mycteroperca phenax	0	0.1	0	0	0	0.1	0	0	0	0	*	*
		(0.2)				(0.3)					(0.2)	(0.2)

Appendix 3.4 continued.

Taxon	1997	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
	(1.7)	(7.0)		(6.2)	(6.4)	(5.2)	(4.4)	(2.8)	(3.7)	(4.1)	(7.5)	(5.3)
Pagrus pagrus	0	01	0	0	0	0	0	0	0	0	0	*
		(0.2)										(0.1)
Priacanthus arenatus	0	0	0	0	0	0	0	0	0	0	0	0
Pseudupeneus maculatus	0.6	0.2	0	0.1	1.0	1.4	0.6	0.5	0.2	0.4	0.5	0.6
	(0.8)	(0.5)		(0.4)	(1.5)	(1.9)	(0.9)	(0.6)	(0.4)	(0.6)	(0.7)	(1.0)
Rhomboplites aurorubens	0	0	0	0	0	0	0	0	0	0	0	0
Scomberomoru s sp.	0	0.1	0	0	0.1	0.1	0.1	0	0	0.1	*	*
		(0.3)			(0.)	(0.3)	(0.3)			(0.2)	(0.2)	(0.2)
Scomberomoru s regalis	0	0	0	0	0.2	0	0.2	0	0	0	0.1	*
					(0.5)		(0.4)				(0.3)	(0.2)
Scombridae	0	0	0	0	0	*	0	0	0	0	0	*
						(0.2)						(0.1)
Seriola sp.	0	0	0	0	0	0.2	0.1	0.1	0.1	0.1	0	0.1
						(0.4)	(0.5)	(0.2)	(0.2)	(0.2)		(0.2)
Serranidae	0.2	0.2	0	0.1	0	*	0	0	0	0	0	*
	(0.4)	(0.5)		(0.3)		(0.2)						(0.2)
Sparidae	0	0.1	0	0	0.1	0.1	0	0	0	0	0	*
		(0.3)			(0.5)	(0.4)						(0.2)
Sparisoma sp.	0.9	0.2	0	0.1	0.2	0	0	0.1	0.2	0.3	0.1	0.2
	(1.0)	(0.4)		(0.5)	(0.5)			(0.3)	(0.4)	(0.7)	(0.3)	(0.5)
Sparisoma viride	0.4	0.3	0	0.2	0.6	0.2	0.2	0.4	0.1	0.1	0.1	0.3
	(0.9)	(0.6)		(0.4)	(0.8)	(0.4)	(0.4)	(0.6)	(0.3)	(0.2)	(0.3)	(0.5)

Appendix 3.4. continued

Taxon	1997	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Sphyraena barracuda	0.1 (0.3)	0.2 (0.4)	0	0.1 (0.5)	0	0.1 (0.3)	0.1 (0.2)	0	0.1 (0.2)	0.1 (0.2)	0.1 (0.3)	0.1 (0.3)
Trachinotus falcatus	0	0.1 (0.3)	0	0	0	0	0	0	0	0.3 (0.8)	0	* (0.3)
Diversity H' log10	1.1	1.2	0.2	1.0	1.2	1.1	1.0	1.2	1.1	1.1	1.0	
Species richness d	25.0	34.0	2.0	26.0	28.0	29.0	25.0	29.0	26.0	31.0	28.0	
Evenness J	0.8	0.8	0.8	0.7	0.8	0.8	0.8	0.8	0.8	0.7	0.7	
Average species abundance per year	6.8	7.4	0.1	3.5	4.1	6.3	4.0	4.8	3.5	7.0	7.2	-
SD regardless of species	17.8	14.7	0.4	10.4	8.8	14.5	11.8	10.4	8.2	18.8	19.6	
Number of sites read for	12	18	1	14	17	23	17	19	20	19	23	

A.3.5 Average and standard deviation of fish species observed in the Tortugas South Ecological Reserve (TSER). # Indicates one sighting therefore no standard deviation can be calculated.

Taxon	2002	2007	2009	2010	2011	Total
Acanthurus coeruleus	1.0 (0.8)	0	0	0	0.4 (0.9)	0.4 (0.7)
Acanthurus sp.	2.3 (2.6)	0.5 (0.7)	0	0	0	0.6 (1.5)
Anisotremus virginicus	0	0	0	0	0	0
Balistes sp.	0	0	0	0	0	0
Calamus sp.	1.3 (1.0)	1.0 (1.4)	2.0 #	0	3.4 (4.4)	1.6 (2.7)
Calamus calamus	1.0 (1.4)	0	0	0.3 (0.5)	0.4 (0.6)	0.4 (0.8)
Carangidae	0	0	0	0	0	0
Caranx bartholomaei	0	0	0	0	0.2 (0.5)	0.1 (0.3)
Caranx crysos	0	0	0	0.5 (1.0)	0	0.1 (0.5)
Caranx ruber	0.5 (0.6)	0	0	0	0	0.1 (0.3)
Centropristis ocyurus	0	0	0	0	0	0
Cephalopholis cruentata	0.5 (1.0)	0	0	0	0	0.1 (0.5)
Cephalopholis fulva	0.3 (0.5)	0	0	0	0	0.1 (0.3)
Diplectrum formosum	0	0	0	0	0	0
Diplectrum sp.	0	0	0	0	0	0
Epinephelus sp.	0.5 (0.6)	0	0	0	0.4 (0.6)	0.3 (0.5)
Epinephelus morio	0.8 (0.5)	0.5 (0.7)	0	0.8 (0.5)	0.6 (0.9)	0.6 (0.6)
Euthynnus alletteratus	0	0	0	0.3 (0.5)	0	0.1 (0.3)
Haemulon sp.	4.5 (8.4)	3.5 (3.5)	2.0 #	0	0.2 (0.5)	1.8 (4.3)
Haemulon plumieri	0.8 (1.0)	0	0	0	0.2 (0.5)	0.3 (0.6)
Holocentrus sp.	0.5 (0.6)	0	0	0	0	0.1 (0.3)
Halichoeres sp.	0.3 (0.5)	0	0	0	0	0.1 (0.3)
Kyphosus sp.	0	0	0	0	0	0

Appendix 3.5 continued

Taxon	2002	2007	2009	2010	2011	Total
Labroidei	2.5	0	0	0	0	0.6
	(1.7)					(1.4)
Lachnolaimus maximus	1.0	0	2.0	0	1.2	0.8
	(0.8)		#		(1.8)	(1.2)
Lactophrys sp.	0.5	0	0	0	0	0.1
	(0.6)					(0.3)
Lutjanus analis	0.5	0.5	0	0.3	3.6	1.4
	(0.6)	(0.7)		(0.5)	(3.7)	(2.5)
Lutjanus griseus	0.3	0	0	0	0	0.1
	(0.5)					(0.3)
Lutjanus sp.	0	0	1.0	0	0.2	0.1
			#		(0.5)	(0.3)
Lutjanus synagris	0	0	0	0	0	0
Malacanthus plumieri	0	2.0	1.0	1.5	0.6	0.9
		(0)	#	(0.6)	(0.6)	(0.8)
Mulloidichthys martinicus	0	0	0	0	0	0
Mycteroperca bonaci	0	0	0	0	0	0
Mycteroperca sp.	0.3	0	0	0	0.2	0.1
	(0.5)				(0.5)	(0.3)
Mycteroperca phenax	0	0	0	0	0	0
Ocyurus chrysurus	0.8	0	0	0.3	1.2	0.6
	(0.5)			(0.5)	(2.7)	(1.5)
Pagrus pagrus	0	0	0	0	0	0
Priacanthus arenatus	0.3	0	0	0	0	0.1
	(0.5)					(0.3)
Pseudupeneus maculatus	0.8	3.0	0	0.5	1.0	1.0
	(0.5)	(2.8)		(0.6)	(1.2)	(1.3)
Rhomboplites aurorubens	0	0	0	0	0	0
Scomberomorus sp.	0	0	0	0	0	0
Scombridae	0	0	0	0	0	0
Seriola sp.	0.3	0	0	0.8	0.6	0.4
	(0.5)			(0.5)	(0.9)	(0.6)
Serranidae	0.3	0	0	0	0	0.1
	(0.5)					(0.3)
Sparidae	0	0	0	0	0	0
Sparisoma sp.	0.3	0	0	0	0	0.1
	(0.5)					(0.3)
Sparisoma viride	0.5	0	0	0	0.2	0.2
	(0.6)				(0.5)	(0.4)

