4-1-2013

Florida Keys Population Abundance Estimates for Nine Coral Species Proposed for Listing Under the U.S. Endangered Species Act

Steven Miller
Nova Southeastern University, smiller@nova.edu

William F. Precht
Dial Cordy and Associates Inc.

Leanne M. Rutten
Nova Southeastern University

Mark Chiappone
Nova Southeastern University, mc191@nova.edu

Florida Keys Population Abundance Estimates for Nine Coral Species Proposed for Listing Under the U.S. Endangered Species Act

April 2013

Steven L. Miller, William F. Precht, Leanne M. Rutten and Mark Chiappone

Nova Southeastern University Oceanographic Center

Technical Series Report (submitted)

Florida Keys Population Abundance Estimates for Nine Coral Species Proposed for Listing Under the U.S. Endangered Species Act

Steven L. Miller[1], William F. Precht[2], Leanne M. Rutten[1] and Mark Chiappone[1]

Nova Southeastern University Oceanographic Center[1] 8000 North Ocean Drive, Dania Beach, FL 33037 USA. [2]Dial Cordy and Associates, Inc., Miami Lakes, Florida, USA

Suggested Citation

Miller SL, Precht WF, Rutten LM, Chiappone M (2013) Florida Keys Population Abundance Estimates for Nine Coral Species Proposed for Listing Under the U.S. Endangered Species Act. Technical Series Report. Nova Southeastern University Oceanographic Center, Technical Series Report, 1(1). Dania Beach, FL. 85 pp.

Acknowledgments

Field surveys that provided the data used in this report were supported by NOAA's National Marine Sanctuary and Coral Reef Conservation Programs, the U.S. Department of Commerce's 1535 Endangered Species Act Projects, and NOAA's National Undersea Research Program at the University of North Carolina at Wilmington. Logistical support was provided by: NOAA's Aquarius Reef Base through the University of North Carolina at Wilmington, NOAA/Florida Keys National Marine Sanctuary, National Park Service/Biscayne National Park, the Keys Marine Lab (FIO/USF); the South Florida and Caribbean Monitoring Network of the National Park Service; and Nova Southeastern University's Oceanographic Center. J. Ault and S. Smith of RSMAS-University of Miami assisted with the sampling design and site allocations. The most recent field surveys in the Florida Keys in the Florida Keys National Marine Sanctuary and Biscayne National Park were conducted under Permits FKNMS-2010-077 and National Park Service/BNP BISC-2012-SCI-0011, respectively.

Dedication

This report is dedicated to the memory of our friend and colleague Dr. Brian Keller, who helped direct, fund, and inspire the work presented in this report. His vision, insights, and support were instrumental in sustaining our work.

<u>Cover photo</u>. A subset of the coral species in the Florida Keys petitioned under the U.S. Endangered Species Act. Upper left: *Acropora palmata*, Upper right: *Agaricia lamarcki*, Lower left: *Montastraea faveolata*, Lower right: *Mycetophyllia ferox*.

Contents

Executive Summary

This report presents abundance and size-class distribution estimates for nine coral species in the Florida Keys and Dry Tortugas, all of which are proposed for listing or reclassification under the U.S. Endangered Species Act (ESA). The lack of population data for these species was highlighted as a deficiency in the ESA Review Process by the Biological Review Team (BRT) in their Status Review (Brainard et al. 2011) and also by the National Oceanic and Atmospheric Administration (NOAA) (Federal Register 2012). Field sampling protocols were adapted from Aronson et al. (1994) and the Atlantic and Gulf Rapid Reef Assessment program (Kramer and Lang 2003) to measure population-level metrics of scleractinian corals, with population data analyses following Smith et al. (2011). The data in this report are based on focused surveys for *Acropora* corals in the Florida Keys during 2005, 2007, and 2012; and for all scleractinian coral species during 2005, 2009, and 2012; and for all coral species in the Dry Tortugas for 2006 and 2008. Colony density within belt transects and size measurements were obtained for each species present. Statistical estimation procedures for population abundance metrics – means (e.g. coral density) and totals (e.g. coral abundance) – for a two-stage stratified random sampling design were adapted from Cochran (1977), and computations were carried out using SAS statistical software. Domain-wide estimates are presented in this report.

<u>Abundant and Common Species (6)</u>: *Acropora cervicornis, A. palmata, Dichocoenia stokesi, Montastraea annularis, M. faveolata, and M. franksi.*

Population estimates for *Acropora cervicornis* in the Florida Keys appear stable and large, ranking as high as 15th among all corals in the Florida Keys, with over 10 million colonies estimated in 2012. There is no evidence of continued decline since the 2006 ESA Threatened Listing. The size structure of the population also remains unchanged over the period of our study in the Florida Keys. Population estimates for the Dry Tortugas are smaller, with large variance terms. The presence of large population numbers in southeast Florida, plus *A. cervicornis* is known to contain genotypes resistant to white band disease, restoration activities are becoming increasingly effective, and there is increasing evidence that populations are recovering at multiple locations throughout the Caribbean, all suggests that the proposal to reclassify *A. cervicornis* to Endangered is not warranted.

Population estimates for *Acropora palmata* in the Florida Keys appear stable since 2005, but remain much reduced overall since declines started in the late 1970s. Relative to the abundance of other corals in the region, *A. palmata* is among the least abundant. The size class distribution of the Florida Keys

population includes both small and large individuals. Relative to *A. cervicornis*, the population status of *A. palmata* in south Florida is two-orders of magnitude smaller, with most of the population reduced to a handful of high-density thickets. This contrasts with the distribution of *A. cervicornis,* which is found throughout the Keys and in multiple habitat types. On a positive note, *A. palmata* has recently become a focus of coral restoration efforts, with increasingly large numbers growing and thriving in offshore nurseries and with successful transplants made to offshore reefs in the upper Keys. Because of large population declines throughout its range and its restricted shallow habitat distribution, we agree with the 2006 assessment to list this species as Threatened under the ESA (Hogarth 2006). Since 2006, however, this species has been relatively stable in Florida and there are no new data that warrants the reclassification of this species to Endangered.

Dichocoenia stokesi is among the most common corals in the Florida Keys and Dry Tortugas. Population abundance estimates approached 100 million colonies in 2005, with no trends apparent in the Florida Keys since then. In the Dry Tortugas, absolute numbers exceeded 12 million in 2006 (SE 4.1 million) and 7 million (SE 1.1 million) in 2008. The large population numbers, even after the White Plague Type II epidemic, its broad distribution among multiple habitat types, especially hard-bottom habitats, its high relative abundance among all corals in the region, and the presently low prevalence of White Plague Type II, all suggest that the proposed listing of *D. stokesi* to Threatened status is not warranted.

In the Florida Keys, *Montastraea annularis* is relatively common and was ranked in the middle among corals in terms of abundance in 2005 (30 out of 47), moving up significantly in 2009 to 13th out of 43, and 12th out of 40 in 2012. Population numbers in 2005 were 5.6 million (SE 1.7), with 11.5 million (SE 2.5 million) in 2009, and 24 million (SE 10.1 million) in 2012. No evidence of decline was observed in total population number. In the Dry Tortugas, *M. annularis* was ranked among the least common corals, near the bottom in 2006 (41 out of 43) and 2008 (31 out of 40). The larger number of *M. annularis* in the Florida Keys, exclusive of the Dry Tortugas, is related to the greater abundance of shallow patch reefs in the former area, where the species is most commonly found. This habitat type is uncommon in the Dry Tortugas. With over 6,000 patch reefs in the Florida Keys and the large number of corals present, listing this species as Endangered is not warranted.

In the Florida Keys, *Montastraea faveolata* is one of the top-ten most abundant scleractinian corals. Population estimates were 39.7 million (SE 8 million) in 2005, 21.9 million (SE 7 million) in 2009, and 47 million (SE 14.5 million) in 2012. The size-class distributions and partial mortality estimates for *M. faveolata* are similar among years. In the Dry Tortugas, *M. faveolata* ranked seventh most abundant in

2006 and fifth most abundant in 2008, with population numbers of 36.1 million (SE 20 million) and 30 million (SE 3.3 million), respectively. Size class distributions are similar to what was seen in the Florida Keys. With the large number of colonies present, especially in the smaller and medium size classes, and the wide distribution of the species in the region, among multiple habitat types and depths, listing of the species as Endangered is not warranted.

In the Florida Keys, *Montastraea franksi* is relatively common and typically found in deeper habitats than *M. faveolata* and *M. annularis*. The species is ranked in the middle, among all corals in the Florida Keys, ranging from 26th in 2005, to 32nd in 2009, and 33rd in 2012 (Figure 3-1). Absolute numbers for 2005 were 8 million (SE 2.2 million), for 2009 0.3 million (SE 214,000), and for 2012 0.4 million (SE 0.3 million). The apparent decline that occurred in 2009 and the similar value in 2012 are due to changes in the allocation scheme and logistics after 2005, where deeper sites were not surveyed. In the Dry Tortugas, *M. franksi* is one of the most common corals, ranking 4th in 2006 and 8th in 2008. Absolute population numbers in the Dry Tortugas are 79 million (SE 19 million) in 2006 and 18.1 million (SE 4.1 million). These population estimates document that *M. franksi* is relatively uncommon in shallower reef habitats through the Florida Keys, but common in deeper reef habitats. We have also seen *M. franksi* in patch reef habitats. With large population numbers, listing the species as Endangered is not warranted.

<u>Uncommon to Common Species (2)</u>: *Agaricia lamarcki* and *Mycetophyllia ferox.*

In the Florida Keys, *Agaricia lamarcki* ranked 35 out of 47 in 2005, it was absent from sampling in 2009, and it ranked 37th out of 40 in 2012. *Mycetophyllia ferox* ranked 39th out of 47 in 2005, 43rd out of 43 in 2009 and 40th out of 40 in 2012. Population estimates for *A. lamarcki* were 3.1 million (SE 1.0 million) in 2005, they were absent in 2007, and 0.2 million (SE 0.2 million) in 2012. This suggests a decline over the seven year period, but few deep sites were sampled in 2007 and 2012 and more work needs to be done to get a reliable population estimate. For *M. ferox*, the population estimates were 1.0 million (SE 0.5 million) in 2005, 9,500 (SE 9,500) in 2009, and 7,000 (SE 7,000) in 2012. The decline in 2009 and 2012 is explained similarly for *M. ferox*, based on sampling deeper coral reef habitats in 2005. The depth preference for these two species was evident in the Dry Tortugas, where we allocated more samples to deeper sites. Both species improved in their relative abundance ranking and populations numbers. For *A. lamarcki*, its ranking jumped to 12th out of 43 in 2006 and 22nd out of 40th in 2008. Populations estimates were 14.3 million (SE 2.6 million) in 2006 and 2.1 million (SE 0.5) in 2008. For *M. ferox,* its abundance ranking approved slightly, to 35th out of 43 in 2006 and 30th out of 40 in 2008. Population estimates were 0.9 million (SE 0.4 million) in 2006 and 0.5 million (SE 0.2 million) in 2008. While these two species are relatively uncommon in shallow habitats, their large population numbers in the deeper coral habitats of the Dry Tortugas do not warrant listing as Endangered under ESA.

<u>Rare Species (1)</u>: *Dendrogyra cylindrus.*

Dendrogyra cylindrus is uncommon throughout the Florida Keys and Dry Tortugas. It differs from the above two species (*Agaricia lamarcki* and *Mycetophyllia ferox*) in that it is typically found in shallower coral reef habitats. It is naturally rare in the Florida Keys and Dry Tortugas. Our sample allocation schemes did not optimize for this species. In the Florida Keys, *Dendrogyra cylindrus* ranked 47th out of 47 in 2005, with a population estimate of 23,000 (SE 23,000) and 41st out of 43 in 2009, with a population estimate of 25,000 (SE 25,000). In 2012, no colonies were encountered. Despite the low population estimate, it is well-known that there are several spectacular stands of this species in the Florida Keys that appear in good condition. This species was not seen in the Dry Tortugas in 2006 and 2008. While our population data are limited for this species, without evidence of significant decline, listing the species as Endangered is not warranted.

It is important to note that these population estimates for the Florida Keys are for a region that is considered marginal for coral reef development, certainly through the Holocene where active reef growth in the Florida Keys is restricted to a relatively small area of the total hard bottom area, plus most corals (including the ones discussed in this report) are at or near their northern geographic limit of distribution in Florida – they are all widely distributed throughout the Caribbean. Further, the total coral reef habitat in the Florida Keys represents a small percentage of area (approximately 3 percent) relative to the larger Caribbean, and about 27 percent of total reef area in the U.S. Caribbean. In other words, the population estimates for these species in the Florida Keys must be considered extremely conservative estimates. As such, our results do not support the NOAA-NMFS proposal to list or reclassify these nine Atlantic coral species under ESA as Threatened or Endangered.

1. Introduction

A significant loss of coral cover in recent decades has occurred globally (Wilkinson 1992), throughout the Indo-Pacific (Bruno and Selig 2007), the Caribbean (Gardner et al 2003, Schutte et al. 2010), and in the Florida Keys (Somerfield et al. 2008). The declines have been documented largely by monitoring programs and by meta-analyses where total percent coral cover is the primary metric (Gardner et al 2003, Bruno and Selig 2007, Schutte et al 2010). Coral cover works well as a metric in status and trends programs because it is relatively straightforward to measure, precise, and comparisons over time and among sites are easily made. What constitutes an ecologically relevant change in cover is somewhat arbitrary, but a 30% decline is probably significant (Connell 1997). However, cover is not a population-based metric. To understand the ecological significance of coral decline, or recovery, population data are needed that quantify species abundances and size-class distributions (Meesters et al. 2001). For example, cover estimates (total or by species) are static and generate similar results for a large number of small colonies, or a small number of large colonies, but size-frequency distributions in a population document processes related to recruitment, growth, and mortality. The processes that shape the patterns seen in size-frequency distributions of coral populations can reveal information about the long-term consequences of environmental change, disturbance, and resilience (Bak and Meesters 1997, 1998, Crabbe 2009).

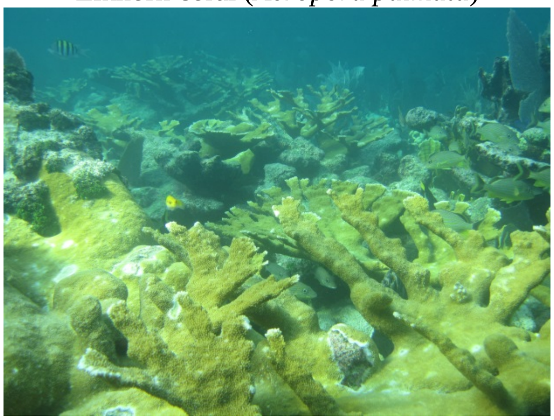
This report presents abundance and size-class frequency distributions for nine coral species (Table 1-1 and Figures 1-1 through 1-2) in the Florida Keys (Figure 1-3), all of which are proposed for listing or reclassification under the U.S. Endangered Species Act (ESA). The lack of population data for these species was highlighted as a deficiency in the ESA Review Process by the Biological Review Team (BRT) in their Status Review (Brainard et al. 2011) and also by the National Oceanic and Atmospheric Administration (Federal Register 2012). Two of the species, *Acropora palmata* and *A. cervicornis*, were listed in 2006 as Threatened under the ESA (Hogarth 2006), largely based upon a summary produced by the Atlantic *Acropora* BRT (2005), and are under consideration for reclassification in 2013 to Endangered. Justification for the initial listing of *A. palmata* and *A. cervicornis* was largely based on widespread population declines due to white band disease and coral bleaching. It was also noted that range-wide population declines occurred over a relatively short time period (i.e. from the late 1970s to the late 1980s). Declines in percent cover data or after-the-fact anecdotal information (Aronson and Precht 2001a, 2001b, Precht et al. 2002) were used to document declines and substantially informed the ESA review process that led to their listing as Threatened under the ESA. The two species were not considered Endangered in 2006 due to estimates of large remaining populations, large and intact geographic distributions, reproductive potential, and evidence of limited recovery. They are now candidate species for reclassification to Endangered Status under ESA based on reports of additional decline, continued threats from coral bleaching and disease, increasing apparent threats due to ocean acidification, and a revised determination that their distribution throughout the Caribbean, while intact, is now limited and an adds to their risk of extinction (Brainard et al. 2011, Federal Register 2012).

The seven additional wider Caribbean scleractinian coral species are under review in 2013 for ESA listing as Threatened or Endangered, based on their vulnerabilities to ocean warming and acidification, coral disease, demographics related to declines and life history factors, as well as their geographic ranges. Five of these species (*Acropora cervicornis*, *A. palmata*, *Montastraea annularis*, *M. faveolata*, and *M. franksi*) are considered the primary reef-building corals in the wider Caribbean region, including south Florida (Precht and Miller 2007).

It is important to note that corals in the Florida Keys and Dry Tortugas are at their northern limit of geographic distribution for reef development in the western Atlantic. They are subject to all of the same stressors and disturbances experienced by corals throughout the wider Caribbean (Precht and Miller 2007); with the additional stress caused by cold fronts that periodically kill large numbers of corals in the Florida Keys (Lirman et al. 2011) and Dry Tortugas (Davis 1982, Porter et al. 1982, Jaap and Hallock 1990). Further, coral reef habitat in the Florida Keys represents approximately 3 percent of total coral reef habitat in the Caribbean (Spalding and Grenfell 1997; Burke and Maidens 2004). The populations assessments presented here are thus an extremely conservative estimate of the total population numbers for each of nine species, since they are found throughout Caribbean. It is also important to note that our monitoring program began in the late 1990s (pilot studies), long after major declines had already occurred in the region, specifically the loss of *D. antillarum* and *Acropora palmata and A. cervicornis* (see Precht and Miller 2007).

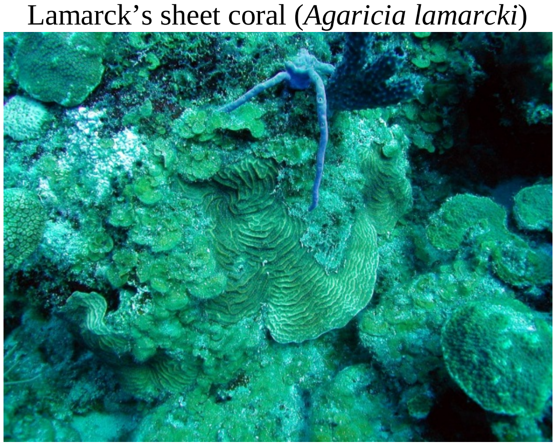
Figure 1-1. Caribbean/Atlantic/Gulf of Mexico candidate coral species in the genera *Acropora*, *Agaricia*, *Dichocoenia*, *Dendrogyra*, and *Mycetophyllia* proposed for endangered or threatened status under the U.S. Endangered Species Act.

Elkhorn coral (*Acropora palmata*)

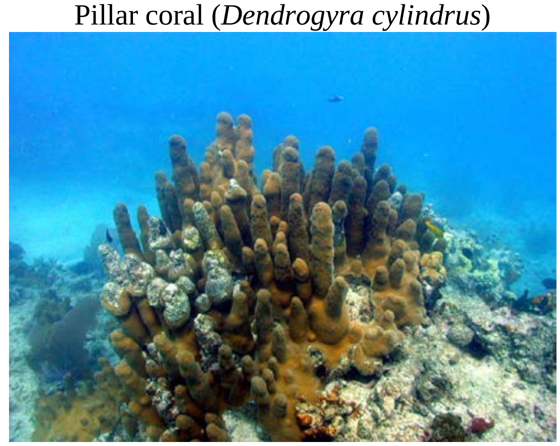


Staghorn coral (*A. cervicornis*)

Lamarck's sheet coral (*Agaricia lamarcki*)



Pillar coral (*Dendrogyra cylindrus*)



Elliptical star coral (*Dichocoenia stokesi*)

Rough cactus coral (*Mycetophyllia ferox*)

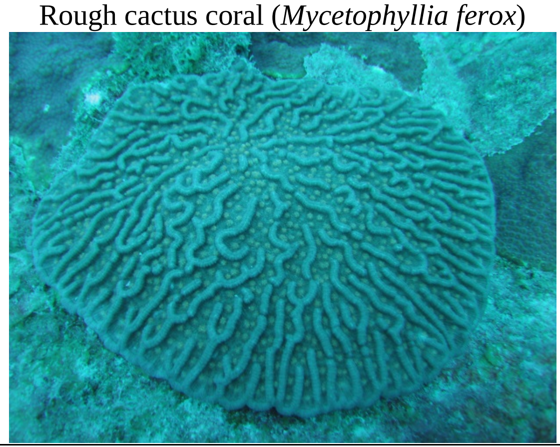


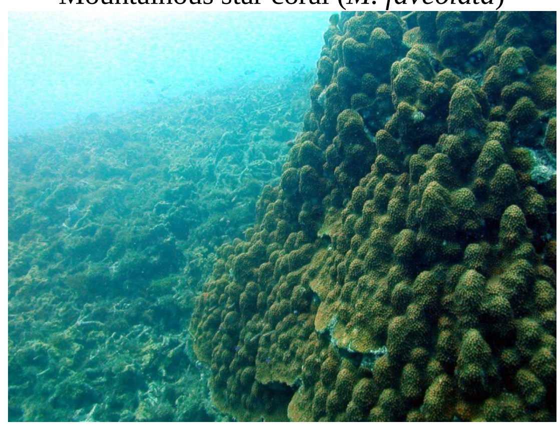
Figure 1-2. Caribbean/Atlantic/Gulf of Mexico coral species in the Genus *Montastraea* proposed for endangered or threatened status under the U.S. Endangered Species Act.

Boulder star coral (*Montastraea annularis*)

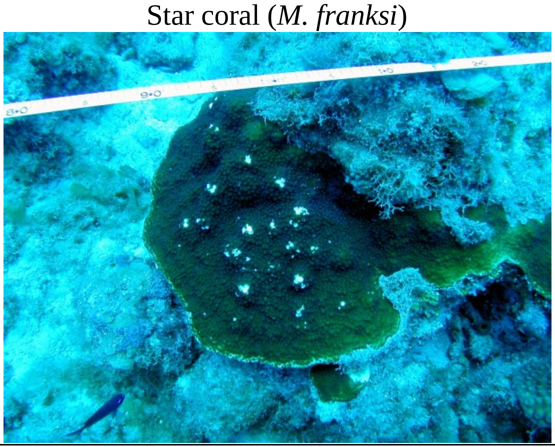
Boulder star coral (*M. annularis*)

Mountainous star coral (*M. faveolata*)

Mountainous star coral (*M. faveolata*)



Star coral (*M. franksi*)



Star coral (*M. franksi*)

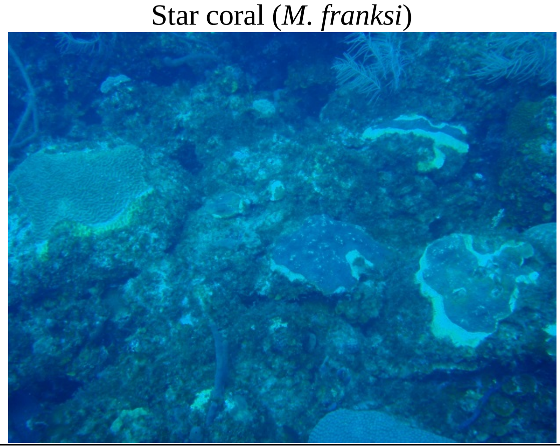


Figure 1-3. The Florida Keys and Dry Tortugas study area, with existing Federal (NOAA, NPS, and FWS) and State managed areas, where surveys for scleractinian corals and other benthic coral reef organisms were undertaken during 1999-2012.

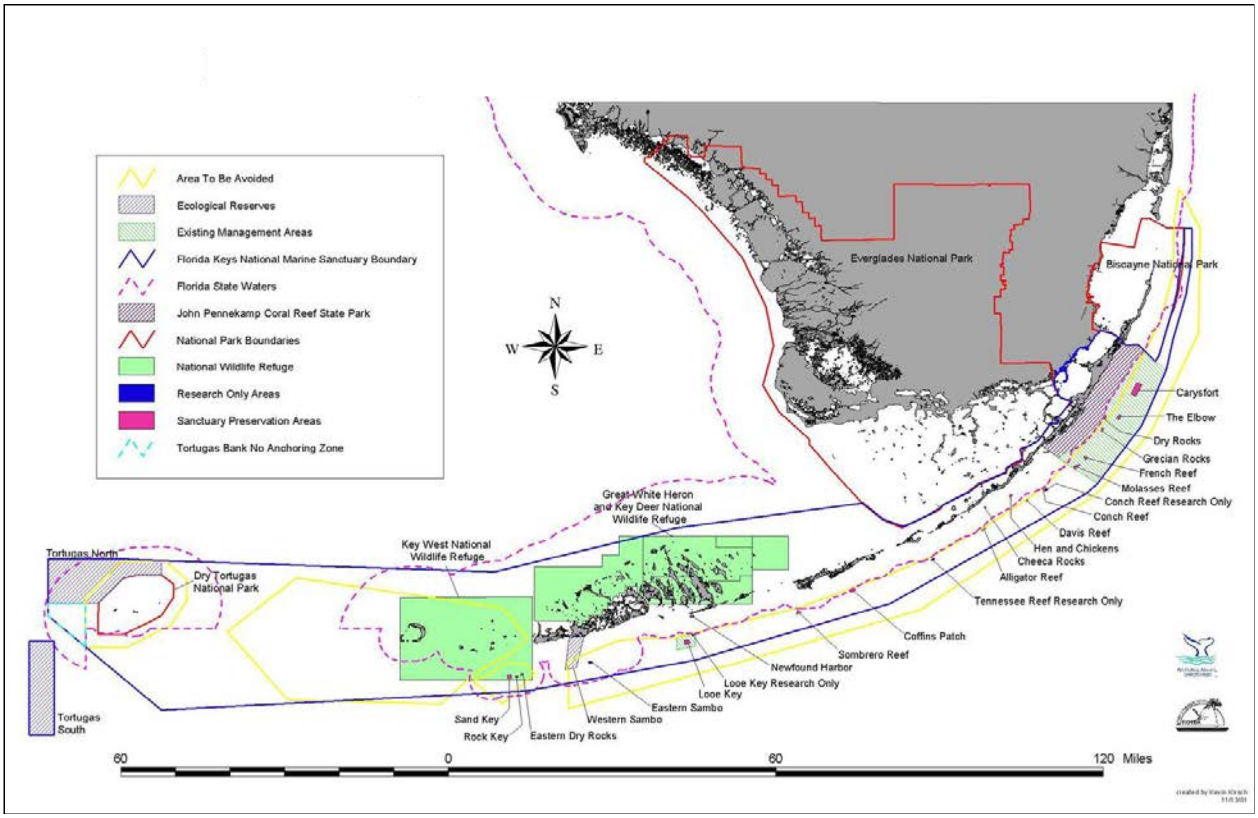


Table 1-1. Caribbean/Atlantic/Gulf species proposed as Endangered or Threatened under the U.S. Endangered Species Act. Scientific and common names based upon Cairns et al. (2002).

