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coral reef paper

A quantitative assessment of the status of benthic communities on US Atlantic coral reefs using a novel standardized approach

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ADVANCES IN ECOSYSTEM-SCALE CORAL REEF VISUAL SURVEYS

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ABSTRACT.—As coral reefs decline globally, the need for an objective approach to quantify the status and trends of corals has become increasingly important. Empirical data on predisturbance conditions are rare, and integrating data from multiple and disparate survey designs and methods can be analytically challenging. Our goal was to conduct a holistic, data-driven evaluation of the status of corals and benthic communities in US Atlantic coral reef jurisdictions: Florida, Flower Garden Banks, Puerto Rico, and the US Virgin Islands. A quantitative approach based upon standardized data was used to compare the change in multiple indicators of coral condition (hard coral, macroalgae, and crustose coralline algae cover, coral density, and old mortality) from historic to current conditions in each geographic region. For each indicator, historic, reference baseline conditions from long-term monitoring data or literature data were first identified, reviewed, and classified on a categorical scale from Very Good to Critical by regional experts to account for condition changes that pre-dated current monitoring data. A reference-centering approach then allowed for categorization of statistical changes from historic to current conditions on the same scale to produce results that could be communicated to a broad audience. Our findings show continued declines for multiple indicators in all regions except Flower Garden Banks, illustrate particularly dire declines from regions that had been impacted by Stony Coral Tissue Loss Disease at the most recent monitoring included in this study, and demonstrate the increasingly critical need for effective coral reef conservation.

Ecosystem status assessments play an important role in the conservation of the world's coral reefs. Global stressors, including increasing ocean temperature and acidification, and local stressors, such as overfishing and poor water quality, have all contributed to a decrease in live coral and reef accretion globally (Hoegh-Guldberg et al. 2007, Perry et al. 2013). Recently, an emergent coral disease in Florida and the Caribbean known as Stony Coral Tissue Loss Disease (SCTLD) has accelerated decline on many reefs (Walton et al. 2018, Brandt et al. 2021). The continued degradation of coral reef ecosystems has increased the need to clearly communicate their current condition to the public and policy makers, prompting the need to assess and succinctly report current reef conditions more frequently. Effective reporting for policy makers and managers must be clearly communicated and comparable to benchmarks that allow evaluation of their efficacy. This underlies the urgency with which managers and policy makers need to understand ecosystem status so they can implement science-based conservation actions (Fletcher et al. 2014). Furthermore, the efficacy of management actions can be more easily evaluated by demonstrating progress over time when compared to a standardized status benchmark at a reference time point (Messer et al. 1991, Carruthers et al. 2013).

Evaluating the status of an ecosystem is complex (Fletcher et al. 2014) and relies on key indicators that not only represent a healthy, functional ecosystem (Landres et al. 1988, Lindenmayer 1999), but are also sensitive enough to accurately detect change between reference baselines and current conditions (Messer et al. 1991, Walz 2000). Ecosystem status assessments typically rely on qualitative expert opinion to select condition datasets, indicators, sites, and/or metric scoring to determine change from idealized reference conditions to current conditions (McField and Kramer 2007, Fletcher et al. 2014, Montenero et al. 2021, Santavy et al. 2022). An approach that engages local expertise is particularly helpful to steps that are inherently qualitative (e.g., identification of data, indicator selection) and can increase the efficacy of conservation efforts (Pretty 1995, Freebairn and King 2003). However, qualitative approaches introduce bias, such as by shifting baselines of users' perceptions that can skew historical knowledge of the natural world (Pauly 1995, Knowlton and Jackson 2008). Inclusion of historical records can help overcome the problem of shifting baselines (Moore et al. 1999, McClenachan 2009, McClenachan et al. 2017). Application of a quantitative approach for analyses and scoring should minimize bias to produce results that are more objective.

Defining quantitative reference baselines to detect change can be limited by a lack of monitoring data that are not only representative of current and predisturbance conditions, but are also comparable and representative across the spatial extent of the study. Although evidence of human impacts to coral reefs dates back to the 18th century (McClenachan et al. 2017), limited quantitative data exist on the state of coral reefs before the 1970s, well after the onset of anthropogenic impacts (Cramer et al. 2012, Roff and Mumby 2012). In lieu of limited temporal baselines, space-for-time comparisons have been used for coral reef fish, where spatial reference locations are available that represent an "ideal" baseline—removed from direct human impacts or unfished areas (Friedlander and DeMartini 2002, Sandin et al. 2008, Grove et al. 2023). However, coral declines are not only attributed to local stressors, but also to regional stressors, such as coral disease (Alvarez-Filip et al. 2022), and global climate stressors. No remote or "ideal" baseline is immune to global climate impacts, such as

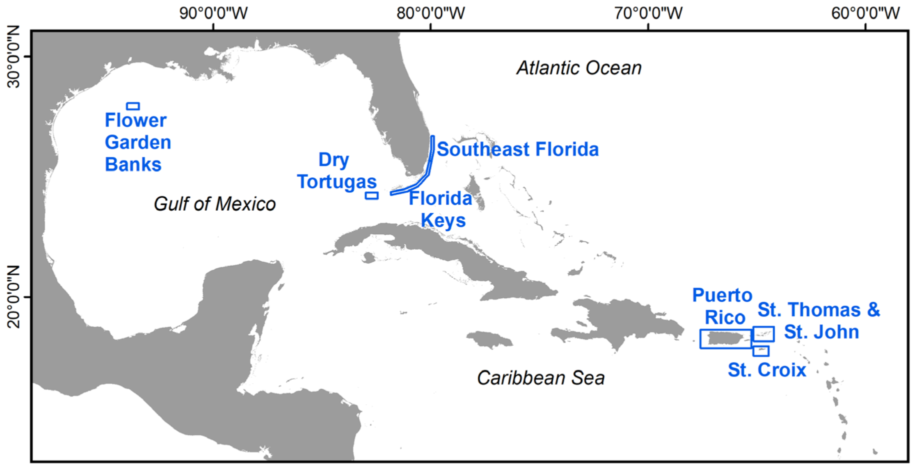

Figure 1. Coral reef jurisdictions in the US Atlantic surveyed by NOAA's National Coral Reef Monitoring Program (NCRMP). The Florida jurisdiction includes Southeast Florida, the Florida Keys, and the Dry Tortugas regions. The US Virgin Islands jurisdiction includes St. Thomas and St. John region, and St. Croix region.

ocean warming (Bruno and Valdivia 2016), which is why the most objective baseline is the best-available historical data that exists for a given region.

Multiple programs have been established since the 1990s and 2000s to monitor the condition of coral reefs in the US Atlantic (inclusive of the US Caribbean and Gulf of Mexico) states and territories (Fig. 1, Online Table S1). Although these programs have been invaluable to coral reef management, comparing data between different programs and across geographic regions has been challenging due to variable methodologies, such as survey design (e.g., fixed sites or stratified random), survey years, and metrics (Online Table S1). A new evaluation approach was needed to incorporate multiple disparate benthic datasets in the US Atlantic into larger ecosystem status assessments.

Here, we describe a repeatable, standardized approach to assess the current status of coral reef benthic communities in the US Atlantic and to quantify change over time relative to a historic baseline. This primarily quantitative approach is broadly applicable to multiple coral reef regions and is sufficiently robust to incorporate regionally variable coral monitoring programs, limited historic data, and regionally-specific qualitative input.

Methods

Overview.—The overall approach was primarily quantitative, informed by regionally specific qualitative input. First, potential indicators were selected to represent population and community level change for corals in each region. Next, the most recent available data for each indicator were used to represent current conditions. Then, historic reference conditions (i.e., baseline values) for each indicator were identified per region according to review criteria. For each indicator, standardized values for the reference conditions were statistically compared to standardized values for current conditions; a statistically significant difference indicated a change

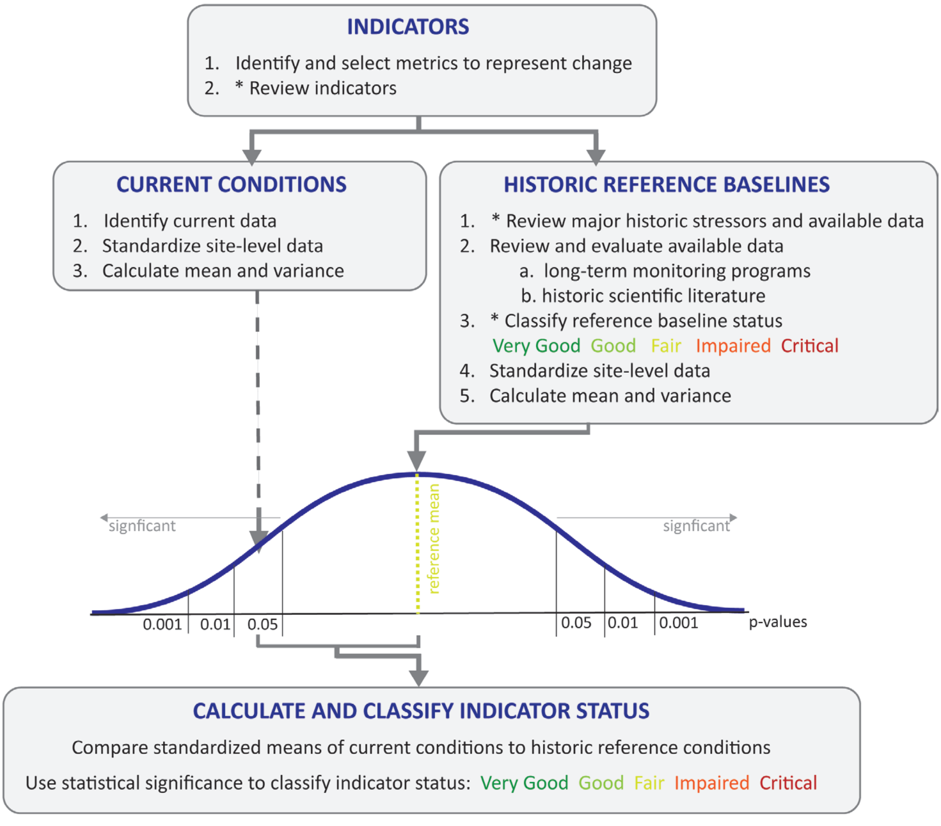

Figure 2. The process workflow in which indicators were selected, current conditions were calculated and standardized, and historic reference baselines were identified, reviewed, categorized, calculated, and standardized. A Student's *t*-test was used to test for significant differences between the standardized values for the reference conditions and current conditions. If the difference was significant, the degree of significance (*P*-value) indicated the categorization of the current conditions. An asterisk (*) indicates where local experts for each geographic region provided qualitative input. This example shows a reference baseline of Fair, and a significant difference between reference and current values categorizes the current condition as Impaired (Table 2).