Appendix 3.5 continued

Taxon	2002	2007	2009	2010	2011	Total
Sphyraena barracuda	0	0	0	0	0.2 (0.5)	0.1 (0.3)
Trachinotus falcatus	0	0	0	0	0	0
Diversity H' log10	1.2	0.7	0.7	0.9	1.0	
Species richness d	26.0	7.0	5.0	9.0	18.0	
Evenness J	0.9	0.9	1.0	0.9	0.8	
Average abundance of species per year	1.8	0.4	0.2	0.4	1.5	-
SD regardless of sp.	3.2	1.4	0.5	1.1	3.6	-
Number of sites read for	4	2	1	4	5	-

A.3.6 Average and standard deviation of fish species observed in the Tortugas Bank Open Access (TBO). * Indicates value below 0.05.

Taxon	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Acanthurus coeruleus	1.0	0	2.3	1.3	0.3	1.0	0.5	0	0.7	0.2	0.7
	(1.7)		(2.9)	(1.5)	(0.7)	(2.0)	(0.8)		(1.2)	(0.4)	(1.5)
Acanthurus sp.	1.0	0	0.5	0	0.7	0	0.2	0	1.7	0.7	0.4
	(1.7)		(1.2)		(1.3)		(0.4)		(1.5)	(1.2)	(1.0)
Anisotremus virginicus	0	0	0	0	0	0	0.2	0	0	0	*
							(0.4)				(0.1)
Balistes sp.	0	0	0	0	0	0	0.2	0	0	0	*
							(0.4)				(0.1)
Calamus sp.	0.3	0	0.8	0	0.4	0.5	0.3	0	2.3	0.5	0.5
	(0.6)		(1.3)		(1.1)	(1.0)	(0.5)		(4.0)	(0.8)	(1.2)
Calamus calamus	0.3	0	0.3	0.8	0	0.3	0.3	0	0.7	0	0.2
	(0.6)		(0.8)	(1.5)		(0.5)	(0.5)		(0.6)		(0.6)
Carangidae	0	0	0	0	0	0	0	0	0	0	0
Caranx bartholomaei	0	0.6	0.7	0	0	0	0	0.3	0	0.3	0.2
		(1.8)	(1.0)					(0.6)		(0.8)	(0.9)
Caranx crysos	1.3	1.0	0	0.5	0.1	0	0.2	0	0	0	0.3
	(2.3)	(2.8)		(1.0)	(0.4)		(0.4)				(1.3)
Caranx ruber	0	0	0	0	0	0.3	0.2	0	0.3	0	*
						(0.5)	(0.4)		(0.6)		(0.2)
Centropristis ocyurus	0	0	0	0	0	0	0	0	0	0	0
Cephalopholis cruentata	0	0	0.2	0	0.1	0.3	0.2	0	0	0	*
			(0.4)		(0.4)	(0.5)	(0.4)				(0.3)
Cephalopholis fulva	0	0	0	0	0	0	0	0	0	0	0
Diplectrum formosum	0	0	0	0	0.1	0.3	0	0	0	0.5	0.1
					(0.4)	(0.5)				(1.2)	(0.5)
Diplectrum sp.	0	0	0	0	0	0	0	0	0	0	0

Appendix 3.6 continued

Taxon	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Epinephelus sp.	0	0	0	0	0	0.3 (0.5)	0	0	0	0	* (0.1)
Epinephelus morio	0	0.4 (0.5)	0.8 (0.8)	0.5 (1.0)	0.4 (0.5)	0	0.3 (0.5)	0.3 (0.6)	0	0.2 (0.4)	0.3 (0.6)
Euthynnus alletteratus	0	0	0	0	0	0	0	0	0	0	0
Haemulon sp.	0	0	7.3 (17.0)	2.5 (5.0)	0	0	0	0	0	0	1.1 (6.1)
Haemulon plumieri	0.3 (0.6)	0	0.7 (1.6)	2.3 (4.5)	1.6 (3.3)	0.3 (0.5)	0.7 (1.2)	1.0 (1.0)	0.3 (0.6)	1.5 (1.9)	0.9 (2.0)
Holocentrus sp.	0	0	0.3 (0.52)	0	0	0.3 (0.50)	0	0	0	0	0.1 (0.2)
Halichoeres sp.	0.7 (0.6)	0	0.2 (0.4)	0	0.1 (0.4)	0	0	0	0	0	0.1 (0.3)
Kyphosus sp.	0	0	0	0	0	0	0.3 (0.8)	0	0	0	* (0.3)
Labroidei	0.7 (0.6)	0.1 (0.4)	0	0.3 (0.5)	0.1 (0.4)	0	0	0	0	1.5 (2.4)	0.3 (0.9)
Lachnolaimus maximus	0	0	0.3 (0.5)	0	0	0	0.2 (0.4)	0.3 (0.6)	0	0.2 (0.4)	0.1 (0.3)
Lactophrys sp.	0	0	0.2 (0.4)	0	0	0	0.2 (0.4)	0	0	0.2 (0.4)	0.1 (0.2)
Lutjanus analis	0.3 (0.6)	0	0.7 (0.5)	0	0.1 (0.4)	0.8 (0.5)	0.2 (0.4)	1.0	0.7 (0.6)	0.3 (0.5)	0.3 (0.5)
Lutjanus griseus	0	0	0	0	0	1.3 (2.50)	0.2 (0.41)	0	0	0	0.1 (0.72)
Lutjanus sp.	0	0	0.2 (0.4)	0	0	0	0.5 (0.8)	0	0	0	0.1 (0.3)

Appendix 3.6 continued

Taxon	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Malacanthus plumieri	0	0	0.2	0	0.1	0	0	1.3	0	0	0.1
			(0.4)		(0.4)			(1.2)			(0.4)
Mulloidichthys martinicus	0	0	0	0	0	0	0	0	0	0	0
Mycteroperca bonaci	0	0	0	0	0	0	0	0	0	0	0
Mycteroperca sp.	0	0	0	0	0	0	0	0	0	0	0
Mycteroperca phenax	0	0	0	0	0	0	0	0	0	0	0
Ocyurus chrysurus	0.3	1.5	1.8	0	0.3	0.8	4.5	2.3	1.7	1.3	1.5
	(0.6)	(2.7)	(3.1)		(0.8)	(1.0)	(6.2)	(3.2)	(2.1)	(1.0)	(2.9)
Pagrus pagrus	0	0	0	0	0	0	0	0	0	0	0
Priacanthus arenatus	0	0	0	0	0	0	0	0	0	0	0
Pseudupeneus maculatus	0.3	0	1.2	0.5	0.3	0	1.0	0.7	0.7	0.3	0.5
	(0.6)		(0.4)	(1.0)	(0.5)		(1.1)	(0.6)	(0.6)	(0.8)	(0.7)
Rhomboplites aurorubens	0	0	0	0	0	0	0	0	0	0	0
Scomberomorus sp.	0.3	0.1	0	0.3	0	0.3	0	0	0.3	0	0.1
	(0.6)	(0.4)		(0.5)		(0.5)			(0.6)		(0.3)
Scomberomorus regalis	0	0	0	0	0	0	0.5	0	0	0	0.1
							(0.6)				(0.2)
Scombridae	0	0	0.2	0	0	0	0	0	0	0	*
			(0.4)								(0.1)
Seriola sp.	0	0.5	0	0	0	0	0.3	0	0	0	0.1
		(1.1)					(0.8)				(0.5)
Serranidae	0	0	0	0	0	0	0	0	0	0	0
Sparidae	0	0	0	0	0	0	0	0	0	0	0
Sparisoma sp.	0	0	0.2	0.3	0	0	0.2	0.7	0	0.5	0.2
			(0.4)	(0.5)			(0.4)	(0.6)		(0.6)	(0.4)

