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# Demographic shifts in stony coral populations in a remote reef system in the Southern Gulf of Mexico during the 2023-2024 marine heat wave

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**Research Article**

**Keywords:** Coral populations, demography, digital underwater photogrammetry, marine heatwaves, response to thermal stress

**Posted Date:** March 3rd, 2026

**DOI:** https://doi.org/10.21203/rs.3.rs-8904068/v1

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**Additional Declarations:** No competing interests reported.

# Abstract

We evaluated shifts in coverage and colony size structure distribution of scleractinian corals at a remote reef system in the Southern Gulf of Mexico during the 2023–2024 marine heatwave. Permanent parcels established at three reef sites (9–20 m depth) were surveyed in the summer of 2023 and 2024 using underwater digital photogrammetry. The obtained orthomosaics were analysed in Geographical Information Systems (GIS) to estimate species richness, area of live tissue cover ($m^2$), and colony partial mortality. Probability density curves based on log10-transformed colony area measurements were constructed, and skewness and kurtosis estimated. Colony size classes for each species were categorised I small, medium, and large based on observed size ranges quartiles. Non-parametric statistical tests were used to identify significant changes ($p < 0.05$) over time. The temperature data showed that the DHW values reached 19.3°C-weeks during the study period. The coral community comprised 13 taxa, dominated by reef building corals. Live coral tissue decreased by 4.7% over time. *Colpophyllia natans*, *Orbicella annularis*, *O. faveolata*, and *O. franksi* lost coverage and presented changes in population size structure, characterised by the loss of large colony sizes and an increase in small colonies. *Montastraea Cavernosa* and *Diploria labyrinthiformis*, showed a shift in the size distribution, with no coverage changes. *Pseudodiploria strigosa*, *Siderastrea siderea*, and *Porites astreoides* were tolerant to thermal stress. Partial mortality remained relatively stable, except for D. *labyrinthiformis*. Coral populations showed differential responses to unprecedented thermal stress, underscoring the need for conservation strategies, even in regions with low anthropogenic disturbance.


# 1. Introduction

Globally, 2023 was documented as the second warmest year since instrumental temperature records began (Lopez et al., 2025), with oceanic heat content reaching record highs. This has led to persistent warming in ocean basins, where the water column in the tropical Atlantic Ocean is heating at a faster rate than the global average (Cheng et al., 2024). Thermal stress has severe effects on coral reefs, as it increases the risk of mass bleaching events and widespread mortality, decreases coral growth and reproduction rates, and favours the proliferation of coral diseases, thereby threatening their persistence (Eakin et al., 2019; Hughes et al., 2018). In the summer of 2023, the now defined fourth global coral bleaching event or 2023–2024 marine heatwave, affected coral reefs across the globe, underscoring the need to assess the accumulated and excessive heat impacts, particularly on understudied coral communities (Reimer et al., 2024).

Recent evaluations of heatwave-induced stress on coral reefs in tropical regions have predominantly focused on the Caribbean and Pacific areas, with limited data available from the Gulf of Mexico. In places such as the Florida Keys, nearly 100% of corals bleached in several locations. However, related mortality rates were low, particularly for reefs located further from land (Neely et al., 2024). Bleaching has been observed to decrease with increasing depth and the consequent reduction in temperature in the water column (Goodbody-Gringley & Chequer, 2025). Species also show different responses to heat

disturbance, with stress tolerant species (Darling et al., 2012) showing higher bleaching rates (Goodbody-Gringley & Chequer, 2025) but rapid recovery (Lachs et al., 2024; Morais et al., 2024).

The importance of including demographic data in coral studies has long been recognized for evaluating the condition of coral communities, reef health, and response to environmental variation (Bak & Meesters, 1998; Dietzel et al., 2020; Edmunds & Riegl, 2020; Lachs et al., 2021; Meesters et al., 2001). Changes in coral size structure can be related to demographic processes such as mortality, growth, and recruitment (Bernard et al., 2023; Vallès et al., 2025). Colony size is also associated with differences in life-history traits, with larger colonies exhibiting lower total mortality and growth rates but higher partial mortality, fecundity (Bak & Meesters, 1998; Chen et al., 2025; Hughes & Connell, 1987), and bleaching under thermal stress (Winslow et al., 2024). Coral colony size structure is associated with local differences in environmental conditions and anthropogenic disturbance. Coral populations with larger sizes have been observed in areas with higher temperatures and salinity (Bauman et al., 2013) but lower anthropogenic disturbance (Bak & Meesters, 1998; Bernard et al., 2023; Hernández-Landa et al., 2020). Coral size-frequency distributions can be quantitatively assessed using statistical metrics such as skewness, mean, and mode, which reflect recruitment and mortality patterns (Bak & Meesters, 1998; Dietzel et al., 2020; Lachs et al., 2021). Identifying the allocation of coral cover allows for a better understanding of population dynamics under disturbance (Edmunds & Riegl, 2020).

Most studies addressing the size structure of coral populations have used belt transects, quadrats, and free diver rover techniques to obtain a limited number of linear measurements (e.g. maximum height or diameter) on observed colonies over spatially limited areas (Bak & Meesters, 1998; Bauman et al., 2013; Bernard et al., 2023; Pisapia et al., 2025; Vallès et al., 2025). However, the recent incorporation on coral reef studies of photogrammetry combined with Geographic Information Systems, hereafter referred to as underwater digital photogrammetry (UWP), permits measurements on large sample sizes and broad spatial scales (Barrera-Falcon et al., 2021; Ferrari et al., 2016; Hernández-Landa et al., 2020). Furthermore, given that processing and data analysis occur in the laboratory, area calculation on coral colonies is more accurate as the shape and boundaries are better represented by the polygon vertices in a spatially explicit context.

Cayo Arenas is a small remote reef system in the Gulf of Mexico located about 160 km from the continent, that was recently incorporated in a new Marine Protected Area. Given its location, it is subjected to limited anthropogenic disturbance compared to other reefs closer to the continent. This provides an opportunity to evaluate changes in coral populations in response to the extreme heat associated with the 2023 marine heatwave. Here, we 1) characterized the scleractinian coral community on reefs between 10–20 m depth in Cayo Arenas using UWP and 2) assessed changes in the size structure of colonies based on live tissue coverage of scleractinian corals in relation to the 2023–2024 marine heatwave.

