2018 Port Everglades Function Assessment assumptions

The purpose of this analysis is to provide the justification and documentation for the input parameters for a Habitat Equivalency Analysis (HEA) or Uniform Mitigation Assessment Method (UMAM) for compensatory mitigation for the unavoidable direct and indirect impacts associated with the Port Everglades Navigation Improvements Project. (PENIP)

Impact distances (meters [m] from Federal Navigation Channel)

The U.S. Army Corps of Engineers, Jacksonville District (Corps) has identified zones of impact based upon distance from Federal Navigation Channel at Port Everglades in order to rationally consider indirect effects associated with the PENIP. The justification for delineation of these zones is outlined below and is largely based upon post- construction monitoring surveys associated with the 2015 Miami Harbor Deepening Project, previous dredging at Port Everglades and peer-reviewed published literature where available.

0-150m

Miami Harbor Deepening Project Post-Construction Monitoring Survey: Post-construction monitoring documented that 67-93% of monitored corals within 30 meters (m) of the channel edge ("channelside") located on the north side of the middle reef had partial mortality associated with sedimentation (DCA 2015c). In addition, based upon data from previous hopper dredging operations in the Baltic, Gajewski and Uscinowicz (1993) noted that the main deposition of sand from hopper dredge overflow was confined to distances within 150 m on each side of the dredge.

151-450m

Miami Harbor Deepening Project Post-Construction Survey: Other than the results from the northern middle reef (AKA Reef 2), all of the other hardbottom and reef areas at Miami Harbor that are perpendicular to the channel had smaller sediment impact areas (R3N – 250m; R3S – 50m; R2S – 400m; HBN – 400m; HBS – 400m) based on the impact delineation cueing on the visual presence and of depth of fine, grey, clay-like sediments (Figures 1 and 2).

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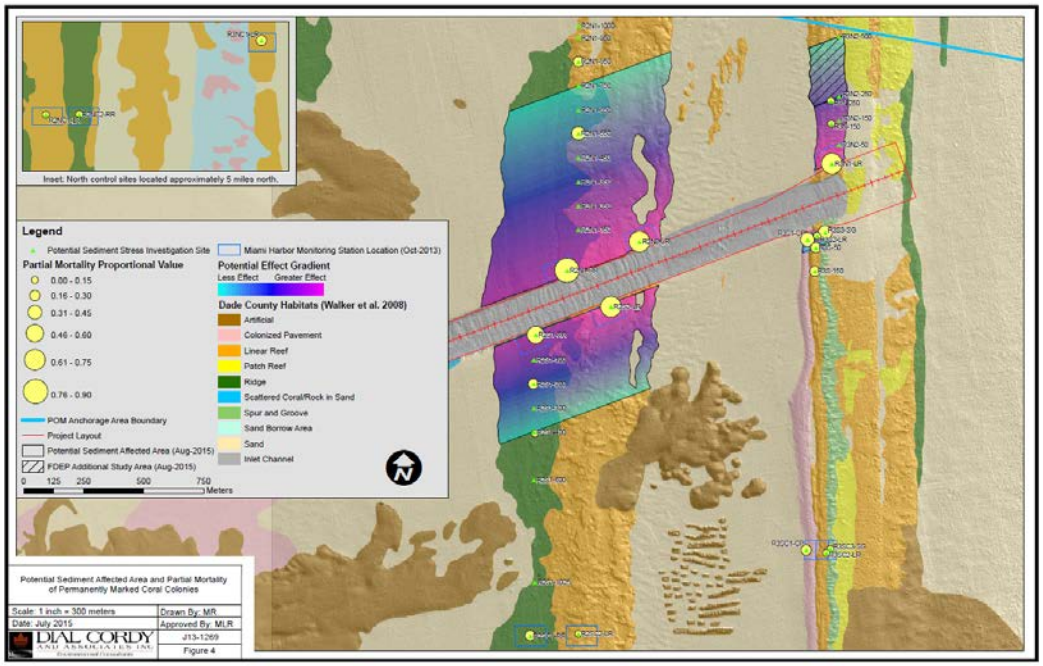

Figure 1 - Sedimentation Delineation Middle and Outer Reefs - Miami Harbor (DCA 2015b)

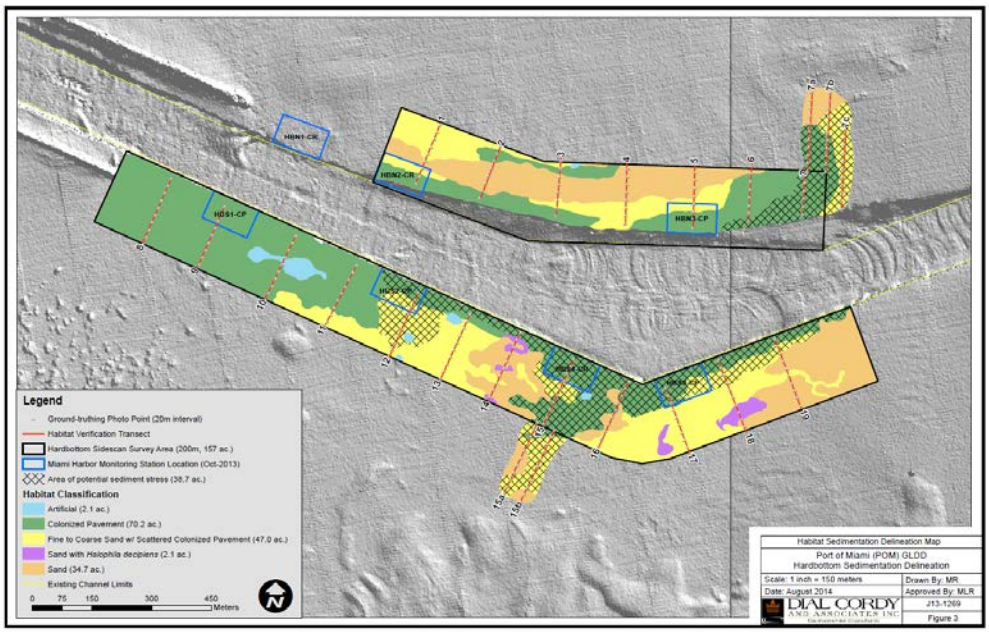

Figure 2- Sedimentation Delineate Nearshore Hardbottom - Miami Harbor (DCA 2015a)

<u>451-750m</u>

Miami Harbor Deepening Project Post-Construction Survey: The maximum distance where fine sediment was documented north of the entrance channel on the northern middle reef (AKA Reef 2) was 650m in the Miami Harbor sedimentation impact delimitation report (DCA 2015b):

> *R2N1-750 was designated as un-impacted because partial mortality of corals was not observed during qualitative surveys, and no pockets of clay-like material were found at the site. In*

addition, several healthy colonies of Acropora cervicornis were found at R2N1-750. R2N1-650 was designated as potentially impacted because partial mortality on corals in low lying areas was observed and pockets of clay-like material were present at the site.

Additionally, Miller et al. (2016) states:

Results indicate increased sediment accumulation, severe in certain times and places, and an associated biological response (e.g., higher prevalence of partial mortality of corals) extended up to 700 m from the channel…

Use of the 750m value is consistent with both of these findings and serves as a good basis for a maximum impact assessment across all of the reef lines perpendicular to the Port Everglades channel.

A study performed by Newell and Siederer (2003) in the United Kingdom (under high current velocities) showed that, in most cases, coarse material up to sand-size particles settles within 200 to 600 m of the point source of discharge, depending on depth of water, tidal velocity, and the velocity of flow from the discharge pipe.

During the 1980-1981 Port Everglades expansion, monitoring of corals north and south of the channel showed no adverse effect of the project associated with excavation with a cutterhead dredge (Figure 3) with upland disposal. Although the dredged material was pumped upland, there is still turbidity and sedimentation associated with the operation of the cutterhead itself. Baseline monitoring was conducted 19 days prior to dredging (April 1980), and post-construction monitoring 109 days (April 1981) after dredging was complete in December 1980. The closest monitored coral north of the channel was at Station 10 – approximately 690m north of the channel (Figure 4). The closest monitored coral south of the channel was at station 3, 100m from the channel edge. Station 15 was approximately 345m south of the existing channel, and station 7 was on the 3rd reef, beyond the eastern limit of the channel, approximately 300m south of the proposed channel extension.

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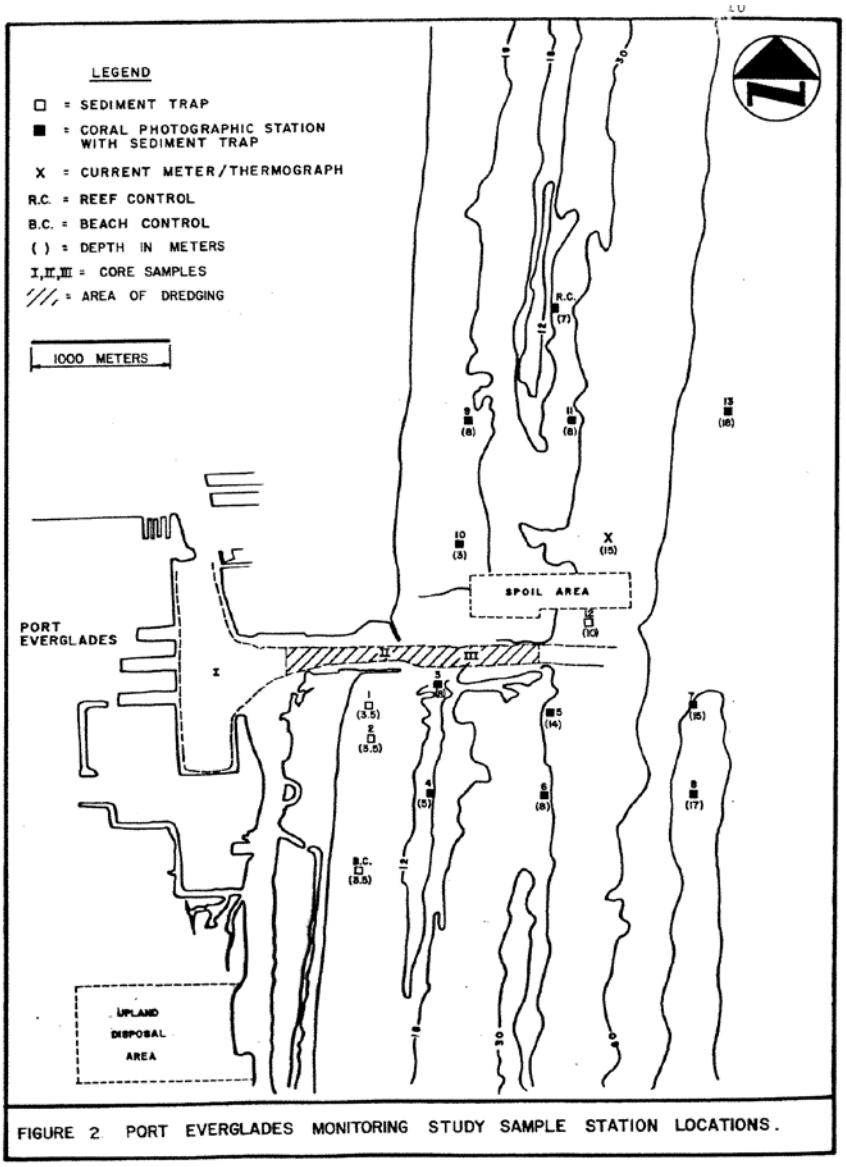