in the indicator status over time. Expert-driven input was primarily incorporated in each region to (1) provide feedback on indicators selected to represent ecosystem status and trends, (2) identify available baseline data and categorize the condition that baseline data represent relative to the temporal timing of regional stressors, and (3) provide feedback on quantitative results relative to expert opinion of ecosystem status. Panels of local experts included stakeholders from federal, state, and territorial resource management agencies, universities, and nongovernmental organizations specific to each region. Each of these steps is detailed below.

Indicators.—Indicators were selected to represent coral reef benthic communities in the US Atlantic (Figs. 1 and 2). Indicator selections were based on (1) ecological value to represent coral reef benthic community status and trends, (2) availability of data to represent current conditions (Online Table S3), (3) availability of data to represent the best available historical reference baselines (Online Table S1), and (4) vetting by local coral reef experts specific to each region (Fig. 2). Of the benthic metrics initially considered for inclusion (Online Table S3), five were ultimately used for the assessment: cover (%) of corals, crustose coralline algae (CCA), and macroalgae, density of coral colonies (≥4 cm colony size; corals × m^{-2}), and percentage of old mortality on coral colonies. Old mortality is considered a proxy for loss of reproductive biomass within coral populations and defined as the percentage of nonliving coral colony surface area, or exposed skeletal structure, that could be eroded or colonized by organisms that are not easily removed (e.g., turf algae). For the coral density and old mortality indicators, reference and current condition values were calculated from a subset of scleractinian coral species specific to each region (Online Table S4). Species were considered to be of high ecological value (e.g., reef-building, listed as threatened in the US Endangered Species Act; Federal Register 2014) and were selected with input from local experts. Although survey methods for benthic cover differ by monitoring program, studies have shown that no significant difference exists in cover values from methods included here (e.g., digital or point-intercept collection methods in either line or belt transects; Nadon and Stirling 2006, Jokiel et al. 2015). Other metrics of population and community status from monitoring programs were not included as indicators due to survey design limitations (i.e., survey timing was independent of episodic disturbances, such as disease and bleaching, and recent mortality).

Current Conditions.—Current conditions were primarily represented by data from the National Oceanic and Atmospheric Administration's (NOAA) National Coral Reef Monitoring Program (NCRMP), which started in 2013, uses survey methods consistent between geographic regions, and has a statistically robust sample size (Fig. 2, Online Table S1). All five benthic indicators were represented in NCRMP data (Online Table S2). In regions where the expert panel recommended inclusion of non-NCRMP data and indicator data were available, local long-term monitoring (LTM) data also supplemented NCRMP data to represent current conditions.

For each indicator, the mean of surveyed primary sample units (PSUs; i.e., sampled sites) within each stratum h (Table 3) was calculated as

$$\bar{y}_h = \frac{1}{n_h} \sum_j y_{hj} \qquad \text{Eq. 1}$$

where $\bar{y}_h$ is the stratum-level mean, n_h is the stratum number of sampled PSUs, and y_{hj} is the site-level observation (Smith et al. 2011). Next, indicator mean density $\bar{y}_{st}$ for the NCRMP survey domain was calculated as the sum of the weighted means of each stratum:

$$\bar{y}_{st} = \sum_h w_h \bar{y}_h \qquad \text{Eq. 2}$$

where w_h is the proportion of stratum area within the sampling domain. The stratum weighting factor was calculated as

$$w_h = \frac{N_h}{\sum_h N_h} \qquad \text{Eq. 3}$$

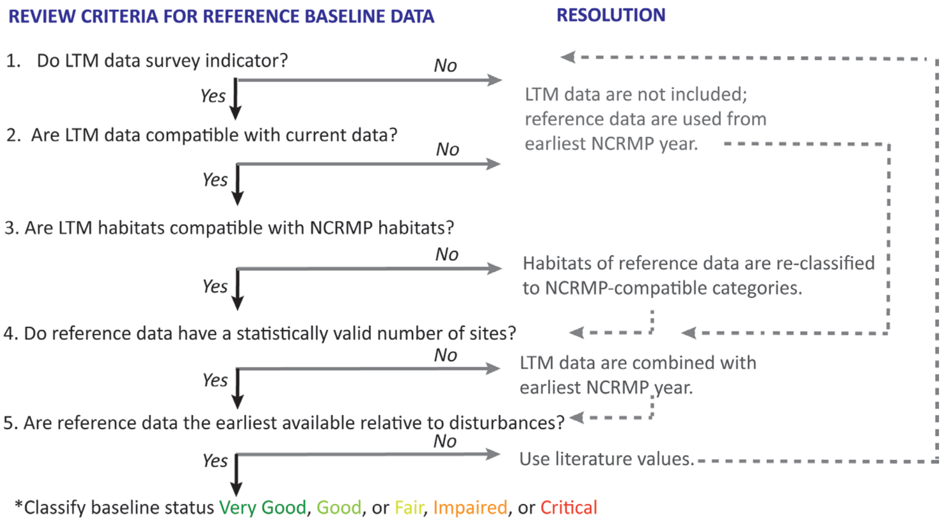

Figure 3. Review criteria for evaluating data from long-term monitoring (LTM) programs and historic scientific literature for potential inclusion into the historic reference baseline. At each step, if data did not meet review criteria, alternate resolutions are described (in gray). At any step, the baseline reference review process can lead to termination of the dataset for inclusion. If no baseline reference dataset met all criteria, the indicator was not scored.

where N_h is the total possible number of PSUs in stratum h. Associated stratum-level and domain-level variances were computed following Lohr (2010).

A standardized approach was used to categorize each indicator in each region. Separately for each indicator, PSU observations y_{hj} of NCRMP data representing current conditions were standardized by scaling to the reference mean value for a given stratum.

$$\tilde{y}_{hj} = y_{hj} - \bar{y}_{h,ref} \qquad \text{Eq. 4}$$

The reference-centered site-level observations $\tilde{y}_{hj}$ were then used to calculate standardized stratum-level means and standardized domain estimates with Eqs. 1–3, substituting $\tilde{y}_{hj}$ for y_{hj}. Note that Eq. 4 shifts the mean of the distribution of y_{hj} observations relative to the reference mean but does not affect the sample variance of the distribution.

Reference Baselines.—Multiple datasets from local LTM programs were reviewed for potential applicability to represent reference baselines in each respective region (Figs. 2 and 3, Online Table S1). Review criteria for potential reference data were as follows (Fig. 3):

Indicators were surveyed by LTM programs.

1. Data in each respective region were compatible with the NCRMP sampling program (i.e., values for current conditions). To test the comparability of data between LTM programs and NCRMP (i.e., current conditions) for each region, pairwise *t*-tests with Bonferroni correction of indicator yearly means were conducted between the NCRMP data and all LTM data for contemporaneous sampling years at the jurisdictional or regional spatial scale (Fig. 3).

Significant results indicated incompatible indicator values between programs, and those LTM data were not subsequently included. LTM program data were from sampling programs with a statistically rigorous number of sites in most regions (Online Table S1).

2. Data included a statistically valid number of sites for the included years and habitat types. To add additional statistical power when the number of survey sites were limited, LTM data were combined with early year(s) of NCRMP data to represent historic baselines (Table 1, Online Tables S1 and S2).
3. Survey site habitat classifications were comparable with habitat data for current condition sites (i.e., NCRMP habitat classifications). Site-level data were classified by habitat type and scored separately to acknowledge ecological differences in habitats.
4. Data represented a time period considered by the expert panel to be less affected by anthropogenic pressures, such as rising ocean temperature and land-based sources of pollution. In many regions, the expert panel considered the early years of many of the LTM programs to represent reef communities prior to high impact disturbance events. However, for regions where regional expert panels considered that LTM efforts did not precede major disturbance events (e.g., the emergence of white band disease or pre-*Diadema antillarum* die off), reference baseline data values were aggregated from available historic scientific literature. When all reference data review criteria were met by regional coral LTM program data, these LTM data were used as all or part of reference data to represent baseline values (Fig. 3).
5. Reference baselines were categorized from Very Good to Critical with input from regional experts (Figs. 2 and 3). This qualitative information incorporated local knowledge on the magnitude of impacts from stressors prior to earliest available data per indicator and accounted for the varying years of historic baseline condition per indicator and region. The mean and variance of site level reference values were standardized as described previously (Eqs. 1–4).

