Research Square

Preprints are preliminary reports that have not undergone peer review.
They should not be considered conclusive, used to inform clinical practice,
or referenced by the media as validated information.

Severe ecological and functional simplification of coral communities following the combined impacts of the SCTLD and the fourth global bleaching event at Banco Chinchorro

Jhoselinne Buenrostro-Muñoz
National Autonomous University of Mexico
Esmeralda Pérez-Cervantes
National Autonomous University of Mexico
Lorenzo Álvarez-Filip
lorenzo@cmarl.unam.mx

National Autonomous University of Mexico

Research Article

Keywords: Thermal stress, disease outbreaks, Mesoamerican Reef, community reshape, coral calcification

Posted Date: April 16th, 2026

DOI: https://doi.org/10.21203/rs.3.rs-9337587/v1

License: This work is licensed under a Creative Commons Attribution 4.0 International License. Read Full License

Additional Declarations: No competing interests reported.

Severe ecological and functional simplification of coral communities following the combined impacts of the SCTLD and the fourth global bleaching event at Banco Chinchorro

Jhoselinne Buenrostro-Muñoz[1,2,*], Esmeralda Pérez-Cervantes[1], Lorenzo Álvarez-Filip[1,3]

[1] Biodiversity and Reef Conservation Laboratory, Instituto de Ciencias del Mar y Limnología, Unidad Académica de Sistemas Arrecifales, Universidad Nacional Autónoma de México, Puerto Morelos 77580, Quintana Roo, México

[2] Posgrado de Ciencias Biológicas, Universidad Nacional Autónoma de México, Ciudad de México, 04530, México.

[3] Laboratorio Nacional de Biología del Cambio Climático, SECIHTI, Ciudad de México, México

Corresponding author: lorenzo@cmarl.unam.mx

Abstract

The Banco Chinchorro Biosphere Reserve, an offshore reef complex in the western Caribbean, is recognized as one of the healthiest systems in the region due to its high biodiversity and isolation from land-based threats. However, in 2023, it faced two devastating events: an outbreak of stony coral tissue loss disease (SCTLD) and the strongest marine heatwave on record. This study investigates how these events impacted the ecological and functional integrity of the reef system by analyzing changes in coral communities at eight sites across the reef bank before, during, and after both events. Coral condition and composition of coral communities, as well as their overall calcification rates, were assessed using a minimum of six 10 m² belt transects for each survey period. Results revealed a halving of coral colony numbers over two years. While density losses were evident across almost all species, key reef-building species like *Pseudodiploria strigosa*, and *Agaricia tenuifolia* experienced the most severe reductions. Additionally, we report local extirpation of species such as *Dendrogyra cylindrus* and *Meandrina meandrites*. The disproportionate coral mortality has reshaped communities into simpler, more homogeneous assemblages, largely dominated by opportunistic and stress-tolerant corals. This decline indicates a critical loss of geo-ecological functionality, resulting in significant reductions in coral community calcification. Understanding the response mechanisms of coral communities to the cumulative effects of global change and disease outbreaks is essential for predicting how coral reefs will respond to rapidly increasing climatic pressures.

Keywords: Thermal stress; disease outbreaks; Mesoamerican Reef; community reshape, coral calcification.

Introduction

Caribbean coral reefs are experiencing unprecedented rates of degradation and heightened pressure from environmental and anthropogenic disturbances, leading to rapid declines in the abundance and cover of reef-building corals (Najeeb et al. 2025). Between the 1980s and 2024, the region lost over 80% of its live hard coral cover, primarily driven by anthropogenic disturbances and disease outbreaks (González-Barrios et al. 2023; Schaffelke et al. 2025); yet changes in absolute cover alone do not fully capture this decline (Alvarez-Filip et al. 2013). Increasing environmental pressures also result in drastic shifts in species abundance and taxonomic composition, leaving many reefs dominated by tolerant scleractinian corals characterized by flattened morphologies, brooding reproduction, high turnover, and resistance to bleaching and disease (Darling et al. 2012; Alvarez-Filip et al. 2013; González-Barrios and Álvarez-Filip 2018; Lima et al. 2023). The resilience of Caribbean coral communities is further compromised by reduced coral recruitment rates, largely driven by newly dominant corals, leading to the predominance of weedy species (Muller et al. 2014; Edmunds 2021; Díaz-Talamantes et al. 2025). Thus, coral declines, accompanied by changes in species composition and functional traits, are compromising key geo-ecological functions such as structural complexity, reef growth, and the capacity to track sea level rise (Denis et al. 2017; Estrada-Saldívar et al. 2019; González-Barrios et al. 2021; Alvarez-Filip et al. 2022). Ultimately, these rapid changes in coral communities cascade through trophic networks, altering fish and invertebrate communities even before structural loss occurs (Kayal et al. 2012; Richardson et al. 2018).

Over the last decade, two events have severely undermined the integrity of Caribbean reefs. First, the stony coral tissue loss disease (SCTLD) outbreak has been spreading across the region since its first documentation in Florida in 2014, becoming the most devastating coral epizootic to date (Precht et al. 2016; Alvarez-Filip et al. 2019). While SCTLD affects nearly 30 coral species, large and massive taxa are the most affected, exhibiting high disease prevalence, accelerated rates of tissue loss, and marked population declines (Estrada-Saldívar et al. 2019; Clark et al. 2021; Guzmán-Urieta et al. 2026). Second, the 2023–2024 marine heatwave (MHW) — the most intense on record for the region, with peaks exceeding 20 degree-heating weeks (DHW) — caused widespread mortality (Reimer et al. 2024). Early reports indicate substantial coral-cover losses and severe declines of branching and foliose species such as *Agaricia tenuifolia* and *Acropora palmata* (Cáceres and Sánchez 2015; Neely et al. 2024; Birkart and Álvarez-Filip 2025; Manzello et al. 2025; Bon et al. 2026). Together, these lethal events have pushed several species toward the verge of extinction (Wilson and Edmunds 2026) and are likely driving a regime shift toward simplified communities dominated by small, resilient colonies, compounding long-term declines since the 1970s.

A close examination reveals that different species are particularly susceptible to each of these disturbances. SCTLD is associated with high mortality in species with stress-tolerant life-history traits, characterized by large, massive colonies, slow growth, and diffuse spawning

with high fecundity, whose contribution to the formation of the physical framework of the reef is essential for reef ecosystem functioning (Alvarez-Filip et al. 2011; Darling et al. 2012). By contrast, species most susceptible to thermal stress are primarily those with thin tissues and plate-like colony morphologies, which typically exhibit weedy life-history traits. Meanwhile, massive species, considered stress-tolerant, have shown greater population-level resistance to thermal stress mortality, although they often display early signs of stress and partial tissue mortality (Darling et al. 2012; Manzello et al. 2018; Muñiz-Castillo et al. 2024). However, the combined effect of both stressors is likely to increase coral susceptibility and mortality rates across multiple species. For example, coral bleaching primarily leaves colonies immunocompromised, thereby increasing their vulnerability to disease outbreaks and further exacerbating the effects of large-scale thermal stress events (Reed et al. 2010; Palmer et al. 2011; Raymundo et al. 2025). Furthermore, we can reasonably assume that disease colonies are also more vulnerable to heat stress, as diseases disrupt the physiological balance of the healthy coral microbiota, leaving corals immunocompromised both internally and externally (Daszak et al. 2001; Aeby et al. 2020). Therefore, we might expect that species exhibiting null, low, or moderate susceptibility to SCTLD will be more prone to bleaching-induced mortality (Cróquer et al. 2021; Bon et al. 2026).

The net outcome of both stressors occurring simultaneously is therefore likely to trigger substantial changes in coral community composition, affecting a wide range of species with different life-history strategies and varying susceptibility to either or both stressors. This could alter the trait composition of ecological communities, compromise the calcification potential, and undermine reefs' capacity to maintain their three-dimensional structure (Alvarez-Filip et al. 2013; González-Barrios and Álvarez-Filip 2018; Urbina-Barreto et al. 2021). In 2023, Banco Chinchorro in the Mexican Caribbean was hit by SCTLD and the 2023–2024 marine heatwave in quick succession. SCTLD was first reported in early 2023, just a few months before anomalously high thermal stress spread across the western Caribbean. Here, we analyzed data from eight reef sites distributed across the bank, sampled before the disturbances, during the onset of the SCTLD outbreak, and after the events, to assess coral-community responses to their combined impacts and the consequences for reef functioning, quantified as changes in community calcification. This study, therefore, aims to quantify shifts in assemblage composition and density, the processes of simplification and homogenization, and the resulting effects on calcification rates across morpho-functional groups.