Figure 3 - 1980 Monitoring Locations (CSA 1981a)

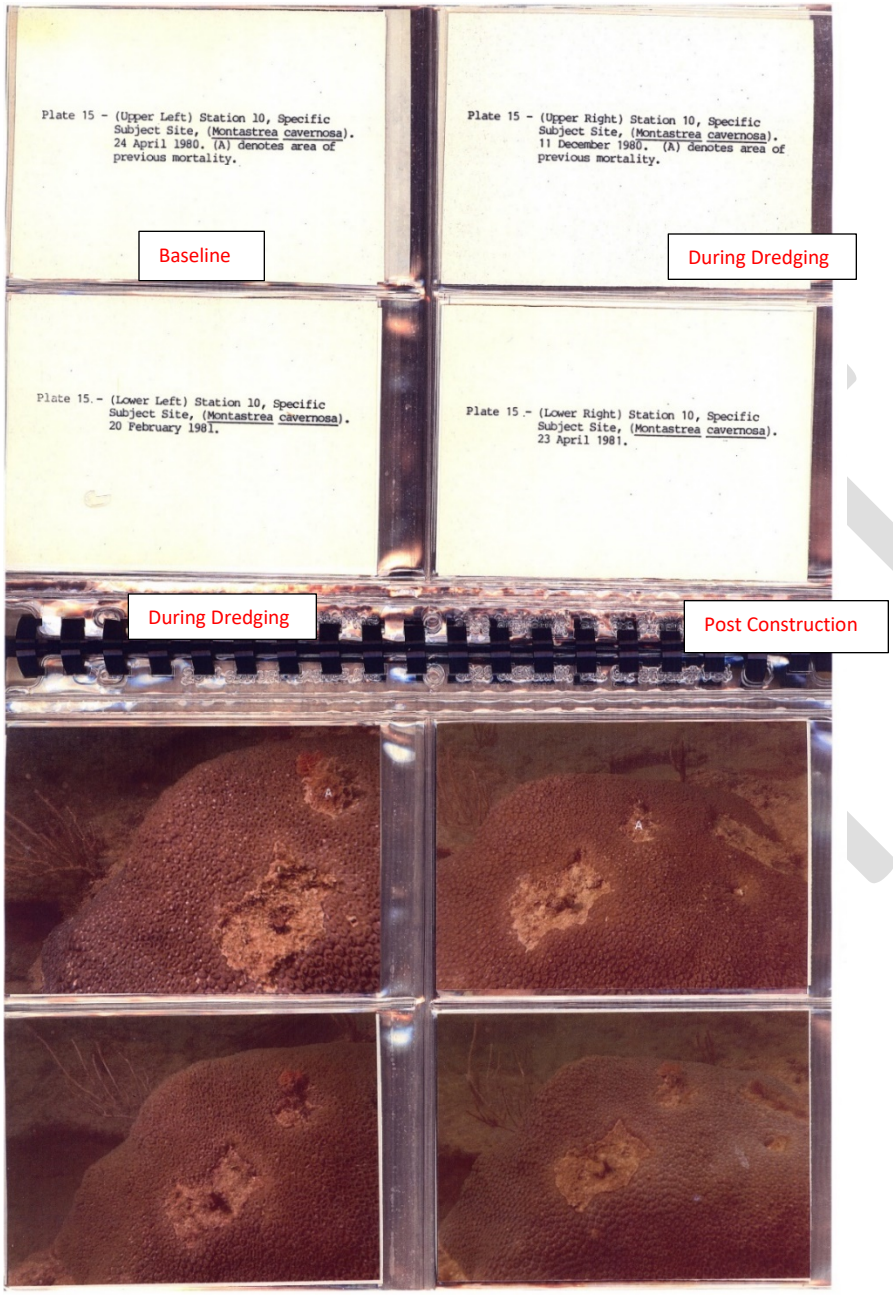

Figure 4- Monitored Coral at Station 10 - 690m north of channel during 1980-81 expansion (CSA 1981b)

751-1200m

In previous Corps dredge projects within southeastern Florida, no impact was documented beyond 750 m. In addition, a search for peer-reviewed literature associated with dredge projects and potential effects on corals and hardbottom organisms resulted in no additional literature that could be used to assist to delineate potential impact areas.

Based on all of this information, the Corps believes that the data supports the establishment of four impact zones associated with sedimentation impacts of 1-150m; 151-450m; 451-750m and 751-1,200m associated with the PENIP.

Impact Timeframes

The Corps has identified impact timeframes in order to allocate indirect effects associated with the Port Everglades Navigation Improvements Project. The justification for delineation of these timeframes is outlined below and is largely based upon post- construction monitoring surveys associated with the 2015 Miami Harbor Deepening Projects and peer-reviewed published literature where available.

Functional Group Coverage as an Indicator of Impact –

Miami Harbor's outer entrance channel was extended through the 3rd reef between 1991 and 1993. The first baseline hardbottom habitat assessment for Miami Harbor General Reevaluation Report was conducted in 2000 which examined the habitats within a zone 2,500 feet wide including the existing channel (Figure 5). Based on the 2000 survey, the biota of the three outer reef tracts were consistent with the overall assemblage of scleractinian corals, sponges, and gorgonians found offshore of Miami-Dade, Broward, and Palm Beach Counties (DCA 2001) at the time of the survey

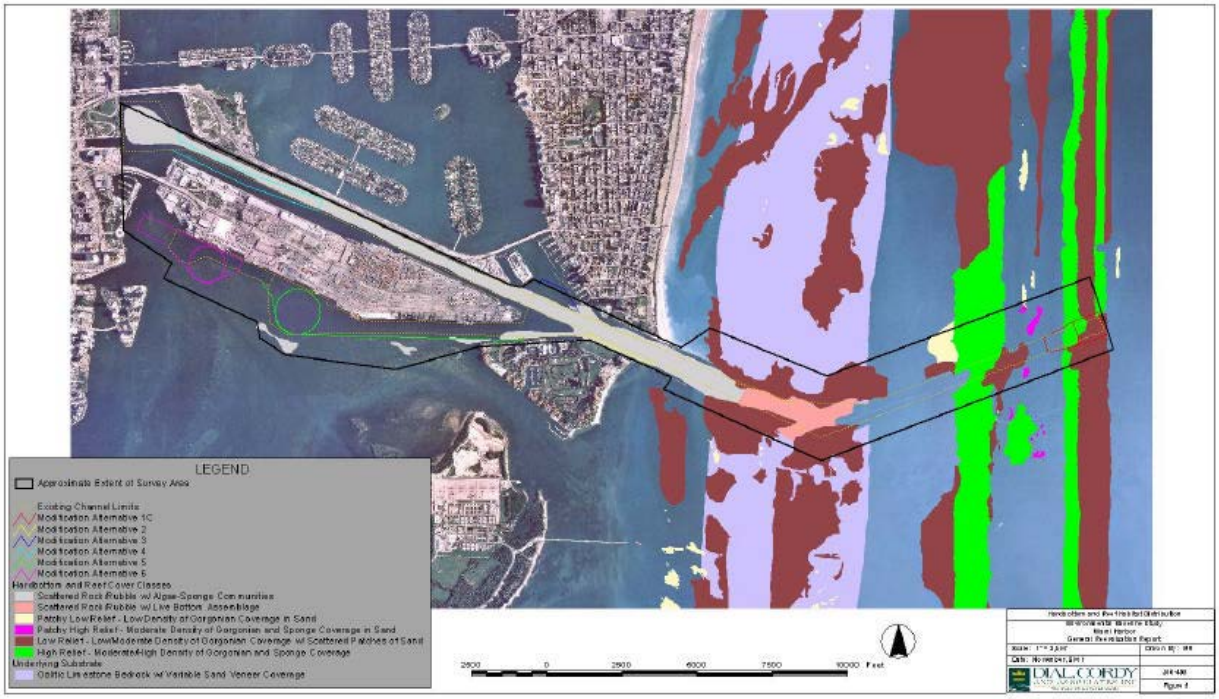

Figure 5 - Miami Harbor Baseline Report - Hardbottom and Reef Habitat Distribution (DCA 2001)

Additionally, the 2010 Miami Harbor baseline study indirect-effect sites and reference sites sampled during the "Pilot Study" had similar functional group percent cover and densities to other reef areas in southeastern Florida that have been characterized by Gilliam et al (2006), Gilliam (2007) and Moyer et al. (2003) (Figure 6), confirming the previous findings in the 2000 survey in much more detail. These results support the conclusion that seven years after construction of the 1991-93 project, it did not result in permanent impacts to the ecosystem in the vicinity of the channel in a manner that significantly changed the functional group composition or densities of those ecosystems.