Appendix 3.6 continued

Taxon	2002	2003	2004	2005	2007	2008	2009	2010	2011	2012	Total
Sphyraena barracuda	0	0	0	0	0.1 (0.4)	0	0.3 (0.5)	0	0.3 (0.6)	0.2 (0.4)	0.1 (0.3)
Trachinotus falcatus	0	0	0	0	0	0	0.2 (0.4)	0	0	0	* (0.1)
Diversity H' log10	1.0	0.7	1.0	0.9	1.0	1.0	1.1	0.9	1.0	1.0	
Species richness d	12.0	7.0	21.0	11.0	16.0	13.0	22.0	9.0	12.0	15.0	
Evenness J	0.9	0.9	0.8	0.9	0.9	0.9	0.8	0.9	0.9	0.9	
Average abundance of species per year	0.4	0.7	2.3	0.8	0.8	0.5	1.4	0.5	0.6	1	-
SD regardless of species	(0.9)	(2.2)	(6.6)	(2.1)	(1.9)	(1.1)	(3.9)	(1.3)	(1.4)	(2.2)	-
Number of sites read for	3	8	6	4	7	4	6	3	3	6	-

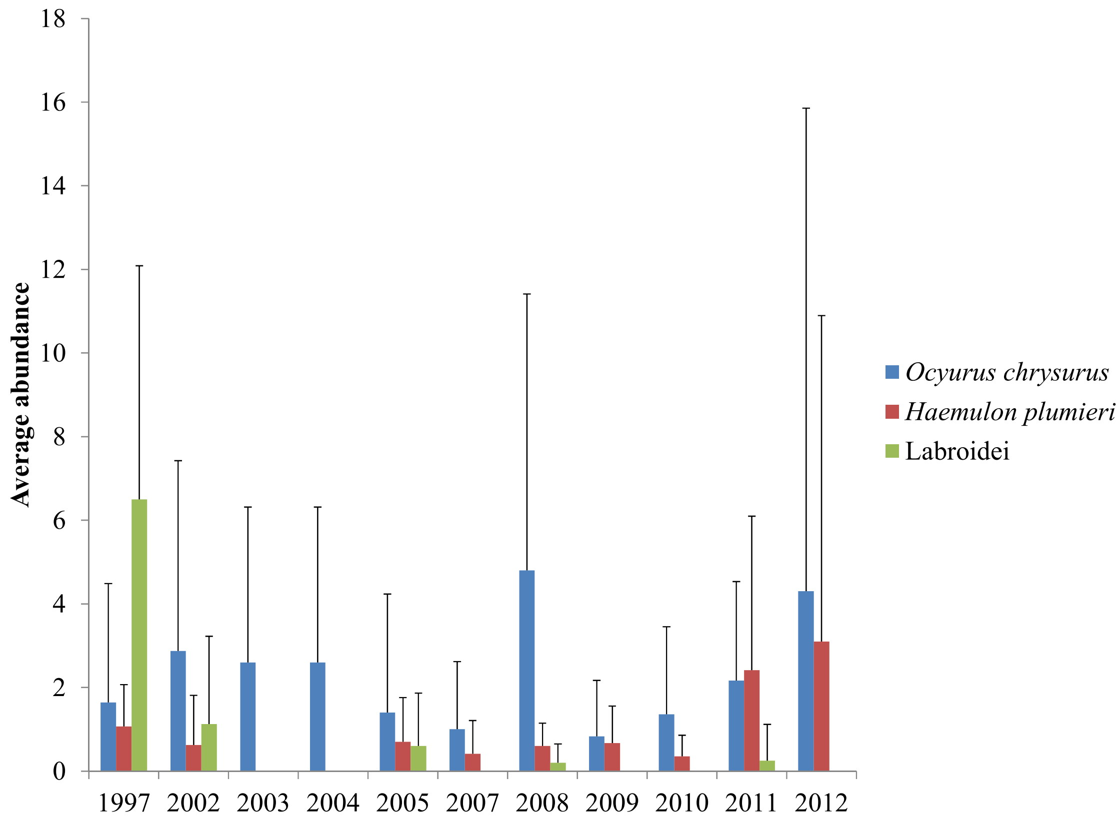
Appendix.3.7 Average and standard deviation of fish species observed in Pulley Ridge. * Indicates value below 0.05.

Taxon	2005	2007	2008	2009	2010	2011	2012	Total
Acanthurus coeruleus	0	0	0	0	0	0	0	0
Acanthurus sp.	0.1	0	0	0	0	0	0	*
	(0.2)							(0.1)
Anisotremus virginicus	0	0	0	0	0	0	0	0
Balistes sp.	0.2	0	0.4	0.2	0.2	0.1	0.2	0.2
	(0.4)		(1.0)	(0.4)	(0.8)	(0.2)	(0.4)	(0.5)
Calamus sp.	0	0.1	0	0	0.1	0.1	0.1	*
		(0.3)			(0.3)	(0.2)	(0.2)	(0.2)
Calamus calamus	0	0	0	0	0	0	0	0
Carangidae	0	0	0	0	0	0	0	0
Caranx bartholomaei	0	0	0	0	0	0	0	0
Caranx crysos	0	0	0	0	0.1	0	0.1	*
					(0.3)		(0.2)	(0.1)
Caranx ruber	0	0	0	0	0	0	0	0
Centropristis ocyurus	0	0	0	0	0	0	0	0
Cephalopholis cruentata	0	0	0	0	0	0	0	0
Cephalopholis fulva	0	0	0	0	0	0	0	0
Diplectrum formosum	0	0	0	0	0	0	0	0
Diplectrum sp.	0	0	0	0	0	0	0	0
Epinephelus sp.	0	0	0	0	0	0	0	0
Epinephelus morio	0.1	0.1	0.1	0.2	0.1	0.1	0.3	0.1
	(0.3)	(0.2)	(0.3)	(0.4)	(0.3)	(0.3)	(0.4)	(0.3)
Euthynnus alletteratus	0.1	0	0.1	0.1	0	0.1	0	*
	(0.2)		(0.3)	(0.3)		(0.3)		(0.2)
Haemulon sp.	0.1	0	0	0	0	0	0.1	0.03
	(0.3)						(0.2)	(0.2)
Haemulon plumieri	0.1	0.1	0	0	0	0	0	*
	(0.3)	(0.2)						(0.2)
Halichoeres sp.	0	0	0	0	0	0	0	0
Kyphosus sp.	0	0	0	0	0	0	0	0
Labroidei	0	0	0	0	0	0	0	0
Lachnolaimus maximus	0.1	0	0	0	0	0	0	*
	(0.2)	0	0	0	0	0	0	(0.1)
Lactophrys sp.	0	0.1	0	0	0	0	0	0.01
		(0.2)						(0.1)
Lutjanus analis	0.3	0.4	0.1	0.3	0.2	0.2	0.5	0.3
	(0.5)	(0.5)	(0.3)	(0.6)	(0.4)	(0.4)	(0.8)	(0.5)
Lutjanus griseus	0	0	0	0	0	0	0	0