Calculation and Categorization of Current Status.—A standardized approach was used to categorize each indicator in each region. For evaluating current conditions with respect to the reference, the null hypothesis is $y_{current} = \bar{y}_{ref}$, i.e., $\overset{\cong}{\bar{y}} = 0$. A *t*-test was used to determine whether the standardized mean $\overset{\cong}{\bar{y}}$ was different from zero (Fig. 2). A scoring rubric based on *P*-values was applied, for which the standardized reference value score was assigned as the mean value. The reference condition categorized by the expert panel as either Very Good, Good, Fair, Impaired, or Critical was considered the mean value. *P*-values > 0.05 indicated no detectable change from the mean (Table 2, Fig. 2). The other categorical scores were distributed between the remaining *P*-values based on rank (Table 2, Fig. 2). For the detrimental indicators of old mortality and macroalgae, an inverse scale was used to assign the score (i.e., increases in either indicator had lower scores and decreases had higher scores). All analyses were performed in R (v3.6.1; R Core Team 2019) using the custom R packages ncrmp.benthic.analysis (Groves and Viehman 2023) and ncrmp.benthics.statusreport (Groves 2019).

Table 1. Datasets used for each indicator (or group of indicators) in each geographic jurisdiction, region, and habitat type specific to region to represent current conditions at the time of the study and to represent historic reference baselines. Literature data are detailed in Table S5.

Region	Indicator(s)	Habitat type	Data used for current conditions	Data used for reference baselines
Flower Garden Banks				
- - -	Coral cover	- - -	NCRMP 2015, 2018; FGB 2015–2017	FGB LTM 1992–2013
- - -	Macroalgae cover	- - -	- - -	- - -
- - -	CCA cover	- - -	NCRMP 2015, 2018; FGB 2015–2017	NCRMP 2013
- - -	Coral density, old mortality	- - -	NCRMP 2015, 2018; FGB 2017	NCRMP 2013
Puerto Rico				
- - -	Coral cover, macroalgae cover, CCA cover, coral density, old mortality	- - -	NCRMP 2016	NCRMP 2014
US Virgin Islands				
St. Thomas and St. John	Coral cover, macroalgae cover, CCA cover	High relief reef	NCRMP 2015, 2017	TCRMP, NPS 1999–2005
St. Thomas and St. John	Coral cover, macroalgae cover, CCA cover	Low relief reef	NCRMP 2015, 2017	TCRMP, NPS 1999–2005
St. Thomas and St. John	Coral density, old mortality	High relief reef	NCRMP 2015, 2017	NCRMP 2013
St. Thomas and St. John	Coral density, old mortality	Low relief reef	NCRMP 2015, 2017	NCRMP 2013
St. Croix	Coral cover, macroalgae cover, CCA cover	High relief reef	NCRMP 2017	TCRMP, NPS 1999–2005
St. Croix	Coral cover, macroalgae cover, CCA cover	Low relief reef	NCRMP 2017	NCRMP 2015
St. Croix	Coral density, old mortality	High relief reef	NCRMP 2017	NCRMP 2015
St. Croix	Coral density, old mortality	Low relief reef	NCRMP 2017	NCRMP 2015
Florida				
Southeast Florida	Coral cover	- - -	NCRMP 2016, 2018	Literature data 1979–1992
Southeast Florida	Macroalgae cover	- - -	NCRMP 2016, 2018	SE-CREMP 2003–2005
Southeast Florida	CCA cover	- - -	NCRMP 2016, 2018	NCRMP 2014
Southeast Florida	Density	- - -	NCRMP, DRM 2016–2018	DRM 2005–2007
Southeast Florida	Old mortality	- - -	- - -	- - -
Florida Keys	Coral cover	Patch reef	NCRMP 2018	Literature data 1974–1999; CREMP 1996–1999
Florida Keys	Coral cover	Bank reef	NCRMP 2016, 2018	Literature data 1974–1999; CREMP 1996–1999
Florida Keys	Macroalgae cover	Patch reef	NCRMP 2018	CREMP 1996–1999
Florida Keys	Macroalgae cover	Bank reef	NCRMP 2016, 2018	CREMP 1996–1999
Florida Keys	CCA cover	- - -	NCRMP 2016, 2018	NCRMP 2014
Florida Keys	Coral density, old mortality	- - -	NCRMP, DRM 2016–2018	DRM 2005–2007
Dry Tortugas	Coral cover	Mid-high relief reef	NCRMP 2016, 2018	Literature data 1975–1999; CREMP 1996–1999
Dry Tortugas	Coral cover	Low relief reef	NCRMP 2016, 2018	Literature data 1975–1999; CREMP 1996–1999
Dry Tortugas	Macroalgae cover, CCA cover	- - -	NCRMP 2016, 2018	NCRMP 2014
Dry Tortugas	Coral density, old mortality	- - -	NCRMP, DRM 2016–2018	DRM 2007, 2009

Table 2. Example scoring rubric where the reference baseline value has been categorized by local experts to be in Fair condition where an increase in indicator is a positive reef attribute. The indicator score is based on the statistical comparison of the standardized reference baseline value (mean) to the standardized status value (deviation from the mean).

Standardized status domain	*P*-value	Score
>0	$P < 0.01$	Very Good
>0	$P < 0.05$	Good
=0	$P > 0.05$, not significant	Fair
<0	$P < 0.05$	Impaired
<0	$P < 0.01$	Critical

Results

Based on the reference review criteria framework (Fig. 3), sources for reference values varied between indicators and between regions, and included LTM data (Florida, Flower Garden Bank and US Virgin Islands), historic survey data from peer-reviewed scientific literature (Florida; Online Table S5), and domain estimates from the earliest year of NCRMP sampling (all jurisdictions; Table 1). Results specific to each region are detailed below.

Flower Garden Banks (FGB).—Coral cover on the Flower Garden Banks has consistently remained over 55% since monitoring began in 1989 (Gittings et al. 1992, Aronson et al. 2005, Johnston et al. 2018), despite a severe bleaching event on both the East and West Flower Garden Banks in September 2016 (Johnston et al. 2018). Local experts classified reference baselines as Very Good for coral cover, coral density, and old mortality, and Good for CCA cover (Table 4). Reference values for coral cover and CCA cover used LTM data from 2009 to 2013, when the Flower Garden Banks National Marine Sanctuary began leading monitoring efforts and changed methods for measuring CCA cover, in combination with 2013 NCRMP data. When FGB monitoring began in 1989, the mean cover of macroalgae was low, less than 5% (Gittings et al. 1992), and has since risen to about 30%, where it has remained

Table 3. Consolidated habitat classification for current strata in US Virgin Islands and Florida.

Regions	Consolidated habitat classification	NCRMP habitats
US Virgin Islands		
St. Thomas and St. John; St. Croix	High relief reefs	Aggregate reef, bedrock, patch reef, unclassified hard bottom
St. Thomas and St. John; St. Croix	Low relief reefs	Pavement, scattered coral and rock in sand
Florida		
Southeast Florida	Hardbottom	Deep ridge complex, linear inner reef, linear middle reef, linear outer reef, near shore reef
Florida Keys	Patch reefs	Inshore patch reef, mid channel patch reef, offshore patch reef
Florida Keys	Bank reefs	Fore reef deep low-relief, fore reef mid-channel linear relief, fore reef shallow linear reef, high-relief reef (spur and groove)
Dry Tortugas	Mid-high relief reefs	Continuous high relief reef, continuous mid-relief reef, isolated high-relief reef
Dry Tortugas	Low relief reefs	Continuous low-relief reef, isolated low-relief reef, spur and groove low-relief reef

Table 4. Flower Garden Banks reference baselines were calculated and standardized for each indicator. The status of each reference baseline was categorized using local expert input. Standardized reference values were compared to standardized current condition values using a *t*-test, and the *P*-value result was used to categorize the indicator score.

Indicator	Reference baseline value and category	Current condition	*P*-value	Indicator score
Coral cover (%)	56.8, Very good	59.9	$P > 0.05$	Very Good
Macroalgae cover (%)	- - -	- - -	- - -	- - -
CCA cover (%)	3.3, Good	2.7	$P > 0.05$	Good
Coral density (corals m^{-2})	5.9, Very good	5.2	$P > 0.05$	Very Good
Old mortality (%)	4.8, Very good	5.5	$P > 0.05$	Very Good

relatively consistent since 2009 (Johnston et al. 2018). However, because these LTM data for macroalgae did not meet the historic data review criteria for a statistically significant sample size ($n = 2$; Fig. 3), and the panel was unable to agree on a categorization of status for more recent data, this indicator was not scored. Coral density has remained high over time at FGBNMS, at over 5 colonies m^{-2} (Johnston et al. 2018; Online Table S1). The coral density reference was based on NCRMP 2013 data, as FGBNMS LTM did not include coral density as a metric until 2015 (Online Table S1).

There were no significant differences ($P > 0.05$) between standardized reference baselines and current values for all indicators (Table 4). Therefore, the indicator scores for coral cover ($t = 1.99$; df = 70, $P > 0.05$), coral density ($t = 2.03$, df = 33, $P > 0.05$), and old mortality ($t = 2.03$, df = 33, $P > 0.05$) remained categorized as Very Good, and CCA as Good ($t = 1.99$, df = 70, $P > 0.05$).

Puerto Rico.—In Puerto Rico, LTM data did not meet review criteria for compatibility with contemporaneous NCRMP values for any common indicator metrics (Fig. 3, Online Table S1; pairwise t-test with Bonferroni correction, $t = -0.28$, df = 3, $P < 0.05$) due to multiple differences in survey designs between programs. Because review criteria were not met for inclusion of LTM data, NCRMP historic reference values were limited to NCRMP 2014 data, and current values were limited to NCRMP 2016 data (Tables 1, Online Table S2).