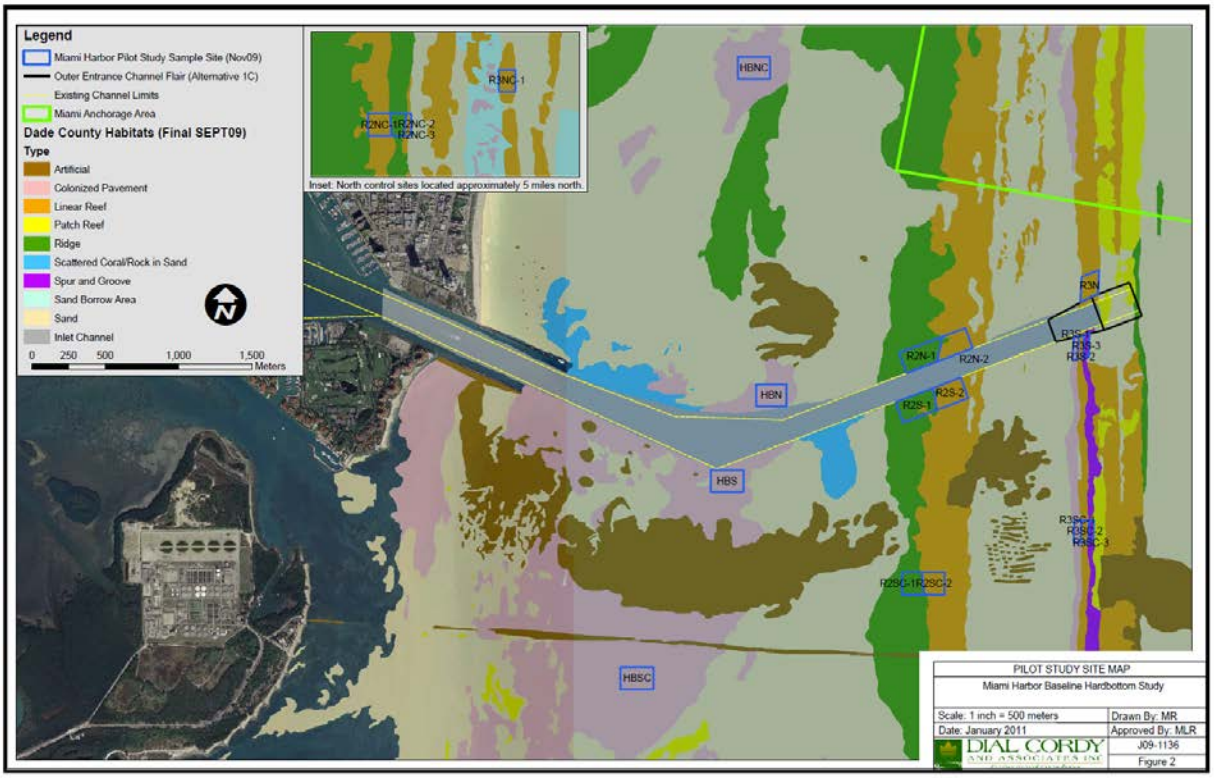

Figure 6 - Miami Harbor Baseline Hardbottom Study - Pilot Study Site Map (DCA 2012)

<u>Sediment Depth as an Indicator of Recovery –</u>

A review of the sediment depth data collected during and after construction at Miami Harbor show that high levels of sediment deposited onto the adjacent reef framework are not permanent and are removed from the system in a relatively short period of time (DCA 2017). In September, 2015, National Marine Fisheries Service (NMFS) conducted dives and measured average sediment depth (without regard to grain size of type of sediment) on the middle reef, north of the Miami Harbor entrance channel. Those sediment values and NMFS' corresponding impact assessment category are included in the 2nd and 3rd column of Table 1. As part of the post-construction impact assessment conducted for the Miami Harbor project, sediment depth was collected at 1 meter intervals along the entire length of both transects (102 measurements per transect) for a total of 1,006 measurements. The average sediment depth for each monitoring location is included in column 2&3 of Table 1 (These data are currently under review by NMFS using photographic comparisons and to date and may show a high degree of error). Only one point out of 1,006 near-channel sediment assessment locations (0.09%) was characterized as fine sediment and exceeded the 3 centimeter (cm) guideline for restoration efforts. The distribution of sediments as recorded in the 2016-2017 impact assessment surveys suggest that the qualitatively documented "clay-like" material that was used to delineate the potential sedimentation effects areas of 2014-2015 were a temporary impact of the project. No "clay-like material" was recorded during these surveys. Given that over a year and a half passed since construction activities, it is likely these sediments have been incorporated into naturally occurring reef sediments, or otherwise dispersed (Griffin 1974, Blair et. al 1994, DCA 2015a, b).

Table 1 - Average Sediment Depths by Location Comparison

Tables 15 & 17 Indirect Impact assessment final report combined with NMFS April 2016 report

Row Labels (the numbers refer to distance in meters from channel)	Average Sediment Depth Dec 2015 (cm)	Average Sediment Depth Sept 2016 (cm)	Recruit Density
R2N1-RR		0.56	1.2
R2N1-LR		2.19**	1.2
R2N-75-RR		0.44	1.4
R2N-75-LR		0.49	1.2
R2N-100-RR	1.78		
R2N-100-HR	1.65		
R2N-150-RR		0.57	0.7
R2N-150-LR		1.0	0.6
R2N-200-LR	1.68		
R2N-200-HR	4.2		
R2N-250-RR		0.66	1.2
R2N-250-LR		0.6	2.2
R2N-300-RR	0.72		
R2N-300-LR	0.92		
R2N-350-RR		0.46	0.6
R2N-350-LR		0.4	0.8
R2N-450-RR		0.45	0.2
R2N-450-LR		1.0	0.9
R2N-500-LR	1.3		
R2N-500-HR	0.95		
R2N-550-RR		0.49	0.6
R2N-550-LR		0.5	0.9
R2N-650-RR		0.58	0.6
R2N-650-LR		0.5	0.3
R2N-700-HR	2.1		
R2N-750-RR		0.49	0.6
R2N-750-LR		0.6	0.8
R2N-850-RR		0.6	1.4
R2N-875-LR		0.5	0.8
R2NC2-RR		0.3	1.6
R2NC1-LR		0.8	2.6
R2NC3-LR		0.4	1.4
R2N-LR	0.32		
R2N-HR	0.30		

** Sites ending in 00 are NMFS sites, sites at 50 or 75 are Corps sites.*

***In all but one site, the average sediment depth is less than 1 cm, and, at this one site exceeding 1 cm (R2N2-LR), the average sediment depth is 2.19 cm. This site contained the deepest accumulation of sediment observed. However, as corroborated by the video data, this site also contains two distinct sand channels (a naturally occurring low relief area that traps sand) dominated by coarse (i.e., not dredge-related) sand accumulated in waves. Adjusting for the presence of the sand channels, i.e.*

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removing the data taken from the sand channels and calculating only sediment depths over hardbottom, the average sediment depth at this site is 0.75 cm.

In addition to using the monitoring data collected in association with the Miami Harbor Deepening Project, the Corps also relied on peer-reviewed literature and monitoring reports from previous dredging projects in southeast Florida (Broward and Dade Counties) as well as an impact assessment associated with Hurricane Andrew (Blair et al. 1994) that documented fine, silty clay material on the reefs offshore of Miami-Dade County. Table 2 includes examples from peer-reviewed studies and project monitoring reports for reported recovery timeframes associated with elevated sediment levels.

Table 2 - Recovery timeframe Durations by Sediment depth range

Literature	Sediment Depths	Location	Reported Recovery time
Marszalek 1981	>1cm in 2 hours	Nearshore & Reefs 2 & 3 Miami-Dade County	2 months (dredging pulses between 1977-1980)
Blair et al. 1994	1 to 4cm silty clay material	Offshore Miami-Dade County (offshore reefs)	2 months
Welch 2002	2 cm	Reef 2 and 3 Miami-Dade County	1 week
Welch 2002	2.9 cm	Reef 2 and 3 Miami-Dade County	"soon after dredging stopped"
Welch 2002	1.14 cm	Reef 2 and 3 Miami-Dade County	"within a few months"
Prekel et al. 2008	Beach quality sand 6 to 8 cm of sand	Nearshore hardbottom Broward County	18 months post construction
DERM 2014	3.7 cm (BAL6)	Reef 2 and 3 Miami-Dade County	6 months

Based on the 20-month timeframe (Sept 2015 cease dredging – May 2017 sediment depth data collection) associated with the Miami Harbor data, the 18-month timeframe associated with monitoring from the Broward County shore protection project and the other monitoring reports that detail even shorter recovery times, the minimum impact recovery timeline would be two years. However, this may be extended to be conservative.

In summary, based on reviews of the data from Miami Harbor, as well as other southeast Florida projects, it appears that the recovery timeframe for hardbottom/reef habitats adjacent to dredging projects or natural events where high levels of sediment are introduced to the water column, ranges from a minimum of two to a maximum of seven years.

Coral Recruits as an Indicator of Recovery –

It is commonly assumed that sedimentation on hardbottom habitats can impede larval coral settlement and recruit development in the area of sedimentation (a recruit is a colony less than 3 cm in diameter). This assumption is not proven out by project surveys and monitoring data. The 2000 and 2010 (DCA 2001, DCA 2012) baseline surveys demonstrated that functional group coverage near the Port Miami channel is not significantly different than seen elsewhere in southeast Florida and the results of the 1-year post-construction monitoring (specifically the additional indirect area monitoring (DCA 2017)) that found in many of the areas where sediment deposition was documented a few months after dredging was complete, sediment levels were at or near those at the control sites within 20 months of the end of

construction (using the date of the LAST survey for the entire survey timeframe). Also every monitoring site had documented coral recruits documented with densities ranging from 0.2 to 2.2 recruits/m^2 (mean 0.91 ±0.45), while controls ranged from 1.4-2.6 recruits/m^2 (mean 1.87 ± 0.64) (Column 4, Table 1).

Additionally, the NMFS Coral Restoration program advised the Corps that "ideal substrate for outplanting of *Acropora* includes areas of relatively flat, consolidated substrate, devoid of thick sediment cover (< 3 cm in depth), and limited cover of space competitors like *Palythoa* and soft corals." This suggests that in general *Acropora cervicornis* can adequately survive, grow and prosper (outplant success criteria) in areas with sediment depths ranging up to 3cm (Tom Moore, Oct 2016 email to Laurel Reichold). If that information were included in the analysis, all impact sites with less than 3 cm of sediment would be considered appropriate *A. cervicornis* habitat. Additionally, for non-Acroporid corals, Lirman et al. (2003) has established that sediment depths must exceed 5cm to prevent coral community development in Florida Bay. Lirman also documents that there are some species that appear to be adapted for high sediment environments, and that they tend to be brooding corals with high recruitments levels, including *Siderastrea radians*, the fifth most common coral found near the Port Everglades channel. The prevalence of this coral demonstrates that this habitat is a high sediment system that is selecting for sediment tolerant species with high recruitment rates.

Lastly, the recently completed RECON survey at Port Everglades documented a large number of recruits at surveyed sites approximately a week following the passage of Hurricane Irma. Nearly four times the number of corals were surveyed in the "CP north inner reef" habitat after the passage of the storm than at sites of the same habitat surveyed before the storm, and nearly all of the variability was due to small corals <3cm. The colonies were too large to have settled out immediately after the storm, so their documentation demonstrates that they are present below the algal mat and sediment at the channelside sites, and that the storm exposed them by clearing off the algal mat and sediment (pers comm, Dial Cordy & Associates April, 2017). This information is included in the draft RECON report that is not yet available to the agencies, but will be provided as soon as it is approved for released.

