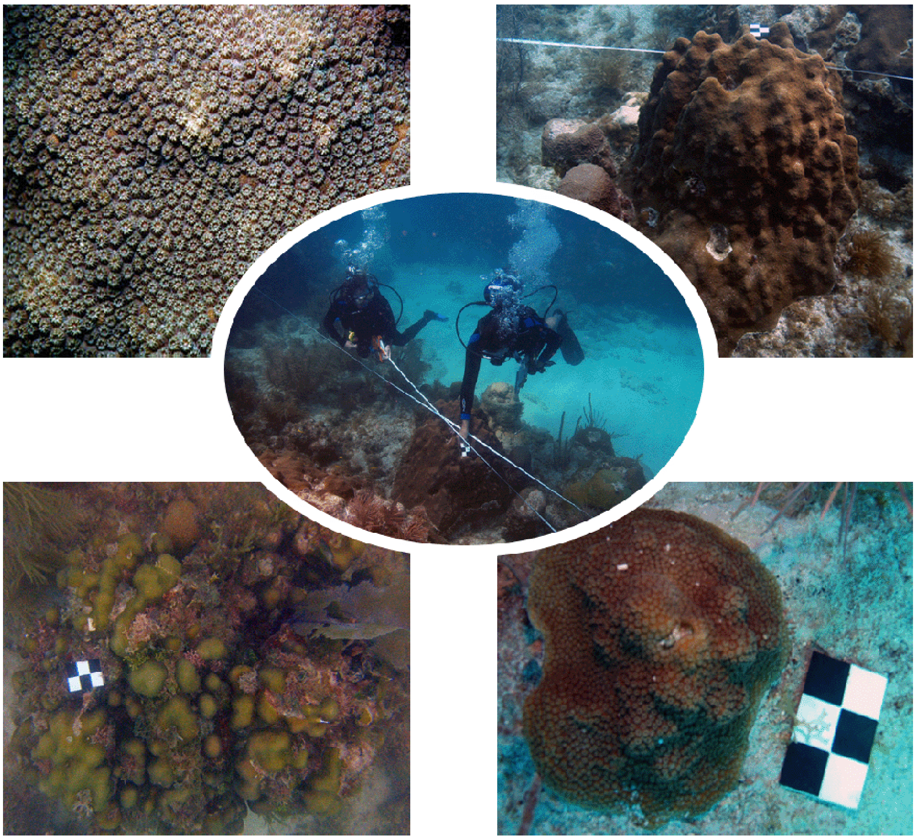
Florida Fish And Wildlife Conservation Commission
Fish And Wildlife Research Institute

Coral Reef Evaluation & Monitoring Project

2009 Annual Report



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2009 CREMP Annual Report

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EXECUTIVE SUMMARY

The purpose of the Coral Reef Evaluation and Monitoring Project (CREMP) is to monitor the status and trends of selected reefs in the Florida Keys National Marine Sanctuary (FKNMS). CREMP assessments have been conducted annually at fixed sites since 1996 and data collected provides information on the temporal changes in benthic cover and diversity of stony corals and associated marine flora and fauna. The core field methods continue to be underwater videography and timed coral species inventories. Findings presented in this report include data from 109 stations at 37 sites sampled from 1996 through 2008 in the Florida Keys and 1999 through 2008 in the Dry Tortugas. The report describes the annual differences (between 2007 and 2008) in the percent cover of major benthic taxa (stony corals, octocorals, sponges, and macroalgae), mean coral species richness and the incidence of stony coral conditions. Additionally, it examines the long-term trends of the major benthic taxa, five coral species that are the most spatially abundant (*Montastraea annularis* complex, *Montastraea cavernosa*, *Colpophyllia natans*, *Siderastrea siderea*, and *Porites astreoides*) and the clionaid sponge, *Cliona delitrix.*

In 2008, mean benthic cover values in the Florida Keys (*N*=97 stations) were 13.6% for octocorals, 12.6% for macroalgae, 6.6% for stony corals and 2.2% for sponges. In the Dry Tortugas (*N*=12 stations) cover was 8.7% for octocorals, 12.3% for macroalgae, 10.3% for stony corals and 1.4% for sponges. From 2007 to 2008 the cover of octocorals and macroalgae increased in the Florida Keys while cover remained similar for stony corals and sponges. Cover for all taxa remained similar between years in the Dry Tortugas. No significant differences in mean coral species richness (number of species per station) were observed between 2007 and 2008 in the Florida Keys and Dry Tortugas. The long-term trends of the four major benthic taxa varied. Throughout the Florida Keys, stony coral and sponge cover has significantly declined from 1996 to 2008. During this time octocoral cover has significantly increased, and no trend has been observed for macroalgae. The trends within the Dry Tortugas mostly mirror those occurring in the Florida Keys; the lone divergence being octocoral cover has decreased there. The demise of stony corals is reflected by the declines in cover of four of the five most spatially prominent species. Overall trends in cover for *M. annularis* complex, *M. cavernosa*, *C. natans*, and *P. astreoides* were all negative. Only the trend for *S. siderea* indicated cover for this species remained similar. To date, the mass bleaching event during the 1997/1998 El Nino Southern Oscillation (ENSO) resulted in the most substantial reductions in cover. However, during the last decade, after effects of the 1997/1998 ENSO had dissipated, the overall trends remain negative for these four species. This is mostly attributed to a regional effect of the Lower Keys and the Dry Tortugas, whereas the sites in the Middle or Upper Keys do not exhibit this trend.

It has been widely reported that following the disappearance of stony corals on reefs in the Caribbean and Western Atlantic large-scale shifts to macroalgae or sponge dominance have occurred. While CREMP has recorded single year spikes or ephemeral blooms after major disturbances (e.g. after the 1997/1998 ENSO and 2005 hurricane season) a prolonged shift towards increased macroalgal cover has not occurred. Likewise, the slow but steady decreases in sponge cover as well as decreases in *Cliona delitrix* suggest that sponges may be vulnerable to the same stressors that affect stony corals. Instead, CREMP findings support a transition to octocorals at many sites in the Florida Keys. The transition is most apparent in the shallow forereef where Elkhorn coral, *Acropora palmata*, and the blade fire coral, *Millepora complanata*,

were previously abundant. Although the demise of *A. palmata* mostly predates the implementation of CREMP, the mortality of the few remnant *A. palmata* colonies on shallow forereefs during or shortly after the 1997/1998 ENSO has been followed by significant increases in octocoral cover in this habitat. Considering that octocoral cover has rebounded twice following major disturbances while little or no recovery of stony coral cover has occurred (even during intervals lacking major perturbations), reefs in the Florida Keys have likely entered into a new alternate state where octocorals are replacing stony corals as the dominant taxa.

INTRODUCTION

The purpose of the Coral Reef Evaluation and Monitoring Project (CREMP) is to monitor the status and trends of selected coral reefs, patch reefs, and hardbottom areas in the Florida Keys National Marine Sanctuary (FKNMS). CREMP was initiated in 1995 as part of the Water Quality Protection Program (WQPP) which mandated a comprehensive monitoring program be established for corals, seagrasses and water quality in the FKNMS. CREMP assessments have been conducted annually at 34 fixed sites since 1996 and data collected provide information on the temporal changes in spatial cover and diversity of stony corals and associated marine flora and fauna. CREMP is one of the longest tenured monitoring programs in the State of Florida and has been instrumental in documenting how a variety of perturbations have resulted in both widespread and localized changes to benthic community composition in the FKNMS. Some changes are due to direct disturbances that have occurred since CREMP was initiated, but many are likely the result of centuries' old anthropogenic modifications to the South Florida environment well before the designation of the Sanctuary in 1990 or the inception of CREMP in 1995.

The Florida Keys contains a small year-round resident population of about 70,000 people with a present day economy that is heavily reliant on tourism and fisheries. The Florida Reef Tract is considered the most widely used and heavily exploited coral reef ecosystem on the planet (Kruczynski 1999; Johns et al. 2001; U.S. Census 2010) with human exploitation dating back to the 1500's. As early as the mid 1800's, Key West had already become one of the most populated cities in Florida due to a prolific commercial fishing industry for conch, sponges, and turtles. Green turtle and sponge fisheries were major marine exports during the late 19th and early 20th century but overexploitation of both fisheries led to their collapse by the 1920's. Since then other commercial and recreational fisheries targeting a wide range of fin- and shellfish (e.g. grouper, snapper, lobster, and stone crab) have expanded. Through the 1900's intense fishing pressure on many of these resources has forced management agencies to establish catch limits and enact fishery closures (e.g. Goliath grouper, Nassau grouper and Queen Conch). Today fishing continues for many species, however there is a general consensus that by the 1970's and 80's many of the Florida Keys fisheries had reached their peak (Tilmant et al. 1989; Bohnsack and Smith 1994; Ault et al. 1998; Ault et al. 2005; McClenachan 2008).

Major population increases and onshore development began around the turn of the 20th century with the completion of the Florida East Coast Railway and, in 1912, the completion of the Overseas Railroad that connected Key West to the mainland. Later, in 1938, the Overseas Highway and a water pipeline from the mainland were constructed, setting the stage for intensive coastal development projects in the Keys during the following decades. To meet the demands of a rapidly growing population and the expanding agricultural economy of South Florida, hydrographic features around the region were permanently changed via the building of dams, canals, and miles of impervious roadways. During this time, the removal of large areas of mangroves, wetlands, and seagrasses severely reduced the volume and altered the chemistry of water flowing out of the Florida Everglades into Florida Bay and eventually across the Florida Reef Tract (Light and Dineen 1994; McIvor 1994; Fourqurean and Robblee 1999). As Key West became a center for tourism in the 1980's large cruise ships began visiting the island and now bring over 350,000 visitors annually.

Deteriorating water quality in South Florida and the Florida Keys was first recognized in the 1970's. In addition to the altered flow and chemistry of water leaving the Everglades, sewage pollution from local development is often emphasized as the leading contributor to declining water quality in the Florida Keys (Lapointe et al. 2004). In the 1980's and 1990's nutrients and bacteria associated with storm water run-off and waste water treatment were recognized as threats to the ecosystem (Lapointe 1990; Lapointe and Clark 1992; Lapointe and Matzie 1996; Szmant and Forrester 1996; Porter et al. 1999; Paul et al. 2000; Lipp et al. 2002; Griffin et al. 2003). Wastewater entering nearshore waters caused algal blooms (Lapointe and Clark 1992; Lapointe et al. 1994; Lapointe 1997) and led to physiological problems in several species, most notably the reproductive failure of nearshore Queen Conch, an iconic and commercially important species (Glazer and Quintero 1998). The presence of fecal coliform bacteria has been detected in nearshore waters and corals (Paul et al. 1997; Lipp et al. 2002) and has been linked to coral diseases (Patterson et al. 2002; Sutherland and Ritchie 2004). Although coral diseases were first documented in Florida in the 1970's, the higher prevalence of some diseases in recent decades may be due to increased nutrient availability in conjunction with a greater frequency of warm sea surface temperature anomalies (Richardson 1998; Bruno et al. 2003; Lesser et al. 2007). In addition, it is believed that eutrophication increases the severity and duration of coral disease outbreaks and lowers the resilience of coral reefs to recover after disturbance events (McCook 2001; Bruno et al. 2004).

Historically, the shallow forereefs of the Florida Keys have caused numerous shipwrecks. Many of the most prominent reefs of the FKNMS are named after vessels that collided with them. As recently as the 1980's, a number of groundings occurred that destroyed large sections of reef. Most notably, within an 18-day period in 1989, three large ship groundings occurred in iconic areas of the Florida Keys. The M/V Alec Owen Maitland, a 130 meter freighter, crushed over 1.5 km^2 of coral habitat near Carysfort Reef, the M/V Elpis, a 143 meter freighter, completely destroyed 2.5 km^2 of coral habitat near Elbow Reef, and the M/V Mavro Ventranic, another large freighter, decimated the reef environment near Pulaski Shoal. The cumulative impacts of these groundings, in conjunction with reports of deteriorating water quality throughout the region, were the primary factors leading to the establishment of the FKNMS.

Florida's coral reefs are also subject to naturally occurring disturbances. Florida is influenced by the El Nino Southern Oscillation (ENSO) in the Pacific Ocean, which brings periods of low winds, little rainfall and extremely high water temperatures to the Florida Keys. Extensive bleaching of Florida's coral reefs occurred at least twice, in 1983 and 1987, prior to CREMP monitoring, although minor incidences of bleaching were observed during other years (Aronson and Miller 2005). CREMP also documented a severe bleaching event in 1997/1998 that resulted in substantial mortality (Somerfield et al. 2008). While coral communities can recover from mild bleaching events, the frequency and intensity of these events has increased during the last two decades (Hoegh-Guldberg 1999) and is expected to intensify even further (Baker et al. 2008). While most bleaching events are associated with thermal stress during ENSO years, other factors such as ultraviolet radiation, turbidity, and eutrophication can trigger a bleaching response (Glynn and Colgan 1993; Brown 1997; Fitt et al. 2001; Manzello et al. 2007; Wagner et al. 2010). Conversely, prolonged periods of low water temperatures have been observed due to Florida's subtropical location. Occasional influxes of extremely cold arctic air migrates southward and can result in widespread coral mortality when water temperatures become depressed below the 16°C threshold for extended periods of time (Roberts et al. 1982).

Tropical cyclone activity has always influenced South Florida, but historical records only reliably date back to the late 1800's. In the early 1900's several major hurricanes impacted the state of Florida including the Labor Day Hurricane of 1935, which made landfall in the Florida Keys as a category 5 hurricane and is still the strongest hurricane on record at the time it made landfall in the United States (National Hurricane Center 2010). Approximately 150 tropical storms or hurricanes affected the state of Florida from 1950 to 1995. Six of these storms made landfall as category 3 hurricane or higher in South Florida between 1950 and 1970. Only one storm of such intensity made landfall in the region (Hurricane Andrew, 1992) between 1970 and 1995 (National Hurricane Center 2010). However, the 2004 and 2005 hurricane seasons were particularly volatile, especially in the Dry Tortugas, with five hurricanes and one tropical storm directly impacting the region (FWRI and DTNP 2010). Hurricanes are naturally occurring events from which coral reef organisms are adapted to survive and recover. Hurricanes can propagate new individuals via asexual reproduction of storm-induced fragments (Gilmore and Hall 1976; Shinn 1976; Fong and Lirman 1995; Fong and Lirman 1997; Lirman 2000) and under the right conditions, a rapid recovery can occur within several years after the passage of major storms (Shinn 1976). However, combined with other chronic stressors such as periodic disease outbreaks, deteriorating water quality, or elevated sea temperatures, hurricanes can have severe, long lasting effects on coral reef habitats.

The interconnected oceanography of the Caribbean basin and Western Atlantic threatens the Florida Reef Tract with natural borne marine pandemics. The Caribbean-wide mass mortality of the long-spined sea urchin, *Diadema antillarum,* affected the Florida Reef Tract in the summer of 1984 after the disease was first observed at Panama in January 1983 (Lessios et al. 1984). Prevailing surface currents (e.g. Yucatan & Gulf streams) transported the disease rapidly throughout much of the Caribbean and the mortality was estimated at 95% of the population in nearly all locations (Lessios et al. 1984). Similarly, white-band disease (WBD) has been a primary cause of *Acropora* mortality in Florida (Precht and Aronson 1997; Precht and Miller 2006), as well as in the Caribbean and Western Atlantic (Aronson and Precht 2001). It was first documented in the U.S. Virgin Islands (Robinson 1973) but had spread to the Florida Reef Tract by the late 1970's and early 1980's. Mass mortalities of spatially-dominant acroporid species (*A. palmata* and *A. cervicornis*) due to WBD were observed at several reefs, and consequently, cover of these species in Florida was estimated to have declined >90% by the late 1990's (Miller et al. 2002; Precht et al. 2002) prior to CREMP.

Coral reef resources in the Florida Keys were first publicly recognized as degraded as early as 1960 when the John Pennekamp Coral Reef State Park was established. Additional management efforts were initiated when the public began to call for more protection. The Key Largo National Marine Sanctuary, adjacent to Pennekamp Park, was established in 1975, and the Looe Key National Marine Sanctuary was established in 1981. Although these two marine sanctuaries encompassed only a small fraction of the Florida Keys' marine environment, they laid the foundation for broader protection of the resource. On November 16, 1990, after a century of heavy exploitation, coastal development, and numerous catastrophic ship groundings, Congress passed a bill that established the Florida Keys National Marine Sanctuary and Protection Act.

The Act designated approximately 2,800 square nautical miles of state and federal waters in the Keys as the Florida Keys National Marine Sanctuary. The FKNMS included the previously established marine protected areas, state parks, and wildlife refuges but also helped designate another 24 special protection areas (SPAs) and four ecological reserves as no-take areas. In conjunction with the EPA WQPP, the FKNMS has also built advanced wastewater treatment facilities to remediate deteriorating water quality.

CREMP was established in 1996 with the primary goal of examining temporal changes in coral benthic communities in the FKNMS and to provide information to management organizations on the status of coral reef resources. Reports provided by CREMP have assisted in evaluating the efficacy of the SPAs and have aided in the permit review process for coastal construction projects and collection of corals in the FKNMS. Since monitoring was initiated, CREMP has documented changes in benthic community composition due to both large-scale and localized disturbances in the environment. These records include widespread coral mortality that occurred in association with the 1997/1998 El Nino and following multiple hurricanes that impacted the region in 2004 and 2005. More localized events include the 2001 harmful algal bloom in the Gulf of Mexico/Florida Bay area that dramatically reduced coral cover at two CREMP sites as well as a disease outbreak which reduced the cover of stony corals at one site in the Dry Tortugas between 2001 and 2003. While it's difficult to distinguish what proportion of the changes observed by CREMP is directly due to present day stressors as opposed to the anthropogenic disturbances propagated over the last century, CREMP has predominately observed a decline in the cover and diversity of corals in the FKNMS. CREMP has observed multiple years where coral cover remained similar, but little recovery was observed during these intervals. This report describes the long-term trends observed by CREMP and provides insight as to how benthic community structure in the Keys has changed since 1996 and may be shaped in the future. Additionally, the report outlines the changes in benthic cover and coral diversity between 2007 and 2008. The results presented in this report include data from 34 sites sampled from 1996 through 2008 in the Florida Keys, and from three sites sampled from 1999 through 2008 in the Dry Tortugas.

METHODS

Site Selection and Station Setup

Sampling locations were initially chosen in 1994 using a stratified random sampling procedure (US EPA EMAP). Forty reef sites were selected within the FKNMS and four permanent stations were installed at each site in 1995. Each station is approximately 22m long x 2m wide and is demarcated by two permanent station markers. Site selection was stratified across four habitat types defined as nearshore hardbottom, patch reefs, shallow spur and groove reefs (~3 to 6m depth), and deep spur and groove reefs (~10 to 20m depth). Although sites were randomly selected, station placement was designed to monitor specific stony coral populations within the selected habitats (e.g. stands of *Acropora palmata* on shallow forereef habitats). In 1999, an additional 12 stations at three sites were installed in the Dry Tortugas. As of 1999, CREMP monitoring included a total of 172 stations at 43 sites.

A statistical analysis of CREMP's experimental design assisted in streamlining the sampling effort by decreasing the number of stations sampled beginning in 2001. As a result of the reduction, all CREMP sites presently consist of two to four monitoring stations. Also starting in 2001, hardbottom sites were removed from the effort due to consistently low coral cover. Monitoring of additional hardbottom locations was discontinued in subsequent years and only one site has been retained (Content Keys). Content Keys is located in an area commonly referred to as the backcountry, the gulf-side area of the Lower Florida Keys region, and has been more appropriately re-designated as a backcountry patch reef. The current CREMP sampling effort now includes 109 stations at 37 sites. Stratified by habitat, CREMP monitors six stations at two backcountry patch reef sites, 10 patch reef sites with 30 stations, 12 shallow reef sites with 39 stations and 13 deep reef sites with 34 stations (Table 1).

The core field methods for CREMP continue to be the underwater videography of three transects and a timed station species inventory at each sampling station (Figure 1). Image analysis from video provides relative estimates of benthic community composition and traditional station species inventories provide information on coral species richness, presence/absence of disease and bleaching, and *Diadema* abundance.

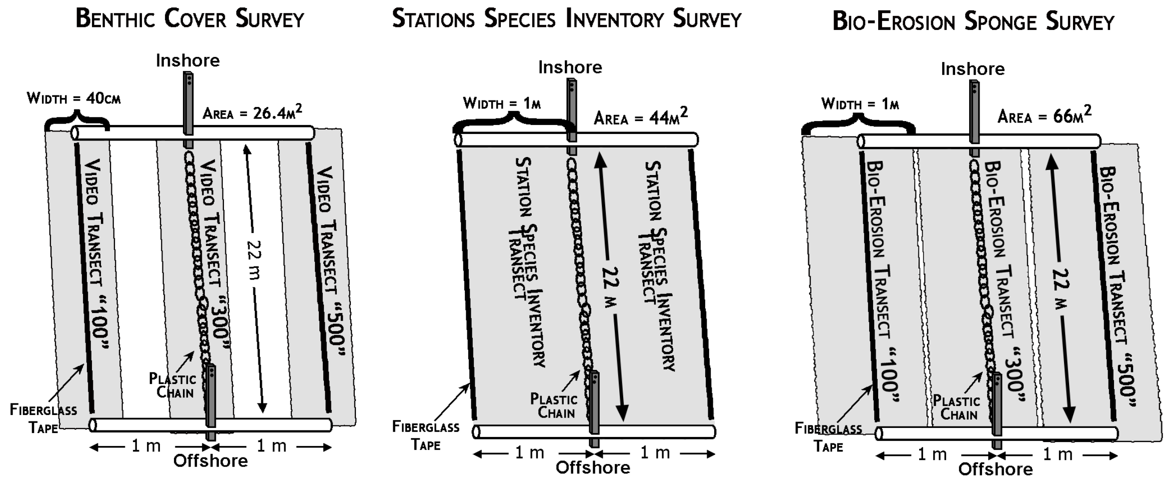


Figure 1: CREMP sites consist of two to four monitoring stations delineated by permanent markers. Stations are approximately 2m x 22m and are generally perpendicular to the shoreline. Three video (benthic survey) transects, a station species inventory (SSI), and a clioniad sponge survey are conducted annually at each station.

Table 1. Location, reef type, and number of stations sampled at 34 CREMP sites monitored from 1996 through 2008 in the Florida Keys and three sites from 1999 through 2008 in the Dry Tortugas.

Region	Habitat/Reef	Site	No. of Stations
Dry Tortugas	Deep	Bird Key Reef	4
Dry Tortugas	Deep	Black Coral Rock	4
Dry Tortugas	Patch	White Shoal	4
Lower Keys	Backcountry Patch	Content Keys	3
Lower Keys	Backcountry Patch	Smith Shoal	3
Lower Keys	Deep	Eastern Sambo Deep	3
Lower Keys	Deep	Looe Key Deep	3
Lower Keys	Deep	Rock Key Deep	2
Lower Keys	Deep	Sand Key Deep	3
Lower Keys	Deep	Western Sambo Deep	3
Lower Keys	Shallow	Eastern Sambo Shallow	3
Lower Keys	Shallow	Looe Key Shallow	3
Lower Keys	Shallow	Rock Key Shallow	4
Lower Keys	Shallow	Sand Key Shallow	3
Lower Keys	Shallow	Western Sambo Shallow	3
Lower Keys	Patch	Cliff Green	2
Lower Keys	Patch	Jaap Reef	3
Lower Keys	Patch	West Washer Women	2
Lower Keys	Patch	Western Head	3
Middle Keys	Deep	Alligator Deep	2
Middle Keys	Deep	Sombrero Deep	2
Middle Keys	Deep	Tennessee Deep	2
Middle Keys	Shallow	Alligator Shallow	3
Middle Keys	Shallow	Sombrero Shallow	4
Middle Keys	Shallow	Tennessee Shallow	3
Middle Keys	Patch	Dustan Rocks	3
Middle Keys	Patch	West Turtle Shoal	4
Upper Keys	Deep	Carysfort Deep	2
Upper Keys	Deep	Conch Deep	2
Upper Keys	Deep	Molasses Deep	3
Upper Keys	Shallow	Carysfort Shallow	3
Upper Keys	Shallow	Conch Shallow	3
Upper Keys	Shallow	Grecian Rocks	4
Upper Keys	Shallow	Molasses Shallow	3
Upper Keys	Patch	Admiral	4
Upper Keys	Patch	Porter Patch	3
Upper Keys	Patch	Turtle	2

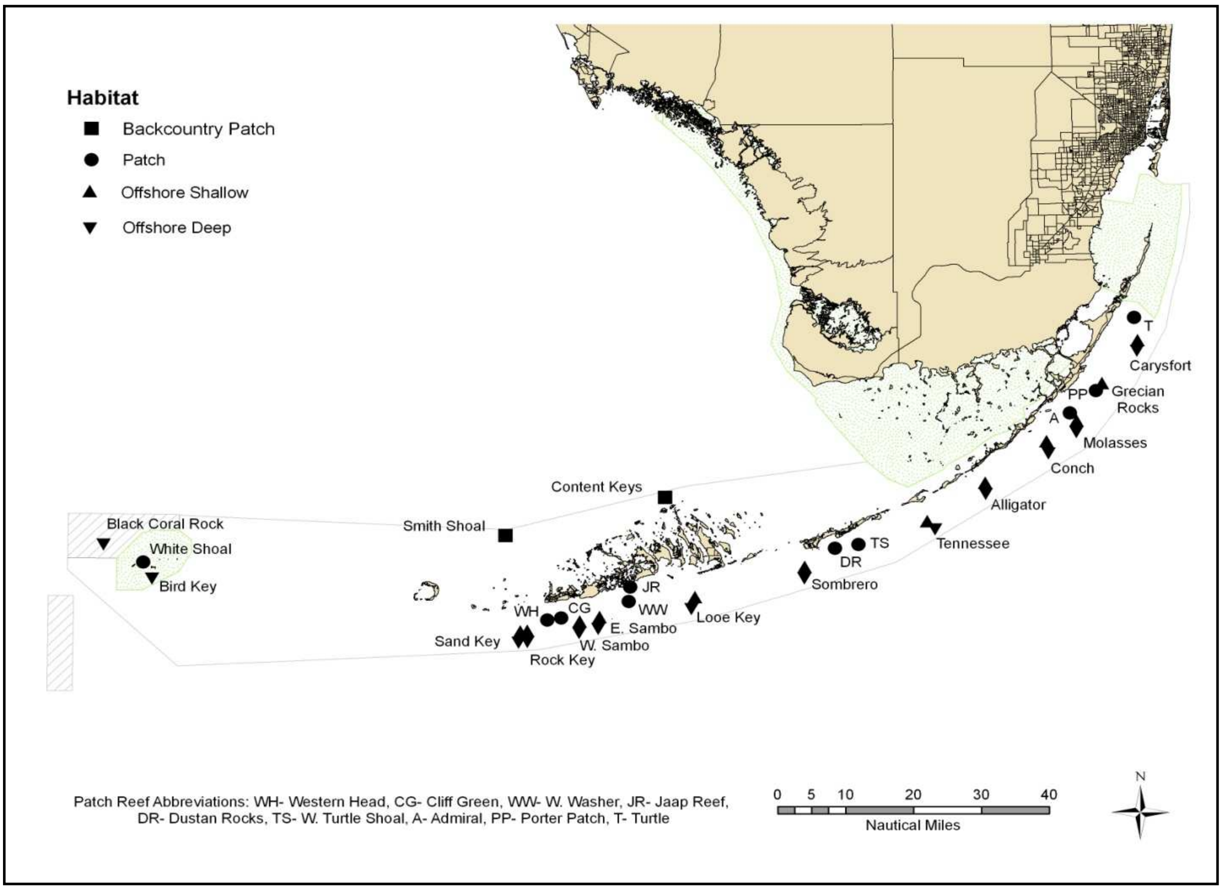


Figure 2. Location of 34 CREMP sites sampled from 1996 through 2008 in the Florida Keys and three sites sampled from 1999 through 2008 in the Dry Tortugas.

Station Species Inventory (SSI)

CREMP has used an established protocol since 1996. Station species inventories comprise a census of all stony corals (Milleporina and Scleractinia), presence/absence of coral disease or bleaching, and counts of *Diadema antillarum* within a station ($44m^2$). Two divers conduct simultaneous, 20 minute inventories within the survey station and enter the data on underwater data sheets. Diseases are categorized as black band, white complex (including white plague, white band, white pox), and other (dark spot, yellow band, etc). Upon completion of the survey, observers compare data underwater (maximum of five minutes) to confirm the stony coral species richness. In the lab, data forms are reviewed, edited, and entered into the FWRI coral database. Data are validated using established coral monitoring quality assurance/quality control protocols.

Video Transects

Video is captured using a SONY TRV 900 to film three 40cm x 22m transects at each station (Figure 1). All transects are filmed at a distance of 40cm above the reef to yield a ~40cm wide image. A convergent laser light system aids in maintaining the camera at the appropriate 40cm distance above the reef surface. Prior to filming each transect, the videographer films a clapperboard that provides information on the date and location of each transect. Filming is conducted perpendicular to the substrate and follows a chain which marks the center of each transect. The videographer maintains a steady swim speed that roughly equals four meters per minute (~4-5min to complete one 22m transect).

In the lab, video is captured digitally and each filmed transect is separated into about 9,000 frames. RavenView™, an image processing program, is used to rejoin a small subset of the images to create a mosaic. Overlap between the images is set to 8% or less, resulting in 60-75 nearly abutting images per transect. The images are then formatted for PointCount '99 image analysis software. Fifteen random points are placed on each image and under each point observers identify benthic taxa (e.g. stony coral to species, octocoral, zoanthid, sponge, seagrass and macroalgae) and substrate. To ensure identification of benthic fauna is consistent across observers, an inter-observer file is completed by all counters prior to image analysis. After all images are analyzed, the data are checked for quality assurance and entered into the Microsoft Access database.

Bioeroding (Clionaid) Sponge Survey

Data on the abundance and cover of three clionaid sponges (*Cliona delitrix, C. lampa*, and *C. caribbaea*) are collected. Clionaid sponge data are acquired using a 1m belt transect centered on the video transects. A PVC pole is held perpendicular to the survey tape and the area of each clionaid sponge is enumerated with a 40cm x 40cm quadrat divided into $25cm^2$ (5cm x 5cm) grids. The total number of $25m^2$ grids occupied by the sponge is used to estimate the cover for each sponge enumerated. Data are entered into the FWRI coral database and are used to quantify the total clionaid spatial coverage at each station. Clionaid area (cm^2/m^2) is calculated for each station by summing the total area of $25cm^2$ grids enumerated and dividing this value by total size of the station.

Statistical & Descriptive Analyses

The results presented in this report include data from 109 stations at 37 sites sampled from 1996 through 2008 in the Florida Keys and from 1999 through 2008 in the Dry Tortugas (Table 1). Where appropriate, the results are presented FKNMS-wide (data pooled for all stations), by habitat, region, site, and for specific region*habitat groupings (e.g. Middle Keys patch reefs). The regions are defined as Upper Keys or "UK" (north Key Largo to Conch Reef), Middle Keys or "MK" (Alligator Reef to Moser Channel), Lower Keys or "LK" (Looe Key to Smith Shoal), and the Dry Tortugas or "DT" (Dry Tortugas to Tortugas Banks), and habitats defined as offshore deep reefs (OD), offshore shallow reefs (OS), patch reefs (P), and backcountry patch reefs (BCP). The results presented here may vary slightly from those reported in previous CREMP reports. Slight differences for values such as percent cover of benthic taxa or mean species richness are due to the differences in the total number of stations used for the analyses in this report compared to those of previous years. Values for all years have been recalculated using the current number of stations. Additionally, two sites, Content Keys and Smith Shoal, have had their habitat grouping re-designated to backcountry patch reef. These sites were formerly classified as a hardbottom and a patch reef site, respectively, and were included in the analyses for these habitats in previous CREMP reports.

Mean species richness was calculated as the number of species per station. To determine if mean species richness was significantly different across all years in the Florida Keys (*N*=97 stations) and Dry Tortugas (*N*=12 stations), repeated measures ANOVA were used. Post hoc Tukey tests (Tukey-Kramer or Holm-Sidak) were used to identify significant differences between 2007 and 2008 and across all years of the project. Long-term trends in mean species richness at the habitat and regional level (*N*=number of stations for each habitat or region) were examined using a generalized mixed model regression in SAS v9.2. Richness data for each station was log-transformed and trends were identified as increasing or decreasing if the slope of the curve was significantly differed from zero.

Changes in benthic cover variables between 2007 and 2008 were examined for macroalgae, octocorals, sponges, and stony corals in SAS v9.2. To test for single year differences of mean percent cover within region*habitat groupings, a Kenward-Roger mixed model ANOVA was used with year and region*habitat groupings (and their interaction) as fixed effects and sites within region*habitat groupings treated as a random effect. Point count data from video transects were pooled for individual stations and square root-transformed. F-tests were used to identify differences in benthic cover means between years for each region*habitat grouping, and to identify differences among region*habitat groupings within years. Only data from the Florida Keys were analyzed because having only three Dry Tortugas sites limit the power of the analysis. For the Florida Keys, the analysis included 10 region*habitat groupings from 34 sites (*N*=number of stations for each region*habitat grouping). Differences in percent cover for the Dry Tortugas are described at the site level.

Long-term trends in benthic cover variables (stony coral, macroalgae, octocoral, sponge), five sentinel stony coral species (*Colpophyllia natans, Montastraea annularis, Montastraea cavernosa, Porites astreoides,* and *Siderastrea siderea*), and clionaid area were examined using a generalized mixed model regression in SAS v9.2. Percent cover data for the benthic variables and sentinel coral species for each station were pooled and square root-transformed. Data for clionaid area (cm^2/m^2) were log-transformed. Stations were nested within sites and sites nested within region*habitat groupings to provide long-term trend information at the site and region*habitat level. Regression lines were calculated from 1996-2008 to understand how the benthic variables and sentinel coral species cover have changed throughout the history of the project. Additionally, regressions were calculated from 1999-2008 to specifically assess how the benthic variables and sentinel coral species have responded in the aftermath of the 1997/1998 ENSO. Long-term trends were calculated for clionaid cover beginning in 2001 when CREMP initiated clionaid monitoring. For all datasets, a regression analysis for each site or region*habitat grouping was calculated from annual percent cover values and the slope was identified as increasing or decreasing by t-tests demonstrating that the slope was significantly different from zero. The analysis included a total of 12 region*habitat groupings from 34 sites in the Florida Keys and three sites in the Dry Tortugas (*N*=number of stations for each region*habitat grouping). Region*habitat groupings and the corresponding number of stations and sites can be found in Table 1. To reduce the possibility of Type I errors due to repeating the same test on multiple region*habitat groupings or sites, a Bonferroni correction was used to adjust the p-value for identifying a trend as significantly increasing or decreasing. At the region*habitat level the p value was adjusted to $p \leq 0.004$ and at site level to $p < 0.002$.

RESULTS & DISCUSSION

2007 vs. 2008

<u>*All Benthic Cover Variables*</u>

When pooled for all stations in the Florida Keys (*N*=97) mean percent cover values were greatest for octocorals and macroalgae (Figure 3). Macroalgal cover did not significantly change between 2007 and 2008 (p=0.52; df=1, 90; F=0.41), remaining similar at 11.3% and 12.7%, respectively. Octocoral cover did significantly increase (p<0.001, df=1, 90; F=15.58), rising from 12.1% in 2007 to 13.6% in 2008. There was no significant change in stony coral cover (p=0.78; df=1, 90; F=0.07) between years with values of 6.7% in 2007 and 6.6% in 2008. Sponges, which make the smallest contribution to benthic cover, significantly increased between years (p=0.015; df=1, 90; F=6.09) rising from 1.9% in 2007 to 2.2% in 2008 (Figure 3).

When combined for both years and pooled for all stations within each habitat type, back country patch reefs averaged the highest macroalgal cover (49.9%). Deep forereefs were next (14.7%) followed by patch reefs (8.6%) and then shallow forereef sites (6.6%). In contrast, octocoral cover was most abundant at patch reefs (20.3%) and shallow forereefs (12.0%). The greatest stony coral cover was found at patch reefs (16.0%), then shallow (3.9%) and deep forereef habitats (2.4%), followed by backcountry patch reefs (2.0%). Sponge cover was highest at patch reefs (3.6%) and offshore deep reefs (3.0%) but contributed <1.0% cover to shallow sites (0.6%) and backcountry patch reefs (0.8%).

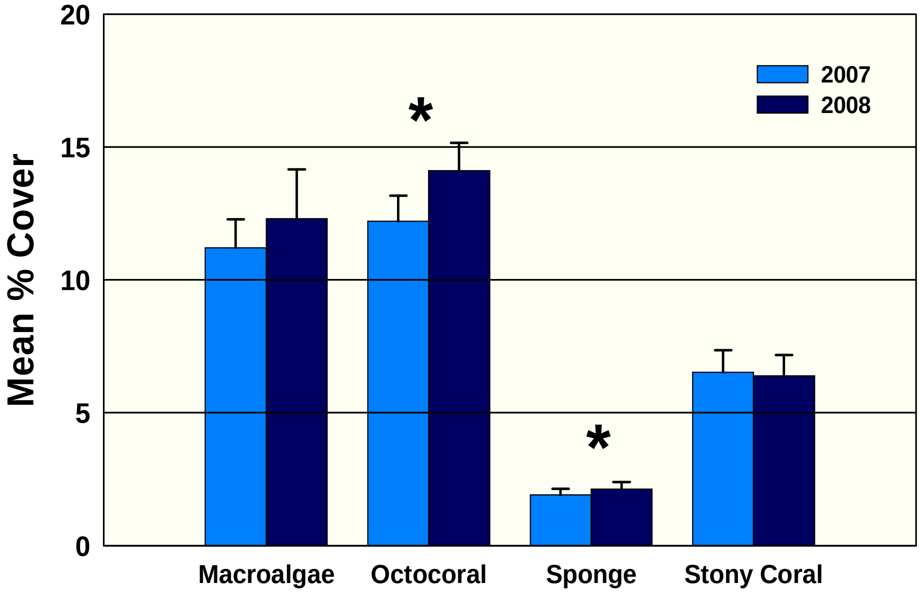


Figure 3. Mean (± SE) of macroalgae, octocoral, sponge and stony coral cover for 2007 and 2008 pooled for all stations (*N* = 97) in the Florida Keys. Error bars represent the standard error of the mean. Asterisks (*) denote a significant difference in percent cover between 2007 and 2008.

When combined for both years and pooled for all stations within each region, macroalgae was the most spatially abundant benthic taxa in the Lower Keys (15.4%), while octocorals provided the greatest cover in the Middle (18.6%) and Upper Keys (17.5%). Stony coral cover was similar across the three regions, ranging from 5.7% in the Middle Keys to 7.0% in the Upper Keys. Lower Keys stony coral cover was 6.9%. Likewise, sponge cover was similar across all three regions ranging from 1.6% in the Upper Keys to 3.0% in the Middle Keys.

When combined for both years and pooled for all stations within each region*habitat grouping, macroalgae was highest at Lower Keys backcountry patch reefs (49.9%), which had the lowest mean cover values for the other three benthic fauna groups. Octocoral and sponge cover were highest at Middle Keys patch reefs at 31.0% and 5.1%, respectively. The highest stony coral cover values were found on the Lower Keys patch reefs (19.7%). For a complete list of benthic cover values for all sites and region*habitat groupings, please see Appendices 1 and 2.

For all stations in the Dry Tortugas (*N*=12), macroalgae was the most prevalent benthic taxa in both years (Figure 4). Macroalgal cover was 14.8% in 2007 and decreased slightly to 12.5% in 2008. Stony coral was the next highest contributor to benthic cover, averaging 9.8% in 2007 and 10.3% in 2008, followed by octocorals with 8.2% cover in 2007 and 8.7% in 2008. Similar to the Keys, sponges made the smallest contribution to benthic cover and averaged <1.5% for both years.

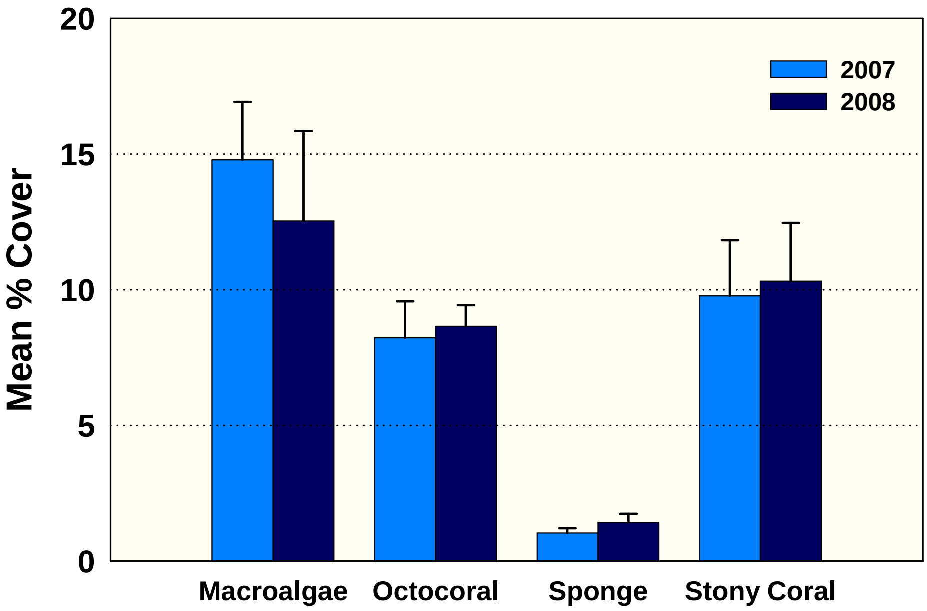


Figure 4. Mean (± SE) of macroalgae, octocoral, sponge and stony coral cover for 2007 and 2008 pooled for all stations (*N* = 12) in the Dry Tortugas. Comparisons across years were not statistically examined.

