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RESEARCH

Changes in feeding behavior of Pacific crown-of-thorns starfish (*Acanthaster* cf. *solaris*) following mass coral bleaching in the northern Great Barrier Reef, Australia

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Abstract The Great Barrier Reef, like most reef ecosystems, is increasingly subject to major acute disturbances, including population irruptions of crown-of-thorns starfish (CoTS) and climate-induced coral bleaching. Given their increasing incidence, acute disturbances are likely to occur simultaneously or successively, though interactive effects of major disturbances are generally unknown. This study explores changes in the feeding behavior of CoTS during an emerging population irruption at Lizard Island that coincided with the 2024 mass bleaching, using in situ survey data. We conducted Scooter-Assisted Large Area Diver-Based surveys to investigate changes in CoTS demography and feeding, and point-intercept transects to examine changes in coral cover. From 2023 to 2025, there was a 49% decline in coral cover at Lizard Island, which was largely attributable to mass bleaching. Daily feeding rates of CoTS significantly declined over the same period, both in terms of the number of coral colonies (42.8% decline) and the combined tissue surface area of all corals consumed for each starfish (46.3% decline). CoTS density increased by 96.1% from 2023 to 2025 despite decreased feeding rates. Additionally, the relative consumption of different coral genera was consistent throughout the study period, with *Acropora* spp. contributing to > 80% of CoTS diet throughout the study. Though the 2024 bleaching event may have suppressed feeding rates and ecological impact of individual CoTS, the longer-term effects of CoTS are likely to conflate with coral loss due to mass bleaching, especially given sustained increases in CoTS densities.

Keywords Population irruption · Crown-of-thorns starfish outbreak · Marine heatwave · Coral reef disturbance · Corallivore feeding rate · Lizard Island

Introduction

Indo-Pacific coral reefs are increasingly subject to major acute disturbances, including population irruptions of crown-of-thorns starfish (CoTS; Pratchett et al. 2014) and climate-induced mass coral bleaching (Hughes et al. 2018). Up until 2010, sustained coral loss on the Great Barrier Reef (GBR) was largely attributed to major tropical storms (cyclones) and recurrent population irruptions of CoTS (Osborne et al. 2011; De'ath et al. 2012; Bellwood et al. 2019), though climate-induced coral bleaching has since emerged as the foremost cause of coral loss (Pratchett et al. 2021; Emslie et al. 2024). Critically, there have been five episodes of mass coral bleaching and mortality since 2016, reflecting sustained increases in ocean temperatures (Emslie et al. 2024). The most recent of these, which occurred in the austral summer of 2024, was the most widespread bleaching event to affect the GBR on record and caused major coral bleaching and mortality across the entire GBR (AIMS LTMP 2025). The latest mass bleaching event (in 2024) also coincided with documented increases in the abundance of Pacific CoTS (*Acanthaster* cf. *solaris*) in the northern GBR (Chandler et al. 2023), signaling the possible start of

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renewed reef-wide population irruptions (Babcock et al. 2020; Uthicke et al. 2024).

Population irruptions of *A.* cf. *solaris* have occurred throughout the western Pacific (Baird et al. 2013; Plass-Johnson et al. 2015; Kayal et al. 2012) and are one of the foremost contributors to long-term coral loss (Bruno and Selig 2007; De'ath et al. 2012; Kayal et al. 2012). On the GBR, recurrent population irruptions have occurred every 14–17 years since at least the 1950s (Pratchett et al. 2014). Past irruptions, which began in approximately 1962, 1979, 1993, and 2009 (Pratchett et al. 2017), all started in the putative initiation box in the northern GBR (14.6°S–17°S) and then moved south along the mid-shelf (Pratchett et al. 2014). CoTS occur naturally on the GBR, but the drivers of population irruptions remain equivocal. There are two main factors considered important in exacerbating, if not initiating population irruptions: terrestrial runoff caused by flood events can increase the availability of the nutrients that supplement limited food availability for CoTS larvae (Fabricius et al. 2010) and overfishing of predators may have reduced predatory limitation on CoTS (Dulvy et al. 2004). Irruption densities of CoTS (> 15 starfish.ha^{-1}) have affected up to 17% of reefs along the length of the GBR during each distinct population irruption (Hoey and Chin 2004) and accounted for a substantial proportion (e.g., 36.7%; Osborne et al. 2011) of sustained coral loss recorded by routine monitoring up until 2017 (see also De'ath et al. 2012; Mellin et al. 2019). Population irruptions of CoTS remain a persistent and continuing threat to coral reefs in this region, and based on the periodicity of past events, renewed population irruptions were expected to begin in 2025–2027 (Babcock et al. 2020). Accordingly, novel and highly resolved sampling methods have revealed increases in CoTS densities in the northern GBR since 2021 (Chandler et al. 2023; Uthicke et al. 2024) that have been overlooked by other sampling methods.