Methods

Study sites and temporality

The Banco Chinchorro Biosphere Reserve is the largest coral reef system in Mexico, with a total area of more than 700 km² (Hernández 2008). Chinchorro was designated a Biosphere Reserve in 1996; it has restricted access, limited human infrastructure, and regulated tourism

(Beltrán-Torres et al. 2003; Torruco and González-Solís 2024). Until late 2022, Chinchorro had not been affected by SCTLD, which reached the Mexican Caribbean in 2018 and spread throughout the continent and Cozumel island in only a few months (Alvarez-Filip et al. 2019, 2022). However, in early 2023, the first reports of the disease were made by recreational divers, which were later confirmed by MPA authorities and scientists. SCTLD spread rapidly across the entire bank, and by May 2023, the disease was confirmed throughout the bank (Fig 1). Just as the disease outbreak was reaching its peak, sea temperature began to rise rapidly, peaking at ~18 DHW by October 2023 (Fig. 1).

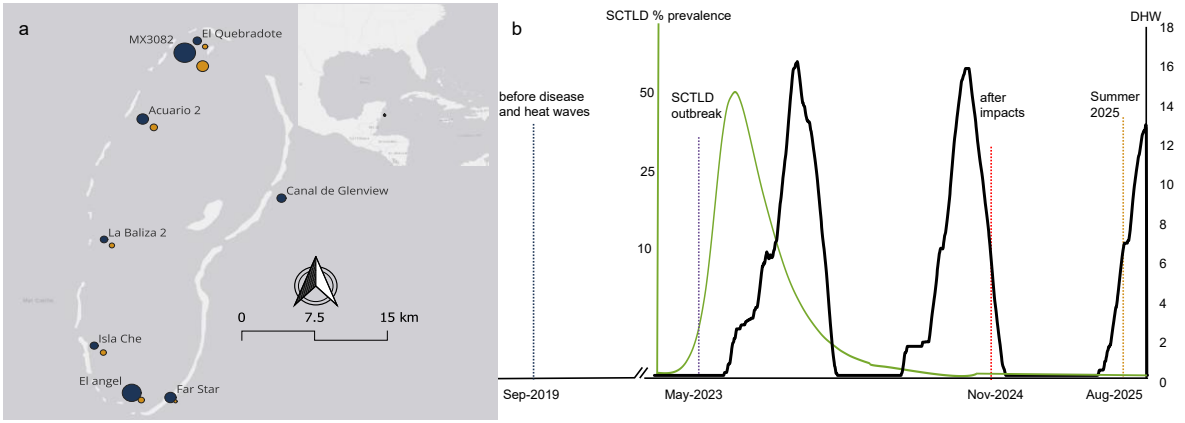


Figure 1. **Location of the study sites and graphical representation of SCTLD outbreak prevalence and the 2023–2024 marine heatwaves in the Banco Chinchorro Biosphere Reserve**. (a) Geographic location of the monitored sites, showing coral community calcification during the first (2019) and last (2025) periods of survey. The inset (upper right) shows the Caribbean region, with the location of Banco Chinchorro marked by a red asterisk. (b) Monitoring periods are shown in vertically dashed lines. The Degree Heating Weeks are shown in the black curve (data obtained NOAA Coral Reef Watch), and the green curve represents the empirical trajectory of SCTLD epizootic constructed using the patterns depicted in Guzman-Urieta et al. 2026. In panel (a), circle size denotes coral colony density (colonies m^2) at each site, and color indicates the monitoring period: September 2019 (blue) and August 2025 (gold). In panel (b), dashed-line colors correspond to the monitoring periods: September 2019 (blue), May 2023 (purple), November 2024 (red), and August 2025 (gold). Left and right axes indicate disease prevalence (%) and degree heating weeks (DHW), respectively.

Field surveys

To evaluate the impacts of the disease outbreak and heat stress we surveyed eight reef sites around Banco Chinchorro (Figure 1) over four periods: Before disturbances (September 2019), during the onset of SCTLD outbreak (May 2023), November 2024 just when thermal stress was diminishing, and in August 2025 (Fig. 1b). At each site, 3 to 11 (mean = 5.48; SD = 1.51) 10 × 1 m belt transects were surveyed following the AGRRA coral protocol to analyze the coral community (Lang et al. 2010). The following data were recorded for each coral colony within the transect, species name, colony size (maximum diameter, perpendicular diameter, and height), percentage bleaching, percentage mortality (recent, transitional, and old), and presence of SCTLD and other diseases. We calculated the prevalence of SCTLD at each site and across all coral species. Colonies with 100% mortality were recorded across all

periods. Colonies exhibiting 100% recent or transitional mortality in the 2023 monitoring were attributed to SCTLD (see Estrada-Saldivar et al. 2021).

Additionally, targeted searches for *Dendrogyra cylindrus* colonies used manta tow (towed snorkel transects at ~1 knot covering ~1 km per tow), roving dives (systematic free-swimming searches along depth contours) and freediving (breath-hold dives into shallow crevices). Identified colonies were tagged at sites, minimizing disturbance to their integrity.

Data analyses

To provide a comprehensive representation of the impacts of both events, we examine changes in colony density, condition, coral community structure, and coral community calcification. Coral density was represented as the number of colonies per m^2 and assessed across all four periods. Furthermore, we also calculated the percentage change in density per site for different taxa (genera or species) and at the regional level was estimated between the survey from before impacts (September 2019) and the one immediate after impacts (November 2024), as this survey was the most the representative of the condition immediate after the disease outbreak and the 2023-24 heat stress.

Coral community structure was described by means of Hill numbers that represent three different axes of diversity: species richness (q0), diversity (q1), and dominance (q2). Calculations were performed on transect-level abundances (colonies), with rarefaction/extrapolation curves appended where necessary to compare samples of differing sizes (iNEXT package; (Hsieh et al. 2016)). Changes in coral community composition between surveyed periods were assessed using bidimensional ordinations and Bray-Curtis dissimilarities. For this procedure, data were square root transformed (Legendre and De Cáceres 2013), and ordinations were prepared with multiple restarts until stress stabilized. Permutational multivariate analysis of variance (PERMANOVA; (Legendre and De Cáceres 2013)) with 999 permutations was used to assess separation between years per site. Multivariate analyses used Bray-Curtis distance in RStudio (4.1.1). The resulting similarity matrices were further analyzed using principal component analysis (R Core Team 2025).

To estimate the effects on the physical functionality of coral reefs, we calculated the potential calcification (i.e., kg $CaCO_3$ m^{-2} yr^{-1}) of the coral assemblages for each period using the sum of the calcification rate of each colony proportional to the sampled reef area (m^2). We used reef-level estimates to calculate the mean coral community calcification for each period. For this, we estimated the calcification rate of each colony within each study site, considering the size, mean annual growth rate (cm yr^{-1}), mean skeletal density (g cm^{-3}), and morphological growth of each species, following the methodology described by Gonzalez-Barrios & Álvarez-Filip 2018. The latter variables were obtained from the literature (see Gonzalez-Barrios & Álvarez-Filip 2018) thus, our calcification estimates do not account for local environmental or ecological variations. For each colony, only the percentage of live tissue

(accounting for disease and partial mortality effects) was considered to adjust the calcification rate (i.e., effective calcification = theoretical species calcification × proportion of live tissue). For each period, the calcification rate was estimated as the annual volume of calcium carbonate accumulated by the colony's living tissue, based on data from the resampled sites.

To evaluate disparities in density, Hill indices, and coral community calcification across reef communities, we employed the non-parametric Kruskal-Wallis test due to non-homogeneous variance in the data. *Post hoc* multiple comparisons were performed using the nonparametric Dunn test, adjusted for Bonferroni. Boxplots and density diagrams show medians and quartiles. For variables of interest (density, Hill indices, coral community calcification rates), central tendency and dispersion measures (median and interquartile range) are presented, with the corresponding statistical tests reported in the figure captions.

