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Extensive coral mortality and critical habitat loss following dredging and their association with remotely-sensed sediment plumes

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ABSTRACT

Dredging poses a potential threat to coral reefs, yet quantifying impacts is often difficult due to the large spatial footprint of potential effects and co-occurrence of other disturbances. Here we analyzed in situ monitoring data and remotely-sensed sediment plumes to assess impacts of the 2013–2015 Port of Miami dredging on corals and reef habitat. To control for contemporaneous bleaching and disease, we analyzed the spatial distribution of impacts in relation to the dredged channel. Areas closer to dredging experienced higher sediment trap accumulation, benthic sediment cover, coral burial, and coral mortality, and our spatial analyses indicate that > 560,000 corals were killed within 0.5 km, with impacts likely extending over 5–10 km. The occurrence of sediment plumes explained ~60% of spatial variability in measured impacts, suggesting that remotely-sensed plumes, when properly calibrated against in situ monitoring data, can reliably estimate the magnitude and extent of dredging impacts.

1. Introduction

An increase in port deepening and widening projects is occurring worldwide to accommodate Neo-Panamax cargo ships following expansion of the Panama canal in 2016 (Ashe, 2018; Good, 2016; Wyss et al., 2012). Several dredging projects in shallow-water ports adjacent to coral reef areas along the eastern seaboard of the United States and the Caribbean have recently been completed, or are planned in the near future (Braley and Doyle, 2017; Good, 2016; Whitefield, 2016). Because of the fragility of coral reef ecosystems and their widespread decline, a critical assessment of environmental impacts, accompanied by an evaluation of best practices for monitoring and reducing them, are key conservation goals in light of continued dredging activities.

Coastal dredging and construction are known to cause significant harm to coral reef ecosystems (Bak, 1978; Dodge and Rimas Vaisnys, 1977; Erftemeijer et al., 2012). Dredging can impact corals and coral habitat through a variety of cause-effect pathways reviewed by Jones et al. (2016), including directly via sedimentation (the deposition of particulate matter on the benthos), and indirectly via increased turbidity and shading from sediment plumes. Sedimentation can potentially impact almost every biological function of corals, from feeding through reproduction (reviewed by Erftemeijer et al., 2012; Fabricius, 2005; Jones et al., 2015; Rogers, 1990). Although some degree of sedimentation is common on coral reefs, and some species may persist in turbid environments through long-term adaptation, high sedimentation is typically detrimental to reef development because corals require a hard substrate on which to settle and grow, and because it is energetically costly to remove sediment from their surfaces, either through ciliary action, mucus production, and/or hydrostatic polyp inflation (Bessell-Browne et al., 2017a; Dodge and Rimas Vaisnys, 1977; Humanes et al., 2017; Riegl and Branch, 1995; Stafford-Smith and Ormond, 1992). Due to the high energetic requirements of self-cleaning and the inability of corals to successfully open their polyps (Riegl and Branch, 1995), corals with ongoing sedimentation impacts may reduce feeding (Abdel-Salam and Porter, 1988; Erftemeijer et al., 2012; Szmant-Froelich et al., 1981). Combined with low light levels (indirectly from increased turbidity or directly as a result of smothering), which inhibit the photophysiology of algal symbionts (Abdel-Salam and Porter, 1988; Philipp and Fabricius, 2003; Piniak, 2007; Telesnicki and Goldberg, 1995; Weber et al., 2006), this can lead to coral starvation (Erftemeijer et al., 2012; Flores et al., 2012; Junjie et al., 2014) and reduced calcification and growth (Edmunds and Davies, 1989; Flores

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et al., 2012; Humanes et al., 2017; Lirman et al., 2003; Miller et al., 2016; Moeller et al., 2016).

Corals exposed to heavy, chronic, or repeated sedimentation can be overwhelmed and unable to successfully rid themselves of sediment (Bak, 1978; Bessell-Browne et al., 2017a; Flores et al., 2012; Marszalek, 1981). When this occurs, corals – particularly those with mounding morphologies – begin to accumulate rejected sediment in "berms", or piles of sediment around the colony perimeter (Miller et al., 2016), making sediment removal even more difficult as the berm increases in height. With enough sedimentation, energetically costly sediment removal mechanisms become exhausted, and corals can become partially or completely buried, resulting in mortality (Lirman et al., 2003; Marszalek, 1981; Miller et al., 2016; Nugues and Roberts, 2003; Riegl, 1995). Mortality commonly occurs first under sediment berms that pile up at colony bases, producing a condition of partial mortality around the base in a "halo" pattern (Marszalek, 1981; Miller et al., 2016).

Sedimentation has also been shown to inhibit coral sexual reproduction in a number of ways (Jones et al., 2015), including by impairing spawning success (Ricardo et al., 2016), fertilization (Ricardo et al., 2015), settlement (Babcock et al., 2002; Ricardo et al., 2017), and recruitment (Moeller et al., 2016). Sediment may also directly remove available recruitment space by covering hard surfaces required for larval settlement (Babcock and Davies, 1991; Ricardo et al., 2017). Recruitment may still be reduced even if sediment is subsequently removed, likely due to the negative impacts of sediment on crustose coralline algae, a key settlement cue (Ricardo et al., 2017). For recently-settled coral recruits, sedimentation tolerance may be at least an order of magnitude lower than for adult corals (Fabricius, 2005), and even relatively low sedimentation rates (16.6 mg $\mathrm{cm^{-2}\,d^{-1}}$) can result in mortality (Moeller et al., 2016). Even sediment that is not deposited on the seabed, but that is moving through the system, is likely to abrade and kill newly-settled coral recruits and other benthic organisms, in addition to blocking photosynthetically active radiation (Storlazzi et al., 2015).

Impacts from sedimentation specifically due to dredging activities can be even more harmful to corals and reef habitat compared to other types of sedimentation. Due to the rapid escalation in sediment load created by sudden commencement of dredging, the typical behavioral, acclimatory, and adaptive responses (for example, selection for particular coral species or morphologies) that normally operate at sites exposed to naturally high sedimentation (Lasker, 1980; Sofonia and Anthony, 2008) may not be able to operate effectively. Moreover, in contrast to other kinds of sedimentation events, such as hurricanes, that generate sediment over hours to days, dredging can generate high sediment conditions for months to years, exceeding the energetic reserves of corals that might otherwise be able to survive acute impacts caused by storms (Flores et al., 2012; Jones et al., 2015; Riegl and Branch, 1995).

The type of sediment released by dredging activities can also be different from naturally occurring sediment (Jones et al., 2016). Dredging sediment is often more fine-grained than natural coarse sediment, and these fine particles can cause higher turbidity (Fourney and Figueiredo, 2017), can take longer to settle out of the water column, can be distributed further (Duclos et al., 2013), and are more harmful to corals (Duckworth et al., 2017; Jones et al., 2015; Nugues and Roberts, 2003; Weber et al., 2006). When deposited on the benthos, this fine sediment may also have an adhesive, clay-like texture that is more resistant to bioturbation and dissipation (Jones et al., 2015), and is more likely to become anoxic (Piniak, 2007; Weber et al., 2006). Dredging can also release sediment from deeper strata than might be disturbed by natural events, generating additional sediment not already existing in the system and with distinct mineralogies compared to those found in reef environments (Saussaye et al., 2017; Swart, 2016). Releasing this sediment may result in acute acidification and/or eutrophication, and, particularly in areas such as shipping channels or ports (Nayar et al., 2007), may also release unwanted contaminants (Eggleton and Thomas, 2004; Jones, 2011; Su et al., 2002), sediment-borne pathogens (Hodgson, 1990; Voss and Richardson, 2006; Weber et al., 2012), or related immune impairment agents. Exposure to dredging plumes has been correlated with a doubling in the prevalence of white syndromes in corals on the Great Barrier Reef (Pollock et al., 2014), suggesting that dredging can either release potential pathogens and/or decrease coral health and compromise immunity.

The Port of Miami shipping channel bisects the Florida Reef Tract, and is immediately surrounded by areas designated under the U.S. Endangered Species Act (ESA) as critical habitat (defined as any area containing the physical and biological features essential to survival) for ESA-listed staghorn and elkhorn corals, of which hundreds of colonies (of *Acropora cervicornis*) were documented in 2013 within 150 m of the channel on the inner reef alone (McCarthy and Spring, 2014). Additional reef coral species listed as threatened under the ESA (including *Orbicella annularis*, *O. faveolata*, *O. franksi*, and *Mycetophyllia ferox*) have also been documented in the area (Dial Cordy and Associates, 2014a). These coral reefs and coral habitat are also designated as Essential Fish Habitat under the Magnuson Stevens Fishery Conservation and Management Act for species managed under the spiny lobster, snapper-grouper, and coral fishery management plans (NOAA Fisheries Service, 2017).

Dredging adjacent to the Florida reef tract took place to widen and deepen the Port of Miami shipping channel between November 20, 2013 and March 16, 2015 (~16 months). Dredged materials, consisting of chopped rock mixed with water, were pumped from the dredge to a spider barge which, in turn, pumped the material into scows (also known as hopper barges). The process of dewatering and overflow of sediment-laden water from the hopper barge deposits fine particles of dredged material into the water column (Jones et al., 2016), which, around the Port of Miami, created sediment plumes with an extent up to ~228 $\mathrm{km^2}$ (Barnes et al., 2015). Ultimately, an estimated 4.2 million $\mathrm{m^3}$ of material was dredged via pipeline, backhoe, and clamshell dredges, destined for a permitted offshore disposal location 2.4 km ESE of the project site at a depth of 120–240 m (Ocean Disposal Database, 2016).

Many dredging projects have historically suffered from data-poor monitoring efforts to determine impacts to coral reefs (Erftemeijer et al., 2012). However, in the case of the Port of Miami, surrounding coral resources were extensively monitored by an environmental consultancy, Dial Cordy and Associates (DCA), on behalf of Great Lakes Dredge and Dock Company, the dredging contractors for the U.S. Army Corps of Engineers (USACE), and on behalf of the Port of Miami (Miami-Dade County). Although this monitoring program concluded that the effects of dredging were minimal and attributed most observed coral mortality to a concomitant regional coral disease outbreak (Dial Cordy and Associates, 2017), state and federal agencies report that dredging impacts were widespread, severe, and long-lasting (Florida Department of Environmental Protection, 2014; Miller et al., 2016; National Marine Fisheries Service, 2016). These conflicting reports prompt the need for a comprehensive analysis of monitoring data to evaluate the contribution of dredging to observed reef impacts. Moreover, the extensive data collected in situ provides a unique opportunity to evaluate whether measured impacts on the benthos are correlated with satellite observations of sediment plumes. Although not a substitute for robust, in situ monitoring, demonstrating such a link would validate the use of remote sensing techniques to monitor and predict dredging and coastal construction impacts to benthic communities where data maybe lacking or unavailable, or where independent data sources are needed (Fisher et al., 2015).

