MIAMI HARBOR DEEP DREDGE PROJECT: A REAPPRAISAL REVEALS SAME RESULTS

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ABSTRACT

As part of the permit conditions for the Miami Harbor expansion project, 26 individual sites consisting of 78 permanent 20-meter long transects (3 per site) were established and some 643 scleractinian corals (including 286 controls) were tagged for repeated monitoring. During each site monitoring visit, in situ coral-condition data were collected by scientific divers for all tagged colonies. Multiple still photographs of each tagged coral colony and downward-facing video of all transects were also collected. Between the fall of 2013 and the summer of 2016, each site was monitored up to 50 times (events) depending upon each site's location with respect to active dredging operations. In situ coral condition data were compared with corresponding still photographs for cross-verification and validation. Combining these in situ coral data with other biotic (video functional assessments) and abiotic metrics (sediment, temperature, etc.) allowed us to differentiate between chronic and acute stressors and whether they were natural or anthropogenic in origin. More specifically, we were able to calculate the prevalence of corals impacted by sedimentation, predation, competition, coral bleaching, and disease. Most importantly, in cases where corals had died, we were able to discern the exact cause of mortality by carefully evaluating the sequence of events recorded (and photographed) prior to their death. Opponents to the project questioned the veracity of the original results. Most of these claims were based on one-off surveys and cherry-picked analysis, using only a small, carefully selected subset of the overall dataset.

Using all the available data collected during the project, the data was reanalyzed, specifically concentrating on the cause of mortality of stony corals. Our results indicate that sedimentation impacts diminished in time, indicative of a pulsed impact. In testing for effects of sediment on coral, octocoral, and sponge abundance, we found no statistically significant evidence for decline at both the site and regional level. Coral morbidity (partial mortality) associated with sedimentation was highest at sites immediately adjacent to the dredge operations, however, few corals actually died (7 colonies, <2%) from project related impacts. The combined losses from sediment burial, bleaching, competitive mortality, and white-band disease accounted for the mortality of only 4.1% of all monitored corals, yet 31.7% of all tagged corals died as a direct result of white-plague disease. Thus, the greatest source of coral mortality was associated with a coral disease outbreak associated with a regional sea-surface temperature thermal anomaly and not local dredging operations.

The regular monitoring of tagged corals at control and near project sites provided the detailed information needed to assign the correct cause of mortality to corals in the project area as opposed to the undocumented assertions of project opponents. The actual monitoring results from the project emphasize the requirement for implementing scientifically-based, not ideologically-based management of natural systems to best understand and protect our fragile coral resources.

Keywords: PortMiami, dredging, sediment, turbidity, coral reefs, monitoring, environmental impact assessment.

INTRODUCTION

Dramatic statements were made by project opponents about the effects of the dredging in Miami, specifically regarding sedimentation impacts to offshore coral resources (Silverstein 2014, 2015). These statements based on a paucity of scientific data and one-off site visits garnered significant media attention (Staletovich 2014a, 2014b, 2015a, 2015b, 2016, Gomez 2015, Nelson 2015, Alverez 2016, Caole Vila 2016). However, results from both long-term compliance and post-construction biological monitoring showed that while some project impacts did occur, these were in accordance with numerous published accounts from earlier dredging programs associated with port infrastructure projects (Sheppard 1980, Brown et al. 1990, Koloi et al. 2005, Adjeroud et al. 2017).

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Since 2013, no less than 35 media articles, blog posts, op-eds, and commentaries have been published about purported dredging impacts at PortMiami. As a result, the view that the PortMiami dredging project was deleterious to the marine environment, especially coral reef hardbottoms and associated seagrass ecosystems have proliferated. However, many of the negative viewpoints, including those published in nationally recognized newspapers are based on supposition and anecdote and not on scientific data (Gomez 2015, Staletovich 2014a, 2014b, Alverez 2016). While bold negative headlines and exaggerated claims may sell newspapers, they are based on unscientific ideological approaches to resource management and often the unsupported assertations of project opponents, many with their own biased agenda. Unfortunately, these unscientific approaches promote circular reasoning, aid in the development of incorrect paradigms, and hinders our ability to preserve the vital resources we are charged with protecting. As a result of the negative press the project received, in conjunction with numerous lawsuits promulgated by local NGO's (Silverstein 2014), the actual scientific results of the project were dismissed by project opponents and regulators. The perception was that the dredging project had far greater environmental impacts than originally anticipated or reported – both in time and space. In response to these concerns, we re-evaluated the project by carefully analyzing the in-situ field data with their corresponding underwater photographs and video collected before, during, and after the project. These results, while essentially indistinguishable from our original analyses (DCA 2015a, 2015b, 2015c, 2015d, 2017), help to form an integrated 'lessons learned' from this project, which in-turn can be used as a practical guide for protection of valuable coral resources on future planned infrastructure projects throughout the topics and sub- tropics.

The Project

The Miami Harbor Phase III Deepening Project was the largest dredging project to be conducted in the region during the past decade. Specifically, Miami Harbor was the first of four eastern seaboard ports to be deepened to accommodate larger post-Panamax vessels. The existing channel, which had been repeatedly dredged since the early 1900's, was deepened from 42 feet to 50 feet. In total, five different dredges removed more than five million cubic yards of limestone rock and sand. Construction began in late 2013 and was completed in the summer of 2015 (USACE 2018).

Biological Monitoring

Dial Cordy and Associates, Inc. (DCA) was contracted to perform the compliance monitoring for the project. This monitoring program was the most detailed ever performed on a US Army Corps of Engineers (USACE) infrastructure project performed to date. In total, 26 permanent monitoring sites, 15 at dredge-adjacent locations and 11 far-field controls were developed by the Florida Department of Environmental Protection (FDEP) for compliance monitoring under FDEP Permit No. 0305721-001-BI as part of the Miami Harbor Phase III Deepening Project (FDEP 2012). Of the 15 dredge-adjacent sites, seven (7) were in nearshore hardbottom habitats, four (4) in inner reef habitats, and four (4) in outer reef habitats (Figure 1). Each monitoring station was established as part of a paired Before-After-Control-Impact (BACI) design (Green and Smith 1997) in which dredge-adjacent sites were paired with a far-field control within their habitat and located on the same side (North or South) of the Miami Harbor channel (gray colored area on Figure 1).

As mandated by the FDEP permit, at each site, three (3) permanent 20-meter transects were established parallel to each other, in a north (0 m) to south (20 m) direction. Transects were spaced at least five meters (5 m) apart. Stainless steel eyebolts (1 cm x 20 cm) were drilled into the bottom at 0 m, 10 m, and 20 m along each transect. Small closed-cell foam floats, coated with anti-fouling paint, were attached to each eyebolt with a short length of nylon braided line to aid in transect relocation. Two floats marked the beginning of each transect, whereas mid and end points were marked with a single float to orient divers and relocate sites.

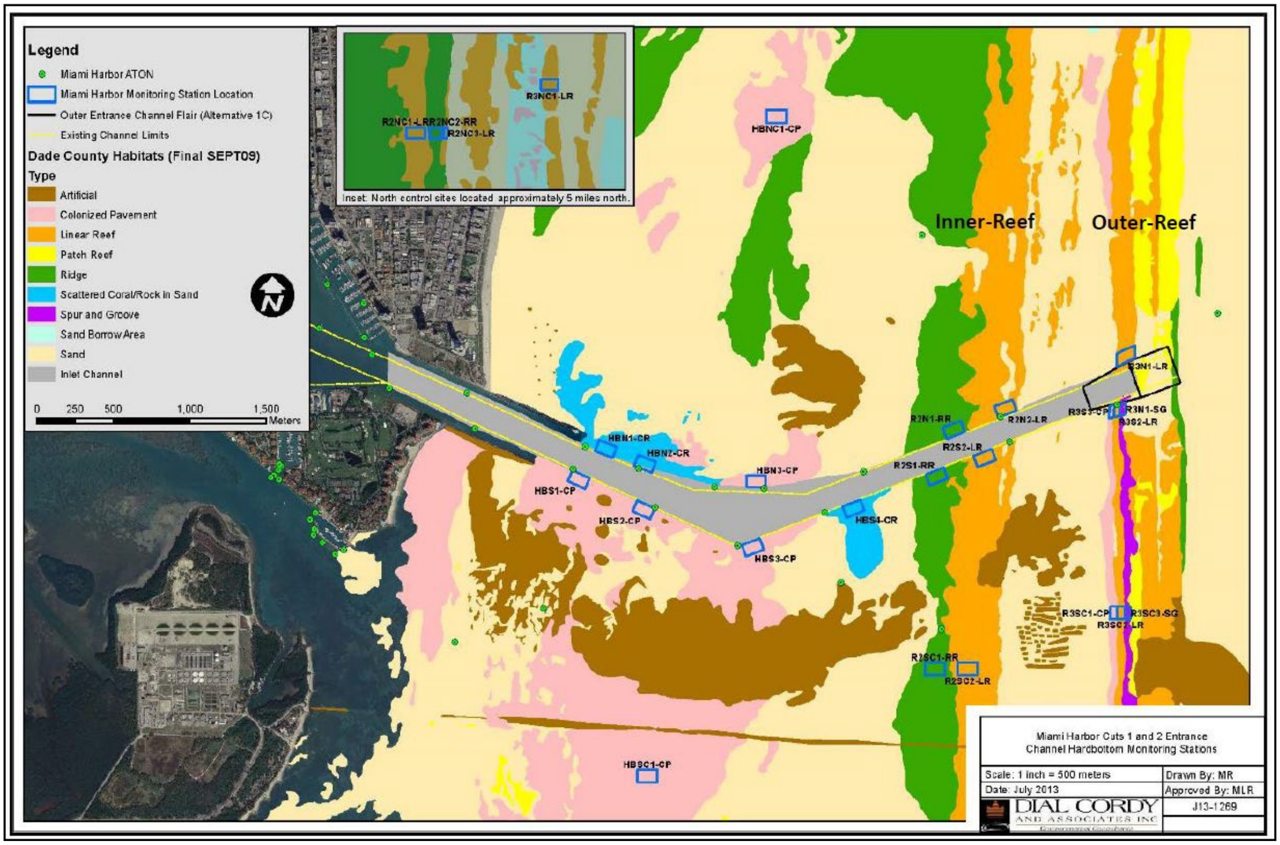


Figure 1. Location map of 26 monitoring sites (blue boxes) associated with the PortMiami Deepening Project. Site abbreviations are noted next to each box. Brown horizontal line across the lower portion of the figure is the location of the outfall pipe trench for the Central Miami-Dade Wastewater Treatment Plant.

At each of the transects, video data was recorded for subsequent analysis in the laboratory. In addition, on each transect, the USACE required a set of up to ten (10) stony coral colonies located within one meter (1 m) of the line were to be monitored for in-situ coral-stress (USACE 2012). Each coral colony that was selected for monitoring was subsequently marked with a numbered yellow plastic tag attached to the hardbottom, next to the colony. The in-situ coral- condition data were collected by trained scientific divers for all tagged stony coral species. Still photographs of each of the permanently marked corals were taken in planar view, so that each colony was present within a single photo frame, along with the permanent marker and scale bar. A total of 643 healthy scleractinian coral colonies from 23 species were initially tagged within the 26 study sites. Most tagged corals were small, ranging from <10–40 cm in diameter. During construction, a monitoring 'event' was triggered anytime the active dredge operations were within 750 m of a site during any given week. In the laboratory, in-situ coral condition data were compared to corresponding still photographs for cross-verification and validation (QA/QC).