# 2. Methods

# 2.1. Study area

The Cayo Arenas reef system (Fig. 1) is part of a series of emergent reefs and shallow coral banks distributed along the Campeche Bank platform in the Southwestern Gulf of Mexico (Tunnell et al., 2007). The seascape is dominated by sandy beds and areas dominated by a calcareous matrix with different coverages of scleractinian corals, octocorals, and macroalgae (Frías-Vega et al., 2025). Three main coral reef development areas that differ in spatial extent, degree of emergence, and exposure to wave and wind energy have been identified. The northeast unit is the largest and includes two emergent sectors, whereas the southeastern unit is smaller and presents a single emergent sector. Both units form an elongated crest along the eastern margin, separated from the western unit by a semi-enclosed channel. The western unit is relatively sheltered from prevailing wave and wind energy and has a vegetated island over sandy sediments and coral rubble (Chávez et al., 2007; Logan et al., 1969).

Located approximately 160 km from the nearest continental coast, Cayo Arenas is considered a remote reef with limited anthropogenic pressures. The area contains some of the best preserved reefs in the Mexican Atlantic, with the dominance of reef building species (e.g. *Orbicella* spp. and *C. natans*), and high herbivory levels driven by the elevated biomass of herbivorous taxa, particularly sea urchins, parrotfish, and surgeonfish (Cabrera-Rivera et al., 2025; Pérez-Cervantes et al., 2017). Cayo Arenas was incorporated in 2024 as one of the core zones of the newly established Southern Gulf of Mexico Reefs National Park (México, 2024).

# 2.2. Data collection and processing

Three reef sites, located between 9 and 20 m depth, were surveyed using an UWP approach (Barrera-Falcon et al., 2021; Hernández-Landa et al., 2020) in August 2023 (prior to the seasonal temperature peak, T1) and in August 2024 (following the 2023–2024 marine heat wave, T2). Cayo Arenas 1 is located on the leeward side of the northeastern reef unit, Bajo Tortugas, in the southern sector of the reef system, and Cayo Arenas 4 is closest to the western reef unit (Fig. 1). At each site, permanent sampling parcels were established by fixing two stainless steel stakes fitted with polycarbonate tags to the substrate, marking the start and end of a central transect. Geographic coordinates and transect orientations were recorded to ensure spatial consistency across sampling years. The parcel area (c.a. 40 x 8 m) was delimited with seven square polyvinyl quadrats (0.3 × 0.3 m) positioned at both ends and the central portion of each transect. These markers also served as references to scale the orthomosaics (see below). Depth was recorded at the central point of each marker. Photographs covering the parcel area were acquired by divers swimming ~ 2 m above the average reef depth at a slow and steady speed. Images were collected using 10-megapixel Canon G12 cameras operated in continuous shooting mode with a self-timer, achieving more than 80% image overlap which is necessary for accurate photogrammetric reconstruction.

Images from each sampling parcel were processed independently using Agisoft Metashape software v2.0.3 to generate orthomosaics. Images were aligned to produce a sparse point cloud, after which the

control markers were used to scale and calibrate the model. The parameters used for alignment were high accuracy, type estimated, key point limit set to 40,000, and tie point limit set to 10,000. An optimised dense point cloud was subsequently generated and filtered to remove erroneous points prior to model reconstruction. Final orthomosaics were produced for each parcel and exported as raster files for the analysis. The obtained orthomosaic scale and root mean square errors were < 1.4 cm and 0.75 pixels, respectively (Supplementary Information, Table 1). For a detailed description of orthomosaic construction, see Rioja-Nieto et al. (2025).

The orthomosaics were analysed using ArcMap v. 10.6, where colonies with a minimum size c.a. 5 cm were identified, coded, and digitised in shapefiles. Species were visually identified to the maximum taxonomic level possible based on (Humann & Deloach, 2002), Coralpedia (https://coralpedia.bio.warwick.ac.uk/) and AGRRA identification guides (https://www.agrra.org/training-and-survey-hub/). Here, a colony was defined as the sum of the area of live tissue (even if not connected) on any autonomous freestanding coral skeleton (Bak & Meesters, 1998). For species with modular or digitate growth forms (O. *annularis*, *P. porites, P. furcata*, and *P. divaricata*) a common skeletal base was considered as a single colony.

The water column temperature was monitored using HOBO data loggers deployed at three sites between 11–20, recording at 30 minutes intervals between July 2023 and September 2025 (Fig. 1). The obtained data was averaged to obtain the daily mean temperature. Data prior to the T1 sampling period was complemented with satellite-derived sea surface temperature (SST, NOAA Coral Reef Watch v3.1, daily 5 km product). Combined in situ and satellite data were used to calculate hotspots and Degree Heating Weeks (DHW °C-weeks), following the Coral Reef Watch method (Liu et al., 2013). The Maximum Monthly Mean (MMM) climatology was derived from the same coral reef watch v3.1 daily 5 km product. Satellite and in situ mean temperatures for an overlapping period of 736 days were compared using a paired t-test and Pearson's correlation analysis.

## 2.3. Coral colonies data analysis

At each sampling site, the shapefiles were used to estimate the area ($m^2$) of live tissue cover for each colony, species richness, and colony abundance. Colony partial mortality was estimated using the persistence index, defined as the ratio of live tissue area*total colony area$^{-2}$. Data for each site was standardised to an area of 250 $m^2$. This sample size is sufficient to characterize 90% of the community species richness (Ochoa Larios, 2025). To examine temporal differences in live tissue coverage and the persistence index over time, a non-parametric Kruskal-Wallis test with Dunn's pairwise comparison with Bonferroni correction was used (Zar, 1984).

The dominant species in terms of abundance were identified using a similarity percentage (SIMPER) analysis, considering a 91% contribution to the total abundance. The size ($m^2$) frequency distributions of the dominant species were visualised using probability density curves based on $\log_{10}$-transformed colony area measurements. Two-sample Kolmogorov-Smirnov tests were used to identify changes in the size distribution over time, and skewness and kurtosis were calculated. To evaluate changes in live

tissue allocation over time, colony size classes for each species were categorised as small (Q1), medium (Q2 and Q3), and large (Q4), corresponding to the size quartiles based on the pooled (T1 and T2) observed size range. Pearson's chi-square test of independence and standardised residuals analysis (McHugh, 2013) were used to identify significant differences (standard value ± 1.96) between size classes for the species over time.