Here, we apply rigorous, data-driven, statistical methods to DCA's in situ monitoring data to determine whether impacts to reef corals and habitat occurred as a result of dredging operations at the Port of Miami. Specifically, we investigate 1) whether dredging activities impacted the quality and quantity of corals and coral habitat, 2) whether dredging-related impacts can be distinguished from other regional disturbances that occurred contemporaneously, such as bleaching and disease, and

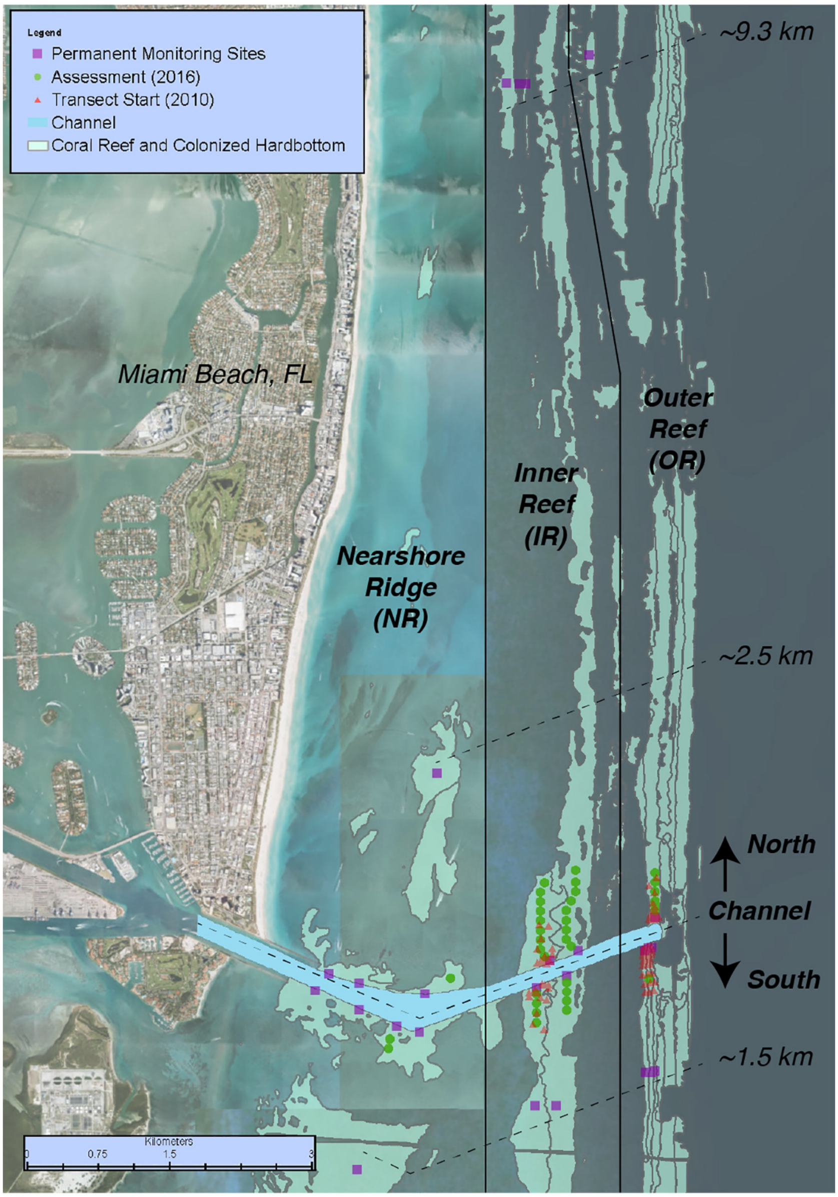

Fig. 1. Map of coral reef habitats and monitoring locations around the Port of Miami shipping channel. Permanent monitoring sites (n = 26) where data were collected throughout dredging are indicated by purple squares, while non-permanent transects used for coral density and/or sediment depth measurements at various distances from the channel are indicated by red triangles (2010; before dredging) and green circles (2016–2017; after dredging). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

3) whether biological responses measured in situ can be predicted by remote sensing of dredging sediment plumes. Finally, we extrapolate from these data to estimate total coral losses and the full spatial extent of impacts from the Port of Miami dredging.

2. Methods

2.1. *Study design and data provenance*

The Port of Miami shipping channel, where dredging occurred, cuts across three tracts of coral reef and colonized hardbottom (Fig. 1), referred to (from west to east) as the nearshore ridge (NR), inner reef (IR), and outer reef (OR; terminology follows Walker (2009) and Miller et al. (2016), but note that USACE and DCA reports often refer to the inner reef as the middle reef). On each of these reefs, DCA monitored regions to the north (N) and south (S) of the channel (n = 6 regions). Within each region, monitoring was conducted in areas both adjacent to and away from the channel (n = 12 areas, Fig. 1). The monitoring areas away from the channel ranged from intermediate distances of ~1.3–2.4 km (NNR, SNR, SIR, SOR) to farther distances of 9.4 km away (NIR, NOR); monitoring areas immediately adjacent to the channel were located within 18–48 m (median = 23 m). Each monitoring area contained 1–3 replicate sites (n = 26 permanent monitoring sites; Dial Cordy and Associates, 2014a, 2014b). At each site, DCA (1) deployed sediment traps to measure sediment accumulation throughout dredging; (2) recorded video transects to analyze changes in benthic cover using Coral Point Count with extensions (CPCe; Kohler and Gill, 2006); and (3) tagged individual corals to monitor their condition over time (details below).

In addition to these data collected at permanent monitoring sites, data were also collected at a range of distances from the edge of the channel out to several hundred meters away (Fig. 1) at various timepoints before (Dial Cordy and Associates, 2012) and ~2 years after dredging (Dial Cordy and Associates, 2017). The metrics collected at these non-permanent sites included the density of corals (recorded both before and after dredging) and the depth of sediment (only recorded ~2 years after dredging). In addition to the DCA datasets, we also analyzed the presence of a sediment plume from dredging operations as detected by satellite imagery (details below).

All data (except remote sensing) were collected by DCA on behalf of Great Lakes Dredge and Dock, the USACE, or the Port of Miami, and were provided to the Florida Department of Environmental Protection (FDEP) for compliance with FDEP Permit No. 0305721-001-BI. We subsequently obtained these data by public records requests under the Florida Sunshine Laws. Data were supplied to FDEP in various spreadsheets which we integrated and standardized for downstream statistical analysis using R code (all data and analysis code is available on Github (http://github.com/jrcunning/pom-dredge) and archived at Zenodo (Cunning, 2019)). We did not independently verify DCA data entry from field notes, photographs, or videos, and we did not repeat intermediate data processing steps (e.g., CPCe analysis). We did, however, correct instances of data entry error (e.g., dates and species identifications) when such errors were apparent from context, and these modifications were also made using R code for transparency and reproducibility.

Our analyses focused on sediment trap accumulation, benthic sediment cover, and tagged coral condition throughout the dredging project, as well as correlations among these measured impacts and remotely-sensed sediment plumes. We also analyzed lasting impacts on the density of scleractinians and the depth of sediment in reef habitat approximately 2 years post-dredging. In the sections below, we describe for each analyzed dataset: 1) how DCA collected and processed samples and/or data (with reference to DCA reports for further detail); and 2) how we conducted downstream statistical analyses to quantify impacts to corals and reef habitat.

2.2. Sediment plume detection

Plumes of sediment in the water column (Fig. 2A) were detected using satellite imagery following the methods of (Barnes et al., 2015). Coordinates corresponding to the 26 permanent monitoring sites were mapped onto satellite imagery to determine presence or absence of a sediment plume on each day for which data were available (dependent on weather conditions and image quality). For each monitoring region, a binomial generalized additive model was used to model the frequency of sediment plume presence over time during dredging operations. To estimate the spatial extent of dredging impacts on the reef, the presence of sediment plumes during the dredging period was quantified for pixels along three north-south transects centered over the NR (longitude = 80.115°W), IR (80.0997°W), and OR (80.0894°W), positioned at 250 m intervals from 15 km south of the channel to 15 km north ($n = 396$ pixels).

2.3. Sediment trap accumulation

Sediment trap accumulation was measured by DCA using three sediment traps (1″ inner diameter PVC pipe with 500 mL collection bottle) deployed at each permanent monitoring site on a continuous basis between 2013-10-15 and 2015-07-20, for intervals that varied among sites and ranged from 10 to 89 days ($n = 1287$ sediment samples in total). Each sediment sample was separated into coarse and fine components using a U.S. Standard #230 sieve (the cut-point between very fine sand and coarse silt; Wentworth, 1922), dried at 150 °F for ≥24 h, and weighed to the nearest 0.01 g (Dial Cordy and Associates, 2015a, 2015b). Masses were divided by the duration of trap deployment to calculate sediment accumulation rates in g day^{-1}. We analyzed only 'fine' trapped sediment, as this is more likely derived from dredging than natural processes. Due to disparities between sediment trap accumulation and benthic sediment deposition (Storlazzi et al., 2011), these data should be interpreted as sediment inputs that are either depositing or moving through the system.

To analyze these data, we used a Poisson generalized additive model to estimate the fine sediment trap accumulation rate in each monitoring area as a smooth function of time (using the midpoint of each sampling interval), weighted by trap deployment duration. To calculate total trap accumulation at each of the 12 monitoring areas during the dredging project, we summed the fitted daily accumulation rates for all days between 2013-11-20 (beginning of dredging) and 2015-03-23 (one week after dredging was completed).

2.4. Benthic sediment cover

The proportion of the benthos occupied by sediment was monitored by DCA using video transects recorded along three 20 m transects at each monitoring site between 2013-11-07 and 2016-08-20 ($n = 1772$ video transects; dates recorded vary among sites since data were collected at a site only when dredge operations were within 750 m). Between 27 and 80 still frames (median = 40) were extracted from each video transect and analyzed using CPCe with 10 points overlaid on each frame. Points overlaying tape, wand, or shadow were excluded, resulting in a range of 1–10 points analyzed per frame (median = 8). Points were classified into a range of benthic categories, of which we focus on "sand", which was used to indicate presence of sediment (Dial Cordy and Associates, 2015b). Two sites (HBN1 and HBN2) were omitted because they are considered a different habitat type (referred to as scattered coral/rock in sand) and/or were influenced by a "sand wave" during baseline surveys, reducing the frequency of data collection (Dial Cordy and Associates, 2014b).

To analyze these data, we used a binomial generalized additive mixed model to estimate sediment cover as a smooth function of time, with site and transect as random factors. Models were fitted only to portions of the time series with < 20-week gaps between data points.