In summary, presence or absence of recruits may not be a good indicator of the role of sediment in the hardbottom/reef habitat, at least not in association with Port Everglades, a high sediment system, as it appears there are species that can recruit in this high sediment system.

Percentage loss of function –
0-150m – 10% loss

In attempting to assess the permanence of the impacts due to temporary habitat burial associated with the dredged material that settled on the habitats surrounding the channel, the relationship between CTB and sand in the weekly functional group monitoring was used to visualize the amount of sediment being introduced into the system and how quickly that material either washed out of the local system or was incorporated into the sediments already in the system through bioturbation and the growth of turf (Goatley and Bellwood 2013). In the last week of dredging (March 16, 2015), sediment levels at R2N2-RR were approximately 70%, while CTB levels were approximately 15%. The remaining 15% makes up all of the other functional groups (corals, soft corals, sponges, etc.). Previous to that, sediment levels at R2N2-RR reached levels in excess of 80% (August/September 2014) (Figure 7). During the one-year post

construction monitoring (August 2016), the percentage of sediment cover at R2N2 had decreased to 19%, a significant decrease and a continued downward trend. Throughout the channel-side sites, this same trend is observed.

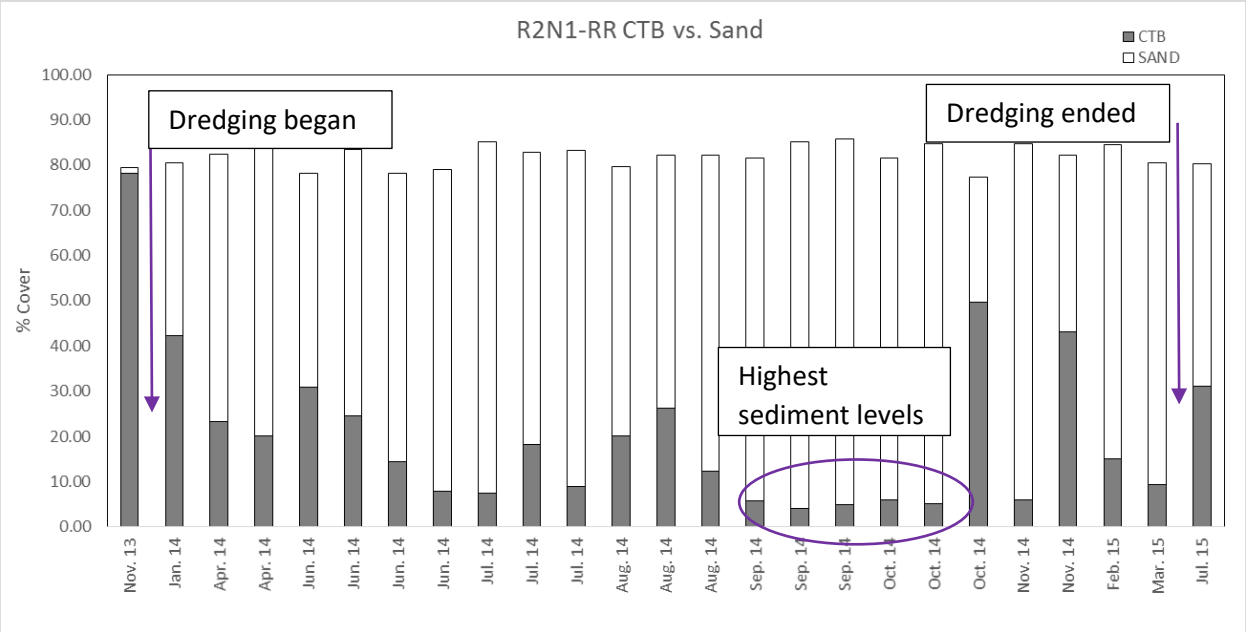

Figure 7 - R2N2-LR CTB vs Sand Analysis

A comparison of functional group mean percent cover at the permanent stations (within 30 meters of the channel edge) pre-project and one-year post-construction shows that in general, scleractinian coral cover decreased approximately the same amount at both channel-side and control sites, octocoral and sponge densities increased channel-side, yet decreased at the control sites (Table 3). There was an increase in sediment cover across the channel-side sites, and a corresponding decrease in CTB cover.

Looking at a short-term scale, a review of the mean functional group percentage coverage comparing the pre-construction baseline condition to the one-year post-construction documents no significant change across all of the functional groups, when compared to their paired controls. Before the project was constructed, during the baseline assessment completed in 2013, the mean percent cover of benthic invertebrates was approximately 15% of the bottom at the channel-side sites during baseline surveys: scleractinians (0.88%), octocorals (9.27%), sponges (4.48%) and zoanthids (0.54%), while CTB and sand comprised the remaining 84.1% of the benthic cover. One year after the project was complete (2016), the mean percent cover of benthic invertebrates was approximately 16% of the bottom at channel-side sites: scleractinians (0.62%), octocorals (9.97%), sponges (4.7%) and zoanthids (0.46%), while CTB and sand comprised the remaining 84.09% of the bottom at channel-side sites.

Table 3 - Mean Functional Group Coverage Baseline vs 1 year Post-Construction (Green = increase and Red = decrease)

Functional Group	Pre-construction		1-Year Post-construction		Delta Channelside	% change	Delta Control	% change
	Channel-side	Control	Channel-side	Control				
Scleractinian corals	0.88	1.90	0.62	1.45	0.26	29.5	0.45	23.7
Octocorals	9.27	14.78	9.97	12.62	0.7	7.6	2.16	14.6
Sponges	4.48	4.87	4.7	3.31	0.22	4.9	1.56	32
Zoanthids	0.54	1.73	0.46	1.91	0.08	14.8	0.18	10.4
Macroalgae	0.18	0.42	0.00	0.45	0.18	100	0.03	7.1
CTB	70.48	64.62	54.76	68.33	15.72	22.3	3.71	5.7
Sediment	13.62	11.26	29.33	11.92	15.71	115.3	0.66	5.9

Marszelak's (1981) review of impacts to habitats on the middle reef associated with dredging sand borrow areas for sand associated with the Miami-Dade County shore protection project, provides this information:

> *Approximately 1 cm of sediment was deposited in less than two hours. Alcyonarians were sediment-free and appeared unaffected by the sediment loading. Scleractinian corals were actively cleansing themselves of sediments; most were partly covered with a layer of mucus and sediment and the polyps were distended and swollen. Strands of mucus, presumably derived from the corals, were observed in the water column and adhering to the erect sea whips. Sponges were covered with a layer of sediment.*
>
> *Monitoring of the affected area over several months revealed that the alcyonarians, mainly sea whips, appeared to be least affected by sediment loading. The upright growth form and flexibility coupled with the cleansing action of currents most likely accounts for the observed high tolerance to siltation. Sponges which at times were completely covered with silt eventually "sloughed-off" the sediments, which appeared to be bound in a mucoid material. No permanent damage to sponges was observed during the monitoring period. The scleractinian corals also recovered from the siltation event and appeared normal except for pale spots on some of the colonies due to the loss of zooxanthellae. Many of the small colonies of D. stokesii, M. cauernosa and other hemispherical forms showed a band of dead tissue adjacent to the substrate, buried beneath the layer of silt which covered the reef surface. The most obvious impact of siltation in the vicinity of the monitor grid was on the limestone substrate rather than on the macrobenthos. It is assumed that most of the attached and endolithic microorganisms colonizing the reef-rock were unable to survive burial by the silt layer. A change in color of the reef surface from white (caused by the recently deposited calcareous silt) to the normal mottled brown color was visible approximately two months after the silt layer had been deposited, indicating that recolonization by microorganisms was in progress. Some of the silt was removed by storm generated waves during the winter months when dredging operations had ceased.*

It should be noted that shore protection projects in the early 1980s did not have to abide by current sand requirements (AKA Florida sand rule) that limits the amount of fine material that could be placed on the beach, and there were no restrictions on overflow associated with these activities. A report from the Corps' geotechnical branch documenting the percentage fines in each of the five contracts from the

Miami-Dade County shore protection project between 1977-1980 documents a range of fines from 4-16% passing the #200 sieve with a total average across the contracts of 11%, more than double the amount of fines that would be in current shore protection project sand sources, so the effects documented by Marszelak contain a higher percentage of fine material and that should be taken into account when reviewing the impacts described therein, when assessing impacts of that project to "beach quality sand".

Assessing impacts of Partial Mortality in Functional Assessments

Given the small percentage of the reef that is covered by scleractian corals at Port Everglades based on 2006 survey data (1.4 corals/m^2; mean of less than one percent across all of the sampled areas, including controls), and the potential for partial mortality of some portion of those few corals associated with sedimentation, the overall loss of some portion of the 1% scleractian coral cover (Table 4) has a very small potential impact on a functional assessment looking at the overall impact of the PENIP on the habitats at and surrounding Port Everglades. This partial mortality can be used to support the 10% functional loss within 150 m of the channel, as this is the zone most likely to have partial mortality associated with sedimentation impacts to scleractian corals.

Table 4 - Percent Cover scleractian corals at Port Everglades 2006 Surveys (DCA 2012)

	Percent Cover		
Reef 2	Zone 1	Zone 2	Zone 3
Scleractian corals	0.3	0.41	NA
Reef 3	Zone 1	Zone 2	Zone 3
Scleractian corals	0.74	0.75	1.15
Reef 3 Previously Impacted	Zone 1	Zone 2	Zone 3
Scleractian corals	0.34	0.44	0.74
Reef 3 Control	Zone 1	Zone 2	Zone 3
Scleractian corals	1	0.53	0.5

It is recommended that instead of a landscape HEA, where the habitat is deemed to be homogeneous in characterization, a population HEA (Milon and Dodge 2001) be conducted looking at the functional loss and recovery of each of the nine main functional groups (Figure 8) found in the habitats surrounding Port Everglades based on their proportional make-up of the overall habitat.