In Puerto Rico, a 2005 bleaching and disease event was estimated to have caused a 40%–60% decrease in coral cover (Eakin et al. 2010), with the reef building corals in the *Orbicella* species complex experiencing some of the highest levels of bleaching abundance and prevalence across the archipelago (García-Sais et al. 2008). Reference baselines for all indicators in Puerto Rico were categorized as Fair by local experts (Table 5). Mean macroalgae cover increased from 22.1% in 2014 to 23.7% in 2016, and coral cover in 2016 was 5.9%, a slight decline (0.5%) from 2014 (Table 5). There were no significant differences between standardized current and historic reference values for coral cover, macroalgae cover, CCA cover, and old mortality; therefore, the status of these indicators remained categorized as Fair ($t = 1.97$, df = 220, $P > 0.05$, respectively). There was a significant difference between standardized coral density and historic reference coral density; therefore, the status of coral density declined from Fair to Impaired ($t = 1.98$, df = 88, $P < 0.05$; Table 5).

US Virgin Islands (USVI).—Reference baselines for cover-based indicators used USVI LTM data from 1999 to 2005 (Table 1). These data and years were identified by local experts based on two major bleaching events in the USVI, in 2005 (Eakin et al.

Table 5. Puerto Rico reference baselines were calculated and standardized for each indicator. The status of each reference baseline was categorized using local expert input. Standardized reference baseline values were compared to standardized current condition values using a *t*-test, and the *P*-value result was used to categorize the indicator score.

Indicator	Reference baseline value and category	Current condition	*P*-value	Indicator score
Coral cover (%)	6.4, Fair	5.9	$P > 0.05$	Fair
Macroalgae cover (%)	22.1, Fair	23.7	$P > 0.05$	Fair
CCA cover (%)	4.2, Fair	3.8	$P > 0.05$	Fair
Coral density (corals m^{-2})	1.9, Fair	1.5	$P < 0.05$	Impaired
Old mortality (%)	17.4, Fair	14.7	$P > 0.05$	Fair

2010) and in 2012 (although the 2012 event was limited to mesophotic depths >30 m; Smith et al. 2016), since initial benthic monitoring began in 1999. The 2005 bleaching event was followed by an unprecedented disease outbreak, both of which caused a 60% decrease in coral cover across the territory (Miller et al. 2009). For the coral density and old mortality indicators, LTM data were not available, so the reference baselines used earliest available NCRMP data (Tables 1, Online Table S2).

All USVI LTM sites were reviewed and classified into comparable NCRMP habitat categories (Tables 1 and 3, Fig. 3, Online Table S1). Expert input, site-level data (i.e., coral cover and density), and NOAA benthic habitat maps were used to inform the classification. In the USVI, the majority (about 75%) of LTM sites (Online Table S1) were located in high-relief habitats with high coral; these corresponded to NCRMP habitat categories of aggregate reef, patch reef, and bedrock (Table 3). Few USVI LTM sites were in low-relief, low-coral habitats (Table 3). Further, the sample size of USVI LTM sites was small, and did not meet the review criteria for sufficient sample size for each individual habitat classification in high-relief habitats and for all combined low-relief habitats (Fig. 3). Therefore, USVI LTM sites were grouped into a high-relief habitat classification (rather than individual habitat categories), and only compared with current conditions in NCRMP sites in high-relief habitat and grouped into a high-relief habitat classification (Table 1). In the low-relief habitat classification, reference baselines did not include LTM data and were limited to the first year of NCRMP data in each region (Table 1).

US Virgin Islands: St. Thomas and St. John.—The mean reference value of 22.9% (1999–2005) for coral cover in high relief reefs in the St. Thomas and St. John region was categorized as Good, and the mean value for current conditions showed a decline to 8.5% (Table 6). Results showed that the standardized reference condition was significantly different than the current condition for coral cover (t = 3.82, df = 21, $P <$ 0.001); therefore, the current condition was categorized as Critical (Table 6). Also in high relief reef habitats, mean macroalgae cover increased from 19.6%, categorized as Fair, to 24.5%. The comparison of standardized values indicated a significant change from the reference value (t = 2.13, df = 21, P < 0.05); therefore, the current status for macroalgae was categorized as Impaired (Table 6). CCA cover and old mortality both showed a significant difference between standardized reference baselines and current conditions, improving from Fair to Very Good (CCA: t = 2.83, df = 21, P < 0.01; old mortality: t = 2.63, df = 93, P < 0.01). Coral density showed a significant decrease from standardized reference baselines, categorized as Fair, to current conditions (t = 1.98, df = 93, P < 0.05); therefore, the current status was considered Impaired (Table

Table 6. For the US Virgin Islands, reference baselines were calculated and standardized for each indicator for each habitat type in each region. The status of each reference baseline was categorized using local expert input. Values of current conditions were calculated and standardized for each indicator. Standardized reference baseline values were compared to standardized current condition values using a t-test, and the P-value result was used to categorize the indicator score.

Indicator	Habitat type	Reference baseline value and category	Current condition	P-value	Indicator score
St. Thomas and St. John					
Coral cover (%)	High relief reefs	22.9, Good	8.5	$P < 0.001$	Critical
Coral cover (%)	Low relief reefs	3.8, Fair	3.5	$P > 0.05$	Fair
Macroalgae cover (%)	High relief reefs	19.6, Fair	24.5	$P < 0.05$	Impaired
Macroalgae cover (%)	Low relief reefs	21.3, Fair	21.7	$P > 0.05$	Fair
CCA cover (%)	High relief reefs	1.4, Fair	4.7	$P < 0.01$	Very Good
CCA cover (%)	Low relief reefs	2.3, Fair	1.8	$P > 0.05$	Fair
Coral density (corals m^{-2})	High relief reefs	3.7, Fair	3.2	$P < 0.05$	Impaired
Coral density (corals m^{-2})	Low relief reefs	2.2, Fair	1.8	$P > 0.05$	Fair
Old mortality (%)	High relief reefs	23.1, Fair	15.8	$P < 0.01$	Very Good
Old mortality (%)	Low relief reefs	22.4, Fair	19.2	$P > 0.05$	Fair
St. Croix					
Coral cover (%)	High relief reefs	22.9, Fair	9.6	$P < 0.001$	Critical
Coral cover (%)	Low relief reefs	5.2, Fair	4.3	$P > 0.05$	Fair
Macroalgae cover (%)	High relief reefs	19.6, Fair	22.9	$P < 0.05$	Impaired
Macroalgae cover (%)	Low relief reefs	14.5, Fair	19	$P > 0.05$	Fair
CCA cover (%)	High relief reefs	1.4, Fair	2.9	$P < 0.001$	Very Good
CCA cover (%)	Low relief reefs	3.2, Fair	2.3	$P < 0.05$	Impaired
Coral density (corals m^{-2})	High relief reefs	2.1, Impaired	3.8	$P < 0.001$	Very Good
Coral density (corals m^{-2})	Low relief reefs	2.1, Fair	2.1	$P > 0.05$	Fair
Old mortality (%)	High relief reefs	21.1, Fair	16.8	$P > 0.05$	Fair
Old mortality (%)	Low relief reefs	16.7, Fair	13.7	$P > 0.05$	Fair

6). In low relief reef habitats, all indicators were initially assessed as Fair, and no statistically significant change was detected (t = 1.98, df = 93, $P > 0.05$; Table 6). Habitat-specific values for each indicator were weighted by reef area and combined for a region-wide categorization of Impaired for coral cover, macroalgae cover, and density, and Good for CCA cover and old mortality.

US Virgin Islands: St. Croix.—In the St. Croix region, coral cover declined in high relief habitats from the mean reference baseline value of 22.9% to the mean current value of 9.6% (Table 6). Statistical comparisons of standardized reference baseline values, categorized as Fair, to current values indicated a significant decline (t = 4.01, df = 21, $P < 0.001$); the current status was therefore categorized as Critical. The status of macroalgae cover in high coral habitats also declined from Fair to Impaired (t = 2.12, df = 21, $P < 0.05$) as actual values of macroalgae increased from 19.6% to 22.9% (Table 6). Scores for CCA cover and coral density both improved in high coral habitats, with CCA cover increasing from Fair to Very Good (t = 2.92, df = 21, $P < 0.01$) and coral density from Impaired to Very Good (t = 3.54, df = 41, $P < 0.001$). There was no change in the status of old mortality, which remained Fair (t = 2.01, df = 41, $P > 0.05$; Table 6).

For low relief habitats, mean reference baselines for coral cover (5.2%) and density (2.1 corals × m^{-2}) were both classified as Fair by local experts, as were all other reference baselines (Table 6). There was no statistically significant change between standardized reference values and standardized current condition values for coral cover (t = 1.97, df = 178, P > 0.05), density (t = 2.01, df = 79, P > 0.05), old mortality (t = 1.99, df =79, P > 0.05), or macroalgae (t = 1.97, df = 178, P > 0.05); therefore, the status of all remained categorized as Fair. The status for CCA cover declined from Fair to Impaired (t = 1.97, df = 178, P < 0.05; Table 6). Habitat-specific values for each indicator were weighted by reef area and combined for a region-wide categorization of Fair for coral cover and density, Impaired for CCA cover and macroalgae cover, and Good for old mortality.

Florida.—In Florida, coral reef declines initially began prior to the inception of LTM programs. Local experts therefore advised the inclusion of literature-sourced data for coral cover in combination with early years of LTM data (Fig. 3). A comprehensive literature review identified coral cover data from 1974 to 1999 that were combined with LTM data for historic references for Southeast Florida, Florida Keys, and the Dry Tortugas regions (Table 1, Online Table S5). Literature-sourced coral cover data had been acquired by comparable field methods used by current LTM programs (e.g., point-intercept transects and image analyses), and were considered to be from individual sites unless text described site-level replication. For all other indicators, the literature review did not identify sufficient historic data to meet the review criteria for reference data, and LTM data were used to represent reference baselines (Fig. 3). For Southeast Florida, reference data were uniformly classified as hardbottom habitat rather than by reef type because habitat identification in the literature was limited, and grouping was needed to meet the review criteria for sample size (Tables 1, 3; Fig. 3). In the Florida Keys and the Dry Tortugas, reference data were classified into broad habitat types that corresponded to the habitat classifications used in current condition data (Tables 1, 3; Fig. 3).