Macroalgal cover

Although there was no significant change in the overall cover of macroalgae in the Keys between 2007 and 2008, the mixed model ANOVA yielded a significant interaction effect between year and region*habitat ($p<0.001$; df=10, 90; $F=3.51$). This result suggests changes in macroalgae were localized or restricted to certain habitat*region groupings. For example, in the Lower Keys, macroalgal cover significantly increased on backcountry patch reefs from 27.2% in 2007 to 72.6% in 2008 ($p<0.001$; df=1, 90; F=14.39), but significantly decreased in deep (19.8% to 12.2%, p=0.007; df=1, 90; F=7.46) and shallow forereef habitats (7.3% to 2.9%, p=0.009; df=1, 90; F=7.05). All other region*habitat groupings showed no difference across years.

Differences in percent cover were not examined statistically for the Dry Tortugas. Only descriptive analyses are presented for Bird Key Reef, Black Coral Rock (deep sites) and White Shoal (patch reef). Two sites, White Shoal and Black Coral Rock, decreased in macroalgae percent cover from 2007 to 2008 (17.2% to 13.6% and 10.2% to 1.2%, respectively). Bird Key Reef increased from 17.0% to 22.8%.

Octocoral Cover

There was a significant interaction effect between year and region*habitat, indicating that octocoral cover did not increase at all region*habitat groupings. Mean octocoral percent cover was significantly higher in 2008 than in 2007 at three region*habitat groupings (Lower Keys shallow and deep sites, and Middle Keys shallow sites). The remaining region*habitat groupings demonstrated no change. Octocoral cover increased from 5.4% to 7.1% (p=0.001; df=10, 90; F=10.54) in the Lower Keys shallow forereef, and from 8.3% to 9.9% (p=0.007; df=10, 90; F=7.58) in the Lower Keys deep forereef. The shallow forereef in the Middle Keys increased from 13.8% to 18.1% ($p<0.001$; df=10, 90; F=21.39).

Differences in percent cover were not examined statistically for the Dry Tortugas. For the three Dry Tortugas sites, octocoral percent cover increased slightly from 2007 to 2008 at White Shoal (6.1% to 8.8%) and at Black Coral Rock (6.4% to 7.4%). Bird Key Reef had a decrease in cover, from 12.2% in 2007 to 9.7% in 2008.

Sponge Cover

Similar to octocorals, there was a significant interaction effect between year and region*habitat, indicating that sponge cover did not increase at all region*habitat groupings. Mean sponge cover significantly increased from 2007 to 2008 in two region*habitat groupings: Lower Keys deep sites from 1.5% to 2.7% ($p<0.001$; df=10, 90; F=22.44) and Middle Keys deep reefs from 2.3% to 3.4% (p=0.01; df=10, 90; F=5.85).

Differences in percent cover were not examined statistically for the Dry Tortugas. At White Shoal and Bird Key Reef, sponge cover changed very little: <0.1% from 2007 to 2008. However, sponge cover at Black Coral Rock increased from 1.6% in 2007 to 2.7% in 2008.

Stony Coral Cover

Although there was no significant change in the overall cover of stony corals in the Keys, a significant interaction effect suggests that one or more region*habitat groupings significantly changed from 2007 to 2008 (p=0.04; df=10, 90; F=2.02). A significant increase in coral cover was found at the Middle Keys shallow forereefs where cover changed from 1.2% to 1.7% (p=0.01; df=10, 90; F=6.88).

Differences in percent cover were not examined statistically for the Dry Tortugas. In the Dry Tortugas, stony coral percent cover increased at Black Coral Rock (from 17.5% to 18.5%). Stony coral is the largest contributor to benthic fauna at Black Coral Rock. Stony coral percent cover remained similar between 2007 and 2008 at White Shoal (~1.5%) and Bird Key Reef (~10.5%).

Stony Coral Species Richness

In the Florida Keys, mean coral species richness was 13.9±0.45 (SE) and 13.6±0.44 species per station in 2007 and 2008 respectively (*N*=97). Results from the post hoc Tukey-Kramer pairwise comparisons demonstrated there was no significant difference between 2007 and 2008 (Tukey-Kramer q=0.795; p>0.05); however, richness in both years was significantly higher than 2006 (Tukey-Kramer; q=4.8 and 4.1; p<0.05). In 2006, mean species richness was 12.7±0.42 species per station, which is the lowest value recorded since the inception of CREMP in 1996. This low occurred after a record number of tropical systems impacted the Keys during a 16 month period between 2004 and 2005. Mean species richness in 2008 was lower than in 1996 (Friedman's ANOVA; x^2=300.35, df=12, p<0.001). Mean richness values can be clustered into four groups (Figure 5). Values reported for 2007 and 2008 are similar to those reported for 2003-2005 and 1999 (Figure 5). Richness values during these years, however, are significantly lower than those from1996-1998 and 2000-2002 (Figure 5). The period from 1996-1998 represents the peak in mean species richness observed by CREMP. Although the overall trend is negative, an encouraging sign is that mean species richness has rebounded twice after 1999 and 2006.

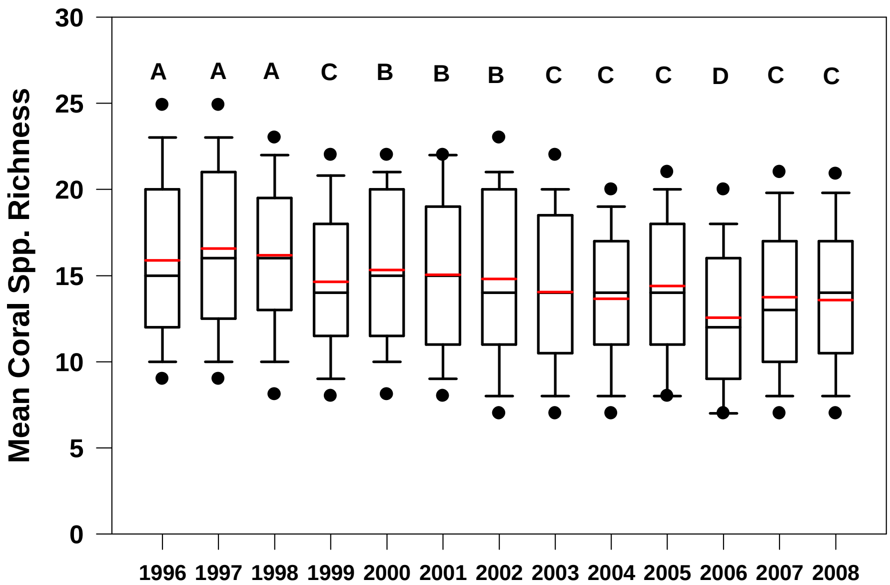


Figure 5. Boxplot of annual mean species richness in the Florida Keys (*N* = 97). Lower point, error bar and bound of the box show the 5^{th}, 10^{th}, and 25^{th} percentiles of the data respectively. The solid black bar within each box represents the median. The solid red bar within each box is the mean. Upper bound of the box, error bar and upper point show the 75^{th}, 90^{th}, and 95^{th} percentiles of the data, respectively. Letters above each box indicate summarized results of Tukey-Kramer pairwise comparisons. Letters denote significant differences in mean species richness across year(s) with group A>B>C>D.

In the Dry Tortugas, mean coral species richness was 18.9±0.82 (SE) and 19.2±0.84 species per station in 2007 and 2008, respectively (N=12). There was no significant difference between the two years (Holm-Sidak, t=0.44; p>0.05), but as in the Florida Keys, richness in both years was significantly higher than in 2006 (Holm-Sidak, t=3.53; p<0.05). Also similar to the Keys, overall mean species richness has declined since monitoring was initiated in the Dry Tortugas (Friedman's ANOVA; x^2=48.15, df=9, p<0.001). Although not as high as in 1999 and 2001, species richness values for 2007 and 2008 are comparable to those observed in 2002 and 2003 (Figure 6). Mean species richness in the Dry Tortugas reached its lowest level between 2004 and 2006. This may be in part due to a coral disease outbreak that occurred at Bird Key Reef in 2001, but the successive and violent hurricane seasons of 2004 and 2005 likely played a role in temporarily removing some species from the sites between 2004 and 2006. After three years in which no major disturbances affected the region, coral diversity appears to be recovering.

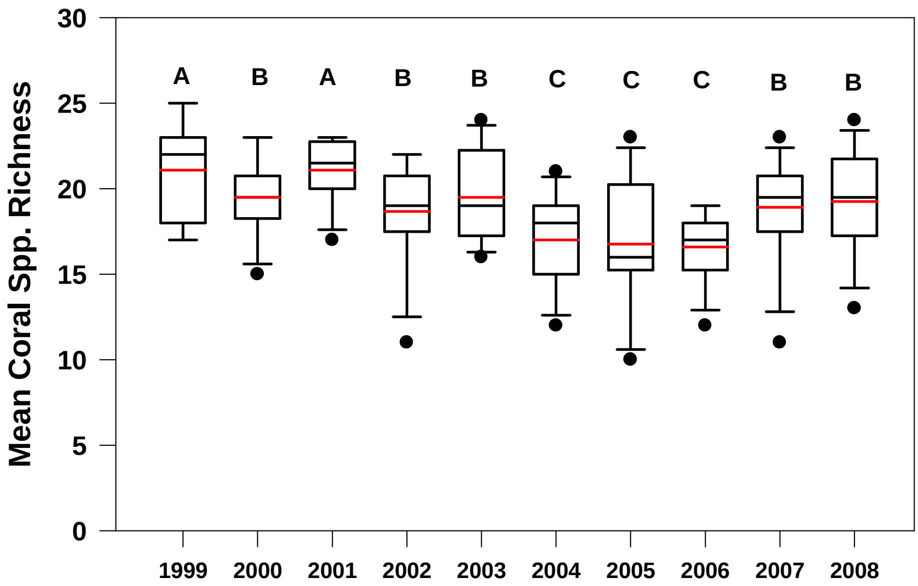


Figure 6. Boxplot of annual mean species richness Dry Tortugas (N = 12). Lower point, error bar and bound of the box show the 5th, 10th, and 25th percentiles of the data respectively. The solid black bar within each box represents the median. The solid red bar within each box is the mean. Upper bound of the box, error bar and upper point show the 75th, 90th, and 95th percentiles of the data, respectively. Letters above each box indicate summarized results of Tukey-Kramer pairwise comparisons. Letters denote significant differences in mean species richness across year(s) with group A>B>C.

The long-term trends show mean species richness has declined across all regions and in all habitat types (Figures 7 and 8). By region, the largest decreases have occurred in the Lower Keys with an average decrease of ~2.6 species per station from 1996 to 2008. By habitat, the largest decreases have occurred at the backcountry patch reefs with an average decrease of ~5.0 species per station from 1996 to 2008. The Dry Tortugas (19.4±0.85) maintains the highest diversity of any region and the deep sites have the highest diversity of any habitat (16.7±0.6 SE). When pooled for all stations in the Florida Keys (N=97), mean species richness has declined by

~2.3 species per station since 1996 and by ~1.8 species per station (N=12) in the Dry Tortugas since 1999.

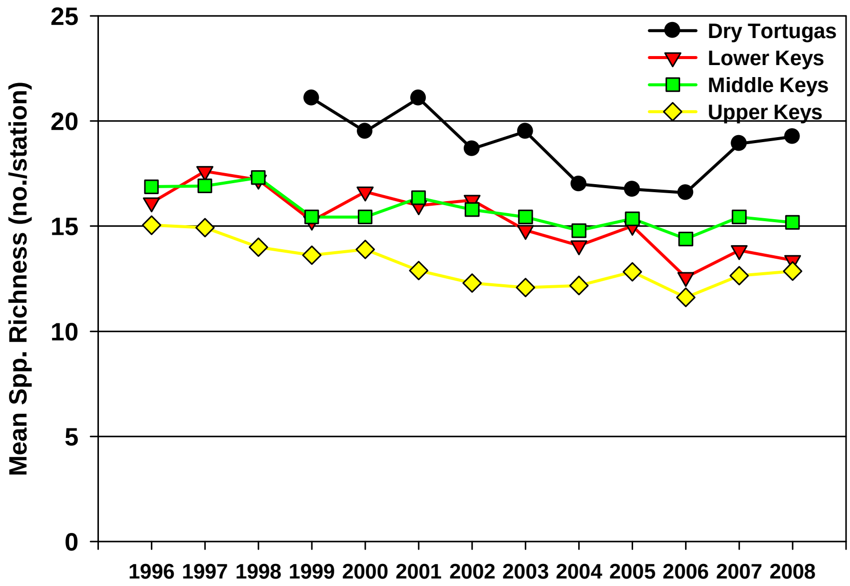


Figure 7. Mean species richness by region from 1996-2008 in the Florida Keys and 1999- 2008 in the Dry Tortugas. The total number of stations sampled annually for each region was 12 in the Dry Tortugas, 46 in the Lower Keys, 23 in the Middle Keys, and 28 in the Upper Keys. A mixed model regression indicated a declining trend in all habitats ($p<0.001$).

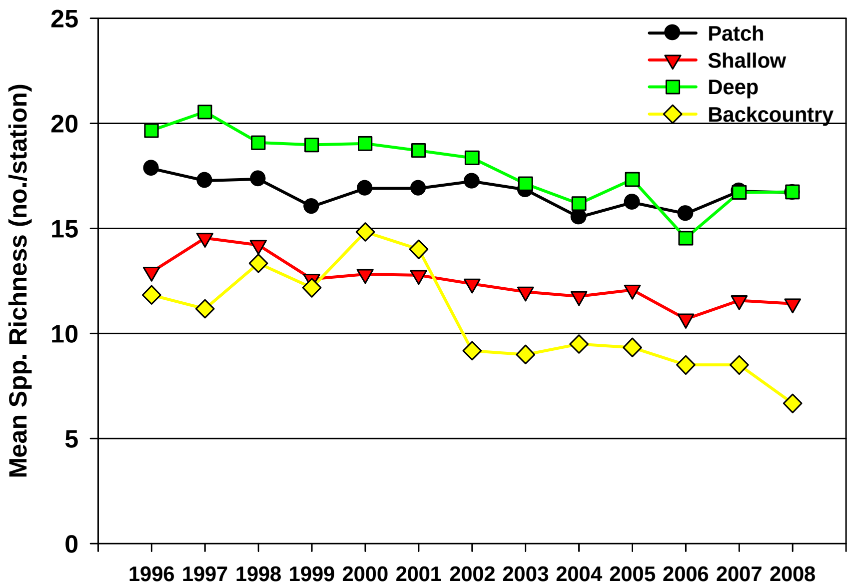


Figure 8. Mean species richness by habitat from 1996-2008 in the Florida Keys. The total number of stations sampled annually for each habitat was 26 at patch reefs, 39 at shallow forereefs, 26 at deep forereefs, and 6 at backcountry patch reefs. A mixed model regression indicated a declining trend in all habitats ($p<0.001$).

Siderastrea siderea was the most widely distributed species, occurring at all 109 stations in 2008. The next four most recorded species were *Porites astreoides* (107 stations), *Millepora alcicornis* (107), *Agaricia agaricites* (99) and *Montastraea cavernosa* (93). A majority of the species observed by CREMP have decreased in presence at stations in the Florida Keys. The species that have had the greatest decrease in presence since 1996 are *Favia fragum* (33 stations), *Mycetophyllia lamarckiana* (21), *Acropora cervicornis* (17), *Colpophyllia natans* (15) and *Porites porites* (15). In contrast, only a few species have increased in distribution, including *Agaricia fragilis* (13 stations), *Stephanocoenia intersepta* (9), *Siderastrea siderea* (6), and *Siderastrea radians* (4). The absolute change in the presence/absence for each species, pooled for all stations from 1996 to 2008 is demonstrated in Figure 9.

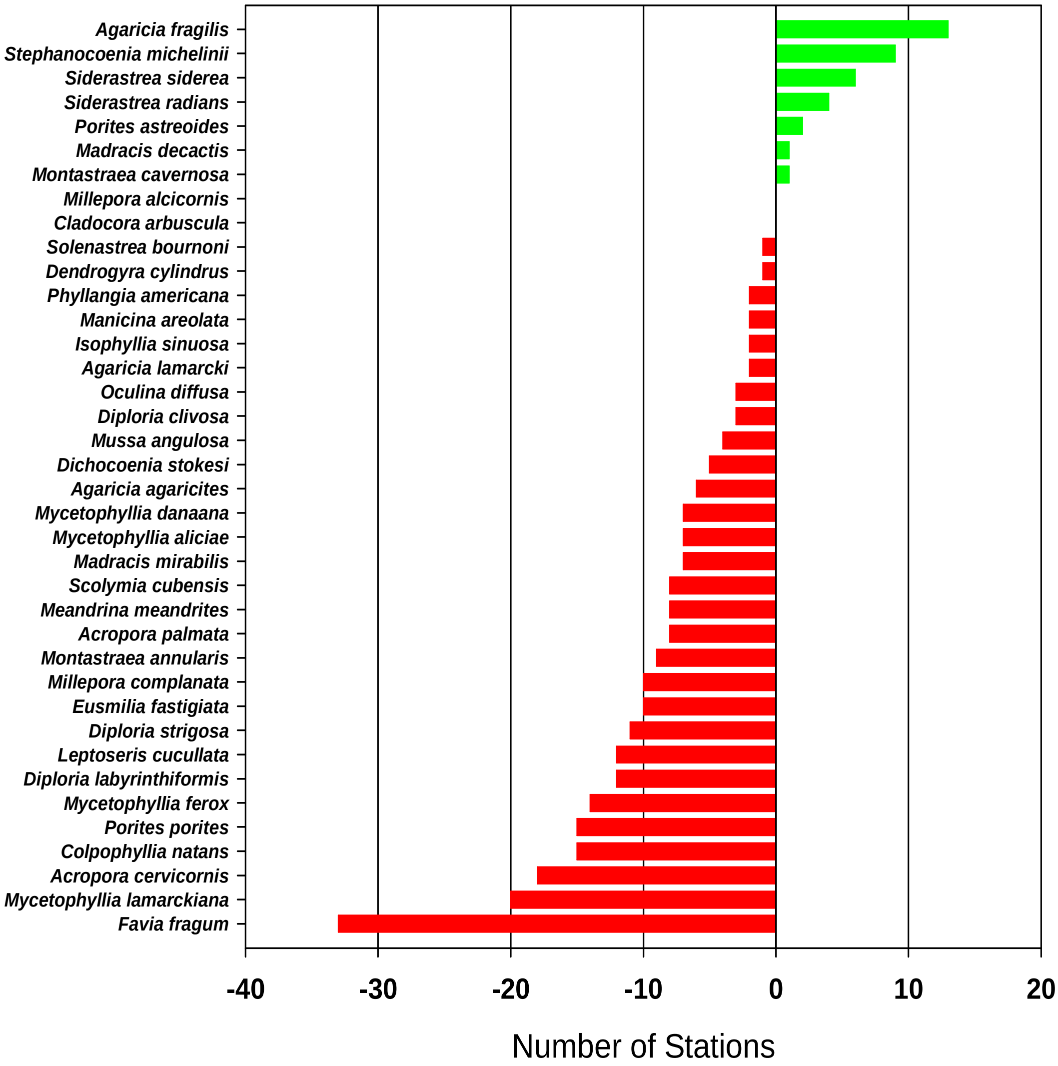
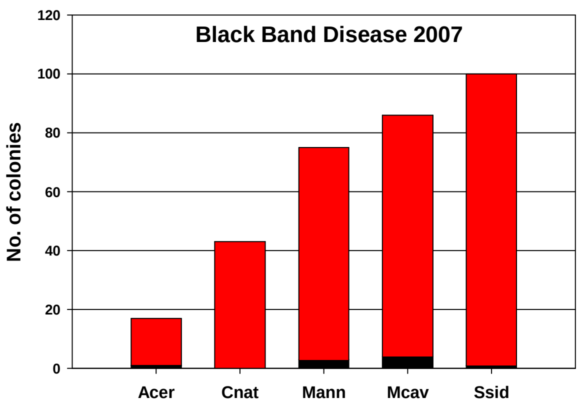
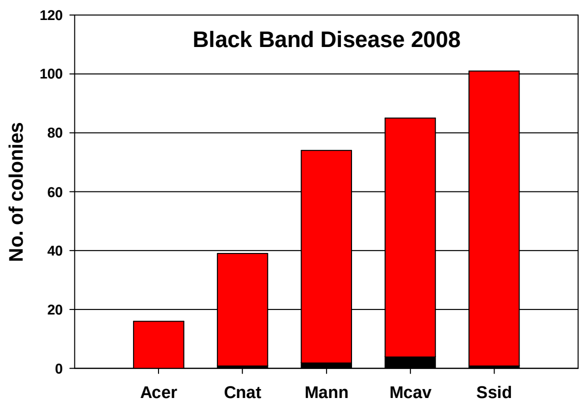


Figure 9. Change in the presence/absence (number of stations) of individual coral species from 1996 to 2008 in the Florida Keys (*N*=97 stations).

Stony Coral Condition

CREMP records the incidence of four coral conditions, including bleaching and three stony coral disease categories: black band disease (BBD), white diseases (e.g. white plague, white band, and white pox), and other diseases (e.g. dark spot, yellow band, and red blotch). In 2007, when pooled for all stations in the Florida Keys and Dry Tortugas (*N*=109), BBD occurred at nine stations, bleaching at 63, other diseases at 77, and white diseases at 46. The incidence of all four conditions declined from 2007 to 2008: BBD to seven stations, bleaching to 31, other diseases to 67, and white diseases to 32. The incidence of disease and bleaching varied among coral species. Of the four conditions recorded, BBD affected the fewest number of coral species in both 2007 and 2008 (four species each year; Figure 10). BBD primarily attacks large boulder and brain corals, with the highest incidence occurring on *Montastraea annularis* complex and *M. cavernosa*. Bleaching was the most common condition recorded during the 2007 survey, affecting 80% (32 of the 40) of the observed coral species (Figure 10). The incidence of bleaching was much lower in 2008, affecting 17 species of the 41 observed (41.5%). Other diseases were observed on 13 species in 2007 and eight species in 2008 (Figure 10). The species most commonly affected by other diseases were *Siderastrea siderea*, *Stephanocoenia intersepta*, and *M. annularis* complex. The number of species affected by white diseases, as well as the number of stations where the disease was recorded for each species, remained relatively consistent between 2007 and 2008 (Figure 10). The species most affected by white diseases were *Agaricia agaricites*, *M. annularis*-complex, *M. cavernosa*, *Porites astreoides*, *S. intersepta, and S. siderea*.





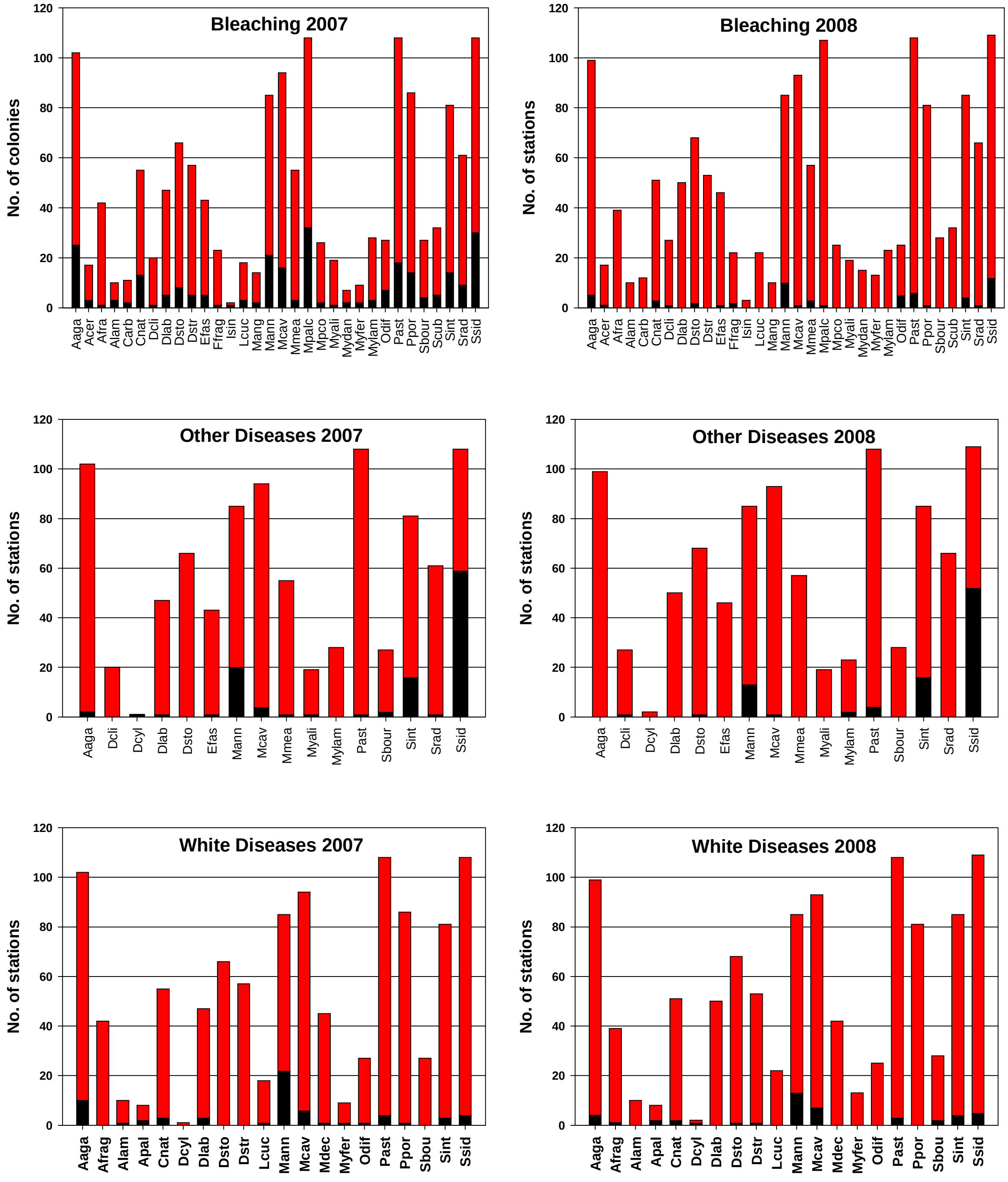


Figure 10. Coral species affected by black band disease, bleaching, other diseases, and white diseases in CREMP stations in 2007 and 2008. Red bars show the number of stations where the species was present. Dark bars indicate the number of stations where the species was observed with the disease/condition. *N*=109 stations in the Florida Keys and Dry Tortugas.

Throughout the project, BBD has been the least common of the four conditions (Figure 11). Although BBD peaked at 19 stations in 1998, it has not been recorded at more than 11 stations within a single year since (Figure 11). Peak incidence of both white diseases and other diseases occurred in 2002 (92 and 99 stations, respectively). Since then the incidence of both disease categories has declined. The decreasing incidence of white syndromes may be partially linked to the decline of *Acropora cervicornis* and *A. palmata* across CREMP stations (see species richness section). Both species of Acroporid corals are commonly targeted by white diseases and have dramatically reduced their populations (Precht et al. 2002, Miller et al. 2002). Bleaching incidence has followed a similar trend to white and other diseases, peaking in 2002 at 90 stations, with decreasing incidence since then (Figure 11). It is important to note that observations of bleaching and disease by CREMP occur throughout the entire field season (May through August) rather than strictly during the peak heating periods of August and September, when bleaching is reportedly most prevalent. Because CREMP records the incidence of disease and bleaching at the station level, but does not estimate the prevalence or severity of the condition of the corals within a station, the results presented here provide a general reference to stony coral condition during a particular year, but do not necessarily reflect peak bleaching years or disease outbreaks.

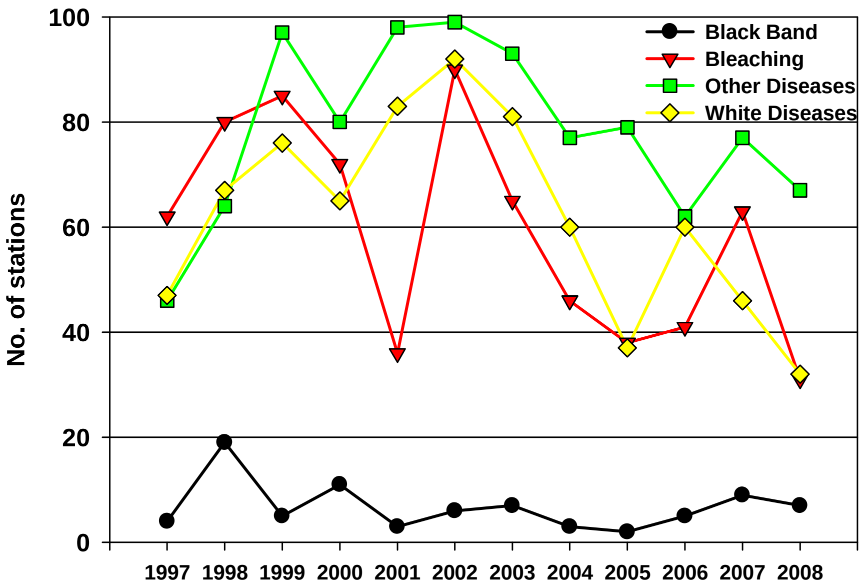


Figure 11. Incidence of black band disease, white diseases (e.g. white plague, white band, and white pox), other diseases (e.g. dark spot, yellow band, and red blotch), and bleaching. *N*=109 stations in the Florida Keys and Dry Tortugas.

Long-term Trends

<u>*All Benthic Cover Variables*</u>

In the Florida Keys (*N*=97 stations), the long-term trends in the mean percent cover of the four major benthic groups (macroalgae, octocorals, stony corals, sponges) differ from 1996 to 2008. Stony corals and sponges have significantly declined, octocorals have significantly increased, and no significant change in macroalgae has been observed (Figure 12). Over the project's duration, coral cover has declined ~48%, from 12.7% in 1996 to 6.6% in 2008 (Figure 12). During this period coral cover reached its lowest level, 6.4%, in 2006. Sponges have always provided the smallest contribution to biotic cover, but like stony corals, demonstrate a negative trend since the project's inception. Sponge cover has decreased from 3.3% in 1999 to 2.2% in 2008, with the lowest value for sponge cover attained in 2006 (1.4%). Although octocoral cover initially declined during the first four years of the project, it has steadily increased every year since except in 2006. Octocoral cover reached a minimum value of 8.9% in 2000, and a maximum value of 13.7% in 2005. In 2008, mean octocoral cover throughout the Florida Keys was 13.6%, a value only slightly below the 2005 peak. Macroalgae has been the most variable component of biotic cover. When combined across all years, mean macroalgal cover is ~11%. The highest cover was attained in 1998 (18.9%) while the lowest cover was found in 2005 (7.4%).

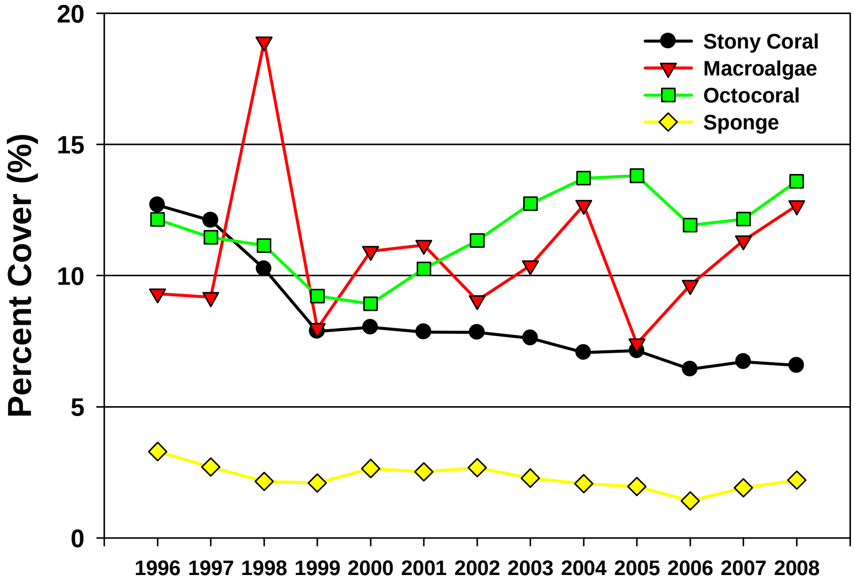


Figure 12. Mean annual percent cover for the four major benthic taxa recorded in CREMP image analysis. Mean percent cover is pooled from 97 stations in the Florida Keys. A mixed model regression indicates a decreasing trend for stony corals and sponges ($p<0.001$), an increasing trend for octocorals ($p<0.001$), and no trend for macroalgae ($p>0.05$).

In the Dry Tortugas (*N*=12 stations), the long-term trends in mean percent cover from 1999 to 2008 are decreasing for stony corals, sponges and octocorals. There was no significant change in macroalgal cover (Figure 13). From 1999 to 2008, coral cover declined ~45% from 18.9% to 10.3% (Figure 13). As in the Keys, coral cover reached its lowest level, 8.8%, in 2006. Sponge cover decreased from 3.6% in 1999 to 1.4% in 2008. Macroalgae has been extremely variable annually, but mean macroalgal cover is ~11% when averaged across all years. The minimum value of 5.7% was observed in 2005 while a maximum value of 14.7% was recorded in 2007. In contrast to the Keys, octocoral cover has not increased. Octocoral cover remained relatively unchanged from 1999 to 2003 (~12.2%), then decreased from 2003 to 2006 to a low of 7.6%, and has subsequently increased to a value of 8.7% in 2008.

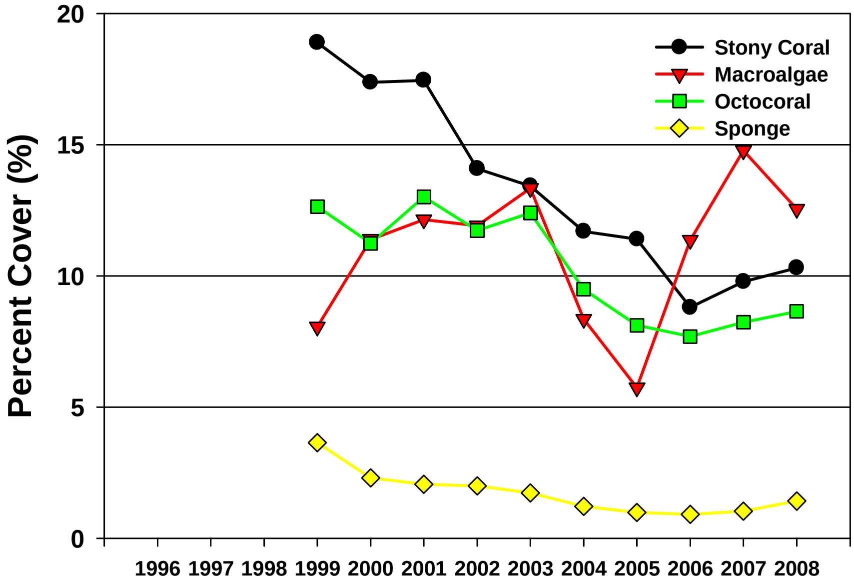


Figure 13. Mean annual percent cover for the four major benthic taxa recorded in CREMP image analysis. Mean percent cover is pooled from 12 stations in the Dry Tortugas. A mixed model regression indicates a decreasing trend for stony corals, octocorals, and sponges ($p<0.001$) and no trend for macroalgae ($p>0.05$).

Multiple factors have likely contributed to the observed changes in benthic cover during the last 13 years. Most notable was the protracted ENSO of 1997/1998. It is well documented that this event caused extensive bleaching and mortality in the wider Caribbean and Western Atlantic and was responsible for dramatically reducing coral cover in the Keys (Somerfield et al. 2008). It is also evident that the impact of this ENSO reverberated across multiple taxonomic groups. In addition to the declines in stony coral cover, CREMP recorded consecutive years with decreases in octocoral and sponge cover between 1997 and 1999. The cause of mortality most likely varied across the different taxonomic groups. Declines in stony corals may have been directly due to the inability to recover from bleaching or may have been caused by a subsequent disease infection following bleaching. Although not documented during the 1997/1998 ENSO, studies since have demonstrated that disease outbreaks occur following mass bleaching events because elevated water temperatures increase pathogen virulence and the physiological stress endured by corals during bleaching weakens their ability to fight off infection (Harvell et al. 2002; Bruno et al. 2007; Miller et al. 2009). The expulsion of sponge symbionts leading to their bleaching has been concomitant during times of coral bleaching (Vicente 1990; Cowart et al. 2006) and may have been responsible for the drop in sponge cover. It is more difficult to ascertain the mechanisms that facilitated the decrease in octocoral cover. Massive bleaching events for octocorals have not been documented like those for stony corals and sponges; however, they are highly susceptible to Aspergillosis caused by the fungus *Aspergillus sydowii* (Nagelkerken 1974; Smith et al. 1996; Nagelkerken 1997; Nagelkerken et al. 1997; Geiser et al. 1998) and red band disease caused by cyanobacteria of the genus *Oscillatoria* spp. (Santavy and Peters 1997; Richardson 1998). It is possible that elevated sea temperatures facilitated an increase in these conditions. Confounding the identification of the causes of mortality is the fact that two hurricanes struck the Keys during this time. Hurricane Georges was especially destructive because it followed on the heels of the 1997/1998 ENSO, making landfall in Key West in September of 1998. Hurricane Irene struck a year later in 1999. Benthic organisms that escaped direct physical damage may have eventually succumbed to sedimentation or scouring because they were already surviving in a sub-optimal condition. It is impossible to identify what proportion of the mortality was due to bleaching, disease, or storm damage during these years, but the most substantial loss of stony coral, octocoral and sponge cover the project has recorded followed this period of combined disturbances. In isolation, hurricanes have resulted in reductions to benthic cover but not to the extent witnessed between 1997 and 1999. For example, following the volatile hurricane season of 2005, CREMP observed a reduction in benthic cover for all three taxonomic groups, but these declines were minor compared to the declines following the combined impacts of the 1997/1998 ENSO and Hurricane Georges.

Recurring patterns of how benthic communities respond to periods of major disturbances are evident. As discussed above, all three major taxa groups are extremely vulnerable to acute disturbances. Declines in the spatial coverage of these taxa create a favorable environment for macroalgae to proliferate and rapidly colonize newly available substrate. CREMP has recorded several instances where the decrease in coral and sponge cover was concomitant with an increase in macroalgal cover. This shift was observed in 1998 when macroalgal cover doubled from 9.2% to 18.8% and to a lesser extent in 2000. This pattern also repeated itself after the 2005 hurricane season. Conversely, even though hurricanes reduce coral cover by breaking or crushing stony corals or ripping octocoral holdfasts from the substrate, the heavy scouring and wave action also removes macroalgae. Two of the lowest values in macroalgal cover were observed during or immediately after periods when hurricanes directly impacted the region. CREMP data also suggests that sponges and octocorals have the ability to recover after major disturbances while stony corals are less resilient. Octocoral and sponge cover began to rebound only two years after the 1997/1998 ENSO and the passages of Hurricanes Georges and Irene. This trend was evident again after 2006.

Stony Coral Cover

In addition to evaluating the overall trends, trends in benthic cover were examined systematically by region*habitat grouping and by site to determine if the observed trends were localized or widespread (*N* varied and was dependent upon the number of stations within a region*habitat grouping or site). Of the 12 region*habitat designations, all but one has significantly decreased in coral cover since 1996 (Bonferroni adjusted, $p<0.004$). The lone exception was the Middle Keys patch reefs (Figure 14; Table 2). Coral cover for this region*habitat grouping was 15.8% in 1996 and 14.3% in 2008. It reached its highest point in 2002 (16.3%). This trend is remarkable considering how severe the declines were in all other areas during the first four years of monitoring. The Lower Keys backcountry patch reefs also fared well until cover collapsed at these two sites following a diatom algal bloom in 2001 (Hu et al. 2003). Between 2001 and 2002, coral cover in this region*habitat grouping dropped from 10.0% to 2.7% and has never recovered. The most notable losses in coral cover between 1996 and 1999 occurred at the shallow sites. A majority of CREMP shallow sites are located on shallow spur and groove forereefs where *Acropora palmata* was once abundant. Both *A. palmata* and *Millepora complanata* suffered mass mortalities due to bleaching, disease and hurricanes that occurred during this time period (Miller et al. 2002; Patterson et al. 2002; Precht and Miller 2006; Somerfield et al. 2008). Although much of the *Acropora palmata* cover on shallow forereefs had already disappeared before CREMP sampling began, the remaining stands were reduced from 3.0% to 0.4% during these four years (Somerfield et al. 2008, Miller et al. 2002). The deep forereefs also had substantial reductions in coral cover, from 6.7% to 3.7%, during the first four years of monitoring. Losses in this habitat were driven by the mortality of large boulder and brain corals like *Montastraea* spp., *Colpophyllia natans,* and *Diploria* spp. Stony coral cover at Patch reefs in the Lower Keys dropped from 27.5% in 1996 to 19.7% in 1999, and patch reef cover in the Upper Keys dropped from 16.3% in 1996 to 11.9% in 1999. Like deep forereef sites, patch reefs primarily consist of large reef-building corals (*Montastraea* spp., *Colpophyllia natans,* and *Diploria* spp.) but contain the highest coral cover values of any habitat in the Keys.