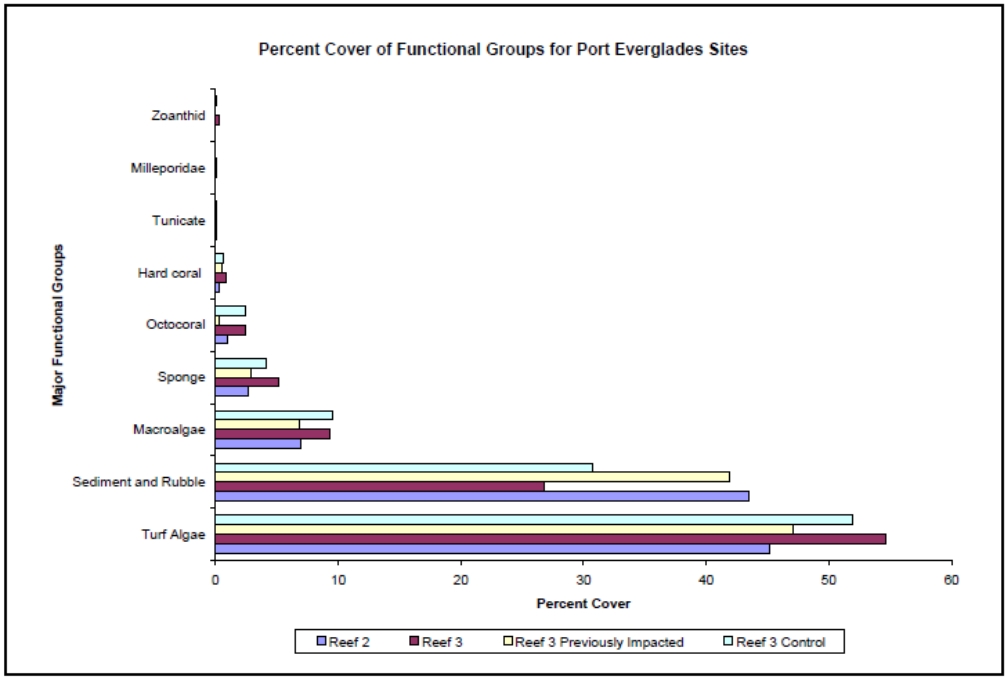

Figure 8 - Mean percent cover data for nine functional groups at four reef sites off Port Everglades in 2006

<u>151-450m – 5% loss/ 451-750m – 2% loss</u>

In a longer term assessment, a comparison of mean cover for soft corals and sponges collected in 2010 during the PED phase of the project and the one-year post construction impact assessment, indirect footprint areas was conducted (Figures 9 and 10). Again, this review focuses on middle reef north (also referred to as the reef 2 in the project impact assessments) as it was the area with the highest level of potential impact throughout the project area.

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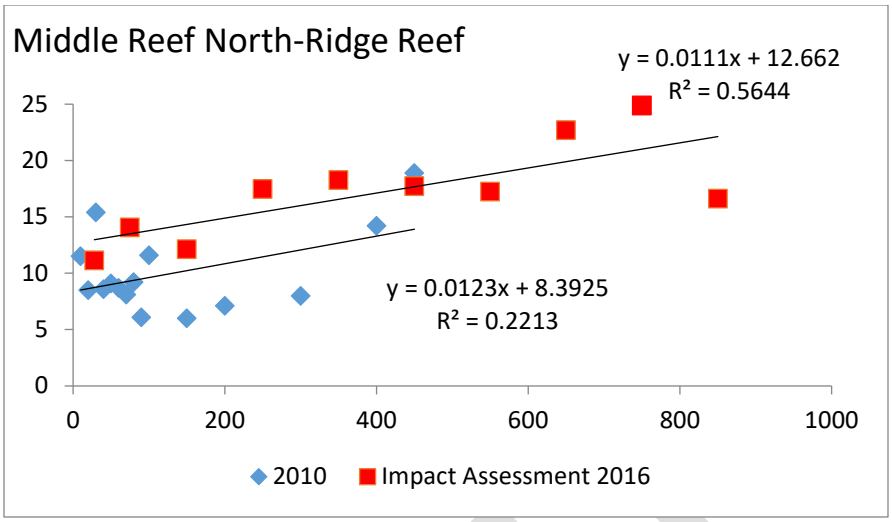

Figure 9. Octocoral densities with respect to site distance from the channel as measured during 2010 baseline survey (blue) and 2016 impact assessment survey (red).

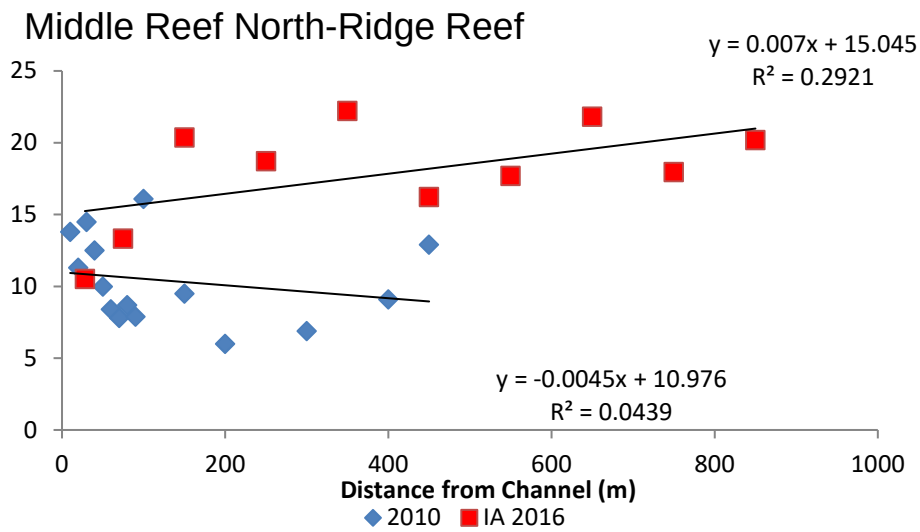

Figure 10- Sponge densities with respect to site distance from the channel as measured during 2010 baseline survey (blue) and 2016 impact assessment survey (red).

Because octocoral and sponge densities from the middle reef north in 2016 were consistent with both 2010 middle reef data and because the analysis of permanent site octocoral and sponge densities found no significant interaction between site and time period with respect to octocoral and sponge densities, it suggests that there was no significant impact to octocoral or sponge densities located along the northern linear middle reef due to construction activities. This same trend was seen on the middle reef south of the channel and on both north and south sides of the channel on the outer reef (AKA Reef 3).

751+m – no loss of function.

No adverse effects of the project were recorded beyond 750m.

All of this data has been used to establish impact zones and will also be used to calculate up front compensatory mitigation associated with unavoidable indirect impacts associated with sedimentation associated with construction of PENIP, providing reasonable assurance to regulators that the unavoidable impacts are adequately replaced.

Artificial Reef Functional Gain

Previous analyses of the functional value of artificial reefs as compensatory mitigation for unavoidable impacts associated with coastal construction projects have determined that the artificial reefs do not mimic the natural habitat functions that they were designed to replace. However, these studies often compare the artificial reefs to "natural reefs" and not to the actual habitat that was proposed to be impacted (Prekel et al. 2008; Gilliam 2012). This is a flawed approach, as compensatory mitigation is designed to replace the originally impacted area, not to serve as functional replacements beyond those of the impacted area. To do so would be ecosystem restoration, not compensatory mitigation. In the case of navigation channels, the areas surrounding the channels are degraded due to poor water quality from land-based sources of pollution (Craft 2008, Futch et al. 2011).

The Corps reviewed documentation of the only other compensatory mitigation project associated with a harbor deepening project in southeast Florida to help provide information on what may be expected from a reef boulder mitigation project associated with channel construction; the Corps reviewed the artificial reef constructed for the Miami Harbor's 1991-93 entrance channel deepening project (also referred to as the 1993 artificial reef). The 1991-93 Miami Harbor deepening project is very similar to the proposed Port Everglades expansion, as it cut through the outer reef to lengthen and deepen the channel.

Additionally to assess how the artificial reef compares to the impacted habitat, the Corps used the same criteria used by FDEP in their UMAM rule for assessing mitigation lift. "…difference in relief and structural composition (i.e., shape and placement of boulders) between natural hardbottom and the artificial reef, communities are expected to differ with respect to their community structure (species richness, taxonomic composition, and size-class distribution).

Coral reef succession is dependent upon conditions at the colonization site. Karlson and Hurd (1993) consider reef habitats to be space limited habitats for sessile invertebrates like scleractinian corals and octocorals. These organisms need disturbance like a hurricane to open up new habitat for larval settlement, fragment attachment or colony expansion. Unlike traditional terrestrial succession explained by Odum (1969), reefs in southeast Florida that are adjacent to inlets are in a constant state of disturbance due to their proximity to coastal communities, poorer water quality associated with inlets and land-based sources of pollution, navigation channels, physical damage due to groundings/anchoring, thermal events, bleaching, etc. This means that new habitat for reef organisms is always being opened up for pioneering species that can survive the more challenging conditions. As the channel moves away from the coast and approaches deeper water, further offshore, some of these perturbations may decrease in intensity (tidal influence of the coastal inlet), decreasing the high turnover of the channelside environment and thus following the more traditional succession pattern shown in Odum, 1969 and the climax community described in Goreau, 1959.

For the Port Everglades functional assessment assumptions, the Corps limited current assumptions to information from actual in water monitoring of the "1993 Artificial Reefs" and the artificial reefs constructed for the Port Miami Phase III project in 2013/2014.

"1993 Artificial Reef"

In 1991, the hardbottom/reef habitats surrounding the Miami Harbor Entrance Channel were characterized by 37 different biotic non-fish species (DERM 1991). Construction of the Miami Harbor Channel extension through the outer reef was completed in 1993 and construction of the artificial reef began in September 1995 and was completed in November 1996 (DERM 2004). There was no transplantation of any organisms to the rocks after placement at the reef site. DERM conducted an assessment of artificial reef success in 2003 (report in March 2004), seven years after construction was complete. In the assessment DERM took biological characterization data from pre-dredge surveys of impact areas in 1990-1991 and compared the artificial reefs biota to those reports. Species assessed included fish and benthic communities. The age of these communities at the time of the report was nine (9) years (1995-2003). These reefs were composed of two segments: POM A (pre-fabricated reef modules) and POM B (limestone boulders). All of the analysis included herein is limited to POM B, the boulders (in two configurations – piles and rows), as that is the material proposed to be used for artificial reef construction at Port Everglades.

The boulder reef at Site POM-B was characterized by 77 species, with taxonomic groups not seen in the channel being present on the reefs. Hard coral species richness increased by 28%; octocoral species richness increased by 43%. Overall species richness of the boulder reefs was 52% higher on the artificial reef than on the reef that was impacted by channel extension in 1993. An additional survey was conducted by DERM in 2007 in association with a Florida Fish and Wildlife Conservation Commission grant looking at the functional group coverage of the reefs after 13 years in the water. Functional group coverage data for the impact site in a pre-dredge condition were not available from DERM's records for comparison. A summary of the results of the Miami channel surveys in 1991 and the POM-B artificial reef boulders is included below (DERM 2004; DERM 2007) (Table 5, Table 6).