Southeast Florida.—In Southeast Florida, coral reefs have been impacted by a number of stressors (Blair and Flynn 1989, Dustan et al. 2008, Walker and Gilliam 2013), such as water quality and human population density (Tomascik and Sander 1985), dredging, sediment impacts to reefs, a cold-water event in 2010 (Lirman et al. 2011), an increase in bleaching and disease (Goldberg 1989, Cunning et al. 2019), and SCTLD (first observed in 2014; Studivan et al. 2022). The reference coral cover (1979–1992; Online Table S5) was 5%, categorized as Fair by local experts, and the current coral cover value was 1.2%. Results of t-tests comparing standardized current and reference baseline values (t = 2.64, df = 73, P < 0.01) categorized the current status as Critical (Table 7). The mean reference baseline macroalgae cover (Table 1) was classified as Good (4.4%; Table 7). There was a significant difference between the standardized reference and current values for macroalgae cover (t = 3.76, df = 23, P < 0.001); therefore, the macroalgae status categorization changed from Good to Critical. CCA cover did not show a significant difference between the standardized reference and current values, and therefore remained Impaired (t = 2.01, df = 48, P > 0.05). Mean coral density declined from 0.6 corals × m^{-2} (2005–2007), categorized as Impaired, to 0.4 corals × m^{-2} (2016–2018); however, the difference between standardized values of reference baseline to current values was not significant. (t = 1.98, df = 98, P > 0.05), and the current status was categorized as Impaired (Table 7). The old mortality indicator was not scored for Southeast Florida because the mean values for

Table 7. For Florida, reference baselines were calculated and standardized for each indicator for each habitat type in each region. The status of each reference baseline was categorized using local expert input. Values of current conditions were calculated and standardized for each indicator. Standardized reference baseline values were compared to standardized current condition values using a *t*-test, and the *P*-value result was used to categorize the indicator score.

Indicator	Habitat type	Reference baseline value and category	Current condition	*P*-value	Indicator score
Southeast Florida					
Coral cover (%)	- - -	5.3, Fair	1.2	$P < 0.01$	Critical
Macroalgae cover (%)	- - -	4.4, Good	26.5	$P < 0.001$	Critical
CCA cover (%)	- - -	1.4, Impaired	1.1	$P > 0.05$	Impaired
Coral density (corals m^{-2})	- - -	0.6, Impaired	0.4	$P > 0.05$	Impaired
Old mortality (%)	- - -	- - -	- - -	- - -	- - -
Florida Keys					
Coral cover (%)	Bank reef	19.7, Good	4.8	$P < 0.001$	Critical
Coral cover (%)	Patch reef	13.1, Good	9.2	$P < 0.05$	Fair
Macroalgae cover (%)	Bank reef	13.4, Good	27.4	$P < 0.001$	Critical
Macroalgae cover (%)	Patch reef	8.4, Good	14.9	$P < 0.01$	Impaired
CCA cover (%)	- - -	6.2, Good	1.7	$P < 0.001$	Critical
Coral density (corals m^{-2})	- - -	1.5, Impaired	2.4	$P < 0.001$	Very Good
Old mortality (%)	- - -	16.9, Fair	15.7	$P > 0.05$	Fair
Dry Tortugas					
Coral cover (%)	Mid-high relief reefs	28, Very Good	9.5	$P < 0.001$	Impaired
Coral cover (%)	Low relief reefs	3.6, Good	2.3	$P < 0.05$	Fair
Macroalgae cover (%)	- - -	40.9, Impaired	42.2	$P > 0.05$	Impaired
CCA cover (%)	- - -	4.2, Good	2.7	$P < 0.01$	Impaired
Coral density (corals m^{-2})	- - -	1.6, Fair	1.4	$P > 0.05$	Fair
Old mortality (%)	- - -	12.7, Fair	13.7	$P > 0.05$	Fair

both coral cover and density were extremely low for both current values and reference baselines (Table 7). Therefore, the population sample size was insufficient for statistical comparisons of old mortality.

Florida Keys.—Numerous disease outbreaks have been documented in the Florida Keys since 1976 (Gladfelter 1982, Aronson and Precht 2001). Substantial declines in coral cover in reef habitats began in the 1980s (Porter and Meier 1992, Ogden et al. 1994, Lapointe et al. 2019), and continued in seven major bleaching events between 1987 and 2015 (Manzello 2015), as well as a cold-water event in 2010 (Lirman et al. 2011). In 2016, SCTLD was spreading through the Florida Keys region (Muller et al. 2020), and by 2018 had reached the entire Keys region. In the Florida Keys, the mean reference values for coral cover on bank reefs (19.7%) and on patch reefs (13.1%) were each categorized as Good by the local expert panel (Table 7, Online Table S5). Due to the timing of SCTLD spread, values for current conditions were limited to 2018 for both bank reef and patch reef habitats, with a mean current value of coral cover of 4.8% on bank reefs and 9.2% on patch reefs (Table 7). Due to significant differences in standardized reference and current values for coral cover on bank reefs (t = 3.36, df = 137, $P < 0.001$) and on patch reefs (t = 2.02, df = 36, $P < 0.05$), the current status of coral cover was categorized as Critical on bank reefs and Fair on patch reefs (Table 7). On bank reefs in the Florida Keys, the mean reference macroalgae cover was 13.4% on bank reefs and 8.4% on patch reefs, both of which were categorized as Good (Table 7). Differences between standardized reference values and current values were significant for both bank (t = 2.07, df = 22, P < 0.001) and patch reefs (t = 3.16, df = 10, P < 0.01); consequently, the current status was categorized as Critical and Impaired, respectively (Table 6). Habitat-specific values were weighted by reef area and combined for a Keys-wide categorization of Impaired for both coral cover and macroalgae cover.

NCRMP 2014 data were used to create the reference value for CCA cover due to limited availability of older data (Table 1). Mean reference baseline values for CCA cover in the Florida Keys were 6.2% (Table 7), considered Good by local experts. Mean current conditions for CCA had declined to 1.7%, and the difference between standardized reference and current values was significant (t = 3.31, df = 376, P < 0.001), resulting in a CCA status categorized as Critical (Table 7). Coral density had a mean reference value of 1.6 corals × m^{-2} (2005–2007; Table 1), categorized as Impaired, and current values of 2.4 corals × m^{-2} (2016–2018; Table 1). There was a significant difference between standardized reference baselines and current conditions for coral density (t = 1.96, df = 254, P < 0.001), resulting in a current status of Very Good (Table 7). For old mortality, the mean reference value of 16.9% was categorized as Fair, and the current value was 15.7% (Tables 1 and 7). There was no significant difference between standardized reference baselines and current conditions for old mortality (t = 1.96, df = 260, P > 0.05), so the Fair categorization did not change (Table 7).

Dry Tortugas.—Disturbance events causing coral mortality in the Dry Tortugas have been described since the late 1800s (Mayer 1903, Porter et al. 1982, Jaap et al. 1989). These disturbances have been major contributors to coral decline (Davis 1982, Kuffner et al. 2020), particularly to Acroporid corals, with mortality of up to 96% on shallow reefs (Porter et al. 1982). In the Dry Tortugas, the mean reference value for coral cover on mid-relief and high relief reefs was 28% and 2.6% on low relief reefs (1975–1999), categorized as Very Good and Good, respectively (Table 7). The mean current (2018) coral cover was 9.5% on mid-high relief reefs and 2.3% on low relief reefs (Table 7). The differences in standardized reference and current values for coral cover was significant for both mid-high relief reefs (t = 2.01, df = 46, P < 0.001), and low relief reefs (t = 2.04, df = 30, P < 0.05), resulting in a current status of Impaired and Fair, respectively (Table 7). Values for the two habitats were weighted by reef area and combined to produce a current coral cover status for the Dry Tortugas of Fair.

The mean reference value for macroalgae cover was high (2014; 40.9%), considered Impaired, and mean current value for macroalgae cover was 42.2% in 2016–2018 (Tables 1, 7). No significant difference was detected between standardized reference and current values (t = 1.98, df = 105, P > 0.05), so the current status remained Impaired. The reference mean CCA cover was 4.2%, classified as Good, and the current condition was 2.7%. The difference between standardized reference and current values was significant (t = 2.62, df = 105, P < 0.01), resulting in a current status of Impaired. Reference values for mean coral density (1.6 corals × m^{-2}) and old mortality (12.7%; 2007–2009; Tables 1 and 7) were classified as Fair, and current values were 1.4 corals × m^{-2} and 13.7%, respectively (2016–2018). There was no significant difference in standardized reference values and current values for coral density (t = 2.02, df = 40, P > 0.05) and old mortality (t = 2.02, df = 40, P > 0.05), respectively, so the current status remained Fair.

Discussion

This study furthered the development of holistic ecosystem evaluations for coral reef benthic communities. Key developments in the analyses approach were: (1) identification of LTM and literature-based baseline data for specific indicators as a reference for measuring changes in resource condition in each US coral reef region, (2) qualitative classification of available reference baseline data relative to the historic context of coral reef decline in each region based on local expertise, (3) application of a quantitative, standardized statistical approach to incorporate a range of disparate baseline data sources and years, and (4) application of a quantitative benchmark to evaluate the change in resource condition over time. Classified, color-coded results are straightforward to interpret across the suite of indicators and geographic regions and identified trends and challenges both specific to individual regions and broadly across the US coral reef ecosystems.