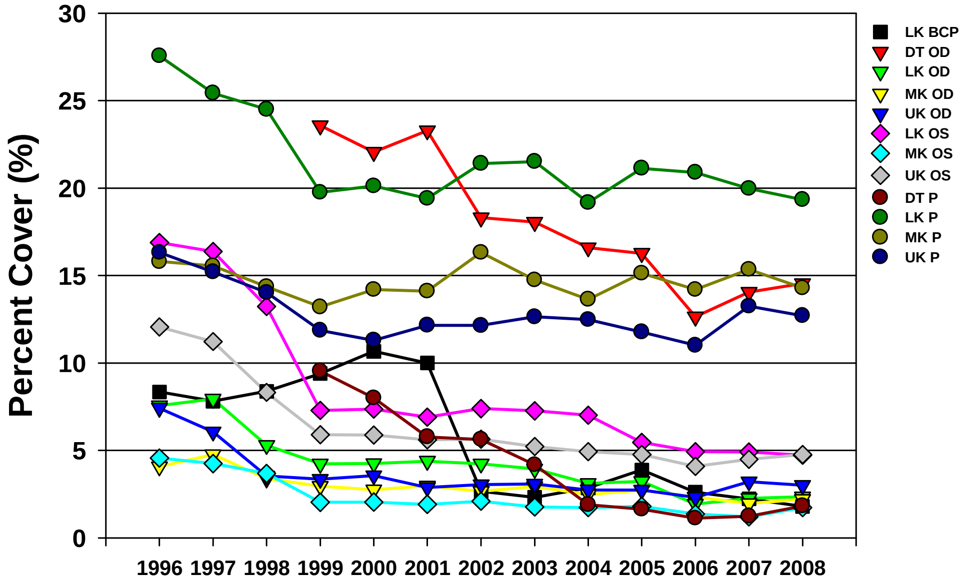


Figure 14. Mean annual percent stony coral cover for 12 region*habitat groupings in the Florida Keys and Dry Tortugas. *N* for each region*habitat grouping varies (see methods). The trend for each region*habitat grouping was determined from a mixed model regression. The direction all slopes is reported in Tables 2 & 3.

At the site level, long-term trends reflect those at the region*habitat level. Twenty-four of 34 sites (70%) in the Florida Keys show a decreasing trend in coral cover since 1996 (Table 4, Appendix 5). Five of eight sites that did not have a significant change were patch reefs while the other three were Upper Keys deep sites. All shallow forereef sites show a negative trend which is indicative of the mass mortalities and lack of recovery of *A. palmata*, and in some instances, *M. complanata* in this habitat. No sites show an increasing trend in coral cover.

Because of the substantial coral mortality incurred during the first three years of the project, the inclusion of these data will always produce a negative trajectory on the overall trends in coral cover. In order to evaluate how the benthic community responded after the 1997/1998 ENSO, we created a new baseline using the values from 1999 as a starting point. Most of the acute mortality associated with 1997/1998 ENSO and Hurricane Georges had abated by 1999. Although the incidence of coral diseases may have remained high, coral cover did increase slightly between 1999 and 2000, so it is reasonable to assume that the major declines in cover from these events had ceased. If a recovery in coral cover was to begin, 1999 appears to be an appropriate starting point.

Table 2. Long-term trends of the four major benthic taxa recorded in CREMP image analysis. Trends were determined for 10 region*habitat groupings from a mixed model regression between 1996 and 2008. Interpretation of trends for each region*habitat grouping are based on Bonferroni corrected p values for repeated testing (adjusted p≤0.004).

Reg*Hab	Stony Corals	Macroalgae	Octocorals	Sponges
LK BCP	decreasing	increasing	no change	no change
LK OD	decreasing	decreasing	no change	no change
LK OS	decreasing	no change	increasing	decreasing
LK P	decreasing	no change	increasing	no change
MK OD	decreasing	no change	no change	no change
MK OS	decreasing	no change	increasing	decreasing
MK P	no change	no change	no change	decreasing
UK OD	decreasing	no change	increasing	no change
UK OS	decreasing	no change	increasing	decreasing
UK P	decreasing	no change	no change	no change

Table 3. Long-term trends of the four major benthic taxa recorded in CREMP image analysis. Trends were determined for 12 region*habitat groupings from a mixed model regression between 1999 and 2008. Interpretation of trends for each region*habitat grouping are based on a Bonferroni corrected p value adjusted for repeated testing (adjusted p≤0.004).

Reg*Hab	Stony Corals	Macroalgae	Octocorals	Sponges
DT OD	decreasing	no change	decreasing	decreasing
DT P	decreasing	no change	decreasing	decreasing
LK BCP	decreasing	increasing	no change	decreasing
LK OD	decreasing	decreasing	increasing	decreasing
LK OS	decreasing	no change	increasing	decreasing
LK P	no change	no change	increasing	no change
MK OD	decreasing	no change	increasing	no change
MK OS	decreasing	no change	increasing	decreasing
MK P	no change	no change	increasing	decreasing
UK OD	no change	no change	increasing	no change
UK OS	decreasing	no change	increasing	decreasing
UK P	no change	no change	no change	no change

Table 4. Long-term trends of the four major benthic taxa recorded in CREMP image analysis for all sites from 1996-2008 and 1999-2008. Thirty-four sites were sampled from 1996 and 37 sites since 1999. Trends were determined from mixed model regression. Interpretation of trends for each site is based on a Bonferroni corrected p value adjusted for repeated testing (adjusted p<0.002).

	Stony Corals		Macroalgae		Octocorals		Sponges	
	96-08	**99-08**	**96-08**	**99-08**	**96-08**	**99-08**	**96-08**	**99-08**
Decreasing	24	15	1	1	1	2	4	4
No Change	10	21	31	34	23	19	28	33
Increasing	0	1	2	2	10	16	0	0

Unfortunately, the timeline adjustment improves the results very little. Since 1999 coral cover at eight of the twelve region*habitat groupings continued to decline (Table 3). This result includes the shallow and deep reef habitats in all regions except the Upper Keys deep sites (Table 3). Patch reefs in all regions, except in the Dry Tortugas, are the only areas that have not had significant coral declines during the last decade. At some locations, substantial losses of coral cover have continued to occur since 1999. As described previously, the 2001 diatom algal bloom coincided with a decline in cover at the backcountry patch reef sites from 10.0% to 2.7% within a single year. Since 1999 coral cover at the two Dry Tortugas deep reef sites has dropped from 23.6% to 14.5%. This result is due to a disease outbreak on *Montastraea annularis* complex and *Colpophyllia natans* that occurred between 2001 and 2003, and to the physical destruction of six storms passing within 100 miles of the Dry Tortugas between August 2004 and October 2005 (FWRI & DRTO, 2010). The region*habitat grouping for patch reefs in the Dry Tortugas consists of only one site (White Shoal) and cover there was diminished when the remaining *Acropora cervicornis* was struck by disease in 2003 and then decimated by the 2004 hurricanes (FWRI & DRTO, 2010). In all, of the 37 sites monitored by CREMP since 1999, 15 (40%) still continue a declining trend in coral cover, 21 (57%) show no change, while one site (3%) did increase in coral cover (Table 4, Appendix 6). Although some sites may demonstrate no significant change in coral cover over the last decade, significant decreases may be less detectable because coral cover values are so low. For example, the Middle Keys shallow sites (Sombrero and Tennessee) had a coral cover value of 2.0% after 1999, but had dropped to 1.7% by 2008.

The most notable instances of coral cover declines since 1999 have occurred in the Dry Tortugas and Lower Keys regions. Only two sites show a declining trend in the Middle and Upper Keys (Appendix 7). Sixteen of 19 (84%) sites in the Dry Tortugas and Lower Keys show a negative trend in cover. While linkages between disturbance events and coral decline are easier to propose in the Dry Tortugas (FWRI & DRTO, 2010), it is more complicated to distinguish why the Lower Keys sites have not fared as well as those in the Middle and Upper Keys. Fourteen of the 16 sites in the Florida Keys with a decreasing trend in coral cover from 1999 to 2008 were based in the Lower Keys (Appendix 7). One reason for this decline could be due to forereef sites in the Lower Keys possessing higher cover values beginning in 1999 as compared to those in the Middle or Upper Keys. If the variance around the interannual mean cover values is equal for most sites (e.g. ±1% cover), significant changes in cover would be easier to detect at sites with higher coral cover values than at those with lower ones. Although the proportional change in cover at two sites could be similar (e.g. a 25% decrease in cover), the absolute change in cover would be greater at sites with higher cover. This scenario would yield a significant trend at a site starting with 8% cover but not necessarily for one with only 2% cover. Although the discrepancies in coral cover across sites should be considered, anthropogenic activities related to the thriving tourism industry of Key West need to be considered and may be responsible for the trends observed in the Lower Keys. Key West is the cruise ship port for the entire Keys region and the most intensively developed island.

The one site that showed a trend of increasing stony coral cover was Jaap Reef. Another site, Molasses Shallow, did show a significant increase in cover since 1999, however it was slightly over the Bonferroni corrected p value of $p<0.002$ ($p=0.01$). Both sites are worth discussing because they are dissimilar in reef geology and community structure, yet both show trends that may be indicative of a slow, gradual recovery of the spatially dominant species that exist within these habitats. Jaap Reef is a high relief nearshore hardbottom composed of a monotypic stand of *Montastraea annularis* complex. Molasses Shallow represents a typical shallow spur and groove reef formation of the Florida Keys that was previously dominated by *Acropora palmata*. The cover of the sentinel species at each of these sites had significantly declined by 2000. *Montastraea annularis* cover decreased from 28% to 14% at Jaap Reef and *A. palmata* decreased from 6% to <0.6% at Molasses Shallow. Since that time, minor increases in cover have been detected and the cover of these two species has averaged 17% and 1.5%, respectively, over the last three years (2006 through 2008). Why corals at these two particular sites have responded positively since 2000 is difficult to explain, but this pattern demonstrates why sites need to be evaluated independently of one another. It should also be pointed out that while the trend over the last decade is positive, cover at these sites is still far below the amount that was recorded in 1996. If anything, these trends suggest the time required for *M. annularis* complex and *A. palmata* to return to levels of 1996 is on the order of decades, given no further substantial bleaching or other large scale disturbances occur.

Macroalgal cover

First, it is important to consider that CREMP image analysis does not identify individual macroalgae species and lumps calcareous, fleshy, and filamentous forms into a single macroalgae category. Annual observations of macroalgae can fluctuate because of variability in the timing of annual sampling, the potential of sampling during a periodic algal bloom, and the ephemeral nature of many algal species that occur on coral reefs. These factors can confound interannual observations of macroalgae, but interpretation of long-term patterns do appear to be consistent. In terms of the overall trend for the region, CREMP results indicate that macroalgal cover has remained relatively similar. Contrary to reports for other Caribbean and Western Atlantic regions (Hughes 1994; Aronson 2000; Rogers and Miller 2006), reefs in the Florida Keys have not undergone a phase-shift towards macroalgal dominance. Although some increases in macroalgal cover have persisted in specific region*habitat groupings, these instances are localized and do not reflect the broader trend for the entire region.

Seven of the ten region*habitat groupings show no long-term trend in macroalgal cover since 1996 (Table 2, Figure 15). One grouping, the Lower Keys backcountry patch reefs, indicates an increasing trend while the Lower Keys deep forereef demonstrates a decreasing trend (Table 2). Even when the first three years of data are excluded, the trends remain the same for all region*habitat groupings. Examining the results at the site level supports the region*habitat grouping trends, where >90% of all sites indicate no trend in macroalgal cover dating back to 1996 or to 1999 (Table 4; Appendices 7 and 8).

The results of both the region*habitat and site levels time–series analyses suggest that the entire region has avoided a prolonged phase-shift towards increased macroalgal cover. While single year spikes or short term blooms have been observed after major disturbances (e.g. after 1997 and 2005), they have not been sustained. Whether due to periodic scouring by hurricanes, herbivorous fish abundance, or other top-down controls, macroalgal cover values remain between 10% and 15% in most areas. Whether these cover values represent an ecologically balanced level of macroalgae on Keys reefs can be debated, but the target level for fleshy macroalgae on Mesoamerican reefs (Belize, Mexico, and Honduras) is set at 15% (HealthyReefs.Org 2010). Most areas in the Keys repeatedly fall under this target for macroalgal cover. In addition, recent evidence demonstrates that mean Caribbean macroalgal cover calculated from 1996-2005 was ~21%, a value nearly double that of the Florida Keys (Schutte et al. 2010). Although it has been previously proposed that Keys reefs have undergone a phase shift towards macroalgae (Maliao et al. 2008), the long-term trends beyond 2000 do not corroborate this finding. Maliao et al. (2008) only used the first five years of CREMP data (1996-2000) and while the brief increase in macroalgal cover following the mass bleaching event lead to this conclusion, elevated levels of macroalgal cover (like those of 1998) have not been observed since.

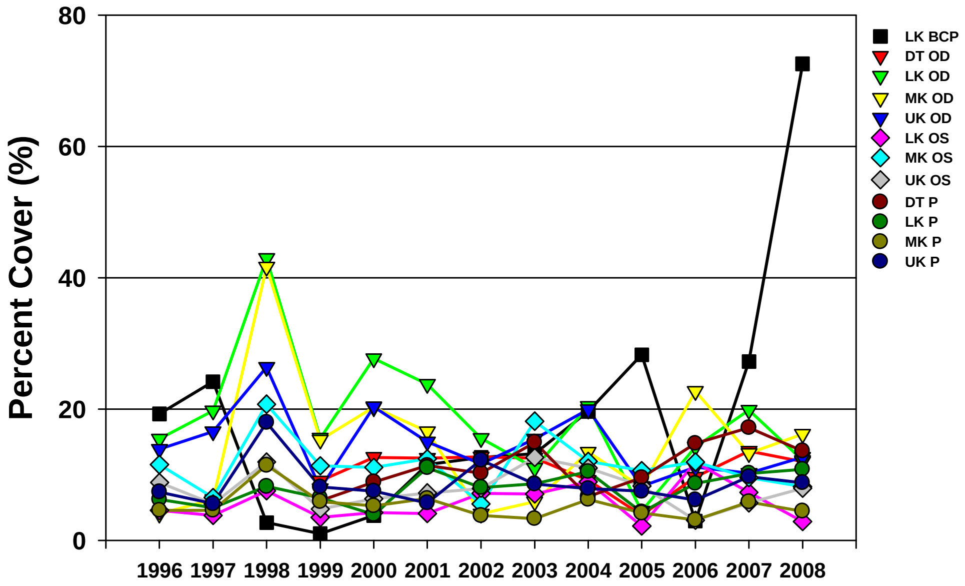


Figure 15. Mean annual percent macroalgal cover for 12 region*habitat groupings in the Florida Keys and Dry Tortugas. *N* for each region*habitat grouping varies (see methods). The trend for each region*habitat grouping was determined from a mixed model regression. The direction for all slopes is reported in Tables 2 and 3.

Deep forereef sites consistently have the highest macroalgal cover and are the most susceptible to periodic blooms (Figure 15). Upwelling is known to bring excess nutrients to the Florida continental shelf and can deliver up to 40 times the amount of nitrogen and phosphorus as compared to terrestrial/nearshore inputs (Leichter et al. 2003). Upwelling varies in space and time so periodic blooms may not uniformly affect all regions of the Keys. This variability also makes assigning causality to periodic blooms challenging. Pulses in macroalgal cover can also be attributed to temperature (Lirman and Biber 2000). It may be no coincidence that the highest macroalgal values observed by CREMP occurred after the extremely warm water temperatures of the 1997/1998 ENSO. Both upwelling and temperature can work simultaneously to foster conditions characteristic of macroalgal blooms, albeit in spatially distinct locales. Since upwelling is usually accompanied by cooler water reaching the continental shelf, CREMP should be able to better distinguish between warm water temperature anomalies and upwelling in the future now that subsurface temperature loggers have been deployed at a subset of CREMP sites throughout the FKNMS.

Octocoral Cover

Similar to macroalgal cover, CREMP image analysis does not differentiate across octocoral species and pools encrusting, branching, and flabellate forms into a single octocoral category. It should also be mentioned that unlike corals, macroalgae, and sponges, quantification of octocoral cover provides an estimation of both canopy and benthic (e.g. encrusting forms) cover. This is important to remember when interpreting the long-term trends for octocorals for at least two reasons. First, canopy cover can increase without requiring additional substrate or the recruitment of new colonies. Thus, octocoral cover can increase without a net gain of individuals. Second, octocoral community structure varies greatly by habitat and by site in the Florida Keys. Because all octocorals are grouped into a single category, a variety of species and/or growth forms may be responsible for apparently similar trends at different sites. Because CREMP pools the results for all phenotypes, interpretation and discussion of trends are only summarized at the gross taxa level.

Five of the ten region*habitat groupings show an increasing trend in octocoral cover since 1996 (Table 2, Figure 16). This trend includes all shallow forereef groupings and the Upper Keys deep and Lower Keys patch reef designations. All other region*habitat groupings demonstrated no change in cover. Considering the decline in octocorals during the first several years of monitoring, the long-term trends demonstrate a resiliency other benthic fauna groups (e.g. stony corals and sponges) have not shared. For most region*habitat groupings, octocoral cover reached low points in 1999 or 2000 (Figure 16). Octocoral cover has been trending positively for all region*habitat groupings except those in the Dry Tortugas and the Lower Keys backcountry patch reefs (Table 2). When using 1999 as a baseline to evaluate recovery after the 1997/1998 ENSO in the Florida Keys, it is clearly evident that a majority of sites are gaining in octocoral cover. This result is most apparent at shallow forereef locations. Of the 12 shallow sites CREMP monitors, ten show an increasing trend ($p<0.002$) in octocoral cover from 1999 to 2008. The two other shallow sites, Carysfort Reef and Rock Key, also had considerable gains in octocoral cover but the regression yielded values slightly above the Bonferroni adjusted p value. In all, 16 of 37 sites (43%) have been increasing in octocoral cover since 1999, while 19 (51%) have not changed, and only 2 (6%) have decreased (Appendix 10). The two sites that decreased in octocoral cover are the deep sites in the Dry Tortugas. Dating back to 1996, when the trend includes the mortality associated with the first five years, only one site shows a decrease in cover suggesting that octocoral cover at nearly all sites has fully recovered (Appendix 9).

Octocorals are susceptible to physical damage from storms and disease (Yoshioka 1994; Cerrano et al. 2000). Mean octocoral cover (pooled for all stations) declined gradually between 1996 and 2000 but there was a substantial decrease after 1998. Another large decline also occurred between 2005 and 2006. During 1998 and 2005, powerful storms (e.g. Hurricane Georges and Wilma) impacted the Keys. Physical abrasion during storms causes partial mortality when branches are severed and whole mortality when holdfasts are detached from the substrate (Coma et al 2004). The amount of partial or whole mortality inflicted varies greatly by depth (because storm surge is usually strongest at shallow forereef locations) and predominately causes complete mortality at these sites due to holdfast detachment (Yoshioka and Yoshioka 1991). The decline in octocoral cover after 1998 may also be linked to thermal stress following the 1997/1998 ENSO. Although CREMP has no direct evidence to support this hypothesis, the 1999 mass mortality of octocorals in the Mediterranean was attributed to extremely high water temperature anomalies which facilitated the onset of a disease outbreak (Cerrano et al. 2000). It would not be unreasonable to assume that disease also contributed, in some part, to the mortality observed because the relationship between elevated sea temperatures and increased disease virulence has been substantiated (Harvell et al. 1999; Bruno et al. 2007).

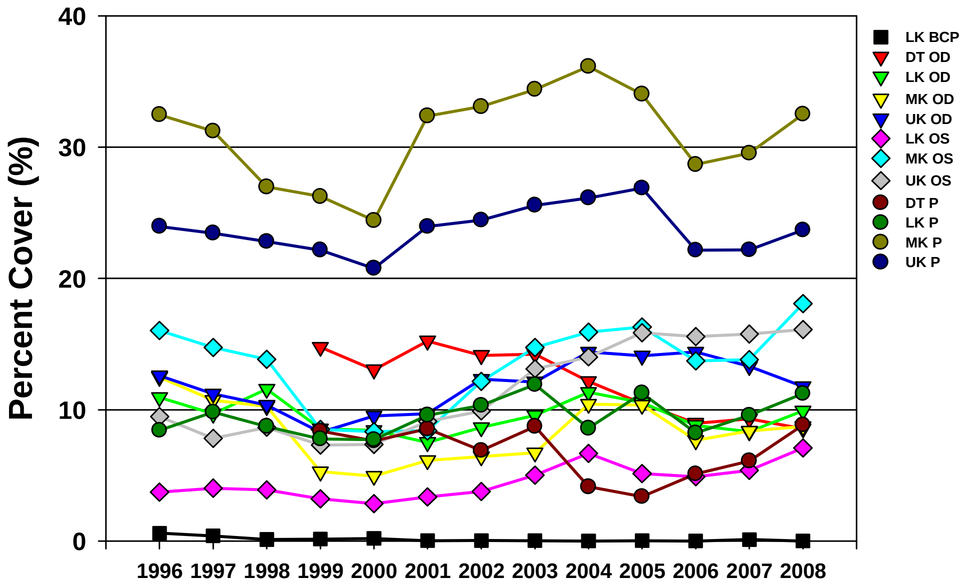


Figure 16. Mean annual percent octocoral cover for 12 region*habitat groupings in the Florida Keys and Dry Tortugas. *N* for each region*habitat grouping varies (see methods). The trend for each region*habitat grouping was determined from a mixed model regression. The direction for all slopes is reported in Tables 2 and 3.

The time required for octocoral cover to be restored after a disturbance is highly dependent upon the type of mortality inflicted. Regeneration can be much faster if it begins from already established colonies that only suffered partial mortality. The onset of recovery is delayed when the settlement and growth of new recruits is required. The slow and gradual increase in octocoral cover at many reefs may reflect the latter recovery process. It is likely that hurricanes, in part, were responsible for reductions in octocoral cover after 1998 and 2005 by stripping whole colonies from the substrate. It would be expected that re-establishing octocoral cover to previous levels would require several years of recruitment and growth. In addition, CREMP image analysis favors large, spatially dominant organisms because the probability of a random point landing on smaller taxa is reduced. Even if recruitment of octocorals was initially high following a hurricane it would only contribute a minor fraction to overall benthic cover because of the small size of recruits. An increase in octocoral cover would likely require several years of uninterrupted growth before CREMP image analysis would begin to detect the increase in cover. This lapse explains why octocoral cover values after a disturbance remain stagnant for a year or two before gradually increasing in many region*habitat groupings.

The most dramatic increases in octocoral cover have occurred at the shallow forereef sites. In 2008, cover at most shallow forereef sites had not only recovered to 1996 levels but exceeded them. When combined for all sites in this habitat designation, octocoral cover has increased from

8.7% in 1996 to 12.9% in 2008. The increase is even more impressive considering octocoral cover had declined to 5.8% by 2000. Prior to 2000, *Acropora palmata* and *Millepora complanata* were still the two predominant stony coral species at many shallow forereef locations. As discussed above, both of these species suffered mass mortalities due to bleaching and disease during the early years of CREMP monitoring and have shown little or no recovery since. The loss of these species opened up large areas of substrate for other organisms to colonize. Octocorals appear readily adapted to persist in the harsh shallow forereef environment and can tolerate wave action and intense ultraviolet radiation at the shallow depth. The transition from stony coral to octocoral dominated communities has been observed before, however all examples are exclusive to the Pacific (Endean et al. 1988; Fox et al. 2003; Stobart et al. 2005). Phase shifts from stony coral to macroalgae or sponge dominated assemblages have been reported for Belize, Jamaica and other Caribbean/Western Atlantic locations (Aronson 2005; Rogers and Miller 2006; Norstrom et al. 2009), but the long-term trends reported here suggest a shift to octocoral dominated communities. Although this trend is readily apparent on shallow forereefs, and is concurrent with the demise of stony corals, increasing octocoral cover at some deep forereef and patch reef sites suggests the transition is starting within other habitats in the Florida Keys.

Sponge Cover

Like macroalgae and octocoral categories, all growth forms are pooled into a single sponge category during CREMP image analysis. Of the four major taxonomic groups CREMP identifies, sponges generally are the smallest contributor to biotic cover at most sites. Six of the ten region*habitat groupings show a negative trend in sponge cover since 1996 (Table 2). Middle and Upper Keys deep forereefs and Lower and Upper Keys patch reefs did not change in sponge cover (Figure 17, Table 3). None of the region*habitat groupings show a positive trend in cover since 1996. The outlook does not change if 1999 is used as a starting point. All but four of the 12 region*habitat groupings show a decreasing trend in sponge cover (Table 3) with the same four region*habitat groupings exhibiting no change in cover. Both region*habitat groupings in the Dry Tortugas show a declining trend since 1999. When examined by site, only four of the 34 sites show a negative trend in cover since 1996 (Appendix 11). The same four sites are the only sites to show a decreasing trend when analyzed from 1999 to 2008 (Appendix 12). This result suggests that sponge cover is only marginally decreasing at many sites, but when these sites are clustered within their specific region*habitat grouping, a stronger negative trend is revealed for the larger spatial area.

Sponges, similar to stony corals and octocorals, are vulnerable to disease, bleaching, sedimentation, physical damage from storms, and harmful algal blooms. Many of these factors have contributed to the decline of sponge cover at localized and regional spatial scales. Widespread declines were recorded after the 1997/1998 ENSO. Between 1997 and 1999 sponge cover in the Florida Keys was reduced from 2.7% to 2.1%. Most notable were the declines at the deep forereef locations. *Xestospongia muta* provide a large portion of the sponge cover in these habitats and are susceptible to both disease and bleaching (Cowart et al. 2006; Lopez-Legentil et al. 2008). Bleaching of *X. muta* has been observed on other reefs during periods of severe thermal stress and lack of recovery may have been a cause of the decline in sponge cover observed from 1997 to 1999. Another regional pattern of decline was observed following the 2005 hurricane season. Several large storms like Wilma directly impacted the Florida Keys. Hurricane Wilma alone inundated parts of the Keys with >10' storm surge. Wave action and heavy storm surge increase sedimentation which smother and clog sponge oscules, resulting in suffocation (Schmahl 1990). Localized events have also influenced sponge cover at specific areas. The harmful algal bloom that spread across the West Florida shelf and Florida Bay in 2001 decimated sponge cover on Lower Keys backcountry patch reefs. This diatom bloom significantly reduced the cover of sponges (much like stony corals) from 3.3% in 2001 to 0.6% in 2002 in this region*habitat grouping (Figure 17). Algal and cyanobacteria blooms occur periodically in Florida Bay and also have virulent effects on sponge populations and seagrass communities (Butler IV 1995).

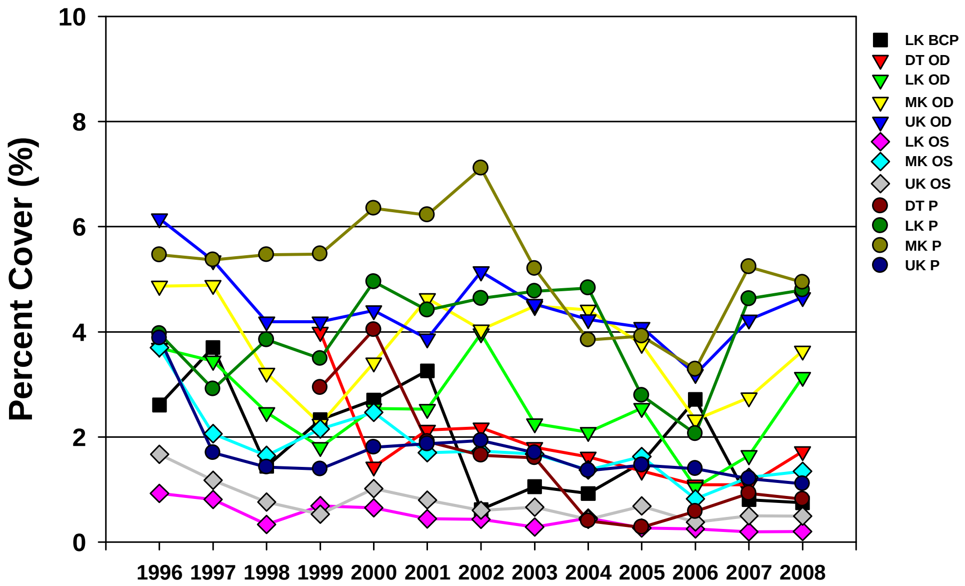


Figure 17. Mean annual percent sponge cover for 12 region*habitat groupings in the Florida Keys and Dry Tortugas. *N* for each region*habitat grouping varies (see methods). The trend for each region*habitat grouping was determined from a mixed model regression. The direction for all slopes is reported in Tables 2 and 3.

It is noteworthy that none of the region*habitat groupings nor individual sites have shown an increase in sponge cover since 1996 or 1999. Sponges on other Caribbean reefs have rapidly increased in cover or abundance in areas previously occupied by stony corals following a major disturbance (Williams et al. 1999; Aronson et al. 2002; Rutzler 2002; Weil et al. 2002). Although this transition was reported for the Florida Keys, it was mostly drawn from nearshore hardbottom environments undergoing eutrophication, rather than from patch and forereef habitats (Ward-Paige et al. 2005). It may be the frequency of major disturbances affecting the Florida Keys that limits the potential transition from stony coral to sponge dominated communities. It also appears, at least in the interim, that sponges along the Florida reef tract are at competitive disadvantage in colonizing newly available substrate compared to the ability and success of octocorals.

Sentinel Stony Coral Species

To further evaluate the decline of stony corals, long-term trends were analyzed from the percent cover data for five sentinel coral species. Following the decline of *Acropora palmata* in the late 1990s, *Montastraea annularis* complex (includes all *annularis* subspecies), *Montastraea cavernosa*, *Colpophyllia natans*, *Siderastrea siderea*, and *Porites astreoides* represent the five most spatially-dominant scleractinians in the Florida Keys and Dry Tortugas. Of these five species, *M. annularis* complex has historically maintained the highest spatial coverage of any coral species. At project inception, *M. annularis* complex cover was greater than 5% in the Florida Keys and more than 8% in the Dry Tortugas (Figure 18). The next most spatially abundant species is *M. cavernosa,* with cover ranging between 2% and 4%, depending upon the region. For both *Montastraea* spp., coverage is higher in the Dry Tortugas than in the Florida Keys. *Colpophyllia natans, S. siderea*, and *P. astreoides* round out the top five, comprising between 0.5% and 2% cover and represent the third through fifth most spatially abundant corals. When percent cover data are pooled for all stations in the Florida Keys (*N*=97) from 1996 to 2008, and in the Dry Tortugas (N=12) from 1999 to 2008, *M. annularis* complex, *M. cavernosa*, *C. natans*, and *P. astreoides* exhibit decreasing trends in spatial cover (Figures 18 and 19). *Siderastrea siderea* is the only species to buck the negative trend as cover for this species has remained relatively similar throughout the project. Although the trends in this report present an overall decline for *M. annularis* complex, *M. cavernosa*, and *C. natans*, the decrease in cover for these species is often driven by the death of a few, large individuals. The size-specific mortality of large boulder and brain corals colonies is not a new phenomenon and has been described by Porter and Meier (1992) in Florida and by Hughes et al. (2000) in Jamaica. These three species are all broadcast spawners, and poor recruitment rates have not been able to mediate the loss of cover for these massive species (Hughes and Tanner 2000; Miller et al. 2000). Surprisingly, even the brooding species, *P. astreoides*, shows an overall decreasing trend. This is not a massive, reef building coral but a “weedier” species that is adept at quickly colonizing available substrate after disturbances. In some parts of the Caribbean, *Porites astreoides* has increased in relative abundance and mediated the loss of coral cover (Green et al. 2008) but this scenario is not reflected by the CREMP trends observed in the Florida Keys and Dry Tortugas. *Siderastrea siderea* appears to be the most tolerant of these five sentinels, and has shown no response to the acute and chronic stressors that have reduced the cover of the other four species.

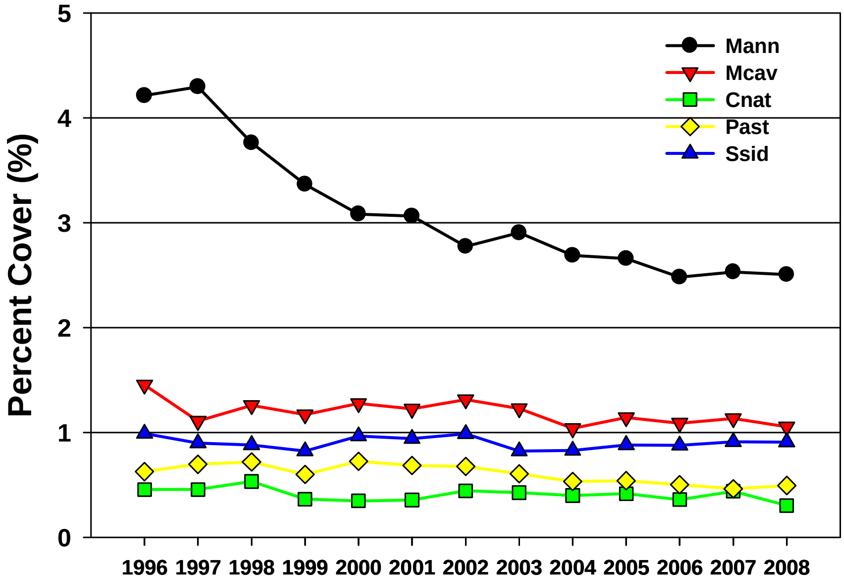


Figure 18. Mean annual percent cover of *Montastraea annularis* complex (Mann), *M. cavernosa* (Mcav), *Colpophyllia natans* (Cnat), *Porites astreoides* (Past), and *Siderastrea siderea* (Ssid) in the Florida Keys (*N*=97). Mean percent cover is pooled from 97 stations in the Florida Keys. A mixed model regression indicates trends for all species were decreasing ($p<0.002$) except *Siderastrea siderea*.

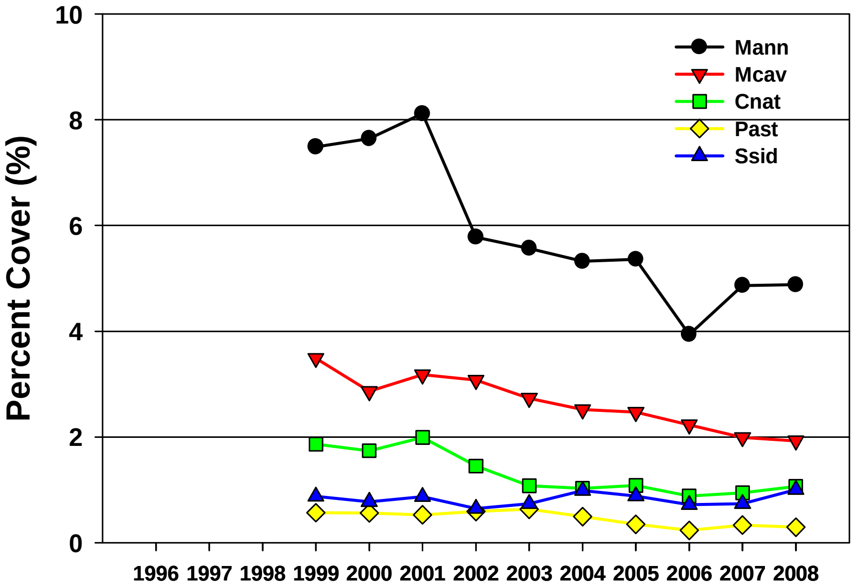


Figure 19. Mean annual percent cover *Colpophyllia natans* (Cnat), *Montastraea annularis* complex (Mann), *M. cavernosa* (Mcav), *Porites astreoides* (Past), and *Siderastrea siderea* (Ssid) in the Dry Tortugas (*N*=12). Mean percent cover is pooled from 12 stations in the Dry Tortugas. A mixed model regression indicates trends for all species were decreasing ($p<0.002$) except *Siderastrea siderea*.

Montastraea annularis complex

Montastraea annularis complex has suffered the largest loss of spatial cover during the CREMP monitoring timeframe, with widespread declines occurring in most regions and habitats. Since 1996, *M. annularis* complex cover has had a negative trend at seven of ten region*habitat groupings (Table 5). These trends are upheld across all habitat types in the Lower Keys region, and at deep forereefs in the Middle and Upper Keys. Even at patch reefs in the Middle and Upper Keys, the p values ($p=0.008$ for both regions) were only slightly above the adjusted Bonferroni corrected value ($p \leq 0.004$) and is supportive of a decreasing trend in both region*habitat groupings. Of all the decreases observed during the initial years, the most dramatic occurred at the Lower Keys patch reefs when cover was reduced from ~11.6% in 1997 to ~6.0% in 2000. When the analyses are conducted starting in 1999 (after the immediate effects of the 1997/1998 ENSO are removed), the number of declining region*habitat groupings decreases from seven to five (Table 6). Since 1999, all habitat types in the Lower Keys show a continued decline except for the patch reefs. The widespread decline in *M. annularis* is also clearly evident when data are analyzed at the site level: 15 of 34 (44%) sites have had a significant reduction in cover since 1996 (Appendix 13). Site-level declines were less extreme after 1999, yet a third of all sites (12 of 37 sites) continued to lose *M. annularis* complex cover at a significant rate (Appendix 14). The most alarming rate of loss since 1999 has occurred in the Lower Keys and Dry Tortugas deep sites. When combined with the results of the Lower Keys shallow forereefs, the trends suggest a strong regionalized effect that may be tied to the intense development of the Lower Keys. *Montastraea annularis* is highly susceptible to disease outbreaks, bleaching, and hurricane damage, and the synergistic effect of these multiple stressors may be exacerbated by additional stressors related to intensive coastal development in the Lower Keys. The decline of this species has been widely reported throughout the Caribbean even on remote, isolated reefs (Hughes 1994; Gardner et al. 2003; Edmunds 2007).

Table 5. Long-term trends for: *Montastraea annularis* complex (Mann), *M. cavernosa* (Mcav), *Colpophyllia natans* (Cnat), *Porites astreoides* (Past), and *Siderastrea siderea* (Ssid). Trends were determined for 10 region*habitat groupings from a mixed model regression from 1996-2008. Interpretation of trends for each region*habitat grouping are based on Bonferroni corrected p values for repeated testing (adjusted $p \leq 0.004$).

Reg*Hab	Mann	Mcav	Cnat	Past	Ssid
LK BCP	decreasing	no change	no change	decreasing	no change
LK OD	decreasing	decreasing	decreasing	decreasing	no change
LK OS	decreasing	decreasing	decreasing	decreasing	no change
LK P	decreasing	decreasing	no change	no change	no change
MK OD	decreasing	decreasing	no change	no change	no change
MK OS	no change	decreasing	no change	no change	no change
MK P	no change	no change	no change	no change	no change
UK OD	decreasing	decreasing	no change	no change	no change
UK OS	decreasing	decreasing	no change	no change	no change
UK P	no change	no change	no change	no change	no change

Table 6. Long-term trends for *Montastraea annularis* complex (Mann), *M. cavernosa* (Mcav), *Colpophyllia natans* (Cnat), *Porites astreoides* (Past), and *Siderastrea siderea* (Ssid). Trends were determined for 12 region*habitat groupings from a mixed model regression from 1999-2008. Interpretation of trends for each region*habitat grouping are based on Bonferroni corrected p values for repeated sampling (adjusted p≤0.004).

Reg*Hab	Mann	Mcav	Cnat	Past	Ssid
DT OD	decreasing	decreasing	decreasing	decreasing	no change
DT P	no change	no change	decreasing	no change	no change
LK BCP	decreasing	no change	no change	decreasing	no change
LK OD	decreasing	decreasing	decreasing	no change	no change
LK OS	decreasing	no change	no change	decreasing	no change
LK P	no change	no change	no change	no change	no change
MK OD	decreasing	no change	no change	no change	no change
MK OS	no change	no change	no change	no change	no change
MK P	no change	no change	no change	no change	no change
UK OD	no change	no change	no change	no change	no change
UK OS	no change	decreasing	no change	no change	no change
UK P	no change	no change	no change	no change	no change

Table 7. Long-term trends for *Montastraea annularis* complex (Mann), *M. cavernosa* (Mcav), *Colpophyllia natans* (Cnat),*Porites astreoides* (Past), and *Siderastrea siderea* (Ssid) at CREMP sites from 1996-2008 and 1999-2008. Trends were determined for 34 sites sampled since 1996 and 37 sites since 1999 from a mixed model regression. Interpretation of trends for each site are based on Bonferroni corrected p values for repeated sampling (adjusted p<0.002). The trends at each site are shown in Appendices 13-22.