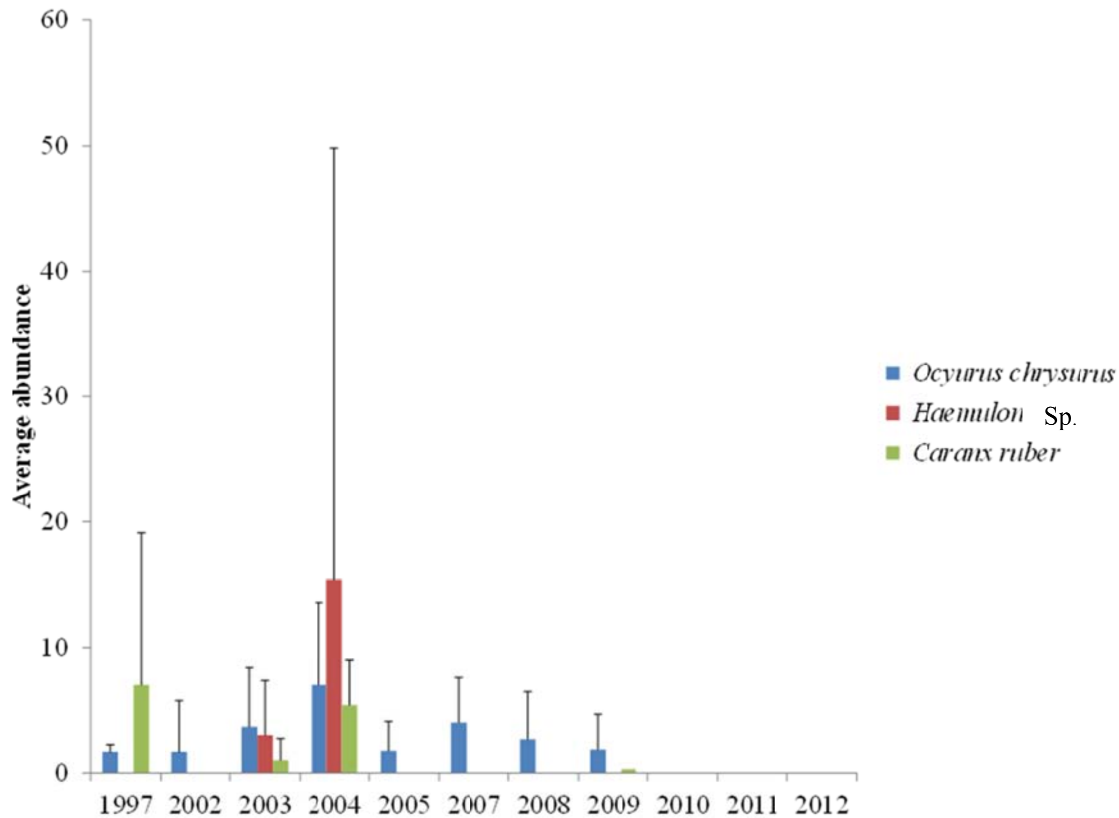
Appendix 3.7 continued

Taxon	2005	2007	2008	2009	2010	2011	2012	Total
Lutjanus sp.	0	0	0	0	0	0.1	0	*
						(0.2)		(0.1)
Lutjanus synagris	0	0	0	0	0	0	0	0
Malacanthus plumieri	1.1	0.4	0.1	0.1	0.4		0.2	0.3
	(1.2)	(0.8)	(0.3)	(0.3)	(0.7)		(0.5)	(0.7)
Mulloidichthys martinicus	0	0	0	0	0	0	0	0
Mycteroperca bonaci	0	0	0	0	0	0.1	0.1	*
						(0.24)	(0.22)	(0.13)
Mycteroperca sp.	0	0	0	0	0	0	0	0
Mycteroperca phenax	0	0	0	0	0	0.3	0	*
						(1.2)		(0.5)
Ocyurus chrysurus	0	0	0	0	0	0	0	0
Pagrus pagrus	0	0	0	0	0	0	0	0
Priacanthus arenatus	0	0	0	0	0	0	0	0
Pseudupeneus maculatus	0.3	0.2	0.3	0.2	0.2	0	0.3	0.2
	(0.6)	(0.4)	(0.6)	(0.6)	(0.4)		(0.9)	(0.6)
Rhomboplites aurorubens	0	0	0	0	0	13.7	2.8	2.4
						(33.2)	(9.4)	(13.7)
Scomberomorus sp.	0	0	0	0	0	0	0	0
Scomberomorus regalis	0	0	0	0	0	0	0	0
Scombridae	0	0	0	0	0	0	0	0
Seriola sp.	0.2	0.2	0.2	0.4	0.1	0.5	0.5	0.3
	(0.5)	(0.5)	(0.4)	(0.8)	(0.3)	(0.9)	(1.2)	(0.7)
Serranidae	0	0	0	0	0	0	0	0
Sparidae	0	0	0	0	0	0	0	0
Sparisoma sp.	0	0	0	0	0	0	0	0
Sparisoma viride	0	0	0	0	0	0	0	0
Sphyraena barracuda	0.1	0	0	0	0.1	0	0.1	*
	(0.2)				(0.3)		(0.2)	(0.2)
Trachinotus falcatus	0	0	0	0	0	0	0	0
Diversity H' log10	0.9	0.8	0.7	0.8	0.9	0.2	0.7	
Species richness d	12.0	8.0	7.0	7.0	9.0	10.0	12.0	
Evenness J	0.8	0.9	0.8	0.9	0.9	0.2	0.6	
Average abundance of species per year	1.0	0.5	0.4	0.4	0.4	5.2	1.9	-
SD regardless of species	3.2	1.6	1.2	1.1	1.0	32.9	8.0	-
Number of sites read for	20	19	16	13	13	17	20	-