2.5. Tagged coral condition

DCA observers tagged and monitored individual coral colonies throughout the project, recording a variety of condition codes that reflected coral health and/or impacts of sedimentation (Dial Cordy and Associates, 2015b). The complete collated dataset contained $n = 23{,}537$ observations of 650 tagged corals at 26 permanent monitoring sites between 2013-10-14 and 2016-08-20 (dates vary among sites since data were collected from a site only when dredge operations were within 750 m). We analyzed coral sediment burial as the occurrence of either "PBUR" (partial burial) or "BUR" (complete burial) condition codes over time using a binomial generalized additive mixed model for each monitoring area, with site, transect, and species as random factors.

To quantify partial mortality due to sedimentation, we analyzed occurrence of the "PM" condition code (defined by DCA as partial mortality specifically due to sedimentation; Dial Cordy and Associates, 2015a) using a binomial generalized linear mixed model (GLMM) with site, transect, and species as random factors. A colony was counted as having experienced partial mortality due to sedimentation if it was recorded with the "PM" condition code at any time during dredging operations. The same analysis was conducted for total mortality (condition code = "DEAD"); however, since some corals may have died due to a concomitant disease outbreak, we additionally analyzed total mortality for the subset of tagged coral species not observed with disease during monitoring (*Acropora cervicornis*, *Agaricia agaricites*, *Agaricia lamarcki*, *Madracis decactis*, *Mycetophyllia* spp., *Porites astreoides*, *Porites porites*, *Siderastrea siderea*, *Stephanocoenia intersepta*), which lowered statistical power but allowed us to isolate potential dredging impacts from disease-related mortality. Further estimates of total mortality are derived from changes in coral density (Section 2.8).

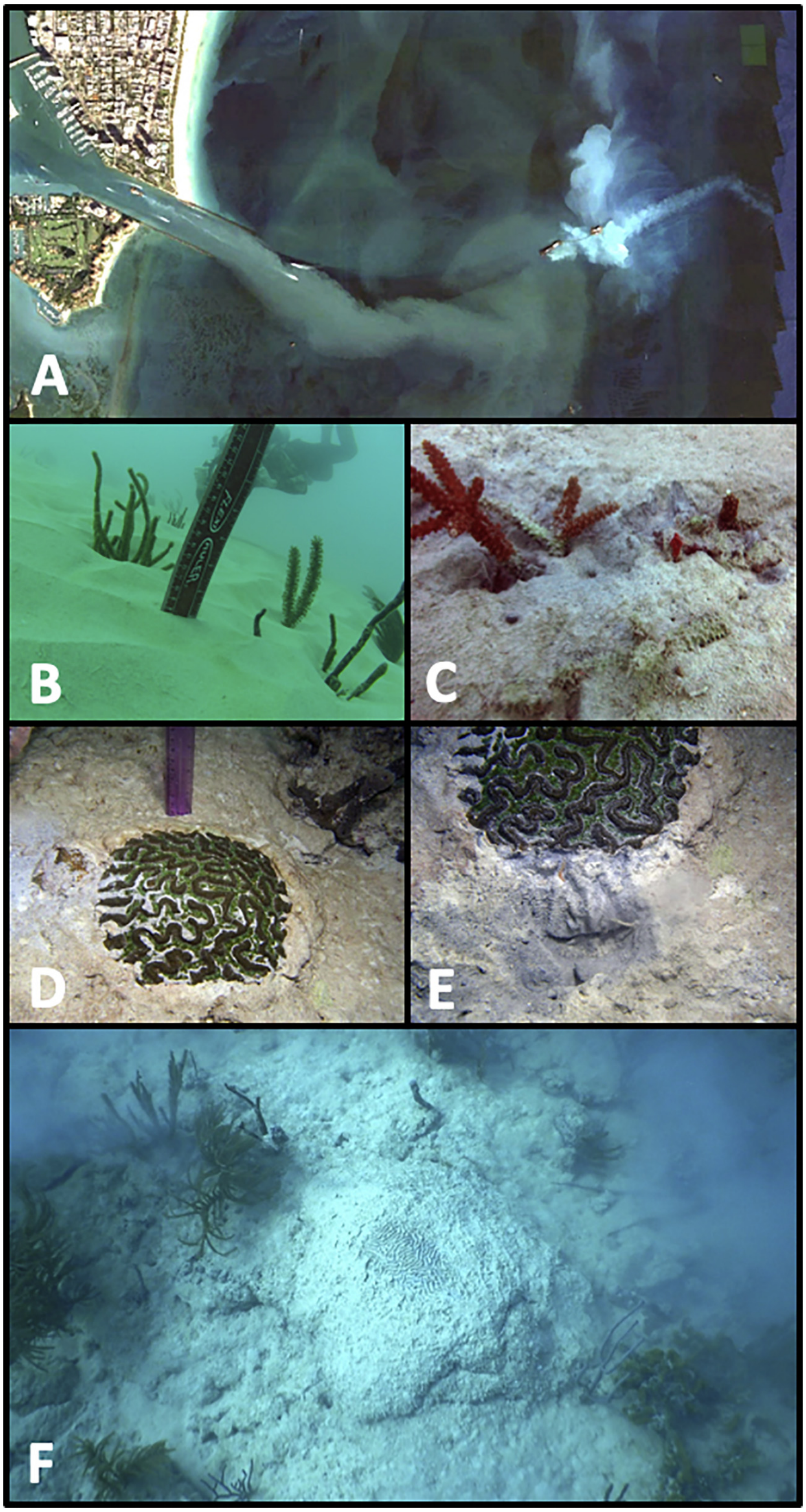

Fig. 2. Representative photographs of sediment plumes and benthic impacts. (A) Sediment plumes spread from dredging activity in the channel on 2013-12-30 (photo: Google Earth). (B) Sediment covers the benthos on the NIR reef < 250 m from the channel on 2015-01-30, with the tops of gorgonians protruding from ~7 cm of sediment (photo: R Silverstein). (C) A colony of *A. cervicornis* is partially buried in sediment 150 m from the channel on the NIR in September 2014 (photo: Coastal Systems International). (D) A *Colpophyllia natans* colony is partially buried with a berm of rejected sediment around its perimeter on 2014-07-22 (photo: (Florida Department of Environmental Protection, 2014)). (E) The same *C. natans* colony after sediment was removed from the base of the colony, revealing significant partial mortality. (F) Dead coral skeletons on the NIR reef < 150 m from the channel on 2015-06-23. Sediment was removed from the top of the large colony in the center to reveal skeletal structure. (Photo: R Silverstein).

2.6. Relationships among measurements

For each of the independent in situ measurements made at permanent monitoring areas (plume presence, fine sediment accumulation, benthic sediment cover, partial or complete coral burial, and partial coral mortality from sediment), we calculated aggregated metrics for each area and analyzed their pairwise correlations. Dredge plume presence was calculated as the proportion of days that a dredge plume was present (Fig. 3); sediment accumulation was calculated as the total accumulation of fine sediment in traps during the project (kg m^{-2}; Fig. 5); benthic sediment cover was calculated as the mean daily proportion of the benthos covered by sediment (Fig. 6); coral burial was calculated as the mean daily probability of coral burial (Fig. 7); coral mortality was calculated as the cumulative probability of partial

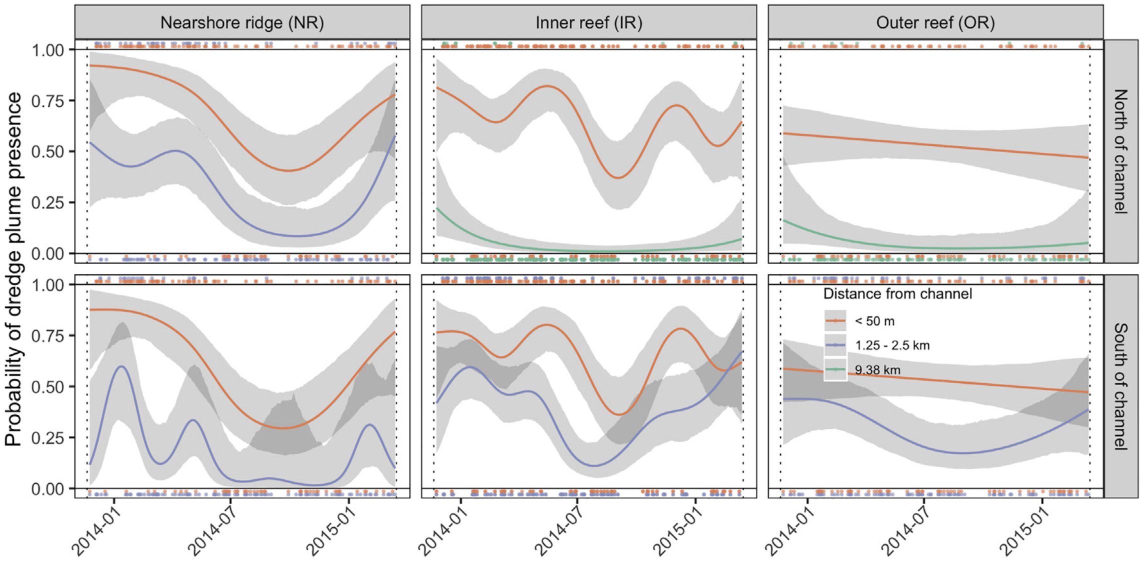

Fig. 3. Presence of a sediment plume as detected by satellite data throughout the dredging project. Smooth lines are GAM fits for each monitoring area (± 95% CI), colored according to distance from channel. Points in margins indicate the presence or absence of the dredge plume on a given date (data not available for all dates). Vertical dotted lines indicate the beginning (2013 − 11 − 20) and end (2015-03-16) of dredging operations.

mortality from sediment among tagged corals (Fig. 8). Metrics derived from time series (e.g., benthic sediment cover, coral burial) only include dates for which data were available from all monitoring areas. Pairwise linear regressions were performed for these aggregated metrics, and R^2 values calculated to determine the proportion of variability of the response explained by each predictor.

To estimate the magnitude of potential benthic impacts based on the sediment plume presence between 15 km south and 15 km north of the channel (Section 2.2), linear transformations using these regression models were made based on the proportion of days that a plume was detected at a given pixel.

2.7. Sediment depth

Sediment depth was measured by DCA ~2 years after dredging (between 2016-09-12 and 2017-05-30; Dial Cordy and Associates, 2017) at 1 m intervals along two perpendicular 50 m transects centered at increasing distances from the channel within the linear reef habitat (as defined by Walker, 2009). The distribution of sediment depth measurements was highly positively skewed, and therefore a quantile regression approach was taken. For each transect (containing $n = 51$ measurements), quantiles were computed to reflect the depth of sediment recorded in 1%, 10%, 25%, and 50% of measurements. These quantiles were then analyzed by linear regression with reef, direction, and log(distance from channel) as predictors.