Results

Changes in coral condition and colony density

In total, we censused 8,299 coral colonies across eight sites and four periods. In 2019, before the impacts, 2,075 colonies were surveyed, and only a minimal proportion of diseases and full colony mortality, while only a small proportion showed signs of bleaching (Fig. 2a). During the first months of 2023 3,325 colonies were surveyed, and 19.84% already showed signs of SCTLD, but only 3.31% presented evidence of thermal stress (Fig. 2b). From the 1,507 colonies surveyed in November 2024, 35.76% showed damage dominated by heat stressed colonies (34.92%) whereas SCTLD was practically absent (0.39%) (Fig. 2b). Although thermal stress persisted in 2025 (Fig. 1b), the proportion of stressed colonies in the 1,016 surveyed colonies decreased notably to 14.86%, contrasting with the high susceptibility observed in 2024, possibly amplified by prior SCTLD impact. Despite fewer stressed colonies monitored in 2025 than in 2024, it is important to note that statistical tests detected no significant differences in density (Table S1) between these years.

Our results revealed a halving of coral colony numbers on Banco Chinchorro attributable to the combined impacts of SCTLD and the heatwaves. Intense and prolonged disturbances reduced colony density from six to fewer than three colonies per m² (Kruskal–Wallis test: $\chi^2 = 58.761$, df = 3, $p \leq 0.001$; Fig. 2). Post hoc analysis indicates that in November 2024 differed significantly from September 2019 and from May 2023 ($p < 0.001$). Similarly, August 2025 differed significantly from September 2019 and May 2023. No differences September 2019, nor between November 2024 and August 2025 after Bonferroni correction, as shown in Figure 2b.

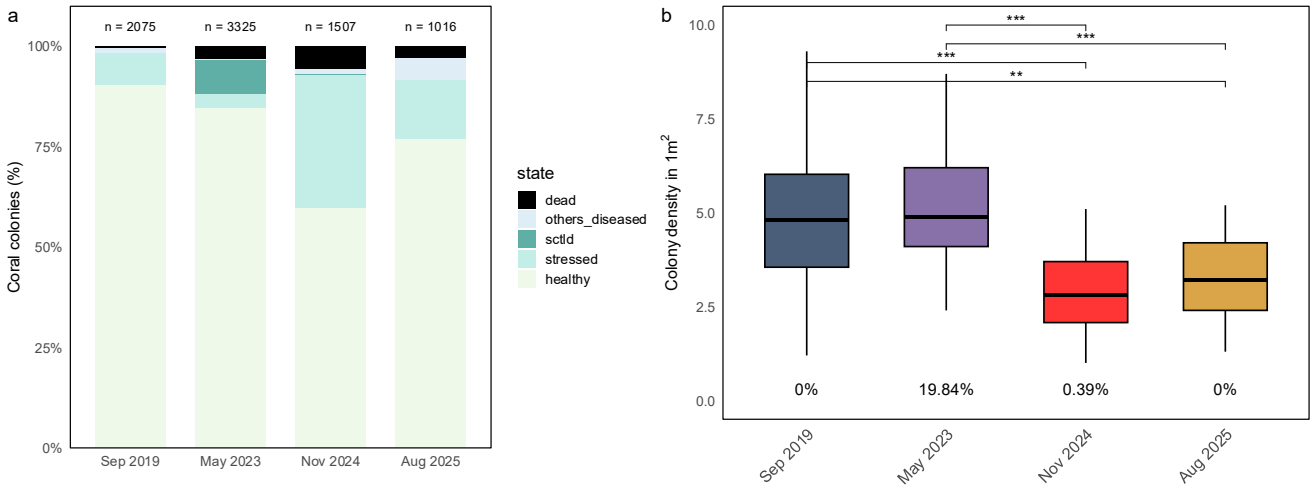


Figure 2. Changes in coral colony density and health status at Banco Chinchorro across monitoring periods. (a) Proportional composition of colony health states, including healthy, stressed colonies showing signs of SCTLD, dead colonies, and colonies affected by other diseases. (b) Boxplots of coral colony density (colonies m²) among periods. In panel (a), colors represent the proportion of colonies in each condition: healthy (pale mint green), stressed (pastel aqua), showing signs of SCTLD (teal), dead (black), and affected by other diseases (ice blue). In panel (b), boxplots show the median (horizontal line), first and third quartiles (box height), and minimum and maximum values (whiskers; outliers excluded). Percentages below the boxplots indicate the proportion of colonies exhibiting signs of SCTLD in each period. Colors correspond to the monitoring period: September 2019 (blue), May 2023 (purple), November 2024 (red), and August 2025 (gold). Estimates are based on colony density per m².

Impacts on coral diversity and community composition

All three metrics of effective diversity changed substantially due to the impacts of the SCTLD outbreak and the two marine heat waves. Prior to the disturbances, reefs exhibited the highest values for species richness, diversity, and evenness (as opposed to dominance; Fig. 3a). Post hoc analysis indicates a significant ($p < 0.001$) decrease in all these metrics by May 2023, during the outbreak of SCTLD, as a result of the rapid loss of some of the most highly susceptible species to SCTLD, such as *Meandrina* spp., *Diploria labyrinthiformis*, and *Pseudodiploria* spp., which was accompanied by a concomitant increase in dominance and decreases in evenness. The November 2024 period accentuated these patterns of loss, with the most pronounced reduction in diversity and the greatest decline in evenness, reflecting a considerable dominance of species that were better able to withstand multiple stressors, primarily those in the genus *Porites* (Fig. 4). In this regard, this period marks the greatest loss of diversity and variability, alongside marked increases in dominance within the coral community structure (Fig. 3b). The overall trend in this November 2024 period indicates a massive loss of species and a reduction in ecological complexity. By 2025, the consequences of the disturbances show a pattern in which all reefs maintained very low values of richness, diversity, and evenness. Therefore, this period exhibits a simplified community structure very similar to that of the November 2024 period (Fig. 3).

The principal component analysis showed a drastic homogenization in coral communities following the disturbances. Multivariate analysis of variance (PERMANOVA, Bray–Curtis distance, 999 permutations) revealed significant differences in coral genera composition among the four evaluated periods (September 2019, May 2023, November 2024 and August 2025) across all analyzed sites ($p = 0.001$). The time factor explains 9.3% of the total variation in community structure (see supplementary Table 2). In September 2019, the sites exhibited distinct community compositions, primarily explained by exposure levels: exposed reefs (Canal de Glenview and Far Star) showed lower diversity and richness than protected ones (e.g. Acuario 2 and MX3082; Fig. 3b), with differences driven by genera such as *Eusmilia, Dichocoenia, Isophyllia, Meandrina, Dendrogyra, Mycetophyllia, Diploria and Pseudodiploria.* Following the disturbances in May 2023 and November 2024, severe and statistically significant reductions in diversity and variability were observed (Fig. 3b), linked to the loss of rare and infrequent genera; this was reflected in a contraction of multivariate space, with 2024–2025 communities converging in a reduced multidimensional space along PC1. These patterns indicate a clear trend toward communities with increased relative abundance of *Porites* and *Stephanocoenia*, with no evidence of significant recovery (Fig. 3b).