This evaluation demonstrates that corals and benthic communities in all US Atlantic coral reef regions, except for Flower Garden Banks National Marine Sanctuary, continue historic trends of decline (e.g., Porter et al. 1999, Aronson and Precht 2001, García-Sais et al. 2008, Miller et al. 2009, Toth et al. 2014). Our findings indicate region-specific changes in coral indicators (Tables 4–7), as well as changes that differ by habitat types. Where habitat types were differentiated, specifically in the Florida Keys, Dry Tortugas, and USVI (Tables 3, 6, 7), low coral habitats show less decline than high coral habitats. This likely reflects species-specific changes in the coral community due to declines in reef-building species. These changes between habitats and between regions are consistent with the findings of Grove et al. (2023) that demonstrate the reference-centering status evaluations of coral reef fish communities are more statistically robust at the habitat or region level than at the site level.

Given continued disturbances, such as SCTLD, and limited recovery, the status of corals has worsened substantially since the last year of data included in these analyses (2018). In 2018, SCTLD was limited to Southeast Florida and the Florida Keys (Walton et al. 2018, Muller et al. 2020, Kolodziej et al. 2021); however, since then, the Dry Tortugas, Puerto Rico, and US Virgin Islands have since been severely impacted, as well as much of the Caribbean (Brandt et al. 2021, Alvarez-Filip et al. 2022). SCTLD first appeared in the US Virgin Islands and Puerto Rico reefs in 2019 (Brandt et al. 2021, Weil et al. 2021), and major hurricanes Irma and Maria impacted reefs in these regions in 2017 causing damage from waves (Viehman et al. 2020) and land-based sediment runoff (Takesue et al. 2021). SCTLD arrived in the Dry Tortugas in 2021 (Grove et al. 2022) and was fully epidemic by 2022, followed by impacts from Hurricane Ian in 2022. Flower Garden Banks reported disease-like lesions in 2022 that have not been confirmed as SCTLD to date (Johnston et al. 2023).

Ecosystem assessments should apply quantitative analyses that are transparent, repeatable in new regions, and can include the input of local experts as needed, with results that are easy to communicate (Game et al. 2013, Borja et al. 2016). The reference-centering approach described here utilizes data-driven decisions informed by local knowledge of the expert panel. However, the inclusion of indicators that address current threats and stressors remains an ongoing research challenge, especially in rapidly changing ecosystems (Messer et al. 1991). As stressors, particularly

SCTLD, continue to drive coral declines in the Caribbean, coral communities are becoming more homogeneous as more reef-building species are becoming rare or even functionally extinct (Burman et al. 2012, Walton et al. 2018, Costa et al. 2021, Heres et al. 2021, Hayes et al. 2022). Although coral cover is broadly used and has the most long-term historic data availability (Obura et al. 2019), it has limited ability to represent demographic shifts in population and community dynamics (Brito-Millán et al. 2019, Edmunds and Riegl 2020). In this study, reef-building coral species were included in the density and mortality indicators; however, the density metric is confounded by colony size, as increased partial mortality has resulted in higher densities of small corals and lower density of large corals (Grove et al. 2022, Hayes et al. 2022). Furthermore, larger corals have lower mortality rates than smaller corals, so disproportionate mortality of larger corals, such as through bleaching or SCTLD, reduces the recovery capacity of a reef (Speare et al. 2022). Although low coral mortality is a positive reef attribute from an ecological perspective, results showed limited changes in old mortality within most regions. This may be due to: (1) a combination of limited historic data and limited expert perspective, (2) cumulative effects of old mortality over a limited span of years, (3) difficulty in identifying severely degraded old dead corals in the field (e.g., bioerosion, colonization), and (4) disproportionate mortality by size and species. To improve representation of community and population declines in reef-building, disease-susceptible, and rare coral species, future ecosystem condition assessments should include indicators that represent diversity and population dynamics (e.g., colony size in combination with density) of individual species impacted by stressors.

Inclusion of multiple long-term datasets highlights data gaps in current long-term monitoring programs that should be addressed to improve future holistic ecosystem assessments. Limited historic and current monitoring data are available to quantify disturbance impacts and recovery (or lack thereof) from the introduction of new corals. Episodic event-based mortality, such as hurricanes and bleaching events, can have major impacts on coral condition (Edmunds and Gray 2014, Viehman et al. 2020, Madden et al. 2023) and are likely to increase in the changing climate. Successful coral recruitment can drive reef recovery (Graham et al. 2011), and recruitment has been used as an indicator in resilience assessments (McClanahan et al. 2012). As coral restorations efforts begin to ramp up in these jurisdictions (e.g., NOAA 2019), restoration monitoring data should also be included to evaluate the contribution of restoration to ecosystem recovery. Data on event-driven mortality and on new corals entering the populations will allow more robust assessments of coral reefs in future years.

The reference-centering approach allows for inclusion of multiple long-term monitoring datasets that varied by data availability, region, indicator, and data source. These reference baselines can be used in future temporal evaluations. However, qualitative categorizations of reference datasets by expert panels still potentially represent shifting baselines, as evidenced in comparisons between published declines and expert categorizations, and in many cases the earliest reference data pre-dated experts (e.g., Online Table S5). Temporal changes were difficult to detect when data availability was limited and baseline data were collected within a few years of the current values (e.g., Puerto Rico indicators). The addition of predisturbance or decades-old data, when available, to ecosystem status evaluations may outweigh any potential limitations of using such data by providing quantitative information that improves the understanding of coral reefs prior to monitoring (Alagona et al. 2012). However, inclusion of literature-derived values for reference baselines necessitates review of sample design (or lack thereof), site selection (potentially targeted or opportunistic), survey methods, habitat heterogeneity, or lack of information about any of the aforementioned (Hughes et al. 2021).

The evaluations of corals and benthic communities described here were incorporated into NCRMP status assessments for each jurisdiction as well as a national assessment, all of which also included evaluations of fish (Grove et al. 2023), climate, and socioeconomics, and were the first in a series of future status assessments of US coral reefs (Towle et al. 2022). The reference-centering approach can be updated with current data and indicators for future regional and national assessments. Continued declines of corals and reef communities, exacerbated by dramatic disease impacts, underscore the critical need for the improved effectiveness of coral reef management and conservation.

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Literature Cited

Alagona P, Sandlos J, Wiersma Y. 2012. Past imperfect: using historical ecology and baseline data for conservation and restoration projects in North America. Environmental Philosophy. 9(1):49–70. Available from: https://www.jstor.org/stable/26169395

Alvarez-Filip L, González-Barrios FJ, Pérez-Cervantes E, Molina-Hernández A, Estrada-Saldívar N. 2022. Stony coral tissue loss disease decimated Caribbean coral populations and reshaped reef functionality. Communications Biology. 5(1):440. https://doi.org/10.1038/s42003-022-03398-6

Aronson RB, Precht WF. 2001. White-band disease and the changing face of Caribbean coral reefs. Hydrobiologia. 460:25–38. https://doi.org/10.1023/A:1013103928980

Aronson RB, Precht WF, Murdoch TJT, Robbart ML. 2005. Long-term persistence of coral assemblages on the Flower Garden Banks, Northwestern Gulf of Mexico: implications for science and management. Gulf Mex Sci. 23(1):84–94. https://doi.org/10.18785/goms.2301.06

Blair S, Flynn B. 1989. Biological monitoring of hard bottom reef communities off Dade County Florida: community description. *In:* Jaap WC, editor. Diving for science. Proceedings of the 10th annual scientific diving symposium. American Academy of Underwater Sciences. p. 9–24.

Borja A, Elliott M, Andersen JH, Berg T, Carstensen J, Halpern BS, Heiskanen A-S, Korpinen S, Lowndes JSS, Martin G, et al. 2016. Overview of integrative assessment of marine systems: the ecosystem approach in practice. Front Mar Sci. 3:20. https://doi.org/10.3389/fmars.2016.00020

Brandt ME, Ennis RS, Meiling SS, Townsend J, Cobleigh K, Glahn A, Quetel J, Brandtneris V, Henderson LM, Smith TB. 2021. The emergence and initial impact of Stony Coral Tissue

Loss Disease (SCTLD) in the United States Virgin Islands. Front Mar Sci. 8:715329. https://doi.org/10.3389/fmars.2021.715329

Brito-Millán M, Vermeij MJA, Alcantar EA, Sandin SA. 2019. Coral reef assessments based on cover alone mask active dynamics of coral communities. Mar Ecol Prog Ser. 630:55–68. https://doi.org/10.3354/meps13128

Bruno JF, Valdivia A. 2016. Coral reef degradation is not correlated with local human population density. Sci Rep. 6:29778. https://doi.org/10.1038/srep29778

Burman SG, Aronson RB, van Woesik R. 2012. Biotic homogenization of coral assemblages along the Florida reef tract. Mar Ecol Prog Ser. 467:89–96. https://doi.org/10.3354/meps09950

Carruthers TJB, Beckert K, Schupp CA, Saxby T, Kumer JP, Thomas J, Sturgis B, Dennison WC, Williams M, Fisher T, et al. 2013. Improving management of a mid-Atlantic coastal barrier island through assessment of habitat condition. Estuar Coast Shelf Sci. 116:74–86. https://doi.org/10.1016/j.ecss.2012.08.012

Costa SV, Hibberts SJ, Olive DA, Budd KA, Long AE, Meiling SS, Miller MB, Vaughn KM, Carrión CI, Cohen MB, et al. 2021. Diversity and disease: the effects of coral diversity on prevalence and impacts of Stony Coral Tissue Loss Disease in Saint Thomas, U.S. Virgin Islands. Front Mar Sci. 8:682688. https://doi.org/10.3389/fmars.2021.682688

Cramer KL, Jackson JBC, Angioletti CV, Leonard-Pingel J, Guilderson TP. 2012. Anthropogenic mortality on coral reefs in Caribbean Panama predates coral disease and bleaching. Ecol Lett. 15(6):561–567. https://doi.org/10.1111/j.1461-0248.2012.01768.x

Cunning R, Silverstein RN, Barnes BB, Baker AC. 2019. Extensive coral mortality and critical habitat loss following dredging and their association with remotely-sensed sediment plumes. Mar Pollut Bull. 145:185–199. https://doi.org/10.1016/j.marpolbul.2019.05.027

Davis GE. 1982. A century of natural change in coral distribution at the Dry Tortugas: a comparison of reef maps from 1881 and 1976. Bull Mar Sci. 32(2):608–623.