Population irruptions of CoTS also contribute to shifts in coral composition (Pratchett 2010), owing to their selective feeding on specific coral types, especially *Acropora* spp. (Keesing 2021; Foo et al. 2024). When their preferred prey corals are locally depleted, or when population irruptions are particularly severe, CoTS will consume less preferred taxa, such as massive *Porites* (Foo et al. 2024). Increasing predominance of *Acropora* corals (e.g., Emslie et al. 2024), attributable to their enhanced recovery capacity in the aftermath of severe disturbances (Linares et al. 2011; Pratchett et al. 2020), will also make coral assemblages even more vulnerable to subsequent population irruptions of CoTS (Millican et al. 2024). However, decreases in cover of preferred corals could hinder or even prevent population irruptions from occurring in the future (Millican et al. 2024).

Increasing coincidence of population irruptions of CoTS with severe marine heatwaves and coral bleaching make it challenging to disentangle the ecological effects of these disturbances (Keesing et al. 2019). Notably, interactions among these major disturbance events have yet to be fully considered. CoTS are sensitive to elevated temperatures (Hue et al. 2020; Lang et al. 2021, 2022) but it is unclear whether changing environmental conditions will moderate or exacerbate the frequency and/or intensity of population irruptions (Uthicke et al. 2015) or otherwise cause changes in the distribution and extent of impacts (e.g., Su et al. 2025). In the short-term, it is likely that mass coral bleaching and mortality will affect the availability and accessibility of different coral prey, which could have ramifications for CoTS behavior (Ling et al. 2020), as well as individual condition and fitness (Caballes et al. 2016). This study explores inter-annual changes in CoTS density and feeding behavior from 2023 to 2025 at Lizard Island, where increasing densities of CoTS (Chandler et al. 2023) coincided with a severe marine heatwave that caused extensive, but localized, mass coral bleaching and mortality in the northern GBR.

Methods

Study site

This study was conducted at Lizard Island (14°40'S, 145°27'E), in the northern Great Barrier Reef, Australia. This reef has been subjected to recurrent population irruptions of CoTS and is also located within the putative initiation box (Pratchett et al. 2014) where reef-wide population irruptions are suggested to start. More specifically, population irruptions in 1992 and 2010 were first recorded in the vicinity of Lizard Island (Pratchett 2005; Vanhatalo et al. 2017) before spreading predominantly southward. Coral assemblages at Lizard Island have also been subjected to significant and increasing heat stress, causing mass coral bleaching and mortality (e.g., Tebbett et al. 2022; Garing et al. 2025; Raoult et al. 2025). In 2024, sea surface temperatures exceeded the local maximum monthly mean throughout January–March, resulting in maximum accumulated heat stress of ~ 6 °C-Heating Weeks, which caused extensive bleaching and mortality of hard corals at some sites (Garing et al. 2025; Raoult et al. 2025). Other events that can cause widespread coral mortality, such as sedimentation due to runoff, outbreaks of coral disease, and tropical cyclones, have not been reported for Lizard Island during the timeframe of this study.