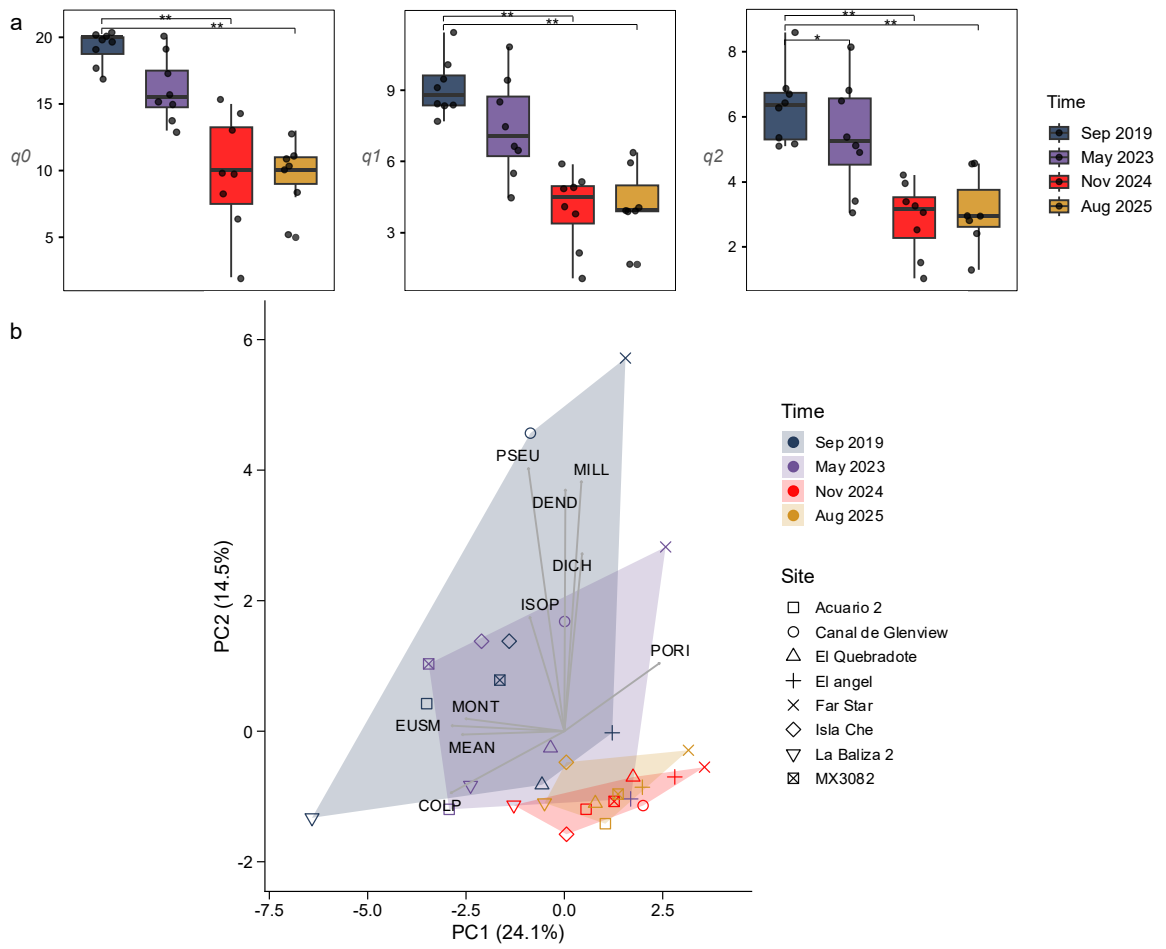


Figure 3. Changes in coral community composition across eight reef sites of the Banco Chinchorro Biosphere Reserve (RBBC) among monitoring periods. a) Boxplots showing patterns of alpha and beta diversity at the species level across site × period, based on Hill numbers (*q0*, *q1* and *q2*) (q0: $\chi^2 = 22.866$, df = 3, $p < 0.001$; q1: $\chi^2 = 20.525$, df = 3, $p < 0.001$; q2: $\chi^2 = 19.295$, df = 3, $p < 0.001$). b) Principal component biplots of coral assemblages at the genus level as a function of site × period (PERMANOVA, Bray–Curtis; F = 9.40, $R^2 = 0.093$, p = 0.001). In panel (a), geometric symbols represent reef sites, while vectors indicate the absolute contributions of colony density by genus. Color polygons depict the multivariate dispersion of sites within each period in the PC1–PC2 space. Genus codes are as follows: COLP (*Colpophyllia*), DEND (*Dendrogyra*), DICH (*Dichocoenia*), EUSM (*Eusmilia*), ISOP (*Isophyllia*), MEAN (*Meandrina*), MILL (*Millepora*), MONT (*Montastraea*), PORI (*Porites*), PSEU (*Pseudodiploria*), STEP (*Stephanocoenia*). In panel (b), points represent individual reef sites, while boxplots show the median (horizontal line), first and third quartiles (box height), and minimum and maximum values (whiskers). In both panels, colors correspond to the monitoring period: September 2019 (blue), May 2023 (purple), November 2024 (red), and August 2025 (gold).

Patterns of loss at genera level

A detailed analysis of genus-level patterns reveals different responses to combined disturbances from the SCTLD outbreak and thermal stress when comparing changes in density, occurrence and percentage of dead tissue for the main genera in Chinchorro (Fig. 4). The first group comprises genera that experienced complete losses in both colony density and presence at previously recorded sites. Notably, *M. meandrites* exhibits a concerning trend of likely complete extirpation from Chinchorro. In 2023, 76 colonies were recorded, of which 32 were already dead, 17 were diseased, and only 27 were healthy. By 2024, only five colonies were recorded, all dead and exhibiting signs of old mortality. Likewise, for *Mycetophyllia,* we recorded nine healthy colonies in 2019, and by 2024, only one dead colony was observed in the surveys.

For *D. cylindrus*, the data reveal an extreme population decline. In 2019, 52 colonies were recorded using different survey protocols (see methods). During the outbreak of SCTLD in April–May 2023 surveys, standardized monitoring detected only two colonies and tagged 39 additional. Of these, 14 showed signs of SCTLD and 25 appeared healthy at the time. By August 2025, all but one tagged colony was found to be dead. In addition, we found one additional colony with ~1% live tissue. Overall, these results indicate a near-total loss of the 2019-recorded colonies within the monitored area.

A second group includes taxa that underwent drastic and widespread reductions in density and disappeared from numerous sites where they had previously been recorded (Fig. 4a-b). This group includes many important reef-building corals, such as *P. strigosa*, *E. fastigiata*, *M. alcicornis*, *M. complanata*, *A. palmata*, *A. cervicornis*, *D. labyrinthiformis*, *D. stokesi*, and *C. natans*, which have historically been dominant in Chinchorro and elsewhere in the Caribbean. Interestingly, the mean percentage of partial mortality in live colonies across these species did not differ substantially between 2019 and 2024 (Fig. 4c), suggesting that these species experienced little partial mortality due to the impacts, and that most colonies that died from the combined effects of disturbances suffered complete tissue mortality. Additionally, the remaining colonies in 2024 exhibited smaller sizes compared to 2019 (Fig.

S1), indicating greater susceptibility of larger colonies, a pattern consistent with findings from other Caribbean regions.

The third group, formed by genera such as *Agaricia*, *Montastraea*, *Orbicella* and *Siderastrea*, exhibited moderate reductions in colony density (33–50%), although the occurrence of these genera persisted at most reef sites. Yet a consistent increase in the percentage partial mortality between 2019 and 2024 was evident for *Agaricia* and *Siderastrea*, while *M. cavernosa* and *Orbicella* seems to be relatively less affected, as the % of colony mortality in 2019 remained similar to that in 2024 (Fig. 4). Lastly, there is a group of genera, where minimal effects were observed due to the impacts, in this group, mainly the genus *Porites*, but also to some degree the genus *Stephanocoenia*, emerged as the winners in the system. In the case of *Porites,* it maintained its presence at all monitored sites and increased in density by 19.4%, with no significant change in the percentage of partial tissue mortality.