Temperature data was analysed in R 4.5.2. Statistical analyses and data visualization were performed using Python 3.12.12, with the Matplotlib, Seaborn and SciPy libraries.

# 3. Results

Thermal conditions showed pronounced seasonal variability, with sustained periods of elevated seawater temperature (c.a. 31°C) during the summers of 2023 and 2024 (Fig. 2), and values around 24°C in winter. Thermal stress began to build up immediately after the first sampling period, reaching maximum DHW values of 19.3°C-weeks in November 2023, after the first sampling period. The collection of data in 2024 occurred just before the increase in DHW values and reached 17.6°C-weeks in November of that year (Fig. 2). The temperature curves obtained from the satellite and the combination of the HOBO-satellite datasets showed similar patterns with a high correlation (Pearson´s $r = 0.995$). However, the in situ daily temperature data was higher ($p < 0.05$).

The scleractinian coral community at Cayo Arenas, considering the three studied reefs located between 9–20 m, is composed of 13 taxa (Supplementary information, Table 2). The species *O. faveolata*, *M. cavernosa*, *P. astreoides, O. annularis, O. franksi, C. natans, S. siderea, P. strigosa, D. labyrinthiformis* and the digitate porites were the most abundant in both sampling periods, with > 95% of the total number of colonies observed (Fig. 3). In general, most species showed similar or lower numbers of colonies during the T2 sampling period. However, for *P. astreoides*, *O. franksi, C. natans*, and *D. labyrinthiformis*, a higher number was observed. The species *Agaricia agaricites*, *S. radians*, and *Madracis decactis* were uncommon, with $\leq$ 10 colonies observed onduringach period o

The total live coral tissue coverage was 24.1% for T1 and 19.4% for T2, with a significant decrease of 4.7% over time ($p < 0.05$). A decrease ($p < 0.05$) was observed for *C. natans*, *O. annularis, O. faveolata, O. franksi* and the digitate Porites (Fig. 4a). No other changes in coverage were observed ($p > 0.05$). However, *D. labyrinthiformis* showed a marginal p-value (0.062, Supplementary Information, Table 3). With the exception of *D. labyrinthiformis* in T2, the average persistence index of the coral community (Fig. 4b) was above 0.92, indicating low colony partial mortality. Between the T1 and T2 sampling periods, the persistence index increased for *Eusmilia fastigata*, *M. cavernosa*, and *O. annularis* and decreased for *D. labyrinthiformis* and *O. faveolata* ($p < 0.05$). Apart from *D. labyrinthiformis*, these changes were small. The number of observed colonies of *A. agaricites, S. radians*, and *M. decactis* was too low per sampling period and was not included in the analysis.

For the year 2023, eight species were dominant (*O. faveolata, M. cavernosa, O. franksi, O. annularis, P. strigosa, C. natans, S. siderea and P. astreoides*) accounting for c.a. 90% of live tissue coverage

contribution to the scleractinian coral community. In 2024, this diminished to six species, where *O. faveolata, M. cavernosa, O. franksi, O. annularis, C. natans* were maintained, and *D. labyrinthiformis* was added.

The colony size distribution was analysed for the dominant species. The populations of *O. franksi*, *C. natans, O. annularis, D. labyrinthiformis, O. faveolata and M. cavernosa* (Fig. 5a) changed between sampling periods ($p < 0.05$). All the Orbicella spp, *C. natans, and M. cavernosa* increased in skewness and kurtosis (Fig. 5b), which indicates a shift on the distribution towards smaller sized colonies and values more grouped towards mean sizes. In *D. labyrinthiformis*, the skewness and kurtosis values decreased, indicating a shift towards median and large values.

When analysing changes in the size of the colonies divided in quartiles, an increase in the frequency of small colonies was observed for *O. franksi, C. natans* and *O. annularis*. The last two also showed a decrease in the large colony category. In *M. cavernosa*, the small and large categories decreased, whereas medium-sized colonies increased. *D. labyrinthiformis* only showed a decrease in the medium sized colonies. *O. faveolata* had an increase in the frequency of medium sized colonies and a decreased in the large ones (Fig. 6).

# 4. Discussion

The scleractinian coral community of Cayo Arenas is characterized by a high coverage of reef-building species, similar to that observed on other reefs in the region (Correa et al., 2025; Webb et al., 2025). Compared to reefs in the Gulf of Mexico that are closer to the coast (Correa et al., 2025), Cayo Arenas shows lower species richness and coverage, but similar species dominance, with *Orbicella* spp., *M. cavernosa*, *C. natans* and *P. astreoides* dominating the reefscape. During the 2023–2024 marine heatwave, overall coral cover experienced much lesser declines than what is reported to occur in the Caribbean during the same marine heatwave event (Bon et al., 2025; Doherty et al., 2025; Thompson et al., 2025). Yet scleractinian corals showed a differential response to the unprecedented marine heatwave, with stress-tolerant species that form large and massive colonies experiencing the greatest changes in terms of population size structure and coverage, while other massive species and weedy corals underwent relatively minor changes. Reef building species are important for maintaining the reef structural framework (Courtney et al., 2016; González-Barrios & Álvarez-Filip, 2018). Therefore, their loss and/or changes in size distribution can affect key ecological processes that maintain biodiversity, such as the provision of refuge and habitat, predator-prey interactions, and recruitment (Darling et al., 2017; Graham & Nash, 2013; Rogers et al., 2018).

The most affected species in terms of tissue mortality and changes in population size structure were *C. natans, O. annularis, O. faveolata*, and *O. franksi*. The first three showed an increase in the frequency of smaller or medium sized colonies coupled with a decrease in large colonies, whereas *O. franksi* only showed an increase in the smaller sized category. This is likely to affect recruitment. A lower number of large colonies compromises the maintenance and recovery rates of their populations, as smaller

colonies tend to show lower fecundity (Bak & Meesters, 1998; Hughes & Connell, 1987). Furthermore, as these are dominant reef building species, the capacity to maintain reef structural complexity which depends on coral coverage (Barrera-Falcón et al., 2025), will also be compromised. The population of *O. franksi* might show a faster recovery rate than that of the other species, as large colonies were not affected. The decline in coral colony size observed here is consistent with observations after thermal disturbances in other regions, such as the Indian Ocean, South Pacific, and Caribbean (Brandt, 2009; Pisapia et al., 2025; Speare et al., 2022). The increase in small and/or medium sized colonies suggests a higher survivability of smaller sizes under thermal stress. Smaller colonies have been observed to show lower bleaching induced mortality than larger colonies (Depczynski et al., 2013; Speare et al., 2022; Winslow et al., 2024).