2.8. Scleractinian abundance

To detect changes in scleractinian abundance, we analyzed the number of corals counted by DCA along belt transects at varying distances from the channel before dredging (in 2010 and 2013; Dial Cordy and Associates, 2014a, 2014b) and ~2 years after dredging (late 2016/early 2017; Dial Cordy and Associates, 2017). In 2010 and 2016–17, all corals ≥1 cm were recorded, while in 2013 only corals ≥3 cm were recorded. For some observations, diameter information was missing, so we assumed these corals were ≥3 cm. Across all of these timepoints, 11,166 scleractinians were counted along 482 transects, comprising 33 species, including ESA-listed *A. cervicornis*, *M. ferox*, *O. annularis*, *O. faveolata*, and *O. franksi*.

We analyzed the density of 'large' (defined as ≥3 cm) corals as a function of distance from channel in each monitoring area using a Poisson generalized linear mixed model, with timepoint (before or after dredging) as an additional fixed factor, and site, transect, and survey date as random factors. For the SIR, one data point was a highly influential outlier (Cook's distance = 1.34, while all other points < 0.12), and was removed from the analysis. The density of 'small' corals (< 3 cm) was analyzed in the same way. For each monitoring region, fitted values were used to test for differences in scleractinian density before vs. after dredging both adjacent to (20 m) and further away from (300 m) the channel.

Fitted values for declines in coral density (corals m^{-2}) at 1 m intervals moving away from the channel in each monitoring region (sufficient data available only for the IR and NOR) were multiplied by the total area of reef habitat at that distance within each region in order to estimate the total number of corals lost due to dredging activity. Area was calculated in ArcGIS as the aggregate sum of coral reef and colonized hardbottom habitat using mapping data from (Walker, 2009).

3. Results

3.1. Sediment plume detection

The frequency of sediment plume (e.g., Fig. 2A) detection by satellite was higher in the permanent monitoring areas closer to the channel (Fig. 3). In late 2013, after dredging commenced, plumes occurred with very high frequency (77–92% of days) near the channel on the NR and IR, and were also frequent on the OR (58% of days), as well as all monitoring areas within 1.25–2 km (33–50% of days; Fig. 3). In contrast, sediment plumes were detected on only 14–17% of days during the same time period 9.4 km away. In 2014 through the end of dredging in 2015, sediment plumes were almost never detected (3–4% of days) 9.4 km away, but still occurred with variable but high frequency at all intermediate distance monitoring areas (16–38% of days) and channelside areas (53–65% of days; Fig. 3).

3.2. Sediment trap accumulation rates

Fine sediment accumulation measured by sediment traps was highest at locations near the channel, and in particular, on the NR and IR (Fig. 4, 5). Fine sediment accumulation rates showed peaks in late 2013, spring 2014, and late 2014, and began to decline after dredging ended (Fig. 4). The highest rates of fine sediment accumulation were

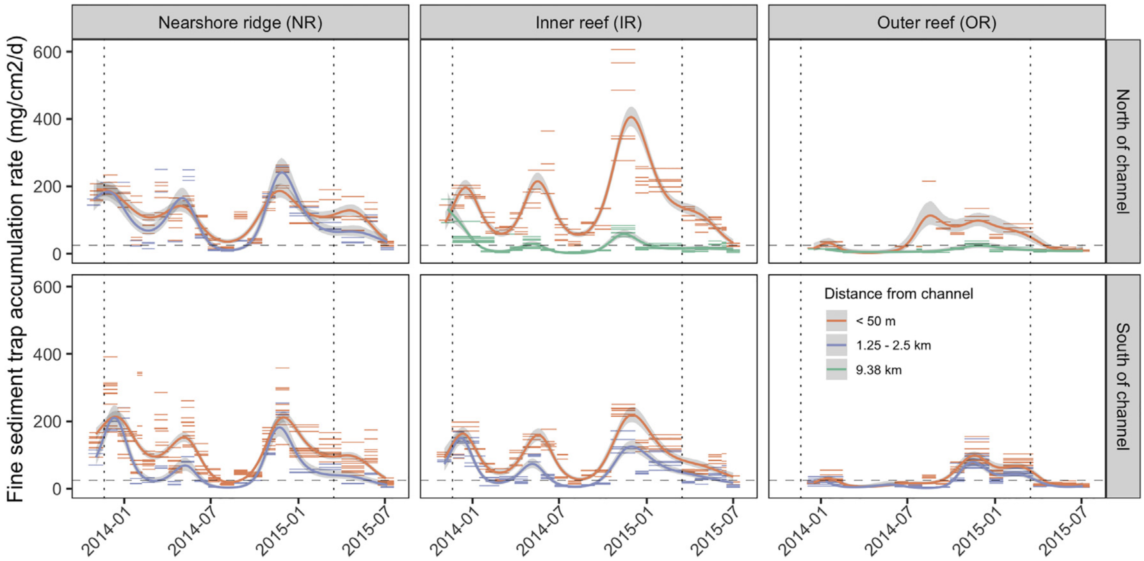

Fig. 4. Accumulation rates of fine sediment in sediment traps throughout the dredging project. Horizontal line segments indicate measured rates of fine sediment accumulation in each trap over each deployment period. Smooth lines are GAM fits for each monitoring area (± 95% CI), colored according to distance from channel. Vertical dotted lines indicate the beginning (2013-11-20) and end (2015-03-16) of dredging operations. The horizontal dashed line indicates a threshold of 25 mg cm^{-2} d^{-1}; sediment deposition rates exceeding this threshold over 30 days may cause severe stress leading to mortality (Nelson et al., 2016).

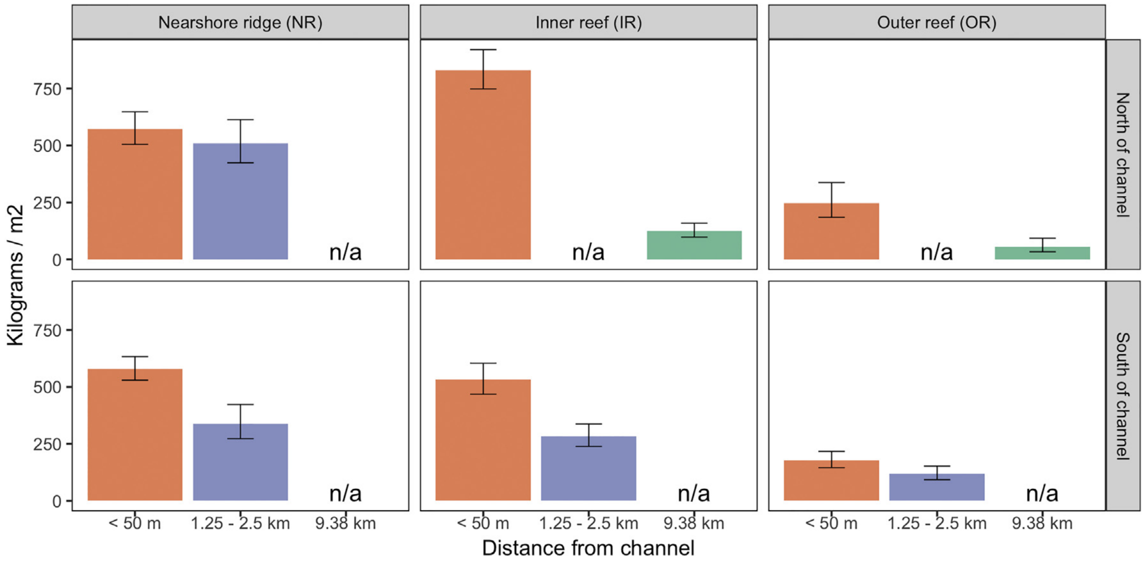

Fig. 5. Total amount of fine sediment accumulated in sediment traps in each monitoring area throughout dredging operations. Bars indicate the sum of fitted daily fine sediment accumulation rates between 2013-11-20 and 2015-03-16 (Fig. 4) for each monitoring area (± 95% CI). Bars are colored corresponding to distance from channel, and 'n/a' indicates areas that were not monitored.

recorded in late 2014 when the NIR channelside area received over 400 mg cm^{-2} d^{-1} on average, with one trap measuring 606 mg cm^{-2} d^{-1} over a 51-day period. During the same time period, traps 9.4 km away from the channel on the NIR measured 10 times less sediment (58.3 mg cm^{-2} d^{-1}). The total fine sediment accumulation over the entire dredging project (Fig. 5) was highest at the NIR channelside area (830 kg m^{-2}).

3.3. Benthic sediment cover

Sediment cover on the benthos (e.g., Fig. 2B) increased during dredging, with channelside areas naturally low in sediment cover (0–10%; IR and OR) becoming 50–90% covered in sediment for most of the duration of dredging (Figs. 6, S1). Intermediate-distance monitoring areas (1.25–2.5 km away) also experienced increases in sediment cover, peaking at 50–70% in late 2014. By contrast, areas located 9.4 km away were typically < 25% sediment, and never exceeded 50%. Mean daily percent sediment cover (± 1 s.d.) was 61.1 ± 16.0% near the channel, 35.8 ± 21.6% at intermediate distances, and 15.4 ± 8.5% at 9.4 km away. Sediment cover over time was temporally correlated with sediment trap accumulation rates for most areas (Fig. S2; median correlation coefficient = 0.51 with median lag time = 33 days), indicating that sediment deposition drove the observed increases in sediment cover.