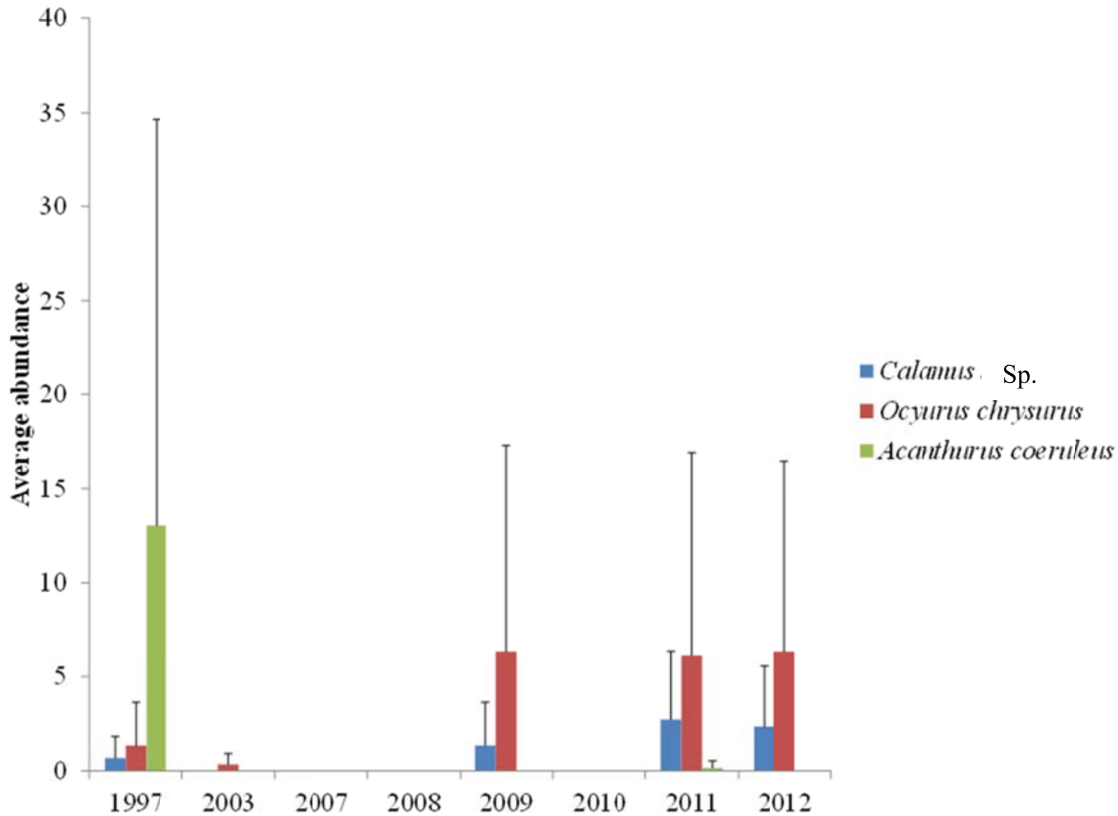
Appendix 4



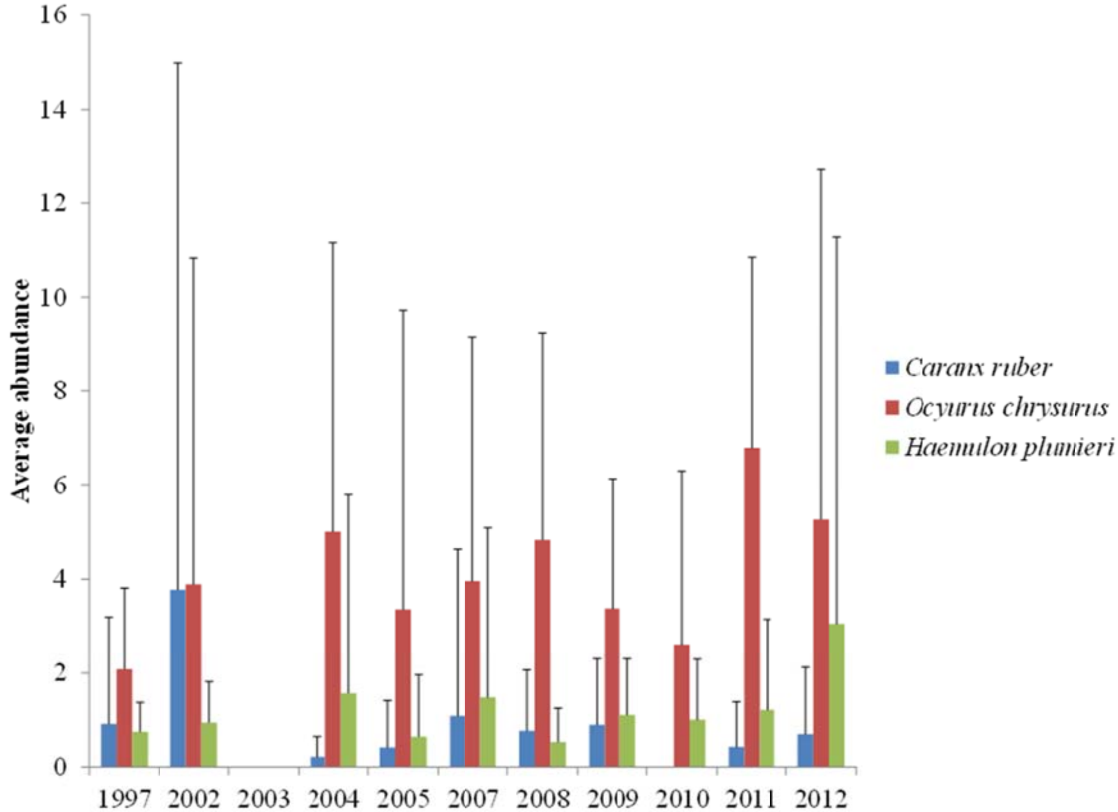
Appendix Fig. (4.1). Average abundance and standard deviation of *Ocyurus chrysurus*, *Haemulon plumieri*, and the suborder Labroidei in the Florida Keys National Marine Sanctuary (FKNMS). Abundance metrics included sites where species were not observed.



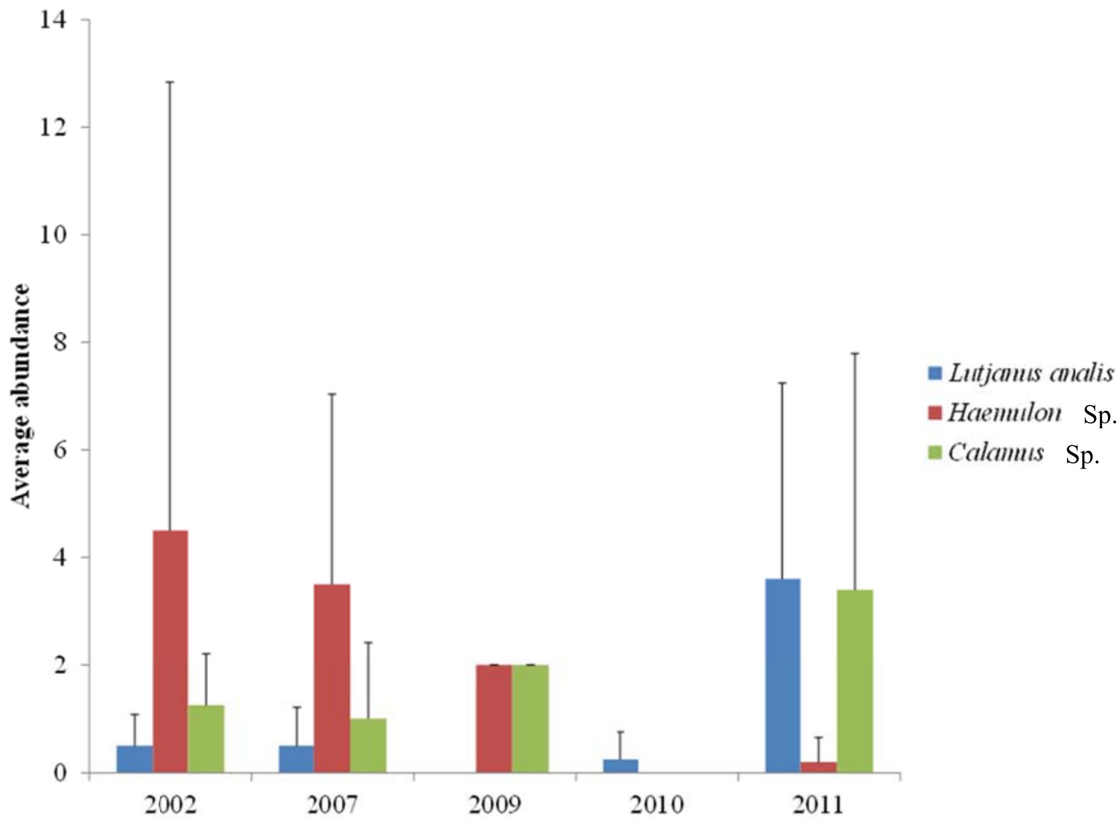
Appendix Fig. (4.2). Average abundance and standard deviation of *Ocyurus chrysurus*, *Haemulon* sp., and *Caranx ruber* in the Dry Tortugas Research Natural Area (DRTO RNA). Abundance metrics included sites where species were not observed.