A shift towards a more homogenous size distribution and loss of medium sized colonies was observed for *M. Cavernosa* and *D. labyrinthiformis*, respectively, with no changes in coverage. The latter also showed a significant decrease in the persistence index (increase in partial mortality) over time. These species seem vulnerable to thermal stress, and our observations suggest a differential response according to colony size, where the species are able to maintain their coverage by the balance of growth and death processes. Webb et al. (2025) observed stable coverage of these species across several regions in response to increased DHW.

The species *P. strigosa*, *S. siderea*, and *P. astreoides* were tolerant to thermal stress, as no changes in coverage, size distribution, or persistence index were observed. *P. strigosa* has been previously described as highly susceptible to thermal stress (Brandt, 2009). However, *P. astreoides* and *S. siderea* are considered susceptible but have a high capacity for recovery after bleaching (Brandt, 2009; Webb et al., 2025). The latter has shown increased coverage over the past few years in the tropical Western Atlantic (Webb et al., 2025). Our results support the hypothesis that the differential response at the species level is dependent on local environmental conditions, including the presence of other chronic stressors (Bauman et al., 2013; Pisapia et al., 2025; Webb et al., 2025), and the availability and/or capability of the species to host thermally tolerant symbiodinium clades (Brandt, 2009; Claar et al., 2020).

Most of the affected species described here are considered stress tolerant (Darling et al., 2012) and are located in an isolated reef system with limited anthropogenic disturbance. Although coral coverage loss was not as catastrophic as in other regions, and partial mortality was low and relatively stable, the observed changes underscore that current environmental change is surpassing coral physiological capacity to survive, even for communities expected to be resilient given their low exposure to anthropogenic stress and high seasonal thermal variation (Baskett et al., 2010; Claar et al., 2020). The need to effectively enforce regulations that limit anthropogenic disturbance in areas such as Cayo Arenas is crucial for the maintenance of coral reefs in the region, as marine heatwaves are expected to increase in frequency and intensity (Werner et al., 2026).

The DHW were estimated using in situ and satellite derived data. This provided a more realistic measurement of the temperature environment, considering local conditions (e.g. currents and upwelling) that change the temperature dynamics and cannot be measured with satellites (Reid et al., 2019). Nonetheless, in situ and satellite data were highly correlated, underscoring the applicability of satellite SST measurements for estimating heat stress in the region. Prior to the T1 data collection, no DHW were observed. However, for T2, the coral community at the study site had been affected by values above 19°C-week. A DHW value of 16°C-week or more is commonly considered to cause severe multi-species mortality (van Woesik et al., 2022). Therefore, our observations can be mainly related to accumulated heat stress, as no other major disturbances (e.g. hurricanes or disease) were observed during the study period.

Our observations are based on coral communities located between 10–20 m, and do not consider species from shallower areas, such as Acroporids, which are known to be highly susceptible to thermal stress and have undergone mass mortality on shallow reefs across the Caribbean (Birkart & Álvarez-Filip, 2025; Manzello et al., 2025). In our 2024 surveys, these species showed high mortality in shallow reef areas. The use of UWP allowed us to obtain information on the size change of colonies over time in a spatially explicit context and assess the coral populations response to thermal stress, complementing observations based on coral coverage. Understanding the allocation of live tissue provides the necessary information to better understand coral structural and functional variation. Given the spatial resolution of UWP analysis (a few centimetres), the use of permanent parcels is necessary for studies considering assessments over time. Small misalignments in parcel data collection were related to an increase in the number of observed colonies in T2. Nevertheless, given the large sample size per period (> 1800 colonies), the results were not affected.

# Declarations

Author Contributions

R.R.N. Conceptualization, supervision, investigation, formal analysis, methodology, validation, visualization, writing original draft, writing review and editing, funding acquisition. A.L.B. formal analysis, methodology, writing original draft, writing review and editing. E.B.F. Investigation, methodology, visualization, validation, formal analysis. V.C.O. writing original draft, formal analysis. D.R.C. Investigation, formal analysis, writing review and editing. C.C.V. Investigation and data curation. L.A.F. Funding acquisition, writing review and editing. R.H.L. Investigation, writing review and editing.

Statements and Declarations

The authors declare have no competing interests to declare that are relevant to the content of this article.

Funding

This research was funded by the Programa de Apoyo a Proyectos de Investigación e Inovación Tecnológica (PAPIIT) grants number IG201323 and IG100426.

Acknowledgements

We thank Nazly Ochoa Larios for her assistance during shapefile construction process.