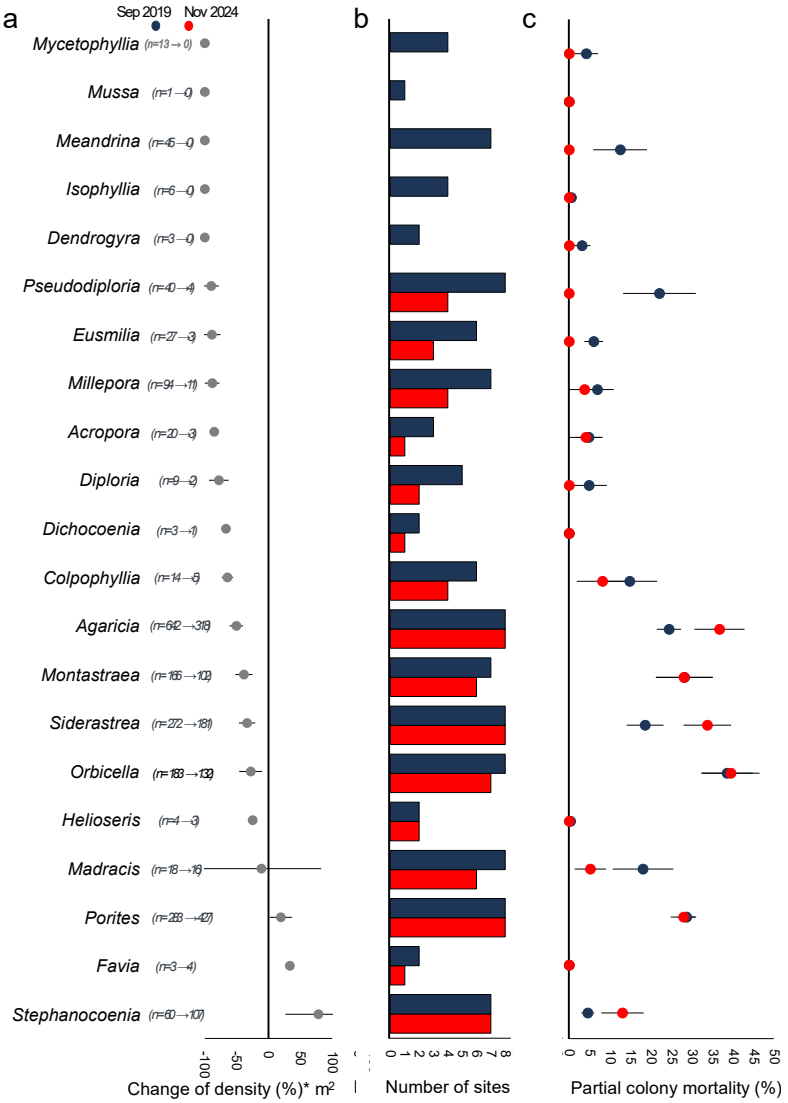


Figure 4. Changes at the genus level Banco Chinchorro between the *pre*-period (2019; blue) and the *post-impact* period (2024; red). (a) Percentage change in colony density by genus between periods. The total number of colonies (n) is indicated for each genus, with values shown as *Sep 2019 → Nov 2024*. The solid vertical black line denotes zero change; points to the left indicate declines in colony density, whereas points to the right indicate increases relative to the *before* period. (b) Spatial comparison of reef sites where each genus was present at each period. (c) Comparison of mean partial colony mortality (%) between periods. In panels (a) and (c), circles represent mean values and vertical lines indicate the standard error of the mean.

Coral community calcification

Although there were marked significant differences in community calcification between 2019 and 2024, no such significance was observed for the estimated in 2024 and 2025 (Fig. 5a; Table S3). This indicates that, nine months after the most severe disturbances ended, coral communities continued to calcify at approximately the same level as at the end of the 2023–2024 heatwave. Calcification community thus declined from an average of 6.68 to 3.98 kg $CaCO_3$ m^{-2} $year^{-1}$ (Fig. 5). Moreover, by summer 2025, the coral community calcification showed a pattern of increased variance, suggesting that most surviving coral colonies across genera calcify at low rates, resulting in distributions skewed towards low values. Finally, there was significant variation in net carbonate production among coral communities and reef types, according to the Kruskal–Wallis test (χ^2=27.902, df=6, p≤0.001). Overall, this widespread reduction in community calcification reflects not only shifts in coral community structure but also a diminished reef-building capacity.

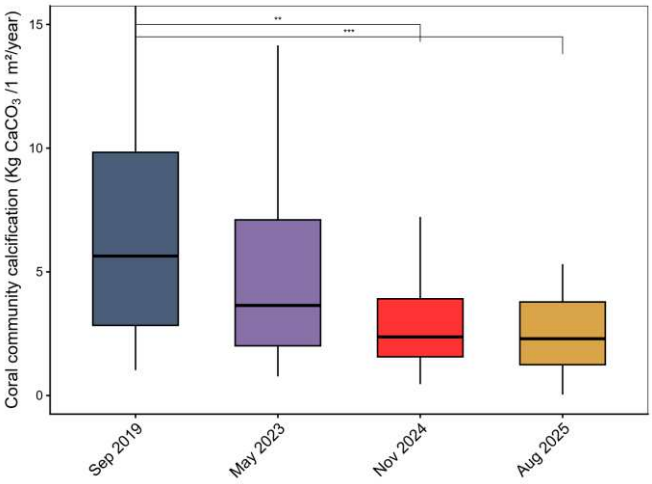


Figure 5. **Decline in coral community calcification ($CaCO_3$) across the four evaluated periods**. Boxplot showing significant variation among reef types (Kruskal–Wallis test, χ^2= =27.902, df=6, p≤0.001). Significant differences are evident between 2019 and 2024, with stabilisation in 2024 and 2025, and increased variance in summer 2025.

Discussion

Heatwaves are rapidly destabilizing marine ecosystems, a trend worsened by disease outbreaks and other human impacts (Smith et al. 2023; Trégarot et al. 2024). Our results

show that the most extreme heat stress on record in the Caribbean, combined with a highly lethal disease outbreak, had catastrophic effects on Banco Chinchorro reefs. Coral abundance fell by roughly half over just two years, with some species such as *D. cylindrus* and *M. meandrites* disappearing from our surveys. Key reef-building acroporids, large massive species (e.g., *P. strigosa* and *C. natans*), and foliose *A. tenuifolia* experienced severe population declines. Losses were nonrandom across size classes, disproportionately affecting larger colonies. In some species, the mean colony size declined by nearly 50%. Overall, mortality transformed communities into simpler, more similar assemblages dominated by opportunistic, stress-tolerant corals, largely *P. astreoides*, and drove an almost 40% decline in community calcification, a proxy for the capacity to accrete calcium carbonate and maintain reef structure.

In the Caribbean, disease outbreaks have been a leading driver of coral decline, with ecological impacts comparable to large-scale bleaching events (Harvell et al. 2007; González-Barrios et al. 2023). We show that the combined effects of these two mortality drivers likely amplify damage beyond their independent impacts. Evidence includes extreme losses of brain corals, which are highly susceptible to SCTLD, exceeding 90% in our study, substantially higher than mortality attributed to only SCTLD (30–70%) in the same region (Alvarez-Filip et al. 2022; Camacho-Vite et al. 2022). Furthermore, the preferential loss of large colonies indicates that consecutive intense disturbances increase vulnerability, probably via interactions between physiological stress and disease: disease-driven shifts in coral microbiota can weaken host responses to prolonged thermal stress (McDevitt-Irwin et al. 2017; Putnam 2021), suggesting disease may degrade physiological condition and raise susceptibility to heat (Bruno et al. 2007; Muller et al. 2008), although our data cannot fully dissociate their relative contributions. This resulted in a subset of remnant colonies, predominantly small in size (Fig. S1), with no recent signs of tissue loss. The persistence of smaller colonies may relate to reduced pathogen exposure, lower initial stress loads, and greater physiological acclimation capacity (Li et al. 2021; Palacio-Castro et al. 2025). Overall, high-intensity, successive disturbances reduce coral condition (Fig. 2) and recovery potential, increase mortality (Fig. 4), and drive functional reef degradation.

The SCTLD outbreak and the 2023 heatwave not only reduced coral abundance but also consistently restructured community composition over a two-year period. Our results show that even species with relatively stress-tolerant life-history traits, such as *Agaricia* spp. And *Siderastrea* spp., experienced substantial abundance declines, albeit less severe than those in highly susceptible corals. These shifts were accompanied by a marked decline in effective diversity, with pronounced reductions in species richness (from 19 to 9 species) and evenness, signaling strong community simplification at the regional scale (Fig. 3). In parallel, the increase in relative abundance of species like *P. astreoides* and *S. intersepta* aligns with a replacement process toward more stress-tolerant taxa. These species help maintain live coral cover and occupy available substrate, potentially limiting macroalgal colonization and

reducing substrate exposure to bioerosion processes. However, their contribution is primarily limited to space occupation, without compensating for the loss of structural volume and three-dimensional complexity provided by branching, foliose, and massive corals (Logan 1984; Alvarez-Filip et al. 2013; Kuffner and Toth 2016; Toth et al. 2018; Lima et al. 2023). Overall, these results indicate that disturbances do not affect all species equally, favoring certain life-history traits while others decline or disappear. Under this framework, these dynamics may indicate either a transient phase in the coral community following disturbance or a shift toward an alternative stable state characterized by the dominance of disturbance-tolerant species under increasingly frequent and intense perturbation regimes, ultimately depending on the frequency and intensity of future stress events (Baum et al. 2023; Turner and Seidl 2023; Edmunds 2024).