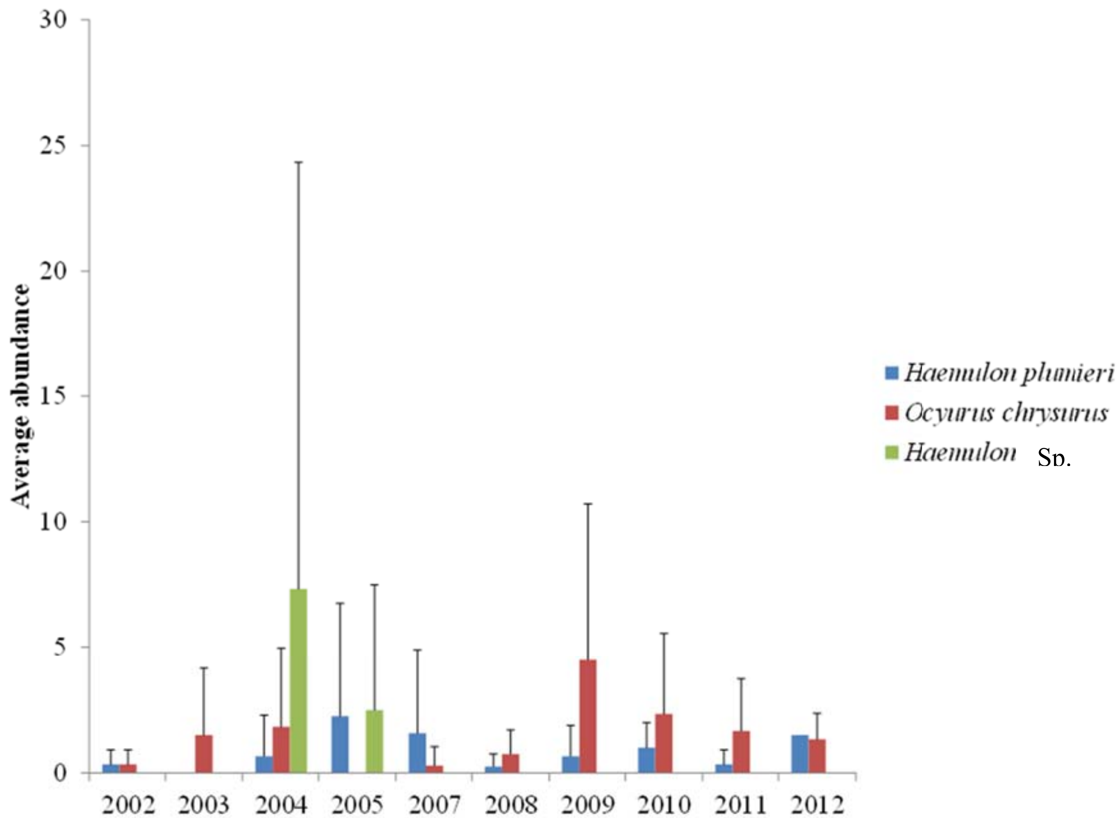
Appendix Fig. (4.3). Average abundance and standard deviation of *Calamus* sp., *Ocyurus chrysurus*, and *Acanthurus coeruleus*, in the Dry Tortugas National Park (DRTO). Abundance metrics included sites where species were not observed.



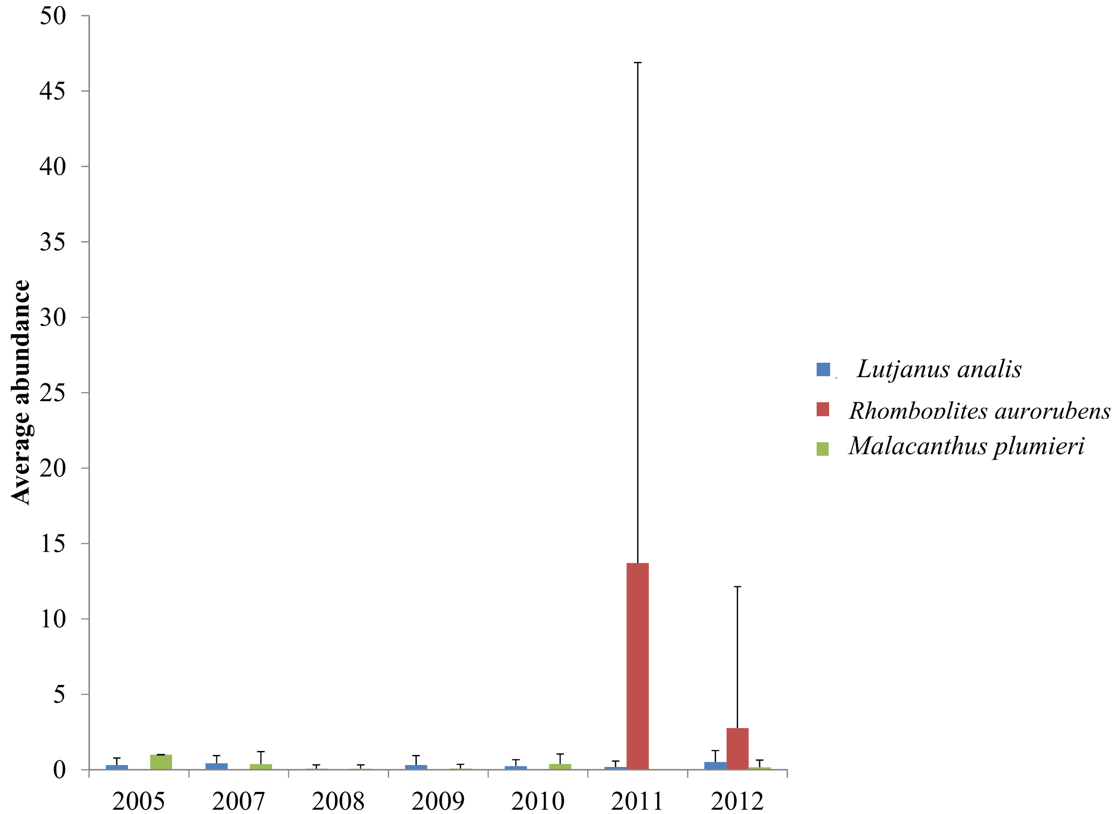
Appendix Fig. (4.4). Average abundance and standard deviation of *Caranx ruber*, *Ocyurus chrysurus*, and *Haemulon plumieri*, in the Tortugas North Ecological Reserve (TNER). Abundance metrics included sites where species were not observed.



Appendix Fig. (4.5). Average abundance and standard deviation of *Lutjanus analis*, *Haemulon* sp., and *Calamus* sp. in the Tortugas South Ecological Reserve (TSER). Abundance metrics included sites where species were not observed.

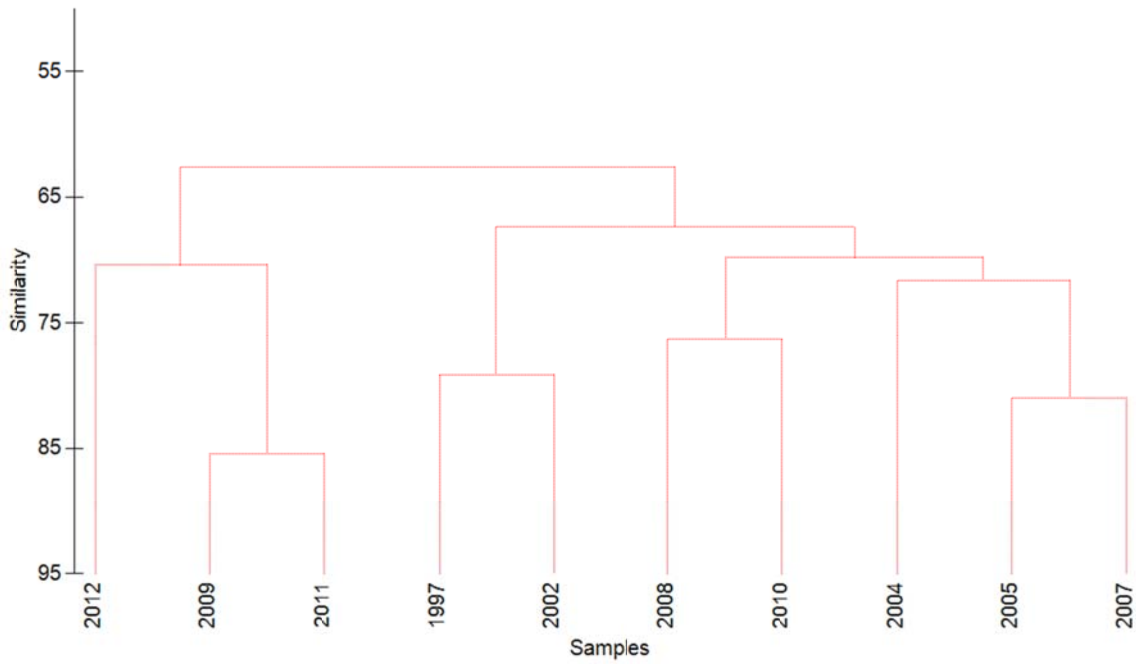


Appendix Fig. (4.6). Average abundance and standard deviation of *Haemulon plumieri*, *Ocyurus chrysurus*, and *Haemulon* sp. in the Tortugas Bank Open Access (TBO). Abundance metrics included sites where species were not observed.