Dustan P, Fauth J, Pante E, Banks K, Vargas-Angel B, Downs C. 2008. Using cellular diagnostics to link land-based sources of pollution with coral reef degradation in South Florida. Ft. Lauderdale Florida: Proceedings of the 11th International Coral Reef Symposium. p. 495–499.

Eakin CM, Morgan JA, Heron SF, Smith TB, Liu G, Alvarez-Filip L, Baca B, Bartels E, Bastidas C, Bouchon C, et al. 2010. Caribbean corals in crisis: record thermal stress, bleaching, and mortality in 2005. PLOS ONE. 5(11):e13969. https://doi.org/10.1371/journal.pone.0013969

Edmunds PJ, Gray SC. 2014. The effects of storms, heavy rain, and sedimentation on the shallow coral reefs of St. John, US Virgin Islands. Hydrobiologia. 734(1):143–158. https://doi.org/10.1007/s10750-014-1876-7

Edmunds PJ, Riegl B. 2020. Urgent need for coral demography in a world where corals are disappearing. Mar Ecol Prog Ser. 635:233–242. https://doi.org/10.3354/meps13205

Federal Register. 2014. Final rule. Endangered and threatened wildlife and plants: final listing determinations on proposal to list 66 reef-building coral species and to reclassify elkhorn and staghorn corals. National Marine Fisheries Service (NMFS), National Oceanic and Atmospheric Administration (NOAA), Commerce. p. 53851–54123.

Fletcher PJ, Kelble CR, Nuttle WK, Kiker GA. 2014. Using the integrated ecosystem assessment framework to build consensus and transfer information to managers. Ecol Indic. 44:11–25. https://doi.org/10.1016/j.ecolind.2014.03.024

Freebairn DM, King CA. 2003. Reflections on collectively working toward sustainability: indicators for indicators! Aust J Exp Agric. 43(3):223–238. https://doi.org/10.1071/EA00195

Friedlander AM, DeMartini EE. 2002. Contrasts in density, size, and biomass of reef fishes between the northwestern and the main Hawaiian islands: the effects of fishing down apex predators. Mar Ecol Prog Ser. 230:253–264. https://doi.org/10.3354/meps230253

Game ET, Kareiva P, Possingham HP. 2013. Six common mistakes in conservation priority setting. Conserv Biol. 27(3):480–485. https://doi.org/10.1111/cobi.12051

García-Sais JR, Appeldoorn R, Bruckner A, Caldow C, Christensen JD, Lilyestrom C, Monaco ME, Sabater J, Williams E, Diaz E. 2008. The state of coral reef ecosystems of the Commonwealth of Puerto Rico. *In:* The state of coral reef ecosystems of the United States and Pacific freely associates states. Silver Spring, Maryland: NOAA Technical Memorandum NOS NCCOS 73. p. 91–134.

Gittings SR, Boland GS, Deslarzes KJP, Hagman DK, Holland BS. 1992. Long-term monitoring at the East and West Flower Garden Banks. New Orleans, Louisiana: US Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Gladfelter WB. 1982. White-band disease in *Acropora palmata*: implications for the structure and growth of shallow reefs. Bull Mar Sci. 32(2):639–643.

Goldberg W. 1989. Biological effects of beach restoration in South Florida: the good, the bad, and the ugly. Beach Preservation Technology. 88:19–28.

Graham N, Nash K, Kool J. 2011. Coral reef recovery dynamics in a changing world. Coral Reefs. 30:283–294. https://doi.org/10.1007/s00338-010-0717-z

Grove LJW, Blondeau J, Cain E, Davis IM, Edwards KF, Groves SH, Hile SD, Langwiser C, Siceloff L, Swanson DW, et al. 2022. National Coral Reef Monitoring Program biological monitoring summary Florida: 2020–2021. Coral Reef Conservation Program; NOAA National Centers for Coastal Ocean Science Technical Memorandum 44. https://doi.org/10.25923/9jns-v916

Grove LJW, Smith S, Blondeau J, Ault J. 2023. Quantifying ecosystem status of the reef fish community from a large-scale probability survey in southern Florida. Bull Mar Sci. (in preparation)

Groves SH. 2019. ncrmp.benthics.statusreport (R Package v1.0.0.001). Accessed 23 January, 2023. Available from: https://github.com/MSE-NCCOS-NOAA/NCRMP.benthics.statusreport

Groves SH, Viehman TS. 2023. ncrmp.benthic.analysis (R package v1.0.0). Accessed 23 January, 2023. Available from: https://github.com/MSE-NCCOS-NOAA/NCRMP_benthics

Hayes NK, Walton CJ, Gilliam DS. 2022. Tissue loss disease outbreak significantly alters the Southeast Florida stony coral assemblage. Front Mar Sci. 9:975894. https://doi.org/10.3389/fmars.2022.975894

Heres MM, Farmer BH, Elmer F, Hertler H. 2021. Ecological consequences of Stony Coral Tissue Loss Disease in the Turks and Caicos Islands. Coral Reefs. 40:609–624. https://doi.org/10.1007/s00338-021-02071-4

Hoegh-Guldberg O, Mumby PJ, Hooten AJ, Steneck RS, Greenfield P, Gomez E, Harvell CD, Sale PF, Edwards AJ, Caldeira K, et al. 2007. Coral reefs under rapid climate change and ocean acidification. Science. 318(5857):1737–1742. https://doi.org/10.1126/science.1152509

Hughes AC, Orr MC, Ma K, Costello MJ, Waller J, Provoost P, Yang Q, Zhu C, Qiao H. 2021. Sampling biases shape our view of the natural world. Ecography. 44(9):1259–1269. https://doi.org/10.1111/ecog.05926

Jaap WC, Lyons WG, Dustan P, Halas JC. 1989. Stony coral (*Scleractinia* and *Milleporina*) community structure at Bird Key Reef, Ft. Jefferson National Monument, Dry Tortugas, Florida. State of Florida, Department of Natural Resources, Florida Marine Research Institute. No. 46.

Johnston MA, Sterne TK, Blakeway RD, MacMillan J, Nuttall MF, Hu X, Embesi JA, Hickerson EL, Schmahl GP. 2018. Long-term monitoring at East and West Flower Garden Banks: 2017 annual report. Galveston, Texas: US Department of Commerce, National Oceanic and Atmospheric Administration, Flower Garden Banks National Marine Sanctuary.

Johnston MA, Studivan MS, Enochs IC, Correa AMS, Besemer N, Eckert RJ, Edwards K, Hannum R, Hu X, Nuttall M, et al. 2023. Coral disease outbreak at the remote Flower Garden Banks, Gulf of Mexico. Front Mar Sci. 10:1111749. https://doi.org/10.3389/fmars.2023.1111749

Jokiel PL, Rodgers KS, Brown EK, Kenyon JC, Aeby G, Smith WR, Farrell F. 2015. Comparison of methods used to estimate coral cover in the Hawaiian Islands. PeerJ. 3:e954. https://doi.org/10.7717/peerj.954

Knowlton N, Jackson JBC. 2008. Shifting baselines, local impacts, and global change on coral reefs. PLoS Biol. 6(2):e54. https://doi.org/10.1371/journal.pbio.0060054

Kolodziej G, Studivan MS, Gleason ACR, Langdon C, Enochs IC, Manzello DP. 2021. Impacts of Stony Coral Tissue Loss Disease (SCTLD) on coral community structure at an inshore patch reef of the Upper Florida Keys using photomosaics. Front Mar Sci. 8:682163. https://doi.org/10.3389/fmars.2021.682163

Kuffner IB, Stathakopoulos A, Toth LT, Bartlett LA. 2020. Reestablishing a stepping-stone population of the threatened elkhorn coral *Acropora palmata* to aid regional recovery. Endanger Species Res. 43:461–473. https://doi.org/10.3354/esr01083

Landres P, Verner J, Thomas J. 1988. Ecological uses of vertebrate indicator species: a critique. Conserv Biol. 2(4):316–328. https://doi.org/10.1111/j.1523-1739.1988.tb00195.x

Lapointe BE, Brewton RA, Herren LW, Porter JW, Hu C. 2019. Nitrogen enrichment, altered stoichiometry, and coral reef decline at Looe Key, Florida Keys, USA: a 3-decade study. Mar Biol. 166(8):108. https://doi.org/10.1007/s00227-019-3538-9

Lindenmayer DB. 1999. Future directions for biodiversity conservation in managed forests: indicator species, impact studies and monitoring programs. For Ecol Manage. 115:277–287. https://doi.org/10.1016/S0378-1127(98)00406-X

Lirman D, Schopmeyer S, Manzello D, Gramer LJ, Precht WF, Muller-Karger F, Banks K, Barnes B, Bartels E, Bourque A, et al. 2011. Severe 2010 cold-water event caused unprecedented mortality to corals of the Florida Reef Tract and reversed previous survivorship patterns. PLOS ONE. 6(8):e23047. https://doi.org/10.1371/journal.pone.0023047

Lohr SL. 2010. Sampling: design and analysis. Boston: Brooks/Cole.