	MANN		**MCAV**		**CNAT**		**PAST**		**SSID**	
	96-08	**99-08**	**96-08**	**99-08**	**96-08**	**99-08**	**96-08**	**99-08**	**96-08**	**99-08**
Decreasing	15	12	6	5	5	3	5	5	0	0
No Change	19	25	28	32	29	33	28	32	33	37
Increasing	0	0	0	0	0	1	1	0	1	0

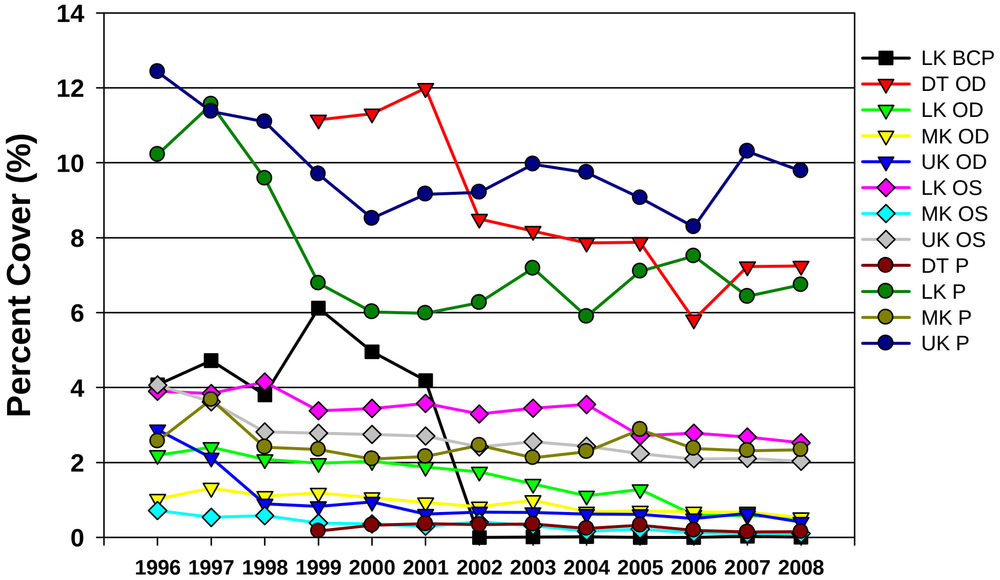


Figure 20. Mean annual percent cover of *Montastraea annularis* complex at 12 region*habitat groupings in the Florida Keys and Dry Tortugas. *N* varies for each region*habitat grouping (see methods). Slope direction for each region*habitat grouping is reported in Tables 5 and 6.

<u>*Montastraea cavernosa*</u>

The long-term trends for *Montastraea cavernosa* are similar to those for *M. annularis* complex. The overall trajectory (pooled for all stations) in the Florida Keys or the Dry Tortugas is negative whether the analysis starts in 1996 or 1999. Since 1996, seven of ten region*habitat groupings in the Florida Keys showed a decline in *M. cavernosa* cover (Table 5). Only three region*habitat groupings have historically had cover values >1.0%: Middle Keys patch reefs, Lower Keys patch reefs and Dry Tortugas deep sites (Figure 21). Of the seven region*habitat groupings showing a decreasing trend since 1996, only the Lower Keys patch reefs started with cover values >1.0%. When data from 1996-1998 are excluded from the analysis, only three of the 12 region*habitat groupings show a decline in percent cover (Table 6). Of the three region*habitat groupings showing a decreasing trend since 1999, only the Dry Tortugas deep sites started with cover values above 1.0%. This area has experienced a steep decline in cover, decreasing from 5.2% in 1999 to 2.9% in 2008. The difference in trends if the baseline begins prior to or after 1999 suggests the stress associated with 1997/1998 ENSO was primarily responsible for the reduction of *M. cavernosa* cover at many locations. For example, the number of sites where the cover of *M. cavernosa* was declining in the Florida Keys was reduced by 50% after 1999 (Appendices 15 and 16). Although this result may be partially due to diminished cover values for *M. cavernosa* at some sites, the cover of this species does not show a decreasing trend since 1999 in the region*habitat groupings where it is most abundant: Middle and Lower Keys patch reefs. Thus, the trends for *M. cavernosa* may be directed by a species-specific vulnerability to large-scale bleaching events, but the coral may be less sensitive to other stressors that continue to reduce the cover *Montastraea annularis* complex. A consistency across both *Montastraea* spp. is that both corals appear to have fared better on patch reefs than at deeper sites since 1999 (Appendices 13 through 16).

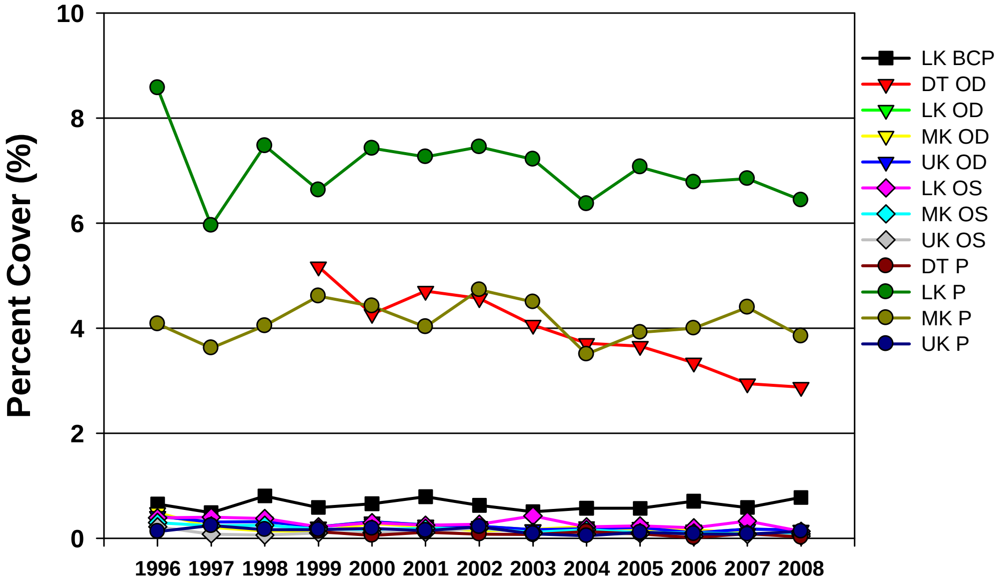


Figure 21. Mean annual percent cover of *Montastraea cavernosa* at 12 region*habitat groupings in the Florida Keys and Dry Tortugas. *N* varies for each region*habitat grouping (see methods). Slope direction for each region*habitat grouping is reported in Tables 5 and 6.

<u>*Colpophyllia natans*</u>

Similar to *Montastraea cavernosa*, *Colpophyllia natans* cover is highest on patch reefs in the Middle and Lower Keys and the deep sites in the Dry Tortugas, ranging between 1.0% and 2.8%, while cover values for all other are region*habitat groupings are <1.0% (Figure 22). When pooled for all stations, long-term trends are negative in the Florida Keys and Dry Tortugas whether the analysis starts in 1996 or 1999. The declines at the region*habitat level are not as widespread as for *Montastraea annularis* and *M. cavernosa*. At the region*habitat level, only deep and shallow forereefs in the Lower Keys indicate a decrease in cover since 1996 (Table 5). This trend continues from 1999-2008 on the Lower Keys deep reefs and is evident in both Dry Tortugas habitats. The noticeable decline in cover between 2001 and 2003 in the Dry Tortugas was due to a disease outbreak at Bird Key Reef that reduced *C. natans* cover from 3.5% in 2001 to 0.7% in 2003 at this site (Morrison et al. 2008). *Colpophyllia natans* is vulnerable to a variety of diseases and disease outbreaks have occurred following major disturbance events (Whelan et al. 2007). Also similar to *M. cavernosa*, *C. natans* cover was strongly influenced by the 1997/1998 ENSO. For example, when examined by site in the Florida Keys from 1996-2008, five sites show a negative trend (Appendix 17). After 1999, the number of sites showing a decreasing trend is reduced to three but two of the sites were in the Dry Tortugas. Only one site continued to decline in the Florida Keys after 1999 (Appendix 18). On a positive note, cover has increased at one site, Western Head, a Lower Keys patch reef. Like *M. cavernosa*, this species appears more vulnerable to large-scale bleaching events and may be more tolerant of chronic stressors.

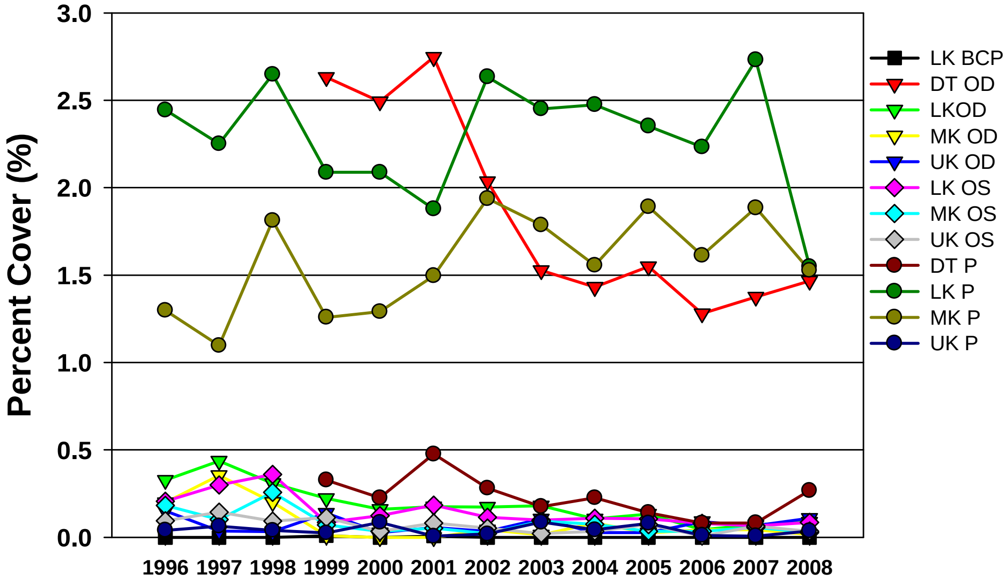


Figure 22. Mean annual percent cover of *Colpophyllia natans* at 12 region*habitat groupings in the Florida Keys and Dry Tortugas. *N* varies for each region*habitat grouping (see methods). Slope direction for each region*habitat grouping is reported in Tables 5 and 6.

<u>*Porites astreoides*</u>

Porites astreoides is found in a variety of habitats. The overall trend for *P. astreoides* (pooled for all stations) in the Florida Keys or for the Dry Tortugas is negative whether the timeline starts in 1996 or 1999. The trend in the Florida Keys is principally driven by three region*habitat groupings that show a significant decline when analyzed from either 1996 or 1999: Lower Keys deep and shallow forereef sites and backcountry patch reefs. When analyzed from 1999 to 2008, three region*habitat groupings show a significant decline: the Lower Keys shallow forereef and backcountry patch reefs and the Dry Tortugas deep sites. Although the 1997/1998 ENSO event reduced the cover of the large boulder and brain corals discussed above, the cover of *P. astreoides* did not significantly change early in the project except at the Lower Keys deep reefs. Of the three region*habitat groupings that had negative trends from 1996 to 2008, this was the only grouping in which cover did not show a significant trend when analyzed from 1999 to 2008. The other two groupings in the Florida Keys, as well as Dry Tortugas deep sites, had negative trends from1999 to 2008, demonstrating a large proportion of *P. astreoides* mortality has been recent. At both the Dry Tortugas deep sites and Lower Keys patch reefs, precipitous and consecutive annual declines in *P. astreoides* cover were observed starting in 2003 and continued through 2006 (Figure 23). At the backcountry patch reefs, the collapse in cover coincided with the diatom algal bloom in 2001. Site level results reflect the patterns of loss observed at the region*habitat level, with five sites indicating a negative trend in both time series from 1996 to 2008 or 1999 to 2008 (Appendices 19 and 20). This pattern indicates that the most severe losses of *P. astreoides* have occurred following the 1997/1998 ENSO event. No consistent pattern was found among the sites with a decreasing trend, as at least one site in nearly all regions and habitats had a negative trend (Appendix 20).

The results presented here are somewhat surprising based upon the findings reported elsewhere in the Caribbean. Given both the acute and chronic nature of the stressors influencing the Florida Reef Tract, corals with r-selected life histories would be expected to thrive. *Porites astreoides* is considered a faster growing, "weedier" species that has high recruitment rates because of its brooding reproductive strategy (McGuire 1998; Knowlton 2001). This is an ideal solution for surviving in periodically disturbed habitats. An increase in the relative abundance and cover of *P. astreoides* has been documented on multiple reefs in the Caribbean during the last several decades (Green et al. 2008), and would be expected in the Florida Keys due to the widespread declines of acroporid corals and *Montastraea annularis* complex. Recruitment rates and juvenile abundance appear to be high for *P. astreoides* on Florida reefs (Miller et al. 2000; Keller and Donahue 2006)) but survival of larger adult colonies must be low since cover has remained similar or declined at many locations. Although *Porites astreoides* is reported to be resistant to acute stressors such as warm water anomalies (Gates et al. 1990) and some chronic stressors such as sedimentation (Gleason 1998), the synergistic effect of multiple chronic stressors such as nutrient enrichment, over-fishing, and sedimentation do inhibit the recruitment and survival of smaller colonies for many species (Bak and Meesters 1999; Mumby et al. 2007). It can only be suggested that the level of chronic stressors influencing the Florida Keys and Dry Tortugas are strong enough to have deleterious impacts on *P. astreoides*. Another possibility is that the increasing shift to octocoral dominance may be limiting the recruitment and survival of *P. astreoides* at some locations, especially on shallow forereefs. Direct spatial competition and allelopathic interactions between octocorals and stony corals have been shown to inhibit the recruitment of stony coral larvae (Maida et al. 1995; Maida et al. 2001).

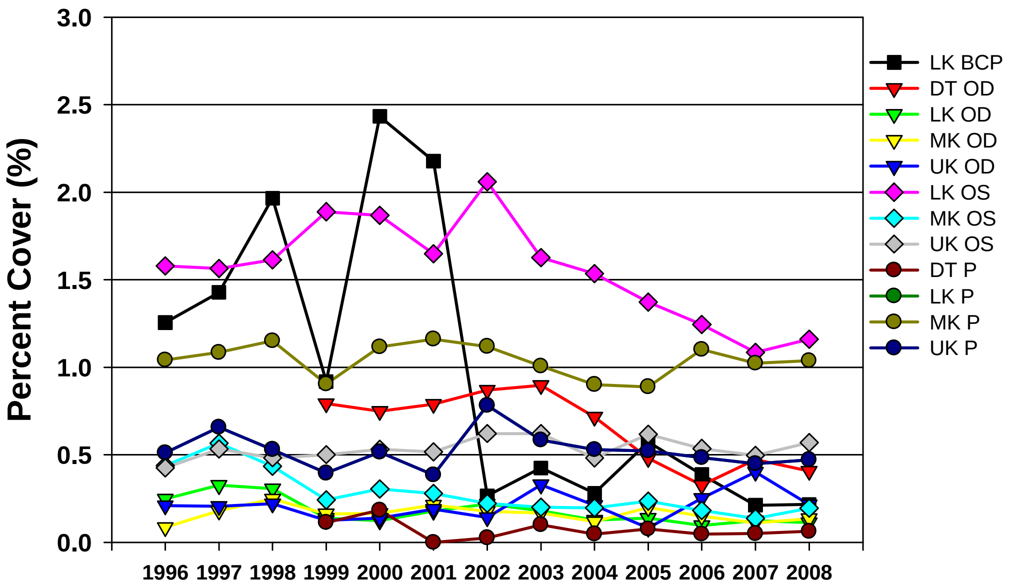


Figure 23. Mean annual percent cover of *Porites astreoides* at 12 region*habitat groupings in the Florida Keys and Dry Tortugas. *N* varies for each region*habitat grouping (see methods). Slope direction for each region*habitat grouping is reported in Tables 5 and 6.

<u>*Siderastrea siderea*</u>

Siderastrea siderea is the only species of the five sentinel stony corals that does not show an overall decreasing trend in cover whether pooled for all stations in the Florida Keys or the Dry Tortugas, regardless of analyses started in 1996 or 1999. Mean annual percent cover for *S. siderea* has remained relatively unchanged since 1996 (Figures 18 and 19). All ten region*habitat groupings indicate no trend in cover in the Florida Keys since 1996 and the trend in cover for all 12 region*habitat groupings has remained flat in the Florida Keys and Dry Tortugas since 1999. Results at the site level mimic those of the region*habitat groupings. Thirty-three of 34 sites demonstrate no change in cover from 1996 to 2008 and all 37 sites show no change in cover from 1999 to 2008 (Table 7; Appendices 21 and 22). The only site to show a change in cover was Cliff Green, a Lower Keys patch reef, and the trend at this site was positive (Appendix 21). *Siderastrea siderea* mean cover was highest on patch reefs in the Florida Keys, especially in the Middle and Lower Keys (between 2.1% and 4.0%), followed by deep sites where cover ranges between 0.5% and 1.5%.

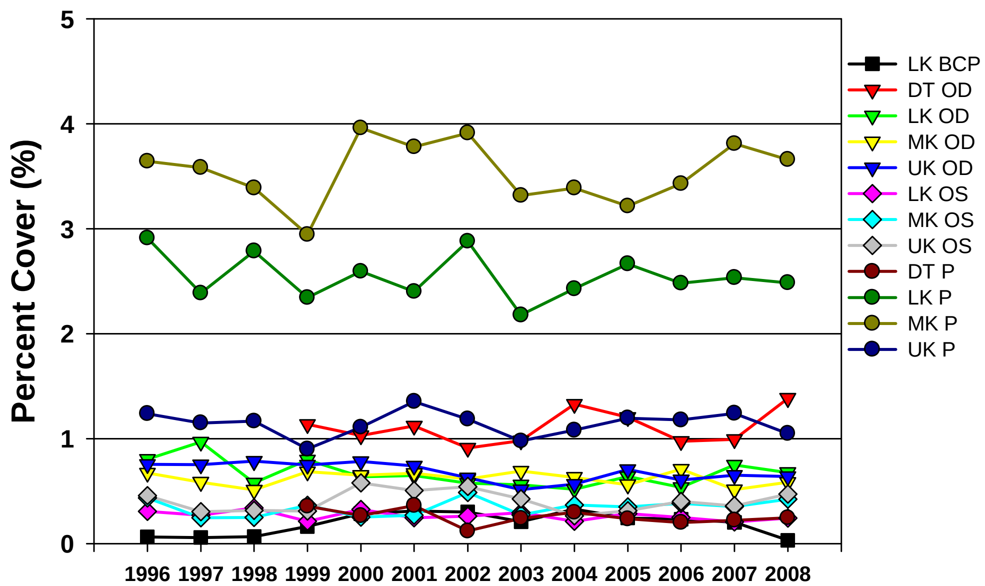


Figure 24. Mean annual percent cover of *Siderastrea siderea* at 12 region*habitat groupings in the Florida Keys and Dry Tortugas. *N* varies for each region*habitat grouping (see methods). Slope direction for each region*habitat grouping is reported in Tables 5 and 6.

Of the five sentinel coral species examined, *Siderastrea siderea* is the most resilient in the Florida Keys and Dry Tortugas. Unlike the other large, framework building corals, *Montastraea* spp. and *Colpophyllia natans*, dramatic reductions in the cover of *S. siderea* did not occur following the 1997/1998 ENSO. *Siderastrea siderea* appears tolerant of adverse conditions during these extreme disturbances and the fact that the long-term trend in cover has remained unchanged at nearly all locations suggests the species is also resistant to chronic stressors. *Siderastrea siderea* is widely distributed around Florida, is common on shallow, nearshore carbonate Gulf of Mexico platform reefs on the Florida West coast, and is also found at the very northern extension of the Florida Reef Tract in Martin County (Colella et al. 2008; Gilliam

2009). Reefs in both of these localities frequently experience turbid conditions, wide fluctuations in sea temperature, and are subject to terrestrial inputs and runoff from adjacent estuaries (Riegl and Dodge 2008). The relatively high abundance of *S. siderea* in both stable and harsh environments lends evidence to the highly adaptive nature of this species. It may be unexpected that *S. siderea* would be the most suited to conditions in the Florida Keys and Dry Tortugas, but changes in coral species composition can be regionally and locally specific. Although *P. astreoides* has partially mediated the decrease in coral cover on many reefs in the Caribbean (Green et al. 2008), a transition from *Acropora cervicornis* to *Agaricia tenuifolia* has been observed in Belize (Aronson and Precht 2001) and from *A. cervicornis* to *Porites porites* in the Bahamas (Curran et al. 1994). If a shift in coral community structure is to occur in the Florida Keys, it certainly appears that *S. siderea* would be the viable candidate to become the most spatially abundant coral.

<u>*Clionaid Cover*</u>

Colony area (cm^2/m^2) and abundance (number of colonies per station) were recorded for three species of clionaid sponges (*Cliona delitrix, C. lampa,* and *C. caribbaea)*. Since 2001, the mean clionaid area, when pooled from all stations (*N*=109) for all three species, has ranged between four and $8cm^2/m^2$ annually. Mean clionaid area was highest in 2001 ($7.9\pm1.9cm^2/m^2$) and reached its lowest point in 2003 (4.8 ± 1.4 cm^2/m^2). After 2003, clionaid area steadily climbed until 2006 and dipped slightly in 2007 to $5.4\pm1.5cm^2/m^2$ before slightly increasing to 5.9 ± 1.7 cm^2/m^2 in 2008.

At the species level, when pooled for all stations (*N*=109), both mean area and abundance for *C. delitrix* was higher than those of *C. caribbaea* and *C. lampa*. *Cliona delitrix* mean area has fluctuated between $2\text{-}5cm^2/m^2$ annually, *C. caribbaea* between $1\text{-}3cm^2/m^2$, and *C. lampa* from $0\text{-}3cm^2/m^2$. *Cliona lampa* has been the most variable of all species, decreasing to nearly zero in 2002 while increasing to $>2cm^2/m^2$ in 2006. *Cliona lampa* is only present at a few CREMP sites in the Florida Keys, and is predominately found at one of the Lower Keys backcountry patch reefs, Content Keys. In 2001, this site was directly impacted by the harmful algal bloom that killed much of the benthic fauna at these sites, and in 2002, *C. lampa* was not observed at Content Keys. In general, *C. delitrix* area and abundance has been decreasing since 2001. Although the area of *C. caribbaea* and *C. lampa* fluctuated from 2001 through 2008, the relative abundance of the two species has remained similar (Figure 25). The decline in spatial cover of *C. delitrix* is most likely concomitant with decreasing abundance. Between 2001 and 2008, *Cliona delitrix* abundance nearly dropped in half.

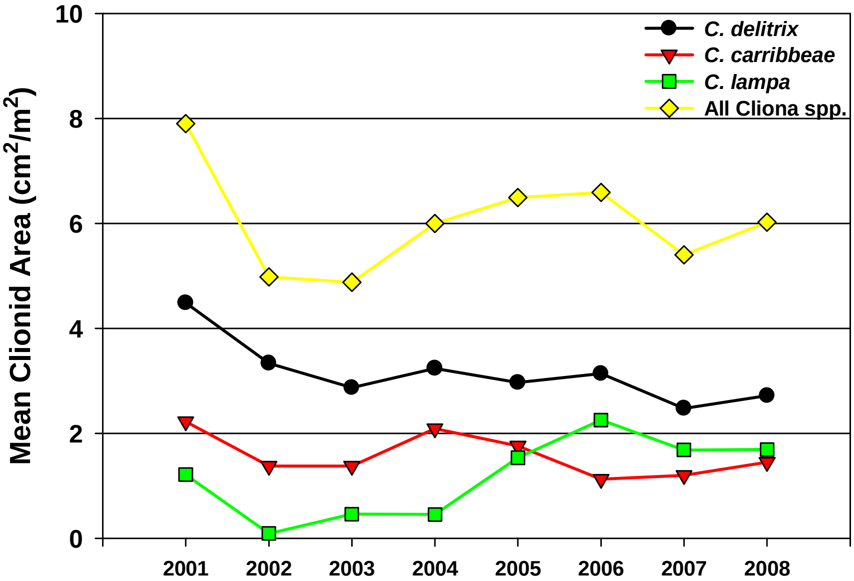


Figure 25. Mean area for *Cliona delitrix*, *C. caribbaea* and *C. lampa* for all CREMP stations surveyed in the Florida Keys and Dry Tortugas from 2001 to 2008 (N=109 stations).

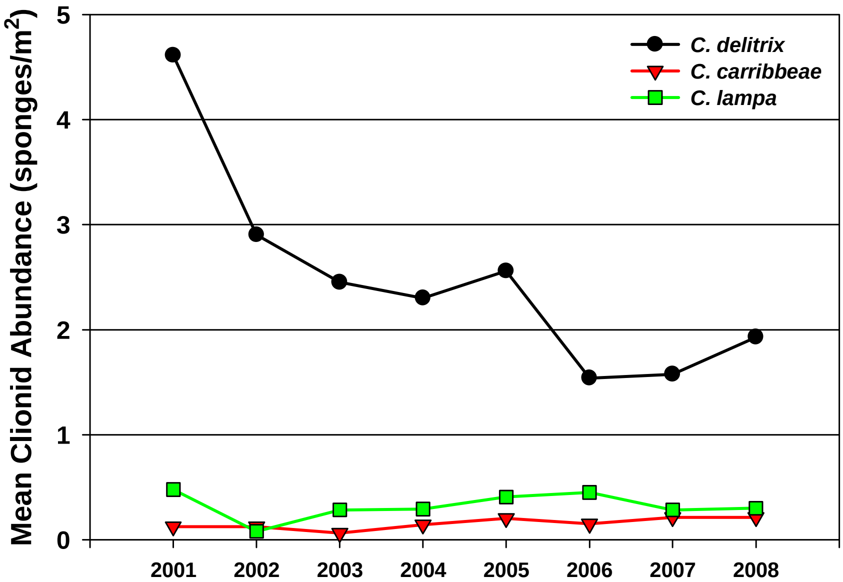


Figure 26. Mean abundance for *Cliona delitrix*, *C. caribbaea* and *C. lampa* for all CREMP stations surveyed in the Florida Keys and Dry Tortugas from 2001 to 2008 (N=109 stations).

To evaluate the long-term trends of clionaid cover, *Cliona delitrix* was selected as a proxy because it was the most widely distributed and spatially dominant clionaid species within CREMP stations. When the data is pooled for all stations in the Florida Keys and Dry Tortugas (N=109), there is an overall decreasing trend from 2001 to 2008 ($p<0.001$). Three of the 12 region*habitat groupings had a significant negative trend in mean *C. delitrix* area between 2001 and 2008, Middle and Upper Keys offshore deep reefs and Lower Keys patch reefs (Table 8). All other region*habitat groupings did not indicate a trend, but the results for Lower Keys and Dry Tortugas deep sites suggest that *C. delitrix* cover is decreasing in these areas as well (Table 8). The highest cover of *C. delitrix* was observed at the patch reefs in the Lower Keys (Figure 27), while shallow forereef sites generally had the lowest mean area. By site, five of the 37 sites showed a decrease in *C. delitrix* cover: two Lower Keys sites, Cliff Green patch reef and Smith Shoal backcountry patch reef, and three deep forereef sites, Bird Key Reef in the Dry Tortugas, Sombrero Reef in the Middle Keys, and Carysfort Reef in the Upper Keys (Table 9; Appendix 23). No change in mean area was detected at the other 32 sites.

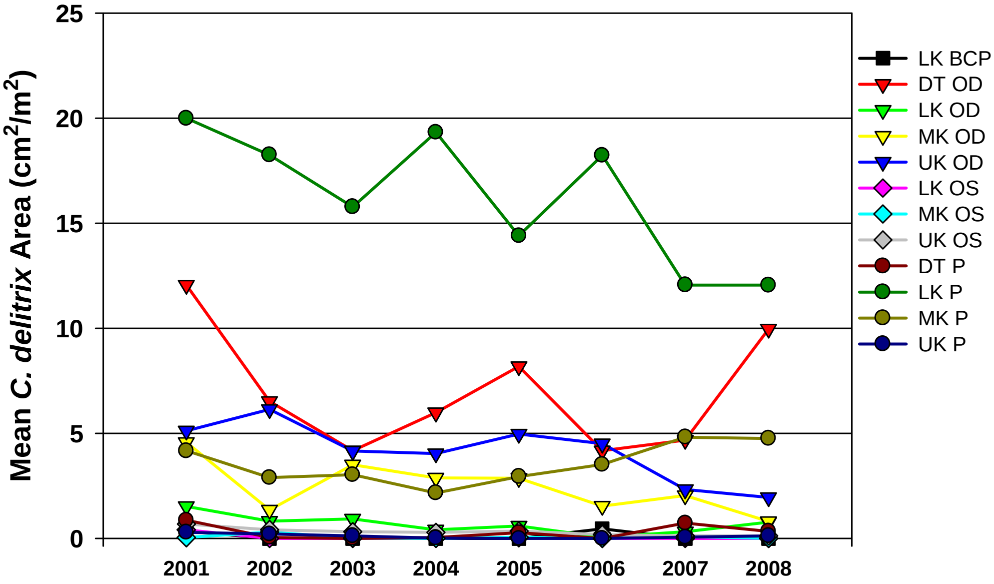


Figure 27. Mean *Cliona delitrix* area (cm^2/m^2) for 12 region*habitat groupings in the Florida Keys and Dry Tortugas from 2001 to 2008 (*N*=109 stations). Trends in *C. delitrix* area were calculated from a mixed model regression. Slope direction for each region*habitat grouping is reported in Table 8.

Table 8. Long-term trends of mean *Cliona delitrix* area (cm^2/m^2) from 2001-2008. Trends for 12 region*habitat groupings in the Florida Keys and Dry Tortugas were determined with a mixed model regression. Interpretation of trends for each region*habitat grouping based on Bonferroni corrected p values for repeated testing (adjusted p≤0.004).

Reg*Hab	P-value	Trend
LK BCP	0.013	no change
DT OD	0.008	no change
LK OD	0.006	no change
MK OD	<0.001	Decreasing
UK OD	<0.001	Decreasing
LK OS	0.338	no change
MK OS	0.591	no change
UK OS	0.144	no change
DT P	0.917	no change
LK P	<0.001	Decreasing
MK P	0.422	no change
UK P	0.576	no Change

Table 9. Long-term trends of mean *Cliona delitrix* area (cm^2/m^2) by site (*N*=37) in the Florida Keys and Dry Tortugas from 2001 to 2008. Trends for each site were determined with a mixed model regression. Interpretation of trends for each site based on Bonferroni corrected p values for repeated testing (adjusted p<0.002).

Trend	Number of Sites
Decreasing	5
No Change	32
Increasing	0

The trends for mean *Cliona delitrix* area mirror those for overall sponge cover in the Florida Keys and Dry Tortugas. A majority of the region*habitat groupings show mean *C. delitrix* area remaining similar while a few indicate a declining trend in cover. Previous reports in the literature have suggested that clionaid cover was increasing in the Florida Keys due to nutrient enrichment of the nearshore waters (Ward-Paige et al. 2005). While CREMP data confirms that the greatest abundance and cover of clionaids exist on reefs in closest proximity to highly urbanized cities like Key West (e.g. Lower Keys patch reefs), CREMP results do not support that clionaid cover is increasing in the Florida Keys. The hypothesis that clionaid cover increases with deteriorating water quality has been used elsewhere because sponges are filter feeders and can store excess nutrients in their tissues (Ward-Paige et al. 2005; Chaves-Fonnegra et al. 2007). While it can only be speculated at this point, CREMP findings, using *C. delitrix* as a proxy, may suggest the level of nutrients entering nearshore waters has been reduced as a result of advanced wastewater and sewage treatments facilities built in the Florida Keys during the last two decades. Alternatively, the decrease of clionaid cover at some locations may be related to the decline of massive star and boulder corals, such as *M. cavernosa* and the *M. annularis* complex. The region*habitat groupings that contain the highest *C. delitrix* area also possess many of the highest coral cover values for these species (see Sentinel coral species section). *Montastraea* spp. are frequently targeted by *C. delitrix*, which slowly erodes their skeletons. It is probably no coincidence that sites that have experienced significant declines in *C. delitrix* area (e.g. Bird Key Reef, Carysfort Deep, Cliff Green, Smith Shoal) have also had dramatic reductions *M. annularis* complex cover (see Appendices). Although it is difficult to identify specific drivers of the observed trends in *C. delitrix* cover, it is most likely some combination of changing water quality and decreasing coral cover.

CONCLUSIONS

Throughout the Caribbean, coral reefs have been in a state of decline for the last 40 years with reports of many reefs losing more than half of their stony coral cover (Gardner et al. 2003). Starting in the late 1970s and continuing through the 1980s, the loss of stony corals in the Florida Keys was due in large part to major stressors such as the white band disease outbreak on acroporid corals and periodic episodes of bleaching (Causey 2001; Precht and Miller 2006). Since the project's inception in 1996, CREMP has continued to document how these acute events have been correlated with negative impacts to Florida's coral reefs. Without question, the greatest regional declines observed by CREMP were associated with the 1997/1998 ENSO and the 2005 Hurricane season. However, more localized events have contributed to reduced coral cover at a much smaller spatial scale. For example, a diatom bloom decimated stony coral cover on backcountry patch reefs in 2001, and a coral disease outbreak between 2001 and 2003 at Bird Key Reef in the Dry Tortugas resulted in substantial mortality to *Montastraea annularis* complex and *Colpophyllia natans* at that site. Although some of these acute stressors are naturally occurring phenomena (e.g. hurricanes), the long history of anthropogenic activities in South Florida has likely played a role in coral decline. Even during periods that lack major disturbances, coral cover has remained similar or continued to decline with only a few examples of recovery. Chronic stressors related to the development and extraction of South Florida's natural resources (e.g. overfishing, sedimentation, nutrient enrichment) may be preventing widespread stony coral recovery and may contribute to the ongoing decline in cover, yet at a less pronounced rate when compared to that of major acute disturbances.

This report summarizes how coral reef community structure has changed at CREMP sites over the last 13 years. Assuming that current environmental conditions surrounding the Florida Keys remain similar and major periodic disturbances continue, the synthesis provided here provides insight into how reefs in Florida Keys may be transformed in the years ahead. In contrast to many areas in the Caribbean, CREMP has not observed a shift from stony coral to macroalgal dominance (Hughes 1994; Rogers and Miller 2006). Although previous research has led to the conclusion that coral reefs in the Florida Keys were undergoing a shift to macroalgal or sponge dominated states (Ward-Paige et al. 2005; Maliao et al. 2008), the results from 13 years of data indicate that octocorals are becoming the most abundant taxa. This transition is most apparent at shallow forereef habitats and succeeds the demise of *Acropora palmata*. This trend has also begun at sites where large *Montastraea* spp. corals have declined (e.g. patch reefs and deep forereefs) and suggests the transition may be widespread. Identifying the reasons behind the transition to octocorals and why they have recovered faster than other benthic fauna after major disturbances is difficult to determine. Twice during the tenure of CREMP, octocoral cover has declined and rebounded. Although there is little information regarding the status and trends of octocoral cover prior to the inception of CREMP, octocorals were reported to exceed stony coral species richness and colony density at some localities in the Florida Keys prior to 1996 (Wheaton and Jaap 1988; Chiappone and Sullivan 1997). It is possible that the shift toward octocorals commenced many decades ago in the Florida Keys starting with the decline in acroporid corals in the 1970's and 1980's, and CREMP is simply documenting the manifestation of this gradually developing transition. It is also uncertain what consequences this transition will bring to Florida Keys coral reefs. Octocorals are direct spatial competitors with stony corals and may limit their recovery by reducing the settlement of coral planulae (Maida et al. 1995; Maida et al. 2001). Alternatively, it is feasible that octocoral dominated benthic habitats will continue to support diverse and abundant fish and invertebrate assemblages, albeit unconventional, as they provide critical habitat and serve as an important food source for a variety of predators (Wolff et al. 1999; Gratwicke et al. 2006).

While the general consensus is that the future of coral reefs is one of doom and gloom there are some reasons for optimism in the Florida Keys. It is important to understand that changes in stony coral communities take place over long temporal scales. The current state of reefs in the Florida Keys is the culmination of at least 100 years of intense fishing pressure and 50 years of physical changes to the coastal environment. Conversely, actions taken to remediate environmental perturbations such as fishing regulations (e.g. catch limits, closures, marine reserves, etc.) and improving water quality (e.g. advanced wastewater treatment in the Florida Keys and the restoration of the Florida Everglades) have been enacted only recently. Even though global climate change continues to threaten coral reefs, the alleviation of local stressors may temper impacts from severe acute stressors (e.g. mass bleaching events) and aid in the recovery following the disturbance. Since 1999, a few sites have shown a positive trend in coral cover (e.g. Jaap Reef and Molasses Shallow) indicating that some recovery may be occurring at isolated locations. Although the increases in stony coral cover at these sites are minor, corrective actions to improve water quality and restore fish populations may prove beneficial over longer temporal scales. While results like these are encouraging, continued monitoring by CREMP will provide the necessary data to evaluate the efficacy of these management actions and if they are reversing the long history of decline of scleractinian corals in the Florida Keys.

LITERATURE CITED

Aronson A, Precht W, Toscano M, and Koltes K. 2002. The 1998 bleaching event and its aftermath on a coral reef in Belize. Marine Biology 141(3):435-447.

Aronson RB, Macintyre, I. G., Wapnick, C. M., O'Neill, M. W. 2005. Phase Shifts, Alternative States, and the Unprecedented Convergence of Two Reef Systems. Ecology [Ecology] 85(7):1876-1891.

Aronson RB, and Precht WF. 2001. White-band disease and the changing face of Caribbean coral reefs. Hydrobiologia 460(1-3):25-38.

Aronson RB, Precht, W. F. 2000. Herbivory and algal dynamics on the coral reef at Discovery Bay, Jamaica. Limnology and Oceanography 45(1):251-255.

Ault JS, Bohnsack JA, and Meester G. 1998. A retrospective (1979-1996) multispecies assessment of coral reef fish stocks in the Florida Keys Fishery Bulletin 96(3):19.

Ault JSB, J.A., Smith SG, and Luo J. 2005. Towards Sustainable Multispecies Fisheries in the Florida, USA, Coral Reef Ecosystem Bulletin of Marine Science 76(2).

Bak RPM, and Meesters EH. 1999. Population Structure as a Response of Coral Communities to Global Change. American Zoologist: From the Symposium Coral Reefs and Environmental Changes?Adaptation, Acclimation, or Extinction presented at the annual Meeting of the Society for Comparative and Integrative Biology, January 3-7, 1998, at Boston, Massachusetts 39(1):56-65.

Baker AC, Glynn PW, and Riegl B. 2008. Climate change and coral reef bleaching: An ecological assessment of long-term impacts, recovery trends and future outlook. Estuarine, Coastal and Shelf Science 80(4):435-471.

Bohnsack JA, and Smith WFG. 1994. The impact of fishing on coral reefs. Global aspects of coral reefs: Health, hazards and history pp C8 C12-Proceedings of the colloquium on global aspects of coral reefs, Miami, 1993 76(2):196-200.

Brown BE. 1997. Coral bleaching: causes and consequences. Coral Reef 16(1):129-138.

Bruno JF, Aronson RB, Precht WF, Glynn PW, Harvell CD, Kaufman L, Rogers CS, Shinn EA, and Valentine JF. 2004. Causes of Coral Reef Degradation Science 302(5650):2.

Bruno JF, Petes LE, Harvell DC, and Hettinger A. 2003. Nutrient enrichment can increase the severity of coral diseases. Ecology Letters 6(12):1056-1061.

Bruno JF, Selig ER, Casey KS, Page CA, Willis BL, Harvell CD, Sweatman H, and Melendy AM. 2007. Thermal stress and coral cover as drivers of coral disease outbreaks. PLoS Biol 5(6):1-8.

Butler IV MJ. 1995. Cascading disturbances in Florida Bay, USA: cyanobacteria blooms, sponge mortality, and implications for juvenile spiny lobsters Panulirus argus. Marine Ecology Progress Series 129:119-125.

Causey B. 2001. Lessons learned from the intensification of coral bleaching from 1980–2000 in the Florida Keys, USA. Salm RV, Coles SL (eds) Coral bleaching and marine protected areas: Proc Workshop 'Mitigating coral bleaching impact through MPA design' (Bishop Museum, Honolulu, HI) Asia Pacific Coastal Marine Program Report 0102.

Cerrano C, Bavestrello G, Bianchi CN, Cattaneo-vietti R, Bava S, Morganti C, Morri C, Picco P, Sara G, Schiaparelli S et al. . 2000. A catastrophic mass-mortality episode of gorgonians and other organisms in the Ligurian Sea (North-western Mediterranean), summer 1999. Ecology Letters 3(4):284-293.

Chaves-Fonnegra A, Zea S, and Gomez ML. 2007. Abundance of the excavating sponge *Cliona delitrix* in relation to sewage discharge at San Andres Island, sw Caribbean, Colombia. Bol Invest Mar Cost 36:63-78.

Chiappone M, and Sullivan JR. 1997. Rapid assessment of reefs in the Florida Keys: Results from a synoptic survey. Proceedings from the 8th International Coral Reef Sypmosium 2:1509-1514.

Colella M, Richardson B, Johnson D, Ruzicka RR, Callahan MK, Bertin M, Wheaton JW, Schmitt S, and Sheridan N. 2008. Assessment of Population and Community Structure of Sessile Macro Invertebrates following a Benthic Mortality Event in the Eastern Gulf of Mexico. Fish and Wildlife Research Institute. St. Petersburg, FL. p 49.

Cowart JD, Henkel TP, McMurray SE, and Pawlik JR. 2006. Sponge orange band (SOB): a pathogenic-like condition of the giant barrel sponge, Xestospongia muta. Coral Reef 24(4):513-514.

Curran HA, Smith DP, Meigs LC, Pufall AE, and Greer ML. 1994. The health and short-term change of two coral patch reefs, Fernandez Bay, San Salvador Island, Bahamas. In: Ginsburg RN (ed) Proc Colloquium on Global Aspects of Coral Reefs: Health, Hazards, and History, 1993 Rosenstiel School of Marine and Atmospheric Science, University of Miami, FL,. p 147–153.