Field sampling

Pacific CoTS (*Acanthaster* cf. *solaris*) were surveyed at Lizard Island using the Scooter-Assisted Large Area Diver-Based (SALAD) method following Chandler et al. (2023). Two divers using Yamaha (500Li) Seascooters (with a surface float attached housing a GPS unit [Garmin eTrex 10]) surveyed parallel 5 m wide transects between the reef crest (2–4 m) and upper reef slope (5–9 m). Transects did not have a set length and were run parallel to the reef crest. The mean length of SALAD surveys in the present study was 737 m and the maximum was > 2 km. Due to the length of these surveys, specific sampling sites were not considered. Instead, successive, non-overlapping surveys were conducted to cover as much of the reef edge as possible. When feeding scars were observed, the divers stopped and conducted concentrated searches to detect relevant CoTS. Where CoTS were detected, the diameter of the starfish (if found) and the sizes of the feeding scars were recorded (single maximum linear measurement, following Chandler et al. 2024). CoTS that were found during surveys were removed, and no repeat sampling of CoTS was performed during this study. Surveys were conducted primarily during the austral summer, but some 2023 surveys were conducted during autumn and winter. Densities of CoTS were calculated based on the number of CoTS located per SALAD survey, standardized for the total search area (quantified from survey track data recorded using the GPS), following Chandler et al. (2023). CoTS generally exhibit cryptic behavior during the day (De'ath and Moran 1998), particularly during marine heatwaves and bleaching episodes, making them difficult to detect. Therefore, inferred CoTS densities were calculated using CoTS observed during SALAD surveys in addition to distinct sets of CoTS feeding scars which were observed but could not be attributed to any CoTS which had been accounted for, following Chandler et al. (2023). Estimating inferred CoTS densities from feeding scars likely still provides an underestimate of the true density, as CoTS can defer feeding for 3–4 days and remain hidden in the reef matrix (Chandler et al. 2025), and there is a possibility of multiple CoTS occurring in close proximity where it is difficult to discern their independent feeding scars. At low-to-moderate CoTS densities observed in the present study, there is reasonable capacity to attribute specific sets of feeding scars to single individuals (Chandler et al. 2023).

To test for inter-annual changes in diet composition of CoTS, all corals with conspicuous recent feeding scars were identified to genus. For genera with considerable morphological variability (e.g., *Acropora* and *Porites*) corals were further distinguished based on gross morphology, following Chandler et al. (2024). CoTS feeding rates were then estimated based on both i) the number of distinct colonies on which feeding scars were apparent, and ii) the combined tissue surface area (SA) of individual feeding scars on each different coral, which were standardized to provide daily feeding rates, following Chandler et al. (2025). The tissue SA of individual feeding scars was calculated from linear measurements of the maximum diameter of each distinct feeding scar (Chandler et al. 2024), that account for inherent differences in the shape and complexity of different corals. Formulas for specific morphotaxa were used when available, and generalized formulas based on morphology were used for coral taxa which did not have a specified formula (Table S1). Photogrammetry can provide greater precision and resolution in estimating the tissue surface area of distinct feeding scars, but prior research showed that single linear estimates combined with conversion coefficients calculated for different coral morphotaxa provide a very good approximation of tissue SA (Chandler et al. 2024). Thus, we measured linear dimensions here to enable greater sampling effort than would have otherwise been possible (i.e., if we had used photogrammetry to precisely estimate tissue surface area for every feeding scar).

Interannual variation in the cover and composition of hard corals (order Scleractinia) were assessed using 50 m point-intercept transects (PITs), following Linares et al. (2011). Two replicate PITs were haphazardly conducted on both the reef crest (2–4 m) and slope (5–9 m) at relevant reefs where SALAD surveys were conducted. To quantify coral cover and composition, the occurrence and identity of corals (using same taxa and morphological categories described above) were recorded beneath each of 100 uniformly spaced points (i.e., 50 cm apart) along each PIT. It is important to note that these coral surveys were not designed to discern the independent contributions of bleaching-induced coral mortality versus CoTS feeding. Rather, the study was focused on assessing changes in CoTS feeding following mass coral bleaching (in 2024) and coral depletion (in 2025). Therefore, the data collected on coral cover and composition represents important contextual information to enrich inferences made in respect to our primary variable of interest (i.e., CoTS feeding). CoTS that were found during PIT surveys were also measured and relevant feeding scar data were recorded following the methodology described above. These data were included in CoTS size and feeding rate analyses only.