Comparisons of far-field reference (control) sites and channel-side (compliance) sites were made weekly during construction activities to satisfy permit conditions. These included: (1) evaluating benthic organisms (scleractinian corals, octocorals, sponges, etc.) for standing sediment that was not removed by normal currents or wave action; and (2) evaluating scleractinian corals along each transect for additional indications of sedimentation stress such as excessive mucus, extruded polyps, and color changes (bleaching or paling). All scleractinian corals on each transect were assessed for each of the health parameters and assigned a health level of '0' or '1' for each parameter (a score of '0' would indicate no observed stress, while a '1' would be indicated for an observed stress parameter. Comparisons are then made between the channel-side sites and their paired far-field controls. Reference and channel-side site mean values were then compared using a two-sample t-test. If the channel-side corals showed a higher frequency of stress than corals at their respective reference sites (two-sample t-test ≤0.05), the weekly impact was considered significant.

Over the course of the project from 2013 - 2016, the DCA scientific dive team performed >10,850 project-related dives. During these dives, >35,000 in situ coral observations were made and >75,000 photographs were recorded of the 643 tagged colonies. In addition, each of the 20-m transects were videographed each time a site was monitored. Approximately, 3,000 video-transects were recorded and analyzed. Functional group ecological assessments of these transects were performed using CPCe coral point count software (Kohler and Gill 2006) by trained analysts. A total of ~40 non-overlapping still frames were captured for each transect and 10 random dots were generated and overlaid on each captured image. This resulted in using some 1,200,000 data points in the functional analysis (DCA 2015a, 2015b, 2017).

Project Goals

One of the primary goals of PortMiami monitoring was to separate and quantify local dredge-related impacts from regional trends in coral community health. A critical component of the repeated measures tagged coral surveys was to examine the causes of coral stress experienced during construction and post-construction periods and determine if this 'stress' was caused or exacerbated by dredging activities.

The tagged corals repeated measures design used in this study was based on resampling replicates (e.g. sites) over time. The repeated measures design tests for interactions of time with treatment. In such designs, impacts are determined by the variability of time trends among sites. Thus, the repeated measurement of experimental units over time is now central to many environmental impact investigations on coral reefs (Green and Smith 1997). In the case of dredging impacts on corals, there may be a number of possible outcomes resulting from repeated measures sampling. In one case, the cumulative effects of chronic sedimentation (press impacts) may not be expressed until well after the commencement of the dredging operations. In this case we would expect to see coral health decline with time. In a second scenario, impacts due to sedimentation may occur over a short period of time (pulsed impacts). In this second scenario we would expect to see differences among the affected corals diminish as time passes. In a third case, other natural stressors (i.e. regional bleaching events, disease outbreaks, and major storms) may be acting simultaneously or synergistically causing coral health to be affected by sources other than dredge operations. A key point in all three cases is that many impacts may not be detected by monitoring if the response variable (coral) is measured at only a single point in time, the temporal intervals are too widely spaced to distinguish between the various causes, or if the measurements are made after the stress events have ceased. Thus, a closely spaced repeated measure design reduces the chances of drawing invalid conclusions leading from the logical fallacy that 'correlation does not imply causation.' Combining these repeated measures data with other collected abiotic metrics (sediment, turbidity, water temperature, wave and current data, etc.) allows environmental scientists and regulators to differentiate between chronic and acute stressors, both natural and project related. These data allow for more precise calculations of project-related impacts which, in-turn, can help guide future projects.

Most importantly, in cases where impacts occur, it should be possible to differentiate and discern the actual proximal cause of the impact to the coral reef community by carefully evaluating the sequence of events recorded prior to and during the impact. As previously noted, it is important to know whether the dredging project associated with the deepening and expansion of PortMiami had an impact by locally exacerbating coral stress ultimately leading to coral mortality.

Results: PortMiami Re-Evaluation

The Role of Dredging

Functional group percent cover data describe the overall composition of benthic organisms and abiotic cover at a site. Project-related sites were assessed in terms of the percent cover of corals, octocorals, sponges, zoanthids, macroalgae, CTB (crustose coralline algae, turf and bare), sand, and other during baseline and impact assessment periods. The mean percent cover of benthic invertebrates was approximately 17% of the bottom at the channel-side sites during baseline surveys: stony corals (0.88%), octocorals (10.01%), sponges (5.01%) and zoanthids (1.13%), while CTB and sand comprised the remaining 83% of the benthic cover. During impact assessment surveys the mean percent cover of benthic invertebrates was again 17% of the bottom at channel-side sites: stony corals (0.51%), octocorals (9.18%), sponges (5.78%) and zoanthids (1.13%), while CTB and sand comprised the remaining 83% of the bottom at channel-side sites. The functional group percent cover analysis documented that mean cover of corals, octocorals, sponges and zoanthids was within a standard error of baseline values at channel-side and control locations in each of the sampled habitats. Results of benthic invertebrate cover (stony corals, octocorals and sponges) from the SECREMP monitoring program (Gilliam et al. 2018) for sites in Miami-Dade County are essentially invariant with the functional group percent cover reported above (see also DCA 2017). Even though regional coral cover was < 1%, we noted a 37% reduction in coral cover over the course of the project using the video transect data. This is remarkably similar to the overall mortality observed (36%) using our tagged coral data discussed below.

Temporary impacts due to the project were documented as increased levels of sand cover and nearly reciprocal declines in CTB cover at channel-side locations. These differences were greater than changes documented at the control sites over the same time period. Overall, mean sand cover increased at channel-side sites from 13.6% to 29.3% (15.7% increase) from baseline to impact assessment surveys in comparison to a 0.6% increase in mean sand cover at control sites. A corresponding decline in the mean cover of CTB was also measured between baseline and impact assessment periods with a decline in mean CTB cover declining from 70.5% to 54.8% (-15.7%) at channel-side locations compared to a 3.7% increase in mean CTB cover at control sites. Increased sand cover was spatially restricted to inner reef and southern hardbottom channel-side sites during the impact assessment surveys (DCA 2015c, 2015d). These increases in sand cover documented at channel-side sites were within the variability of sand cover documented over time at the control sites. At control locations, sand cover varied as much as 68.3% over the course of project monitoring due to seasonal variability. The range mean sand cover at control site locations was 40.0% over the course of the project. The increased sand cover documented at channel-side sites during the impact assessment survey is within the range of control site variability. The increase in sand cover channel-side is expected to be ephemeral, as sand cover at channel-side sites has declined since construction completion and continues to trend downward toward baseline values.

By late 2016, 31 of the original 643 tagged corals were missing. A temporal evaluation of the remaining 612 corals confirms earlier results that show coral mortality associated with sediment stress and burial, while present, was minimal. In total, 219 corals died (36%) over the course of the project and only 7 colonies; < 2% of all tagged corals died as a result of sediment burial. An additional six (6) corals died as a direct result of competitive interactions, four (4) corals died as a result of coral bleaching, three (3) colonies of *Acropora cervicornis* died from white-band disease, five (5) corals died from an unknown disease referred to as receding margin syndrome (a.k.a. Turf/Algal/Sediment, TAS code in AGRRA 2018), while 194 corals died from impacts associated with white-plague disease.

At some sites, we observed significant but localized project related partial mortality. Partial mortality directly associated with sediment was documented at PortMiami and affected 64.8% of corals at the channel-side sites and 19.4% of corals at reference sites over the course of the project (DCA 2017). This was commonly observed as a halo of dead tissue (generally <10% of total live tissue) surrounding the very base of the colony (Figure 2). Across the inner reef sites, R2N1-RR recorded the highest percentage of corals affected by partial mortality (93%), R2N2-LR, R2S1-RR and R2S2-LR all exhibited the next highest percentage of corals with partial mortality due to sediment (63%). The two north control sites (R2NC1-LR and R2NC2-RR) had the lowest percentage of corals affected by partial mortality due to sediment (7%). The two south control sites had 30% (R2SC1-RR) and 8% (R2SC2-LR) of corals affected by partial mortality due to sediment. At the outer reef sites, more than 70% of all tagged corals at R3N1-LR exhibited partial mortality due to sediment, while R3NC1-LR had 29% of corals affected by partial mortality. The south side of the channel at the outer reef sites exhibited less sediment-related partial mortality when compared to the north channel-side outer reef site. R3S2-LR had the lowest percentage with only 4% of corals with partial mortality due to sediment, while R3S1-CP and R3S3-SG had percentages of 42% and 36% respectively. R3SC2-LR had the lowest percentage of partial mortality (0%) among the south controls while R3SC1- CP (17%) and R3SC3-SG (13%) exhibited higher percentages.

The finding that dredging did not result in mass coral mortality related to sediment-stress is also consistent with other historical dredging projects performed throughout the south Florida region (Antonius 1974, Courtney et al. 1974, Griffin 1974, Marszalek 1982, Spring and Hodell 2011). These studies, especially those performed in the early 1970's, set a solid foundation upon which dredging impacts could be compared and contrasted in waters adjacent to coral reefs.

In one study, the Harbor Branch Foundation initiated a field research project to investigate the impact of dredge and fill operations on the Florida Keys reef ecosystem (Antonius 1974, Griffin 1974). Their results suggested that little sediment was transported from nearshore dredging operations in the Keys out to the reefs. Some of the most important observations in the Griffin (1974) study include the following:

- The area of relatively intense plume, turbidity greater than 40 mg/l, rarely extended more than 100-200 m from the dredge.

- Concentration vs. distance plots show that the plume suspensate settles normally, with surface concentration declining in a logarithmic manner and gradually fading into the background turbidity. In general, the area of plume influence rarely exceeds the limits of an area extending about 500 m down current from the active dredge.

- Natural turbidity varied moderately in time and space. These natural variations are related to wind stress, resulting in higher turbidity especially during the winter and spring.

- Waves and currents wafted nearly all of the fine-grained dredge effluent out of the project area within a few months following cessation of dredge operations.

- Considering the natural turbidity level and the measured spread of effluent from the dredge, it seems that the patch reef studied was too distant to have been affected by the dredging. In other words, a reef situated at least one-half mile from a dredge project in the Keys is not likely to be affected. This conclusion coincides with the results of the biological team (Antonius 1974). They monitored the health of the patch reef at the initiation of the project (November 1972) and after its termination (November 1973). Based on a quantitative quadrat surveys, they reported that no detectable changes had occurred and that the percentages of live and dead coral were identical before and after dredging.