Madden IA, Mariwala A, Lindhart M, Narayan S, Arkema KK, Beck MW, Baker JW, Suckale J. 2023. Quantifying the fragility of coral reefs to hurricane impacts: a case study of the Florida Keys and Puerto Rico. Environ Res Lett. 18(2):024034. https://doi.org/10.1088/1748-9326/acb451

Manzello DP. 2015. Rapid recent warming of coral reefs in the Florida Keys. Sci Rep. 5(1):16762. https://doi.org/10.1038/srep16762

Mayer AG. 1903. The Tortugas, Florida, as a station for research in biology. Science. 17:190–192. https://doi.org/10.1126/science.17.422.190

McClanahan TR, Donner SD, Maynard JA, MacNeil MA, Graham NAJ, Maina J, Baker AC, Alemu IJB, Beger M, Campbell SJ, et al. 2012. Prioritizing key resilience indicators to support coral reef management in a changing climate. PLOS ONE. 7(8):e42884. https://doi.org/10.1371/journal.pone.0042884

McClenachan L. 2009. Documenting loss of large trophy fish from the Florida Keys with historical photographs. Conserv Biol. 23(3):636–643. https://doi.org/10.1111/j.1523-1739.2008.01152.x

McClenachan L, O'Connor G, Neal BP, Pandolfi JM, Jackson JB. 2017. Ghost reefs: nautical charts document large spatial scale of coral reef loss over 240 years. Sci Adv. 3(9):e1603155. https://doi.org/10.1126/sciadv.1603155

McField M, Kramer PR. (with contributions by Gorrez M, McPherson M). 2007. Healthy reefs for healthy people: a guide to indicators of reef health and social well-being in the Mesoamerican reef region. Healthy Reefs for Healthy People Initiative. The Smithsonian Institution. 208 p.

Messer JJ, Linthurst RA, Overton WS. 1991. An EPA program for monitoring ecological status and trends. Environ Monit Assess. 17(1):67–78. https://doi.org/10.1007/BF00402462

Miller J, Muller E, Rogers C, Waara R, Atkinson A, Whelan KRT, Patterson M, Witcher B. 2009. Coral disease following massive bleaching in 2005 causes 60% decline in coral cover on reefs in the US Virgin Islands. Coral Reefs. 28(4):925. https://doi.org/10.1007/s00338-009-0531-7

Montenero K, Kelble C, Broughton K. 2021. A quantitative and qualitative decision-making process for selecting indicators to track ecosystem condition. Mar Policy. 129:104489. https://doi.org/10.1016/j.marpol.2021.104489

Moore MM, Wallace Covington W, Fulé PZ. 1999. Reference conditions and ecological restoration: a southwestern Ponderosa Pine perspective. Ecol Appl. 9(4):1266–1277. https://doi.org/10.1890/1051-0761(1999)009[1266:RCAERA]2.0.CO;2

Muller EM, Sartor C, Alcaraz NI, van Woesik R. 2020. Spatial epidemiology of the Stony Coral Tissue Loss Disease in Florida. Front Mar Sci. 7:163. https://doi.org/10.3389/fmars.2020.00163

Nadon M-O, Stirling G. 2006. Field and simulation analyses of visual methods for sampling coral cover. Coral Reefs. 25:177–185. https://doi.org/10.1007/s00338-005-0074-5

National Oceanic and Atmospheric Administration. 2019. Restoring seven iconic reefs: a mission to recover the coral reefs of the Florida Keys. National Oceanic and Atmospheric Administration. 45 p.

Obura DO, Aeby G, Amornthammarong N, Appeltans W, Bax N, Bishop J, Brainard RE, Chan S, Fletcher P, Gordon TAC, et al. 2019. Coral reef monitoring, reef assessment technologies, and ecosystem-based management. Front Mar Sci. 6:580. https://doi.org/10.3389/fmars.2019.00580

Ogden JC, Porter JW, Smith NP, Szmant AM, Jaap WC, Forcucci D. 1994. A long-term interdisciplinary study of the Florida Keys seascape. Bull Mar Sci. 54(3):1059–1071.

Pauly D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. Trends Ecol Evol. 10(10):430. https://doi.org/10.1016/S0169-5347(00)89171-5

Perry CT, Murphy GN, Kench PS, Smithers SG, Edinger EN, Steneck RS, Mumby PJ. 2013. Caribbean-wide decline in carbonate production threatens coral reef growth. Nat Commun. 4:1402. https://doi.org/10.1038/ncomms2409

Porter JW, Battey JF, Smith GJ. 1982. Perturbation and change in coral reef communities. Proc Natl Acad Sci USA. 79(5):1678–1681. https://doi.org/10.1073/pnas.79.5.1678

Porter JW, Lewis SK, Porter KG. 1999. The effect of multiple stressors on the Florida Keys coral reef ecosystem: a landscape hypothesis and a physiological test. Limnol Oceanogr. 44:941–949. https://doi.org/10.4319/lo.1999.44.3_part_2.0941

Porter JW, Meier OW. 1992. Quantification of loss and change in Floridian reef coral populations. Am Zool. 32:625–640. https://doi.org/10.1093/icb/32.6.625

Pretty JN. 1995. Participatory learning for sustainable agriculture. World Dev. 23(8):1247–1263. https://doi.org/10.1016/0305-750X(95)00046-F

R Core Team. 2019. R: a language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. Available from: https://www.R-project.org/

Roff G, Mumby PJ. 2012. Global disparity in the resilience of coral reefs. Trends Ecol Evol. 27(7):404–413. https://doi.org/10.1016/j.tree.2012.04.007

Sandin SA, Smith JE, DeMartini EE, Dinsdale EA, Donner SD, Friedlander AM, Konotchick T, Malay M, Maragos JE, Obura D, et al. 2008. Baselines and degradation of coral reefs in the Northern Line Islands. PLOS ONE. 3(2):e1548. https://doi.org/10.1371/journal.pone.0001548

Santavy DL, Jackson SK, Jessup B, Gerritsen J, Rogers C, Fisher WS, Weil E, Szmant A, Cuevas-Miranda D, Walker BK, et al. 2022. A biological condition gradient for coral reefs in the US Caribbean Territories: Part I. Coral narrative rules. Ecol Indic. 138:108805. https://doi.org/10.1016/j.ecolind.2022.108805

Smith SG, Swanson DW, Chiappone M, Miller SL, Ault JS. 2011. Probability sampling of stony coral populations in the Florida Keys. Environ Monit Assess. 183(1–4):121–138. https://doi.org/10.1007/s10661-011-1912-2

Smith TB, Gyory J, Brandt ME, Miller WJ, Jossart J, Nemeth RS. 2016. Caribbean mesophotic coral ecosystems are unlikely climate change refugia. Glob Change Biol. 22(8):2756–2765. https://doi.org/10.1111/gcb.13175

Speare KE, Adam TC, Winslow EM, Lenihan HS, Burkepile DE. 2022. Size-dependent mortality of corals during marine heatwave erodes recovery capacity of a coral reef. Glob Change Biol. 28(4):1342–1358. https://doi.org/10.1111/gcb.16000

Studivan MS, Rossin AM, Rubin E, Soderberg N, Holstein DM, Enochs IC. 2022. Reef sediments can act as a Stony Coral Tissue Loss disease vector. Front Mar Sci. 8:815698. https://doi.org/10.3389/fmars.2021.815698

Takesue RK, Sherman C, Ramirez NI, Reyes AO, Cheriton OM, Ríos RV, Storlazzi CD. 2021. Land-based sediment sources and transport to southwest Puerto Rico coral reefs after Hurricane Maria, May 2017 to June 2018. Estuar Coast Shelf Sci. 259:107476. https://doi.org/10.1016/j.ecss.2021.107476

Tomascik T, Sander F. 1985. Effects of eutrophication on reef-building corals. Mar Biol. 87(2):143–155. https://doi.org/10.1007/BF00539422

Toth LT, van Woesik R, Murdoch TJT, Smith SR, Ogden JC, Precht WF, Aronson RB. 2014. Do no-take reserves benefit Florida's corals? 14 years of change and stasis in the Florida Keys National Marine Sanctuary. Coral Reefs. 33(3):565–577. https://doi.org/10.1007/s00338-014-1158-x

Towle EK, Donovan EC, Kelsey H, Allen ME, Barkley H, Blondeau J, Brainard RE, Carew A, Couch CS, Dillard MK, et al. 2022. A National Status Report on United States coral reefs based on 2012–2018 data from National Oceanic and Atmospheric Administration's National Coral Reef Monitoring Program. Front Mar Sci. 8:812216. https://doi.org/10.3389/fmars.2021.812216

Viehman TS, Nemeth M, Groves S, Buckel C, Griffin S, Field D, Moore T, Moore J. 2020. Coral assessment and restoration in the U.S. Caribbean after 2017 hurricanes. NOAA National Centers for Coastal Ocean Science Technical Memorandum 278. https://doi.org/10.25923/7r0b-wc52

Walker BK, Gilliam DS. 2013. Determining the extent and characterizing coral reef habitats of the northern latitudes of the Florida Reef Tract (Martin County). PLOS ONE. 8(11):e80439. https://doi.org/10.1371/journal.pone.0080439

Walton CJ, Hayes NK, Gilliam DS. 2018. Impacts of a regional, multi-year, multi-species coral disease outbreak in Southeast Florida. Front Mar Sci. 5:323. https://doi.org/10.3389/fmars.2018.00323

Walz R. 2000. Development of environmental indicator systems: experiences from Germany. Environ Manage. 25(6):613–623. https://doi.org/10.1007/s002670010048

Weil E, Hernández-Delgado E, Gonzalez M, Williams S, Suleiman-Ramos S, Figuerola M, Metz-Estrella T. 2021. Spread of the new coral disease "SCTLD" into the Caribbean: implications for Puerto Rico. Reef Encounter. 34:38–43.