Statistical analysis

Generalized linear mixed effects models (GLMMs) were used to examine changes among years (fixed categorical factor with three levels [2023, 2024, and 2025]) in inferred CoTS density (explicitly accounting for feeding scars), CoTS size, CoTS feeding rates (tissue area consumed and number of feeding scars per day), percent cover of hard corals, and percent cover of *Acropora* spp. Inferred density was modeled using a tweedie distribution with a log link function. CoTS size, the combined tissue surface area of corals consumed by each CoTS per day, and the number of feeding scars per day of each CoTS were modeled using gamma distributions with log link functions. Both total hard coral cover and *Acropora* spp. cover were modeled using beta distributions with logit link functions. In all models, survey site identity was also included as a random effect to account for any lack of spatial independence among replicate surveys (note any temporal patterns were accounted for by the fixed effects term of year). Model fit and assumptions were evaluated using plots and tests derived from simulated residuals (*DHARMa* package; Hartig 2024), which were satisfactory in all cases. Post-hoc pairwise tests, based on a Tukey's correction, were conducted to examine within factor level differences. All GLMMs were conducted in R (version 4.4.1; R Core Team 2024) using the *glmmTMB* package (Brooks et al. 2017; McGillyCuddy et al. 2025). Post hoc analyses were performed using the *emmeans* package (Lenth 2025) and post hoc results can be found in Table S2.

The relative abundance of hard coral taxa (i.e., relative availability of CoTS prey) was compared between study years with a permutational multivariate analysis of variance (PERMANOVA) based on a Bray–Curtis dissimilarity matrix. Another PERMANOVA based on a Bray–Curtis dissimilarity matrix was used to compare the relative surface area of each coral taxa consumed by CoTS (i.e., CoTS diet composition) between study years. Additionally, permutational analysis of multivariate dispersions (PERMDISPs) were used to test for differences in the variability of coral abundance and coral consumption among years. PERMANOVAs were performed using the adonis2 function and PERMDISPs were performed using the betadisper function, both from the *vegan* package (Oksanen et al. 2025) in R.

Results and discussion

Inferred densities of CoTS recorded at Lizard Island increased from 13.55 CoTS.ha^{-1} (± 1.66 SE) in 2023, to 16.51 CoTS.ha^{-1} (± 2.24 SE) in 2024, and 26.57 CoTS.ha^{-1} (± 4.58 SE) in 2025 (Fig. 1a). These significant and sustained increases in CoTS densities among years (Table 1) further extend increases recorded from 2020 to 2022 (Chandler et al. 2023), indicating the start of renewed population irruptions in this region (see also Uthicke et al. 2024). Critically, densities recorded in 2024, and especially 2025, exceed the nominal threshold (15 CoTS.ha^{-1}) used to distinguish population irruptions on the GBR (Moran and De'ath 1992), representing the fifth distinct episode of population irruptions since the 1960s. Moreover, 14–15 years have elapsed since the start of the last documented irruption in the vicinity of Lizard Island in 2010 (Vanhatalo et al. 2017). While CoTS densities increased substantially from 2023 to 2025, the average size of CoTS recorded in 2023 (421.74 mm ± 8.19 SE, $n = 69$) was very similar to that recorded in 2025 (422.44 mm ± 6.06 SE, $n = 127$; Fig. 1b) and did not vary significantly among years (Table 1), though mean CoTS size was larger in the present study compared to what was observed in previous studies (Chandler et al. 2023). The number of large (> 450 mm maximum diameter) starfish increased over time, which is expected given the growth of established starfish (Pratchett et al. 2014), but this was offset by simultaneous increases in the number of smaller starfish (< 300 mm diameter), indicating that there has been continued CoTS recruitment at Lizard Island, contributing to the increasing densities.

Daily feeding rates of individual CoTS exhibited significant and unexpected declines from 2023 to 2024 and remained low in 2025. These interannual changes in feeding rates were apparent based on both (1) the estimated mean number of feeding scars per day made by individual CoTS (Fig. 1c), which declined from 1.45 colonies (± 0.10 SE) in 2023 to 0.90 colonies (± 0.10 SE) in 2024 and was still reduced in 2025 (0.83 colonies ± 0.08), and (2) the combined tissue SA of feeding scars (Fig. 1d) which declined from 2,503.60 cm^2 (± 230.51 SE) in 2023 to 1,312.74 cm^2 (± 242.69 SE) in 2024 and remained at this lower level into 2025 (1,344.05 cm^2 ± 189.74 SE). While there is evidence that efficient predators can affect CoTS densities (e.g., Kayal & Lenihan 2025, Doll et al. 2026), it is unlikely that predators impacted the observed reduction in CoTS feeding rates due to increase in CoTS density over the study period. Individual feeding rates of CoTS are known to vary seasonally (e.g., Keesing and Lucas 1992), which is likely due to metabolic suppression at low temperatures. Similarly, elevated temperatures (that led to mass coral bleaching in 2024) may lead to changes in feeding and activity patterns (Lang et al. 2021), as shown for coral reef fishes (Johansen et al. 2013). Lab studies have shown that CoTS metabolism decreases when they are exposed to high temperatures (Lang et al. 2021, 2022), however the effects of thermal stress on CoTS feeding rates, especially in the presence of bleaching corals, have not yet been investigated. Such effects are, however, expected to be short-lived, due to the temporary nature of marine heatwaves, whereas feeding rates of CoTS appeared to be suppressed throughout 2024 and 2025.