Subsequently, additional studies from dredging projects in the region (Courteney et al. 1974, Marszalek 1982) have replicated these earlier results. The key points raised by these studies include the following:

- Turbidity plumes visibly extend from the dredge generally in the direction of prevailing currents.
- Silt and sand settle rapidly from the turbidity plume forming a visible layer of sediment on the reef surface.
- Octocorals are the most tolerant of the reef macrofauna to sediment loading and dredging induced turbidity.
- Sponges that were covered in silt-sized particles eventually "sloughed-off" the sediment which appeared to be bound in a mucoid material.
- No permanent damage to sponges were observed.
- All coral species present in the study area (southeast Florida) showed similar tolerance to dredging.
- Tissue loss in stony corals (partial mortality) was generally confined to a rim around the base of the colony.
- The greatest impacts were found immediately adjacent to the cutter head dredge operations.
- While scleractinian corals appeared the most impacted of the reef macrofauna – no mass mortality of corals occurred.

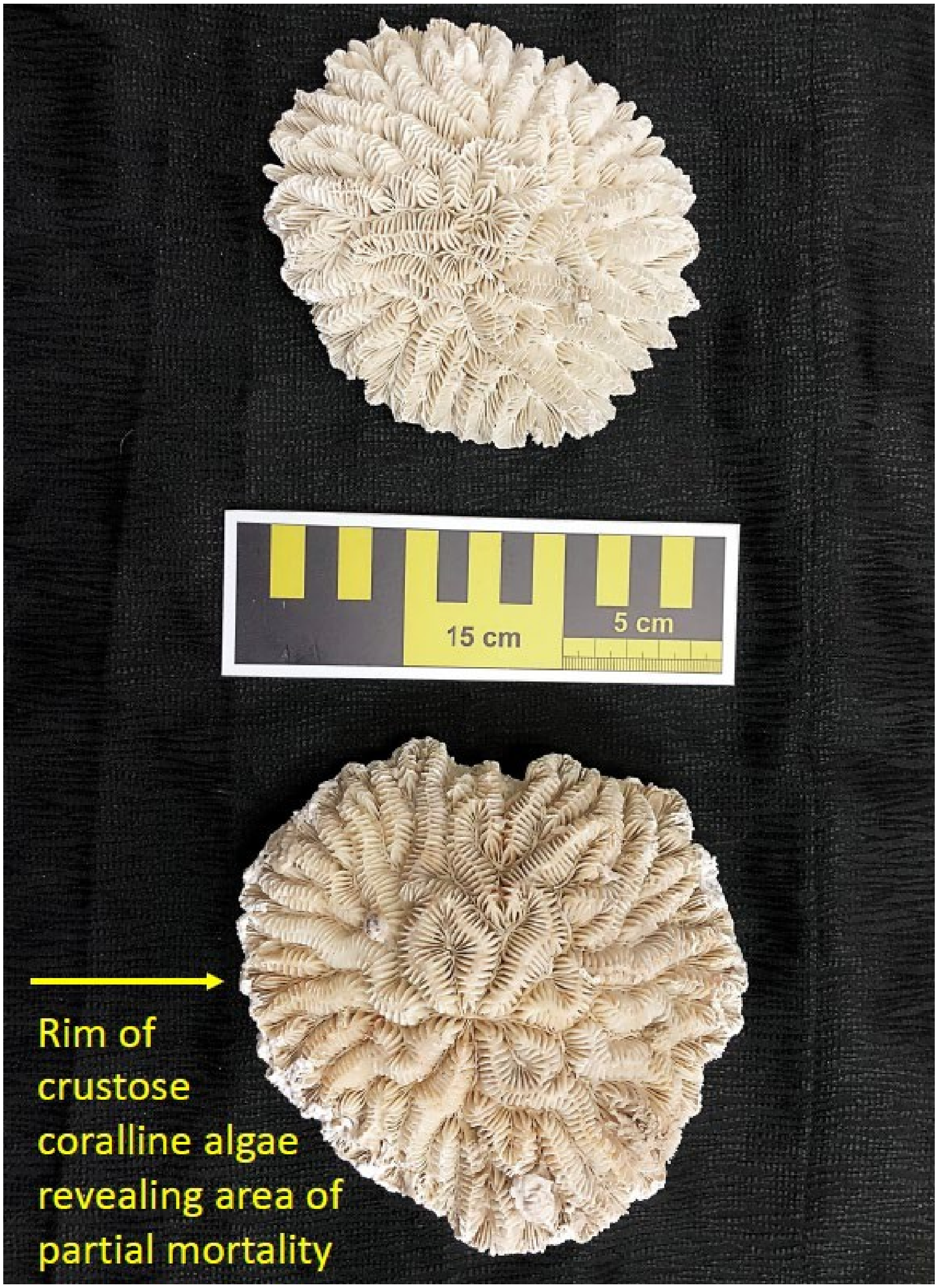


Figure 2. Photographs of *Meandrina meandrites*, both from the inner-reef killed by white-plague disease. Coral in top of photograph is from southern far-field control monitoring site (R2SC2-LR). Coral in bottom of photo is from channel-side compliance monitoring site (R2N2-LR). Note the narrow rim (halo) of crustose coralline algae that colonized area of partial mortality associated with sediment burial around base of colony prior to subsequent whole-colony mortality associated with regional disease outbreak.

The Role of Regional Thermal Anomaly – Coral Bleaching & Disease

The greatest impacts associated with coral mortality were not project related and could be attributed to a catastrophic, regional coral bleaching/disease outbreak. Anomalously high sea-surface temperatures recorded during the summer of 2014 contributed to a significant region-wide coral bleaching event that started in August 2014 (NOAA 2014a, 2014b, Manzello 2015), with a second, subsequent bleaching event occurring in 2015 (NOAA 2015; Figure 3). After the 2014 bleaching event, the monitoring program noted a white-plague coral disease (a.k.a. Stony Coral Tissue Loss Disease) outbreak spreading throughout the project area, initiating in the vicinity of the southern control sites (Precht et al. 2016, DCA 2017, FKNMS 2018).

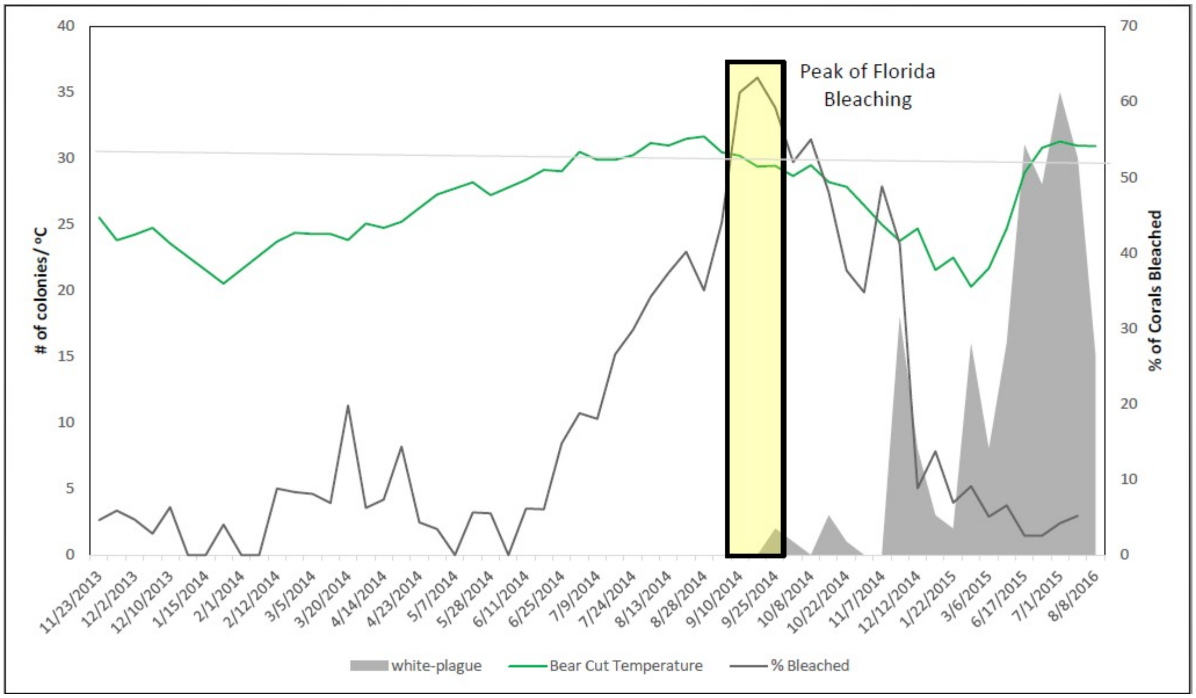


Figure 3. Graph showing the relationship between measured sea surface temperature (SST) and prevalence of coral bleaching and white-plague disease at our 26 repeated-measures monitoring stations. Temperature was downloaded from NOAA National Data Buoy Center, Virginia Key Station (VAKF1), located off the southern tip of Virginia Key in Bear Cut, The Virginia Key Station is denoted by solid green line (NOAA 2018). Horizontal line across the graph at 30.4 °C represents regional coral bleaching threshold from Manzello et al. (2007). Coral bleaching prevalence is the proportion of bleached tagged coral colonies recorded during daily surveys. Yellow bar highlights the period of peak coral bleaching. White-plague disease prevalence (including recently dead corals) is shown as solid grey curve. Note that the peak white-plague disease prevalence lags behind peak coral bleaching, which in-turn lags behind peak SSTs. This is identical to the onset and progression of a white-plague disease outbreak following the 2005 coral bleaching event in the U.S.V.I. (Miller et al. 2009).

By the end of the project, white-plague disease was widespread across all hardbottom, inner reef, and outer reef stations accounting for >85 percent of the total hard coral mortality at these sites (n=194 colonies). Importantly, nearly equivalent levels of disease-related mortality were observed between far-field control (29.4%) and dredge-adjacent locations (33.6%). These results confirmed a regional-scale disease outbreak (Precht et al. 2016), as opposed to one driven by project activities. Similar monitoring results were recorded throughout Miami-Dade County by other researchers throughout this period (Carsey et al. 2016, CSi 2016, DERM 2016, Hayes et al. 2017, Walton et al. 2018, Gilliam et al. 2018). Significant coral mortality associated with a regional white-plague coral disease outbreak was observed at channel-side and control locations over the course of project monitoring, starting in the fall of 2014 (Figures 4 & 5). This is contrary to reports that indicted the Miami Harbor dredging program for initiating and exacerbating the impacts of this disease outbreak (Miller et al. 2016).