Table 5 Miami Harbor Pre-Dredge Survey vs 2003 Mitigation Survey (DERM 2004)

Functional Group	1991 Pre-Dredging # of Taxa	POM A # of Taxa	POM B # of Taxa
Algae	Not Noted	5*	5*
Anemone	Not Noted	1	0
Bivalve	1	2	2
Bryozoan	Not Noted	1	2
Corallimorph	Not Noted	1	0
Echinoderm	1	2	0
Scleractinian Coral	18	18	25
Hydroid	Not Noted	1	1
Hyrdocoral	Not Noted	1	1
Octocoral	8	1	14
Sponge	9	27	28
Tunicate	Not Noted	6	4
Zoanthid	Not Noted	0	0
Totals	37	61	77

Table 6 Miami Harbor Artificial Reef Functional Group Percent Cover in 2007 (DERM 2007)

Functional Group	POM B (boulders) % cover
Algae	76.54
Scleractinian Coral	4.04
Milleporidae	0.38
Octocoral	11.73
Sponges	5.14
Ascidaria	0.11
Zoanthid	0.04
Other live	0.23

As noted in Table 7, both of the 1993 artificial reefs have more octocoral percent cover (POM boulder piles 4.83%; POM bounder rows 11.73%) than the outer reef (R3N 1.02%; R3S 6.89%) and serve as a good indicator that given time (in this case seven years), the diversity of octocorals can equal or exceed that of the originally impacted area. Octocorals are one of the two dominant functional groups on southeast Florida reefs

Sponges are the second dominant functional group on southeast Florida reefs. Sponge percent cover on the 1993 artificial reef (POM boulder piles 10.49%/ boulder row 5.14%) exceeded that found on the outer reef (R3N 4.22%; R3S 4.8%) at both bounder reef locations. This serves as another indicator that these reefs were providing equal or greater function for sponges than the reef next to the channel.

Monitoring of the 1993 artificial reef in 2004 and 2007 provide data after a longer timeframe than is usually observed for mitigation reefs, thus providing more info on how these reefs can provide many of the functions lost or impacted associated with harbor expansion projects and associated unavoidable impacts.

Miami Harbor Phase III Artificial Reef (2013/2014 reef)

As previously stated, the artificial reef constructed as compensatory mitigation for the Miami Harbor Phase III project was deployed in fall of 2013 and consists of north and south components (1.782 acres and 9.816 acres respectively). Relocation of 566 non-*Acropora* coral colonies to the northern half of the northern artificial reef site between April and May of 2014. For this functional assessment, the Corps is using May 2014 as beginning timeframe for reef function assessment and we are comparing collected data from the "2013 artificial reef" to the Baseline Reef Report (DCA 2012) of the ecosystem surrounding Miami Harbor prepared for the Feasibility Study (Table 7), particularly data for the outer reef, which was the impacted area of the project that the mitigation was constructed for. All data and graphics regarding the Phase III artificial reef are taken from the Year 3 Artificial Reef Monitoring Report (CSI 2017).

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Table 7 Functional Group Comparison – Percent Cover

		Algae	Scleractinian	Octocoral	Sponge	Sand*	Hydroid	Tunicate	Bare substrate	Other
Reef 3 (2012)										
	R3N	45.13	0.32	1.02	4.22	49.24	0.07			
	R3S	79.55	0.21	6.89	4.80	7.89	0.65			
POM P3 Artificial Reef										
2015	ARN	82.43	.95	0.04	2.31	11.85	0.87	0.25	0.13	1.40
	ARS	81.67	.20	0.06	1.36	13.86	0.65	0.22	0.60	1.13
	AR combined	81.79	.32	0.06	1.51	13.53	0.91	0.22	0.52	1.17
2016	ARN	70.63	1.64	0.03	8.54	14.74	1.06	0.46	0.21	2.70
	ARS	75.71	0.92	0.16	5.45	14.24	0.62	0.29	0.65	1.97
	AR combined	74.86	1.04	0.14	5.96	14.32	0.70	0.31	0.57	2.09
2017	ARN	83.19	1.29	0.13	2.11	11.24	0.15	0.24	0.00	1.66
	ARS	83.84	1.05	0.24	3.15	10.09	0.17	0.12	0.01	1.34
	AR combined	83.73	1.09	0.22	2.97	10.28	0.16	0.14	0.01	1.39
1993 Reef										
2007	POM B (Boulder Row)	76.54	4.04	11.73	5.14	1.79 (as sand, pavement, rubble)	0.04	0.11	NA	0.23
	POM B (Boulder piles)	79.9	3.57	4.83	10.49	0.27 (as sand, pavement, rubble)	0.04	0.00	NA	0.49

* Reef 3 assessment lumped sand/rubble and bare together. The value for this is listed under the sand category.

Highlighted values exceed the impact area baseline

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Within three years or placement, hard coral species richness, diversity, and evenness on the artificial reef are all becoming similar to the outer reef. The 2013 artificial reef includes 89.47% of the hard coral species found on the outer reef as of the third year monitoring event. The two most abundant species of hard coral on the artificial reef (*Porites astreoides* and *Siderastrea* spp.) were the same prevalent species as recorded on the outer reef and recorded in other studies of southeast Florida outer reefs (Walker, 2014). Also, nine (9) of the ten (10) most prevalent scleractinian corals on the outer reef were all observed on the artificial reef including: *Siderastrea* spp., *Porites astreoides*, *Montastraea cavernosa*, *Stephanocoenia intersepta*, *Madracis decactis*, *Dichocoenia stokesii*, *Porites porites*, *Agaricia agaricites*, and *Meandrina meandrites*. Only two (2) species of coral *Dendrogyra cylindrus*, and *Solenastrea hyades*), observed on the outer reef, have not yet been observed on the artificial reef. During the outer reef survey, only one small colony (4 cm) of *Dendrogyra cylindrus* was observed. Recruitment and survival rates of *D. cylindrus* is low. There is a severe population bottleneck for *D. cylindrus* during dispersal and settlement of larvae, the causes for which have yet to be identified (Marhaver et al 2015). *D. cylindrus* appears to be most effective in propagation by fragmentation. As a result, the likelihood of *D. cylindrus* recruiting to the artificial reef may also be low as there are no existing *D. cylindrus* colonies immediately surrounding the artificial reef and no *D. cylindrus colonies were transplanted to the North Artificial Reef.* Since the year 3 artificial reef survey was conducted, NMFS and FWC have documented the die-off of more than 99% of the *D.cylundrus* in southeast Florida due to a region-wide disease outbreak. During the year 3 monitoring survey, four (4) hard coral species were observed within a quadrat on the artificial reef that were not present on the outer reef: *Agaricia lamarcki, Colpophyllia natans, Phyllangia americana,* and *Porites furcata.* Thus as of the year 3 survey the artificial reef has a greater species richness than the outer reef (Table 8).

Table 8 Scleractinian Coral Species Richness/ Density

		Species Richness	Density (Colonies/m2)	Shannon-Wiener Diversity Index	Shannon-Wiener Evenness Index
Reef 3					
	R3N		1.0	1.94	0.74
	R3S		1.3	1.60	0.55
	Combined	19	1.18	1.82	0.62
2013 Artificial Reef					
2015	ARN	12	3.44	1.23	0.22
	ARS	12	0.20	1.72	0.39
	AR combined	17	0.74	1.41	0.24
2016	ARN	18	8.29	1.18	0.41
	ARS	18	2.36	1.30	0.46
	AR combined	22	3.35	1.27	0.42
2017	ARN	18	7.28	1.66	0.58
	ARS	19	4.75	1.43	0.42
	AR combined	21	5.17	1.55	0.49

Highlighted values exceed the impact area baseline

One likely contributor to the success of the hard coral species on the artificial reef is the presence of urchin on the reefs. Multiple juvenile long-spined urchin (*Diadema antillarum*) were observed on many of the transects during the year 3 monitoring on the 2013 artificial reef, but no *D. antillarum* were observed on the outer reef during the baseline hardbottom survey (DCA 2012). Increased densities,

survivorship, and growth rates of juvenile coral colonies have been observed in areas with higher populations of *D. antillarum* (Lessios 2016) as they reduce the amount of algal cover that negatively affects coral recruitment (Vermeij 2005).

The average hard coral density recorded on the artificial reef was 5.17 colonies/m^2, which is greater than that recorded on the outer reef (1.18 colonies/m^2; Table 8). The north artificial reef had much higher hard coral density than the outer reef sites, with 7.28 colonies/m^2 recorded. The observed differences in recruitment between the northern and southern artificial reefs may be due to a multitude of factors, including proximity to donor sites with corals that are actively spawning, prevailing currents at the time of coral spawning, benthic mortality, and benthic settlement based on the presence of the 566 relocated hard coral colonies placed on the north artificial reef site (no coral colonies were transplanted to the south artificial reef), which could be contributing to coral recruitment through spawning or other settlement cues for planktonic coral larvae (Abelson 2006; Moulding 2005). Continued monitoring may clarify this current difference in hard coral density between the artificial reef sites.

As expected on a newly constructed artificial reef, hard coral size was skewed to colonies less than 5 cm in diameter (Table 9; Table 10; Figure 11). The size-class distribution on the artificial reef is lower than the outer reef, which is predictable in the beginning stages of recruitment, but each monitoring survey has shown a size-class distribution more similar to the outer reef. Future monitoring will most likely continue to show a wider size-class distribution more similar to the outer reef as the recruits continue to grow and become established colonies (Moulding 2005). Growth rates of hard coral recruits is slow and can take years before recruits contribute to increased coral cover and reef health; however, the presence of a higher density of coral recruits and the survival of juveniles colonies is a positive sign that the artificial reef is, and will continue to be, successful (Edmunds 2007; Moulding 2005).