Temporal declines in the feeding rates of CoTS may therefore reflect changes in the distribution and abundance of coral prey in the aftermath of the mass-bleaching (sensu Ling et al. 2020). However, there was no significant change in the proportional use of different coral prey among years (PERMANOVA; $F = 2.0783$, $\text{df} = 148$, $p = 0.059$; Fig. 2) nor in the variability of CoTS diet composition among years (PERMDISP; $F = 1.7611$, $p = 0.1755$). Partial mortality of coral colonies, which can be caused by bleaching events (Burn et al. 2022), could help explain the conservation of the pre-bleaching coral community structure despite the reduction in total coral cover. Importantly, *Acropora* spp. (both staghorn and other growth forms) accounted for > 80% of coral prey consumed in each year, even though there were moderate, but sustained declines in their availability (Fig. 2). This reflects the widely reported feeding preference for

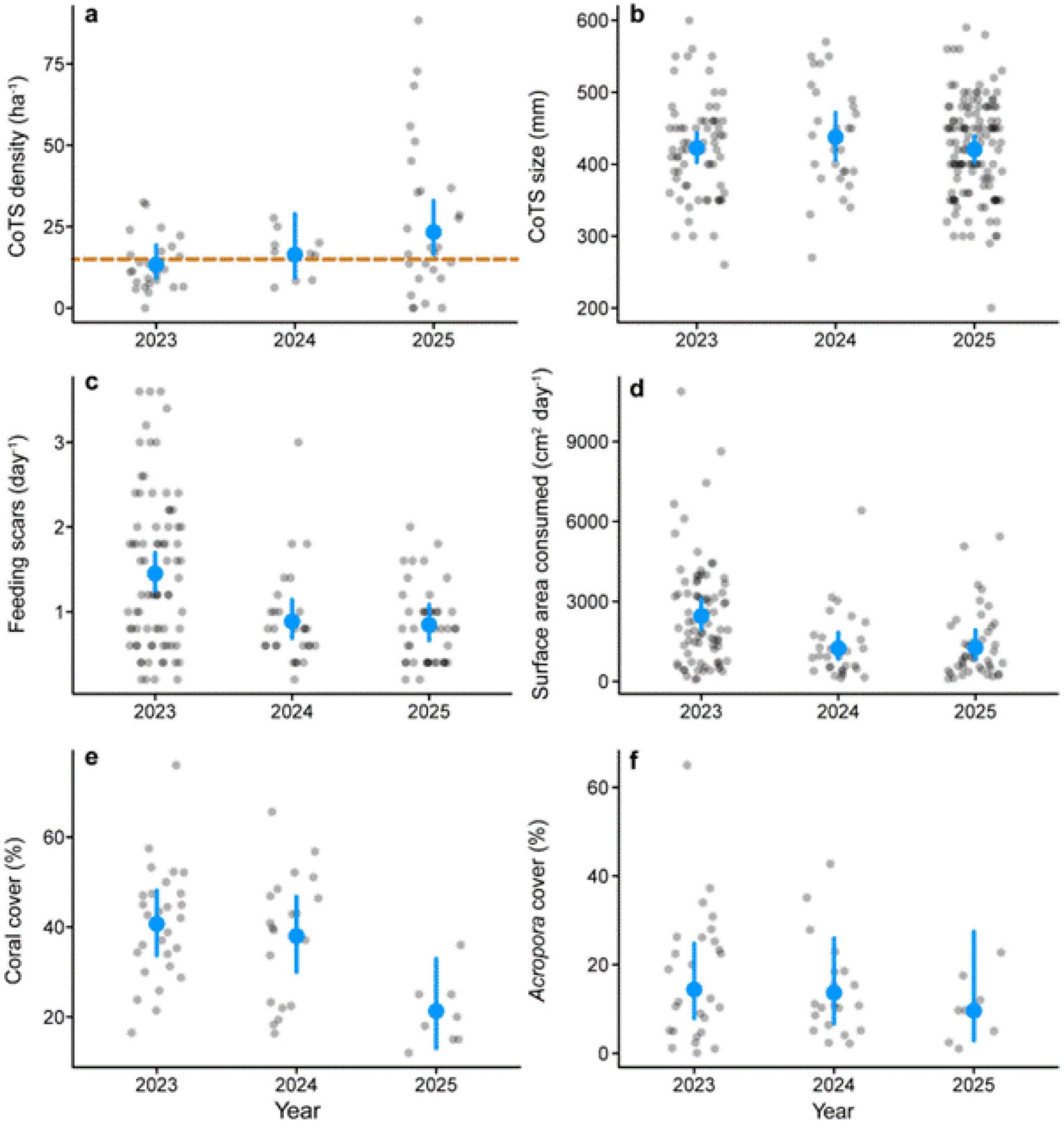

Fig. 1 Temporal variation in CoTS density (**a**), CoTS size (**b**), feeding scars made per day by individual CoTS (**c**), surface area of coral tissue consumed per day by individual CoTS (**d**), percent cover of hard corals (**e**), and percent cover of *Acropora* spp. The blue points denote the predicted mean and the line ranges the 95% confidence intervals from generalized linear mixed-effects models (GLMMs). Smaller gray points show the raw data. The orange dashed line is the accepted threshold for CoTS population irruptions (Moran and De'ath 1992)