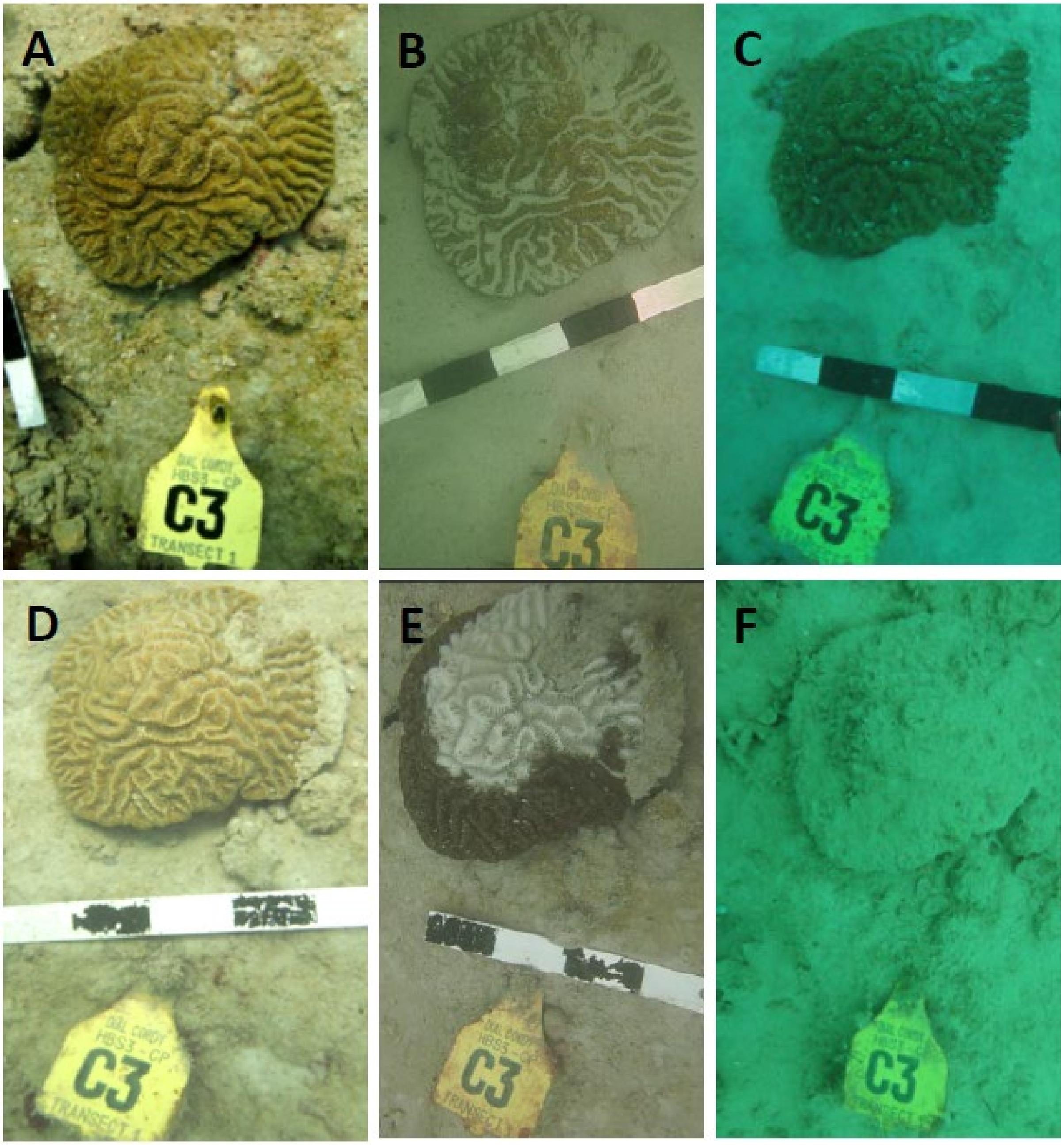


Figure 4. Photographic montage of tagged *Meandrina meandrites* colony at channel-side hardbottom site (HBS3-CP-T1-C3). (A) Photograph of coral during initial baseline surveys on October 22, 2013; (B) coral during compliance week 4 of monitoring, December 12, 2013 revealing peak of sediment stress at that site including partial burial of colony, note blanket of fine sediment covering the bottom; (C) colony showing partial mortality associated with sediment burial during compliance week 18 on March 20, 2014; (D) coral showing signs of bleaching (paling) associated with warm-water thermal stress during compliance week 42 on September 05, 2014, note the bottom is no longer covered in blanket of fine sediment and the residual rim of partial mortality on right-side of colony is exposed from earlier, acute sediment stress; (E) coral showing the signs of active white-plague disease during compliance week 62 on January 22, 2015, also note that coral had regained its color associated with the return of zooxanthellae associated with seasonal cooling of water temperatures; (F) total colony mortality from white-plague disease, photo taken during compliance week 69 performed on March 8, 2015.

None of the 26 sites surveyed had levels of white-plague disease mortality that were higher than regional estimates of disease-related mortality (DCA 2017). These results show the limited influence of the local environment, including conditions immediately adjacent to a long-term dredging project, on the fate of corals throughout the regional white-plague disease epidemic.

The highly conserved patterns of species-level mortality observed throughout the southeastern Florida (Hayes et al. 2017, Lunz et al. 2017, Neely 2017, FKNMS 2018) suggest genetic or physiological differences between coral species were likely key determinants of white-plague disease survivorship both in Miami-Dade County and regionally. To date, the etiology of the disease remains a mystery.

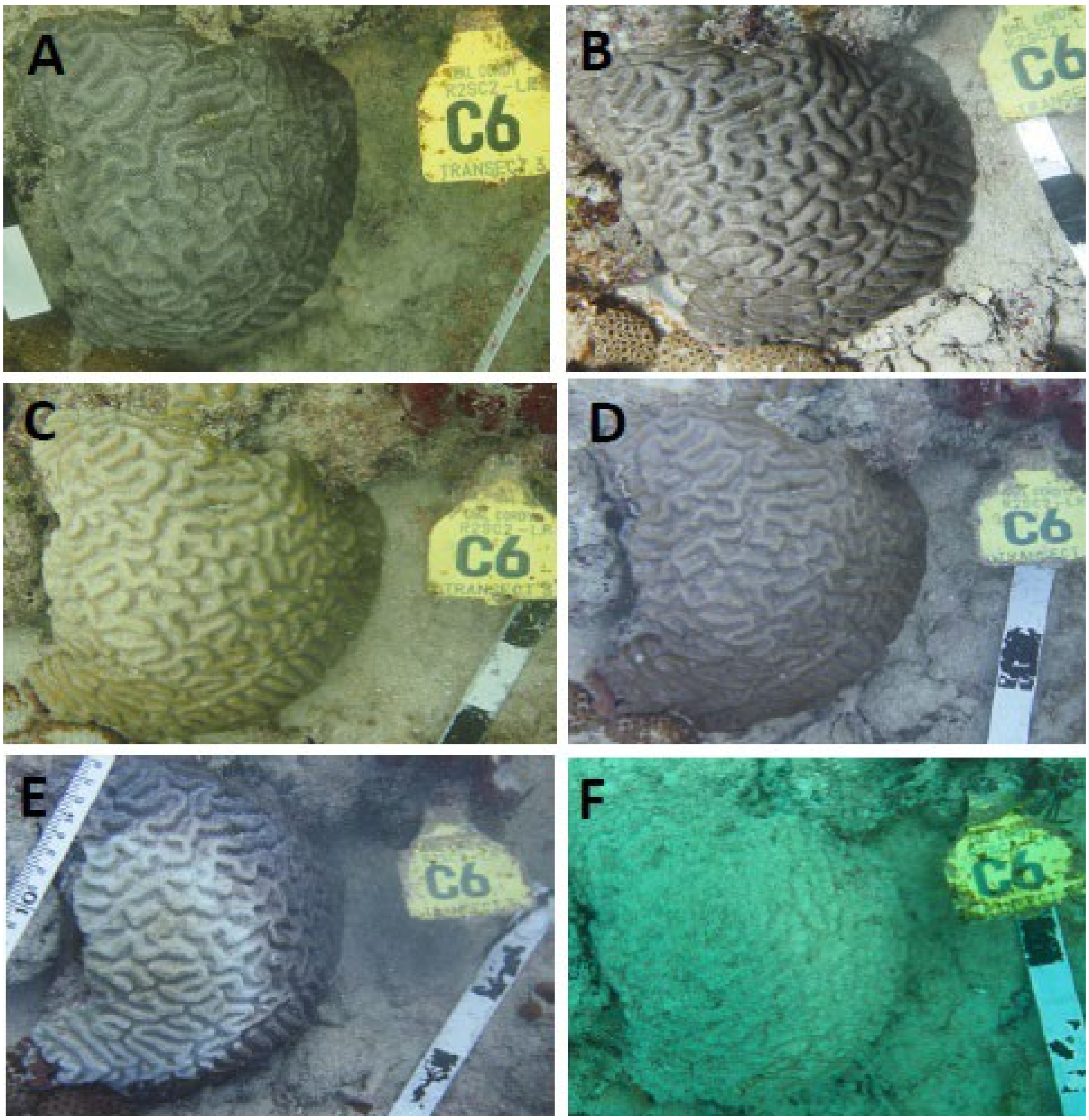


Figure 5. Photographic montage of tagged *Colpophyllia natans* colony at southern far-field control site (R2SC2-LR-T3-C6). (A) Photograph of coral during initial baseline surveys on December 2, 2013; (B) coral during compliance week 35 of monitoring, July 18, 2014; (C) coral showing signs of bleaching (paling) associated with warm-water thermal stress during compliance week 44 on September 18, 2014; (D) colony during compliance week 48 on October 17, 2014 showing post-bleaching return of zooxanthellae associated with seasonal cooling of water temperatures; (E) coral showing the signs of active white-plague disease during compliance week 56 on December 12, 2014, (F) total colony mortality from white-plague disease, photo taken during compliance week 69 performed on March 17, 2015, note the coral is covered by fine-grained sediment. Like the *M. meandrites* colony shown in Figure 4, this coral shows the typical sequence of coral bleaching (paling) followed by infection with white- plague disease resulting in whole coral mortality. Disease susceptible species like *M. meandrites* and *C. natans* show the same level of impact irrespective of location (channel-side or far-field controls).

Comparison with Other Projects

Presently, the only other in situ analysis showing a purported relationship between dredging and increased coral disease prevalence is an example from the Gorgon Dredging Project at Montebello and Barrow Islands in Western Australia where approximately 7.6 million tons of marine sediment were removed over an 18-month period from May 19, 2010 to November 7, 2011 (Chevron Australia 2011). Mean levels of coral disease at high dredge-plume exposure (two sites) near the dredge operation were found to be significantly higher (7.26% of corals) than the mean level of coral disease at sites with lower exposure (3.1% of corals) recorded during a one-off coral survey performed more than a year after completion of dredging activities in December 2012 (Pollock et al. 2014). While Pollock et al. (2014) conclude that the presence of higher mean disease prevalence at sites located near the dredging operation is a clear link between sediment and disease, their site-level data does not support this conclusion. The spatially distinct pattern of greater disease-infection at high exposure sites with levels of disease diminishing with distance presented by Pollock et al. (2014) was not observed at Port Miami (DCA 2017), comments in Miller et al. (2016) notwithstanding.

In the Pollock et al. (2014) study, out of the five sites that were categorized as either heavy (two sites) or moderate dredge-plume exposure (three sites), only one heavy exposure site (50% of those surveyed) and one moderate exposure site (33.3% of those surveyed) had rates of coral disease that were outside the range observed at low or no exposure sites. Levels of exposure to dredging were based on the daily extent of dredge plumes analyzed from satellite imagery (Evans et al. 2012). The combined site-level results show that 60% of moderate and heavy exposure sites did not show elevated disease prevalence. Even more interesting, the site with highest dredge-plume exposure (347 days of turbidity stress) did not have levels of disease that were elevated above the range observed at low or no exposure sites (0-9 days of dredge-plume exposure; Pollock et al. 2014). Pollock et al. (2014) provided no explanation as to how or why the site with highest exposure would have the same level of coral disease as a low or no exposure location.