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# Figures

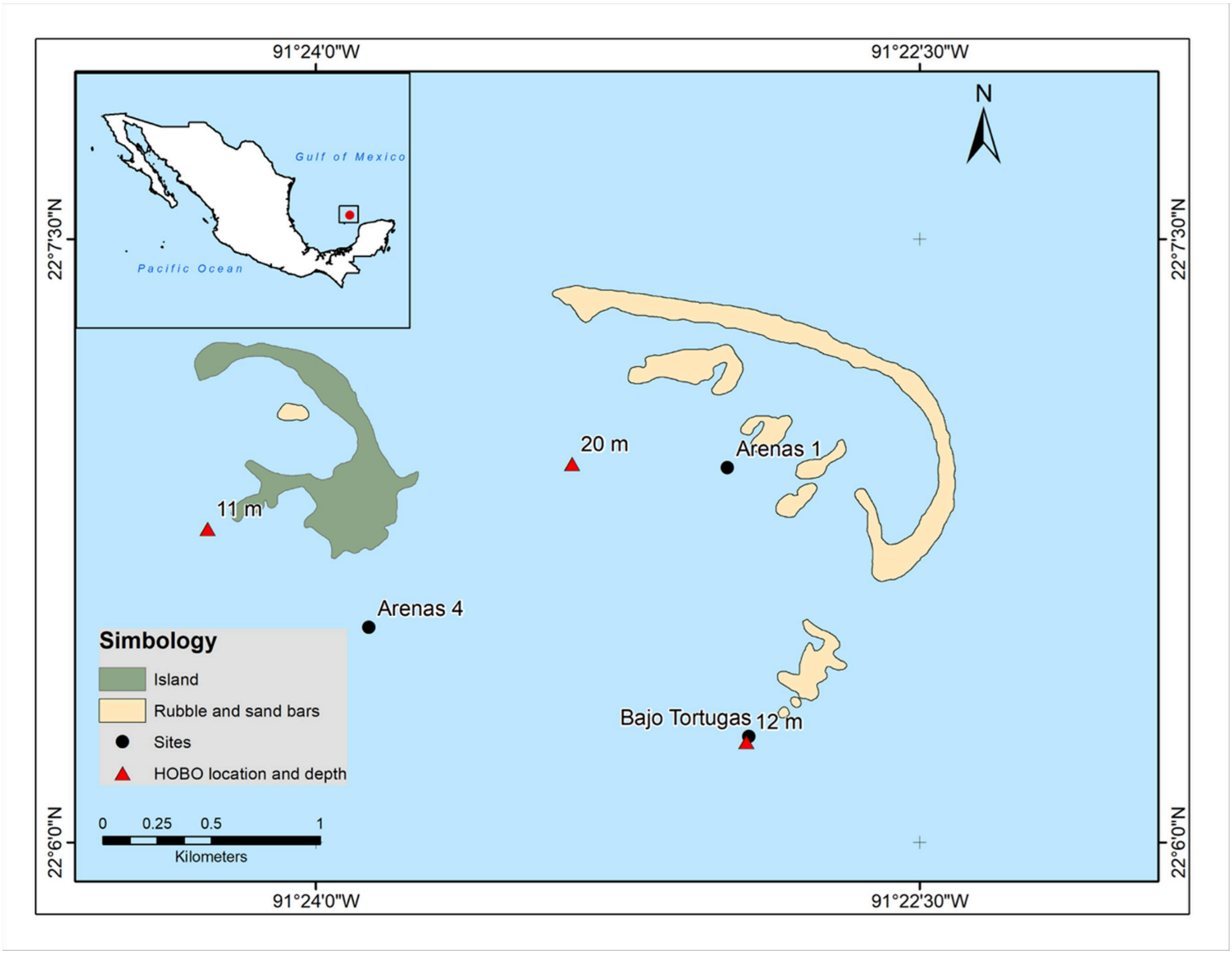
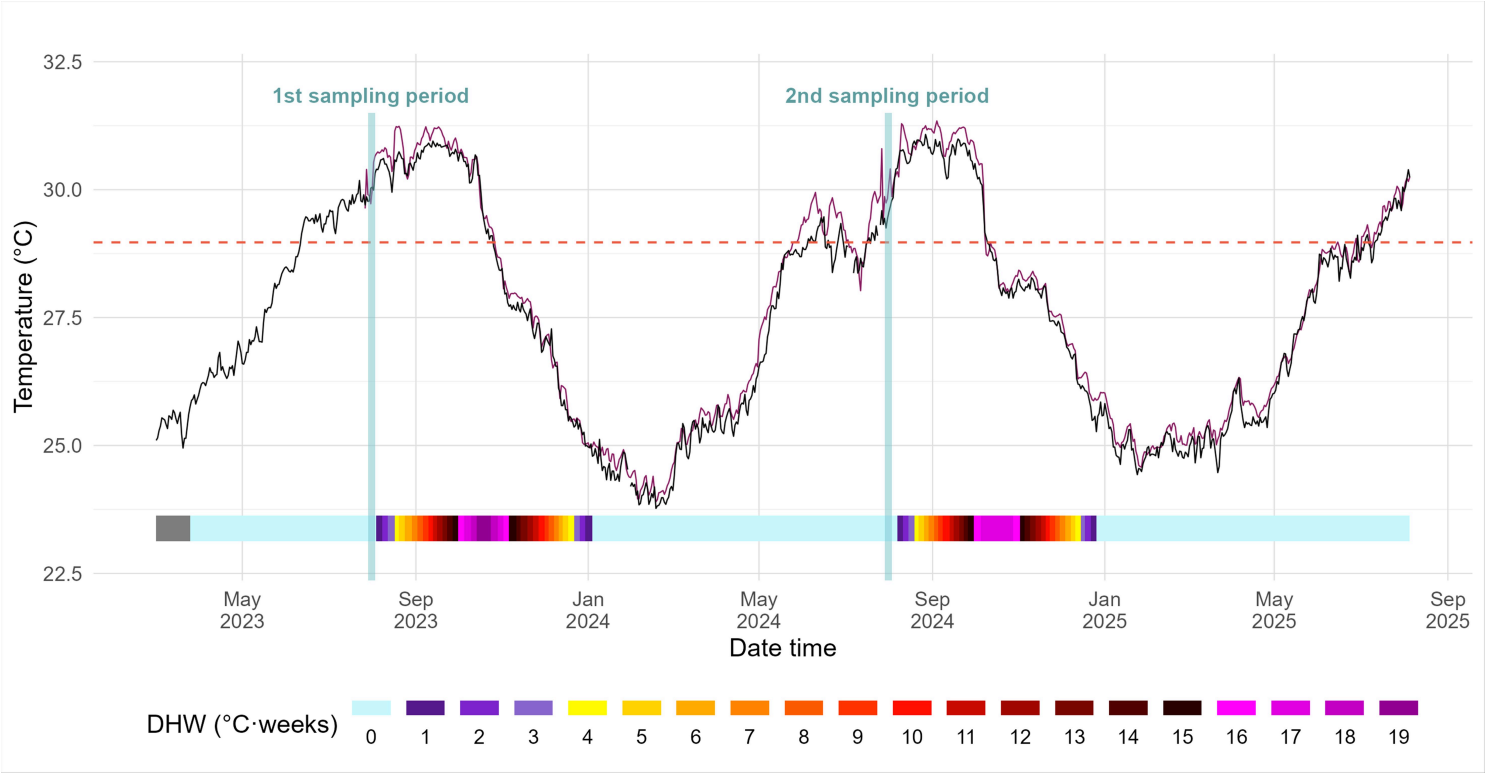