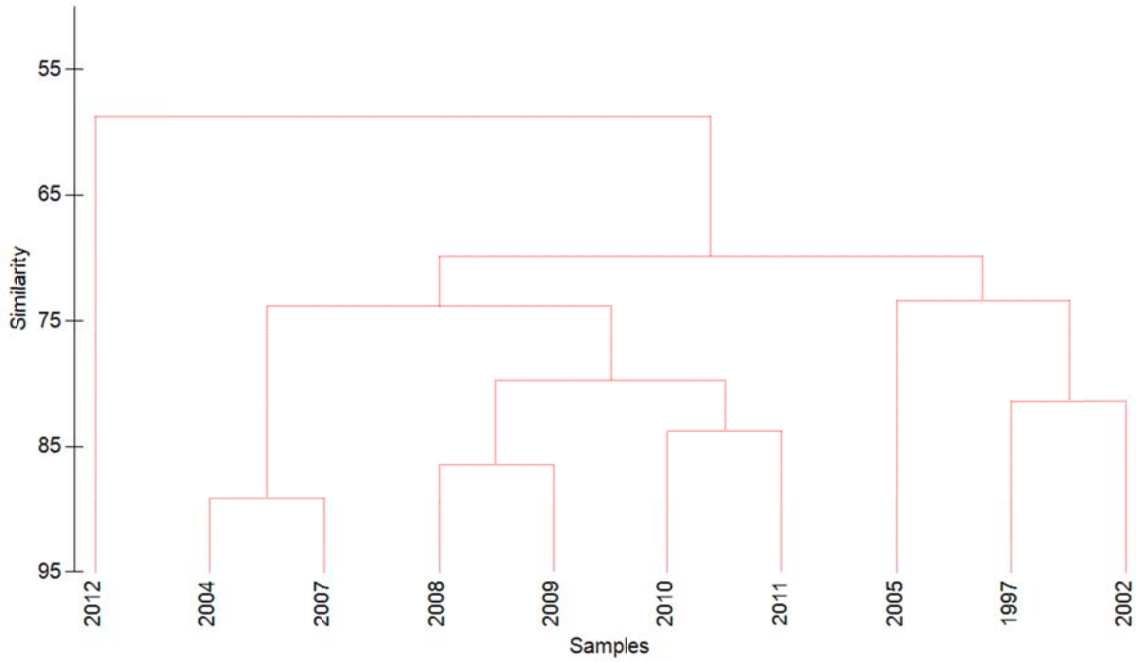


Appendix Fig. (4.7). Average abundance and standard deviation of *Lutjanus analis*, *Rhombopolites aurorubens*, and *Malacanthus plumieri* in Pulley Ridge. Abundance metrics included sites where species were not observed.

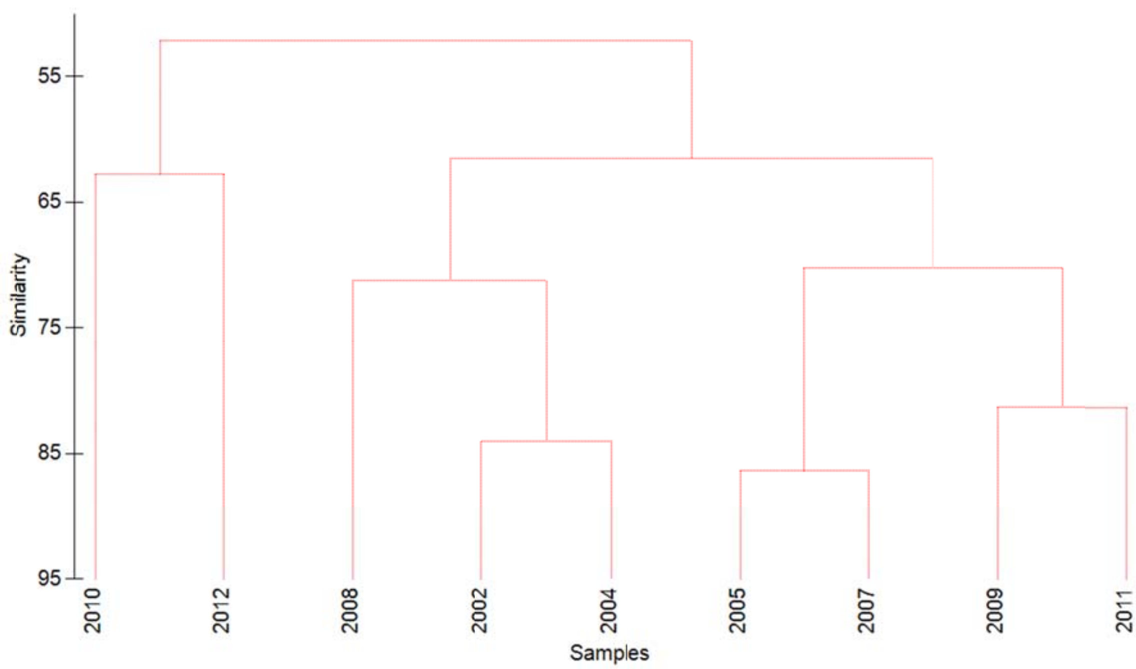
Appendix 5



Appendix Fig.(5.1) Dendrogram of the Bray-Curtis cluster analysis for the habitat composition within the Florida Keys National Marine Sanctuary (FKNMS).