Second, the most prevalent coral disease observed by Pollock et al. (2014) was white-syndrome. White-syndrome occurs predominantly on acroporid corals (Work and Aeby 2011). Why is this important? Increased disease infection, especially white-syndromes attributed to dredging in Pollock et al. (2014) followed a strong region-wide, La Niña-induced bleaching event that produced temperatures exceeding bleaching thresholds from December 2010 - April 2011, resulting in extensive coral morbidity and mortality (Bureau and Meteorology 2012, Moore et al. 2012, Pearce and Feng 2012, Joesephitis et al. 2012). The Joesephitis et al. (2012) study analyzed the reefs at Montebello and Barrow during the 2010-2011 regional thermally-induced bleaching event. They noted that differences in the amount of coral cover and the extent of bleaching were largely attributable to differences in *Acropora* spp. abundance between sites with acroporid corals showing higher levels of bleaching incidence. Therefore, it stands to reason that if the acroporids were the species that where preferentially diseased by white-syndromes in the Pollock et al. (2014) study, and those were the same species that were preferentially bleached in the Joesephitis et al. (2012) study, the two events may be inextricably linked (see Bruno et al. 2007). Unfortunately, there are no species- specific coral disease data presented in Pollock et al. (2014) limiting our ability to understand if the disease preferentially occurred in acroporid species that were more susceptible to bleaching during the 2010-2011 event. Remarkably, this major thermal anomaly and concomitant regional coral bleaching event was not mentioned or cited in the Pollock et al. (2014) manuscript.

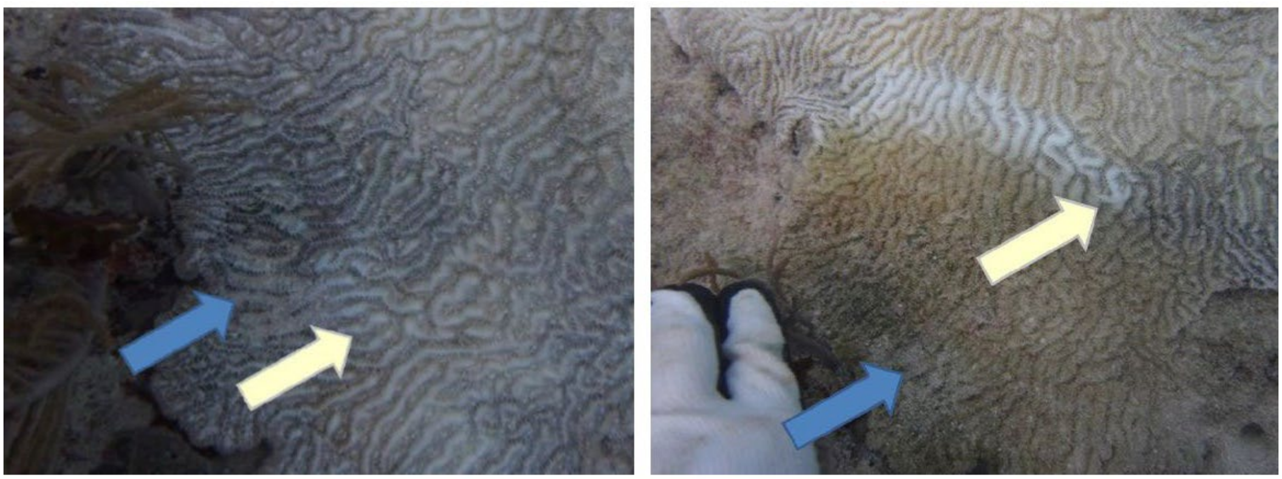
Third, a study performed by Onton et al. (2011) on reefs off the Western Australian coast showed that disease prevalence varied at both regional and local scales. Most striking in this analysis is that the results of Pollock et al. (2014) performed after the completion of a major dredge project (3.1 – 7.26%) are essentially invariant from the pre-dredging background values (1.1 – 7%) presented in Onton et al. (2011) on reefs from the same region.

Recently, using the results of a repeated measures monitoring program undertaken during the dredging operations, Stoddart et al. (2019) showed the frequency of coral disease through time. In their study, 60 corals were assessed repeatedly at each of four dredging 'impact' sites (< 1 km from dredging), and four 'reference' sites (> 20 km from dredging) over an 18-month period. Contrary to the results of Pollock et al. (2014), the frequency of occurrence of coral disease (usually <5% of corals) was not significantly impacted by dredging. Moreover, Stoddart et al. (2019)

note that thermal bleaching over the period 2010 to 2016 has had an order of magnitude greater impact on coral loss than any dredging-related impacts. Clearly, results from one-off surveys, especially when performed post hoc, do little to engender confidence in broad, sweeping conclusions regarding the role of dredging on enhancing coral disease prevalence (Pollock et al. 2014; see also Miller et al. 2016).

Fighting Unsubstantiated Claims and Pedestrian Science

DCA scientists first observed the onset of disease at one of the southern far-field control sites (R2SC2-LR) located off Virginia Key on September 26, 2014. On that day, four tagged corals had white-plague disease-like signs (originally reported as an unknown condition by our scientific divers). This was the first time we observed epidemic levels of disease (>5% prevalence) at one of our study sites (see Precht et al. 2016; DCA 2017). Because the disease started at the far-field project controls south of the PortMiami dredging operation, and the fact that we witnessed the spread of the disease away from this area (both south away from the dredging operation and north towards it), it was apparent that this disease outbreak was unrelated to the dredging project (Figure 6).



September 26, 2014 October 8, 2014

Figure 6. Close-up of initial observation of partial mortality (blue arrow) and first appearance of the white disease line (light yellow arrow) from partially bleached *Meandrina meandrites* coral located at R2SC2-LR-T2-C1 on September 26, 2014 and again with well-defined white-plague disease line (light yellow arrow) and continued partial mortality now covered in turf algae (blue arrow) on October 8, 2014 (from DCA 2017). Note that in photo on right that the bleached portion of the colony (upper portion of colony) has regained color (zooxanthellae) in response to seasonal cooling of water temperatures.

Even though the data clearly showed the disease initiated at the southern far-field control sites, numerous reports implicated the dredging project as the direct cause for the disease. For instance, Ward (2015) noted "recent dredging in Miami, USA has led to disease, smothering and death of the corals in the region." Once unsubstantiated claims are published, they are perpetuated by the media, NGOs, and project opponents making it impossible 'to put the genie back in the bottle.' Even state regulators made biased claims based on conjecture and innuendo – not science. For instance, a Regional Administrator with one of the regulatory agencies with purview over the project stated the following in an email entitled "White Plague Coral Disease Outbreak in Miami" dated July 1, 2015: "I am admittedly biased in my presumption that new(ish) sediment in the system is in part or wholly triggering this event. I would like to discuss what (if anything) we might be able to do to collect useful data that we can use to prove…my (biased) presumption."

Earlier, in May 2015, the National Marine Fisheries Service (NMFS) had rendered a similar opinion about cumulative impacts of the PortMiami Project before the dredging program had been completed, and before NMFS scientists performed in-water impact assessment surveys. Specifically, without any quantitative analysis, the agency stated in a formal letter to the USACE that 'damage' to the reef "greatly exceeds projections, perhaps more than ten times over," and "there is no indication the effects will be temporary" (Gomez 2015, Nelson 2015, Figure 7).


CBS 4 NEWS

Local

NOAA Says Port Miami Dredge Disaster For Reef

August 18, 2015 4:40 PM

MIAMI (CBSMiami) — In a project all but finished, the Army Corps of Engineers has dredged the channels at and leading to Port Miami for nearly two years, but another U.S. government agency says the Corps wreaked environmental havoc in the process.

Figure 7. Lead headline from local media outlet highlighting the claims of the NMFS before completion of project (see Nelson 2015).

Subsequent to this letter, a contract scientist working for the NMFS shadowed DCA divers during their monitoring at sites R2N1-RR and R2N2-LR on May 19, 2015. In a letter report dated June 17, 2015 (NMFS 2015), numerous claims were made based on one-off observations and photographs collected during the May 19th site visit. For example, the following photo was tendered as evidence that the dredging was responsible for the mortality of tagged corals at the DCA monitoring sites (Figure 8). Specifically, the figure caption in the NMFS report reads as follows "Figure 5a. Likely colonies of *Meandrina meandrites* smothered by fine sediment from the Port of Miami expansion dredging. Note that species identification after burial is complicated, however the morphology of the ridges and septa are distinct to *M. meandrites*." First, the coral in the photograph is actually a colony *of Pseudodiploria strigosa* and not *M. meandrites* (see Figure 9).

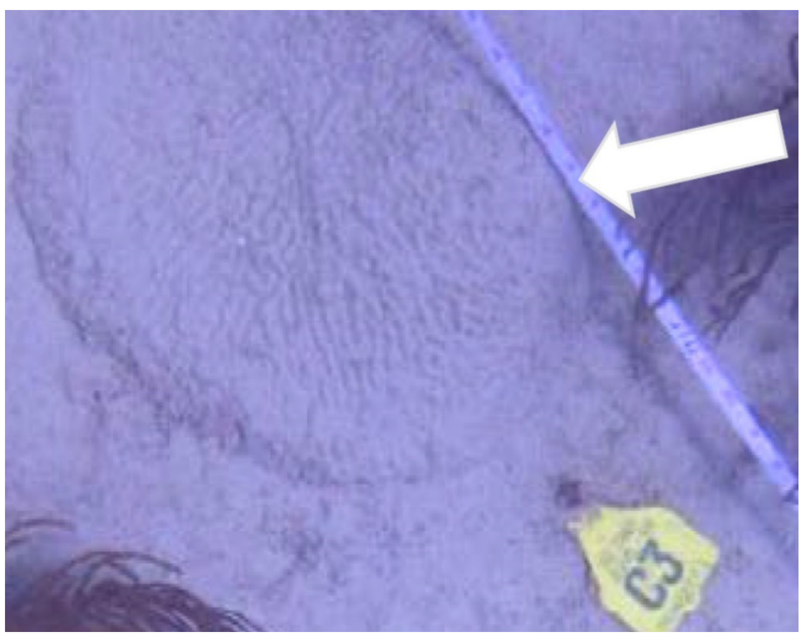


Figure 8. Image taken from Figure 5a in NMFS report (NMFS 2015). This tagged colony was coral number 3 on transect 1 at site R2N1-RR. Compare this photo with montage shown in our Figure 9 below.

While this correction may seem pedantic, getting the ID of the coral species correct in an official government document is important, especially when it is consequential to the question being addressed. We also question how one can make a determination as to the cause of mortality based on a one-off observation made long after the coral has died? The report is rife with similar errors and misstatements.