Proposed Status	**Scientific name**
Endangered	
Staghorn coral	*Acropora cervicornis* (Lamarck, 1816)
Elkhorn coral	*Acropora palmata* (Lamarck, 1816)
Pillar coral	*Dendrogyra cylindrus* Ehrenberg, 1834
Boulder star coral	*Montastraea annularis* (Ellis and Solander, 1786)
Mountainous star coral	*Montastraea faveolata* (Ellis and Solander, 1786)
Star coral	*Montastraea franksi* (Gregory, 1895)
Rough cactus coral	*Mycetophyllia ferox* Wells, 1973
Threatened	
Lamarck's sheet coral	*Agaricia lamarcki* Milne Edwards and Haime, 1851
Elliptical star coral	*Dichocoenia stokesi* Milne Edwards and Haime, 1848

2. Study Area and Survey Methods

2.1 Florida Keys Environmental Setting

The Florida Keys comprise an archipelago of limestone islands spanning more than 360 km from south of Miami to the Dry Tortugas. With the exception of isolated banks in the Flower Gardens area in the northwestern Gulf of Mexico, the Florida Keys ecosystem represents the only region of extensive coral reef development in the continental U.S. (Jaap 1984). The islands are part of the larger south Florida shelf, a submerged Pleistocene platform 6-35 km wide and generally < 12 m deep (Lidz et al. 2003). The primary influences on the distribution and development of Florida Keys reefs are paleotopography and fluctuating sea level (Shinn et al. 1989; Lidz et al. 2003). Bedrock throughout south Florida is Pleistocene limestone, either exposed on the seafloor or lying underneath Holocene reefs and sands (Shinn et al. 1989). Proceeding seaward from the shorelines of the Pleistocene islands, a nearshore rock ledge extends ~2.5 km from the shoreline, with the seabed consisting of hard-bottom, seagrass, and isolated inshore patch reefs (FMRI 1998). Seaward of the island platform is Hawk Channel, a broad trough-like depression dominated by mostly non-coralline, non-oolitic grainstone, dotted with several thousand patch reefs whose distribution is affected by the number and width of tidal passes connecting Florida Bay and the Atlantic Ocean (Marszalek et al. 1977; Shinn et al. 1989). Bands of rock ridges exist further offshore along the outer shelf and on the upper slope from 30-40 m depth before the shelf tapers off into the Straits of Florida. The semi-continuous offshore reef tract is emergent in places, in which Holocene reefs sit atop a ridge of Pleistocene corals (~86-78 ka), forming a shelf-margin ledge (Lidz et al. 2003), with a series of outlier reefs seaward of this main reef tract at 30-40 m depth (Lidz 2006). Like inner shelf margin patch reefs, the distribution of platform margin reefs reflects exchange processes between Florida Bay and the Atlantic Ocean (Marszalek et al. 1977; Shinn et al. 1989), which is related to the size and orientation of the Pleistocene islands and thus the presence and size of tidal passes, as well as the proximity of the Florida Current to the platform margin (Pitts 1994; Smith 1994).

2.2 Coral Population Survey Design

The sampling domain for the coral data presented in this report included most of the ocean-side hard-bottom and coral reef types from northern Biscayne National Park to southwest of Key West, as well as the Dry Tortugas. The Florida Keys and Dry Tortugas survey areas included along-shelf and cross-shelf gradients of hard-bottom and coral reef habitats. To control for spatial variation in population abundance metrics, the survey domain was divided into strata based upon habitat types, geographic regions, and management zones (Miller et al. 2002). Cross-shelf habitat types were designated using regional benthic habitat maps (FMRI 1998). The hard-bottom and coral reef habitat classification scheme accounted for features that correlate with benthic fauna distributions, including cross-shelf position, topographic complexity, and the proportion of sand interspersed among hard-bottom structures. A geographic regional stratification variable (i.e. upper, middle, and lower Keys) was used to account for oceanographic and geological features in the Florida Keys that influence the distribution, community dynamics, and biotic composition of reefs (Marszalek et al. 1977, Shinn et al. 1977). Management zones, including the Florida Keys National Marine Sanctuary (FKNMS) no-take marine reserves, were incorporated as a third stratification variable that delineated areas open and closed to consumptive activities. Figures 2-1 and 2-2 illustrate the spatial distribution of sampling locations in the Florida Keys and Dry Tortugas. Results are presented here for the larger survey domains of the Florida Keys and Dry Tortugas, only. Results based on habitat and management zone stratifications have been presented elsewhere for *Acropora palmata* and *A, cervicornis* (Miller et al. 2008) and are in preparation for the larger suite of corals found in the region.

A geographic information system containing digital layers for benthic habitat (FMRI 1998), bathymetry (National Geophysical Data Center, Silver Spring, Maryland), and no-take marine reserve boundaries (Florida Keys National Marine Sanctuary, Marathon, Florida) was used to facilitate delineation of the sampling survey area, strata, and sample units. Map resolution was such that the survey domain was divided into a grid with individual cells of size 200 m by 200 m (40,000 m^2) that defined unique habitat types. Grid cells or sites 200-m x 200-m in dimension were used to randomly select sites from the combination of habitat type, regional sector, and management zone combinations. Habitats were designated using regional benthic habitat maps (FMRI 1998). The habitat classification scheme accounted for features that correlate with benthic fauna distributions, including cross-shelf position, topographic complexity, and the proportion of sand interspersed among hard-bottom structures. The habitat strata incorporated hard-bottom and coral reef habitat types from the island platform inshore of Hawk Channel to ~15 m depth along the reef tract in the Florida Keys. Sampling in the Dry Tortugas region during 2006 and 2008 included a diversity of habitat types on the shallower bank encompassed mostly by Dry Tortugas National Park, as well as deeper (15-27 m) hard-bottom and coral reef habitats on the Tortugas Bank further to the west. Figures 2-3 to 2-5 illustrate representative examples of the various hard-bottom and coral reef habitat types surveyed. Coral surveys during this period did not encompass nearshore hard-bottom, hard-bottom/seagrass matrix, seagrass beds, and bare sand. From northern Biscayne National Park to SW of Key West, the habitats sampled were inshore and mid-channel patch reefs, offshore patch reefs, back reef rubble, shallow (< 6 m) hard-bottom, inner line reef tract spur and groove from Grecian

Rocks northward to Turtle Reef, shallow (< 6 m) high-relief spur and groove along the platform margin, and deeper fore-reef habitats from 6-15 m depth. Deeper fore-reef habitats encompassed continuous, low-relief hard-bottom, patchy hard-bottom, and low-relief spur and groove. In the Dry Tortugas region, habitats sampled included shallow to deeper hard-bottom, patch reefs, high-relief spur and groove, fore-reef terrace, and low-relief spur and groove habitats (Tables 2-3 and 2-4).

A two-stage sampling scheme following Cochran (1977) was employed to account for the disparity in size between the grid cell minimum mapping unit (40,000 m^2) and the belt transect area surveyed for corals (10 or 15 m^2). Grid cells containing reef habitats were designated as primary sample units. Belt transects were designated as the second-stage sample units (SSU). The size of an individual primary sampling unit allowed divers to swim to the location of any given second-stage sampling unit from a moored vessel. The conceptual layout of the two-stage stratified random sampling design is shown in Figure 2-6. The survey area was divided into sub-regions termed strata. Each stratum was further subdivided into primary sample units, and each primary unit was again subdivided into second-stage sample units. Note that each primary- and second-stage sample unit contains a fixed amount of area; thus, the sum of second-stage sample units within primary units of all strata equals the total survey area. The strata areas and corresponding number of possible primary sample units in the Florida Keys survey area are given in Table 2-3. Selection of primary and second-stage samples within a given stratum was carried out in two stages. First, the primary units to be sampled were randomly selected without replacement from the complete list of *Nh* units using a discrete uniform probability distribution (Law and Kelton 2000), which assigned equal selection probability to each primary unit. Second, a similar procedure was used to select second-stage units to be sampled from the total possible units within a primary unit.

2.3 Field Methodology

Field sampling protocols were adapted from Aronson et al. (1994) and the Atlantic and Gulf Rapid Reef Assessment program (Kramer and Lang 2003) to capture population-level metrics of scleractinian corals, including presence-absence, station (transect) frequency of occurrence, density, size, condition, and population abundance estimates structured by habitat, size class, and colony conditions (Swanson 2011). The data in this report focus on the scleractinian corals, with particular reference to ESA-candidate species, in the Florida Keys surveyed during 2005, 2007, 2009 and 2012, and in the Dry Tortugas for 2006 and 2008. Stony coral colonies were separated by size into juvenile (< 4 cm max. diameter) and non-juvenile (> 4 cm) adult life stages, following Bak and Engel (1979) and others. Colony density within belt transects and size measurements (maximum diameter, maximum height, and perpendicular diameter) were obtained for adults of each species present. An individual colony was considered to be a continuous skeletal unit, so that a colony that was part of the same skeleton but divided into two or more separate pieces of live tissue was still considered to be one colony.

The underwater surveys consisted first of locating randomly selected, pre-determined coordinates with a differential global positioning system. A Garmin® global positioning system receiver (model GPS76) was used to determine the position at each site. For each sampling year, the targeted list of sites varied based upon logistics and objectives. If the original waypoint was not the intended habitat type, based on visual assessment by a snorkeler, the closest alternate site was sampled instead. Once on-site, a two-person diver team orients two to four transect tapes 25-m (1999-2002) or 15-m (2005-2012) in length along the bottom. In both the Florida Keys and Dry Tortugas, a 0.8-m swath along each of two 25-m transects was surveyed for all scleractinian corals > 4 cm maximum diameter. In 2009 and 2012 in the Florida Keys, as well as in 2006 and 2008 in the Dry Tortugas, two replicate 10-m x 1-m belt transects per site were surveyed. In the Florida Keys (excluding the Dry Tortugas) during 2007 and 2012 optimized *Acropora* surveys encompassed four 15-m x 1-m belt transects, and two 15-m x 1-m belt transects per site, respectively. Table 2-1 lists the sampling effort by year for *Acropora* corals, Table 2-2 lists the sampling effort for those years in which all coral species, including *Acropora* corals, were surveyed, while Table 2-3 lists the physical characteristics of habitats sampled. During 2005 in the Florida Keys (excluding the Dry Tortugas), the sampling effort for *Acropora* corals was the same as for all other scleractinian coral species; specifically, paired 25-m x 0.4 m (1999-2002) or 10-m x 1-m (2005) transects were surveyed per site for coral density, size, and condition. Finally, Table 2-4 details the sampling effort by habitat and by year for *Acropora* and non-*Acropora* coral species in the Florida Keys and Dry Tortugas.

Coral surveys involved colony counting, measurements of colony dimensions, estimates of percent live vs. dead, and assessments of condition (e.g. bleaching, disease, overgrowth, and predation). Each colony greater than 4 cm in maximum diameter was identified, measured, and assessed for condition. All scleractinian colonies located within the belt transect were included in the survey, even if a portion of the colony extended outside of the boundaries of the belt transect. Individual colonies were identified as continuous skeletal units, regardless of whether the skeletal unit contained multiple patches of separate live tissue. Only colonies containing live tissue were included in the survey. Colony size was recorded using 10-cm incremental classes, to facilitate rapid assessment. Size class 0 was used to record the maximum diameter of species that have a small maximum size, such as *Favia fragum* and *Scolymia* spp., which would otherwise be excluded due to the overall adult (non-juvenile) size class lower-limit of 4 cm.

There is no upper limit imposed on the maximum diameter size classes. Mortality was recorded using 20% incremental classes and included visual estimates of recent and long-term tissue death.

Each colony was also assessed for condition (limited summary data are presented in this report). Any colonies with lighter tissue coloration than normal were assessed for bleaching. Partially pale and pale colonies were not included in the bleaching data analyses, although their condition was recorded. Mottling, or small patterns of light and dark discolorations often found on colonies of *Siderastrea siderea*, was also recorded, but not included in the bleaching data analyses. Only disease conditions that were actively causing tissue death or lesions on a colony were recorded. If a colony showed signs of a disease that could not be clearly identified, the condition was recorded as unknown disease. If a colony contained patches of necrotic issue with no identifiable cause, it was recorded as necrosis. Dark-spot condition/syndrome was recorded as a disease, even though it does not typically result in lesions or rapid tissue death. Overgrowth of coral tissue by another organism (e.g. algae, sponges, gorgonians, *Palythoa*, and other corals) was noted only if overgrowth by the organism was clearly causing tissue death or lesions. Overgrowth of organisms onto dead portions of a colony was not recorded, nor was overgrowth or shading of live tissue with no resulting lesions or tissue death.

Physical impacts, such a sediment scour, contact with other organisms, and fishing gear damage (e.g. trap rope abrasion) were recorded as abrasion. The presence of boring sponges such as *Cliona delitrix* was recorded if a sponge was actively causing tissue death lesions, but was not recorded if a sponge was only visible on dead portions of a colony. The presence of damselfish nests or gardens was recorded whenever they were found adjacent to, or surrounded by, live tissue. Likewise, fish bites/scrapes were only recorded if they were found on live tissue. Whenever gastropods were observed on a coral colony, the identity and total length of each individual was noted, regardless of whether the gastropods were actively feeding on live coral tissue. However, only gastropods actively feeding on live coral tissue were recorded as a mortality condition. Apparent gastropod feeding scars with no gastropods present was recorded as unknown mortality. Any tissue death that could not be attributed to disease, abrasion, boring sponges, or predation was also recorded as unknown mortality.

2.4 Statistical Analyses

Statistical estimation procedures for population abundance metrics – means (e.g. coral density) and totals (e.g. coral abundance) – for a two-stage stratified random sampling design were adapted from Cochran (1977), and computations were carried out using SAS statistical software. Animal density (colonies per station of transect) was the principal metric used to develop and evaluate the statistical sampling design. Survey-wide mean and variance estimates of density were obtained from weighted averages of strata means and variances. A stratum weighting factor was the proportion of the stratum area relative to the overall survey area. Stratum abundance (absolute number of colonies) was estimated by multiplying stratum density by stratum area. The same principle was used to estimate the variance of stratum abundance. Survey-wide abundance and associated variance were obtained by summing the respective strata estimates over all strata. Prevalence of conditions, including percent live vs. dead, were estimated as the proportion of individuals within a population afflicted with the specific condition (Gerstman 2003). Coral density and abundance calculations were based upon the number of corals recorded within the stations (i.e. within each of the belt transects). First, coral density (no. colonies per m^2) was calculated for each station. Next, mean coral density and variance were calculated for each site, using the coral densities of the two stations. The mean site-level coral densities and variances were then used to calculate mean stratum-level (habitat, management zones, and habitat by management zone) coral densities and variances. Finally, stratum-level and domain abundance estimates were calculated based upon the stratum-level coral densities and variances, as well as the proportional areas of each stratum within the sampling domain. Only domain-wide estimates are presented in this report.

Figure 2-1. Sampling locations for *Acropora* corals during 2005, 2007, and 2012 (left) and for all scleractinian coral species during 2005, 2009, and 2012 (right).

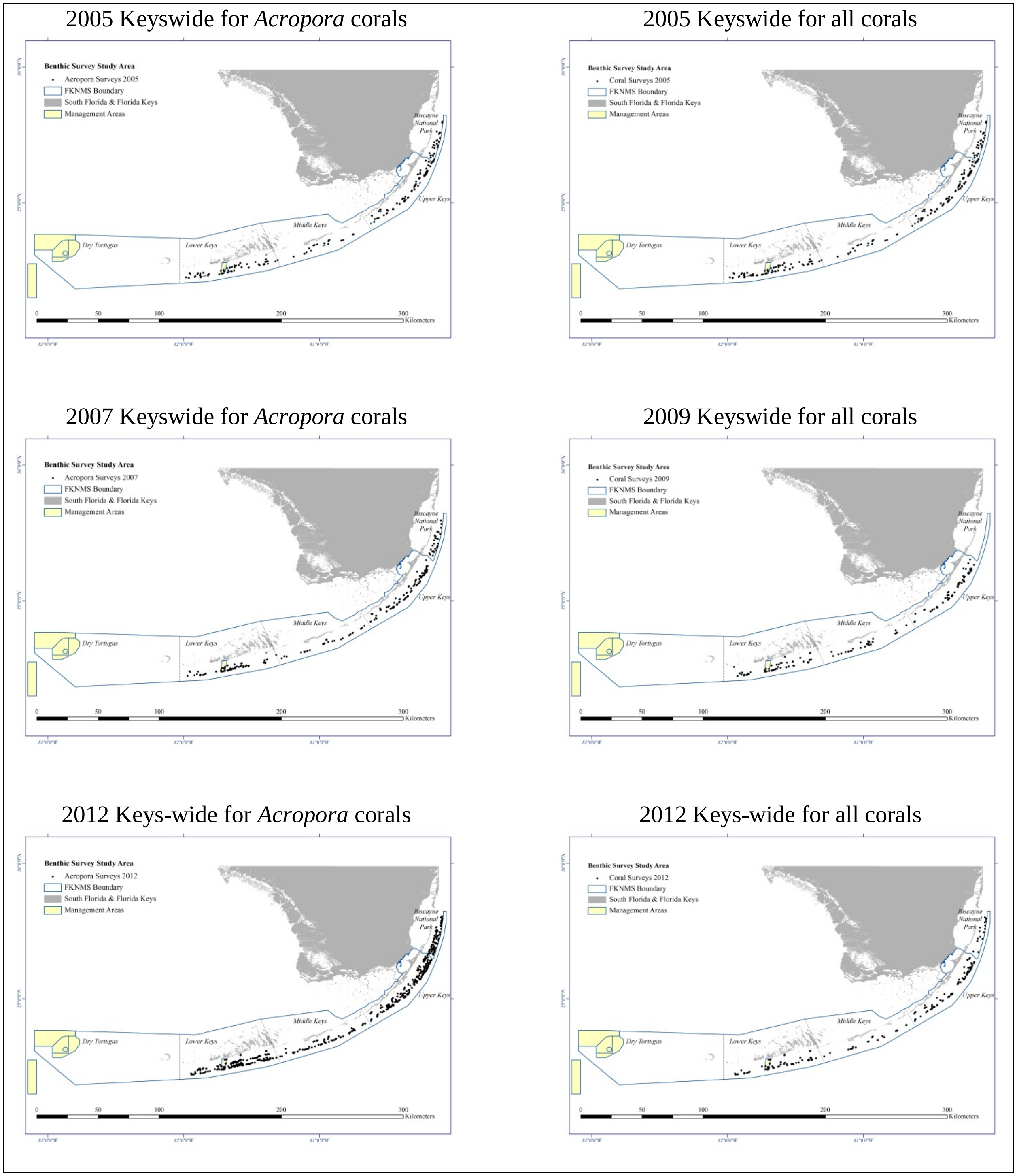


Figure 2-2. Sampling locations for *Acropora* corals and other scleractinian coral species in the Dry Tortugas region during 2006 (top) and 2008 (bottom).

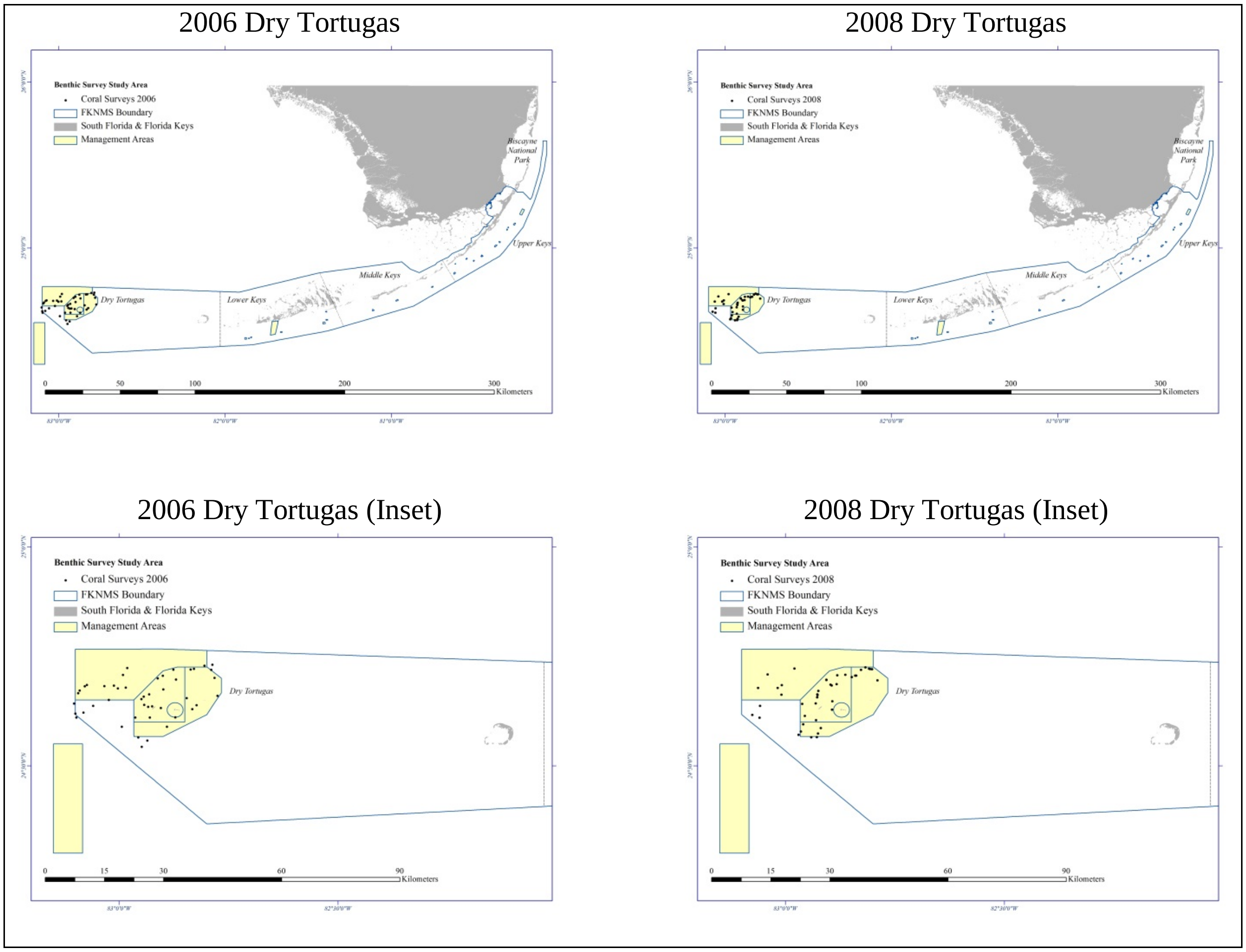


Figure 2-3. Representative examples of inshore, mid-channel and offshore patch reefs, as well as reef rubble habitats sampled in the Florida Keys and Dry Tortugas.

Inshore patch reef
Cheeca Rocks SPA, 2.4-3.7 m
24° 54.252'N, 80° 36.896'W

Mid-channel patch reef
Basin Hill Shoals, 2.4-4.3 m
25° 14.412'N, 80° 15.868'W

Offshore patch reef
West of S. Carysfort Reef, 1.5-2.7 m
25° 12.752'N, 80° 13.797'W

Offshore patch reef
Watsons Reef area, 11.0-12.2 m
25° 10.067'N, 80° 15.239'W

Back-reef rubble
Molasses Reef SPA, 1.5-3.0 m
25° 00.708'N, 80° 22.691'W

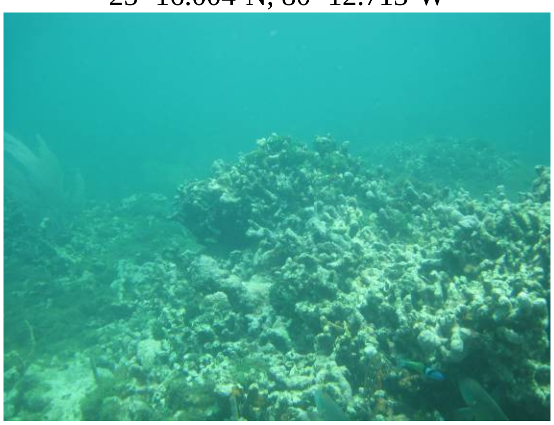
Back-reef rubble
Pickles Reef, 2.7-3.4 m
24° 59.480'N, 80° 24.978'W

Figure 2-4. Representative examples of shallow (< 6 m) low-relief hard-bottom sites and high-relief spur and groove habitats sampled in the Florida Keys and Dry Tortugas during.

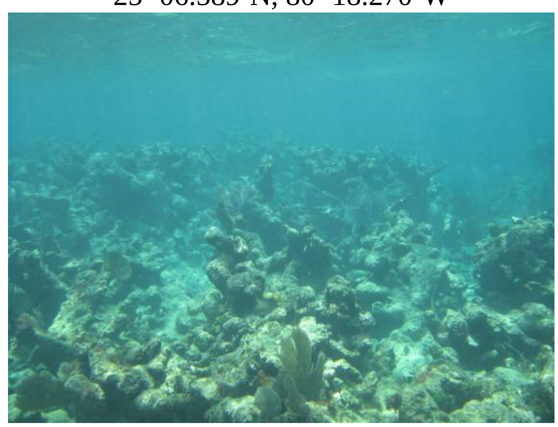
Low-relief hard-bottom
Carysfort Reef SPA, 3.0-3.7 m
25^{o} 13.399'N, 80^{o} 12.772'W

Low-relief hard-bottom
North of Watsons Reef, 4.0-5.5 m
25^{o} 11.468'N, 80^{o} 14.254'W

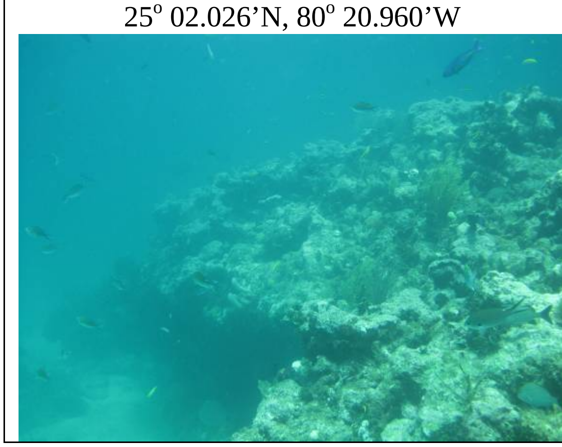
Inner line reef tract spur and groove
Turtle Rocks, 3.7-4.3 m
25^{o} 16.004'N, 80^{o} 12.713'W



Inner line reef tract spur and groove
Grecian Rocks SPA, 1.8-3.7 m
25^{o} 06.589'N, 80^{o} 18.270'W



Platform margin high-relief spur and groove
French Reef SPA, 4.9-8.5 m
25^{o} 02.026'N, 80^{o} 20.960'W



Platform margin high-relief spur and groove
Elbow Reef SPA, 2.1-5.5 m
25^{o} 08.572'N, 80^{o} 15.475'W

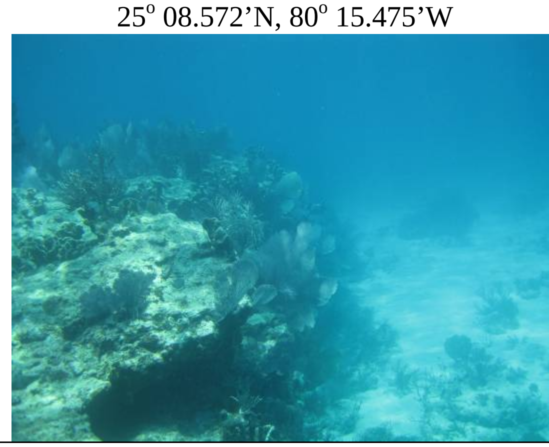


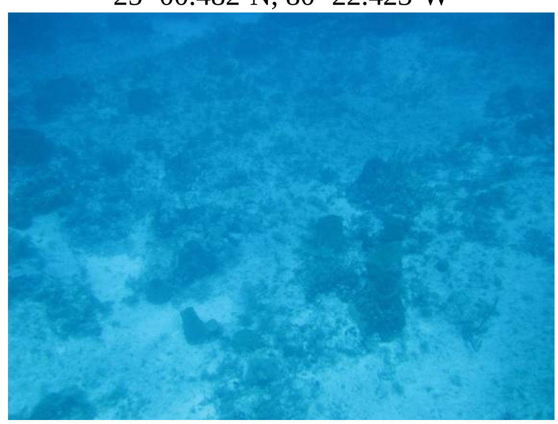
Figure 2-5. Representative examples of deeper (6-27 m) fore-reef habitats sampled in the Florida Keys and Dry Tortugas.