The disproportionate loss of massive corals with large colonies, as well as rare and low-abundance species, resulting in a restructuring of the coral community and its functional traits. Consistent with these changes, community calcification declined by ~40%, with 2025 data showing no recovery toward pre-disturbance levels (Fig. 5). This pattern indicates persistent deterioration in reef-building potential that cannot be explained solely by shifts in coral cover, suggesting that the loss of key functions—rather than live tissue area reduction *per se*—is limiting reef accretion (Álvarez-Filip et al. 2013). In this context, reductions of this magnitude have been linked to transitions toward low or even zero net growth states in coral reefs, particularly under simultaneous disturbance scenarios (Perry et al. 2015, 2018), implying that the studied communities may are transitioning into erosional regimes more quickly than previously predicted, a threshold projected to be reached in over a decade (Perry et al. 2025).

This deterioration appears driven by multiple mechanisms acting in concert. Selective mortality disproportionately eliminated major carbonate production contributors, while community composition shifts favored encrusting or plate-like morphologies, smaller colonies, and overall reduced carbonate production (Álvarez-Filip et al. in review). Moreover, although some functionally important colonies survived, their exposure to consecutive thermal stress events suggests physiological trade-offs, where greater bleaching tolerance may be associated with reduced calcification and carbon translocation (Yao et al. 2023; Grottoli et al. 2025). Consequently, even if coral cover stabilizes or partially recovers, reef functional integrity may remain compromised: loss of builder corals diminishes habitat structural complexity, limiting refuges and settlement substrates for organisms and affecting associated communities (Kayal et al. 2012; Richardson et al. 2018). Degradation of branching corals into unstable fragments exacerbates these effects by hindering recruitment and promoting ecosystem degradation trajectories (Dove et al. 2020; Hoegh-Guldberg et al. 2023). Ultimately, sustained reductions in coral community calcification carry direct geomorphological implications, as diminished capacity to maintain physical reef-framework

growth heightens vulnerability and reduces reefs' ability to keep pace with sea-level rise (Perry et al. 2013, 2015, 2025; Perry and Morgan 2017).

Certainly, the situation facing the reefs of Banco Chinchorro represents an unprecedented loss. Protecting remaining coral communities, together with interventions aiming to restore the populations of affected species, is essential to halt their degradation not only in this biosphere reserve but across the wider Caribbean region (Knowlton et al. 2021; Reimer et al. 2024; Suggett et al. 2024). A range of direct interventions can help maintain or recover the capacity of these systems to respond to disturbances, particularly in contexts where adaptive limits have been exceeded (Mora et al. 2016); for example, branching coral restoration has been shown to promote the recovery of coral calcification at the community level over relatively short time scales (<5 years) (Lange et al. 2024; Toth et al. 2025). In this context, it is essential to prioritize ecological restoration actions aimed at conserving species, their interactions, and reef functioning (Barredo et al. 2021; Suggett et al. 2024). One example of these efforts is the rescue of *D. cylindrus* coral fragments by researchers and reserve staff in Chinchorro, with some fragments maintained in aquaria and others left in situ. Given the severity of the 2023–2024 disturbances, combining species-based rescue with broader, function-oriented restoration offers a strong opportunity to rebuild reef structure and processes (Darling et al. 2017). Integrative restoration that includes multiple functional groups can restore population abundance, interaction networks, and key biogeochemical and ecological processes (e.g., carbonate accretion, nutrient cycling, herbivory), and can be implemented at the population, community, or ecosystem level depending on objectives (Croquer et al. 2026). Centering efforts on ecosystem functioning increases the likelihood of long-term persistence (Peixoto and Voolstra 2023), while complementary measures such as balanced tourism policies and community-scale restoration can further aid recovery (Lamb et al. 2015; Rinkevich 2021; De et al. 2022). Crucially, all these measures will be effective only in the long term if accompanied by substantial reductions in local and global stressors.

Data availability

The data that support the findings of this study are available within the paper and in the Supplementary Information. Any additional data not found within are available from the authors upon request.

References

Aeby GS, Howells E, Work T, Abrego D, Williams GJ, Wedding LM, Caldwell JM, Moritsch M, Burt JA (2020) Localized outbreaks of coral disease on Arabian reefs are linked to extreme

temperatures and environmental stressors. Coral Reefs 39:829–846 https://doi.org/10.1007/s00338-020-01928-4

Alvarez-Filip L, Carricart-Ganivet JP, Horta-Puga G, Iglesias-Prieto R (2013) Shifts in coral-assemblage composition do not ensure persistence of reef functionality. Scientific reports 3:3486 https://doi.org/10.1038/srep03486

Alvarez-Filip L, Dulvy NK, Côté IM, Watkinson AR, Gill JA (2011) Coral identity underpins architectural complexity on Caribbean reefs. Ecological Applications 21:2223–2231 https://doi.org/10.1890/10-1563.1

Alvarez-Filip L, Estrada-Saldívar N, Pérez-Cervantes E, Molina-Hernández A, González-Barrios FJ (2019) A rapid spread of the stony coral tissue loss disease outbreak in the Mexican Caribbean. PeerJ 7:e8069 https://doi.org/10.7717/peerj.8069

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 5:440 https://doi.org/10.1038/s42003-022-03398-6

Baum JK, Claar DC, Tietjen KL, Magel JMT, Maucieri DG, Cobb KM, McDevitt-Irwin JM (2023) Transformation of coral communities subjected to an unprecedented heatwave is modulated by local disturbance. Sci Adv 9:eabq5615 https://doi.org/10.1126/sciadv.abq5615

Beltrán-Torres AU, Muñoz-Sánchez L, Carricart-Ganivet JP (2003) Effects of hurricane Keith at a patch reef on Banco Chinchorro, Mexican Caribbean. Bulletin of Marine Science 73:187–196

Birkart LV, Álvarez-Filip L (2025) 2023 global heatwave causes mass mortality of a keystone coral on shallow Western Atlantic reefs. Iscience 28: https://doi.org/10.1016/j.isci.2025.113722

Bon M, Kayal M, Dromard CR, Desrosiers C (2026) Impact of the 2023 coral bleaching event in Martinique, Eastern Caribbean. Coral Reefs 45:167–180 https://doi.org/10.1007/s00338-025-02694-x

Cáceres SM, Sánchez JA (2015) Growth strategies of an abundant reef-building coral in the southern Caribbean (Undaria tenuifolia). Revista de la Academia Colombiana de Ciencias Exactas, Físicas y Naturales 39:348–357 https://doi.org/10.18257/raccefyn.195

Clark AS, Williams SD, Maxwell K, Rosales SM, Huebner LK, Landsberg JH, Hunt JH, Muller EM (2021) Characterization of the microbiome of corals with stony coral tissue loss disease along Florida's coral reef. Microorganisms 9:2181 https://doi.org/10.3390/microorganisms9112181

Cróquer A, Weil E, Rogers CS (2021) Similarities and differences between two deadly Caribbean coral diseases: white plague and stony coral tissue loss disease. Frontiers in Marine Science 8:709544 https://doi.org/10.3389/fmars.2021.709544

Darling ES, Alvarez-Filip L, Oliver TA, McClanahan TR, Côté IM (2012) Evaluating life-history strategies of reef corals from species traits. Ecology Letters 15:1378–1386 https://doi.org/10.1111/j.1461-0248.2012.01861.x

505 Daszak P, Cunningham AA, Hyatt AD (2001) Anthropogenic environmental change and the
506 emergence of infectious diseases in wildlife. Acta tropica 78:103–116

507 Denis V, Ribas-Deulofeu L, Sturaro N, Kuo C-Y, Chen CA (2017) A functional approach to the
508 structural complexity of coral assemblages based on colony morphological features.
509 Scientific reports 7:9849 https://doi.org/10.1038/s41598-017-10334-w

510 Díaz-Talamantes R, Pérez-Cervantes E, Álvarez-Filip L (2025) Juveniles reveal natural recovery
511 potential of Caribbean coral species after a widespread disease die-off. Communications
512 Earth & Environment 6:1010 https://doi.org/10.1038/s43247-025-02975-x

513 Dove SG, Brown KT, Van Den Heuvel A, Chai A, Hoegh-Guldberg O (2020) Ocean warming and
514 acidification uncouple calcification from calcifier biomass which accelerates coral reef
515 decline. Communications Earth & Environment 1:55 https://doi.org/10.1038/s43247-020-
516 00054-x