Fortunately, in this instance, because of the repeated measures monitoring program required under the PortMiami permit compliance monitoring contract, we could evaluate the history of that coral over the duration of the dredging project. Photographs were taken of this specific coral during 48 separate monitoring site visits between the fall of 2013 and the summer of 2015. The abridged montage below (Figure 9), shows the sequence of events that led to the eventual mortality of this coral. Note that mortality was caused by white-plague disease that initiated following a regional thermal stress (bleaching) event and not sediment burial as proffered in the NMFS report (see also Figures 4 & 5, and Precht et al. 2016 for additional examples). As previously mentioned, while some corals in close proximity to active dredge operations were periodically impacted by sediment leading to increased levels of stress (i.e. mucus production) and partial mortality, these corals showed no increase in susceptibility to coral bleaching or disease.

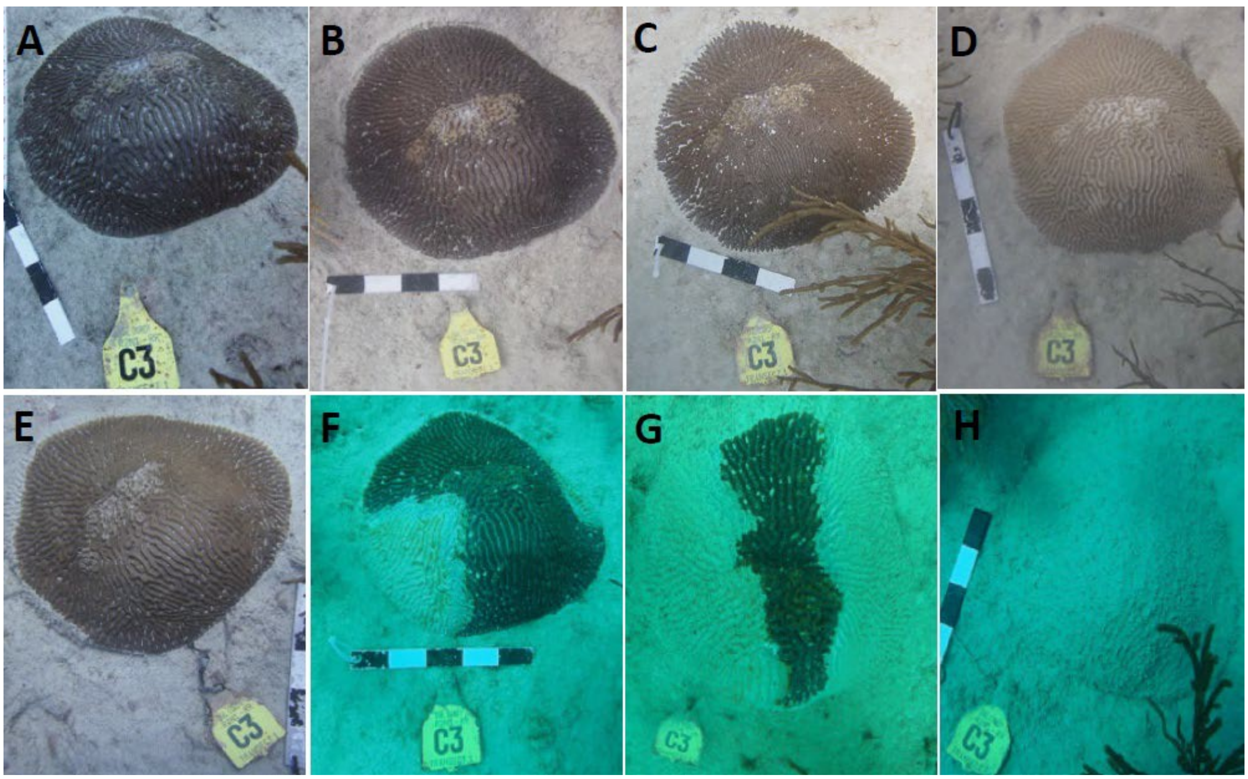


Figure 9. Temporal photographic montage of *Pseudodiploria strigosa* colony (R2N1-RR-T1-C3). This is the same channel-side coral as depicted in Figure 8 above. (A) Photograph of coral during initial baseline surveys on November 18, 2013 prior to initiation of dredge activities; (B) coral during compliance week 21 of monitoring, April 15, 2014; (C) coral during compliance week 40 on August 21, 2014 showing some sediment stress from dredging activities with a rim of partial mortality around the very base of colony; (D) coral showing signs of bleaching (paling) associated with warm-water thermal anomaly during compliance week 46 on October 2, 2014; (E) coral showing residual halo of partial mortality during compliance week 52 on November 13, 2014, note that the coral has regained its color (zooxanthellae) in response to seasonal cooling of water temperatures; (F) coral showing progressive signs of active white-plague disease during compliance week 66 on February 25, 2015; (G) colony showing >60% mortality associated with progression of white-plague disease during compliance week 69, March 18, 2015; and (H) total colony mortality, note the coral is covered by fine-grained sediment, photo taken during post-construction surveys performed on June 18, 2015.

In December 2015, staff from the NMFS performed quantitative field studies at PortMiami and their associated far-field control sites. Like Pollock et al. (2014), these one-off surveys were completed long after the completion of dredging operations. Their report (NMFS 2016), and subsequent publication (Miller et al. 2016), which has numerous errors, directly contradicts the project reports produced by DCA (DCA 2015a, 2015b).

The first error of note includes a pseudo-replicated design of their transects. The term pseudo-replication was coined by Hurlbert (1984) to refer to the use of inferential statistics to test for treatment effects with data from experiments where either the treatments are not replicated, or the replicates are not statistically independent. Replication refers to having more than one experimental (or observational) unit with the same treatment.

Each unit with the same treatment is called a replicate. Most models for statistical inference require true replication. For the 'transect' design used in this monitoring protocol, pseudo-replication is more about how we can report and interpret/extrapolate the sample size. For instance, each site has two 50 m long transects that start at a fixed point; one with a north-south orientation and the other with an east-west orientation (Miller et al. 2016). At the level of the 'site' these two transects are not true replicates but are pseudo-replicates of each other because they start (touch) at a common point. Even if these two 50 m long transects are then subdivided into smaller blocks/samples these units must be treated as subsamples. For instance, if we analyze these data into 10 m long individual segments or blocks, we do not have 10 replicate samples (n=10) but a sample size of one (n=1). The consequences of doing statistical inference using pseudo-replicates includes non-existent site variability or site variability that is significantly underestimated. This will result in confidence intervals that are too small and an inflated probability of committing Type I statistical errors (falsely concluding there is an effect when in fact there is none).

Second, the data was collected at 1 m increments along each pseudo-replicated transect accounting for 100 data points for each of their sites. Miller et al. (2016) refers to these as line-intercept transects (LIT) when in reality they are linear-point intercept (LPI) transects (Hill and Wilkinson 1994, Rogers et al. 1994). However, the data furnished by NMFS to the USACE from their LPI surveys collected in their December 2015 surveys include 10 of 11 sites where there are more than 100 points per transect. At one site (R2N-200-LR) there are 148 observations (data points) on a 100 m long L-shaped transect. At another site (R2N-300-LR), their count data reveal 40 out of 121 points of sediment over hardbottom (33%). However, the data in their supplemental material furnished with the paper (http://dx.doi.org/10.7717/peerj.2711#supplemental-information) shows 45% of that site designated as sediment over hardbottom (Supplemental Table 1 in Miller et al. 2016). Even more problematic are their coral counts which reveal live coral cover at these sites ranging between 12 - 38%. This is an area where coral cover is routinely measured in fractions of a percent (<1%; Gilliam et al. 2018). We are perplexed by these data and their subsequent analysis. Also, the failure to use standard LPI protocol (Ohlhorst et al. 1988, Aronson and Precht 1995) and why there were more than 100 points counted at each site are not explained in either their methods or results.

Third, their site closet to dredge-plume exposure (R2N-100-RR) showed that 10% of corals had some evidence of partial mortality associated with sediment burial (halos) while 52% of the site was characterized as having 'sediment over hardbottom.' However, their site furthest from the dredging operation (R2N-700-LR) showed 11% of corals with evidence of sediment halos while 53% of the site was characterized as having sediment over hardbottom (Supplemental Table 1 in Miller et al. 2016). Mathematical model projections of turbidity plumes from cutter head dredges (Henriksen 2009) reveals that most sediment settles within the first 150 m and follows an exponential decay away from the active dredge (see also Griffin 1974). Miller et al. (2014) provided no explanation on the mechanisms that would allow the site within 100 m of the dredge to have the same level of partial mortality and sediment cover as a site 700 m away.

Figure 10. NMFS photo showing partial burial of the octocoral *Eunicea* sp. by some 14 cm of reef sediment. Note the white plates of *Halimeda* admixed (floating) throughout the coarse-grained reef sediment.

The most problematic issue with the NMFS (2016) report and subsequent publication (Miller et al. 2016), like with the NMFS (2015) report discussed earlier, was inferring causality using one-off site observations, measurements or photographs. For instance, authors of the NMFS reports have repeatedly used the following photograph (Figure 10 above) taken in April 2016, almost a year after cessation of dredging activities as clear and direct evidence of significant deposition of project related sediments resulting in partial burial of octocorals throughout the project area (Karazsia 2016). However, close examination of the sediments in the photograph show typical, locally sourced reef derived sediments, with abundant coarse, unbroken plates (segments) of the calcareous green alga *Halimeda incrassata* and not fine-grained, silty sediments that would typically be associated with cutter-head dredge operations.

A montage of photographs (see Figure 11 below) taken throughout the Caribbean and southeast Florida shows similar occurrences of octocorals being buried by reef-derived sand. This is a commonly observed phenomenon on reefs the world over. Given that the area in the NMFS photograph was not repeatedly surveyed before, during, and after dredging operations, the ultimate cause leading to the burial of these organisms cannot be determined with any certainty.