Table 9 Scleractinian Coral Size – Does not include any relocated corals

Site		0-5cm	6-10cm	11-25cm	>26cm	Total
	R3N	155	110	38	3	306
	R3S	275	77	47	4	403
	R3 Combined	430	187	85	7	709
Artificial Reef						
2015						
	ARN	271	2	1	1	275
	ARS	75	2	1	1	79
	AR Combined	346	4	2	2	354
2016						
	ARN	633	29	0	1	633
	ARS	943	0	1	0	944
	AR Combined	1,576	29	1	1	1607
2017						
	ARN	463	115	3	1	582
	ARS	1,838	62	1	0	1,901
	AR Combined	2,301	177	4	1	2,483

Table 10 Average Hard Coral Colony Size (cm)

Site	2012	3-year (2017)	2-year (2016)	1-year (2015)
R3N	6.80			
R3S	5.60			
R3 Combined	6.12			
ARN		3.96	2.77	1.73
ARS		2.75	1.73	2.11
AR Combined		3.03	2.16	1.81

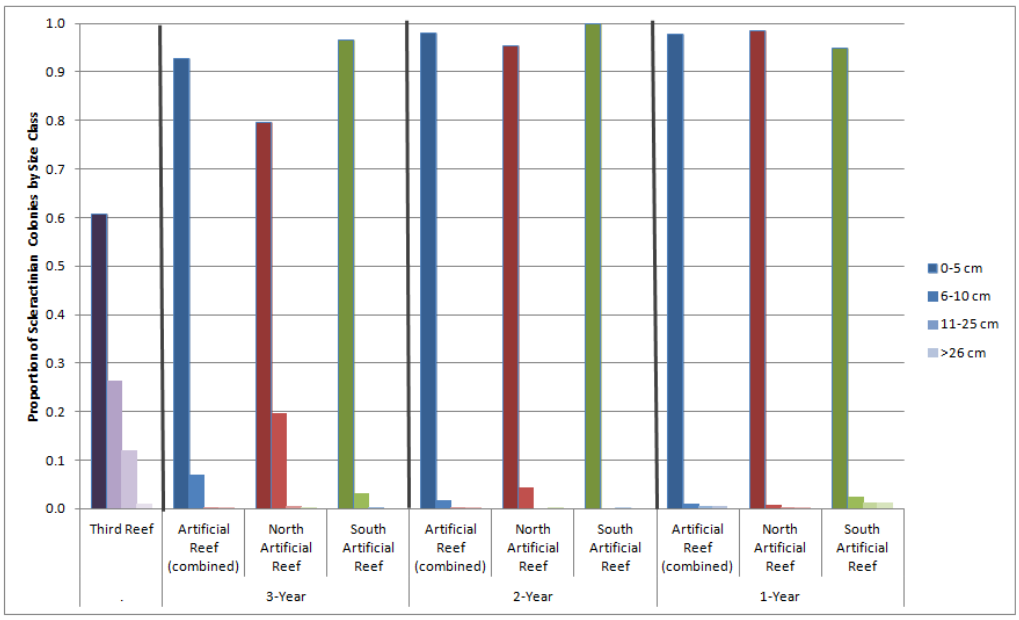

Figure 11 Proportion of Hard Coral Colonies by Size Class and Site

Octocoral genera richness, diversity, and evenness on the artificial reef are currently all less than what was recorded on the outer reef. Only 44.44%, or four (4) of the nine (9), octocoral species observed on the Outer Reef were present on the artificial reef during the year 3 monitoring survey. *Pseudopterogorgia* was the most abundant octocoral genus observed at the artificial reef and *Eunicea* was the second most abundant, which were also the predominant octocoral genera present at the outer reef (DCA 2012). While not all nine (9) of the octocorals observed on the outer reef have been observed within the quadrats monitored as of the year 3 monitoring survey for the artificial reef (*Briareum*, *Erythropodium*, *Plexaura*, *Pseudoplexaura*, and *Pterogorgia* were not yet recorded), the recent survey identified two (2) additional species of octocoral (*Muriceopsis and Plexaurella*) on the artificial reef that were not previously observed on the outer reef. While the Octocoral genera has not yet met the permit-defined success criteria (75% of the genera from outer reef must be on the artificial reef) the artificial reef is proving to be a healthy thriving community that supports the same dominant octocoral species as the outer reef, while offering habitat to other species of octocoral not observed during the outer reef survey.

Average octocoral density for the artificial reef, 0.39 colonies/m^2, was lower than on the outer reef, 0.70 colonies/m^2 (Table 11). The south artificial reef had a higher number of colonies (176) and a higher density (0.44) of octocorals than the north artificial reef (12 colonies and 0.15 density, respectively). The

observed differences in the current octocoral density and species diversity may be partly due to different survival rates as a result of the environmental conditions, mainly sedimentation load and current velocities (Perkol-Finkel 2009). Other factors such as low recruitment rates or the small sample size collected on the north artificial reef may be affecting the observed octocoral densities, especially the lower octocoral density observed on the north artificial reef versus the south artificial reef. Octocoral size-class distribution was skewed to the smaller size classes, consistent with the hard coral size-class distribution on a newly developing reef; the year 3 monitoring survey showed a size-class distribution more similar to the outer reef than during the year 1 monitoring survey (Table 12; Figure 12). Future monitoring will most likely continue to show a size-class distribution more similar to the outer reef as the recruits continue to grow.

Table 11 Octocoral Species Richness/ Density

		Species Richness	Density (Colonies/m2)	Shannon-Wiener Diversity Index	Shannon-Wiener Evenness Index
Reef 3					
	R3N	9	1.0	1.18	0.54
	R3S		0.4	1.06	0.54
Artificial Reef					
2015	ARN	1	0.03	0	0.0
	ARS	2	0.05	0.53	0.76
	AR combined	2	0.04	0.50	0.72
2016	ARN	1	0.04	0	0
	ARS	5	0.24	0.57	0.41
	AR combined	5	0.21	0.56	0.35
2017	ARN	3	0.15	0.57	0.52
	ARS	6	0.44	0.73	0.41
	AR combined	6	0.39	0.73	0.41

Highlighted values exceed the impact area baseline

Table 12 Percentage of Octocoral Size Class

Site	0-5cm	6-10cm	11-25cm	>26cm
R3N	9%	14%	45%	32%
R3S	15%	15%	25%	45%
R3 Combined	11%	14%	39%	36%
Artificial Reef				
2015				
ARN	100%	0%	0%	0%
ARS	94%	6%	0%	0%
AR Combined	95%	5%	0%	0%
2016				
ARN	67%	0%	33%	0%
ARS	66%	26%	7%	1%
AR Combined	66%	25%	8%	1%
2017				
ARN	42%	17%	42%	0%
ARS	31%	30%	35%	5%
AR Combined	31%	29%	35%	4%

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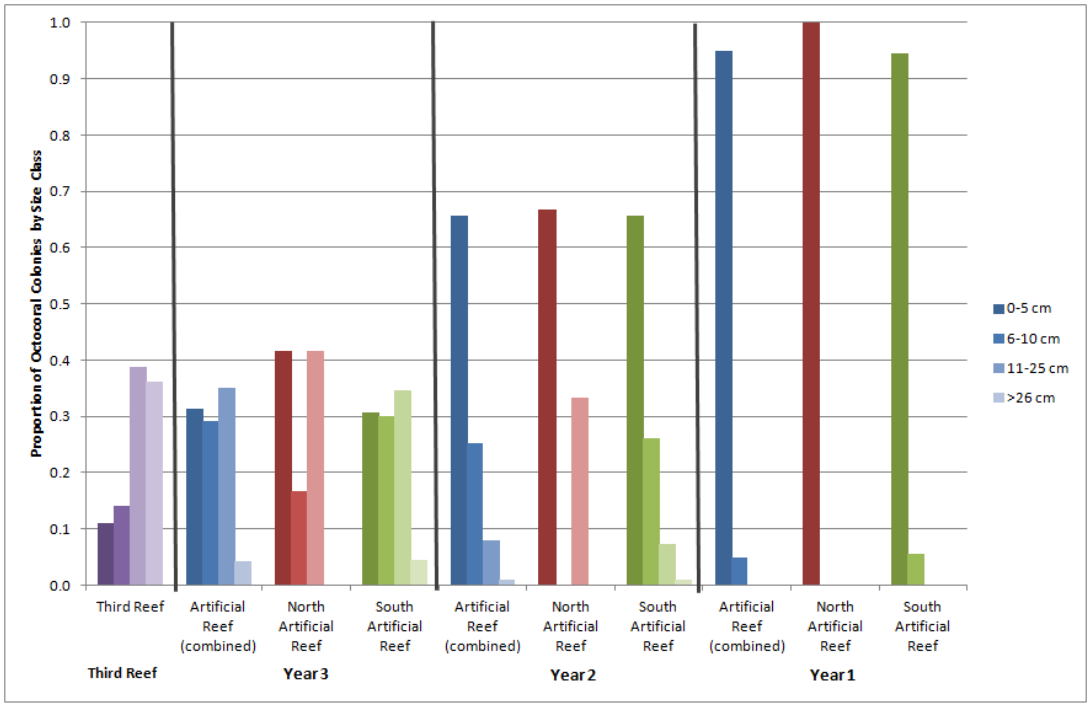

Figure 12 - Proportion of Octocoral Colonies by Size Class and Site.

The average sponge density on the artificial reef was 10.75 colonies/m^2 during the year 3 monitoring, which was greater than recorded on the outer reef, 5.2 colonies/m^2 (Table 10; Figure 13) (DCA 2012). Due to the way the sponge data was initially collected for the outer reef (they were not identified to genus and analyzed in depth) sponges were not able to be used as one of the FDEP permit's comparative benthic groups (hard coral, octocoral, and algae) used to assess the development of the artificial reef. It is apparent that the artificial reef is supporting a large population of sponges, with sponge densities currently double what they were on the outer reef.

Table 13 Sponge Species Richness/Density

		Number of Colonies	Density (Colonies/m2)
Reef 3			
	R3N	2,178	7.3
	R3S	967	3.2
	Combined	3,145	5.24
Artificial Reef			
2015	ARN	73	0.91
	ARS	298	0.75
	AR combined	371	0.77
2016	ARN	556	6.95
	ARS	3,565	8.91
	AR combined	4,121	8.59
2017	ARN	837	10.46
	ARS	4,323	10.81
	AR combined	5,160	10.75

Highlighted values exceed the impact area baseline

Twenty-two (22) genera of algae were observed at the artificial reef, which is greater than the five (5) genera of algae observed at the outer reef (Table 14). All, 100%, of the five (5) algae species on the outer reef were observed on the artificial reef. The south reef had a higher generic richness of algae than the north reef with 22 and 14 genera present, respectively. Turf algae was the most predominant algae at the artificial reef and was present on all transects. Turf algae is an important component in development of artificial reefs by providing food to many reef-associated invertebrates and fishes (Palmer-Zwahlen 1994), however, it can be detrimental to the colonization of coral (Vermeij 2005). The artificial reef has a greater macroalgae diversity than the outer reef with 17 more genera of macroalgae on the artificial reef as compared to the outer reef.