Edmunds PJ. 2007. Evidence for a decadal-scale decline in the growth rates of juvenile scleractinian corals. Marine Ecology Progress Series 341:1-13.

Endean R, Cameron AM, and DeVantier LM. 1988. Acanthaster planci predation on massive corals: the myth of rapid recovery of devasted reefs. Proceedings of the 6th International Coral Reef Symposium, Australia 2:143-148.

Fitt WK, Barbara BE, Warner ME, and Dunne RP. 2001. Coral bleaching: interpretation of thermal tolerance limits and thermal thresholds in tropical corals. Coral Reefs 20(1):51-65.

Fong P, and Lirman D. 1995. Hurricanes Cause Population Expansion of the Branching Coral Acropora palmata (Scleractinia): Wound Healing and Growth Patterns of Asexual Recruits. Marine Ecology 16(4):317-335.

Fong P, and Lirman D. 1997. Patterns of Damage to the Branching Coral Acropora palmata Following Hurricane Andrew: Damage and Survivorship of Hurricane-Generated Asexual Recruits. Journal of Coastal Research 13(1):67-72.

Fourqurean JW, and Robblee MB. 1999. Florida Bay: A history of recent ecological changes. Estuaries and Coasts 22(2B):345-357.

Fox HE, Pet JS, Dahuri R, and Caldwell RL. 2003. Recovery in rubble fields: long-term impacts of blast fishing. Marine Pollution Bulletin 46:1024-1031.

FWRI and DTNP. 2010. Long Term Monitoring & Assessment of Selected Coral Reef Sites In the Dry Tortugas National Park. 2006 Annual Report. Fish and Wildlife Research Institute. St. Petersburg, FL. p 35.

Gardner TA, Cote IM, Gill JA, Grant A, and Watkinson AR. 2003. Long-tern region-wide declines in Caribbean corals. Science 301:958-960.

Gates RD, Baghdasarian G, and Muscatine L. 1990. Temperature stress causes host cell detachment in symbiotic cnidarians: Implications for coral bleaching. Biological Bulletin, Marine Biological Laboratory, Woods Hole 182(3):324-332.

Geiser DM, Taylor JW, Ritchie KB, and Smith GW. 1998. Cause of sea fan death in the West Indies. Nature 394:137-138.

Gilliam DS. 2009. 2008 Year 6 Final Report. Southeast Florida Coral Reef Evaluation and Monitoring Project Florida DEP report #G0099:39.

Gilmore MD, and Hall BR. 1976. Life history, growth habits, and constructional roles of Acropora cervicornis in the patch reef environment. Journal of Sedimentary Research 46(3):519-522.

Glazer R, and Quintero I. 1998. Observations on the sensitivity of queen conch to water quality: Implications for coastal development. Proceedings of the Gulf and Caribbean Fisheries Institute(50):78-93.

Gleason DF. 1998. Sedimentation and distributions of green and brown morphs of the Caribbean coral Porites astreoides. Journal of Experimental Marine Biology and Ecology 230(1):73-89

Glynn PW, and Colgan MW. 1993. Sporadic disturbances in fluctuating coral reef environments: El Nino and coral reef development in the eastern Pacific. American Zoologist 32(6):707-718.

Gratwicke B, Petrovic C, and Speight MR. 2006. Fish distribution and ontogenetic habitat preferences in non-estuarine lagoons and adjacent reefs. Environmental Biology of Fishes 76:191-210.

Green DH, Edmunds PJ, and Carpenter RC. 2008. Increasing relative abundance of Porites astreoides on Caribbean reefs mediated by an overall decline in coral cover. Marine Ecology Progress Series 359:1-10.

Griffin DW, Donaldson KA, Paul JH, and Rose JB. 2003. Pathogenic human viruses in coastal waters. Clinical Microbiology Reviews 16(1):129-143.

Harvell CD, Burkholder JM, Colwell RR, Epstein PR, Grimes DJ, Hofmann EE, Lipp EK, Osterhaus ADME, Overstreet RM, Porter JW et al. . 1999. Emerging Marine Disease-Climate Links and Anthropogenic Factors. Science 285(3):1505-1510.

Harvell DC, Mitchell CE, Ward JR, Altizer S, Dobson AP, Ostfeld RS, and Samuel MD. 2002. Climate Warming and Disease Risks for Terrestrial and Marine Biota. Science 296(5576):2158-2162.

HealthyReefs.Org. 2010. Report card for the Mesoamerican reef. An evaluation of ecosystem health. Healthy Reefs for Healthy People. p 44.

Hoegh-Guldberg O. 1999. Climate change, coral bleaching and the future of the world's coral reefs. Marine & Freshwater Research 50(8):839-866.

Hu C, Hackett KE, Callahan MK, Andrefouet S, Wheaton JL, Porter JW, and Muller-Karger FE. 2003. The 2002 ocean color anomaly in the Florida Bight: A cause of local coral reef decline? Geophysical Research Letters 30(3):1151-1154.

Hughes TP. 1994. Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. Science 265(5178):1547-1551.

Hughes TP, and Tanner JE. 2000. Recruitment failure, life histories, and long-term decline of Caribbean corals. Ecology 81(8):2250-2263.

Johns GM, Leeworthy VR, Bell FW, and Bonn MA. 2001. Socioeconomic Study of Reefs in Southeast Florida. Hazen and Sawyer Environmental Engineers & Scientists. p 348.

Keller B, and Donahue S. 2006. 2002 - 03 Florida Keys National Marine Sanctuary Science Report: An Ecosystem Report Card After Five Years of Marine Zoning. Marine Sanctuaries Conservation Series NMSP-06-12 Marine Sanctuaries Conservation Series NMSP-06-12 US

Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Sanctuary Program, Silver Spring, MD. p 358.

Knowlton N. 2001. The future of coral reefs. Proceedings from the National Academy of Science USA 98:5419-5425.

Kruczynski B. 1999. Water quality concerns in the Florida Keys: Sources, Effects, and Solutions. Florida Keys National Marine Sanctuary Water Quality Protection Program p68.

Lapointe BE. 1990. Caribbean coral reefs: Are they becoming algal reefs? Sea Frontiers(March-April):83-91.

Lapointe BE. 1997. Nutrient thresholds for bottom-up control of macroalgal blooms on coral reefs in Jamaica and southeast Florida. Limnology and Oceanography 45:1119-1131

Lapointe BE, Barile PJ, and Matzie WR. 2004. Anthropogenic nutrient enrichment of seagrass and coral reef communities in the Lower Florida Keys: discrimination of local versus regional nitrogen cycles. Journal of Experimental Marine Biology and Ecology 308:23-58.

Lapointe BE, and Clark MW. 1992. Nutrient Inputs from the Watershed and Coastal Eutrophication in the Florida Keys. Estuaries 15(4):465-476.

Lapointe BE, and Matzie WR. 1996. Effects of stormwater nutrient discharges on eutrophication processes in nearshore waters of the Florida keys. Estuaries and coasts 19(2):422-435.

Lapointe BE, Tomasko DA, and Matzie WR. 1994. Eutrophication and the trophic state classification of seagrass communities in the Florida Keys. Bulletin of Marine Science 54(3):21.

Leichter JJ, Stewart HL, and Miller SL. 2003. Episodic Nutrient Transport to Florida Coral Reefs. Limnology and Oceanography 48(4):1394-1407.

Lesser MP, Bythell JC, Gates RD, Johnstone RW, and Hoegh-Guldberg O. 2007. Are infectious diseases really killing corals? Alternative interpretations of the experimental and ecological data. Journal of Experimental Marine Biology and Ecology 346:36-44.

Lessios HA, Cubit JD, Robertson DR, Shulman MJ, Parker MR, Garrity SD, and Levings SC. 1984. Mass mortality of Diadema antillarum on the Caribbean coast of Panama. Coral reefs Heidelberg etc 3(4):173-182.

Light SS, and Dineen WJ. 1994. Water Control in the Everglades: A historical perspective. In: Everglades: the ecosystem and its restoration 47-84.

Lipp EK, Jarrell JL, Griffin DW, Lukasik J, Jacukiewicz J, and Rose JB. 2002. Preliminary evidence for human fecal contamination in corals of the Florida Keys, USA. Marine Pollution Bulletin 44(7):666-670.

Lirman D. 2000. Fragmentation in the branching coral Acropora palmata (Lamarck): growth, survivorship, and reproduction of colonies and fragments. Journal of Experimental Marine Biology and Ecology 251(1):41-57.

Lirman D, and Biber P. 2000. Seasonal Dynamics of Macroalgal Communities of the Northern Florida Reef Tract. Botanica marina 43(4):305-314.

Lopez-Legentil S, Song B, McMurray SE, and Pawlik JR. 2008. Bleaching and stress in coral reef ecosystems: hsp70 expression by the giant barrel sponge Xestospongia muta. Molecular Ecology 17(7):1840-1849.

Maida M, Sammarco PW, and Coll JC. 1995. Effects of soft corals on scleractinian coral recruitment. I: Directional allelopathy and inhibition of settlement. Marine Ecology Progress Series 121:191-202.

Maida M, Sammarco PW, and Coll JC. 2001. Effects of soft corals on Scleractinian coral recruitment. 2: Allelopathy, spat survivorship and reef community structure. Marine Ecology 22(4):397-414.

Maliao RJ, Turingan RG, and Lin J. 2008. Phase-shift in coral reef communities in the Florida Keys National Marine Sanctuary (FKNMS), USA. Marine biology 154(5):841-853.

Manzello DP, Brandt M, Smith TB, Lirman D, Hendee JC, and Nemeth RS. 2007. Hurricanes benefit bleached corals. Proceedings of the National Academy of Sciences 104(29):12035-12039.

McClenachan L. 2008. Documenting Loss of Large Trophy Fish from the Florida Keys with Historical Photographs. Conservation Biology 23(3).

McCook LJ, Wolanski, E., Spagnol, S. 2001. Modelling and visualizing interactions between natural disturbances and eutrophication as causes of coral reef degradation. Monographs Aquatic Ecology and Autecology 113-125.

McGuire MP. 1998. Timing of larval release by *Porites astreiodes* in the northern Florida Keys. Coral Reefs 17:369-375.

McIvor CC. 1994. Food Habits of mangrove fishes: A comparison along estuarine gradients in Northeasetern Florida Bay. Bulletin of Marine Science 54(3):881-899.

Miller J, Muller E, Rogers C, Waara R, Atkinson A, Whelan KRT, Patterson M, and Witcher B. 2009. Coral disease following massive bleaching in 2005 causes 60% decline in coral cover on reefs in the US Virgin Islands. Coral Reefs 28(4):925-937.

Miller MW, Bourque AS, and Bohnsack JA. 2002. An analysis of the loss of acroporid corals at Looe Key, Florida, USA: 1983-2000. Coral Reefs 21(2):179-182.

Miller MW, Weil E, and Szmant AM. 2000. Coral recruitment and juvenile mortality as structuring factors for reef benthic communities in Biscayne National Park, USA. Coral Reefs 19:115-123.

Mumby PJ, Harborne AR, Williams J, Kappel CV, Brumbaugh DR, Micheli F, Holmes KE, Dahlgren CP, Paris CB, and Blackwell PG. 2007. Trophic cascade facilitates coral recruitment in a marine reserve. Proceedings of the National Academy of Science 104:8362-8367.

Nagelkerken I, Buchan K, Smith GW, Bonair K, Bush P, Garzon-Ferreira, J. B, L., Gayle P, Harvell CD, Heberer C et al. . 1997. Widespread disease in Caribbean sea fans I. Spreading and general characteristics. Proc 8th Int Coral Reef Sym I:679-682.

Nagelkerken I, Buchan, K., Smith, G.W., Bonair, K., Bush, P., Garzon-Ferreira, J., Botero, L., Gayle, P., Harvell, C.D., Heberer, C., Kim, K., Petrovic, C., Pots, L., Yoshioka, P. 1997. Widespread disease in Caribbean sea fans II. Pattern of infection, and tissue loss. Mar Ecol Prog Ser 160:255-263.

Nagelkerken WP. 1974. Studies on the fauna of Curacao and other Caribbean Islands: No. 147. Onthe occurrence of fishes in relation to corals in Curacao. Uitg Natuurwet Stud Suriname Ned Antillen no 77:118-141.

National Hurricane Center. 2010. http://www.nhc.noaa.gov/.

Norstrom AV, Nystrom M, Lokrantz J, and Folke C. 2009. Alternative states on coral reefs: beyondcoral–macroalgal phase shifts. Marine Ecology Progress Series 363:295–306.

Patterson KL, Porter JW, Ritchie KB, Polson SW, Mueller E, Peters EC, Santavy DL, and Smith GW. 2002. The etiology of white pox, a lethal disease of the Caribbean elkhorn coral, *Acropora palmata.* Proceedings of the National Academy of Sciences, USA 99(13):8725-8730.

Paul JH, Rose JB, Jiang S, Kellogg, C., and Shinn EA. 1997. Occurrence of fecal indicator bacteria in surface waters and the subsurface aquifer in Key Largo, Florida. Applied and Environmental Microbiology 61(6):2235-2241.

Paul JH, Rose JB, Jianga SC, Zhoua X, Cochrana P, Kellogga C, Kanga JB, Griffina D, Farrah S, and Lukasik J. 2000. Evidence for groundwater and surface marine water contamination by waste disposal wells in the Florida Keys. Pergamon 31(6):1448-1454.

Porter JW, Harvell CD, Kim K, Burkholder JM, Colwell RR, Epstein PR, Grimes DJ, Hofmann EE, Lipp EK, Osterhaus ADME et al. . 1999. Emerging Marine Diseases--Climate Links and Anthropogenic Factors. Science 285(5433):1505-1510.

Precht W, Bruckner A, Aronson R, and Bruckner R. 2002. Endangered acroporid corals of the Caribbean. Coral Reefs 21(1):41-43.

Precht WF, and Aronson RB. 1997. Stasis, Biological Disturbance, and Community Structure of a Holocene Coral Reef,. Paleobiology 23(3):326-346.

Precht WF, and Miller SL. 2006. Ecological Shifts along the Florida Reef Tract: The Past as a Key to the Future. In: Geological Approaches to Coral Reef Ecology. Aronson RB, editor. Springer, NY. p 237-312.

Richardson LL. 1998. Coral diseases: what is really known? Trends in Ecology & Evolution 13(11):438-443.

Riegl BM, and Dodge RE. 2008. *Coral Reefs of the World 1.* Coral Reefs of the USA. 2008: Springer.

Roberts HH, Rouse LJ, Walker ND, and Hudson JH. 1982. Cold-water stress in Florida Bay and northern Bahamas; a product of winter cold-air outbreaks Journal of Sedimentary Research 52(1):10.

Robinson A. 1973. Natural vs. Visitor-related Damage to Shallow Water Corals: Recommendations for Visitor Management and the Design of Underwater Nature. NPS Symposium Series 1.

Rogers CS, and Miller J. 2006. Permanent 'phase shifts' or reversible declines in coral reef cover? Lack of recovery of two reefs in St. John, US Virgin Islands Marine Ecology Progress Series 306:103-114.

Rutzler K. 2002. Impact of crustose clionid sponges on Caribbean reef corals. Acta Geology Hispanica 37:61-72.

Santavy DL, and Peters EC. 1997. Microbial pests: coral disease in the Western Atlantic. Proc 8th Int Coral Reef Sym 1:607-612.

Schmahl GP. 1990. Community structure and ecology of sponges associated with four southern Florida coral reefs. in: K Rützler, ed New perspectives in sponge biology Smithson Inst Press:376–383.

Schutte VGW, Selig ER, and Bruno JF. 2010. Regional spatio-temporal trends in Caribbean coral reef benthic communities. Marine Ecology Progress Series 402:115-122.

Shinn EA. 1976. Coral reef recovery in Florida and the Persian Gulf. Environmental Geology 1(4):241-254.

Smith GW, Harvell CD, and Kim K. 1996. Observations on the pathogenesis of sea fans infected with *Aspergillus* sp. Revista de biologia tropical [Rev Biol Trop] 46:205-208.

Somerfield PJ, Jaap WC, Clarke KR, Callahan M, Hackett K, Porter J, Lybolt M, Tsokos C, and Yanev G. 2008. Changes in coral reef communities among the Florida Keys, 1996–2003. Coral Reefs 27(4):951-965.

Stobart B, Teleki K, Buckley R, Downing N, and Callow M. 2005. Coral recovery at Aldabra Atoll, Seychelles:five years after the 1998 bleaching event. Philosophical Transaction of the Royal Society A 363:251-255.

Sutherland KP, and Ritchie KM. 2004. White Pox Disease of the Carribean Elkhorn coral, *Acropora palmata*. In: Coral Health and Disease Springer-Verlag (Berlin) E Rosenberg and Y Loya (Eds) Chapter: 16:289-297.

Szmant AM, and Forrester A. 1996. Water column and sediment nitrogen and phosphorus distribution patterns in the Florida Keys, USA. Coral Reef 15(1):21-41.

Tilmant JT, Rutherford ES, and Thue EB. 1989. Fishery harvest and population dynamics of red drum (*Sciaenops oscellatus*) from Florida Bay and adjacent waters. Bulletin of Marine Science 44(1):126-138.

U.S. Census. 2010. United States Census Bureau. http://wwwcensusgov/.

Vicente VP. 1990. Response of sponges with autotrophic endosymbionts during the coral-bleaching episode in Puerto Rico (West Indies). Coral Reefs 8(4):199-202.

Wagner DE, Kramer P, and Woesik RV. 2010. Species composition, habitat, and water quality influence coral bleaching in southern Florida Marine Ecology Progress Series 408:65-78.

Ward-Paige CA, Risk MJ, Sherwood OA, and Jaap WC. 2005. Clionid sponge surveys on the Florida Reef Tract suggest land-based nutrient inputs. Marine Pollution Bulletion 51(5-7):570-579.

Weil E, Hernandez-Delgado EA, Bruckner AW, Ortiz AL, and Nemeth MRH. 2002. Distribution and status of Acroporid coral (Scleractinia) populations in Puerto Rico. In: Bruckner, AW (ed) NOAA Tech Memo NMFS-OPR-24(Silver Spring, MD):71-98.

Wheaton JL, and Jaap WC. 1988. Corals and other prominent benthic Cnidaria of Looe Key National Marine Sanctuary, Florida. Florida Marine Research Publications 43:1-25.

Williams EHJ, Bartels PJ, and Bunkley-Williams L. 1999. Predicted disappearance of coral-reef ramparts: a direct result of major ecological disturbances. global Change Biology 5(8):839-844.

Wolff N, Grober-Dunsmore R, Rogers CS, and Beets J. 1999. Management implications of fish trap effectiveness in adjacent coral reef and gorgonian habitats Environmental Biology of Fishes 55:81-90.

Yoshioka PM. 1994. Size-specific life history pattern of a shallow-water gorgonian. Journal of Experimental Marine Biology and Ecology 184(1):111-122.

Yoshioka PM, and Yoshioka BB. 1991. A comparison of the survivorship and growth of shallow-water gorgonian species of Puerto Rico. Marine Ecology Progress Series 69:253-260.

APPENDICES

Appendix 1. Percent cover values for major benthic taxa categories for all years at all region*habitat groupings.

Region	Habitat	Sample Year	Macroalgae	Stony Corals	Octocorals	Sponges	Zoanthids
DT	**OD**	1999	9.1	23.6	14.8	4.0	0.1
		2000	12.6	22.1	13.0	1.4	0.0
		2001	12.5	23.3	15.2	2.1	0.0
		2002	12.7	18.3	14.1	2.2	0.0
		2003	12.5	18.0	14.2	1.8	0.1
		2004	9.2	16.6	12.2	1.6	0.0
		2005	3.8	16.3	10.5	1.3	0.0
		2006	9.6	12.6	9.0	1.1	0.0
		2007	13.6	14.0	9.3	1.1	0.0
		2008	12.0	14.5	8.6	1.7	0.0
DT	**P**	1999	6.1	9.5	8.4	2.9	0.0
		2000	8.9	8.0	7.6	4.0	0.0
		2001	11.4	5.8	8.5	1.9	0.0
		2002	10.3	5.6	6.9	1.7	0.0
		2003	14.9	4.2	8.7	1.6	0.0
		2004	6.7	1.9	4.1	0.4	0.0
		2005	9.6	1.6	3.4	0.3	0.0
		2006	14.8	1.1	5.1	0.6	0.0
		2007	17.2	1.2	6.1	0.9	0.0
		2008	13.6	1.8	8.8	0.8	0.0
LK	**BCP**	1996	19.3	8.3	0.6	2.6	0.0
		1997	24.2	7.8	0.4	3.7	0.0
		1998	2.7	8.4	0.1	1.4	0.0
		1999	1.0	9.4	0.2	2.3	0.1
		2000	3.8	10.7	0.2	2.7	0.0
		2001	11.6	10.0	0.0	3.3	0.0
		2002	12.6	2.7	0.0	0.6	0.0
		2003	13.2	2.3	0.0	1.1	0.0
		2004	19.7	2.9	0.0	0.9	0.0
		2005	28.3	3.9	0.0	1.5	0.0
		2006	2.9	2.6	0.0	2.7	0.0
		2007	27.2	2.2	0.1	0.8	0.0
		2008	72.6	1.8	0.0	0.7	0.0
LK	**OD**	1996	15.5	7.6	10.9	3.7	0.3
		1997	19.8	7.9	9.7	3.4	0.3
		1998	43.0	5.3	11.6	2.5	0.1
		1999	15.6	4.2	8.5	1.8	0.2
		2000	27.7	4.3	8.4	2.5	0.2
		2001	23.8	4.4	7.5	2.5	0.1
		2002	15.5	4.2	8.7	4.0	0.2
		2003	11.0	3.9	9.6	2.3	0.1
		2004	20.4	3.1	11.3	2.1	0.2
		2005	4.5	3.2	10.6	2.5	0.2
		2006	14.2	1.9	8.8	1.0	0.1

		2007	19.8	2.3	8.3	1.6	0.2
		2008	12.2	2.4	9.9	3.1	0.2
LK	**OS**	1996	4.6	16.9	3.7	0.9	4.5
		1997	3.8	16.4	4.0	0.8	4.8
		1998	7.6	13.2	3.9	0.3	4.5
		1999	3.5	7.3	3.2	0.7	4.2
		2000	4.3	7.4	2.8	0.7	4.4
		2001	4.1	6.9	3.4	0.4	4.5
		2002	7.2	7.4	3.8	0.4	5.3
		2003	7.1	7.3	5.0	0.3	5.1
		2004	8.9	7.0	6.7	0.5	4.4
		2005	2.2	5.5	5.1	0.3	3.7
		2006	11.9	4.9	4.9	0.2	3.8
		2007	7.3	4.9	5.4	0.2	4.0
		2008	2.9	4.7	7.1	0.2	3.9
LK	**P**	1996	6.2	27.6	8.4	4.0	0.0
		1997	5.0	25.4	9.8	2.9	0.0
		1998	8.2	24.5	8.7	3.8	0.1
		1999	6.5	19.7	7.8	3.5	0.0
		2000	3.9	20.1	7.7	5.0	0.1
		2001	11.1	19.4	9.6	4.4	0.0
		2002	8.1	21.4	10.3	4.6	0.0
		2003	8.6	21.5	11.9	4.8	0.0
		2004	10.5	19.2	8.6	4.8	0.0
		2005	4.3	21.1	11.3	2.8	0.0
		2006	8.7	20.9	8.2	2.1	0.1
		2007	10.2	20.0	9.6	4.6	0.1
		2008	10.8	19.3	11.2	4.8	0.0
MK	**OD**	1996	4.0	4.1	12.5	4.9	0.5
		1997	6.2	4.8	10.7	4.9	0.1
		1998	41.7	3.4	10.3	3.2	0.0
		1999	15.3	3.0	5.3	2.2	0.0
		2000	20.3	2.8	4.9	3.4	0.0
		2001	16.6	3.0	6.1	4.6	0.0
		2002	4.1	2.7	6.4	4.0	0.0
		2003	5.9	2.9	6.7	4.5	0.0
		2004	13.5	2.5	10.4	4.4	0.0
		2005	8.1	2.7	10.4	3.8	0.0
		2006	22.7	2.3	7.7	2.3	0.0
		2007	13.3	2.0	8.4	2.7	0.0
		2008	16.2	2.2	8.7	3.6	0.0
MK	**OS**	1996	11.5	4.6	16.0	3.7	6.9
		1997	6.5	4.2	14.8	2.1	7.5
		1998	20.7	3.7	13.8	1.6	6.4
		1999	11.4	2.0	8.5	2.2	6.6
		2000	11.1	2.0	8.3	2.5	7.3
		2001	12.4	1.9	8.4	1.7	7.0
		2002	5.5	2.1	12.2	1.7	8.2
		2003	18.2	1.8	14.7	1.7	6.5
		2004	12.0	1.7	15.9	1.4	6.7

		2005	10.6	1.8	16.3	1.6	6.0
		2006	11.9	1.4	13.7	0.8	5.5
		2007	9.6	1.2	13.8	1.2	5.2
		2008	8.2	1.7	18.1	1.3	4.9
MK	**P**	1996	4.6	15.8	32.5	5.5	2.0
		1997	4.6	15.6	31.2	5.4	2.3
		1998	11.4	14.4	27.0	5.5	1.8
		1999	5.9	13.2	26.2	5.5	2.2
		2000	5.3	14.2	24.4	6.3	2.3
		2001	6.4	14.1	32.4	6.2	1.8
		2002	3.8	16.3	33.1	7.1	2.2
		2003	3.3	14.7	34.4	5.2	2.2
		2004	6.2	13.6	36.1	3.8	1.6
		2005	4.2	15.1	34.0	3.9	1.8
		2006	3.1	14.2	28.7	3.3	2.0
		2007	5.9	15.3	29.5	5.2	2.4
		2008	4.4	14.3	32.5	4.9	1.7
UK	**OD**	1996	13.9	7.4	12.6	6.2	0.0
		1997	16.6	6.1	11.2	5.4	0.2
		1998	26.4	3.5	10.4	4.2	0.0
		1999	7.9	3.4	8.3	4.2	0.2
		2000	20.4	3.6	9.5	4.4	0.0
		2001	15.0	2.9	9.7	3.9	0.0
		2002	11.7	3.0	12.3	5.1	0.1
		2003	15.5	3.1	12.1	4.5	0.0
		2004	19.9	2.7	14.4	4.2	0.1
		2005	8.2	2.7	14.1	4.1	0.1
		2006	11.1	2.3	14.4	3.2	0.0
		2007	10.2	3.2	13.3	4.2	0.1
		2008	12.7	3.0	11.8	4.7	0.0
UK	**OS**	1996	8.8	12.1	9.5	1.7	2.3
		1997	5.6	11.2	7.8	1.2	3.2
		1998	11.9	8.3	8.7	0.8	2.7
		1999	4.8	5.9	7.3	0.5	2.6
		2000	6.3	5.9	7.4	1.0	2.9
		2001	7.2	5.6	9.1	0.8	2.5
		2002	7.9	5.6	9.9	0.6	3.2
		2003	12.6	5.2	13.1	0.7	3.1
		2004	11.0	4.9	14.0	0.4	2.6
		2005	8.3	4.8	15.9	0.7	2.7
		2006	3.1	4.1	15.6	0.4	2.7
		2007	5.7	4.5	15.8	0.5	2.8
		2008	7.9	4.8	16.1	0.5	3.4
UK	**P**	1996	7.4	16.3	24.0	3.9	0.3
		1997	5.6	15.2	23.4	1.7	0.4
		1998	17.9	14.0	22.8	1.4	0.3
		1999	8.1	11.9	22.2	1.4	0.4
		2000	7.5	11.3	20.8	1.8	0.4
		2001	5.7	12.2	23.9	1.9	0.4
		2002	12.2	12.2	24.4	1.9	0.4

	2003	8.6	12.6	25.6	1.7	0.3
	2004	7.9	12.5	26.1	1.4	0.3
	2005	7.5	11.8	26.9	1.5	0.4
	2006	6.2	11.0	22.1	1.4	0.3
	2007	9.7	13.2	22.2	1.2	0.3
	2008	8.8	12.7	23.7	1.1	0.3

Appendix 2. Percent cover values for major benthic taxa categories for all years at all CREMP sites.

Site Name and Location	Sample Year	Macroalgae	Stony Corals	Octocorals	Sponges	Zoanthids
Admiral	1996	3.4	30.0	25.7	0.2	0.1
25.0447 °N	1997	3.7	27.6	23.0	0.4	0.1
80.3948 °W	1998	7.4	26.6	26.8	0.3	0.1
	1999	5.0	22.5	23.8	0.4	0.1
	2000	7.2	20.0	23.3	0.9	0.1
	2001	2.7	21.2	25.0	0.4	0.1
	2002	6.6	21.6	24.4	0.4	0.2
	2003	6.6	23.1	28.0	0.6	0.2
	2004	3.5	23.1	27.7	0.5	0.1
	2005	6.1	21.7	30.3	0.5	0.1
	2006	2.7	19.9	23.7	0.6	0.1
	2007	8.9	24.6	23.0	0.5	0.1
	2008	5.6	23.3	26.9	0.3	0.1
Alligator Deep	1996	0.9	1.9	16.4	2.9	0.0
24.8452 °N	1997	11.3	2.3	14.6	3.3	0.1
80.6209 °W	1998	25.3	1.2	15.5	2.9	0.0
	1999	1.1	1.3	9.5	2.8	0.0
	2000	12.4	1.1	7.0	3.2	0.0
	2001	2.6	1.3	7.5	3.9	0.0
	2002	7.4	1.3	9.2	2.9	0.0
	2003	3.4	1.1	9.7	4.3	0.0
	2004	1.8	0.9	14.9	4.5	0.1
	2005	3.6	1.1	14.8	3.3	0.0
	2006	8.0	0.8	10.4	2.0	0.0
	2007	2.5	1.0	13.2	1.6	0.0
	2008	1.5	1.3	13.0	2.5	0.0
Alligator Shallow	1996	26.2	2.0	15.2	7.7	1.3
24.8461 °N	1997	14.4	2.1	14.0	0.9	0.6
80.6236 °W	1998	25.0	1.4	15.3	0.8	0.4
	1999	28.4	0.9	8.2	1.9	0.9
	2000	19.7	1.0	6.2	1.2	0.7
	2001	38.1	0.6	6.4	0.4	0.6
	2002	13.9	0.5	10.5	0.7	0.9
	2003	37.6	0.3	11.2	0.4	0.7
	2004	22.3	0.6	14.0	0.4	0.6
	2005	15.8	0.9	17.8	0.7	0.7
	2006	10.1	0.4	15.9	0.7	1.4
	2007	14.3	0.3	17.0	0.5	1.3
	2008	8.0	0.5	20.1	0.6	1.2
Bird Key Reef	1999	11.0	20.6	19.6	2.0	0.0
24.6117 °N	2000	14.0	21.5	17.1	0.7	0.0
82.8702 °W	2001	10.6	22.3	17.5	0.5	0.0
	2002	21.0	15.0	18.8	1.0	0.0

	2003	1.8	15.0	17.4	0.6	0.0
	2004	0.6	12.6	14.3	0.5	0.0
	2005	6.5	10.8	13.1	0.4	0.0
	2006	2.6	9.5	12.3	0.5	0.0
	2007	17.0	10.6	12.2	0.6	0.0
	2008	22.8	10.6	9.7	0.7	0.0
Black Coral Rock	1999	7.1	26.5	10.0	6.0	0.2
24.6993 °N	2000	11.3	22.6	9.0	2.1	0.0
83.0022 °W	2001	14.4	24.3	13.0	3.8	0.0
	2002	4.5	21.6	9.4	3.3	0.0
	2003	23.3	21.1	11.1	3.0	0.1
	2004	17.7	20.6	10.0	2.8	0.1
	2005	1.2	21.8	7.9	2.3	0.0
	2006	16.7	15.8	5.7	1.7	0.0
	2007	10.2	17.5	6.4	1.6	0.0
	2008	1.2	18.5	7.4	2.7	0.0
Carysfort Deep	1996	24.7	14.3	8.4	4.0	0.0
25.2208 °N	1997	31.0	11.0	10.2	2.1	0.3
80.2099 °W	1998	20.1	5.4	9.2	0.6	0.0
	1999	12.8	5.2	6.6	0.8	0.6
	2000	27.3	6.4	9.3	0.8	0.0
	2001	17.7	4.8	9.8	0.9	0.0
	2002	9.2	5.0	14.4	2.2	0.0
	2003	31.1	5.0	10.1	0.6	0.0
	2004	44.5	3.6	10.6	1.2	0.0
	2005	6.6	3.7	10.6	1.0	0.0
	2006	9.3	3.6	9.5	0.8	0.0
	2007	9.3	4.9	12.1	1.6	0.0
	2008	10.0	4.1	10.4	1.3	0.0
Carysfort Shallow	1996	6.0	10.9	8.3	0.4	2.7
25.2210 °N	1997	3.3	10.9	7.5	0.2	4.6
80.2103 °W	1998	4.5	7.5	8.0	0.2	4.0
	1999	1.1	3.6	6.0	0.0	2.9
	2000	2.9	3.5	6.7	0.2	3.1
	2001	2.8	3.7	9.0	0.1	3.2
	2002	2.4	4.1	8.9	0.1	4.9
	2003	1.3	3.3	11.3	0.0	5.3
	2004	1.1	3.1	10.1	0.0	4.1
	2005	0.7	3.5	9.4	0.1	4.2
	2006	0.9	2.5	9.5	0.1	4.1
	2007	1.1	2.5	9.1	0.1	3.2
	2008	3.1	2.9	9.7	0.1	5.2
Cliff Green	1996	0.7	19.6	22.4	7.8	0.0
24.5036 °N	1997	1.7	16.3	21.8	4.9	0.0
81.7677 °W	1998	1.7	15.6	25.1	4.1	0.1
	1999	0.4	16.4	22.3	5.9	0.0
	2000	1.0	14.9	26.1	4.7	0.0

	2001	1.0	14.8	25.8	6.0	0.1
	2002	0.1	16.8	33.6	5.9	0.1
	2003	2.3	15.0	34.2	5.0	0.1
	2004	2.8	11.3	24.6	5.0	0.1
	2005	0.2	13.7	29.7	2.8	0.0
	2006	0.7	12.4	22.0	2.9	0.3
	2007	1.2	13.3	23.3	7.4	0.3
	2008	0.8	12.3	30.8	6.8	0.0
Conch Deep	1996	6.4	5.0	9.4	5.8	0.0
24.9519 °N	1997	13.2	3.5	8.6	4.3	0.0
80.4513 °W	1998	33.3	3.2	8.3	6.4	0.0
	1999	10.0	2.6	7.8	3.4	0.0
	2000	30.0	1.7	6.2	3.4	0.0
	2001	22.1	2.4	6.9	4.2	0.0
	2002	19.7	2.4	9.9	5.0	0.0
	2003	13.5	2.1	9.8	5.2	0.0
	2004	10.9	2.7	13.4	5.3	0.0
	2005	15.0	2.6	10.8	4.6	0.0
	2006	8.9	1.8	13.8	3.0	0.0
	2007	11.2	2.9	13.2	4.9	0.0
	2008	16.2	3.3	11.6	5.6	0.0
Conch Shallow	1996	18.8	5.9	8.6	3.2	0.9
24.9559 °N	1997	8.3	5.9	5.7	2.9	1.2
80.4574 °W	1998	24.1	4.1	6.4	0.6	1.5
	1999	10.9	3.0	4.2	0.5	1.1
	2000	8.3	3.5	3.0	1.3	1.3
	2001	11.7	3.3	3.4	1.0	0.9
	2002	13.0	2.4	6.1	0.3	1.3
	2003	32.9	2.2	7.8	0.5	1.0
	2004	17.4	1.6	10.2	0.3	1.1
	2005	25.0	1.1	15.8	0.6	1.2
	2006	4.5	1.1	13.9	0.4	0.8
	2007	9.2	1.4	14.2	0.3	1.0
	2008	14.1	1.6	14.6	0.3	1.2
Content Keys	1996	31.3	1.2	0.0	1.9	0.0
24.8221 °N	1997	33.4	1.1	0.1	5.7	0.0
81.4889 °W	1998	0.8	1.4	0.0	1.7	0.0
	1999	1.1	1.4	0.0	2.6	0.0
	2000	7.3	1.2	0.0	2.5	0.0
	2001	20.7	2.3	0.0	3.5	0.0
	2002	0.3	0.5	0.0	0.2	0.0
	2003	2.6	0.6	0.0	0.6	0.0
	2004	31.3	0.4	0.0	0.5	0.0
	2005	55.6	0.6	0.0	0.8	0.0
	2006	5.1	0.3	0.0	3.3	0.0
	2007	35.2	0.4	0.2	0.4	0.0
	2008	84.6	0.2	0.0	1.0	0.0

Dustan Rocks	1996	8.3	17.0	34.8	3.9	1.2
24.6895 °N	1997	4.3	16.5	37.9	3.1	0.9
81.0302 °W	1998	22.3	15.5	26.0	1.9	0.7
	1999	9.3	14.8	28.1	2.9	0.8
	2000	9.6	16.2	27.4	3.0	0.8
	2001	11.9	14.6	38.7	3.3	0.7
	2002	7.6	17.1	40.7	3.2	0.7
	2003	7.0	16.1	36.7	2.3	0.7
	2004	12.6	15.2	37.7	1.5	0.8
	2005	7.0	16.1	37.4	1.8	0.9
	2006	4.0	16.9	28.2	1.3	0.7
	2007	7.1	18.0	32.0	2.5	0.7
	2008	9.4	17.2	34.8	2.5	0.6
Eastern Sambo Deep	1996	6.0	8.5	9.0	6.0	0.1
24.4884 °N	1997	11.5	8.0	9.7	4.4	0.1
81.6659 °W	1998	49.1	6.2	9.5	2.7	0.0
	1999	7.8	5.9	9.3	2.1	0.1
	2000	17.5	6.4	8.0	2.6	0.0
	2001	17.6	5.9	8.4	3.0	0.0
	2002	15.1	5.9	10.1	6.1	0.0
	2003	13.6	5.2	7.9	1.7	0.0
	2004	26.6	4.3	10.0	1.7	0.0
	2005	10.9	4.6	11.2	2.9	0.0
	2006	5.6	2.9	11.3	1.2	0.0
	2007	11.1	3.1	7.0	2.2	0.0
	2008	8.4	2.9	9.4	6.2	0.0
Eastern Sambo Shallow	1996	8.9	18.5	1.1	1.0	10.0
	1997	7.4	16.2	0.8	1.6	9.8
24.4917 °N	1998	4.1	12.1	0.9	0.9	8.6
81.6636 °W	1999	1.4	6.0	0.5	0.4	9.0
	2000	1.9	5.8	0.5	1.2	9.0
	2001	3.3	5.2	0.9	0.3	11.0
	2002	5.3	5.7	1.6	0.5	11.9
	2003	9.6	5.2	2.1	0.5	10.4
	2004	15.3	4.5	2.3	0.7	8.9
	2005	2.4	3.8	1.8	0.2	7.1
	2006	3.3	3.4	2.6	0.4	8.6
	2007	9.5	3.2	3.6	0.2	8.9
	2008	6.8	3.4	5.1	0.2	9.0
Grecian Rocks	1996	4.0	17.8	11.2	1.8	2.3
25.1075 °N	1997	7.4	16.3	9.7	1.0	2.7
80.3069 °W	1998	13.7	13.6	10.6	1.3	2.4
	1999	6.0	11.2	11.0	1.0	2.9
	2000	11.2	11.2	10.7	1.3	3.0
	2001	11.7	10.3	12.4	0.8	2.8
	2002	12.8	10.2	13.3	1.1	2.7
	2003	11.1	9.9	15.9	1.4	2.6

	2004	15.1	8.9	17.5	0.9	1.9
	2005	5.2	8.6	16.8	1.1	2.0
	2006	3.4	7.8	18.3	0.6	1.9
	2007	5.6	8.0	18.6	0.9	2.5
	2008	7.0	8.3	20.7	0.9	2.5
Jaap Reef	1996	16.0	30.7	1.3	0.5	0.1
24.5857 °N	1997	9.8	33.5	1.9	1.7	0.0
81.5826 °W	1998	18.1	27.3	1.2	3.5	0.0
	1999	15.9	18.1	0.7	2.7	0.0
	2000	12.1	15.9	0.4	4.1	0.2
	2001	26.1	16.0	1.1	1.4	0.0
	2002	25.9	17.3	0.7	1.7	0.0
	2003	26.3	21.4	1.1	1.4	0.0
	2004	31.4	18.6	0.9	1.5	0.0
	2005	13.7	20.4	2.0	1.4	0.1
	2006	23.8	21.2	0.9	0.9	0.0
	2007	27.3	18.9	1.0	1.2	0.0
	2008	33.1	20.8	1.2	1.0	0.0
Looe Key Deep	1996	17.3	8.9	15.3	4.8	0.3
24.5423 °N	1997	8.0	8.5	12.5	5.7	0.4
81.4145 °W	1998	46.3	5.7	15.8	3.1	0.1
	1999	13.3	5.5	10.4	2.0	0.3
	2000	33.8	5.5	9.6	4.2	0.2
	2001	23.7	6.6	10.4	3.3	0.2
	2002	6.3	5.3	9.5	6.6	0.2
	2003	7.7	4.8	12.9	4.2	0.0
	2004	18.0	3.9	14.1	4.1	0.2
	2005	4.8	4.5	14.6	3.9	0.2
	2006	36.5	2.9	7.5	1.4	0.2
	2007	24.0	2.9	9.7	2.2	0.1
	2008	34.4	2.6	10.9	3.5	0.2
Looe Key Shallow	1996	7.1	22.1	9.1	1.6	4.2
24.5453 °N	1997	0.2	21.6	9.5	0.8	4.7
81.4066 °W	1998	10.8	22.0	8.0	0.4	4.8
	1999	4.1	16.6	8.8	0.9	4.3
	2000	9.3	16.9	6.7	0.6	5.4
	2001	4.4	17.6	7.0	0.7	4.2
	2002	10.4	17.1	8.3	0.8	5.3
	2003	13.8	17.2	10.7	0.3	5.5
	2004	7.8	16.3	14.2	0.5	4.5
	2005	6.4	14.3	12.5	0.7	4.2
	2006	3.0	14.6	9.8	0.2	4.7
	2007	8.6	14.5	9.2	0.4	3.9
	2008	2.9	12.6	14.4	0.4	4.3
Molasses Deep	1996	10.7	3.0	20.0	8.6	0.1
25.0072 °N	1997	5.6	3.7	14.8	9.7	0.4
80.3756 °W	1998	25.7	2.0	13.6	5.5	0.1