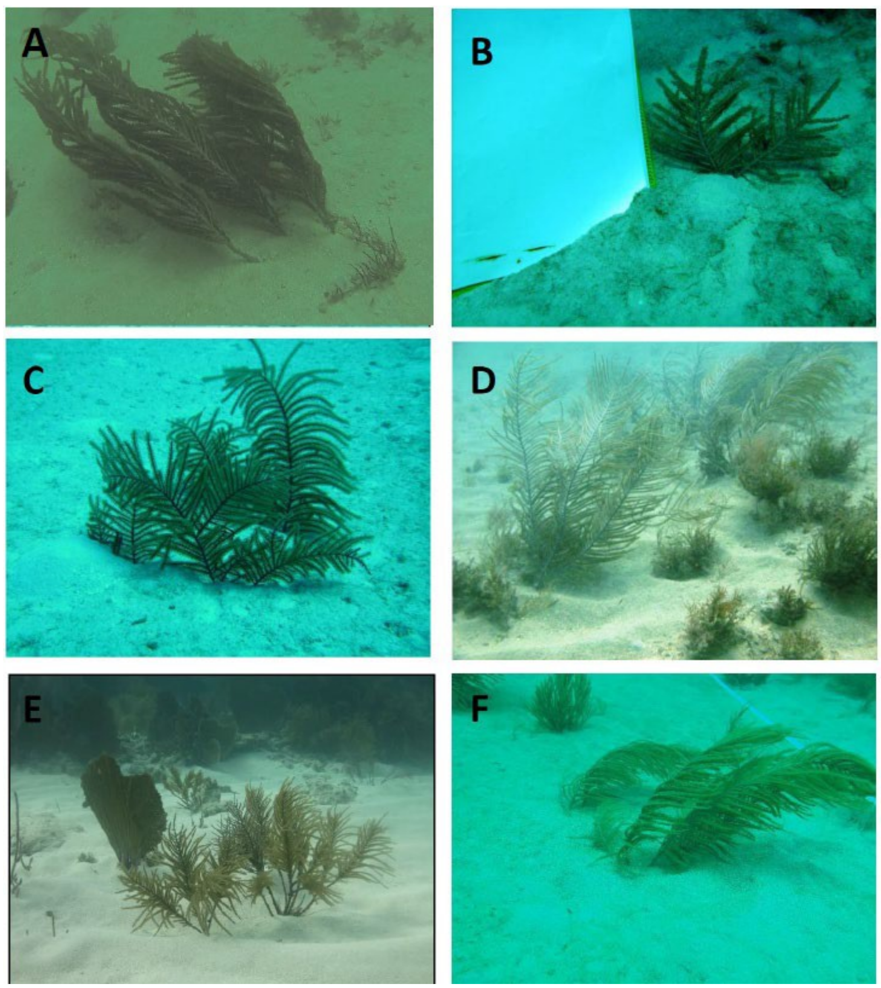


Figure 11. (A) Photograph of octocoral being buried by sand. Photo taken during the establishment of PortMiami hardbottom project sites in September 2013, some three months before initiation of dredging operations. (B) Partial burial of octocoral by fine-grained sediments in low energy patch reef setting in Biscayne National Park. Photo taken in August 2014 by W.Precht. (C) Partial burial of an octocoral by fine-grained sediments in groove between two reef-spurs. Photo taken in Grand Cayman, B.W.I. by W. Precht in February 2016. (D) Harbottom in the process of being engulfed by migrating sands. Note migrating sand waves burying both octocoral and algal resources. Photo taken on Inner Reef adjacent to and just south of the active shipping channel at PortMiami in May 2011 by Dr. Phil Frank of Terramar Environmental Services, Inc. (E) Photo of migrating sands in the process of burying live octocorals at Crocker Reef in the Florida Keys National Marine Sanctuary. Photo taken by Dr. Ilsa Kuffner of the USGS in 2013. (F) Photo of octocorals buried by migrating sediment in an area on the Inner Reef north of Port Everglades in Broward County, FL. Photo by W. Precht in July 2017.

For instance, the figure below is taken directly from NMFS (2016). Their caption reads "Sedimentation impacts observed on transect R2N-200-LR. *Psuedodiploria strigosa* shows partial mortality with a sediment halo." While the coral clearly shows a mortality halo, this was not caused by sediment burial but by the progressive mortality associated with white-plague disease which is still clearly active and visible in the photograph. The cause for the mortality crown on the top of the coral in this photo is unknown. The failure of NMFS scientists to identify the difference between stressors in the field, in this case the difference between sediment stress and coral disease, is extremely disquieting.

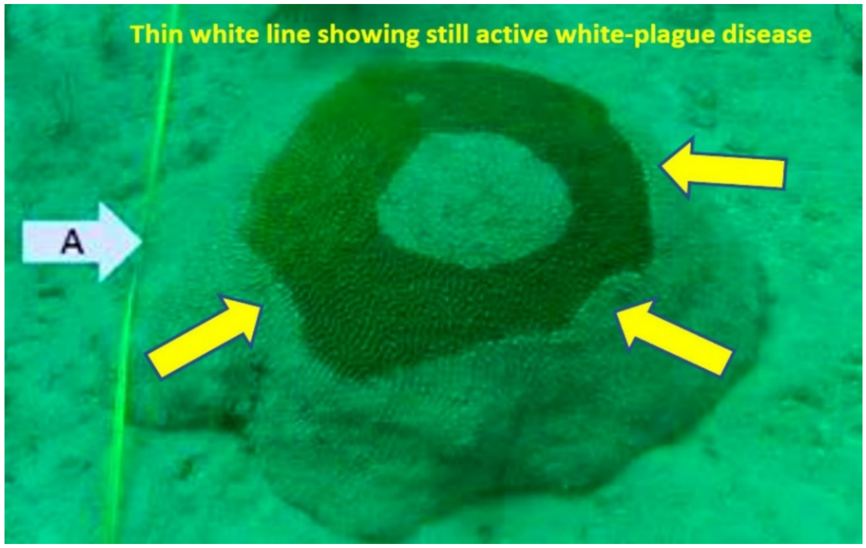


Figure 12. Colony of *Pseudodiploria strigosa* showing halo of coral mortality taken from Figure 8 in NMFS (2016). Yellow arrows point to active white-plague disease boundary progressively moving across the surface of the colony.

In addition to the field data collected by NMFS staff in December 2015, they performed multiple analysis on the data and photographs collected by DCA over the course of the PortMiami project (DCA, 2015a, 2015b). The result of both these efforts are used to detail the purported negative impacts associated with the PortMiami deepening project (Miller et al. 2016). Due to the fact that Miller et al. (2016) performed this work on the mast of the NMFS, the U.S. government agency responsible for stewardship of the oceans, their report (NMFS 2016) and subsequent publication (Miller et al. 2016) was extolled by the local media, a number of NGOs, and vocal project opponents as prima facie evidence for a link between dredging stress and coral mortality. Exploiting the environmental concerns of the local citizenry about the PortMiami deep dredge project, local NGOs and the media used the Miller et al. (2016) manuscript as proof that the PortMiami dredging project had far greater impacts than anticipated or reported by the USACE (see Alverez 2016, Caole Vila 2016, Staletovich 2016). This reliance, based on who did the report and not the actual science performed in the report, is known as the 'fallacy of authoritarian bias' or appeal to authority (argumentum ad verecundiam). This argument also pitted two federal agencies against each other. One, the NMFS, is viewed as the agency responsible for protecting the environment, while the other, the USACE, is commonly perceived as the agency responsible for harming the environment. This is a false dichotomy. Executive Order No.13089 signed by President Bill Clinton on June 11, 1998, stated "[a]ll Federal agencies whose actions may affect U.S. coral reef ecosystems must affirmatively act to become aware of and to protect the nation's coral reefs." This Executive Order thus places federal agencies under an "affirmative obligation to not only avoid harming coral reefs within U.S. waters but also to protect and enhance those reefs, subject to only limited exceptions." This order pertains to all federal agencies irrespective of whether they are with the Department of Commerce (NMFS) or the Department of Defense (USACE; see Craig 2000).

Moreover, by gaining control of the local media through release of internal NMFS reports and letters, project opponents followed a time-tested recipe for how to spin the NMFS results in their favor, inhibiting important lessons learned that ultimately impeded scientific progress all while casting a negative cloud over the project, the USACE, and its contractors (Oreskes and Conway 2010).

In addition to the problems with their collected field data described above, the following discussion carefully addresses numerous additional flaws in the Miller et al. (2016) manuscript based on their analysis of data collected by DCA as part of the FDEP permit compliance monitoring program (DCA 2015a, 2015b). At PortMiami, Miller et al. (2016) found increased coral mortality at two (2) dredge-adjacent sites when compared to their paired controls and attributed these differences to the added stress of local dredging. While the raw data collected by DCA and presented in Miller et al. (2016) are accurate, the limited scope of their analysis that only presented data from four (4) out of the 26 sites available from an open-access dataset is nothing short of scientific malpractice (see The Royal Society 2012, Boulton 2013). In addition, the channel-side sites they selected had the highest rates of disease-related coral mortality while their paired controls had some of the lowest (DCA 2017). They also note that six corals at the two channel-side sites (11.5%) died as a direct result of sediment burial. However, a total of only seven corals died at all 15 channel-side sides (2.0%). This material omission of data in Miller et al. (2016), fundamentally impacted their results and biased their interpretation.

Miller et al. (2016) also failed to address significant differences in the species-susceptibility of the coral communities within and between the sites they evaluated. This is a fatal flaw in their analysis and compromises their primary conclusions. Specifically, the two (2) dredge-adjacent sites analyzed by Miller et al. (2016) were composed of corals that were significantly more susceptible to white-plague disease than those at their paired controls (DCA 2017). For example, at one of their dredge-adjacent sites, 57.7% of the tagged coral community (15 out of 24 corals) were susceptible to white-plague disease; at the paired control only 25.0% (7 out of 28 corals) were disease- susceptible (Precht et al. 2016). Miller et al. (2016) stated that there was an “increased risk of disease and death (>double) in the immediate vicinity of the dredged channel, in comparison with project-chosen reference reef.” Remarkably, this comment directly corroborates the percentages of disease susceptibility between channel-side and control sites used in Miller et al. (2016). Thus, the failure of Miller et al. (2016) to account for the species-specific susceptibility of corals to mortality from coral disease has created a circumstance in which published rates of coral mortality attributed to dredging are artificially inflated.

None of the near-dredge sites had levels of coral mortality that were higher than predicted from regional mortality data; a result that shows no exacerbation of disease-related mortality at near-dredge sites during the regional 2014-2017 white-plague disease outbreak. This included corals that had expressed partial mortality from dredging prior to disease onset. Most importantly, taking into account disease-susceptibility of surveyed coral communities not only explained why certain sites had high disease-related mortality but also explained why certain sites located directly-adjacent to long-term dredging had little or no disease-related mortality over a two-year period when compared to higher rates of disease at paired, far-field controls. As previously mentioned, the combined losses from sediment burial, bleaching, competitive mortality, and white-band disease accounted for the mortality of only 4.1% of all monitored corals, yet a further 194 corals, 31.7% of all tagged corals died as a direct result of white-plague disease. When you add the 11 coral colonies that were still actively diseased during our last site visit in 2016 increases the total of white-plague disease impacted corals to 205, or 33.5%. This loss due to white-plague disease is essentially the same as the region-wide losses (35%) described for the SECREMP monitoring program during the same time period (D. Gilliam personal communication, Gilliam et al. 2018).

Miller’s conclusion that proximity to dredging exacerbated coral disease-mortality, is poorly validated when applied to the entire PortMiami dataset. Disease-related mortality was higher than controls at seven (7) sites, lower at seven (7) sites and equal in one (1) paired example, suggesting that dredging could only have exacerbated disease mortality at 46.7% of the sites. These results are invariant with the results of a coin toss experiment (Underwood 1997). In contrast, when coral community composition is accounted for, and regional species-susceptibility to white-plague disease is modeled as the primary driver of white-plague disease mortality, levels of mortality were within predicted range at 14 out of 15 dredge-adjacent sites, and at 25 out of 26 project sites including far-field controls; an accuracy rate of 93% and 96% in both cases

Miller et al. (2016) also noted that the pattern of increased disease-related mortality at channel-side sides is consistent with the highest sediment plume exposure area taken from satellite images (Barnes et al. 2015). The project data, however, does not support this conclusion (see preceding paragraph). Additionally, the notion that turbidity measured from satellite images is a proxy for sedimentation is poorly validated in the peer-reviewed literature. Turbidity is a simple measurement of light scattering and used for water quality purposes and therefore should not be used as a surrogate for sedimentation impacts. Miller et al. (2016) incorrectly calls these turbidity plumes – sediment plumes. While nephelometric turbidity observations can be successfully used in quantitative sediment studies, these data can only be obtained if the relevant turbidity vs. in situ suspended sediment concentration data is collected and a regression equation is obtained (Ouillon et al. 2004). Thus, ground-truthing satellite images using an algorithm approach (Gramer and Hendee 2018) is required to obtain a consistent quantitative estimate of sediment concentration and accordingly, the potential impact a dredge plume may have on benthic organisms (Evans et al. 2012). These data were not collected during the PortMiami monitoring program.