Acropora by *A.* cf. *solaris* (e.g., Ling et al. 2020; Foo et al. 2024), which may be explained by the relatively high tissue SA that is consumed by CoTS when feeding on these complex corals (Chandler et al. 2024). Notably, control of CoTS populations on the GBR is performed by culling, where divers manually remove starfish from reefs (Matthews et al. 2024). Mass bleaching events could provide opportunities for managers to enact CoTS control measures on

Table 1 Summary of results from generalized linear mixed-effects models (GLMMs) used to examine changes in coral cover and crown-of-thorns starfish (CoTS) density, size, and feeding

Response variable	Model	Predictor variable	Estimate	SE	*z* value	*p* value
CoTS density	Tweedie GLMM	Intercept	2.5632	0.185	13.895	< 0.001
		2024	0.213	0.331	0.644	0.5198
		2025	0.582	0.245	2.376	0.0175
CoTS size	Gamma GLMM	Intercept	6.047	0.024	255.850	< 0.001
		2024	0.035	0.044	0.810	0.419
		2025	− 0.005	0.032	− 0.160	0.869
Feeding scars (day^{-1})	Gamma GLMM	Intercept	0.38615	0.070	5.535	< 0.001
		2024	− 0.496	0.132	− 3.743	< 0.001
		2025	− 0.569	0.123	− 4.627	< 0.001
Surface area (day^{-1})	Gamma GLMM	Intercept	7.831	0.108	72.700	< 0.001
		2024	− 0.677	0.209	− 3.230	0.0012
		2025	− 0.662	0.212	− 3.130	0.0018
Hard coral cover	Beta GLMM	Intercept	− 0.381	0.137	− 2.790	0.005
		2024	− 0.114	0.212	− 0.537	0.591
		2025	− 0.944	0.306	− 3.086	0.002
Acropora spp. cover	Beta GLMM	Intercept	− 1.814	0.303	− 5.992	< 0.001
		2024	− 0.067	0.465	− 0.144	0.885
		2025	− 0.482	0.647	− 0.745	0.457

Year was treated as a categorical fixed effect. SE = standard error

concentrated groups of starfish as they are drawn towords surviving preferred prey corals.