3.4. Burial of corals

There was only one record of partial burial by sediment (out of 1211 observations of tagged corals, 0.08%) prior to the start of dredging. After dredging began, the probability of partial or complete coral burial

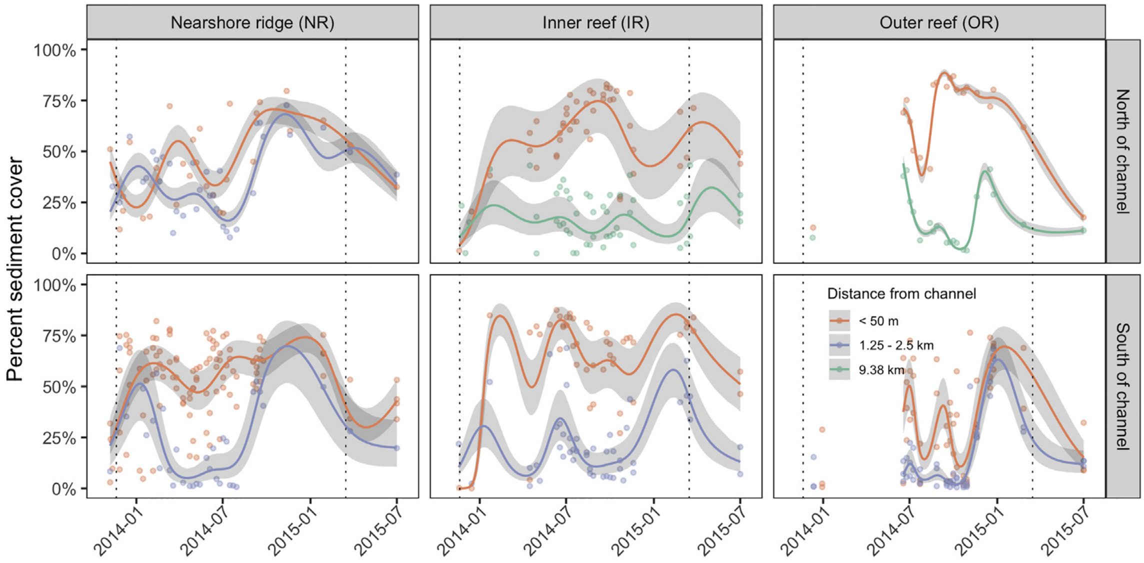

Fig. 6. Percent sediment cover at each monitoring area during dredging operations. Points indicate the mean percent sediment cover for each transect measured by CPCe analysis, and smooth lines show GAMM fits for each monitoring area (± 95% CI). Fitted lines are colored by distance from channel. Vertical dotted lines indicate the beginning (2013-11-20) and end (2015-03-16) of dredging operations.

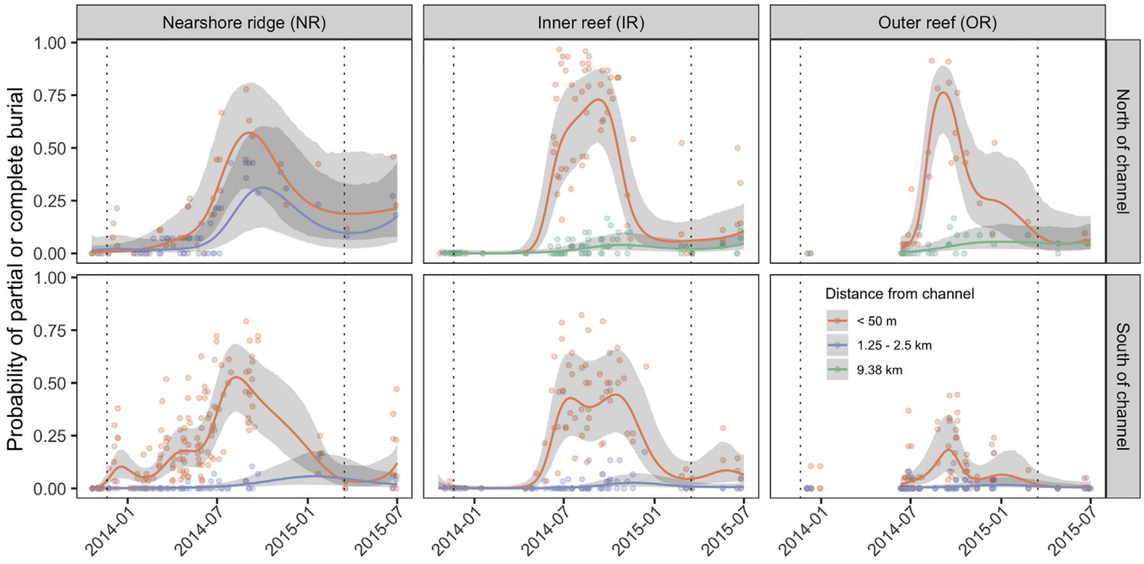

Fig. 7. Probability of partial or complete coral burial by sediment in each monitoring area during dredging operations. Points indicate the proportion of living tagged corals in each monitoring area observed on a given date as either partially or completely buried in sediment. Lines represent GAMM fits for each monitoring area (± 95% CI) colored by distance from channel. Vertical dotted lines indicate the beginning (2013-11-20) and end (2015-03-16) of dredging operations.

(e.g., Fig. 2C, D) rose sharply in areas, first on the NR in late 2013/early 2014, and then on the IR and OR in mid to late 2014 (Fig. 7). The probability of partial or complete burial reached 57–76% at all northern channelside areas by late August/early September 2014. At southern channelside areas, probabilities peaked at 44.5–52.7% on the NR and IR, but reached only 18.3% on the OR. At all intermediate and far distances, probabilities remained below 6%, with the exception of the NNR area (2.3 km away), which peaked at 31.2%.

3.5. Coral mortality

Partial mortality due to sedimentation (e.g., Fig. 2E) was frequently observed for tagged corals, especially in channelside areas, where the cumulative prevalence was 56.8–74.7% (except the SOR at 24.1%; Fig. 8). In addition to partial mortality due to sedimentation, tagged corals also experienced total mortality (e.g., Fig. 2F), although causes of total coral mortality were not necessarily known. To eliminate disease as a possible cause of death and isolate total mortality due to sedimentation, we analyzed data from only those coral species not observed with disease by DCA during the monitoring period. Among these non-disease-susceptible tagged corals, the probability of total mortality was 36% on the NIR adjacent to the channel, but only 7% at a distance of 9.4 km (Fig. S4). Other monitoring regions also showed trends of higher partial and total mortality in areas closer to the channel. Mortality of non-tagged corals is presented in Section 3.8.

3.6. Correlations among metrics in permanent monitoring areas

Measured impacts of dredging were highly correlated across the 12 monitoring areas (Fig. 9). Areas where sediment plumes were more frequently detected had higher rates of sediment trap accumulation, higher proportions of the benthos covered in sediment, higher

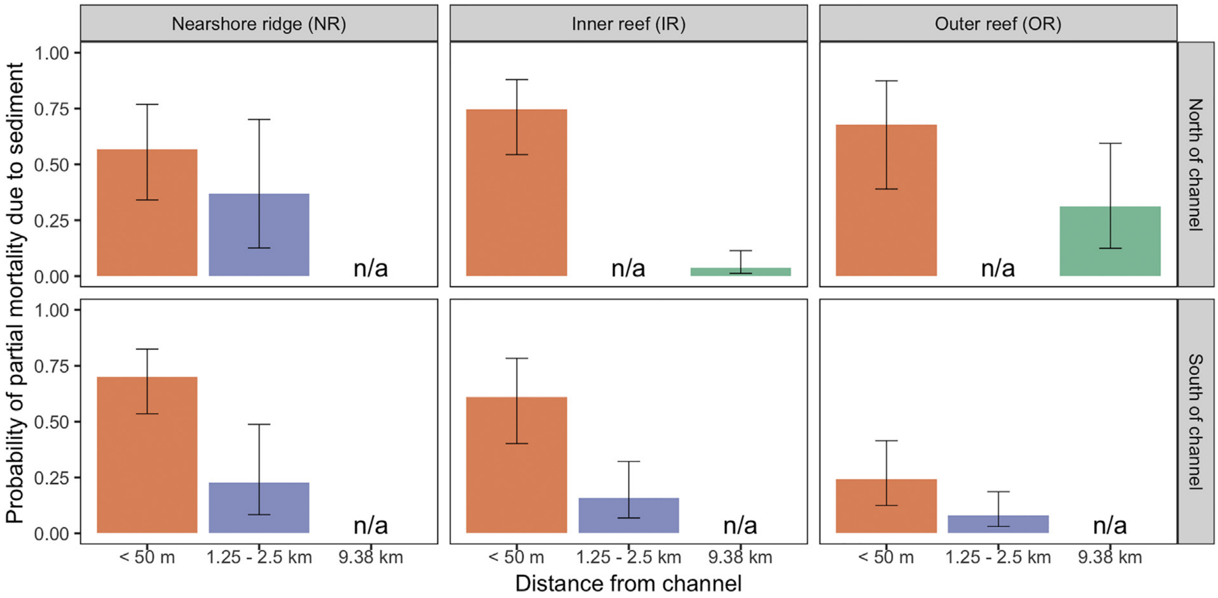

Fig. 8. Coral partial mortality due to sedimentation in each monitoring area throughout dredging operations. Bars indicate the predicted probability (± 95% CI) of tagged corals being observed with the condition code PM (partial mortality due to sedimentation) at any point through March 2015 (the final month of dredging operations). Bar colors correspond to distance from channel, and 'n/a' indicates areas that were not monitored.

probabilities of corals being partially or completely buried in sediment, and higher rates of coral partial mortality. Each of these metrics were highly positively correlated with each other (R^2 values between 0.51 and 0.90), and were statistically significant (all p-values < 0.01).

The frequency of sediment plume presence predicted an average of 62% of the spatial variability among all the other analyzed parameters, including sediment trap accumulation (52%), mean benthic sediment cover (73%), mean probability of coral burial (65%), and cumulative probability of partial mortality to corals (59%) across permanent monitoring sites. Using this predictive power, we estimated the likely magnitude and extent of these impacts based on the occurrence of sediment plumes in satellite imagery over each reef ranging from 15 km south to 15 km north of the channel (Fig. 10). The frequency of sediment plumes was ~55–70% near the channel, and decreased with distance from channel, with a skew toward higher frequency to the north. Based on these data, probable dredging-related impacts within 2–3 km include a ~4- to 9-fold increase in fine sediment input (an additional ~200–500 kg/m^2 depositing or moving through the system), a ~2- to 4-fold increase in benthic sediment (covering an additional 15–40% of the reef, on average, for 16 months), and a ~5- to 12-fold increase in coral partial mortality (affecting an additional ~20–55% of corals; Fig. 10). Sediment plumes further predict a doubling in partial mortality out to ~5 km south and ~9 km north of the channel, and some level of impact as far as ~10 km south and ~15 km north (the full range of plume detection).

3.7. *Alteration of habitat 2 years post-dredging*

Two years after dredging, significant portions of the benthos to the north of the channel were found to be covered in deep sediment (Fig. 11). One quarter of the NOR near the channel was covered by 10 cm or more of sediment, with pockets (~1% of the reef area) up to 30 cm deep. One quarter of the NIR was under 3 cm or more of sediment, with pockets over 10 cm deep. Based on sediment depth data for the NIR and NOR, the percent of area 20 m from the channel covered in at least 1 cm of sediment was 47% and 55%, respectively, while at 500 m, this decreased to 12% and 16%. While sediment depth data was not collected pre-dredging, three sites were surveyed for benthic sediment cover both before dredging and again two years later (Fig. S5): two channelside sites with very low sediment cover before dredging (0.2 and 1.2%) had significantly higher sediment cover two years post-dredging (19.4 and 34.0%; fold-changes of 14 and 156), while a third site located ~1 km away on the SIR had no change in sediment cover (21.8% to 21.5%).