	1999	0.8	2.3	10.5	8.3	0.2
	2000	3.8	2.6	13.0	9.0	0.0
	2001	5.3	1.4	12.3	6.5	0.1
	2002	6.1	1.7	12.7	8.3	0.2
	2003	1.9	2.3	16.5	7.8	0.1
	2004	4.4	2.0	19.2	6.2	0.2
	2005	3.1	1.9	20.9	6.7	0.3
	2006	15.3	1.6	19.9	5.7	0.1
	2007	10.2	1.8	14.6	6.3	0.3
	2008	11.8	1.6	13.3	7.1	0.1
Molasses Shallow	1996	8.0	11.7	9.2	1.2	3.3
25.0089 °N	1997	2.8	10.1	7.7	0.6	4.5
80.3753 °W	1998	4.9	6.3	9.0	0.7	3.0
	1999	0.8	3.9	6.9	0.5	3.3
	2000	1.3	3.5	8.0	1.2	4.1
	2001	1.3	3.6	10.6	1.3	2.8
	2002	1.8	4.4	10.2	0.7	4.1
	2003	5.8	3.9	16.5	0.5	3.8
	2004	9.1	4.9	17.1	0.3	3.6
	2005	3.3	4.6	21.1	0.9	3.7
	2006	3.4	3.8	19.6	0.4	4.2
	2007	6.9	4.9	20.2	0.5	4.5
	2008	7.9	5.1	17.9	0.6	4.8
Porter Patch	1996	9.0	3.4	14.3	10.6	0.4
25.1032 °N	1997	9.3	3.3	14.2	3.3	0.7
80.3243 °W	1998	32.6	3.1	13.9	2.4	0.4
	1999	13.1	2.7	14.6	2.7	0.5
	2000	6.2	3.6	12.4	3.0	0.4
	2001	12.4	3.9	17.5	3.7	0.6
	2002	15.9	3.3	17.3	3.7	0.6
	2003	14.7	3.1	14.5	2.9	0.2
	2004	16.4	2.5	18.4	2.5	0.3
	2005	10.6	2.7	15.2	2.7	0.4
	2006	12.1	2.4	11.6	2.2	0.3
	2007	9.2	3.3	12.2	2.3	0.2
	2008	16.8	2.9	12.8	2.0	0.3
Rock Key Deep	1996	1.2	4.8	13.7	4.4	0.5
24.4532 °N	1997	15.9	8.0	10.2	3.4	0.4
81.8568 °W	1998	30.7	4.4	12.9	2.9	0.4
	1999	9.6	3.9	10.0	2.4	0.3
	2000	14.8	3.1	9.3	2.4	0.3
	2001	21.8	3.2	6.6	3.3	0.3
	2002	3.8	4.3	7.8	3.0	0.4
	2003	5.5	3.3	13.6	2.7	0.3
	2004	7.1	2.7	13.4	2.0	0.2
	2005	1.6	2.7	12.3	2.6	0.2
	2006	5.5	1.7	7.9	0.8	0.3

	2007	11.6	2.6	9.6	1.5	0.3
	2008	12.8	2.5	8.3	2.4	0.3
Rock Key Shallow	1996	1.7	11.5	3.1	0.9	4.3
24.4546 °N	1997	2.5	10.5	4.4	0.6	4.8
81.8584 °W	1998	9.1	7.4	5.1	0.2	4.6
	1999	5.2	3.3	3.1	0.4	4.2
	2000	4.0	3.0	3.8	0.5	4.2
	2001	5.7	2.8	4.2	0.2	4.7
	2002	10.5	4.0	3.7	0.2	5.0
	2003	2.5	4.0	5.5	0.3	5.3
	2004	3.9	3.7	6.6	0.3	5.1
	2005	0.5	2.3	4.4	0.2	4.4
	2006	15.4	1.7	4.2	0.1	3.9
	2007	6.6	1.7	4.9	0.2	4.4
	2008	1.8	1.8	5.3	0.1	4.0
Sand Key Deep	1996	2.1	4.5	11.6	1.9	0.9
24.4517 °N	1997	34.7	5.8	11.0	1.8	0.4
81.8798 °W	1998	32.3	4.8	14.9	3.1	0.3
	1999	12.7	2.5	11.2	2.4	0.2
	2000	29.5	2.8	13.3	2.7	0.3
	2001	26.6	2.6	9.7	2.3	0.2
	2002	7.4	2.5	11.6	2.7	0.5
	2003	11.5	2.7	10.3	2.2	0.4
	2004	7.1	2.3	12.4	1.8	0.5
	2005	2.3	1.9	10.7	2.3	0.5
	2006	17.6	1.0	12.9	1.1	0.3
	2007	19.9	1.6	11.4	1.6	0.6
	2008	2.6	2.2	12.9	2.6	0.6
Sand Key Shallow	1996	1.1	12.9	2.8	0.6	0.7
24.4520 °N	1997	4.4	13.2	3.3	0.4	1.2
81.8775 °W	1998	2.5	10.1	2.4	0.1	1.1
	1999	0.4	5.7	2.1	1.1	1.1
	2000	1.4	6.1	1.5	0.4	1.1
	2001	3.2	4.9	2.4	0.1	0.9
	2002	1.8	5.6	2.1	0.4	1.6
	2003	0.8	4.7	2.4	0.3	1.9
	2004	1.7	5.1	4.5	0.5	1.3
	2005	0.2	4.2	3.3	0.0	0.9
	2006	27.7	3.2	4.3	0.2	0.9
	2007	5.2	3.4	3.8	0.1	1.4
	2008	0.2	4.3	5.4	0.1	1.0
Smith Shoal	1996	7.2	15.4	1.2	3.4	0.0
24.7197 °N	1997	14.9	14.6	0.7	1.7	0.1
81.9195 °W	1998	4.6	15.4	0.2	1.2	0.0
	1999	1.0	17.4	0.3	2.0	0.2
	2000	0.3	20.2	0.4	2.9	0.0
	2001	2.5	17.7	0.0	3.0	0.0

	2002	24.9	4.8	0.1	1.0	0.0
	2003	23.8	4.0	0.1	1.5	0.0
	2004	8.0	5.3	0.0	1.3	0.0
	2005	1.0	7.2	0.0	2.2	0.0
	2006	0.8	4.9	0.0	2.1	0.0
	2007	19.3	4.1	0.0	1.2	0.0
	2008	60.6	3.4	0.0	0.5	0.0
Sombrero Deep	1996	7.6	3.7	10.8	3.9	1.3
24.6231 °N	1997	4.0	4.6	7.1	3.3	0.1
81.1105 °W	1998	54.0	3.8	6.7	2.2	0.0
	1999	25.2	2.8	2.1	1.7	0.0
	2000	31.2	2.0	2.0	3.1	0.0
	2001	25.7	3.2	4.2	4.9	0.0
	2002	1.9	2.2	2.7	2.4	0.0
	2003	11.1	3.0	3.3	3.0	0.0
	2004	10.2	3.0	6.4	2.0	0.0
	2005	9.6	2.5	5.5	3.1	0.0
	2006	40.6	1.6	3.9	1.4	0.0
	2007	23.9	1.3	3.3	1.5	0.0
	2008	29.3	1.5	3.6	2.6	0.0
Sombrero Shallow	1996	5.0	7.1	10.1	1.3	15.6
24.6259 °N	1997	4.7	6.2	10.6	2.9	17.5
81.1101 °W	1998	20.8	5.5	9.5	1.7	15.1
	1999	4.8	3.1	7.1	2.3	15.0
	2000	9.0	2.6	8.6	2.7	17.0
	2001	2.1	2.7	8.2	1.7	15.9
	2002	2.8	3.2	12.6	1.5	18.6
	2003	12.9	2.8	15.6	1.1	15.3
	2004	10.6	2.5	15.2	1.3	15.5
	2005	5.6	2.3	13.6	1.7	13.5
	2006	14.4	2.1	10.8	0.9	11.7
	2007	6.5	2.0	11.6	1.2	11.0
	2008	8.0	2.5	16.2	1.4	10.3
Tennessee Deep	1996	3.6	6.6	10.3	7.9	0.3
24.7527 °N	1997	3.5	7.5	10.4	8.1	0.0
80.7578 °W	1998	45.6	5.1	8.6	4.6	0.0
	1999	19.6	4.7	4.2	2.2	0.0
	2000	17.2	5.2	5.8	3.9	0.1
	2001	21.4	4.4	6.7	5.1	0.0
	2002	2.8	4.5	7.4	6.7	0.0
	2003	3.2	4.7	7.3	6.1	0.0
	2004	28.4	3.5	10.0	6.7	0.0
	2005	11.1	4.6	10.7	4.8	0.0
	2006	19.6	4.7	8.7	3.6	0.0
	2007	13.3	3.6	8.6	5.1	0.0
	2008	17.8	3.7	9.5	5.8	0.0
Tennessee Shallow	1996	5.6	3.8	24.7	2.9	0.9

24.7450 °N	1997	1.1	3.8	21.1	2.1	1.0
80.7812 °W	1998	16.4	3.5	18.1	2.5	0.9
	1999	3.0	1.8	10.8	2.2	1.0
	2000	5.3	2.3	10.1	3.5	1.1
	2001	0.5	2.3	10.7	3.0	1.5
	2002	0.8	2.2	13.3	3.0	1.6
	2003	5.7	1.9	17.1	3.7	0.7
	2004	3.7	1.8	18.8	2.4	1.1
	2005	12.1	2.1	18.5	2.5	1.3
	2006	10.5	1.4	15.5	0.9	1.2
	2007	8.9	1.1	13.6	1.9	1.4
	2008	8.7	1.9	18.5	2.0	1.3
Turtle	1996	12.8	8.3	35.0	1.2	0.7
25.2947 °N	1997	3.7	8.3	38.2	2.0	0.5
80.2191 °W	1998	17.1	5.4	28.1	2.2	0.7
	1999	6.8	4.4	30.1	1.4	0.8
	2000	10.0	5.5	28.3	1.9	0.9
	2001	1.6	6.6	31.4	2.1	0.7
	2002	17.9	6.4	35.3	2.2	0.7
	2003	3.3	5.9	37.3	2.2	0.8
	2004	3.9	6.1	34.6	1.3	0.7
	2005	5.7	5.5	37.6	1.6	0.9
	2006	4.1	6.1	34.8	1.8	0.8
	2007	12.1	5.5	35.6	1.0	0.7
	2008	3.3	6.3	33.7	1.3	0.9
West Turtle Shoal	1996	1.8	14.9	30.7	6.6	2.7
24.6993 °N	1997	4.8	14.9	26.2	7.1	3.4
80.9669 °W	1998	3.3	13.5	27.7	8.1	2.6
	1999	3.4	12.0	24.8	7.4	3.3
	2000	2.0	12.7	22.2	8.8	3.4
	2001	2.3	13.7	27.6	8.4	2.6
	2002	0.9	15.7	27.4	10.0	3.4
	2003	0.5	13.8	32.7	7.4	3.2
	2004	1.4	12.4	35.0	5.6	2.3
	2005	2.1	14.4	31.5	5.5	2.5
	2006	2.4	12.2	29.0	4.8	2.9
	2007	4.9	13.4	27.7	7.3	3.6
	2008	0.7	12.1	30.8	6.7	2.5
West Washer Women	1996	3.1	31.9	17.5	2.2	0.0
24.5475 °N	1997	1.9	25.2	24.2	1.0	0.0
81.5866 °W	1998	8.3	27.0	16.2	2.8	0.0
	1999	4.9	23.6	15.4	1.3	0.0
	2000	0.4	25.4	11.8	3.2	0.0
	2001	2.5	23.4	19.2	3.1	0.0
	2002	0.8	29.1	16.3	2.8	0.0
	2003	0.6	23.9	21.3	2.2	0.1
	2004	1.5	24.6	16.5	2.8	0.1

	2005	0.5	25.3	21.3	2.7	0.0
	2006	2.7	26.8	17.4	2.0	0.1
	2007	5.8	25.7	18.9	3.0	0.0
	2008	1.6	21.3	21.9	3.0	0.0
Western Head	1996	2.2	26.9	0.2	6.0	0.0
24.4977 °N	1997	4.4	23.5	0.1	4.0	0.0
81.8056 °W	1998	2.7	26.0	0.4	4.7	0.4
	1999	2.3	21.1	0.1	4.1	0.0
	2000	0.1	24.3	0.0	7.1	0.0
	2001	8.5	23.3	0.8	7.3	0.0
	2002	0.3	23.5	0.4	7.9	0.1
	2003	0.4	24.4	1.6	9.7	0.0
	2004	0.6	21.3	0.3	9.4	0.0
	2005	0.2	24.0	1.6	4.2	0.0
	2006	2.8	22.3	0.3	2.7	0.0
	2007	2.0	21.7	2.8	7.3	0.0
	2008	1.3	21.4	1.1	8.4	0.0
Western Sambo Deep	1996	45.9	10.2	5.9	1.5	0.0
24.4785 °N	1997	27.4	9.4	5.2	2.0	0.0
81.7161 °W	1998	52.3	5.2	5.3	0.7	0.0
	1999	32.5	3.3	2.3	0.3	0.0
	2000	38.7	3.1	2.2	0.8	0.0
	2001	28.7	3.3	2.3	0.9	0.0
	2002	41.3	3.2	4.0	1.1	0.0
	2003	14.8	3.4	4.5	0.7	0.0
	2004	39.0	2.3	7.5	0.8	0.0
	2005	1.8	2.3	4.6	0.9	0.0
	2006	3.0	1.0	4.2	0.5	0.0
	2007	29.8	1.2	4.3	0.6	0.0
	2008	2.9	1.7	7.6	0.8	0.0
Western Sambo Shallow	1996	5.0	21.1	2.7	0.6	3.5
	1997	5.0	22.4	2.0	0.7	3.7
24.4797 °N	1998	11.0	16.5	2.6	0.1	3.4
81.7170 °W	1999	5.8	6.2	1.6	0.8	2.3
	2000	4.9	6.5	1.4	0.7	2.4
	2001	3.4	5.3	2.0	0.9	1.7
	2002	6.8	5.7	3.3	0.3	2.5
	2003	10.1	6.2	4.2	0.2	2.2
	2004	17.6	6.6	5.8	0.4	2.0
	2005	1.9	3.8	3.9	0.3	1.7
	2006	9.0	2.9	3.8	0.4	1.2
	2007	6.9	2.7	5.6	0.1	1.1
	2008	3.0	2.6	5.9	0.2	1.0
White Shoal	1999	6.1	9.5	8.4	2.9	0.0
24.6416 °N	2000	8.9	8.0	7.6	4.0	0.0
82.8964 °W	2001	11.4	5.8	8.5	1.9	0.0
	2002	10.3	5.6	6.9	1.7	0.0

	2003	14.9	4.2	8.7	1.6	0.0
	2004	6.7	1.9	4.1	0.4	0.0
	2005	9.6	1.6	3.4	0.3	0.0
	2006	14.8	1.1	5.1	0.6	0.0
	2007	17.2	1.2	6.1	0.9	0.0
	2008	13.6	1.8	8.8	0.8	0.0

Appendix 3. Percent cover values for five sentinel coral species for all years at all region*habitat groupings.

Region	Habitat	Sample Year	*Montastraea annularis*	*Montastraea cavernosa*	*Colpophyllia natans*	*Siderastrea siderea*	*Porites astreoides*
DT	**OD**	1999	11.1	5.2	2.6	1.1	0.8
		2000	11.3	4.3	2.5	1.0	0.7
		2001	12.0	4.7	2.7	1.1	0.8
		2002	8.5	4.6	2.0	0.9	0.9
		2003	8.2	4.1	1.5	1.0	0.9
		2004	7.9	3.7	1.4	1.3	0.7
		2005	7.9	3.7	1.5	1.2	0.5
		2006	5.8	3.3	1.3	1.0	0.3
		2007	7.2	2.9	1.4	1.0	0.5
		2008	7.2	2.9	1.5	1.4	0.4
DT	**P**	1999	0.2	0.1	0.3	0.4	0.1
		2000	0.3	0.1	0.2	0.3	0.2
		2001	0.4	0.1	0.5	0.4	0.0
		2002	0.3	0.1	0.3	0.1	0.0
		2003	0.4	0.1	0.2	0.2	0.1
		2004	0.2	0.1	0.2	0.3	0.0
		2005	0.3	0.1	0.1	0.2	0.1
		2006	0.2	0.0	0.1	0.2	0.0
		2007	0.1	0.1	0.1	0.2	0.1
		2008	0.2	0.0	0.3	0.2	0.1
LK	**BCP**	1996	4.1	0.7	0.0	0.1	1.3
		1997	4.7	0.5	0.0	0.1	1.4
		1998	3.8	0.8	0.0	0.1	2.0
		1999	6.1	0.6	0.0	0.2	0.9
		2000	5.0	0.7	0.0	0.3	2.4
		2001	4.2	0.8	0.0	0.3	2.2
		2002	0.0	0.6	0.0	0.3	0.3
		2003	0.0	0.5	0.0	0.2	0.4
		2004	0.0	0.6	0.0	0.3	0.3
		2005	0.0	0.6	0.0	0.2	0.6
		2006	0.0	0.7	0.0	0.2	0.4
		2007	0.0	0.6	0.0	0.2	0.2
		2008	0.0	0.8	0.0	0.0	0.2
LK	**OD**	1996	2.2	0.9	0.3	0.8	0.2
		1997	2.4	0.6	0.4	1.0	0.3
		1998	2.1	0.5	0.3	0.6	0.3
		1999	2.0	0.5	0.2	0.8	0.1
		2000	2.0	0.6	0.2	0.6	0.1
		2001	1.9	0.6	0.2	0.6	0.2
		2002	1.7	0.7	0.2	0.6	0.2
		2003	1.4	0.5	0.2	0.6	0.2
		2004	1.1	0.4	0.1	0.5	0.1
		2005	1.3	0.4	0.1	0.6	0.1

		2006	0.6	0.3	0.0	0.5	0.1
		2007	0.6	0.2	0.1	0.7	0.1
		2008	0.5	0.3	0.1	0.7	0.1
LK	**OS**	1996	3.9	0.4	0.2	0.3	1.6
		1997	3.8	0.4	0.3	0.3	1.6
		1998	4.1	0.4	0.4	0.3	1.6
		1999	3.4	0.2	0.1	0.2	1.9
		2000	3.4	0.3	0.1	0.3	1.9
		2001	3.6	0.2	0.2	0.2	1.6
		2002	3.3	0.3	0.1	0.3	2.1
		2003	3.5	0.4	0.1	0.3	1.6
		2004	3.5	0.2	0.1	0.2	1.5
		2005	2.7	0.2	0.1	0.3	1.4
		2006	2.8	0.2	0.1	0.2	1.2
		2007	2.7	0.3	0.1	0.2	1.1
		2008	2.5	0.1	0.1	0.2	1.2
LK	**P**	1996	10.2	8.6	2.4	2.9	0.5
		1997	11.6	6.0	2.3	2.4	0.7
		1998	9.6	7.5	2.6	2.8	0.5
		1999	6.8	6.6	2.1	2.3	0.4
		2000	6.0	7.4	2.1	2.6	0.5
		2001	6.0	7.3	1.9	2.4	0.4
		2002	6.3	7.4	2.6	2.9	0.8
		2003	7.2	7.2	2.5	2.2	0.6
		2004	5.9	6.4	2.5	2.4	0.5
		2005	7.1	7.1	2.4	2.7	0.5
		2006	7.5	6.8	2.2	2.5	0.5
		2007	6.4	6.8	2.7	2.5	0.4
		2008	6.7	6.4	1.5	2.5	0.5
MK	**OD**	1996	1.0	0.5	0.2	0.7	0.1
		1997	1.3	0.2	0.4	0.6	0.2
		1998	1.1	0.1	0.2	0.5	0.2
		1999	1.2	0.1	0.0	0.7	0.2
		2000	1.1	0.3	0.0	0.7	0.2
		2001	0.9	0.2	0.0	0.7	0.2
		2002	0.8	0.1	0.0	0.6	0.2
		2003	1.0	0.2	0.0	0.7	0.2
		2004	0.7	0.2	0.1	0.6	0.1
		2005	0.7	0.2	0.0	0.6	0.2
		2006	0.7	0.2	0.0	0.7	0.1
		2007	0.7	0.1	0.0	0.5	0.1
		2008	0.5	0.1	0.0	0.6	0.1
MK	**OS**	1996	0.7	0.3	0.2	0.4	0.4
		1997	0.5	0.2	0.1	0.2	0.6
		1998	0.6	0.2	0.3	0.3	0.4
		1999	0.4	0.2	0.1	0.4	0.2
		2000	0.4	0.2	0.0	0.3	0.3

		2001	0.3	0.2	0.1	0.3	0.3
		2002	0.4	0.2	0.0	0.5	0.2
		2003	0.3	0.1	0.1	0.3	0.2
		2004	0.2	0.2	0.1	0.4	0.2
		2005	0.2	0.1	0.0	0.4	0.2
		2006	0.1	0.1	0.0	0.4	0.2
		2007	0.1	0.1	0.1	0.4	0.1
		2008	0.1	0.1	0.0	0.4	0.2
MK	**P**	1996	2.6	4.1	1.3	3.6	1.0
		1997	3.7	3.6	1.1	3.6	1.1
		1998	2.4	4.0	1.8	3.4	1.2
		1999	2.3	4.6	1.3	2.9	0.9
		2000	2.1	4.4	1.3	4.0	1.1
		2001	2.2	4.0	1.5	3.8	1.2
		2002	2.5	4.7	1.9	3.9	1.1
		2003	2.1	4.5	1.8	3.3	1.0
		2004	2.3	3.5	1.6	3.4	0.9
		2005	2.9	3.9	1.9	3.2	0.9
		2006	2.4	4.0	1.6	3.4	1.1
		2007	2.3	4.4	1.9	3.8	1.0
		2008	2.3	3.8	1.5	3.7	1.0
UK	**OD**	1996	2.9	0.4	0.2	0.8	0.2
		1997	2.1	0.3	0.0	0.8	0.2
		1998	0.9	0.3	0.0	0.8	0.2
		1999	0.8	0.2	0.1	0.7	0.1
		2000	0.9	0.3	0.0	0.8	0.1
		2001	0.6	0.3	0.1	0.7	0.2
		2002	0.7	0.2	0.0	0.6	0.1
		2003	0.7	0.2	0.1	0.5	0.3
		2004	0.6	0.2	0.0	0.6	0.2
		2005	0.6	0.2	0.0	0.7	0.1
		2006	0.5	0.1	0.1	0.6	0.3
		2007	0.6	0.2	0.1	0.7	0.4
		2008	0.4	0.2	0.1	0.6	0.2
UK	**OS**	1996	4.1	0.2	0.1	0.5	0.4
		1997	3.6	0.1	0.1	0.3	0.5
		1998	2.8	0.1	0.1	0.3	0.5
		1999	2.8	0.1	0.1	0.3	0.5
		2000	2.7	0.1	0.0	0.6	0.5
		2001	2.7	0.1	0.1	0.5	0.5
		2002	2.4	0.2	0.1	0.5	0.6
		2003	2.5	0.1	0.0	0.4	0.6
		2004	2.4	0.1	0.0	0.3	0.5
		2005	2.2	0.1	0.1	0.3	0.6
		2006	2.1	0.0	0.0	0.4	0.5
		2007	2.1	0.1	0.1	0.4	0.5
		2008	2.0	0.0	0.0	0.5	0.6

UK	**P**	1996	12.4	0.1	0.0	1.2	0.4
		1997	11.4	0.2	0.1	1.1	0.3
		1998	11.1	0.2	0.0	1.2	0.5
		1999	9.7	0.2	0.0	0.9	0.3
		2000	8.5	0.2	0.1	1.1	0.4
		2001	9.2	0.1	0.0	1.4	0.4
		2002	9.2	0.2	0.0	1.2	0.4
		2003	10.0	0.1	0.1	1.0	0.6
		2004	9.7	0.0	0.0	1.1	0.5
		2005	9.1	0.1	0.1	1.2	0.4
		2006	8.3	0.1	0.0	1.2	0.5
		2007	10.3	0.1	0.0	1.2	0.5
		2008	9.8	0.1	0.0	1.0	0.5

Appendix 4. Percent cover values for five sentinel coral species for all years at all CREMP sites.

Site Name and Location	Sample Year	*Montastraea annularis*	*Montastraea cavernosa*	*Colpophyllia natans*	*Siderastrea siderea*	*Porites astreoides*
Admiral	1996	27.0	0.0	0.1	1.5	0.2
25.0447 °N	1997	24.9	0.0	0.1	1.2	0.2
80.3948 °W	1998	24.1	0.0	0.0	1.4	0.5
	1999	20.8	0.0	0.0	1.0	0.3
	2000	18.2	0.1	0.0	0.8	0.3
	2001	19.3	0.1	0.0	1.3	0.2
	2002	19.5	0.1	0.0	1.4	0.3
	2003	21.4	0.0	0.0	1.1	0.3
	2004	21.1	0.0	0.0	1.2	0.4
	2005	19.6	0.0	0.0	1.4	0.4
	2006	17.9	0.1	0.0	1.1	0.5
	2007	22.4	0.0	0.0	1.3	0.5
	2008	21.2	0.0	0.0	1.2	0.5
Alligator Deep	1996	0.1	0.1	0.0	0.1	0.0
24.8452 °N	1997	0.4	0.1	0.0	0.2	0.0
80.6209 °W	1998	0.2	0.0	0.0	0.1	0.0
	1999	0.1	0.0	0.0	0.2	0.0
	2000	0.1	0.1	0.0	0.1	0.1
	2001	0.1	0.0	0.0	0.2	0.0
	2002	0.0	0.0	0.0	0.2	0.0
	2003	0.0	0.0	0.0	0.1	0.0
	2004	0.0	0.0	0.0	0.1	0.0
	2005	0.0	0.0	0.0	0.2	0.0
	2006	0.0	0.0	0.0	0.2	0.0
	2007	0.0	0.0	0.0	0.2	0.0
	2008	0.0	0.0	0.0	0.2	0.0
Alligator Shallow	1996	0.0	0.1	0.0	0.1	0.3
24.8461 °N	1997	0.0	0.1	0.0	0.0	0.3
80.6236 °W	1998	0.0	0.0	0.0	0.0	0.2
	1999	0.0	0.0	0.0	0.1	0.1
	2000	0.0	0.0	0.0	0.1	0.0
	2001	0.0	0.0	0.1	0.1	0.0
	2002	0.0	0.0	0.0	0.1	0.0
	2003	0.0	0.0	0.0	0.1	0.0
	2004	0.0	0.0	0.0	0.1	0.0
	2005	0.0	0.0	0.0	0.1	0.0
	2006	0.0	0.0	0.0	0.1	0.0
	2007	0.0	0.0	0.0	0.1	0.0
	2008	0.0	0.0	0.0	0.0	0.0
Bird Key Reef	1999	10.5	1.9	3.0	1.9	0.9
24.6117 °N	2000	11.5	2.5	2.9	1.6	1.0
82.8702 °W	2001	11.6	2.2	3.5	1.7	1.0
	2002	7.8	2.5	1.1	1.2	1.3
	2003	7.1	1.9	0.7	1.7	1.1
	2004	6.7	1.4	0.6	2.0	0.9
	2005	5.5	1.5	0.4	1.7	0.5

	2006	4.8	1.8	0.6	1.4	0.2
	2007	6.1	1.0	0.7	1.6	0.4
	2008	5.9	0.8	0.6	2.1	0.4
Black Coral Rock	1999	11.8	8.4	2.3	0.4	0.6
24.6993 ^{o}N	2000	11.1	6.0	2.1	0.4	0.5
83.0022 ^{o}W	2001	12.3	7.2	2.0	0.5	0.5
	2002	9.2	6.7	3.0	0.7	0.4
	2003	9.2	6.2	2.4	0.2	0.7
	2004	9.0	6.0	2.2	0.7	0.6
	2005	10.3	5.8	2.7	0.7	0.5
	2006	6.8	4.9	2.0	0.5	0.5
	2007	8.3	4.9	2.1	0.4	0.6
	2008	8.6	4.9	2.3	0.7	0.4
Carysfort Deep	1996	7.4	1.0	0.4	1.3	0.2
25.2208 ^{o}N	1997	5.4	0.6	0.1	1.4	0.4
80.2099 ^{o}W	1998	1.7	0.6	0.1	1.9	0.4
	1999	1.9	0.3	0.4	1.6	0.1
	2000	2.3	0.9	0.1	1.7	0.3
	2001	1.3	0.6	0.2	1.3	0.4
	2002	1.8	0.6	0.1	1.0	0.4
	2003	1.8	0.4	0.3	0.9	0.8
	2004	1.5	0.4	0.1	0.8	0.3
	2005	1.3	0.5	0.1	1.4	0.1
	2006	1.3	0.2	0.3	1.0	0.3
	2007	1.7	0.5	0.1	1.0	0.9
	2008	0.8	0.4	0.3	1.0	0.3
Carysfort Shallow	1996	0.8	0.2	0.0	0.2	0.9
25.2210 ^{o}N	1997	0.1	0.0	0.0	0.2	1.3
80.2103 ^{o}W	1998	0.2	0.0	0.0	0.1	1.2
	1999	0.2	0.0	0.0	0.1	1.3
	2000	0.1	0.1	0.0	0.4	1.7
	2001	0.2	0.0	0.0	0.7	1.7
	2002	0.3	0.2	0.0	0.4	2.0
	2003	0.1	0.0	0.0	0.2	1.8
	2004	0.1	0.0	0.0	0.2	1.5
	2005	0.2	0.0	0.0	0.2	2.0
	2006	0.1	0.0	0.0	0.4	1.6
	2007	0.1	0.0	0.0	0.3	1.4
	2008	0.2	0.0	0.0	0.3	1.7
Cliff Green	1996	2.8	9.9	0.8	1.2	0.2
24.5036 ^{o}N	1997	4.7	5.7	2.3	1.3	0.6
81.7677 ^{o}W	1998	3.1	6.7	2.7	1.4	0.3
	1999	2.7	8.0	2.0	1.4	0.2
	2000	2.0	7.7	2.0	2.0	0.1
	2001	2.1	7.1	2.2	1.6	0.2
	2002	2.0	7.5	2.9	2.2	0.6
	2003	1.6	7.3	2.5	1.4	0.2
	2004	0.8	5.3	1.6	1.3	0.1
	2005	0.9	6.7	1.6	2.5	0.2

	2006	1.0	6.8	1.4	2.2	0.0
	2007	0.3	7.3	2.4	2.1	0.2
	2008	0.9	5.5	1.2	2.5	0.2
Conch Deep	1996	0.9	0.1	0.0	0.9	0.3
24.9519 °N	1997	1.0	0.2	0.0	0.5	0.2
80.4513 °W	1998	0.8	0.2	0.0	0.4	0.2
	1999	0.6	0.3	0.0	0.5	0.2
	2000	0.5	0.1	0.0	0.4	0.1
	2001	0.6	0.1	0.0	0.9	0.1
	2002	0.2	0.1	0.0	0.7	0.0
	2003	0.2	0.0	0.0	0.5	0.0
	2004	0.4	0.1	0.0	0.7	0.3
	2005	0.6	0.0	0.0	0.6	0.1
	2006	0.2	0.1	0.0	0.7	0.3
	2007	0.2	0.0	0.0	0.7	0.2
	2008	0.4	0.0	0.0	0.8	0.3
Conch Shallow	1996	0.0	0.0	0.0	0.3	0.0
24.9559 °N	1997	0.0	0.0	0.0	0.2	0.0
80.4574 °W	1998	0.0	0.0	0.0	0.3	0.0
	1999	0.0	0.0	0.0	0.2	0.0
	2000	0.0	0.0	0.0	0.3	0.0
	2001	0.0	0.0	0.0	0.3	0.0
	2002	0.0	0.0	0.0	0.3	0.0
	2003	0.0	0.0	0.0	0.2	0.0
	2004	0.0	0.0	0.0	0.2	0.0
	2005	0.0	0.0	0.0	0.2	0.0
	2006	0.0	0.0	0.0	0.3	0.0
	2007	0.0	0.0	0.0	0.2	0.0
	2008	0.0	0.0	0.0	0.2	0.0
Content Keys	1996	0.0	0.0	0.0	0.1	0.5
24.8221 °N	1997	0.0	0.0	0.0	0.0	0.4
81.4889 °W	1998	0.0	0.0	0.0	0.0	0.5
	1999	0.0	0.0	0.0	0.3	0.7
	2000	0.0	0.0	0.0	0.0	0.6
	2001	0.0	0.0	0.0	0.4	1.2
	2002	0.0	0.0	0.0	0.2	0.2
	2003	0.0	0.0	0.0	0.1	0.2
	2004	0.0	0.0	0.0	0.1	0.1
	2005	0.0	0.0	0.0	0.1	0.2
	2006	0.0	0.0	0.0	0.1	0.1
	2007	0.0	0.0	0.0	0.1	0.1
	2008	0.0	0.0	0.0	0.0	0.1
Dustan Rocks	1996	5.1	3.6	0.6	4.4	2.1
24.6895 °N	1997	5.5	3.2	0.4	3.8	1.4
81.0302 °W	1998	4.6	3.8	0.6	3.7	2.1
	1999	4.4	3.7	0.8	3.7	1.2
	2000	4.3	4.6	0.7	4.0	1.4
	2001	3.8	3.8	0.6	3.6	1.5
	2002	4.7	4.4	0.9	4.1	1.1

	2003	4.3	4.6	0.5	3.6	1.5
	2004	4.5	3.4	0.8	3.8	1.3
	2005	5.4	3.5	0.7	3.5	1.2
	2006	4.8	4.3	0.6	3.9	1.5
	2007	4.8	5.0	0.8	4.5	1.4
	2008	4.7	4.3	0.8	4.2	1.6
Eastern Sambo Deep	1996	3.4	1.5	0.3	1.2	0.4
24.4884 °N	1997	2.8	1.4	0.7	1.7	0.3
81.6659 °W	1998	3.0	1.0	0.5	0.9	0.4
	1999	2.4	1.3	0.3	1.3	0.1
	2000	3.5	1.4	0.0	0.8	0.3
	2001	2.4	1.3	0.2	0.9	0.2
	2002	2.8	1.6	0.1	0.7	0.3
	2003	2.1	1.3	0.3	0.7	0.1
	2004	1.8	1.0	0.2	0.8	0.1
	2005	1.5	1.0	0.2	1.1	0.3
	2006	0.7	0.7	0.0	1.1	0.1
	2007	0.7	0.6	0.2	1.0	0.2
	2008	0.4	0.5	0.2	0.9	0.2
Eastern Sambo Shallow	1996	0.3	0.0	0.0	0.0	3.7
	1997	0.4	0.0	0.0	0.1	3.6
24.4917 °N	1998	0.2	0.1	0.0	0.2	4.2
81.6636 °W	1999	0.3	0.0	0.0	0.2	5.0
	2000	0.4	0.0	0.0	0.0	5.1
	2001	0.5	0.1	0.0	0.2	4.3
	2002	0.1	0.0	0.0	0.2	5.1
	2003	0.7	0.1	0.0	0.1	4.0
	2004	0.5	0.1	0.0	0.1	3.6
	2005	0.2	0.0	0.0	0.1	3.2
	2006	0.2	0.0	0.0	0.2	2.8
	2007	0.2	0.0	0.0	0.2	2.6
	2008	0.1	0.0	0.0	0.1	2.8
Grecian Rocks	1996	10.9	0.4	0.3	1.0	0.5
25.1075 °N	1997	10.2	0.1	0.3	0.6	0.6
80.3069 °W	1998	8.1	0.1	0.3	0.6	0.5
	1999	7.6	0.2	0.4	0.7	0.5
	2000	8.3	0.2	0.0	1.2	0.3
	2001	8.0	0.3	0.2	0.6	0.2
	2002	7.0	0.3	0.1	1.1	0.4
	2003	7.3	0.2	0.0	1.0	0.4
	2004	6.8	0.2	0.1	0.5	0.3
	2005	6.4	0.2	0.1	0.6	0.4
	2006	6.2	0.1	0.0	0.7	0.4
	2007	5.9	0.2	0.2	0.6	0.4
	2008	5.8	0.1	0.1	1.1	0.4
Jaap Reef	1996	26.8	0.8	0.7	0.4	1.0
24.5857 °N	1997	29.7	0.2	0.6	0.0	1.3
81.5826 °W	1998	25.1	0.6	0.4	0.0	0.7
	1999	16.5	0.3	0.3	0.0	0.8

	2000	14.0	0.1	0.4	0.2	0.9
	2001	14.4	0.0	0.5	0.0	0.7
	2002	15.3	0.2	0.6	0.1	0.8
	2003	19.2	0.2	0.6	0.2	0.8
	2004	16.4	0.3	0.6	0.1	0.8
	2005	18.6	0.1	0.5	0.0	1.0
	2006	19.8	0.0	0.4	0.0	0.7
	2007	17.3	0.2	0.6	0.1	0.6
	2008	18.5	0.1	0.3	0.1	0.9
Looe Key Deep	1996	2.4	1.5	0.6	1.3	0.3
24.5423 °N	1997	3.4	1.0	0.5	1.3	0.3
81.4145 °W	1998	2.7	0.8	0.5	0.6	0.3
	1999	2.9	0.6	0.2	1.1	0.1
	2000	2.3	0.6	0.3	1.1	0.1
	2001	3.0	0.8	0.2	0.9	0.2
	2002	2.1	0.6	0.3	0.9	0.2
	2003	1.6	0.5	0.2	0.9	0.1
	2004	1.5	0.3	0.1	0.6	0.2
	2005	2.1	0.5	0.1	0.8	0.1
	2006	1.3	0.3	0.0	0.6	0.2
	2007	0.8	0.3	0.0	1.0	0.1
	2008	0.7	0.4	0.0	0.8	0.0
Looe Key Shallow	1996	16.0	1.3	0.7	0.4	0.5
24.5453 °N	1997	16.1	1.3	1.5	0.5	0.5
81.4066 °W	1998	16.6	1.7	1.4	0.5	0.4
	1999	13.6	0.8	0.4	0.1	0.7
	2000	13.7	1.0	0.6	0.2	0.4
	2001	14.5	0.7	0.8	0.4	0.3
	2002	13.5	1.0	0.5	0.3	0.6
	2003	14.0	1.2	0.4	0.4	0.4
	2004	13.8	0.7	0.4	0.3	0.4
	2005	11.4	0.7	0.5	0.4	0.3
	2006	11.9	0.8	0.4	0.3	0.4
	2007	11.5	1.1	0.4	0.3	0.4
	2008	9.8	0.5	0.4	0.4	0.5
Molasses Deep	1996	0.4	0.1	0.1	0.1	0.1
25.0072 °N	1997	0.0	0.2	0.0	0.3	0.0
80.3756 °W	1998	0.2	0.1	0.0	0.1	0.1
	1999	0.1	0.1	0.0	0.2	0.1
	2000	0.0	0.0	0.0	0.2	0.0
	2001	0.0	0.1	0.0	0.1	0.0
	2002	0.0	0.0	0.0	0.2	0.1
	2003	0.0	0.1	0.0	0.2	0.1
	2004	0.0	0.0	0.0	0.2	0.1
	2005	0.0	0.1	0.0	0.0	0.1
	2006	0.0	0.0	0.0	0.1	0.1
	2007	0.0	0.1	0.1	0.2	0.1
	2008	0.0	0.0	0.0	0.1	0.1
Molasses Shallow	1996	2.2	0.3	0.0	0.1	0.2