517 Edmunds PJ (2021) Recruitment hotspots and bottlenecks mediate the distribution of corals on a
518 Caribbean reef. Biology letters 17: https://doi.org/10.1098/rsbl.2021.0149

519 Edmunds PJ (2024) Why keep monitoring coral reefs? BioScience 74:552–560
520 https://doi.org/10.1093/biosci/biae046

521 Estrada-Saldívar N, Jordán-Dalhgren E, Rodríguez-Martínez RE, Perry C, Alvarez-Filip L (2019)
522 Functional consequences of the long-term decline of reef-building corals in the Caribbean:
523 evidence of across-reef functional convergence. Royal Society Open Science 6:
524 https://doi.org/10.1098/rsos.190298

525 González-Barrios FJ, Álvarez-Filip L (2018) A framework for measuring coral species-specific
526 contribution to reef functioning in the Caribbean. Ecological indicators 95:877–886
527 https://doi.org/10.1016/j.ecolind.2018.08.038

528 González-Barrios FJ, Cabral-Tena RA, Alvarez-Filip L (2021) Recovery disparity between coral
529 cover and the physical functionality of reefs with impaired coral assemblages. Global
530 Change Biology 27:640–651 https://doi.org/10.1111/gcb.15431

531 González-Barrios FJ, Estrada-Saldívar N, Pérez-Cervantes E, Secaira-Fajardo F, Álvarez-Filip L
532 (2023) Legacy effects of anthropogenic disturbances modulate dynamics in the world's
533 coral reefs. Global Change Biology 29:3285–3303 https://doi.org/10.1111/gcb.16686

534 Grottoli AG, Hulver AM, Thurber RV, Toonen RJ, Schmeltzer ER, Kuffner IB, Barott KL, Baums
535 IB, Castillo KD, Chapron L (2025) Future of coral bleaching research. BioScience 75:585–
536 598 https://doi.org/10.1093/biosci/biaf066

537 Guzmán-Urieta EO, Jordán-Dahlgren E, Alvarez-Filip L (2026) Epidemiological analysis reveals
538 coral species affected by stony coral tissue loss disease present a similar epizootic
539 progression despite differences in susceptibility and population impact. PloS one
540 21:e0339054 https://doi.org/10.1371/journal.pone.0339054

541 Hernández (2008) Hernández: El manejo sostenible de los recursos... - Google Académico.
542 https://scholar.google.com/scholar_lookup?&title=El%20manejo%20sostenible%20de%20l
543 os%20recursos%20pesqueros%20de%20la%20Reserva%20de%20la%20Biosfera%20Banc
544 o%20Chinchorro&publication_year=2008&author=Hern%C3%A1ndez%2CA&author=Ro

dr%C3%ADguez-Zaragoza%2CFA&author=Garc%C3%ADa%2CMC&author=Castro%2CJM&author=Medina-Flores%2CJ

Hoegh-Guldberg O, Skirving W, Dove SG, Spady BL, Norrie A, Geiger EF, Liu G, De La Cour JL, Manzello DP (2023) Coral reefs in peril in a record-breaking year. Science 382:1238–1240 https://doi.org/10.1126/science.adk4532

Hsieh TC, Ma KH, Chao A (2016) iNEXT: an R package for rarefaction and extrapolation of species diversity (H ill numbers). Methods Ecol Evol 7:1451–1456 https://doi.org/10.1111/2041-210X.12613

Kayal M, Vercelloni J, Lison de Loma T, Bosserelle P, Chancerelle Y, Geoffroy S, Stievenart C, Michonneau F, Penin L, Planes S (2012) Predator crown-of-thorns starfish (Acanthaster planci) outbreak, mass mortality of corals, and cascading effects on reef fish and benthic communities. https://doi.org/10.1371/journal.pone.0047363

Kuffner IB, Toth LT (2016) A geological perspective on the degradation and conservation of western Atlantic coral reefs. Conservation Biology 30:706–715 https://doi.org/10.1111/cobi.12725

Lang JC, Marks KW, Kramer PA, Kramer PR, Ginsburg RN (2010) AGRRA protocols version 5.4. Atlantic and Gulf Rapid Reef Assessment Program, Florida, USA 1–31

Lange ID, Razak TB, Perry CT, Maulana PB, Prasetya ME, Lamont TA (2024) Coral restoration can drive rapid reef carbonate budget recovery. Current Biology 34:1341–1348 https://doi.org/10.1016/j.cub.2024.02.009

Legendre P, De Cáceres M (2013) Beta diversity as the variance of community data: dissimilarity coefficients and partitioning. Ecology Letters 16:951–963 https://doi.org/10.1111/ele.12141

Lima ALR, Eggertsen LM, Teixeira JLS, Schiavetti A, Félix-Hackradt FC, Hackradt CW (2023) The influence of marine protected areas on the patterns and processes in the life cycle of reef fishes. Rev Fish Biol Fisheries 33:893–913 https://doi.org/10.1007/s11160-023-09761-y

Logan A (1984) Interspecific aggression in hermatypic corals from Bermuda. Coral Reefs 3:131–138

Manzello DP, Cunning R, Karp RF, Baker AC, Bartels E, Bonhag R, Borreil A, Bourque A, Brown KT, Bruckner AW, Corbett B, D'Alessandro M, Dahlgren C, Dilworth J, Geiger E, Gilliam DS, Gomez M, Hanson G, Harrell C, Hesley D, Huebner LK, Kenkel CD, Koch HR, Kuehl J, Kuffner IB, Ladd MC, Lee S, Lesneski KC, Lewan A, Lirman D, Liu G, Matsuda SB, Montoya-Maya PH, Moore J, Muller EM, Nedimyer K, Parkinson JE, Ruzicka R, Spadaro J, Spady BL, Stein J, Unsworth JD, Walter C, Wen ADE, Williams DE, Williams SD, Williamson OM (2025) Heat-driven functional extinction of Caribbean *Acropora* corals from Florida's Coral Reef. Science 390:361–366 https://doi.org/10.1126/science.adx7825

Manzello DP, Enochs IC, Kolodziej G, Carlton R, Valentino L (2018) Resilience in carbonate production despite three coral bleaching events in 5 years on an inshore patch reef in the Florida Keys. Mar Biol 165:99 https://doi.org/10.1007/s00227-018-3354-7

585 Muller EM, Rogers CS, Van Woesik R (2014) Early signs of recovery of Acropora palmata in St.
586 John, US Virgin Islands. Mar Biol 161:359–365 https://doi.org/10.1007/s00227-013-2341-2

587 Muñiz-Castillo AI, Rivera-Sosa A, McField M, Chollett I, Eakin CM, Enríquez S, Giró A, Drysdale
588 I, Rueda M, Soto M (2024) Underlying drivers of coral reef vulnerability to bleaching in
589 the Mesoamerican Reef. Communications Biology 7:1452 https://doi.org/10.1038/s42003-
590 024-07128-y

591 Najeeb S, Khan RAA, Deng X, Wu C (2025) Drivers and consequences of degradation in tropical
592 reef island ecosystems: strategies for restoration and conservation. Frontiers in Marine
593 Science 12:1518701 https://doi.org/10.3389/fmars.2025.1518701

594 Neely KL, Nowicki RJ, Dobler MA, Chaparro AA, Miller SM, Toth KA (2024) Too hot to handle?
595 The impact of the 2023 marine heatwave on Florida Keys coral. Frontiers in Marine
596 Science 11:1489273 https://doi.org/10.3389/fmars.2024.1489273

597 Palmer CV, McGinty ES, Cummings DJ, Smith SM, Bartels E, Mydlarz LD (2011) Patterns of
598 coral ecological immunology: variation in the responses of Caribbean corals to elevated
599 temperature and a pathogen elicitor. Journal of Experimental Biology 214:4240–4249
600 https://doi.org/10.1242/jeb.061267

601 Perry CT, Alvarez-Filip L, Graham NA, Mumby PJ, Wilson SK, Kench PS, Manzello DP, Morgan
602 KM, Slangen AB, Thomson DP (2018) Loss of coral reef growth capacity to track future
603 increases in sea level. Nature 558:396–400 https://doi.org/10.1038/s41586-018-0194-z