Low-relief hard-bottom
Whistle Buoy, 6.7-8.5 m
25° 17.411’N, 80° 10.412’W

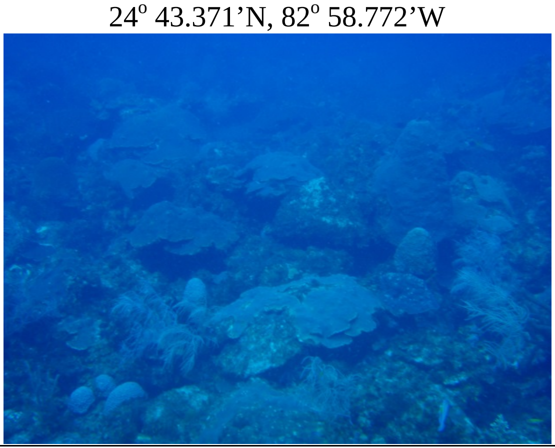
Low-relief spur and groove
SW of Davis Reef, 10.7-11.3 m
24° 55.148’N, 80° 30.428’W

Patchy hard-bottom
Whistle Buoy, 10.4-11.3 m
25° 17.596’N, 80° 10.351’W

Low-relief spur and groove
Molasses Reef SPA, 12.2-13.4 m
25° 00.482’N, 80° 22.423’W



Reef terrace
Little Tortugas Bank, 22.6-24.4 m
24° 43.371’N, 82° 58.772’W



Patchy hard-bottom
SW Dry Tortugas National Park, 18.6-18.9 m
24° 34.257’N, 82° 58.204’W

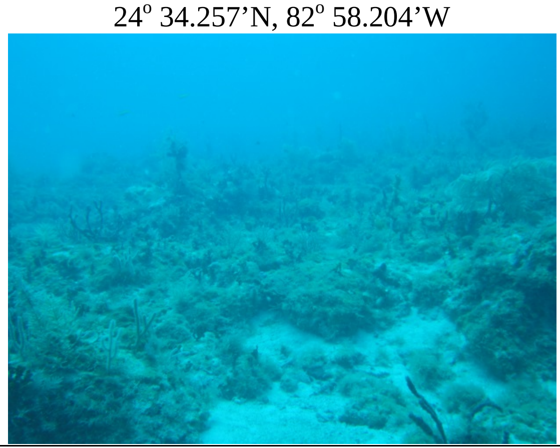


Figure 2.6. The two-stage stratification designed for coral population surveys in the Florida Keys and Dry Tortugas: (A) incorporates habitat type (cross-shelf position and depth), geographic region (along-shelf position), and management zone, utilizing a grid of 200-m x 200-m cells overlain onto existing habitat and bathymetry maps. (B) The example below demonstrates the two-stage stratification approach, where first- or primary-stage units shown as squares within a targeted habitat type are randomly selected based upon the three stratification variables. (C) An enlarged view of the sample grid with the arrow indicating a 200-m x 200-m cell containing a targeted benthic habitat type. (D) An enlarged view of one sample cell where second-stage units (transects) are deployed at random GPS points within a particular cell.

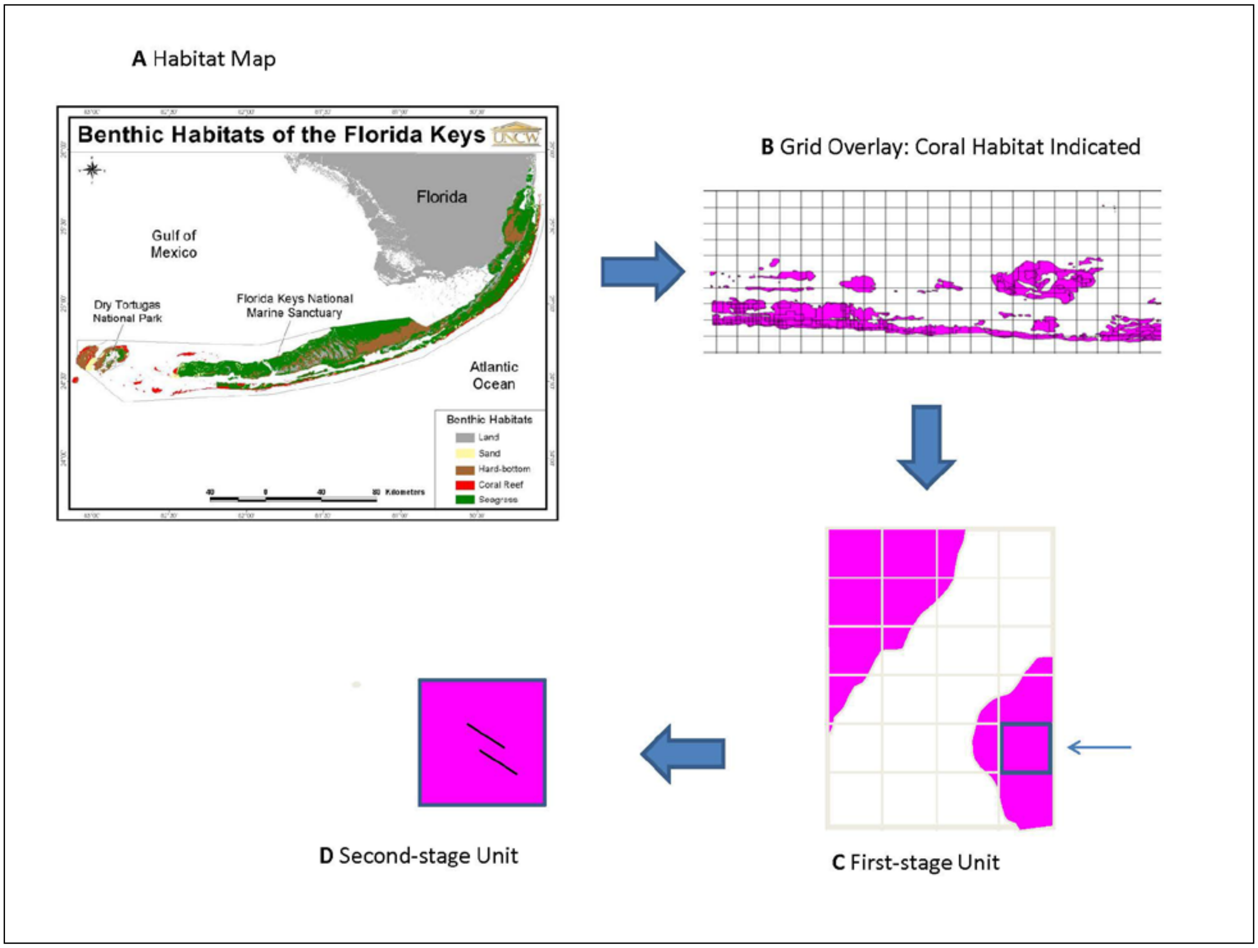


Figure 2-7. Benthic survey methods used to sample coral populations in the Florida Keys.

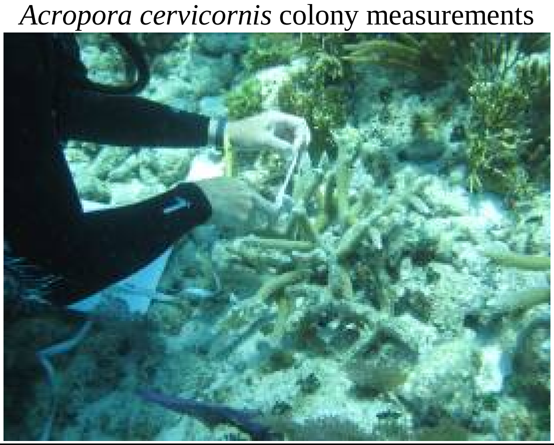
Underwater slate for data collection

Disentangling corals from marine debris

Using the 0.5-m scale bar along a transect

Transect deployment

Acropora cervicornis colony measurements



Belt transect survey

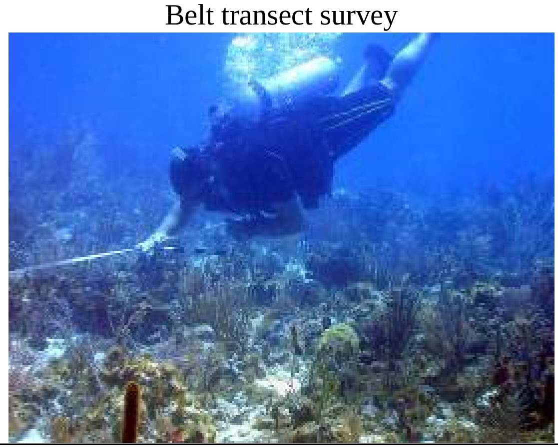


Table 2-1. Sampling effort for *Acropora* corals in the Florida Keys and Dry Tortugas. The number of sites represents the number of 200-m x 200-m grid cells sampled, while the number of sites available is based upon mapped hard-bottom and coral reef habitat.

Geographic coverage	Year	Presence-absence	Frequency of occurrence		Density/size/condition	
		No. sites (no. available)	No. transects	Area surveyed (m^2)	No. transects	Area surveyed (m^2)
Fowey Rocks-Key West	2005	195 (12,379)	780	46,800	390	3,900
	2007	231 (10,104)	924	13,860	924	13,860
	2012	600 (13,744)	1,200	18,000	1,200	18,000
Dry Tortugas	2006	46 (8,801)	184	2,760	92	920
	2008	43 (7,951)	168	2,520	86	860

Data do not include nearshore hard-bottom (4 sites surveyed in 2000 in the lower Keys, deeper (15-28 m) fore-reef sites (25) surveyed Keys-wide in 2005, and seagrass/hard-bottom matrix (3 sites surveyed in 2006 in the upper Keys and 4 sites surveyed in 2007 in the upper Keys). Note that the Key Largo-Alligator Reef surveys in 1999-2001, 2005, 2007-09, and 2010-2012 are a subset of the Keys-wide effort during those years. ns = Not sampled.

Table 2-2. Sampling effort for non-*Acropora* corals in the Florida Keys and Dry Tortugas. The number of sites represents the number of 200-m x 200-m grid cells sampled, while the number of sites available is based upon mapped hard-bottom and coral reef habitat.

Geographic coverage	Year	Presence-absence	Frequency of occurrence		Density/size/condition	
		No. sites	No. transects	Area surveyed (m^2)	No. transects	Area surveyed (m^2)
Fowey Rocks-Key West	2005	195 (12,379)	780	46,800	390	3,900
	2009	160 (8,678)	640	9,600	320	3,200
	2012	202 (13,744)	404	4,040	404	4,040
Dry Tortugas	2006	46 (8,801)	184	2,760	92	920
	2008	43 (7,951)	168	2,520	86	860

Data do not include nearshore hard-bottom (4 sites surveyed in 2000 in the lower Keys, deeper (15-28 m) fore-reef sites (25) surveyed Keys-wide in 2005, and seagrass/hard-bottom matrix (4 sites surveyed in 2007 in the upper Keys). ns = Not sampled.

Table 2-3. Hard-bottom and coral reef habitat areas (A_h, km^2), proportional habitat areas (%), and physical characteristics (mean ± 1 SE, range) of Florida Keys (northern Biscayne National Park to Key West, top) and Dry Tortugas region (bottom) habitats surveyed for *Acropora* corals and other ESA candidate species during 1999-2012.

Florida Keys (northern Biscayne National Park to SW of Key West)

Habitat type (no. sites)	**Habitat area (km^2)**	**Sample depth (m)**			**Max. vertical relief (cm)**	
	A_h (%)	**Mean. min**	**Mean max.**	**Range**	**Mean**	**Range**
Inshore patch reefs (24)	8.80 (1.7)	2.4 ± 0.2	4.1 ± 0.2	0.9-7.3	135 ± 10	44-235
Mid-channel patch reefs (379)	125.56 (24.9)	4.0 ± 0.1	5.5 ± 0.1	0.6-10.4	82 ± 2	15-263
Offshore patch reefs (384)	73.52 (14.6)	5.2 ± 0.1	6.5 ± 0.1	1.2-13.4	71 ± 2	22-255
Reef rubble (75)	14.16 (2.8)	3.7 ± 0.2	4.4 ± 0.2	1.2-8.8	27 ± 2	8-69
Shallow (< 6 m) hard-bottom (197)	41.00 (8.1)	4.4 ± 0.1	5.3 ± 0.1	1.5-8.5	42 ± 1	18-94
Inner line reef tract (82)	3.92 (0.8)	2.9 ± 0.1	4.9 ± 0.1	0.6-8.0	106 ± 5	45-240
High-relief spur and groove (309)	10.88 (2.2)	3.6 ± 0.1	5.8 ± 0.1	0.6-14.9	106 ± 3	31-310
Deeper (> 6 m) hard-bottom (182)	95.92 (19.1)	8.0 ± 0.1	9.1 ± 0.1	5.5-13.7	43± 1	10-105
Patchy hard-bottom in sand (105)	33.76 (6.7)	7.9 ± 0.1	8.8 ± 0.2	4.3-13.7	44 ± 1	17-80
Low-relief spur and groove (278)	95.92 (19.1)	9.8 ± 0.1	11.0 ± 0.1	5.8-18.0	51 ± 1	14-133
All habitats (2,015)	503.44 (100.0)	5.5 ± 0.1	6.9 ± 0.1	0.6-18.0	70 ± 1	8-310

Dry Tortugas region (National Park, Tortugas Bank and Riley's Hump, excluding the Marquesas)

Habitat type (no. sites)	**Habitat area (km^2)**	**Sample depth (m)**			**Max. vertical relief (cm)**	
	A_h (%)	**Mean. min**	**Mean max.**	**Range**	**Mean**	**Range**
Patch reefs (< 6 m) (6)	7.88 (2.4)	4.3 ± 0.2	6.0 ± 0.3	3.7-7.0	22 ± 5	17-26
Patch reefs (6-15 m) (3)	16.48 (4.9)	8.6 ± 0.7	9.9 ± 0.9	7.3-11.3		
Patch reefs (15-21 m) (0)	3.76 (1.1)					
Low-relief hard-bottom (< 6 m) (4)	28.36 (8.5)	3.4 ± 0.4	4.3 ± 0.2	2.4-4.9	50 ± 5	40-58
Low-relief hard-bottom (6-15 m) (7)	43.88 (13.1)	12.7 ± 1.0	13.3 ± 1.1	7.3-15.5	34	34
Low-relief hard-bottom (15-21 m) (6)	73.48 (21.9)	17.6 ± 1.1	18.7 ± 1.0	16.2-21.6	48	48
Low-relief hard-bottom (21-33 m) (0)	54.24 (16.2)					
Low-relief spur & groove (< 6 m) (0)	1.44 (0.4)					
Low-relief spur & groove (6-15 m) (4)	7.68 (2.3)	11.1 ± 1.2	13.2 ± 1.3	8.5-15.5	69 ± 24	40-116
Low-relief spur & groove (15-21 m) (0)	2.4 (0.7)					
Low-relief spur & groove (21-33 m) (0)	0.08 (0.0)					
Patchy hard-bottom (< 6 m) (1)	3.12 (0.9)	5.2	5.8	5.2-5.8		
Patchy hard-bottom (6-15 m) (10)	22.24 (6.6)	12.6 ± 0.7	13.2 ± 0.7	8.8-15.8	40 ± 7	27-58
Patchy hard-bottom (15-21 m) (6)	9.04 (2.7)	19.1 ± 0.6	20.0 ± 0.6	18.0-21.0	38 ± 1	38-39
Patchy hard-bottom (21-33 m) (1)	3.12 (0.9)	22.3	23.8	22.3-23.8		
High-relief spur & groove (< 6 m) (0)	1.08 (0.3)					
High-relief spur & groove (6-15 m) (8)	2.2 (0.7)	6.1 ± 0.6	8.5 ± 0.8	6.1-12.5	141 ± 24	106-186
High-relief spur & groove (15-21 m) (0)	1.04 (0.3)					
High-relief spur & groove (21-33 m) (0)	0.76 (0.2)					
Reef knoll (6-15 m) (2)	0.16 (0.0)	9.3 ± 0.2	13.0 ± 2.0	7.3-14.9		
Reef knoll (15-21 m) (5)	0.4 (0.1)	17.8 ± 0.9	20.3 ± 1.1	15.8-22.9	133 ± 23	105-178
Reef knoll (21-33 m) (0)	0.12 (0.0)					
Reef terrace (6-15 m) (0)	0.08 (0.0)					
Reef terrace (15-21 m) (10)	5.28 (1.6)	17.3 ± 0.6	19.0 ± 0.6	15.2-22.3	106 ± 8	76-124
Reef terrace (21-33 m) (8)	11.12 (3.3)	22.6 ± 0.4	23.8 ± 0.5	21.3-26.2	170 ± 18	153-188
Medium-profile reef (6-15 m) (3)	1.48 (0.4)	3.6 ± 0.4	15.7 ± 2.4	2.7-5.8	85 ± 7	78-92
Medium-profile reef (15-21 m) (17)	4.12 (1.2)	11.8 ± 0.6	13.3 ± 0.6	6.4-16.5	83 ± 14	45-170
Medium-profile reef (21-33 m) (14)	2.16 (0.6)	19.4 ± 0.9	20.4 ± 0.9	15.2-26.5	74 ± 18	56-92
Rocky outcrops (6-15 m) (0)	0.52 (0.2)					
Rocky outcrops (15-21 m) (0)	6.4 (1.9)					
Rocky outcrops (21-33 m) (7)	20.96 (6.3)	21.8 ± 1.7	22.9 ± 1.2	20.1-24.1		
All habitats (122)	335.08 (100.0)	13.7 ± 0.6	15.0 ± 0.6	2.4-25.9	83 ± 7	17-188

Table 2-4. Sampling effort by habitat type and year in the Florida Keys (northern Biscayne National Park to Key West) and in Dry Tortugas region. First number = sites sampled (i.e. 200-m x 200-m grid cells), second number = proportion of total sampling effort.

Acropora coral surveys in the Florida Keys (northern Biscayne National Park to SW of Key West)

Habitat type	2005	2007	2012
Inshore patch reefs	4 (2.0)	1 (0.4)	8 (1.3)
Mid-channel patch reefs	47 (22.9)	35 (15.2)	153 (25.5)
Offshore patch reefs	27 (13.2)	42 (18.2)	122 (20.3)
Reef rubble	0 (0)	0 (0)	29 (4.8)
Low-relief hard-bottom (< 6 m)	18 (8.8)	25 (10.8)	52 (8.7)
Inner line reef tract	5 (2.4)	8 (3.5)	18 (3.0)
High-relief spur and groove	19 (9.3)	51 (22.1)	62 (10.3)
Low-relief hard-bottom (6-15 m)	23 (11.2)	15 (6.5)	42 (7.0)
Patchy hard-bottom in sand (6-15 m)	11 (5.4)	21 (9.1)	32 (5.3)
Low-relief spur and groove (6-15 m)	16 (7.8)	33 (14.3)	82 (13.7)
Low-relief spur and groove (15-22 m)	16 (7.8)	0 (0)	0 (0)
Low-relief spur and groove (22-27 m)	9 (9.3)	0 (0)	0 (0)

Non-Acropora coral surveys in the Florida Keys (northern Biscayne National Park to SW of Key West)

Habitat type	2005	2009	2012
Inshore patch reefs	4 (2.0)	4 (2.5)	8 (4.0)
Mid-channel patch reefs	47 (22.9)	22 (13.8)	25 (12.4)
Offshore patch reefs	27 (13.2)	28 (17.5)	29 (14.4)
Reef rubble	0 (0)	0 (0)	21 (10.4)
Low-relief hard-bottom (< 6 m)	18 (8.8)	17 (10.6)	12 (5.9)
Inner line reef tract	5 (2.4)	6 (3.8)	7 (3.5)
High-relief spur and groove	19 (9.3)	36 (22.5)	36 (17.8)
Low-relief hard-bottom (6-15 m)	23 (11.2)	11 (6.9)	12 (5.9)
Patchy hard-bottom in sand (6-15 m)	11 (5.4)	6 (3.8)	12 (5.9)
Low-relief spur and groove (6-15 m)	16 (7.8)	30 (18.8)	40 (19.8)
Low-relief spur and groove (15-22 m)	16 (7.8)	0 (0)	0 (0)
Low-relief spur and groove (22-27 m)	9 (9.3)	0 (0)	0 (0)

Coral surveys in the Dry Tortugas region

Habitat type	2006	2008
Patch reefs (< 6 m)	4 (8.7)	2 (4.7)
Patch reefs (6-15 m)	3 (6.5)	0 (0)
Low-relief hard-bottom (< 6 m)	0 (0)	3 (7.0)
Low-relief hard-bottom (6-15 m)	5 (10.9)	1 (2.3)
Low-relief hard-bottom (15-21 m)	3 (6.5)	1 (2.3)
Low-relief spur & groove (6-15 m)	0 (0)	3 (7.0)
Patchy hard-bottom (< 6 m)	0 (0)	0 (0)
Patchy hard-bottom (6-15 m)	5 (10.9)	4 (9.3)
Patchy hard-bottom (15-21 m)	3 (6.5)	2 (4.7)
Patchy hard-bottom (21-33 m)	0 (0)	0 (0)
High-relief spur & groove (6-15 m)	1 (2.2)	3 (7.0)
Reef knoll (6-15 m)	1 (2.2)	0 (0)
Reef knoll (15-21 m)	1 (2.2)	3 (7.0)
Reef terrace (15-21 m)	3 (6.5)	5 (11.6)
Reef terrace (21-33 m)	5 (10.9)	2 (4.7)
Medium-profile reef (6-15 m)	4 (8.7)	11 (25.6)
Medium-profile reef (15-21 m)	5 (10.9)	3 (7.0)
Medium-profile reef (21-33 m)	2 (4.3)	0 (0)
Rocky outcrops (15-21 m)	1 (2.2)	0 (0)

3. Florida Keys Individual Species Accounts: Results and Discussion

3.1 Introduction

The Biological Review Team (Brainard et al. 2011) noted that quantitative abundance estimates were available for only a few of the 82 candidate coral species, with coral percent cover, often only to genus level, most commonly reported in various monitoring programs. Further, the BRT noted that most of the candidate species from the wider Caribbean were too rare (e.g. *Dendrogyra cylindrus*) to document meaningful trends, or were identified only to genus (e.g. *Mycetophyllia and Agaricia*), or were often misidentified (e.g. *Montastraea annularis* complex).

Results presented below for the Florida Keys and Dry Tortugas contribute to the ESA review process because they provide, in some cases for the first time, population estimates over time, with variance terms, for the nine candidate species in the wider Caribbean. The data presented include multiple sampling events (2005 - 2012) and encompass most of the shallow-water (< 30 m) coral reef and hard-bottom habitats on the south Florida shelf, from south of Miami to the Dry Tortugas. Although this considerable data set includes information for each species on presence-absence, density, size classes, and condition, in a stratified design (e.g. habitat, along-shelf position, and management zone), the focus of the following results section is on domain-wide (total) population abundance estimates for the Florida Keys (Miami south to Key West) and the Dry Tortugas.

3.2 Total Population Estimates for Corals in the Florida Keys

Keys-wide population numbers for all coral species are provided in Figure 3-1, for three sample years, specifically 2005, 2009 and 2012. Species codes for Figures 3-1 and 3-2 are listed in Table 3-1. In 2012, with an estimate of over 2.4 billion scleractinian corals in the Florida Keys, the top-ten most abundant species account for over 92% of all scleractinian coral colonies in the Keys, as defined by the boundaries of the Florida Keys National Marine Sanctuary (excluding the Dry Tortugas). The next ten most abundant species account for just over six percent, while the remaining 19 species comprise the remainder. The top five species have population numbers that measure in the hundreds of millions. The most abundant species in all three years, *Siderastrea siderea*, was estimated at nearly 800 million colonies in 2012. The nine ESA candidate species are found throughout the rankings, without any correlation to abundance. One ESA candidate species for ESA Threatened status, *Dichocoenia stokesi*, is a top ten species for all years sampled, with over 80 million colonies estimated in 2012. *Montastraea faveolata* is a candidate species for ESA Endangered status and is ranked in the top ten for the last two sample periods, with over 47 million colonies estimated in 2012. Another *Montastraea* species, *M. annularis* is ranked 12th and 11th, with over 24 million colonies estimated in 2012. Notable is *Acropora cervicornis*, ranked as high as 15th, with over 9 million colonies estimated in 2012, which is proposed for re-classification to Endangered status. The other candidate species fall out in the middle to lower third of the ranking.

For the Dry Tortugas, population numbers are provided in Figure 3-2, for two sample years, 2006 and 2008. Differences in the ranking with results from the Keys-wide data are largely due to the fact that we sampled more sites in deeper coral reef habitats, including the deep terrace habitat in the Dry Tortugas this is not found in the Florida Keys (Table 2-4). In 2006, with over 720 million colonies estimated in the Dry Tortugas, the top ten most abundant species accounted for 87% of all coral colonies. The next ten most abundant species accounted for just over ten percent, with the next 22 species comprising the remainder. The most abundant species both years, *Porites astreoides and Montastraea cavernosa*, were estimated at over 100 million colonies in both years. *Siderastrea siderea*, the most common coral in the Keys, was ranked third in the Dry Tortugas. In 2008, over 560 million colonies were estimated in the Dry Tortugas, with the top ten species accounting for 86% of all coral colonies. The next ten species accounted for about 12% of all colonies, and the remaining 19 species comprised the remainder. As for the Florida Keys, it is apparent that the nine ESA candidate species in the Dry Tortugas are found throughout the rankings without any correlation to abundance. Of particular note are *M. faveolata* and *M. franksi*, both of which are candidate species for Endangered status, which are top-ten species in the Dry Tortugas area in both 2006 and 2008. In addition, *Agaricia lamarcki*, uncommon throughout the Florida Keys (related to less sampling in deeper habitats), is ranked 11th in 2006 and 21st in 2008. The lower number in 2008 is due primarily to less sampling in deeper (> 20 m) hard-bottom and coral reef habitats (Table 2-4).

Lower total numbers of corals in the Dry Tortugas is partly explained by its smaller total area (335 km^2), about two thirds the area of the Florida Keys (503 km^2). In addition, there are substantial differences in the area and proportion of different hard-bottom and coral reef habitat types between the Dry Tortugas and the rest of the Florida Keys (FMRI 1998, Franklin et al. 2003). For example, continuous and patchy hard-bottom, which does not support high densities of many corals, in the Dry Tortugas region comprises about 72% of the total hard-bottom and reef habitat available, while in the rest of the Florida Keys is about 34% (Table 2-3).

Table 3-1. Scleractinian coral species codes for Figures 3-1 and 3-2.