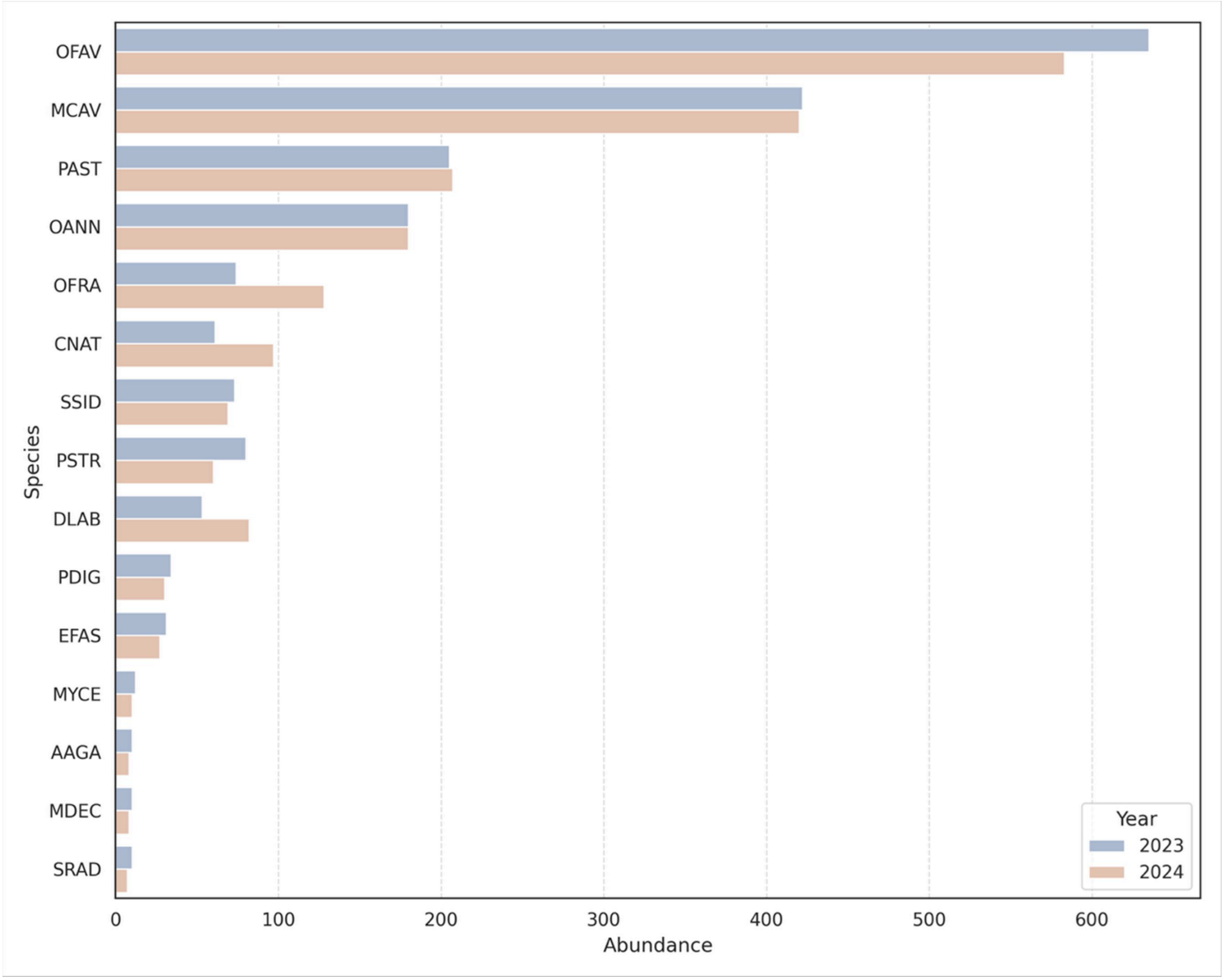

**Figure 1**

Location of sampling sites and temperature data loggers (HOBO) on the Cayo Arenas reef in the Gulf of Mexico. The maximum depths for the sites are as follows: Arenas 1 (10 m), Arenas 4 (20 m), and Bajo Tortugas (14 m).