25.0089 °N	1997	2.1	0.2	0.3	0.1	0.3
80.3753 °W	1998	1.3	0.1	0.0	0.2	0.2
	1999	1.7	0.1	0.0	0.1	0.2
	2000	0.7	0.1	0.1	0.1	0.2
	2001	0.9	0.1	0.1	0.3	0.3
	2002	0.8	0.2	0.1	0.1	0.2
	2003	1.1	0.1	0.0	0.1	0.3
	2004	1.3	0.1	0.0	0.1	0.1
	2005	1.0	0.1	0.2	0.1	0.2
	2006	0.7	0.0	0.0	0.2	0.2
	2007	1.1	0.1	0.1	0.2	0.2
	2008	0.9	0.1	0.0	0.2	0.2
Porter Patch	1996	0.6	0.0	0.0	1.0	0.1
25.1032 °N	1997	0.3	0.1	0.1	1.1	0.0
80.3243 °W	1998	0.6	0.1	0.0	0.7	0.1
	1999	0.7	0.2	0.0	0.7	0.0
	2000	0.5	0.2	0.0	1.5	0.0
	2001	0.7	0.1	0.0	1.4	0.1
	2002	0.8	0.2	0.0	0.9	0.1
	2003	0.5	0.1	0.0	1.0	0.0
	2004	0.4	0.1	0.0	0.7	0.1
	2005	0.5	0.0	0.0	1.1	0.1
	2006	0.3	0.0	0.0	1.0	0.0
	2007	0.5	0.0	0.0	1.2	0.1
	2008	0.4	0.0	0.0	1.0	0.0
Rock Key Deep	1996	1.3	0.9	0.6	0.5	0.0
24.4532 °N	1997	2.2	0.9	0.8	0.9	0.6
81.8568 °W	1998	1.2	1.0	0.4	0.5	0.4
	1999	1.4	0.5	0.4	0.3	0.4
	2000	0.7	1.1	0.1	0.3	0.2
	2001	1.0	0.7	0.2	0.3	0.3
	2002	1.0	1.6	0.3	0.6	0.3
	2003	0.8	0.6	0.0	0.3	0.5
	2004	0.5	0.4	0.0	0.5	0.3
	2005	0.6	0.5	0.3	0.3	0.2
	2006	0.2	0.5	0.0	0.5	0.2
	2007	0.6	0.4	0.0	0.7	0.2
	2008	0.3	0.4	0.1	0.6	0.2
Rock Key Shallow	1996	0.4	0.1	0.0	0.2	1.3
24.4546 °N	1997	0.3	0.2	0.0	0.2	1.2
81.8584 °W	1998	0.4	0.0	0.0	0.1	1.1
	1999	0.4	0.0	0.0	0.1	1.2
	2000	0.3	0.1	0.0	0.2	0.9
	2001	0.3	0.1	0.0	0.1	1.1
	2002	0.3	0.0	0.0	0.1	1.4
	2003	0.8	0.3	0.0	0.1	0.6
	2004	0.4	0.1	0.0	0.1	1.1
	2005	0.2	0.0	0.0	0.1	0.8
	2006	0.1	0.1	0.0	0.1	0.8

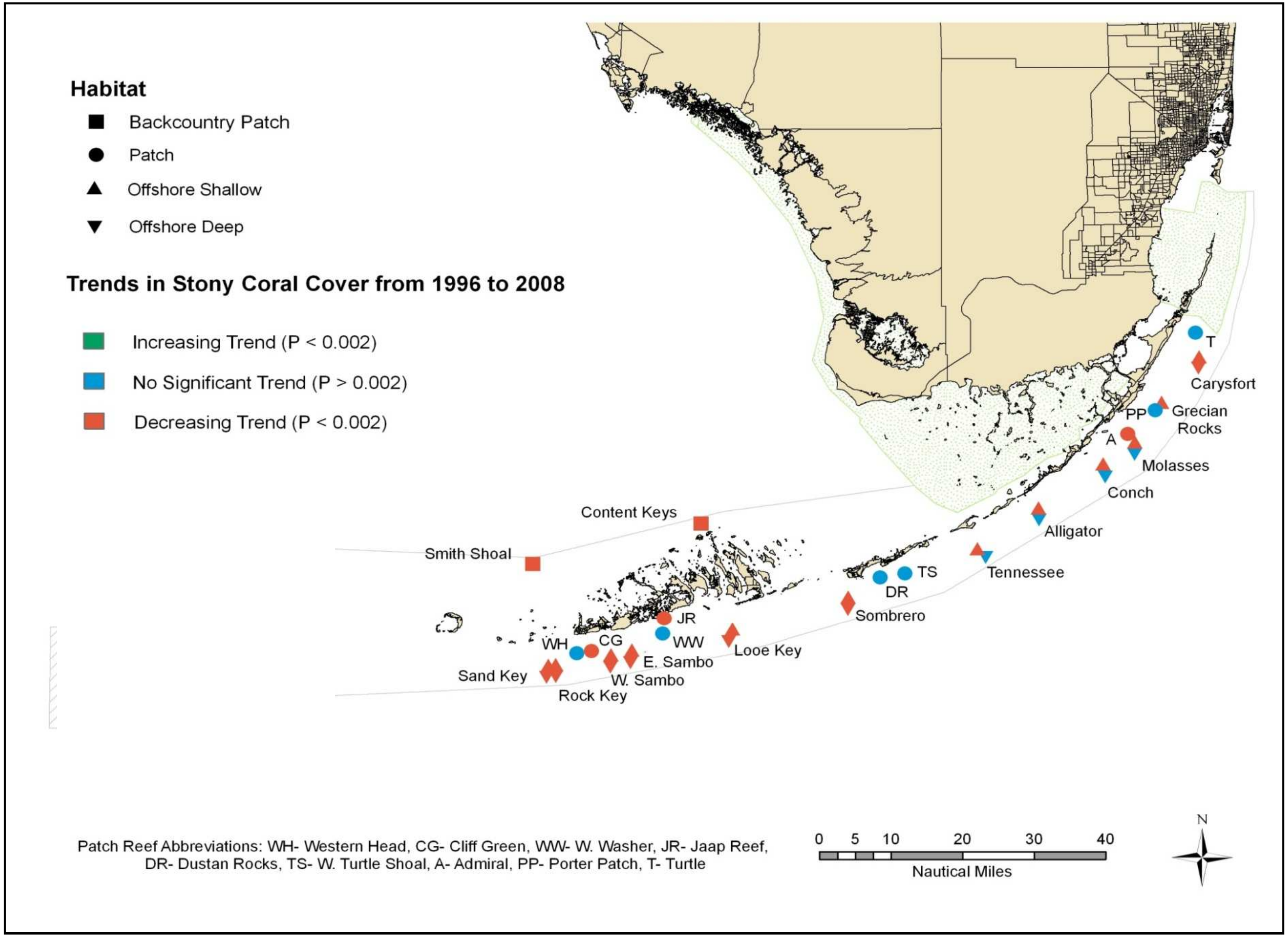
	2007	0.2	0.0	0.0	0.1	0.7
	2008	0.0	0.0	0.0	0.1	0.7
Sand Key Deep	1996	1.2	0.3	0.3	0.5	0.0
24.4517 °N	1997	1.2	0.0	0.2	0.4	0.1
81.8798 °W	1998	1.2	0.0	0.1	0.4	0.1
	1999	0.8	0.0	0.2	0.5	0.0
	2000	1.0	0.0	0.3	0.5	0.0
	2001	0.8	0.0	0.2	0.5	0.0
	2002	0.9	0.0	0.2	0.4	0.0
	2003	0.8	0.0	0.1	0.4	0.0
	2004	0.6	0.0	0.1	0.3	0.0
	2005	0.5	0.0	0.0	0.5	0.0
	2006	0.0	0.0	0.1	0.3	0.1
	2007	0.2	0.0	0.1	0.6	0.0
	2008	0.2	0.0	0.1	0.7	0.0
Sand Key Shallow	1996	2.3	0.5	0.1	0.6	0.7
24.4520 °N	1997	1.8	0.5	0.0	0.4	0.6
81.8775 °W	1998	3.3	0.2	0.1	0.5	0.4
	1999	2.6	0.3	0.0	0.3	0.6
	2000	2.6	0.5	0.0	0.9	0.7
	2001	2.7	0.3	0.0	0.4	0.7
	2002	2.4	0.4	0.0	0.7	0.8
	2003	1.6	0.5	0.0	0.6	0.7
	2004	2.4	0.3	0.0	0.3	0.8
	2005	1.5	0.5	0.0	0.8	0.7
	2006	1.7	0.2	0.0	0.4	0.7
	2007	1.8	0.6	0.0	0.3	0.4
	2008	2.9	0.2	0.0	0.3	0.6
Smith Shoal	1996	8.1	1.3	0.0	0.0	2.0
24.7197 °N	1997	9.4	1.0	0.0	0.1	2.4
81.9195 °W	1998	7.6	1.6	0.0	0.1	3.4
	1999	12.2	1.2	0.0	0.0	1.2
	2000	9.9	1.3	0.0	0.5	4.2
	2001	8.4	1.6	0.0	0.3	3.2
	2002	0.0	1.3	0.0	0.4	0.4
	2003	0.0	1.0	0.0	0.3	0.6
	2004	0.0	1.1	0.0	0.6	0.4
	2005	0.0	1.1	0.0	0.4	1.0
	2006	0.0	1.4	0.0	0.3	0.7
	2007	0.1	1.2	0.0	0.3	0.3
	2008	0.0	1.6	0.0	0.0	0.3
Sombrero Deep	1996	0.6	0.5	0.5	1.2	0.0
24.6231 °N	1997	0.9	0.4	1.0	1.0	0.1
81.1105 °W	1998	0.5	0.0	0.6	1.0	0.2
	1999	0.5	0.2	0.0	1.3	0.2
	2000	0.3	0.2	0.0	1.1	0.0
	2001	0.6	0.2	0.0	1.1	0.1
	2002	0.3	0.2	0.1	1.0	0.0
	2003	0.4	0.3	0.0	1.4	0.1

	2004	0.4	0.4	0.2	1.3	0.1
	2005	0.3	0.1	0.1	0.8	0.2
	2006	0.1	0.2	0.1	0.7	0.1
	2007	0.1	0.2	0.0	0.8	0.0
	2008	0.1	0.1	0.0	0.8	0.1
Sombrero Shallow	1996	1.8	0.5	0.4	0.9	0.7
24.6259 °N	1997	1.3	0.4	0.2	0.6	0.9
81.1101 °W	1998	1.3	0.4	0.6	0.6	0.8
	1999	0.9	0.3	0.1	0.8	0.5
	2000	0.8	0.3	0.0	0.4	0.6
	2001	0.7	0.3	0.0	0.6	0.6
	2002	0.9	0.3	0.0	1.0	0.5
	2003	0.8	0.2	0.1	0.6	0.5
	2004	0.3	0.3	0.1	0.7	0.5
	2005	0.5	0.2	0.1	0.7	0.5
	2006	0.3	0.2	0.1	0.8	0.3
	2007	0.2	0.2	0.0	0.7	0.3
	2008	0.2	0.2	0.0	0.9	0.4
Tennessee Deep	1996	2.4	0.9	0.1	0.8	0.3
24.7527 °N	1997	2.6	0.2	0.1	0.6	0.4
80.7578 °W	1998	2.6	0.4	0.0	0.4	0.5
	1999	2.9	0.2	0.0	0.6	0.3
	2000	2.9	0.5	0.0	0.7	0.4
	2001	2.1	0.4	0.0	0.7	0.5
	2002	2.1	0.2	0.0	0.7	0.5
	2003	2.5	0.2	0.0	0.6	0.3
	2004	1.6	0.2	0.1	0.4	0.2
	2005	1.9	0.4	0.0	0.7	0.4
	2006	1.9	0.3	0.0	1.2	0.3
	2007	2.0	0.0	0.1	0.5	0.3
	2008	1.5	0.1	0.0	0.8	0.3
Tennessee Shallow	1996	0.1	0.2	0.1	0.1	0.2
24.7450 °N	1997	0.1	0.2	0.0	0.1	0.4
80.7812 °W	1998	0.2	0.3	0.0	0.0	0.2
	1999	0.0	0.2	0.1	0.1	0.1
	2000	0.1	0.2	0.1	0.1	0.2
	2001	0.1	0.3	0.1	0.1	0.1
	2002	0.1	0.2	0.1	0.2	0.1
	2003	0.0	0.1	0.1	0.1	0.0
	2004	0.1	0.2	0.1	0.2	0.0
	2005	0.0	0.1	0.0	0.2	0.1
	2006	0.1	0.0	0.0	0.2	0.1
	2007	0.0	0.1	0.1	0.1	0.1
	2008	0.1	0.1	0.0	0.1	0.1
Turtle	1996	0.9	0.5	0.0	1.1	1.3
25.2947 °N	1997	1.0	0.8	0.1	1.1	0.8
80.2191 °W	1998	0.9	0.5	0.0	1.4	1.0
	1999	1.0	0.5	0.1	0.9	0.7
	2000	1.0	0.3	0.4	1.1	1.0

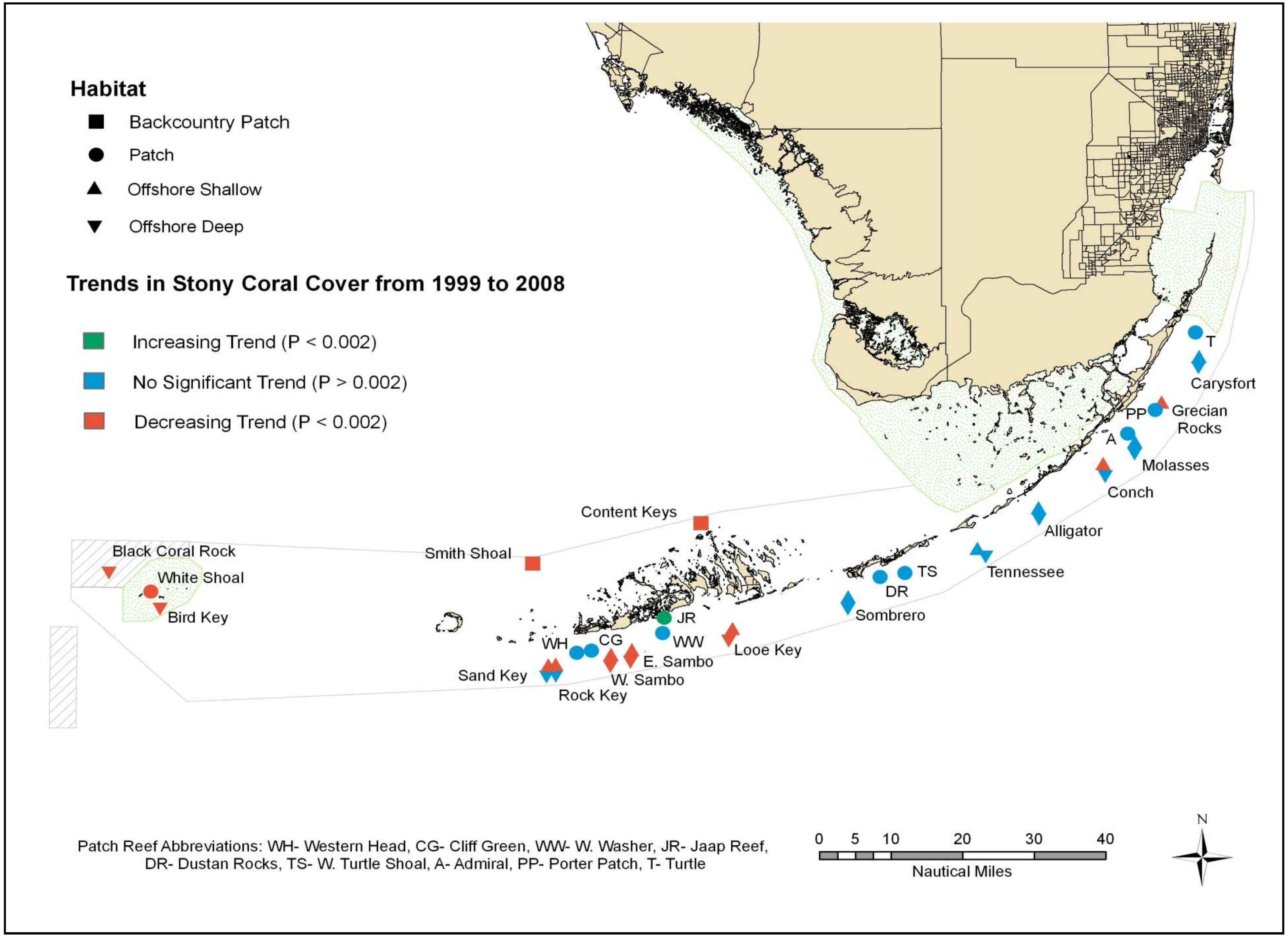
	2001	1.6	0.4	0.0	1.3	1.2
	2002	1.3	0.6	0.1	1.2	1.2
	2003	1.2	0.2	0.3	0.6	1.8
	2004	1.1	0.1	0.1	1.3	1.2
	2005	0.9	0.5	0.4	1.0	1.2
	2006	1.0	0.1	0.0	1.6	1.0
	2007	0.7	0.3	0.0	1.2	1.2
	2008	0.9	0.5	0.1	0.9	1.3
West Turtle Shoal	1996	0.7	4.4	1.8	3.1	0.2
24.6993 °N	1997	2.3	3.9	1.6	3.4	0.8
80.9669 °W	1998	0.8	4.2	2.7	3.2	0.4
	1999	0.8	5.3	1.6	2.3	0.6
	2000	0.4	4.3	1.7	3.9	0.9
	2001	0.9	4.2	2.2	3.9	0.9
	2002	0.8	4.9	2.7	3.8	1.1
	2003	0.5	4.4	2.7	3.1	0.6
	2004	0.6	3.6	2.1	3.1	0.6
	2005	1.0	4.3	2.8	3.0	0.7
	2006	0.6	3.7	2.3	3.1	0.8
	2007	0.4	4.0	2.7	3.3	0.8
	2008	0.5	3.5	2.1	3.3	0.6
West Washer Women	1996	3.4	10.6	6.8	7.9	0.6
24.5475 °N	1997	5.9	5.0	2.7	6.2	0.4
81.5866 °W	1998	3.2	8.2	6.0	7.1	0.8
	1999	4.0	6.4	4.7	6.8	0.5
	2000	3.9	7.0	5.1	6.6	1.0
	2001	4.1	7.2	3.5	6.1	0.6
	2002	4.4	8.7	6.1	7.4	1.0
	2003	2.9	6.8	5.2	4.9	1.1
	2004	2.8	6.6	5.7	6.7	0.9
	2005	3.4	7.1	5.0	6.9	0.6
	2006	4.0	7.9	4.4	7.5	0.9
	2007	4.5	7.4	5.1	6.7	0.8
	2008	3.0	7.4	3.0	5.3	0.6
Western Head	1996	3.1	14.1	2.4	3.3	0.2
24.4977 °N	1997	1.8	12.5	3.5	2.9	0.2
81.8056 °W	1998	2.7	14.3	2.6	3.6	0.3
	1999	1.7	12.2	2.3	2.4	0.0
	2000	2.2	14.8	1.8	2.7	0.2
	2001	1.3	14.7	2.0	2.9	0.1
	2002	1.3	13.8	2.2	3.1	0.7
	2003	1.8	14.4	2.4	2.9	0.2
	2004	0.8	13.0	2.8	2.7	0.3
	2005	2.2	14.2	2.9	2.6	0.2
	2006	1.9	12.8	3.2	1.8	0.3
	2007	0.9	12.8	3.5	2.5	0.3
	2008	1.3	12.7	2.0	3.0	0.2
Western Sambo Deep	1996	2.4	0.0	0.0	0.5	0.4

24.4785 °N	1997	2.4	0.0	0.1	0.5	0.3
81.7161 °W	1998	2.1	0.0	0.1	0.5	0.3
	1999	2.1	0.1	0.1	0.5	0.2
	2000	2.1	0.0	0.1	0.4	0.1
	2001	1.9	0.0	0.1	0.5	0.3
	2002	1.7	0.2	0.0	0.3	0.3
	2003	1.6	0.0	0.2	0.4	0.3
	2004	1.0	0.0	0.1	0.3	0.2
	2005	1.5	0.1	0.1	0.3	0.1
	2006	0.7	0.0	0.0	0.1	0.0
	2007	0.7	0.0	0.0	0.3	0.1
	2008	0.9	0.0	0.0	0.4	0.1
Western Sambo Shallow	1996	1.7	0.1	0.3	0.4	1.8
	1997	1.7	0.0	0.1	0.2	2.1
24.4797 °N	1998	1.5	0.0	0.4	0.5	2.1
81.7170 °W	1999	1.0	0.1	0.0	0.4	2.2
	2000	1.2	0.0	0.1	0.3	2.5
	2001	1.0	0.1	0.2	0.2	2.1
	2002	1.1	0.0	0.1	0.2	2.6
	2003	1.1	0.1	0.1	0.3	2.7
	2004	1.6	0.0	0.1	0.3	1.9
	2005	1.2	0.0	0.1	0.2	2.0
	2006	0.8	0.0	0.0	0.3	1.6
	2007	0.6	0.1	0.0	0.3	1.5
	2008	0.5	0.0	0.0	0.4	1.3
White Shoal	1999	0.2	0.1	0.3	0.4	0.1
24.6416 °N	2000	0.3	0.1	0.2	0.3	0.2
82.8964 °W	2001	0.4	0.1	0.5	0.4	0.0
	2002	0.3	0.1	0.3	0.1	0.0
	2003	0.4	0.1	0.2	0.2	0.1
	2004	0.2	0.1	0.2	0.3	0.0
	2005	0.3	0.1	0.1	0.2	0.1
	2006	0.2	0.0	0.1	0.2	0.0
	2007	0.1	0.1	0.1	0.2	0.1
	2008	0.2	0.0	0.3	0.2	0.1

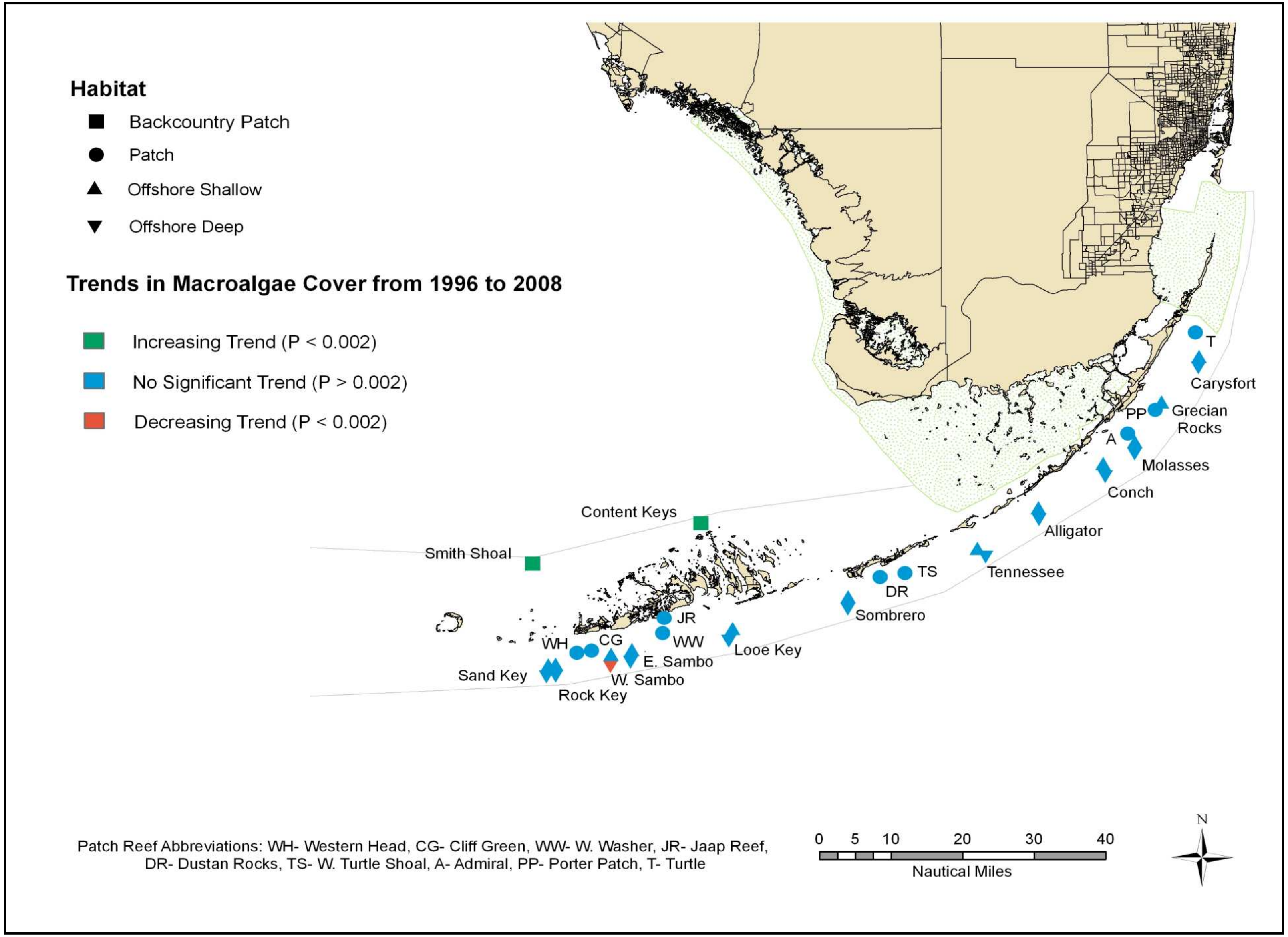
Appendix 5. Long-term trends in stony coral cover 1996 to 2008.



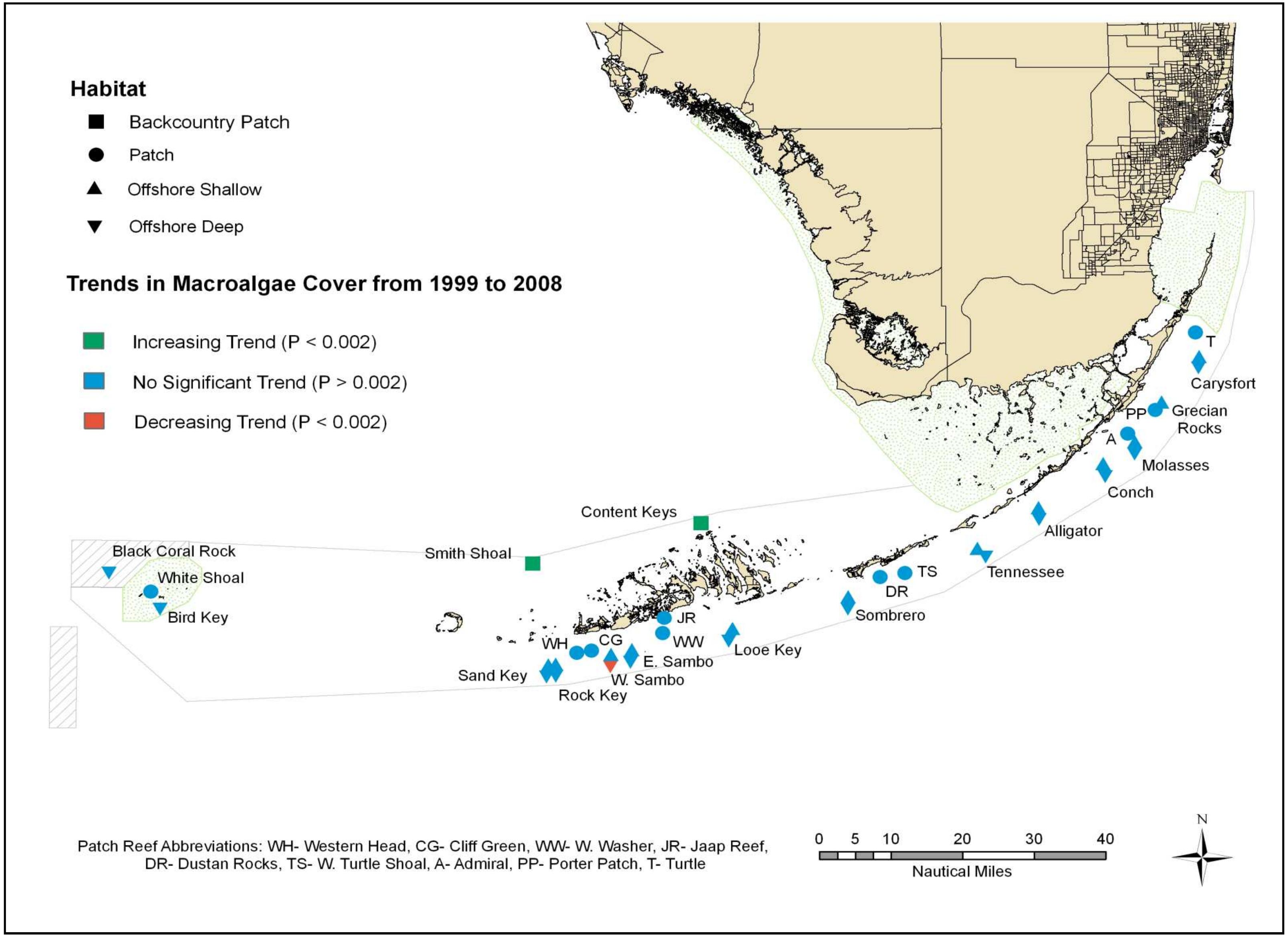
Appendix 6. Long-term trends in stony coral cover from 1999 to 2008.



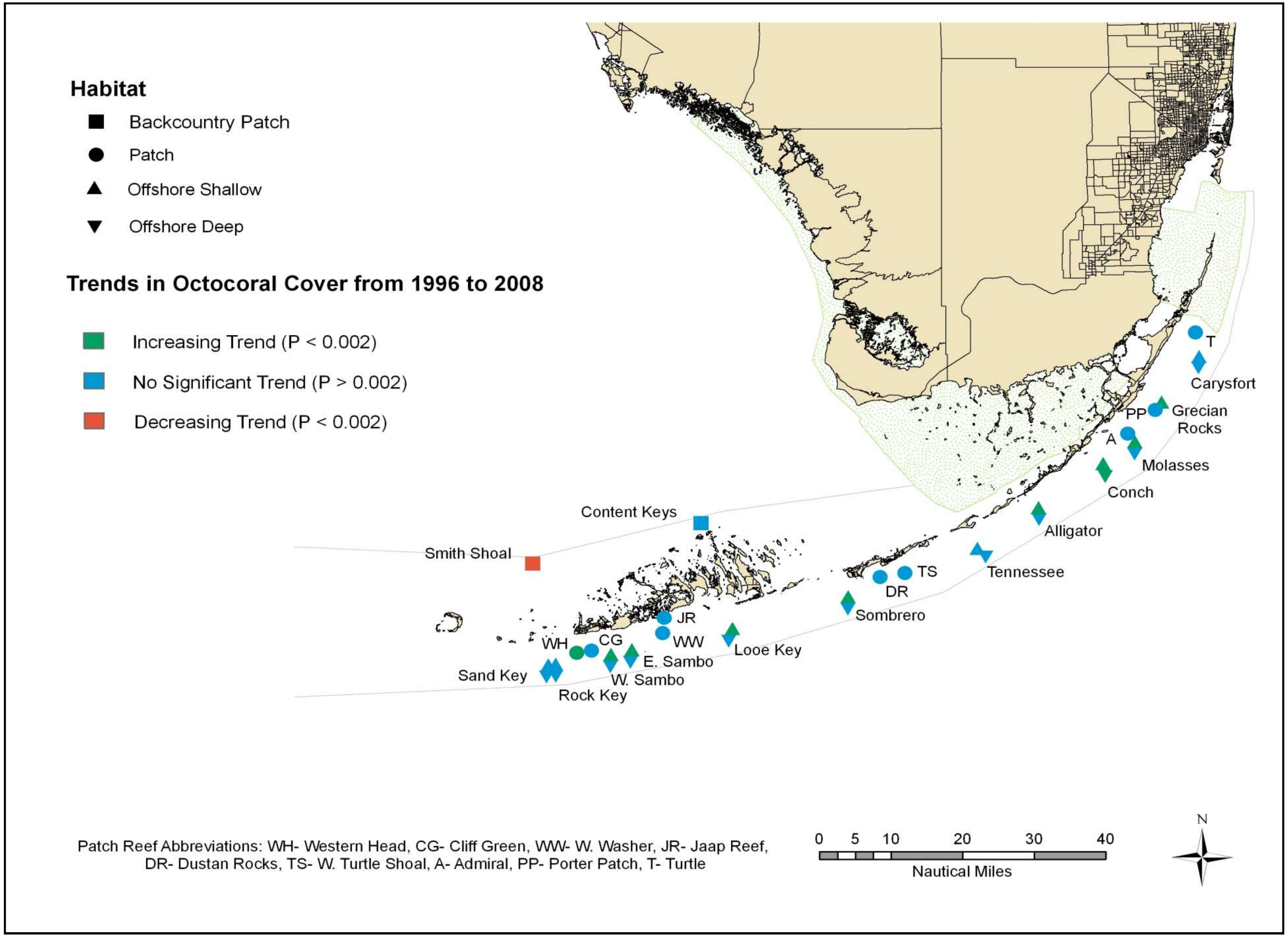
Appendix 7. Long-term trends in macroalgal cover from 1996 to 2008.



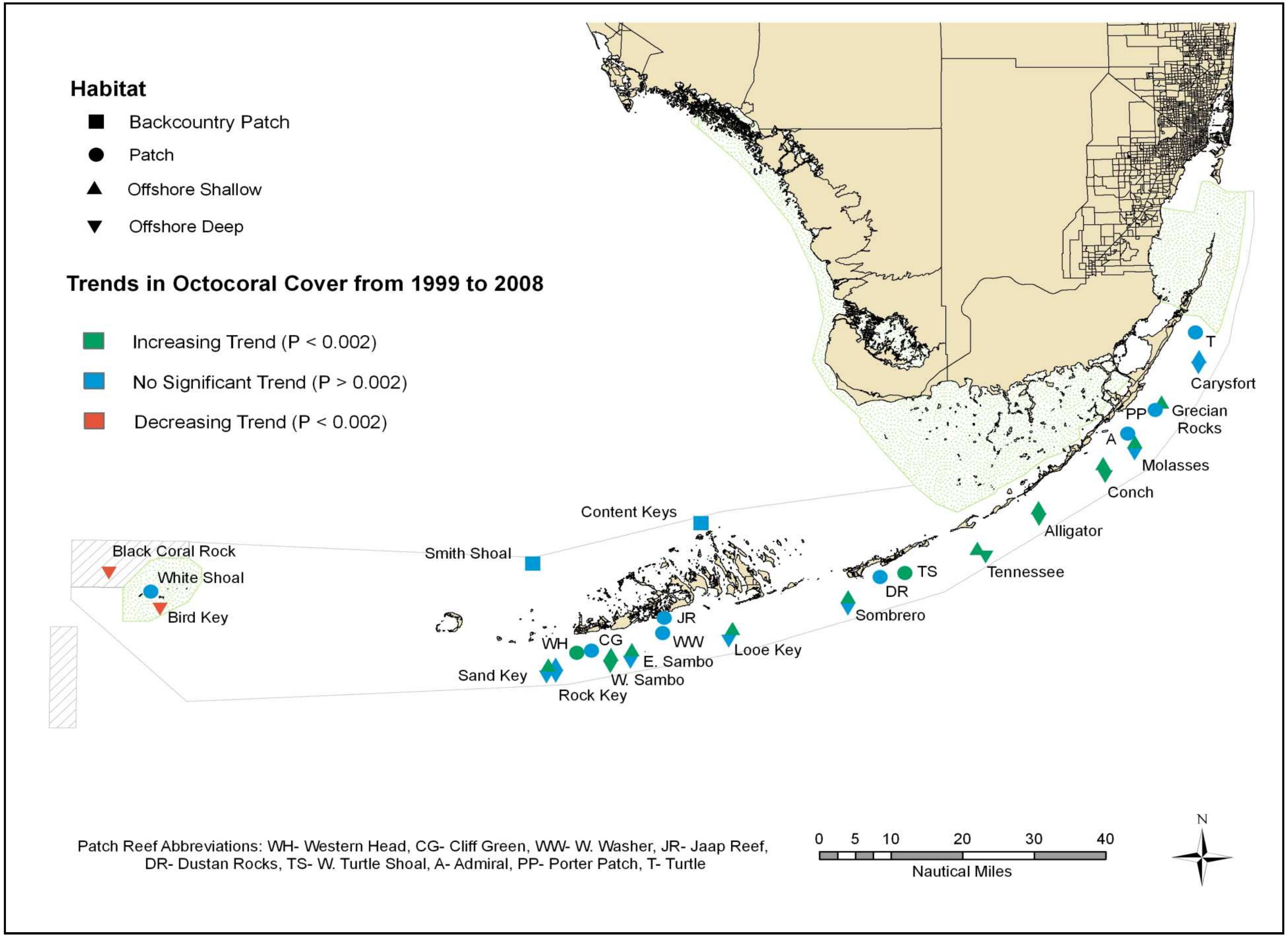
Appendix 8. Long-term trends in macroalgal cover from 1999 to 2008.



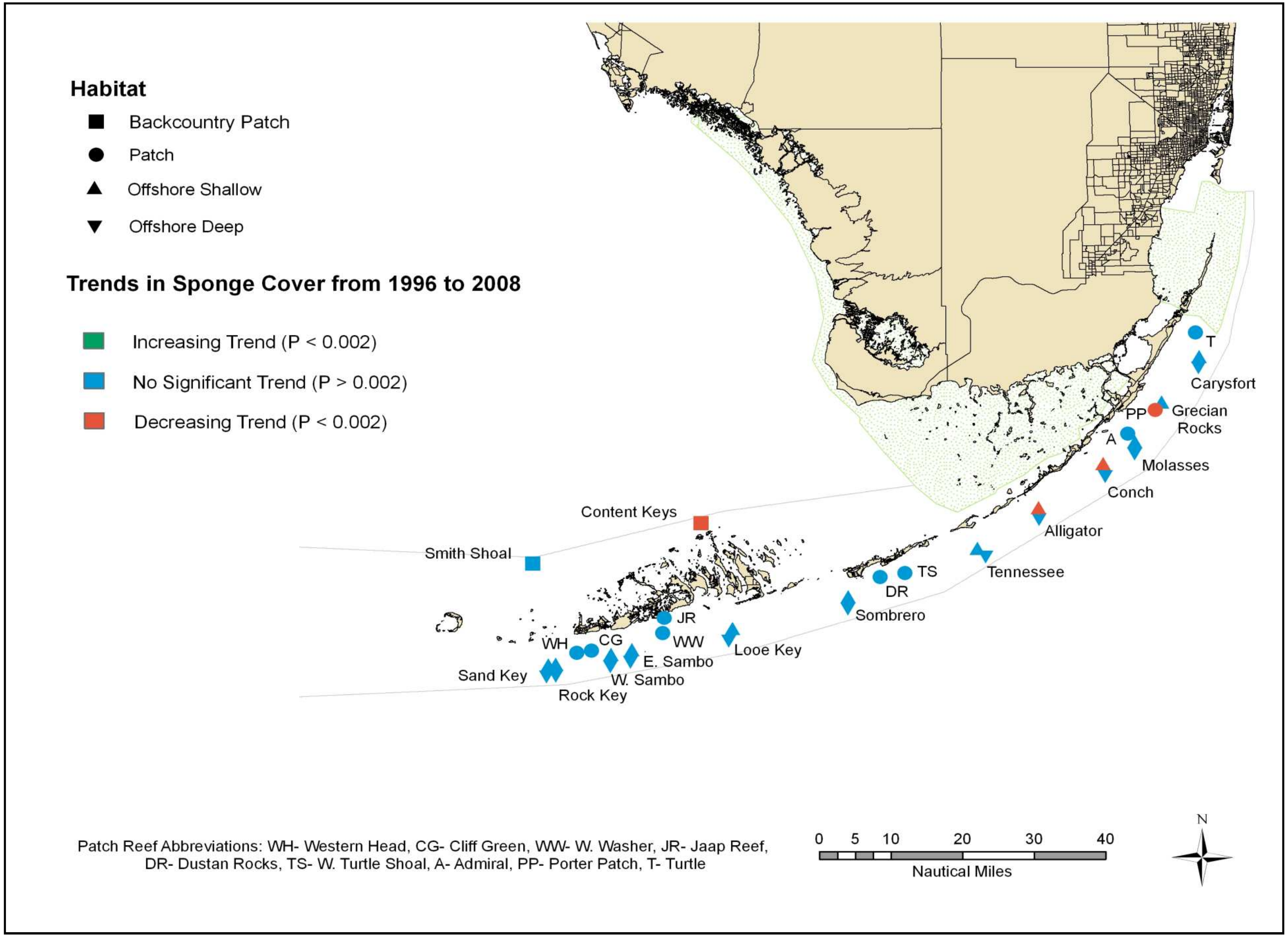
Appendix 9. Long-term trends in octocoral cover from 1996 to 2008.