ACRV = *Acropora cervicornis*	LCUC = *Leptoseris cucullata*	MDAN = *Mycetophyllia danaana*
APAL = *Acropora palmata*	MCAR = *Madracis carmabi*	MFER = *Mycetophyllia ferox*
AAGR = *Agaricia agaricites*	MDEC = *Madracis decactis*	MLAM = *Mycetophyllia lamarckiana*
AFRA = *Agaricia fragilis*	MFOR = *Madracis formosa*	ODIF = *Oculina diffusa*
AHUM = *Agaricia humilis*	MMIR = *Madracis mirabilis*	PAST = *Porites astreoides*
ALAM = *Agaricia lamarcki*	MSEN = *Madracis senaria*	PBRA = *Porites branneri*
CARB = *Cladocora arbuscula*	MARE = *Manicina areolata*	PCOL = *Porites colonensis*
CNAT = *Colpophyllia natans*	MMEA = *Meandrina meandrites*	PDIV = *Porites porites divaricata*
DCYL = *Dendrogyra cylindrus*	MALC = *Millepora alcicornis*	PFUR = *Porites porites furcata*
DSTK = *Dichocoenia stokesi*	MCOM = *Millepora complanata*	PPOR = *Porites porites porites*
DCLV = *Diploria clivosa*	MANN = *Montastraea annularis*	SCOL = *Scolymia* spp.
DLAB = *Diploria labyrinthiformis*	MCAV = *Montastraea cavernosa*	SRAD = *Siderastrea radians*
DSTR = *Diploria strigosa*	MFAV = *Montastraea faveolata*	SSID = *Siderastrea siderea*
EFAS = *Eusmilia fastigiata*	MFRA = *Montastraea franksi*	SBOU = *Solenastrea bournoni*
FFRG = *Favia fragum*	MANG = *Mussa angulosa*	SHYA = *Solenastrea hyades*
IRIG = *Isophyllastrea rigida*	MALI = *Mycetophyllia aliciae*	SMIC = *Stephanocoenia michelini*
ISIN = *Isophyllia sinuosa*		

Figure 3-1. Rank-order abundance of all scleractinian coral species in the Florida Keys, exclusive of the Dry Tortugas, during 2005, 2009 and 2012. Species marked with an asterisk (*) are ESA-candidate species.

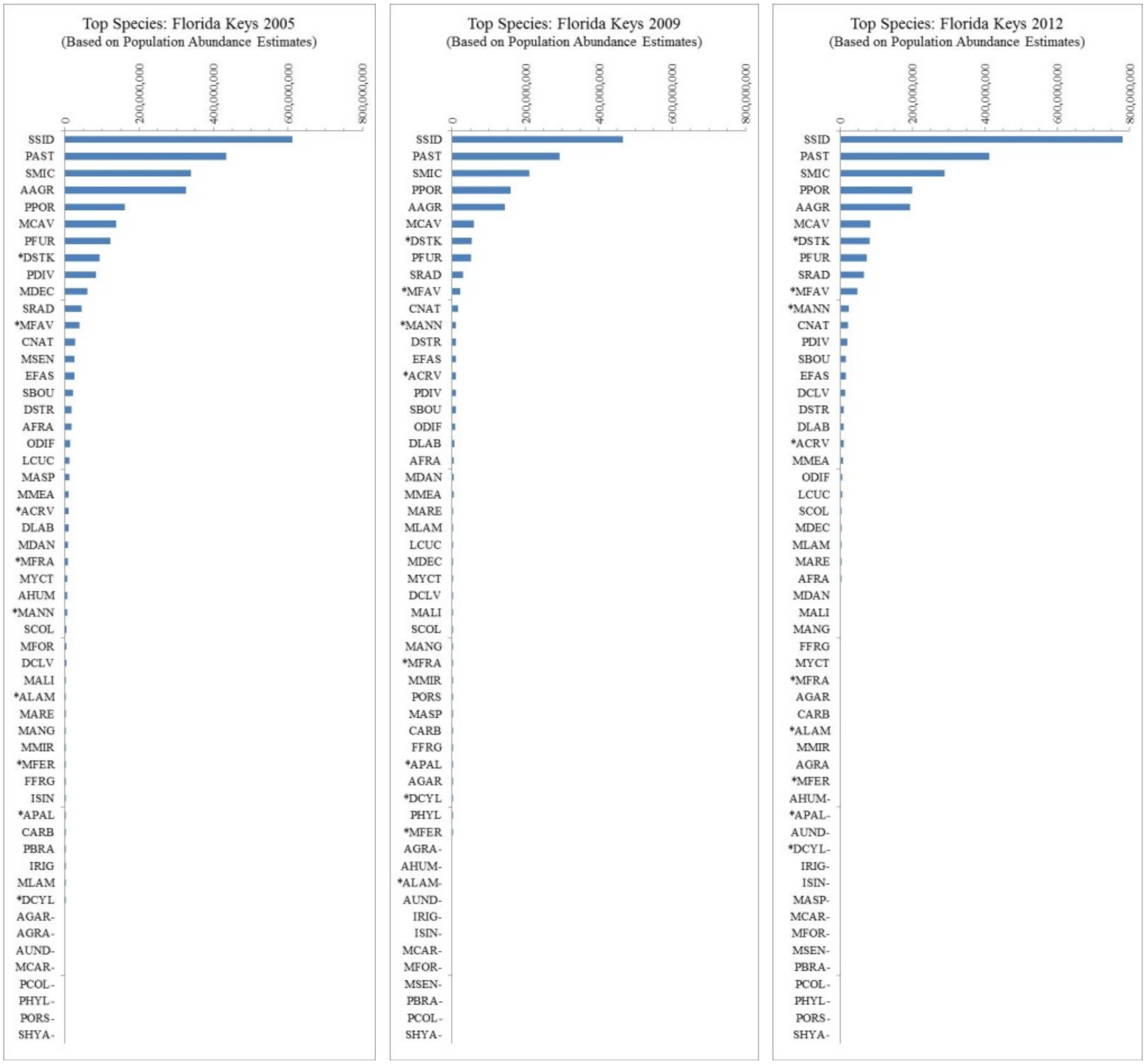
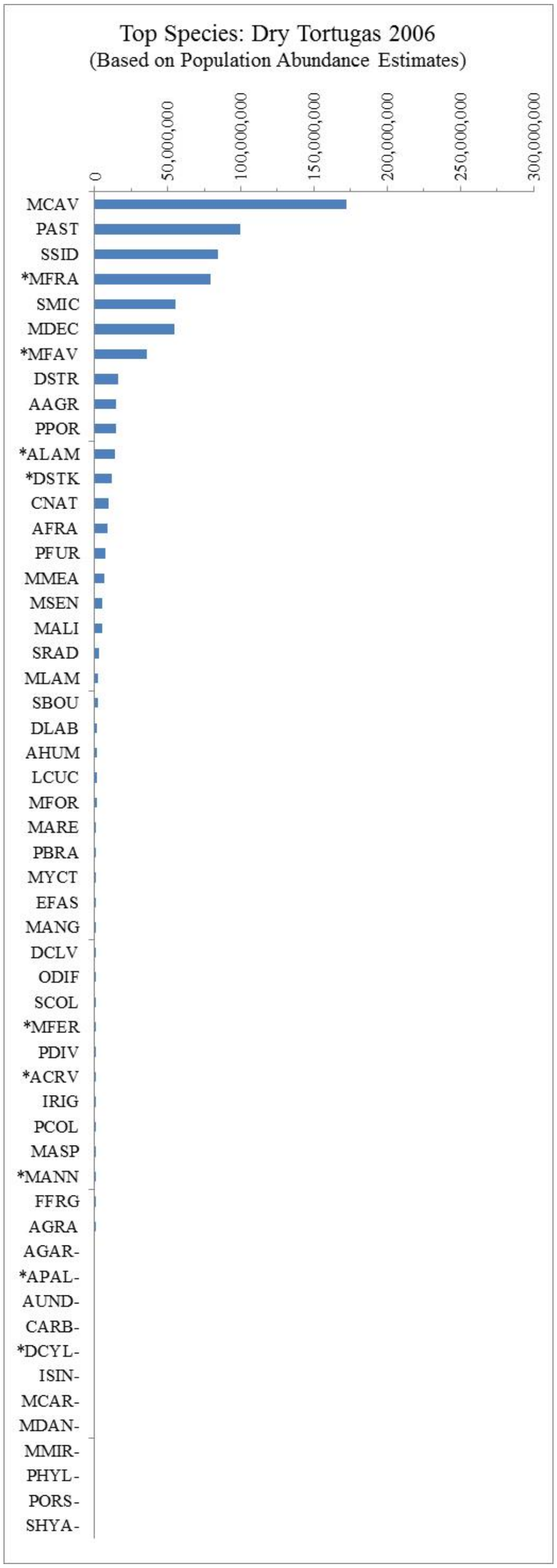


Figure 3-2. Rank-order abundance of all scleractinian coral species in the Dry Tortugas region during 2006 and 2008. Species marked with an asterisk (*) are ESA-candidate species.



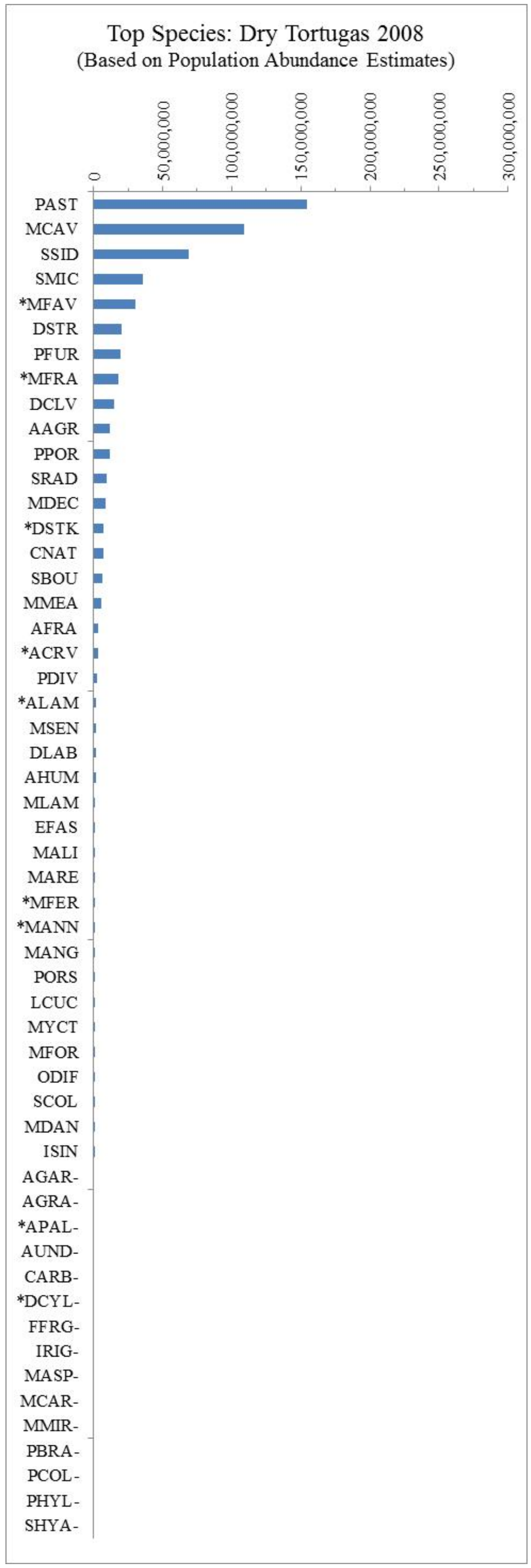


Table 3-2. Population abundance estimates (± 1 standard error) of *Acropora* corals (staghorn coral, *A. cervicornis* and elkhorn coral, *A. palmata*) in the Florida Keys during (top) and Dry Tortugas. Precision (coefficient of variation = SE/mean * 100) estimates are show below the population abundance estimates.

Florida Keys (northern Biscayne National Park to SW of Key West)

Species	**2005**	**2007**	**2012**
Acropora cervicornis	10,217,794 ± 4,579,629 (44.8)	6,925,400 ± 2,413,721 (34.9)	10,022,743 ± 3,129,312 (31.2)
A. palmata	551,000 ± 463,326 (84.1)	1,013,704 ± 337,209 (33.3)	467,614 ± 258,111 (55.2)

Dry Tortugas region

Species	**2006**	**2008**
Acropora cervicornis	358,000 ± 358,000 (100.0)	3,532,900 ± 2,923,368 (82.7)
A. palmata	0 ± 0	0 ± 0

Table 3-3. Population abundance estimates (± 1 standard error) of non-*Acropora*, ESA-candidate scleractinian coral species in the Florida Keys (top) and Dry Tortugas region (bottom) during 1999-2012. Precision (coefficient of variation = SE/mean * 100) estimates are show below the population abundance estimates.

Florida Keys (northern Biscayne National Park to SW of Key West)

Species	**2005**	**2009**	**2012**
Dichocoenia stokesi	92,770,519 ± 13,090,398 (14.1)	53,791,417 ± 9,694,112 (18.0)	81,622,462 ± 10,011,077 (12.3)
Montastraea annularis	5,649,281 ± 2,704,814 (47.9)	11,500,050 ± 4,504,510 (39.2)	24,286,629 ± 12,404,060 (51.1)
M. faveolata	39,693,866 ± 8,052,925 (20.3)	21,931,600 ± 6,974,491 (31.8)	47,277,937 ± 14,510,239 (30.7)
M. franksi	8,020,284 ± 3,543,905 (44.2)	262,900 ± 244,348 (92.9)	376,986 ± 366,141 (97.1)
Agaricia lamarcki	3,064,250 ± 1,310,063 (42.7)	0 ± 0	210,400 ± 210,400 (100.0)
Mycetophyllia ferox	985,123 ± 701,587 (71.2)	9,500 ± 9,499 (100.0)	7,000 ± 6,999 (100.0)
Dendrogyra cylindrus	22,800 ± 22,800 (100.0)	54,400 ± 54,399 (100.0)	0 ± 0

Dry Tortugas region

Species	**2006**	**2008**
Dichocoenia stokesi	12,101,600 ± 4,129,871 (34.1)	7,112,534 ± 1,055,946 (14.8)
Montastraea annularis	0 ± 0	476,679 ± 348,137 (73.0)
M. faveolata	36,121,978 ± 4,814,250 (13.3)	29,966,218 ± 3,336,944 (11.1)
M. franksi	79,030,861 ± 19,028,585 (24.1)	18,173,977 ± 4,087,802 (22.5)
Agaricia lamarcki	14,350,464 ± 2,636,519 (18.4)	2,135,758 ± 518,565 (24.3)
Mycetophyllia ferox	485,500 ± 395,073 (81.4)	480,222 ± 215,973 (45.0)
Dendrogyra cylindrus	0 ± 0	0 ± 0

3.3 Population Estimates for the Nine ESA Candidate Species in the Florida Keys

Abundance and size class frequency data for the nine ESA candidate species are presented in Table 3-2 for *Acropora palmata* and *A. cervicornis*, and Table 3-3 for the other seven candidate species. Size class frequency data are presented in Figures 3-3 through 3-5 for *A. palmata* and *A. cervicornis,* and Figures 3-7 through 3-17, for the other seven candidate species. Total abundance and abundance by size classes for *A. palmata* and *A. cervicornis* are presented for 2005, 2007, and 2012. For the other seven ESA candidate species, abundance and size class frequency data are presented for the Florida Keys, exclusive of the Dry Tortugas, for the years 2005, 2009, and 2012. Results from the Dry Tortugas are presented for all ESA candidate species, for the years 2006 and 2008.

3.4 Genus *Acropora* (Family Acroporidae)

The Threatened ESA listing in 2006 for *Acropora palmata* and *A. cervicornis* was based on an assessment of stressors that identified disease, elevated sea surface temperature and hurricanes as threats severe enough to put the two species at risk of extinction within the foreseeable future (*Acropora* BRT 2005). The threats were not considered severe enough to warrant an ESA Endangered (imminent threat of extinction throughout all or part of its range) listing due to an assessment that concluded that existing populations contained high numbers of individuals, the large geographic range of the species was intact, and reproductive potential (sexual and by fragmentation) was considered sufficient to sustain the populations (Hogarth 2006).

The proposed reclassification of *Acropora palmata* and *A. cervicornis* to Endangered in 2012 (Federal Register 2012) is based on a general assessment that determined their high susceptibility to bleaching caused by warming sea surface temperatures, stress from ocean acidification that may be occurring, disease, and continued threats from local stressors. Further, the Federal Register Report cites numerous references that appeared after 2006 to suggest that: (1) populations continue to decline (Lundgren 2008, Muller et al. 2008, Williams et al. unpublished data; Williams et al. 2008, Colella et al. 2012, Rogers and Muller et al. 2012); (2) sexual recruitment is limited (Williams et al. 2008); (3) thermal stress can impair recruitment (Randall and Szmant 2009); (4) ocean acidification can negatively impact recruitment in *A. palmata* (Albright et al. 2012); and (5) genetic variability is reduced in existing populations (Baums et al. 2006). The Federal Register Report also notes that the geographic ranges of both species have not changed, but considers the loss of local populations likely, which will reduce their geographic ranges. Contrary to the 2005 *Acropora* Biological Review Team (BRT) that considered the range of these two species geographically large, the most recent BRT (Brainard et al. 2011) determined that the entire Caribbean is sufficiently limited in geographic scale to be a factor that increases the extinction risk of all corals, including both Atlantic *Acropora* species. However, this notion that corals in the Caribbean are at higher risk because of its limited size is at odds with genetic data. For instance, Baums et al. (2005, 2006) identified two isolated populations of *A. palmata* in the Caribbean. They noted that because the eastern and western *A. palmata* populations appear to differ in their genotypic diversity and may also differ demographically, conservation strategies need to be tailored to local conditions. Similarly, Vollmer and Palumbi (2007) noted that for *A. cervicornis* targeted conservation efforts need to be scaled to encompass smaller, not larger, areas. Finally, Murdoch and Aronson (1999) noted that the coral assemblages of the Florida Reef Tract exhibited different degrees of variability in cover depending on the spatial scale surveyed. They found that coral cover varied little within and among sites on a single reef, yet it varied substantially from reef to reef. These data show that while it is clear that regional-scale processes such as bleaching and disease are acting on all these reefs simultaneously, no two reefs or areas respond the same to these disturbances.

While the BRT (Brainard et al. 2011) and Federal Register proposal (2012) included a few references that suggest that Atlantic *Acropora* coral recovery may be occurring to a limited extent, there is an extensive literature that suggests recovery and persistence may be more significant. For example, Keck et al. (2005) describe an abundant population of *Acropora cervicornis* in Roatán (Honduras). Schelton et al. (2006) determined that *A. palmata* populations in South Caicos, Turks and Caicos Islands, southeastern Bahamas, were in relatively good condition. Mayor et al. (2006) discuss the recovery of *A. palmata* at Buck Island, St. Croix, U.S. Virgin Islands, previously devastated by white band disease. Precht and Aronson (2006) discuss the details of recovering *Acropora* populations along the north coast of Jamaica and they also highlight the locations of recovering *Acropora* populations throughout the Caribbean; Idjadi et al. (2006) monitored recovery of *A. cervicornis* in Jamaica, suggesting the persistence of large *Montastraea* colonies may have helped facilitate the process; Zubillaga et al. (2008) identified recovering *A. palmata* in Los Roques, Venezuela, based upon population and genetic data; Riegl et al. (2008) described stands of *A. cervicornis* off Honduras, where offshore populations exhibited good survivorship after a major bleaching event in 2008; Lirman et al. (2010) described a “megapopulation” of *A. cervicornis* in the Dominican Republic, measuring 2 ha with large interlocking colonies; Grober-Dunsmore et al. (2006) and Rogers and Muller (2010) monitored recovering *A. palmata* populations in St. Johns, USVI, but expressed concern about disease and bleaching; Macintyre and Toscano (2007) identified recovering populations of *A. palmata* along the Belizean Barrier Reef; S. Griffin (pers. comm.) reported a significant and vibrant stand *A. palmata* in Puerto Rico (Vega Baja Reef); Lidz and Zawada

(2013) reported *A. cervicornis* at Pulaski Shoal, Dry Tortugas, where it was previously absent. Northward expansion of *A. cervicornis* in Florida to include large stands of *A. cervicornis* (Vargas-Angel et al. 2003) was discussed by Precht and Aronson (2004) as possibly related to present-day warming. The relatively recent discovery of *A. palmata* at the Flower Garden Banks National Marine Sanctuary (Zimmer et al. 2006) is also notable. These references suggest that since the eastern Caribbean coral bleaching event of 2005, and in the northern Florida Keys localized Hurricane damage during 2005 (Williams et al. 2008), there has been a general trend of *Acropora* stasis and recovery, including the expansion of local populations throughout the Caribbean.

It is also noteworthy that while coral bleaching and ocean acidification remain serious threats, the acclimatization potential of corals to increased temperatures is an active area of research (Jones and Berkelmans 2010, Baker et al. 2004), with a focus on identifying heat-resistant phenotypes, and corals have metabolic mechanisms that offer resilience to lowered carbonate saturation state, resulting from increased pH (McCulloch et al. 2012), suggesting a complicated story. Disease resistance in *Acropora cervicornis* was also described, with approximately 6% of the genotypes studied resistant to White Band Disease (Vollmer and Kline 2008). Genetic variability was also considered a concern contributing to extinction risk in *A. palmata* and *A. cervicornis*, but Reyes and Schiza (2010) described significant fine-scale genetic variability in *A. palmata* and *A. cervicornis* in Puerto Rico, and Hemmond and Vollmer (2010) measured high levels of standing genetic diversity in Florida, relative to the greater Caribbean, suggesting Florida populations have sufficient genetic variation to be viable and resilient to environmental disturbance and disease. *A. palmata* is also reported to have mechanisms that help maintain genomic integrity, in the face of considerable stress experienced over potentially long life spans, which contributes to its evolutionary success (Polato et al. 2011).

Finally, it is important to consider the geological context for these two species in Florida, because it helps explain aspects of their recent ecology. When conditions have been favorable in Florida, as they were during the middle Holocene, *Acropora* corals have dominated the shallow-reef community. When conditions have deteriorated, as in the Pleistocene, head corals have dominated and persisted. For instance, during the last interglacial period (~125 thousand years ago) conditions across the south Florida platform were inimical to the growth of *Acropora* corals (Precht and Miller 2007) yet they are still found, albeit rare, as fossils in the Key Largo Limestone (Precht and Goodwin 2010). Species replacements and range expansions in the past, especially of the *Acropora* corals, emphasize the resilience of reef ecosystems and the individualistic responses of coral species to rapid environmental change (Aronson and

Precht 2001a; Precht and Miller 2007). For Florida at least, the present reef community assemblage, highlighted by diminished *Acropora* populations, is not unique in space or in time.

In southeastern Florida, a series of submerged, shore-parallel, fossil reef terraces reveal a precedent for the recent range expansion of *Acropora* (Precht and Aronson, 2004). This nearly continuous barrier reef system extended northward from Miami to Palm Beach County in the early to middle Holocene (Banks et al. 2007, Finkl and Andrews 2008). The internal architectures of these reefs are replete with *Acropora* corals and the shore-parallel terraces represent a series of back-stepped reefs (Precht et al., 2000). During the Holocene thermal maximum (COHMAP 1988, Ruddiman and Mix 1991, Lin et al. 1997, Kerwin et al. 1999, Haug et al. 2001), SSTs were warmer than today in the western Atlantic, and during this period *Acropora*-dominated reefs were common along the southeastern coast of Florida (Lighty 1977, Lighty et al. 1978, Precht and Aronson 2004). In apparent response to climatic cooling in the late Holocene (de Menocal et al. 2000, Jessen et al. 2005), the northern limits of the *Acropora* species contracted 150 km south to Fowey Rocks (Precht and Aronson 2004). In historical times, Fowey Rocks was the northernmost emergent reef of the Florida reef tract, as well as the northernmost extent of *A. palmata* (Vaughan 1914, Jaap 1984, Porter 1987, Shinn et al. 1989, ABR Team 2005). Similar range expansion and contraction of a barrier reef dominated by *A. palmata* was noted off Abaco Island in the northernmost Bahamas (Lighty et al. 1980, Macintyre 2007).

The above citations and results, plus the population data presented below, do not support the proposal to reclassify the two Atlantic *Acropora* species as Endangered. There is little data to suggest that the population status of the two species has significantly changed since 2006, and there is added evidence that recovery has occurred at multiple locations. Further, successful coral restoration projects, throughout southeast Florida and the Caribbean (Johnson et al. 2011, Hollarsmith et al. 2012; Precht et al. 2012), including established nurseries that contain potentially thermal tolerant genotypes (Bowden-Kerby and Carne 2012), suggest that despite the continued presence of all the stressors identified by the BRT (Brainard et al. 2011), these corals thrive in offshore nurseries and survive when transplanted to multiple reef locations.

3.4.1 *Acropora cervicornis* (Lamarck, 1816)

Populations of *Acropora cervicornis* are known to have gone through boom-bust cycles in the Florida Keys (Jaap 1998). For instance, the longest historical record available for the Florida reef tract is derived from maps of community distributions in the Dry Tortugas, meticulously prepared by Alexander Agassiz in 1881 and then redrawn from new surveys in 1976 (Agassiz 1883, Davis 1982). A number of interesting changes in coral community structure occurred over that century, all of which could be attributed to natural system variability. Thousands of square meters of *A. palmata* were alive in 1881, but they were largely reduced to alga-covered rubble by the late 1970s, with only two living patches (~600 m^2) in 1976. *A. cervicornis* cover was not common in the late 1880s, according to Agassiz's map; it dominated many locations by 1976 (Davis 1982), only to suffer >90% morality during the winter cold fronts of 1977 described earlier (see Porter et al. 1982). Since our surveys began, population estimates for *Acropora cervicornis* in the Florida Keys appear stable and large (Table 3-2), with approximately 10 million (SE 3.1 million) colonies in 2012. There is no evidence of continued decline since the 2006 ESA Threatened Listing. The size structure of the population also remains unchanged over the period of our study in the Florida Keys (Figure 3-3), with most of the population comprised of relatively small colonies.

In the Dry Tortugas (Figure 3-4), we have sampled less frequently and at fewer sites, consequently the variance terms are quite large, with 3.5 million (SE 2.9 million) colonies in 2008. We have not yet visited the Dry Tortugas with an optimized sampling scheme for *Acropora cervicornis,* as we have done in the Florida Keys for 2007 and 2012. Still, it is apparent that in the Dry Tortugas, after large declines caused by the same factors responsible for decline throughout the Caribbean, plus a major mortality event from cold water (Porter et al. 1982), that remnant populations exist, with evidence of recovery (Lidz and Zawada 2013). However, most colonies are relatively small, less than 60 cm in maximum diameter.

In the Florida Keys and Dry Tortugas, partial mortality (the percent of dead tissue on a colony) is variable among size classes and among years (Figures 3-3 and 3-4). In the Florida Keys in 2005, partial mortality increased with size class of corals and was highest among the largest colonies. Similar patterns were observed in 2007 and 2008, though percentages were highest in 2005. Partial mortality in the Dry Tortugas appeared similar to that seen in the Florida Keys, but because few sites were visited the variance terms are high. More work is needed to accurately describe the population. Multiple factors are responsible for partial mortality. For *Acropora cervicornis* in 2007, no disease was evident, but 1.9% of colonies exhibited signs of damselfish predation, and 2.6% in 2012; snail predation was 1.3% in 2012.