3.8. *Impacts to coral populations*

Two years after dredging, the density of corals ≥3 cm at ~20 m from the channel was significantly reduced in all reef areas (Fig. 12); these reductions in coral density ranged from ~26–43% (SIR, SOR, NIR) up to 50–64% (NNR, SNR, NOR). Where sufficient data exist, declines in coral density lessened moving away from the channel, such that at 300 m away, changes in coral density were not statistically significant (except for a marginally significant ($p < 0.1$) decline of 43% on the NOR).

The density of small corals (1–2 cm diameter) near the channel was even more reduced two years after dredging (Fig. 13). On the NIR, SIR, and NOR, small corals declined by ~80% (NIR: 78.8%, $p < 0.01$; SIR: 80.1%, $p < 0.1$; NOR: 78.3%, but not significant). At 300 m away from the channel, small corals were still reduced by 72.3% on the NIR ($p < 0.1$), and, though not significant, were 67.3% lower on the SIR, and 62.8% lower on the NOR. The same analyses were also repeated with only coral species that were not observed with disease to isolate dredging-related impacts; these analyses produced very similar results (Figs. S6, S7). As additional evidence of the disproportionate loss of small corals in these areas, size-frequency distributions of the coral populations within 100 m of the channel showed a higher mean size of corals after dredging at NIR ($p < 0.0001$), SIR ($p < 0.0001$), and SOR ($p < 0.001$) relative to before dredging (Fig. S8).

By multiplying the declines in coral density by the total reef area in each region (where data exist, i.e., at three of six monitoring areas and only out to 500 m), we estimate that over half a million corals were killed during the dredging period (Table 1). In some cases, the distances in Table 1 are less (192 m on the SIR and 349 m on the NIR) because there was no estimated decline in density beyond that point (Fig. 11). The greatest losses (> 400,000 corals, ~71.5%) occurred in the small size class (< 3 cm) on the inner reef (Table 1).

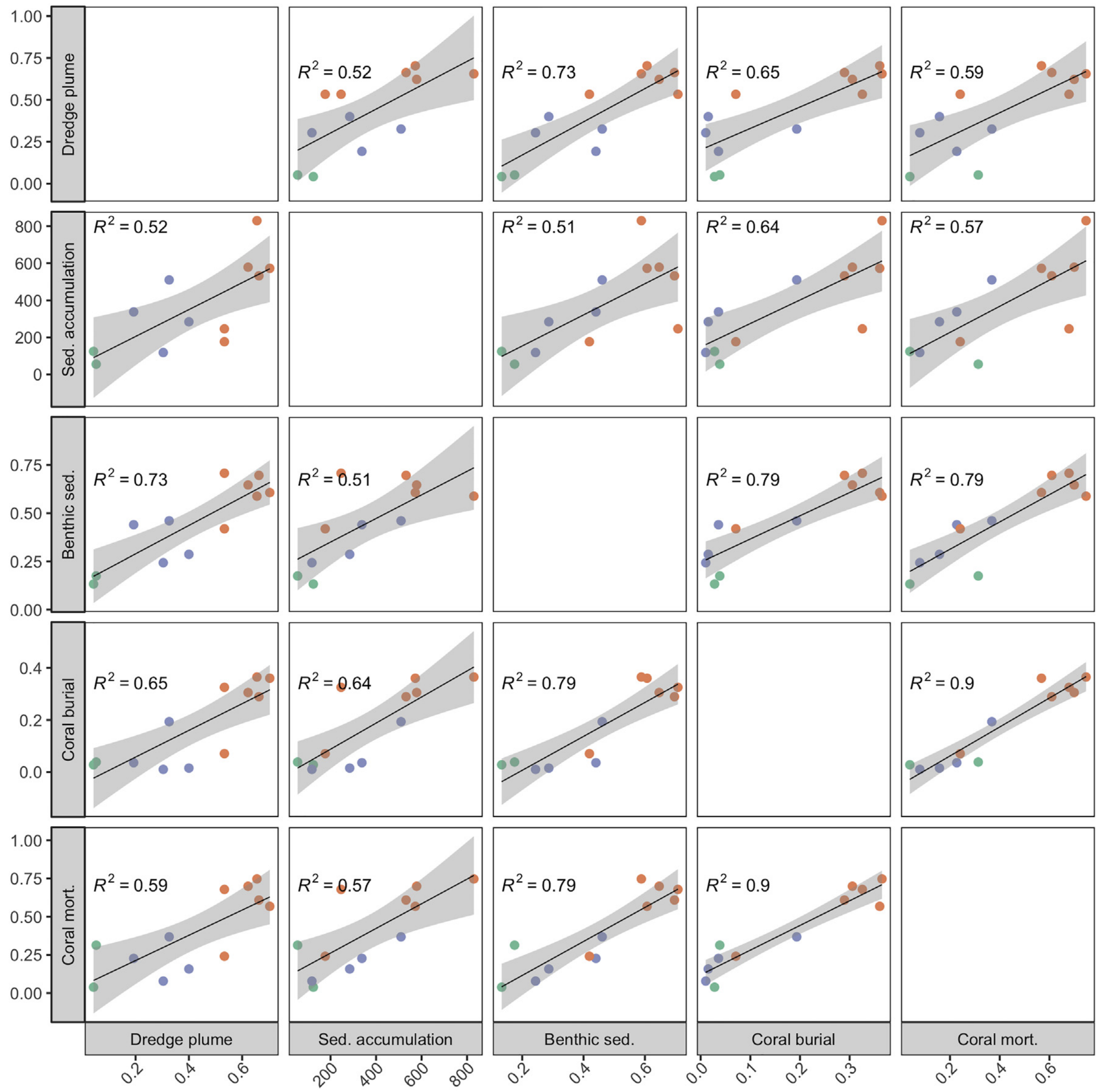

Fig. 9. Correlations among multiple metrics measured across monitoring areas. Points represent monitoring area ($n = 12$) colored by distance from channel (green ≤ 50 m; blue = 1.25–2.5 km; red = 9.38 km). Metrics are as follows: Dredge plume = proportion of days dredge plume present (Fig. 3); Sed. accumulation = total fine sediment trap accumulation during project (kg m^{-2}; Fig. 6); Benthic sed. = mean daily proportion benthic sediment cover during project (Fig. 5); Coral burial = mean daily probability of coral burial during project (Fig. 7); Coral mort. = probability of partial mortality due to sedimentation among tagged corals (Fig. 8). Metrics derives from time series (e.g., Benthic sed., Coral burial) only include dates for which data were available from all monitoring areas. Each panel displays a linear regression and the proportion of variation in the response explained by the predictor (R-squared). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

4. Discussion

4.1. Impacts from dredging at the Port of Miami

These analyses reveal significant impacts of dredging-related sedimentation on corals and the quality and quantity of coral habitat surrounding the Port of Miami shipping channel. Sediment plumes were detected with high frequency within several kilometers to the north and south of the channel (Figs. 3, 10), indicating that potential impacts from dredging were widespread and sustained over the 16-month project duration. Sediment trap data indicate that nearly ~830 kg of fine sediment per m^2 were input to reef habitats near the channel (Fig. 5), and that high sediment inputs occurred out to at least 2.5 km (Figs. 4, 5). Modeling work by Nelson et al. (2016) suggests that sedimentation rates > 25 mg cm^{-2} d^{-1} for 30 days (i.e., 7.5 kg m^{-2} over a single 30-day period) are likely to cause severe coral stress and mortality; over the 20 months of sediment trap monitoring, this threshold was exceeded, on average, 84.0% of the time at sites adjacent to the channel, 66.7% of the time at intermediate distance sites (1.25–2.5 km), and 15.2% of the time at the farthest sites (9.4 km; Fig. 4).

As some of this sediment deposited on the benthos, reef habitat initially low in sediment cover became 50–90% covered in sediment during dredging operations (Fig. 6). In addition to major impacts to existing benthic organisms (discussed below), the consequences of reef habitat being mostly buried for ~16 months likely include a significant reduction in larval recruitment. With at least two spawning and recruitment cycles (2014 and 2015) potentially affected by high sedimentation and suspended sediment, it is likely that area reefs experienced reduced fertilization, recruitment, and juvenile survivorship (e.g., Babcock et al., 2002; Fabricius, 2005; Moeller et al., 2016; Ricardo et al., 2015, 2016). Therefore, in addition to the effects documented here, dredging likely had detrimental, but as-yet-unquantified, effects on the ability of corals at these sites to undergo successful sexual reproduction. Indeed, elevated turbidity – even for a single month – has

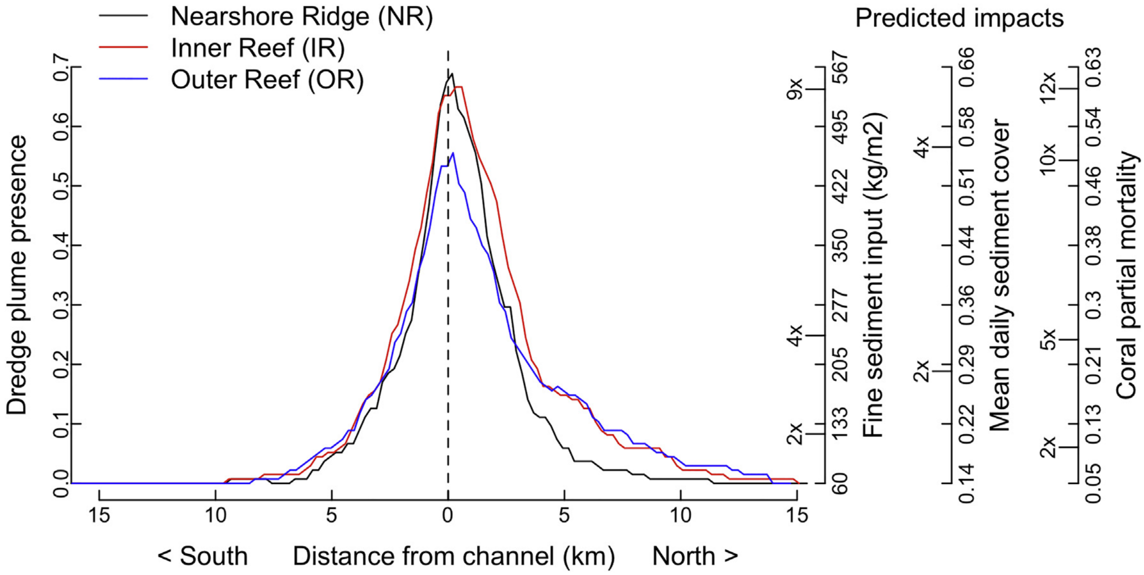

Fig. 10. Predicted magnitude and extent of dredging impacts based on dredge plume occurrence. Lines indicate the frequency of dredge plume presence throughout dredging operations based on satellite imagery along linear transects from 15 km north to 15 km south of the channel on the nearshore ridge, inner, and outer reefs. Axes on the right show predicted impacts based on linear transformations of dredge plume presence according to the models in Fig. 9. Additional ticks are added to transformed axes to indicate relative fold-changes ('2×', etc.) above baseline values in the absence of a plume. The baseline values at the bottom of the transformed axes correspond to the y-intercepts of the regressions in Fig. 9, i.e., the level of impact expected with a sediment plume is entirely absent.

been found to reduce coral recruitment from 80% to 10% (Fourney and Figueiredo, 2017). Surveys conducted on affected reefs near the Port of Miami dredging in 2014 found no coral recruits due to fine sediment covering the benthos (Florida Department of Environmental Protection, 2014).