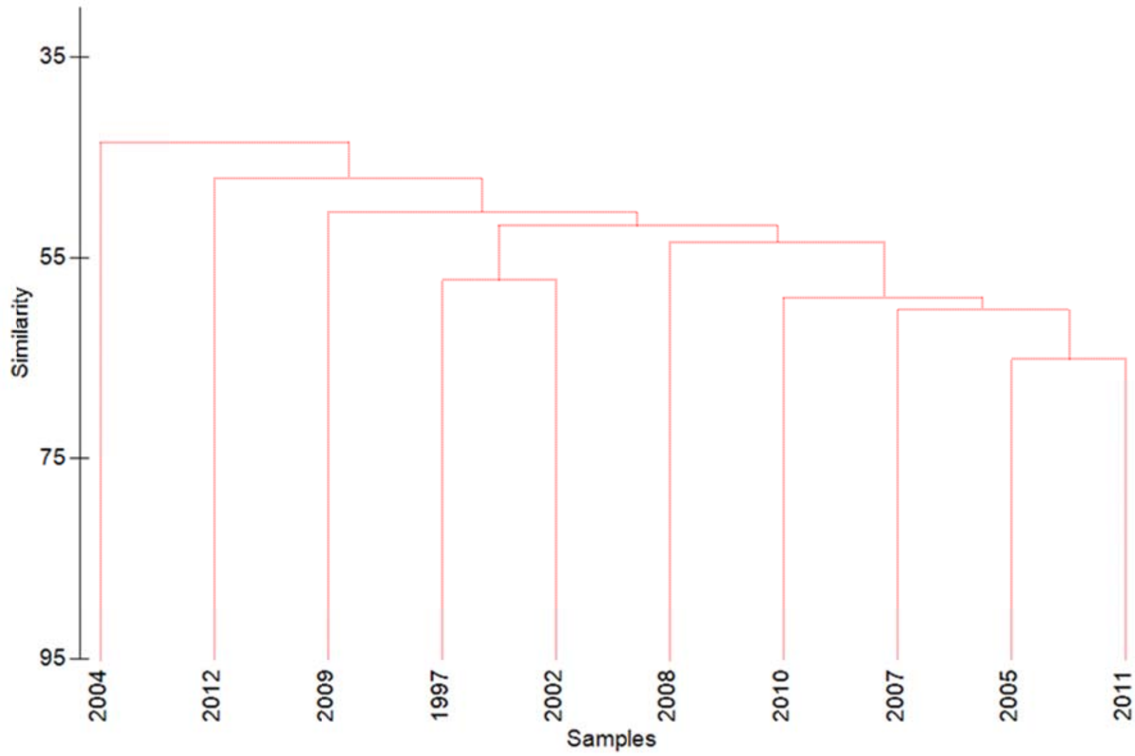


Appendix Fig.(5.2) Dendrogram of the Bray-Curtis cluster analysis for the habitat composition within the Tortugas North Ecological Reserve (TNER).

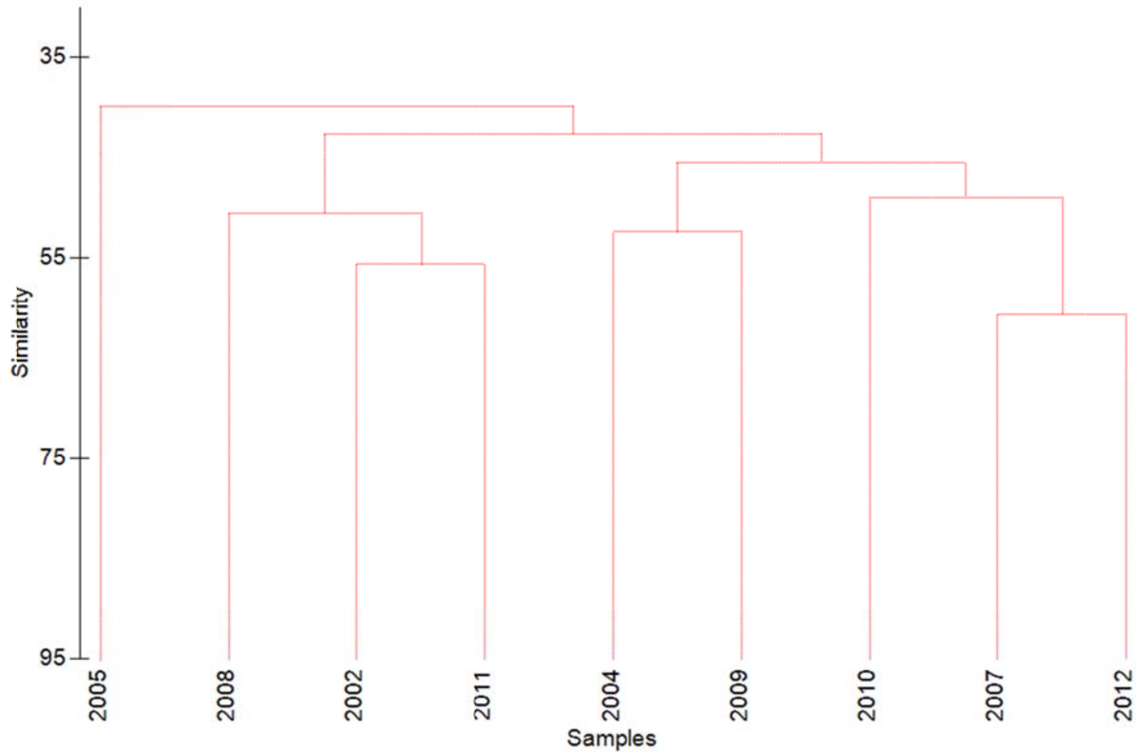


Appendix Fig.5.3 Dendrogram of the Bray-Curtis cluster analysis for the habitat composition within the Tortugas Bank Open Access (TBO).

Appendix 6



Appendix Fig. (6.1) Dendrogram of the Bray-Curtis cluster analysis for the fish community within the Florida Keys National Marine Sanctuary (FKNMS).



Appendix Fig. (6.2) Dendrogram of the Bray-Curtis cluster analysis for the fish community within the Tortugas Bank Open Access (TBO).

Appendix 7.0 Dissimilarity table from SIMPER analysis is shown below. Percentage contributions to dissimilarity between pairs of management regimes were provided for each taxon. The dissimilarity cumulative of percentage contribution accounting for 50% of the overall dissimilarity between the pairs of blocks are represented in bold text. ♦ Indicates the taxa that had ratios of dissimilarity over standard deviation greater than or equal to 1.0 in most comparisons, i.e. the best discriminating taxa.

Taxon	Highest	Middle	Lowest	Lowest vs. Middle	Lowest vs. Highest	Middle vs. Highest
♦*Ocyurus chrysurus*	1.60	1.32	2.49	**7.91**	**7.64**	**6.62**
♦*Haemulon plumieri*	0.76	0.45	1.33	**5.66**	**4.94**	**4.28**
Labroidei	0.50	0.44	0.69	**5.04**	**4.87**	**5.80**
♦*Haemulon* sp.	0.84	0.52	0.65	**5.21**	**5.41**	**7.94**
♦*Acanthurus coeruleus*	0.67	0.41	1.06	**5.01**	**4.68**	**4.32**
Caranx ruber	0.54	0.61	0.39	**3.91**	2.77	**5.97**
Rhomboplites aurorubens	0	0	0.36	1.62	1.63	0
♦*Calamus* sp.	0.69	0.23	1.18	**6.01**	**5.10**	**5.17**
Pseudupeneus maculatus	0.65	0.44	0.86	3.70	3.77	**4.39**
Lutjanus analis	0.48	0.39	0.96	3.73	**4.12**	3.18
Epinephelus morio	0.69	0.29	0.66	2.76	2.23	**4.91**
Acanthurus sp.	0.51	0.43	0.66	3.20	3.47	3.89
Calamus calamus	0.45	0.27	0.56	2.47	2.63	2.66
Caranx bartholomaei	0.15	0.12	0.80	**5.05**	**4.77**	2.48
Caranx crysos	0.06	0.21	0.96	**6.07**	**6.16**	1.97
Sparisoma viride	0.26	0.21	0.41	2.29	1.97	2.34
Cephalopholis cruentata	0.37	0.07	0.24	1.28	1.39	2.75
Sparisoma sp.	0.18	0.19	0.36	1.89	1.77	1.91

Appendix 7.0 continued

Taxon	Highest	Middle	Lowest	Lowest vs. Middle	Lowest vs. Highest	Middle vs. Highest
Diplectrum formosum	0.05	0.15	0.61	3.30	3.32	1.80
Malacanthus plumieri	0.38	0.02	0.30	2.15	2.83	3.18
Lachnolaimus maximus	0.42	0.12	0.25	1.24	1.94	2.76
Seriola sp.	0.15	0.12	0.37	2.55	2.07	2.21
Lutjanus griseus	0.18	0	0.35	2.16	2.23	1.39
Lutjanus synagris	0.09	0	0.36	1.75	2.03	0.73
Halichoeres sp.	0.14	0.15	0.25	1.86	1.80	1.78