The impact of these various stressors may restrict the development of larger colonies and ultimately thickets, since partial mortality has greater impact on smaller colonies (see Precht et al. 2010). Still, substantial population numbers remain in southeast Florida millions of colonies), along with sufficient standing genetic diversity and connectivity among populations, relative to the greater Caribbean, to suggest that Florida populations are viable and resilient to environmental disturbance and disease

(Hemmond and Vollmer 2010). The presence of such large population numbers in southeast Florida, the fact that *Acropora cervicornis* is known to contain genotypes resistant to white band disease (Vollmer and Kline 2008), the probable expansion northward of viable populations (Vargas-Angel et al. 2003, Precht and Aronson 2004), the success that restoration activities are having in the region (Grablow et al. 2010, 2011, Nedimyer et al. 2010, Precht and Nedimyer 2010, Precht et al. 2012), and increasing evidence that populations are recovering at multiple locations throughout the Caribbean (Keck et al. 2005, Precht and Aronson 2006, Idjadi et al. 2006, Riegl et al. 2008, Lirman et al. 2010, Lidz and Zawada 2013), suggests that a reclassification to Endangered status is not supported by the population data and is thus unwarranted.

Figure 3-3. *Acropora cervicornis* colony abundance by skeletal unit size class (max. diameter, cm) in the Florida Keys (northern Biscayne National Park to SW of Key West) during 2005 (top), 2007 (middle), and 2012 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.

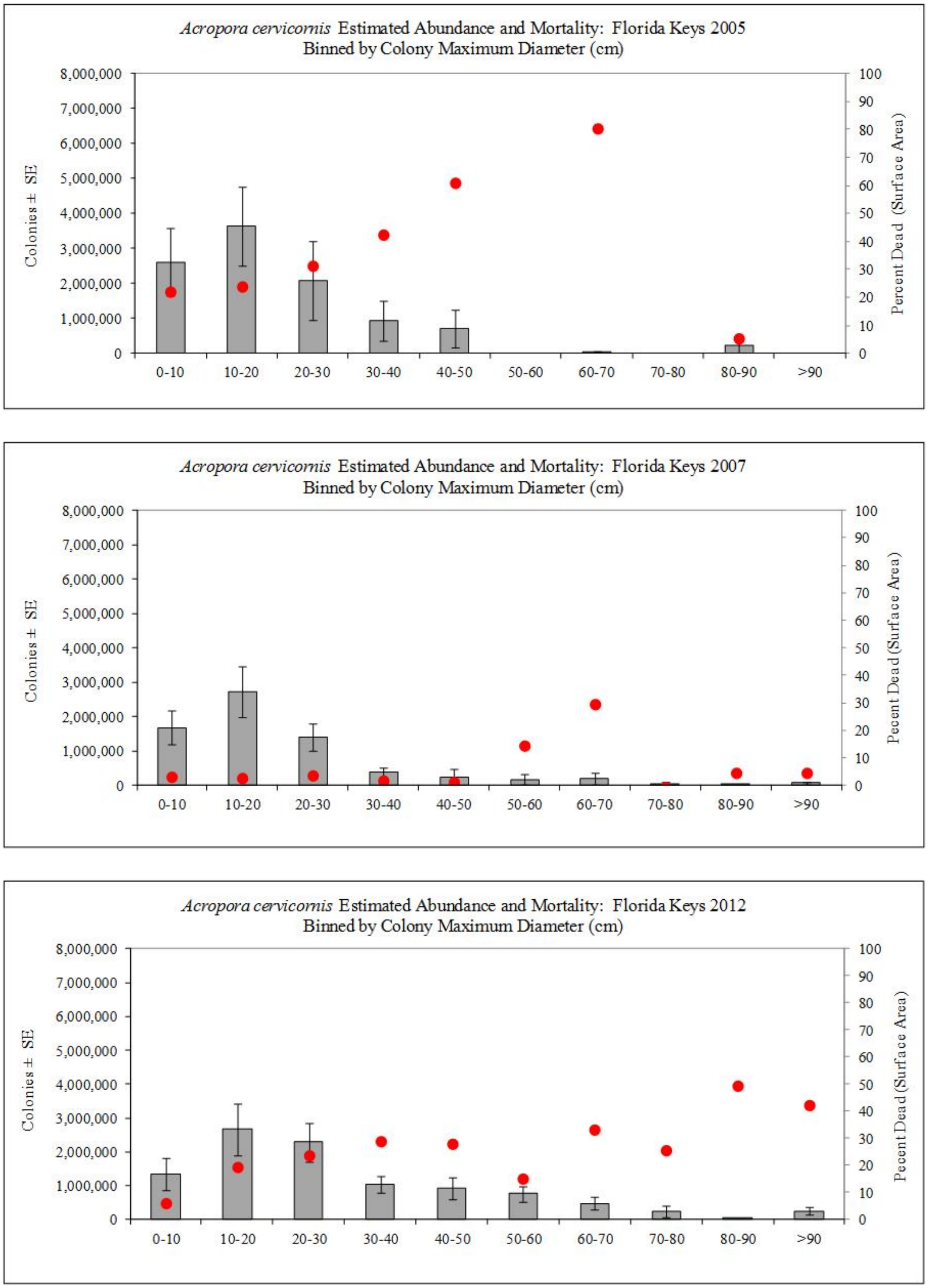
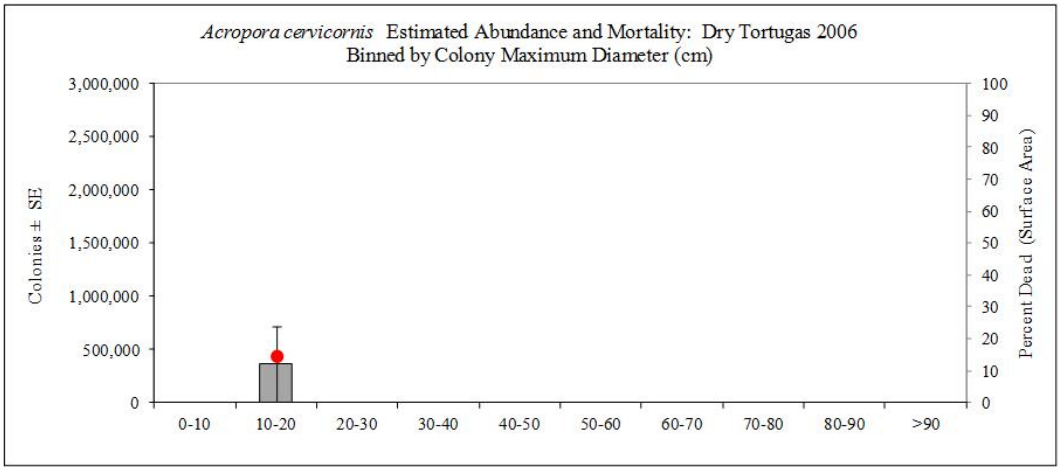
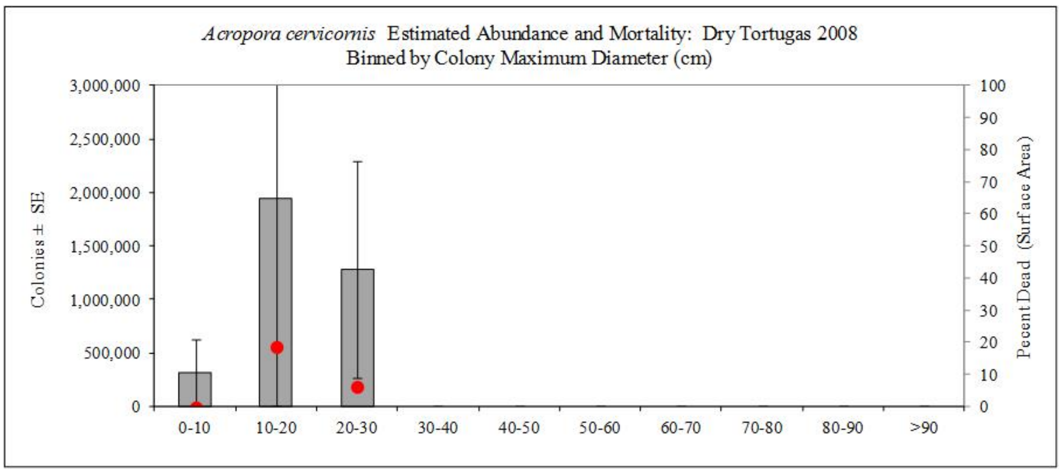


Figure 3-4. *Acropora cervicornis* colony abundance by skeletal unit size class (max. diameter, cm) in the Dry Tortugas region during 2006 (top) and 2008 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.





3.4.2 *Acropora palmata* (Lamarck, 1816)

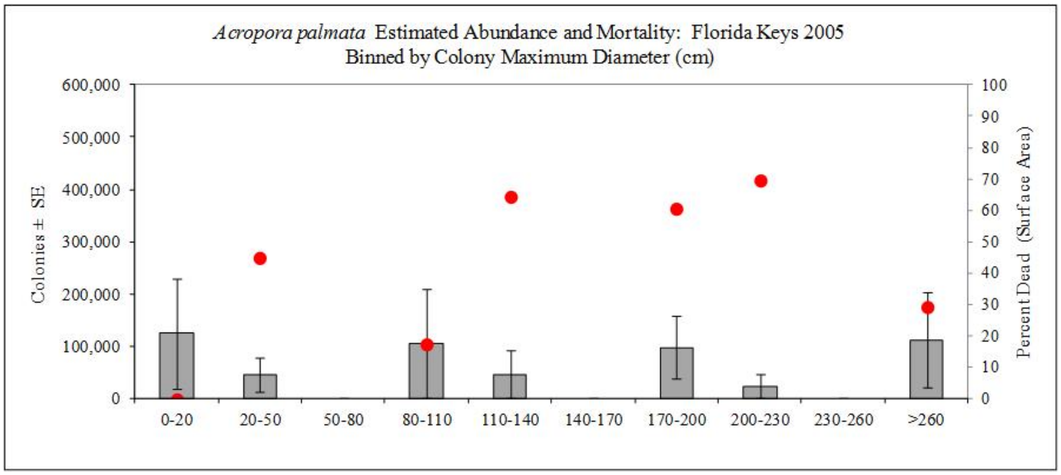
The U.S. Environmental Protection Agency (EPA), in conjunction with the Florida Marine Research Institute (FMRI) and NOAA, started the first Keys-wide status-and-trends program in 1996 (Jaap et al. 2001). This Coral Reef Ecosystem Monitoring Program (CREMP) documented coral cover during the period 1996 to the present. Although it was spatially extensive, the program began long after substantial coral loss had already occurred, particularly the demise of the branching corals *Acropora palmata* and *A. cervicornis*. Of the 40 stations in the study, only seven contained *A. palmata* in 1996. At these seven sites, the average loss of *A. palmata* was 85% between 1996 and 1999 (Patterson et al. 2002). These results might appear alarming; however, they need to be placed in perspective. The average percent cover of *A. palmata* at the seven sites was around ~5% in 1996 and had declined to ~2% by 1999, which is an absolute change in coral cover of only 3%. One of the stations sampled included the reef at Rock Key. There, *A. palmata* cover dropped from <4% to slightly less than 2%. This is the same reef that Enos (1977) described as having no living *A. palmata* in the early 1970s. It could well be that these small populations of *A. palmata* represent residual, transient, or marginal populations that are highly susceptible to both biotic and physical disturbances including disease, predation, storms, cold fronts, and coral bleaching events. Interestingly, the period 1996 to 2000 of the Jaap and Patterson studies spanned two major coral bleaching events in 1997 to 1998, continued outbreaks of pests and disease, and the passage of a major hurricane directly over many of the sampling stations. Unfortunately, in the case of the CREMP data, reliance on results from a few locations makes it difficult to expand their significance to regional population status and trends (Porter et al. 2012). This also applies to their species richness and frequency of occurrence data.

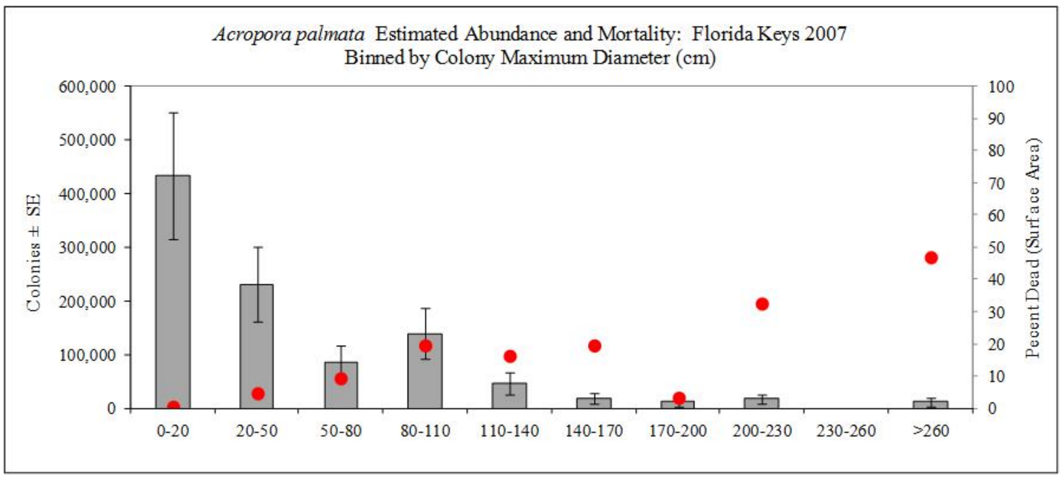
Population estimates for *Acropora palmata* in the Florida Keys appear stable since 2005 (Table 3-1), but remain much reduced overall since declines started in the late 1970s, due to a combination of factors that are well known (Precht and Miller 2007). In 2012, we estimated 467,000 (SE 258,000) colonies in the Florida Keys, with a relatively high coefficient of variance of 55. Relative to the abundance of other corals in the region, *A. palmata* is among the least abundant, ranking among corals that are naturally rare in abundance (Figure 3-1). Our sampling program did not detect any *A. palmata* in the Dry Tortugas during 2006 or 2008 although a few colonies were detected during an early sample effort that was part of a pilot program in 1999. Shinn et al. (1977) noted that *A. palmata* was historically exceedingly rare in the Dry Tortugas. This contrasts dramatically with the historical condition in Florida, where *A. palmata* was previously a major framework-building species of the shallow-most reef crest and shallow high-relief spur-and-groove habitats (Jaap 1984; Shinn et al. 1989). Our population estimates have relatively high variance terms associated with the means, which reflects the difficulty of sampling species that are uncommon to rare, plus are restricted to a few locations (clumped).

The size class distribution of the Florida Keys population includes both small and large individuals (Figure 3-5), with a majority in the smaller size classes since 2007. Partial mortality was high across all size classes in 2005, reduced somewhat in 2007 with higher mortality seen in larger colony sizes, and then high again in 2012 with the smallest and largest sized colonies exhibiting less partial mortality than those in the middle size classes (Figure 3-5). Factors affecting *A. palmata* in the Florida Keys are well described (Williams and Miller 2011). During 2012, of the *A. palmata* colonies measured, no disease was evident, but 5.5% were impacted by active damselfish predation, 1.1% by snail predation, and 7.1% by *Cliona* sponge boring.

Relative to *Acropora cervicornis*, the population status of *A. palmata* in south Florida is significantly different. First, there is a two-order of magnitude difference in population size. In addition, *A. palmata* exhibits a much more limited habitat distribution compared to its congener, with most of the population reduced to a handful of high-density thickets (Miller et al. 2008). The preferred habitat type for *A. palmata*, today and historically, is largely limited to shallow high-relief spur-and-groove habitats, which comprise a small percentage (< 2%) of the total area in the Florida Keys (Table 2.3, see FMRI 1998). These areas are the named reefs found on nautical charts, about two dozen in total. Today, *A. palmata* is limited to perhaps a dozen of these sites, where remnant populations of large colonies and thickets remain (see L. Precht et al. 2010), such as South Carysfort Reef, Elbow Reef, Horseshoe Reef, French Reef, Molasses Reef, Sand Island, Grecian Rocks, and Looe Key (Figure 3-6). Note that all of these reefs, except for Looe Key are in the upper Florida Keys (discussed in Precht and Miller 2007). This contrasts with the distribution of *A. cervicornis*, which is found throughout the Keys and in multiple habitat types. On a positive note, *A. palmata* has recently become a focus of coral restoration efforts (Nedimeyer et al. 2010), with increasingly large numbers growing and thriving in offshore nurseries and with successful transplants recently made to offshore reefs in the upper Keys. Because of large population declines throughout its range and its restricted shallow habitat distribution, we agree with the 2006 assessment (Hogarth 2006) to list this species as Threatened under the ESA (Precht et al. 2004). Since 2006, however, this species has been relatively stable in Florida and there is no new data that warrants the re-reclassification of this coral to Endangered.

Figure 3-5. *Acropora palmata* colony abundance by skeletal unit size class (max. diameter, cm) in the Florida Keys (northern Biscayne National Park to SW of Key West) during 2005 (top), 2007 (middle), and 2012 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.





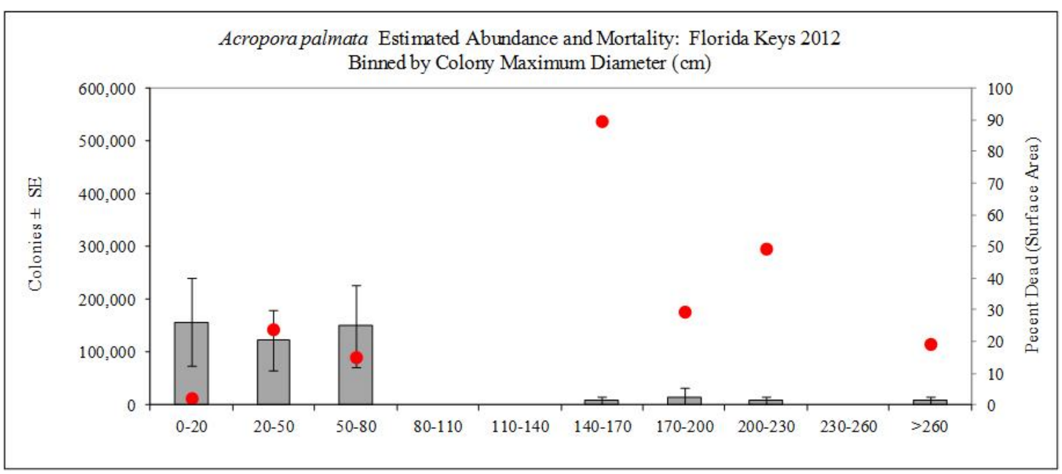
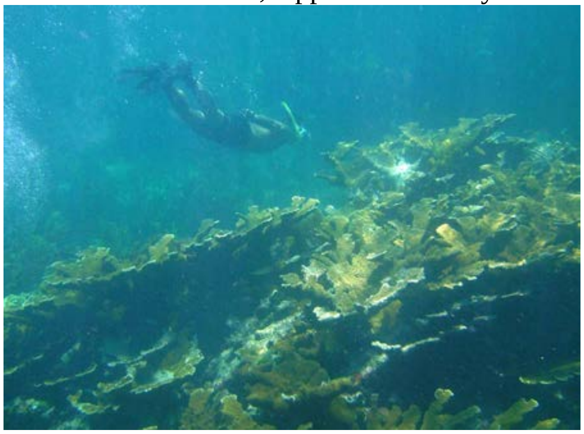


Figure 3-6. Examples of extant thickets of elkhorn coral (*Acropora palmata*) in the Florida Keys. Although specific reefs (e.g. the reef flat at Carysfort) experienced declines in elkhorn coral due to the disease in the 1970s, many stands persist at several reefs along the Florida Reef Tract.

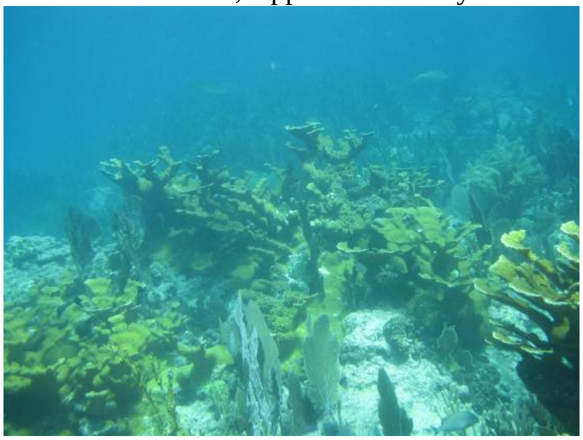
South Carysfort Reef, Upper Florida Keys

Horseshoe Reef, Upper Florida Keys



Grecian Rocks, Upper Florida Keys

Elbow Reef, Upper Florida Keys



3.5 Genus *Dichocoenia* (Family Meandrinidae)

3.5.1 *Dichocoenia stokesi* Milne Edwards and Haime, 1848

Dichocoenia stokesi is referenced as uncommon by the BRT (Brainard et al. 2011, citing Vernon 2002). The BRT also notes that the coral is the ninth most abundant coral in south Florida (Wagner et al. 2010). White Plague Type II disease was identified as a major threat to *D. stokesi* (Richardson et al. 1998), where an epidemic in the Florida Keys was reported to kill 75 percent of all colonies at several reefs in 1995 (Richardson et al. 1998, Richardson and Voss 2005). It is this apparent susceptibility to disease that identified *D. stokesi* as a candidate species for ESA listing as Threatened, where the BRT stated that the species was unlikely to recover from dramatic declines caused by disease, when combined with bleaching events. It is important to note that the declines referenced by the BRT occurred only in Florida and results were from only several reefs. Similar declines were not reported Keys-wide or throughout the Caribbean. In addition, this coral appears extremely resistant to coral bleaching events and has never been observed bleached in the Florida Keys since our surveys began in 1999. Additional threats to the species identified in the ESA process include moderate vulnerability to ocean warming and acidification, a narrow geographic distribution (limited to the Caribbean), and inadequacy of regulatory mechanisms.

Our population results contrast dramatically with Vernon (2002) and support the results instead of Wagner et al. (2010). In the Florida Keys, for all sample periods, *Dichocoenia stokesi* is among the most common corals, ranking eighth or better (Figure 3-1). The absolute numbers are staggering (Table 3-2), approaching 100 million colonies in 2005, with no trends apparent since then (Table 3-2). The species is relatively small (Figure 3-7) and rarely exceeds 40 cm in diameter. Size class distributions remained similar among the three sample periods (2005, 2009, and 2012) and larger colonies typically exhibited more partial mortality (Figures 3-7). Of the 502 colonies counted, measured, and assessed for condition in 2012, 1.8% exhibited predation by *Coralliophila* snails. Of particular importance is that no examples of active White Plague type II or any other disease-like conditions were noted.

In the Dry Tortugas, *Dichocoenia stokesi* was consistently ranked among the most common corals (12th in 2006 and 14th in 2008). Absolute numbers exceeded 12 million in 2006 (SE 4.1 million) and 7 million (SE 1.1 million) in 2008, less than what is seen in the Florida Keys. The 2006 size class distribution includes most colonies in the smallest size class, with higher partial mortality in the larger corals (Figure 3-7). In 2008, corals in the two smallest size classes predominate, with partial mortality similar to what was seen in 2005 (Figure 3-8). Total population size decreased in 2008, but not significantly and was still large, exceeding 7 million colonies.

The large population numbers, even after the White Plague Type II epidemic during the 1990s, its broad distribution among multiple habitat types, especially hard-bottom habitats, its high relative abundance among all corals in the region (including a top ten species for recruitment, unpublished data), and the low prevalence of White Plague Type II, all suggest that the proposed listing of *Dichocoenia stokesi* to Threatened status is not supported by the population data and is thus unwarranted.

Figure 3-7. *Dichocoenia stokesi* colony abundance by skeletal unit size class (max. diameter, cm) in the Florida Keys (northern Biscayne National Park to SW of Key West) during 2005 (top), 2009 (middle), and 2012 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.

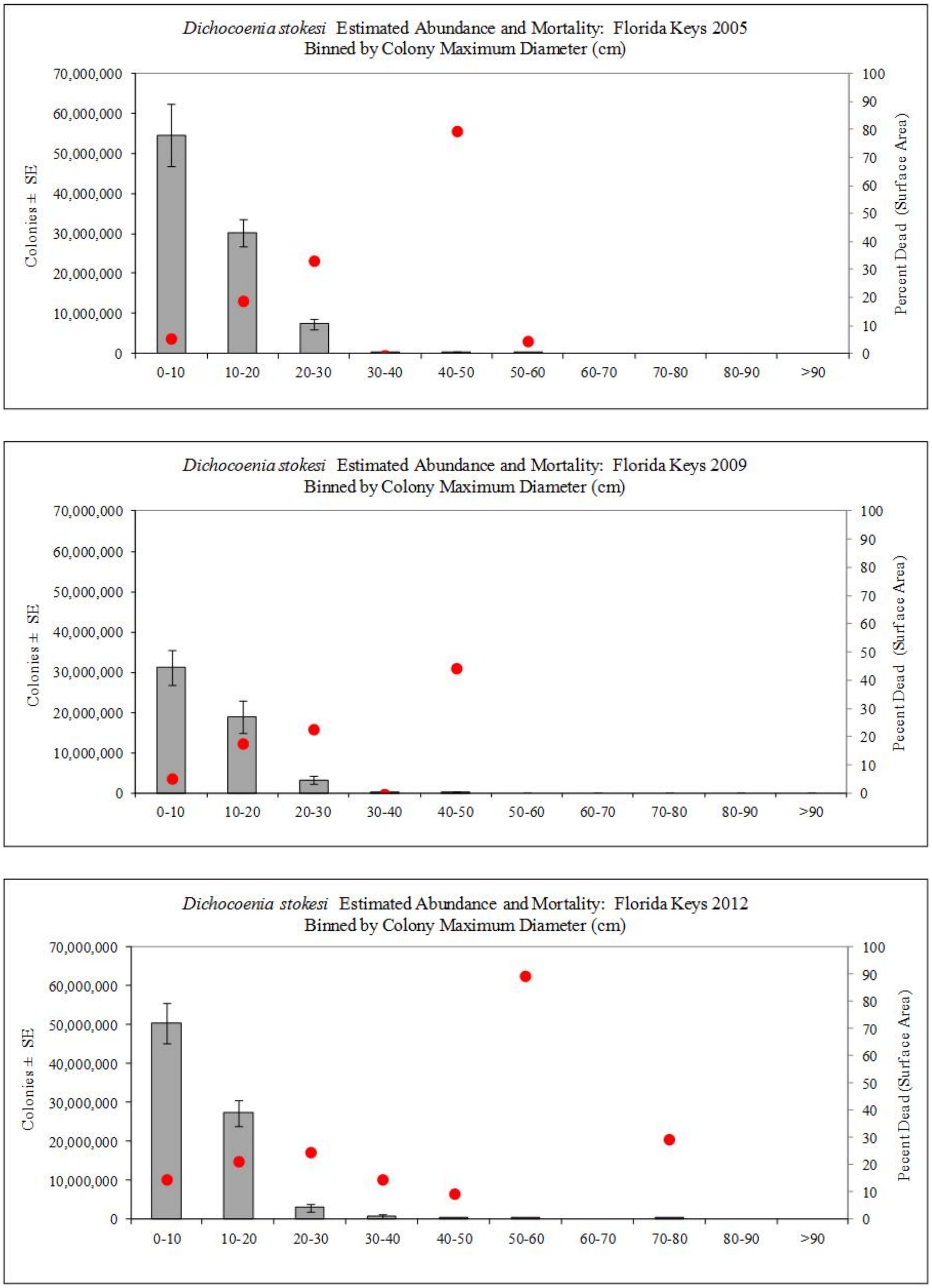
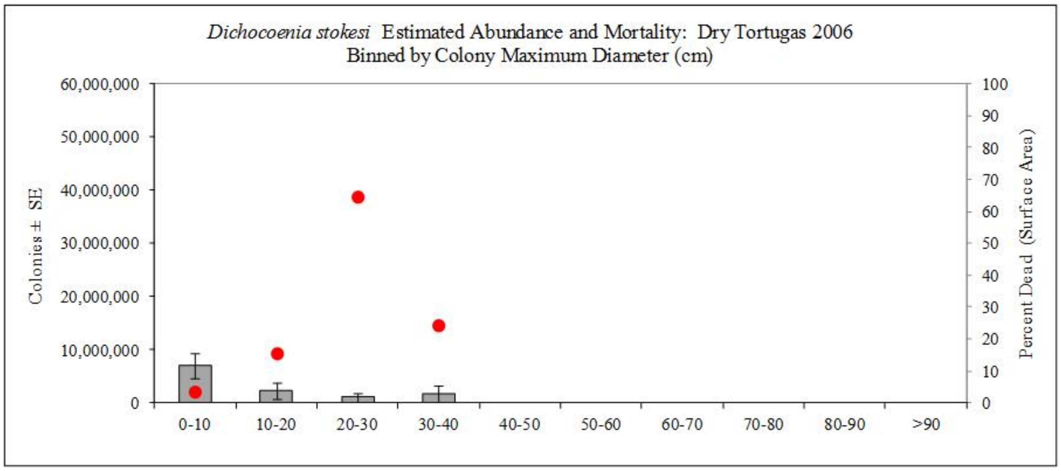
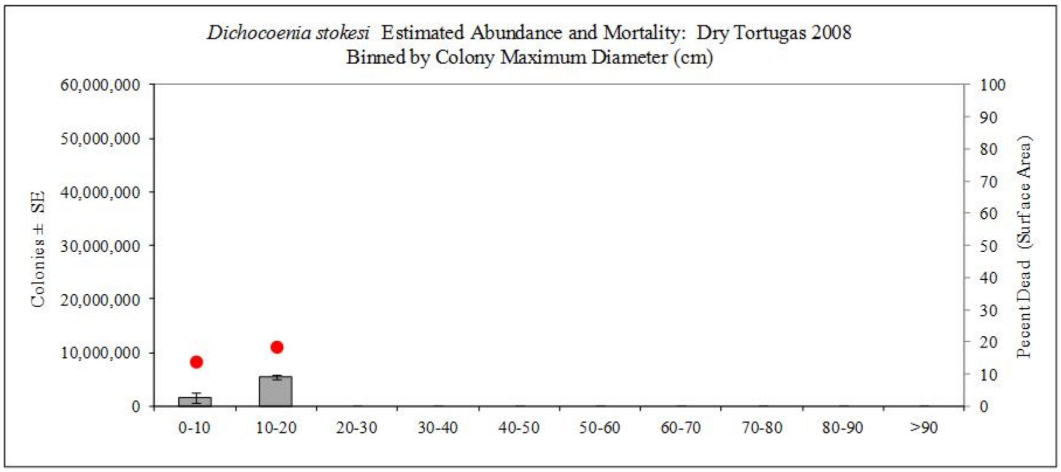


Figure 3-8. *Dichocoenia stokesi* colony abundance by skeletal unit size class (max. diameter, cm) in the Dry Tortugas region during 2006 (top) and 2008 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.