Table 14 Algae Presence

		Number of genera
Reef 3		
	Combined	5
Artificial Reef		
2015	ARN	11
	ARS	17
	AR combined	18
2016	ARN	13
	ARS	25
	AR combined	26
2017	ARN	14
	ARS	22
	AR combined	22

In conclusion, artificial reefs serve very similar habitat functions as natural hardbottom/reef habitat adjacent to large inlets that are often impacted by harbor expansion projects, as well as perturbations associated with storms, and inlet flushing. A functional replacement timeframe for artificial reefs in South Florida should be between seven and ten years, based on the documented colonization of the 1993 artificial reef and the 2013/2014 artificial reef. All of the major functional group densities on the artificial reefs have either exceeded the reference natural habitat (sponges, scleractinian corals, octocorals (1993 reef)), or are quickly approaching those densities (octocorals 2013 reef). And as time passes, the sizes of these organisms will continue to grow, approaching the average size of the reference habitat (scleractinian corals – Table 10; octocorals – Figure 12). The functional gain over the seven to ten years should be on the order of 80-90% due to the density and percent cover of colonizing the artificial reefs. As stated previously, artificial reefs for compensatory mitigation should be compared to the habitat they are mitigating for, not to reference habitats not subject to the perturbations of channelside reef environments.

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Literature Cited

Abelson, A. 2006. Artificial Reefs VS Coral Transplantation as Restoration Tools for Mitigating Coral Reef Deterioration: Benefits, Concerns, and Proposed Guidelines. *Bulletin of Marine Science* 78(1): 151-159.

Blair, S.M., McIntosh, T.L. and Mostkoff, B.J., 1994. Impacts of Hurricane Andrew on the offshore reef systems of central and northern Dade County, Florida. Bulletin of Marine Science, 54(3), pp.961-973.

Craft, J.A. 2008. Using nearshore macrobenthos as environmental indicators adjacent to a major navigation inlet: Port Everglades inlet, Florida. Proceedings of the 11th Annual International Coral Reef Symposium. Ft Lauderdale, FL

CSA 1981a. Environmental Monitoring Associated with the Port Everglades Harbor Deepening Project of 1980. Final Report. June 1981. Volume 1.

CSA 1981b. Environmental Monitoring Associated with the Port Everglades Harbor Deepening Project of 1980. June 1981. Volume II (Section A): Appendices.

CSI 2017. 3-Year Annual Artificial Reef Monitoring Report for Miami Harbor Phase III Federal Channel Expansion. Contract No. E12-SEA-03/Project No 1999-027. October 2017.

DCA 2001. Environmental Baseline Study, Miami Harbor, General Reevaluation Report. Final Report. November 2001.

DCA 2009. Benthic and Fish Community Assessment at Port Everglades Harbor Entrance Channel. Final Report. December 2009.

DCA 2012. Miami Harbor Baseline Hardbottom Study. Revised Final. March 2012.

DCA 2015a. Monthly Delineation of Potential Sedimentation Effect Area Within Nearshore Hardbottom (January 2015) Port of Miami Phase III Federal Channel Expansion Project. FDEP Permit #0305721-001-BI April 13, 2015. Final Report.

DCA 2015b. Delineation of Potential Sedimentation Effect Area Within Middle & Outer Reef Habitats. Port of Miami Phase III Federal Channel Expansion Project FDEP Permit #0305721-001-BI August, 2015. Final Report.

DCA 2015c. Miami Harbor Phase III Federal Channel Expansion Project. FDEP Permit #0305721-001-BI. Quantitative Post-Construction Analysis for Middle and Outer Reef Benthic Communities. November 2015. Final Report.

DCA. 2017. Miami Harbor Phase III Federal Channel Expansion Project Permit No. 0305721-001-BI Impact Assessment for Hardbottom Middle and Outer Reef Benthic Communities at Cross Sites. Final November 2017

DERM 1991. Port of Miami Biological Assessment. September 19, 1991.

DERM 2004. Miami-Dade Seaport Department Mitigation Artificial Reefs. Assessment of Prefabricated Artificial Reef Modules (POM-A) and Linear Limerock Boulder areas (POM-B) Report prepared

DRAFT

for Miami-Dade Seaport Department By Miami-Dade Department of Environmental Resources Management Restoration & Enhancement Section.

DERM 2007. Baseline Biological Monitoring of Miami-Dade Limerock Boulder Reefs. Final Report. Submitted to Florida Fish and Wildlife Conservation Commission in fulfillment of grant agreement FWCC-06121.

DERM. 2014. 1 Year Quantitative Post Construction Benthic Monitoring in Association with the Offshore Borrow Area utilized for Contract E: Miami-Dade Beach Nourishment. Submitted in compliance with: FDEP JCP Permit File #0295427-001JC and associated AGREEMENT between FDEP and Miami-Dade County Miami-Dade County Department of Regulatory and Economic Resources Division of Environmental Resources Management

Edmunds, P.J. 2007. Evidence For A Decadal-Scale Decline In The Growth Rates Of Juvenile Scleractinian Corals. *Marine Ecology Progress Series* 341:1-13.

Futch, J.C., D.W Griffin, K. Banks, and E.K. Lipp. 2011. Evaluation of sewage source and fate on southeast Florida coral reefs. *Marine Pollution Bulletin*. 62: 2308-2316.

Gajewski, L.S. and S. Uscinowicz. 1993. Hydrologic and Sedimentologic Aspects of Mining Aggregate from the Slupsk Bank (Baltic Sea). Marine Georesources and Geotechnology. Volume 11, pp 229-244.

Gilliam, D.S. 2007. Southeast Florida Coral Reef Evaluation and Monitoring Project 2006 Year 4 Final Report. Prepared for: Florida Fish and Wildlife Conservation Commission, Fish & Wildlife Research Institute, Florida Department of Environmental Protection. Pp. 31.

Gilliam, D.S. 2012. A Study to Evaluate Reef Recovery Following Injury and Mitigation Structures Offshore Southeast Florida: Phase II. Nova Southeastern University Oceanographic Center. Dania Beach, Florida.

Gilliam, D.S., R.E. Dodge, R.E. Spieler, L.K.B. Jordan, and J.C. Walczak. 2006. Marine Biological Monitoring in Broward County, Florida: Year 6 Annual Report. Technical Report DPEP 04-01. Ft. Lauderdale, Florida: Broward County, Board of County Commissioners.

Goatley, Christopher H and David R. Bellwood. 2013. Ecological Consequences of Sediment on High-Energy Coral Reefs. PLOS One. Vol 8. Issue 10. October 2013.

Goreau, Thomas, F. 1959. The ecology of Jamaican coral reefs. I. Species composition and zonation. Ecology. Volume 40, Issue 1. Pg 67-90.

Griffin, G. 1974. Case history of a typical dredge-fill project in the northern Florida Keys — effects on water clarity, sedimentation rates and biota. Final Report: Fort Pierce, Florida Harbor Branch Foundation Inc.

Karlson, R.H and L.E. Hurd. 1993. Disturbance, coral reef communities, and changing ecological paradigms. Coral Reefs. Volume 12: 117-125

Lessios, H.A. 2016. The Great *Diadema antillarum* Die-Off: 30 Years Later. *Annual Review of Marine Science* 8:267-83.

DRAFT

Lirman, D., Orlando, B., Maciá, S., Manzello, D., Kaufman, L., Biber, P. and Jones, T. 2003. Coral communities of Biscayne Bay, Florida and adjacent offshore areas: diversity, abundance, distribution, and environmental correlates. Aquatic Conserv: Mar. Freshw. Ecosyst., 13: 121–135. doi:10.1002/aqc.552

Marhaver, K., Vermeij, M.J. and Medina, M.M. 2015. Reproductive Natural History and Successful Juvenile Propagation of the Threatened Caribbean Pillar Coral *Dendrogyra cylindrus*. BMC Ecology 15:9.

Marszelak, D.S. 1981. Impact of Dredging on a Subtropical Reef Community, Southeast Florida, USA. Proceedings of the Fourth International Coral Reef Symposium, Manila. Vol. 1.

Miller, M. W., Karazsia, J., Groves, C. E., Griffin, S., Moore, T., Wilber, P., & Gregg, K. (2016). Detecting sedimentation impacts to coral reefs resulting from dredging the PortMiami, Florida USA. *PeerJ*, *4*, e2711. http://doi.org/10.7717/peerj.2711

Milon, J.W., and R.E. Dodge. 2001. Applying habitat equivalency analysis for coral reef damage assessment and restoration. *Bulletin of Marine Science*. Volume 69, Number 2. Pages 975-988.

Moore, Tom. 2016. Email personal communication. October 03, 2016.

Moulding, A.L. 2005. Coral recruitment patterns in the Florida Keys. Rev. biol. trop, San José, v. 53, supl. 1, p. 75-82, May 2005. Available from http://www.scielo.sa.cr/scielo.php?script=sci_arttext&pid=S0034-77442005000300013&lng=en&nrm=iso. Access on 20 Apr. 2018.

Moyer, R. P., B. Riegl, et al. 2003. "Spatial patterns and ecology of benethic communities on a high-latitude South Florida (Broward County, USA) reef system." Coral Reefs **22**: 447-464

Newell, R.C. & Seiderer, L.J. 2003. Ecological impacts of marine aggregate dredging on seabed resources. Marine Ecological Surveys Limited. October 2003.

NOAA-NMFS. 2016. NMFS Assessment of Sedimentation Impacts resulting from Port of Miami Expansion. Final Report.

Odum, EP. 1969. The strategy of ecosystem development. Science 164:262-270.

Palmer-Zwahlen, M.L. and D.A. Aseltine. 1994. Successional Development of the Turf Community on a Quarry Rock Artificial Reef. *Bulletin of Marine Science* 55(2-3): 902-923.

Perkol-Finkel, S and Y. Benayahu. 2009. The role of differential survival patterns in shaping corals communities on neighboring artificial and natural reefs. Journal of Experimental Marine Biology and Ecology. Volume 369, pages 1-7

Prekel, S., L. Fisher and S. Higgins. 2008. Habitat Similarity between an artificial reef and the Surrounding Natural Hardbottom in Broward County, Florida, USA. Proceedings of the 11th International Coral Reef Symposium, Ft. Lauderdale, FL 7-11 July 2008.

Vermeij, M. J. A. 2005. Substrate Composition And Adult Distribution Determine Recruitment Patterns In A Caribbean Brooding Coral. *Marine Ecology Progress Series.* Vol. 295: 123-133.

DRAFT

Walker, B.K. and K. Klug. 2014. Southeast Florida Shallow-Water Habitat Mapping & Coral Reef Community Characterization. Florida DEP Coral Reef Conservation Program report. Miami Beach, FL. Pp 83.

Welch, L. 2002. Memorandum to Steve Blair, Chief. Restoration and Enhancement Section. Summary of Sedimentation Effects on Coral Reefs Adjacent to Beach Renourishment Borrow Areas in Miami-Dade County. September 9, 2002.

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