Cover of hard corals declined through the course of this study, but mainly from 2024 to 2025 (Fig. 1e, Table 1), whereby mean cover declined slightly from 40.66% (± 2.36 SE) in 2023 to 38.28% (± 3.14 SE) in 2024 and significantly declined to 20.75% (± 2.74 SE) in 2025. Given relatively low densities of CoTS recorded at Lizard Island, combined with apparent suppression of individual feeding rates in 2024–2025, it is difficult to attribute much of the localized coral loss to CoTS. Pratchett (2010), for example, recorded only moderate levels of coral loss (28.8% coral cover decline) at CoTS densities of < 25 CoTS.ha^{-1} during population irruptions at Lizard Island in the 1990s. Local coral loss recorded between 2024–2025 is, therefore, most likely due to coral mortality in the aftermath of severe mass bleaching that occurred in early 2024 (Garing et al. 2025; Raoult et al. 2025). Raoult et al. (2025) documented extensive bleaching in March 2024 in shallow reef environments at two sites on opposing sides of Lizard Island and further showed that many of the bleached corals (mainly *Acropora* spp.) had died by June 2024. This aligns with previous work showing coral mortality may be highly protracted in the aftermath of major bleaching events, depending on the species (Baird and Marshall 2002).

Overall cover of *Acropora* spp. was 17.11% (± 2.72SE) in 2023, compared to 14.16% (± 2.44SE) in 2024 and 10.11% (± 2.62SE) in 2025. Despite these apparent declines, there was no significant change in *Acropora* among years detected (Fig. 1f, Table 1). Moreover, coral composition in general did not vary significantly among years (PERMANOVA; $F = 1.775$, df = 55, $p = 0.056$; Fig. 2) nor was there a significant difference in the variability in coral community composition among years (PERMDISP; $F = 2.428$, $p = 0.098$). Coral composition changes observed in this study were relatively subtle compared to those in other parts of the GBR (e.g., Byrne et al. 2025), with our results for Lizard Island aligning with those of Garing et al. (2025) which showed shifts in the composition of corals at Lizard Island due to the 2024 bleaching event, especially in slope habitats, were more subtle relative to previous compositional changes. It is apparent that there were few transects with high (> 20%) cover of *Acropora* spp. in 2025, compared to 2023 and 2024 (Fig. 1f), which likely spatially constrained prey availability for CoTS. *Acropora* spp. tend to be the first and worst affected coral taxa during mass bleaching and mortality (Hughes et al. 2019), such that increasing frequency of mass bleaching events could lead to widespread declines in the abundance of *Acropora* spp. (Pratchett et al. 2020) and greatly constrain availability of preferred prey for CoTS. CoTS will travel to find its preferred prey (Ling et al. 2020), which could help to explain why we saw increased density of CoTS on reefs which still have populations of *Acropora* spp. CoTS will consume other coral taxa if preferred coral prey are not available (Pratchett et al. 2014), but increased consumption of non-preferred prey (e.g., *Porites* spp.) leads to declines in individual condition and fitness (Caballes et al. 2016). Moderate declines in the abundance of *Acropora* spp. recorded at Lizard Island are probably insufficient to moderate future increases in local CoTS densities, though