3.6 Genus *Montastraea* (Family Faviidae)

The BRT (Brainard et al. 2011) notes that the *Montastraea annularis* species complex is generally considered to be abundant if not dominant on Caribbean coral reefs, at least historically. Declines are relatively well documented for Florida (Dustan and Halas 1987, Dupont et al. 2008, Smith et al. 2008), based on monitoring programs that largely measured percent coral cover. We also have evidence that declines occurred in some habitat types (not reported here), but overall the population remains in the tens of millions colonies in the Florida Keys alone (Figures 3.1 and 3.2).

The BRT notes particular concern over the declines, citing the low growth rates and low recruitment rates as increasing the risk of extinction. However, confusion abounds in the literature regarding identification of the three sibling species within the *M. annularis* species complex – unlike for other ESA candidate species in the wider Caribbean. For instance, the data regarding growth form and corallite morphology is confounded in the *M. annularis* species complex due to high regional inter- and intraspecific variability, polymorphism, phenotypic changes related to: habitat type, depth, energy, sediment stress, and light (Dustan 1975, Foster 1979, 1980, Graus and Macintyre 1983); differential growth rates due to variations in depth (Bosscher and Meesters 1992) and habitat type (Hudson 1981); and reproductive hybridization within and between species in the complex (Szmant et al. 1997, Budd and Pandolfi 2004). The recent proposed listing of corals from the *M. annularis* species complex to Endangered status, under the U.S. Endangered Species Act (Brainard et al. 2011) requires accurate taxonomic, ecological, and biologic data to determine if the listing is warranted, as well as to identify appropriate conservation efforts for each of these individual sibling species. Our population data presented below are not confounded by taxonomic uncertainty.

3.6.1 *Montastraea annularis* Ellis and Solander, 1786

In the Florida Keys, *Montastraea annularis* is relatively common and was ranked in the middle among corals in terms of abundance in 2005 (30 out of 47), moving up significantly in 2009 to 13 out of 43, and 12 out of 40 in 2012 (Figure 3-1). In terms of population numbers, in 2005 our estimate was 5.6 million (SE 1.7), with 11.5 million (SE 2.5 million) in 2009, and 24 million (SE 10.1 million) in 2012 (Table 3-2). While variance terms are high, no evidence of decline was observed in total population number. While reports of decline in the Keys exist (Dustan and Halas 1987, Dupont et al. 2008, Smith et al. 2008), they are typically reported for a limited number of sites and they are from habitats that traditionally had the highest cover to begin with - and thus are most likely to show decline (Hughes 1992).

The size class frequency distribution for *Montastraea annularis* in the Florida Keys, exclusive of the Dry Tortugas, appears similar among the three sample periods (Figure 3-9). There is no evidence of a recruitment event to explain the increasing population sizes in 2009 and 2012, though in 2012 there are proportionally more individuals in the largest size class. Partial mortality appears similar across years, where it is lower among the smallest size classes and increases then flattens out among the larger size classes. Maximum values do not exceed 70%. While this number is relatively high it reflects an aspect of its growth form, where columns are mostly live tissue at their tops, but not deeper into the colony. Unfortunately, research to document the impact of partial mortality on the population structure and numbers of this species is absent. In 2012, no disease was noted, including black-band disease. Snail predation was recorded on 2.5% of the sampled colonies.

In the Dry Tortugas, *Montastraea annularis* was ranked among the least common corals, near the bottom in 2006 (41 out of 43) and not much better in 2008 (31 out of 40). We could not even calculate a variance term for the absolute population in 2006. In 2008 the population estimate was 0.5 million (SE 0.3 million). Size class frequency figures are not particularly informative when numbers are low (Figure 3-10). We have some population data from a pilot study collected over several years (1999-2002) that confirms the low ranking (43 out of 49), with a similarly high variance term associated with the population estimate of 1.0 million (SE 0.7 million).

The larger number of *Montastraea annularis* in the Keys is related to the greater abundances of shallow patch reefs, where the species is most commonly found (e.g. Basin Hills Shoals, Mosquito Bank, Cheeca Rocks). This habitat type is uncommon in the Dry Tortugas (Table 2.3), where we have found *M. annularis* at only a few locations, such as Little Africa, and other shallow patch reefs. While there have been population declines of this species in some specific habitats, its multi-habitat distribution coupled with a broad depth distribution makes this coral less susceptible than other coral species with limited habitat distribution and confined depth ranges such as *Acropora palmata*. With over 6,000 patch reefs in the Florida Keys (FMRI 1998) and the large number of corals present, listing this species as Endangered is not supported by the population data and is thus unwarranted.

Figure 3-9. *Montastraea annularis* colony abundance by skeletal unit size class (max. diameter, cm) in the Florida Keys (northern Biscayne National Park to SW of Key West) during 2005 (top), 2009 (middle), and 2012 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.

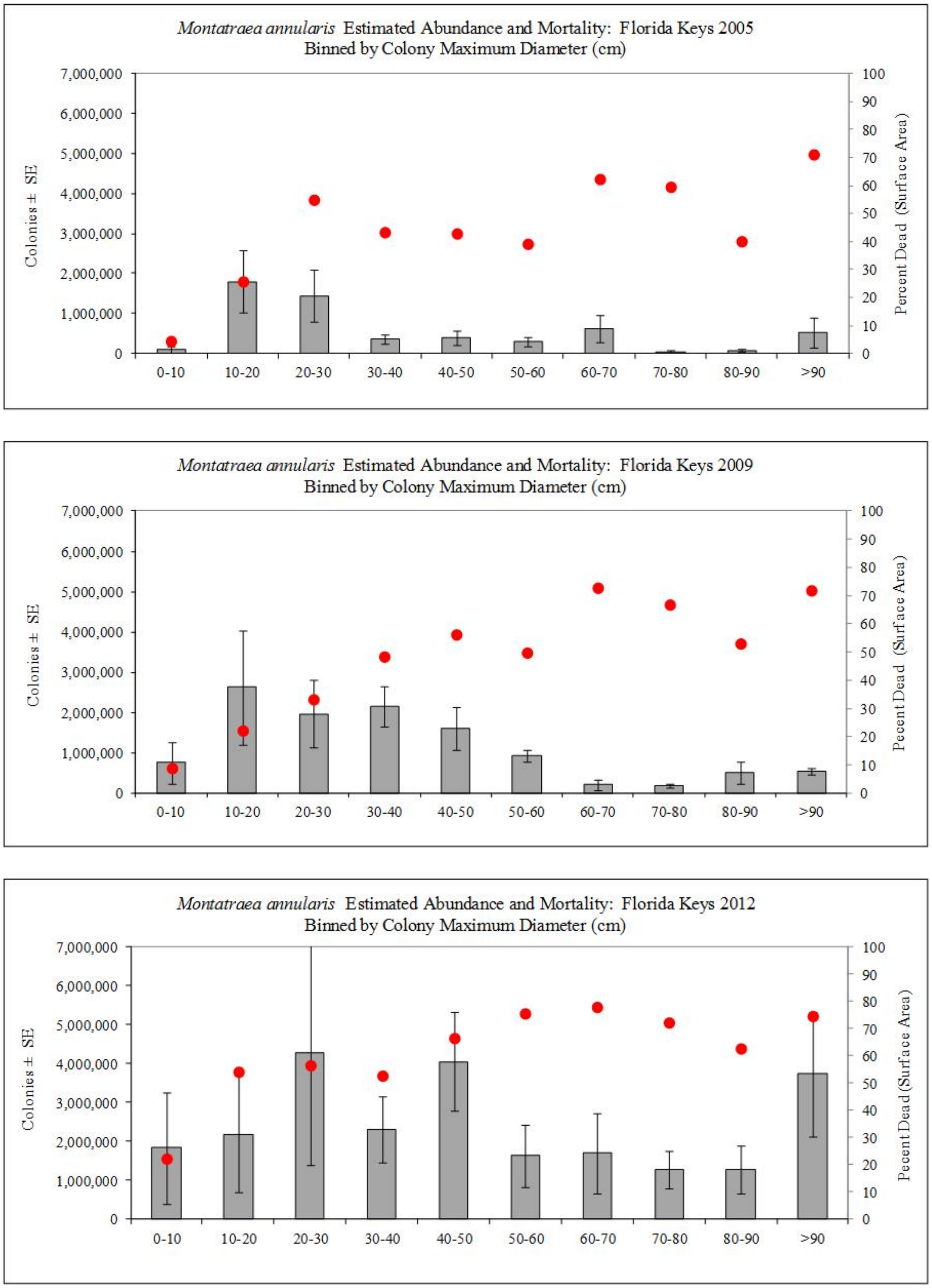
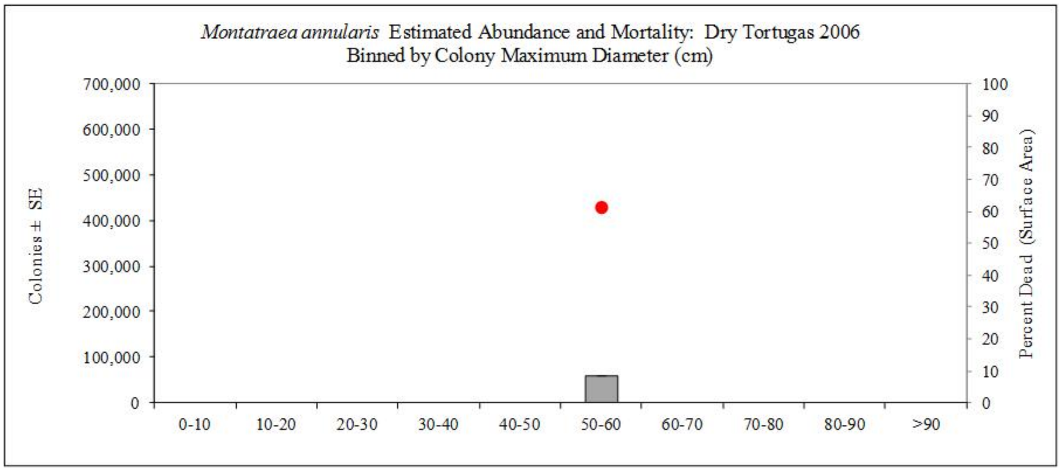
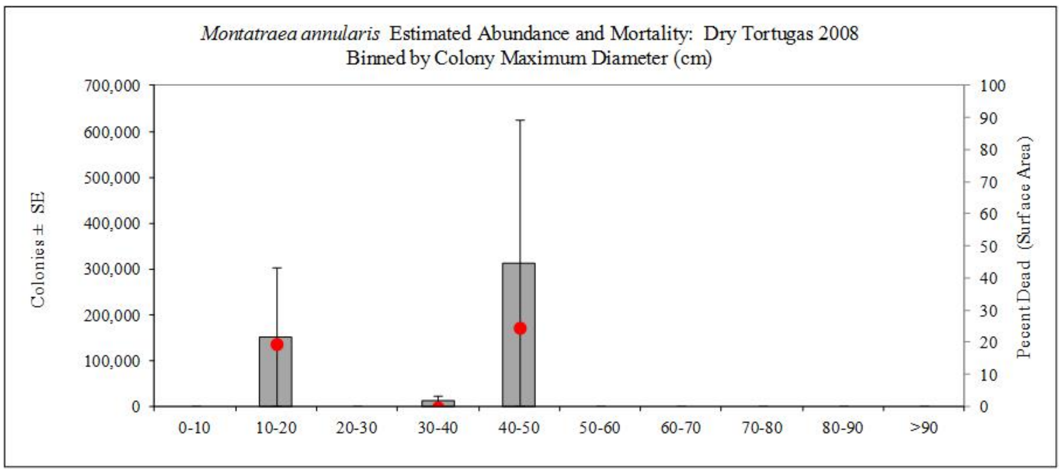


Figure 3-10. *Montastraea annularis* colony abundance by skeletal unit size class (max. diameter, cm) in the Dry Tortugas region during 2006 (top) and 2008 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.





3.6.2 *Montastraea faveolata* Ellis and Solander, 1786

In the Florida Keys, *Montastraea faveolata* is one of the top ten most abundant scleractinian corals (Figure 3-1). It is the second most abundant candidate species, behind *Dichocoenia stokesi* and well ahead of *Acropora cervicornis* in ranking (Figure 3-1). For the three sample periods, a non-significant decline was observed between 2005 and 2012, when in 2009 the sample allocation included a reduced number of sites. Absolute numbers estimated for the population were 39.7 million (SE 8 million) in 2005, 21.9 million (SE 7 million) in 2009, and 47 million (SE 14.5 million) in 2012 (Table 3-2). The size-class distributions and partial mortality estimates for *M. faveolata* are similar among years, even for 2009 (Figure 3-10). Interestingly, there are two peaks in the distribution, for corals of relatively small size (20-30cm diameter) and for corals greater than 90 cm diameter. In 2012, the distribution flattens somewhat, but the peak in the largest size class remains. It is well known that sexual recruitment is extremely rare for this species and has been historically; as such, the smallest size class is never abundant. Partial mortality does not appear to show any trends among size classes (red dots in Figure 3-11), but was noticeably higher in 2012 among the largest size class. Disease was not present in our 2012 sampling (365 colonies measured), though 1.9% exhibited snail predation. While declines have been documented for a few locations in the Florida Keys (Dustan and Halas 1987, Dupont et al. 2008, Smith et al. 2008), there are still millions of individuals across multiple size classes, with a peak among the largest colonies.

Related to relative abundance, the situation is even better in the Dry Tortugas, where *Montastraea faveolata* is ranked seventh most abundant in 2006 and fifth most abundant in 2008. Absolute population numbers are 36.1 million (SE 20 million) and 30 million (SE 3.3 million), respectively (Table 3-2). Size class distributions (Figure 3-12) are similar to what was seen in the Florida Keys, with the exception that partial mortality appears to be smaller among the smaller size classes, especially in 2008.

With the large number of colonies present, especially in the smaller and medium size classes, and the wide distribution of the species in the region, listing of the species as Endangered is not supported by the population data. Concern about the loss of the largest individuals in the population, based largely on anecdotal information, is a notable impact to the population. These largest colonies may be hundreds of years old and they have achieved iconic status for good reason – they are spectacular features wherever they exist. However, their decline represents only a modest loss in total numbers and a minor shift in the size-class frequency distribution for the species. In addition, like its congener *M. annularis*, its broad depth and multi-habitat distribution make it less susceptible to species collapse. Listing this species as Endangered is therefore not supported by the population data and is thus unwarranted.

Figure 3-11. *Montastraea faveolata* colony abundance by skeletal unit size class (max. diameter, cm) in the Florida Keys (northern Biscayne National Park to SW of Key West) during 2005 (top), 2009 (middle), and 2012 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.

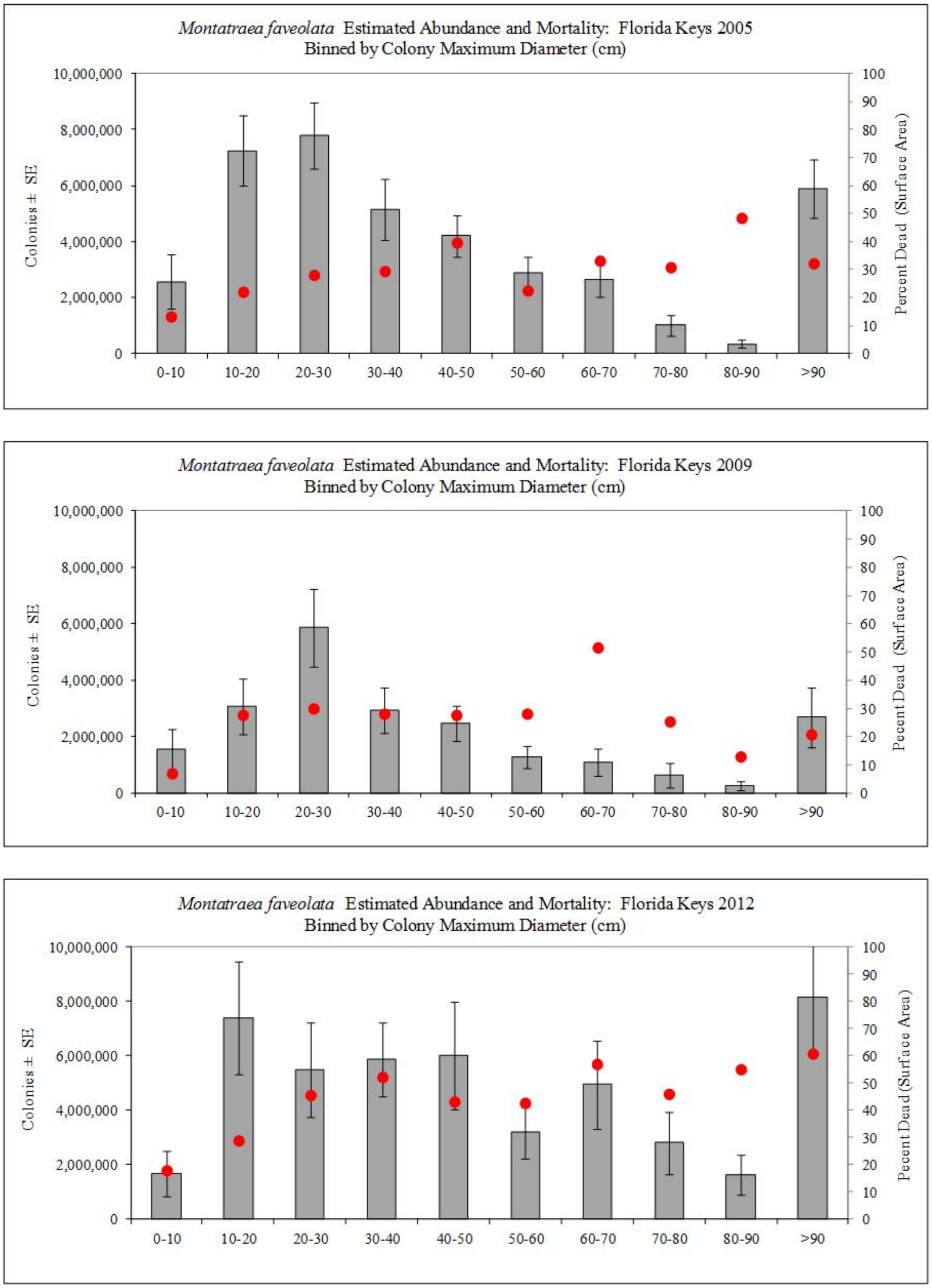
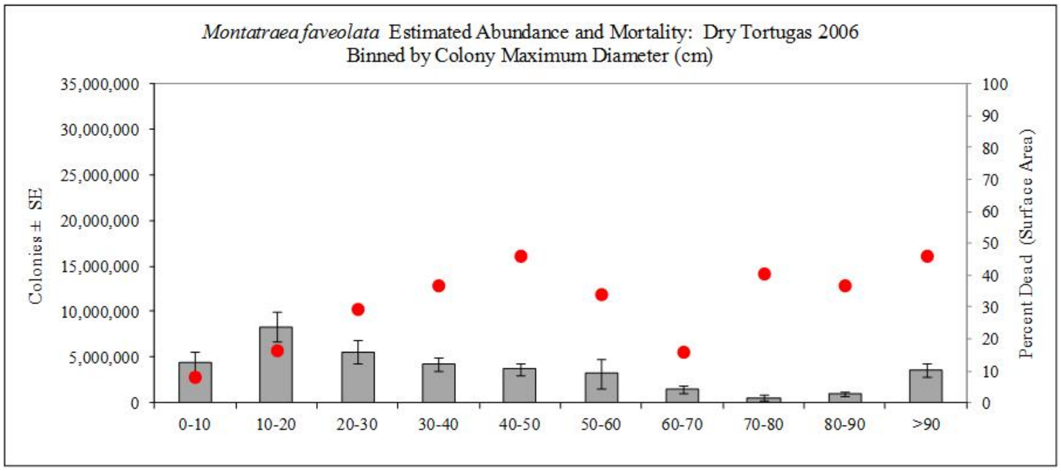
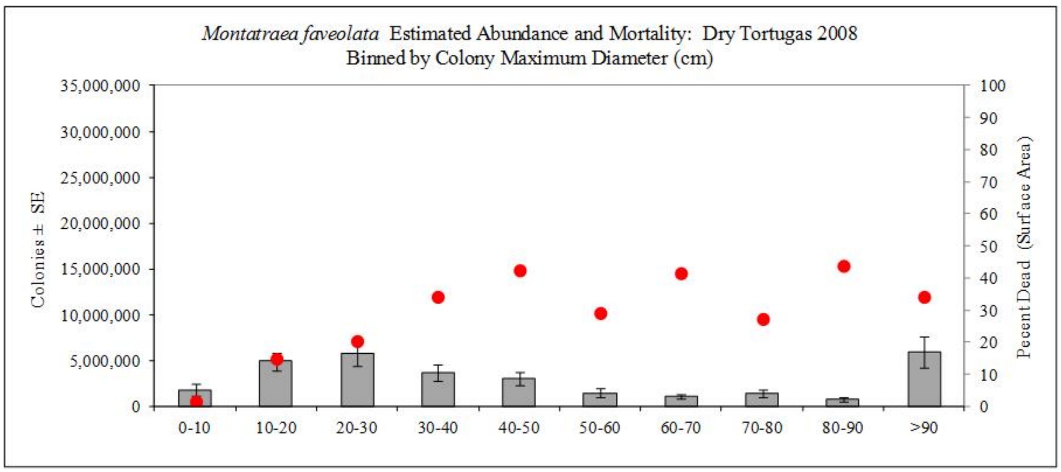


Figure 3-12. *Montastraea faveolata* colony abundance by skeletal unit size class (max. diameter, cm) in the Dry Tortugas region during 2006 (top) and 2008 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.





3.6.3 *Montastraea franksi* Gregory, 1895

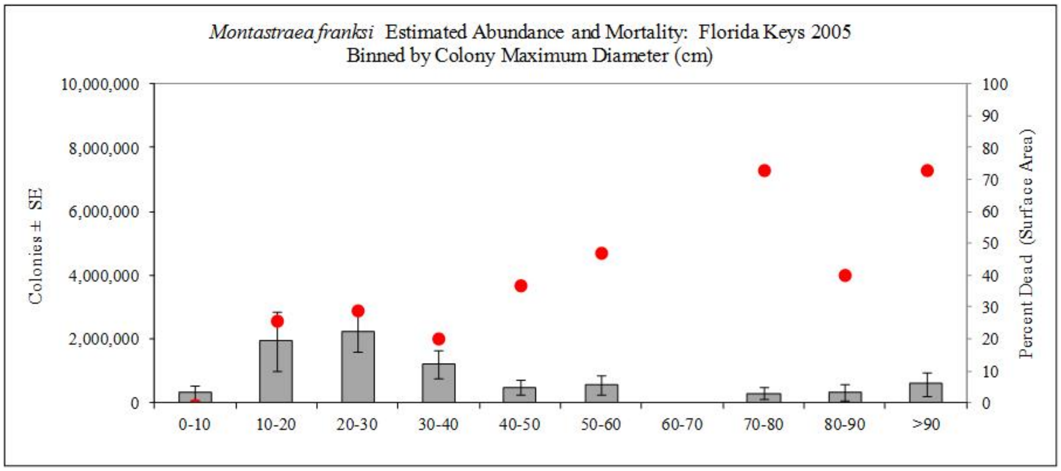
In the Florida Keys, *Montastraea franksi* is relatively common and typically found in deeper habitats than *M. faveolata* and *M. annularis*. The species is ranked in the middle, among all corals in the Florida Keys, ranging from 26th in 2005, to 32nd in 2009, and 33rd in 2012 (Figure 3-1). None of these sample periods included deeper reef habitats in the sampling regime, due to time constraints diving deeper. Absolute numbers (Table 3-2) for 2005 were 8 million (SE 2.2 million), for 2009 0.3 million (SE 214,000), and for 2012 0.4 million (SE 0.3 million). The apparent decline that occurred in 2009 and the similar value in 2012 are due to changes in the allocation scheme and logistics after 2005, where deeper sites were not able to be surveyed. This is why we note in our sampling effort that sometimes different habitat types were sampled. Results from the Tortugas (below) confirm the importance of deeper reef habitats for this species.