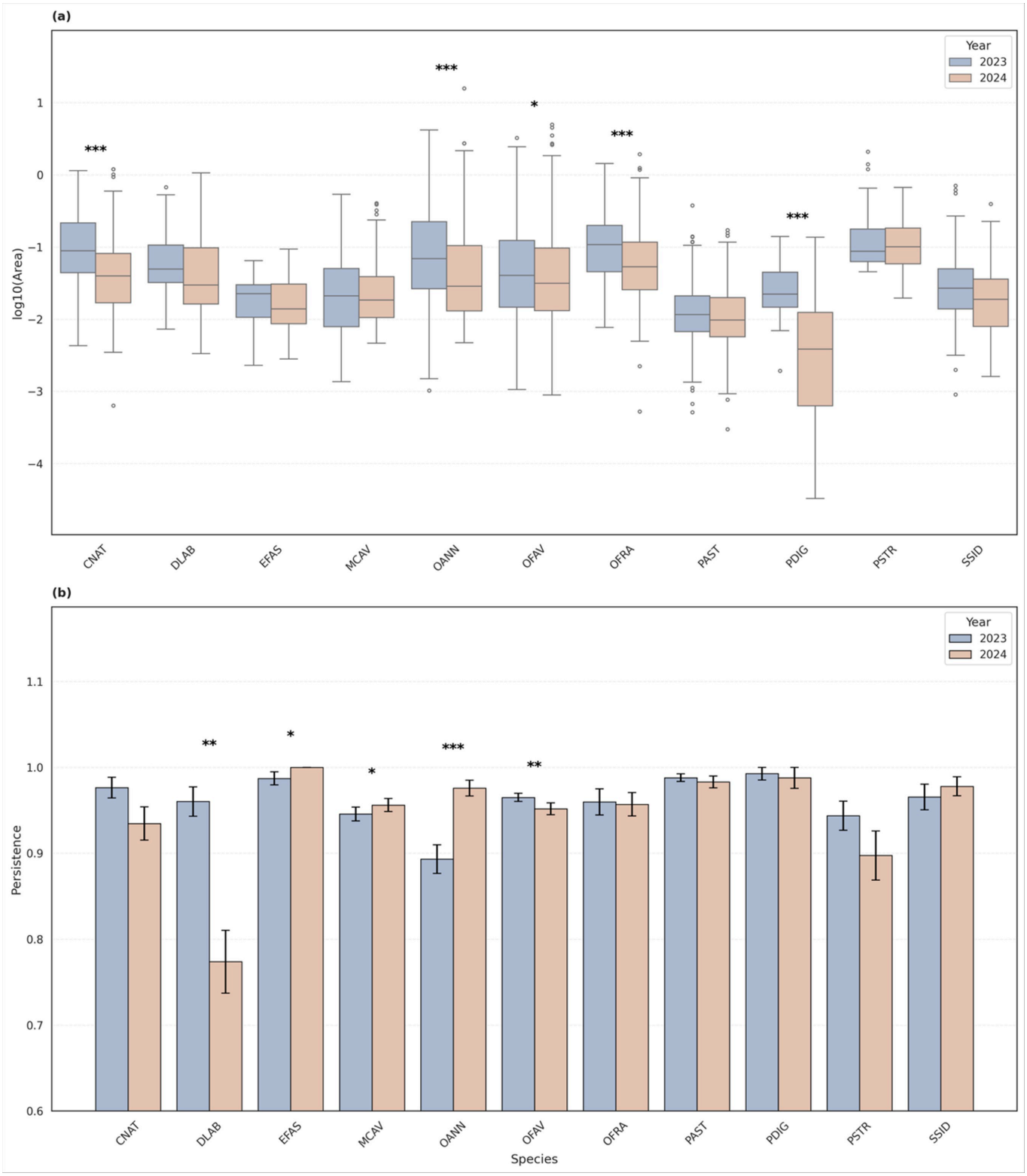

## Figure 2

Time series of seawater temperature (°C) from May 2023 to September 2025. The purple line represents the daily mean in situ temperature obtained from the HOBO data loggers. The black line shows the satellite-derived sea surface temperature (SST). The dashed red line indicates the Maximum Monthly Mean (28.9 °C) estimated from the SST NOAA Coral Reef Watch v3.1 daily 5 km product. Vertical shaded blue bars mark field sampling periods. The band at the bottom represents the Degree Heating Weeks (DHW, °C-weeks), indicating the period of thermal stress in 2023 and 2024.




## Figure 3

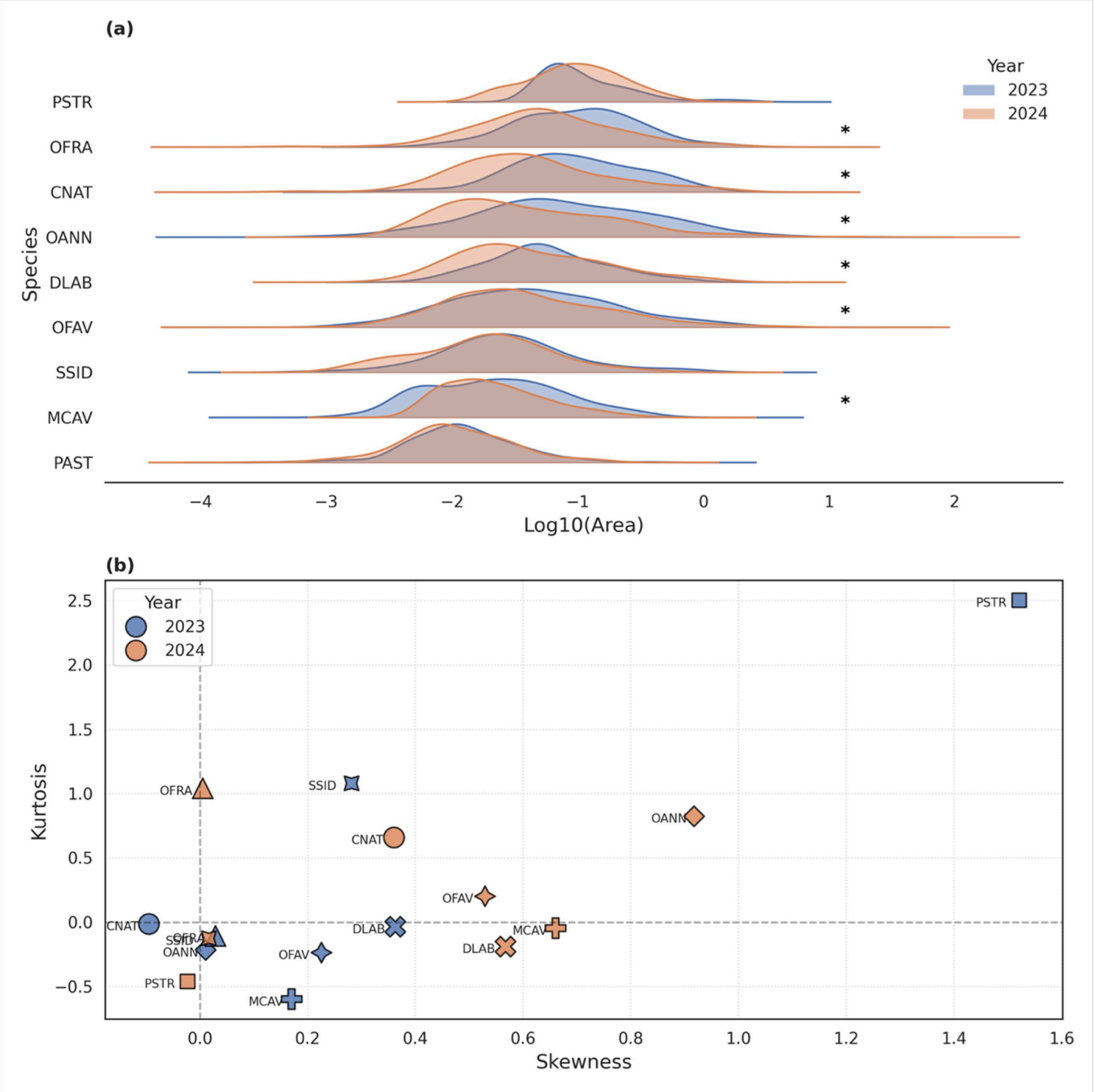
Total number of coral colonies per sampling period observed in permanent parcels at three reef sites in Cayo Arenas (750 $m^2$ total area, per year). Species codes were defined according to the AGGRA protocol: OFAV (*Orbicella faveolata*), MCAV (*Montastraea cavernosa*), PAST (*Porites astreoides*), OANN (*Orbicella annularis*), OFRA (*Orbicella franksi*), CNAT (*Colpophyllia natans*), SSID (*Siderastrea siderea*), PSTR (*Pseudodiploria strigosa*), DLAB (*Diploria labyrinthiformis*), PDIG (Digitate *Porites* spp.), EFAS (*Eusmilia fastigiata*), MYCE (*Mycetophyllia* spp.), AAGA (*Agaricia agaricites*), SRAD (*Siderastrea radians*), MDEC (*Madracis decactis*).