As the benthos became covered in sediment, corals were also buried. By mid to late 2014, ~50–75% of tagged corals adjacent to the channel (except on the SOR) were partially or completely buried in sediment (Fig. 7). Consistent with these rates of burial, ~50–75% of tagged corals adjacent to the channel (except the SOR) also suffered partial mortality due to sedimentation (Fig. 8). In populations of colonial organisms, partial mortality (e.g., of large colonies) is as important as whole-colony mortality (e.g., of small colonies) in driving changes in total tissue biomass (Meesters et al., 1996), and in determining changes in population-level parameters like fecundity (Denley and Metaxas, 2016). Colonies are also less likely to recover from partial mortality in high sediment conditions (Meesters et al., 1992).

Whole-colony mortality was assessed in multiple ways: by direct observations of tagged corals, and by changes in total coral density before and after dredging. However, unlike the other metrics analyzed here, complete mortality has the potential to be confounded by regional bleaching and/or disease events that were occurring contemporaneously (Walton et al., 2018). Therefore, to distinguish dredging impacts from these regional disturbances, we analyzed (1) spatial patterns in proximity to dredging, (2) impacts to coral species not affected by disease, and (3) impacts to coral size classes with varying

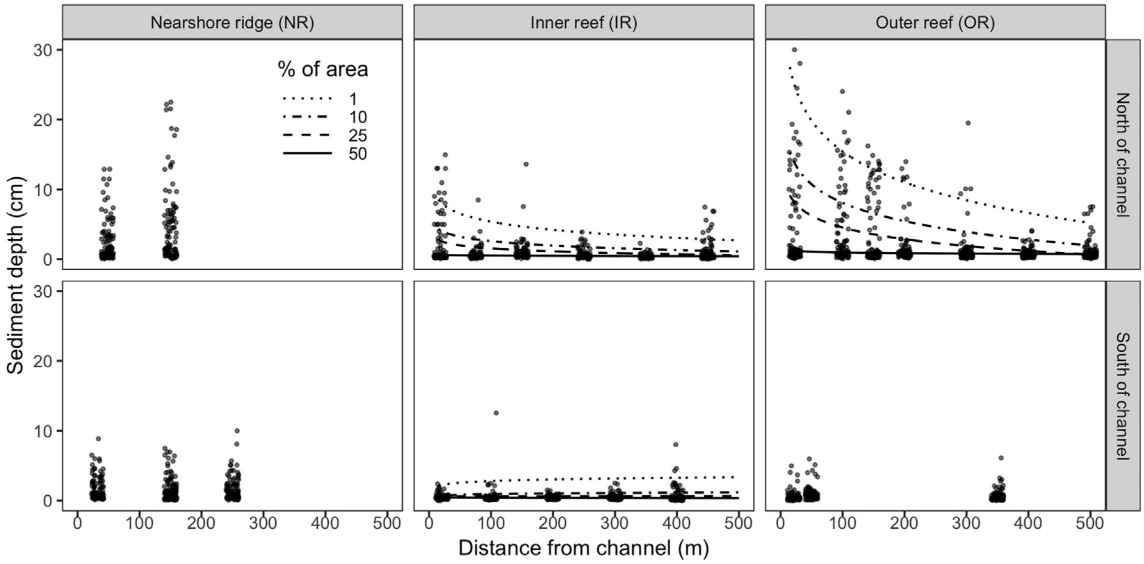

Fig. 11. Depth of sediment in relation to channel measured ~2 years after dredging operations. Dots represent individual sediment depth measurements along transects, jittered around the distance from the center of the transect to the channel in each monitoring region. Lines represent quantile regressions across transects as a function of log(distance from channel), indicating that at a given distance from the channel (on the x-axis), sediment with a particular minimum depth (on the y-axis) covered X% of the reef (where X is the quantile of the regression, e.g. 1%, 10%, etc.).

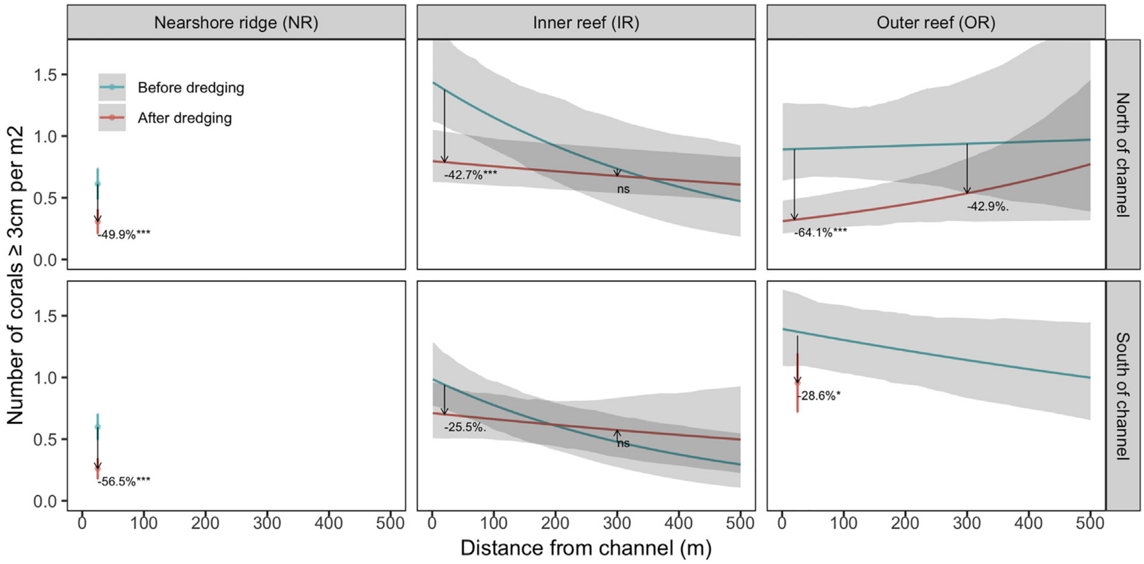

Fig. 12. Density of corals ≥ 3 cm within 500 m from the channel in each monitoring region before and after dredging operations. Data 'before' dredging were collected in 2009 and 2013, and data after dredging were collected in 2016–2017. GLMM fits are shown as lines for the regions and times in which data were collected at > 2 points beyond 50 m, and otherwise as points including only data from within 50 m of the channel. Shaded regions and error bars represent 95% CI's. Text annotations indicate the results of tests for differences between timepoints at 20 m and 300 m from the channel (*** = $p < 0.0001$; ** = $p < 0.001$; * = $p < 0.01$; . = $p < 0.1$; ns = not significantly different).

vulnerability to sediment burial. Among the tagged coral species not affected by disease during monitoring (Section 2.5), complete mortality was 5 × higher near the channel (36%) compared to 9 km away (7%) on the NIR (Fig. S4). Consistent with this high mortality near the channel, we also found 25–80% reductions in total coral density after dredging, and these declines lessened in severity moving away from the channel (Figs. 12, 13). This spatial pattern of coral loss suggests these impacts were caused directly by dredging or its interaction with other stressors (e.g., Bessell-Browne et al., 2017b), since bleaching or disease alone would not be expected to cause more severe mortality closer to the channel. Even coral species never observed with disease, when analyzed separately, showed the same decline in density and relationship with distance from the channel (Figs. S6, S7), providing additional evidence that dredging, not disease, was the driving factor. Furthermore, 1–2 cm corals, which could more easily be buried by sediment due to their small size, suffered disproportionate declines of up to 80% (which were also more severe closer to the channel), again implicating sediment burial as the cause. Finally, the observed declines in coral density around the Port of Miami far exceeded the regional estimate of 11.6% decline (2013–2016) as a result of bleaching and disease (Walton et al., 2018). This extensive coral mortality increased directly with proximity to the channel, affected even disease-resistant species, and disproportionately impacted small corals. Taken together, these findings cannot be explained by regional bleaching or disease, and instead indicate local disturbance in the channel as causing the majority of coral loss. While in theory, some other unidentified factor associated

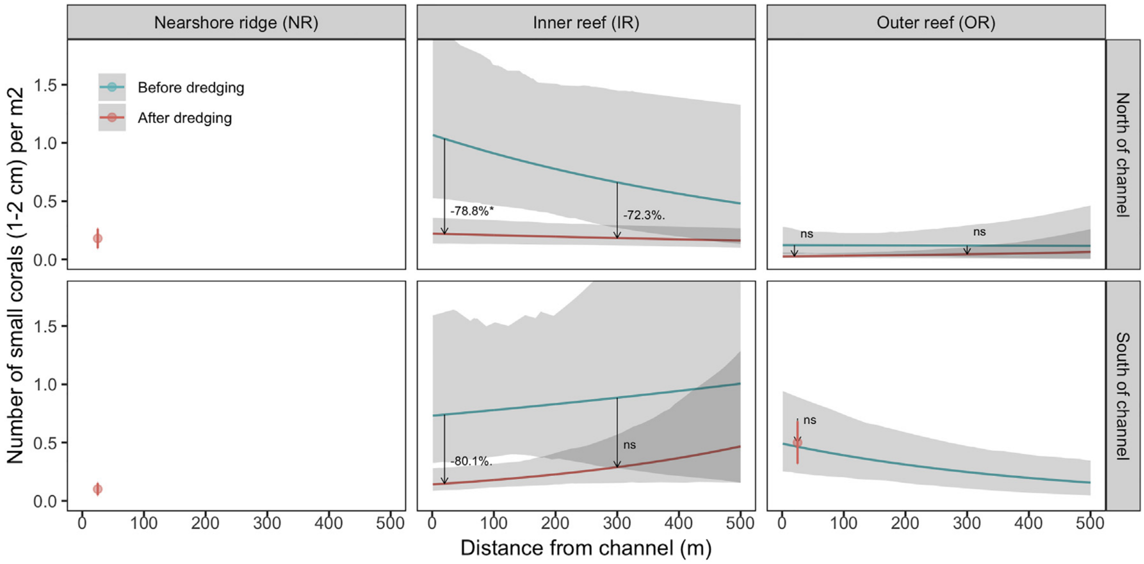

Fig. 13. Density of small corals (1–2 cm) within 500 m from the channel in each monitoring region before and after dredging operations. Data 'before' dredging were collected in 2010, and data after dredging were collected in 2016–2017. GLMM fits are shown as lines for the regions and times in which data were collected at > 2 points beyond 50 m. Shaded regions and error bars represent 95% CI's. Text annotations indicate the results of tests for differences between timepoints at 20 m and 300 m from the channel (*** = $p < 0.0001$; ** = $p < 0.001$; * = $p < 0.01$; . = $p < 0.1$; ns = not significantly different).