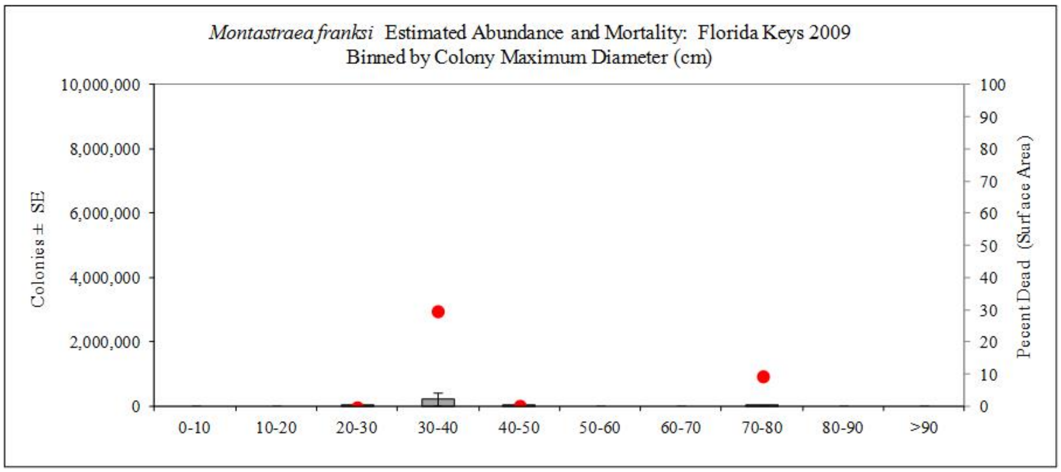
The 2005 size class distribution figure for the Florida Keys is the only one that provides useful information (Figure 3-13). The majority of the population is relatively small, 10-40cm diameter, but a sizeable number of colonies are found larger than 90-cm diameter. A clear increasing trend with size is seen in partial mortality. Without knowing that deeper sites were not sampled in 2009 and 2012, the high partial mortality seen in 2005 might be interpreted to explain the absence of corals in later years – that would be wrong. We have modified sample allocation schemes among years in an attempt to assess the distribution and abundance of corals throughout the sanctuary. The primary aim of the program is to monitor the no-take zones in the Florida Keys National Marine Sanctuary, most of which are found within relative shallow waters. Over time, we are also establishing baselines in multiple habitat types that will allow us to detect change over time, if we are able to allocate samples in sufficiently large numbers.

In the Dry Tortugas, where we were able to sample deeper coral reef habitats, *Montastraea franksi* is one of the most common corals, ranking 4th in 2006 and 8th in 2008 (Figure 3-1). Absolute population numbers (Table 3-2) in the Dry Tortugas are 79 million (SE 19 million) in 2006 and 18.1 million (SE 4.1 million); these differences are related to sample allocation differences between the two time periods. While total numbers appear lower in 2008, the size class distribution is similar to what was observed in 2006 (Figure 3-14). Few corals are seen in the smallest size class, increasing somewhat to about 50-60 cm diameter, then decreasing until a peak among corals larger than 90 cm diameter. Partial mortality was similar for both years, increasing with increasing size classes.

These results document that *Montastraea franksi* is relatively uncommon in shallower reef habitats through the Florida Keys, but common in deeper reef habitats. We have also seen *M. franksi* in patch reef habitats. Listing this species as Endangered is therefore not supported by the population data and is thus unwarranted.

Figure 3-13. *Montastraea franksi* colony abundance by skeletal unit size class (max. diameter, cm) in the Florida Keys (northern Biscayne National Park to SW of Key West) during 2005 (top), 2009 (middle), and 2012 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.





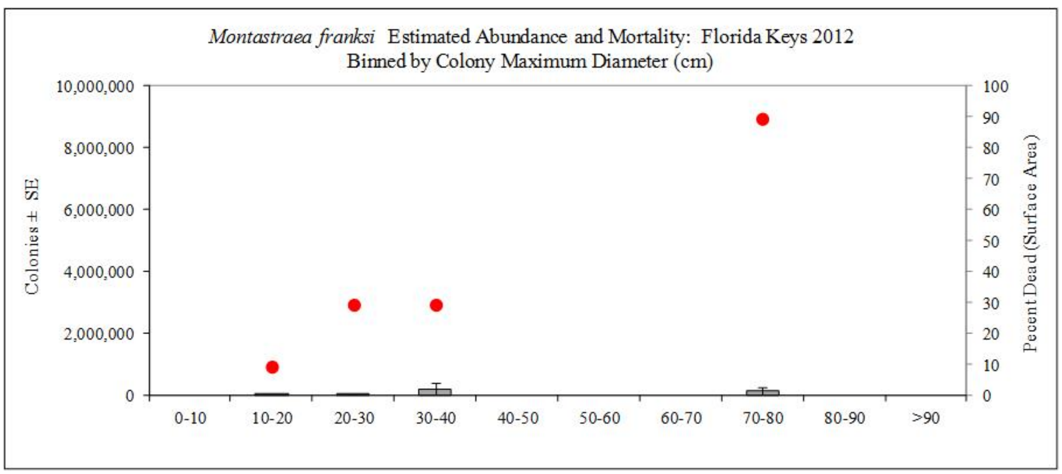
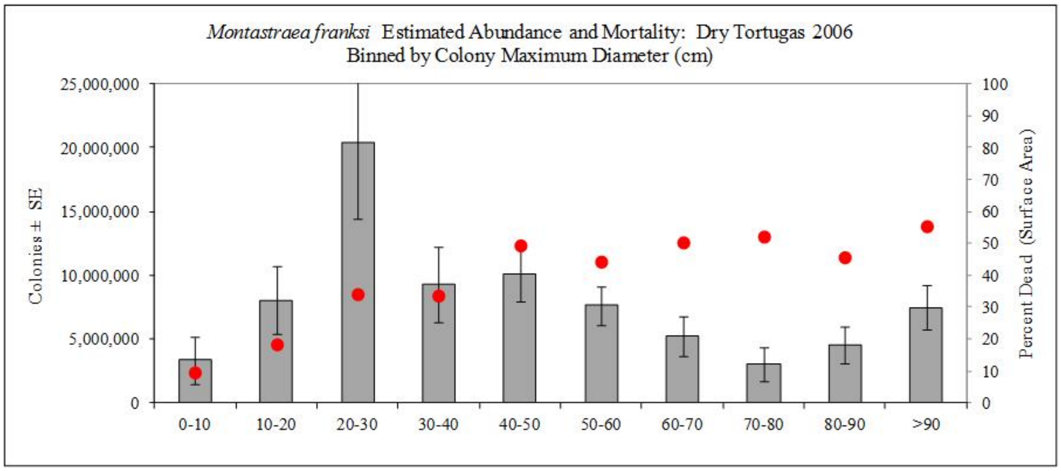
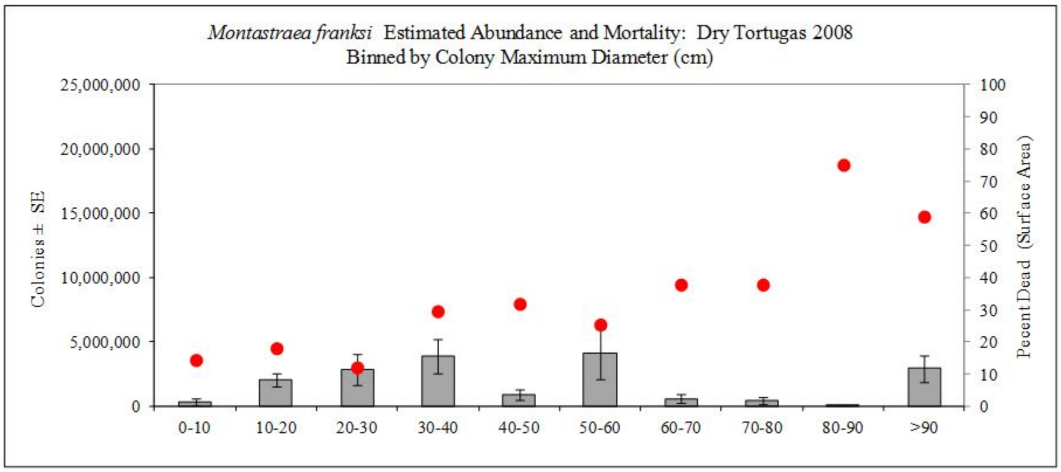


Figure 3-14. *Montastraea franksi* colony abundance by skeletal unit size class (max. diameter, cm) in the Dry Tortugas region during 2006 (top) and 2008 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.





3.7 Genus *Agaricia* (Family Agariciidae) and Genus *Mycetophyllia* (Family Mussidae)

3.7.1 *Agaricia lamarcki* ME & H, 1851 and *Mycetophyllia ferox* Wells, 1973

Agaricia lamarcki and Mycetophyllia ferox are presented together because they exhibit similar traits. They are both rare and they are more common in deeper coral reef habitats. Our sampling efforts were not optimized for such species.

In the Florida Keys, where few sites were sampled in deeper habitats, both species were ranked toward the bottom, or last, in terms of coral abundance (Figure 3-1). *Agaricia lamarcki* was ranked 35 out of 47 in 2005, it was absent from our sampling in 2009, and it ranked 37th out of 40 in 2012. *Mycetophyllia ferox* was ranked 39th out of 47 in 2005, 43rd out of 43 in 2009 and 40th out of 40 in 2012. Population estimates for the two species, even ranked as low as they are, suggest that there are still many corals present (Table 3-2). For *A. lamarcki*, the population estimates were 3.1 million (SE 1.0 million) in 2005, they were absent in 2007, and 0.2 million (SE 0.2 million) in 2012. This suggests a decline over the seven year period, but few deep sites were sampled in 2007 and 2012 and more work needs to be done to get a reliable population estimate. For *M. ferox* (Table 3-2), the population estimates are 1.0 million (SE 0.5 million) in 2005, 9,500 (SE 9,500) in 2009, and 7,000 (SE 7,000) in 2012. The decline in 2009 and 2012 is explained similarly for *M. ferox,* based on sampling deeper coral reef habitats in 2005.

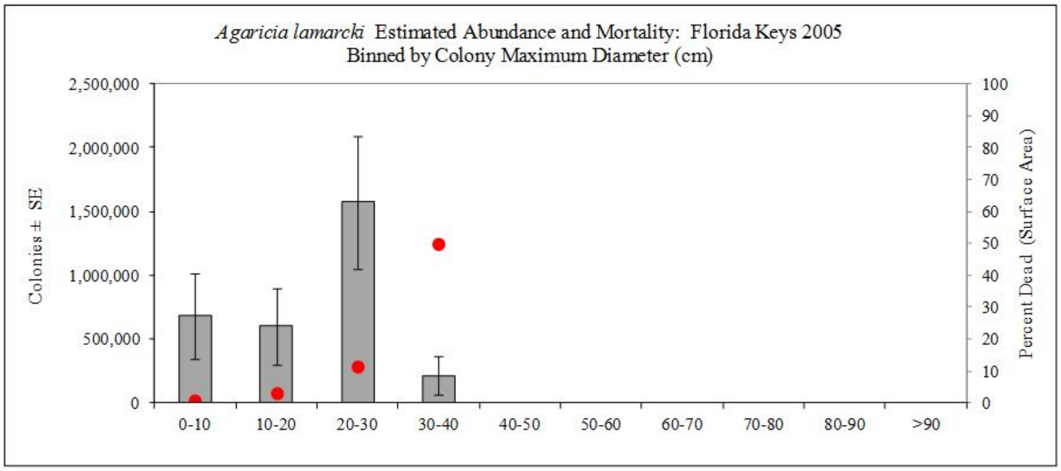
Size class distributions are only provided for 2005, for both species (Figure 3-15). Not enough corals were sampled in 2009 and 2012 to produce distributions. For *Agaricia lamarcki*, the most abundant size class was in 20-30cm diameter, with as many colonies as the rest of the population. Partial mortality was highest in the largest size class, at 50 percent. For *Mycetophyllia ferox*, the most common size class included a peak at 10-20cm diameter, with as many colonies as the rest of the population. Partial mortality was highest in the largest size class, at 50%.

The depth preference for these two species was evident in the Dry Tortugas, where we allocated more samples to deeper sites. Both species improved in their relative abundance ranking and populations numbers. For *Agaricia lamarcki*, its ranking jumped to 12th out of 43 in 2006 and 22nd out of 40th in 2008 (Figure 3-1). Populations estimates were 14.3 million (SE 2.6 million) in 2006 and 2.1 million (SE 0.5) in 2008 (Table 3-2). The smaller estimate in 2008 is mostly explained by fewer sites allocated to deeper habitats. For *Mycetophyllia ferox*, its abundance ranking approved slightly, to 35th out of 43 in 2006 and 30th out of 40 in 2008. Population estimates were 0.9 million (SE 0.4 million) in 2006 and 0.5 million (SE

0.2 million) in 2008. Size class distributions for *A. lamarcki* in the Dry Tortugas (Figure 3-15) include many more and larger corals than seen in the Florida Keys. Partial mortality was also lower. Fewer larger corals were seen in 2008. For *Mycetophyllia ferox*, the size class distributions in the Dry Tortugas (Figure 3-16) differed somewhat from the Florida Keys. In 2006, numbers decreased with increasing size, while in 2008 there were fewer corals in the smallest size class and many more and larger corals seen than in 2006. Partial mortality in 2006 and 2008 was less than 20 percent in all size classes, except for a peak in one larger size class (30-40 cm diameter) in 2008 of nearly 70 percent. Fewer corals were also seen in this size class compared to the next smaller and larger size classes.

It is worth noting that corals in southeast Florida are generally near their northern limit of distribution. Whether or not this explains the lower population numbers for these two species in the Florida Keys, compared to the Dry Tortugas, is unknown. Sampling fewer deeper sites in the Keys probably explains the smaller numbers. Work related to understanding the latitudinal distributions of corals in the Dry Tortugas and Florida Keys is in progress. While these two species are relatively uncommon in shallow habitats, listing these species as Endangered is not supported by their large population numbers in the deeper coral habitats of the Dry Tortugas, and is thus unwarranted.

Figure 3-15. *Agaricia lamarcki* (top) and *Mycetophyllia ferox* (bottom) colony abundance by skeletal unit size class (max. diameter, cm) in the Florida Keys (northern Biscayne National Park to SW of Key West) during 2005, with the average percent colony mortality shown in red (right-handed scale) for each size class.



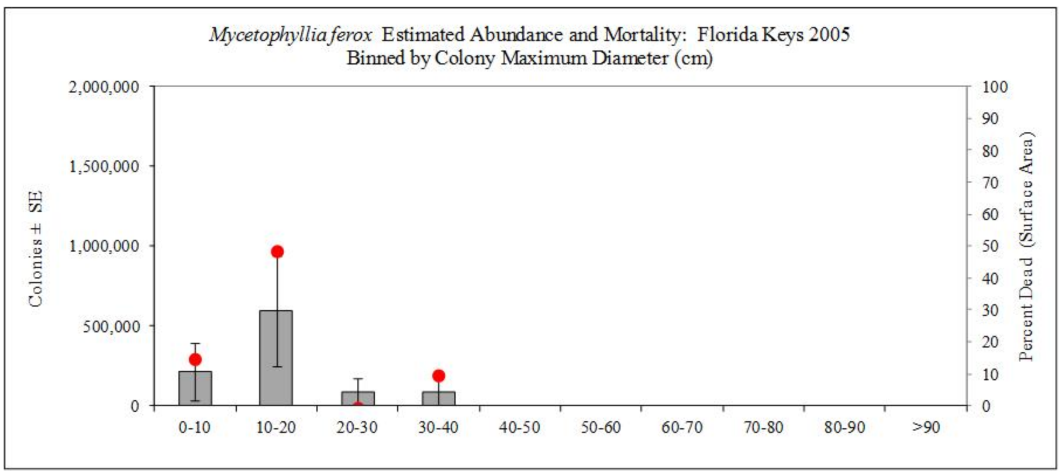
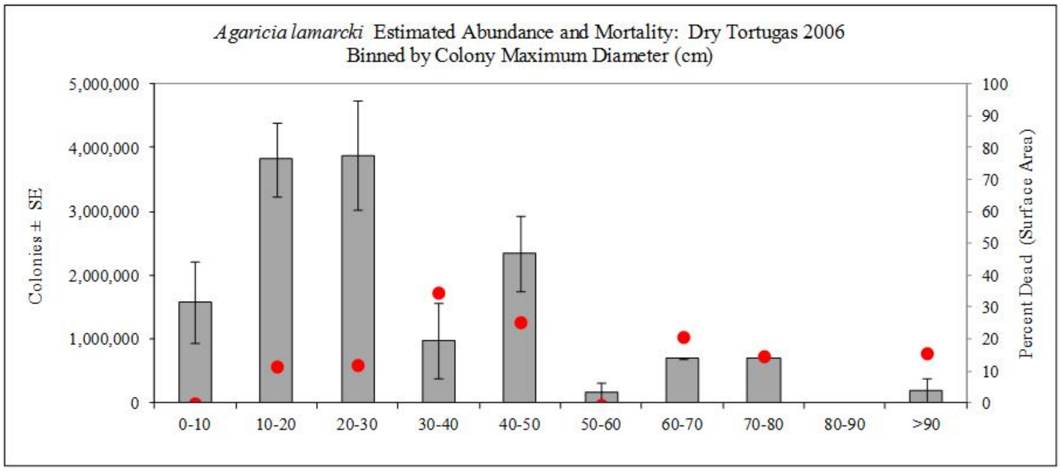


Figure 3-16. *Agaricia lamarcki* colony abundance by skeletal unit size class (max. diameter, cm) in the Dry Tortugas region during 2006 (top) and 2008 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.



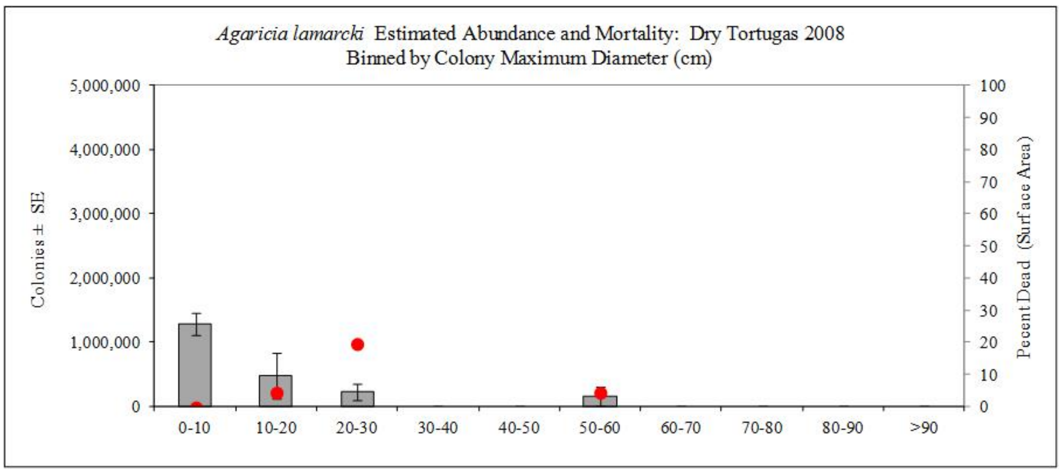
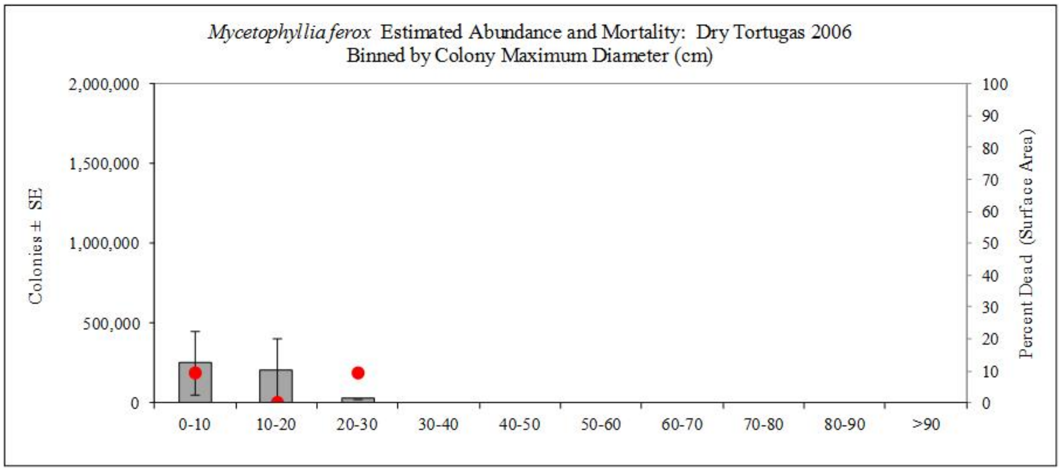
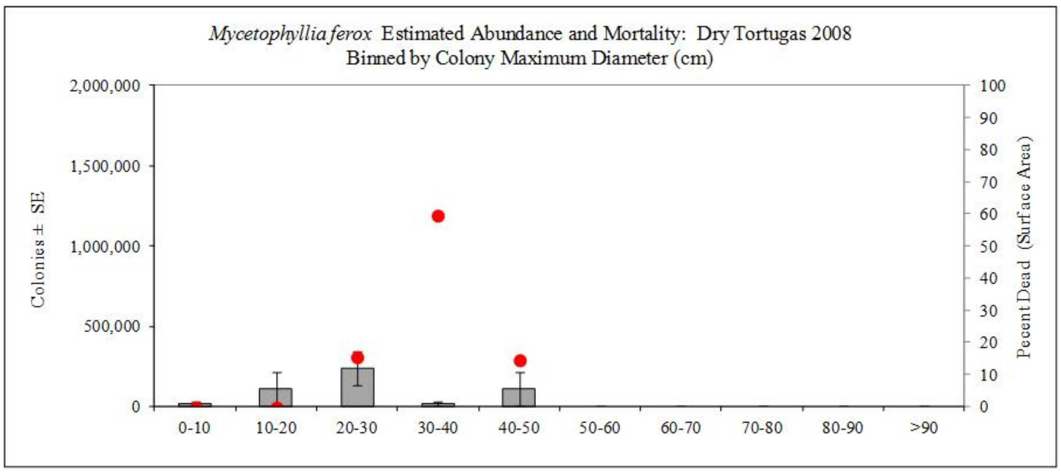


Figure 3-17. *Mycetophyllia ferox* colony abundance by skeletal unit size class (max. diameter, cm) in the Dry Tortugas region during 2006 (top) and 2008 (bottom), with the average percent colony mortality shown in red (right-handed scale) for each size class.





3.5 Genus *Dendrogyra* (Family Meandrinidae)

3.5.1 *Dendrogyra cylindrus* Ehrenberg, 1834

Dendrogyra cylindrus (pillar coral) is uncommon throughout the Florida Keys and Dry Tortugas. It is an iconic coral species due to its dramatic presentation, with vertical pillars and the potential to form dense stands (Figure 3-17). It differs from the above two species (i.e. *Agaricia lamarcki* and *Mycetophyllia ferox*) in that it is typically found in shallower coral reef habitats. It is naturally rare in the Florida Keys and Dry Tortugas. Our sample allocation schemes did not optimize for this species.

As noted by the BRT (Brainard et al. 2011), rarer species have variable traits that allow them to persist across ecologic and geologic time, and perhaps do not warrant the concern about their status in the absence of significant declines. For instance, *Dendrogyra cylindrus* was listed in a volume on Rare and Endangered Biota of Florida (Antonius 1994) solely because of their limited abundance and not based on population declines or known threats specific to this coral. However, when rare species are also iconic or charismatic, such as *D. cylindrus*, then management and societal interests become highly relevant. It is when previously abundant species become rare, due to stress events that kill large numbers in a population, or when habitats are altered that do not allow populations to sustain themselves, that urgency is applied to management.

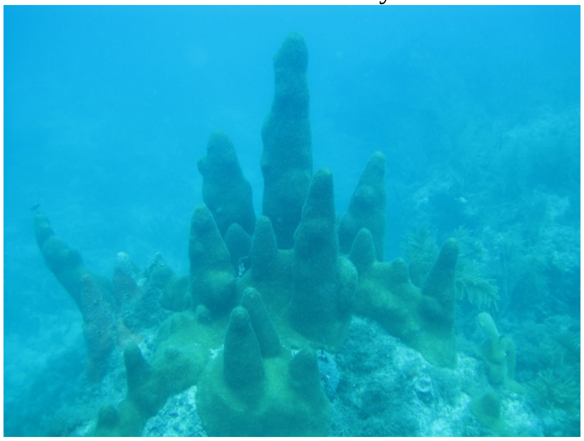
In the Florida Keys, *Dendrogyra cylindrus* ranked 47th out of 47 in 2005, with a population estimate of 23,000 (SE 23,000) and 41st out of 43 in 2009, with a population estimate of 25,000 (SE 25,000). In 2012, despite surveying 600 sites from northern Biscayne National Park to southwest of Key West, no pillar coral colonies were encountered (Figure 3-1, Table 3-2). Of particular note is the high variability in the population abundance estimates. Despite the low population estimate, it is well-known that there are several spectacular stands of this coral in the Florida Keys that appear in good condition (Figure 3-18). This presents a case for non-random monitoring to assess the status and trends of these spectacular coral assemblages, which to our knowledge is not happening. Too few colonies were measured to present a size class distribution, but in 2005 all corals were in the 70-80cm diameter size class with low partial mortality (< 2 %) In 2009, all corals were in the greater than 90 cm diameter size class. This species was not seen in the Dry Tortugas in 2006 and 2008. In pilot studies conducted over several years in the Dry Tortugas (1999-2002), *D. cylindrus* was rarely encountered and ranked 49th out of 49 corals.

Our sampling program was not optimized for rare species such as *Dendrogyra cylindrus*. The species is naturally rare and apparently clumped in its distribution, when forming larger stands. Anecdotal information suggests that several known stands have not declined, with the exception of one that was located at Conch Reef. If *D. cylindrus* is listed as Endangered because it is rare, without evidence of significant decline, then the majority of coral species in the Caribbean meet the same criteria. Using similar criteria, the IUCN Coral Red List only warranted a vulnerable status for this species (Aronson et al. 2008). While our population data are limited for this species, large extant stands exist without evidence of significant decline. Thus, listing the species as Endangered is not warranted.

Figure 3-18. Examples of extant stands of pillar coral (*Dendrogyra cylindrus*) in the Florida Keys. Although this species is naturally rare, large colonies still remain and appear in relatively good condition.

Large pillar coral colony at Marker 32 in the lower Florida Keys

Large pillar coral colony at Rock Key in the lower Florida Keys



Pillar coral stand at Pickles Reef in the upper Florida Keys

Pillar coral stand near Turtle Rocks, northern Key Largo

4. Conclusion

An important element of the Endangered Species Act is development of a Recovery Plan for species that are listed, either Threatened or Endangered. The intent of listing is to implement actions that protect and help recover species, so that they can eventually be removed from the list. For *Acropora palmata* and *A. cervicornis*, listed as Threatened in 2006, a Recovery Plan has not been published. What explains a seven-year delay in writing a Recovery Plan? While we don't know for certain, one explanation might be that outside of restoration work, which involves nurseries and transplanting corals to reefs, and is already an active area of development, there is nothing within the ESA that provides meaningful additional protection or recovery. For instance, data from throughout the Caribbean indicate that no form of local stewardship or management could have protected these *Acropora* populations from their major sources of mortality or changed the overall trajectory of coral loss during the past few decades (Precht et al. 2004). In addition, there are millions of *A. cervicornis* individuals remaining, just in southeast Florida. And the 2006 Federal Listing estimated potential numbers for *A. cervicornis* in the billions throughout the Caribbean. While fewer numbers remain for *A. palmata*, extinction is clearly not imminent.