## Figure 4

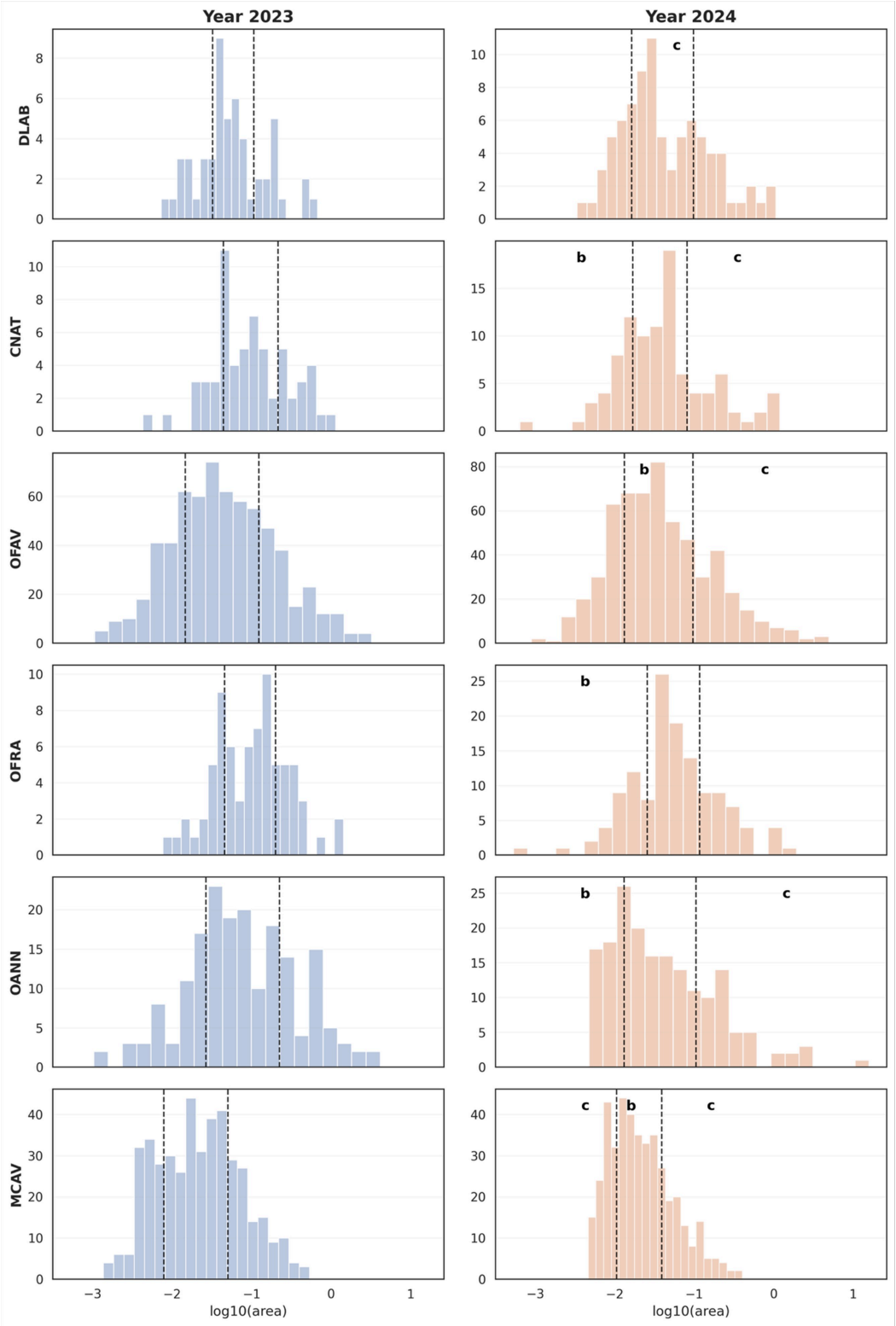
(a) $Log_{10}$ live tissue area for scleractinian corals in 2023 and 2024. Black lines represent the median, boxes represent the interquartile range, and whiskers are 1.5 times below the first and above the third quartiles, respectively. (b) Average persistence index (live tissue area * colony total area$^{-2}$); bars indicate the standard error. *** p<0.001, ** p<0.01, *p<0.05. Species codes are the same as those used in Figure

3. The number of observed colonies of *A. agaricites, S. radians*, and *M. decactis* was <10 in one sampling period and were not included in the analysis.




## Figure 5

(a) Probability density curves of Log10 size distribution ($m^2$) of coral populations colonies of dominant species in the Cayo Arenas reef system. * indicates a significant change ($p<0.05$) between T1 and T2. (b) Scatter plot of the kurtosis and skewness distribution values for the dominant species.




## Figure 6

Transformed (log10) colony size frequency of dominant species that showed significant changes ($p<0.05$) in size distribution between 2023-2024. The vertical dotted lines indicate the boundaries between small (Q1), medium (Q2 and Q3), and large (Q4) quartiles, considering the pooled (T1 and T2) size distribution for each species. The letters above the size distributions indicate a (no change, $p>0.05$), b (increase, $p<0.05$), and c (decrease, $p<0.05$) in the number of colonies belonging to each size class.

## Supplementary Files

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

- SupplementaryinformationTable1.docx
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