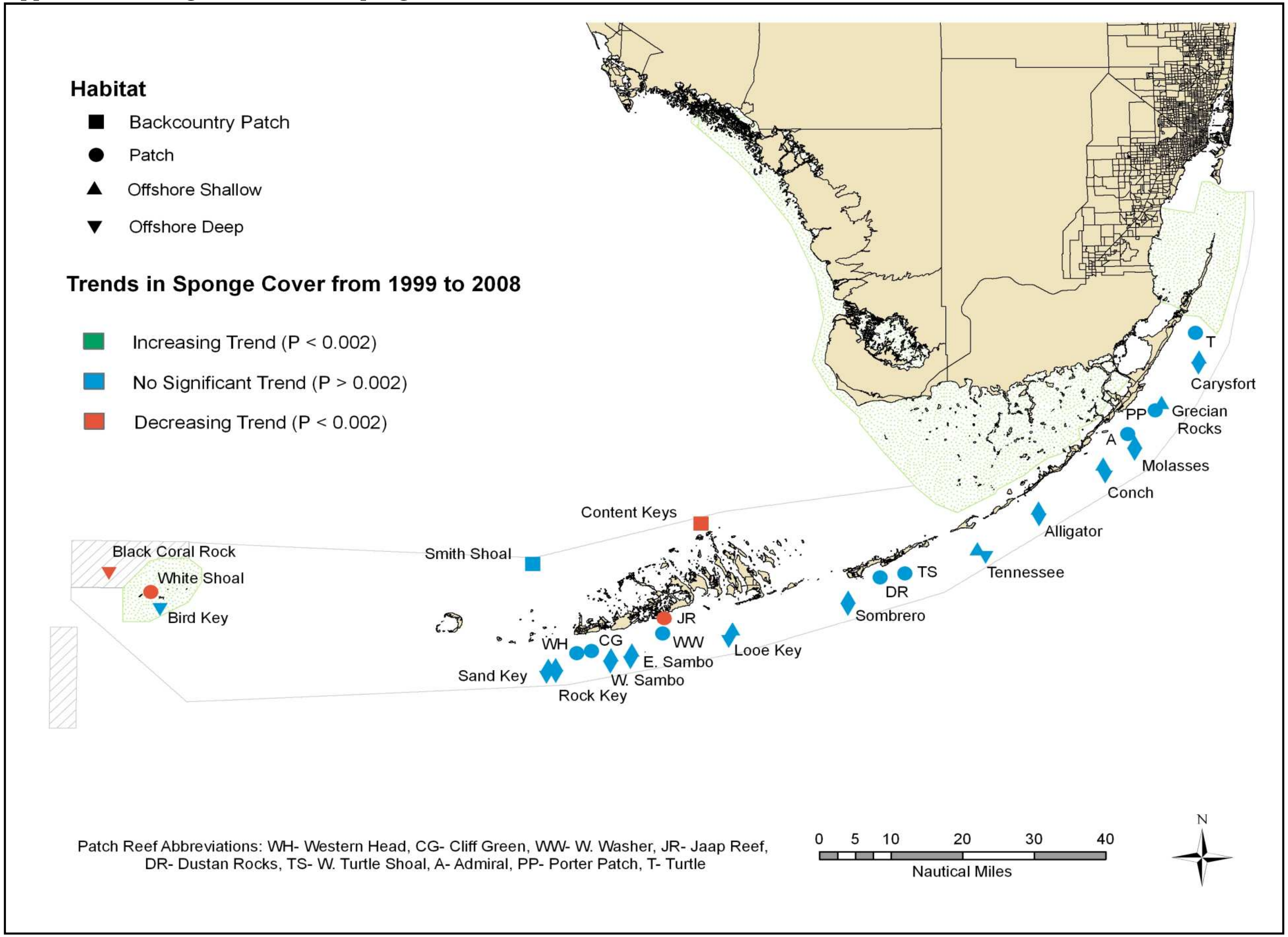
Appendix 10. Long-term trends in octocoral cover from 1999 to 2008.



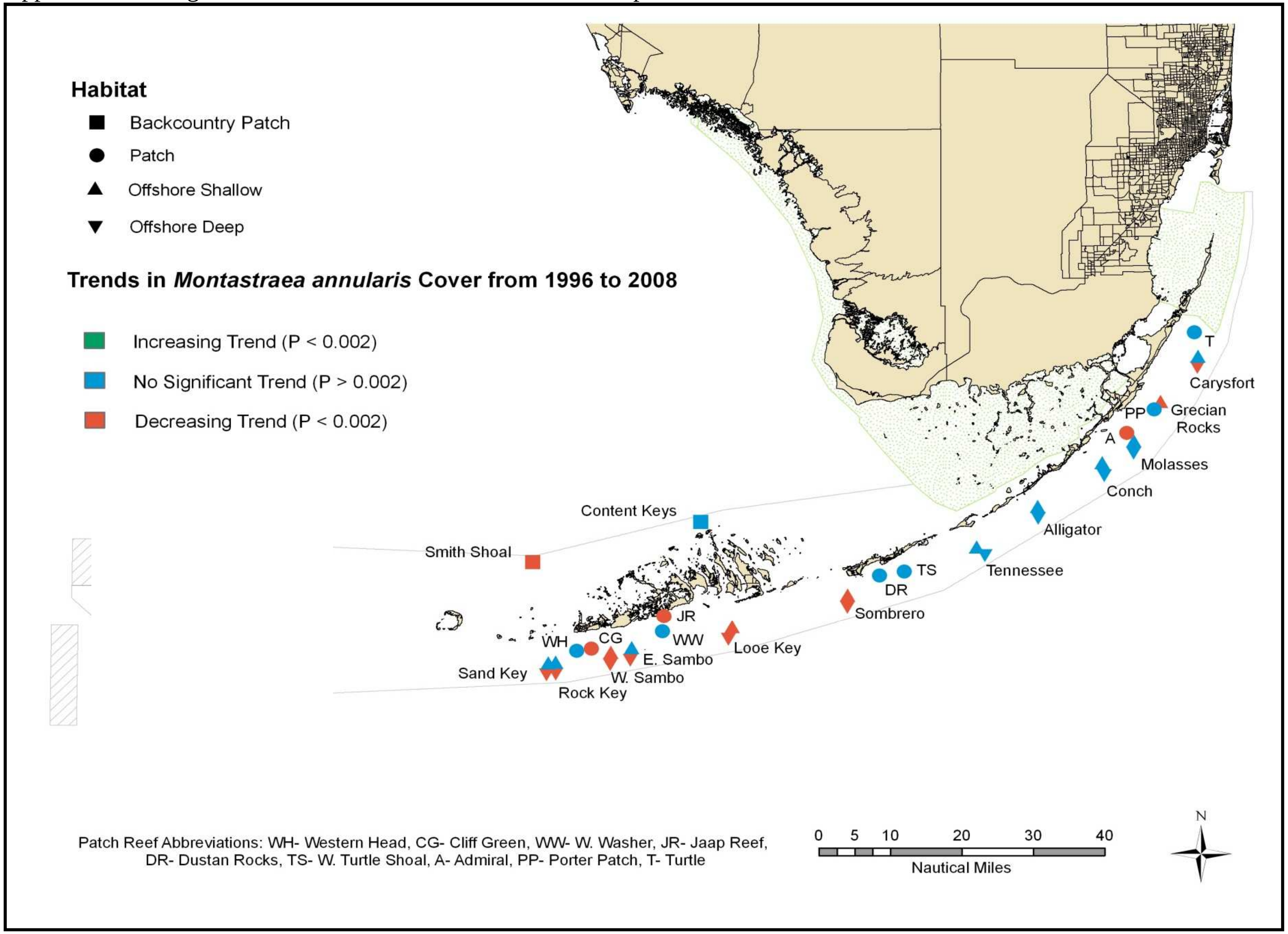
Appendix 11. Long-term trends in sponge cover from 1996 to 2008.



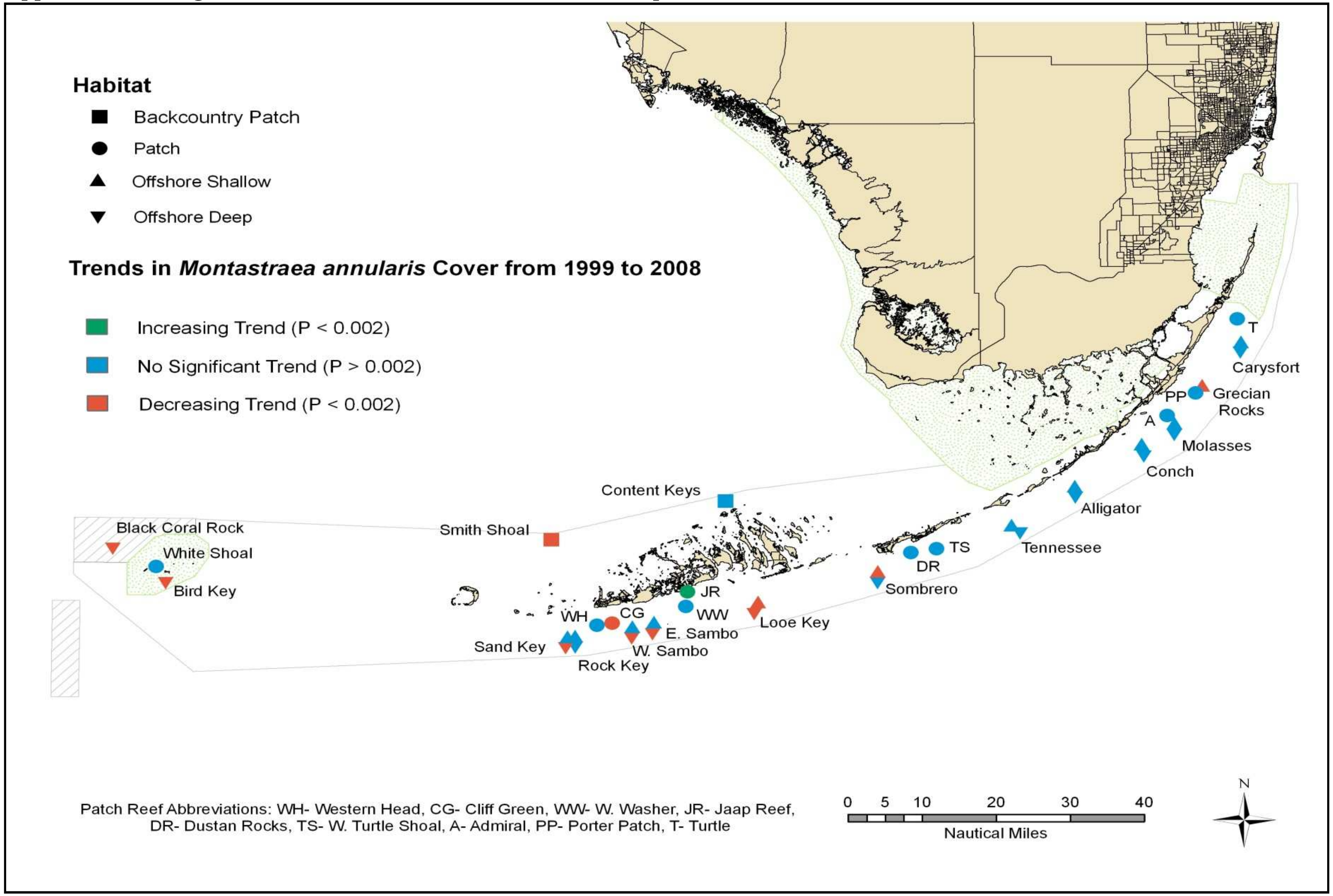
Appendix 12. Long-term trends in sponge cover from 1999 to 2008.



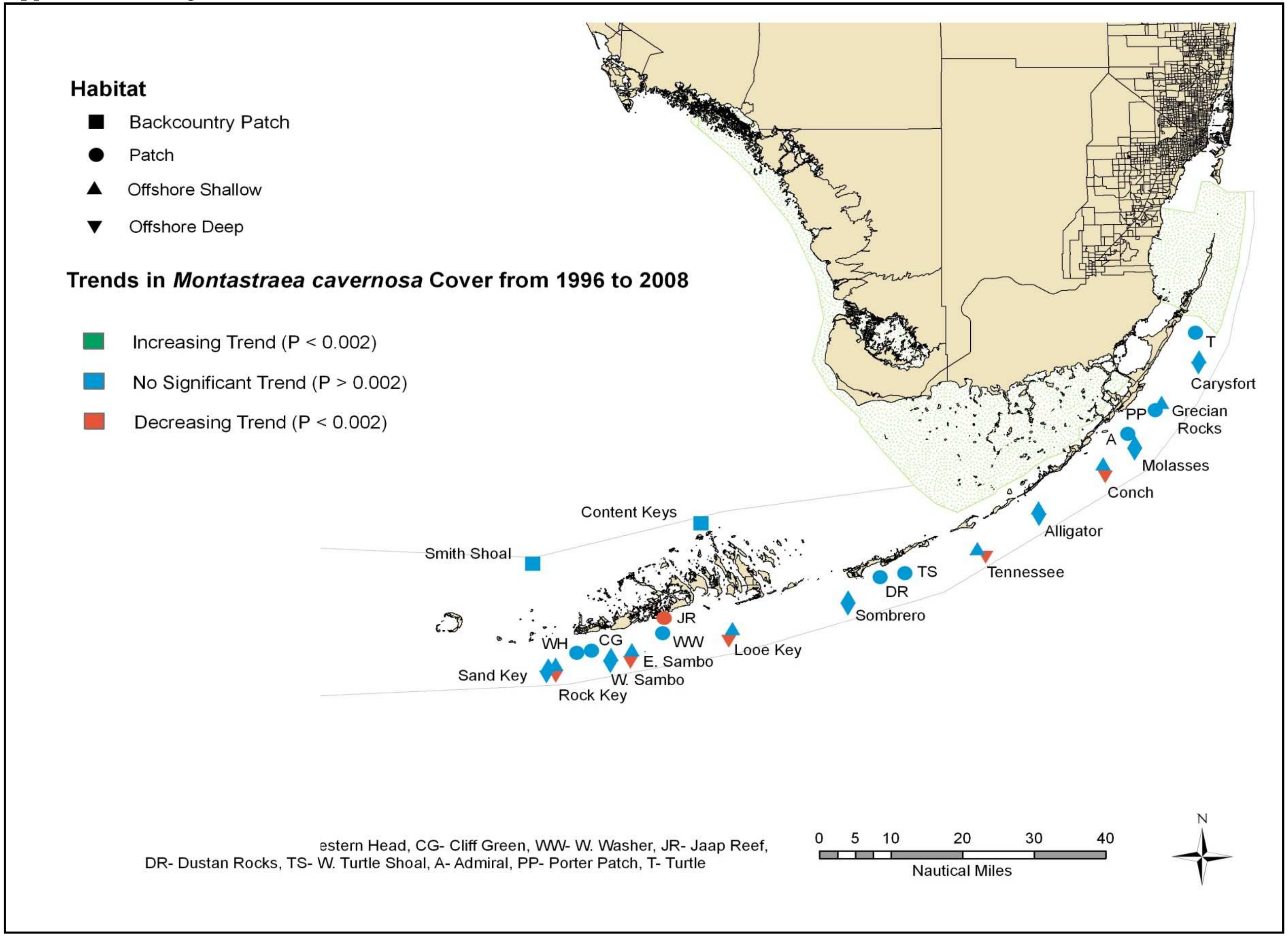
Appendix 13. Long-term trends in *Montastraea annularis* complex cover from 1996 to 2008.



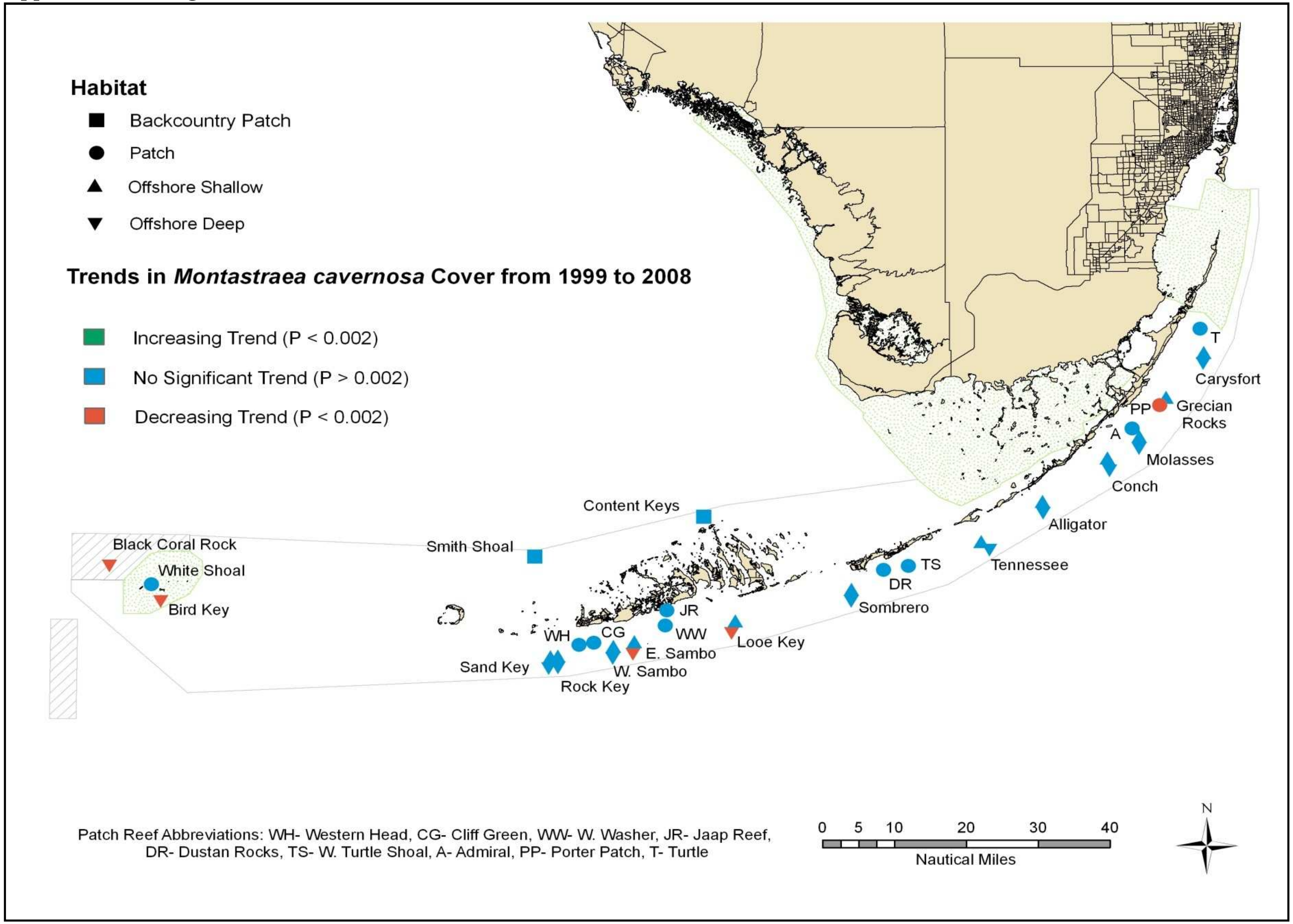
Appendix 14. Long-term trends in *Montastraea annularis* complex cover from 1999 to 2008.



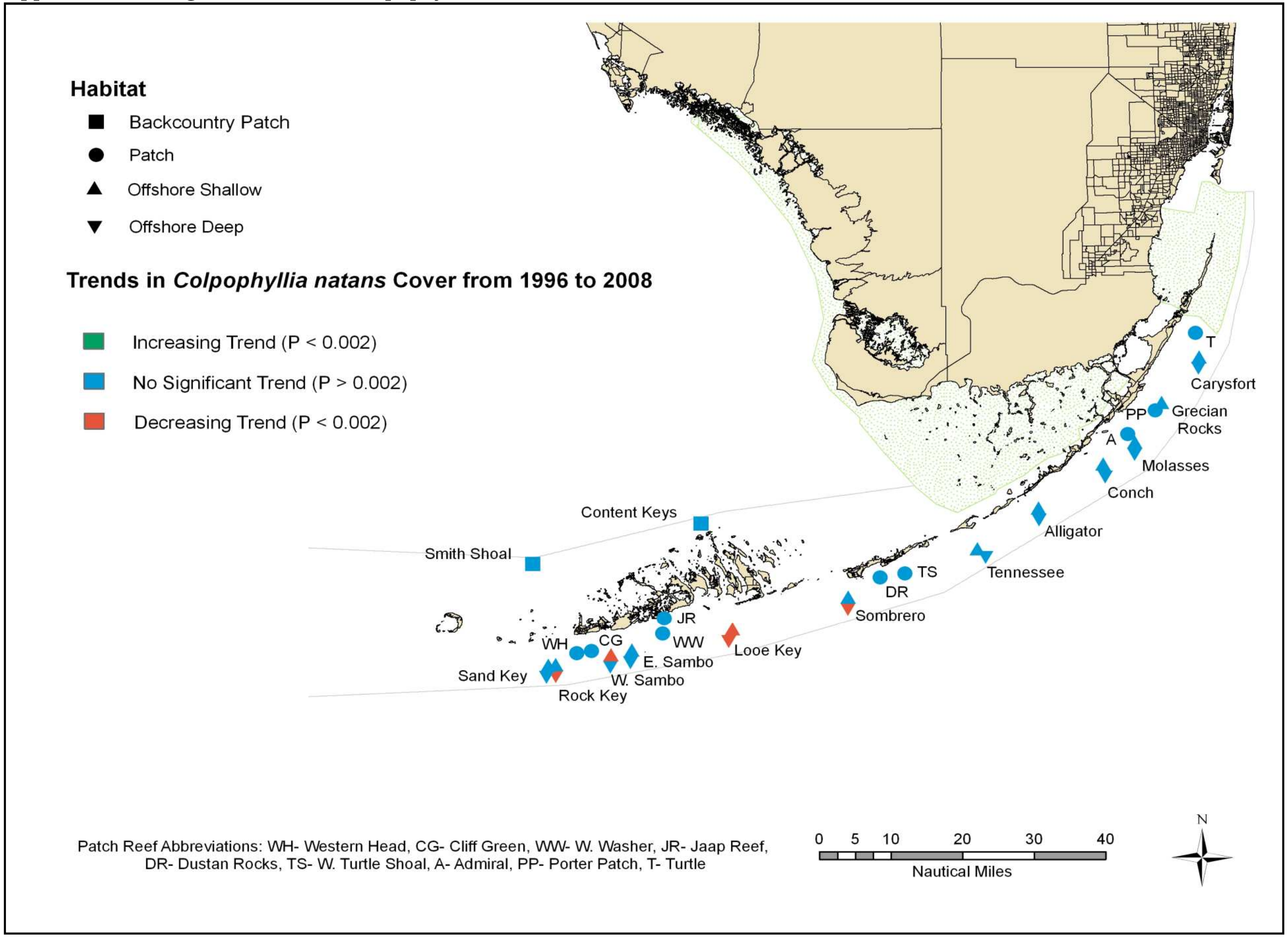
Appendix 15. Long-term trends in *Montastraea cavernosa* cover from 1996 to 2008.



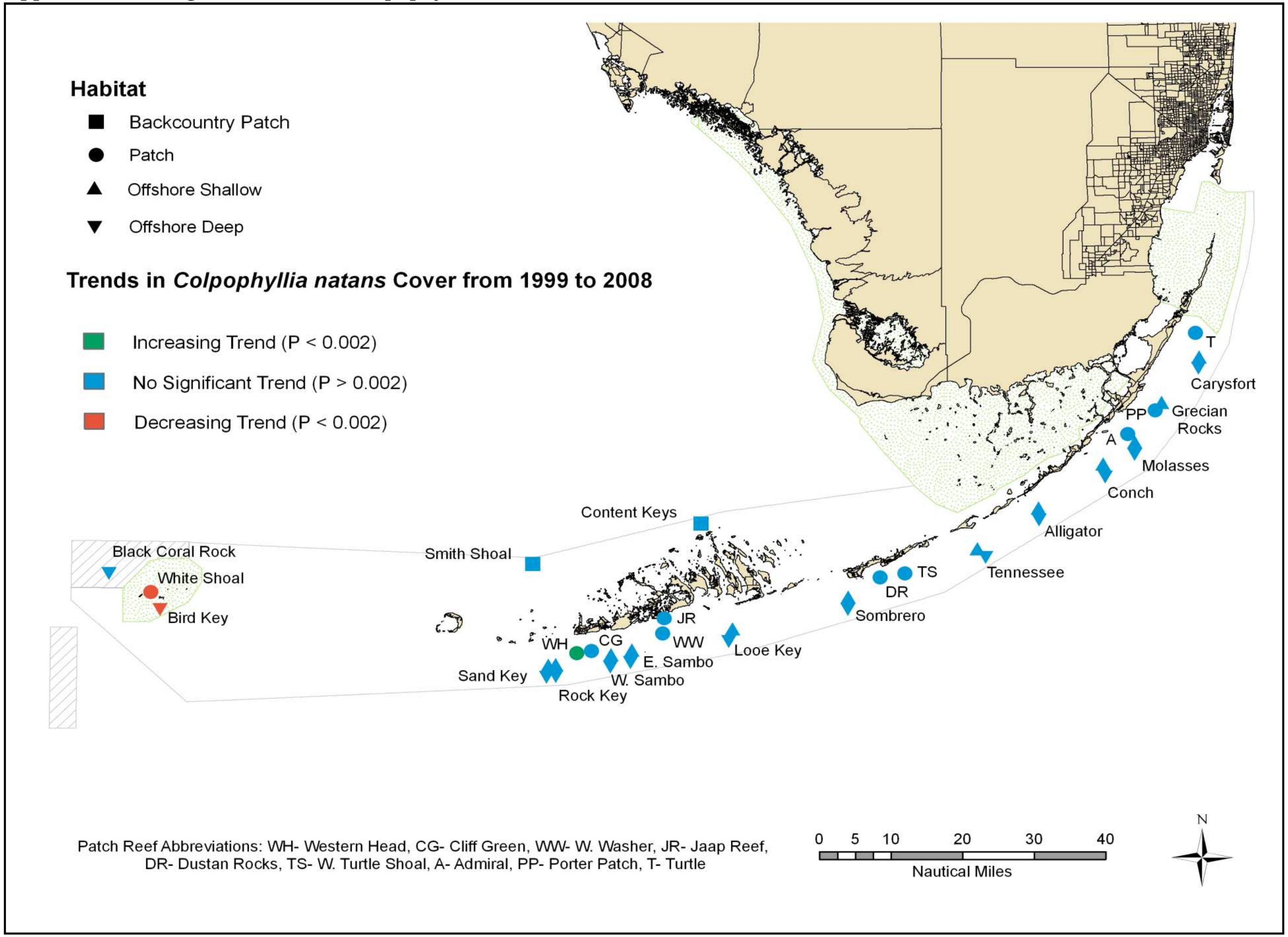
Appendix 16. Long-term trends in *Montastraea cavernosa* cover from 1999 to 2008.



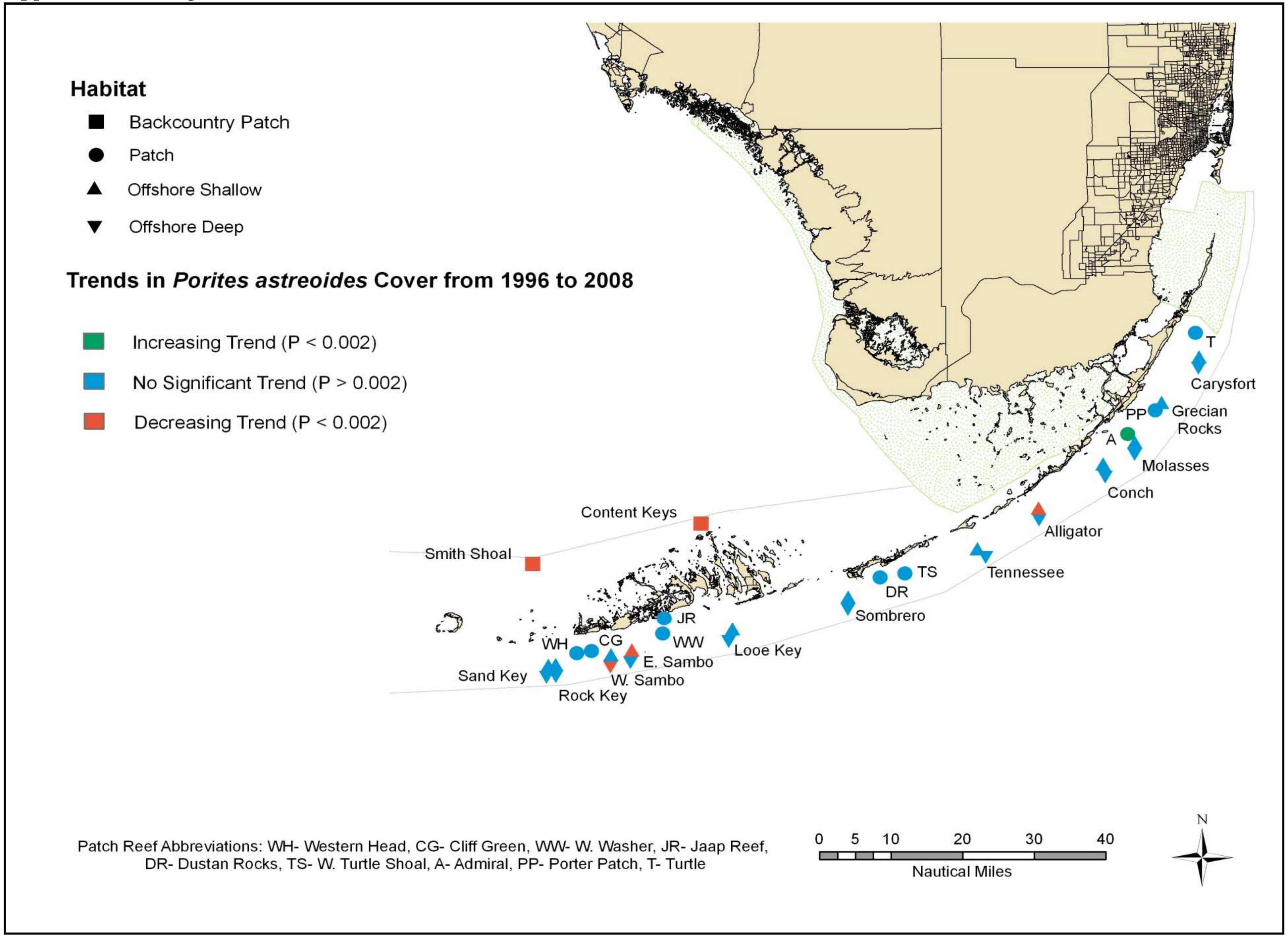
Appendix 17. Long-term trends in *Colpophyllia natans* cover from 1996 to 2008.



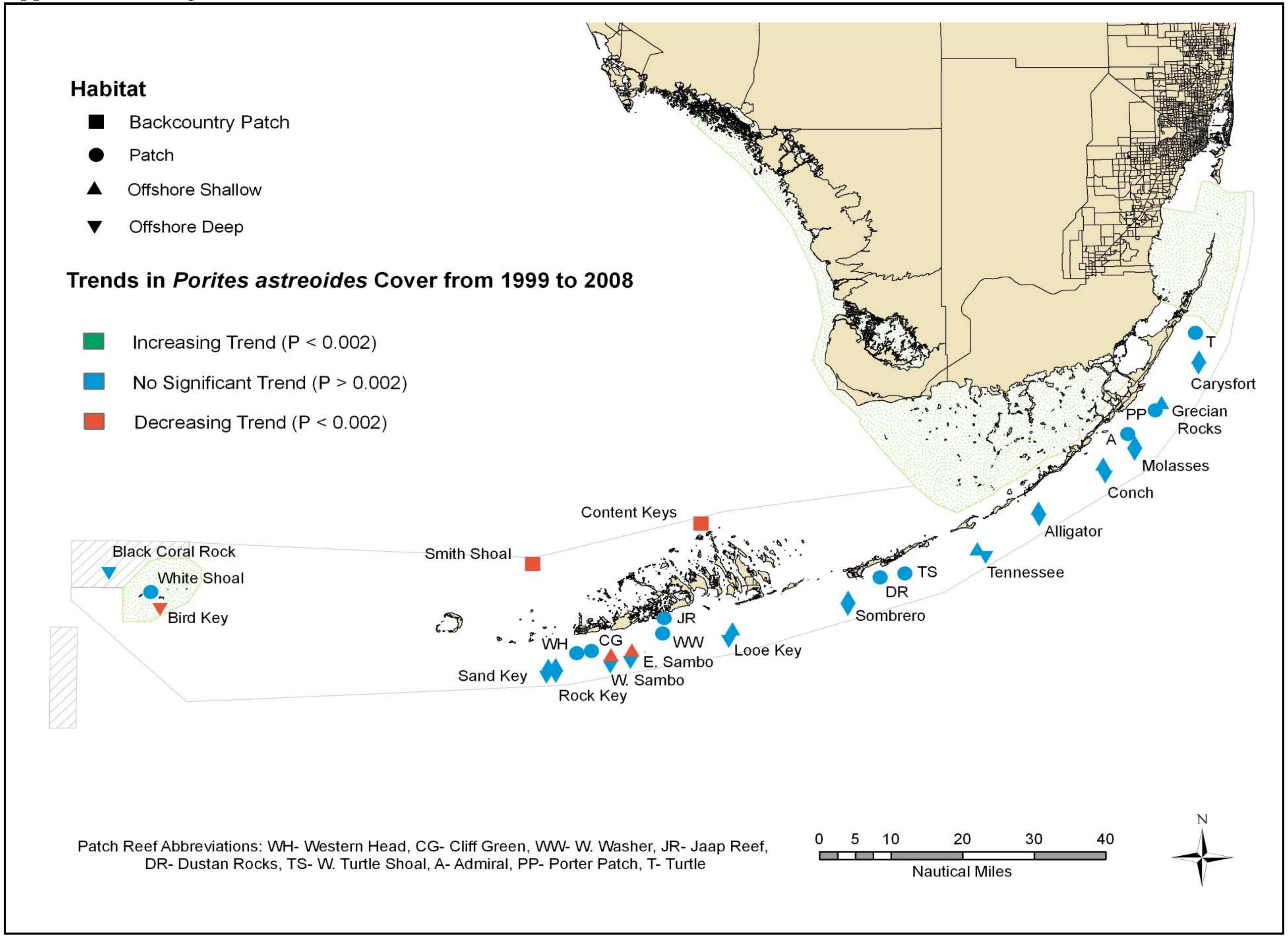
Appendix 18. Long-term trends in *Colpophyllia natans* cover from 1999 to 2008.



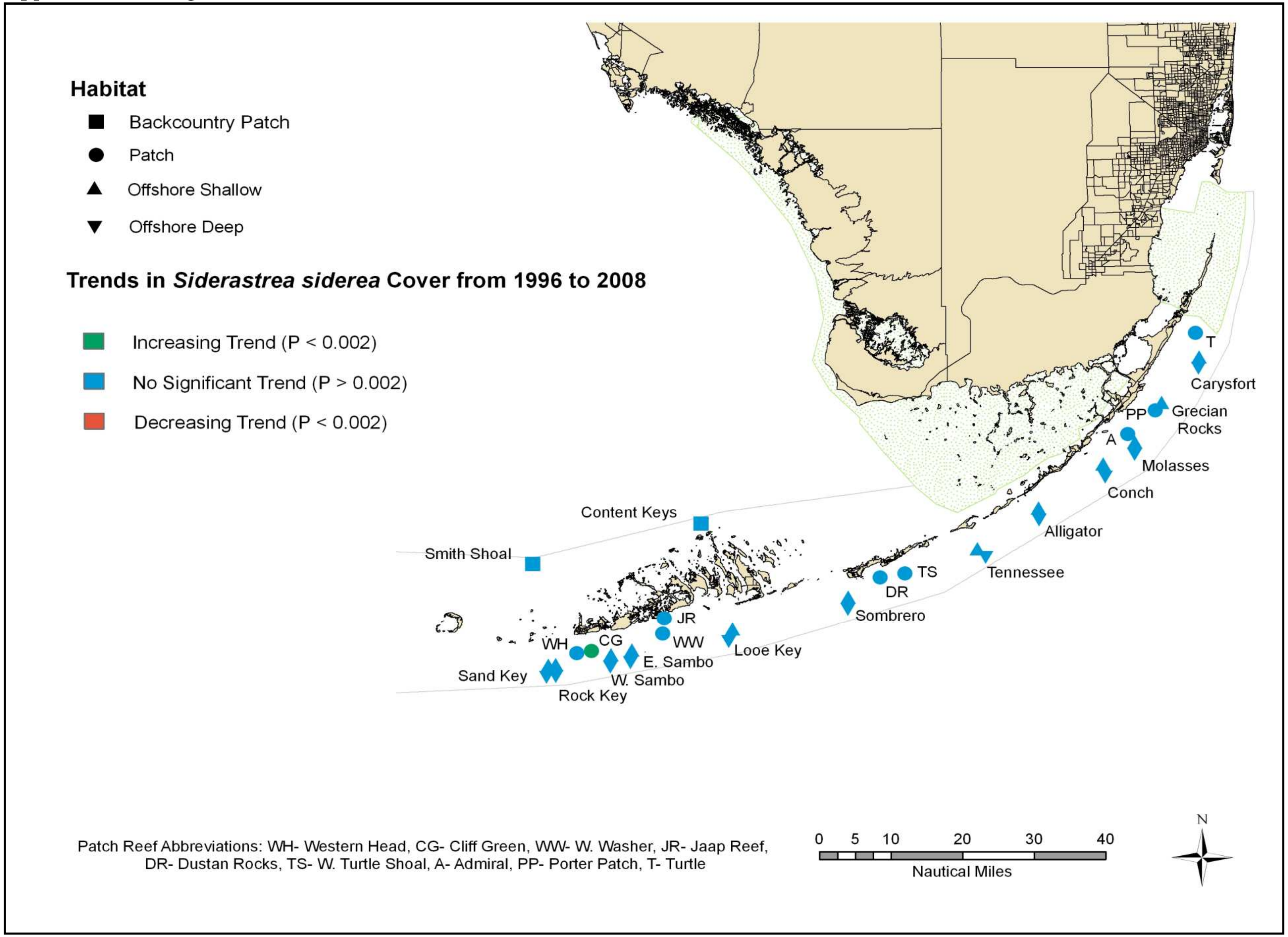
Appendix 19. Long-term trends in *Porites astreoides* cover from 1996 to 2008.



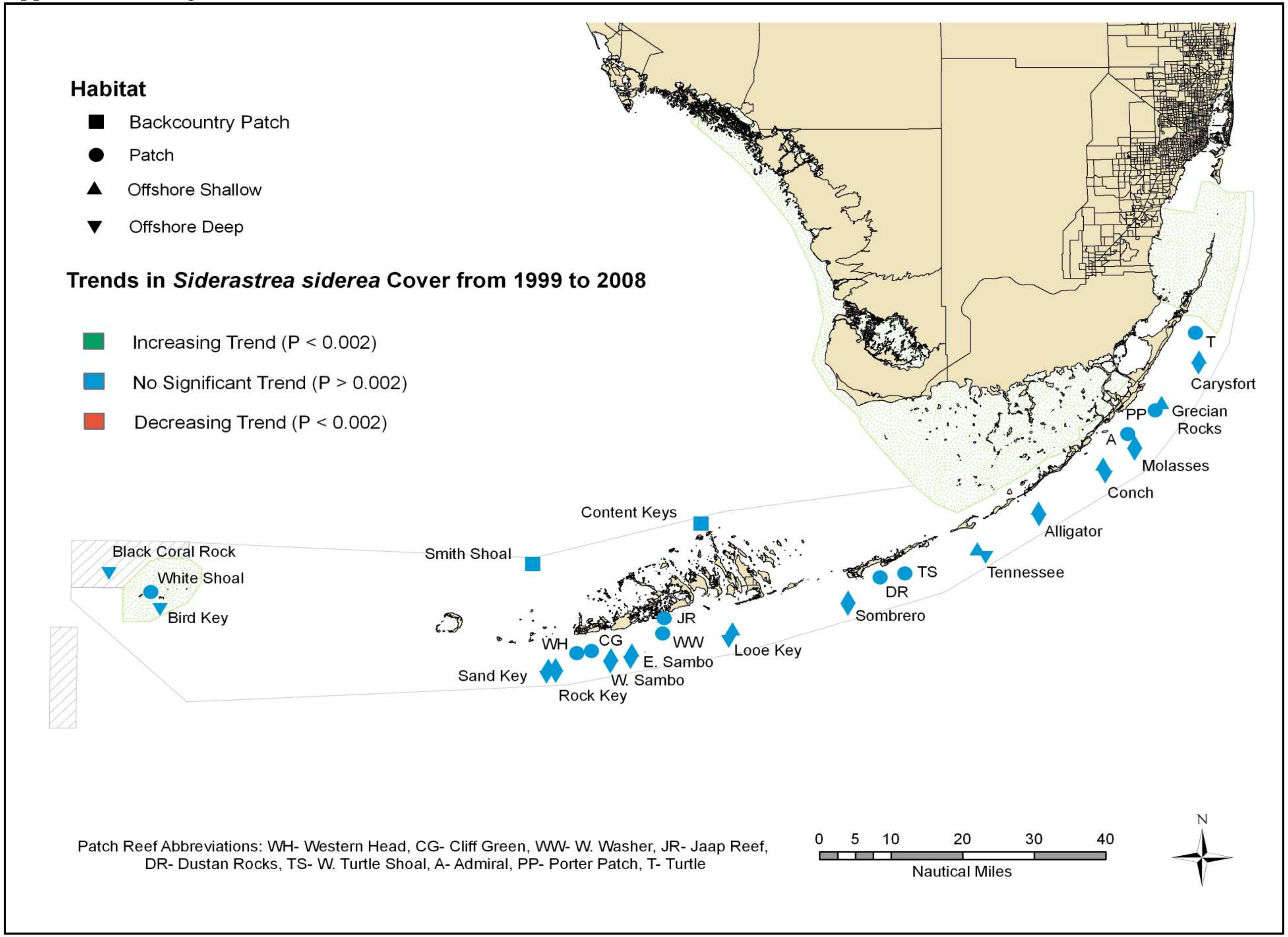
Appendix 20. Long-term trends in *Porites astreoides* cover from 1999 to 2008.



Appendix 21. Long-term trends in *Siderastrea siderea* cover from 1996 to 2008.



Appendix 22. Long-term trends in *Siderastrea siderea* cover from 1999 to 2008.



Appendix 23. Long-term trends in *Cliona delitrix* cover from 2001 to 2008.