A reclassification to Endangered for *Acropora palmata* and *A. cervicornis* has the potential to significantly increase regulatory authority over the species, especially related to taking, but in reality nothing beneficial results that has the potential to increase population numbers in a meaningful manner. Taking or collecting or even damage from development does not represent a meaningful threat to these species, and that's typically where ESA could be most relevant. This is perhaps a harsh assessment, but when measured against the population data, application of ESA to management of these species seems misplaced and off target.

An alternative view about coral species recovery, outside of ESA, is to ask what conditions are required for recovery to occur in corals that have experienced significant population decline. The recent demise of *Acropora* in the Caribbean is far from an extinction event, yet declines that occurred are ecologically relevant. So, in ecological time (decades), it is important to ask whether or not natural recovery might occur. It is reasonable to suggest that the prolific growth rates of the acroporids (see Shinn 1966, 1976), along with sexual recruitment, are sufficient to repopulate all of the habitats occupied in Florida, and throughout the Caribbean, within a decade or two (see Idjadi et al. 2006; Precht and Aronson 2006). We just don't know what special set of conditions are required for such recovery to occur. At larger time scales, but still ecological, there is concern that bleaching and ocean acidification will eventually reduce population numbers further, though such losses have not been seen since their original ESA listing as

Threatened. Counter to concern about loss, is recovery and stasis among existing populations, persistence of large numbers of individuals in populations, especially for *A. cervicornis* as we report here for Florida, and successful restoration activities that suggest potential for increasing population numbers in a meaningful manner (also for *A. palmata)*. Further, in the face of global warming, a northward expansion of *A. palmata* and *A. cervicornis,* and perhaps other corals, may occur, mimicking the conditions of the mid-Holocene (Precht and Aronson 2004). The recent occurrence of *A. cervicornis* thickets off Fort Lauderdale presents an interesting case. Are these populations a harbinger of impending global change, or do they merely represent the temporary range expansion of remnant stands? Monitoring and assessment programs will eventually answer the question.

Corals such as the *Montastraea annularis* species complex that have broad depth and habitat distributions are less vulnerable to extinction than corals with restricted distributions. Thus, their potential listing is contrary to their ecology, especially in light of their remaining substantial population numbers both in Florida and throughout their range. In addition, the listing of species that are presently rare and have historically been rare also seems to be contrary to the biology and ecology of these species. The potential of listing rare coral species just because they are rare may set precedence from which there is no escape and may lead to a slippery slope in which all rare coral species might be listed.

It is important to note that these population estimates for the Florida Keys are for a region that is considered marginal for coral reef development, certainly through the Holocene where active reef growth in the Florida Keys is restricted to a relatively small area of the total hard bottom area, plus most corals (including the ones discussed in this report) are at or near their northern geographic limit of distribution in Florida – they are all widely distributed throughout the Caribbean. Further, the total coral reef habitat in the Florida Keys represents a small percentage of area (approximately 3 percent) relative to the larger Caribbean, and about 27 percent of total reef area in the U.S. Caribbean. In other words, the population estimates for these species in the Florida Keys must be considered extremely conservative estimates. As such, our results do not support the NOAA-NMFS proposal to list or reclassify these nine Atlantic coral species. Based on population data presented in this Technical Report, the Listings and Reclassifications are not warranted.

5. References

Acropora Biological Review Team (BRT) (2005) Atlantic *Acropora* Status Review Document. Report to National Marine Fisheries Service, Southeast Regional Office, St. Petersburg, FL, 152 p

Agassiz A (1883) The Tortugas and Florida reefs. Mem Amer Acad Arts Sci 11:107–134

Albright R, Mason B, Langdon C (2008) Effect of aragonite saturation state on settlement and post-settlement growth of *Porites astreoides* larvae. Coral Reefs 27:485-490

Albright RA, Mason B, Miller M, Langdon C (2010) Ocean acidification compromises recruitment success of the threatened Caribbean coral, *Acropora palmata*. Proc Natl Acad Sci 107:20400-20404

Antonius A (1994) Endangered pillar coral, *Dendrogyra cylindrus* (Ehrenberg). In: Rare and endangered biota of Florida series, Volume IV: Invertebrates. Deyrup M, Franz R (eds), University of Florida Press, Gainesville, FL

Aronson R, Bruckner A, Moore J, Precht B, Weil E (2008) *Dendrogyra cylindrus*. In: IUCN 2012. IUCN Red List of Threatened Species, Version 2012.2. Online at: www.iucnredlist.org

Aronson RB, Precht WF (2001a) Evolutionary paleoecology of Caribbean coral reefs. Pages 171-233 In: Evolutionary paleoecology: The ecological context of macroevolutionary change. Allmon WD, Bottjer DJ (eds), Columbia University Press, NY

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

Ault JS, Diaz GA, Smith SG, Luo J, Serafy JE (1999) An efficient sampling survey design to estimate pink shrimp population abundance in Biscayne Bay, Florida. N Amer J Fish Mgmt 19:696-712

Bak RPM, Meesters EH (1997) Coral diversity, populations and ecosystem functioning. Pages 27-38 In: In: Proceedings of the 6th International Conference on Coelenterate Biology. Den Hartog JC (ed), National Museum Natural History, Leiden

Bak RPM, Meesters EH (1998) Coral population structure: the hidden information of colony size-requency distributions. Mar Ecol Prog Ser 162:301–306

Bak RPM, Nieuwland G, Meesters EH (2009) Coral growth rates revisited after 31 years: What is causing lower extension rates in *Acropora palmata*? Bull Mar Sci 84:287-294

Banks KW, Riegl B, Shinn EA, Piller WE, Dodge RE (2007) Geomorphology of the southeast Florida continental reef tract (Miami-Dade, Broward, and Palm Beach Counties). Coral Reefs 26:617-633

Baums I, Miller MW, Hellberg ME (2005) Regionally isolated populations of an imperiled Caribbean coral, *Acropora palmata*. Mol Ecol 14(5):1377-1390

Baums I, Miller MW, Hellberg ME (2006) Geographic variation in clonal structure in a reefbuilding coral, *Acropora palmata*. Ecol Monogr 76(4):503-519

Bosscher H, Meesters EH (1992) Depth related changes in the growth rate of *Montastrea annularis*. Proc 7th Int Coral Reef Symp, Guam, 1:507-512

Brainard RE, Birkeland C, Eakin CM, McElhany P, Miller MW, Patterson M, Piniak GA (2011) Status review report of 82 candidate coral species petitioned under the U.S. Endangered Species Act. USDOC, NOAA Tech Mem NOAA-TM-NMFS-PIFSC-27, 530 p +1 Appendix

Bowden-Kerby A, Carne L (2012) Thermal tolerance as a factor in Caribbean *Acropora* restoration. Proc 12th Intl Coral Reef Symp, Cairns, 20A, 5 p

Bruckner AW (2002) Proceedings of the Caribbean *Acropora* workshop: Potential application of the U.S. Endangered Species Act as a conservation strategy. NOAA Tech Mem NMFS-OPR-24, Silver Spring, MD, 199 p

Budd AF, Pandolfi JM (2004) Overlapping species boundaries and hybridization within the *Montastraea annularis* reef coral complex in the Pleistocene of the Bahama Islands. Paleobiology, 30(3):396-425

Burke L, Maidens J (2004) Reefs at risk in the Caribbean. World Resources Institute, Wash. DC: 84 p

Cairns SD, Calder DR, Brinckmann-Voss A, Casrto CB, Fautin DG, Pugh PR, Mills CE, Jaap WC, Arai MN, Haddock SHD, Opresko DM (2002) Common and scientic names of aquatic invertebrates from the United States and Canada: Cnidaria and Ctenophora, 2nd edition. American Fisheries Society, Bethesda, MD, 115 p

Connell JH (1997) Disturbance and recovery of coral assemblages. Proc 8th Intl Coral Reef Symp, Panama, 1:9-22

Cochran WG (1977) Sampling techniques, 3rd ed. Wiley, NY

COHMAP Members (1988) Climatic changes of the last 18,000 years: observations and model simulations. Science 241:1043-1052

Crabbe MJC (2009) Scleractinian coral population size structures and growth rates indicate coral resilience on the fringing reefs of North Jamaica. Mar Env Res 67:189-198

Davis GE (1982) A century of natural change in coral distribution at the Dry Tortugas: A comparison of reef maps from 1881 and 1976. Bull Mar Sci 32:608–623

de Menocal P, Ortiz J, Guilderson T, Sarnthein M (2000) Coherent high- and low-latitude climate variability during the Holocene Warm Period. Science 288:2198-2202

Dustan P (1975) Growth and form in the reef-building coral *Montastrea annularis*. Mar Biol 33: 101-107

Dustan P, Halas JC (1987) Changes in the reef-coral community of Carysfort Reef, Key Largo, Florida: 1974 to 1982. Coral Reefs 6:91-106

Edmunds PJ (2007) Evidence for a decadal-scale decline in the growth rates of juvenile scleractinian corals. Mar Ecol Prog Ser 341:1-13

Enos P (1977) Holocene sediment accumulations of the south Florida shelf margin. Pages 1-130 In: Quaternary sedimentation in South Florida, Memoir 147. Enos P, Perkins RD (eds), Geological Society of America, Boulder, CO

Federal Register (2012) Endangered and Threatened Wildlife and Plants: Proposed Listing Determinations for 82 Reef-Building Coral Species; Proposed Reclassification of *Acropora palmata* and *A. cervicoronis*. 77 FR 73219. https://federalregister.gov/a/2012-29350

Finkl CW, Andrews JL (2008) Shelf geomorphology along the southeast Florida Atlantic continental platform: barrier coral reefs, nearshore bedrock, and morphosedimentary features. J Coast Res 24:823-849

FMRI (Florida Marine Research Institute) (1998) Benthic habitats of the Florida Keys. FMRI Tech Rep TR-4. FDEP, St. Petersburg, 53 p

Foster AB (1979) Phenotypic plasticity in the reef corals *Montastraea annularis* (Ellis & Solander) and *Siderastrea siderea* (Ellis & Solander). J Exp Mar Biol Ecol 39: 25-54

Foster, A. B. (1980). Environmental variation in skeletal morphology within the Caribbean reef corals *Montastraea annularis* and *Siderastrea siderea*. Bull Mar Sci 30(3):678-709

Franklin EC, Ault JS, Smith SG, Luo J, Meester GA, Diaz GA, Chiappone M, Swanson DW, Miller SL, Bohnsack JA (2003) Benthic habitat mapping in the Tortugas region, Florida. Mar Geod 26:19-34

Gerstman BB (2003) Epidemiology kept simple: An introduction to class and modern epidemiology, 2nd ed. Wiley-Liss, NY.

Grablow K, Nedimyer K, Northrop A, Precht W (2011) Restoration techniques for threatened acroporid corals in Florida and the Caribbean. Page 82 In: Abstracts of the 21st Biennial Conference of the Coastal and Estuarine Research Federation, November 2011, Daytona Beach, FL

Grablow K, Nedimyer K, Northrop A, Precht W (2010) *Acropora cervicornis* restoration: Coral Restoration Foundation's seven-year summary for the upper Keys. Page 69 In: Linking Science to Management – Conference and Workshop on the Florida Keys Marine Ecosystem. Duck Key, FL. Online at: http://conference.ifas.ufl.edu/floridakeys/Program%20Book.pdf

Grablow K, Nedimyer K, Northrop A, Precht W (2010) Comparative growth and survival of *Acropora cervicornis* on concrete disk versus line nurseries. Page 70 In: Linking Science to Management – Conference and Workshop on the Florida Keys Marine Ecosystem. Duck Key, FL. Online at: http://conference.ifas.ufl.edu/floridakeys/Program%20Book.pdf

Graus RR, Macintyre IG (1982) Variations in the growth forms of the reef coral *Montastraea annularis* (Ellis and Solander): a quantitative evaluation of growth response to light distribution using computer simulation. In: Rützler K, Macintyre IG (eds) The Atlantic barrier reef ecosystem at Carrie Bow Cay, Belize. 1. Structure and communities. Smithson Contrib Mar Sci 12:441–464

Grober-Dunsmore R, Bonito V, Frazer TK (2006) Potential inhibitors to recovery of *Acropora palmata* populations in St. John, US Virgin Islands. Mar Ecol Prog Ser 321:123–132

Haug GH, Hughen KA, Sigman DM, Peterson LC, Rohl U (2001) Southward migration of the Intertropical Convergence Zone through the Holocene. Science 293:1304-1308

Hughes TP (1992) Monitoring of coral reefs: a bandwagon? Reef Encounter 11:9-12

Hemmond EM, Vollmer SV (2010) Genetic diversity and connectivity in the threatened staghorn coral (*Acropora cervicornis*) in Florida. PLoS ONE 5(1):e8652.oi:10.1371/journal.pone.0008652

Hollarsmith JA, Griffin SP, Moore TD (2012) Success of outplanted *Acropora cervicornis* colonies in reef restoration. Proc 12th Intl Coral Reef Symp, Cairns, 20A, 5 p

Idjadi JA, Lee SC, Bruno JF, Precht WF, Allen-Requa L, Edmunds PJ (2006) Rapid phase-shift reversal on a Jamaican coral reef. Coral Reefs 25:209-211

Jaap WC (1984) The ecology of the south Florida coral reefs: A community profile. US Fish Wildl Serv, Washington DC

Jaap WC (1998) Boom-bust cycles in *Acropora*. Reef Encounter 23:12–13

Jaap WC, Hallock P (1990) Coral reefs. Pages 574-616 In: Ecosystems of Florida. Meyers RL, Ewel JJ (eds), University of Central Florida Press, Orlando, FL

Jaap WC, Halas JC, Muller RG (1988) Community dynamics of stony corals (Scleractinia and Milleporina) at Key Largo National Marine Sanctuary, Key Largo, Florida during 1981-1986. Proc 6th Intl Coral Reef Symp, Townsville, 2:237-243

Jaap WC, Porter JW, Wheaton J, Hackett K, Lybolt M, Callahan MK, Tsokos C, Yanev G (2001) EPA/FKNMS Coral Reef Monitoring Project: Updated executive summary, 1996–2000. Report to Steering Committee, August 2001

Jessen CA, Rundgren M, Bjorck S, Hammarlund D (2005) Abrupt climatic changes and an unstable transition into a late Holocene thermal decline: a multiproxy lacustrine record from southern Sweden. J Quaternary Sci 20:349-362

Johnson ME, Lustic C, Bartels E, Baums IB, Gilliam DS, Larson L, Lirman D, Miller MW, Nedimyer K, Schopmeyer S (2011) Caribbean *Acropora* restoration guide: Best practices for propagation and population enhancement. The Nature Conservancy, Arlington, VA

Kerwin MW, Overpeck JT, Webb RS, DeVernal A, Rind DH, Healy RJ (1999) The role of oceanic forcing in mid-Holocene Northern Hemisphere climatic change. Paleoceanography 14:200-210

Keck J, Houston RS, Purkis S, Riegl BM (2005) Unexpectedly high cover of *Acropora cervicornis* on offshore reefs in Roatán (Honduras). Coral Reefs 24:509

Lidz BH (2006) Pleistocene corals of the Florida Keys: Architects of imposing reefs-Why? J Coast Res 22:750-759

Lidz BH, Reich CG, Shinn EA (2003) Regional Quaternary submarine geomorphology in the Florida Keys. Geol Soc Amer Bull 115:845-866

Lidz BH, Zawada (2013) Possible return of *Acropora cervicornis* at Pulaski Shoal, Dry Tortugas National Park, Florida. J Coast Res 29(2):256-271

Lighty RG (1977) Relict shelf-edge Holocene coral reef: southeast coast of Florida. Proc 3rd Intl Coral Reef Symp, Miami, 2:215-221

Lighty RG, Macintyre IG, Neumann AC (1980) Demise of a Holocene barrier-reef complex, northern Bahamas. Geol Soc Amer Abstracts Programs 12:471

Lighty RG, Macintyre IG, Stuckenrath R (1978) Submerged early Holocene barrier reef south-east Florida shelf. Nature 276: 59-60

Lin HL, Peterson LC, Overpeck JT, Trumbore SE, Murray DW (1997) Late Quaternary climate change from δ18O records of multiple species of planktonic foraminifera: high-resolution records from the anoxic Cariaco Basin, Venezuela. Paleoceanography 12(3):415-427

Lirman D, Bowden-Kerby A, Schopmeyer S, Huntington B, Thybert T, Gough M, Gough T, Gough R, Gough Y (2010) A window to the past: documenting the status of one of the last remaining “megapopulations” of the threatened staghorn coral *Acropora cervicornis* n the Domincan Republic. Aquat Conserv 20(7):773-781

Lirman D, Schopmeyer S, Manzello D, Gramer LJ, Precht WF, Muller-Karger F, Thanner S (2011) Severe 2010 cold-water event caused unprecedented mortality to corals of the Florida Reef Tract and reversed previous survivorship patterns. PLoS One 6(8): e23047

Macintyre, I.G., 2007. Demise, regeneration, and survival of some western Atlantic reefs, in: Aronson, R.B. (Ed.). Geological Approaches to Coral Reef Ecology. Springer-Verlag, New York, pp. 181-200.

Macintyre IG, Toscano M (2007) The Elkhorn Coral *Acropora palmata* is coming back to the Belize Barrier Reef. Coral Reefs 26:757

Marszalek DS, Babashoff G, Noel MR, Worley DR (1977) Reef distribution in south Florida. Proc 3rd Intl Coral Reef Symp, Miami, 2:223-229

Mayor PA, Rogers CS, Hillis-Starr ZM (2006) Distribution and abundance of elkhorn coral, *Acropora palmata*, and prevalence of white-band disease at Buck Island Reef National Monument, St. Croix, US Virgin Islands. Coral Reefs 25: 239-242

Meesters EH, Hilterman M, Kardinaal M, deVries M, Bak RPM (2001) Colony size-frequency distributions of scleractinian coral populations: spatial and interspecific variation. Mar Ecol Prog Ser 209:43-54

Miller RG (1981) Simultaneous statistical inference. Springer-Verlag, NY

Miller SL, Chiappone M, Rutten LM, Swanson DW (2008) Population status of *Acropora* corals in the Florida Keys. Proc 11th Intl Coral Reef Symp, Ft. Lauderdale, 775-779

Miller SL, Swanson DW, Chiappone M (2002) Multiple spatial scale assessment of coral reef and hard-bottom community structure in the Florida Keys National Marine Sanctuary. Proc 9th Intl Coral Reef Symp, Bali, 1:69-77

Murdoch TJT, Aronson RB (1999) Scale-dependent spatial variability of coral assemblages along the Florida reef tract. Coral Reefs 18:341-351.

Nedimyer K, Grablow K, Northrop K, Precht W (2010) Producing *Acropora palmata* in offshore coral nurseries for reef restoration. Page 128 In: Linking Science to Management – Conference and Workshop on the Florida Keys Marine Ecosystem. Duck Key, FL. Online at: http://conference.ifas.ufl.edu/floridakeys/Program%20Book.pdf

Patterson K, Porter J, Ritchie K, Polson S, Mueller E, Peters E, Santavy D, Smith G (2002) The etiology of white pox, a lethal disease of the Caribbean elkhorn coral, *Acropora palmata*. Proc Natl Acad Sci 99:8725.

Polato NR, Vera JC, Baums IB (2011) Gene discovery in the threatened elkhorn coral: 454 sequencing of the *Acropora palmata* transcriptome. PLoS ONE 6(12): e28634. doi:10.1371/journal.pone.0028634

Porter JW (1987) Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (south Florida) - reef-building corals. US Fish Wildlife Serv Biol Rep, 82(11.73), 23 p

Porter JW, Battey JF, Smith JG (1982) Perturbation and change in coral reef communities. Proc Natl Acad Sci USA 79:1678–1681

Porter JW, Meyers MK, Ruzicka R, Callahan M, Colella M, Kidney J, Rathbun S, Sutherland KP (2012) Catastrophic loss of *Acropora palmata* in the Florida Keys: Failure of the ‘Sorcerer’s Apprentice Effect’ to aid recovery following the 2005 Atlantic hurricane season. Proc 12th Intl Coral Reef Symp, Cairns, 19B

Pitts PA (1994) An investigation of near-bottom flow patterns along and across Hawk Channel, Florida Keys. Bull Mar Sci 54:610-620.

Precht L, Gintert B, Lirman D (2010) Assessing populations of the threatened elkhorn coral, *Acropora palmata*, at Horseshoe and South Carysfort Reefs within the Florida Keys National Marine Sanctuary. Page 143 In: Linking Science to Management – Conference and Workshop on the Florida Keys Marine Ecosystem. Duck Key, FL. Online at: http://conference.ifas.ufl.edu/floridakeys/Program%20Book.pdf

Precht WF, Aronson RB (2004) Climate flickers and range shifts of reef corals. Front Ecol Environ 2(6):307-314

Precht WF, Aronson RB (2006) Death and resurrection of Caribbean coral reefs: a palaeoecological perspective. Pages 40–77 In: Coral reef conservation. Cˆote´ I, Reynolds J (eds), Cambridge University Press, Cambridge, UK

Precht WF, Aronson RB, Moody RM, Kaufman L (2010) Changing patterns of microhabitat utilization by the threespot damselfish, *Stegastes planifrons*, on Caribbean Reefs. PLoS ONE 5(5): e10835. doi:10.1371/journal.pone.0010835

Precht WF, Bruckner A, Aronson RB, Bruckner R (2002) Endangered acroporid corals of the Caribbean: Coral Reefs 21(1):41-42

Precht WF, Goodwin WB (2010) First Report of *Acropora palmata* from the Pleistocene Key Largo Limestone. Page 145 In: Linking Science to Management – Conference and Workshop on the Florida Keys Marine Ecosystem. Duck Key, FL. Online at: http://conference.ifas.ufl.edu/floridakeys/Program%20Book.pdf

Precht WF, Goodwin WB, Nedimyer K (2012) Landscape-scale approaches show much promise for future coral reef restoration projects. Page 434 In: Tropical connections – South Florida's marine environment. Kruczynski WL, Fletcher PJ (eds), IAN Press, Cambridge, MD

Precht WF, Miller SL (2007) Ecological shifts along the Florida Reef Tract: The past is the key to the future. Pages 237-312 In: Geological approaches to coral reef ecology. Aronson RB (ed), Springer, NY

Precht WF, Macintyre IG, Dodge RE, Banks K, Fisher L (2000) Backstepping of Holocene reefs along Florida's east coast. Proc 9th Intl Coral Reef Symp, Bali, 321

Precht WF, Nedimyer K (2010) Seascape-scale approaches to restoring coral reefs and the future of restoration in the Florida Keys National Marine Sanctuary. Page 147 In: Linking Science to Management – Conference and Workshop on the Florida Keys Marine Ecosystem. Duck Key, FL. Online at: http://conference.ifas.ufl.edu/floridakeys/Program%20Book.pdf

Precht WF, Robbart M Aronson RB (2002) The potential listing of *Acropora* species under the U.S. Endangered Species Act. Mar Pollut Bull 49:534-536

Reyes JD, Schizas NV (2010) No two reefs are created equal: fine-scale population structure in the threatened coral species *Acropora palmata* and *A. cervicornis*. Aquatic Biology 10:69–83

Riegl B, Purkis SJ, Keck J, Rowlands GP (2008) Monitored and modeled coral population dynamics and the refuge concept. Mar Pollut Bull 58:24–38

Richardson LL , Goldberg WM, Kuta KG, Aronson RB, Smith GW, Ritchie KB, Halas JC, Feingold JS. Miller SM (1998) Florida 's mystery coral-killer identified. Nature 392:557-558

Richardson, LL, Goldberg WM, Carlton RG, and Halas JC**.** 1998. Coral disease outbreak in the Florida Keys: Plague Type II. Rev Biol Trop 46 Suppl 5:198-198

Rogers CS and Muller EM (2012) Bleaching, disease and recovery in the threatened scleractinian coral *Acropora palmata* in St John, USVI: 2003-2010. Coral Reefs 31:807-819

Ruddiman WF, Mix AC (1991) The north and equatorial Atlantic at 9000 and 6000 yr B.P.. Pages 94-124 In: Global climates since the last glacial maximum. Wright Jr HE, Kutzbach JE, Webb III T, Ruddiman WF, Street-Perrott FA, Bartlein PJ (eds), University of Minnesota Press, Minneapolis, MN

Schelten C, Brown S, Gurbisz CB, Kautz B, Lentz JA (2006) Status of *Acropora palmata* populations off the coast of South Caicos, Turks and Caicos Islands. Proc Gulf Carib Fish Inst 57: 665-678

Shinn EA, Hudson JH, Halley RB, Lidz B (1977) Topographic control and accumulation rate of some Holocene coral reefs: South Florida and Dry Tortugas. Proc 3rd Intl Coral Reef Symp, Miami, 2:1-7

Shinn EA, Hudson JH, Robbin DM, Lidz B (1981) Spurs and grooves revisited: construction versus erosion Looe Key Reef, Florida. Proc 4th Intl Coral Reef Symp, Manila, 1:475-483

Shinn EA, Lidz BH, Kindinger JL, Hudson JH, Halley RB (1989) Reefs of Florida and the Dry Tortugas. U.S. Geological Survey, St. Petersburg, 53 p

Smith SG, Swanson DW, Chiappone M, Miller SL, Ault JS (2011) Probability sampling of stony coral populations in the Florida Keys. Environ Monit Assess 183:121-138

Swanson DW (2011) Spatial dynamics of coral populations in the Florida Keys. Open Access Dissertations, Paper 626, http://scholarlyrepository.miami.edu/oa_dissertations/626

Szmant AM, Miller MW (2005) Settlement preferences and post-settlement mortality of laboratory cultured and settled larvae of the Caribbean hermatypic corals *Montastraea faveolata* and *Acropora palmata* in the Florida Keys, USA. Proc 5th Intl Coral Reef Congr, Tahiti, 4:295-300

Szmant AM, Weil E, Miller MW, Colon DE (1997) Hybridization within the species complex of the scleractinan coral *Montastraea annularis*. Mar Biol 129:561-572

Vargas-Angel B, Thomas JD, Hoke SM (2003) High-latitude *Acropora cervicornis* thickets off Fort Lauderdale, Florida. Coral Reefs 22:465-473

Vaughan TW (1914) Investigations of the geology and geologic processes of the reef tracts and adjacent areas of the Bahamas and Florida. Carnegie Institution of Washington Yearbook 12:1-183

Vollmer SV, Kline DI (2008) Natural disease resistance in threatened staghorn corals. PLoS ONE 3(11): e3718. doi:10.1371/journal.pone.0003718

Vollmer SV, Palumbi SR (2007) Restricted gene flow in the Caribbean staghorn coral *Acropora cervicornis*: implications for the recovery of endangered reefs. Journal of Heredity 98(1):40–50

Walker ND, Roberts HH, Rouse LJ Jr, Huh OK (1982) Thermal history of reef-associated environments during a record cold-air outbreak event. Coral Reefs 1:83–87

Williams DE, Miller MW (2011) Attributing mortality among drivers of population decline in *Acropora palmata* in the Florida Keys (USA). Coral Reefs 31:369-382

Zimmer B, Precht WF, Hickerson EL, Sinclair J (2006) Discovery of *Acropora palmata* at the Flower Garden Banks National Marine Sanctuary, northwestern Gulf of Mexico. Coral Reefs 25:192

Zubillaga AL, Márquez LM, Cróquer A, Bastidas C (2008) Ecological and genetic data indicate recovery of the endangered coral *Acropora palmata* in Los Roques, Southern Caribbean. Coral Reefs 27:63–72