Miller et al. (2016) also used the tagged corals from their limited set of our time-series photographs (four sites, two channel-side and two controls), for analysis of live tissue loss over time. Miller et al. (2016) stated "the live tissue area was quantified from the best-matched photo (angle and orientation) of each colony from the pre-construction and from the post-construction phase... " However, these photographic data collected by DCA under the mandate of the FDEP permit conditions were never intended for this type of post hoc analysis. Accurate comparison of images of the same coral taken at different times is highly dependent on the precise positioning of the camera (Rogers et al. 1994). Specifically, the placement of the camera being held by a free-swimming scientific diver from visit-to-visit results in a shift in the location of a colony within the image frame, which can cause apparent growth or shrinkage where none has in fact occurred (see Figures 4, 5 & 9 for examples of multiple photos of the same colony taken over time; note the slightly different perspective in each photograph).

Clearly, when photographing coral species for long-term growth assessment measurements (tissue loss or gain), a framer should be used to ensure the camera focal plane is parallel to the coral being measured, the camera-to-subject distance is constant (fixed), and the same camera and lens is consistently used. Permanent site markers (e.g. pins fixed into drilled holes) surrounding the coral colony of interest also helps to ensure precise positioning of the framer each time the colony is subsequently photographed (Hill and Wilkinson 1994). None of these methods were used in photographing the 643 tagged coral colonies associated with this project. Another significant issue with these photographs is that wide-angle lenses suffer from peripheral distortion and very steep perspective (i.e. subjects closer to the camera appear much larger than those more distant). Steepened perspective and optical distortion cause the object to appear 'warped' making accurate growth measurements of three-dimensional colonies technically difficult to achieve, even when the camera is precisely repositioned every time with a framer and fixed focal length. For this reason alone, calibrating and extracting size measurements for three-dimensional colonies can be challenging even when the monitoring program is designed and implemented correctly. We also tried to do a similar analysis (DCA 2017) with the same limitations. In the case of the post hoc data presented in Miller et al. (2016), however, the inherent statistical error in the collection of these data are greater than the actual measured differences rendering their conclusions regarding tissue loss over time meaningless.

Because this 'appeal to authority' strongly influenced the dialogue regarding purported impacts of the PortMiami project, the harmful and widespread influence of climate change on the initiation and overall impact of bleaching and subsequent white-plague disease outbreak was not addressed or acknowledged as a contributing factor to observed coral morality. The consequence of this logical fallacy is an inaccurate scientific and public perception of the actual effects of dredging on local coral assemblages (e.g. Staletovitch 2016) and a corresponding lack of understanding of the deadly impacts of climate change on coral reef resources throughout southeastern Florida. This discounting of climate-related impacts plays directly into the rhetoric and false claims of climate deniers by undercutting compelling scientific evidence showing the direct linkage between increased temperature, coral bleaching and coral disease (see Bradley 2011). This also results in an inaccurate scientific and public perception of the effects of dredging on local coral assemblages (Staletovitch 2016). Such an inaccurate perception, in turn, influences environmental regulation and policy (van Tuijn 2015, Stein 2017), slows scientific discovery, and forms the foundation of nuisance law suits (CBD 2016) that include attempts to derail port infrastructure projects, including in areas absent of corals and coral reefs (IDR 2017).

CONCLUSIONS

Management Implications

The quantification of the impacts responsible for causing coral morbidity and mortality remains a major hurdle for managing dredging projects adjacent to coral reefs. Here, we demonstrate an application of a commonly used monitoring method to accurately quantify both the species and community-level impacts associated with the PortMiami dredging project. Using repeated measures data from 26 sites off Miami-Dade County, Florida allowed us to differentiate causes of coral stress and mortality as well as local (project) versus regional (natural) impacts. In testing for effects of sediment on coral, octocoral, and sponge abundance (cover), we found no statistically significant evidence for decline at both the site and regional level. However, coral morbidity (partial mortality) associated with sedimentation was highest at sites immediately adjacent to the dredge operations. These results suggest that management actions that reduce sediment impacts should ultimately reduce coral stress and concomitant non-lethal impacts to corals and other benthic organisms across sites benefitting the ecosystem. However, in dissecting the components of observed coral mortality, we found that contact with sediment or proximity to the dredge did not cause the wide-spread disintegration of the coral community. In addition, there was no relationship between sediment contact (partial burial) and proximity to dredge operations with coral disease prevalence or disease related coral mortality. This is contrary to the commonly held notion that sediment stress from dredging is linked with coral disease.

Lessons Learned

The detailed scientific analysis of the data collected from the PortMiami Phase III Deepening Project has provided a basis from which a detailed 'lessons learned' has been established. It is hoped that these lessons provide an effective framework for the future environmental management of construction projects where corals and coral reefs may be potentially harmed. This is especially relevant, with future projects such as the deepening of Port Everglades and the expansion of a cruise ship pier in Grand Cayman looming on the horizon. The following bullet points highlight some of the major lessons we identified from the daily compliance monitoring of the PortMiami project:

- There should be a panel of experts convened that provide consensus input to the project (Delphi Technique), especially during pre-project planning (Okoli and Pawlowski 2004). These experts should not be active stakeholders in the project.
- Collect better (more) background ecological data in advance of construction activities including the use of 2D and 3D coral mosaics to better define benthic resources in the project area (Lirman et al. 2007, Precht et al. 2016).
- Repeat the 2D and 3D photo mosaics both during and after construction at key locations to evaluate changes to the benthic resources in time and space.
- Develop accurate hydrodynamic models showing predicted dispersal of dredge plumes prior to construction (Savoli et al. 2013).
- Evaluate, a priori, the construction equipment, methods, and best management practices (BMPs) that should be employed to minimize environmental impacts (Huston and Huston 1976, MICCI 2008, Foster et al. 2010, USCRTF 2016).
- Increase area of surveillance during the compliance monitoring program to assure no impacts are missed or go unreported (see DCA 2010).
- Use a census monitoring technique of the reef habitats that focuses on a regression-based approach moving away from the active dredge channel (DCA 2010). Post surveys following the dredging operation will allow comparison with the pre-dredging data. Impacts of the dredging operation, should they occur, will be detectable as a significant relationship between distance from the channel and the magnitude of change between the pre- and post-impact habitat states.

- Implement a BACI design repeated measures tagged coral monitoring program but require the weekly monitoring of all sites irrespective of the location or position of the dredge. This will assure that natural events such as coral bleaching, disease outbreaks, or impacts from storms are not missed or underestimated. Impacts would be detectable as significant interaction terms of ANOVA between time (before versus after dredging) and treatment (indirect impact versus control/reference).
- Perform detailed in situ coral recruit (< 4cm) surveys during baseline and post-construction periods.
- The newest technology needs to be used to assist and supplement the in-water scientific diving operations. These include but are not limited to the use of fixed optical/acoustic turbidity/sediment sensors spread throughout the project site, the use of real-time underwater video feeds at key locations, and the collection and interpretation of daily drone video footage for turbidity plume monitoring. Drone captured images are a powerful tool in the management and communication of the environmental aspects associated with dredging programs (Ramírez-Macías et al. 2018).
- There should be a liaison between the contractors and regulators to assure for the accounting and dissemination of all collected monitoring data throughout all phases of the project. These data should be placed on a project specific website available to all project stakeholders.
- There should be an independent 'third-party' contractor that provides QA/QC oversite of the monitoring programs implemented during the project to eliminate the perception of bias on either side.
- Applying an adaptive decision process to the monitoring program is likely to enhance management outcomes (Williams and Brown 2014). However, adaptive management is a tool that must be used efficiently (in real time) to effectively work. This entails developing feedback loops in the monitoring program that triggers a response at the first sign of a problem. Therefore, changes in construction methodologies as a response to adaptive management recommendations must be performed in a timely and efficient manner (within hours to days – not weeks to months).
- There is a need for all regulatory agencies to fully appreciate the extent and nature of the actual site conditions rather than relying on assumptions either presumed or inferred by project opponents. This will require that the agencies be actively engaged throughout the project and not performing field inspections until well after purported impacts have occurred or worse yet long after the project has been completed.
- Communication and interaction between all stakeholders and relevant government agencies needs to be open and transparent throughout all project phases. This open and transparent communication will be necessary to avoid and resolve conflicts between the various groups, many with disparate goals and interests.
- Establish cooperative and collaborative environmental education programs and public workshops to disseminate project information while promoting the protection and conservation of environmental resources.
- Publish the results of the project by incorporating both its successes and failures. We will learn more from our failures, because failure reveals the inadequacies in our project design (Precht and Robbart 2006). This will set a path for improved protection of coral reef resources on future projects.

Remember that all the lessons learned listed above take resources, time, and money which are all borne by the project sponsors and ultimately each one of us as taxpayers. The same goes for nuisance lawsuits filed by project opponents. Thus, it is imperative that accurate, hypothesis-driven, science-based monitoring programs need to inform all management decisions. This is critical to overall project success.

Summary

The presentation of these results is not intended to say that dredging does not or will not affect nearby coral resources, some impacts are inevitable (Foster et al. 2010, Erftemeijer et al. 2012) and clearly there were impacts associated with the PortMiami dredging program (see DCA 2015a; 2015b; 2015c; 2015d; 2017). These results aim to point out that the strength of the regional disease event was so devastating that the impact of the dredging project did not significantly impact rates of whole colony mortality – locally or regionally. This is contrary to the purported impacts in Miller et al. (2016) and those impacts espoused in numerous op-eds and media publications

(Silverstein 2015, Alverez 2016, Staletovich 2016). In other words, if corals had not been followed through time to document the impacts of the dredging due to sediment, it would be impossible to identify those factors from post hoc metrics of coral mortality alone.

Using limited one-off surveys to differentiate between natural and anthropogenic causes (in this case sediment impacts from dredging) is not well supported. Especially surveys performed long after the active dredge operations have ceased. Using a BACI approach to impact assessment allows for hypotheses to be tested and either validated or falsified. Thus, regular monitoring of tagged corals at far-field control and near project sites provided the necessary detailed information needed to assign the correct cause of mortality to corals in the project area as opposed to the undocumented assertions of project opponents and regulators who conducted one-off surveys or worse yet, used selectively biased, cherry-picked data collected by others. Specifically, using a repeated measures approach to monitoring allowed us to isolate the effects of dredging activities from natural variation caused by other events such as coral bleaching and disease. Such an approach to environmental monitoring reduces the uncertainty that underlies the documentation of effects of anthropogenic impacts. Future port expansion projects proposed throughout the tropics demand that such programs are employed, and the results integrated into an on-going, ever-evolving adaptive management approach towards resource protection.

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