Table 1
Estimated coral loss two years after dredging within 500 m of the dredged channel. Losses were estimated by multiplying declines in density per m^2 (see Figs. 12, 13) by the reef area in each region at increasing distance from the channel out to a maximum distance of 500 m (the extent of data collection), or the distance at which density estimates before and after dredging intersected (whichever was lesser). Coral density data were only available for three of six reef regions. 'Out to (m)' and 'From area (km^2)' indicate the distance from channel, and total area, within which the estimate of coral loss took place.

Direction	Reef	Size class	Corals lost	Out to (m)	From area (km^2)
Northern	Inner	< 3 cm	203,249	500	0.86
Northern	Inner	≥ 3 cm	72,982	349	0.75
Southern	Inner	< 3 cm	197,917	500	1.51
Southern	Inner	≥ 3 cm	16,912	192	1.3
Northern	Outer	< 3 cm	11,082	500	1.87
Northern	Outer	≥ 3 cm	58,636	500	1.87
Total			**560,778**		

with the channel could have produced this channel effect, all lines of evidence point to dredging activity as the direct cause.

By multiplying the declines in coral density (where data exist) by the total area of coral habitat (Walker, 2009), these data indicate that at least half a million corals were lost within 500 m of the channel as a result of dredging activities (Table 1). Moreover, this may be a conservative estimate of total coral loss, because it only includes areas where sufficient data were collected (i.e., only three out of six reef regions, and only out to a distance of 500 m from the channel). Impacts likely occurred in the other three reef regions and at distances > 500 m, as our analyses below indicate. Consequently, the actual amount of coral lost as a result of dredging is likely to be much higher (millions of colonies).

In addition to these significant coral losses, surviving corals and portions of coral colonies likely also suffered sublethal impacts including growth reductions (confirmed empirically by Miller et al. (2016)), depletion of energy and lipid reserves, and decreased immunity associated with the physical challenges of sediment removal and stress from the eutrophication, acidification, and light attenuation caused by suspended sediment (Erftemeijer et al., 2012; Fourney and Figueiredo, 2017; Jones et al., 2015; Pollock et al., 2014; Riegl and Branch, 1995). Corals in the vicinity of the plume or that suffered direct sedimentation also may have been more susceptible to the regional disease outbreak (Pollock et al., 2014; Stoddart et al., 2019; Voss and Richardson, 2006), and this could be investigated directly in future research.

Our combined analysis of a number of independent physical and biological metrics reveals highly significant positive correlations between sediment plume presence, sediment trap accumulation, areal sediment cover, coral burial, and coral partial mortality. This indicates that, despite their independent nature, each of these metrics is a good predictor of the others across the spatial extent of monitoring (Fig. 9). Indeed, correlations between these metrics are not surprising, because they reflect a causal sequence of events linking dredging activity to coral mortality, in which dredge plumes spread from the site of origin, sediment is deposited on the benthos, and this sediment buries and kills corals. Nevertheless, explicitly quantifying these relationships is important because we show that 62% of the variability in sediment impacts to reef corals and habitat measured in situ can be explained by the occurrence of remotely-sensed sediment plumes. While this relationship will vary from project to project, it suggests that when calibrated against an appropriate set of in situ monitoring data (such as was done here), the footprint and magnitude of impact of dredging operations can be estimated based on satellite observations.

At the Port of Miami, we used the predictive power of these remotely-sensed sediment plumes to extrapolate potential impacts beyond the range of in situ data collection, greatly expanding our ability to assess impacts over spatial scales that are much larger than the original design of the monitoring program. Based on the occurrence of sediment plumes spanning ~25 km of the Florida Reef Tract to the north and south of the channel (Fig. 10), we can predict that dredging operations input hundreds of kilograms of fine sediment per m^2 and buried > 25% of the reef as far as 3 km away, doubled coral partial mortality from sediment 5–10 km away, and had potential impacts as far as 15 km from the dredged channel.

Dredging-related impacts were not only widespread, but also long-lasting, as indicated by sediment cover and depth measurements taken ~2 years after dredging. Although comparable sediment cover data prior to dredging exist for only three sites (making it difficult to quantify changes comprehensively), these data show that two years later, sediment cover was unchanged at one site 1.25 km away, but remained elevated ten- to a hundred-fold at two sites near the channel (Fig. S5). Sediment depth was also not measured prior to dredging, but the very low sediment cover measured on the inner reef (0–1%; Fig. S5) suggests that depths at these locations must have also been near zero. Two years after dredging, > 1 cm of sediment covered ~50% of the NIR and NOR near the channel, but only ~12–16% of the reef 500 m away (Fig. 10); this spatial pattern suggests that the deeper sediment near the channel resulted from dredging. These persistent sediment deposits are particularly detrimental as these areas are designated as critical habitat for threatened *Acropora* corals. Even a "thin veneer" of sediment may impair settlement in some corals (Ricardo et al., 2017), and standing sediment deeper than 1 cm is considered to render an area non-functional as recruitable coral habitat (National Marine Fisheries Service, 2016). In the *Acropora* Recovery Plan, habitat of suitable quality is defined as hard substrate free of sediment cover as required for larval recruitment, with recruitment failure identified as a key conservation challenge (National Marine Fisheries Service, 2015). Therefore, the impacts documented here represent a significant and long-lasting loss of living resources and critical habitat for these ESA-listed corals, as well as other reef species.

4.2. Recommendations for future dredging near coral reefs

These findings are particularly relevant in light of future dredging operations planned in the vicinity of coral reefs. Before the Port of Miami dredging project commenced, environmental assessments predicted "temporary", "localized", and "insignificant" sedimentation impacts to corals and coral habitat, stretching only as far as 150 m from the channel (Florida Department of Environmental Protection, 2012; National Marine Fisheries Service, 2011; U.S. Army Corps of Engineers, 2004). Impacts were not expected to exceed the permitted 7.07 acres of coral habitat impact (by direct removal of habitat for channel expansion; Florida Department of Environmental Protection, 2012). In contrast, we show sedimentation impacts were directly observed at least 2.5 km from the channel (Figs. 3–8), with predicted impacts (based on sediment plume occurrence) as far as 10–15 km away (Fig. 10). The impacts described here show that the project exceeded the pre-construction predictions in terms of: (1) geographic area, by one to two orders of magnitude; (2) severity, with hundreds of thousands to millions of corals either partially or totally killed, including those listed as threatened on the Endangered Species Act; and (3) permanence, lasting at least two years post-dredging. These impacts show that the measures put in place to ensure only temporary and insignificant impacts (Florida Department of Environmental Protection, 2012; National Marine Fisheries Service, 2011; U.S. Army Corps of Engineers, 2004) were insufficient to protect corals and coral habitat.

To avoid similar outcomes, future dredging projects should extend their predicted areas of impact and related monitoring and mitigation. For example, "intermediate" distance locations (1.25–2.5 km from the channel) were originally designed as "control" locations for near-channel sites (Florida Department of Environmental Protection, 2012), but were located within the area of dredging impact. A lack of sufficient pre-dredging monitoring data also hinders an even more comprehensive assessment of coral habitat impacts. While we did not directly analyze turbidity measurements, turbidity limits established at 29 NTUs above background (Florida Department of Environmental Protection, 2012) were only reported to have been exceeded on very few occasions, yet severe impacts to surrounding coral reefs still occurred. Based on these findings, we support Fourney and Figueiredo (2017) in recommending a reduction of US-EPA allowable turbidity from 29 NTU above background to < 7 NTU near coral reefs. We also echo other authors' recommendations of a moratorium on all sediment-releasing activities before, during, and after coral spawning periods, which also coincide with times of year when thermal stress is expected to occur (Bessell-Browne et al., 2017a; Miller et al., 2016; Moeller et al., 2016).

Coral reefs in Florida and elsewhere are under increasing threats from a range of global threats, particularly climate change, that are not easily within the scope of local management actions. Given these declines, additional attention must be paid to protecting the corals that remain and preventing avoidable impacts from human activities, such as dredging that, unlike climate change, can be managed at the local level.

5. Conclusions

This report describes a spatially explicit statistical approach to estimate impacts to coral health and reef resources as a result of dredging at the Port of Miami from 2013 to 2015. Multiple, independent datasets, ranging from remotely-sensed dredge plumes to benthic sediment cover to partial mortality of tagged corals, all show strong increases with proximity to the dredging site, and are also highly correlated with one another. Taken together, these approaches indicate that local dredging-related sedimentation, not regional disturbances such as disease or bleaching, was the cause of these observed impacts. The extensive data collected by DCA allowed us to establish, for the first time, direct, quantitative links between remotely-sensed sediment plumes and in situ benthic impacts, revealing potential effects of the Port of Miami dredging along a 25 km segment of the northern Florida Reef Tract (from 10 km south of the channel to 15 km north). These dredging activities resulted in a 10- to 100-fold increase in sediment cover on nearby reefs, and a likely loss of over a million corals from affected areas in the vicinity. The geographic scope, longevity, and severity of these impacts far exceeded pre-dredging predictions, indicating a pressing need to re-evaluate environmental thresholds, monitoring, mitigation, adaptive management, and enforcement to avoid similar harm to corals and coral habitat as a result of dredging activities in the future.

Acknowledgments

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Declaration of competing interest

Miami Waterkeeper was a co-plaintiff in an Endangered Species Act litigation regarding impacts to coral reefs resulting from the 2013–15 dredging activities at the Port of Miami. This litigation concluded in August 2018. At time of submission (December 2018) Miami Waterkeeper is a co-plaintiff in another environmental litigation regarding dredging at Port Everglades (Ft. Lauderdale, FL). Author R. Silverstein is currently the Executive Director and Waterkeeper of Miami Waterkeeper.

Appendix A. Supplementary information

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2019.05.027.

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