604 Perry CT, de Bakker DM, Webb AE, Comeau S, Harvey BP, Cornwall CE, Alvarez-Filip L, Pérez-
605 Cervantes E, Morris J, Enochs IC (2025) Reduced Atlantic reef growth past 2° C warming
606 amplifies sea-level impacts. Nature 646:619–626 https://doi.org/10.1038/s41586-025-
607 09439-4

608 Perry CT, Morgan KM (2017) Bleaching drives collapse in reef carbonate budgets and reef growth
609 potential on southern Maldives reefs. Scientific reports 7:40581
610 https://doi.org/10.1038/srep40581

611 Perry CT, Murphy GN, Graham NA, Wilson SK, Januchowski-Hartley FA, East HK (2015) Remote
612 coral reefs can sustain high growth potential and may match future sea-level trends.
613 Scientific Reports 5:18289 https://doi.org/10.1038/srep18289

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

617 Precht WF, Gintert BE, Robbart ML, Fura R, Van Woesik R (2016) Unprecedented disease-related
618 coral mortality in Southeastern Florida. Scientific reports 6:31374
619 https://doi.org/10.1038/srep31374

620 Raymundo LJ, Andersen MD, Rouzé H (2025) Coral restoration in a stressful environment:
621 Disease, bleaching, and dysbiosis in Acropora aspera in Guam, Micronesia. iScience 28:
622 https://doi.org/10.1016/j.isci.2025.112244

623 Reed KC, Muller EM, van Woesik R (2010) Coral immunology and resistance to disease. Diseases
624 of aquatic organisms 90:85–92 https://doi.org/10.3354/dao02213

Reimer JD, Peixoto RS, Davies SW, Traylor-Knowles N, Short ML, Cabral-Tena RA, Burt JA, Pessoa I, Banaszak AT, Winters RS, Moore T, Schoepf V, Kaullysing D, Calderon-Aguilera LE, Wörheide G, Harding S, Munbodhe V, Mayfield A, Ainsworth T, Vardi T, Eakin CM, Pratchett MS, Voolstra CR (2024) The Fourth Global Coral Bleaching Event: Where do we go from here? Coral Reefs 43:1121–1125 https://doi.org/10.1007/s00338-024-02504-w

Richardson LE, Graham NAJ, Pratchett MS, Eurich JG, Hoey AS (2018) Mass coral bleaching causes biotic homogenization of reef fish assemblages. Global Change Biology 24:3117–3129 https://doi.org/10.1111/gcb.14119

Schaffelke B, Brigdale A, Dallison T, Koss J, McField M, Petit A, Planes S, Rocle N, Staub F, Towle E (2025) The status and trends of coral reefs around the world–connecting science to conservation action. https://doi.org/10.5194/oos2025-938

Smith KE, Burrows MT, Hobday AJ, King NG, Moore PJ, Sen Gupta A, Thomsen MS, Wernberg T, Smale DA (2023) Biological Impacts of Marine Heatwaves. Annu Rev Mar Sci 15:119–145 https://doi.org/10.1146/annurev-marine-032122-121437

Torruco D, González-Solís A (2024) Unusual formations in the B. R. Chinchorro Bank: Miniature atolls. Marine Ecology 45:e12817 https://doi.org/10.1111/maec.12817

Toth LT, Johnson SA, Lyons EO, Spadaro J, Stathakopoulos A, Bloomer S, Mallon J, Jenkins CM, Williams SD, Combs I, Craig Z, Muller E (2025) Coral restoration can drive rapid increases in reef accretion potential. Sci Rep 15:28353 https://doi.org/10.1038/s41598-025-04818-3

Toth LT, Kuffner IB, Stathakopoulos A, Shinn EA (2018) A 3,000-year lag between the geological and ecological shutdown of Florida's coral reefs. Global Change Biology 24:5471–5483 https://doi.org/10.1111/gcb.14389

Trégarot E, D'Olivo JP, Botelho AZ, Cabrito A, Cardoso GO, Casal G, Cornet CC, Cragg SM, Degia AK, Fredriksen S (2024) Effects of climate change on marine coastal ecosystems–A review to guide research and management. Biological Conservation 289:110394 https://doi.org/10.1016/j.biocon.2023.110394

Turner MG, Seidl R (2023) Novel Disturbance Regimes and Ecological Responses. Annu Rev Ecol Evol Syst 54:63–83 https://doi.org/10.1146/annurev-ecolsys-110421-101120

Urbina-Barreto I, Chiroleu F, Pinel R, Fréchon L, Mahamadaly V, Elise S, Kulbicki M, Quod J-P, Dutrieux E, Garnier R (2021) Quantifying the shelter capacity of coral reefs using photogrammetric 3D modeling: From colonies to reefscapes. Ecological Indicators 121:107151 https://doi.org/10.1016/j.ecolind.2020.107151

Wilson B, Edmunds PJ (2026) Survival, rarity, and extinction in tropical stony corals. Conservation Biology 40:e70200 https://doi.org/10.1111/cobi.70200

Yao L, Kershaw S, Shen S, Wang X (2023) A new reef classification model with insights into Phanerozoic evolution of reef ecosystems. Sedimentology 70:1886–1914 https://doi.org/10.1111/sed.13099

(2025) Citing RStudio. https://support.posit.co/hc/en-us/articles/206212048-Citing-RStudio

ERDDAP - NOAA Coral Reef Watch Operational Daily Near-Real-Time Global 5-km Satellite Coral Bleaching Monitoring Products - Data Access Form. https://coastwatch.pfeg.noaa.gov/erddap/griddap/NOAA_DHW.html

Acknowledgments

We thank Ana Molina Hernández, Raúl Tecalco Rentería, Sara Melo Merino, Nuria A. Estrada Saldivar, David Alberto Macías Díaz, Irvin L. Chávez Estrada, E. Omar Guzmán Urieta, Yair Ohrel Alonso Romero, Andrea Rodríguez Jiménez, and Dulce M. Tapia Aguirre for their support in the field and for processing part of the data presented in this study. We also thank Melina Soto and the Healthy Reefs Initiative for their logistical and financial support. This study was conducted with the permission and support of the Mexican Commission of Natural Protected Areas and the Reserva de la Biosfera Banco Chinchorro. J.B.-M. thanks the Secretaría de Ciencia, Humanidades, Tecnología e Innovación for the scholarship (No. 1037778) to pursue her PhD studies. This article forms part of the requirements for J.B.-M. to obtain the PhD degree in the Posgrado en Ciencias Biológicas, UNAM.

Funding

Open access funding provided by Universidad Nacional Autónoma de México, UNAM.

Author information

Authors and Affiliations

Biodiversity and Reef Conservation Laboratory, Instituto de Ciencias del Mar y Limnología, Unidad Académica de Sistemas Arrecifales, Universidad Nacional Autónoma de México, Puerto Morelos 77580, Quintana Roo, México. Posgrado de Ciencias Biológicas, Universidad Nacional Autónoma de México, Ciudad de México, 04530, México.

Jhoselinne T. Buenrostro-Muñoz

Biodiversity and Reef Conservation Laboratory, Instituto de Ciencias del Mar y Limnología, Unidad Académica de Sistemas Arrecifales, Universidad Nacional Autónoma de México, Puerto Morelos 77580, Quintana Roo, México. Laboratorio Nacional de Biología del Cambio Climático, SECIHTI, Ciudad de México, México.

Lorenzo Álvarez-Filip

Biodiversity and Reef Conservation Laboratory, Instituto de Ciencias del Mar y Limnología, Unidad Académica de Sistemas Arrecifales, Universidad Nacional Autónoma de México, Puerto Morelos 77580, Quintana Roo, México.

Esmeralda Pérez-Cervantes

Contributions

Conceptualization was performed by J.B.-M and L.A-F; field survey logistics was coordinated by L.A-F and E.P-C; field work was conducted by L.A-F and E.P-C; the analyzes of the data were made by J.B.-M and L.A-F; manuscript writing was performed by J.B.-M and L.A-F, revised and approved by all other authors.

Corresponding author

Correspondence to Lorenzo Álvarez-Filip.

Ethics declarations

Conflict of interest

The authors have no conflict of interest to declare that is relevant to the content of this article.

Supplementary Information

Below is the link to the electronic supplementary material.

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- TableS1.pdf
- TableS2.pdf
- Supplementarymaterial.pdf