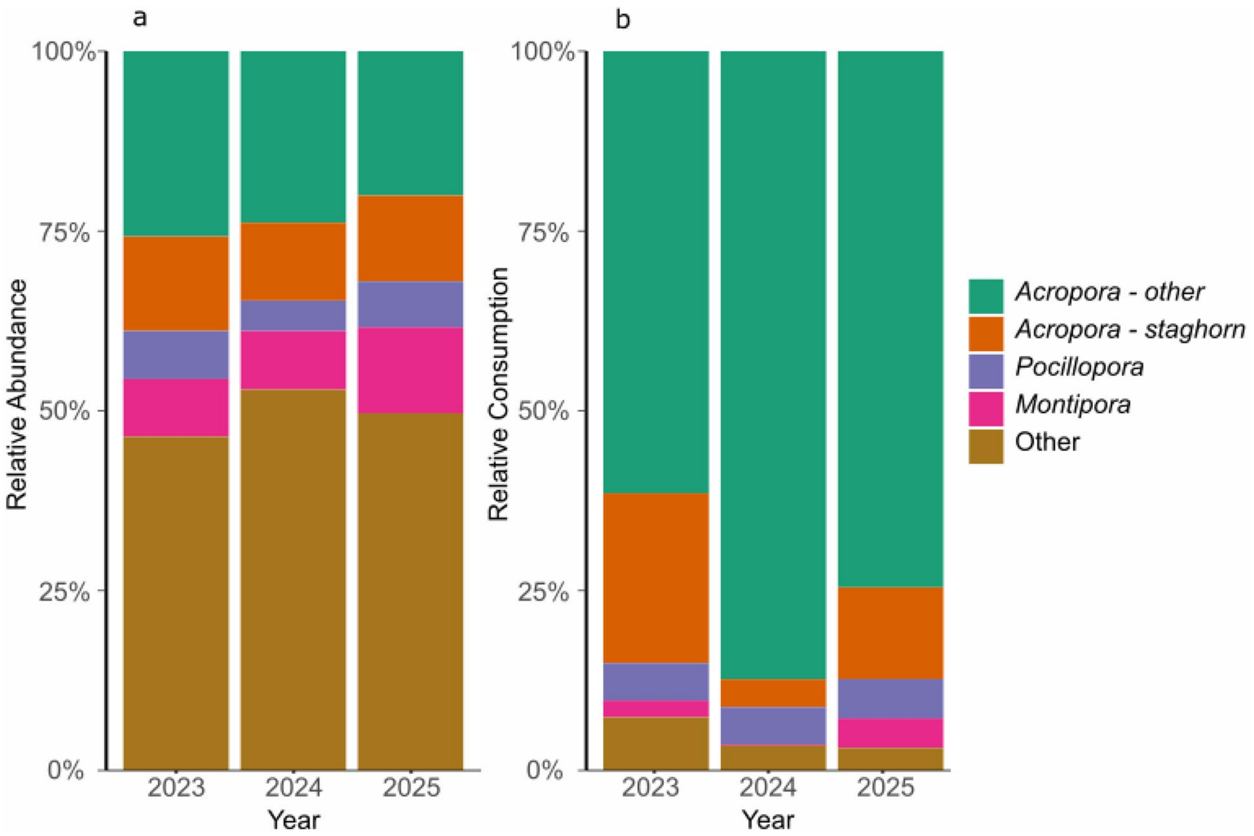

Fig. 2 Stacked bar charts showing the relative availability (**a**) and relative consumption by crown-of-thorns starfish (b; based on surface area of coral tissue consumed) of hard corals

sustained increases in CoTS are likely to further compound local coral loss, with disproportionate effects on *Acropora* spp. (Pratchett et al. 2014).

Conclusion

CoTS densities increased significantly at Lizard Island from 2023 to 2025, reaffirming the emergence of renewed population irruptions in the northern GBR (Chandler et al. 2023; Uthicke et al. 2024). However, individual feeding rates of CoTS declined significantly between 2023 and 2024, coinciding with mass coral bleaching and mortality (Garing et al. 2025; Raoult et al. 2025). Elevated temperatures that caused mass coral bleaching in 2024 may have led to metabolic suppression that caused declines in feeding activity of CoTS, and/or depletion of prey corals constrained their potential feeding rates. Given observed and projected increases in the densities of CoTS at Lizard Island, it is likely that their sustained feeding on prey corals will significantly conflate recent coral mortality caused by climate-induced coral bleaching. There is also a very high likelihood that there will be further incidences of mass coral bleaching and mortality (Hughes et al. 2018) that may moderate or surpass ecological effects of CoTS. Interactions between mass coral bleaching and population irruptions of CoTS needs careful consideration and further evaluation, especially given the projected increases in the coincidence of these major acute disturbances.

Data access

SALAD data on size and abundance of CoTS are being made available via eAtlas data repository (Pratchett et al. 2025), but specific data used in this study can be made available via direct requests to the corresponding author.

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Author contributions LTL, PCD, and MSP devised and formulated the study. LTL, PCD, MRG, and MSP collected the data. LTL and SBT analyzed the data. LTL and MSP wrote the first draft, and all authors read, edited, and approved the final manuscript.

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Data availability SALAD data on size and abundance of CoTS are being made available via eAtlas data repository ([Pratchett]https:/eatlas.org.au/datasets et al. 2025), but specific data used in this study can be made available via direct requests to the corresponding author.

Declarations

Conflict of interest Morgan Pratchett is the Editor in Chief of Coral Reefs and Sterling Tebbett is a Topic Editor at Coral Reefs, however, neither had any role in the peer review and publication decisions regarding this manuscript.

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